APPENDIX E.9

AQUATIC EFFECTS MONITORING REPORTS

APPENDIX E.9.1

2018 CREMP MONITORING REPORT

(PART 1 OF 3)

minnow environmental inc.

A Trinity Consultants Company

Mary River Project 2018 Core Receiving Environment Monitoring Program Report

Part 1 of 3 (Sections 1 to 3)

Prepared for:
Baffinland Iron Mines Corporation
Oakville, Ontario

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March 2019

Mary River Project 2018
Core Receiving Environment
Monitoring Program Report

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EXECUTIVE SUMMARY

The Mary River Project is an operating high-grade iron mine located in the Qikiqtani Region of northern Baffin Island, Nunavut. Owned and operated by Baffinland Iron Mines Corporation (Baffinland), the mine began commercial operation in 2015. Mining activities at the Mary River Project include open pit ore extraction, ore haulage, stockpiling, crushing, and screening, followed by transport by truck to Milne Port for subsequent seasonal loading onto bulk carrier ships for transfer to international markets. No milling or additional processing of the ore is conducted on-site and therefore no tailings are produced at the Mary River Project. Mine waste management facilities at the Mary River Project thus consist simply of a mine rock dump and surface runoff collection/sedimentation ponds currently situated near the mine rock dump and ore stockpile areas. In addition to periodic discharge of treated effluent from the mine waste rock disposal area, other potential mine inputs to aquatic systems located adjacent to the mine include runoff and dust from ore (crusher) stockpiles, discharge of treated sewage effluent, runoff and explosives residue from quarry operations, deposition of fugitive dust generated by mine activities, and general mine site runoff.

Under the terms and conditions of the Project's Type 'A' Water Licence issued by the Nunavut Water Board, Baffinland was required to develop and implement an Aquatic Effects Monitoring Plan (AEMP) at the Mary River Project. In order to meet the AEMP objectives for the Mary River Project, Baffinland developed a Core Receiving Environment Monitoring Program (CREMP) to provide a basis for the evaluation of mine-related influences on water quality, sediment quality, and/or aquatic biota (including phytoplankton, benthic invertebrates, and fish). The primary receiving systems that serve as the focus for the CREMP include the Camp Lake system (i.e., Camp Lake tributaries 1 and 2, Camp Lake), the Sheardown Lake system (i.e., Sheardown Lake tributaries 1, 9 and 12; Sheardown Lake NW and Sheardown Lake SE), and the Mary River and Mary Lake system. The CREMP has implemented an effects-based approach using standard environmental effects monitoring techniques as the basis for the evaluation of potential mine-related effects within the mine primary receiving systems on an annual frequency since the commencement of commercial mine production/operation in 2015.

The results of the 2018 CREMP indicated some mine-related influences on water and sediment quality of a few of the mine primary receiver systems, but no ecologically significant, adverse, mine-related effects to biota were identified based on comparisons to applicable reference conditions or baseline data. Within the Camp Lake system, mine-related effects on water quality were apparent as elevated concentrations of copper at the CLT1 north branch only, chloride, manganese, molybdenum, nitrate, potassium, sodium, sulphate, and uranium at the CLT1 main stem, and chloride, manganese, molybdenum, sodium, sulphate, and uranium at Camp Lake,

based on comparisons to reference conditions and/or to baseline data. Arsenic and manganese concentrations were elevated within littoral sediment of Camp Lake compared to reference lake sediments and to Camp Lake baseline data. Active quarrying (QMR2 Quarry) in the watershed was a possible source of these parameters to waterbodies of the Camp Lake system. Nevertheless, no adverse effects to phytoplankton, benthic invertebrates, or arctic charr (*Salvelinus alpinus*) were indicated at mine-exposed areas of the Camp Lake system in 2018, which was consistent with concentrations of most metals being below the applicable water and sediment quality guidelines (WQG and SQG, respectively) at these waterbodies.

Within the Sheardown Lake system, mine-related effects on water quality were apparent only at Sheardown Lake Tributary 1 (SDLT1) and both basins of Sheardown Lake. At SDLT1, aqueous concentrations of manganese, nickel, nitrate, sodium, strontium, sulphate, total dissolved solids, and uranium were elevated compared to concentrations at reference areas and during applicable baseline studies, but only copper concentrations were above WQG in 2018. At Sheardown Lake NW, aqueous concentrations of chloride, manganese, molybdenum, sulphate, and uranium were elevated compared to Reference Lake 3 in 2018 and/or to baseline data, whereas at Sheardown Lake SE, copper and molybdenum concentrations were elevated compared to reference conditions in 2018 and/or to baseline data. However, no parameters were elevated above WQG at either basin of Sheardown Lake in 2018. Metal concentrations in sediment at littoral and profundal habitats of the Sheardown Lake basins were very similar to concentrations observed for the same habitat types at Reference Lake 3 in 2018, suggesting no marked mine-related influences on sediment metal concentrations. No ecologically significant and/or adverse effects to phytoplankton, benthic invertebrates, or arctic charr were indicated at mine-exposed areas of Sheardown Lake Tributaries 1, 9, and 12, Sheardown Lake NW, or Sheardown Lake SE in 2018, which was consistent with concentrations of most metals being below the applicable WQG and SQG at these waterbodies.

Within the Mary River/Mary Lake system, mine-related effects on water quality were apparent only as slightly elevated concentrations of manganese and sulphate at mine-exposed areas of Mary River. Similarly, mine-related effects on sediment quality only included slight elevation of manganese concentrations at littoral habitat of Mary Lake. However, no adverse effects to phytoplankton, benthic invertebrates, or arctic charr were indicated at mine-exposed areas of Mary River and/or Mary Lake in 2018 which, similar to the other mine receiving systems, was consistent with concentrations of most metals being below the applicable WQG and SQG.

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ACRONYMS AND ABBREVIATIONS

AEMP – Aquatic Effects Monitoring Plan

ANCOVA – Analysis of Covariance

ANOVA – Analysis of Variance

CES – Critical Effect Size

CPUE – Catch Per Unit Effort

CREMP – Core Receiving Environment Monitoring Program

CSQG – Canadian Sediment Quality Guidelines

CWQG – Canadian Water Quality Guidelines

DOC –Dissolved Organic Carbon

EEM – Environmental Effects Monitoring

FFG – Functional Feeding Group

GPS – Global Positioning System

HPG – Habitat Preference Group

HSD – Honestly Significant Difference

KS –Kolmogorov-Smirnov

MRTF – Mary River Tributary-F

NW – Northwest

PEL – Probable Effect Level

PSQG – Provincial Sediment Quality Guideline

PWQO – Provincial Water Quality Objective

QA/QC – Quality Assurance / Quality Control

SE – Southeast

SEL – Severe Effect Level

SQG – Sediment Quality Guideline

TDS – Total Dissolved Solids

TKN – Total Kjeldahl Nitrogen

TOC – Total Organic Carbon

TSS – Total Suspended Solids

UTM – Universal Transverse Mercator

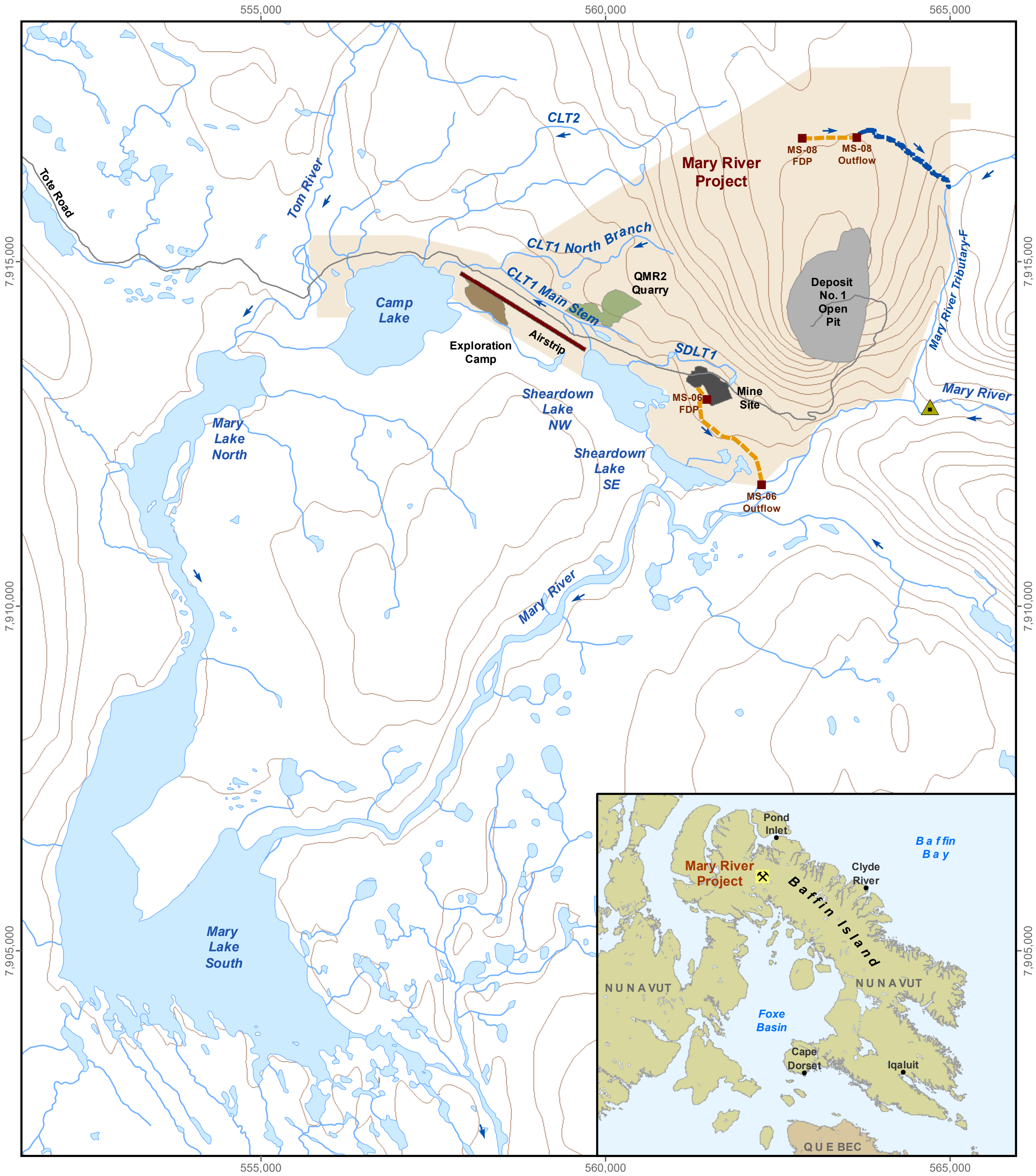
WQG – Water Quality Guideline

YOY –Young of the Year

1 INTRODUCTION

The Mary River Project, owned and operated by Baffinland Iron Mines Corporation (Baffinland), is a high-grade iron ore mining operation located in the Qikiqtani Region of northern Baffin Island, Nunavut (Figure 1.1). Open pit mining, including pit bench development, ore haulage and stockpiling, and the crushing and screening of high-grade iron ore, commenced at the Mary River Project in mid-September 2014. For the initial mining stages, as much as 4.2 million tonnes (Mt) of crushed/screened ore generated at the Project's mine site is transported annually by truck to Milne Port, which is located approximately 100 km north of the mine site. At Milne Port, the ore is stockpiled before being loaded onto bulk carrier ships for transport to international markets during the summer ice-free period. No milling or additional ore processing is conducted on-site, and thus no tailings are produced at the Mary River Project. All waste rock generated at the Project is deposited at a waste rock stockpile facility at the mine site (Figure 1.1). Ore processing areas and the waste rock stockpile facility are equipped with surface water management ponds to control and analyze runoff prior to discharge. Potential inputs to aquatic systems from Project operations at the mine site include effluent discharges from surface water management ponds (MS-06, MS-08) and sewage treatment plants, fugitive dust emissions from mining, crushing, and trucking operations, explosives residue, and sediment deposition from general mine site runoff.

Under terms and conditions of a Type 'A' Water Licence issued by the Nunavut Water Board (No. 2AM-MRY1325 Amendment No. 1), Baffinland developed an Aquatic Effects Monitoring Plan (AEMP) for the Mary River Project. A key objective of the AEMP was to provide data and information to allow the evaluation of short- and long-term effects of the Project on aquatic ecosystems. To meet this objective, Baffinland developed a Core Receiving Environment Monitoring Program (CREMP) to assess potential mine-related influences on water quality, sediment quality and biota (including phytoplankton, benthic invertebrates and fish) at aquatic environments located near the mine (Baffinland 2015; KP 2014; NSC 2014). The primary receiving systems that are the focus for the CREMP include the Camp Lake system (Tributaries 1 and 2, Camp Lake), the Sheardown Lake system (Tributaries 1, 9 and 12; Sheardown Lake NW and Sheardown Lake SE), Mary River, and Mary Lake (Figure 1.1). Over the initial three years of mine operation, the CREMP studies indicated some effects of the Mary River Project mine operations on water quality and sediment quality of receiving waterbodies, but these effects were confined to single tributaries feeding into each of Camp and Sheardown lakes, as well as near the immediate outlets of these tributaries to each respective lake (Minnow 2016a, 2017, 2018). No adverse mine-related effects to phytoplankton, benthic invertebrate, or fish were indicated at any of the Camp, Sheardown, or Mary lake systems in 2015, 2016, or 2017 based on comparisons



LEGEND

- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Discharge Line
- Overland Effluent Channel
- QMR2 Quarry
- Airstrip
- Exploration Camp
- Mine Site
- Open Pit
- Mary River Project

Baffinland Iron Mines Corporation, Mary River Project Location

0 1.25 2.5 5 km

Map Projection: UTM Zone 17N NAD 1983

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Figure 1.1

Document Path: S:\Projects\187202\187202.0025 - Baffinland 2018 AEMP\4 - GIS\CREMP Study\18-25 Figure 1.1 Location.mxd to representative reference waterbodies and to available pre-mine baseline data for each lake system (Minnow 2016a, 2017, 2018).

This report presents the methods and results of the 2018 CREMP, including an evaluation of potential Mary Lake Project-related influences on chemical and biological conditions at mine-exposed waterbodies through the fourth full year of mine operation. As in the three previous studies, the 2018 Mary River Project CREMP included water quality monitoring, sediment quality monitoring, phytoplankton monitoring, benthic invertebrate community assessment, and an arctic charr (*Salvelinus alpinus*) fish population assessment. The 2018 CREMP was implemented in accordance with the original study design (Baffinland 2015) with the exception of the continued use of a reference creek benthic invertebrate community study area that was originally added in 2016 to improve the program's ability to evaluate mine-related influences on stream biota (Minnow 2016b, 2017, 2018).

2 METHODS

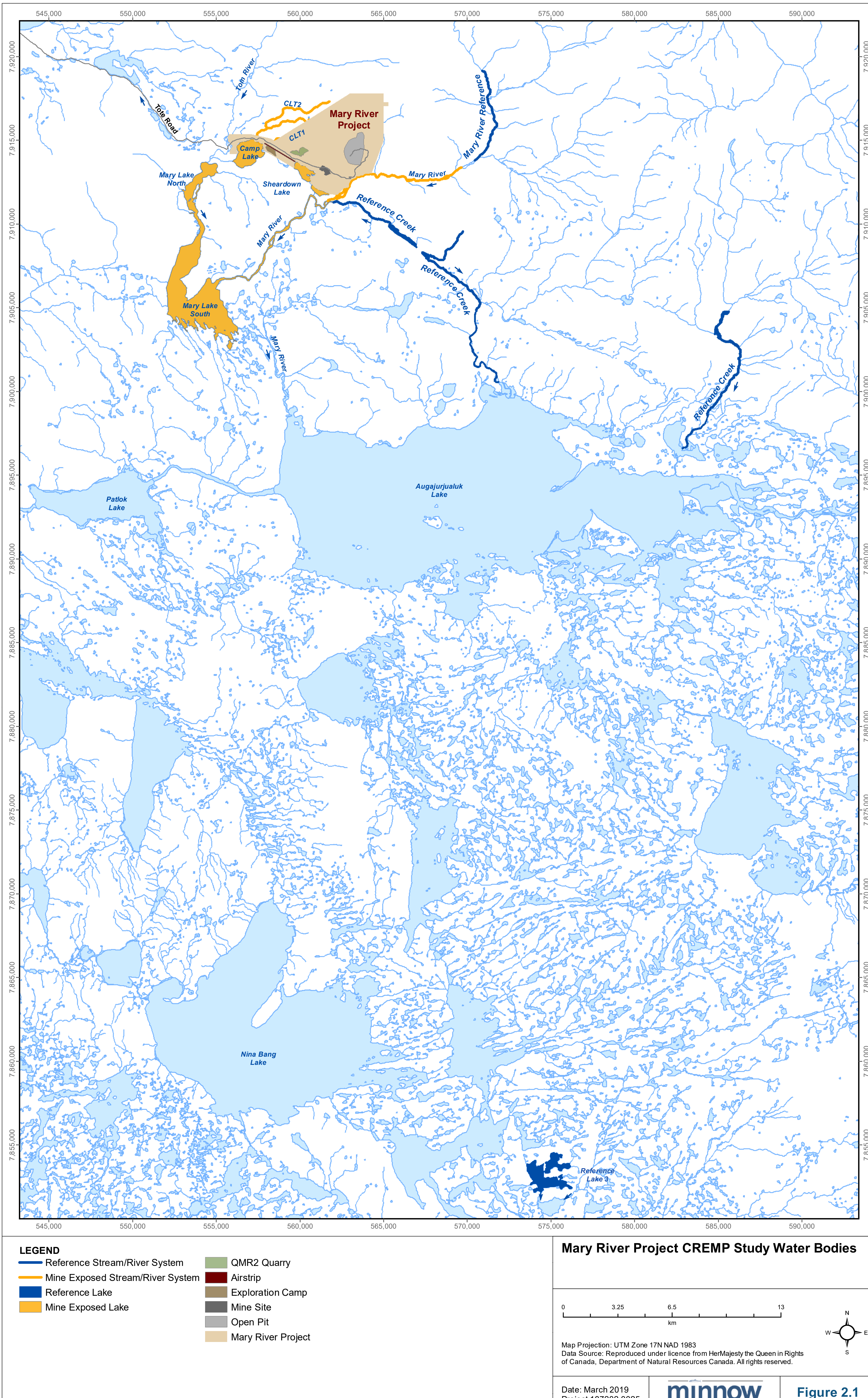
2.1 Overview

The Mary River Project CREMP includes water quality monitoring, sediment quality monitoring, phytoplankton (chlorophyll-a) monitoring, benthic invertebrate community assessment, and fish population assessment (Baffinland 2015). In 2018, water quality and phytoplankton monitoring was conducted by Baffinland environment department personnel over four separate sampling events, including a lake ice-cover event (April 13th to 23rd) and open-water season events corresponding to Arctic spring (freshet), summer, and autumn (June 30th to July 3rd, July 29th to August 12th, and August 19th to 27th, respectively). Sediment quality, benthic invertebrate community, and fish population sampling was conducted by Minnow Environmental Inc. (Minnow) personnel with assistance from Baffinland environment department staff from August 15th to 29th 2018, the seasonal timing of which was consistent with monitoring conducted for previous baseline (2005 to 2013), mine construction (2014), and mine operational (2015, 2016, 2017) studies. Similar to previous CREMP studies, the 2018 study included field sampling and standard laboratory quality assurance/quality control (QA/QC) for individual water quality, sediment quality, and benthic invertebrate community study components to allow for an assessment of the overall quality of each respective data set (Appendix A).

The 2018 CREMP study areas included the same mine-exposed and reference waterbodies established in the original design documents (Baffinland 2015; KP 2014; NSC 2014) and the same reference creek and lake that was added to the program in 2015 (Figure 2.1). To simplify the discussion of results, the mine-exposed study areas were separated by lake catchment as follows:

- the Camp Lake system (Camp Lake Tributaries 1 and 2, and Camp Lake);
- the Sheardown Lake system (Sheardown Lake Tributaries 1, 9, and 12, Sheardown Lake Northwest [NW], and Sheardown Lake Southeast [SE]); and,
- the Mary River/Mary Lake system.

Reference Lake 3, which served as a reference waterbody for lentic (lake) environments beginning in the 2015 CREMP study, was again used as the reference lake for the 2018 study. Reference Lake 3 is located approximately 62 km south of the Mary River Project (Figure 2.1), and is well outside the area of mine influence. Streams used as reference areas in the current and previous CREMP included an unnamed tributary to the Mary River and two unnamed tributaries to Angajurjuatuk Lake, all of which are located southeast of the mine (Figure 2.1). As in the previous CREMP studies, an area of Mary River located well upstream of current Baffinland



Document Path: S:\Projects\187202\187202.0025 - Baffinland 2018 AEMP\4 - GIS\CREMP Study\18-25 Figure 2.1 Study Waterbodies.mxd mine activity (i.e., GO-09) served as a reference area for the mine-exposed portion of Mary River in the 2018 study (Figure 2.1).

2.2 Water Quality

2.2.1 General Design

Surface water quality monitoring was conducted by Baffinland environment department personnel at the sampling locations and frequencies stipulated in the Mary River Project CREMP design (Baffinland 2015; KP 2014). The surface water sampling was conducted at as many as 57 stations per sampling period (Table 2.1; Figures 2.2 and 2.3), and included collection of *in situ* measurements and water chemistry data.

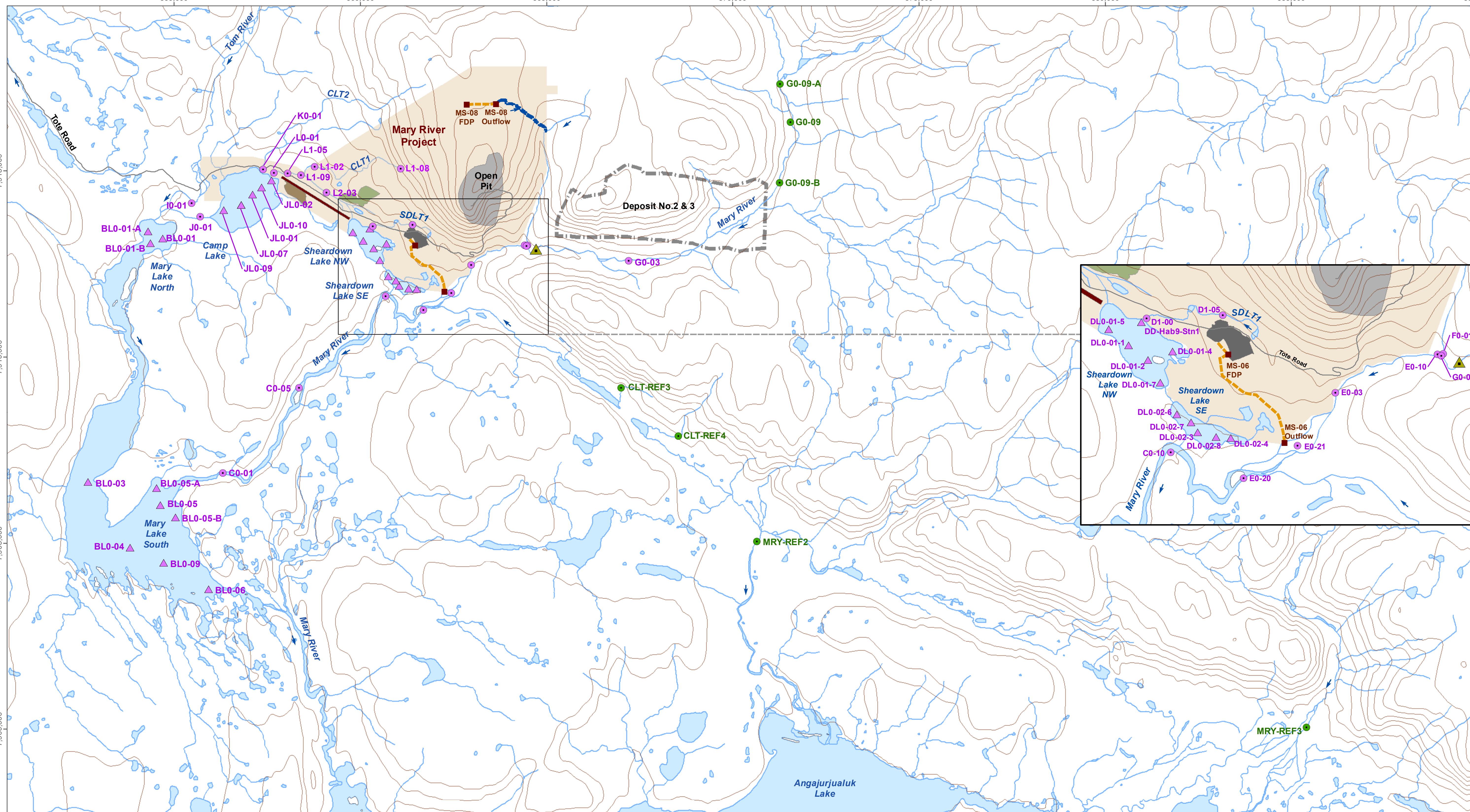
2.2.2 In situ Water Quality Measurement Data Collection and Analysis

In situ measurements of water temperature, dissolved oxygen, pH, specific conductance (i.e., temperature standardized measurement of conductivity), and turbidity were taken at the bottom of the water column at all lotic (i.e., creek, river) stations and as a vertical profile at one metre intervals at each lentic (i.e., lake) water quality monitoring station during routine monitoring conducted by Baffinland. These *in situ* measurements were also collected at the surface and bottom (i.e., approximately 30 cm above the water-sediment interface) at all lake benthic invertebrate community (benthic) stations during the fall biological sampling completed by Minnow, with the exception of turbidity measurements. The *in situ* measurements were collected using one of three YSI ProDSS (Digital Sampling System) meters equipped with a 4-Port sensor (YSI Inc., Yellow Springs, OH). Meter readings for pH, specific conductance, and turbidity were checked against standard solutions and calibrated as necessary in the morning on the day in which sampling was to be completed, prior to field sampling. Dissolved oxygen concentration readings were checked and calibrated at a greater frequency through each sampling day in response to changing sampling conditions (e.g., changes in elevation, barometric pressure, and/or ambient temperature). During the winter ice-cover sampling event, a gas-powered, 15 centimetre (6-inch) diameter ice auger was used to access the water column at all lake water quality monitoring stations. All ice shavings were removed from the auger hole prior to the collection of *in situ* measures. To avoid confounding influences associated with snow/ice melt in the auger hole, the *in situ* measurements were collected beginning just below the ice layer. Additional supporting observations of water colour and clarity were recorded at the time of water quality and biological sampling at all benthic stations, and Secchi depth was measured at all lake stations during ice-free periods using the methods outlined in Wetzel and Likens (2000).

Table 2.1: Mary River Project CREMP Water Quality and Phytoplankton Monitoring Station Coordinates and Annual Sampling Schedule

Study System	Water Body	Station ID	UTM Zone 17N, NAD83		Ref. Data Set [a]	Sampling Season			
			Easting	Northing		Winter (Apr. - May)	Spring (June)	Summer (July)	Fall (Aug. - Sept.)
Reference Areas	Creek Reference	CLT-REF3	567004	7909174	na	-	✓	✓	✓
		CLT-REF4	568533	7907874		-	✓	✓	✓
		MRY-REF3	585407	7900061		-	✓	✓	✓
		MRY-REF2	570650	7905045		-	✓	✓	✓
	Reference Lake 3	REF-03-W1	575642	7852666	na	-	-	✓	✓
		REF-03-W2	574836	7852744		-	-	✓	✓
		REF-03-W3	574158	7853237		-	-	✓	✓
	Mary River Reference	G0-09-A	571264	7917344	na	-	✓	✓	✓
		G0-09	571546	7916317		-	✓	✓	✓
		G0-09-B	571248	7914682		-	✓	✓	✓
Camp Lake System	Camp Lake Tributaries	I0-01	555470	7914139	a	-	✓	✓	✓
		J0-01	555701	7913773		-	✓	✓	✓
		K0-01	557390	7915030		-	✓	✓	✓
		L0-01	557681	7914959		-	✓	✓	✓
		L1-02	558765	7915121		-	✓	✓	✓
		L1-05	558040	7914935		-	✓	✓	✓
		L1-08	561076	7915068		-	✓	✓	✓
		L1-09	558407	7914885		-	✓	✓	✓
		L2-03	559081	7914425		-	✓	✓	✓
	Camp Lake	JL0-01	557108	7914369	b	✓	-	✓	✓
		JL0-02	557615	7914750		✓	-	✓	✓
		JL0-07	556800	7914094		✓	-	✓	✓
		JL0-09	556335	7913955		✓	-	✓	✓
		JL0-10	557346	7914562		✓	-	✓	✓
Sheardown Lake System	Sheardown Tributary 1	D1-00	560329	7913512	a	-	✓	✓	✓
		D1-05	561397	7913558		-	✓	✓	✓
	Sheardown Lake NW	DD-Hab9-Stn1	560259	7913455	b	✓	-	✓	✓
		DL0-01-1	560080	7913128		✓	-	✓	✓
		DL0-01-2	560353	7912924		✓	-	✓	✓
		DL0-01-4	560695	7913043		✓	-	✓	✓
		DL0-01-5	559798	7913356		✓	-	✓	✓
		DL0-01-7	560525	7912609		✓	-	✓	✓
	Sheardown Lake SE	DL0-02-3	561046	7911915	b	✓	-	✓	✓
		DL0-02-4	561511	7911832		✓	-	✓	✓
		DL0-02-6	560756	7912167		✓	-	✓	✓
		DL0-02-7	560952	7912054		✓	-	✓	✓
		DL0-02-8	561301	7911846		✓	-	✓	✓
Mary River and Mary Lake System	Mary River	G0-03	567204	7912587	c	-	✓	✓	✓
		G0-01	564459	7912984		-	✓	✓	✓
		F0-01	564483	7913015		-	✓	✓	✓
		E0-21	562444	7911724		-	✓	✓	✓
		E0-20	561688	7911272		-	✓	✓	✓
		E0-10	564405	7913004		-	✓	✓	✓
		E0-03	562974	7912472		-	✓	✓	✓
		C0-10	560669	7911633		-	✓	✓	✓
		C0-051	558352	7909170		-	✓	✓	✓
		C0-01	556305	7906894		-	✓	✓	✓
	Mary Lake (North Basin)	BL0-01	554691	7913194	b	✓	-	✓	✓
		BL0-01-A	554300	7913378		✓	-	✓	✓
		BL0-01-B	554369	7913058		✓	-	✓	✓
	Mary Lake (South Basin)	BL0-03	552680	7906651	b	✓	-	✓	✓
		BL0-04	553817	7904886		✓	-	✓	✓
		BL0-05	554632	7906031		✓	-	✓	✓
		BL0-06	555924	7903760		✓	-	✓	✓
		BL0-05-A	554530	7906478		✓	-	✓	✓
		BL0-05-B	555034	7905692		✓	-	✓	✓
		BL0-09	554715	7904479		✓	-	✓	✓

[a] Reference data applicable to indicated study area include a - lotic reference stations; b - lentic reference stations; and, c - Mary River upstream stations.



LEGEND

Water Monitoring Stations

- Lake - Mine Exposed
- Stream - Mine Exposed
- Stream - Reference
- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Discharge Line
- Overland Effluent Channel
- QMR2 Quarry
- Airstrip
- Exploration Camp
- Mine Site
- Open Pit
- Mary River Project
- Lease Boundary For Deposit No. 2 & 3
- Contours (20 m)

0 1.5 3 6
km

Map Projection: UTM Zone 17N NAD 1983
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N W E S

Mary River Project, CREMP Routine Water Quality and Phytoplankton Monitoring Station Locations

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Figure 2.2

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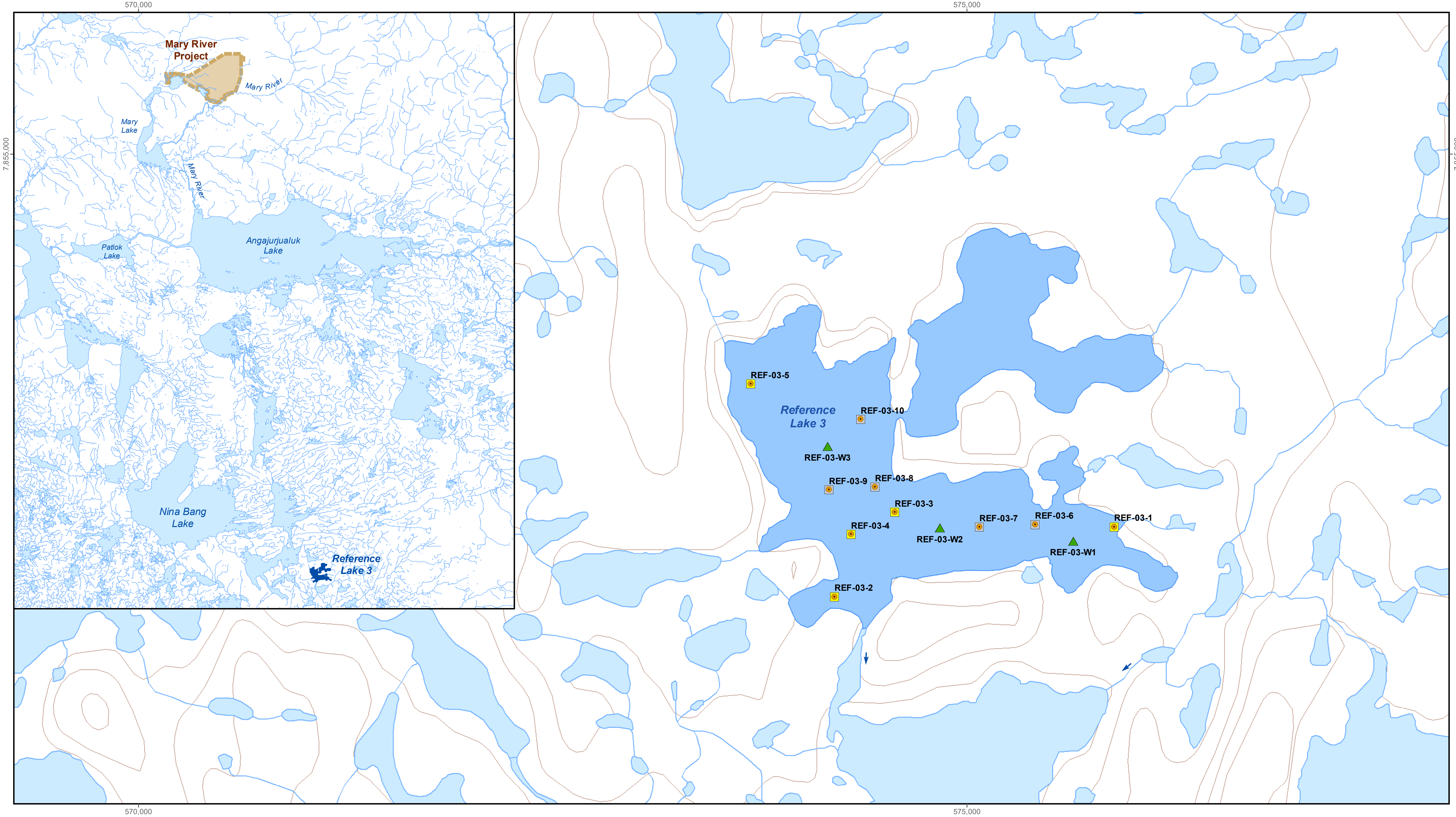

LEGEND

- Sediment and Benthic Monitoring Location
- Littoral Sampling Depth
- Profundal Sampling Depth
- Water Quality and Phytoplankton Monitoring Station
- Reference Lake

0 0.35 0.7 1.4
km

N W E S

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Mary River Project CREMP Reference Lake 3 Monitoring Station Locations

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Figure 2.3

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In situ water quality data collected at the mine-exposed study streams, rivers and lakes were compared to respective reference area data, to applicable water quality guidelines (WQG[1]; dissolved oxygen concentrations and pH only), and, for pH and conductivity, to baseline data. *In situ* water quality data were compared spatially within each system (i.e., from upstream- to downstream-most stations) using both qualitative and statistical approaches. For the statistical analysis, raw data and log-transformed data were assessed for normality and homogeneity of variance prior to conducting comparisons between (pair-wise) or among (multiple-group) applicable like-habitat mine-exposed and reference study area groups using Analysis-of-Variance (ANOVA). The selection of whether untransformed or log-transformed data were used for the ANOVA tests was determined based on which data best met the assumptions of ANOVA. In instances where normality could not be achieved through data transformation, non-parametric Mann-Whitney U-test and Kruskal-Wallis H-test statistics were applied using the raw data to validate the pair-wise and multiple-group ANOVA statistical results, respectively. Similarly, in instances in which variances of normal data could not be homogenized by transformation, Student's t-tests assuming unequal variance were applied using either raw or log-transformed data to validate the pair-wise ANOVA statistical results. In cases in which multiple-group comparisons were conducted, Tukey's Honestly Significant Difference (HSD) and Tamhane's pair-wise *post hoc* tests were implemented for homogenous and non-homogenous data, respectively All statistical comparisons were conducted using SPSS Version 12.0 software (SPSS Inc., Chicago, IL).

Vertical profiles of the *in situ* measurements taken from lake stations were plotted and visually assessed to evaluate potential thermal or chemical (i.e., using specific conductance) stratification and the corresponding depths associated with any distinct layering. The occurrence of a thermocline was conservatively assessed as a ≥0.5˚C change in temperature per 1 m incremental change in depth[2]. The vertical profile data collected at the mine-exposed study lakes were compared to that of Reference Lake 3 for each seasonal monitoring event using profile data averaged for each incremental depth below the water surface at each lake. At each study lake, spatial and seasonal differences in the vertical profile plots were evaluated to provide a better understanding of natural conditions and/or mine-related influences on within-lake water quality. Additional evaluation of the *in situ* dissolved oxygen concentration and pH data included comparisons to water quality guidelines (WQG)[1].

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[1] Canadian Environmental Quality Guidelines (CCME 1999, 2017) were used as the primary source for WQG, including those for pH and dissolved oxygen concentrations.

[2] Wetzel (2001) defines the thermocline as a ≥1˚C change in temperature per 1 m incremental change in depth, and thus a ≥0.5˚C change in temperature per 1 m incremental change in depth was considered highly conservative.

2.2.3 Water Chemistry Sampling and Data Analysis

Surface water chemistry samples were collected from both lotic and lentic environments (Table 2.1). At lotic stations, the water chemistry samples were collected from approximately mid-water column by hand directly into pre-labeled sample bottles which, for those requiring preservation, were pre-dosed or dosed in the field with required chemical preservatives. At lentic stations, two water chemistry samples were collected, one approximately 1 m below the surface (or just below the ice layer for the winter sampling event) and the other from approximately 1 m above the bottom, using a vertically-oriented 2.2 L TT Silicon Kemmerer bottle (Wildco Supply Co., Yulee, FL). During the winter sampling event, the water column was accessed at the same time and using the same methods as described above for the *in situ* measurements. Lake water collected using the beta-bottle was transferred directly into sample bottles that had been either pre-dosed or were subsequently dosed with the required chemical preservatives. In cases in which filtration was required (e.g., for dissolved metals), filtration was conducted in the field using methods consistent with AEMP standard operating procedures (Baffinland 2015).

Following collection, the water chemistry samples were placed into coolers in the field and maintained at cool temperatures for shipment to the analytical laboratory. Field water chemistry sampling QA/QC included trip blanks, field blanks, and the collection of equipment blanks and field duplicates with replication conducted on as many as 10% of the total samples collected for each CREMP sampling event (Appendix A). The water chemistry samples were shipped on ice to ALS Canada Ltd. (ALS; Waterloo, ON) for analysis of pH, conductivity, hardness, total suspended solids (TSS), total dissolved solids (TDS), anions (alkalinity, bromide, chloride, sulphate), nutrients (ammonia, nitrate, nitrite, total Kjeldahl nitrogen [TKN], total phosphorus), dissolved and total organic carbon (DOC and TOC, respectively), mercury, total and dissolved metals, and phenols using standard laboratory methods.

The water chemistry data were compared: i) among mine-exposed and reference areas for each study lake catchment (Table 2.1); ii) spatially and seasonally at each mine-exposed waterbody; iii) to applicable WQG for the protection of aquatic life (Table 2.2); iv) to site specific water quality benchmarks developed for the Mary River Project AEMP (Intrinsik 2014); and, v) to baseline water quality data. For data screening, and to simplify discussion of results, the magnitude of elevation in parameter concentrations was calculated as the mine-exposed area mean concentration divided by the respective reference station/area mean concentration. Similarly, for temporal comparisons, the magnitude of elevation in parameter concentrations was calculated by dividing the individual mine-exposed station/area 2018 mean concentrations by the baseline (2005 to 2013 data) mean concentration for each parameter. The resulting magnitude of elevation in

Table 2.2: Water Quality Guidelines Used for the Mary River Project 2015 to 2018 CREMP Studies

Parameters		Units	Water Quality Guideline (WQG)[a]	Criteria Source[a]	Supporting Information and/or Calculations Used to Derive Hardness Dependent Criteria
Conventionals	pH (lab)	pH	6.5 - 9.0	CWQG	-
Nutrients and Organics	Nitrate	mg/L	13	CWQG	-
	Nitrite	mg/L	0.06	CWQG	-
	Total Phosphorus	mg/L	0.020	PWQO	Total phosphorus objective is 0.030 mg/L for lotic (rivers, streams) environments, and 0.020 mg/L for lentic (lake) environments.
	Phenols	mg/L	0.001	PWQO	-
Anions	Chloride (Cl)	mg/L	120	CWQG	-
	Sulphate (SO_4)	mg/L	218	BCWQG	Sulphate guideline is hardness (mg/L $CaCO_3$) dependent as follows: 128 mg/L at 0 - 30 hardness, 218 mg/L at 31 - 75 hardness, 309 mg/L at 76 - 180 hardness, and 429 mg/L at 181 - 250 hardness. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
Total Metals	Aluminum (Al)	mg/L	0.100	CWQG	-
	Antimony (Sb)	mg/L	0.020	PWQO	-
	Arsenic (As)	mg/L	0.005	CWQG	-
	Beryllium (Be)	mg/L	0.011	PWQO	-
	Boron (B)	mg/L	1.5	CWQG	-
	Cadmium (Cd)	mg/L	0.00012	CWQG	Cadmium guideline is hardness (mg/L $CaCO_3$) dependent. For hardness between 17 and 280 mg/L, the cadmium guideline is calculated using the equation Cd (ug/L) = $10^{(0.83[\log(hardness)] - 2.46)}$. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
	Chromium (Cr)	mg/L	0.0089	CWQG	-
	Cobalt (Co)	mg/L	0.001	PWQO	-
	Copper (Cu)	mg/L	0.002	CWQG	Copper guideline is hardness (mg/L $CaCO_3$) dependent. At hardness <82 mg/L and >180 mg/L, the copper guideline is 2 and 4 ug/L, respectively. For hardness ranging from 82 - 180 mg/L, the copper guideline (ug/L) = 0.2 * $e^{(0.8545[\ln(hardness)] - 1.463)}$. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
	Iron (Fe)	mg/L	0.30	CWQG	-
	Lead (Pb)	mg/L	0.002	CWQG	Lead guideline is hardness (mg/L $CaCO_3$) dependent. At hardness <60 mg/L and >180 mg/L, the lead guideline is 1 and 7 ug/L, respectively. For hardness ranging from 60 - 180 mg/L, the lead guideline (ug/L) = $e^{(1.273[\ln(hardness)] - 4.705)}$. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
	Manganese (Mn)	mg/L	0.935	BCWQG	Manganese guideline is hardness (mg/L $CaCO_3$) dependent, and calculated using the equation Mn (ug/L) = 0.0044 * (hardness) + 0.605. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with hardness of 75 mg/L.
	Mercury (Hg)	mg/L	0.000026	CWQG	-
	Molybdenum (Mo)	mg/L	0.073	CWQG	-
	Nickel (Ni)	mg/L	0.077	CWQG	Nickel guideline is hardness (mg/L $CaCO_3$) dependent. At hardness <60 mg/L and >180 mg/L, the nickel guideline is 25 and 150 ug/L, respectively. For hardness ranging from 60 - 180 mg/L, the nickel guideline (ug/L) = $e^{(0.76[\ln(hardness)] + 1.06)}$. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
	Selenium (Se)	mg/L	0.001	CWQG	-
	Silver (Ag)	mg/L	0.00025	CWQG	-
	Thallium (Tl)	mg/L	0.0008	CWQG	-
	Tungsten	mg/L	0.030	PWQO	-
	Uranium (U)	mg/L	0.015	CWQG	-
	Vanadium (V)	mg/L	0.006	PWQO	-
	Zinc (Zn)	mg/L	0.030	CWQG	-

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME1999, 2017) was selected where a CCME guideline exists. Where no CCME guideline exists, the selected criteria is the lowest of either the Ontario Provincial Water Quality Objective (PWQO; OMOE 1994) or the British Columbia Water Quality Guideline (BCWQG; BCMOE 2013), as available.

parameter concentrations was qualitatively assigned as slightly, moderately, or highly elevated compared to reference and/or baseline conditions using the categorization described in Table 2.3.

Table 2.3: Magnitude of Elevation Categorizations for Water and Sediment Chemistry Comparisons

Categorization	Magnitude of Elevation Criterion
Slightly elevated	Concentration 3-fold to 5-fold higher at mine-exposed area versus the reference area or baseline data, as applicable.
Moderately elevated	Concentration 5-fold to 10-fold higher at mine-exposed area versus the reference area or baseline data, as applicable.
Highly elevated	Concentration ≥ 10-fold higher at effluent-exposed area versus the reference area or baseline data, as applicable.

Applicable WQG included the Canadian Water Quality Guidelines (CWQG; CCME 1999, 2017) or, for parameters with no CWQG, the most conservative (i.e., lowest) criterion available from established Ontario Provincial Water Quality Objectives (PWQO; OMOEE 1994) or British Columbia Water Quality Guidelines (BCWQG; BCMOE 2006, 2017). The water quality guidelines are abbreviated simply as 'WQG' in this report, although it is recognized that in certain cases the values presented may represent water quality 'objectives'. For those water quality guidelines that are hardness dependent, the hardness of the individual sample was used to calculate the water quality guideline for the specific parameter according to established formulae (Table 2.2). The water chemistry data were also compared to site specific water quality benchmarks developed for the Mary River Project AEMP (Intrinsik 2014). The Mary River Project AEMP water chemistry benchmarks were derived using an evaluation of background (i.e., baseline) water chemistry data together with existing generic water quality guidelines that consider aquatic toxicity thresholds. The AEMP benchmarks were developed to inform management decisions under the AEMP assessment approach and management response framework (Baffinland 2015). An elevation in parameter concentration above the respective AEMP benchmark may trigger various actions (e.g., sampling design modifications, additional statistical assessment, considerations for mitigation, etc.) to better understand and potentially mitigate effects resulting from elevated concentrations of the parameter of concern (Baffinland 2015). Water chemistry data for key parameters (i.e., parameters with concentrations that were notably higher at mine-exposed areas compared to reference areas, that were historically identified as site-specific parameters of concern, and/or that were above WQG and/or AEMP benchmarks) were plotted to evaluate changes in concentrations in 2018 compared to baseline (2005 to 2013 data) and previous mine construction (2014) and operational (2015, 2016, 2017) studies.

2.3 Sediment Quality

2.3.1 General Design

Sediment quality monitoring under the Mary River Project CREMP focuses primarily on assessing potential mine-related effects to the sediment of lake environments based on a gradient design (Baffinland 2015; KP 2014, 2015). Sediment quality sampling was conducted at five to ten stations per study lake for physical and chemical characterization as outlined under the CREMP, with additional characterization of physical sediment properties conducted at four to six stations per study lake to support the benthic invertebrate community analysis (Table 2.4; Figure 2.4). The lake sediment stations were designated as littoral or profundal based on a sample collection cut-off depth of 12 m, which was used to define lake zonation during the baseline characterization studies (KP 2014, 2015). Sediment quality sampling of lotic (stream and river) habitats is conducted once every three years under the CREMP[3], and because sediment quality sampling of lotic habitat was last conducted in 2017, no sediment was collected at stream and river habitats in 2018.

2.3.2 Sample Collection and Laboratory Analysis

Sediment at study lakes was collected for physical and chemical characterization using a gravity corer (Hoskin Scientific Ltd., Model E-777-00) outfitted with a clean 5.1 cm inside-diameter polycarbonate tube. From each retrieved core sample containing an intact, representative sediment-water interface, the surficial two centimetres of sediment was manually extruded upwards into a graded core collar, sectioned with a stainless steel core knife, and placed into a pre-labeled plastic sample bag. Samples from three to four cores treated in this manner were composited to create a single sample at each station. Supporting measurements of total core sample length and depths of visually-apparent redox boundaries/horizons, as well as notes regarding sediment texture and colour for each visible horizon, general sediment odour (e.g., hydrogen sulphide), and presence of algae or plants on or in the sediment, were recorded for each core sample. For QA/QC purposes, a field duplicate 'split' sample was collected at all study lakes using the same coring methods discussed above but eight rather than four replicate core samples were taken to create the split sample (Table 2.4; Appendix A). Following collection, all sediment samples were placed into a cooler, transported to the mine, and stored under cool conditions until shipment to the analytical laboratory.

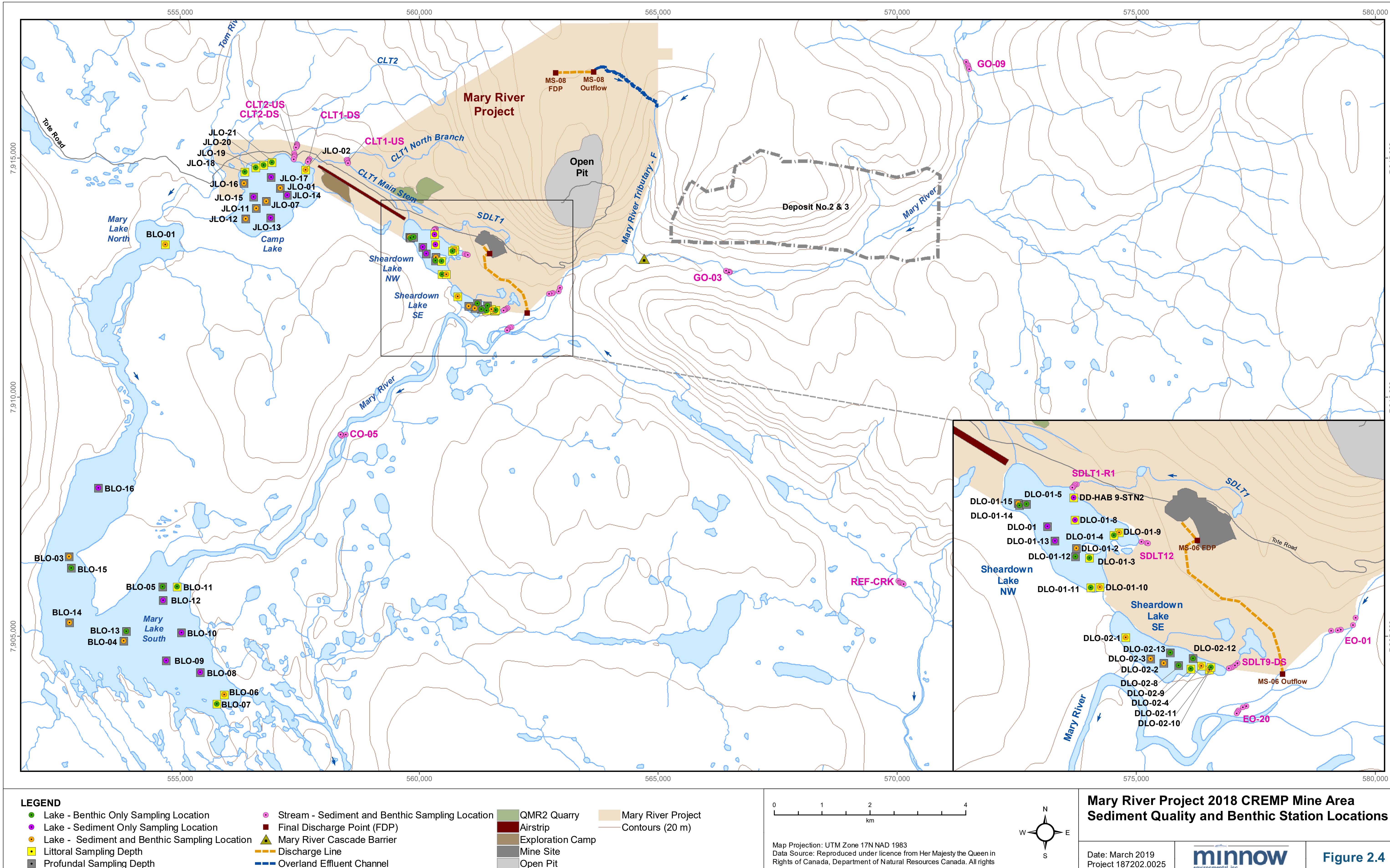
Upon completion of the biological monitoring field program, sediment samples were shipped to ALS (Waterloo, ON). Physical characterization of samples included percent moisture and particle

[3] The three year schedule for sampling of sediment at lotic habitat was based on a recommendation by regulators following the submission of the 2016 CREMP.

Table 2.4: Lake Sediment Quality and Benthic Invertebrate Community Monitoring Station Coordinates Used for the Mary River Project CREMP 2018 Study

Waterbody	Station Code	UTM Zone 17W		Sampling Habitat	Sample Type		
		Easting	Northing		Sediment Core[a]	Sediment petite-Ponar[a]	Benthic Invertebrate
Reference Lake 3	REF-03-1	575992	7852992	littoral	✓	-	✓
	REF-03-2	574200	7852330	littoral	✓	-	✓
	REF-03-3	574564	7852840	littoral	✓	-	✓
	REF-03-4	574301	7852705	littoral	✓	-	✓
	REF-03-5	573694	7853613	littoral	✓	-	✓
	REF-03-6	575411	7852766	profundal	✓	-	✓
	REF-03-7	575076	7852750	profundal	✓	-	✓
	REF-03-8	574445	7852992	profundal	✓	-	✓
	REF-03-9	574168	7852975	profundal	✓	-	✓
	REF-03-10[b]	574358	7853400	profundal	✓	-	✓
Camp Lake	JLO-02	557627	7914748	littoral	✓	-	✓
	JLO-01[b]	557092	7914370	profundal	✓	-	✓
	JLO-14	557246	7914224	profundal	✓	-	-
	JLO-17	556900	7914594	profundal	✓	-	-
	JLO-21	556926	7914911	littoral	-	✓	✓
	JLO-20	556750	7914850	littoral	-	✓	✓
	JLO-19	556587	7914801	littoral	-	✓	✓
	JLO-07	556803	7914095	profundal	✓	-	✓
	JLO-18	556357	7914706	littoral	-	✓	✓
	JLO-16	556335	7914470	profundal	✓	-	✓
	JLO-15	556542	7914184	profundal	✓	-	-
	JLO-11	556594	7913946	profundal	✓	-	✓
	JLO-13	556896	7913751	profundal	✓	-	-
	JLO-12	556378	7913728	profundal	✓	-	✓
Sheardown Lake Northwest (NW)	DLO-01-5	559806	7913348	profundal	✓	-	✓
	DLO-01-14	559821	7913328	profundal	-	✓	✓
	DLO-01-15	559884	7913340	profundal	-	✓	✓
	DD-HAB 9-STN2	560325	7913400	littoral	✓	-	-
	DLO-01-8	560338	7913192	littoral	✓	-	-
	DLO-01	560079	7913132	profundal	✓	-	-
	DLO-01-13	560151	7912997	profundal	✓	-	-
	DLO-01-2[b]	560350	7912927	profundal	✓	-	✓
	DLO-01-12	560339	7912852	profundal	-	✓	✓
	DLO-01-9	560746	7913076	littoral	✓	-	✓
	DLO-01-4	560696	7913049	littoral	-	✓	✓
	DLO-01-3	560471	7912838	littoral	-	✓	✓
	DLO-01-11	560482	7912563	littoral	-	✓	✓
	DLO-01-10	560570	7912566	littoral	✓	-	✓
Sheardown Lake Southeast (SE)	DLO-02-1[b]	560807	7912099	littoral	✓	-	✓
	DLO-02-11	561585	7911799	littoral	✓	-	✓
	DLO-02-10	561602	7911821	littoral	-	✓	✓
	DLO-02-4	561512	7911833	littoral	✓	-	✓
	DLO-02-12	561433	7911905	profundal	-	✓	✓
	DLO-02-9	561414	7911806	littoral	-	✓	✓
	DLO-02-8	561300	7911839	profundal	-	✓	✓
	DLO-02-13	561222	7911958	profundal	-	✓	✓
	DLO-02-2	561161	7911858	profundal	✓	-	✓
	DLO-02-3	561039	7911898	profundal	✓	-	✓
Mary Lake	BLO-01	554690	7913186	littoral	✓	-	✓
	BLO-16	553289	7908092	profundal	✓	-	-
	BLO-03	552679	7906660	profundal	✓	-	✓
	BLO-15	552723	7906419	profundal	-	✓	✓
	BLO-14	552688	7905282	profundal	✓	-	✓
	BLO-05	554635	7906033	profundal	-	✓	✓
	BLO-11	554942	7906033	littoral	-	✓	✓
	BLO-12	554644	7905742	profundal	✓	-	-
	BLO-13	553879	7905094	profundal	-	✓	✓
	BLO-04[b]	553820	7904893	profundal	✓	-	✓
	BLO-10	555033	7905065	profundal	✓	-	-
	BLO-09	554707	7904486	profundal	✓	-	-
	BLO-08	555424	7904239	profundal	✓	-	-
	BLO-07	555767	7903583	littoral	-	✓	✓
	BLO-06	555925	7903771	littoral	✓	-	✓

[a] Sediment core samples analyzed for particle size, TOC and total metals. Petite-ponar sediment grab samples analyzed for particle size only.
[b] Duplicate sediment core sample collected for quality control/quality assurance (QA/QC).



Mary River Project 2018 CREMP Mine Area Sediment Quality and Benthic Station Locations

Date: March 2019
Project 187202.0025

minnow environmental inc.

Figure 2.4

Document Path: S:\Projects\187202\187202.0025 - Baffinland 2018 AEMP\4 - GIS\CREMP Study\18-25 Figure 2.4 Sediment and Benthic.mxd size analyses, and chemical characterization included analyses of total organic carbon (TOC) and total metals including mercury. Standard laboratory methods were used for all physical and chemical sediment analyses.

2.3.3 Data Analysis

Sediment quality data from the mine-exposed lakes were compared to reference lake data, to applicable sediment quality guidelines/AEMP benchmarks and, where applicable, to baseline sediment quality data. Sediment physical characteristics (i.e., moisture, particle size) and TOC data were statistically summarized based on separate calculation of mean, standard deviation, standard error, minima, and maxima for littoral and profundal habitat at each study lake. These data were compared statistically between applicable mine-exposed and reference lakes using the same tests, transformations (with the exception that logit transformations were conducted for dependent proportional data rather than log transformations), assumptions, and software described previously for the statistical evaluation of *in situ* water quality (see Section 2.2.3).

The sediment chemistry data from the mine-exposed lakes were initially assessed to identify potential gradients in sediment metal concentrations with distance from known or suspected sources of mine-related deposits to the lake. For each sediment chemistry parameter, the data were separately averaged for littoral and profundal habitat at each lake and then compared between each respective mine-exposed and reference lake based on proportional elevation in parameter concentrations. The magnitude of elevation in average parameter concentrations between the mine-exposed and reference lakes was calculated and compared as described previously (Section 2.2.3; Table 2.3).

Sediment chemistry data collected at lake environments were compared to applicable Canadian Sediment Quality Guidelines (CSQG; CCME 1999) probable effect levels (PEL) or, for parameters with no CSQG, to Ontario Provincial Sediment Quality Guidelines (PSQG; OMOE 1993) severe effect levels (SEL). The sediment quality guidelines used for the 2018 CREMP were abbreviated simply as 'SQG', although it is recognized that the values presented may represent either national PEL or Ontario provincial SEL guidelines. The 2018 lake environment sediment chemistry data analyses also included comparisons to Mary River Project AEMP sediment quality benchmarks that were derived using baseline sediment chemistry data for each mine-exposed lake and existing generic CSQG interim or PSQG lowest effect level sediment quality guidelines (Intrinsik 2014, 2015). As indicated previously, the AEMP benchmarks were developed to inform management decisions under the AEMP assessment approach and management response framework (Baffinland 2015). An elevation in parameter concentration above the AEMP benchmark may trigger various actions to better understand and potentially mitigate effects resulting from elevated concentrations of the parameter of concern (Baffinland 2015).

Sediment chemistry data for key parameters (i.e., parameters with concentrations that were notably higher at mine-exposed areas compared to the reference area, that have been identified as site-specific parameters of concern in previous studies, and/or those with concentrations above SQG and/or AEMP benchmarks) were plotted to evaluate potential changes in parameter concentrations among the 2018 data, baseline data (2005 to 2013), and previous 2015 to 2017 mine operation period data. In addition, as described previously, the magnitude of elevation was calculated for all parameters using the 2018 data and baseline data for each individual study lake using the same calculation (and categorization description) as described previously (Section 2.2.3; Table 2.3).

2.4 Biological Assessment

2.4.1 Phytoplankton

The Mary River Project CREMP uses measures of aqueous chlorophyll-a concentrations to assess potential mine-related influences to phytoplankton. Because chlorophyll-a is the primary pigment of phytoplankton (i.e., algae and other photosynthetic microbiota suspended in the water column), aqueous chlorophyll-a concentrations are often used as a surrogate for evaluating the amount of photosynthetic microbiota in aquatic environments (Wetzel 2001). Chlorophyll-a samples were collected by Baffinland environmental department staff at the same stations and same time, as well as with the same methods and equipment, as described for the collection of water chemistry samples (Table 2.1; Figures 2.2 and 2.3; Section 2.2.3). The chlorophyll-a samples were collected into 1 L glass amber bottles and maintained in a cool and dark environment prior to submission to ALS (Mary River On-Site Laboratory, NU). On the same day of collection, the laboratory filtered the samples through a 0.45 micron cellulose acetate membrane filter assisted by a vacuum pump. Following filtration, the membrane filter was wrapped in aluminum foil, inserted into a labelled envelope, and then frozen. At the completion of field collections for the seasonal sampling event, the filters were shipped frozen to ALS in Waterloo, ON for chlorophyll-a analysis using standard methods. The field QA/QC applied during chlorophyll-a sampling was similar to that described for water chemistry sampling (see Section 2.2.3).

The CREMP study design also stipulates the collection of phytoplankton community samples for archiving (NSC 2014, 2015). In the event that water quality, chlorophyll-a, and/or other biological components indicate potential mine-related effects to primary productivity at a specific mine-exposed waterbody, the phytoplankton community samples may be processed to further investigate the nature of mine-related effects to phytoplankton biomass and community structure (e.g., taxonomic composition, richness, density). To date, none of the archived phytoplankton community samples have been processed (2006 to 2017). In 2018, phytoplankton community samples were collected using the same methods described in the CREMP (NSC 2014) and, as in the past, these samples were not processed, but were archived for potential future analysis.

The analysis of aqueous chlorophyll-a concentrations closely mirrored the approach used to evaluate the water quality data. Briefly, chlorophyll-a concentrations were compared: i) between respective mine-exposed and reference areas; ii) spatially and seasonally at each mine-exposed waterbody; iii) to AEMP benchmarks; and, iv) to baseline data. Comparisons of chlorophyll-a concentrations between the mine-exposed and reference areas were based on both qualitative and statistical approaches, the latter of which used the same parametric and/or non-parametric statistics, as appropriate, as described previously for statistical analysis of *in situ* water quality data (Section 2.2.2). An AEMP benchmark chlorophyll-a concentration of 3.7 µg/L was established for the Mary River Project (NSC 2014), and therefore the 2018 chlorophyll-a concentration data were compared to this benchmark to assist with the determination of potential mine-related enrichment effects at waterbodies influenced by mine operations. A mine-related effect on the productivity of a waterbody of interest was assessed as a chlorophyll-a concentration above the AEMP benchmark, the representative reference area, and/or the respective waterbody baseline condition.

2.4.2 Benthic Invertebrate Community

2.4.2.1 General Design

The Mary River Project CREMP benthic invertebrate community (benthic) survey outlines a habitat-based approach for characterizing potential mine-related effects to benthic biota of lotic (stream/river) and lentic (lake) environments (NSC 2014). Lotic areas sampled for benthic invertebrates included Camp Lake Tributaries 1 and 2 at historically established areas located upstream and downstream of the Milne Inlet Tote Road, Sheardown Lake Tributaries 1, 9 and 12 near their respective outlets, and Mary River upstream (two areas) and downstream (three areas) of the mine site (Table 2.5; Figure 2.4). Benthic samples were also collected at a reference creek located within the same unnamed tributary to Angajurjualuk Lake that is used for reference water quality sampling (Stations CLT-REF4 and MRY-REF2) as part of the 2018 CREMP to augment the original study design (Table 2.5; Figure 2.4). This reference creek, referred to as Unnamed Reference Creek herein, was initially sampled as part of the benthic invertebrate community assessment in the 2016 CREMP (see Minnow 2017). Consistent with the federal Environmental Effects Monitoring (EEM) program (Environment Canada 2012), five stations were sampled at each lotic study area with the exception of Sheardown Lake Tributary 12, where only three stations were sampled due to limited habitat available for sampling using conventional gear suitable for erosional habitat. As in 2015, 2016, and 2017, the level of replication used for lotic benthic sampling in 2018 was greater than specified under the original CREMP design in order to

Table 2.5: Stream and River Benthic Invertebrate Community Monitoring Station Coordinates Used for the Mary River Project CREMP 2018 Study

Lake System	Waterbody	Station Code	Station Type	UTM Zone 17W, NAD83	
				Easting	Northing
Angajurjualuk Lake	Unnamed Tributary	REF-CRK-B1	Reference	570025	7906148
		REF-CRK-B2	Reference	570060	7906115
		REF-CRK-B3	Reference	570093	7906110
		REF-CRK-B4	Reference	570121	7906099
		REF-CRK-B5	Reference	570137	7906086
Camp Lake	Camp Lake Tributary 1	CLT1-US-B1	Reference	558502	7914967
		CLT1-US-B2	Reference	558488	7914963
		CLT1-US-B3	Reference	558494	7914930
		CLT1-US-B4	Reference	558509	7914903
		CLT1-US-B5	Reference	558517	7914890
		CLT1-DS-B1	Mine-Exposed	557710	7914978
		CLT1-DS-B2	Mine-Exposed	557693	7914957
		CLT1-DS-B3	Mine-Exposed	557686	7914944
		CLT1-DS-B4	Mine-Exposed	557678	7914932
		CLT1-DS-B5	Mine-Exposed	557672	7914917
	Camp Lake Tributary 2	CLT2-US-B1	Reference	557441	7915291
		CLT2-US-B2	Reference	557451	7915275
		CLT2-US-B3	Reference	557450	7915251
		CLT2-US-B4	Reference	557441	7915237
		CLT2-US-B5	Reference	557423	7915215
		CLT2-DS-B1	Mine-Exposed	557392	7915104
		CLT2-DS-B2	Mine-Exposed	557398	7915053
		CLT2-DS-B3	Mine-Exposed	557400	7915032
		CLT2-DS-B4	Mine-Exposed	557997	7915008
		CLT2-DS-B5	Mine-Exposed	557377	7914971
Sheardown Lake Northwest (NW)	Sheardown Lake Tributary 1 (Reach 1)	SDLT1-R1-B1	Mine-Exposed	560352	7913522
		SDLT1-R1-B2	Mine-Exposed	560338	7913520
		SDLT1-R1-B3	Mine-Exposed	560328	7913507
		SDLT1-R1-B4	Mine-Exposed	560320	7913497
		SDLT1-R1-B5	Mine-Exposed	560313	7913493
	Sheardown Lake Tributary 12	SDLT12-B1	Mine-Exposed	560953	7912988
		SDLT12-B2	Mine-Exposed	561003	7912975
		SDLT12-B3	Mine-Exposed	561016	7912971
Sheardown Lake Southeast (SE)	Sheardown Lake Tributary 9	SDLT9-DS-B1	Mine-Exposed	561848	7911860
		SDLT9-DS-B2	Mine-Exposed	561825	7911838
		SDLT9-DS-B3	Mine-Exposed	561798	7911824
		SDLT9-DS-B4	Mine-Exposed	561785	7911816
		SDLT9-DS-B5	Mine-Exposed	561767	7911812
Mary Lake	Mary River	GO-09-B1	Reference	571447	7917010
		GO-09-B2	Reference	571479	7916946
		GO-09-B3	Reference	571489	7916919
		GO-09-B4	Reference	571499	7916883
		GO-09-B5	Reference	571503	7916858
		GO-03-B1	Mine-Exposed	566489	7912626
		GO-03-B2	Mine-Exposed	566509	7912616
		GO-03-B3	Mine-Exposed	566491	7912605
		GO-03-B4	Mine-Exposed	566425	7912630
		GO-03-B5	Mine-Exposed	566425	7912642
		EO-01-B1	Mine-Exposed	562944	7912281
		EO-01-B2	Mine-Exposed	562922	7912214
		EO-01-B3	Mine-Exposed	562806	7912171
		EO-01-B4	Mine-Exposed	562778	7912165
		EO-01-B5	Mine-Exposed	562717	7912158
		EO-20-B1	Mine-Exposed	561930	7911460
		EO-20-B2	Mine-Exposed	561895	7911447
		EO-20-B3	Mine-Exposed	561858	7911420
		EO-20-B4	Mine-Exposed	561848	7911408
		EO-20-B5	Mine-Exposed	561841	7911393
		CO-05-B1	Mine-Exposed	558465	7909208
		CO-05-B2	Mine-Exposed	558387	7909183
		CO-05-B3	Mine-Exposed	558365	7909214
		CO-05-B4	Mine-Exposed	558355	7909224
		CO-05-B5	Mine-Exposed	558359	7909209

provide consistency with EEM standards (Minnow 2016a). To the extent possible, previously established lotic benthic stations were incorporated into the 2018 sampling program to provide comparability to historical baseline information.

At lentic environments, benthic sampling was conducted at the 40 previously established CREMP stations among the four mine-exposed study lakes (i.e., ten stations in each of Camp, Sheardown NW, Sheardown SE and Mary lakes), as well as at the same ten stations established at Reference Lake 3 during the 2015 study (Table 2.5; Figures 2.3 and 2.4). Analysis of benthic data collected at Reference Lake 3 from 2015 to 2017 indicated that, similar to temperate lakes (Ward 1992), depth-related influences on benthic invertebrate community structure (e.g., density and richness) occurs naturally in lakes of the region (Minnow 2016a, 2017, 2018). Analysis of benthic data collected from Reference Lake 3 in 2018 provided on-going confirmation of the occurrence of natural depth-related influences on benthic invertebrate community structure in area lakes (Appendix B). Because of the occurrence of natural depth-related differences in benthic invertebrate communities, the benthic stations at each mine-exposed and reference lake were categorized as littoral zone (2 to 12 m depth) or profundal zone (>12 m depth) stations based on station depth (Table 2.5). To the extent possible, five littoral and five profundal stations were designated for each study lake based on the previously established suite of CREMP lentic benthic stations[4] in order to provide temporal continuity with the baseline studies and the original CREMP design (Table 2.4; Figure 2.4), as well as to allow data analysis in accordance with EEM standards. Specifically, the sampling of five stations from each zone at each study area ensures adequate statistical power to detect ecologically meaningful differences in benthic metrics of ± two standard deviations of a comparable reference area mean at an α and β of 0.10 (Environment Canada 2012).

2.4.2.2 Sample Collection and Laboratory Analysis

Two types of sampling equipment and methods were employed during the 2018 CREMP benthic survey to reflect different habitat types as follows:

- at **lotic (stream/river) stations** (i.e., predominantly cobble and/or gravel substrate in flowing waters), benthic samples were collected using a Surber sampler (0.0929 m^2 sampling area) outfitted with 500 µm mesh. At each erosional station, one sample representing a composite of three Surber sampler grabs (i.e., 0.279 m^2 area) was collected to ensure that each sample was representative of habitat conditions. A concerted effort

[4] At Sheardown Lake SE, depths greater than 12 m deep are spatially limited, and thus the five deepest CREMP stations were designated as profundal despite one of the five being less than 12 m deep. At Mary Lake, six of the CREMP stations occurred at depths well greater than 12 m and thus were all designated as profundal, with the four remaining stations designated as littoral.

was made to ensure that water velocity and substrate characteristics were comparable among respective lotic mine-exposed and reference study area stations to minimize natural influences on community variability. Once all three sub-samples were collected at each respective station, all material gathered in the Surber sampler net was transferred to a plastic sampling jar to which both external and internal station identification labels were affixed.

- at **lentic (lake) stations** (i.e., predominantly soft silt-sand, silt and/or clay substrates with variable amounts of organics), benthic sampling was conducted using a Petite Ponar grab sampler (15.24 x 15.24 cm; 0.023 m^2 sampling area). A single sample, consisting of a composite of five grabs (i.e., 0.115 m^2 sampling area) was collected at each station with care taken to ensure that each grab was acceptable (i.e., that the grab captured sufficient surface material and was full to each edge). Any incomplete grabs were discarded. For each acceptable grab, the Petite Ponar was thoroughly rinsed and the material then field-sieved through 500 µm mesh. Following sieving of all five grabs, the retained material was carefully transferred into a plastic sampling jar to which both external and internal station identification labels were affixed.

Following collection, the benthic samples were preserved to a level of 10% buffered formalin in ambient water. Supporting measurements and information collected at each replicate grab location for lotic stations included sampling depth, water velocity, and description of aquatic vegetation/algae presence. In addition, *in situ* water quality at the bottom of the water column and global positioning system (GPS) coordinates was collected and recorded at each lotic benthic station. Supporting information recorded at each lake benthic station included substrate description, presence of aquatic vegetation/algae, sampling depth, *in situ* water quality measurements near the water column surface and bottom, and GPS coordinates. All GPS coordinates were collected in Universal Transverse Mercator (UTM) units using a hand-held portable Garmin GPS72 (Garmin International Inc., Olathe, KS) device based on 1983 North America Datum (NAD 83).

Benthic samples were submitted to and processed by Zeas Inc. (Nobleton, ON) using standard sorting methods. Upon arrival at the laboratory, a biological stain was added to each benthic sample to facilitate greater sorting accuracy. The samples were washed free of formalin in a 500 µm sieve and the remaining sample material was then examined under a stereomicroscope at a magnification of at least ten times by a technician. All benthic invertebrates were removed from the sample debris and placed into vials containing 70% ethanol according to major taxonomic groups (i.e., order or family levels). A senior taxonomist later enumerated and identified the benthic organisms to the lowest practical level (typically genus or species) utilizing up-to-date taxonomic keys. Quality assurance/quality control (QA/QC) conducted during the laboratory processing of benthic samples included organism recovery and sub-sampling checks on as many as 10% of the total samples collected for the 2018 CREMP (Appendix A).

2.4.2.3 Data Analysis

Benthic data were evaluated separately for lotic, lentic littoral, and lentic profundal habitat data sets. Benthic invertebrate communities were evaluated using summary metrics of mean invertebrate abundance (or "density"; average number of organisms per m^2), mean taxonomic richness (number of taxa, as identified to lowest practical level), Simpson's Evenness Index (E), and the Bray-Curtis Index of Dissimilarity. Simpson's Evenness was calculated using the Krebs method (Smith and Wilson 1996) and Bray-Curtis Index was calculated using the formula provided in Environment Canada (2012). Additional comparisons were conducted using percent composition of dominant/indicator taxa, functional feeding groups, and habitat preference groups (calculated as the abundance of each respective group relative to the total number of organisms in the sample). Dominant/indicator taxonomic groups were defined as those groups representing, on average, greater than 5% of total organism abundance for a study area or any groups considered important indicators of environmental stress. Functional feeding groups (FFG) and habitat preference groups (HPG) were assigned based on Pennak (1989), Mandaville (2002), and/or Merritt et al. (2008) descriptions/designations for each taxon.

Statistical comparisons of all applicable benthic invertebrate community indices and community composition endpoints were conducted using the same tests described for the *in situ* water quality comparisons (see Section 2.2.2). Pair-wise differences between the mine-exposed and reference areas were preferentially tested using ANOVA on untransformed, normally distributed data. However, in the event that data were determined to be non-normal, a suite of transformations[5] including $\log_{10}$, square root, and fourth root was applied to the data and evaluated for normality. The transformation that resulted in normal data with lowest skew and kurtosis values was then used for statistical testing using ANOVA. In instances where normality could not be achieved through data transformation, non-parametric Mann-Whitney U-tests were used to validate the statistical results from pair-wise ANOVA tests. All statistical comparisons were conducted using R programming (R Foundation for Statistical Computing, Vienna, Austria).

An effect on benthic invertebrate communities was defined as a statistically significant difference between any paired mine-exposed and reference areas at a p-value of 0.10. For each endpoint showing a significant difference, a magnitude of difference was calculated between study area

[5] Non-normal dependent proportional benthic data were subject to a modified probit transformation that better accounted for nil (or near-zero) values in the statistical analysis than the other indicated transformations.

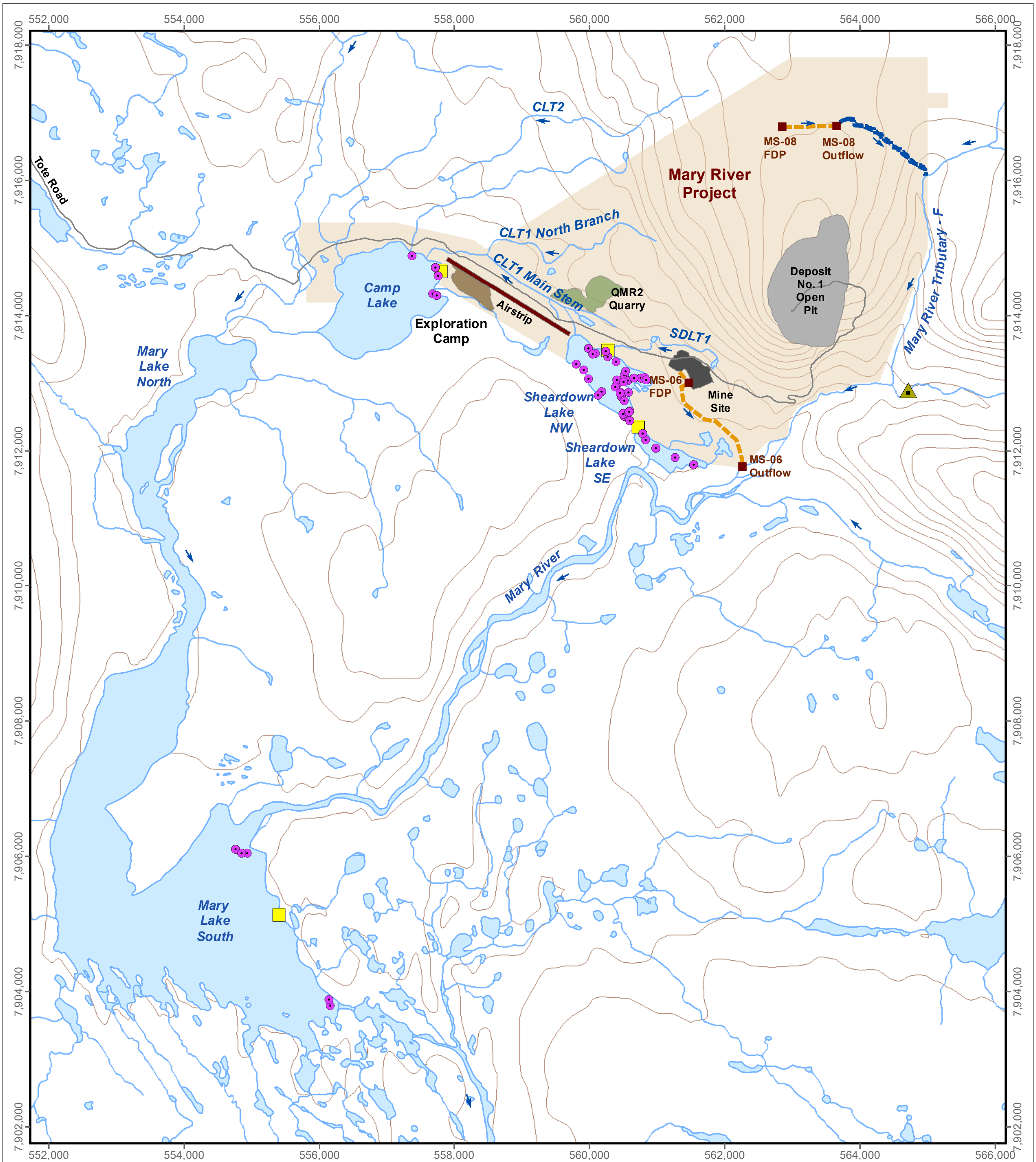
means. Because the benthic survey was designed to have sufficient power to detect a difference (effect size) of ± two standard deviations (SD), the magnitude of the difference was calculated to reflect the number of reference mean standard deviations (SD_{REF}) using equations provided by Environment Canada (2012). A Critical Effect Size for the benthic invertebrate community study (CES_{BIC}) of ± 2 SD_{REF} was used to define ecologically relevant 'effects', which is analogous to differences beyond those expected to occur naturally between two areas that are uninfluenced by anthropogenic inputs (i.e., between pristine reference areas; see Munkittrick et al. 2009, Environment Canada 2012).

Temporal comparisons included statistical evaluations among the baseline and 2015 to 2018 data for primary benthic metrics (i.e., density, richness, Simpson's Evenness), dominant invertebrate groups, and FFG using uni-variate tests (e.g., ANOVA) and pair-wise *post hoc* tests, as appropriate. The temporal statistical comparisons were conducted using the same tests, transformations, assumptions, and software described above for the *in situ* water quality comparisons (see Section 2.2.2). For study areas that contained data for multiple years (i.e., 3 or more), Tukey's HSD *post hoc* tests were used in instances in which normal data showed equal variance, and Tamhane's *post hoc* tests were used in instances in which normal data showed unequal variance. Similar to the 2018 within-year statistical analyses, the magnitude of difference was calculated for endpoints that differed significantly between years in the *post hoc* tests and was compared to the benthic survey CES_{BIC} of within two standard deviations of the baseline year mean (abbreviated as ±2 $SD_{BL\text{-}year}$).

2.4.3 Fish Population

2.4.3.1 General Design

The Mary River Project CREMP fish population survey outlines a non-lethal sampling design to evaluate potential mine-related effects to the fish population (e.g., age structure, growth, condition) at the mine-exposed lakes (NSC 2014, 2015). The fish population survey targeted arctic charr (*Salvelinus alpinus*) primarily because this species is the only abundant fish common to all of the mine's regional lakes, sufficient baseline catch and measurement data is available for this species to allow application of a before-after statistical evaluation, and because of this species importance as an Inuit subsistence food source. The approach employed for the CREMP fish population survey closely mirrored the recommended EEM approach for non-lethal sampling (Environment Canada 2012). Specifically, the fish population survey targeted the collection of approximately 100 arctic charr from nearshore lake habitat and 100 arctic charr from littoral/profundal lake habitat. The four mine-exposed study lakes used for the fish population survey were the same as those used to document baseline conditions, namely Camp, Sheardown NW, Sheardown SE, and Mary lakes (Figure 2.5). Unlike in previous CREMP studies, a sufficient



LEGEND

Fishing Sampling Location

- Electrofishing
- Gill Net
- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Discharge Line
- Overland Effluent Channel

Mary River Project 2018 CREMP Fish Survey Sampling Locations

0 1.25 2.5 5 km

Map Projection: UTM Zone 17N NAD 1983

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Date: March 2019

Project 187202.0025

minnow environmental inc.

Figure 2.5

Document Path: S:\Projects\187202\187202.0025 - Baffinland 2018 AEMP\4 - GIS\CREMP Study\18-25 Figure 2.5 Fish Survey Sampling Locations.mxd number of arctic charr were captured at Reference Lake 3 nearshore and littoral/profundal areas to statistically evaluate of potential health effects on arctic charr populations at the mine-exposed lakes. Therefore, the 2018 CREMP fish population survey included separate comparison of arctic charr collected at nearshore and littoral profundal habitats in 2018 between the mine-exposed and reference lakes, as well as comparisons of fish captured at nearshore and littoral/profundal zones of individual mine-exposed lakes before-and-after the commencement of the Mary River Project commercial mine operations.

2.4.3.2 Sample Collection

Nearshore areas of the study lakes used for the CREMP study were sampled for arctic charr using a battery powered backpack electrofishing unit (Model LR-24, Smith-Root Inc., Vancouver, WA). An electrofishing team, consisting of the backpack electrofisher operator and a single netter, conducted a single fishing pass at one shoreline reach of each study lake and three to five reaches at each lotic study area. The number of passes conducted at each lake/lotic study area was dependent upon catch success, with more passes required in instances in which target numbers were not cumulatively attained. All fish captured during each pass were retained in buckets of aerated water. At the conclusion of each pass, total fishing effort (i.e., electrofishing seconds) was recorded to allow calculation of time-standardized catch. All captured fish were identified to species and enumerated, with any non-target species subsequently released alive at the area of capture. All captured arctic charr were temporarily retained for processing using methods described below (Section 2.4.3.3). Additional supporting information collected for each electrofishing pass included recording the GPS coordinates at the points of commencement and completion of electrofishing activities and a description of the sampled habitat.

Littoral/profundal areas of the study lakes were sampled for arctic charr using experimental (gang index) gill nets. Multiple-panel, 2 m high gill nets with total lengths ranging from 61 to 91 m (200’ to 300’) and bar mesh sizes ranging from 38 to 76 mm (1.5” to 3”) were set on the bottom for short durations (approximately 0.8 to 3.3 hours per set; mean of 1.9 hours) during daylight hours only. Upon retrieval of each net, all captured fish were identified to species, enumerated, and processed (see below) separately for each individual gill net panel mesh size. For each gill net set, information including mesh sizes, duration of sampling, sampling depth range, GPS coordinates, and habitat descriptions were recorded.

2.4.3.3 Field and Laboratory Processing

Following completion of each electrofishing pass and retrieval of each individual gill net panel, all captured arctic charr were subject to processing in the field. For all live captures, the external condition of each individual was assessed visually for the presence of any deformities, erosions, lesions, and tumors (DELT), in addition to evidence of external and/or internal parasites. All observations were recorded on field sheets, with supporting photographs taken as appropriate. Each fish was then subject to measurement of fork and total length to the nearest millimetre using a standard measuring board. Following length measurements, fish captured using the electrofishing unit were individually weighed to the nearest milligram using an Ohaus Model 123 Scout-Pro analytical balance (Ohaus Corp., Pine Brook, NJ) with a surrounding draft shield. For arctic charr captured in gill nets, individuals were weighed using Pesola™ spring scales (Pesola AG, Baar Switzerland) demarcated at intervals of 1 to 2% of the total scale range and providing accuracy of ±0.3% of the fish mass. The Pesola™ spring scale for individual weight measurement of gill-net captured fish was selected so that the fish weight was near the top of the scale's range to ensure that measurements achieved a resolution near 1%. All live arctic charr captured by electrofishing and gill netting methods that were not selected for the collection of aging structures were released near the location of capture following these individual measurements of length and weight.

As specified for EEM non-lethal fish population surveys (see Environment Canada 2012), approximately 10% of the targeted number of arctic charr captured using electrofishing methods were sacrificed for collection of aging structures. Otoliths were removed from all sacrificed individuals and incidental mortalities for age determination. Upon removal, these aging structures were wrapped in wax paper, placed inside envelopes labelled with the fish identification, and then dried for storage. Age structures (otoliths) were shipped to North Shore Environmental Services (NSES; Thunder Bay, ON) for age determination. At the laboratory, otoliths were prepared for aging using a "crack and burn" method. The prepared otolith samples were mounted on a glass slide using a mounting medium and examined under a compound microscope using transmitted light to determine fish age. For each structure, the age and edge condition was recorded along with a confidence rating for the age determination.

2.4.3.4 Data Analysis

Fish community data from the mine-exposed and reference study areas were compared based on total catch and catch-per-unit-effort (CPUE) for each sampling method. Electrofishing CPUE was calculated as the number of fish captured per electrofishing minute for each lake nearshore or lotic study area, and gill netting CPUE was calculated as the number of fish captured per 100 metre·hours of net used for each study lake. Temporal comparison of fish community assemblage was conducted using electrofishing CPUE and gill netting CPUE to evaluate relative changes in fish catches at mine area lakes between mine baseline and through the years of mine operation.

Arctic charr population health was assessed separately for electrofishing and experimental gill netting data sets. Initial data analysis included the plotting of length frequency distributions so that, together with appropriate aging data, young-of-the-year (YOY) individuals could be distinguished from the older juvenile/adult life stages (electrofishing data set), or various size/age classes could be distinguished from one another (gill netting data set). Where sample sizes allowed, the YOY age class was assessed separately from the older juvenile/adult age classes for fish survey endpoints between the individual mine-exposed lakes and Reference Lake 3. Fish size endpoints of fork length and fresh body weight were summarized by separately reporting mean, median, minimum, maximum, standard deviation, standard error, and sample size by size class (if possible) for each study area. The recorded measurement endpoints were used as the basis for evaluating four response categories (survival, growth, reproduction and energy storage; Table 2.6) according to the procedures outlined for EEM by Environment Canada (2012). Length-frequency distributions were compared between mine-exposed and reference lakes or between lotic study areas for data collected in 2018, and for before-after analysis using data collected in 2018 and the combined baseline period, using a non-parametric two-sample Kolmogorov-Smirnov (KS) test. Potential differences in reproductive success between paired study areas were based on evaluation of the relative proportion of arctic charr YOY between the mine-exposed and reference areas, and by comparing the results of KS tests conducted with and without YOY individuals included in the data sets.

Mean fork length and body weight were compared between mine-exposed and reference study areas in 2018, and between 2018 and the mine baseline period, using ANOVA, with data evaluated for normality and homogeneity of variance before applying parametric statistical procedures. In cases where data did not meet the assumptions of ANOVA despite log-transformation, a non-parametric Mann-Whitney U-test was also performed to test for/validate significant differences between study areas or study periods indicated by the ANOVA tests, as appropriate. Fork length, body weight, and body weight adjusted to a designated fork length, were plotted using boxplots. In the boxplots, the box represents the 25th percentile, median, and 75th percentile values, and the whiskers represent the minimum and maximum values. However, values that were 1.5 times the height of the box beyond the 25th percentile or 75th percentile were plotted as individual points and the whiskers were truncated to the next observation in the dataset.

Body weight at fork length (condition) was compared using Analysis-of-Covariance (ANCOVA). Prior to conducting the ANCOVA tests, scatter plots of all variable and covariate combinations were examined to identify outliers, leverage values or other unusual data. The scatter plots were also examined to ensure there was adequate overlap between the 2018 mine-exposed and reference/mine-exposed baseline data sets, and that there was a linear relationship between the variable and the covariate. In order to verify the existence of a linear relationship, each

Table 2.6: Fish Population Survey Endpoints Examined for the Mary River Project CREMP 2018 Study

Response Category	Endpoint	Statistical Procedure[c,d,e]	Critical Effect Size
Survival	Length-frequency distribution[a]	K-S Test	not applicable
	Age[a,f]	ANOVA	not applicable
Energy Use (size)	Size (fresh body weight)[b]	ANOVA	25%
	Size (fork length)[b]	ANOVA	25%
Energy Use (growth)	Size-at-age (body weight against age)[a,f]	ANCOVA	25%
	Size-at-age (fork length against age)[b,f]	ANCOVA	25%
Energy Use (reproduction)	Relative abundance of YOY (% composition)[b]	K-S Test	not applicable
Energy Storage	Condition (body weight against length)[a]	ANCOVA	10%

[a] Endpoints used for determining "effects" as designated by statistically significant difference between mine-exposed and reference areas (Environment Canada 2012).

[b] These analyses are for informational purposes and significant differences between exposure and reference areas are not necessarily used to designate an effect (Environment Canada 2012).

[c] ANOVA (Analysis of Variance) used except for non-normal data, where Mann Whitney U-test may have been used.

[d] ANCOVA (Analysis of Covariance). For the ANCOVA analyses, the first term in parentheses is the endpoint (dependent variable Y) that is analyzed for an effluent effect. The second term in parentheses is the covariate, X (age, weight, or length).

[e] K-S Test (Kolmogorov-Smirnov test).

[f] Endpoints which were applied to reduced data sets, including sacrificed fish and/or mortalities.

relationship was tested using linear regression analysis by area and evaluated at an alpha level of 0.05. If it was determined that there was no significant linear regression relationship between the variable and covariate for the 2018 mine-exposed and/or reference/mine-exposed baseline data sets, then the ANCOVA was not performed. Once it was determined that ANCOVA could be used for statistical analysis of the data, the first step in the ANCOVA analysis was to test whether the slopes of the regression lines for the 2018 mine-exposed and reference/baseline data sets were equal. This was accomplished by including an interaction term (dependent × covariate) in the ANCOVA model and evaluating if the interaction term was significantly different, in which case the regression slopes would not be equal between data sets and the resulting ANCOVA would provide spurious results. In such cases, two methodologies were employed to assess whether a full ANCOVA could proceed. In order of preference these were: 1) removal of influential points using Cook's distance and re-assessment of equality of slopes; and, 2) Coefficients of Determination that considered slopes equal regardless of an interaction effect (Environment

Canada 2012). For the Coefficients of Determination, the full ANCOVA was completed to test for main effects, and if the r^2 value of both the parallel regression model (interaction term) and full regression model were greater than 0.8 and within 0.02 units in value, the full ANCOVA model was considered valid (Environment Canada 2012). If both methods proved unacceptable, the magnitude of effect was estimated at both the minimum and maximum overlap of covariate variables between areas (Environment Canada 2012). This results in a statistically significant interaction effect (slopes are not equal), but the calculation of the magnitude of difference at the minimum and maximum values of covariate overlap is not assigned statistical difference as it would for a full ANCOVA model. If the interaction term was not significant (i.e., homogeneous slopes between the two populations), then the full ANCOVA model was run without the interaction term to test for differences in adjusted means between the two data sets. The adjusted mean was then used as an estimate of the population mean based on the value of the covariate in the ANCOVA model.

For endpoints showing significant data set differences, the magnitude of difference between 2018 mine-exposed and reference data or the baseline data was calculated as described by Environment Canada (2012) using mean (ANOVA), adjusted mean (ANCOVA with no significant interaction) or predicted values (ANCOVA with significant interaction). The anti-log of the mean, adjusted mean, or predicted value was used in the equations for endpoints that were $\log_{10}$-transformed. In addition, the magnitude of difference for ANCOVA with a significant interaction was calculated for each of the minimum and maximum values of the covariate. If there was no significant difference indicated between data sets, the minimum detectable effect size was calculated as a percent difference from the reference mean/mine-exposed baseline mean for ANOVA or adjusted reference mean/mine-exposed baseline mean for ANCOVA at alpha = beta = 0.10 using the square root of the mean square error (generated during either the ANOVA or ANCOVA procedures) as a measure of variability in the sample population based on formulae provided by Environment Canada (2012). Finally, if outliers or leverage values were observed in a data set (or sets) upon examination of scatter plots and residuals, then the values were removed and ANOVA or ANCOVA tests were repeated and presented only for the reduced data sets. Similar to the Critical Effect Sizes (CES) applied to the benthic invertebrate community survey, a fish population survey CES magnitude of difference of $\pm$ 25% was applied to general endpoints (CES_G) of survival, growth, reproduction and relative liver size, and a magnitude of difference of $\pm$ 10% was applied for condition (CES_C) to define any ecologically relevant differences, consistent with those recommended for EEM (Table 2.6; Munkittrick et al. 2009; Environment Canada 2012).

Finally, an *a priori* power analysis was completed to determine appropriate fish sample sizes for future surveys as recommended by Environment Canada (2012). These analyses were completed based on the mean square error values generated during the ANOVA or ANCOVA procedures and were calculated with alpha and beta set equally at 0.10 for the analysis. Two main assumptions served as the basis for the power analysis. The first assumption was that the fish caught in each of the effluent-exposed and reference areas were representative of the population at large (i.e., similar distribution and variance with respect to the parameters examined). The second assumption was that the characteristics of the populations as a whole would not change substantially prior to the next study. The power analysis results were reported as the minimum sample size (number of fish/area) required to detect a given magnitude of difference (effect size) between the mine-exposed and reference area/baseline populations for each endpoint. The magnitude of difference was presented as a percentage decrease or increase of the reference area/baseline mean for each endpoint as measured during the fish population study using the observed pooled standard deviation of the residuals from the t-test or parallel slope ANCOVA model.

3 CAMP LAKE SYSTEM

3.1 Camp Lake Tributary 1 (CLT1)

3.1.1 Water Quality

Camp Lake Tributary 1 (CLT1) dissolved oxygen (DO) concentrations were consistently at or above saturation at the north branch and main stem stations during all spring, summer, and fall monitoring events (Appendix Tables C.1 to C.3). Dissolved oxygen concentration and percent saturation at the CLT1 north branch and lower main stem stations were higher than at the reference creek, and well above the WQG lowest acceptable concentration for early life stages of cold-water biota (i.e., 9.5 mg/L), at the time of biological sampling in August 2018 (Figure 3.1; Appendix Table C.13). No consistent spatial patterns in pH were shown with progression downstream through the CLT1 north branch (Stations L1-08 to L1-02) and main branch (Stations L2-03 to L0-01) stations during all spring, summer, and fall monitoring events (Appendix Tables C.1 to C.3). Although pH was significantly lower at the CLT1 north branch and lower main stem study areas compared to Unnamed Reference Creek, no significant differences in pH were indicated between the two CLT1 study areas in August 2018, suggesting no substantial influence of the Milne Inlet Tote Road on in-stream pH (Figure 3.1; Appendix Table C.13). The pH at all CLT1 stations/study areas was also consistently within WQG limits, suggesting adverse effects on biota were unlikely as a result of the slight difference in pH between CLT1 and Unnamed Reference Creek.

Specific conductance within CLT1 was generally highest in the upper main stem (Station L2-03) and lowest in the north branch (Stations L1-02 and -08), with intermediate values observed at the lower main stem stations reflecting mixing of these two branches and suggesting a mine-related source affecting water quality of the CLT1 upper main stem (Appendix Tables C.1 to C.3, C.14). Specific conductance was consistently higher at the CLT1 north branch and main stem stations compared to the CREMP lotic reference stations over the spring, summer, and fall sampling events (Appendix Tables C.1 to C.3), and was also significantly higher at the CLT1 study areas compared to Unnamed Reference Creek during the August 2018 biological study (Figure 3.1). In addition, specific conductance was significantly higher at the CLT1 lower main stem than at the north branch (Appendix Table C.13). These results further corroborated the occurrence of a mine-related source affecting water quality of CLT1, primarily in the main stem of the tributary.

Water chemistry of the CLT1 north branch was similar to the reference creek stations in 2018 with the exception of moderately higher (i.e., 5- to 10-fold) sulphate concentrations and slightly higher (i.e., 3- to 5-fold) total molybdenum and potassium concentrations during the spring sampling event (Table 3.1; Appendix Tables C.14 and C.15). Parameter concentrations were below

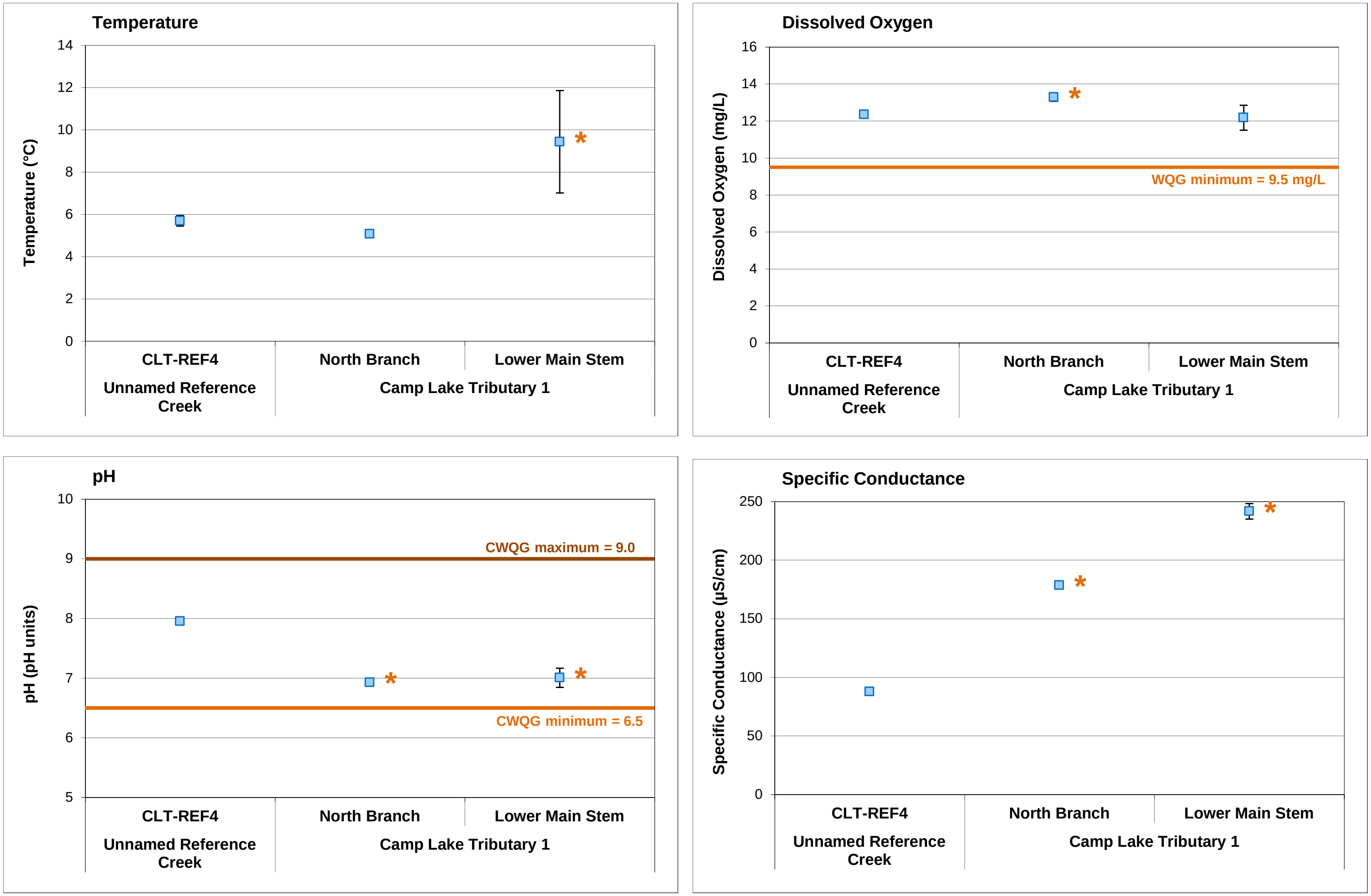


Figure 3.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 1 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2018

Note: An asterisk (*) next to data point indicates mean value differs significantly from the Unnamed Reference Creek mean.

Table 3.1: Water Chemistry at Camp Lake Tributary (CLT) Monitoring Stations During Fall (late August and September) Sampling, Mary River Project CREMP, 2018

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creek Average (n=4)	North Branch CLT1		Upper Main Stem	Lower Main Stem CLT1			CLT-2
						L1-08	L1-02	L2-03	L1-09	L1-05	L0-01	K0-01
					Fall 2018	25-Aug-2018	25-Aug-2018	26-Aug-2018	25-Aug-2018	18-Aug-2018	18-Aug-2018	19-Aug-2018
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	97	107	171	350	218	204	218	195
	pH (lab)	pH	6.5 - 9.0	-	7.88	8.06	8.07	8.03	8.26	8.20	8.19	8.23
	Hardness (as $CaCO_3$)	mg/L	-	-	47	62	89	146	103	104	108	108
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	3.2	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	53	60	76	195	110	90	105	124
	Turbidity	NTU	-	-	2.4	0.3	0.2	5.6	1.1	1.4	1.3	1.8
	Alkalinity (as $CaCO_3$)	mg/L	-	-	43	58	83	122	92	92	90	104
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.021	0.045	<0.020	0.508	0.059	0.106	0.147	<0.020
	Nitrate	mg/L	13	13	<0.020	0.025	0.022	3.530	0.625	0.629	0.571	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	0.0224	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	0.66	<0.15	0.21	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.5	3.0	2.6	5.5	3.4	2.9	2.8	3.6
	Total Organic Carbon	mg/L	-	-	2.0	2.9	3.3	5.9	4.0	4.4	4.1	2.4
	Total Phosphorus	mg/L	0.020^{α}	-	0.0035	<0.0030	<0.0030	0.0067	<0.0030	<0.0030	<0.0030	<0.0030
	Phenols	mg/L	0.004^{α}	-	<0.0010	<0.0010	0.0012	0.0045	<0.0010	<0.0010	<0.0010	0.0011
Anions	Bromide (Br)	`	-	-	0.1	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.6	1.2	1.3	21.7	7.2	7.6	8.1	2.2
	Sulphate (SO_4)	mg/L	218^{β}	218	2.7	4.0	3.8	13.8	5.6	6.2	6.2	4.5
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.054	0.013	0.007	0.108	0.031	0.033	0.035	0.052
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	0.00012	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0059	0.0086	0.0103	0.0144	0.0120	0.0118	0.0100	0.0113
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	0.027	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	<0.000010	0.000035	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	9.8	12.1	18.6	29.1	23.0	21.2	21.2	22.0
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009^{α}	0.0040	<0.00010	<0.00010	<0.00010	0.00031	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.0009	0.0022	0.0021	0.0014	0.0019	0.0019	0.0016	0.0015
	Iron (Fe)	mg/L	0.30	0.326	0.046	<0.030	<0.030	0.314	0.077	0.077	0.073	0.054
	Lead (Pb)	mg/L	0.001	0.001	0.00008	<0.000050	<0.000050	0.00021	<0.000050	0.00005	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	<0.0010	<0.0010	0.0030	0.0015	0.0018	0.0017	0.0014
	Magnesium (Mg)	mg/L	-	-	5.3	7.5	10.8	18.0	12.3	12.3	11.1	13.2
	Manganese (Mn)	mg/L	0.935^{β}	-	0.0007	0.0004	0.0004	0.0250	0.0053	0.0041	0.0032	0.0016
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00029	0.00078	0.00065	0.00300	0.00109	0.00089	0.00082	0.00032
	Nickel (Ni)	mg/L	0.025	0.025	0.00053	<0.00050	0.00064	0.00139	0.00088	0.00087	0.00079	0.00063
	Potassium (K)	mg/L	-	-	0.6	1.8	1.7	3.3	2.0	2.2	1.9	1.5
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.8	0.8	0.8	1.2	0.9	0.9	0.9	1.0
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	1.2	0.5	1.2	13.8	4.0	4.1	3.4	1.7
	Strontium (Sr)	mg/L	-	-	0.0094	0.0078	0.0092	0.0289	0.0202	0.0189	0.0182	0.0127
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.0019	0.0023	0.0018	0.0201	0.0052	0.0051	0.0042	0.0014
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME 1999, 2017) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2017). See Table 2.2 for information regarding WQG criteria.

[b] AEMP Water Quality Benchmarks developed by Intrinsik (2013) using baseline water quality data specific to the Camp Lake tributary system.

(shaded) Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

applicable WQG and watercourse-specific AEMP benchmarks at the CLT1 north branch in 2018 except for copper, which was marginally above only the WQG during the summer and fall sampling events (Table 3.1; Appendix Table C.14). Temporal comparisons indicated that parameter concentrations at the CLT1 north branch in fall 2018 were within the range of those measured during the mine baseline (2005 to 2013) period with the exception of higher total copper concentrations, which were consistently greater at the CLT1 north branch stations in all years of commercial mine production, since 2015, than during mine baseline studies (Figure 3.2; Appendix Figure C.2). Overall, only a minor influence on water quality, reflected mainly by a slight elevation in copper concentrations, was indicated at the CLT1 north branch following the commencement of commercial mine production.

Hardness and concentrations of total dissolved solids (TDS), total ammonia, nitrate, nitrite, total Kjeldahl nitrogen (TKN), chloride, sulphate, and several metals including iron, manganese, molybdenum, potassium, sodium, strontium, and uranium, were slightly to highly elevated (i.e., 3-fold to ≥10-fold higher, respectively) at the upstream-most CLT1 main stem station (L2-03) compared to average reference creek station water chemistry in at least two of the three seasonal sampling events (Table 3.1; Appendix Tables C.14 and C.15). On average, concentrations of nitrate, chloride, and sulphate, as well as total concentrations of manganese, molybdenum, potassium, and sodium, were elevated at the CLT1 lower main stem (i.e., stations L1-09, L1-05 and L0-01) compared to respective average concentrations at the reference creek stations (Appendix Table C.14). Notably, the magnitude of elevation in concentrations of the above parameters compared to the reference creek stations was substantially lower at the lower main stem stations compared to the upper main stem, reflecting the influence of CLT1 main stem dilution from the north branch (Appendix Table C.14).

Within the CLT1 upper main stem (i.e., Station L2-03), total aluminum and iron concentrations were above respective WQG and watercourse-specific AEMP benchmarks during the spring sampling event, and in addition to total uranium and phenol concentrations, were also above WQG during one or both of the summer and fall sampling events in 2018 (Table 3.1; Appendix Table C.14). Total aluminum concentrations were also above WQG and/or AEMP benchmarks at the MRY-REF3 lotic reference station during the spring and summer 2018 sampling events (Appendix Table B.2). Notably, higher turbidity was evident at the CLT1 main stem and MRY-REF 3 lotic reference stations than at the other mine-exposed and reference creek stations, which in turn suggested that elevation in total aluminum and iron concentrations compared to WQG/AEMP benchmarks reflected association of these metals with suspended particulate matter (Appendix Tables B.2 and C.14). This was corroborated by evaluation of the dissolved concentrations of aluminum and iron, which showed similar average concentrations between CLT1 stations and the reference creek stations and suggested that the key source(s) of aluminum

Figure 3.2: Temporal Comparison of Water Chemistry at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for Mine Baseline (2005 to 2013), Construction (2014) and Operational (2015 to 2018) Periods During Fall

Notes: Values represent mean ± SD. Lotic reference stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4). Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.2 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to the Camp Lake Tributaries.

and iron may not be mine-related (Appendix Tables C.4, C.16, and C.17). In contrast, dissolved concentrations of uranium were elevated at the CLT1 upper main stem compared to the lotic reference creek station average despite elevated turbidity, suggesting a mine-related source of uranium to CLT1 (Appendix Table C.17).[6]

Temporal comparisons of CLT1 main stem water chemistry data indicated that, of the parameters shown to be elevated relative to the reference creek stations in 2018, hardness and concentrations of TDS, chloride, and total strontium were comparable to or only slightly higher than concentrations recorded during the baseline period (Figure 3.2; Appendix Figure C.2). However, nitrate, TKN, and sulphate concentrations, as well as total iron, manganese, molybdenum, sodium, and uranium concentrations, were consistently higher during the mine operational years, including 2018, compared to the mine baseline period at the CLT1 upper main stem and at least one of the three CLT1 lower main stem stations (Figure 3.2; Appendix Figure C.2). Higher parameter concentrations at the CLT1 main stem stations following the initiation of commercial mine production potentially reflected blasting/excavating activity (including associated dust generation) at mine quarry QMR2[7], as well as fugitive dust generation from increased truck usage on the Milne Inlet Tote Road, compared to the baseline period. Notably higher concentrations of nitrogen-based compounds (e.g., ammonia, nitrate, nitrite, TKN) in 2018 at CLT1 were consistent with the deposition of explosives residue from the QMR2 quarry. Overall, mine-related influences on water quality of the CLT1 main stem were primarily evidenced by elevated specific conductance and hardness, as well as concentrations of nitrate, TKN, chloride, sulphate, and total metals including manganese, molybdenum, potassium, sodium, and uranium, at the upper main stem, though with the exception of uranium, none were elevated above applicable WQG or AEMP benchmarks. Although aluminum and iron concentrations were elevated above WQG and AEMP benchmarks at CLT1, similar occurrence at the reference areas suggested that these elevations were likely natural.

3.1.2 Phytoplankton

Chlorophyll-a concentrations at the upper-most CLT1 north branch station (Station L1-08) were lower than the average concentration among reference creek stations for spring, summer, and fall sampling events in 2018 (Figure 3.3). However, chlorophyll-a concentrations further downstream at the CLT1 north branch, nearer to the mine (i.e., Station L1-02), were generally

[6] On average, dissolved concentrations of manganese, molybdenum, potassium, sodium, and strontium were also elevated at CLT1 upper and/or lower main stem stations compared to respective averages from the lotic reference creek stations, supporting the analysis of total metal concentrations that suggested a mine-related source of these metals.

[7] The QMR2 quarry is used to provide material for mine infrastructure projects (e.g., road construction).

comparable to reference creek chlorophyll-a concentrations for each individual sampling event, suggesting no marked differences in phytoplankton abundance between the CLT1 north branch and the reference creek stations (Figure 3.3).

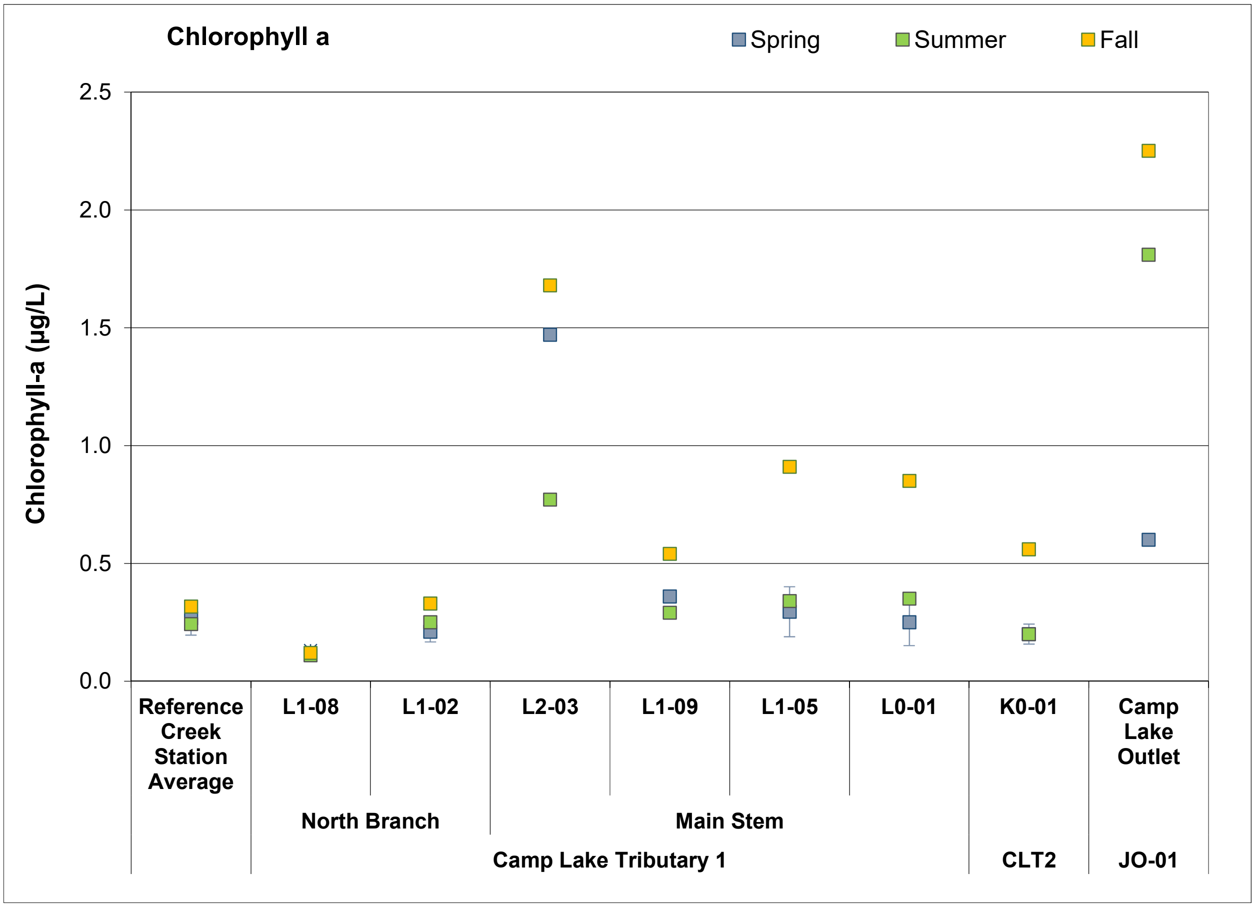


Figure 3.3: Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT1) and Tributary 2 (CLT2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2018

Note: Reference creek data represented by average (± SD; n = 4) calculated from CLT-REF and MRY-REF stations.

Within the CLT1 main stem, chlorophyll-a concentrations were highest at upstream-most Station L2-03 during spring, summer, and fall sampling events in 2018 (Figure 3.3). On average, chlorophyll-a concentrations were significantly higher at the CLT1 main stem stations compared to the reference creek stations during the fall sampling event, but not during the spring and summer sampling events (Appendix Table E.2). Relatively high chlorophyll-a concentrations at Station L2-03 and in the CLT1 lower main stem during fall sampling potentially reflected higher nutrient (e.g., nitrate) concentrations compared to average concentrations under reference conditions (Appendix Tables C.14 and C.15). Nevertheless, chlorophyll-a concentrations at all CLT1 north branch and main stem monitoring stations were well below the AEMP benchmark of 3.7 µg/L for all seasonal sampling events in 2018 (Figure 3.3). Similar to the reference creek stations, chlorophyll-a concentrations observed at all CLT1 stations in 2018 suggested low (i.e., oligotrophic) phytoplankton productivity based on Dodds et al. (1998) trophic status classification for stream environments (i.e., chlorophyll-a < 10 µg/L). This trophic status classification was also consistent with an ultra-oligotrophic to oligotrophic WQG categorization (CCME 2017) for CLT1 based on aqueous total phosphorus concentrations typically less than 10 µg/L at each CLT1 north branch and main stem station during all spring, summer, and fall sampling events (Appendix Table C.14).

Temporal comparisons of the CLT1 chlorophyll-a data indicated that concentrations at the North Branch Stations L1-08 and L1-02 in fall 2018 were similar to, or lower than, those observed in the fall during the baseline period (i.e., 2005 to 2013; Figure 3.4). At the CLT1 main stem, chlorophyll-a concentrations were higher in mine operational years from 2015 to 2018 than during the mine baseline period with the exception of at the CLT1 mouth (Station L0-01; Figure 3.4). In addition, among the years of mine operation, chlorophyll-a concentrations were highest in either 2017 or 2018 at CLT1 lower main stem stations, but nevertheless were continuously lower than the AEMP benchmark of 3.7 µg/L (Figure 3.4). Overall, spatial and temporal analyses of chlorophyll-a concentrations suggested that the mine operation may have contributed to slightly higher phytoplankton abundance at CLT1 main stem stations, but not at the north branch or at the mouth of the main stem, compared to reference conditions. As indicated above, higher phytoplankton abundance within the CLT1 main stem was consistent with the occurrence of higher nutrient concentrations (e.g., nitrate) compared to the reference creeks. This suggested that slightly greater phytoplankton abundance at the CLT1 main stem was the result of current mine operations. Despite slightly greater phytoplankton abundance at CLT1 over time, the watercourse has remained oligotrophic since the commencement of commercial mine operation.

3.1.3 Benthic Invertebrate Community

3.1.3.1 Upstream North Branch (CLT1 US)

Benthic invertebrate density was significantly higher, and Simpson's Evenness significantly lower, at the CLT1 upstream (north branch) study area compared to the Unnamed Reference Creek at magnitudes that were ecologically significant (Table 3.2). Lower evenness at the CLT1 upstream study area reflected disproportionately high densities of a number of Chironomidae (non-biting midges) genera, including *Cricotopus* and *Pseudokieferiella* (Appendix Table F.5). In addition to these differences, differences in benthic invertebrate community assemblage were suggested between study areas based on significant differences in Bray-Curtis Index (Table 3.2). The main differences in dominant benthic invertebrate groups included significantly lower relative

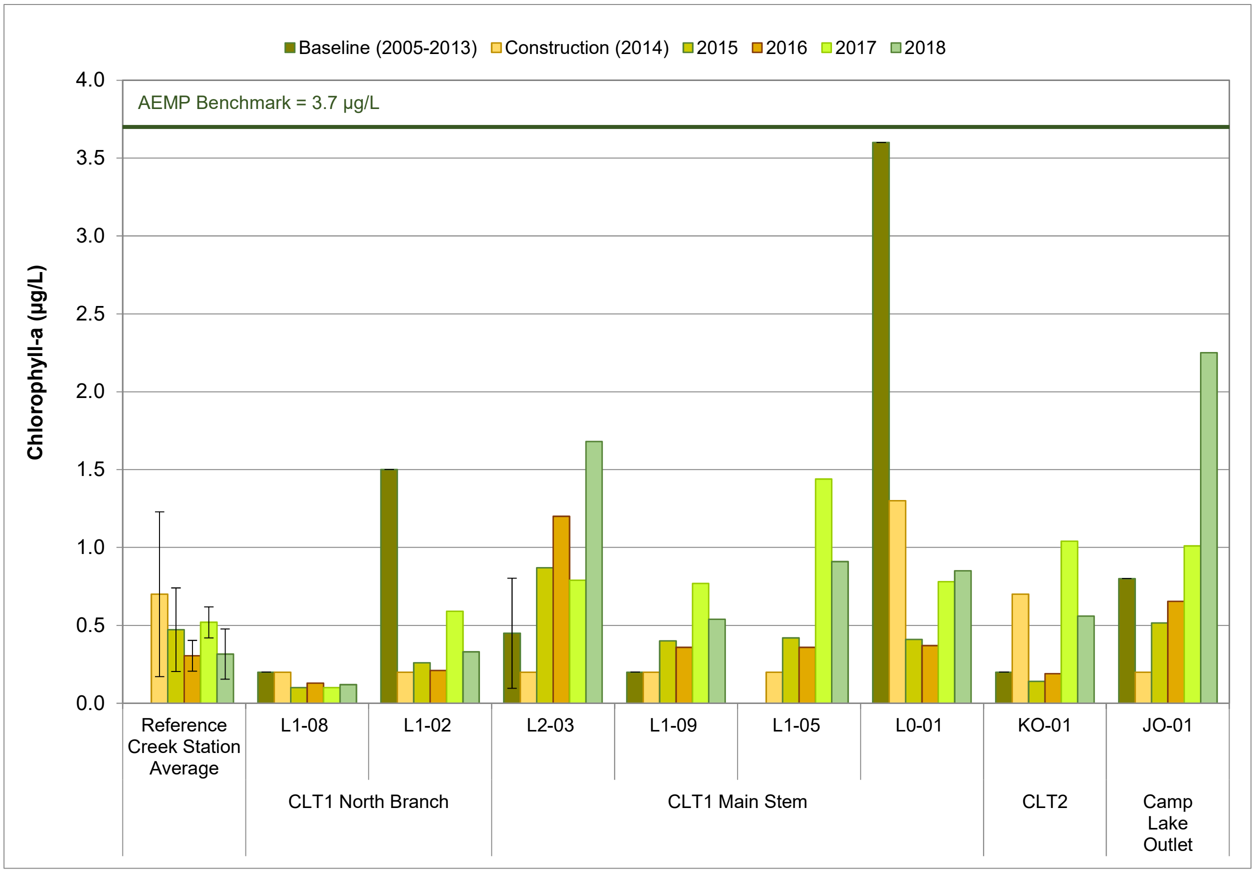


Figure 3.4: Temporal Comparison of Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2018) Periods during Fall

Note: Reference creek data represented by average (± SD; n = 4) calculated from CLT-REF and MRY-REF stations.

Table 3.2: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 1 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2018

Metric	Data Transform-ation	Overall 3-Area Comparison		Pair-wise, *post hoc* comparisons[a]				
		Significant Difference Among Areas?	P-value	Study Area	Mean	Standard Deviation (SD)	Magnitude of Difference (SD)	Pairwise Comparison
Density (No. per m^2)	none	YES	0.008	Reference Creek	526	152	-	a
				CLT1 Upstream	1,410	536	5.8	b
				CLT1 Downstream	787	316	1.7	a
Richness (No. of Taxa)	log	YES	0.003	Reference Creek	19.8	3.0	-	a
				CLT1 Upstream	17.2	1.5	-0.9	a
				CLT1 Downstream	14.4	1.1	-1.8	b
Simpson's Evenness	none	YES	0.007	Reference Creek	0.960	0.011	-	a
				CLT1 Upstream	0.877	0.058	-7.4	b
				CLT1 Downstream	0.856	0.050	-9.3	b
Bray-Curtis Index	none	YES	< 0.001	Reference Creek	0.208	0.092	-	a
				CLT1 Upstream	0.766	0.042	6.1	b
				CLT1 Downstream	0.772	0.035	6.1	b
Nemata (% of community)	square root	YES	0.003	Reference Creek	6.0	2.0	-	a
				CLT1 Upstream	1.6	0.7	-2.3	b
				CLT1 Downstream	2.6	1.8	-1.7	b
Oligochaeta (% of community)	rank	YES	0.017	Reference Creek	0.0	0.0	-	a
				CLT1 Upstream	1.2	0.6	nc	b
				CLT1 Downstream	1.2	1.0	nc	b
Hydracarina (% of community)	log	NO	0.101	Reference Creek	3.3	1.3	-	a
				CLT1 Upstream	5.3	1.3	1.6	a
				CLT1 Downstream	3.6	1.5	0.3	a
Ephemeroptera (% of community)	fourth root	YES	0.006	Reference Creek	5.7	2.1	-	a
				CLT1 Upstream	1.2	1.1	-2.1	b
				CLT1 Downstream	0.4	0.7	-2.5	b
Chironomidae (% of community)	log	YES	< 0.001	Reference Creek	60.4	4.5	-	a
				CLT1 Upstream	86.8	4.2	5.9	b
				CLT1 Downstream	85.9	4.1	5.7	b
Metal Sensitive Chironomids (% of community)	none	YES	0.041	Reference Creek	12.4	4.2	-	a,b
				CLT1 Upstream	17.8	4.8	1.3	a
				CLT1 Downstream	8.6	6.0	-0.9	b
Simuliidae (% of community)	rank	YES	0.008	Reference Creek	15.9	4.6	-	a
				CLT1 Upstream	0.5	0.5	-3.4	b
				CLT1 Downstream	0.4	0.5	-3.4	b
Tipulidae (% of community)	fourth root	NO	0.377	Reference Creek	2.8	1.5	-	a
				CLT1 Upstream	2.9	2.3	0.1	a
				CLT1 Downstream	4.9	3.1	1.4	a
Collector-Gatherer FFG (% of community)	none	YES	0.040	Reference Creek	71.5	4.0	-	a
				CLT1 Upstream	54.6	13.5	-4.2	b
				CLT1 Downstream	59.5	8.3	-3.0	a,b
Filterer FFG (% of community)	square root	YES	< 0.001	Reference Creek	16.0	4.5	-	a
				CLT1 Upstream	0.5	0.5	-3.4	b
				CLT1 Downstream	0.4	0.5	-3.5	b
Shredder FFG (% of community)	log	YES	< 0.001	Reference Creek	4.8	2.8	-	a
				CLT1 Upstream	39.3	13.2	12.5	b
				CLT1 Downstream	35.5	7.7	11.1	b
Clinger HPG (% of community)	fourth root	YES	0.006	Reference Creek	22.6	5.0	-	a
				CLT1 Upstream	42.4	12.7	4.0	b
				CLT1 Downstream	35.6	7.2	2.6	b
Sprawler HPG (% of community)	square root	YES	0.091	Reference Creek	66.2	5.0	-	a
				CLT1 Upstream	51.9	13.9	-2.8	b
				CLT1 Downstream	55.8	6.7	-2.1	a,b
Burrower FFG (% of community)	0.0000	YES	0.028	Reference Creek	11.2	1.9	-	a
				CLT1 Upstream	5.7	2.6	-2.8	b
				CLT1 Downstream	8.6	3.5	-1.3	a,b

Indicates a statistically significant difference for respective comparison (p-value ≤ 0.1).

Blue shaded values indicate significant difference (ANOVA p-value ≤ 0.10) that was also outside of a Critical Effect Size of ±2 SD_{REF}, indicating that the difference between the mine-exposed area and reference area was ecologically meaningful.

[a] *Post hoc* analysis of 1-way ANOVA among all areas protected for multiple comparisons.

abundance of Ephemeroptera (mayflies), Nemata (roundworms), and Simuliidae (blackflies), and conversely, significantly higher relative abundance of Chironomidae (non-biting midges), at the CLT1 north branch compared to the reference creek based on magnitudes of difference outside of the benthic invertebrate community critical effect size (CES_{BIC}) of ±2 reference area standard deviations (SD_{REF}; Table 3.2). Notably, the relative abundance of metal-sensitive chironomids did not differ significantly between the CLT1 north branch and the reference creek (Table 3.2), suggesting that the community composition differences between watercourses were unrelated to differing metal concentrations.

Assessment of benthic invertebrate functional feeding groups (FFG) indicated significantly higher relative abundance of shredders and significantly lower relative abundance of collector-gatherers and filterers at the CLT1 north branch compared to Unnamed Reference Creek (Table 3.2). The differences in FFG composition potentially reflected differences in the type or amount of in-stream vegetation between watercourses. For instance, a greater density of shredders (including *Cricotopus* midges) at the CLT1 north branch may have reflected greater abundance of bryophytes, which serve as a food source for shedders, compared to the reference creek where greater abundance of periphyton may have contributed to a greater relative abundance of collector-gatherer and filterer FFG (Table 3.2; Appendix Table F.1). Collectively, the data suggested that the differences in benthic invertebrate community assemblage between the CLT1 north branch and Unnamed Reference Creek were unrelated to metal concentrations, and likely reflected differences in the types and/or abundance of in-stream vegetation between these study areas.

Temporal comparisons of the CLT1 north branch benthic invertebrate community data indicated that density, Simpson's Evenness, and the relative abundance of key dominant taxonomic groups and FFG did not show any consistent type and/or direction of significant differences for years of mine operation, including 2018, compared to baseline data collected in both 2007 and 2011 (Figure 3.5; Appendix Tables F.7 and F.8). Notably, richness was the only endpoint that differed significantly in years of mine operation (2017 and 2018 only) compared to both years in which baseline data were collected (i.e., 2007 and 2011), but because higher richness was indicated at the CLT1 north branch in 2017 and 2018, this difference was not consistent with an influence typically associated with mine operation (Figure 3.5; Appendix Tables F.7 and F.8). Overall, the temporal evaluation indicated no adverse mine-related influences on benthic invertebrates of the CLT1 north branch since the commencement of commercial mine operations in 2015.

3.1.3.2 Downstream Lower Main Stem (CLT1 DS)

The benthic invertebrate community at the lower main stem of Camp Lake Tributary (CLT1 DS), downstream of the Tote Road crossing, showed significantly lower richness and Simpson's

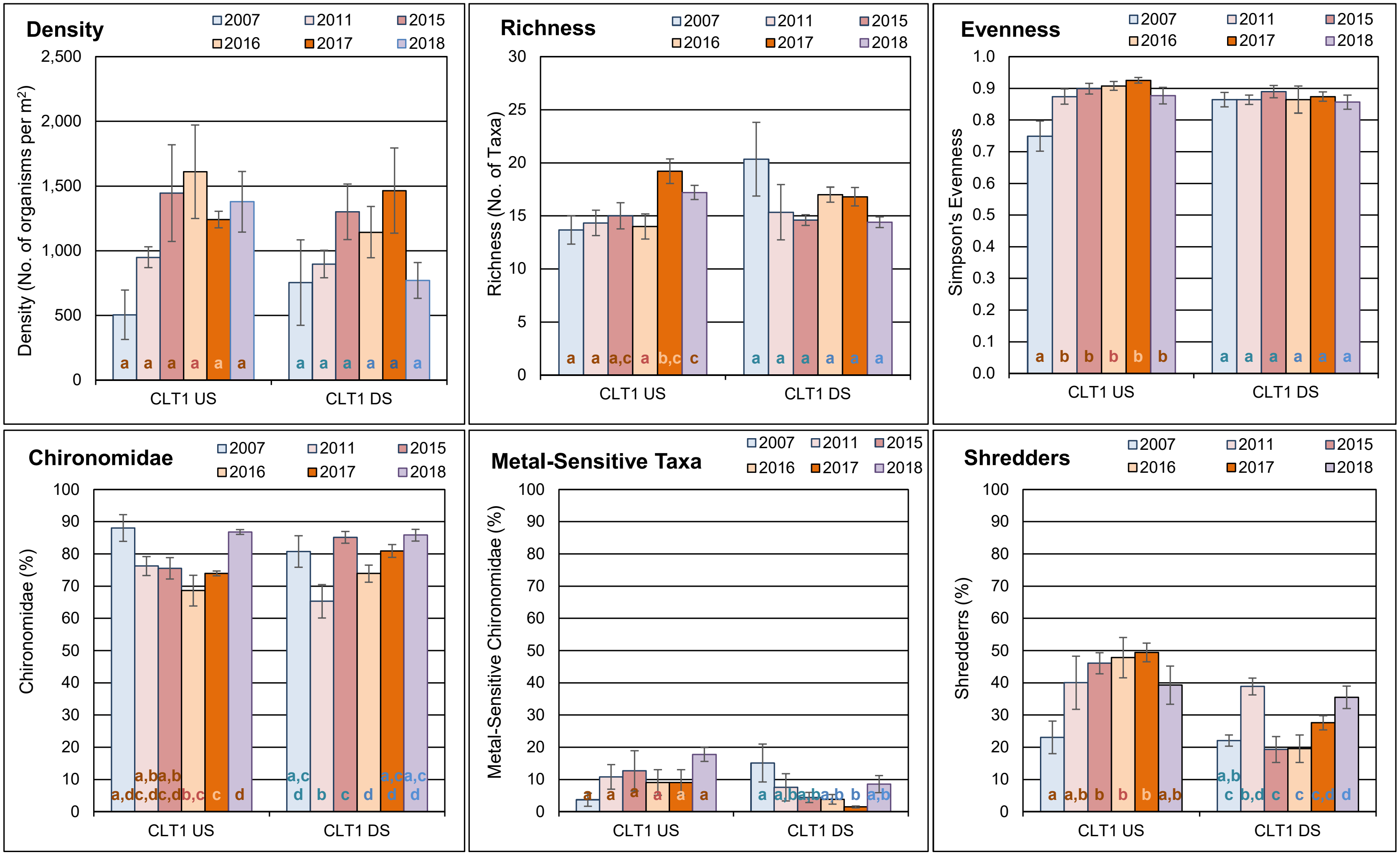


Figure 3.5: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Camp Lake Tributary 1 Study Areas among Mine Baseline (2007, 2011) and Operational (2015 to 2018) Periods

Note: The same like-coloured letter inside bars indicates no significant difference between/among study years for respective community endpoint.

Evenness compared to Unnamed Reference Creek in 2018 (Table 3.2). In addition, the benthic invertebrate community assemblage at the CLT1 lower main stem differed from the reference creek as suggested by significant differences in Bray-Curtis Index and composition of dominant taxonomic groups and FFG (Table 3.2). Because no significant difference in the relative abundance of metal-sensitive chironomids was indicated between the CLT1 lower main stem and reference area (Table 3.2), the community composition differences between these study areas did not appear to be related to differences in metal concentrations. The key differences in benthic invertebrate dominant group and FFG composition between the CLT1 lower main stem and reference creek study areas were very similar to those shown between the CLT1 north branch and reference creek study areas, suggesting a similar mechanism for the differences in benthic invertebrate community composition at the CLT1 north branch and lower main stem study areas compared to the reference creek. Specifically, the differences in benthic invertebrate community composition between the CLT1 lower main stem and reference area likely reflected higher and lower abundance of bryophytes and periphyton, respectively, at CLT1 (Appendix Table F.1). No significant differences in water depth or substrate embeddedness were shown between CLT1 and Unnamed Reference Creek study areas (Appendix Tables F.2 and F.3) and therefore differing types and/or abundance of in-stream vegetation was the most plausible habitat variable contributing to differences in the benthic invertebrate community between CLT1 and the reference creek.

Benthic invertebrate density, richness, and the relative abundance of metal-sensitive chironomids were significantly lower at the CLT1 lower main stem compared to north branch study areas (Table 3.2), but the magnitude of these differences were within a CES_{BIC} of ±2 SD of the north branch mean indicating that these differences were not ecologically meaningful. Other metrics, including Simpson's Evenness, the relative abundance of all dominant groups except metal-sensitive chironomids, all FFG, and all habitat preference groups (HPG), did not differ significantly between the lower main stem and north branch study areas of CLT1 (Table 3.2). In turn, this indicated no substantial influences to the benthic invertebrate community of CLT1 associated with the Tote Road crossing (BG01).

Temporal comparison of the CLT1 lower main stem data indicated no significant differences in benthic invertebrate density, richness, Simpson's Evenness, or the proportion of metal-sensitive chironomids during individual years of mine operation (2015 to 2018) compared to both years in which mine baseline data were collected (i.e., 2007 and 2011; Figure 3.5; Appendix Tables F.9 and F.10). In addition, no consistent types and/or direction of differences in the relative abundance of dominant groups or FFG were indicated between 2018 and years in which baseline data were collected at the CLT1 lower main stem (Figure 3.5; Appendix Tables F.9 and F.10).

Overall, these results suggested no substantial changes in benthic invertebrate community features between the mine operational and mine baseline periods at the CLT1 lower main stem.

3.1.4 Integrated Summary

3.1.4.1 Upstream North Branch (CLT1 US)

Potential mine-related effects on water quality of the CLT1 north branch in 2018 included elevated molybdenum, potassium, and sulphate concentrations compared to average concentrations at the reference creek, but only during the spring sampling event. Although total copper concentrations were not particularly elevated at the CLT1 north branch compared to reference conditions, concentrations at the CLT1 north branch were marginally above WQG in the summer and fall of 2018. Total copper concentrations at the CLT1 north branch were also consistently elevated in each of the four years of commercial mine operation (2015 to 2018) compared to concentrations shown during mine baseline studies, indicating a mine-related source of copper to the CLT1 north branch. No substantial mine development has occurred in the CLT1 north branch watershed, and therefore sources of copper, molybdenum, potassium, and sulphate to the watercourse potentially included fugitive dust from the mine and/or natural minerology of the bedrock/overburden in the region of the mine.

Chlorophyll-a concentrations (a surrogate for phytoplankton abundance) at the CLT1 north branch were comparable to concentrations at the reference creek stations in 2018, and to concentrations recorded at the north branch during mine baseline studies. Chlorophyll-a concentrations at the CLT1 north branch were also consistently well below the AEMP benchmark in 2018, and were indicative of oligotrophic conditions typical of Arctic watercourses. Benthic invertebrate density was significantly greater, and Simpson's Evenness significantly lower, at the CLT1 north branch compared to the reference creek in 2018. However, these differences appeared to be related to differing habitat conditions between watercourses that included greater amounts of in-stream vegetation at the CLT1 north branch. This was supported by no ecologically significant differences in the relative abundance of metal-sensitive chironomids between the CLT1 north branch and reference creek in 2018, and by no significant differences in density, Simpson's Evenness, and relative abundance of any dominant taxonomic groups or FFG between mine operational (2015 to 2018) and baseline periods. Therefore, despite total copper concentrations above WQG, no adverse effects on phytoplankton and benthic invertebrates were indicated at the CLT1 north branch since the commencement of commercial mine operations in 2015.

3.1.4.2 Downstream Main Stem (CLT1 DS)

At the CLT1 main stem, mine-related influences on water quality were evident as elevated conductivity, hardness, and concentrations of chloride, nitrate, sulphate, TKN, and total metals including manganese, molybdenum, potassium, sodium, and uranium, based on comparisons to reference creek water quality data and to CLT1 main stem baseline study data. Of these, uranium was the only parameter observed at concentrations elevated above WQG and AEMP benchmarks specific to the Camp Lake Tributaries that appeared to be related to the mine operations. The occurrence of higher parameter concentrations at the CLT1 main stem stations since the initiation of commercial mine production was likely mainly attributable to blasting/excavating activity (including associated dust generation) at mine quarry QMR2, but also to fugitive dust generation from increased truck usage on the Project roads since the mine baseline period.

Despite evidence of continued mine-related influence on water quality of the CLT1 main stem, including nitrate and TKN concentrations, chlorophyll-a concentrations at the CLT1 main stem were generally higher than at the reference creek in 2018, and within the CLT1 main stem, were also higher in all years of mine operation from 2015 to 2018 than during the mine baseline period. The occurrence of relatively high chlorophyll-a concentrations at the CLT1 main stem not only suggested that metal concentrations including uranium were not highly bioavailable to phytoplankton, but that elevated nitrate/TKN concentrations may have contributed to a slight biological enrichment of the watercourse. Nevertheless, chlorophyll-a concentrations at the CLT1 main stem were well below the AEMP benchmark and were reflective of oligotrophic conditions typical of Arctic watercourses. Although benthic invertebrate community richness, Simpson's Evenness, and general composition differed significantly between the CLT1 lower main stem and Unnamed Reference Creek communities in 2018, the weight-of-evidence indicated that natural differences in in-stream bryophyte (moss) growth between watercourses largely accounted for these differences. This was supported by no ecologically significant differences in relative abundance of metal-sensitive chironomids between the CLT1 main stem and reference creek benthic invertebrate communities in 2018, and by no consistent type and/or direction of differences in benthic invertebrate community endpoints between mine operational (2015 to 2018) and baseline studies. Notably, no ecologically significant differences in benthic invertebrate community endpoints were indicated between the CLT1 north branch (upstream) and main stem (downstream) study areas, suggesting no substantial influences to the benthic invertebrate community of CLT1 related to the Milne Inlet Tote Road water crossing (BG01). Overall, no adverse mine-related effects to phytoplankton or benthic invertebrates were indicated within the CLT1 lower main stem since the commencement of commercial mine operation in 2015.

3.2 Camp Lake Tributary 2 (CLT2)

3.2.1 Water Quality

Camp Lake Tributary 2 (CLT2) DO saturation levels were consistently high at Station KO-01 in 2018, and were similar to mean DO saturation observed among the reference creek stations during all seasonal sampling events (Appendix Tables C.1 to C.3). *In situ* DO concentrations were higher at the CLT2 upstream and downstream study areas than at Unnamed Reference Creek, and were well above the WQG lowest acceptable concentration for the protection of sensitive stages of cold-water biota, at the time of biological sampling in August 2018 (Figure 3.6). Aqueous pH at the CLT2 upstream and downstream study areas was generally slightly higher (i.e., more alkaline) than at the reference creeks, but consistently well within WQG limits during the spring, summer, and fall water sampling events (Appendix Tables C.1 to C.3). However, during biological sampling in August 2018, pH at the CLT2 study areas was significantly lower than at Unnamed Reference Creek, and near the WQG lower limit (Figure 3.6). No significant difference in pH was indicated between CLT2 study areas located upstream and downstream of the Milne Inlet Tote Road suggesting that this road crossing did not markedly influence pH of CLT2 (Figure 3.6). *In situ* specific conductance was significantly higher at CLT2 compared to Unnamed Reference Creek, and was also significantly higher downstream compared to upstream of the Milne Inlet Tote Road at CLT2 during August 2018 biological sampling (Figure 3.6; Appendix Table C.19), suggesting a Tote Road related influence on water quality at CLT2.

The CLT2 (Station KO-01) water chemistry exhibited highly elevated (i.e., ≥10-fold) sulphate concentrations and slightly elevated (i.e., 3- to 5-fold) hardness, alkalinity, and concentrations of TDS and potassium compared to the average from reference creek stations during the spring 2018 sampling event (Appendix Tables B.2, C.14, and C.15). However, water chemistry was similar at CLT2 and the reference creek stations during the summer and fall sampling events in 2018 with the exception of only a moderate elevation (i.e., 5- to 10-fold) in the concentration of sulphate at CLT2 in summer 2018 (Appendix Table C.15). Despite elevation of the parameters indicated above at CLT2, aqueous concentrations of all parameters, including sulphate, were consistently well below WQG and AEMP benchmarks at the CLT2 monitoring station in 2018 (Table 3.1; Appendix Table C.14). Temporal comparisons of CLT2 water chemistry data indicated that conductivity and all parameter concentrations in fall 2018 were generally within the range of those that occurred during the mine baseline period (2005 to 2013; Appendix Tables C.15 and C.18) and within the range of those shown over the 2015 to 2017 mine operation period (Figure 3.2; Appendix Figure C.2). In consideration of all spatial and temporal data, the 2018 water chemistry data suggested only a minor mine-related influence on aqueous conductivity within the CLT2 system compared to applicable reference and mine baseline conditions.

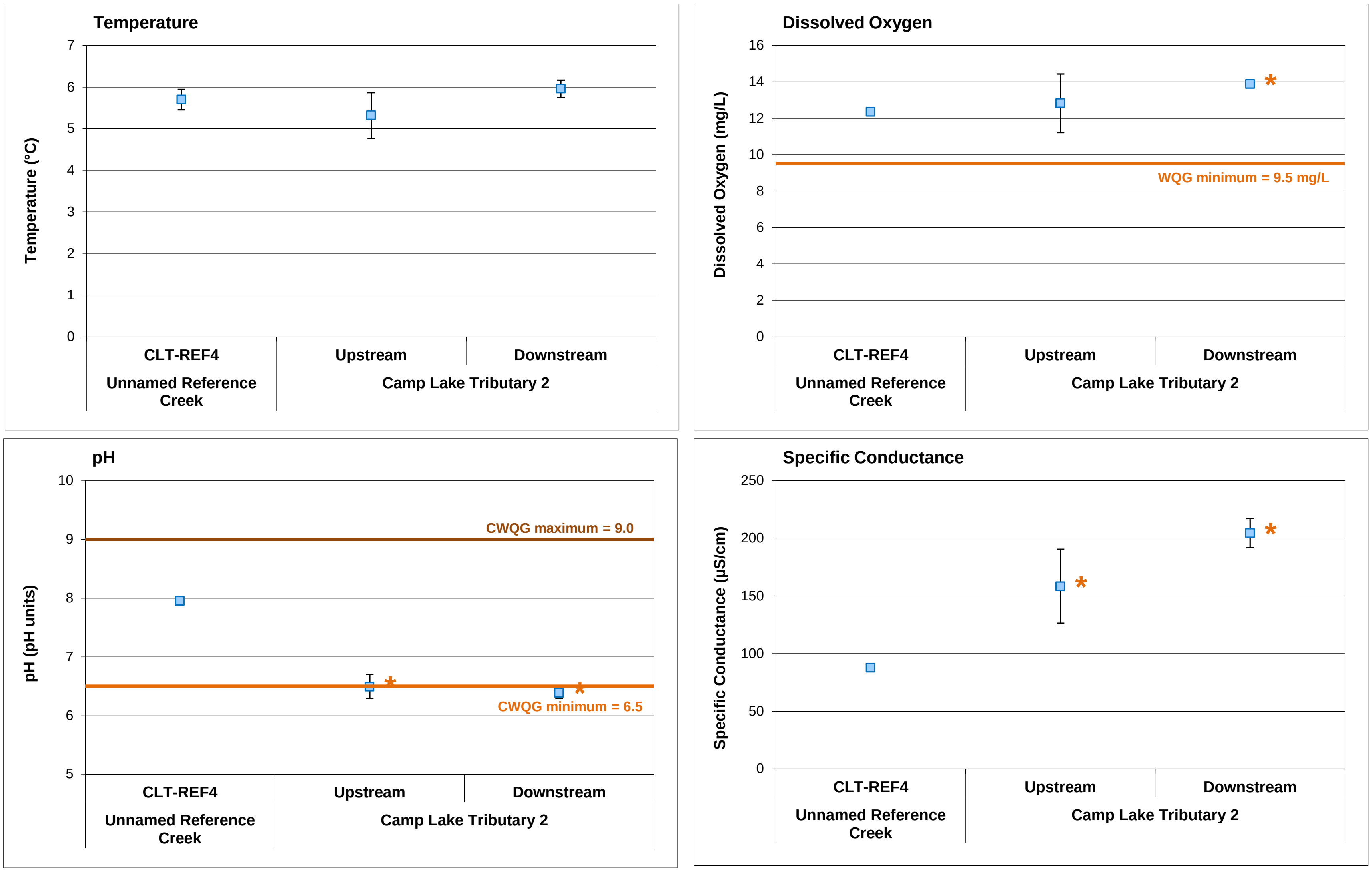


Figure 3.6: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 2 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2018

Note: An asterisk (*) next to data point indicates mean value differs significantly from the Unnamed Reference Creek mean.

3.2.2 Phytoplankton

Chlorophyll-a concentrations at CLT2 (Station KO-01) were slightly lower than average concentrations observed at the reference creeks during spring and summer sampling events, but higher than concentrations at the reference creeks during the fall sampling event in 2018 (Figure 3.3). Concentrations of nutrients, including total ammonia, nitrate, and total phosphorus, showed no marked differences between CLT2 and the reference creek stations during the fall sampling event (Appendix Tables C.14 and C.15), and therefore the occurrence of higher chlorophyll-a concentrations within lower CLT2 in fall 2018 did not appear to be related to a nutrient enrichment influence. Notably, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L during each of the 2018 sampling events at CLT2. Low phytoplankton productivity, indicative of oligotrophic conditions, was also suggested at CLT2 based on comparison of chlorophyll-a concentrations to Dodds et al (1998) trophic status classification for creek environments. This productivity classification was supported by a WQG categorization of ultra-oligotrophic to oligotrophic based on mean aqueous total phosphorus concentrations below 10 µg/L at CLT2 during all spring, summer, and fall sampling events (Table 3.1; Appendix Table C.14). Temporal comparisons indicated higher chlorophyll-a concentrations in 2018 compared to the mine baseline period and, similar to the CLT1 lower main stem, higher chlorophyll-a concentrations in 2017 and 2018 compared to the two previous years of commercial mine operation, at lower CLT2 during fall sampling (Figure 3.4). For the reasons indicated above, higher chlorophyll-a concentrations at CLT2 in fall 2018 did not appear to be associated with a mine-related change in nutrient concentrations over time, and thus may have simply reflected natural seasonal/temporal variation in chlorophyll-a concentrations

3.2.3 Benthic Invertebrate Community

At Camp Lake Tributary 2 (CLT2), sampling was conducted upstream and downstream of the Tote Road (areas CLT2 US and CLT2 DS, respectively) to assess potential mine-related influences to the benthic invertebrate community. Benthic invertebrate density and richness were significantly lower at the CLT2 study areas compared to Unnamed Reference Creek, the differences of which were generally ecologically meaningful (Table 3.3). Differences in community composition were also indicated between the CLT2 and Unnamed Reference Creek study areas based on significantly higher Bray-Curtis Index at both CLT2 study areas (Table 3.3). Similar to CLT1, the key differences in community dominant group composition included significantly lower relative abundance of mayflies, roundworms, and blackflies, and conversely, a significantly higher relative abundance of Chironomidae, at the CLT2 study areas compared to

Table 3.3: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 2 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2018

Metric	Data Transform-ation	Overall 3-Area Comparison		Pair-wise, *post hoc* comparisons[a]				
		Significant Difference Among Areas?	P-value	Study Area	Mean	Standard Deviation (SD)	Magnitude of Difference (REF_{SD})	Pairwise Comparison
Density (No. per m²)	fourth root	YES	< 0.001	Reference Creek	526	152	-	a
				CLT2 Upstream	169	66	-2.3	b
				CLT2 Downstream	128	53	-2.6	b
Richness (No. of Taxa)	none	YES	0.002	Reference Creek	19.8	3.0	-	a
				CLT2 Upstream	15.0	3.7	-1.6	b
				CLT2 Downstream	11.2	1.9	-2.8	b
Simpson's Evenness	rank	NO	0.185	Reference Creek	0.960	0.011	-	a
				CLT2 Upstream	0.923	0.046	-3.3	a
				CLT2 Downstream	0.904	0.091	-5.0	a
Bray-Curtis Index	fourth root	YES	< 0.001	Reference Creek	0.208	0.092	-	a
				CLT2 Upstream	0.661	0.084	4.9	b
				CLT2 Downstream	0.713	0.093	5.5	b
Nemata (% of community)	square root	YES	0.028	Reference Creek	6.0	2.0	-	a
				CLT2 Upstream	1.1	1.6	-2.5	b
				CLT2 Downstream	2.7	3.0	-1.7	a,b
Oligochaeta (% of community)	rank	NO	0.266	Reference Creek	0.0	0.0	-	a
				CLT2 Upstream	2.4	3.4	-2.7	a
				CLT2 Downstream	1.1	2.5	-1.9	a
Hydracarina (% of community)	square root	NO	0.808	Reference Creek	3.3	1.3	-	a
				CLT2 Upstream	4.7	4.4	1.1	a
				CLT2 Downstream	3.5	4.0	0.2	a
Ephemeroptera (% of community)	rank	YES	0.001	Reference Creek	5.7	2.1	-	a
				CLT2 Upstream	0.0	0.0	-2.7	b
				CLT2 Downstream	0.0	0.0	-2.7	b
Chironomidae (% of community)	log	YES	< 0.001	Reference Creek	60.4	4.5	-	a
				CLT2 Upstream	80.2	3.9	4.4	b
				CLT2 Downstream	84.1	6.0	5.3	b
Metal Sensitive Chironomids (% of community)	square root	NO	0.364	Reference Creek	12.4	4.2	-	a
				CLT2 Upstream	9.5	5.0	-0.7	a
				CLT2 Downstream	8.0	7.0	-1.0	a
Simuliidae (% of community)	none	YES	< 0.001	Reference Creek	15.9	4.6	-	a
				CLT2 Upstream	6.6	5.5	-2.0	b
				CLT2 Downstream	0.9	2.1	-3.3	b
Tipulidae (% of community)	none	NO	0.908	Reference Creek	2.8	1.5	-	a
				CLT2 Upstream	2.3	1.5	-0.3	a
				CLT2 Downstream	2.6	2.4	-0.1	a
Collector-Gatherer FFG (% of community)	none	YES	0.030	Reference Creek	71.5	4.0	-	a
				CLT2 Upstream	73.2	8.6	0.4	a
				CLT2 Downstream	83.4	6.5	3.0	b
Filterer FFG (% of community)	none	YES	< 0.001	Reference Creek	16.0	4.5	-	a
				CLT2 Upstream	6.6	5.5	-2.1	b
				CLT2 Downstream	0.9	2.1	-3.3	b
Shredder FFG (% of community)	none	YES	< 0.095	Reference Creek	4.8	2.8	-	a
				CLT2 Upstream	12.9	5.3	2.9	b
				CLT2 Downstream	7.7	7.3	1.0	a,b
Clinger HPG (% of community)	square root	YES	0.036	Reference Creek	22.6	5.0	-	a
				CLT2 Upstream	24.2	8.2	0.3	a
				CLT2 Downstream	12.5	6.9	-2.0	b
Sprawler HPG (% of community)	none	YES	0.023	Reference Creek	66.2	5.0	-	a
				CLT2 Upstream	69.6	6.5	0.7	a
				CLT2 Downstream	79.5	8.4	2.6	b
Burrower FFG (% of community)	log	YES	0.026	Reference Creek	11.2	1.9	-	a
				CLT2 Upstream	6.2	2.5	-2.6	b
				CLT2 Downstream	8.0	2.8	-1.7	a,b

Indicates a statistically significant difference for respective comparison (p-value ≤ 0.1).

Blue shaded values indicate significant difference (ANOVA p-value ≤ 0.10) that was also outside of a Critical Effect Size of ±2 SD_{REF}, indicating that the difference between the mine-exposed area and reference area was ecologically meaningful.

[a] *Post hoc* analysis of 1-way ANOVA among all areas protected for multiple comparisons.

the reference creek based on magnitudes of difference outside of the CES_{BIC} of ±2 SD_{REF} (Table 3.3). However, no significant difference in the relative abundance of metal-sensitive chironomids was indicated between the CLT2 and reference creek study areas (Table 3.3), suggesting that the community composition differences between watercourses were unlikely related to metal concentrations. Notably, no significant differences in density, richness, Simpson's Evenness, or the relative abundance of dominant invertebrate groups were indicated between the CLT2 upstream and downstream study areas, indicating no substantial influences to the benthic invertebrate community of CLT2 associated with the Tote Road water crossing (CV225; Table 3.3).

Temporal comparisons indicated no consistent ecologically significant differences in any benthic invertebrate community endpoints at the CLT2 upstream and downstream study areas during years of mine operation (2015 to 2018) compared to 2007 baseline data with the exception of Simpson's Evenness (Figure 3.7; Appendix Tables F.14 and F.15). Because high Simpson's Evenness is normally associated with a diverse, healthy benthic invertebrate community, the occurrence of significantly higher Simpson's Evenness at the CLT2 upstream study area from 2015 to 2018 compared to 2007 was not consistent with an adverse influence related to recent mine operations. In turn, this suggested no adverse mine-related influences on the benthic invertebrate community of CLT2 since the commencement of commercial mine operations in 2015.

3.2.4 Integrated Summary

Potential mine-related effects on water quality of CLT2 in 2018 included slightly elevated conductivity and sulphate concentrations compared to average reference area conditions. However, water chemistry at CLT2 was comparable between 2018 and the mine baseline period, suggesting that natural regional variability in water chemistry among lotic environments likely accounted for differing conductivity and sulphate concentrations between CLT2 and the reference creek stations. Aqueous concentrations of all parameters were consistently well below applicable WQG and site-specific AEMP benchmarks at CLT2 in all years of mine operation from 2015 to 2018.

Chlorophyll-a concentrations at CLT2 varied seasonally from those observed at reference creek stations in 2018, but were consistently well below the AEMP benchmark and were indicative of oligotrophic conditions characteristic of Arctic watercourses. Although chlorophyll-a concentrations at CLT2 were higher in 2018 than during the mine baseline period, nutrient concentrations within CLT2 were comparable between 2018 and the mine baseline studies. This suggested that the differences in chlorophyll-a concentrations between 2018 and the baseline studies likely reflected natural seasonal/temporal variation. The benthic invertebrate community

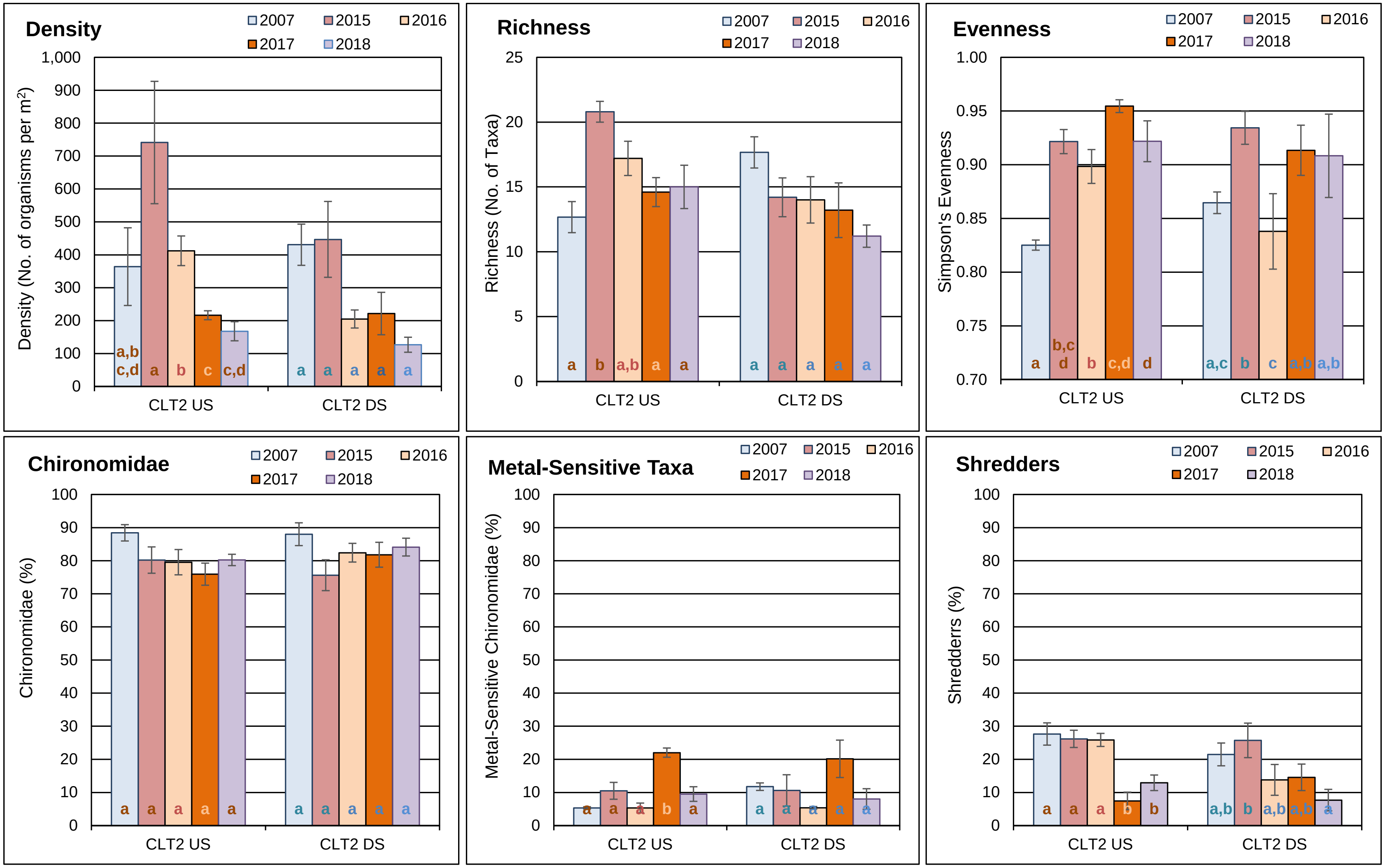


Figure 3.7: **Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Camp Lake Tributary 2 Study Areas among Mine Baseline (2007) and Operational (2015 to 2018) Periods**

Note: The same like-coloured letter inside bars indicates no significant difference between/among study years for respective community endpoint.

of CLT2 exhibited significantly lower density and richness, and significantly different composition, than Unnamed Reference Creek in 2018. However, no significant difference in the relative abundance of metal-sensitive chironomids was indicated at CLT2 compared to the reference creek in 2018. In addition, no ecologically significant differences in any benthic invertebrate community endpoints were consistently indicated between the mine operational and baseline studies at CLT2 with the exception of higher Simpson's Evenness following commencement of commercial mine operation. Because high Simpson's Evenness is normally associated with a more diverse, healthy benthic invertebrate community, the occurrence of significantly higher Simpson's Evenness at the CLT2 in years of mine operation was not indicative of an adverse influence related to the mine. Notably, no significant differences in benthic invertebrate community endpoints occurred between the CLT2 upstream and downstream study areas, indicating no substantial influences to the benthic invertebrate community of CLT2 associated with the Tote Road water crossing (CV225). Overall, similar to the findings of the three previous CREMP studies, the chlorophyll-a and benthic invertebrate community data indicated no adverse mine-related effects to biota of CLT2 since commercial mine operations commenced in 2015.

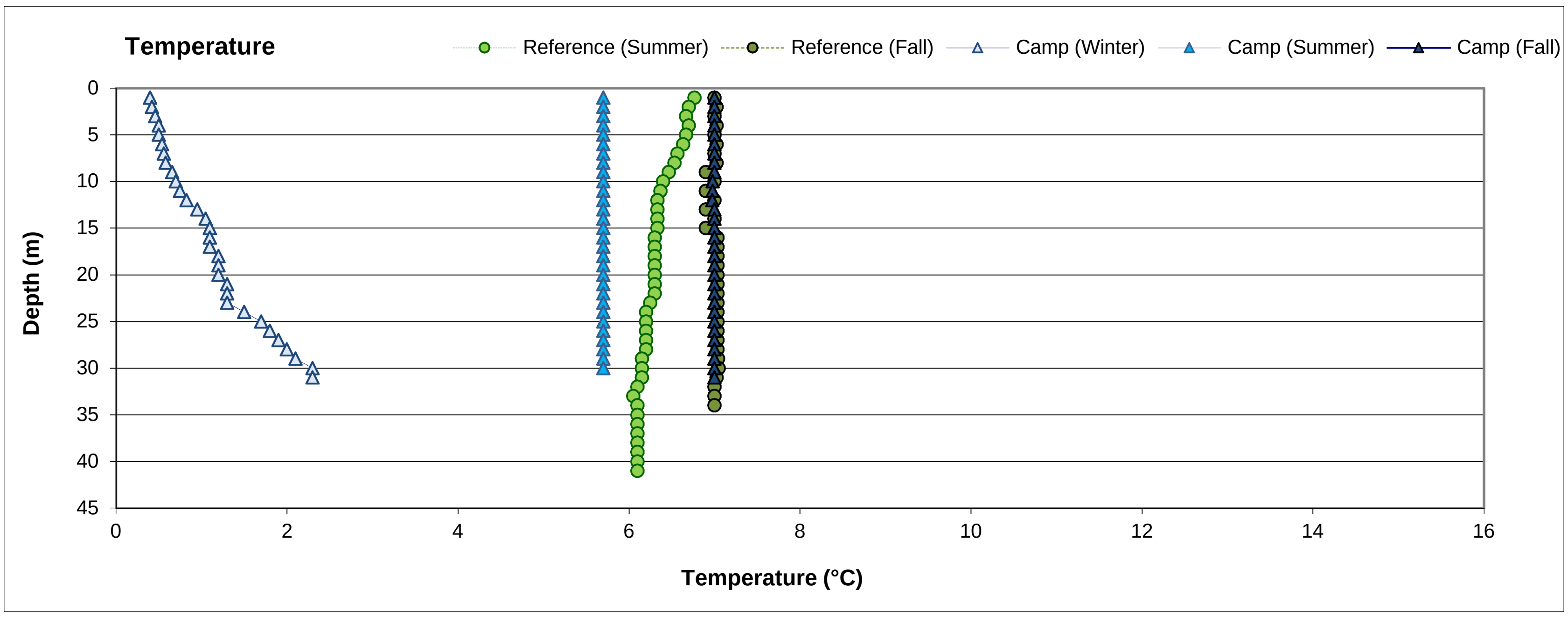
3.3 Camp Lake (JLO)

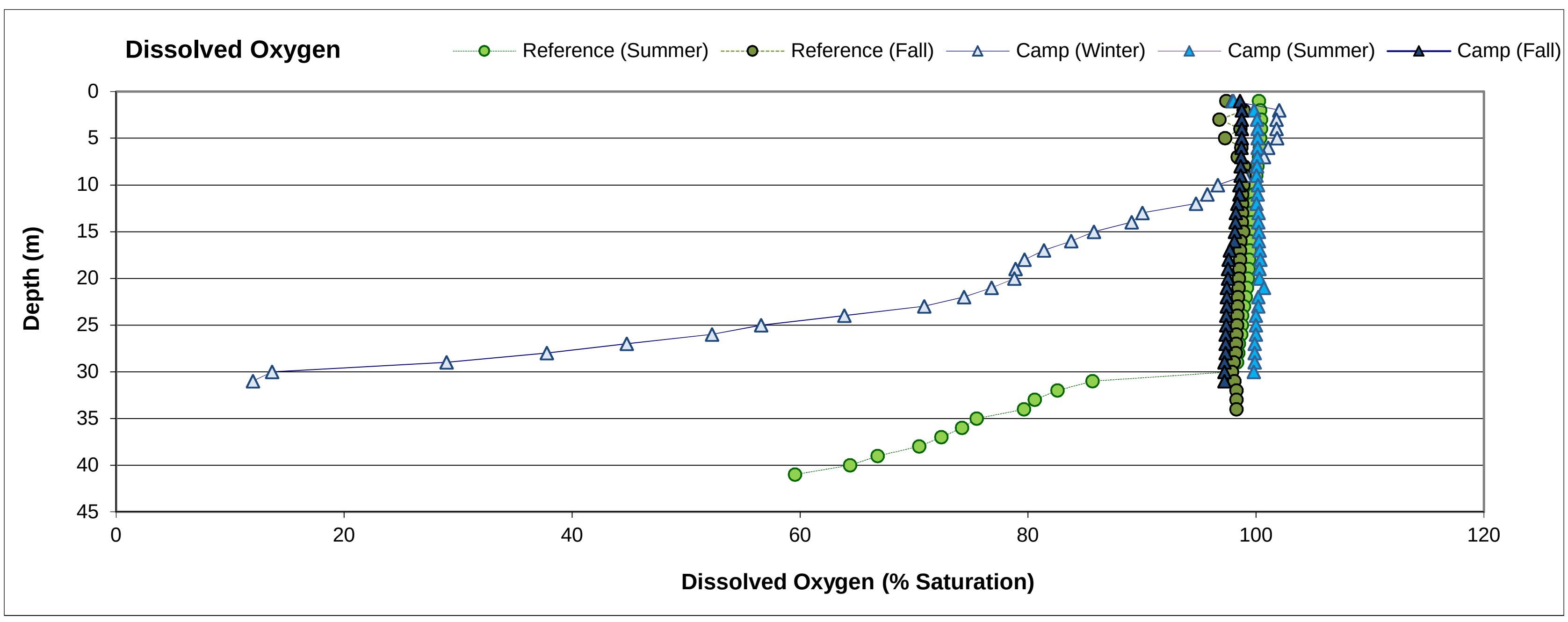
3.3.1 Hydraulic Retention Time

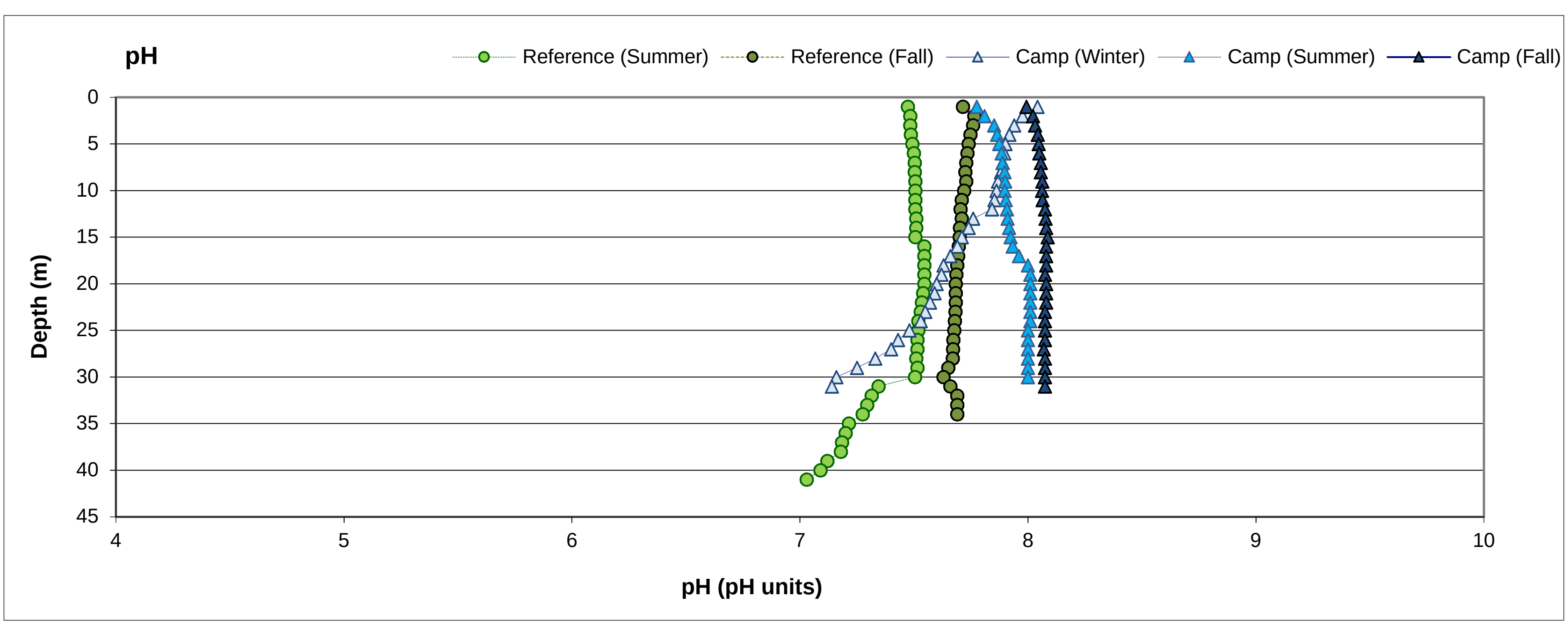
A hydraulic retention time of 416 ± 184 days was estimated by Minnow (2018) for Camp Lake using mean annual watershed runoff extrapolated from CLT1 and CLT2 flow monitoring stations and a lake volume of 27.5 million cubic metres.

3.3.2 Water Quality

In situ water quality profiles conducted at Camp Lake showed no substantial spatial differences in water temperature, dissolved oxygen, pH or specific conductance with progression from the CLT1 inlet to the lake outlet during any of the winter, summer or fall seasonal sampling events in 2018 (Appendix Figures C.3 to C.6). The 2018 Camp Lake water column profiles indicated a slight increase in temperature from surface to bottom (i.e., <2˚C) during the winter sampling event, but no changes in temperature with depth, including no indication of thermal stratification, during the summer and fall sampling events (Figure 3.8). The average temperature profile at Camp Lake closely mirrored that observed at Reference Lake 3 for the summer and fall sampling events in 2018 (Figure 3.8). No significant differences in water temperature near the bottom of the water column were indicated between Camp Lake and Reference Lake 3 for littoral and profundal stations sampled during August 2018 biological monitoring (Figure 3.9; Appendix Tables C.24 and C.25).







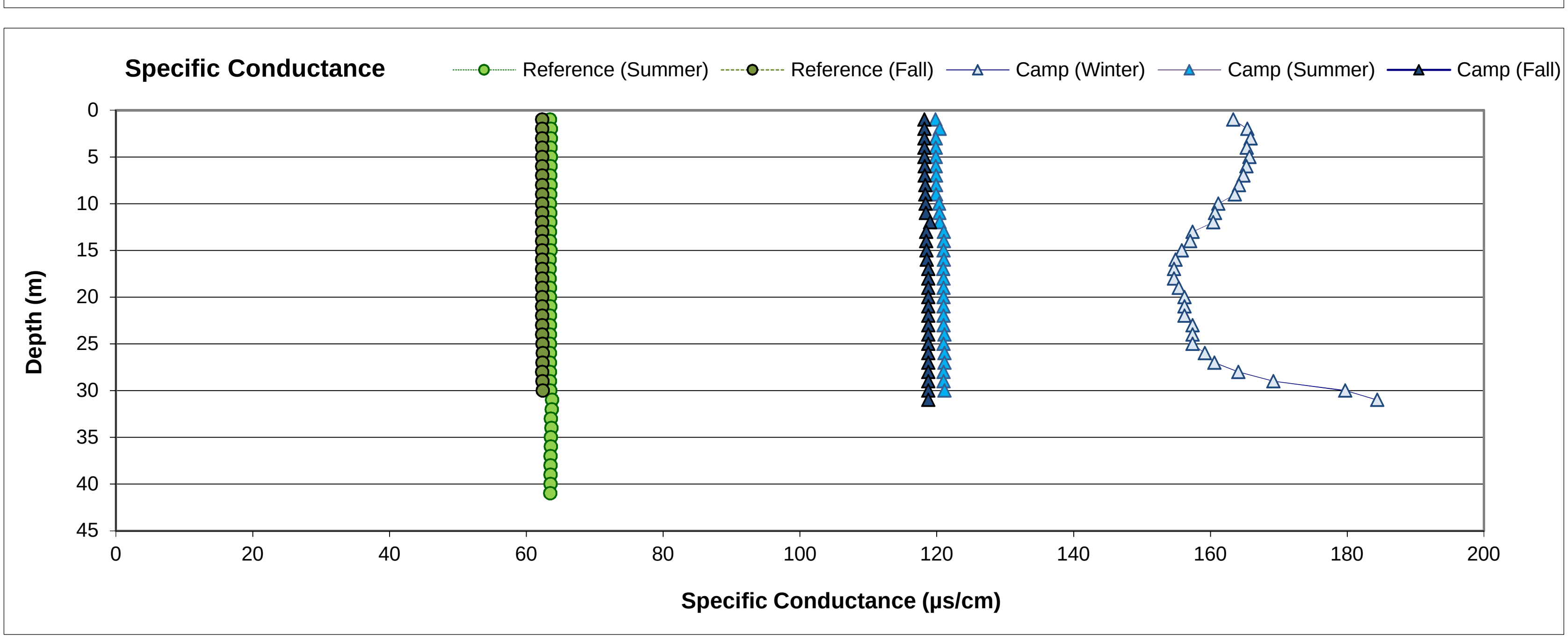


Figure 3.8: Average *In Situ* Water Quality with Depth from Surface at Camp Lake (JLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2018

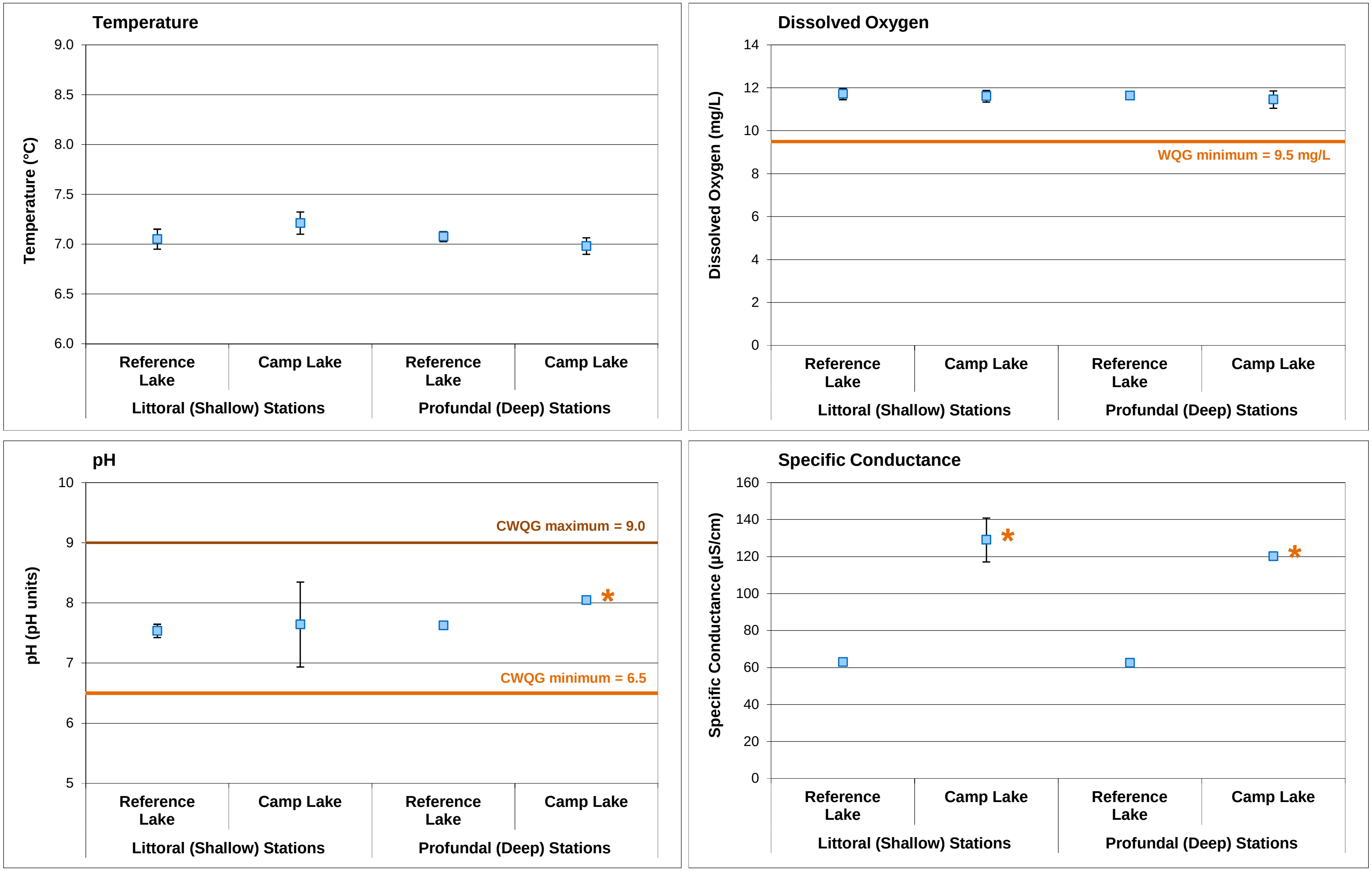


Figure 3.9: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake (JLO) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2018

Note: An asterisk (*) next to data point indicates mean value differs significantly from the Reference Lake 3 mean for the respective littoral or profundal station type.

Dissolved oxygen profiles conducted at Camp Lake in 2018 showed declining saturation levels with increased depth beginning at approximately 5 m below surface in the winter, but otherwise showed no appreciable changes from surface to bottom, and generally reflected the DO profiles observed at Reference Lake 3 for comparable depths, during summer and fall 2018 (Figure 3.8). The Camp Lake DO profiles from 2018 were comparable to those observed in winter, summer, and fall from 2015 to 2017 at Camp Lake. Dissolved oxygen levels near the bottom of the water column at littoral and profundal sampling depths of Camp Lake were generally fully saturated, and did not differ significantly from those at Reference Lake 3, during August 2018 biological sampling (Figure 3.9; Appendix Table C.25). In addition, dissolved oxygen concentrations/saturation levels at Camp Lake were well above the WQG minimum for the protection of sensitive stages of cold water biota (i.e., 9.5 mg/L or 54%, respectively) during all seasonal sampling events in 2018 except at water depths greater than approximately 25 m in winter (Figures 3.8 and 3.9). This suggested that dissolved oxygen concentrations were not likely limiting to biota at Camp Lake for the majority of the year.

In situ profiles of pH and specific conductance showed no marked step changes from the surface to bottom of the Camp Lake water column, indicating the absence of any chemical stratification (Figure 3.8). Although the bottom pH at profundal stations of Camp Lake was significantly higher than at Reference Lake 3 during the August 2018 biological study (Appendix Table C.25), the mean incremental difference between lakes was small (i.e., 0.5 pH units) and all pH values were consistently within WQG limits (Figure 3.9), suggesting that the pH difference between lakes was not ecologically meaningful. Specific conductance was consistently higher at Camp Lake than at Reference Lake 3 in summer and fall 2018, the difference of which was shown to be significant during the August 2018 biological study (Figures 3.8 and 3.9), and suggested a mine-related influence on water quality. Secchi depth readings, which served as a proxy for water clarity, were significantly lower (i.e., shallower) at Camp Lake compared to Reference Lake 3 during the 2018 August biological study (Appendix Table C.25; Appendix Figure C.7). No spatial gradient in Secchi depth readings was apparent with progression from the CLT inlet to the lake outlet stations in fall 2018 at Camp Lake (Appendix Table C.23).

Water chemistry data collected at Camp Lake in 2018 showed no distinct spatial differences with progression from the CLT inlets to the lake outlet during any of the winter, summer or fall sampling events (Table 3.4; Appendix Table C.26), suggesting that the lake waters were well mixed laterally. A slight elevation (i.e., 3- to 5-fold higher) in turbidity and concentrations of chloride, total aluminum, total manganese, and total uranium was evident at Camp Lake compared to Reference Lake 3 during the summer and/or fall 2018 sampling events (Table 3.4; Appendix Table C.27). Of the three metals indicated above, concentrations of dissolved manganese and

Table 3.4: Water Chemistry at Camp Lake (JLO) and Reference Lake 3 (REF3) Monitoring Stations[a], Mary River Project CREMP, August 2018

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3)	Camp Lake Stations					
						JL0-02	JL0-10	JL0-01	JL0-07	JL0-09	J0-01 Camp Lake Outlet
					Fall 2018	21-Aug-18	21-Aug-18	20-Aug-18	20-Aug-18	20-Aug-18	27-Aug-18
Conventionals	Conductivity (lab)	umho/cm	-	-	75	137	137	137	136	137	141
Conventionals	pH (lab)	pH	6.5 - 9.0	-	7.65	8.05	8.04	8.05	8.06	8.04	8.10
Conventionals	Hardness (as $CaCO_3$)	mg/L	-	-	35	70	70	68	70	69	70
Conventionals	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
Conventionals	Total Dissolved Solids (TDS)	mg/L	-	-	46	60	52	78	74	83	90
Conventionals	Turbidity	NTU	-	-	0.51	0.60	0.52	0.77	0.73	0.90	0.51
Conventionals	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	62	60	58	58	57	63
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.044	<0.020	<0.020	0.028	0.026	<0.020	0.044
Nutrients and Organics	Nitrate	mg/L	13	13	<0.020	0.025	<0.020	<0.020	<0.020	<0.020	<0.020
Nutrients and Organics	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
Nutrients and Organics	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.16	0.15	0.25	<0.15	<0.15	0.16	<0.15
Nutrients and Organics	Dissolved Organic Carbon	mg/L	-	-	2.9	2.6	2.4	2.5	2.5	2.5	2.2
Nutrients and Organics	Total Organic Carbon	mg/L	-	-	3.8	2.7	2.6	2.5	2.5	2.6	2.8
Nutrients and Organics	Total Phosphorus	mg/L	0.020[α]	-	0.0049	0.0041	0.0041	0.0053	0.0036	0.0041	0.0030
Nutrients and Organics	Phenols	mg/L	0.004[α]	-	0.0011	0.0019	<0.0010	0.0015	<0.0010	<0.0010	<0.0010
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
Anions	Chloride (Cl)	mg/L	120	120	1.3	3.8	3.7	3.7	3.7	3.7	3.8
Anions	Sulphate (SO_4)	mg/L	218[β]	218	3.7	3.9	3.8	3.8	3.8	3.8	3.9
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0043	0.0129	0.0068	0.0131	0.0096	0.0098	0.0077
Total Metals	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
Total Metals	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
Total Metals	Barium (Ba)	mg/L	-	-	0.0064	0.0068	0.0070	0.0067	0.0068	0.0068	0.0071
Total Metals	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
Total Metals	Calcium (Ca)	mg/L	-	-	7.2	14.0	13.6	13.5	13.2	13.6	14.0
Total Metals	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
Total Metals	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
Total Metals	Copper (Cu)	mg/L	0.002	0.0022	0.00076	0.00093	0.00076	0.00087	0.00087	0.00094	0.00083
Total Metals	Iron (Fe)	mg/L	0.30	0.326	<0.030	<0.030	<0.030	<0.030	<0.030	<0.030	<0.030
Total Metals	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
Total Metals	Magnesium (Mg)	mg/L	-	-	4.3	8.3	8.8	8.4	8.3	8.3	8.3
Total Metals	Manganese (Mn)	mg/L	0.935[β]	-	0.00064	0.00181	0.00120	0.00175	0.00170	0.00187	0.00208
Total Metals	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
Total Metals	Molybdenum (Mo)	mg/L	0.073	-	0.00014	0.00032	0.00032	0.00031	0.00030	0.00031	0.00038
Total Metals	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00061	0.00056	0.00062	0.00058	0.00060	0.00067
Total Metals	Potassium (K)	mg/L	-	-	0.9	1.2	1.3	1.2	1.2	1.2	1.1
Total Metals	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
Total Metals	Silicon (Si)	mg/L	-	-	0.42	0.34	0.32	0.32	0.33	0.32	0.33
Total Metals	Sodium (Na)	mg/L	-	-	0.9	1.7	1.8	1.6	1.6	1.6	1.5
Total Metals	Strontium (Sr)	mg/L	-	-	0.0081	0.0106	0.0102	0.0104	0.0102	0.0105	0.0117
Total Metals	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
Total Metals	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
Total Metals	Uranium (U)	mg/L	0.015	-	0.00026	0.00093	0.00076	0.00085	0.00085	0.00084	0.00087
Total Metals	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
Total Metals	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030

[a] Values presented are averages from samples taken from the surface and the bottom of the water column at each station.

[b] Canadian Water Quality Guideline (CCME 1999, 2017) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2017). See Table 2.2 for information regarding WQG criteria.

[c] AEMP Water Quality Benchmarks developed by Intrinsik (2013) using baseline water quality data (2006 - 2013) specific to Camp Lake.

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the applicable AEMP benchmark.

dissolved uranium also showed slight elevation at Camp Lake compared to Reference Lake 3 in 2018 suggesting that the elevation in manganese and uranium concentrations at Camp Lake was mine-related. In contrast, because dissolved aluminum concentrations were not elevated at Camp Lake compared to Reference Lake 3, the elevation in total aluminum concentrations at Camp Lake was likely associated with suspended particulate matter as reflected in higher turbidity (Appendix Tables C.26 and C.27). Total aluminum showed a moderately strong significant positive correlation with turbidity from the Camp Lake 2018 water chemistry data whereas dissolved aluminum did not ($r = 0.67$ and -0.10, respectively; Appendix Table C.28), supporting the notion that aluminum was associated with suspended particulate material in Camp Lake and thus was unlikely to be bioavailable. Concentrations of all parameters were below applicable WQG and AEMP benchmarks at Camp Lake during all sampling events in 2018 with the exception of phenol and total phosphorus concentrations, each of which were above respective WQG near the bottom at one or both of Stations JLO-09 and JLO-10 in winter (Appendix Table C.26). These stations are relatively shallow (i.e., <15 m deep) and showed no substantial change in profile characteristics between surface and bottom during the winter sampling event (Appendix Figures C.3 to C.6), and therefore the reason for higher concentrations of phenol and total phosphorus at the bottom of the water column of these stations in winter was unclear (e.g., no occurrence of low DO concentrations, low pH, or high specific conductance that might facilitate parameter migration from sediments to overlying waters).

Temporal comparisons of Camp Lake water chemistry data indicated that, of the parameters shown to be elevated at CLT1 in fall 2018, most showed near consistent increases over the mine baseline (2005 to 2013) and/or mine operational period (2015 to 2018), including conductivity, hardness, and total concentrations of chloride, manganese, molybdenum, sodium, strontium, sulphate, and uranium (Figure 3.10; Appendix Figure C.8). Other parameters, including iron, nitrate, TDS, and TKN, showed no consistent direction of change between the mine baseline and operational periods. Total aluminum concentrations showed a marked step increase at four of the five Camp Lake stations in fall 2018 compared to concentrations reported during mine baseline and previous years of mine operation (Figure 3.10), the reason for which was unclear. Notably, concentrations of all of the parameters indicated above have consistently been well below WQG and AEMP benchmarks through all years of mine construction and operation at Camp Lake (Figure 3.10; Appendix Figure C.8).

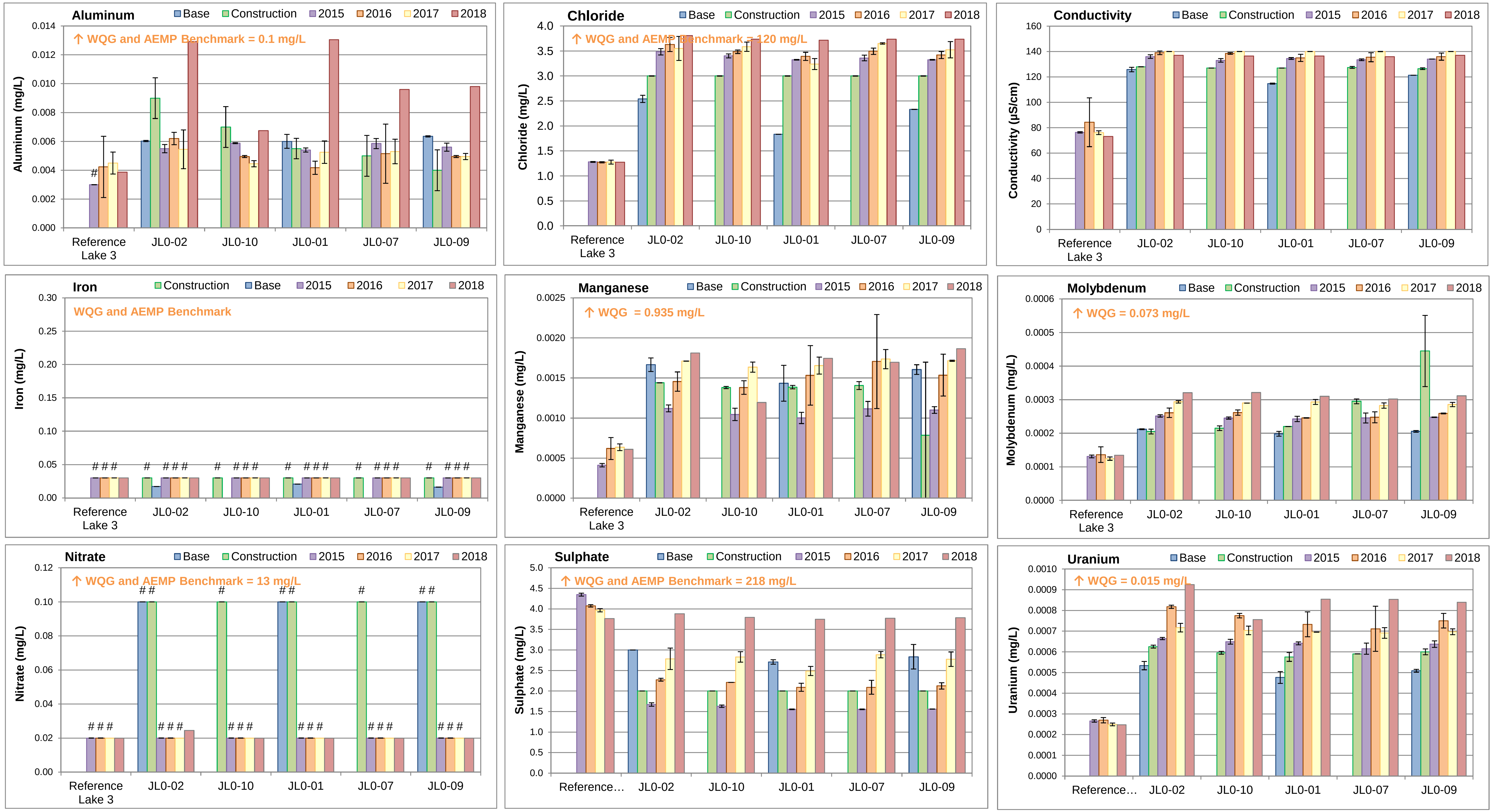


Figure 3.10: Temporal Comparison of Water Chemistry at Camp Lake (JLO) for Mine Baseline (2005 to 2013), Construction (2014) and Operational (2015 to 2018) Periods during Fall

Notes: Values represent mean ± SD. Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.2 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Camp Lake.

3.3.3 Sediment Quality

Surficial sediment (i.e., top 2 cm) collected at the Camp Lake coring stations in 2018 was characterized primarily by silt loam with low total organic carbon (TOC) content, except at Stations JLO-12 and JLO-16 where sand constituted the predominant substrate material (Figure 3.11; Appendix Table D.6). Surficial sediment at littoral stations of Camp Lake contained significantly less clay content, but otherwise showed similar particle size, than at Reference Lake 3 (Appendix Table D.7). However, TOC content in sediment at littoral and profundal stations of Camp Lake was significantly lower than at Reference Lake 3 (Figure 3.11; Appendix Table D.7). A surficial and/or sub-surface layer of oxidized material (likely iron hydroxide or oxy-hydroxides), visible as reddish-orange to orange-brown substrate, was commonly observed in sediments of Camp Lake (Appendix Tables D.5 and D.6). Similar observations of oxidized material were made at Reference Lake 3 (Appendix Tables D.1 and D.2), suggesting the natural occurrence of iron (oxy)hydroxides in the sediment of lakes within the mine local study area. Substrates of Camp Lake exhibited minor, sporadic blackening at sediment depths greater than 2 cm at some stations, suggesting occasional incidence of reducing conditions within substrates of the lake. However, no strongly defined redox boundaries were identified visually, and no noticeable sulphidic odours potentially associated with reducing sediment conditions were detected at Camp Lake littoral and profundal stations in 2018 (Appendix Tables D.5 and D.6). Qualitative observations suggestive of reducing sediment conditions were similar between Camp Lake and Reference Lake 3 in 2018 (Appendix Tables D.1, D.2, D.5 and D.6), which indicated that factors leading to reduced sediment conditions were comparable between lakes.

No spatial gradients in sediment metal concentrations were evident with progression from stations located nearest to the CLT1 inlet to those located near the outlet of Camp Lake in 2018, although concentrations of a number of metals were higher at stations located in the half of the lake located closest to the CLT1 inlet (Appendix Table D.7). Arsenic and manganese concentrations were slightly elevated (i.e., 3- to 5-fold higher) in sediment at the single Camp Lake littoral station (i.e., Station JLO-02) compared to respective average concentrations in sediment at Reference Lake 3 littoral stations (Table 3.5; Appendix Table D.8). Iron, manganese, and nickel concentrations were above respective SQG, and arsenic, iron, nickel, and phosphorus concentrations were above respective AEMP benchmarks, in sediment at the Camp Lake littoral station (Table 3.5). Of these metals, the average concentration of iron was also above SQG, and the average concentration of copper was above the Camp Lake AEMP benchmark, in littoral sediment at Reference Lake 3 (Table 3.5). Because Camp Lake littoral station JLO-02 is located near the inlet from CLT1, this suggested that mine-influenced flow from this tributary potentially contributed to elevation of the metals indicated above in sediment at this location. Metal concentrations in profundal sediment of Camp Lake were comparable to those of Reference Lake 3 in 2018

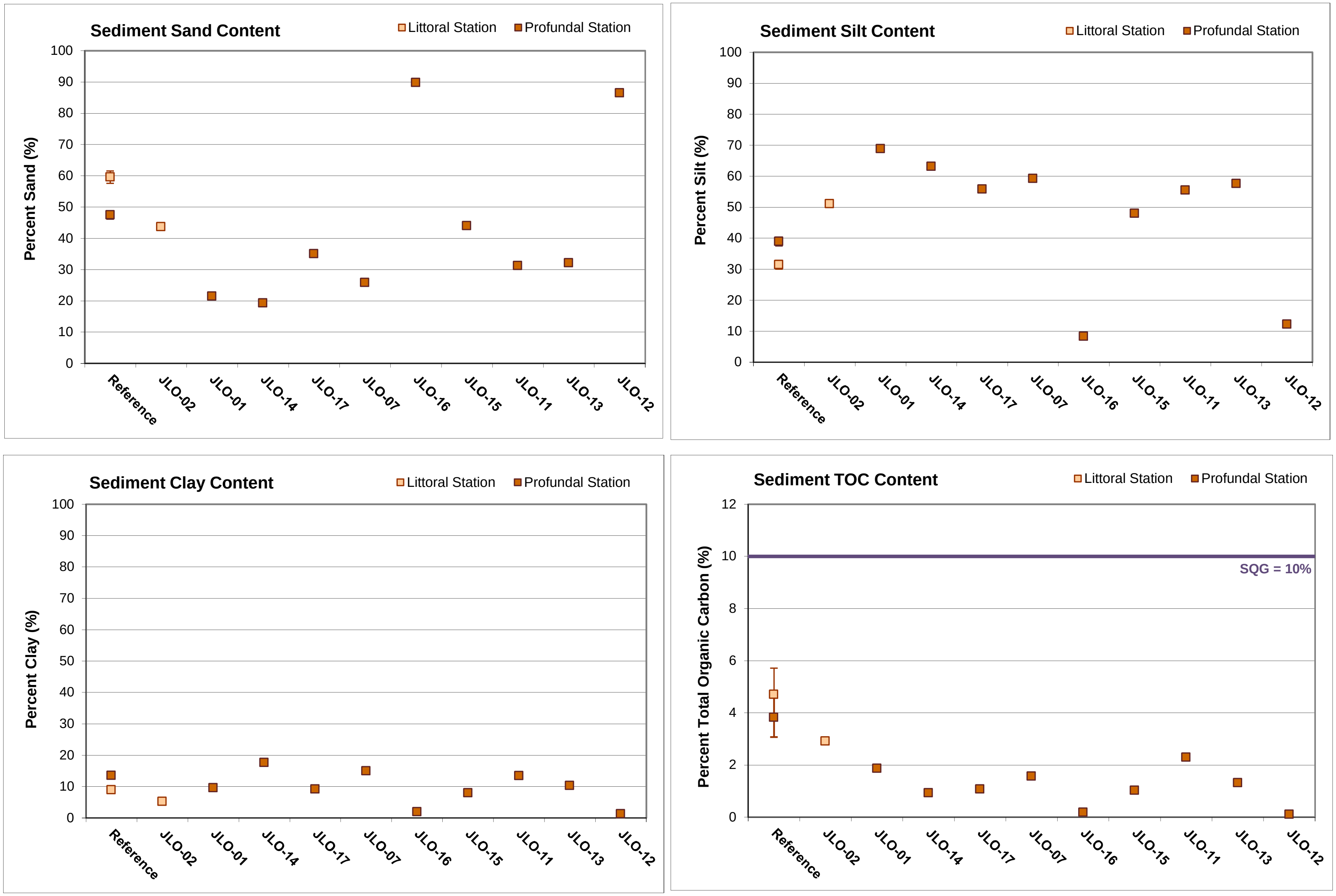


Figure 3.11: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Camp Lake (JLO) Sediment Monitoring Stations and to Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2018

Table 3.5: Sediment Total Organic Carbon and Metal Concentrations at Camp Lake (JLO) and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2018

	Analyte	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b]	Littoral Stations: Reference Lake (n = 5) Average ± Std. Error	Littoral Stations: Camp Lake (n = 1) Average ± Std. Error	Profundal Stations: Reference Lake (n = 5) Average ± Std. Error	Profundal Stations: Camp Lake (n = 9) Average ± Std. Error
Total Organic Carbon		%	10[α]	-	4.70 ± 1.01	2.91 ± 0.52	3.82 ± 0.75	1.15 ± 0.24
Metals	Aluminum (Al)	mg/kg	-	-	17,880 ± 1,993	18,600 ± 0	24,420 ± 3,494	16,447 ± 1,992
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	0.10 ± 0.00	<0.10 ± 0.00	<0.10 ± 4.90654E-18
	Arsenic (As)	mg/kg	17	5.9	5.25 ± 0.95	***9.62*** ± 0.00	***6.07*** ± 0.78	4.45 ± 0.69
	Barium (Ba)	mg/kg	-	-	133 ± 25	164 ± 0	152 ± 23	70 ± 10
	Beryllium (Be)	mg/kg	-	-	0.68 ± 0.08	0.89 ± 0.00	0.87 ± 0.12	0.85 ± 0.12
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.30 ± 0.00	<0.20 ± 0.00	0.27 ± 0.02
	Boron (B)	mg/kg	-	-	13.9 ± 1.6	19.7 ± 0.0	15.6 ± 2.2	21.4 ± 2.7
	Cadmium (Cd)	mg/kg	3.5	1.5	0.195 ± 0.044	0.237 ± 0.000	0.197 ± 0.005	0.163 ± 0.027
	Calcium (Ca)	mg/kg	-	-	5,480 ± 804	5,310 ± 0	5,584 ± 664	5,553 ± 1,371
	Chromium (Cr)	mg/kg	90	98	58.9 ± 7.7	76.5 ± 0.0	77.3 ± 11.0	67.5 ± 6.9
	Cobalt (Co)	mg/kg	-	-	11.70 ± 1.40	25.60 ± 0.00	17.42 ± 2.37	16.51 ± 2.11
	Copper (Cu)	mg/kg	110[α]	50	***73.9*** ± 11.0	49.6 ± 0.0	***96*** ± 14.7	39.9 ± 5.5
	Iron (Fe)	mg/kg	40,000[α]	52,400	46,700 ± 9,489	***65,100*** ± 0	50,900 ± 7,115	31,011 ± 3,413
	Lead (Pb)	mg/kg	91	35	16.4 ± 2.1	19.8 ± 0.0	19.5 ± 2.8	18.3 ± 2.5
	Lithium (Li)	mg/kg	-	-	26.0 ± 2.7	30.4 ± 0.0	36.1 ± 5.0	29.2 ± 3.9
	Magnesium (Mg)	mg/kg	-	-	11,104 ± 1,352	15,800 ± 0	15,394 ± 2,199	13,034 ± 872
	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	640 ± 60	2,390 ± 0	1,279 ± 115	1,132 ± 222
	Mercury (Hg)	mg/kg	0.486	0.17	0.0433 ± 0.0111	0.0418 ± 0.0000	0.0650 ± 0.0121	0.0264 ± 0.0050
	Molybdenum (Mo)	mg/kg	-	-	3.84 ± 0.86	2.35 ± 0.00	2.57 ± 0.27	0.81 ± 0.11
	Nickel (Ni)	mg/kg	75[α,β]	72	42.9 ± 5.9	***83.4*** ± 0.0	53.8 ± 6.6	61.7 ± 6.2
	Phosphorus (P)	mg/kg	2,000[α]	1,580	1,305 ± 272	***1,650*** ± 0	1,188 ± 118	941 ± 91
	Potassium (K)	mg/kg	-	-	4,134 ± 469	4,100 ± 0	5,660 ± 796	4,010 ± 504
	Selenium (Se)	mg/kg	-	-	0.66 ± 0.14	0.48 ± 0.00	0.81 ± 0.15	0.31 ± 0.03
	Silver (Ag)	mg/kg	-	-	0.15 ± 0.02	0.11 ± 0.00	0.26 ± 0.04	0.12 ± 0.01
	Sodium (Na)	mg/kg	-	-	319.8 ± 43	185 ± 0	433 ± 62	196 ± 29
	Strontium (Sr)	mg/kg	-	-	12.2 ± 1.5	10.1 ± 0.0	13.8 ± 1.6	12.9 ± 1.7
	Thallium (Tl)	mg/kg	-	-	0.450 ± 0.063	0.564 ± 0.000	0.754 ± 0.091	0.429 ± 0.062
	Tin (Sn)	mg/kg	-	-	<2.0 ± 0.0	<2.0 ± 0.0	2.1 ± 0.0	<2.0 ± 0.0
	Titanium (Ti)	mg/kg	-	-	1,155 ± 132	1,180 ± 0	1,388 ± 163	925 ± 99
	Uranium (U)	mg/kg	-	-	13.4 ± 2.3	6.41 ± 0.0	24.5 ± 3.9	4.89 ± 0.7
	Vanadium (V)	mg/kg	-	-	58.3 ± 6.9	67.0 ± 0.0	72.7 ± 9.4	56.7 ± 6.7
	Zinc (Zn)	mg/kg	315	135	81.36 ± 10.2	66.4 ± 0.0	99 ± 14.2	51.5 ± 6.3
	Zirconium (Zr)	mg/kg	-	-	4.1 ± 0.8	6.4 ± 0.0	3.9 ± 0.5	5.2 ± 0.8

[a] Canadian Sediment Quality Guideline for the protection of aquatic life, probable effects level (PEL; CCME 2017) except those indicated by α (Ontario Provincial Sediment Quality Objective [PSQO], severe effect level (SEL); OMOE 1993) and β (British Columbia Working Sediment Quality Guideline [BCSQG], probable effects level (PEL; BCMOE 2017)).

[b] AEMP Sediment Quality Benchmarks developed by Intrinsik (2013). The indicated values are specific to Camp Lake.

(grey shading) Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

(Table 3.5; Appendix Table D.8). Although mean concentrations of iron and manganese were above respective SQG in profundal sediment at Camp Lake, mean concentrations of these metals were also above SQG in profundal sediment at Reference Lake 3 (Table 3.5) indicating naturally high concentrations of iron and manganese in sediment of lakes in the mine local study area. Concentrations of arsenic, copper, and nickel were above respective Camp Lake AEMP benchmarks in sediment at profundal stations located in the half of the lake located closest to the CLT1 outlet, but on average, were below the applicable benchmarks (Table 3.5; Appendix Table D.7). Of these latter metals, average concentrations of arsenic and copper were also above Camp Lake AEMP benchmarks in profundal sediment at Reference Lake 3 (Table 3.5), indicating naturally high concentrations of these metals in sediment of local study area lakes.

Temporal comparisons indicated that average metal concentrations in sediment at Camp Lake littoral and profundal stations were comparable between 2018 and the baseline period for each respective station type, the only exceptions of which were slightly higher (i.e., 3- to 5-fold greater) arsenic and manganese concentrations in sediment at the single Camp Lake littoral station in 2018 (Figure 3.12; Appendix Table D.8).[8] Average metal concentrations in sediment at Camp Lake littoral and profundal stations in 2018 were typically within the range of those observed from 2015 to 2017 (Figure 3.12). In addition, no pattern of consistently higher metal concentrations has occurred in Camp Lake sediment over the 2015 to 2018 period of mine operation (Figure 3.12). Overall, with the exception of a step-increase in arsenic and manganese concentrations shown at the littoral station closest to the CLT1 inlet to Camp Lake in 2015, and taking reference lake data into consideration, no substantial changes to sediment metal concentrations have been observed at Camp Lake littoral and profundal stations following the commencement of commercial mine operations in 2015.

3.3.4 Phytoplankton

Camp Lake chlorophyll-a concentrations showed no clear spatial gradients with distance from the CLT1 inlet to the lake outlet stations during any of the winter, summer, or fall sampling events in 2018 (Figure 3.13). Chlorophyll-a concentrations differed significantly among seasons at Camp Lake, with highest and lowest concentrations occurring during the fall and winter sampling events, respectively (Figure 3.13). On average, chlorophyll-a concentrations at Camp Lake were significantly higher than at Reference Lake 3 during the summer and fall sampling events (Appendix Tables E.7 and E.8), suggesting greater phytoplankton abundance at Camp Lake.

[8] Reported sediment boron concentrations from 2015 to 2018 were considerably higher (i.e., 10- to 70-fold) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences.

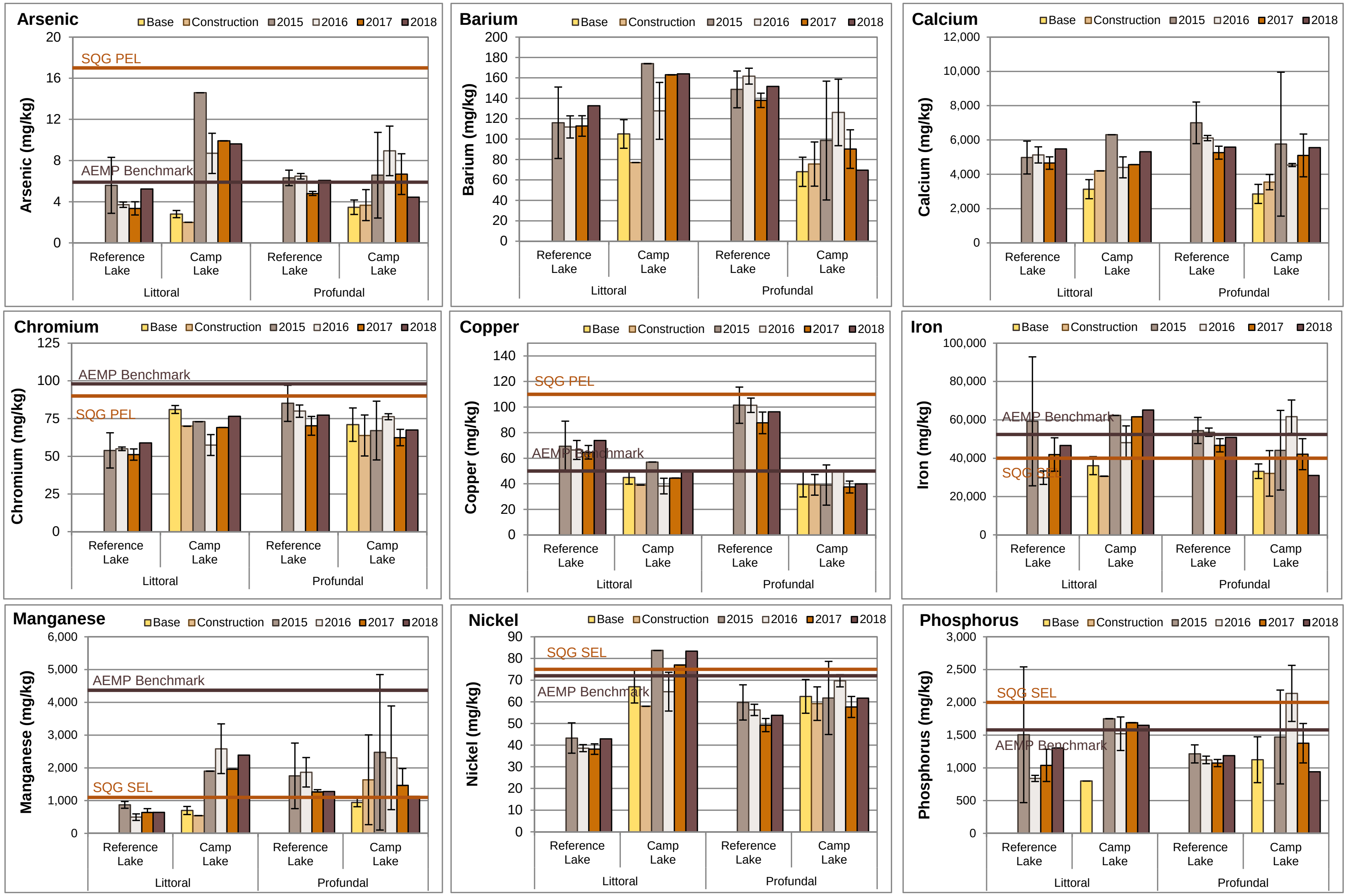


Figure 3.12: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Camp Lake and Reference Lake 3 for Mine Baseline (2005 to 2013), Construction (2014) and Operational (2015 to 2018) Periods, Mary River Project CREMP, 2018

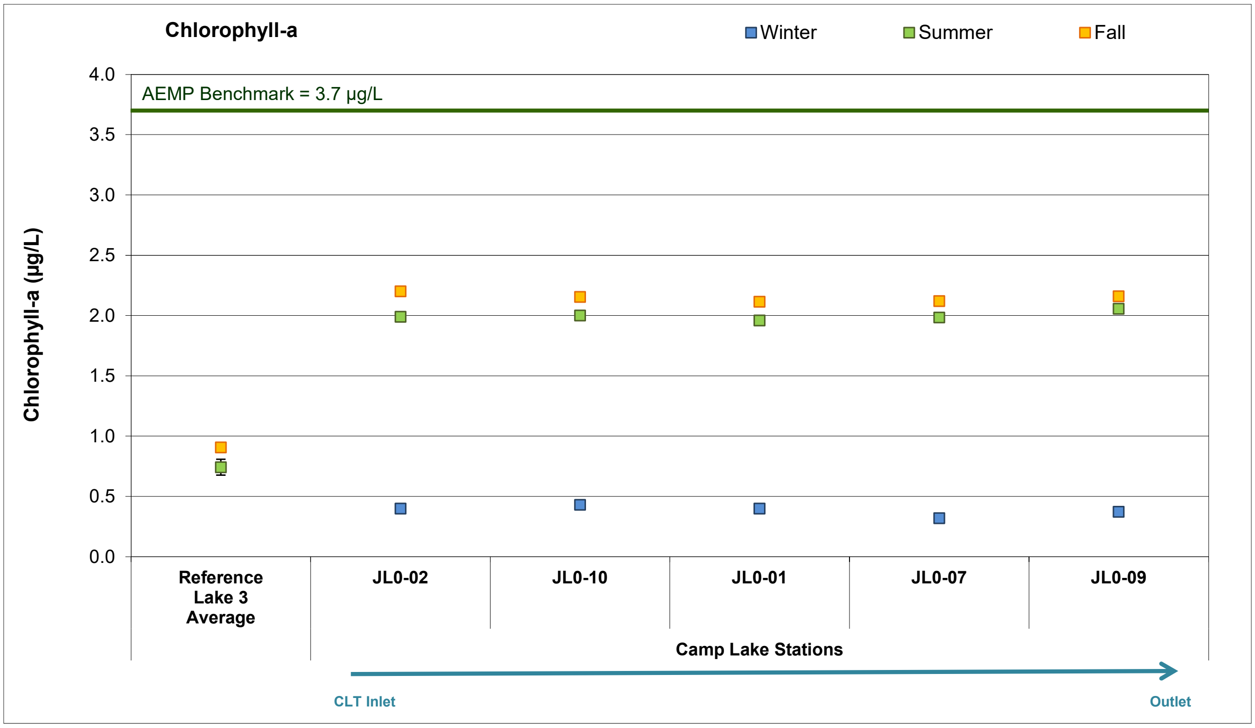


Figure 3.13: Chlorophyll-a Concentrations at Camp Lake (JLO) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2018

Notes: Values are averages of samples taken from the surface and the bottom of the water column at each station. Reference values represent mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2018.

However, the Camp Lake chlorophyll-a concentrations were consistently well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events in 2018 (Figure 3.13). Average chlorophyll-a concentrations at Camp Lake suggested relatively low phytoplankton abundance and an oligotrophic status based on comparison to Wetzel (2001) lake trophic classifications using chlorophyll-a concentrations. This trophic status classification was also consistent with an ultra-oligotrophic to oligotrophic CWQG categorization for Camp Lake based on mean aqueous total phosphorus concentrations below 10 µg/L during all 2018 sampling events (Table 3.4; Appendix Table C.26).

Temporal comparisons of the Camp Lake chlorophyll-a data did not indicate any consistent significant differences between years of mine construction (2014) and mine operation (2015 to 2018) for seasonal data collected in winter, summer, or fall (Figure 3.14). However, average chlorophyll-a concentrations were significantly higher during summer and fall sampling events in 2018 compared to the previous three years of commercial mine operation (Figure 3.14; Appendix Table E.9). No changes in nitrate concentrations were evident at Camp Lake among the four years of commercial mine operation (Figure 3.10), and therefore the occurrence of higher chlorophyll-a concentrations at Camp Lake in 2018 did not appear to be related to mine-related

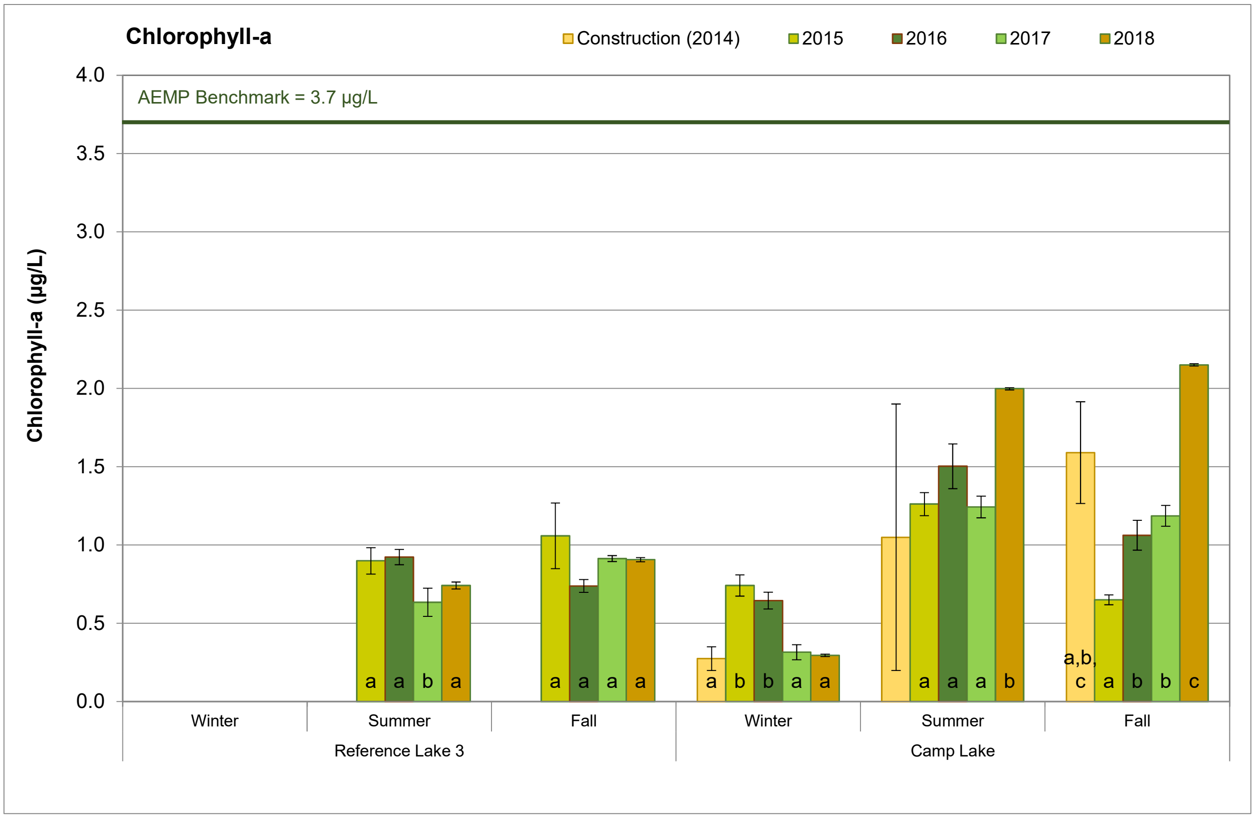


Figure 3.14: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Camp Lake and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2018) Periods (mean ± SE)

Note: Bars with the same letter at the base do not differ significantly between years for the applicable season.

nutrient inputs. No chlorophyll-a baseline (2005 to 2013) data are available for Camp Lake, precluding comparisons to conditions prior to the mine construction period.

3.3.5 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral and profundal habitat of Camp Lake compared to like-habitat stations at Reference Lake 3 (Tables 3.6 and 3.7). For both habitat types, the magnitude of difference in density was ecologically meaningful based on a CES_{BIC} outside of ±2 SD_{REF}. Although no significant difference in richness was indicated between lakes at littoral stations, richness was significantly higher at Camp Lake profundal habitat compared to like-habitat at Reference Lake 3 by a magnitude outside of the CES_{BIC} of ±2 SD_{REF} (Tables 3.6 and 3.7). In addition to these differences, benthic invertebrate community structure differences were indicated between Camp Lake and Reference Lake 3 by significantly differing Bray-Curtis Index for both littoral and profundal habitat types (Tables 3.6 and 3.7). However, no ecologically meaningful differences in the relative abundance of any benthic invertebrate dominant groups

Table 3.6: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JLO) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	none	YES	0.009	ANOVA	6.1	Reference Lake 3	1,045	258	116	696	1,000	1,391
						Camp Lake Littoral	2,623	1,006	450	1,200	3,043	3,539
Richness (Number of Taxa)	none	NO	0.102	ANOVA	1.5	Reference Lake 3	10.8	2.3	1.0	7.0	11.0	13.0
						Camp Lake Littoral	14.2	3.4	1.5	9.0	15.0	18.0
Simpson's Evenness (E)	log10	NO	0.337	ANOVA	0.3	Reference Lake 3	0.825	0.103	0.046	0.720	0.816	0.939
						Camp Lake Littoral	0.851	0.057	0.025	0.784	0.858	0.930
Bray-Curtis Index	square-root	YES	< 0.001	ANOVA	4.9	Reference Lake 3	0.313	0.092	0.041	0.178	0.358	0.394
						Camp Lake Littoral	0.768	0.046	0.020	0.704	0.758	0.821
Nemata (%)	square-root	NO	0.450	ANOVA	-0.5	Reference Lake 3	7.1	8.8	3.9	0.0	3.4	21.3
						Camp Lake Littoral	2.8	3.2	1.5	0.0	1.4	7.7
Ostracoda (%)	fourth-root	YES	< 0.001	ANOVA	-1.3	Reference Lake 3	23.9	18.3	8.2	3.4	20.6	53.3
						Camp Lake Littoral	0.4	0.6	0.3	0.0	0.0	1.5
Chironomidae (%)	none	YES	0.023	ANOVA	1.3	Reference Lake 3	66.9	22.2	10.0	35.5	73.8	91.4
						Camp Lake Littoral	95.4	4.0	1.8	89.1	95.6	99.0
Metal-Sensitive Chironomidae (%)	log10	YES	0.085	ANOVA	-1.0	Reference Lake 3	36.5	19.6	8.8	17.8	27.5	60.1
						Camp Lake Littoral	17.4	18.5	8.3	2.9	14.0	49.1
Collector-Gatherers (%)	none	NO	0.353	ANOVA	0.6	Reference Lake 3	55.6	19.0	8.5	33.0	57.5	79.2
						Camp Lake Littoral	67.3	18.6	8.3	39.4	66.7	89.9
Filterers (%)	log10	YES	0.094	ANOVA	-0.9	Reference Lake 3	33.9	18.7	8.4	15.5	24.9	56.6
						Camp Lake Littoral	16.7	17.8	7.9	2.9	13.6	47.1
Shredders (%)	none	YES	0.010	ANOVA	-1.6	Reference Lake 3	7.0	2.6	1.1	2.9	7.5	9.4
						Camp Lake Littoral	2.9	1.0	0.4	1.4	3.2	3.7
Clingers (%)	square-root	YES	0.018	ANOVA	-1.4	Reference Lake 3	36.1	18.4	8.2	17.1	26.9	58.3
						Camp Lake Littoral	9.5	11.2	5.0	0.0	4.5	28.1
Sprawlers (%)	none	NO	0.938	ANOVA	0.0	Reference Lake 3	51.9	17.7	7.9	29.5	52.5	71.8
						Camp Lake Littoral	52.7	12.0	5.4	34.8	52.4	67.2
Burrowers (%)	square-root	YES	0.006	ANOVA	4.1	Reference Lake 3	12.0	6.4	2.8	6.9	11.1	22.6
						Camp Lake Littoral	37.8	17.2	7.7	19.5	37.8	65.2

[a] Magnitude calculated by comparing the difference between the reference area and mine-exposed area means divided by the reference area standard deviation.

Grey shading indicates statistically significant difference between study areas based on p-value≤ 0.10.

Blue shaded values indicate significant difference (ANOVA p-value ≤ 0.10) that was also outside of a Critical Effect Size of ±2 SD_{REF}, indicating that the difference was ecologically meaningful.

Table 3.7: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JLO) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	fourth-root	YES	0.004	ANOVA	5.8	Reference Lake 3	377	155	69	104	452	470
						Camp Lake Profundal	1,270	616	275	887	1,043	2,365
Richness (Number of Taxa)	log10	YES	0.008	ANOVA	2.1	Reference Lake 3	5.4	1.3	0.6	4.0	6.0	7.0
						Camp Lake Profundal	8.2	0.8	0.4	7.0	8.0	9.0
Simpson's Evenness (E)	log10	NO	0.841	ANOVA	-0.3	Reference Lake 3	0.455	0.296	0.132	0.218	0.296	0.933
						Camp Lake Profundal	0.373	0.120	0.054	0.218	0.380	0.499
Bray-Curtis Index	none	YES	0.018	ANOVA	1.8	Reference Lake 3	0.224	0.304	0.136	0.0505	0.109	0.763
						Camp Lake Profundal	0.765	0.273	0.122	0.448	0.951	0.981
Nemata (%)	rank	NO	1.000	Mann-Whitney U	0.1	Reference Lake 3	2.5	3.8	1.7	0.0	0.0	8.7
						Camp Lake Profundal	2.9	5.6	2.5	0.0	0.0	12.9
Hydracarina (%)	none	NO	0.225	t-test (unequal)	-0.6	Reference Lake 3	3.7	3.8	1.7	0.0	3.9	8.7
						Camp Lake Profundal	1.2	1.0	0.4	0.0	1.0	2.6
Ostracoda (%)	fourth-root	YES	0.071	ANOVA	-0.9	Reference Lake 3	3.1	2.9	1.3	0.0	2.0	7.5
						Camp Lake Profundal	0.7	1.5	0.7	0.0	0.0	3.4
Chironomidae (%)	rank	NO	0.151	Mann-Whitney U	0.9	Reference Lake 3	90.8	4.9	2.2	82.7	92.2	95.7
						Camp Lake Profundal	95.2	7.9	3.5	81.1	98.5	99.1
Metal-Sensitive Chironomidae (%)	log10	NO	0.985	ANOVA	-0.3	Reference Lake 3	11.4	16.8	7.5	2.3	3.9	41.4
						Camp Lake Profundal	6.6	3.0	1.3	3.1	5.9	11.0
Collector-Gatherers (%)	rank	NO	0.548	Mann-Whitney U	0.4	Reference Lake 3	89.8	13.6	6.1	66.3	96.2	100.0
						Camp Lake Profundal	95.6	2.9	1.3	91.2	96.9	98.3
Filterers (%)	square-root	NO	0.671	ANOVA	-0.3	Reference Lake 3	6.5	10.5	4.7	0.0	3.7	25.0
						Camp Lake Profundal	3.0	3.7	1.7	0.0	2.1	8.8
Clingers (%)	square-root	NO	0.556	ANOVA	-0.4	Reference Lake 3	10.2	13.6	6.1	0.0	3.9	33.6
						Camp Lake Profundal	4.5	4.7	2.1	0.9	2.6	12.5
Sprawlers (%)	fourth-root	YES	0.083	ANOVA	-1.5	Reference Lake 3	79.3	26.8	12.0	32.7	90.4	100.0
						Camp Lake Profundal	38.4	44.4	19.8	2.2	9.6	93.2
Burrowers (%)	none	YES	0.046	t-test (unequal)	3.3	Reference Lake 3	10.6	14.1	6.3	0.0	5.6	33.6
						Camp Lake Profundal	57.1	41.8	18.7	5.9	85.3	90.9

[a] Magnitude calculated by comparing the difference between the reference area and mine-exposed area means divided by the reference area standard deviation.

Grey shaded values indicate statistically significant difference between study areas based on p-value ≤ 0.10.

Blue shaded values indicate significant difference (ANOVA p-value ≤ 0.10) that was also outside of a CES of ±2 SD_{REF}, indicating that the difference was ecologically meaningful.

were indicated between Camp Lake and Reference Lake 3 for either habitat type. Therefore, the differences in Bray-Curtis Index indicated between Camp Lake and Reference Lake 3 likely reflected the combination of higher benthic invertebrate density and richness at Camp Lake as well as differences in the densities of individual taxa of the various dominant benthic invertebrate groups between lakes. Because higher benthic invertebrate density and richness are not consistent with adverse influences typically associated with mine operations, factors other than metal concentrations likely accounted for the differences in benthic invertebrate community structure between Camp Lake and Reference Lake 3. This was supported by no ecologically meaningful difference in the relative abundance of metal-sensitive Chironomidae shown between Camp Lake and Reference Lake 3 for both habitat types (Tables 3.6 and 3.7), which suggested that the difference in benthic invertebrate community structure between lakes was unlikely to be associated with differences in metal concentrations. Notably, aqueous metal concentrations were below WQG and AEMP benchmarks at Camp Lake (Appendix Table C.26), and metal concentrations in sediment were generally below SQG at Camp Lake with the exception of iron and manganese, which were also above SQG at Reference Lake 3 (Table 3.5).

The subtle differences in benthic invertebrate community structure between Camp Lake and Reference Lake 3 also did not appear to be related to differences in food resources between lakes as demonstrated by no ecologically meaningful differences in FFG between lakes for either habitat type (Tables 3.6 and 3.7). Rather, an ecologically significant higher relative abundance of burrowing invertebrates at littoral and profundal habitat of Camp Lake compared to Reference Lake 3 suggested that the differences in community structure between lakes may have been related to natural differences in substrate properties between lakes. The key differences in substrate properties of benthic stations between lakes that was common to both littoral and profundal habitats was significantly lower content of moisture and total organic carbon (TOC) in sediment at Camp Lake (Appendix Table F.20). These properties suggested that substrate at Camp Lake was more compact than that at Reference Lake 3. Because substrate compactness is an important factor influencing inhabitation by burrowing invertebrates (Ward 1992), greater substrate compactness at Camp Lake may have accounted for the subtle benthic invertebrate community assemblage differences compared to Reference Lake 3.

Temporal comparisons did not indicate any consistent ecologically significant differences in general community effect indicators of density, richness, and Simpson's Evenness at littoral and profundal habitats of Camp Lake between the mine baseline (2007, 2013) and individual years of commercial mine operation since 2015 (Figure 3.15; Appendix Tables F.22 and F.23). Similarly, no significant differences in benthic invertebrate dominant taxonomic groups or FFG were consistently indicated between baseline and mine operational years for littoral habitat at Camp Lake (Figure 3.15; Appendix Table F.22). Despite a significantly lower relative abundance of

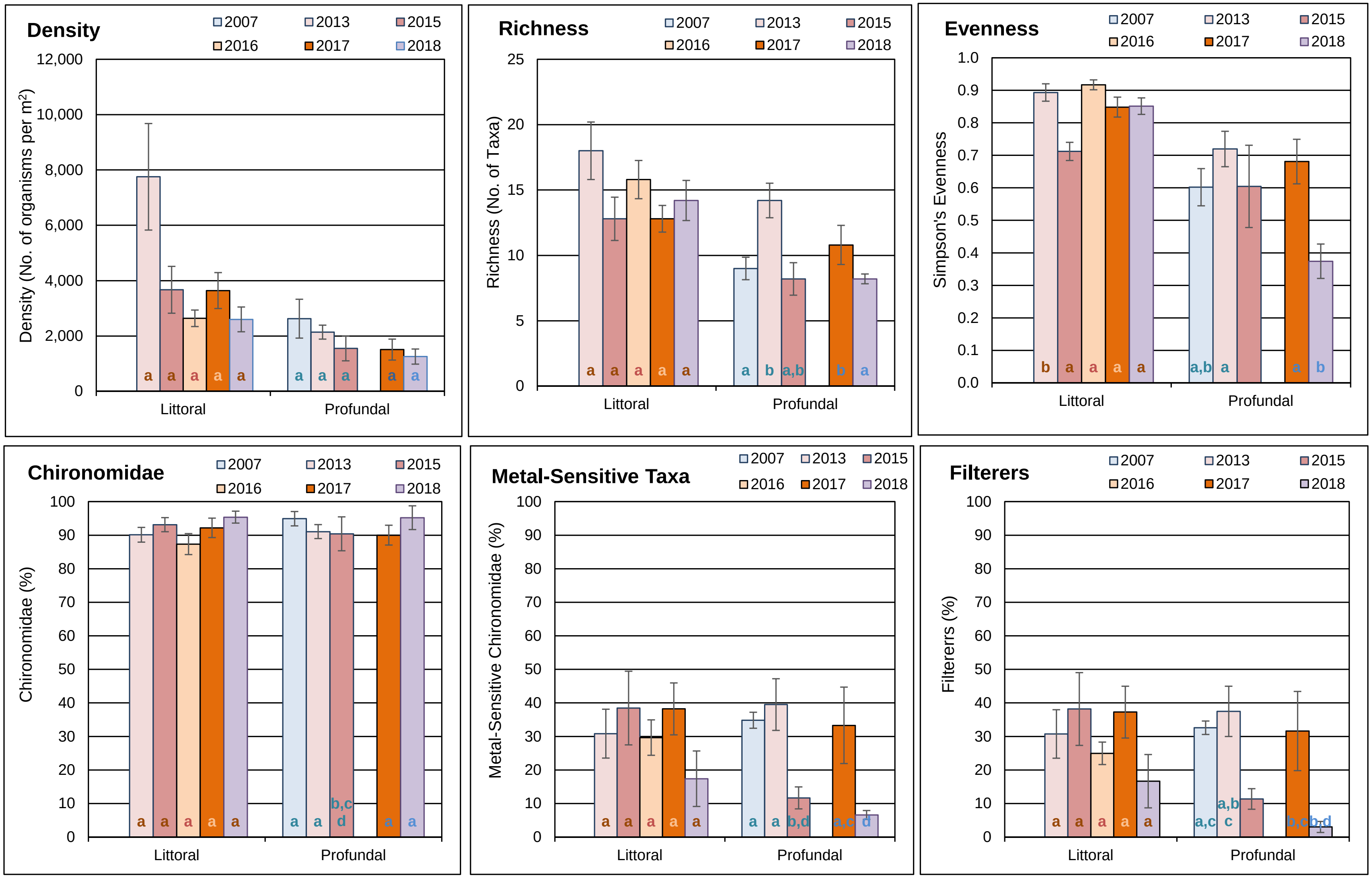


Figure 3.15: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Camp Lake Littoral and Profundal Study Areas among Mine Baseline (2007, 2013) and Operational (2015 to 2018) Periods

Note: The same like-coloured letter inside bars indicates no significant difference between/among study years for respective community endpoint.

metal-sensitive chironomids and FFG in mine operational years of 2015 and 2018 compared to the 2007 and 2013 baseline data for profundal habitat at Camp Lake, similar differences were not indicated between 2017 and either of the 2007 or 2013 baseline data (Appendix Table F.23). This indicated that the study-to-study differences in community features at profundal stations of Camp Lake were likely the result of sampling artifacts (e.g., differences in sampling station locations and/or replication among studies) or natural temporal variability among studies unrelated to potential influences from commercial mine operation. Overall, consistent with only minor changes in water and sediment quality since the mine baseline period, no significant changes in benthic invertebrate community features were indicated at littoral and profundal habitat of Camp Lake following the commencement of commercial mine operation in 2015.

3.3.6 Fish Population

3.3.6.1 Camp Lake Fish Community

The Camp Lake fish community was represented by arctic charr (*Salvelinus alpinus*) and ninespine stickleback (*Pungitius pungitius*), reflecting the same fish species composition as that observed at Reference Lake 3 (Table 3.8). A higher density of arctic charr was suggested at Camp Lake compared to Reference Lake 3 based on both greater electrofishing total catch-per-unit-effort (CPUE) from shallow rocky nearshore habitat, and greater gill netting CPUE from deeper littoral/profundal habitat at Camp Lake (Table 3.8). In turn, this suggested higher fish productivity at Camp Lake compared to Reference Lake 3, and was consistent with the chlorophyll-a and benthic invertebrate community results which indicated higher phytoplankton abundance and greater benthic invertebrate density at Camp Lake. Ninespine stickleback, which were first recorded in Camp Lake in 2016 (Minnow 2017), appeared to exhibit similar abundance at rocky nearshore habitat of Camp Lake and Reference Lake 3 based on comparable electrofishing CPUE for this species (Table 3.8). The electrofishing and gill netting CPUE for arctic charr at Camp Lake in 2018 were within or greater than, respectively, the range of those observed during baseline (2005 to 2013) studies (Figure 3.16). In addition, the CPUE of arctic charr was greater than those observed during each of the previous three years of mine operation for each respective collection method (Figure 3.16). In turn, this suggested no decline in the relative abundance of arctic charr at nearshore or littoral/profundal habitats of Camp Lake compared to the mine baseline period or since the commencement of commercial mine operations in 2015.

Table 3.8: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2018

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	101	2	103	2
		CPUE	1.59	0.02	1.61	
	Gill netting	No. Caught	34	0	34	
		CPUE	0.38	0	0.38	
Camp Lake	Electrofishing	No. Caught	109	1	110	2
		CPUE	7.81	0.07	7.89	
	Gill netting	No. Caught	94	0	94	
		CPUE	4.29	0	4.29	

[a] Catch-per-unit-effort (CPUE) for electrofishing represents the number of fish captured per electrofishing minute, and for gill netting represents the number of fish captured per 100 m hours of net deployed.

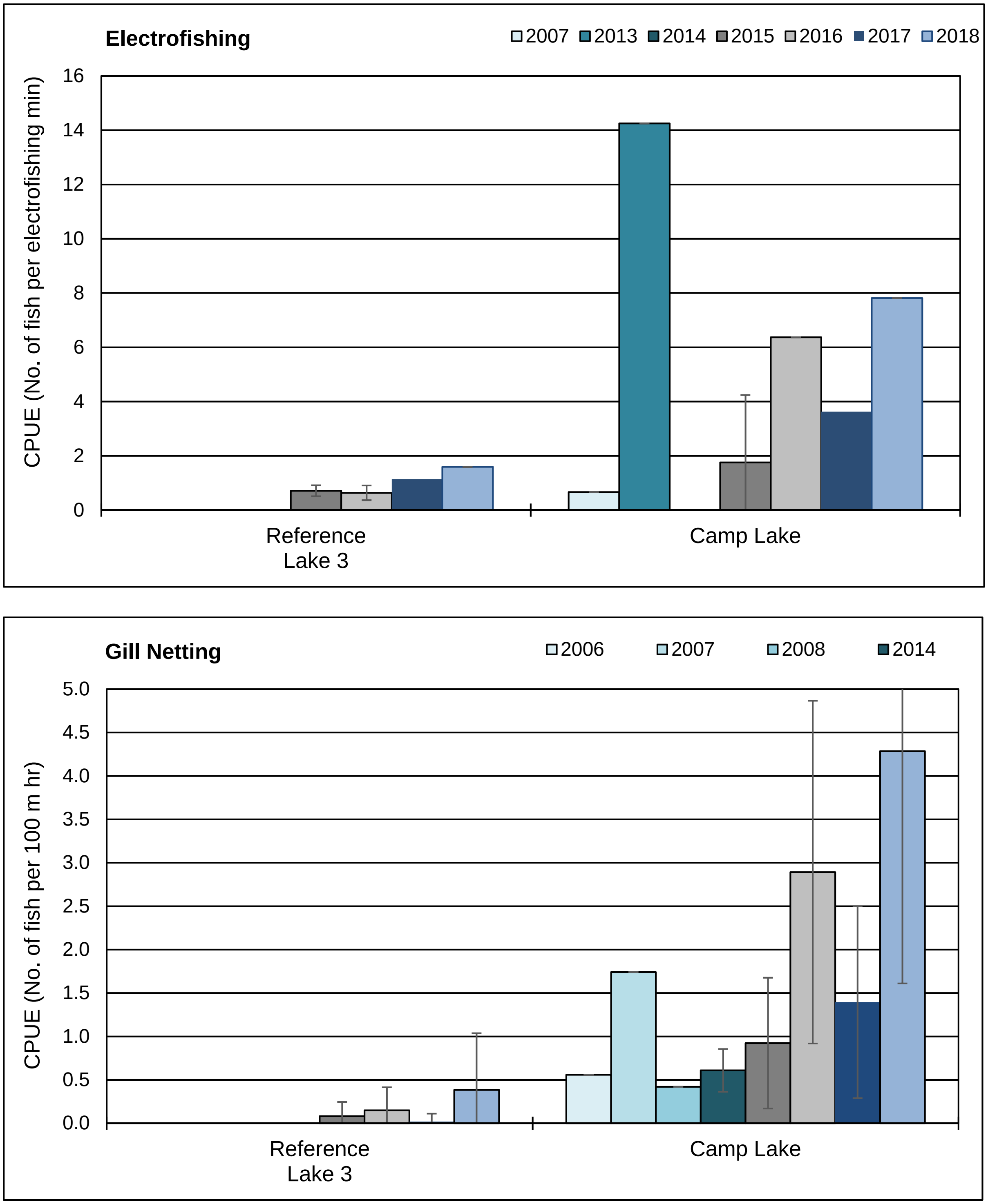


Figure 3.16: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, 2006 to 2018

Note: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015 to 2018) mine phases.

3.3.6.2 Camp Lake Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Camp Lake nearshore arctic charr population (i.e., fish captured by electrofishing) were assessed based on a control-impact analysis using 2018 data from Camp Lake and Reference Lake 3, as well as a before-after analysis using Camp Lake 2018 and baseline (2013) data. A total of 109 and 100 arctic charr were sampled at nearshore habitat of Camp Lake and Reference Lake 3, respectively, in August 2018, for the control-impact analysis. Young-of-the-year (YOY) were distinguished from older (non-YOY) age classes at a fork length cut-off of 4.5 cm for the Camp Lake and Reference Lake 3 data sets based on the evaluation of length-frequency distributions coupled with supporting age determinations (Figure 3.17). Due to the low number of arctic charr YOY captured at Camp Lake (i.e., two), fish population comparisons focused on non-YOY individuals, where applicable, to limit confounding influences of naturally differing weight-at-length relationships between YOY and non-YOY individuals on data interpretation.

The length-frequency distribution for the nearshore arctic charr differed significantly between Camp Lake and Reference Lake 3 (Table 3.9), reflecting the occurrence of very few YOY and greater numbers of larger individuals captured at Camp Lake (Figure 3.17). Non-YOY arctic charr captured at the Camp Lake nearshore were significantly longer (40%) and heavier (135%) than those captured at Reference Lake 3 nearshore (Table 3.9; Appendix Table G.6). However, condition (i.e., weight-at-length relationship) of non-YOY arctic charr was significantly lower at Camp Lake than Reference Lake 3 at a magnitude outside of the ecologically meaningful Critical Effect Size for condition of ±10% (referred to herein as CES_C; Table 3.9; Appendix Table G.6). The occurrence of lower arctic charr condition at Camp Lake may have reflected influences associated with greater densities (e.g., intraspecific competition) and/or greater number of larger sized individuals (e.g., natural size-dependent differences) compared to Reference Lake 3.

Temporal comparisons of the Camp Lake nearshore non-YOY arctic charr data indicated significantly different length-frequency distribution between the 2018 study and the 2013 baseline study (Table 3.9). In addition, non-YOY arctic charr captured at the nearshore of Camp Lake in 2018 were significantly shorter (-28%), lighter (-56%) and of lower condition (-9%) than those captured during the 2013 baseline study (Table 3.9; Appendix Table G.7). Similar differences in nearshore non-YOY arctic charr size and condition were indicated at Camp Lake from 2015 to 2017 compared to the 2013 baseline data (Table 3.9). In all studies from 2015 to 2018, the magnitude of difference in non-YOY arctic charr condition compared to the 2013 baseline data was just within the CES_C of ±10% (Table 3.9). This suggested that the differences in non-YOY arctic charr energy use in each year of mine operation compared to the baseline period was within

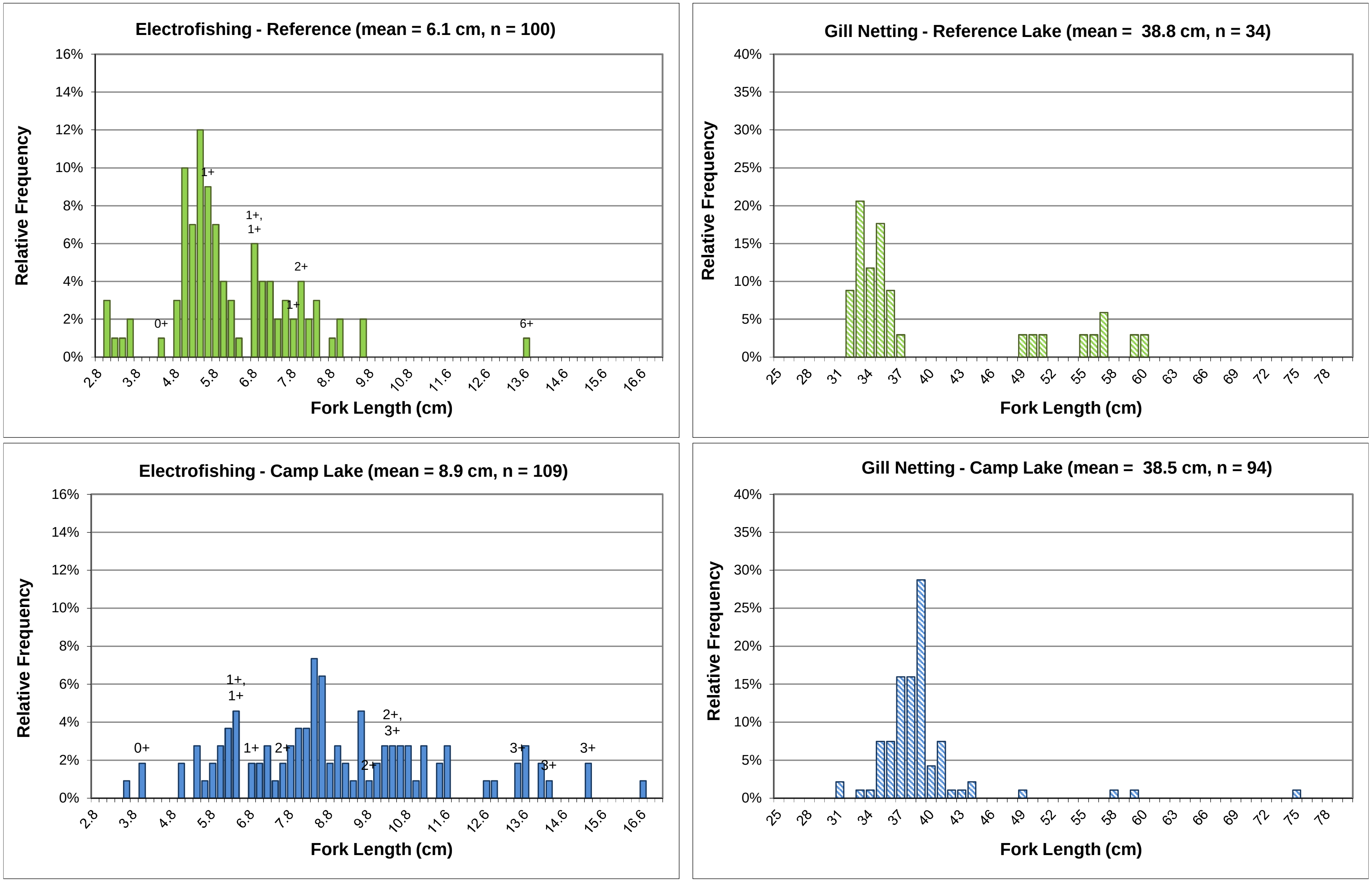


Figure 3.17: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2018

Note: Fish ages are shown above the bars, where available.

Table 3.9: Summary of Statistical Results for Arctic Charr Population Comparisons between Camp Lake and Reference Lake 3 from 2015 to 2018, and between Camp Lake Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]							
			versus Reference Lake 3				versus Camp Lake baseline period data [b]			
			2015	2016	2017	2018	2015	2016	2017	2018
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	**Yes (+41%)**	No	**Yes (+17%)**	**Yes (+40%)**	**Yes (-15%)**	**Yes (-32%)**	**Yes (- 35%)**	**Yes (-28%)**
		Size (mean weight)	**Yes (+176%)**	No	**Yes (+51%)**	**Yes (+135%)**	**Yes (-42%)**	**Yes (-71%)**	**Yes (-74%)**	**Yes (-56%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	No	**Yes (-6%)**	No	**Yes (-14%)**	**Yes (-6%)**	**Yes (-10%)**	**Yes (-10%)**	**Yes (- 9%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	-	-	-	-	**Yes (+48%)**	**Yes (+58%)**	**Yes (+ 46%)**	**-**
	Energy Use	Size (mean fork length)	-	-	-	**Yes (+10%)**	**Yes (+6%)**	No	**Yes (+12%)**	**Yes (+15%)**
		Size (mean weight)	-	-	-	**Yes (+46%)**	No	No	**Yes (+37%)**	**Yes (+46%)**
		Growth (fork length-at-age)	-	-	-	-	No	**Yes (nc)**	No	-
		Growth (weight-at-age)	-	-	-	-	No	**Yes (nc)**	No	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+12%)**	No	**Yes (-3%)**	No	No

[a] Values in parentheses indicate direction and magnitude of any significant differences.
[b] Baseline period data included 2013 nearshore electrofishing data and 2006, 2008, and 2013 littoral/profundal gill netting data. nc = non-calculable magnitude.
[c] Due to low catches of arctic charr in gill nets at Reference Lake 3 in 2015, 2016, and 2017, no comparison of fish health was conducted for gill netted fish.

the range of variability expected to occur naturally between years at waterbodies uninfluenced by human activity. No consistent differences in nearshore arctic charr size or condition were indicated between Camp Lake and Reference Lake 3 from 2015 to 2018, but in instances in which differences occurred, arctic charr from Camp Lake tended to be significantly larger and of lower condition (Table 3.9). Notably, nearshore arctic charr sampled at Reference Lake 3 were significantly larger and of greater condition in 2018 compared to all previous years from 2015 to 2017, the magnitude of difference in condition of which was near or outside of the CES_C of ±10% (Appendix Table B.12). This suggested that year-to-year variability in condition of nearshore arctic charr can be naturally high at local study area lakes.

Littoral/Profundal Arctic Charr

Mine-related influences on the Camp Lake littoral/profundal arctic charr population (i.e., fish captured by gill netting) was assessed based on a control-impact analysis using 2018 data from Camp Lake and Reference Lake 3, as well as a before-after analysis of Camp Lake 2018 versus baseline (combined 2006, 2007, 2008, and 2013) data. A total of 94 and 34 arctic charr were sampled from littoral/profundal habitat of Camp Lake and Reference Lake 3, respectively, in August 2018, for the control-impact analysis. The length-frequency distribution for littoral/ profundal arctic charr differed significantly between Camp Lake and Reference Lake 3, reflecting the occurrence of relatively larger fish at Camp Lake (Table 3.9; Figure 3.17). Littoral/profundal arctic charr captured at Camp Lake were significantly longer (10%) and heavier (46%) than those captured at Reference Lake 3 (Table 3.9; Appendix Table G.12). In addition, the condition of arctic charr captured at littoral/profundal areas of Camp Lake was significantly higher, and at an ecologically meaningful absolute magnitude greater than 10%, than those sampled at Reference Lake 3 (Table 3.9; Appendix Table G.12).

Temporal comparisons of arctic charr data collected from Camp Lake littoral/profundal areas indicated significantly different length-frequency distribution of arctic charr in 2018 compared to the combined baseline data set (i.e., 2006, 2007, 2008, and 2013 studies; Table 3.9). Although fork length and fresh body weight were significantly greater for arctic charr captured at Camp Lake in 2018 compared to the baseline period, no significant difference in condition was indicated between 2018 and the baseline period at Camp Lake (Table 3.9). The 2018 comparisons to baseline conditions were generally consistent with those of the three previous CREMP studies, and collectively indicated no ecologically meaningful differences in condition of spawning-sized arctic charr at Camp Lake between the mine operational years and the baseline period.

3.3.7 Integrated Summary

Potential mine-related influences on water quality of Camp Lake in 2018 included slightly elevated chloride, manganese, and uranium concentrations compared to Reference Lake 3, as well as slightly higher conductivity, hardness, and concentrations of chloride, manganese, molybdenum, sodium, strontium, sulphate, and uranium, compared to Camp Lake baseline data. However, parameter concentrations at Camp Lake were consistently well below WQG and AEMP benchmarks from 2015 to 2018.[9] In sediment of Camp Lake, arsenic and manganese concentrations were elevated at the single littoral station compared to Reference Lake 3 in 2018 and to concentrations observed during the baseline period. However, no metals were elevated in sediment at Camp Lake profundal stations compared to Reference Lake 3 in 2018, nor to concentrations shown in mine baseline studies. Iron and manganese were observed at concentrations above SQG at the Camp Lake littoral station and on average at profundal stations, but average concentrations of these metals were also above SQG at Reference Lake 3 indicating natural elevation of these metals in sediments of regional lakes. Within Camp Lake, arsenic, copper, iron, nickel, and phosphorus concentrations were above AEMP benchmarks at the lone littoral station, as were arsenic, copper, and nickel concentrations in sediment at profundal stations located closest to the CLT1 outlet in 2018. Average concentrations of arsenic and copper were also above the Camp Lake AEMP benchmarks in profundal sediment at Reference Lake 3, indicating naturally high concentrations of these metals in sediment of local study area lakes. Overall, recent mine operations appeared to contribute to higher chloride, manganese, molybdenum, sodium, sulphate, and uranium concentrations in water, as well as to slightly higher arsenic, nickel, and phosphorus concentrations in sediment of Camp Lake. However, concentrations of these parameters generally remained below applicable water or sediment quality guidelines from 2015 to 2018 with the exception of nickel, which was slightly above the SQG at the littoral station.

Camp Lake chlorophyll-a concentrations were significantly higher than at Reference Lake 3 in 2018 suggesting greater primary production at Camp Lake. However, Camp Lake chlorophyll-a concentrations remained well below the AEMP benchmark during all seasonal sampling events, and suggested oligotrophic conditions typical of Arctic waterbodies. Temporal evaluation of the chlorophyll-a data suggested higher chlorophyll-a concentrations in 2018 compared to previous studies, but no changes to the trophic status of Camp Lake since mine operations commenced at the Mary River Project. Benthic invertebrate density and richness were significantly higher in littoral and/or profundal habitat of Camp Lake compared to Reference Lake 3 in 2018, but similar relative abundance of dominant taxonomic groups, including metal-sensitive chironomids, was indicated between lakes. No ecologically significant differences in density, richness, Simpson's Evenness, dominant taxonomic groups, or FFG were consistently indicated between the mine

[9] Total phenol and phosphorus concentrations were above WQG near the bottom of two stations at Camp Lake in 2018, but appeared to be anomalies (see Section 3.3.2).

baseline (2007, 2013) period and all individual years of mine operation from 2015 to 2018. Analysis of Camp Lake arctic charr populations suggested greater fish abundance compared to Reference Lake 3 in 2018, and no decline in the numbers of arctic charr in 2018 compared to the Camp Lake baseline studies. Although arctic charr captured at the nearshore of Camp Lake exhibited significantly lower condition compared to those captured at Reference Lake 3 in 2018, as well as to those captured at Camp Lake during the mine baseline studies, the magnitude of these differences were generally within the range of variability expected to occur naturally (i.e., ±10% of reference condition). Spawning-sized arctic charr captured at Camp Lake showed significantly greater condition than those captured at Reference Lake 3, but were similar in condition to those captured at Camp Lake during baseline studies. Overall, the chlorophyll-a, benthic invertebrate community, and arctic charr fish population data all suggested no adverse mine-related influences to the biota of Camp Lake since the commencement of commercial mine operation at the Mary River Project in 2015.