APPENDIX E.9

AQUATIC EFFECTS MONITORING REPORTS

APPENDIX E.9.1

2017 CREMP MONITORING REPORT

(Part 1)

Mary River Project 2017
Core Receiving Environment Monitoring Program Report

Part 1 of 3
(Chapters 1 - 3)

Prepared for:
Baffinland Iron Mines Corporation
Oakville, Ontario

Prepared by:
Minnow Environmental Inc.
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Mary River Project 2017
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 European 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 waste rock stockpile and surface runoff collection ponds currently situated near the mine waste rock stockpile and ore stockpile areas. In addition to periodic discharge of treated effluent from the mine waste rock stockpile 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 terms and conditions of a 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 potential mine-related influences on water quality, sediment quality, and/or 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 2017 CREMP indicated some mine-related influences on water and sediment quality of a few of the mine primary receiver systems, but ecologically significant mine-related effects to biota were generally restricted to only a single tributary within the Sheardown Lake system. Within the Camp Lake system, mine-related effects on water quality were apparent as elevated concentrations of copper (CLT1 north branch only), iron, nitrate, and sulphate (CLT1 main stem only), and chloride, manganese, molybdenum, sodium, and uranium at Camp Lake Tributary 1 and Camp Lake based on comparison to reference conditions and 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 pit) in the watershed was likely a key 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 2017, which was consistent with concentrations of most metals below 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, where aqueous concentrations of nitrate, sodium, and sulphate were elevated compared to concentrations at reference areas and during applicable baseline studies. Sedimentation was evident as deposits of reddish-brown silt material characterized by iron concentrations above SQG at Sheardown Lake tributaries 1 and 12, but within lake environments, only manganese concentrations were elevated in sediment of Sheardown Lake SE compared to reference lake and baseline conditions. Sheardown Lake Tributary 12 was the only waterbody sampled under the CREMP at which differences in benthic invertebrate community structure could be definitively linked to a mine-related influence. At this tributary, changes in the benthic invertebrate community assemblage relative to reference conditions and baseline studies appeared to be related to potential flow reduction and/or sedimentation. No adverse effects to phytoplankton, benthic invertebrates, or arctic charr were indicated at mine-exposed areas of Sheardown Lake tributaries 1 and 9, Sheardown Lake NW, or Sheardown Lake SE in 2017, which was consistent with concentrations of most metals below applicable WQG and SQG at these waterbodies.

Within the Mary River/Mary Lake system, mine-related effects on water quality were apparent only as elevated concentrations of ammonia, nitrate, and sulphate at a tributary of Mary River (MRTF) that receives treated mine effluent, whereas mine-related effects on sediment quality only included slight elevation of manganese concentrations at Mary Lake. However, no adverse effects to phytoplankton, benthic invertebrates, or arctic charr were indicated at mine-exposed areas of MRTF, Mary River, or Mary Lake in 2017 which, similar to the other mine receiving systems, was consistent with concentrations of most metals below applicable WQG and SQG.

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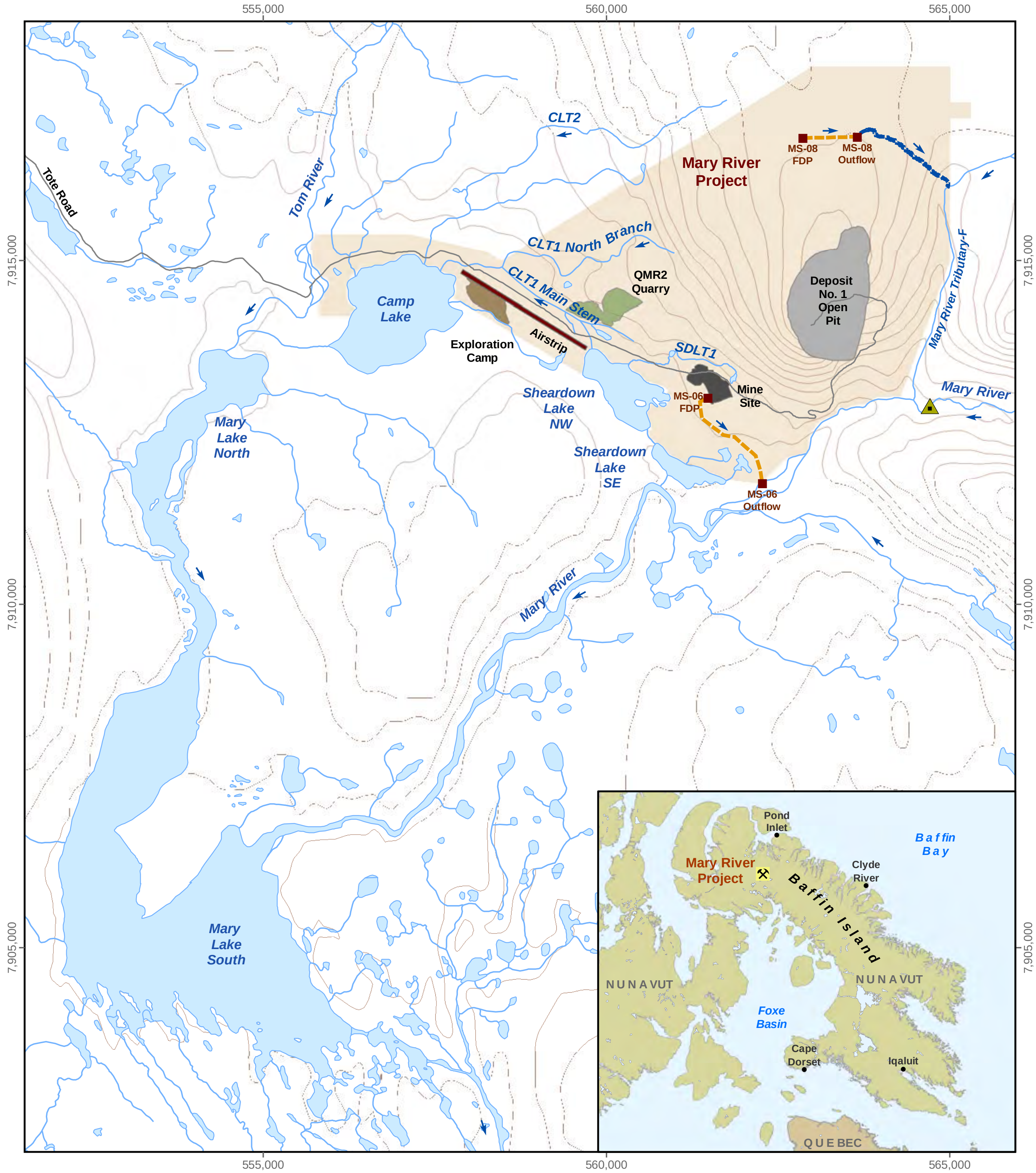
1 INTRODUCTION

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 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 European 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. The mine waste management facilities include a mine waste rock stockpile and surface runoff collection ponds currently situated near the mine waste rock stockpile and ore stockpile areas. In addition to periodic discharge of treated effluent from the mine waste rock disposal area to the Mary River system, other potential mine inputs to aquatic systems located adjacent to the mine include runoff and dust from ore (crusher) stockpiles located on the mine site within the Sheardown Lake catchment, treated mine camp sewage effluent discharge to Mary River, runoff and explosives residue from quarry operations to the Camp Lake catchment, deposition of fugitive dust generated by mine activities, and 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 2014; KP 2014a; NSC 2014). The primary receiving systems that serve as 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 two years of mine operation, the CREMP studies indicated some effects of Baffinland 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). No adverse mine-related effects to phytoplankton, benthic invertebrate, or fish were suggested at any of the Camp, Sheardown, or Mary lake systems in 2015 or 2016 based on comparisons to



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

N E S W

Map Projection: UTM Zone 17N NAD 1983
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Figure 1.1

Document Path: S:\Projects\177202\177202.0033 - Baffinland 2017\2 - Mapping\CREMP Maps\17-33 Fig 1.1 Location.mxd representative reference waterbodies and to available pre-mine baseline data for each lake system (Minnow 2016a, 2017).

This report presents the methods and results of the 2017 CREMP, including an evaluation of potential Mary Lake Project-related influences on chemical and biological conditions at mine-exposed waterbodies through the third full year of mine operation. As in the two previous studies, the 2017 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 CREMP was designed as an iterative series of monitoring and interpretative phases, with the results of previous studies used to inform the direction of future monitoring. The 2017 CREMP was implemented in accordance with the original study design (Baffinland 2014) with the exception of the continued use of a reference creek benthic invertebrate community study area that was originally added to the program in 2016 (Minnow 2016a, 2017). Data and results of federal Environmental Effects Monitoring (EEM) conducted at the Mary River Project in 2017 have also been included in this CREMP report to consolidate all pertinent biological information and assist with the evaluation of potential mine-related effects.

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 2014). In 2017, water quality and phytoplankton monitoring was conducted by Baffinland environment department personnel over four separate sampling events, including an ice-cover event (April 11th - 17th) and open-water season events corresponding to Arctic spring (freshet), summer, and autumn (July 7th - 9th, July 21st - August 10th, and August 20th - September 2nd, 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 16th - 30th 2017, the seasonal timing of which was consistent with monitoring conducted for previous baseline (2005 – 2013), mine construction (2014), and mine operational (2015, 2016) studies. Similar to previous CREMP studies, the 2017 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 2017 CREMP study areas included the same mine-exposed and reference waterbodies established in the original design documents (Baffinland 2014; KP 2014a; NSC 2014) and the same reference 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 2017 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 any potential 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

LEGEND

- Reference Stream/River System
- Mine Exposed Stream/River System
- Reference Lake
- Mine Exposed Lake
- QMR2 Quarry
- Airstrip
- Exploration Camp
- Mine Site
- Open Pit
- Mary River Project

Mary River Project CREMP Study Water Bodies

0 3.25 6.5 13
km

N W E S

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Figure 2.1

Document Path: S:\Projects\177202\177202.0033 - Baffinland 2017\2 - Mapping\CREMP Maps\17-33 Fig 2.1 Study Waterbodies.mxd current Baffinland mine activity (i.e., GO-09) served as a reference area for the mine-exposed portion of Mary River in the 2017 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 2014; KP 2014a). 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.

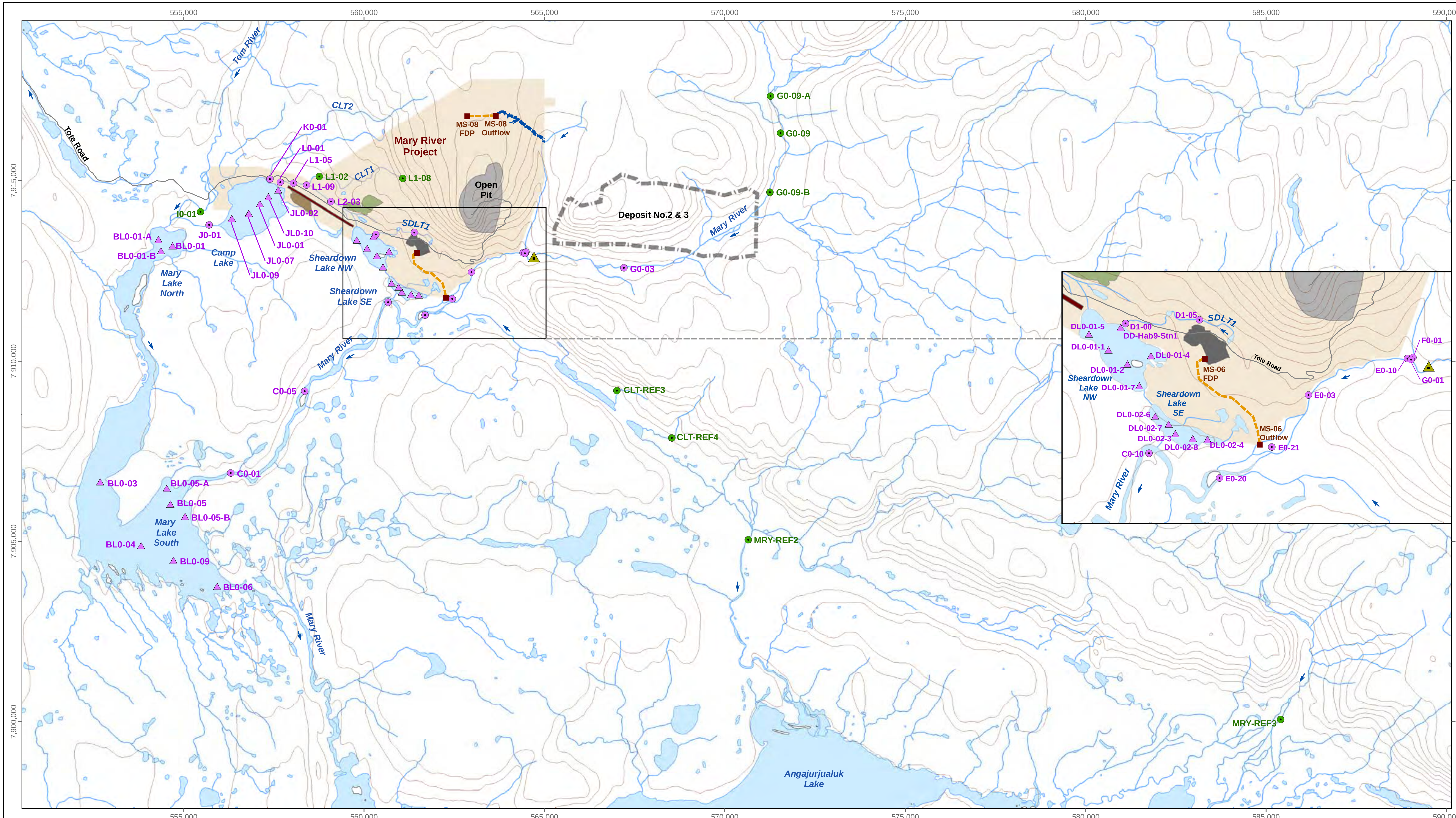
2.2.2 *In Situ* Water Quality Measurement Data Collection and Analysis

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 a YSI ProDSS (Digital Sampling System) meter equipped with a 4-Port sensor and, for top-bottom measurements collected at lake benthic invertebrate community stations, a YSI 556 MDS (Multiparameter Display System) meter equipped with YSI 6820 sonde (YSI Inc., Yellow Springs, OH). Meter readings for pH, specific conductance, and turbidity were checked against standard solutions and calibrated as necessary on the day of field sampling. Dissolved oxygen concentration readings were checked and calibrated at 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 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	Lotic 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

N
W E
S

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

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Figure 2.2

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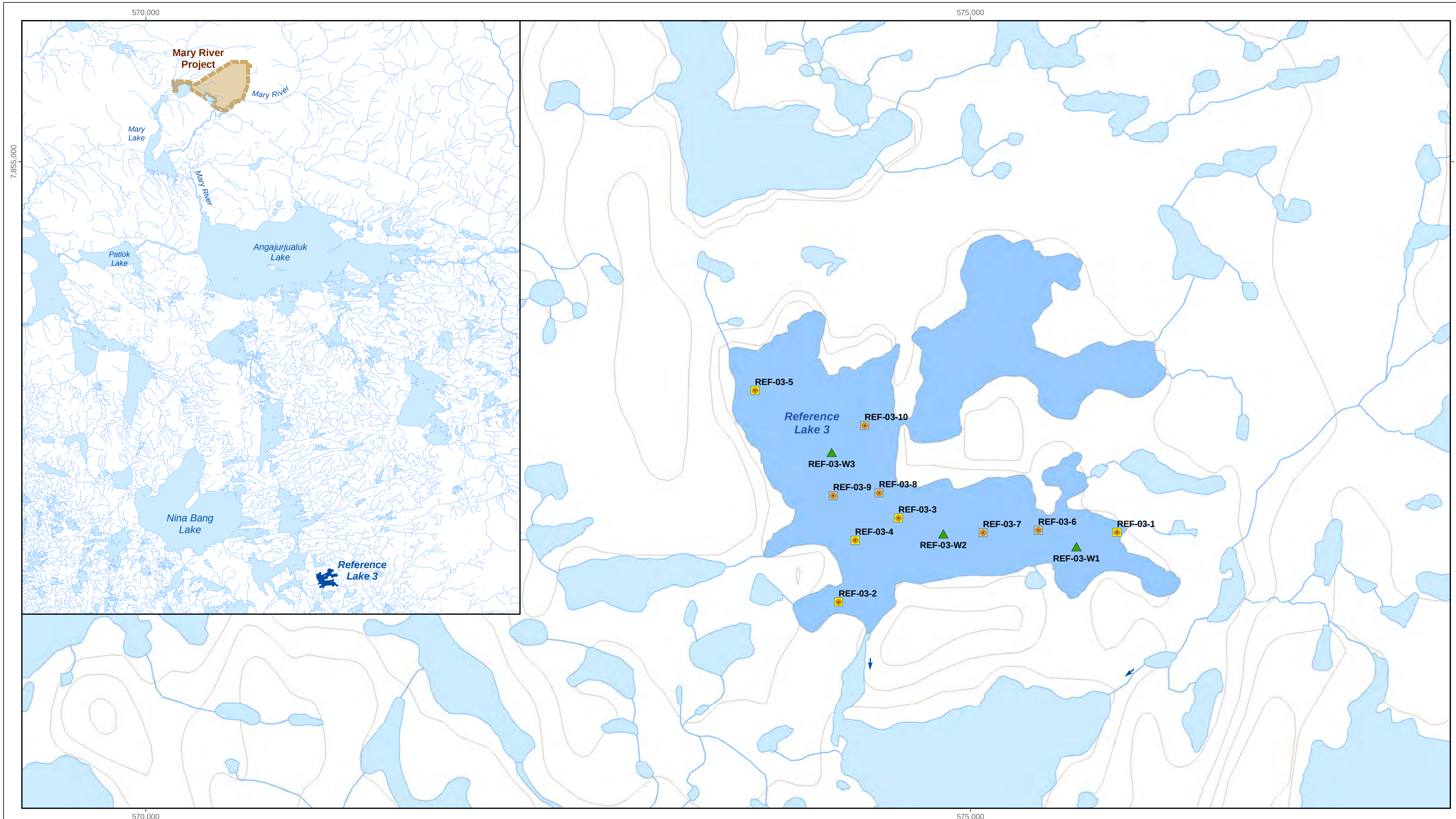

LEGEND

- 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

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Mary River Project CREMP Reference Lake 3 Monitoring Station Locations

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Figure 2.3

Document Path: S:\Projects\177202\177202.0033 - Baffinland 2017\2 - Mapping\CREMP Maps\17-33 Fig 2.3 Reference Lake 3.mxd

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]; pH and dissolved oxygen concentrations 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) or 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, dissolved oxygen, or chemical (i.e., pH and/or 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 the reference lake for each seasonal monitoring event using profile data averaged for each incremental depth below the water surface among lake stations by season. 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 WQG[1].

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[1] Canadian Environmental Quality Guidelines for the protection of aquatic life (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 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 non-metallic, 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 pre-dosed with required chemical preservatives, where appropriate, except those requiring field filtration. For samples that required filtration (e.g., for dissolved metals), filtration was conducted in the field using methods consistent with AEMP standard operating procedures (Baffinland 2014).

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.

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 2017 mean concentrations by the baseline (2005 - 2013) mean concentration for each parameter. The resulting magnitude of dissimilarity in parameter

Table 2.2: Water Quality Guidelines Used for the Mary River Project 2015 to 2017 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 Environment 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.

concentrations were 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 mine-exposed area versus the reference area or baseline data, as applicable.

Applicable WQG included the Canadian Water Quality Guidelines for the protection of aquatic life (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 be 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 2014). 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 2014). 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 2017 compared to baseline (2005 – 2013) and previous mine construction (2014) and operational (2015, 2016) studies.

Correlation analysis was conducted to evaluate potential relationships between physical variables known to affect water quality (turbidity, TSS) and total metal concentrations. Spearman Rank correlation coefficients (r_s) were generated from untransformed water quality data collected at each waterbody of interest over the entire 2017 sampling season using SPSS Version 12.0 software. For correlations shown to be significant (i.e., p-value ≤ 0.05), the relative strength of the relationship was described according to definitions provided by Fowler et al. (1998). Particular emphasis was placed on correlations that were considered strong (absolute r_s ranging from 0.70 to 0.89) to very strong (absolute r_s ranging from 0.90 to 1.00) for identifying potential causal associations.

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 2014; KP 2014a, 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 is the value that was used to define lake zonation during the baseline characterization studies (KP 2014a, 2015). Sediment quality sampling was also conducted at up to five stations from each of the eight stream and five river study areas used to assess mine-related effects to benthic invertebrate communities (Table 2.5; Figure 2.4). This level of effort exceeded the requirements for sampling at the Camp Lake tributaries (three stations), Sheardown Lake tributaries (six stations), and Mary River (four stations) outlined in the original CREMP design (KP 2014a). Notably, all stream and river study areas were previously observed to contain limited depositional habitat (KP 2015; Minnow 2016a,b, 2017) and a general absence of any substantial accumulation of fine sediments, and therefore sediment sampling for chemical characterization was generally restricted to shoreline and interstices of large, coarse substrate material (e.g., cobbles, boulders) within the applicable study areas.

2.3.2 Sample Collection and Laboratory Analysis

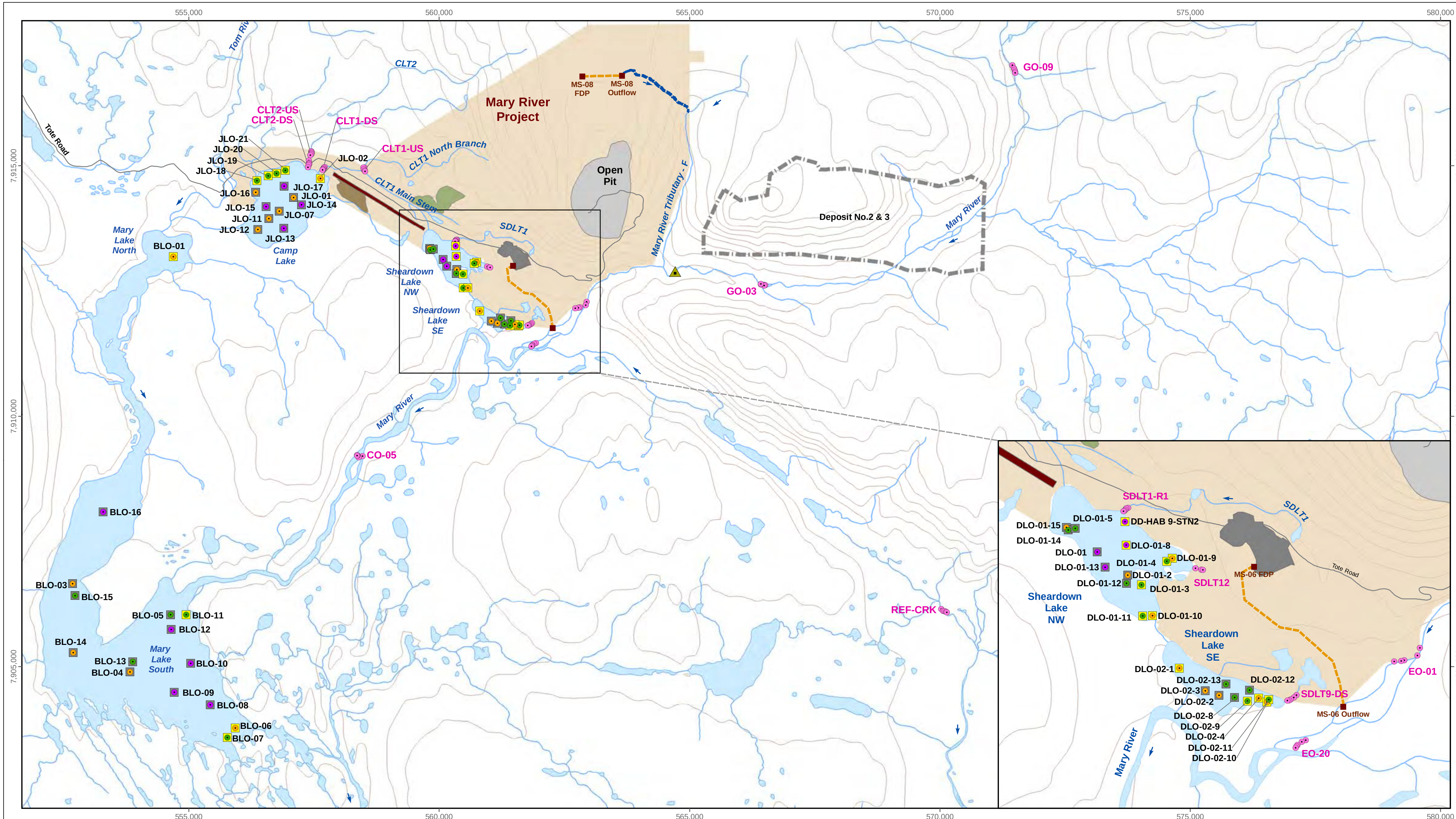
Sediment at study lakes was collected for physical and chemical characterization using a gravity corer (Hoskin Scientific Ltd., Model E-777-00) outfitted with a clean 5.1 cm inside-diameter polycarbonate tube. From each retrieved core sample containing an intact, representative sediment-water interface, the surficial two centimetres of sediment was manually extruded

Table 2.4: Lake Sediment Quality and Benthic Invertebrate Community Monitoring Station Coordinates Used for the Mary River Project CREMP 2017 Study

Waterbody	Station Code	UTM Zone 17W		Sampling Habitat	Sample Type		
		Easting	Northing		Sediment Core[a]	Sediment petite-Ponar[a]	Benthic Invertebrate
Reference Lake	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 Southwest (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 sample analyzed for particle size only.

[b] Duplicate sediment core sample collected for quality control/quality assurance (QA/QC).



LEGEND

- Lake - Benthic Only Sampling Location
- Lake - Sediment Only Sampling Location
- Lake - Sediment and Benthic Sampling Location
- Littoral Sampling Depth
- Profundal Sampling Depth
- Stream - Sediment and Benthic Sampling Location
- 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
- Contours (20 m)

0 1 2 4
km

N E S W

Map Projection: UTM Zone 17N NAD 1983
Data Source: Reproduced under licence from Her Majesty the Queen in Rights of Canada, Department of Natural Resources Canada. All rights

Mary River Project 2017 CREMP Mine Area Sediment Quality and Benthic Station Locations

Date: March 2018
Project 177202.0033

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Figure 2.4

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Table 2.5: Stream and River Benthic Invertebrate Community Monitoring Station Coordinates Used for the Mary River Project CREMP 2017 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 Southwest (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

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 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 any 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 twice the number of replicate core samples taken (Table 2.4; Appendix A). Sediment at stream/river habitats was collected for chemical characterization using a stainless steel scoop. At each station, sediment sampling focused on those locations containing the finest grain size available, including the channel margins and downstream of/underneath large boulders within the active channel. One sample, representing a composite of a variable number of scoopfuls, was collected directly into a pre-labelled 0.95 L plastic sealable bag at each station. 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 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. Deposited sediment quality data from mine-exposed stream/river study areas were compared only to applicable reference area data and SQG because no AEMP benchmarks or baseline data were available. Sediment physical characteristics (i.e., moisture, particle size) and TOC data were statistically summarized based on separate calculation of mean, standard deviation, standard error, minimum and maximum 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 2.1.1).

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 the magnitude of 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 2.1.2; Table 2.3). The same approach was applied to the stream/river sediment chemistry data using applicable mine-exposed and reference area sediment chemistry data.

Sediment chemistry data were compared to applicable Canadian Sediment Quality Guidelines (CSQG; CCME 1999, 2017) 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 2017 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 2017 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 2014). 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 2014).

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 2017 data, baseline (2005 – 2013) data, and previous 2015 - 2016 mine operation period data. In addition, as described previously, the magnitude of elevation was calculated for all parameters between 2017 and baseline data for each individual study lake using the same calculation (and categorization description) as described previously (Section 2.1.2; Table 2.3).

Notably, the applicability of lotic sediment chemistry monitoring data to the interpretation of lotic benthic invertebrate community data was considered minimal given the fact that fine sediment composes much less than 5% of available substrate at the lotic environments (extrapolation of the data suggests that silt and clays compose less than 0.5% of available habitat) and that benthic invertebrates collected for the CREMP do not inhabit these fine sediments. By extension, because fish species inhabiting lotic environments of the area largely rely on benthic invertebrates as a food source, the applicability of sediment chemistry monitoring data to understanding effects on fish was also considered minimal. Because sufficient amounts of fine sediment were able to be collected at only 3 of 23 lotic stations during the baseline period (KP 2014a,b), no temporal comparison of the stream/river sediment chemistry data was conducted.

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 abundance of photosynthetic microbiota in aquatic environments (Wetzel 2001). Chlorophyll-a samples were collected by Baffinland environmental department personnel at the same stations and same time as the collection of water chemistry samples (Table 2.1; Figures 2.2 and 2.3). In addition, the water samples for chlorophyll-a analyses were collected using the same methods and equipment as described for water chemistry samples (Section 2.1.2). 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 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 2.1.2).

The CREMP study design also stipulates the collection of phytoplankton community samples for archiving (NSC 2014, 2015a). 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 (i.e., taxonomic composition, richness, density). To date, none of the archived phytoplankton community samples have been processed (2006 - 2016). In 2017, 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 usage.

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 described previously (Section 2.1). An AEMP benchmark chlorophyll-a concentration of 3.7 µg/L was established for the Mary River Project (NSC 2014), and therefore the 2017 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 (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 2017 CREMP, augmenting 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). Environmental Effects Monitoring (EEM) benthic invertebrate community data collected at an unnamed tributary to Mary River (referred to as Mary River Tributary-F [MRTF]) downstream (effluent-exposed) and upstream (reference) of the primary mine effluent discharge have also been included in this CREMP report to consolidate all available benthic information for the mine receiving environment (Appendix Table F.1; Figure 2.4). Consistent with the federal EEM program, five stations were sampled at each CREMP and EEM 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 and 2016, the level of replication used for lotic benthic sampling in 2017 was greater than specified under the original CREMP design in order to provide consistency with EEM standards (Minnow 2016a). To the extent possible, previously established lotic benthic stations were incorporated into the 2017 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 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 in both 2015 and 2016 indicated that, similar to temperate lakes (Ward 1992), depth-related differences in benthic invertebrate community structure (e.g., density and richness) occurs naturally in lakes of the Baffinland region (Minnow 2016a, 2017). Additional sampling conducted at Reference Lake 3 in 2017 confirmed the occurrence of natural depth-related differences in 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 (2-12 m depth) or profundal (>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[3] 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. Furthermore, the sampling of five stations from each zone at each study area ensures provide adequate statistical power to detect ecologically meaningful differences in benthic metrics of ± two standard deviations 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 2017 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

[3] 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.

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 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, substrate size, an estimate of substrate embeddedness, and a description of macrophyte/algae presence. In addition, *in situ* water quality at the bottom of the water column and collection/recording of global positioning system (GPS) coordinates was conducted at each lotic benthic station. Supporting information recorded at each lake benthic station included substrate description, presence of aquatic macrophytes/ 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 2017 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 presented 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 and untransformed, normally distributed data. However, in the event that data were determined to be non-normal, a suite of transformations, including $\log_{10}$, square root, fourth root, and modified probit, was applied to the data and evaluated for normality[4]. The transformation that resulted in the highest p-value from a Shapiro-Wilks test of the residuals from t-tests was applied to the data for statistical testing using ANOVA and *post hoc* tests, as applicable. In instances where normality could not be achieved through data transformation, non-parametric Kruskal-Wallis H-test and Mann-Whitney U-tests were used to validate the statistical results from multiple group and pair-wise ANOVA tests, respectively. All

[4] 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.

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, the magnitude of difference was calculated between study area 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 any 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). The use of a CES_{BIC} of ± 2 SD_{REF} in defining ecologically significant differences or changes over time for the Mary River Project CREMP was further supported by temporal analysis of available site-specific reference creek and reference lake data (Appendix B).

Temporal comparisons included statistical evaluations among the baseline, 2015, 2016, and 2017 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. 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 2017 within-year statistical analyses, the magnitude of difference was calculated for endpoints that differed significantly between years in the *post hoc* tests and 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

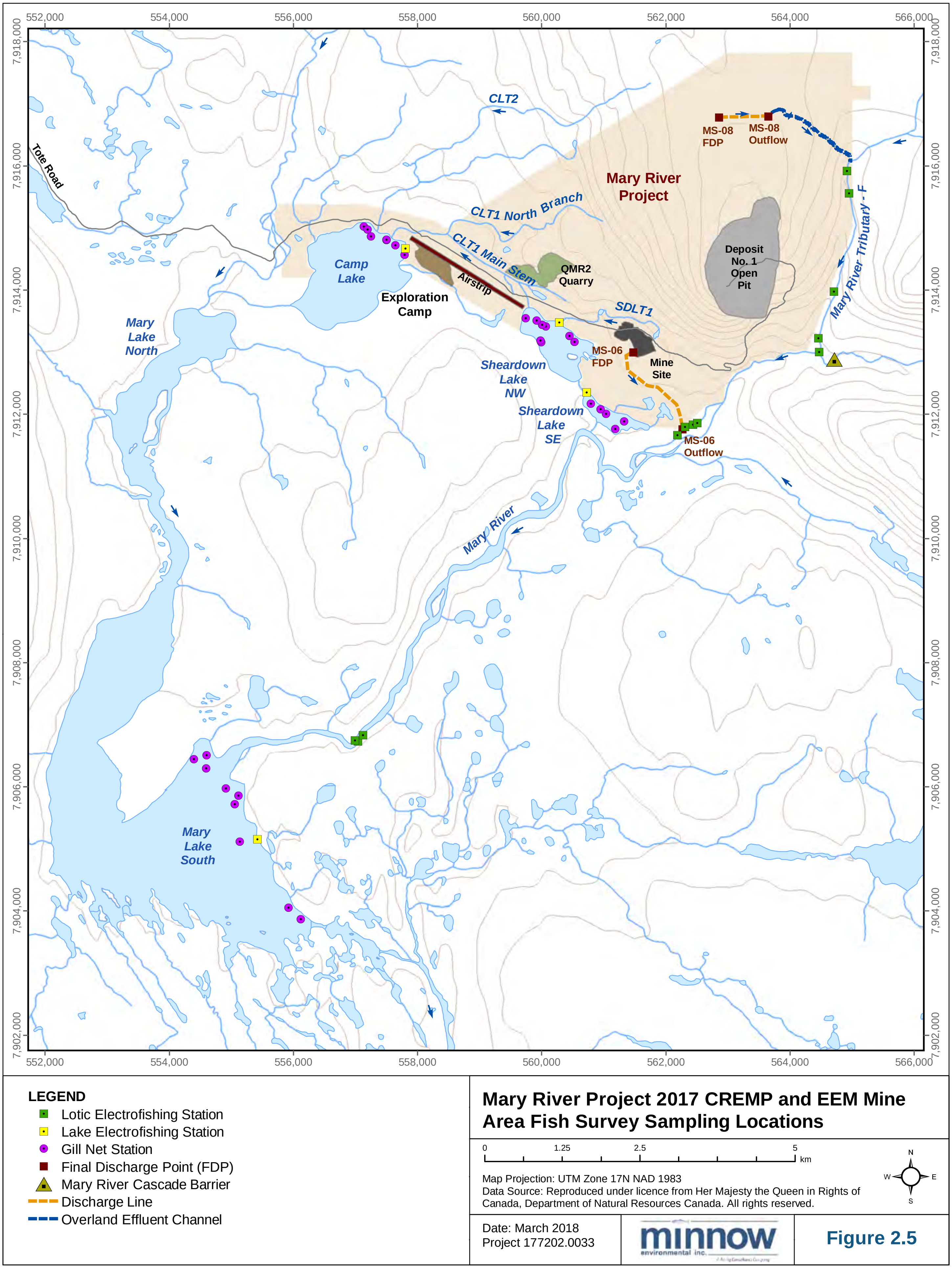
2.4.3.1 General Design

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, 2015a). The fish population survey targeted arctic charr (*Salvelinus alpinus*) primarily because this species is the only abundant fish common to the mine's regional lakes, sufficient baseline catch and measurement data were 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). Although the 2017 study also targeted arctic charr from Reference Lake 3 as a basis for the evaluation of potential mine-related influences to the fish population, similar to the 2015 and 2016 CREMP studies, low numbers of arctic charr were captured from the littoral/profundal zone of the reference lake in 2017. Thus, the 2017 CREMP fish population survey focused on comparisons of fish collected at the nearshore of the mine-exposed and reference lakes, as well as on 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.

In addition to the CREMP data, EEM fish population survey data collected at MRTF and Mary River near- and far-field mine-exposed areas in 2017 have been included in this CREMP report to consolidate all available fish population information applicable to the mine receiving environment (Figure 2.5).

2.4.3.2 Sample Collection

Nearshore areas of the study lakes used for the CREMP study, and streams/rivers used for the EEM 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 cumulative target numbers were not 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.



Document Path: S:\Projects\177202\177202.0033 - Baffinland 2017\2 - Mapping\CREMP Maps\17-33 Fig 2.5 Fish Survey Sampling Locations.mxd

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 – 91 m (200' – 300') and bar mesh sizes ranging from 38 – 76 mm (1.5" – 3") were set on the bottom for short durations (approximately 0.8 – 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 size, 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) or 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 1-2% 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. Aging structures were also removed from all arctic charr that died incidentally during experimental gill net sampling, reflecting approximately 13% of the total catch numbers. Otoliths were removed from all sacrificed individuals and incidental mortalities for age determination. Upon removal, these aging structures were wrapped separately in wax paper, placed inside envelopes labelled with the fish identification, and then dried for storage. For all incidental mortalities from experimental gill netting, in addition to removal of aging structures, fish were dissected to determine sex and for removal of the liver and whole gonads for weight measurement. These organs were weighed to the nearest milligram using an

Ohaus Model 123 Scout-Pro balance outfitted with a surrounding draft shield. During processing, these fish were also inspected for internal abnormalities (e.g., parasites, lesions, tumours, etc.) and descriptions were recorded as appropriate.

Age structures (otoliths) were shipped to North Shore Environmental Services (NSES; Thunder Bay, ON; CREMP samples) or AAE Tech Services Inc. (LaSalle, MB; EEM samples) for age determination. At the laboratory, otoliths were prepared for aging using a “crack and burn” method (NSES) or read whole after polishing three to five seconds with an appropriately graded wet-dry sandpaper and a drop of clove oil to enhance translucency (AAE). 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 the current year of mine operation.

Arctic charr population health was assessed separately for electrofishing and experimental gill netting data sets. Initial data analysis for the non-lethal survey included plotting length frequency distributions as described by Bonar (2002) and Gray et al. (2002), so that, together with appropriate aging data, 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 comparison of fish survey endpoints between the individual mine-exposed lakes and the reference lake (CREMP data) and lotic study areas (EEM data). 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 2017 (CREMP and EEM data),

Table 2.6: Fish Population Survey Endpoints Examined for the Mary River Project CREMP 2017 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.

and for before-after analysis using data collected in 2017 and during the combined baseline period (CREMP data only), 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 2017, and between 2017 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. 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 2017 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 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 2017 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 2017 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 2017 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 formula 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) was 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 – C.3). Dissolved oxygen concentrations and percent saturation at the CLT1 north branch and lower main stem stations were comparable and similar to the reference creek at the time of biological sampling in August 2017 (Figure 3.1; Appendix Table C.13). Notably, DO concentrations were well above the WQG minimum limit for cold-water biota early life stages (i.e., 9.5 mg/L) at all CLT1 stations in 2017 (Appendix Figure C.1; Figure 3.1), suggesting that mine activity had not adversely affected DO concentrations. No consistent spatial patterns in pH were evident 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 – C.3). Although pH was significantly higher at the CLT1 north branch compared to Unnamed Reference Creek, no significant differences in pH were indicated between the CLT1 north branch and lower main stem study areas in August 2017 suggesting no substantial influence of the Tote Road on 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 the CLT1 north branch and Unnamed Reference Creek.

Conductivity of CLT1 was consistently 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 CLT1 system dilution influences and suggesting a mine-related source affecting water quality of the CLT1 upper main stem (Appendix Tables C.1 – C.3, C.14). Although specific conductance was significantly higher at the lower main stem than at the north branch of CLT1, specific conductance at both of these CLT1 study areas was significantly lower than at Unnamed Reference Creek during the August 2017 biological study (Figure 3.1). During spring and summer sampling events, conductivity was clearly elevated at the CLT1 upper and lower main stem stations compared to the CREMP lotic reference stations (Appendix Tables C.1, C.2 and C.14), suggesting that reference area conductivity can vary substantially among seasons.

Water chemistry of the CLT1 north branch was similar to the reference creek stations in 2017 with the exception of a 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). In addition, parameter concentrations were below applicable WQG and watercourse-specific AEMP

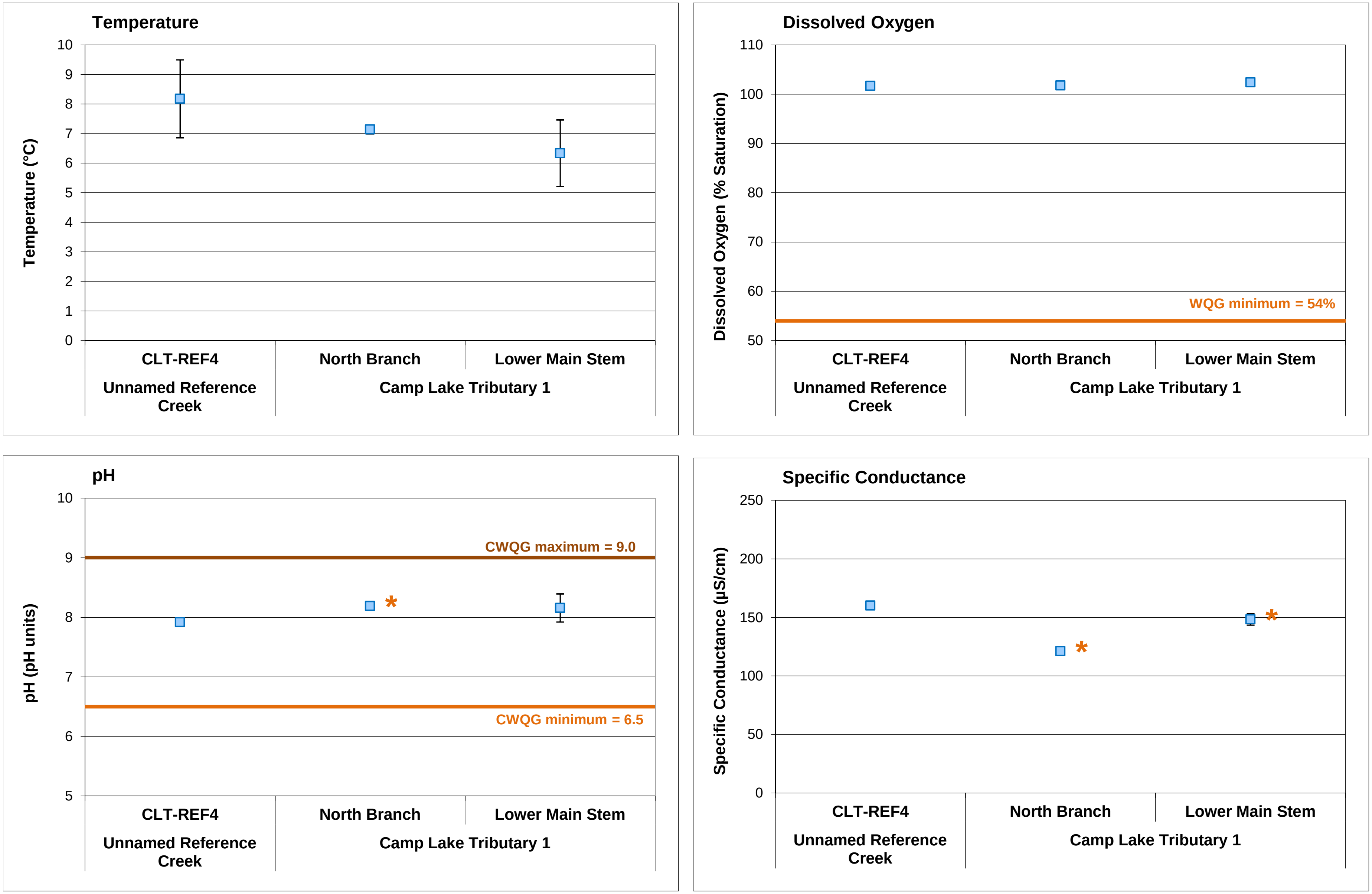


Figure 3.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 1 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

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 (August-September) Sampling, Mary River Project CREMP, 2017

	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 2017	1-Sep-2017	27-Aug-2017	27-Aug-2017	27-Aug-2017	27-Aug-2017	27-Aug-2017	27-Aug-2017
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	116	135	188	336	231	245	255	253
	pH (lab)	pH	6.5 - 9.0	-	7.90	8.03	8.13	8.18	8.14	8.18	8.09	8.20
	Hardness (as $CaCO_3$)	mg/L	-	-	55	65	94	153	114	117	124	129
	Total Suspended Solids (TSS)	mg/L	-	-	4.1	<2.0	<2.0	8.8	2.7	3.7	15.7	2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	60	73	96	169	90	114	152	134
	Turbidity	NTU	-	-	6.1	0.7	1.1	17.3	4.1	4.3	18.9	1.8
	Alkalinity (as $CaCO_3$)	mg/L	-	-	52	63	84	140	102	110	109	102
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.0265	<0.020	<0.020	0.032	<0.020	0.066	<0.020	<0.020
	Nitrate	mg/L	13	13	0.063	0.041	0.109	0.539	0.180	0.161	0.164	0.111
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	<0.15	0.24	0.47	0.30	0.22	0.17	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.6	2.3	2.6	5.0	3.3	3.5	3.4	2.6
	Total Organic Carbon	mg/L	-	-	1.7	2.1	2.8	5.1	3.5	4.0	3.5	2.8
	Total Phosphorus	mg/L	0.020[α]	-	0.0078	0.0072	0.0034	0.0096	0.0177	0.0060	0.0126	0.0044
	Phenols	mg/L	0.004[α]	-	<0.0010	0.0015	<0.0010	<0.0010	0.0016	<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	<0.10
	Chloride (Cl)	mg/L	120	120	2.5	2.0	2.6	19.9	8.6	9.8	10.9	3.1
	Sulphate (SO_4)	mg/L	218[β]	218	4.1	4.1	7.3	9.4	7.3	6.0	6.5	26.4
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	***0.208***	0.022	0.032	***0.395***	0.100	0.179	***0.620***	0.063
	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	0.00012	<0.00010	<0.00010	0.00014	<0.00010	<0.00010	0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0076	0.0089	0.0112	0.0153	0.0132	0.0139	0.0175	0.0126
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00010	<0.00050	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Bismuth (Bi)	mg/L	-	-	<0.000050	<0.00050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	0.013	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	11.2	12.7	19.0	29.8	22.6	23.5	23.7	25.3
	Chromium (Cr)	mg/L	0.0089	0.0089	0.00074	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	0.00060	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.0040	0.00015	<0.00010	<0.00010	0.00034	0.00012	0.00016	0.00033	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.0013	***0.0025***	0.0021	0.0016	0.0019	0.0020	0.0022	0.0011
	Iron (Fe)	mg/L	0.30	0.326	0.222	<0.030	<0.050	***0.705***	0.204	0.323	***0.821***	0.082
	Lead (Pb)	mg/L	0.001	0.001	0.00021	<0.000050	<0.000050	0.00060	0.00015	0.00022	0.00065	0.00006
	Lithium (Li)	mg/L	-	-	0.0014	<0.0010	<0.0010	0.0031	0.0021	0.0025	0.0030	0.0016
	Magnesium (Mg)	mg/L	-	-	6.5	7.9	11.4	19.9	13.7	14.6	15.0	16.2
	Manganese (Mn)	mg/L	0.935[β]	-	0.0033	0.0006	0.0019	0.0456	0.0142	0.0184	0.0243	0.0029
	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	-	0.00034	0.00074	0.00070	0.00141	0.00087	0.00088	0.00071	0.00035
	Nickel (Ni)	mg/L	0.025	0.025	0.00068	<0.00050	0.00067	0.00135	0.00092	0.00108	0.00144	0.00054
	Potassium (K)	mg/L	-	-	0.8	1.9	1.9	3.0	2.2	2.3	2.5	1.6
	Selenium (Se)	mg/L	0.001	-	<0.000050	<0.0010	<0.000050	0.000078	0.000055	<0.000050	0.000053	<0.000050
	Silicon (Si)	mg/L	-	-	1.3	0.8	1.0	1.8	1.2	1.4	2.0	1.1
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000050	<0.000010	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Sodium (Na)	mg/L	-	-	1.8	0.5	1.4	9.9	3.6	3.9	4.1	2.0
	Strontium (Sr)	mg/L	-	-	0.0113	0.0075	0.0095	0.0256	0.0237	0.0277	0.0274	0.0146
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00001	<0.00010	<0.000010	0.00001	<0.000010	<0.000010	0.00002	<0.000010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	0.0123	<0.010	<0.0020	0.0210	<0.0060	0.0103	0.0418	<0.0040
	Uranium (U)	mg/L	0.015	-	0.0021	0.0029	0.0021	0.0088	0.0033	0.0036	0.0035	0.0019
	Vanadium (V)	mg/L	0.006[α]	0.006	0.0007	<0.0010	<0.00050	0.0008	<0.00050	0.0006	0.0011	<0.00050
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	0.0034	<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.

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

benchmarks at the CLT1 north branch in 2017 except for copper, which was above both criteria at the upper-most station 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 2017 were within the range of those measured during the mine baseline (2005 – 2013) period with the exception of higher total copper concentrations, which were consistently elevated in all years since commercial mine production commenced in 2015 (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.

Water chemistry at the CLT1 main stem indicated hardness and concentrations of total dissolved solids (TDS), nitrate, total Kjeldahl nitrogen (TKN), organic carbon, 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). However, on average, only concentrations of chloride and total manganese were elevated at the CLT1 lower main stem (i.e., stations L1-09, L1-05 and L0-01) compared to respective reference creek station average concentrations (Appendix Table C.14), reflecting natural dilution of the CLT1 main stem from the north branch. Total aluminum and iron concentrations were above respective WQG and watercourse-specific AEMP benchmarks at the CLT1 upper main stem and downstream-most lower main stem station in 2017, but only during the fall sampling event. These metals also occurred at concentrations above WQG and AEMP benchmarks at the MRY-REF3 lotic reference station during the fall 2017 sampling event (Appendix Table B.2). Notably, abnormally high turbidity was evident at the CLT1 main stem and MRY-REF 3 lotic reference stations during the fall sampling event (Table 3.1), suggesting that elevations in total aluminum and iron concentrations during the fall sampling event were related to suspended particulate matter. Of the latter, only dissolved iron concentrations at the CLT1 main stem stations were elevated above average concentrations at the lotic reference area (Appendix Table C.17), suggesting that elevated iron concentrations in CLT1 reflected a mine-related source.

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 2017, 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, sulphate, and total iron, manganese, molybdenum, sodium, and uranium concentrations were consistently higher during the mine operational years, including 2017, compared to the mine baseline period at all four CLT1 main stem stations (Figure 3.2; Appendix

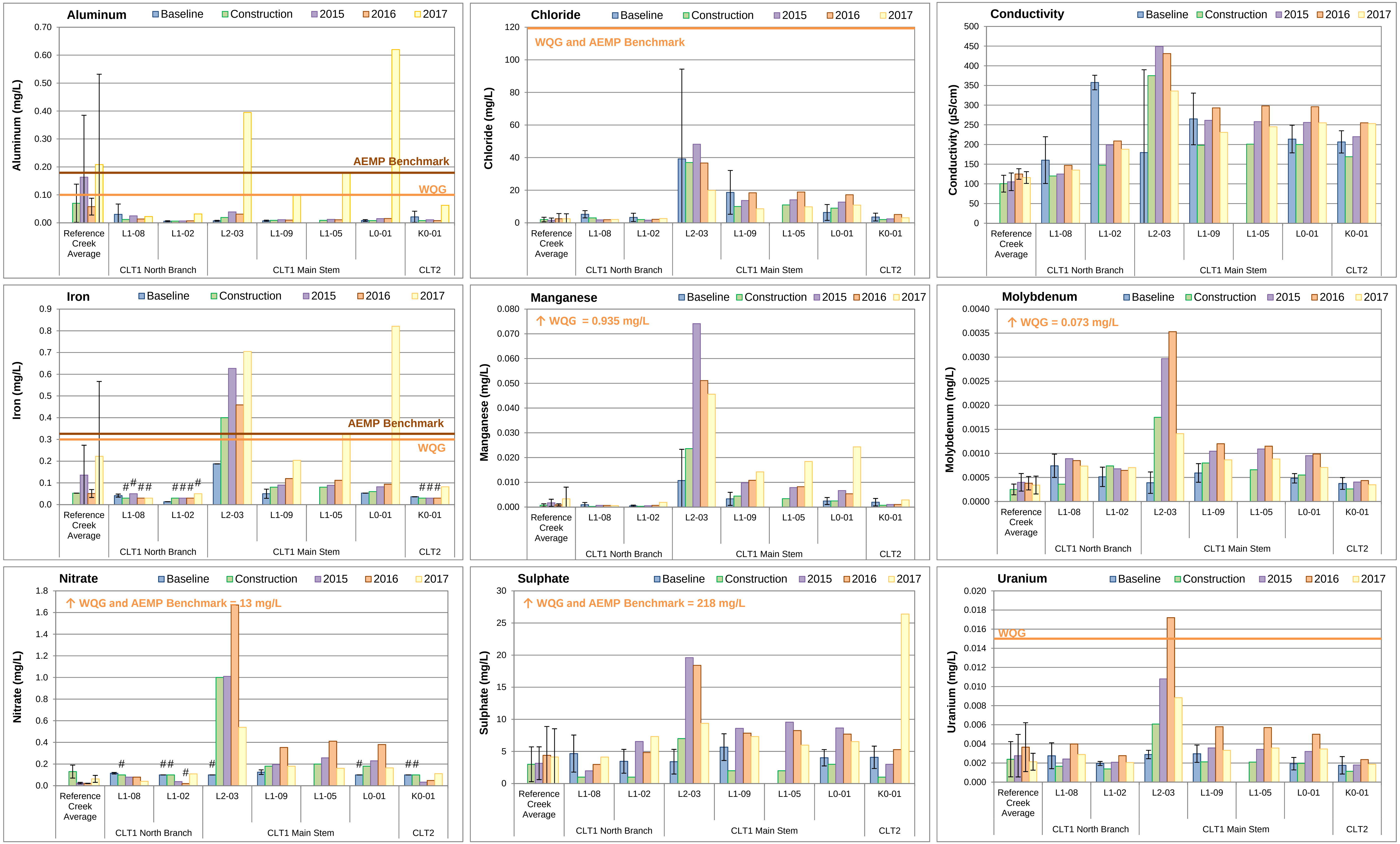


Figure 3.2: Temporal Comparison of Water Chemistry at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for Mine Baseline (2005 - 2013), Construction (2014) and Operational (2015 - 2017) Periods During Fall.

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.

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[5], as well as fugitive dust generation from increased truck usage on the Tote Road, compared to the baseline period. Notably, total aluminum and iron concentrations were particularly elevated at the CLT1 main stem stations in fall 2017 compared to the baseline period and the two previous CREMP studies, but as discussed above, likely reflected elevated suspended particulate concentrations (i.e., turbidity) as similarly observed at the lotic reference stations. This was supported by the evaluation of dissolved metal fractions, which indicated no substantial difference in dissolved aluminum and iron concentrations between 2017 and the baseline period for all spring, summer, and fall sampling events (Appendix Table C.18), suggesting that these parameters were largely associated with suspended materials. Collectively, mine-related influences on water quality of the CLT1 main stem were primarily evidenced by elevated hardness and concentrations of nitrate, TKN, chloride, sulphate, and total metals including manganese, molybdenum, potassium, sodium, strontium, and uranium, at the upper main stem, though none were elevated above applicable WQG or AEMP benchmarks.

3.1.2 Sediment Quality

Deposited sediment at CLT1 upstream (north branch; CLT1-US) and downstream (lower main stem; CLT1-DS) study areas was visually characterized as predominantly coarse sand (Appendix Table D.7). In-stream substrate at both CLT1 study areas was composed mainly of cobble material (i.e., substrate diameter 6 – 25 cm), with sand constituting only a trace amount (i.e., <1%) of the material observed at the sediment surface (Appendix Table F.1). As a result, deposited sediment suitable for chemical characterization (i.e., sand and finer substrate sizes) was collected mainly from shoreline/streambank areas at the upstream north branch study area, and from underneath large cobble/boulders that were manually overturned at the downstream lower main stem study area (Appendix Table D.7). Sediment total organic carbon (TOC) content was low (i.e., <1%) at both CLT1 study areas, but nevertheless was slightly elevated (i.e., 3- to 5-fold higher) at the CLT1 upstream study area compared to average lotic reference conditions suggesting a more depositional environment at the former (Table 3.2; Appendix Table D.10).

Metal concentrations in deposited sediment at CLT1 upstream and downstream study areas were generally elevated compared to respective average lotic reference area metal concentrations (Table 3.2; Appendix Table D.10). Most notably, concentrations of aluminum, chromium, cobalt, copper, iron, magnesium, manganese, molybdenum, nickel, and potassium concentrations were highly elevated (i.e., ≥10-fold higher) in deposited sediment at one or both of the CLT1 study

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

Table 3.2: Sediment Total Organic Carbon and Metal Concentrations at Camp Lake Tributary 1 (CLT1) and Lotic Reference Area Sediment Monitoring Stations, Mary River Project CREMP, August 2017

Parameter	Units	Sediment Quality Guideline (SQG)[a]	Lotic Reference Stations		Camp Lake Tributary 1	
			Unnamed Reference Creek (REFCRK; n = 5)	Mary River Reference (GO-09; n = 5)	Upstream CLT1-US (n = 5)	Downstream CLT1-DS (n = 9)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC	%	10[α]	<0.10 ± 0.00	<0.10 ± 0.00	0.41 ± 0.18	0.23 ± 0.15
Aluminum (Al)	mg/kg	-	418 ± 126	763 ± 271	6,824 ± 2,489	4,426 ± 2,081
Antimony (Sb)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Arsenic (As)	mg/kg	17	0.12 ± 0.02	0.16 ± 0.02	0.65 ± 0.14	0.61 ± 0.15
Barium (Ba)	mg/kg	-	2 ± 0.3	4 ± 1.2	16 ± 5.1	24 ± 11
Beryllium (Be)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	0.27 ± 0.11	0.17 ± 0.08
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0.01	0.26 ± 0.11
Boron (B)	mg/kg	-	<5.0 ± 0	<5.0 ± 0	9.5 ± 4.3	5.0 ± 0.0
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	<0.020 ± 0	0.043 ± 0.013	0.061 ± 0.009
Calcium (Ca)	mg/kg	-	214 ± 44	842 ± 508	2,682 ± 560	2,384 ± 1,217
Chromium (Cr)	mg/kg	90	1.4 ± 0.4	3.4 ± 1.0	28.9 ± 6.9	18.4 ± 4.5
Cobalt (Co)	mg/kg	-	0.32 ± 0.09	0.67 ± 0.19	5.73 ± 1.68	4.10 ± 1.46
Copper (Cu)	mg/kg	110[α]	0.8 ± 0.5	1.3 ± 0.5	18 ± 3.7	13.5 ± 5.1
Iron (Fe)	mg/kg	40,000[α]	1,240 ± 475	2,826 ± 1,271	14,480 ± 2,663	23,240 ± 6,196
Lead (Pb)	mg/kg	91	0.7 ± 0.2	1.1 ± 0.1	3.9 ± 0.9	5.1 ± 2.1
Lithium (Li)	mg/kg	-	<2.0 ± 0.0	2.2 ± 0.5	12.6 ± 5.5	5.3 ± 2.1
Magnesium (Mg)	mg/kg	-	333 ± 116	826 ± 521	8,110 ± 2,920	5,054 ± 2,584
Manganese (Mn)	mg/kg	1,100[α,β]	10 ± 2.4	22 ± 7.1	134 ± 45	195 ± 80
Mercury (Hg)	mg/kg	0.486	0.0050 ± 0	<0.0050 ± 0	0.0067 ± 0.0016	0.0053 ± 0.0007
Molybdenum (Mo)	mg/kg	-	<0.10 ± 0.00	0.11 ± 0.01	0.15 ± 0.05	1.10 ± 0.50
Nickel (Ni)	mg/kg	75[α,β]	0.8 ± 0.2	1.9 ± 0.7	18.6 ± 5.7	14.3 ± 5.2
Phosphorus (P)	mg/kg	2,000[α]	61 ± 16	113 ± 49	227 ± 50	210 ± 72
Potassium (K)	mg/kg	-	106 ± 13	168 ± 77	1,310 ± 580	1,712 ± 784
Selenium (Se)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	0.20 ± 0	0.20 ± 0
Silver (Ag)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	0.10 ± 0	0.10 ± 0
Sodium (Na)	mg/kg	-	<50 ± 0	<50 ± 0	67 ± 17	59 ± 14
Strontium (Sr)	mg/kg	-	1.2 ± 0.2	1.9 ± 0.3	2.7 ± 0.4	2.8 ± 0.8
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	<0.050 ± 0.000	<0.050 ± 0	0.094 ± 0.033	0.093 ± 0.040
Tin (Sn)	mg/kg	-	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0
Titanium (Ti)	mg/kg	-	36 ± 16	109 ± 26	396 ± 100	316 ± 122
Uranium (U)	mg/kg	-	0.2 ± 0.1	0.30 ± 0.0	0.8 ± 0.2	0.84 ± 0.4
Vanadium (V)	mg/kg	-	1.9 ± 0.7	5.0 ± 2.2	26.2 ± 4.6	17.4 ± 4.5
Zinc (Zn)	mg/kg	315	2.0 ± 0.0	3.7 ± 1.3	15 ± 5.1	21 ± 7.5
Zirconium (Zr)	mg/kg	-	1.1 ± 0.1	1.7 ± 0.3	1.9 ± 0.7	2.1 ± 0.4

[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)).

Indicates parameter concentration above Sediment Quality Guideline (SQG).

areas compared to average lotic reference area concentrations (Appendix Table D.10). However, of these metals, only iron, manganese, and potassium, together with barium and zinc, occurred at concentrations ≥1.5 times greater at the downstream area than at the upstream area of CLT1 (Table 3.2), potentially reflecting an additional influence of the Tote Road and other mine sources on metal concentrations of deposited sediment at the lower main stem study area. Interestingly, the spatial pattern in metal concentrations of deposited sediment at CLT1 suggested that the primary source of metals was the north branch watershed, which differed from that shown for influences on water quality where the lower main stem watershed appeared to be the dominant source of metals (Section 3.1.1). Despite elevation in metal concentrations of deposited sediment at CLT1 compared to average lotic reference area conditions, concentrations of all metals were well below applicable Sediment Quality Guidelines (SQG) at all CLT1 upstream and downstream stations (Table 3.2; Appendix Tables D.8 and D.9). No baseline sediment metal concentration data were collected at the CLT1 study areas, and thus no evaluation of potential mine-related influences on sediment quality following commencement of commercial mine operations could be conducted.

3.1.3 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 2017 (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 comparable to reference creek chlorophyll-a concentrations for each individual sampling event, suggesting no marked differences in phytoplankton productivity between the CLT1 north branch and the reference creek stations (Figure 3.3). Within the CLT1 main stem, chlorophyll-a concentrations were highest at upstream-most Station L2-03 during the spring and summer sampling events, but were comparable among the upper and lower main stem stations during the fall sampling event in 2017 (Figure 3.3). Chlorophyll-a concentrations were consistently significantly higher at the CLT1 main stem stations compared to the reference creek stations for each of the spring, summer, and fall sampling events in 2017 (Appendix Table E.2), potentially reflecting an outcome of higher nutrient (e.g., nitrate) concentrations at the CLT1 main stem stations compared to average 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 2017 (Figure 3.3). Similar to the reference creek stations, chlorophyll-a concentrations observed at all CLT1 stations in 2017 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

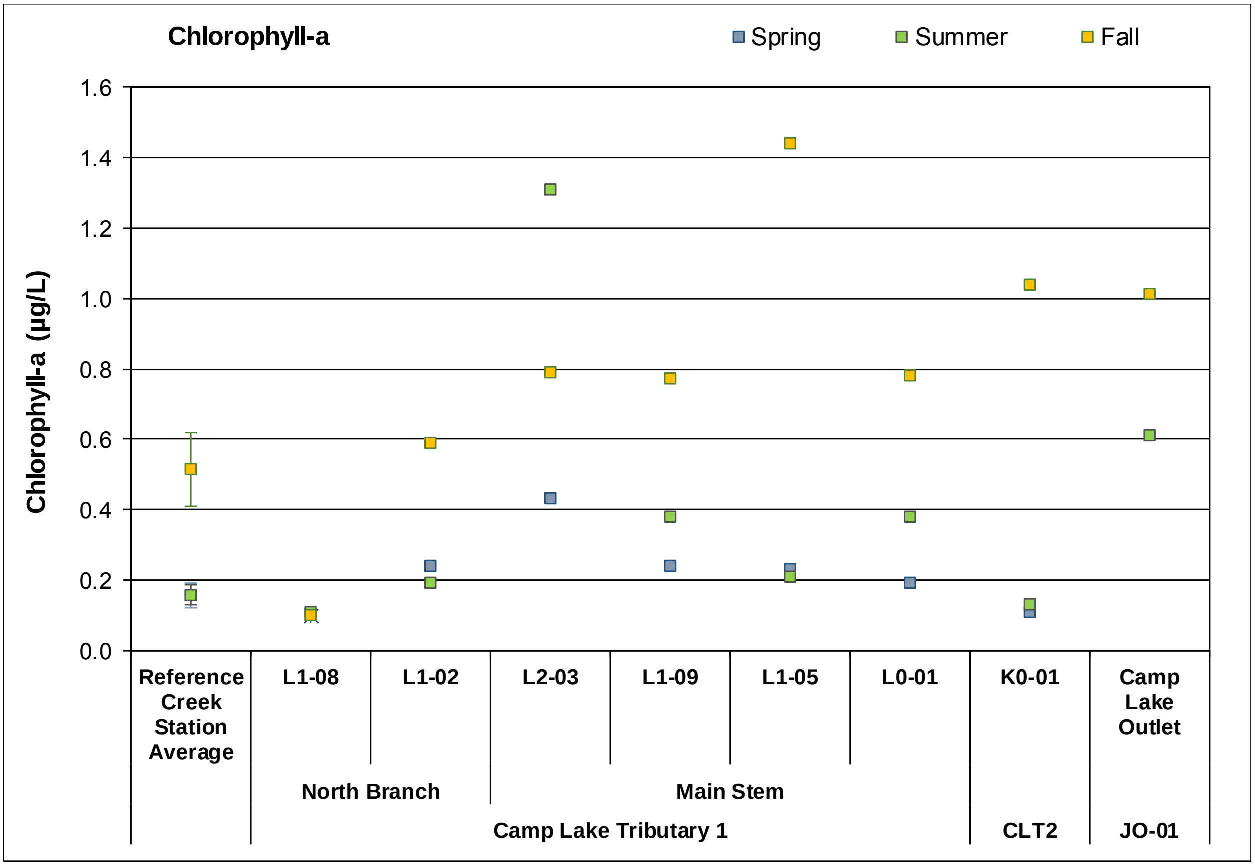


Figure 3.3: Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT1) and Tributary 2 (CLT2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2017

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

'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 stations during all spring, summer, and fall sampling events in 2017 (Appendix Table C.14).

Temporal comparisons of the CLT1 chlorophyll-a data indicated that concentrations at the north branch in fall 2017 were similar to, or lower than, those observed in the fall during the baseline (2005 – 2013) period (Figure 3.4). At the CLT1 main stem, chlorophyll-a concentrations were generally higher in mine operational years from 2015 – 2017 than during the mine baseline period with the exception of at the CLT1 mouth (Station L0-01; Figure 3.4). The spatial and temporal analyses of chlorophyll-a concentrations suggested that mine operation may have contributed to slightly higher phytoplankton productivity at CLT1 main stem stations, but not at the north branch or at the mouth of the main stem. As indicated above, higher phytoplankton productivity within the CLT1 main stem was consistent with the occurrence of higher nutrient concentrations (e.g.,

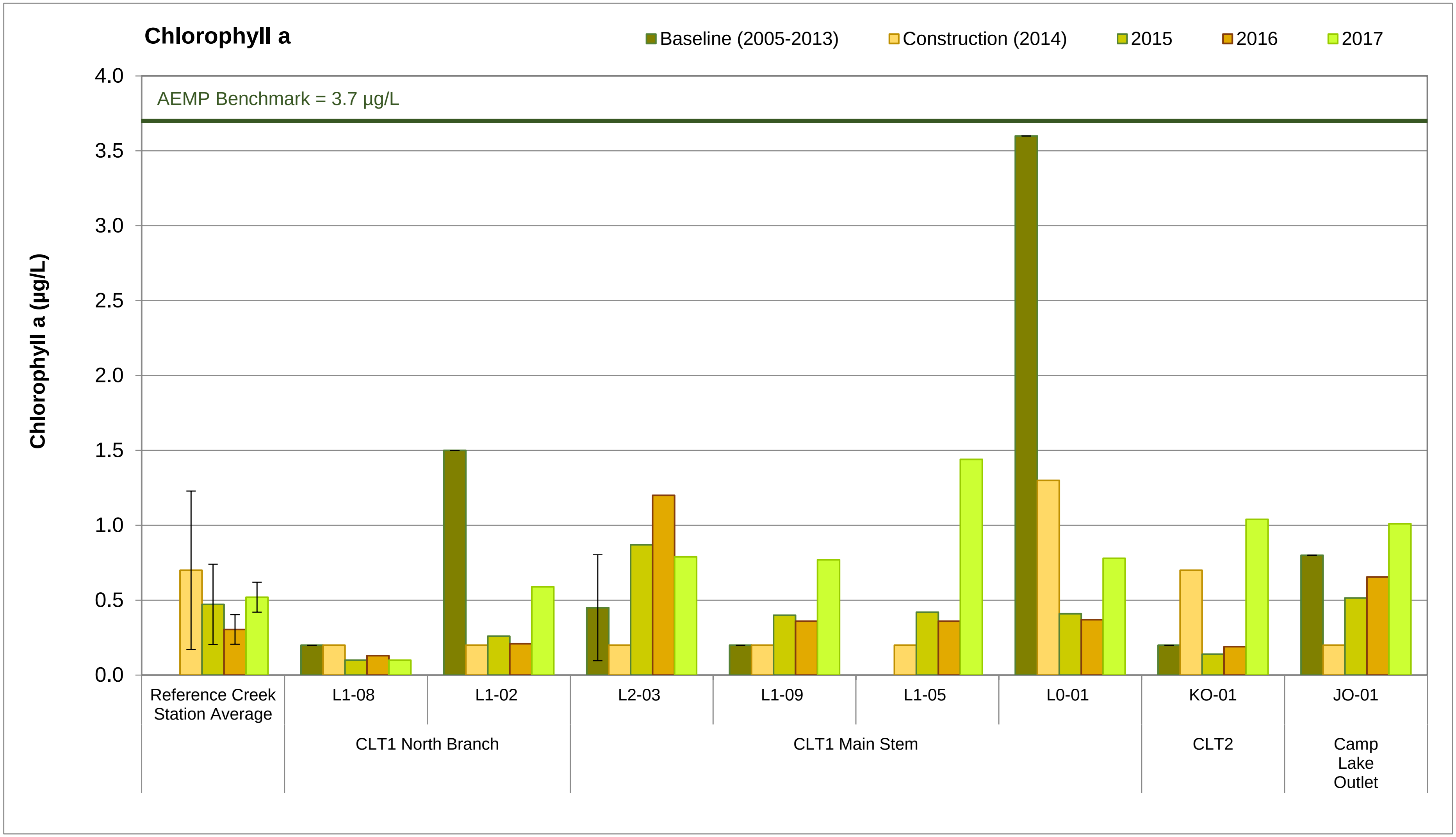


Figure 3.4: Temporal Comparison of Chlorophyll a Concentrations at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015, 2016, and 2017) Periods during Fall

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

nitrate) than at the reference creeks, although (see Section 3.1.1). This suggested that slightly greater phytoplankton productivity at the CLT1 main stem was the result of current mine operations and specifically, the introduction of nutrients to the system as a result of active quarrying at the QMR2 pit. Despite slightly greater phytoplankton productivity at CLT1 over time, the watercourse has remained ‘oligotrophic’ since the commencement of commercial mine operation.

3.1.4 Benthic Invertebrate Community

Upstream North Branch (CLT1 US)

Benthic invertebrate density, richness, and Simpson’s Evenness did not differ significantly between the CLT1 upstream (north branch) and Unnamed Reference Creek study areas (Table 3.3). However, differences in community assemblage were suggested between these study areas based on significant differences in Bray-Curtis Index (Table 3.3). Ecologically significant lower relative abundance of Ephemeroptera (mayflies) and Simuliidae (blackflies), and conversely, higher relative abundance of Chironomidae (non-biting midges) and Tipulidae (crane flies), was indicated 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 3.3). Of these groups, only absolute densities of mayflies and crane flies showed significant, ecologically meaningful, differences between the CLT1 north branch and Unnamed Reference Creek (Appendix Table F.8). Notably, the relative abundance of metal-sensitive chironomids did not differ significantly between the CLT1 north branch and the reference creek, suggesting that the community composition differences between watercourses were unrelated to metal concentrations. Assessment of benthic invertebrate functional feeding groups (FFG) indicated significantly higher relative abundance of shredders at the CLT1 north branch, suggesting the presence of greater amounts of living and/or decomposing large leafy/woody vegetation, compared to Unnamed Reference Creek (Table 3.3). In addition, significantly lower proportions of FFG collector-gatherers and filterers were indicated at the CLT1 north branch compared to the reference creek (Table 3.3). The differences in FFG composition potentially reflected differences in in-stream vegetation types/abundance between watercourses, which included higher bryophyte (moss) and lower periphyton abundance at the CLT1 north branch compared to the reference creek (Appendix Table F.1). Specifically, a greater density of shredders (including *Tipula* crane flies) 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 3.3; Appendix Table F.1). Collectively, the data suggested that differences in benthic invertebrate

Table 3.3: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 1 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2017

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	Magnitude of Difference (SD)	Pairwise Comparison
Density (No. per m^2)	log	NO	0.8399	Reference Creek	1,972	1,861	-	a
				CLT1 Upstream	1,242	143	-0.4	a
				CLT1 Downstream	1,465	735	-0.3	a
Richness (No. of Taxa)	log	YES	0.0224	Reference Creek	21.2	1.9	-	a
				CLT1 Upstream	19.2	2.6	-1.0	a,b
				CLT1 Downstream	16.8	1.9	-2.3	b
Simpson's Evenness	none	YES	0.0038	Reference Creek	0.935	0.017	-	a
				CLT1 Upstream	0.925	0.019	-0.5	a
				CLT1 Downstream	0.874	0.033	-3.6	b
Bray-Curtis Index	rank	YES	< 0.001	Reference Creek	0.305	0.205	-	a
				CLT1 Upstream	0.729	0.027	2.1	b
				CLT1 Downstream	0.777	0.029	2.3	c
Nemata (% of community)	none	NO	0.1301	Reference Creek	8.8	3.6	-	a
				CLT1 Upstream	4.1	2.1	-1.3	a
				CLT1 Downstream	4.6	5.0	-1.2	a
Oligochaeta (% of community)	fourth root	YES	0.0142	Reference Creek	1.0	1.4	-	a
				CLT1 Upstream	1.1	1.3	0.1	a,b
				CLT1 Downstream	5.0	2.7	2.9	b
Hydracarina (% of community)	square root	YES	0.0181	Reference Creek	4.8	2.3	-	a
				CLT1 Upstream	7.6	1.1	1.3	b
				CLT1 Downstream	4.0	1.4	-0.3	a
Ephemeroptera (% of community)	fourth root	YES	< 0.001	Reference Creek	13.8	6.0	-	a
				CLT1 Upstream	1.0	0.6	-2.1	b
				CLT1 Downstream	0.4	0.6	-2.2	c
Chironomidae (% of community)	none	YES	< 0.001	Reference Creek	37.9	11.2	-	a
				CLT1 Upstream	74.0	1.7	3.2	b
				CLT1 Downstream	80.9	4.5	3.9	b
Metal Sensitive Chironomids (% of community)	fourth root	YES	0.0785	Reference Creek	5.7	5.6	-	a,b
				CLT1 Upstream	7.2	5.2	0.3	a
				CLT1 Downstream	1.5	0.7	-0.7	b
Simuliidae (% of community)	fourth root	YES	< 0.001	Reference Creek	28.3	6.6	-	a
				CLT1 Upstream	1.2	1.3	-4.1	b
				CLT1 Downstream	0.3	0.5	-4.2	b
Tipulidae (% of community)	fourth root	YES	< 0.001	Reference Creek	0.7	0.5	-	a
				CLT1 Upstream	8.4	1.5	17.0	b
				CLT1 Downstream	3.9	3.1	7.1	c
Collector-Gatherer FFG (% of community)	none	YES	< 0.001	Reference Creek	58.8	7.2	-	a
				CLT1 Upstream	38.8	7.1	-2.8	b
				CLT1 Downstream	67.2	6.4	1.2	a
Filterer FFG (% of community)	fourth root	YES	< 0.001	Reference Creek	28.7	6.5	-	a
				CLT1 Upstream	1.3	1.5	-4.2	b
				CLT1 Downstream	0.3	0.5	-4.4	b
Shredder FFG (% of community)	none	YES	< 0.001	Reference Creek	6.7	5.5	-	a
				CLT1 Upstream	49.5	6.4	7.8	b
				CLT1 Downstream	27.6	4.9	3.8	c
Clinger HPG (% of community)	none	YES	0.0007	Reference Creek	40.1	7.8	-	a
				CLT1 Upstream	52.5	7.2	1.6	b
				CLT1 Downstream	28.9	5.7	-1.4	c
Sprawler HPG (% of community)	none	YES	0.0029	Reference Creek	49.0	9.2	-	a
				CLT1 Upstream	33.8	7.9	-1.7	b
				CLT1 Downstream	57.6	8.6	0.9	a
Burrower FFG (% of community)	none	NO	0.4433	Reference Creek	10.7	4.5	-	a
				CLT1 Upstream	13.7	2.1	0.7	a
				CLT1 Downstream	13.4	4.7	0.6	a

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

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, richness, Simpson's Evenness, and relative abundance of key dominant groups and FFG did not show any consistent type and/or direction of significant differences in any years of mine operation, including 2017, compared to baseline data collected in both 2007 and 2011 (Figure 3.5; Appendix Table F.9). Notably, higher density, richness (2017 only) and Simpson's Evenness were indicated at the CLT1 north branch in years of mine operation than during years (2007 and 2011) in which baseline data were collected (Figure 3.5; Appendix Table F.9). Therefore, the temporal evaluation indicated no adverse mine-related influences on the benthic invertebrate community of the CLT1 north branch since the commencement of commercial mine operations in 2015.

Downstream Lower Main Stem (CLT1 DS)

The benthic invertebrate community at the lower main stem of Camp Lake Tributary (CLT1 DS), just downstream of the Tote Road, showed significantly lower richness and Simpson's Evenness compared to Unnamed Reference Creek in 2017 (Table 3.3). 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 invertebrate groups and FFG (Table 3.3). Because no significant difference in the relative abundance of metal-sensitive chironomids was indicated between the CLT1 lower main stem and reference area (Table 3.3), the community composition differences between these study areas appeared to be unrelated to differences in metal concentrations. The key differences in benthic invertebrate composition between the CLT1 lower main stem and reference study areas were very similar to those shown between the CLT1 north branch and reference creek, suggesting a similar mechanism for 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). Notably, because substrate with significantly larger diameter and greater embeddedness was sampled at CLT1 compared to Unnamed Reference Creek (Appendix Tables F.3 and F.4), differences in habitat may have also contributed to the indicated differences in benthic invertebrate community compositional features between CLT1 and the reference creek.

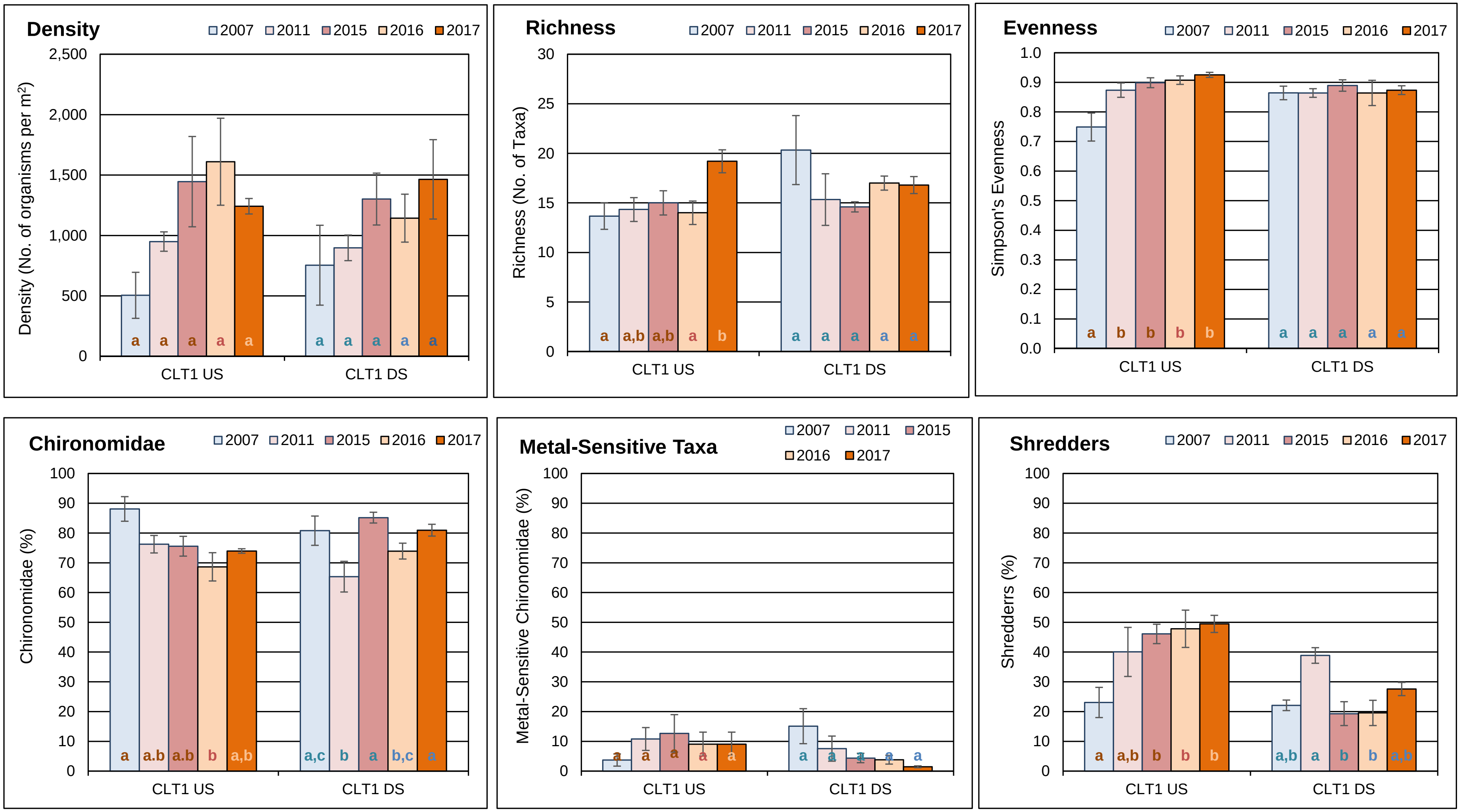


Figure 3.5: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Camp Lake Tributary 1 Study Areas among Mine Baseline (2007, 2011) and Operational (2015, 2016, 2017) Periods

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

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 between individual years of mine operation (2015, 2016, 2017) and the mine baseline (2007, 2011 data) period (Figure 3.5; Appendix Table F.10). In addition, no consistent types and/or direction of differences in the relative abundance of dominant groups or FFG were indicated between 2017 and years in which baseline data were collected at the CLT1 lower main stem (Figure 3.5; Appendix Table 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.

3.1.5 Integrated Effects Evaluation

Upstream North Branch (CLT1 US)

Potential mine-related effects on water quality of the CLT1 north branch in 2017 included slightly elevated molybdenum and potassium concentrations compared to average reference creek data, but only during the spring sampling event. Although CLT1 north branch total copper concentrations were not particularly elevated compared to reference conditions, concentrations at the CLT1 north branch were above WQG and the CLT AEMP benchmark in summer and fall in 2017. In addition, total copper concentrations were consistently elevated compared to the 2005 - 2013 baseline data in each of the three years of commercial mine operation, indicating a mine-related source of copper to the CLT1 north branch. Deposited sediment at the CLT1 north branch contained elevated concentrations of several metals compared to Unnamed Reference Creek, but concentrations of all metals were well below SQG and deposited sediment material composed less than 1% of surficial bed material throughout CLT1. No substantial mine development has occurred in the CLT1 north branch watershed, and therefore hypothesized sources of metals to the watercourse potentially include fugitive dust from the mine and/or natural minerology of the bedrock/overburden in the region of the mine.

Despite copper concentrations above WQG, chlorophyll-a concentrations (a surrogate for phytoplankton abundance) at the CLT1 north branch were comparable to those of the reference creek stations in 2017, and to those during the baseline period, all of which were well below the AEMP benchmark and suggested oligotrophic conditions typical of Arctic watercourses. In addition, no ecologically significant differences in primary benthic invertebrate community endpoints (i.e., density, richness, and Simpson' Evenness) or in the relative abundance of metal-sensitive chironomids between the CLT1 north branch and reference creek in 2017, nor any consistent significant differences in benthic metric type or direction between mine operational (2015 – 2017) and baseline periods. In turn, this suggested that despite aqueous total copper concentrations above the applicable AEMP benchmark, concentrations did not adversely affect phytoplankton and benthic invertebrates of the CLT1 north branch. Overall, similar to the findings of the two previous CREMP studies, no adverse mine-related effects to biota of the CLT1 north branch were indicated in 2017.

Downstream Main Stem (CLT1 DS)

At the CLT1 main stem, mine-related influences on water quality were evident as elevated conductivity and concentrations of chloride, nitrate, sulphate, TDS, TKN, and several metals including iron, manganese, molybdenum, potassium, sodium, strontium, and uranium at the upstream-most station (Station L2-03) compared to reference creek station data in 2017. However, downstream of the confluence with the north branch, only chloride and total manganese concentrations were elevated at the CLT1 main stem stations (i.e., stations L1-01, L1-05, and L1-09) compared to average reference creek conditions in 2017. Nevertheless, concentrations of iron, manganese, molybdenum, nitrate, sodium, sulphate, TKN, and uranium were consistently elevated in 2015, 2016, and 2017 at the CLT1 main stem stations compared to the baseline period. As hypothesized in previous CREMP studies, quarrying activity at the QMR2 pit was likely a key source of the parameters shown to be elevated at CLT1 main stem stations.

Despite evidence of continued mine-related influence on water quality of the CLT1 upper main stem in 2017, parameter concentrations were below applicable WQG and site-specific AEMP benchmarks with the exception of aluminum and iron, which were above their respective benchmarks during the fall sampling event in 2017. Notably, total aluminum and iron concentrations were also elevated above AEMP benchmarks at one of the four lotic reference creek stations in fall 2017. High turbidity at the CLT1 main stem and reference creek in fall 2017 indicated a potential causal link to the high total concentrations of aluminum and iron, and evaluation of dissolved concentrations of these metals indicated that the source of iron was likely related to mine operations. Similar to the north branch, deposited sediment at the CLT1 main stem contained elevated concentrations of several metals compared to Unnamed Reference Creek, but concentrations of all metals were well below SQG and deposited sediment material composed a very small proportion of surficial bed material within the CLT1 main stem, limiting the exposure of sediment metal concentrations to in-stream biota.

Chlorophyll-a concentrations at the CLT1 main stem were generally highest at the upstream-most Station L2-03, were significantly higher than the reference creek average during all three seasonal sampling events in 2017, and were higher in 2015, 2016, and 2017 than during the mine baseline period. The occurrence of relatively high chlorophyll-a concentrations at the CLT1 main stem not only suggested that concentrations of aluminum, iron, uranium, and other metals were not highly bioavailable at the CLT1 upper main stem, but that elevated nitrate concentrations may have contributed to 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 2017, the weight-of-evidence indicated that natural differences in in-stream bryophyte (moss) growth between watercourses 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 2017, and no consistent significant differences in benthic metric type or direction between the mine operational (2015 – 2017) and baseline (2007, 2011) studies. Thus, no adverse mine-related effects to phytoplankton and benthic invertebrate biota of the CLT1 lower main stem were indicated in 2017 based on comparison to reference creek conditions and to baseline data.

3.2 Camp Lake Tributary 2 (CLT2)

3.2.1 Water Quality

Camp Lake Tributary 2 (CLT2) dissolved oxygen saturation levels were consistently high at Station KO-01 in 2017, and were similar to mean DO saturation observed among the reference creek stations during all seasonal sampling events (Appendix Tables C.1 – C.3). *In situ* DO concentrations at the CLT2 upstream and downstream study areas did not differ significantly from those at Unnamed Reference Creek, nor from each other, at the time of biological sampling in August 2017 (Appendix Table C.19), and were all above the WQG minimum limit for protection of sensitive stages of cold-water biota (Figure 3.6). Aqueous pH at both CLT2 study areas was slightly higher (i.e., more alkaline) than the average among lotic reference stations, but was consistently well within WQG limits (Appendix Tables C.1 – C.3). No significant difference in pH was indicated between CLT2 study areas located upstream and downstream of the Tote Road (Figure 3.6). *In situ* specific conductance was significantly higher at CLT2 compared to Unnamed Reference Creek, but did not differ significantly upstream and downstream of the Tote Road during the August 2017 at the time of biological sampling (Figure 3.6).

Water chemistry at CLT2 (Station KO-01) was similar to the reference creek stations during spring, summer, and fall sampling events in 2017 with the exception of slightly to moderately higher (i.e., 5- to 10-fold) sulphate concentrations (Table 3.1; Appendix Table C.15). In addition, aqueous concentrations of all parameters, including sulphate, were consistently well below established WQG and AEMP benchmarks at the CLT2 monitoring station in 2017 (Table 3.1; Appendix Table C.14). Temporal comparisons of CLT2 water chemistry data indicated that parameter concentrations in fall 2017 were generally within the range of those measured during the mine baseline period (2005 – 2013; Appendix Tables C.15 and C.18) and were not unlike

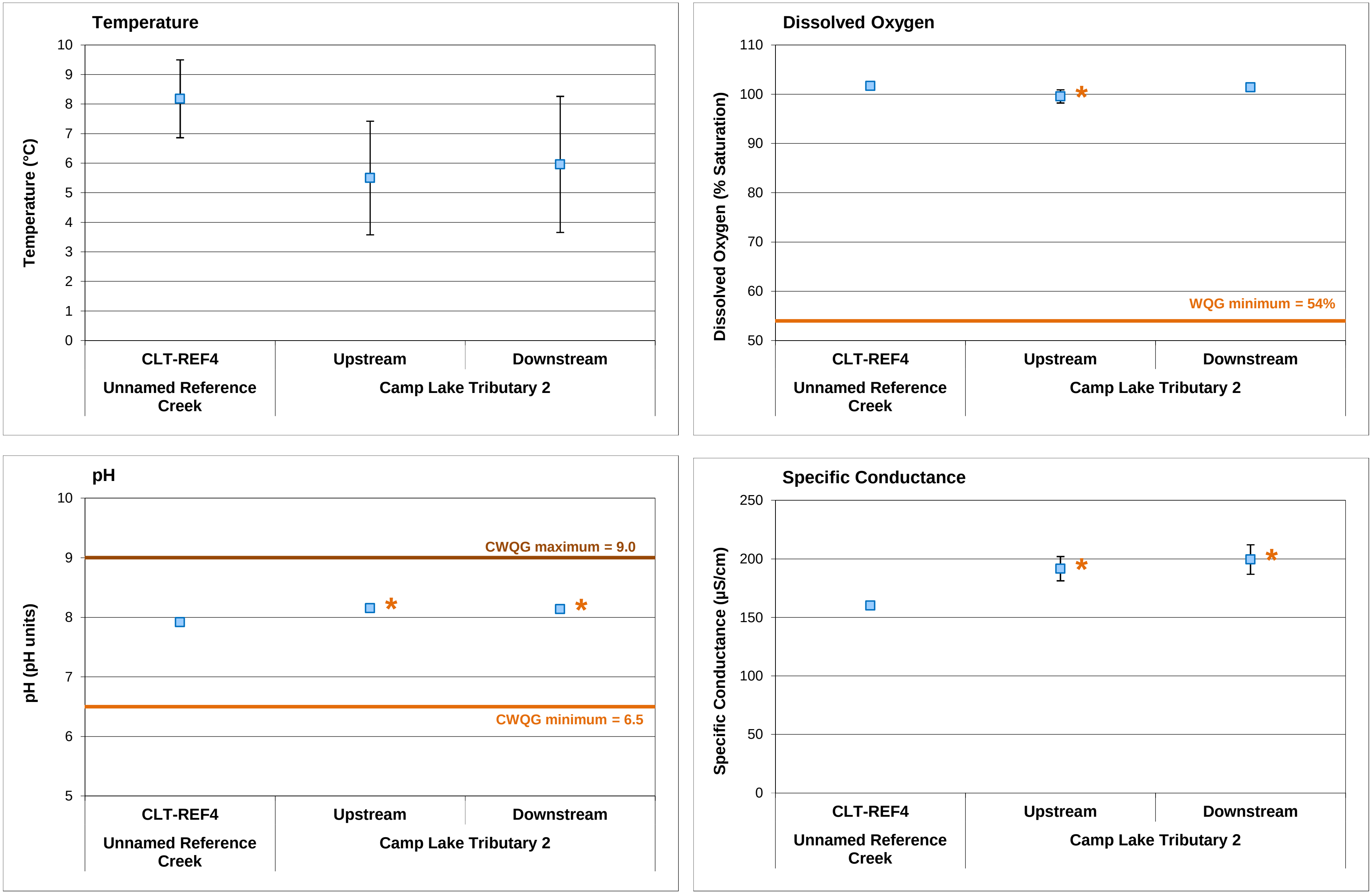


Figure 3.6: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 2 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

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

those observed during the 2014 mine construction and 2015 – 2016 mine operation period (Figure 3.2; Appendix Figure C.2). Collectively, the 2017 water chemistry data suggested only minor mine-related influence on aqueous conductivity and sulphate concentrations within the CLT2 system compared to applicable reference and mine baseline conditions.

3.2.2 Sediment Quality

Deposited sediment at CLT2 upstream (CLT2-US) and downstream (CLT2-DS) study areas was visually characterized as coarse to very coarse sand (Appendix Table D.7). Similar to CLT1, the in-stream substrate at both CLT2 study areas was composed mainly of cobble material (i.e., substrate diameter 6 – 25 cm), with sand constituting only a trace amount (i.e., <1%) of the material observed at the sediment surface of the upstream area, and approximately 5% of material at the downstream area (Appendix Table F.1). Accordingly, deposited sediment was collected mainly from shoreline/streambank areas at the CLT2 study areas (Appendix Table D.7). Sediment TOC content was low (i.e., ~0.1%) at both CLT2 study areas, and comparable to average lotic reference area TOC content suggesting similar depositional characteristics among the CLT2 and lotic reference study areas (Table 3.4; Appendix Table D.10).

Deposited sediment at CLT2 showed slightly (i.e., 3- to 5-fold higher) to moderately (i.e., 5-fold to 10-fold higher) concentrations of aluminum, calcium, chromium, cobalt, copper, iron, magnesium, manganese, nickel, potassium, and vanadium compared to respective average concentrations at the reference creek (Table 3.4; Appendix Table D.10). Of these metals, only copper, iron, and vanadium, as well as zinc, occurred at concentrations ≥1.5 times higher at the downstream area at than at the upstream area of CLT2 (Table 3.4), potentially reflecting greater influence of the Tote Road on metal concentrations of deposited sediment within the lower CLT2 watercourse. However, concentrations of all metals were well below applicable SQG at all upstream and downstream stations at CLT2 (Table 3.4; Appendix Tables D.11 and D.12). Notably, metal concentrations in deposited sediment at the CLT2 upstream and downstream study areas were consistently lower than those at the CLT1 study areas, potentially indicating reduced influences with greater distance from the mine. No baseline sediment metal concentration data were collected at the CLT2 study areas, and thus no evaluation of potential mine-related influences on sediment quality following commencement of commercial mine operations could be conducted.

3.2.3 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 2017 (Figure 3.3). Nutrient concentrations, including ammonia, nitrate, and total phosphorus, showed

Table 3.4: Sediment Total Organic Carbon and Metal Concentrations at Camp Lake Tributary 2 (CLT2) and Lotic Reference Area Sediment Monitoring Stations, Mary River Project CREMP, August 2017

Parameter	Units	Sediment Quality Guideline (SQG)[a]	Lotic Reference Stations		Camp Lake Tributary 2	
			Unnamed Reference Creek (REFCRK; n = 5)	Mary River Reference (GO-09; n = 5)	Upstream CLT2-US (n = 5)	Downstream CLT2-DS (n = 5)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD
Total Organic Carbon	%	10[α]	<0.10 ± 0.00	<0.10 ± 0.00	0.11 ± 0.01	0.12 ± 0.03
Aluminum (Al)	mg/kg	-	418 ± 126	763 ± 271	1,664 ± 314	1,650 ± 509
Antimony (Sb)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Arsenic (As)	mg/kg	17	0.12 ± 0.02	0.16 ± 0.02	0.35 ± 0.07	0.36 ± 0.12
Barium (Ba)	mg/kg	-	1.8 ± 0.3	3.7 ± 1.2	5.7 ± 1.1	5.5 ± 1
Beryllium (Be)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0.00
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0.00
Boron (B)	mg/kg	-	<5.0 ± 0	<5.0 ± 0	<5.0 ± 0	<5.0 ± 0.0
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	<0.020 ± 0	<0.020 ± 0	0.021 ± 0.003
Calcium (Ca)	mg/kg	-	214 ± 44	842 ± 508	1,534 ± 511	1,536 ± 747
Chromium (Cr)	mg/kg	90	1.4 ± 0.4	3.4 ± 1.0	9.5 ± 2.1	12.3 ± 9.0
Cobalt (Co)	mg/kg	-	0.32 ± 0.09	0.67 ± 0.19	1.83 ± 0.37	1.96 ± 0.93
Copper (Cu)	mg/kg	110[α]	0.8 ± 0.5	1.3 ± 0.5	4.5 ± 0.7	6.5 ± 6.1
Iron (Fe)	mg/kg	40,000[α]	1,240 ± 475	2,826 ± 1,271	5,088 ± 1,784	8,254 ± 8,263
Lead (Pb)	mg/kg	91	0.7 ± 0.2	1.1 ± 0.1	1.5 ± 0.4	1.6 ± 0.5
Lithium (Li)	mg/kg	-	<2.0 ± 0.0	2.2 ± 0.5	3.1 ± 0.4	3.3 ± 0.9
Magnesium (Mg)	mg/kg	-	333 ± 116	826 ± 521	2,402 ± 552	2,408 ± 917
Manganese (Mn)	mg/kg	1,100[α,β]	10 ± 2.4	22 ± 7.1	58 ± 12	59 ± 21
Mercury (Hg)	mg/kg	0.486	<0.0050 ± 0	<0.0050 ± 0	<0.0050 ± 0	<0.0050 ± 0.0000
Molybdenum (Mo)	mg/kg	-	<0.10 ± 0.00	0.11 ± 0.01	<0.10 ± 0.00	<0.10 ± 0.00
Nickel (Ni)	mg/kg	75[α,β]	0.8 ± 0.2	1.9 ± 0.7	5.9 ± 1.1	6.4 ± 3.4
Phosphorus (P)	mg/kg	2,000[α]	61 ± 16	113 ± 49	117 ± 18	130 ± 58
Potassium (K)	mg/kg	-	106 ± 13	168 ± 77	392 ± 66	386 ± 110
Selenium (Se)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0
Silver (Ag)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Sodium (Na)	mg/kg	-	<50 ± 0	<50 ± 0	<50 ± 0	<50 ± 0
Strontium (Sr)	mg/kg	-	1.2 ± 0.2	1.9 ± 0.3	1.8 ± 0.3	1.8 ± 0.4
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	<0.050 ± 0	<0.050 ± 0	<0.050 ± 0.000	<0.050 ± 0.000
Tin (Sn)	mg/kg	-	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0
Titanium (Ti)	mg/kg	-	36 ± 16	109 ± 26	137 ± 36	146 ± 52
Uranium (U)	mg/kg	-	0.20 ± 0.08	0.30 ± 0.03	0.23 ± 0.05	0.27 ± 0.13
Vanadium (V)	mg/kg	-	1.9 ± 0.7	5.0 ± 2.2	8.4 ± 2.0	13.4 ± 11.4
Zinc (Zn)	mg/kg	315	2.0 ± 0.0	3.7 ± 1.3	4.2 ± 0.9	8.1 ± 4.1
Zirconium (Zr)	mg/kg	-	1.1 ± 0.1	1.7 ± 0.3	1.4 ± 0.2	1.5 ± 0.4

[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)).

Indicates parameter concentration above Sediment Quality Guideline (SQG).

no marked differences between CLT2 and the reference creek stations during the fall sampling event in 2017 (Appendix Tables C.14 and C.15), and therefore the occurrence of higher chlorophyll-a concentrations within lower CLT2 in fall 2017 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 2017 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 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 2017 compared to the mine baseline period and to the two previous years of mine operation at lower CLT2 during fall sampling (Figure 3.4). For the reasons indicated above, higher chlorophyll-a concentrations at CLT2 in fall 2017 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.

3.2.4 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 each significantly lower at both CLT2 study areas compared to Unnamed Reference Creek, although only the difference in richness was ecologically meaningful (Table 3.5). Differences in community composition were also indicated by a significantly higher Bray-Curtis Index at both CLT2 study areas compared to the Unnamed Reference Creek. Ecologically significant differences in the relative abundance of various dominant benthic invertebrate groups, including lower proportion of mayflies and blackflies, and higher proportion of chironomids (including metal-sensitive taxa), were indicated between CLT2 and the reference creek (Table 3.5). However, in terms of absolute densities, the only ecologically significant difference was a lower mayfly density at the CLT2 study areas compared to the Unnamed Reference Creek (Appendix Table F.14). Similarly, although the relative abundance of collector-gatherer and filterer FFG and clinger and sprawler habitat preference groups (HPG) differed significantly between CLT2 and reference creek study areas, absolute densities of all FFG and HPG did not differ significantly at magnitudes outside of CES_{BIC} (Table 3.5; Appendix Table F.14). The reason(s) for the variable differences in benthic invertebrate community relative abundances versus absolute densities between the CLT2 and reference creek study areas were unclear based on the available data. However, because metal-sensitive chironomids were present at CLT2 in significantly higher relative abundance and

Table 3.5: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 2 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2017

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	Magnitude of Difference (SD)	Pairwise Comparison
Density (No. per m^2)	log	YES	0.0002	Reference Creek	1,972	1,861	-	a
				CLT2 Upstream	216	30	-0.9	b
				CLT2 Downstream	222	144	-0.9	b
Richness (No. of Taxa)	none	YES	0.0050	Reference Creek	21.2	1.9	-	a
				CLT2 Upstream	14.6	2.5	-3.4	b
				CLT2 Downstream	13.2	4.7	-4.2	b
Simpson's Evenness	none	NO	0.1803	Reference Creek	0.935	0.017	-	a
				CLT2 Upstream	0.955	0.013	1.2	a
				CLT2 Downstream	0.913	0.052	-1.2	a
Bray-Curtis Index	log	YES	< 0.001	Reference Creek	0.305	0.205	-	a
				CLT2 Upstream	0.782	0.019	2.3	b
				CLT2 Downstream	0.794	0.102	2.4	b
Nemata (% of community)	modified probit	YES	0.0142	Reference Creek	8.8	3.6	-	a
				CLT2 Upstream	1.0	1.4	-2.2	b
				CLT2 Downstream	3.2	4.4	-1.6	a,b
Oligochaeta (% of community)	modified probit	NO	0.2304	Reference Creek	1.0	1.4	-	a
				CLT2 Upstream	3.4	6.4	1.7	a
				CLT2 Downstream	4.8	6.9	2.8	a
Hydracarina (% of community)	none	NO	0.1479	Reference Creek	4.8	2.3	-	a
				CLT2 Upstream	8.0	4.2	1.4	a
				CLT2 Downstream	3.3	4.0	-0.6	a
Ephemeroptera (% of community)	none	YES	< 0.001	Reference Creek	13.8	6.0	-	a
				CLT2 Upstream	1.1	1.7	-2.1	b
				CLT2 Downstream	0.9	1.0	-2.2	b
Chironomidae (% of community)	none	YES	< 0.001	Reference Creek	37.9	11.2	-	a
				CLT2 Upstream	75.9	7.5	3.4	b
				CLT2 Downstream	81.8	8.4	3.9	b
Metal Sensitive Chironomids (% of community)	modified probit	YES	0.0031	Reference Creek	5.7	5.6	-	a
				CLT2 Upstream	22.0	3.1	2.9	b
				CLT2 Downstream	20.2	12.6	2.6	b
Simuliidae (% of community)	modified probit	YES	< 0.001	Reference Creek	28.3	6.6	-	a
				CLT2 Upstream	6.5	3.1	-3.3	b
				CLT2 Downstream	3.5	3.1	-3.8	b
Collector-Gatherer FFG (% of community)	none	YES	0.0045	Reference Creek	58.8	7.2	-	a
				CLT2 Upstream	75.6	3.9	2.3	b
				CLT2 Downstream	77.1	10.5	2.6	b
Filterer FFG (% of community)	modified probit	YES	< 0.001	Reference Creek	28.7	6.5	-	a
				CLT2 Upstream	6.5	3.1	-3.4	b
				CLT2 Downstream	3.5	3.1	-3.9	b
Shredder FFG (% of community)	none	NO	0.1833	Reference Creek	6.7	5.5	-	a
				CLT2 Upstream	7.4	5.9	0.1	a
				CLT2 Downstream	14.6	8.9	1.4	a
Clinger HPG (% of community)	modified probit	YES	0.0112	Reference Creek	40.1	7.8	-	a
				CLT2 Upstream	22.4	2.7	-2.3	b
				CLT2 Downstream	22.1	10.6	-2.3	b
Sprawler HPG (% of community)	none	YES	0.0008	Reference Creek	49.0	9.2	-	a
				CLT2 Upstream	70.9	5.3	2.4	b
				CLT2 Downstream	69.0	7.0	2.2	b
Burrower FFG (% of community)	none	NO	0.5877	Reference Creek	10.7	4.5	-	a
				CLT2 Upstream	6.7	5.0	-0.9	a
				CLT2 Downstream	8.9	8.2	-0.4	a

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

(blue) 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

similar density than at the Unnamed Reference Creek, no adverse metal-related influences to the benthic invertebrate community were indicated at the CLT2 study areas (Table 3.5). In turn, this suggested that the other differences in benthic invertebrate community structure between CLT2 and the reference creek likely reflected natural variability. Notably, no significant differences in density, richness, Simpson's Evenness, or the relative abundance of dominant invertebrate groups, FFG, or HPG were indicated between the CLT2 upstream and downstream study areas, indicating no adverse influences to the benthic invertebrate community of CLT2 associated with the Tote Road crossing (Table 3.5).

Temporal comparisons indicated no consistent ecologically significant differences in any benthic invertebrate community endpoints at the CLT2 upstream and downstream study areas over the three years of mine operation (2015, 2016, 2017) compared to 2007 baseline data with the exception of Simpson's Evenness (Figure 3.7; Appendix Tables F.15 and F.16). 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 in 2015, 2016, and 2017 compared to 2007 was not consistent with an adverse influence related to recent mine operations. This suggested that differences in benthic invertebrate community endpoints between CLT2 and Unnamed Reference Creek in 2017 were most likely related to natural differences in habitat between watercourses, and that no appreciable changes to the benthic invertebrate community of CLT2 have occurred since commercial mine operations commenced in 2015.

3.2.5 Integrated Effects Evaluation

Potential mine-related effects on water quality of CLT2 in 2017 included slightly elevated conductivity and concentrations of sulphate compared to reference creek averages. However, because CLT2 water chemistry in 2017 was comparable to the 2005 - 2013 baseline data, natural regional variability in water chemistry among lotic environments likely accounted for seemingly elevated conductivity and sulphate at CLT2 in 2017 compared to the reference creek stations. Aqueous concentrations of all parameters were consistently well below applicable WQG and site-specific AEMP benchmarks at CLT2 during the 2015, 2016, and 2017 years of mine operation. Deposited sediment at CLT2 contained elevated concentrations of aluminum, calcium, chromium, cobalt, copper, iron, magnesium, manganese, nickel, potassium, and vanadium compared to the reference creek, but concentrations of all metals were well below SQG and deposited sediment material composed less than 5% of surficial bed material in CLT2.

Chlorophyll-a concentrations at CLT2 varied seasonally from those shown at reference creek stations in 2017, but were consistently well below the AEMP benchmark and reflective of oligotrophic conditions characteristic of Arctic watercourses. Although CLT2 chlorophyll-a

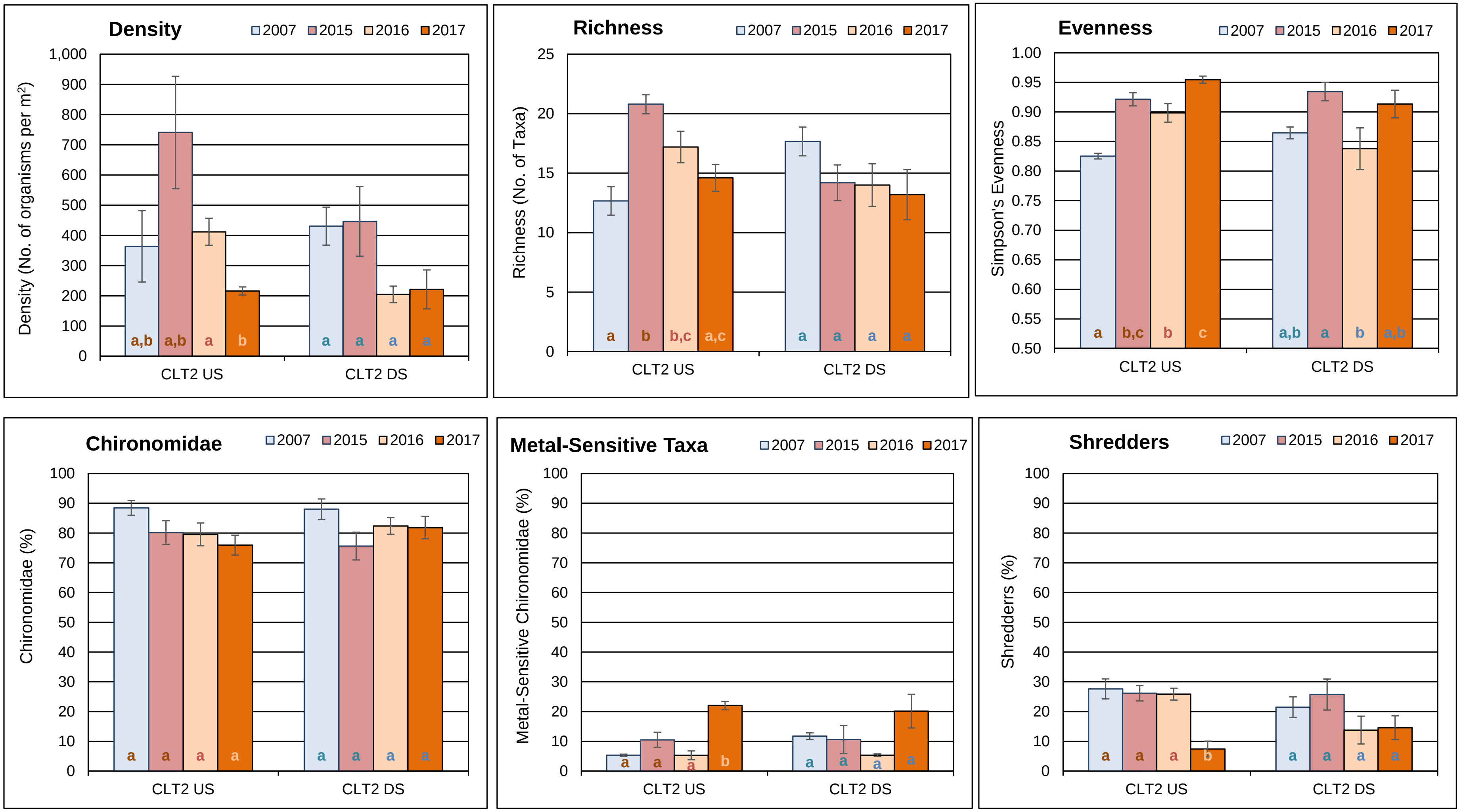


Figure 3.7: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Camp Lake Tributary 2 Study Areas among Mine Baseline (2007) and Operational (2015, 2016, 2017) Periods

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

concentrations were higher in 2017 than during the mine baseline period and the two previous CREMP studies, no change in nutrient concentrations was indicated at CLT2 in 2017 and therefore natural seasonal/temporal variation in chlorophyll-a concentrations potentially accounted for higher concentrations in 2017. The benthic invertebrate community of CLT2 exhibited significantly lower density and richness, and significantly different composition, than Unnamed Reference Creek in 2017, but these differences appeared to be related to natural habitat differences between watercourses. This was supported by the occurrence of significantly higher relative abundance of metal-sensitive chironomids at CLT2 than at the reference creek in 2017. In addition, no ecologically significant differences in benthic invertebrate community endpoints were consistently indicated at CLT2 between years of mine operation and the 2007 baseline study 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 upstream study area in years of mine operation compared to 2007 was not consistent with an adverse influence related to recent mine operations. Collectively, similar to the findings of the two previous CREMP studies, the chlorophyll-a and benthic invertebrate community data indicated no adverse mine-related effects to biota of CLT2 in 2017.

3.3 Camp Lake (JLO)

3.3.1 Hydraulic Retention Time

A hydraulic retention time of 416 ± 184 days was estimated for Camp Lake using mean annual watershed runoff (2007 – 2016 data) extrapolated from CLT1 and CLT2 flow monitoring stations (Stations H05 and H04, respectively) and a lake volume of 27.5 million cubic metres (from NSC 2015b).

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 2017 (Appendix Figures C.3 - C.6). The 2017 Camp Lake water temperature profiles showed a slight increase in temperature with depth (i.e., <2˚C) during the winter sampling event, and a weakly stratified condition marked by a thermocline at the 20 – 23 m depth during fall sampling that mirrored the fall temperature profile pattern at Reference Lake 3 (Figure 3.8). On average, water temperature near the bottom of the water column at littoral and profundal stations of Camp Lake was cooler than at Reference Lake 3 (Figure 3.9; Appendix Tables C.24 – C.25). Although bottom water temperatures at Camp Lake littoral stations differed significantly from respective

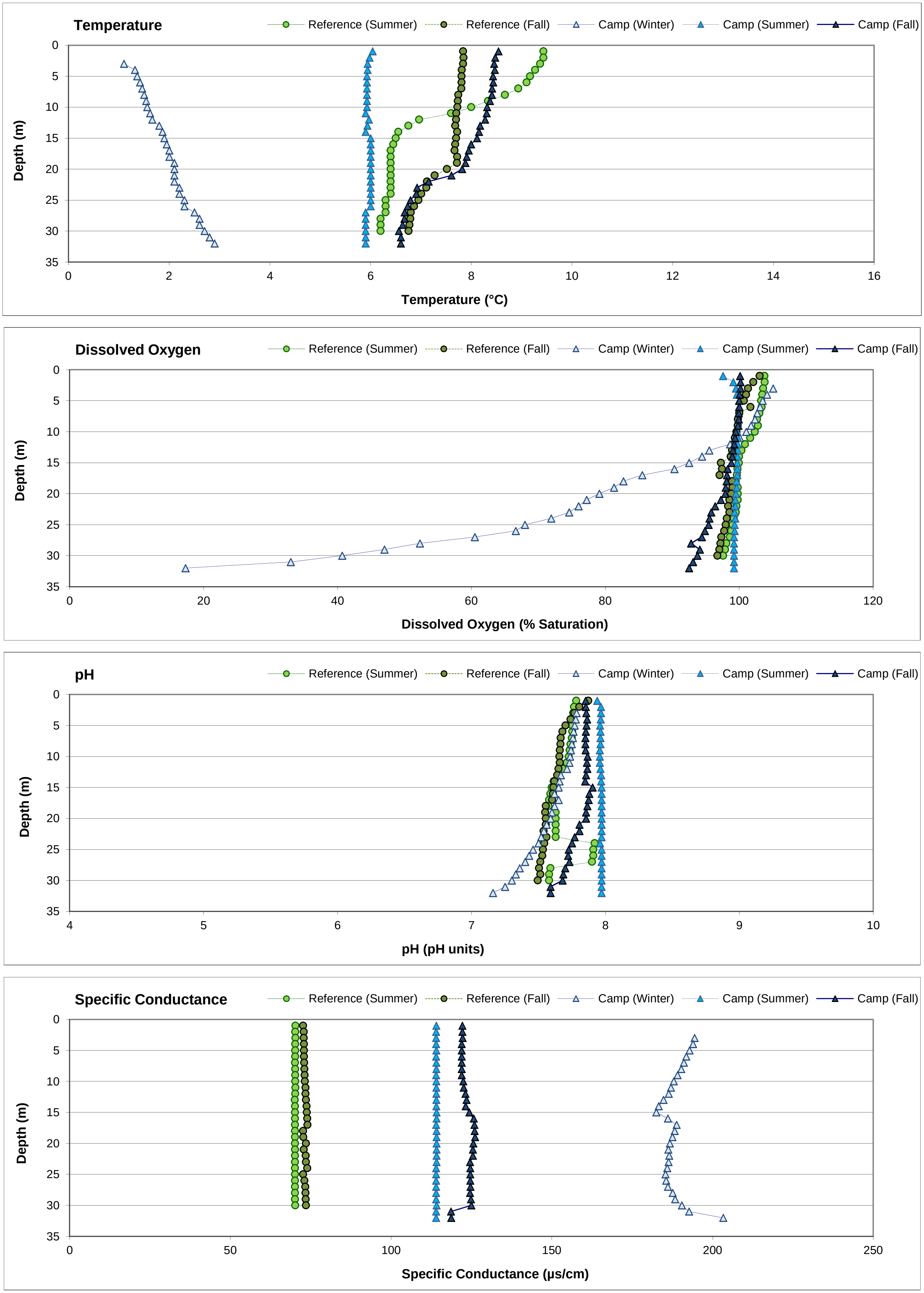


Figure 3.8: Average *In Situ* Water Quality with Depth from Surface at Camp Lake (JLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2017

stations at Reference Lake 3, the small incremental difference in water temperature (i.e., 0.4˚C) was unlikely to result in meaningful ecological differences between lakes.

Dissolved oxygen profiles conducted at Camp Lake in 2017 showed declining saturation levels with increased depth beginning at approximately 11 m below surface in the winter, but showed no appreciable changes from surface to bottom during summer or fall 2017, reflecting the dissolved oxygen profiles at Reference Lake 3 (Figure 3.8) and observations from Camp Lake in 2015 and 2016. Dissolved oxygen conditions 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 fall sampling in 2017 (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 2017 except at water depths greater than approximately 30 m in winter (Figures 3.8 and 3.9). This suggested that dissolved oxygen concentrations were not likely to be limiting to biota at Camp Lake for the entire lake volume for the majority of the year.

In situ profiles of pH and specific conductance showed no substantial change from the surface to bottom of the Camp Lake water column, indicating the absence of chemical stratification (Figure 3.8). Although the bottom pH at littoral and profundal stations of Camp Lake was significantly higher than at the reference lake during the fall sampling event (Appendix Table C.25), the mean incremental difference between lakes was very small (i.e., 0.5 pH units) and all pH values were consistently within WQG limits (Figure 3.9). Specific conductance was significantly higher at Camp Lake compared to the reference lake during fall sampling in 2017 (Figure 3.9). However, because mean specific conductance at Camp Lake was intermediate to that of the reference creek and river stations (i.e., range from 55 – 168 µS/cm), the occurrence of higher specific conductance at Camp Lake compared to the reference lake likely reflected natural phenomena. 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 2017 fall sampling event (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 2017 at Camp Lake (Appendix Table C.23).

Water chemistry data collected at Camp Lake in 2017 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 3.6; Appendix Table C.26), suggesting that the lake waters were well mixed laterally. Slight elevation (i.e., 3- to 5-fold higher) in chloride and total manganese concentrations was evident at Camp Lake compared to the reference lake during the summer 2017 sampling

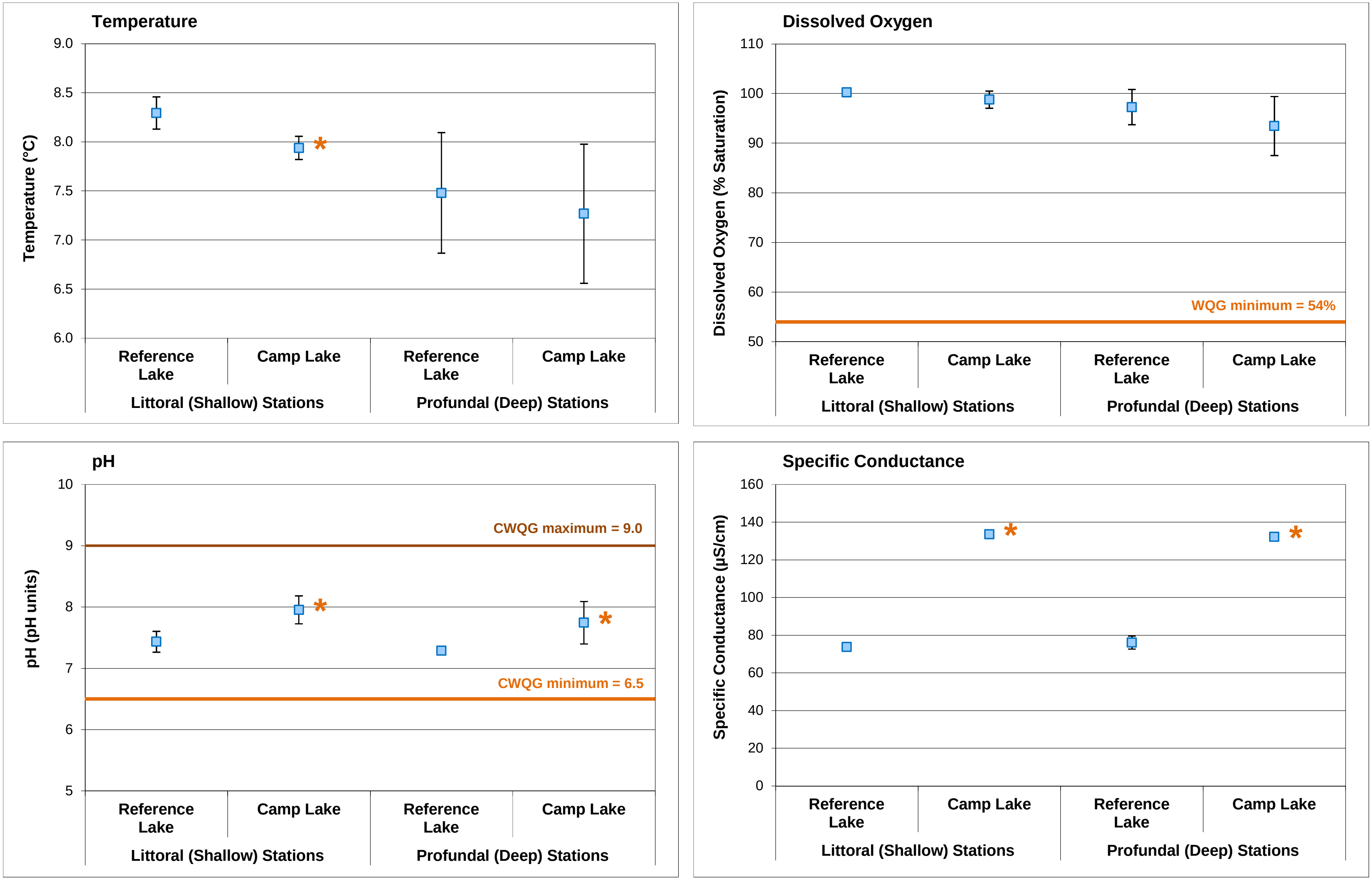


Figure 3.9: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake (JLO) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

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.

Table 3.6: Water Chemistry at Camp Lake (JLO) and Reference Lake 3 (REF3) Monitoring Stations[a], Mary River Project CREMP, August 2017

	Parameters	Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3) Fall 2017	Camp Lake Stations JL0-02 26-Aug-17	JL0-10 25-Aug-17	JL0-01 26-Aug-17	JL0-07 26-Aug-17	JL0-09 25-Aug-17	J0-01 Camp Lake Outlet 1-Sep-17
Conventionals	Conductivity (lab)	umho/cm	-	-	76	140	140	140	140	140	139
	pH (lab)	pH	6.5 - 9.0	-	7.76	8.05	8.06	8.06	7.97	8.02	8.02
	Hardness (as $CaCO_3$)	mg/L	-	-	35	68	66	67	66	66	66
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	33	61	59	64	62	59	69
	Turbidity	NTU	-	-	0.69	0.51	0.76	0.47	0.42	0.67	0.67
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31	66	63	64	66	64	66
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.020	<0.020	<0.020	<0.020	0.026	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	0.16	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.7	1.8	1.8	1.8	1.7	1.7	1.7
	Total Organic Carbon	mg/L	-	-	2.8	1.8	1.8	1.7	1.7	1.8	1.7
	Total Phosphorus	mg/L	0.020[α]	-	0.0040	0.0052	<0.0030	0.0053	0.0058	<0.0030	0.0051
	Phenols	mg/L	0.004[α]	-	0.0020	0.0019	<0.0010	<0.0010	0.0013	0.0010	0.0019
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.3	3.6	3.6	3.2	3.7	3.5	3.7
	Sulphate (SO_4)	mg/L	218[β]	218	4.0	2.8	2.8	2.5	2.9	2.8	3.0
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0045	0.0055	0.0045	0.0053	0.0053	0.0050	0.0047
	Antimony (Sb)	mg/L	0.020[α]	-	<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	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0064	0.0066	0.0063	0.0065	0.0064	0.0065	0.0065
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	6.9	13.6	13.4	13.3	13.3	13.2	13.2
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.00079	0.00082	0.00082	0.00097	0.00081	0.00084	0.00083
	Iron (Fe)	mg/L	0.30	0.326	<0.030	<0.030	<0.030	<0.030	<0.030	<0.030	0.057
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	0.0015	0.0013	0.0014	0.0013	0.0013	0.0012
	Magnesium (Mg)	mg/L	-	-	4.4	8.5	8.4	8.4	8.3	8.2	8.2
	Manganese (Mn)	mg/L	0.935[β]	-	0.00063	0.00171	0.00164	0.00166	0.00174	0.00172	0.00277
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00012	0.00029	0.00029	0.00029	0.00028	0.00029	0.00030
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	0.00059	0.00057	0.00059	0.00059	0.00057	0.00075
	Potassium (K)	mg/L	-	-	0.9	1.1	1.1	1.1	1.1	1.1	1.1
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.41	0.32	0.32	0.33	0.36	0.32	0.34
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.8	1.4	1.4	1.4	1.4	1.4	1.4
	Strontium (Sr)	mg/L	-	-	0.0078	0.0102	0.0101	0.0100	0.0100	0.0100	0.0099
	Thallium (Tl)	mg/L	0.0008	0.0008	<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
	Titanium (Ti)	mg/L	-	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00025	0.00072	0.00070	0.00070	0.00069	0.00070	0.00077
	Vanadium (V)	mg/L	0.006[α]	0.006	<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

[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.

event, but no parameters were elevated at Camp Lake compared to the reference lake in fall 2017 (Table 3.6; Appendix Table C.27). However, evaluation of dissolved manganese indicated comparable concentrations between Camp Lake and the reference lake during the summer and fall sampling events in 2017. Concentrations of total manganese, together with total aluminum, showed a significant positive correlation with turbidity at Camp Lake using all 2017 data (r = 0.69 and 0.66, respectively; Appendix Table C.28), suggesting that these metals were associated with suspended particulate material in Camp Lake and thus were unlikely to be bioavailable. Notably, concentrations of all parameters were well below established WQG and AEMP benchmarks at Camp Lake during all sampling events in 2017 (Table 3.6; Appendix Table C.26), further indicating that parameter concentrations at Camp Lake were unlikely to adversely affect biota.

Temporal comparisons of Camp Lake water chemistry data indicated that, of the parameters shown to be elevated at CLT1 in 2017, only conductivity and total concentrations of chloride, molybdenum, sodium, and less so, strontium and uranium, showed continuous increases over the mine baseline, construction and operational periods (Figure 3.10; Appendix Figure C.8). Other parameters, including hardness, iron, manganese, nitrate, and sulphate, showed no consistent direction of change between the mine baseline and operational periods. Notably, all parameter concentrations were consistently well below WQG and AEMP benchmarks through all years of mine construction and operation at Camp Lake (e.g., Appendix Table C.26).

3.3.3 Sediment Quality

Surficial sediment (i.e., top 2 cm) collected at the Camp Lake coring stations in 2017 was composed mainly of sandy 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). 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 Table D.13). Similar substrate was observed at Reference Lake 3 (Appendix Table D.3), 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 2017. Qualitative observations suggestive of reducing sediment conditions were similar between Camp Lake and Reference Lake 3 in 2017 (Appendix Tables D.3 and D.13), which indicated that factors leading to reduced sediment conditions were comparable between lakes.

Figure 3.10: Temporal Comparison of Water Chemistry at Camp Lake (JLO) for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015 - 2017) Periods During Fall

Notes: Values represent mean (± SD) of surface and bottom measures. 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.

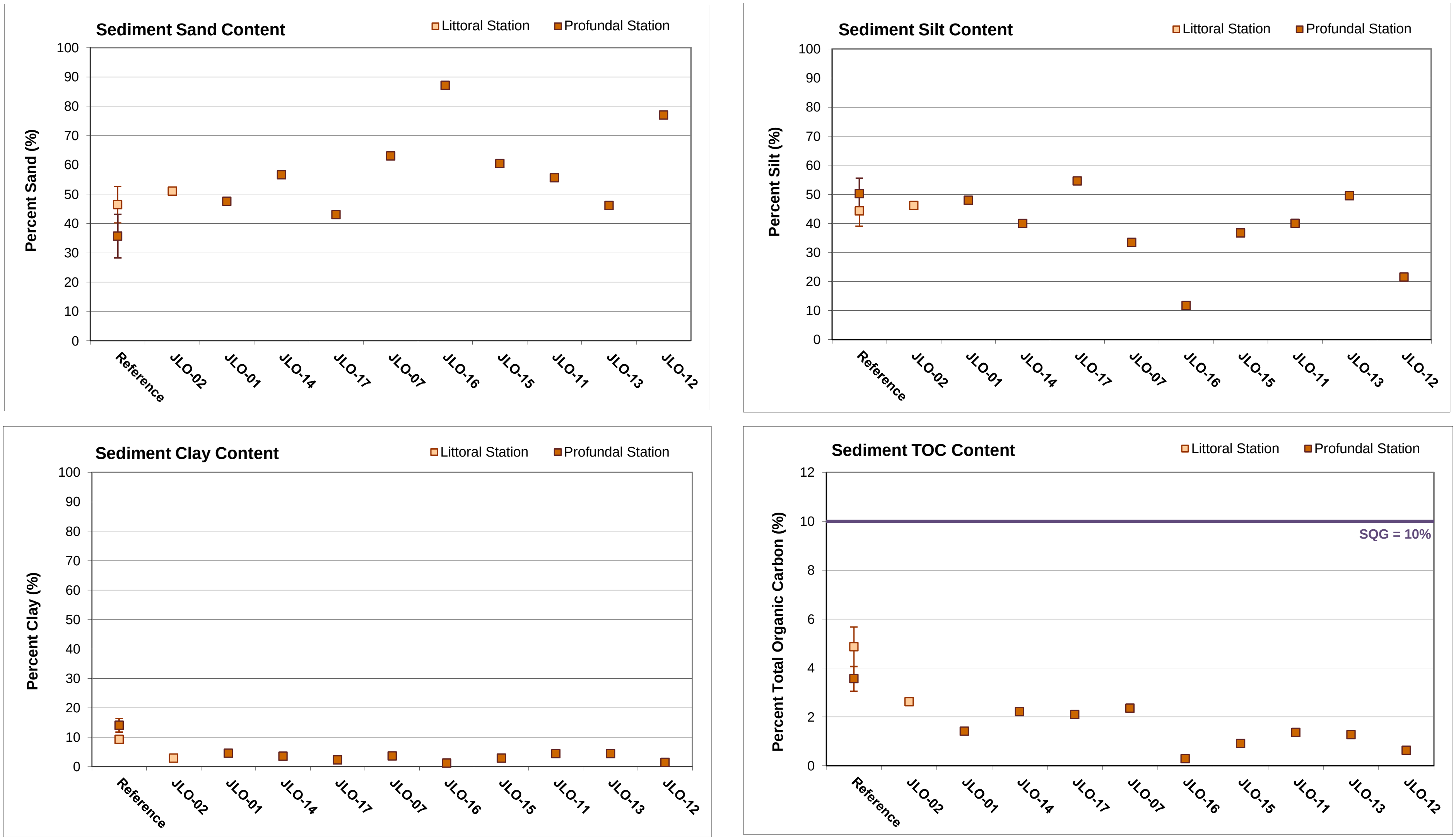


Figure 3.11: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Camp Lake (JLO) Sediment Monitoring Stations and to Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2017

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 2017 (Appendix Table D.15). Arsenic and manganese concentrations were slightly elevated (i.e., 3- to 5-fold higher) at the single Camp Lake littoral station (i.e., Station JLO-02) compared to sediment at Reference Lake 3 littoral stations (Table 3.7; Appendix Table D.16). Manganese, iron, and nickel concentrations were above respective SQG, and arsenic, iron, nickel, and phosphorus were above respective AEMP benchmarks at the Camp Lake littoral station (Table 3.7). Of these metals, average iron concentrations were also above SQG in littoral sediment of Reference Lake 3 (Table 3.7). 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. Notably, metal concentrations were considerably higher in sediment of Camp Lake than in deposited sediment at either of the Camp Lake tributary study areas (Appendix Table D.18), reflecting the depositional nature of Camp Lake versus erosional characteristics of the tributaries. Metal concentrations in the profundal sediment of Camp Lake were comparable to those of the reference lake in 2017 (Table 3.7; Appendix Table D.16). Although mean iron and manganese concentrations were above respective SQG at Camp Lake profundal stations, mean concentrations of these metals in profundal sediment of Reference Lake 3 were also above SQG in 2017 (Table 3.7) indicating naturally high concentrations of iron and manganese in sediment of lakes in the mine local study area. On average, only arsenic concentrations were above applicable AEMP benchmarks in profundal sediment of Camp Lake in 2017 (Table 3.7).

Temporal comparisons indicated that average metal concentrations in sediment at Camp Lake littoral and profundal stations were comparable between 2017 and the baseline period for each respective station type, the only exception of which was slightly higher (i.e., 3- to 5-fold greater) arsenic concentrations in sediment at the single Camp Lake littoral station in 2017[6] (Figure 3.12; Appendix Table D.16). Average metal concentrations in sediment at Camp Lake littoral and profundal stations in 2017 were typically within the range of those observed from 2015 and 2016, with no consistently higher concentrations occurring that would suggest an increasing trend since the commencement of commercial mine operations (Figure 3.12; Appendix Table D.18). Notably, sediment metal concentrations at Camp Lake in 2017 were very similar to those measured in 2015, when the same station locations were sampled (Figure 3.12). Overall, with the exception of a step-increase in arsenic concentrations shown at the littoral station closest to the CLT1 inlet

[6] Reported sediment boron concentrations in 2015, 2016 and 2017 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.

Table 3.7: Sediment Total Organic Carbon and Metal Concentrations at Camp Lake (JLO) and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2017

	Analyte	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b]	Littoral Stations		Profundal Stations	
					Reference Lake (n = 5)	Camp Lake (n = 1)	Reference Lake (n = 5)	Camp Lake (n = 9)
					Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error
Total Organic Carbon		%	10[α]	-	4.87 ± 0.81	1.32 ± 0.52	3.60 ± 0.52	1.39 ± 0.24
Metals	Aluminum (Al)	mg/kg	-	-	14,720 ± 736	15,000 ± 0	22,140 ± 1,652	13,908 ± 1,501
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	0.13 ± 0.00	<0.10 ± 0.00	<0.10 ± 0
	Arsenic (As)	mg/kg	17	5.9	3.35 ± 0.64	***9.92*** ± 0.00	4.80 ± 0.20	***6.68*** ± 1.98
	Barium (Ba)	mg/kg	-	-	113 ± 10	163 ± 0	138 ± 7	90 ± 19
	Beryllium (Be)	mg/kg	-	-	0.57 ± 0.02	0.80 ± 0.00	0.85 ± 0.06	0.75 ± 0.08
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.28 ± 0.00	<0.20 ± 0.00	0.25 ± 0.02
	Boron (B)	mg/kg	-	-	12.1 ± 0.8	16.8 ± 0.0	16.6 ± 1.5	18.6 ± 1.9
	Cadmium (Cd)	mg/kg	3.5	1.5	0.182 ± 0.032	0.203 ± 0.000	0.179 ± 0.024	0.133 ± 0.018
	Calcium (Ca)	mg/kg	-	-	4,656 ± 362	4,560 ± 0	5,262 ± 377	5,101 ± 1,247
	Chromium (Cr)	mg/kg	90	98	51.1 ± 3.7	69.1 ± 0.0	70.2 ± 6.3	62.4 ± 5.5
	Cobalt (Co)	mg/kg	-	-	10.63 ± 0.79	21.30 ± 0.00	15.98 ± 1.14	15.97 ± 1.86
	Copper (Cu)	mg/kg	110[α]	50	***64.7*** ± 5.3	44.5 ± 0.0	***88*** ± 8.5	37.5 ± 4.7
	Iron (Fe)	mg/kg	40,000[α]	52,400	41,960 ± 8,713	***61,600*** ± 0	46,740 ± 3,447	42,100 ± 8,092
	Lead (Pb)	mg/kg	91	35	12.4 ± 0.6	17.5 ± 0.0	16.9 ± 1.0	16.3 ± 2.1
	Lithium (Li)	mg/kg	-	-	24.0 ± 0.9	26.2 ± 0.0	35.0 ± 2.1	25.1 ± 2.8
	Magnesium (Mg)	mg/kg	-	-	10,256 ± 714	13,800 ± 0	14,660 ± 1,126	12,373 ± 742
	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	639 ± 115	1,960 ± 0	1,266 ± 70	1,465 ± 512
	Mercury (Hg)	mg/kg	0.486	0.17	0.0440 ± 0.0081	0.0483 ± 0.0000	0.0528 ± 0.0089	0.0327 ± 0.0064
	Molybdenum (Mo)	mg/kg	-	-	3.50 ± 0.97	2.29 ± 0.00	2.90 ± 0.20	1.35 ± 0.49
	Nickel (Ni)	mg/kg	75[α,β]	72	38.2 ± 2.4	***77.0*** ± 0.0	49.3 ± 3.0	57.6 ± 4.8
	Phosphorus (P)	mg/kg	2,000[α]	1,580	1,039 ± 246	***1,690*** ± 0	1,073 ± 54	1,377 ± 301
	Potassium (K)	mg/kg	-	-	3,754 ± 212	3,670 ± 0	5,694 ± 366	3,691 ± 402
	Selenium (Se)	mg/kg	-	-	0.61 ± 0.12	0.46 ± 0.00	0.59 ± 0.12	0.34 ± 0.05
	Silver (Ag)	mg/kg	-	-	0.13 ± 0.01	0.12 ± 0.00	0.20 ± 0.03	0.12 ± 0.01
	Sodium (Na)	mg/kg	-	-	284.2 ± 21	174 ± 0	410 ± 36	197 ± 32
	Strontium (Sr)	mg/kg	-	-	10.0 ± 0.5	8.1 ± 0.0	12.7 ± 0.8	10.8 ± 1.3
	Thallium (Tl)	mg/kg	-	-	0.346 ± 0.036	0.458 ± 0.000	0.657 ± 0.038	0.374 ± 0.047
	Tin (Sn)	mg/kg	-	-	2.1 ± 0.1	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0
	Titanium (Ti)	mg/kg	-	-	997 ± 51	884 ± 0	1,288 ± 56	744 ± 62
	Uranium (U)	mg/kg	-	-	11.0 ± 1.0	5.43 ± 0.0	23.5 ± 2.3	4.31 ± 0.6
	Vanadium (V)	mg/kg	-	-	47.8 ± 2.0	57.0 ± 0.0	67.2 ± 3.9	50.4 ± 5.2
	Zinc (Zn)	mg/kg	315	135	67.32 ± 2.5	57.6 ± 0.0	91 ± 6.8	47.7 ± 5.1
	Zirconium (Zr)	mg/kg	-	-	3.9 ± 0.4	6.6 ± 0.0	3.4 ± 0.1	4.6 ± 0.5

[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.

Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

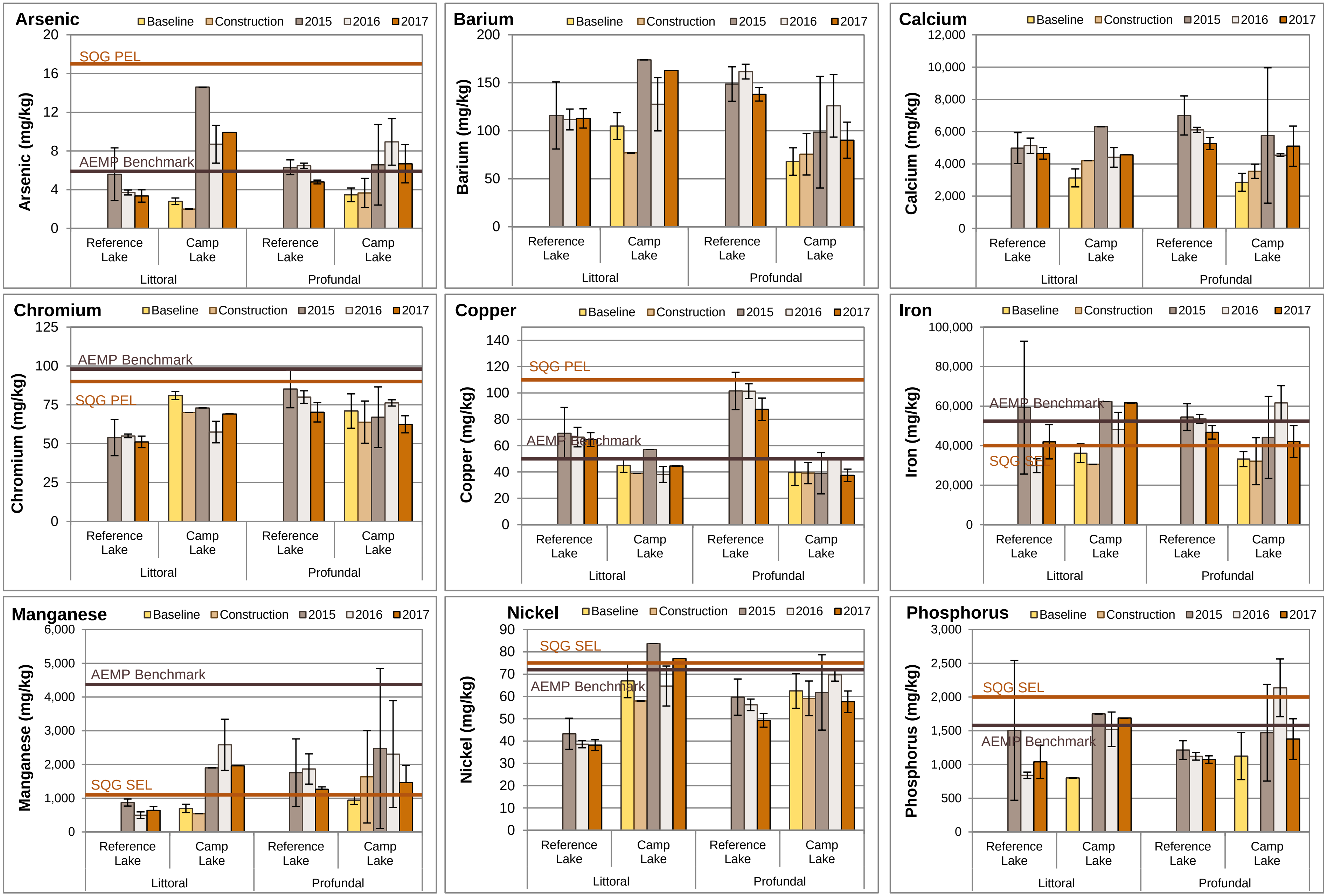


Figure 3.12: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Camp Lake and Reference Lake 3 for Mine Baseline (2005 - 2013), Construction (2014) and Operational (2015, 2016, and 2017) Periods, Mary River Project CREMP, 2017

to Camp Lake in 2015, and taking reference lake data into consideration, no substantial changes to sediment metal concentrations were indicated at Camp Lake littoral and profundal stations following the commencement of Baffinland 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 2017, although concentrations were somewhat lower at stations located nearer the lake outlet during the winter sampling event (Figure 3.13). Chlorophyll-a concentrations did not differ significantly between summer and fall sampling events, but concentrations during both of these sampling events were each significantly higher than during the winter sampling event at Camp Lake in 2017 (Appendix Table E.6). 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 density at Camp Lake. 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 2017 (Figure 3.13). Camp Lake mean chlorophyll-a concentrations in 2017 suggested low phytoplankton productivity and an ‘oligotrophic’ trophic status based on comparison to Wetzel (2001) lake classification 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 2017 lake 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 among the mine construction (2014) and operational (2015, 2016, 2017) years for seasonal data collected in winter, summer, and fall (Figure 3.14). In addition, annual average chlorophyll-a concentrations did not differ significantly among the most recent four years (Appendix Table E.9), suggesting no changes in the trophic status of Camp Lake since mine operations commenced. No chlorophyll-a baseline (2005 – 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 3.8 and 3.9). 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

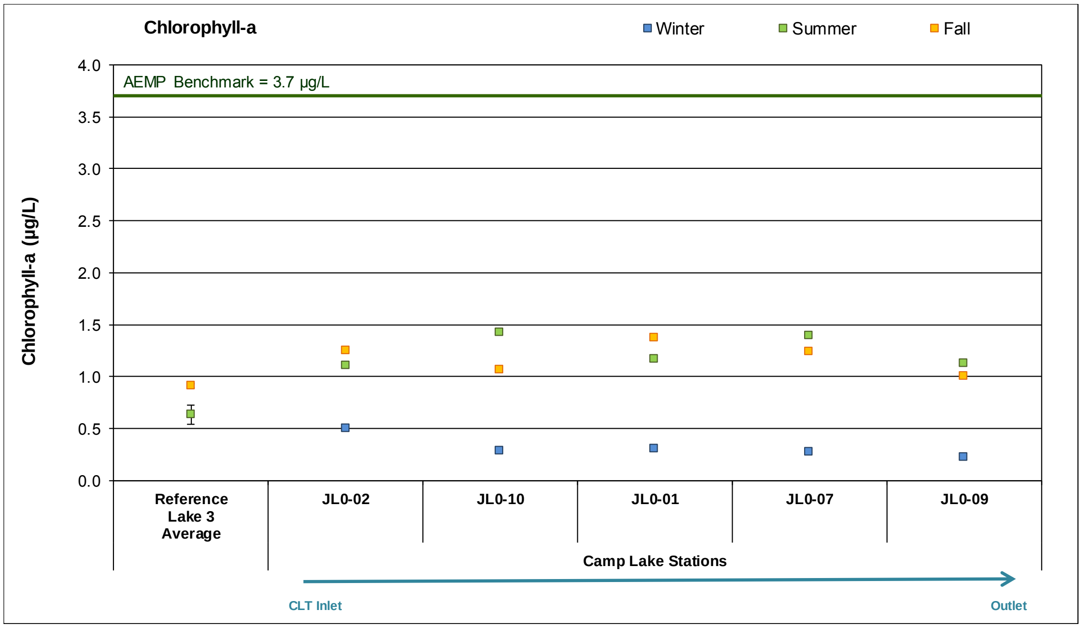


Figure 3.13: Chlorophyll-a Concentrations at Camp Lake (JLO) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2017

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 2017.

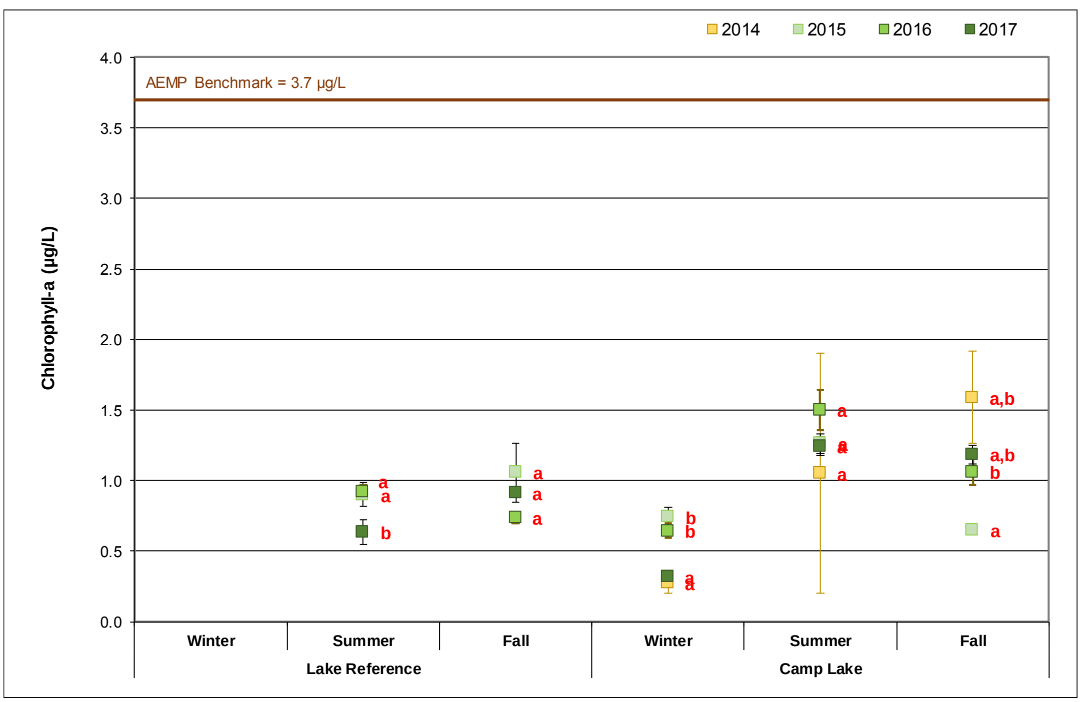


Figure 3.14: Chlorophyll-a Concentration Seasonal Comparison among 2014, 2015, 2016, and 2017 (mean ± SE) at Camp Lake Phytoplankton Monitoring Stations, Mary River Project CREMP

Notes: Data points with the same letter on the right do not differ significantly between years for the applicable season.

Table 3.8: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JLO) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	square root	YES	0.018	ANOVA	2.5	Reference Lake 3	1,489	850	380	372	2,618
						Camp Lake Littoral	3,642	1,449	648	2,377	5,790
Richness (Number of Taxa)	log	NO	0.781	ANOVA	0.2	Reference Lake 3	12.4	2.5	1.1	10.0	15.0
						Camp Lake Littoral	12.8	2.3	1.0	10.0	15.0
Simpson's Evenness (E)	none	YES	< 0.001	ANOVA	0.3	Reference Lake 3	0.807	0.142	0.063	0.605	0.934
						Camp Lake Littoral	0.848	0.068	0.031	0.751	0.938
Bray-Curtis Index	none	YES	< 0.001	ANOVA	3.2	Reference Lake 3	0.384	0.120	0.054	0.203	0.536
						Camp Lake Littoral	0.763	0.060	0.027	0.690	0.821
Nemata (%)	square root	NO	0.866	ANOVA	0.1	Reference Lake 3	3.9	3.3	1.5	0.8	9.1
						Camp Lake Littoral	4.2	4.2	1.9	0.0	8.8
Hydracarina (%)	square root	NO	0.173	ANOVA	-0.9	Reference Lake 3	5.3	3.0	1.4	1.8	8.1
						Camp Lake Littoral	2.7	3.4	1.5	0.0	8.6
Ostracoda (%)	square root	YES	< 0.001	ANOVA	-2.1	Reference Lake 3	38.8	18.4	8.2	22.3	63.9
						Camp Lake Littoral	0.2	0.3	0.2	0.0	0.8
Chironomidae (%)	square root	YES	0.003	ANOVA	2.3	Reference Lake 3	51.8	17.9	8.0	29.5	67.9
						Camp Lake Littoral	92.2	6.5	2.9	82.8	98.6
Metal-Sensitive Chironomidae (%)	none	YES	0.049	ANOVA	1.7	Reference Lake 3	15.5	13.4	6.0	0.5	37.0
						Camp Lake Littoral	38.2	17.3	7.7	14.9	53.8
Collector-Gatherers (%)	log	YES	0.056	ANOVA	-1.4	Reference Lake 3	73.9	16.0	7.2	56.0	95.5
						Camp Lake Littoral	50.8	17.4	7.8	34.7	74.7
Filterers (%)	none	YES	0.049	ANOVA	1.7	Reference Lake 3	14.7	13.3	5.9	0.5	36.4
						Camp Lake Littoral	37.3	17.3	7.7	13.4	52.4
Shredders (%)	log(X+1)	NO	0.598	ANOVA	-0.4	Reference Lake 3	4.2	6.6	3.0	0.0	15.9
						Camp Lake Littoral	1.8	1.8	0.8	0.0	4.6
Clingers (%)	none	NO	0.128	ANOVA	1.3	Reference Lake 3	19.8	12.8	5.7	2.7	38.1
						Camp Lake Littoral	36.9	18.4	8.2	7.4	54.3
Sprawlers (%)	log	YES	0.033	ANOVA	-1.8	Reference Lake 3	72.6	13.9	6.2	55.7	92.9
						Camp Lake Littoral	48.1	18.5	8.3	33.4	79.2
Burrowers (%)	log	YES	0.082	ANOVA	2.5	Reference Lake 3	7.5	3.0	1.4	4.5	12.0
						Camp Lake Littoral	15.1	8.0	3.6	5.4	27.5

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

Gray 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 3.9: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JLO) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	log	YES	< 0.001	ANOVA	42.8	Reference Lake 3	149	32	14	113	190
						Camp Lake Profundal	1,510	844	377	509	2,549
Richness (Number of Taxa)	fourth root	YES	0.002	ANOVA	4.4	Reference Lake 3	4.2	1.5	0.7	2.0	6.0
						Camp Lake Profundal	10.8	3.3	1.5	8.0	16.0
Simpson's Evenness (E)	none	NO	0.791	ANOVA	-0.2	Reference Lake 3	0.704	0.105	0.047	0.597	0.843
						Camp Lake Profundal	0.681	0.154	0.069	0.520	0.861
Bray-Curtis Index	log	YES	< 0.001	ANOVA	5.4	Reference Lake 3	0.239	0.114	0.051	0.111	0.376
						Camp Lake Profundal	0.850	0.128	0.057	0.629	0.949
Nemata (%)	square root	YES	0.017	ANOVA	nc	Reference Lake 3	0.0	0.0	0.0	0.0	0.0
						Camp Lake Profundal	7.1	6.2	2.8	0.0	12.2
Hydracarina (%)	none	YES	0.065	ANOVA	-1.0	Reference Lake 3	8.2	5.5	2.5	0.0	15.0
						Camp Lake Profundal	2.5	2.2	1.0	0.0	5.3
Ostracoda (%)	log(X+1)	NO	0.264	ANOVA	-0.6	Reference Lake 3	2.8	4.1	1.8	0.0	8.9
						Camp Lake Profundal	0.3	0.6	0.3	0.0	1.4
Chironomidae (%)	log	NO	0.816	ANOVA	0.1	Reference Lake 3	89.0	7.6	3.4	81.6	100.0
						Camp Lake Profundal	90.0	6.6	2.9	82.9	99.2
Metal-Sensitive Chironomidae (%)	square root	YES	0.097	ANOVA	2.4	Reference Lake 3	12.0	8.9	4.0	0.0	20.2
						Camp Lake Profundal	33.3	25.5	11.4	9.7	72.7
Collector-Gatherers (%)	none	NO	0.151	ANOVA	-4.9	Reference Lake 3	84.3	4.1	1.9	79.8	90.5
						Camp Lake Profundal	64.2	28.1	12.6	20.2	88.2
Filterers (%)	none	YES	0.079	ANOVA	2.7	Reference Lake 3	6.5	9.3	4.2	0.0	20.2
						Camp Lake Profundal	31.6	26.4	11.8	8.6	72.7
Clingers (%)	none	YES	0.028	ANOVA	3.7	Reference Lake 3	14.6	4.9	2.2	9.5	20.2
						Camp Lake Profundal	33.1	27.8	12.4	9.0	78.3
Sprawlers (%)	square root	YES	0.021	ANOVA	-4.6	Reference Lake 3	81.4	6.3	2.8	74.8	90.5
						Camp Lake Profundal	52.1	23.7	10.6	14.5	79.3
Burrowers (%)	square root	YES	0.063	t-test (unequal variance)	2.9	Reference Lake 3	3.9	3.7	1.7	0.0	8.0
						Camp Lake Profundal	14.9	8.3	3.7	7.2	28.9

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

Gray 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 Critical Effect Size of ±2 SD_{REF}, indicating that the difference was ecologically meaningful.

like-habitat at the reference lake by a magnitude outside of the CES_{BIC} of ±2 SD_{REF} (Tables 3.8 and 3.9). 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 3.8 and 3.9). Higher benthic invertebrate density and richness at Camp Lake compared to the reference lake, which would contribute to the differences in Bray-Curtis Index, was not consistent with adverse influences typically associated with mine operations. This suggested that factors other than metal concentrations likely accounted for the differences in benthic invertebrate community structure between Camp Lake and Reference Lake 3. Notably, for the above general benthic invertebrate community metrics that differed significantly, a greater magnitude of difference was indicated between study lakes for profundal habitat than for littoral habitat (Tables 3.8 and 3.9). In turn, this suggested that habitat variables associated with greater depth likely contributed to more pronounced differences in benthic invertebrate community structure between lakes.

The key differences in benthic invertebrate community structure at magnitudes outside of CES_{BIC} included significantly lower and higher relative abundance of Ostracoda (seed shrimp) and Chironomidae (non-biting midges), respectively, at littoral habitat of Camp Lake compared to the reference lake (Tables 3.8 and 3.9). The relative abundance of metal-sensitive Chironomidae was also significantly higher at Camp Lake than at Reference Lake 3 for both habitat types (Tables 3.8 and 3.9), suggesting that the difference in benthic invertebrate community structure between lakes was unlikely associated with differences in metal concentrations. This was supported by water quality monitoring data that showed aqueous metal concentrations below WQG and AEMP benchmarks at Camp Lake (Appendix Table C.26), and by sediment quality monitoring data that showed sediment metal concentrations were generally below SQG at Camp Lake with the exception of iron and manganese, which were also above SQG at Reference Lake 3 (Table 3.7).

The differences in benthic invertebrate community structure between Camp Lake and Reference Lake 3 were consistent with differences in food resources between lakes. For instance, greater benthic invertebrate density and significantly higher relative abundance of the filterer FFG suggested higher phytoplankton productivity at Camp Lake than at Reference Lake 3, which was supported by observations of significantly lower Secchi depth and significantly higher summer and fall chlorophyll-a concentrations at Camp Lake in 2017 (Appendix Tables C.25, E.7 – E.8). Similarly, significantly lower relative abundance of Ostracoda and the collector-gatherer FFG at Camp Lake compared to the reference lake may have been related to significantly lower sediment TOC (Appendix Table F.21), which serves as a key food source for these groups. These observations suggested that benthic invertebrates of Camp Lake utilize autochthonous food sources (i.e., primary producers) to a greater extent than at Reference Lake 3, and that Camp Lake was naturally more productive than Reference Lake 3.

Temporal comparisons did not indicate any consistent ecologically significant differences in general community descriptors of density, richness, and Simpson's Evenness at littoral and profundal habitats of Camp Lake between the mine baseline (2007, 2013) period and individual years since the commencement of commercial mine operation (2015, 2016, 2017; Figure 3.15; Appendix Tables F.23 and F.24). Similarly, no significant differences in benthic invertebrate dominant taxonomic groups or FFG were indicated between baseline and mine operational years at littoral habitat of Camp Lake (Figure 3.15; Appendix Table F.23). Despite a significantly differing relative abundance of metal-sensitive chironomids and FFG between mine operation year 2015 and the 2007 and 2013 baseline data at profundal habitat of Camp Lake, similar differences were not indicated between 2017 and either of the 2007 or 2013 baseline data (Appendix Table F.24). 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 in 2017 (Table 3.10). 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 on greater gill netting CPUE from deeper littoral/profundal habitat at Camp Lake in 2017 (Table 3.10). In turn, this suggested higher fish productivity at Camp Lake compared to Reference Lake 3, and was consistent with chlorophyll-a results which indicated higher phytoplankton productivity 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 in 2017 (Table 3.10). Camp Lake electrofishing and gill netting CPUE for arctic charr in 2017 were within the range of those observed during baseline (2005 - 2013) studies, as well as to those during the previous two years of mine operation, for each respective collection method (Figure 3.16). In turn, this suggested no

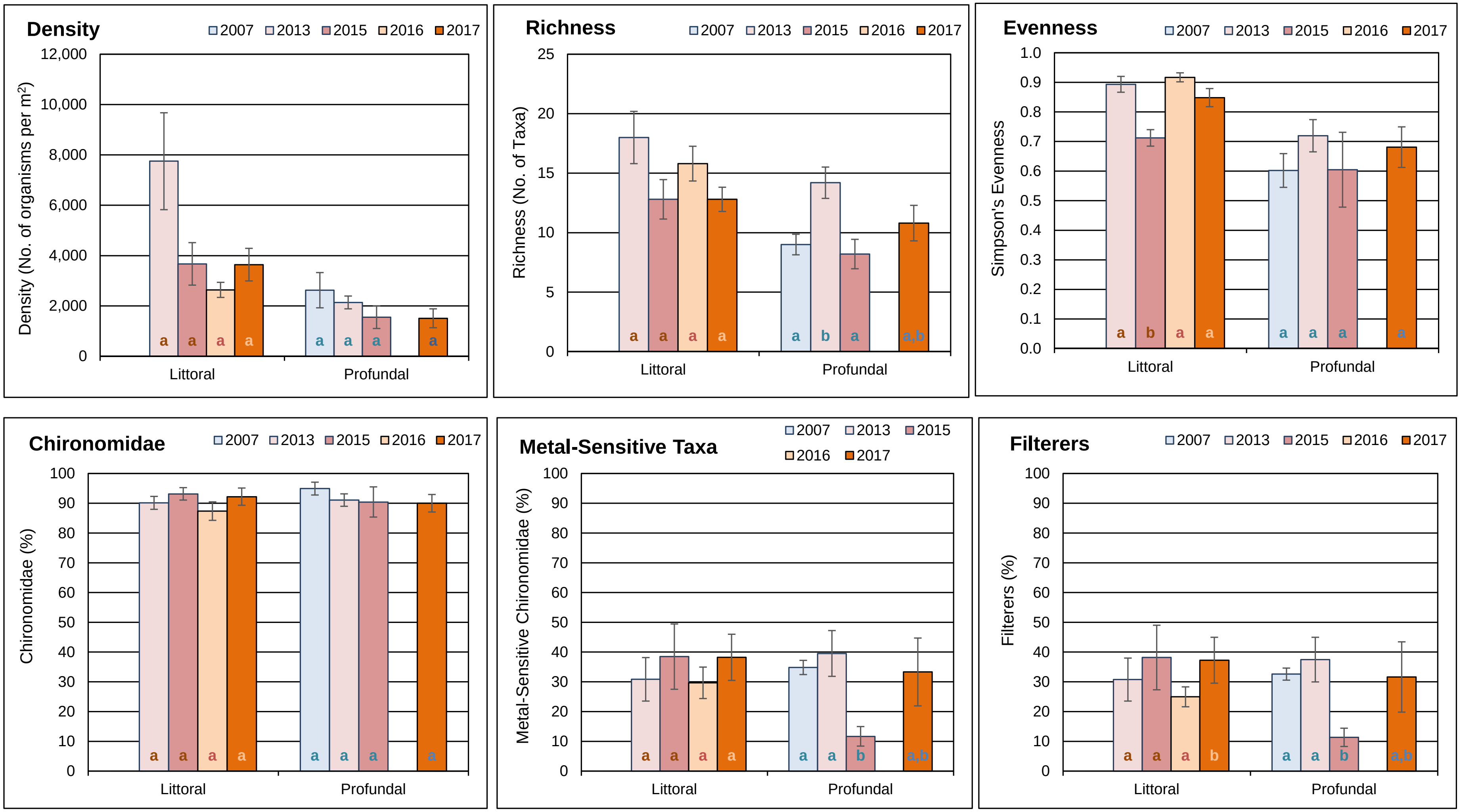


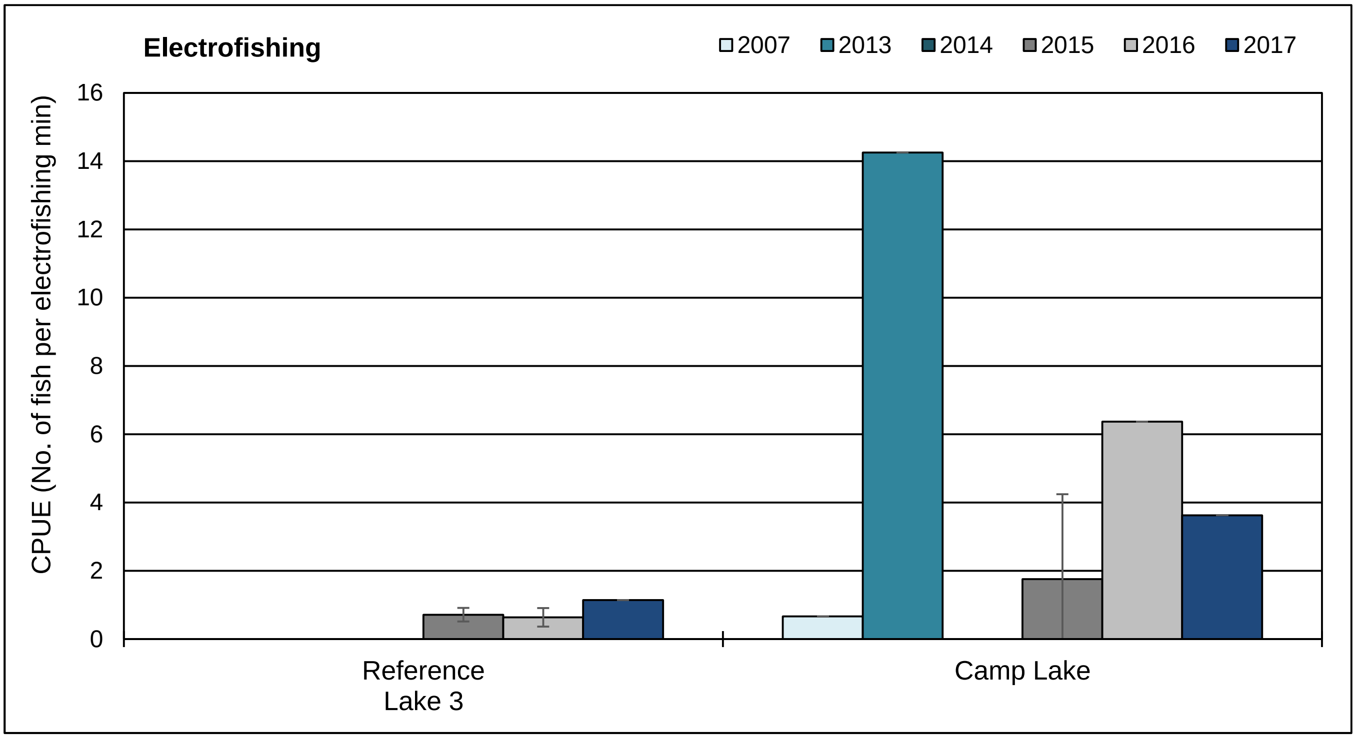
Figure 3.15: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Camp Lake Littoral and Profundal Study Areas among Mine Baseline (2007, 2013) and Operational (2015, 2016, 2017) Periods

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

Table 3.10: 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 2017

Lake	Method [a]		Arctic Charr	Nine-spine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	100	11	111	2
		CPUE	1.03	0.11	1.14	
	Gill netting	No. Caught	1	0	1	
		CPUE	0.02	0	0.02	
Camp Lake	Electrofishing	No. Caught	97	4	101	2
		CPUE	3.48	0.14	3.62	
	Gill netting	No. Caught	96	0	96	
		CPUE	1.40	0	1.40	

[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.



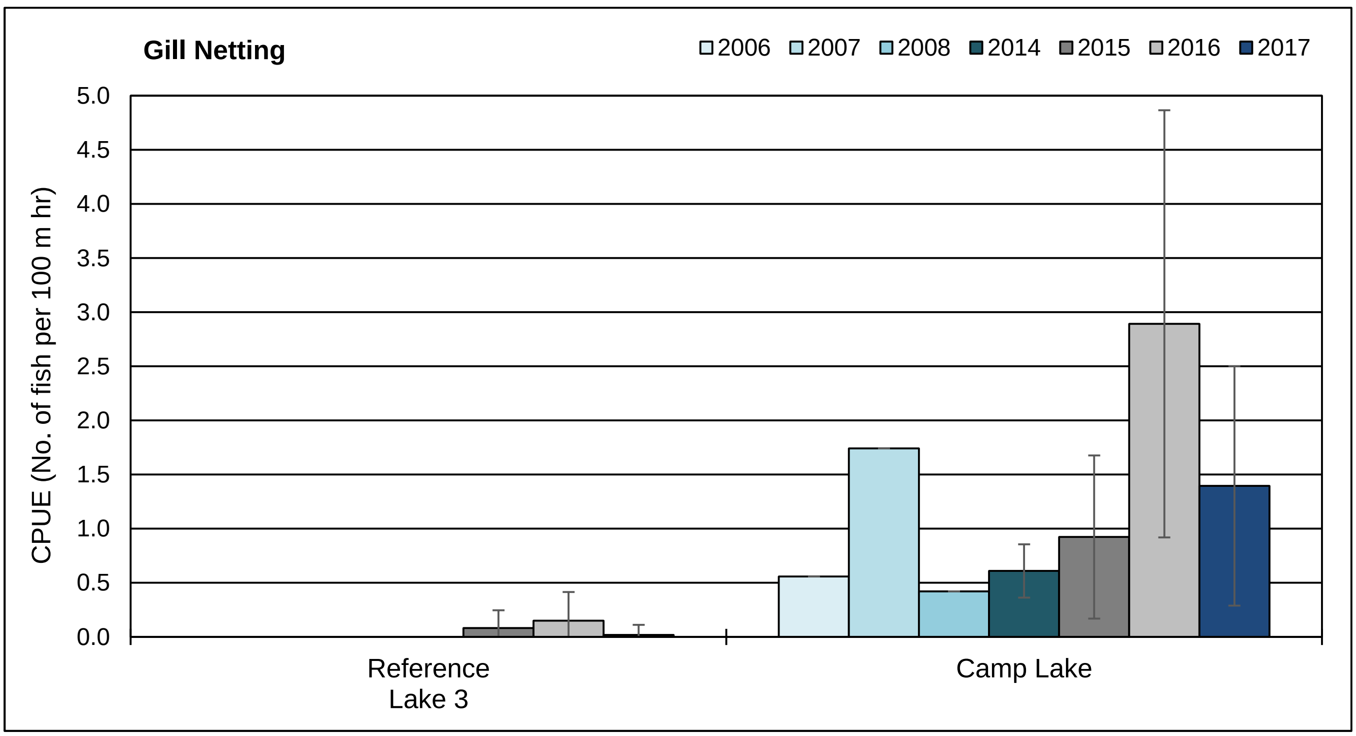


Figure 3.16: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, 2006 - 2017

Note: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015, 2016, 2017) mine phases.

substantial changes 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.

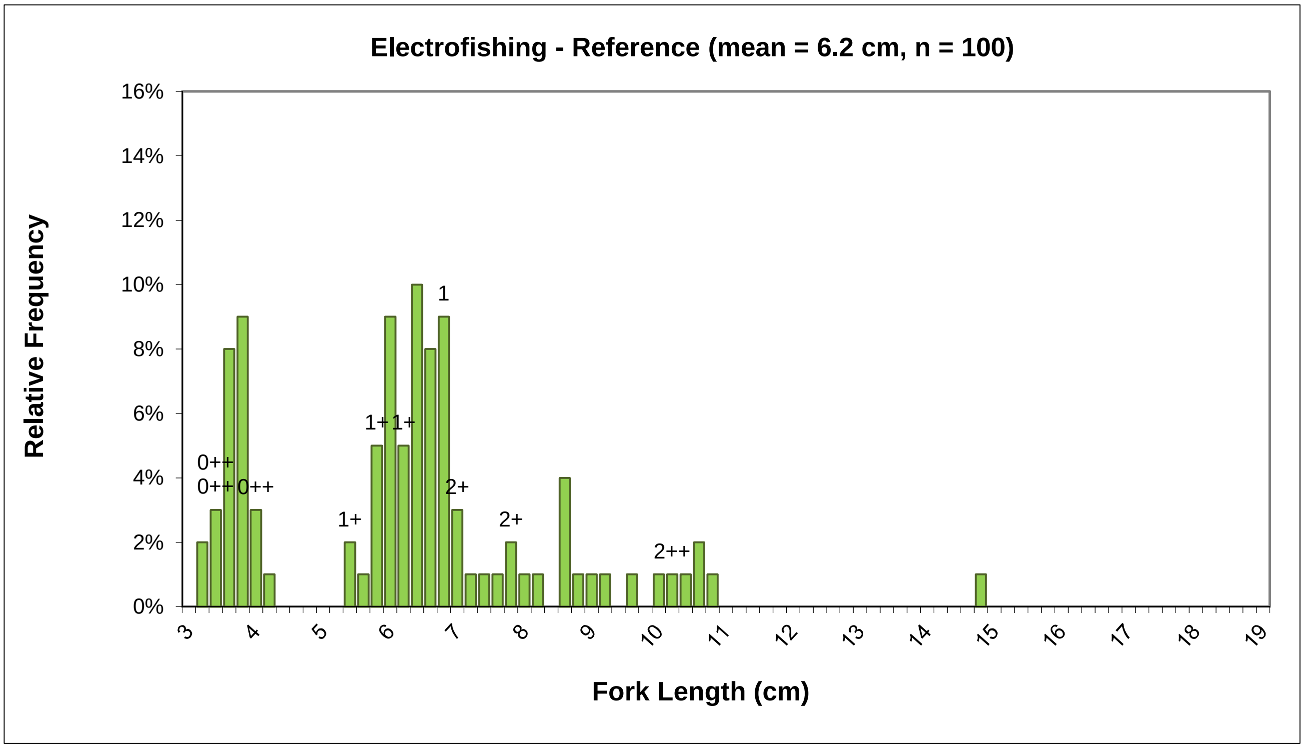
3.3.6.2 Camp Lake Fish Population Assessment

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 2017 data from Camp Lake and Reference Lake 3, as well as a before-after analysis using Camp Lake 2017 and baseline (2013) data. A total of 97 and 100 arctic charr were captured at nearshore habitat of Camp Lake and Reference Lake 3, respectively, in August 2017, 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 5.0 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 were conducted using only 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 3.11), 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 (17%) and heavier (51%) than those captured at the reference lake nearshore (Table 3.11; Appendix Table G.6). However, condition (i.e., weight-at-length relationship) of non-YOY arctic charr did not differ significantly between Camp Lake and the reference lake (Table 3.11; Appendix Table G.6). Consequently, no substantial differences in the overall health of nearshore non-YOY arctic charr was suggested between the Camp Lake and Reference Lake 3 populations in 2017.

Temporal comparisons of the Camp Lake nearshore non-YOY arctic charr data indicated significantly different length-frequency distribution between the 2017 study and the 2013 baseline study (Table 3.11). In addition, non-YOY arctic charr captured at the nearshore of Camp Lake in 2017 were significantly shorter (-35%), lighter (-74%) and of lower condition (-10%) than those captured during the 2013 baseline study (Table 3.8; Appendix Table G.7). Similar differences in nearshore non-YOY arctic charr size and condition were demonstrated in 2015 and 2016 compared to the 2013 baseline data (Table 3.11). In each of the individual 2015, 2016, and 2017 studies, the magnitude of difference in non-YOY arctic charr condition compared to the 2013 baseline data was just within the Critical Effect Size (CES) of ±10% (referred to herein as CES_C;



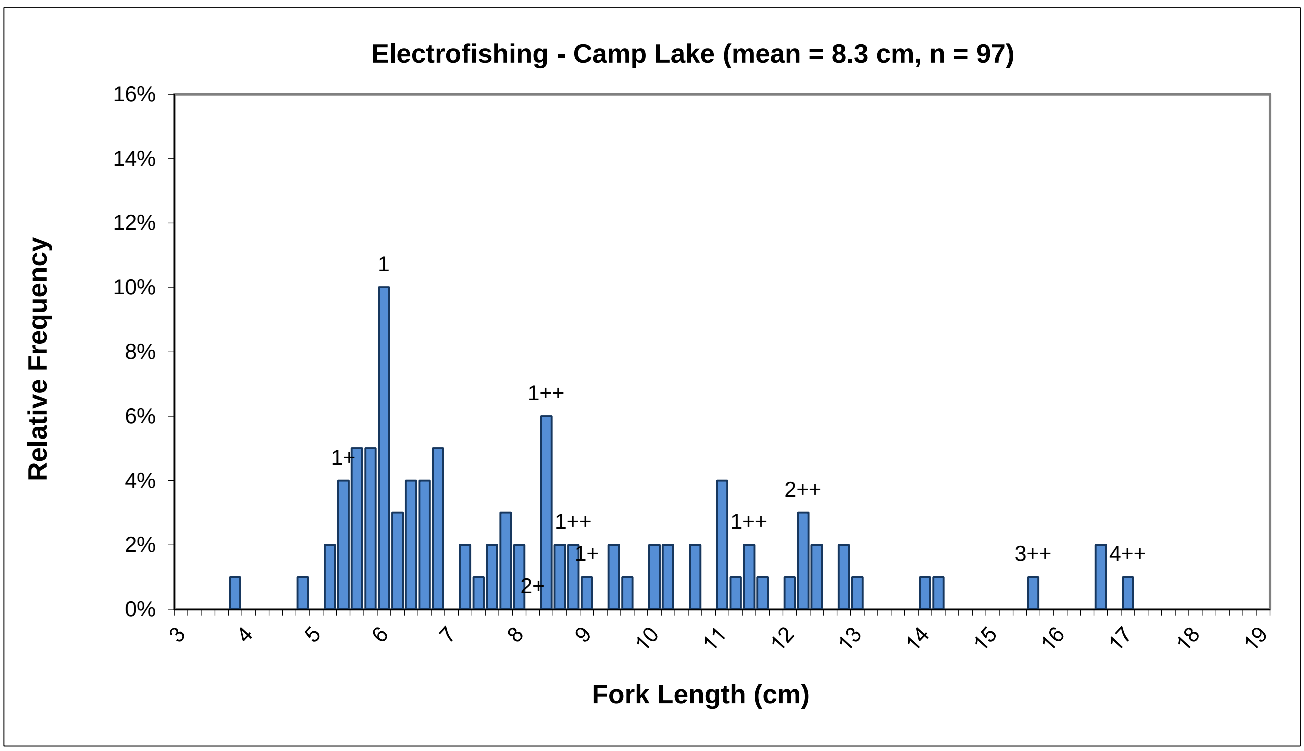


Figure 3.17: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2017

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

Table 3.11: Summary of Statistical Results for Arctic Charr Population Comparisons between Camp Lake and Reference Lake 3 in 2015, 2016, and 2017, 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	2015	2016	2017
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	-	-
	Energy Use (non-YOY)	Size (mean weight)	**Yes (+176%)**	No	**Yes (+51%)**	**Yes (-42%)**	**Yes (-71%)**	**Yes (-74%)**
		Size (mean fork length)	**Yes (+41%)**	No	**Yes (+17%)**	**Yes (-15%)**	**Yes (-32%)**	**Yes (- 35%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	No	**Yes (-6%)**	No	**Yes (-6%)**	**Yes (-10%)**	**Yes (-10%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**
		Age	-	-	-	**Yes (+48%)**	**Yes (+58%)**	**Yes (+ 46%)**
	Energy Use	Size (mean weight)	-	-	-	No	No	**Yes (+37%)**
		Size (mean fork length)	-	-	-	**Yes (+6%)**	No	**Yes (+12%)**
		Growth (weight-at-age)	-	-	-	No	**Yes (nc)**	No
		Growth (fork length-at-age)	-	-	-	No	**Yes (nc)**	No
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	No	**Yes (-3%)**	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.

Table 3.11). 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 the range of variability expected to occur naturally between years at waterbodies uninfluenced by human activity.

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 using a before-after analysis of Camp Lake 2017 versus baseline (combined 2006, 2007, 2008, and 2013) data. Similar to the two previous CREMP studies, despite a total of 96 arctic charr captured by gill netting at littoral/profundal areas of Camp Lake and application of similar fishing effort, the sample size of arctic charr from Reference Lake 3 was very small (i.e., 1) in August 2017, precluding a control-impact analysis for the determination of mine-related effects using the littoral/profundal catch data. Biological information collected from arctic charr mortalities encountered during the 2017 Camp Lake littoral/profundal sampling suggested that 23% of the population was represented by non-spawners of reproductive age (referred to simply as non-spawners herein; Appendix Table G.12). The average age, length, and weight of non-spawners was comparable to that of female spawners (Appendix Table G.12) indicating that, typical of high Arctic systems, individual arctic charr do not spawn yearly at Camp Lake. Liver somatic index (LSI) did not differ significantly between non-spawners and female spawners (ANOVA; $p = 0.22$), suggesting similar energy reserves available for gamete development between these groups. Internal body cavity parasites were present in all arctic charr incidental mortalities (Appendix Table G.12), which could contribute to biennial or longer frequency between spawning events for arctic charr in the mine local study area lakes as a result of lower energy applied towards gamete production stemming from infections with parasites. High incidence of internal parasites in arctic charr of the Mary River Project mine area lakes was noted in baseline studies (NSC 2014, 2015a) as well as in each of the two previous CREMP studies (Minnow 2016a, 2017).

Temporal comparisons of arctic charr data collected from Camp Lake littoral/profundal areas indicated significantly different length-frequency distribution of arctic charr in 2017 compared to the combined baseline data set (i.e., 2006, 2007, 2008, and 2013 studies; Table 3.11). The differences in length-frequency distributions were consistent with the analysis of significantly older arctic charr at Camp Lake in 2017 compared to the baseline period (Table 3.11; Appendix Table G.13). Fork length and fresh body weight were significantly larger in arctic charr captured at Camp Lake in 2017 compared to the baseline period. However, arctic charr of spawning size showed no significant differences in growth or condition between 2017 and the baseline period at Camp Lake (Table 3.11). These results were consistent with those of the two previous CREMP studies, which collectively indicated no ecologically meaningful differences in size, growth, or condition of spawning-sized arctic charr at Camp Lake between the mine operational years and the baseline period.

3.3.7 Integrated Effects Evaluation

Potential mine-related influences on water quality of Camp Lake in 2017 included slightly elevated chloride and manganese concentrations compared to the reference lake, as well as slightly higher conductivity and concentrations of chloride, molybdenum, sodium, and less so, strontium and uranium, compared to 2005 - 2013 baseline data. However, in all cases, parameter concentrations at Camp Lake were consistently well below WQG and AEMP benchmarks in 2015, 2016 and 2017. Sediment arsenic and manganese concentrations were elevated at the single Camp Lake littoral station compared to the reference lake in 2017 and, for arsenic, to concentrations observed during the baseline period. However, no metals were elevated in sediment at Camp Lake profundal stations compared to the reference lake in 2017, nor to concentrations shown in mine baseline studies. Although spatial analysis was limited by the collection of sediment chemistry from only a single littoral station at Camp Lake under the AEMP, elevated metal concentrations at this station suggested that mine-influenced flow from CLT1 was likely the source of these metals. 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 the reference lake indicating natural elevation of these metals in sediments of regional lakes. Arsenic, iron, nickel, and phosphorus concentrations were above AEMP benchmarks at the littoral station, as was the average arsenic concentration at profundal stations at Camp Lake in 2017. Overall, recent mine operations appeared to contribute to higher chloride, molybdenum, manganese, and sodium concentrations in water, as well as to slightly higher arsenic, nickel, and phosphorus concentrations in sediment of Camp Lake. However, concentrations of these parameters remained below applicable guidelines with the exception of nickel, which was slightly above the SQG at the littoral station, suggesting a low potential for adverse effects to biota of Camp Lake.

Camp Lake chlorophyll-a concentrations were significantly higher than at the reference lake in 2017 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 indicated no changes to the trophic status of Camp Lake since mine operations commenced at the Mary River Project. Benthic invertebrate density, richness, Simpson's Evenness, and relative abundance of metal-sensitive chironomids was higher at Camp Lake than at the reference lake in 2017. In addition, no ecologically significant differences in primary benthic invertebrate community metrics, dominant taxonomic groups, or FFG were consistently indicated between the mine baseline (2007, 2013) period and individual years since the commencement of commercial mine operation (2015, 2016, 2017). Analysis of Camp Lake arctic charr populations suggested greater fish abundance compared to the reference lake in 2017, but similar numbers of arctic charr in 2017 relative to the Camp Lake baseline studies. No significant, ecologically meaningful, differences in arctic charr condition were indicated between Camp Lake and the reference lake in 2017, nor between Camp Lake arctic charr collected in 2017 compared to the baseline period, for nearshore and littoral/profundal arctic charr populations. Collectively, 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.