SECTION 7

FISH HEALTH

TABLE OF CONTENTS

7 FISH HEALTH

The fish health component of the AEMP is conducted every three years. It was conducted in 2012 and will be conducted again in 2015 and reported in the 2015 AEMP Annual Report.

SECTION 8

FISH COMMUNITY MONITORING

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Term	Definition
AICc	Akaike's Information Criterion
ANCOVA	analysis of covariance
AEMP	Aquatic Effects Monitoring Program
ANOVA	analysis of variance
ARGR	Arctic Grayling
Authorization	Authorization SC00196 as part of the *Fisheries Act*
AWCPUE	area-weighted catch per unit effort
BsM	Broad-scale Fish Community Monitoring
BURB	Burbot
CI	confidence interval
CISC	Cisco
COC	chain-of-custody
CL	cleithrum
CPUE	catch per unit effort
De Beers	De Beers Canada Inc.
df	degrees of freedom
DO	dissolved oxygen
e	mathematical constant e
EA	Environmental Assessment
EAR	Environmental Assessment Report
EEM	Environmental Effects Monitoring
e.g.	for example
FL	fork length
FR	fin ray
Fstat	F statistic
Golder	Golder Associates Ltd.
Hr	hour
HSD	Honesty Significance Difference
ID	identification number
i.e.,	that is
K-S	Kolmogorov-Smirnov
K-W	Kruskal-Wallis
LK13	Lake 13
LKCH	Lake Chub
LKTR	Lake Trout
Ln	natural log (base e)
LNSC	Longnose Sucker
LTH	Lake Trout per hectare
Mine	Snap Lake Mine
MVLWB	Mackenzie Valley Land and Water Board
N	number of samples

Term	Definition
n/a	not available
NAD	North American Datum
NEL	Northeast Lake
npar	the number of parameters of a model
NRPK	Northern Pike
NWT	Northwest Territories
OT	otoliths
P	probability
PIT	Passive Integrated Transponder
QA	quality assurance
QC	quality control
RNWH	Round Whitefish
SD	standard deviation
SE	standard error
SL	Snap Lake
SLSC	Slimy Sculpin
SR	studentized residual
SSD	statistical significant difference
TDS	total dissolved solids
TL	total length
UTM	Universal Transverse Mercator
W	body weight in grams

LIST OF SYMBOLS

Term	Definition
ΔAICc	difference in AICc values in comparison to the full, individual-lake model
α	condition factor from weight-length relationship
β	shape factor from weight-length relationship
$AGE_{Preferred}$	age using a preferred structure
$AGE_{Secondary}$	age using a secondary or less preferred structure
eggs/female	eggs per female
eggs/kg	eggs/kilogram
K	condition factor
k	von Bertalanffy growth coefficient
L	fork length
L_∞	asymptotic length, or length at infinity
L_t	total length of a fish at age t
ln	natural log
M	natural mortality
M′	natural annual mortality
r^2	coefficient of determination

Term	Definition
t_o	extrapolated age at which theoretical length is zero
ω	rate of growth in length
W	total body weight

UNITS OF MEASURE

Term	Definition
%	percent
>	greater than
<	less than
°	degrees
±	plus or minus
°C	degrees Celsius
°C/yr	degrees Celsius per year
µS/cm	microSiemens per centimetre
Adj r^2	coefficient of determination
cm	centimetre
g	gram
ha	hectares
kg	kilogram
kg/ha	kilogram per hectare
km	kilometre
km^2	square kilometres
km/hr	kilometres per hour
m	metre
mg/L	milligrams per litre
mm	millimetre
Mm^3	million cubic metres
yr	year

8 FISH COMMUNITY MONITORING

8.1 Introduction

8.1.1 Background

De Beers Canada Inc. (De Beers) is required to monitor the fish community in Snap Lake. The first fish community monitoring program occurred in 2009, when a modified Broad-scale Fish Monitoring (BsM) protocol (Sandstrom et al. 2009) was attempted in Snap Lake and one reference lake, Northeast Lake (De Beers 2010a), which at the time was required by the *Fisheries Act* Authorization (#SC00196) for Snap Lake Mine (Mine). This report presents the results of the second fish community monitoring program, as required for the Mine in Water Licence MV2011L2-0004. A modified[1] BsM protocol was employed again for the fish population monitoring program in 2013 in Snap Lake, Northeast Lake, and an additional reference lake, Lake 13, as per the updated Aquatic Effects Monitoring Program (AEMP) Design Plan (De Beers 2014). The AEMP Design Plan was updated in 2012 and finalized in 2013, such that large-bodied fish surveys are included to monitor fish populations and community composition, as well as fish tissue chemistry[2] (De Beers 2014).

Lake Trout (*Salvelinus namaycush*) and Round Whitefish (*Prosopium cylindraceum*) have been included in large-bodied fish surveys at the Mine in the past to document fish health and fish tissue chemistry in Snap Lake at baseline (1999), during early operations (2004), and as part of the AEMP (De Beers 2002, 2005, 2010a). Because Lake Trout and their prey (including Round Whitefish) were identified as a valued component during the Environmental Assessment (EA; De Beers 2002), additional discussion of these species is included herein.

8.1.2 Objective

The objective of the large-bodied fish community monitoring program is to determine whether treated effluent discharged from the Mine is having an effect on the fish community. Specific conditions in Water Licence MV2011L2-0004 that apply to the fish community component of the AEMP for the Mine (Part G, Schedule 6; Item 1a [iv] and 1d of MVLWB 2013) are:

a) Monitoring for the purpose of measuring Project-related effects on the following components of the Receiving Environment:

iv *fish population, and year class strength and community composition using standard methods;*

---

[1] Although the recommended BsM gear configuration is a double gang joined at the ends of the spanners, De Beers deployed single gangs in both 2009 and 2013 to minimize incidental fish mortalities as required by the Water Licence.

[2] The fish tissue chemistry component is presented in Section 9 of the AEMP report.

d) *Procedures to minimize the impacts of the AEMP on fish populations and fish habitat.*

The fish community monitoring program was designed to meet the above conditions by answering the following key question:

- Will the fish community be affected by the changes in water quality in Snap Lake and will any change be greater than predicted in the Environmental Assessment Report (EAR)?

The EAR predicted that the operation of the Mine would not result in any significant changes to the fish community in Snap Lake (De Beers 2002). The fish community monitoring program, therefore, considers numerous fisheries measurements, or metrics, in 2013 Snap Lake compared to the reference lakes as indicators of change in the fish community. This includes fish catch and effort, community composition (i.e., species richness and abundance), biological characteristics (i.e., length, weight, and age), length and age structure, condition, growth, fecundity, and mortality in each lake. One additional evaluation included a review of the mortality caused by the BsM in Snap Lake, to address the condition of minimizing impacts of the AEMP in Water Licence MV2011L2-0004. Temporal trends in fish community metrics will be assessed in the 2016 AEMP Re-evaluation Report.

8.1.3 Supporting Studies

A series of supporting studies were completed in 2013, including the studies listed below:

- a stable isotope study, inclusive of prey items of fish species, in Snap Lake (Section 11.4);
- a Lake Trout mark recapture study in Snap Lake (Section 11.3); and,
- an evaluation of seasonal changes in Lake Trout thermal habitat in Snap Lake, Northeast Lake, and Lake 13 during the ice free period (Appendix 8F).

An overview of these additional study programs is considered herein.

8.1.3.1 Stable Isotope Investigation of Snap Lake Food Chain

The stable isotope study targeted all of the fish species captured in Snap Lake in 2013, as well as lower trophic organisms such as zooplankton, phytoplankton, periphyton, and littoral and profundal macroinvertebrates. The study assessed the diets of fish in Snap Lake to determine the food web structure in the lake (Section 11.4).

8.1.3.2 Lake Trout Population Abundance

Populations of Lake Trout in low productivity subarctic waters are thought to be typically low in abundance with limited capacity to sustain exploitation (Power 1978). The BsM method has been predicted to result in less than 2 percent (%) mortality to fish populations in lakes sampled using the method (Sandstrom 2013, pers. comm.). To accurately determine the mortality rate of Lake Trout associated with the BsM method in Snap Lake, a mark recapture study was initiated in 2012 and completed in 2013 (Section 11.3). A density estimate of Lake Trout in Snap Lake (i.e., fish per hectare [ha]) was also determined from the mark recapture study, which provided an opportunity to compare Lake Trout density in Snap Lake relative to other lakes in North America. The results of the Lake Trout mark recapture study are reported in Section 11.3 and the implications to Lake Trout mortality in Snap Lake caused by the BsM are reviewed in Section 8.5.

8.1.3.3 Thermal Habitat for Lake Trout

Shallow lakes, such as Snap Lake, receive high levels of solar insolation during the summer. Peak water temperatures may be increasing due to increasing air temperatures, particularly in Canada's north (Schindler et al. 1996). Lake Trout are a cold water stenotherm, and are highly sensitive to increases in temperature (Plumb and Blanchfield 2009). This sensitivity can affect habitat suitability, availability, and use, with potential negative implications to growth and reproduction. An assessment of the relative availability of suitable thermal habitat within the three study lakes was performed to help with interpretation of the 2013 fish community monitoring results with respect to influences of climate change versus Mine-related effects. Although the lakes in the Snap Lake AEMP are relatively small, the factors affecting thermal dynamics are unclear (Adrian et al. 2009), requiring that the thermal habitat for each lake be examined on an individual lake basis. To evaluate seasonal changes in the amount of thermal habitat for Lake Trout in the three study lakes, arrays of temperature loggers suspended at a range of depths from a fixed mooring anchored at the deepest part of each lake during the summer of 2013 were used (Section 2). The results of the thermal habitat analysis of the logger arrays are provided in Appendices 8E to 8F, and the potential implications of thermal habitat to Lake Trout are discussed in Section 8.5. Thermal habitat is also discussed in Section 11.3 (Lake Trout Population Estimate).

8.2 Methods

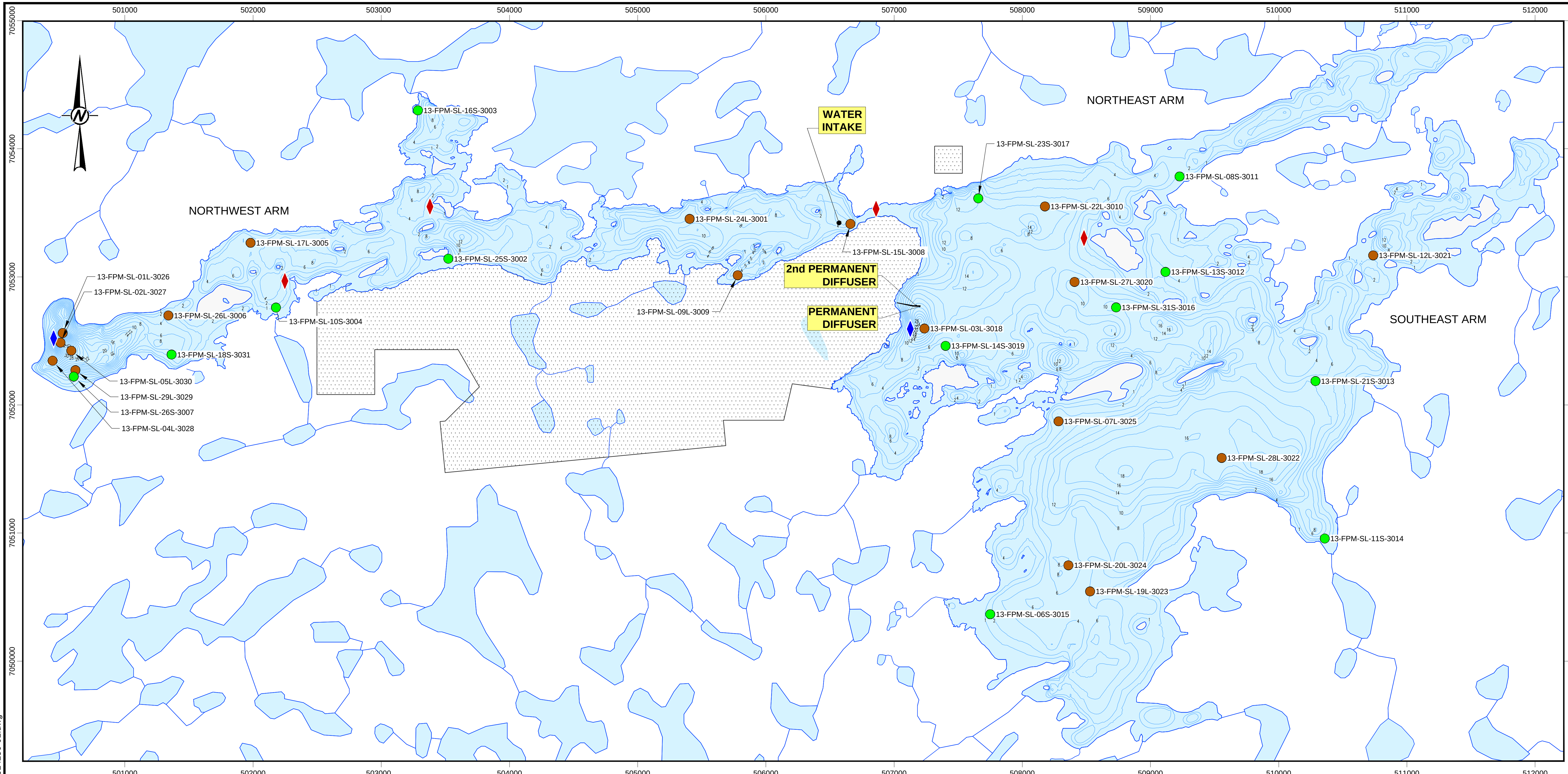
8.2.1 Sampling Areas

The study areas for the 2013 AEMP fish community monitoring consisted of Snap Lake, Northeast Lake, and Lake 13 (Section 1, Figure 1-1). Individual net set locations within the study lakes are presented in Figure 8-1 to 8-3.

8.2.1.1 Snap Lake

Snap Lake has a surface area of approximately 15.7 square kilometres (km^2), equivalent to 1,566 hectares (ha, and an estimated volume exceeding 80 million cubic metres (Mm^3).[3] Snap Lake is shallow, with a mean depth of approximately 5 metres (m), and is well mixed with little evidence of thermal stratification during open-water conditions. There are two deep areas that have depth greater than 20 m: one deep area is located in the main basin near the diffuser, and the other deep area is located in the northwest arm (Figure 8-1). Snap Lake is very clear, having a Secchi depth of 6 to 7 m (Section 3.4.1), and is classified as an oligo-mesotrophic lake because of low to moderate concentrations of nutrients and organic productivity (De Beers 2012a). The ice-free or open-water season for Snap Lake generally runs from July to October, with the lake ice-covered for the period of November to June. The duration of the open-water and ice-covered periods have been consistent for the past seven years (De Beers 2006, 2007, 2008, 2009, 2010b, 2011, 2012b, 2013).

[3] There is some uncertainty about total lake volume due to changes in water level, and this value of 80 Mm^3 is currently under review.



L:\2013\1349\13-1349-0001\Phase 4200\Task 4250\13134900014250-01.dwg

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- Fish Community Monitoring Station - Large Mesh Gillnet
- Fish Community Monitoring Station - Small Mesh Gillnet
- Temperature Logger Location - Shallow
- Temperature Logger Location - Deep

REFERENCES

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

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

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

SNAP LAKE MINE

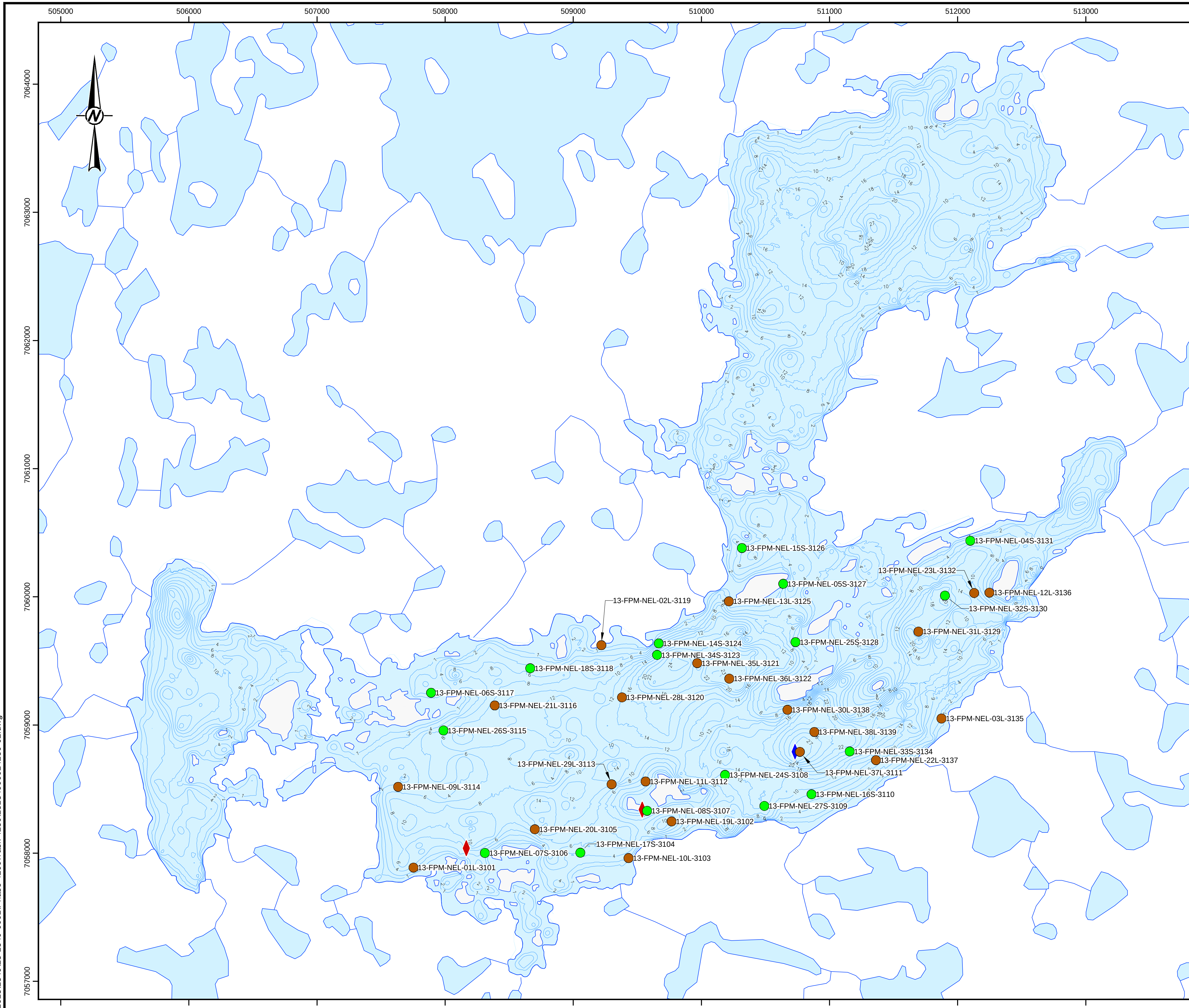
Bathymetric Map of Snap Lake Showing Locations of Fish Community Gill Net Sites and Water Temperature Stations, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83
500 0 500 SCALE 1:30,000 METRES	DE BEERS GROUP OF COMPANIES
FILE No: 13134900014250-01	DATE: February 4, 2014
JOB No: 14-1349-0004	REVISION No: A
OFFICE: GOLDER VICTORIA	DRAWN: JEF CHECK: PMC

Figure 8-1

8.2.1.2 Northeast Lake

Northeast Lake is located 10 km northeast of Snap Lake and is a relatively small lake, with a surface area of approximately 17.7 km^2 or 1,769 ha, and an approximate volume of 125.8 Mm^3 (Section 1, Figure 1-2). Northeast Lake is shallow with a mean depth of 7.1 m and a maximum depth of 30 m (Figure 8-2).



LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Fish Community Monitoring Station - Large Mesh Gillnet
- Fish Community Monitoring Station - Small Mesh Gillnet
- Temperature Logger Location - Shallow
- Temperature Logger Location - Deep

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.

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

Grid is displayed in Transverse Mercator, Datum : NAD83, Coordinate system : UTM zone 12.

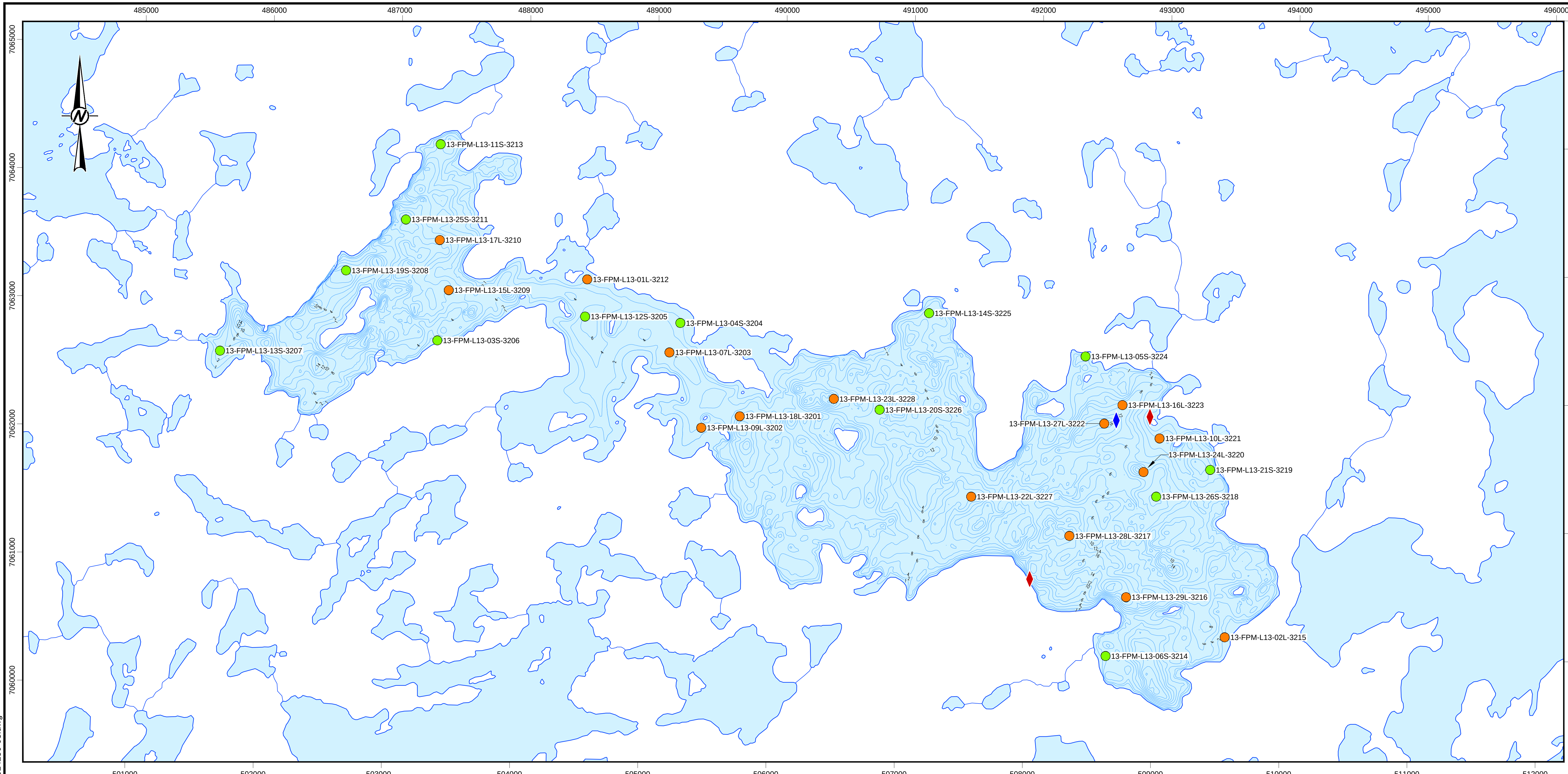
SNAP LAKE MINE

Bathymetric Map of Northeast Lake Showing Locations of Fish Community Gill Net Sites and Water Temperature Stations, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES	
500 0 500 SCALE 1:30,000 METRES			
FILE No: 13134900014250-02		DATE: February 4, 2014	
JOB No: 14-1349-0004	REVISION No: A	Figure 8-2	
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

8.2.1.3 Lake 13

Lake 13 is located 10 km northwest of Snap Lake and is a relatively small and shallow lake, with a surface area of approximately 10.9 km^2 or 1,088 ha, and an approximate volume of 61 Mm^3 (Section 1, Figure 1-2). The mean depth of Lake 13 is 5.6 m, and the maximum depth is 22 m (Figure 8-3).



L:\2013\1349\13-1349-0001\Phase 4200\Task 4250\13134900014250-03.dwg

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Fish Community Monitoring Station - Large Mesh Gillnet
- Fish Community Monitoring Station - Small Mesh Gillnet
- Temperature Logger Location - Shallow
- Temperature Logger Location - Deep

REFERENCES

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

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

NOTES

Map moved to projection : Transverse Mercator, Datum : NAD83, Coordinate system : UTM zone 12.

SNAP LAKE MINE

Bathymetric Map of Lake 13 Showing Locations of Fish Community Gill Net Sites and Water Temperature Stations, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83
500 0 500 SCALE 1:30,000 METRES	DE BEERS GROUP OF COMPANIES
FILE No: 13134900014250-03	DATE: February 4, 2014
JOB No: 14-1349-0004	REVISION No: A
OFFICE: GOLDER VICTORIA	DRAWN: JEF / CHECK: PMC

Figure 8-3

8.2.2 Timing of Sampling

The fish community monitoring program occurred in July of 2013, approximately 20 days after ice-off, when fish were expected to be most active and randomly distributed throughout the water column. The program consisted of 13 consecutive days of fishing effort, from July 6 to July 18, 2013. Each lake was fished on the following dates:

- Lake 13: July 6 to 10, 2013;
- Northeast Lake: July 11 to 16, 2013; and,
- Snap Lake: July 12 to 18, 2013.

8.2.3 Field Methods

8.2.3.1 Broad-scale Fish Community Monitoring Netting Protocol

The BsM protocol specifies a combination of large and small mesh gill nets (Appendix 8A, Table 8A-4) spanning a range of mesh sizes in each gang to target a broad range of fish sizes and species (Sandstrom et al. 2009). Net gangs of large or small mesh nets were set on the lake bottom at a range of depths, as prescribed in the BsM protocol, perpendicular to depth contours, with the number of nets set in each lake dependent on lake area and maximum depth. Large and small mesh nets were not set in the same area to reduce potential capture bias for large fish that might be attracted to small fish caught in the small mesh nets. Small mesh ranged in size from 60 to 840 millimetres (mm), and large mesh ranged from 160 to 1,020 mm.

Nets were set for a minimum of 18 hours. Catches were standardized by converting to number of fish caught per 24-hour period. It was not possible to include dawn and dusk periods in the sampling effort, as per the BsM protocol, due to perpetual daylight in the northern hemisphere in July. All sampling was completed in daylight.

Sampling effort was allocated as equally as possible in all regions of each lake and was spatially stratified by water depth (Figures 8-1, 8-2, and 8-3; Appendix 8A, Tables 8A-1, 8A-2, and 8A-3) using gear as described in Appendix 8A, Table 8A-4. The number of net deployments was determined by the lake surface area and maximum water depth. The number of net deployments by mesh size for the three individual lakes, each with an 18-hour set duration, is shown in Table 8-1.

Table 8-1 Summary of the Water Depths and Target Number of 18 hour Net Deployments for Small and Large Mesh Gill Nets for All Lakes During the 2013 Snap Lake AEMP Fish Community Survey

Design		Number of Net Deployments					
Strata ID	Depth (m)	Snap Lake		Northeast Lake		Lake 13	
		Large Mesh	Small Mesh	Large Mesh	Small Mesh	Large Mesh	Small Mesh
1	1 to 3	2	4	3	5	2	4
2	3 to 6	4	4	5	5	4	4
3	6 to 12	4	3	5	4	4	3
4	12 to 20	3	2	4	3	3	2
5	20 to 35	3	n/a	4	n/a	3	n/a
6	35 to 50	2	n/a	n/a	n/a	n/a	n/a
Total		**18**	**13**	**21**	**17**	**16**	**13**

ID = identification number; m = metre; n/a = not applicable; AEMP = Aquatics Effects Monitoring Program.

The following information was recorded for each day of netting on each lake:

- site number – a unique code that identifies each sampling site;
- effort number – a unique code that identifies each net deployment;
- set date and time (mm/dd/yr; 24-hour format);
- lift date and time; (mm/dd/yr; 24-hour format);
- gear specific parameters (e.g., net type, small or large mesh);
- water depth of each gill net set (e.g., minimum and maximum in metres);
- Universal Transverse Mercator (UTM) co-ordinates of each fishing effort (North American Datum [NAD] 83; Zone12 V);
- environmental conditions, including air temperature in degrees Celsius (°C), cloud cover (%), wind direction in degrees (°), and speed in kilometres per hour (km/hr);
- water quality profile taken once daily from each lake during fishing operations, including dissolved oxygen (DO) in milligrams per litre (mg/L), water temperature (°C), pH, and conductivity in microSiemens per centimetre (µs/cm); and,
- number and species of fish captured and observed.

Captured fish were sorted by sample effort number and gill net size category (i.e., large or small mesh), but not by net panel or mesh size within a category.

8.2.3.2 Fish Processing

Fish were held in a tote box filled with fresh lake water if alive at the time of capture and still in good condition after removal from the gill net. The fish were processed as described below, and released if they appeared capable of surviving. If survival was in doubt, fish were sacrificed with a blow to the head and processed lethally. All dead fish were held on ice prior to examination to reduce the rate of tissue degeneration.

All captured fish, whether dead or alive, were visually examined externally and any features of the fish that did not appear normal (i.e., wounds, tumours, parasites, fin fraying, gill parasites, or lesions) were photographed and recorded (Appendix 8B-2). External examinations were completed following the recommendations outlined in Environment Canada (2012). Information on maturity, sex, and overall health was also recorded.

For each fish that was live-released, the following information was collected and recorded:

- species;
- fork length, plus or minus (±) 1 mm (if applicable);
- total length (± 1 mm);
- total body weight (± 0.1 gram [g]);
- sex (if evident, otherwise recorded as unknown);
- life stage (if evident, otherwise recorded as unknown);
- external health assessment (an examination was conducted on each fish of the eyes, gills, thymus, skin, body form, fins, and operculum); and,
- photographs of any fish with abnormal external features (Appendix 8B-2).

The first three leading fin rays of the left pelvic fin ray were removed from each fish for aging purposes. In addition to the above, Lake Trout were also examined for the presence of a Passive Integrated Transponder (PIT) tag and a visual check made for the presence of adipose or left pelvic fin clips as related to the mark recapture study (see Section 11.3);

In addition to the above information, dead fish were also examined internally, and the following information was collected and recorded:

- internal pathology (i.e., liver and kidney colour, fat content);
- sex;
- state of reproductive development (i.e., maturity categories as outlined in Table 8-2);
- stomach contents (% fullness);
- whole liver weight (± 0.1 g);
- whole gonad weight (± 0.1 g); and,
- fecundity (female only).

Aging structures (i.e., sagittal otoliths, scales, pelvic fin rays, and/or cleithra) were removed from each fish for aging purposes. In addition to the above observations, muscle, liver, and kidney samples were also collected from Lake Trout and Round Whitefish from all three study lakes for tissue chemistry (see Section 9) and muscle samples were collected from all Snap Lake fish for stable isotope analysis (see Section 11.4).

Table 8-2 Fish Maturity Categories and Associated Criteria Used During the 2013 Snap Lake AEMP Fish Community Survey

Life Stage	Maturity Stage	Definition and Morphological Criteria
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
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

Note: categories based on visual observation in the field; AEMP = Aquatics Effects Monitoring Program.

The internal condition of all dead fish was observed and recorded immediately following the opening of the body cavity (i.e., tissue colour and condition). The liver was removed first; liver weight was recorded and the whole liver was placed in a labelled zip-lock bag. Stomach fullness was observed and recorded, along with a general description of gut contents and parasite loads. Stomachs from the first five fish of each species examined with greater than 50% subjective stomach fullness were removed and stored in a labelled, leak-proof vial filled with 10% buffered formalin. Stomach contents were archived upon completion of the program.

Fish sex and sexual maturity were recorded as per the maturity stages outlined in Table 8-2, and the gonads were examined for any abnormalities. Both right and left gonad lobes were removed and the total gonad weight was recorded. For females, approximately 5 g of ovarian tissue was removed from the developed portion of the left and right ovaries, weighed, and placed in a leak-proof vial with 10% buffered formalin for fecundity analysis (i.e., number of eggs per fish).

Appropriate aging structures for the species in question (e.g., sagittal otoliths, cleithra, scales, or pectoral fin rays) were collected for age determination from each fish according to the methods outlined by Mackay et al. (1990). Sagittal otoliths were the primary and preferred aging structure for most fish species collected, as otoliths are a more accurate reflection of age for older fish than scales and fin rays which tend to underestimate the age of older fish. Accordingly, attempts were made to collect sagittal otoliths from all dead fish wherever possible. If two sagittal otoliths could not be collected, a secondary aging structure was collected.

All information was recorded on the catch record field data sheet. Measurements were taken in the field and the live fish were released near the capture location.

8.2.4 Age Determination

If two sagittal otoliths could not be collected from a fish, a secondary aging structure was collected and regression analysis was used to convert fin ray age to otolith age prior to statistical analysis (Sokal and Rohlf 1995). All aging structures were preserved, stored in individually labelled envelopes, and shipped according to standard protocols to North/South Consultants Inc. in Winnipeg, Manitoba for analysis. Otoliths and fin rays were used for Lake Trout, Round Whitefish, Longnose Sucker (*Catostomus catostomus*), Arctic Grayling (*Thymallus arcticus*), and Lake Chub (*Couesius plumbeus*); for Northern Pike (*Esox lucius*), age was determined from cleithra and fin rays. For Burbot (*Lota lota*) and Cisco (*Coregonus artedii*), fish were aged using otoliths.

Otoliths and fin rays were processed using a thin-sectioning technique. Sectioning was done using a Minitom low speed sectioning saw. Prior to sectioning, fin rays were dipped in Cold Cure epoxy resin and allowed to harden. Otolith and fin ray sections (0.5 to 0.75 mm thickness) were permanently mounted onto a glass microscope slide using Cytoseal-60 and interpreted by visual examination using a Leica MS5 dissecting microscope and transmitted light. Round Whitefish otoliths were analyzed whole by interpreting annuli on the surface of the structure under a Leica MS5 dissecting microscope (MacKay et al. 1990). Cleithra were boiled and cleaned to remove connective tissue and were aged by counting annuli (unaided) under light. All aging structures were interpreted a minimum of two times. A subsample consisting of 10% of all aging structures was examined by a second fishery technician and ages compared between technicians. In cases of disagreement between observers on a fish age, the mean age of both estimates was used to provide an age for that fish.

If the preferred aging structure for an individual fish species was not available or was too damaged to use, age was instead predicted using a linear regression relationship (Sokal and Rohlf 1995) developed using the preferred and the secondary aging structures for each species where both aging structures were collected (Table 8-3), as follows:

$$AGE_{Preferred} = a + b\,(AGE_{Secondary}) \qquad \text{[Equation 8.1]}$$

where: $Age_{preferred}$ = the predicted age for the preferred structure
a = intercept of the line
b = slope of line
$Age_{secondary}$ = age as determined from secondary aging structure

Age was predicted for 10 Lake Trout, 7 Round Whitefish, 5 Northern Pike, and 1 Lake Chub. The resulting regression relationships between the preferred and secondary aging structures were highly significant (Table 8-3).

Table 8-3 Statistical Relationships Between Ages Determined From Preferred and Secondary Aging Structures for Fish Species Collected From All Lakes[(a)] During the July 2013 Snap Lake AEMP Fish Community Survey

Species	Aging Structure	Preferred/ Secondary	N[(b)]	Range (yr)	Mean (yr)	SD (yr)	Regression Equation	*P*-value	Adj. r^2
LKTR	Otolith	Preferred	57	5 to 43	18	10	OT Age = 0.9807 + 1.4423 (FR Age)	<0.0001	0.68
	Fin ray	Secondary		3 to 25	12	6			
RNWH	Otolith	Preferred	56	1 to 10	4	2	OT Age = 0.6803 + 1.0045 (FR Age)	<0.0001	0.62
	Fin ray	Secondary		1 to 8	4	2			
LKCH	Otolith	Preferred	23	1 to 11	5	3	OT Age = 1.129 + 0.733 (FR Age)	<0.0001	0.68
	Fin ray	Secondary		1 to 12	5	3			
NRPK	Cleithra	Preferred	9	1 to 14	9	4	CL Age = 0.4598 + 0.9626 (FR Age)	0.0005	0.82
	Fin ray	Secondary		1 to 15	9	4			

(a) Data from Snap Lake, Northeast Lake, and Lake 13 were all combined for the age determinations from secondary aging structures.
(b) Value represents the number of fish from which both primary and secondary aging structures were available and used to establish the regression equation to predict otolith age from secondary aging structures.
LKTR = Lake Trout; RNWH = Round Whitefish; LKCH = Lake Chub; NRPK = Northern Pike; N = number of samples; OT = otoliths; FR = fin ray; CL = cleithrum; yr = year; SD = standard deviation; *P* = probability; Adj. r^2= coefficient of determination; < = less than; AEMP = Aquatics Effects Monitoring Program.

8.2.5 Data Analysis

Data for species collected in 2013 were statistically compared among Snap Lake, Northeast Lake, and Lake 13. These comparisons were performed to provide a basis for evaluating the current status and observed changes within Snap Lake, and for addressing the question of whether the fish community in Snap Lake has been affected by Mine-related changes in water quality.

Where statistical tests were performed, data were screened for statistical outliers using regression analysis. A key requirement in regression analysis is the homoscedastic distribution of errors, or equal scatter within the range of data included in the regression (Hoffman 2004). Examining studentized residual (SR) values, or the residuals of the regression normalized to an estimate of variance, provided a means to quantitatively screen for outliers (Hoffman 2004). In general, any SR value beyond ± 3 was examined more closely and tended to be removed from the data set and subsequent analyses. Outliers can be found in Appendix 8G, Table 8G-2. Analysis of covariance (ANCOVA) (Sokal and Rohlf 1995) was used to compare the relationship between weight and length for Lake Trout and Round Whitefish among lakes. If a significant interaction existed between a lake and a covariate, ANCOVA could still proceed if the coefficient of determination (r^2) of the full regression model (i.e., including the interaction term) was greater than 0.8 and only slightly greater (i.e., 0.02 or less) than the r^2 of the reduced regression model (i.e., interaction term removed) (Barrett et al. 2010). Individuals for each species were first pooled across all fish and then analyzed separately by sex.

8.2.5.1 Fish Catch and Effort

An area-weighted catch per unit effort (AWCPUE) (Sandstrom et al. 2009) was calculated for each species within each lake where more than 10 individuals were caught in at least one of the three study lakes. The AWCPUE was derived by multiplying average conventional catch per unit effort (CPUE) within a particular depth stratum by the proportion of the total lake area within that stratum, and summing across all of the depth strata fished. The area of each depth stratum within a given lake was calculated from the hypsometric curve for that lake. Hyposometric curves were developed using existing bathymetry records using AutoCAD Civil 3D. The total AWCPUE for each species for either large or small mesh nets was calculated by taking the sum of the AWCPUEs for the area-weighted depth strata for either mesh size. Effort was standardized to 24 hours.

For fish species with total catches below 10 within a given lake, a CPUE was calculated by dividing the number of fish caught by the number of total hours of effort for the lake (e.g., fish/hr).

To determine size selectivity of both large and small mesh nets for all species combined, length bins of 20 mm width (i.e., increasing size from 0 to 1,020 mm) were used to calculate the length frequencies of each species caught throughout the program. Length frequency distributions were constructed for each species for each small and large mesh nets.

8.2.5.2 Community Composition

Community composition was assessed as the number of fish species present in a lake. The type of gill net in which species were captured was also considered, as individual small mesh, large mesh, and all mesh (i.e., pooled data). To evaluate changes in community composition, species collected in previous studies and therefore confirmed as resident species in the lakes were considered herein; however, quantitative comparisons of community composition over time were not performed.

8.2.5.3 Biological Characteristics of the Catch

Descriptive statistics, including arithmetic mean, median, minimum, maximum, standard deviation (SD), and standard error (SE) were calculated for fork length, total length, body weight, and age for each species, within each lake.

Descriptive statistics were based on all fish collected and not differentiated by sex, except for Lake Trout and Round Whitefish. Lake Trout and Round Whitefish data were pooled across all fish (i.e., including juveniles of unknown sex), and were also analyzed separately by sex (i.e., confirmed males and female were grouped and analyzed together).

Analysis of variance (ANOVA) (Sokal and Rohlf 1995) was used to test for differences among lakes within species for fork length (mm), body weight (g), body condition (K), and age. Normality and equality of variance of data were tested prior to statistical analyses. If the data were either non-normal or did not demonstrate homogeneity of variance, data were log-transformed and re-tested. If the data still did not meet the assumptions of normality and equal variance, a non-parametric test was used (e.g., Kruskal-Wallis) (Sokal and Rohlf 1995). Where ANOVAs indicated significant differences, Tukey's honestly significant difference (HSD) test was performed to determine which lakes differed (Sokal and Rohlf 1995). All tests were considered significant at $P = 0.10$.

8.2.5.4 Length and Age Structure

The non-parametric two-sample Kolmogorov-Smirnov (K-S) test (Sokal and Rohlf 1995) was used to compare length- and age-frequency distributions among lakes for Lake Trout and Round Whitefish. Comparisons were made by pooling all fish, as well as separating fish by sex.

Ricker (1975) defined year class strength as: "*the fish spawned or hatched in a given year.*" Although low numbers of age-0 and age-1 fish were captured during the monitoring program, year class strength indices could be evaluated from population age structure data. For example, the relative contribution of age classes to the population of each lake was assessed. This was done by tabulating and plotting the percentage of the catch at each age against age-class. The distributions of fish in each age class were compared using the K-S test to examine differences in age-class structure between the lakes as a potential indication of historical differences in year class strength. Raw catch data (i.e., real numbers of fish at age, uncorrected for set duration) were used for this analysis.

8.2.5.5 Condition and Growth

Two measures of condition were derived based on whole fish including Fulton's condition factor (K), and the weight-length relationship.

Fulton's K

Fulton's condition factor (K) was calculated using the following equation (Ricker 1975):

$$K = \left(\frac{W}{L^3}\right) \times 10^5 \qquad \text{[Equation 8.2]}$$

where: K = Fulton's condition factor
W = total body weight (g)
L = fork length (mm)

Difference in Fulton's condition among lakes was tested by ANCOVA, using total body weight and fork length as the covariates.

Weight-Length Relationship

Using weight and length, the condition (α) and shape (β) were estimated using linear regression on the following model (Quinn and Deriso 1999):

$$\mathrm{Ln}\, W = \mathrm{Ln}\, \alpha + \beta\, \mathrm{Ln}\, L + \mathcal{E} \qquad \text{[Equation 8.3]}$$

where: W = total body weight (g)
L = fork length (mm)
α = the intercept, which in this case is known as the condition
β = the slope, which in this case is known as the shape factor
Ɛ = additive process error.

If β, referred to as shape, is less than or greater than 3, fish are growing allometrically (becoming less or more rotund with length respectively) whereas if β equals 3 it indicates isometric growth, indicating growth is occurring with unchanged body proportions.

Variation in Lake Trout growth was examined using the length-at-age relationship among the three lakes, using the von Bertalanffy growth equation (Quinn and Deriso 1999) as follows:

$$L_t = L_\infty(1 - e^{-k(t-t_0)}) \quad \text{[Equation 8.4]}$$

where: L_t = the total length of a fish at age t (mm)
L_∞ = the asymptotic length (mm) or length at infinity of a fish (i.e., the theoretical length to which a fish would grow possible to grow infinitely old)
k = the von Bertalanffy growth coefficient (i.e., the fraction by which the gap between L_t and L_∞ is closed each year)
t = age
t_0 = theoretical age at 0 length
e = mathematical constant e.

For Round Whitefish, the three-parameter model could not converge due lack of small, young fish data. Hence, the t_0 term was removed from the Round Whitefish model:

$$L_t = L_\infty(1 - e^{-k(t)}) \quad \text{[Equation 8.5]}$$

where: L_t = the total length of a fish at age t (mm)
L_∞ = the asymptotic length (mm) or length at infinity of a fish (i.e., the theoretical length to which a fish would grow possible to grow infinitely old)
k = the von Bertalanffy growth coefficient (i.e., the fraction by which the gap between L_t and L_∞ is closed each year)
t = age
e = mathematical constant e.

The rate of growth for Lake Trout from each lake was calculated from k and L_∞ as follows:

$$\omega = k \times L_\infty \quad \text{[Equation 8.6]}$$

where: ω = rate of growth in length (mm)
k = von Bertalanffy growth coefficient
L_∞ = asymptotic length (mm).

The analysis was implemented in the statistical environment R, v. 3.0.1 (R Core Team 2013) using packages “FSA” (Ogle 2010) and “nlstools” (Baty and Delignette-Muller 2013), and the “nls” function for non-linear regression. A fully specified, lake-specific von Bertalanffy curve was estimated, where each lake was assumed to have a separate value of L_∞, t_0, and *k*. In addition, three curves were estimated assuming that every two lakes were not statistically significant, i.e., 1) common parameters for Lake 13 and Snap Lake (but another set of parameters for Northeast Lake), 2) common parameters for Lake 13 and Northeast Lake (but a separate set of parameters for Snap Lake), and 3) a common set of parameters for Northeast Lake and Snap Lake, but a separate set for Lake 13 differed. The curves from these two-lake combined models were compared to the fully specified model (individual-lake) using F-tests on models' residual sums of squares and corrected Akaike's Information Criterion (AICc)[4] of the simplified model vs. the full model:

$$AICc = -2 \times LL + 2 \times npar + \frac{2 \times npar(npar+1)}{N - npar - 1} \quad \text{[Equation 8.7]}$$

where: N = the number of data points
npar = the number of parameters of a model
LL = the maximized log-likelihood of a model (Sakamoto et al. 1986).

Models with lower AICc values are preferred, as they indicate better fit and/or have fewer parameters, decreasing the risk of overfitting. If AICc value of the simplified model was lower and the F-test indicated that the simplified model was not significantly different from the full model ($\alpha \geq 0.05$), the combined lakes in the simplified model were concluded to not have significantly different growth curves.

Once the final curves were estimated, mean values and 95% confidence intervals were calculated for each model parameter. Plots of growth curves were provided for each species at each lake, if applicable, or for each species at a combination of lakes, if no significant difference was found among lakes. Plotting was performed in R using the packages "ggplot2" (Wickham 2009).

8.2.5.6 Fecundity

Fecundity (i.e., eggs/female) was calculated for seasonally developed Lake Trout and Round Whitefish using one of two equations depending on whether a sub-sample was taken or not.

Subsample taken:

$$\text{Fecundity} = \frac{\text{\# of eggs in subsample} \times \text{lobe weight (lab)} \times \text{total ovary weight (fresh)}}{\text{subsample weight (lab)} \times \text{lobe weight (fresh)}} \quad \text{[Equation 8.8]}$$

No subsample taken:

$$\text{Fecundity} = \frac{\text{\# of eggs in sample} \times \text{total ovary weight (fresh)}}{\text{lobe weight (fresh)}} \quad \text{[Equation 8.9]}$$

Note: all weights are in grams (g).

[4] The Akaike information criterion is a measure of the relative quality of a statistical model, for a given set of data; it provides a means for model selection. The AIC deals with the trade-off between the goodness of fit of the model and the complexity of the model; it offers a relative estimate of the information lost when a given model is used to represent the process that generates the data.

Fecundity was also expressed as the number of eggs per kilogram (i.e., eggs/kg), which normalizes fecundity relative to fish weight. The relationship between fecundity (number of eggs) and fork length (mm) was also compared among the three study lakes as per Koops et al. (2004) by ANCOVA for both Lake Trout and Round Whitefish. Summary statistics were calculated for both measures of fecundity.

8.2.5.7 Mortality

Natural Mortality

Natural mortality rate within each of the three study lakes for Lake Trout and Round Whitefish was estimated from a catch curve, which was created based on the number of fish in each age-class caught during BsM sampling regardless of sex or stage of sexual maturity (Quinn and Deriso 1999). Specifically, mortality rate was estimated as the difference from 1 of the antilog of the slope of the negative linear relationship between the natural log (Ln) of the number of fish caught and age for fish older than the median age in the population.

Sampling Mortality

Mortality due to the sampling method was estimated for Lake Trout. It could not be estimated for Round Whitefish as there was no population estimate for this species.

The sampling (i.e., instantaneous) mortality imposed by the BsM study in 2013 on the fishable (greater than 250 mm fork length) population of Lake Trout in Snap Lake was estimated and then added to the natural mortality estimated from the catch curve, to determine total mortality. Instantaneous mortality for the BsM method was calculated as the percentage (%) the total number of Lake Trout collected with the BsM protocol in 2013, represented of the estimated number of fishable Lake Trout in Snap Lake in 2012, based on a mark recapture study. This was then divided by three to get an annualized value, as the BsM method is only used every three years.

The rates of harvest for the BsM method was calculated as the kilogram of Lake Trout per hectare (kg/ha) by summing the total weight of Lake Trout removed from Snap Lake in 2013 using the BsM protocol and dividing this by the area (ha) of the lake.

8.2.6 Age of Maturity

Age of maturity for fish species was to be calculated. However, due to unequal sample sizes, this was not straightforward. Further consideration of age of maturity will be given in the AEMP Re-evaluation Report in 2016 where data may be pooled across years to provide a more robust calculation.

8.2.7 Water Quality

Water temperature, DO, pH, and conductivity were collected from vertical profiles at each net set location within Snap Lake, Lake 13, and Northeast Lake during July 2013 (Figures 8-1 to 8-3) using a YSI 650

Multiparameter Display System water quality sonde with a YSI 600 Quick Sample multi-parameter water quality probe, as per methods described in Section 3. Measurements were made at one- to two-metre depth intervals throughout the entire water column (Appendix 8A, Table 8A-6). Profiles were started at 0.3 m below the surface and continued every 1.0 to 2.0 m, depending on total depth, until a depth of 0.5 m above the bottom. To obtain an entire profile, the water quality probe was sequentially lowered to the appropriate depth, held in place until the sensor stabilized and measurements were recorded, and then lowered to the next depth until the final profile depth had been reached. At the time of water quality sampling, information on wind direction, intensity, and wave height was also recorded.

8.3 Data Management and Quality Control

Quality assurance and quality control (QA/QC) procedures were implemented to ensure field sampling and laboratory analyses produced valid data. Data were transcribed into electronic databases that were verified prior to analyses.

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 preprinted 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 (COC) forms were used to track the shipment of samples. Field equipment was calibrated before each use throughout the field program as per manufacturer's specifications. Detailed methods are provided below.

Field data forms were reviewed for completeness and accuracy daily by field crew. All data were entered into a Microsoft Office 2010 Excel spreadsheet (Appendix 8B, Table 8B-1). All data were entered into a Microsoft Office 2007 Access relational database. Upon completion of data entry, each data table in the database was QC'd using a series of error checking queries as a secondary level of quality control. Finally, 10% of the Sampling Effort and Fish Biological data were then manually verified against the hard copy data forms as a third level of quality control.

Individual QA/QC procedures were undertaken by each laboratory that performed specialty analyses for the aging and fecundity analysis as follows:

- **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 8C).
- **Fecundity** – One out of every 10 fecundity samples was recounted by a second, independent individual. If the recount of the sample was within 10% of the initial count, the initial count was regarded as acceptable and no recounts of the remaining samples were required. If the recount was not within 10% of the initial count, the initial count was regarded as unacceptable and the remaining nine samples were recounted. This procedure was repeated until recounts were within 10% of the previous count. The results of the fecundity analyses are presented in the fecundity report (Appendix 8D)

The appendices provide the final reported results following internal QA/QC procedures implemented by responsible laboratories and subsequent QA/QC procedures implemented upon receipt of the data. Results were screened visually upon initial receipt and any unusual results (e.g., anomalously high or low relative to the rest of the samples) were flagged and the laboratory was asked to confirm their accuracy.

Data entry review involved checking a minimum of 10% of the entered data for accuracy, data entry errors, transcription errors, and invalid data. Checking was done by a second, independent individual. If an error was found, all data underwent a complete QA check (i.e., every datum checked) by the second independent person.

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 person.

8.4 Results

8.4.1 Fish Catch and Effort

The number of large mesh gill net sets in 2013 by lake was 18 for Snap Lake, 21 for Northeast Lake, and 15 for Lake 13. The total large mesh gill netting efforts corresponded to 318.7 hours for Snap Lake, 501.2 hours for Northeast Lake, and 306.3 hours for Lake 13. The number of net sets for small mesh gill net sets was 13 for Snap Lake, 17 for Northeast Lake, and 13 for Lake 13. The total small mesh gill netting efforts corresponded to 229.4 hours for Snap Lake, 445.4 hours for Northeast Lake, and 275.2 hours for Lake 13. The target minimum number of total net sets per lake was achieved for Snap Lake (n = 31) and Northeast Lake (n = 38), but was one net set short of the target 29 for Lake 13. Catch data and raw effort data are presented in Appendices 8A and 8B-1.

Lake Trout, Round Whitefish, Lake Chub, Longnose Sucker, and Northern Pike were collected in the greatest numbers among the study lakes (Table 8-4), and as such are considered in further detail in the following sections. Arctic Grayling, Cisco, and Burbot were caught infrequently; therefore, it was not possible to discern spatial patterns of relative abundance among the study lakes. Burbot and Arctic Grayling were only captured in Snap Lake and Northeast Lake, while Cisco were only captured in Lake 13 (Table 8-4). Slimy Sculpin (*Cottus cognatus*) were not captured in any of the lakes in 2013.

Relative abundance based on AWCPUEs in Snap Lake was higher than in the reference lakes for all species present in Snap Lake (Table 8-5). The AWCPUE for Lake Trout caught in large mesh gill nets was 2.7 times higher for Snap Lake compared to either Northeast Lake or Lake 13. Similarly, for small mesh gill nets, Lake Trout AWCPUE was 1.5 to 2.8 times higher for Snap Lake compared to the two reference lakes (Table 8-5). The two reference lakes Lake Trout AWCPUEs were similar for large mesh gill nets, but Lake Trout AWCPUE was nearly two times greater for Lake 13 than Northeast Lake. Lake Trout CPUE was greatest in large mesh nets set in the 1 to 3 m and 6 to 12 m depth strata in Snap Lake, while the greatest Lake Trout CPUE in Northeast Lake was in the 12 to 20 m depth strata, and in the 3 to 6 m depth strata in Lake 13 (Table 8-6).

Similar to Lake Trout, AWCPUEs for Round Whitefish in Snap Lake were higher for both large and small mesh nets compared to either Northeast Lake or Lake 13 (Table 8-5). The AWCPUE for Round Whitefish caught in large mesh gill nets was 3.2 times higher for Snap Lake compared to Northeast Lake, and 1.3 times higher compared to Lake 13. Similarly, for small mesh gill nets, Round Whitefish AWCPUE was 2.6 times higher for Snap Lake compared to Northeast Lake, and 1.3 times higher compared to Lake 13.

Snap Lake Round Whitefish AWCPUEs were more than triple Northeast Lake values for large mesh and more than double for small mesh gill nets (Table 8-5). Round Whitefish CPUE was greatest in large mesh nets set in the 3 to 6 m and 6 to 12 m depth strata in Snap Lake, while the greatest Round Whitefish CPUE in Northeast Lake was for small mesh nets in the 3 to 6 m depth strata, and in the 3 to 6 m depth strata in Lake 13 (Table 8-6).

The small mesh AWCPUE for Lake Chub in Snap Lake was 2.24 fish/overnight set and was approximately 4.5 times higher than for Northeast Lake (0.49 fish/overnight set); only one Lake Chub was caught in Lake 13 (Table 8-5).

In large mesh nets, the Longnose Sucker AWCPUE was higher for Snap Lake compared to Northeast Lake (Table 8-5). Longnose Sucker were not captured in Northeast Lake in small mesh nets or in Lake 13 in either size net. Forty-six percent of the total Longnose Sucker catch in Snap Lake occurred in one net set.

The Northern Pike AWCPUE for large mesh nets was higher in Lake 13 compared to Northeast Lake, but the opposite pattern was observed for small mesh nets. Northern Pike are not present in Snap Lake.

Table 8-4 Summary of the Number of Fish Captured, and Catch Rate for Small or Large Mesh Gill Nets from All Lakes Combined During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Net Type	Total Effort (hr)	Species	Total Catch	Fish/hr
Large mesh	1,125.98	LKTR	127	0.113
		RNWH	75	0.067
		NRPK	16	0.014
		LKCH	0	0.000
		LNSC	13	0.012
		ARGR	4	0.004
		BURB	0	0.000
		CISC	0	0.000
Small mesh	949.99	LKTR	49	0.052
		RNWH	78	0.082
		NRPK	6	0.006
		LKCH	51	0.054
		LNSC	37	0.039
		ARGR	2	0.002
		BURB	1	0.001
		CISC	2	0.002

LKTR = Lake Trout, RNWH = Round Whitefish, NRPK = Northern Pike, LNSC = Longnose Sucker, LKCH = Lake Chub, ARGR = Arctic Grayling, BURB = Burbot, CISC = Cisco; hr = hour; AEMP = Aquatics Effects Monitoring Program.

Table 8-5 Summary of the Area-Weighted Catch Per Unit Effort of Fish Species Captured in Large or Small Mesh Nets in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species[a]	Net Type	Lake	AWCPUE[b]
LKTR	Large mesh	Snap Lake	4.32
		Northeast Lake	1.59
		Lake 13	1.56
	Small mesh	Snap Lake	1.67
		Northeast Lake	0.60
		Lake 13	1.13
RNWH	Large mesh	Snap Lake	2.84
		Northeast Lake	0.89
		Lake 13	2.31
	Small mesh	Snap Lake	2.24
		Northeast Lake	0.87
		Lake 13	1.72
NRPK	Large mesh	Snap Lake	n/a
		Northeast Lake	0.58
		Lake 13	0.61
	Small mesh	Snap Lake	n/a
		Northeast Lake	0.21
		Lake 13	0.10
LNSC	Large mesh	Snap Lake	1.36
		Northeast Lake	0.02
		Lake 13	n/a
	Small mesh	Snap Lake	2.80
		Northeast Lake	0.00
		Lake 13	n/a
LKCH	Small mesh	Snap Lake	2.24
		Northeast Lake	0.49
		Lake 13	0.05

(a) Only includes species where at least 10 individuals captured; (b) CPUE weighted by the area of each BsM depth strata and standardized to fish/net type/overnight set.
n/a = Not Applicable; species not present; LKTR = Lake Trout, RNWH = Round Whitefish, NRPK = Northern Pike, LNSC = Longnose Sucker, LKCH = Lake Chub; AWCPUE= area-weighted catch per unit effort; AEMP = Aquatics Effects Monitoring Program.

Table 8-6 Catch Per Unit Effort of Lake Trout and Round Whitefish Captured by Depth Strata Using Small or Large Mesh Gill Nets Set in Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Net Type	Stratum	Lake Trout CPUE[(a)]			Round Whitefish CPUE[(a)]		
		Snap Lake	Northeast Lake	Lake 13	Snap Lake	Northeast Lake	Lake 13
Small mesh	1 to 3 m	1.00	0.30	1.75	2.50	0.50	3.25
	3 to 6 m	2.25	0.70	1.25	3.00	2.20	2.25
	6 to 12 m	1.50	0.63	1.00	1.75	0.38	0.33
	12 to 20 m	2.00	0.67	0.00	1.00	0.67	2.50
Large mesh	1 to 3 m	5.00	1.00	0.50	1.00	0.67	3.00
	3 to 6 m	3.67	1.42	2.67	4.67	1.67	3.67
	6 to 12 m	5.00	1.75	1.50	3.50	0.50	1.50
	12 to 20 m	2.75	2.25	0.80	0.50	1.13	0.00
	20 to 35 m	3.33	1.75	0.00	0.00	0.00	0.00
	35 to 50 m	2.50	n/a	n/a	0.00	n/a	n/a

(a) CPUE calculated as average net catch per depth strata; CPUE = catch per unit effort; n/a = not applicable; m = metre; AEMP = Aquatics Effects Monitoring Program.

8.4.2 Community Composition

Five fish species were collected during the July 2013 fish community monitoring program from Snap Lake and Lake 13, and seven species were collected from Northeast Lake (Table 8-7). Lake Trout, Round Whitefish, and Lake Chub were captured in all lakes. Longnose Sucker and Arctic Grayling were only captured in Snap Lake and Northeast Lake, while Northern Pike were only captured in Northeast Lake and Lake 13, and Burbot were only captured in Northeast Lake. Cisco were captured in Lake 13 for the first time in 2013.

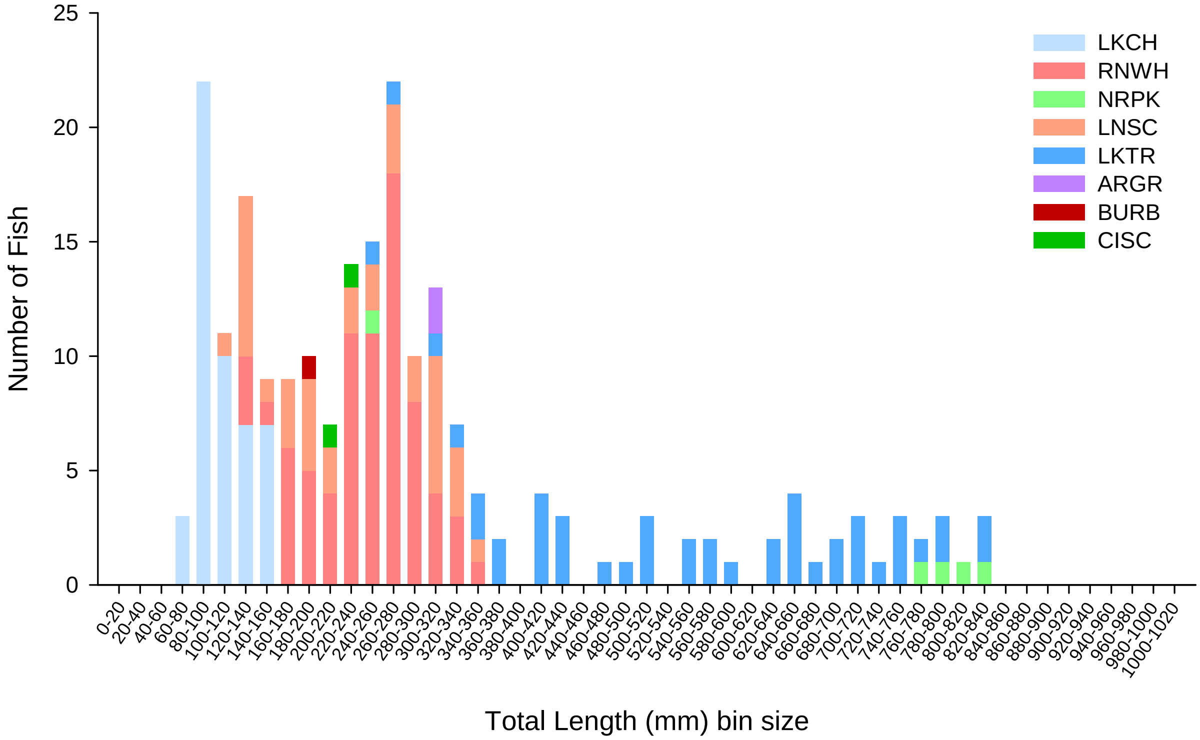
With the exception of Lake Chub, Burbot, and Cisco, there was considerable overlap among species and size ranges within species of fish captured by either small or large mesh nets in the AEMP lakes (Figures 8-4 and 8-5). Most fish captured in small mesh nets ranged from 60 to 380 mm with two possible groupings evident (80 to 220 mm and 220 to 380 mm). Catches in large mesh nets had a bimodal distribution with most fish in the 200 to 380 mm and 600 to 900 mm size ranges. Although small mesh nets captured few small Round Whitefish, there was generally no bias in Round Whitefish size captured between either net type. Lake Trout size ranges were similar (small mesh = 240 to 840 mm; large mesh = 220 to 980 mm), but large mesh nets were more effective at capturing greater numbers of Lake Trout, especially in the larger size ranges.

Table 8-7 Community Composition, Number of Fish Caught in Large or Small Mesh Nets, and their Frequency of Occurrence Separated by Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Lake	Species	Small Mesh		Large Mesh		All Meshes	
		n	%	N	%	n	%
Snap Lake	LKTR[(a)]	21	16.9	67	59.8	88	37.3
	RNWH[(a)]	30	24.2	32	28.6	62	26.3
	NRPK	0	0.0	0	0.0	0	0.0
	LKCH[(a)]	34	27.4	0	0.0	34	14.4
	LNSC[(a)]	37	29.8	13	11.6	50	21.2
	ARGR[(a)]	2	1.6	0	0.0	2	0.8
	BURB[(a)]	0	0.0	0	0.0	0	0.0
	CISC	0	0.0	0	0.0	0	0.0
	NNST	0	0.0	0	0.0	0	0.0
	SLSC[(a)]	0	0.0	0	0.0	0	0.0
	All Species	**124**	**100**	**112**	**100**	**236**	**100**
Northeast Lake	LKTR	13	24.1	41	55.4	54	42.2
	RNWH	20	37.0	20	27.0	40	31.3
	NRPK	4	7.4	10	13.5	14	10.9
	LKCH	16	29.6	0	0.0	16	12.5
	LNSC	0	0.0	1	1.4	1	0.8
	ARGR	0	0.0	2	2.7	2	1.6
	BURB	1	1.9	0	0.0	1	0.8
	CISC	0	0.0	0	0.0	0	0.0
	NNST	0	0.0	0	0.0	0	0.0
	SLSC	0	0.0	0	0.0	0	0.0
	All Species	**54**	**100**	**74**	**100**	**128**	**100**
Lake 13	LKTR	15	31.3	19	39.6	34	35.4
	RNWH	28	58.3	23	47.9	51	53.1
	NRPK	2	4.2	6	12.5	8	8.3
	LKCH	1	2.1	0	0.0	1	1.0
	LNSC	0	0.0	0	0.0	0	0.0
	ARGR	0	0.0	0	0.0	0	0.0
	BURB	0	0.0	0	0.0	0	0.0
	CISC	2	4.2	0	0.0	2	2.1
	NNST	0	0.0	0	0.0	0	0.0
	SLSC	0	0.0	0	0.0	0	0.0
	All Species	**48**	**100**	**48**	**100**	**96**	**100**

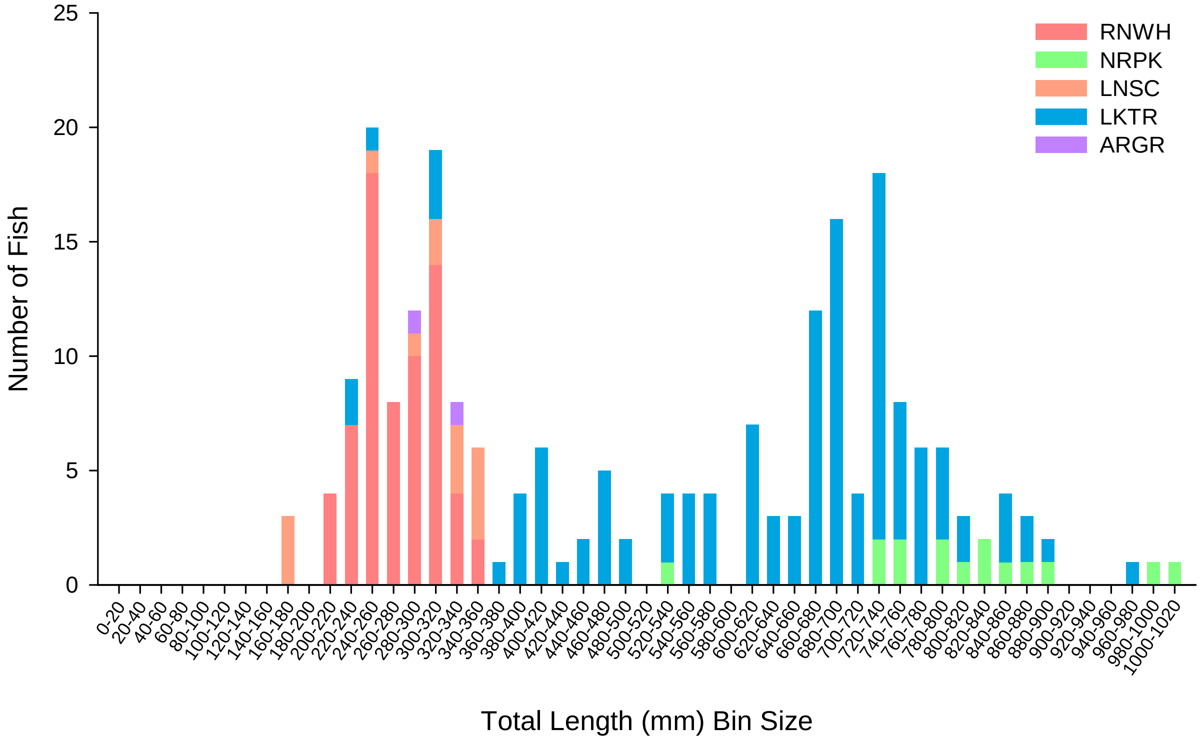
(a) Species that have been historically collected in Snap Lake.
LKTR = Lake Trout, RNWH = Round Whitefish, NRPK = Northern Pike, LNSC = Longnose Sucker, LKCH = Lake Chub, ARGR = Arctic Grayling, BURB = Burbot, CISC = Cisco, NNST = Ninespine Stickleback, SLSC = Slimy Sculpin; n = number of samples; %=percent; AEMP = Aquatics Effects Monitoring Program.

Figure 8-4 Length Frequency Distributions of Fish Species Captured Across All Lakes in Small Mesh Gill Nets During the July 2013 Snap Lake AEMP Fish Community Monitoring Program



mm = millimetre; LKCH = Lake Chub; RNWH = Round Whitefish; NRPK = Northern Pike; LNSC = Longnose Sucker; LKTR = Lake Trout; ARGR = Arctic Grayling; BURB = Burbot; CISC = Cisco; AEMP = Aquatics Effects Monitoring Program.

Figure 8-5 Length Frequency Distributions of Species Captured Across All Lakes in Large Mesh Gill Nets During the July 2013 Snap Lake AEMP Fish Community Monitoring Program



mm = millimetre. RNWH = Round Whitefish; NRPK = Northern Pike; LNSC = Longnose Sucker; LKTR = Lake Trout; ARGR = Arctic Grayling; AEMP = Aquatics Effects Monitoring Program

8.4.3 Biological Characteristics

Summary statistics for length, weight, and age for Lake Trout and Round Whitefish are presented in Table 8-8. Biological data of other species caught during the 2013 fish community monitoring program are presented in Appendix 8B, Table 8B-1. Summary statistics for all species are presented in Appendix 8G, Table 8G-1. Summary statistics for condition are presented in Section 8.4.5.

Lake Trout

There were relatively few significant differences in Lake Trout age, length, or weight among lakes (Table 8-9); of those present, most involved differences between Snap Lake and Lake 13. Lake Trout age did not differ significantly among lakes for all fish, or when analyzed separately by sex. When all Lake Trout were tested, Lake Trout from Snap Lake were significantly shorter (P = 0.017), but were not heavier than Lake Trout from Lake 13, and were not different in length or weight compared to Lake Trout from Northeast Lake (Table 8-9). Male Lake Trout from Snap Lake did not differ from Lake Trout from Lake 13 or Northeast Lake for length or weight. Female Lake Trout did not differ from female Lake Trout from Lake 13 in length or weight, but were significantly shorter (*P* = 0.090) and lighter (*P* = 0.019) from Snap Lake than from Northeast Lake (Table 8-9).

Round Whitefish

Round Whitefish from Snap Lake were significantly longer, heavier, and older than Round Whitefish from Lake 13 and Northeast Lake when all fish or males only were considered (Table 8-9). Female Round Whitefish from Snap Lake were not significantly different than females from the reference lakes. Length and weight of Round Whitefish from Northeast Lake and Lake 13 did not differ, whether all fish were combined or analyzed separately by sex. Male Round Whitefish from Lake 13 were significantly longer (*P* = 0.057) than those from Northeast Lake. When all Round Whitefish were combined, those from Northeast Lake were older (*P* = 0.051) than those from Lake 13 (Table 8-9).

Table 8-8 Summary of Fork Length, Total Length, Body Weight, and Age for Lake Trout Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Group	Variable	Units	Lake	N	Median	Mean	SD	SE	Minimum	Maximum
LKTR	All	Fork length	mm	Lake 13	32	613	588	107	19	342	770
				Northeast Lake	54	602	546	174	24	189	870
				Snap Lake	87	553	525	135	15	183	801
		Total length	mm	Lake 13	32	670	648	118	21	375	828
				Northeast Lake	54	664	601	188	26	218	942
				Snap Lake	84	599	569	143	16	205	875
		Body weight	g	Lake 13	32	2,570	2,393	995	176	400	4,360
				Northeast Lake	54	2,500	2,398	1,610	219	76	7,400
				Snap Lake	85	2,080	1,997	1,404	152	67	7,420
		Age	years	Lake 13	33	19	20	10	1.7	6	43
				Northeast Lake	54	20	20	10	1.4	4	43
				Snap Lake	76	14	17	9	1.0	5	40
	Male	Fork length	mm	Lake 13	21	605	577	118	26	342	770
				Northeast Lake	26	626	595	143	28	260	870
				Snap Lake	45	578	545	137	20	281	801
		Total length	mm	Lake 13	21	666	633	128	28	375	828
				Northeast Lake	26	683	656	156	31	290	942
				Snap Lake	45	623	598	148	22	318	875
		Body weight	g	Lake 13	21	2,510	2,279	1,049	229	400	4,360
				Northeast Lake	26	2,845	2,824	1,587	311	180	7,400
				Snap Lake	45	2,340	2,351	1,624	242	240	7,420
		Age	years	Lake 13	21	18	18	8	1.7	6	35
				Northeast Lake	26	21	23	10	1.9	5	41
				Snap Lake	45	15	19	9	1.4	5	40
	Female	Fork length	mm	Lake 13	4	625	585	100	50	437	652
				Northeast Lake	17	645	588	142	34	304	798
				Snap Lake	25	537	509	98	20	340	644
		Total length	mm	Lake 13	4	688	647	110	55	486	727
				Northeast Lake	17	708	645	150	36	342	852
				Snap Lake	25	597	559	108	22	377	704
		Body weight	g	Lake 13	4	2,580	2,288	847	424	1,050	2,940
				Northeast Lake	17	3,260	2,681	1,419	344	300	5,140
				Snap Lake	25	2,080	1,694	882	176	470	3,150
		Age	years	Lake 13	4	26	24	12	6.2	7	37
				Northeast Lake	17	21	21	10	2.4	7	43
				Snap Lake	25	15	17	7	1.5	8	32
RNWH	All	Fork length	mm	Lake 13	48	215	212	41	6	100	299
				Northeast Lake	33	204	209	47	8	130	290
				Snap Lake	61	234	230	36	5	153	299
		Total length	mm	Lake 13	48	234	232	46	7	109	336
				Northeast Lake	33	223	228	51	9	143	316
				Snap Lake	61	258	252	38	5	168	326
		Body weight	g	Lake 13	48	87	97	54	8	8	300
				Northeast Lake	33	77	97	64	11	20	240
				Snap Lake	60	130	137	60	8	33	270
		Age	years	Lake 13	50	3	3	2	0.2	1	8
				Northeast Lake	33	4	4	2	0.3	2	8
				Snap Lake	57	5	5	2	0.3	2	10
	Male	Fork length	mm	Lake 13	21	214	218	16	3	194	251
				Northeast Lake	14	202	203	36	10	135	279
				Snap Lake	25	238	237	35	7	171	299
		Total length	mm	Lake 13	21	234	238	19	4	211	275
				Northeast Lake	14	221	221	39	10	147	303
				Snap Lake	25	260	260	37	7	196	326
		Body weight	g	Lake 13	21	87	96	29	6	64	170
				Northeast Lake	14	76	83	42	11	24	190
				Snap Lake	25	130	148	62	12	61	270
		Age	years	Lake 13	23	3	3	1	0.2	2	6
				Northeast Lake	14	3	4	2	0.4	2	8
				Snap Lake	24	4	5	2	0.4	3	10
	Female	Fork length	mm	Lake 13	17	236	238	28	7	189	299
				Northeast Lake	11	261	252	30	9	192	290
				Snap Lake	30	238	235	29	5	178	293
		Total length	mm	Lake 13	17	256	260	32	8	208	336
				Northeast Lake	11	284	275	32	10	210	316
				Snap Lake	30	261	258	31	6	196	320
		Body weight	g	Lake 13	17	116	133	60	15	73	300
				Northeast Lake	11	170	159	57	17	68	240
				Snap Lake	30	140	140	53	10	50	240
		Age	years	Lake 13	17	4	4	2	0.4	3	8
				Northeast Lake	11	5	5	2	0.6	3	8
				Snap Lake	28	5	6	2	0.4	2	9

LKTR = Lake Trout; RNWH = Round Whitefish; N= number of samples; mm = millimetre; g = gram; SD = standard deviation; SE = standard error; AEMP = Aquatics Effects Monitoring Program.

Table 8-9 Results of Statistical Comparisons Among Lakes for Fork Length, Total Length, Body Weight, and Age Either Combined or by Sex of Lake Trout and Round Whitefish Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Survey

Species	Group	Biological Metric	Analysis Type	Interaction Statistic		Intercept Statistic		SSD	Pairwise Comparison (*P*-value)					
				$Fstat_{(\alpha, df1, df2)}$	*P*-value	$Fstat_{(\alpha, df, df2)}$	*P*-value		LK13 vs NEL	Direction of Difference	LK13 vs SL	Direction of Difference	SL vs NEL	Direction of Difference
LKTR	All	Fork length	K-W	n/a	n/a	n/a	0.0601	Y	0.5114	LK13 = NEL	0.0168	SL <LK13	0.1714	SL = NEL
		Body weight	K-W	n/a	n/a	n/a	0.1151	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
		Age	$ANOVA^{log}$	n/a	n/a	$1.5184_{(0.1, 2, 160)}$	0.2222	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
	Male	Fork length	K-W	n/a	n/a	n/a	0.3462	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
		Body weight	ANOVA	n/a	n/a	$1.5460_{(0.1, 2, 88)}$	0.2188	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
		Age	$ANOVA^{log}$	n/a	n/a	$1.9268_{(0.1, 2, 89)}$	0.1516	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
	Female	Fork length	ANOVA	n/a	n/a	$2.5825_{(0.1, 2, 43)}$	0.0873	Y	0.9983	LK13 = NEL	0.4584	SL = LK13	0.0902	SL <NEL
		Body weight	ANOVA	n/a	n/a	$4.0549_{(0.1, 2, 43)}$	0.0244	Y	0.8002	LK13 = NEL	0.5853	SL = LK13	0.0190	SL <NEL
		Age	$ANOVA^{log}$	n/a	n/a	$1.4893_{(0.1, 2, 43)}$	0.2369	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
RNWH	All	Fork length	K-W	n/a	n/a	n/a	0.0276	Y	0.3949	LK13 = NEL	0.0210	SL >LK13	0.0387	SL >NEL
		Body weight	ANOVA	n/a	n/a	$9.5656_{(0.1, 2, 137)}$	0.0001	Y	0.9242	LK13 = NEL	0.0002	SL >LK13	0.0039	SL >NEL
		Age	K-W	n/a	n/a	n/a	0.0000	Y	0.0511	LK13 <NEL	0.0000	SL >LK13	0.0340	SL >NEL
	Male	Fork length	K-W	n/a	n/a	n/a	0.0092	Y	0.0569	LK13 >NEL	0.0351	SL >LK13	0.0113	SL >NEL
		Body weight	K-W	n/a	n/a	n/a	0.0015	Y	0.1620	LK13 = NEL	0.0049	SL >LK13	0.0027	SL >NEL
		Age	K-W	n/a	n/a	n/a	0.0030	Y	0.7251	LK13 = NEL	0.0008	SL >LK13	0.0323	SL >NEL
	Female	Fork length	ANOVA	n/a	n/a	$1.3299_{(0.1, 2, 55)}$	0.2729	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
		Body weight	$ANOVA^{log}$	n/a	n/a	$0.7305_{(0.1, 2, 55)}$	0.4863	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
		Age	K-W	n/a	n/a	n/a	0.1016	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL

ANOVA = Analysis of Variance; $ANOVA^{log}$ = Analysis of Variance performed on log data; K-W = Kruskal-Wallis; SSD = statistically significant difference (α= 0.1); n/a = not applicable; LK13 = Lake 13; SL = Snap Lake; NEL = Northeast Lake; < = less than; > = greater than; AEMP = Aquatics Effects Monitoring Program.

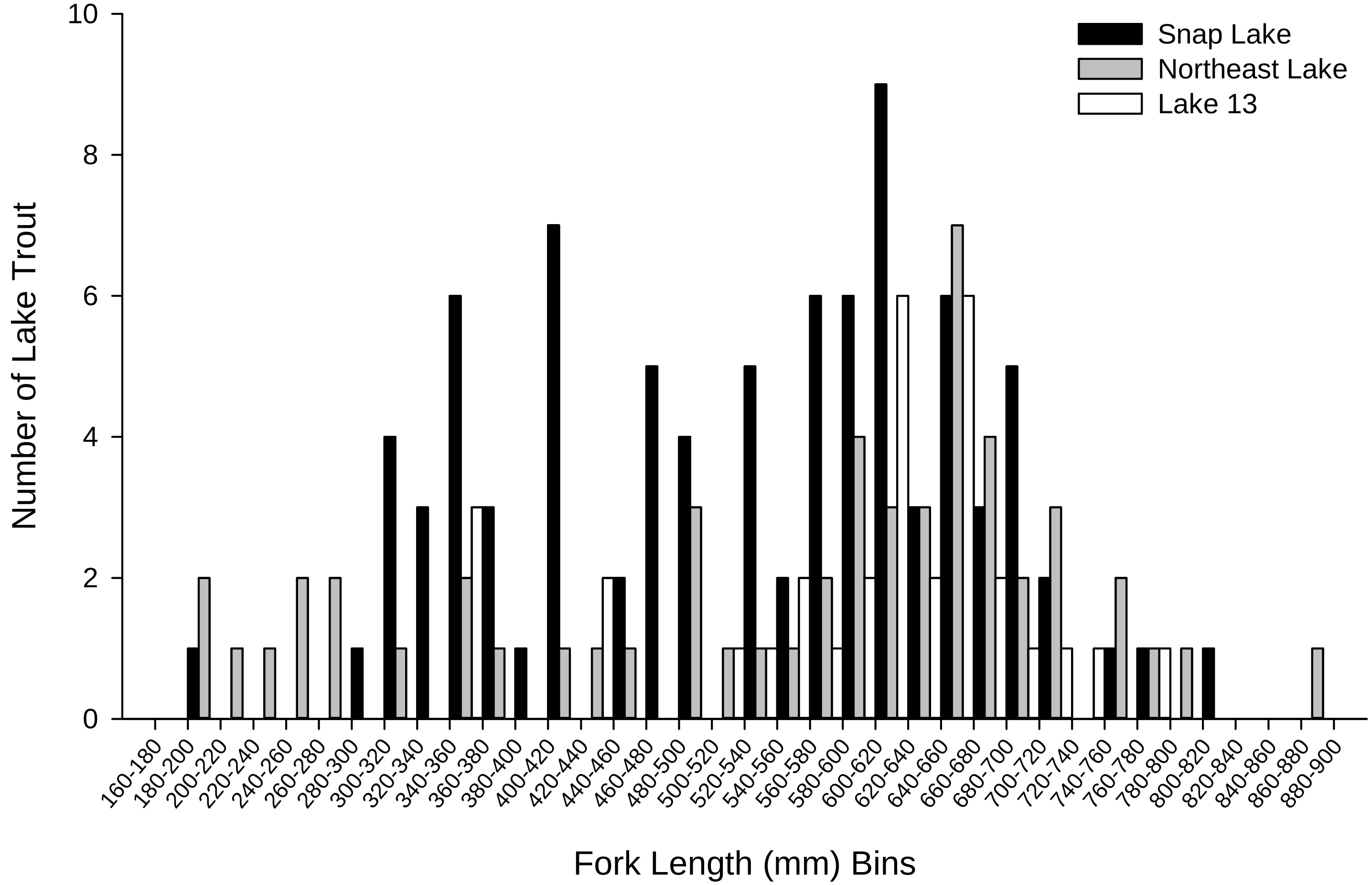
8.4.4 Length and Age Structure

Lake Trout

Length frequency distributions for all Lake Trout combined were significantly different between Snap Lake and Northeast Lake ($P = 0.096$) and Lake 13 ($P = 0.014$) (Figure 8-6; Table 8-10). Female Lake Trout length frequency distribution for Snap Lake was significantly different from Northeast Lake (P = 0.002), but not Lake 13 (Table 8-10). A large portion of fish in Snap Lake fell into one of two length modes between 300 and 500 mm or between 520 and 720 mm (Figure 8-6). The majority of fish from Northeast Lake and Lake 13 fell into a single mode between 560 to 700 mm (Figure 8-6). Lake Trout smaller than 180 mm fork length were not captured in any of the three study lakes.

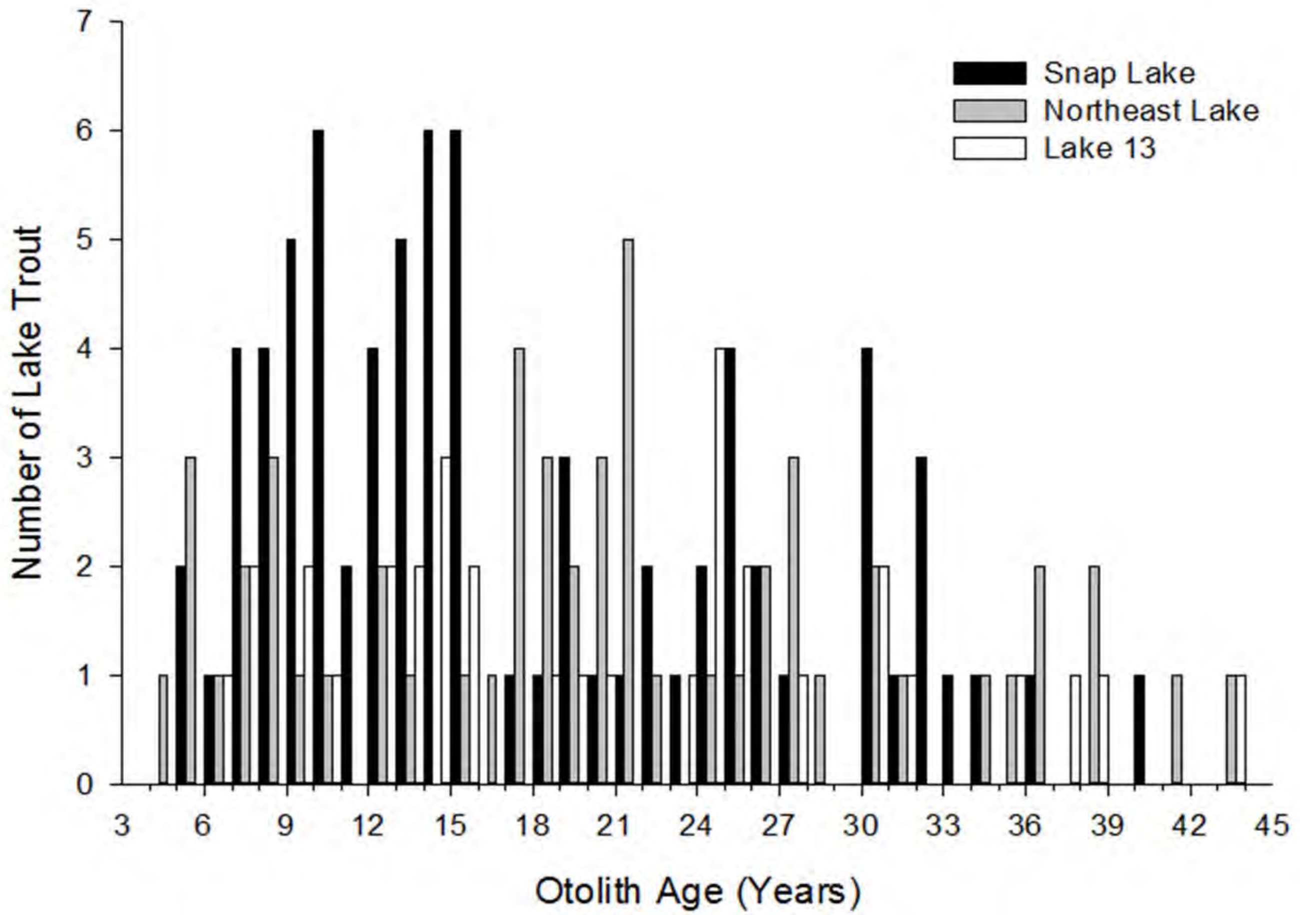
Age frequencies of Lake Trout were not significantly different between Lake 13 and Northeast Lakes nor between Lake 13 and Snap Lake, regardless of whether all fish were combined or analyzed separately by sex (Figure 8-7; Table 8-10). Northeast Lake and Snap Lake had statistically different age frequencies for all fish combined (P = 0.009) and males only (P = 0.025) (Table 8-10). Lake Trout from Snap Lake were younger than Lake Trout from Northeast Lake; the majority of Lake Trout in Snap Lake were between age 7 and age 16, with strong age 26, 31, and 33 age classes, while Lake Trout from Northeast Lake were between age 18 and age 28 (Figure 8-7). No Lake Trout between age 1 and 3 were caught in any of the lakes.

Figure 8-6 Fork Length Frequencies of Lake Trout Captured in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program



mm = millimetre; AEMP = Aquatics Effects Monitoring Program.

Figure 8-7 Age Frequencies of Lake Trout Captured in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program



AEMP = Aquatics Effects Monitoring Program

Table 8-10 Summary Kolmogorov-Smirnov Pair-wise Tests of Homogeneity of Total Length and Age Frequency of Lake Trout and Round Whitefish Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Group	Distribution	*P*-Value for K-S Test Results Comparisons		
			LK13 vs NEL	LK13 vs SL	NEL vs SL
LKTR	All	Length frequency	0.379	0.014	0.096
		Age frequency	0.642	0.506	0.009
	Males	Length frequency	0.734	0.432	0.337
		Age frequency	0.162	0.965	0.025
	Females	Length frequency	0.562	0.159	0.002
		Age frequency	0.603	0.407	0.143
RNWH	All	Length frequency	0.134	0.033	0.023
		Age frequency	0.304	0.001	0.220
	Males	Length frequency	0.052	0.021	0.036
		Age frequency	0.998	0.006	0.188
	Females	Length frequency	0.308	0.984	0.136
		Age frequency	0.393	0.041	1.000

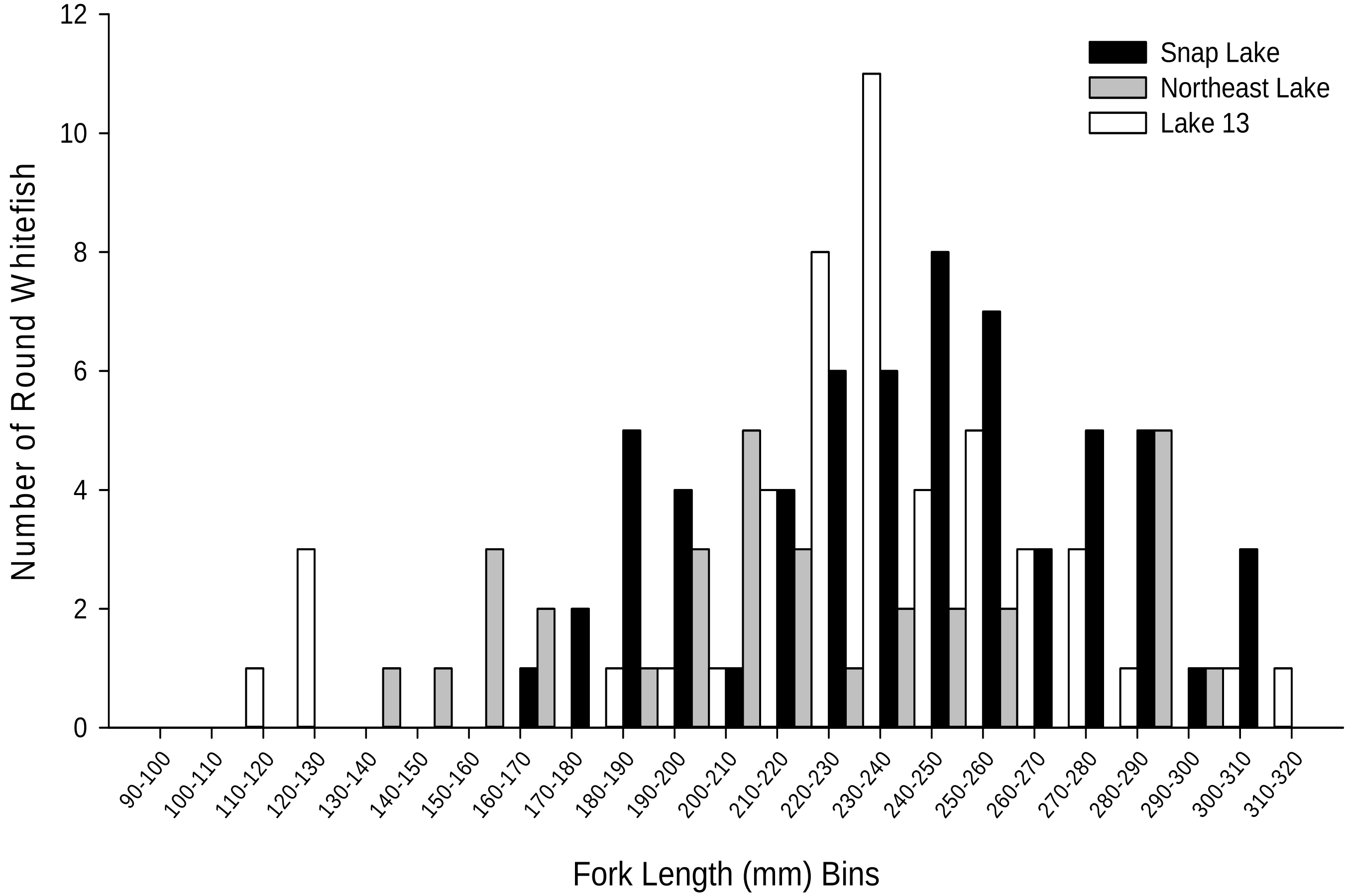
LKTR = Lake Trout; RNWH = Round Whitefish; LK13 = Lake 13; SL = Snap Lake; NEL = Northeast Lake; *P* = probability; K-S= Kolmogorov-Smirnov; AEMP = Aquatics Effects Monitoring Program.

Round Whitefish

For Round Whitefish, when all fish were combined, length frequencies were significantly different between Snap Lake and Lake 13 (*P* = 0.033) or Northeast Lake (*P* = 0.023) (Figure 8-8; Table 8-10). Lake 13 and Northeast Lake Round Whitefish length frequencies did not differ. When sexes were analyzed separately, male length frequency differed among all lakes; Snap Lake was significantly different from Lake 13 (P = 0.033), and Northeast Lake (P = 0.036), and Lake 13 was significantly different from Northeast Lake (*P* = 0.052) (Table 8-10). Round Whitefish female length frequency did not differ among lakes. Round Whitefish caught in Snap Lake had a narrower range of lengths (153 to 299 mm) compared to Northeast Lake (130 to 290 mm) and Lake 13 (100 to 299 mm) (Figure 8-8).

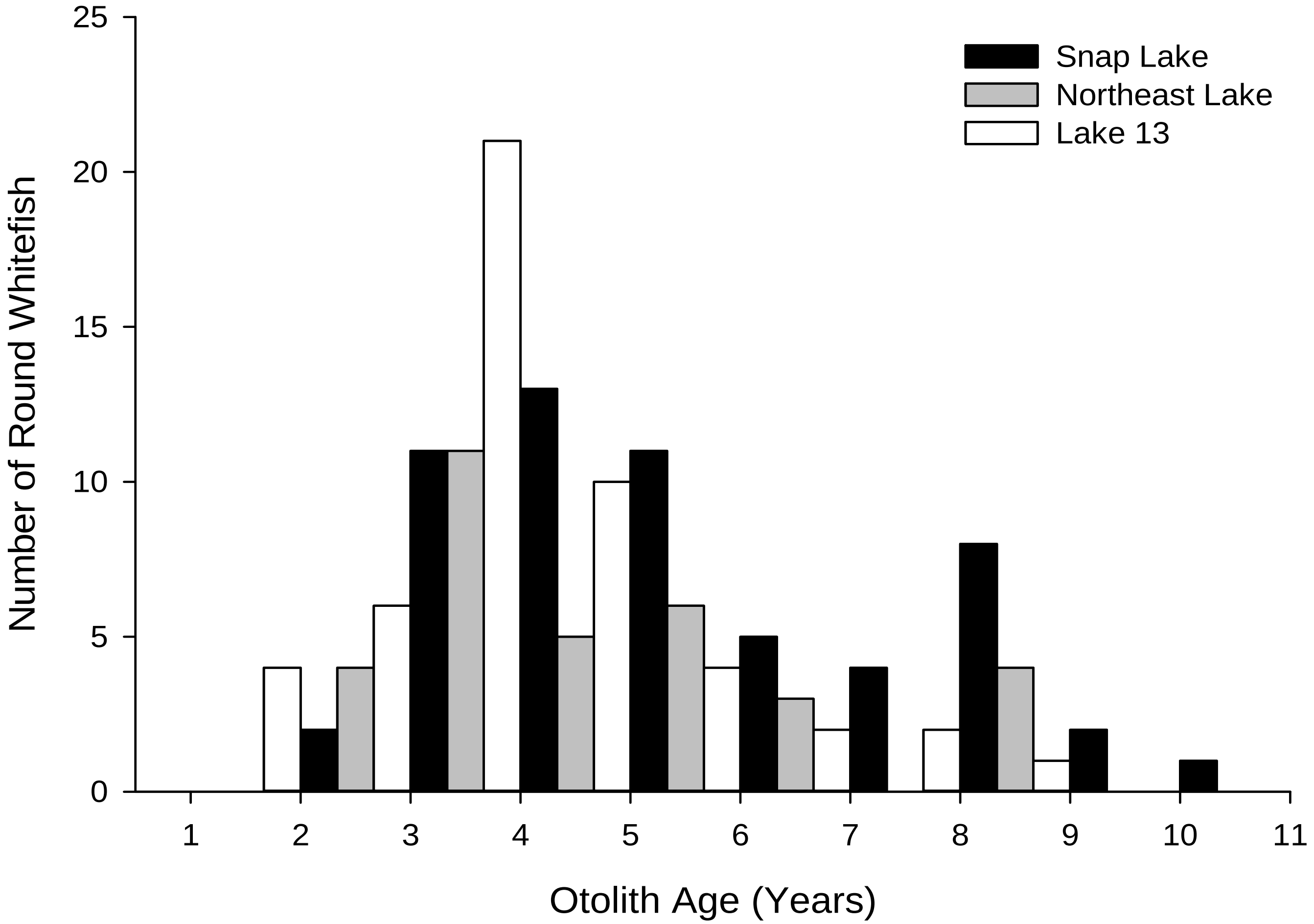
The age frequencies of Round Whitefish from Snap Lake differed from Lake 13 regardless of whether all fish were combined (P = 0.001) or analyzed separately by males (P = 0.006) or females (P = 0.041) (Figure 8-9, Table 8-10). The majority of Snap Lake Round Whitefish ages were evenly distributed across age 3, 4, 5, and 8; however, most Round Whitefish from Lake 13 were either age 3 or age 4, and most fish from Northeast Lake were age 3.

Figure 8-8 **Fork Length Frequencies of Round Whitefish Captured in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program**



mm = millimetre; AEMP = Aquatics Effects Monitoring Program.

Figure 8-9 Age Frequencies of Round Whitefish Captured in Snap Lake, Northeast Lake, or Lake 13 During the July 2013 Snap Lake AEMP Fish Community Monitoring Program



AEMP = Aquatics Effects Monitoring Program

8.4.4.1 Year Class Strength

As noted above, for Lake Trout length frequency was different between Snap Lake and reference lakes but age frequency was not (Figure 8-7; Table 8-10). For Round Whitefish, length frequencies were significantly different between Snap Lake and Lake 13 or Northeast Lake (Figure 8-8; Table 8-10). The age frequencies of Round Whitefish from Snap Lake differed from Lake 13 (Figure 8-9, Table 8-10).

Analysis of year class strength was difficult because of the small within age class sample size relative to the number of age classes present in the lakes (Table 8-11 and Table 8-12). In other words, there was a broad age demographic (e.g., 2 to 45 years for Lake Trout) with very few individuals within each age class (Table 8-11).

Table 8-11 Age Structure by Number and Percentage of Individuals in each Age Class for Lake Trout in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Age	Snap Lake		Northeast Lake		Lake 13	
	Number of Individuals	%	Number of Individuals	%	Number of Individuals	%
2	1	1.14	1	1.85	—	—
3	7	7.95	4	7.41	6	17.65
4	3	3.41	3	5.56	5	14.71
5	3	3.41	2	3.70	—	—
6	3	3.41	4	7.41	1	2.94
7	5	5.68	2	3.70	3	8.82
8	4	4.55	7	12.96	—	—
9	4	4.55	2	3.70	3	8.82
10	4	4.55	3	5.56	1	2.94
11	2	2.27	—	—	—	—
12	1	1.14	1	1.85	—	—
13	7	7.95	—	—	1	2.94
14	6	6.82	—	—	1	2.94
15	3	3.41	1	1.85	2	5.88
16	—	—	1	1.85	—	—
17	2	2.27	2	3.70	—	—
18	1	1.14	—	—	1	2.94
19	1	1.14	1	1.85	1	2.94
20	1	1.14	3	5.56	—	—
21	1	1.14	2	3.70	—	—
22	2	2.27	—	—	—	—
23	1	1.14	—	—	1	2.94
24	2	2.27	—	—	1	2.94
25	3	3.41	1	1.85	1	2.94
27	1	1.14	2	3.70	—	—
28	—	—	1	1.85	—	—

Table 8-11 Age Structure by Number and Percentage of Individuals in each Age Class for Lake Trout in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program

	Snap Lake		Northeast Lake		Lake 13	
Age	Number of Individuals	%	Number of Individuals	%	Number of Individuals	%
30	3	3.41	—	—	1	2.94
31	1	1.14	—	—	—	—
32	1	1.14	—	—	—	—
33	1	1.14	—	—	—	—
34	1	1.14	1	1.85	—	—
35	—	—	1	1.85	—	—
36	1	1.14	1	1.85	—	—
38	—	—	1	1.85	1	2.94
43	—	—	1	1.85	—	—
Total	**88**		**54**		**34**	

— = No fish captured in that age group; % = percent; AEMP = Aquatics Effects Monitoring Program.

Table 8-12 Age Structure by Number and Percentage of Individuals in each Age Class for Round Whitefish in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program

	Snap Lake		Northeast Lake		Lake 13	
Age	Number of Individuals	%	Number of Individuals	%	Number of Individuals	%
1	—	—	—	—	5	9.80
2	3	4.84	3	7.50	6	11.76
3	4	6.45	7	17.50	12	23.53
4	9	14.52	3	7.50	5	9.80
5	4	6.45	7	17.50	3	5.88
6	2	3.23	—	—	2	3.92
7	1	1.61	—	—	2	3.92
8	7	11.29	2	5.00	1	1.96
9	5	8.06	1	2.50	—	—
10	1	1.61	—	—	—	—
11	1	1.61	1	2.50	—	—
12	5	8.06	1	2.50	2	3.92
13	1	1.61	1	2.50	2	3.92
14	—	—	—	—	1	1.96
15	4	6.45	—	—	1	1.96
17	—	—	1	2.50	—	—
18	—	—	2	5.00	—	—
19	1	1.61	—	—	—	—
20	—	—	—	—	1	1.96
21	—	—	2	5.00	—	—
22	—	—	1	2.50	—	—

Table 8-12 Age Structure by Number and Percentage of Individuals in each Age Class for Round Whitefish in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Age	Snap Lake		Northeast Lake		Lake 13	
	Number of Individuals	%	Number of Individuals	%	Number of Individuals	%
24	—	—	—	—	3	5.88
25	—	—	—	—	1	1.96
26	1	1.61	1	2.50	—	—
27	—	—	—	—	1	1.96
32	1	1.61	—	—	—	—
35	—	—	—	—	1	1.96
36	—	—	1	2.50	—	—
37	—	—	—	—	1	1.96
38	—	—	1	2.50	—	—
40	1	1.61	—	—	—	—
43	—	—	—	—	1	1.96
Total	**62**		**40**		**51**	

% = percent; — = No fish captured in that age group; AEMP = Aquatics Effects Monitoring Program.

8.4.5 Condition and Growth

Lake Trout

Fulton's condition was significantly greater for Lake Trout from Snap Lake than Lake 13, but was not different from Lake Trout from Northeast Lake when all fish were combined (Table 8-13 and 8-14). Fulton's condition for all fish combined was significantly greater for Lake Trout from Northeast Lake than Lake Trout from Lake 13 (Table 8-14). When fish were separated by sex and analyzed separately, the same statistical differences were present for male Lake Trout as were present for all fish combined (Table 8-14). There were no differences in Fulton's condition for female Lake Trout among Snap Lake, Lake 13, or Northeast Lake. In summary, Lake Trout generally had greater condition in Snap Lake and Northeast Lake, except among females where condition was similar across study lakes, with a limited sample size of four female fish from Lake 13.

The Lake Trout weight-length relationship for Snap Lake indicated allometric growth (plumber with increased length) is occurring (Table 8-15). The regression relationships were highly significant for all Lake Trout pooled or separated by sex. All Lake Trout from Snap Lake and Northeast Lake, and males only from Lake 13 became more rotund with increasing length (i.e., $\beta > 3$). Lake 13 females and pooled Lake Trout, in contrast, became less rotund or more slender with increasing length (i.e., $\beta < 3$) but the sample size of females was very small (n=4). In summary, growth rate of Lake Trout from Snap Lake and Northeast Lake were similar to each other, and fish were becoming more rotund with length. In Lake 13, however, fish were becoming less rotund with length. These differences between lakes were not substantive as the growth coefficient (k), length at infinity (L_∞) and theoretical length at age 0 (t_0) showed broad overlap across lakes based on 95% confidence intervals.

Growth curves based on Lake 13 data were found to be significantly different from both Northeast Lake and Snap Lake (Table 8-16); therefore, a pooled model was not possible. A single von Bertalanffy curve was fit to the combined Northeast Lake and Snap Lake data, and Lake 13 data were used to fit a separate growth curve (Figure 8-10, Table 8-16 and 8-17). These results indicate that based on the data used in this analysis, the growth curves from Northeast Lake and Snap Lake are not distinguishable and are best described by a single model; whereas, the growth curve from Lake 13 is different from Northeast and Snap Lake.

Round Whitefish

For Round Whitefish, Fulton's condition was significantly greater for Snap Lake compared to Lake 13 when all fish were combined or analyzed separately for males or females (Tables 8-13 and 8-14). Snap Lake Round Whitefish also had a greater Fulton's condition compared to Round Whitefish from Northeast Lake when all fish were combined, or analyzed separately for males, but not females (Table 8-14). There was no difference in Fulton's condition between reference lakes whether all fish were combined or separated by sex and analyzed separately. In summary, Snap Lake Round Whitefish had higher condition than Round Whitefish from either Lake 13 or Northeast Lake.

The Round Whitefish weight-length relationship in Snap Lake indicates allometric growth is occurring in all lakes (Table 8-15). The regression relationships were highly significant for all Round Whitefish pooled or separated by sex. All Round Whitefish from Northeast Lake and Lake 13 are becoming more rotund with increasing length (i.e., $\beta > 3$). Snap Lake male Round Whitefish, however, are becoming less rotund with increasing length (i.e., $\beta < 3$). In summary, although growth rate of Round Whitefish differed among lakes, these differences were not substantive in that the growth coefficient (k), length at infinity (L_∞) and theoretical length at age 0 (t_0) showed broad overlap across lakes based on 95% confidence intervals.

Growth curves based on Snap Lake data were not significantly different from those based on Northeast Lake or Lake 13 data (Table 8-18). However, the growth curve based on Lake 13 was significantly different than the Northeast Lake growth curve; therefore, data from the three lakes could not be pooled into a single model (Figure 8-11, Table 8-17 and 8-18). This indicates that, based on the data currently available for this assessment, growth rates are measurably different among the three lakes requiring separate growth models for each lake.[5]

[5] The use of separate models for Round Whitefish is justified, despite a lower difference in Akaike information criterion (ΔAICc) (difference in AICc values in comparison to the full, individual-lake model) value for the model of Snap Lake and Northeast Lake grouped, because the two reference lakes are significantly different from each other.

Table 8-13 Summary of Fulton's Condition for Lake Trout and Round Whitefish Captured in Snap Lake, Northeast Lake, and Lake 13 During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Sex	Lake	N	Median	Mean	SD	SE	Minimum	Maximum
LKTR	All	Lake 13	32	1.09	1.09	0.13	0.02	0.86	1.37
		Northeast Lake	54	1.16	1.14	0.13	0.02	0.81	1.47
		Snap Lake	85	1.17	1.17	0.13	0.01	0.73	1.46
	Male	Lake 13	21	1.05	1.08	0.13	0.03	0.89	1.37
		Northeast Lake	26	1.16	1.16	0.12	0.02	0.97	1.47
		Snap Lake	45	1.20	1.20	0.12	0.02	0.94	1.46
	Females	Lake 13	4	1.08	1.11	0.11	0.05	1.02	1.26
		Northeast Lake	17	1.18	1.17	0.13	0.03	0.83	1.35
		Snap Lake	25	1.17	1.15	0.12	0.02	0.95	1.42
RNWH	All	Lake 13	48	0.89	0.90	0.10	0.02	0.69	1.12
		Northeast Lake	33	0.92	0.91	0.10	0.02	0.65	1.06
		Snap Lake	60	1.03	1.03	0.09	0.01	0.86	1.37
	Male	Lake 13	21	0.89	0.90	0.09	0.02	0.73	1.11
		Northeast Lake	14	0.94	0.92	0.10	0.03	0.65	1.06
		Snap Lake	25	1.05	1.06	0.09	0.02	0.90	1.22
	Females	Lake 13	17	0.95	0.94	0.11	0.03	0.72	1.12
		Northeast Lake	11	0.96	0.95	0.06	0.02	0.83	1.03
		Snap Lake	30	1.03	1.02	0.09	0.02	0.86	1.37

LKTR = Lake Trout; RNWH = Round Whitefish; N = number of samples; SD = standard deviation; SE = standard error; AEMP = Aquatics Effects Monitoring Program.

Table 8-14 Summary of Results of Statistical Comparisons of Fulton's Condition for Lake Trout and Round Whitefish Captured in Snap Lake, Northeast Lake, and Lake 13 during the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Group	Analysis Type	Interaction Statistic		Intercept Statistic		SSD	Pairwise Comparison (P-value)					
			$Fstat_{(\alpha, df1, df2)}$	P-value	$Fstat_{(\alpha, df, df2)}$	P-value		LK13 vs NEL	Direction of Difference	LK13 vs SL	Direction of Difference	SL vs NEL	Direction of Difference
LKTR	All	$ANCOVA^{log}$	$1.2611_{(0.05, 2, 163)}$	0.2861	$8.5441_{(0.1, 2, 165)}$	0.0003	Y	0.0045	LK13 <NEL	0.0001	SL >LK13	0.2131	SL = NEL
	Males	$ANCOVA^{log}$	$0.6238_{(0.05, 2, 86)}$	0.5383	$8.7663_{(0.1, 2, 88)}$	0.0003	Y	0.0210	LK13 <NEL	0.0001	SL >LK13	0.0943	SL >NEL
	Females	$ANCOVA^{log}$	$1.4333_{(0.05, 2, 39)}$	0.2508	$0.8677_{(0.1, 2, 41)}$	0.4275	N	n/a	LK13 = NEL	n/a	SL = LK13	n/a	SL = NEL
RNWH	All	$ANCOVA^{log}$	$0.7588_{(0.05, 2, 133)}$	0.4703	$25.1475_{(0.1, 2, 135)}$	0.0000	Y	0.1345	LK13 = NEL	0.0000	SL >LK13	0.0000	SL >NEL
	Male	$ANCOVA^{log}$	$3.2444_{(0.05, 2, 53)}$	0.0469*	$19.8632_{(0.1, 2, 55)}$	0.0000	Y	0.1359	LK13 = NEL	0.0000	SL >LK13	0.0008	SL >NEL
	Females	$ANCOVA^{log}$	$0.3170_{(0.05, 2, 51)}$	0.7298	$5.9043_{(0.1, 2, 53)}$	0.0048	Y	0.9282	LK13 = NEL	0.0035	SL >LK13	0.0165	SL >NEL

LKTR = Lake Trout; RNWH = Round Whitefish; ANCOVA = Analysis of Covariance; $ANCOVA^{log}$ = log of Analysis of Covariance; SSD = statistical significant difference (α= 0.1); α = condition factor from weight-length relationship; n/a = not applicable; LK13 = Lake 13; SL = Snap Lake; NEL = Northeast Lake; Fstat = F statistic *P* = probability, df = degrees of freedom; <= less than; >= greater than; * = met the conditions of Barrett et al. (2010) and ANCOVA proceeded; Y = yes; N = no; AEMP = Aquatics Effects Monitoring Program.

Table 8-15 Summary of Statistics for the Weight-Length Relationships for Lake Trout and Round Whitefish Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Lake	Group	N	Size Range (FL - mm)	Weight-Length Regression Equation	*P*-value	Adj. r^2
LKTR	Snap Lake	All	84	183 to 801	Ln W = -11.9996 + 3.1029 Ln FL	<0.001	0.99
		Males	45	281 to 801	Ln W = -12.1982 + 3.1372 Ln FL	<0.001	0.99
		Females	25	340 to 644	Ln W = -11.6748 + 3.0484 Ln FL	<0.001	0.97
	Northeast Lake	All	53	189 to 870	Ln W = -12.1428 + 3.1223 Ln FL	<0.001	0.99
		Males	26	260 to 870	Ln W = -11.9675 + 3.0935 Ln FL	<0.001	0.99
		Females	16	304 to 798	Ln W = -11.5369 + 3.0302 Ln FL	<0.001	0.99
	Lake 13	All	32	342 to 770	Ln W = -11.2489 + 2.9708 Ln FL	<0.001	0.96
		Males	21	342 to 770	Ln W = -11.5060 + 3.0093 Ln FL	<0.001	0.97
		Females	4	437 to 652	Ln W = -8.4332 + 2.5315 Ln FL	0.001	1.00
RNWH	Snap Lake	All	59	153 to 299	Ln W = -11.9487 + 3.0846 Ln FL	<0.001	0.98
		Males	25	171 to 299	Ln W = -10.7679 + 2.8734 Ln FL	<0.001	0.97
		Females	29	178 to 293	Ln W = -12.9610 + 3.2661 Ln FL	<0.001	0.98
	Northeast Lake	All	32	130 to 290	Ln W = -12.7276 + 3.2119 Ln FL	<0.001	0.99
		Males	13	135 to 279	Ln W = -11.6442 + 3.0129 Ln FL	<0.001	0.98
		Females	11	192 to 290	Ln W = -12.8347 + 3.2297 Ln FL	<0.001	0.98
	Lake 13	All	48	100 to 299	Ln W = -12.5249 + 3.1677 Ln FL	<0.001	0.98
		Males	21	194 to 251	Ln W = -14.8336 + 3.5963 Ln FL	<0.001	0.89
		Females	17	189 to 299	Ln W = -12.1411 + 3.1024 Ln FL	<0.001	0.90

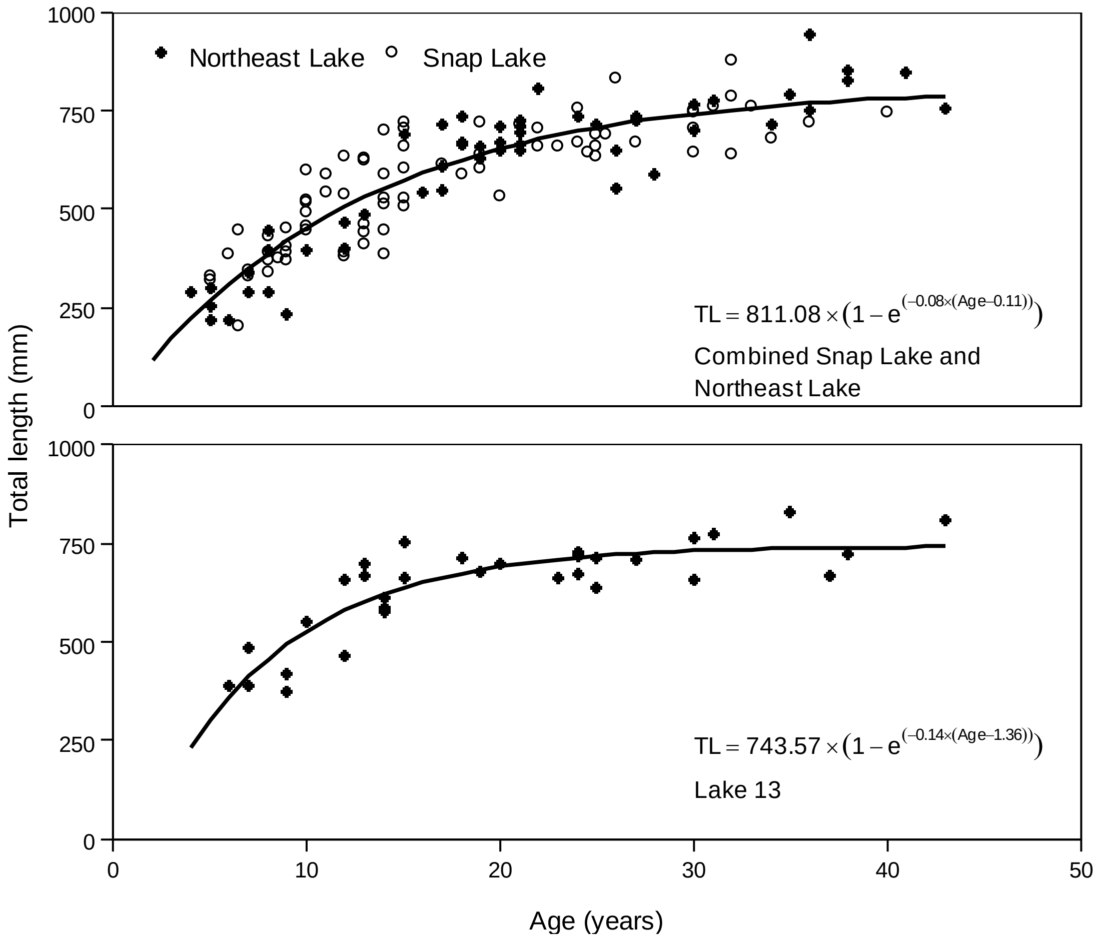
N = number of samples; FL = fork length in mm (millimetre); W = body weight in g (grams); *P* = probability; Adj r^2 = coefficient of determination; Ln = natural log (base e) AEMP = Aquatics Effects Monitoring Program.

Table 8-16 Comparison of Von Bertalanffy Growth Curves for Lake Trout Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Model	N	npar	*P*-value	ΔAICc
Full model (individual-lake)	163	9	n/a	0
Snap Lake = Northeast Lake	163	6	0.148	-1.01
Snap Lake = Lake13	163	6	0.023	3.41
Lake 13 = Northeast Lake	163	6	0.002	8.99

N = number of data points; npar = number of variables; p-value = the significance of a model comparison F-test; ΔAICc = difference in AICc values in comparison to the full, individual-lake model; n/a = not applicable; AEMP = Aquatics Effects Monitoring Program.

Figure 8-10 Length-Age Relationship of Lake Trout Captured in Snap Lake, Northeast Lake and Lake 13 during the July 2013 Snap Lake AEMP Fish Community Monitoring Program



Note: Data from Northeast Lake and Snap Lake were pooled for a single curve.
mm = millimetre; TL =total length; e = mathematical constant e; AEMP = Aquatics Effects Monitoring Program.

Table 8-17 Summary of Von Bertalanffy Growth Curve Coefficients and Rate of Growth for Lake Trout and Round Whitefish Captured in All Lakes during the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Lake	Coefficient	Mean	95% CI
LKTR	Lake 13	k	0.14	0.07 to 0.24
		L_∞	743.57	696.85 to 823.64
		t_0	1.36	-4.77 to 3.82
		ω ($L_\infty \times$ k)	106.48	n/a
LKTR	Combined Northeast Lake and Snap Lake	k	0.08	0.06 to 0.11
		L_∞	811.08	763 to 880.77
		t_0	0.11	-1.90 to 1.53
		ω ($L_\infty \times$ k)	66.88	n/a
RNWH	Lake 13	k	0.53	0.42 to 0.67
		L_∞	293.16	271.18 to 321.24
		ω ($L_\infty \times$ k)	155.37	n/a
	Northeast Lake	k	0.39	0.29 to 0.51
		L_∞	296.01	268.49 to 332.75
		ω ($L_\infty \times$ k)	115.44	n/a
	Snap Lake	k	0.44	0.36 to 0.54
		L_∞	298.69	281.79 to 319.02
		ω ($L_\infty \times$ k)	131.42	n/a

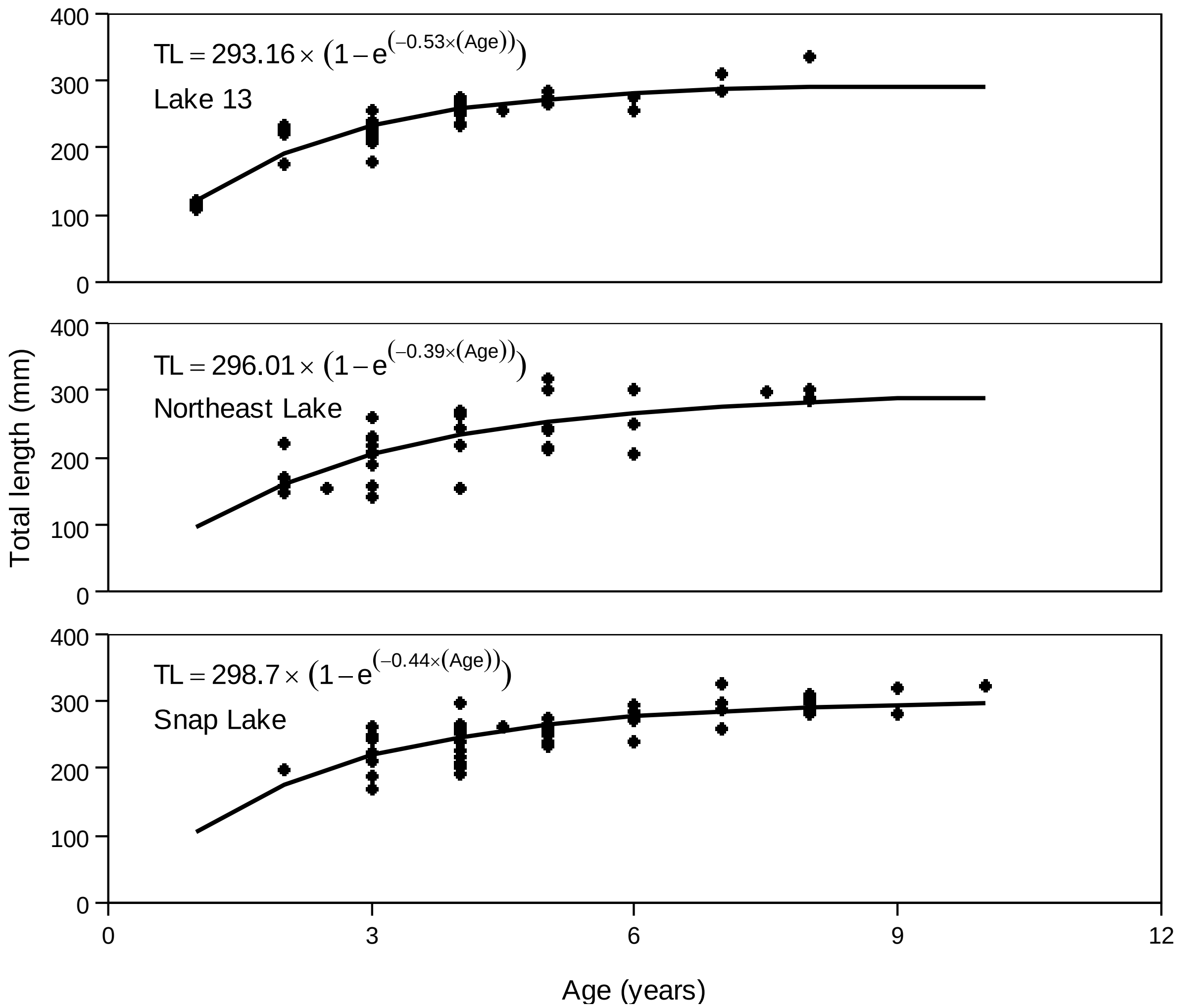
LKTR = Lake Trout; RNWH = Round Whitefish; k = von Bertalanffy growth coefficient; L_∞= length at infinity; t_0 = theoretical length at age 0; ω = rate of growth in length; n/a = not applicable; % = percent; AEMP = Aquatics Effects Monitoring Program.

Table 8-18 Comparison of Von Bertalanffy Growth Curves for Round Whitefish Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Model	N, npar	npar	*p*-value	ΔAICc
Full model (individual-lake)	276	9	n/a	0
Snap Lake = Northeast Lake	276	6	0.15	-0.363
Snap Lake = Lake13	276	6	0.12	0.08
Lake 13 = Northeast Lake	276	6	<0.001	11.98

N = number of data points; npar = number of variables; p-value = the significance of a model comparison F-test; ΔAICc = difference in AICc values in comparison to the full, individual-lake model; n/a = not applicable; < = less than; AEMP = Aquatics Effects Monitoring Program.

Figure 8-11 Length-Age Relationship of Round Whitefish Captured in Lake 13, Northeast Lake, and Snap Lake during the July 2013 Snap Lake AEMP Fish Community Monitoring Program



mm = millimetre; TL =total length; e = mathematical constant e; AEMP = Aquatics Effects Monitoring Program.

8.4.6 Fecundity

Lake Trout

There was considerable variability in fecundity (i.e., eggs/female) in the study lakes; variability was reduced when fecundity was expressed as eggs/kg (Table 8-19). There was a significant relationship between Lake Trout fecundity and fork length for Snap Lake (Table 8-20). Statistical tests were not performed to compare Snap Lake Lake Trout fecundity to the reference lakes when calculated as eggs/female, due to differences in size of fish collected among the lakes, and a small sample size for Lake 13 (i.e., n = 4). Reference lakes contained a greater proportion of longer fish, often times outside the range of lengths represented in the Snap Lake female fish for which fecundity was analyzed.

Table 8-19 Summary Statistics for Fecundity of Lake Trout and Round Whitefish Captured in Lake 13, Northeast Lake, or Snap Lake During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Variable	Lake	N	Mean	SD	SE	Minimum	Maximum
LKTR	Eggs/female	Lake 13	4	9,069	4,195	2,097	3,277	12,940
		Northeast Lake	9	11,287	6,669	2,223	4,398	24,734
		Snap Lake	22	5,687	3,800	810	1,572	19,565
	Eggs/kg	Lake 13	4	3,819	568	284	3,121	4,401
		Northeast Lake	9	3,818	3,259	1,086	1,128	12,184
		Snap Lake	22	3,189	1,466	313	694	8,581
RNWH	Eggs/female	Lake 13	16	2,693	1,456	364	395	5,492
		Northeast Lake	10	3,017	1,385	438	836	4,913
		Snap Lake	17	2,440	1,132	275	712	4,493
	Eggs/kg	Lake 13	16	20,244	9,197	2,299	3,950	36,613
		Northeast Lake	10	17,457	4,959	1,568	7,741	22,714
		Snap Lake	17	13,816	4,311	1,046	7,417	21,395

LKTR = Lake Trout; RNWH = Round Whitefish; N= number of samples; SD = standard deviation; SE = standard error; kg = kilogram; eggs/kg = eggs per kilogram; eggs/female = eggs per female; AEMP = Aquatics Effects Monitoring Program.

Table 8-20 Summary of Results of Statistical Comparisons of Fecundity for Lake Trout and Round Whitefish Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Species	Variable	Analysis Type	Interaction Statistic		Intercept Statistic		SSD	Pairwise Comparison (P-value)					
			$Fstat_{(\alpha, df1, df2)}$	P-value	$Fstat_{(\alpha, df, df2)}$	P-value		LK13 vs NEL	Direction of Difference	LK13 vs SL	Direction of Difference	SL vs NEL	Direction of Difference
LKTR	Eggs by FL	ANCOVA	$0.4957_{(0.05,1,20)}$	0.4895	$3.0527_{(0.1,1,21)}$	0.0952	Y	nt	nt	n/a	SL <LK13	nt	nt
RNWH	Eggs by FL	ANCOVA	$0.2956_{(0.05,2,36)}$	0.0702	$4.7640_{(0.1,2,38)}$	0.0143	Y	0.0721	LK13 >NEL	0.0040	SL <LK13	0.4231	SL = NEL

LKTR = Lake Trout; RNWH = Round Whitefish; FL = fork length; nt = not tested due to non-significant regression relationship for Northeast Lake; n/a = not applicable, the P-value of the intercept statistic applies to the Snap lake versus Lake 13 comparison; ANCOVA = Analysis of Covariance; Fstat = F statistic; α = condition factor from weight-length relationship; df = degrees of freedom; LK13 = Lake 13; NEL = Northeast Lake; SSD = statistical significant difference; p-value = probability value; Y = yes; AEMP = Aquatics Effects Monitoring Program.

Round Whitefish

There was considerable variability in fecundity (i.e., eggs/female) in the study lakes; similar to Lake Trout, Round Whitefish fecundity variability was reduced when fecundity was expressed as eggs/kg (Table 8-19). There was a significant relationship between Round Whitefish fecundity and total length for Snap Lake (Table 8-20). Round Whitefish fecundity was significantly different among lakes (Table 8-20). Fecundity relative to length for Round Whitefish from Snap Lake was significantly less than Lake 13, while Round Whitefish from Lake 13 had significantly greater fecundity relative to length than Northeast Lake. There was no difference in fecundity of Round Whitefish between Snap Lake and Northeast Lake (Table 8-20).

8.4.7 Mortality

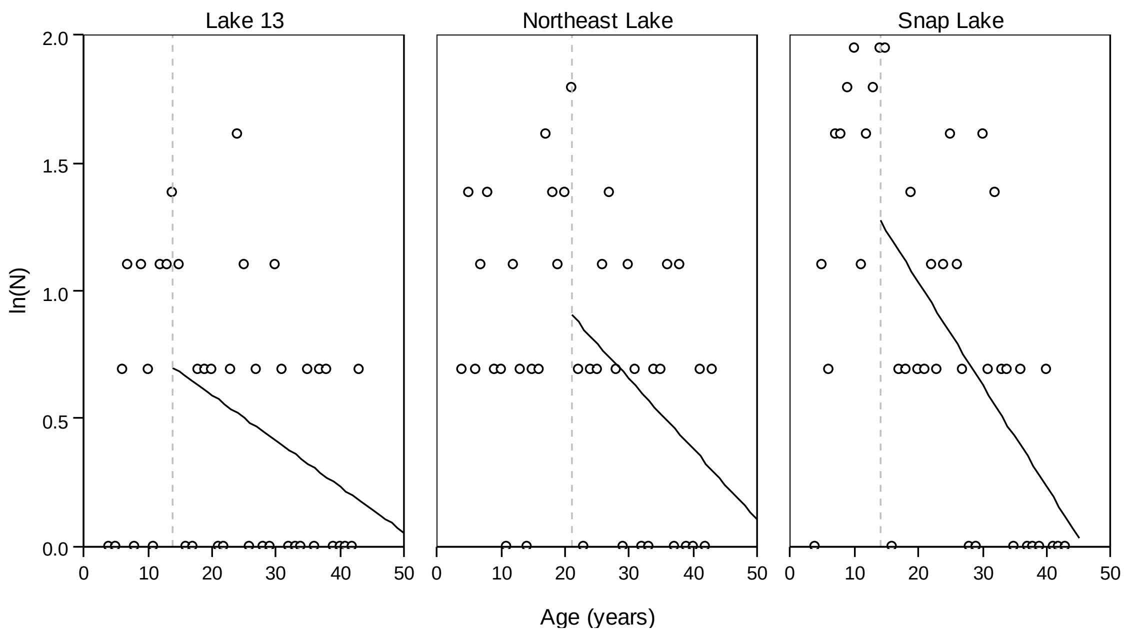
Natural Mortality

Catch curve analysis appeared to be a more accurate method of calculating natural mortality (M) than that derived based on growth in length and length at infinity (L_∞) values from the von Bertalanffy growth curve. Natural annual mortality (M′) estimated using the population rate of growth in length (ω) and length at infinity (L_∞) (Shuter et al. 1998) performed reasonably well when compared with the von Bertalanffy growth curve for Snap Lake Lake Trout (i.e., M = 4% vs M′ = 4.5%). For both Lake 13 (M = 1.8% vs M′ = 6.2%) and Northeast Lake (M = 2.8% vs M’ = 4.4%), M′ was consistently higher than M.

Lake Trout

Lake Trout mortality estimates derived from the catch curves for each lake (i.e., the slope of the line; Figure 8-12) showed among-lake variation that ranged from 1.8% per year in Lake 13 to 4.0% per year for Snap Lake; however, since confidence limits overlapped across lakes (Table 8-21), mortality rates were not significantly different. The mortality rate for Snap Lake was 4% per year, while Northeast Lake was 2.8% per year, and Lake 13 was 1.8% per year (Table 8-21).

Figure 8-12 Relationship Between Ln(N) and Age of Lake Trout Captured in Lake 13, Northeast Lake, and Snap Lake during the July 2013 Snap Lake AEMP Fish Community Monitoring Program



ln(N) = natural log of number (N) of fish; dashed vertical line represents the descending limb of the catch curve and corresponds to the age at which fish numbers begin to decline; this is the point from which mortality was estimated; AEMP = Aquatics Effects Monitoring Plan.

Table 8-21 Summary of Natural Mortality Rate and Confidence Intervals for Lake Trout Captured in All Lakes during the July 2013 Snap Lake AEMP Fish Community Survey

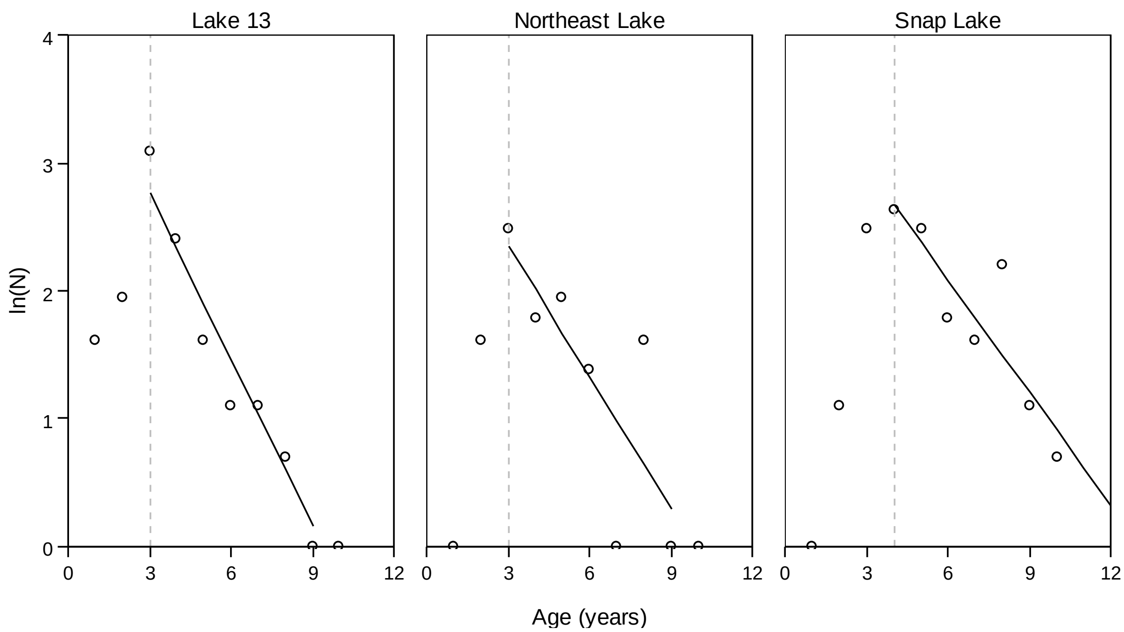
Lake	Mortality (%/Year)	Upper 95% CI	Lower 95% CI
Lake 13	1.8	3.9	0
Northeast Lake	2.8	6.0	0
Snap Lake	4.0	6.2	1.8

CI = confidence interval; % = percent; AEMP = Aquatics Effects Monitoring Program.

Round Whitefish

Round Whitefish mortality estimates derived from the catch curves for each lake (i.e., the slope of the line; Figure 8-13) showed among-lake variation that ranged from 29.3% in Snap Lake, 34.2% in Northeast Lake, and 43.3% in Lake 13. Overlapping confidence limits across lakes (Table 8-22) indicated mortality rates were not significantly different among lakes.

Figure 8-13 Relationship Between Log and Age of Round Whitefish Captured in Lake 13, Northeast Lake, and Snap Lake during the July 2013 Snap Lake AEMP Fish Community Monitoring Program



ln(N) = natural log of number (N) of fish; dashed vertical line represents the descending limb of the catch curve and corresponds to the age at which fish numbers begin to decline; this is the point from which mortality was estimated; AEMP = Aquatics Effects Monitoring Program.

Table 8-22 Summary of Natural Mortality Rate and Confidence Interval for Round Whitefish Captured in All Lakes During the July 2013 Snap Lake AEMP Fish Community Survey

Lake	Mortality (%/Year)	Upper 95% CI	Lower 95% CI
Lake 13	43.3	53.5	33.1
Northeast Lake	34.2	56.8	11.7
Snap Lake	29.3	47.2	11.4

CI = confidence interval; % = percent; AEMP = Aquatics Effects Monitoring Program.

Sampling Mortality

The BsM study resulted in the collection of 88 Lake Trout in 2013 and the estimated mean population size for 2012 was 1,589 Lake Trout (95% confidence interval = 1,151 to 2,299) based on a mark recapture study (Section 11.3). The instantaneous mortality imposed by the BsM method was estimated at 5.6%. Combining the mortality associated with the BsM collections with the natural annual mortality (approximately ~4%) of Lake Trout in Snap Lake from the catch-curve analysis produced a total annual mortality estimate of 9.6%. This mortality rate assumes the BsM study is conducted annually when, in fact, it is only conducted every three years, so the annualized mortality resulting from the recurring BsM assessment would be approximately 1.9%, which is within the 2% mortality initially predicted for use of the BsM method for Snap Lake. The corresponding combined total annual mortality would be approximately 6%.

The Lake Trout harvest rates of the 2013 program were 0.11 kg/ha for Snap Lake, 0.07 kg/ha for Northeast Lake, and 0.07 kg/ha for Lake 13.

8.4.8 Water Quality

Water quality profiles measured at each gill net set location during the fish community monitoring program from July 6 to 18, 2013, are presented in Appendix 8A, Table 8A-6. Lake 13 was sampled from July 6 to July 10, 2013; Northeast Lake was sampled from July 11 to July 16; and Snap Lake was sampled from July 10 to July 18, 2013.

8.4.9 Summary

There were numerous differences between Snap Lake and the reference lakes in the 2013 fish community monitoring program (Table 8-23); however, there were no indications that fish in Snap Lake differed in a systematic way from fish in the two reference lakes that would suggest an effect from the Mine.

Table 8-23 Summary of Results for the July 2013 Snap Lake AEMP Fish Community Monitoring Program

Metric	Endpoint	Result Summary
Community Composition	Species presence and catchability	• Six of seven documented fish species sampled in Snap Lake in 2013[(a)] • Gill net method not suitable for all fish species; additional gear needed to supplement sampling
Abundance	AWCPUE	• Relative abundance[(b)] in Snap Lake was higher than in the reference lakes for all species present in Snap Lake
Size of Fish	Length	• Lake Trout (all fish combined) in Snap Lake were shorter than in Lake 13 but not different than Northeast Lake • Round Whitefish (all fish and males) in Snap Lake were longer than in reference lakes
	Weight	• Lake Trout females were lighter in Snap Lake than in Lake 13 • Round Whitefish (all fish and males) males and all fish combined were heavier in Snap Lake then in the reference lake
	Condition	• Lake Trout (all fish and males) condition in Snap Lake was significantly greater than Lake 13, and (males only) Northeast Lake; condition was less in Lake 13 than Northeast Lake for all fish and males only • Round Whitefish (all fish, males and females) condition in Snap Lake was signficantly greater than both reference lakes; condition was not different between the reference lakes
Age		• Lake Trout age varied from 4 to 43 years; age did not differ significantly among lakes for all fish, or when analyzed separately by sex • Round Whitefish age varied from 2 to 10 years; Snap Lake fish (all fish and males) were older than reference lake fish, and Northeast Lake fish (all fish) were older than Lake 13 fish
Length Frequency		• Lake Trout (all fish) length frequency distributions were significantly different between Snap Lake andboth reference lakes; Snap Lake was different from Northeast Lake for females only; there was a greater porportion of short Lake Trout in Snap Lake than the reference lakes • Round Whitefish (all fish) length frequency distributions were significantly different between Snap Lake and both reference lakes; Snap Lake was also different than both reference lakes for males only
Year Class Strength		• Inconclusive. Too few fish in each age class to calculate properly
Growth Rate		• Lake Trout growth rates in Snap Lake and Northeast Lake are similar, while Lake 13 Lake Trout growth is different • Round Whitefish growth rates are different among all lakes
Mortality Rate	Natural	• Mortality rates of Lake Trout and Round Whitefish were relatively similar for each species between all three study lakes
	Due to sampling	• Mortality of the sampling is approximately 2% over 3 years which appearst to be within a sustainable harvest level
Fecundity		• Lake Trout fecundity was less in Snap Lake than in Lake 13, but not different that Northeast Lake; Lake Trout fecundity was greater in Lake 13 than in Northeast Lake • Round Whitefish fecundity was less in Snap Lake than in Lake 13

a) Burbot were not captured in the BsM (Broad-scale Fish Community Monitoring) netting program but with setlines during the Stable Isotope Special Study (Section 11.4).

b) based on area weighted catch per unit effort (AWCPUE)

AEMP = Aquatics Effects Monitoring Program

8.5 Discussion

8.5.1 Community Composition

Seven fish species have been previously documented in Snap Lake: Arctic Grayling, Burbot, Lake Chub, Lake Trout, Longnose Sucker, Round Whitefish, and Slimy Sculpin (De Beers 2002; Golder 2005a). In 2013, five species were captured including Arctic Grayling, Lake Chub, Lake Trout, Longnose Sucker, and Round Whitefish. The absence of Slimy Sculpin may be due either to the limitations of the sampling gear used (i.e., gill nets) in capturing this species or the gear being deployed in habitats not preferred by Slimy Sculpin. Slimy Sculpin have been caught in other lakes using the BsM method but only in low numbers (Brekke 2014, pers. comm.), as has been the case until recently for Snap Lake (De Beers 2013). Burbot were not captured in the gill nets, but were captured in the 2013 Stable Isotope Special Study (Section 11.4) where setlines were used.

For Northeast Lake, seven fish species were recorded during the 2005 Candidate Reference Lakes study (Golder 2005a): Arctic Grayling, Burbot, Lake Chub, Lake Trout, Ninespine Stickleback, Northern Pike, and Round Whitefish. During the 2009 population monitoring program conducted as part of the AEMP (De Beers 2010a), eight species were captured including the above listed species and Longnose Sucker. With the exception of Ninespine Stickleback, all species recorded in 2009 were captured in 2013. The absence of Ninespine Stickleback from the catch in Northeast Lake may either reflect a limitation of the gear used or the habitat in which it was deployed.

Lake 13 was sampled during the 2005 Candidate Reference Lakes study (Golder 2005b) and five fish species were recorded: Lake Chub, Lake Trout, Ninespine Stickleback, Northern Pike, and Round Whitefish. In 2013, five fish species were captured: Lake Chub, Lake Trout, Northern Pike, Round Whitefish, and Cisco. The capture of Cisco represented a new species for Lake 13. As in Northeast Lake, the absence of Ninespine Stickleback from Lake 13 in 2013 may either reflect a limitation of the gear used or the habitat in which it was deployed.

Additional species of fish could have been captured if double gang nets had been set in 2013, rather than single gangs. A study near Yellowknife, NWT, captured small-bodied fish using double gang sets under the BsM protocol (Brekke 2014, pers. comm.). Although the recommended BsM gear configuration is a double gang joined at the ends of the spanners, De Beers deployed single gangs to minimize incidental fish mortalities in both 2009 and 2013 programs. This deployment configuration was selected because of the small size of the lakes (i.e., less than 18 km^2), narrow depth strata, and ultimately, because fish population size was unknown for key species such as Lake Trout or Round Whitefish in the study lakes (prior to the Lake Trout Population estimate on Snap Lake in 2013). Since fish populations in the study lakes have never sustained such intense fishing pressure and because Water Licence (MV2011L2-0004) requires measures to minimize impact on the lakes, it was determined in 2009 that single gang deployments was the best option (De Beers 2010a). Given that the mortality due to the BsM program was approximately 2% and the total mortality of Lake Trout was estimated at 6%, which results in an acceptable level of harvest (see below), increasing harvest with a double gang deployment is not being considered.

Assuming the next fish community monitoring program in 2016 uses the single-gang deployment, the 2016 BsM protocol will require supplementary, targeted, non-lethal sampling with other gear types to accurately characterize species composition in study lakes. Although low absolute abundance may explain low catches of some species, gill nets may not be appropriate for species such as Burbot, which are more readily collected using baited setlines, or Slimy Sculpin and Ninespine Stickleback, which are more susceptible to capture with either minnow traps (Jackson and Harvey 1997), trawls (Elrod and O'Gorman 1991), or back pack electrofishing (Reid et al. 2009). Applying a sufficient amount of BsM gill netting effort in the study area lakes to achieve reliable relative abundance estimates of Burbot, Slimy Sculpin, and Northern Stickleback would result in higher mortalities of other species. Although Slimy Sculpin have reportedly been captured by BsM nets in other lakes, catch rates are low (Brekke 2014, pers. comm.). Slimy Sculpin were challenging to capture in Snap Lake in previous years in either minnow traps or boat electrofishing (De Beers 2010a). Attempts will be made in 2016 to sample with backpack electrofishing in areas focussed near inlet streams to document presence/absence of the species. It is unlikely the method can be used to quantify abundance of the species.

8.5.2 Abundance

Relative abundance based on the BsM method is assumed to track absolute abundance (Sandstrom et al. 2009) and, as such, species at low absolute abundance should be less numerous in BsM nets than species that are at high absolute abundance. The BsM study in 2013 produced large catches of Lake Trout and Round Whitefish, relatively small but consistent catches of Lake Chub, and intermittent catches of Longnose Sucker, Burbot, Arctic Grayling, and Cisco, but did not capture Slimy Sculpin or Ninespine Stickleback. Low captures of Arctic Grayling may reflect the preference by this species for habitats (e.g., tributary mouths) other than those in the open lake where sampling efforts were focused in the present study.

Catch rates of fish varied among lakes. Catch rate of Lake Trout for Snap Lake was over twice that of either Northeast Lake or Lake 13, and this likely indicates that the absolute population size of Lake Trout in the two reference lakes was lower. The reasons for the differences in catch rates are unknown but are speculated to be a eutrophication response to the Mine's effluent (see Section 12) or unrelated to the mine in terms of difference in top predators in each lake, or habitat use and the patchy nature of the fish community (De Beers 2010a) or differences such as temperature, which may be influenced by climate change (discussed below in Section 8.5.3).

It was assumed that a higher abundance of fish in Snap Lake should have resulted in lower growth as a result of density dependent effects (i.e., less food per individual resulting in less growth). However in the case of Snap Lake, higher abundance may be compensated for by higher abundance of Lake Trout prey including Lake Chub, Lake Whitefish (*Coregonus clupeaformis*), and Longnose Sucker (see Section 11.4). For Round Whitefish, it may be possible that the increased abundance of prey items also compensates for the higher abundance, but the pattern was less clear, a slight eutrophication effect from the Mine may be possible if the study considered abundance alone. However, when all the study metrics are considered together, there is no discernable pattern suggesting a consistent Mine-related effect.

Northern Pike, a known top predator (Scott and Crossman 1973), was captured in Northeast Lake and Lake 13 but not Snap Lake. Mills et al. (2002) reported a much lower abundance of Lake Trout in an Experimental Lakes Area lake that contained Northern Pike compared to four other lakes where Northern Pike were absent. Carl et al. (1990) hypothesized that the presence of Northern Pike can suppress Lake Trout abundance, and such an effect could be mediated through reduced prey availability. The stable isotope study (Section 11.4) indicated the suspected prey of Lake Trout consisted primarily of Round Whitefish, Lake Chub, and Longnose Sucker, and the relative abundance of these species (based on catch rates) was lower in Northeast Lake and Lake 13 compared to Snap Lake. Therefore, higher Lake Trout abundance in Snap Lake could be due to the lack of Northern Pike, compounded by the higher abundance of prey species in the reference lakes.

Despite the higher relative abundance of Lake Trout in Snap Lake compared to the reference lakes, on the basis of absolute abundance, abundance of Lake Trout in Snap Lake is lower than in other lakes, particularly for a population that is relatively unexploited outside of periodic AEMP assessments. A density of 1.0 Lake Trout per hectare (LTH) was calculated for 2012 based on the estimated area of Snap Lake (1,566 ha) and the median mark recapture estimate of 1,589 fishable (greater than 250 mm fork length) Lake Trout. On the basis of lake area, Lake Trout absolute abundance in Snap Lake was low compared to other lakes in North America but there is overall little published data on lakes of a similar size to Snap Lake.

Although estimates of Lake Trout population abundance for lakes in the Northwest Territories are limited, Burr (1997) reported that densities of mature Lake Trout for six Alaska lakes at similar northern latitudes to the study lakes ranged from 3.1 to 32.8 LTH. For eight unexploited lakes in the Experimental Lakes Area of northwestern Ontario, Lake Trout density ranged from 7.6 to 23.8 LTH, although these lakes are considerably smaller (16 to 54 ha) than Snap Lake (1,600 ha). Payne et al. (1990) indicated that Lake Trout densities are typically higher in small lakes than large lakes. Fish productivity is expected to be lower in northern lakes like Snap Lake, with lower overall primary and secondary productivity (Downing and Plante 1993) compared to more southerly lakes, such as those in the Experimental Lakes Area.

Although exploitation has often been associated with low Lake Trout abundance and the periodic AEMP assessments represent a form of exploitation, the low density of Lake Trout in Snap Lake is not considered to result from exploitation. Based on the BsM netting program, Lake Trout AWCPUE for Snap Lake was higher than the AWCPUE for Northeast Lake, even though Northeast Lake has been subject to the same number of large-scale assessments as Snap Lake. Lake Trout AWCPUE in Snap Lake was also higher than for Lake 13, a lake which had not been assessed prior to this study and has been subject to virtually no other form of exploitation.

8.5.3 Thermal Habitat and Abundance

A factor that may affect Lake Trout abundance is thermal habitat. Lake Trout are a cold water stenotherm, and although there is considerable variation with respect to Lake Trout thermal habitat use, temperatures above 15°C appear to be unsuitable for extended periods of time (Plumb and Blanchfield 2009). Christie and Regier (1988) reported that the sustained yield of Lake Trout for a series of north temperate lakes was related to summer measures of either the amount of thermal habitat area or thermal habitat volume that was within their preferred thermal niche of 8°C to 12°C. At the time that the BsM program was completed in early summer of 2013, there were no restrictions on thermal habitat for Lake Trout in any of the study lakes (Appendix 8F). However, over the entire summer and fall period, the upper portion of the three study lakes warmed above 15°C (Appendix 8F; also see Section 11.4 for figures of Snap Lake).

Based on data obtained from a vertical temperature logger array in Snap Lake in 2013 (Appendix 8E), there was a gradual deepening of the 15°C isotherm in the main basin of the lake. As a result, less than 5% of the current estimated lake volume was less than 15°C for a short time period in mid-July, and less than 1% of the lake volume was less than 15°C for a three week period in August. With decreasing depth, water temperatures became increasingly unsuitable (i.e., greater than 15°C) for increased periods of time during the summer (Appendix 8F, Figures 8F-2 and 8F-3). The only areas in Snap Lake where there were no restrictions on the thermal suitability of habitat for Lake Trout was a deeper area close to the diffuser in the main basin (Section 11.4 Figure 11.4-9), and in a deep hole at the western-most portion of the northwest arm (Section 11.4, Figure 11.4-10).

By comparison, the amount of thermally suitable habitat for Lake Trout reached a low of 10% of the total lake volume in Northeast Lake. The minimum amount of thermally suitable habitat in Lake 13 was close to 0%, but only for a three day period in August.

The restrictions on the availability of thermally suitable habitat observed for the study lakes may be greater than for other lakes at lower latitudes; in northwestern Ontario, at least 20% to 40% of the lake volume fell below the 15°C benchmark over a two year period for a Lake Trout lake (Plumb and Blanchfield 2009). This minimum lake volume above 15°C is much greater than the volume that was available in Snap Lake, Northwest Lake, or Lake 13.

Whether the amount of suitable thermal habitat available for Lake Trout in the three lakes during 2013 represents conditions during an average year is not clear. Air temperatures in the Canadian north are on a long-term warming trend, and as lake temperatures are correlated with air temperatures, warmer lake temperatures are also expected (Schindler et al. 1990, 1996). For Snap Lake, the depth of the 15°C isotherm in early August (i.e., prior to August 15) has shown some fluctuation during the seven-year period 2006 to 2013, but 2013 conditions did not appear anomalous. In two of the seven years, there was no water at less than 15°C in the water column in August, but in the other five years, the 15°C isotherm was located within 3 m of the bottom of the lake, as occurred in 2013. Small lakes such as Snap Lake are likely to be more responsive to the effects of climate change than large lakes.

Despite an apparent limited availability of appropriate thermal habitat for Lake Trout in Snap Lake during the summer when most growth is expected to occur, the use of littoral resources by Lake Trout appeared high. The estimated proportion of carbon in the Lake Trout diet from littoral sources located in shallow areas of the lake was 75% (Section 11.4). The most productive area of the littoral zone of oligotrophic lakes is often found at a depth of less than 3 m (Keast and Harker 1977). In Snap Lake, Lake Trout may feed heavily in the littoral zone while thermal conditions are acceptable and then seek refuge in colder parts of the lake once the littoral zone becomes too warm (see Snucins and Gunn 1995; Mackenzie-Grieve and Post 2006).

Mobile predators such as Lake Trout have been shown to be a major factor in governing the stability of food webs, governing the movement of energy and nutrients through littoral and pelagic areas (Dolson et al. 2009; Post et al. 2000; McCann et al. 2005; Rooney et al. 2006). As such, the trend of increasing air temperatures and probable concurrent effect on lake temperatures may further reduce the degree of connection between littoral and pelagic zones of lakes like Snap Lake. Given the large range of effects that could be caused by temperature increases in Snap Lake to resident Lake Trout and the Snap Lake ecosystem of which Lake Trout are an integral and modifying factor, ongoing temperature monitoring on Snap Lake will be essential to be able to separate the effects of increased lake temperature from the effects of the Mine.

8.5.4 Size of Fish

The attributes of Lake Trout from Snap Lake, including condition, survival, age at maturity, growth, and L_∞, were similar when compared to Lake Trout from Northeast Lake or Lake 13, suggesting there is no obvious effect of the Mine on Snap Lake. These attributes do appear to be indicative of populations at relatively high density, but not close to the carrying capacity for these lakes.

Lake Trout from all study lakes were more slender than the average reported for 63 other Lake Trout populations in southern British Columbia and Quebec (McDermid et al. 2010), and were in the lower third of the 58 Lake Trout populations used for development of a standard weight equation for this species (β = 3.25; Piccolo et al. 1993). Consistent with a presumed high density, Lake Trout from Snap Lake, Northeast Lake, and Lake 13 suffered much lower natural annual mortality (M' = 1.8% to 4.0%) than typically reported for other Lake Trout populations in Canada (minimum M = 17%; Healey 1978; Martin and Olver 1980). Conversely, the Lake Trout in Snap Lake, Northeast Lake, and Lake 13 grew to an L_∞ (747 to 819 mm) that was near the high end of the range for Lake Trout (maximum fork length = 500 to 1,000 mm; Healey 1978; Martin and Olver 1980), and much higher than the range in L_∞ calculated based on lake area alone (610 to 632 mm; Shuter et al. 1998). These values are not consistent with a high density population with low natural annual mortality.

In northern latitudes across a broad geographic range, Lake Trout reportedly grow slower, mature later, live longer, and experience lower total mortality than at southern latitudes (Martin 1952; Martin and Olver 1980; McDermid et al. 2010). However, biotic factors, such as food availability and predation, can moderate the effects of climatic conditions and influence mortality, growth, and maturity of Lake Trout populations (Martin and Olver 1980).

The natural mortality rate for Round Whitefish in these study lakes was estimated at 30% 40%. Little has been published about mortality and population factors for Round Whitefish. This level of mortality is comparable to other whitefish species such as Lake Whitefish (Ebener et al. 2010). Length at infinity (L_∞;

370 mm fork length) for an anadromous population of Round Whitefish from Eastern James Bay-Hudson Bay was higher than Round Whitefish from Snap Lake (285 mm fork length), Northeast Lake (283 mm fork length), and Lake 13 (275 mm fork length) (Morin et al. 1982).

8.5.5 Fecundity

Average fecundity of Lake Trout from Snap Lake on a per fish basis was generally much higher relative to other northern populations (Healey 1978), although the reasons and implications to Lake Trout life history in Snap Lake of this are unclear. It was not possible to compare fecundity of Lake Trout among the three study lakes because fecundity was related to size, and there was little overlap in the size of Lake Trout females among lakes. Using relationships contained in Healey (1978) between fecundity (eggs/female) and fork length, calculated fecundity (eggs/female) for a 55 centimetre (cm) fork length Lake Trout (i.e., the midpoint of the Snap Lake data) for other Lake Trout lakes would be as follows: Lake Ontario (18,249 eggs), Lake Superior (2,572 eggs), Great Slave (3,486 eggs), Opeongo (1,297 eggs), Alexie (2,266 eggs), Chitty (2,110 eggs), Drygeese (1,369 eggs). With the exception of Lake Ontario where estimated fecundity was approximately four-fold higher, the estimates for the remaining lakes were less than Snap Lake (4,320 eggs).

Lake Trout fecundity responds positively to exploitation (Healey 1978). Using the relationship in Fitzsimons and O'Gorman (1996) between fecundity (eggs/female) and total length (mm), for a 60 cm long Lake Trout, the fecundity for a Lake Trout from Snap Lake (i.e., 4,320) was higher than a Lake Ontario Lake Trout (3,030), even after taking account of the effect of exploitation. This indicates that Lake Trout fecundity in Snap Lake is among the highest of that reported for North American populations.

Fecundity of Round Whitefish for the AEMP lakes ranged from 2,440 eggs/female for Snap Lake to 3,017 for Northeast Lake. Round Whitefish fecundity was within the range reported for Newfound Lake (New Hampshire) (2,200 to 9,445 eggs/female; Normandeau 1963), and Lake Superior (1,076 to 11,888 eggs/female; Bailey 1963).

8.5.6 Age of Fish

The age structure of male Lake Trout in Snap Lake was different from Northeast Lake. The males in Northeast Lake were younger than those in Snap Lake. Conversely Round Whitefish in Snap Lake were similar in age structure to those in Northeast Lake, but older than the Round Whitefish captured in Lake 13. Overall no young Lake Trout (i.e., ages 1 to 3), and few young Round Whitefish (i.e., < 3 y) were captured in any of the lakes; however, insufficient numbers of Lake Trout and Round Whitefish were captured within each age class to allow for an analysis of year class strength. A previous study which utilized a variety of sampling gear also found that juvenile Lake Trout and Round Whitefish were not abundant in any of the lakes (De Beers 2005). As such, the lack of juvenile Lake Trout and Round Whitefish may indicate that fish are utilizing different habitat during the spring than where the sampling gear is being set, rather than Mine-related effects.

8.5.7 Mortality

Natural Mortality

Mortality rates for Lake Trout and Round Whitefish were estimated, including both natural mortality and mortality due to the BsM sampling program. Annual natural mortality of Lake Trout was greater in Snap Lake (4%) compared to Northeast Lake (2.8%) and Lake 13 (1.8%); however, the differences were not statistically significant. The Lake Trout mortality rate calculated for Snap Lake, while based only on a single year of data, appears unlikely to be biased because the age frequency distribution did not suggest the presence of a single strong year class (Ricker 1975).

Natural mortality for Lake Trout from Snap Lake was less than the average annual mortality of 17% reported by Healey (1978) for 24 separate estimates of annual natural mortality for 15 different populations of Lake Trout. Healey (1978) stated that an annual mortality rate in the range of 20% to 30% was typical of unexploited Lake Trout populations; however, these rates may have been biased high due to the aging structures used (i.e., scales). For southern Ontario lakes, Shuter et al. (1998) present a range of annual mortality from 11% to 22%, for fish aged either with fin rays or otoliths that show considerably less bias than scales (Campana 2001). The method of estimating mortality for these lakes was catch curve analysis, similar to this study. For montane lakes in Alberta, where aging was performed using fin rays, annual natural mortality was approximately 20% for lakes where Lake Trout maximum age did not exceed 20 years, and 10% for lakes where the maximum age exceeded 20 years (Sullivan 2014, pers. comm.). For one montane lake located in Jasper National Park, where no fishing was believed to occur and where fish lived in excess of 25 years, natural mortality was 8%. For Lake Mistassini, a relatively unexploited lake located in northern Quebec, natural mortality was 5%, based on catch curve analysis of otolith aged fish (Hansen et al. 2012). Nine lakes located in the Experimental Lakes Area of northwestern Ontario that experienced no exploitation had natural mortality that ranged from 9% to 22%, based on Jolly-Seber estimates for fish aged using fin rays (Mills et al. 2002).

Mortality Due to Sampling

Mortality due to the sampling method was estimated for Lake Trout. It could not be estimated for Round Whitefish as there was no population estimate data for this species.

Given the relatively low natural mortality of Lake Trout in the study lakes, even with the level of exploitation associated with the BsM program, total mortality appears to be within the limits of a sustainable population based on conditions in Snap Lake. The instantaneous mortality imposed by the BsM method was estimated at 5.6%. The total annual mortality estimate of natural mortality plus BsM mortality was 9.6%. However, the BsM method is only used once every three years; therefore, the annualized mortality resulting from the recurring BsM assessment would be approximately 1.9%, which is within the 2% mortality initially predicted for use of the BsM method for the Snap Lake AEMP fish community monitoring program. The combined total annual mortality, assuming a three-year program cycle, would be approximately 6%. This level of mortality is less than reportedly sustainable by other Lake Trout populations whether exploited (Healey 1978; Shuter et al. 1998) or unexploited (Mills et al. 2002).

The amount of Lake Trout biomass removed using the BsM method from the study lakes considering their area, appears to be within a level sustainable by Lake Trout populations. The level of harvest resulting from the BsM method in 2013 for the three study lakes (Snap Lake -0.11 kg/ha, Northeast Lake -0.07 kg/ha, and Lake 13 -0.07 kg/ha) were all lower and fell below the range of suggested harvest of exploited populations of Lake Trout in northern waters of 0.2 to 0.5 kg/ha (Healy 1978).

8.5.8 Water Quality

At the time of sampling, water quality did not appear to play a role in the observed variation in Lake Trout among the three lakes. During sampling, both DO and water temperatures were similar among lakes and well within established criteria for the maintenance of aquatic life. Dissolved oxygen levels in all lakes were consistently higher than the DO criterion of 7 mg/L which was established based on metabolic scope for activity and power capacity of juvenile Lake Trout (Evans 2007). Water temperatures throughout the entire water column of all three lakes were less than the established upper temperature limit of 15°C (Plumb and Blanchfield 2009) during the period of fish sampling. Variation in pH among lakes was low. Despite the higher total dissolved solids (TDS) in Snap Lake compared to the two reference lakes (Section 3), relative abundance of Lake Trout was approximately twice as high in Snap Lake compared to the reference lakes. This suggests that the present TDS levels in Snap Lake are not limiting to the Lake Trout population.

8.5.9 Broad-scale Fish Community Monitoring Methodology

The data and analyses presented suggest that year class strength may not be an appropriate endpoint for monitoring impacts of the Mine on fish populations in Snap Lake. Catch per unit effort, comparison of fish size, and mortality and growth rate may be more sensitive indices of population-level change associated with the Mine. These parameters may be less sensitive to sample size issues, and are easily calculated from data collected in a standard monitoring program. The use of such metrics should be explored as alternative endpoints for monitoring population change over time.

The BsM program is a standardized, lethal sampling program that measures numerous fisheries metrics that may be variable between lakes; the level of change in any of these metrics that would suggest a Mine effect is not well defined and may require review in the AEMP Re-evaluation Report in 2016. This is the second BsM program for Snap Lake, and temporal trends in the fish community will be considered in the future as additional data sets are collected.

8.6 Conclusions

On the basis of the 2013 fish community monitoring program, there were no indications that fish in Snap Lake differed in a systematic way from fish in the two reference lakes that would suggest an effect from the Mine. There were differences between the two references lakes, suggesting natural variability accounts for changes observed in the area.

Some differences identified in the 2013 fish community monitoring program were as follows:

- Fish community composition in Snap Lake is similar to previous years. One exception is that Slimy Sculpin were not sampled, likely due to their low abundance and the use of the gill net method. Additional gear types may be needed to sample the full community composition.
- As expected, fish community composition is different in the three lakes, including Northern Pike, a top predator, in Northeast Lake and Lake 13.
- The relative abundance of Lake Trout, Round Whitefish, and Longnose Sucker was higher in Snap Lake than the reference lakes. This suggests the current level of TDS is not limiting to the fish community.
- When all fish were combined, total length of Lake Trout from Snap Lake was significantly shorter than that of Lake Trout from Lake 13 but did not differ in total length from Northeast Lake.
- There was a greater proportion of shorter Lake Trout captured in Snap Lake than Northeast Lake and Lake 13. However, the fish were stouter (higher condition). Round Whitefish were longer, heavier, and older in Snap Lake than the reference lakes.
- Lake Trout fecundity in the study lakes was high relative to other lakes in North America.
- Round Whitefish fecundity was significantly different among lakes. Fecundity of Round Whitefish from Snap Lake was significantly less than Lake 13, while Round Whitefish from Lake 13 had significantly greater fecundity than Northeast Lake. There was no difference in fecundity of Round Whitefish between Snap Lake and Northeast Lake.
- Utilizing the Lake Trout population estimate data (Section 11.3), the mortality to Lake Trout due to a BsM program every three years was calculated to be approximately 2% of the fishable population. The natural and sampling mortality together were estimated at 6% in the year of the BsM program, which results in an acceptable level of harvest of Lake Trout over a three year period.[6]
- Lake temperatures warmed over the summer in each study lake and reduced the “habitat optimums” of key fish species such as Lake Trout. This may affect fish populations’ ability to feed and grow in the future, independent of the Mine operation.

[6] Increasing harvest to double gang deployment as is normally required by the BsM protocol is not being considered. The modified BsM protocol with the additional gear for smaller-sized fish is deemed an appropriate sampling method for the next sampling program in Snap Lake in 2016. This modified method (single gang nets with additional gear) will be used on the downstream lakes in 2014.

Despite the existence of these differences between fish from Snap Lake and the reference lakes, the conclusion is made that no changes directly attributed to the Mine were identified in the fish community monitoring program. Fish were healthy and abundant in Snap Lake in 2013.

8.6.1 Key Question 1: Will the Fish Community be Affected by the Changes in Water Quality in Snap Lake and Will any Change be Greater than Predicted in the EAR?

The EAR predicted that the operation of the Mine would not result in any detectable changes to the fish community in Snap Lake. Although there were some significant differences in fish population metrics examined between Snap Lake and the reference lakes, these differences could reasonably be attributed to natural variation, differences in sample methods, or sample size/composition effects. Based on the results of this study, there have been no discernable changes to the fish community of Snap Lake that could be directly attributed to Mine-related changes in water quality.

8.7 Recommendations

Given the limitations of the BsM method in capturing species such as Burbot, Arctic Grayling, Slimy Sculpin, and Ninespine Stickleback in the study lakes based on 2013 results, consideration should be given to the addition of alternative methods that, in conjunction with the BsM, would provide a more effective means of indexing population metrics of these species, while being cognizant of the need to control incidental mortality. Slimy Sculpin were challenging to capture in Snap Lake in previous years; attempts will be made in 2016 to sample with backpack electrofishing in areas near inlet streams.

8.8 References

Adrian R, O'Reilly CM, Zagarese H, Baines SB, Hessen DO, Keller W, Livingstone DM, Sommaruga R, Straile D, Donk EV, Weyhenmeyer GA, Winder M. 2009. Lakes as sentinels of climate change. Limnol Oceanogr 54: 2283.

Bailey MM. 1963. Age, growth and maturity of Round Whitefish of the Apostle Islands and Isle Royale regions, Lake Superior. Fish. Bull. 63: 63–75.

Barrett T, Tingley M, Munkittrick K, Lowell R. 2010. Dealing with heterogeneous regression slopes in analysis of covariance: new methodology applied to environmental effects monitoring fish survey data. Environ Monit Assess 166: 279-291.

Baty F, Delignette-Muller ML. 2013. nlstools: tools for nonlinear regression diagnostics.

Brekke L. Fisheries Biologist. 2014. Aboriginal Affairs and Northern Development Canada (AAND), Yellowknife, NWT, Canada. E-mail. March 5, 2014.

Burr JM. 1997. Growth, density, and biomass of Lake Trout in Arctic and Subarctic Alaska. Am Fish Soc Sympos 19: 109-118.

Campana SE. 2001. Accuracy, precision and quality control in age determination, including a review of the use and abuse of age validation methods. J Fish Biol 59: 197-242.

Carl LM, Christie W, Deacon L, Hulsman D, Loftus D., Maraldo D, Marshall T, Ryan P.1990. Fish Community and Environmental Effects on Lake Trout: Lake Trout Synthesis. Ontario Ministry of Natural Resources. Toronto, ON, Canada. 47 p.

Christie GC, Regier HA. 1988. Measures of optimal thermal habitat and their relationship to yields for four commercial fish species. Can J Fish Aquat Sci 45: 301-314.

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

De Beers. 2005. Aquatic Effects Monitoring Program for the Snap Lake Diamond Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2006. 2005 Annual Report in Support of the Aquatic Effects Monitoring Program (AEMP) Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2007. 2006 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2008. 2007 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2009. 2008 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2010a. 2009 Fish Population Monitoring Program Report, Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2010b. 2009 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2011. 2010 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012a. Aquatic Effects Monitoring Program Re-evaluation Report: Snap Lake AEMP. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012b. 2011 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2013. 2012 Annual Report in Support of the Aquatics Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

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

Dolson R, McCann K, Rooney N, Ridgway M. 2009. Lake morphometry predicts the degree of habitat coupling by a mobile predator. Oikos 118: 1230-1238.

Downing JA, Plante C. 1993. Production of fish populations in lakes. Can J Fish Aquat Sci 50: 110-120.

Ebener MP, TO Brenden, ML Jones. 2010. Estimates of fishing and natural mortality rates for four Lake Whitefish stocks in Northern Lakes Huron and Michigan. Journal of Great Lakes Research 36: 110–120.

Elrod JH, O'Gorman, R. 1991. Diet of juvenile Lake Trout in southern Lake Ontario in relation to abundance and size of prey fishes, 1979–1987. Trans Am Fish Soc 120: 290-302.

Environment Canada. 2012. Metal Mining Technical Guidance for Environmental Effects Monitoring (EEM). National EEM Office, Ottawa, ON, Canada.

Evans DO. 2007. Effects of hypoxia on scope-for-activity and power capacity of Lake Trout (*Salvelinus namaycush*). Can J Fish Aquat Sci 64: 345-361.

Fitzsimons JD, O'Gorman R. 1996. Fecundity of hatchery Lake Trout in Lake Ontario. J Great Lakes Res 22: 304-309.

Golder. 2005a. Reference Lake Desktop Screening for the Snap Lake Project. Submitted to De Beers Canada Inc., Yellowknife, NWT, Canada.

Golder. 2005b. Field Investigation and Reference Lake Selection for the Snap Lake Project. Submitted to De Beers Canada, Inc., Yellowknife, NWT, Canada.

Hansen MJ, Nate NA, Krueger CC, Zimmerman MS, Kruckman HG, Taylor WW. 2012. Age, growth, survival, and maturity of Lake Trout morphotypes in Lake Mistassini, Quebec. Trans Am Fish Soc 141: 1492-1503.

Healey MC. 1978. The dynamics of exploited Lake Trout populations and implications for management. J Wildl Manage 42: 307-328.

Hoffmann JP. 2004. Generalized Linear Models: An Applied Approach. Pearson Education Inc., Boston MA, USA.

Jackson DA, Harvey HH. 1997. Qualitative and quantitative sampling of lake fish communities. Can J Fish Aquat Sci 54: 2807-2813.

Keast A, Harker J. 1977. Fish distribution and benthic invertebrate biomass relative to depth in an Ontario Lake. Environ Biol Fish 2: 235-240.

Koops MA, Hutchings JA, McIntyre TM. 2004. Testing hypotheses about fecundity, body size and maternal condition in fishes. Fish Fisher 5: 120-130.

MacKay WC, Ash GR, Norris HJ. 1990. Fish Ageing Methods for Alberta. RL & L. Environmental Services.. Edmonton, AB, Canada.

Mackenzie-Grieve JL, Post JR. 2006. Thermal habitat use by Lake Trout in two contrasting Yukon Territory lakes. Trans Am Fish Soc 135: 727-738.

Martin NV. 1952. A study of the Lake Trout, *Salvelinus namaycush*, in two Algonquin Park, Ontario, lakes. Trans Am Fish Soc 81: 111-137.

Martin NV, Olver CH. 1980. The Lake Charr, *Salvelinus namaycush*. Charrs: salmonid fishes of the genus *Salvelinus*, pp 205-277. In: Balon, E.K. (ed.), Charrs: Salmonid Fishes of the Genus *Salvelinus*. Dr. W. Junk, The Hague, pp. 205–277.

McCann KS, Rasmussen JB, Umbanhowar J. 2005. The dynamics of spatially coupled food webs. Ecol Lett 8: 513-523.

McDermid JL, Shuter BJ, Lester NP. 2010. Life history differences parallel environmental differences among North American Lake Trout (*Salvelinus namaycush*) populations. Can J Fish Aquat Sci 67: 314-325.

Mills KH, Chalanchuk SM, Allan DJ. 2002. Abundance, annual survival, and recruitment of unexploited and exploited Lake Charr, *Salvelinus namaycush*, populations at the Experimental Lakes Area, northwestern Ontario. Environ Biol Fish 64: 281-292.

Morin R, Dodson JJ,.Power G. 1982. Life history variations of anadromous cisco, lake whitefish ***(****Coregonus clupeaformis),* and round whitefish ***(****Prosopium cylindraceum****)*** populations of eastern James-Hudson Bay. Can J Fish Aguat Sci 39: 958-967.

MVLWB (Mackenzie Valley Land and Water Board). 2013. Mackenzie Valley Land and Water Board Water License # MV2011L2-0004. Yellowknife, NWT, Canada.

Normandeau DA. 1963. The Life History of the Round Whitefish *Prosopium cylindraceum* (Pallas) of Newfound Lake. Doctoral dissertation, University of New Hampshire, Durham, NH, USA.

Ogle, DH. 2010. FSA: Fisheries Stock Analysis, R package version 0.3.5.

Payne NR, Korver RM, MacLennan DS, Nepszy SJ, Shuter BJ, Stewart TJ, Thomas ER. 1990. The Harvest Potential and Dynamics of Lake Trout Populations in Ontario. Lake Trout Synthesis Population Dynamics Working Group Report. Ontario Ministry of Natural Resources, Toronto, ON, Canada.

Piccolo JJ, Hubert WA, Whaley RA. 1993. Standard weight equation for Lake Trout. N Am J Fish Manage 13: 401-404.

Plumb JM, Blanchfield PJ. 2009. Performance of temperature and dissolved oxygen criteria to predict habitat use by Lake Trout (*Salvelinus namaycush*). Can J Fish Aquat Sci 66: 2011-2023.

Post DM, Pace ML, Hairston NG. 2000. Ecosystem size determines food-chain length in lakes. Nature 405: 1047-1049.

Power G. 1978. Fish population structure in Arctic lakes. J Fish Res Board Can 35: 53-59.

Quinn TJ, Deriso RB. 1999. Quantitative Fish Dynamics. Oxford University Press, New York, NY, USA, pp 542.

R Core Team. 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Reid SM, Yunker G, Jones NE, 2009. Evaluation of single-pass backpack electric fishing for stream fish community monitoring. Fish Manage Ecol 16: 1–9.

Ricker WE. 1975. Computation and interpretation of biological statistics of fish populations. Bull Fish Res Board Canada 191: 382 p. Ottawa ON, Canada.

Rooney N, McCann K, Gellner G, Moore JC. 2006. Structural asymmetry and the stability of diverse food webs. Nature 442: 265-269.

Sakamoto Y, Ishiguro M, and Kitagawa G. 1986. Akaike Information Criterion statistics. D. Reidel Publishing Company.

Sandstrom S, Rawson M, Lester N. 2009. Manual of Instructions for Broad-scale Fish Community Monitoring; Using Large Mesh Gill nets and Small Mesh Gill nets. Ontario Ministry of Natural Resources. Peterborough, ON, Canada.

Sandstrom S. Fisheries Biologist. 2013. Ontario Ministry of Natural Resources, Bracebridge, ON, Canada. E-mail. June, 2013

Schindler DW, Frost, TM, Mills KH, Chang PSS, Davies IJ, Findlay L, Malley DF, Shearer JA, Turner MA, Garrison PJ, Watras CJ, Webster K, Gunn JM, Brezonik PL, Swenson WA. 1990. Comparisons between experimentally-and atmospherically-acidified lakes during stress and recovery. Proceedings of the Royal Society of Edinburgh. Section B. Biological Sciences 97: 193-226.

Schindler DW, Curtis PJ, Parker BR, Stainton MP. 1996. Consequences of climate warming and lake acidification for UV-B penetration in North American boreal lakes. Nature 379: 705-708.

Schindler DW, Frost, TM, Mills KH, Chang PSS, Davies IJ, Findlay L, Malley DF, Shearer JA, Turner MA, Garrison PJ, Watras CJ, Webster K, Gunn JM, Brezonik PL, Swenson WA. 1990. Comparisons between experimentally-and atmospherically-acidified lakes during stress and recovery. Proc Royal Soc Edinburgh Sec B Biol Sci 97: 193-226.

Scott WB, Crossman EJ. 1973. Freshwater fishes of Canada. Fish Res Board Can Bulletin 183 966 p.

Shuter BJ, Jones ML, Korver RM, Lester NP. 1998. A general, life history based model for regional management of fish stocks: the inland Lake Trout (*Salvelinus namaycush*) fisheries of Ontario. Can J Fish Aquat Sci 55: 2161-2177.

Snucins EJ, Gunn JM. 1995. Coping with a warm environment: behavioral thermoregulation by Lake Trout. Trans Am Fish Soc 124: 118-123.

Sokal RR, Rohlf FJ. 1995. Biometry: The Principles and Practice of Statistics in Biological Sciences. WH Freeman and Company, New York, NY, USA.

Wickham, H. 2009. ggplot2: Elegant Graphics for Data Analysis. Springer New York.

SECTION 9

FISH TISSUE CHEMISTRY

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Plan
ANCOVA	analysis of covarience
$ANCOVA_{log}$	analysis of covariance on log_{10} transformed data
ANOVA	analysis of variance
$ANOVA_{log}$	analysis of variance on log_{10} transformed data
CFIA	Canadian Food Inspection Agency
De Beers	De Beers Canada Inc.
DL	detection limit
DQO	data quality objective
EAR	Environmental Assessment Report
EEM	Environmental Effects Monitoring
e.g.	for example
FIN	Fish Identification Number
i.e.	that is
IQR	interquartile range
K-W	Kruskall-Wallis
K-S	Kolmogorov-Smirnov
L13	Lake 13
LKTR	Lake Trout
Mine	Snap Lake Mine
M-W	Mann-Whitney
MVLWB	Mackenzie Valley Land and Water Board
n	sample count
nc	not calculated (more than 50%)
nd	not determined (at least 50% of data values were <DL)
ND	not detected
nt	not tested (at least 50% of data values were <DL for each group)
NEL	Northeast Lake
P	probability
p	p-value
QA	quality assurance
QC	quality control
RNWH	Round Whitefish
RPD	relative percent difference
SD	standard deviation
SL	Snap Lake
SR	studentized residual
t	two-sample t-test
t_{log}	two-sample t-test on log_{10} transformed data
USEPA	United States Environmental Protection Agency
WOE	weight of evidence
↑	increase
↓	decrease

UNITS OF MEASURE

Term	Definition
%	percent
>	greater than
<	less than
±	plus or minus
g	gram
mg/kg	milligrams per kilogram
mm	millimetre
ww	wet weight

9 FISH TISSUE CHEMISTRY

9.1 Introduction

In 2013, De Beers Canada Inc. (De Beers) implemented the field component of the Snap Lake Mine (Mine) Aquatic Effects Monitoring Program (AEMP), as required by Type A Water Licence MV2011L2-0004 (MVLWB 2013). The scope of the AEMP is based on the approved study design document submitted to the Mackenzie Valley Land and Water Board (MVLWB) on March 28, 2013, which was approved with conditions on November 29, 2013. A fish health survey was completed in July 2012, which targeted Lake Chub (*Couesius plumbeus*) and included a small-bodied fish tissue chemistry survey (De Beers 2013). The most recent large-bodied fish tissue survey was conducted in 2009 under the previous AEMP Design Plan (De Beers 2005). This section presents the results of the first large-bodied fish tissue chemistry survey conducted under the Mine's revised AEMP in 2013.

9.1.1 Background

The AEMP Design Plan was updated in 2012 and finalized in 2013 (herein referred to as the 2013 AEMP Design Plan; De Beers 2014). Lake Trout (*Salvelinus namaycush*) and Round Whitefish (*Prosopium cylindraceum*) were included in large-bodied fish surveys conducted in 1999, 2004, and 2009 to document fish tissue chemistry in Snap Lake (De Beers 2002, 2005, 2010). No patterns in large-bodied fish tissue chemistry were observed across these years, and no changes beyond predictions made in the Environmental Assessment Report (EAR) were observed in fish tissue chemistry (De Beers 2012). Lake Trout and Round Whitefish were retained as study species in the fish tissue chemistry program of the updated AEMP Design Plan in 2013.

A second reference lake was also proposed in the 2013 AEMP Design Plan (De Beers 2014). Subsequently, Reference Lake 13 (Lake 13) was added to the large-bodied fish tissue chemistry survey in 2013.

A small-bodied fish survey using Lake Chub (*Couesius plumbeus*) was also added to the fish tissue chemistry program in the 2013 AEMP Design Plan 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 2014). The results of the most recent small-bodied fish survey are provided in the 2012 Annual Report for the Snap Lake AEMP (De Beers 2013). The next scheduled small-bodied fish health and fish tissue program is in 2015.

The 2013 AEMP Design Plan requires the calculation of a "normal range" of fish tissue chemistry in the local study lakes to determine whether any changes observed in Snap Lake during the 2013 fish tissue chemistry survey were beyond the range of natural variability observed in the reference lakes and in Snap Lake before Mine development. Previous fish tissue chemistry surveys at Snap Lake (De Beers 2013) compared the Snap Lake mean to a normal range defined as the mean plus or minus (±) 2 standard deviations (SD); however, this approach has been reassessed. The normal range for fish tissue chemistry parameters is calculated herein as the range of tissue chemistry concentrations found pre-development and in local reference lakes, and is calculated for each parameter in each species of fish and each tissue type. The normal range calculation for the 2013 data is considered to be more sensitive and encompasses a narrower range of concentrations, thereby providing a better “early warning” of changes in fish tissue chemistry.

9.1.2 Objectives

The objective of the large-bodied fish tissue chemistry survey is to determine whether treated effluent discharged from the Mine is having an effect on fish tissue chemistry with the potential to limit fish use by humans. 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 (v) and 1d of MVLWB (2013)] 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 two key questions:

- Are tissue parameter concentrations in fish from Snap Lake increasing relative to baseline?
- Are tissue parameter concentrations in fish from Snap Lake increasing relative to reference lakes?

Fish tissue concentrations from 2013 are herein compared to baseline and reference lakes. An increase in tissue parameter concentrations in Lake Trout or Round Whitefish relative to baseline would provide an early warning of effects on fish usability. Temporal trends in tissue concentrations will be assessed in the 2016 AEMP Re-evaluation Report.

9.2 Methods

9.2.1 Fish Collection and Laboratory Analyses

Fish tissue samples were collected from adult fish sampled in the Fish Community monitoring (Section 8) from one exposure lake (Snap Lake) and two reference lakes (Northeast Lake and Lake 13), following the methods outlined in Section 8.2. Ten liver, kidney, and muscle tissues of selected Lake Trout and Round Whitefish were collected for each of the three lakes and submitted for tissue chemistry analyses; an additional Lake Trout was sampled from Northeast Lake, resulting in a sample size of 11 for Lake Trout liver, kidney, and muscle tissues for Northeast Lake.

Tissues were removed from Lake Trout and Round Whitefish, weighed, placed in separate zip-lock bags and labelled, including fish identification number, tissue type, and analyses required. Tissues were removed using a clean stainless steel filleting knife; the skin was removed from each muscle fillet. Contamination of samples was controlled by covering the work area with clean plastic wrap which was changed after each dissection; utensils were rinsed in 5 percent (%) nitric acid between fish to avoid cross contamination. Sex, length, and weight of each fish submitted for tissue chemistry analyses are provided in Table 9-1; further details are found in Section 8 and Appendix 8F.

Samples were analyzed by ALS Canada Ltd. (ALS; Burnaby, British Columbia) for metals[1] and lipid concentrations as listed in the 2013 AEMP Design Plan (De Beers 2014). The parameters analyzed (primarily metals) and their respective detection limits (DLs) are listed in Table 9-2; where the DL varied greatly among samples, the range of the DL is presented. Data are reported as milligrams per kilogram wet weight (mg/kg ww).

Table 9-1 Lake Trout and Round Whitefish Samples used in the Fish Tissue Chemistry Survey, 2013 AEMP

Fish Identification Number (FIN)	Species	Lake	Sex	Fork Length (mm)	Total Length (mm)	Total Body Weight (g)
SL-13-U-L13-LKTR-28001	Lake Trout	Lake 13	M	600	656	2,540
SL-13-U-L13-LKTR-28023			M	355	389	420
SL-13-U-L13-LKTR-28033			M	647	717	2,770
SL-13-U-L13-LKTR-28052			M	424	463	840
SL-13-U-L13-LKTR-28060			M	690	764	2,940
SL-13-U-L13-LKTR-28092			M	522	575	1,710
SL-13-U-L13-LKTR-28094			F	437	486	1,050
SL-13-U-L13-LKTR-28100			F	608	669	2,470
SL-13-U-L13-LKTR-28101			M	679	755	2,880
SL-13-U-L13-LKTR-28102			M	649	714	3,590
SL-13-U-NEL-LKTR-27001		Northeast Lake	M	650	713	3,090
SL-13-U-NEL-LKTR-27002			F	686	753	3,260
SL-13-U-NEL-LKTR-27003			F	501	550	1,630
SL-13-U-NEL-LKTR-27033			M	500	554	1,470
SL-13-U-NEL-LKTR-27035			M	710	774	3,500
SL-13-U-NEL-LKTR-27045			F	352	397	570
SL-13-U-NEL-LKTR-27074			M	566	627	2,320
SL-13-U-NEL-LKTR-27100			F	703	766	3,900
SL-13-U-NEL-LKTR-27104			F	745	806	5,140
SL-13-U-NEL-LKTR-27109			M	752	825	4,110
SL-13-U-NEL-LKTR-27125			M	761	849	5,160
SL-13-U-SL-LKTR-26001		Snap Lake	M	331	370	490
SL-13-U-SL-LKTR-26004			M	579	636	2,340
SL-13-U-SL-LKTR-26095			M	653	720	3,760

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

Table 9-1 Lake Trout and Round Whitefish Samples used in the Fish Tissue Chemistry Survey, 2013 AEMP

Fish Identification Number (FIN)	Species	Lake	Sex	Fork Length (mm)	Total Length (mm)	Total Body Weight (g)
SL-13-U-SL-LKTR-26103	Lake Trout	Snap Lake	F	402	440	690
SL-13-U-SL-LKTR-26127			F	619	692	2,760
SL-13-U-SL-LKTR-26162			M	703	752	3,940
SL-13-U-SL-LKTR-26176			M	801	875	7,420
SL-13-U-SL-LKTR-26183			F	569	629	2,160
SL-13-U-SL-LKTR-26225			M	674	743	4,160
SL-13-U-SL-LKTR-26229			F	585	640	2,360
SL-13-U-L13-RNWH-28003	Round Whitefish	Lake 13	F	299	336	300
SL-13-U-L13-RNWH-28027			M	231	254	114
SL-13-U-L13-RNWH-28029			F	236	260	130
SL-13-U-L13-RNWH-28030			M	248	274	170
SL-13-U-L13-RNWH-28035			F	189	208	73
SL-13-U-L13-RNWH-28049			M	194	211	64
SL-13-U-L13-RNWH-28071			F	215	237	84
SL-13-U-L13-RNWH-28085			F	285	311	240
SL-13-U-L13-RNWH-28095			M	216	230	90
SL-13-U-NEL-RNWH-27004	Round Whitefish	Northeast Lake	F	261	284	170
SL-13-U-NEL-RNWH-27007			F	275	300	210
SL-13-U-NEL-RNWH-27020			M	220	242	103
SL-13-U-NEL-RNWH-27043			M	279	303	190
SL-13-U-NEL-RNWH-27046			M	189	207	69
SL-13-U-NEL-RNWH-27078			M	199	218	72
SL-13-U-NEL-RNWH-27103			F	239	261	130
SL-13-U-NEL-RNWH-27108			F	223	244	92
SL-13-U-NEL-RNWH-27140			F	290	316	240
SL-13-U-NEL-RNWH-27141			F	276	302	210
SL-13-U-SL-RNWH-26002		Snap Lake	M	274	298	230
SL-13-U-SL-RNWH-26003			F	217	239	105
SL-13-U-SL-RNWH-26018			F	183	200	55
SL-13-U-SL-RNWH-26097			F	269	293	200
SL-13-U-SL-RNWH-26099			M	242	265	160
SL-13-U-SL-RNWH-26131			F	276	302	210
SL-13-U-SL-RNWH-26164			M	283	310	240
SL-13-U-SL-RNWH-26168			M	299	326	270
SL-13-U-SL-RNWH-26170			M	227	249	130
SL-13-U-SL-RNWH-26230			M	204	224	83

SL = Snap Lake; NEL = Northeast Lake; LK13 = Lake 13; LKTR = Lake Trout; RNWH = Round Whitefish; mm = millimetres; g = grams.

Table 9-2 Analytical Detection Limits for Kidney, Liver and Muscle Tissue in Lake Trout and Round Whitefish, 2013

Parameter	Detection Limit (mg/kg ww, unless otherwise noted)		
	Kidney	Liver	Muscle
% Moisture	0.1	0.1	0.1
Aluminum (Al)	0.4	0.4	0.4
Antimony (Sb)	0.002	0.002	0.002
Arsenic (As)	0.004	0.004	0.004
Barium (Ba)	0.01	0.01/0.02[a]	0.01
Beryllium (Be)	0.002/0.004[a]	0.002/0.004[a]	0.002
Bismuth (Bi)	0.002/0.004[a]	0.002/0.004[a]	0.002
Boron (B)	0.2/0.4[a]	0.2/0.4[a]	0.2
Cadmium (Cd)	0.002	0.002	0.002
Calcium (Ca)	0.5/5	0.5/5	0.5/5
Cesium (Cs)	0.001	0.001	0.001
Chromium (Cr)	0.01	0.01	0.01
Cobalt (Co)	0.004	0.004	0.004
Copper (Cu)	0.01	0.01	0.01
Gallium (Ga)	0.004/0.008[a]	0.004/0.008[a]	0.004
Iron (Fe)	0.2	0.2	0.2
Lead (Pb)	0.004	0.004/0.008[a]	0.004
Lithium (Li)	0.02/0.04[a]	0.02/0.04[a]	0.02
Magnesium (Mg)	1/10	1/10	1/10
Manganese (Mn)	0.004	0.004	0.004
Mercury (Hg) – Lake Trout	0.001 to 0.1	0.001 to 0.12	0.001 to 0.01
Mercury (Hg) – Round Whitefish	0.001 to 0.1	0.001 to 0.01	0.001 to 0.008
Molybdenum (Mo)	0.004	0.004	0.004
Nickel (Ni)	0.01	0.01	0.01
Phosphorus (P)	5/50	5/50	5/50
Potassium (K)	20/200	20/200	20/200
Rhenium (Re)	0.002/0.004[a]	0.002/0.004[a]	0.002
Rubidium (Rb)	0.01	0.01	0.01
Selenium (Se)	0.02	0.02	0.02
Silver (Ag)	0.001/0.002[a]	0.001	0.001
Sodium (Na)	20/200	20/200	20/200
Strontium (Sr)	0.01	0.01	0.01
Tellurium (Te)	0.004/0.008[a]	0.004/0.008[a]	0.004
Thallium (Tl)	0.0004	0.0004	0.0004
Thorium (Th)	0.002/0.004[a]	0.002/0.004[a]	0.002
Tin (Sn)	0.02/0.04[a]	0.02/0.04	0.02
Uranium (U)	0.0004	0.0004/0.0008[a]	0.0004
Vanadium (V)	0.02/0.04[a]	0.02/0.04[a]	0.02
Yttrium (Y)	0.002/0.004[a]	0.002/0.004[a]	0.002
Zinc (Zn)	0.1	0.1	0.1

Table 9-2 Analytical Detection Limits for Kidney, Liver and Muscle Tissue in Lake Trout and Round Whitefish, 2013

Parameter	Detection Limit (mg/kg ww, unless otherwise noted)		
Zirconium (Zr)	0.04/0.08[(a)]	0.04/0.08[(a)]	0.04
% Lipid Content – Lake Trout	0.5 to 6.8	0.5 to 1.22	0.5 to 0.97
% Lipid Content – Round Whitefish	-	0.5 to 5.49	0.5 to 0.93

a) One sample was analyzed at the higher detection limit due to sample heterogeneity that interfered with achieving the desired analytical detection limit.

% = percent; mg/kg ww = milligrams per kilogram wet weight; - = not analyzed due to insufficient sample size.

9.2.2 Data Analyses

Statistical analyses were performed to assess changes in fish tissue chemistry and used in the Weight of Evidence (WOE) assessment (Section 12). The results of the statistical comparisons and normal range comparisons were used to determine whether any results required follow-up action (Section 13).

9.2.2.1 Data Handling

Values reported below the laboratory DL were reviewed prior to use in any calculations. When a majority of the concentrations for a parameter were below the DL (i.e., greater than [>] 50% of results less than [<] DL), the mean and SD were not calculated and the results for that parameter were reported as "not determined." When a majority of the concentrations for a parameter were above the DL (i.e., >50% of results >DL), the non-detects were substituted with one-half the DL (USEPA 2000) for the calculation of the mean and SD, and for statistical comparisons. When a majority of the concentrations for a parameter were below the DL in one lake, but a majority of the concentrations for the same parameter were above the DL in another lake, statistical comparisons were conducted using a non-parametric test that tested for differences in the median instead of the mean (Section 9.2.2.2).

Descriptive statistics, including sample size, DL, minimum, maximum, median, arithmetic mean, and SD were reported by fish species and lake for each parameter.

Censored Box Plots

Fish tissue chemistry data were plotted by species and lake (Appendix 9D). A boxplot was used to plot a data set with eight or more observations, while individual values were plotted when sample sizes were less than eight or when few or no concentrations were above the DL. This approach was used because boxplots can provide a misleading representation of the data distribution when sample sizes are small. The boxes in the box plots were defined using the 25th percentile, the 75th percentile, and the median. The lower whisker was defined as the minimum concentration within 1.5 times the interquartile range (IQR) below the 25th percentile; the upper whisker was defined as the maximum concentration within 1.5 times the IQR above the 75th percentile. Concentrations outside of the range of the whiskers were plotted as individual values. The boxplots were censored[2] at the maximum DL, such that concentrations below the DL are not shown (Helsel 2005). The approximate proportion (e.g., <25% or 25% to 50%) of concentrations below detection can be determined by the sections of the boxplot that are censored. When individual values were plotted, concentrations below the DL were represented by open symbols at half the DL.

9.2.2.2 Statistical Analyses

Statistical analyses were conducted to test for differences among sampling areas in the concentrations of parameters measured in Lake Trout and Round Whitefish. Statistical analyses were also conducted to test for differences in concentrations between Snap Lake in 2013 and pooled reference or baseline lakes, and Snap Lake at baseline (i.e., 1999 and 2004 Snap Lake data, when available).

Four statistical comparisons were conducted:

- 2013 Snap Lake mean compared to the individual 1999 and 2004 Snap Lake means for muscle tissue (i.e., baseline mean; Section 9.4.3);
- 2013 Northeast Lake mean compared to 2013 Lake 13 mean to determine whether reference data could be pooled (Section 9.4.1);
- 2013 Snap Lake mean compared to the pooled 2013 reference mean (Section 9.4.1); and,
- 2013 Snap Lake mean compared to the individual 2013 reference lake means when reference lakes could not be pooled (Section 9.4-1).

Statistical tests were considered significant at an alpha of 0.1 (i.e., probability [*P*] <0.1).

Prior to statistical analyses, tests were conducted to assess the assumptions of normality and equality of variances for parametric tests. All concentrations were $\log_{10}$ transformed and subsequently tested as both raw (untransformed) data and $\log_{10}$ transformed data. The goodness of fit of each data set to a normal distribution was tested using a Kolmogorov-Smirnov (K-S) test. The assumption of group variances being equal was tested using Levene's test.

Mercury and selenium tend to biomagnify (i.e., accumulate to a higher degree in top trophic level species, and larger/older individuals). The relationship between mercury or selenium and body size (i.e., fish length) was assessed using linear regression. When the regression was found to be significant (P <0.05), length was included as a covariate in the statistical analysis. A second analysis was performed if the regression was significant, where measured concentrations of mercury and selenium were extrapolated for a fish of a standard length of 600 millimetres (mm) for Lake Trout and 240 mm for Round Whitefish,

[2] Data are considered "censored" when the value of a measurement is only partially known; values below analytical detection limits are a primary example of censored data.

which correspond to the mean length for each species, rounded to the nearest tenth. Additional information is provided in Appendix 9A regarding the predicted mercury and selenium calculations for a fish of standard length (i.e., $mercury^{predicted}$ and $selenium^{predicted}$). Statistical tests were conducted on predicted and measured mercury and selenium concentrations. Measured mercury and selenium concentrations were tested using length as a covariate. If the mercury-length or selenium-length regression relationships were not significant, statistical analysis was only performed on mercury and selenium concentrations (i.e., without a covariate), and predicted concentrations were not calculated.

Studentized residuals (SR) were used as a screening tool for identifying statistical outliers. Observations with SR>|4| were considered statistical outliers and were removed from relevant statistical tests. Statistical testing was performed only on outlier-removed datasets. Statistical analyses were conducted using the Minitab 16 software (Minitab Inc., State College, Pennsylvania).

Two-Sample t-tests

Two-sample t-tests were used to assess differences between the 2013 Snap Lake mean concentration and the 2013 pooled reference mean concentration for each parameter. When the assumptions of normality and equality of variances could not be met (even after $\log_{10}$ transformation), the non-parametric Mann-Whitney test was conducted to assess differences in median concentrations between areas.

Analysis of Variance

An analysis of variance (ANOVA) was used to assess differences among the 2013 Snap Lake mean concentration and the individual 2013 reference lake mean concentrations for each parameter, including $mercury^{predicted}$ and $selenium^{predicted}$. When the overall ANOVA was significant (P <0.1), pairwise comparisons (Tukey's honestly significant differences method) were used to determine which sampling areas were different. When the assumptions of normality and equality of variances could not be met (even after $\log_{10}$ transformation), the non-parametric Kruskal-Wallis (K-W) test was conducted to assess differences in median concentrations between areas. When the overall K-W test was significant (P <0.1), pairwise comparisons (Dunn's Method; Dunn 1964) were used to determine which sampling areas were different.

Analysis of Covariance

Analysis of covariance (ANCOVA) was used to assess differences in concentrations of mercury and selenium among sampling areas when body size was identified as a significant covariate. When fish length was not identified as a significant predictor of selenium or mercury concentrations, total body weight was assessed as a potential covariate. When total body weight was not identified as a significant predictor of selenium or mercury concentrations, differences among sampling areas in parameter concentrations were tested using the relevant two-sample t-test or ANOVA.

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 (P <0.05) between sampling lakes and the covariate (i.e., assumption of homogeneity of slopes was satisfied), then an ANCOVA was performed and predicted means were calculated.

If a significant difference (P <0.1) was detected in the ANCOVA analysis for differences among the 2013 Snap Lake mean concentration and the reference site mean concentration, pairwise comparisons (Tukey's honestly significant differences method) were conducted to determine which sampling areas were different. When the assumptions of normality and equality of variances could not be met, a rank ANCOVA (Conover and Iman 1982) was conducted. The rank ANCOVA is the same as the ANCOVA, with data values substituted by their ranks.

Magnitude of Difference

The magnitude of the difference between Snap Lake and the reference lakes for parameters analyzed without a covariate was calculated by expressing the difference as a percentage of the mean of the pooled reference lakes as follows:

$$Magnitude\ Difference\ (\%) = \frac{(Exposure\ Mean - Pooled\ Reference\ Mean)}{Pooled\ Reference\ Mean} \times 100.$$ [Equation 9-1]

The magnitude of the difference between reference lakes analyzed without a covariate was calculated as the relative percent difference (RPD) as follows:

$$RPD\ (\%) = \frac{|Reference\ 1\ Mean - Reference\ 2\ Mean|}{Pooled\ Reference\ Mean} \times 100.$$ [Equation 9-2]

The magnitude of the differences between Snap Lake and the reference lakes for parameters analyzed with a covariate (i.e., calculated using predicted means) were calculated as follows:

$$Magnitude\ Difference\ (\%) = \frac{(Exposure\ Mean_{adjusted} - Reference\ Mean_{adjusted})}{Reference\ Mean_{adjusted}} \times 100.$$ [Equation 9-3]

The magnitude of the difference between reference lakes analyzed with a covariate was calculated as the RPD as follows:

$$RPD\ (\%) = \frac{|Reference\ 1\ Mean_{adjusted} - Reference\ 2\ Mean_{adjusted}|}{Pooled\ Reference\ Mean_{adjusted}} \times 100.$$ [Equation 9-4]

When the concentrations were $\log_{10}$ transformed for statistical analysis, the RPD and magnitude difference were calculated using the antilog (i.e., transformed as 10^x) of the mean calculated on the $\log_{10}$ transformed concentrations.

9.2.2.3 Normal Range

The details of the calculation of the normal range are found in Appendix 9A. A summary of the calculation is provided below. The data used in the calculation of normal range for muscle tissue involved data from the reference lakes in 1999, 2004, 2009, and 2013 and in Snap Lake in 1999 and 2004. For kidney, the normal range is calculated using only the 2013 reference lake data, as there are no baseline kidney tissue chemistry data. For liver, the normal range is calculated using the 2013 reference lake data and the 1999 Snap Lake, and reference lake data for Lake Trout. The 1999 Round Whitefish liver tissue concentrations were measured on composite samples and were not included in the normal range calculations. The reference lakes included in the normal range calculations were Northeast Lake, Lake 13, and a reference lake from the baseline surveys known as “Reference Lake.”

The normal range was not calculated when all concentrations were below the DL. When at least one concentration for a given parameter was above the DL, and the concentrations for that parameter were also detected in 2013 samples, the non-detected samples from 1999, 2004, or 2009 were randomly assigned a concentration based on the shape of the 2013 data distribution. This was done such that historical data with varying DLs could be included in the normal range calculation (see Appendix 9A for details). If fewer than 50% of the concentrations were above the DL in every available year for the data used in the calculation of the normal range (i.e., including 2013 samples), then a normal range was not defined.

Normal range was calculated using a 95% prediction interval for the mean of the sample size achieved in Snap Lake in 2013 (i.e., sample size of 10). If data were not normally distributed, transformations were performed to achieve normality (i.e., logarithm or Box-Cox power transformation). When normality could not be achieved after a data transformation, a resampling technique was used to obtain an estimate of the normal range for the mean of a sample size achieved in Snap Lake in 2013 (Appendix 9A, Section 9B1.3.2). Mean concentrations of each parameter in Snap Lake in 2013 were compared to the calculated normal range for the mean (based on means and normal ranges calculated on a transformed scale) (Appendix 9A). Mean concentrations in Snap Lake in 2013 above the normal range for the mean were noted as normal range exceedances.

9.2.2.4 Guideline Comparison

Metal concentrations in the muscle tissue of Lake Trout and Round Whitefish were compared to available national guidelines for human consumption, where guidelines were available. The Canadian Food Inspection Agency (CFIA) and Health Canada guidelines state fish collected for commercial use may contain a maximum of 0.5 mg/kg 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 mg/kg ww in fish tissue for human consumption, while lead must be below 0.5 mg/kg ww (CFIA 2009). Fish from Snap Lake are not sold commercially; therefore, these guidelines are considered for comparison purposes only and are not considered within the AEMP response framework.

9.2.2.5 Action Levels

Following the 2013 AEMP Design Plan (De Beers 2014), fish tissue chemistry data were reviewed to determine whether any Low Action Levels were triggered. For fish tissue, the Low Action Levels were defined as follows:

- Significance Threshold – Fish Safe to Eat:
  - **Low Action Level:** Mean fish tissue concentration outside of the normal range.
- Significance Threshold – Ecological Integrity Maintained:
  - **Low Action Level:** Statistically significant difference (P <0.1) in fish tissue chemistry that is beyond the normal range *and* change is in direction and of magnitude indicative of impairment to fish health.

Herein a statistically significant difference in fish tissue chemistry is defined as a difference from reference lakes and, where baseline data were available[3], a difference from baseline. This interpretation of “statistically significant difference” is a deviation from the definition provided in the 2013 AEMP Design Plan (De Beers 2014), where only comparison to reference lakes was considered. This change was made to place equal importance on differences from baseline as on differences from reference lakes in accordance with the key questions of the study. This change was necessary given the updated approach to calculating normal range for fish tissue chemistry, which compares the 2013 Snap Lake mean to a prediction interval; the normal range includes both reference lake and baseline data (Section 9.2.2.3 and Appendix 9A).

The inclusion of both baseline and reference lake data and the revised calculation of the normal range required a modification to the definition of the Low Action Level. If the concentrations of fish tissue parameters from Snap Lake were significantly different than reference lakes, but were not different than baseline, they were not considered further as potential Low Action Level triggers. A Low Action Level was triggered only if the Snap Lake 2013 parameter concentration was statistically different than both the reference lakes and baseline, and the mean was outside the normal range (as defined in Section 9.2.2.3 and Appendix 9A).

Revised Action Level for Significance Threshold – Fish Safe to Eat and Ecological Integrity Maintained

- **Low Action Level:** Muscle tissue is statistically different in Snap Lake from the reference lakes and baseline, and the mean is outside the normal range.

Muscle parameters with concentrations that were significantly different than reference lakes and baseline, and outside the normal range, were considered Low Action Level triggers that may require follow-up action (Section 9.5.1). Lake Trout and Round Whitefish liver and kidney data were not considered Low Action Level triggers due to limited or absent baseline and reference lake data for these tissue types.

---

[3] Muscle chemistry Low Action Level triggers were considered using reference lake and baseline data.

Kidney and liver data were, however, used to support the review and interpretation of muscle Low Action Level triggers (Section 9.5.1).

9.2.2.6 Other Data Considerations

A qualitative comparison of Snap Lake 2013 summary statistics with summary statistics from other years (i.e., 1999, 2004 or 2009 data) is provided in Section 9.5.2.

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. Historical data were reviewed for consistency of units of measurement (i.e., mg/kg and wet weight versus dry weight) and of DLs before inclusion in the 2013 data set.

9.3.1 Overview of Procedures

Field and laboratory equipment was calibrated throughout the field program following manufacturer's 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 preprinted 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 shipment and receipt of samples.

Laboratory QA/QC included analysis of a series of sample blanks, spikes, and duplicates. Maximum allowable differences in sample parameters were reported and any data quality objective (DQO) exceedances were noted in the laboratory results report.

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 unrealistic data values. If a data entry error was found, data underwent a zero tolerance QC check, where every datum was checked.

Results of statistical analyses were independently reviewed by a qualified scientist with appropriate technical qualifications. Tables containing data summaries and statistical results were reviewed, and values verified by a second, independent, individual.

9.3.2 Summary of Quality Assurance/ Quality Control Results

The QA/QC review provided the following information:

- Lipids data were unavailable for some tissue samples due to sample volume constraints.
- A number of method blanks had elevated concentrations of nickel and zinc, which corresponded with concentrations in tissue samples; re-digestion was not possible due to limited sample volume. However, these results were not inconsistent with similar tissue types from other, unaffected digestion batches and thus the tissue sample data were considered valid.
- There was evidence of heterogeneity in some of the kidney and liver samples, but homogenization procedures applied to these samples met laboratory criteria for applicability.

Data quality objectives were established based on an average tissue type, and were not specifically derived for each individual tissue type; data that did not meet DQOs were excluded from the data analyses. Data that were excluded were one Lake Trout liver sample for chromium, one Round Whitefish liver sample for chromium and lead, and one Lake Trout muscle sample for chromium (Appendix 9B). The remaining data were accepted and included in subsequent data analyses, subject to statistical screening methods as outlined in Section 9.2.2.1.

9.4 Results

Summary statistics and statistical comparisons for the 2013 fish tissue chemistry analyses were completed for Lake Trout and Round Whitefish from Snap Lake, Northeast Lake, and Lake 13 including comparisons to baseline and reference data, where applicable (Tables 9-3 to 9-8). Additional supporting information provided in appendices is listed below:

- Appendix 9A – Normal Range Determination;
- Appendix 9B – Fish Tissue Chemistry Data;
- Appendix 9C – Summary Statistics for Fish Tissue Chemistry Data; and,
- Appendix 9D – Fish Tissue Chemistry Summary Plots.

Appendix 9A provides a detailed description of how the normal range was calculated for all parameters, and includes examples and results summary tables of normal ranges. Appendix 9B contains the analytical data. Appendix 9C contains the summary statistics for the reference lakes, including minimum, maximum, median, mean, and SD. Statistical outliers that were removed from the statistical tests and calculation of normal range are also presented in Appendix 9C. Appendix 9D contains graphical plots for each parameter, grouped by tissue type.

9.4.1 Baseline Comparisons

Statistical analyses were conducted to test for differences in parameter concentrations in Lake Trout and Round Whitefish muscle tissue in Snap Lake in 2013 relative to Snap Lake in 1999 and 2004, where sufficient data existed.

Lake Trout

Concentrations of parameters in Lake Trout muscle tissue that were significantly lower in 2013 relative to baseline were:

- Cesium, copper, iron, mercury$^{\text{predicted}}$, rubidium, and selenium (Table 9-7).

Concentrations of parameters in Lake Trout muscle tissue that were significantly higher in 2013 compared to baseline were:

- Phosphorus, potassium, and strontium (Table 9-7).

Round Whitefish

Concentrations of parameters in Round Whitefish muscle tissue that were significantly lower in 2013 compared to baseline were:

- Mercury$^{\text{predicted}}$, nickel, rubidium, selenium, and zinc (Table 9-8).

Concentrations of parameters in Round Whitefish muscle tissues that were significantly higher in 2013 compared to baseline were:

- Cesium, sodium, and strontium (Table 9-8).

Table 9-3 Summary Statistics and Statistical Comparisons to Reference Sites and Normal Range for Lake Trout Kidney Tissue Collected from Snap Lake in 2013

	2013 Snap Lake Summary Statistics								Comparisons to Reference																Normal Range Comparison				
									2013 NEL vs 2013 Lake 13			2013 Snap Lake vs 2013 Pooled Reference				2013 Snap Lake vs 2013 Lake 13				2013 Snap Lake vs 2013 NEL			2013 Snap	Normal Range	% of 2013 Reference	Normal Range[b]	
Parameter	n	DL[a]	%>DL	Minimum[a]	Maximum[a]	Median[a]	Mean[a]	SD[a]	Test	p	%	Test	p	↑/↓	%	Test	p	↑/↓	%	p	↑/↓	%	Back Transformed Mean[a]	for Mean of sample of size n[a]	Data >DL	Exceedance	Below
Aluminum	10	0.40	100	0.62	2.64	1.35	1.54	0.77	t	0.066	33	-	-	-	-	ANOVA	0.024	↓	-40	0.685	-	-16	0.803	(0.734, 0.910)	-	-	-
Antimony	10	0.0020	40	<0.0020	0.0052	<0.0020	nd	nd	MW	0.009	nc	-	-	-	-	KW	0.082	↓	nc	0.364	-	nc	-	-	-	-	-
Arsenic	10	0.0040	100	0.008	0.0277	0.0167	0.0166	0.0063	t_{log}	0.001	88	-	-	-	-	$ANOVA_{log}$	<0.001	↓	-76	0.031	↓	-49	0.0166	(0.0240, 0.0754)	43	-	X
Barium	10	0.010	100	0.012	0.224	0.033	0.069	0.075	MW	0.596	13	MW	0.139	-	26	-	-	-	-	-	-	-	0.029	(0.037, 0.062)	-	-	X
Beryllium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Bismuth	10	0.0020	50	<0.0020	0.0037	<0.0020	nd	nd	t	0.419	16	t	0.005	↓	-46	-	-	-	-	-	-	-	nd(<0.0024)	(0.0022, 0.0047)	-	-	-
Boron	10	0.20	0	<0.20	<0.20	<0.20	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Cadmium	10	0.002	100	0.146	1.240	0.359	0.411	0.315	t_{log}	0.028	62	-	-	-	-	KW	0.768	-	21	0.256	-	-35	0.411	(0.239, 0.693)	-	-	-
Calcium	10	0.5/5	100	98	212	141	145	34	t_{log}	0.712	7	t_{log}	0.387	-	-11	-	-	-	-	-	-	-	142	(116, 219)	-	-	-
Cesium	10	0.0010	100	0.0174	0.0909	0.0754	0.0711	0.0211	t	0.345	19	t	0.210	-	21	-	-	-	-	-	-	-	0.0711	(0.0377, 0.0800)	-	-	-
Chromium	10	0.010	70	<0.010	0.220	0.022	0.038	0.065	t	0.437	21	t_{log}	0.001	↓	-70	-	-	-	-	-	-	-	0.038	(0.036, 0.103)	-	-	-
Cobalt	10	0.0040	100	0.0564	0.2240	0.1065	0.1137	0.0504	t_{log}	0.218	29	t_{log}	0.633	-	-9	-	-	-	-	-	-	-	0.1021	(0.0751, 0.1705)	-	-	-
Copper	10	0.010	100	0.670	0.963	0.787	0.803	0.096	t_{log}	0.254	1	t	0.615	-	-6	-	-	-	-	-	-	-	0.803	(0.734, 0.910)	-	-	-
Gallium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Iron	10	0.2	100	69	331	144	164	74	t_{log}	0.447	19	t_{log}	0.126	-	-27	-	-	-	-	-	-	-	164	(137, 334)	-	-	-
Lead	10	0.0040	20	<0.0040	0.01	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	29	-	-
Lithium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	14	-	-
Magnesium	10	1/10	100	110	148	130	129	13	MW	0.805	2	t_{log}	0.754	-	-1	-	-	-	-	-	-	-	128	(118, 144)	-	-	-
Manganese	10	0.004	100	0.253	0.480	0.348	0.354	0.075	t	0.254	17	t_{log}	0.369	-	-9	-	-	-	-	-	-	-	0.354	(0.301, 0.491)	-	-	-
Mercury	10	0.001 to 0.010	100	0.093	2.310	0.807	1.013	0.746	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mercury[predicted (c)]	10	0.001 to 0.010	100	0.383	1.126	0.864	0.852	0.223	$ANCOVA_{log}$	0.016	55	-	-	-	-	$ANCOVA_{log}$	0.042	↑	51	0.540	-	-12	0.819	(0.490, 1.108)	-	-	-
Molybdenum	10	0.0040	100	0.0305	0.0875	0.0523	0.0520	0.0182	t_{log}	0.695	6	t_{log}	0.880	-	2	-	-	-	-	-	-	-	0.0472	(0.0369, 0.0612)	-	-	-
Nickel	10	0.010	100	0.097	0.677	0.292	0.290	0.176	t	0.005	63	-	-	-	-	ANOVA	0.785	-	21	0.064	↓	-37	0.290	(0.202, 0.511)	-	-	-
Phosphorus	10	5/50	100	1910	2580	2310	2297	236	logt	0.860	1	t_{log}	0.791	-	1	-	-	-	-	-	-	-	2297	(2003, 2552)	-	-	-
Potassium	10	20/200	100	2260	2820	2410	2490	207	t	0.438	5	t	0.977	-	0	-	-	-	-	-	-	-	2490	(2176, 2796)	-	-	-
Rhenium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Rubidium	10	0.01	100	4.84	7.26	5.90	5.91	0.75	t	0.023	28	-	-	-	-	$ANOVA_{log}$	0.062	↓	-39	0.722	-	-18	5.91	(6.63, 10.63)	-	-	X
Selenium	10	0.020	100	0.865	4.220	2.195	1.989	0.977	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Selenium[predicted (c)]	10	0.0010	100	1.201	3.570	1.903	2.003	0.690	$ANCOVA_{log}$	0.152	22	$ANCOVA_{log}$	0.112	-	-19	-	-	-	-	-	-	-	1.909	(1.806, 3.069)	-	-	-
Silver	10	0.0010	70	<0.0010	0.0033	0.0019	0.0019	0.0011	t_{log}	0.071	66	-	-	-	-	$ANOVA_{log}$	0.190	-	-46	0.997	-	3	0.0015	(0.0010, 0.0038)	-	-	-
Sodium	10	20/200	100	1430	2470	2290	2157	320	t	0.567	3	MW	0.057	↑	7	-	-	-	-	-	-	-	2157	(1846, 2194)	-	-	-
Strontium	10	0.010	100	0.417	0.952	0.695	0.676	0.192	t_{log}	0.436	12	t_{log}	0.001	↑	58	-	-	-	-	-	-	-	0.626	(0.316, 0.519)	-	X	-
Tellurium	10	0.0040	40	<0.0040	0.0174	<0.0040	nd	nd	MW	0.028	nc	-	-	-	-	KW	0.638	-	nc	0.086	-	nc	-	-	57	-	-
Thallium	10	0.0004	100	0.0104	0.0483	0.0332	0.0320	0.0124	t	0.070	27	-	-	-	-	$ANOVA_{log}$	0.005	↑	76	0.123	-	33	0.0320	(0.0153, 0.0271)	-	X	-
Thorium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Tin	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Uranium	10	0.00040	100	0.00065	0.00259	0.00190	0.00171	0.00068	MW	0.526	10	t_{log}	0.302	-	-23	-	-	-	-	-	-	-	0.00152	(0.00114, 0.00344)	-	-	-
Vanadium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	14	-	-
Yttrium	10	0.0020	30	<0.0020	0.004	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	38	-	-
Zinc	10	0.1	100	15.8	40.7	19.8	22.5	7.6	t_{log}	0.131	12	MW	0.459	-	10	-	-	-	-	-	-	-	19.8	(17.7, 22.1)	-	-	-
Zirconium	10	0.040	0	<0.040	<0.040	<0.040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 600 mm.

n = sample count; DL = detection limit;% = percent; "<" = less than; ">" = greater than; "-" = not tested; nd = not determined (at least 50% of data values were <DL); nc = not calculated (more than 50% of values were <DL for at least one group); nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; $ANOVA_{log}$ = analysis of variance on log_{10} transformed data; $ANCOVA_{log}$ = analysis of covariance on log_{10} transformed data; K-W = Kruskal-Wallis test; M-W = Mann Whitney test; NEL = Northeast Lake; SD = standard deviation; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data;↑/↓= statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline; X = present .

Table 9-4 Summary Statistics and Statistical Comparisons to Reference Sites and Normal Range for Round Whitefish Kidney Tissue Collected from Snap Lake in 2013

	2013 Snap Lake Summary Statistics								Comparisons to Reference														Normal Range Comparison				
									2013 NEL to 2013 Lake 13			2013 Snap Lake vs 2013 Pooled Reference				2013 Snap Lake vs 2013 Lake 13				2013 Snap Lake vs 2013 NEL			2013 Snap	Normal Range	% of 2013 Reference	Normal Range[b]	
Parameter	n	DL[a]	%>DL	Minimum[a]	Maximum[a]	Median[a]	Mean[a]	SD[a]	Test	p	%	Test	p	↑/↓	%	Test	p	↑/↓	%	p	↑/↓	%	Back Transformed Mean[a]	for Mean of sample of size n[a]	Data >DL	Exceedance	Below
Aluminum	10	0.40/0.08	100	1.10	9.08	1.76	2.44	2.37	t_{log}	0.933	3	MW	0.148	-	37	-	-	-	-	-	-	-	1.85	(0.78, 2.28)	-	-	-
Antimony	10	0.0020/0.0040	30	<0.0020	0.0059	<0.0020	nd	nd	MW	0.060	nc	-	-	-	-	KW	0.660	-	nc	0.150	-	nc	-	-	32	-	-
Arsenic	10	0.0040/0.0080	100	0.0103	0.0559	0.0240	0.0270	0.0138	t	0.088	27	-	-	-	-	ANOVA	0.335	-	-22	0.990	-	3	0.0270	(0.0215, 0.0389)	-	-	-
Barium	10	0.010/0.020	100	0.051	0.339	0.096	0.116	0.086	t	0.350	22	t_{log}	<0.001	↓	-62	-	-	-	-	-	-	-	0.116	(0.171, 0.407)	-	-	X
Beryllium	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	
Bismuth	10	0.0020/0.0040	60	<0.0020	0.0030	0.0023	0.0020	0.0008	MW	0.357	24	MW	0.006	↓	-49	-	-	-	-	-	-	-	0.0020	(0.0023, 0.0057)	-	-	X
Boron	10	0.20/.040	0	<0.20	<0.20	<0.20	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	21	-	-
Cadmium	10	0.0020/0.040	100	0.132	0.835	0.356	0.437	0.237	MW	0.025	59	-	-	-	-	KW	0.071	↑	66	0.599	-	-10	0.437	(0.173, 0.586)	-	-	-
Calcium	10	5	100	78	2930	284	554	856	t_{log}	0.089	95	-	-	-	-	$ANOVA_{log}$	0.033	↓	-74	0.660	-	-35	280	(263, 1786)	-	-	-
Cesium	10	0.0010/0.0020	100	0.0126	0.0397	0.0267	0.0276	0.0079	t	0.192	20	t	0.002	↑	47	-	-	-	-	-	-	-	0.0276	(0.0138, 0.0238)	-	X	-
Chromium	10	0.010/0.020	80	<0.010	0.076	0.031	0.036	0.024	MW	0.504	53	t_{log}	0.141	-	-42	-	-	-	-	-	-	-	0.020	(0.023, 0.073)	-	-	X
Cobalt	10	0.0040/0.0080	100	0.266	0.841	0.466	0.498	0.185	t	0.526	18	t	0.045	↑	48	ANOVA	0.553	-	64	0.488	-	36	0.498	(0.169, 0.502)	-	-	-
Copper	10	0.010/0.020	100	0.903	1.800	1.090	1.205	0.292	t	0.068	33	-	-	-	-	-	-	-	-	-	-	-	1.205	(0.826, 1.619)	-	-	-
Gallium	10	0.0040/0.0080	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Iron	10	0.2/0.4	100	127	291	227	216	57	t	0.167	23	t	0.065	↑	26	-	-	-	-	-	-	-	216	(120, 221)	-	-	-
Lead	10	0.0040/0.0080	100	0.0047	0.0409	0.0106	0.0134	0.0108	t	0.892	2	t_{log}	0.555	-	12	-	-	-	-	-	-	-	0.0134	(0.0072, 0.0131)	-	X	-
Lithium	10	0.020/0.040	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	26	-	-
Magnesium	10	10	100	103	240	180	176	38	t	0.947	1	t	0.182	-	-10	-	-	-	-	-	-	-	176	(167, 223)	-	-	-
Manganese	10	0.004/0.0080	100	0.273	1.550	0.458	0.652	0.414	t_{log}	0.028	54	-	-	-	-	$ANOVA_{log}$	0.012	↓	-52	0.654	-	-18	0.549	(0.562, 1.363)	-	-	X
Mercury	10	0.001 to 0.010	100	0.066	0.352	0.162	0.182	0.088	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mercury[predicted(c)]	10	0.001 to 0.010	100	0.084	0.241	0.169	0.164	0.047	$ANCOVA_{log}$	0.177	28	$ANCOVA_{log}$	0.054	↑	37	-	-	-	-	-	-	-	0.157	(0.080, 0.163)	-	-	-
Molybdenum	10	0.0040/0.0080	100	0.052	0.203	0.126	0.132	0.047	t	0.149	31	t	<0.001	↑	91	-	-	-	-	-	-	-	0.132	(0.043, 0.095)	-	X	-
Nickel	10	0.010/0.020	100	0.101	0.412	0.238	0.253	0.113	t	0.027	36	-	-	-	-	ANOVA	0.074	↑	54	0.887	-	8	0.253	(0.143, 0.261)	-	-	-
Phosphorus	10	50	100	1890	3950	2900	2945	553	t_{log}	0.634	4	t_{log}	0.160	-	-10	-	-	-	-	-	-	-	2788	(2789, 3637)	-	-	X
Potassium	10	200	100	2470	5410	3370	3517	769	t	0.042	19	-	-	-	-	ANOVA	0.525	-	11	0.576	-	-8	3517	(2916, 4118)	-	-	-
Rhenium	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Rubidium	10	0.01/0.020	100	3.21	8.48	5.32	5.15	1.54	t_{log}	0.058	17	-	-	-	-	$ANOVA_{log}$	0.062	↓	-22	0.722	-	-7	4.62	(4.99, 6.57)	-	-	X
Selenium[(d)]	10	0.020/0.040	100	0.905	2.150	1.565	1.571	0.448	t	0.085	38	-	-	-	-	ANOVA	0.282	-	36	0.882	-	-7	1.571	(0.871, 2.005)	-	-	-
Silver	10	0.0010/0.0020	30	<0.0010	0.0018	<0.0010	nd	nd	MW	0.028	nc	-	-	-	-	KW	0.012	↓	nc	0.887	-	nc	-	-	53	-	-
Sodium	10	200	100	960	1980	1375	1417	272	t	0.773	4	t	0.251	-	11	-	-	-	-	-	-	-	1417	(1002, 1545)	-	-	-
Strontium	10	0.010/0.020	100	0.391	7.160	0.745	1.736	2.094	t_{log}	0.048	106	-	-	-	-	$ANOVA_{log}$	0.194	-	-57	0.919	-	20	1.021	(0.563, 3.565)	-	-	-
Tellurium	10	0.0040/0.0080	90	<0.0080	0.0101	0.0074	0.0074	0.0023	t	0.056	nc	-	-	-	-	ANOVA	0.208	-	48	0.851	-	-9	-	-	74	-	-
Thallium	10	0.0004/0.0008	100	0.0103	0.0336	0.0158	0.0182	0.0074	t	0.236	23	t_{log}	<0.001	↑	107	-	-	-	-	-	-	-	0.0182	(0.0060, 0.0122)	-	X	-
Thorium	10	0.0020/0.0040	30	<0.0020	0.0035	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Tin	10	0.020/0.040	10	<0.020	0.023	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	11	-	-
Uranium	10	0.00040/0.00080	100	0.00234	0.02010	0.00717	0.00995	0.00686	t_{log}	0.028	120	-	-	-	-	$ANOVA_{log}$	0.321	-	90	0.485	-	-39	0.00837	(0.00336, 0.01852)	-	-	-
Vanadium	10	0.020/0.040	60	<0.020	0.080	0.029	0.032	0.022	MW	0.930	nc	MW	0.149	-	nc	-	-	-	-	-	-	-	-	-	47	-	-
Yttrium	10	0.0020/0.0040	60	<0.0020	0.0075	0.0024	0.0029	0.0020	MW	0.863	nc	MW	0.617	-	nc	-	-	-	-	-	-	-	-	-	53	-	-
Zinc	10	0.1/0.2	100	9.2	58.8	20.9	22.0	13.8	MW	0.014	88	-	-	-	-	KW	0.011	↓	-58	0.793	-	7	17.2	(17.6, 35.7)	-	-	X
Zirconium	10	0.040/0.080	0	<0.040	<0.040	<0.040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 240 mm.

d) Predicted selenium concentration was not calculated due to a non-significant regression relationship between selenium concentration and fish length.

n = sample count; DL = detection limit;% = percent; “<” = less than; “>” = greater than; “-“ = not tested; nd = not determined (at least 50% of data values were <DL); nc = not calculated (more than 50% of values were <DL for at least one group); nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; $ANOVA_{log}$ = analysis of variance on log_{10} transformed data; $ANCOVA_{log}$ = analysis of covariance on log_{10} transformed data; K-W = Kruskal-Wallis test; M-W = Mann Whitney test; NEL = Northeast Lake; SD = standard deviation; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data; ↑/↓= statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline; X = present.

Table 9-5 Summary Statistics and Statistical Comparisons to Reference Sites and Normal Range for Lake Trout Liver Collected from Snap Lake in 2013

	2013 Snap Lake Summary Statistics								Comparisons to Reference														Normal Range Comparison				
									2013 NEL vs 2013 Lake 13			2013 Snap Lake vs 2013 Pooled Reference				2013 Snap Lake vs 2013 Lake 13				2013 Snap Lake vs 2013 NEL			2013 Snap Lake	Normal Range	% of 2013 Reference	Normal Range[b]	
Parameter	n	DL[a]	%>DL	Minimum[a]	Maximum[a]	Median[a]	Mean[a]	SD[a]	Test	p	%	Test	p	↑/↓	%	Test	p	↑/↓	%	p	↑/↓	%	Back Transformed Mean[a]	for Mean of sample of size n[a]	Data >DL	Exceedance	Below
Aluminum	10	0.40/0.08	70	0.20	3.75	0.44	0.77	1.10	t_{log}	0.917	4	t_{log}	0.037	↓	-50	-	-	-	-	-	-	-	0.52	(0.52, 1.84)	-	-	X
Antimony	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Arsenic	10	0.0040/0.0080	80	<0.0040	0.0755	0.0067	0.0134	0.0222	t_{log}	<0.001	99	-	-	-	-	$ANOVA_{log}$	<0.001	↓	-79	0.164	-	-46	0.0067	(0.0070, 0.0242)	-	-	X
Barium	10	0.010/0.020	30	<0.010	0.446	<0.010	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	15	-	-
Beryllium	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Bismuth	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	65	-	-
Boron	10	0.20/.040	0	<0.20	<0.20	<0.20	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Cadmium	10	0.0020/0.040	100	0.0324	0.1650	0.0573	0.0655	0.0378	t	0.827	5	t_{log}	0.020	↓	-40	-	-	-	-	-	-	-	0.0588	(0.0675, 0.151)	-	-	X
Calcium	10	0.5/5	100	40.7	104.0	67.5	68.2	20.9	t_{log}	0.814	5	t_{log}	0.021	↓	-33	-	-	-	-	-	-	-	63.9	(67.0, 136.8)	-	-	X
Cesium	10	0.0010/0.0020	100	0.0141	0.1390	0.0701	0.0679	0.0351	MW	0.049	35	-	-	-	-	ANOVA	0.891	-	9	0.114	-	55	0.0679	(0.0384, 0.0682)	-	-	-
Chromium	10	0.010/0.020	40	<0.010	0.125	<0.010	nd	nd	t_{log}	0.002	156	-	-	-	-	KW	<0.001	↓	-89	0.073	↓	-60	nd(<0.0261)	(0.0192, 0.1028)	-	-	-
Cobalt	10	0.0040/0.0080	100	0.062	0.121	0.085	0.090	0.018	t	0.259	25	t	0.064	↓	-39	-	-	-	-	-	-	-	0.0896	(0.0849, 0.1751)	-	-	-
Copper	10	0.010/0.020	100	1.69	12.10	4.03	5.04	3.43	t_{log}	0.097	35	-	-	-	-	$ANOVA_{log}$	<0.001	↓	-68	0.007	↓	-54	4.13	(6.98, 14.31)	-	-	X
Gallium	10	0.0040/0.0080	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Iron	10	0.2/0.4	100	79	311	196	191	66	t_{log}	0.796	10	t_{log}	0.103	-	-38	-	-	-	-	-	-	-	178	(146, 555)	-	-	-
Lead	10	0.0040/0.0080	20	<0.0040	0.0077	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	35	-	-
Lithium	10	0.020/0.040	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Magnesium	10	1/10	100	108	187	149	150	28	t	0.398	6	t	0.826	-	-1	-	-	-	-	-	-	-	150	(132, 173)	-	-	-
Manganese	10	0.004/0.0080	100	0.84	1.88	1.32	1.31	0.34	MW	0.038	26	-	-	-	-	ANOVA	0.317	-	-16	0.746	-	10	1.26	(1.00, 1.52)	-	-	-
Mercury	9	0.001 to 0.120	100	0.050	0.773	0.404	0.418	0.251	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mercury[predicted (c)]	9	0.001 to 0.120	100	0.059	0.64	0.461	0.397	0.197	$ANCOVA_{log}$	0.187	43	$ANCOVA_{log}$	0.888	-	4	-	-	-	-	-	-	-	0.318	(0.343, 0.859)	-	-	X
Molybdenum	10	0.0040/0.0080	100	0.093	0.181	0.115	0.128	0.032	t	0.179	17	t	0.169	-	-14	-	-	-	-	-	-	-	0.128	(0.072, 0.154)	-	-	-
Nickel	10	0.010/0.020	100	0.013	0.182	0.042	0.069	0.059	t_{log}	0.265	31	t_{log}	0.679	-	-11	-	-	-	-	-	-	-	0.050	(0.035, 0.073)	-	-	-
Phosphorus	9	5/50	100	2320	4270	2875	2984	577	MW	0.245	5	t	0.632	-	3	-	-	-	-	-	-	-	3102	(2625, 3223)	-	-	-
Potassium	10	20/200	100	1920	3640	2580	2649	543	t	0.043	17	-	-	-	-	ANOVA	0.743	-	1	0.416	-	6	2649	(2182, 2983)	-	-	-
Rhenium	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-		-	-	-	-	-	-	-	-	0	-	-
Rubidium	10	0.01/0.020	100	4.97	13.20	7.28	7.89	2.82	t	<0.001	47	`	-	-	-	ANOVA	0.000	↓	-40	0.980	-	-3	7.89	(8.40, 12.28)	-	-	X
Selenium	10	0.020/0.040	100	1.34	4.11	2.04	2.27	1.02	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-
Selenium[predicted(c)]	10	0.020/0.040	100	1.22	3.63	1.81	2.32	0.93	$ANCOVA_{log}$	0.207	21	$ANCOVA_{log}$	0.259	-	-16		-	-	-	-	-	-	2.16	(1.75, 3.02)	-	-	-
Silver	10	0.0010/0.0020	100	0.0017	0.0613	0.0153	0.0255	0.0235	t_{log}	<0.001	126	-	-	-	-	$ANOVA_{log}$	0.001	↓	-82	0.376	-	-41	0.0144	(0.0206, 0.0779)	-	-	X
Sodium	10	20/200	100	1140	1720	1395	1406	196	t	0.608	5	t	0.366	-	7	-	-	-	-	-	-	-	1406	(1083, 1544)	-	-	-
Strontium	10	0.010/0.020	100	0.123	0.403	0.192	0.228	0.100		0.748	6	t_{log}	0.029	↑	43	-	-	-	-	-	-	-	0.228	(0.091, 0.187)	-	X	-
Tellurium	10	0.0040/0.0080	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Thallium	10	0.0004/0.0008	100	0.0265	0.1360	0.0985	0.0882	0.0380	t	0.646	9	t	0.007	↑	60	-	-	-	-	-	-	-	0.0806	(0.0295, 0.0637)	-	X	-
Thorium	10	0.0020/0.0040	10	<0.0020	0.0045	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Tin	10	0.020/0.040	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Uranium	10	0.00040/0.000 80	50	<0.00040	0.00142	<0.00040	nd	nd	t_{log}	0.622	24	t_{log}	0.227	-	-38	-	-	-	-	-	-	-	-	-	60	-	-
Vanadium	10	0.020/0.040	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	15	-	-
Yttrium	10	0.0020/0.0040	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	30	-	-
Zinc	10	0.10/0.20	100	18.4	28.0	22.4	22.9	3.1	t	0.035	19	-	-	-	-	$ANOVA_{log}$	0.0002	↓	-31	0.0656	↓	-16	22.9	(23.3, 32.1)	-	-	X
Zirconium	10	0.040/0.080	0	<0.040	<0.040	<0.040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 600 mm.

n = sample count; DL = detection limit;% = percent; "<" = less than; ">" = greater than; "-" = not tested; nd = not determined (at least 50% of data values were <DL); nc = not calculated (more than 50% of values were <DL for at least one group); nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; $ANOVA_{log}$ = analysis of variance on log_{10} transformed data; $ANCOVA_{log}$ = analysis of covariance on log_{10} transformed data; K-W = Kruskal-Wallis test; M-W = Mann Whitney test; NEL = Northeast Lake; SD = standard deviation; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data; ↑/↓= statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline; X = present.

Table 9-6 Summary Statistics and Statistical Comparisons to Reference Sites and Normal Range for Round Whitefish Liver Collected from Snap Lake in 2013

	2013 Snap Lake Summary Statistics								Comparisons to Reference															Normal Range Comparison				
									2013 NEL vs 2013 Lake 13			2013 Snap Lake vs 2013 Pooled Reference				2013 Snap Lake vs 2013 Lake 13				2013 Snap Lake vs 2013 NEL			2013 Snap Lake	Normal Range	% of 2013 Reference	Normal Range[b]	
Parameter	n	DL[a]	%>DL	Minimum[a]	Maximum[a]	Median[a]	Mean[a]	SD[a]	Test	p	%	Test	p	↑/↓	%	Test	p	↑/↓	%	p	↑/↓	%	Back Transformed Mean[a]	for Mean of sample of size n[a]	Data >DL	Exceedance	Below
Aluminum	10	0.40	70	<0.40	1.29	0.55	0.66	0.45	t_{log}	0.765	9	t_{log}	0.622	-	-12	-	-	-	-	-	-	-	0.52	(0.35, 1.01)	-	-	-
Antimony	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Arsenic	10	0.0040	90	<0.0040	0.0483	0.0290	0.0288	0.0164	t_{log}	0.168	48	t_{log}	0.044	↓	-49	-	-	-	-	-	-	-	0.0178	(0.0238, 0.0733)	-	-	X
Barium	10	0.010	10	<0.010	0.066	<0.010	nd	nd	MW	0.744	nc	MW	<0.001	↓	nc	-	-	-	-	-	-	-	nd(<0.016)	(0.013, 0.074)	-	-	-
Beryllium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Bismuth	10	0.0020	20	<0.0020	0.0027	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	65	-	-
Boron	10	0.20	0	<0.20	<0.20	<0.20	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Cadmium	10	0.0020	100	0.0424	0.1330	0.0673	0.0770	0.0263	t_{log}	0.258	17	t_{log}	0.703	-	5	-	-	-	-	-	-	-	0.0722	(0.0534, 0.0903)	-	-	-
Calcium	10	0.5/5	100	37.8	203.0	96.7	98.1	50.1	t_{log}	0.141	36	t_{log}	0.183	-	-37	-	-	-	-	-	-	-	74.7	(78.6, 151.0)	-	-	X
Cesium	10	0.0010	100	0.0124	0.0844	0.0249	0.0321	0.0231	t	0.031	39	-	-	-	-	ANOVA$_{log}$	0.575	-	37	0.020	↑	103	0.024	(0.0135, 0.0235)	-	X	-
Chromium	10	0.010	70	<0.010	0.166	0.016	0.038	0.055	t_{log}	<0.001	416	-	-	-	-	ANOVA$_{log}$	<0.001	↓	-90	0.394	-	85	0.015	(0.008, 0.125)	-	-	-
Cobalt	10	0.0040	100	0.0411	0.2990	0.0870	0.1085	0.0731	t	0.037	59	-	-	-	-	ANOVA$_{log}$	0.790	-	-18	0.450	-	50	0.0852	(0.0557, 0.1254)	-	-	-
Copper	8	0.010	100	1.97	2.68	2.23	2.28	0.28	t	0.044	26	-	-	-	-	ANOVA$_{log}$	0.009	↓	-30	0.534	-	-11	2.28	(2.25, 3.63)	-	-	-
Gallium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-			-	-	-	-	0	-	-
Iron	10	0.2	100	94	349	145	174	84	t	0.032	35	-	-	-	-	ANOVA$_{log}$	0.855		9	0.032	↑	54	174	(91, 168)	-	X	-
Lead	10	0.0040	20	<0.0040	0.0083	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	35	-	-
Lithium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Magnesium	10	1/10	100	110	221	185	180	32	t	0.483	5	t	0.002	↓	-18	-	-	-	-	-	-	-	180	(197, 243)	-	-	X
Manganese	10	0.004	100	0.67	2.48	1.61	1.59	0.47	t	0.016	31	-	-	-	-	ANOVA$_{log}$	0.001	↓	-41	0.283	-	-19	1.59	(1.75, 2.85)	-	-	X
Mercury	9	0.001 to 0.010	100	0.087	0.174	0.126	0.127	0.029	t[e]	0.501	10	nt	-	-	-	ANOVA$_{log}$	-	-	-	0.062	↑	42	-	-	-	-	-
Mercury[predicted (c)]	9	0.001 to 0.010	100	0.091	0.162	0.127	0.125	0.025	-	-	-	ANCOVA$_{log}$	0.018	↑	36	ANCOVA$_{log}$	0.006	↑	30	-	-	-	0.123	(0.069, 0.119)	-	X	-
Molybdenum	10	0.0040	100	0.106	0.200	0.170	0.168	0.026	t	0.697	4	t	0.121	-	-11	-	-	-	-	-	-	-	0.168	(0.158, 0.221)	-	-	-
Nickel	10	0.010	100	0.018	0.083	0.045	0.049	0.021	t	0.054	23	-	-	-	-	ANOVA	0.077	↓	-26	0.871	-	-7	0.049	(0.047, 0.073)	-	-	-
Phosphorus	10	5/50	100	2280	4550	3820	3615	668	t	0.139	8	t	0.016	↓	-13	-	-	-	-	-	-	-	3615	(3779, 4528)	-	-	X
Potassium	10	20/200	100	2040	4050	3650	3422	669	t	0.888	1	t	0.143	-	-9	-	-	-	-	-	-	-	3422	(3321, 4220)	-	-	-
Rhenium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Rubidium	10	0.01	100	4.64	9.53	6.65	6.72	1.32	t	0.005	46	-	-	-	-	ANOVA$_{log}$	<0.001	↓	-36	0.643	-	-10	6.54	(6.76, 11.79)	-	-	X
Selenium[d]	10	0.020	100	1.00	2.86	1.37	1.49	0.54	t	0.843	2	ANOVA$_{log}$	0.527	-	-6	-	-	-	-	-	-	-	1.49	(1.3, 1.81)	-	-	-
Silver	10	0.0010	90	<0.0010	0.0871	0.0030	0.0149	0.0274	MW	0.005	157	-	-	-	-	ANOVA$_{log}$	0.292	-	-86	0.270	-	166	0.0043	(0.0012, 0.0156)	-	-	-
Sodium	10	20/200	100	1020	2390	1345	1403	385	t	0.047	-	-	-	-	-	ANOVA$_{log}$	0.963	-	2	0.115	-	19	1403	(1085, 1413)	-	-	-
Strontium	10	0.010	100	0.117	1.130	0.269	0.334	0.303	MW	0.102	53	MW	0.646	-	-44	-	-	-	-	-	-	-	0.217	(0.179, 0.432)	-	-	-
Tellurium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Thallium	10	0.0004	100	0.0231	0.1080	0.0699	0.0604	0.0248	t	0.053	68	-	-	-	-	ANOVA	0.786	-	-15	0.226	-	73	0.0538	(0.0208, 0.0703)	-	-	-
Thorium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Tin	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Uranium	10	0.00040	90	<0.00040	0.00763	0.00213	0.00254	0.00224	t	0.013	60	-	-	-	-	ANOVA	0.929	-	13	0.130	-	-39	0.00254	(0.00181, 0.00469)	-	-	-
Vanadium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	15	-	-
Yttrium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	30	-	-
Zinc	10	0.10	100	18.9	36.4	22.9	23.9	5.2	MW	0.014	174	-	-	-	-	KW	0.008	↓	-93	0.618	-	-1	23.1	(23.7, 45.4)	-	-	X
Zirconium	10	0.040	0	<0.040	<0.040	<0.040	nd	nd	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 240 mm.

d) Predicted selenium concentration was not calculated due to a non-significant regression relationship between selenium concentration and fish length.

e) ANCOVA not conducted for Snap Lake vs Lake 13 as regression slopes are significantly different (p = 0.040) and Lake 13 slope is negative.

n = sample count; DL = detection limit;% = percent; "<" = less than; ">" = greater than; "-" = not tested; nd = not determined (at least 50% of data values were <DL); nc = not calculated (more than 50% of values were <DL for at least one group); nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; ANOVA$_{log}$ = analysis of variance on $\log_{10}$ transformed data; ANCOVA$_{log}$ = analysis of covariance on $\log_{10}$ transformed data; K-W = Kruskal-Wallis test; M-W = Mann Whitney test; NEL = Northeast Lake; SD = standard deviation; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on $\log_{10}$ transformed data; ↑/↓ = statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline; X = present.

Table 9-7 Summary Statistics and Statistical Comparisons to Reference Sites (2013), Baseline (1999 and 2004), and Normal Range for Lake Trout Muscle Tissue Collected from Snap Lake in 2013

	2013 Snap Lake Summary Statistics								Comparisons to Baseline							
									2013 Snap Lake vs 1999 Snap Lake				2013 Snap Lake vs 2004 Snap Lake			
Parameter	N	DL[a]	%>DL	Minimum[a]	Maximum[a]	Median[a]	Mean[a]	SD[a]	Test	p	↑/↓	%	Test	p	↑/↓	%
Aluminum	10	0.40	10	<0.40	0.69	<0.40	nd	nd	nt	-	-	-	nt	-	-	-
Antimony	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Arsenic	10	0.0040	90	<0.0040	0.0146	0.0070	0.0083	0.0043	nt	-	-	-	nt	-	-	-
Barium	10	0.010	40	<0.010	0.063	<0.010	nd	nd	nt	-	-	-	nt	-	-	-
Beryllium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Bismuth	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Boron	10	0.20	0	<0.20	<0.20	<0.20	nd	nd	nt	-	-	-	nt	-	-	-
Cadmium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Calcium	10	0.5/5	100	48	317	65	98	82	nt	-	-	-	t	0.316	-	13
Cesium	10	0.0010	100	0.0261	0.1440	0.1200	0.1055	0.0436	t	0.953	-	-1	MW	0.012	↓	-30
Chromium	10	0.010	70	<0.010	0.175	0.043	0.057	0.056	nt	-	-	-	nt	-	-	
Cobalt	10	0.0040	20	<0.0040	0.0143	<0.0040	nd	nd	nt	-	-	-	nt	-	-	
Copper	10	0.010	100	0.200	0.443	0.235	0.264	0.076	MW	0.004	↓	-54	t_{log}	0.009	↓	-40
Gallium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	-	nt	-	-	
Iron	10	0.20	100	1.61	4.47	3.13	3.13	0.76	nt	-	-	-	t_{log}	0.041	↓	-30
Lead	10	0.0040	20	<0.0040	0.0102	<0.0040	nd	nd	nt	-	-	-	nt	-	-	
Lithium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	-	nt	-	-	
Magnesium	10	1/10	100	194	291	255	252	27	nt	-	-	-	t	0.074	-	9
Manganese	10	0.004	100	0.050	0.139	0.084	0.093	0.032	t	0.891	-	2	t	0.557	-	9
Mercury	9	0.001 to 0.010	100	0.046	0.767	0.363	0.372	0.217	nt	-	-	-	nt	-	-	
Mercury[predicted (c)]	9	0.001 to 0.010	100	0.280	0.573	0.315	0.366	0.103	$ANCOVA_{log}$	0.011	↓	-37	$ANCOVA_{log}$	0.001	↓	-38
Molybdenum	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	-	nt	-	-	
Nickel	10	0.010	90	<0.010	0.022	0.013	0.013	0.004	t	0.059	-	-26	nt	-	-	
Phosphorus	10	5/50	100	1980	2540	2270	2278	179	nt	-	-	-	t	0.001	↑	17
Potassium	10	20/200	100	3280	4470	3945	3932	420	nt	-	-	-	t	0.011	↑	14
Rhenium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	
Rubidium	10	0.01	100	5.26	10.60	7.55	7.65	1.48	t_{log}	0.061	↓	-22	t	0.001	↓	-28
Selenium[d]	10	0.020	100	0.213	0.437	0.332	0.320	0.069	MW	0.140	-	10	$ANCOVA_{log}$	<0.001	↓	-27
Silver	10	0.0010	0	<0.0010	<0.0010	<0.0010	nd	nd	nt	-	-	-	nt	-	-	
Sodium	10	20/200	100	154	342	267	272	51	nt	-	-	-	MW	0.405	-	-5
Strontium	10	0.010	100	0.060	0.632	0.093	0.175	0.181	MW	0.006	↑	335	MW	0.033	↑	146
Tellurium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	-	nt	-	-	
Thallium	10	0.0004	100	0.00355	0.01410	0.00879	0.00936	0.00348	nt	-	-	-	nt	-	-	
Thorium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	
Tin	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	-	nt	-	-	
Uranium	10	0.00040	0	<0.00040	<0.00040	<0.00040	nd	nd	nt	-	-	-	nt	-	-	
Vanadium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	-	nt	-	-	
Yttrium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	
Zinc	10	0.10	100	2.38	4.26	3.06	3.10	0.60	t_{log}	0.184	-	-12	t_{log}	0.136	-	-13
Zirconium	10	0.040	0	<0.040	<0.040	<0.040	nd	nd	nt	-	-	-	nt	-	-	

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 600 mm.

d) Predicted selenium concentration was not calculated due to a non-significant regression relationship between selenium concentration and fish length.

n = sample count; DL = detection limit;% = percent; "<" = less than; ">" = greater than; "-" = not tested; nd = not determined (at least 50% of data values were <DL); nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; $ANOVA_{log}$ = analysis of variance on log_{10} transformed data; $ANCOVA_{log}$ = analysis of covariance on log_{10} transformed data; MW = Mann Whitney test; NEL = Northeast Lake; SD = standard deviation; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data; ↑/↓ = statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline.

Table 9-7 Summary Statistics and Statistical Comparisons to Reference Sites (2013), Baseline (1999 and 2004), and Normal Range for Lake Trout Muscle Tissue Collected from Snap Lake in 2013

	Comparisons to Reference															Normal Range Comparison				
	2013 NEL vs 2013 Lake 13			2013 Snap Lake vs 2013 Pooled Reference				2013 Snap Lake vs 2013 Lake 13				2013 Snap Lake vs 2013 NEL			2013 Snap	Normal Range	% of 2013 Reference	Normal Range[b]	
Parameter	Test	p	%	Test	p	↑/↓	%	Test	p	↑/↓	%	p	↑/↓	%	Back Transformed Mean[a]	for Mean of sample of size n[a]	Data >DL	Exceedance	Below
Aluminum	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Antimony	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Arsenic	t_{log}	0.060	63	-	-	-	-	$ANOVA_{log}$	<0.001	↓	-75	0.036	↓	-54	0.007	(0.008, 0.029)	-	-	X
Barium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	nd(<0.0130)	(0.009, 0.04)	-	-	-
Beryllium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Bismuth	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Boron	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Cadmium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Calcium	t_{log}	0.145	34	t	0.035	↓	-33	-	-	-	-	-	-	-	76	(70, 166)	-	-	-
Cesium	t_{log}	0.189	29	MW	0.342	-	1	-	-	-	-	-	-	-	0.099	(0.080, 0.134)	-	-	-
Chromium	t_{log}	0.002	153	-	-	-	-	$ANOVA_{log}$	0.434	-	-47	0.240	-	117	0.031	(0.028, 0.125)	-	-	-
Cobalt	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	33	-	-
Copper	nt	0.770	3	t_{log}	0.383	-	8	-	-	-	-	-	-	-	0.250	(0.242, 0.444)	-	-	-
Gallium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Iron	t	0.884	2	t	0.228	-	-14	-	-	-	-	-	-	-	3.01	(3.09, 5.11)	-	-	X
Lead	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	24	-	-
Lithium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Magnesium	t	0.178	5	t	0.189	-	5	-	-	-	-	-	-	-	249	(224, 264)	-	-	-
Manganese	t_{log}	0.109	23	t_{log}	0.700	-	5	-	-	-	-	-	-	-	0.087	(0.074, 0.11)	-	-	-
Mercury	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mercury[predicted (c)]	$ANCOVA_{log}$	0.037	49	-	-	-	-	$ANCOVA_{log}$	0.562	-	37	0.273	-	-16	0.355	(0.338, 0.647)	-	-	-
Molybdenum	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Nickel	MW	0.693	1	MW	0.506	-	14	-	-	-	-	-	-	-	0.012	(0.016, 0.060)	-	-	X
Phosphorus	t	0.486	2	t	0.758	-	1	-	-	-	-	-	-	-	2278	(2028, 2389)	-	-	-
Potassium	t	0.469	3	t	0.427	-	3	-	-	-	-	-	-	-	3932	(3464, 4042)	-	-	-
Rhenium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Rubidium	t	0.001	33	-	-	-	-	ANOVA	<0.001	↓	-40	0.223	-	-16	7.65	(9.38, 12.96)	-	-	X
Selenium[d]	$ANCOVA_{log}$	0.001	33	-	-	-	-	ANOVA	0.004	↑	38	0.961	-	-2	0.315	(0.251, 0.398)	-	-	-
Silver	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	24	-	-
Sodium	t	0.527	6	MW	0.123	-	11	-	-	-	-	-	-	-	272	(231, 311)	-	-	-
Strontium	t_{log}	0.218	36	t_{log}	0.921	-	-3	-	-	-	-	-	-	-	0.121	(0.050, 0.198)	-	-	-
Tellurium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Thallium	t_{log}	0.534	10	t_{log}	0.013	↑	47	-	-	-	-	-	-	-	0.0094	(0.0051, 0.0084)	-	**X**	-
Thorium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Tin	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Uranium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Vanadium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Yttrium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Zinc	MW	0.205	2	t_{log}	0.399	-	-6	-	-	-	-	-	-	-	3.02	(3.14, 4.36)	-	-	X
Zirconium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 600 mm.

d) Predicted selenium concentration was not calculated due to a non-significant regression relationship between selenium concentration and fish length.

DL = detection limit;% = percent; "<" = less than; ">" = greater than; "-" = not tested; nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; $ANOVA_{log}$ = analysis of variance on log_{10} transformed data; $ANCOVA_{log}$ = analysis of covariance on log_{10} transformed data;MW = Mann Whitney test; NEL = Northeast Lake; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data; ↑/↓ = statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline; X = present.

Table 9-8 Summary Statistics and Statistical Comparisons to Reference Sites (2013), Baseline (1999 and 2004), and Normal Range for Round Whitefish Muscle Tissue Collected from Snap Lake in 2013

Parameter	2013 Snap Lake Summary Statistics								Comparisons to Baseline							
									2013 Snap Lake vs 1999 Snap Lake				2013 Snap Lake vs 2004 Snap Lake			
	n	DL[(a)]	%>DL	Minimum[(a)]	Maximum[(a)]	Median[(a)]	Mean[(a)]	SD[(a)]	Test	p	↑/↓	%	Test	p	↑/↓	%
Aluminum	10	0.40	30	<0.40	0.63	<0.40	nd	nd	nt	-	-	-	nt	-	-	-
Antimony	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Arsenic	10	0.0040	100	0.007	0.016	0.011	0.011	0.003	nt	-	-	-	nt	-	-	-
Barium	10	0.010	90	<0.010	0.036	0.016	0.016	0.008	nt	-	-	-	nt	-	-	-
Beryllium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Bismuth	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Boron	10	0.20	0	<0.20	<0.20	<0.20	nd	nd	nt	-	-	-	nt	-	-	-
Cadmium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Calcium	10	0.5/5	100	115	323	200	215	74	nt	-	-	-	t	0.361	-	-17
Cesium	10	0.0010	100	0.0282	0.0884	0.0630	0.0609	0.0190	t	0.044	↑	38	nt	-	-	-
Chromium	10	0.010	60	<0.010	0.061	0.020	0.024	0.022	nt	-	-	-	nt	-	-	-
Cobalt	10	0.0040	90	<0.0040	0.0136	0.0079	0.0081	0.0033	nt	-	-	-	nt	-	-	-
Copper	10	0.010	100	0.243	0.465	0.288	0.307	0.068	t_{log}	0.122	-	6	t	0.016	-	-28
Gallium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	-	nt	-	-	-
Iron	10	0.20	100	2.76	4.88	3.79	3.83	0.72	nt	-	-	-	MW	0.587	-	6
Lead	10	0.0040	30	<0.0040	0.0087	<0.0040	nd	nd	nt	-	-	-	nt	-	-	-
Lithium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	-	nt	-	-	-
Magnesium	10	1/10	100	306	335	321	322	10	nt	-	-	-	t	0.861	-	0
Manganese	10	0.004	100	0.095	0.259	0.143	0.152	0.046	MW	0.770	-	-13	t	0.204	-	19
Mercury	9	0.001 to 0.010	100	0.043	0.116	0.075	0.078	0.023	nt	-	-	-	nt	-	-	-
Mercury[predicted (c)]	9	0.001 to 0.010	100	0.049	0.093	0.066	0.073	0.016	ANCOVA$_{log}$	0.505	-	6	ANCOVA$_{rank}$	<0.001	↓	-39
Molybdenum	10	0.0040	10	<0.0040	0.0386	<0.0040	nd	nd	nt	-	-	-	nt	-	-	-
Nickel	10	0.010	70	<0.010	0.025	0.014	0.014	0.007	t	0.007	↓	-48	nt	-	-	-
Phosphorus	10	5/50	100	2500	2960	2710	2736	142	nt	-	-	-	t	0.880	-	0
Potassium	10	20/200	100	4230	5030	4785	4732	265	nt	-	-	-	t	0.113	-	4
Rhenium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Rubidium	10	0.01	100	4.89	8.33	6.59	6.58	1.14	t	0.148	-	-10	t	<0.001	↓	-32
Selenium[(d)]	10	0.020	100	0.247	0.371	0.297	0.297	0.037	MW	0.241	-	-18	MW	0.022	↓	-20
Silver	10	0.0010	0	<0.0010	<0.0010	<0.0010	nd	nd	nt	-	-	-	nt	-	-	-
Sodium	10	20/200	100	219	440	308	296	72	nt	-	-	-	MW	0.028	↑	32
Strontium	10	0.010	100	0.283	0.837	0.565	0.562	0.187	MW	0.014	↑	79	t	0.755	-	-6
Tellurium	10	0.0040	0	<0.0040	<0.0040	<0.0040	nd	nd	nt	-	-	-	nt	-	-	-
Thallium	10	0.0004	100	0.0069	0.0257	0.0109	0.0123	0.0055	nt	-	-	-	nt	-	-	-
Thorium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Tin	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	-	nt	-	-	-
Uranium	10	0.00040	0	<0.00040	<0.00040	<0.00040	nd	nd	nt	-	-	-	nt	-	-	-
Vanadium	10	0.020	0	<0.020	<0.020	<0.020	nd	nd	nt	-	-	-	nt	-	-	-
Yttrium	10	0.0020	0	<0.0020	<0.0020	<0.0020	nd	nd	nt	-	-	-	nt	-	-	-
Zinc	10	0.10	100	3.44	5.21	4.25	4.22	0.58	t_{log}	0.027	↓	-16	t	0.470	-	5
Zirconium	10	0.040	0	<0.040	<0.040	<0.040	nd	nd	nt	-	-	-	nt	-	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 240 mm.

d) Predicted selenium concentration was not calculated due to a non-significant regression relationship between selenium concentration and fish length.

n = sample count; DL = detection limit;% = percent; "<" = less than; "-" = not tested; nd = not determined (at least 50% of data values were <DL); nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; ANOVA$_{log}$ = analysis of variance on log_{10} transformed data; ANCOVA$_{log}$ = analysis of covariance on log_{10} transformed data; MW = Mann Whitney test; NEL = Northeast Lake; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data; ↑/↓= statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline.

Table 9-8 Summary Statistics and Statistical Comparisons to Reference Sites (2013), Baseline (1999 and 2004), and Normal Range for Round Whitefish Muscle Tissue Collected from Snap Lake in 2013

Parameter	Comparisons to Reference														Normal Range Comparison				
	2013 NEL vs 2013 Lake 13			2013 Snap Lake vs 2013 Pooled Reference				2013 Snap Lake vs 2013 Lake 13				2013 Snap Lake vs 2013 NEL			2013 Snap	Normal Range	% of 2013 Reference	Normal Range[b]	
	Test	P	%	Test	p	↑/↓	%	Test	p	↑/↓	%	p	↑/↓	%	Back Transformed Mean[a]	for Mean of sample of size n[a]	Data >DL	Exceedance	Below
Aluminum	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Antimony	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	16	-	-
Arsenic	t	0.052	43	-	-	-	-	ANOVA	0.452	-	-18	0.476	-	27	0.011	(0.007, 0.014)	-	-	-
Barium	t	0.886	3	t	<0.001	↓	-61	-	-	-	-	-	-	-	0.015	(0.032, 0.089)	-	-	X
Beryllium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Bismuth	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Boron	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Cadmium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-
Calcium	t	0.157	26	t	0.173	-	-20	-	-	-	-	-	-	-	204	(156, 365)	-	-	-
Cesium	MW	0.072	13	-	-	-	-	logANOVA	0.088	↑	40	<0.001	↑	77	0.0554	(0.0271, 0.0449)	-	X	-
Chromium	t_{log}	0.558	36	t_{log}	0.071	-	-59	-	-	-	-	-	-	-	0.016	(0.019, 0.082)	-	-	X
Cobalt	t_{log}	0.044	60	-	-	-	-	$ANOVA_{log}$	0.801	-	-15	0.237	-	53	0.0075	(0.00440, 0.0095)	-	-	-
Copper	t	0.006	36	-	-	-	-	$ANOVA_{log}$	0.213	-	-17	0.233	-	19	0.295	(0.253, 0.443)	-	-	-
Gallium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Iron	t	0.062	23	-	-	-	-	ANOVA	0.489	-	-11	0.552	-	12	3.73	(3.21, 4.79)	-	-	-
Lead	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	21	-	-
Lithium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Magnesium	MW	0.102	2	MW	0.024	↑	6	-	-	-	-	-	-	-	322	(290, 321)	-	X	-
Manganese	t	<0.001	46	-	-	-	-	ANOVA	0.003	↓	-32	0.798	-	8	0.144	(0.121, 0.210)	-	-	-
Mercury	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mercury[predicted (c)]	$ANCOVA_{log}$	0.381	9	ANCOVA	0.006	↑	30	-	-	-	-	-	-	-	0.072	(0.051, 0.108)	-	-	-
Molybdenum	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	23	-	-
Nickel	t	0.183	24	t	0.798	-	5	-	-	-	-	-	-	-	0.011	(0.016, 0.067)	-	-	X
Phosphorus	t	0.773	1	t	0.034	↑	6	-	-	-	-	-	-	-	2736	(2451, 2753)	-	-	-
Potassium	MW	0.595	1	MW	0.066	↑	4	-	-	-	-	-	-	-	4732	(4166, 4647)	-	X	-
Rhenium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Rubidium	t_{log}	<0.001	32	-	-	-	-	$ANOVA_{log}$	0.001	↓	-28	0.964	-	-2	6.47	(6.91, 9.10)	-	-	X
Selenium[d]	t	0.006	41	-	-	-	-	ANOVA	0.310	-	21	0.077	↓	-20	0.298	(0.278, 0.414)	-	-	-
Silver	-	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	11	-	-
Sodium	t	0.518	8	t	0.864	-	2	-	-	-	-	-	-	-	296	(212, 300)	-	-	-
Strontium	t	0.212	25	t	0.901	-	2	-	-	-	-	-	-	-	0.540	(0.252, 0.808)	-	-	-
Tellurium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Thallium	t_{log}	0.430	16	t_{log}	<0.001	↑	128	-	-	-	-	-	-	-	0.0123	(0.0035, 0.0074)	-	X	-
Thorium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Tin	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Uranium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Vanadium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Yttrium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-
Zinc	t_{log}	0.013	38	-	-	-	-	$ANOVA_{log}$	0.040	↓	-17	0.763	-	-6	4.15	(3.91, 5.46)	-	-	-
Zirconium	nt	-	-	nt	-	-	-	-	-	-	-	-	-	-	-	-	0	-	-

a) Units = milligram per kilogram wet weight (mg/kg ww).

b) Normal range exceedance or below = 2013 Snap Lake back transformed mean value above the upper bound of the normal, or below the lower bound of the normal.

c) Predicted concentration for a fish length of 240 mm.

d) Predicted selenium concentration was not calculated due to a non-significant regression relationship between selenium concentration and fish length.

n = sample count; DL = detection limit;% = percent; "<" = less than; ">" = greater than; "-" = not tested; nt = not tested (at least 50% of data values were <DL for each group); ANOVA = analysis of variance; $ANOVA_{log}$ = analysis of variance on log_{10} transformed data; $ANCOVA_{log}$ = analysis of covariance on log_{10} transformed data; KW = Kruskal-Wallis test; MW = Mann Whitney test; NEL = Northeast Lake; p = p-value; t = two-sample t-test; t_{log} = two-sample t-test on log_{10} transformed data; ↑/↓= statistically significant increase/decrease in parameter in Snap Lake relative to reference or baseline.

9.4.2 Reference Lake Comparisons

Northeast Lake Compared to Lake 13

Statistical analyses were conducted to test for differences in parameter concentrations in Lake Trout and Round Whitefish kidney, liver, and muscle tissues between the reference lakes in 2013. Visual comparisons of the reference lake data are presented in Appendix 9D (kidney tissue plots are presented in Figures 9D2-1 through 9D2-39, liver tissue plots are presented in Figures 9D3-1 through 9D3-39, and muscle tissue plots are presented in Figures 9D4-1 through 9D4-39). Summary statistics for the reference lakes are presented in Appendix 9C (kidney summary statistics are presented in Tables 9C-1 and 9C-2, liver summary statistics are presented in Tables 9C-3 and 9C-4, and muscle summary statistics are presented in Tables 9C-5 and 9C-6).

Some parameters were significantly different between Northeast Lake and Lake 13.

Lake Trout

Parameters that were present in greater concentrations in Northeast Lake relative to Lake 13 in Lake Trout tissue were:

- Kidney: cadmium, mercury$^{\text{predicted}}$, nickel, tellurium, and thallium (Table 9-3 and Table 9C-1); and
- Muscle: mercury$^{\text{predicted}}$ and selenium (Table 9-7 and Table 9C-5).

Parameters that were present in lower concentrations in Northeast Lake relative to Lake 13 in Lake Trout tissue were:

- Kidney: aluminum, antimony, arsenic, rubidium, and silver (Table 9-3 and Table 9C-1);
- Liver: cesium, chromium, copper, manganese, potassium, and zinc (Table 9-5 and Table 9C-3); and,
- Muscle: arsenic, chromium, and rubidium (Table 9-7 and Table 9C-5).

Round Whitefish

Parameters that were present in greater concentrations in Northeast Lake relative to Lake 13 in Round Whitefish tissue were:

- Kidney: cadmium, copper, nickel, potassium, selenium, tellurium, and uranium (Table 9-4 and Table 9C-2); and,
- Muscle: copper, iron, selenium, and zinc (Table 9-8 and Table 9C-6).

Parameters that were present in lower concentrations in Northeast Lake relative to Lake 13 in Round Whitefish tissue were:

- Kidney: antimony, arsenic, calcium, manganese, rubidium, silver, strontium, and zinc (Table 9-4 and Table 9C-2);
- Liver: arsenic, magnesium, mercury^{predicted}, and phosphorus (Table 9-6 and Table 9C-4); and,
- Muscle: arsenic, cesium, and cobalt (Table 9-8 and Table 9C-6).

Snap Lake Compared to Reference Lakes

Statistical analyses were conducted to test for differences in parameter concentrations in Lake Trout and Round Whitefish kidney, liver, and muscle tissues in Snap Lake and the pooled reference lakes. If reference lakes could not be pooled, differences between Snap Lake and the individual reference lakes were examined. Visual comparisons of Snap Lake 2013 data are presented in Appendix 9D (kidney tissue plots are presented in Figures 9D2-1 through 9D2-39, liver tissue plots are presented in Figures 9D3-1 through 9D3-39, and muscle tissue plots are presented in Figures 9D4-1 through 9D4-39). As in previous years, fish tissue chemistry was variable between lakes and between species. There were statistically significant differences in some parameters between lakes; some parameters decreased in Snap Lake relative to reference lakes, and some increased in Snap Lake relative to reference lakes. Results of statistical comparisons between Snap Lake and the reference lakes are summarized below. The magnitude of differences is presented, along with the results of the statistical tests, in Tables 9-3 through 9-8.

Lake Trout

Parameters with significantly lower concentrations in Lake Trout tissues from Snap Lake relative to the reference lakes in 2013 were:

- Kidney: aluminum, antimony, arsenic, bismuth, chromium, nickel, and rubidium (Table 9-3);
- Liver: aluminum, arsenic, cadmium, calcium, chromium, cobalt, copper, rubidium, silver, and zinc (Table 9-5); and,
- Muscle: arsenic, calcium, and rubidium (Table 9-7).

Parameters with significantly greater concentrations in Lake Trout tissues from Snap Lake in 2013 were:

- Kidney: mercury^{predicted}, sodium, strontium, and thallium (Table 9-3);
- Liver: strontium and thallium (Table 9-5); and,
- Muscle: selenium and thallium (Table 9-7).

Round Whitefish

Parameters with significantly lower concentrations in Round Whitefish tissues from Snap Lake relative to the reference lakes in 2013 were:

- Kidney: barium, bismuth, calcium, manganese, rubidium, silver, and zinc (Table 9-4);
- Liver: arsenic, barium, chromium, copper, magnesium, manganese, nickel, phosphorus, rubidium, and zinc (Table 9-6); and,
- Muscle: barium, manganese, rubidium, selenium, and zinc (Table 9-8).

Parameters with significantly greater concentrations in Round Whitefish tissues from Snap Lake in 2013 compared to reference were:

- Kidney: cadmium, cesium, cobalt, iron, mercury^{predicted}, molybdenum, nickel, and thallium (Table 9-4);
- Liver: cesium, iron, mercury, and mercury^{predicted} (Table 9-6); and,
- Muscle: cesium, magnesium, mercury^{predicted}, phosphorus, potassium, and thallium (Table 9-8).

9.4.3 Normal Range

Mean parameter concentrations in Lake Trout and Round Whitefish kidney, liver, and muscle tissue from Snap Lake in 2013 were compared to the normal range for the mean of each parameter (Section 9.2.2.3 and Appendix 9A).

In 2013, most parameters in Snap Lake liver, kidney, and muscle tissues of Lake Trout and Round Whitefish were within the normal range. There were 35 occurrences where Snap Lake fish were below the normal range, 17 in Lake Trout and 18 in Round Whitefish. There were 16 occurrences where Snap Lake fish were above the normal range, 5 in Lake Trout and 11 in Round Whitefish.

Lake Trout

Mean parameter concentrations in Lake Trout tissues from Snap Lake in 2013 that were below the normal range were:

- Kidney: arsenic, barium, and rubidium (Table 9-3);
- Liver: aluminum, arsenic, cadmium, calcium, copper, mercury^{predicted}, rubidium, silver, and zinc (Table 9-5); and,
- Muscle: arsenic, iron, nickel, rubidium, and zinc (Table 9-7).

Mean parameter normal range exceedances in Lake Trout from Snap Lake in 2013 were:

- Kidney: strontium and thallium (Table 9-4);
- Liver: strontium and thallium (Table 9-6); and,
- Muscle: thallium (Table 9-8).

Round Whitefish

Mean parameter concentration in Round Whitefish tissues from Snap Lake in 2013 that were below the normal range were:

- Kidney: barium, bismuth, chromium, manganese, phosphorus, rubidium, and zinc (Table 9-4);
- Liver: arsenic, calcium, magnesium, manganese, phosphorus, rubidium, and zinc (Table 9-6); and,
- Muscle: barium, chromium, nickel, and rubidium.

Mean parameter normal range exceedances in Round Whitefish from Snap Lake in 2013 were:

- Kidney: cesium, lead, molybdenum, and thallium (Table 9-4);
- Liver: cesium, iron, and mercury^{predicted} (Table 9-6); and,
- Muscle: cesium, magnesium, potassium, and thallium (Table 9-8).

9.4.4 Guideline Comparisons

There were no CFIA (2009) arsenic (3.5 mg/kg ww) or lead (0.5 mg/kg ww) guideline exceedances in Lake Trout or Round Whitefish tissues in 2013.

As at baseline, Lake Trout from each of the study lakes had kidney, liver, and muscle mercury concentrations above the CFIA (2009) guideline of 0.5 mg/kg ww. Only one Round Whitefish had liver tissue mercury concentrations above the CFIA (2009) guideline.

Lake Trout

The number and percentage of Lake Trout kidney, liver, and muscle tissue samples with measured concentrations of mercury above the CFIA (2009) commercial consumption guideline in each study lake were:

- Kidney: 7 out of 10 (70%) in Snap Lake, 9 out of 11 (82%) in Northeast Lake, and 5 out of 10 (50%) in Lake 13;
- Liver: 5 out of 10 (50%) in Snap Lake, 7 out of 11 (64%) in Northeast Lake, and 3 out of 10 (30%) in Lake 13; and,

- Muscle: 2 out of 10 (20%) in Snap Lake, 6 out of 11 (55%) in Northeast Lake, and 1 out of 10 (10%) in Lake 13.

Round Whitefish

One Round Whitefish from Snap Lake had a liver tissue sample with measured mercury concentrations above the CFIA (2009) commercial consumption guideline. This is consistent with the 1999 results where composite liver samples from Snap Lake were above the CFIA mercury guideline (De Beers 2002). There were no Round Whitefish liver tissue samples above the CFIA (2009) guideline in the reference lakes.

9.4.5 Summary

There were numerous statistically significant differences in tissue chemistry in 2013 in both Lake Trout and Round Whitefish from Snap Lake. These differences were evident across tissue types and relative to baseline, reference lakes, and the normal range (Table 9-9). The magnitude of the differences between Snap Lake and the reference lakes was variable. In some cases, the differences were small; for example, the magnitude of the difference between phosphorus in Snap Lake Round Whitefish muscle versus the reference lakes Round Whitefish was 6%. In other cases, the magnitude of the difference was large; for example, the magnitude of the difference between thallium in Snap Lake Round Whitefish concentration and the reference lakes was 128% (Tables 9-3 to 9-8). There were also numerous differences from one reference lake to another (Section 9.4.2).

Table 9-9 Summary of Statistically Significant Differences and Normal Range Exceedances in Lake Trout and Round Whitefish Tissue Chemistry Parameters Collected from Snap Lake in 2013

Parameters	Lake Trout				Round Whitefish			
	Comparisons to reference in 2013			Comparison to baseline	Comparisons to reference in 2013			Comparison to baseline
	Kidney	Liver	Muscle	Muscle	Kidney	Liver	Muscle	Muscle
Aluminum	↓	↓↓	-	-	-	-	-	-
Antimony	↓	-	-	-	-	-	-	-
Arsenic	↓↓	↓↓	↓↓	nt	-	↓↓	-	nt
Barium	-	-	-	-	↓↓	↓	↓	-
Bismuth	↓	-	-	-	↓↓	-	-	-
Cadmium	-	↓↓	-	-	↑	-	-	-
Calcium	-	↓↓	↓	-	↓	-	-	-
Cesium	-	-	-	-	↑↑	↑↑	↑↑	↑↑
Chromiuim	↓	↓	-	-	-	↓	-	-
Cobalt	-	↓	-	-	↑	-	-	-
Copper	-	↓↓	-	↓	-	↓	-	-
Iron	-	-	-	↓	↑	↑↑	-	-
Magnesium	-	-	-	-	-	↓↓	↑	-
Manganese	-	-	-	-	↓↓	↓↓	↓	-
Mercury	-	-	-	-	-	↑	-	-
Mercury[predicted]	↑	-	-	↓	↑	↑↑	↑	↓
Molybdenum	-	-	-	-	↑↑	-	-	-
Nickel	↓	-	-	-	↑	↓	↓	↓↓
Phosphorus	-	-	-	↑	-	↓↓	↑	-
Potassium	-	-	-	↑	-	-	↑↑	-
Rubidium	↓↓	↓↓	↓↓	↓↓	↓↓	↓↓	↓	↓↓
Selenium	-	-	↑	↓	-	-	↓	↓
Selenium[predicted]	-	-	nc	nc	nc	nc	nc	nc
Silver	-	↓↓	-	-	↓	-	-	-
Sodium	↑	-	-	-	-	-	-	↑
Strontium	↑↑	↑↑	-	↑	-	-	-	↑
Thallium	↑↑	↑↑	↑↑	nt	↑↑	-	↑↑	nt
Zinc	-	↓↓	-	-	↓↓	↓↓	↓	↓

Note: "-" indicates no change; ↓ or ↑ indicates a statistically significant difference in the direction indicated; ↑↑ or ↓↓indicates a statistically significant difference in the direction indicated that is also beyond normal range.

Solid boldly boxed (▬) cells indicate Low Action Levels that have been triggered based on elevated concentrations (De Beers 2014); **dashed boldly boxed (- - -)** cells indicate parameters that are not considered Low Action Level triggers because concentrations are not elevated, but consistently decreased concentrations across tissue types and species that are also below normal range are noted.

nt = not tested due to lack of sufficient baseline data; nc = not calculated due to lack of a significant regression relationship between selenium and fork length.

9.5 Discussion

The present (2013) large-bodied fish tissue chemistry study reported statistically significant differences in fish tissue parameters between Snap Lake, Northeast Lake, and Lake 13. Differences between 2013 Snap Lake and baseline or reference lake muscle parameter concentrations that were also above normal range were considered within the AEMP Response Framework as Low Action Level exceedances, and are discussed in Section 9.5.1. Statistically significant differences between Snap Lake muscle parameter concentrations that were greater than reference lakes, but not greater than baseline, are discussed in Section 9.5.2. Statistically significant differences between Snap Lake liver and kidney parameters that were determined to be less than reference lake concentrations in 2013, and that were below the normal range, were not considered Low Action Level triggers; these are also discussed in Section 9.5.2.

9.5.1 Action Level Triggers

A revised Low Action Level is proposed for fish tissue on the basis of the revised normal range (Appendix 9A). It is thought the new level is conservative and appropriate for “early-warning” in large-bodied fish tissue focussing on parameters differing from baseline, reference, and outside the normal range. Parameters in muscle tissue were assessed as Low Action Level triggers as follows:

- Concentration in Snap Lake in 2013 was statistically different from the Snap Lake baseline, if available;
- Concentration in Snap Lake in 2013 was statistically different from the 2013 reference lake concentration (either pooled or individual reference lakes); and,
- Concentration in Snap Lake in 2013 was above the normal range.

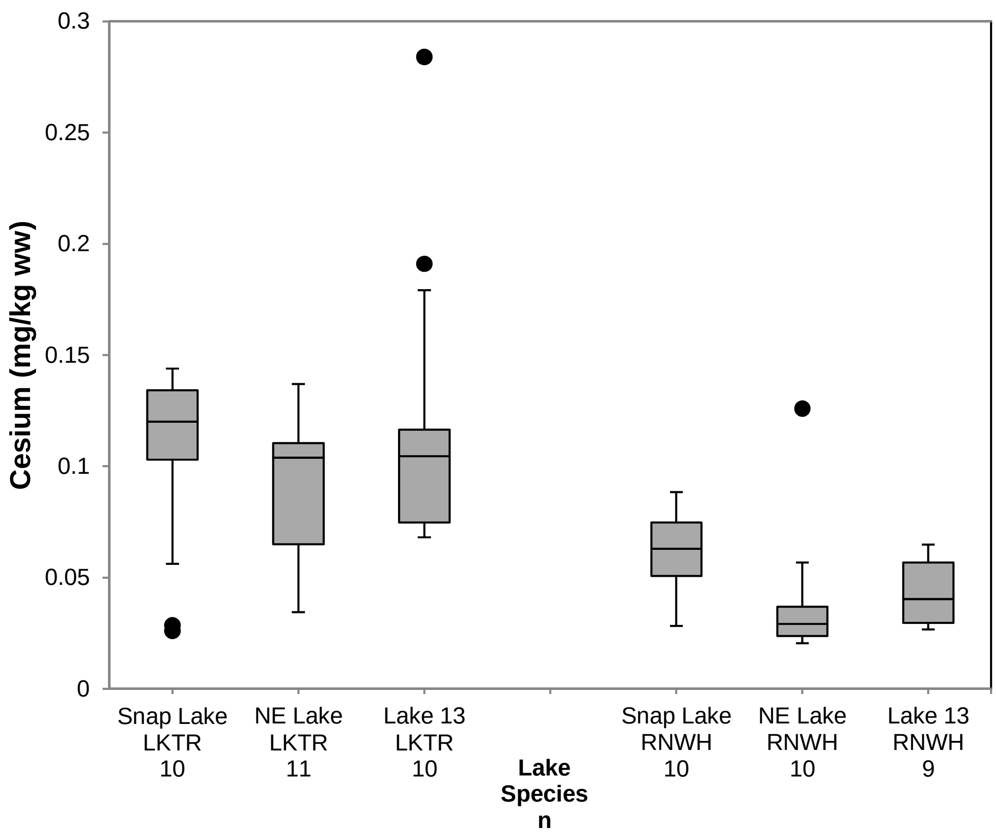
Parameters that met the above conditions were considered Low Action Level exceedances under the Fish Health categories of Toxicological Impairment and Nutrient Enrichment and Fish Safe to Eat (Tables 9-7 through 9-9). In 2013, two parameters, cesium and thallium, triggered a Low Action Level on the basis of elevated tissue concentrations (Table 9-7 through 9-9).

Cesium

Cesium was significantly elevated in Round Whitefish liver, kidney, and muscle tissue in 2013, was significantly elevated relative to baseline in muscle tissue, and was outside of the normal range for each tissue type (Tables 9-4, 9-6, and 9-8; summarized in Table 9-9; Appendix 9A Figures 9A-35, 9A-83 and 9A-125). A review of the 2013 Snap Lake results and recent years data (i.e., 2004, 2009) was conducted (Table 9-10). Mean Round Whitefish muscle cesium concentration in 2013 was only slightly greater than in Snap Lake in 2004 (approximately 0.05 mg/kg in 2004 up to 0.06 mg/kg in 2013; Table 9-10).

Muscle cesium concentrations in Lake Trout from Snap Lake in 2013 were not different than the pooled reference lake cesium concentrations although they were significantly lower in Snap Lake in 2004 (Table 9-7; Figure 9-1). Mean cesium concentration in Lake Trout kidney and liver tissue in 2013 were not different than the reference lakes (Table 9-9). In 2012, cesium concentrations were not different in Lake Chub from Snap Lake relative to the reference lakes (De Beers 2013).

Figure 9-1 Cesium Concentration in the Muscle Tissue of Lake Trout and Round Whitefish Collected from Snap Lake, Northeast Lake, and Lake 13, 2013



mg/kg ww = milligrams per kilogram wet weight; n = sample size; LKTR = Lake Trout; RNWH = Round Whitefish; NE = Northeast Lake. This figure is the same as in Appendix 9D4-10, and is repeated here for ease of reference.

The reference lakes varied in their cesium concentrations. Liver cesium concentrations were significantly different between the two reference lakes in 2013 for both fish species (magnitude of difference 39%), as was muscle cesium concentration in Round Whitefish (magnitude of difference 13%). Round Whitefish liver cesium concentrations in Snap Lake and Lake 13 were not different from each other (Table 9-6 and Figure 9D3-10).

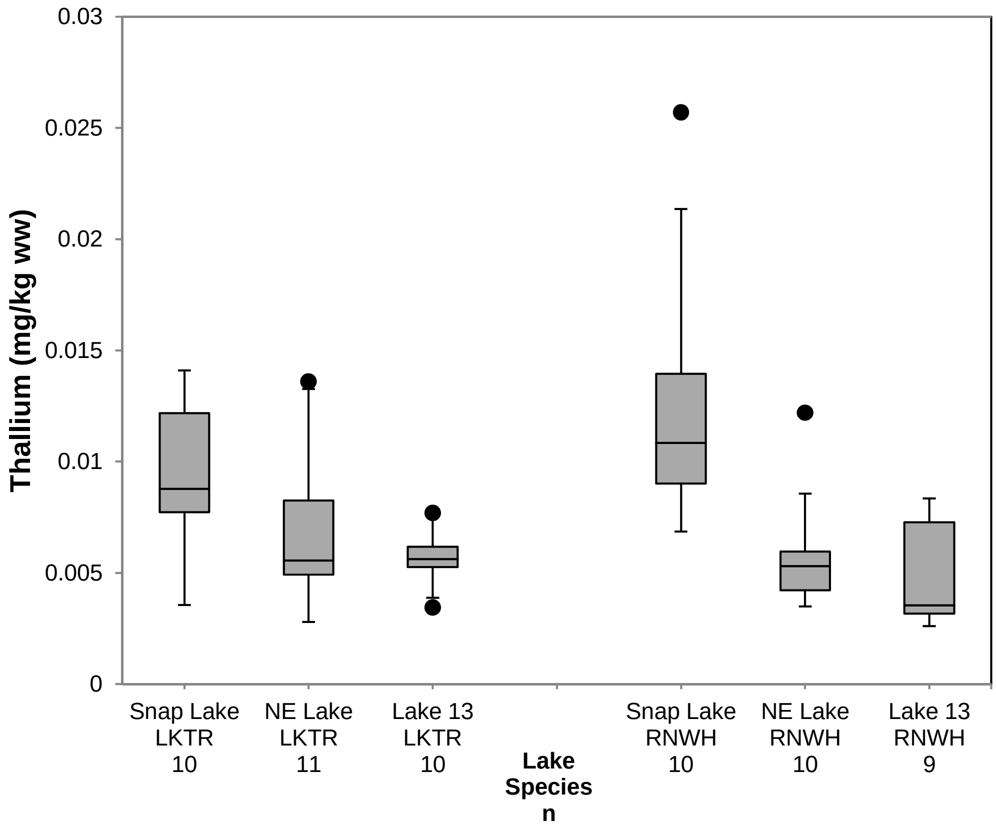
Cesium is an alkali metal with properties similar to rubidium and potassium. Cesium is generally considered to be less toxic than other metals because of its similarity to potassium; there appears to be a correlation between the amount of potassium in water and the amount of cesium accumulated in fish (Phillips and Russo 1978). Cesium concentrations were not predicted to increase in Snap Lake fish tissue due to Mine operations (De Beers 2002). Cesium is occasionally detected in treated effluent (Appendix 3E, Figure 3E-30); however the DL is variable and the number of detections is very low (two) (Appendix 3E, Table 3E-2). Cesium has never been detectable in Snap Lake or local reference lakes waters since monitoring began in 2004 (Appendix 3G, Figure 3G-34). Differences in cesium concentrations in fish tissues may result from non-Mine-related inputs; for instance, the two reference lakes were significantly different from one another. Despite the lack of a clear link to the Mine through effluent, cesium is conservatively considered a Low Action Level trigger for Round Whitefish.

Thallium

Thallium concentrations were elevated and beyond the normal range in both Lake Trout and Round Whitefish muscle (Figure 9-2) and kidney tissues (Appendix 9D, Figure 9D-32). Thallium concentrations in 2013 Snap Lake fish muscle tissue were not compared to baseline due to insufficient data above the detection limit in the baseline data set; however, all muscle thallium concentrations were below the DL in 2004, compared to measurable concentrations in 2009 and 2013 (Table 9-10). Given consistency in trends across species and tissue types in elevated thallium concentrations (Tables 9-7 through 9-9), and measurable concentrations following baseline, the differences in thallium concentrations relative to reference lakes and the normal range were considered sufficient evidence to trigger a Low Action Level for thallium in muscle tissue of both Lake Trout and Round Whitefish in Snap Lake.

Mean thallium concentration in muscle of Round Whitefish was lower in 2013 relative to 2009 (Table 9-10). The opposite was observed in Lake Trout, where 2013 mean muscle thallium concentrations were greater than 2009 (Table 9-10). In 2012, thallium concentrations were also significantly greater in Lake Chub carcasses from Snap Lake relative to the reference lakes (De Beers 2013).

Figure 9-2 Thallium Concentration in the Muscle Tissue of Lake Trout and Round Whitefish Collected from Snap Lake, Northeast Lake, and Lake 13, 2013



Note: This figure is the same as Appendix 9D4-32, and is repeated here for ease of reference.

mg/kg ww = milligrams per kilogram wet weight; n = sample size; LKTR = Lake Trout; RNWH = Round Whitefish; NE Lake = Northeast Lake.

Thallium is a naturally occurring metal that is generally considered to be less toxic than other metals such as mercury or lead. Thallium has an ionic radius similar to potassium and, therefore, may potentially interact with potassium ion transport channels for uptake into fish tissues (Borgmann et al. 1998). There is evidence that thallium and cesium may biomagnify in fish tissues (Lin et al. 2001; Gantner et al. 2009).

It is unclear whether thallium concentrations in fish tissues can be directly linked to the Mine. Thallium is detectable in treated effluent, although in most years the concentrations were below laboratory DLs (Appendix 3E, Figure 3E-46). In 2013, a new lower DL for thallium in water was available and very low concentrations of thallium were noted in treated effluent. Thallium is consistently undetected in Snap Lake and local reference lake waters (Appendix 3G, Figure 3G-49). Thallium concentrations in fish muscle were not different between the reference lakes for either Lake Trout or Round Whitefish, but were different for Lake Trout kidney and Round Whitefish liver. Despite the lack of a clear link to the Mine, thallium is conservatively considered a Low Action Level trigger given the magnitude of the difference in concentration between lakes (e.g., magnitude of the difference in Round Whitefish muscle in Snap Lake was 128% greater than the pooled reference lakes).

Table 9-10 Temporal Comparison for Fish Tissue Parameters that Exceed Low Action Levels

Parameter	Species	Tissue Type	Mean ± Standard Deviation (mg/kg ww)			Are the temporal trends consistent between species?	Are 2013 results consistent among tissue types?	Low Action Level exceedence?
			2013 Snap Lake	2009 Snap Lake	2004 Snap Lake			
Cesium	RNWH	Muscle	0.0609 ± 0.0190	-	0.052 ± 0.0488	no	yes	yes
Thallium	RNWH	Muscle	0.0123 ± 0.0055	0.014 ± 0.0076	<0.04	no	yes	yes
	LKTR	Muscle	0.0094 ± 0.0035	0.0075 ± 0.0041	<0.04		yes	yes

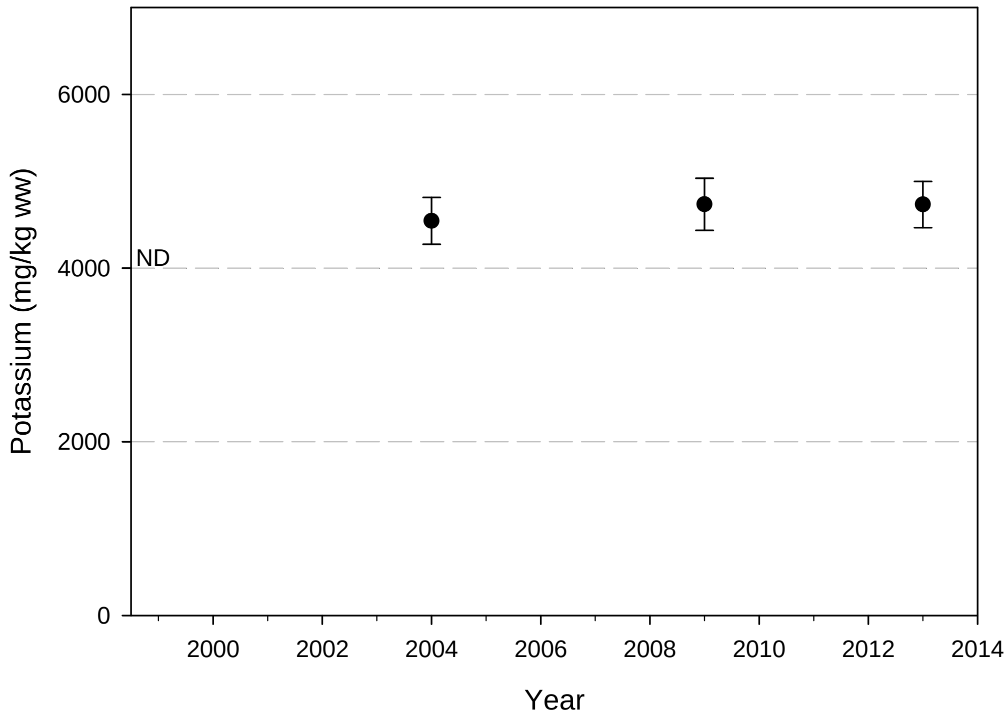
RNWH = Round Whitefish; LKTR = Lake Trout; mg/kg ww = milligrams per kilograms wet weight; ± = plus or minus; < = less than; - = no data.

9.5.2 Other Parameters

Some metals were observed to be increasing, although inconsistently among lakes and tissue types: potassium, iron, mercury, and strontium. Rubidium and zinc decreased over time.

Round Whitefish muscle potassium concentration was significantly elevated in Snap Lake relative to the reference lakes, and was beyond the normal range, but was not significantly different than at baseline (Tables 9-8 and 9-9, Appendix 9D, Figure 9D4-24). While potassium is a nutrient and is essential for healthy cellular function in fish (i.e., integral to the maintenance of ionic homeostasis, nerve transmission, and cellular metabolism), increased potassium concentrations could be indicative of impaired kidney function and an inability to excrete potassium efficiently, or increased uptake from the surrounding environment (Wood et al. 2012a). There are increased concentrations of potassium in Snap Lake water due to effluent discharge containing elevated total dissolved solids (Section 3); however, there are no obvious fish kidney abnormalities indicative of impairment possibly due to potassium (Section 8 in the present report, and Section 7 in De Beers 2013). Potassium concentrations have varied little over time in Round Whitefish muscle tissue in Snap Lake (Figure 9-3), and 2013 potassium concentrations in fish tissues are not significantly different than baseline. Thus, potassium concentration in Round Whitefish muscle does not trigger a Low Action Level exceedance.

Figure 9-3 Potassium Concentrations (Mean ± Standard Deviation) in Muscle Tissue of Round Whitefish Collected from Snap Lake in 2004, 2009, and 2013



± = plus or minus; mg/kg ww = milligrams per kilogram wet weight; ND = not detected.

Iron was elevated in Round Whitefish liver in Snap Lake compared to reference lakes (9% higher than Lake 13, and 54% higher than Northeast Lake) and was outside of the normal range. It is difficult to interpret these results because there is only one other year of baseline liver data (1999) measured on composite livers and not individual samples (Table 9-11). Further, the two reference lakes were also different from each other (magnitude of difference was 35%), and the range of variability in both Northeast Lake and Lake 13 in 2013 largely overlaps with the 2013 Snap Lake data (Appendix 9D, Figure 9D3-15). Therefore, despite the observed differences in Snap Lake compared to reference lakes, mean iron concentration in Round Whitefish liver in 2013 does not trigger a Low Action Level exceedance.

Similarly, mercury^{predicted} concentration was greater in the kidney, liver, and muscle tissue of Round Whitefish and kidney tissue of Lake Trout in Snap Lake in 2013 than in the reference lakes, and mercury^{predicted} was also above the normal range in Round Whitefish liver tissue (Table 9-9). Mercury^{predicted} concentration was not different in Lake Trout liver or muscle tissue in Snap Lake in 2013 than in the reference lakes, and was not above normal range in any Lake Trout tissues (Table 9-9). The mercury^{predicted} concentration in both Lake Trout and Round Whitefish muscle tissue was significantly lower than baseline. The 2013 fish tissue mercury results suggest that mercury concentrations in Snap Lake in 2013 were greater than reference lake mercury concentrations, particularly in Round

Whitefish (Table 9-11), but the mean mercury concentration, as estimated for a standard sized fish, has not increased from baseline. Mercury concentrations were not elevated in Lake Chub carcasses in 2012 relative to the reference lakes (De Beers 2013). Fish tissue mercury concentrations do not trigger a Low Action Level exceedance.

Lake Trout mean liver strontium concentration was substantially greater in 2013 relative to baseline (Table 9-11). Strontium concentration in Lake Trout kidney and liver tissue was significantly greater than reference lakes and was beyond the normal range (Table 9-9), and both Lake Trout and Round Whitefish muscle strontium concentrations were greater than baseline (Table 9-9). Strontium concentrations in 2013 were not different than reference lakes in any tissue type in Round Whitefish. Strontium concentrations were also elevated in Lake Chub carcasses from Snap Lake relative to the reference lakes in 2012 (De Beers 2013). Strontium is preferentially accumulated in bone tissues of fish (Carraca et al. 1990), and can substitute for calcium in physiological processes and bind to calcium-binding proteins in fish, potentially modulating intracellular calcium regulation and transport (Wood et al. 2012b). The changes observed in strontium concentrations in fish may be correlated with the observed decreases in calcium concentrations in Lake Trout liver and muscle tissue, and in Round Whitefish kidney (Table 9-9). Despite the observed differences in strontium concentrations in Lake Trout kidney and liver tissue from Snap Lake compared to reference lakes, there is no comparable response in Lake Trout muscle tissue and any Round Whitefish tissue. Fish tissue strontium concentrations do not trigger a Low Action Level exceedance.

In contrast to the above parameters, rubidium and zinc decreased over time. Mean Lake Trout and Round Whitefish rubidium concentrations were significantly reduced beyond the normal range in both kidney and liver tissues, and were significantly reduced in muscle tissues relative to the reference lakes and baseline (Table 9-9). However, rubidium concentrations were not different in Lake Chub carcasses from Snap Lake relative to the reference lakes in 2012 (De Beers 2013). As per the definition of a Low Action Level for fish tissue (Section 9.2.2.5), a decrease in a parameter that is not beyond normal range does not trigger a response within the AEMP Response Framework (De Beers 2014). Therefore, decreased rubidium in Lake Trout and Round Whitefish tissue do not trigger a Low Action Level.

Similarly, mean liver zinc concentrations were significantly lower than reference mean concentrations and lower than the normal range for both Lake Trout and Round Whitefish. Zinc concentrations in Snap Lake fish tissue in 2013 and 1999 (De Beers 2002) were relatively consistent (Table 9-11). Zinc is an essential metal and is required for the efficient metabolism of proteins, carbohydrates, and lipids and is also involved in immune responses, nervous system function, and cell signalling (Wood et al. 2012a). The changes observed in 2013 liver and kidney zinc concentrations are unlikely to be metabolically limiting due to the small magnitude of change (Table 9-3 through 9-8) observed in zinc concentrations in Snap Lake relative to the reference lakes. Thus, consistent with the definition of a Low Action Level for fish tissue (Section 9.2.2.5), lower fish tissue zinc concentrations do not trigger a Low Action Level exceedance.

Table 9-11 Temporal Comparison of Additional Fish Tissue Parameters

Parameter	Species	Tissue type	Mean ± standard deviation (mg/kg ww)		Are the 2013 results consistent between species?	Are 2013 results consistent among tissue types?
			2013 Snap Lake (n)[a]	1999 Snap Lake (n)		
Iron	RNWH	Liver	174 ± 84	-	no	yes
Mercury	RNWH	Liver	0.229 ± 0.321	0.13 ± 0.04 (5)	no	yes
Strontium	LKTR	Liver	0.228 ± 0.100	0.075 ± 0.028 (11)	no	yes
Rubidium	LKTR	Liver	7.89 ± 2.82	10.25 ± 1.97 (11)	yes	yes
		Kidney	5.91 ± 0.75	-		
	RNWH	Liver	6.72 ± 1.32	7.94 ± 1.68 (5)		yes
		Kidney	5.15 ± 1.54	-		
Zinc	LKTR	Liver	22.9 ± 3.1	22.3 ± 3.6 (11)	yes	no
	RNWH	Kidney	22.0 ± 13.8	-		yes
		Liver	23.9 ± 5.2	24.2 ± 1.5 (5)		

a) 2013 Snap Lake sample size is 10 for all parameters.

RNWH = Round Whitefish; LKTR = Lake Trout; n = sample size; mg/kg ww = milligrams per kilogram wet weight; ± = plus or minus; - = data not available or not appropriate for comparison.

9.6 Conclusions

9.6.1 Key Question 1: Are tissue metal concentrations in fish from Snap Lake increasing relative to reference lakes?

Results from the large-bodied fish tissue chemistry survey conducted in 2013 indicate that tissue concentrations of some parameters in fish from Snap Lake have increased relative to the reference lakes.

For Lake Trout, parameters that had significantly higher concentrations in Snap Lake compared to the reference lakes in 2013 were thallium (in all three tissue types) and strontium (in kidney and liver tissue). The magnitude of differences in Snap Lake relative to the reference lakes were sometimes large for thallium (33% to 128%), and consistently smaller for strontium (36% to 58%). The 2013 mean liver and muscle concentration of thallium, and liver and kidney concentration of strontium, also exceeded the normal range.

For Round Whitefish, parameters that had significantly higher concentrations in Snap Lake in 2013 compared to reference were cesium and mercury (in all three tissue types), iron (in kidney and liver), and thallium (in kidney and muscle). The magnitude of differences in Snap Lake relative to the reference lakes was larger for cesium (40% to 103%) and thallium (107% to 128%), than for mercury (30% to 42%) and iron (26% to 54%). The 2013 mean concentrations of cesium in all tissue types, and thallium in kidney and muscle, also exceeded the normal range.

Other parameters were not significantly greater than reference lake concentrations, and seven parameters had significantly lower concentrations in Snap Lake in 2013 compared to reference lakes (i.e., aluminum, arsenic, barium, manganese, rubidium, silver, and zinc).

Fourteen parameters were significantly different between the reference lakes in both Lake Trout and Round Whitefish tissues (i.e., arsenic, cadmium, cesium, chromium, copper, manganese, nickel, potassium, rubidium, selenium, silver, tellurium, thallium, and zinc). Two parameters were significantly different between the reference lakes in only Lake Trout (i.e., antimony and mercury). Six parameters were significantly different between the reference lakes in only Round Whitefish (i.e., calcium, cobalt, iron, sodium, strontium, and uranium).

There was no evidence of negative effects on fish health, condition, or abundance (Section 8), or risk to human health (Section 9.4.4) due to changes in fish tissue metal concentrations.

9.6.2 Key Question 2: Are tissue metal concentrations in fish from Snap Lake increasing relative to baseline?

The large-bodied fish tissue chemistry survey conducted in 2013 indicates that few parameters in muscle tissue have increased relative to baseline. Only muscle tissue chemistry was tested against baseline.

Concentrations of strontium in Lake Trout muscle tissue were significantly higher in 2013 in Snap Lake than Snap Lake baseline. Phosphorus and potassium concentrations were also significantly higher in Snap Lake muscle tissue from Lake Trout; however, the magnitude of the differences relative to baseline were not large for either phosphorus (17%) or potassium (14%).

Both cesium and strontium concentrations in Round Whitefish muscle tissue were significantly higher in 2013 in Snap Lake than Snap Lake baseline. The magnitude of the difference was 38% for cesium and 79% for strontium. Cesium concentrations also exceeded the normal range. Sodium was significantly higher relative to baseline in Round Whitefish muscle, with a magnitude difference of 32%.

All other parameters were not significantly greater than baseline concentrations in muscle tissue. Parameters that had significantly lower concentrations in Snap Lake in 2013 compared to baseline were cesium, iron, magnesium, mercury, and selenium in Lake Trout muscle, and mercury, nickel, rubidium, selenium, and zinc in Round Whitefish muscle.

9.7 Recommendations

As per the 2013 AEMP Design Plan (De Beers 2014), the next fish tissue chemistry study is the small-bodied Lake Chub survey in 2015. The next large-bodied fish tissue chemistry survey is scheduled to occur in 2016. An additional fish program is scheduled in 2014 in three lakes downstream of Snap Lake, which will include a fish tissue chemistry component. Future fish programs will collect bone and archive the samples for analysis of strontium. These frequencies of future monitoring are considered appropriate to capture early warning signs of any changes occurring in Snap Lake fish tissue chemistry, while balancing the need to minimize mortality to the fish populations in the study lakes.

9.8 References

Borgmann U, Cheam V, Norwood WP, Lechner J. 1998. Toxicity and bioaccumulation of thallium in *Hyalella azteca*, with comparison to other metals and prediction of environmental impact. Environ Pollut 99: 105-114.

Carraca S, Ferreira A, Coimbra J. 1990. Sr transfer factor between different levels in the trophic chain in two dams of Douro rivers (Portugal). Water Res 24: 1497–1508.

CFIA (Canadian Food Inspection Agency). 2009. Canadian Food Inspection Agency Fish Products Standards and Methods Manual: Appendix 3 Canadian Guidelines for Chemical Contaminants and Toxins in Fish and Fish Products. Ottawa, ON, Canada.

Conover WJ, Iman RL. 1982. Analysis of covariance using the rank transformation. Biometrics 38: 715-724.

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

De Beers. 2005. 2004 Annual Report in Support of the Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence MV2001L2-0002. Submitted to the Mackenzie Valley Land and Water Board, Yellowknife, NWT, Canada.

De Beers. 2010. 2009 Annual Report in Support of the Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence MV2001L2-0002. March 2010. Submitted to the Mackenzie Valley Land and Water Board, Yellowknife, NWT, Canada.

De Beers. 2012. Aquatic Effects Monitoring Program Re-evaluation Report: Snap Lake AEMP. Submitted to the Mackenzie Valley Land and Water Board, Yellowknife, NWT, Canada.

De Beers. 2013. 2012 Annual Report. Aquatic Effects Monitoring Program Snap Lake Mine. Type A Water Licence MV2001L2-0002. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2014. 2013 Aquatic Effects Monitoring Program Design Plan. Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board, Yellowknife, NWT, Canada.

Dunn DJ. 1964. Multiple comparisons using rank sums. Technometrics 5: 241-252.

Gantner N, Power M, Babaluk JA, Reist JD, Kock G, Lockhart LW, Soloman KR, Muir DCG. 2009. Temporal trends of mercury, cesium, potassium, selenium and thallium in Arctic char (*Salvelinus alpinus*) from Lake Hazen, Nunavut, Canada: effects of trophic position, size, and age. Environ Toxicol Chem 28: 254-26.

Helsel DR. 2005. Nondetects and Data Analysis: Statistics for Censored Environmental Data. John Wiley and Sons. Hoboken, NJ, USA.

Lin TS, Nriagu J, Wang XQ. 2001. Thallium concentration in Lake Trout from Lake Michigan. Bull Environ Contam Toxicol 67: 921-925.

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

Phillips, GR, Russo RC. 1978. Metal Bioaccumulation in Fishes and Aquatic Invertebrates, a Literature Review. US Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Bozeman, MT, USA.

USEPA (United States Environmental Protection Agency). 2000. Guidelines for Data Quality Assessment. Practical Methods for Data Analysis. Washington, DC, USA. EPA/600/R-96/084.

Wood CM, Farrell AP, Brauner CJ. 2012a. Homeostasis and Toxicology of Essential Metals. Elvsevier, New York, USA.

Wood CM, Farrell AP, Brauner CJ. 2012b. Homeostasis and Toxicology of Non-Essential Metals. Elvsevier, New York, USA.

SNAP LAKE MINE
FISH TASTING ANNUAL REPORT

September 11-12, 2013

DE BEERS
GROUP OF COMPANIES

Table of Contents

INTRODUCTION

Since 2005, Elders have gathered at the Snap Lake Mine in September to share their lifetime of experience and knowledge during an annual Fish Tasting. This year's event took place on the 11th and 12th September.

De Beers Canada Inc. (De Beers) is committed to acknowledging and implementing traditional knowledge. This is an important event that demonstrates how we are living up to the commitment to mine diamonds safely and profitably, without harm to people or the environment. The Fish Tasting is an informal gathering of representatives of involved communities and De Beers' staff to evaluate the condition of large bodied fish (Lake Trout) in Snap Lake by way of tasting and visual inspection. The principal objective of the fish tasting is provide the opportunity to the Elders to determine, in terms of the traditional knowledge and experience, if the flavour and texture of the fish in the lake remains acceptable or human consumption.

Environmental Agreement Commitment #42 from the Information Technical Session, Day 2 (Dec.18/03 p. 91) states that:

> *De Beers will have an annual fish and caribou tasting at site with elders.*
> *The Mackenzie Valley Land and Water Board Water License #MV 2001L2-0002, Part G, Section 2(b) (IV) states:*
>
> *Part G: Conditions applying to aquatic effects monitoring plan.*
> *2. The AEMP shall include, but not limited to the following:*
> *b) A process for measuring the Project-related effects on*
> *Iv. The taste of fish, to be completed with the communities, due to changes in water quality in Snap Lake;*

Elders attend from several communities within the Northwest Territories: Tli Cho Government, Yellowknives Dene First Nation; Lutsel K'e Dene First Nation; and, North Slave Métis Alliance. This year, we welcomed to Snap Lake two new communities represented by elders from the Northwest Territories Métis Nation and Deninu K'ue Nation.

Two elders from different community groups are selected based on an annual rotation schedule and 1one on-site community member/employee volunteers are assigned to the catching of the fish the day prior to the actual fish tasting event. The designated team is then assisted by De Beers staff to catch Lake Trout needed for tasting, using gill nets and/or fishing rods.

The next day, all invited elders, interpreters and observers from the Snap Lake Environmental Monitoring Agency (SLEMA) travel to the Mine to participate in the tasting.

The fish are examined by the elders and then cleaned, with elders examining the internal organs to make sure they look normal and healthy. The fish are then filleted and prepared for to be cooked. New to this year's event, Paul Vecsei, Fisheries Biologist from Golder Associates, joined as an observer in order to ensure that scientific and traditional knowledge observations can be shared across the two spheres. For the official tasting, fillets are boiled in water, eaten without addition of salt, pepper, oil or butter. The manners in which the fish are prepared along with the parameters agreed upon by elders include the following: taste, texture and overall fish health. Elders were interviewed and their comments are documented in the final report.

The Elders are welcomed to collect berries found on the tundra and take home any leftover fish.

As in previous Fish Tasting events, the First Nations groups harvested the plentiful cranberries that grow in the area which were prepared along with bannock and served alongside the fish.

PROCEEDINGS: 2013 FISH TASTING

<u>2013 Sep 11 - Day One of Fishing</u>

Two elders arrive to site a day prior to the event to fish and set gill nets for the fish tasting event the next day.

Weather: 11 degrees Celsius, 220 degree winds at 4 knots; 2,300 Ft Scattered Cloudy Periods, Sunny- Calm
Fisherman: Ernest Boucher & Wayne Langenhan
Assistants: Guylaine Gueguen (Environment Technician) and Freddy Kotchilea (DBC Site Services employee and Behcho Ko Community Member)

1415 hrs- Fishing Start Time

Freddy Kotchilea- Boat Driver
Ernest Boucher requested to troll with a 'Five of Diamond' lures on Ugly stick fishing rod. It was decided to troll on the main basin of Snap Lake until the best spot was selected to set the nets were found.

After trolling for no longer than 10 minutes, Ernest caught a trout east through the narrows into the main basin of Snap Lake. Estimated weight of Lake Trout was 7 pounds. First Lake Trout (and only trout caught by fishing rod) was at 1452 hours.

1552 hrs- First Net Installation (100 yard Length)

The first net was set out in Snap Lake in the main basin between a cluster of islands east of the mine site. It was tied to shore and extended out between two islands. The net was 4.5 inch plastic mesh, 100 yards long and 6 foot wide. The line had a led line and buoys/floats on the opposing line. The net rested afloat on the surface of the water and was anchored on the opposing end to where it was tied off on an island. It was easy to set (Location 1 _Waypoints: 12V 0508940, 7053449)

1619 hrs - Second Net Installation (25 yard length)

Second Net was installed in the water, not too far from the first location, except the net was tied to shore and anchored in the lake towards the main basin (Not between two islands). Weather was beautiful, lake was very calm. The net was also 4.5" mesh, but had cotton mesh (Location 2 waypoints: 12 V 0508981, 7052851).

1626 hrs - Returned to shore after agreeing to check nets in the morning.

<u>2013 Sep 12 - Day 2 of Fishing</u>

Weather: Very Windy (34.78 km/hr)

Fisherman: Ernest Boucher & Wayne Langenham
Assistants: Guylaine Gueguen (Environment Technician) & Freddy Kotchilea (DBC Site Services employee and Community Member)

Start Time: 7:25 am
End Time: 8:17 am

Net Retrieval from Second Location: 3 Lake Trout caught

Net Retrieval from First Location: 13 Lake Trout caught (1 Lake Trout was released as it was still alive and healthy)

<u>2013 Sep 12 - Fish Tasting Event</u>

After all fish were caught they were brought back to site to be weighed and measured the remainder of our guests arrived to site that same morning. Tom Bradbury escorted the elders to the kitchen for breakfast. Shortly thereafter, site orientation was given to the group by Bruce Spencer, DBC Training Coordinator. A power point presentation was given by Guylaine Gueguen to provide background for the new community members that had never attended a previous Fish Tasting at the Snap Lake Mine. A short Q&A session took place allowing the elders to ask questions with Environmental personnel present.

When the site and Fish Tasting orientation was completed, the elders were escorted to the diamond sale that was taking place at the mine site. The diamond sale occurs once a year, and it just so happened that the diamond sale representatives were on site during the same time as the Fish Tasting Event. This provided an opportunity for the elders to view the product mined at Snap Lake.

FISH TASTING

The Gazebo was set up with refreshments, coffee, snacks and all accessories to fillet and cook the fish. The fish were filleted and observations made. Paul Vecsei, Fisheries Biologist from Golder Associates was present to dialogue with the Elders and provides scientific answers to questions posed by anyone attending the events. Paul was a great addition to the event as he provided insight into the observations of the Elders and his contribution highlighted how well traditional knowledge and science can complement each other. He also assisted with the filleting of the fish as we had 17 in total. DeBeers Snap Lake Mine Environmental staff interviewed the Elders while filleting the fish and recorded comments and observations (Appendix B). No abnormalities were observed. Cysts in the stomach lining were observed; however, Paul Vecsei assured the Elders that this is commonly found in Lake Trout.

The fish was then boiled on the fire top and tasted by all Elder's attending the event. Comment of the Fish taste and texture was recorded by Environmental personnel (Appendix B).

The event took place mostly around the picnic tables outside filleting the 17 Lake Trout. Several Elders sat by the fire and shared stories in their native tongue. Bertha Catholique from Lutsel' Ke, fried some bannock served with various jams, butter and Rogers Golden Syrup, while a pot of tea infused with Labrador tea (*Ledum groenlandicum*) was steeped on the fire.

Darren Raymond and Peter Mooney, representing senior management at Snap Lake Mine, stopped by to engage with the Elders around a cup of tea and bannock.

The event ended with the Elders picking cranberries and selecting which fillets of Lake Trout they wanted to bring home with them.

2013 FISH TASTING PHOTO GALLERY

Ernest Boucher_ Fisherman from Lutsel K'e Dene First Nation

Wayne Langenhan_ Fisherman from North Slave Métis Alliance


Day 1_ First Fish Caught by Ernest Boucher from Lutsel K'e Dene First Nation

Freddy Kotchilea, SLM Waste Management Technician and Behcho Ko Community Member, netting a Lake Trout

Day 1_ Installation of the first gill net into Snap Lake

Day 2_ Early morning gill net retrieval

17 Lake Trout caught in total, 1 Lake Trout released

Elders and visitors observing diamonds at the Snap Lake Diamond Sale

Elders and visitors arriving at the fish tasting site in the early afternoon

Fish are filleted and examined by Elders, Visitors and Fish Biologist

Each fish were carefully examined for the any abnormalities and overall fish health

Paul Vecsei, Fish Biologist, attends to provide scientific explanations

Boiling the Fish on the Fire for the Initial Fish Tasting

Bannock Fried by Bertha Catholique to be enjoyed alongside the fish

Tom Bradbury, DBC Permitting Coordinator, frying the remainder of the fish

Madelyn Drybones, a regular attendee of the Fish Tasting Event, enjoying the weather

Peter Mooney, DCB Operations Manager, and Ernest Boucher from Lutsel K'e Dene First Nation

Sitting Along-side the fire, drinking tea and sharing stories

Elder's Completed their visit with some cranberry picking

APPENDIX A: FISH TASTING ATTENDEES

Robert Beaulieu	Deninu Kué First Nation
Leonard Beaulieu	Deninu Kué First Nation
Angus Beaulieu	Northwest Territories Métis Nation
Ernest Boucher (Fisherman)	Lutsel K'e Dene First Nation
Madelaine Drybones	Lutsel K'e Dene First Nation
Eddie Fabian	Northwest Territories Métis Nation
Wayne Langenhan (Fisherman)	North Slave Métis Alliance
Michel Louis Rabesca	Tli Cho Government
Mike Francis	Yellowknives Dene First Nation
George Tatsiechele	Yellowknives Dene First Nation
Bertha Catholique	Chipewyan Interpreter for Madelaine Drybones
Zhong Liu	Snap Lake Environmental Monitoring Agency
Paul Vecsei	GOLDER Associates_ Fisheries Biologist_
Darren Raymond	DBC Safety, Health, Environment & Risk Manager
Peter Mooney	DBC Operations Manager
Bruce Spencer	DBC Training Coordinator
Michelle Peters	DBC Environment & Monitoring Superintendent
Guylaine Gueguen	DBC Environmental Technician
Tom Bradbury	DBC Permitting Coordinator
Freddy Kotchilea	DBC Waste Management Technician and Behcho Ko Community Member

APPENDIX B: FISH CHARACTERIZATION TABLE

Date	Fishing Method	Species Type	Weight (lbs)	Fork Length (inches)
11-Sep-13	Rod	Trout	6.60	27.25
12-Sep-13	4.5" Mesh_ Cotton Gill Net	Trout	5.7	24.0
12-Sep-13	4.5" Mesh_ Cotton Gill Net	Trout	3.65	20.5
12-Sep-13	4.5" Mesh_ Cotton Gill Net	Trout	8.25	26.0
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	6.8	19.35
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	2.0	18.3
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	5.5	24.5
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	7.5	26.5
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	6.25	25.5
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	3.45	19.16
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	6.15	25.25
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	6.45	25.5
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	5.8	26.0
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	3.1	19.16
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	2.75	7.45
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	14.5	1.20
12-Sep-13	4.5" Mesh_ Plastic Gill Net	Trout	19.0	3.10

APPENDIX C: 2013 PREPARATION & INTERVIEW/COMMENTS FROM ELDERS

Preparation

The whole round fish will be reviewed and assessed for health. This would include taking a photograph, verifying the internal organs and general observations when the fish are being prepared for cooking.

1. Preparation of the fish will be only by boiling. Each individual fish will be boiled separately in water that has not been used for the preparation of any prior fish.
2. No cooking medium (oil, butter margarine) spices, seasoning, salt, pepper etc will be applied to the fish.

Fish Health Observation

First Trout Filleted by Ernest Boucher.

Ernest Boucher- “Liver Looks Good” “It is a female. It has already spawned”.
Madelaine Drybones “Liver looks different. Probably because it is spawning”

- Spawning happens earlier at Snap Lake than in Fort Resolution
- Best cooked on fire
- Texture of the fish, ok
- Female fish flesh softer due to the fish spawning

Second Trout Filleted by Ernest Boucher.

Ernest Boucher- “Another female. Few days away from spawning. Full of eggs”
Leonard Beaulieu- “Trout is pale, maybe because they are old”. “Better looking fish than in the Great Slave Lake”.

- The remaining trout were filleted by various community members: Madelaine Drybones, Ernest Boucher was teaching Freddy Kotchilea how to filet the fish, Paul Vecsei assisted. All Lake Trout were filleted, and Elder’s took the remaining fish home with them.

- The fish was boiled and roasted over the fire, and everyone had a taste.

Comments from the Elders after the Fish Tasting-Boiled

- Wayne Beaulieu “I tried the trout without salt and with salt, and they both were good. That was good fish man”.
- Ernest Boucher “Good. Excellent!”
- Madelaine Drybones “Tastes different than the fish in the Great Slave Lake. It tastes more ‘Mossy’. This is probably because the lakes are not connected around here’. – Not true
- Bertha (Translator for Madelaine) - “There is lots of good fish in these lakes”.
- Paul Vecsei-“Fish spawn every two years. Most of the fish filleted today, the eggs are still very small, meaning that the eggs are still developing. The meat is different in males and female fish. When the females spawn, they lack fat and the meat can seem quite ‘Raggedy”. If the meat seems very saturated, it is due to the fish drowning in the nets. If there are cysts present in the fish’ stomach lining, this is more common than not. We should not be alarmed by this”.

APPENDIX D: GOLDER ASSOCIATE FISH TASTING REPORT

Golder Associates

TECHNICAL MEMORANDUM

DATE October 9, 2013 **PROJECT No.** 13-1349-0001-1200

TO Alexandra Hood, Environment & Permitting Superintendant
Michelle Peters, Environmental Monitoring Superintendant
De Beers Canada Inc.

CC Lasha Young, Hilary Machtans, Peter Chapman

FROM Paul Vecsei **EMAIL** Paul_Vecsei@golder.com

DE BEERS SNAP LAKE FISH TASTING 2013 – GOLDER OBSERVATIONS

Golder Associates Ltd. (Golder), at the request of De Beers Canada Inc. (De Beers), participated in one day of the fish-tasting program of the Aquatic Effects Design Program (AEMP). Observations by Golder staff are presented herein.

BACKGROUND

De Beers hosts an annual fish tasting program at Snap Lake as part of the AEMP and Environmental Agreement. The program originally had involvement of both scientists and Elders, and community members. It evolved to include only Elders and community members such that it was a community-led program. In 2012, some fishery questions arose during the tasting, particularly about the types of parasites in fish. After discussions in 2013, it was thought that there may be benefit to having a fish biologist attend the fish tasting to be 'on-call' for technical support should any questions regarding fish health or fish populations arise. As such, De Beers had Golder send one fish biologist to attend and observe and provide any technical comments as required.

2013 FISH TASTING OBSERVATIONS

The fish tasting was held on September 11, 2013. De Beers staff and Elders collected fish in gill nets one day in advance. Numerous fish were captured. On the basis of the biologist observations, the fish-tasting was successful in terms of the number of fish captured and the fact that most attendees commented that the fish taste was 'very good' or 'outstanding'. Detailed results of the tasting will be reported in the annual AEMP report.

Elders did not have technical questions in 2013. There were no questions about parasite load. The Golder biologist observed a low level of parasites in Lake Trout stomach walls (cysts), which are thought to be normal for the species and the lake. The level of parasites is reported every three years in the fish health and fish population chapters of the AEMP report. The biologist also observed that some of the Lake Trout gonads were underdeveloped and small and others were larger with developed eggs; the smaller underdeveloped gonads are likely from fish that would not spawn in fall 2013 (so called 'skip-spawners'). This is also thought to be normal for the lake and the latitude, and has been reported in previous AEMP reports.

Golder Associates Ltd.
9, 4905 - 48 Street Yellowknife, Northwest Territories, Canada X1A 3S3
Tel: +1 (867) 873 6319 Fax: +1 (867) 873 6379 www.golder.com
Golder Associates: Operations in Africa, Asia, Australasia, Europe, North America and South America

A final observation is that the fish may have been in the gill net for a relatively long time. This can sometimes lead to changes in fish texture and taste. It may be advisable in future to check the gill nets more frequently such that any changes to fish texture or taste are not related to the sampling method. This is a minor concern and it is understood that checking nets is often dependent on weather and logistics.

In summary, Golder had one biologist participate in the 2013 fish tasting. The program was well run, appeared to be well received by participants, and no obvious fish health issues were observed by the biologist as fish were filleted.

Thank you for the opportunity to participate in this year's fish tasting. We trust that this letter meets your requirements. If you have any questions or require clarification please contact Paul Vecsei at (867) 873-6319 or Lasha Young at (780) 930-2855.

GOLDER ASSOCIATES LTD.

Hilary Machtans

Hilary Machtans, MSc, on behalf of
Paul Vecsei, PhD
Fisheries Biologist

Hilary Machtans

Hilary Machtans, MSc
Senior Fisheries Biologist

PV/HM/PMC

SECTION 11.1

LITTORAL ZONE SPECIAL STUDY

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Program
C	carbon
Chl *a*	Chlorophyll *a*
CV	coefficient of variation
DIC	dissolved inorganic carbon
DO	dissolved oxygen
DOC	dissolved organic carbon
EEM	Environmental Effects Monitoring
ELA	Experimental Lakes Area
GF/C	glass fiber filter type C
K_z	light attentuation coefficient
MVLWB	Mackenzie Valley Land and Water Board
Mine	Snap Lake Mine
N	nitrogen
NEL	Northeast Lake
%SI	percent surface irradiance
PAR	photosynthetically active radiation
P	phosphorus
QA	quality assurance
QC	quality control
SCUBA	self-contained underwater breathing apparatus
SD	standard deviation
SE	standard error
Si	silica
TDP	total dissolved phosphorus
TDN	total dissolved nitrogen
TN	total nitrogen
TP	total phosphorus
UofA	University of Alberta Biogeochemical Analytical Services Laboratory

LIST OF SYMBOLS

Term	Definition
%	percent
°	degree

UNITS OF MEASURE

Term	Definition
°C	degrees Celsius
cells/cm^2	cells per square centimetre
cm	centimetre
cm^2	square centimetre
ha	hectare
h	hour
kHz	kilohertz
L	litre
m	metre
mm	millimetre
mL	millilitre
µg	microgram
µg/L	microgram per litre
µg/cm^2	micrograms per square centimetre
µm	micrometre
µmol	micromole
µmol/L	micromoles per litre
µmol/cm^2	micromoles per square centimetre
µmol photons/s/m^2	micromoles of photons per second per square metre
µS/cm	microSiemens per centimetre
org/m^2	organisms per square metre
x	times

11 SPECIAL STUDIES

11.1 Littoral Zone Special Study

11.1.1 Introduction

The Littoral Zone Special Study is a three-year study with the objective of evaluating the feasibility of conducting littoral zone monitoring under the Aquatic Effects Monitoring Program (AEMP). In 2012, a preliminary assessment of the littoral zone of Snap Lake and Northeast Lake was completed, focusing mainly on determining the best means of sampling this zone. Littoral zone sampling continued in August 2013 and results are described in this section. This study will be repeated again in 2014, and a full evaluation of the data from all three years will be included in the 2014 AEMP report.

11.1.1.1 Background

Importance of the Littoral Zone

The littoral zone is one of the most diverse and complex areas of any lake ecosystem (Turner 1993). It is the near-shore region and is the link between the catchment area of the lake and the open-water area (Wetzel 2001). Unlike the open-water, which is relatively homogenous, the littoral zone can be a heterogeneous 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 can 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 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 (i.e., open-water) 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 could 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). Epilithon in the littoral zone is a complex biological biofilm consisting of algae, bacteria, and detritus attached to submerged rock surfaces in most aquatic ecosystems, called the epilithon (Wetzel 2001).

Photosynthetically Active Radiation

Photosynthetically active radiation (PAR) is the specific band of solar radiation that is used by plants for photosynthesis. The intensity of solar radiation within the water column influences aquatic life, such as phytoplankton, epilithic algae, and macrophytes, all of which rely on light for photosynthesis and growth. The euphotic zone is the area of the water column that extends from the surface of the water to a depth where PAR is approximately 1 percent (%) of light measured at the surface.

Littoral Nutrients

The attached algal component of rock-associated biofilm is called epilithic algae (Wetzel 2001). It obtains nutrients mostly by diffusion from the overlying water (Kahlert and Petterson 2002). The influence of nutrient concentrations in the surrounding water is greater on epilithic algae than on plant-associated or sediment-associated algae.

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 for epilithic algae. 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 were 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 could 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 could 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, can retain relatively high nutrient concentrations in the epilithon (Sand-Jensen 1983).

The approximate molar ratios that the nutrients in the epilithic algae and associated bacteria and detritus can be found in and 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 can serve as an 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 can be used to 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 supplied to the epilithic algae via a complex mixture of bacteria and detritus that are apart of the epilithon, rendering interpretation of such ratios more complex in the littoral zone than in the pelagic zone.

Littoral Invertebrates

Littoral invertebrates are small aquatic animals that lack backbones (i.e., insect larvae, crustaceans, worms, leeches, snails, and clams). They live on the bottom in the near-shore region (e.g., among, under, or on the surface of rocks, burrowing into or on the surface of sediments, and associated with aquatic plants). Littoral invertebrates form diverse communities, and can consist of thousands of organisms per square metre (org/m^2). Snails are common if calcium concentrations in the water are high, because snails use calcium in shell development, which is the case in Snap Lake.

Littoral invertebrates provide a seasonal food source to egg-laying fish and other small transient fish species (Moss 2005). They also influence epilithic algal biomass and community composition through feeding on epilithon (Lamberti and Moore 1984), and can act as a link between primary producers and fish (Wetzel 2001). As such, they can be useful for monitoring the environmental status of shallow lakes (Rosenberg and Resh 1993).

History of Littoral Zone Sampling 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 MV2001 L2-0002: MVLWB 2004). Periphyton is equivalent to attached algae, which encompasses all forms of attached algae; in Snap Lake the greatest proportion of attached algae is epilithic algae. 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 (Mine).

The 2004 Special Study was designed to assess the feasibility of epilithic algae sampling in Snap Lake, 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.

A three year Littoral Zone Special Study was initiated again in Snap Lake in 2012, following recent AEMP findings of an apparent enrichment effect in the plankton and deep water benthic invertebrate communities, without measurable changes in total phosphorus (TP) concentrations in the water column. This could indicate that P is being intercepted and retained by the littoral zone, with a consequent increase in productivity, which is reflected throughout Snap Lake. In addition, high total organic carbon concentrations in the sediments (i.e., close to 20%) and low dissolved oxygen (DO) concentrations could be an indication of littoral zone material affecting other areas of Snap Lake.

11.1.1.2 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. Specific objectives were to:

- determine the most appropriate and cost effective method for routine sampling during the AEMP;
- determine the importance of the littoral zone to overall productivity in Snap Lake and a reference area, Northeast Lake;
- 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 Mine-related effects; and,
- determine whether littoral data can provide useful additional 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 2014a) 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:

- **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?
- **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?
- **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?
- **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?

11.1.2 Methods

11.1.2.1 Sampling Locations and Timing

The Littoral Zone Special Study was designed on the basis of the 2004 baseline Periphyton Special Study in Snap Lake. In 2012, a subset of the 2004 stations were re-sampled in Snap Lake, and new stations were selected in Northeast Lake to collect reference lake data. An additional station, SNAP LZ08, was added in the northwest arm of Snap Lake in 2013. Five stations in the main basin of Snap Lake, three stations in the northwest arm of Snap Lake, and two stations in Northeast Lake were sampled in 2013 (Figure 11.1-1; Table 11.1-1). An additional three stations in Northeast Lake (NEL LZ03, NEL LZ04, and NEL LZ05) were scheduled to be sampled in 2013, but inclement weather prevented sampling in these areas. Each epilithic algal and littoral invertebrate sampling station was located on cobble or boulder substratum within the euphotic zone, at a depth of 2 metres (m), below the wave-washed zone.

Samples were collected from August 15 to 20, 2013. A mid-August sampling period was selected to replicate the timing of sampling in 2004 and 2012 and to allow sampling during the period of maximum productivity, which is typically in August in the sub-arctic region.

Table 11.1-1 Littoral Zone Sampling Stations in Snap Lake and Northeast Lake in 2004, 2012, and 2013

Lake Area	2004 Station Name[a]	2012 Station Name	2013 Station Name	Zone	Easting	Northing
Main Basin of Snap Lake	PERI 4	SNAP LZ01	SNAP LZ01	12V	507250	7053242
	-	SNAP LZ02	SNAP LZ02	12V	508741	7053978
	PERI 7	SNAP LZ03	SNAP LZ03	12V	508024	7051210
	-	SNAP LZ04	SNAP LZ04	12V	509070	7050770
	PERI 8	SNAP LZ05	SNAP LZ05	12V	509615	7053028
Northwest Arm of Snap Lake	PERI 3	SNAP LZ06	SNAP LZ06	12V	503754	7053448
	PERI 1	SNAP LZ07	SNAP LZ07	12V	502191	7052714
	-	-	SNAP LZ08	12V	506108	7053643
Northeast Lake	-	NEL LZ01	NEL LZ01	12V	508736	7059712
	-	NEL LZ02	NEL LZ02	12V	509921	7059851
	-	NEL LZ03	NEL LZ03[b]	12V	511697	7058828
	-	NEL LZ04[c]	NEL LZ04[b]	12V	TBD	TBD
	-	NEL LZ05[c]	NEL LZ05[b]	12V	TBD	TBD

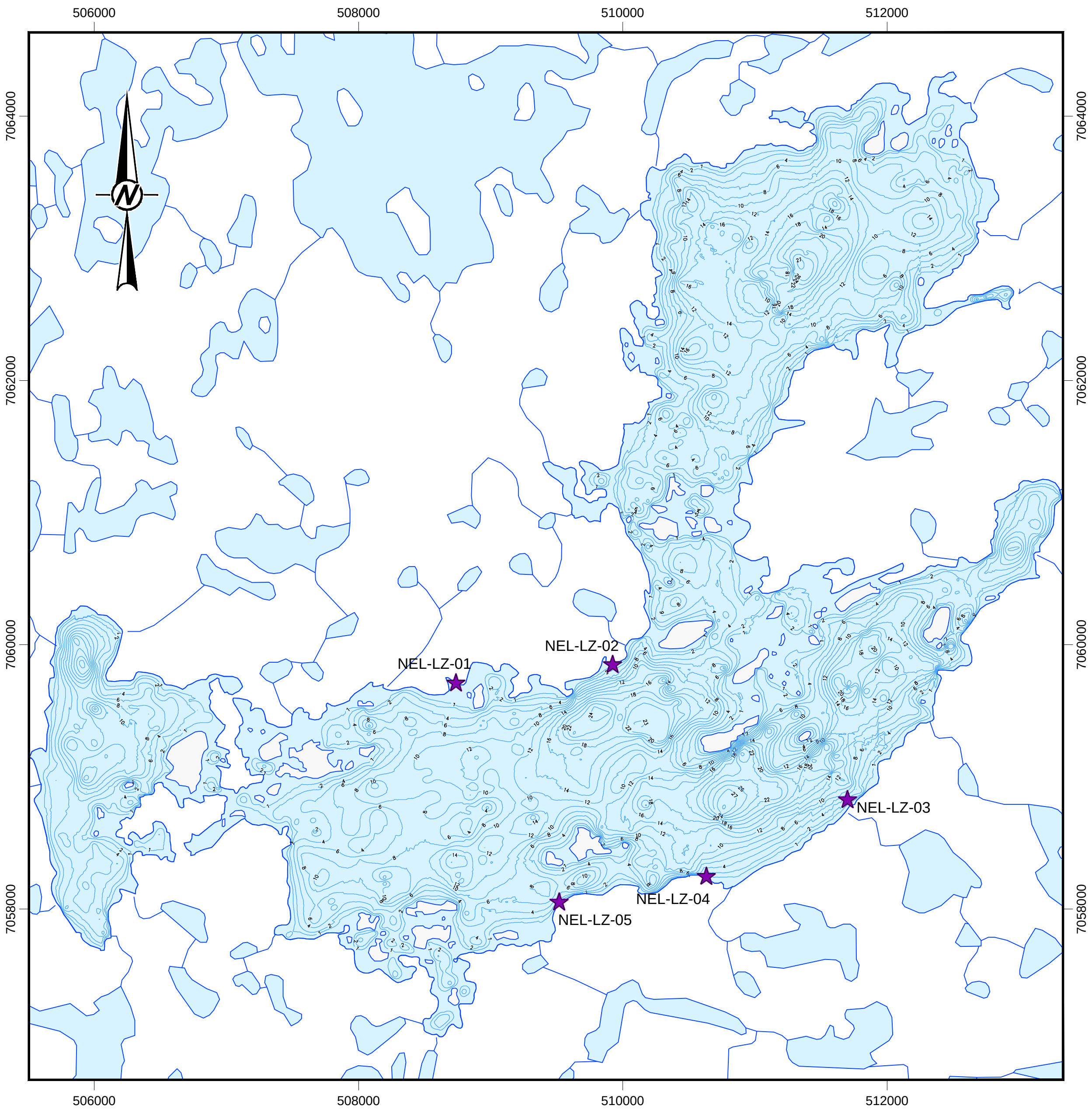
Notes: Universal Transverse Mercator coordinates.

a) Only a subset of the 2004 periphyton stations corresponded to the 2012 and 2013 stations.

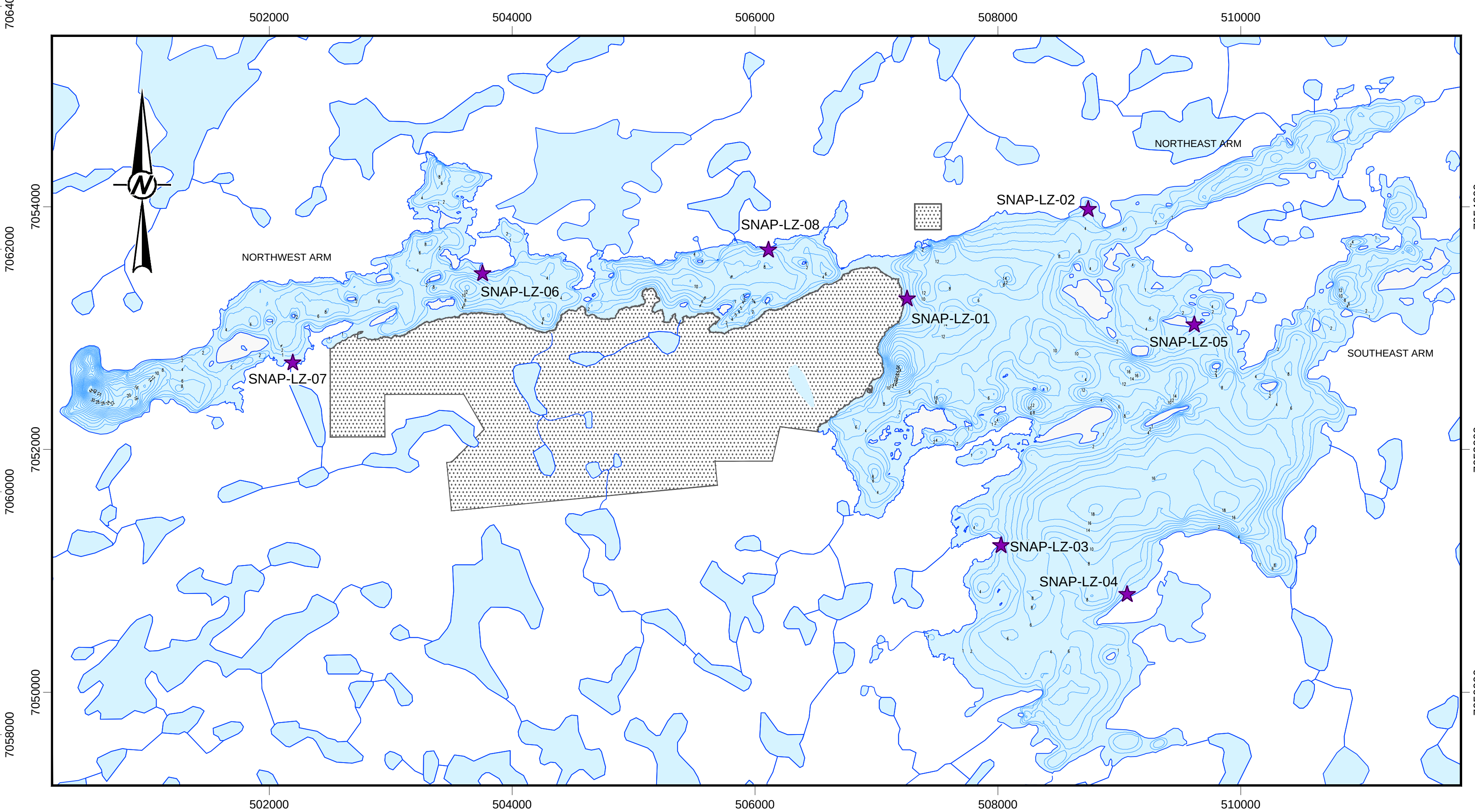
b) Stations were not sampled in 2013 due to inclement weather.

c) Stations were not sampled in 2012 due to inclement weather.

PERI = Periphyton sampling station; SNAP LZ = Snap Lake littoral zone sampling station; NEL LZ = Northeast Lake littoral zone sampling station; - = not sampled; TBD = to be determined.



NORTHEAST LAKE



SNAP LAKE

L:\2013\1349\13-1349-0001\Phase 7000\1313490001-7010-11.1-1.dwg

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- Littoral Study Station

NORTHEAST LAKE REFERENCE

NTS topographic map 75 M/10 É 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 REFERENCE

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

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

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

SNAP LAKE MINE

Littoral Zone Special Study Monitoring Stations in Snap Lake and Northeast Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83	DE BEERS GROUP OF COMPANIES
SCALE 1 0 1 KILOMETRES		
FILE No: 1313490001-7010-11.1-1		DATE: March 18, 2014
JOB No: 14-1349-0004	REVISION No: A	Figure **11.1-1**
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC

11.1.2.2 Sampling Methods

Supporting Environmental Variables

Field water quality parameters (water temperature, DO, pH, and conductivity, herein referred to as conductivity) were measured at each littoral zone station using a YSI 600-QS multi-meter. Light levels were measured at each station using a LI-COR LI-1400 light meter with a spherical light sensor to simultaneously measure upwelling (light reflected back from below the sensor) and downwelling (light entering the water above the sensor). The LI-COR light meter measured PAR as micromoles of photons per second per square metre (μmol photons/s/m^2). Light measurements were recorded at the surface and at the littoral sampling depth of 2 m. LI-COR light meter readings were not recorded at NEL LZ01 during the August littoral program because of inclement weather.

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). 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-Litre (L) plastic bottles. The TDP and TDN samples were then filtered through 0.45-micrometer (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 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 boulder surfaces in areas of low slope (i.e., less than a 10 degree [°] angle, assessed visually). Samples were removed from the rock surfaces by divers using a self-contained underwater breathing apparatus (SCUBA)-based technique. 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 (Appendix 11.1A). *In situ* scraping-brush samplers, based on a design 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).

At each station, three composite samples, each consisting of five 5-cm^2 scrapings (sub-samples), were collected using 60-mL syringes (Table 11.1-2 and Figure 11.1-2). 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 11.1B). A one litre water sample was also taken at each station for use during in-lab sample preparation of the epilthic samples. Only two samples were collected at NEL LZ01 due to inclement weather; one sample from SNAP LZ03 was lost during sample processing when the Whirlpak bag broke.

Table 11.1-2 Littoral Zone Sampling Program, 2013

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	28[(b)]
		QC samples (recounted by Plankton R Us)	2	1	-	2
		Chlorophyll *a*	10	3	Yes	58[(b)]
		Particulate C/N	10	3	Yes	58[(b)]
		Particulate P	10	3	Yes	58[(b)]
Littoral Invertebrates	2 m	Littoral invertebrate community composition (sweep method)	10	1	No	10
		QC samples	1	1	-	1
Littoral Invertebrates	2 m	Littoral invertebrate community composition (Hester-Dendy samplers)	10	3	No	39[(c)] 28[(d),(e)]
		QC samples	3	1	-	3

a) Includes both Snap Lake and Northeast Lake samples.

b) Only two samples analyzed at each of SNAP LZ03 and NEL LZ01. SNAP LZ03C was lost during processing and NEL LZ01C could not be collected due to inclement weather.

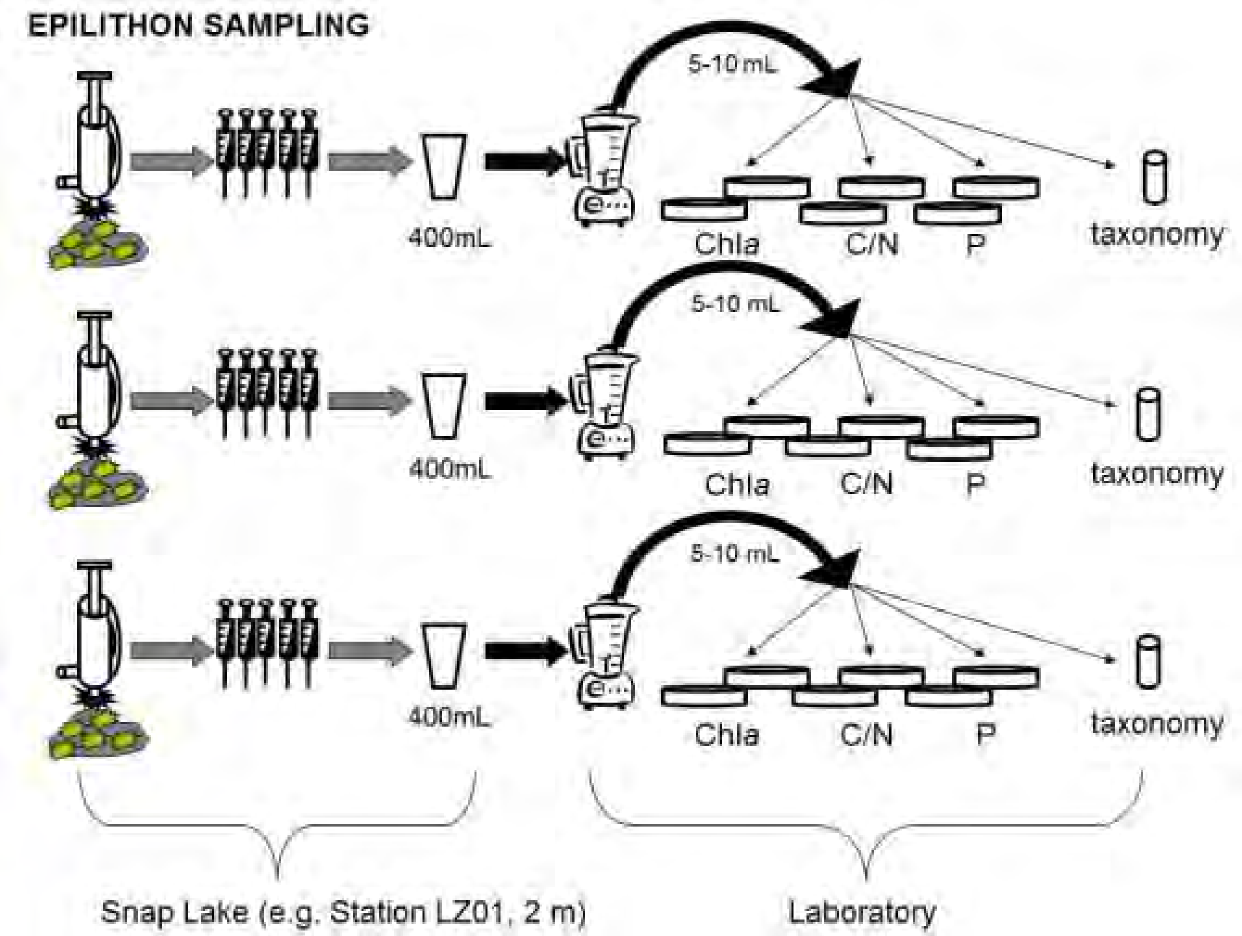
c) Number of Hester-Dendy samplers deployed in July 2013.

d) Only one of three Hester-Dendy samplers was collected at SNAP LZ03. Two samplers were found above the water line. Hester-Dendy samplers were deployed, but not collected, at NEL LZ03, NEL LZ04, and NEL LZ05 due to inclement weather.

e) Number of Hester-Dendy samplers retrieved in August 2013.

"-" = not applicable or not collected; C = carbon, N = nitrogen; P = phosphorus; DIC = dissolved inorganic carbon; DOC = dissolved organic carbon; m = metre; QC = quality control.

Figure 11.1-2 Overview of Epilithic Algae Sample Collection Methods, 2013



Note: Three samples were collected at each station and each sample was made up of five syringes.
mL = millilitre; Chl *a* = Chlorophyll *a*; C = carbon; N = nitrogen; P = phosphorus.

Invertebrates

Sweep Net Method

Sweep net sampling was conducted to collect sufficient material to identify taxa present in the littoral zones of Snap Lake and Northeast Lake, and to make field observations that would allow the refinement of the sampling method. A comparison of two different sieve sizes (i.e., 250 and 500 µm) for collection was completed in 2012 (De Beers 2013). It was determined that a 250 µm mesh sieve would be used during future littoral zone sampling and sample processing because this mesh retained substantially more invertebrate taxa, thereby providing a more accurate representation of the littoral invertebrate community. Littoral invertebrates were collected at a depth of 2 m, after epilithic algal sample collection, and approximately 50 m away from the epilithic algal sampling area to avoid disturbance.

One littoral invertebrate sample was collected at each station, for a total of ten samples (Table 11.1-2). 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 enough 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, PhD, in Summerland, British Columbia, for taxonomic analyses.

Field observations were also made regarding the presence of heavier, shelled, or cased organisms that could not be efficiently collected by the sweep method thereby resulting in selectivity of the method toward small, light, easily-dislodged invertebrates.

Hester-Dendy Artificial Substrate Samplers

The sweep net method did not allow a quantitative assessment of the invertebrates present in the littoral zone; therefore, Hester-Dendy artificial substrate samplers were used to quantitatively sample the invertebrate community in 2013. The Hester-Dendy samplers were deployed in the littoral zone at a depth of approximately 2 m and retrieved after a suitable invertebrate colonization period. The samplers consisted of 14 square plates of tempered Masonite with an area of 7.6 cm^2. Plates were separated by nylon spacers using a configuration of 8 single spacers, 1 double spacer, 2 triple spacers, and 2 quadruple spacers to provide varying distances between plates. The total surface area for the sampler was 160 cm^2.

The Hester-Dendy samplers were deployed on July 5 and 6, 2013 at each littoral zone sampling station in Snap Lake and Northeast Lake. A rope was placed through the eye of each sampler, and attached to a rock located on shore. Three samplers were deployed at each littoral sampling station in Snap Lake (SNAP LZ01 to LZ08) and Northeast Lake (NEL LZ01 to LZ05), for a total of 39 samplers.

Following a six-week colonization period, the Hester-Dendy samplers were removed by divers from August 15 to 20, 2013. To retrieve each sampler, the diver placed a 250 µm mesh bag around the Hester-Dendy sampler. The supporting ropes were cut, the bag gently closed and the sampler brought to the surface and placed in a plastic tub filled with lake water for transport back to the laboratory for sample processing. Of the 39 samplers deployed, 28 were retrieved because two samplers at SNAP LZ03 had washed up onshore and were no longer available to invertebrates for colonization, and inclement weather conditions prevented retrieval of 3 samplers at each of the Northeast Lake stations (i.e., NEL LZ03, NEL LZ04, and NEL LZ05.

Supporting environmental information was collected at each sampling area when samples were deployed and retrieved:

- Hester-Dendy sampler deployment and retrieval date and time;
- a sketch of the location of each sampler;
- weather conditions (i.e., air temperature, wind velocity, percent cloud cover, and precipitation [presence/absence]);
- field water quality measurements (i.e., DO, pH, conductivity, and water temperature) at the time of deployment;
- water depth and distance from shore;
- visual estimate of substrate size;
- sediment characteristics (i.e., colour, odour, organic content, and evidence of anoxia) at the time of retrieval;
- site characteristics near the sampler; and,
- photographs of each station and the Hester-Dendy sampler deployment and retrieval.

11.1.2.3 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. Low light levels were used during sample processing to avoid light damage to the chlorophyll samples. Each sample was prepared following protocols provided by Dr. Michael Turner as 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 material (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 with a magnetic stir bar which was placed on to Nuova II stirrer using a setting of seven. Lake water, collected from the surface at the littoral zone sampling stations, was used to bring the final volume to 400 mL. Ten millilitre 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 millimetre (mm) diameter 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, before 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. Twenty-mL subsamples were removed and transferred to 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. One hundred millilitres of the remaining sample was transferred to a Whirlpak bag and frozen as an archive sample for future analyses.

Littoral Invertebrate Sample Preparation

Hester-Dendy invertebrate samplers were removed from their collection bags, dismantled, and rinsed in their corresponding tubs to remove any clinging organisms using a soft-bristled brush. The water and sample material from each tub was then rinsed through a 250 µm mesh screen. The material retained on the screen was then rinsed into a pre-labelled sample bottle and preserved in 10% neutral buffered formalin. Samples were sent to J Zloty, PhD, in Summerland, BC for identification and enumeration of invertebrates.

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 kilohertz (kHz) on a Sonifer cell Disruptor, Model W140 from Heat Systems, Ultrasonic Inc. for up to two 15 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 125 times (x) 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 250 µm mesh 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 250 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; Thorp 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.

11.1.2.4 Data Analyses

Supporting Environmental Variables

The Littoral Zone Special Study was designed to answer the key questions listed in Section 11.1.1.2. An overview of the data analysis approach associated with each of the four key questions is provided in Table 11.1-3.

A qualitative review of the data was completed. Summary statistics (i.e., arithmetic mean, median, minimum, maximum, standard deviation [SD] and standard error [SE]) were calculated for particulate C, N, P, chlorophyll *a*, and epilithic algal biomass and abundance.

Averages for stations in 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. Averages from Snap Lake were compared to those from 2012 and 2004, while averages in Northeast Lake were compared to those from 2012. The mean plus or minus standard error (± SE) for the settled volume of particulate material was calculated for each station, and was compared to values from 2012.

The percent of total lake area available for epilithic algal colonization was calculated using the following formula:

$$\text{Area of littoral zone (\%)} = \frac{\text{Area of the lake with depths from 0 to 4 m} \times 100}{\text{Total lake surface area}} \qquad \text{[Equation 11.1-1]}$$

All light measurements were expressed as a percentage of the surface irradiance value (%SI) calculated as follows:

$$\%\ SI = (I_z/I_0) \times 100 \qquad \text{[Equation 11.1-2]}$$

where Iz and I0 are irradiance (μmol photons/s/m^2) at depth z (m) and at the surface, respectively.

Table 11.1-3 Overview of Analysis Approach for Littoral Zone Special Study Key Questions, 2013

Key Question	Overview of Analysis Approach
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?	This question will be answered after three years of the Littoral Zone Special Study. An annual assessment of the within-station variability was based on the 2012 and 2013 data. The CV among the samples was calculated for each station (Appendix 11.1C, Table 11.1-5) for particulate C, N, P, ratios of C, N, and P, chlorophyll *a*, and epilithic algal abundance and biomass. In addition, within-lake variability was described by examining among-station variability and spatial trends in Snap Lake and Northeast Lake.
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, 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 11.1C, Table 11.1-5). 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 2013. Values in the main basin and northwest arm of Snap Lake were also compared to values from 2012 and baseline (2004) values, while values in Northeast Lake were compared to values from 2012. Nutrient ratios were also compared to values reported in the literature (Healey and Hendzel 1980; Hillebrand and Sommer 1999; Elser et al. 2000) that indicate nutrient status and food quality.
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, 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 (Appendix 11.1C, Table 11.1-5). 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 2013. Total biomass and abundance in Snap Lake and Northeast Lake were compared to values from 2012. Relative abundance and biomass in Snap Lake were compared to values from 2012 and baseline (2004) values, while values in Northeast Lake were compared to values from 2012.
4. 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?	Total density, relative density, functional feeding group density, taxa richness, evenness, and relative densities of the major invertebrate and Diptera taxa, collected by Hester-Dendy samplers, were calculated for each station. Stations in the main basin of Snap Lake were compared to those in the northwest arm of Snap Lake and Northeast Lake in 2013. Relative densities of the major invertebrate taxa, collected by the sweep net method, 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 addition, data obtained from sweep net samples in 2013 were compared to data from 2012.

CV = coefficient of variation; SE = standard error; C = carbon; N = nitrogen; P = phosphorus.

Once light enters the water, light intensity decreases logarithmically with water depth (Wetzel 2001). This is quantified by the extinction or attenuation coefficient, which is the fraction of light that is absorbed per metre and is related to reflection, refraction or scattering, and absorption by water, dissolved compounds, and suspended particles. The maximum depth of the euphotic zone depends on the extent of this light attenuation in the water column. The vertical light attenuation coefficient (K_z) was calculated to compare light attenuation through the water column at different stations. Light attenuation was calculated using the transformed Beer-Lambert equation as follows:

$$K_z = -[\ln(Iz/I0)]/z \quad \text{[Equation 11.1-3]}$$

where K_z is the attenuation coefficient at a specific depth z (m), and Iz and I0 are irradiance (μmol photons/s/m^2) at depth z and at the surface, respectively.

Littoral Zone Nutrients

Molar ratios were calculated for each particulate nutrient parameter (i.e., C, N, and P). Particulate nutrient concentrations in micrograms per square centimetre (µg/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 (± SE) for each molar ratio (C:N, C:P, and N:P) was calculated for each station. The ratios from 2013 were compared to those from 2012 and 2004 in Snap Lake, and 2012 in Northeast Lake. 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). Particulate nutrient (i.e., C, N, P) concentrations from 2013 in Snap Lake and Northeast Lake were compare to those from 2012.

The amount of carbon on the rock surface, in micromoles per square centimetre (µmol/cm^2), was divided by the chlorophyll *a* concentration in micrograms per square centimeter (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 (± SE) was calculated for each station. The ratios from 2013 were compared to those from 2012 and 2004 in Snap Lake, and 2012 in Northeast Lake. Chlorophyll *a* concentrations from 2012 and 2013 were compared for both lakes.

The percent algal carbon associated with the epilithic algae and associated bacteria and detritus was also calculated for each station. Values for Snap Lake were compared among 2004, 2012, and 2013, while values from Northeast Lake were compared between 2012 and 2013. Percent algal carbon was calculated by 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 Algae

Total abundance and biomass of epilithic algae were calculated for each station in Snap Lake and Northeast Lake and compared to values from 2012. Epilithic algal abundance and biomass data were divided into groups based on taxonomic results:

- Cyanobacteria;
- Chlorophyceae (chlorophytes);
- Bacillariophyceae (diatoms); and,
- “others” (i.e., Chrysophyceae, Dinophyceae, and Euglenophyceae).

The relative proportion accounted for by each group, based on both abundance and biomass, was calculated separately for each station. Stations in the main basin of Snap Lake were compared to those in the northwest arm of Snap Lake and Northeast Lake. In addition, relative abundances and biomass in Snap Lake were compared to values from 2004 and 2012, while those in Northeast Lake were compared to values from 2012.

Within-Station and Among-Lake Variation in the Eplithon

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 described by examining station-to-station variability and spatial trends in Snap Lake and Northeast Lake.

Invertebrates

Littoral invertebrate data were reviewed and non-littoral organisms, such as calanoid copepods (Calanoida), cyclopoid copepods (Cyclopoida), water fleas (Cladocera), and terrestrial invertebrates were removed from the data set before data analysis. True fly (Diptera) pupae were also removed before data analysis.

The littoral invertebrate sweep net data from 2012 and 2013 were compared and evaluated qualitatively (i.e., based on presence or absence), while the Hester-Dendy data from 2013 were converted from abundance (number per sample) to density, or number of organisms per square metre (org/m^2), and evaluated quantitatively. The following invertebrate summary variables were calculated for each littoral zone station:

- total invertebrate density (Hester-Dendy samples only);
- taxonomic richness (Hester-Dendy samples only);
- Simpson's index of diversity (Hester-Dendy samples only);
- evenness (Hester-Dendy samples only);
- community composition (i.e., relative densities of major invertebrate taxa from Hester-Dendy and sweep net samples);
- Diptera composition (i.e., relative densities of major Diptera taxa from Hester-Dendy and sweep net samples); and,
- Relative density of functional feeding groups (Hester-Dendy samples only).

Density estimates can be used to evaluate the number of organisms present on the rock surface; greater densities mean more food for fish. Increases in nutrients and attached algae can equate to increased invertebrate density; conversely, toxicity can reduce the overall density of invertebrates. Richness is the total number of taxonomic groups within a station. It provides an indication of the diversity of littoral invertebrates in an area; a higher richness value usually indicates a more healthy and balanced community.

Simpson's index of diversity measures the proportional distribution of organisms in the community, given that not all organisms have the same success in the environment. Certain conditions could favour one organism over another (Simpson 1949). Simpson's index of diversity values range between 0 and 1, where lower values indicate a community dominated by fewer taxonomic groups (less diverse); these are often referred to as stressed communities. Values close to 1 indicate a community consisting of more taxa that are more evenly distributed among the taxonomic groups present. Simpson's index of diversity was calculated using the formula provided by Krebs (1999), as recommended by Environment Canada (2012) for Environmental Effects Monitoring (EEM) programs:

$$D = 1 - \sum_{i=1}^{S} (p_i)^2 \qquad \text{[Equation 11.1-4]}$$

where D = Simpson's index of diversity; S = the total number of taxa; and p_i = the proportion of the i^{th} taxon.

Evenness is an index recommended by Environment Canada (2012) for analyzing EEM data. It is a measure of how evenly the total invertebrate density is distributed among the taxa present at the site. Evenness is also expressed as a value between 1 and 0, with one representing high evenness and zero representing low evenness. Evenness was calculated using the formula provided by Smith and Wilson (1996):

$$E = 1/ \sum_{i=1}^{S} (p_i)^2 / S \qquad \text{[Equation 11.1-5]}$$

where E = Evenness; p_i = the proportion of the i^{th} taxon; and S = the total number of taxa.

To determine community composition, the littoral invertebrate data were divided into major taxonomic groups, and densities of major invertebrate taxa were calculated for each station in Snap Lake and Northeast Lake. Major taxa that collectively represented less than 10% of the total density in all areas were combined into the "Others" group. The "Others" category differed depending on whether samples were collected from Hester-Dendy samplers (Others = Cnidaria, Nematoda, Nematomorpha, Pelecypoda, Plecoptera, and Trichoptera) or by the sweep net method (Others = Cnidaria, Nematoda, Pelecypoda, and Trichoptera).

Diptera (mostly Chironomidae) dominated most samples at all stations. Consequently, densities of each subfamily or tribe within the Diptera were plotted to compare sampling areas. If subfamilies or tribes collectively represented less than 10% of the total density in all areas, they were combined into the "Others" group (i.e., Ceratopogoninae, Dasyheleinae, Empididae, Pseudochironomini, Prodiamesinae, and Diamesinae).

Functional feeding groups (i.e., collector-gatherers, scrapers, filterers, predators, herbivores, shredders, parasites, suspension feeders, deposit feeders, and grazers) were assigned to the lowest taxonomic level 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.

11.1.3 Quality Assurance and Quality Control

11.1.3.1 Overview of Procedures

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. The QA/QC protocols are designed so that field sampling, laboratory analyses, data entry, data analyses, and report preparation activities produced 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 11.1A.

11.1.3.2 Summary of Results

The methods used during this study to collect the littoral zone samples are widely accepted in-lake methods (Turner et al. 1987). The certified commercial divers were scientifically-trained in 2013. Divers were provided with an extensive briefing about the work that was required and how it should be conducted, including the importance of consistency in sampling. Sampler bias and field technique inaccuracies improved in 2013 compared to in 2012 (Appendix 11.1A). Issues with the sampling technique in 2012 could have also caused an under-representation of cyanobacteria by biasing the samples towards firmly attached algae rather than light, flocculent forms, which were observed in the photographs. Collections performed by scientifically-trained divers during the 2013 program resolved these issues (Appendix 11.1A, Appendix 11.1B).

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 the 2004, 2012, and 2013 data, when comparisons among years were needed. The use of proportional data allows for comparisons to be made among stations, between lakes, and among years (Appendix 11.1A).

A species-level presence or absence assessment of the community was not performed because different taxonomists analyzed the samples in 2004 compared to 2012 and 2013. 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 occurred in 2013; however, results were not available at the time that this report was being prepared. These results will be provided following the three years of this Special Study.

Based on results for field blanks and split samples, potential QC issues were identified for particulate carbon, particulate nitrogen, particulate phosphorus, and chlorophyll *a* (Appendix 11.1A). None of the field blank results from 2013 exceeded the QC criterion, and no replicate values were removed from further analyses. Of the split samples, 15 results (18% of total) showed a relative difference of more than 20%, resulting in QC flags. Overall, the QC results of the taxonomy data indicated that the 2013 littoral zone data were of acceptable quality and no data were invalidated. Detailed QA/QC results are provided in Appendix 11.1A.

11.1.4 Results

11.1.4.1 Supporting Environmental Variables

Snap Lake and Northeast Lake are small, shallow arctic lakes (Table 11.1-4). In 2013, 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 134 days of open-water in 2013 (Section 2). 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 are of similar size.

Table 11.1-4 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)]	%	48	59	40

a) Area of littoral zone (%) = (Area of lake with depths from 0 to 4 m / Total lake surface area) x 100.

ha = hectares; % = percent.

Northeast Lake (1,843 hectares [ha]) has a slightly larger surface area compared to Snap Lake (1,566 ha; Table 11.1-4). 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 (Tables 11.1-4 and 11.1-5) 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%), and Northeast Lake (40%; Table 11.1-4).

Irradiance measurements were collected in the littoral zone in 2013. The percent of surface irradiance available at a depth of two metres ranged from 10% to 28% (Table 11.1-5). Irradiance tended to be higher in the main basin of Snap Lake (16% to 28%) compared to the northwest arm (10% to 22%) and Northeast Lake (17%), but this could be a reflection of the lower ambient light conditions on the days that Northeast Lake and the northwest arm of Snap Lake were sampled (Table 11.1-5).

Table 11.1-5 Littoral Zone Light Data in Snap Lake and Northeast Lake, August 2013

Lake/Area	Station	Surface	Sample Depth (2 m)	Surface Irradiance at 2 m	Vertical Light Attenuation (*Kz*)
		(µmol photons/m²/sec)	(µmol photons/m²/sec)	(%)	(µmol photons/m²/sec)
Main Basin of Snap Lake	SNAP LZ01	1,585	260	16	0.9
	SNAP LZ02	3,330	868	26	0.7
	SNAP LZ03	3,140	512	16	0.9
	SNAP LZ04	3,020	850	28	0.6
	SNAP LZ05	2,950	780	26	0.7
Northwest Arm of Snap Lake	SNAP LZ06	1,150	120	10	1.1
	SNAP LZ07	775	173	22	0.8
	SNAP LZ08	1,610	352	22	0.8
Northeast Lake	NEL LZ01	-	-	-	-
	NEL LZ02	530	90	17	0.9

m = metre; % = percent; "-" = not measured because of inclement weather; µmol photons/m²/sec = micromoles of photons per square metre per second.

At the time of sampling in mid-August 2013, there was little variation in mean temperature and DO among lakes at the sampling locations and sampling areas (Table 11.1-6). Mean conductivity in Snap Lake was higher in 2013 compared to 2012, but followed a similar trend in both years, with higher conductivity observed in the main basin (462 µS/cm) and northwest arm (194 µS/cm) of Snap Lake compared to Northeast Lake (21 µS/cm). Mean conductivity in Northeast Lake in 2012 (22 µS/cm) and 2013 (21 µS/cm) was similar to that observed in Snap Lake in 2004 (26 µS/cm). Differences among areas were noted in pH; average pH values were higher in the main basin (7.8) and northwest arm (7.6) of Snap Lake compared to Northeast Lake (7.0; Table 11.1-6). The pH value (7.6) at SNAP LZ01, the station closest to the diffuser, was the lowest in the main basin, but was similar to the range observed in the northwest arm of Snap Lake (7.5 to 7.7; Appendix11.1C, Table 11.1C-1).

Concentrations of DOC in Snap Lake ranged from 2,900 micrograms per litre (µg/L) in the northwest arm of Snap Lake to 5,440 µg/L in the main basin of Snap Lake (Table 11.1-6). The lowest concentrations of DOC were observed in Northeast Lake (1,450 µg/L). Higher concentrations of DOC were observed in Snap Lake in 2013 (2,900 to 5,440 µg/L) compared to in 2012 (2,350 to 4,640 µg/L).

Concentrations of DIC in 2013 were similar to those observed in 2012, with the highest values observed in the northwest arm of Snap Lake (4,300 µg/L), followed by Northeast Lake (3,450 µg/L), and the main basin of Snap Lake (3,360 µg/L; Table 11.1-6). The station closest to the diffuser, SNAP LZ01, had the highest concentration of DIC (4,100 µg/L), and the lowest concentration of DOC (5,200 µg/L) in the main basin of Snap Lake (Appendix11.1C, Table 11.1C-2).

These lakes have naturally low nutrient concentrations, which do not support abundant phytoplankton populations (De Beers 2002). Whole-lake mean TN concentrations in 2013 were greater in the main basin of Snap Lake (2,258 µg/L) compared to the northwest arm (549 µg/L) and Northeast Lake (162 µg/L), which reflects the input of the Mine discharge to Snap Lake; TDN followed the same pattern as TN (Table 11.1-6). Mean TP concentrations in 2013 were lower in the main basin of Snap Lake and Northeast Lake (3.0 and 3.5 µg/L, respectively) compared to the northwest arm of Snap Lake (4.0 µg/L). In addition, mean TP concentrations in the main basin of Snap Lake in 2012 and 2013 were lower than reported in 2004 (3.2 µg/L). Mean TDP has remained below the detection limit throughout the study. Open-water nutrient concentrations measured as part of the plankton component (Section 5) show that both Northeast Lake and Snap Lake are P-limited systems.

Table 11.1-6 Water Chemistry at the Littoral Zone Sampling Stations in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013

Parameter	Units	Snap Lake 2004[a]	Snap Lake 2012[b]		Northeast Lake 2012	Snap Lake 2013[b]		Northeast Lake 2013
			Main Basin	Northwest Arm		Main Basin	Northwest Arm	
Mean Temperature	°C	-	16.6	15.6	16.2	18.1	17.7	17.6
Mean Dissolved Oxygen	mg/L	-	9.9	10.4	9.8	9.3	9.0	9.1
Mean Conductivity	µS/cm	26	397	125	22	462	194	21
Mean pH	-	7.0	7.7	7.4	7.3	7.8	7.6	7.0
Mean Dissolved Inorganic Carbon	µg/L	-	3,575	4,350	3,400	3,360	4,300	3,450
	µmol/L[c]	-	280	363	283	280	358	287
Mean Dissolved Organic Carbon	µg/L	-	4,640	2,350	1,733	5,440	2,900	1,450
	µmol/L[c]	-	387	196	144	453	241	121
Mean Total Nitrogen	µg/L	1,524	2,407	426	164	2,258	549	162
	µmol/L[c]	109	172	30	12	161	39	12
Mean Total Phosphorus	µg/L	3.2	2.0	4.0	2.3	3.0	4.0	3.5
	µmol/L[c]	0.10	0.06	0.13	0.07	0.10	0.13	0.11
Mean Total Dissolved Nitrogen	µg/L	1,318	1,920	344	174	1,966	566	105
	µmol/L[c]	94	137	25	12	140	40	7
Mean Total Dissolved Phosphorus	µg/L	-	<4[d]	<4[d]	<4[d]	<4[d]	<4[d]	<4[d]
	µmol/L[c]	-	<0.13[d]	<0.13[d]	<0.13[d]	<0.13[d]	<0.13[d]	<0.13[d]

a) 2004 information based on the annual open-water whole lake average data provided by the water quality component (De Beers 2005).

b) 2012 and 2013 information is based on surface water collected during the August programs at littoral zone sampling stations.

c) Molar concentrations were calculated by dividing the nutrient concentrations by their respective atomic values.

d) Detection limit for parameter.

°C = degrees Celsius; mg/L = milligrams per litre; µg/L = micrograms per litre; µmo/L = micromoles per litre; µS/cm = microSiemens per centimetre; - = data unavailable or not measured; < = less than.

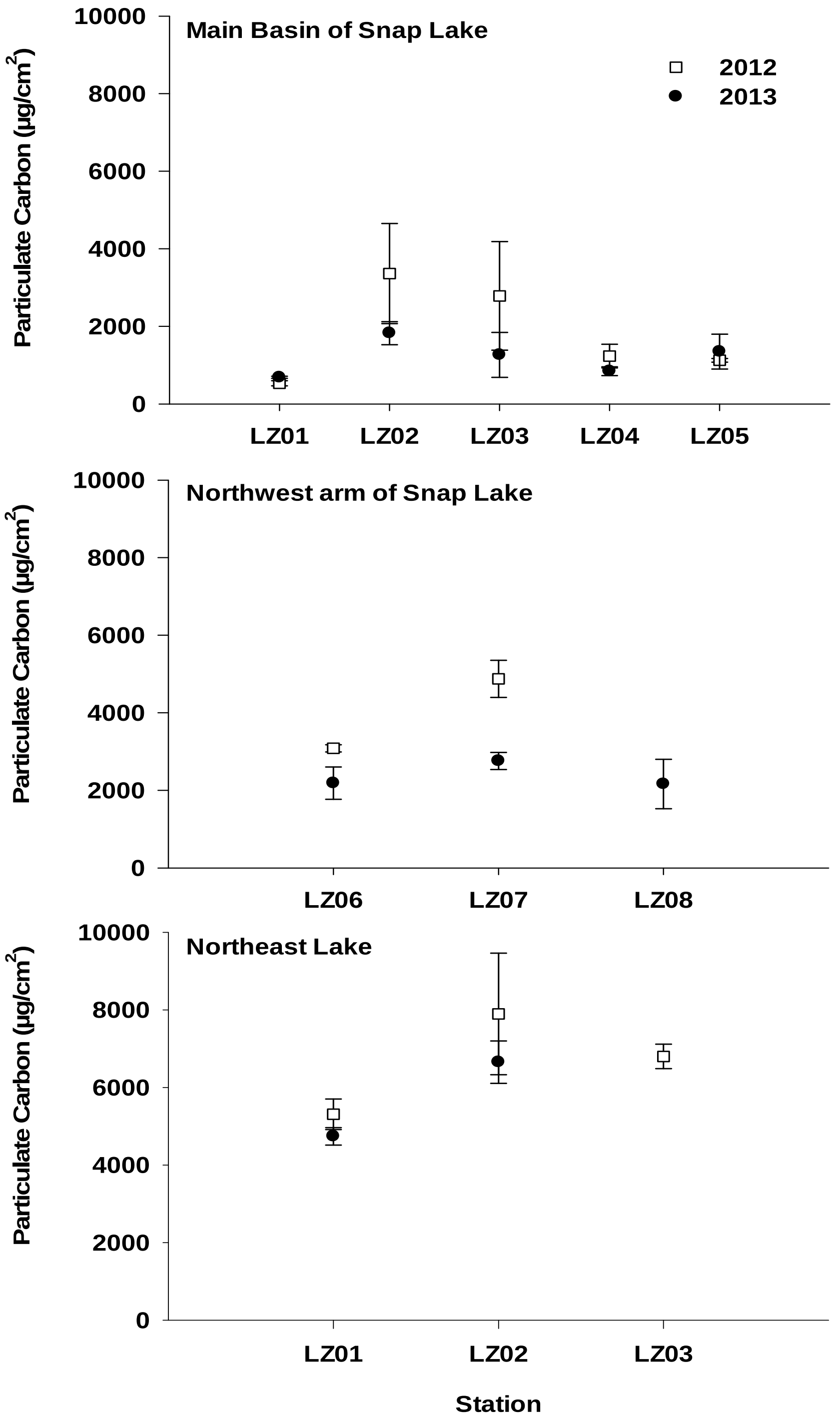
11.1.4.2 Nutrient Concentrations in the Epilithon

Mean epilithic particulate nutrients tended to be higher in Northeast Lake and the northwest arm of Snap Lake compared to the main basin (Figures 11.1-3 to 11.1-5). Epilithic particulate C concentrations (685 to 1,825 µg/cm^2) were lower in the main basin of Snap Lake compared to the northwest arm of Snap Lake (2,186 to 2,757 µg/cm^2) and Northeast Lake (4,740 to 6,652 µg/cm^2; Figure 11.1-3). The lowest epilithic particulate C concentrations were observed at SNAP LZ01 (685 µg/cm^2), the station closest to the diffuser, while the highest epilithic particulate C concentrations were observed at NEL LZ02 (6,652 µg/cm^2). Epilithic particulate C concentrations were similar between years, with the exception of SNAP LZ07, where concentrations were lower in 2013 compared to 2012.

Mean epilithic particulate P concentrations were lower and less variable in 2013 than in 2012 (Figure 11.1-4). Epilithic particulate P concentrations ranged from 1.8 to 3.0 µg/cm^2 in the main basin of Snap Lake, from 5.0 to 7.0 µg/cm^2 in the northwest arm of Snap Lake, and from 7.6 to 7.8 µg/cm^2 in Northeast Lake. Mean epilithic particulate N concentrations were lower and less variable in 2013 than in 2012 (Figure 11.1-5). Epilithic particulate N concentrations ranged from 63 to 174 µg/cm^2 in the main basin of Snap Lake, from 152 to 274 µg/cm^2 in the northwest arm of Snap Lake, and from 338 to 676 µg/cm^2 in Northeast Lake.

Spatial patterns were similar among the three epilithic particulate nutrients measured (Figures 11.1-3 to 11.1-5). Within-station variability was higher at SNAP LZ02 and SNAP LZ03 in the main basin of Snap Lake, SNAP LZ06 (except for C) in the northwest arm of Snap Lake, and NEL LZ02 in Northeast Lake. The high variability at NEL LZ02 was caused by higher mean particulate nutrient concentrations in sample NEL LZ02A compared to NEL LZ02B and NEL LZ02C (Appendix11.1C, Table 11.1C-3).

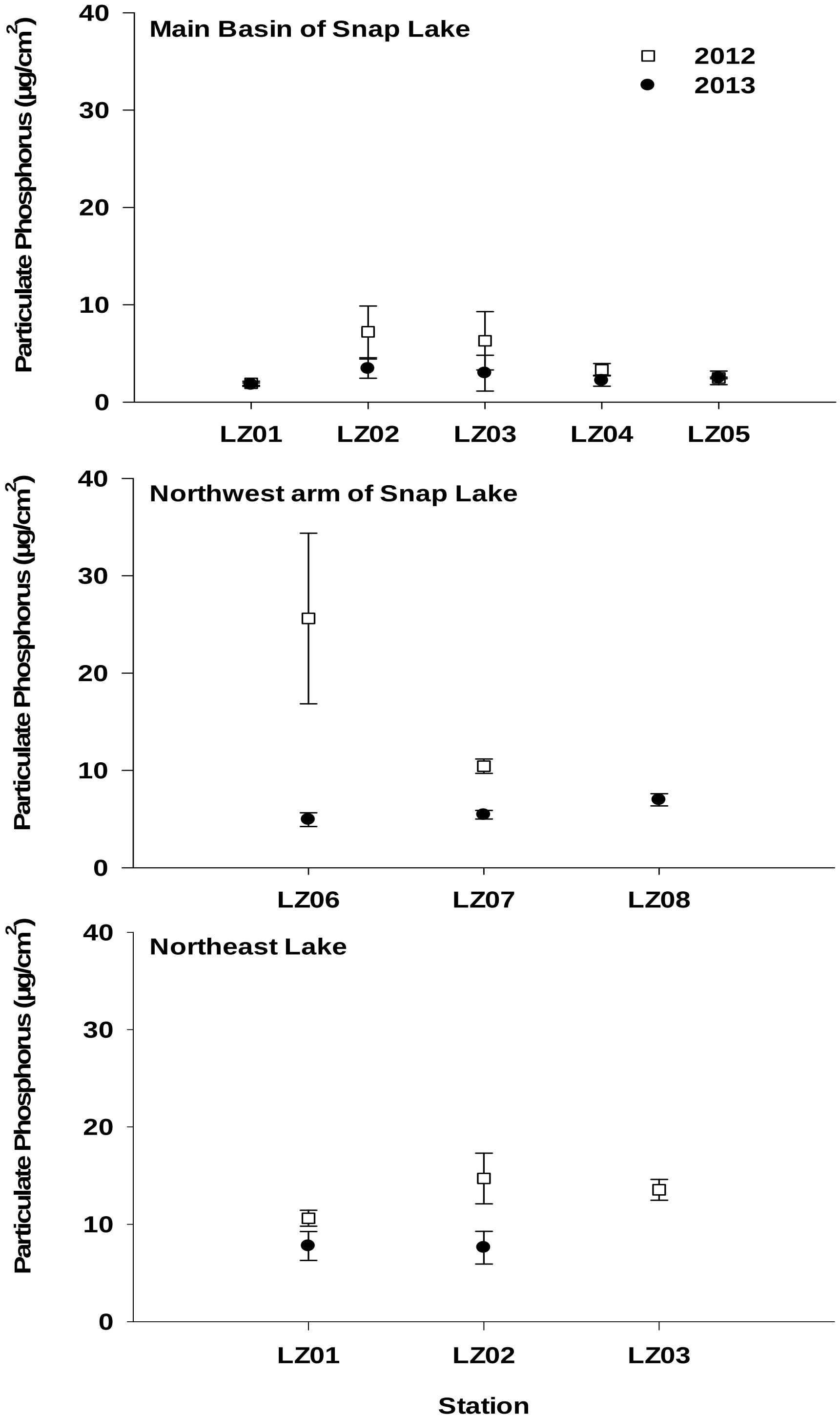
Figure 11.1-3 Spatial Trends in Epilithic Particulate Carbon in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the mean. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

µg/cm^2 = micrograms per square centimetre.

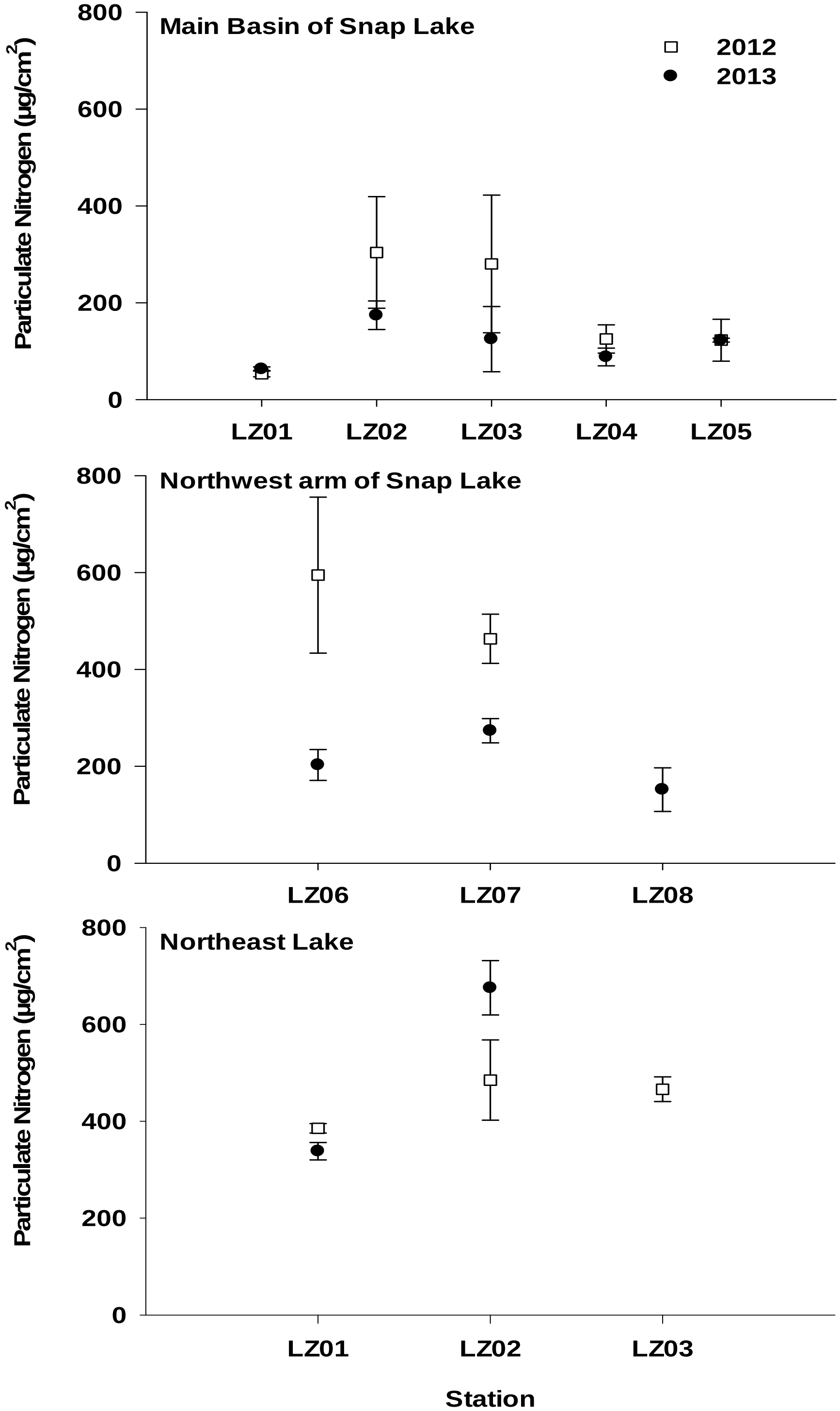
Figure 11.1-4 Spatial Trends in Epilithic Particulate Phosphorus in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the mean. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

µg/cm^2 = micrograms per square centimetre.

Figure 11.1-5 Spatial Trends in Epilithic Particulate Nitrogen in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the mean. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.
µg/cm^2 = micrograms per square centimetre.

11.1.4.3 Molar Ratios of Nutrients in Epilithic Algae

C:N ratio

The C:N molar ratios in Snap Lake and Northeast Lake did not show a clear spatial pattern in 2013. The mean C:N molar ratios in Snap Lake in 2013 were lower than in 2004, but generally similar and less variable compared to 2012 (Figure 11.1-6). The mean C:N molar ratios ranged from 15 to 17 in 2004, and from 11 to 16 in 2013. In 2012 and 2013, molar ratios tended to be higher in Northeast Lake compared to the main basin of Snap Lake; however, differences were observed between the two stations in Northeast Lake in 2013; the mean C:N ratio at NEL LZ01 (16) was higher than at NEL LZ02 (11).

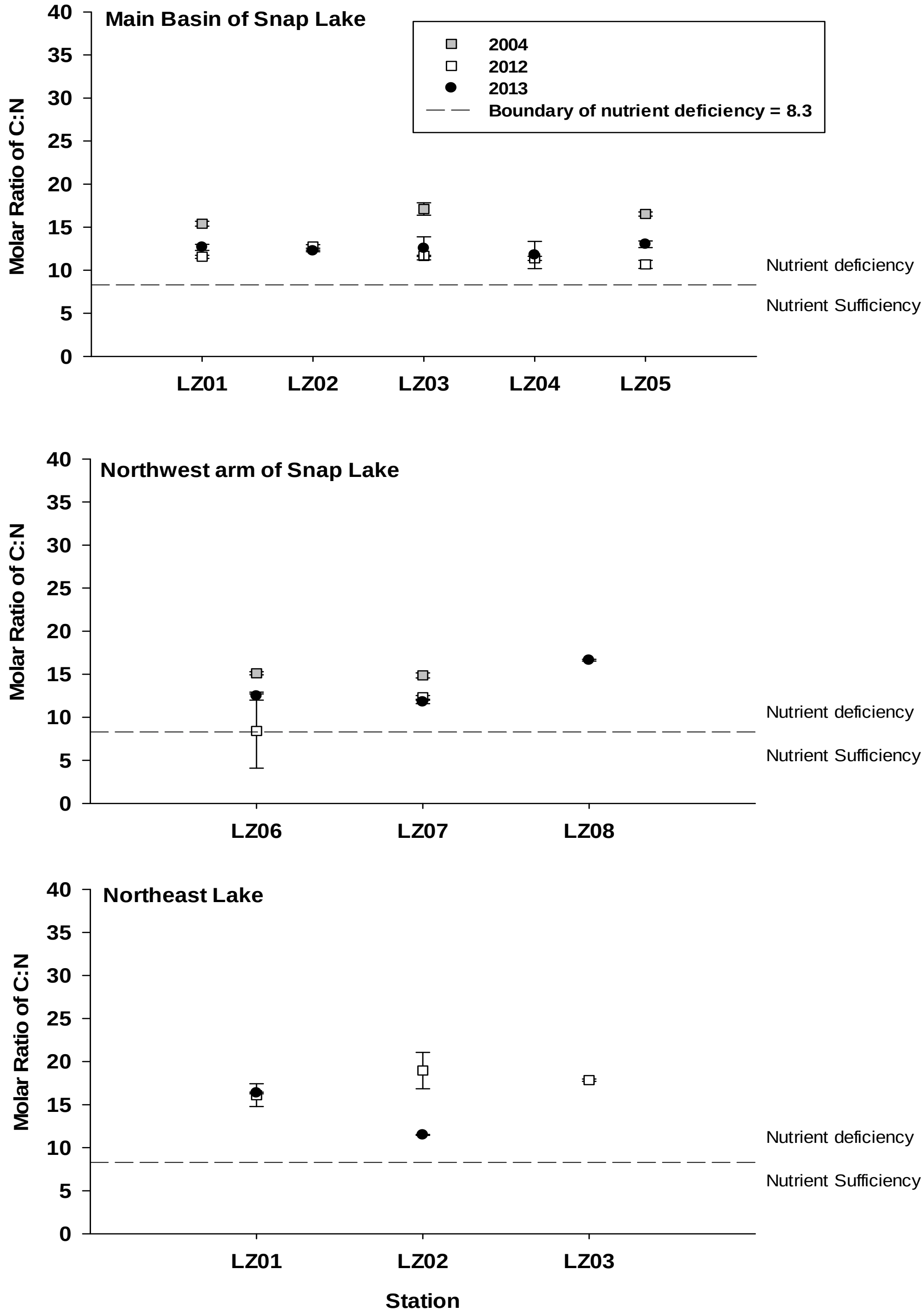
C:P Ratio

Mean C:P molar ratios in Northeast Lake (1,617 to 2,674) were higher than in the main basin (1,001 to 1,567) and northwest arm (818 to 1,315) of Snap Lake (Figure 11.1-7). Variability around each mean value was greater in Northeast Lake, particularly at NEL LZ02. With the exception of SNAP LZ02, C:P molar ratios were similar at all stations in the main basin of Snap Lake, ranging from 1,001 to 1,365. The mean C:P molar ratio at SNAP LZ02 (1,567) was higher and more variable than at all other stations in the main basin and northwest arm of Snap Lake.

N:P Ratio

Mean N:P molar ratios in 2013 tended to be higher in Northeast Lake (99 to 233) compared to the main basin (79 to 128) and northwest arm (49 to 112) of Snap Lake (Figure 11.1-18); however, the molar ratio at NEL LZ02 (233) was highly variable and more than twice as high as the molar ratio at NEL LZ01 (99). The lowest N:P molar ratio in the main basin was observed at SNAP LZ01 (79), while the highest was observed at SNAP LZ02 (128).

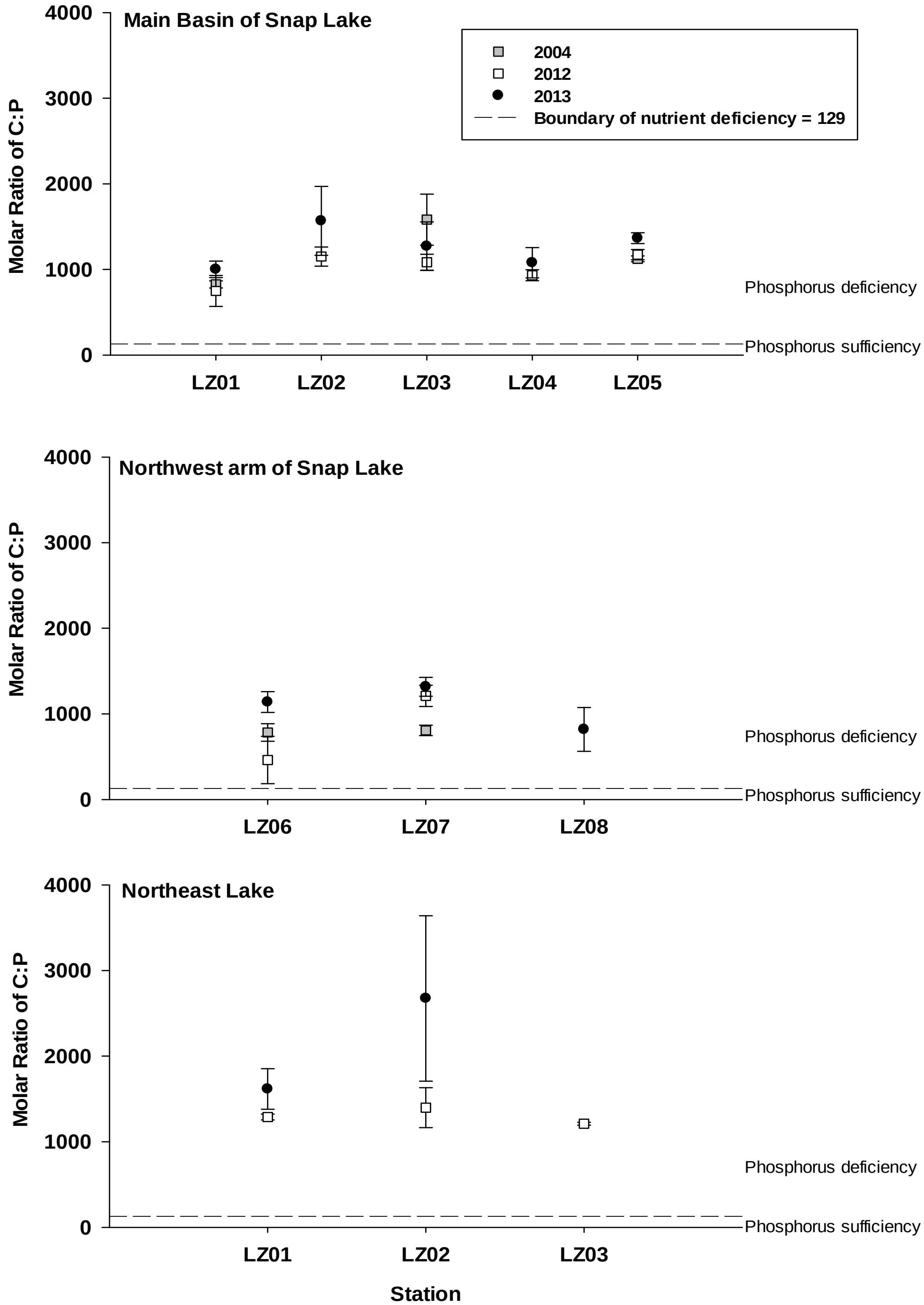
Figure 11.1-6 Spatial Trends in Epilithic Molar Ratios of Carbon to Nitrogen in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Error bars represent standard error of the mean; boundary of nutrient deficiency based on Healey and Hendzel (1980). SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

C= carbon; N = nitrogen.

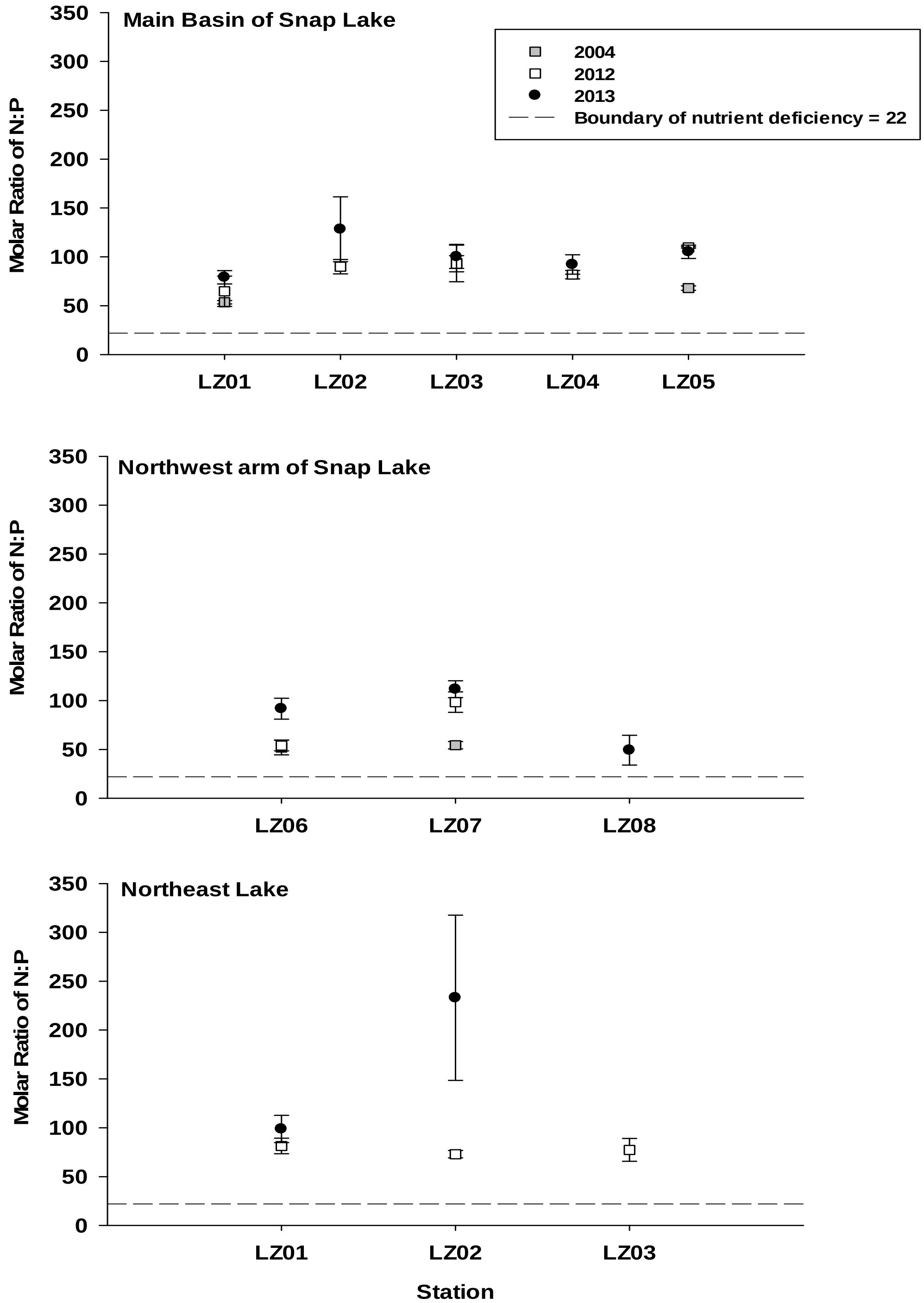
Figure 11.1-7 Spatial Trends in Epilithic Molar Ratios of Carbon to Phosphorus in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Error bars represent standard error of the mean; boundary of nutrient deficiency based on Healey and Hendzel (1980). SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

C= carbon; P = phosphorus.

Figure 11.1-8 Spatial Trends in Epilithic Molar Ratios of Nitrogen to Phosphorus in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Error bars represent standard error of the mean; boundary of nutrient deficiency based on Hillebrand and Sommer (1999). SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

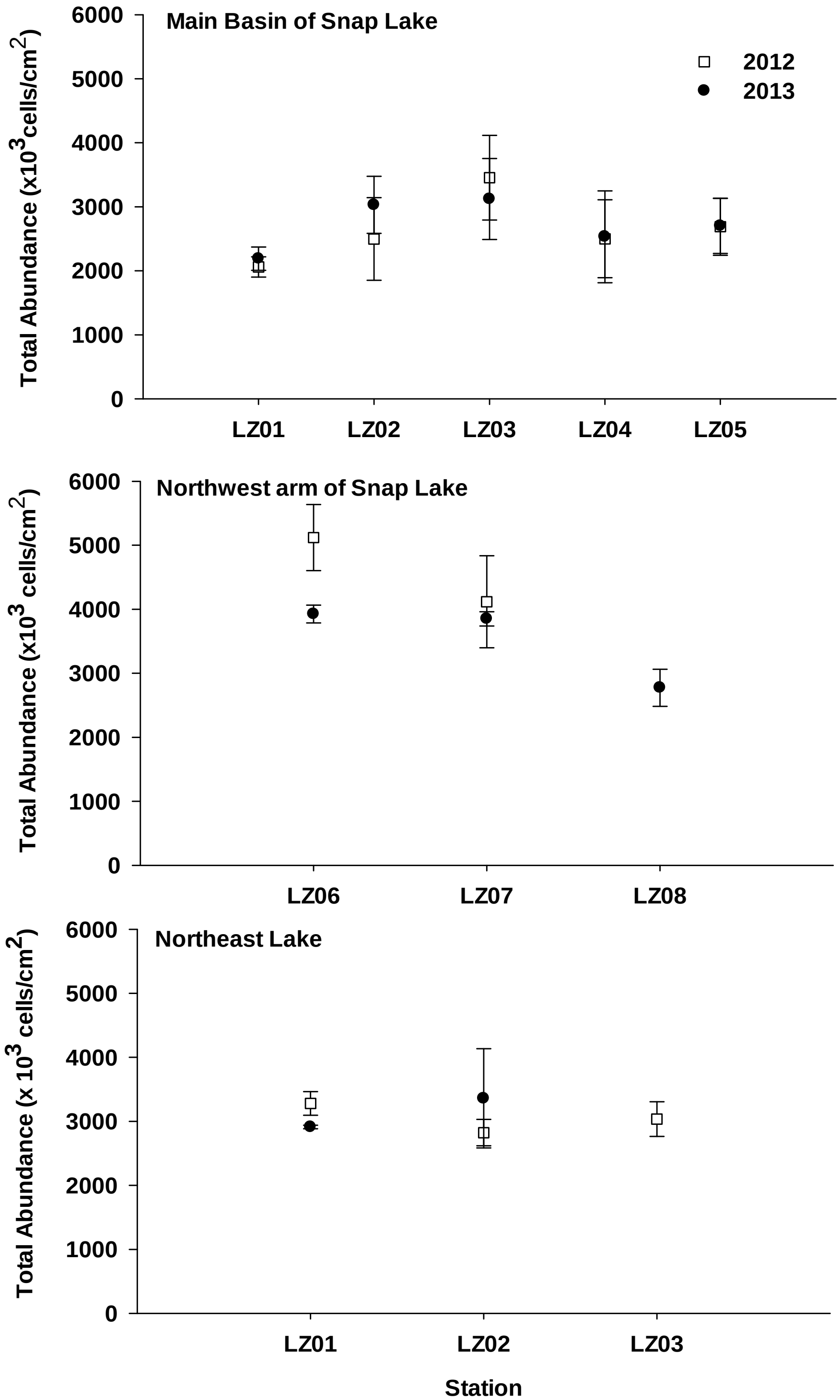
N = nitrogen; P = phosphorus.

11.1.4.4 Epilithic Algal Abundance and Biomass

Mean epilithic algal abundance in the main basin of Snap Lake in 2013 ranged from 2,189,205 to 3,925,017 cells per square centimetre (cells/cm^2; Figure 11.1-9). These values were similar to abundances observed in Northeast Lake, which ranged from 2,912,360 to 3,358,933 cells/cm^2. The lowest mean epilithic algal abundance was observed at SNAP LZ01 (2,189,205 cells/cm^2), and the highest abundance was observed at SNAP LZ06 (3,925,017 cells/cm^2). Overall, variability in Snap Lake tended to be high compared to Northeast Lake. Mean epilithic algal abundances in 2013 were generally similar, but less variable, than those observed in 2012. An exception to this occurred at SNAP LZ06, where mean epilithic algal abundances were higher in 2012 (5,120,107 cells/cm^2) compared to 2013 (3,925,017 cells/cm^2).

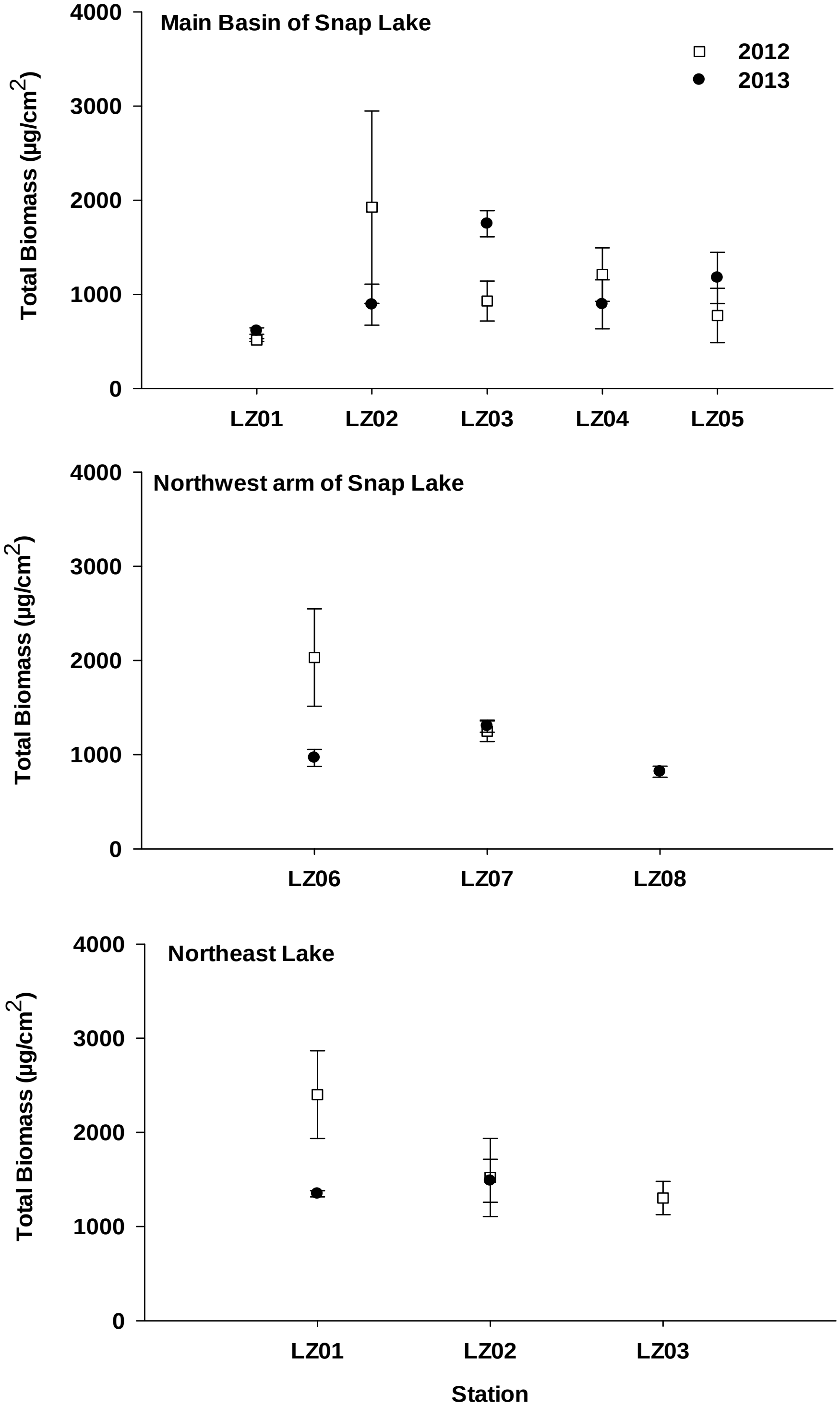
In 2013, mean epilithic algal biomass in the main basin was variable and ranged from 611 at SNAP LZ01 to 1,749 µg/cm^2 at SNAP LZ03 (Figure 11.1-10; Appendix 11.1C, Table 11.1C-4). With the exception of SNAP LZ03, biomass was lower in Snap Lake (611 to 1,302 µg/cm^2) compared to Northeast Lake (1,348 to 1,487 µg/cm^2). Biomass was generally similar among years, but decreased from 2012 to 2013 at SNAP LZ06 and NEL LZ01, and increased at SNAP LZ03.

Figure 11.1-9 Epilithic Algal Abundance in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the mean. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.
cells/cm^2 = cells per square centimetre; x = times.

Figure 11.1-10 Epilithic Algal Biomass in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the mean. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.
µg/cm² = micrograms per square centimetre.

11.1.4.5 Settled Volume of Particulate Material and Percent Viable Algae

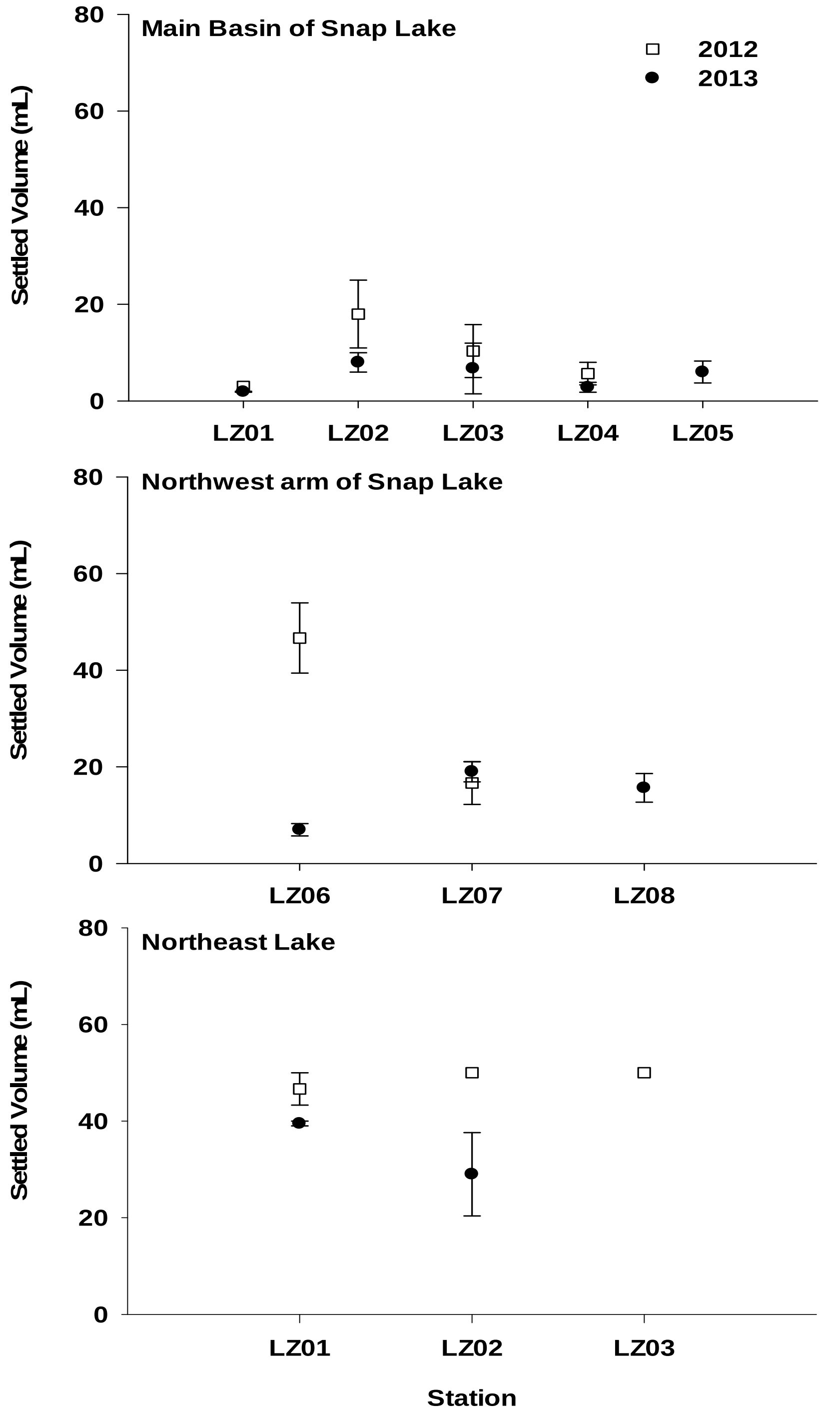
The settled volume of particulate material was lower in the main basin of Snap Lake (1.9 to 8.0 mL) compared to the northwest arm (7.0 to 15.7 mL) and Northeast Lake (29.0 to 39.5 mL; Figure 11.1-11). Similar to 2012, higher settled volumes of particulate material were observed at SNAP LZ02 (8.0 mL) and SNAP LZ03 (6.7 mL), compared to the other stations in the main basin, and the lowest settled volume of particulate material was observed at station SNAP LZ01 (1.9 mL), the station closest to the diffuser. Settled volumes of particulate material in the northwest arm of Snap Lake and Northeast Lake tended to decrease from 2012 to 2013, but remained similar in the main basin of Snap Lake. In addition, variation among stations in the northwest arm and Northeast Lake was high compared to stations in the main basin of Snap Lake.

Mean chlorophyll *a* concentration followed a similar pattern as settled volume in 2013, and was lower in the main basin (1.2 to 3.9 $\mu g/cm^2$) compared to the northwest arm (1.5 to 13.5 $\mu g/cm^2$) and Northeast Lake (15.1 to 16.3 $\mu g/cm^2$; Figure 11.1-12). The lowest mean chlorophyll *a* concentration was observed at SNAP LZ01, the station closest to the diffuser. Chlorophyll *a* concentrations were more variable in 2013 than in 2012, particularly in the northwest arm of Snap Lake and Northeast Lake. Chlorophyll *a* concentrations in Northeast Lake increased from 2012 to 2013, but generally remained similar in Snap Lake.

Northeast Lake and the main basin of Snap Lake had similar C:chlorophyll *a* ratios, with values ranging from 33 to 53 in the main basin, and from 28 to 50 in Northeast Lake (Figure 11.1-13). Ratios tended to be lower in the northwest arm of Snap Lake, ranging from 16 to 40. The highest mean C:chlorophyll *a* ratio was observed at SNAP LZ01, the station closest to the diffuser, while the lowest ratio was observed at SNAP LZ06. Ratios of C:chlorophyll *a* decreased, and became less variable, in the main basin and Northeast Lake from 2012 to 2013, but remained similar in the northwest arm. This decrease was most pronounced in Northeast Lake, where values in 2013 were up to five times lower in 2013 compared to 2012. Since 2004, the C:chlorophyll *a* ratios have decreased at all stations, except for SNAP LZ01, where C:chlorophyll *a* ratios have remained similar among all three years.

The mean percentage of viable algal carbon was similar between the northwest arm of Snap Lake and Northeast Lake, ranging from 1.2% to 2.3%. In contrast, values in the main basin of Snap Lake were higher and ranged from 2.4% to 8.4%. The highest, and most variable, viable algal carbon value was at SNAP LZ03 in the main basin of Snap Lake (8%). The percentage of viable algal carbon has increased in Snap Lake, particularly in the main basin, since 2004 (Figure 11.1-14). Values were in similar ranges from 2012 to 2013 in both lakes, with the exception of SNAP LZ03 and SNAP LZ07, where values were higher in 2013.

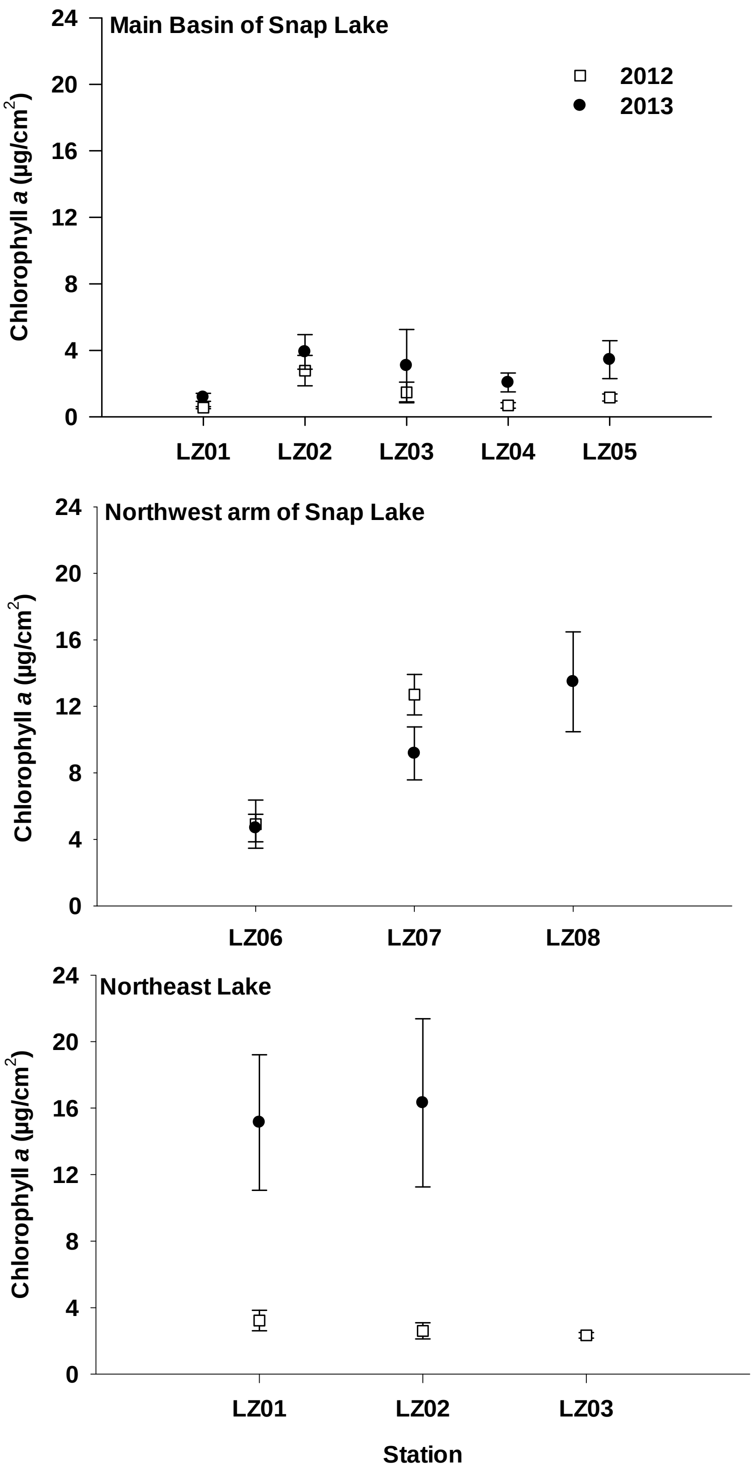
Figure 11.1-11 Settled Volume of Particulate Material in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the mean. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

mL = millilitre.

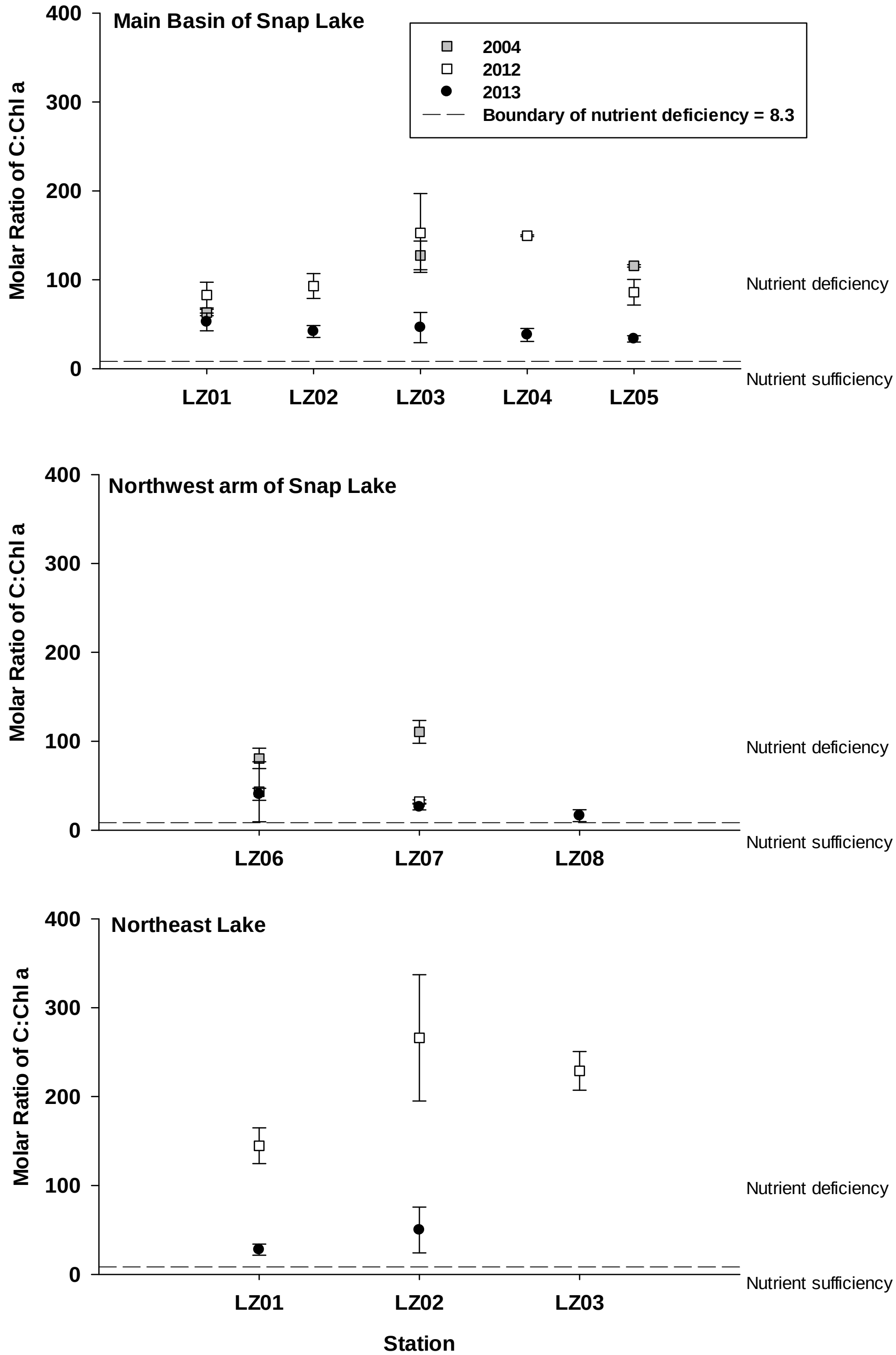
Figure 11.1-12 Spatial Trends in Epilithic Algae Chlorophyll *a* Concentrations, in Snap Lake and Northeast Lake, August 2012 and 2013



Note: Error bars represent standard error of the means. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

µg/cm^2 = micrograms per square centimetre.

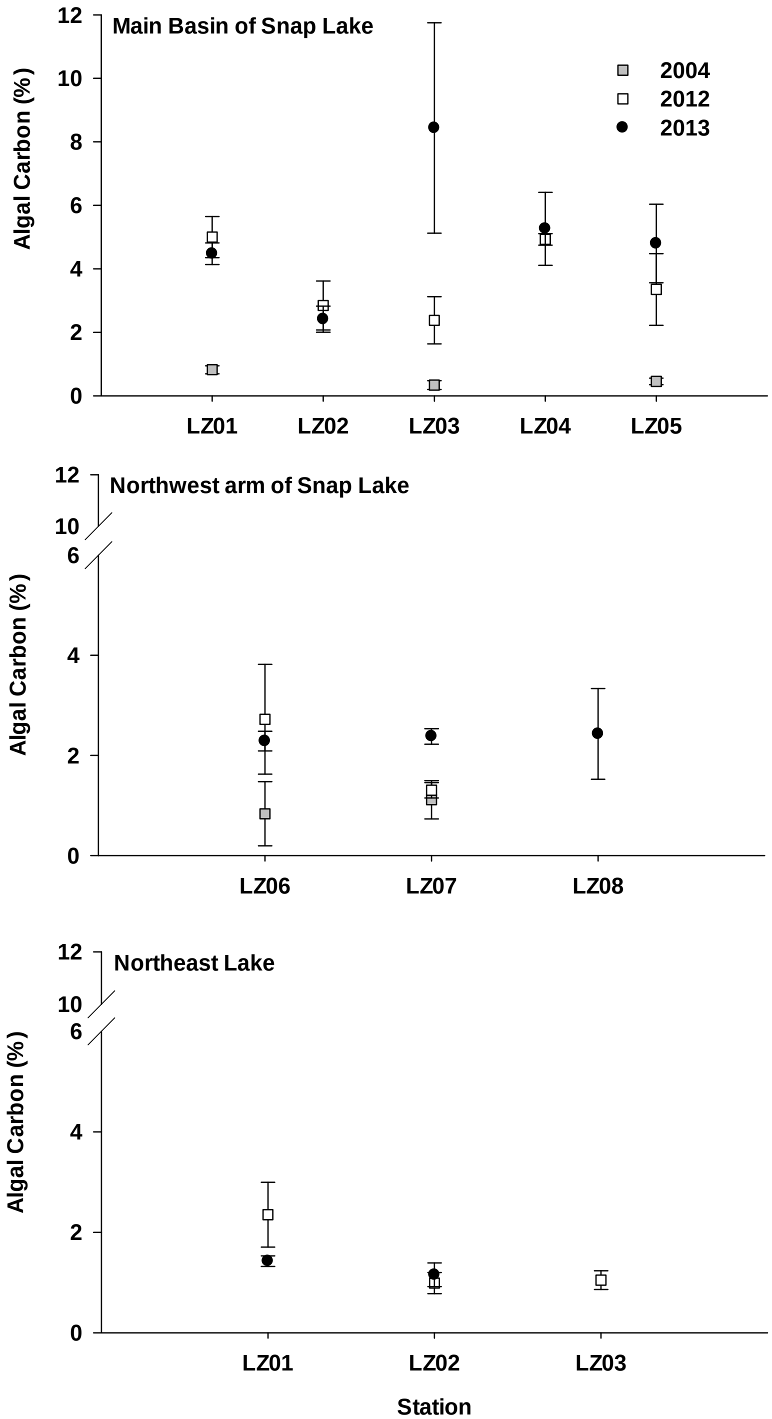
Figure 11.1-13 Spatial Trends in the Ratio of Epilithic Carbon to Chlorophyll *a* in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Error bars represent standard error of the means; boundary of nutrient deficiency based on Healey and Hendzel (1980). SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

C = carbon; Chl *a* = chlorophyll *a*.

Figure 11.1-14 Percentage of Epilithic Algal Carbon in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Error bars represent standard error of the means. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

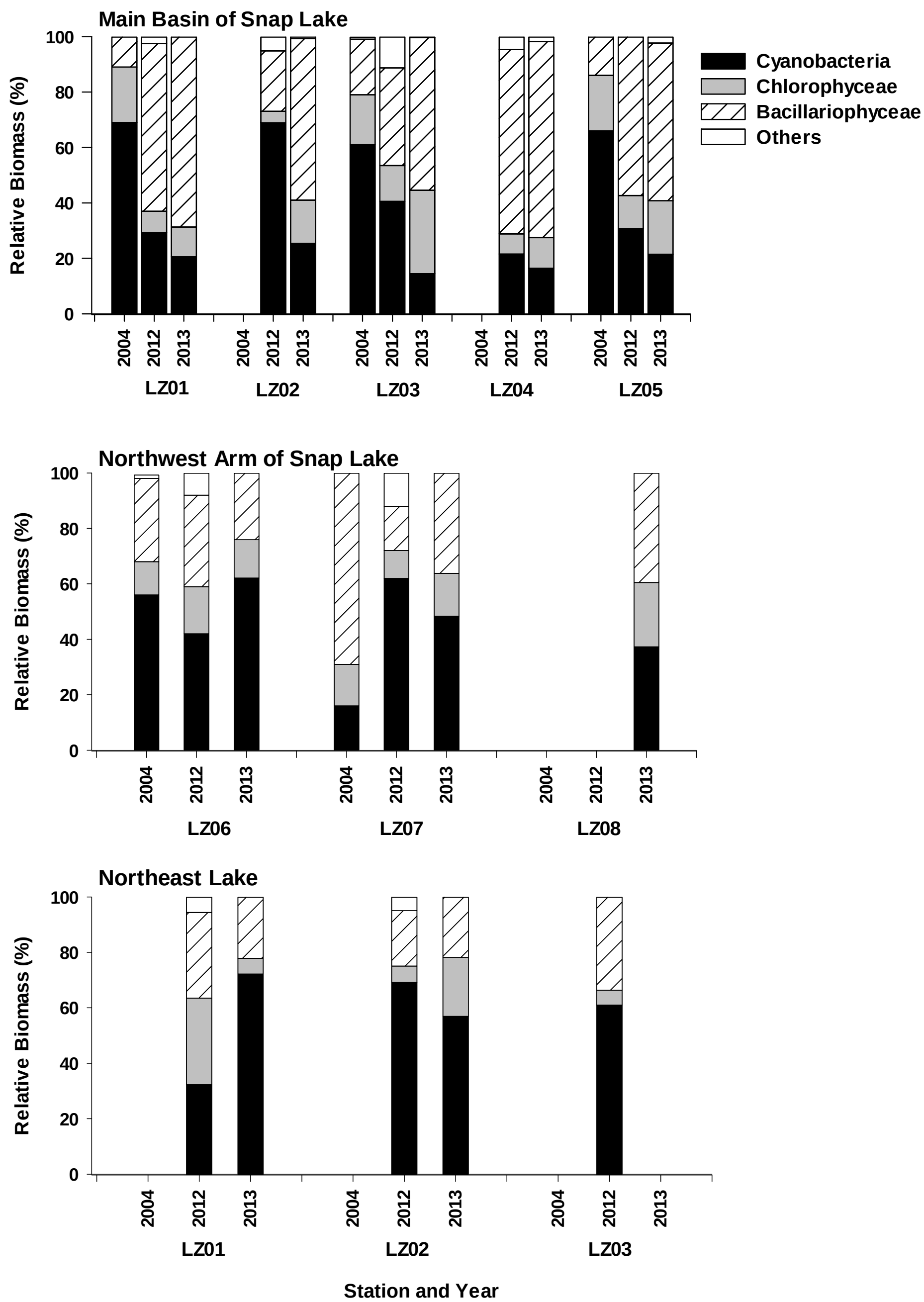
% = percent.

11.1.4.6 Epilithic Algal Community Composition

In 2004, epilithic algal abundance was dominated by either cyanobacteria or diatoms at all stations in Snap Lake (Figure 11.1-15). Cyanobacteria dominated the epilithic algae in both Snap Lake and Northeast Lake in 2012, but only the northwest arm of Snap Lake and Northeast Lake in 2013. In the main basin, diatoms increased in abundance from 2004 to 2013, and were the dominant group in 2013. The relative abundance of chlorophytes has been variable, but has generally increased at each station from 2004 to 2013, and has been similar among areas. “Others” have been rare and have accounted for less than 1% of the relative abundance in both lakes.

Relative epilithic algal biomass was dominated by cyanobacteria at all stations in 2004, with the exception of SNAP LZ07 in the northwest arm, where diatoms were the dominant group (Figure 11.1-16). Diatoms and chlorophytes were present in 2004, but at low relative biomass compared to cyanobacteria. In 2012 and 2013, the relative biomass of the major epilithic algal groups differed among sampling areas. In 2012, three stations in the main basin of Snap Lake (SNAP LZ01, SNAP LZ04, and SNAP LZ05) were diatom-dominated, while two stations (SNAP LZ02 and SNAP LZ03) were dominated by cyanobacteria. In contrast, all stations in the main basin of Snap Lake were diatom-dominated in 2013, with secondary dominance by cyanobacteria. Cyanobacteria remained the dominant group in Northeast Lake and the northwest arm of Snap Lake.

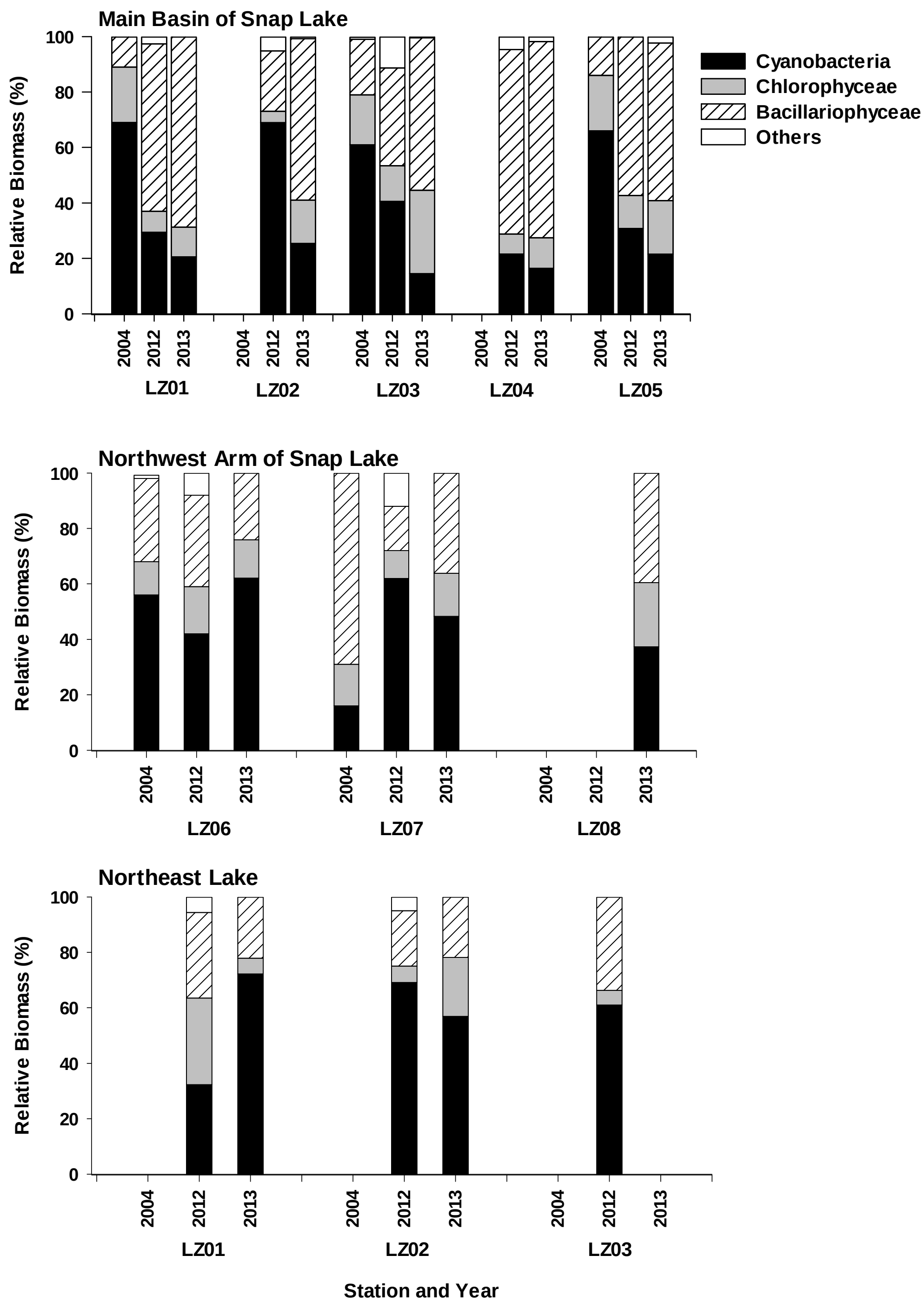
Figure 11.1-15 Relative Abundance of Major Epilithic Algal Groups in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Sampling did not occur in Northeast Lake in 2004. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

% = percent.

Figure 11.1-16 Relative Biomass of Major Epilithic Algal Groups in Snap Lake in 2004, 2012, and 2013, and Northeast Lake in 2012 and 2013



Note: Sampling did not occur in Northeast Lake in 2004. SNAP LZ08 was not sampled in 2012, and NEL LZ03 was not sampled in 2013 due to inclement weather.

% = percent.

11.1.4.7 Within-Station Variation in the Epilithon

Within-station variation was high at most stations in 2013 (Appendix 11.1C, Table 11.1C-7). 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) and 10 stations, 38 samples showed within-station variation greater than 20%. The greatest variation based on all measured parameters was observed at SNAP LZ03 in the main basin of Snap Lake, with CVs ranging from 11% to 100% (Appendix 11.1C, Table 11.1C-5). The least amount of variation was observed at SNAP LZ02, where CVs ranged from 25% to 50%. There were also lake-specific differences; the main basin of Snap Lake showed the highest variability (41%), while the northwest arm and Northeast Lake had the same mean CV value (23%).

11.1.4.8 Littoral Zone Invertebrate Community: Quantitative Assessment

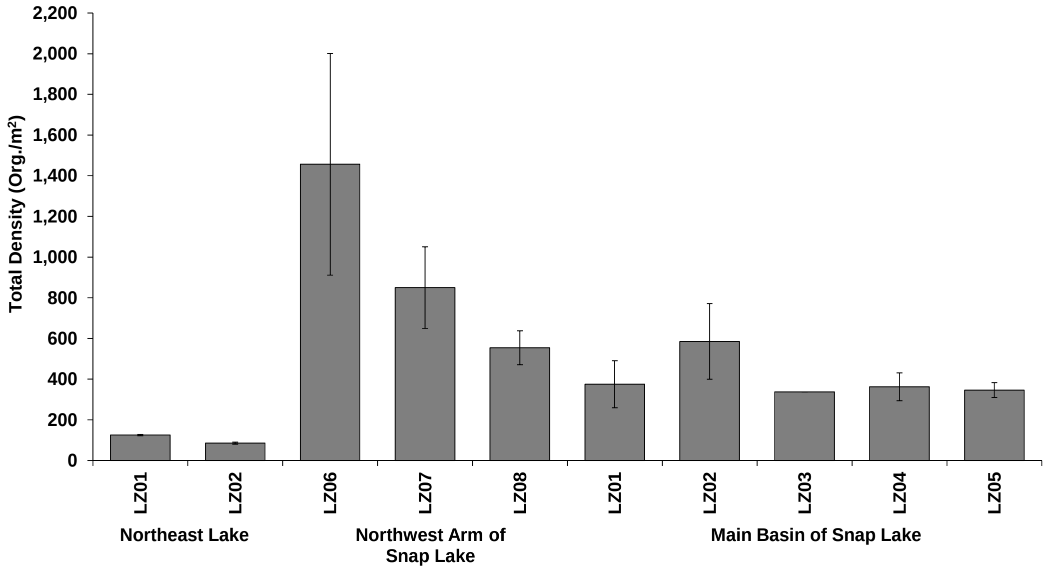
Mean littoral invertebrate density was variable within stations and among areas in Snap Lake, particularly in the northwest arm (Figure 11.1-17). Mean density in the northwest arm (554 to 1,456 org/m^2) and main basin (345 to 545 org/m^2) of Snap Lake was greater than in Northeast Lake (85 to 125 org/m^2). The variation in density among stations in the northwest arm of Snap Lake was influenced by a high density of Nadidae (aquatic worms) and *Tanytarsus* (midges) at SNAP LZ06, and a high density of Nadidae at SNAP LZ06 (Appendix 11.1C, Table 11.1C-6).

Richness ranged from low to moderate and varied within and among areas (Figure 11.1-18). Total richness ranged from 9 to 27 taxa per station in the main basin of Snap Lake, from 21 to 27 taxa per station in the northwest arm of Snap Lake, and from 11 to 14 taxa per station in Northeast Lake. With the exception of SNAP LZ03, total richness in Snap Lake was greater than in Northeast Lake. Overall, taxa richness was in the expected range for subarctic lakes (Beaty et al. 2006).

Simpson’s index of diversity was generally high, ranging from 0.80 to 0.91 in the main basin of Snap Lake, from 0.83 to 0.91 in the northwest arm of Snap Lake, and from 0.78 to 0.84 in Northeast Lake (Figure 11.1-19). Evenness was variable and ranged from low to moderate in the areas sampled: 0.38 to 0.65 in the main basin of Snap Lake, 0.28 to 0.41 in the northwest arm of Snap Lake, and 0.32 to 0.59 in Northeast Lake (Figure 11.1-20). Simpson’s index of diversity and the evenness data indicate that a few taxa usually accounted for most of the total density observed at each station.

The common invertebrate taxa observed in the littoral zones of Snap Lake and Northeast Lake were Oligochaeta (aquatic worms), Hydracarina (aquatic mites), Ostracoda (ostracods), Gastropoda (snails), Ephemeroptera (mayflies), and Diptera (true flies, all in the Chironomidae, or midge family; Appendix 11.1C, Table 11.1C-6). At the lowest taxonomic level of identification, 48 taxa were identified in Snap Lake and 17 taxa in Northeast Lake. The most common taxa in Snap Lake were *Gyraulus* sp., *Valvata* sp., *Endochironomus* sp., *Tanytarsus* sp., and *Corynoneura* sp. The most common taxa in Northeast Lake were *Dicrotendipes* sp., *Tanytarsus* sp., *Psectrocladius* sp., *Micropsectra*/*Tanytarsus* sp., *Mystacides* sp., and *Thienemannimyia* sp. (Appendix 11.1C, Table 11.1C-6).

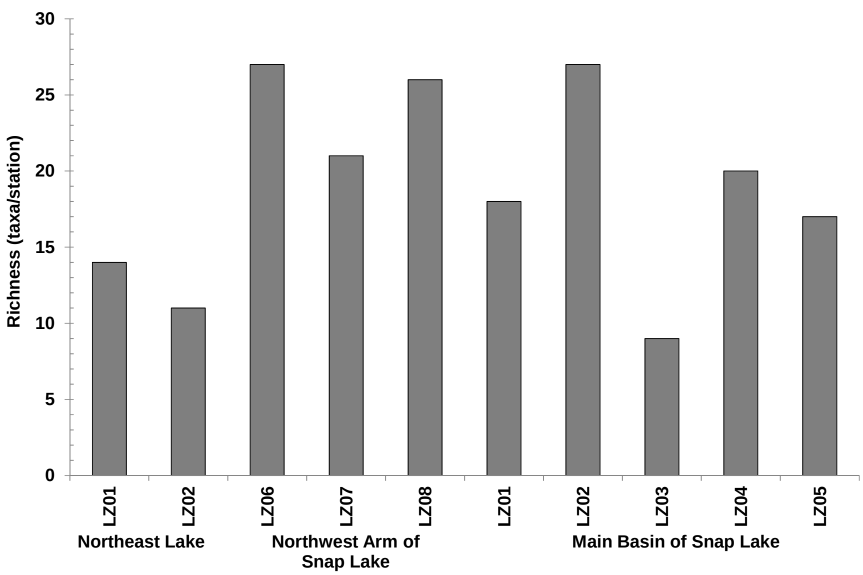
Figure 11.1-17 Mean Littoral Invertebrate Density, Collected by Hester-Dendy Samplers, in Snap Lake and Northeast Lake, August 2013



Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. Error bars represent standard error of the means. Only one of three Hester-Dendy samplers was collected at SNAP LZ03 (two samplers were found above the water line). Hester-Dendy samplers were deployed, but not collected, at NEL LZ03, NEL LZ04, and NEL LZ05 due to inclement weather.

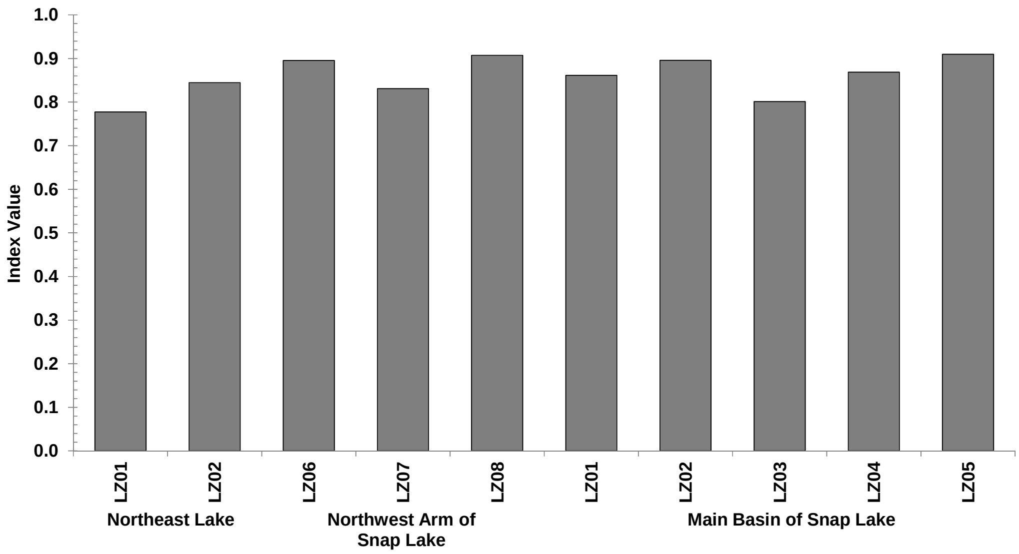
org/m^2 = number of organisms per square metre.

Figure 11.1-18 Littoral Invertebrate Species Richness, Collected by Hester-Dendy Samplers in Snap Lake, and Northeast Lake, August 2013



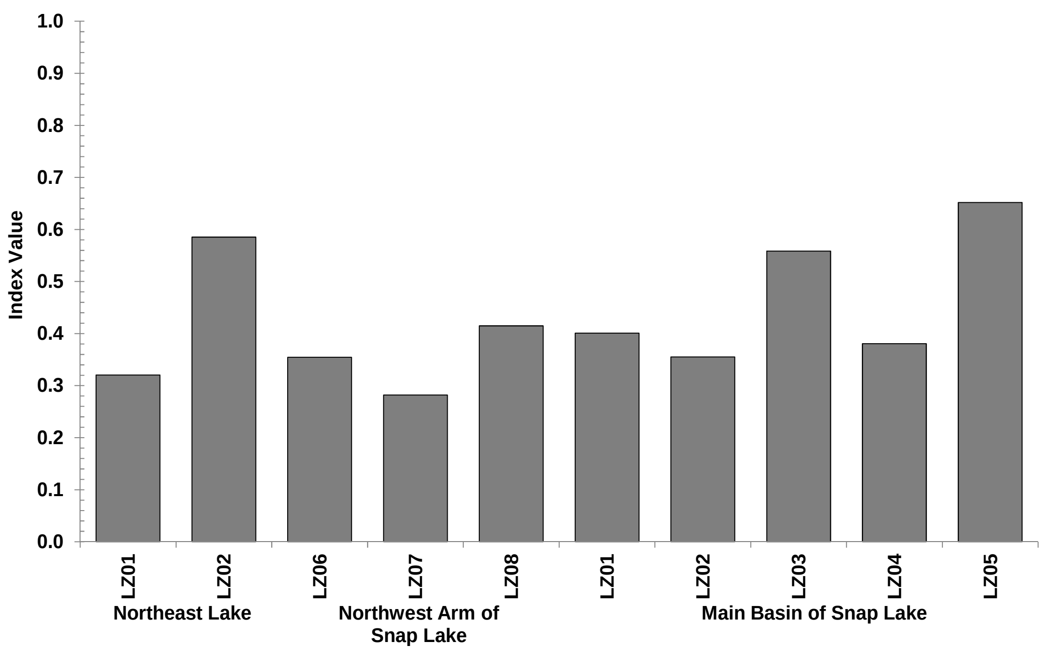
Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. Only one of three Hester-Dendy samplers was collected at SNAP LZ03 (two samplers were found above the water line). Hester-Dendy samplers were deployed, but not collected, at NEL LZ03, NEL LZ04, and NEL LZ05 due to inclement weather.
taxa/station = number of taxa per station.

Figure 11.1-19 Simpson's Index of Diversity of Littoral Invertebrates, Collected by Hester-Dendy Samplers in Snap Lake and Northeast Lake, 2013



Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. Only one of three Hester-Dendy samplers was collected at SNAP LZ03 (two samplers were found above the water line). Hester-Dendy samplers were deployed, but not collected, at NEL LZ03, NEL LZ04, and NEL LZ05 due to inclement weather.

Figure 11.1-20 Evenness of Littoral Invertebrate Communities, Collected by Hester-Dendy Samplers in Snap Lake and Northeast Lake, 2013



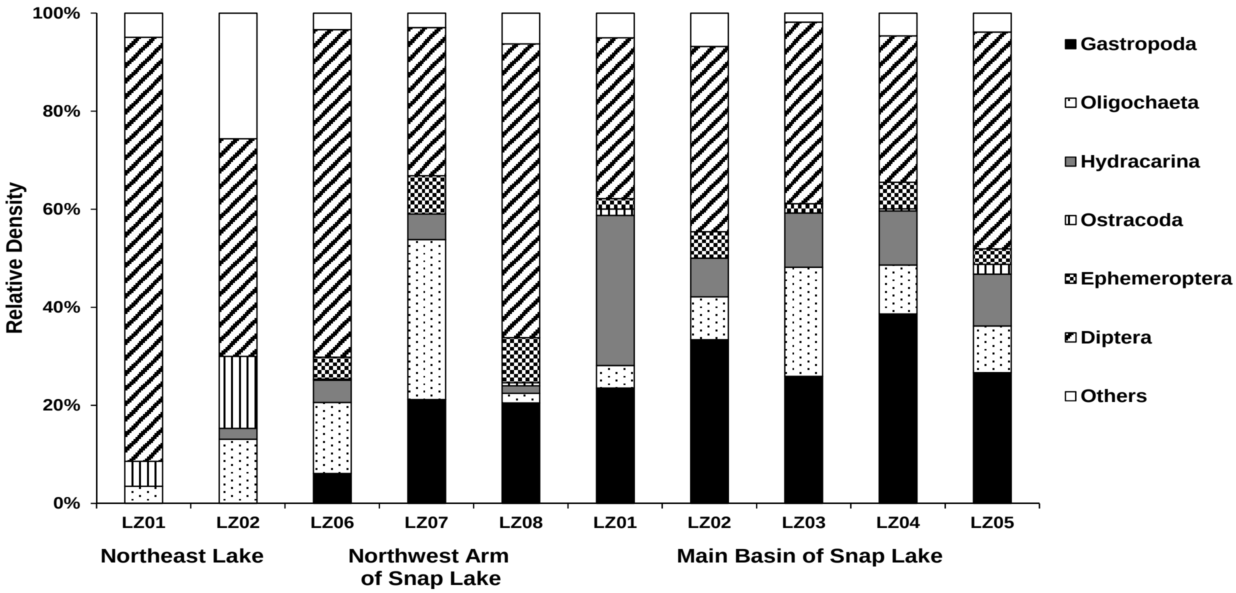
Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. Only one of three Hester-Dendy samplers was collected at SNAP LZ03 (two samplers were found above the water line). Hester-Dendy samplers were deployed, but not collected, at NEL LZ03, NEL LZ04, and NEL LZ05 due to inclement weather.

Diptera (mostly midges) dominated the littoral invertebrate communities in both Snap Lake and Northeast Lake in 2013. Littoral invertebrate communities in the main basin of Snap Lake were more evenly distributed among groups compared to the northwest arm of Snap Lake and Northeast Lake (Figure 11.1-21). Higher relative densities of Hydracarina and Gastropoda were observed in the main basin of Snap Lake compared to other areas, with the highest Hydracarina relative density (31%) observed at SNAP LZ01, the station closest to the diffuser. Gastropods, Ephemeroptera, Oligochaeta, and Hydracarina were absent, or present only at low densities, in Northeast Lake. In contrast, ostracods were primarily found in Northeast Lake, and were in low densities (less than 2%) in Snap Lake. A large proportion of “Others” (26%) was observed at NEL LZ02, due to a large density of a Trichopteran (caddisfly); *Mystacides* sp. (Appendix 11.1C, Table 11.1C-6).

The relative densities of Chironomidae taxa were highly variable in Snap Lake, particularly in the main basin (Figure 11.1-22). The stations closest to the diffuser, SNAP LZ01 and LZ02, contained relatively fewer Orthocladiinae, but more Tanytarsini, compared to other stations in the main basin of Snap Lake. Stations SNAP LZ03 and SNAP LZ04 were dominated by Orthocladiinae, while SNAP LZ05 was dominated by Chironomini. The main basin differed from the northwest arm by having fewer Chironomini and more Tanytarsini or Orthocladiinae, depending on the station. Communities in Northeast Lake were dominated by Orthocladiinae. The relative density of Chironomini was lower in Northeast Lake compared to Snap Lake.

The distribution of littoral invertebrates into functional feeding groups indicated that the community of Snap Lake was relatively balanced and diverse, particularly in the main basin (Figure 11.1-23). Communities in the main basin consisted mostly of collector-gatherers, scrapers, shredders and predators, although other groups were represented. Communities in the northwest arm of Snap Lake consisted mostly of collector-gatherers, shredders, and scrapers. Collector-gatherers were more abundant, and parasites and deposit-suspension feeders were less abundant in the northwest arm compared to the main basin of Snap Lake. The highest percentage of parasites and predators, and the lowest percentage of shredders were observed at SNAP LZ01, the station closest to the diffuser. The Northeast Lake community was dominated by collector-gatherers and shredders, although other groups were also present (Figure 11.1-23). Herbivores were present at higher relative densities in Northeast Lake compared to Snap Lake.

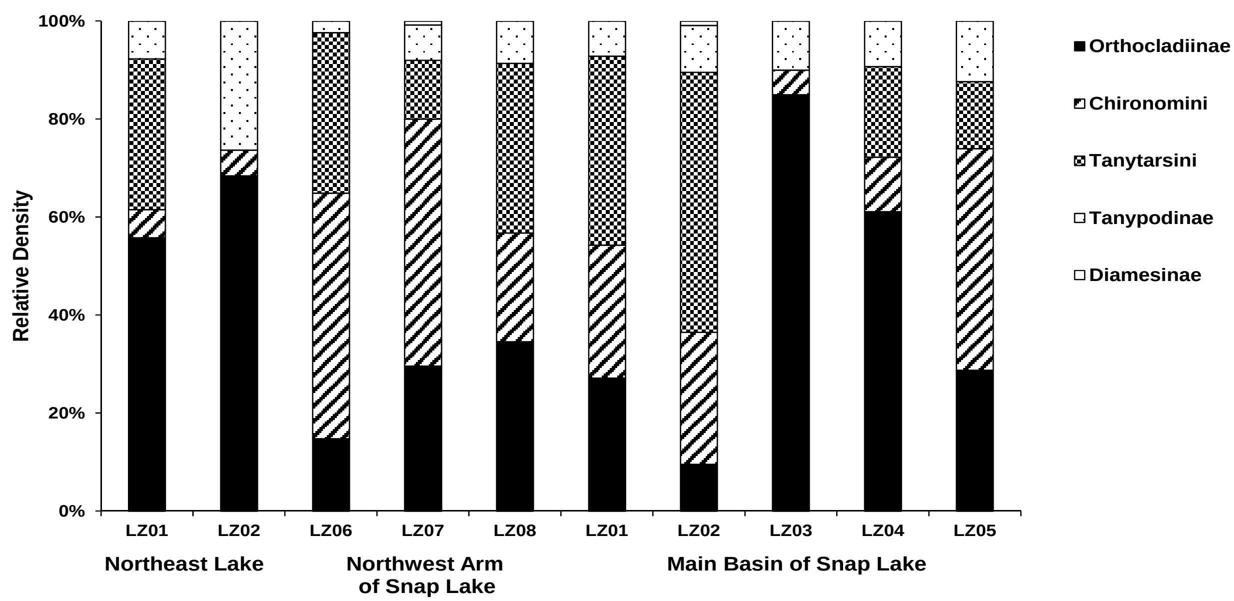
Figure 11.1-21 Relative Densities of Major Littoral Invertebrate Groups, Collected by Hester-Dendy Samplers in Snap Lake and Northeast Lake, August 2013



Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. NEL LZ03 was not sampled in 2013 due to inclement weather.

% = percent.

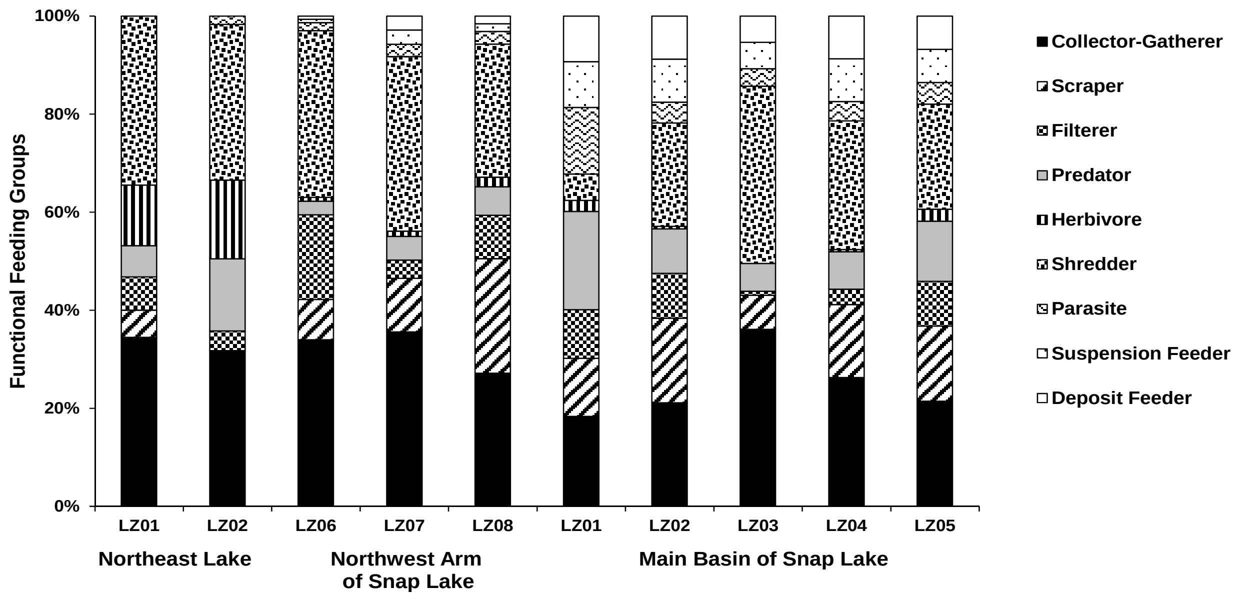
Figure 11.1-22 Relative Densities of Diptera Taxa, Collected by Hester-Dendy Samplers in Snap Lake and Northeast Lake, August 2013



Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. NEL LZ03 was not sampled in 2013 due to inclement weather.

% = percent.

Figure 11.1-23 Relative Density of Invertebrate Functional Feeding Groups in Hester-Dendy Samples Collected in Snap Lake and Northeast Lake, August 2013



Note: Stations are arranged from furthest from the diffuser to closest to the diffuser in both directions from the main basin of Snap Lake LZ01 station. NEL LZ03 was not sampled in 2013 due to inclement weather.

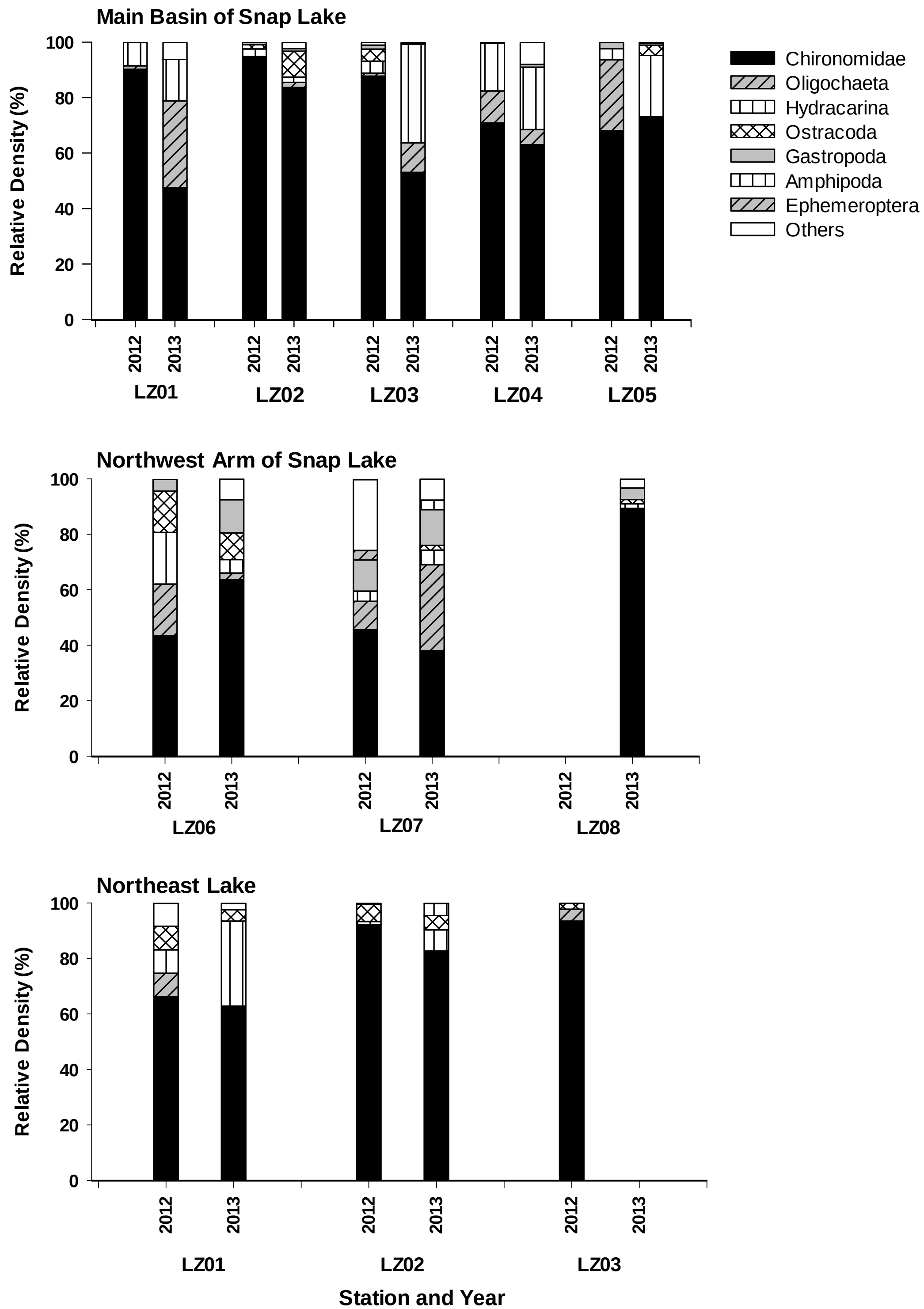
% = percent.

11.1.4.9 Littoral Zone Invertebrate Community: Qualitative Assessment

Littoral zone invertebrate community composition in sweep net samples was variable between years and among stations, but communities were generally dominated by Diptera (mostly midges) in both Snap Lake and Northeast Lake (Figure 11.1-24; Appendix 11.1C, Table 11.1C-7). Diptera were relatively less abundant in the samples in the northwest arm of Snap Lake compared to the main basin and Northeast Lake. The relative densities of less dominant taxa were highly variable among stations and years. The relative density of Hydracarina increased in the main basin of Snap Lake and Northeast Lake between 2012 and 2013. Ephemeroptera were found at one station (SNAP LZ07) in 2013, but at low relative densities (3.4%).

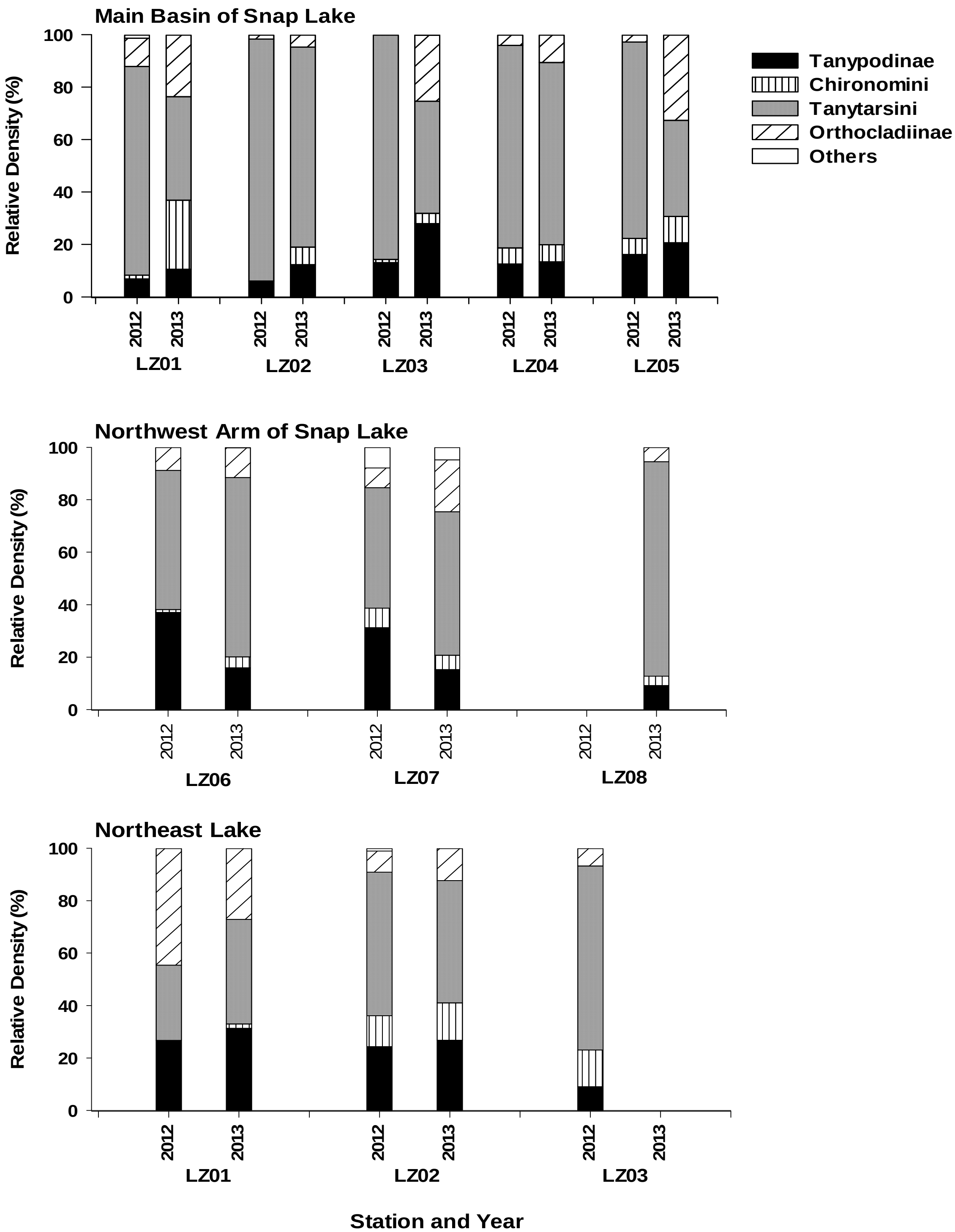
Tanytarsini were the most abundant Diptera taxa at most stations in 2012 and 2013 (Figure 11.1-25). Chironomini relative densities at SNAP LZ01, the station closest to the diffuser, was higher in 2013 compared to 2012. Differences were also noted in the relative densities of Diptera taxa in the main basin: Orthocladiinae, Tanypodinae, and Chironomini were more abundant, and Tanytarsini were less abundant in 2013. Differences among years in the northwest arm and Northeast Lake were less pronounced.

Figure 11.1-24 Relative Densities of Major Littoral Invertebrate Groups, Collected Using a Sweep Net in Snap Lake and Northeast Lake, 2012 and 2013



Note: SNAP LZ08 was not sampled in 2012 and station NEL LZ03 was not sampled in 2013 due to inclement weather.
% = percent.

Figure 11.1-25 Relative Densities of Littoral Diptera taxa, Collected Using a Sweep Net, in Snap Lake and Northeast Lake, 2012 and 2013



Note: SNAP LZ08 was not sampled in 2012 and station NEL LZ03 was not sampled in 2013 due to inclement weather.

% = percent.

11.1.5 Discussion

11.1.5.1 Littoral Zone Nutrients

The DIC concentrations observed in Snap Lake and Northeast Lake in 2012 and 2013 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 could also be the case in Snap Lake and Northeast Lake, as the littoral zone nutrient ratio data indicated nutrient deficiency, particularly in P. The C:N, C:P, N:P, and C:chlorophyll *a* molar ratios in Snap Lake and Northeast Lake indicated nutrient deficiency in 2012 and 2013, and the C:P and N:P nutrient molar ratios indicated that the limiting nutrient is P.

The molar ratios of C:N and C:P in 2013 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 ratios in the main basin of Snap Lake remained similar throughout the study, while those in the northwest arm of Snap Lake and Northeast Lake increased. This could indicate both a decrease in P-deficiency and an increase in food quality in Snap Lake, particularly in the main basin. The discrepancy observed between Snap Lake and Northeast Lake from 2004 to 2013 could be an indication of Mine-related P additions to the Snap Lake main basin, as opposed to the northwest arm, but further observations of Northeast Lake are needed to verify this. The C:N molar ratios in 2012 and 2013 were lower at all stations in Snap Lake compared to 2004, indicating a change towards greater nutrient sufficiency. This could 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 and 2013 compared to 2004, could be an indication of a Mine-related nutrient enrichment effect in Snap Lake.

11.1.5.2 Epilithic Algae and Associated Bacteria and Detritus

The settled volume of particulate material at each station can be used as a measure of the amount of material, both biotic (living and dead) and abiotic on rock surfaces. 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 of Snap Lake. The lowest settled volumes were measured at SNAP LZ01, the station in the main basin closest to the diffuser.

Increases in mean chlorophyll *a* concentrations were observed in Northeast Lake, but not Snap Lake, from 2012 to 2013. The increase in chlorophyll *a* was not accompanied by an increase in the percentage of algal carbon or epilithic algal biomass. However, the C:chlorophyll *a* ratio did demonstrate a decrease from 2012 to 2013 in the main basin of Snap Lake and Northeast Lake, indicating that the epilithic algae contained proportionally more chlorophyll *a* in these areas in 2013 compared to 2012 (i.e., they were in better condition in 2013).

The percentage algal C observed at all stations in Snap Lake in 2004, in the northwest arm of Snap Lake in 2004, 2012, and 2013, and in Northeast Lake in 2012 and 2013 were similar to values observed in undisturbed lakes at the ELA (i.e., 1% to 5%; Hille 2008). Values observed in the main basin of Snap Lake in 2012 and 2013 (i.e., values greater than 5%) were similar to those observed in a system receiving P-loading at the ELA (Hille 2008). The higher proportion of viable algal C in 2012 and 2013 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. This could also be an indication of a Mine-related nutrient enrichment effect in Snap Lake, but more data are needed from Northeast Lake to confirm or refute this possibility.

Epilithic algal community composition differed among years 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 and 2013. In 2004, relative abundance and biomass were dominated by either cyanobacteria or diatoms at all stations in Snap Lake. In 2012 and 2013, stations in the main basin of Snap Lake were generally diatom-dominated with secondary dominance by cyanobacteria. Stations in the northwest arm of Snap Lake and Northeast Lake were dominated by cyanobacteria. Overall, conditions in the main basin of Snap Lake appear to be favouring diatoms over cyanobacteria.

With the observed increase in N-loading and the 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 2013, relative percent cyanobacteria biomass decreased in the main basin of Snap Lake, while the relative percent biomass of chlorophytes and diatoms increased. The increase in diatom biomass throughout the study 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 cyanobacteria biomass in the main basin of Snap Lake can also be linked to the observed increases in food quality from 2004 to 2013 (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 2012).

11.1.5.3 Within-Station and Among-Lake Variation in the Epilithin

It is important to understand the degree of station-to-station variation when interpreting changes in epilithic algal biomass and composition. A low (less than 20%) CV, 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 CVin nutrient concentrations and biomass could translate to inaccurate interpretation of effects, and prevent meaningful statistical analyses.

In 2013, five stations were sampled in the main basin of Snap Lake, three stations in the northwest arm of Snap Lake, and two stations in Northeast Lake. Within each of these stations, three sub-areas were sampled to examine within-station variation. Within-station variation was high at most stations in 2013. Improved sampling techniques used in 2013 indicated that the variation could be due to the high natural variability that is associated with epilithon and the littoral zone in general.

Natural epilithic algae, along with their associated bacteria and detritus, are inherently variable and include a combination of extremely diverse microhabitats, which vary in developmental stage (Robinson 1983) and composition among stations. Variation can also be caused by changes in the influence 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).

11.1.5.4 Littoral Zone Invertebrates

Snap Lake supported a higher total density and richness of littoral zone invertebrates compared to Northeast Lake, but Simpson's diversity was high and evenness was low to moderate in both lakes, indicating that a few taxa accounted for most of the organisms present.

Diptera (all in the Chironomidae family) were the dominant taxa in Snap Lake and Northeast Lake which is similar to what was observed in the deep-water benthic invertebrate community (De Beers 2014b). Dominance of benthic invertebrate communities by the Chironomidae family is expected in the subarctic region, where Snap Lake is located (Beaty et al. 2006; Northington et al. 2010). Communities in Snap Lake were more diverse and had a greater proportion of Hydracarina and Gastropoda, and a smaller proportion of Ostracoda and Diptera, compared to Northeast Lake. The increased calcium concentrations in Snap Lake water (De Beers 2013, 2014b) likely resulted in increased snail (Gastropod) populations in the littoral zone of Snap Lake.

The relative densities of Chironomidae taxa in the littoral zone were highly variable in Snap Lake, particularly in the main basin, and were dominated by Tanytarsini, Orthocladiinae, or Chironomini. Communities closest to the diffuser differed from the rest of Snap Lake by generally having fewer Orthocladiinae. The deep-water benthic invertebrate community had high relative densities of both the Chironomini and Tanytarsini tribes (De Beers 2012). Communities in the littoral zones in Northeast Lake were less variable than Snap Lake and were dominated by Orthocladiinae.

Based on the samples collected in 2013, there were differences in littoral invertebrate functional feeding group composition between Snap Lake and Northeast Lake; compared to Northeast Lake, the main basin of Snap Lake contained a greater diversity of functional feeding groups, more suspension feeders, deposit feeders, scrapers, and parasites, and fewer herbivores. The observation of fewer herbivores in Snap Lake indicates that changes in epilithic algal communities are not a result of increased herbivory in the area. Community composition in terms of functional feeding groups was similar at all stations in the main basin of Snap Lake, indicating similar food type and availability to invertebrates throughout the lake's perimeter. An exception to this was at SNAP LZ01, where the relative density of Hydracarina was up to 2.7 times higher than at the other stations.

Littoral invertebrates can exert a strong top-down influence on 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). The stable isotope special study (Section 11.5) aids in understanding the extent to which littoral grazers use and incorporate epilithic algal biomass, which also aids in the interpretation of how the Mine could be affecting the epilithic algal community.

The quantitative Hester-Dendy sampling method proved to be a better method for collecting representative littoral taxa in 2013. The qualitative sweep-net method cannot provide the necessary information needed to provide an accurate understanding of the littoral invertebrate communities in Snap Lake and Northeast Lake.

11.1.6 Conclusions

11.1.6.1 Key Question 1: Can littoral zone 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?

Littoral zone monitoring can be conducted in Snap and Northeast Lakes. Differences were apparent in the epilithic algal community and its associated bacteria and detritus between 2004, 2012, and 2013, and between Snap Lake and Northeast Lake in 2012 and 2013. Continuation of this Special Study for another year, as planned, is required to adequately answer the question of whether possible Mine-related changes can be detected.

11.1.6.2 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. The molar ratios of C:N and C:P indicate 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 was similar in 2012 and 2013, but has decreased since 2004, indicating a change towards greater nutrient sufficiency. This could indicate Mine-related nutrient enrichment of the epilithic algae and associated bacteria in the main basin of Snap Lake. If so, this could also indicate availability of additional food for invertebrates and fish.

The percentage of viable algal carbon has increased in Snap Lake, particularly in the main basin, since 2004. This 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 could also be an indication of Mine-related nutrient enrichment. Continuation of this Special Study for an additional year is required before possible Mine-related changes on food quality for invertebrates and fish can be adequately assessed.

11.1.6.3 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 among years 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 and 2013, 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. Diatom biomass has increased in the main basin of Snap Lake, but not in the other areas. This could indicate a Mine-related effect, but continuation of this Special Study for an additional year is required before the question of possible Mine-related effects can be adequately answered.

11.1.6.4 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?

Invertebrate community composition differed between Snap Lake and Northeast Lake in 2013; the Snap Lake littoral community was more diverse and contained fewer Chironomidae, but more Hydracarina and Gastropods, compared to Northeast Lake. In addition, the relative proportion of Chironomidae taxa differed between the two Lakes. 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), and the shift in community dominance from Chironomidae to Gastropoda and Hydracarina in Snap Lake may indicate Mine-related environmental effects. Furthermore, the increase in Gastropod density in Snap Lake could be due to Mine-related increases in calcium concentrations. Continuation of this special study for an additional year is required before the question of possible Mine-related effects can be adequately answered.

11.1.7 Recommendations

Based on the 2013 sampling program, it is recommended that Hester-Dendy artificial substrate samplers should be used to quantitatively sample littoral invertebrates, and the sweep-net sampling method should be discontinued.

11.1.8 References

Beaty SR, Fortino K, Hershey AE. 2006. Distribution and growth of benthic macroinvertebrates among different patch types of the littoral zones of tow Arctic lakes. Freshw Biol 51: 2347-2361.

Cattaneo A. 1987. Periphyton in lakes in different trophy. Can J Fish Aquat Sci 44: 296-303.

Clifford H. 1991. Aquatic Invertebrates of Alberta. University of Alberta Press, Edmonton, AB, Canada.

Coffman WP, Ferrington LC Jr. 1996. Chironomidae. In Merritt RW, Cummins KW (eds), An Introduction to the Aquatic Insects of North America. 3rd Edition. Kendall/Hunt Publishing Company, Dubuque, IA, USA, pp 635-754.

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

De Beers. 2005. 2004 Annual Report in Support of the Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence (MV22001L2-0002). Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012. 2011 Annual Report in Support of the Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence (MV2011L2-0004). Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2013. 2012 Annual Report in Support of the Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence (MV2011L2-0004). Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

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

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

Elser JJ, Sterner RW, Galford AE, Chrzanowski TH, Findlay DL, Mills KH, Paterson MJ, Stainton MP, Schindler DW. 2000. Pelagic C:N:P stoichiometry in a eutrophied lake: responses to a whole-lake food web manipulation. Ecosystems 3:293-307.

Environment Canada. 2002. Revised Guidance for Sample Sorting and Subsampling Protocols for Environmental Effects Monitoring (EEM) Benthic Invertebrate Community Surveys. National EEM Office, Ottawa, ON, Canada.

Environment Canada. 2012. Metal Mining Technical Guidance Document for Aquatic EEM. National EEM Office, Ottawa, ON, Canada.

Epler JH. 2001. Identification Manual for the Larval Chironomidae (Diptera) of North and South Carolina. Special publication SJ2001-SP13. North Carolina Department of Environment and Natural Resources, Division of Water Quality, Raleigh, NC, USA.

Fairchild WG, Lowe RL. 1984. Artificial substrates which release nutrients: effects on periphyton and invertebrate succession. Hydrobiologia 114: 29-37.

Findlay D, Kling HJ. 1979. A Species List and Pictorial Reference to the Phytoplankton Species of Central and Northern Canada. Parts I and II. Can Fish Mar Serv MS Report 1503: 619pp.

Frost PC, Elser JJ, Turner MA. 2002. Effects of caddisfly grazers on the elemental composition of epilithon in a boreal lake. J N Am Benthol Soc 21:54-63.

Gibbons WN, Munn MD, Paine MD. 1993. Guidelines for Monitoring Benthos in Freshwater Environments. Environment Canada, North Vancouver, BC, Canada.

Haney JF. 1987. Field studies on zooplankton-cyanobacteria interaction. New Zealand J Mar Fresh Res 21:467-475.

Healey FP, Hendzel LL. 1980. Physiological indicators of nutrient deficiency in lake phytoplankton. Can J Fish Aquat Sci 37:442-453.

Hille KA. 2008. Does aquaculture impact benthic algal ecology? MSc Thesis, University of Manitoba, Winnipeg, MB, Canada.

Hillebrand H, Sommer U. 1999. The nutrient stoichiometry of benthic microalgal growth: redfield proportions are optimal. Limnol Oceangr 44:440-446.

Hustedt F. 1930. Bacillariophyta (Diatomeae). In A Pascher (ed), Die Susswasser-Flora Mitteleuropas. Heft 10. Sweite Auglage. Fisher, Jena, Germany.

Kahlert M, Petterson K. 2002. The impact of substrate and lake trophy on the biomass and nutrient status of benthic algae. Hydrobiologia 498: 161-169.

Kathman RD, Brinkhurst RO. 1998. (Revised May 1999). Guide to the Freshwater Oligochaetes of North America. Aquatic Resources Center, Thompsons Station, TN, USA.

Krebs, C. J. 1999. Ecological Methodology. Benjamin/Cummings. Menlo Park, CA, USA.

Lamberti GA, Moore JW. 1984. Aquatic insects as primary consumers. In VH Resh, DM Rosenberg (eds), The Ecology of Aquatic Insects. Prager, New York, NY, USA.

Maschwitz DE, Cook EF. 2000. Revision of the Nearctic species of the Genus *Polypedilum* Kieffer (Diptera: Chironomidae) in the Subgenus P. (*Polypedilum*) Kieffer and P. (*Urespedilum*) Oyewo and Saether. Ohio Biol Surv Bull 12: 1-135.

McAlpine JF, Peterson BV, Shewell GE, Teskey HJ, Vockeroth JR, Wood DM (coords). 1981. Manual of Nearctic Diptera. Volume 1. Agriculture Canada Research Branch Monograph 27, Ottawa, ON, Canada.

Merritt RW, Cummins KW (eds). 1996. An Introduction to the Aquatic Insects of North America. 3rd Edition. Kendall/Hunt Publishing Company, Dubuque, IA, USA.

Merritt RW, Cummins KW, Berg MB (eds). 2008. An Introduction to the Aquatic Insects of North America. 4th Edition. Kendall/Hunt Publishing Company, Dubuque, IA, USA.

Moss B. 2005. The littoral and profundal communities of lakes. In Moss B (ed), Ecology of Fresh Waters: Man and Medium, Past to Future. Blackwell Publishing, Liverpool, UK.

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

Nauwerck A. 1963. Die Beziehungen zwischen Zooplankton und Phytoplankton in See Erken. Syme Bot Upsaliensis 17: 1-163.

Northington RM, Keyse MD, Beaty SR, Whalen SC, Sokol ER, Hershey AE. 2010. Benthic secondary production in eight oligotrophic Arctic Alaskan lakes. J N Am Benthol Soc 29: 465-479.

Oliver DR, Roussel ME. 1983. The Genera of Larval Midges of Canada (Diptera: Chironomidae). Part 11. The Insects and Arachnids of Canada. Agriculture Canada Research Branch, Ottawa, ON, Canada.

Owen BB, Afzal M, Cody WR. 1978. Staining preparations for phytoplankton and periphyton. Brit Phycol J 13: 155-160.

Patrick R, Reimer CW. 1966. The Diatoms of the United States. Monogr Acad Nat Sci Philadelphia 13: 688pp.

Pennak RW. 1989. Fresh-water Invertebrates of the United States. 3rd Edition. John Wiley & Sons, New York, NY, USA.

Riber HH, Sorensen JP, Kowalczewski A. 1983. Exchange of phosphorus between water, macrophytes and epiphytic periphyton in the littoral of Mikolajskie Lake, Poland. In RG Wetzel (ed), Periphyton of Freshwater Ecosystems. Dr W Junk Publishers, The Hague, Netherlands.

Riber HH, Wetzel RG. 1987. Boundary-layer and internal diffusion effects on phosphorous fluxes in lake periphyton. Limnol Oceanogr 32: 1181-1194.

Robinson GGC. 1983. Methodology: the key to understanding periphyton. In RG Wetzel (ed), Periphyton of Freshwater Ecosystems, Dr. W Junk Publishers, The Hague, Netherlands.

Rosenberg DM, Resh VH (eds). 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall, New York, NY, USA.

Sand-Jensen K. 1983. Physical and chemical parameters regulating growth of periphytic communities. In RG Wetzel (eds), Periphyton of Freshwater Ecosystems, Dr W Junk Publishers, The Hague, Netherlands.

Schindler DW. 1974. Eutrophication and recovery in experimental lakes: implications for lake management. Science 184: 8978-899.

Schindler DW. 1978. Factors regulating phytoplankton production and standing crop in the world's freshwaters. Limnol Oceanogr 23: 478-486.

Simpson, E. H. 1949. Measurement of diversity. Nature 163: 688-688.

Smith B, Wilson CM. 1996. A consumer's guide to evenness indices. Oikos 76: 70-82.

Smith JE, Smith CM, Hunter CL. 2001. An experimental analysis of the effects of herbivory and nutrient enrichment on benthic community dynamics on a Hawaiian reef. Coral Reefs 19: 332-342.

Soponis AR. 1977. A revision of the Nearctic species of *Orthocladius* Van Der Wulp (Diptera: Chironomidae). Mem Entomol Soc Can 102: 1-187.

Stevenson RJ, Bothwell ML, Lowe RL. 1982. Production and chlorophyll concentration of epipelic and epilithic algae in fertilized and non-fertilized subarctic lakes. Hydrobiologia 126: 213-219.

Strayer D, Likens GE. 1986. An energy budget for the zoobenthos of Mirror Lake, New Hampshire. Ecology 67: 303-313.

Thorp JH, Covich AP (eds). 2001. Ecology and Classification of North American Freshwater Invertebrates. 2nd Edition. Academic Press, San Diego, CA, USA.

Turner MA. 1993. The Ecological Effects of Experimental Acidification Upon Littoral Algal Associations of Lakes in the Boreal Forest. PhD Dissertation, University of Manitoba, Winnipeg, MB, Canada

Turner MA, Jackson MB, Findlay DL, Graham RW, DeBruyn ER, Vandermeer EM. 1987. Early responses of periphyton to experimental lake acidification. Can J Fish Aquat Sci 44: 135-149.

Turner MA, Howell TE, Robinson GGC, Campbell P, Hecky RE, Schindler EU. 1994. Roles of nutrients in controlling growth of epilithon in oligotrophic lakes of low alkalinity. Can J Fish Aquat Sci 51: 2784-2793.

Vadeboncoeur YM, Vander Zanden J, Lodge DM. 2002. Putting the lake back together: reintegrating benthic pathways into lake food web models. Bioscience 52:44-53

Vollenweider RA. 1974. A Manual on Methods for Measuring Primary Production in Aquatic Environments. Burgess and Son (Abingdon) Ltd, Edinburgh, Scotland, UK.

Wehr JD, Sheath RG (eds). 2003. Freshwater Algae of North America: Ecology and Classification. Academic Press, San Diego, CA, USA.

Wetzel RG. 2001. Limnology 3rd edition. Elsevier Science Academic Press, New York, NY, USA.

Wiederholm T (ed). 1983. Chironomidae of the Holarctic Region. Keys and Diagnoses. Part 1. Larvae. Entomol Scand Suppl 19: 1-457.

Wiggins GB. 1996. Larvae of the North American Caddisfly Genera (Trichoptera). 2nd Edition. University of Toronto Press, Toronto, Ontario, Canada.

SECTION 11.2

PICOPLANKTON SPECIAL STUDY

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Program
De Beers	De Beers Canada Inc.
EAR	Environmental Assesment Report
Golder	Golder Associates Ltd.
MVEIRB	Mackenzie Valley Environmental Impact Review Board
MVLWB	Mackenzie Valley Land and Water Board
Mine	Snap Lake Mine
N	nitrogen
P	phosphorus
QA	quality assurance
QC	quality control
RPD	relative percent difference
SE	standard error
TDS	total dissolved solids
TSS	total suspended solids

UNITS OF MEASURE

Term	Definition
±	plus or minus
cells/mL	cells per millilitre
mL	millilitres
%	percent
µm	micrometres

11.2 Picoplankton Special Study

11.2.1 Introduction

While phytoplankton community metrics can be useful indicators of environmental change, because of their rapid response to changes in nutrients or other substances, picoplankton abundance can be used as an even earlier indicator of nutrient changes. A special study to incorporate monitoring of picoplankton and augment the existing phytoplankton monitoring program (Section 5) was recommended in 2007 (De Beers 2008). This special study was implemented in 2008 and has provided insight and information supporting the current trends observed within the phytoplankton community (De Beers 2009, 2010, 2011, 2012a, 2013). Continuation of this special study was recommended as part of the AEMP re-evaluation (De Beers 2012b) and included in the AEMP Design Plan (De Beers 2014).

There is one impact hypotheses examined in the AEMP for the Mine that is addressed in the picoplankton special study, this is the nutrient enrichment hypothesis. Eutrophication could occur due to the release of nutrients (primarily phosphorus [P] and nitrogen [N], and, for some species, total dissolved solids [TDS]) to Snap Lake.

11.2.1.1 Background

Picoplankton are the smallest size category of plankton ranging between 0.2 and 2.0 micrometres (µm). Picoplankton includes two major groups, free–living heterotrophic bacteria and small autotrophic phytoplankton, with the most ubiquitous being pico-cyanobacteria. Flagellates are flagellated autotrophic, mixotrophic, and heterotrophic phytoplankton that are larger in size (i.e., nanometre size class ranging from 2.0 to 20 µm) than the hetertrophic bacteria and pico-cyanobacteria. Flagellates are often the primary predator of picoplankton (Stockner 1991).

These organisms 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 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 P and N (Stockner and Shortreed 1994; Schallenberg and Burns 2001). In addition, picoplankton provides a rich food source for zooplankton, which ultimately translates into food resources for fish. 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 pressure (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).

11.2.1.2 Objective

The primary objective of the 2013 Picoplankton Special Study was to monitor changes in picoplankton abundance in Snap Lake, Northeast Lake, and Lake 13. The results of this special study provide supporting information to the phytoplankton component of the AEMP (Section 5). Analyses of the 2013 Picoplankton Special Study data addressed the following two key questions:

- Key Question 1: 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?
- Key Question 2: How do any observed changes in the picoplankton community compare to changes observed in the phytoplankton community?

11.2.2 Methods

11.2.2.1 Sampling Locations and Timing

The Picoplankton Special Study was completed in conjunction with the open-water plankton monitoring program between July 7, 2013 and September 15, 2013 (Section 5). Sampling occurred at the same plankton stations in Snap Lake, Northeast Lake, and Lake 13 (Section 5.2: Table 5-1; Figures 5-1 to 5-3).

11.2.2.2 Field Methods

To accurately assess seasonal variability of the picoplankton and flagellate communities in Snap Lake, Northeast Lake, and Lake 13, sampling occurred monthly during the open-water period between July and September. A summary of the sampling events completed in Snap Lake, Northeast Lake, and Lake 13 is presented in Table 11.2.1.

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 millilitres (mL) of buffered formalin, while flagellate samples were field-preserved with 2.5 mL of Lugol’s solution.

Table 11.2.1 Summary of Picoplankton Community Sampling Events in Snap Lake, Northeast Lake, and Lake 13, 2013

Variable	July 7 to 16, 2013 (n)	August 8 to 14, 2013 (n)	September 5 to 15, 2013 (n)
Snap Lake – Main Basin			
Picoplankton[(a)]	5	5	5
Flagellates	5	5	5
Snap Lake – Northwest Arm			
Picoplankton[(a)]	4	4	4
Flagellates	4	4	0
Northeast Lake			
Picoplankton[(a)]	5	5	5
Flagellates	5	5	5
Lake 13			
Picoplankton[(a)]	4	5	5
Flagellates	5	5	5

Notes:
a) Picoplankton includes abundance of heterotrophic bacteria and pico-cyanobacteria.
n = number of samples

Picoplankton samples were submitted to Advanced Eco-solutions Inc. in Liberty Lake, WA, USA for analysis of abundance of heterotrophic bacteria and pico-cyanobacteria. Flagellate samples were submitted to Eco-logic Ltd. in West Vancouver, BC for analysis of flagellate abundance.

11.2.2.3 Sample Sorting and Taxonomic Identification

Heterotrophic bacteria and pico-cyanobacteria were processed and enumerated using epi-fluorescence microscopy techniques as described by MacIsaac and Stockner (1981, 1993) and Stockner (2005).

Prior to quantitative enumeration of the flagellates, the samples were gently shaken for 60 seconds, carefully poured into 25-mL settling chambers, and allowed to settle for a minimum of six hours. Enumeration was 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). Taxonomic references were the compendia of Prescott (1978), Canter-Lund and Lund (1995), and Wehr and Sheath (2003).

11.2.2.4 Data Analyses

Data analyses methods were designed to answer the two key questions as outlined in Table 11.2.2. Specific details relevant to the data analyses methods to address each key question are provided in Sections 11.2.2.5 and 11.2.2.6.

Table 11.2.2 Overview of Analysis Approach for Picoplankton Special Study Key Questions

Key Question	Overview of Analysis Approach
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?	A qualitative review of the picoplankton and flagellate data was completed to evaluate changes in abundance and determine (a) whether there was growth inhibition, and (b) whether this was related to nutrient enrichment within Snap Lake. Quantitative comparisons (i.e., statistical tests) will be completed as part of the AEMP four-year re-evaluation report in 2016.
How do any observed changes in the picoplankton community compare to changes observed in the phytoplankton community?	Visual assessments of the spatial and temporal trends observed in the picoplankton, flagellate, and phytoplankton communities were conducted.

11.2.2.5 Key Question 1: 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?

A qualitative review of the picoplankton and flagellate data was completed as part of the 2013 annual AEMP report. The review evaluated changes in abundance and determined whether there was growth inhibition and, if so, whether this was related to nutrient enrichment within Snap Lake. Quantitative comparisons (i.e., statistical analyses) will be completed as part of the four-year re-evaluation of the AEMP in 2016 as outlined in the 2013 AEMP Design Plan (De Beers 2014).

11.2.2.6 Key Question 2: How do any observed changes in the picoplankton community compare to changes observed in the phytoplankton community?

Visual assessments of the spatial and temporal trends observed in the picoplankton and flagellates were conducted. A qualitative comparison to the phytoplankton communities was also completed.

11.2.3 Quality Assurance and Quality Control

11.2.3.1 Overview of Procedures

Quality assurance and quality control (QA/QC) procedures were applied during all aspects of the picoplankton component to check 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.

Split samples for heterotrophic bacteria and pico-cyanobacteria were analyzed to assess the taxonomist's counting efficiency. The inherent variability associated with the picoplankton samples makes the establishment of a QC threshold value difficult. For the purposes of the picoplankton QC assessment, samples were flagged and assessed further if there was a relative percent difference (RPD) greater than

50 percent (%) between the field and QC samples. Flagged data were not automatically rejected because of exceedance of the acceptance criterion; rather, they were evaluated on a case-by-case basis, as some level of within-station variability is expected for these types of 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 2008 to 2013 data for a range comparison. If the data were outside the corresponding 2008 to 2013 range, laboratory re-analysis occurred. If laboratory re-analysis confirmed the results, the outlier points were retained in the final data set unless there was a technically defensible reason to exclude them.

The data were also reviewed for unusually high or low values (i.e., greater or less than 10 times typical lake values), which would suggest erroneous results. Unusually high or low results were invalidated on a case-by-case basis. Invalidated data were retained in Appendix 11A 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.

11.2.3.2 Summary of QA/QC Results

In general, there was consistency between the field sample and QC sample results for the heterotrophic bacteria and pico-cyanobacteria (Appendix 11.2A; Table 11.2A-1). The RPD exceeded 50% for heterotrophic bacteria from SNAP11A in August (167%) and for pico-cyanobacteria from NEL05 in July (67%). Further follow-up assessment of these samples determined the results for the field samples were within the 2008 to 2013 ranges and were deemed valid.

The following flagellate samples had RPDs exceeding 50% at one of the major taxonomic group levels (Appendix 11.2A; Table 11.2A-2):

- SNAP11A in September for dinoflagellates (RPD = 200%);
- NEL05 in July for chlorophytes (RPD = 67%); and,
- Lake13-05 in September for chryso-cryptophytes (RPD = 86%).

An RPD greater than 50% in total flagellate abundance was identified in one sample from Lake 13 (Station LK13-05) in September (RPD 79%). Further follow-up assessment of these samples determined the results for the field sample were within the 2008 to 2013 range and deemed to be valid.

11.2.4 Results

Appendix 11.2A contains detailed results from all sampling events for the picoplankton program as follows:

- Appendix 11.2A, Table 11.2A-3 – picoplankton (i.e., heterotrophic bacteria and pico-cyanobacteria) enumeration data; and,
- Appendix 11.2A, Table 11.2A-4 – flagellate taxonomic and enumeration data.

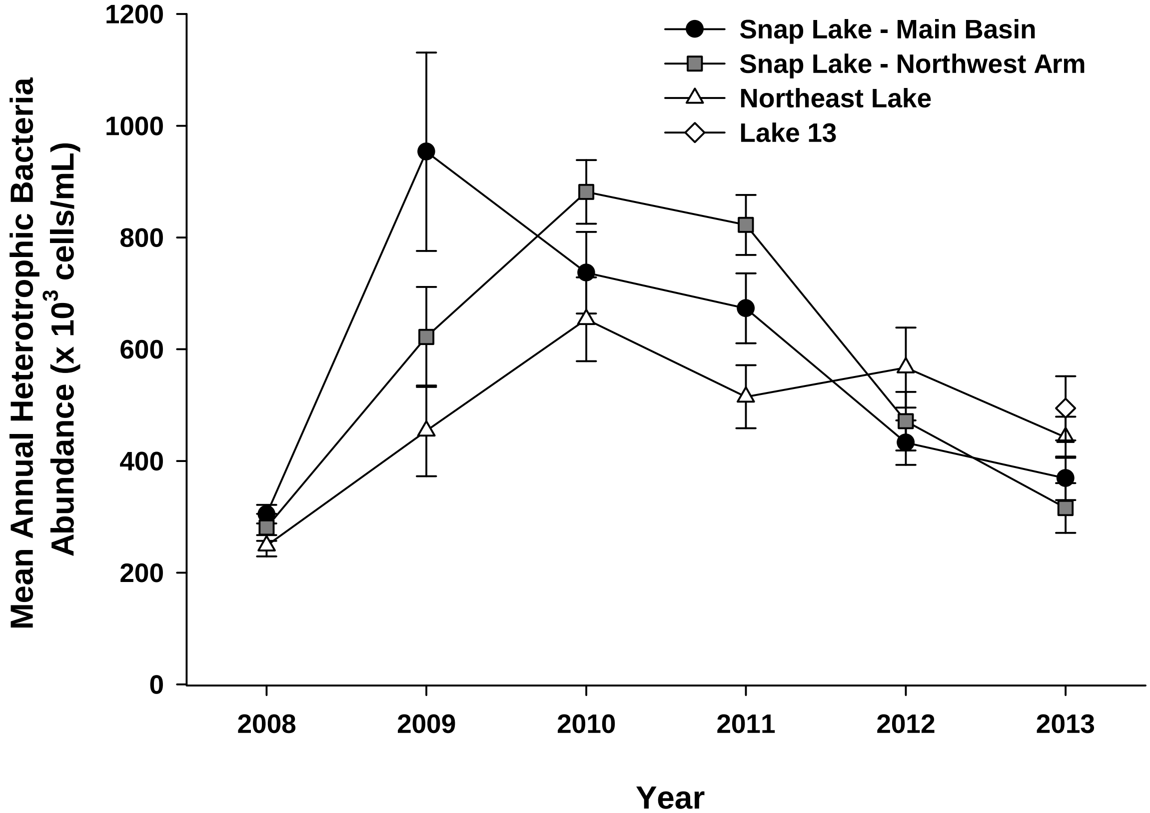
11.2.4.1 Heterotrophic Bacteria

Mean heterotrophic bacterial abundances in the main basin and northwest arm of Snap Lake were greater than in Northeast Lake between 2008 and 2011 (Figure 11.2-1). However, a decreasing trend in heterotrophic bacterial abundance has been observed in the main basin since 2009 and the northwest arm since 2010. In 2013, mean plus or minus standard error (± SE) annual heterotrophic bacterial abundances were higher in the main basin of Snap Lake (368,965 ± 44,495 cells per millilitre [cells/mL]) compared to the northwest arm (315,943 ± 39,021 cells/mL). Although mean (± SE) annual heterotrophic abundance has been higher in Northeast Lake (442,524 ± 36,632 cells/mL) compared to both areas of Snap Lake starting in 2012, overall no temporal trend was observed. Heterotrophic bacteria were not sampled in Lake 13 prior to 2013; therefore, no temporal trends can be determined. However, mean (± SE) annual heterotrophic bacterial abundance in Lake 13 (494,439 ±57,350 cells/mL) was within the range of Northeast Lake.

In general, heterotrophic bacterial abundances peaked in July and decreased over the open-water season at most stations in Snap Lake (Figure 11.2-2). In contrast, Northeast Lake exhibited peaks in heterotrophic bacterial abundance in September; high abundance values were also observed in July at all stations. More variability was observed in Lake 13 and no clear seasonal pattern was present. In August, the heterotrophic bacterial abundance at Station LK13-04 was almost double that of the highest abundances recorded at other stations in the lake.

A weak spatial pattern relative to the diffuser was observed in the main basin of Snap Lake (Figure 11.2-2), where stations closest to the diffuser had lower abundance values in July and August. This weak spatial pattern in the main basin of Snap Lake was absent in September. No spatial pattern relative to the diffuser was evident in the northwest arm of Snap Lake.

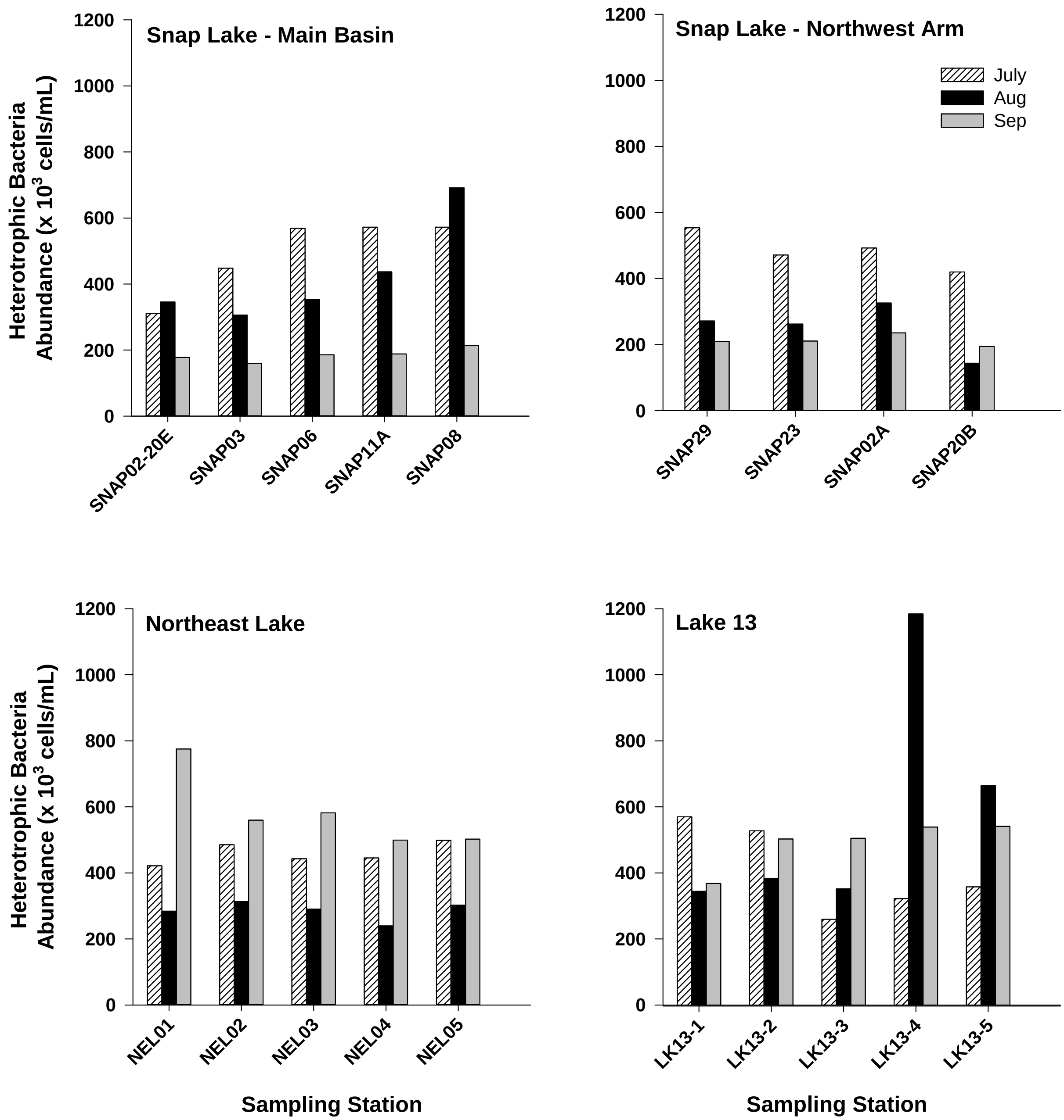
Figure 11.2-1 Temporal Trends in Mean Annual Heterotrophic Bacterial Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, 2008 to 2013



Note: Error bars represent standard error of the mean. Sampling in Northeast Lake did not include an August sampling session until 2011. Heterotrophic bacteria were not sampled in Lake 13 prior to 2013.

cells/mL = cells per millilitre.

Figure 11.2-2 Seasonal and Spatial Trends in Heterotrophic Bacterial Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake and Lake 13, 2013



Note: Stations in Snap Lake are arranged from closest to the diffuser (SNAP02-20E) to farthest from the diffuser (SNAP20B) in the main basin and northwest arm of Snap Lake.

cells/mL = cells per millilitre.

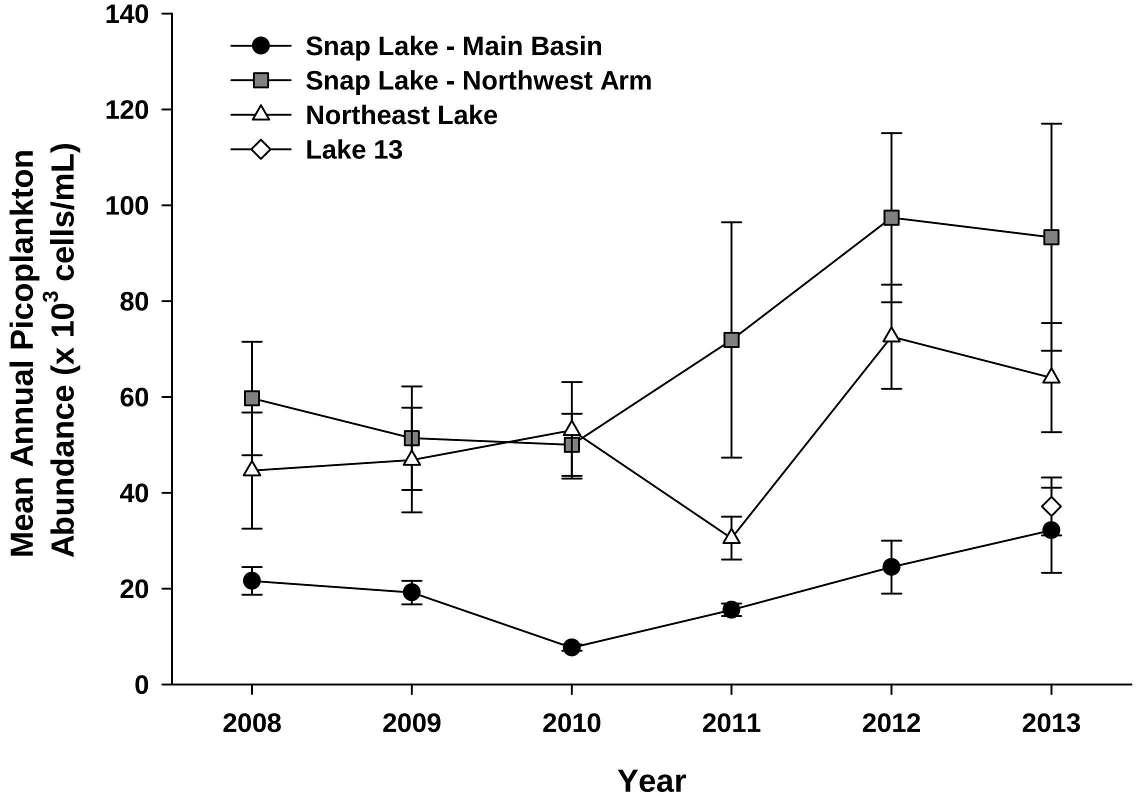
11.2.4.2 Pico-cyanobacteria

Mean annual pico-cyanobacteria abundances in the main basin of Snap Lake have remained relatively unchanged between 2008 and 2013 (15,547 ± 1,303 to 32,140 ± 8,889 cells/mL), with the exception of 2010, when there was a slight decrease (7,666 cells/mL; Figure 11.2-3). Since 2008, mean (± SE) annual pico-cyanobacteria abundances have been consistently higher in Northeast Lake (30,511 ± 4,451 to 72,571 ± 10,874 cells/mL) and the northwest arm of Snap Lake (49,976 ± 6,489 to 97,420 ± 17,656 cells/mL) compared to the main basin of Snap Lake. Mean annual pico-cyanobacteria abundances in Northeast Lake and the northwest arm of Snap Lake were similar from 2008 to 2010. In 2011, mean (± SE) annual pico-cyanobacteria abundances in Northeast Lake exhibited a decrease (30,511 ± 4,451 cells/mL), while mean (± SE) annual abundance in the northwest arm of Snap Lake increased (71,886 ± 24,588 cells/mL). In 2012, mean pico-cyanobacteria abundances increased in both Northeast Lake and in the northwest arm of Snap Lake. In 2013, slight decreases compared to the previous year were observed in the mean (± SE) annual pico-cyanobacteria abundance in Northeast Lake (64,013 ± 11,388) and the northwest arm of Snap Lake (93,333 ± 23,680). Pico-cyanobacteria were not sampled in Lake 13 prior to 2013; therefore, no temporal trend can be determined. However, in 2013, Lake 13 had similar mean annual pico-cyanobacterial abundance as the main basin of Snap Lake (37,139 ± 6,034 cells/mL), which was lower than the values observed in Northeast Lake or the northwest arm of Snap Lake.

In general, seasonal peaks in pico-cyanobacteria abundances occurred in September in the main basin and northwest arm of Snap Lake as well as in Northeast Lake in 2013 (Figure 11.2-4), with the exception of SNAP02A and NEL01, which peaked in August. In Lake 13, seasonal peaks in pico-cyanobacteria abundances were observed mainly in August.

There was no discernable spatial pattern of pico-cyanobacteria abundance relative to the diffuser in the main basin of Snap Lake (Figure 11.2-4). However, pico-cyanobacteria abundances were generally lower at the main basin stations compared with stations in the northwest arm.

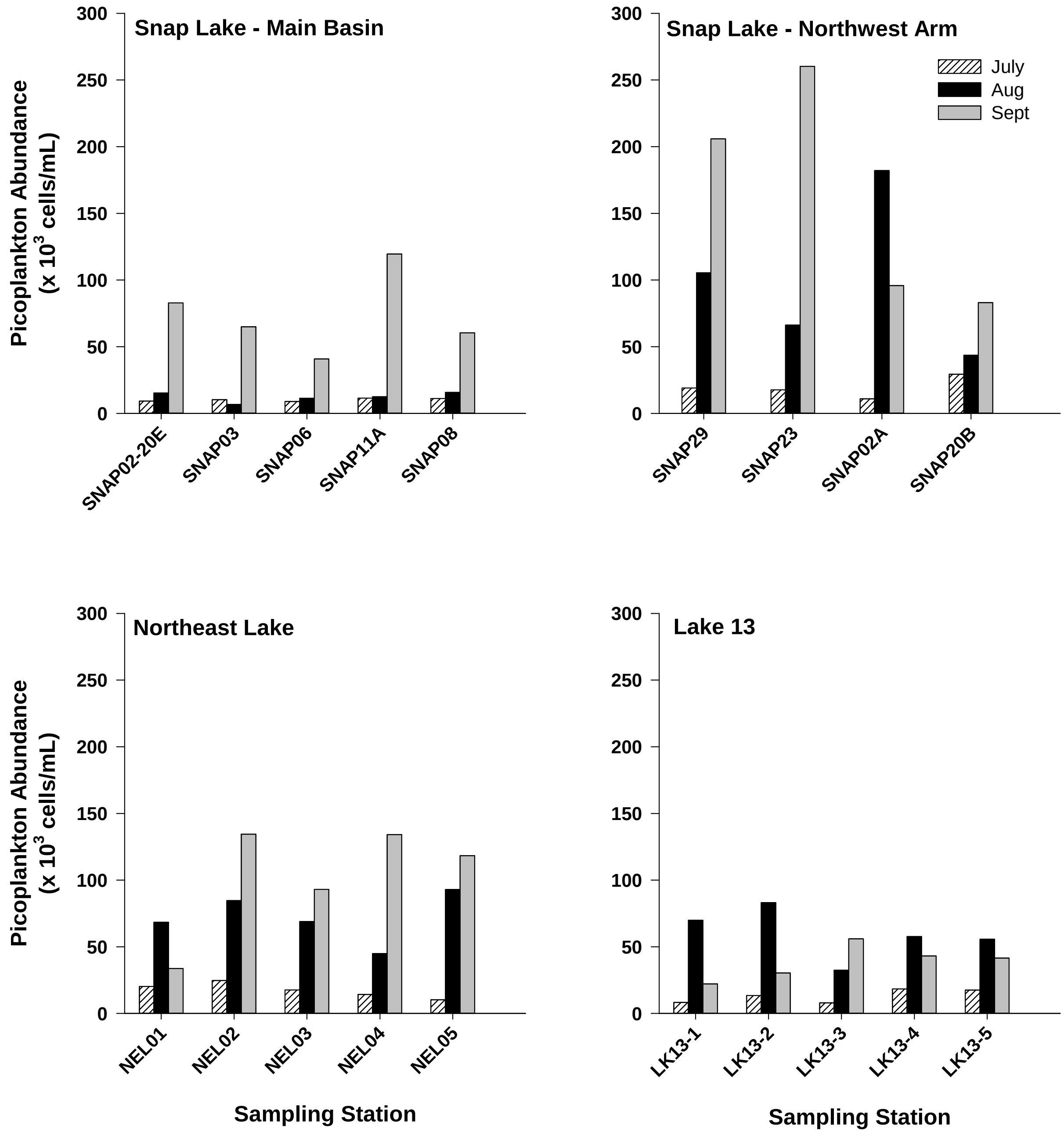
Figure 11.2-3 Temporal Trends in Mean Annual Pico-cyanobacteria Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, 2008 to 2013



Note: Error bars represent standard error of the mean. Sampling in Northeast Lake did not include an August sampling session until 2011. Pico-cyanobacteria were not sampled in Lake 13 prior to 2013.

cells/mL = cells per millilitre.

Figure 11.2-4 Seasonal and Spatial Trends in Pico-cyanobacteria Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, 2013



Note: Stations are arranged from closest to the diffuser (SNAP02-20E) to farthest from the diffuser (SNAP20B) in the main basin and northwest arm of Snap Lake.

cells/mL = cells per millilitre.

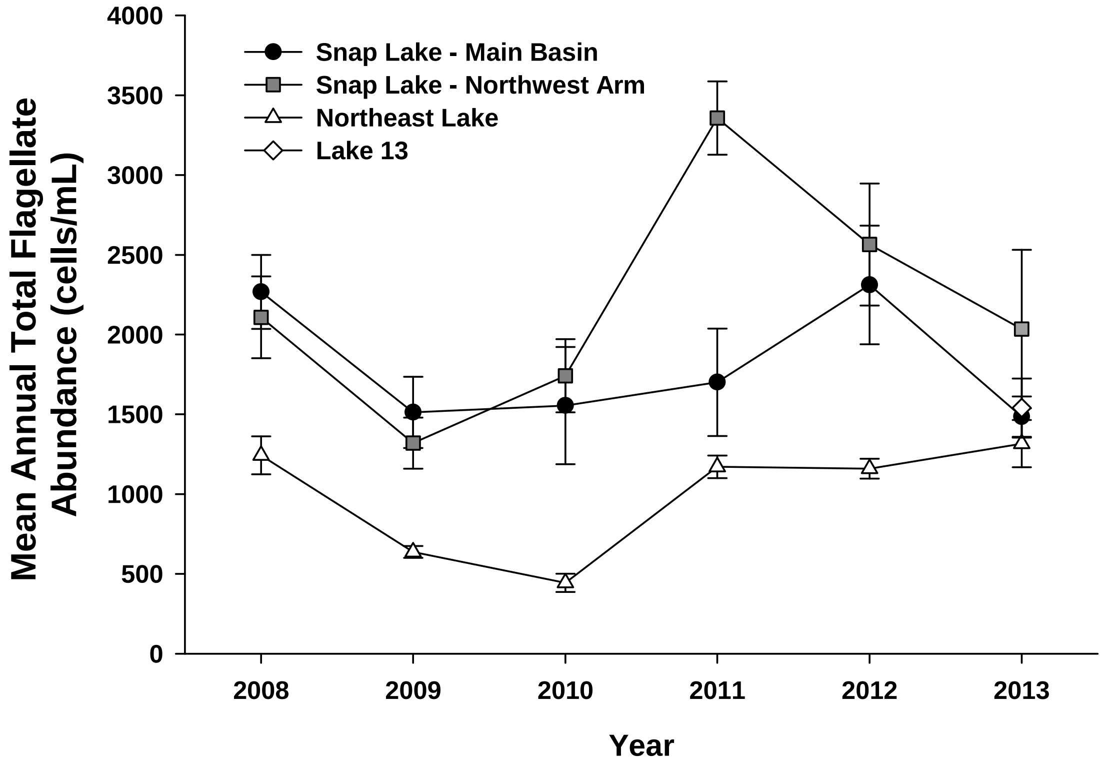
11.2.4.3 Flagellates

As in previous years, the chryso-cryptophyte group dominated the flagellate communities in Snap Lake and Northeast Lake in 2013 (Appendix 11.2A, Table 11.2A-4). Overall, the difference between mean annual flagellate abundance in Northeast Lake (1,316 ± 147 cells/mL) and the main basin of Snap Lake (1,484 ± 126 cells/mL) was less pronounced than in past years (Figure 11.2-5). Flagellate abundance in the northwest arm remained highest (2,034 ± 497 cells/mL) relative to the other lakes. However, the difference between the main basin and northwest arm of Snap Lake was more pronounced in 2013 than in previous years, with the exception of 2011 when flagellate abundance doubled in the northwest arm (3,357 ± 337 cells/mL) relative to the main basin (1,700 ± 230 cells/mL) (Figure 11.2-5). Flagellates were not sampled in Lake 13 prior to 2013; therefore, no temporal trend can be determined. However, in 2013, mean annual flagellate abundance in Lake 13 was similar to that in the main basin of Snap Lake (1,539 ± 185 cells/mL).

In 2013, seasonality of flagellate abundance was variable in Snap Lake and Lake 13. Most stations in the main basin of Snap Lake and Lake 13 exhibited peak flagellate abundances in August, although flagellate abundance remained high at a number of the stations in Lake 13 in September (Figure 11.2-6). In the northwest arm of Snap Lake, peak flagellate abundance occurred in September. Northeast Lake flagellates were most abundant during July, and a consistent seasonal pattern of decreasing abundance through the open-water season was evident.

No spatial relationship in flagellate abundance relative to the diffuser in the main basin of Snap Lake was observed in 2013. Particularly high flagellate abundance values were observed at Station SNAP02A in August (5,007 cells/mL) and September (6,142 cells/mL), with values triple the others observed anywhere else in Snap Lake (Figure 11-6). However, these high values did not correspond to abnormally low heterotrophic bacteria or pico-cyanobacterial abundances.

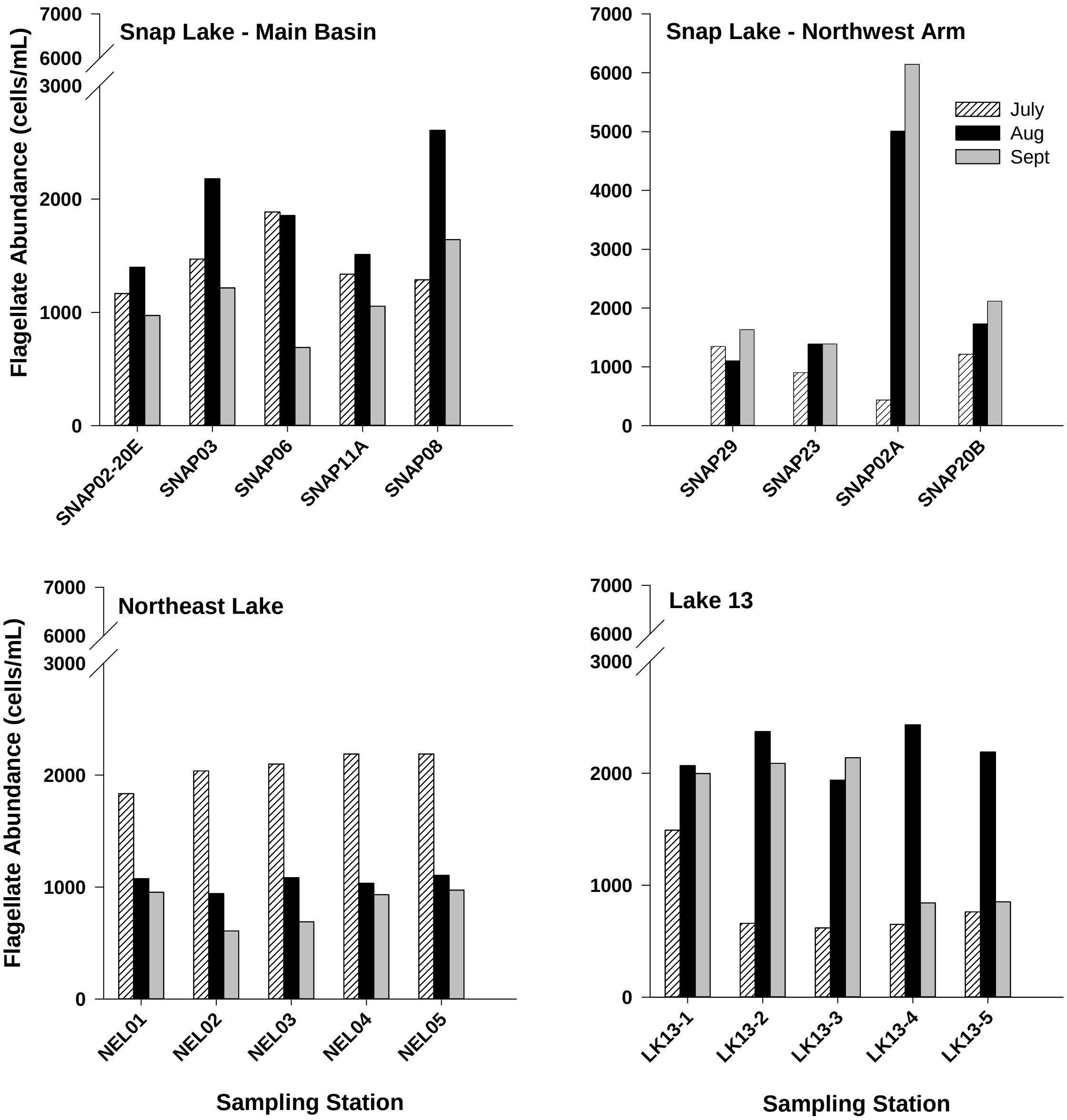
Figure 11.2-5 Temporal Trends in Mean Annual Flagellate Abundance in the Main Basin and Northwest Arm of Snap Lake, Northeast Lake, and Lake 13, 2008 to 2013



Note: Error bars represent standard error of the mean. Sampling in Northeast Lake did not include an August sampling session until 2011. Flagellates were not sampled in Lake 13 prior to 2013.

cells/mL = cells per millilitre.

Figure 11.2-6 Spatial and Temporal Flagellate Abundance in the Main Basin and Northwest Arm of Snap Lake, and Northeast Lake, 2008 to 2012 and Lake 13, 2013



Note: Stations are arranged from closest to the diffuser (SNAP02-20E) to farthest from the diffuser (SNAP20B) in the main basin and northwest arm of Snap Lake.

cells/mL = cells per millilitre.

11.2.5 Discussion

Picoplankton are divided into two major groups: free-living heterotrophic bacteria; and small autotrophic pico-cyanobacteria (Drakare 2002). Flagellated heterotrophic phytoplankton are larger and graze on picoplankton (Hall et al. 1993). Picoplankton, like phytoplankton, are sensitive to changes in environmental conditions and make for valuable biological monitoring tools. Changes in nutrient content, light availability, grazing, and concentrations of dissolved organic carbon affect the interactions of heterotrophic bacteria, pico-cyanobacteria, and larger flagellated phytoplankton.

Heterotrophic bacteria showed weak spatial patterns in the main basin of Snap Lake, with stations closest to the diffuser generally having lower heterotrophic bacterial abundances. Although there was no spatial pattern in pico-cyanobacteria abundance relative to the diffuser, stations within the main basin of Snap Lake had lower abundances compared to the northwest arm stations.

In 2013, there was no discernable spatial pattern in the flagellate abundances in relation to the diffuser in Snap Lake. In general, flagellate abundances in the main basin of Snap Lake were similar to the reference lakes. This is reflective of earlier sampling years and in contrast to the 2012 results, where closer proximity to the diffuser suggested an impact on flagellate abundance in Snap Lake.

Both heterotrophic bacteria and flagellate abundances decreased in 2013 within Snap Lake. Overall, heterotrophic bacteria abundances were lower in Snap Lake when compared to the reference lakes. In addition, flagellate abundance was lower in the main basin of Snap Lake than in the northwest arm, but similar to both reference lakes. Seasonal peaks occurred earlier in the season for heterotrophic bacteria (i.e., July), whereas pico-cyanobacteria and flagellates peaked in August or September.

Possible responses to nutrient enrichment within Snap Lake, particularly the main basin, were observed in lower abundances of heterotrophic bacteria, pico-cyanobacteria, and flagellate abundances in 2013. Although these findings are supportive of the expected trend of growth inhibition due to Mine-related increases in N and P (Stockner and Shortreed 1994; Schallenberg and Burns 2001), the weak spatial patterns, in conjunction with the phytoplankton results (Section 5), suggest that Snap Lake may be returning to a stable state similar to baseline; however, a lack of baseline picoplankton data prevent confirmation of this trend. 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).

11.2.6 Conclusions

11.2.6.1 Key Question 1: 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?

From 2008 to 2011, heterotrophic bacteria abundances in the main basin and northwest arm of Snap Lake were greater than in Northeast Lake. In 2012, heterotrophic bacterial abundance in the main basin and northwest arm of Snap Lake decreased while an increase occurred in Northeast Lake. In 2013, this decrease in heterotrophic bacteria abundance continued in the main basin and the northwest arm of Snap Lake. No clear temporal trend has been evident in Northeast Lake since 2010. Temporal trends cannot be assessed in Lake 13; however, in 2013, heterotrophic bacteria abundances were comparable to Northeast Lake.

In contrast, between 2008 and 2013, mean annual pico-cyanobacteria abundances were greater in Northeast Lake and the northwest arm of Snap Lake than in the main basin of Snap Lake. While there has been some variability over time, a general increasing trend was observed in Northeast Lake and the northwest arm of Snap Lake until 2013, when mean annual pico-cyanobacteria decreased in both areas. Since monitoring began in 2008, mean annual pico-cyanobacteria abundances have remained similar in the main basin of Snap Lake, with the exception of 2010, when a decrease in mean annual pico-cyanobacteria abundance was observed. Temporal trends cannot be assessed in Lake 13; however, pico-cyanobacteria abundances in this lake in 2013 were similar to values observed in the main basin of Snap Lake.

Lower heterotrophic bacteria, pico-cyanobacteria, and flagellate abundances in the main basin of Snap Lake suggest potential growth inhibition and Mine-related nutrient enrichment. Although weak spatial patterns, in conjunction with the phytoplankton results (Section 5), suggest that Snap Lake may be returning to a new stable state; however, a lack of baseline picoplankton data prevents confirmation of this trend.

11.2.6.2 Key Question 2: How do any 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. Changes observed in 2012 and 2013 suggest other changes in water quality (e.g., increased TDS and alkalinity) or additional supporting environmental factors may also be influencing the phytoplankton (Section 5) and picoplankton communities. In 2013, heterotrophic bacteria abundance continued to decrease within Snap Lake and a weak spatial trend in relation to the diffuser was evident in the main basin of Snap Lake. Mean annual total phytoplankton biomass also continued to decrease in the main basin of Snap Lake (Section 5), which may imply Mine-related factors (e.g., alteration in TDS and total suspended solids [TSS]) are influencing the populations.

Flagellate abundance in 2013 showed a similar declining trend in Snap Lake as heterotrophic bacteria and phytoplankton. No spatial trend relative to the diffuser was observed in 2013.

11.2.7 Recommendations

Based on the results to date, no changes are required for the picoplankton program. The inclusion of this special study will be re-assessed during the next AEMP re-evaluation in 2016.

11.2.8 References

Canter-Lund H, Lund JWG. 1995. Freshwater Algae: Their Microscopic World Explored. BioPress Ltd., Bristol, UK.

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

De Beers. 2004. Environmental Agreement, De Beers Snap Lake Diamond Project. Agreement between Government of Canada, Government of the Northwest Territories, and Aboriginal Bands. Signed May 31, 2004. Yellowknife, NWT, Canada.

De Beers. 2008. 2007 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2009. 2008 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2010. 2009 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2011. 2010 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012a. 2011 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012b. Aquatic Effects Re-evaluation Report. Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence (MV2011L2-0002). Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2013. 2012 Annual Report in Support of the Aquatics Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

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

Drakare S. 2002. The Role of Picophytoplankton in Lake Food Webs. Comprehensive Summaries of Uppsala dissertations from the Faculty of Science and Technology 763. Uppsala, Sweden.

Hall JA, Barrett DP, James M. 1993. The importance of phytoflagellate, heterotrophic flagellate and ciliate grazing on bacteria and picophytoplankton sized prey in a coastal marine environment. J Plankton Res 15:1075-1086.

Lund JWG, Kippling C, le Cren ED. 1958. The inverted microscope method of estimating algal numbers and the statistical basis for the estimation by counting. Hydrobiologia 11: 144-170.

MacIsaac EA, Stockner JG. 1981. Current trophic state and potential impacts of coal mine development on productivity of Middle Quinsam and Long Lake. Can Tech Rep Fish Aquat Sci 1381: 1-63.

MacIsaac EA, Stockner JG. 1993. Enumeration of phototrophic picoplankton by auto-fluorescence microscopy. In PF Kemp, BF Sherr, EB Sherr, J Cole (eds), Handbook of Methods in Aquatic Microbial Ecology. Lewis Publishing, Boca Raton, FL, USA, pp187-197.

Munawar M, Weisse T. 1989. Is the ‘microbial loop’ an early warning indicator of anthropogenic stress? Hydrobiologia 188:163-174.

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

MVLWB. 2012. Mackenzie Valley Land and Water Board Licence #MV2011L2-0004. Yellowknife, NWT, Canada.

Prescott GW. 1978. How to Know the Freshwater Algae. McGraw-Hill Science, New York, NY, USA.

Schallenberg M, Burns CW. 2001. Tests of autotrophic picoplankton as early indicators of nutrient enrichment in an ultra-oligotrophic lake. Freshw Biol 46: 27-37.

Stockner JG. 1991. Autotrophic picoplankton in freshwater ecosystems: The view from the summit. Int Rev Gesamten Hydrobiol 76: 483-492.

Stockner JG. 2005. Reservoir Monitoring Manual Dworshak Reservoir. Prepared for US Army Corps of Engineers. Walla Walla, WA, USA.

Stockner JG, Shortreed KS. 1994. Autotrophic picoplankton community dynamics in a pre-alpine lake in British Columbia, Canada. Hydrobiologia 274: 133-142.

Utermohl H. 1958. Zur vervollkommnung der quantitative phytoplankton-metodik. Mitt Int Verh Limnol 9: 1-28.

Wehr JD, Sheath RG. 2003. Freshwater Algae of North America. Academic Press, San Diego, CA, USA.

SECTION 11.3

DOWNSTREAM LAKES SPECIAL STUDY

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

Acronyms

Abbreviation	Definition
AEMP	Aquatic Effects Monitoring Program
ALS	ALS Canada Ltd.
APHA	American Public Health Association
BOD	biochemical oxygen demand
$CaCO_3$	calcium carbonate
CCME	Canadian Council of Ministers of the Environment
De Beers	De Beers Canada Inc.
DL	detection limit
DO	dissolved oxygen
DQO	data quality objective
DSL1	Downstream Lake 1
DSL2	Downstream Lake 2
EAR	Environmental Assessment Report
ERA	ecological risk assessment
Flett	Flett Research Ltd.
GEMSS	Generalized Environmental Modelling System for Surfacewaters
GF/C	glass fibre filter type C
GIS	global information system
Golder	Golder Associates Ltd
GPS	global positioning system
H1	Hydrology Station 1
H2	Hydrology Station 2
HCO_3	bicarbonate
ID	identification number
i.e.	that is
ISQG	interim sediment quality guideline
KING	King Lake
LCB	Lac Capot Blanc
Maxxam	Maxxam Analytics Inc
MDS	Multiparameter Display System
Mine	Snap Lake Mine
N	nitrogen
NAD	North American Datum
NTU	nephelometric turbidity units
P	phosphorus
PEL	Probable Effect Level
PVC	polyvinyl chloride
QA	quality assurance
QC	quality control
QS	quick sample
RPD	relative percent difference
SCN	sample control number

Abbreviation	Definition
SD	standard deviation
SIO_2	silicate
SQG	sediment quality guideline
SSWQO	site-specific water quality objective
TDS	total dissolved solids
TN	total nitrogen
TP	total phosphorus
TOC	total organic carbon
UofA	University of Alberta Biogeochemcial Analytical Service Laboratory
UTM	Universal Transverse Mercator
WQG	water quality guideline
wt	weight
3-D	three dimensional

Units of Measure

Abbreviation	Definition
ºC	degrees Celsius
%	percent
<	less than
>	greater than
±	plus or minus
≤	less than or equal to
µm	micrometre
µg/L	micrograms per litre
µS/cm	microSiemens per centimetre
cm	centimetre
km	kilometre
L	litre
m	metre
mg/L	milligrams per litre
mg-N/L	milligrams as nitrogen per litre
mg-P/L	milligrams as phosphorus per litre
mg/kg	milligrams per kilogram
mL	millilitre
mm	millimetre

11.3 Downstream Lakes Special Study

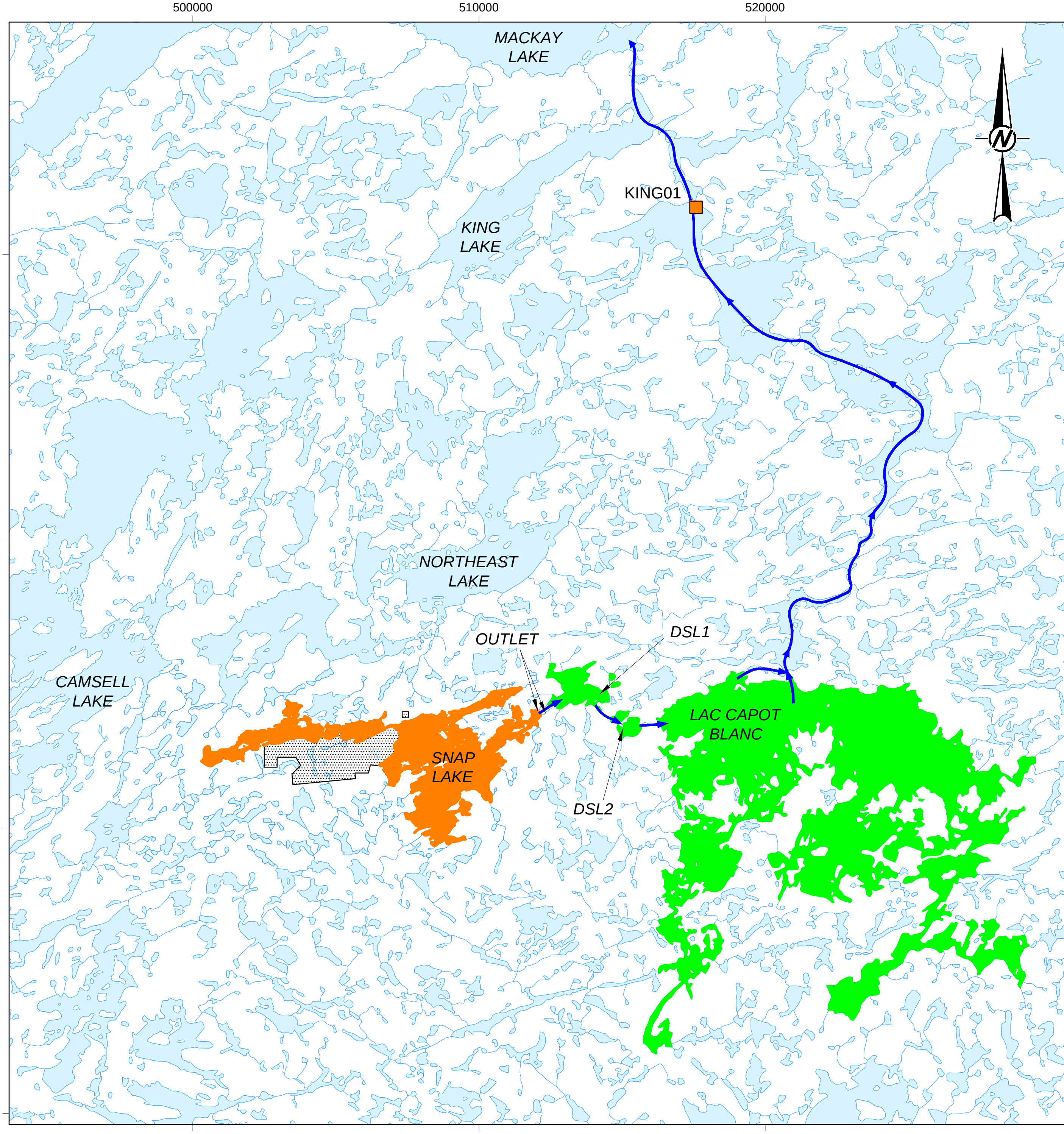
11.3.1 Introduction

11.3.1.1 Background

Treated effluent is becoming evenly mixed throughout the main basin of Snap Lake. As such, the focus of the Aquatic Effects Monitoring Program (AEMP) has shifted from evaluating spatial and seasonal trends in Snap Lake to monitoring trends over time and changes downstream of Snap Lake. The AEMP currently includes one long-term downstream monitoring station at King Lake (KING01), approximately 25 kilometres (km) downstream of Snap Lake (Figure 11.3-1). Initial reconnaissance work was recently initiated to investigate the extent of treated effluent immediately downstream of Snap Lake. The timeline for data collection downstream of Snap Lake was:

- 2011 - reconnaissance work initiated to investigate the location and extent of the treated effluent plume downstream of Snap Lake.
- 2012 - characterization of conditions downstream of Snap Lake continued as part of a special study under the AEMP. In addition, the AEMP re-evaluation was completed and included a recommendation to continue monitoring downstream of Snap Lake. Formal monitoring requirements were proposed as part of the Downstream Lakes Special Study in the 2013 AEMP Design Plan (De Beers 2014).
- 2013 – the 2013 AEMP Design Plan (De Beers 2014) was approved and monitoring was completed with the objectives of further documenting the extent of the treated effluent downstream of Snap Lake, assessing current water quality and sediment quality conditions in the first three lakes downstream of Snap Lake, and completing bathymetric coverage in Lac Capot Blanc.

Results of the initial 2011 and 2012 downstream monitoring indicated the plume was observed throughout the first two downstream lakes, and near the inlet of Lac Capot Blanc. Field conductivity approached background concentrations approximately 6 km downstream of Snap Lake in 2011 and 2012. In the Environmental Assessment Report (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 (De Beers 2002).



LEGEND

- AEMP Water Quality Station
- Snap Lake Mine Footprint
- Monitoring Area
- 2013 Downstream Lakes Special Study Area
- Waterbody
- Watercourse
- Flow Direction

NOTE

DSL1 First lake downstream of Snap Lake

DSL2 Second lake downstream of Snap Lake

REFERENCE

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

SNAP LAKE MINE

Downstream Lakes Overview, 2013

PROJECTION: UTM Zone 12

DATUM: NAD83

4 0 4

SCALE 1:200,000 KILOMETRES

DE BEERS GROUP OF COMPANIES

FILE No: 13134900016200A11.2-1

DATE:

JOB No: 14-1349-0004

REVISION No: A

OFFICE: GOLDER VICTORIA

DRAWN: JEF

CHECK: PMC

Figure 11.3-1

L:\2013\1349\13-1349-0001\Phase 6200\Task 6220\13134900016200A11.2-1.dwg Apr 16, 2014 – 11:18am

Results of the 2013 Downstream Lakes Special Study are provided herein (Section 11.3.5). This information will be used to refine future AEMP sampling locations downstream of Snap Lake and inform future prediction updates.

11.3.1.2 Objectives

The objectives of the 2013 Downstream Lakes Special Study were primarily to document the extent of the treated effluent downstream of Snap Lake relative to the EAR predictions and characterize sediment and water quality in the first three lakes downstream of Snap Lake. Analyses of the 2013 Downstream Lakes Special Study data addressed the following two key questions:

- Key Question 1: What is the spatial extent of the treated effluent plume downstream of Snap Lake (i.e., plume delineation)?
- Key Question 2: What are the current water and sediment quality characteristics in the three downstream lakes?

In addition to answering the key questions, bathymetric coverage in Lac Capot Blanc was expanded and downstream water quality predictions were updated as per recommendations in the 2012 AEMP report and 2013 AEMP Design Plan (De Beers 2013a, 2014).

11.3.1.3 Monitoring Study Area

Three lakes located immediately downstream of Snap Lake were surveyed during the 2013 Downstream Lakes Special Study, based on evidence of treated effluent in these lakes in 2011 and 2012 (De Beers 2012, 2013a). 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 11.3A, Photos 11.3A-1 and 11.3A-2) 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 11.3-1).

11.3.2 Field and Laboratory Methods

As per the 2013 AEMP Design Plan, field programs were conducted in the downstream lakes during winter, summer, and fall 2013 (De Beers 2014). The program is outlined in Table 11.3-1; a brief overview is provided below, followed by detailed methods for each component in Sections 11.3.2.1 to 11.3.2.6.

Supporting information, including additional bathymetric transects and continuous temperature logger data were collected. Instantaneous field measurements (i.e., temperature, dissolved oxygen [DO], pH, and specific conductivity [herein referred to as conductivity]), continuous conductivity data (i.e., installation of data sondes), and water quality samples were collected in support of Key Question 1. Field measurements and water and sediment samples were collected in support of Key Question 2.

Table 11.3-1 Sampling Program for the 2013 Downstream Lakes Special Study

Lake	Station	UTM Coordinates (NAD 83; Zone 12)[(a)]		Depth (m)	Winter	Open Water	
		Easting	Northing		May	July[(b)]	September
DSL1	Inlet DSL1	512309	7054239	0.3	Ice thickness No field measurements collected as site was frozen to bottom	Surface field measurements	Surface field measurements
	DSL1-1	513403	7054940	14.5	Ice thickness Water column field profile Mid-depth water sample	Water column field profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample; temperature logger (deep)	Same as July + sediment sample
	DSL1-2	512373	7054285	3.6	-	-	Sediment sample
	DSL1-3	513881	7054380	7.7	-	-	Sediment sample
	Outlet DSL1	514220	7054061	0.3	Ice thickness No field measurements collected as site was frozen to bottom	Surface field measurements	Surface field measurements
DSL2	Inlet DSL2	514813	7053637	0.3	Ice thickness No field measurements collected as site was frozen to bottom	Surface field measurements	Surface field measurements
	DSL2-1	515197	7053418	6.8	Ice thickness Water column field profile Mid-depth water sample	Water column field profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample; temperature logger (deep)	Same as July + sediment sample
	DSL2-2	514809	7053529	2.7	-	-	Sediment sample
	DSL2-3	515573	7053581	3.0	-	-	Sediment sample
	Outlet DSL2	515841	7053640	0.3	-[(c)]	Surface field measurements	Surface field measurements
Lac Capot Blanc	Inlet 1 LCB	516297	7053611	0.5	Ice thickness No field measurements collected as site was frozen to bottom	Surface field measurements Surface grab water sample Installed conductivity data sonde	Surface field measurements Surface grab water sample Removed conductivity data sonde
	LCB-C1[(d)]	516715	7054019	6.0	-	-	Field measurements (surface, bottom)
	LCB-C3[(d)]	516803	7053666	8.0	-	-	Field measurements (surface, bottom)
	LCB-C5[(d)]	516890	7053313	4.0	-	-	Field measurements (surface, bottom)
	LCB-G1[(d)]	518204	7053546	10.0	-	-	Field measurements (surface, bottom)
	LCB-G2[(d)]	518004	7053671	10.0	-	-	Field measurements (surface, bottom)
	LCB-G3[(d)]	517336	7053803	4.0	-	-	Field measurements (surface, bottom)
	LCB-1	518411	7053352	13.9	Ice thickness Water column field profile Mid-depth water sample	Water column profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample; temperature logger (deep)	Same as July + sediment sample
	LCB-2 LCB-2A[(e)]	523699	7053890	12.5	Ice thickness Water column field profile Mid-depth water sample (LCB-2)[(d)]	Water column profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample	Same as July + sediment sample
	LCB-3	525105	7052028	11.0	Ice thickness Water column field profile Mid-depth water sample	Water column profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample	Same as July + sediment sample
	LCB-4	522634	7049277	12.0	-	Water column profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample	Same as July + sediment sample
	LCB-5	519151	7053987	23.4	Dissolved oxygen profile	Water column profile	Water column profile
	LCB-6	520657	7055515	9.5	Ice thickness Water column field profile Mid-depth water sample	-	-

Table 11.3-1 Sampling Program for the 2013 Downstream Lakes Special Study

Lake	Station	UTM Coordinates (NAD 83; Zone 12)[a]		Depth (m)	Winter	Open Water	
		Easting	Northing		May	July[b]	September
Lac Capot Blanc (continued)	LCB-6*	520723	7055445	0.3	-	Surface field measurements Surface water sample	Surface field measurements Surface water sample
	LCB-7	520796	7054145	12.7	-	Water column profile Mid-depth water sample Depth-integrated nutrient and chlorophyll sample	Same as July + sediment sample
	Outlet 1 LCB	519241	7055428	0.5	Ice thickness Surface field measurements	Surface field measurements Installed conductivity data sonde	Surface field measurements Removed conductivity data sonde
	Outlet 2/ 2a LCB [f]	520743	7055292	0.5	Ice thickness Surface field measurements	Surface field measurements Installed conductivity data sonde	Surface field measurements Removed conductivity data sonde

a) Universal Transverse Mercator (UTM) Coordinates; North American Datum (NAD) 83; Zone 12V.

b) The 2013 AEMP Design Plan specified that only field measurements be collected at lake outlets in July (De Beers 2014); however, mid-depth water samples were collected and analyzed for conventional parameters, major ions, and nitrogen nutrients to support on-going modelling efforts.

c) Outlet DSL2 could not be located during the winter program under the snow. It was assumed to be frozen to bottom based on conditions of other inlets and outlets in DSL1 and DSL2.

d) Stations were sampled to identify the leading edge of the plume in Lac Capot Blanc during open-water conditions.

e) LCB-2 was moved 200 m northeast and re-named to LCB-2A in July 2013 to provide adequate depth for sediment sampling.

f) Two separate channels, split by an island, were identified in Outlet 2 of Lac Capot Blanc in July 2013 (Appendix 11.3A, Photos 11.3A-9 and 11.3A-10). The channels were referred to as Outlet 2a LCB and Outlet 2b LCB; the latter was too shallow to sample.

LCB-6* = time constraints and logistical issues forced crew to sample LCB-6 from shore, rather than mid-channel.

- = no samples collected; m = metre; DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc.

11.3.2.1 Bathymetry

Sufficient bathymetric coverage was achieved during the 2011 and 2012 surveys; therefore, no additional data were collected for DSL1 and DSL2. Additional transects were required from Lac Capot Blanc to update the existing map, with a focus on the northeast basin and southern sections of the west and east basins (Appendix 11.3B). Bathymetry transects in Lac Capot Blanc were completed in a grid fashion August 16 to 18, 2013, using a Garmin sonar coupled with a global positioning system (GPS) unit (sonar/GPS).

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 (i.e., approximately 50 to 100 metres [m] apart) 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.

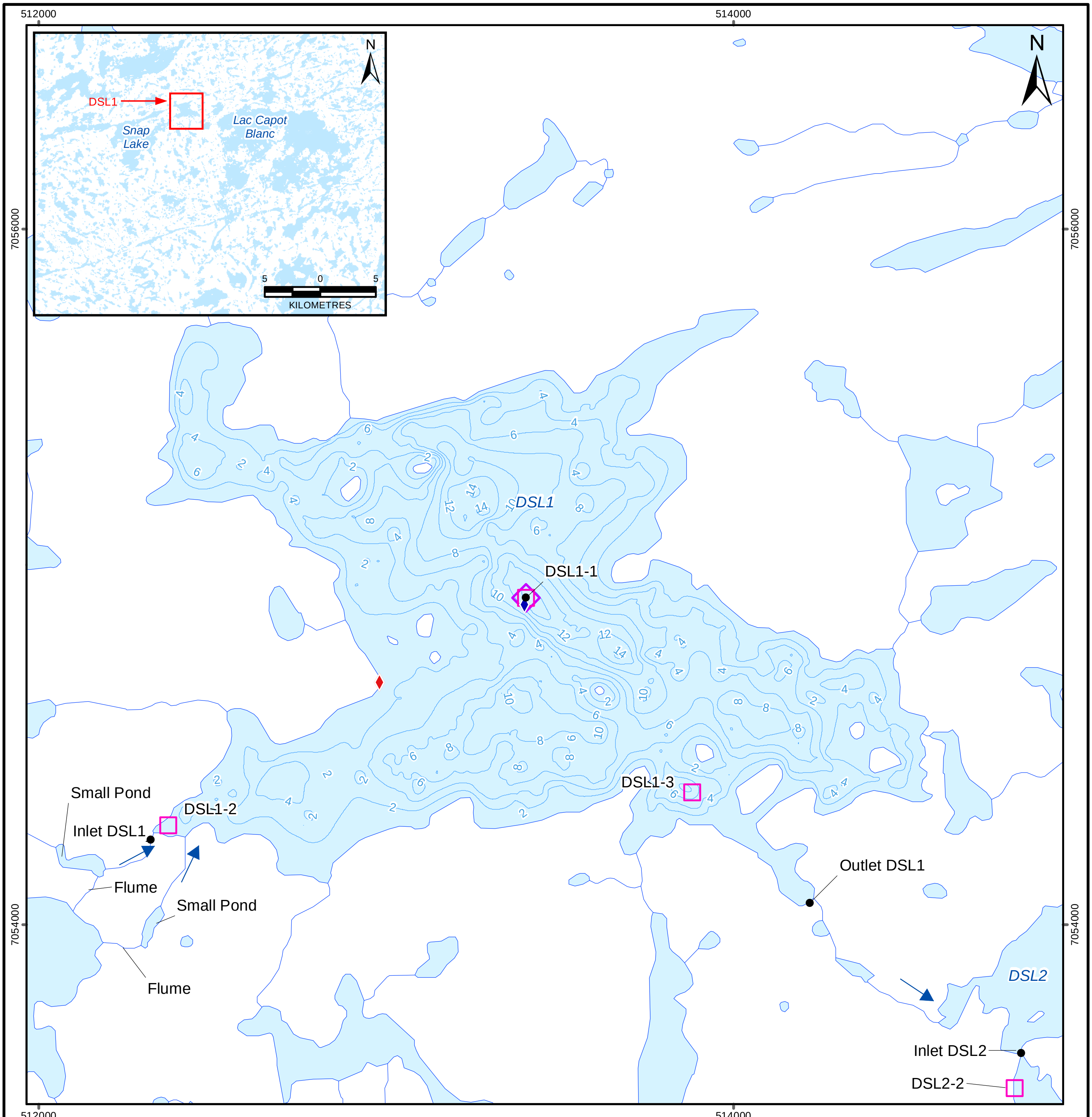
11.3.2.2 Supporting Environmental Variables

The following supporting environmental information was recorded:

- sampling date and time;
- weather conditions (air temperature, wind velocity, and wind direction);
- GPS coordinates recorded as Universal Transverse Mercator (UTM);
- water depth;
- Secchi depth;
- vertical profiles of water temperature, DO, pH, and conductivity, measured at discrete intervals (Section 11.3.2.3); and,
- water temperature data using three temperature loggers (Onset TidbiT Water Temperature Loggers – UTBI-001).

Temperature loggers were installed in the three downstream lakes in July and removed in September 2013 (July 12 to September 9, 2013 for DSL1 and DSL2; July 14 to September 9, 2013 for Lac Capot Blanc). The temperature loggers were programmed to record water temperature hourly.

One shallow site location (i.e., less than 1 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 m above the bottom substrate. The locations of the temperature loggers are shown in Figures 11.3-2 to 11.3-4.



LEGEND

- Plankton Sample Station
- Sediment Quality Sample Station
- Water Quality Sample Station
- Contour (2 m)
- Flow Direction
- Watercourse
- Waterbody

Temperature Logger

- Deep
- Shallow

NOTES

DSL1: First lake downstream of Snap Lake
DSL2: Second lake downstream of Snap Lake

REFERENCES

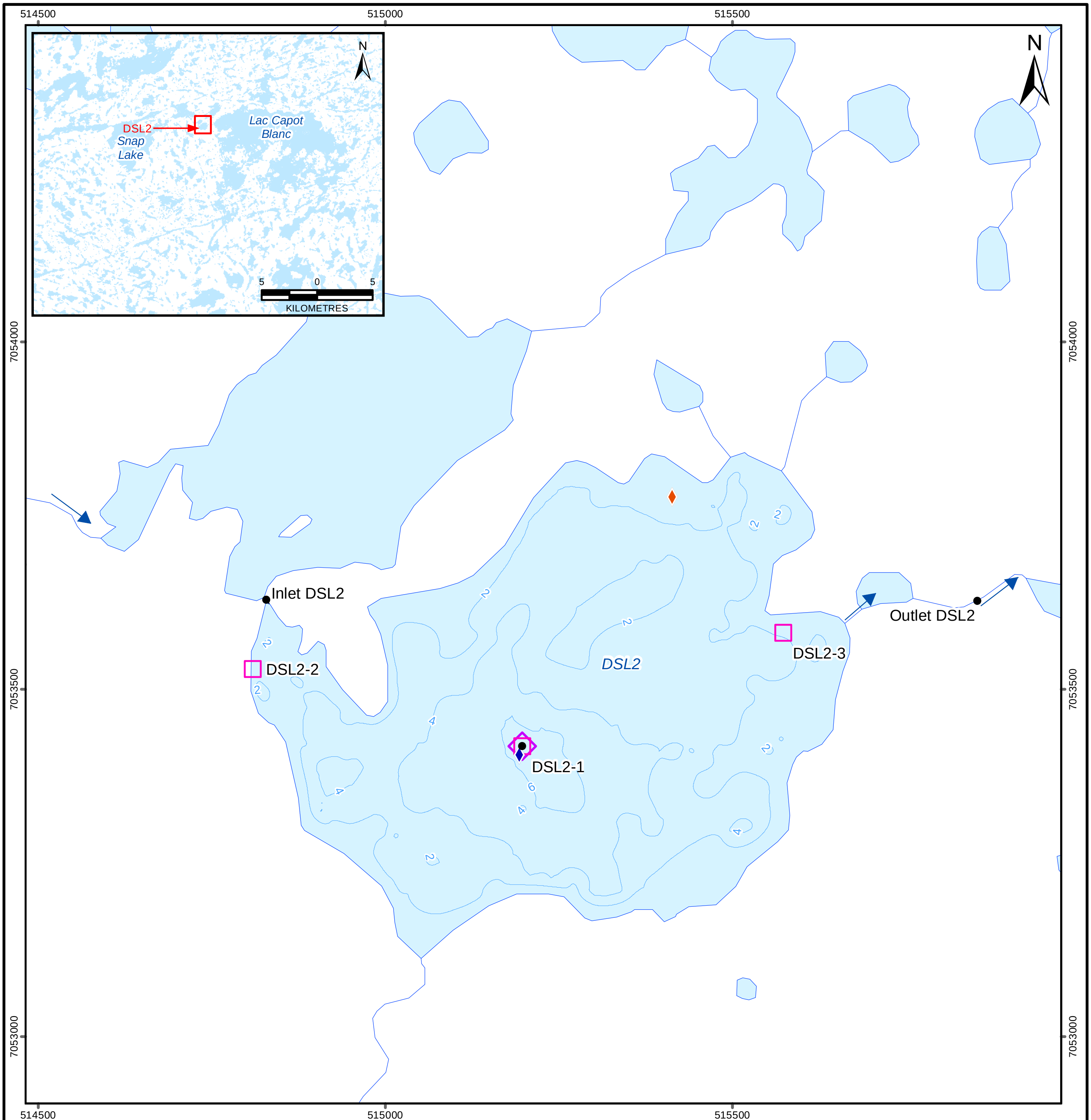
Base Data Source: CanVec

SNAP LAKE MINE

2013 Downstream Lakes Special Study, Downstream Lake 1

PROJECTION: UTM Zone 12		DATUM: NAD83	DE BEERS GROUP OF COMPANIES
350 0 350 SCALE 1:16,000 METRES			
FILE No: 1413490004-7001_11.2-2			DATE: March 28, 2014
JOB NO: 14-1349-0004	REVISION NO: A		Figure **11.3-2**
OFFICE: GOLD-CAL	DRAWN: JC	CHECK: PMC	

I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Fig11.3-2_DSL1_Sampling_FINAL.mxd



I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Fig11.3-3_DSL2_Sampling_FINAL.mxd

LEGEND

Plankton Sample Station	
Sediment Quality Sample Station	
Water Quality Sample Station	
Contour (2 m)	
Flow Direction	
Watercourse	
Waterbody	

Temperature Logger

- Deep
- Shallow

NOTES

DSL2: Second lake downstream of Snap Lake

REFERENCES

Base Data Source: CanVec

SNAP LAKE MINE

2013 Downstream Lakes Special Study, Downstream Lake 2

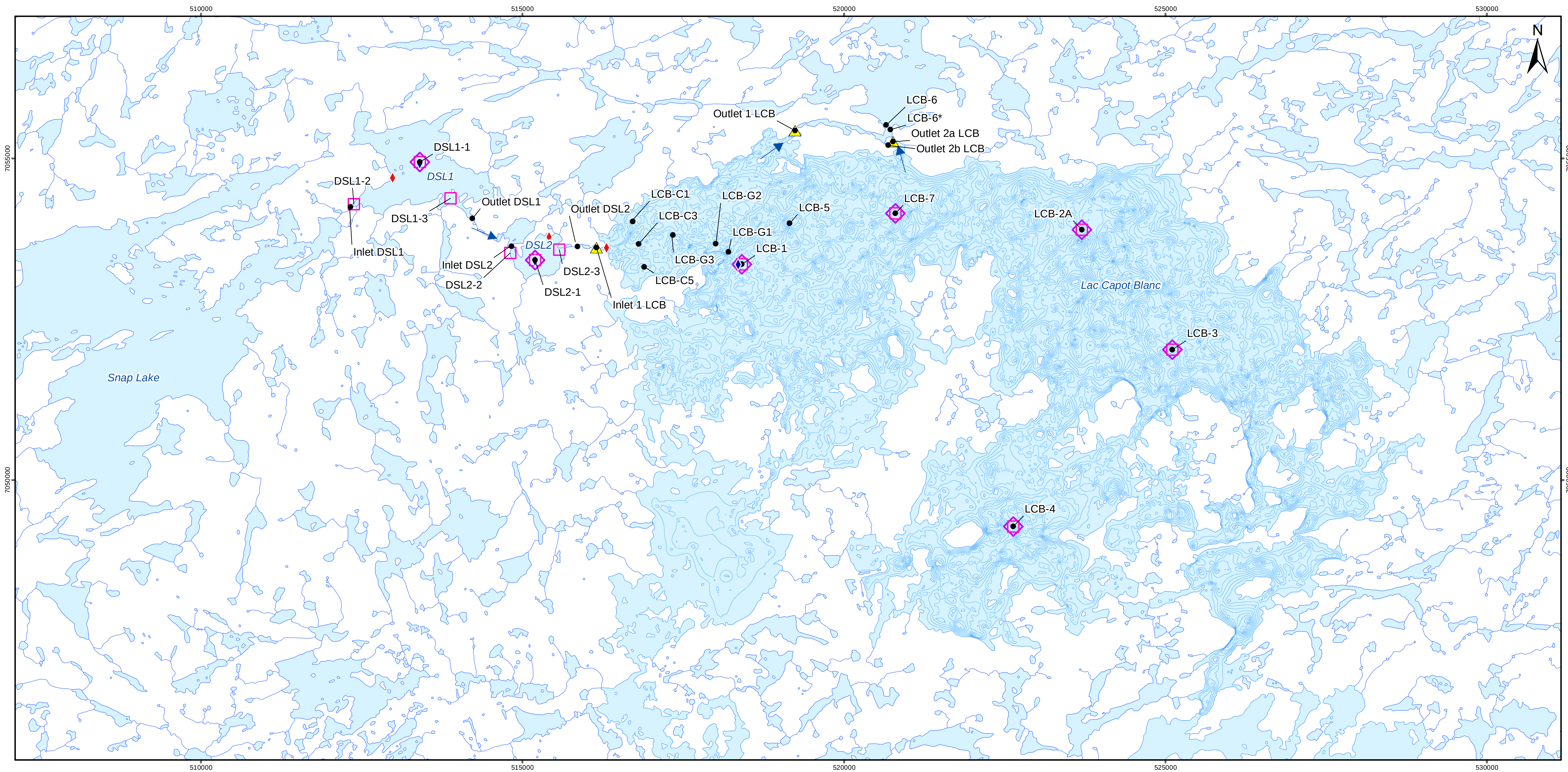
PROJECTION:	DATUM:
UTM Zone 12	NAD83

175 0 175

SCALE 1:8,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No:			DATE:
1413490004-7001_11.2-3			March 28, 2014
JOB NO: 14-1349-0004	REVISION NO: A		Figure 11.3-3
OFFICE: GOLD-CAL	DRAWN: AW	CHECK: PMC	



LEGEND

- Contour (2 m)
- Flow Direction
- Watercourse
- Waterbody
- Conductivity Logger
- Plankton Sample Station
- Sediment Quality Sample Station
- Water Quality Sample Station

Temperature Logger

- Deep
- Shallow

REFERENCES

Base data source: CanVec

NOTES

DSL1: First lake downstream of Snap Lake
DSL2: Second lake downstream of Snap Lake

SNAP LAKE MINE

2013 Downstream Lakes Special Study, Lac Capot Blanc

PROJECTION: UTM Zone 12	DATUM: NAD83		DE BEERS GROUP OF COMPANIES
1,000 0 1,000 SCALE 1:60,000 METRES			
FILE No: 1313490007-7001_11.2-4			DATE: March 28, 2014
JOB NO: 13-1349-0007	REVISION NO: A		Figure 11.3-4
OFFICE: GOLD-CAL	DRAWN: VV	CHECK: PMC	

I:\CLIENTS\DE_BEERS\Snap_Lake\Mapping\MXD\AEMP2013\Fig11.3-4_DSL3_Sampling_FINAL.mxd

11.3.2.3 Water Quality Field Measurements

To collect continuous, real-time conductivity measurements on an hourly basis, data sondes (HOBO Conductivity Data Sonde – U24-001) were installed at Inlet 1 Lac Capot Blanc and Outlets 1 and 2a of Lac Capot Blanc on July 14, 2013 (Appendix 11.3A, Photo 11.3A-3; Figure 11.3-4). On August 15, 2013, the conductivity data sondes were retrieved, downloaded, and redeployed; the units were removed between September 10 and 12, 2013 in advance of winter ice formation.

Instantaneous spot 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. Surface field measurements were collected at the inlet and outlet of DSL1 and DSL2 and the inlet and outlet channels of Lac Capot Blanc.

Water column profile data (i.e., readings taken at multiple depths) were collected at one station in DSL1 and DSL2 and six stations in Lac Capot Blanc (Table 11.3-1; Figures 11.3-2 to 11.3-4). A 30-m cable was connected to the YSI meter for depth profiles. Methods used to collect spot field measurements and water column profiles were:

- for inlet and outlet stations, spot field measurements were collected just below the surface of the water column (i.e., 0.1 to 0.3 m below surface if depth permitted), winter spot field measurements were collected 0.1 to 0.3 m below the bottom of the ice if depth permitted;
- 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 m within the water column.

A modified profile method (i.e., measuring conductivity 0.3 m below surface and 0.3 m above bottom) was completed at stations along a transect from the inlet of Lac Capot Blanc at 250 m, 500 m, and 1.5 km to delineate the leading edge of the treated effluent plume (Table 11.3-1; Figure 11.3-4).

Light penetration 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.

11.3.2.4 Water Sample Collection

Water samples were collected at nine stations, including one mid-depth sample in DSL1 and DSL2, a surface grab sample from the inlet of Lac Capot Blanc, and six mid-depth samples from Lac Capot Blanc (Table 11.3-1). The surface grab water sample was collected from the middle of the watercourse at 0.3 m below the water surface. The mid-depth water samples were collected using a Kemmerer water sampler, after taking profile measurements.

During the ice-covered 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. Water from the Teflon and polyvinyl chloride (PVC) Kemmerer samplers were poured into individual 4-litre (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.

During open-water season, 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-L biochemical oxygen demand (BOD) bottle from either the Teflon or PVC Kemmerer sampler for transport back to the De Beers water processing facility. Sample bottles from Maxxam Analytics Inc. (Maxxam) and Flett Research Ltd. (Flett) were triple-rinsed with sample water before filling, with the exception of glass bottles. Sample bottles from ALS Canada Ltd. (ALS) were not rinsed before filling as per their instructions. 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 the appropriate samples after filtering. Water samples collected in May and September were submitted to the appropriate analytical laboratories for analysis of conventional parameters, major ions, nutrients, and total metals and metalloids, consistent with the AEMP parameter suite (De Beers 2012). Samples collected in July were analyzed for conventional parameters, major ions, and nitrogen nutrients only. The 2013 AEMP Design Plan specified that only field measurements be collected at lake outlets in July (De Beers 2014); however, to support on-going modelling efforts in the downstream lakes, water samples were collected at various locations in July, but were analyzed for a reduced parameter suite.

The QA and QC procedures for the 2013 Downstream Lakes Special Study were consistent with the protocols provided in the AEMP (Appendix 3A) and De Beers QA/QC Plan (De Beers 2008). In total, nine QC samples were collected between May and September 2013: one travel blank; two field blanks; three equipment blanks; one set of split samples; and, two sets of duplicate samples (Appendix 11.3C, Table 11.3C-1). Further details on QA/QC sampling, scheduling and handling are provided in Appendix 11.3C.

11.3.2.5 Depth-Integrated Nutrient and Chlorophyll a Collection

Depth-integrated total phosphorus (TP), total nitrogen (TN), and chlorophyll *a* and *c* samples were collected within the euphotic zone, which is the depth of water in a lake that is exposed to sufficient sunlight for photosynthesis, at one station in DSL1, one station in DSL2, and five stations in Lac Capot Blanc (Table 11.3-1). A single composite water sample was collected at each station using a Kemmerer water sampler. Discrete water samples were collected at 2 m intervals within the top 6-m (i.e., surface, 2, 4, and 6 m), to maintain consistency with sampling techniques employed in Snap Lake. Equal volumes of water from each depth were combined in a clean plastic bucket and mixed to create a homogeneous composite sample. A 250 millilitre (mL) sub-sample was collected for TP and TN analysis in a clear Nalgene bottle. Samples were frozen and shipped to the University of Alberta Biogeochemical Analytical Service Laboratory (UofA) in Edmonton, Alberta, where analyses were completed.

The remaining composite sample water was used to fill a 1-L amber Nalgene bottle for filtration for chlorophyll *a* samples. Duplicate samples were collected at each station. Approximately 500 mL of water per sample was filtered onto a 47 millimetre (mm) glass fibre (GF/C) filter using a glass filter tower and vacuum pump. Each filter was removed using forceps, folded in half, wrapped in aluminum foil, and frozen. Chlorophyll *a* samples were shipped to UofA for analysis.

11.3.2.6 Sediment Sample Collection

Sediment samples were collected at three stations in each of DSL1 and DSL2, and five stations in Lac Capot Blanc (Table 11.3-1). Within DSL1 and DSL2, sampling stations were located as close as possible to the inlet and outlet of each lake[1] while still being able to sample fine-grained depositional material, and near the approximate middle of each lake. Lac Capot Blanc stations were located in areas where water depth was in the 10 to 15 m range. Sediment samples were collected after plankton sampling, water quality vertical profiles, and water quality sampling were completed at each station, so that disturbed sediments would not adversely affect any water column sampling. Procedures for sample collection, which were the same as those used for routine AEMP sediment sampling in Snap Lake and the reference lakes, are summarized below. Sediment samples were collected using an Ekman grab.

Three sediment grabs 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 for each station. 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-1 in Lac Capot Blanc). Two 250 mL glass jars were filled from the composite sample for nutrients, carbon, and total metals analyses. A 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 ALS for analyses of the routine AEMP suite of sediment chemistry parameters.

11.3.3 Data Analysis Methods

Supporting Information – Bathymetry and Temperature Monitoring

Bathymetry data (i.e., depth and GPS positions) recorded by the Garmin sonar were downloaded and stored electronically. Those files were then transcribed onto a bathymetric contour map using geographic information system (GIS) software. Temperature data recorded by the loggers were downloaded using onset HoboWare Pro software, exported to Microsoft Excel, then plotted to identify trends.

[1] It was originally intended that sediment samples would be collected at the same inlet and outlet stations used for water quality monitoring, but those locations were too shallow and of unsuitable substrate; therefore, the sediment stations were re-located to areas of suitable substrate. The field sampling records identified these inlet and outlet stations by the same names used for water quality sampling (but at different coordinates), but they were subsequently renamed as follows: Inlet DSL1 became DSL1-2; Outlet DSL1 became DSL1-3; Inlet DSL2 became DSL2-2; and, Outlet DSL2 became DSL2-3.

Key Question 1: What is the spatial extent of the treated effluent plume downstream of Snap Lake (i.e., plume delineation)?

Field measurements of conductivity were primarily used to map the spatial patterns of the treated effluent plume downstream of Snap Lake. The extent of the plume was assessed by plotting:

- Water quality data with distance downstream, and visually examining the data to identify the location of the plume. Figures showing the plume as a snap-shot in time were prepared to show the spatial patterns in water quality. For those figures, conductivity between sampling stations (i.e., inlet tributaries, in-lake stations, outlet tributaries) was estimated using an inverse distance weighted method of interpolation, in a GIS figure similar to that used in Snap Lake (Section 3).
- Vertical profiles to investigate the portion of the water column in each downstream lake potentially influenced by treated effluent.
- Continuous, real-time conductivity measurements from the data sondes to compare measurements among locations and identify changes over the open-water season.

Key Question 2: What are the current water and sediment quality characteristics in the three downstream lakes?

Water quality data collected from DSL1, DSL2, and Lac Capot Blanc in each season were compared to baseline concentrations in Snap Lake (i.e., Snap Lake normal range; Section 3) and AEMP benchmarks applicable to Snap Lake and downstream lakes, which refers to a collective list of generic water quality guidelines (WQGs) (i.e., CCME 1999 with updates through 2013) and EAR benchmarks (De Beers 2002). Where possible, data were reviewed to identify potential changes for stations sampled over multiple years.

Sediment quality data for each lake were summarized separately in terms of the whole-lake mean, minimum, and maximum for each parameter. Sediment quality data were compared to the interim sediment quality guidelines (ISQGs) and Probable Effect Levels (PELs) developed by Canadian Council of Ministers of the Environment (CCME) (1999 with updates through 2013) for protection of freshwater aquatic life. The CCME ISQGs and PELs are currently available for seven metals analyzed for the Snap Lake AEMP (Section 11.3.5.2). 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. Mean sediment parameter concentrations for each lake were also compared to Snap Lake normal ranges, which were calculated as the mean plus or minus (±) 2 standard deviations (SD) from baseline sediment quality data in Snap Lake (see Section 4).

11.3.4 Quality Assurance and Quality Control

The QA/QC quality control procedures for the 2013 Downstream Lakes Special Study (i.e., field methods, laboratory analyses, data management and analyses, and reporting) were consistent with the protocols provided in the AEMP (Appendix 3A) and De Beers QA/QC Plan (De Beers 2008). 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.

The field measurements and analytical results for the water sampling programs completed as part of 2013 Downstream Lakes Special Study were validated separately from the remainder of the AEMP QA/QC analysis, but followed similar procedures. Details of QA/QC procedures, and results for QC samples collected as part of 2013 Downstream Lakes Special Study, are provided in Appendix 11.3C.

11.3.5 Results

11.3.5.1 Supporting Information

Bathymetry

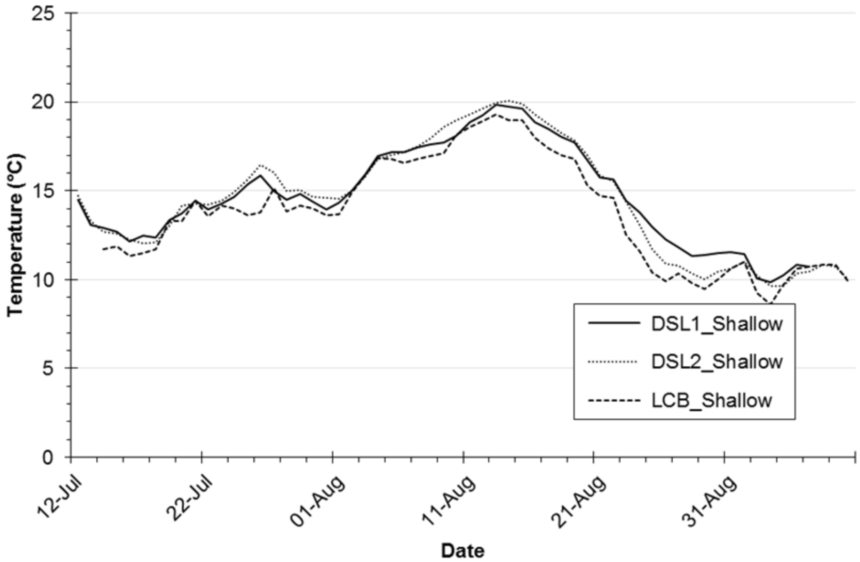
The bathymetric update at Lac Capot Blanc was completed on the northeastern basin and southern areas immediately adjacent to the west and east basins (Appendix 11.3B, Figure 11.3B-1). Depth of near-shore waters ranged from 0 to 2 m. The majority of the lake had an average water depth of 4 to 8 m. A few scattered areas had water depths in the 18 to 22 m range with the deepest area located in the middle of the east basin, at 44 to 46 m.

Water Temperature Monitoring

Water temperature data collected during the open-water season from the temperature loggers installed in DSL1, DSL2, and Lac Capot Blanc are presented in Figures 11.3-5 (shallow sites) and 11.3-6 (deep sites, surface and bottom). The shallow temperature logger for Lac Capot Blanc was installed July 14, 2013. The water temperature recorded in Lac Capot Blanc was cooler than in DSL1 and DSL2 for the duration of the monitoring period. The warmest peak for all three lakes occurred in mid-August, followed by a cooling trend over the remainder of the season.

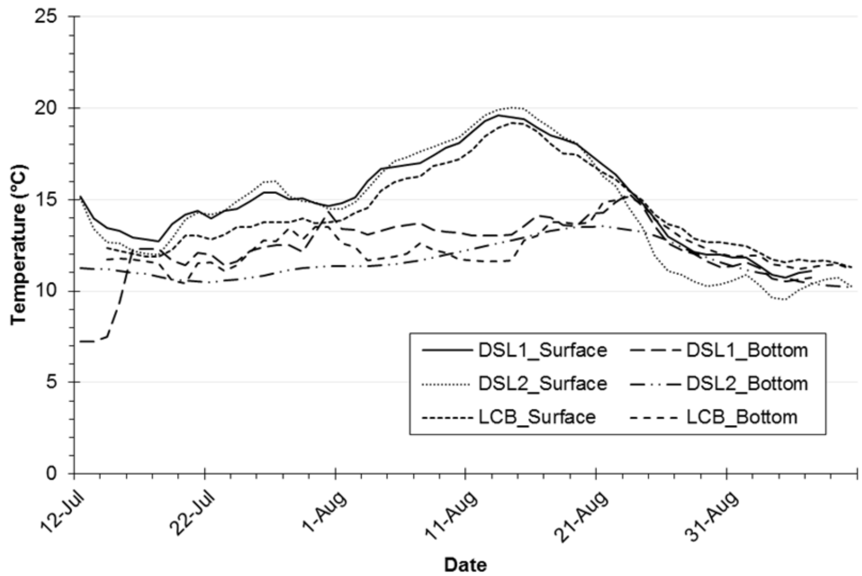
Surface water temperatures for the deep sample sites followed a similar pattern as the shallow locations, with water temperatures in Lac Capot Blanc being slightly cooler than in DSL1 and DSL2, and the warmest temperatures occurring in mid-August with a general cooling over the remainder of the season (Figure 11.3-6). Maximum temperatures at the deep stations occurred later in the season (i.e., end of August) compared to the surface and shallow locations. The deep temperature logger in DSL2 showed a very linear temperature increase with few peaks or valleys; temperatures ranged from approximately 10 degrees Celsius (ºC) to 14ºC. Water temperatures at the deep locations in DSL1 and Lac Capot Blanc were relatively stable, with a slight cooling off toward the end of the August (Figure 11.3-6).

Figure 11.3-5 Water Temperature at Shallow Sample Sites, July to September 2013



Note: total depth was less than 1.0 m; loggers were set at 0.5 m depth.
DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; °C = degrees Celsius; m = metre.

Figure 11.3-6 Water Temperature at Deep Sample Sites, July to September 2013



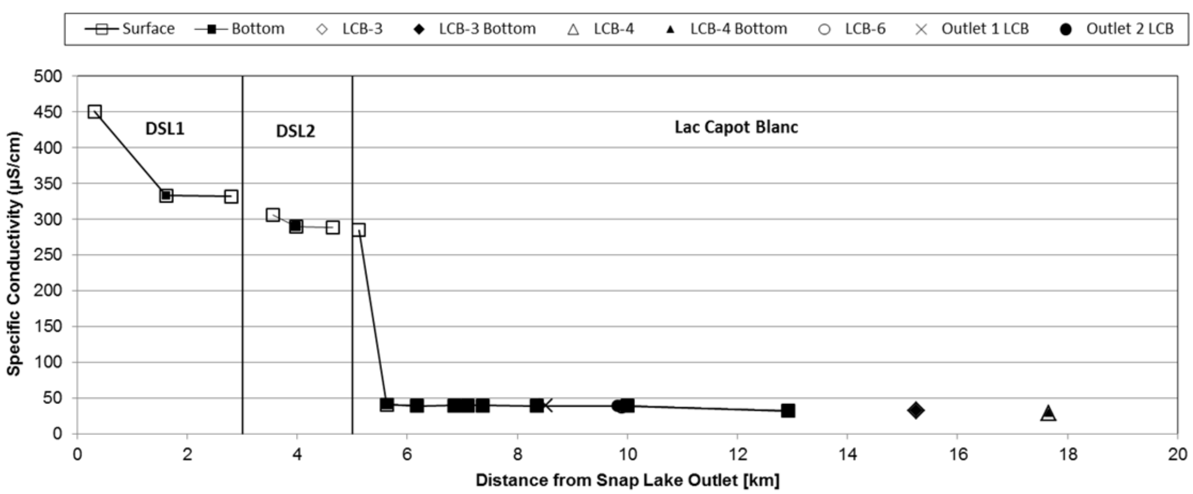
Notes: Surface logger depth was 0.3 m below water surface; deep logger was located at 1.0 m above bottom substrate. Lac Capot Blanc surface logger depth was set at 1.0 m below surface.
DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; °C = degrees Celsius; m = metre.

11.3.5.2 Key Question 1: What is the spatial extent of the treated effluent plume downstream of Snap Lake (i.e., plume delineation)?

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 the September sampling program in 2013. Emphasis was placed on conductivity, an indirect electrical measurement for the Mine-related constituents including total dissolved solids (TDS), nitrate, and major ions. Field conductivity measurements were compared with those measured in other reference areas, such as Northeast Lake, where conductivity has been consistently below 30 microSiemens per centimetre (µS/cm). As well, sampling stations located in Lac Capot Blanc, farthest from Snap Lake (i.e., LCB-3 and LCB-4), were lower than or comparable to 30 µS/cm and, therefore, considered background or reference values (Figure 11.3-7). Conductivity values above 30 µS/cm were assumed to be influenced to some degree by treated effluent exposure.

Figure 11.3-7 Field Conductivity Downstream of Snap Lake, September 2013

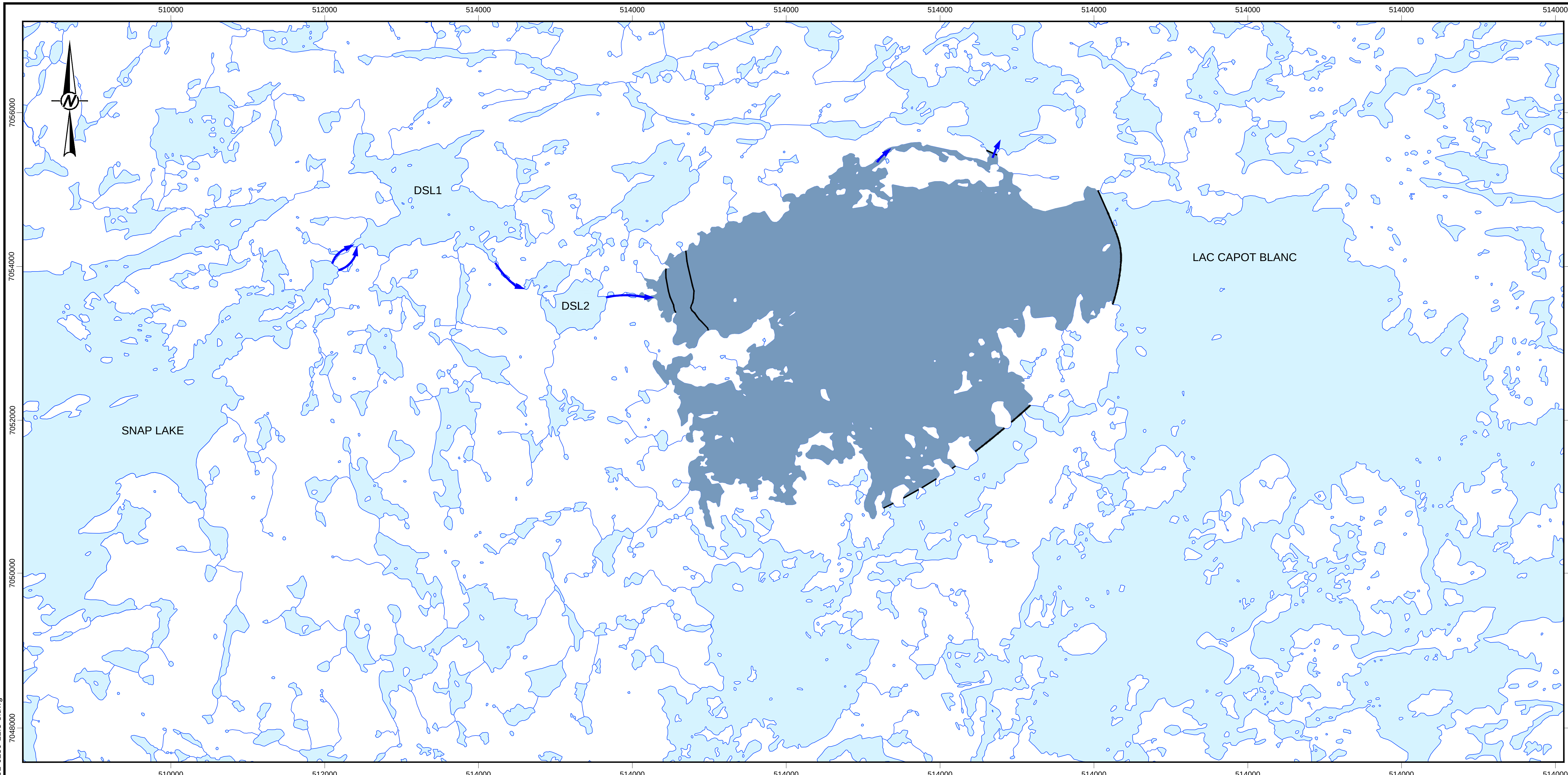


DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; µS/cm = microSiemens per centimetre; km = kilometre.

Field conductivity measurements at sampling stations Inlet DSL1, Inlet DSL2, and Inlet 1 LCB were 451, 306, and 285 µS/cm, respectively, in September 2013 (Figure 11.3-7). Conductivity notably decreased from 285 µS/cm to 41 µS/cm between station Inlet 1 LCB and LCB-C3, which is located approximately 650 m from the inlet of Lac Capot Blanc (Figure 11.3-7). Conductivity gradually decreased to background at LCB-2A (approximately 5 km east from the inlet of Lac Capot Blanc), and was consistent at the farthest Lac Capot Blanc stations, LCB-3 and LCB-4 (32 and 29 µS/cm, respectively). Conductivity at LCB-6, which is located immediately north (downstream) of the Lac Capot Blanc outlets (Figure 11.3-4), was 38 µS/cm in September 2013.

Results indicate that the treated effluent extends to a larger area of Lac Capot Blanc in 2013 compared to 2011 and 2012 (Figure 11.3-8). The field conductivity decreased to background levels within 50 m, 650 m, and 5 km of the inlet of Lac Capot Blanc in 2011, 2012, and 2013, respectively (De Beers 2012, 2013a). Treated effluent was evident, in total, approximately 11 km downstream of Snap Lake in 2013 (into Lac Capot Blanc). In the EAR (De Beers 2002), parameter concentrations associated with the treated effluent discharge were conservatively predicted to reach near background concentrations 44 km downstream of Snap Lake by the end of operations, assuming maximum concentrations during operations.

Vertical patterns were assessed using field measurements collected at the water column profile stations in DSL1, DSL2, and Lac Capot Blanc during the 2013 Downstream Lakes Special Study (Figure 11.3-9 and Appendix 11.3D, Table 11.3D-1). Field conductivity measurements were elevated near the bottom of the water column at DSL1-1 in May and July 2013 and at DSL2-1 in May 2013. The results indicate that the denser treated effluent tends to sink to the bottom of the water column in DSL1 and DSL2 during the ice-covered season, and then vertically mixes throughout the water column during open-water when wind-driven mixing occurs (Figure 11.3-9). A similar trend was observed in Lac Capot Blanc at the station located closest to the inlet (LCB-1) during the ice-covered season. However, once mixing occurred during the open-water season, conductivity was relatively well-mixed throughout the water column, with higher concentrations at stations located closer to the inlet (LCB-1, LCB-5, and LCB-7); stations farther from the inlet were at or below background concentrations (Figure 11.3-9).



L:\2013\1349\13-1349-0001\Phase 6200\Task 6230\1313490001-6230-11.3-8.dwg

LEGEND

- Watercourse
- Flow Path
- Waterbody
- Approximate Spatial Extent of Treated Effluent Plume Delineated by Year (2011, 2012, 2013)

REFERENCES

Base Vector Digital CANVEC 1:50K Data obtained from Geogratis online É 1992

Datum: NAD83 Projection: UTM Zone 12.

NOTES

DSL1: First lake downstream of Snap Lake
DSL2: Second lake downstream of Snap Lake

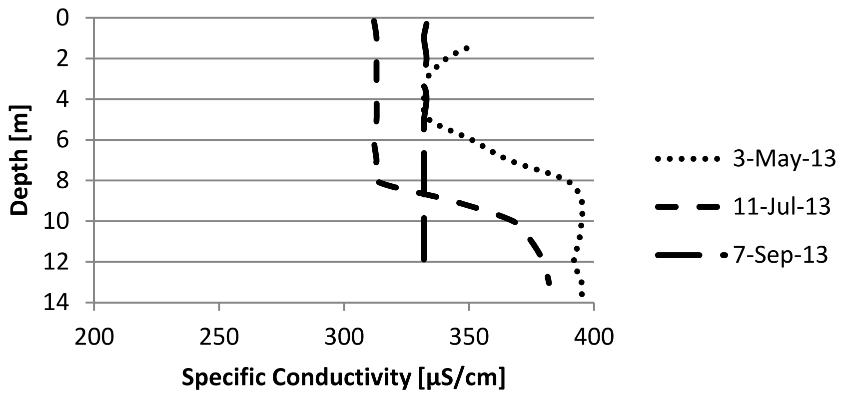
SNAP LAKE MINE

Zone of Treated Effluent Influence in 2013

PROJECTION: UTM Zone 12	DATUM: NAD83		DE BEERS GROUP OF COMPANIES
500 0 500 SCALE 1:30,000 METRES			
FILE No: 1313490001-6230-11.3-8			DATE: December 5, 2013
JOB No: 14-1349-0004	REVISION No: A		Figure 11.3-8
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

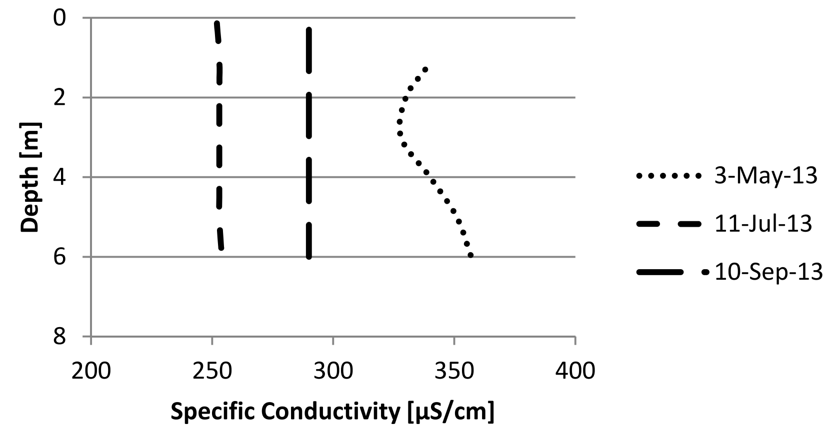
Figure 11.3-9 Vertical Profile Measurements from the Downstream Lakes, 2013

a) Downstream Lake 1 (DSL1), station DSL1-1



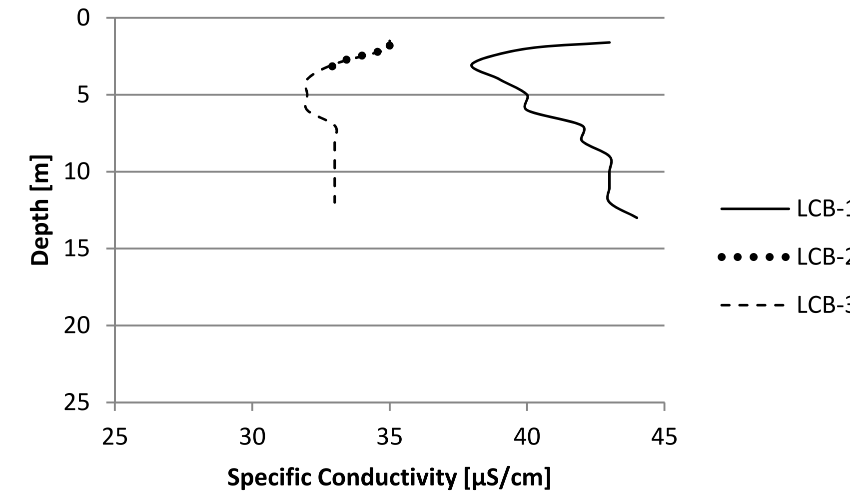
m = metre; µS/cm = microSiemens per centimetre; Jul = July; Sep = September
DSL1 = Downstream Lake 1

b) Downstream Lake 2 (DSL2), station DSL2-1



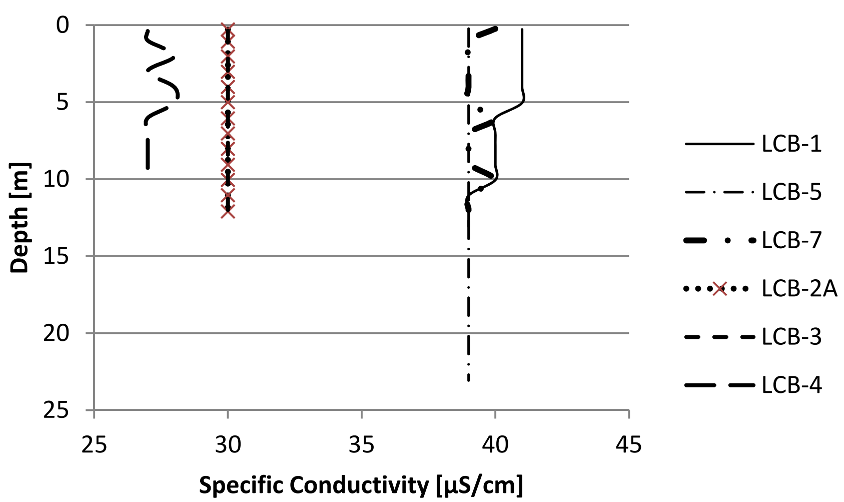
m = metre; µS/cm = microSiemens per centimetre
DSL2 = Downstream Lake 2

c) Lac Capot Blanc, May 2013



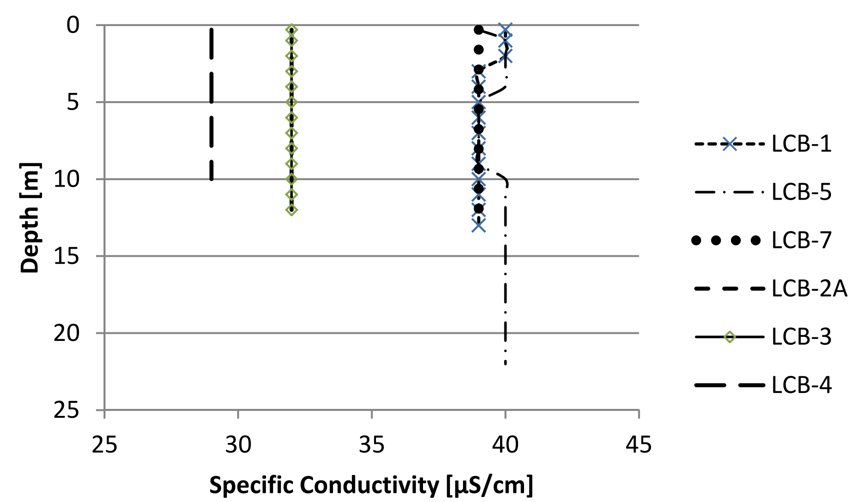
Note: Stations are plotted in order of increasing distance from the inlet of Lac Capot Blanc.
m = metre; µS/cm = microSiemens per centimetre; LCB = Lac Capot Blanc

d) Lac Capot Blanc, July 2013



Note: Stations are plotted in order of increasing distance from the inlet of Lac Capot Blanc.
Data from Stations LCB-2A and LCB-3 overlap.
m = metre; µS/cm = microSiemens per centimetre; LCB = Lac Capot Blanc.

e) Lac Capot Blanc, September 2013

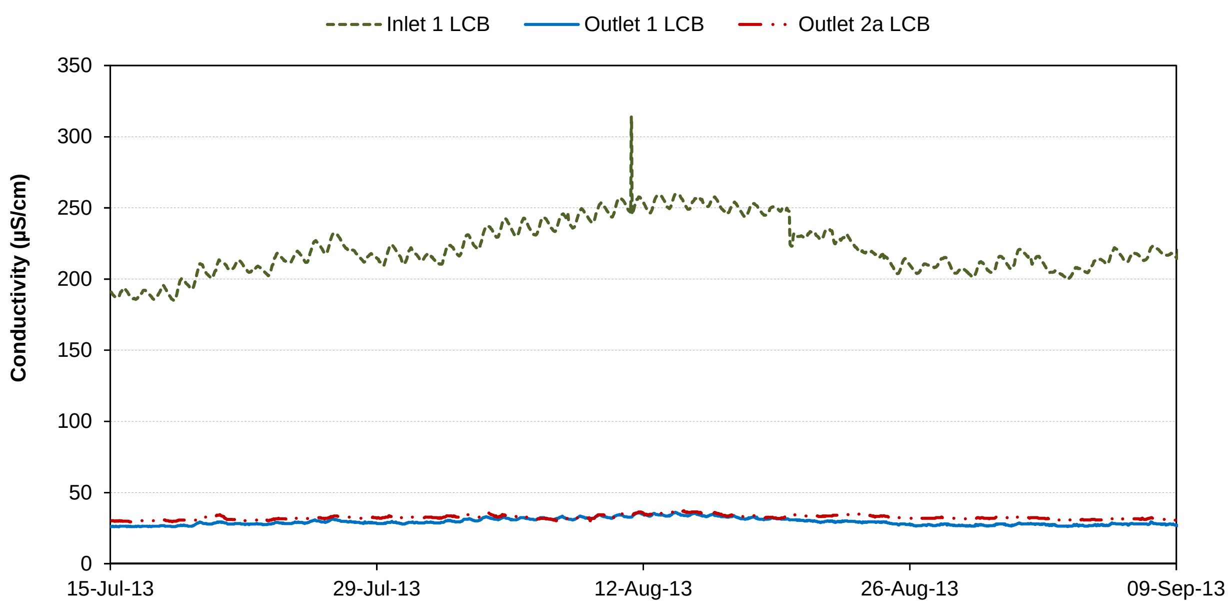


Note: Stations are plotted in order of increasing distance from the inlet of Lac Capot Blanc.
Data from Stations LCB-2A and LCB-3 overlap, as well as data from LCB-1 and LCB-7.
m = metre; µS/cm = microSiemens per centimetre; LCB = Lac Capot Blanc.

Continuous Conductivity Data

Conductivity measurements recorded by the data sondes installed at the inlet and two outlets of Lac Capot Blanc support that the treated effluent plume is evident at the inlet of Lac Capot Blanc, but mixes rapidly prior to reaching the outlets (Figure 11.3-10). Average conductivity at the inlet (Inlet 1 LCB) over the open-water period was 221 µS/cm, compared to 29 µS/cm and 32 µS/cm at Outlet 1 LCB and Outlet 2a LCB, respectively. Maximum conductivity measurements occurred in mid-August at all three locations, corresponding with the time-frame when maximum temperature values were observed (Figures 11.3-5 and 11.3-6).

Figure 11.3-10 Continuous Conductivity Measurements, 2013



Note: The source of the conductivity spike (i.e., conductivity >300 µS/cm) in August at Inlet 1 LCB is unknown.
LCB = Lac Capot Blanc; µS/cm = microSiemens per centimetre; > = greater than.

Degree of Change between 2012 and 2013

Field conductivity measurements at the inlet and the outlet of each lake were higher in 2013 compared to 2012. The percent change from 2012 to 2013 ranged from 16 percent (%) to 22% (Table 11.3-2), which was comparable to the percent change in 2012 from 2011 (De Beers 2013a).

Table 11.3-2 Change in Conductivity from 2012 to 2013

Lake	Station	Conductivity [µS/cm]		Percentage Increase (2012 to 2013)[b]
		2012	2013[a]	
DSL1	Inlet DSL1	377	451	20%
	Outlet DSL1	273	332	22%
DSL2	Inlet DSL2	263	306	16%
	Outlet DSL2	241	289	20%
Lac Capot Blanc	Inlet 1 LCB	240	285	19%
	Outlet 2a LCB	33	39	18%

a) Instantaneous field conductivity measurements from September were compared.

b) The percentage increase was calculated: ([2013 field conductivity – 2012 field conductivity] / [2012 field conductivity]) x 100.

DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; µS/cm = microSiemens per centimetre; % = percent.

11.3.5.3 Key Question 2: What are the current water and sediment quality characteristics in the three downstream lakes?

Water Quality

Results of the analyses performed on the 2013 downstream lakes water samples are reported in Appendix 11.3E, along with information for AEMP benchmark comparisons. Table 11.3-3 provides the range observed in each downstream lake (minimum and maximum concentrations) as well as comparisons to Snap Lake baseline normal ranges for each analyte. Quality control results are provided in Appendix 11.3C.

Quality Control Summary

The QC results from the 2013 downstream lakes program indicated that:

- The relative percent differences (RPD) between duplicate samples were generally within 20% for most parameters (Appendix 11.3C, Tables 11.3C-2 and 11.3C-3). Within-site variability and field sampling precision was rated as low and high, respectively in May with the notable differences in 2% of the total number of parameters analyzed. In July, within-site variability and field sampling precision were both rated as moderate with notable differences in 21% of the total number of parameters analyzed in the duplicate samples.
- The results of split samples from ALS and Maxxam were generally comparable (Appendix 11.3C, Table 11.3C-4). The analytical precision was rated as moderate in 2013 with the notable differences (i.e., RPD values greater than 20%) in 11% of the total number of parameters analyzed in the split sample.
- Blank results were generally reported below the detection limits (DL), or less than 10% of the minimum lake concentrations in 2013 with exception of six parameters (Appendix 11.3C, Table 11.3C-5 and Table 11.3C-6). For three parameters, total copper, total organic phosphorus (calculated), and total phosphorus, the potential contamination was isolated to the blank samples. Concentrations of aluminum and antimony in the field blank, and boron in the travel blank, were detectable and greater than 10% of the minimum lake concentrations in 2013, which is consistent

with the results reported in the comprehensive AEMP QA/QC assessment in 2013 (Appendix 3A). Potential contamination of these parameters may have occurred and data for those parameters should be interpreted with this limitation in mind. Further information on the overall blank contamination, as it relates to the entire water quality component of the AEMP, is provided in Appendix 3A.

Overall, the water quality data collected during the 2013 Downstream Lakes Special Study is considered to be of acceptable quality and adequate to address the objectives of the program.

Water Quality Results

Similar to field conductivity, concentrations of Mine-related constituents including TDS, nitrate, and major ions were higher in DSL1, DSL2, and Inlet 1 LCB compared to those measured at most stations of Lac Capot Blanc in 2013 (Appendix 11.3E, Table 11.3E-1). 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, lithium, molybdenum, nickel, rubidium, strontium, and uranium, which are also characteristic of the treated effluent (Appendix 11.3E, Table 11.3E-1).

Water quality data collected from the downstream lakes were compared to AEMP benchmarks, which refers to a list of generic WQGs (e.g., CCME 1999 with updates) and EAR benchmarks (De Beers 2002). Most of the parameters measured at the downstream lakes in 2013 were below the AEMP benchmarks, with the exception of field pH, fluoride, and lead (Appendix 11.3E, Table 11.3E-1). Field pH concentrations were lower than the optimal range for aquatic life (pH 6.5 to 9.0) at LCB-1, DSL2, and LCB-6 in May 2013. Fluoride concentrations were above the AEMP benchmark[2] (0.12 mg/L) at DSL1-1 in July and at DSL2-1 in July and September. More than half of the samples (i.e., 63%) collected in Snap Lake in 2013 were higher than the fluoride AEMP benchmark. Therefore, elevated fluoride concentrations in the downstream lakes were likely due to the discharge of treated effluent, which contains fluoride from groundwater sources. However, the increase of fluoride is associated with elevated calcium and hardness, which are expected to reduce the potential for toxicity effects associated with fluoride.

The total lead concentration at LCB-2A was above the AEMP benchmark in September 2013. This sample was re-tested and the total lead result was confirmed by the laboratory. As a follow-up, the dissolved lead concentration was analyzed, and the result was below the DL, indicating the majority of the lead was associated with suspended material in the sample. The total lead concentrations were generally below the DLs in DSL1, DSL2, and in all other samples collected from Lac Capot Blanc. Therefore, the AEMP benchmark exceedance of total lead concentration may be attributed to an anomalous result in that sample, rather than a Mine-related effect.

[2] As described in Section 3.4.3, a site-specific water quality objective (SSWQO) of 2.46 mg/L was proposed for fluoride (De Beers 2013b).

Maximum pH, TDS, major ions, nitrogen parameters, and eight metals (i.e., barium, boron, lithium, lead, molybdenum, rubidium, strontium, and uranium) were above the normal range of Snap Lake (mean concentration ± 2 SD) in DSL1, DSL2, and Lac Capot Blanc (Table 11.3-3). With the exception of total lead, those parameters have also been identified as increasing in Snap Lake since the Mine started discharging in 2004 (Section 3.4.4), indicating that treated effluent exposure is likely the main contributor to the elevated concentrations. The maximum total lead concentration may be attributed to an anomalous result in that sample, rather than a Mine-related effect.

Table 11.3-3 Water Quality Summary for Downstream Lakes, 2013

Field Parameters	Units	Downstream Lake 1 (DSL1)		Downstream Lake 2 (DSL2)		Lac Capot Blanc (LCB)		Snap Lake Normal Range (Mean ± 2 SD)[a]	Comparison to Snap Lake Normal Ranges
		min	max	min	max	min	max		
Conventional Parameters									
Laboratory pH	-	7.4	7.5	7.4	7.4	6.8	7.4	6.3 to 6.9	Above
Total Dissolved Solids, calculated (Lab) [b]	mg/L	150	187	120	184	13	138	6 to 21	Above
Turbidity-Unfiltered	NTU	0.19	0.47	0.37	0.49	0.22	0.41	0.1 to 1.1	Within
Major Ions									
Bicarbonate, as HCO_3	mg/L	19	27	17	31	7	18	4 to 12	Above
Calcium	mg/L	28	39	22	38	2	28	0.7 to 2.1	Above
Chloride	mg/L	72	87	58	84	3	65	0.2 to 1.3	Above
Fluoride	mg/L	0.10	**0.13**	0.11	**0.13**	0.06	0.12	0.03 to 0.06	Above
Hardness, as $CaCO_3$	mg/L	85	118	68	116	9	87	2.5 to 9.5	Above
Magnesium	mg/L	3.7	4.9	3.1	5.3	0.7	3.9	0.3 to 1.0	Above
Potassium	mg/L	1.4	1.8	1.2	2.1	0.4	1.4	0.1 to 0.8	Above
Reactive Silica, as SiO_2	mg/L	0.5	1.5	0.5	1.8	0.1	0.9	0.2 to 0.7	Above
Sodium	mg/L	15	20	12	21	1	16	0.3 to 0.9	Above
Sulphate	mg/L	13	16	10	14	1	12	0.8 to 4.8	Above
Total Alkalinity, as $CaCO_3$	mg/L	16	22	14	25	5	15	2.8 to 9.6	Above
Nutrients and Carbons									
Nitrate, as N, calc'd	mg-N/L	0.83	0.95	0.47	0.49	<0.006	0.49	0.006 to 0.054	Above
Nitrate/Nitrite, as N	mg-N/L	0.83	0.95	0.48	0.49	<0.006	0.49	0.006 to 0.046	Above
Nitrite, as N	mg-N/L	<0.002	0.005	0.003	0.003	<0.002	<0.002	0.002 to 0.002	Above
ortho-Phosphate, as P	mg-P/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.001 to 0.005	Within
Total Ammonia, as N	mg-N/L	0.013	0.048	0.034	0.254	<0.005	0.016	0.002 to 0.06	Above
Total Dissolved Phosphorus	mg-P/L	<0.001	<0.001	<0.001	0.0012	<0.001	0.0011	0.001 to 0.014	Within
Total Kjeldahl Nitrogen	mg-N/L	0.17	0.24	0.12	0.30	<0.05	0.17	0.05 to 0.66	Within
Total Organic Carbon	mg/L	3.0	3.2	3.1	3.3	2.6	4.0	1 to 5.7	Within
Total Phosphorus	mg-P/L	0.003	0.004	0.002	0.003	0.001	0.004	0.001 to 0.018	Within
Total Metals and Metalloids									
Aluminum	µg/L	1.1	4.2	2.0	6.5	1.0	5.8	3.5 to 14	Within
Antimony	µg/L	0.03	0.35	0.03	0.19	<0.02	0.70	0.03 to 0.83	Within
Arsenic	µg/L	0.09	0.11	0.10	0.11	0.06	0.09	0.02 to 0.29	Within
Barium	µg/L	14	22	13	24	3	12	0.92 to 4.72	Above
Beryllium	µg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.1	Within
Bismuth	µg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.03	Within
Boron	µg/L	30	40	26	37	2.7	26	1 to 4.8	Above

Table 11.3-3 Water Quality Summary for Downstream Lakes, 2013

Field Parameters	Units	Downstream Lake 1 (DSL1)		Downstream Lake 2 (DSL2)		Lac Capot Blanc (LCB)		Snap Lake Normal Range (Mean ± 2 SD)[a]	Comparison to Snap Lake Normal Ranges
		min	max	min	max	min	max		
Cadmium	µg/L	<0.005	<0.005	<0.005	<0.005	<0.005	0.007	0.05	Within
Cesium	µg/L	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.1	Within
Chromium	µg/L	<0.06	<0.06	<0.06	<0.06	<0.06	0.373	0.06 to 0.81	Within
Cobalt	µg/L	0.01	0.02	<0.01	0.01	<0.01	0.01	0.04 to 0.19	Within
Copper	µg/L	0.37	0.41	0.29	0.39	<0.1	0.41	0.5 to 2.8	Within
Iron	µg/L	3	17	3	16	2	15	5.7 to 52	Within
Lead	µg/L	<0.01	<0.01	<0.01	<0.01	<0.01	**1.2**	0.05 to 0.79	Above (LCB)
Lithium	µg/L	5.6	8.7	4.9	7.9	0.7	4.8	0.1 to 3.3	Above
Manganese	µg/L	2.7	5.5	2.9	3.5	1.5	5.4	2 to 9.7	Within
Mercury (Flett)	µg/L	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	0.0012	0.01	Within
Molybdenum	µg/L	0.43	0.54	0.21	0.35	<0.05	0.33	0.06 to 0.14	Above
Nickel	µg/L	0.23	0.41	0.14	0.31	<0.06	0.28	0.08 to 1.2	Within
Rubidium	µg/L	3	3.7	2.8	3.8	1.1	2.7	0.4 to 1.9	Above
Selenium	µg/L	<0.04	<0.04	<0.04	<0.04	<0.04	<0.04	0.1	Within
Silver	µg/L	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	0. 1	Within
Strontium	µg/L	429	556	345	511	20	344	4.1 to 13	Above
Thallium	µg/L	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.03	Within
Titanium	µg/L	<0.1	<0.1	<0.1	0.4	<0.1	0.2	0.1 to 0.5	Within
Uranium	µg/L	0.10	0.29	0.18	0.29	0.04	0.27	0.05	Above
Vanadium	µg/L	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	0.05 to 0.31	Within
Zinc	µg/L	1.5	2.2	1.2	1.5	<0.8	2.6	0.8 to 5.3	Within

Note: **Bold** values are above AEMP benchmarks (refer to Appendix 11.3E for detailed results). Maximum and minimum values and concentrations in DSL 1, DSL 2 and Lac Capot Blanc from the 2013 reporting period are presented.

a) Normal range is based on data collected prior to 2004 in Snap Lake, with the upper and lower range calculated as the mean concentration ± 2 SD. For parameters which were typically below the detection limits, the detection limit was used as the normal range.

b) Total dissolved solids calculated (lab) refers to laboratory-calculated total dissolved solids concentrations adapted from Methods 1030 E in Standard Methods for the Examination of Water and Wastewater, 21st Edition (APHA 2005).

AEMP= Aquatic Effects Monitoring Program; ± = plus or minus; min = minimum; max = maximum; SD = standard deviation; DSL 1 = Downstream Lake 1; DSL 2 = Downstream Lake 2; LCB = Lac Capot Blanc; HCO_3 = bicarbonate; $CaCO_3$ = calcium carbonate; SiO_2 = silicate; N = nitrogen; P = phosphorus; Flett = Flett Research Ltd.; < = less than; mg/L = milligrams per litre; NTU = nephelometric turbidity units; mg-P/L = milligrams as phosphorus per litre; mg-N/L = milligrams as nitrogen per litre; µg/L = micrograms per litre.

Secchi Depths, Depth-Integrated Nutrients, and Chlorophyll Concentrations

In the downstream lakes, Secchi depths ranged from 7.0 to 8.5 m in July and 6.5 to 10.2 m in September (Table 11.3-4). In DSL2 Secchi depths reached the maximum water column depth of 7.0 m, while in DSL1 and Lac Capot Blanc Secchi depths ranged from 6.5 to 10.0 m with maximum water column depths at sample stations ranging from 10.6 to 14.0 m.

Total nitrogen concentrations decreased with distance downstream from Snap Lake (Table 11.3-4). Little variation in TN concentrations was observed between July and September at each station. Mean concentrations of TN were highest in DSL1 (1 milligrams as nitrogen per litre [mg-N/L]), approximating concentrations observed in Snap Lake (2.22 mg-N/L in the main basin and 0.93 mg-N/L in the northwest arm), followed by DSL2 (0.74 mg-N/L), and Lac Capot Blanc (0.17 mg-N/L). The TN concentrations were similar among stations in Lac Capot Blanc and comparable to mean TN concentrations observed in Northeast Lake (0.18 mg-N/L) and Lake 13 (0.20 mg-N/L).

Total phosphorus concentrations were similar among the three downstream lakes during July and September (Table 11.3-4). In July all depth-integrated TP concentrations were below the DL of0.003 mg-P/L. In September, concentrations of TP ranged from 0.003 to 0.006 mg-P/L, similar to concentrations observed in Snap Lake (0.002 mg-P/L in the main basin and 0.004 mg-P/L in the northwest arm) and Northeast Lake (0.002 mg-P/L). The TP concentrations in Lake 13 were higher (0.008 mg-P/L) than those observed in the other lakes.

Table 11.3-4 Secchi Depth, Depth-Integrated Nutrients, and Chlorophyll *a* and *c* in the Downstream Lakes

Lake	Station	Date	Maximum Depth (m)	Secchi Depth (m)	Depth-Integrated			
					Total Nitrogen (mg-N/L)	Total Phosphorus (mg-P/L)	Chlorophyll *a* (µg/L)	Chlorophyll *c* (µg/L)
DSL1	DSL1-1	11-Jul-13	13.6	8.0	0.99	<0.003	1.08	0.42
		7-Sep-13	13.1	7.8	1.00	0.006	1.54	<0.004
DSL2	DSL2-1	11-Jul-13	7.0	Bottom	0.70	<0.003	1.18	0.01
		10-Sep-13	7.0	Bottom	0.79	0.003	1.49	0.06
Lac Capot Blanc	LCB-1	13-Jul-13	14.0	7.5	0.17	<0.003	1.59	0.16
		11-Sep-13	13.2	6.5	0.19	0.004	1.87	0.20
	LCB-2A	13-Jul-13	12.5	8.3	0.15	<0.003	0.92	0.06
		11-Sep-13	12.9	10.2	0.15	0.003	1.38	<0.004
	LCB-3	13-Jul-13	12.3	8.5	0.14	<0.003	0.75	0.02
		11-Sep-13	12.4	10.0	0.17	0.003	1.35	0.03
	LCB-4	13-Jul-13	10.6	8.3	0.17	<0.003	1.21	0.07
		11-Sep-13	10.8	8.0	0.20	0.004	1.77	<0.004
	LCB-7	13-Jul-13	12.7	8.0	0.14	<0.003	1.74	0.08
		11-Sep-13	12.4	8.2	0.18	0.004	1.55	<0.004

m = metre; mg-N/L = milligrams as nitrogen per litre; mg-P/L = milligrams as phosphorus per litre; µg/L = micrograms per litre; DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; > = greater than; < = less than.

Chlorophyll *a* concentrations among the three downstream lakes were similar (Table 11.3-4). Mean chlorophyll *a* concentrations in DSL1 (1.31 micrograms per litre [µg/L]), DSL2 (1.34 µg/L), and Lac Capot Blanc (1.43 µg/L) were higher than those observed in the main basin of Snap Lake (1.44 µg/L), and lower than those observed in the northwest arm of Snap Lake (2.26 µg/L). Chlorophyll *a* concentrations in the downstream lakes were similar to those observed in Northeast Lake (1.74 µg/L) and Lake 13 (1.49 µg/L).

Mean chlorophyll *c* concentrations were greatest in DSL1 (0.21 µg/L), followed by DSL2 (0.04 µg/L), and Lac Capot Blanc (0.06 µg/L; Table 11.3-4). Mean chlorophyll *c* concentrations in the downstream lakes were within the range observed in the main basin of Snap Lake (0.02 µg/L), and lower than those observed in the northwest arm (0.30 µg/L), but higher than those observed in Northeast Lake (0.01 µg/L) and Lake 13 (0.01 µg/L).

Sediment Quality

Results of the sediment chemistry analyses performed on the 2013 downstream lakes sediment samples are reported in Table 11.3-5, along with information for SQG comparisons. Table 11.3-6 provides the summary statistics for each downstream lake (mean, minimum, maximum concentrations), as well as comparisons to Snap Lake baseline normal ranges for each analyte. Complete results, including results for the field duplicate sample, are provided in Appendix 11.3F. All results are presented on a dry weight basis, except for moisture content.

Quality Control Summary

Holding times were met for all analyses, and none of the target analytes were detected in the method blanks. The specified DLs were met for all analytes except that the DLs for available ammonium, available nitrate, and available phosphate had to be increased in some samples because of sample matrix effects. The DL for available ammonium was increased from 1.0 to 1.6 milligrams per kilogram (mg/kg) nitrogen (N) for the DSL1-3 and LCB-3 samples; this may have affected the result for Station LCB-3, as available ammonium was undetected. The DL for available nitrate was increased from 4.0 to 6.0 mg/kg N for the LCB-1 field duplicate, LCB-4, and LCB-7 samples; this did not affect data quality as available nitrate was undetected in all sediment samples from the three downstream lakes.

The DL for available phosphate was increased from 2.0 to 4.0 mg/kg phosphorus (P) for the LCB-1, LCB-1 field duplicate, LCB-4, and LCB-7 samples; this may have affected the result for Station LCB-7 as available phosphate was undetected.

Results for laboratory duplicate analyses were within specified data quality objectives (DQOs), which were expressed as RPDs ranging from 20% to 40% depending on the analyte; RPDs for the laboratory duplicates were less than 10% for all analytes. Results for the field duplicate sample collected from the Lac Capot Blanc station (LCB-1) are provided in Appendix 11.3F. Relative percent differences (RPDs) for the original and field duplicate samples met the DQO of less than or equal to (≤) 20% RPD, except for three parameters; the RPDs for available ammonium, available potassium, and lithium were 34%, 26%, and 21%, respectively. Results for analyses of laboratory reference materials met the applicable DQOs (60% to 140% recovery, 70% to 130% recovery, or 80% to 120% recovery) for each analyte.

Table 11.3-5 Sediment Quality Guideline Comparison Results for Downstream Lakes, 2013

Lake Name	Units (dw)	Detection Limits	CCME SQGs		Downstream Lake 1 (DSL1)			Downstream Lake 2 (DSL2)			Lac Capot Blanc (LCB)				
Station ID					DSL1-2	DSL1-1	DSL1-3	DSL2-2	DSL2-1	DSL2-3	LCB-1	LCB-2A	LCB-3	LCB-4	LCB-7
Sample ID (Golder SCN)			ISQG	PEL	2013-9101	2013-9102	2013-9103	2013-9104	2013-9105	2013-9106	2013-9107	2013-9108	2013-9109	2013-9110	2013-9111
Physical															
Fines (Silt + Clay)	%	0.1	-	-	94.5	98.7	80.9	85.1	96.9	96.3	96.2	92.9	98.4	91.0	99.0
Total Organic Carbon	%	0.1	-	-	17.3	18.6	17.5	17.3	21.2	19.7	14.1	9.17	8.93	10.0	13.5
Nutrients															
Available Ammonium-N	mg/kg	1	-	-	4.9	1.8	3.4	1.6	1.8	4.0	1.2	1.5	<1.6	<1.0	1.6
Available Nitrate-N	mg/kg	4	-	-	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<4.0	<6.0	<6.0
Total Kjeldahl Nitrogen	mg/kg	0.02	-	-	1.65	1.50	1.38	1.39	1.51	1.56	1.05	0.701	0.663	0.735	0.992
Total Nitrogen	mg/kg	0.02	-	-	1.64	1.51	1.43	1.45	1.63	1.57	1.07	0.729	0.680	0.759	1.04
Available Phosphate-P	mg/kg	2	-	-	7.1	3.7	7.4	15.6	<2.0	11.2	20.5	4.1	7.2	13.0	<4.0
Available Potassium	mg/kg	20	-	-	221	151	140	142	99	132	82	71	73	86	96
Available Sulfate-S	mg/kg	3	-	-	646	564	246	250	170	113	43.6	27.5	37.1	48.1	77.7
Metals															
Aluminum	mg/kg	50	-	-	17,300	19,600	12,900	14,500	15,300	13,300	14,800	16,300	18,400	17,400	15,900
Antimony	mg/kg	0.10	-	-	0.14	0.13	<0.10	<0.10	<0.10	<0.10	0.11	0.11	<0.10	0.12	0.17
Arsenic	mg/kg	0.10	5.9	17	3.02	4.13	2.27	2.68	3.12	2.84	2.30	5.46	2.73	2.26	5.41
Barium	mg/kg	0.50	-	-	70.1	72.8	58.5	71.5	64.3	73.6	89.9	202	131	119	154
Beryllium	mg/kg	0.20	-	-	1.27	1.71	1.03	1.76	1.62	1.23	0.88	0.81	0.99	0.75	1.05
Bismuth	mg/kg	0.20	-	-	0.54	0.74	0.46	0.37	0.52	0.40	0.26	0.31	0.30	0.24	0.35
Boron	mg/kg	2.0	-	-	13.8	15.2	19.1	12.5	10.3	13.7	13.6	9.3	8.4	12.8	10.7
Cadmium	mg/kg	0.10	0.6	3.5	**0.66**	0.54	0.35	0.41	0.43	0.39	0.50	**0.66**	0.45	0.33	0.59
Calcium	mg/kg	100	-	-	9,630	5,790	4,780	6,220	6,350	4,890	3,390	2,830	2,620	2,750	3,230
Cesium	mg/kg	0.10	-	-	2.13	1.81	1.69	2.08	1.74	1.95	1.84	2.07	2.33	2.36	1.72
Chromium	mg/kg	0.50	37.3	90	28.3	31.3	24.0	23.9	26.7	23.5	31.0	36.4	**40.2**	**47.7**	30.4
Cobalt	mg/kg	0.10	-	-	10.8	26.1	8.02	7.10	15.9	8.72	6.15	13.4	8.97	8.86	15.9
Copper	mg/kg	0.50	35.7	197	**74.0**	**108**	**66.9**	**59.0**	**110**	**70.1**	**66.9**	**63.9**	**66.2**	**52.8**	**78.8**

Table 11.3-5 Sediment Quality Guideline Comparison Results for Downstream Lakes, 2013

Lake Name	Units (dw)	Detection Limits	CCME SQGs		Downstream Lake 1 (DSL1)			Downstream Lake 2 (DSL2)			Lac Capot Blanc (LCB)				
Station ID					DSL1-2	DSL1-1	DSL1-3	DSL2-2	DSL2-1	DSL2-3	LCB-1	LCB-2A	LCB-3	LCB-4	LCB-7
Sample ID (Golder SCN)			ISQG	PEL	2013-9101	2013-9102	2013-9103	2013-9104	2013-9105	2013-9106	2013-9107	2013-9108	2013-9109	2013-9110	2013-9111
Iron	mg/kg	50	-	-	31,500	64,100	19,100	16,000	46,700	15,800	17,500	38,300	28,700	26,400	65,000
Lead	mg/kg	0.50	35	91.3	7.05	7.27	6.49	11.6	9.14	8.42	7.69	8.03	7.08	7.01	10.3
Lithium	mg/kg	0.50	-	-	38.7	21.2	26.9	28.5	20.8	28.4	25.0	30.3	32.4	35.3	17.4
Magnesium	mg/kg	20	-	-	4,200	3,830	3,800	3,820	3,550	4,480	4,170	5,190	5,700	7,250	3,860
Manganese	mg/kg	1.0	-	-	241	369	173	167	148	158	210	2900	967	397	2550
Mercury	mg/kg	0.050	0.17	0.49	<0.050	0.064	<0.050	<0.050	<0.050	0.052	<0.050	<0.050	<0.050	<0.050	<0.050
Molybdenum	mg/kg	0.10	-	-	5.18	15.6	5.16	3.21	10.4	2.79	5.03	6.25	5.08	3.46	9.51
Nickel	mg/kg	0.50	-	-	39.9	37.0	29.3	32.5	40.8	34.5	24.5	33.2	31.5	30.5	30.9
Phosphorus	mg/kg	50	-	-	1,610	1,950	775	953	782	602	790	1,050	1,070	572	1,210
Potassium	mg/kg	50	-	-	1,580	1,540	1,490	1,440	1,210	1,470	1,740	2,300	2,380	3,350	1,620
Rubidium	mg/kg	1.0	-	-	15.8	12.7	13.2	14.8	11.9	14.2	14.4	17.0	18.5	22.6	12.9
Selenium	mg/kg	0.10	-	-	1.12	1.64	0.76	0.74	1.26	0.71	0.93	0.89	0.90	0.67	1.22
Silver	mg/kg	0.20	-	-	<0.20	0.24	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20
Sodium	mg/kg	100	-	-	370	450	370	370	480	390	160	150	160	160	150
Strontium	mg/kg	1.0	-	-	116	84.6	65.2	90.0	91.7	67.2	32.8	25.7	23.6	23.4	29.6
Thallium	mg/kg	0.050	-	-	0.163	0.157	0.131	0.138	0.126	0.134	0.142	0.375	0.222	0.202	0.274
Tin	mg/kg	2.0	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
Titanium	mg/kg	1.0	-	-	297	253	287	227	272	373	244	336	324	550	215
Uranium	mg/kg	0.050	-	-	17.4	27.2	17.9	34.9	41.8	25.5	30.8	29.3	35.6	24.6	37.2
Vanadium	mg/kg	0.20	-	-	26.4	32.7	25.2	22.9	26.3	23.6	29.0	34.2	38.6	43.5	31.2
Zinc	mg/kg	5.0	123	315	110	**153**	104	118	**135**	119	88.4	**124**	92.6	77.3	**140**

Note: **Bold** values are above the CCME ISQG.

CCME = Canadian Council of Ministers of the Environment; SGQ = sediment quality guideline; ISQG = interim sediment quality guideline; SCN= sample control number; PEL = Probable Effect Level; DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; - not applicable; % = percent; mg/kg = milligrams per kilogram; < = less than the detection limit; wt = weight; Golder = Golder Associates Ltd; ID= identification number.

Table 11.3-6 Sediment Quality Summary for Downstream Lakes, 2013

Station ID	Units (dry wt)	Downstream Lake 1 (DSL1)			Downstream Lake 2 (DSL2)			Lac Capot Blanc (LCB)			Snap Lake Normal Range (Mean ± 2SD)	Comparison to Snap Lake Normal Ranges
		min	mean	max	min	mean	max	min	mean	max		
Physical												
Fines (Silt + Clay)	%	80.9	91.4	98.7	85.1	92.8	96.9	91.0	95.5	99.0	87.3 to 100.3	Within
Total Organic Carbon	%	17.3	17.8	18.6	17.3	19.4	21.2	08.9	11.1	14.1	9.9 to 29.1	Within
Nutrients												
Available Ammonium-N	mg/kg	1.80	3.37	4.90	1.60	2.47	4.00	0.50	1.12	1.60	13.9 to 87.3	Below (DSL1, DSL2, and LCB)
Available Nitrate-N	mg/kg	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.4	3.0	0 to 68.8	Within
Total Kjeldahl Nitrogen	mg/kg	1.38	1.51	1.65	1.39	1.49	1.56	0.66	0.83	1.05	0.70 to 2.17	Within
Total Nitrogen	mg/kg	1.43	1.53	1.64	1.45	1.55	1.63	0.68	0.86	1.07	0.85 to 2.21	Within
Available Phosphate-P	mg/kg	3.7	6.1	7.4	1.0	9.3	15.6	2.00	9.36	20.5	0 to 38.8	Within
Available Potassium	mg/kg	140	171	221	99	124	142	71	82	96	27.7 to 156	Above (DSL1)
Available Sulfate-S	mg/kg	246	485	646	113	178	250	28	47	78	0 to 233	Above (DSL1)
Metals												
Aluminum	mg/kg	12,900	16,600	19,600	13,300	14,367	15,300	14,800	16,560	18,400	8,539 to 21,326	Within
Antimony	mg/kg	0.05	0.11	0.14	0.05	0.05	0.05	0.05	0.11	0.17	0.10 to 0.10	Above (DSL1 and LCB); Below (DSL2)
Arsenic	mg/kg	2.27	3.14	4.13	2.68	2.88	3.12	2.26	3.63	5.46	1.24 to 4.41	Within
Barium	mg/kg	59	67	73	64	70	74	90	139	202	0 to 834	Within
Beryllium	mg/kg	1.03	1.34	1.71	1.23	1.54	1.76	0.75	0.90	1.05	0.51 to 1.44	Above (DSL2)
Bismuth	mg/kg	0.46	0.58	0.74	0.37	0.43	0.52	0.24	0.29	0.35	0.40 to 0.65	Below (LCB)
Boron	mg/kg	13.8	16.03	19.10	10.30	12.17	13.70	8.40	10.96	13.60	2.8 to 23.4	Within
Cadmium	mg/kg	0.35	0.52	0.66	0.39	0.41	0.43	0.33	0.51	0.66	0.34 to 1.05	Within
Calcium	mg/kg	4,780	6,733	9,630	4,890	5,820	6,350	2,620	2,964	3,390	2,924 to 5,510	Above (DSL1 and DSL2)
Cesium	mg/kg	1.69	1.88	2.13	1.74	1.92	2.08	1.72	2.06	2.36	0.48 to 3.29	Within
Chromium	mg/kg	24.0	27.87	31.30	23.50	24.70	26.70	30.40	37.14	47.70	17.6 to 55.0	Within
Cobalt	mg/kg	8.0	15.0	26.1	7.1	10.6	15.9	6.2	10.7	15.9	6.6 to 16.6	Within
Copper	mg/kg	67	83	108	59	80	110	53	66	79	75 to 124	Below (LCB)
Iron	mg/kg	19,100	38,233	64,100	15,800	26,167	46,700	17,500	35,180	65,000	4,874 to 44,426	Within

Table 11.3-6 Sediment Quality Summary for Downstream Lakes, 2013

Station ID	Units (dry wt)	Downstream Lake 1 (DSL1)			Downstream Lake 2 (DSL2)			Lac Capot Blanc (LCB)			Snap Lake Normal Range (Mean ± 2SD)	Comparison to Snap Lake Normal Ranges
		min	mean	max	min	mean	max	min	mean	max		
Lead	mg/kg	6.5	6.9	7.3	8.4	9.7	11.6	7.0	8.0	10.3	2.4 to 8.6	Above (DSL2)
Lithium	mg/kg	21.2	28.9	38.7	20.8	25.9	28.5	17.4	28.1	35.3	3.3 to 38.7	Within
Magnesium	mg/kg	3,800	3,943	4,200	3,550	3,950	4,480	3,860	5,234	7,250	591 to 6,854	Within
Manganese	mg/kg	173	261	369	148	158	167	210	1,405	2,900	96 to 478	Above (LCB)
Mercury	mg/kg	0.03	0.04	0.06	0.03	0.03	0.05	0.03	0.03	0.03	0.05 to 0.06	Below (DSL1, DSL2, and LCB)
Molybdenum	mg/kg	5.2	8.6	15.6	2.8	5.5	10.4	3.5	5.9	9.5	1.9 to 17.3	Within
Nickel	mg/kg	29.3	35.4	39.9	32.5	35.9	40.8	24.5	30.1	33.2	26.6 to 56.6	Within
Phosphorus	mg/kg	775	1,445	1,950	602	779	953	572	938	1,210	594 to 2,994	Within
Potassium	mg/kg	1,490	1,537	1,580	1,210	1,373	1,470	1,620	2,278	3,350	0 to 3,650	Within
Rubidium	mg/kg	12.7	13.9	15.8	11.9	13.6	14.8	12.9	17.1	22.6	0.6 to 26.7	Within
Selenium	mg/kg	0.76	1.17	1.64	0.71	0.90	1.26	0.67	0.92	1.22	0.10 to 0.10	Above (DSL1); Below (DSL2 and LCB)
Silver	mg/kg	0.10	0.15	0.24	0.10	0.10	0.10	0.10	0.10	0.10	0.20 to 0.20	Below (DSL1, DSL2, and LCB)
Sodium	mg/kg	370	397	450	370	413	480	150	156	160	139 to 345	Above (DSL1 and DSL2)
Strontium	mg/kg	65.2	88.6	116.0	67.2	83.0	91.7	23.4	27.0	32.8	15.7 to 39.1	Above (DSL1 and DSL2)
Thallium	mg/kg	0.13	0.15	0.16	0.13	0.13	0.14	0.14	0.24	0.38	0 to 0.41	Within
Tin	mg/kg	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	2.00 to 2.00	Below (DSL1, DSL2, and LCB)
Titanium	mg/kg	253	279	297	227	291	373	215	334	550	98 to 822	Within
Uranium	mg/kg	17.4	20.8	27.2	25.5	34.1	41.8	24.6	31.5	37.2	3.5 to 14.6	Above (DSL1, DSL1, and LCB)
Vanadium	mg/kg	25.2	28.1	32.7	22.9	24.3	26.3	29.0	35.3	43.5	16.6 to 46.4	Within
Zinc	mg/kg	104	122	153	118	124	135	077	104	140	72 to 298	Within

% = percent; mg/kg = milligrams per kilogram; ± = plus or minus; SD = standard deviation; wt = weight; ID = identification number; DSL1 = Downstream Lake 1; DSL2 = Downstream Lake 2; LCB = Lac Capot Blanc; min = minimum; max = maximum.

Sediment Chemistry Results

Sediments from the three downstream lakes sampling stations consisted primarily of fine-grained material (silt and clay), 81% to 99% fines. The total organic carbon (TOC) concentrations ranged from 8.9% to 21.2%. 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 downstream lakes or showed a net decrease with increasing distance downstream, except that mean concentrations of nine analytes increased with distance downstream: available phosphate, barium, chromium, magnesium, manganese, potassium, titanium, uranium, and vanadium. Similar patterns were observed for available phosphate, chromium, and uranium in 2012, when only one station was sampled in each lake.

None of the metals concentrations were above PELs; however, concentrations of four metals were above their respective ISQGs: cadmium at Stations DSL-2 and LCB-2A; chromium at Stations LCB-3 and LCB-4; copper at all stations; and, zinc at Stations DSL1-1, DSL2-1, LCB-2A, and LCB-7. Concentrations of these metals have also been above their respective ISQGs in sediment from Snap Lake and Northeast Lake in previous years of AEMP monitoring, reflecting the natural enrichment of the region.

The minimum, mean, and maximum concentrations of target analytes were calculated for each downstream lake. Concentrations not detected in the analyses were replaced with values equal to half of their respective DL. The mean values were used to compare concentrations of target analytes in each downstream lake with their respective Snap Lake baseline normal ranges. Concentrations of available potassium, available sulphate, antimony, beryllium, calcium, lead, manganese, selenium, sodium, strontium, and uranium were above Snap Lake normal ranges at one or more downstream lakes. Concentrations of available ammonium, antimony, bismuth, copper, mercury, selenium, silver, and tin were below Snap Lake normal ranges at one or more downstream lakes.

11.3.6 Downstream Water Quality Prediction Summary

It was recommended in the 2012 Annual AEMP report that the downstream water quality predictions be revisited, so that mixing and other processes could be considered (De Beers 2013a). For the EAR, 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 consider mixing patterns within each of the lakes, or provide time-varying estimates of concentrations at particular nodes, nor did it account for settling or the time of travel through the Lockhart River system.

More rigorous predictions (including timing and movement of the treated effluent plume) have now been completed to support the downstream lakes monitoring program development. A mass-balance model of DSL1 and DSL2 was set up in GoldSim and a three-dimensional (3-D) hydrodynamic model was developed for Lac Capot Blanc in the same model platform that was used to predict water quality in Snap Lake (Generalized Environmental Modelling System for Surfacewaters [GEMSS]). The hydrodynamic model was used to predict temperature and TDS concentrations at various points in Lac Capot Blanc, including near the inlet and outlet, in deeper areas, and as whole-lake averages. A steady-state model, similar to that used in the EAR, was used to predict TDS concentrations in lakes downstream of Lac Capot Blanc. Detailed methods and results will be provided in Water Licence Amendment documentation in 2014.

11.3.7 Summary and Conclusions

11.3.7.1 Key Question 1: What is the spatial extent of the treated effluent plume downstream of Snap Lake (i.e., plume delineation)?

Evidence of the treated effluent was detected throughout DSL1 and DSL2 and near the inlet of Lac Capot Blanc in 2013. Treated effluent extended approximately 5 km from the inlet of Lac Capot Blanc and approximately 11 km downstream from Snap Lake's outlet in 2013 (Figure 11.3-7). Based on the 2013 conductivity values the area influenced by treated effluent increased in size. The field conductivity decreased to background levels within 50 m and within 650 m of the inlet of Lac Capot Blanc in 2011 and 2012, respectively (De Beers 2012, 2013a).

11.3.7.2 Key Question 2: What are the current water and sediment quality characteristics in the three downstream lakes?

Based on the field measurements collected in 2013 (Appendix 11.3D, Table 11.3D-1), the water in DSL1 and DSL2, and Lac Capot Blanc was well oxygenated, and varied from slightly acidic to slightly alkaline. Field pH measurements at stations DSL2-1, LCB-1, and LCB-6 were lower than the optimal range for aquatic life (pH 6.5 to 9.0) during ice-covered season.

Concentrations of TDS, nitrate, and major ions were elevated in DSL1, DSL2, and Inlet 1 LCB (Appendix 11.3E, Table 11.3E-1) and decreased at LCB-1 in Lac Capot Blanc in 2013 (Appendix 11.3E, Table 11.3E-1), indicating that the influence of the treated effluent extends beyond the inlet of Lac Capot Blanc. The same decreasing trend was also observed in barium, boron, lithium, molybdenum, nickel, rubidium, strontium, and uranium, which are also characteristic of the treated effluent (Appendix 11.3E, Table 11.3E-1). Results indicate that the influence from the Mine was reduced as total watershed areas and inflows to the downstream lakes increased.

Most parameters in downstream lakes in 2013 were below the AEMP benchmarks, with the exception of fluoride and lead (Appendix 11.3E, Table 11.3E-1). Fluoride concentrations were above the aquatic life guideline (i.e., 0.12 mg/L) at two stations (DSL1-1 and DSL2-1) in July and at DSL2-1 in September 2013. Although the primary source of fluoride was treated effluent, increases in fluoride concentrations correspond with elevated calcium and hardness, which are expected to reduce the potential for toxicity effects associated with fluoride The total lead concentration at LCB-2A in September 2013 was also above its AEMP benchmark; however, the exceedance was attributed to an anomalous result in that sample, rather than a Mine-related effect.

Total nitrogen concentrations decreased with distance downstream of Snap Lake. Within Lac Capot Blanc, TN concentrations were similar among all stations and were similar to concentrations observed in Northeast Lake and Lake 13. Total phosphorus concentrations were similar among the three downstream lakes and were similar to concentrations observed in Snap Lake and Northeast Lake (Section 3). Chlorophyll *a* concentrations were similar among the three downstream lakes and comparable to concentrations observed in Northeast Lake and Lake 13 (Section 5), but were lower than those observed in the northwest arm of Snap Lake and higher than those observed in the main basin of Snap Lake. Chlorophyll *c* concentrations were greatest in DSL1, followed by Lac Capot Blanc, and DSL2. Chlorophyll *c* concentrations observed in the downstream lakes were similar to those observed in Northeast Lake and

Lake 13, but were higher than those observed in the main basin of Snap Lake and lower than those observed in the northwest arm of Snap Lake.

Based on the results reported for 2013 (Table 11.3-4), sediments from the three downstream lakes sampling stations consisted primarily of fine-grained material (silt and clay) and TOC concentrations ranged from 8.9% to 21.2%. 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 downstream lakes or showed a net decrease with increasing distance downstream, except that mean concentrations of nine analytes increased with distance downstream: available phosphate, barium, chromium, magnesium, manganese, potassium, titanium, uranium, and vanadium. Similar patterns were observed for available phosphate, chromium, and uranium in 2012, when only one station was sampled in each lake.

None of the sediment metals concentrations were above PELs; however, concentrations of four metals were above their respective ISQGs: cadmium at two stations; chromium at two stations; copper at all stations; and, zinc at four stations. Concentrations of these metals have also been above their respective ISQGs in sediment from Snap Lake and Northeast Lake in previous years' AEMP monitoring, reflecting the natural enrichment of the region.

Sediment concentrations of available potassium, available sulphate, antimony, beryllium, calcium, lead, manganese, selenium, sodium, strontium, and uranium were above Snap Lake normal ranges at one or more downstream lakes. Concentrations of available ammonium, antimony, bismuth, copper, mercury, selenium, silver, and tin were below Snap Lake normal ranges at one or more downstream lakes.

11.3.8 Recommendations

As per the 2013 AEMP Design Plan (De Beers 2014), monitoring in the downstream lakes, DSL1, DSL2, and Lac Capot Blanc, will continue in order to evaluate current conditions, investigate dispersion of treated effluent associated with Mine discharge, and support future model updates. Specific recommendations for improvements are provided below.

Monitoring Current Conditions

As per the 2013 AEMP Design Plan (De Beers 2014), the 2014 to 2016 Downstream Lakes Special Study will continue to gather information on the downstream spatial extent of the treated effluent plume and on water and sediment quality on an annual basis. Monitoring in 2014 will be similar to that done in 2013, but will also include sampling of biotic components including benthic invertebrates, plankton, and fish. Recommendations for improvements or to address data gaps are:

- Measuring the main point source inflows to and outflows from DSL1, DSL2, and Lac Capot Blanc when water quality samples are collected to determine whether the water balances developed for the downstream lakes are representative of conditions.
- Recording ice thickness routinely in DSL1, DSL2, and Lac Capot Blanc, if conditions allow. Ice formation and melting dates and ice thickness drive salt rejection and freshwater replacement in the downstream lakes models, which in turn affects mixing and overall concentrations.
- Documenting fish habitat characteristics in the streams connecting the downstream lakes.

Scope of Study

As part of the AEMP re-evaluation and study design update in 2016, the scope of monitoring as part of the AEMP will be assessed. At that time, it will be determined whether monitoring in the Downstream Lakes will remain as a special study or be incorporated into the core AEMP program. Specific sampling locations and procedures will be provided based on updated modelling results as well as information collected during these first few years of special study investigation.

11.3.9 References

APHA (American Public Health Association). 2005. Standard Methods for the Examination of Water and Wastewater, 21st Edition, Washington, DC, USA.

CCME (Canadian Council of Ministers of the Environment). 1999. Canadian Environmental Quality Guidelines, 1999. Canadian Environmental Quality Guidelines Summary Table, with updates to 2013. Winnipeg, MB, Canada. Available at: http://st-ts.ccme.ca/. Accessed November 2013.

Chapman PM, Mann GS. 1999. Sediment quality values (SQVs) and ecological risk assessment (ERA). Mar Pollut Bull 38:339-344.

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

De Beers. 2008. 2008 Quality Assurance and Quality Control (QA/QC) Plan, Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012. 2011 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2013a. 2012 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2013b. Development of Fluoride Benchmark for Aquatic Life in Snap Lake. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

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

SECTION 11.4

LAKE TROUT POPULATION ESTIMATE SPECIAL STUDY

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Acronym	Definition
AEMP	Aquatic Effects Monitoring Program
BsM	Broad-scale Community Monitoring
CBFWA	Columbia Basin Fish and Wildlife Authority
CPUE	catch per unit effort
De Beers	De Beers Canada Inc.
DO	dissolved oxygen
e.g.	for example
FL	fork length
G	fish girth
Golder	Golder Associates Ltd.
i.e.	that is
ITL	fish total length in inches
LCL	lower credibility limit
LTH	Lake Trout per hectare
M	total number of individuals marked in first sample
m	number of marked fish recaptured in second sample
MC	Monte Carlo error
MCMC	Markov Chain Monte Carlo
Mine	Snap Lake Mine
MVLWB	Mackenzie Valley Land and Water Board
n	total number of fish (marked and unmarked) in second sample
N	total number of individuals in the population
NAD	North American Datum
NWT	Northwest Territories
QA	quality assurance
QC	quality control
R	a free software programming language and software environment for statistical computing and graphics
rjags	JAGS (Just Another Gibbs Sampler) is a program for the analysis of Bayesian models using Markov chain Monte Carlo (MCMC) in R
SD	standard deviation
the Study	Lake Trout Population Estimate Special Study
TL	total length
UCL	upper credibility limit
UTM	Universal Transverse Mercator
W	weight

UNITS OF MEASURE

Unit	Definition
ºC	degrees Celsius
%	percent
±	plus or minus
<	less than
>	greater than
µS/cm	microSiemens per centimetre
cm	centimetre
g	gram
ha	hectare
in	inch
kHz	kiloHertz
km	kilometre
km^2	square kilometres
km/hr	kilometres per hour
m	metre
m^3	cubic metres
mg/L	milligrams per litre
mm	millimetre

11.4 Lake Trout Population Estimate Special Study

11.4.1 Introduction and Objectives

11.4.1.1 Background

Specific Water Licence conditions relevant to the Lake Trout Population Estimate Special Study (the Study) component of the Aquatic Effects Monitoring Program (AEMP) in the Water Licence MV2011L2-0004 [Part G, Schedule 6, Item 1a (iv) and 1d of MVLWB (2013)] are:

a) Monitoring for the purpose of measuring Project-related effects on the following components of the Receiving Environment:

iv. fish population and community composition using standard methods;

d) Procedures to minimize the impacts of the AEMP on fish populations and fish habitat.

To sample the fish community within Snap Lake, a Broad-scale Community Monitoring (BsM) netting program (Sandstrom et al. 2009), which is standardized to lake area with respect to the number and mesh size of gill nets used and the depths over which netting occurs, is conducted every three years. This method requires that gill nets be set overnight to obtain a representative sample which, in 2013, resulted in the mortality of approximately 88 Lake Trout (*Salvelinus namaycush*) in Snap Lake. The effects of mortality due to the BsM method are assumed to be low, based on data from Ontario lakes where, on average, the BsM methodology results in the mortality of less than 2 percent (%) of the fishable population (Sandstrom 2013, pers. comm.). However, this assumption has not been validated for any of the lakes within the Snap Lake AEMP. Accordingly, the objective of the Study was to estimate the population abundance of fishable Lake Trout (greater than 250 millimetres [mm] fork length [FL]) in Snap Lake. With this information, a decision could be made as to whether the level of monitoring-based mortality could be sustained by the Snap Lake population and whether this incremental increase in mortality could potentially affect the ability to identify Snap Lake Mine (Mine)-related effects (see Section 8, Fish Community Monitoring). The Study was endorsed by federal and provincial fisheries professionals at a 2012 De Beers workshop and, in 2012, was included as a special study component of the AEMP.

In addition to providing information to assess monitoring-related mortality, the population abundance estimate was expected to provide a reference point (e.g., Lake Trout per hectare [LTH]) for the relative abundance estimate (e.g., catch per unit effort [CPUE]) of Lake Trout from the BsM program. This reference point could then be used to make comparisons with other lakes where estimates of the absolute abundance of fishable Lake Trout have been made, as a means of determining the relative productivity of the Snap Lake population. For lakes in Saskatchewan, Alberta, and Ontario, the CPUE of Lake Trout caught in large mesh nets, as used in the BsM program, was correlated with the fishable population abundance determined by mark-recapture (e.g., LTH; Sullivan 2013, pers. comm.). This indicates that standardized gill netting programs such as the BsM have the potential to provide an effective means of gauging relative population status although, as discussed above, there is a need to determine whether catches are sustainable, and whether the numbers of fish removed from the lake affect an ability to identify project effects. These questions are addressed in the Fish Community Monitoring section of this AEMP report (Section 8).

The Study was conducted over a two year period (2012 and 2013). In 2012, fish were marked and recaptured over two approximately one week periods in July (July 11 to 17) and August (August 21 to 26). In 2013, fish were recaptured from July 5 to 11. The recaptures made in August 2012 and July 2013 were used to derive the estimate of population abundance.

11.4.2 Objectives

The objectives of the Study were as follows.

- In the 2012 marking sessions - capture, mark, and live release a sample of approximately 400 Lake Trout from Snap Lake.
- Use an active fishing method (e.g., angling) for capturing fish and a fish processing protocol with low direct and residual mortality. Mark fish so that each individual could be identified on recapture, the loss of marks would be low, and this loss could be independently verified by means of a secondary mark.
- In the 2013 recapture session, capture as large a sample of Lake Trout as possible (ideally 700 fish) from all parts of the lake where fish were distributed and had an equal opportunity for capture.
- Using data from the multiple mark-recapture sample design, estimate the population abundance of fishable Lake Trout in Snap Lake, and the level of confidence surrounding this estimate.

Analysis of information provided by the Study addressed the following key question:

- How many Lake Trout of fishable size (greater than 250 mm FL), are estimated to be in Snap Lake and what is the level of confidence of that estimate?

11.4.3 Methods

11.4.3.1 Study Area

Snap Lake is located 220 kilometres (km) northeast of Yellowknife, Northwest Territories. The lake is 30 km south of MacKay Lake and 100 km south of Lac de Gras, where the Diavik and Ekati diamond mines are located (Section 1, Figure 1-1). Snap Lake has a surface area of approximately 16 square kilometres (km^2) (1,600 hectares [ha]) and a volume of 79 million cubic metres (m^3). Snap Lake is shallow, with a mean depth of approximately 5 metres (m) and, for the most part, is well-mixed with little evidence of thermal stratification during open-water conditions. Two exceptions are deeper areas greater than 20 m, one located in the main basin and one in the northwest arm that were stratified at depths greater than 20 m during the summer. Snap Lake is clear, with a Secchi disc visibility of 6 to 7 m (Section 3.4.1) and is classified as oligo-mesotrophic because of low to moderate nutrient availability and organic productivity (De Beers 2012a). The open-water season generally extends from July to October, with the lake being ice-covered from November to June. The durations of the open water and ice-covered periods have been consistent over the past seven years (De Beers 2006, 2007, 2008, 2009, 2010, 2011, 2012b, 2013).

11.4.3.2 Fish Collections

The target for the 2012 marking program was established at 400 Lake Trout. This target was based on the estimated number of Lake Trout in Snap Lake and the sample size of marked fish required to obtain a statistically valid estimate of abundance (Robson and Regier 1964). A low abundance (e.g., 1 fish per ha; 1,600 fishable Lake Trout) was assumed based on Snap Lake's northern location and relatively low aquatic productivity (Downing and Plante 1993). Using the sample size charts developed by Robson and Regier (1964) and the target of 400 marked fish, an estimated 700 fish would need to be captured and examined for marks to provide a population estimate with 95% credibility intervals and an accuracy level of plus or minus (±) 10%.

During 2012, angling was used to collect Lake Trout for marking (Table 11.4-1). All angling was conducted from a boat using two or three rods equipped with either a large spoon (i.e., 4 to 6 inches [in] long by 2 to 3 in wide) or metal jig (0.8 to 1.8 ounce) (Appendix 11.4A, Photos 11.4A-1 and 11.4A-2). All lures had a single unbaited barbless hook. On the first day of angling in July 2012, trolling two or three lines behind a boat moving at 4 to 6 kilometres per hour (km/hr) was evaluated as a means of collecting fish. Due to low fish catches using this method and lack of large targets (presumed to be Lake Trout) on the Lowrance Mark 5 Pro Dual Frequency fish finder in the areas trolled, trolling was discontinued after several hours of effort. The crew then proceeded to survey the lake and identify areas that contained aggregations of large targets on the fish finder. Results of these surveys indicated Lake Trout were essentially aggregated in two deep (e.g., greater than 20 m depth) areas of the lake, likely due to colder temperatures in these locations. During July 2012, the thermocline was located relatively deep and close to the bottom with temperatures in the epilimnion in excess of 15 degrees Celsius (°C), a temperature that exceeds an upper thermal criterion established for Lake Trout based on biotelemetry (Plumb and Blanchfield 2009). Because thermal structure during July 2012 was considerably different from that normally prevailing after ice out when there is little to no stratification and temperatures are well below 15°C, a different sampling approach was required than that normally used for BsM sampling. The BsM program uses a series of randomly set gillnets set across multiple depth strata, and conducted immediately after ice out in late June when fish are believed to be well dispersed and randomly distributed. In July 2012, suitable habitat was restricted to near bottom, requiring a different sampling approach.

Based on the finding that most Lake Trout were located near bottom and associated with deep water areas that appeared to provide thermal refuge, angling effort was switched exclusively to these two deep water areas with vertical jigging used as the sole method of capture. Although this shift in methodology from trolling to jigging resulted in a smaller area of the lake being sampled than was originally planned, this was necessitated by the aggregated nature of Lake Trout during 2012 and the need to meet the marking target. The jigging method of angling involved dropping a metal jig or round headed jig or spoon to the bottom of the lake, and then retrieving it while alternately raising and lowering the rod while reeling in the slack line. The 2012 angling locations are described in Table 11.4-1 and general locations fished and locations where Lake Trout were collected are shown in Figure 11.4-1.

Table 11.4-1 Fish Sampling Locations in Snap Lake, Summer 2012

Sampling Location	Sample Method	UTM Coordinates(a)	
		Easting	Northing
Main Basin	Vertical Jigging	507118	7052611
Main Basin	Vertical Jigging	507155	7052773
Northwest Arm	Vertical Jigging	500410	7052575
Northwest Arm	Vertical Jigging	500462	7052651

Note: Refer to Figure 11.4-1 for specific sample locations.

a) North American Datum (NAD), Zone 12V. The Universal Transverse Mercator (UTM) coordinates represent the general sampling area.

In 2013, a combination of jigging and trolling were used to recapture tagged Lake Trout. This change in sample methodology from the 2012 marking sessions (that used jigging only), was initially devised to cover a larger area of the lake in an attempt to reduce the spatial sampling bias inherent in the 2012 marking sessions and increase recaptures of fish tagged in 2012. This decision was supported by initial sample surveys in July 2013 that revealed a much broader distribution of Lake Trout in Snap Lake (based on sonar targets and initial captures) likely due to cooler water temperatures in 2013 (see Section 2, Site Characterization and Supporting Environmental Variables). Vertical jigging was used in the two deepest areas of the lake as was done in 2012. Trolling was used to collect Lake Trout from a larger area of Snap Lake during 2013, including several arms off the main basin. The 2013 angling locations are shown in Figure 11.4-2 and presented in Table 11.4-2.

Table 11.4-2 Fish Sampling Locations in Snap Lake, Summer 2013

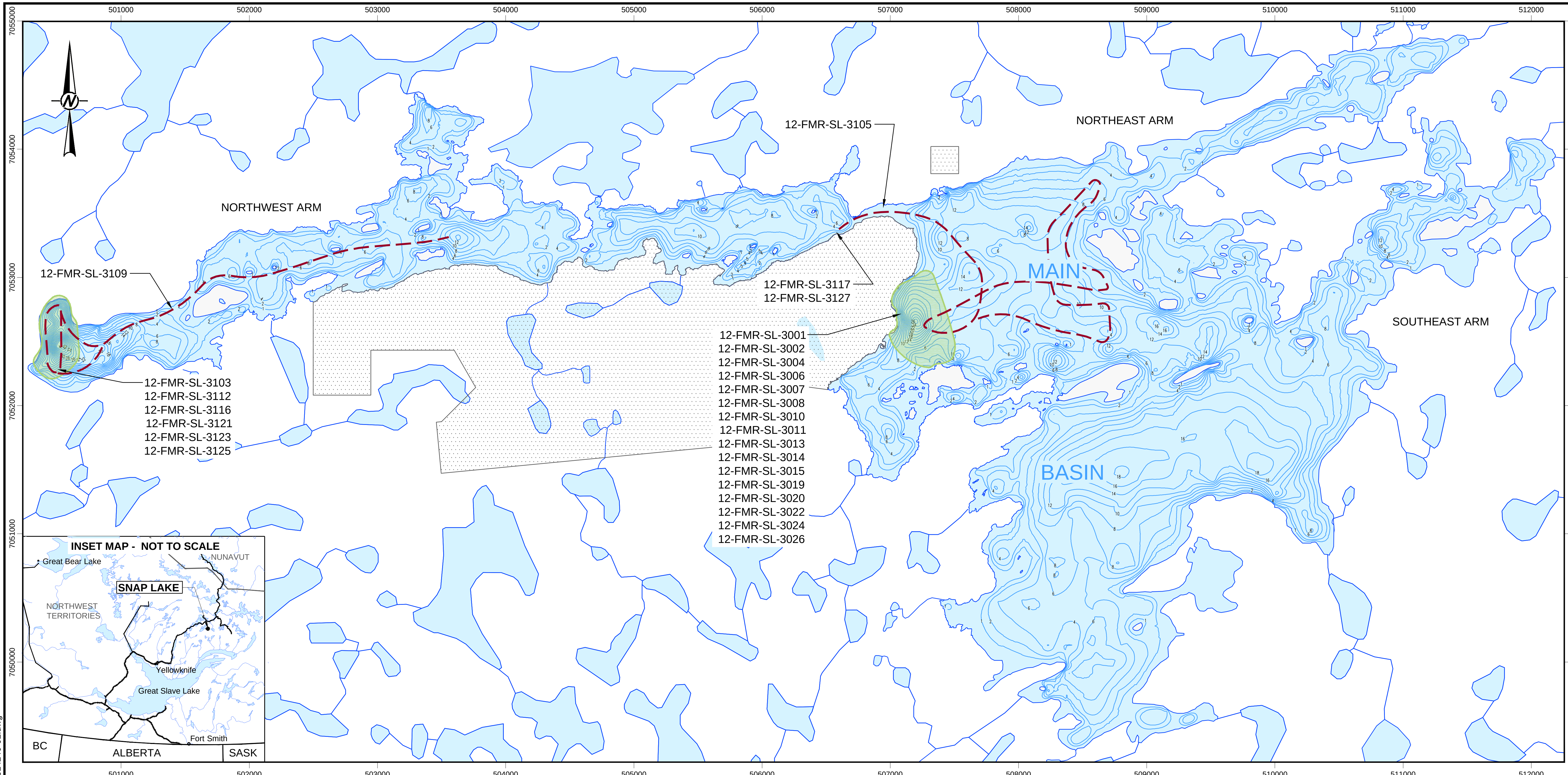
Sampling Location	Sample Method	UTM Coordinates(a)			
		Start		End	
		Easting	Northing	Easting	Northing
Main Basin	Vertical Jigging	507155	7052575	-	-
Northwest Arm	Trolling	500681	7052228	500329	7052304
Northeast Arm	Trolling	508267	7053328	510661	7054455
Southwest Arm	Trolling	508774	7053691	511372	7054788
Northwest Arm	Trolling	500681	7052228	500387	7052902
Main Basin (south end)	Trolling	509021	7051398	508331	7050548
Southeast Arm	Trolling	510481	70527425	512055	7053843

Note: Start and end refer to the trolling transect points of origin. Refer to Figure 11.4-2 for specific locations.

a) North American Datum (NAD), Zone 12V. The Universal Transverse Mercator (UTM) coordinates represent the general sampling area for angling.

- = not applicable.

.



L:\2013\1349\13-1349-0001\Phase 4200\Task 4240\13134900014240-02.dwg

LEGEND

- depth Contour (m)
- Watercourse
- Trolling
- Waterbody
- Snap Lake Mine Footprint
- Area Angled During Investigation

12-FMR-SL-3101

YEAR — FISH MARK/ RECAPTURE — SNAP LAKE — EFFORT #

REFERENCES

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

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

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

SNAP LAKE MINE

Lake Trout Population Estimate Sampling Sites in Snap Lake, 2012

PROJECTION: UTM Zone 12	DATUM: NAD83

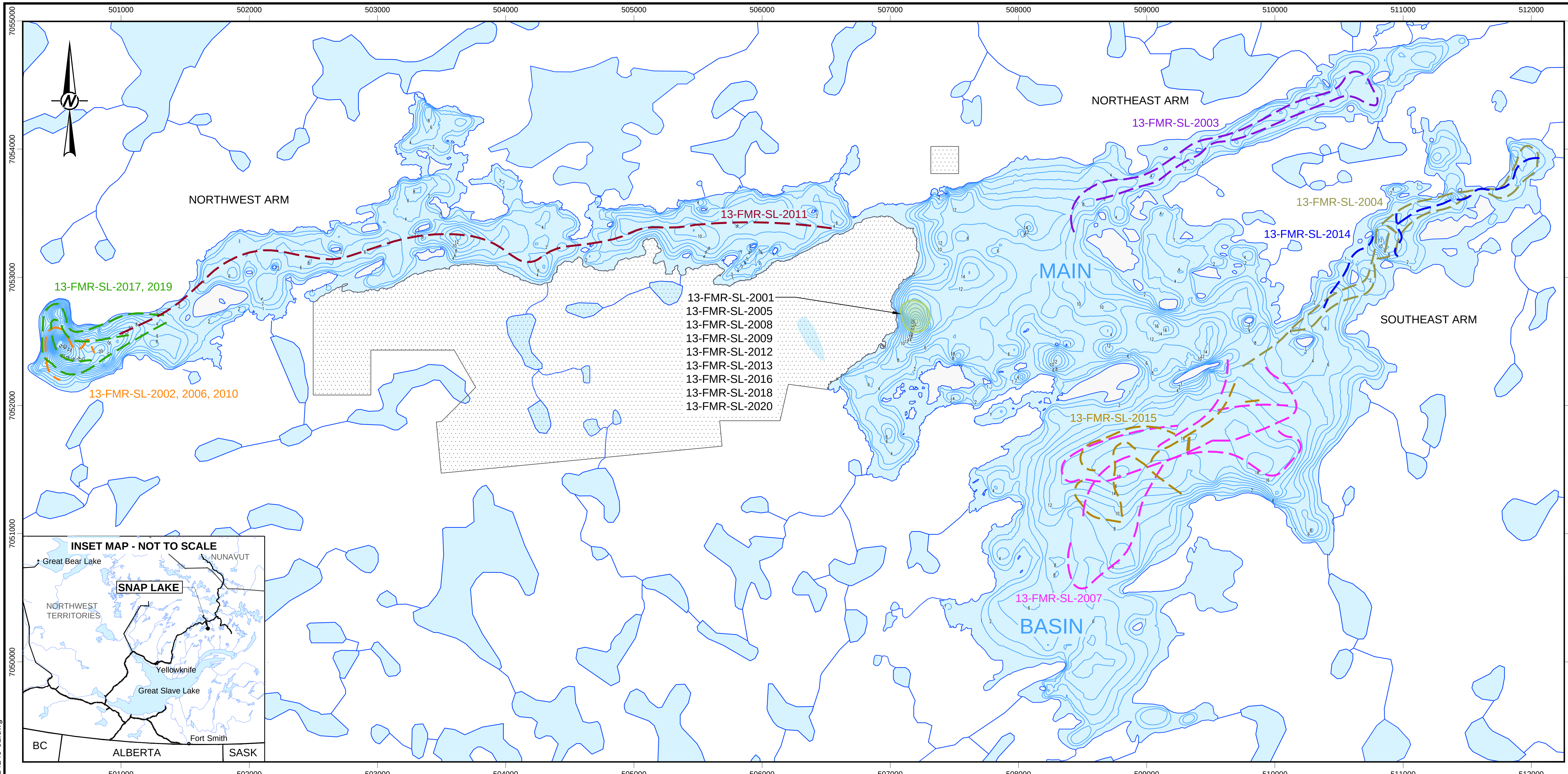
500 0 500

SCALE 1:30,000 METRES

DE BEERS GROUP OF COMPANIES

FILE No: 13134900014240-02		DATE: November 5, 2013
JOB No: 14-1349-0004	REVISION No: A	
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC

Figure 11.4-1



L:\2013\1349\13-1349-0001\phase 4200\task 4240\13134900014240-01.dwg

LEGEND

- depth Contour (m)
- Watercourse
- Trolling
- Waterbody
- Snap Lake Mine Footprint
- Area Angled During Investigation

13-FMR-SL-2001

YEAR — FISH MARK/ RECAPTURE — SNAP LAKE — EFFORT #

REFERENCES

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

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

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

SNAP LAKE MINE

Lake Trout Population Estimate Sampling Sites in Snap Lake, 2013

PROJECTION: UTM Zone 12	DATUM: NAD83
500 0 500 SCALE 1:30,000 METRES	DE BEERS GROUP OF COMPANIES
FILE No: 13134900014240-01	DATE: November 5, 2013
JOB No: 14-1349-0004	REVISION No: A
OFFICE: GOLDER VICTORIA	DRAWN: JEF / CHECK: PMC

Figure 11.4-2

11.4.3.3 Fish Processing and Marking

All Lake Trout collected were, while still hooked, gently guided into a partially submerged cloth holding cradle at the side of the boat where they were measured for FL and total length (TL) (± 1 mm) and weighed to the nearest gram (g) using a digital scale. Technical difficulties with the digital weigh scale at the start of the July program in 2012 required that weight (W) be estimated according to Lum (2013).

$$\mathbf{W = ITL^2 \times G \times 0.0007 \times 0.453592} \qquad \text{[Equation 11.4-1]}$$

Where: W = weight (g);

ITL = fish total length in inches (as calculated from the TL measured in mm); and,

G = fish girth in mm measured just posterior to the dorsal fin

After the fish was measured and while still resting in the cradle, it received a PIT tag (134.2 kiloHertz [kHz]; length 12.5 mm, diameter 2 mm) (Appendix 11.4A, Photos 11.4A-3 and 11.4A-4). The PIT tag was injected into the abdominal cavity posterior to the pectoral fin using a plastic syringe style implanter fitted with a 3.2 centimetre (cm) long non-replaceable needle (2 mm bore) according to the procedures outlined in the PIT Tag Marking Procedures Manual (CBFWA 1999; Appendix 11.4A, Photos 11.4A-5 to 11.4A-8). Fish were oriented ventral side up and held in the cradle by one individual, while a second individual loaded the PIT tag into the needle and injected it into the fish between the posterior tip of the pectoral fin and the anterior point of the pelvic girdle, 1 to 2 mm off the mid-ventral line. Prior to release, the PIT tag number was confirmed by scanning the fish using a digital PIT tag reader. As a precaution against losing PIT tag information stored on the reader, the PIT tag number of each fish was also manually recorded on the appropriate data sheet along with the corresponding fish measurement data.

As a check against PIT tag loss, the adipose fin of each PIT tagged fish was partially removed as a secondary mark (Appendix 11.4A, Photo 11.4A-9). The excised fin was stored in a separate labelled vial, placed in a freezer, and archived for potential future analyses (e.g., genetics or stable isotope). The total processing time for each fish from the time when the fish first entered the cradle to when it was released, was typically less than one minute. Tagged fish were released in the same area of the lake where they were collected. Upon release, tagged fish typically swam of their own volition out of the fully submerged cradle, away from the boat, and towards the bottom (Appendix 11.4A, Photos 11.4A-10 to 11.4A-14). Fish that did not immediately swim away were held in the submerged cradle until they fully recovered and swam away. If there was any doubt the fish would survive after being released, it was sacrificed by a blow to the head and processed for additional life history information. During recapture, fish were again led into the fish cradle and scanned for a PIT tag, examined for an adipose clip, and released back into the lake. In 2013, newly captured unmarked fish that were not PIT tagged had their adipose fin removed as in 2012 and the left pelvic fin removed for aging. This marking procedure allowed the identification of within-session recaptures in 2013 that were not PIT tagged.

11.4.3.4 Short-term Survival Assessment of PIT Tagged Fish

To assess the short-term effect of PIT tagging on fish survival, three PIT tagged and three non-PIT tagged (control) fish were held for 29 hours in net pens (1.0 m width x 1.0 m length x 1.0 m depth) next to the shore in 1.0 m of water (Appendix 11.4A, Photos 11.4A-15 and 11.4A-16). With the exception of the three fish that received a PIT tag, all fish were caught, handled, and processed in the same manner. After the holding period their condition (i.e., swimming upright, skin appearance, and gill ventilation rate) was noted and, if deemed healthy, the fish were released back to the lake.

11.4.3.5 Aging Determination

To provide representative aging samples from marked fish, the leading ray of the left pelvic fin was removed from each live fish collected in August 2012 and stored in an individually-labelled envelope. In the office, the ray was coated in epoxy resin and allowed to set and harden. Duplicate sections of the ray (0.7 mm thick) were then cut using a Struers Minitom low speed sectioning saw. Sections were mounted on a glass slide with Cytoseal 60 and read under a compound microscope with transmitted light. A subsample (10%) of all fin rays collected was examined by a second fishery technician and, if ages between technicians differed, the structure was re-examined and a mutually agreed-upon age was assigned. Results of the age comparisons are provided in Appendix 11.4B.

11.4.3.6 Water Quality Data

To provide representative measures of water quality parameters of temperature (°C), dissolved oxygen (DO) (milligrams per litre [mg/L]), pH, and conductivity (microSiemens per centimetre [µS/cm]) for Snap Lake during the Study, profiles were collected at AEMP stations SNAP 20B and SNP02-20e (Section 3, Figure 3-1) within Snap Lake during July 2012 and 2013. Measurements were made using a YSI 650 Multiparameter Display System water quality sonde with a YSI 600 Quick Sample multi-parameter water quality probe. In 2012, measurements were made at 1 m depth intervals throughout the entire water column as part of the regular AEMP monitoring. In 2013, the same water quality measurements were made by fisheries personnel at the same stations and depths as sampled in 2012 (AEMP stations SNAP 20B and SNP02-20e) using the same YSI sonde as used in 2012 (Appendix 11.4C). The YSI sonde was calibrated for each of the parameters measured according to the manufacturer's instructions and with valid calibration standards as appropriate. The DO membrane on the sonde was replaced prior to use, if required.

Dissolved oxygen levels were related to the DO criterion (7 mg/L) established for Lake Trout based on metabolic scope-for-activity and power capacity of juvenile Lake Trout (Evans 2007). No criteria for pH and conductivity are known to exist for Lake Trout.

To evaluate temporal variation in the thermal structure of Snap Lake in 2013, water temperature data were also collected using a vertical array of temperature loggers (Onset Tidbit Water Temperature Loggers – UTBI-001) suspended from fixed moorings in each of the two deepest locations within Snap Lake. The upper-most logger of an array was positioned 0.3 m below surface, and the remaining loggers spaced at 3.0 intervals with the bottom logger set at 0.3 m above the substrate. The array located at AEMP water quality station Snap 20B in the northwest arm consisted of 15 loggers, while the array located at AEMP water quality station SNP02-20e near the diffuser in the main basin, consisted of ten loggers. The moorings and associated arrays were deployed in July 2013 and retrieved in September 2013 (Appendix 11.4D). Locations of the temperature arrays can be found in Section 2, Figure 2-1.

11.4.4 Population Abundance Estimation

A Bayesian probability implementation of the Petersen method (Pine et al. 2003) for closed population mark-recapture data was used to estimate the abundance of Lake Trout in Snap Lake in 2012 and 2013. The analysis was implemented using the statistical environment R, v. 3.0.1 (R Development Core Team 2013), interfaced with JAGS v. 3.3.0 (Plummer 2003) through the rjags package (Plummer 2013). JAGS distributions and functions are defined in Table 11.4-3.

Table 11.4-3 JAGS Distributions and Functions used in the Bayesian Models

Distribution/function	Description
dbin(p, n)	Binomial distribution with n trials and p probability of success
dnorm(μ, τ)	Normal distribution with a mean μ and 1/variance τ
log(x)	Natural logarithm function
logit(x)	Logit function

The classic Petersen model for two capture sessions assumes a closed population and equal capture probability and estimates the total number of individuals in the population (*N*) with the formula

$$N = Mn/m \qquad \text{[Equation 11.4-2]}$$

Where: ***M*** is the number of individuals marked during the first sample;
n is the total number of fish (marked and unmarked) in the second sample; and,
m is the number of marked fish recaptured in the second sample.

In the Study, a Petersen mark-recapture experiment was extended to account for marking in July 2012, marking and recapture in August 2012, and recapture in July 2013. In addition, the model was expanded to include a survival rate for the population. Four separate Bayesian models were constructed for this analysis:

1) survival estimated from catch curve data for fish collected in August 2012, with abundance estimated based on three angling sessions – July 2012, August 2012, and July 2013;
2) survival input as a constant rate of 0.9, simulating high survival in the adult population, with abundance estimated based on three angling sessions – July 2012, August 2012, July 2013;
3) survival estimated from catch curve data for:
    a. fish collected in August 2012, with abundance estimated based on two angling sessions – July 2012 and August 2012, and one gill-netting session in July 2013 (2013 BsM program; see Section 8); and,
    b. fish collected in August 2012, with abundance estimated based on a single angling session (total number marked in July 2012 and August 2012 combined), and one gill-netting session in July 2013 (2013 BsM program; see Section 8).

In the Bayesian implementation of a Petersen mark-recapture experiment, catchability was estimated based on the number of recaptures in each of the two recapture sessions (August 2012 and July 2013; see Table 11.4-4 for a full list of the parameters used, specified by model). Catchability for trolling and jigging was assumed to be equal. The total number of fish present at both recapture sessions was modelled as a binomial function of the number of untagged fish captured during the session and the catchability value. Survival was derived from the descending limb (ages 7 to 29) of the catch curve based on catch-at-age data for Lake Trout caught by angling in August 2012. For the analysis, recruitment was assumed to be equal to the annual mortality rate (i.e., the population was assumed to be in equilibrium).

Table 11.4-4 Variables and Parameters in the Bayesian Analysis of Fish Abundance

Variable/parameter	Model Number	Description
Marking parameters		
Jul12	1, 2, 3	Number of fish marked in July 2012
Aug12	1, 2, 3	Number of fish marked in August 2012
SJulAug12	1, 2, 3, 4	Number of fish marked in July/August 2012 and survived to July 2013
Recapture parameters		
JulAug12Jul13	1, 2, 3, 4	Fish marked in July and August 2012, and recaptured in July 2013
pAng	1, 2, 3	The catchability of Lake Trout using angling
pNet	3, 4	The catchability of Lake Trout using gill nets
S	1, 2, 3, 4	Survival between years
R	1, 2, 3, 4	Recruitment to the sampled population
Unmarked fish parameters		
uAug12	1, 2, 3	Number of unmarked fish captured in August 2012
uJul13	1, 2, 3, 4	Number of unmarked fish captured in July 2013
Abundance estimates		
etaUAug12	1, 2, 3	Logarithm of expected abundance of unmarked fish in August 2012
etaUJul13	4	Logarithm of expected abundance of unmarked fish in July 2013
UAug12	1, 2, 3	Abundance estimate for unmarked fish in August 2012
SUJul13	1, 2, 3	Number of unmarked fish that survived from August 2012 to July 2013
NUJul13	1, 2, 3	Number of fish recruited to the sampled population in July 2013
TotAug12	1, 2, 3	Abundance estimate for all fish (marked + unmarked) in August 2012
TotJul13	1, 2, 3, 4	Abundance estimate for all fish (marked + unmarked – deaths + recruits) in July 2013

The analysis was not stratified by year or marking site (i.e., catchability was modelled as a constant parameter, rather than a random variable). The prior distributions for all parameters were vague or uninformative (Table 11.4-5). The complete model specification used is shown in Table 11.4.6, and the model code is provided in Appendix 11.4E.

Table 11.4-5 Prior Probability Distributions in the Bayesian Analysis of Fish Abundance

Variable/Parameter	Model Number	Description
pAng	1, 2, 3	dunif(0, 1)
pNet	3, 4	dunif(0, 1)
etaUAug12	1, 2, 3	dnorm(0.0,1.0E-6)
etaUJul13	4	dnorm(0.0,1.0E-6)

Table 11.4-5 Prior Probability Distributions in the Bayesian Analysis of Fish Abundance

Variable/Parameter	Model Number	Description
JulAug12Jul13	1, 2	dbin(pAng, round(SJulAug12))
JulAug12Jul13	3, 4	dbin(pNet, round(SJulAug12))
uAug12	1, 2, 3	dbin(pAng, round(UAug12))
uJul13	1, 2	dbin(pAng, round(SUJul13))
uJul13	3	dbin(pNet, round(SUJul13))
uJul13	4	dbin(p, round(UJul13))

Table 11.4-6 Dependencies between Variables and Parameters in the Bayesian Analysis of Fish abundance

Variable/parameter	Model Number	Dependency
UAug12	1, 2, 3	round(exp(etaUAug12))
UJul13	4	round(exp(etaUJul13))
R	1, 2, 3, 4	1 – S
SJulAug12	1, 2, 3, 4	JulAug12*S
SUJul13	1, 2, 3	S*(round(UAug12) - Aug12)
NUJul13	1, 2, 3	R*(round(UAug12) - Aug12)
TotAug12	1, 2, 3	UAug12 + Jul12
TotJul13	1, 2, 3, 4	SUJul13 + Jul13 + NUJul13

Mean and median values of abundance estimates and 95% credibility intervals were calculated in R. The Monte Carlo error (MC) for each parameter estimate was recorded. The MC error quantifies the variability in the estimates that is due to the sampling error in the simulation-based solution for Bayesian analysis. Simulation run lengths were chosen such that the MC error was less than 5% of the posterior standard deviation for a parameter (Kery 2010). The posterior distributions, which were estimated using Gibbs sampling (Kery 2010), were derived from 5,000 Markov Chain Monte Carlo (MCMC) simulations and thinned from three MCMC chains of 104 iterations in length. Model convergence was confirmed by ensuring that R-hat (the Gelman-Rubin Brooks potential scale reduction factor) was less than 1.1 for each of the parameters in the model (Kery 2010).

11.4.5 Quality Assurance and Quality Control

As part of routine Quality Assurance (QA) and Quality Control (QC) for field operations, equipment was calibrated and samples were collected by experienced personnel and labelled, preserved as required, and shipped according to standard protocols. Specific work instructions that outlined each field task in detail were provided to field personnel by the task manager, and these were reviewed before any sampling occurred. Detailed field notes were recorded in waterproof field books and on pre-printed waterproof field data sheets and maps in either pencil or indelible ink. Data sheets and sample labels were checked at the end of each field day for completeness and accuracy. Chain-of-custody forms were used to track the shipment of fin rays for aging to North/South Consultants Inc.

All sampling related to the Lake Trout mark-recapture program was uniquely numbered with Universal Transverse Mercator (UTM) coordinates of angling start and end geographic locations along with gear type and water depth.

In the field, data forms were reviewed for accuracy daily by crew leads. Data were entered into Microsoft Excel when field crews returned to the office. A review of data entry involved checking a minimum of 10% of the entered data for accuracy, data entry errors, transcription errors, and invalid data. Checking was done by a second, independent individual. If an error was found, all data underwent a complete QA check (i.e., every datum checked) by the second independent individual. Upon completion of the data entry QA, each table generated from the database was reviewed for accuracy using a series of error checking routines as a secondary level of QC. All statistical results were independently reviewed by a second statistician within Golder Associates Ltd (Golder). Tables with summary data and statistical results were also checked and values verified by a second reviewer as were all appendices.

Appendix 11.4F provides the aging results for fin rays reported by North/South Consultants and subject to their internal QA/QC procedures. Upon receipt of the data, Golder staff visually screened the data for QA/QC and found no errors.

11.4.6 Results

11.4.6.1 Fish Collections

In 2012, when fishing was restricted to the deeper and colder areas of Snap Lake, 340 Lake Trout were captured by angling; of these, 295 were marked with a PIT tag and released and 37 were recaptured (Table 11.4-7). Mortalities over the two sampling periods due to hooking injuries (e.g., bleeding gills) averaged 3.4%. There was no mortality among three PIT tagged fish held overnight in net pens; although, one of three non-PIT tagged control fish died of an unknown cause. Water temperature during the holding period in the net pens averaged 15.5ºC (Appendix11.4 A, Photos 11.4A-17 and 11.4A-18).

Table 11.4-7 Numbers of Lake Trout Marked and Recaptured by Angling During the 2012 to 2013 Study Period

Sample Period	Number Marked	Number Recaptured	Time of Original Marking
July 2012	208[(a)]	25	July 2012
August 2012	87[(a)]	11	July 2012
		1	August 2012
July 2013	100[(b)]	16	July + August 2012
		3	July 2013
Total marked (PIT tagged)	**295**		

a) Marked by PIT tag and fin clip.

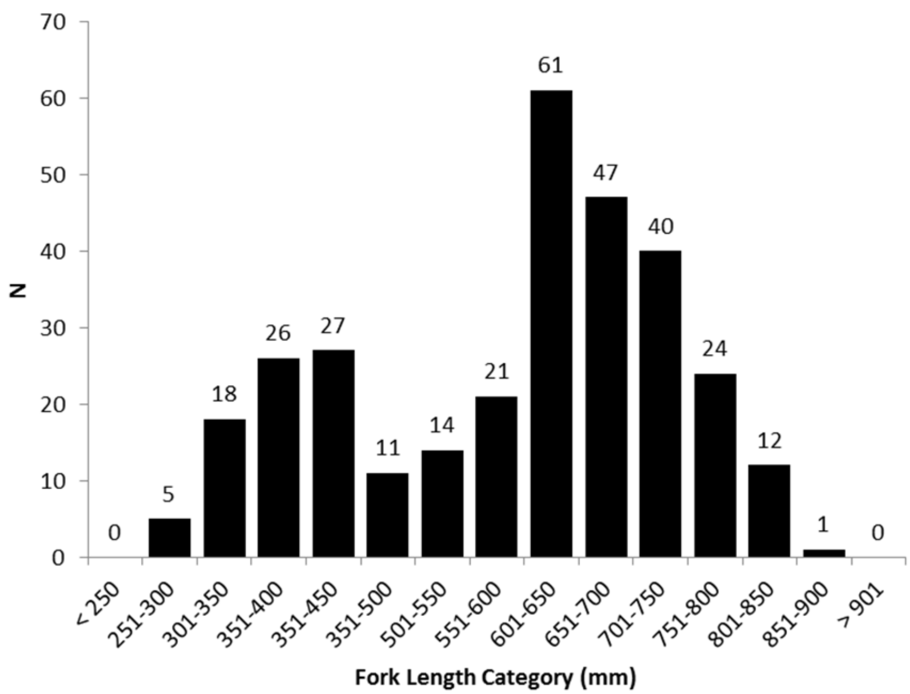
b) Marked by fin clip only, not PIT tagged

In total, 212 (72%) of the 295 Lake Trout marked in 2012 were captured in the main basin (Figure 11.4-1). The remaining 83 (28%) Lake Trout were captured in the northwest arm. In 2013, when fishing occurred throughout the lake including several arms off the main basin, 122 fish were collected, 14 of which had PIT tags and an additional two fish (12.5% of PIT tagged fish) that had been PIT tagged but had lost the tag (identified based on the presence of an adipose fin clip). Most fish collected in 2013 came from the main basin (70%) with fewer from the northwest arm (30%) and no fish collected from either the northeast or southeast arms. Total numbers of fish captured per sampling event, CPUE, and fish mortality for 2012 and 2013 are summarized in Appendix 11.4G.

Fish angled from the main basin of Snap Lake in July 2012, August 2012, and July 2013 comprised 75%, 71%, and 70%, respectively, of all fish caught. Of the fish marked in 2012 and recaptured in 2013, most fish were recaptured in the same area as initially captured, predominantly the main basin (86%). One fish collected in the northwest arm in 2012 was recaptured in the main basin in 2013. As well as producing the greatest numbers of fish during in each sampling session, the main basin had an angler CPUE that was consistently higher than the northwest arm (Appendix 11.4G).

Fish captured in 2012 (both sessions combined) had a mean length of 630 mm FL (range = 270 to 860 mm FL; Figure 11.4-3) and a mean weight of 2,620 g (range = 212 to 7,674 g). The mean age of Lake Trout, based on the fin rays of the 93 fish captured in August 2012, was 13 years (range = 5 to 29 years; Figure 11.4-4).

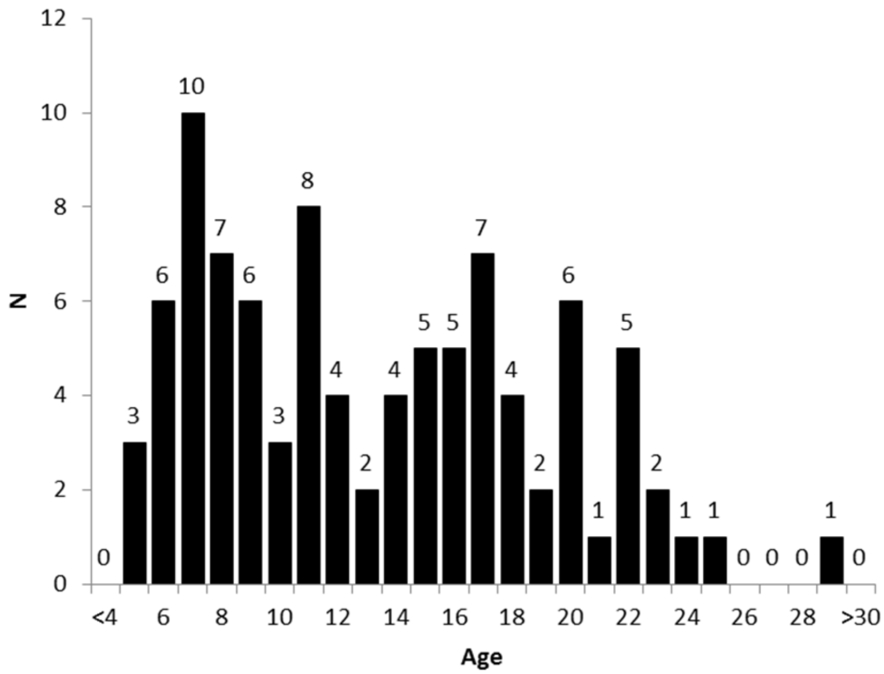
Figure 11.4-3 Length Frequency of Lake Trout Caught by Angling in Snap Lake, July and August 2012



Note: values above bars are number of fish per fork length category.

mm = millimetre; > = greater than; < = less than; N = total number of individuals in the population.

Figure 11.4-4 Age Frequency Based on Fin Rays of Lake Trout Caught by Angling in Snap Lake, August 2012



Note: values above bars are number of fish aged within a given age.

> = greater than; < = less than; N = total number of individuals in the population.

11.4.6.2 Lake Trout Population Abundance

A total of 295 Lake Trout were marked with a PIT tag in 2012: one was recaptured in July 2012; 12 were recaptured in August 2012; and, 16 were recaptured in July 2013, although three of the latter were multiple recaptures (Table 11.4-8).

Table 11.4-8 Numbers of Lake Trout Marked and Recaptured by Angling and Marked by Angling and Recaptured Using Gill Nets in the 2013 Broad-scale Community Monitoring Program

Sampling	Number Marked	Number Recaptured	Time of Original Marking
July 2012	208[(a)]	25	July 2012
August 2012	87[(a)]	12	July + August 2012
July 2013	100[(b)]	16	July + August 2012
2013 BsM	77[(c)]	11[(d)]	July + August 2012

a) Marked by PIT tag and fin clip.

b) Marked by fin clip only, not PIT tagged.

c) Sampled dead from gill nets without a PIT tag.

d) Sampled dead from gill nets with a PIT tag.

BsM = Broad-scale Community Monitoring.

For the three models (1, 3, 4) where survival was estimated from the catch curve, median catchability values were similar (range = 0.056 to 0.067) with the lowest catchability estimated from the gill netting that was part of the BsM netting described in Section 8. The BsM gill netting program used for the fish community assessment method on Snap Lake occurred the week after the July 2013 angling recapture session was completed (July 12 to 18).

The estimates of population abundance were relatively unaffected by the period tagged fish were at large, the estimate of fish survival, or whether recaptures were based strictly on angling, a combination of angling and netting, or netting alone (Table 11.4-9). The median population abundance of Lake Trout in 2012 (median = 1,671 fish; lower credibility limit [LCL] 1,238, upper credibility limit [UCL] 2,379) that used the catch curve survival of 0.722 and the two angling sessions in 2012 was similar to the 2013 estimate (median = 1,589 fish; LCL 1,151, UCL 2,299) that used the same survival but all three angling sessions (2012 to 2013). The population abundance values modelled with a higher survival value (0.9) for 2012 (median = 1,841 fish; LCL 1,360, UCL 2,633) and 2,013 (median = 1,812; LCL 1,331, UCL 2,604) were similar to the estimate that used the lower survival for two or three angling sessions. The estimate for 2013 that used all three angling sessions (median = 1,589 fish) was only marginally higher than the estimate that used the two 2012 angling sessions and the 2013 netting session (median = 1,562 fish; LCL 1,113, UCL 2,337). When catches across the two 2012 angling sessions were combined and the estimate made with the 2013 netting session, the population abundance for 2013 (median = 1,609 fish; LCL 983, UCL 2,971) was slightly higher than that for three angling sessions (median = 1,589 fish) and that for two angling sessions and the 2013 netting session (median = 1,562 fish).

Table 11.4-9 Estimated Survival, Catchability, and Population Abundance of Lake Trout in Snap Lake in 2012 and 2013 Based on Four Different Bayesian Population Estimation Models

Model	Variable	Mean	SD	95% LCL	Median	95% UCL
1	Survival	0.722	0.045	0.631	0.723	0.805
	Catchability – angling	0.067	0.012	0.045	0.066	0.093
	Population estimate – 2012	1,707	292	1,238	1,671	2,379
	Population estimate – 2013	1,625	294	1,151	1,589	2,299
2	Survival	0.9	0	0.9	0.9	0.9
	Catchability – angling	0.060	0.011	0.040	0.059	0.083
	Population estimate – 2012	1,881	326	1,360	1,841	2,633
	Population estimate – 2013	1,852	326	1,331	1,812	2,604
3	Survival	0.713	0.046	0.619	0.714	0.799
	Catchability – angling	0.057	0.012	0.036	0.056	0.084
	Catchability – netting	0.061	0.012	0.040	0.060	0.088
	Population estimate – 2012	1,690	313	1,205	1,647	2,420
	Population estimate – 2013	1,605	316	1,113	1,562	2,237
4	Survival	0.713	0.046	0.619	0.715	0.800
	Catchability – netting	0.056	0.017	0.029	0.055	0.093
	Population estimate – 2013	1,708	519	983	1,609	2,971

SD = standard deviation; 95% LCL = 95% lower credibility limit, 95% UCL = 95% upper credibility limit.

11.4.6.3 Water Quality

Water Temperature

Sonde Measurements: July 2012 and July 2013

The thermal structure of Snap Lake in July 2012 differed from July 2013 and these differences were evident in both the main basin and the northwest arm (see Appendix 11.4C for a summary of all water quality data collected in 2012 and 2013). Water temperature in July 2012 in the northwest arm was above 15°C (the upper limit of temperature suitability for Lake Trout; Plumb and Blanchfield 2009) but only near the surface whereas, in July 2013, temperatures at or above 15°C occurred at depths of 2 m and shallower (Figure 11.4-5). In 2013, the average depth of the thermocline (e.g., zone of greatest change in temperature with change in depth) in the northwest arm was 6 m, almost 4 m shallower than in 2012 (12 m). Coupled with the shallower thermocline in 2013 was a colder hypolimnion with temperatures that averaged 1.5°C colder in 2013 than in 2012. For the main basin of Snap Lake, water temperatures during July were similar from the surface to 16 m depth during both 2012 and 2013 sampling sessions (Figure 11.4-5). The approximate depth of the thermocline in 2012 (22 m) was approximately 5 m deeper than in 2013 (17 m). Water temperature in the hypolimnion was approximately 1°C colder in 2012 (3.5°C) than in 2013 (4.5°C).

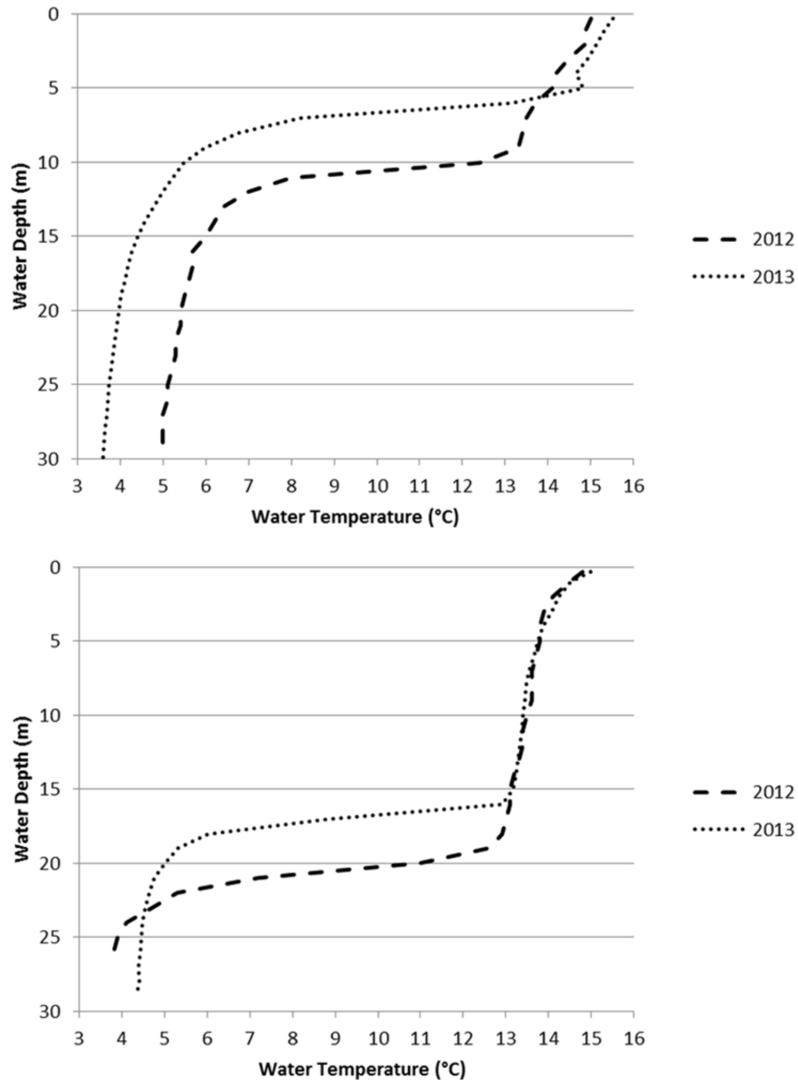
Fixed Mooring Measurements: July to September 2013

The much broader (July to September) and more continuous (24 hours per day) timeline provided by the vertical array of temperature loggers in the northwest arm and main basin provided a more complete picture of the temperature dynamics in Snap Lake although only general weekly trends are reported here. The data are presented in Appendix 11.4D; a detailed analysis can be found in Section 2. The implications to Lake Trout are discussed in Section 8.

In the northwest arm, thermal stratification was a persistent feature throughout the period of observation with limited change in the location of the thermocline, which ranged in depth from approximately 7 to 10 m (Figure 11.4-6). From approximately July 14 onward, there was little change in temperature below the 10 m depth interval. During this time period, water temperature averaged approximately 13°C at 10 m and declined to approximately 4°C at 39 m, the maximum depth of the northwest arm. Surface temperature (less than 5 m depth) increased through the period of observation and exceeded 15°C for much of August.

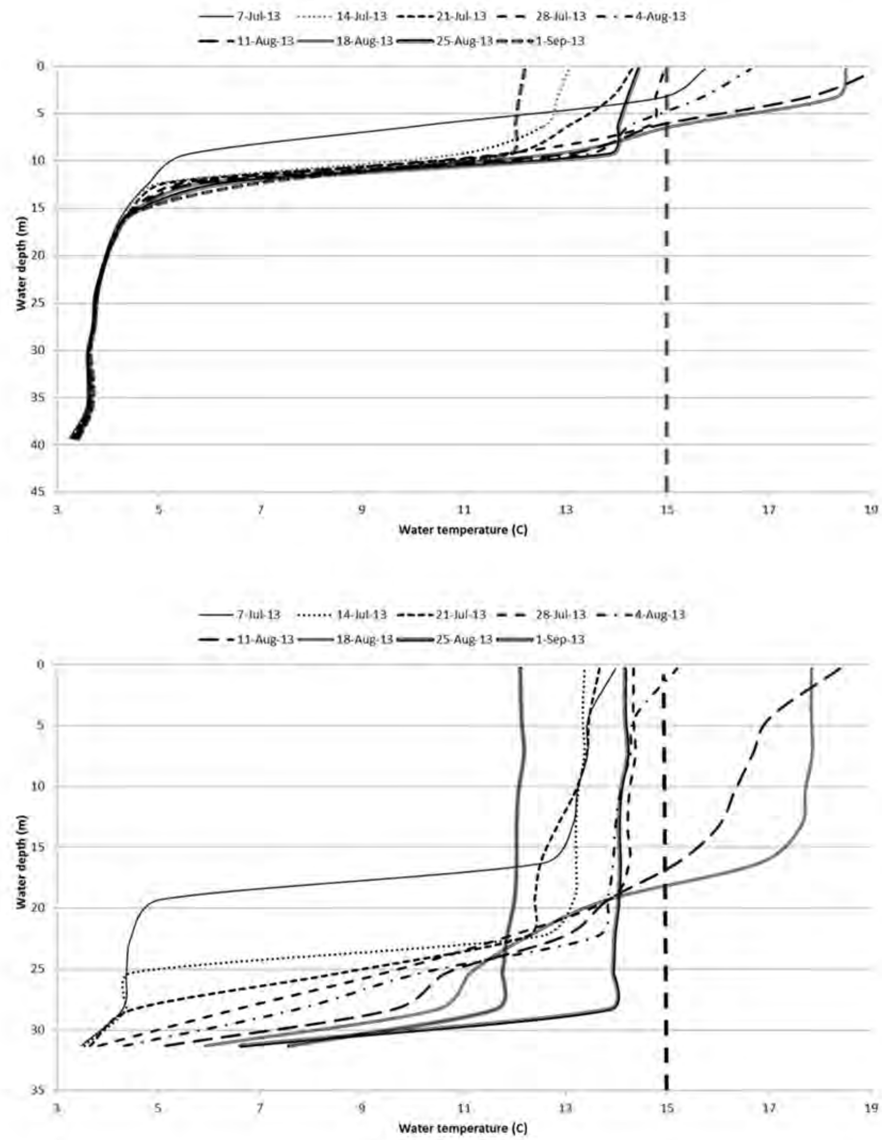
In the main basin, there was a marked deepening in the thermocline from a depth of approximately 17 m on July 7, to close to bottom (31.0 m) by August 18, when the greatest temperature change occurred within 5 m of the bottom (Figure 11.4-6). The upper mixed layer underwent progressive warming as well as continuous deepening throughout the summer. In mid-August, the temperature of the mixed surface waters of the main basin was in excess of 15°C.

Figure 11.4-5 Water Temperature Profiles of the Northwest Arm (upper panel) and the Main Basin (lower panel) of Snap Lake, July 2012 and July 2013



m = metre; °C = degrees Celsius.

Figure 11.4-6 Open Water Temperature Profile Collected from a Vertical Array of Temperature Loggers in the Northwest Arm (upper panel) and Main Basin (lower panel) of Snap Lake, July to September 2013



Note: A 15°C isotherm shown as a reference (dashed vertical line).

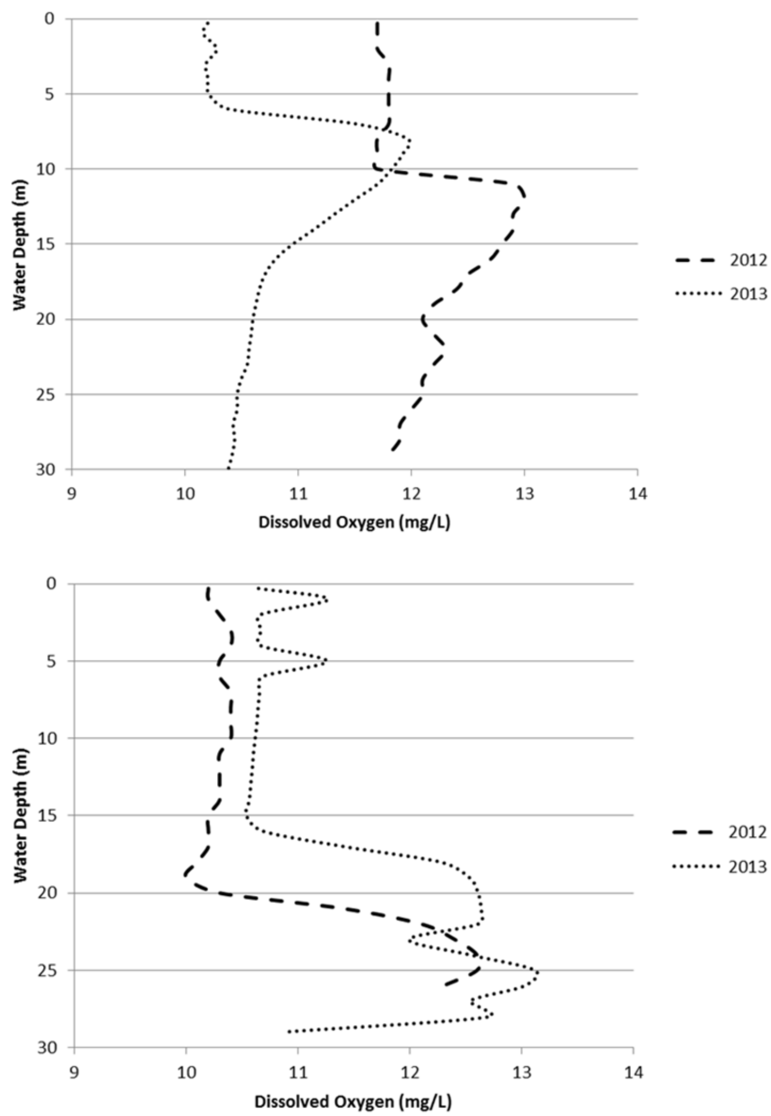
m = metre; °C = degrees Celsius; Jul; = July; Aug = August; Sep = September.

Dissolved Oxygen

Dissolved oxygen profiles varied between the northwest arm (Figure 11.4-7) and the main basin of Snap Lake, although general patterns were similar across 2012 and 2013 within the two zones. Dissolved oxygen profiles for the northwest arm in July 2012 and July 2013 were similar in that they showed a distinct elevation at mid-water depths. However, DO concentrations in 2012 were approximately 1.5 to 2 mg/L higher at all depths compared to 2013, although in both years DO was consistently at or above

7 mg/L, a concentration below which Lake Trout are negatively affected (Evans 2007). In contrast to the vertical profile evident in the northwest arm, in the main basin there was a distinct elevation in DO concentrations near bottom in both 2012 and 2013. Unlike the northwest arm, DO in the main basin was consistently lower in 2012 compared to 2013 by approximately 0.5 mg/L, but still at or above 7 mg/L.

Figure 11.4-7 Dissolved Oxygen Profiles for the Northwest Arm (upper panel) and Main Basin (lower panel) of Snap Lake, July 2012 and 2013

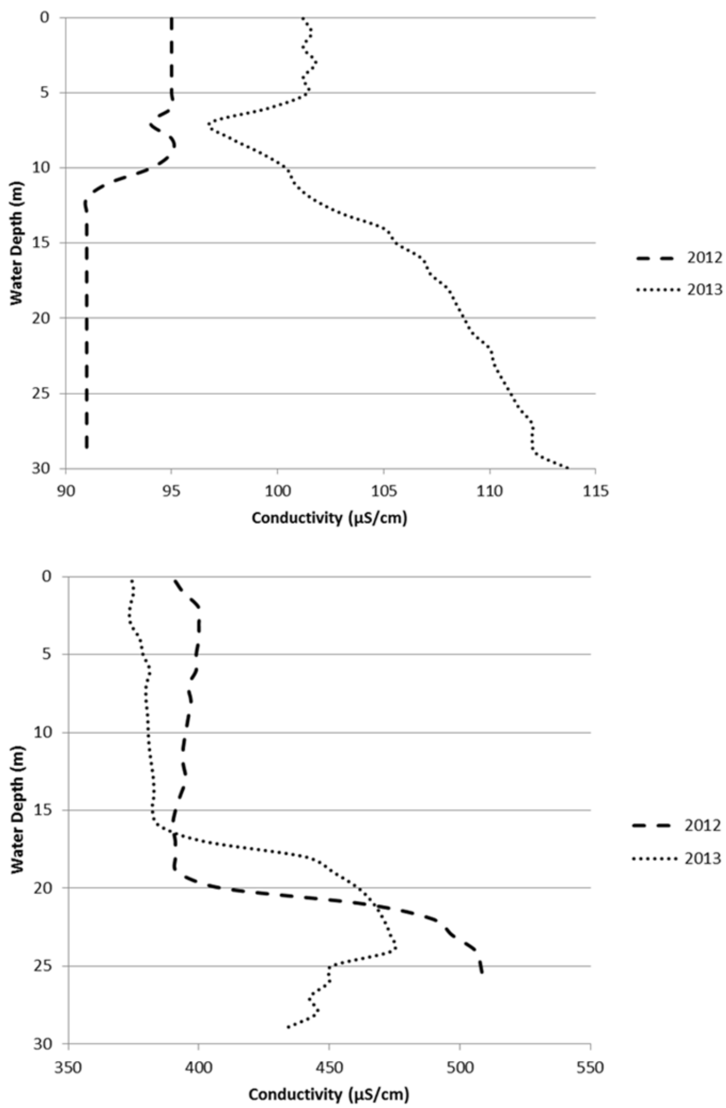


m = metre; mg/L = milligrams per litre.

Conductivity

Conductivity measured in the northwest arm in both July 2012 and July 2013 was less than 115 µS/cm regardless of depth (Figure 11.4-8); for the main basin, conductivity was above 350 µS/cm throughout the water column for both July 2012 and July 2013 (Figure 11.4-8). This represents over a three-fold difference between the two areas. In both July 2012 and July 2013, there was evidence of negative stratification (e.g., increased conductivity with depth) in the main basin. In the northwest arm; however, negative stratification was only evident in 2013; for 2012, mild positive stratification was evident. Based on the thermal structure in the lake, increases in conductivity appear to be associated with the hypolimnion.

Figure 11.4-8 Conductivity Profiles for the Northwest Arm (upper panel) and Main Basin (lower panel) of Snap Lake, July 2012 and 2013



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

11.4.7 Discussion

The LCL and UCL of the Snap Lake Lake Trout population abundance estimate were 1,151 and 2,299, respectively, which represents an approximate 40% ± variance around the median abundance estimate of 1,589 Lake Trout. This high variance reflects the low median recapture rate throughout the Study (6.7% of fish recaptured between marking sessions; LCL = 4.5%, UCL = 9.3%). The low catchability may reflect the active collection method used in the Study that was more spatially and temporally limited when compared to passive gear such as gill nets. However, the catchability for gill nets (5.6%, LCL 2.9%, UCL 9.3%) was similar to that for angling. Angling was necessary to avoid the known higher instantaneous and potential higher post-release mortality associated with the capture of Lake Trout using gill nets. Although limited in scope the results of the overnight holding study did not indicate that the capture, handling, and tagging methods used in the Study resulted in any short-term mortality. In addition, any fish seriously injured by angling were easily recognized and were not marked and released. Conversely, gill netted fish often have internal injuries that are not readily apparent during tagging and may suffer extensive scale loss, factors that may result in mortality days or weeks after release. Increased levels of post-release mortality of marked fish would have resulted in an overestimate of the Lake Trout population.

The population abundance of Lake Trout in Snap Lake was relatively invariant both temporally and across different collection gears, which indicated high confidence in the Peterson estimates derived, and that the assumptions for that estimate were met. To meet the assumptions of a Peterson mark-recapture experiment, the initial intention of the Study program was to sample a relatively large area of the lake and capture and tag a sufficient number of Lake Trout that were representative of the population in Snap Lake. With a sufficiently large sample of fish at recapture and high catchability of marked fish, the larger the initial number of Lake Trout marked, the greater the precision of the population estimate. The specific assumptions of a Peterson mark-recapture experiment that are relevant to the present study design and potentially influence the population estimate derived are discussed below:

1. *Sampling must be random:* Differences in fish distribution between the mark and recapture sessions that required changes to the locations sampled and the methods used between sessions likely resulted in a violation of the random sampling assumption, although analysis of the data collected indicated the effect of this violation on the resultant estimate was likely low based on the similarity of Lake Trout abundance estimates for 2012 when Lake Trout were highly aggregated to those of 2013 when Lake Trout were by contrast highly disaggregated (see Section 11.4.6.2).

2. *Marked fish must mix completely with unmarked fish between sampling events and every individual must have an equal probability of capture:* The differences in sample design and methods between sampling sessions should not have violated these assumptions. It was assumed that both marked and unmarked fish undertook the same redistribution patterns in response to the changes in temperature regimes that followed a sampling session. Although there appeared to be limited movement between the northwest arm and main basin, Lake Trout were collected from both areas during all three sampling sessions. Consequently, this assumption was met.

3. *Capture, handling, and marking does not result in post-release mortality of marked fish*: The selection of angling and the use of the in-water holding trough during tagging were specifically designed to reduce post-tagging mortality. Results of the post-tagging holding study did not indicate the occurrence of post-release mortality. This assumption was met.

4. *Every fish has an equal probability of being captured during the subsequent sampling event(s):* Angling has been shown to cause subsequent hook avoidance that could affect the probability of equal capture of marked and unmarked fish. However, this is a transitory effect (Waters 1960; Askey et al. 2006) that in the present study was reduced to the degree possible (as dictated by the sampling window) by having at least three weeks between angling sessions in 2012 and one year between the 2012 and 2013 sessions. Angling avoidance has also been associated with prolonged post-capture holding periods (Askey et al. 2006); however, in the present study fish were generally released within one minute of capture. This assumption was met.

5. *The ratio of marked to unmarked animals must not change:* Common causes of violations of this assumption are due to immigration, emigration, loss of marks, mortality, and recruitment. Considering the Snap Lake system is essentially closed, immigration and emigration were not considered as sources of bias. Although a proportion of marked fish appeared to lose their PIT tags, the use of a secondary mark (e.g., adipose fin clip) allowed the identification of these individuals for the purposes of the model. To the degree possible, potential mortality and recruitment effects were incorporated into the Bayesian model and this assumption was met.

Lake Trout are a cold water stenotherm, and although there is considerable variation with respect to Lake Trout thermal habitat use, temperatures above 15°C appear to be unsuitable for extended periods based on telemetry (Plumb and Blanchfield 2009). Lake Trout are known to behaviorally thermoregulate when thermal conditions become limiting and they move to habitats offering thermal refuge (Snucins and Gunn 1995; Mackenzie-Grieve and Post 2006). The thermal profile of Snap Lake likely had a strong influence on the vertical and spatial distribution of Lake Trout. During the 2012 marking sessions, surface water temperatures were relatively warm and almost all Lake Trout were captured within two relatively small areas that represented the deepest and coldest areas of the lake. In 2013, surface water temperatures were cooler and fish were more widely distributed throughout the lake. However, their distributions in both years were likely representative of the Lake Trout population in Snap Lake as a whole and did not result in a substantive bias in the population estimate.

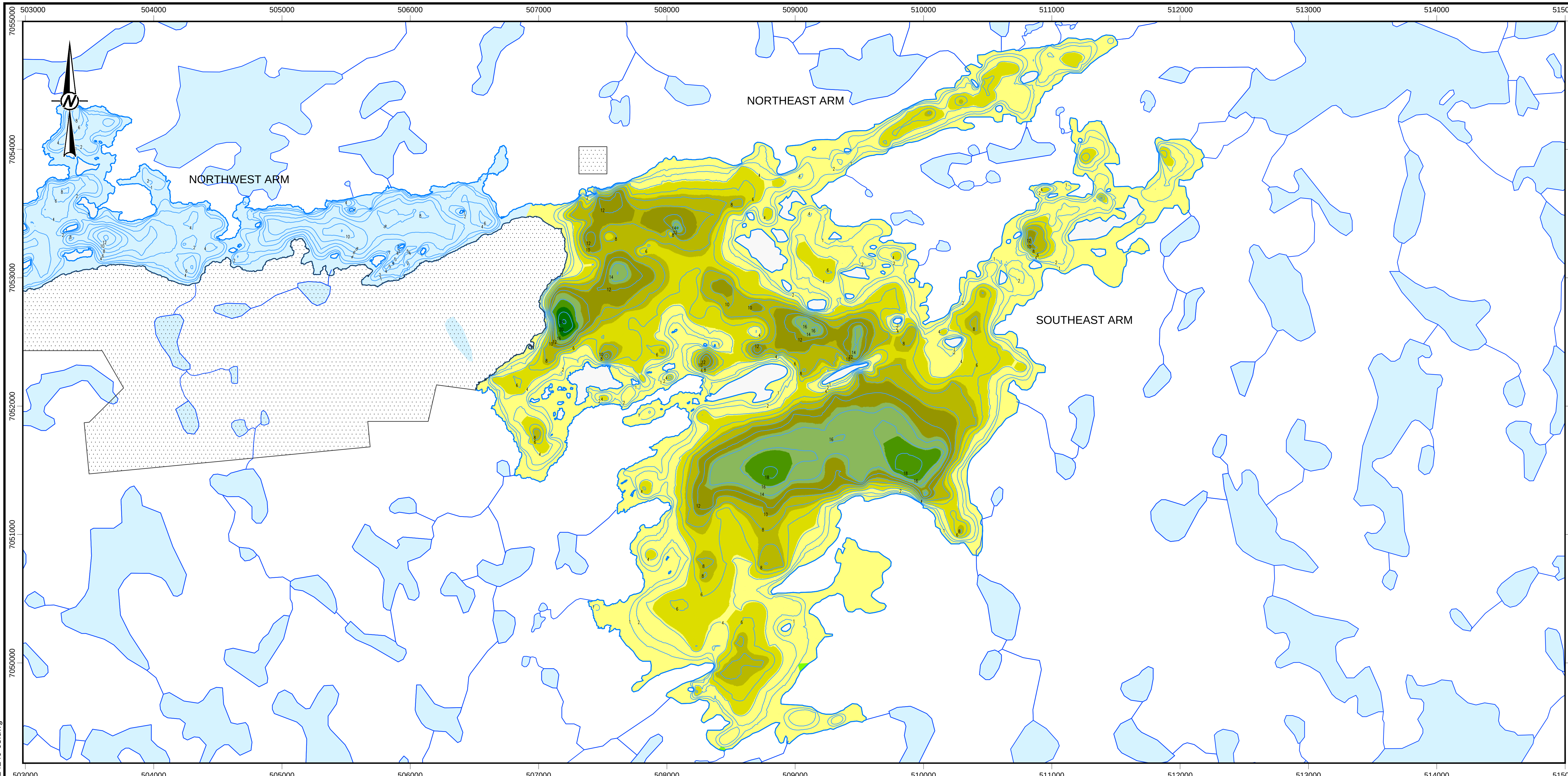
The distribution of Lake Trout in Snap Lake was not apparently influenced by DO levels as these were always close to full saturation throughout the lake and well above critical thresholds. Conductivity did not appear to have an influence on fish distribution as much higher concentrations than those recorded in either basin appear to be required to cause avoidance for fish (Pimentel and Bulkley 1983).

Relative to the area of Snap Lake (1,600 ha), the median estimate of 1,589 fishable (greater than 250 mm FL) Lake Trout represents a relatively low abundance on a per unit area basis (1.0 LTH^{-1}) compared to other Lake Trout lakes in the reviewed literature. For eight unexploited lakes in the Experimental Lakes Area of northwestern Ontario, Lake Trout abundance ranged from 7.6 to 23.8 LTH, whereas for one heavily exploited lake, which also contained Northern Pike (*Esox lucius*), abundance was 0.6 LTH (Mills et al. 2002). This low abundance in lakes with high exploitation rates was similar to that reported for Crean Lake (0.17 LTH) where exploitation was high due to a commercial fishery (Melville 2005). The lakes reported in Mills et al. (2002) are much smaller (16 to 54 ha) than Snap Lake (1,600 ha). Payne et al. (1991) suggested that Lake Trout densities are higher in small lakes than large lakes. For example, Lake Trout abundance in Pend Oreille Lake (38,300 ha) was 0.93 LTH although, in this instance, there was an unspecified amount of commercial exploitation (Hansen et al. 2008). For western Lake Superior, the largest lake by area in the world, Lake Trout abundance averaged 7.6 LTH (Nieland 2006). Although fish productivity is expected to be lower in northern boreal lakes with lower overall primary and secondary productivity (Downing and Plante 1993), there are few estimates of Lake Trout population size for lakes in the Northwest Territories and the methods used are not comparable with those used in the Study. At similar latitudes Burr (1997) reported that abundance of mature Lake Trout ranged from 3.1 to 32.8 LTH for six unexploited Alaskan lakes and 0.6 LTH for one Alaskan lake with high exploitation.

The low density of Lake Trout in Snap Lake is unlikely to reflect exploitation since the only source of exploitation is the periodic assessments connected with the Snap Lake AEMP. Based on the BsM netting program, Lake Trout CPUE in Snap Lake was higher than CPUEs in two adjacent reference lakes (Northeast Lake and Lake 13) and one of these lakes (Lake 13), has had virtually no exploitation (see Section 8). A more likely explanation is related to the limited amount of summer habitat in the lake that is suitable for this species. Christie and Regier (1988) reported that the sustained yield of Lake Trout for a series of north-temperate lakes was related to summer measures of either the amount of thermal habitat area or thermal habitat volume that was within the preferred thermal niche (8°C to 12°C) of Lake Trout. Based on data obtained from a vertical temperature logger array in Snap Lake in 2013, there was a gradual deepening in the depth of the 15°C isotherm in the main basin. As a result, for a short period in mid-July less than 5% of the lake volume was less than 15°C and, for a three week period in August, less than 1% of the lake volume was less than 15°C. With decreasing depth, water temperatures became unsuitable (greater than 15°C) for increasing periods of time during the summer (Figures 11.4-9 and 11.4-10). It was only for deeper areas close to the diffuser in the main basin (Figure 11.4-9) and a deep hole at the westernmost portion of the northwest arm (Figure 11.4-10) that there were no restrictions on the thermal suitability of habitat for Lake Trout. By comparison, for a Lake Trout lake in northwestern Ontario, at least 20 to 40% of the lake volume fell within the less than 15°C benchmark over a two year period (Plumb and Blanchfield 2009). According to temperature measurements made with sondes in 2012 and 2013, Snap Lake temperatures were cooler in 2013 than 2012 such that availability of water less than 15°C may be even less during warmer years than was observed in 2013. Whether water temperatures in 2013 were “normal” is unclear. For Snap Lake, during the seven year period 2006 to 2013, the depth of the 15°C isotherm in early August (prior to August 15) showed some fluctuation but 2013 did not appear to be anomalous; there was no water at less than 15°C in the water column in two of the seven years but for the other five years, the 15°C isotherm was located within 3 m of the bottom of the lake as occurred in 2013. Air temperatures in the Canadian north are on a long-term warming trend and, as lake temperature is correlated with air temperature, warmer lake temperatures can also be expected (Schindler et al. 1990, 1996).

Median survival of Lake Trout in Snap Lake based on fin ray ages and calculated from the descending limb of the August 2012 catch curve was 72.2%$\cdot$year^{-1} (range = 63.0% to 80.5%) Although survival was relatively high, this value was still less than that reported for other populations having low or no exploitation. For nine lakes in the Experimental Lakes Area of northwestern Ontario, Lake Trout survival averaged 83%$\cdot$year^{-1} and ranged from 69 to 91%$\cdot$year^{-1}, with the lowest survival (69%) associated with high experimental exploitation and the presence of Northern Pike (Mills et al. 2002). Shuter et al. (1998) reported a range in survival of 78% to 89%$\cdot$year^{-1} for several unexploited Lake Trout stocks in southern Ontario.

The mark recapture study conducted in Snap Lake provided an absolute estimate of Lake Trout abundance and provides a benchmark against which future estimates of absolute abundance of Lake Trout can be compared. However this estimate is only meaningful for Lake Trout greater than 250 FL and bears no relationship to the abundance of other fish species in Snap Lake.



L:\2013\1349\13-1349-0001\phase 4200\task 4240\13134900014240-03.dwg

LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- * The number of days that areas based on depth (m) in Snap Lake main basin are considered unsuitable (>15°C) thermal habitat for Lake Trout between 6 July and 7 September, 2013.

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).

Elevations Table

# of Unsuitable Days*	From Depth	To Depth	Color
23-26	0	4.3	
22	4.3	7.3	
20	7.3	10.3	
18	10.3	13.3	
14	13.3	16.3	
4	16.3	19.3	
0	19.3	43	

SNAP LAKE MINE

Thermal Suitability of the Main Basin of Snap Lake for Lake Trout (July 6 to September 7, 2013)

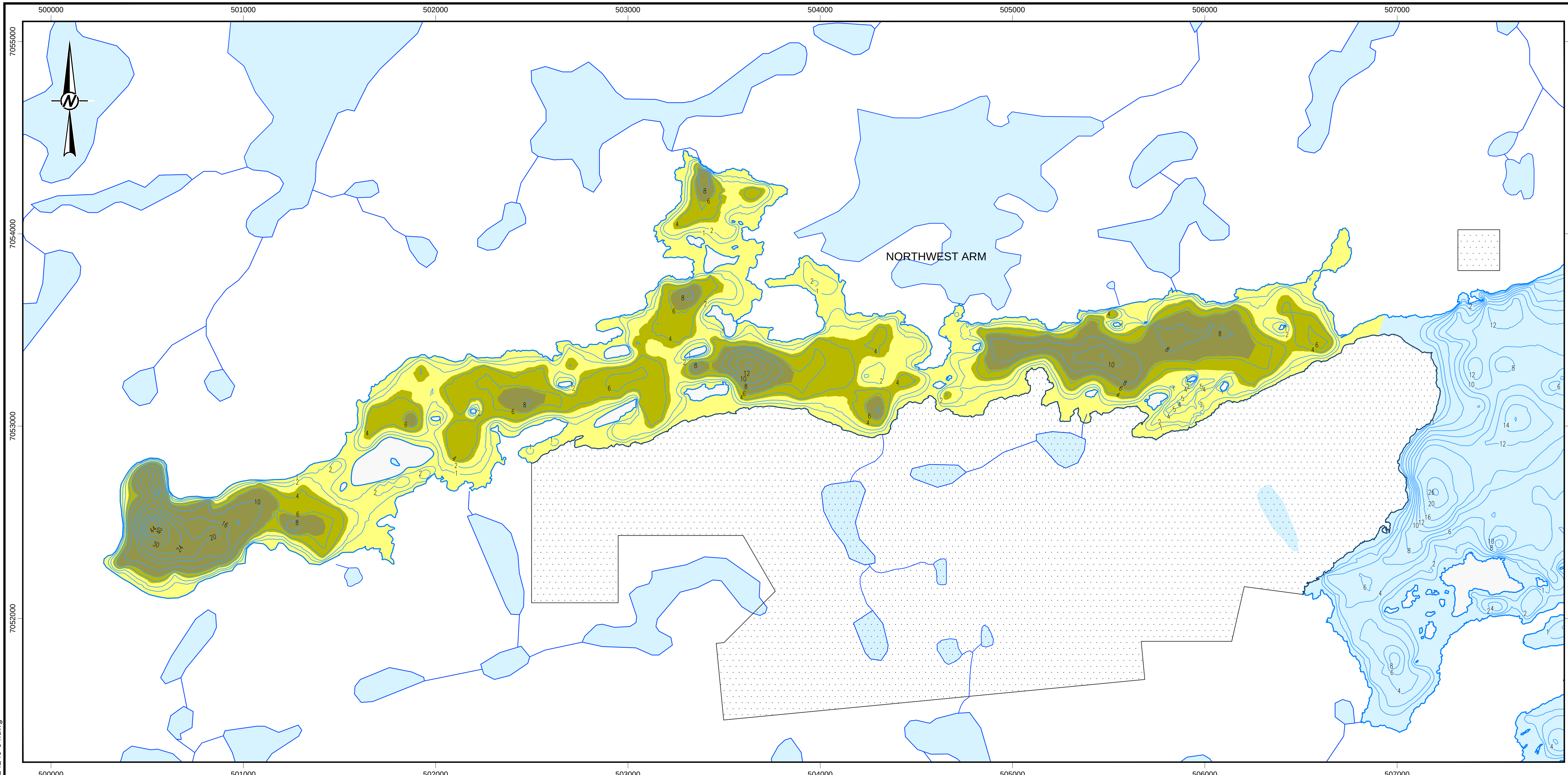
PROJECTION: UTM Zone 12
DATUM: NAD83

500 0 500
SCALE 1:30,000 METRES

DE BEERS
GROUP OF COMPANIES

FILE No: 13134900014240-03
DATE: December 10, 2013
JOB No: 14-1349-0004
REVISION No: A
OFFICE: GOLDER VICTORIA
DRAWN: JEF
CHECK: LH

Figure 11.4-9



LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint
- * The number of days that areas based on depth (m) in Snap Lake Northwest arm are considered unsuitable (>15°C) thermal habitat for Lake Trout between 6 July and 7 September, 2013.

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).

Elevations Table

# of Unsuitable Days*	From Depth	To Depth	Color
23-27	0	3.3	
11	3.3	6.3	
0	6.3	43	

SNAP LAKE MINE

Thermal Suitability of the Northwest Arm of Snap Lake for Lake Trout (July 6 to September 7, 2013)

PROJECTION: UTM Zone 12	DATUM: NAD83
400 0 400 SCALE 1:20,000 METRES	DE BEERS GROUP OF COMPANIES
FILE No: 13134900014240-04	DATE: December 2, 2013
JOB No: 14-1349-0004	REVISION No: A
OFFICE: GOLDER VICTORIA	DRAWN: JEF CHECK: LH

Figure 11.4-10

L:\2013\1349\13-1349-0001\phase 4200\task 4240\13134900014240-04.dwg

11.4.8 Conclusions

The objectives of the Study were met as follows:

- The Study resulted in the capture, marking, and live releasing of 295 Lake Trout from Snap Lake. Although this was less than the target of 400 Lake Trout, the number marked was sufficient to warrant the implementation of the 2013 recapture component.
- The Study used an active fishing method (e.g., angling) for capturing fish and a fish processing protocol that had low direct and residual mortality. Individual marked fish were readily identified on recapture; the loss of marks was low and independently verified by an adipose fin clip.
- In the 2013 recapture session, fishing was completed in all parts of the lake so all fish had an equal opportunity for capture and fish were captured from all parts of the lake where fish were distributed Although the target of 700 fish was not met, the 117 Lake Trout captured in 2013 and the 16 recaptures were sufficient to estimate the abundance of Lake Trout and the level of confidence surrounding this estimate.

Despite the relatively low productivity of Lake Trout in Snap Lake, the present Study indicated it is possible to obtain an estimate of Lake Trout population abundance in Snap Lake using a moderate level of effort. The credibility interval associated with this estimate, while relatively wide (± 40%), should be sufficiently narrow to allow an assessment of the effects of mortality arising from the standardized gill net assessment (BsM) monitoring method (see Section 8).

Analysis of information provided by the Study addressed the following key question:

How many Lake Trout of fishable size (greater than 250 mm FL), are estimated to be in Snap Lake and what is the level of confidence of that estimate?

11.4.8.1 Key Question 1: How many Lake Trout of fishable size (greater than 250 mm FL) are estimated to be in Snap Lake, and what is the level of confidence in that estimate?

The median estimate of Lake Trout of fishable size (greater than 250 mm FL) in Snap Lake in 2012 was 1,589 with 95% credibility interval between 1,151 and 2,299, respectively. Accordingly there is a 95% probability that population abundance is between 1,151 and 2,299 fish.

11.4.9 References

Askey PJ, Richards, SA Post, JR, Parkinson, EA. 2006. Linking angling catch rates and fish learning under catch-and-release regulations. N Am J Fish Manage 26: 1020–1029.

Burr JM. 1997. Growth, density, and biomass of lake trout in Arctic and subarctic Alaska. In Reynolds J (ed), Fish Ecology in Arctic North America. Amer Fish Soc Symp 19: 109-118. Bethesda, MD, USA.

CBFWA (Columbia Basin Fish and Wildlife Authority PIT Tag Steering Committee). 1999. PIT Tag Marking Procedures Manual Version 2.0. Seattle, WA, USA

Christie GC, Regier HA. 1988. Measures of optimal thermal habitat and their relationship to yields of four commercial fish species. Can J Fish Aquat Sci 45: 301-314.

De Beers (De Beers Canada Inc). 2006. 2005 Annual Report in Support of the Aquatic Effects Monitoring Program (AEMP) Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2007. 2006 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2008. 2007 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2009. 2008 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2010. 2009 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2011. 2010 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012a. Aquatic Effects Re-evaluation Report. Aquatic Effects Monitoring Program for the Snap Lake Project. Water Licence (MV22001L2-0002). Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2012b. 2011 Annual Report in Support of the Aquatic Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

De Beers. 2013. 2012 Annual Report in Support of the Aquatic Effects Monitoring Program Water Licence (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

Downing JA, Plante C. 1993. Production of fish populations in lakes. Can J Fish Aquat Sci 50: 110–120.

Evans DO. 2007. Effects of hypoxia on scope-for-activity and power capacity of lake trout (*Salvelinus namaycush*). Can J Fish Aquat Sci 64: 345–361.

Hansen M J, Horner NJ, Liter M, Peterson MP, Maiolie MA. 2008. Dynamics of an increasing lake trout population in Lake Pend Oreille, ID, USA. N Am J Fish Manage 28: 1160–1171.

Kery M. 2010. Introduction to WinBUGS for ecologists. A Baeysian approach to regression, ANOVA, mixed models and related analyses. Academic Press, Burlington, MA, USA.

Lum L. 2013. Calculation of fish weight from girth. See http://www.downtimecharters.com/calc-fishweight-lum.htm. Accessed November 2013.

Mackenzie-Grieve JL, Post JR. 2006. Thermal habitat use by lake trout in two contrasting Yukon territory lakes. Trans Am Fish Soc 135: 727-738.

MVLWB (Mackenzie Valley Land and Water Board). 2013. Mackenzie Valley Land and Water Board Water Updated Type A Water Licence MV2011L2-0004. Yellowknife, NWT, Canada.

Melville GE. 2005. Development of an assessment approach for remnant lake salmonid populations: the Crean Lake example. Aquat Living Resour 18: 313–317.

Mills KH, Chalanchuk SM, Allan DJ. 2002. Abundance, annual survival, and recruitment of unexploited and exploited lake char, *Salvelinus namaycush*, populations at the Experimental Lakes Area, northwestern Ontario. Env Biol Fish 64: 281–292.

Nieland JL. 2006. Modeling the sustainability of lake trout fisheries in eastern Wisconsin waters of Lake Superior. MSC thesis, University of Wisconsin, Madison, WI, USA.

Payne NR, Korver RM, MacLennan DS, Nepszy SJ, Shuter BJ, Stewart TJ, Thomas ER. 1991. The harvest potential and dynamics of lake trout populations in Ontario. Lake Trout Synthesis: Population Dynamics Working Group. Ontario Ministry of Natural Resources. Toronto, ON, Canada

Pimentel R, Bulkley RV, 1983. Concentrations of total dissolved solids preferred or avoided by endangered Colorado river fishes. Trans Am Fish Soc 112: 595-600.

Pine WE, Pollock KH, Hightower JE, Kwak TJ, Rice JA. 2003. A review of tagging methods for estimating fish population size and components of mortality. Fisheries 28: 10–23.

Plumb JM, Blanchfield PJ. 2009. Performance of temperature and dissolved oxygen criteria to predict habitat use by lake trout (*Salvelinus namaycush*). Can J Fish Aquat Sci 66: 2011-2023.

Plummer M. 2003. JAGS: a program for analysis of Bayesian graphical models using Gibbs sampling, Proceedings of the 3rd International Workshop on Distributed Statistical Computing (DSC 2003), March 20–22, Vienna, Austria. ISSN 1609-395X.

Plummer M. 2013. rjags: Bayesian graphical models using MCMC. R package version 3-10. http://CRAN.R-project.org/package=rjags

R Development Core Team 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Robson DS, Regier HA. 1964. Sample size in Petersen mark-recapture experiments. Trans Am Fish Soc 93: 215-226.

Sandstrom S. 2013. Biologist. Ontario Ministry of Natural Resources, Bracebridge, ON, Canada. Telephone conversation. June 15, 2013.

Sandstrom S, Rawson M, Lester N. 2009. Manual of instructions for broad-scale fish community monitoring; using North American (NA1) and Ontario Small Mesh (ON2) gillnets. Ontario Ministry of Natural Resources, Peterborough, ON, Canada. Version 2011.1 35 pp + appendices.

Sullivan S. 2013. Fisheries science specialist. Alberta Government, Edmonton, AB, Canada. E-mail. September 12, 2013.

Schindler DW, Beaty KG, Fee, EJ, Cruikshank DR, Debruyn ER, Findlay DL, Linsey GA, Shearer JA, Stainton MP, Turner MA. 1990. Effects of climatic warming on lakes of the central boreal forest. Science 250(4983): 967–970.

Schindler DW, Bayley SE, Parker BR, Beaty KG, Cruikshank DR, Fee EJ, Schindler EU, Stainton MP. 1996. The effects of climatic warming on the properties of boreal lakes and streams at the Experimental Lakes Area, northwestern Ontario. Limnol Oceanogr 41: 1004–1017.

Shuter BJ., Jones ML, Korver RM, Lester NP. 1998. A general life history based model for regional management of fish stocks: the inland lake trout (*Salvelinus namaycush*) fisheries of Ontario. Can J Fish Aquat Sci 55: 2161-2177.

Snucins EJ, Gunn JM. 1995. Coping with a warm environment: behavioral thermoregulation by lake trout. Trans Am Fish Soc 124: 118–123.

Waters TF. 1960. The development of population estimate procedures in small trout lakes. Trans Am Fish Soc 9: 287-294.

SECTION 11.5

STABLE ISOTOPE FOOD WEB ANALYSIS SPECIAL STUDY

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ACRONYMS

Term	Definition
ANCOVA	analysis of covariance
BsM	broad scale community assessment protocol
BURB	Burbot
C	carbon
C:N	carbon-nitrogen ratio
CADD	caddisflies
CHIR	chironomids
CI	confidence interval
EPHE	Ephemeroptera
FING	fingernail clams
GAST	Gastropoda (snails)
LKTR	Lake Trout
LNSC	Longnose Sucker
MANCOVA	multivariate analysis of covariance
Mine	Snap Lake Mine
N	nitrogen
n	number
OLIG	oligochaetes
PERI	periphyton
PHYT	phytoplankton
P-value	statistical significance
QA	quality assurance
QC	quality control
r	correlation coefficient
RDCT	R Development Core Team
RNWF	Round Whitefish
SD	standard deviation
SEAb	standard ellipse areas
SIAR	stable isotope analysis in R (SIAR)-A statistical package that takes data on organism isotopes and fits a Bayesian model to their dietary habits based upon a Gaussian likelihood
UTM	Universal Transverse Mercator
ZOOP	zooplankton

UNITS OF MEASURE

Term	Definition
±	plus or minus
,	less than
°C	degree Celsius
%	percent
‰	per thousand or per mil
cm	centimetre
g	gram
km	kilometre
L	litre
m	metre
mg	milligram
mL	millilitre
mm	millimetre
m/s	metres per second
µm	micrometre

11.5 Stable Isotope Food Web Analysis Special Study

11.5.1 Introduction

Understanding trophic niches and the food web structure within which they reside, and the myriad of trophic interactions that can exist within a lake is essential for effective fisheries management and identifying the potential impacts of development in northern lakes where knowledge of the structure and function of aquatic food webs is often limited to non-existent. The trophic niche, which describes the relative positions of populations, species, or functional biotic groups in an ecosystem (Hutchinson 1957), can be sensitive to species interactions and habitat availability; collapses in niche size have been associated with loss of habitat resulting in ecosystem fragmentation (Layman et al. 2007). In such instances, species that are trophic generalists may be less susceptisble to changes in trophic structure or habitat availability because of an ability to shift among alternative food resources (i.e., they maintain a broad niche) or habitats, whereas other species may tend to be trophic specialists, feeding on fish as top predators (i.e., they have a more limited niche) possibly in a restricted habitat, despite being capable of feeding on a wide variety of prey items (e.g., invertebrates, fish). As a result they may be more susceptible to effects on their specialized food sources (Lepak et al. 2006). Understanding whether species in a lake are generalists or specialists is important for evaluating the potential impacts of development that may affect single or multiple parts of the food web depending on the relative sensitivities of its members.

Relatively little is known about trophic relationships among fish species in Snap Lake. In ecosystems, fish often play a pivotal role as integrators and controllers of littoral-benthic and pelagic food webs due to their high mobility, rapid behavioural responses, and flexible feeding on both benthic (e.g., aquatic insect larvae and molluscs) and pelagic (e.g., crustaceans, zooplankton) prey (Polis et al. 1997; Vadeboncoeur et al. 2002; Vander Zanden and Vadeboncoeur 2002). Coupling between benthic and pelagic food webs is particularly evident in small, unproductive high latitude lakes such as Snap Lake, where fish need to cope with limited and seasonally fluctuating food resources (Schindler and Scheuerell 2002; Christoffersen et al. 2008). The relative importance of benthic and pelagic components has not been established for Snap Lake; such information is important for understanding potential effects of the Snap Lake Mine (Mine).

In small, high latitude lakes, littoral-benthic food webs (i.e., the energy flow from benthic algae to benthivorous fishes via littoral-benthic macroinvertebrates) have frequently been shown to be of particular importance for the lakes' total production and ecosystem function (Welch and Kalff 1974; Hecky and Hesslein 1995; Sierszen et al. 2003; Karlsson and Bystrom 2005). Pelagic phytoplankton production in high latitude lakes tends to be constrained because of low dissolved nutrients, with the resulting clear water leading to extensive illuminated littoral areas suitable for photosynthetic benthic algae, although such a balance can be reversed by nutrient enrichment (Liboriussen and Jeppesen 2003; Vadeboncoeur et al. 2003).

Benthic invertebrate communities drive food chains in northern lakes (Sierszen et al. 2003), but availability of littoral-benthic species to fish may be affected by lake morphometry and the proximity of littoral habitat to coldwater habitat, particularly for coldwater stenotherms like Lake Trout (Dolson et al. 2009). Moreover, the relative importance of benthic invertebrates may, under conditions of eutrophication associated with excessive nutrient addition, decline in favour of a shift in primary productivity from benthic algae to pelagic phytoplankton. Such regime shifts can result in changes in biodiversity and food web structure given the importance of benthic energy pathways to a wide array of fish species (Vander Zanden and Vadenboncoeur 2002; Vadenboncoeur et al. 2003).

Understanding trophic relationships has, in the past, involved laborious collections of biota and extensive examination of stomach contents to establish patterns of spatial and temporal importance to the fish species occupying a lake. Such approaches are expensive to carry out and require very large collections to take into account the multiple sources of variation and as a result, can be a major source of mortality. As an alternative, stable isotope analyses of ratios of ^{15}N:^{14}N and ^{13}C:^{13}C in consumer tissues can provide information on long-term feeding behaviour and trophic interactions, trophic position and niche width and can differentiate between littoral-benthic and pelagic sourced energy (Jackson et al. 2011). Isotope ratios (i.e., $\delta^{15}N$ and $\delta^{13}C$) in muscle tissues of fish in temperate lakes principally reflect the food consumed during the spring and summer growth period (Perga and Gerdeaux 2005) and have been established as important tools for research on trophic niches (Bearhop et al. 2004). Although gut content analysis can be used to examine trophic structure and feeding relationships, it represents a very short period of foraging history, and often provides limited to no useful information towards understanding trophic interactions. This is because empty stomachs are common, assimilation efficiencies of prey items are unknown and may provide different contributions to growth of consumer tissues. Stable isotope analysis, in contrast, overcomes this limitation because it reflects a diet that has been assimilated into consumer tissue and can be compared with the $\delta^{15}N$ and $\delta^{13}C$ values of prey tissues to determine the relative importance of different prey items in the food web. As a result, a stable isotope approach can provide more accurate insights into diet, trophic structure, niche width and the relative importance of littoral-benthic and pelagic energy for the fish community in Snap Lake with a considerably smaller, less expensive, and less intrusive sampling effort. Thus, the stable isotope approach was used to answer the following two Key Questions about the aquatic food web at Snap Lake:

- **Key Question 1:** What eats what in Snap Lake?
- **Key Question 2:** Is the Snap Lake food web planktonically or benthically driven?

11.5.2 Methods

11.5.2.1 Study Site and Species

Snap Lake (63°26’20”N, 110°52'00 W) is a small (1,566 hectares [ha]), shallow (average depth 5 metres [m]), oligo- to meso-trophic lake located about 220 kilometres (km) north-east of Yellowknife, Northwest Territories in the Aylmer Lake/Lockhart River system (Figure 11.5-1). The lake is situated in the High Sub Arctic ecoclimatic zone. Mean daily temperatures range from the January low of -35 degree Celsius (°C) to the July high of 25°C (Section 2). The lake drains east, via a chain of lakes of variable size into the

Lockhart River system which eventually discharges into the eastern basin of Great Slave Lake. Snap Lake supports seven species of fish: one omnivore (Lake Chub, *Couesius plumbeus*) three benthivores (Longnose Sucker, *Catostomus catostomus*; Round Whitefish, *Prosopium cylindraceu*; and, Slimy Sculpin, *Cottus cognatus),* two piscivores (Burbot, *Lota lota* and Lake Trout, *Salvelinus namaycush*), and one insectivore (Arctic Grayling, *Thymallus arcticus*).

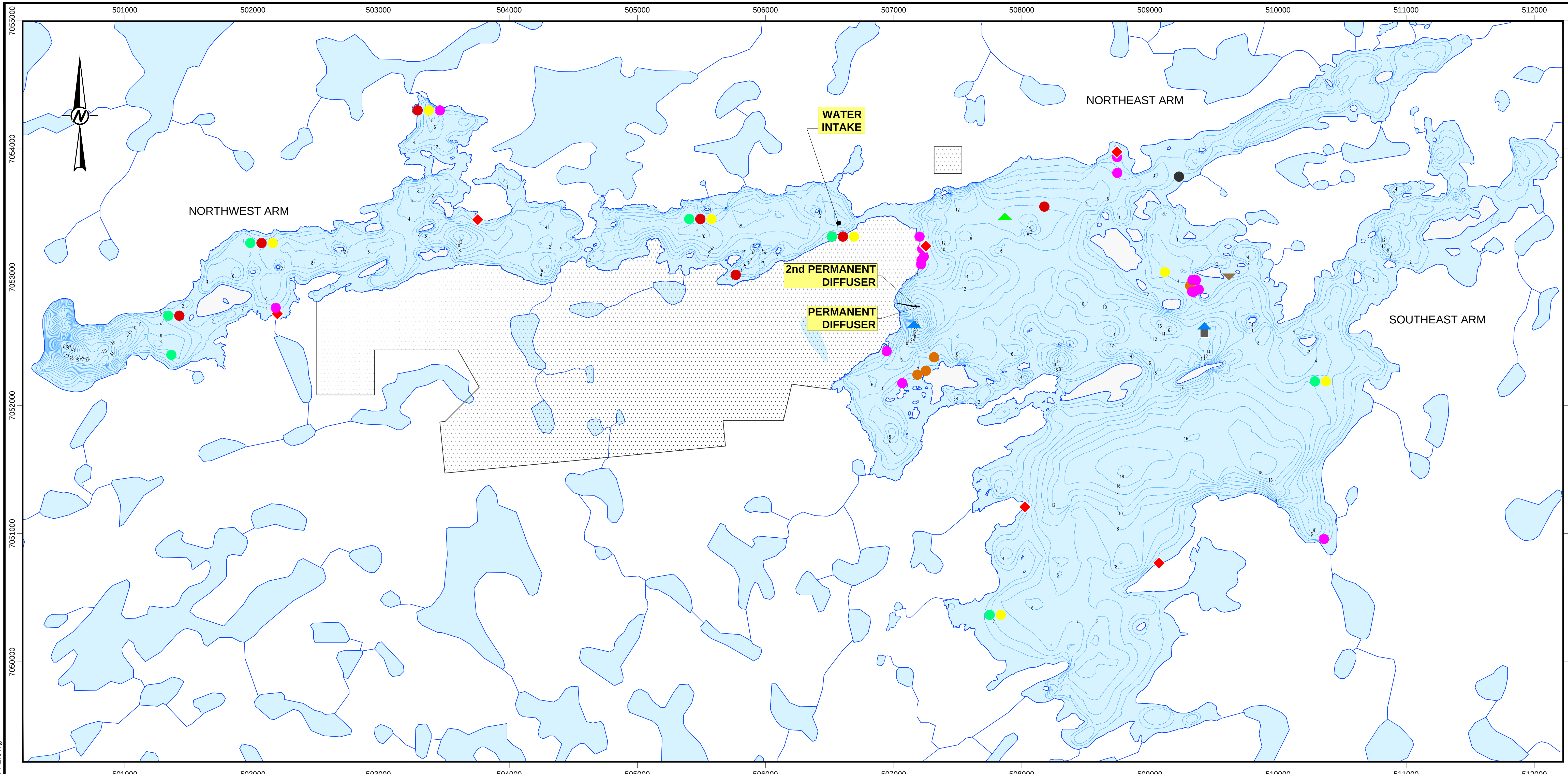
11.5.2.2 Sample Collection

Fish

Lake Trout, Longnose Sucker, Round Whitefish, and Arctic Grayling were collected from a range of depths (2.1 to 13.8 m, 1.0 to 10.1 m, 1.8 to 10.1 m, and 1.0 to 2.9 m, respectively) during the July 2013 Fish Community Monitoring Program using the Broad Scale Community Assessment Protocol (hereafter B_SM) (Sandstrom et al. 2011). Additional frozen archive samples of Lake Trout and Round Whitefish collected with gillnets in 1999 were also included to evaluate temporal changes. During 2013, fish were collected using small (13 to 38 millimetres [mm]) and large (38 to 127 mm) mesh gill nets. One Lake Trout was also collected in August using a set line in 5 m of water. Lake Chub were collected using a combination of methods to obtain a large enough sample, including the BsM sampling and minnow traps. Minnow traps containing wheat products were fished in August at depths ranging from 1.5 to 3 m. In addition, Burbot were collected using set lines at depths ranging from 2 to 5 m and baited with dead Cisco (*Coregonus artedii)* that had been salted to reduce the rate of breakdown. The sample sizes for each fish species were 10 for Lake Chub and Longnose Sucker, 11 for Lake Trout, 9 for Round Whitefish, 2 for Arctic Grayling, and 5 for Burbot. All fish were weighed and measured at the time of capture, and otoliths removed for age determination. A 0.5 milligram (mg) sample of skinless muscle tissue was removed from the epaxial musculature for stable isotope analysis, placed in a labeled vial, and kept in a freezer or on ice until it could be frozen with dry ice.

Zooplankton

A single zooplankton sample was collected at each of three AEMP sampling stations: SNP02-20E, SNAP06, and SNAP03. Each sample consisted of a composite of five vertical hauls. Haul depths ranged from 12 to 25 m. Zooplankton samples were collected using a 0.5 m diameter, 2.5 m long, 153 micrometre (µm) mesh plankton net. After each of the five vertical hauls, the net was rinsed from the outside in with lake water to prevent contamination with surface plankton and concentrate the sampled zooplankton in a dolphin-bucket attached at the cod end of the net. The material collected in the dolphin-bucket was then rinsed into a 500 millilitre (mL) bottle with distilled water, to prevent introduction of new zooplankton. Material collected from each of the five vertical hauls was combined into a single composite sample at each station in a 500 mL dark glass jar. The same procedure was carried out at each of the other two stations. Each composite zooplankton sample was transferred from the 500 mL bottle onto a piece of 153 µm Nitex fabric held in a Buchner funnel, and the water removed from the sample using a weak vacuum until the sample took on a semi-dry appearance. A coarse-level visual inspection of the samples suggested that they consisted mostly of copepods and cladocerans.



LEGEND

- Depth Contour (m)
- Watercourse
- Waterbody
- Snap Lake Mine Footprint

Invertebrates

- Periphyton
- Phytoplankton
- Zooplankton
- Littoral benthos
- Profundal benthos

Fish

- Lake Trout
- Round Whitefish
- Lake Chub
- Longnose Sucker
- Burbot
- Arctic Grayling

REFERENCES

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

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

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

SNAP LAKE MINE

Sampling Sites for Fish and Invertebrates Collected in 2013 for Stable Isotope Special Study

PROJECTION: UTM Zone 12	DATUM: NAD83		DE BEERS GROUP OF COMPANIES
500 0 500 SCALE 1:30,000 METRES			
FILE No: 1313490001-7010-11.4-1			DATE: March 4, 2014
JOB No: 14-1349-0004	REVISION No: A		Figure 11.5-1
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

L:\2013\1349\13-1349-0001\Phase 7000\1313490001-7010-11.4-1.dwg

Phytoplankton

Phytoplankton samples were collected from the euphotic zone (depth less than 10 m) at the same three stations as the zooplankton samples. Depth-integrated composite samples were collected using a 2 litre (L) Beta-bottle. Lake water was collected every 2 m from the surface to a depth of approximately 10 m. Water from each depth was combined in a 20 L cooler and transported to the Snap Lake Environment Laboratory. Water from each of the three coolers (corresponding to each station) was filtered through a set of sieves of decreasing mesh size to separate out the different phytoplankton size fractions. The sieves were a 153 µm sieve to remove zooplankton, a 53 µm sieve to collect large phytoplankton, and 25 µm and 10 µm sieves to collect small phytoplankton. It was the intention to evaluate the 10 to 25 µm, and 25 to 53 µm fractions separately, but because of low biomass these two fractions were pooled. A total of three samples were collected for this phytoplankton size fraction (10 to 53 µm). Each of the three phytoplankton samples were transferred from the sieves using lake water onto a glass fiber filter (Type GF/C) and the water removed with a weak vacuum.

Epilithic algae (periphyton)

Epilithic algal samples were collected along the shoreline of Snap Lake within the littoral zone at a depth of 2 m. A specialized SCUBA-assisted scraping-brush sampler (Turner et al. 1983) was used to collect epilithic samples from the upper surfaces of individual rocks. Three epilithic samples were collected from a single location resulting in replication within the area but not within the lake. Each of the epilithic samples was filtered onto a glass fiber filter.

Littoral macroinvertebrates

Samples of whole Ephemeroptera, caddisflies, and snails were collected along an approximate 500 m long portion of the shoreline by SCUBA divers using a battery operated electric pump suction device fitted with a 500 µm collection bag. For each taxon, sufficient material was collected for three individual samples collected sequentially to avoid pseudoreplication. A representative sample of all of the sizes present was taken for each taxon.. Shells were removed from snails in the laboratory while samples were still frozen.

Profundal macroinvertebrates

For chironomids, oligochaetes, and fingernail clams, samples were collected offshore at three AEMP sampling stations: SNAP03, SNAP06, and SNAP12. Samples were collected with an epibenthic sled fitted with a 500 µm mesh 25 centimetre (cm) diameter net (Wildco). The sled was manually pulled along the bottom at a speed of approximately 0.5 metres per second (m/s) for 50 m and then retrieved. The contents of the collection bag on the sled were transferred to a 5-L bucket and brought to the Snap Lake Environment Laboratory. Contents of the 5-L bucket were then screened and transferred to a shallow tray where individual organisms were picked out with foreceps. Sufficient material was collected for each taxon for the three individual samples. Shells were removed from fingernail clams in the laboratory while samples were still frozen.

Sample Processing of Invertebrates

Samples for all invertebrate taxa with the exception of epilithic algae that were filtered directly onto filter paper, were held in labeled glass vials until sufficient material was collected; any remaining water was removed by decanting the vial. All samples were weighed to check that greater than 0.5 mg of material was collected for each taxon. Samples were then stored on ice until they could be frozen using dry ice.

11.5.2.3 Stable Isotope Analyses

Stable isotopes are expressed using the delta notation (δ) and measured as parts per thousand (‰) differences between the isotope ratio of the sample and that of a defined international standard according to the formula:

$$\delta X = \{(R_{sample} - R_{standard})/R_{standard}] \times 1000 \qquad \text{[Equation 11.5-1]}$$

where $\delta X = {}^{15}N$ or ${}^{13}C$
R is the ratio of ${}^{15}N/{}^{14}N$ or ${}^{13}C/{}^{12}C$.

Samples depleted in the heavier isotopes (${}^{15}N$ or ${}^{13}C$) in comparison to the standard have lower delta values. Samples that are enriched in the heavier isotopes in comparison to the standard have higher delta values. All international standards are set at 0‰ by convention. Standards used to compute all values reported here included carbonate rock from the Pee Dee Belemnite formation (Craig 1957) and nitrogen gas in the atmosphere (Mariotti 1983).

All samples were freeze dried, ground to a fine powder, and the powder stored until analyzed. Approximately 1 mg of freeze dried, ground material was used in the simultaneous analyses of stable carbon and nitrogen isotopes. All analyses were performed on a Thermo Finnigan continuous flow stable isotope mass spectrometer coupled to a Carlo Erba Elemental Analyzer at the Environmental Isotope Laboratory at the University of Waterloo (Waterloo, ON, Canada). The international Atomic Energy Agency N1 and CH6 standards, respectively, were used to determine the accuracy of $\delta^{15}N$ or $\delta^{13}C$ values measured as the mean difference ± one standard deviation of repeat measures of the standards [$\delta^{15}N$ = 20.31 ± 0.10‰ (n = 3) and $\delta^{13}C$ = -10.45 ± 0.07‰ (n = 3)]. Precision was measured by repeat analyses of in house standards for $\delta^{15}N$ (0.77‰) and $\delta^{13}C$ (-25.35‰) as the mean ± one standard deviation [$\delta^{15}N$ = 0.77 ± 0.07‰ (n = 6) and $\delta^{13}C$ = -25.35 ± 0.01‰ (n = 6)]. Sample reproducibility was measured by repeat analyses of samples as the mean difference ± one standard deviation of the difference between duplicate analyses of randomly selected standards [$\delta^{15}N$ = 0.19 ± 0.14‰ (n = 20) and $\delta^{13}C$ = 0.16 ± 0.15‰ (n = 20)]. Fish muscle did not require normalizing $\delta^{13}C$ values for lipid content because the carbon to nitrogen ratio was less than 3.5:1 for all large-bodied fish collected from Snap Lake (Post et al. 2007).

11.5.2.4 Stomach Content Analyses

To provide information on diet for this study during the BsM program, stomachs of all fish sampled were opened and examined for the presence of fish. Identifiable fish were enumerated by species.

11.5.2.5 Data Analyses

Preliminary assessment of species data indicated that δ^{15}N and δ^{13}C values covaried with fish length for most large bodied fish (Figure 11.5-1); length was considered an indicator of fish age. The association between δ^{15}N and δ^{13}C values and fish total length was evaluated using Pearson product–moment correlation. Fish muscle δ^{15}N and δ^{13}C values were compared among species using multivariate analysis of covariance (MANCOVA) after accounting for the effect of species length (as an index of fish age). To accomplish this, species length was first log transformed to meet a homogeneity of variance assumption. Since the length of different species did not overlap, the log transformed length values were centered by subtraction of the species mean to place these values on a common scale. A common scale was necessary so the adjusted mean value results from MANCOVA made biological sense. Univariate tests were also used to assess differences among fish muscle with analysis of covariance (ANCOVA). Tukey adjustments for simultaneous multi-way contrasts were used to detect differences among fish species. Due to the low number of samples collected, multivariate and univariate analyses were not completed for invertebrate prey species. Inferences about differences among invertebrate δ^{15}N and δ^{13}C values were based on overlap of standard errors; however, note that such overlaps do not necessarily indicate statistical significance.

Muscle tissue of Lake Trout and Round Whitefish collected as part of 1999 baseline monitoring studies at Snap Lake were compared to samples of the same species collected in 2013 to evaluate temporal changes. The use of fork length was included as a covariate for this comparison because total length was not measured during the 1999 baseline studies. Multivariate and univariate differences were assessed using MANCOVA and ANCOVA. All statistical analyses were completed using the CAR (Fox and Weisberg 2011), MULTCOMP (Hothorn et al. 2008), and LSMEANS (Russell 2013) packages in R statistical software (RDCT 2008).

To describe the aquatic community and food web in Snap Lake and allow comparisons with other aquatic ecosystems, ecosystem attributes were calculated from the stable isotope data:

- percent littoral-benthic carbon for fish: δ^{13}Cfish - δ^{13}C clam/ δ^{13}C snail - δ^{13}C clam;
- trophic position of large bodied fish: δ^{15}N fish - δ^{15}N baseline/3.4; and,
- food chain length: 2+((δ^{15}N Lake Trout - δ^{15}N clam)+3.4).

Bayesian isotopic mixing models were used to describe the likely sources of prey comprising diets for each of the fish species (Parnell et al. 2010; Parnell and Jackson 2013). This approach considered the mean and variability of values of prey items and their carbon and nitrogen concentrations, and fish species specific isotopic discrimination factors predicted from individual mean $\delta^{15}N$ and $\delta^{13}C$ (Caut et al. 2009). Based on a model in Caut et al. (2009) relating discrimination factor to mean isotopic concentration for each of $\delta^{15}N$ and $\delta^{13}C$, predicted discrimination factors of $\delta^{15}N$ and $\delta^{13}C$ respectively were calculated for Burbot (2.57‰ and 0.83‰), Lake Chub (3.21‰ and 1.55‰), Lake Trout (2.16‰ and 1.74‰), Longnose Sucker (3.17‰ and 1.39‰), and Round Whitefish (2.73‰ and 2.05‰). To account for variability in prey-consumer isotopic discrimination, a standard deviation of 0.99 and 0.67 for $\delta^{15}N$ and $\delta^{13}C$, respectively, was estimated from studies of Trout species contributing to the discrimination model (see Appendix A of Caut et al. 2009) and were assumed for all fish species. For each fish species, the potential prey items to include in the mixing models were identified based on results of previous gut contents studies at Snap Lake, scientific literature, and expert opinion. Archived prey samples from 1999 were not available to reconstruct diets of Lake Trout and Round Whitefish sampled in 1999, so $\delta^{15}N$ and $\delta^{13}C$ values of 2013 prey sources were assumed.

Bayesian standard ellipse areas (SEA_b) were estimated from $\delta^{15}N$ and $\delta^{13}C$ values for each fish species to describe diet niche (Jackson et al. 2011). Posterior distributions of diet and SEA_b models were defined by 10,000 simulated samples. Differences in Lake Trout and Round Whitefish SEA_b were compared by calculating the proportions of SEA_b posterior samples of 1999 that were smaller than 2013. All Bayesian analyses of diet were completed using the SIAR package (Parnell et al. 2010; Parnell and Jackson 2013) in R statistical software (RDCT 2008).

11.5.3 Quality Assurance and Quality Control

The Quality Assurance (QA) and Quality Control (QC) procedures used in this study were developed such that field sampling, laboratory analyses, data entry and analyses, and report preparation produced technically sound and scientifically defensible results. As part of routine QA/QC for field operations, samples were collected by experienced personnel and labelled, preserved as required, and shipped according to standard protocols. Specific work instructions that outlined each field task in detail were provided to field personnel by the task manager, and these were reviewed before any sampling occurred. Detailed field notes were recorded in waterproof field books and on pre-printed waterproof field data sheets and maps in either pencil or indelible ink. Data sheets and sample labels were checked at the end of each field day for completeness and accuracy. Chain-of-custody forms were used to track the shipment of fin rays for aging, which were sent to North/South Consultants Inc (Winnipeg).

All sampling related to this Stable Isotope Special Study was uniquely numbered with Universal Transverse Mercator (UTM) coordinates along with gear type and water depth.

In the field, data forms were reviewed for accuracy daily by crew leads. Data were entered into Microsoft Excel when field crews returned to the office. A review of data entry involved checking a minimum of 10% of the entered data for accuracy, data entry errors, transcription errors, and invalid data. Checking was done by a second, independent individual. If an error was found, all data underwent a complete QA check (i.e., every datum checked) by the second independent individual. Upon completion of the data entry QA, each table generated from the database was reviewed for accuracy using a series of error checking routines as a secondary level of QC. All statistical results were independently reviewed by a second statistician within Golder. Tables with summary data and statistical results were also checked and values verified by a second reviewer, as were all appendices.

Appendices 11.5A and 11.5B provide the data for fish and invertebrates, respectively. The aging results for fin rays reported by North/South Consultants and subject to their internal QA/QC procedures are reported in Appendix 11.5A. Information on the QA/QC procedures and results of precision and repeatability analyses for the Waterloo University Isotope Laboratory are given above. Upon receipt of the data, Golder staff visually screened the data for QA/QC and found it to have no outliers.

11.5.3.1 Sample Collection

Samples of fish and potential prey items were collected from Snap Lake during July and August 2013 and July 1999 (Figure 11.5-1). Fish collected in 1999 and 2013 varied in total length, fork length, weight, and estimated age (Table 11.5-1). Among collection years, Lake Trout sampled in 1999 were longer, weighed more, and were older and less variable in these metrics than those sampled in 2013. Round Whitefish sampled in 1999 and 2013 were nearly identical in fork length and weight although slightly older in age in 1999 compared to 2013.

Table 11.5-1 Mean Values (± 1 SD) of Length, Weight, and Age of Large-bodied Fish at Snap Lake, 1999 and 2013

Common Name (n)	Year	Total Length (mm)	Fork Length (mm)	Weight (g)	Age (years)
Arctic Grayling (2)	2013	310.0 (9.9)	-	290.0 (28.3)	2.5 (0.7)
Burbot (5)	2013	464.2 (72.4)	-	473.2 (112.5)	11.0 (1.4)
Lake Chub (10)	2013	110.3 (18.9)	-	10.6 (4.5)	4.7 (1.8)
Lake Trout (9)	1999	-	576.3 (99.8)	2,302.8 (1,079.3)	17.0 (3.9)
Lake Trout (11)	2013	516.0 (122.6)	468.6 (114.1)	1,433.6 (1,102.6)	11.3 (5.0)
Longnose Sucker (10)	2013	295.2 (25.3)	-	306.0 (104.6)	11.4 (4.0)
Round Whitefish (15)	1999	-	263.7 (32.3)	191.3 (68.2)	7.1 (1.1)[(a)]
Round Whitefish (9)	2013	284.2 (29.6)	258.9 (28.8)	187.8 (54.9)	5.3 (2.4)

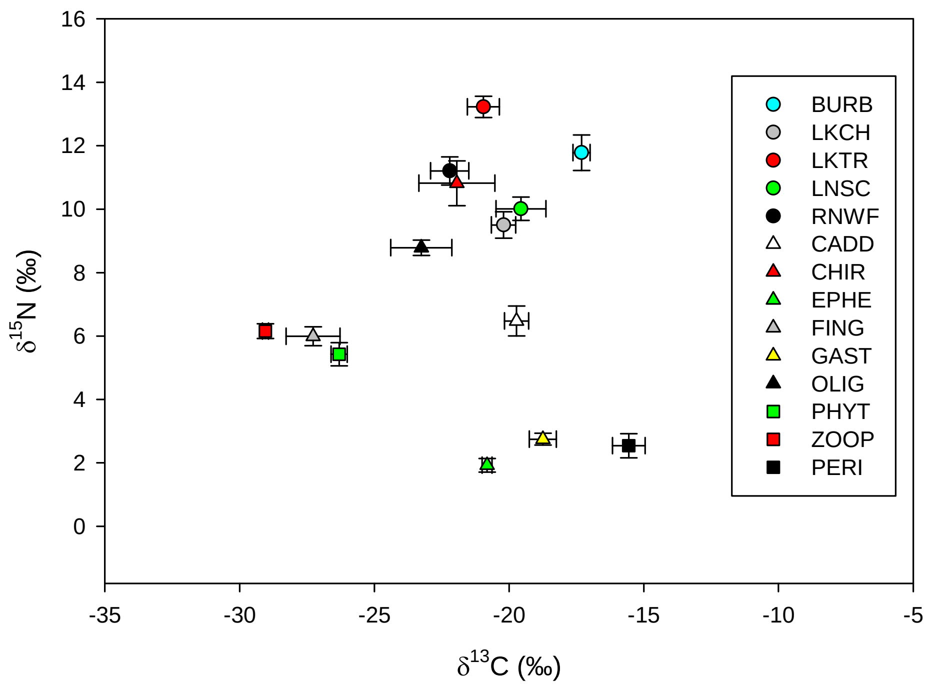
a) age was not estimated for one Round Whitefish in 1999 (n = 14).

n= number, mm = millimetre; g = gram; SD = standard deviation; a dash indicates no data.

11.5.3.2 $\delta^{15}N$ and $\delta^{13}C$ Data for Aquatic Taxa

Results of the stable isotope analyses indicated $\delta^{15}N$ and $\delta^{13}C$ values were variable among aquatic taxa in Snap Lake (Figures 11.5-2 and 11.5-3; Table 11.5-2). The results of correlation analysis between $\delta^{15}N$ and $\delta^{13}C$ values and total length of large-bodied fish sampled in 2013 are presented in Table 11.5-3.

Figure 11.5-2 Mean (± 1SE) δ^{15}N and δ^{13}C of Burbot, Lake Chub, Lake Trout, Longnose Sucker, Round Whitefish, Caddisfly, Chironomids, Ephemeroptera, Fingernail Clams, Snails, Oligochaetes, Phytoplankton, Zooplankton, and Periphyton Sampled from Snap Lake, 2013



‰ = per mil; BURB = Burbot; LKCH = Lake Chub; LKTR = Lake Trout; LNSC = Longnose Sucker; RNWF = Round Whitefish; CADD = caddisflies; CHIR = chironomids; EPHE = Ephemeroptera; FING = fingernail clams; GAST = Gastropoda (snails); OLIG = oligochaetes; PHYT = phytoplankton; ZOOP = zooplankton; PERI = periphyton.

Table 11.5-2 Mean (± 1 SD) δ^{15}N, δ^{13}C, Nitrogen, Carbon, and Carbon-Nitrogen Ratios of Aquatic Taxa from Snap Lake, 1999 and 2013

Common Name (n)	Year	δ^{15}N (‰)	(‰)	N (%)	C (%)	C:N
Arctic Grayling (2)	2013	9.18 (0.61)	-20.29 (0.05)	14.31 (0.53)	47.00 (0.51)	3.29 (0.09)
Burbot (5)	2013	11.78 (1.25)	-17.31 (0.71)	15.13 (0.34)	48.42 (0.20)	3.20 (0.07)
Lake Chub (10)	2013	9.50 (1.32)	-20.21 (1.45)	13.94 (0.46)	45.82 (1.49)	3.29 (0.11)
Lake Trout (9)	1999	13.17 (1.18)	-22.60 (1.13)	15.36 (0.69)	48.87 (1.31)	3.18 (0.13)
Lake Trout (11)	2013	13.23 (1.11)	-20.96 (1.97)	14.75 (0.87)	48.36 (2.13)	3.28 (0.14)
Longnose Sucker (10)	2013	10.01 (1.17)	-19.56 (2.94)	13.94 (1.26)	44.87 (2.95)	3.23 (0.11)
Round Whitefish (15)	1999	11.38 (1.08)	-22.23 (1.21)	14.75 (1.29)	48.67 (4.60)	3.30 (0.18)
Round Whitefish (9)	2013	11.21 (1.33)	-22.21 (2.14)	13.74 (0.86)	45.81 (3.62)	3.33 (0.10)
Caddisflies (3)	2013	6.48 (0.82)	-19.72 (0.78)	8.16 (0.70)	48.70 (1.35)	6.00 (0.64)
Chironomids (3)	2013	10.82 (1.22)	-21.94 (2.44)	9.96 (0.58)	46.93 (2.47)	4.73 (0.50)
Ephemeroptera (3)	2013	1.92 (0.37)	-20.82 (0.32)	10.57 (1.38)	50.38 (1.45)	4.83 (0.70)
Fingernail clams (3)	2013	5.99 (0.52)	-27.28 (1.73)	9.41 (0.21)	48.17 (1.09)	5.12 (0.09)
Gastropods (3)	2013	2.75 (0.32)	-18.75 (0.87)	6.71 (0.12)	37.16 (0.62)	5.54 (0.19)

Table 11.5-2 Mean (± 1 SD) δ^{15}N, δ^{13}C, Nitrogen, Carbon, and Carbon-Nitrogen Ratios of Aquatic Taxa from Snap Lake, 1999 and 2013

Common Name (n)	Year	δ^{15}N (‰)	(‰)	N (%)	C (%)	C:N
Oligochaetes (3)	2013	8.78 (0.42)	-23.26 (1.97)	10.16 (1.85)	47.24 (1.69)	4.75 (0.82)
Periphyton (10)	2013	2.54 (1.19)	-15.55 (1.92)	2.34 (0.33)	28.14 (4.96)	12.01 (1.40)
Phytoplankton (4)	2013	5.42 (0.73)	-26.31 (0.60)	2.09 (0.84)	26.82 (2.25)	15.27 (8.30)
Zooplankton (3)	2013	6.16 (0.41)	-29.04 (0.20)	8.16 (0.22)	52.36 (0.76)	6.42 (0.08)

n= number, ‰ = per mil; %= percent; N = nitrogen; C = carbon; C:N = Carbon-Nitrogen Ratios; SD = standard deviation.

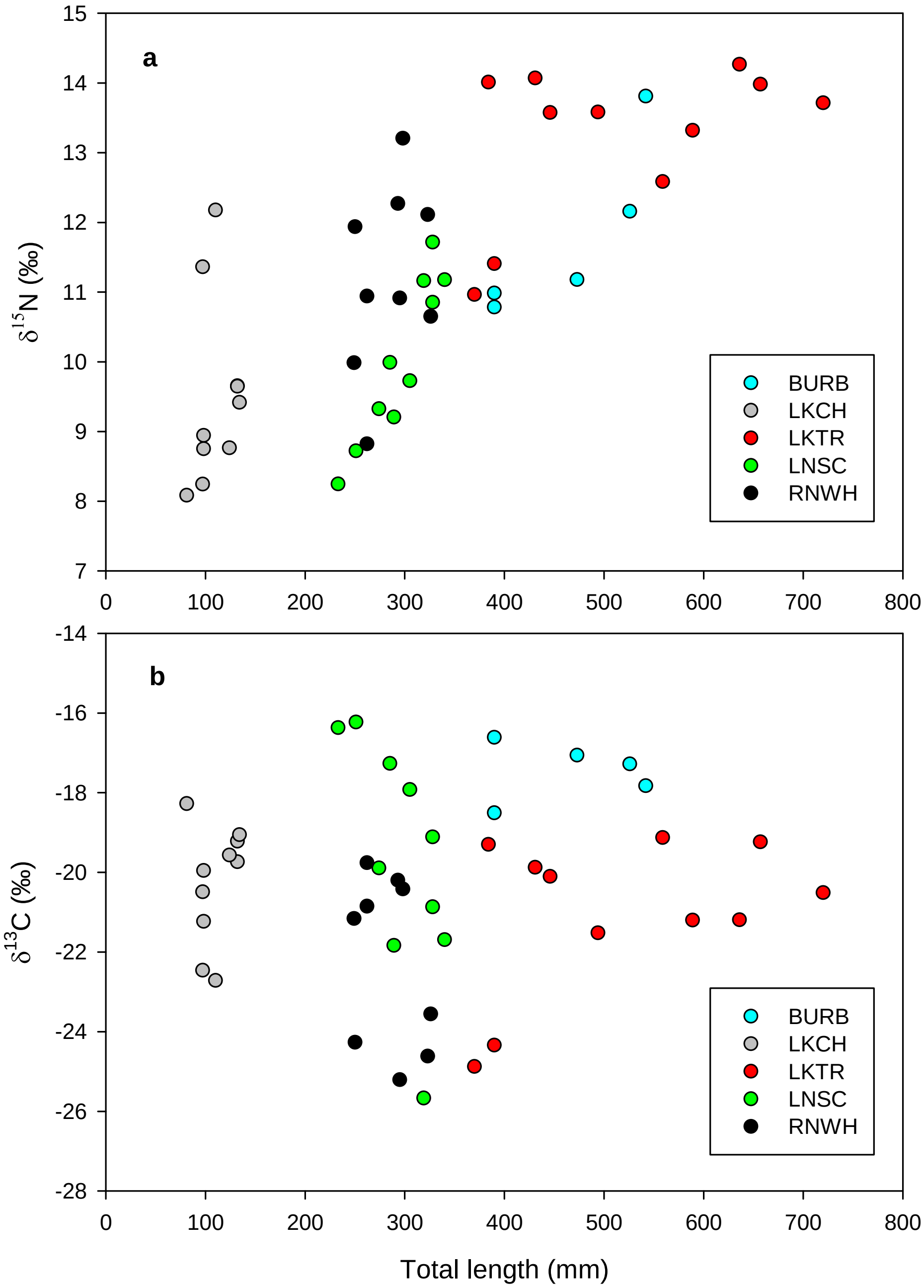
Table 11.5-3 Correlation Coefficient and Statistical Significance Results of Values of δ^{15}N and δ^{13}C with Total Length of Large-bodied Fish at Snap Lake, 2013

Common Name (n)	δ^{15}N		δ^{13}C	
	r	P-value	r	P-value
Burbot (5)	0.85	0.07	-0.02	0.98
Lake Chub (10)	0.19	0.59	0.23	0.52
Lake Trout (11)	0.49	0.13	0.37	0.26
Longnose Sucker (10)	0.93	<0.01	-0.64	<0.05
Round Whitefish (9)	0.39	0.30	-0.40	0.29

n= number; r = correlation coefficient; *P*-value = statistical Significance; N = nitrogen; C = carbon.

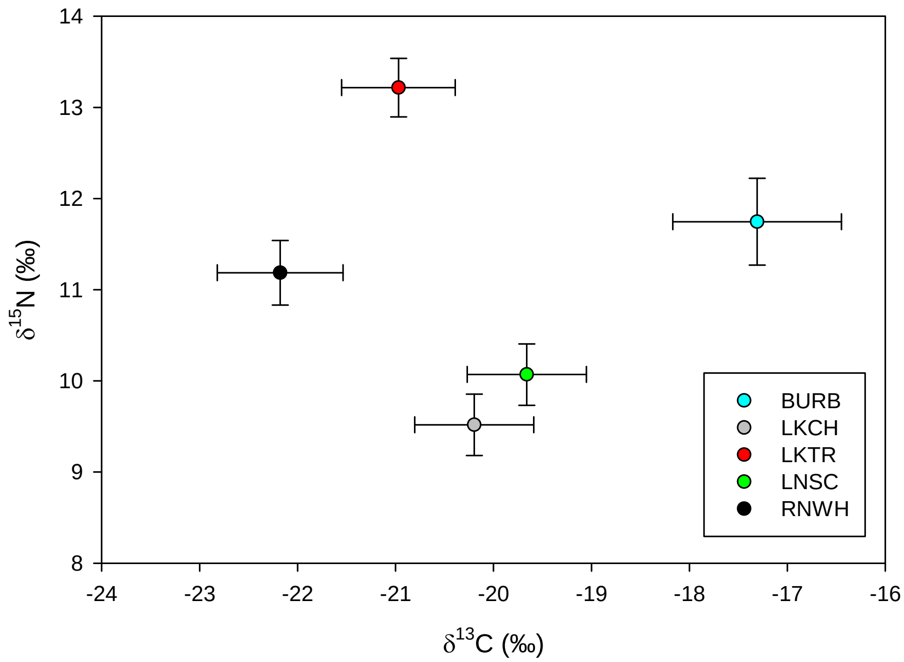
Multivariate analyses indicated that δ^{15}N and δ^{13}C values of fish differed among species after controlling for the effect of total length (MANCOVA, $F_{8,76}$ =10.12 , P <0.01). Univariate differences were detected for δ^{15}N (ANCOVA, $F_{4,39}$ = 18.9 , P <0.01) and δ^{13}C (ANCOVA, $F_{4,39}$ = 4.9, P <0.01) values among fish species after controlling for total length (Figures 11.5-3 and 11.5-4).

Figure 11.5-3 Relationship between $\delta^{15}N$ (a) or $\delta^{13}C$ (b) and Total Length for Burbot, Lake Chub, Lake Trout, Longnose Sucker, and Round Whitefish Collected from Snap Lake, 2013



‰ = per mil; mm = millimetre; BURB = Burbot; LKCH = Lake Chub; LKTR = Lake Trout; LNSC = Longnose Sucker; RNWF = Round Whitefish; N = nitrogen; C = carbon.

Figure 11.5-4 Mean (± 1 SE) Muscle $\delta^{15}N$ and $\delta^{13}C$ Values of Burbot, Lake Chub, Lake Trout, Longnose Sucker, and Round Whitefish from Snap Lake, 2013, After Controlling for Species Total Length



‰ = per mil; BURB = Burbot; LKCH = Lake Chub; LKTR = Lake Trout; LNSC = Longnose Sucker; RNWF = Round Whitefish; N = nitrogen; C = carbon.

11.5.3.3 Variation in Diet Among Fish Species

Bayesian stable isotope mixing models indicated that the composition of diets of fish were variable among species (Table 11.5-4). For Burbot, the estimated contributions to diet were relatively consistent among the prey items considered except for fingernail clam, which was lower than for the other fish and invertebrate prey species considered. Prey items for Lake Chub were also relatively consistent except for a lower estimated proportion of zooplankton (8%). Approximately 50% of Lake Trout diet was derived from the combination of Burbot, Round Whitefish, and chironomids; all other sources were less than 11%. For Longnose Sucker, 61% of diet was derived from caddisflies and oligochaetes. All other prey items for Longnose Sucker were lower than 11%. Round Whitefish diet was heavily derived from chironomids, but oligochaetes and zooplankton were all greater than 15% of diet. The credibility intervals of the proportions that each prey item contributed to diet indicated there was uncertainty in diet composition of all large-bodied fish. For example, numerous credibility intervals included zero and ranged up to 40% above the lower interval value.

Lake Trout and Burbot were the only species assumed to eat fish including each other. Lake Trout showed a preference for Round Whitefish, whereas the diet for Burbot was more equitable across Round Whitefish, Longnose Sucker, and Lake Chub.

In terms of the prey occupying the littoral, profundal, pelagic zones of Snap Lake, Round Whitefish made the greatest use of profundal benthos, less use of littoral benthos, and no use of pelagic biota. For Longnose Sucker and Lake Chub, use of littoral, profundal, and pelagic biota by the two species was similar although the contribution of pelagic prey was relatively low (8%) (Figure 11.5-5).

Table 11.5-4 Percent (± 95% CI[(a)]) of Prey Items in the Diet of Fish Species Collected from Snap Lake, 2013

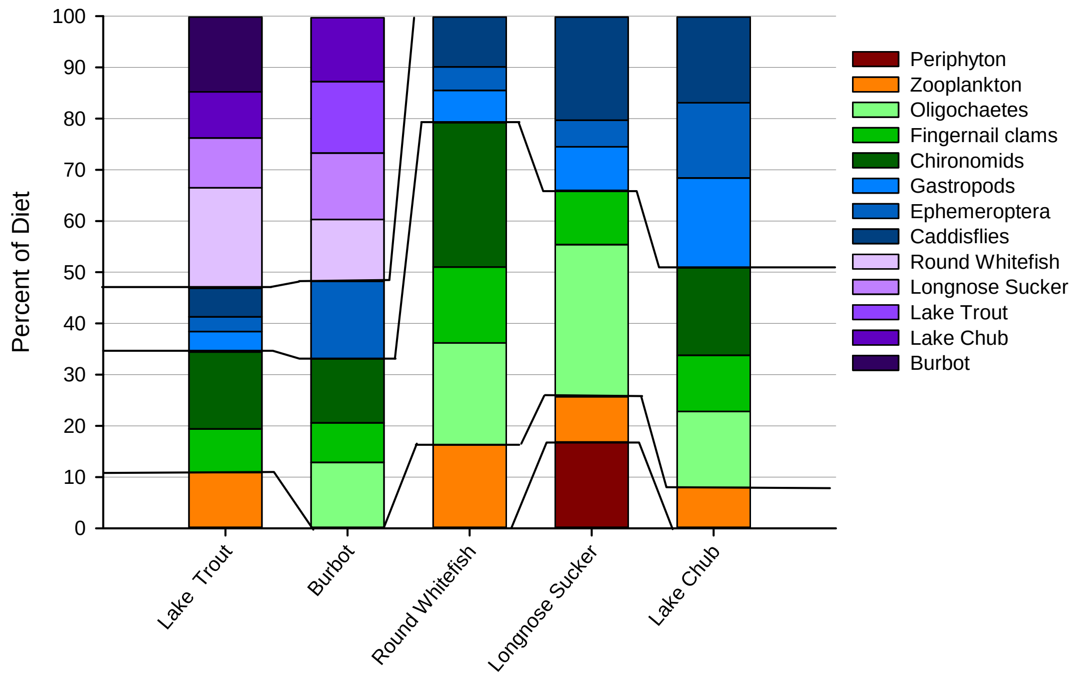
Prey item	Large-bodied fish species				
	Burbot	Lake Chub	Lake Trout	Longnose Sucker	Round Whitefish
Burbot	-	-	14.9 (0.1 – 28.9)	-	-
Lake Chub	12.9 (0.0 – 25.8)	-	9.0 (0.0 – 20.5)	-	-
Lake Trout	13.9 (0.2 – 26.3)	-	-	-	-
Longnose Sucker	13.0 (0.0 – 25.6)	-	9.7 (0.1 – 22.3)	-	-
Round Whitefish	12.1 (0.0 – 24.3)	-	19.6 (0.8 – 39.0)	-	-
Caddisflies	-	17.0 (0.4 – 31.5)	5.6 (0.0 – 15.0)	20.1. (0.0 – 38.9)	9.9 (0.0 – 24.0)
Chironomids	12.5 (0.0 – 25.2)	17.1 (1.8 – 30.6)	15.0 (0.0 – 33.4)	-	28.2(8.6 – 48.6)
Ephemeroptera	15.1 (1.3 – 26.8)	14.7 (0.3 – 27.7)	2.9 (0.0 – 8.6)	5.2 (0.0 – 14.3)	4.6 (0.0 – 12.4)
Fingernail clams	7.7 (0.0 – 19.6)	11.0 (0.0 – 23.6)	8.5 (0.0 – 19.6)	10.8 (0.0 – 25.2)	14.8 (0.0 – 30.7)
Gastropods	-	17.5 (1.3 – 31.6)	4.0 (0.0 – 11.5)	8.5 (0.0 – 21.0)	6.3 (0.0 – 16.6)
Oligochaetes	12.9 (0.0 – 25.5)	14.8 (0.0 – 29.5)	-	29.7 (10.7 – 50.3)	19.9 (0.0 – 38.8)
Phytoplankton	-	-	-	-	-
Zooplankton	-	8.0 (0.0 – 18.5)	10.9 (0.0 – 21.2)	8.8 (0.0 – 21.6)	16.3 (0.4 – 31.3)
Periphyton	-	-	-	16.9 (0.0 – 33.9)	-

Note: '-' denotes that species was not included in diet.

a) reported are Bayesian 95% highest density region intervals.

% = percent; CI = confidence interval.

Figure 11.5-5 Diet Reconstruction for Lake Trout, Burbot, Round Whitefish, Longnose Sucker, and Lake Chub Collected from Snap Lake, 2013



11.5.3.4 Isotopic Niche 2013

Estimates of SEA_b varied among fish (Table 11.5-5). Burbot had the smallest estimated SEA_b and Round Whitefish the largest. Comparison of posterior distributions indicated that 69% of Burbot SEA_b estimates were smaller than for Lake Chub, 58% of Lake Chub estimates were smaller than for Lake Trout, and 50% of Longnose Sucker estimates were smaller than for Round Whitefish.

Table 11.5-5 Mean, Median, and 95% Credible Intervals[(a)] of Standard Ellipse Area (Diet Niche) based on $\delta^{15}N$ and $\delta^{13}C$ Values of Large-bodied Fish from Snap Lake, 2013

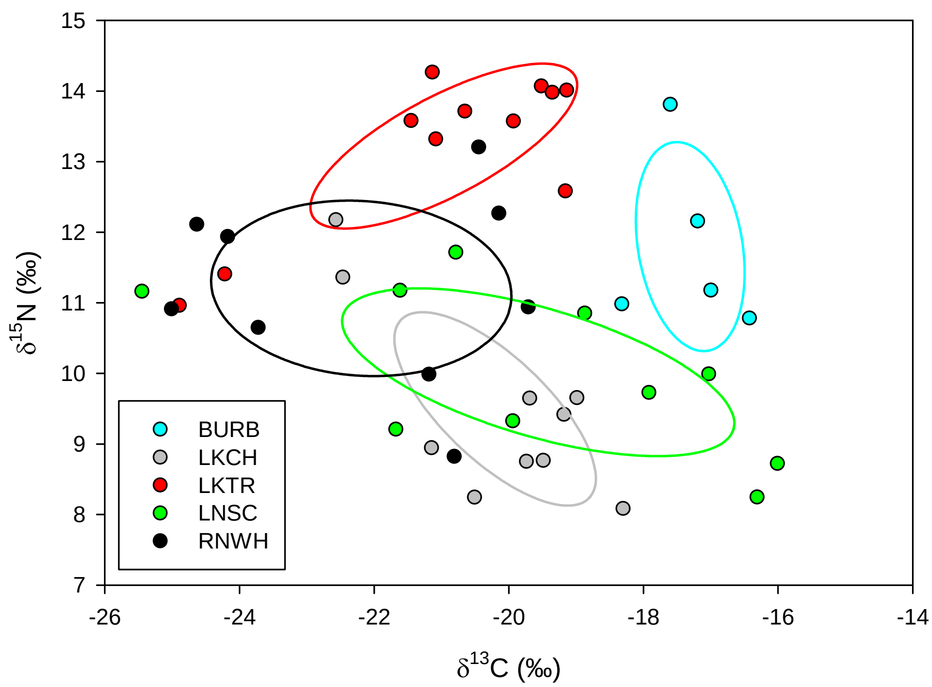
Common name	Statistics of Bayesian standard ellipse area			
	Mean	Median	95% Lower	95% Upper
Burbot	3.71	3.27	1.19	7.23
Lake Chub	4.48	4.2	2.08	7.31
Lake Trout	4.92	4.63	2.38	7.99
Longnose Sucker	8.64	8.11	4.01	14.30
Round Whitefish	8.78	8.16	3.91	14.9

a) reported are Bayesian 95% highest density region intervals.

% = percent.

The shape and orientation of SEA_bs varied among species and indicated overlap of SEA_b between Lake Chub, Longnose Sucker, and Round Whitefish, and between Round Whitefish and Lake Trout (Figure 11.5-6). No species overlapped with Burbot SEA_b.

Figure 11.5-6 Bayesian Standard Ellipse Areas (Diet Niche) of $\delta^{15}N$ and $\delta^{13}C$ Muscle Values of Burbot, Lake Chub, Lake Trout, Longnose Sucker, and Round Whitefish from Snap Lake, 2013

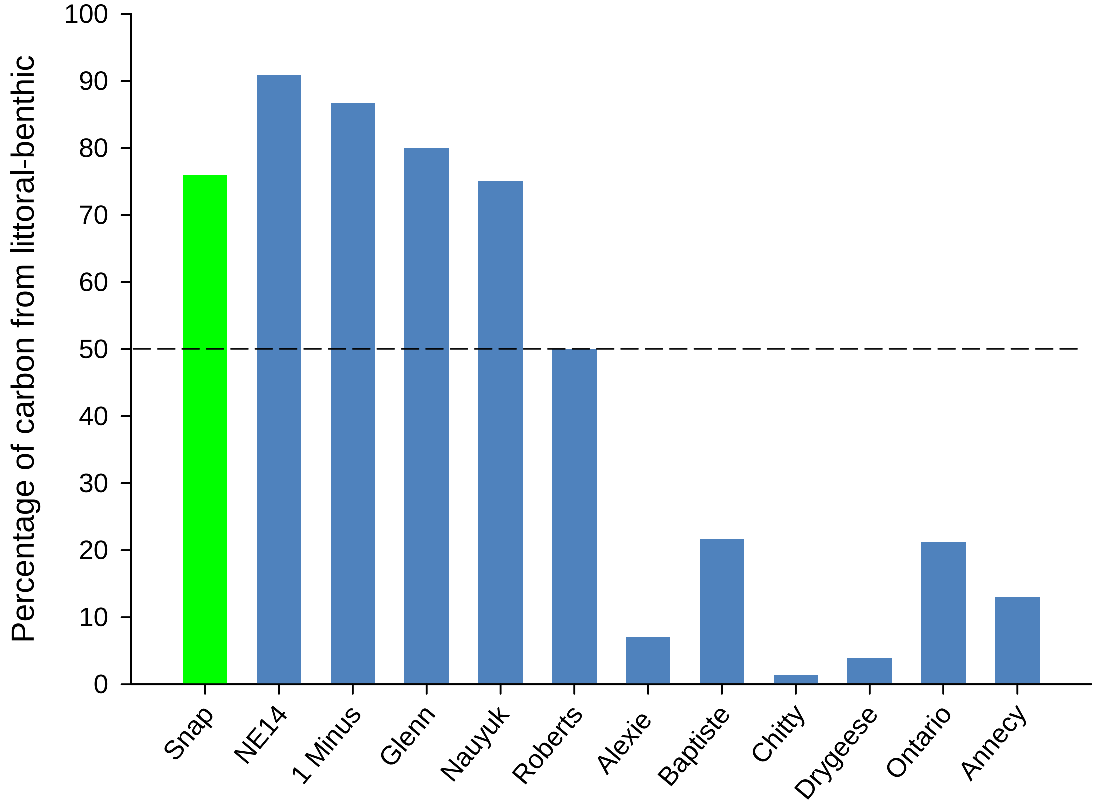


‰ = per mil; BURB = Burbot; LKCH = Lake Chub; LKTR = Lake Trout; LNSC = Longnose Sucker; RNWF = Round Whitefish; N = nitrogen; C = carbon.

11.5.3.5 Carbon Source, Trophic Position and Food Chain Length

Approximately 75% of carbon for Lake Trout in Snap Lake was derived from littoral-benthic sources (Figure 11.5-7).

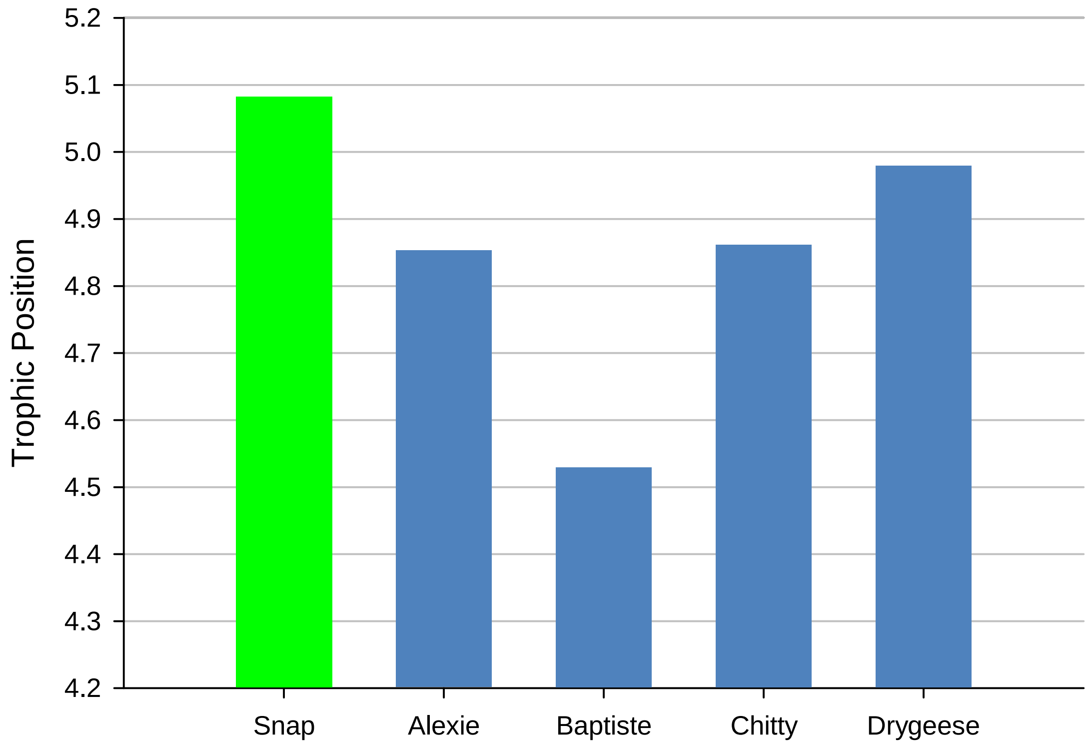
Figure 11.5-7 Percentage of Carbon from Littoral-Benthic Sources for Lake Trout from Snap Lake and Five High Latitude Lakes (percent littoral-benthic carbon: δ^{13}Cfish - δ^{13}C clam/ δ^{13}C snail - δ^{13}C clam



Sources: Lakes NE-14 and 1 Minus - Sierszen et al. 2003, Glen, Nauyuk, and Roberts Lakes - Swanson et al. 2011, Alexie, Baptiste, Chitty, Drygeese Lakes - Cott et al. 2011, Lake Ontario – Fitzsimons 2014, Lake Annecy - Janjua and Gerdeaux 2011.Note: dashed line indicates 50% littoral-benthic carbon.

Lake Trout exhibited relatively high carbon use from littoral-benthic sources, as well as a relatively high trophic position that exceeded that of other fish species; Lake Trout from other lakes had trophic positions which ranged up to one trophic position below Lake Trout from Snap Lake (Figure 11.5-8).

Figure 11.5-8 Trophic Position of Lake Trout in Snap Lake Compared to Lake Trout from Four Lakes within 250 km of Snap Lake



Source: Alexie, Baptistes, Chitty, Drygeese - Cott et al. 2011.

Note: Trophic position = δ^{15}N fish - δ^{15}N baseline/3.4

km = kilometre.

11.5.3.6 Temporal Patterns in δ^{15}N and δ^{13}C Values, Isotopic Niche and Diet for Lake Trout and Round Whitefish

The results of correlation analysis indicated a significant association between δ^{15}N and fork length of Round Whitefish and between δ^{13}C and fork length for Lake Trout, but only for samples collected during 1999 (Table 11.5-6). Multivariate analyses indicated that δ^{15}N and δ^{13}C values differed among years after controlling for the effect of fork length for Lake Trout (MANCOVA, $F_{2,16} = 4.1$, $P = 0.04$), but not for Round Whitefish (MANCOVA, $F_{2,20} = 1.05$, $P = 0.34$).

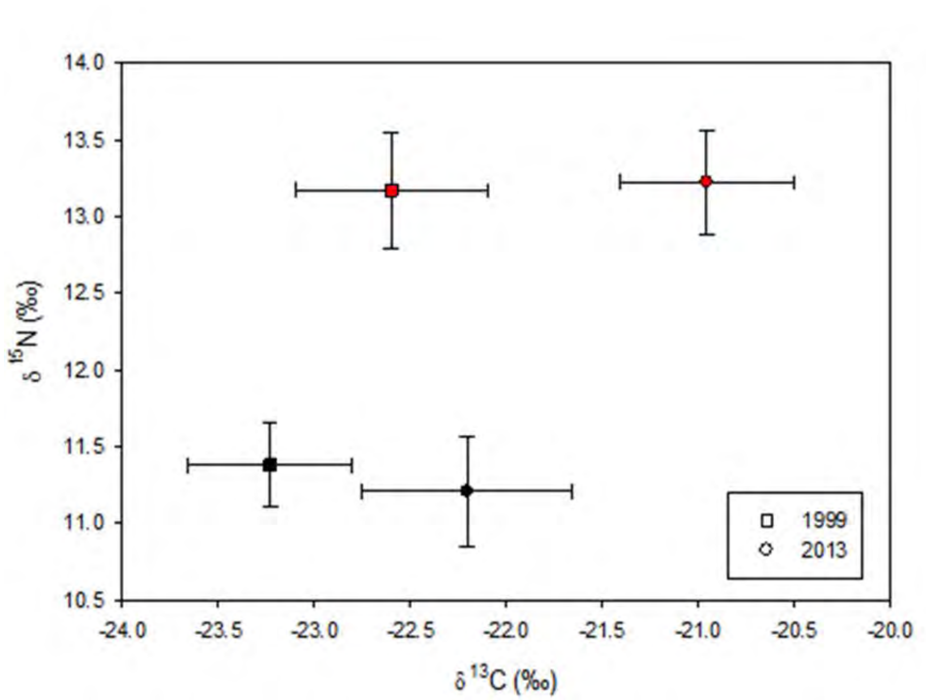
Table 11.5-6 Correlation Coefficient and Statistical Significance Results of Values of $\delta^{15}N$ and $\delta^{13}C$ with Fork Length of Lake Trout and Round Whitefish from Snap Lake, 1999 and 2013

Common Name (n)	Year	$\delta^{15}N$		$\delta^{13}C$	
		r	P-value	r	P-value
Lake Trout (9)	1999	-0.05	0.90	0.63	0.07
Lake Trout (11)	2013	0.48	0.13	0.38	0.26
Round Whitefish (15)	1999	0.47	0.07	-0.01	0.96
Round Whitefish (9)	2013	0.47	0.20	0.14	0.72

n = number; r = correlation coefficient; *P*-value = statistical evidence ; N = nitrogen; C = carbon.

Univariate tests indicated that $\delta^{15}N$ did not vary between years for Round Whitefish (ANCOVA, $F_{2,21}$ = 0.14, P = 0.71) or Lake Trout (ANCOVA, $F_{2,17}$ = 0.11, P = 0.92). For $\delta^{13}C$ no differences between years were detected for Round Whitefish (ANCOVA, $F_{2,21}$ = 2.18, P = 0.15), whereas for Lake Trout there were differences (ANCOVA, $F_{2,17}$ = 5.86, P = 0.03) after controlling for fork length (Figure 11.5-9).

Figure 11.5-9 Muscle Mean (± 1 SE) $\delta^{15}N$ and $\delta^{13}C$ Values for Lake Trout (Red) and Round Whitefish (Black, After Controlling for Fork Length) collected from Snap Lake, 1999 and 2013



‰ = per mil; N = nitrogen; C = carbon.

Under the assumption that the $\delta^{15}N$ and $\delta^{13}C$ isotopic composition of prey items was the same as in 2013, Bayesian stable isotope mixing models indicated that the composition of diets of Lake Trout and Round Whitefish sampled during 1999 were variable (Table 11.5-7). In 1999, approximately 20% of Lake Trout diet was estimated to be from Round Whitefish or zooplankton, whereas chironomids and fingernail clams contributed 12% and 13%, respectively. All other prey items were estimated to be less than 8% of Lake Trout diet. The highest contributions to Round Whitefish diet were chironomids and zooplankton at 35% and 30%, respectively. Oligochaetes and fingernail clams were estimated to be 15% and 12% of Round Whitefish diet. All other prey items were estimated to be less than 4% of Round Whitefish diet.

The estimated proportions of zooplankton in the diets of Lake Trout and Round Whitefish in 2013 were lower by almost a half in 1999. Of the littoral benthos, Lake Trout consumed less in 2013 compared to 1999 but there was no change in profundal benthos. For Round Whitefish, more littoral benthos was consumed in 2013 whereas for profundal benthos there was only a small change of less than 2%.

Table 11.5-7 Percent (± 95% CI[(a)]) of Prey Items in the Diets of Lake Trout and Round Whitefish from Snap Lake, 1999

Prey item	Fish species	
	Lake Trout	Round Whitefish
Burbot	7.4 (0.0 – 18.5)	-
Lake Chub	7.2 (0.0 – 18.4)	-
Lake Trout	-	-
Longnose Sucker	7.4 (0.0 – 18.6)	-
Round Whitefish	20.0 (0.4 – 39.9)[(b)]	-
Caddisflies	4.8 (0.0 – 13.2)	3.8 (0.0 – 10.7)
Chironomids	12.1 (0.0 – 26.9)	34.6 (15.5 – 51.4)
Ephemeroptera	3.3 (0.0 – 9.6)	1.9 (0.0 – 5.1)
Fingernail clams	13.4 (0.0 – 28.2)	12.2 (0.0 – 30.6)
Gastropods (snails)	4.0 (0.0 – 11.2)	2.6 (0.0 – 7.1)
Oligochaetes	-	14.8 (0.0 – 36.2)
Phytoplankton	-	-
Zooplankton	20.5 (3.9 – 36.1)	30.2 (12.1 – 46.3)
Periphyton	-	-

'-' denotes that species was not included in diet.

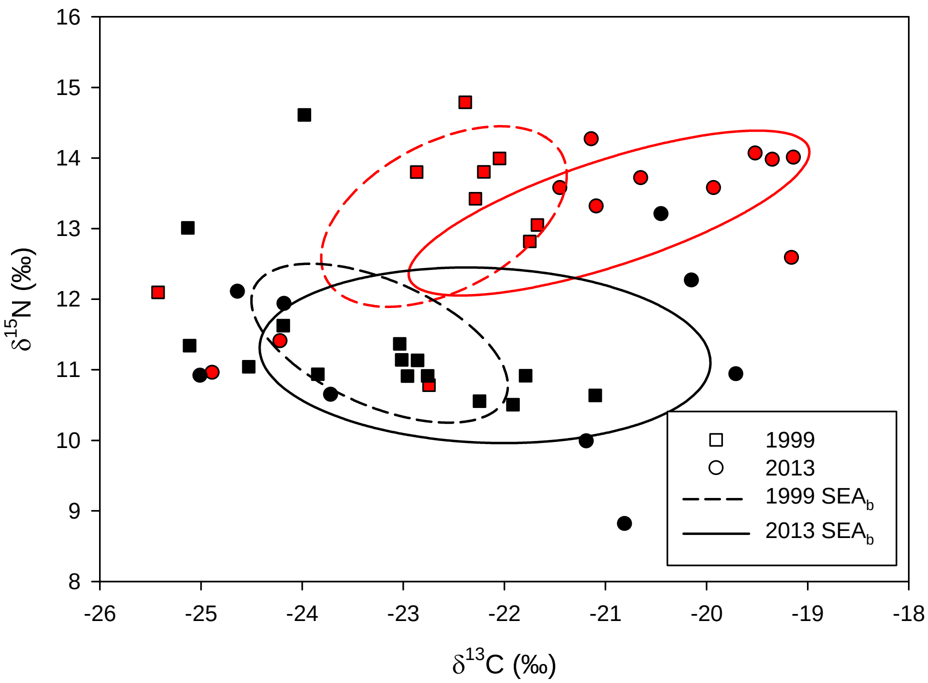
a) Reported are Bayesian 95% highest density region intervals.

b) Based on 1999 Round Whitefish $\delta^{15}N$ and $\delta^{13}C$ values.

% = percent; CI = confidence interval.

The mean (± 95% CI) and median values of SEA_b of Lake Trout in 1999 were 4.37 (95% CI: 1.96 to 7.37) and 4.06, respectively. Mean and median values for Round Whitefish during 1999 were 3.79 (95% CI: 20.6 to 5.84) and 3.62, respectively. The estimated mean SEA_bs of Lake Trout and Round Whitefish varied in shape and area between 1999 and 2013 (Figure 11.5-10). Comparison of the posterior distribution of SEA_bs within species among years indicated that 62% of SEA_bs of 1999 Lake Trout were smaller than SEA_bs for 2013 Lake Trout. For Round Whitefish, 98% of SEA_bs were smaller for 1999 than SEA_bs for 2013. There was evidence of overlap for both Lake Trout and Round Whitefish between SEA_bs of 1999 and 2013.

Figure 11.5-10 Bayesian Standard Ellipse Area for Lake Trout (Red) and Round Whitefish (Black) Collected from Snap Lake, 1999 and 2013



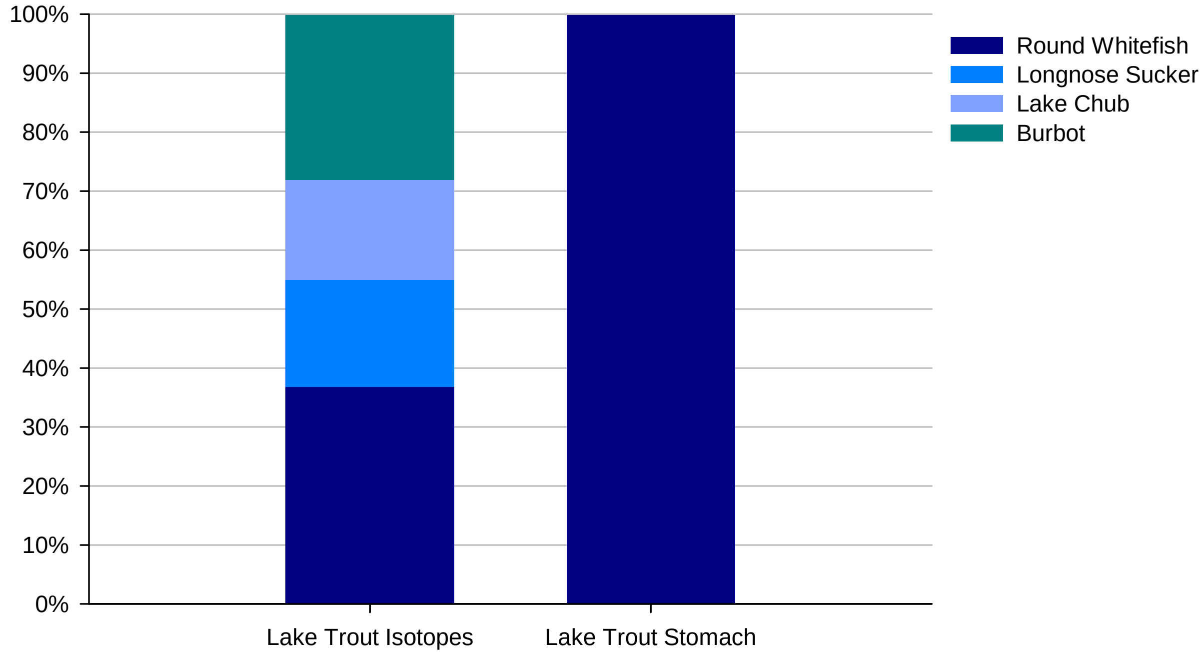
‰ = per mil; SEA_b = Bayesian Standard Ellipse Area; N = nitrogen; C = carbon.

11.5.3.7 Stomach Content Analyses

Of 89 Lake Trout and five Burbot stomachs examined, only 5 Lake Trout contained fish. Of these five Lake Trout, three stomachs contained a single Round Whitefish with one stomach containing two Round Whitefish. The fifth Lake Trout was found to have two Lake Trout in its stomach. Burbot stomachs were devoid of fish prey. The Whitefish prey observed in Lake Trout stomachs were approximately 40% of the length of their predators.

Compared to stable isotopes data, Lake Trout stomach content analysis indicated a more conservative degree of fish consumption, with only one fish species in the diet. The stable isotope mixing model supported consumption of Burbot, Longnose Sucker, Lake Chub, and Round Whitefish (Figure 11.5-11).

Figure 11.5-11 Fish Species Consumption by Lake Trout Collected from Snap Lake in 2013, Based on a Stable Isotope Mixing Model and Stomach Content Analysis



% = percent.

11.5.4 Discussion

Based on diet reconstruction from the mixing model, Lake Trout from Snap Lake appear to be generalist feeders and hence contribute to the integration of littoral and pelagic trophic pathways in the lake. Generalist fish species, because they can undergo rapid changes in their feeding behaviour and habitat use, are likely the most important couplers of littoral and pelagic food web compartments in lakes (Vander Zanden and Vadeboncoeur 2002; Rooney et al. 2006). Generalist foraging by fish is particularly evident in high-latitude lakes where consumers must adapt to seasonal changes in prey availability, light, and temperature (Hecky and Hesslein 1995; Power et al. 2008). Generalist fish also play a major role in carbon and nutrient cycling in lakes, for example by feeding on benthic prey but providing nutrients to pelagic phytoplankton in dissolved form (Vanni 2002; Glaholt and Vanni 2005). From a whole community perspective, generalist foraging of fish such as Lake Trout may increase food web stability by decreasing consumer-resource oscillations (Vadeboncoeur et al. 2005; Rooney et al. 2006; Kratina et al. 2012).

As a generalist, the diet of Snap Lake Lake Trout is much broader than has been reported for other northern Lake Trout populations that showed a higher degree of specialization. For Arctic lakes, it was reported that Lake Trout fed predominantly on fish (Johnson 1976) including Lake Whitefish (*Coregonus clupeaformis*), Cisco (C. *artedii*) and Slimy Sculpin (*Cotttus cognatus*). In contrast, in the Toolik area lakes of Alaska, diet studies indicated that molluscs were a major food source for Lake Trout whereas Slimy Sculpin comprised only 12% of the diet based on percent occurrence (Hershey 1990; Merrick et al. 1991, 1992). These studies, however, were based on stomach content analysis and molluscs having a calcareous shell which may have been much more resistant to digestion than softer fish tissue. Shells are indigestible and therefore are more easily identified in stomach contents than soft-bodied prey, which digest more rapidly (Kionka and Windell 1972). The ingestion and assimilation of different prey items are not often equal, which is a commonly identified problem when interpreting stomach content data (Barton et al. 2005; Campbell et al. 2009). Stable isotopes provide a time-integrated measure of an organism's trophic position, account for temporal and spatial variation in feeding at multiple levels of the food web, and detect trophic interactions that are otherwise unobservable. Since stable isotopes are believed to reflect actual assimilation of prey items by an organism, they have seen broad application in assigning trophic pathways in aquatic and terrestrial food webs as was the case in this study (Hecky and Hesslein 1995).

The lack of a positive relationship between δ^{15}N and size suggests that Lake Trout remained generalists across the range of lengths examined and did not show the ontogenetic changes in δ^{15}N associated with more southerly populations at between 150 and 400 mm in total length; this size of Lake Trout has been associated with a transition from a strictly invertebrate diet to a diet containing increasing amounts of fish (Becker 1983; Madenjian et al. 1998; Mittelbach and Persson 1998; Cott et al. 2011). For four lakes approximately 250 km south of Snap Lake, Cott et al. (2011) reported a weak but positive relationship between δ^{15}N and body mass of Lake Trout.

The precision and accuracy of stable isotopes to identify the contribution of particular prey items, including their application in mixing models, depends on obtaining samples of all potential prey items from the field, which is not always possible. For instance, it was not possible to collect samples of Slimy Sculpin, which appear to be extremely rare in Snap Lake (De Beers 2013).

Lake Trout diets for both juveniles and adults appear to reflect prey abundance (Elrod and O'Gorman 1991; Madenjian et al. 1998). In Lake Ontario the consumption of Slimy Sculpin by Lake Trout was proportional to numerical abundance in the lake and hence Lake Trout did not selectively feed on Slimy Sculpin. In addition, after Slimy Sculpin became rare in Lake Ontario, their importance in Lake Trout diets also declined (Elrod and O'Gorman 1991; Mills et al. 2003; Rush et al. 2012). Thus, while Slimy Sculpin were not included in mixing model calculations for Lake Trout, the contribution of Slimy Sculpin if any, should be low based on their low abundance at Snap Lake.

Although cannibalism was observed in two Lake Trout collected in the BsM sampling, the occurrence of cannibalism is expected to be limited based on the absence of a positive relationship between δ^{15}N and total length. Hobson and Welch (1995) used a 3.7‰ stepwise increase between intermediate and large Arctic Char (*Salvelinus alpinus*) to infer cannibalism of large sized char on intermediate sized char. In the present study, the range in δ^{15}N in Lake Trout was generally less than 3‰.

The accuracy of mixing models depends on whether diet tissue discrimination factors for a species are appropriate. Small variations in the values used for the discrimination factors used may lead to important differences in the output of isotopic-mixing models (Ben-David and Schell 2001). Diet tissue discrimination factors of 1‰ for δ^{13}C and 3.4‰ for δ^{15}N have been widely applied for freshwater fish across several species and irrespective of diet isotopic values (Post 2002). For fish, Caut et al. (2009)

found significant relationships between carbon and nitrogen discrimination factors and diet isotopic ratios, leading them to propose the use of Diet-Dependent Discrimination Factors as a means of deriving discriminations factors appropriate to the diet in question. Although Auerswald et al. (2010) have criticized the methods proposed by Caut et al (2009), these criticisms seem largely baseless (Caut et al. 2010).

The use of mixing models provided insights into temporal and spatial variation in the diet of Snap Lake Lake Trout than was possible using conventional stomach analysis that is based on the diet evident at the time of capture and in a sense only provides a snap shot for one time and place. Using mixing models for this study, it was estimated that Lake Trout were consuming Round Whitefish, Burbot, Longnose Sucker, and Lake Chub to varying degrees, whereas stomach contents analysis only detected Round Whitefish. Similarly, for invertebrates mixing models estimated Lake Trout to be consuming most of the invertebrate taxa in Snap Lake including profundal and littoral benthos whereas based on stomachs only chironomids and tricopterans were being eaten.

The diet represented in this study likely represents fish diet during the previous year because of the turnover rate of isotopes and the fact that Lake Trout growth in northern populations is probably highly seasonal (Morbey et al. 2010). The isotope turnover rate defines the delay necessary for the isotope composition of a consumer to reach equilibrium with that of their food source. Isotopic turnover is mediated by two general processes: the dilution of existing mass by new mass synthesized from recently consumed prey (i.e., growth) and the replacement or conversion of existing tissue using material synthesized from recent diet (i.e., metabolism; Hesslein et al. 1993). For European Whitefish (*Coregonus laravetus*) at the latitude of central Europe, Perga and Gerdeaux (2005) reported that muscle exhibited a slow and discontinuous turnover such that it provided a long-term integrated image of the isotope composition of food consumed from March to September, 7 months out of 12, during which nutrients were allocated to growth. Although the stomachs of European Whitefish were not empty over the other five months, their isotope composition was not reflected in the muscle, and it was concluded that the nutrients from the food were allocated to basal metabolism and to gonad growth. As winters last longer at northern latitudes where growth of Lake Trout is slower than at southern latitudes, growth would be expected to occur over a more restricted period (June to August) and, as a result, stable isotopes would reflect diet over this more restricted period (McDermid et al. 2010).

With regards to tissue turnover of isotopes, Weidel et al. (2011) contended, based on ^{13}C, that dynamic $\delta^{13}C$ models with a metabolic tissue replacement term were better supported than models predicting isotopic change from growth alone. Based on the average weight (1,400 g) of Lake Trout collected from Snap Lake for this study and using the predictive relationship in Weidel et al. (2011) for turnover rate relative to body mass, a turnover rate for $\delta^{13}C$ of 165 days is predicted. This would correspond to a period of approximately five to six months prior to July when most samples were collected. Turnover rates of $\delta^{15}N$ appear to reflect the same general patterns as seen for $\delta^{13}C$ (Buchheister and Latour 2010). In winter, cold temperatures and a paucity of prey are likely to generate a pattern of slow (or no) growth, typical of freshwater temperate fishes (Garvey et al. 2004; Byström et al. 2006). In contrast, the warmer isothermal conditions associated with spring would decrease foraging costs, promote extensive foraging, and allow winter energy losses to be reversed and allow for new somatic growth (Henderson et al. 2000).

Despite limited availability of appropriate thermal habitat for Lake Trout in Snap Lake during the summer when most growth is expected to occur, the use of littoral resources by Lake Trout appeared high given that the estimated proportion of carbon from littoral sources was 75%. Based on the summer of 2013, which was not atypical in terms of air temperature, there was a progressive deepening in the 15°C isotherm (Section 8), the upper preferred temperature limit for Lake Trout (Plumb and Blanchfield 2009). This was based on the temperature record derived from a vertical array of temperature loggers at a fixed mooring in the deepest part of the lake. During most of August, the volume of the lake having a temperature below 15°C was less than 1%. Morbey et al. (2006) reported that, although Lake Trout spend most of their time in favourable temperatures within the metalimnion of a lake, they make frequent excursions, albeit of short duration (i.e., in the order of 15 minutes), into lake temperatures exceeding their optimum for physiological performance. Their frequent use of nearshore habitats suggested that feeding in littoral areas was common. The most productive area of the littoral zone of oligotrophic lakes is often less than 3 m in depth (Keast and Harker 1977). Nevertheless, access to and feeding in littoral areas is strongly affected by lake morphometry. Dolson et al. (2009) found that, where habitat use is restricted by temperature, Lake Trout made greater use of littoral resources in circular lakes than lakes that were more reticulate in shape such as Snap Lake. Differences were modified by the relative distance from cold-water refugia in the lake. Although the strength of habitat coupling by mobile predators such as Lake Trout has been shown to be a major factor in governing the stability of food webs (Post et al. 2000; McCann et al. 2005; Rooney et al. 2006), the trend of increasing air temperatures and concurrent effect on lake temperatures may reduce the degree of habitat coupling between littoral and pelagic zones of lakes.

The Snap Lake food chain was clearly benthically driven and similar to other high latitude low productivity lakes (Vadenboncoeur et al. 2003). It was estimated that approximately 75% of carbon in Lake Trout was derived from littoral sources, which is similar to Lake Trout from other lakes at high latitude including Glenn (80%), Nauyuk (75%), and Roberts (50%) lakes in the Northwest Territories (NWT) (Swanson et al. 2011). and Lake NE14 (92%) and Lake I Minus (88%) in Alaska (Sierszen et al. 2003). Partly this is due to benthic algae in the littoral zone; Vander Zanden et al. (2011) reported that the littoral contribution to fish carbon was weakly correlated to the littoral contribution to whole lake primary productivity. It was only for Longnose Sucker where it was possible to measure a direct contribution of carbon in benthic algae to carbon in fish tissue. The amount of carbon contributed by littoral sources may be dependent on the presence of a pelagic prey fish. The amount of littoral carbon for Lake Trout Snap Lake like other northern lakes in NWT and Alaska was much higher than for four lakes (Alexie, Baptiste, Chitty, Drygeese) 250 km south of Snap Lake where littoral carbon ranged from 2% to 20% and Cisco a pelagic forage fish were important in diets (Cott et al. 2011). Increased use of pelagic over littoral forage fish by Lake Trout has been related to changes in lake morphometry with more reticulate lakes favouring the use of pelagic prey as Lake Trout have reduced access to littoral habitats that are increasingly physically removed from mid- and deep-water cold-water refugia (Dolan et al. 2009).

Although much of the carbon contributing to Lake Trout in this study was believed to derive from benthic sources there is large natural variation and often overlap in the $\delta^{13}C$ of terrestrial and aquatic resources (Gu et al. 2011) making it difficult to resolve the relative contribution of each. Basal resources in lake ecosystems originate from three distinct pools: as autochthonous primary production in pelagic (open-water) and benthic (bottom) habitats, and as allochthonous (terrestrial) primary production in adjacent terrestrial habitats. In low-productivity lakes such as Snap Lake that dominate many northern regions, the relative availability of these three resources is controlled by terrestrial inputs and their effects on light attenuation (Ask et al. 2009; Karlsson et al. 2009). Low nutrient concentrations in these systems limit pelagic phytoplankton production. Benthic algae, in contrast, can access nutrients from sediment pore waters (at least on soft substrates) and are thus light, rather than nutrient, limited. Under these conditions, benthic primary production substantially contributes to and may even dominate whole-lake autochthonous production (Vadeboncoeur et al. 2003, 2008; Ask et al. 2009). The use of hydrogen stable isotope ratios (δ^2H), because there is a large separation between terrestrial and aquatic material, has revealed substantial allochthonous resource utilization by zooplankton and fish and zoobenthic consumers with terrestrial subsidies to food webs being more important in smaller than larger systems (Caraco et al. 2010; Cole et al. 2011; Solomon et al. 2011; Karlsson et al. 2012; Wilkinson et al. 2013).

Lake Trout and Burbot are the two top predators in Snap Lake; Lake Trout had a slightly higher trophic position. Cott et al. (2011) reported that the mean trophic position for Lake Trout and Burbot did not differ among four boreal lakes 250 km south of Snap Lake. The indication that Lake Trout in Snap Lake had a higher trophic level may have been a reflection of the distinctiveness of the isotopic niches of the two species. There was evidence of both indirect competition in that both species used the same prey species although in different amounts, and direct competition in the form of predation. Throughout their North American range, Burbot co-occur with Lake Trout and often overlap in habitats and diets (Scott and Crossman 1973). In the Laurentian Great Lakes, Burbot and Lake Trout are both predators of fish in their shared limnetic environment (Schram et al. 2006; Jacobs et al. 2010). If Lake Trout stocks in Snap Lake became depleted, Burbot densities could increase and impede Lake Trout recovery through competition for food sources or through predation (Schram et al. 2006; Jacobs et al. 2010).

The trophic position of Lake Trout in Snap Lake, a measure of food chain length, was higher than that of four boreal lakes (Alexie, Baptiste, Chitty, Drygeese) (Cott et al. 2011) albeit within one trophic level (i.e., within a $\delta^{15}N$ of 3.4‰). Post et al. (2000) reported that food chain length increased with ecosystem size, and the area of the four boreal lakes was approximately one-quarter the area of Snap Lake. The percentage of carbon from littoral sources for these four lakes ranged from 2% to 20%. The higher trophic position of Snap Lake Lake Trout is liklely unrelated to lake productivity as food chain length was reported to be unrelated to lake productivity (Vander Zanden et al. 1999; Post et al. 2000). For Lake Trout, Vander Zanden et al. (2000) reported that trophic level did not increase appreciably as a function of size, which they attributed to a weak predator and prey size relationship as well as there being no relationship between prey fish trophic level and body size.

Trophic structure of the Snap Lake food web has been maintained since the Mine opened and in fact niche size for both lake Trout and Round Whitefish has increased between 1999 and 2013 with no indication of ecosystem fragmentation (Layman et al. 2007). It appears based on an increase in $\delta^{13}C$ of over 2‰ for Lake Trout that the food web is increasingly reliant on littoral-benthic as opposed to pelagic carbon sources. Oligotrophic systems in the absence of changes to the food web show limited temporal change and mostly less than 1‰ for both $\delta^{13}C$ and $\delta^{15}N$ (Jangua and Gerdeaux 2011). However, for Snap Lake, according to archival samples of Lake Trout that were collected in 1999 before the Mine became operational and assuming no change in the stable isotope signatures of prey species, there has been no change in the Snap Lake food chain length based on $\delta^{15}N$ over the period the Mine has been operating but a change in $\delta^{13}C$. The increase in $\delta^{13}C$ values for Lake Trout may indicate an increased reliance on littoral-benthic over pelagic carbon energy sources. Given that Vander Zanden et al. (2011) observed a significant, albeit weak, relationship between the benthic contribution to fish body carbon and the benthic algal contribution to whole-lake primary productivity, the increase in $\delta^{13}C$ values measured for Lake Trout may be a reflection of increased benthic algae in Snap Lake and its consumption by members of the Snap Lake food web (Section 11). Direct consumption of benthic algae was reflected in the mixing model results for Longnose Sucker (20%), which comprised about 10% of Lake Trout diet. Elsewhere, increases in measured $\delta^{13}C$ values have been associated with increased use of nearshore littoral resources. For Lake Ontario Lake Trout, the addition of Round Goby (*Neogobius melanostomus*), a species that is reliant on nearshore carbon, to their diet resulted in an increase in $\delta^{13}C$ values measured for Lake Trout; this increase was on the order of 2‰ to 3‰ (Rush et al. 2012). Similarly, for Lake Whitefish, increased reliance on littoral prey was associated with a 3‰ increase in $\delta^{13}C$ (Rennie et al. 2009).

11.5.5 Conclusions

Stable isotopes and mixing models were successfully used to characterize the food web of Snap Lake providing estimates of consumption for fish species in the lake. For the same sampling effort, the stable isotope study provided considerably more information about the food web than conventional stomach content analysis. To achieve the same level of spatial and temporal resolution of diet, through stomach content analysis, would have required a much greater sampling effort with much greater mortality, and would be associated with increased costs. Two key questions were addressed by this study as follows.

11.5.5.1 Key Question 1: What Eats What in Snap Lake?

In Snap Lake, both Lake Trout and Burbot were generalists consuming both fish and invertebrates (profundal, littoral, and pelagic); Lake Trout were the top predator based on trophic position. Round Whitefish, Longnose Sucker, and Lake Chub consumed mixtures of pelagic, profundal, and littoral organisms.

11.5.5.2 Key Question 2: Is the Snap Lake Food Web Planktonically Driven or Benthically Driven?

The Snap Lake food web is benthically driven with an estimated 75% of the carbon in Lake Trout coming from benthic sources.

11.5.5.3 Recommendation

The Snap Lake food web has been demonstrated to be predominantly benthically driven. No additional studies are recommended at this time.

11.5.6 References

Ask J, Karlsson J, Persson L, Ask P, Bystrom P, Jansson M. 2009. Terrestrial organic matter and light penetration: Effects on bacterial and primary production in lakes. Limnol Oceanogr 54: 2034–2040.

Auerswald K, Wittmer MHOM, Zazzo A, Schaufele R, Schnyder H. 2010. Biases in the analysis of stable isotope discrimination in food webs. J Appl Ecol 47: 936–941.

Barton DR, Johnson RA., Campbell L, Petruniak J, Patterson M. 2005. Effects of round gobies (*Neogobius melanostomus*) on dreissenid mussels and other invertebrates in eastern Lake Erie, 2002–2004. J Great Lakes Res 31 (Suppl. 2): 252–261.

Bearhop S, Adams CE, Waldron S, Fuller RA, Macleod H. 2004. Determining trophic niche width: A novel approach using stable isotope analysis. J Anim Ecol 73: 1007–1012.

Becker GC. 1983. Fishes of Wisconsin. University of Wisconsin Press, Madison, WI, USA.

Ben-David M, Schell DM. 2001. Mixing models in analyses of diet using multiple stable isotopes: A response. Oecologia 127: 180–184.

Buchheister A, Latour RJ. 2010. Turnover and fractionation of carbon and nitrogen stable isotopes in tissues of a migratory coastal predator, Summer Flounder (*Paralichthys dentatus*). Can J Fish Aquat Sci 67: 445–461.

Byström P, Andersson J, Kiessling A, Eriksson LO. 2006. Size and temperature dependent foraging capacities and metabolism: Consequences for winter starvation mortality in fish. Oikos 115: 43–52.

Campbell LM, Thacker R, Barton D, Muir DCG, Greenwood D, Hecky RE. 2009. Re-engineering the Eastern Lake Erie littoral food web: The trophic function of non-indigenous Ponto– Caspian species. J Great Lakes Res 35: 224–231.

Caraco N, Bauer JE, Cole JJ, Petsch SS, Raymond P. 2010. Millennial-aged organic carbon subsidies to a modern river food web. Ecology 91: 2385–2393.

Caut S, Angulo E, Courchamp. F. 2009. Variation in discrimination factors ($\Delta^{15}N$ and $\Delta^{13}C$): The effect of diet isotopic values and applications for diet reconstruction. J Appl Ecol 46: 443-453.

Caut S, Angulo E, Courchamp F, Figuerola J. 2010. Trophic experiments to estimate isotope discrimination factors. J Appl Ecol 47: 948–954.

Christoffersen KS, Jeppesen E, Moorhead DL, Tranvik LJ. 2008. Food-web relationships and community structures in high-latitude lakes. In: Vincent WF, Laybourn-Parrym J (eds), Polar Lakes and

Rivers: Limnology of Arctic and Antarctic Aquatic Ecosystems. Oxford University Press, New York, NY, USA, pp 269-289.

Cole JJ, Carpenter SR, Kitchell JF, Pace ML, Solomon CT, Weidel B. 2011. Strong evidence for terrestrial support of zooplankton in small lakes based on stable isotopes of carbon, nitrogen and hydrogen. PNAS 108: 1975–1980.

Cott PA, Johnston TA, Gunn JM. 2011. Food web position of Burbot relative to Lake Trout, Northern Pike, and Lake Whitefish in four sub-arctic boreal lakes. J Appl Ichthyol 27 (Suppl. 1): 49–56.

Craig H. 1957. Isotopic standards for carbon and oxygen and correction factors for mass spectrometric analysis of carbon dioxide. Geochim Cosmochim Acta 12: 133–149

De Beers. 2013. 2012 Annual Report in Support of the Aquatics Effects Monitoring Program Water License (MV2001L2-0002), Snap Lake Project. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

Dolson R, McCann K, Rooney N, Ridgway M. 2009. Lake morphometry predicts the degree of habitat coupling by a mobile predator. Oikos 118: 1230–1238.

Elrod JH, O'Gorman R. 1991. Diet of juvenile Lake Trout in southern Lake Ontario in relation to abundance and size of pry fishes, 1979-1987. Trans Amer Fish Soc. 120: 290-302.

Fitzsimons J. 2014. The ontogeny of thiamine deficiency in Lake Ontario Lake Trout. Unpublished. Calgary, AB, Canada

Fox J, Weisberg S. 2011. An R Companion to Applied Regression. Second edition. Thousand Oaks CA: Sage. URL: http://socserv.socsci.mcmaster.ca/jfox/Books/Companion.

Garvey JE, Ostrand KG, Wahl DH. 2004. Energetics, predation, and ration affect size-dependent growth and mortality of fish during winter. Ecology 85: 2860–2871.

Glaholt SP Jr, Vanni MJ. 2005. Ecological responses to simulated benthic-derived nutrient subsidies mediated by omnivorous fish. Freshw Biol 50: 1864-1881.

Gu B, Schelske CL, Waters MN. 2011. Patterns and controls of seasonal variability of carbon stable isotopes of particulate organic matter in lakes. Oecologia 165: 1083–1094.

Hecky RE, Hesslein RH. 1995. Contributions of benthic algae to lake food webs as revealed by stable isotope analysis. J N Am Benthol Soc 14: 631–653.

Henderson BA, Trivedi T, Collins N. 2000. Annual cycle of energy allocation to growth and reproduction of Yellow Perch. J Fish Biol 57: 122–133.

Hershey AE. 1990. Snail populations in arctic lakes: Competition mediated by predation? Oecologia 82: 26–32.

Hesslein RH, Hallard KA, Ramlal P. 1993. Replacement of sulfur, carbon, and nitrogen in tissue of growing Broad Whitefish *Coregonus nasus* in response to a change in diet traced by ^{34}S, ^{13}C, and ^{15}N. Can J Fish Aquat Sci 50: 2071–2076.

Hobson KA, Welch HE. 1995. Cannibalism and trophic structure in a high arctic lake: Insights from stable-isotope analysis. Can J Fish Aquat Sci 52: 1195–1201.

Hothorn T, Bretz F, Westfall P. 2008. Simultaneous inference in general parametric models. Biometrical J 50: 346-363.

Hutchinson GE. 1957. Concluding remarks. CSH Symp. Quant Biol 22: 415–427.

Jackson A., Inger R, Parnell AC, Bearhop S. 2011. Comparing isotopic niche widths among and within communities: SIBER-Stable isotope Bayesian ellipses in R. J Anim Ecol 80: 595-602.

Jacobs GR, Madenjian CP, Bunnell DB, Holuszko JD. 2010. Diet of Lake Trout and Burbot in northern Lake Michigan during spring: evidence of ecological interaction. J Great Lakes Res 36: 312–317.

Janjua MY, Gerdeaux D. 2011. Evaluation of food web and fish dietary niches in oligotrophic Lake Annecy by gut content and stable isotope analysis. Lake Reserv Manage 27: 115–127.

Johnson L. 1976. Ecology of arctic populations of Lake Trout, *Salvelinus namaycush*, Lake Whitefish, *Coregonus clupeaformis*, Arctic Char, *S. alpinus*, and associated species in unexploited lakes of the Canadian Northwest Territories. J Fish Res Board Can 33: 2459–2488.

Karlsson J, Bystrom P. 2005. Littoral energy mobilization dominates energy supply for top consumers in subarctic lakes. Limnol Oceanogr 50: 538–543.

Karlsson J, Bystrom P, Ask J, Ask P, Persson L, Jansson M. 2009. Light limitation of nutrient-poor lake ecosystems. Nature 460: 506–509.

Karlsson J, Christensen TR, Crill P. Forster J, Hammarlund D, Jackowicz-Korczynski M, Kokfelt U, Roehm C, Rosen P. 2012. Quantifying the relative importance of lake emission in the carbon budget of a subarctic catchment, J. Geophys. Res., 115, G03006

Keast A, Harker J. 1977. Strip counts as a means of determining densities and habitat utilization patterns in lake fishes. Environ Biol Fish 1: 181–188.

Kionka BC, Windell JT. 1972. Differential movement of digestible and indigestible food fractions in Rainbow Trout, *Salmo gairdneri*. Trans Am Fish Soc 101: 112–115.

Kratina P, Greig HS, Thompson PL, Carvalho-Pereira TSA, Shurin JB. 2012. Warming modifies trophic cascades and eutrophication in experimental freshwater communities. Ecology 93: 1421–1430.

Layman CA, Quattrochi JP, Peyer CM, Allgeier JE. 2007. Niche width collapse in a resilient top predator following ecosystem fragmentation. Ecol Lett 10: 937–944.

Lepak JM, Kraft CE, Weidel BC. 2006. Rapid food web recovery in response to removal of an introduced apex predator. Can J Fish Aquat Sci 63: 569–575.

Liboriussen L, Jeppesen E. 2003. Temporal dynamics in epipelic, pelagic and epiphytic algal production in a clear and a turbid shallow lake. Freshw Biol 48: 418–431.

Madenjian CP, DeSorcie TJ, Stedman RM. 1998. Ontogenic and spatial patterns in diet and growth of Lake Trout in Lake Michigan. Trans Am Fish Soc 127: 236–252.

Mariotti A. 1983. Atmospheric nitrogen is a reliable standard for natural d15N abundance measurements. Nature 303: 685–687.

McCann KS, Rasmussen JB, Unbanhowar J. 2005. The dynamics of spatially coupled food webs. Ecol Lett 8: 513–523.

McDermid J, Shuter B, Lester N. 2010. Life history differences parallel environmental differences among North American Lake Trout (*Salvelinus namaycush*) populations. Can J Fish Aquat Sci 67: 314–325.

Merrick GW, Hershey AE, McDonald ME. 1991. Lake Trout, *Salvelinus namaycush,* control of snail density and size distribution in an arctic lake. Can J Fish Aquat Sci 48: 498-502.

Merrick GW, Hershey AE, McDonald ME. 1992. Salmonid diet and the size, distribution, and density of benthic invertebrates in an arctic lake. Hydrobiologia 240: 225–233.

Mills EL, Casselman JM, Dermott R, Fitzsimons JD, Gal G, Holeck KT, Hoyle JA, Johannsson OE, Lantry BF, Makarewicz JC, Millard ES, Munawar IF, Munawar M, O'Gorman R, Owens RW, Rudstam LG, Schaner T, Stewart TJ. 2003. Lake Ontario: Food web dynamics in a changing ecosystem (1970– 2000). Can J Fish Aquat Sci 60: 471–490.

Mittelbach GG, Persson L. 1998. The ontogeny of piscivory and its ecological consequences. Can J Fish Aquat Sci 55: 1454–1465.

Morbey YE, Addison P, Shuter BJ, Vascotto K. 2006. Within-population heterogeneity of habitat use by Lake Trout *Salvelinus namaycush*. J Fish Biol 69: 1675–1696.

Morbey YE, Couture P, Busby P, Shuter BJ. 2010. Physiological correlates of seasonal growth patterns in Lake Trout *Salvelinus namaycush*. J Fish Biol 77: 2298–2314.

Parnell A, Jackson A. 2013. SIAR: Stable Isotope Analysis in R. R package version 4.2. URL: http://CRAN.R-project.org/package=siar.

Parnell A, Inger R, Bearhop S, Jackson AL. 2010. Source partitioning using stable isotopes: Coping with too much variation. PLoS One 5:e9672.

Perga ME, Gerdeaux D. 2005. Are fish what they eat all year round? Oecologia 14: 598–606.

Plumb JM, Blanchfeld PJ. 2009. Performance of temperature and dissolved oxygen criteria to predict habitat use by Lake Trout *Salvelinus namaycush*. Can J Fish Aquat Sci 66: 2011–2023.

Polis GA, Anderson WB, Holt RD. 1997. Toward an integration of landscape and food web ecology: the dynamics of spatially subsidized food webs. Annual Rev Ecol Syst 28**:** 289–316.

Post DM. 2002. Using stable isotopes to estimate trophic position: Models, methods, and assumptions. Ecology 83: 703–718.

Post DM, Pace ML, Hairston NGJ. 2000. Ecosystem size determines food-chain length in lakes. Nature 405: 1047–1049.

Post DM, Layman CA, Albrey D, Takimoto G, Quattrochi J, Montaña CG. 2007. Getting to the fat of the matter: Models, methods and assumptions for dealing with lipids in stable isotope analyses. Oecolgia 152: 179-189.

Power M, Reist JD, Dempson JB. 2008. Fish in high-latitude arctic lakes. In: Vincent, WF, Laybourn-Parrym, J (eds), Polar Lakes and Rivers, Limnology of Arctic and Antarctic Aquatic Ecosystems. Oxford University Press, Oxford, UK, pp 249–68.

RDCT (R Development Core Team). 2008. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing. Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Rennie MD, Sprules WG, Johnson TB. 2009. Resource switching in fish following a major food web disruption. Oecologia 159: 789–802.

Rooney N, McCann K, Gellner G, Moore JC. 2006 Structural asymmetry and the stability of diverse food webs. Nature 442: 265–269.

Rush SA, Paterson G, Johnson TB, Drouillard KG, Haffner GD, Hebert CE, Arts MT McGoldrick DJ, Backus SM, Lantry BF, Lantry JR, Schaner T, Fisk AT. 2012. Long-term impacts of invasive species on a native top predator in a large lake system. J Freshw Ecol 57: 2342–2355.

Russell LV. 2013. Lsmeas: Least-squares Means. R package version 1.10-4. URL: http://CRAN.R-project.org/package=lsmeans

Sandstrom S, Rawson M, Lester N. 2011. Manual of Instructions for Broad-scale Fish Community Monitoring; Using North American (NA1) and Ontario Small Mesh (ON2) Fillnets. Ontario Ministry of Natural Resources, Peterborough, ON, Canada. Version 2011.1 35 pp + appendices.

Schindler DE, Scheuerell MD. 2002. Habitat coupling in lake ecosystems. Oikos 98: 177–189.

Schram ST, Johnson TB, Seider MJ. 2006. Burbot consumption and relative abundance in the Apostle Island region of Lake Superior. J Great Lakes Res 32: 798–805.

Scott WB, Crossman EJ. 1973. Freshwater Fishes of Canada. Fisheries Research Board of Canada, Ottawa, ON, Canada, 966 pp.

Sierszen ME, McDonald ME, Jensen DA. 2003. Benthos as the basis for arctic lake food webs. Aquat Ecol 37: 437–445.

Solomon CT, Carpenter SR, Clayton MK, Cole JJ, Coloso JJ, Pace ML, Vander Zanden M J, Weidel BC. 2011. Terrestrial, benthic, and pelagic resource use in lakes: Results from a three-isotope Bayesian mixing model. Ecology 92: 1115–1125.

Swanson HK, Kidd KA, Reist JD, Trudel M. 2011. Quantifying importance of marine prey in the diets of two partially anadromous fishes. Can J Fish Aquat Sci 68: 2020–2028.

Turner MA, Schindler DW, Graham DRW.1983 Photosynthesis-irradiance relationships of epilithic algae measured in the laboratory and in situ. In: Wetzel RG (ed), Periphyton of Freshwater Ecosystems. Dr W Junk Publishers,The Hague, Netherlands, pp 73-87.

Vadeboncoeur Y, Vander Zanden MJ, Lodge DM. 2002. Putting the lake back together: Reintegrating benthic pathways into lake food web models. Bioscience 52**:** 44–54.

Vadeboncoeur Y, Jeppesen E, Vander Zanden MJ, Schierup HH, Christoffersen K, Lodge DM. 2003. From Greenland to Green Lakes: Cultural eutrophication and the loss of benthic pathways in lakes. Limnol Oceanogr 48: 1408–1418.

Vadeboncoeur Y, McCann KS, Vander Zanden MJ, Rasmussen JB. 2005. Effects of multi-chain omnivory on the strength of trophic control in lakes. Ecosystems 8: 682–693.

Vadeboncoeur Y, Peterson G, Vander Zanden MJ, Kalff J. 2008. Benthic algal production across lake size gradients: interactions among morphometry, nutrients, and light. Ecology 89: 2542–2552.

Vander Zanden MJ, Vadeboncoeur Y. 2002. Fishes as integrators of benthic and pelagic food webs in lakes. Ecology 83**:** 2152–2161.

Vander Zanden MJ, Shuter BJ, Lester N, Rasmussen JB. 1999. Patterns of food chain length in lakes: a stable isotope study. Am Nat 154: 406–416.

Vander Zanden MJ, Shuter BJ, Lester N, Rasmussen JB. 2000. Within- and among-population variation in the trophic position of a pelagic predator, Lake Trout *Salvelinus namaycush*. Can J Fish Aquat Sci 57: 725–731.

Vander Zanden MJ, Vadeboncoeur Y, Chandra S. 2011. Fish reliance on littoral-benthic resources and the distribution of primary production in lakes. Ecosystems 14: 894–903.

Vanni MJ. 2002. Nutrient cycling by animals in freshwater ecosystems. Ann Rev Ecol System 33: 341–370.

Weidel B, Carpenter S, Kitchell J, Vander Zanden M. 2011. Rates and components of carbon turnover in fish muscle: Insights form bioenergetics models and a whole-lake 13c addition. Can J Fish Aquat Sci 68: 387–399.

Welch HE, Kalff J. 1974. Benthic photosynthesis and respiration in char lake. J Fish Res Board Can 31: 609–620.

Wilkinson GM, Carpenter SR, Cole JJ, Pace ML, Yang C. 2013. Terrestrial support of pelagic consumers: patterns and variability revealed by a multi-lake study. Freshw Biol 58: 2037–2049.

SECTION 12

WEIGHT OF EVIDENCE

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

List of Acronyms

AEMP	Aquatic Effects Monitoring Program
BsM	Broad-scale Monitoring
CCME	Canadian Council of Ministers of the Environment
CPUE	catch per unit effort
De Beers	De Beers Canada Inc.
EAR	Environmental Assessment Report
ISQG	Interim Sediment Quality Guideline
Mine	Snap Lake Mine
MVLWB	Mackenzie Valley Land and Water Board
N	nitrogen
P	phosphorus
PEL	Probable Effect Level
SAB	Science Advisory Board for Contaminated Sites in British Columbia
SD	standard deviation
SSWQO	site-specific water quality objective
TDS	total dissolved solids
TKN	total Kjeldahl nitrogen
TOC	total organic carbon
WOE	weight of evidence
WQ	water quality

Units of Measure

%	percent
<	less than
↑	increase
↑/↓	Rating 1
↑↑/↓↓	Rating 2
↑↑↑/↓↓↓	Rating 3
↓	decrease

12 WEIGHT OF EVIDENCE 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; the weight of evidence (WOE) integration examines the linkages between exposure and resulting biological responses as required by the Mine's Water Licence (MVLWB 2013). Specifically, Schedule 6, Part G, Conditions Applying to Aquatic Effects Monitoring of the Water Licence MV2011L2-0004 (MVLWB 2013), 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 that requirement by conducting a WOE integration of the measures of contaminant and nutrient exposure, and biological response described in the findings of the AEMP Component Sections. The integration approach is based on Section 7 of the AEMP Design Plan for Snap Lake (De Beers 2014), and supports the AEMP Response Framework by distinguishing between nutrient enrichment and toxicological impairment as the cause of any observed biological responses. When Action Level conditions are met for a given biological component, the WOE integration informs which Action Level group is triggered (i.e., Action Levels for Toxicological Impairment, for Nutrient Enrichment, or both), and then contributes this system understanding to inform response planning.

The WOE integration follows principles described in the scientific literature (e.g., Chapman and Anderson 2005; McDonald et al. 2007), provincial, and federal guidance in Canada (e.g., Environment Canada and Ontario Ministry of the Environment 2008; SAB 2008; Environment Canada 2012). It examines, qualitatively, the strength of evidence indicating that enrichment effects and/or toxicity effects are occurring in Snap Lake.

12.1 Approach

Weight of Evidence 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" (Environment Canada 2012).

This definition encompasses a range of practice, ranging from best professional judgment assessments to complex quantitative methods (Environment Canada 2012). It is an established and accepted method for integrating environmental assessment data (e.g., Chapman and Anderson 2005; McDonald et al. 2007; Chapman and Smith 2012). Guidance documents on WOE methods have been developed and are in use in Canada both provincially (e.g., Environment Canada and Ontario Ministry of the Environment 2008; SAB 2008) and federally (e.g., Environment Canada 2012).

In general terms, in this WOE integration, the endpoint results for each AEMP component were *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 WOE integration indicating the degree of support for alternative hypotheses regarding the type of effect in Snap Lake. Key aspects of the approach were:

- It indicates the relative degree of support that the AEMP findings provide for two alternative hypotheses: *nutrient enrichment* versus *toxicological impairment*.
- Hypotheses are examined for each broad ecosystem component included in the AEMP for Snap Lake.
- 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.
- This information is integrated 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 12.1.1 to 12.1.5.

12.1.1 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 AEMP Design Plan (De Beers 2014); a brief overview focussed on components relevant to the WOE integration is provided below.

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[1]" of Snap Lake has occurred.

The Mine-related stressors of potential concern relevant to WOE integration for Snap Lake are:

- total dissolved solids (TDS) and its constituent ions;
- metals[2]; and,
- the nutrients phosphorus (P) and nitrogen (N).

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 main sources of nutrients in Snap Lake are: 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, phosphorus mainly in treated domestic waste water, and potentially in surface runoff. Calcium, a component of TDS can also be a nutrient for zooplankton and benthic invertebrates.

The EAR (De Beers 2002) 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 WOE 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 to 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).

---

[1] The term "pollution" is used to indicate contamination that results in adverse biological effects to populations or communities of organisms.

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

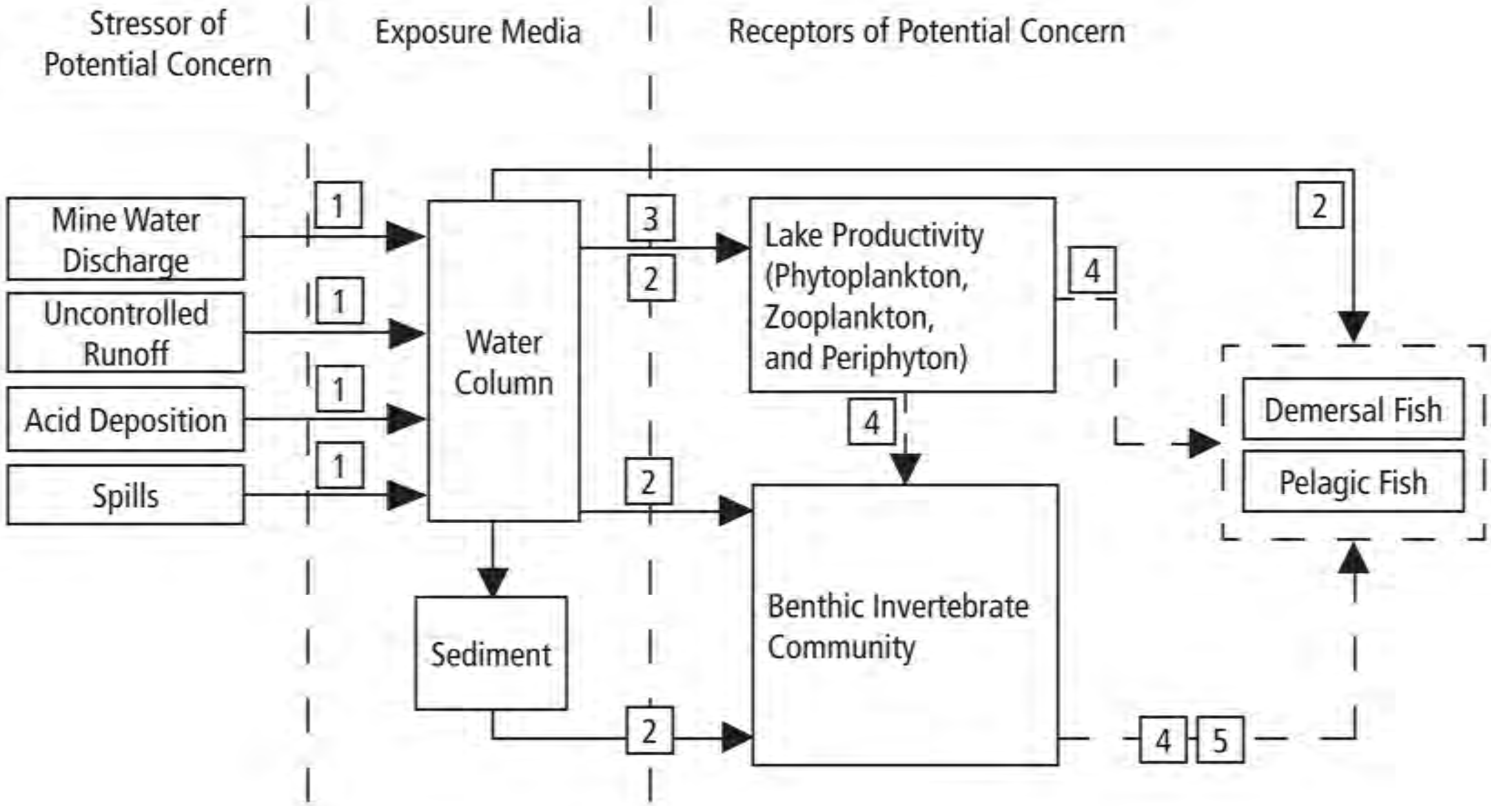
Of the above phytoplankton, zooplankton, benthic invertebrates, and fish are included in the AEMP because they represent direct effects to the biological community of Snap Lake. Periphyton are currently the subject of special studies (Section 11.1) and are not yet included in standard AEMP monitoring.

The pathways by which the sources identified above say 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 12-1). 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; 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, resulting 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.



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

SNAP LAKE MINE

Conceptual Site Model
Relevant Exposure Pathways

PROJECTION: N/A		DATUM: N/A	DE BEERS GROUP OF COMPANIES
SCHEMATIC ONLY, NOT TO SCALE			
FILE No: 1313490001-7010_12-1			DATE: April 12, 2013
JOB No: 14-1349-0004	REVISION No: A		Figure **12-1**
OFFICE: GOLDER VICTORIA	DRAWN: JEF	CHECK: PMC	

L:\2013\1349\13-1349-0001\Phase 7000\1313490001-7010_12-1.dwg

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.

The conceptual model 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.

The WOE integration provides a systematic approach for distinguishing between these two hypotheses. It is anticipated that these would be the two main types of effects resulting from treated effluent release.

12.1.2 Endpoints

The 2013 AEMP included parameters and testing representing the following types of information: water quality and chronic toxicity at the edge of the treated effluent mixing zone (nutrients and chemical contaminants); sediment quality (limited stations); fish tissue chemistry; plankton community; benthic invertebrate community and fish community.

In the AEMP Design Plan (De Beers 2014), the parameters and biological variables measured in these components have been 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).
- ***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, fish health, and fish community monitoring.

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.

Sediment quality monitoring for the whole of Snap Lake main basin was not conducted in 2013; therefore, the effect summary for 2013 sediment quality was based on the results observed in 2012. Sediment quality tends to integrate fluctuations in treated effluent loadings and water quality over time and, as indicated in previous AEMP studies for Snap Lake (e.g., De Beers 2013), change more slowly than water quality. Therefore, the 2012 sediment findings were deemed to provide an appropriate representation of potential sediment-borne exposure to toxicants and nutrients, in the absence of basin-wide monitoring in 2013.

A fish community monitoring study is conducted every three years. This was the first year that the full program was completed under the AEMP Design Plan (De Beers 2014) and because this program is new to the AEMP, the interpretation of fish community information is not considered sufficiently well-understood to be formally incorporated into the WOE approach. As such, the results of the fish community monitoring study were considered as supporting information but no formal WOE ranking for fish health and community monitoring was made for 2013. The understanding of the fish community monitoring study this year (2013), and in 2016 (the next time it is conducted) will help to inform the inclusion of 2016 fish community monitoring findings in a formal WOE ranking for fish health and community monitoring in the 2016 AEMP report.

12.1.3 Endpoint Response Ratings

The starting point for the WOE 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 12.1.4), 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 2013 AEMP is presented in Table 12-1. 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:

- **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.
- **Rating 2** – This rating includes situations where greater changes, responses, or trends in exposure (i.e., outside normal range[3]), 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.
- **Rating 3** – This rating indicates the strongest level response in exposure or biological response endpoints. None of the endpoints in the WOE integration conducted in the 2013 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, benthic, or fish community monitoring). 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 as to 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 achieved, then it was 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.

Note that 2013 fish community monitoring findings were not rated but rather considered qualitatively as supporting information based on the rationale described in Section 12.1.2.

[3] "Normal Range" is determined based on +/- 2SD in Snap Lake Main Basin baseline and +/- 2SD in reference lakes, and/or other appropriate considerations.

Table 12-1 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)	less than EAR prediction	greater than AEMP Benchmark[b]	greater than 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 greater than baseline normal range[d]	
	Toxicity at edge of mixing zone	no persistent toxicity	sublethal toxicity observed at edge of mixing zone in two 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)	less than EAR prediction	greater than AEMP benchmark	greater than 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 greater than baseline normal range	
Exposure – Sediment Quality (potential toxicants)	**Comparison to benchmarks** (where they exist)	less than ISQG	greater than ISQG	greater than 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]	
Exposure – Fish Tissue Chemistry (potential toxicants)	**Snap Lake compared to reference lakes**	no difference	difference in mean concentration	Snap Lake mean greater than normal range	Rating 2 in both endpoints
	Snap Lake compared to baseline	no difference	difference in mean concentration	Snap Lake mean greater than normal range	

Table 12-1 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 – 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)	
Field Biological Responses – Benthic Invertebrate 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** Condition, Relative Gonad Size, Relative Liver Size	Small-bodied fish health not included in the 2013 AEMP			
	Fish community monitoring Endpoints to be developed	Ratings to be developed by 2016 – fish community monitoring findings included as supporting information for 2013			

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; CCME = Canadian Council of Ministers of the Environment; EAR = Environmental Assessment Report; ISQG = Interim Sediment Quality Guideline; PEL = Probable Effect Level; SD = standard deviation.

12.1.4 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 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. This means 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).

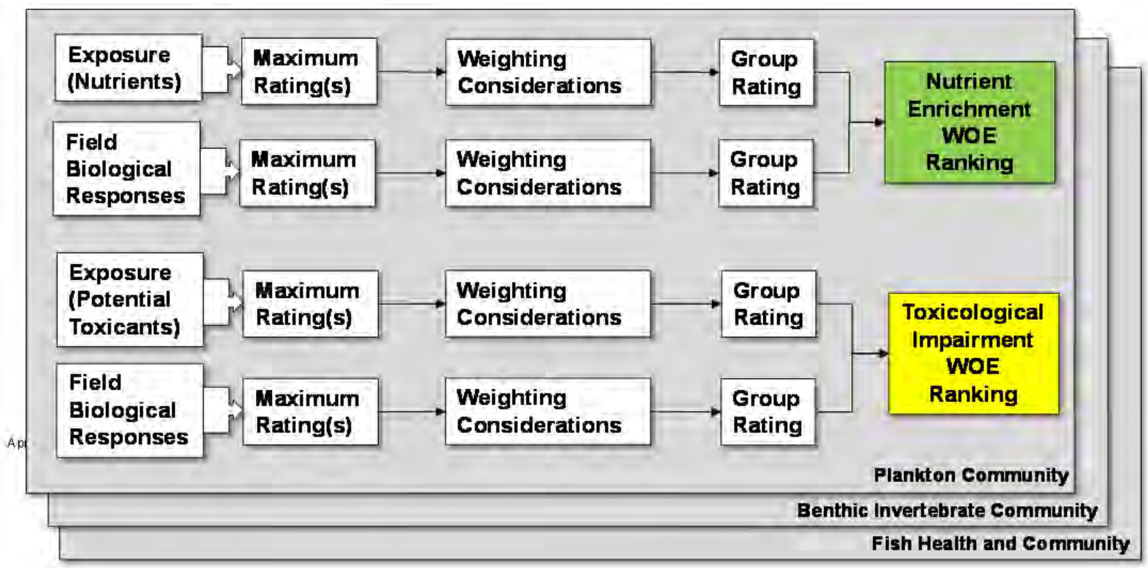
Direction considerations were applied to field biological response endpoints only to reflect the degree of support that an observed biological response contributes to the two alternative hypotheses. These considerations were contingent on the observed direction of change or relationship. For example, 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 12.2.1.

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 2013. 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 WOE integration, a discussion of the rationale was provided.

12.1.5 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 12-2 provides a graphical summary of the overall integration process.

Figure 12-2 Conceptual Integration Process Applied in the Weight of Evidence Assessment



WOE = weight of evidence.

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 WOE 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 WOE 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 WOE integration and the AEMP Response Framework described in the AEMP Design Plan. The WOE 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.

Weight of Evidence rankings were not determined for fish health and community for the 2013 AEMP. This year, 2013, was the first year the full program was completed under the AEMP Design Plan (De Beers 2014). Endpoints and ratings for fish community monitoring data have not yet been developed and therefore the fish community monitoring results were considered qualitatively to provide context for the rating results for fish tissue chemistry.

12.2 Results

12.2.1 Endpoint Rating Results

Tables 12-2 to 12-6 provide the endpoint summaries for each AEMP component (with the exception of sediment quality and fish community monitoring, which are discussed in the text). The endpoint summaries categorize the responses for the endpoints associated with each AEMP component according to the response ratings presented in Table 12-1. These endpoint summary results have been formulated based on the data analyses and interpretation described in Sections 3 through 6, and Sections 8 and 9.

Additional discussion of the endpoint responses relevant to each hypothesis is provided in the WOE analyses in Sections 12.2.2 and 12.2.3.

Table 12-2 Water Quality Endpoint Summary

Endpoint Group	Parameter Grouping	List of Parameters	Endpoint Ratings: Comparison to Benchmarks	Endpoint Ratings: Trends in Snap Lake Compared to Reference Lakes	Endpoint Ratings: Comparison of Snap Lake Main Basin to Normal Range
Toxicants					
Parameters with Toxicological Benchmarks	major ions	chloride, fluoride, nitrate	↑	↑	↑↑
		ammonia, calcium[(a)], magnesium[(a)], nitrite, sulphate, TDS (calculated)	no response	↑	↑↑
	Metals	boron[(b)], molybdenum[(b)], nickel[(b)], strontium[(c)], uranium[(b)]	no response	↑	↑↑
		aluminum, arsenic, cadmium, chromium (total and hexavalent), copper, iron, lead, mercury, selenium, silver, thallium, zinc	no response	no response	no response
Parameters without Toxicological Benchmarks	major ions	potassium, sodium, silica	n/a	↑	↑↑
	Metals	barium, lithium, rubidium	n/a	↑	↑↑
		manganese	n/a	↑	↑
		cobalt	n/a	↑	no response
		antimony, titanium, vanadium	n/a	no response	no response
Nutrients					
Parameters with Enrichment Benchmarks	major ions	calcium[(a)], TDS (calculated),	no response	no response	no response
	nitrogen compounds	ammonia[(a)], nitrate[(a)]	no response	no response	no response
	phosphorus compounds	total phosphorus	no response	no response	no response
Parameters without Enrichment Benchmarks	major ions	silica	n/a	↑	↑↑
	nitrogen compounds	nitrite, total nitrogen	n/a	↑	↑↑
		TKN	n/a	↑	↑
	phosphorus compounds	dissolved inorganic phosphorus, ortho-phosphate, total dissolved phosphorus	n/a	no response	no response
	carbon compounds	TOC	n/a	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) EAR prediction used as AEMP benchmark for these constituents.

b) Parameter concentration is well-below the benchmark, suggesting that the trends and differences from normal range are of low toxicological significance.

c) There is no EAR prediction or AEMP benchmark for strontium but observed concentrations are below the recommended SSWQO.

n/a = not applicable for this parameter grouping; TDS = total dissolved solids; TKN = total Kjeldahl nitrogen; TOC = total organic carbon.

Table 12-3 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 12-4 Fish Tissue Chemistry Endpoint Summary

Endpoint Group	List of Parameters	Endpoint Ratings	
		Snap Lake Compared to Reference Lakes	Snap Lake Compared to Baseline
Lake Trout Muscle	phosphorus	no response or decrease	↑
	potassium	no response or decrease	↑
	selenium	↑	↓
	strontium	no response or decrease	↑
	thallium	↑↑	not tested[(a)]
	remaining parameters	no response or decrease	no response or decrease
Round Whitefish Muscle	cesium	↑↑	↑↑
	magnesium	↑	no response or decrease
	mercury	↑	↓
	phosphorus	↑	no response or decrease
	potassium	↑↑	no response or decrease
	sodium	no response or decrease	↑
	strontium	no response or decrease	↑
	thallium	↑↑	not tested[(a)]
	remaining parameters	no response or decrease	no response or decrease

Note: ↓ or ↑ indicates a statistically significant difference in the direction indicated; ↑↑ or ↓↓indicates a statistically significant difference in the direction indicated that is also beyond normal range. A decrease/negative change or relationship in Snap Lake was only rated if it provided context for an observed increase in the parameter in question (i.e., for selenium and mercury, the decreases compared to baseline provided context for observed increases relative to reference lakes).

a) There are insufficient baseline data for thallium to allow for statistical testing with baseline (<50% of samples above detection limit); therefore, no Rating determination was possible for “compared to baseline”.
< = less than.

Table 12-5 Plankton Community Endpoint Summary

Endpoint Group	Endpoint	Rating	Description	Hypothesis Supported
Phytoplankton				
Chlorophyll a	trends in Snap Lake compared to reference lakes	no response	-	-
Abundance	trend in Snap Lake compared to reference lakes	↑	increased from baseline but has remained relatively consistent from 2007 to 2013	enrichment
	Snap Lake compared to baseline	↑	2013 abundance was six times greater than baseline	
Biomass	trend in Snap Lake compared to reference lakes	↑ and ↓	there was a clear increasing trend from 2004 to 2009, followed by a decreasing trend from 2009 to 2013 back to near baseline values	
	Snap Lake compared to baseline	↑	2013 biomass value was 1.5 times greater than baseline	
Community	community structure	↑↑/↓↓	changes in relative biomass/abundance at functional group level (shift from chrysophyte-cyanobacteria dominated community to diatom dominated back to chrysophyte-cyanobacteria dominated community)	enrichment
Zooplankton				
Abundance	trend in Snap Lake compared to reference lakes	no response	-	toxicity
	Snap Lake compared to baseline	↓	abundance was 1.6 times lower in 2013 compared to baseline	
Biomass	trend in Snap Lake compared to reference lakes	↓	overall decrease from baseline but variable since 2009	
	Snap Lake compared to baseline	↓	biomass was 1.4 times lower in 2013 compared to baseline	
Community	community structure	↑↑/↓↓	changes in relative biomass/abundance at functional group level (shift from calanoid copepods back to cyclopoid copepods)	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.

“-“= description of response not necessary for non-responsive endpoints.

Table 12-6 Benthic Invertebrate Community Endpoint Summary

Endpoint Group	Endpoint	Rating	Description	Hypothesis Supported
Total Density Richness Simpson's Diversity Index	trends in Snap Lake compared to reference lakes	no response	-	-
	Snap Lake compared to Northeast Lake	no response	-	-
Evenness	trends in Snap Lake compared to reference lakes	↑	decreasing evenness trend in Northeast Lake, but evenness relatively constant in main basin of Snap Lake	either
	Snap Lake compared to Northeast Lake	no response	-	-
Density of Dominant Taxa	trends in Snap Lake compared to reference lakes	↓ and ↑ (Pisidiidae)	variable trend (decrease from 2010 to 2012 followed by an increase in 2013) in main basin of Snap Lake compared to a relatively constant density in Northeast Lake and decrease in Lake 13	enrichment
	Snap Lake compared to Northeast Lake	↓ (*Micropsectra*)	*Micropsectra* density statistically lower in main basin of Snap Lake compared to Northeast Lake	either
		↑ (*Valvata* and *Tanytarsus*)	*Valvata* and *Tanytarsus* density statistically higher in main basin of Snap Lake compared to Northeast Lake	enrichment
Community	community structure	↑/↓ (relative abundance)	higher relative density of Pisiidiidae and lower relative density of total Chironomidae in the main basin of Snap Lake in 2013 and relative to Northeast Lake and Lake 13	enrichment

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.

"-"=description of response not necessary for non-responsive endpoints.

12.2.1.1 Exposure Components

Water Quality

For ease of interpretation and presentation of the summary for water quality, the parameters were grouped into two overall categories: parameters with benchmarks; and, parameters without benchmarks, with a further distinction between toxicological benchmarks and enrichment benchmarks. 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 included in the groupings for both types of responses.

A summary of endpoint ratings for water quality is provided in Table 12-2. Chloride, fluoride, and nitrate were above AEMP benchmarks, but were below the recommended site-specific water quality objectives, which resulted in a Rating 1 for Comparison to Benchmarks. Major ions (including chloride, fluoride, and nitrate), TDS, and several metals exhibited an increasing trend in Snap Lake compared to reference lakes (maximum of Rating 1) and concentrations were statistically higher compared to the normal range (maximum of Rating 2).

No parameters regarded as nutrients (e.g., phosphorus and nitrogen compounds, certain major ions) exceeded AEMP benchmarks (for enrichment) or EAR predictions in Snap Lake, where benchmarks and predictions were available. However, nitrogen compounds, TDS, and some major ions exhibited an increasing trend in Snap Lake compared to reference lakes, resulting in a Rating 1 for this endpoint. The parameters that exhibited an increasing trend were also statistically higher in the main basin of Snap Lake compared to the normal range, resulting in a maximum Rating 2 for all but total Kjeldahl nitrogen (TKN; which was Rating 1).

Treated Effluent Toxicity

Endpoint ratings for toxicity tests carried out on treated effluent from the edge of the mixing zone are summarized in Table 12-3. There was no indication of treated effluent toxicity at the edge of the mixing zone in 2013 (i.e., rating of “no response” for each test).

Sediment Quality

As per the AEMP Redesign Plan, sediment was not collected as part of the 2013 AEMP; therefore, the endpoint ratings from the 2012 AEMP were used for the purposes of the WOE analysis. This assumed that sediment quality had not changed between 2012 and 2013 to a degree that would change WOE conclusions. This assumption was considered reasonable given that sediment quality tends to integrate fluctuations in treated effluent loadings and water quality over time, and as indicated in previous AEMP studies for Snap Lake (e.g., De Beers 2013), sediment quality changes slower than water quality.

In 2012, the most pronounced responses for potential toxicants in sediments were found for bismuth, selenium, sodium, and strontium, which each exhibited increasing temporal trends and had concentrations in the main basin of Snap Lake that were beyond the baseline normal range. The toxicological significance of these differences were considered uncertain because there are no Canadian Council of Ministers of the Environment (CCME) Interim Sediment Quality Guidelines (ISQGs) or Probable Effect Level (PELs) for these metals. However, the trends and differences for these metals indicated potential toxicant exposure resulting in a maximum classification of Rating 2.

With respect 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 were deemed to indicate a potential enrichment "signature" in the water column. In contrast, total organic carbon (TOC) was naturally high in Snap Lake and Northeast Lake with no apparent differences between the two.

Fish Tissue Chemistry

Endpoint ratings for fish tissue chemistry are summarized in Table 12-4. For Lake Trout muscle, selenium and thallium tissue concentrations in 2013 were significantly greater in Snap Lake compared to reference lakes, with the difference in concentrations resulting in Rating 1 for selenium and Rating 2 for thallium. However, the selenium concentrations in Snap Lake were lower than baseline (Rating 1 for a decrease) suggesting that the Snap Lake versus reference difference is not caused by the Mine. Phosphorus, potassium, and strontium tissue concentrations were significantly higher in Snap Lake compared to baseline, resulting in a Rating 1.

For Round Whitefish muscle, several analytes (cesium, magnesium, phosphorus, potassium, and thallium) were significantly greater in Snap Lake in 2013 compared to reference lakes. Of these, cesium, potassium, and thallium also beyond the normal range resulting in the Rating 2 for these parameters. Magnesium, phosphorus, and potassium were either not different or lower compared to baseline. Cesium, sodium, and strontium tissue concentrations were significantly greater in Snap Lake in 2013 relative to baseline, with the difference in cesium concentrations resulting in a Rating 2 for this endpoint because it was beyond the normal range. Mercury had mixed results being higher in Snap Lake than the reference lakes (Rating 1) but lower than baseline (Rating 1 for a decrease). This suggested that the Snap Lake versus reference difference is not caused by the Mine. Statistical comparisons of thallium concentrations between Snap Lake and baseline were not made due to insufficient baseline data for this metal.

12.2.1.2 Biological Response Components

For each endpoint where a response was observed in 2013, 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 WOE integration; answers could be *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 benthic 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, and 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.

Plankton Community

A summary of ratings for plankton community endpoints is provided in Table 12-5. The rationale for the rating results was:

- In Snap Lake, phytoplankton abundance and biomass in 2013 were both higher than the baseline normal range. In Snap Lake, relative to reference lakes, phytoplankton abundance has exhibited an overall increasing trend but has remained relatively consistent from 2007 to 2013. Compared to reference lakes, the trend in phytoplankton biomass has been equivocal, with an increasing trend prior to 2009 followed by a decreasing trend from 2009 to 2013 back to near baseline conditions. A maximum of Rating 1 was applied to phytoplankton abundance and biomass endpoints. The increased abundance and biomass of phytoplankton appeared to support an explanation of enrichment.
- A phytoplankton community shift has been evident in Snap Lake since 2006. Cyanobacteria were the dominant group in 2006. From 2007 to 2012, the relative proportion of cyanobacteria biomass decreased and the community shifted to a diatom-chrysophyte co-dominated community; however, the relative proportion of cyanobacteria increased from 2.5% in 2012 to 20% in 2013. This change at the functional group level resulted in a Rating of 2 for phytoplankton community structure. The type of shift in the phytoplankton community appeared consistent with enrichment.
- Zooplankton abundance in Snap Lake was lower in 2013 compared to baseline normal range, although the difference was considered subtle (1.6 times lower). There was no trend observed in Snap Lake zooplankton abundance compared to reference lakes. There has been an overall

decreasing trend in zooplankton biomass in Snap Lake compared to reference lakes since 2009 although the trend has been variable. Zooplankton biomass in Snap Lake was lower than baseline conditions but, similar to abundance, the difference is considered subtle (1.4 times lower). A maximum of Rating 1 was given to zooplankton abundance and biomass endpoints. If enrichment or toxicity were the only factors affecting zooplankton biomass and abundance, then the decreased biomass and abundance would be consistent with very mild toxicological impairment, but these changes could also be explained by other factors such as food supply, predation, or inter-annual variation in regional factors (e.g., temperature, light).

- A community shift has also been evident for zooplankton since 2004. From 2004 to 2009, calanoid copepod dominance decreased with increasing dominance of rotifers. From 2009 to 2013, calanoid copepod dominance began to increase, by 2013 accounting for about 60% of the overall community. This change at the functional group level resulted in a Rating of 2 for zooplankton community structure. It appears that the zooplankton community shift has paralleled that for phytoplankton (suggesting an enrichment cause).

Benthic Invertebrate Community

Ratings for benthic invertebrate community endpoints are summarized in Table 12-6; and specific considerations applied for the ratings were:

- There has been a variable trend in the Pisidiidae density in Snap Lake compared to reference lakes with a decrease from 2010 to 2013 followed by an increase in 2013. The trend was given a Rating 1, but overall is considered equivocal given the variability.
- In 2013, *Microtendipes* density was significantly lower in Snap Lake compared to Northeast Lake while the densities of *Valvata* and *Tanytarsus* were significantly higher. The lower density of *Microtendipes* observed in Snap Lake compared to Northeast Lake supports an explanation of toxicity, or possibly enrichment, in the case that other dominant species were outcompeting *Microtendipes*. The higher density of *Valvata* and *Tanytarsus* combined with the apparent community shift based on relative abundance of dominant taxa supports an explanation of nutrient enrichment. Rating 1 was applied for these benthic invertebrate community endpoints; the responses were considered mild and consistent with EAR predictions.
- Evenness exhibited an increasing trend in Snap Lake relative to Northeast Lake, resulting in a Rating 1. 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.

Fish Community Monitoring

The fish community monitoring program examined a range of fish population parameters that included relative abundance (as determined by the Broad-scale Monitoring [BsM]), size (length and weight), growth, age composition, survival, and reproductive capacity (age at maturity and fecundity) among the study lakes. Of these, the most relevant parameters for eventual inclusion into the WOE are expected to be abundance (catch per unit effort [CPUE] as indicated by the BsM), growth (weight-length-age relationships), and species composition (limited to those species that are accurately indexed by the BsM). In the 2013 study, some statistically significant differences were noted between fish population parameters from Snap Lake and the reference lakes. However, those differences could reasonably be attributed to natural variation or differences in fish community structure in each lake and are not thought to support either the Nutrient Enrichment or Toxicant Hypotheses. There were no discernable changes to the Snap Lake fish community attributable to Mine-related changes.

12.2.2 Toxicological Impairment Weight of Evidence Analysis

The WOE integration describing the integration for potential toxicological impairment of the plankton community, the benthic invertebrate community, and the fish community is summarized in Table 12-7.

12.2.2.1 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 2013 the water quality parameters that exhibited the strongest and most consistent responses in Snap Lake were chloride, fluoride, and nitrate. Chloride, fluoride, and nitrate each had exceedances of their respective CCME water quality guidelines, combined with increasing trends in Snap Lake and concentrations that were outside of the baseline normal range (maximum Rating of 2). However, since the primary source of chloride, fluoride, and nitrate is the treated effluent, increases in these parameters are associated with elevated calcium and hardness, which reduce the potential for toxicity effects associated with these (and other) parameters. 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. These parameters did not exceed their respective site-specific water quality objectives, meaning that 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.
- ***Field Biological Responses*:** For the plankton community in 2013, the zooplankton community exhibited responses most consistent with mild toxicological impairment. There was an indication of a slightly lower zooplankton abundance and biomass in the main basin of Snap Lake in 2013 than was present at baseline, and this was combined with a functional-group community shift. These 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 responses appear more likely due to enrichment followed by a compensatory community shift, especially given that phytoplankton biomass in Snap Lake remains above baseline. Based on these findings, the phytoplankton endpoint group was judged to be at Rating 0 overall with respect to this hypothesis.

Table 12-7 Weight of Evidence Integration for the Toxicological Impairment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Key Supporting Evidence/Rationale	Group Rating	WOE Rank and Rationale
Plankton Community					
Water Quality (Exposure)	comparison to benchmarks	↑	concentrations of chloride, fluoride, and nitrate exceed AEMP benchmarks	↑	**WOE Rank 1** -WQ exposure has increased for key parameters and is exceeding generic conservative guidelines, but not site-specific water quality objectives. There was no indication of treated effluent toxicity at the edge of the mixing zone in 2013. -Phytoplankton response does not appear consistent with toxicological impairment; it is better explained by nutrient enrichment and a community shift. Zooplankton response is consistent with mild toxicological impairment, but could also be related to trophic dynamics such as top-down feeding pressure and/or bottom-up changes in food supply.
	trends in Snap Lake compared to Reference Lakes	↑	concentrations of chloride, fluoride, and nitrate, other major ions, TDS, and multiple metals have increasing trends relative to Northeast Lake		
	comparison to baseline normal range	↑↑	concentrations of chloride, fluoride, and nitrate, other major ions, TDS, and multiple metals exceed baseline concentrations		
	toxicity at edge of mixing zone	no response	-		
WOE Phytoplankton Community (Field Biological Response)	chlorophyll *a*	no response	-	0	
	abundance	(↑)	increased abundance does not indicate toxicity		
	biomass	(↑) and ↓	biomass trend is indicative of enrichment		
	community structure	↑↑/↓↓	community shift is indicative of enrichment rather than toxicity		
Zooplankton Community (Field Biological Response)	abundance	↓	slightly lower abundance could indicate toxicity	↓	
	biomass	↓	slightly lower biomass could indicate toxicity		
	community structure	↑↑/↓↓	community shift parallels the shift for phytoplankton which is associated with enrichment		

Table 12-7 Weight of Evidence Integration for the Toxicological Impairment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Key Supporting Evidence/Rationale	Group Rating	WOE Rank and Rationale
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 2013. -In 2012, multiple sediment metals were displaying increasing temporal trends and were beyond the baseline normal range for Snap Lake Main Basin. However, none of the metals that exceeded the ISQG were indicating differences from reference or baseline conditions, or trends in Snap Lake. -The overall benthic community response does not appear consistent with toxicological impairment, but the possibility of selective toxicity to *Micropsectra* cannot be ruled out. The overall response is better explained by nutrient enrichment or inter-annual variation.
Sediment Quality (Exposure)	overall assessment	↑↑	in 2012 (De Beers 2013), most pronounced responses were found for bismuth, selenium, sodium, and strontium. Sediment quality guidelines were not exceeded		
Benthic Invertebrate Community (Field Biological Response)	total density richness Simpson's Diversity Index	no response	-	↓	
	evenness	(↑)	change does not indicate toxicity		
	density of dominant taxa (trends in Snap Lake compared to Reference Lakes)	(↑) and ↓ (Pisidiidae density)	variable trend is indicative of a community shift but unlikely to be related to toxicity		
	density of dominant taxa (Snap Lake compared to Northeast Lake)	↓ (*Micropsectra* density)	lower density in Snap Lake might indicate selective toxicity to this species		
		(↑) (*Valvata* and *Tanytarsus* density)	increased abundance does not indicate toxicity		
	community structure	↑/↓	change to community structure is most likely explained by enrichment		

Table 12-7 Weight of Evidence Integration for the Toxicological Impairment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Key Supporting Evidence/Rationale	Group Rating	WOE Rank and Rationale
Fish Health and Community					
Fish Tissue Chemistry – Lake Trout Muscle (Exposure)	Snap Lake compared to Reference Lakes	↑↑	thallium was higher in Snap Lake compared to baseline and exceeded normal range	↑↑	-WOE Rank not estimated because endpoints and ratings for fish community are still under development. -Exposure endpoints indicated an increase in some metals in fish tissue. Cesium and thallium are the two metals showing the highest changes in Snap Lake relative to reference and baseline, but it was uncertain how these increased metal concentrations are connected to the Mine. Strontium is consistently elevated in both species relative to baseline and has a clear linkage to treated effluent release. -Fish community monitoring findings did not indicate any impairment response to the increased metals accumulation in Snap Lake.
	Snap Lake compared to Baseline	↑	phosphorus, potassium, and strontium were higher in Snap Lake compared to baseline.		
Fish Tissue Chemistry – Round Whitefish Muscle (Exposure)	Snap Lake compared to Reference Lakes	↑↑	cesium, potassium, and thallium were higher in Snap Lake compared to reference and exceeded normal range		
	Snap Lake compared to Baseline	↑↑	cesium was higher in Snap Lake compared to baseline and exceeded normal range		
		↑	sodium and strontium were higher in Snap Lake compared to baseline		
Small-bodied fish health (Field Biological Response)	Not included in the 2013 AEMP				
Fish Community (Field Biological Response)	Endpoints and ratings have yet to be developed. The fish community monitoring findings indicated that there were no discernable changes to the Snap Lake fish community attributable to Mine-related increases in exposure to substances of toxicological concern.				

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; ISQG = Interim Sediment Quality Guideline; TDS = total dissolved solids; WQ = water quality; WOE = weight of evidence.

Integration of the endpoint groups for plankton community exposure and field biological responses indicates that: water quality was altered in Snap Lake in 2013 including multiple parameters, which could potentially cause toxicological impairment in the plankton community; and, concurrent with this, a slight decrease in zooplankton abundance and biomass combined with a function group-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 2013 was judged to be at WOE Rank 1.

12.2.2.2 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. The maximum classification for sediment quality in 2012 (Rating 2) was also retained overall for the benthic invertebrate community exposure endpoint group.
- ***Field Biological Responses*****:** The pattern of response in the benthic invertebrate community that could be indicative of toxicity was a slight decrease in *Micropsectra* density in Snap Lake compared to Northeast Lake, consistent with a mild impairment response for this species (Rating 1). This response could also be the result of the shift in phytoplankton community since 2009 (i.e., changing food supply), top-down interactions (i.e., predation), or competition with other dominant species which have increased in relative abundance (Rating 1 for community structure). However, in the absence of a clear alternative explanation for the mild responses, Rating 1 was retained for the benthic invertebrate community endpoint group. The remaining benthic invertebrate responses were not indicative of toxicity.

Integration of the endpoint groups indicates that sediment and water quality have been altered in the main basin of Snap Lake including multiple parameters that could potentially cause toxicological impairment in benthic invertebrates. This increased exposure is concurrent with a mild impairment response in one taxonomic group of the benthic invertebrate community. In the absence of other influences unrelated to toxicity or enrichment, this could be due to toxicant exposure. Given the *a priori* weighting considerations discussed in Section 12.1.4, 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.

12.2.2.3 Fish Health and Community

A WOE ranking was not made for fish health and community for the 2013 because small-bodied fish health was not monitored and endpoints and ratings for fish community monitoring have yet to be developed. Rather a qualitative discussion is presented here.

The endpoint findings for exposure (tissue chemistry) in Lake Trout and Round Whitefish were rated and summarized to provide context for the tissue chemistry results.

- ***Exposure*****:** Cesium, strontium, and thallium were identified as parameters in fish tissue in 2013 with the strongest and most consistent increases in Snap Lake relative to reference lakes or baseline conditions (maximum response of Rating 2). Thallium was non-responsive with respect to water quality ratings, and cesium was not reported in water and generally not in effluent; therefore, a link between water concentrations and fish tissue concentrations cannot be discerned. Neither cesium nor thallium were responsive with respect to sediment quality ratings in 2012. Although the link to water quality and sediment quality is unclear for these parameters, the evidence still suggests a difference in Snap Main basin, implying that the changes could be due to the Mine despite no evidence of such a linkage. Strontium has increased relative to baseline in both Lake Trout and Round Whitefish. Water quality and sediment quality also indicate elevated strontium in Snap Lake with the 2013 water concentration being above the baseline normal range.
- ***Field Biological Responses*****:** There were no discernable changes to the Snap Lake fish community, in terms of abundance (CPUE as indicated by the BsM), growth (weight-length-age relationships), and species composition, that were attributable to Mine-related changes in water quality or metals accumulation.

The maximum of Rating 2 was applied overall for the fish tissue chemistry results for 2013, however, this rating should be considered uncertain given the uncertain linkage of cesium and thallium to the Mine. The consistent strontium increases in Snap Lake fish tissue relative to baseline that had a clear linkage to the Mine resulted in Rating 1 for this parameter. There was no evidence of a Mine-related response in the fish community suggesting that the increased metals exposure was not causing any impairment to Lake Trout and Round Whitefish. This suggests negligible evidence for toxicological impairment of fish health and community.

12.2.3 Nutrient Enrichment Weight of Evidence Analysis

The WOE integration describing the evidence for nutrient enrichment of the plankton community and the benthic invertebrate community is summarized in Table 12-8.

Table 12-8 Weight of Evidence Integration for the Nutrient Enrichment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Key Supporting Evidence/Rationale	Group Rating	WOE Rank and Rationale
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. -However, Snap Lake is expected to be phosphorus limited, and phosphorus compounds do not indicate any increase 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. Subtle changes could also be related to trophic dynamics such as top-down feeding pressure and/or bottom-up changes in food supply.
	trends in Snap Lake compared to Reference Lakes	↑	concentrations of calcium, silica, TDS, and nitrogen compounds have upward trends relative to Northeast Lake		
	comparison to baseline normal range	↑↑	mean concentrations of calcium, silica, TDS, and nitrogen compounds in Snap Lake are above normal range		
Phytoplankton Community (Field Biological Response)	chlorophyll *a*	no response	-	↑↑	
	abundance	↑	increased abundance suggests enrichment		
	biomass	↑ and (↓)	trend is indicative of an enrichment response combined with a compensatory community shift.		
	community structure	↑↑/↓↓	moderate shift in community structure is most likely related to enrichment		
Zooplankton Community (Field Biological Response)	abundance	(↓)	lower abundance does not indicate enrichment	0	
	biomass	(↓)	lower biomass does not indicate enrichment.		
	community structure	↑↑/↓↓	change to community structure is most likely explained as a response to phytoplankton enrichment		

Table 12-8 Weight of Evidence Integration for the Nutrient Enrichment Hypothesis

Endpoint Group	Endpoint	Maximum Response Rating	Key Supporting Evidence/Rationale	Group Rating	WOE Rank and Rationale
Benthic Invertebrate Community					
Water Quality (Exposure)	overall assessment	↑↑	see plankton community (above)	↑	**WOE Rank 1** -Enrichment is apparent from water quality. -In 2012, there was 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 appears more consistent with nutrient enrichment than with toxicological impairment. *Valvata* and *Tanytarsus* densities have increased in Snap Lake leading to a shift in community structure. The decreased *Microspsectra* density may be due to this community shift.
Sediment Quality (Exposure)	overall assessment	↑↑	in 2012 (De Beers 2013), nitrogen compounds were elevated in sediments. TOC was considered high but did not differ between Snap Lake and Northeast Lake.		
Primary Productivity (Exposure)	chlorophyll *a* and phytoplankton biomass	↑ and (↓)	phytoplankton biomass trend suggests enrichment - see plankton community (above)		
Benthic Invertebrate Community (Field Biological Response)	total density richness Simpson's Diversity Index	no response	-	↑	
	evenness	↑	difference is equivocal but could indicate enrichment		
	density of dominant taxa (trends in Snap Lake compared to Reference Lakes)	↑ and (↓) (Pisidiidae density)	variable trend is indicative of a community shift related to enrichment		
	density of dominant taxa (Snap Lake compared to Northeast Lake)	(↓) (*Micropsectra* density)	lower density in Snap Lake generally to not indicate enrichment but could be due to an enrichment-related community shift		
		↑ (*Valvata* and *Tanytarsus* density)	increased abundance is consistent with enrichment		
	community structure	↑/↓	change to community structure is most likely explained by enrichment		

Endpoint Group	Endpoint	Maximum Response Rating	Key Supporting Evidence/Rationale	Group Rating	WOE Rank and Rationale
Fish Health and Community					
Water Quality (Exposure)	overall assessment	↑↑	see plankton community (above)	↑	-Nutrient concentrations have increased in Snap Lake and appear to have caused a shift in the phytoplankton community. -However, in 2013 this enrichment did not appear to have caused an increase in zooplankton or benthic invertebrate food supply for fish. - Fish community findings do not indicate any enrichment-related response.
Primary Productivity (Exposure)	chlorophyll *a* and phytoplankton biomass	↑ and (↓)	phytoplankton biomass trend suggests enrichment - see plankton community (above)		
Food Supply (Exposure)	zooplankton abundance and biomass	(↓)	lower abundance and biomass do not indicate enrichment		
	benthic invertebrate total density	no response	-		
Small-bodied fish health (Field Biological Response)	Not included in the 2013 AEMP				
Fish Community (Field Biological Response)	Endpoints and ratings have yet to be developed. The fish community findings indicated that there were no discernable changes to the Snap Lake fish community attributable to Mine-related enrichment.				

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.

↑/↓ = Rating 1; ↑↑/↓↓ = Rating 2; TDS = total dissolved solids; TOC = total organic carbon; WOE = weight of evidence; WQ = water quality.

12.2.3.1 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 considered the main indicator of exposure for the plankton community. Nutrient enrichment in Snap Lake was indicated by increasing trends and concentrations beyond the normal range for nitrogen compounds (nitrite, nitrate, ammonia, and TKN), and TDS (including calcium which can be a nutrient for zooplankton and benthic invertebrates). The concentrations of TDS and nitrogen compounds beyond the normal range classified as Rating 2, whereas the trends classified as Rating 1 for these parameters. Snap Lake is expected to be phosphorus-limited and phosphorus compounds did not show increases in Snap Lake. Based on this consideration, the Rating 2 for nitrogen compounds and TDS was considered to overstate the potential influence of these parameters on enrichment of the plankton community. Therefore, Rating 1 was applied, overall, for this exposure endpoint group.
- ***Field Biological Responses*: ** Enrichment of the phytoplankton community appears to be occurring as indicated by the trends in phytoplankton biomass and community shift in the phytoplankton community. The likely explanation for these changes is an enrichment-caused biomass increase followed by a compensatory community shift that then reduced biomass. The community shift at the functional group level resulted in Rating 2; this rating was applied overall for phytoplankton. In contrast, the pattern of response for zooplankton (decreased biomass relative to baseline) did not appear consistent with enrichment (resulting in a Rating 0 for this endpoint group), but this does not outweigh the conclusion that enrichment appears to be occurring in the phytoplankton community. A possible explanation for the lack of an apparent enrichment response in the zooplankton community is predation pressure.

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 back to chrysophyceae-cyanobacteria. These findings are consistent with an overall WOE Rank of 2 (moderate support) for the Nutrient Enrichment Hypothesis for the plankton community.

12.2.3.2 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 nitrogen compounds, and TDS (Rating 1, 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. 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). Total organic carbon (TOC) is naturally high in Snap Lake and Northeast Lake with no apparent differences between the two. Thus, although water quality and 2012 sediment quality indicate chemical enrichment of Snap Lake, the 2013 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 clearest pattern of response in the benthic invertebrate community was that of higher densities of two dominant taxa (*Valvata* and *Tanytarsus*) in Snap Lake compared to Northeast Lake, with a concurrent decrease in *Micropsectra*. A minor community shift toward Pisiididae from Chironomidae was also apparent. These changes were each at Rating 1 and were considered consistent with a response to nutrient enrichment. Thus, for 2013, the benthic invertebrate community was also considered to be at Rating 1, 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 2013. 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. Despite the lack of direct evidence for increased food supply, the pattern of response in the benthic invertebrate community is consistent with that expected under a mild response to nutrient enrichment, and there is evidence indicating an enrichment response in the phytoplankton community. These findings were considered consistent with an overall WOE Rank of 1 (weak support) for the Nutrient Enrichment Hypothesis for the benthic invertebrate community. In general, the degree of support for nutrient enrichment provided by these endpoint results was considered stronger than that provided for toxicological impairment.

12.2.3.3 Fish Health and Fish Community Monitoring

Similar to Section 12.2.2.3, the endpoints ratings for Exposure are combined with a qualitative discussion of the 2013 fish community findings, below:

- ***Exposure:*** Although phytoplankton community enrichment has occurred in Snap Lake, the density and abundance of zooplankton and benthic invertebrates do not indicate 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 combined with a phytoplankton response, but no increased food supply for fish.
- ***Field Biological Responses*:** There were no discernable changes to the Snap Lake fish community, in terms of abundance (CPUE as indicated by the BsM), growth (weight-length-age relationships), and species composition, that were attributable to Mine-related changes in nutrient exposure or food supply.

While, the exposure endpoints indicate that nutrient concentrations have increased in Snap Lake and appear to have caused a shift in the phytoplankton community, this enrichment did not appear to have caused an increase in zooplankton or benthic invertebrate food supply for fish. The fish community monitoring findings do not indicate any enrichment-related response indicating negligible evidence for enrichment of fish health and community.

12.2.4 Summary

Both hypotheses regarding the nature of possible effects in Snap Lake were potentially supported based on the results of the 2013 AEMP.

For the Toxicological Impairment Hypothesis, the results of the WOE integration of exposure and field biological responses resulted in the following WOE rankings:

- ***Plankton Community*** – WOE Rank 1;
- ***Benthic Invertebrate Community*** – WOE Rank 1; and,
- ***Fish Health and Community*** – WOE Rank not estimated (refer to discussion in Section 12.1.1) but 2013 findings do not suggest any toxicological impairment of fish health and community.

Increased exposure to potential toxicants by plankton and benthic invertebrates in Snap Lake was indicated by: AEMP benchmark exceedances combined with increasing trends or differences from the normal range in water quality (chloride, fluoride, nitrate, and TDS); and, 2012 sediment quality results (bismuth, selenium, sodium, and strontium). Biological responses consistent with toxicological impairment were a mild decrease in zooplankton abundance and biomass combined with a species-level community shift, and a decrease in *Micropsectra* density. 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, changes in food supply, or inter-annual variability. Therefore, the classification of WOE Rank 1 for plankton and benthic invertebrates is considered conservative and likely to be a false-positive finding. The responses of the phytoplankton community were not consistent with this hypothesis and there was no response indicative of toxicological impairment in the fish community.

In summary, the conditions in Snap Lake for 2013 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.

For the Nutrient Enrichment Hypothesis, the results of the WOE integration of exposure and field biological responses resulted in the following WOE rankings:

- ***Plankton Community*** – WOE Rank 2;
- ***Benthic Invertebrate Community*** – WOE Rank 1; and,
- ***Fish Health and Community Monitoring*** – WOE Rank not estimated (refer to discussion in Section 12.1.1) but 2013 findings do not suggest any enrichment of fish health and community.

Increased exposure to potential nutrients by plankton and benthic invertebrates of Snap Lake was indicated by increasing trends or differences from the normal range in water quality (TDS and 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 zooplankton there was very little evidence of enrichment-related responses. For benthic invertebrates, biological responses consistent with nutrient enrichment were a general increase in the density of dominant taxa (*Valvata* and *Tanytarsus*) combined with a shift in community structure. There was no response indicative of enrichment in the fish community.

The AEMP findings for Snap Lake for 2013 provided moderate evidence for enrichment of the plankton community and mild evidence of enrichment in the benthic invertebrate community.

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 2013, 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. Based on these findings it can be concluded that, although there was a weak to moderate support for each hypothesis for certain ecosystem components, the evidence supports nutrient enrichment over toxicological impairment. Also, there appears to be no impairment of the structure and function of the Snap Lake ecosystem through 2013.

No Action Levels were triggered for the biological components of Snap Lake in the 2013 AEMP; this is considered appropriate given the generally weak responses in zooplankton and benthic invertebrates, somewhat stronger response in phytoplankton that was consistent with that anticipated in the environmental assessment, and lack of a Mine-related response in the fish community.

Low Action Levels were triggered for fish tissue chemistry (cesium and thallium), but direct linkage to the Mine via water quality and treated effluent is considered uncertain. However, the presence of these metals at concentrations different from the reference lakes, and in the case of cesium, different from baseline, suggest the need for further investigation.

12.3 References

Chapman PM, Anderson J. 2005. A decision-making framework for sediment contamination. Integr Environ Assess Manage 1: 163-173.

Chapman PM, Smith M. 2012. Assessing, managing and monitoring contaminated aquatic sediments. Mar Pollut Bull 64: 2000-2004.

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

De Beers. 2013. Snap Lake Mine Aquatic Effects Monitoring Program – 2012 Annual Report. Yellowknife, NWT, Canada.

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

Environment Canada and Ontario Ministry of the Environment. 2008. Canada-Ontario Decision-making Framework for Assessment of Great Lakes Contaminated Sediment. Ottawa, ON, Canada.

Environment Canada. 2012. Federal Contaminated Sites Action Plan (FCSAP) Ecological Risk Assessment Guidance. Prepared by Azimuth Consulting Group. Vancouver, BC, Canada.

McDonald BG, de Bruyn AMH, Wernick BG, Patterson L, Pellerin N, Chapman PM. 2007. Design and application of a transparent and scalable weight-of-evidence framework: An example from Wabamun Lake, Alberta, Canada. Integr Environ Assess Manage 3: 476-483.

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

SAB (Science Advisory Board for Contaminated Sites in British Columbia). 2008. Detailed Ecological Risk Assessment in British Columbia – Technical Guidance. Submitted to the Ministry of Environment, Victoria, BC, Canada.

Wenning RJ, Ingersoll CG, Batley G, Moore DW (Eds). 2005. Use of Sediment Quality Guidelines and Related Tools for the Assessment of Contaminated Sediments. SETAC Press, Pensacola, FL, USA.

SECTION 13

ACTION LEVELS

TABLE OF CONTENTS

LIST OF TABLES

LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Program
EAR	Environmental Assessment Report
CCME	Canadian Council of Ministers of the Environment
HC	Health Canada
ISQG	interim sediment quality guidelines
LR	lysine-arginine
Mine	Snap Lake Mine
P	probability
PEL	probable effect level
SSWQO	site-specific water quality objective
SD	standard deviation
SNP	Surveilliance Network Program
TBD	to be determined
TDS	total dissolved solids
TK	traditional knowledge
WOE	weight of evidence
WQG	water quality guideline

UNITS OF MEASURE

Term	Definition
>	greater than
<	less than
%	percent
±	plus or minus
µg/L	micrograms per litre

13 ACTION LEVELS

13.1 Introduction

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

The goal of the response framework is to systematically respond to monitoring results, as necessary, to identify the potential for significant adverse effects and undertake necessary mitigation actions. This is accomplished by implementing appropriate mitigation at predefined Action Levels, which are triggered before a significant adverse effect can occur. Changes from baseline data, reference lake data, or deviations from the range of natural variability are all considered in the determination of whether or not an Action Level is triggered.

The Action Levels were based on the 2013 AEMP Design Plan (De Beers 2014). This AEMP reports represents the first time the response framework was applied to the AEMP. Action Levels are discussed in detail in each section of the AEMP, and are summarized below.

13.2 Approach

The 2013 AEMP Design Plan provided tabular summaries of the proposed Significance Thresholds and Action Levels (De Beers 2014). Each table includes the following information:

- **Key Information** - Summarizes which measurement endpoints are assessed for each assessment endpoint.
- **Negligible** - The conditions under which the Low Action Level would not yet be reached.
- **Low Action Level** - The conditions under which the Low Action Level would be reached.
- **Comment/Rationale** – The rationale for the Low Action Level.

The Action Levels for the categories of Drinking Water and Fish Safe to Eat are presented in Table 13-1. The Action Levels for the category of Ecological Stability are presented in Table 13-2 for Toxicological Impairment and Table 13-3 for Nutrient Enrichment.

Table 13-1 Proposed Action Levels – Drinking Water and Fish Safe to Eat

Tiered Action Level	Drinking Water for Humans *Water Must be Drinkable*	Fish Consumption by Humans *Fish Safe to Eat*
Key Information	Drinking water parameters (metals, nutrients, and major ions) measured in AEMP samples (all stations) and SNP samples (Station SNP 02-15 only) Microcystin-LR measured in AEMP samples (all stations) and SNP samples (Station SNP 02-15 only)	Fish taste and texture (TK input) Metal concentrations in edible fish tissue
Negligible	Drinking water parameters <75% Health Canada human health and aesthetic drinking WQG **AND** Microcystin-LR <75% of Health Canada human health drinking WQG **AND** Drinking water parameters <75% CCME wildlife health WQG	Taste and texture good (TK input) **AND** Metals in edible fish tissue below 75% of upper limit of normal range[(a)]
Low	Drinking water parameters at any location are above 75% of Health Canada human health or aesthetic drinking WQG **OR** Microcystin-LR at any location is above 75% of Health Canada human health drinking WQG **OR** Drinking water parameters at any location are above 75% of CCME wildlife health WQG,	Fish taste and/or texture not acceptable (TK input) **OR** Metals in edible fish tissue above 75% of upper limit of normal range[(a)].
Medium	*TBD*[(b)]	*TBD*[(b)]
High	*TBD*[(b)]	*TBD*[(b)]

Table 13-1 Proposed Action Levels – Drinking Water and Fish Safe to Eat

Tiered Action Level	Drinking Water for Humans *Water Must be Drinkable*	Fish Consumption by Humans *Fish Safe to Eat*
Comment/Rationale	Action Levels for drinking water exclude consideration of coliforms. Health Canada recommends disinfection of all surface waters prior to consumption. Action Levels apply to any one drinking water parameter in any one sample collected from any location in Snap Lake. CCME livestock watering guidelines will be used for wildlife health. Microcystin-LR concentrations from depth-integrated AEMP samples and mid-depth samples from one SNP station (SNP 02-15, the drinking water intake for Snap Lake) will be considered. Temporal (i.e., changes over time) and spatial (e.g., proximity to the camp water intake) trends will be considered when recommending action. See bullets in Section 6.4.2 for details.	Negligible Action Level of “fish taste and texture is good” is based on a satisfactory outcome from the annual fish tasting program The Low Action Level of “fish taste and/or texture is not acceptable” is based on any one fish receiving a ‘not good/unacceptable rating’ from any one participant of the fish tasting program The Low Action Level of “metals in edible fish tissue” is based on the mean concentration for any metal in Snap Lake fish tissue observed above 75% of the upper limit of normal range See bullets in Section 6.4.2 for details.

a) Normal Range for fish endpoints is defined as the 95% prediction interval for the mean (see Appendix 9A).

b) TBD = to be determined if Low Action Level is reached.

AEMP = Aquatic Effects Monitoring Program; SNP = Surveillance Network Program; < = less than; % = percent; TK = Traditional Knowledge; CCME = Canadian Council Ministers of the Environment; WQG = water quality guideline; TBD = to be determined; LR = lysine-arginine.

Table 13-2 Proposed Action Levels - Toxicological Impairment

Tiered Action Level	Water Quality (substances of potential toxicological concern and measured toxicity) *Ecological Integrity Maintained*		Sediment Quality *Ecological Integrity Maintained*
Key Information	Differences between Snap Lake and reference lakes or normal range AEMP Benchmarks	Toxicity results for edge of mixing zone	Differences between Snap Lake and reference lakes or normal range CCME ISQGs
Negligible	Concentration not exceeding AEMP Benchmarks[(a)] where they exist, or if exceeding, not due to Mine **AND** Within normal range lake-wide	No persistent sublethal toxic effects to test organisms in mixing zone samples	Not exceeding CCME ISQG or, if exceeding, not due to the Mine **AND** Within normal range lake-wide
Low	Concentration greater than normal and reference range lake-wide supported by a temporal trend **AND** Exceeding 75% of AEMP Benchmark[(a)] at the edge of the mixing zone (i.e., diffuser station)	Persistent sublethal toxic effects to test organisms in mixing zone samples **OR** Sublethal toxic effects for Fish Early Life Stage test in mixing zone samples	Exceeding 75% of ISQG in Snap Lake as a result of Mine operation **AND** Greater than normal range
Medium	*TBD*[(b)]	*TBD*[(b)]	*TBD*[(b)]
High	*TBD*[(b)]	*TBD*[(b)]	*TBD*[(b)]

Table 13-2 Proposed Action Levels - Toxicological Impairment

Tiered Action Level	Water Quality (substances of potential toxicological concern and measured toxicity) *Ecological Integrity Maintained*	Sediment Quality *Ecological Integrity Maintained*
Comment/Rationale	AEMP Benchmarks refers to benchmarks currently used in the AEMP to which substance concentrations are compared (i.e., EAR benchmarks and CCME guidelines). Exceeding 75% of AEMP Benchmark at the edge of the mixing zone (i.e., diffuser station) = the average concentration from the three diffuser stations (i.e., SNP 02-20d, e, f) in any one sampling event is >75% of the AEMP Benchmark. Lake-wide refers to all locations in the Main Basin. Temporal (i.e., changes over time) and spatial (i.e., proximity to diffuser) trends will be considered when recommending action. Persistent sublethal toxicity is defined as two concurrent or two consecutive sublethal test results (i.e., sublethal toxic effects). Sublethal toxic effects are defined as IC25 less than highest test concentrations (i.e., <100% for *C. dubia* and <97% for *P. subcapitata.* Fish Early Life Stage test indicates results from the 30-day test. See bullets in Section 6.4.3 for details.	ISQG is highly protective so is an appropriate trigger value. This will be triggered based on comparison of mean concentration from main basin stations. See bullets in Section 6.4.3 for details.

Table 13-2 Proposed Action Levels - Toxicological Impairment

Tiered Action Level	Plankton Community *Ecological Integrity Maintained*	Benthic Community *Ecological Integrity Maintained*	Fish Health *Ecological Integrity Maintained*	Fish Community *Ecological Integrity Maintained*
Key Information	Differences between Snap Lake and reference lakes or normal range	Differences between Snap Lake main basin and reference lakes or normal range; trends over time in Snap Lake main basin and reference lakes	Differences between Snap Lake and reference lakes or normal range	Differences between Snap Lake and reference lakes or normal range
Negligible	No persistent decline beyond the normal range in total phytoplankton biomass or cladoceran abundance and biomass	No statistically significant changes (P>0.1) in Snap Lake main basin extending below the normal range for richness and densities of dominant taxa **AND** No divergence of trends in richness and densities of dominant taxa in Snap Lake main basin compared to reference lakes	No changes in fish health endpoints [(c)] or fish tissue chemistry in Snap Lake beyond the normal range **AND** Changes are of magnitude[(d)] that would not indicate an impairment to fish health	No indication from catch rates of a change[(e)] in number of fish of any species from Snap Lake
Low	A persistent decline beyond the normal range in total phytoplankton biomass within the main basin of Snap Lake **OR** A persistent decline beyond the normal range in cladoceran abundance **or** biomass within the main basin of Snap Lake	Statistically significant changes (P<0.1) in Snap Lake main basin extending below the normal range for richness **OR** Statistically significant changes(P<0.1) in Snap Lake main basin extending below the normal range for densities of dominant taxa **OR** Downward trend in richness and densities of dominant taxa in Snap Lake main basin, but not in reference lakes	Statistically significant difference (P<0.1) in fish health endpoints[(c)] or fish tissue chemistry that is beyond normal range **AND** Change is in direction, and of magnitude[(d)], that is indicative of an impairment to fish health	Indication from catch rates of a change[(e)] in number of fish of a species from Snap Lake
Medium	*TBD*[(b)]	*TBD*[(b)]	*TBD*[(b,f)]	
High	*TBD*[(b)]	*TBD*[(b)]	*TBD*[(b,f)]	

Table 13-2 Proposed Action Levels - Toxicological Impairment

Tiered Action Level	Plankton Community *Ecological Integrity Maintained*	Benthic Community *Ecological Integrity Maintained*	Fish Health *Ecological Integrity Maintained*	Fish Community *Ecological Integrity Maintained*
Comment/Rationale	Plankton communities are inherently variable therefore persistent trends need to be observed before action is taken. Persistent is defined as a sustained increase or decrease equal to or greater than three years. The normal range is defined as the background data (2004) mean ± 2 SDs. See bullets in Section 6.4.3 for details.	Toxicity generally causes a downward trend in richness and density of benthic invertebrates. The normal range is defined as ± 2 SD of the mean of reference stations and unaffected stations (identified based on conductivity as and effluent tracer) in Snap Lake during the early years of the mine. Dominant taxa are defined as those accounting for more than 5% of the total invertebrates across all stations. See bullets in Section 6.4.3 for details.	See bullets in Section 6.4.3 for details.	

Note: “Normal Range” is currently determined based on ± 2SD in Snap Lake Main Basin baseline and ±2SD in reference lakes, and/or other appropriate considerations.

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

b) TBD – to be determined if Low Action Level is reached.

c) Key fish health endpoints are: condition, relative gonad size, and relative liver size. They will be assessed between Snap Lake and the reference lakes.

d) Definition of a magnitude of change that is indicative of impairment to fish health is based on the critical effect sizes defined by Environment Canada’s Metal Mining Effluent Regulations Guidance Document (Environment Canada 2012) and refers to an increase or a decrease in fish health endpoints.

e) Definition of “change” to be developed, but anticipates comparison of relative abundance (i.e., catch per unit effort) between lakes.

f) It is anticipated that fish health and fish community would be combined at the Medium and High Action Levels.

EAR = Environmental Assessment Report; AEMP = Aquatic Effects Monitoring Program; CCME = Canadian Council of Ministers of the Environment; ISQG = interim sediment quality guideline; P = probability; PEL = Probable Effect Level; Mine = Snap Lake Mine; > = greater than; % = percent; ± = plus or minus; SD = standard deviation; TBD = to be determined.

Table 13-3 Proposed Action Levels – Nutrient Enrichment

Tiered Action Level	Water Quality (Nutrients) *Ecosystem Function*	Plankton Community *Ecosystem Function*
Key Information	Differences between Snap Lake and reference lakes or normal range AEMP Benchmarks and site-specific benchmarks	Differences between Snap Lake and reference lakes or normal range
Negligible	Consistent with EAR prediction (De Beers 2002) **AND** If AEMP Benchmark exists, below the benchmark	No consistent ecologically-important changes in richness and community structure
Low	Exceeding EAR Predictions supported by temporal trend **AND** Exceeding >75% AEMP Benchmark, if it exists	Persistent increase beyond the normal range in total phytoplankton or zooplankton biomass in the main basin of Snap Lake **AND** Minor shift in phytoplankton or zooplankton community composition (based on major[b] groups) in the main basin of Snap Lake
Medium	*TBD*[a]	*TBD*[a]
High	*TBD*[b]	*TBD*[a]
Comment/Rationale	Whole-lake average concentrations (main basin only) will be compared against maximum whole-lake average concentrations predicted in the EAR and updated predictions. Comparisons to new predictions will be made; however, the comparisons to the EAR predictions will be prioritized . AEMP Benchmark for total phosphorus = Mesotrophic status defined by phosphorus levels of 10.9 -95.6 µg/L (Wetzel 2001). The low action level refers to > 75% of the low end of this range (i.e., 10.9 µg/L) (see text).	Plankton communities are inherently variable therefore persistent trends need to be observed before action is taken. Persistent is defined as a sustained increase or decrease equal to or greater than three years. The normal range is defined as background data (2004) mean ± 2 SDs. See bullets in Section 6.4.3 for details.

Table 13-3 Proposed Action Levels – Nutrient Enrichment

Tiered Action Level	Benthic Community *Ecological Integrity Maintained*	Fish Health *Ecological Integrity Maintained*	Fish Community *Ecological Integrity Maintained*
Key Information	Differences between Snap Lake main basin and reference lakes or normal range; trends over time in Snap Lake main basin and reference lakes	Differences between Snap Lake and reference lakes or normal range	Differences between Snap Lake and reference lakes or normal range
Negligible	No statistically-significant changes (P>0.1) in Snap Lake main basin extending beyond the normal range for richness and densities of dominant taxa **AND** No divergence of trends in richness and densities of dominant taxa in Snap Lake compared to reference lakes	No changes in fish health endpoints or fish tissue chemistry in Snap Lake beyond the normal range **AND** Changes are of magnitude[c] that would not indicate an impairment to fish health	No indication from catch rates of a change[d] in number of fish of any species from Snap Lake
Low	Statistically significant changes(P<0.1) in Snap Lake main basin extending beyond the normal range for richness **OR** Statistically-significant changes (P<0.1) in Snap Lake main basin extending beyond the normal range for densities of dominant taxa **OR** Upward trend in richness and densities of dominant taxa in Snap Lake, but not reference lakes	Statistically significant difference (P<0.1) in fish health endpoints or fish tissue chemistry that is beyond normal range **AND** Change is in direction, and of magnitude[c], that is indicative of an impairment to fish health	Indication from catch rates of a change[d] in number of fish of a species from Snap Lake
Medium	*TBD*[a]	*TBD*[a,e]	
High	*TBD*[a]	*TBD*[a,e]	

Table 13-3 Proposed Action Levels – Nutrient Enrichment

Tiered Action Level	Benthic Community *Ecological Integrity Maintained*	Fish Health *Ecological Integrity Maintained*	Fish Community *Ecological Integrity Maintained*
Comment/Rationale	Mild nutrient enrichment generally causes an upward trend in richness and density of benthic invertebrates. The normal range is defined as ± 2 SD of reference stations and unaffected stations (identified based on conductivity as and effluent tracer) in Snap Lake during the early years of the mine. See bullets in Section 6.4.3 for details	Tissue chemistry parameters which are relevant to the nutrient enrichment criteria are sodium, potassium and phosphorus (as listed in Section 4,Table 4.8-1).	

Note: "Normal Range" is determined based on ±2SD in Snap Lake Main Basin baseline and ± 2SD in reference lakes, and/or other appropriate considerations.

a) TBD = to be determined if Low Action Level is reached.

b) "Major" indicates a change at the Class level of biological organization for phytoplankton and a combination of Phylum and Order levels for zooplankton.

b) Key fish health endpoints are: condition, relative gonad size, and relative liver size. They will be assessed between Snap Lake and the reference lakes.

c) Definition of a magnitude of change that is indicative of impairment to fish health is based on the critical effect sizes defined by Environment Canada's Metal Mining Effluent Regulations Guidance Document (Environment Canada 2012) and refers to an increase or a decrease in fish health endpoints.

d) Definition of "change" to be developed, but anticipates comparison of relative abundance (i.e., catch per unit effort) between lakes.

e) It is anticipated that fish health and fish community would be combined at the Medium and High Action Levels.

De Beers. 2002. Snap Lake Diamond Project: Environmental Assessment Report. Submitted to the Mackenzie Valley Environmental Impact Review Board.

AEMP = Aquatic Effects Monitoring Program; EAR = Environmental Assessment Report; SD = standard deviation; % = percent; TBD = to be determined; > = greater than; µg/L = micrograms per litre; P = probability.

13.3 Suggested Responses

Table 13-4 provides a summary of suggested responses to be taken (Actions) when an Action Level is reached. For any Action Level, the following AEMP “Best Practices” will be followed each year when interpreting the AEMP findings:

- assess cause/linkage to Mine;
- examine trends;
- predict trends and predict time to reach a potential next Action Level, where appropriate;
- examine weight of evidence (WOE) assessment for strength of linkage between exposure, toxicity, and field biological responses;
- examine ecological significance; and,
- confirm that existing benchmarks are appropriate, and revise if warranted.

Additional responses detailed in the Response Plan will depend on the component affected (e.g., water quality, plankton community), the likely cause of the effect as determined in the WOE assessment (i.e., toxicological impairment versus nutrient enrichment), and the type and magnitude of effect.

Table 13-4 Suggested Types of Actions to be Taken if an Action Level is Exceeded

Action Level	Suggested Types of Actions
Negligible	**Response Actions that would be taken:** • AEMP best practices
Low	**Response Actions that would be taken:** • AEMP best practices • Confirm Low Action level • Set Medium and High Action Levels • Develop Response Plan **Potential additional Response Actions:** • Revise Low Action Level, if warranted and scientifically defensible • Set site-specific benchmarks, if appropriate • If trending towards Medium, identify potential mitigation options • Increase monitoring frequency for plankton, benthos, and/or fish to confirm findings • Desk-top or field special study to examine ecological significance, causation, and/or linkage to Mine

Table 13-4 Suggested Types of Actions to be Taken if an Action Level is Exceeded

Action Level	Suggested Types of Actions
Medium	**Response Actions that would be taken:** • AEMP Best practices • Develop Response Plan • Confirm Medium Action Level • If Medium Action Level confirmed, implement mitigation(s) to stop or slow trend **Potential additional Response Actions:** • Desk-top or field special study(ies) to examine ecological significance, causation, and/or linkage to Mine • Maintain increased monitoring frequency for plankton, benthos, and/or fish to confirm that mitigation is working • Refine Medium and High Action Levels if warranted and scientifically defensible
High	**Response Actions that would be taken:** • AEMP Best practices • Confirm High Action level • Develop Response Plan • If High Action Level confirmed, implement appropriate mitigations on a priority basis to reverse trend **Potential additional Response Actions:** • Special study(ies) to examine effectiveness of mitigation, and long-term monitoring of mitigation effectiveness • Special study(ies) to examine ecological significance and reversibility, causation, and/or linkage to Mine

AEMP (Aquatic Effects Monitoring Program) Best Practices: evaluate causation/linkage to Mine; examine trends; predict trends where appropriate; examine WOE assessment linkage between exposure, toxicity, and field biological responses; examine ecological significance; confirm that existing benchmarks are appropriate and revise if warranted.

13.4 Action Level Assessment

13.4.1 Water Safe to Drink and Fish Safe to Eat

No Action Levels related to Water Safe to Drink in Snap Lake were triggered in 2013.

Low Action Levels related to Fish Safe to Eat were triggered for two parameters: cesium and thallium:

- cesium concentration in muscle tissue of Round Whitefish in Snap Lake was above the normal range; and,
- thallium concentrations in muscle tissue of Round Whitefish and Lake Trout in Snap Lake were above the normal range.

13.4.2 Ecological Function – Toxicological Impairment

13.4.2.1 Water Quality

Low Action Levels related to toxicological impairment that could affect ecological function in Snap Lake were triggered for three parameters for water quality: chloride, fluoride, and nitrate:

- Maximum monthly concentrations were above 75 percent (%) of generic AEMP benchmarks (see Section 3.4.3).
- Concentrations were increasing over time in Snap Lake (see Section 3.4.4).
- Concentrations were greater in Snap Lake relative to both reference lakes (see Section 3.4.5).

Results of toxicity testing did not trigger action levels because toxicity testing did not show any toxic effects to test organisms in the mixing zone samples (see Section 3.4.3).

A Total Dissolved Solids (TDS) Response Plan, which included TDS, chloride, and fluoride, and a Nitrogen Response Plan, which included ammonia and nitrate, were submitted in December 2013 as part of the Water Licence Amendment (De Beers 2013a and b, respectively). A TDS Response Plan and Nitrogen Response Plan are requirements of the Water Licence (MV2011L2-0004). Total dissolved solids is predicted to be exceeded between January 2014 and January 2015 (De Beers 2013c). Ammonia did not trigger an Action Level in 2013 but was included in the Nitrogen Response Plan as a requirement of the Water Licence.

The TDS Response Plan as required under the Water Licence MV2011L2-0004, describes the tasks that De Beers has completed and is in the process of completing in response to increasing TDS, chloride, and fluoride concentrations in Snap Lake:

- determine sources of TDS, chloride, and fluoride loadings to Snap Lake;
- provide current and ongoing management practices to reduce TDS, chloride, and fluoride loadings to Snap Lake;
- recommend TDS, chloride, and fluoride site-specific water quality objectives (SSWQOs) in Snap Lake that are protective of aquatic life and consider factors that make the salts less toxic;
- propose concentrations of TDS, chloride, and fluoride that are not to be exceeded in the discharge to Snap Lake (i.e., effluent quality criteria applied at the last point of discharge);
- update modelling predictions; and,
- provide a water management strategy for the life of the Mine.

The Nitrogen Response Plan describes the tasks that De Beers has completed and is in the process of completing in response to increasing nitrate and ammonia concentrations in Snap Lake:

- determine sources of nitrate and ammonia loadings to Snap Lake including reviewing and improving explosives management practices;
- provide current and ongoing management practices to reduce nitrate and ammonia loadings to Snap Lake;
- recommend a SSWQO for nitrate and a water quality guideline (WQG) for ammonia in Snap Lake protective of aquatic life and consider exposure and toxicity modifying factors;
- propose concentrations of nitrate and ammonia that are not to be exceeded in the discharge to Snap Lake (i.e., effluent quality criteria applied at the last point of discharge);
- update modelling predictions; and
- discuss options to reduce nitrogen loadings in the discharge to Snap Lake.

13.4.2.2 Fish Health and Tissue Chemistry

A fish health survey was not completed in 2013; the next fish health survey will be performed in 2015.

Low Action Levels related to toxicological impairment that could affect ecological function in Snap Lake were triggered for two parameters for fish tissue chemistry: cesium and thallium:

Cesium

- Concentrations in Snap Lake were statistically different from baseline in Round Whitefish muscle tissue (see Section 9.4.1).

- Concentrations in Snap Lake were statistically different from reference lake concentrations in Round Whitefish liver, kidney, and muscle tissue (see Section 9.4.2).
- Concentrations in Snap Lake were above the normal range in Round Whitefish liver, kidney, and muscle tissue (see Section 9.4.3).

Thallium

- Concentrations in Snap Lake were elevated relative to baseline in both Lake Trout and Round Whitefish muscle tissue; however, statistical comparisons could not be completed as baseline concentrations were all below the detection limit (see Section 9.4.1).
- Concentrations in Snap Lake were statistically different from reference lake concentrations in Lake Trout and Round Whitefish kidney and muscle tissue (see Section 9.4.2).
- Concentrations in Snap Lake were above the normal range in Lake Trout and Round Whitefish kidney and muscle tissue (see Section 9.4.3).

Response plans for cesium and thallium will be submitted in 2014.

13.4.3 Ecological Function – Nutrient Enrichment

No Action Levels related to Nutrient Enrichment that could affect ecological function in Snap Lake were triggered in 2013.

13.5 Conclusions

Low Action Levels related to “Fish Safe to Eat” in Snap Lake were triggered for two fish tissue parameters: cesium and thallium. A Low Action Level for “Fish Safe to Eat” is the level for which risk assessment activities are initiated. A response plan for cesium and thallium will be submitted in 2014.

Low Action Levels related to toxicological impairment in Snap Lake were triggered for three water quality parameters: chloride, fluoride, and nitrate, and two fish tissue chemistry parameters: cesium and thallium. Response plans for chloride, nitrate, and fluoride were submitted as required by the Water Licence (De Beers 2013a, b).

No Action Levels related to Water Safe to Drink or Nutrient Enrichment were triggered.

13.6 References

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

De Beers. 2013a. Total Dissolved Solids Response Plan Report. Yellowknife, NWT, Canada.

De Beers. 2013b. Nitrogen Response Plan Report. Yellowknife, NWT, Canada.

De Beers. 2013c. Snap Lake Hydrodynamic and Water Quality Model Report. Submitted to the Mackenzie Valley Land and Water Board. Yellowknife, NWT, Canada.

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

Wetzl RG. 2001. Limnology 3rd edition. Elsevier Science Academic Press, New York, NY, USE.

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

SECTION 14

RECOMMENDATIONS

TABLE OF CONTENTS

LIST OF ACRONYMS

Term	Definition
AEMP	Aquatic Effects Monitoring Program
BsM	Broad-scale Monitoring
DO	dissolved oxygen
Mine	Snap Lake Mine
TDS	total dissolved solids

14 RECOMMENDATIONS

Where available, each section of the 2013 Aquatic Effects Monitoring Program (AEMP) provided recommendations for consideration. These recommendations are detailed below.

Section 2 - Site Characterization and Supporting Environmental Variables

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

Section 3 - Water Quality

- Implement the recommendations from the Quality Assurance/ Quality Control assessment (outlined in Appendix 3A), which focuses on investigating potential contamination and variability between samples. These recommendations include reducing variability between field dissolved oxygen (DO) and Winkler titration DO, and continuing to consider options for minimizing holding time issues for parameters with sensitive holding times (e.g. freezing samples), and discussing analytical procedures with the laboratories, particularly for antimony, to determine potential sources and/or interferences that may be contributing to measured blank concentrations.
- Continue to periodically investigate the accuracy and precision of analyzing total phosphorus by the analytical laboratories currently used in the AEMP program and the potential for streamlining the collection of nutrient data by the water quality and plankton components (outlined in Appendix 3B). A limited number of nutrient spike samples should routinely be sent to the primary laboratories used for nutrient analyses in the AEMP as an on-going and independent check of the accuracy of nutrient results. Recommendations are completing one season of split sampling at plankton stations, and sending split samples to the two primary laboratories that provide nutrient analyses for the water quality and plankton sections. The split samples are intended to provide the plankton component with sufficient overlapping data to merge historical plankton nutrient data analyzed by University of Alberta Biogeochemical Analytical Service Laboratory with future plankton data recommended to be analyzed by ALS Canada Ltd. The splits samples will also be used to confirm whether the lack of differences between mid-depth and depth-integrated samples for nutrients is applicable in Northeast Lake.
- Identify the potential cause(s) of high turbidity at SNP 02-15 by assessing whether sampling procedures or the location or condition of the water intake structure may be introducing turbidity in samples collect at SNP 02-15. Based on those findings, review whether data from SNP 02-15 are appropriate to determine whether water in Snap Lake is safe to drink.
- Give consideration to parameters with concentrations that have increased beyond the normal range in Snap Lake, but for which there are either no relevant AEMP benchmarks (i.e., barium, lithium, rubidium), or the recommended site-specific benchmark has not yet been accepted (i.e., strontium). It is recommended that available toxicological literature be reviewed to determine the implications of increases in total barium, lithium, and rubidium on aquatic life.

- Continue to make necessary adjustments to loadings and predictions for total dissolved solids (TDS) and other treated effluent-related parameters. The re-evaluation of the predicted loadings and consequences to the water quality in Snap Lake are being conducted because the concentrations of TDS and other treated effluent-related parameters are directly correlated to increased loadings.

Section 4 - Sediment Quality

- Continue to use Northeast Lake and Lake 13 as reference lakes to assess long-term regional trends, but exclude the anomalous LK13-03 station from calculation of mean parameter concentrations for Lake 13.

Section 6 - Benthic Invertebrate Community

- Lake 13 should only be used for comparisons of trends over time with the main basin of Snap Lake. Differences in the benthic invertebrate community in Lake 13 compared to both Northeast Lake and the main basin of Snap Lake render it unsuitable for direct comparisons to the main basin of Snap Lake.
- Lake 13 data should be excluded from the calculation of the normal range for comparison to the main basin of Snap lake, because its inclusion would increase the upper limit of the normal range, reducing the potential to detect an enrichment effect in Snap Lake. Northeast Lake data from fall 2009 onward should continue to be used for estimating the normal range.
- Effects on the main basin of Snap Lake should be evaluated by comparing Northeast Lake to Snap Lake and evaluating trends over time in reference lakes to those in the main basin of Snap Lake.
- Composite samples, consisting of six individually sieved Ekman grabs combined into a single sample, should be collected at all stations beginning with the next benthic sampling program. Previous data indicate that six replicates at a station are sufficient to capture within station variability.
- Station SNAP07 should be excluded from the calculation of summary statistics for benthic invertebrate variables and statistical comparisons between Northeast Lake and the main basin of Snap Lake. SNAP07 is located near-shore in the northeast arm compared to other stations in the main basin of Snap Lake, which are in the open-water. Also, it is at the shallow end of the depth range required for benthic invertebrate stations. These two factors may have contributed to the different benthic invertebrate community observed at this station in 2013.

Section 8 - Fish Community Monitoring

- Given the limitations of the Broad-scale Monitoring (BsM) method in capturing species such as Burbot, Arctic Grayling, Slimy Sculpin, and Ninespine Stickleback in the study lakes based on 2013 results, consideration should be given to the addition of alternative methods that, in conjunction with the BsM, would provide a more effective means of indexing population metrics of these species, while being cognizant of the need to control incidental mortality.
- Slimy Sculpin were challenging to capture in Snap Lake in previous years; attempts will be made in 2016 to sample with backpack electrofishing in areas near inlet streams.

Section 9 - Fish Tissue Chemistry

- As per the 2013 AEMP Design Plan, the next fish tissue chemistry study is the small-bodied Lake Chub survey in 2015. The next large-bodied fish tissue chemistry survey is scheduled to occur in 2016. An additional fish program is scheduled in 2014 in three lakes downstream of Snap Lake, which will include a fish tissue chemistry component. The frequencies of future monitoring are considered appropriate to capture early warning signs of any changes occurring in Snap Lake fish tissue chemistry, while balancing the need to minimize mortality to the fish populations in the study lakes.
- Future fish programs will collect bone and archive the samples for analysis of strontium.

Section 11.1 - Littoral Zone Special Study

- Based on the 2013 sampling program, it is recommended that Hester-Dendy artificial substrate samplers should be used to quantitatively sample littoral invertebrates, and the sweep-net sampling method should be discontinued.

Section 11.2 – Picoplankton Special Study

- Based on the results to date, no changes are required for the picoplankton program. The inclusion of this special study will be re-assessed during the next AEMP re-evaluation in 2016.

Section 11.3 - Downstream Lakes Special Study

- Measuring the main point source inflows to and outflows from DSL1, DSL2, and Lac Capot Blanc when water quality samples are collected to determine whether the water balances developed for the downstream lakes are representative of conditions.
- Recording ice thickness routinely in DSL1, DSL2, and Lac Capot Blanc, if conditions allow. Ice formation and melting dates and ice thickness drive salt rejection and freshwater replacement in the downstream lakes models, which in turn affects mixing and overall concentrations.
- Documenting fish habitat characteristics in the streams connecting the downstream lakes.
- As part of the AEMP re-evaluation and study design update in 2016, the scope of monitoring as part of the AEMP will be assessed. At that time, it will be determined whether monitoring in the Downstream Lakes will remain as a special study or be incorporated into the core AEMP program. Specific sampling locations and procedures will be provided based on updated modelling results, as well as information collected during these first few years of special study investigation.

Section 11.5 - Stable Isotope Food Web Analysis Special Study

- The Snap Lake food web has been demonstrated to be predominantly benthically driven. No additional studies are recommended at this time.

SECTION 15

CLOSURE

TABLE OF CONTENTS

15 CLOSURE

This report was prepared by the undersigned, and reviewed by Alex Hood (Environmental Permitting Superintendent) and Michelle Peters (Environmental Monitoring Superintendent), De Beers Snap Lake Mine. Golder and De Beers would like to thank site staff including Gail Seto and Erin Rowlands (Senior Environmental Technician Supervisors), Tom Bradbury (Permitting Coordinator) Matt Bayly, Melissa Leclair, Andrea Hrynkiw, Guylaine Gueguen, and Andre Boulanger (Environmental Technicians) for field logistical support, coordinating the fish tasting activities, and providing site information. Additional Golder staff contributed to this report; we gratefully acknowledge the following contributors: Robin Bourke and Chris Madland (Site Characterization and Environmental Variables); Giovanna Diaz, Jonathon Love, and Alison Snow (Water Quality); Danny Lee (Sediment Quality); Dave Hasek (Plankton); Clayton James, Sima Usvyatsov, and Larry Hildebrand (Fish Community Monitoring); Tim Barrett (Fish Tissue Chemistry); Shevelle Hutt (Littoral Zone Special Study); Jill LaPorte (Picoplankton Special Study); Paul Vecsei (Lake Trout Population Special Study); Amy Ofukany (Stable Isotopes Special Study); James Dwyer (Weight of Evidence); Joel Farah, Carmen Walker, and Vanessa Vallis (Drafting and GIS); Elodie Taniere, Ada Ma, and Mark Jaferllari (Report Production); and Karin Lintner, Howard Peters, and Claire Fossey (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.

GOLDER ASSOCIATES LTD.

Report prepared by:

Orignal Signed
Andre Bachteram, M.Sc.
Aquatic Biologist

Report reviewed by:

Original Signed
Peter Chapman, Ph.D.
Principal, Senior Environmental Scientist

Orignal Signed
Dan Coulton, Ph.D.
Wildlife Biologist

Orignal Signed
Tamara Darwish, M.Sc.
Fisheries Biologist

Orignal Signed
John Fitzsimons, B.Sc.
Senior Ecologist

Orignal Signed
Kelly Hille, M.Sc.
Aquatic Biologist

Orignal Signed
Tatiana Leclerc, B.Sc.
Environmental Scientist

Orignal Signed
Alison Humphries, M.Sc.
Water Quality Specialist

Original Signed
Tasha Hall, B.Sc.
Associate, Senior Water Quality Specialist

Original Signed
Zsolt Kovats, M.Sc.
Associate, Senior Aquatic Ecologist

Original Signed
Katherine Harris, M.Sc.
Senior Aquatic Ecologist

Original Signed
Hilary Machtans, M.Sc.
Senior Fisheries Biologist

Original Signed
Kristine Mason, M.Sc.
Associate, Senior Fisheries Biologist

Original Signed
Lasha Young, M.Sc.F.
Associate, Project Manager

Orignal Signed
Rie MacIntosh, B.Sc.
Water Quality Specialist

Orignal Signed
Cathy McPherson, B.Sc.
Senior Environmental Scientist

Orignal Signed
Dale Robertson, BNR.Sc.
Fisheries Biologist

Orignal Signed
Rainie Sharpe, Ph.D.
Fisheries Biologist, Ecotoxicologist

Orignal Signed
Ryan Stevenson, M.R.M.
Senior Environmental Scientist