APPENDIX G

Supporting Studies and Reports

APPENDIX G.1

CREMP Report

minnow environmental inc.
A Trinity Consultants Company

Mary River Project 2020
Core Receiving Environment Monitoring Program Report

Part 1 of 3
(Sections 1 to 7)

Prepared for:
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Mary River Project 2020
Core Receiving Environment Monitoring Program Report

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EXECUTIVE SUMMARY

The Mary River Project (the 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 Project include open pit ore extraction, ore haulage, stockpiling, crushing, and screening, followed by transport by truck to Milne Port for subsequent seasonal loading onto bulk carrier ships for transfer to international markets. No milling or additional processing of the ore is conducted on-site and therefore no tailings are produced at the Project. Mine waste management facilities at the mine site include a mine waste rock stockpile and surface runoff collection/containment ponds situated near the mine waste rock stockpile and ore stockpile areas. In addition to periodic discharge of treated effluent from these facilities to the Mary River system, other potential mine inputs to aquatic systems located adjacent to the Mine Site include runoff and dust from ore (crusher) stockpiles located within the Sheardown Lake catchment, treated sewage effluent discharge to Mary River, runoff and explosives residue deposition from quarry operations within the Camp Lake catchment, deposition of fugitive dust generated by mine activities, and general mine site runoff.

Under the terms and conditions of the Project's Type 'A' Water Licence issued by the Nunavut Water Board, Baffinland was required to develop and implement an Aquatic Effects Monitoring Plan (AEMP) at the Mine Site. In order to meet the AEMP objectives, Baffinland developed a Core Receiving Environment Monitoring Program (CREMP) to provide a basis for the evaluation of mine-related influences on water quality, sediment quality, and/or aquatic biota (including phytoplankton, benthic invertebrates, and fish). The primary receiving systems that serve as the focus for the CREMP include the Camp Lake system (i.e., Camp Lake tributaries 1 and 2, Camp Lake), the Sheardown Lake system (i.e., Sheardown Lake tributaries 1, 9, and 12, Sheardown Lake northwest basin, and Sheardown Lake southeast basin), and the Mary River and Mary Lake system. Potential mine related effects within the mine primary receiving systems have been assessed annually under the CREMP since the commencement of commercial mine operation in 2015 using a combination of comparisons to site-specific benchmarks for water and sediment quality developed for the AEMP and application of an effects-based approach using standard environmental effects monitoring techniques. Annual results from the CREMP are applied within a four-step Assessment Approach and Management Response Framework designed for the Mary River Project AEMP to then guide management response decisions related to changes in parameter concentrations and/or aquatic biota attributable to mine operations.

The results of the 2020 CREMP indicated mine-related influences on water and sediment quality at some of the primary receiving systems, but no ecologically significant, adverse, mine-related effects to biota were identified at any of the receiving waterbodies based on comparisons to applicable reference and/or baseline conditions. Within the Camp Lake system, copper concentrations were elevated above site-specific AEMP water quality benchmarks at the north branch of Camp Lake Tributary 1 (CLT1) in 2020, but because this elevation in copper concentrations did not appear to be mine-related, a low action response to identify the source of copper to the CLT1 north branch using expanded water quality monitoring is recommended. At the CLT1 upper main stem, iron concentrations were elevated above AEMP water quality benchmarks, concentrations at reference creeks, and concentrations during baseline, indicating a mine-related change that prompted a low action response recommendation to establish assess effects on biota within the upper main stem through the establishment of benthic invertebrate community sampling stations. At Camp Lake Tributary 2, no changes in concentrations of AEMP benchmark parameters occurred relative to background or to baseline and no adverse biological effects were indicated in 2020, and thus no adjustments to the existing AEMP are recommended. At Camp Lake, arsenic concentrations were elevated within littoral sediment compared to reference lake sediments and to Camp Lake baseline data. No mine-related sources of arsenic to the Camp Lake system have been evident currently or in the past, and therefore a low action response to harmonize sediment quality and benthic invertebrate community monitoring stations using increased replication at littoral habitat of Camp Lake is recommended. No

Within the Sheardown Lake system, copper concentrations were elevated above site-specific AEMP water quality benchmarks at Sheardown Lake Tributary 1 (SDLT1) in 2020, but because this elevation in copper concentrations did not appear to be mine-related, a low action response is recommended to identify the source of copper to SDLT1 using expanded water quality monitoring. No mine-related changes to phytoplankton or benthic invertebrates were indicated at Sheardown Lake tributaries 9 and 12 in 2020, but because water quality is not monitored at these tributaries under the current AEMP, a low action response to add a water quality monitoring station at each of these two tributaries is recommended to improve the ability of the program to interpret biological data in the future. At the Sheardown Lake northwest (NW) and southeast (SE) basins, water quality consistently met AEMP benchmarks and, despite arsenic, chromium, iron, manganese, and/or nickel concentrations above AEMP benchmarks for sediment quality at one or both basins, concentrations of all these metals were comparable to those of background and/or basin-specific baseline indicating no mine-related change in metal concentrations. Because concentrations of metals in Sheardown Lake sediment were similar to those shown at the reference lake and/or baseline, it is recommended that consideration be given to updating the AEMP sediment quality benchmarks for Sheardown Lake to reflect both reference lake and baseline data.

Within the Mary River/Mary Lake system, no mine-related effects to water quality were indicated based on comparison to reference areas and to baseline data. An AEMP benchmark for sediment quality was exceeded for manganese at a single profundal station at Mary Lake in 2020, but based on the isolated occurrence of this exceedance and the fact that average manganese concentrations in sediment at Mary Lake were not elevated compared to concentrations at the reference lake or to those during baseline, no mine-related change in manganese concentrations were indicated at Mary Lake. Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no changes to AEMP monitoring at Mary River/Mary Lake are recommended.

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ACRONYMS AND ABBREVIATIONS

AEMP – Aquatic Effects Monitoring Plan

ANCOVA – Analysis-of-Covariance

ANOVA – Analysis-of-Variance

BCWQG – British Columbia Water Quality Guidelines

CES – Critical Effect Size

cm – Centimetre

CPUE – Catch-Per-Unit-Effort

CREMP – Core Receiving Environment Monitoring Program

CSQG – Canadian Sediment Quality Guidelines

CWQG – Canadian Water Quality Guidelines

dbRDA – Distance-Based Redundancy Analysis

DELT – Deformities, Erosions, Lesions, And Tumors

DOC – Dissolved Organic Carbon

DSS – Digital Sampling System

EEM – Environmental Effects Monitoring

ERP – Early Revenue Phase

FFG – Functional Feeding Group

GPS – Global Positioning System

HPG – Habit Preference Group

HSD – Honestly Significant Difference

KS – Kolmogorov-Smirnov

L – Litre

MDL – Method Detection Limit

MRTF – Mary River Tributary-F

Mt – Million Tonnes

NAD 83 – 1983 North American Datum

NSES – North Shore Environmental Services

NU – Nunavut

NWB – Nunavut Water Board

PEL – Probable Effect Level

PSQG – Ontario Provincial Sediment Quality Guidelines

PWQO – Ontario Provincial Water Quality Objectives

QA/QC – Quality Assurance/Quality Control

SD – Standard Deviation

SEL – Severe Effect Levels

SQG – Sediment Quality Guidelines

TDS – Total Dissolved Solids

TKN – Total Kjeldahl Nitrogen

TMAES – Trinity Minnow Aquatic Environmental Services

TOC – Total Organic Carbon

TSS – Total Suspended Solids

UTM – Universal Transverse Mercator

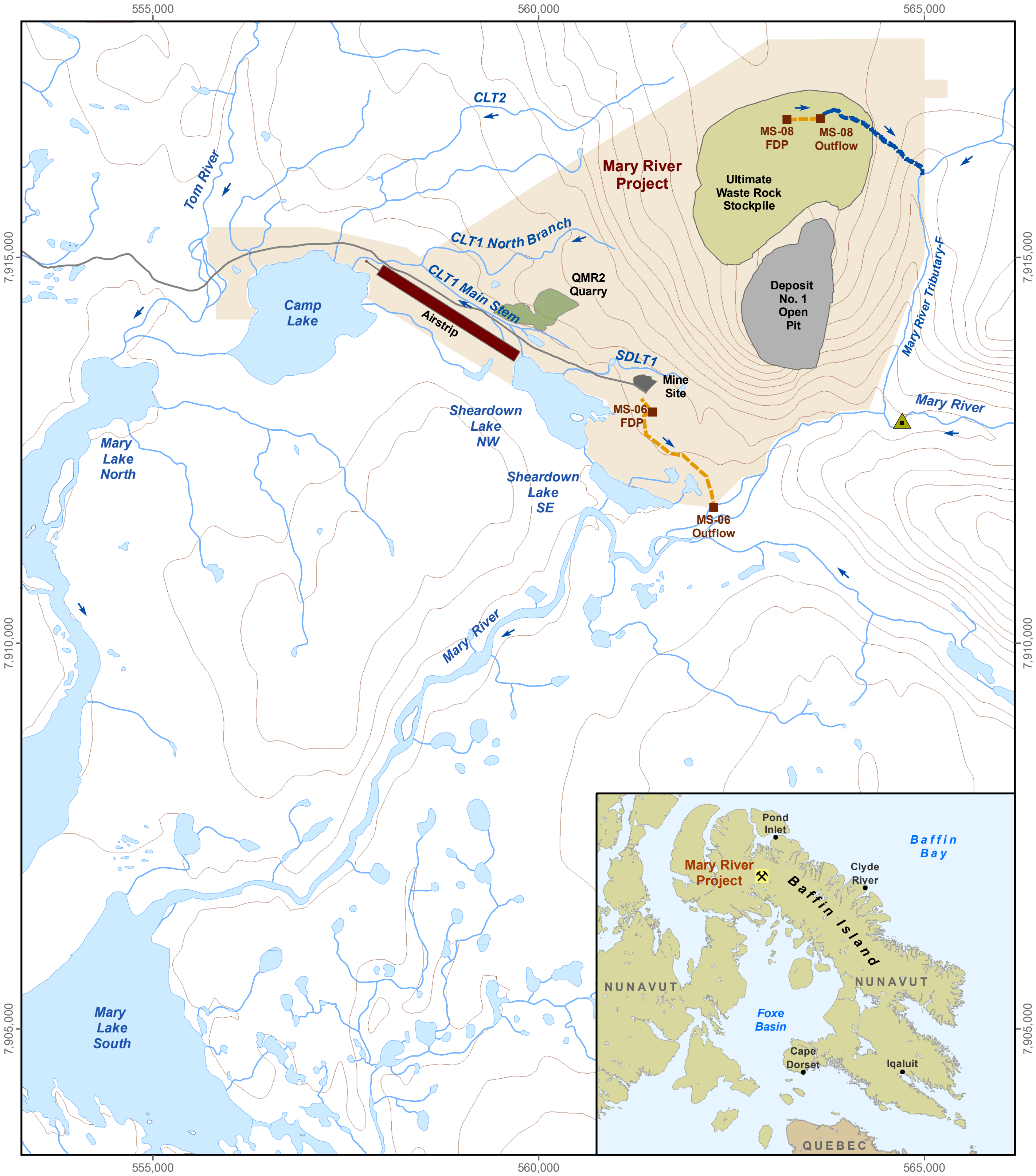
WQG – Water Quality Guidelines

YOY – Young-Of-The-Year

1 INTRODUCTION

The Mary River Project (the 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 (NU) (Figure 1.1). Commercial open pit mining, including pit bench development, ore haulage, and ore stockpiling, as well as the crushing and screening of high-grade iron ore, commenced at the Project Mine Site in 2015. In the current mining phase, referred to as the Early Revenue Phase (ERP), up to 6 million tonnes (Mt) of crushed/screened ore is mined annually at the Project. Ore from the Project Mine Site is transported in haul trucks along the Milne Inlet Tote Road to Milne Port, located approximately 100 km north of the Mine Site, where it is stockpiled. At Milne Port, the ore is loaded onto bulk carrier ships for transport to international markets during the shipping season. No milling or additional ore processing is conducted at the Mine Site, and thus no tailings are produced at the Project. Mine waste management facilities at the Mary River Project thus consist simply of a mine waste rock stockpile and surface runoff collection/containment ponds currently situated near the mine waste rock stockpile and ore stockpile areas. In addition to periodic discharge of treated effluent from these facilities 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 sewage effluent discharge to Mary River, runoff and explosives residue deposition from quarry operations to the Camp Lake catchment, deposition of fugitive dust generated by mine activities, and general Mine Site runoff.

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



LEGEND

- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Discharge Line
- Overland Effluent Channel
- Ultimate Deposit No. 1 Pit Limits
- Ultimate Waste Rock Stockpile Limits
- Mine Site Complex Pad
- Airstrip
- Mary River Project
- QMR2 Quarry

Baffinland Iron Mines Corporation, Mary River Project Location

0 1.25 2.5 5 km

N E S W

Map Projection: UTM Zone 17 W NAD 1983
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Figure 1.1

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(Minnow 2016a, 2017, 2018, 2019). No adverse mine-related effects to phytoplankton, benthic invertebrates, or fish were indicated at any of the Camp Lake, Sheardown Lake, or Mary Lake systems from 2015 to 2019 based on comparisons to reference waterbodies and to available pre-mine baseline data for each lake system (Minnow 2016a, 2017, 2018, 2019).

This report presents the methods and results of the 2020 CREMP, including an evaluation of potential Project-related influences on chemical and biological conditions at mine-exposed waterbodies through the sixth full year of mine operation. As in the five previous years, the 2020 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 2020 CREMP was implemented in accordance with the original study design (Baffinland 2015) with the exception of the continued use of a reference creek benthic invertebrate community study area added to the program in 2016 to provide improved ability for the evaluation of mine-related influences on stream biota (Minnow 2016b, 2017, 2018, 2019, 2020).

2 METHODS

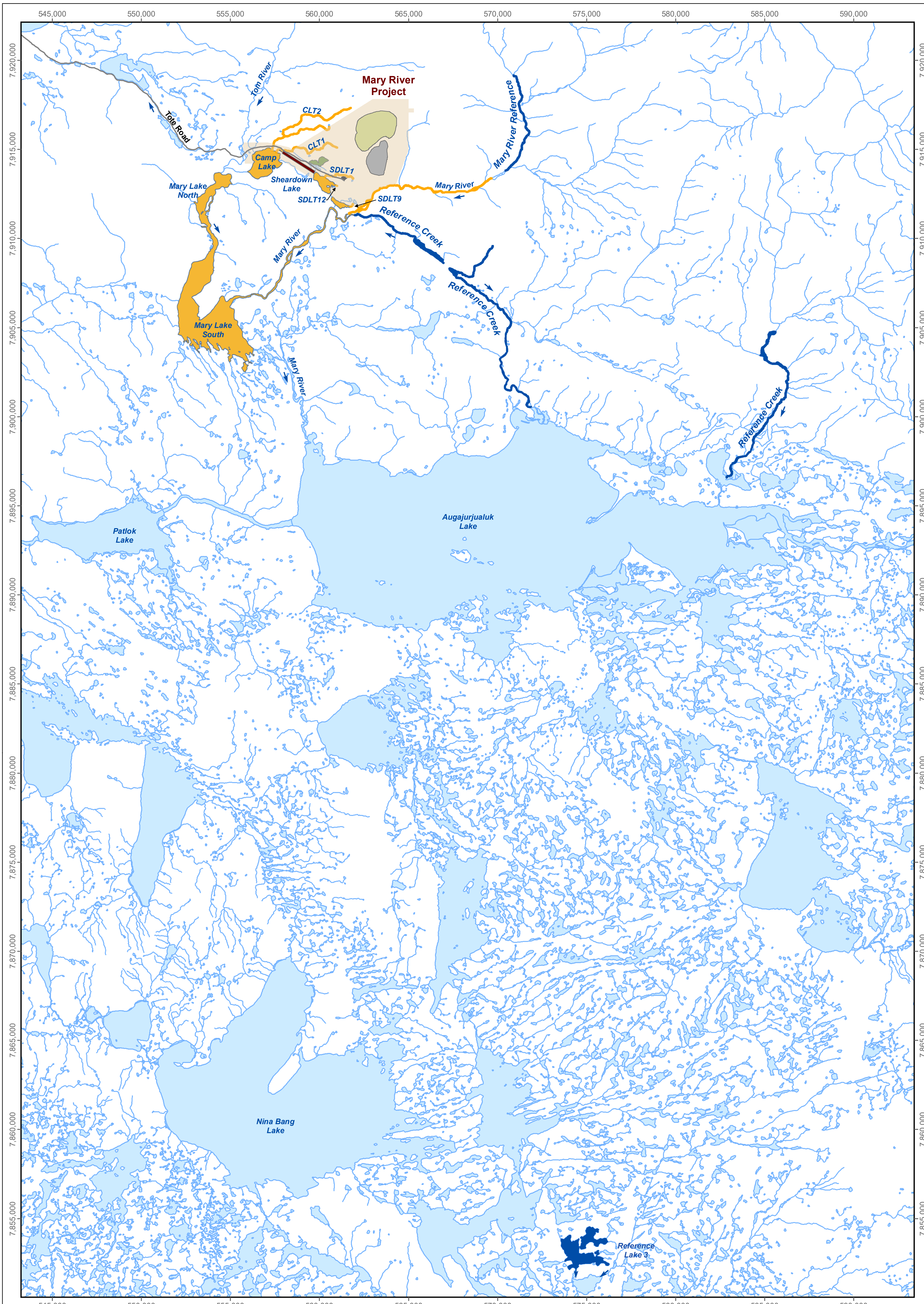
2.1 Overview

The CREMP includes water quality monitoring, sediment quality monitoring, phytoplankton (chlorophyll-a) monitoring, benthic invertebrate community assessment, and fish population assessment (Baffinland 2015). In 2020, water quality and phytoplankton monitoring were conducted by Baffinland environment department personnel over four separate sampling events, including a lake ice-cover event (April 12th to 24th) and open-water season events corresponding to Arctic spring (freshet), summer, and fall (June 2nd to 4th, July 28th to August 3rd, and August 26th to 30th, respectively). Sediment quality, benthic invertebrate community, and fish population sampling was conducted by Trinity Minnow Aquatic Environmental Services (TMAES) personnel with assistance from Baffinland environment department personnel from August 8th to 20th 2020, the seasonal timing of which was consistent with monitoring conducted for previous baseline (2005 to 2013), mine construction (2014), and mine operational (2015 to 2019) studies. Similar to previous CREMP studies, the 2020 study included field sampling and standard laboratory quality assurance/quality control (QA/QC) for the water quality and benthic invertebrate community study components to allow for an assessment of the overall quality of each respective data set (Appendix A).

The 2020 CREMP study areas included the same mine-exposed and reference waterbodies established in the original design documents (Baffinland 2015) 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 NW, and Sheardown Lake 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 2020 study. Reference Lake 3 is located approximately 62 km south of the Mine Site (Figure 2.1), well outside the area of mine influence. Streams used as reference areas in the current and previous CREMP included an unnamed tributary to the Mary River and two unnamed tributaries to Angajurjualuk Lake, all of which are located southeast of the Mine Site (Figure 2.1). Similar to previous CREMP studies, an area of Mary River located well upstream of current mine activity



LEGEND

- Reference Stream/River System
- Mine Exposed Stream/River System
- Reference Lake
- Mine Exposed Lake
- Ultimate Deposit No. 1 Pit Limits
- Ultimate Waste Rock Stockpile Limits
- Mine Site Complex Pad
- Airstrip
- QMR2 Quarry
- Mary River Project

Mary River Project CREMP Study Waterbodies

0 3.25 6.5 13
km

N E S W

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Figure 2.1

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.1 Study Waterbodies.mxd (i.e., GO-09) served as a reference area for the mine-exposed portion of Mary River in the 2020 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 CREMP design (Baffinland 2015). The surface water sampling was conducted at as many as 57 stations during each sampling event (Table 2.1; Figures 2.2 and 2.3) and included collection of *in situ* measurements and water chemistry data. The evaluation of potential mine-related effects on surface waters in the vicinity of the Project was based upon comparisons of those parameters for which AEMP benchmarks have been developed to applicable reference data, to available baseline data, and to guidelines that included site-specific AEMP benchmarks. The AEMP benchmarks were developed to aid in defining effects of the Project on surface water quality, and to guide management response decisions to elevations above the benchmarks within a four-step Assessment Approach and Management Response Framework (Baffinland 2015).

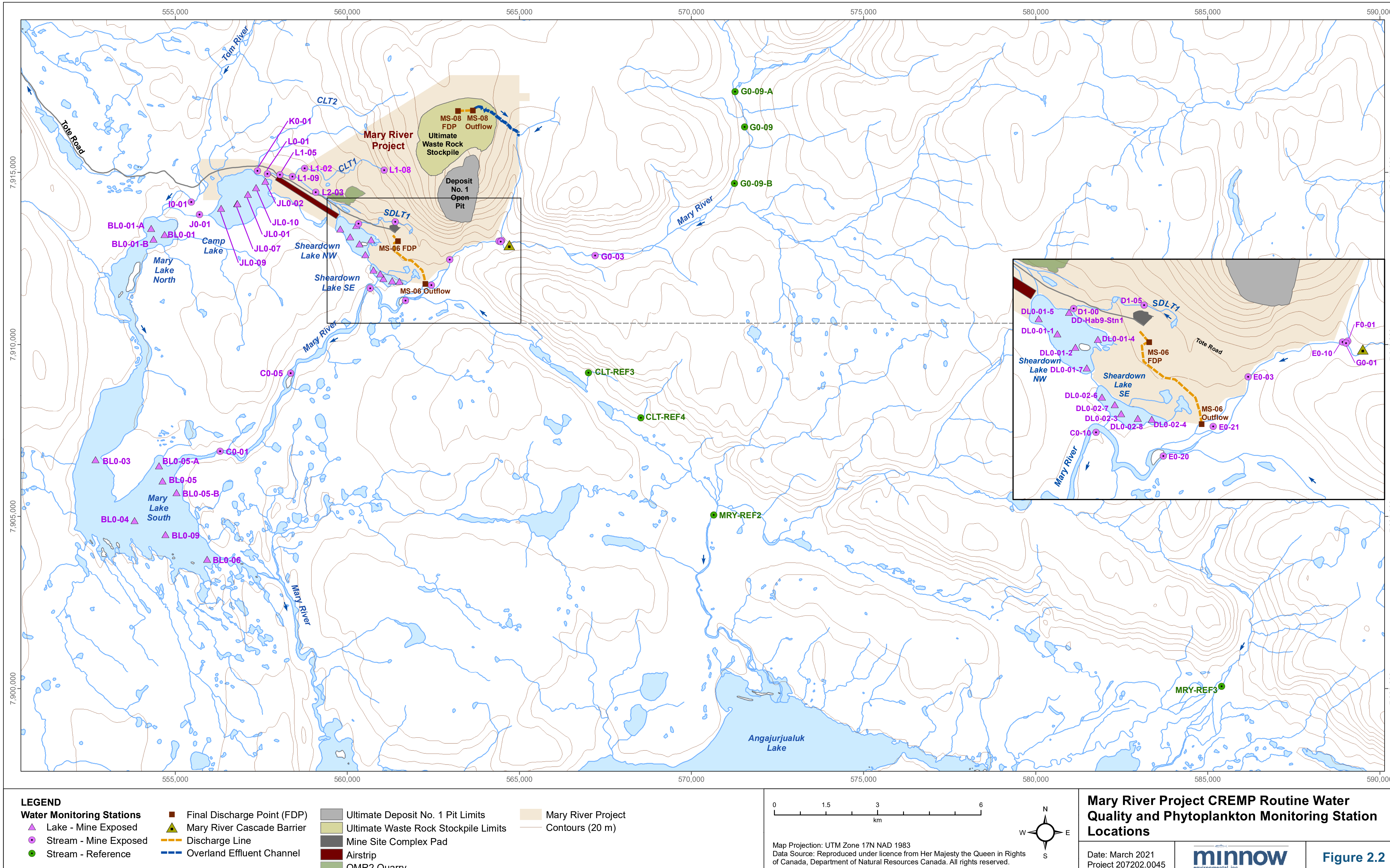
2.2.2 *In situ* Water Quality Measurement Data Collection and Analysis

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 (m) intervals at each lentic (i.e., lake) water quality monitoring station during routine monitoring conducted by Baffinland personnel. These *in situ* measurements were also collected at the surface and bottom (i.e., approximately 30 centimetres [cm] above the water-sediment interface) at all lake benthic invertebrate community (benthic) stations during biological sampling conducted in August by TMAES personnel, except for turbidity measurements. The *in situ* measurements were collected using one of three YSI ProDSS (Digital Sampling System) meters equipped with a 4-Port sensor (YSI Inc., Yellow Springs, OH). Meter readings for pH, specific conductance, and turbidity were checked against standard solutions and calibrated as necessary the morning of the day in which sampling was to be completed, prior to 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-cm (6-inch) diameter ice auger was used to access the water column at lake water quality monitoring stations. Ice shavings were removed from the auger hole prior to the collection of

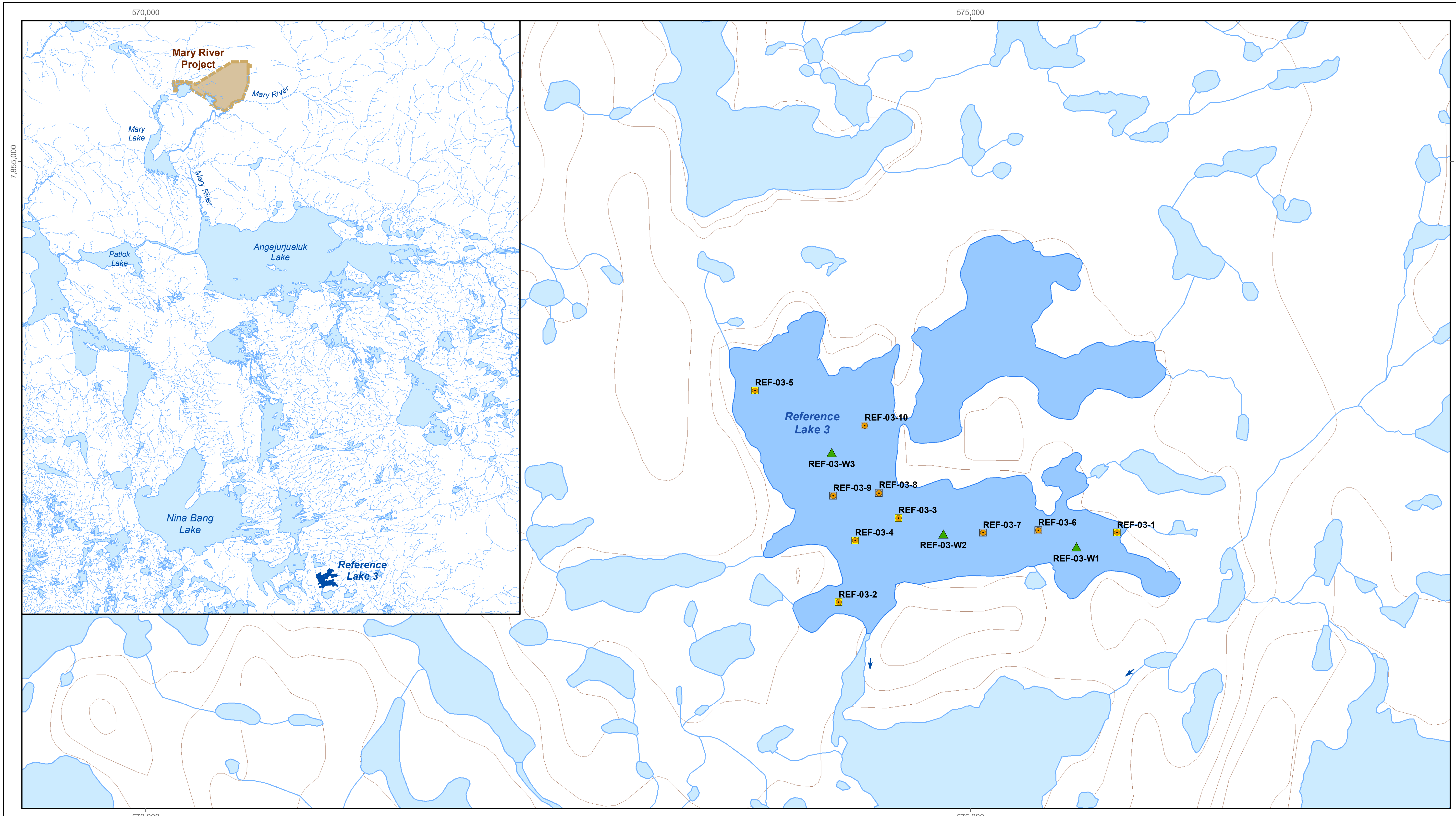
Table 2.1: Mary River Project CREMP Water Quality and Phytoplankton Monitoring Station Coordinates and Annual Sampling Schedule

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



Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.2 Water Quality and Phyto.mxd



LEGEND

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

0 0.35 0.7 1.4
km

N W E S

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Mary River Project CREMP Reference Lake 3 Monitoring Station Locations

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Figure 2.3

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.3 Reference Lake 3.mxd *in situ* measures. To avoid confounding influences associated with snow/ice melt in the auger hole, the *in situ* measurements were collected 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).

In situ water quality data collected at the mine-exposed study streams, rivers, and lakes were compared to respective reference area data, applicable water quality guidelines (WQG[1]; dissolved oxygen concentrations and pH only), and, for pH and conductivity, 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 untransformed or log-transformed data 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-tests and Kruskal-Wallis H-tests were used to conduct pair-wise and multiple-group comparisons, respectively, on rank transformed data. Similarly, in instances in which variances of normal data could not be homogenized by transformation, Student's t-tests assuming unequal variance were used for pair-wise comparisons. In cases in which multiple-group comparisons were conducted, normally distributed data were subject to Tukey's Honestly Significant Difference (HSD) and Tamhane's pair-wise *post hoc* tests for homogenous and non-homogenous data, respectively. All statistical comparisons were conducted using R programming (R Foundation for Statistical Computing, Vienna, Austria).

Vertical profiles of the *in situ* measurements taken from lake stations were plotted and visually assessed to evaluate potential thermal or chemical stratification and the corresponding depths associated with distinct layering. The occurrence of a thermocline was conservatively assessed as a ≥0.5˚C change in temperature per 1 m change in depth[2]. The vertical profile data collected at the mine-exposed study lakes were compared to those of the reference lake for each seasonal monitoring event using profile data averaged for each incremental depth below the water surface at each lake. At each study lake, spatial and seasonal differences in the vertical profile plots were

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[1] Canadian Environmental Quality Guidelines (CCME 1999, 2019) 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 change in depth. Through discussions regarding the CREMP in 2017, regulatory agencies requested that a ≥0.5˚C change in temperature per 1 m change in depth be used to conservatively define a thermally stratified condition.

evaluated to provide a better understanding of natural conditions and/or mine-related influences on within-lake water quality.

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, water chemistry samples were collected from approximately mid-water column by hand directly into pre-labeled sample bottles that were triple rinsed with ambient water. For samples requiring preservation, chemical preservatives were added to the samples before capping the bottles, or for sample bottles that were pre-dosed with chemical preservatives, the bottle was filled using a sample transferred from a separate bottle. 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 litre (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 Kemmerer bottle was transferred directly into sample bottles that had been pre-dosed with required chemical preservatives, where appropriate, except those requiring field filtration. In cases in which filtration of lotic and lentic station water samples was required (e.g., for dissolved metals), filtration was conducted in the field using methods consistent with AEMP standard operating procedures (Baffinland 2015).

Following collection, water chemistry samples were placed into coolers in the field and maintained at cool temperatures prior to shipment to the analytical laboratory. Water chemistry sampling QA/QC included trip blanks, field blanks, and the collection of equipment blanks and field duplicates at an approximate rate of 5% of the total number of samples collected for each CREMP sampling event (Appendix A). 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.

For parameters in which water chemistry AEMP benchmarks have been developed, 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) and/or to site-specific water quality benchmarks developed for the Mary River Project AEMP (Intrinsik 2014); and, iv) to baseline water quality data. For data screening, and to simplify discussion of results, parameter concentration

Table 2.2: Water Quality Guidelines Used for the Mary River Project 2015 to 2020 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	3	CWQG	-
	Nitrite	mg/L	0.06	CWQG	-
	Total Phosphorus	mg/L	0.020 or 0.030	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 to 30 hardness, 218 mg/L at 31 to 75 hardness, 309 mg/L at 76 to 180 hardness, and 429 mg/L at 181 to 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) = $10^{(0.83[\log(\text{hardness}] -2.46)}$. 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 to 180 mg/L, the copper guideline (ug/L) = 0.2 * $e^{(0.8545[\ln(\text{hardness}] - 1.463)}$. 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 to 180 mg/L, the lead guideline (ug/L) = $e^{(1.273[\ln(\text{hardness}] - 4.705)}$. 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 to 180 mg/L, the nickel guideline (ug/L) = $e^{(0.76[\ln(\text{hardness}] + 1.06)}$. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
	Selenium (Se)	mg/L	0.001	CWQG	-
	Silver (Ag)	mg/L	0.00025	CWQG	-
	Thallium (Tl)	mg/L	0.0008	CWQG	-
	Tungsten	mg/L	0.030	PWQO	-
	Uranium (U)	mg/L	0.015	CWQG	-
	Vanadium (V)	mg/L	0.006	PWQO	-
	Zinc (Zn)	mg/L	0.030	CWQG	-

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME1999, 2019) 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 2019), as available.

enrichment factors were calculated as the mine-exposed area mean concentration divided by the respective reference station/area mean concentration. Similarly, for temporal comparisons, the parameter concentration enrichment factor was calculated by dividing the 2020 mean parameter concentration at a mine-exposed station/area by the baseline (2005 to 2013 data) mean concentration. The resulting enrichment factors 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: Enrichment Factor Categories for Water and Sediment Chemistry Comparisons

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

Applicable WQG included the Canadian Water Quality Guidelines (CWQG; CCME 1999, 2019) 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, 2019). Water quality guidelines are abbreviated simply as 'WQG' in this report, although it is recognized that in certain cases the values presented may represent water quality 'objectives'. For WQGs that are hardness dependent, the hardness of the individual sample was used to calculate the WQG for the specific parameter according to established formulae (Table 2.2). Water chemistry data were also compared to site-specific water quality benchmarks developed for the Mary River Project AEMP (Intrinsik 2014). The AEMP water chemistry benchmarks were derived using an evaluation of background (i.e., baseline) water chemistry data together with existing generic WQGs that consider aquatic toxicity thresholds. These benchmarks were developed to inform management decisions under the AEMP assessment approach and management response framework (Baffinland 2015). An elevation in concentration of a parameter 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 (Baffinland 2015). Water chemistry data for key parameters (i.e., parameters with AEMP benchmarks, or with concentrations that were higher at mine-exposed areas compared to reference areas) were plotted to evaluate changes in concentrations between baseline (2005 to 2013 data) and mine operational (2015 to 2020) years.

2.3 Sediment Quality

2.3.1 General Design

Sediment quality monitoring for the CREMP was designed to assess potential mine-related effects to the sediment of lake environments using a gradient-based approach (Baffinland 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 cut-off depth of 12 m, the value of which was used to define lake zonation during baseline characterization studies (KP 2014a, 2015). Sediment quality sampling was also conducted at three 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). Stream sediment sampling in the Camp Lake tributaries, Sheardown Lake tributaries, and Mary River is required every three years as outlined in the original CREMP design (KP 2014a; Baffinland 2015). All stream and river study areas were previously observed to contain limited depositional habitat and a general absence of substantial accumulation of fine sediments (KP 2015; Minnow 2016a,b, 2017, 2018). As a result, sediment sampling for chemical characterization was generally restricted to the shoreline and interstices of large, coarse substrate material (e.g., cobbles, boulders) within the applicable study areas. Similar to water quality, the evaluation of potential mine-related effects on sediments in Project area lakes focused on the use of established AEMP benchmarks to define Project-related effects.

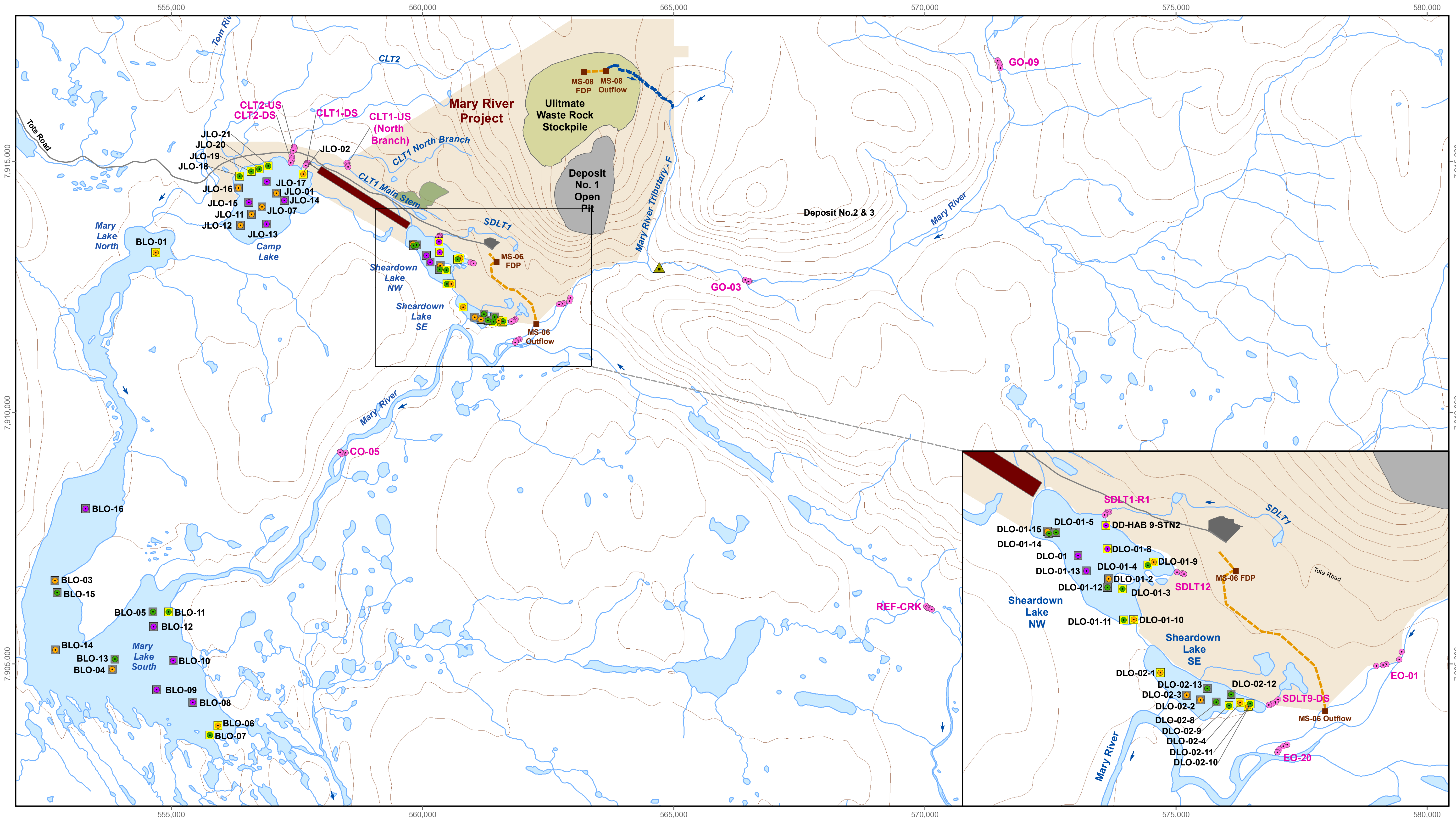
2.3.2 Sample Collection and Laboratory Analysis

Sediment at the 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 top two cm of sediment was manually extruded upwards into a graded core collar, sectioned with a stainless-steel core knife, and placed into a pre-labeled plastic sample bag. Samples from three to four cores treated in this manner were composited to create a single sample at each station. Supporting measurements of total core sample length and depths of visually apparent redox boundaries/horizons, as well as notes regarding sediment texture and colour for each visible horizon, general sediment odour (e.g., hydrogen sulphide), and presence of algae or plants on or in the sediment, were recorded for each core sample. Sediment from stream/river (erosional) habitats was collected for chemical characterization using a stainless-steel spoon. Sediment sampling from erosional habitats focused on locations containing the finest grain sizes available, including channel margins and downstream of large

Table 2.4: Lake Sediment Quality and Benthic Invertebrate Community Monitoring Station Coordinates Used for the Mary River Project 2020 CREMP Study

Waterbody	Station Code	UTM Zone 17W		Sampling Habitat	Sample Type		
		Easting	Northing		Sediment Core[a]	Sediment petite-Ponar[a]	Benthic Invertebrate
Reference Lake 3	REF-03-1	575889	7852752	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	574358	7853400	profundal	✓	-	✓
Camp Lake	JLO-02	557627	7914748	littoral	✓	-	✓
	JLO-01	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	560350	7912927	profundal	✓	-	✓
	DLO-01-12	560339	7912852	profundal	-	✓	✓
	DLO-01-9	560746	7913076	littoral	✓	-	✓
	DLO-01-4	560696	7913049	littoral	-	✓	✓
	DLO-01-3	560471	7912838	littoral	-	✓	✓
	DLO-01-11	560482	7912563	littoral	-	✓	✓
	DLO-01-10	560570	7912566	littoral	✓	-	✓
Sheardown Lake Southeast (SE)	DLO-02-1	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	553820	7904893	profundal	✓	-	✓
	BLO-10	555033	7905065	profundal	✓	-	-
	BLO-09	554707	7904486	profundal	✓	-	-
	BLO-08	555424	7904239	profundal	✓	-	-
	BLO-07	555767	7903583	littoral	-	✓	✓
	BLO-06	555925	7903771	littoral	✓	-	✓

[a] Sediment core samples analyzed for particle size, TOC and total metals. Petite-ponar sediment grab samples analyzed for particle size only.



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
- Ultimate Deposit No. 1 Pit Limits
- Ultimate Waste Rock Stockpile Limits
- Mine Site Complex Pad
- Airstrip
- QMR2 Quarry
- Mary River Project
- Contours (20 m)

0 1 2 4 km

N E S W

Map Projection: UTM Zone 17N NAD 1983
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Mary River Project 2020 CREMP Mine Area Sediment Quality and Benthic Station Locations

Date: March 2021
Project 207202.0045

minnow environmental inc.

Figure 2.4

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.4 Sediment and Benthic.mxd

Table 2.5: Stream and River Sediment and Benthic Invertebrate Community Monitoring Station Identifiers and Coordinates Used for the Mary River Project CREMP 2020 Study

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

boulders within the active channel. One sample, representing a composite of a variable number of spoonfuls, was collected directly into a pre-labelled plastic sample 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 field program, sediment samples were shipped to ALS (Waterloo, ON) for analysis using standard laboratory methods. Physical characterization of samples included percent moisture and particle size analyses, and chemical characterization included analyses of TOC and total metals (including mercury).

2.3.3 Data Analysis

Sediment quality data from the mine-exposed lakes, creeks, and rivers were compared to like-habitat reference area data, applicable sediment quality guidelines/AEMP benchmarks and, when available, baseline sediment quality data. Sediment physical characteristics (i.e., moisture, particle size) and TOC data collected at study area lakes were summarized based on calculation of mean, standard deviation, standard error, minima, and maxima for littoral and profundal habitat. The data from the mine-exposed lakes were compared to the reference lake data using the same statistical tests, data transformations, test assumptions, and statistical software described previously for the statistical evaluation of *in situ* water quality (see Section 2.2.3).

The sediment chemistry data from the mine-exposed lakes were initially assessed to identify potential gradients in metal concentrations with distance from known or suspected sources of mine-related deposits to the lake. For each mine-exposed lake, creek, or river study area, data for each sediment chemistry parameter were separately averaged for each habitat type (e.g., littoral and profundal habitat in lakes) and then compared between like-habitat mine-exposed and reference areas using enrichment factors calculated and compared as described previously for evaluation of water chemistry (Section 2.2.3; Table 2.3). Sediment chemistry data collected at lake environments were compared to applicable Canadian Sediment Quality Guidelines (CSQG; CCME 1999) probable effect levels (PEL) or, for parameters with no CSQG, to Ontario Provincial Sediment Quality Guidelines (PSQG; OMOE 1993) severe effect levels (SEL), collectively referred to as ‘SQG’ throughout this document. The 2020 lake sediment chemistry data analyses included comparisons to Mary River Project AEMP sediment quality benchmarks that were derived using baseline sediment chemistry data for each mine-exposed lake and existing generic CSQG interim or PSQG lowest effect level sediment quality guidelines (Intrinsik 2014, 2015). As indicated previously, the AEMP benchmarks were developed to inform management decisions under the AEMP assessment approach and management response framework (Baffinland 2015). An increase in concentration above the

AEMP benchmark may trigger various actions to better understand and potentially mitigate effects (Baffinland 2015).

Sediment chemistry data for key parameters (i.e., parameters with concentrations that were notably higher at mine-exposed areas compared to the reference area, that have been identified as site-specific parameters of concern in previous studies, and/or those with concentrations above SQG and/or AEMP benchmarks) were plotted to evaluate potential changes in concentrations from 2020 relative to baseline (2005 to 2013) and earlier in the period of mine operation (2015 to 2019). In addition, as described previously, enrichment factors were calculated between the 2020 and baseline data for each individual study lake using the same calculation (and categorization description) as described previously (Section 2.2.3; Table 2.3).

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

The CREMP study design also stipulates the collection of phytoplankton community samples for archiving (Baffinland 2015). If water quality, chlorophyll-a, and/or other biological components indicate potential mine-related effects on primary productivity at a specific mine-exposed waterbody, the phytoplankton community samples may be processed to further investigate the nature of potential mine-related effects on phytoplankton biomass and community structure (e.g., taxonomic composition, richness, density). To date, none of the archived phytoplankton community samples have been processed (2006 to 2019). In 2020, phytoplankton community samples were collected using the same methods described in the CREMP (Baffinland 2015) and, as in the past, these samples were not processed, but were archived for potential future use.

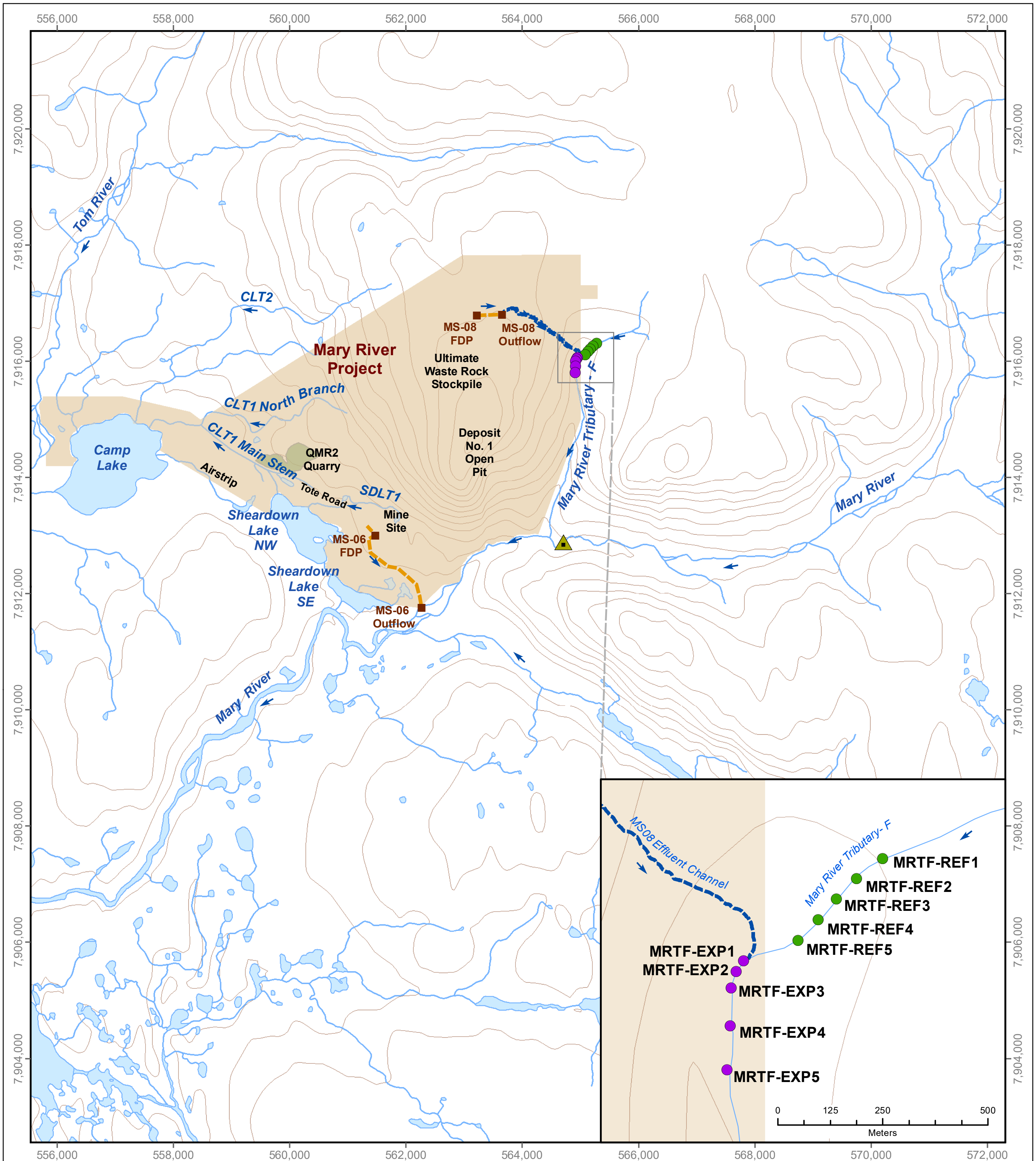
The analysis of aqueous chlorophyll-a concentrations closely mirrored the approach used to evaluate the water quality data. Chlorophyll-a concentrations were compared: i) between respective mine-exposed and reference areas; ii) spatially and seasonally at each mine-exposed waterbody; iii) to AEMP benchmarks; 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 was based on the same statistical tests, data transformations, test assumptions, statistical software, and alpha (p-value) for defining differences as described previously for statistical analysis of *in situ* water quality data (Section 2.2.2). An AEMP benchmark chlorophyll-a concentration of 3.7 µg/L was established for the Mary River Project (Baffinland 2015). The 2020 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 was defined as a chlorophyll-a concentration above the AEMP benchmark, the concentration measured in a representative reference area, and/or the respective waterbody baseline condition.

2.4.2 Benthic Invertebrate Community

2.4.2.1 General Design

The CREMP benthic invertebrate community (benthic) survey design outlines a habitat-based approach for characterizing potential mine-related effects to benthic biota of lotic (stream/river) and lentic (lake) environments (Baffinland 2015). Lotic areas sampled for benthic invertebrates included Camp Lake Tributaries 1 and 2 at historically established areas located upstream and downstream of the Milne Inlet Tote Road, Sheardown Lake Tributaries 1, 9, and 12 near their respective outlets, and Mary River upstream (two areas) and downstream (three areas) of the Mine Site (Table 2.5; Figure 2.4). Benthic samples were also collected at a reference creek located within the same unnamed tributary to Angajurjualuk Lake that is used for reference water quality sampling (Stations CLT-REF4 and MRY-REF2) as part of the 2020 CREMP to augment the original study design (Table 2.5; Figure 2.4). This reference creek, referred to as Unnamed Reference Creek herein, was initially sampled as part of the benthic invertebrate community assessment in the 2016 CREMP (see Minnow 2017). 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 (Figure 2.5). Consistent with the federal EEM program, the CREMP incorporated sampling at five benthic stations at each lotic study area except for 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 studies conducted from 2015 to 2019, the level of replication used for lotic benthic sampling in 2020 was greater than specified under the original CREMP design to provide consistency with EEM standards (Minnow 2016a). To the extent possible, the same station locations used in previous studies were sampled in 2020 to provide continuity among historical baseline and recent studies.

In lentic environments, benthic sampling was conducted at the 40 previously established stations described in the CREMP study design among the four mine-exposed study lakes (i.e., ten stations in each of Camp, Sheardown NW, Sheardown SE and Mary lakes), as well as at the same ten stations established at Reference Lake 3 during the 2015 study (Table 2.4; Figures 2.3 and 2.4). Analysis of benthic data collected at Reference Lake 3 from 2015 to 2019 indicated that, similar to temperate lakes (Ward 1992), depth-related influences on benthic invertebrate community structure (e.g., density and richness) occur naturally in lakes of the study region (Minnow 2016a, 2017, 2018, 2019, 2020). Analysis of benthic data collected from Reference Lake 3 in 2020 provided on-going confirmation of the occurrence of natural depth-related influences on benthic invertebrate community structure in area lakes (Appendix B). Because of the occurrence of natural depth-related differences in benthic invertebrate communities, the benthic stations at each mine-exposed and reference lake were categorized as littoral zone (2-12 m depth) or profundal zone (>12 m depth) stations based on station depth (Table 2.4). To the extent possible, five littoral and five profundal stations were designated for each study lake



LEGEND

Benthic Invertebrate Station

- Effluent-exposed
- Reference
- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Discharge Line
- Overland Effluent Channel
- Mary River Project

Mary River Tributary-F Benthic Station Locations Used for the Mary River Project Second EEM Study, August 2020

0 1.5 3 6 km

Map Projection: UTM Zone 17 WN NAD 1983

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Date: March 2021

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Figure 2.5

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.5 Benthic Station Locations.mxd based on the previously established suite of CREMP lentic benthic stations[3] 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. The sampling of five stations from each zone at each study area ensured adequate statistical power to detect ecologically meaningful differences in benthic metrics of ± two standard deviations (SDs) of a comparable reference area mean using an equal α and β of 0.10 (Environment Canada 2012).

2.4.2.2 Sample Collection and Laboratory Analysis

Two types of equipment and methods were used during the 2020 CREMP benthic survey to sample the different types of habitat encountered as follows:

- at **lotic (stream/river) stations** (i.e., predominantly cobble and/or gravel substrate in flowing waters), benthic samples were collected using a Surber sampler (0.0929 m^2 sampling area) outfitted with 500-µm mesh. At each erosional station, one sample representing a composite of three Surber sampler grabs (i.e., 0.279 m^2 area) was collected to ensure adequate representation of the habitat. A concerted effort was made to ensure that water velocity and substrate characteristics were comparable among respective mine-exposed and reference study area stations to minimize natural influences on community variability. Once all three sub-samples were collected at each station, all material gathered in the Surber sampler net was transferred to a plastic sampling jar which was labelled with both an external and internal station identifier.
- 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). 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 which was labelled with both an external and internal station identifier.

Following collection, benthic samples were preserved to a level of 10% buffered formalin in ambient water. Supporting measurements and information collected at each replicate grab

[3] At Sheardown Lake SE, depths greater than 12 m 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 greater than 12 m and thus were all designated as profundal, with the four remaining stations designated as littoral.

location for lotic stations included sampling depth, water velocity, and description of aquatic vegetation/algae presence. In addition, *in situ* water quality at the bottom of the water column and 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 vegetation/algae, sampling depth, *in situ* water quality 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 American 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 examined under a stereomicroscope at a magnification of at least ten times by a technician. 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. The 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 2020 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, and the Bray-Curtis Index of Dissimilarity. Simpson’s Evenness was calculated using the Krebs method (Smith and Wilson 1996). Additional comparisons were conducted using percent composition of dominant/indicator taxa, functional feeding groups (FFG), and habit preference groups (HPG; percent composition of taxa and groups were 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. The FFG and HPG were assigned based on Pennak (1989), Mandaville (2002), and/or Merritt et al. (2008) descriptions/designations for each taxon.

Statistical comparisons of benthic invertebrate community metrics and community composition endpoints, with the exception of Bray-Curtis Index, 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 Student's t-tests on untransformed, normally distributed data. However, if data were determined to be non-normal, transformations including $\log_{10}$ and $\log_{10}(x+1)$ were applied to the data and evaluated for normality. The transformation that resulted in normal data with lowest skew and kurtosis values was then used for statistical testing using Students t-tests. In instances where normality could not be achieved through data transformation, non-parametric Mann-Whitney U-tests were used for the pair-wise comparisons on rank transformation. Statistical comparisons were conducted using R programming (R Foundation for Statistical Computing, Vienna, Austria). An effect on benthic invertebrate communities was defined as a significant difference between any paired mine-exposed and reference areas at a p-value of 0.10. For each endpoint that differed significantly, a 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 SD, the magnitude of the difference was calculated to reflect the number of reference mean standard deviations (SD_{REF}) using equations provided by Environment Canada (2012). A Critical Effect Size for the benthic invertebrate community study (CES_{BIC}) of ± 2 SD_{REF} was used to define ecologically relevant 'effects', which is analogous to differences beyond those expected to occur naturally between two areas that are uninfluenced by anthropogenic inputs (i.e., between pristine reference areas; see Munkittrick et al. 2009; Environment Canada 2012).

The Bray-Curtis Index was used to evaluate community level differences between study areas, and was computed and assessed statistically using procedures recommended for federal EEM studies (i.e., Borcard and Legendre 2013). Specifically, community level differences between study areas were assessed in a pairwise fashion using ln-transformed abundance data, and with homogeneity of group variance calculated according to the PERMDISP2 procedure provided by Anderson (2006). A Mantel Test and distance-based Redundancy Analysis (dbRDA) was then used to determine potential differences in community structure between study areas using R statistical software (as per Borcard and Legendre 2013).

Temporal comparisons included statistical evaluations among the baseline and 2015 to 2020 data for primary benthic metrics (i.e., density, richness, Simpson's Evenness), dominant invertebrate groups, and FFG using univariate 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 based on a multiple group analysis (see Section 2.2.2). Tukey's HSD *post hoc* tests were used in instances where normal data showed equal variance, and Tamhane's *post hoc* tests were used in instances where normal data showed unequal variance (for the multiple group temporal comparisons). Similar to the 2020 within-year statistical analyses, the magnitude of difference was calculated for endpoints that differed significantly between years in the *post hoc* tests, which was then compared to the benthic survey CES_{BIC} of within SDs of the baseline year mean (abbreviated as ±2 $SD_{BL\text{-}year}$).

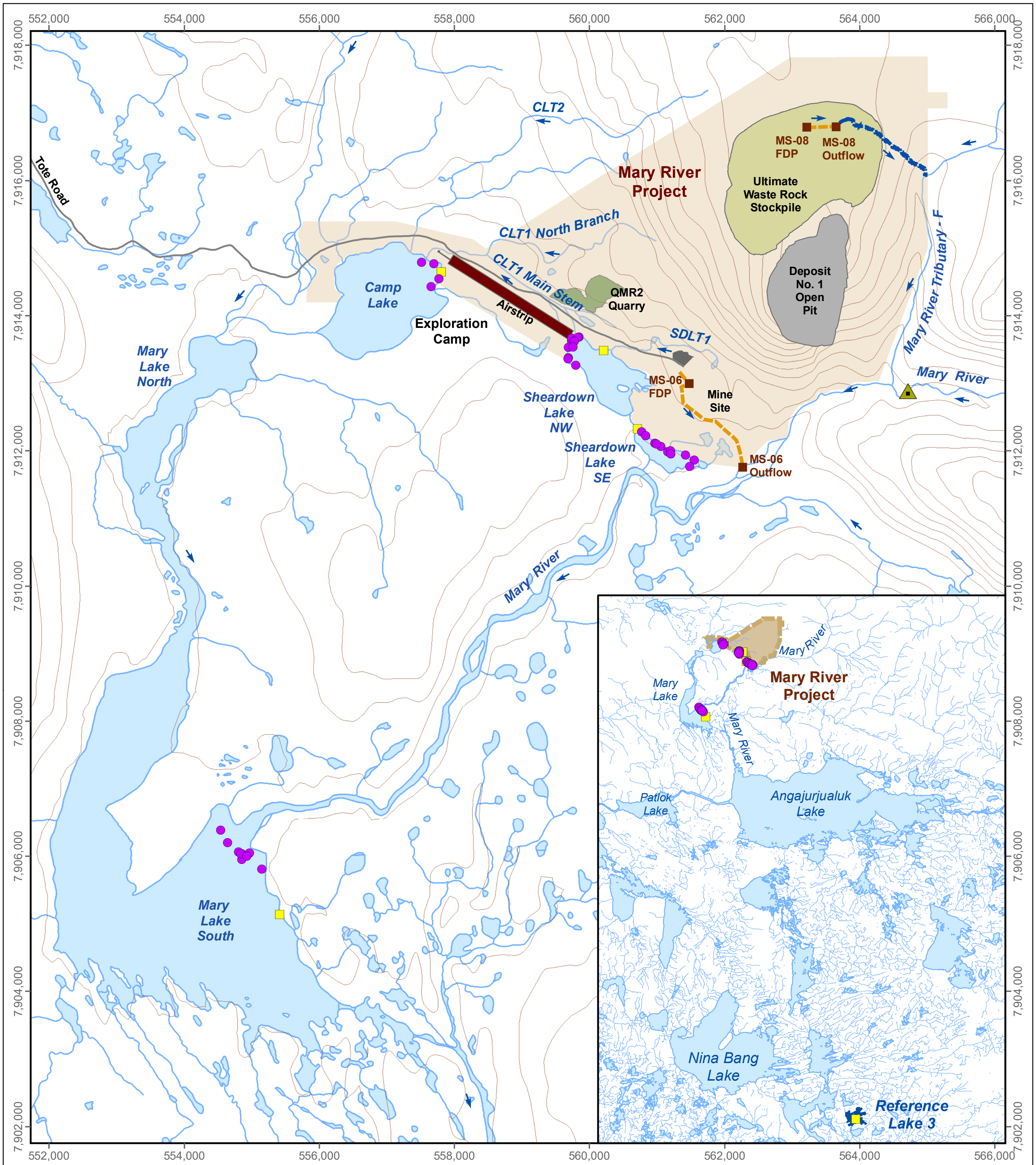
2.4.3 Fish Population

2.4.3.1 General Design

The CREMP fish population survey outlines a non-lethal sampling design to evaluate potential mine-related effects on the fish population (e.g., age structure, condition) at the mine-exposed lakes (Baffinland 2015). The fish population survey targeted arctic charr (*Salvelinus alpinus*) primarily because this species is the most abundant in the mine's regional lakes and sufficient baseline catch and measurement data exist to allow application of a before-after statistical evaluation. Arctic charr are also important 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 2.6). Unlike CREMP studies conducted from 2015 to 2017, enough arctic charr were captured at Reference Lake 3 nearshore and littoral/profundal areas to allow statistical evaluation of potential health effects on arctic charr populations at the mine-exposed lakes. Therefore, the 2020 CREMP fish population survey included separate comparisons of arctic charr collected at nearshore and littoral/profundal habitats between the mine-exposed lakes and reference lake, as well as comparisons of fish from 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 and fish tissue data collected from Mary River near- and far-field mine-exposed areas, as well as an unnamed tributary to Angajurjualuk Lake in 2020 (Figure 2.7; Minnow 2020) have been summarized in this CREMP report to consolidate all available fish population information applicable to the mine receiving environment.

2.4.3.2 Sample Collection

Nearshore areas of 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



LEGEND

- Electrofishing
- Gill Netting
- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Ultimate Deposit No. 1 Pit Limits
- Ultimate Waste Rock Stockpile Limits
- Mine Site Complex Pad
- Airstrip
- Discharge Line
- Overland Effluent Channel
- QMR2 Quarry
- Mary River Project
- Contours (20 m)

Mary River Project 2020 CREMP Fish Survey Sampling Locations

0 1.25 2.5 5 km

N W E S

Map Projection: UTM Zone 17W NAD 1983
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Date: March 2021
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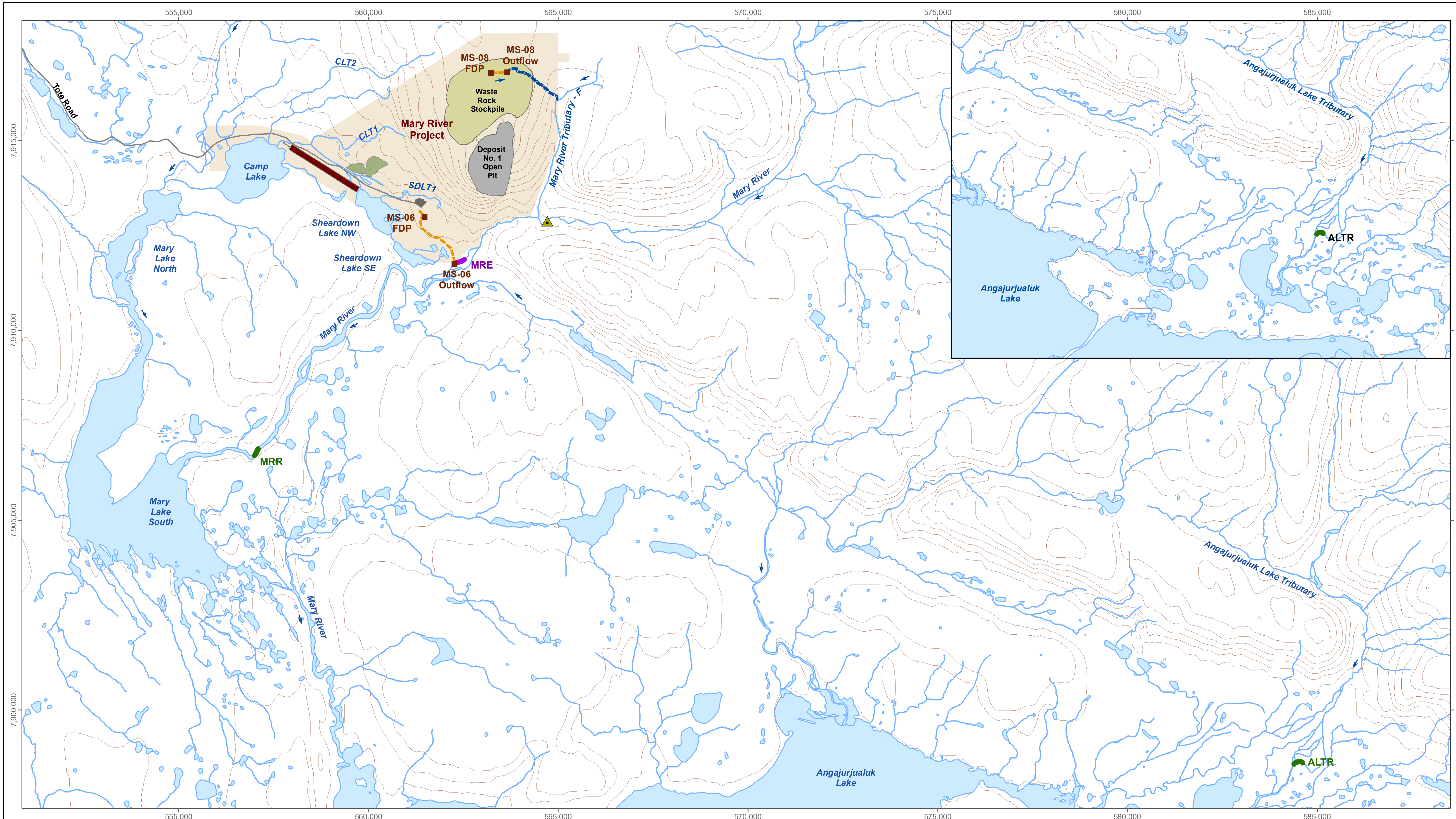
Figure 2.6

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.6 Fish Survey Sampling Locations.mxd



LEGEND

Fish Survey Sampling Location

- Effluent-Exposed
- Reference
- Discharge Line
- Final Discharge Point (FDP)
- QMR2 Quarry
- Ultimate Deposit No. 1 Pit Limits
- Ultimate Waste Rock Stockpile Limits
- Mine Site Complex Pad
- Airstrip
- Mary River Project
- Contours (20 m)
- Tote Road

0 1.5 3 6
km

N W E S

Map Projection: UTM Zone 17 W NAD 1983
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Fish Survey Sampling Locations for the Mary River Project Second EEM, August 2020

Date: March 2021
Project 207202.0045

Figure 2.7

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 207202.0045 - AEMP\4 - GIS\Baffinland 2020 CEMP\20-45 Figure 2.7 Fish Survey Sampling Locations for the Mary River Project Second EEM.mxd backpack electrofisher operator and a single netter, conducted a single fishing pass at up to two shoreline reaches of each study lake and each lotic study area (Figure 2.6). The number of passes conducted at each lake/lotic study area was dependent upon catch success, with an additional pass required in instances where target sample numbers were not cumulatively attained. All fish captured during each pass were retained in buckets containing 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, following which any non-target species were released alive at the area of capture. All captured arctic charr were temporarily retained for processing using methods described in Section 2.4.3.3. For each electrofishing pass, GPS coordinates were recorded at the boundaries of each electrofishing reach.

Littoral/profundal areas of the study lakes were sampled for arctic charr using experimental (gang index) gill nets. Multiple-panel, 2 m high gill nets with total lengths ranging from 61 to 91 m (200’ to 300’) and bar mesh sizes ranging from 38 to 76 mm (1.5” to 3”) were set on the bottom for short durations (range from 0.5 to 2.9 hours per set; average of 1.4 hours) during daylight hours. 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, and GPS coordinates 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 fish was assessed visually for the presence of any deformities, erosions, lesions, and tumors (DELT), in addition to evidence of external and/or internal parasites. All observations were recorded on field sheets, with supporting photographs taken as appropriate. Each fish was then subject to measurement of fork and total length to the nearest millimetre using a standard measuring board. Following length measurements, fish captured by electrofisher 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 by gill net, individuals were weighed using Pesola™ spring scales (Pesola AG, Baar Switzerland) demarcated at intervals of 1 to 2% of the total scale range and providing accuracy of ±0.3% of the fish mass. The Pesola™ spring scale for individual weight measurement of gill-net captured fish was selected so that the fish weight was near the top of the scale’s range to ensure that measurements achieved a resolution near 1%. All live arctic charr that were not selected for the collection of aging structures were released near the location of capture following measurements.

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 age structures. Otoliths were removed from all sacrificed individuals for age determination. Upon removal, otoliths were wrapped separately in wax paper, placed inside envelopes labelled with the fish identification, and then dried for storage. Otoliths were shipped to North Shore Environmental Services (NSES; Thunder Bay, ON) for age determination. At the laboratory, otoliths were prepared for aging using a "crack and burn" method. The prepared 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 otolith, the age and edge condition were 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 qualitatively using electrofishing CPUE and gill netting CPUE to evaluate relative changes in fish catches at mine area lakes between mine baseline and individual years of mine operation from 2015 to 2020.

Arctic charr population health was assessed separately for electrofishing and experimental gill netting data sets. Initial data analysis included the plotting of length frequency distributions so that, together with appropriate age data, young-of-the-year (YOY) individuals could be distinguished from the older juvenile/adult life stages (electrofishing data set), or various size/age classes could be distinguished from one another (gill netting data set). Where sample sizes allowed, the YOY age class was assessed separately from the older juvenile/adult age classes for lake nearshore fish survey endpoints. 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 age class (if possible) for each study area. 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 the mine-exposed lakes and the reference lake or between lotic study areas using data

Table 2.6: Fish Population Survey Endpoints Examined for the Mary River Project CREMP 2020 Study

Response Category	Endpoint	Statistical Procedure[c,d,e]	Critical Effect Size
Survival	Length-frequency distribution[a]	K-S Test	not applicable
Energy Use (size)	Size (fresh body weight)[b]	ANOVA	25%
	Size (fork length)[b]	ANOVA	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-tests were 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).

collected in 2020, and between the combined baseline period and 2020 for individual lakes (i.e., before-after analysis), 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 using data collected in 2020, and between the mine baseline period and 2020. Data were evaluated for normality and homogeneity of variance before applying parametric statistical tests such as ANOVA. In cases where data did not meet the assumptions of ANOVA despite log-transformation, a non-parametric Mann-Whitney U-test was used to test for differences between study areas or study periods. 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 that there was adequate overlap between the 2020 mine-exposed and reference area data, or between the 2020 mine-exposed and baseline data, and that there was a linear relationship between the variable and the covariate. 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 2020 mine-exposed area and the reference data or mine-exposed area baseline data, then the ANCOVA was not performed.

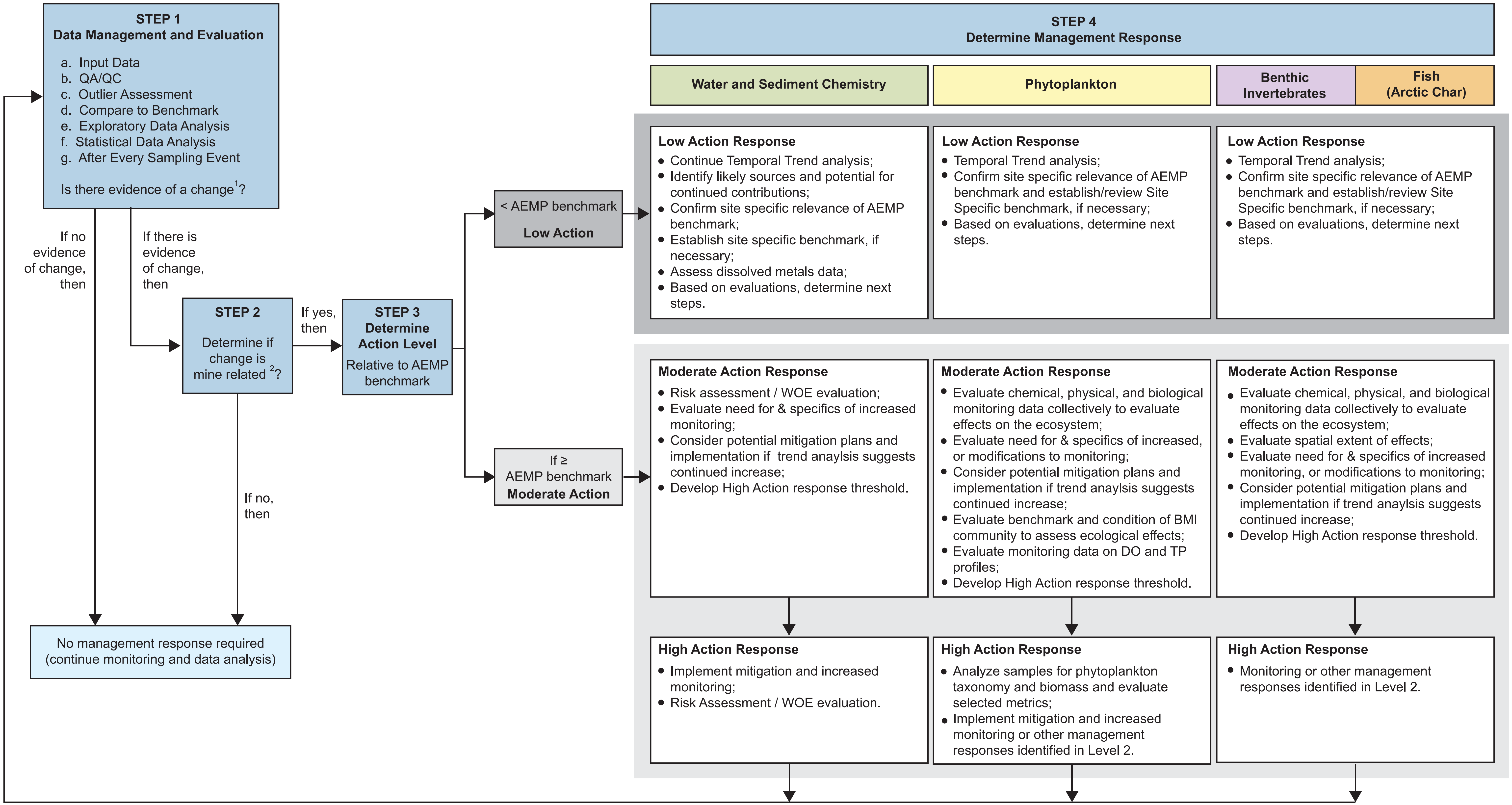
Once it was determined that ANCOVA could be used for statistical analysis, the first step in the ANCOVA was to test whether the slopes of the regression lines between data sets were equal. This was accomplished by including an interaction term (dependent × covariate) in the ANCOVA model and evaluating 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, the options considered to determine if a full ANCOVA could proceed included 1) removal of influential points using Cook's distance and re-assessment of equality of slopes; and/or, 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, a statistically significant interaction effect (slopes are not equal) was noted, and the magnitude of effect was estimated at both the minimum and maximum overlap of covariate variables between areas (Environment Canada 2012). 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 differences, the magnitude of difference between 2020 mine-exposed and reference data or between 2020 and 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. If there was no significant difference 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 for both the complete and reduced data sets. Similar to the CES applied to the benthic invertebrate community survey, a magnitude of difference of ± 10% was applied for condition (CES_C), to define 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. Two main assumptions served as the basis for the power analysis. The first assumption was that the fish caught in each of the mine-exposed and reference areas in 2020, or at mine-exposed areas in 2020 and baseline, 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 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 ANOVA or parallel slope ANCOVA model.

2.5 Effects Assessment

The objective of the Mary River Project 2020 CREMP was to evaluate potential mine-related influences on chemical and biological conditions at aquatic environments located near the mine following the sixth full year of mine operation. The 2020 CREMP incorporated an effects-based approach that included standard EEM techniques to provide rigorous evaluation of potential mine-related effects at key waterbodies that receive mine-related deposits from various mine effluents, surface runoff, and aerial deposition of dust originating from mine operations. Under this approach, water quality and sediment quality data were used to support the interpretation of phytoplankton, benthic invertebrate community, and fish population survey data collected at mine-exposed areas of the Camp Lake, Sheardown Lake, Mary River, and Mary Lake systems. The evaluation of potential mine-related effects within these systems was based upon comparisons of the 2020 data to applicable reference data, to available baseline data, and to guidelines that included site-specific AEMP benchmarks. The latter were developed to guide management response decisions within a four-step Assessment Approach and Management Response Framework as outlined in the Mary River Project AEMP (Figure 2.8; Baffinland 2015). An effects determination was conducted for all key waterbodies located within each of the Camp Lake, Sheardown Lake, Mary River, and Mary Lake systems which included summarization of instances in which the Mary River Project AEMP benchmarks for water quality and sediment quality were exceeded at waterbodies examined under the CREMP. Based on weight-of-evidence that considered incidences in which the AEMP benchmarks were exceeded and corroboration of adverse influences on aquatic biota based on the results of biological monitoring, the effects determination identified potential biological effects at these waterbodies in 2020 and, where appropriate, provided recommendation(s) for future study to assist Baffinland with decisions regarding appropriate management actions.



Notes:
1. Statistical or qualitative change when compared to:
   a) benchmark,
   b) baseline values,
   c) temporal or spatial trends
2. Mine related changes are a result of the mine and associated facilities including but not limited to effects from effluent discharges and dust deposition that are distinguished from natural causes or variation.

Baffinland Mary River Project AEMP Data Assessment Approach and Response Framework

Date: March 2021
Project 207202.0045

minnow environmental inc.

Figure 2.8

3 CAMP LAKE SYSTEM

3.1 Camp Lake Tributary 1 (CLT1)

3.1.1 Water Quality

Camp Lake Tributary 1 (CLT1) dissolved oxygen was consistently near full saturation at the north branch and main stem stations during all spring, summer, and fall monitoring events, and were comparable to, or slightly higher than, concentrations at the reference creeks (Appendix Tables C.1 to C.3; Figure 3.1). In addition, dissolved oxygen concentrations at CLT1 north branch and lower main stem stations were above the WQG lowest acceptable concentration for early life stages of cold-water biota (i.e., 9.5 mg/L) at the time of biological sampling in August 2020 (Figure 3.1; Appendix Table C.12). No consistent spatial patterns in pH were shown with progression downstream through the CLT1 north branch (Stations L1-08 to L1-02) and main stem (Stations L2-03 to L0-01) stations for each of the spring, summer, and fall monitoring events (Appendix Tables C.1 to C.3). Although pH was significantly higher at CLT1 compared to Unnamed Reference Creek during the fall sampling event in August 2020, the pH at all CLT1 stations was consistently within WQG limits in 2020 in all spring, summer, and fall sampling events (Figure 3.1; Appendix Tables C.1 to C.3). No significant difference in pH was indicated between CLT1 north branch and lower main stem in August 2020, indicating no substantial influence of the Milne Inlet Tote Road on in-stream pH (Figure 3.1; Appendix Table C.12).

Specific conductance at CLT1 was generally highest in the upper main stem (Station L2-03) and lowest in the north branch (Stations L1-02 and L1-08), with intermediate values observed at the lower main stem stations reflecting mixing of these two branches and suggesting a potential mine-related source affecting water quality of the CLT1 upper main stem (Appendix Tables C.1 to C.3, and C.12). Specific conductance was consistently higher at CLT1 compared to the reference creek stations over the spring, summer, and fall sampling events in 2020 (Appendix Tables C.1 to C.3, and C.13), and was also significantly higher at CLT1 compared to Unnamed Reference Creek during the August 2020 biological study (Figure 3.1). No significant difference in specific conductance was indicated between the CLT1 north branch and lower main stem in August 2020 (Appendix Table C.12), suggesting that the source of elevated specific conductance was unrelated to the Milne Inlet Tote Road but rather was associated with the upper portion of the CLT1 system.

At the CLT1 north branch stations (L1-08 and L1-02), water chemistry met AEMP benchmarks and WQG in 2020 except for copper concentrations, which were elevated relative to one or both criteria during the spring, summer, and fall sampling events (Table 3.1; Appendix Table C.14). However, like most parameters, copper concentrations at the CLT1 north branch were not

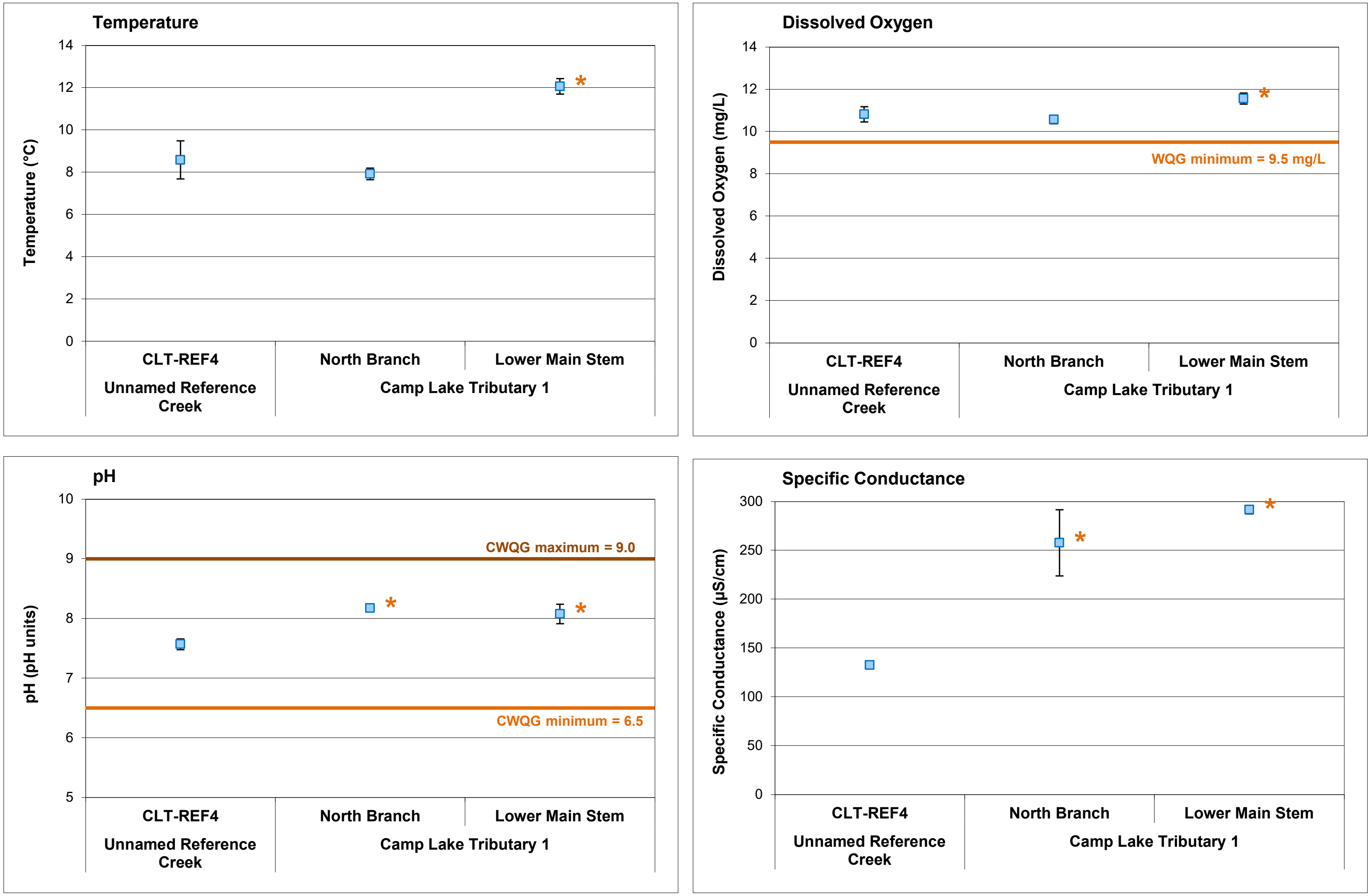


Figure 3.1: Comparison of In Situ Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 1 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2020

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

Table 3.1: Mean Water Chemistry at Camp Lake Tributary 1 (CLT1) Monitoring Stations During Spring, Summer, and Fall, Mary River Project CREMP, 2020

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creeks (n=4)			North Branch			Upper Main Stem (L2-03)			Lower Main Stem		
					Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	55	134	175	111	192	216	258	393	449	155	282	300
	pH (lab)	pH	6.5 - 9.0	-	7.63	8.01	8.05	8.04	8.14	8.14	8.16	8.01	8.11	8.19	8.23	8.23
	Hardness (as $CaCO_3$)	mg/L	-	-	23.6	57.05	82.6	52.2	90.5	112.5	103	147	191	70	128	149
	Total Suspended Solids (TSS)	mg/L	-	-	3.2	2.7	2	2.35	2	2	9.9	<2.0	<2.0	2.2	2.0	2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	85	85	99	86	101	54	158	191	236	110	132	110
	Turbidity	NTU	-	-	1.87	6.62	2.49	1.21	0.19	0.18	15.80	2.38	2.62	2.31	0.79	0.88
	Alkalinity (as $CaCO_3$)	mg/L	-	-	24	61	69	53	94	101	93	138	156	69	122	133
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.01	0.01225	0.01	0.01	0.01	0.016	0.064	0.024	0.036	0.010	0.010	0.013
	Nitrate	mg/L	3	3	0.020	0.062	0.076	0.051	0.074	0.075	1.03	1.520	1.710	0.149	0.280	0.350
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.005	0.005	0.005	0.0089	0.0099	0.0124	0.005	0.005	0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.02	0.15	0.15	0.05	0.15	0.15	1.04	0.32	0.54	0.15	0.16	0.18
	Dissolved Organic Carbon	mg/L	-	-	1.93	3.44	2.31	2.65	3.01	2.51	4.79	6.01	8.00	3.41	4.76	3.79
	Total Organic Carbon	mg/L	-	-	2.23	3.05	2.14	3.50	3.85	2.86	5.87	7.48	6.86	4.68	5.51	4.08
	Total Phosphorus	mg/L	0.030^{α}	-	0.0045	0.0065	0.0039	0.0037	0.0030	0.003	0.0197	0.0093	<0.0030	0.0082	0.0031	0.0030
	Phenols	mg/L	0.004^{α}	-	0.0010	0.0010	0.0021	0.0010	0.0013	0.0010	<0.0010	0.0012	0.0018	0.0010	0.0010	0.0013
Anions	Bromide (Br)	`	-	-	0.10	0.1	0.1	0.1	0.1	0.1	<0.10	<0.10	<0.10	0.1	0.1	0.1
	Chloride (Cl)	mg/L	120	120	1.2	4.07	7.09	1.42	3.01	4.53	16.4	27.70	34.4	4.4	11.2	14.7
	Sulphate (SO_4)	mg/L	218^{β}	218	1.31	5.52	9.25	2.57	5.17	7.25	8.22	16.40	19.60	3.48	7.97	11.13
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0775	***0.3106***	0.0593	0.0194	0.0087	0.0080	***0.2700***	0.0495	0.0979	0.0566	0.0154	0.0124
	Antimony (Sb)	mg/L	0.020^{α}	-	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	<0.00010	<0.00010	<0.00010	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.00010	0.00013	0.00010	0.00010	0.00010	0.00010	0.00018	0.00015	0.00014	0.00010	0.00010	0.00010
	Barium (Ba)	mg/L	-	-	0.00362	0.00948	0.01031	0.00758	0.01215	0.01365	0.01240	0.01510	0.01800	0.00889	0.01470	0.01620
	Beryllium (Be)	mg/L	0.011^{α}	-	0.0005	0.0004	0.0005	0.0005	0.0005	0.0005	<0.00050	<0.00050	<0.00050	0.0005	0.0005	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0003875	0.0005	0.0005	0.0005	0.0005	<0.00050	<0.00050	<0.00050	0.0005	0.0005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.01	0.010	0.010	0.010	0.017	0.025	0.023	0.010	0.012	0.012
	Cadmium (Cd)	mg/L	0.00012	0.00008	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	<0.000010	<0.000010	<0.000010	0.00001	0.00001	0.00001
	Calcium (Ca)	mg/L	-	-	4.9	11.8	16.5	10.2	17.8	21.8	20.6	29.4	35.7	14.1	25.3	28.1
	Chromium (Cr)	mg/L	0.0089	0.0089	0.00050	0.00082	0.00050	0.00050	0.00050	0.00050	0.00059	<0.00050	<0.00050	0.00050	0.00050	0.00050
	Cobalt (Co)	mg/L	0.0009^{α}	0.0040	0.00010	0.00016	0.00010	0.00010	0.00010	0.00010	0.00027	0.00019	0.00024	0.00010	0.00010	0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.00071	0.00115	0.00102	0.00206	***0.00222***	0.00216	0.00164	0.00134	0.00159	0.00185	0.00194	0.00196
	Iron (Fe)	mg/L	0.30	0.326	0.077	0.2425	0.06625	0.030	0.030	0.030	***0.420***	***0.423***	***0.522***	0.080	0.123	0.100
	Lead (Pb)	mg/L	0.001	0.001	0.000107	0.000226	0.000092	0.000053	0.000050	0.000050	0.000987	0.000092	0.000222	0.000102	0.000050	0.000050
	Lithium (Li)	mg/L	-	-	0.0010	0.0011	0.0010	0.0010	0.0013	0.0013	0.0033	0.0042	0.0042	0.0015	0.0026	0.0026
	Magnesium (Mg)	mg/L	-	-	2.86	6.7	9.6	6.4	11.2	13.9	13.1	19.1	24.2	8.8	15.7	18.2
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00136	0.00300	0.00102	0.00078	0.00063	0.00061	0.01970	0.03230	0.03820	0.00338	0.00846	0.00766
	Mercury (Hg)	mg/L	0.000026	-	0.0000050	0.000005	0.000005	0.000005	0.000005	0.000005	<0.0000050	<0.0000050	<0.0000050	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00015	0.00045	0.00057	0.00050	0.00116	0.00127	0.00199	0.00361	0.00385	0.00073	0.00132	0.00148
	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00070	0.00057	0.00055	0.00057	0.00057	0.00711	0.00139	0.00165	0.00082	0.00108	0.00100
	Potassium (K)	mg/L	-	-	0.45	0.93	1.04	1.43	2.30	2.56	2.85	4.05	4.40	1.67	2.64	2.80
	Selenium (Se)	mg/L	0.001	-	0.0010	0.0007625	0.001	0.001	0.001	0.001	<0.0010	<0.0010	<0.0010	0.001	0.001	0.001
	Silicon (Si)	mg/L	-	-	0.62	1.25	0.87	0.67	0.92	0.96	1.23	0.94	1.08	0.80	1.12	1.11
	Silver (Ag)	mg/L	0.00025	0.0001	0.000010	0.00002	0.00001	0.00001	0.00001	0.00001	<0.000010	<0.000010	<0.000010	0.00001	0.00001	0.00001
	Sodium (Na)	mg/L	-	-	0.83	2.76	3.97	0.68	1.52	1.84	9.30	16.00	19.00	2.43	5.82	7.00
	Strontium (Sr)	mg/L	-	-	0.00488	0.01391	0.01850	0.00630	0.01185	0.01465	0.02020	0.03110	0.03850	0.01100	0.02310	0.02707
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00010	0.00008	0.00010	0.00010	0.00010	0.00010	<0.00010	<0.00010	<0.00010	0.00010	0.00010	0.00010
	Tin (Sn)	mg/L	-	-	0.00010	0.0001	0.0001	0.0001	0.0001	0.0001	<0.00010	<0.00010	<0.00010	0.0001	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.011	0.0241	0.0100	0.0100	0.0100	0.0100	<0.010	<0.010	<0.010	0.0100	0.0100	0.0100
	Uranium (U)	mg/L	0.015	-	0.00045	0.00405	0.00737	0.00091	0.00454	0.00781	0.01500	0.02470	0.03320	0.00265	0.00736	0.01022
	Vanadium (V)	mg/L	0.006^{α}	0.006	0.0010	0.0012	0.0010	0.0010	0.0010	0.0010	<0.0010	<0.0010	<0.0010	0.0010	0.0010	0.0010
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.003	0.003	0.003	0.003	0.003	0.0034	<0.0030	<0.0030	0.003	0.003	0.0030

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

[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.

particularly elevated compared to the reference creek stations, with only total molybdenum and potassium concentrations showing slight elevation (i.e., 3- to 5-fold) at the CLT1 north branch but only during the spring sampling event in 2020 (Table 3.1; Appendix Tables C.14 and C.15). Total copper concentrations at the CLT1 north branch were, on average, higher in the spring of 2019 and 2020 compared to baseline, and in fall of all years of commercial mine production from 2015 to 2020 compared to baseline, but concentrations during summer were comparable between years of mine production from 2015 to 2020 and baseline (Appendix Figure C.2). Therefore, only a minor influence on water quality, reflected mainly by a slight elevation in copper concentrations, was indicated at the CLT1 north branch since commercial mine production commenced at the project in 2015.

At the CLT1 upper main stem (Station L2-03), mean concentrations of aluminum and iron were above their respective AEMP benchmarks in the spring sampling event and all spring, summer, and fall sampling events, respectively, in 2020 (Table 3.1). The total concentration of uranium was also elevated above WQG at the upper main stem in both summer and fall 2020 (Table 3.1). In addition to iron and uranium concentrations, chloride, nitrate, and total and dissolved manganese, molybdenum, and sodium concentrations were moderately (i.e., 5-fold to 10-fold) to highly (i.e., ≥10-fold) elevated at the CLT1 upper main stem compared to average concentrations at the reference creeks in two or more seasonal sampling events in 2020 (Table 3.1; Appendix Table C.15). Although total concentrations of aluminum and several other parameters were elevated at the CLT1 upper main stem during the spring sampling event compared to AEMP benchmarks and/or concentrations at the reference creeks, the elevation in these parameters appeared to be related to suspended minerals in the water column as indicated by elevated turbidity at the time of sampling (Appendix Table C.14).[4] In contrast, dissolved concentrations of iron, manganese, molybdenum, sodium, and uranium were moderately to highly elevated at the upper main stem in two or more seasonal sampling events in 2020, and thus elevations of these metals (and chloride and nitrate) appeared to reflect a mine-related source. Of those parameters with AEMP benchmarks, only iron, manganese, nitrate, and sulphate concentrations were elevated at the CLT1 upper main stem in 2020 compared to baseline, of which only the concentration of iron was above site-specific AEMP benchmarks (Appendix Figure C.2). Molybdenum and uranium concentrations, which do not have AEMP

[4] Total aluminum and iron concentrations were also above AEMP benchmarks and/or WQG at the MRY-REF3 lotic reference station in 2020, where higher turbidity typically occurs compared to the other reference creek stations (Appendix Table B.2). This suggested natural elevation of these metals in regional watercourses as a result, in part, of naturally greater amount of mineral/particulate matter suspended in the water at this station. Evaluation of dissolved concentrations of aluminum showed similar average concentrations between CLT1 stations and the reference creek stations, corroborating that total aluminum concentrations were associated with turbidity and suggesting that mine operations were not a key source of aluminum to the system (Appendix Tables C.4, C.16, and C.17).

benchmarks, also showed elevated concentrations at the CLT1 upper main stem in 2020 compared to baseline (Appendix Figure C.2).

At the CLT1 lower main stem (Stations L1-09, L1-05, and L0-01), water chemistry met all AEMP benchmarks and WQG over the duration of spring, summer, and fall sampling events in 2020, including for copper and iron concentrations which were elevated above the AEMP benchmarks at the north branch and upper main stem, respectively (Table 3.1; Appendix Table C.14). Nevertheless, manganese, nitrate, and uranium concentrations were moderately elevated at the CLT1 lower main stem compared to the reference creeks in one of the three seasonal sampling events in 2020 (Appendix Table C.15). Of those parameters with AEMP benchmarks, only copper, nitrate, and sulphate showed elevated concentrations in 2020 compared to baseline at the lower main stem (Appendix Figure C.2), of which the elevation in copper likely reflected a north branch source. Similar to the upper main stem, molybdenum and uranium concentrations, which do not have applicable AEMP benchmarks, were also elevated at the CLT1 lower main stem in 2020 compared to baseline (Appendix Figure C.2).

Higher iron, manganese, molybdenum, nitrate, and uranium concentrations at the CLT1 main stem and/or lower stem stations following the initiation of commercial mine operation potentially reflected blasting/excavating activity (including associated dust generation) at the Mine Site QMR2 Quarry[5], as well as fugitive dust generation from increased truck usage on the Milne Inlet Tote Road, compared to the baseline period. The relatively high concentrations of nitrate over years of mine operation at CLT1 were consistent with the deposition of explosives residue from blasting at the QMR2 Quarry as the source of these compounds. Concentrations of total molybdenum and uranium were highest at CLT1 main stem stations in 2019 and 2020 compared to all previous years of mine operation, but concentrations of these parameters generally remained well below WQG suggesting low potential for biological effects (Appendix Figure C.2). Overall, mine-related influences on water quality of the CLT1 were primarily reflected as elevated conductivity and concentrations of copper at the north branch, and elevated concentrations of nitrate, chloride, sulphate, and total metals including manganese, molybdenum, sodium, and uranium, at the upper main stem station. Despite elevation of parameter concentrations at the CLT1 north branch and upper main stem, none were elevated above applicable AEMP benchmarks or WQG at the lower main stem prior to discharge to Camp Lake.

3.1.2 Sediment Quality

In-stream substrate at CLT1 upstream (north branch; CLT1-US) and downstream (lower main stem; CLT1-DS) study areas was composed mainly of cobble material (i.e., substrate with

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

diameters of 6 to 25 cm), with sand constituting only a trace amount (i.e., <1%) of the material observed at the sediment surface (Minnow 2018). Sediment sampled for chemistry analysis at both CLT1 study areas was predominantly composed of medium-sized coarse sand (Appendix Table D.7). The TOC content of the sampled sediment was generally low (i.e., <2%) at both CLT1 study areas, but was elevated (i.e., 7.5 and 13.5-fold higher at CLT1-US and CLT1-DS, respectively) compared to average lotic reference conditions suggesting a slightly more depositional environment at CLT1 (Table 3.2; Appendix Tables D.8 to D.10).

Metal concentrations in sediment from CLT1-US and CLT1-DS were generally elevated compared to those measured at lotic reference areas (Appendix Table D.10). This was particularly the case for aluminum, barium, copper, magnesium, manganese, molybdenum, nickel, potassium, and zinc, for which mean concentrations were five-fold or greater at one or both of CLT1-US and CLT1-DS compared to the average at reference areas (Table 3.2; Appendix Table D.10). Of these metals, only manganese, molybdenum, and potassium, together with uranium and zirconium, occurred at concentrations 1.5 times or greater at the downstream area compared to the upstream area (Table 3.2), potentially reflecting an influence of the Milne Inlet Tote Road or other mine sources on metal concentrations in sediment at CLT1-DS. Despite higher metal concentrations in sediment at CLT1-US and CLT1-DS compared to average lotic reference conditions, concentrations of all metals were well below applicable SQG at both CLT1 study areas (Table 3.2; Appendix Tables D.8 and D.9).

3.1.3 Phytoplankton

Chlorophyll-a concentrations at the upper-most CLT1 north branch station (Station L1-08) were lower than the mean concentration among reference creeks for spring, summer, and fall sampling events in 2020 (Figure 3.2). However, chlorophyll-a concentrations farther downstream within the north branch (i.e., Station L1-02) were generally comparable to chlorophyll-a concentrations at the reference creeks for all seasonal sampling events, suggesting no marked differences in phytoplankton abundance between the CLT1 north branch and the reference creek stations (Figure 3.2).

Within the CLT1 main stem, chlorophyll-a concentrations were generally highest at upstream-most Station L2-03 during spring, summer, and fall sampling events in 2020 (Figure 3.2). On average, chlorophyll-a concentrations were higher, but did not differ significantly, between the CLT1 main stem and reference creek stations during the spring and fall sampling events, but were significantly lower at the CLT1 main stem during the summer sampling event (Appendix Table E.2). Relatively high chlorophyll-a concentrations at Station L2-03 and in the CLT1 lower main stem during spring and summer sampling events potentially reflected higher nutrient (e.g., nitrate) concentrations compared to the reference creeks (Appendix Tables

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 2020

Parameter	Units	SQG[a]	Lotic Reference Stations		Camp Lake Tributary 1	
			Unnamed Reference Creek (REFCRK; n = 3)	Mary River Reference (GO-09; n = 3)	Upstream CLT1-US (n = 3)	Downstream CLT1-DS (n = 3)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC	%	10[α]	0.12 ± 0.035	0.11 ± 0.012	0.88 ± 0.20	1.57 ± 1.72
Aluminum (Al)	mg/kg	-	584 ± 185	2,757 ± 1,141	7,390 ± 537	5,847 ± 1,242
Antimony (Sb)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Arsenic (As)	mg/kg	17	0.22 ± 0.11	0.38 ± 0.10	0.69 ± 0.12	0.67 ± 0.17
Barium (Ba)	mg/kg	-	2.72 ± 0.722	12.6 ± 5.05	17.8 ± 3.22	26.2 ± 5.97
Beryllium (Be)	mg/kg	-	<0.10 ± 0	0.14 ± 0.040	0.28 ± 0.032	0.23 ± 0.045
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0
Boron (B)	mg/kg	-	<5.0 ± 0	5.4 ± 0.75	9.3 ± 0.85	5.5 ± 0.81
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	<0.020 ± 0	0.047 ± 0.0078	0.049 ± 0.0042
Calcium (Ca)	mg/kg	-	494 ± 249	2,750 ± 894	2,660 ± 230	3,703 ± 948
Chromium (Cr)	mg/kg	90	7.79 ± 5.39	13.6 ± 4.34	26.7 ± 2.40	21.2 ± 6.39
Cobalt (Co)	mg/kg	-	0.953 ± 0.558	2.40 ± 0.758	6.40 ± 0.864	5.27 ± 1.42
Copper (Cu)	mg/kg	110[α]	1.21 ± 0.899	4.45 ± 2.50	23.9 ± 11.1	11.6 ± 4.95
Iron (Fe)	mg/kg	40,000[α]	12,493 ± 9,700	11,063 ± 2,423	22,833 ± 5,773	26,533 ± 9,287
Lead (Pb)	mg/kg	91	1.49 ± 0.546	3.07 ± 0.857	4.13 ± 0.654	5.73 ± 2.39
Lithium (Li)	mg/kg	-	<2.0 ± 0	5.0 ± 2.3	11.2 ± 0.462	7.8 ± 1.4
Magnesium (Mg)	mg/kg	-	444 ± 165	2,810 ± 1,212	8,297 ± 274	6,910 ± 1,790
Manganese (Mn)	mg/kg	1,100[α,β]	27.4 ± 14.6	76 ± 29.4	167 ± 34	247 ± 73
Mercury (Hg)	mg/kg	0.486	<0.0050 ± 0	<0.0050 ± 0	0.0051 ± 0.00017	0.0059 ± 0.0015
Molybdenum (Mo)	mg/kg	-	<0.10 ± 0	0.11 ± 0.023	0.29 ± 0.080	1.07 ± 0.436
Nickel (Ni)	mg/kg	75[α,β]	1.76 ± 0.920	6.11 ± 1.99	18.9 ± 1.55	23.4 ± 9.28
Phosphorus (P)	mg/kg	2,000[α]	167 ± 98	350 ± 118	261 ± 27.8	235 ± 84.7
Potassium (K)	mg/kg	-	133 ± 42	750 ± 320	1,260 ± 120	2,127 ± 738
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	68 ± 21	78 ± 6.51	75 ± 22
Strontium (Sr)	mg/kg	-	2.00 ± 0.544	4.72 ± 1.01	2.85 ± 0.195	4.0 ± 1.13
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	<0.050 ± 0	0.068 ± 0.023	0.097 ± 0.021	0.122 ± 0.0321
Tin (Sn)	mg/kg	-	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0
Titanium (Ti)	mg/kg	-	83 ± 47	353 ± 123	442 ± 58	400 ± 60
Uranium (U)	mg/kg	-	0.479 ± 0.247	0.922 ± 0.298	0.898 ± 0.0956	1.52 ± 0.510
Vanadium (V)	mg/kg	-	16.8 ± 12.8	19.5 ± 5.06	24.3 ± 2.54	15.8 ± 4.95
Zinc (Zn)	mg/kg	315	3.0 ± 1.2	10.3 ± 4.31	18.9 ± 2.22	26.3 ± 6.40
Zirconium (Zr)	mg/kg	-	2.1 ± 0.91	5.8 ± 2.0	2.7 ± 0.10	4.7 ± 1.3

Indicates parameter concentration above SQG.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV 2020).

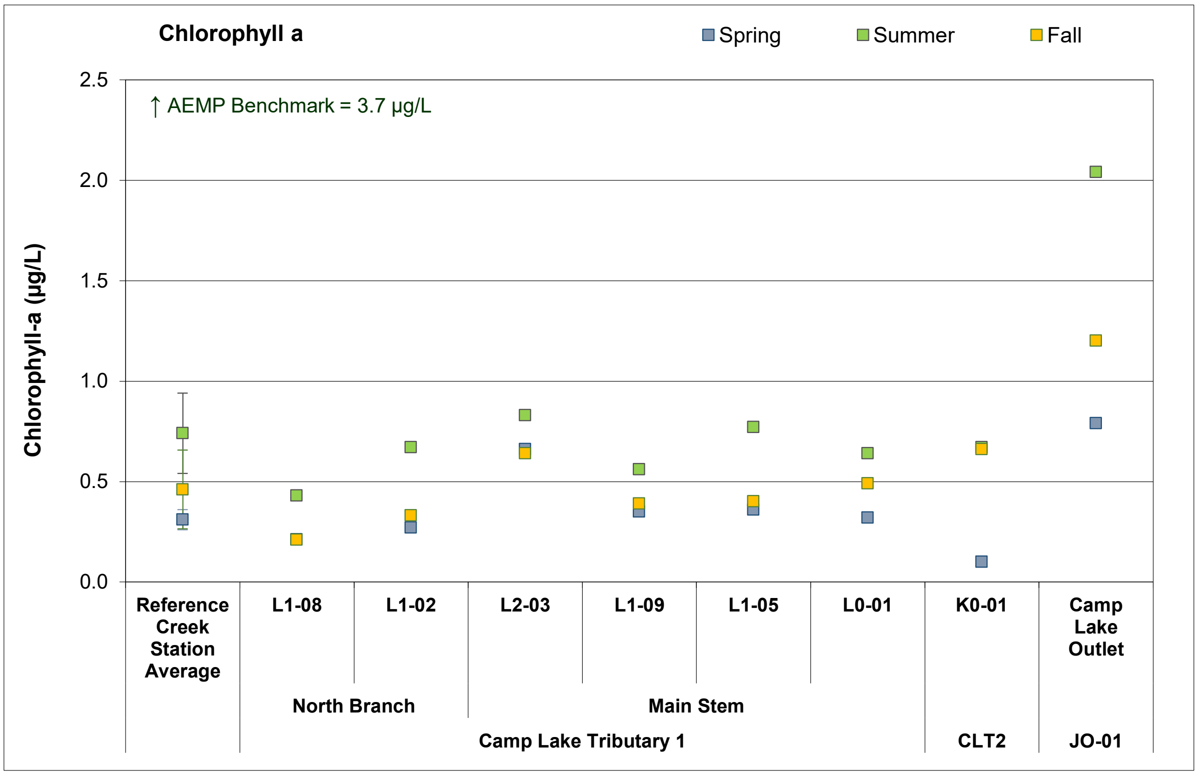


Figure 3.2: Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT1) and Tributary 2 (CLT2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2020

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 2020 (Figure 3.2). Similar to the reference creek stations, chlorophyll-a concentrations at all CLT1 stations in 2020 suggested low (i.e., oligotrophic) phytoplankton productivity based on Dodds et al. (1998) trophic status classification for stream environments (i.e., chlorophyll-a < 10 µg/L). This trophic status classification was also consistent with an 'ultra-oligotrophic' to 'oligotrophic' WQG categorization (CCME 2020) for CLT1 based on aqueous total phosphorus concentrations typically less than 10 µg/L at each CLT1 north branch and main stem station during all spring, summer, and fall sampling events (Appendix Table C.14).

Chlorophyll-a concentrations at the CLT1 north branch in fall 2020 were similar to, or lower than, those observed in the fall during the baseline period (i.e., 2005 to 2013; Figure 3.3). At the CLT1 main stem, chlorophyll-a concentrations were higher in mine operational years from 2015 to 2020 than during the mine baseline period except for at the CLT1 mouth (Station L0-01; Figure 3.3). However, no pattern of increasing chlorophyll-a concentrations was indicated among the years of mine operation at any of the CLT1 north branch or lower main stem stations, and concentrations were continuously lower than the AEMP benchmark of 3.7 µg/L from 2015 to 2020 (Figure 3.3). Overall, the spatial and temporal analyses of chlorophyll-a concentrations suggested that the mine operation may have contributed to slightly higher phytoplankton abundance at CLT1 main stem stations during spring and fall sampling events, but not at the north branch or at the mouth of the main stem compared to reference conditions. As indicated above, higher phytoplankton abundance within the CLT1 main stem was consistent with the occurrence of higher aqueous nutrient concentrations (e.g., nitrate) compared to water quality at the reference creeks. This suggested that slightly greater phytoplankton abundance at the CLT1 main stem was the result of current mine operations and specifically, the introduction of nutrients to the system because of active quarrying at the QMR2 pit. Despite slightly greater phytoplankton abundance at the CLT1 main stem stations than at the reference creeks in spring and fall of 2020, the CLT1 north branch and main stem have remained 'oligotrophic' since the commencement of commercial mine operation in 2015.

3.1.4 Benthic Invertebrate Community

3.1.4.1 Upstream North Branch (CLT1 US)

Benthic invertebrate density at the CLT1 upstream (north branch) was significantly greater than at the reference creek, and no significant differences in richness and Simpson's Evenness were indicated between the CLT1 north branch and Unnamed Reference Creek study areas (Table 3.3; Appendix Figure F.1). Differences in benthic invertebrate community assemblage between the

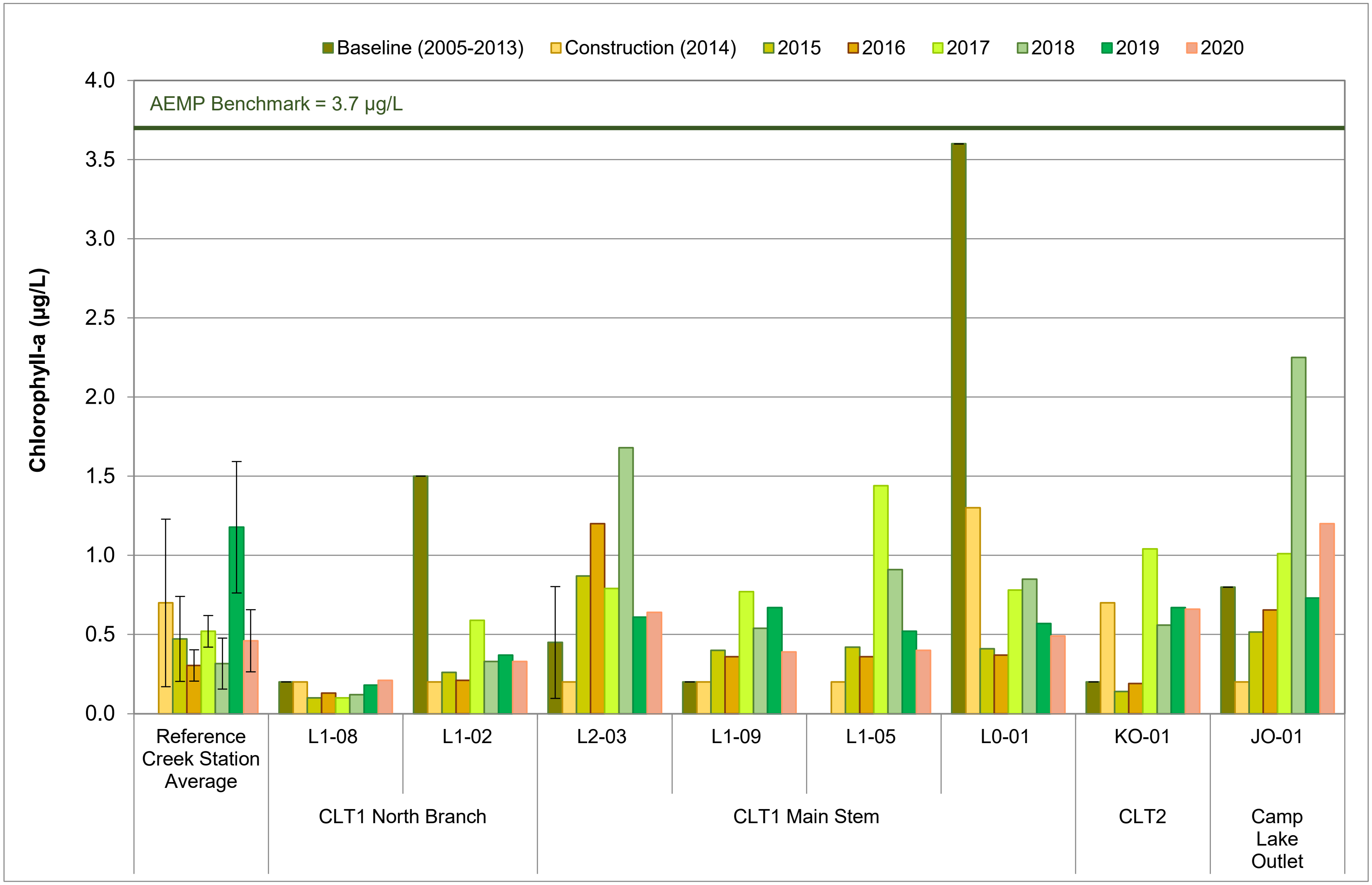


Figure 3.3: Temporal Comparison of Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2020) Periods during Fall

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

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 2020

Metric	Overall 3-Area Comparison				Pair-wise, *post hoc* comparisons				
	Statistical Test[a]	Data Transform-ation	Significant Difference Among Areas?	P-value	Study Area	Mean	Standard Deviation (SD)	Magnitude of Difference (Ref SD)	Pairwise Comparison
Density (No. per m^2)	ANOVA	none	YES	0.009	Reference Creek	713	296	-	a
					CLT1 Upstream	1,635	711	3.1	b
					CLT1 Downstream	626	261	-0.3	a
Richness (No. of Taxa)	ANOVA	none	NO	0.358	Reference Creek	16.0	4.4	-	a
					CLT1 Upstream	19.2	2.5	0.7	a
					CLT1 Downstream	17.4	3.1	0.3	a
Simpson's Evenness	ANOVA	none	NO	0.216	Reference Creek	0.840	0.043	-	a
					CLT1 Upstream	0.884	0.032	1.0	a
					CLT1 Downstream	0.828	0.067	-0.3	a
Nemata (% of community)	ANOVA	log10(x+1)	NO	0.235	Reference Creek	0.7	1.3	-	a
					CLT1 Upstream	1.3	1.1	0.4	a
					CLT1 Downstream	2.1	1.0	1.0	a
Oligochaeta (% of community)	ANOVA	log10	YES	0.004	Reference Creek	1.9	1.5	-	a
					CLT1 Upstream	4.4	1.3	1.6	b
					CLT1 Downstream	11.5	8.4	6.5	b
Hydracarina (% of community)	ANOVA	log10(x+1)	NO	0.286	Reference Creek	4.5	3.7	-	a
					CLT1 Upstream	3.0	2.5	-0.4	a
					CLT1 Downstream	1.6	1.6	-0.8	a
Ostracoda (% of community)	K-W	rank	YES	0.005	Reference Creek	30.6	11.7	-	a
					CLT1 Upstream	0.5	0.6	-2.6	b
					CLT1 Downstream	0.1	0.2	-2.6	b
Chironomidae (% of community)	ANOVA	log10	YES	<0.001	Reference Creek	48.3	12.9	-	a
					CLT1 Upstream	81.7	7.9	2.6	b
					CLT1 Downstream	75.8	7.1	2.1	b
Metal Sensitive Chironomids (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	0.8	1.2	-	a
					CLT1 Upstream	13.8	4.7	10.5	b
					CLT1 Downstream	3.6	1.6	2.2	a
Simuliidae (% of community)	K-W	rank	YES	0.074	Reference Creek	9.8	8.6	-	a
					CLT1 Upstream	0.3	0.3	-1.1	b
					CLT1 Downstream	0.2	0.2	-1.1	b
Tipulidae (% of community)	K-W	rank	YES	0.048	Reference Creek	1.5	2.3	-	a
					CLT1 Upstream	7.4	6.2	2.6	b
					CLT1 Downstream	6.2	2.0	2.0	b
Collector-Gatherer FFG (% of community)	ANOVA	none	YES	<0.001	Reference Creek	80.7	8.8	-	a
					CLT1 Upstream	58.5	8.9	-2.5	b
					CLT1 Downstream	81.6	4.5	0.1	a
Filterer FFG (% of community)	ANOVA	log10(x+1)	YES	0.099	Reference Creek	9.9	8.9	-	a
					CLT1 Upstream	4.4	5.3	-0.6	ab
					CLT1 Downstream	0.9	1.2	-1.0	b
Shredder FFG (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	2.8	2.7	-	a
					CLT1 Upstream	32.8	6.8	11.3	b
					CLT1 Downstream	14.1	4.2	4.3	c
Clinger HPG (% of community)	ANOVA	log10	YES	0.001	Reference Creek	15.8	7.7	-	a
					CLT1 Upstream	33.2	11.6	2.3	b
					CLT1 Downstream	10.3	2.9	-0.7	a
Sprawler HPG (% of community)	ANOVA	none	YES	<0.001	Reference Creek	79.4	6.6	-	a
					CLT1 Upstream	52.6	9.7	-4.1	b
					CLT1 Downstream	69.1	4.8	-1.6	a
Burrower FFG (% of community)	ANOVA	log10	YES	0.001	Reference Creek	4.8	3.3	-	a
					CLT1 Upstream	13.5	4.8	2.6	b
					CLT1 Downstream	20.6	7.4	4.8	b

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

(blue box) 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] Statistical tests include Analysis of Variance (ANOVA) followed by Tukey's Honestly Significant Difference (HSD) post hoc tests, or Kruskal-Wallis H-test (K-W) followed by Mann-Whitney U-test (M-W).

CLT1 north branch and Unnamed Reference Creek, as indicated by significantly differing Bray-Curtis Index (Appendix Table F.7), included ecologically significant[6] greater relative abundance of Chironomidae and Tipulidae dominant groups, and lower relative abundance of Ostracoda, at the CLT1 north branch (Table 3.3). Within the Chironomidae, an ecologically significantly higher relative abundance of metal-sensitive taxa was indicated at the CLT1 north branch than at the reference creek, indicating no adverse influences on biota related to metals within the watercourse. Key differences in FFGs and HPGs, including significantly higher relative abundance of the shredder FFG, clinger HPG, and burrower HPG at the CLT1 north branch, were consistent with greater amounts of in-stream vegetation (e.g., bryophyte mosses) than at the reference creek as reported in previous CREMP studies (e.g., Minnow 2020). No consistent ecologically significant differences in density, richness, Simpson's Evenness, dominant taxonomic groups, or FFGs were indicated at the CLT1 north branch in 2020 compared to baseline studies conducted in 2007 and 2011 (Appendix Tables F.8 and F.9; Appendix Figure F.2). Collectively, the 2020 data suggested that differences in benthic invertebrate community assemblage between the CLT1 north branch and Unnamed Reference Creek reflected differences in the types and/or abundance of in-stream vegetation between these study areas. This was supported by comparisons to baseline, which indicated no ecologically significant changes in benthic invertebrate community metrics at the CLT1 north branch since the commencement of commercial mine operations in 2015.

3.1.4.2 Downstream Lower Main Stem (CLT1 DS)

The benthic invertebrate community at the lower main stem of Camp Lake Tributary 1 (CLT1 DS), downstream of the Milne Inlet Tote Road crossing, did not differ significantly in density, richness, or Simpson's Evenness compared to Unnamed Reference Creek in 2020 (Table 3.3; Appendix Figure F.1). Differences in benthic invertebrate community assemblage between CLT1 DS and the reference creek, as indicated by significantly differing Bray-Curtis Index (Appendix Table F.7), included ecologically significant greater relative abundance of Oligochaeta, Chironomidae, and Tipulidae dominant groups, and lower relative abundance of Ostracoda, at CLT1 DS (Table 3.3). However, similar to the CLT1 north branch, no significant difference in the relative abundance of metal-sensitive Chironomidae and ecologically significant higher relative abundance of the shredder FFG and the burrower HPG occurred at CLT1 DS compared to the reference creek

[6] Ecological significance is defined as a magnitude of difference between the mine-exposed and reference area that is outside of a CES (CES_{BIC}) of ±2 reference area SDs (SD_{REF}) for the benthic invertebrate community metric. Differences outside of the CES_{BIC} are greater than those that would be expected to occur naturally (i.e., between two pristine reference areas), and thus require additional evaluation to determine whether the difference is mine-related considering the direction of response and taking a weight-of-evidence approach that considers the results from other study components (e.g., water chemistry) and benthic invertebrate community endpoints.

in 2020 (Table 3.3). This indicated that the differences in community features between CLT1 DS and Unnamed Reference Creek were unlikely associated with metal concentrations, but rather due to naturally differing habitat (e.g., food resources and/or substrate properties) between study areas. No consistent ecologically significant differences in density, richness, Simpson's Evenness, or dominant taxonomic groups, including the proportion of metal-sensitive chironomids, were indicated at the CLT1 lower main stem in 2020 compared to both of the 2007 and 2011 baseline studies (Appendix Table F.10; Appendix Figure F.2). A significantly higher relative abundance of the collector-gatherer FFG was generally shown at CLT1 DS since 2015 compared to baseline, potentially indicating a shift in food resources available to benthic invertebrates at the lower main stem area over time (Appendix Table F.11). However, the absence of consistent ecologically significant differences in the relative abundance of metal-sensitive taxa in years of mine operation compared to baseline suggested that the FFG differences over time were unrelated to differing metal concentrations.

Between the CLT1 study areas, benthic invertebrate density, shredder FFG relative abundance, and clinger HPG relative abundance were significantly lower downstream than upstream of the Milne Inlet Tote Road crossing, but no significant differences in richness, evenness, or dominant taxonomic groups were indicated between the downstream and upstream areas in 2020 (Table 3.3; Appendix Figure F.1). Similar to differences in community features between the CLT1 north branch and reference creek, these differences in community features between CLT1 study areas reflected lower abundance of in-stream vegetation (e.g., mosses), which serves as a key food resource and habitat for the shredder FFG and clinger HPG, respectively, at the downstream area compared to upstream of the Milne Inlet Tote Road crossing. Therefore, in-stream vegetation was the key contributor to differences in benthic invertebrate density and FFG and HPG composition between the CLT1 lower main stem and upstream study areas in 2020 rather than influences associated with the Milne Inlet Tote Road.

3.1.5 Effects Assessment and Recommendations

3.1.5.1 Upstream North Branch (CLT1 US)

At the CLT1 north branch, the following AEMP benchmarks were exceeded in 2020:

- Aqueous total copper concentration greater than the benchmark of 0.0022 mg/L in spring and summer (0.00221 mg/L and 0.00226 mg/L, respectively) at Station L1-08.

Copper concentrations at the CLT1 north branch in spring of 2019 and 2020, and in fall from 2015 to 2020, were slightly higher than concentrations in respective seasonal sampling events during baseline. However, copper concentrations at the CLT1 north branch during summer sampling events from 2015 to 2020 were comparable to baseline, but were also shown to be above the AEMP benchmark in 29% of samples taken (Appendix Figure C.2). In addition, copper concentrations in spring and fall during years of mine commercial production from 2015 to 2020 were comparable to those shown in summer during baseline. No substantial mine development has occurred in the CLT1 north branch watershed, and thus mine-related sources of copper to this portion of the watercourse potentially included fugitive dust. However, because copper concentrations farther downstream at the CLT1 main stem, closer to sources of dust generation, were below AEMP benchmarks, the source of copper to the CLT1 north branch was likely related natural minerology of the bedrock/overburden in the region of the mine. Metal concentrations in sediment of the CLT1 north branch were well below SQG. In addition, no adverse effects on phytoplankton (chlorophyll-a) or benthic invertebrates of the CLT1 north branch were indicated in 2020, nor during studies conducted since the commencement of commercial mine production in 2015, indicating that copper concentrations above the AEMP benchmark at the CLT1 north branch may not have been biologically available.

Following application of the Mary River Project AEMP Management Response Framework (Figure 2.8), uncertainty in whether a change in copper concentrations has occurred at the CLT1 north branch between the period of commercial mine production and baseline results in a low action response related to copper concentrations above the AEMP benchmark at this watercourse. An expanded spatial water quality sampling program implemented at the CLT1 north branch as a special investigation to identify whether the source(s) of copper to the watercourse reflect natural minerology of the bedrock/overburden within the watershed is recommended as an initial low action response.

3.1.5.2 Downstream Main Stem (CLT1 DS)

At the CLT1 main stem, the following AEMP benchmarks were exceeded in 2020:

- Aqueous total aluminum concentration was greater than the benchmark of 0.179 mg/L in spring at the upper main stem Station L2-03 (0.270 mg/L); and,
- Aqueous total iron concentration was greater than the benchmark of 0.326 mg/L at upper main stem Station L2-03 in spring, summer, and fall (0.420 mg/L, 0.423 mg/L, and 0.522 mg/L, respectively).

Concentrations of all parameters were below AEMP water quality benchmarks at all stations within the lower main stem (i.e., Stations L1-09, L0-05, and L0-01), and metal concentrations in sediment were below SQG at the CLT1 downstream area (CLT1 DS), in 2020. Elevation of total aluminum concentrations above the AEMP water quality benchmark at the upper main stem in spring 2020 was related to suspended mineral material in the water column as reflected by high turbidity in these samples. Because total aluminum concentrations at the CLT1 main stem in

2020 were not elevated compared to the reference creek nor to concentrations at the upper main stem during baseline, the source of aluminum to the CLT1 main stem was likely related to background minerology of material entering the system during spring runoff events. In contrast, iron concentrations at the CLT1 upper main stem in 2020 were elevated compared to concentrations at the reference creek and at CLT1 during baseline, suggesting a mine-related source of iron to the system. Relatively high iron concentrations at the CLT1 main stem following the initiation of commercial mine operation potentially reflected blasting/excavating activity (including associated dust generation) at the Mine Site QMR2 Quarry, as well as fugitive dust generation from increased truck usage on the Milne Inlet Tote Road, compared to the baseline period. Despite elevated iron concentrations at the CLT1 upper main stem, no adverse effects on phytoplankton and benthic invertebrates were indicated at the CLT1 downstream in 2020, suggesting that potential biological effects from elevated iron concentrations were likely limited only to the CLT1 upper main stem and did not extend to the lower main stem or Camp Lake.

Under the Mary River Project AEMP Data Management Response Framework (Figure 2.8), the determination of a mine-related change to a parameter concentration above the AEMP benchmark necessitates a management response (Steps 2 and 3). Because a mine-related elevation in iron concentrations occurred at the CLT1 upper main stem in 2020, but the spatial extent was limited and no biological effects were observed a short distance downstream, consideration for the establishment of benthic invertebrate community sampling stations at the CLT1 upper main stem, close to water quality Station L2-03, is recommended to evaluate possible effects on biota in this portion of the CLT1 system as a low action response.

3.2 Camp Lake Tributary 2 (CLT2)

3.2.1 Water Quality

Camp Lake Tributary 2 (CLT2) dissolved oxygen was consistently near full saturation at the time of spring, summer, and fall monitoring events, and concentrations were comparable to or slightly higher than those at the reference creeks (Appendix Tables C.1 to C.3; Figure 3.4). In addition, dissolved oxygen concentrations at CLT2 were well above the WQG lowest acceptable concentration for early life stages of cold-water biota (i.e., 9.5 mg/L) at the time of biological sampling in August 2020 (Figure 3.4; Appendix Table C.12). Aqueous pH at the CLT2 upstream and downstream study areas was generally slightly higher (i.e., more alkaline) than at the reference creeks but consistently well within WQG limits during the spring, summer, and fall sampling events in 2020 (Appendix Tables C.1 to C.3; Figure 3.4). No significant difference in pH was indicated between CLT2 study areas located downstream and upstream of the Milne Inlet Tote Road suggesting that this road crossing did not markedly influence the pH of CLT2 (Appendix Table C.19). *In situ* specific conductance was consistently higher at CLT2 compared

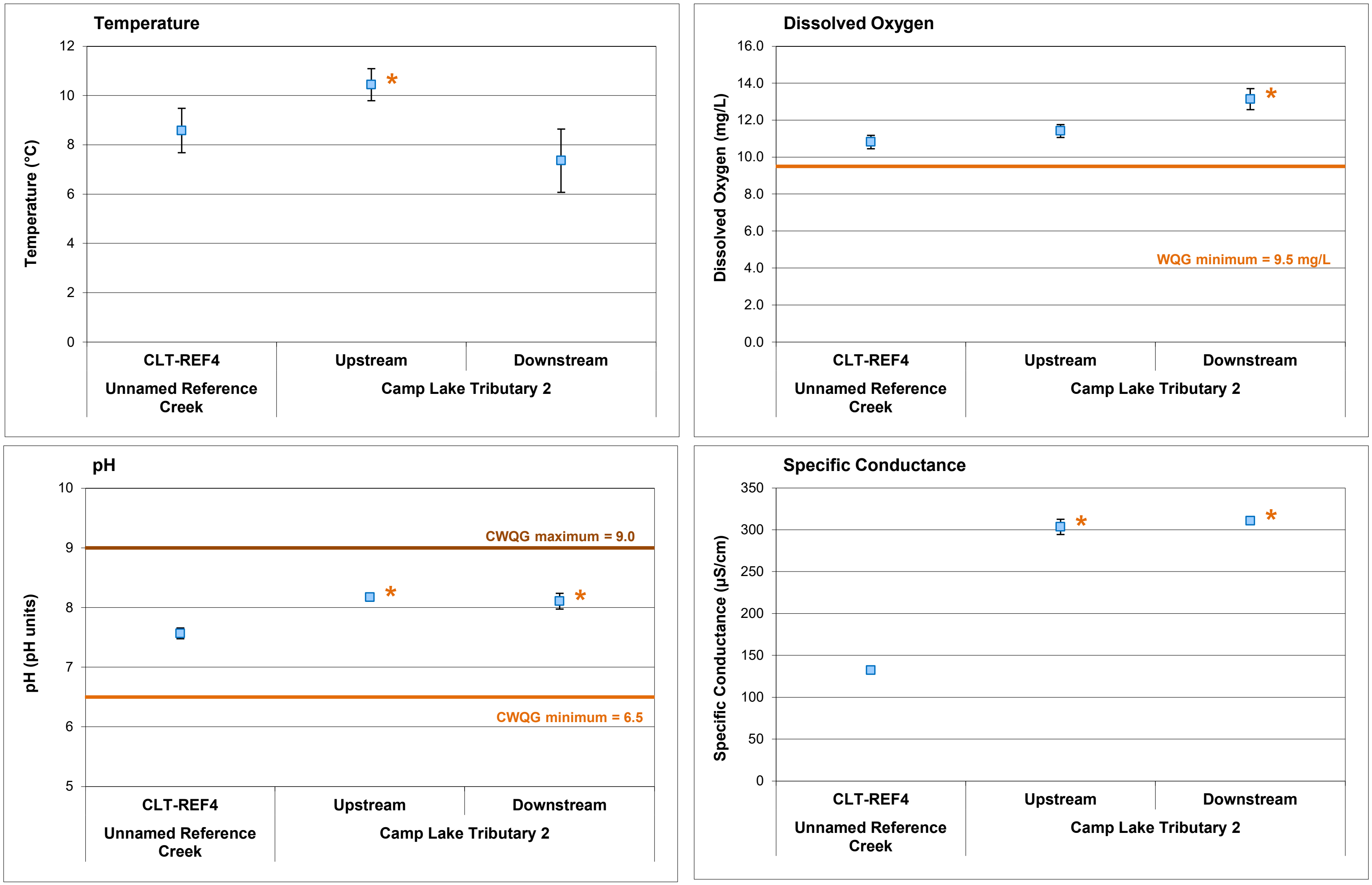


Figure 3.4: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 2 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2020

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

to the reference creeks in 2020, and was also significantly higher downstream compared to upstream of the Milne Inlet Tote Road at CLT2 during August 2020 biological sampling (Figure 3.4; Appendix Table C.19), suggesting a slight influence of the road on water quality at CLT2.

Water chemistry at CLT2 (Station KO-01) met all AEMP benchmarks and WQG in spring, summer, and fall sampling events of 2020 (Table 3.4). Among those parameters with established AEMP benchmarks, nitrate and sulphate concentrations showed moderate elevation (i.e., 5--to 10-fold) at CLT2 compared to mean concentrations at the reference creeks, but only during the spring sampling event in 2020 (Appendix Table C.15).[7] Chloride and sulphate concentrations were the only parameters with established AEMP benchmarks that were higher at CLT2 in 2020 compared to baseline, but concentrations of both of these parameters remained well below the AEMP benchmarks since the commencement of commercial mine operations in 2015 (Appendix Figure C.3). In addition, concentrations of chloride and sulphate at CLT2 were similar to those observed at the reference creeks in 2020, suggesting a natural factor may have accounted for higher concentrations of these parameters at CLT2 since baseline. For those parameters without AEMP benchmarks, only sodium and total and dissolved uranium concentrations showed elevation at CLT2 in 2020 compared to baseline. In consideration of all spatial and temporal (baseline) comparisons, no marked mine-related influence on water quality was indicated within the CLT2 system in 2020.

3.2.2 Sediment Quality

Sediment from CLT2 upstream (CLT2-US) and downstream (CLT2-DS) study areas was visually characterized as medium-sized coarse sand (Appendix Table D.7). The in-stream substrate at both CLT2 study areas was composed mainly of cobble material (i.e., substrate diameter 6 to 25 cm), with sand constituting a trace amount (i.e., <1%) and approximately 5% of the material observed at the sediment surface of the upstream and downstream areas, respectively (Minnow 2018). Mean sediment TOC content was low (i.e., <0.5%) at both CLT2 study areas, but approximately 2 to 3 times greater than the mean TOC content in sediment sampled at the lotic reference areas (Table 3.5; Appendix Table D.10).

Similar to CLT1, mean concentrations of metals in sediment from CLT2 were generally elevated compared to those measured at the lotic reference areas (Appendix Table D.10). This was particularly the case for calcium, copper, magnesium, nickel, and potassium, for which mean concentrations were five-fold or greater at one or both of CLT2-US and CLT2-DS study areas compared to mean concentrations at the lotic reference areas (Table 3.5; Appendix Table D.10).

[7] This statement includes the evaluation of both total and dissolved metal concentrations.

Table 3.4: Mean Water Chemistry at Camp Lake Tributary 2 (CLT2) Monitoring Stations During Spring, Summer, and Fall, Mary River Project CREMP, 2020

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creeks (n=4)			Camp Lake Tributary 2		
					Spring	Summer	Fall	Spring	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	55	134	175	154	300	345
	pH (lab)	pH	6.5 - 9.0	-	7.63	8.01	8.05	8.18	8.34	8.43
	Hardness (as $CaCO_3$)	mg/L	-	-	23.6	57.05	82.6	70	147	167
	Total Suspended Solids (TSS)	mg/L	-	-	3.2	2.7	2	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	85	85	99	120	153	192
	Turbidity	NTU	-	-	1.87	6.62	2.49	0.55	0.20	0.23
	Alkalinity (as $CaCO_3$)	mg/L	-	-	24	61	69	62	134	136
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.01	0.01225	0.01	<0.010	<0.010	<0.010
	Nitrate	mg/L	3	3	0.020	0.062	0.076	0.135	0.047	0.148
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.02	0.15	0.15	0.14	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.93	3.44	2.31	2.81	3.88	3.08
	Total Organic Carbon	mg/L	-	-	2.23	3.05	2.14	3.86	4.36	3.50
	Total Phosphorus	mg/L	0.030$^{\alpha}$	-	0.0045	0.0065	0.0039	0.0075	<0.0030	<0.0030
	Phenols	mg/L	0.004$^{\alpha}$	-	0.0010	0.0010	0.0021	<0.0010	<0.0010	0.0017
Anions	Bromide (Br)	`	-	-	0.10	0.1	0.1	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.2	4.07	7.09	2.4	9.1	13.6
	Sulphate (SO_4)	mg/L	218$^{\beta}$	218	1.31	5.52	9.25	11.80	14.90	26.20
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0775	***0.3106***	0.0593	0.0229	0.0094	0.0089
	Antimony (Sb)	mg/L	0.020$^{\alpha}$	-	0.0001	0.0001	0.0001	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	0.00010	0.00013	0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00362	0.00948	0.01031	0.00856	0.01550	0.01840
	Beryllium (Be)	mg/L	0.011$^{\alpha}$	-	0.0005	0.0004	0.0005	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0003875	0.0005	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.01	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	0.00001	0.00001	0.00001	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	4.9	11.8	16.5	13.9	27.6	31.8
	Chromium (Cr)	mg/L	0.0089	0.0089	0.00050	0.00082	0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009$^{\alpha}$	0.0040	0.00010	0.00016	0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.00071	0.00115	0.00102	0.00116	0.00159	0.00152
	Iron (Fe)	mg/L	0.30	0.326	0.077	0.2425	0.06625	<0.030	<0.030	<0.030
	Lead (Pb)	mg/L	0.001	0.001	0.000107	0.000226	0.000092	<0.000050	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	0.0010	0.0011	0.0010	0.0012	0.0021	0.0019
	Magnesium (Mg)	mg/L	-	-	2.86	6.7	9.6	8.9	16.6	20.3
	Manganese (Mn)	mg/L	0.935$^{\beta}$	-	0.00136	0.00300	0.00102	0.00093	0.00230	0.00077
	Mercury (Hg)	mg/L	0.000026	-	0.0000050	0.000005	0.000005	<0.0000050	<0.0000050	<0.0000050
	Molybdenum (Mo)	mg/L	0.073	-	0.00015	0.00045	0.00057	0.00031	0.00059	0.00073
	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00070	0.00057	0.00052	0.00074	0.00065
	Potassium (K)	mg/L	-	-	0.45	0.93	1.04	1.20	2.10	2.45
	Selenium (Se)	mg/L	0.001	-	0.0010	0.0007625	0.001	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.62	1.25	0.87	0.67	0.86	0.72
	Silver (Ag)	mg/L	0.00025	0.0001	0.000010	0.00002	0.00001	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.83	2.76	3.97	1.85	5.68	7.36
	Strontium (Sr)	mg/L	-	-	0.00488	0.01391	0.01850	0.00886	0.01950	0.02210
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00010	0.00008	0.00010	<0.00010	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	0.00010	0.0001	0.0001	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	0.011	0.0241	0.0100	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00045	0.00405	0.00737	0.00063	0.00326	0.00460
	Vanadium (V)	mg/L	0.006$^{\alpha}$	0.006	0.0010	0.0012	0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.003	0.003	<0.0030	<0.0030	<0.0030

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

[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.

Table 3.5: 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 2020

Parameter	Units	SQG[a]	Lotic Reference Stations		Camp Lake Tributary 2	
			Unnamed Reference Creek (REFCRK; n = 3)	Mary River Reference (GO-09; n = 3)	Upstream CLT2-US (n = 3)	Downstream CLT2-DS (n = 3)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC	%	10[α]	0.12 ± 0.035	0.11 ± 0.012	0.33 ± 0.11	0.26 ± 0.19
Aluminum (Al)	µg/g	-	584 ± 185	2,757 ± 1,141	4,483 ± 2,188	3,057 ± 1,592
Antimony (Sb)	µg/g	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Arsenic (As)	µg/g	17	0.22 ± 0.11	0.38 ± 0.10	0.74 ± 0.28	0.47 ± 0.16
Barium (Ba)	µg/g	-	2.72 ± 0.722	12.6 ± 5.05	15.9 ± 6.70	9.64 ± 4.36
Beryllium (Be)	µg/g	-	<0.10 ± 0	0.14 ± 0.040	0.19 ± 0.072	0.16 ± 0.056
Bismuth (Bi)	µg/g	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0
Boron (B)	µg/g	-	<5.0 ± 0	5.4 ± 0.75	5.3 ± 0.52	<5.0 ± 0
Cadmium (Cd)	µg/g	3.5	<0.020 ± 0	<0.020 ± 0	0.030 ± 0.0071	0.026 ± 0.0067
Calcium (Ca)	µg/g	-	494 ± 249	2,750 ± 894	5,277 ± 1,911	2,190 ± 851
Chromium (Cr)	µg/g	90	7.79 ± 5.39	13.6 ± 4.34	21.6 ± 7.03	15.2 ± 9.49
Cobalt (Co)	µg/g	-	0.953 ± 0.558	2.40 ± 0.758	4.45 ± 1.67	2.70 ± 1.41
Copper (Cu)	µg/g	110[α]	1.21 ± 0.899	4.45 ± 2.50	11.9 ± 4.15	7.46 ± 0.930
Iron (Fe)	µg/g	40,000[α]	12,493 ± 9,700	11,063 ± 2,423	18,067 ± 8,528	13,590 ± 8,265
Lead (Pb)	µg/g	91	1.49 ± 0.546	3.07 ± 0.857	3.49 ± 1.51	3.05 ± 0.879
Lithium (Li)	µg/g	-	<2.0 ± 0	5.0 ± 2.3	6.6 ± 2.6	4.1 ± 1.9
Magnesium (Mg)	µg/g	-	444 ± 165	2,810 ± 1,212	7,047 ± 2,826	4,073 ± 2,065
Manganese (Mn)	µg/g	1,100[α,β]	27.4 ± 14.6	75.7 ± 29.4	143 ± 40	102 ± 50.8
Mercury (Hg)	µg/g	0.486	<0.0050 ± 0	<0.0050 ± 0	<0.0050 ± 0	<0.0050 ± 0
Molybdenum (Mo)	µg/g	-	<0.10 ± 0	0.11 ± 0.023	0.34 ± 0.18	0.47 ± 0.41
Nickel (Ni)	µg/g	75[α,β]	1.76 ± 0.920	6.11 ± 1.99	15.3 ± 7.14	10.2 ± 4.79
Phosphorus (P)	µg/g	2,000[α]	167 ± 98	350 ± 118	254 ± 64	177 ± 68.0
Potassium (K)	µg/g	-	133 ± 42	750 ± 320	1,313 ± 881	1,077 ± 677
Selenium (Se)	µg/g	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0
Silver (Ag)	µg/g	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Sodium (Na)	µg/g	-	<50 ± 0	68 ± 21	63 ± 21	60 ± 17
Strontium (Sr)	µg/g	-	2.00 ± 0.544	4.72 ± 1.01	3.86 ± 1.018	2.51 ± 0.633
Sulphur (S)	µg/g	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	µg/g	-	<0.050 ± 0	0.068 ± 0.023	0.082 ± 0.043	0.070 ± 0.021
Tin (Sn)	µg/g	-	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0
Titanium (Ti)	µg/g	-	83.3 ± 47.4	353 ± 123	336 ± 147	248 ± 111
Uranium (U)	µg/g	-	0.5 ± 0.25	0.922 ± 0.298	0.743 ± 0.474	1.09 ± 0.332
Vanadium (V)	µg/g	-	16.8 ± 12.8	19.5 ± 5.06	16.5 ± 4.03	12.7 ± 8.70
Zinc (Zn)	µg/g	315	3.0 ± 1.2	10.3 ± 4.31	13.9 ± 7.48	17.7 ± 8.30
Zirconium (Zr)	µg/g	-	2.1 ± 0.91	5.8 ± 2.0	3.7 ± 1.3	3.4 ± 1.6

Indicates parameter concentration above SQG.

Notes: TOC = total organic carbon; SQG = sediment quality guideline; n = number of samples; SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life, probable effects level (PEL; CCME 2020), except those indicated by reference mark. α = Ontario Provincial Sediment Quality Objective (PSQO), severe effect level (SEL; OMOE 1993). β = British Columbia Working SQG, PEL (BC ENV 2020).

Despite higher concentrations than at the lotic reference areas, no metals were present at concentrations 1.5 times or greater at the downstream area compared to the upstream area of CLT2 (Table 3.5), suggesting minimal influence of the Milne Port Tote Road on sediment quality at CLT2-DS. Concentrations of all metals were also well below applicable SQG at all CLT2 stations (Table 3.5; Appendix Tables D.11 and D.12). Notably, metal concentrations in sediment from CLT2 were almost always lower than those from CLT1, potentially indicating reduced mine-influence with increasing distance from the mine.

3.2.3 Phytoplankton

Chlorophyll-a concentrations at CLT2 (Station KO-01) were within the range observed at the reference creeks during summer and fall sampling events, but were lower than concentrations at the reference creeks during the spring sampling event in 2020 (Figure 3.2). Concentrations of nutrients, including total ammonia, nitrate, and total phosphorus, were similar or higher at CLT2 compared to the reference creek stations during the spring sampling event (Appendix Tables C.14 and C.15), and therefore the occurrence of lower chlorophyll-a concentrations at CLT2 in spring 2020 did not appear to be related to differing nutrient concentrations. In addition, concentrations of all parameters were below WQG at CLT2 in spring 2020, and thus the lower chlorophyll-a concentrations at CLT2 compared to the reference creeks may have reflected natural variability (Appendix Table C.14). Notably, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L for all sampling events in 2020 at CLT2 (Figure 3.2). 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 CCME (2020) WQG categorization of oligotrophic based on mean aqueous total phosphorus concentrations below 10 µg/L at CLT2 during all spring, summer, and fall sampling events (Table 3.4; Appendix Table C.14). Higher chlorophyll-a concentrations occurred at CLT2 from 2017 to 2020 compared to the mine baseline period for the fall sampling event, but no increasing trend over time was suggested (Figure 3.3). For the reasons indicated above, higher chlorophyll-a concentrations at CLT2 in spring 2020 compared to the baseline period did not appear to be associated with a mine-related change in nutrient concentrations over time, and thus likely reflected natural seasonal/temporal variation in chlorophyll-a concentrations.

3.2.4 Benthic Invertebrate Community

Benthic invertebrate density and richness at both the upstream and downstream study areas of CLT2 did not differ significantly from Unnamed Reference Creek (Table 3.6; Appendix Figure F.3). Evenness at the CLT2 downstream area differed from the reference creek, but the magnitude of this difference was within the CES_{BIC} of ±2 SD_{REF} and positive (Table 3.6), indicating that this

Table 3.6: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 2 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2020

Metric	Overall 3-Area Comparison				Pair-wise, *post hoc* comparisons				
	Statistical Test[a]	Data Transform-ation	Significant Difference Among Areas?	P-value	Study Area	Mean	Standard Deviation (SD)	Magnitude of Difference (Ref SD)	Pairwise Comparison
Density (No. per m^2)	ANOVA	log10	NO	0.941	Reference Creek	713	296	-	a
					CLT2 Upstream	679	325	-0.1	a
					CLT2 Downstream	881	672	0.6	a
Richness (No. of Taxa)	ANOVA	none	NO	0.615	Reference Creek	16.0	4.4	-	a
					CLT2 Upstream	17.4	2.2	0.3	a
					CLT2 Downstream	18.6	5.1	0.6	a
Simpson's Evenness	K-W	rank	YES	0.039	Reference Creek	0.840	0.043	-	a
					CLT2 Upstream	0.879	0.058	0.9	ab
					CLT2 Downstream	0.921	0.033	1.9	b
Nemata (% of community)	K-W	rank	NO	0.215	Reference Creek	0.7	1.3	-	a
					CLT2 Upstream	0.5	0.7	-0.2	a
					CLT2 Downstream	8.2	15.8	5.9	a
Oligochaeta (% of community)	ANOVA	log10(x+1)	NO	0.181	Reference Creek	1.9	1.5	-	a
					CLT2 Upstream	15.3	19.0	9.0	a
					CLT2 Downstream	4.4	5.0	1.7	a
Hydracarina (% of community)	ANOVA	log10	NO	0.844	Reference Creek	4.5	3.7	-	a
					CLT2 Upstream	3.0	1.3	-0.4	a
					CLT2 Downstream	4.7	4.8	0.1	a
Ostracoda (% of community)	K-W	rank	YES	0.007	Reference Creek	30.6	11.7	-	a
					CLT2 Upstream	0.4	0.5	-2.6	b
					CLT2 Downstream	0.3	0.5	-2.6	b
Chironomidae (% of community)	ANOVA	log10	YES	0.024	Reference Creek	48.3	12.9	-	a
					CLT2 Upstream	70.4	14.2	1.7	b
					CLT2 Downstream	75.2	16.5	2.1	b
Metal Sensitive Chironomids (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	0.8	1.2	-	a
					CLT2 Upstream	5.7	3.5	3.9	b
					CLT2 Downstream	11.5	4.2	8.7	c
Simuliidae (% of community)	ANOVA	log10(x+1)	YES	0.068	Reference Creek	9.8	8.6	-	a
					CLT2 Upstream	1.9	1.7	-0.9	b
					CLT2 Downstream	3.1	1.9	-0.8	ab
Tipulidae (% of community)	K-W	rank	NO	0.482	Reference Creek	1.5	2.3	-	a
					CLT2 Upstream	1.8	1.0	0.1	a
					CLT2 Downstream	1.8	2.2	0.1	a
Collector-Gatherer FFG (% of community)	ANOVA	none	NO	0.849	Reference Creek	80.7	8.8	-	a
					CLT2 Upstream	80.5	11.6	0.0	a
					CLT2 Downstream	77.5	8.7	-0.4	a
Filterer FFG (% of community)	ANOVA	log10(x+1)	YES	0.069	Reference Creek	9.9	8.9	-	a
					CLT2 Upstream	1.7	1.5	-0.9	b
					CLT2 Downstream	3.1	2.1	-0.8	ab
Shredder FFG (% of community)	ANOVA	log10(x+1)	YES	0.037	Reference Creek	2.8	2.7	-	a
					CLT2 Upstream	8.5	3.1	2.1	ab
					CLT2 Downstream	11.2	7.0	3.2	b
Clinger HPG (% of community)	ANOVA	log10	NO	0.573	Reference Creek	15.8	7.7	-	a
					CLT2 Upstream	12.0	4.2	-0.5	a
					CLT2 Downstream	16.6	7.1	0.1	a
Sprawler HPG (% of community)	ANOVA	none	NO	0.222	Reference Creek	79.4	6.6	-	a
					CLT2 Upstream	65.1	17.6	-2.2	a
					CLT2 Downstream	66.3	14.1	-2.0	a
Burrower FFG (% of community)	ANOVA	log10	YES	0.044	Reference Creek	4.8	3.3	-	a
					CLT2 Upstream	22.9	16.2	5.5	b
					CLT2 Downstream	17.1	19.0	3.7	ab

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] Statistical tests include Analysis of Variance (ANOVA) followed by Tukey's Honestly Significant Difference (HSD) post hoc tests, or Kruskal-Wallis H-test (K-W) followed by Mann-Whitney U-test (M-W).

difference was not ecologically significant nor indicative of an adverse response, respectively. Differences in community composition were indicated between CLT2 and Unnamed Reference Creek based on differing Bray-Curtis Index (Appendix Table F.7), of which the only ecologically significant differences included significantly higher and lower relative abundance of Chironomidae and Ostracoda dominant groups, respectively, at one or both CLT2 study areas compared to the reference creek (Table 3.6; Appendix Figure F.3). Ecologically significant higher relative abundance of metal-sensitive chironomids was indicated at CLT2 study areas compared to the reference creek (Table 3.6), suggesting that the community composition differences between watercourses were not likely related to metal concentrations. In addition, no ecologically significant differences in benthic invertebrate FFG and HPG were shown at both CLT2 study areas compared to the reference creek (Table 3.6), indicating no substantial differences in food resources and habitat conditions available to benthic invertebrates between CLT2 and Unnamed Reference Creek. No consistent ecologically significant differences in any benthic invertebrate community endpoints were indicated at either of the CLT2 upstream and downstream study areas over years of mine operation (2015 to 2020) compared to 2007 baseline data with the exception of routinely higher evenness at CLT2 (Appendix Tables F.15 and F.16; Appendix Figure F.4). Because high evenness is normally associated with a healthy distribution of benthic invertebrate taxa, the occurrence of significantly higher evenness at CLT2 on a routine basis from 2015 to 2020 compared to baseline was not consistent with an adverse influence related to recent mine operations. Overall, greater evenness and relative abundance of metal-sensitive taxa at CLT2 compared to the reference creek in 2020, as well as no consistent differences in density, richness, and relative abundance of dominant groups and FFG at the CLT2 study areas between mine operational and baseline periods indicated no adverse mine-related effects to benthic invertebrates at CLT2.

Between the CLT2 study areas, no significant differences in benthic invertebrate density, richness, evenness, and relative abundance of dominant taxonomic groups, FFGs, or HPGs were indicated between study areas located downstream and upstream of the Milne Inlet Tote Road crossing in 2020 (Table 3.6; Appendix Figure F.3). Therefore, no effects to benthic invertebrates were evident at CLT2 in 2020 as a result of potential influences associated with the Milne Inlet Tote Road.

3.2.5 Effects Assessment and Recommendations

Water chemistry at CLT2 met all AEMP benchmarks in 2020. In addition, sediment quality met all SQG, and no adverse effects on phytoplankton or benthic invertebrates were indicated at CLT2 in 2020. Under the Mary River Project AEMP Management Response Framework, the absence of a mine-related change in AEMP benchmark parameters over time (or compared to background) requires no further management response (Figure 2.8). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no adjustment to the existing AEMP need be applied at CLT2 as part of the next monitoring program.

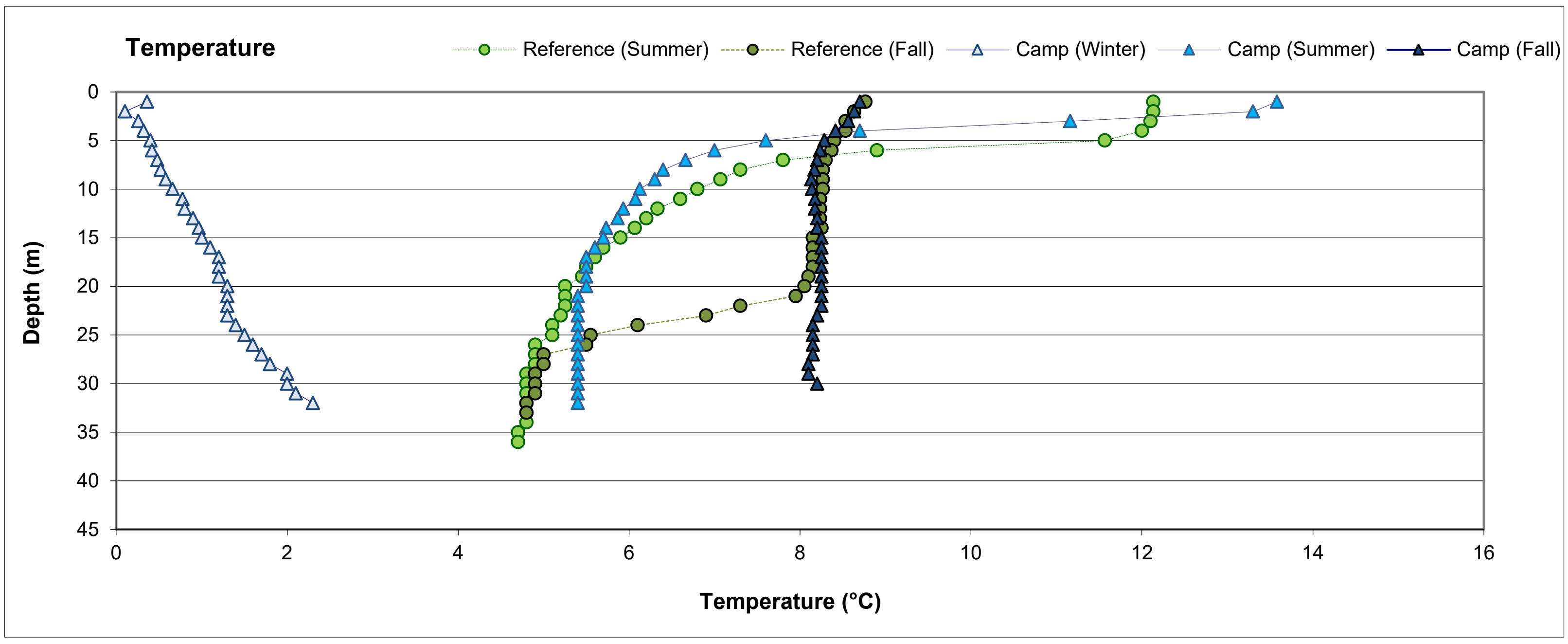
3.3 Camp Lake (JLO)

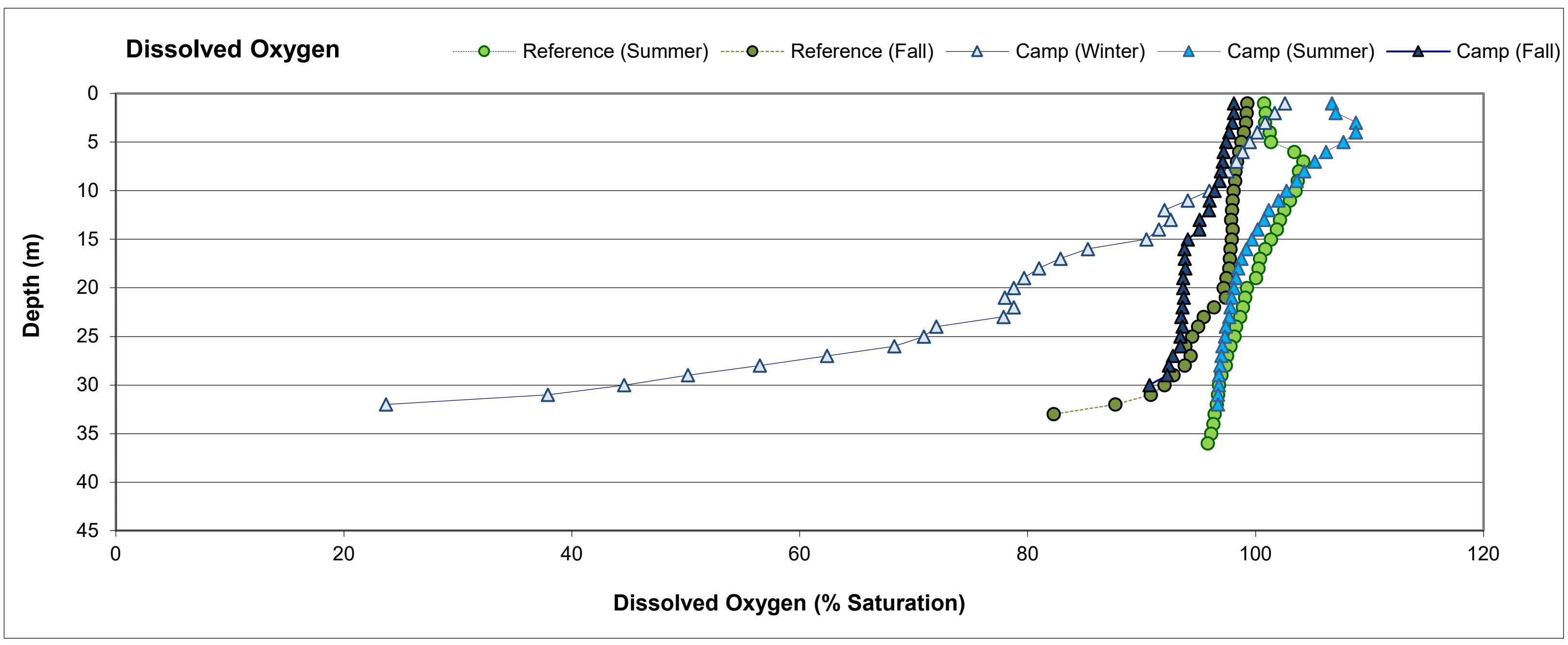
3.3.1 Water Quality

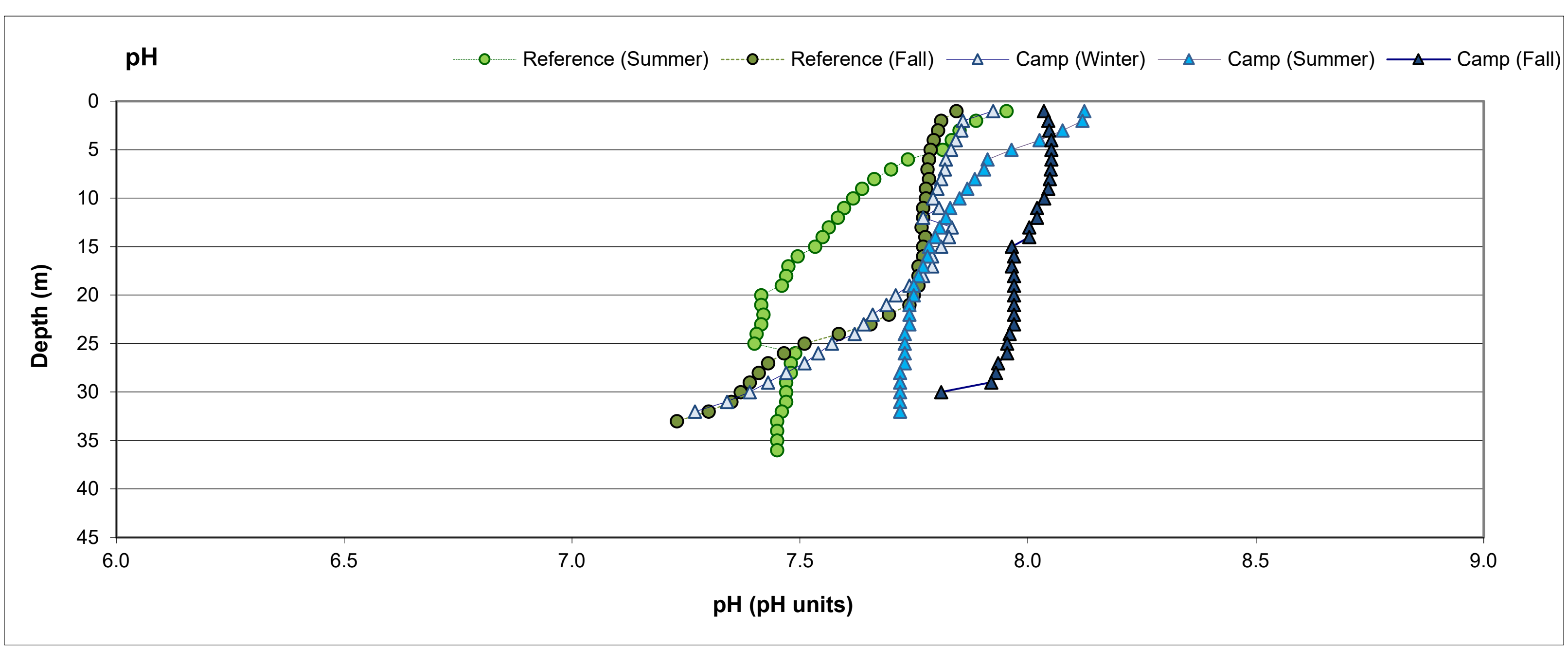
In situ water quality profiles conducted at Camp Lake showed no substantial spatial differences in water temperature, dissolved oxygen, pH or specific conductance with progression from the CLT1 inlet to the lake outlet during any of the winter, summer, or fall seasonal sampling events in 2020 (Appendix Figures C.4 to C.7). The 2020 Camp Lake water column profiles indicated a slight increase in temperature from surface to bottom (i.e., approximately 2˚C) during the winter sampling event, and a distinctly warmer surface layer extending to a depth of approximately 6 metres during the summer sampling event (Figure 3.5). The average temperature profiles at Camp Lake in summer and fall sampling events roughly mirrored those at Reference Lake 3 in 2020 (Figure 3.5). Water temperature near the bottom of the water column was significantly lower at littoral stations of Camp Lake than Reference Lake 3, but did not differ significantly between lakes at profundal stations sampled during August 2020 biological monitoring (Figure 3.6; Appendix Tables C.25).

Dissolved oxygen profiles conducted at Camp Lake in 2020 showed declining saturation levels with increased depth beginning at approximately 10 m below surface in the winter, but otherwise showed relatively minor changes from surface to bottom during the summer and fall that closely reflected the dissolved oxygen profiles observed at Reference Lake 3 (Figure 3.5). Although dissolved oxygen at the bottom of the water column was near full saturation at littoral and profundal sampling depths of Camp Lake, dissolved oxygen concentrations were significantly lower at Camp Lake than at Reference Lake 3 at the time of biological sampling in August 2020 (Figure 3.6; Appendix Table C.25). Dissolved oxygen concentrations 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) during all seasonal sampling events in 2020 except at water depths greater than approximately 25 m in winter (Figure 3.6; Appendix Tables C.20 to C.22). This suggested that dissolved oxygen concentrations were not likely to be limiting to biota at Camp Lake for most of the year, except for the portion of the water column greater than 25 m deep during the winter.

In situ profiles showed decreasing pH with increased depth at Camp Lake and Reference Lake 3, with the changes in pH through the water column at both lakes appearing to coincide with changes in water temperature and, to a lesser extent, dissolved oxygen levels (Figure 3.5). Although pH







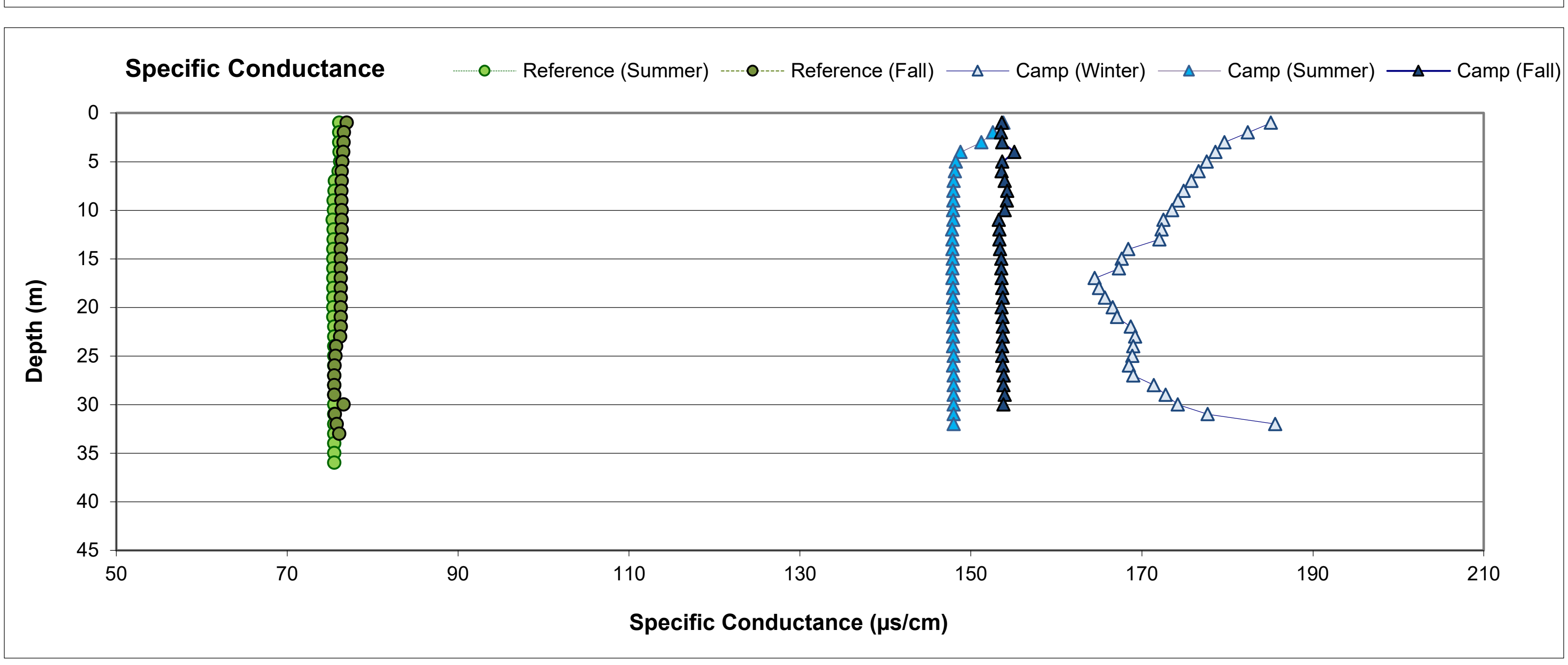


Figure 3.5: Average *In Situ* Water Quality with Depth from Surface at Camp Lake (JLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

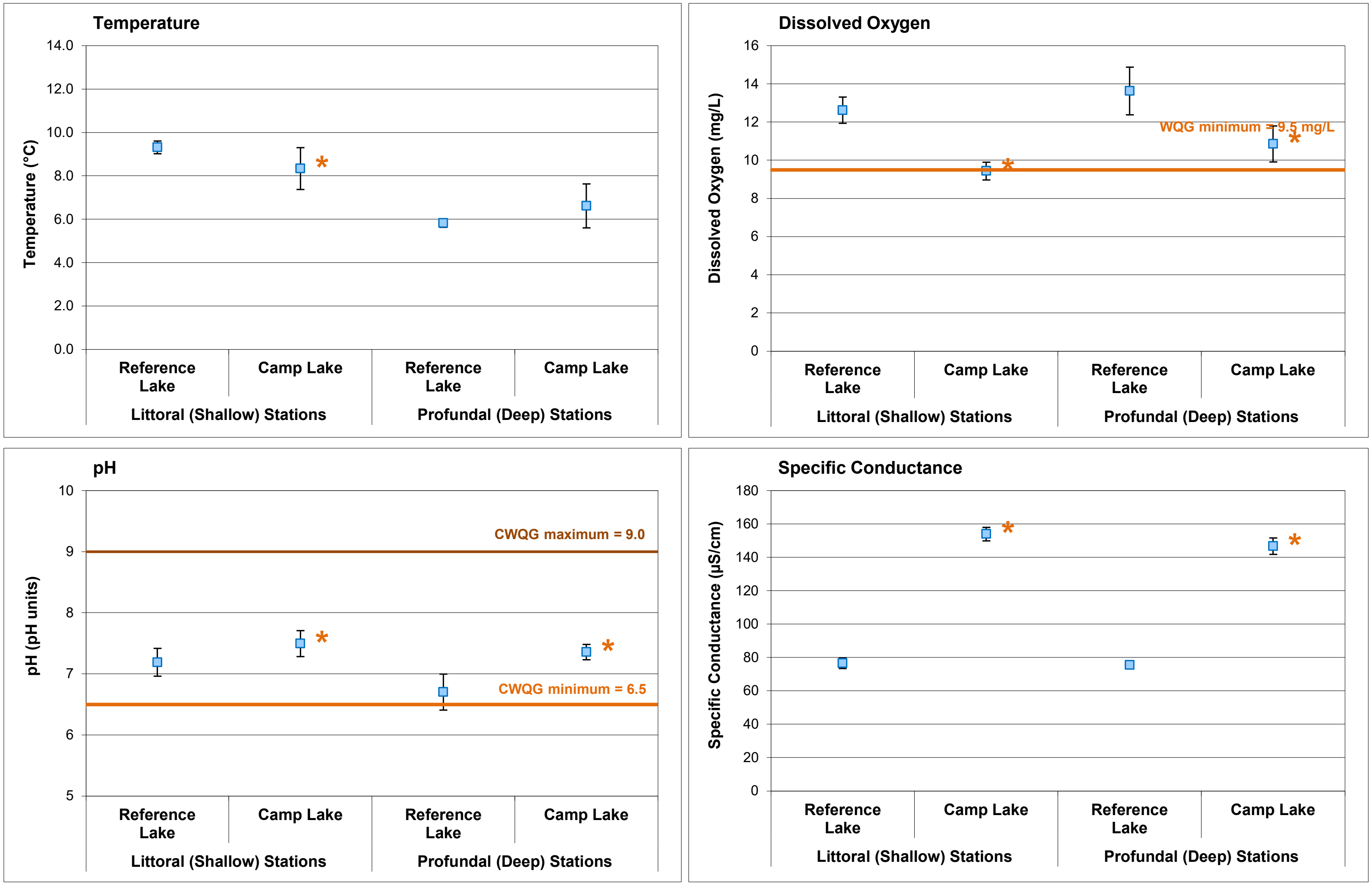


Figure 3.6: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake (JLO) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2020

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.

near the bottom at littoral and profundal stations of Camp Lake were significantly higher than at the reference lake during the August 2020 biological study, the mean incremental difference in pH between lakes was small (i.e., 0.6 pH units) and all pH values were consistently within WQG limits (Figure 3.6, Appendix Table C.26), suggesting that the pH difference between lakes was not ecologically meaningful. Specific conductance profiles showed no marked step changes from the surface to bottom of the Camp Lake water column, indicating the absence of chemical stratification (Figure 3.5). Specific conductance was consistently higher at Camp Lake than at Reference Lake 3 in summer and fall 2020 (Figure 3.5), the difference of which was shown to be significant during the August 2020 biological study (Figure 3.6) and possibly reflected a mine-related influence on water quality. Secchi depth readings, which serve as a proxy for water clarity, were significantly lower at Camp Lake than at Reference Lake 3 during the August 2020 biological study (Appendix Figure C.8) indicating more suspended particulate material in waters of Camp Lake.

Water chemistry at Camp Lake met all AEMP benchmarks and WQG over the duration of spring, summer, and fall sampling events in 2020 (Table 3.7). Among those parameters with established AEMP benchmarks, aluminum and chloride concentrations were moderately (i.e., 5- to 10-fold) and slightly (i.e., 3- to 5-fold) elevated, respectively, at Camp Lake compared to the reference lake (Table 3.7; Appendix Table C.27). Of those parameters without AEMP benchmarks, only total and dissolved manganese, molybdenum, and uranium concentrations were slightly elevated at Camp Lake compared to the reference lake during summer and/or fall sampling events in 2020 (Appendix Tables C.27 and C.29). Concentrations of chloride, sulphate, and total aluminum were elevated at Camp Lake in 2020 compared to baseline, though only during winter and/or summer sampling events (Appendix Figure C.9; Appendix Tables C.27 and C.29). In addition, concentrations of each of these parameters were consistently well below AEMP benchmarks since commercial mine operations commenced in 2015 (Appendix Figure C.9). Overall, comparisons to Reference Lake 3 water chemistry in 2020 and to Camp Lake baseline water chemistry suggested slightly elevated concentrations of chloride, manganese, molybdenum, and uranium at Camp Lake in 2020 which reflected a slight mine-related influence on water quality of the lake. However, because concentrations of all parameters remained well below AEMP benchmarks and WQG since commercial mine operations commenced in 2015, including in 2020, no adverse effects on biota were expected at Camp Lake.

3.3.2 Sediment Quality

Surficial sediment (i.e., top 2 cm) collected at the Camp Lake coring stations in 2020 was primarily composed of silt and sand with low (i.e., 0.3 to 3.4%) TOC content (Figure 3.7; Appendix Table D.15). Surficial sediment at littoral stations of Camp Lake contained significantly more sand

Table 3.7: Mean Water Chemistry at Camp Lake (JLO) and Reference Lake 3 (REF3) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 (n = 3)		Camp Lake Stations (n = 5)		
					Summer	Fall	Winter	Summer	Fall
Conventionals	Conductivity (lab)	umho/cm	-	-	79	79	179	154	142
	pH (lab)	pH	6.5 - 9.0	-	7.66	7.75	7.73	8.05	7.98
	Hardness (as $CaCO_3$)	mg/L	-	-	35	38	96	71	71
	Total Suspended Solids (TSS)	mg/L	-	-	2.0	2.0	2.0	2.0	2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	41	51	115	85	84
	Turbidity	NTU	-	-	0.15	0.15	0.13	0.78	0.31
	Alkalinity (as $CaCO_3$)	mg/L	-	-	46	34	80	67	64
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.010	0.014	0.019	0.005	0.011
	Nitrate	mg/L	3	3	0.020	0.020	0.054	0.043	0.030
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.001	0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	0.16	0.19	0.11	0.16
	Dissolved Organic Carbon	mg/L	-	-	3.3	3.5	2.4	2.4	2.3
	Total Organic Carbon	mg/L	-	-	4.6	3.8	2.7	2.0	2.8
	Total Phosphorus	mg/L	0.020^{α}	-	0.0041	0.0031	0.0044	0.0034	0.0039
	Phenols	mg/L	0.004^{α}	-	0.0010	0.0011	0.0021	0.0028	0.0013
Anions	Bromide (Br)	mg/L	-	-	0.1	0.1	0.1	0.05	0.1
	Chloride (Cl)	mg/L	120	120	1.4	1.4	5.8	4.5	4.5
	Sulphate (SO_4)	mg/L	218^{β}	218	3.6	3.6	5.1	4.6	4.6
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0031	0.0032	0.0119	0.0162	0.0049
	Antimony (Sb)	mg/L	0.020^{α}	-	0.0001	0.0001	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.0001	0.0001	0.0001	0.0001	0.0001
	Barium (Ba)	mg/L	-	-	0.0064	0.0070	0.0089	0.0074	0.0068
	Beryllium (Be)	mg/L	0.011^{α}	-	0.0005	0.0005	0.0005	0.0001	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0005	0.0005	0.00005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.01	0.01	0.01
	Cadmium (Cd)	mg/L	0.00012	0.00008	0.00001	0.00001	0.00001	0.000005	0.00001
	Calcium (Ca)	mg/L	-	-	7.2	7.2	18.6	14.0	13.7
	Chromium (Cr)	mg/L	0.0089	0.0089	0.0005	0.0005	0.0005	0.000117	0.0005
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	0.0001	0.0001	0.0001	0.0001	0.0001
	Copper (Cu)	mg/L	0.002	0.0022	0.00073	0.00075	0.00130	0.00091	0.00084
	Iron (Fe)	mg/L	0.30	0.326	0.03	0.03	0.03	0.03	0.03
	Lead (Pb)	mg/L	0.001	0.001	0.00005	0.00005	0.000068	0.00005	0.00005
	Lithium (Li)	mg/L	-	-	0.0010	0.0010	0.0016	0.0012	0.0011
	Magnesium (Mg)	mg/L	-	-	4.2	4.7	11.9	8.6	8.5
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00080	0.00068	0.00168	0.00290	0.00109
	Mercury (Hg)	mg/L	0.000026	-	0.000005	0.000005	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00013	0.00015	0.00048	0.00041	0.00037
	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00050	0.00082	0.00055	0.00059
	Potassium (K)	mg/L	-	-	0.9	0.9	1.5	1.3	1.2
	Selenium (Se)	mg/L	0.001	-	0.001	0.001	0.001	0.0000516	0.001
	Silicon (Si)	mg/L	-	-	0.50	0.50	0.54	0.46	0.32
	Silver (Ag)	mg/L	0.00025	0.0001	0.00001	0.00001	0.00001	0.00001	0.00001
	Sodium (Na)	mg/L	-	-	0.9	1.0	2.6	1.9	1.9
	Strontium (Sr)	mg/L	-	-	0.0084	0.0082	0.0152	0.0113	0.0102
	Thallium (Tl)	mg/L	0.0008	0.0008	0.0001	0.0001	0.0001	0.00001	0.0001
	Tin (Sn)	mg/L	-	-	0.0001	0.0001	0.000104	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.01	0.01	0.01	0.000763	0.01
	Uranium (U)	mg/L	0.015	-	0.00032	0.00033	0.00128	0.00125	0.00120
	Vanadium (V)	mg/L	0.006^{α}	0.006	0.0010	0.001	0.001	0.0005	0.001
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.003	0.00582	0.00529	0.003

(grey box) Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the applicable AEMP benchmark.

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

[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 to 2013) specific to Camp Lake.

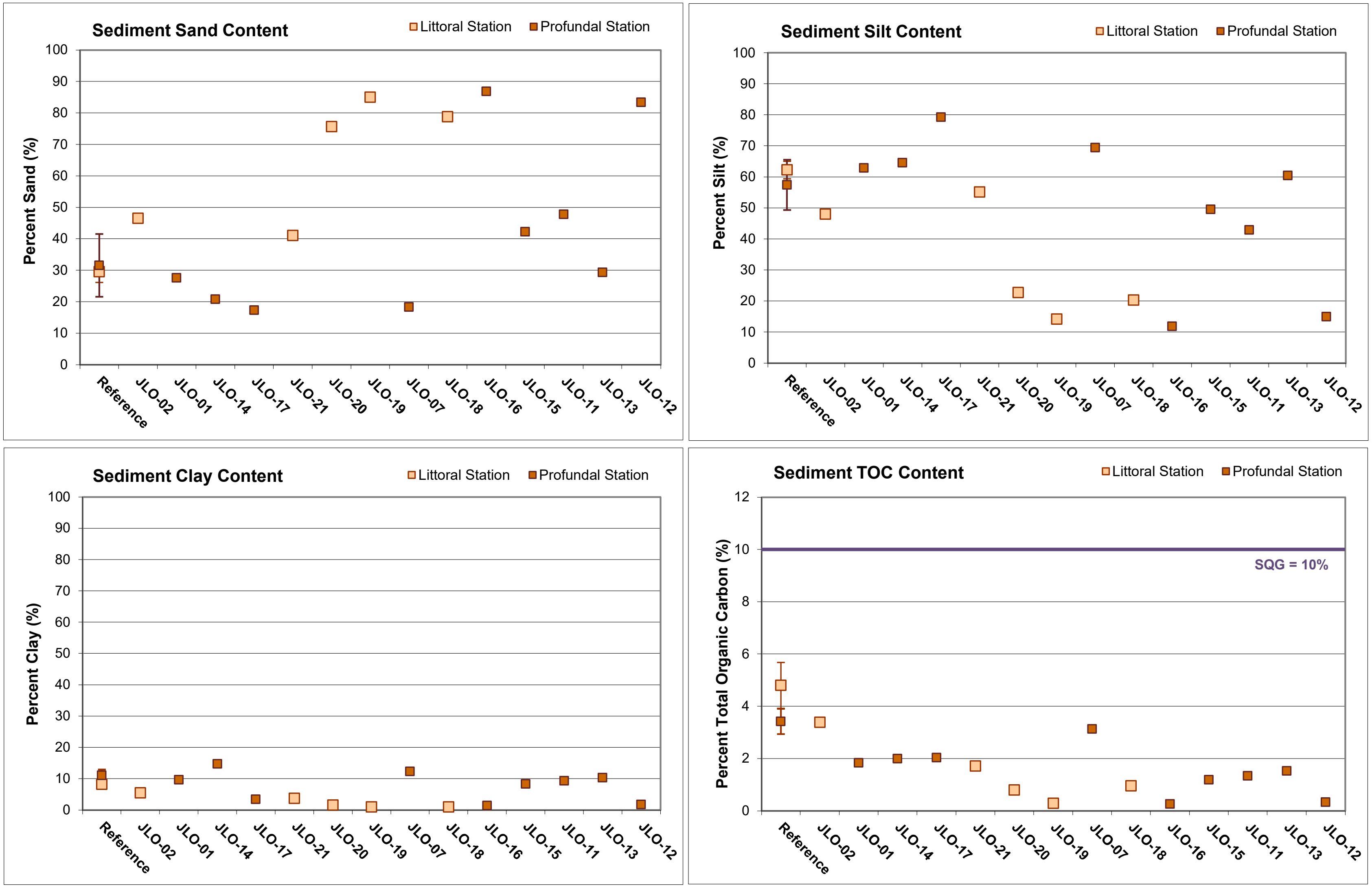


Figure 3.7: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Camp Lake (JLO) Sediment Monitoring Stations and to Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2020

and less silt and clay compared to Reference Lake 3 whereas the particle size of sediment from profundal areas of both lakes did not differ (Appendix Table D.16). The TOC in sediment at littoral and profundal stations of Camp Lake was significantly lower, and sediment was significantly more compact (i.e., lower moisture content), than at the reference lake (Figure 3.7; Appendix Table D.16). 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 observed in sediments at some Camp Lake stations (Appendix Tables D.13 and D.14). Similar observations of oxidized material were made at Reference Lake 3 (Appendix Tables D.3 and D.4), 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 and sulphidic odour was detected in sediment from some stations, suggesting occasional incidence of reducing conditions within substrates of the lake. However, no strongly defined redox boundaries were identified visually in Camp Lake sediments in 2020 (Appendix Table D.14). Qualitative observations suggestive of reducing conditions in sediment were similar between Camp Lake and Reference Lake 3 in 2020 (Appendix Tables D.3, D.4, D.13, and D.14), which indicated that factors leading to these conditions were comparable between lakes.

Evidence of slightly higher concentrations of some metals (e.g., arsenic, copper, iron, molybdenum, nickel, uranium) in sediment at stations located closer to the CLT1 inlet were indicated compared to those located near the outlet of Camp Lake in 2020 (Appendix Table D.15). However, these spatial differences in metal concentrations were most likely attributable to higher TOC and smaller particle size in sediments from stations closest to the CLT1 inlet (Appendix Table D.15) as supported by observations of no spatial changes in water chemistry within Camp Lake. Metal concentrations in littoral and profundal sediment of Camp Lake were comparable (i.e., less than a factor of 3-fold higher) to those of the reference lake in 2020 (Table 3.8; Appendix Table D.17). Iron and manganese concentrations were above their respective SQG, and arsenic, iron, and nickel concentrations were higher than the Camp Lake AEMP benchmarks, in sediment from the Camp Lake littoral station in 2020 (Table 3.8). Mean concentrations of iron and copper were also above SQG and AEMP benchmarks, respectively, in littoral sediments at Reference Lake 3 (Table 3.8). Because station JL0-02 is located near the CLT1 inlet, this suggested that mine-influenced flow from this tributary potentially contributed to higher concentrations of the metals indicated above in sediment at this location. Although the mean concentration of manganese was above the SQG in profundal sediment from Camp Lake, the mean concentration of this metal, as well as iron, were also above SQG in profundal sediment at Reference Lake 3 (Table 3.8) indicating naturally high concentrations of these metals in sediments of the study area. Concentrations of arsenic, copper, iron, manganese,

Table 3.8: Sediment Total Organic Carbon and Metal Concentrations at Camp Lake (JLO) and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2020

	Analyte	Units	SQG[a]	AEMP Benchmark[b]	Littoral Stations: Reference Lake (n = 5) Average ± SD	Littoral Stations: Camp Lake (n = 1)	Profundal Stations: Reference Lake (n = 5) Average ± SD	Profundal Stations: Camp Lake (n = 9) Average ± SD
	TOC	%	10^{α}	-	4.80 ± 1.96	3.39	3.42 ± 1.08	1.51 ± 0.889
Metals	Aluminum (Al)	mg/kg	-	-	16,880 ± 1,785	15,500	21,800 ± 2,185	15,303 ± 5,683
Metals	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	0.11	<0.10 ± 0	<0.10 ± 0
Metals	Arsenic (As)	mg/kg	17	5.9	3.53 ± 1.09	***9.03***	4.07 ± 0.397	5.54 ± 4.51
Metals	Barium (Ba)	mg/kg	-	-	117 ± 22	122	122 ± 18	82.7 ± 57.2
Metals	Beryllium (Be)	mg/kg	-	-	0.65 ± 0.073	0.78	0.80 ± 0.092	0.83 ± 0.33
Metals	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.29	<0.20 ± 0	0.27 ± 0.056
Metals	Boron (B)	mg/kg	-	-	12.2 ± 0.853	17.8	14.7 ± 1.77	24.5 ± 10.4
Metals	Cadmium (Cd)	mg/kg	3.5	1.5	0.173 ± 0.047	0.269	0.148 ± 0.0172	0.159 ± 0.074
Metals	Calcium (Ca)	mg/kg	-	-	5,608 ± 1,247	5,650	5,010 ± 407	5,122 ± 2,362
Metals	Chromium (Cr)	mg/kg	90	98	54.3 ± 4.40	66.2	65.0 ± 6.64	64.6 ± 20.1
Metals	Cobalt (Co)	mg/kg	-	-	10.8 ± 1.64	18.4	15.2 ± 1.56	16.3 ± 5.62
Metals	Copper (Cu)	mg/kg	110^{α}	50	***71.4*** ± 14.2	49.9	***83.8*** ± 11.1	39.6 ± 16.8
Metals	Iron (Fe)	mg/kg	$40,000^{\alpha}$	52,400	50,600 ± 24,939	***61,000***	45,080 ± 4,440	36,833 ± 15,000
Metals	Lead (Pb)	mg/kg	91	35	13.8 ± 0.799	18.9	16.7 ± 1.82	18.0 ± 7.48
Metals	Lithium (Li)	mg/kg	-	-	26.0 ± 2.51	22.4	33.7 ± 3.83	27.3 ± 10.3
Metals	Magnesium (Mg)	mg/kg	-	-	11,440 ± 814	13,400	14,180 ± 1,422	12,476 ± 2,881
Metals	Manganese (Mn)	mg/kg	$1,100^{\alpha,\beta}$	4,370	579 ± 258	1,410	1,230 ± 355	2,063 ± 2,299
Metals	Mercury (Hg)	mg/kg	0.486	0.17	0.0500 ± 0.0178	0.0530	0.0583 ± 0.0164	0.0404 ± 0.0233
Metals	Molybdenum (Mo)	mg/kg	-	-	4.44 ± 3.31	2.45	2.52 ± 0.273	1.52 ± 1.61
Metals	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	72	40.0 ± 3.52	***72.5***	45.0 ± 4.54	61.0 ± 18.5
Metals	Phosphorus (P)	mg/kg	$2,000^{\alpha}$	1,580	1,167 ± 394	1,310	956 ± 47	1,037 ± 510
Metals	Potassium (K)	mg/kg	-	-	4,100 ± 453	4,100	5,338 ± 543	4,171 ± 1,725
Metals	Selenium (Se)	mg/kg	-	-	0.73 ± 0.31	0.49	0.61 ± 0.18	0.37 ± 0.135
Metals	Silver (Ag)	mg/kg	-	-	0.14 ± 0.047	0.12	0.20 ± 0.057	0.13 ± 0.043
Metals	Sodium (Na)	mg/kg	-	-	304 ± 32	203	369 ± 50	227 ± 125
Metals	Strontium (Sr)	mg/kg	-	-	11.6 ± 1.70	9.87	12.3 ± 1.24	12.4 ± 5.93
Metals	Sulphur (S)	mg/kg	-	-	1,400 ± 387	<1,000	1,140 ± 195	1,789 ± 2,367
Metals	Thallium (Tl)	mg/kg	-	-	0.379 ± 0.0415	0.467	0.594 ± 0.094	0.435 ± 0.183
Metals	Tin (Sn)	mg/kg	-	-	<2.0 ± 0	<2.0	<2.0 ± 0	<2.0 ± 0
Metals	Titanium (Ti)	mg/kg	-	-	1,006 ± 109	833	1,136 ± 50	816 ± 259
Metals	Uranium (U)	mg/kg	-	-	11.0 ± 2.41	7.39	19.7 ± 3.76	5.04 ± 2.51
Metals	Vanadium (V)	mg/kg	-	-	54.1 ± 5.40	54.3	63.4 ± 4.89	52.5 ± 18.5
Metals	Zinc (Zn)	mg/kg	315	135	73.1 ± 7.83	59.4	83.8 ± 8.52	50.2 ± 19.2
Metals	Zirconium (Zr)	mg/kg	-	-	4.5 ± 1.0	7.8	3.9 ± 0.32	5.4 ± 3.6

[shaded] Indicates parameter concentration above SQG.

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV 2020).

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

nickel, and phosphorus were above respective Camp Lake AEMP benchmarks in sediment at some profundal stations of Camp Lake, but on average, were below the applicable benchmarks (Table 3.8; Appendix Table D.15). Of these metals, average concentrations of copper were also above the Camp Lake AEMP benchmark in profundal sediment at Reference Lake 3 (Table 3.8), providing further support for naturally elevated concentrations of copper in sediment.

Mean metal concentrations in sediment from Camp Lake littoral and profundal stations were comparable between 2020 and the baseline period for each respective station type (Appendix Table D.17). The only exception was a slightly higher (i.e., 3-fold greater) arsenic concentration in sediment from the single Camp Lake littoral station in 2020 (Figure 3.8; Appendix Table D.17).[8] Metal concentrations in sediment from Camp Lake littoral and profundal stations in 2020 were typically within the range of those observed from 2015 to 2019 (Figure 3.8). In addition, except for slightly higher mean concentrations of arsenic, calcium, and manganese, there was no evidence of consistently higher metal concentrations in Camp Lake sediments over the 2015 to 2020 period of mine operation relative to baseline (Figure 3.8). Overall, no substantial changes in sediment chemistry have been observed at Camp Lake following the commencement of mine operations in 2015.

3.3.3 Phytoplankton

Camp Lake chlorophyll-a concentrations showed no clear spatial gradients with distance from the CLT1 inlet to the lake outlet stations in 2020 (Figure 3.9). Chlorophyll-a concentrations were significantly lower in winter compared to summer and fall at Camp Lake in 2020 (Figure 3.9; Appendix Table E.6). Chlorophyll-a concentrations at Camp Lake did not differ significantly from those at Reference Lake 3 in the summer sampling event, but were significantly higher at Camp Lake in the fall sampling event (Appendix Tables E.7 and E.8). However, chlorophyll-a concentrations at Camp Lake were consistently well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events in 2020 (Figure 3.9). Average chlorophyll-a concentrations at Camp Lake suggested relatively low phytoplankton abundance and an 'oligotrophic' status based on comparison to Wetzel (2001) lake trophic classifications using chlorophyll-a concentrations. This trophic status classification was also consistent with an ultra-oligotrophic to oligotrophic WQG (CCME 2020) categorization for Camp Lake based on

[8] Boron concentrations in sediment from 2015 to 2020 were considerably higher (i.e., 10- to 70-fold) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences.

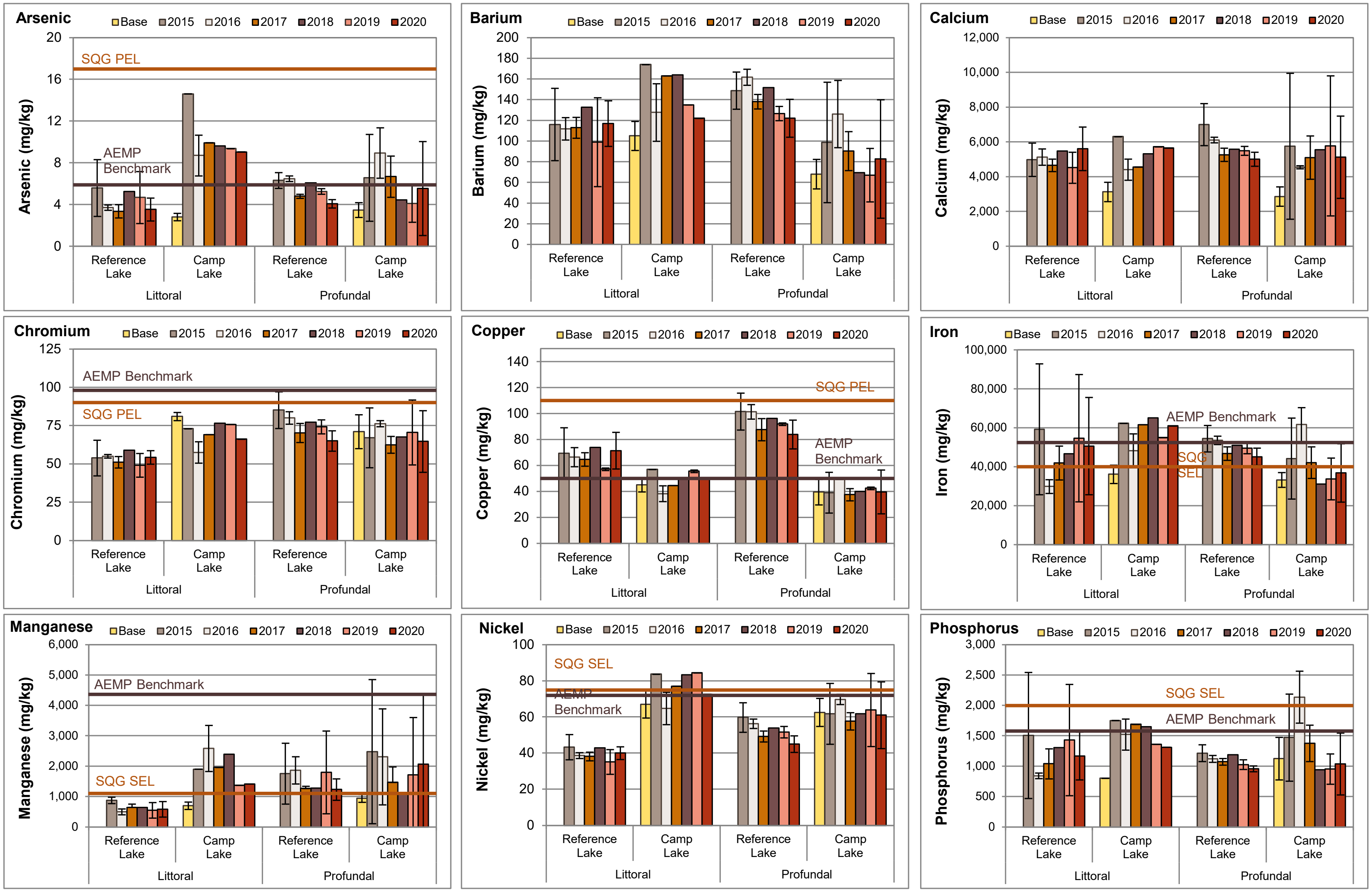


Figure 3.8: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Camp Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2020) Periods

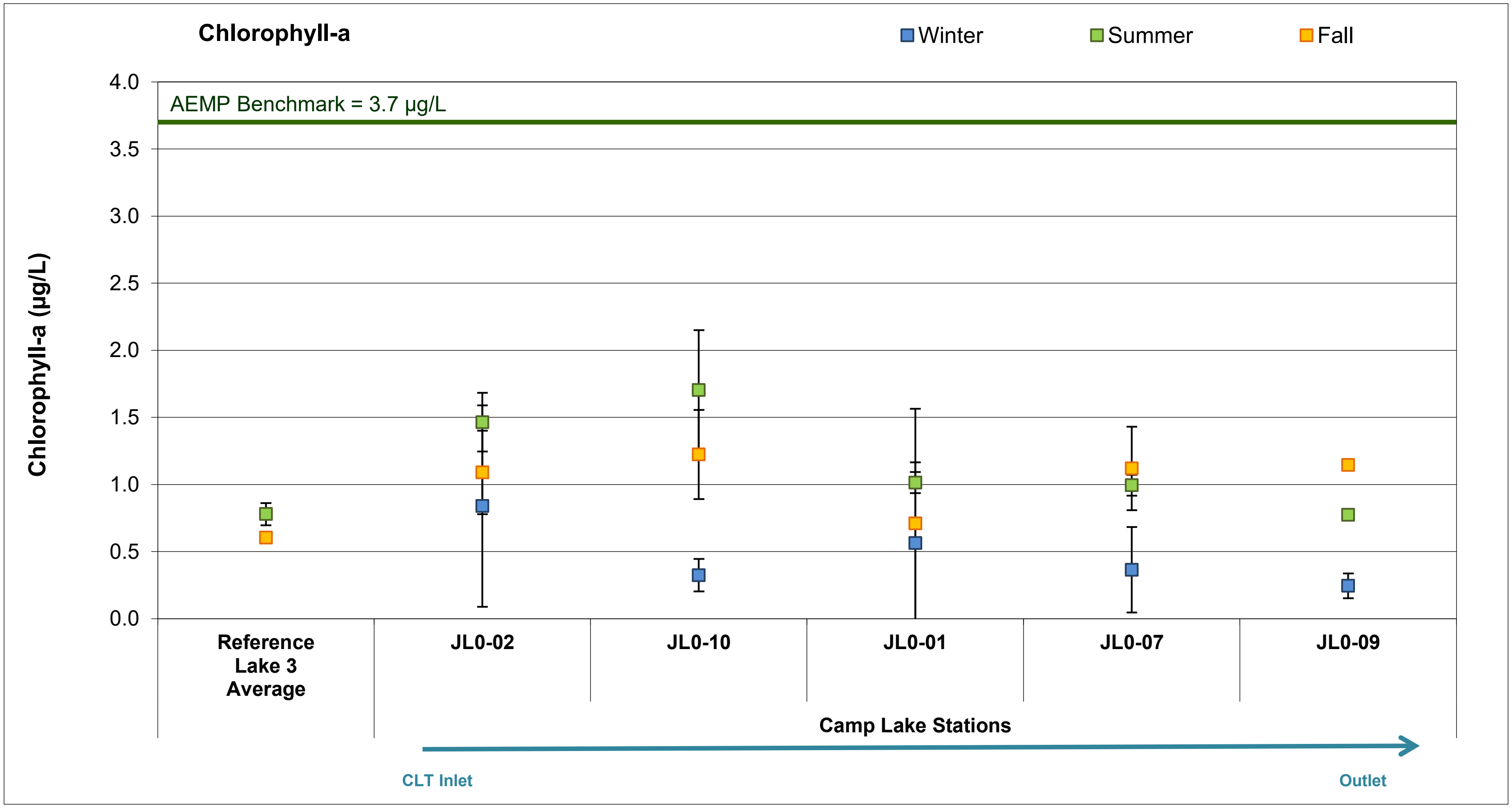


Figure 3.9: Chlorophyll a Concentrations at Camp Lake (JLO) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2020

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 2020.

mean aqueous total phosphorus concentrations below 10 µg/L for all seasonal sampling events (Table 3.7; Appendix Table C.26).

Temporal comparisons of the Camp Lake chlorophyll-a data did not indicate any consistent significant differences between years of mine construction (2014) and mine operation (2015 to 2020) for seasonal data collected in winter, summer, or fall (Figure 3.10). The lack of any consistent directional changes in chlorophyll-a concentrations for any given season among years was consistent with no substantial changes in nutrient (e.g., nitrate) concentrations and water quality generally achieving WQG at Camp Lake for the six years since mine operations commenced. No chlorophyll-a baseline (2005 to 2013) data are available for Camp Lake, precluding comparisons to conditions prior to the mine construction period.

3.3.4 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 3.9 and 3.10). For both habitat types, the difference was ecologically significant based on the magnitude being outside of the CES_{BIC} of ±2 SD_{REF}. No significant differences in richness or evenness were indicated between Camp Lake and the reference lake for either littoral or profundal habitat (Tables 3.9 and 3.10). Bray-Curtis Index differed significantly between Camp Lake and Reference Lake 3 for both littoral and profundal habitat types (Appendix Table F.21), indicating benthic invertebrate community structural differences between lakes. No ecologically significant differences in relative abundance of metal-sensitive Chironomidae were indicated between Camp Lake and Reference Lake 3 (Tables 3.9 and 3.10), which was consistent with metal concentrations in water and sediment of Camp Lake generally below applicable guidelines (Tables 3.7 and 3.8).[9] Therefore, the difference(s) in community structure between lakes appeared unrelated to metal concentrations.

The key differences in benthic invertebrate community composition between Camp Lake and Reference Lake 3 included significantly higher and lower relative abundance of Chironomidae and Ostracoda dominant groups, respectively, at littoral habitat of Camp Lake (Tables 3.9 and 3.10). No ecologically significant differences in FFGs were indicated between Camp Lake and the reference lake (Tables 3.9 and 3.10), suggesting a similar food resource base for benthic invertebrates between lakes. However, an ecologically significant higher relative abundance of the burrower HPG was present at Camp Lake compared to Reference Lake 3 (Tables 3.9

[9] Although mean concentrations of iron and manganese in sediment were above SQG at Camp Lake, the concentrations of these metals in sediment of the reference lake were also above SQG (Table 3.8), indicating natural elevation of iron and manganese in lakes of the study area.

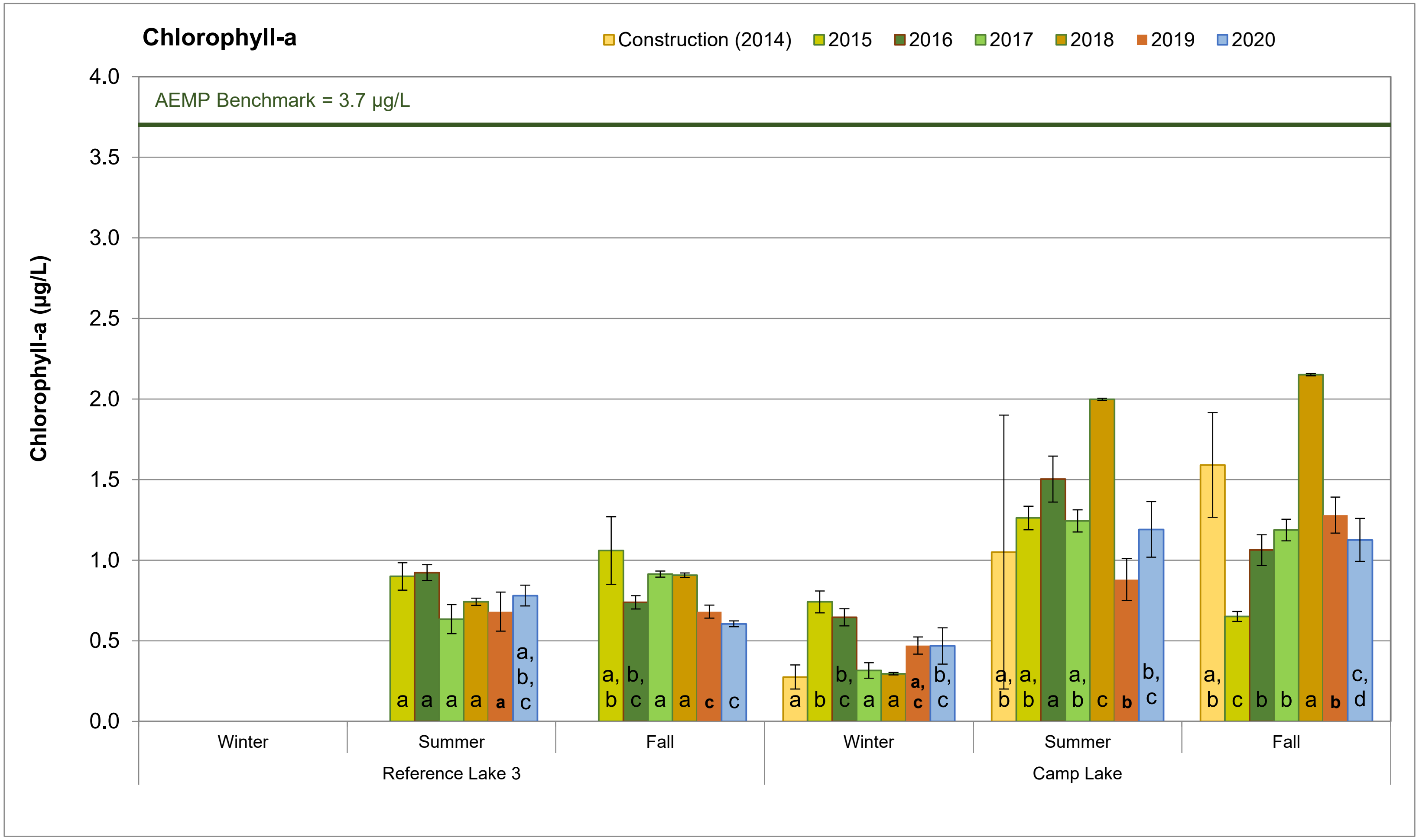


Figure 3.10: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Camp Lake and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2020) Periods (mean ± SE)

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

Table 3.9: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JLO) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-equal	log10	YES	0.003	8.3	Reference Lake 3	1,571	430	193	1,190	1,474	2,310
						Camp Lake Littoral	5,122	2,202	985	2,000	5,474	8,052
Richness (Number of Taxa)	t-equal	log10	NO	0.150	0.8	Reference Lake 3	14.6	2.5	1.1	13.0	14.0	19.0
						Camp Lake Littoral	16.6	1.7	0.7	14.0	17.0	18.0
Simpson's Evenness (E)	t-equal	none	NO	0.378	0.3	Reference Lake 3	0.810	0.110	0.049	0.630	0.847	0.923
						Camp Lake Littoral	0.842	0.044	0.020	0.773	0.858	0.889
Shannon Diversity	t-equal	log10	NO	0.559	0.2	Reference Lake 3	2.710	0.526	0.235	2.080	2.730	3.510
						Camp Lake Littoral	2.820	0.129	0.058	2.700	2.770	3.010
Hydracarina (%)	t-equal	log10	NO	0.568	-1.2	Reference Lake 3	5.3	2.6	1.2	3.5	4.4	9.9
						Camp Lake Littoral	2.0	2.4	1.1	0.3	1.0	6.0
Ostracoda (%)	Mann-Whitney	rank	YES	0.008	-2.5	Reference Lake 3	37.9	14.5	6.5	26.7	36.2	62.6
						Camp Lake Littoral	2.0	1.6	0.7	0.0	2.1	4.1
Chironomidae (%)	Mann-Whitney	rank	YES	0.008	2.6	Reference Lake 3	52.6	15.6	7.0	26.9	59.0	66.4
						Camp Lake Littoral	92.9	2.0	0.9	90.8	92.9	95.8
Metal-Sensitive Chironomidae (%)	t-equal	none	NO	0.882	0.2	Reference Lake 3	28.8	9.5	4.3	15.6	32.5	38.7
						Camp Lake Littoral	30.3	19.7	8.8	9.7	33.7	56.9
Collector-Gatherers (%)	t-equal	log10	NO	0.440	-0.5	Reference Lake 3	63.1	11.4	5.1	53.6	60.3	81.5
						Camp Lake Littoral	56.9	17.7	7.9	34.0	54.9	83.4
Filterers (%)	t-equal	none	NO	0.804	0.3	Reference Lake 3	27.1	9.8	4.4	14.4	29.2	38.0
						Camp Lake Littoral	29.7	20.2	9.0	7.0	33.7	56.2
Shredders (%)	t-unequal	none	YES	0.088	-1.0	Reference Lake 3	3.9	3.3	1.5	0.6	3.2	7.4
						Camp Lake Littoral	0.5	0.7	0.3	0.0	0.2	1.6
Clingers (%)	t-equal	none	NO	0.742	-0.4	Reference Lake 3	31.9	9.3	4.2	17.9	33.5	41.6
						Camp Lake Littoral	28.6	19.7	8.8	4.6	25.5	56.2
Sprawlers (%)	t-equal	log10	YES	0.046	-1.8	Reference Lake 3	57.9	12.1	5.4	41.0	57.2	73.8
						Camp Lake Littoral	36.7	17.8	8.0	22.0	26.6	65.0
Burrowers (%)	t-equal	log10	YES	0.002	5.0	Reference Lake 3	10.2	4.9	2.2	4.6	8.3	17.3
						Camp Lake Littoral	34.7	12.5	5.6	21.8	31.1	55.3

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

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

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

Table 3.10: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JLO) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-equal	log10	YES	0.005	6.4	Reference Lake 3	479	142	63	336	491	681
						Camp Lake Profundal	1,383	656	293	621	1,138	2,345
Richness (Number of Taxa)	t-equal	log10	NO	0.125	1.8	Reference Lake 3	7.0	1.9	0.8	5.0	8.0	9.0
						Camp Lake Profundal	10.4	4.5	2.0	7.0	8.0	18.0
Simpson's Evenness (E)	t-equal	log10	NO	0.346	-1.3	Reference Lake 3	0.731	0.045	0.020	0.689	0.721	0.795
						Camp Lake Profundal	0.673	0.151	0.068	0.512	0.643	0.901
Shannon Diversity	t-equal	log10	NO	0.623	1.2	Reference Lake 3	1.800	0.196	0.088	1.580	1.720	2.030
						Camp Lake Profundal	2.040	0.750	0.336	1.350	2.000	3.280
Hydracarina (%)	t-equal	log10(x+1)	NO	0.494	-0.5	Reference Lake 3	2.8	2.0	0.9	0.0	3.5	5.1
						Camp Lake Profundal	1.9	2.2	1.0	0.0	1.4	5.6
Ostracoda (%)	t-equal	log10(x+1)	NO	0.386	-0.8	Reference Lake 3	8.6	4.1	1.8	3.5	7.7	14.5
						Camp Lake Profundal	5.5	7.1	3.2	0.0	1.5	17.3
Chironomidae (%)	t-equal	none	NO	0.484	0.7	Reference Lake 3	87.9	4.2	1.9	82.3	87.2	92.7
						Camp Lake Profundal	91.0	8.4	3.8	77.6	90.8	98.5
Metal-Sensitive Chironomidae (%)	t-equal	none	YES	0.049	-1.1	Reference Lake 3	31.5	17.6	7.9	7.9	38.0	49.3
						Camp Lake Profundal	11.9	6.9	3.1	1.9	11.1	18.5
Collector-Gatherers (%)	t-equal	log10	YES	0.020	1.6	Reference Lake 3	62.9	15.0	6.7	45.4	56.1	79.0
						Camp Lake Profundal	87.0	9.5	4.3	75.1	90.1	99.3
Filterers (%)	t-equal	none	YES	0.037	-1.2	Reference Lake 3	30.7	17.5	7.8	7.9	38.0	49.3
						Camp Lake Profundal	9.4	7.4	3.3	0.4	9.1	18.5
Shredders (%)	t-unequal	log10(x+1)	NO	0.119	-0.9	Reference Lake 3	2.2	2.3	1.0	0.0	2.5	5.3
						Camp Lake Profundal	0.2	0.5	0.2	0.0	0.0	1.1
Clingers (%)	t-equal	none	YES	0.046	-1.2	Reference Lake 3	33.5	16.9	7.6	13.1	41.5	52.8
						Camp Lake Profundal	12.4	10.6	4.8	0.4	15.4	25.5
Sprawlers (%)	t-unequal	log10	NO	0.267	-0.9	Reference Lake 3	64.8	16.2	7.2	45.5	58.5	87.0
						Camp Lake Profundal	50.0	40.2	18.0	12.3	40.1	93.0
Burrowers (%)	t-unequal	log10(x+1)	YES	0.057	12.4	Reference Lake 3	1.7	2.9	1.3	0.0	0.0	6.7
						Camp Lake Profundal	37.5	32.3	14.4	5.6	34.4	72.3

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

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

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

and 3.10). This difference in HPG between lakes may have reflected the occurrence of more compact (i.e., lower moisture content), sandier sediment at Camp Lake (Appendix Table F.18). Substrate compactness is an important factor influencing inhabitation by burrowing invertebrates (Ward 1992), and thus greater substrate compactness may have accounted for the subtle benthic invertebrate community assemblage differences at Camp Lake compared to Reference Lake 3. Overall, markedly higher benthic invertebrate density without accompanying differences in richness, evenness, and FFG at Camp Lake compared to Reference Lake 3 suggested that Camp Lake was more biologically productive.

No consistent significant differences in general community effect indicators of density, richness, and evenness were indicated at littoral and profundal habitats of Camp Lake over years of mine operation (2015 to 2020) compared to baseline (2007, 2013; Appendix Tables F.22 and F.23; Appendix Figures F.5 and F.6). Similarly, benthic invertebrate dominant taxonomic groups and FFGs over years of mine operation from 2015 to 2020 did not differ significantly from baseline at littoral habitat at Camp Lake (Appendix Table F.22). At profundal habitat of Camp Lake, the relative abundance of metal-sensitive chironomids and the filterer FFG were routinely significantly lower at magnitudes outside of the CES_{BIC} of ±2 SD_{REF} over years of mine operation compared to the 2007 baseline data, but not to the 2013 baseline data (Appendix Table F.23). This indicated that the study-to-study differences in community features at profundal stations of Camp Lake were likely the result of sampling artifacts (e.g., differences in sampling station locations and/or replication among studies) or natural temporal variability among studies and was not related to potential influences from mine operation. Therefore, consistent with only minor changes in water and sediment quality since the mine baseline period, no ecologically significant differences in benthic invertebrate community features were indicated at littoral and profundal habitat of Camp Lake following the commencement of commercial mine operation in 2015.

3.3.5 Fish Population

3.3.5.1 Camp Lake Fish Community

The fish community at Camp Lake was composed of arctic charr (*Salvelinus alpinus*) and ninespine stickleback (*Pungitius*; Table 3.11), reflecting the same fish species observed previously (Minnow 2020). Higher CPUE for arctic charr and ninespine stickleback occurred at Camp Lake compared to Reference Lake 3 suggesting greater densities of both species at Camp Lake (Table 3.11). The higher density of fish at Camp Lake compared to Reference Lake 3 may be linked to greater productivity within Camp Lake based on higher chlorophyll-a concentrations in water (indicative of greater phytoplankton density) and greater benthic invertebrate density. Electrofishing CPUE for arctic charr at Camp Lake in 2020 was within the range observed during baseline studies (2007 to 2013) and over the five previous years

Table 3.11: 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 2020

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	134	1	135	2
		CPUE	2.09	0.016	2.11	
	Gill netting	No. Caught	69	0	69	
		CPUE	0.956	0	0.956	
Camp Lake	Electrofishing	No. Caught	109	18	127	2
		CPUE	4.93	0.814	5.75	
	Gill netting	No. Caught	94	0	94	
		CPUE	14.8	0	14.8	

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

since mine operation commenced (Figure 3.11). In contrast, the CPUE associated with gill netting at Camp Lake in 2020 was substantially greater than baseline and earlier years of mine operations (Figure 3.12). An increase in the gill netting CPUE (almost three times greater than 2019) was also observed at Reference Lake 3 in 2020 (Figure 3.11). Higher gill netting CPUE in 2020 at both Camp Lake and Reference Lake 3 compared to previous studies likely reflected slightly earlier sampling timing in 2020 which, due to warmer water temperatures, may have resulted in greater fish movement and thus higher catches. Because electrofishing is an 'active' fish collection method, the similarity in electrofishing CPUE between 2020 and baseline, and between 2020 and other years of mine operation, suggested no substantial changes in within-lake fish densities at either lake and supported the notion that slight difference in sampling timing among years likely accounted for higher gill netting CPUE in 2020.

3.3.5.2 Camp Lake Fish Population Assessment

Nearshore Arctic Charr

A total of 100 arctic charr were sampled from the nearshore habitat of each of Camp Lake and Reference Lake 3 in August 2020. Arctic charr YOY were distinguished from older (non-YOY) age classes at both lakes using a fork length of 4.3 cm based on the evaluation of length-frequency distributions coupled with supporting age determinations (Figure 3.12; Appendix Tables G.4 and G.5) and historical evaluations (Minnow 2020). Due to limited capture of YOY in Camp Lake (i.e., only 2 of 100 individuals), statistical comparisons focused only on non-YOY individuals. The length-frequency distribution for the nearshore arctic charr differed significantly between Camp Lake and Reference Lake 3 (Table 3.12; Appendix Table G.6) based on fewer YOY and smaller-sized individuals captured at Camp Lake (Figure 3.12). Non-YOY arctic charr from Camp Lake were significantly longer (8%) and heavier (44%) than those from Reference Lake 3 (Table 3.12; Appendix Table G.6). Condition (i.e., weight-at-length) of non-YOY was significantly greater for arctic charr captured at Camp Lake than those from the reference lake, although the magnitude of this difference (7%) was within the CES of ±10% (referred to herein as CES_C), suggesting that this difference was not ecologically significant (Table 3.12; Appendix Table G.6).

Arctic charr non-YOY at Camp Lake were almost consistently significantly longer and heavier than at Reference Lake 3 from 2015 to 2020, indicating consistent presence of larger juveniles at Camp Lake (Table 3.12). In contrast, condition of non-YOY arctic charr showed no consistent differences, and no consistent direction of differences, between Camp Lake and the reference lake from 2015 to 2020 (Table 3.12) suggesting no appreciable differences in fish health between lakes. The length-frequency distribution of non-YOY arctic charr collected from nearshore habitats in 2020 differed from the (2013) baseline study at Camp Lake (Table 3.12).

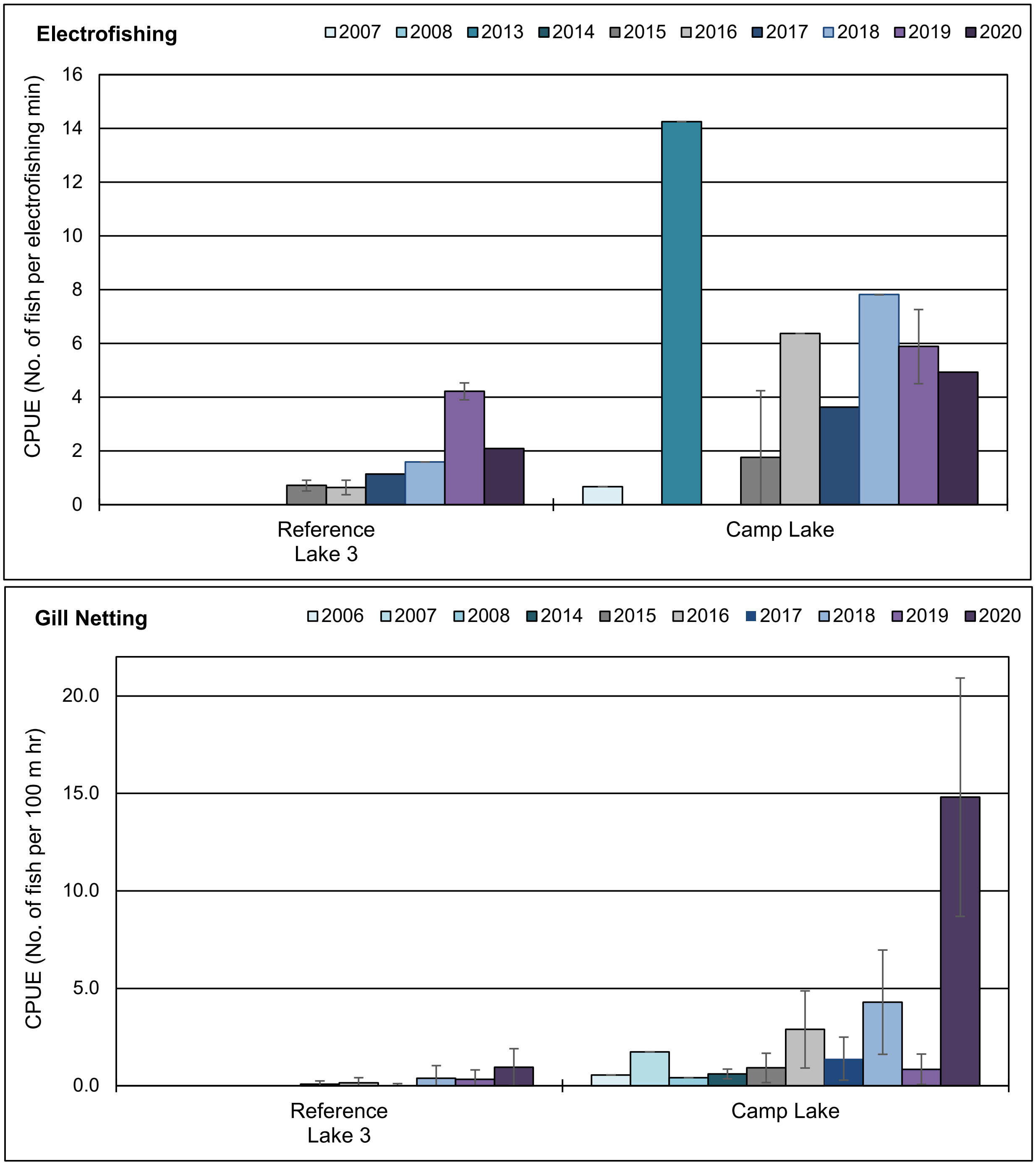


Figure 3.11: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, 2006 to 2020

Note: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015 to 2020) mine phases.

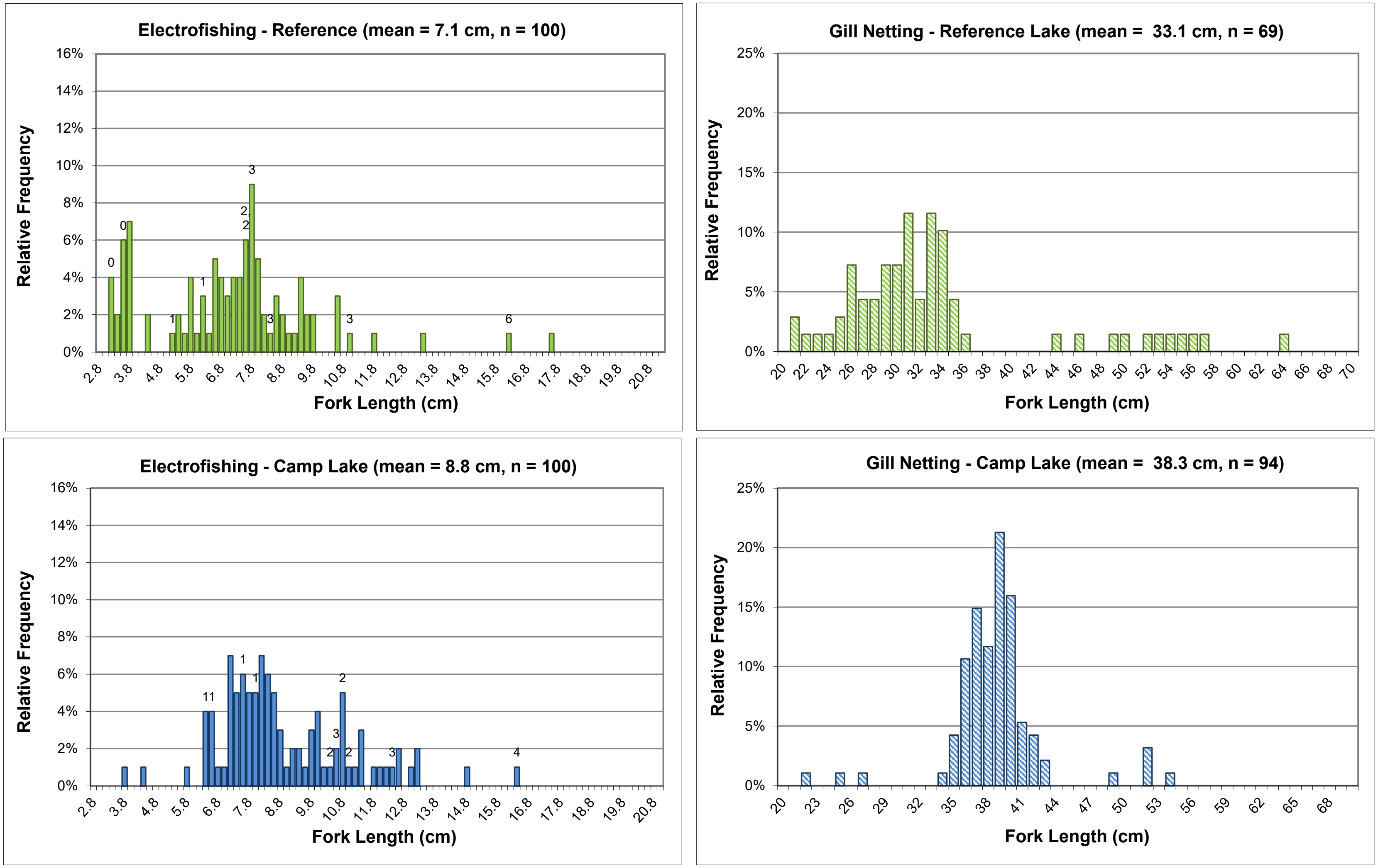


Figure 3.12: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2020

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

Table 3.12: Summary of Statistical Results for Arctic Charr Population Comparisons between Camp Lake and Reference Lake 3 from 2015 to 2020, and between Camp Lake Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]											
			versus Reference Lake 3						versus Camp Lake baseline period data [b]					
			2015	2016	2017	2018	2019	2020	2015	2016	2017	2018	2019	2020
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**
		Age	No	No	No	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	**Yes (+41%)**	No	**Yes (+17%)**	**Yes (+40%)**	**Yes (+10%)**	**Yes (+8%)**	**Yes (-15%)**	**Yes (-32%)**	**Yes (-35%)**	**Yes (-28%)**	No	**Yes (-22%)**
		Size (mean weight)	**Yes (+176%)**	No	**Yes (+51%)**	**Yes (+135%)**	**Yes (+29%)**	**Yes (+44%)**	**Yes (-42%)**	**Yes (-71%)**	**Yes (-74%)**	**Yes (-56%)**	No	**Yes (-52%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	No	**Yes (-6%)**	No	**Yes (-14%)**	**Yes (-7%)**	**Yes (+7%)**	**Yes (-6%)**	**Yes (-10%)**	**Yes (-10%)**	**Yes (-9%)**	**Yes (-11%)**	No
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	-	-	-	-	-	-	**Yes (+48%)**	**Yes (+58%)**	**Yes (+ 46%)**	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	**Yes (+10%)**	**Yes (+28%)**	**Yes (+24%)**	**Yes (+6%)**	No	**Yes (+12%)**	**Yes (+15%)**	**Yes (+17%)**	**Yes (+19%)**
		Size (mean weight)	-	-	-	**Yes (+46%)**	**Yes (+130%)**	**Yes (+129%)**	No	No	**Yes (+37%)**	**Yes (+46%)**	**Yes (+44%)**	**Yes (+47%)**
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+12%)**	**Yes (+6%)**	**Yes (+18%)**	No	**Yes (-3%)**	No	No	No	No

BOLD indicates a significant difference related to the comparison.

[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.

Similar to most previous years of mine operation, non-YOY arctic charr from Camp Lake were significantly shorter and lighter in 2020 than during baseline, but unlike most years, showed no difference in condition between 2020 and baseline (Table 3.12; Appendix Table G.7). Overall, the absence of consistent differences in non-YOY condition between Camp Lake and Reference Lake 3 since 2015, and occurrence of differences near ecologically meaningful thresholds in non-YOY condition at Camp Lake between mine operational and baseline studies, suggested no effects on the health of non-YOY arctic charr at Camp Lake since mine operations commenced in 2015.

Littoral/Profundal Arctic Charr

A total of 94 and 69 arctic charr were sampled from littoral/profundal habitat of Camp Lake and Reference Lake 3, respectively, in August 2020. The length-frequency distribution for littoral/ profundal arctic charr differed significantly between Camp Lake and Reference Lake 3, reflecting the occurrence of relatively larger fish at Camp Lake (Table 3.12; Figure 3.12). Littoral/profundal arctic charr from Camp Lake were significantly longer (24%) and heavier (129%) and had greater body condition (18%) than those captured at the reference lake (Table 3.12; Appendix Table G.6). The absolute magnitude of difference in condition between Camp Lake and Reference Lake 3 was greater than the CES_C of 10%, suggesting an ecologically significant difference. Larger body size and greater body condition of littoral/profundal arctic charr at Camp Lake relative to Reference Lake 3 were consistent with results in the two previous years (Table 3.12), suggesting an on-going difference between these populations.

A significant difference in length-frequency distribution of littoral/profundal arctic charr from Camp Lake was observed between 2020 and the combined baseline data set (i.e., 2006, 2007, and 2008 studies; Table 3.12). Although fork length and body weight were significantly greater for littoral/profundal arctic charr captured at Camp Lake in 2020 and most other years in which the mine was operational compared to the baseline period, no significant differences in body condition have generally been indicated since 2015 (Table 3.12). The occurrence of consistently larger littoral/profundal arctic charr at Camp Lake during mine operational years compared to the reference lake and Camp Lake baseline data, as well as greater condition and no differences in condition in littoral/profundal arctic charr compared to the reference lake and Camp Lake baseline data, respectively, collectively indicated no effects on the health of spawning-sized arctic charr at Camp Lake since mine operations commenced in 2015.

3.3.6 Effects Assessment and Recommendations

At Camp Lake, the following AEMP benchmarks were exceeded in 2020:

- Arsenic concentration in sediment was greater than the benchmark or 5.9 mg/kg at the single Camp Lake littoral monitoring station (JL0-02);
- Iron concentration in sediment was greater than the benchmark of 52,400 mg/kg at the single Camp Lake littoral monitoring station (JL0-02);
- Nickel concentration in sediment was greater than the benchmark of 72 mg/kg at the single Camp Lake littoral monitoring station (JL0-02); and,
- Arsenic, copper, iron, manganese, nickel, and phosphorus concentrations in sediment were above respective benchmarks at individual stations, but on average were below these benchmarks among the Camp Lake profundal stations.

Arsenic concentrations in sediment at the Camp Lake littoral station in 2020 were markedly higher than at the reference lake and compared to baseline, but showed no substantial change from 2015 to 2020 suggesting no on-going source of arsenic to sediment at this station. Although iron concentrations in sediment at the Camp Lake littoral station in 2020 were elevated compared to baseline, similar concentrations of iron were observed in littoral sediment at the reference lake suggesting naturally high background concentrations. Similarly, although nickel concentrations in sediment at the Camp Lake littoral station were elevated compared to concentrations at the reference lake in 2020, no substantial change in nickel concentrations had occurred between 2020 and baseline. At profundal habitat of Camp Lake in 2020, mean concentrations of arsenic, copper, iron, manganese, nickel, and phosphorus in sediment were all comparable to mean concentrations observed at the reference lake, as well as to Camp Lake baseline data, suggesting no changes over time. Thus, only the concentration of arsenic at the Camp Lake littoral station in 2020 was elevated compared to concentrations observed in sediment both at the reference lake in 2020 and at the Camp Lake littoral station at the time of baseline.

No AEMP water quality benchmarks were exceeded at Camp Lake during spring, summer, or fall sampling events in 2020.[10] In addition, no adverse effects on phytoplankton, benthic invertebrates, nor on fish (arctic charr) health were indicated at Camp Lake in 2020 based on comparisons to reference lake conditions and to Camp Lake baseline data. Considering these results within the Mary River Project AEMP Management Response Framework, the potential change in arsenic concentrations in sediment at the littoral station of Camp Lake warrants a low action response. Arsenic concentrations in water at CLT1, CLT2, and Camp Lake have consistently been near or below laboratory Method Detection Limits (MDL) since 2015, and thus

[10] The reported concentration of zinc at the Station JL0-07 surface was above the AEMP benchmark during the summer sampling event but this result appeared to be an anomaly based on an order of magnitude difference in concentration between this station and data reported for all other Camp Lake stations in summer 2020 (Appendix Table C.26).

the mine did not appear to be a source of arsenic to the Camp Lake system. Under the current AEMP, sediment chemistry sampling is conducted only at a single littoral station at Camp Lake (Baffinland 2015), and therefore the current AEMP does not adequately capture variability in sediment chemistry at littoral habitat of Camp Lake. Moreover, sediment chemistry sampling under the current AEMP is not always conducted at the same locations at which benthic invertebrate community sampling is conducted, precluding linkages to be drawn between sediment chemistry and biological responses. Accordingly, as per recommendations provided in the past by Minnow (2016b), a low action response of harmonizing lake sediment quality and benthic invertebrate monitoring stations, focusing primarily on littoral habitat, is recommended to improve the ability of the program to evaluate mine-related effects to biota and potentially allow linkages to be determined between metal concentrations in sediment and benthic invertebrate responses.

4 SHEARDOWN LAKE SYSTEM

4.1 Sheardown Lake Tributaries (SDLT1, SDLT12, and SDLT9)

4.1.1 Water Quality

Dissolved oxygen was consistently near full saturation at each of the Sheardown Lake tributaries during spring, summer, and fall sampling events in 2020 (Appendix Tables C.1 to C.3). Dissolved oxygen concentrations at Sheardown Lake Tributary 1 (SDLT1) and Sheardown Lake Tributary 9 (SDLT9) did not differ significantly from those at Unnamed Reference Creek during the August 2020 biological study (Figure 4.1). Although dissolved oxygen concentrations were significantly lower at Sheardown Lake Tributary 12 (SDLT12) than at Unnamed Reference Creek, the dissolved oxygen concentrations at SDLT12, and both other Sheardown Lake tributaries, were well above the WQG minimum for supporting sensitive life stages of cold-water biota (i.e., 9.5 mg/L) during the August 2020 biological study (Figure 4.1; Appendix Table C.31). *In situ* pH was significantly higher at SDLT1 and SDLT12 compared to Unnamed Reference Creek, whereas pH at SDLT9 did not differ significantly from that at the reference creek during the August 2020 biological study (Figure 4.1). Despite minor differences in pH among the Sheardown Lake tributaries, pH was consistently within WQG limits at each of the Sheardown Lake tributaries and thus slight dissimilarity in pH among areas was unlikely to be ecologically meaningful. Specific conductance at each of the Sheardown Lake tributaries was significantly higher than at Unnamed Reference Creek during the August 2020 biological study (Figure 4.1; Appendix Table C.32). Because specific conductance often serves as an indication of mine-associated influences on water quality (e.g., Environment Canada 2012), these observations suggested a potential mine-related influence on water quality of the SDLT1, SDLT9, and SDLT12 watercourses.

Sheardown Lake Tributary 1 (SDLT1) is the only tributary of the Sheardown Lake system at which routine water chemistry monitoring is conducted, with one monitoring station established in each of the upper and lower reaches of the tributary (i.e., Stations D1-05 and D1-00, respectively; Figure 2.2). Water chemistry of SDLT1 met AEMP benchmarks and WQG in spring, summer, and fall sampling events of 2020 except copper concentrations, which on average were elevated relative to both criteria for all sampling events (Table 4.1; Appendix Table C.33). Among parameters with established AEMP benchmarks, mean chloride, copper, nitrate, and sulphate concentrations were elevated at SDLT1 compared to the reference creeks during at least one sampling event in 2020, with nitrate and sulphate elevated by greatest factors (Table 4.1; Appendix Table C.35). For parameters without AEMP benchmarks, concentrations of total and dissolved molybdenum, potassium, and uranium, and concentrations of dissolved manganese,

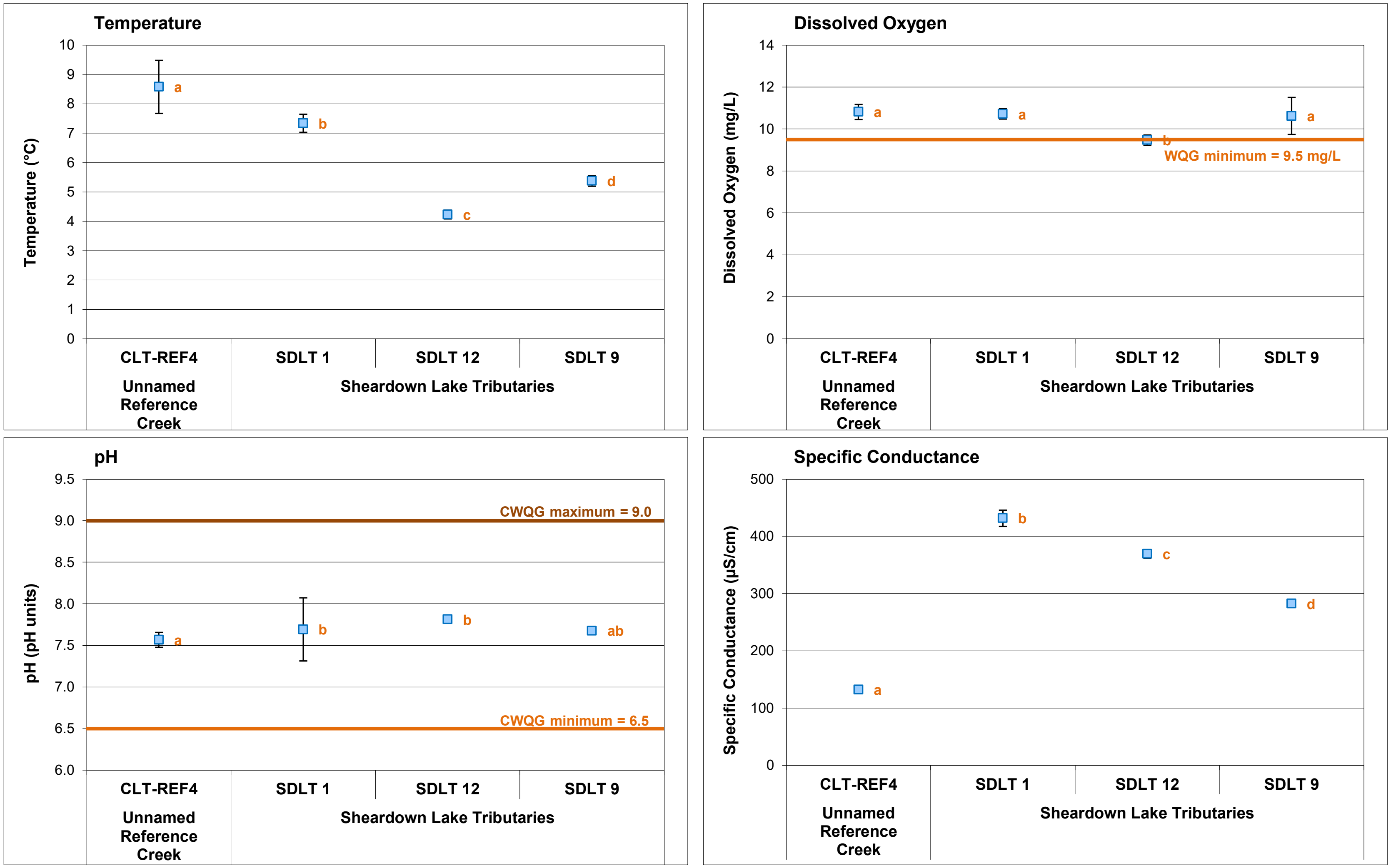


Figure 4.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Tributaries (SDLT) and Unnamed Reference Creek Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2020

Note: The same letter(s) next to data points indicate study area values do not differ significantly.

Table 4.1: Mean Water Chemistry at Sheardown Lake Tributary 1 (SDLT1) Monitoring Stations in Spring, Summer, and Fall, Mary River Project CREMP, 2020

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Bench-mark[b]	Reference Creek (n = 4)			Sheardown Lake Tributary 1 (n = 2)		
					Spring	Summer	Fall	Spring	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	55	134	175	217	338	309
	pH (lab)	pH	6.5 - 9.0	-	7.63	8.01	8.05	8.07	7.94	8.07
	Hardness (as $CaCO_3$)	mg/L	-	-	24	57	83	100	152	155
	Total Suspended Solids	mg/L	-	-	3.2	2.7	2	2.6	2	2
	Total Dissolved Solids	mg/L	-	-	85	85	99	122	198	172
	Turbidity	NTU	-	-	1.87	6.62	2.49	5.37	0.57	0.20
	Alkalinity (as $CaCO_3$)	mg/L	-	-	24	61	69	74	112	111
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.010	0.012	0.010	0.010	0.010	0.010
	Nitrate	mg/L	3	3	0.020	0.062	0.076	0.416	0.900	0.701
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.005	0.005	0.005
	Total Kjeldahl Nitrogen	mg/L	-	-	0.02	0.15	0.15	0.15	0.18	0.23
	Dissolved Organic Carbon	mg/L	-	-	1.9	3.4	2.3	3.4	3.8	3.1
	Total Organic Carbon	mg/L	-	-	2.2	3.1	2.1	4.8	4.9	3.4
	Total Phosphorus	mg/L	0.030$^{\alpha}$	-	0.0045	0.0065	0.0039	0.0058	0.0267	0.0030
	Phenols	mg/L	0.004$^{\alpha}$	-	0.0010	0.0010	0.0021	0.0018	0.0010	0.0013
Anions	Bromide (Br)	mg/L	-	-	0.1	0.1	0.1	0.1	0.1	0.1
	Chloride (Cl)	mg/L	120	120	1.2	4.1	7.1	4.2	8.0	8.4
	Sulphate (SO_4)	mg/L	218$^{\beta}$	218	1.3	5.5	9.2	26.3	52.1	32.3
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.078	***0.311***	0.059	0.095	0.011	0.011
	Antimony (Sb)	mg/L	0.020$^{\alpha}$	-	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.00010	0.0001275	0.0001	0.0001	0.0001	0.0001
	Barium (Ba)	mg/L	-	-	0.0036	0.0095	0.0103	0.0107	0.0151	0.0147
	Beryllium (Be)	mg/L	0.011$^{\alpha}$	-	0.0005	0.0004	0.0005	0.0005	0.0005	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0003875	0.0005	0.0005	0.0005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.010	0.010	0.013	0.016	0.016
	Cadmium (Cd)	mg/L	0.00012	0.00008	0.00001	0.000009	0.000010	0.000026	0.000025	0.000025
	Calcium (Ca)	mg/L	-	-	4.9	11.8	16.5	17.5	27.1	27.5
	Chromium (Cr)	mg/L	0.0089	0.00856	0.00050	0.00082	0.0005	0.0005	0.0005	0.0005
	Cobalt (Co)	mg/L	0.0009$^{\alpha}$	0.004	0.00010	0.0001575	0.0001	0.00011	0.00011	0.0001
	Copper (Cu)	mg/L	0.002	0.0022	0.0007	0.0011	0.0010	***0.0029***	***0.0024***	***0.0023***
	Iron (Fe)	mg/L	0.30	0.326	0.077	0.243	0.066	0.131	0.089	0.061
	Lead (Pb)	mg/L	0.001	0.001	0.00011	0.00023	0.00009	0.00023	0.00005	0.00005
	Lithium (Li)	mg/L	-	-	0.0010	0.0011	0.0010	0.0019	0.0020	0.0019
	Magnesium (Mg)	mg/L	-	-	2.86	6.7	9.6	13.0	20.7	20.4
	Manganese (Mn)	mg/L	0.935$^{\beta}$	-	0.00136	0.00300	0.00102	0.00462	0.00539	0.00293
	Mercury (Hg)	mg/L	0.000026	-	0.000005	0.000005	0.000005	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00015	0.00045	0.00057	0.00351	0.00390	0.00495
	Nickel (Ni)	mg/L	0.025	0.025	0.0005	0.0007	0.0006	0.0015	0.0013	0.0012
	Potassium (K)	mg/L	-	-	0.45	0.93	1.04	2.44	2.84	3.06
	Selenium (Se)	mg/L	0.001	-	0.001	0.0007625	0.001	0.001	0.001	0.001
	Silicon (Si)	mg/L	-	-	0.62	1.25	0.87	1.39	1.42	1.43
	Silver (Ag)	mg/L	0.00025	0.0001	0.00001	0.00002	0.00001	0.00001	0.00001	0.00001
	Sodium (Na)	mg/L	-	-	0.83	2.76	3.97	2.46	3.90	4.20
	Strontium (Sr)	mg/L	-	-	0.0049	0.0139	0.0185	0.0142	0.0200	0.0187
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00010	0.00008	0.00010	0.00010	0.00010	0.00010
	Tin (Sn)	mg/L	-	-	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.0108	0.0241	0.0100	0.0100	0.0100	0.0100
	Uranium (U)	mg/L	0.015	-	0.00045	0.00405	0.00737	0.00371	0.00789	0.01430
	Vanadium (V)	mg/L	0.006$^{\alpha}$	0.006	0.00100	0.00115	0.00100	0.00100	0.00100	0.00100
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.003	0.003	0.0063	0.00565	0.00485

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

[a] Canadian Water Quality Guideline except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]) and β (British Columbia Water Quality Guideline [BCWQG]). See Table 2.3 for information regarding WQG criteria.

[b] AEMP Water Quality Benchmarks developed by Intrinsik (2013) using baseline water quality data adopted from Camp Lake Tributaries.

were moderately (i.e., 5- to 10-fold) to highly (i.e., ≥10-fold) elevated at SDLT1 compared to the reference creeks in at least one of the spring, summer, or fall 2020 sampling events (Appendix Table C.35). Highest elevation in parameter concentrations typically occurred during the spring sampling event (Appendix Tables C.34 and C.35). In addition, higher parameter concentrations were generally observed at lower SDLT1 compared to upper SDLT1, suggesting that additional inputs of metals to SDLT1 occurred with distance downstream of the headwaters at the main mine camp (Appendix Table C.34).

Despite total copper concentrations above the AEMP benchmark and WQG at SDLT1 in 2020, the concentrations of copper during each seasonal sampling event were comparable to those reported at SDLT1 during baseline (Appendix Figure C.11; Appendix Table C.34), suggesting that copper concentrations were naturally high within this tributary prior to commencement of mine operations in 2015. Among the other parameters with established AEMP benchmarks, nitrate and sulphate concentrations were most consistently elevated at SDLT1 in 2020 (and other years of mine operation) compared to baseline (Appendix Figure C.11; Appendix Table C.33). For parameters without AEMP benchmarks, sodium and uranium were the only parameters with elevated concentrations for more than one of the three seasonal sampling events at SDLT1 in 2020 compared to baseline (Appendix Table C.33; Appendix Figure C.11). Overall, the key mine-related influences on water quality of SDLT1 based on comparisons to the reference creeks and baseline included elevated specific conductance and concentrations of molybdenum, nitrate, sulphate, and uranium, although none of the latter four parameters were observed at concentrations above applicable AEMP benchmarks and/or WQG.

4.1.2 Sediment Quality

Sediment from SDLT1 was visually characterized as reddish-brown silt, whereas sediment from SDLT12 was mainly coarse sand and gravel, and sediment from SDLT9 was medium-sized coarse sand (Appendix Table D.18). Natural in-stream substrate at tributaries SDLT1 and SDLT12 is composed almost entirely of cobble and boulder material, but smaller particles tend to deposit interstitially in slow flowing areas and along the shoreline at both tributaries (Minnow 2018). In contrast, small cobble is the primary substrate type at SDLT9, but sand can constitute as much as 5 to 10% of the surficial bed material in this tributary (Minnow 2018). Sediment TOC content was low (i.e., <1%) in samples collected from SDLT1 and SDLT12, but slightly higher (0.8 to 4.0%) in samples from SDLT9 (Appendix Tables D.19, D.21, and D.22). Sediment TOC content was 6-fold higher (on average) at SDLT1 and SDLT12 than observed at lotic reference areas, but 17-fold higher at SDLT9 (Table 4.2; Appendix Table D.20). This suggested a more depositional environment and/or greater suspended sediment loads at the three Sheardown Lake tributaries compared to reference conditions.

Table 4.2: Sediment Total Organic Carbon and Metal Concentrations at Sheardown Lake Tributaries (SDLT1, 12, and 9) and Applicable Reference Creek and River Sediment Monitoring Stations, Mary River Project CREMP, August 2020

Parameter	Units	SQG[a]	Lotic Reference Stations	Sheardown Lake Tributaries		
			Unnamed Reference Creek (REFCRK; n = 3)	Sheardown Trib 1 SDLT1 (n = 3)	Sheardown Trib 12 SDLT12 (n = 3)	Sheardown Trib 9 SDLT9 (n = 3)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC	%	10[α]	0.12 ± 0.035	0.76 ± 0.19	0.75 ± 0.19	2.03 ± 1.69
Aluminum (Al)	mg/kg	-	584 ± 185	15,967 ± 1,601	8,153 ± 807	6,997 ± 3,386
Antimony (Sb)	mg/kg	-	<0.10 ± 0	0.17 ± 0.057	0.22 ± 0.023	0.11 ± 0.012
Arsenic (As)	mg/kg	17	0.22 ± 0.11	3.29 ± 1.13	5.83 ± 1.00	1.65 ± 1.15
Barium (Ba)	mg/kg	-	2.72 ± 0.722	63.6 ± 13.2	17.1 ± 6.11	32.2 ± 19.4
Beryllium (Be)	mg/kg	-	<0.10 ± 0	0.65 ± 0.042	0.68 ± 0.020	0.32 ± 0.20
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	0.48 ± 0.16	0.25 ± 0.040	0.22 ± 0.035
Boron (B)	mg/kg	-	<5.0 ± 0	6.8 ± 1.1	5.5 ± 0.68	8.50 ± 6.06
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	0.143 ± 0.0261	0.051 ± 0.011	0.064 ± 0.040
Calcium (Ca)	mg/kg	-	494 ± 249	3,790 ± 1,060	1,223 ± 643	2,920 ± 1,974
Chromium (Cr)	mg/kg	90	7.79 ± 5.39	33.7 ± 2.25	30.6 ± 0.643	22.7 ± 10.3
Cobalt (Co)	mg/kg	-	0.953 ± 0.558	13.3 ± 1.37	14.6 ± 0.751	6.24 ± 3.07
Copper (Cu)	mg/kg	110[α]	1.21 ± 0.899	24.0 ± 2.16	19.4 ± 0.656	15.7 ± 11.3
Iron (Fe)	mg/kg	40,000[α]	12,493 ± 9,700	152,667 ± 33,546	345,000 ± 53,731	60,133 ± 40,945
Lead (Pb)	mg/kg	91	1.49 ± 0.546	12.4 ± 0.666	5.48 ± 1.05	5.48 ± 3.02
Lithium (Li)	mg/kg	-	<2.0 ± 0	17.8 ± 1.20	8.17 ± 1.69	7.67 ± 3.50
Magnesium (Mg)	mg/kg	-	444 ± 165	14,000 ± 2,166	5,790 ± 957	6,017 ± 3,033
Manganese (Mn)	mg/kg	1,100[α,β]	27.4 ± 14.6	681 ± 51	810 ± 103.9	294 ± 175
Mercury (Hg)	mg/kg	0.486	<0.0050 ± 0	0.0065 ± 0.00067	0.0052 ± 0.00029	0.0120 ± 0.0069
Molybdenum (Mo)	mg/kg	-	<0.10 ± 0	5.73 ± 1.33	3.82 ± 0.248	1.82 ± 1.27
Nickel (Ni)	mg/kg	75[α,β]	1.76 ± 0.920	33.1 ± 1.14	36.4 ± 1.70	24.5 ± 14.8
Phosphorus (P)	mg/kg	2,000[α]	167 ± 98	344 ± 50	252 ± 51	428 ± 155
Potassium (K)	mg/kg	-	133 ± 42	5,817 ± 801	800 ± 385	1,597 ± 827
Selenium (Se)	mg/kg	-	<0.20 ± 0	0.21 ± 0.012	0.27 ± 0.061	0.27 ± 0.12
Silver (Ag)	mg/kg	-	<0.10 ± 0	0.12 ± 0.021	0.10 ± 0	<0.10 ± 0
Sodium (Na)	mg/kg	-	<50 ± 0	126 ± 29	<50 ± 0	63 ± 22
Strontium (Sr)	mg/kg	-	2.00 ± 0.544	3.9 ± 0.49	2.1 ± 0.56	4.06 ± 1.73
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	<0.050 ± 0	0.271 ± 0.0278	0.068 ± 0.017	0.153 ± 0.0785
Tin (Sn)	mg/kg	-	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0
Titanium (Ti)	mg/kg	-	83 ± 47	821 ± 134	218 ± 69	485 ± 139
Uranium (U)	mg/kg	-	0.479 ± 0.247	4.35 ± 1.408	2.33 ± 0.395	1.28 ± 0.946
Vanadium (V)	mg/kg	-	16.8 ± 12.8	27.7 ± 3.64	15.6 ± 1.15	18.0 ± 8.10
Zinc (Zn)	mg/kg	315	3.0 ± 1.2	75.8 ± 8.90	25.2 ± 5.67	23.1 ± 12.5
Zirconium (Zr)	mg/kg	-	2.1 ± 0.91	9.0 ± 1.9	3.5 ± 0.38	2.9 ± 2.1

Indicates parameter concentration above SQG.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV 2020).

Mean concentrations of metals in sediment from both SDLT1 and SDLT12 were generally elevated compared to mean concentrations at lotic reference areas (Table 4.2; Appendix Table D.20). In particular, concentrations of aluminum, arsenic, barium, cobalt, copper, iron, magnesium, manganese, molybdenum, nickel, potassium, and zinc were highly elevated (i.e., ≥10-fold higher) in sediment from one or both of these tributaries compared to the lotic reference areas (Appendix Table D.20). In part, elevated metal concentrations in sediment from SDLT1 and SDLT12 may reflect finer substrate sizes and more depositional features of these tributaries compared to that observed at the lotic reference areas. On average, metal concentrations in sediment from SDLT1 and SDLT12 were below applicable SQG except for iron, which occurred at mean concentrations approximately four- and nine-times higher than the SQG, respectively, at these tributaries (Table 4.2; Appendix Tables D.19 and D.21). Sediment from SDLT9 had highly elevated concentrations of molybdenum relative to mean concentrations at the lotic reference areas, whereas mean concentrations of several other metals including aluminum, arsenic, barium, copper, iron, magnesium, manganese, nickel, and potassium were only moderately higher (i.e., 5-fold to 10-fold) at SDLT9 (Appendix Table D.20). Similar to the other Sheardown Lake tributaries, concentrations of all metals except iron (which was only 1.5 times greater than the SQG, on average) were well below SQG at SDLT9 (Table 4.2; Appendix Table D.22).

4.1.3 Phytoplankton

Among the Sheardown Lake tributaries, phytoplankton (chlorophyll-a) monitoring is conducted only at SDLT1 as part of the Mary River Project CREMP (Table 2.1). Chlorophyll-a concentrations were lower at upper SDLT1 (Station D1 05) compared to near the creek mouth (Station D1 00) during each of the spring, summer, and fall sampling events in 2020 (Figure 4.2). Nitrate, phosphorus, and TKN concentrations were consistently the same or higher near the mouth of SDLT1 in 2020 (Appendix Table C.34), and thus higher chlorophyll-a concentrations near the mouth was in line with typical responses of phytoplankton to higher nutrient concentrations. Chlorophyll-a concentrations at SDLT1 were within the range of variability observed among reference creeks in spring and summer sampling events, but were considerably lower compared to the reference creeks in the summer sampling event (Figure 4.2). Although the latter may have reflected a mine-related influence on phytoplankton abundance occurring seasonally at lower SDLT1, chlorophyll-a concentrations were unusually high at the reference creeks in the summer of 2020 compared to previous years, and thus may not reflect the norm. For all sampling events in 2020, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L at both of the SDLT1 monitoring stations (Figure 4.2). Similar to the reference creeks and Camp Lake tributaries, chlorophyll-a concentrations at SDLT1 were suggestive of oligotrophic, low productivity conditions based on Dodds et al (1998) trophic status

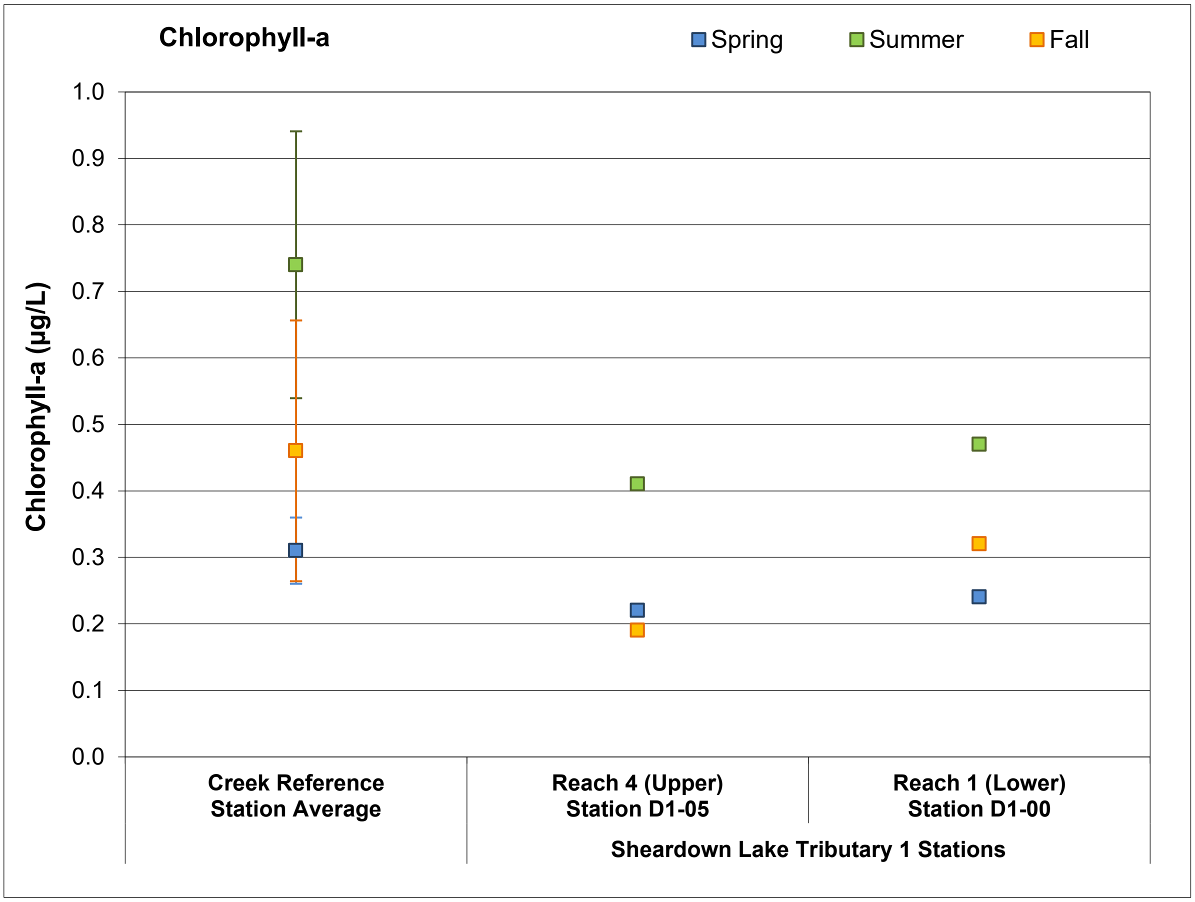


Figure 4.2: Chlorophyll-a Concentrations at Sheardown Lake Tributary 1 Phytoplankton Monitoring Stations, Mary River Project CREMP, 2020

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

classification for stream environments (i.e., chlorophyll-a concentration <10 µg/L). Relatively low chlorophyll-a concentrations at SDLT1 stations in 2020 were also consistent with an oligotrophic categorization using CWQG (CCME 2020) categorization based on aqueous phosphorus concentrations (i.e., concentrations below 10 µg/L; Table 4.1; Appendix Table C.33).

Chlorophyll-a concentrations at SDLT1 stations in fall 2020 were similar to those during the baseline period (Figure 4.3). In addition, no consistent directional changes in chlorophyll-a concentrations were shown at the SDLT1 stations during fall sampling events over the mine baseline (2005 to 2013), construction (2014), and operational (2015 to 2020) periods (Figure 4.3). These results suggested no adverse mine-related influences on phytoplankton productivity at SDLT1 over the past six years of mine operation.

4.1.4 Benthic Invertebrate Community

4.1.4.1 Sheardown Lake Tributary 1 (SDLT1)

The benthic invertebrate community at the lower reach of SDLT1, near the outlet to Sheardown Lake NW, showed significantly lower evenness and significant differences in composition (as indicated by Bray-Curtis Index) compared to Unnamed Reference Creek in 2020 (Table 4.3; Appendix Table F.29). Marked differences in community composition between SDLT1 and the reference creek included significantly higher relative abundance of Nemata and Chironomidae, and significantly lower relative abundance of Hydracarina, Ostracoda, and Simuliidae at SDLT1 (Table 4.3). However, an ecologically significant higher relative abundance of metal-sensitive Chironomidae occurred at SDLT1 compared to Unnamed Reference Creek (Table 4.3), suggesting that metals were not biologically available and/or were not a large contributor to community composition differences between SDLT1 and the reference creek. This result was consistent with concentrations of all metals below WQG at SDLT1, except copper which was slightly above the WQG, in 2020 (Table 4.1). Ecologically significant higher relative abundance of the shredder FFG was indicated at SDLT1 compared to Unnamed Reference Creek, suggesting a greater amount of in-stream vegetation and/or organic debris at SDLT1. In addition, ecologically significant higher relative abundance of the burrower HPG was shown at SDLT1 compared to the reference creek (Table 4.3), possibly indicating physical habitat alteration associated with sedimentation had affected benthic invertebrate community composition at SDLT1 relative to reference conditions.

No consistent ecologically significant differences in density, richness, or evenness were indicated at SDLT1 over years of mine operation (2015 to 2020) compared to baseline (Appendix Figure F.7; Appendix Table F.30). Similarly, no ecologically significant differences in the relative abundance of any dominant taxonomic groups were consistently indicated over years

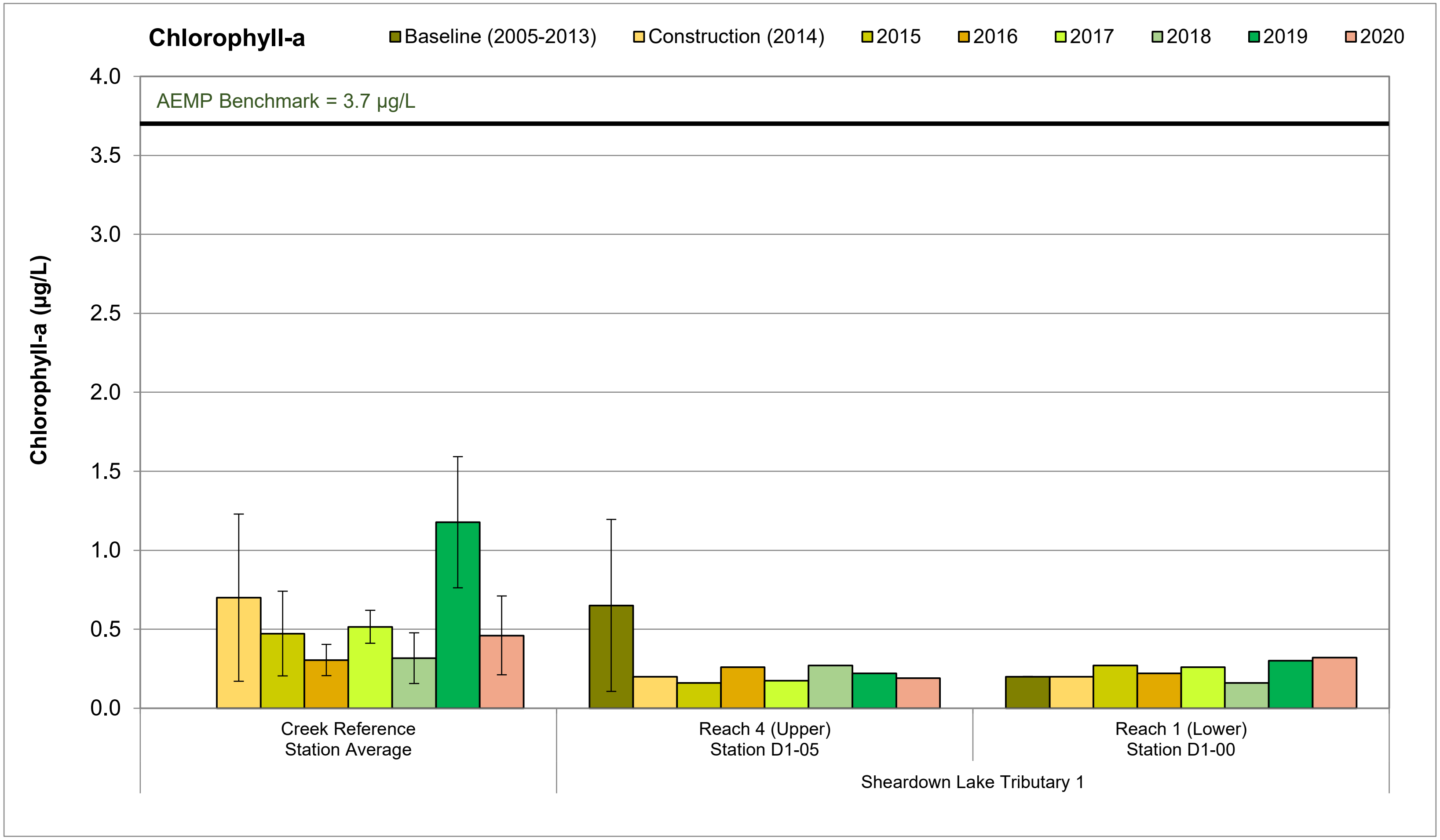


Figure 4.3: Temporal Comparison of Chlorophyll-a Concentrations at Sheardown Lake Tributary 1 for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2020) Periods in the Fall, Mary River Project CREMP

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

Table 4.3: Benthic Invertebrate Community Metric Statistical Comparison Results among the Sheardown Lake Tributaries and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2020

Metric	Overall 4-Area Comparison				Pair-wise, *post hoc* comparisons				
	Statistical Test[a]	Data Transformation	Significant Difference Among Areas?	P-value	Study Area	Mean	Standard Deviation (SD)	Magnitude of Difference (Ref SD)	Different from Reference Creek?
Density (No. per m^2)	ANOVA	log10	YES	0.050	Reference Creek	713	296	-	-
					SDLT1	679	625	-0.1	NO
					SDLT12	1,318	26	2.0	NO
					SDLT9	1,601	841	3.0	YES
Richness (No. of Taxa)	ANOVA	none	NO	0.350	Reference Creek	16.0	4.4	-	-
					SDLT1	13.4	2.9	-0.6	NO
					SDLT12	17.3	4.2	0.3	NO
					SDLT9	18.0	4.7	0.5	NO
Simpson's Evenness	ANOVA	none	YES	0.019	Reference Creek	0.840	0.043	-	-
					SDLT1	0.722	0.097	-2.7	YES
					SDLT12	0.832	0.063	-0.2	NO
					SDLT9	0.855	0.024	0.3	NO
Nemata (% of community)	ANOVA	log10(x+1)	YES	0.016	Reference Creek	0.7	1.3	-	-
					SDLT1	6.2	3.6	4.3	YES
					SDLT12	5.4	1.7	3.7	YES
					SDLT9	3.1	2.3	1.8	NO
Hydracarina (% of community)	K-W	rank	YES	0.004	Reference Creek	4.5	3.7	-	-
					SDLT1	1.1	0.9	-0.9	YES
					SDLT12	0.2	0.2	-1.2	YES
					SDLT9	4.2	1.8	-0.1	NO
Ostracoda (% of community)	K-W	rank	YES	0.001	Reference Creek	30.6	11.7	-	-
					SDLT1	0.0	0.1	-2.6	YES
					SDLT12	0.6	0.1	-2.6	YES
					SDLT9	8.2	2.6	-1.9	NO
Chironomidae (% of community)	ANOVA	none	YES	<0.001	Reference Creek	48.3	12.9	-	-
					SDLT1	85.0	6.9	2.8	YES
					SDLT12	88.6	2.1	3.1	YES
					SDLT9	70.3	4.3	1.7	YES
Metal Sensitive Chironomids (% of community)	ANOVA	log10(x+1)	YES	0.002	Reference Creek	0.8	1.2	-	-
					SDLT1	10.5	5.3	7.9	YES
					SDLT12	0.5	0.4	-0.3	NO
					SDLT9	3.2	3.6	1.9	NO
Simuliidae (% of community)	ANOVA	log10(x+1)	YES	0.020	Reference Creek	9.8	8.6	-	-
					SDLT1	0.0	0.0	-1.1	YES
					SDLT12	0.0	0.0	-1.1	YES
					SDLT9	0.5	0.9	-1.1	YES
Tipulidae (% of community)	ANOVA	log10(x+1)	NO	0.218	Reference Creek	1.5	2.3	-	-
					SDLT1	3.9	2.5	1.0	NO
					SDLT12	1.5	0.7	0.0	NO
					SDLT9	3.1	1.3	0.7	NO
Collector-Gatherer FFG (% of community)	ANOVA	log10	YES	0.002	Reference Creek	80.7	8.8	-	-
					SDLT1	79.4	7.0	-0.1	NO
					SDLT12	82.2	1.4	0.2	NO
					SDLT9	63.1	6.9	-2.0	YES
Shredder FFG (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	2.8	2.7	-	-
					SDLT1	15.8	6.4	4.9	YES
					SDLT12	16.9	1.8	5.3	YES
					SDLT9	29.8	6.8	10.2	YES
Clinger HPG (% of community)	ANOVA	log10	YES	0.010	Reference Creek	15.8	7.7	-	-
					SDLT1	15.1	5.7	-0.1	NO
					SDLT12	16.3	1.9	0.1	NO
					SDLT9	33.4	8.1	2.3	YES
Sprawler HPG (% of community)	ANOVA	none	YES	<0.001	Reference Creek	79.4	6.6	-	-
					SDLT1	72.1	6.0	-1.1	NO
					SDLT12	73.8	0.3	-0.8	NO
					SDLT9	53.0	9.9	-4.0	YES
Burrower FFG (% of community)	ANOVA	log10	YES	0.032	Reference Creek	4.8	3.3	-	-
					SDLT1	12.5	5.8	2.3	YES
					SDLT12	9.9	1.6	1.5	YES
					SDLT9	8.1	2.8	1.0	YES

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] Statistical tests include Analysis of Variance (ANOVA) followed by Tukey's Honestly Significant Difference (HSD) post hoc tests, or Kruskal-Wallis H-test (K-W) followed by Mann-Whitney U-test (M-W).

of mine operation compared to both years of baseline at SDLT1 (Appendix Table F.30).[11] However, consistent differences in FFG composition that included significantly higher relative abundance of collector-gatherers and significantly lower relative abundance of filterers and shredders at SDLT1 beginning in 2018 and 2019, respectively, compared to baseline potentially indicated a shift in the benthic invertebrate food base during more recent years of mine operation. Interestingly, the relative abundance of these FFG at SDLT1 since 2018 has more closely reflected the FFG composition at the reference creek, suggesting that the benthic invertebrate food base at SDLT1 more recently reflects the reference condition than was observed during baseline.

4.1.4.2 Sheardown Lake Tributary 12 (SDLT12)

Benthic invertebrate density, richness, and evenness at SDLT12 did not differ significantly compared to Unnamed Reference Creek, but benthic invertebrate community compositional differences were indicated between SDLT12 and the reference creek in 2020 based on significantly differing Bray-Curtis Index (Table 4.3; Appendix Table F.29). Similar to SDLT1, the differences in community composition included significantly higher relative abundance of Nemata, Chironomidae, shredder FFG, and burrower HPG, and significantly lower relative abundance of Hydracarina, Ostracoda, and Simuliidae, at SDLT12 compared to Unnamed Reference Creek (Table 4.3). However, no significant differences in the relative abundance of metal-sensitive Chironomidae were indicated at SDLT12 compared to the reference creek in 2020 (Table 4.3). In addition, no ecologically significant differences in benthic invertebrate density, richness, evenness, and relative abundance of any dominant taxonomic groups or FFG were consistently indicated at SDLT12 over years of mine operation compared to baseline (Appendix Table F.32; Appendix Figure F.8). The dominant taxon at SDLT12 was the midge *Diplocladius*, which is characteristic of small, cool, slow-flowing or still streams (Armitage et al. 1995; Namayandeh et al 2016). Much lower densities of this midge were present at the reference creek (compare Appendix Tables F.4 and F.31), which suggested that the existence of significantly slower water velocity at SDLT12 compared to Unnamed Reference Creek (Appendix Table F.26) likely accounted for the differences in benthic invertebrate community composition shown between these creeks. Overall, no adverse influences of the mine on benthic invertebrate community structure or food resources were indicated at SDLT12 in 2020 and since the commencement of commercial mine operations in 2015.

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[11] Although the relative abundance of Tipulidae at SDLT1 was consistently significantly lower in years of mine operation compared to baseline data collected in 2008, no significant difference in the relative abundance of this group was indicated in years of mine operation relative to baseline data collected in 2013. In addition, the relative abundance of Tipulidae at SDLT1 from 2016 to 2020 was comparable to that shown at the reference creek in 2020, suggesting that the relative abundance of this group during baseline in 2008 was unusually high.

4.1.4.3 Sheardown Lake Tributary 9 (SDLT9)

Benthic invertebrate density was significantly higher at SDLT9 compared to Unnamed Reference Creek, the magnitude of which was outside of the CES_{BIC} of ±2 SD_{REF} (Table 4.3). In addition, richness, evenness, and relative abundance of all dominant groups, including metal-sensitive Chironomidae, did not differ significantly between SDLT9 and the reference creek at magnitudes considered ecologically meaningful (Table 4.3). Ecologically significant lower relative abundance of collector-gatherer FFG and sprawler HPG, and ecologically significant higher relative abundance of shredder FFG and clinger HPG, were indicated at SDLT9 compared to the reference creek in 2020 (Table 4.3). However, the relative abundance of all FFG, as well as benthic invertebrate density, richness, evenness, and relative abundance of all dominant taxonomic groups, showed no consistent differences over years of mine operation compared to baseline at SDLT9 (Appendix Table F.34; Appendix Figure F.9), indicating that the differences in FFG between SDLT9 and the reference creek in 2020 reflected natural phenomena. For instance, a higher relative abundance of the shredder FFG was consistent with field observations of greater amounts of rooted in-stream vegetation and organic debris, the primary food source for shredders, at SDLT9 compared to the reference creek (Appendix Table F.24). In turn, this suggested that differing amounts and/or types of organic material accounted for the differences in benthic invertebrate community composition between SDLT9 and the reference creek. Overall, no adverse influences of the mine on the benthic invertebrate community structure of SDLT12 were indicated since the commencement of commercial mine operations in 2015, including in 2020.

4.1.5 Effects Assessment and Recommendations

At the SDLT1, the following AEMP benchmarks were exceeded in 2020:

- Aqueous total copper concentration greater than the benchmark of 0.0022 mg/L in spring, summer, and fall monitoring events (i.e., mean of 0.0029 mg/L, 0.0024 mg/L, and 0.0023 mg/L, respectively).

Although copper concentrations at SDLT1 were, on average, slightly higher than at the reference creeks in 2020, the concentration of copper in 2020 was closely comparable to those reported during baseline suggesting natural elevation. Given the proximity to mine operations and evidence of sedimentation, a mine-related source of copper to SDLT1 seems likely, but because no elevation in copper concentrations was indicated at SDLT1 from 2015 to 2020 compared to baseline conditions, copper concentrations at SDLT1 may just naturally be similar to the AEMP benchmark. Biological monitoring conducted at SDLT1 in 2020 indicated no adverse effects to phytoplankton or benthic invertebrates, potentially reflecting copper concentrations at, or just marginally above, the WQG. Because no adverse effects to biota were associated with copper concentrations above the AEMP benchmark at SDLT1, a low action response to identify the likely source(s) of copper to the system is recommended to meet obligations under the AEMP Management Response Framework.

Benthic invertebrate community monitoring at SDLT12 and SDLT9 indicated no adverse influences of the mine on the benthic invertebrate community structure of either watercourse since the commencement of commercial mine operations in 2015, including in 2020. Under the AEMP Management Response Framework, no adjustment to the existing AEMP need be applied at SDLT12 and SDLT9 for the next monitoring program due to the absence of any mine-related changes shown in the benthic invertebrate community shown between the mine-operational period and baseline at these tributaries. However, because routine water quality monitoring is not conducted at either SDLT12 or SDLT9 under the current AEMP (Baffinland 2015), linkages between water chemistry and biological responses are not possible. Therefore, it is recommended that a water quality monitoring station be established at each of these watercourses and that the same AEMP water quality monitoring program implemented at SDLT1 be conducted at SDLT12 and SDLT9 in the future in order to provide water chemistry data to support the interpretation of biological data.

4.2 Sheardown Lake Northwest (DLO-1)

4.2.1 Water Quality

Water quality profiles of *in situ* water temperature, dissolved oxygen, pH, and specific conductance conducted at Sheardown Lake NW in 2020 showed no substantial station-to-station differences during any of the winter, summer, or fall sampling events (Appendix Figures C.12 to C.15). A warmer surface layer was indicated at Sheardown Lake NW during the summer sampling event in 2020 that extended to a depth of approximately 6 metres, but no complete thermal stratification developed in summer or during the winter or fall sampling events (Figure 4.4). Thermal changes with depth at Sheardown Lake NW were very similar to the patterns shown at Reference Lake 3 during the summer and fall sampling events in 2020 (Figure 4.4). The average water temperature at the bottom of the water column at Sheardown Lake NW littoral stations was significantly cooler than at Reference Lake 3 during the August 2020 biological study, but no differences in bottom water temperature were indicated between lakes at profundal sampling depths (Figure 4.5). Dissolved oxygen profiles at Sheardown Lake NW showed a distinct oxycline in winter and from depths of 4 to 11 metres in summer, but no oxycline was evident in fall indicating well mixed conditions (Figure 4.4). The general pattern in dissolved oxygen profiles at Sheardown Lake NW in summer and fall sampling events were similar to those observed at Reference Lake 3 in 2020 (Figure 4.4). Dissolved oxygen concentrations near the bottom of the water column were significantly lower at Sheardown Lake NW littoral and profundal

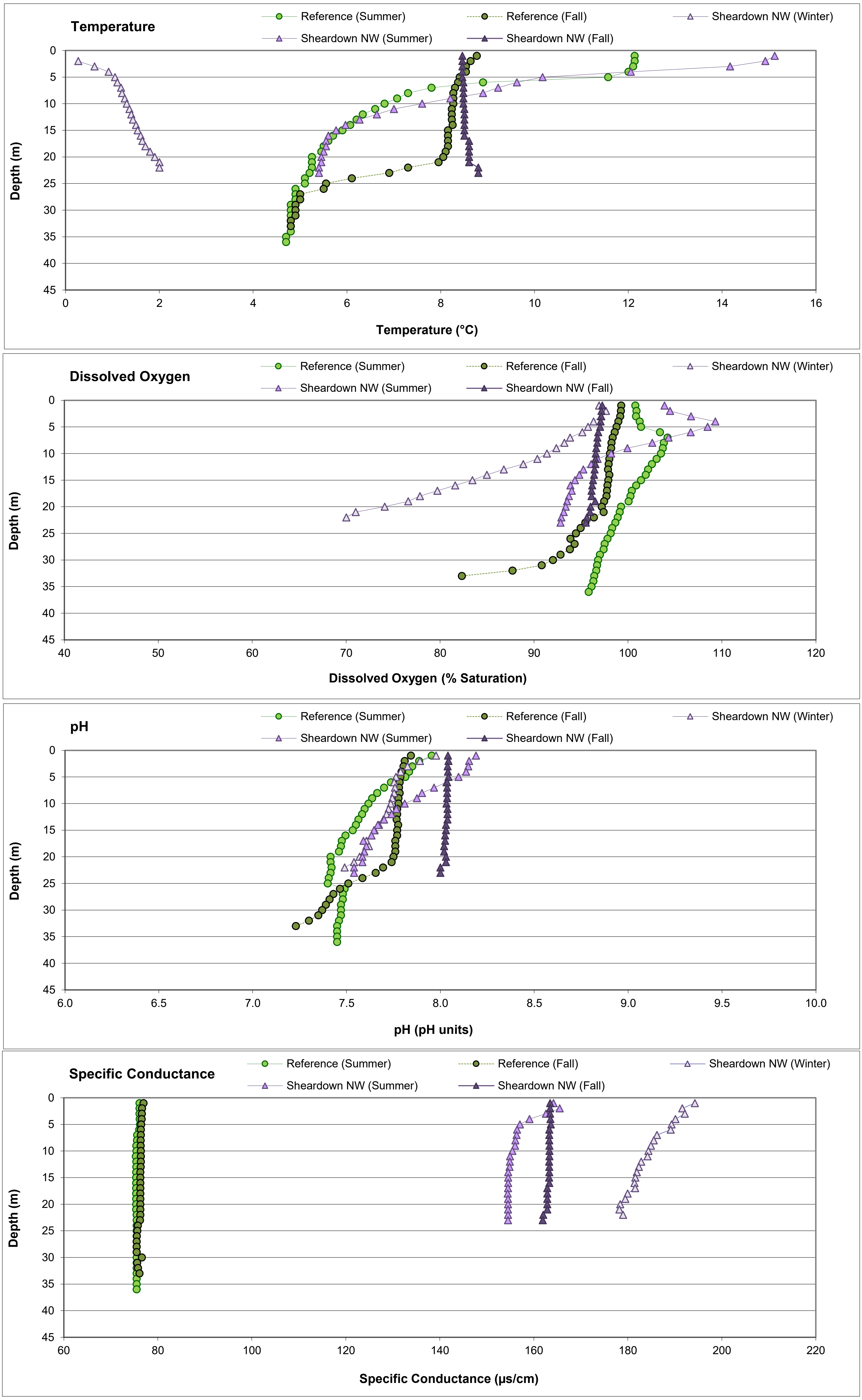


Figure 4.4: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake NW (DLO-01) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

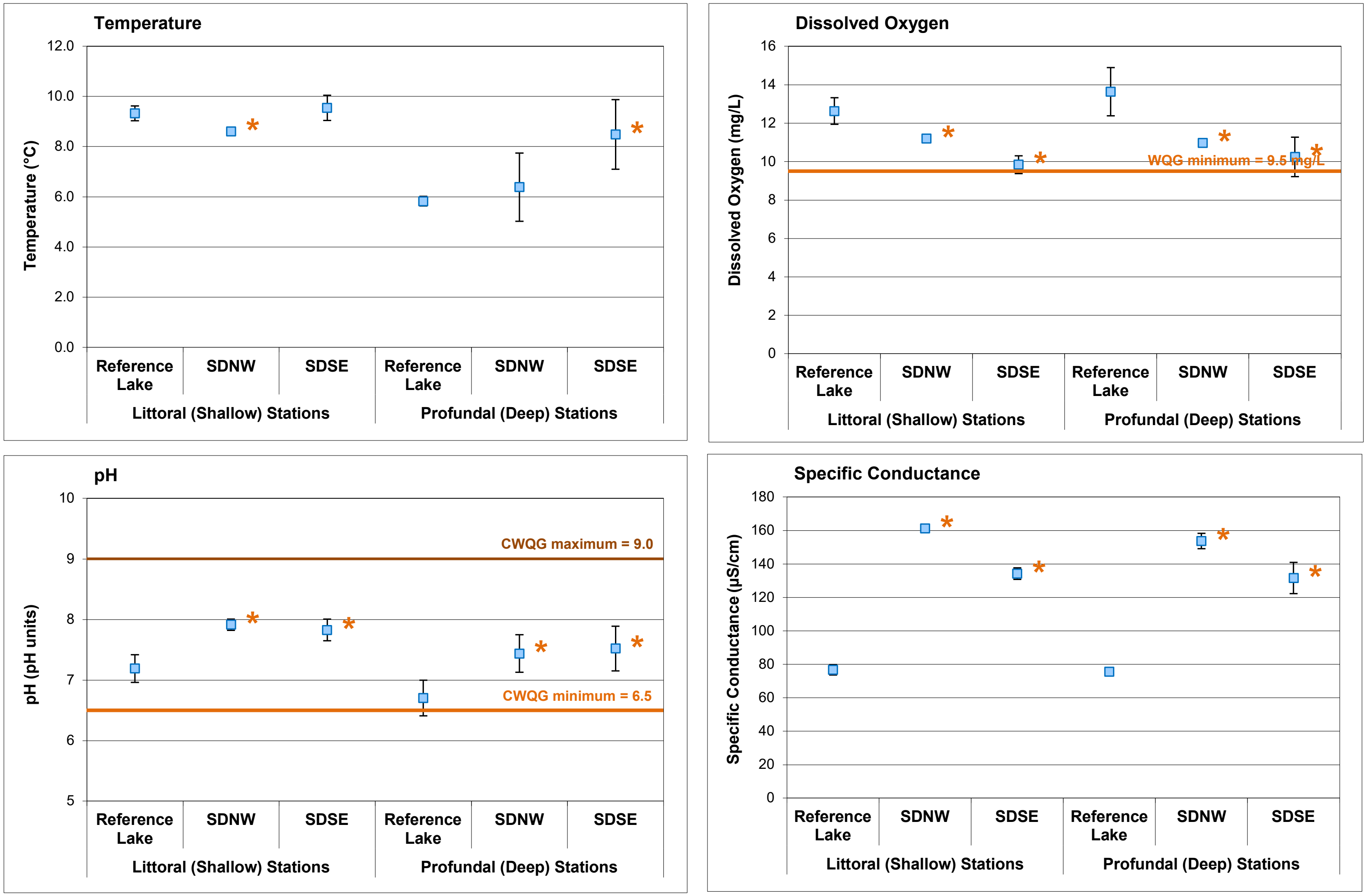


Figure 4.5: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Basins (SDNW and SDSE) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2020

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.

stations than like habitat stations at Reference Lake 3 during the August 2020 biological study (Figure 4.5). However, mean dissolved oxygen concentrations were above the WQG of 9.5 mg/L near the bottom at littoral and profundal stations of Sheardown Lake NW during biological monitoring in August 2020 (Figure 4.5; Appendix Table C.40).

Water column profiles showed decreasing pH with increased depth at Sheardown Lake NW and Reference Lake 3 in 2020, with the changes in pH through the water column at both lakes appearing to coincide with changes in water temperature (Figure 4.4). The pH near the bottom at littoral and profundal stations of Sheardown Lake NW were significantly higher than at respective habitats at the reference lake during the August 2020 biological study (Figure 4.5). However, the mean incremental difference in bottom pH between lakes was less than a pH unit, and pH values were consistently within WQG limits at Sheardown Lake NW (Figure 4.5; Appendix Table C.40), suggesting that the pH difference between lakes was not ecologically meaningful. Specific conductance profiles at Sheardown Lake NW showed no distinct changes with depth during any of the winter, summer, or fall sampling events in 2020, and exhibited similar patterns to those observed at Reference Lake 3 (Figure 4.4). Specific conductance near the bottom of the water column was significantly higher at Sheardown Lake NW littoral and profundal stations compared to the reference lake (Figure 4.5; Appendix Table C.40). Water clarity, as determined through evaluation of Secchi depth, was significantly lower at Sheardown Lake NW than at the reference lake at the time of the August 2020 biological study (Appendix Figure C.8).

Water chemistry at Sheardown Lake NW met all AEMP benchmarks and WQG over the duration of spring, summer, and fall sampling events in 2020 (Table 4.4). Among those parameters with established AEMP benchmarks, aluminum, chloride, nitrate, and sulphate concentrations were elevated by factors greater than three at Sheardown Lake NW compared to the reference lake during the summer and fall sampling events (Table 4.4; Appendix Table C.43). Of those parameters without AEMP benchmarks, turbidity, total manganese concentrations, and total and dissolved molybdenum and uranium concentrations were elevated at Sheardown Lake NW compared to the reference lake during summer and/or fall sampling events in 2020 (Appendix Tables C.43 and C.45). Similar to previous studies, elevated total aluminum and manganese concentrations at Sheardown Lake NW compared to the reference lake in 2020 were associated with suspended material that contributed to elevated turbidity at Sheardown Lake NW (Table 4.4; Appendix Table C.42). Naturally high turbidity[12] at Sheardown Lake NW may reflect backflow received from Mary River that contains relatively high amounts of suspended aluminum--and manganese-bearing particulate minerals. Similar concentrations of dissolved

[12] Turbidity at Sheardown Lake NW in 2020 was comparable to turbidity shown at the lake during baseline (Appendix Table C.44), suggesting that greater turbidity at this lake compared to Reference Lake 3 reflects a natural phenomenon.

Table 4.4: Mean Water Chemistry at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

	Parameters	Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 (n = 3)		Sheardown Lake NW Stations (n = 5)		
					Summer	Fall	Winter	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	79	79	191	164	166
	pH (lab)	pH	6.5 - 9.0	-	7.66	7.75	7.65	7.99	8.02
	Hardness (as $CaCO_3$)	mg/L	-	-	35	38	97	76	80
	Total Suspended Solids (TSS)	mg/L	-	-	2.0	2.0	2.0	2.5	2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	41	51	118	93	92
	Turbidity	NTU	-	-	0.15	0.15	0.12	0.84	0.55
	Alkalinity (as $CaCO_3$)	mg/L	-	-	46	34	76	59	74
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.010	0.014	0.025	0.005	0.012
	Nitrate	mg/L	3	3	0.020	0.020	0.211	0.219	0.211
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.001	0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	0.16	0.17	0.12	0.16
	Dissolved Organic Carbon	mg/L	-	-	3.3	3.5	2.7	1.8	2.0
	Total Organic Carbon	mg/L	-	-	4.6	3.8	3.8	1.8	2.5
	Total Phosphorus	mg/L	0.020[α]	-	0.004	0.003	0.007	0.005	0.007
	Phenols	mg/L	0.004[α]	-	0.0010	0.0011	0.002	0.0018	0.002
Anions	Bromide (Br)	mg/L	-	-	0.1	0.1	0.1	0.05	0.1
	Chloride (Cl)	mg/L	120	120	1.4	1.4	5.2	4.2	4.5
	Sulphate (SO_4)	mg/L	218[β]	218	3.6	3.6	17.1	14.7	15.2
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.0031	0.003	0.005	0.016	0.011
	Antimony (Sb)	mg/L	0.020[α]	-	0.0001	0.0001	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.0001	0.0001	0.0001	0.0001	0.0001
	Barium (Ba)	mg/L	-	-	0.0064	0.00696	0.00919	0.00759	0.00819
	Beryllium (Be)	mg/L	0.011[α]	-	0.0005	0.0005	0.0005	0.0001	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0005	0.0005	0.00005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.0115	0.01225	0.0135
	Cadmium (Cd)	mg/L	0.00012	0.00009	0.00001	0.00001	0.00001	0.000005	0.00001
	Calcium (Ca)	mg/L	-	-	7.2	7.2	18.7	14.1	15.5
	Chromium (Cr)	mg/L	0.0089	0.0089	0.0005	0.0005	0.0005	0.0001	0.0005
	Cobalt (Co)	mg/L	0.0009[α]	0.004	0.0001	0.0001	0.0001	0.0001	0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.00073	0.0008	0.0010	0.0009	0.0008
	Iron (Fe)	mg/L	0.30	0.300	0.03	0.03	0.03	0.02	0.03
	Lead (Pb)	mg/L	0.001	0.001	0.00005	0.00005	0.00005	0.00005	0.00005
	Lithium (Li)	mg/L	-	-	0.0010	0.001	0.0015	0.0014	0.0015
	Magnesium (Mg)	mg/L	-	-	4.2	4.7	11.9	9.2	10.1
	Manganese (Mn)	mg/L	0.935[β]	-	0.00080	0.00068	0.00075	0.00283	0.00132
	Mercury (Hg)	mg/L	0.000026	-	0.000005	0.000005	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00013	0.00015	0.00120	0.00097	0.00101
	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00050	0.00080	0.00072	0.00069
	Potassium (K)	mg/L	-	-	0.9	0.90	1.64	1.33	1.38
	Selenium (Se)	mg/L	0.001	-	0.001	0.001	0.001	0.000052	0.001
	Silicon (Si)	mg/L	-	-	0.50	0.50	0.62	0.63	0.56
	Silver (Ag)	mg/L	0.00025	0.0001	0.00001	0.00001	0.00001	0.00001	0.00001
	Sodium (Na)	mg/L	-	-	0.9	0.96	2.25	1.83	1.96
	Strontium (Sr)	mg/L	-	-	0.0084	0.0082	0.0132	0.0104	0.0108
	Thallium (Tl)	mg/L	0.0008	0.0008	0.0001	0.0001	0.0001	0.00001	0.0001
	Tin (Sn)	mg/L	-	-	0.0001	0.0001	0.0001	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.010	0.010	0.010	0.001	0.010
	Uranium (U)	mg/L	0.015	-	0.00032	0.00033	0.00143	0.00135	0.00153
	Vanadium (V)	mg/L	0.006[α]	0.006	0.0010	0.001	0.001	0.0005	0.001
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.003	0.003	0.003	0.003

[shaded box] Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

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

[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 specific to Sheardown Lake NW.

[d] Benchmark is 0.179 mg/L and 0.173 mg/L for shallow and deep stations, respectively (Intrinsik 2013).

aluminum and manganese between Sheardown Lake NW and Reference Lake 3 in 2020 (and historically) suggested that the mine was unlikely to be the source of these metals (Appendix Tables C.43 and C.45). Sulphate was the only parameter among those with established AEMP benchmarks that was elevated in 2020 compared to baseline at Sheardown Lake NW (Appendix Figure C.16), as were dissolved molybdenum and uranium concentrations among those parameters without AEMP benchmarks (Appendix Tables C.42 and C.44). Overall, a slight mine-related influence on water quality of Sheardown Lake NW was indicated in 2020 as reflected by elevated concentrations of chloride, molybdenum, nitrate, sulphate, and uranium. However, concentrations of all parameters remained well below AEMP benchmarks and WQG since commercial mine operations commenced in 2015, and therefore no adverse biological effects were expected at Sheardown Lake NW.

4.2.2 Sediment Quality

Surficial sediment in Sheardown Lake NW was primarily composed of silt, except at littoral station DLO-01-10, which contained 93% sand (Figure 4.6; Appendix Table D.25). Except for a slightly higher percentage of clay in sediments from profundal stations in Sheardown Lake NW, no differences in sediment particle size were noted relative to stations sharing like-habitat at Reference Lake 3 (Appendix Table D.26). The TOC content in sediment from littoral and profundal stations at Sheardown Lake NW was significantly lower, and sediment was significantly more compact (i.e., lower moisture content), than at the reference lake (Appendix Table D.26). Similar to observations at Reference Lake 3 and Camp Lake, reddish-brown oxidized material was observed on the surface of Sheardown Lake NW sediments at sine stations (Appendix Tables D.23 and D.24). This material occasionally occurred as a thin, distinct layer that was likely principally composed of iron (oxy)hydroxide precipitate. Substrate of Sheardown Lake NW exhibited some blackening (or unusually dark colouration), and traces of a sulphidic odour at some stations at the time of the August 2020 sampling event (Appendix Tables D.23 and D.24), suggesting the occurrence of reducing conditions in the sediment similar to that observed at the reference lake (Appendix Tables D.3 and D.4).

No consistent spatial differences in sediment metal concentrations occurred between Sheardown Lake NW stations located nearest to key tributary inlets (e.g., SDLT1 and SDLT12) and those located near the lake outlet in 2020 (Appendix Table D.25). Although arsenic, cadmium, iron, nickel, and uranium concentrations were highest in sediment at the Sheardown Lake NW station located closest to the outlet of SDLT1 (i.e., Station DD-HAB 9-STN2), higher concentrations of these metals may be related to high TOC content at this location (Appendix Table D.25). Sheardown Lake Tributary 1 was previously identified as a source of iron loadings to the lake (Section 4.1.2), so elevated iron concentrations at this location are not unexpected. Mean metal

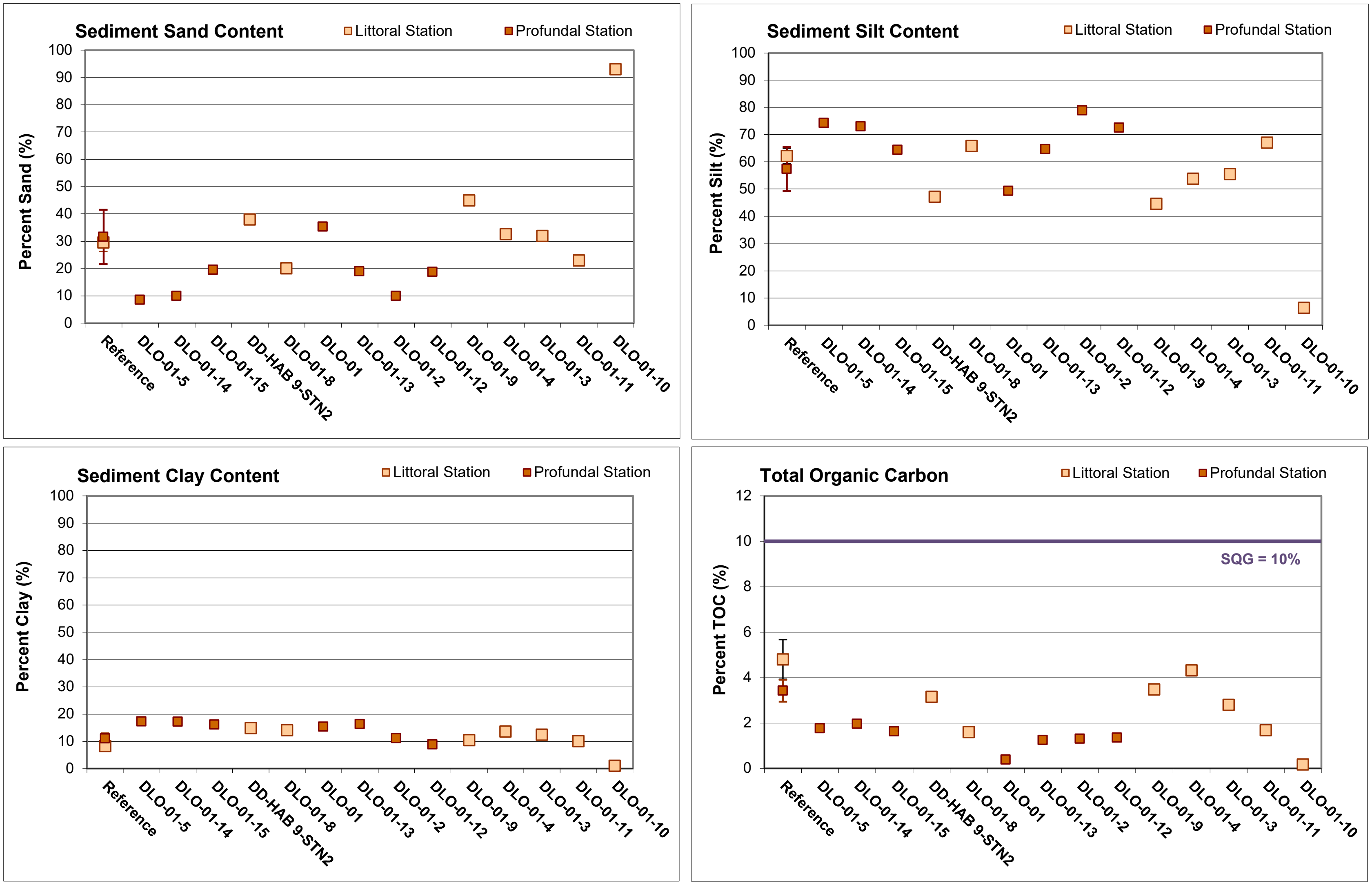


Figure 4.6: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake NW (DLO-01) Sediment Monitoring Stations and Reference Lake 3 (mean ± SE), Mary River Project CREMP, August 2020

concentrations in littoral and profundal sediment of Sheardown Lake NW were very similar to mean concentrations observed at like-habitat in Reference Lake 3, the only exception being slightly elevated (i.e., 3- to 5-fold higher) mean concentration of manganese at the Sheardown Lake NW profundal stations (Table 4.5; Appendix Table D.27). Although average concentrations of iron were above SQG in sediment from littoral stations in Sheardown Lake NW, the average concentration of iron was also above the SQG in sediment from Reference Lake 3 indicating naturally elevated concentrations of iron (Table 4.5). Nickel was also above the SQG in sediment from one littoral station in Sheardown Lake NW (Appendix Table D.25). In addition to iron, manganese was above the SQG in profundal sediments from Sheardown Lake NW and the reference lake, indicating manganese is also naturally elevated in the study area (Table 4.5). Only the mean concentration of manganese in profundal sediment of Sheardown Lake NW was above the lake-specific AEMP benchmark, whereas at the reference lake, mean concentrations of copper, iron, and manganese were elevated above the Sheardown Lake NW AEMP benchmarks in littoral and profundal sediments (Table 4.5).

Metal concentrations in sediment from Sheardown Lake NW in 2020 were comparable to those observed during the mine baseline (2005 to 2013) period (Figure 4.7; Appendix Table D.27).[13] On average, metal concentrations in sediment of Sheardown Lake NW in 2020 were within the range of those observed from 2015 to 2019, except for manganese at the profundal stations where concentrations were higher than historically (although the variability in concentrations among sampling stations was substantial, as indicated by the high standard deviation; Figure 4.7). No continual increase in mean concentrations of any metals were apparent from 2015 to 2020 at the Sheardown Lake NW littoral or profundal stations, including for manganese (Figure 4.7). Overall, no substantial changes in sediment metal concentrations of Sheardown Lake NW have occurred since the commencement of mine operations in 2015.

4.2.3 Phytoplankton

Chlorophyll-a concentrations at Sheardown Lake NW showed no consistent spatial gradients with progression towards the lake outlet among the winter, summer, and fall sampling events in 2020 (Figure 4.8). Chlorophyll-a concentrations differed significantly among seasons at Sheardown Lake NW, with highest and lowest concentrations observed in summer and winter, respectively (Appendix Tables E.5 and E.6) reflecting similar occurrence of highest chlorophyll-a concentrations in the summer sampling event at the reference lake (Appendix Table B.7).

[13] See footnote 6 regarding differences in the concentration of boron in sediment between baseline and recent CREMP studies.

Table 4.5: Sediment Particle Size, Total Organic Carbon, and Metal Concentrations at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2020

Parameter		Units	SQG[a]	AEMP Benchmark[b] (NW, SE)	Littoral		Profundal	
					Reference Lake (n = 5)	Sheardown Lake NW (n = 4)	Reference Lake (n = 5)	Sheardown Lake NW (n = 4)
					Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC		%	10^{α}	-	4.80 ± 1.96	2.10 ± 1.39	3.42 ± 1.08	1.17 ± 0.576
Metals	Aluminum (Al)	mg/kg	-	-	16,880 ± 1,785	15,635 ± 8,686	21,800 ± 2,185	20,675 ± 3,970
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
	Arsenic (As)	mg/kg	17	6.2, 5.9	3.53 ± 1.09	4.05 ± 2.84	4.07 ± 0.397	3.72 ± 0.66
	Barium (Ba)	mg/kg	-	-	117 ± 22.0	87 ± 58	122 ± 18.3	110 ± 41
	Beryllium (Be)	mg/kg	-	-	0.65 ± 0.073	0.88 ± 0.48	0.80 ± 0.092	0.98 ± 0.176
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.23 ± 0.024	<0.20 ± 0	0.24 ± 0.04
	Boron (B)	mg/kg	-	-	12.2 ± 0.853	25.8 ± 14.1	14.7 ± 1.77	29.9 ± 5.78
	Cadmium (Cd)	mg/kg	3.5	1.5, 1.5	0.173 ± 0.047	0.252 ± 0.177	0.148 ± 0.0172	0.253 ± 0.046
	Calcium (Ca)	mg/kg	-	-	5,608 ± 1,247	4,023 ± 1,857	5,010 ± 407	4,270 ± 593
	Chromium (Cr)	mg/kg	90	97, 79	54.3 ± 4.40	60.1 ± 30.9	65.0 ± 6.64	71.3 ± 11.41
	Cobalt (Co)	mg/kg	-	-	10.8 ± 1.64	11.5 ± 6.39	15.2 ± 1.56	15.9 ± 3.32
	Copper (Cu)	mg/kg	110	58, 56	71.4 ± 14.2	38.0 ± 21.8	***83.8*** ± 11.1	42.5 ± 6.55
	Iron (Fe)	mg/kg	$40,000^{\alpha}$	52,200, 34,400	***50,600*** ± 24,939	41,200 ± 21,466	45,080 ± 4,440	43,350 ± 12,934
	Lead (Pb)	mg/kg	91.3	35	13.8 ± 0.799	16.5 ± 8.55	16.7 ± 1.82	19.7 ± 3.94
	Lithium (Li)	mg/kg	-	-	26.0 ± 2.51	29.6 ± 16.3	33.7 ± 3.83	33.2 ± 6.59
	Magnesium (Mg)	mg/kg	-	-	11,440 ± 814	9,930 ± 5,277	14,180 ± 1,422	13,105 ± 2,514
	Manganese (Mn)	mg/kg	$1,100^{\alpha,\beta}$	4,530, 657	579 ± 258	526 ± 380	***1,230*** ± 355	***5,291*** ± 5,948
	Mercury (Hg)	mg/kg	0.486	0.17	0.0500 ± 0.0178	0.0306 ± 0.0198	0.0583 ± 0.0164	0.0332 ± 0.01397
	Molybdenum (Mo)	mg/kg	-	-	4.44 ± 3.31	3.08 ± 1.93	2.52 ± 0.273	5.22 ± 4.69
	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	77, 66	40.0 ± 3.52	56.1 ± 31.1	45.0 ± 4.54	61.4 ± 9.82
	Phosphorus (P)	mg/kg	$2,000^{\alpha}$	1,958, 1,278	1,167 ± 394	832 ± 379	956 ± 47.3	900 ± 94.6
	Potassium (K)	mg/kg	-	-	4,100 ± 453	4,168 ± 2,333	5,338 ± 543	5,288 ± 1,043
	Selenium (Se)	mg/kg	-	-	0.73 ± 0.31	0.37 ± 0.15	0.61 ± 0.18	0.32 ± 0.11
	Silver (Ag)	mg/kg	-	-	0.14 ± 0.047	0.16 ± 0.042	0.20 ± 0.057	0.16 ± 0.024
	Sodium (Na)	mg/kg	-	-	304.2 ± 32	232 ± 122	369 ± 50	283 ± 50
	Strontium (Sr)	mg/kg	-	-	11.6 ± 1.70	9.45 ± 3.72	12.3 ± 1.24	11.40 ± 1.50
	Sulphur (S)	mg/kg	-	-	1,400 ± 387	<1,000 ± 0	1,140 ± 195	<1,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.379 ± 0.0415	0.419 ± 0.242	0.594 ± 0.094	0.514 ± 0.1225
	Tin (Sn)	mg/kg	-	-	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0
	Titanium (Ti)	mg/kg	-	-	1,006 ± 109	934 ± 472	1,136 ± 50	1,194 ± 210.5
	Uranium (U)	mg/kg	-	-	11.0 ± 2.41	6.17 ± 3.68	19.7 ± 3.76	6.37 ± 1.212
	Vanadium (V)	mg/kg	-	-	54.1 ± 5.40	45.8 ± 23.4	63.4 ± 4.89	56.7 ± 9.97
	Zinc (Zn)	mg/kg	315	135	73.1 ± 7.83	53.5 ± 29.0	83.8 ± 8.52	63.3 ± 11.80
	Zirconium (Zr)	mg/kg	-	-	4.5 ± 1.0	10.1 ± 6.4	3.9 ± 0.32	7.03 ± 1.68

(shaded) Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV 2020)

[b] AEMP Sediment Quality Benchmarks developed by Intrinsik (2013) using sediment quality guidelines, background sediment quality data, and method detection limits. The indicated values are specific to the Sheardown Lake basins

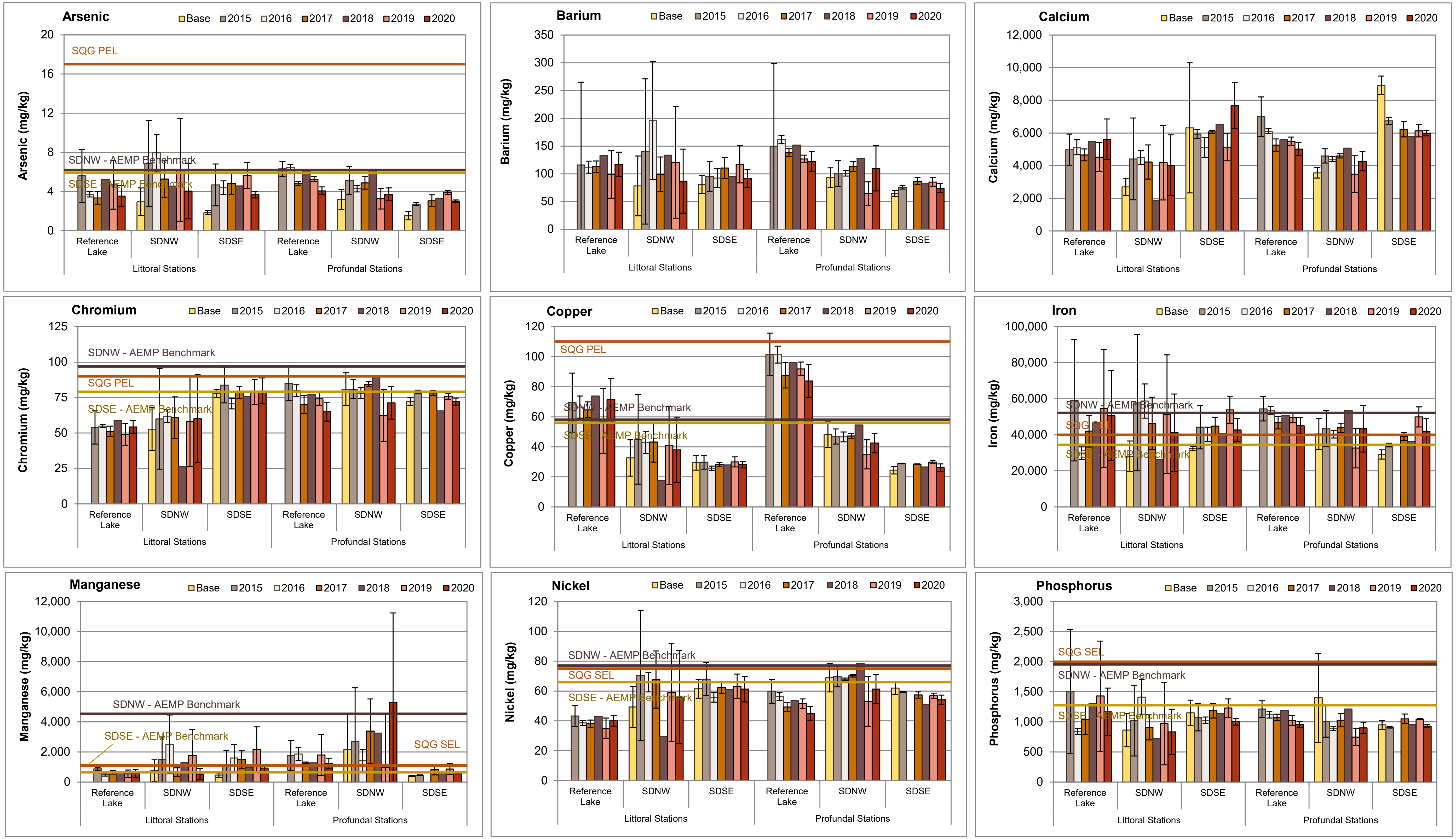


Figure 4.7: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Sheardown Lake NW (SDNW), Sheardown Lake SE (SDSE), and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2020) Periods

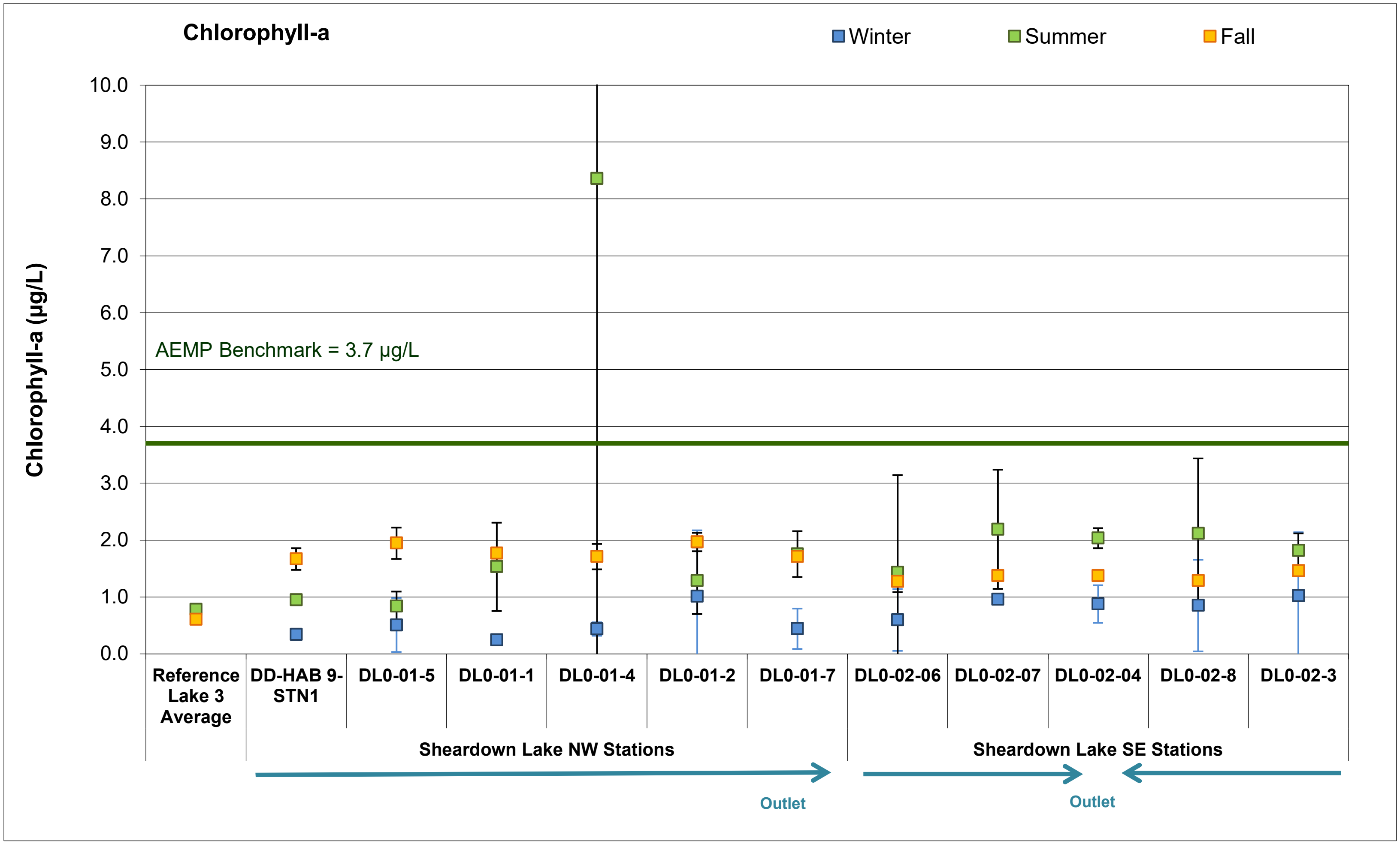


Figure 4.8: Chlorophyll-a Concentrations at Sheardown Lake NW (DLO-1) and Sheardown Lake SE (DLO-2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2020

Notes: Values are averages of samples taken from the surface and the bottom of the water column at each station. Reference values are expressed as mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2020.

Although chlorophyll-a concentrations were significantly higher at Sheardown Lake NW compared to Reference Lake 3 for both the summer and fall sampling events in 2020 (Appendix Tables E.7 and E.8), chlorophyll-a concentrations during each of the winter, summer, and fall sampling events at Sheardown Lake NW were well below the AEMP benchmark of 3.7 µg/L for all samples except one sample in July (Figure 4.8). For the sample that exceeded the benchmark, the chlorophyll-a concentration (15 µg/L) was an order of magnitude higher than any other sample collected in the lake on the same day, suggesting a sampling or laboratory analysis issue rather than an actual increase in phytoplankton abundance. Chlorophyll-a concentrations at Sheardown Lake NW were suggestive of an 'oligotrophic' status using Wetzel (2001) lake trophic status classifications. This trophic status classification was consistent with an oligotrophic categorization for Sheardown Lake NW using CWQG (CCME 2020) classifications based on aqueous total phosphorus concentrations near the surface (i.e., concentrations below 10 µg/L; Table 4.4; Appendix Table C.42).

Chlorophyll-a concentrations at Sheardown Lake NW in 2020 were within the ranges shown among years of mine construction (2014) and previous mine operation (2015 to 2019), and showed no consistent direction of changes for any of the winter, summer, or fall seasons (Figure 4.9; Appendix Table E.11). This suggested no ecologically meaningful changes in the trophic status of Sheardown Lake NW since the onset of mine operations in 2015. No chlorophyll-a data are available for Sheardown Lake NW over the mine baseline period (2005 to 2013), precluding comparisons of Sheardown Lake NW chlorophyll-a data to the period prior to mine construction.

4.2.4 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral and profundal habitats of Sheardown Lake NW compared to like-habitat at Reference Lake 3 at magnitudes outside of the CES_{BIC} of ±2 SD_{REF} (Tables 4.6 and 4.7). Although no significant differences in richness or evenness were indicated between Sheardown Lake NW and the reference lake for either littoral or profundal habitat (Tables 4.6 and 4.7), Bray-Curtis Index differed significantly between Sheardown Lake NW and Reference Lake 3 for both habitat types (Appendix Table F.21). Because no ecologically significant differences (i.e., CES_{BIC} outside of ±2 SD_{REF}) in the relative abundance of any dominant taxonomic groups were indicated between Sheardown Lake NW and the reference lake for either habitat type (Tables 4.6 and 4.7), the difference in Bray-Curtis Index between lakes likely reflected substantially higher benthic invertebrate density at Sheardown Lake NW. The occurrence of higher benthic invertebrate density without an accompanying difference in evenness or dominant taxonomic groups suggested that Sheardown Lake NW was simply more productive than Reference Lake 3. This was supported by the

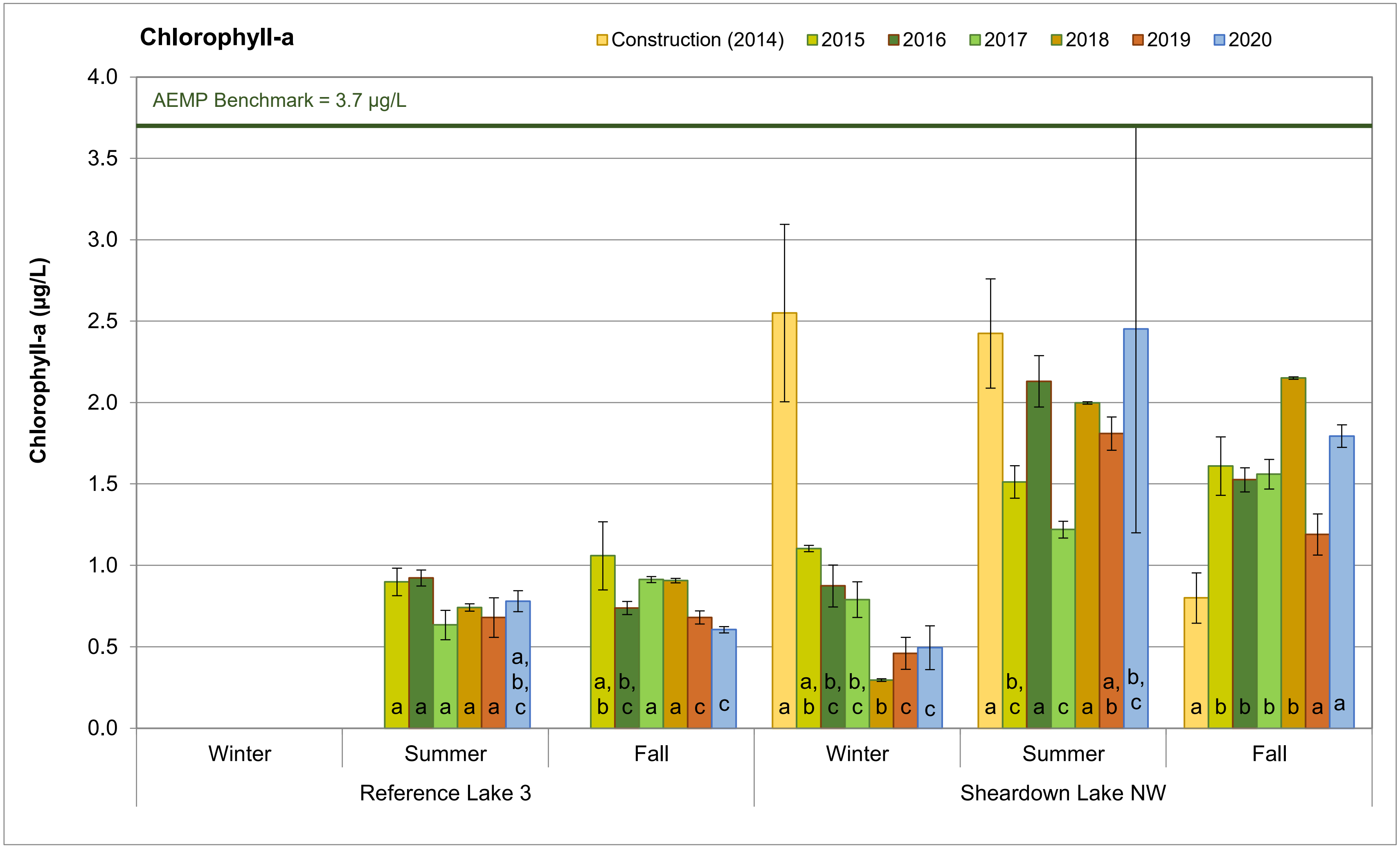


Figure 4.9: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Sheardown Lake NW and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2020) Periods (mean ± SE)

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

Table 4.6: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake NW (DLO-01) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-equal	log10	YES	0.003	11.8	Reference Lake 3	1,571	430	193	1,190	1,474	2,310
						Sheardown NW Littoral	6,631	3,914	1,750	2,250	6,457	12,500
Richness (Number of Taxa)	Mann Whitney	rank	NO	0.202	0.3	Reference Lake 3	14.6	2.5	1.1	13.0	14.0	19.0
						Sheardown NW Littoral	15.4	1.1	0.5	14.0	15.0	17.0
Simpson's Evenness (E)	t-equal	none	NO	0.515	-0.3	Reference Lake 3	0.810	0.110	0.049	0.630	0.847	0.923
						Sheardown NW Littoral	0.773	0.055	0.025	0.718	0.762	0.854
Shannon Diversity	t-equal	log10	NO	0.301	-0.6	Reference Lake 3	2.710	0.526	0.235	2.080	2.730	3.510
						Sheardown NW Littoral	2.410	0.217	0.097	2.260	2.280	2.760
Hydracarina (%)	t-equal	log10	YES	0.065	-1.0	Reference Lake 3	5.3	2.6	1.2	3.5	4.4	9.9
						Sheardown NW Littoral	2.6	3.5	1.6	0.2	0.3	8.1
Ostracoda (%)	t-equal	log10	NO	0.118	-0.8	Reference Lake 3	37.9	14.5	6.5	26.7	36.2	62.6
						Sheardown NW Littoral	25.6	8.6	3.8	13.9	28.0	36.0
Chironomidae (%)	t-equal	none	YES	0.060	1.1	Reference Lake 3	52.6	15.6	7.0	26.9	59.0	66.4
						Sheardown NW Littoral	70.4	9.3	4.2	60.5	70.1	83.7
Metal-Sensitive Chironomidae (%)	t-equal	log10	NO	0.101	-1.4	Reference Lake 3	28.8	9.5	4.3	15.6	32.5	38.7
						Sheardown NW Littoral	15.4	14.3	6.4	2.3	14.4	36.4
Collector-Gatherers (%)	t-equal	none	NO	0.114	1.3	Reference Lake 3	63.1	11.4	5.1	53.6	60.3	81.5
						Sheardown NW Littoral	77.5	14.1	6.3	54.2	83.5	90.8
Filterers (%)	t-equal	log10	NO	0.114	-1.3	Reference Lake 3	27.1	9.8	4.4	14.4	29.2	38.0
						Sheardown NW Littoral	14.6	14.5	6.5	1.3	13.2	36.0
Shredders (%)	t-unequal	none	YES	0.068	-1.1	Reference Lake 3	3.9	3.3	1.5	0.6	3.2	7.4
						Sheardown NW Littoral	0.2	0.2	0.1	0.0	0.0	0.4
Clingers (%)	t-equal	none	YES	< 0.001	-2.5	Reference Lake 3	31.9	9.3	4.2	17.9	33.5	41.6
						Sheardown NW Littoral	8.5	3.8	1.7	3.7	9.6	12.5
Sprawlers (%)	t-equal	none	NO	0.176	-0.8	Reference Lake 3	57.9	12.1	5.4	41.0	57.2	73.8
						Sheardown NW Littoral	48.4	7.7	3.5	40.3	47.1	57.5
Burrowers (%)	t-equal	log10	YES	< 0.001	6.7	Reference Lake 3	10.2	4.9	2.2	4.6	8.3	17.3
						Sheardown NW Littoral	43.1	8.5	3.8	32.4	41.7	54.2

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

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

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

Table 4.7: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake NW (DLO-01) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2019

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-unequal	none	YES	0.021	6.0	Reference Lake 3	479	142	63	336	491	681
						Sheardown NW Profundal	1,326	527	236	802	1,345	2,034
Richness (Number of Taxa)	t-equal	log10	NO	0.581	0.4	Reference Lake 3	7.0	1.9	0.8	5.0	8.0	9.0
						Sheardown NW Profundal	7.8	2.5	1.1	6.0	7.0	12.0
Simpson's Evenness (E)	t-equal	none	NO	0.103	-5.5	Reference Lake 3	0.731	0.045	0.020	0.689	0.721	0.795
						Sheardown NW Profundal	0.486	0.295	0.132	0.236	0.324	0.872
Shannon Diversity	t-equal	log10	NO	0.199	-1.9	Reference Lake 3	1.800	0.196	0.088	1.580	1.720	2.030
						Sheardown NW Profundal	1.420	0.891	0.398	0.677	0.961	2.720
Hydracarina (%)	t-equal	log10(x+1)	NO	0.839	-0.1	Reference Lake 3	2.8	2.0	0.9	0.0	3.5	5.1
						Sheardown NW Profundal	2.6	0.9	0.4	1.3	3.1	3.2
Ostracoda (%)	t-equal	none	NO	0.434	-0.5	Reference Lake 3	8.6	4.1	1.8	3.5	7.7	14.5
						Sheardown NW Profundal	6.6	3.9	1.7	1.1	5.9	11.7
Chironomidae (%)	t-equal	log10	NO	0.268	0.7	Reference Lake 3	87.9	4.2	1.9	82.3	87.2	92.7
						Sheardown NW Profundal	90.8	3.7	1.7	86.2	91.0	95.7
Metal-Sensitive Chironomidae (%)	t-equal	log10	YES	0.002	-1.6	Reference Lake 3	31.5	17.6	7.9	7.9	38.0	49.3
						Sheardown NW Profundal	4.0	4.0	1.8	1.3	2.6	11.1
Collector-Gatherers (%)	t-equal	none	YES	0.003	1.9	Reference Lake 3	62.9	15.0	6.7	45.4	56.1	79.0
						Sheardown NW Profundal	91.9	3.5	1.6	86.8	91.7	96.6
Filterers (%)	t-equal	none	YES	0.008	-1.6	Reference Lake 3	30.7	17.5	7.8	7.9	38.0	49.3
						Sheardown NW Profundal	2.4	4.9	2.2	0.0	0.0	11.1
Clingers (%)	t-equal	log10	YES	0.003	-1.7	Reference Lake 3	33.5	16.9	7.6	13.1	41.5	52.8
						Sheardown NW Profundal	5.2	4.2	1.9	1.3	3.2	12.1
Sprawlers (%)	t-equal	none	NO	0.786	0.3	Reference Lake 3	64.8	16.2	7.2	45.5	58.5	87.0
						Sheardown NW Profundal	69.9	36.7	16.4	16.1	94.2	97.5
Burrowers (%)	t-equal	log10(x+1)	NO	0.146	8.1	Reference Lake 3	1.7	2.9	1.3	0.0	0.0	6.7
						Sheardown NW Profundal	24.9	32.6	14.6	1.3	2.6	71.8

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

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

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

occurrence of no ecologically significant differences in the relative abundance of metal-sensitive chironomids and FFG between lakes (Tables 4.6 and 4.7), which indicated no sediment metal-related influences or effects to available food resources, respectively, on the benthic invertebrate community of Sheardown Lake NW in 2020. Similar to Camp Lake, sediment at Sheardown Lake NW littoral and profundal stations was significantly more compact (i.e., lower moisture content) than like-habitat stations at the reference lake, which potentially contributed to differences in relative abundance of HPG between lakes (Tables 4.6 and 4.7; Appendix Table F.35).

No significant differences in benthic invertebrate density, richness, evenness, relative abundance of dominant groups, and relative abundance of FFG were consistently shown for Sheardown Lake NW littoral stations over years of mine operation (2015 to 2020) compared to baseline studies conducted in 2007, 2008, and 2013 (Appendix Figure F.10; Appendix Table F.37). At profundal stations of Sheardown Lake NW, only the relative abundance of metal-sensitive Chironomidae routinely differed significantly between the mine operational period and both years of mine baseline (Appendix Table F.38). However, because greater relative abundance of metal-sensitive Chironomidae were observed in years of mine operation compared to baseline (Appendix Figure F.11), this temporal difference was not indicative of an adverse effect on the benthic invertebrate community at Sheardown Lake NW. Therefore, consistent with no substantial changes in water and sediment quality since the mine baseline period, no significant differences in benthic invertebrate community features were indicated at littoral and profundal habitat of Sheardown Lake NW since the commencement of commercial mine operation in 2015.

4.2.5 Fish Population

4.2.5.1 Sheardown Lake NW Fish Community

The fish community of Sheardown Lake NW included arctic charr and ninespine stickleback in 2020 (Table 4.8), reflecting the same fish species that were observed historically (Minnow 2020). Arctic charr and ninespine stickleback CPUE were higher at Sheardown Lake NW than at the reference lake in 2020 (Table 4.8), suggesting higher densities and/or productivity of these species at Sheardown Lake NW. A greater relative abundance of fish, together with higher chlorophyll-a concentrations and greater benthic invertebrate density, suggested that overall biological productivity was higher at Sheardown Lake NW than at Reference Lake 3. Arctic charr electrofishing CPUE at Sheardown Lake NW in 2020 was within the range observed over the mine baseline period (2007 to 2013) and previous years of mine operation (2015 to 2019; Figure 4.10). Gill netting CPUE for arctic charr in 2020 was also within the range observed during baseline, but slightly greater than the previous five years of mine operation (Figure 4.10). The similarities in CPUE among study years suggested that the relative abundance of arctic charr

Table 4.8: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Sheardown Lake NW (DLO-01), Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2020

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	134	1	135	2
		CPUE	2.09	0.016	2.11	
	Gill netting	No. Caught	0	0	0	
		CPUE	0.956	0	0.956	
Sheardown Lake Northwest	Electrofishing	No. Caught	118	6	124	2
		CPUE	4.46	0.227	4.69	
	Gill netting	No. Caught	98	0	98	
		CPUE	3.34	0	3.34	
Sheardown Lake Southeast	Electrofishing	No. Caught	115	63	178	2
		CPUE	4.28	2.35	6.63	
	Gill netting	No. Caught	107	0	107	
		CPUE	3.81	0	3.81	

[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.

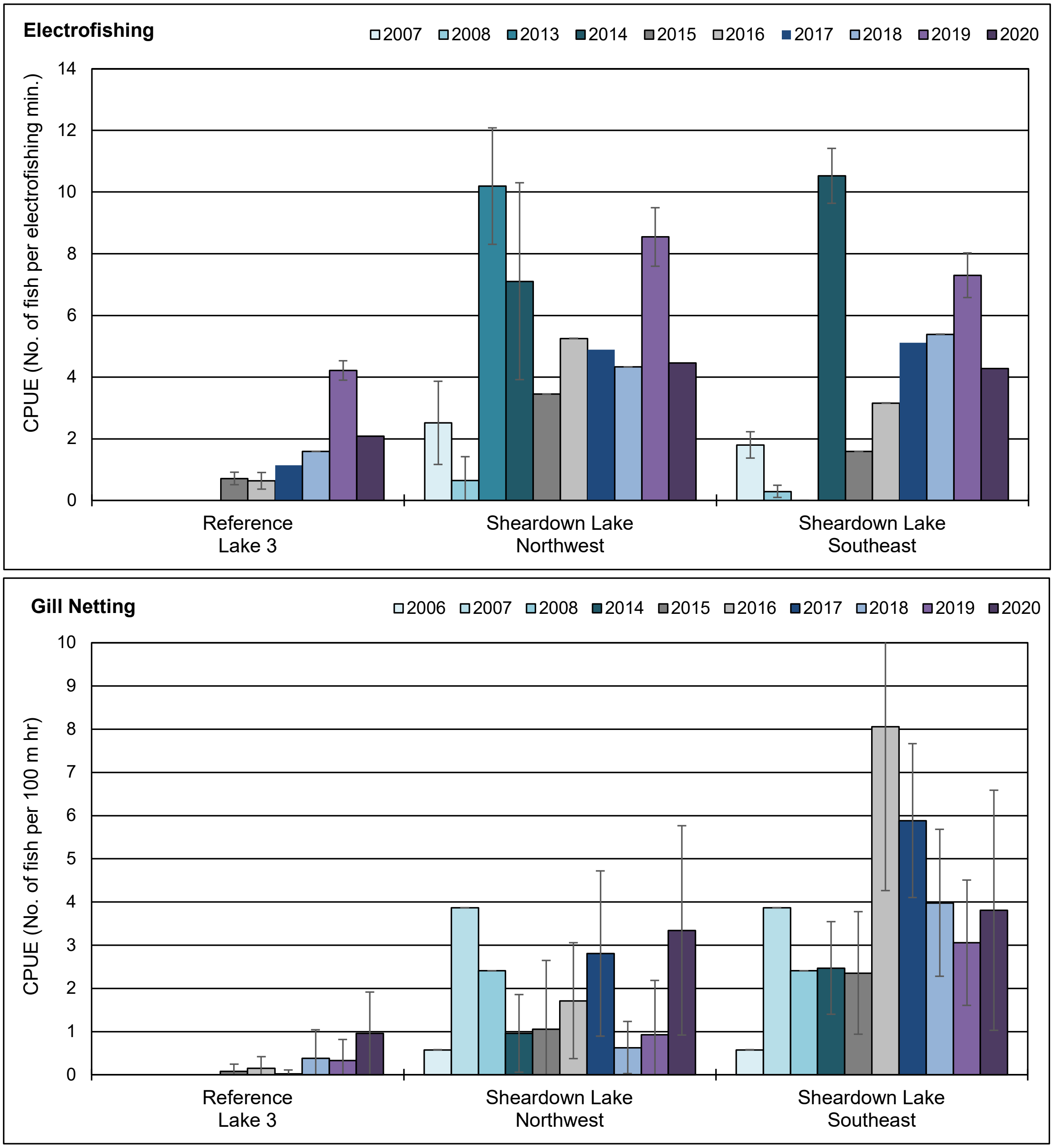


Figure 4.10: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Back-pack Electrofishing and Gill Netting at Sheardown Lake NW (DLO-01) and Sheardown Lake SE (DLO-02), Mary River Project CREMP, 2006 to 2020

Notes: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015 to 2020) mine phases. Lake basins (i.e., NW or SE) were not differentiated historically for baseline gill netting catches.

at the nearshore and littoral/profundal habitats of Sheardown Lake NW has remained similar over time, suggesting the mine has not influenced the size of the arctic charr population in the lake.

4.2.5.2 Sheardown Lake NW Fish Population Assessment

Nearshore Arctic Charr

A total of 100 arctic charr were captured from nearshore habitat of each of Sheardown Lake NW and Reference Lake 3 in August 2020. Distinguishing arctic charr YOY from the older, non-YOY age class was possible using a fork length cut-off of 4.8 cm and 4.3 cm for the Sheardown Lake NW and Reference Lake 3 data sets, respectively, based on evaluation of length-frequency distributions coupled with supporting age determinations (Figure 4.11; Appendix Tables G.4 and G.13). Because greater than ten YOY arctic charr were identified from the Sheardown Lake NW and Reference Lake 3 populations, statistical comparisons of health endpoints were completed separately on both the YOY and non-YOY populations. Length-frequency distributions for the nearshore arctic charr (based on the full dataset, and non-YOY only) differed significantly between Sheardown Lake NW and Reference Lake 3 (Table 4.9). This was primarily due to a greater number of non-YOY and larger fish being captured at Sheardown Lake NW compared to the reference lake (Figure 4.11). Both YOY and non-YOY arctic charr from nearshore habitats in Sheardown Lake NW were significantly larger and had greater condition than those from the reference lake (Table 4.9; Appendix Table G.14). However, the magnitudes of the differences in condition between lakes for both the YOY and noy-YOY populations were not considered ecologically significant because they were within CES_C (i.e., ±10%; Table 4.9; Appendix Table G.14).

Temporal comparisons of nearshore arctic charr populations between Sheardown Lake NW and Reference Lake 3 generally indicated non-YOY were significantly larger but showed no consistent difference/direction of difference in condition at Sheardown Lake NW since 2015 (Table 4.9). Although the lengths and weights of non-YOY arctic charr in years of mine operation (i.e., 2015 to 2020) have not differed consistently relative to the baseline period at Sheardown Lake NW, the condition of non-YOY arctic charr in all years of mine operation has been significantly lower than baseline at magnitudes near or outside of CES_C (i.e., ±10%; Table 4.9). The inconsistent response of non-YOY arctic charr between Sheardown Lake NW and Reference Lake 3 since 2015 compared to between years of mine operation and baseline at Sheardown Lake NW resulted in uncertainty as to whether current mine operations have affected non-YOY arctic charr health at Sheardown Lake NW.

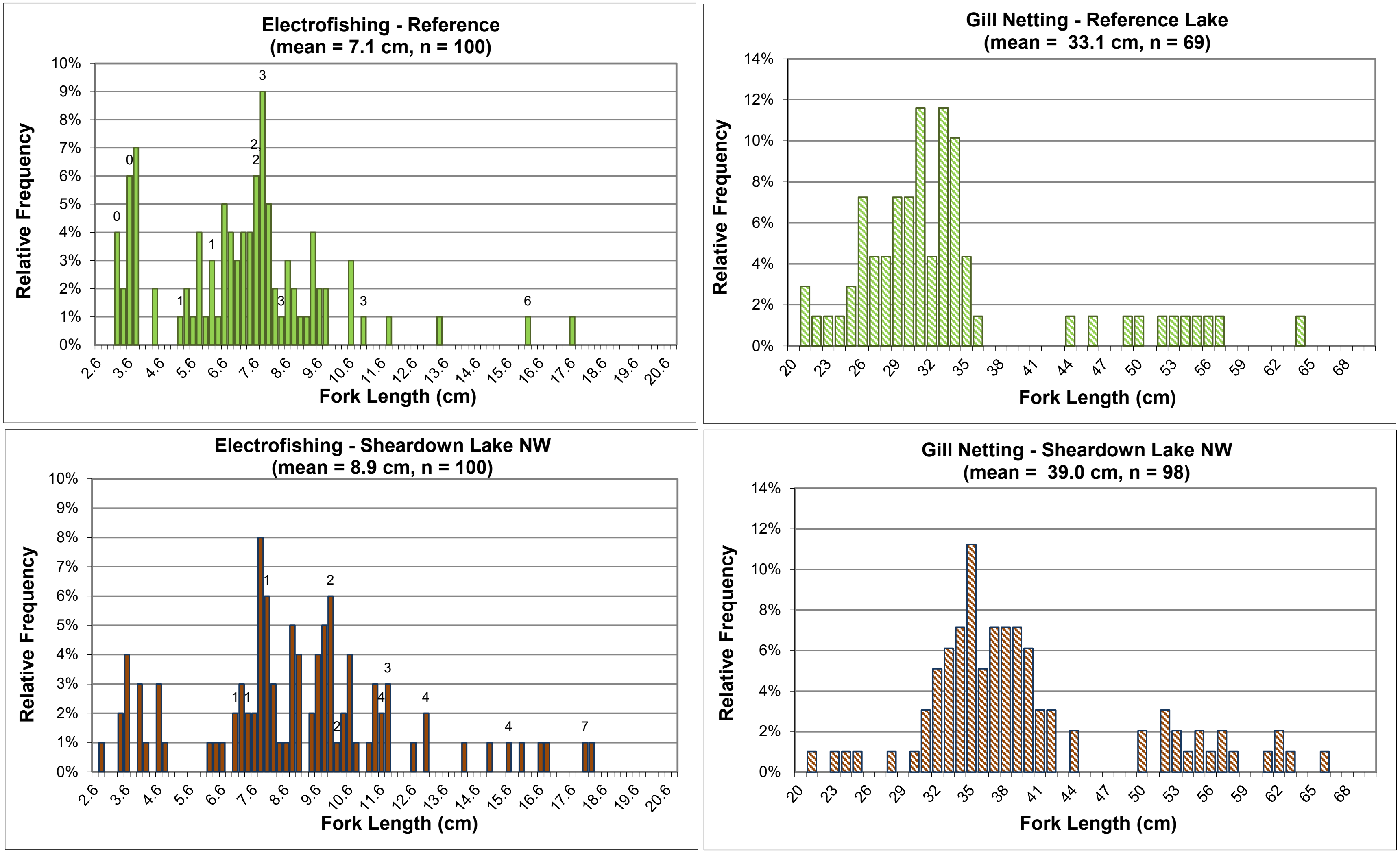


Figure 4.11: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2020

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

Table 4.9: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake NW and Reference Lake 3 from 2015 to 2020, and between Sheardown Lake NW 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 Sheardown Lake NW baseline period data [b]					
			2015	2016	2017	2018	2019	2020	2015	2016	2017	2018	2019	2020
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	-	-	No	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	**Yes (+29%)**	**Yes (+17%)**	**Yes (+20%)**	**Yes (+24%)**	**Yes (-10%)**	**Yes (+22%)**	No	No	No	**Yes (-12%)**	No	**Yes (+13%)**
		Size (mean weight)	**Yes (+121%)**	**Yes (+60%)**	No	**Yes (+83%)**	**Yes (-24%)**	**Yes (+99%)**	No	**Yes (-29%)**	No	**Yes (-50%)**	No	No
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	**Yes (+7%)**	**Yes (-5%)**	**Yes (+4%)**	**Yes (+10%)**	**Yes (-13%)**	**Yes (-12%)**	**Yes (-9%)**	**Yes (-10%)**	**Yes (-13%)**	**Yes (-9%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	No
		Age	-	-	-	-	-	-	**Yes (-35%)**	**Yes (-28%)**	**Yes (-26%)**	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (+22%)**	**Yes (+18%)**	**Yes (-21%)**	**Yes (-14%)**	**Yes (-6%)**	No	No	No
		Size (mean weight)	-	-	-	No	**Yes (+92%)**	**Yes (+94%)**	**Yes (-47%)**	**Yes (-31%)**	**Yes (-9%)**	No	No	No
		Growth (fork length-at-age)	-	-	-	-	-	-	No	No	No	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	No	No	**Yes (+24%)**	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+4%)**	No	**Yes (+11%)**	**Yes (+8%)**	**Yes (+11%)**	**Yes (+6%)**	No	No	No

BOLD indicates a significant difference related to the comparison.

[a] Values in parentheses indicate direction and magnitude of any significant differences.

[b] Baseline period data included 2002, 2005, 2006, 2008, and 2013 nearshore electrofishing data and 2006, 2008 and 2013 littoral/profundal gill netting data.

[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.

Littoral/Profundal Arctic Charr

A total of 98 and 69 arctic charr were sampled from littoral/profundal habitat of Sheardown Lake NW and Reference Lake 3, respectively, in August 2020. The length-frequency distribution for littoral/profundal arctic charr differed significantly between lakes based on greater numbers of larger fish captured at Sheardown Lake NW (Table 4.9; Figure 4.11). Arctic charr captured by gill net at Sheardown Lake NW in 2020 were significantly larger and had greater condition than those captured at Reference Lake 3 (Table 4.9; Appendix Table G.18). The magnitude of difference in condition (11%) was slightly outside of the CES_C of ±10%, suggesting a potentially ecologically meaningful difference.

The differences in size and condition of arctic charr captured from littoral/profundal habitat between Sheardown Lake NW and Reference Lake 3 in 2020 were similar to the differences shown in 2018 and/or 2019, suggesting no appreciable changes in health of littoral/profundal arctic charr at Sheardown Lake NW over time. No significant differences in body size or condition of arctic charr captured from littoral/profundal habitat of Sheardown Lake NW were observed from 2018 to 2020 relative to the baseline period (Table 4.9; Appendix Figure G.10; Appendix Table G.18). From 2015 to 2017, arctic charr sampled from littoral/profundal habitat of Sheardown Lake NW were significantly shorter, lighter, and of greater condition than those captured during the baseline period (Table 4.9). The absence of differences in size and condition of arctic charr from Sheardown Lake NW over the 2018 to 2020 period compared to baseline appeared to reflect closer comparability in fish size between the most recent studies and baseline. In turn, this suggested that assessment of littoral/profundal arctic charr health should use sampling methods that reduce variability in the size of fish sampled to assess potential mine-related effects. Nevertheless, the general absence of consistent ecologically significant differences in condition of arctic charr captured at littoral/profundal areas of Sheardown Lake NW from 2018 to 2020 compared to Reference Lake 3 and Sheardown Lake NW baseline suggested no adverse mine-related influences on the adult arctic charr population of the lake as a result of mine operations.

4.2.6 Effects Assessment and Recommendations

At Sheardown Lake NW, the following AEMP benchmarks were exceeded in 2020:

- Arsenic concentration in sediment was greater than the benchmark of 6.2 mg/kg at one littoral monitoring station (DD-HAB 9-STN2), although the average concentration of arsenic in sediment at littoral stations was below this benchmark;
- Iron concentration in sediment was greater than the benchmark of 52,200 mg/kg at one littoral station (DD-HAB 9-STN2) and one profundal station (DL0-01-5), although average

concentrations of iron in sediment at littoral and profundal stations were below this benchmark;

- Manganese concentration in sediment was greater than the benchmark of 4,530 mg/kg at two profundal stations (DL0-01-2 and DL0-01-5), although the average concentration of manganese in sediment at profundal stations was below this benchmark; and,
- Nickel concentration in sediment was greater than the benchmark of 77 mg/kg at one littoral monitoring station (DD-HAB 9-STN2), although the average concentration of nickel in sediment at littoral stations was below this benchmark.

No AEMP benchmarks for water quality were exceeded over the duration of spring, summer, and fall sampling events in 2020 at Sheardown Lake NW. Lake-specific AEMP benchmarks for sediment quality were exceeded for arsenic, iron, manganese, and nickel at as many as one littoral station and two profundal stations in 2020, but none of these metals were elevated in the sediment of Sheardown Lake NW compared to the reference lake and to concentrations at Sheardown Lake NW during baseline. No adverse effects to phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at Sheardown Lake NW in 2020 based on comparisons to reference conditions and to Sheardown Lake NW baseline conditions. Because no mine-related changes in metal concentrations occurred in sediment at Sheardown Lake NW in 2020, and no adverse effects to biota were associated with concentrations of metals above AEMP benchmarks for sediment quality, a low action response is recommended to meet obligations under the AEMP Management Response Framework for Sheardown Lake NW. Specifically, it is recommended that, because concentrations of metals in Sheardown Lake NW sediment have been similar to those shown at the reference lake, consideration should be given to updating the AEMP sediment quality benchmarks for Sheardown Lake NW to reflect not only baseline data, but also reference lake data.

4.3 Sheardown Lake Southeast (DLO-2)

4.3.1 Water Quality

Vertical profiles of *in situ* water temperature, dissolved oxygen, pH and specific conductance conducted at Sheardown Lake SE showed no substantial within-season station-to-station differences during any of the winter, summer, or fall sampling events in 2020 (Appendix Figures C.17 to C.20). Distinctly cooler water temperature was indicated with depth at the Sheardown Lake SE basin in the summer, as well as at depths greater than 10 metres in the fall, that roughly mirrored gradients observed at Reference Lake 3 during both seasons in 2020 (Figure 4.12). The average water temperature at the bottom of the water column at Sheardown Lake SE littoral stations did not differ significantly from that at the reference lake, unlike at profundal stations

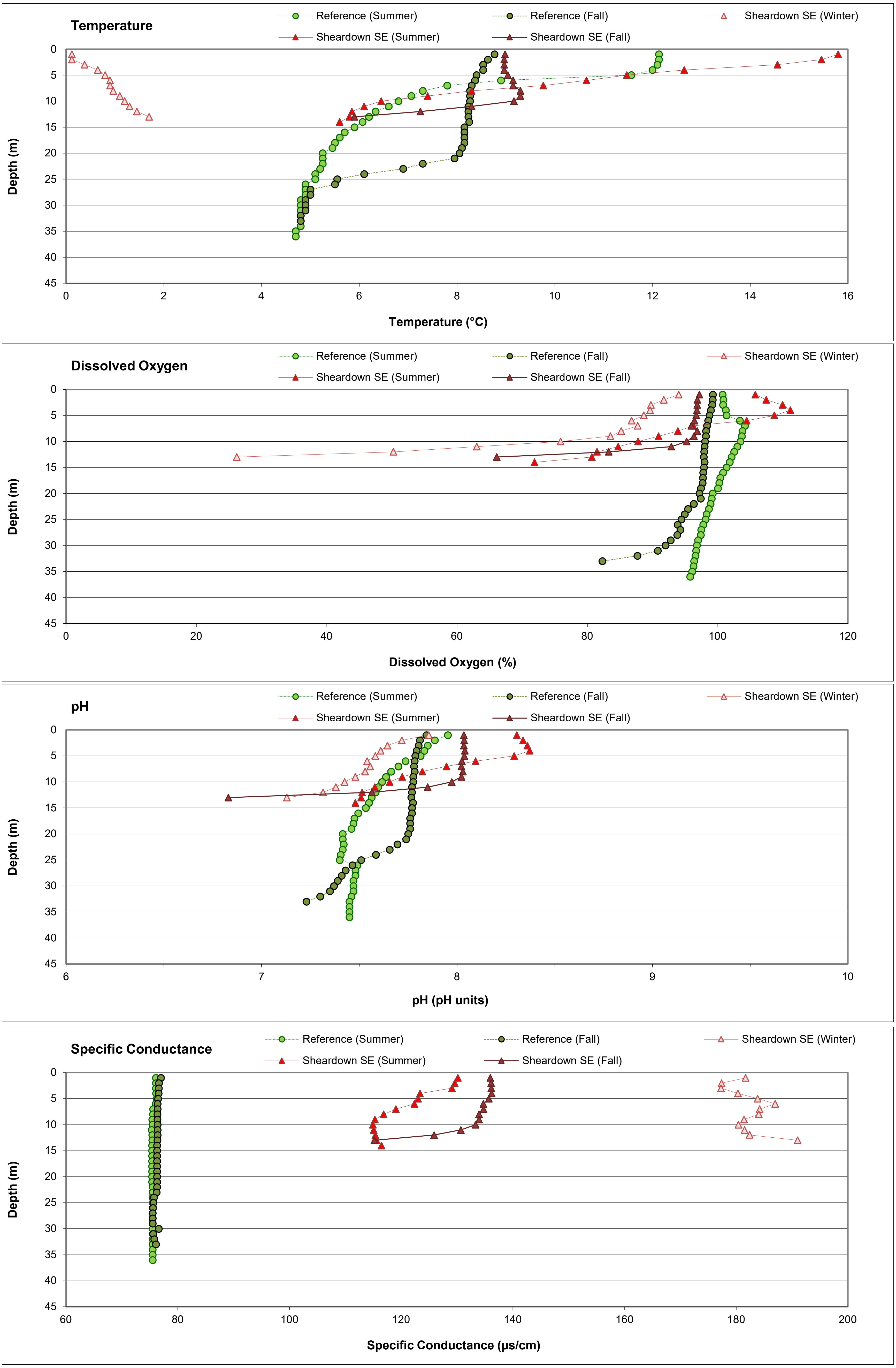


Figure 4.12: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake SE (DLO-02) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

where the water temperature was significantly warmer than at Reference Lake 3 during the August 2020 biological study (Figure 4.5; Appendix Table C.51). Sheardown Lake SE is a smaller and shallower waterbody than Reference Lake 3 (see Figure 2.1; Appendix Table B.1), and therefore heat distribution patterns (i.e., thermal profiles) may be expected to differ naturally between these lakes. Dissolved oxygen profiles conducted at Sheardown Lake SE in 2020 showed a gradient of decreasing saturation levels with increased depth in all but the fall sampling event (Figure 4.12). Dissolved oxygen concentrations near the bottom of the water column were significantly lower at Sheardown Lake SE littoral and profundal stations than like habitat stations at the reference lake during the August 2020 biological study (Figure 4.5). However, mean dissolved oxygen concentrations were above the WQG for the protection of sensitive populations of cold-water species (i.e., 9.5 mg/L) near the bottom at littoral and profundal stations of Sheardown Lake SE at the time of biological monitoring (Figure 4.5; Appendix Table C.51).

Water column profiles showed decreasing pH with increased depth at Sheardown Lake SE and Reference Lake 3 during winter and summer sampling events in 2020, with the changes in pH through the water column at both lakes appearing to coincide with changes in water temperature during each given season (Figure 4.12). The pH near the bottom of the water column at littoral and profundal stations of Sheardown Lake SE were significantly higher than at respective stations at the reference lake during the August 2020 biological study (Figure 4.5). However, the mean incremental difference in bottom pH between lakes was less than a pH unit, and pH values were consistently within WQG limits at Sheardown Lake SE (Figure 4.5; Appendix Table C.51), suggesting that the pH difference between lakes was not ecologically meaningful. Specific conductance at Sheardown Lake SE differed between the bottom and surface of the water column in all seasons (Figure 4.12), and was significantly higher at the littoral and profundal stations of Sheardown Lake SE than at Reference Lake 3 during the August 2020 biological study (Figure 4.5). Secchi depth at Sheardown Lake SE was significantly lower, indicating lower water clarity, than at the reference lake during the August 2020 biological study (Appendix Figure C.8; Appendix Tables C.51). Indeed, water clarity at Sheardown Lake SE was the lowest among all study lakes used for the CREMP in 2020 and historically, which is hypothesized to reflect backflow containing naturally high suspended sediment received from Mary River.

Water chemistry at Sheardown Lake SE met all AEMP benchmarks and WQG over the duration of spring, summer, and fall sampling events in 2020 (Table 4.10). Among those parameters with established AEMP benchmarks, aluminum and nitrate concentrations were elevated by factors greater than three at Sheardown Lake SE compared to Reference Lake 3 during the summer and fall sampling events (Table 4.10; Appendix Table C.43). Of those parameters without AEMP benchmarks, turbidity and total and dissolved manganese, molybdenum, and uranium

Table 4.10: Mean Water Chemistry at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 (n = 3)		Sheardown Lake SE Stations (n = 5)		
					Summer	Fall	Winter	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	79	79	194	128	142
	pH (lab)	pH	6.5 - 9.0	-	7.66	7.75	7.44	8.02	8.01
	Hardness (as $CaCO_3$)	mg/L	-	-	35	38	101	60	68
	Total Suspended Solids (TSS)	mg/L	-	-	2.0	2.0	2.0	2.0	2.2
	Total Dissolved Solids (TDS)	mg/L	-	-	41	51	129	69	77
	Turbidity	NTU	-	-	0.15	0.15	0.23	0.94	2.32
	Alkalinity (as $CaCO_3$)	mg/L	-	-	46	34	79	49	56
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.010	0.014	0.018	0.007	0.013
	Nitrate	mg/L	3	3	0.020	0.020	0.230	0.095	0.089
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.002	0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	0.16	0.15	0.13	0.15
	Dissolved Organic Carbon	mg/L	-	-	3.3	3.5	2.7	2.0	2.1
	Total Organic Carbon	mg/L	-	-	4.6	3.8	2.7	1.7	2.4
	Total Phosphorus	mg/L	0.020^{α}	-	0.004	0.003	0.004	0.006	0.009
	Phenols	mg/L	0.004^{α}	-	0.0010	0.0011	0.001	0.0026	0.001
Anions	Bromide (Br)	mg/L	-	-	0.1	0.1	0.1	0.05	0.1
	Chloride (Cl)	mg/L	120	120	1.4	1.4	5.4	3.3	3.9
	Sulphate (SO_4)	mg/L	218^{β}	218	3.6	3.6	12.4	9.2	9.6
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.0031	0.003	0.006	0.026	0.064
	Antimony (Sb)	mg/L	0.020^{α}	-	0.0001	0.0001	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.0001	0.0001	0.000101	0.0001	0.0001
	Barium (Ba)	mg/L	-	-	0.0064	0.00696	0.01100	0.00641	0.00776
	Beryllium (Be)	mg/L	0.011^{α}	-	0.0005	0.0005	0.0005	0.0001	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0005	0.0005	0.00005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.0116	0.01	0.0114
	Cadmium (Cd)	mg/L	0.00012	0.00009	0.00001	0.00001	0.00001	0.000005	0.00001
	Calcium (Ca)	mg/L	-	-	7.2	7.2	19.1	11.3	13.4
	Chromium (Cr)	mg/L	0.0089	0.0089	0.0005	0.0005	0.0005	0.00013	0.0005
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	0.0001	0.0001	0.0001	0.0001	0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.00073	0.0008	0.0010	0.0008	0.0009
	Iron (Fe)	mg/L	0.300	0.300	0.030	0.030	0.030	0.034	0.064
	Lead (Pb)	mg/L	0.001	0.001	0.00005	0.00005	0.00005	0.00005	0.00006
	Lithium (Li)	mg/L	-	-	0.0010	0.001	0.0013	0.0011	0.0013
	Magnesium (Mg)	mg/L	-	-	4.2	4.7	12.9	6.9	8.3
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00080	0.00068	0.00233	0.00526	0.00361
	Mercury (Hg)	mg/L	0.000026	-	0.000005	0.000005	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00013	0.00015	0.00083	0.00063	0.00064
	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00050	0.00083	0.00059	0.00061
	Potassium (K)	mg/L	-	-	0.9	0.90	1.69	1.03	1.15
	Selenium (Se)	mg/L	0.001	-	0.001	0.001	0.001	0.00005	0.001
	Silicon (Si)	mg/L	-	-	0.50	0.50	0.74	0.55	0.50
	Silver (Ag)	mg/L	0.00025	0.0001	0.00001	0.00001	0.00001	0.00001	0.00001
	Sodium (Na)	mg/L	-	-	0.9	0.96	2.46	1.37	1.71
	Strontium (Sr)	mg/L	-	-	0.0084	0.0082	0.0156	0.0086	0.0102
	Thallium (Tl)	mg/L	0.0008	0.0008	0.0001	0.0001	0.0001	0.00001	0.0001
	Tin (Sn)	mg/L	-	-	0.0001	0.0001	0.0001	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.010	0.010	0.010	0.001	0.010
	Uranium (U)	mg/L	0.015	-	0.00032	0.00033	0.00135	0.00089	0.00118
	Vanadium (V)	mg/L	0.006^{α}	0.006	0.0010	0.001	0.001	0.0005	0.001
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.003	0.004	0.003	0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

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

[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 specific to Sheardown Lake NW.

[d] Benchmark is 0.179 mg/L and 0.173 mg/L for shallow and deep stations, respectively (Intrinsik 2013).

concentrations were elevated at Sheardown Lake SE compared to the reference lake during summer and/or fall sampling events in 2020 (Appendix Tables C.43 and C.54). Similar to the northwest basin of the lake, elevated total aluminum concentrations at Sheardown Lake SE compared to the reference lake in 2020 were associated with influences on water quality of the lake due to backflow received from Mary River that contributed to elevated turbidity at Sheardown Lake SE (Table 4.10; Appendix Table C.43).[9] Similar dissolved aluminum concentrations between Sheardown Lake SE and the reference lake in 2020 (and historically) suggested that the mine was unlikely to be the source of aluminum (Appendix Table C.54). Sulphate was the only parameter among those with established AEMP benchmarks that was elevated in 2020 compared to baseline at Sheardown Lake SE (Appendix Figure C.21), as was dissolved molybdenum concentrations among those parameters without AEMP benchmarks (Appendix Tables C.43 and C.54). Overall, a slight mine-related influence on water quality of Sheardown Lake SE was indicated in 2020 as reflected by elevated concentrations of manganese, molybdenum, nitrate, sulphate, and uranium. With the exception of manganese, concentrations of these parameters were also elevated at Sheardown Lake NW, suggesting a common mine-related source. However, concentrations of all parameters remained well below AEMP benchmarks and WQG at Sheardown Lake SE since commercial mine operations commenced in 2015, and therefore no adverse effects to biota were expected at the southeast basin of Sheardown Lake.

4.3.2 Sediment Quality

Surficial sediment at Sheardown Lake SE was primarily composed of silt with low (i.e., <2%) TOC content (Figure 4.13; Appendix Table D.30). Substrate at littoral stations of Sheardown Lake SE contained significantly less sand, moisture, and TOC content, and significantly greater silt and clay content, compared to littoral stations at Reference Lake 3 (Appendix Table D.31). Sediments from profundal stations at Sheardown Lake SE also had lower TOC and moisture content compared to profundal stations at Reference Lake 3, but no significant differences in particle size occurred (Appendix Table D.31). The relatively high proportion of fines in substrate of Sheardown Lake SE is potentially due to the receipt of Mary River backflow during high flow periods, which can be expected to result in the deposition of high quantities of naturally suspended, fine-grained material. Similar to observations at the other mine-exposed lakes and the reference lake, iron (oxy)hydroxide material was visible in surficial and/or sub-surface substrate at some Sheardown Lake SE stations, in some cases occurring as a thin, distinct layer or floc (Appendix Tables D.28 and D.29). Below the surficial layer, substrates at Sheardown Lake SE exhibited some sporadic blackening suggesting development of reducing conditions; however, no distinct redox boundary was observed (Appendix Tables D.28 and D.29). Observations regarding reducing sediment conditions at Sheardown Lake SE were similar to

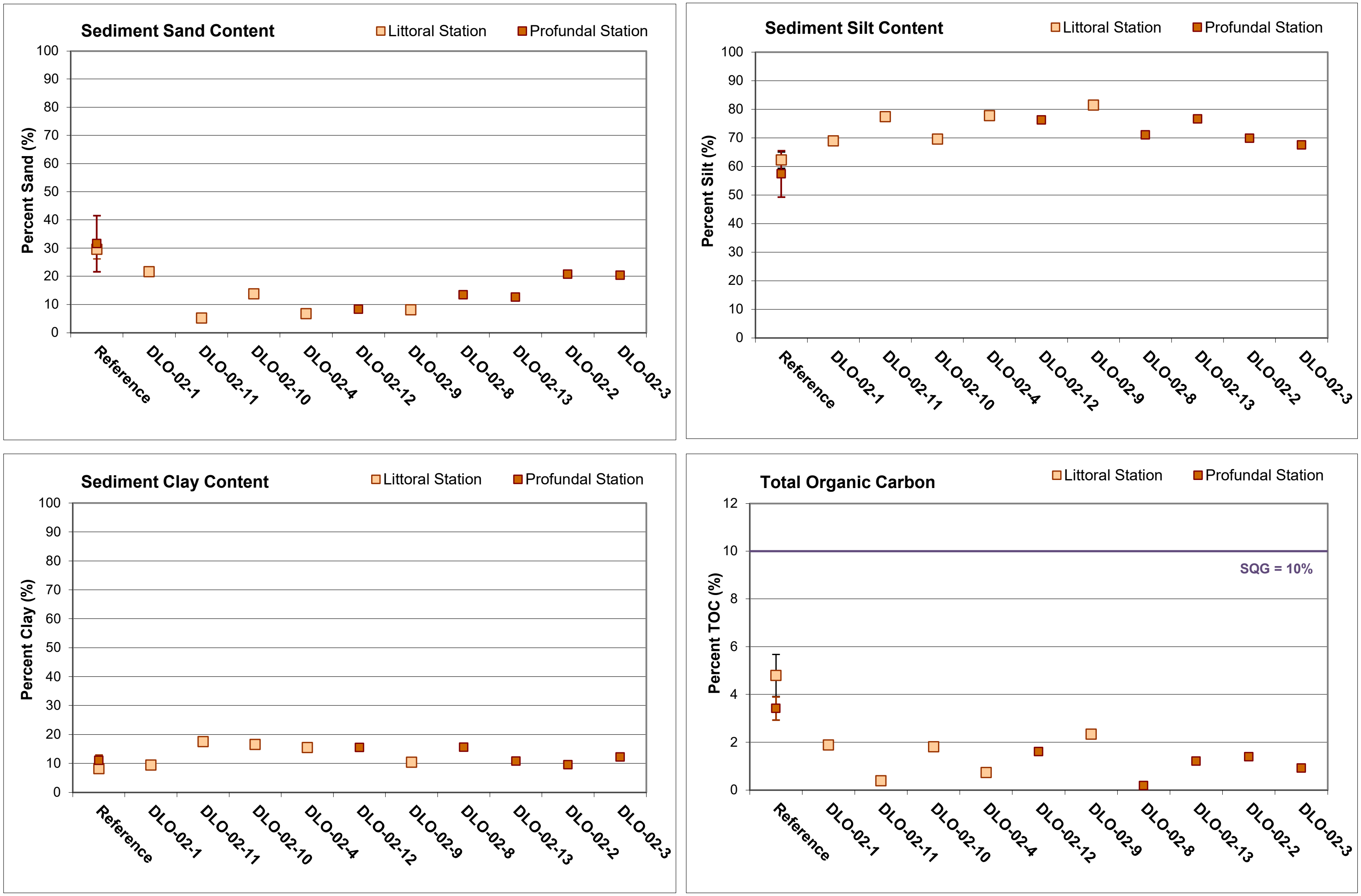


Figure 4.13: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake SE (DLO-02) Sediment Monitoring Stations and Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2020

those made at Reference Lake 3 (Appendix Tables D.3, D.4, D.28, and D.29), suggesting that factors leading to reduced sediment conditions were comparable between lakes.

Metal concentrations in sediment at Sheardown Lake SE showed no clear spatial gradients with progression towards the lake outlet in 2020, suggesting no point sources of metals to the lake (Appendix Table D.30). Metal concentrations were, on average, similar to those observed for like-habitat stations at Reference Lake 3, except for a slight elevation (i.e., 3- to 5-fold) of zirconium concentrations in sediment from Sheardown Lake SE (Table 4.11; Appendix Table D.32). On average, concentrations of iron were above the SQG in Sheardown Lake SE (Table 4.11; Appendix Table D.30). Mean concentrations of iron, as well as chromium and manganese (the latter two metals in sediment from littoral stations only), were also above AEMP benchmarks in sediment in 2020 (Table 4.11; Appendix Table D.30). However, as indicated previously, average concentrations of iron and manganese were also above SQG and AEMP benchmarks at littoral and/or profundal stations of Reference Lake 3 (Table 4.11). This suggested that the elevation of iron and manganese concentrations in sediment relative to SQG and lake-specific AEMP benchmarks may be a natural phenomenon at lakes within the local study area of the mine.

Metal concentrations in sediment of littoral and profundal habitat at Sheardown Lake SE in 2020 were comparable to concentrations observed in the mine baseline period (2005 to 2013), and were also within respective ranges observed in previous years of mine operations (i.e., 2015 to 2019; Figure 4.7; Appendix Table D.32).[14] Thus, no substantial changes to metal concentrations in sediments at Sheardown Lake SE were indicated since the commencement of commercial mine operations in 2015.

4.3.3 Phytoplankton

Chlorophyll-a concentrations at Sheardown Lake SE showed no spatial gradients with closer proximity to the lake outlet during any of the winter, summer, or fall sampling events in 2020 (Figure 4.8). Chlorophyll-a concentrations did not differ significantly between the summer and fall sampling events in 2020, but concentrations in winter were significantly lower than the two open-water seasons (Appendix Table E.6). Similar to Sheardown Lake NW, chlorophyll-a concentrations at Sheardown Lake SE were significantly greater than at the reference lake for both the summer and fall sampling events in 2020 (Appendix Table E.7 and E.8), but concentrations were well below the AEMP benchmark of 3.7 µg/L at all stations and for all sampling events (Figure 4.8). On average, chlorophyll-a concentrations at Sheardown Lake SE

[14] See footnote 8 regarding differences in the concentration of boron in sediment between baseline and recent CREMP studies.

Table 4.11: Sediment Particle Size, Total Organic Carbon, and Metal Concentrations at Sheardown Lake SE (DLO-02), and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2020

Parameter		Units	SQG[a]	AEMP Benchmark[b] (NW, SE)	Littoral		Profundal	
					Reference Lake (n = 5)	Sheardown Lake SE (n = 3)	Reference Lake (n = 5)	Sheardown Lake SE (n = 2)
					Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC		%	10^{α}	-	4.80 ± 1.96	1.43 ± 0.828	3.42 ± 1.08	1.06 ± 0.332
Metals	Aluminum (Al)	mg/kg	-	-	16,880 ± 1,785	17,400 ± 1,114	21,800 ± 2,185	17,550 ± 1,909
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
	Arsenic (As)	mg/kg	17	6.2, 5.9	3.53 ± 1.09	3.67 ± 0.326	4.07 ± 0.397	3.03 ± 0.120
	Barium (Ba)	mg/kg	-	-	117 ± 22.0	92 ± 16	122 ± 18.3	74 ± 8.2
	Beryllium (Be)	mg/kg	-	-	0.65 ± 0.073	0.87 ± 0.079	0.80 ± 0.092	0.84 ± 0.078
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.24 ± 0.031	<0.20 ± 0	0.22 ± 0.021
	Boron (B)	mg/kg	-	-	12.2 ± 0.853	22.3 ± 3.35	14.7 ± 1.77	20.8 ± 2.12
	Cadmium (Cd)	mg/kg	3.5	1.5, 1.5	0.173 ± 0.047	0.102 ± 0.0106	0.148 ± 0.0172	0.090 ± 0.016
	Calcium (Ca)	mg/kg	-	-	5,608 ± 1,247	7,663 ± 1,413	5,010 ± 407	5,990 ± 170
	Chromium (Cr)	mg/kg	90	97, 79	54.3 ± 4.40	***79.7*** ± 9.13	65.0 ± 6.64	72.3 ± 2.47
	Cobalt (Co)	mg/kg	-	-	10.8 ± 1.64	13.7 ± 0.289	15.2 ± 1.56	13.3 ± 0.990
	Copper (Cu)	mg/kg	110	58, 56	71.4 ± 14.2	28.2 ± 2.26	***83.8*** ± 11.1	26.1 ± 2.62
	Iron (Fe)	mg/kg	$40,000^{\alpha}$	52,200, 34,400	***50,600*** ± 24,939	***42,733*** ± 6,352	45,080 ± 4,440	***41,950*** ± 7,000
	Lead (Pb)	mg/kg	91.3	35	13.8 ± 0.799	17.4 ± 1.93	16.7 ± 1.82	15.4 ± 1.273
	Lithium (Li)	mg/kg	-	-	26.0 ± 2.51	32.6 ± 1.88	33.7 ± 3.83	32.2 ± 2.40
	Magnesium (Mg)	mg/kg	-	-	11,440 ± 814	15,400 ± 2,095	14,180 ± 1,422	13,800 ± 1,414
	Manganese (Mn)	mg/kg	$1,100^{\alpha,\beta}$	4,530, 657	579 ± 258	***925*** ± 206	***1,230*** ± 355	602 ± 60.8
	Mercury (Hg)	mg/kg	0.486	0.17	0.0500 ± 0.0178	0.0215 ± 0.00044	0.0583 ± 0.0164	0.0204 ± 0.0059
	Molybdenum (Mo)	mg/kg	-	-	4.44 ± 3.31	1.50 ± 0.280	2.52 ± 0.273	1.40 ± 0.184
	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	77, 66	40.0 ± 3.52	61.3 ± 8.62	45.0 ± 4.54	54.2 ± 3.18
	Phosphorus (P)	mg/kg	$2,000^{\alpha}$	1,958, 1,278	1,167 ± 394	1,004 ± 55.7	956 ± 47.3	929 ± 24.0
	Potassium (K)	mg/kg	-	-	4,100 ± 453	4,343 ± 340	5,338 ± 543	4,565 ± 728
	Selenium (Se)	mg/kg	-	-	0.73 ± 0.31	0.20 ± 0.01	0.61 ± 0.18	0.21 ± 0.0071
	Silver (Ag)	mg/kg	-	-	0.14 ± 0.047	0.12 ± 0.006	0.20 ± 0.057	0.12 ± 0.0141
	Sodium (Na)	mg/kg	-	-	304.2 ± 32	279 ± 31	369 ± 50	261 ± 13.4
	Strontium (Sr)	mg/kg	-	-	11.6 ± 1.70	11.7 ± 1.277	12.3 ± 1.24	10.7 ± 0.778
	Sulphur (S)	mg/kg	-	-	1,400 ± 387	<1,000 ± 0	1,140 ± 195	<1,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.379 ± 0.0415	0.415 ± 0.0598	0.594 ± 0.094	0.364 ± 0.0361
	Tin (Sn)	mg/kg	-	-	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0
	Titanium (Ti)	mg/kg	-	-	1,006 ± 109	1,303 ± 85	1,136 ± 50	1,290 ± 0.0
	Uranium (U)	mg/kg	-	-	11.0 ± 2.41	5.03 ± 0.431	19.7 ± 3.76	4.86 ± 0.658
	Vanadium (V)	mg/kg	-	-	54.1 ± 5.40	51.0 ± 3.97	63.4 ± 4.89	48.3 ± 1.63
	Zinc (Zn)	mg/kg	315	135	73.1 ± 7.83	57.3 ± 1.85	83.8 ± 8.52	54.8 ± 6.43
	Zirconium (Zr)	mg/kg	-	-	4.5 ± 1.0	17.6 ± 1.8	3.9 ± 0.32	17.9 ± 1.8

Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV

[b] AEMP Sediment Quality Benchmarks developed by Intrinsik (2013) using sediment quality guidelines, background sediment quality data, and method detection limits. The indicated values are specific to the Sheardown Lake basins

indicated an 'oligotrophic' status as defined by Wetzel (2001). This trophic status classification was consistent with an oligotrophic categorization for Sheardown Lake SE based on CWQG (CCME 2020) trophic classifications as defined by total phosphorus concentrations (i.e., average concentrations below 10 µg/L; Table 4.10; Appendix Table C.54).

Chlorophyll-a concentrations did not indicate any consistent direction of significant differences between 2020 and the mine construction (2014) period or previous years of mine operation (2015 to 2019) for winter, summer, and fall seasons (Figure 4.14; Appendix Table E.13). The variability in chlorophyll-a concentrations among years at Sheardown Lake SE may reflect the combination of mine-related influences and variable influence of Mary River on Sheardown Lake SE water levels, hydraulic retention time, and/or chemistry among years/seasons. For instance, Mary River discharges into or drains Sheardown Lake SE during high and low flow periods, respectively, the nature of which may affect phytoplankton abundance and/or community structure. No chlorophyll-a baseline (2005 to 2013) data are available for Sheardown Lake SE, precluding comparisons to conditions prior to the mine construction period.

4.3.4 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral and profundal habitats, and richness was significantly higher at profundal habitat, of Sheardown Lake SE compared to like-habitat stations at Reference Lake 3, the differences of which were at magnitudes outside of the CES_{BIC} of ±2 SD_{REF} (Tables 4.12 and 4.13). In addition to these differences, benthic invertebrate community compositional differences were indicated between Sheardown Lake SE and Reference Lake 3 based on significantly differing Bray-Curtis Index for both littoral and profundal habitat types (Appendix Table F.21). However, the only ecologically significant differences in dominant taxonomic groups included higher and lower relative abundance of Chironomidae and Ostracoda, respectively, at littoral stations of Sheardown Lake SE compared to Reference Lake 3 (Table 4.12). No differences in dominant taxonomic groups were indicated between Sheardown Lake SE and the reference lake in 2020 (Tables 4.12 and 4.13). Similar to Sheardown Lake NW, the occurrence of higher benthic invertebrate density without an accompanying difference in evenness or existence of a significantly lower relative abundance of metal-sensitive taxa suggested that Sheardown Lake SE was simply more productive than Reference Lake 3, and was not adversely influenced by mine operations in 2020.

Similar to the other mine-exposed lakes, sediment at littoral and profundal stations of Sheardown Lake SE was significantly more compact (i.e., lower moisture content) than like-habitat stations at the reference lake (Appendix Table F.39). The occurrence of more stable, compact sediment likely accounted for an ecologically significant higher relative abundance of the burrower HPG at Sheardown Lake SE compared to the reference lake (Tables 4.12 and 4.13). In addition to

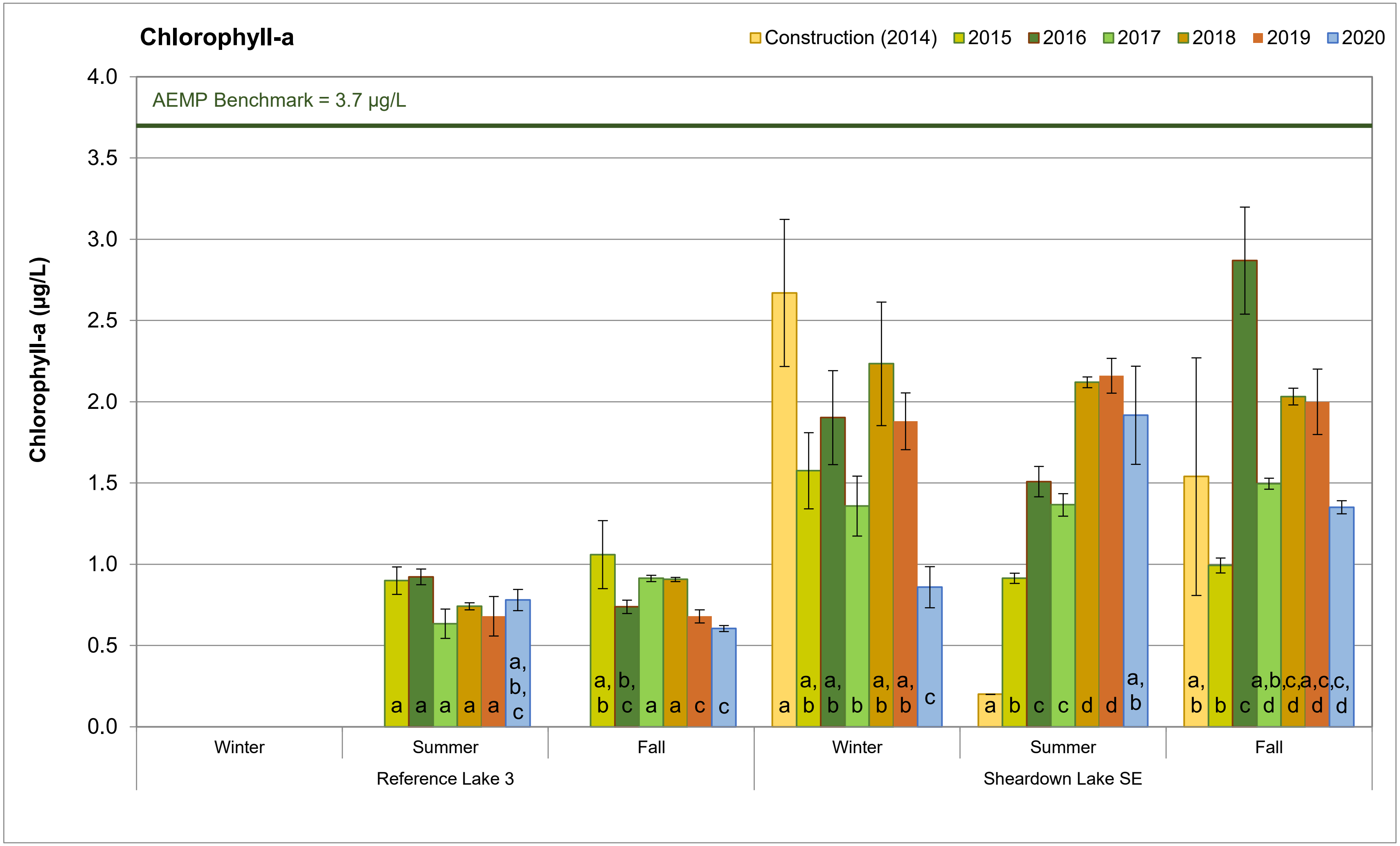


Figure 4.14: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Sheardown Lake SE and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2020) Periods (mean ± SE)

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

Table 4.12: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake SE (DLO-02) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-equal	log10	YES	< 0.001	8.9	Reference Lake 3	1,571	430	193	1,190	1,474	2,310
						Sheardown SE Littoral	5,407	1,391	622	3,216	5,509	6,914
Richness (Number of Taxa)	t-equal	log10	NO	0.305	-1.0	Reference Lake 3	14.6	2.5	1.1	13.0	14.0	19.0
						Sheardown SE Littoral	12.2	1.6	0.7	10.0	13.0	14.0
Simpson's Evenness (E)	t-equal	none	NO	0.799	0.1	Reference Lake 3	0.810	0.110	0.049	0.630	0.847	0.923
						Sheardown SE Littoral	0.826	0.068	0.031	0.722	0.827	0.911
Shannon Diversity	t-equal	log10	NO	0.547	-0.4	Reference Lake 3	2.710	0.526	0.235	2.080	2.730	3.510
						Sheardown SE Littoral	2.520	0.261	0.117	2.220	2.570	2.830
Hydracarina (%)	t-equal	log10	NO	0.167	-0.7	Reference Lake 3	5.3	2.6	1.2	3.5	4.4	9.9
						Sheardown SE Littoral	3.4	2.4	1.1	1.0	2.7	5.9
Ostracoda (%)	t-equal	log10	YES	< 0.001	-2.1	Reference Lake 3	37.9	14.5	6.5	26.7	36.2	62.6
						Sheardown SE Littoral	6.8	3.5	1.6	2.3	8.1	10.3
Chironomidae (%)	t-equal	none	YES	0.001	2.3	Reference Lake 3	52.6	15.6	7.0	26.9	59.0	66.4
						Sheardown SE Littoral	89.2	5.1	2.3	81.9	89.0	96.4
Metal-Sensitive Chironomidae (%)	t-equal	none	YES	0.012	-1.7	Reference Lake 3	28.8	9.5	4.3	15.6	32.5	38.7
						Sheardown SE Littoral	12.3	6.5	2.9	3.0	15.1	19.5
Collector-Gatherers (%)	t-equal	log10	NO	0.258	-0.8	Reference Lake 3	63.1	11.4	5.1	53.6	60.3	81.5
						Sheardown SE Littoral	54.0	14.9	6.7	40.3	50.6	75.3
Filterers (%)	t-equal	none	YES	0.022	-1.5	Reference Lake 3	27.1	9.8	4.4	14.4	29.2	38.0
						Sheardown SE Littoral	12.3	6.5	2.9	3.0	15.1	19.5
Shredders (%)	t-unequal	none	YES	0.060	-1.2	Reference Lake 3	3.9	3.3	1.5	0.6	3.2	7.4
						Sheardown SE Littoral	0.0	0.0	0.0	0.0	0.0	0.0
Clingers (%)	t-equal	none	YES	0.011	-1.8	Reference Lake 3	31.9	9.3	4.2	17.9	33.5	41.6
						Sheardown SE Littoral	15.5	6.4	2.9	4.7	18.1	20.9
Sprawlers (%)	t-equal	log10	NO	0.267	-0.7	Reference Lake 3	57.9	12.1	5.4	41.0	57.2	73.8
						Sheardown SE Littoral	49.7	9.0	4.0	42.4	46.5	65.3
Burrowers (%)	t-equal	none	YES	0.004	5.0	Reference Lake 3	10.2	4.9	2.2	4.6	8.3	17.3
						Sheardown SE Littoral	34.8	13.1	5.9	16.6	35.1	52.9

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

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

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

Table 4.13: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake SE (DLO-02) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-equal	log10	YES	< 0.001	23.9	Reference Lake 3	479	142	63	336	491	681
						Sheardown SE Profundal	3,869	1,304	583	3,026	3,336	6,164
Richness (Number of Taxa)	t-equal	none	YES	0.034	2.4	Reference Lake 3	7.0	1.9	0.8	5.0	8.0	9.0
						Sheardown SE Profundal	11.4	3.4	1.5	6.0	12.0	15.0
Simpson's Evenness (E)	t-equal	none	NO	0.429	0.9	Reference Lake 3	0.731	0.045	0.020	0.689	0.721	0.795
						Sheardown SE Profundal	0.772	0.099	0.045	0.619	0.806	0.869
Shannon Diversity	t-equal	none	NO	0.101	2.3	Reference Lake 3	1.800	0.196	0.088	1.580	1.720	2.030
						Sheardown SE Profundal	2.250	0.501	0.224	1.670	2.570	2.670
Hydracarina (%)	t-equal	none	NO	0.434	-0.4	Reference Lake 3	2.8	2.0	0.9	0.0	3.5	5.1
						Sheardown SE Profundal	1.9	1.3	0.6	0.3	2.5	3.1
Ostracoda (%)	t-equal	log10	NO	0.177	-0.8	Reference Lake 3	8.6	4.1	1.8	3.5	7.7	14.5
						Sheardown SE Profundal	5.3	5.1	2.3	0.5	5.7	13.0
Chironomidae (%)	t-equal	none	NO	0.202	1.1	Reference Lake 3	87.9	4.2	1.9	82.3	87.2	92.7
						Sheardown SE Profundal	92.5	6.2	2.8	84.1	91.2	98.9
Metal-Sensitive Chironomidae (%)	t-equal	none	NO	0.191	-0.7	Reference Lake 3	31.5	17.6	7.9	7.9	38.0	49.3
						Sheardown SE Profundal	19.9	4.7	2.1	11.8	21.6	24.0
Collector-Gatherers (%)	t-equal	log10	NO	0.292	-0.7	Reference Lake 3	62.9	15.0	6.7	45.4	56.1	79.0
						Sheardown SE Profundal	52.5	14.5	6.5	33.6	52.1	70.6
Filterers (%)	t-equal	none	NO	0.212	-0.6	Reference Lake 3	30.7	17.5	7.8	7.9	38.0	49.3
						Sheardown SE Profundal	19.7	4.7	2.1	11.8	21.6	23.8
Clingers (%)	t-equal	none	NO	0.142	-0.8	Reference Lake 3	33.5	16.9	7.6	13.1	41.5	52.8
						Sheardown SE Profundal	20.5	5.3	2.4	12.3	21.7	26.9
Sprawlers (%)	t-equal	log10	YES	0.041	-1.7	Reference Lake 3	64.8	16.2	7.2	45.5	58.5	87.0
						Sheardown SE Profundal	36.9	17.8	8.0	16.8	37.4	55.1
Burrowers (%)	t-unequal	log10(x+1)	YES	0.007	14.2	Reference Lake 3	1.7	2.9	1.3	0.0	0.0	6.7
						Sheardown SE Profundal	42.6	21.3	9.5	19.4	43.0	66.3

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

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

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

differences in sediment properties between lakes, significantly shallower ‘profundal’ sampling depths at Sheardown Lake SE likely contributed to the differences in benthic invertebrate community features compared to the reference lake (Appendix Table F.39). Natural depth-related influences on benthic invertebrate community structure that include lower density and richness at greater depth in lake environments are well documented (Ward 1992; Armitage et al. 1995), and were consistently evident at Reference Lake 3 from 2015 to 2020 (Appendix B) indicating similar patterns in pristine lakes of the Mary River Project region. The maximum depth of Sheardown Lake SE is approximately 14 m (Minnow 2018). Because profundal habitat for the Mary River Project CREMP is defined as water depths ≥12 m, benthic invertebrate community data collected from profundal depths of Sheardown Lake SE (average station depth of 12 m) are not directly comparable to those at Reference Lake 3, where the mean depth of profundal stations is 21 m (Appendix Table F.39). Therefore, the differences in benthic invertebrate community endpoints shown between Sheardown Lake SE and the reference lake in 2020 likely reflected a combination of naturally greater productivity, naturally more compact sediment, and naturally shallower ‘profundal’ sampling depths at Sheardown Lake SE. Moreover, no ecologically significant effects on the relative abundance of metal-sensitive Chironomidae were indicated at Sheardown Lake SE in 2020, suggesting no metal-related influences on the benthic invertebrate community of the lake.

Benthic invertebrate density was routinely significantly lower at Sheardown Lake SE in years of mine operation compared to baseline (Appendix Tables F.41 and F.42). However, no ecologically significant differences in richness, evenness, relative abundance of dominant taxonomic groups, or relative abundance of HPG were consistently shown at littoral or profundal habitat of Sheardown Lake SE over the 2015 to 2020 period of mine operation compared to baseline (Appendix Figures F.12 and F.13; Appendix Tables F.41 and F.42). Because density was the only benthic invertebrate community metric that consistently differed between the mine-operational and baseline period, natural temporal variability among studies (and in particular, high density during the 2007 and 2013 baseline studies) likely accounted for the difference in benthic invertebrate density at Sheardown Lake SE between these periods. Overall, consistent with no substantial differences in water and sediment quality since the mine baseline period, no ecologically significant differences in benthic invertebrate community features were indicated at littoral and profundal habitat of Sheardown Lake SE following the commencement of mine operation in 2015.

4.3.5 Fish Population

4.3.5.1 Sheardown Lake SE Fish Community

The Sheardown Lake SE fish community was composed of arctic charr and ninespine stickleback in 2020 (Table 4.8), the same fish species as observed in previous years (Minnow 2020). Total fish CPUE was greater at Sheardown Lake SE than Reference Lake 3, suggesting higher densities and/or productivity of both arctic charr and ninespine stickleback at Sheardown Lake SE (Table 4.8). Consistent with the other mine-exposed lakes, greater numbers of arctic charr, together with greater density of benthic invertebrates, suggested that productivity was higher at Sheardown Lake SE than at Reference Lake 3. Electrofishing and gill netting CPUE at Sheardown Lake SE in 2020 were both within respective ranges observed during the previous five years of mine operation (i.e., 2015 to 2019), and were generally greater than in baseline studies (i.e., 2006 through 2008; Figure 4.10). Based on these data, arctic charr abundance at nearshore and littoral/profundal habitats may be comparable to, or potentially greater than, baseline at Sheardown Lake SE indicating that the mine has not adversely influenced the number of arctic charr in the lake.

4.3.5.2 Sheardown Lake SE Fish Population Assessment

Nearshore Arctic Charr

A total of 100 arctic charr were captured from nearshore habitat at each of Sheardown Lake SE and Reference Lake 3 in August 2020. Arctic charr YOY were distinguished from non-YOY using fork length cut-offs of 5.1 cm and 4.3 cm for the Sheardown Lake SE and Reference Lake 3 data sets, respectively, based on evaluation of length-frequency distributions coupled with supporting age determinations (Figure 4.15; Appendix Tables G.4 and G.19). Because greater than ten YOY arctic charr were identified from the Sheardown Lake SE and Reference Lake 3 populations, statistical comparisons of health endpoints were completed separately on both the YOY and non-YOY populations. The length-frequency distribution for the whole population of nearshore arctic charr did not differ significantly between Sheardown Lake SE and Reference Lake 3; however, a difference was noted when comparing non-YOY between the two lakes (Appendix Table G.20). This difference reflected slightly larger non-YOY fish captured at Sheardown Lake SE compared to the reference lake (Figure 4.15). Arctic charr YOY and non-YOY from nearshore areas of Sheardown Lake SE were significantly larger and had greater condition than those from Reference Lake 3 (Table 4.14; Appendix Table G.20). The absolute magnitudes of difference in condition were greater than the CES_C of 10% for both age classes at Sheardown Lake SE, suggesting that the differences may be ecologically significant (Table 4.14; Appendix Table G.20).

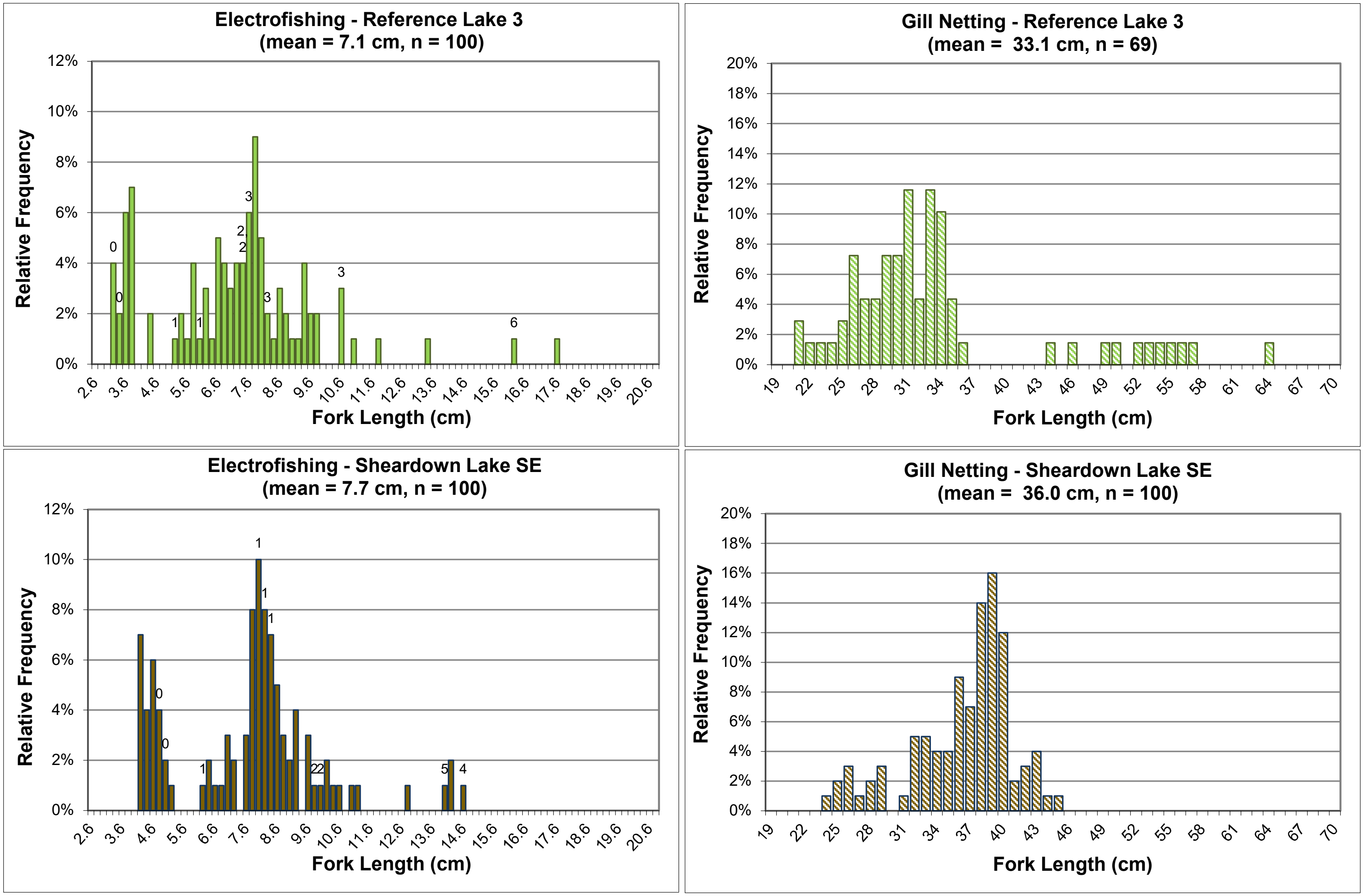


Figure 4.15: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2020

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

Table 4.14: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake SE and Reference Lake 3 from 2015 to 2020, and between Sheardown Lake SE 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 Sheardown Lake SE baseline period data [b]					
			2015	2016	2017	2018	2019	2020	2015	2016	2017	2018	2019	2020
Nearshore Electrofishing	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	-	-	**Yes (+273%)**	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	No	No	**Yes (+12%)**	**Yes (+21%)**	**Yes (-28%)**	**Yes (+7%)**	**Yes (+7%)**	**Yes (-15%)**	**Yes (+19%)**	**Yes (-47%)**	No	**Yes (+30%)**
		Size (mean weight)	No	No	**Yes (+55%)**	**Yes (+59%)**	**Yes (-59%)**	**Yes (+53%)**	No	**Yes (-43%)**	**Yes (+54%)**	No	No	**Yes (+117%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+4%)**	No	**Yes (+9%)**	**Yes (-13%)**	**Yes (+4%)**	**Yes (+14%)**	**Yes (-14%)**	**Yes (-16%)**	No	**Yes (-15%)**	**Yes (-13%)**	No
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	No
		Age	-	-	-	-	-	-	**Yes (-13%)**	No	No	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (+23%)**	**Yes (+21%)**	**Yes (-9%)**	**Yes (-7%)**	**Yes (-5%)**	**Yes (-4%)**	**Yes (-2%)**	No
		Size (mean weight)	-	-	-	No	**Yes (+102%)**	**Yes (+107%)**	**Yes (-26%)**	**Yes (-20%)**	**Yes (-16%)**	**Yes (-16%)**	**Yes (-11%)**	**Yes (-7.0%)**
		Growth (fork length-at-age)	-	-	-	-	-	-	No	No	No	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	**Yes (+18%)**	**Yes (+24%)**	No	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+7%)**	No	**Yes (+14%)**	No	No	**Yes (-6%)**	**Yes (-7%)**	**Yes (-6%)**	**Yes (-5.0%)**

BOLD indicates a significant difference related to the comparison.

[a] Values in parentheses indicate direction and magnitude of any significant differences.

[b] Baseline period data included 2007 nearshore electrofishing data and 2007 and 2008 littoral/profundal gill netting data.

[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.

No consistent directional differences in size or condition were observed in non-YOY arctic charr from nearshore habitat of Sheardown Lake SE compared to the reference lake from 2015 to 2020, although most often larger fish of slightly greater condition occurred at Sheardown Lake SE over this period (Table 4.14). Although before-after analysis of data collected at Sheardown Lake SE in 2020 (mine operation) and 2007 (baseline) was conducted (Appendix Table G.7), poor accuracy in fresh body weight measurements during baseline sampling precluded meaningful data interpretation, and therefore these results were not discussed herein. Overall, the differences in nearshore non-YOY arctic charr size and condition between Sheardown Lake SE and Reference Lake 3 likely reflected natural variability between the two populations over time.

Littoral/Profundal Arctic Charr

A total of 100 and 69 arctic charr were sampled from littoral/profundal habitat of Sheardown Lake SE and Reference Lake 3, respectively, in August 2020. The length-frequency distribution for littoral/profundal arctic charr differed significantly between lakes due to more larger fish being captured at Sheardown Lake SE (Table 4.14; Figure 4.15). Littoral/profundal arctic charr from Sheardown Lake SE were significantly longer, heavier, and had greater condition than those from Reference Lake 3 (Table 4.14; Appendix Table G.24). The absolute magnitude of difference in condition was also above the CES_C of 10%, suggesting potential ecological significance (Table 4.14; Appendix Table G.20).

The differences in size and condition of arctic charr captured from littoral/profundal habitat between Sheardown Lake NW and Reference Lake 3 in 2020 were similar to the differences shown in 2018 and/or 2019, suggesting no appreciable changes in health of littoral/profundal arctic charr at Sheardown Lake NW over time. No difference in length-frequency distribution of arctic charr captured from littoral/profundal habitat of Sheardown Lake SE was shown between 2020 and baseline, although differences were reported historically (Table 4.14). Arctic charr sampled from littoral/profundal habitat of Sheardown Lake SE in years of mine operation (i.e., 2015 to 2020) have almost consistently been smaller than those captured at the time of the mine baseline, but significantly lower condition has only occurred compared to baseline since 2017 (Table 4.14). Differences in arctic charr condition from 2015 to 2020 at Sheardown Lake SE, relative to Reference Lake 3 or Sheardown Lake SE baseline, were generally absent or not ecologically meaningful based on the magnitude of difference being within the CES_C of ±10% (Table 4.14). In turn, this suggested no adverse influences on adult arctic charr at Sheardown Lake SE through the first six years of mine operation.

4.3.6 Effects Assessment and Recommendations

At Sheardown Lake SE, the following AEMP benchmarks were exceeded in 2020:

- Chromium concentration in sediment, on average, was greater than the benchmark of 79 mg/kg at littoral stations;
- Iron concentration in sediment, on average, was greater than the benchmark of 34,400 mg/kg at littoral and profundal stations;
- Manganese concentration in sediment was, on average, greater than the benchmark of 657 mg/kg at littoral stations; and,
- Nickel concentration in sediment was greater than the benchmark of 66 mg/kg at one littoral monitoring station (DL0-02-11), although the average concentration of nickel in sediment at littoral stations was below this benchmark.

No AEMP benchmarks for water quality were exceeded over the duration of spring, summer, and fall sampling events in 2020 at Sheardown Lake SE. Lake-specific AEMP benchmarks for sediment quality were exceeded for chromium, iron, manganese, and nickel concentrations at Sheardown Lake SE in 2020. However, none of these metals occurred at concentrations in sediment of Sheardown Lake SE that were elevated compared to the reference lake, or to concentrations shown at Sheardown Lake SE during the baseline period. In addition, concentrations of these metals were above the Sheardown Lake SE AEMP benchmarks in sediment at the reference lake, suggesting naturally high concentrations of each of the indicated metals in sediments of area lakes. Notably, AEMP benchmarks established for sediment quality at Sheardown Lake SE tend to be lower than SQG, and are generally lower than AEMP benchmarks established for the other mine-exposed lakes (Baffinland 2015). No adverse effects to phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at Sheardown Lake SE in 2020 based on comparisons to reference conditions and to applicable Sheardown Lake SE baseline conditions. Because no mine-related changes in metal concentrations occurred in sediment at Sheardown Lake SE in 2020 and no adverse effects to biota were associated with concentrations of metals above AEMP benchmarks for sediment quality, a low action response is recommended to meet obligations under the AEMP Management Response Framework. Specifically, it is recommended that the relevance of site-specific sediment quality AEMP benchmarks for Sheardown Lake SE be assessed and, if necessary, determined anew taking into consideration data from the reference lake and applicable SQG.

5 MARY RIVER AND MARY LAKE SYSTEM

5.1 Mary River Tributary-F

5.1.1 Water Quality

Mary River Tributary-F (MRTF) dissolved oxygen concentrations did not differ significantly between areas located downstream and upstream of the MS-08 effluent discharge channel (effluent-exposed and reference areas, respectively) and were well above the WQG lowest acceptable concentration for sensitive early life stages of cold-water biota (i.e., 9.5 mg/L) at both areas at the time of EEM sampling (i.e., August 2020). Although pH and specific conductance were each significantly higher at the effluent-exposed area than at the reference area of MRTF, the mean incremental difference between areas for each of these parameters was very small and pH values were well within the WQG acceptable range for the protection of aquatic life (i.e., between 6.5 and 9.0) at the time of EEM sampling. The proportion of effluent within MRTF immediately below the effluent channel confluence was estimated as 2.6% on average, under flow conditions at the time of EEM sampling, based on extrapolation using measures of specific conductance collected in the field (Minnow 2021).

Water chemistry at MRTF met all AEMP benchmarks over the duration of spring, summer, and fall sampling events in 2020 (Table 5.1). Although concentrations of total aluminum and phosphorus were above applicable WQG in spring at the effluent-exposed area of MRTF (i.e., Station F0-01), concentrations of these parameters were also above WQG at reference areas indicating naturally elevated concentrations of aluminum and phosphorus within regional watercourses (Table 5.1). Among those parameters with established AEMP benchmarks, nitrate and sulphate concentrations were consistently elevated at MRTF compared to the Mary River reference area (i.e., G0-09 series stations) in all spring, summer, and fall sampling events (Appendix Tables C.59 and C.61), but remained at concentrations well below AEMP benchmarks and WQG (Table 5.1). Nitrate and sulphate concentrations were also elevated in summer and fall sampling events from 2018 to 2020 compared to baseline at MRTF (Appendix Figure C.23). No other parameters were observed at concentrations that were continually elevated at MRTF in 2020 compared to reference conditions through all seasons, nor compared to baseline, except that total and dissolved concentrations of manganese were elevated compared to reference conditions during the spring sampling event in 2020 (Appendix Tables C.59 and C.61; Appendix Figure C.23). Overall, a slight mine-related influence on water quality was indicated by elevated concentrations of nitrate and sulphate at MRTF in 2020, but concentrations of these parameters (and all others) were routinely well below AEMP benchmarks and WQG since commercial mine operations commenced in 2015.

Table 5.1: Mean Water Chemistry at Mary River Monitoring Stations During Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	G0-09 Reference (n = 3)			G0 Upstream (n = 2)			Mary River Tributary F			E0 Adjacent (n = 4)			C0 Downstream (n = 3)		
					Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Conventionals	Conductivity (lab)	umho/cm	-	-	61	186	248	45	171	228	108	345	403	53	178	245	57	176	234
	pH (lab)	pH	6.5 - 9.0	-	7.74	8.22	8.33	7.64	8.14	8.48	7.89	8.30	8.32	7.67	8.17	8.19	7.66	8.22	8.17
	Hardness (as $CaCO_3$)	mg/L	-	-	28	81	120	20	74	106	49.9	166	211	24	77	117	25	75	112
	Total Suspended Solids (TSS)	mg/L	-	-	2	12.6	2.1	14.6	10.25	2.8	9.8	<2.0	<2.0	8.5	12.5	2.5	7.7	15.1	2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	66	97	129	66	103	119	90	190	224	74	116	128	73	107	127
	Turbidity	NTU	-	-	4.3	26.3	2.8	7.3	34.5	4.7	2.2	0.4	0.6	6.3	38.6	6.2	8.5	29.5	3.3
	Alkalinity (as $CaCO_3$)	mg/L	-	-	29	80	103	21	78	91	39	122	132	27	83	95	24	82	94
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.010	0.010	0.01	0.01	0.01	0.01	<0.010	<0.010	<0.010	0.010	0.010	0.010	0.010	0.010	0.011
	Nitrate	mg/L	3	3	0.021	0.147	0.103	0.030	0.101	0.163	0.187	0.714	1.090	0.043	0.133	0.259	0.089	0.171	0.229
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.005	0.005	0.005	<0.0050	<0.0050	<0.0050	0.005	0.005	0.005	0.005	0.005	0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	0.15	0.15	0.15	0.15	0.15	<0.15	0.32	0.16	0.15	0.15	0.17	0.15	0.15	0.15
	Dissolved Organic Carbon	mg/L	-	-	1.6	4.0	2.6	2.3	4.0	2.3	2.1	1.7	2.0	2.2	3.0	2.2	2.2	2.6	2.3
	Total Organic Carbon	mg/L	-	-	2.2	2.9	2.4	2.7	2.8	2.3	2.9	2.7	2.3	2.9	3.2	2.3	3.3	3.3	2.6
	Total Phosphorus	mg/L	0.020[α]	-	0.0121	0.0215	0.0034	0.0557	0.0225	0.0038	0.0377	<0.0030	<0.0030	0.0480	0.0240	0.0046	0.0365	0.0230	0.0073
	Phenols	mg/L	0.004[α]	-	0.0015	0.0012	0.0010	0.0010	0.0010	0.0010	<0.0010	<0.0010	<0.0010	0.0012	0.0010	0.0010	0.0011	0.0010	0.0010
Anions	Bromide (Br)	mg/L	-	-	0.1	0.1	0.1	0.1	0.1	0.1	<0.10	<0.10	<0.10	0.1	0.1	0.1	0.1	0.1	0.1
	Chloride (Cl)	mg/L	120	120	1.0	8.9	11.8	1.6	7.8	13.2	1.3	13.7	11.7	1.1	7.6	12.7	1.4	7.5	11.7
	Sulphate (SO_4)	mg/L	218[β]	218	0.7	5.5	6.7	0.6	4.9	7.0	11.8	36.0	60.7	2.9	6.4	12.1	2.6	6.1	10.4
Total Metals	Aluminum (Al)	mg/L	0.100	0.966	0.121	***1.046***	0.087	0.230	***1.330***	0.148	0.170	0.034	0.041	0.216	***1.071***	0.173	0.204	***1.202***	0.121
	Antimony (Sb)	mg/L	0.020[α]	-	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	<0.00010	<0.00010	<0.00010	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.00011	0.00021	0.0001	0.000115	0.000245	0.00010	<0.00010	<0.00010	<0.00010	0.00010	0.00022	0.00010	0.00010	0.00023	0.00010
	Barium (Ba)	mg/L	-	-	0.0049	0.0159	0.0140	0.0052	0.0178	0.0148	0.0054	0.0165	0.0179	0.0048	0.0160	0.0158	0.0051	0.0156	0.0148
	Beryllium (Be)	mg/L	0.011[α]	-	0.0005	0.0001	0.0005	0.0005	0.0001	0.0005	<0.00050	<0.00010	<0.00050	0.0005	0.0001	0.0005	0.0005	0.0001	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.00005	0.0005	0.0005	0.00005	0.0005	<0.00050	<0.000050	<0.00050	0.0005	0.00005	0.0005	0.0005	0.00005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.01	0.01	0.01	0.01	<0.010	<0.010	<0.010	0.01	0.01	0.01	0.01	0.01	0.01
	Cadmium (Cd)	mg/L	0.00012	0.00006	0.000028	0.000005	0.000010	0.00001	0.00001	0.00001	<0.000010	<0.0000050	<0.000010	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001
	Calcium (Ca)	mg/L	-	-	6.2	16.0	23.9	4.8	14.5	21.4	9.4	29.4	36.7	4.8	15.4	23.1	5.2	15.2	21.9
	Chromium (Cr)	mg/L	0.0089	0.0089	0.0005	0.00198	0.00050	0.00057	0.00260	0.00050	<0.00050	<0.00050	<0.00050	0.00050	0.00225	0.00050	0.00050	0.00185	0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	0.00010	0.00042	0.00010	0.00017	0.00054	0.00010	0.00016	<0.00010	0.00017	0.00011	0.00047	0.00010	0.00010	0.00041	0.00010
	Copper (Cu)	mg/L	0.002	0.0024	***0.0027***	0.0021	0.0010	0.0009	***0.0025***	0.0012	0.0005	<0.0010	0.0009	0.0007	0.0023	0.0013	0.0008	0.0021	0.0011
	Iron (Fe)	mg/L	0.30	0.874	0.104	***0.941***	0.080	0.281	***1.215***	0.125	0.202	0.028	0.050	0.201	***0.961***	0.176	0.181	***0.920***	0.111
	Lead (Pb)	mg/L	0.001	0.001	0.00023	0.00073	0.00009	0.00034	0.00089	0.00013	0.00021	<0.000050	<0.000050	0.00021	0.00075	0.00017	0.00020	0.00074	0.00009
	Lithium (Li)	mg/L	-	-	0.0010	0.0020	0.0010	0.0010	0.0024	0.0010	<0.0010	0.0026	0.0018	0.0010	0.0021	0.0011	0.0010	0.0020	0.0010
	Magnesium (Mg)	mg/L	-	-	3.4	9.0	13.3	2.6	8.8	12.4	6.7	21.4	27.8	3.1	9.1	13.8	3.2	9.1	13.1
	Manganese (Mn)	mg/L	0.935[β]	-	0.0025	0.0119	0.0013	0.0072	0.0146	0.0017	0.0091	0.0010	0.0018	0.0049	0.0123	0.0024	0.0050	0.0132	0.0024
	Mercury (Hg)	mg/L	0.000026	-	0.000005	0.000005	0.000005	0.000005	0.00000505	0.000005	<0.0000050	<0.0000050	<0.0000050	0.000005	0.000005	0.000005	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00007	0.00043	0.00043	0.00005	0.00038	0.00049	0.00010	0.00049	0.00041	0.00007	0.00055	0.00059	0.00011	0.00040	0.00059
	Nickel (Ni)	mg/L	0.025	0.025	0.00146	0.00144	0.00050	0.00057	0.00204	0.00051	<0.00050	0.00057	<0.00050	0.00050	0.00183	0.00062	0.00056	0.00167	0.00071
	Potassium (K)	mg/L	-	-	0.61	1.65	1.45	0.51	1.71	1.50	0.65	1.81	1.69	0.51	1.62	1.55	0.59	1.56	1.47
	Selenium (Se)	mg/L	0.001	-	0.001	0.00005	0.001	0.001	0.00005	0.001	<0.0010	0.00007	<0.0010	0.001	0.00005	0.001	0.001	0.00005	0.001
	Silicon (Si)	mg/L	-	-	0.69	2.25	0.95	0.76	2.71	0.98	0.64	1.00	1.09	0.77	2.30	1.11	0.75	2.37	0.97
	Silver (Ag)	mg/L	0.00025	0.0001	0.00001	0.00005	0.00001	0.00001	0.00005	0.00001	<0.000010	<0.000050	<0.000010	0.00001	0.00005	0.00001	0.00001	0.00005	0.00001
	Sodium (Na)	mg/L	-	-	0.9	4.6	5.8	0.6	4.0	5.8	0.4	3.2	3.4	0.6	3.8	5.6	0.8	3.9	5.2
	Strontium (Sr)	mg/L	-	-	0.0054	0.0214	0.0252	0.0046	0.0188	0.0244	0.0099	0.0414	0.0373	0.0051	0.0194	0.0257	0.0048	0.0192	0.0237
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00010	0.00002	0.00010	0.00010	0.00003	0.00010	<0.00010	<0.000010	<0.00010	0.00010	0.00003	0.00010	0.00010	0.00003	0.00010
	Tin (Sn)	mg/L	-	-	0.0004	0.0001	0.0001	0.0001	0.0001	0.0001	<0.00010	<0.00010	<0.00010	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.010	0.065	0.010	0.015	0.082	0.010	0.011	0.002	<0.010	0.012	0.065	0.011	0.010	0.074	0.010
	Uranium (U)	mg/L	0.015	-	0.0004	0.0054	0.0072	0.0003	0.0040	0.0065	0.0003	0.0037	0.0048	0.0003	0.0039	0.0061	0.0003	0.0036	0.0054
	Vanadium (V)	mg/L	0.006[α]	0.006	0.0010	0.00192	0.00100	0.00100	0.00247	0.00100	<0.0010	<0.00050	<0.0010	0.00100	0.00206	0.00100	0.00100	0.00178	0.00100
	Zinc (Zn)	mg/L	0.030	0.030	0.013	0.009	0.003	0.003	0.0037	0.003	<0.0030	<0.0030	<0.0030	0.003	0.0041	0.0146	0.0034	0.0031	0.0030

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

BOLD Indicates parameter concentration above the AEMP benchmark.

[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 Mary River

5.1.2 Phytoplankton

Chlorophyll-a concentrations at MRTF were comparable to those reported at upstream reference stations during individual spring, summer, and fall sampling events in 2020 (Appendix Table E.14), and were well below the AEMP benchmark of 3.7 µg/L for each of these sampling events. Low phytoplankton productivity, indicative of oligotrophic conditions, was suggested at MRTF based on comparison of chlorophyll-a concentrations to Dodds et al (1998) trophic status classification for creek environments. Overall, no mine-related influences on phytoplankton density were suggested at MRTF in 2020 based on the chlorophyll-a concentration data.

5.1.3 Benthic Invertebrate Community

No ecologically significant differences in benthic invertebrate density and richness were indicated between the MRTF effluent-exposed and reference study areas during the August 2020 EEM study (Table 5.2)[15]. Significantly higher evenness at the MRTF effluent-exposed area compared to the reference area, as well as significantly differing Bray-Curtis Index between these areas, indicated differing benthic invertebrate assemblages between the effluent-exposed and reference areas of MRTF. The primary difference in community composition between the effluent-exposed and reference areas of MRTF was a significantly lower relative abundance of Chironomidae at the effluent-exposed area, including those considered metal-sensitive (Table 5.2). A lower relative abundance of metal-sensitive Chironomidae at the effluent-exposed area suggested that the difference in benthic invertebrate assemblage between areas was potentially related to mine effluent, but because aqueous metal concentrations at MRTF were mostly below WQG (Table 5.1), a factor (or factors) other than metal concentrations likely accounted for the differences in assemblage between the MRTF study areas. For instance, a significantly higher relative abundance of the filterer FFG at the MRTF effluent-exposed area suggested that organic inputs from the effluent channel may have contributed to community composition differences relative to the MRTF upstream reference area (Minnow 2020). Overall, influences of mine operations on the benthic invertebrate community of MRTF remained uncertain following the August 2020 EEM study, but did not appear to be related to metal concentrations originating from mine effluent and/or operations.

[15] Under the MDMER, metrics of richness, Simpson's Evenness, and Bray-Curtis Index are calculated using family-level taxonomy, and thus the MRTF benthic invertebrate community results discussed herein evaluated metrics calculated using this level of taxonomy. For all monitoring conducted for the Mary River Project CREMP, the above metrics were calculated using lowest-practical-level taxonomy.

Table 5.2: Benthic Invertebrate Community Metric Statistical Comparisons Between Mary River Tributary-F (MRTF) Mine-Exposed (EXP) and Reference (REF) Areas, Mary River Project Second EEM Study, August 2020

Endpoint			Data Transformation	Test	Test P-value	MCT[a]		MOD (REF_{SD})
						REF	EXP	
EEM Effect Indicators		Density (No./m²)	log10	tequal	0.009	324	87.3	-1.6
		Richness (No. of Taxa)	log10	tequal	0.151	12.6	9.17	ns
		Simpson's Evenness	log10	tequal	0.087	0.372	0.538	1
Group Percentage (%)	Taxa	Chironomidae	log10	tequal	0.029	76.3	52.8	-3.9
		Metal Sensitive Chironomidae	log10	tequal	0.005	50.8	21.0	-4.1
		Simuliidae	log10(x+1)	tunequal	0.118	2.8	19.2	ns
		Tipulidae	log10(x+1)	tequal	0.996	16.3	16.2	ns
	FFG	Collector-Gatherer	log10	tunequal	0.054	77.3	53.5	-3.5
		Filterer	log10(x+1)	tequal	0.07	3.3	20.3	11.4

P-value < 0.1.
P-value < 0.1 and MOD < -2.
P-value < 0.1 and MOD > 2.

Note: MOD = Magnitude of Difference = (MCT_{Exp} - $MCT_{Ref)}$/SD_{Ref}. FFG = Functional Feeding Group.

[a] MCT = Measure of Central Tendency; MCT reported as median for rank-transformed data and as back-transformed mean for all other cases.

5.1.4 Effects Assessment and Recommendations

Water chemistry at MRTF (Station F0-01) met all AEMP benchmarks consistently over the duration of spring, summer, and fall sampling events in 2020, and for parameters with established AEMP benchmarks, no changes in concentrations were shown relative to baseline. No adverse effects on phytoplankton were indicated at MRTF in 2020. Biological sampling conducted at MRTF to meet MDMER obligations suggested some differences in benthic invertebrate community assemblages between effluent-exposed and reference areas, but these differences did not appear to be related to metal concentrations originating from mine effluent and/or mine operations (Minnow 2021). Under the Mary River Project AEMP Management Response Framework, the absence of a mine-related change in AEMP benchmark parameters over time (or compared to background) requires no further management response (Figure 2.8). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and to baseline, and no adverse biological effects related to metals were indicated in 2020, no adjustment to the existing AEMP need be applied at MRTF as part of the next monitoring program.

5.2 Mary River

5.2.1 Water Quality

Dissolved oxygen in water at Mary River stations was consistently at or above saturation during all spring, summer, and fall monitoring events, and showed comparable saturation among the G0 09 series reference stations and stations adjacent to (E0 series) and downstream (C0 series) of the Mary River Project for each respective seasonal sampling event in 2020 (Figure 5.1; Appendix Tables C.1 to C.3). Dissolved oxygen concentrations were significantly higher at Mary River benthic study areas located adjacent to (E0-01, E0-20) and downstream (C0-05) of the mine than at the upstream (G0-09, G0-03) study areas in August 2020, suggesting no increased oxygen demand associated with mine operations (Appendix Figure C.22; Appendix Table C.56). In addition, dissolved oxygen concentrations were consistently well above WQG acceptable levels for sensitive life stages of cold-water biota (i.e., 9.5 mg/L) at all Mary River stations in spring, summer, and fall of 2020 (Figure 5.1; Appendix Figure C.18; Appendix Table C.55), indicating that slight differences in dissolved oxygen concentrations among the Mary River study areas were not likely to be ecologically meaningful.

In situ pH at all Mary River mine-exposed stations was generally comparable to pH at the G0-09 series reference stations during the spring, summer, and fall sampling events in 2020 (Figure 5.1). Although significant differences in pH were indicated between area E0-20 adjacent to the mine and the G0-09 reference area, the mean incremental difference in pH between these areas was

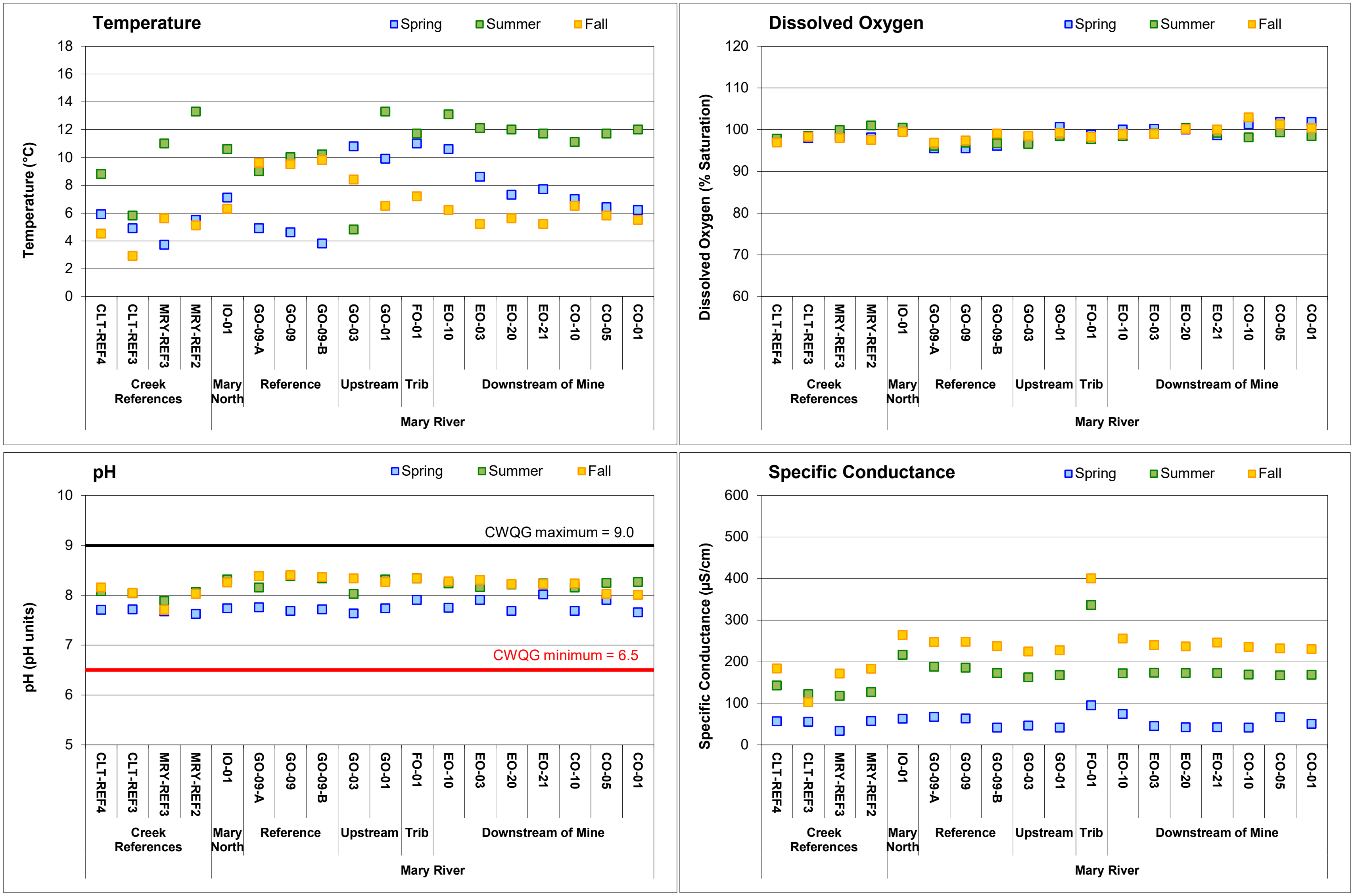


Figure 5.1: Comparison of *In Situ* Water Quality Variables Measured at Mary River Water Quality Monitoring Stations in Spring, Summer, and Fall 2020, Mary River Project CREMP

less than a pH unit, and pH at all Mary River areas were consistently within WQG limits (Figure 5.1; Appendix Table C.57), suggesting that any pH differences among the Mary River study areas were not likely to be ecologically meaningful. Specific conductance was consistently lowest in spring and highest in fall at all Mary River stations (Figure 5.1), reflecting natural seasonal differences related to proportion of flow from surface runoff (e.g., spring snowmelt) and baseflow/groundwater sources. Specific conductance was considerably higher at Mary River Tributary-F than at all other monitoring stations, which suggested that this tributary was a key source of mine-related inputs to Mary River (e.g., MS-08 effluent; Figure 5.1). Within Mary River, specific conductance was significantly higher in the portion of the river adjacent to the mine (immediately downstream of the MRTF confluence) at E0 series stations, but not downstream of the mine at the C0 series sampling locations at the time of biological monitoring in August 2020 (Appendix Figure C.22; Appendix Table C.57), suggesting that mine-related influences on Mary River water quality were of limited spatial scale.

Within Mary River, mean concentrations of aluminum and iron were above their respective AEMP benchmarks at stations located adjacent to (E0 series) and downstream of (C0 series) the mine in 2020, but only during the summer sampling event (Table 5.1). In addition, mean concentrations of total aluminum, copper, and phosphorus were above applicable WQG in spring, summer, and/or fall sampling events in 2020 at the E0 and C0 series stations (Table 5.1). However, in all cases in which the AEMP benchmarks and WQG were exceeded at areas adjacent to and downstream of the mine, the mean concentrations for each of these parameters were similar or higher, and above applicable AEMP benchmark and WQG values, at the Mary River reference stations (G0-09 series) and/or other upstream stations (G0 series) for each given seasonal sampling event (Table 5.1).[16] Relatively high concentrations of these parameters within Mary River at the time of the summer sampling event relative to the spring and fall sampling events appeared to be associated with highly turbid sampling conditions in the summer (Table 5.1). Concentrations of aluminum, copper, iron, and phosphorus were lower at MRTF than at the Mary River reference and mine-exposed stations (Table 5.1), suggesting that this mine-exposed tributary was not a substantial source of these parameters in 2020. Among those parameters with established AEMP benchmarks, nitrate and sulphate concentrations were elevated by factors greater than three only at Station E0-10, downstream of the confluence with MRTF, and only during the spring and fall sampling events in 2020 (Appendix Table C.62). Therefore, elevation in concentrations of nitrate and sulphate in Mary River appeared to be associated with mine deposits to MRTF (e.g., MS-08 effluent). Nitrate, and sulphate to a lesser

[16] Previous CREMP studies also showed total aluminum concentrations above respective WQG and/or AEMP benchmarks at Mary River GO series reference stations, indicating naturally high concentrations of this metal in Mary River.

extent, were also the only two parameters that were elevated at Mary River stations adjacent to (E0 series) and downstream of (C0 series) the mine in 2020 compared to baseline that also did not show an elevation over time at the Mary River reference area (Appendix Figure C.23), indicating that higher concentrations of these parameters was mine-related. Overall, no marked influences on water quality of Mary River were indicated in 2020 as a result of mine operations except for slight enrichment of nitrate and sulphate concentrations near the mine, although to levels that remained well below AEMP benchmarks.

5.2.2 Sediment Quality

Deposited sediment sampled from Mary River study areas was mostly medium-sized coarse sand and some gravel (Appendix Table D.33). Substrate among the Mary River study areas was largely composed of cobble and boulder material with minimal amounts of sand and finer material except at the downstream-most study area C0-05, where medium-sized sand composed approximately 65% of the surficial in-stream substrate (Minnow 2018). Silt precipitate and/or deposits were generally absent from all Mary River mine-exposed study areas during the August 2020 sampling event (Appendix Table D.33). Sediment TOC content was low (i.e., <0.2%) at all Mary River study areas, and generally did not differ between the mine-exposed areas and the upstream reference area (G0-09), suggesting similar depositional characteristics among the Mary River study areas (Table 5.3; Appendix Table D.36).

Metal concentrations in sediment from all Mary River study areas were highly comparable (Table 5.3; Appendix Table D.36). The only notable difference was a slight elevation (i.e., 3-fold) in the concentration of molybdenum at E0-20 compared to the average concentration at the upstream G0-09 reference area (Table 5.3; Appendix Table D.36). Concentrations of metals in deposited sediment were also well below applicable SQG at all Mary River study areas (Table 5.3; Appendix Tables D.34, D.35, and D.37 to D.39).

5.2.3 Phytoplankton

Chlorophyll-a concentrations at Mary River stations located downstream of the mine were generally within the range of, or slightly higher, than the G0 series river reference stations and/or creek reference stations during the 2020 spring, summer, and fall sampling events (Figure 5.2). Chlorophyll-a concentrations were consistently well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events at all Mary River sampling stations in 2020, and were suggestive of low (i.e., oligotrophic) phytoplankton productivity based on Dodds et al (1998) trophic status classification for stream environments. Therefore, no adverse mine-related influences on phytoplankton abundance were indicated at Mary River in 2020. Low to moderate phytoplankton productivity was expected for Mary River reference and mine-exposed stations in

Table 5.3: Sediment Total Organic Carbon and Metal Concentrations at Mary River Mine-Exposed and Reference (GO-09) Sediment Monitoring Stations, Mary River Project CREMP, August 2020

Parameter	Units	SQG[a]	Mary River Reference (GO-09; n = 3)	Upstream GO-03 (n = 3)	Mary River Mine-Exposed Areas		
					Adjacent EO-01 (n = 3)	Adjacent EO-20 (n = 3)	Downstream CO-05 (n = 3)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC	%	10[α]	0.11 ± 0.012	0.11 ± 0.012	0.11 ± 0.010	0.19 ± 0.10	0.12 ± 0.015
Aluminum (Al)	mg/kg	-	2,757 ± 1,141	2,000 ± 688	2,407 ± 438	4,217 ± 2,456	2,468 ± 1,366
Antimony (Sb)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Arsenic (As)	mg/kg	17	0.38 ± 0.10	1.10 ± 1.15	0.41 ± 0.059	0.62 ± 0.12	0.37 ± 0.27
Barium (Ba)	mg/kg	-	12.6 ± 5.05	9.39 ± 3.05	11.3 ± 1.92	20.7 ± 9.70	10.2 ± 5.51
Beryllium (Be)	mg/kg	-	0.14 ± 0.040	0.11 ± 0.023	0.11 ± 0.010	0.24 ± 0.19	0.13 ± 0.044
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	0.24 ± 0.075	0.30 ± 0.18	<0.20 ± 0
Boron (B)	mg/kg	-	5.4 ± 0.75	<5.0 ± 0	<5.0 ± 0	6.7 ± 2.9	<5.0 ± 0
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	<0.020 ± 0	0.026 ± 0.0053	0.038 ± 0.020	<0.020 ± 0
Calcium (Ca)	mg/kg	-	2,750 ± 894	2,080 ± 249	2,353 ± 454	2,887 ± 1,147	2,046 ± 1,384
Chromium (Cr)	mg/kg	90	13.6 ± 4.34	13.7 ± 3.49	18.7 ± 10.8	26.1 ± 5.37	14.1 ± 10.9
Cobalt (Co)	mg/kg	-	2.40 ± 0.758	1.96 ± 0.468	2.58 ± 0.850	3.88 ± 1.21	2.32 ± 1.45
Copper (Cu)	mg/kg	110[α]	4.45 ± 2.50	2.78 ± 1.01	4.05 ± 0.215	7.40 ± 2.27	3.14 ± 1.84
Iron (Fe)	mg/kg	40,000[α]	11,063 ± 2,423	13,633 ± 3,099	16,950 ± 13,939	19,233 ± 6,401	6,443 ± 4,132
Lead (Pb)	mg/kg	91	3.07 ± 0.857	2.74 ± 0.567	2.78 ± 0.477	4.27 ± 1.48	2.30 ± 1.00
Lithium (Li)	mg/kg	-	5.0 ± 2.3	3.5 ± 1.2	3.4 ± 0.57	6.33 ± 4.44	4.8 ± 2.9
Magnesium (Mg)	mg/kg	-	2,810 ± 1,212	1,793 ± 389	2,630 ± 251	4,440 ± 2,669	3,380 ± 2,370
Manganese (Mn)	mg/kg	1,100[α,β]	76 ± 29.4	58.6 ± 14.3	85.4 ± 25.4	137 ± 39	72.5 ± 41.1
Mercury (Hg)	mg/kg	0.486	<0.0050 ± 0	0.0050 ± 0.00006	<0.005 ± 0	<0.0050 ± 0	<0.0050 ± 0
Molybdenum (Mo)	mg/kg	-	0.11 ± 0.023	<0.10 ± 0	0.20 ± 0.085	0.36 ± 0.20	0.12 ± 0.025
Nickel (Ni)	mg/kg	75[α,β]	6.11 ± 1.99	4.83 ± 1.12	8.27 ± 2.03	16.7 ± 6.30	14.2 ± 12.9
Phosphorus (P)	mg/kg	2,000[α]	350 ± 118	400 ± 50.5	383 ± 136	376 ± 76	270 ± 167
Potassium (K)	mg/kg	-	750 ± 320	507 ± 189	617 ± 129	1,167 ± 650	517 ± 297
Selenium (Se)	mg/kg	-	<0.20 ± 0	<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	0.13 ± 0.046	<0.10 ± 0
Sodium (Na)	mg/kg	-	68 ± 21	<50 ± 9	<50 ± 0	67 ± 29	57 ± 12
Strontium (Sr)	mg/kg	-	4.72 ± 1.01	3.99 ± 0.183	3.75 ± 0.376	4.69 ± 1.69	3.28 ± 1.30
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	0.068 ± 0.023	0.053 ± 0.0058	<0.050 ± 0	0.086 ± 0.048	0.062 ± 0.0061
Tin (Sn)	mg/kg	-	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0
Titanium (Ti)	mg/kg	-	353 ± 123	263 ± 50	294 ± 21	396 ± 133	255 ± 141
Uranium (U)	mg/kg	-	0.922 ± 0.298	0.822 ± 0.199	0.785 ± 0.189	1.09 ± 0.328	0.725 ± 0.586
Vanadium (V)	mg/kg	-	19.5 ± 5.06	23.8 ± 4.93	22.7 ± 18.3	25.9 ± 9.83	9.26 ± 5.56
Zinc (Zn)	mg/kg	315	10.3 ± 4.31	7.8 ± 2.3	10.3 ± 1.04	17.0 ± 7.91	9.17 ± 5.46
Zirconium (Zr)	mg/kg	-	5.8 ± 2.0	4.4 ± 1.4	3.8 ± 0.49	5.7 ± 3.1	3.1 ± 1.5

Indicates parameter concentration above SQG.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV 2020).

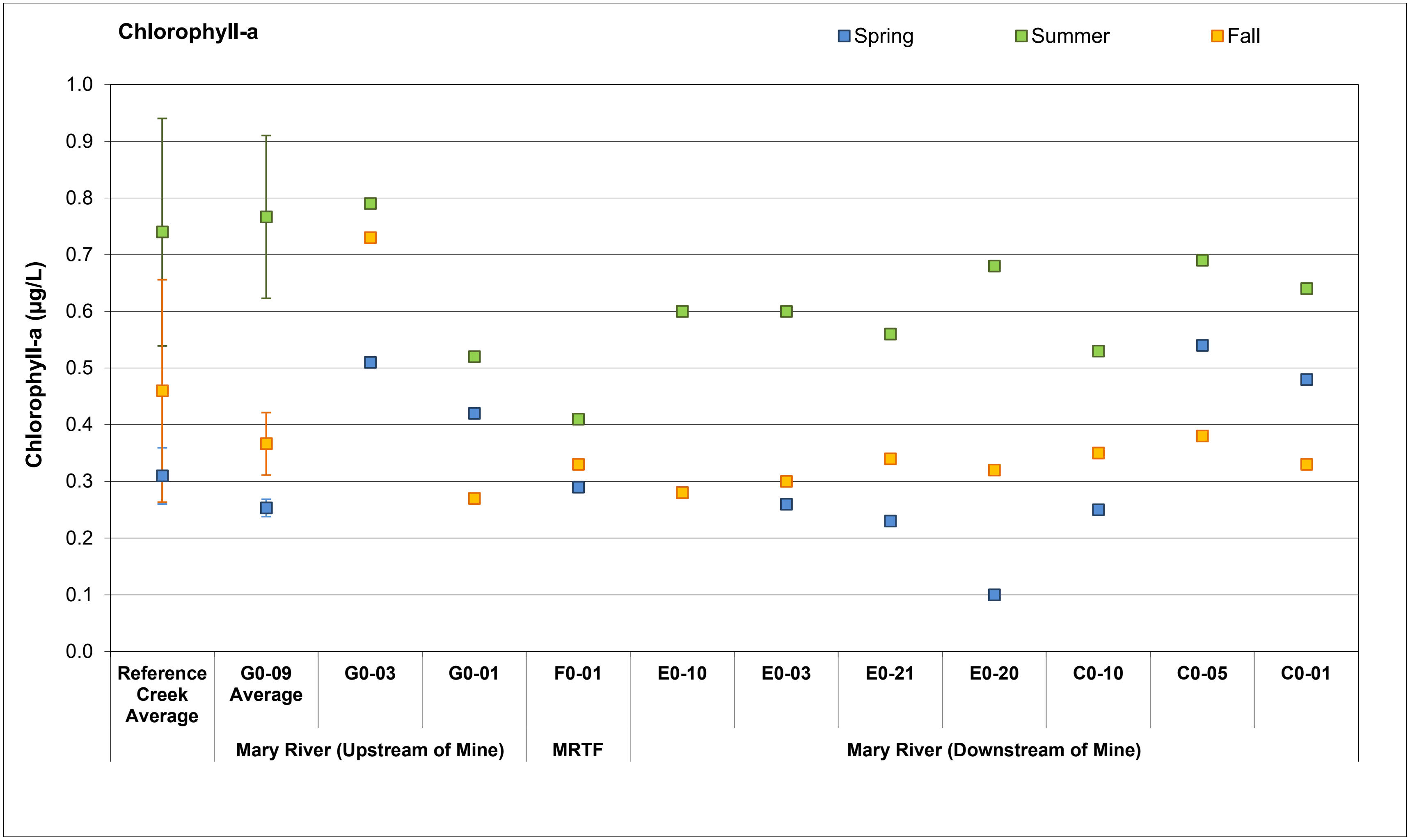


Figure 5.2: Chlorophyll-a Concentrations at Mary River Phytoplankton Monitoring Stations Located Upstream and Downstream of the Mine, Mary River Project CREMP, 2020

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

2020 given ‘oligotrophic’ to ‘mesotrophic’ productivity categorizations based on CWQG classifications that use total phosphorus concentrations to define trophic status (Table 5.1; Appendix Table C.58).

Chlorophyll-a concentrations at Mary River mine-exposed and reference stations in fall 2020 were generally similar to those shown at the time of baseline and previous years of mine operation (Figure 5.3). Chlorophyll-a concentrations in fall 2020 were not disproportionately higher or lower compared to baseline at the mine-exposed stations of Mary River compared to the reference stations, suggesting no adverse change/differences in phytoplankton abundance due to mine-related influences over time.

5.2.4 Benthic Invertebrate Community

The Mary River benthic invertebrate community assessment included a spatial statistical analysis of endpoints among an upstream reference area (G0-09), an upstream area with limited mine exposure (G0-03), two near-field mine-exposed areas located near the mine (E0-01, E0 20), and a far-field cumulative effects mine-exposed area located downstream of the mine (C0-05; see Table 2.5). At the Mary River G0-03 study area, no ecologically significant differences in benthic invertebrate density, richness, evenness, and relative abundance of metal-sensitive taxa were indicated compared to the G0-09 reference area, suggesting no marked influences of the mine operation on the benthic invertebrate community. Some differences in community assemblage were suggested between G0-03 and G0-09 study areas by differing Bray-Curtis Index (Appendix Table F.51) and significantly higher and lower relative abundance of Hydracarina and Chironomidae groups, respectively, at G0-03 in 2020 (Table 5.4). However, the relative abundance of these groups at G0-03 in 2020 did not differ significantly from baseline (Appendix Table F.50), suggesting that the differences in assemblage between G0-03 and G0-09 study areas in 2020 reflected natural variability.

At the near-field mine exposed study areas (i.e., E0-01 and E0-20), no ecologically significant differences in density, richness, evenness, and the proportion of metal-sensitive Chironomidae were indicated relative to the reference study area (Table 5.4; Appendix Table F.50). Differing Bray-Curtis Index suggested differing community composition between the E0 and G0-09 study areas, but no significant differences in the relative abundance of any dominant groups were indicated in 2020 (Table 5.4; Appendix Table F.51). Rather, the differences in community composition at E0-01 and E0-20 compared to the G0-09 reference area appeared to reflect lower relative abundance of the collector-gatherer FFG and the sprawler HPG at one or both of the E0 study areas (Appendix Table F.50), potentially indicating habitat differences between the E0 and G0-09 study areas. No ecologically significant differences in density, richness, evenness (E0-20 study area only) and relative abundance of dominant

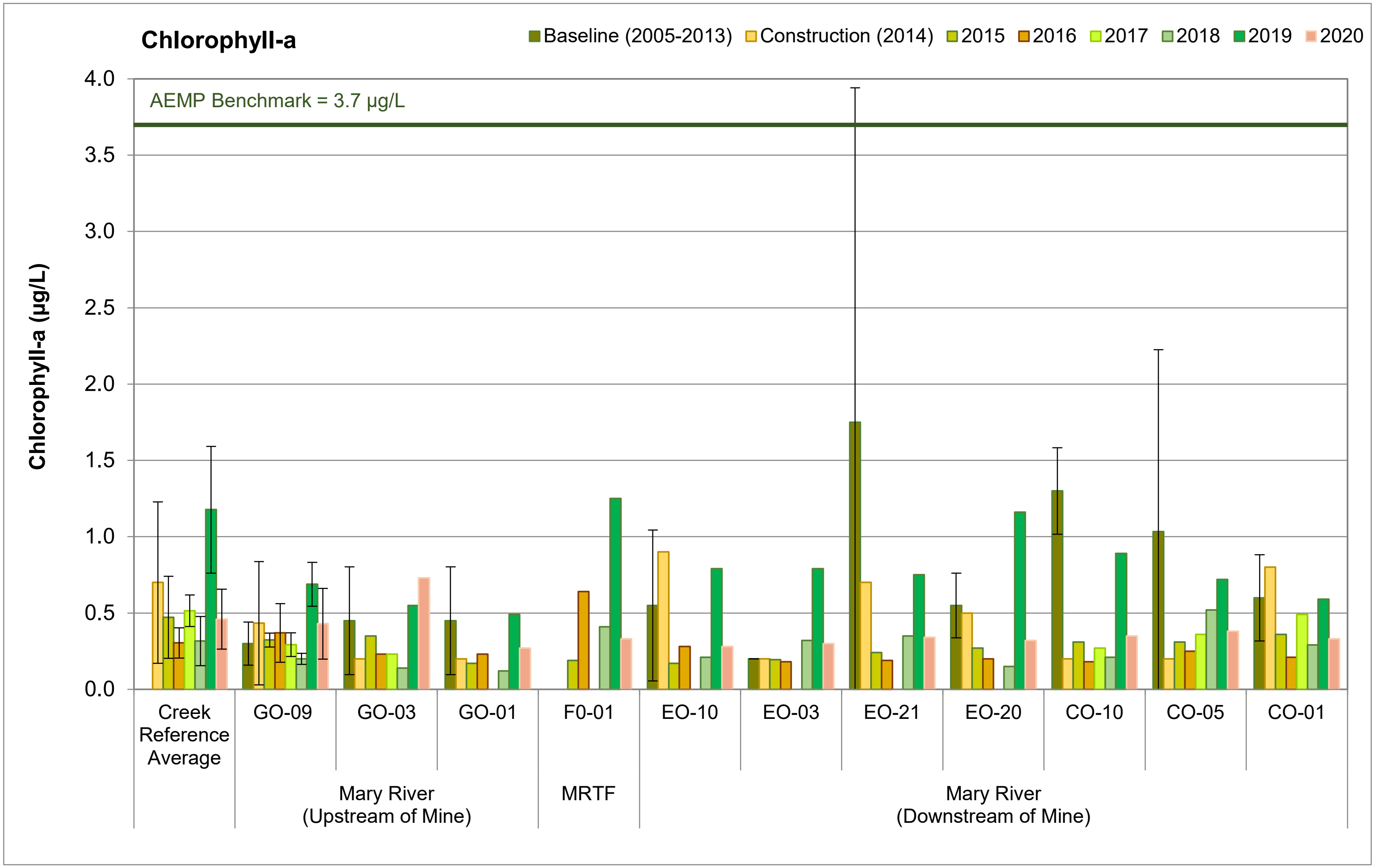


Figure 5.3: Temporal Comparison of Chlorophyll-a Concentrations at Mary River Stations for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2020) Periods during the Fall

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

Table 5.4: Benthic Invertebrate Community Metric Statistical Comparison Results among Mary River Reference (GO-09), Upstream (GO-03), and Mine-Exposed (EO-01, EO-20, CO-05) Study Areas, Mary River Project CREMP, August 2020

Metric	Data Transform-ation	Overall 5-Area Comparison		Pair-wise, post-hoc comparisons[a]				
		Significant Difference Among Areas?	P-value	Area	Mean	Standard Deviation	Effect Size vs. GO-09 Reference	Pairwise Comparison
Density (No. per m²)	rank	YES	0.019	GO-09 Ref	886	831	-	a
				GO-03	196	95.3	-0.8	b
				EO-01	513	420	-0.4	a
				EO-20	1,441	553	0.7	a
				CO-05	906	818	0.0	a
Richness (No. of Taxa)	none	NO	0.710	GO-09 Ref	15.0	2.0	-	a
				GO-03	13.6	3.3	-0.7	a
				EO-01	15.6	2.1	0.3	a
				EO-20	14.0	2.6	-0.5	a
				CO-05	14.2	2.2	-0.4	a
Simpson's Evenness	rank	YES	0.036	GO-09 Ref	0.826	0.153	-	ab
				GO-03	0.918	0.024	0.6	a
				EO-01	0.913	0.016	0.6	a
				EO-20	0.868	0.046	0.3	b
				CO-05	0.839	0.048	0.1	b
Hydracarina (% of community)	rank	YES	0.058	GO-09 Ref	1.1	1.3	-	a
				GO-03	4.3	3.1	2.4	b
				EO-01	3.1	2.9	1.5	ab
				EO-20	0.8	0.4	-0.2	a
				CO-05	6.0	4.9	3.8	b
Chironomidae (% of community)	none	YES	0.002	GO-09 Ref	89.6	5.1	-	a
				GO-03	69.9	13.6	-3.9	b
				EO-01	88.2	9.2	-0.3	a
				EO-20	95.5	2.5	1.2	a
				CO-05	84.8	8.0	-0.9	a
Tipulidae (% of community)	rank	YES	0.003	GO-09 Ref	1.2	0.5	-	b
				G0-03	6.2	5.1	10.6	b
				E0-01	5.7	4.3	9.5	b
				E0-20	0.6	0.9	-1.4	b
				C0-05	1.4	0.9	0.3	b
Metal Sensitive Chironomidae (% of community)	log10	YES	0.043	GO-09 Ref	32.0	23.7	-	a
				GO-03	12.3	5.8	-0.8	ab
				EO-01	11.0	2.5	-0.9	ab
				EO-20	9.4	9.1	-1.0	b
				CO-05	19.4	9.6	-0.5	ab

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

Indicates magnitude of difference outside of the Critical Effect Size of ± 2 SD of respective baseline year mean, suggesting an ecologically meaningful difference in endpoint value between study years.

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

taxonomic groups (including metal-sensitive Chironomids) were indicated between mine operational years (2015 to 2020) and baseline (2007) at either of the E0 near-field study areas (Appendix Tables F.54 and F.56). Although evenness has consistently been significantly higher at an absolute magnitude greater than 2 SDREF in years of mine operation compared to baseline at the E0-01 study area, higher evenness is not associated with an adverse influence and thus was not consistent with effects to the benthic invertebrate community normally attributed to mine operations.

At far-field mine-exposed area CO-05, no ecologically significant differences in benthic invertebrate density, richness, evenness, and relative abundance of metal-sensitive taxa were indicated compared to the G0-09 reference area, suggesting no marked influences of the mine operation on the benthic invertebrate community. Although differences in community assemblage were suggested between the C0-05 and G0-09 study areas by differing Bray-Curtis Index (Appendix Table F.51), only a significantly higher relative abundance of Hydracarina was indicated between these study areas in 2020 (Table 5.4). No ecologically significant differences in density, richness, evenness, and dominant taxonomic groups were indicated at C0-05 for all years of mine operation (2015 to 2020) compared to one or both years of baseline period data (2007 and 2011; Appendix Table F.58). Therefore, no adverse effects of mine-operations on the benthic invertebrate community at Mary River C0-05 were indicated since the commencement of commercial mine operations in 2015.

5.2.5 Fish Community and Population Health

The fish community of Mary River was composed of arctic charr and low numbers of ninespine stickleback, and was comparable to the Angajurjualuk Lake Tributary reference area during the EEM fish survey in August 2020 in terms of both species composition and fish abundance as reflected by CPUE (Table 5.5). Similar fish species composition was indicated within Mary River between EEM studies conducted in 2017 and 2020, and although arctic charr CPUE was higher in 2020, this likely reflected lower water levels resulting in improved sampling efficiency compared to the 2017 study (Minnow 2021).

The length-frequency distribution of arctic charr captured at Mary River adjacent to the mine differed significantly from the distribution shown at the Angajurjualuk Lake Tributary reference area, but did not differ significantly from the distribution shown farther downstream in Mary River near Mary Lake (Table 5.6). Similar length-at-age relationships were indicated for arctic charr sampled at Mary River adjacent to the mine compared to the Angajurjualuk Lake Tributary reference area and Mary River near the outlet to Mary Lake, suggesting that the difference in arctic charr length-frequency distribution between Mary River and Angajurjualuk Lake Tributary was a sampling artifact (Minnow 2021). No significant differences in length, weight, growth

Table 5.5: Summary of Fish Catches at Mary River Project Second EEM Study Fish Population Study Areas, August 2020

Study Area	Total Effort		Summary Statistic Endpoint	Fish Species			Catch Summary	
	Distance Sampled (m)	Electrofishing Seconds		Arctic Char		Ninespine Stickleback	Totals	Total No. Species
				YOY [b]	Non-YOY [b]			
Angajurjualuk Lake Tributary Reference (ALTR)	234	2,226	Total No. Caught	0	104	2	**106**	**2**
			CPUE[a]	0	2.80	0.05	**2.86**	
Mary River Reference (MRR)	200	2,972	Total No. Caught	0	122	35	**157**	**2**
			CPUE[a]	0	2.46	0.71	**3.17**	
Mary River Effluent-Exposed (MRE)	228	2,241	Total No. Caught	0	122	0	**122**	**1**
			CPUE[a]	0	3.27	0	**3.27**	

[a] Electrofishing catch-per-unit-effort (CPUE) represents number of fish captured per minute of electrofishing.

[b] Young-of-the-year (YOY).

Table 5.6: Summary of Arctic Charr Endpoint Statistical Comparison Results Between Effluent-Exposed and Reference Areas Used for the Mary River Project First and Second EEM Studies, August 2017 and 2020

Endpoint [a]	Applicable Critical Effect Size	Sample Size	Statistically Significant Differences Observed?[b]				
			First EEM (2017)		Second EEM (2020)		
			Test	MRE vs MRR	Test	MRE vs ALTR	MRE vs MRR
Survival (Age Frequency Distribution)*	none	100[c]	K-S	No	K-S	**Yes (-22%)**	No
Survival (Age)*	± 25%	20	n/a	n/a	K-W	**Yes (+50%)**	No
Body Size (Fork Length)	none	100[c]	t-test	No	K-W	**Yes (+12%)**	No
		20	n/a	n/a	ANOVA	No	No
Body Size (Body Weight)	none	100	t-test	No	K-W	**Yes (+36%)**	No
		20	n/a	n/a	ANOVA	No	No
Energy Usage (Length-at-age)	none	20	n/a	n/a	ANCOVA	No	No
Energy Usage (Weight-at-age)*	± 25%	20	n/a	n/a	ANCOVA	No	No
Energy Storage (liver weight at body weight)*	± 25%	20	n/a	n/a	ANOVA	No	No
Energy Storage (condition)*	± 10%	100[c]	ANCOVA	**Yes (-4.5%)**	ANCOVA	No	No
		20	n/a	n/a	ANCOVA	No	No

Indicates an absolute magnitude of difference (MOD) greater than applicable Critical Effect Size for fish population survey EEM effect indicators.

Notes: YOY = young-of-the year; MRR = Mary River reference area; MRE = Mary River effluent-exposed area; ALTR = Angajurjualuk Lake Tributary reference area; n/a indicates endpoint not applicable (i.e., endpoint associated with lethal sampling).

[a] Endpoints denoted with an asterisk represent primary EEM endpoints used for the determination of "effects" for a lethal fish population study.

[b] Information provided indicates whether a significant difference occurred between areas (**yes**/no) and the magnitude of difference for any differences (in parentheses).

[c] Sample size varied between areas. In First Study, n=100 at the effluent-exposed and reference area. In Second Study, n=108 for length measures and 100 for weight and condition measures at the effluent-exposed area, and n=100 at the reference areas.

(i.e., body weight-at-age), relative liver size (i.e., liver weight-at-body weight), or condition (i.e., body weight-at-fork length) were indicated for arctic charr sampled at Mary River adjacent to the mine compared to those sampled at either the Angajurjualuk Lake Tributary reference area or Mary River near the outlet to Mary Lake (Table 5.6). In addition, no externally visible abnormalities or parasitic infections were observed on any arctic charr captured at the Mary River effluent exposed area (Minnow 2021). Muscle tissue selenium concentrations in arctic charr captured at Mary River did not differ significantly to those captured at the Angajurjualuk Lake Tributary reference area, and were well below the USEPA (2016) chronic effects criterion of 11.3 mg/kg dry weight for protection of aquatic life (Figure 5.4), suggesting reproductive impairments (e.g., deformities, mortality) to Mary River arctic charr were highly unlikely. Overall, the absence of any significant differences in EEM effect indicators related to growth, relative liver size, condition, and tissue selenium concentrations in arctic charr captured at Mary River compared to applicable reference areas indicated no marked influence of mine operations on the health of arctic charr at Mary River in 2020 (Minnow 2021).

5.2.6 Effects Assessment and Recommendations

At Mary River, the following AEMP benchmarks were exceeded in 2020:

- Aluminum concentration in water was greater than the benchmark of 0.966 mg/L adjacent to (i.e., E0-series stations) and downstream of (i.e., C0-series stations) of the mine during the summer sampling event;
- Copper concentration in in water was greater than the benchmark of 0.0024 mg/L adjacent to (i.e., E0-series stations) and downstream of (i.e., C0-series stations) of the mine during the summer sampling event;
- Iron concentration in water was greater than the benchmark of 0.874 mg/L adjacent to (i.e., E0-series stations) and downstream of (i.e., C0-series stations) of the mine during the summer sampling event; and,
- Lead concentration in water was greater than the benchmark of 0.001 mg/L adjacent to (i.e., E0-series stations) the mine during the summer sampling event.

Within Mary River, concentrations of aluminum, copper, iron, and lead were above respective AEMP benchmarks at stations located adjacent to the mine (i.e., E0 series stations) and, with the exception of lead, downstream of the mine (C0 series) in 2020, but only during the summer

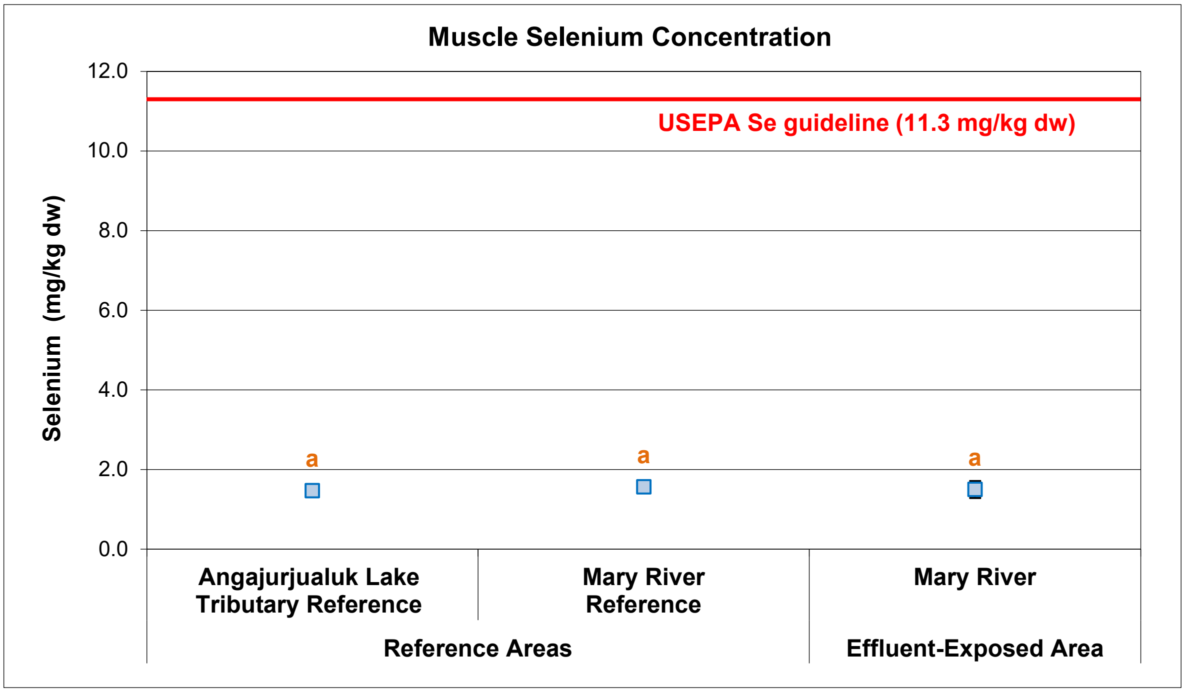


Figure 5.4: Selenium Concentrations (mean ± SD; n = 8) in Dorsal Muscle Tissue of Arctic Charr Captured at Mary River Effluent-Exposed and Reference Study Areas During the Second EEM Study, August 2020

Note: Data points with the same letters do not differ significantly.

sampling event.[17] However, in all cases in which the AEMP benchmarks were exceeded, the concentrations for each of these parameters were similar or higher, and above applicable AEMP benchmarks, at the Mary River reference stations (G0-09 series) and/or upstream stations (G0 series) at the time of the summer sampling event.[18] Relatively high concentrations of these parameters within Mary River at the time of the summer sampling event appeared to be associated with highly turbid conditions compared to the spring and fall sampling events. Concentrations of aluminum, copper, iron, and lead in water at Mary River stations located adjacent to and downstream of the mine in 2020 were comparable to concentrations at each station in each season during baseline. Therefore, no mine-related changes to parameter concentrations were indicated at Mary River mine-exposed stations in 2020 compared to the reference stations and to Mary River baseline data. In addition, metal concentrations in sediment were well below SQG, and no adverse effects on phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at all Mary River mine-exposed areas in 2020. Under the Mary River Project AEMP Management Response Framework, the absence of a mine-related change in AEMP benchmark parameters over time (or compared to background) requires no further management response (Figure 2.8). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no management response (i.e., alteration of existing AEMP) is required for Mary River as part of the next monitoring program.

5.3 Mary Lake

5.3.1 Water Quality

Water quality profiles conducted at the north and south basins of Mary Lake showed similar patterns in water temperature from the surface to bottom as those shown at Reference Lake 3 for summer and fall sampling events in 2020 (Figures 5.5 and 5.6). At the north basin, development of an epilimnion occurred through the surficial 5 m and a hypolimnion was evident at depths greater than approximately 11 m in the fall (Figure 5.5). No distinct thermal layering was evident at the north basin of Mary Lake in the winter and fall, or at the south basin for any seasons although a clear gradient in water temperature was evident in summer (Figures 5.5 and 5.6).

[17] The reported concentration of zinc at Station E0-03 was above the AEMP benchmark during the summer sampling event but this result appeared to be an anomaly based on an order of magnitude difference in concentration between this station and data reported for all other Mary River stations in summer 2020 (Appendix Table C.59).

[18] Cadmium, copper, and zinc concentrations in water were above AEMP benchmarks at the Mary River upstream reference stations (i.e., Station G0-09) in spring 2020, indicating the potential for natural elevation of these parameters in Mary River adjacent to and downstream of the mine.

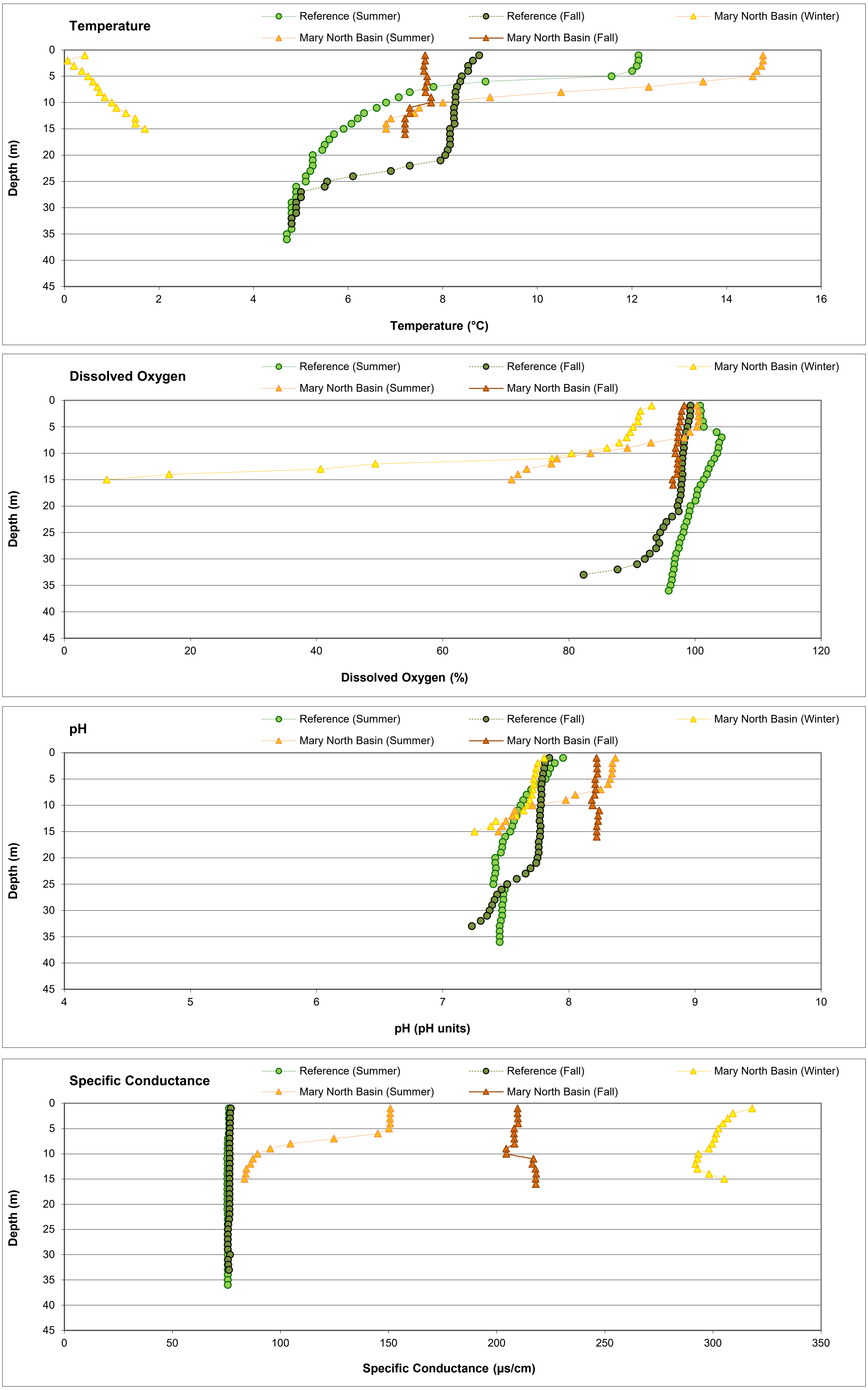
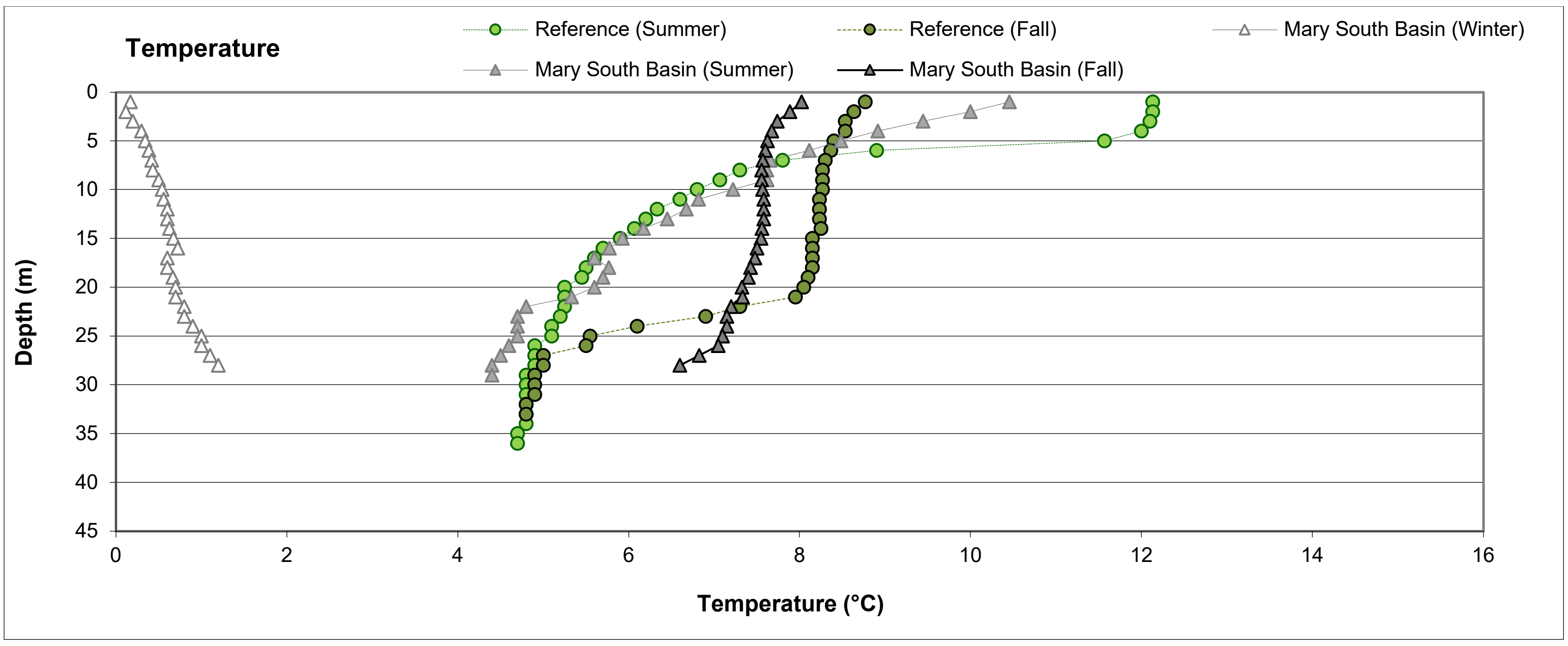
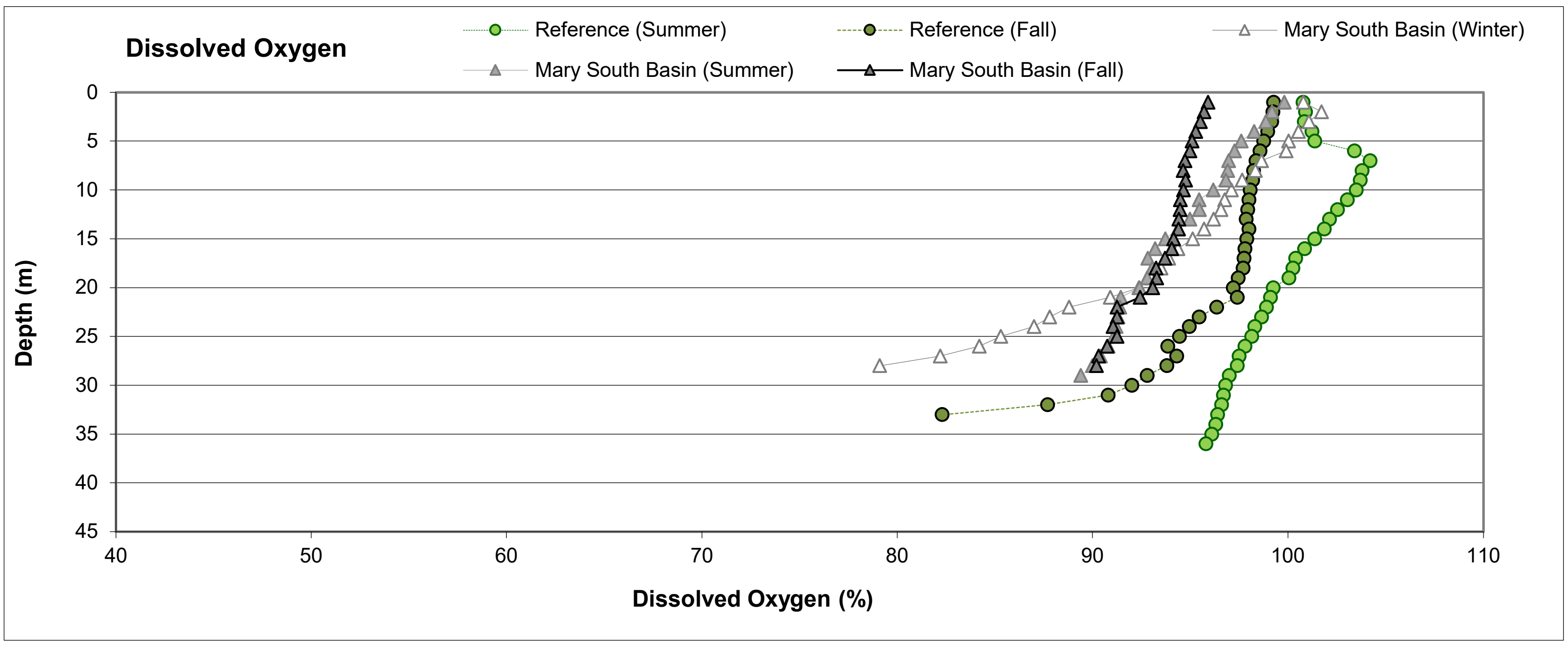
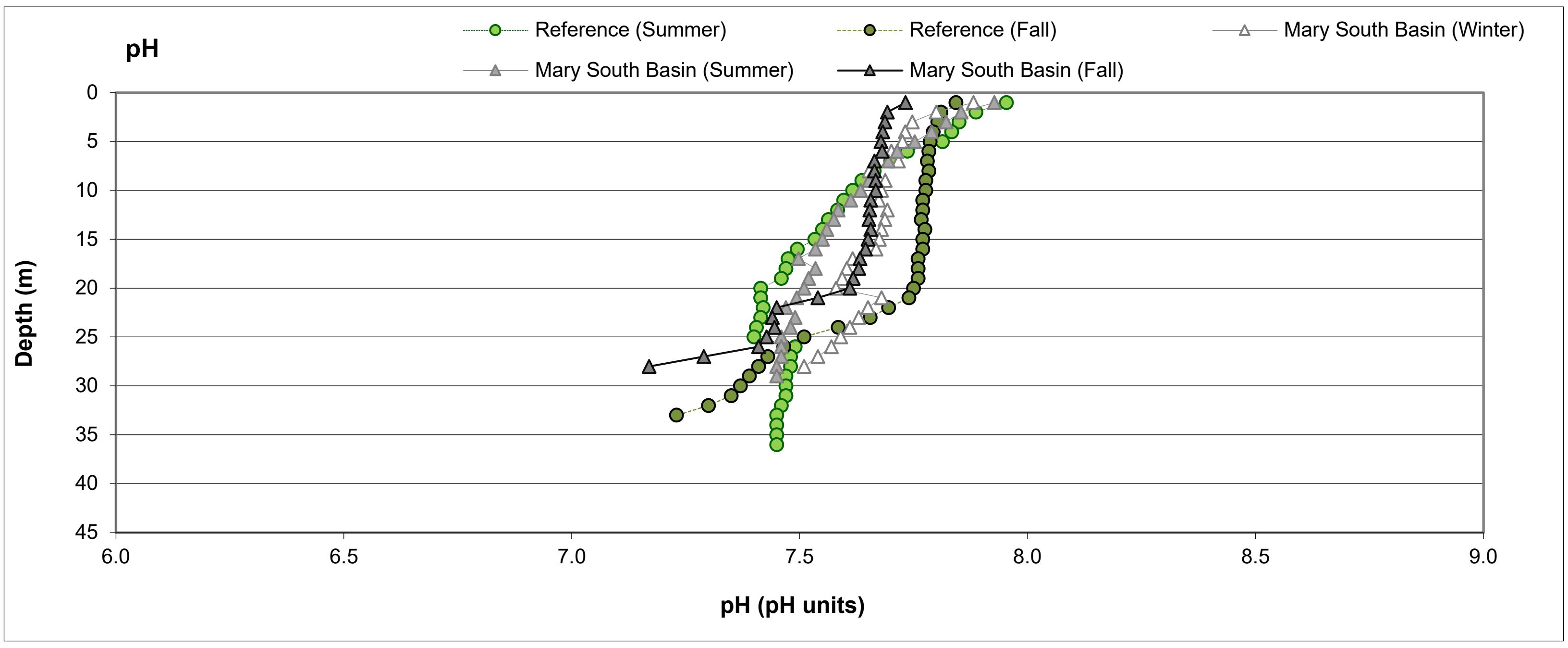


Figure 5.5: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake North Basin (BLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020







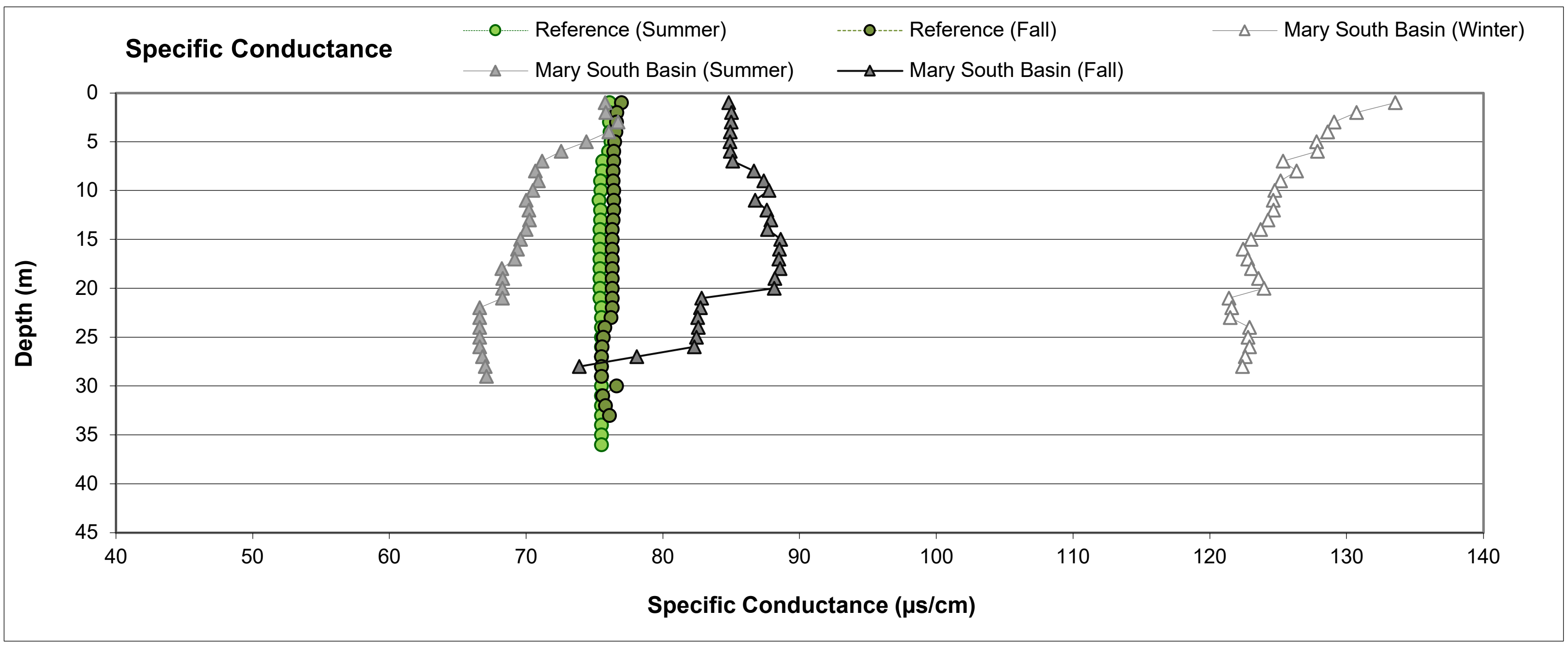


Figure 5.6: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake South Basin (BLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

Water temperatures at the bottom of the water column at Mary Lake littoral and profundal stations did not differ significantly from those at like-habitat stations of Reference Lake 3 during the August 2020 biological study (Figure 5.7; Appendix Table C.65). Dissolved oxygen profiles showed the development of moderate to strong oxyclines extending through the entire water column at both the Mary Lake north and south basins for winter and summer sampling events in 2020 (Figures 5.5 and 5.6). The dissolved oxygen profiles conducted during summer and fall at Mary Lake mirrored similar profiles at the reference lake except for in summer at the north basin. Dissolved oxygen concentrations near the bottom of the water column were significantly lower at Mary Lake littoral and profundal stations than like-habitat stations at the reference lake during the August 2020 biological study. However, dissolved oxygen concentrations were above the WQG for the protection of sensitive populations of cold-water species (i.e., 9.5 mg/L) at all littoral and profundal stations of Mary Lake during the August 2020 study (Figure 5.7; Appendix Table C.65).

Water column profiles showed slightly decreasing pH with increased depth at Mary Lake north and south basins, comparable to those at Reference Lake 3, during winter and summer sampling events in 2020, with the changes in pH through the water column at both lakes appearing to coincide with changes in water temperature during each given season (Figures 5.5 and 5.6). The pH near the bottom of the water column at littoral stations of Mary Lake did not differ significantly from the reference lake, but was significantly higher at profundal stations of Mary Lake compared to the reference lake during the August 2020 biological study (Figure 5.7). The mean incremental difference in bottom pH at profundal stations between lakes was small (less than a pH unit) and pH was consistently within WQG limits at all Mary Lake stations (Figure 5.7, Appendix Table C.65), and therefore this pH difference between lakes was not ecologically meaningful. Specific conductance was substantially higher at the north basin compared to the south basin of Mary Lake (Figures 5.5 and 5.6; Appendix Figure C.28), likely reflecting natural differences in dominant inflow sources to Mary Lake (i.e., Tom River inflow to the north basin, and the Mary River inflow to the south basin) and natural differences in geochemistry associated with these inflows. Specific conductance profiles showed variable changes from the surface to bottom of the water column at the north basin, but were relatively uniform at the south basin, over winter, summer, and fall sampling events in 2020 (Appendix Figure C.28). The differences between basins may have reflected differing influence associated with the dominant inflows to the lake and the station location relative to these inflows. Specific conductance near the bottom of the water column at littoral and profundal stations of Mary Lake did not differ significantly from like-habitat stations at Reference Lake 3 during the August 2020 biological study (Figure 5.7). Water clarity, as determined using Secchi depth

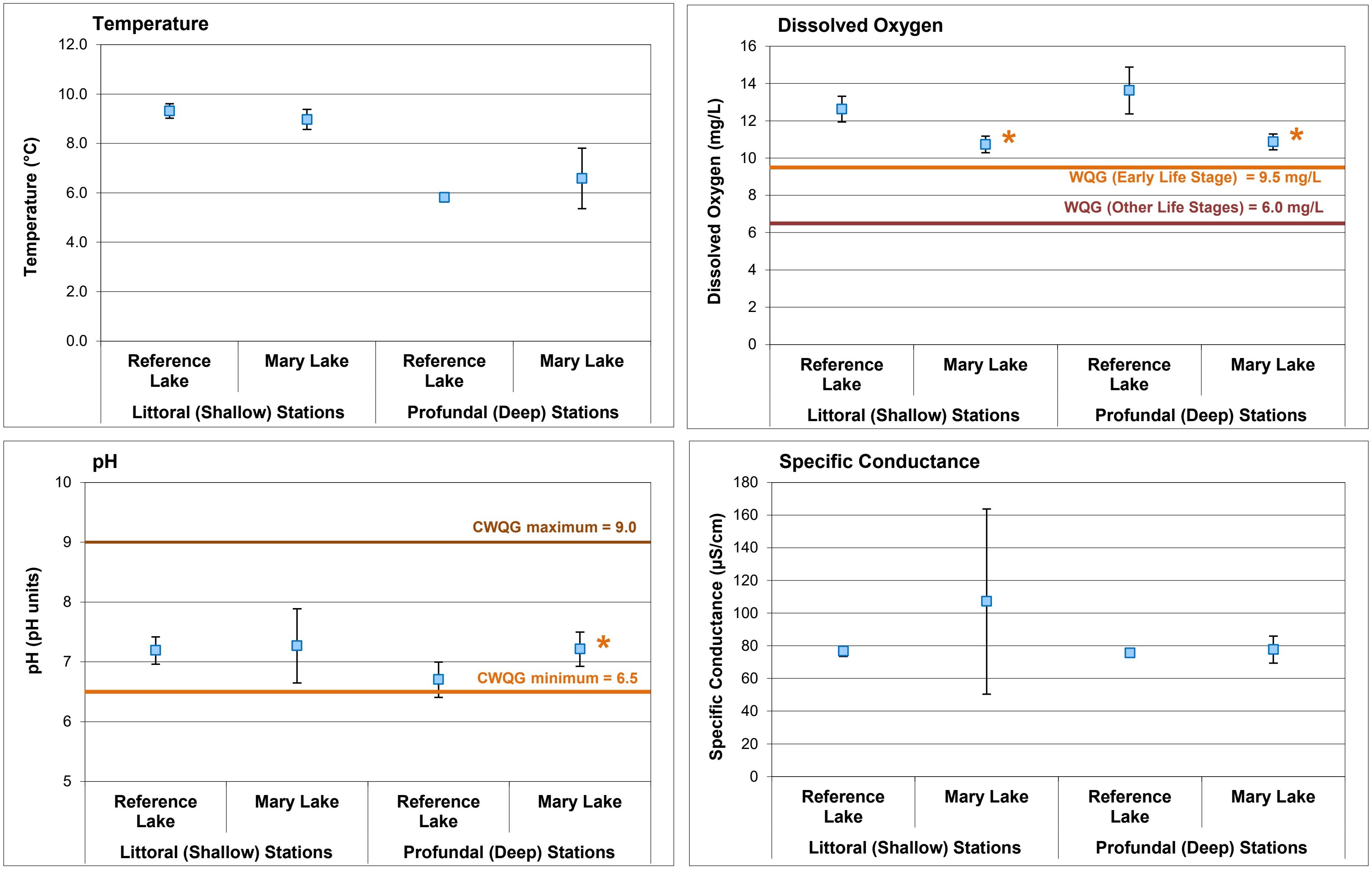


Figure 5.7: Comparison of *In Situ* Water Quality Variables (mean ± SD) Measured at Mary Lake (BLO) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2020

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.

readings, was significantly lower at Mary Lake compared to Reference Lake 3 in August 2020 (Appendix Table C.64; Appendix Figure C.8).

Water chemistry at Mary Lake north and south basins met all AEMP benchmarks and WQG over the duration of spring, summer, and fall sampling events in 2020 (Table 5.7). Among those parameters with established AEMP benchmarks, chloride concentrations were elevated at the north basin, and aluminum concentrations were elevated at both the north and south basins of Mary Lake compared to Reference Lake 3 in the summer and fall sampling events (Table 5.7; Appendix Table C.69). Of those parameters without AEMP benchmarks, turbidity was elevated at the north and south basins of Mary Lake, and total and dissolved manganese and uranium concentrations were elevated at the north basin of Mary Lake compared to the reference lake in both summer and fall sampling events in 2020 (Appendix Tables C.67). Similar to the Sheardown Lake system, elevated total aluminum concentrations at Mary Lake compared to the reference lake in 2020 were connected to naturally higher turbidity at Mary Lake and thus was unrelated to the mine operations.[19] Average concentrations of all parameters, including those with or without established AEMP benchmarks and, for metals, whether in total or dissolved form, were comparable between 2020 and baseline for the Mary River north basin and south basin (Appendix Figure C.29; Appendix Tables C.68, C.70, C.72, and C.73). Overall, mine-related influences on water quality of Mary Lake in 2020 included a slight elevation in chloride, manganese, and uranium concentrations compared to the reference lake. However, the occurrence of water quality below AEMP benchmarks and lack of water chemistry changes over time suggested no adverse mine-related influences on water chemistry of Mary Lake since the initiation of commercial mine operations in 2015.

5.3.2 Sediment Quality

Surficial sediment of the Mary Lake north basin (BLO-01) was mainly composed of silt with low TOC content (Figure 5.8; Appendix Table D.42). Sediments from the Mary Lake south basin also had low TOC content (i.e., <1.5%) and were predominantly composed of silt except at stations BLO-03 (profundal) and BLO-11 (littoral), which contained 92% and 79% sand, respectively (Figure 5.8; Appendix Table D.42). Substrate from Mary Lake was similar to that of Reference Lake 3 in terms of particle size, but had significantly lower TOC and moisture

[19] Turbidity at Mary Lake in 2020 was comparable to turbidity shown at the lake during baseline (Appendix Table C.69), suggesting that greater turbidity at this lake compared to Reference Lake 3 reflected natural phenomena. The occurrence of similar dissolved concentrations of aluminum between Mary Lake and Reference Lake 3 in 2020 (and historically) indicated that aluminum was associated with particulate material suspended in the water column, and thus was unlikely to be associated with mine-related source.

Table 5.7: Mean Water Chemistry at Mary Lake North Basin (BL0-01) and South Basin (BL0) Monitoring Stations[a], During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2020

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 (n = 3)		Mary Lake North Basin Stations (n = 3)			Mary Lake South Basin Stations (n = 7)		
					Summer	Fall	Winter	Summer	Fall	Winter	Summer	Fall
Conventionals	Conductivity (lab)	umho/cm	-	-	79	79	303	136	221	129	78	88
	pH (lab)	pH	6.5 - 9.0	-	7.66	7.75	7.66	8.11	8.21	7.65	7.66	7.73
	Hardness (as $CaCO_3$)	mg/L	-	-	35	38	156	63	106	67	34	41
	Total Suspended Solids (TSS)	mg/L	-	-	2.0	2.0	2.0	2.0	2.0	2.0	2.2	2.1
	Total Dissolved Solids (TDS)	mg/L	-	-	41	51	178	83	130	85	50	52
	Turbidity	NTU	-	-	0.15	0.1	0.1	0.9	0.6	0.1	2.0	1.4
	Alkalinity (as $CaCO_3$)	mg/L	-	-	46	34	132	61	96	59	43	38
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.010	0.014	0.010	0.010	0.010	0.010	0.010	0.010
	Nitrate	mg/L	3	3	0.020	0.020	0.119	0.008	0.044	0.052	0.037	0.043
	Nitrite	mg/L	0.06	0.06	0.005	0.005	0.005	0.001	0.005	0.005	0.008	0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	0.16	0.16	0.11	0.16	0.16	0.15	0.15
	Dissolved Organic Carbon	mg/L	-	-	3.3	3.5	3.5	2.3	2.8	3.1	2.2	1.9
	Total Organic Carbon	mg/L	-	-	4.6	3.8	4.9	1.7	10.5	3.6	3.0	2.3
	Total Phosphorus	mg/L	0.020^{α}	-	0.004	0.003	0.003	0.005	0.008	0.003	0.005	0.005
	Phenols	mg/L	0.004^{α}	-	0.0010	0.001	0.002	0.001	0.001	0.0016	0.0010	0.0011
Anions	Bromide (Br)	mg/L	-	-	0.10	0.10	0.10	0.05	0.10	0.10	0.16	0.10
	Chloride (Cl)	mg/L	120	120	1.4	1.37	17.52	4.18	9.42	4.56	2.75	3.40
	Sulphate (SO_4)	mg/L	218^{β}	218	3.6	3.64	7.32	1.86	4.02	3.87	1.84	2.34
Total Metals	Aluminum (Al)	mg/L	0.100	0.130	0.0031	0.0032	0.0035	0.0254	0.0199	0.0061	0.0648	0.0383
	Antimony (Sb)	mg/L	0.020^{α}	-	0.0001	0.00010	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Barium (Ba)	mg/L	-	-	0.0064	0.0070	0.0145	0.0071	0.0107	0.0070	0.0046	0.0051
	Beryllium (Be)	mg/L	0.011^{α}	-	0.0005	0.0005	0.0005	0.0001	0.0005	0.0005	0.0005	0.0005
	Bismuth (Bi)	mg/L	-	-	0.0005	0.0005	0.0005	0.00005	0.0005	0.0005	0.0005	0.0005
	Boron (B)	mg/L	1.5	-	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
	Cadmium (Cd)	mg/L	0.00012	0.00006	0.00001	0.00001	0.00001	0.000005	0.00001	0.00001	0.00001	0.00001
	Calcium (Ca)	mg/L	-	-	7.2	7.2	31.1	12.7	21.3	13.0	7.0	7.9
	Chromium (Cr)	mg/L	0.0089	0.0089	0.0005	0.0005	0.00050	0.00013	0.00050	0.0005	0.0005	0.0005
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.00073	0.00075	0.00115	0.00092	0.00097	0.00074	0.00059	0.00059
	Iron (Fe)	mg/L	0.30	0.326	0.03	0.030	0.032	0.055	0.030	0.030	0.065	0.040
	Lead (Pb)	mg/L	0.001	0.001	0.00005	0.00005	0.00005	0.00005	0.00005	0.000050	0.000071	0.000050
	Lithium (Li)	mg/L	-	-	0.0010	0.0010	0.0014	0.0010	0.0014	0.0010	0.0010	0.0010
	Magnesium (Mg)	mg/L	-	-	4.24	4.7	19.3	7.7	12.7	7.8	4.1	4.9
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00080	0.00068	0.00392	0.00421	0.00240	0.00051	0.00201	0.00115
	Mercury (Hg)	mg/L	0.000026	-	0.000005	0.000005	0.000005	0.000005	0.000005	0.000005	0.000005	0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.00013	0.00015	0.00037	0.00020	0.00032	0.00024	0.00100	0.00017
	Nickel (Ni)	mg/L	0.025	0.025	0.00050	0.00050	0.00075	0.00052	0.00050	0.00050	0.00050	0.00050
	Potassium (K)	mg/L	-	-	0.86	0.90	1.41	0.87	1.16	0.84	0.57	0.63
	Selenium (Se)	mg/L	0.001	-	0.001	0.001	0.001	0.00005	0.001	0.001	0.001	0.001
	Silicon (Si)	mg/L	-	-	0.495	0.50	1.51	0.67	0.74	0.61	0.53	0.49
	Silver (Ag)	mg/L	0.00025	0.0001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.00001
	Sodium (Na)	mg/L	-	-	0.89	0.96	6.88	2.38	4.33	2.10	1.29	1.60
	Strontium (Sr)	mg/L	-	-	0.0084	0.0082	0.0224	0.0098	0.0163	0.0115	0.0062	0.0070
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00010	0.0001	0.0001	0.0000	0.0001	0.0001	0.0001	0.0001
	Tin (Sn)	mg/L	-	-	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
	Titanium (Ti)	mg/L	-	-	0.01	0.0100	0.0100	0.0009	0.01	0.01	0.01	0.01
	Uranium (U)	mg/L	0.015	-	0.0003	0.0003	0.0045	0.0012	0.00344	0.00135	0.00070	0.00097
	Vanadium (V)	mg/L	0.006^{α}	0.006	0.0010	0.0010	0.0010	0.0005	0.0010	0.0010	0.0010	0.0010
	Zinc (Zn)	mg/L	0.030	0.030	0.0030	0.0030	0.0030	0.0030	0.003	0.003	0.004	0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

[a] Values presented are averages from samples taken from the surface and the bottom of the water column at each lake for the indicated season

[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 to 2013) specific to Mary Lake.

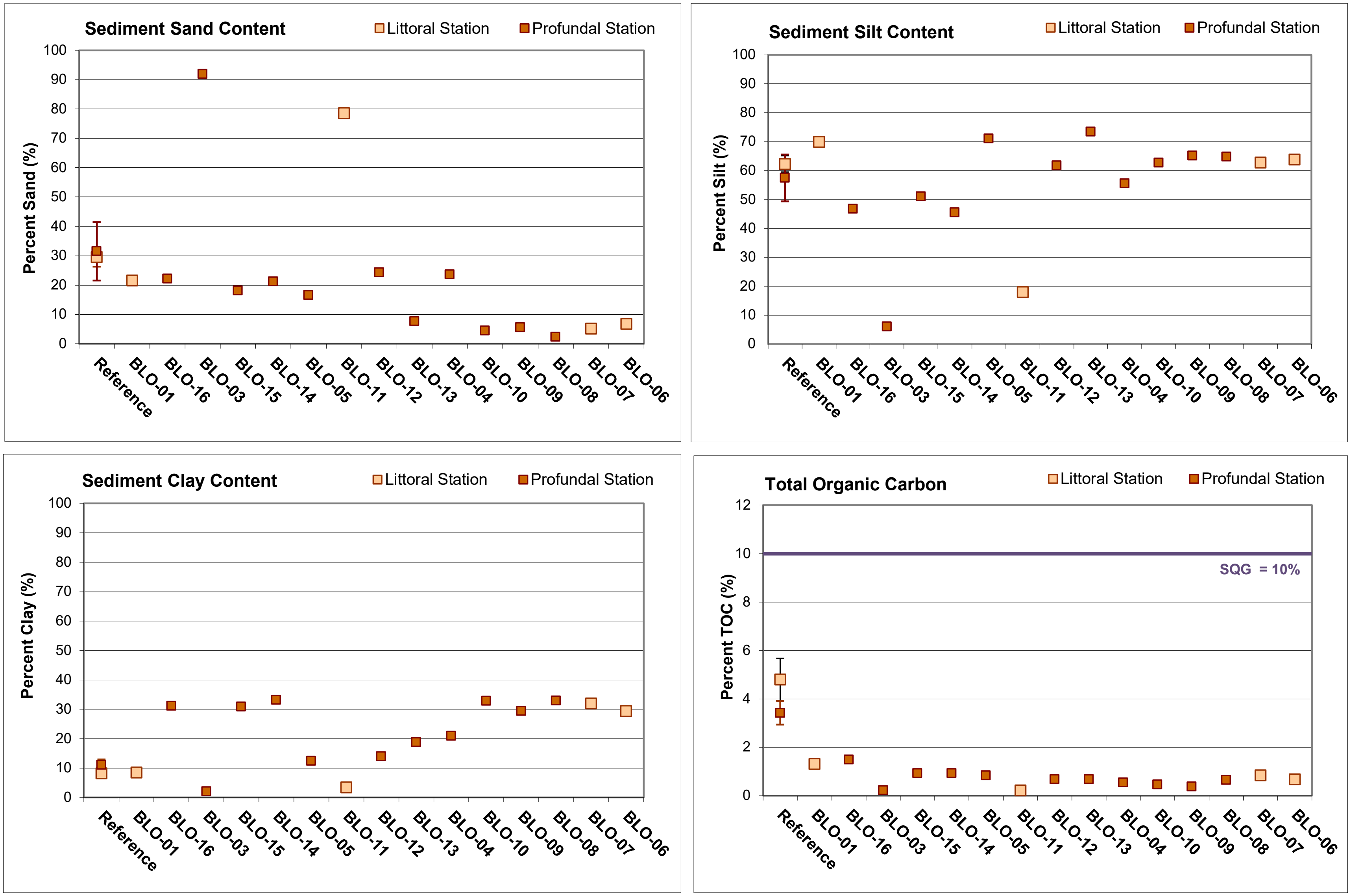


Figure 5.8: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Mary Lake (BLO) Sediment Monitoring Stations and to Reference Lake 3 (mean ± SE), Mary River Project CREMP, August 2020

content for both littoral and profundal habitat (Appendix Table D.43). Reddish-brown coloured iron (oxy)hydroxide material was evident at various sediment sampling locations throughout Mary Lake (Appendix Tables D.40 and D.41), mirroring similar observations at Reference Lake 3 and the other mine-exposed lakes where such material was commonly visible as a thin, distinct layer or floc on or within surficial sediment. Substrate of Mary Lake also commonly contained sub-surface blackening/dark colouration indicating the presence of reduced sediment, but no distinct redox boundaries were observed (Appendix Tables D.40 and D.41). Similar sub-surface reducing conditions were observed in sediment of the reference lake, including the absence of distinct redox boundaries (Appendix Tables D.3 and D.4), suggesting that factors leading to reduced sediment conditions were comparable between lakes.

Metal concentrations in sediment at Mary Lake were comparable to those of Reference Lake 3 in 2020 except for slightly (i.e., 3- to 5-fold) higher concentrations of zirconium at both littoral and profundal habitat (Appendix Table D.44). At the lone Mary Lake north basin station (i.e., BLO-01), concentrations of all metals in sediment were below applicable SQG and lake-specific AEMP benchmarks except for manganese, which exceeded the SQG only (Table 5.8). In the south basin, concentrations of chromium and iron in sediment from the littoral station, and mean manganese concentrations in sediments from the profundal stations, were above applicable SQG but not lake-specific AEMP benchmarks (Table 5.8). Metal concentrations in sediment at the Mary Lake south basin showed no spatial gradients with progression from the Mary River inlet to the lake outlet among the profundal stations (Appendix Table D.42).[20] As indicated previously, mean concentrations of iron and manganese were elevated above SQG in sediment at the reference lake (Table 5.8), suggesting that concentrations of iron and manganese above SQG at Mary Lake likely reflected a natural condition unrelated to mine activity.

Metal concentrations in sediment at littoral and profundal stations of Mary Lake in 2020 have not changed substantially from those observed during the mine baseline (2005 to 2013) period (Figure 5.9; Appendix Table D.44).[21] On average, metal concentrations in sediment from Mary Lake were within the range of those observed from 2015 to 2019 and there was no evidence of an increasing trend over time for any metals (Figure 5.9). Overall, no changes in metal concentrations in sediment were apparent in sediments at Mary Lake since the initiation of commercial mine operations in 2015.

[20] Spatially, the order of sediment quality from closest to Mary River to the lake outlet were as follows: BLO-12, BLO-10, BLO-09, BLO-08, and BLO-06 (Figure 2.4). All of these stations, except BLO-06, were profundal.

[21] See footnote 6 regarding differences in the concentration of boron in sediment between baseline and recent CREMP studies.

Table 5.8: Sediment Total Organic Carbon and Metal Concentrations at Mary Lake North Basin (BLO-01) and South Basin (BLO), and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2020

	Parameter	Units	SQG[a]	AEMP Benchmark[b]	Littoral: Reference Lake (n = 5) Average ± SD	Littoral: Mary Lake North (n = 1)	Littoral: Mary Lake South (n = 1)	Profundal: Reference Lake (n = 5) Average ± SD	Profundal: Mary Lake (South Basin) (n = 8) Average ± SD
TOC		%	10^{α}	-	4.80 ± 1.96	1.31	0.68	3.42 ± 1.08	0.71 ± 0.34
Metals	Aluminum (Al)	mg/kg	-	-	16,880 ± 1,785	15,000	29,600	21,800 ± 2,185	22,616 ± 2,156
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10	<0.10	<0.10 ± 0	<0.10 ± 0
	Arsenic (As)	mg/kg	17	5.9	3.53 ± 1.09	4.82	3.47	4.07 ± 0.397	3.09 ± 0.43
	Barium (Ba)	mg/kg	-	-	117 ± 22.0	79.2	104	122 ± 18.3	86.3 ± 7.97
	Beryllium (Be)	mg/kg	-	-	0.65 ± 0.073	0.72	1.33	0.80 ± 0.092	1.02 ± 0.10
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	<0.20	0.23	<0.20 ± 0	0.24 ± 0.0092
	Boron (B)	mg/kg	-	-	12.2 ± 0.853	18.9	43.9	14.7 ± 1.77	30.1 ± 2.29
	Cadmium (Cd)	mg/kg	3.5	1.5	0.173 ± 0.047	0.108	0.145	0.148 ± 0.0172	0.138 ± 0.0117
	Calcium (Ca)	mg/kg	-	-	5,608 ± 1,247	7,720	5,310	5,010 ± 407	4,213 ± 662
	Chromium (Cr)	mg/kg	90	98	54.3 ± 4.40	60.6	94.1	65.0 ± 6.64	79.6 ± 7.46
	Cobalt (Co)	mg/kg	-	-	10.8 ± 1.64	14.0	18.7	15.2 ± 1.56	15.8 ± 1.30
	Copper (Cu)	mg/kg	110	50	***71.4*** ± 14.2	27.8	36.0	***83.8*** ± 11.1	31.3 ± 3.16
	Iron (Fe)	mg/kg	$40{,}000^{\alpha}$	52,400	50,600 ± 24,939	34,500	46,600	45,080 ± 4,440	39,013 ± 3,068
	Lead (Pb)	mg/kg	91.3	35	13.8 ± 0.799	14.5	25.5	16.7 ± 1.82	20.6 ± 2.15
	Lithium (Li)	mg/kg	-	-	26.0 ± 2.51	29.2	50.7	33.7 ± 3.83	39.5 ± 4.06
	Magnesium (Mg)	mg/kg	-	-	11,440 ± 814	14,500	18,700	14,180 ± 1,422	15,421 ± 1,463
	Manganese (Mn)	mg/kg	$1{,}100^{\alpha,\beta}$	4,370	579 ± 258	1,350	778	1,230 ± 355	1,693 ± 268
	Mercury (Hg)	mg/kg	0.486	0.170	0.0500 ± 0.0178	0.0293	0.0541	0.0583 ± 0.0164	0.0527 ± 0.0074
	Molybdenum (Mo)	mg/kg	-	-	4.4 ± 3.31	0.52	0.92	2.52 ± 0.273	0.98 ± 0.14
	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	72	40.0 ± 3.52	51.7	62.0	45.0 ± 4.54	57.8 ± 5.11
	Phosphorus (P)	mg/kg	$2{,}000^{\alpha}$	1,580	1,167 ± 394	1,100	849	956 ± 47	855 ± 82
	Potassium (K)	mg/kg	-	-	4,100 ± 453	3,470	7,650	5,338 ± 543	5,685 ± 579
	Selenium (Se)	mg/kg	-	-	0.73 ± 0.31	<0.20	0.26	0.61 ± 0.18	0.24 ± 0.008
	Silver (Ag)	mg/kg	-	-	0.14 ± 0.047	<0.10	0.16	0.20 ± 0.057	0.15 ± 0.0104
	Sodium (Na)	mg/kg	-	-	304 ± 32	238	453	369 ± 50	354 ± 35
	Strontium (Sr)	mg/kg	-	-	11.6 ± 1.70	11.1	16.7	12.3 ± 1.24	13.5 ± 1.18
	Sulphur (S)	mg/kg	-	-	1,400 ± 387	<1,000	<1,000	1,140 ± 195	<1,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.379 ± 0.0415	0.298	0.603	0.594 ± 0.094	0.461 ± 0.0472
	Tin (Sn)	mg/kg	-	-	<2.0 ± 0	<2.0	<2.0	<2.0 ± 0	<2.0 ± 0
	Titanium (Ti)	mg/kg	-	-	1,006 ± 109	978	1,970	1,136 ± 50	1,466 ± 142
	Uranium (U)	mg/kg	-	-	11.0 ± 2.41	3.35	8.48	19.7 ± 3.76	7.13 ± 0.771
	Vanadium (V)	mg/kg	-	-	54.1 ± 5.40	48.0	78.6	63.4 ± 4.89	61.7 ± 5.63
	Zinc (Zn)	mg/kg	315	135	73.1 ± 7.83	48.1	86.2	83.8 ± 8.52	67.4 ± 6.56
	Zirconium (Zr)	mg/kg	-	-	4.5 ± 1.0	10.4	25.9	3.9 ± 0.32	20.4 ± 2.7

(grey box) Indicates parameter concentration above SQG.
BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation.

[a] Canadian SQG for the protection of aquatic life probable effects level (PEL; CCME 2020) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BC ENV 2020).

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

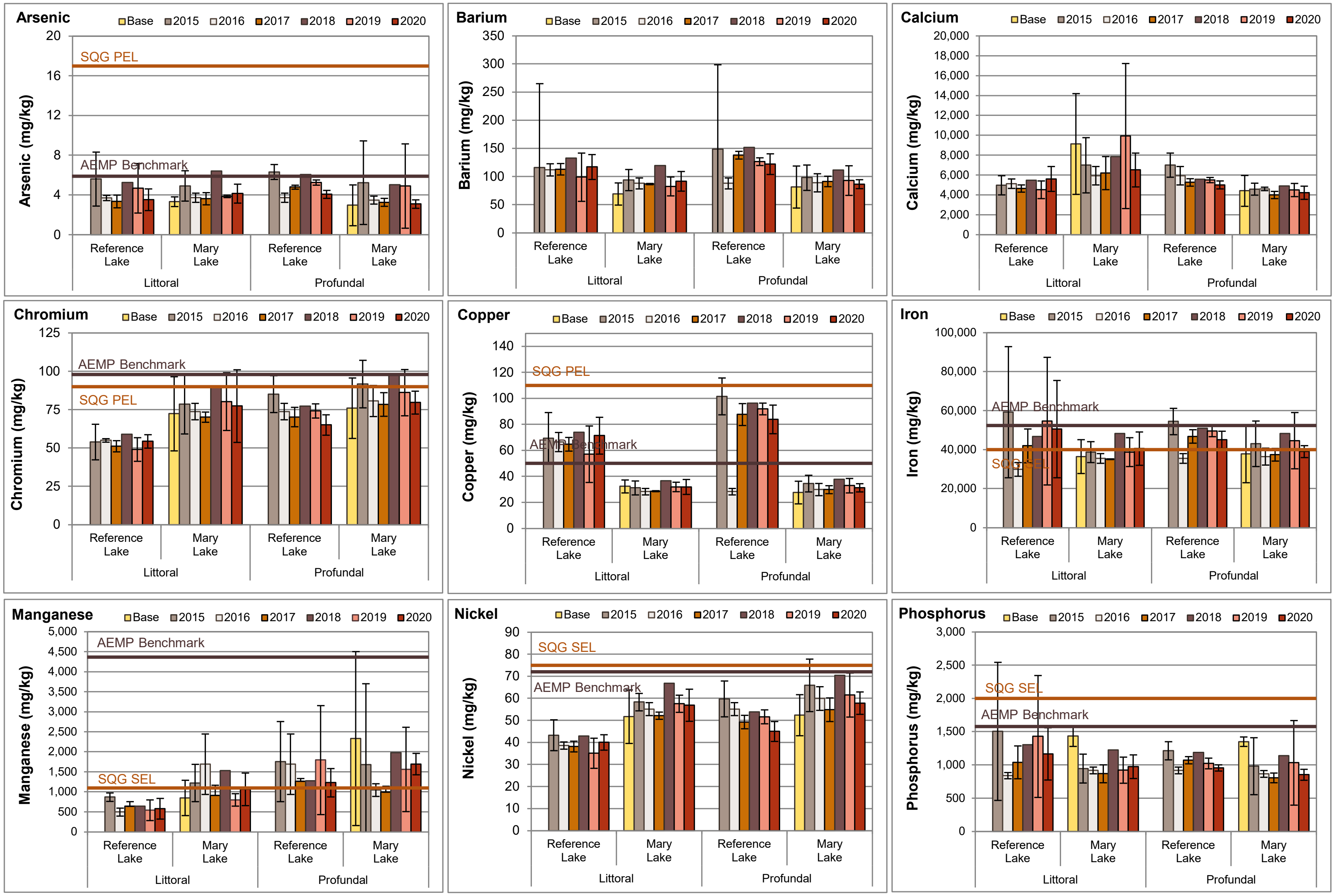


Figure 5.9: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Mary Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2020) Periods

5.3.3 Phytoplankton

Chlorophyll-a concentrations at Mary Lake showed no spatial gradients with distance from either the Tom River inlet or the Mary River inlet towards the lake outlet during any of the winter, summer, or fall sampling events in 2020 (Figure 5.10). Chlorophyll-a concentrations were typically lowest in winter and highest in summer at both the north and south basins of Mary Lake (Figure 5.10). Chlorophyll-a concentrations at the Mary Lake north basin did not differ significantly from those at Reference Lake 3 in fall and summer, or between the Mary Lake south basin and Reference Lake 3 in the fall sampling event, but concentrations at the south basin were significantly higher than at the reference lake at the time of the summer sampling event (Appendix Tables E.7 and E.8). Chlorophyll-a concentrations at the Mary Lake north and south basins were well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events in 2020 (Figure 5.10) and reflected an 'oligotrophic' primary productivity categorization based on Wetzel (2001) classification. This oligotrophic categorization agreed with CWQG trophic status classification that is based on average aqueous total phosphorus concentrations below 10 µg/L (Table 5.7; Appendix Tables C.68 and C.72).

Temporal comparison of Mary Lake chlorophyll-a concentrations, conducted separately for the north and south basins, did not indicate any consistent direction of significant differences between the 2019 data and data from the mine construction (2014) period or previous years of mine operation (2015 to 2019) during any of the winter, summer, or fall seasons (Figure 5.11; Appendix Figure E.1). In addition, annual average chlorophyll-a concentrations have not shown any consistent direction of change (i.e., increase or decrease) over time since the mine was constructed in 2014 (Figure 5.11; Appendix Figure E.1) suggesting no substantial changes in the trophic status of the lake since mine operations commenced at the Mary River Project. No chlorophyll-a baseline (2005 to 2013) data are available for Mary Lake, precluding comparisons to conditions prior to mine construction.

5.3.4 Benthic Invertebrate Community

Benthic invertebrate density, richness, and evenness at littoral and profundal habitat of Mary Lake did not differ significantly compared to like-habitat stations at Reference Lake 3 in 2020 (Tables 5.9 and 5.10). Benthic invertebrate community compositional differences were indicated between Mary Lake and Reference Lake 3 based on significantly differing Bray-Curtis Index for both littoral and profundal habitat types (Appendix Table F.21), but no significant differences in dominant taxonomic groups were indicated between Mary Lake and the reference lake for either habitat (Tables 5.9 and 5.10). Rather, significantly lower relative abundance of the filterer FFG and the clinger HPG at Mary Lake littoral and profundal stations compared to like-habitat stations at Reference Lake 3 (Tables 5.9 and 5.10) suggested that differences in the community

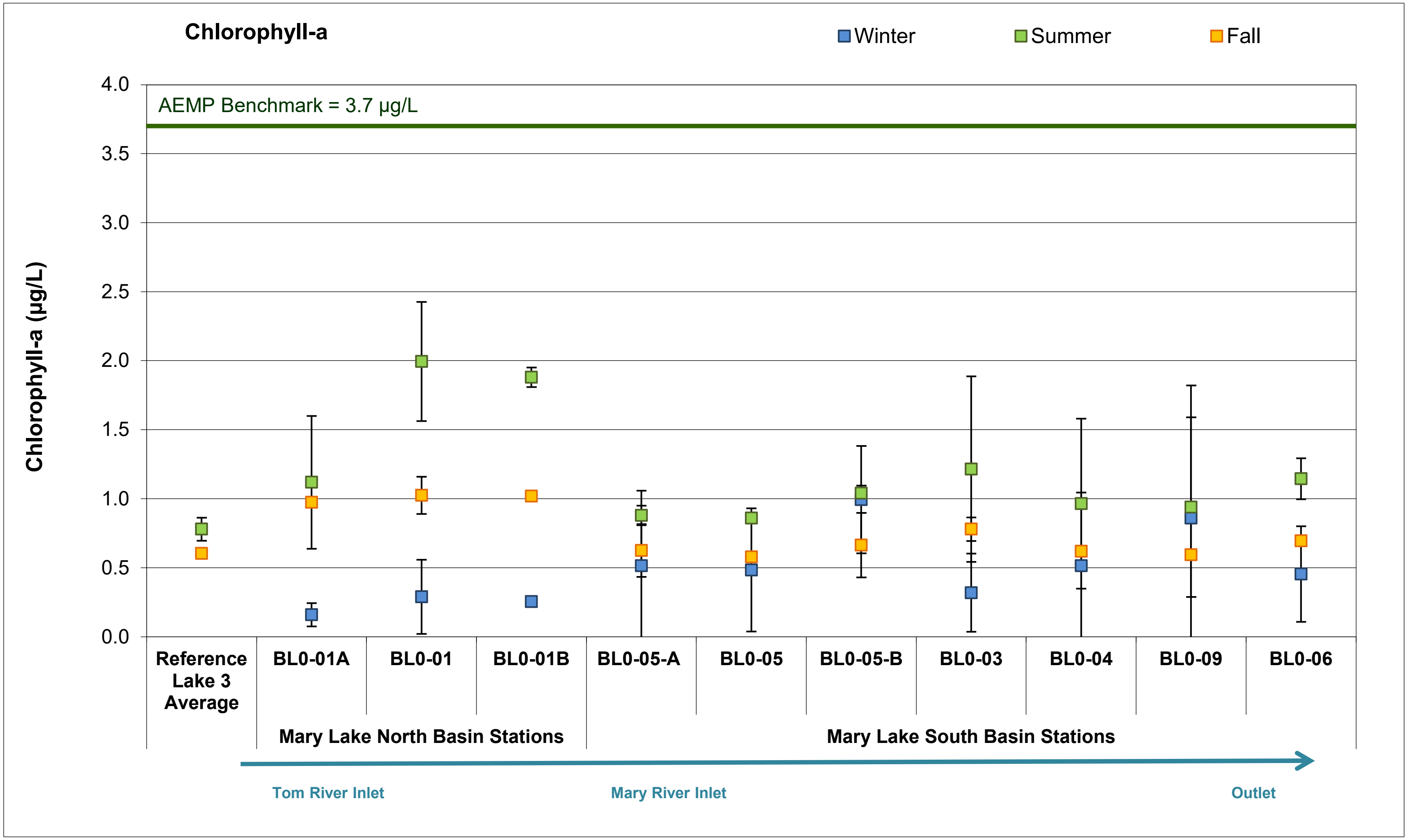


Figure 5.10: Chlorophyll-a Concentrations at Mary Lake (BLO) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2020

Notes: Values presented are averages of samples taken from the surface and the bottom of the water column at each station. Reference lake values represent mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2020.

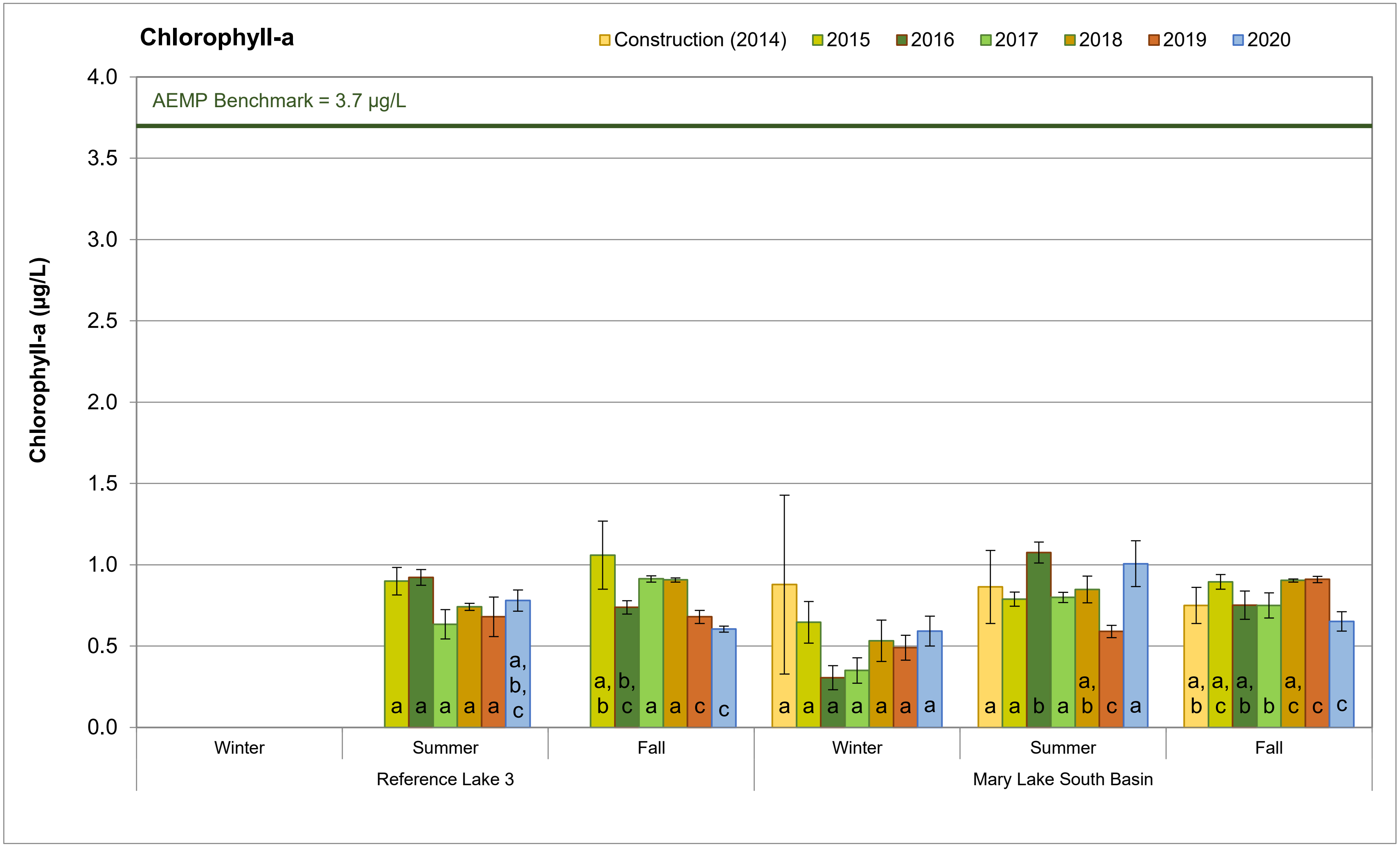


Figure 5.11: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between the Mary Lake South Basin and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2020) Periods (mean ± SE)

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

Table 5.9: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BLO) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tunequal	log10	NO	0.401	5.8	Reference Lake 3	1,571	430	193	1,190	1,474	2,310
						Mary Lake Littoral	4,086	3,955	1,978	966	3,017	9,345
Richness (Number of Taxa)	t-unequal	log10	NO	0.244	-1.2	Reference Lake 3	14.6	2.5	1.1	13.0	14.0	19.0
						Mary Lake Littoral	11.5	4.2	2.1	7.0	11.5	16.0
Simpson's Evenness (E)	t-equal	none	NO	0.464	-0.4	Reference Lake 3	0.810	0.110	0.049	0.630	0.847	0.923
						Mary Lake Littoral	0.765	0.041	0.021	0.707	0.775	0.802
Hydracarina (%)	t-equal	log10	NO	0.241	-0.6	Reference Lake 3	5.3	2.6	1.2	3.5	4.4	9.9
						Mary Lake Littoral	3.7	2.9	1.4	1.7	2.7	7.8
Ostracoda (%)	t-equal	log10	NO	0.606	0.3	Reference Lake 3	37.9	14.5	6.5	26.7	36.2	62.6
						Mary Lake Littoral	42.7	12.7	6.4	23.8	48.2	50.7
Chironomidae (%)	t-equal	none	NO	0.987	0.0	Reference Lake 3	52.6	15.6	7.0	26.9	59.0	66.4
						Mary Lake Littoral	52.5	14.7	7.4	43.8	45.8	74.4
Metal-Sensitive Chironomidae (%)	t-equal	none	YES	0.008	-2.2	Reference Lake 3	28.8	9.5	4.3	15.6	32.5	38.7
						Mary Lake Littoral	8.1	6.6	3.3	3.9	5.2	17.9
Collector-Gatherers (%)	t-equal	none	NO	0.661	0.5	Reference Lake 3	63.1	11.4	5.1	53.6	60.3	81.5
						Mary Lake Littoral	68.8	24.9	12.5	34.5	73.2	94.1
Filterers (%)	t-equal	log10(x+1)	YES	0.009	-2.2	Reference Lake 3	27.1	9.8	4.4	14.4	29.2	38.0
						Mary Lake Littoral	5.5	8.3	4.2	0.0	2.1	17.6
Shredders (%)	t-unequal	none	YES	0.064	-1.1	Reference Lake 3	3.9	3.3	1.5	0.6	3.2	7.4
						Mary Lake Littoral	0.1	0.2	0.1	0.0	0.0	0.4
Clingers (%)	t-equal	log10	YES	0.004	-2.5	Reference Lake 3	31.9	9.3	4.2	17.9	33.5	41.6
						Mary Lake Littoral	8.7	6.2	3.1	3.6	6.8	17.6
Sprawlers (%)	t-equal	log10	YES	0.033	1.9	Reference Lake 3	57.9	12.1	5.4	41.0	57.2	73.8
						Mary Lake Littoral	80.7	11.8	5.9	65.6	83.6	90.2
Burrowers (%)	t-equal	log10	NO	0.582	0.1	Reference Lake 3	10.2	4.9	2.2	4.6	8.3	17.3
						Mary Lake Littoral	10.6	12.1	6.1	2.3	5.7	28.5

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

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

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

Table 5.10: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BLO) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2020

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	p-value	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	t-equal	log10	NO	0.308	2.1	Reference Lake 3	479	142	63	336	491	681
						Mary Lake Profundal	779	452	184	216	625	1,362
Richness (Number of Taxa)	t-equal	log10	NO	0.814	0.3	Reference Lake 3	7.0	1.9	0.8	5.0	8.0	9.0
						Mary Lake Profundal	7.5	2.8	1.2	5.0	6.0	12.0
Simpson's Evenness (E)	t-unequal	none	NO	0.140	-3.2	Reference Lake 3	0.731	0.045	0.020	0.689	0.721	0.795
						Mary Lake Profundal	0.586	0.201	0.082	0.322	0.564	0.834
Hydracarina (%)	t-equal	log10(x+1)	NO	0.166	0.9	Reference Lake 3	2.8	2.0	0.9	0.0	3.5	5.1
						Mary Lake Profundal	4.7	2.1	0.9	2.5	4.3	8.0
Ostracoda (%)	t-unequal	log10	NO	0.810	3.6	Reference Lake 3	8.6	4.1	1.8	3.5	7.7	14.5
						Mary Lake Profundal	23.4	28.9	11.8	1.3	14.2	76.0
Chironomidae (%)	Mann Whitney	rank	NO	0.792	-4.0	Reference Lake 3	87.9	4.2	1.9	82.3	87.2	92.7
						Mary Lake Profundal	71.3	30.2	12.3	16.0	80.5	94.6
Metal-Sensitive Chironomidae (%)	t-equal	none	YES	0.007	-1.5	Reference Lake 3	31.5	17.6	7.9	7.9	38.0	49.3
						Mary Lake Profundal	5.6	5.2	2.1	0.0	4.1	14.7
Collector-Gatherers (%)	t-equal	none	YES	0.006	1.6	Reference Lake 3	62.9	15.0	6.7	45.4	56.1	79.0
						Mary Lake Profundal	86.6	5.9	2.4	76.3	87.7	92.1
Filterers (%)	t-equal	log10(x+1)	YES	0.004	-1.6	Reference Lake 3	30.7	17.5	7.8	7.9	38.0	49.3
						Mary Lake Profundal	3.2	5.3	2.2	0.0	0.7	13.6
Shredders (%)	t-equal	log10(x+1)	NO	0.261	-0.7	Reference Lake 3	2.2	2.3	1.0	0.0	2.5	5.3
						Mary Lake Profundal	0.8	1.9	0.8	0.0	0.0	4.5
Clingers (%)	t-equal	log10	YES	0.036	-1.2	Reference Lake 3	33.5	16.9	7.6	13.1	41.5	52.8
						Mary Lake Profundal	12.5	11.1	4.5	2.7	9.1	32.0
Sprawlers (%)	Mann Whitney	rank	NO	0.177	0.7	Reference Lake 3	64.8	16.2	7.2	45.5	58.5	87.0
						Mary Lake Profundal	75.5	32.9	13.4	9.3	88.5	93.8
Burrowers (%)	Mann Whitney	rank	NO	0.191	3.6	Reference Lake 3	1.7	2.9	1.3	0.0	0.0	6.7
						Mary Lake Profundal	12.0	23.0	9.4	0.0	3.4	58.7

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

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

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

between lakes reflected slight differences in dominant food resources available to benthic invertebrates and/or physical habitat features, respectively. Although the relative abundance of metal-sensitive Chironomidae was significantly lower at Mary Lake than at Reference Lake 3 (Tables 5.9 and 5.10), metal concentrations in water and sediment of Mary Lake were comparable to those at the reference lake (Tables 5.7 and 5.8), suggesting that differences in benthic invertebrate community features between lakes were not related to metal concentrations.

No significant differences in benthic invertebrate density, richness, evenness, relative abundance of dominant groups, and relative abundance of FFG were shown consistently at Mary Lake littoral and profundal habitat over years of mine operation (2015 to 2020) compared to baseline (Appendix Figures F.15 and F.16; Appendix Tables F.61 and F.62). In addition, no significant differences in the relative abundance of metal-sensitive Chironomidae were indicated for years of mine-operation relative to baseline (Appendix Tables F.61 and F.62), indicating that the differences in the relative abundance of this group between Mary Lake and Reference Lake 3 in 2020 likely reflected natural variability. Therefore, consistent with no substantial changes in water and sediment quality since the mine baseline period, no ecologically significant changes in benthic invertebrate community features were indicated at littoral and profundal habitat of Mary Lake since the commencement of commercial mine operation in 2015.

5.3.5 Fish Population

5.3.5.1 Mary Lake (South) Fish Community

Arctic charr and ninespine stickleback were captured in Mary Lake in 2020 (Table 5.6), consistent with the previous five years of sampling (Minnow 2020). Electrofishing and gill netting CPUE were each higher at Mary Lake than at Reference Lake 3 (Table 5.11), suggesting greater densities and/or productivity of both arctic charr and ninespine stickleback at Mary Lake. Consistent with the other mine-exposed lakes, greater numbers of arctic charr together with greater density of benthic invertebrates suggested that overall biological productivity was higher at Mary Lake than at Reference Lake 3. Arctic charr CPUE associated with electrofishing in 2020 at Mary Lake was comparable to highest CPUE from other years of mine operation and substantially greater than baseline monitoring conducted in 2008 (Figure 5.12). Gill netting CPUE at Mary Lake in 2020 was within the range of observed during previous years of mine operation (2015 to 2019), and also greater than CPUE during baseline (2006 and 2007; Figure 5.12). Based on the CPUE data, arctic charr abundances at nearshore and littoral/profundal habitats of Mary Lake were likely comparable to or greater than during the baseline period, indicating no mine-related influences on arctic charr abundance in the lake.

Table 5.11: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Mary Lake (BLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2020

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	134	1	135	2
		CPUE	2.09	0.016	2.11	
	Gill netting	No. Caught	69	0	69	
		CPUE	0.956	0	0.956	
Mary Lake	Electrofishing	No. Caught	105	26	131	2
		CPUE	3.01	0.746	3.76	
	Gill netting	No. Caught	94	0	94	
		CPUE	4.60	0	4.60	

[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.

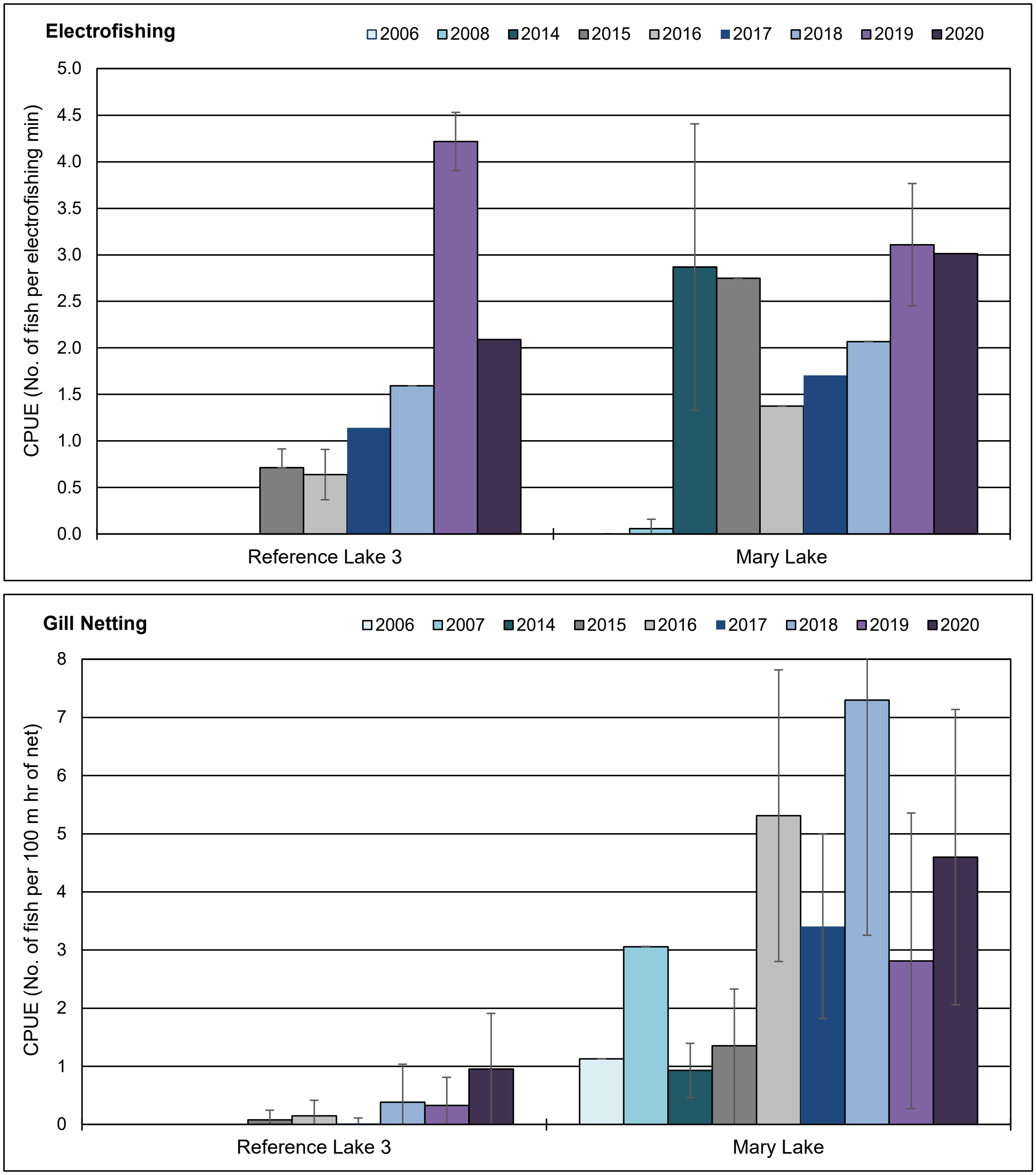


Figure 5.12: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Back-pack Electrofishing and Gill Netting at Mary Lake (BLO), Mary River Project CREMP, 2006 to 2020

Note: Data presented for fish sampling conducted in fall during baseline (2006, 2007), construction (2014), and operational (2015 to 2020) mine phases.

5.3.5.2 Mary Lake (South) Fish Population Assessment

Nearshore Arctic Charr

A total of 100 arctic charr were captured from nearshore habitats in each of Mary Lake and Reference Lake 3 in August 2020. Arctic charr YOY were distinguished from non-YOY using fork length cut-offs of 4.1 cm and 4.3 cm for the Mary Lake and Reference Lake 3 data sets, respectively, based on evaluation of length-frequency distributions coupled with supporting age determinations (Figure 5.13; Appendix Tables G.4 and G.25). However, due to small sample sizes of nearshore arctic charr YOY at Mary Lake (i.e., only two individuals), statistical comparisons of fish health endpoints were conducted using the non-YOY population only. Arctic charr of nearshore habitat showed differing length-frequency distributions between Mary Lake and Reference Lake 3, reflecting fewer YOY and a more limited size distribution of fish at Mary Lake compared to the reference lake (Table 5.12; Figure 5.13; Appendix Table G.26). Arctic charr non-YOY from Mary Lake were similar in size to reference lake fish, and although condition of non-YOY was significantly greater at Mary Lake than at the reference lake, the magnitude of this difference was well within the CES_C of ±10% indicating that this difference was not ecologically meaningful (Table 5.12; Appendix Table G.26). No consistent differences in size or condition of non-YOY arctic charr from nearshore habitat of Mary Lake relative were indicated relative to the reference lake from 2015 to 2020, suggesting that differences between lakes over time reflected natural variability (Table 5.12). No nearshore arctic charr baseline data were collected at Mary Lake, precluding data analysis using a before-after design. Collectively, the data indicated no adverse effects on arctic charr from nearshore areas in Mary Lake since the commencement of mine operations in 2015.

Littoral/Profundal Arctic Charr

A total of 94 and 69 arctic charr were sampled from littoral/profundal habitat of Mary Lake and Reference Lake 3, respectively, in August 2020. The length-frequency distribution for littoral/profundal arctic charr differed significantly between lakes due to a greater number of larger fish being caught at Mary Lake (Table 5.12; Figure 5.13; Appendix Table G.30). Arctic charr sampled from littoral/profundal habitat of Mary Lake were also significantly longer, heavier, and of greater condition than those from Reference Lake 3 in 2020 (Table 5.12; Appendix Table G.30). The absolute magnitude of difference in body condition was greater than the CES_C of 10%, suggesting that this difference may be ecologically significant (Table 5.12; Appendix Table G.30). An on-going significant difference in length-frequency distribution was the only consistent difference shown for arctic charr captured from littoral/profundal habitat of Mary Lake from 2015 to 2020 compared to the reference lake data for the same period and Mary Lake baseline data (Table 5.12; Appendix Table G.30). No consistent differences in arctic charr size and condition

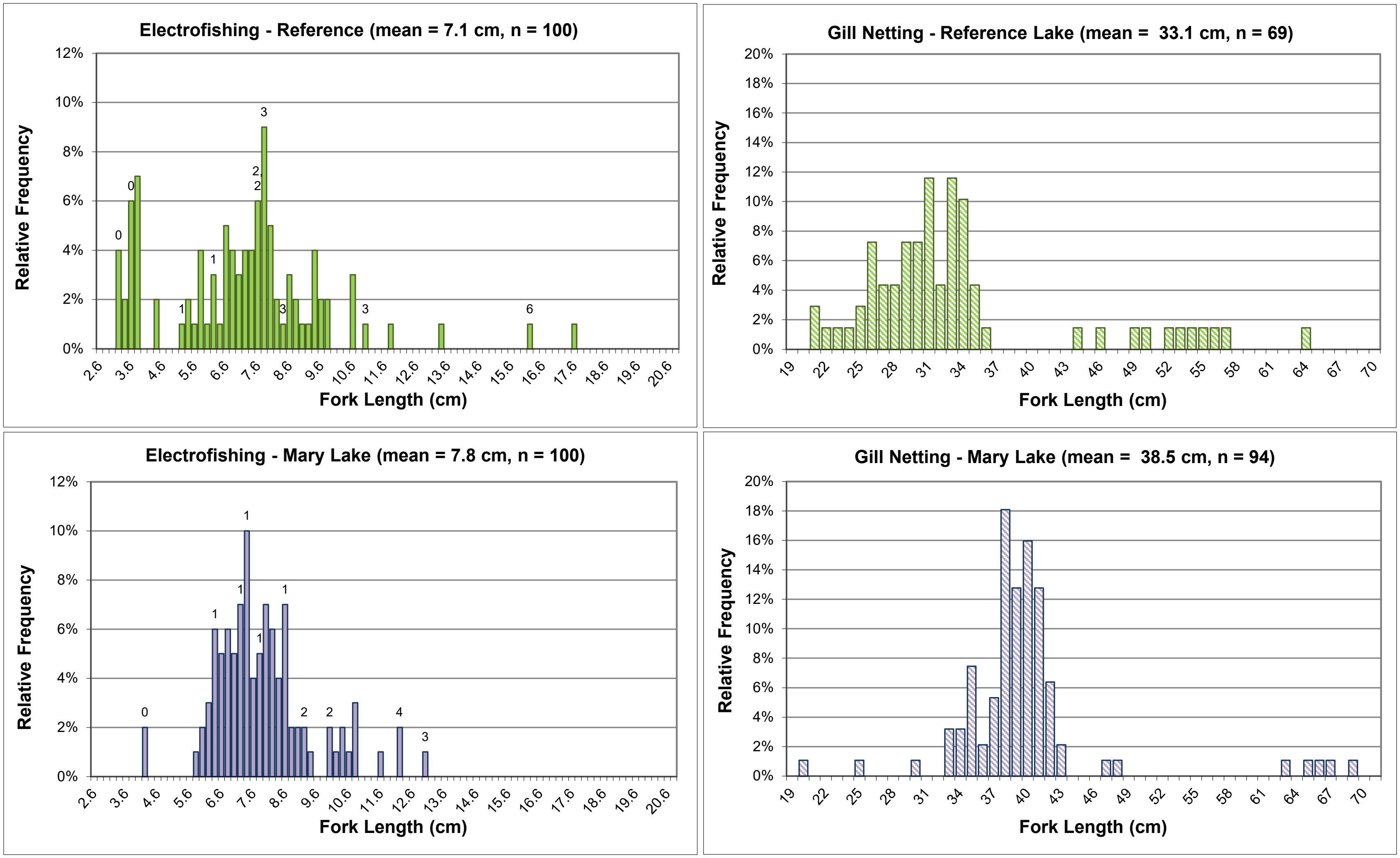


Figure 5.13: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Mary Lake (BLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2020

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

Table 5.12: Summary of Statistical Results for Arctic Charr Population Comparisons between Mary Lake and Reference Lake 3 from 2015 to 2020, and between Mary Lake Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]											
			versus Reference Lake 3						versus Mary Lake baseline period data [b]					
			2015	2016	2017	2018	2019	2020	2015	2016	2017	2018	2019	2020
Electrofishing Samples	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes (-27%)**	-	-	-	-	-	-
		Age	**Yes (-43%)**	No	No	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	No	No	**Yes (+17%)**	**Yes (+10%)**	**Yes (-27%)**	No	-	-	-	-	-	-
		Size (mean weight)	No	No	**Yes (+51%)**	No	**Yes (-61%)**	No	-	-	-	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	No	**Yes (-8%)**	**Yes (+4%)**	**Yes (+2.6%)**	-	-	-	-	-	-
Gill Netting Samples [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes (-64%)**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes (+21%)**
		Age	-	-	-	-	-	-	No	**Yes (-14%)**	No	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	**Yes (+12%)**	**Yes (+24%)**	**Yes (+23%)**	**Yes (+6%)**	No	**Yes (-5%)**	No	**Yes (-4%)**	No
		Size (mean weight)	-	-	-	**Yes (+51%)**	**Yes (+96%)**	**Yes (+118%)**	**Yes (+19%)**	No	**Yes (-9%)**	No	**Yes (-14%)**	No
		Growth (fork length-at-age)	-	-	-	-	-	-	No	**Yes (nc)**	No	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	No	**Yes (nc)**	No	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+3%)**	**Yes (+3%)**	**Yes (+14%)**	No	**Yes (+3%)**	**Yes (+5%)**	**Yes (-3%)**	**Yes (-5%)**	No

BOLD indicates a significant difference related to the comparison.

[a] Values in parentheses indicate direction and magnitude of any significant differences.

[b] No baseline period data collected for nearshore electrofishing; baseline period littoral/profundal gill netting data included combined 2006 and 2007 information.

[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.

endpoints for fish captured at littoral/profundal habitat of Mary Lake have occurred from 2015 to 2020 compared to baseline (Table 5.12). This suggested that natural and/or sampling variability likely accounted for the variable differences in arctic charr health endpoints between years of mine operation and baseline at Mary Lake.

5.3.6 Effects Assessment and Recommendations

At Mary Lake, the following AEMP benchmark was exceeded in 2020:

- Manganese concentration in sediment was greater than the benchmark of 4,370 mg/kg at one profundal monitoring station (BL0-09), although the average concentration of manganese in sediment at profundal stations was below this benchmark.

The AEMP benchmarks for sediment quality were exceeded only at a single profundal station and only for a single parameter (manganese) at Mary Lake in 2020. The isolated occurrence of this exceedance, and the fact that average manganese concentrations in sediment at Mary Lake were not particularly elevated compared to concentrations at the reference lake or to those at Mary Lake during baseline, indicated no mine-related change in manganese concentrations at Mary Lake since commercial mine operations commenced in 2015. No AEMP benchmarks for water quality were exceeded over the duration of spring, summer, and fall sampling events in 2020 at Mary Lake. In addition, no adverse effects on phytoplankton, benthic invertebrates, nor on fish (arctic charr) health were indicated at Mary Lake in 2020 based on comparisons to reference lake conditions and to Mary Lake baseline data. Under the Mary River Project AEMP Management Response Framework, the absence of a mine-related change in AEMP benchmark parameters over time (or compared to background) requires no further management response (Figure 2.8). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no management response (i.e., alteration of existing AEMP) is required for Mary Lake as part of the next monitoring program.

6 CONCLUSIONS

6.1 Overview

The objective of the Mary River Project 2020 CREMP was to evaluate potential mine-related influences on chemical and biological conditions at aquatic environments located near the mine following the sixth full year of mine operation. The CREMP employs an effects-based approach that includes standard environmental effects monitoring (EEM) techniques that were conducted as the basis for determining potential mine-related effects at key receiving waterbodies. Under this approach, water quality and sediment quality data were used to support the interpretation of phytoplankton, benthic invertebrate community, and fish population survey data collected at mine-exposed areas of the Camp Lake, Sheardown Lake, Mary River and Mary Lake systems. The evaluation of potential mine-related effects within these systems was based upon comparisons of the 2020 data to applicable reference data, baseline data, and to guidelines that included site-specific AEMP benchmarks. The latter were developed to guide management response decisions within a four-step Management Response Framework as outlined in the Mary River Project AEMP (Baffinland 2015). An effects determination was conducted for all key waterbodies located within each of the Camp Lake, Sheardown Lake, Mary River, and Mary Lake systems, which was based on weight-of-evidence that considered incidences in which the AEMP benchmarks were exceeded and a commensurate adverse influence on aquatic biota occurred. Where appropriate, recommendations for future study were provided to assist Baffinland with decisions regarding appropriate management actions for cases in which AEMP benchmarks were not achieved. Potential mine-related effects identified in the 2020 CREMP are provided separately below for the Camp, Sheardown and Mary River/Lake systems.

6.2 Camp Lake System

Within the Camp Lake system, AEMP monitoring is conducted at Camp Lake Tributary 1 (CLT1), Camp Lake Tributary 2 (CLT2), and Camp Lake (JL0). At CLT1, AEMP water quality benchmarks were exceeded in 2020 for copper at the north branch, and for aluminum and iron at the main stem portions of the system (Table 6.1). Copper concentrations at the CLT1 north branch were elevated compared to concentrations at reference creeks, but were comparable to those shown during baseline. Although elevated aluminum concentrations at the CLT1 main stem were not attributable to mine operations, iron concentrations at the CLT1 upper main stem in 2020 were elevated compared to those at reference creeks and to baseline suggesting a potential mine-related influence on CLT1 water quality. Metal concentrations in sediment at CLT1 were well below SQG. In addition, no adverse effects on phytoplankton (chlorophyll-a) or benthic invertebrates were indicated at CLT1 in 2020 compared to reference creek and CLT1 baseline

Table 6.1: Summary of AEMP Benchmark Exceedances and Effects Determination for the Mary River Project 2020 CREMP and Monitoring Recommendations Based on the Results

Waterbody	AEMP Benchmark Exceedance	Effects Determination Summary	Recommendation
Camp Lake Tributary 1 (North Branch)	Aqueous total copper concentration greater than 0.0022 mg/L benchmark in spring and summer at the north branch (0.00221 mg/L and 0.00226 mg/L, respectively).	Copper concentrations at the north branch were comparable to those during baseline. No adverse effects on phytoplankton or benthic invertebrates based on comparisons to reference data and to baseline data.	Low action response includes an expanded spatial water quality sampling program to identify the source(s) of copper to the watercourse.
Camp Lake Tributary 1 (Main Stem)	Aqueous total aluminum concentration greater than 0.179 mg/L benchmark in spring at upper main stem (0.270 mg/L). Aqueous total iron concentration greater than 0.326 mg/L benchmark in spring, summer, and fall at upper main stem (0.420 mg/L, 0.423 mg/L, and 0.522 mg/L, respectively).	Aluminum concentrations at CLT1 upper main stem were comparable to reference creeks and to baseline, and thus the change was not mine-related. Iron concentrations at CLT1 upper main stem were higher than background and reference, suggesting a mine-related change. No adverse effects on phytoplankton or benthic invertebrates based on comparisons to reference data and to baseline data at the CLT1 lower main stem.	Low action response includes establishing benthic invertebrate community monitoring stations at CLT1 upper main stem to evaluate/track effects to biota.
Camp Lake Tributary 2	Water quality met all AEMP benchmarks in 2020.	No adverse effects on phytoplankton or benthic invertebrates based on comparisons to reference data and to baseline data.	No changes recommended to monitoring program for CLT2 based on comparison to AEMP benchmarks.
Camp Lake	Sediment arsenic concentration > 5.9 mg/kg benchmark at single littoral monitoring station (9.0 mg/kg). Sediment iron concentration > 52,400 mg/kg benchmark at single littoral monitoring station (61,000 mg/kg). Sediment nickel concentration > 72 mg/kg benchmark at single littoral monitoring station (72.5 mg/kg). Sediment arsenic, copper, iron, manganese, nickel, and phosphorus concentrations above respective benchmarks at individual stations, but below benchmarks on average, at profundal stations.	No AEMP water quality benchmarks were exceeded at Camp Lake in 2020. For all parameters except arsenic, no change in parameter concentration in sediment was shown compared to background and/or baseline, indicating the change was not mine-related. Sediment chemistry is monitored only at a single littoral station at Camp Lake under the AEMP, and thus it is unclear whether the change in arsenic concentration is mine-related (e.g., no identifiable source of arsenic). No adverse effects on phytoplankton, benthic invertebrates, or fish compared to reference data and to baseline conditions.	Low action response to harmonize lake sediment quality and benthic invertebrate monitoring stations, focusing primarily on littoral habitat, to improve the ability of the program to evaluate changes in metal concentrations in littoral sediment and to track mine-related effects to biota.
Sheardown Lake Tributary 1	Aqueous total copper concentration greater than 0.0022 mg/L benchmark in spring, summer, and fall (0.0029 mg/L, 0.0024, and 0.0023 mg/L, respectively).	Copper concentrations at SDLT1 were comparable to those during baseline. No adverse effects on phytoplankton or benthic invertebrates based on comparisons to reference data and to baseline data.	Low action response includes an expanded spatial water quality sampling program to identify the source(s) of copper to the watercourse.
Sheardown Lake Tributaries 9 and 12	Water quality met all AEMP benchmarks in 2020.	No ecologically significant and/or adverse effects on phytoplankton or benthic invertebrate community endpoints based on comparisons to reference data and to baseline data.	Low action response to add water quality monitoring stations to each of these tributaries to assist in determination of effects to biota in the future.
Sheardown Lake Northwest and Southeast basins	Arsenic concentration in sediment greater than AEMP benchmark. Chromium concentration in sediment greater than AEMP benchmark. Iron concentration in sediment greater than AEMP benchmark. Manganese concentration in sediment greater than AEMP benchmark. Nickel concentration in sediment greater than AEMP benchmark.	No AEMP water quality benchmarks were exceeded at Sheardown Lake in 2020. For all parameters, no change in concentration in sediment was shown compared to background and/or baseline, indicating the change was not mine-related.	Low action response to examine the relevance of site-specific sediment quality AEMP benchmarks for Sheardown Lake SE and, if necessary, establish new AEMP benchmarks taking into consideration data from the reference lake and applicable sediment quality guidelines.
Mary River	Aluminum concentration in water greater than AEMP benchmark in summer. Copper concentration in water greater than AEMP benchmark in summer. Iron concentration in water greater than AEMP benchmark in summer. Lead concentration in water greater than AEMP benchmark in summer.	Concentrations of metals in water of Mary River during the summer occurred as a result of high turbidity in 2020, and were comparable to background and/or baseline indicating that the elevated concentrations in 2020 were not mine-related. No adverse effects on phytoplankton, benthic invertebrates, or fish were indicated at Mary River compared to reference data or to baseline conditions.	No changes recommended to monitoring program for Mary River due to exceedances of AEMP benchmarks.
Mary Lake	Manganese concentration in sediment greater than AEMP benchmark at a single profundal station.	No AEMP water quality benchmarks were exceeded at Mary Lake in 2020. Isolated occurrence of this exceedance, and the fact that average concentrations of manganese in sediment were comparable to background and baseline, indicated that change was not mine-related. No adverse effects on phytoplankton, benthic invertebrates, or fish compared to reference data and to baseline conditions.	No changes recommended to monitoring program for Mary Lake due to exceedance of AEMP benchmark.

conditions. Applying the Mary River Project AEMP Management Response Framework, low action responses including implementation of an expanded spatial water quality sampling program to identify the source(s) of copper to the CLT1 north branch, and establishment of benthic invertebrate community sampling stations to evaluate possible mine-related effects on biota in the upper main stem portion of CLT1, are recommended.

At CLT2, water chemistry met all AEMP benchmarks, sediment quality met all SQG, and no adverse effects on phytoplankton or benthic invertebrates were indicated relative to reference creek conditions and CLT2 baseline data in 2020. Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no adjustments to the existing AEMP are recommended.

At Camp Lake, no AEMP water quality benchmarks were exceeded, but arsenic concentrations in sediment at a single littoral station that were above the AEMP sediment quality benchmark possibly indicated a mine-related change in 2020 relative to background and/or baseline conditions (Table 6.1). No adverse effects on phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at Camp Lake in 2020 based on comparisons to reference lake conditions and to Camp Lake baseline data. No identifiable mine-related sources of arsenic to the Camp Lake system were evident, and the current AEMP does not adequately capture variability in sediment chemistry at littoral habitat of Camp Lake. Considering arsenic concentrations in sediment, sources of arsenic to the system, and the current AEMP design, a low action response is recommended at Camp Lake under the AEMP Management Response Framework. To this end, harmonizing lake sediment quality and benthic invertebrate community monitoring stations, focusing on littoral habitat, is recommended to improve the ability of the program to evaluate mine-related effects to sediment quality at littoral areas and to potentially allow linkages to be determined between metal concentrations in sediment and benthic invertebrate community responses in the future.

6.3 Sheardown Lake System

Within the Sheardown Lake system, AEMP monitoring is conducted at Sheardown Lake Tributaries 1, 12, and 9 (SDLT1, SDLT12, and SDLT9, respectively), Sheardown Lake NW (DL0-01) and Sheardown Lake SE (DL0-02). At the Sheardown Lake tributaries, AEMP water quality benchmarks were exceeded in 2020 for copper at SDLT1 (Table 6.1), but because no elevation in copper concentrations was indicated compared to baseline conditions, copper concentrations naturally appeared to be near the AEMP benchmark at this tributary. No adverse effects to phytoplankton or benthic invertebrates were indicated at SDLT1 or at either SDLT12 or SDLT9 in 2020 based on comparison to reference creek concentrations and respective Sheardown Lake Tributary baseline data. Because no adverse effects to biota were associated with copper concentrations above the AEMP benchmark at SDLT1, a low action response to identify the likely source(s) of copper to the system is recommended to meet obligations under the AEMP Management Response Framework. Although no mine-related changes to phytoplankton or benthic invertebrates were indicated at SDLT12 and SDLT9 in 2020, a low action response to add a water quality monitoring station at each of these two tributaries under the AEMP is recommended to improve the ability of the program to interpret biological data in the future.

At Sheardown Lake NW, no AEMP benchmarks for water quality were exceeded in 2020. Lake-specific AEMP benchmarks for sediment quality were exceeded for arsenic, iron, manganese, and nickel in 2020, but none of these metals were elevated in the sediment of Sheardown Lake NW compared to the reference lake and to concentrations at Sheardown Lake NW during baseline (Table 6.1). No adverse effects to phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at Sheardown Lake NW in 2020 based on comparisons to reference conditions and to Sheardown Lake NW baseline conditions. Because no mine-related changes in metal concentrations occurred in sediment at Sheardown Lake NW in 2020, and no adverse effects to biota were associated with concentrations of metals above AEMP sediment quality benchmarks, a low action response is recommended to meet obligations under the AEMP Management Response Framework. Specifically, it is recommended that, because concentrations of metals in Sheardown Lake NW sediment have been similar to those shown at the reference lake, consideration should be given to updating the AEMP sediment quality benchmarks for Sheardown Lake NW to reflect not only baseline data, but also reference lake data.

At Sheardown Lake SE, no AEMP benchmarks for water quality were exceeded in 2020. Lake-specific AEMP benchmarks for sediment quality were exceeded for chromium, iron, manganese, and nickel concentrations at Sheardown Lake SE in 2020 (Table 6.1). However, none of these metals occurred at concentrations in sediment of Sheardown Lake SE that were elevated compared to the reference lake, or to concentrations shown at Sheardown Lake SE during the baseline period. In addition, concentrations of these metals were above the Sheardown Lake SE AEMP benchmarks in sediment at the reference lake, suggesting naturally high concentrations of each of the indicated metals in sediments of area lakes. No adverse effects to phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at Sheardown Lake SE in 2020 based on comparisons to reference conditions and to applicable Sheardown Lake SE baseline conditions. Because no mine-related changes in metal concentrations occurred in sediment at Sheardown Lake SE in 2020 and no adverse effects to biota were associated with concentrations of metals above AEMP benchmarks for sediment quality, a low action response is recommended to meet obligations under the AEMP Management Response Framework. Specifically, it is recommended that the relevance of site-specific sediment quality AEMP benchmarks for Sheardown Lake SE be assessed and, if necessary, determined anew taking into consideration data from the reference lake and applicable sediment quality guidelines.

6.4 Mary River and Mary Lake Systems

Within the Mary River and Mary Lake systems, AEMP monitoring is conducted at Mary River Tributary-F (MRTF), Mary River, and Mary Lake (BL0). At MRTF, no AEMP benchmarks for water quality were exceeded in 2020, and for parameters with established AEMP benchmarks, no changes in concentrations were shown relative to baseline. No adverse effects on phytoplankton were indicated at MRTF in 2020. Biological sampling conducted at MRTF to meet MDMER obligations suggested some differences in benthic invertebrate community assemblages between effluent-exposed and reference areas, but these differences did not appear to be related to metal concentrations originating from mine effluent and/or mine operations (Minnow 2021). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and to baseline, and no adverse biological effects related to metals were indicated in 2020, no changes to the existing sampling program at MRTF are recommended.

At Mary River, concentrations of aluminum, copper, iron, and lead were above respective AEMP benchmarks at stations located adjacent to the mine (i.e., E0 series stations) and, with the exception of lead, downstream of the mine (C0 series) in 2020 (Table 6.1). However, the concentrations for each of these parameters were similar or higher, and above applicable AEMP benchmarks, at the Mary River reference stations (G0-09 series) and/or upstream stations (G0 series), reflecting highly turbid sampling conditions that occurred in 2020. No mine-related changes to parameter concentrations were indicated at Mary River mine-exposed stations in 2020 compared to the reference stations and to Mary River baseline data. In addition, metal concentrations in sediment were well below SQG, and no adverse effects on phytoplankton, benthic invertebrates, and fish (arctic charr) health were indicated at all Mary River mine-exposed areas in 2020. Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no changes to AEMP monitoring at Mary River are recommended as per the AEMP Management Response Framework.

At Mary Lake, no AEMP benchmarks for water quality were exceeded in 2020. Lake-specific AEMP benchmarks for sediment quality were exceeded for manganese concentrations at a single profundal station in 2020 (Table 6.1). The isolated occurrence of this exceedance, and the fact that average manganese concentrations in sediment at Mary Lake were not elevated compared to concentrations at the reference lake or to those at Mary Lake during baseline, indicated no mine-related change in manganese concentrations at Mary Lake since commercial mine operations commenced in 2015. No adverse effects on phytoplankton, benthic invertebrates, nor on fish (arctic charr) health were indicated at Mary Lake in 2020 based on comparisons to reference lake conditions and to Mary Lake baseline data. Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2020, no changes to AEMP monitoring at Mary Lake are recommended as per the AEMP Management Response Framework.

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