APPENDIX E.9

Aquatic Effects Monitoring Reports

APPENDIX E.9.1

2021 CREMP Monitoring Report

Mary River Project 2021 Core Receiving Environment Monitoring Program Report

Part 1 of 2
(Sections 1 to 7)

Prepared for:
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Mary River Project 2021
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. 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 2021 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, only the concentration of copper was elevated above site-specific AEMP water quality benchmarks at the north branch of Camp Lake Tributary 1 (CLT1) in 2021, but copper concentrations at this portion of the tributary were not elevated compared to those during baseline. Special investigation into the source of copper to the CLT1 north branch through spatially expanded sampling indicated a natural source (e.g., bedrock and/or overburden) of copper to this system. Because this elevation in copper concentrations did not appear to be mine-related, and no adverse biological effects were indicated at CLT1, no adjustment to the existing AEMP is required. 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. However, the spatial extent was limited and no biological effects were observed, and therefore continued monitoring of the benthic invertebrate community at CLT1 upper main stem was recommended in 2022 (and future CREMP studies) to assess and track changes in potential effects to biota in the CLT1 upper main stem over time. At Camp Lake Tributary 2 (CLT2), no changes in concentrations of parameters with AEMP benchmarks occurred relative to background or to baseline and no adverse biological effects were indicated in 2021, and thus no adjustments to the existing AEMP are recommended. At Camp Lake, arsenic, copper, iron, manganese, nickel, and phosphorus concentrations were above AEMP sediment quality benchmarks at individual stations, however average concentrations of these metals were below respective benchmarks and were comparable to background and/or baseline concentrations. Because no changes in concentrations of parameters with AEMP sediment quality benchmarks occurred relative to background and baseline, and no adverse biological effects were indicated at Camp Lake in 2021, no adjustments to the existing AEMP were required. However, harmonizing sediment quality and benthic invertebrate community monitoring stations for littoral habitat of Camp Lake is recommended to improve the ability of the CREMP to link sediment quality with biological responses to changes in, for instance, metal concentrations.

Within the Sheardown Lake system, copper concentrations were elevated above site-specific AEMP water quality benchmarks at Sheardown Lake Tributary 1 (SDLT1) in 2021, but copper concentrations were comparable to baseline and no adverse biological effects were indicated at SDLT1. Special investigation involving spatially expanded sampling did not indicated any distinct source of copper to SDLT1, suggesting a naturally occurring source of copper to the system. Because elevated copper concentrations within SDLT1 were determined not to be mine-related, no further management actions are required. Water quality was sampled at Sheardown Lake Tributary 12 (SDLT12) for the first time in fall 2021 and in this single sample, the AEMP water quality benchmarks for total ammonia and nitrate were exceeded.

Because water quality monitoring at SDLT12 was newly added and only a single water quality data point existed, and considering that no adverse effects to biota were indicated at SDLT12 in 2021, a low action response to continue collecting water quality data at SDLT12 is recommended. At Sheardown Lake Tributary 9 (SDLT9), water chemistry met all AEMP benchmarks and no adverse biological effects were indicated in 2021, and thus on-going water quality monitoring is recommended for 2022 and beyond to track potential changes in environmental quality at this tributary over time . At the Sheardown Lake northwest (NW) and southeast (SE) basins, water quality consistently met AEMP benchmarks and, despite arsenic, copper, 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. No adverse effects to phytoplankton, benthic invertebrates, or arctic charr were indicated at Sheardown Lake in 2021 based on evaluation relative to reference conditions and baseline data. 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. The AEMP sediment quality benchmarks for arsenic, chromium, manganese, and nickel concentrations were exceeded at a single station for each of these parameters at Mary Lake in 2021. However, based on the isolated occurrence of these exceedances and the fact that average concentrations of these metals in sediment at Mary Lake were not elevated compared to concentrations at the reference lake or to those during baseline, elevation in concentrations of these metals in sediment of Mary Lake was not mine-related. Because no changes in concentrations of parameters with AEMP benchmarks occurred relative to background and baseline, and no adverse effects to phytoplankton, benthic invertebrates, or arctic charr were indicated in 2021, 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

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

ECCC – Environment and Climate Change Canada

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

MRTF – Mary River Tributary-F

NAD 83 – 1983 North American Datum

NSES – North Shore Environmental Services

NU – Nunavut

NW – Northwest

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

SE – Southeast

SEL – Severe Effect Levels

SQG – Sediment Quality Guidelines

TDS – Total Dissolved Solids

TKN – Total Kjeldahl Nitrogen

TOC – Total Organic Carbon

TSS – Total Suspended Solids

UTM – Universal Transverse Mercator

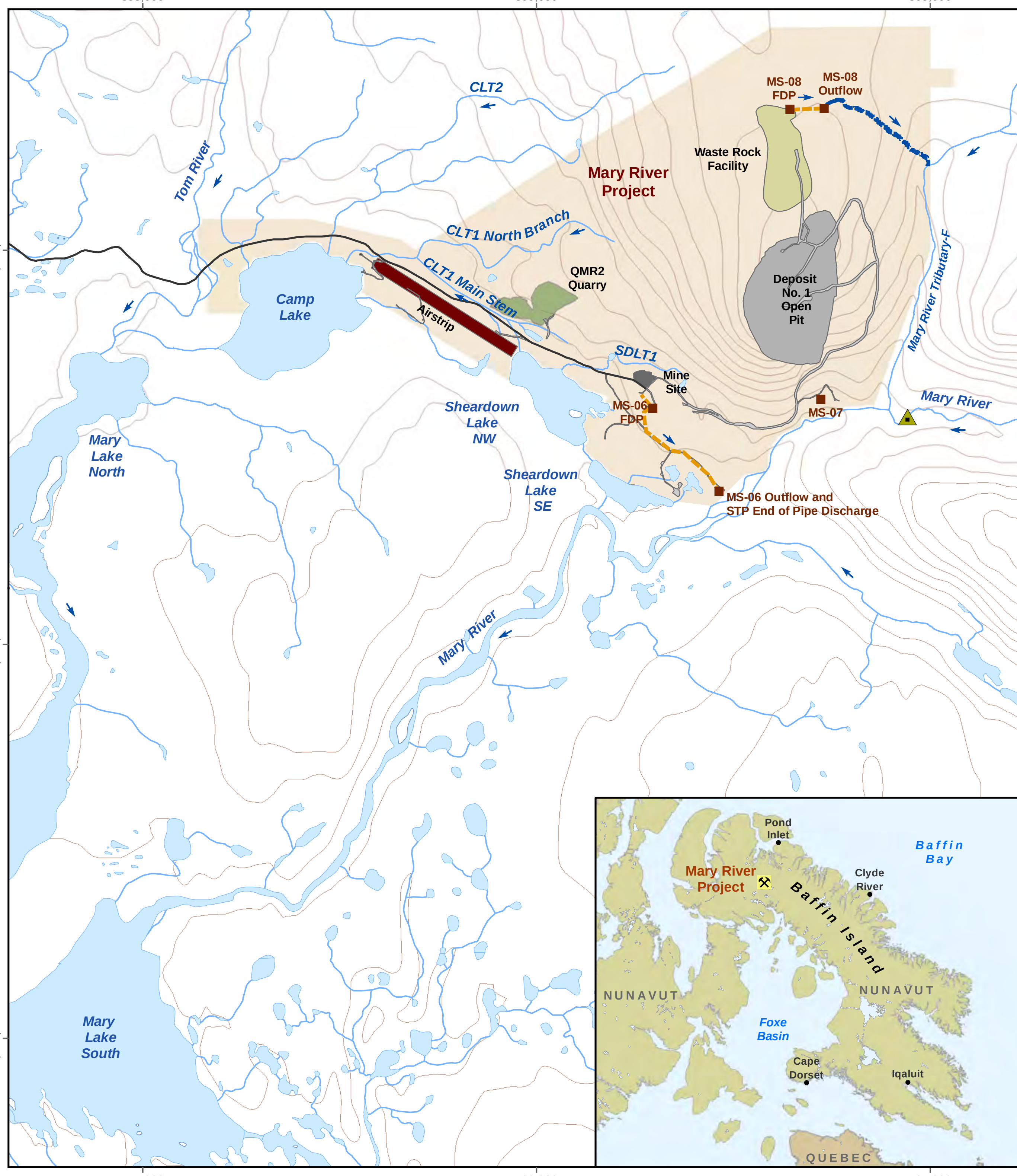
WQG – Water Quality Guidelines

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 Tote Road to Milne Port, located approximately 100 kilometres (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 2014a; 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 six years of mine operation, the CREMP studies have indicated only minimal effects of Project operations on 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



LEGEND

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

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 217202.0075 - AEMP 2021\D - GIS\AEMP 2021\21-75 Figure 1.1 Location.mxd each respective lake (Minnow 2016a, 2017, 2018, 2019, 2020, 2021b). 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 2020 based on comparisons to reference waterbodies and to available pre-mine baseline data for each lake system (Minnow 2016a, 2017, 2018, 2019, 2020, 2021b).

This report presents the methods and results of the 2021 CREMP, including an evaluation of potential Project-related influences on chemical and biological conditions at mine-exposed waterbodies through the seventh full year of mine operation. As in the six previous years, the 2021 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 2021 CREMP was implemented in accordance with AEMP Revision 1 (Baffinland 2015), except for the addition of 11 water quality monitoring stations and two benthic invertebrate community study areas. In 2021, water quality monitoring stations were added at the Camp Lake Tributary 1 north branch and Sheardown Lake tributaries SDLT1, SDLT12, and SDLT9 in 2021 to investigate elevated aqueous total copper concentrations (Minnow 2021b). Also in 2021, a benthic invertebrate community sampling area was added at the CLT1 upper main stem to evaluate possible effects of elevated aqueous total aluminum and total iron concentrations on biota in this portion of the CLT1 system. Since 2016, a reference creek benthic invertebrate community study area has been added to the program to provide improved ability for the evaluation of mine-related influences on stream biota (Minnow 2016b, 2017, 2018, 2019, 2020, 2021b).

2 METHODS

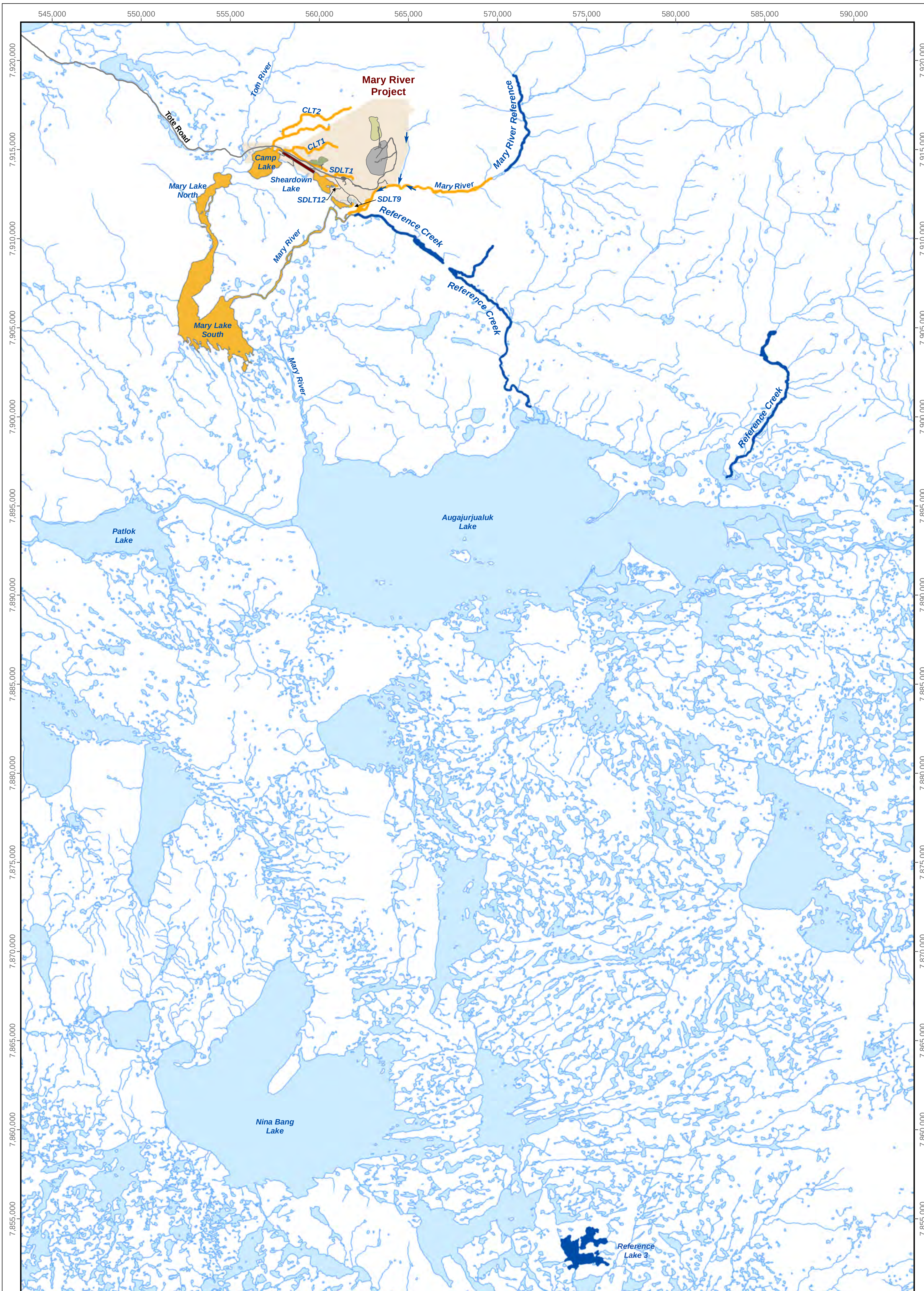
2.1 Overview

The CREMP includes water quality monitoring, sediment quality monitoring, phytoplankton (chlorophyll-a) monitoring, benthic invertebrate community assessment, and fish population assessment (Baffinland 2015). In 2021, water quality and phytoplankton monitoring were conducted by Baffinland environment department personnel over four separate sampling events, including a lake ice-cover event (April 16th to 23rd) and open-water season events corresponding to Arctic spring (freshet), summer, and fall (June 29th to July 2nd, July 22nd to August 1st, and August 26th to September 5th, respectively). Sediment quality, benthic invertebrate community, and fish population sampling was conducted by Minnow Environmental Inc. (Minnow) personnel with assistance from Baffinland environment department personnel from August 7th to 19th 2021, the seasonal timing of which was consistent with monitoring conducted for previous baseline (2005 to 2013), mine construction (2014), and mine operational (2015 to 2020) studies. Similar to previous CREMP studies, the 2021 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 2021 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 (i.e., lake) environments beginning in the 2015 CREMP study, was again used as the reference lake for the 2021 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 Lake
- Reference Stream/River System
- Mine-exposed Stream/River System
- Mine-exposed Lake
- Mary River Project
- Road
- Waste Rock Facility
- Ultimate Deposit No. 1 Pit Limits
- Mine Site Complex Pad
- Airstrip
- QMR2 Quarry

Mary River Project CREMP Study Waterbodies

0 3.25 6.5 13
km

N W E S

Map Projection: UTM Zone 17N NAD 1983
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Figure 2.1

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 217202.0075 - AEMP 2021\D - GIS\AEMP 2021\21-75 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 2021 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 68 stations during each sampling event (Table 2.1 and 2.2; Figures 2.2 and 2.3) and included collection of *in situ* measurements and water chemistry data. Of these 68 stations, 57 are part of the core CREMP design, two were added to the core CREMP design in fall 2021[1], and 9 have been included in the 2021 CREMP as part of special investigations[2] (Section 2.5.2). 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

2.2.2.1 Sample Collection and Laboratory 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

[1] Water quality and phytoplankton monitoring stations were added in fall 2021 to Sheardown Tributary 12 (station LDFG-OUT) and Sheardown Tributary 9 (station MS-C-G). These stations were added as per recommendations made in the Mary River project 2020 CREMP (Minnow 2021b) to provide supporting information for benthic invertebrate community data analysis (see Section 2.5.2).

[2] In 2020, water quality data exceeded the AEMP benchmarks in several instances and resulted in low action responses under the Response Framework at the CLT1 north branch and at the SDLT1 (Minnow 2021b; see Section 2.5.2). Special investigations were conducted in 2021 to follow up on these aqueous concentrations that exceeded AEMP benchmarks. In fall 2021 three new water quality monitoring stations (L1-CUSI-T1, L1-CUSI-T2, and L1-CUSI-A) were sampled to support the investigation of water quality of Camp Lake Tributary 1 (CLT1) north branch. Also in fall 2021, six new water quality monitoring stations (D1-CUSI-A, D1-CUSI-B, D1-CUSI-C, MS-C-A, MS-C-B, and MS-C-F) were sampled to support the investigation of water quality of Sheardown Lake Tributary 1 (SDLT1)

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	J0-01	555701	7913773	a	-	✓	✓	✓
		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 Tributary 12	LDFG-OUT [b]	561021	7912967	a	-	- [c]	- [c]	✓
	Sheardown Tributary 9	MS-C-G [b]	561813	7911830	a	-	- [c]	- [c]	✓
	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		-	✓	✓	✓
	Tom River	I0-01	555470	7914139	a	-	✓	✓	✓
	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.

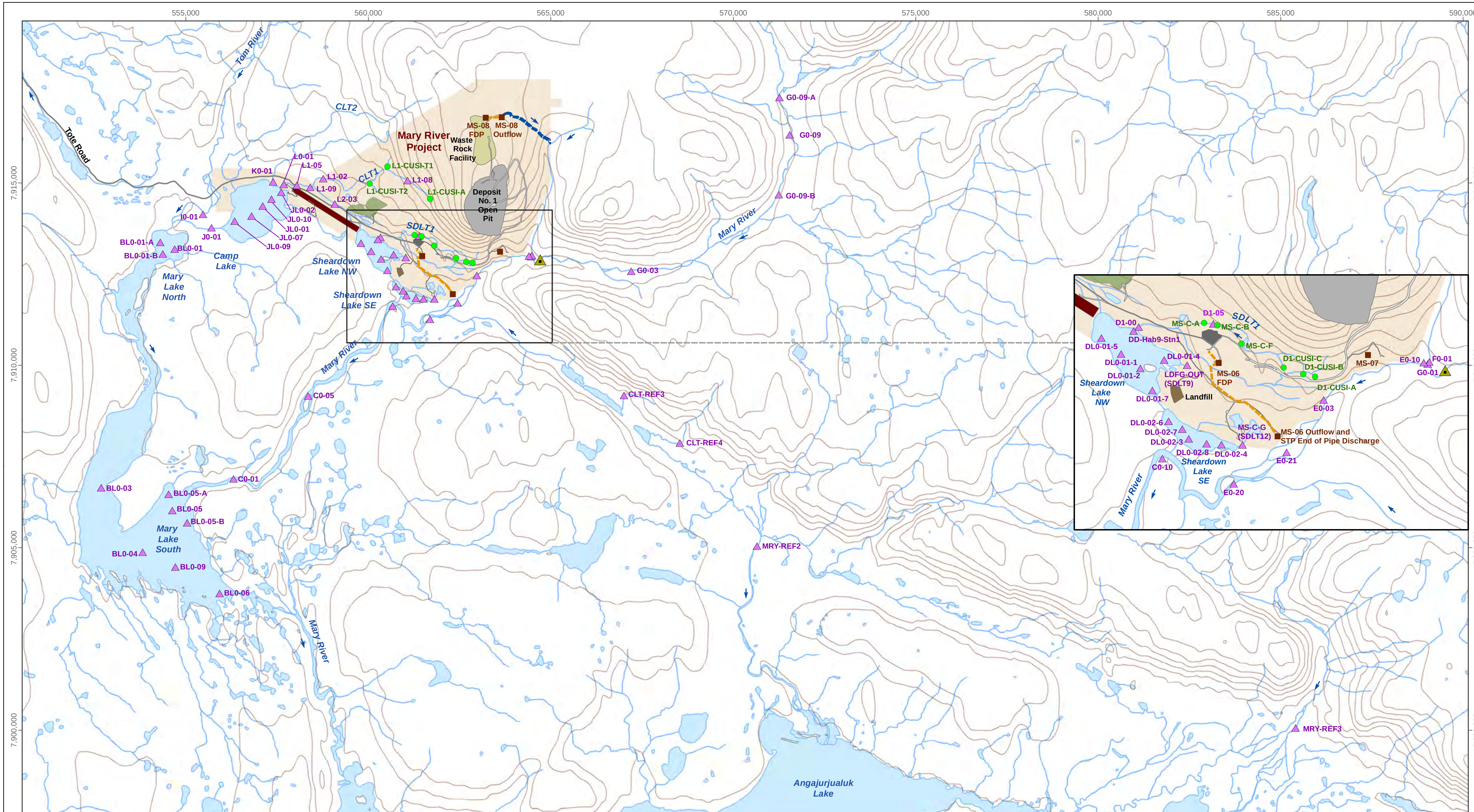
[b] Water quality and phytoplankton monitoring stations were added i n fall 2021 to Sheardown Tributary 12 (station LDFG-OUT) and Sheardown Tributary 9 (station MS-C-G). These stations were added as per recommendations made in the Mary River project 2020 CREMP (Minnow 2021b) to provide supporting information for benthic invertebrate community data analysis.

[c] Stations LDFG-OUT and MS-C-G were added to the CREMP in fall 2021, and therefore were not sampled in earlier seasons. There stations will be sampled in spring, summer, and fall in future years.

Table 2.2: Mary River Project CREMP Water Quality Monitoring Station Coordinates for Special Investigations, Sampled Fall 2021

Study System	Water Body	Station ID	UTM Zone 17N, NAD83	
			Easting	Northing
Camp Lake System	Camp Lake Tributaries	L1-CUSI-T1	560520	7915449
		L1-CUSI-T2	560035	7914977
		L1-CUSI-A	561693	7914565
Sheardown Lake System	Sheardown Tributary 1	D1-CUSI-A	562852	7912804
		D1-CUSI-B	562683	7912840
		D1-CUSI-C	562401	7912932
		MS-C-A	561263	7913571
		MS-C-B	561454	7913537
		MS-C-F	561797	7913278

Note: Locations sampled in fall 2021 based on recommendations made in the Mary River Project 2020 CREMP (Minnow 2021b; Section 2.5.2) to investigate water quality results that exceeded AEMP benchmarks.



LEGEND

- Water Quality Monitoring Station for 2021 Special Investigation
- Water Quality and Phytoplankton Monitoring Station
- Mary River Cascade Barrier
- Final Discharge Point (FDP)
- Mary River Project
- Landfill
- Ultimate Deposit No. 1 Pit Limits
- Mine Site Complex Pad
- Airstrip
- QMR2 Quarry
- Waste Rock Facility
- Road
- Overland Effluent Channel
- Contours (20 m)
- Discharge Line

0 1.5 3 6
km

N
W E
S

Map Projection: UTM Zone 17N NAD 1983
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Mary River Project CREMP Routine Water Quality and Phytoplankton Monitoring Station Locations

Date: March 2022
Project 217202.0075

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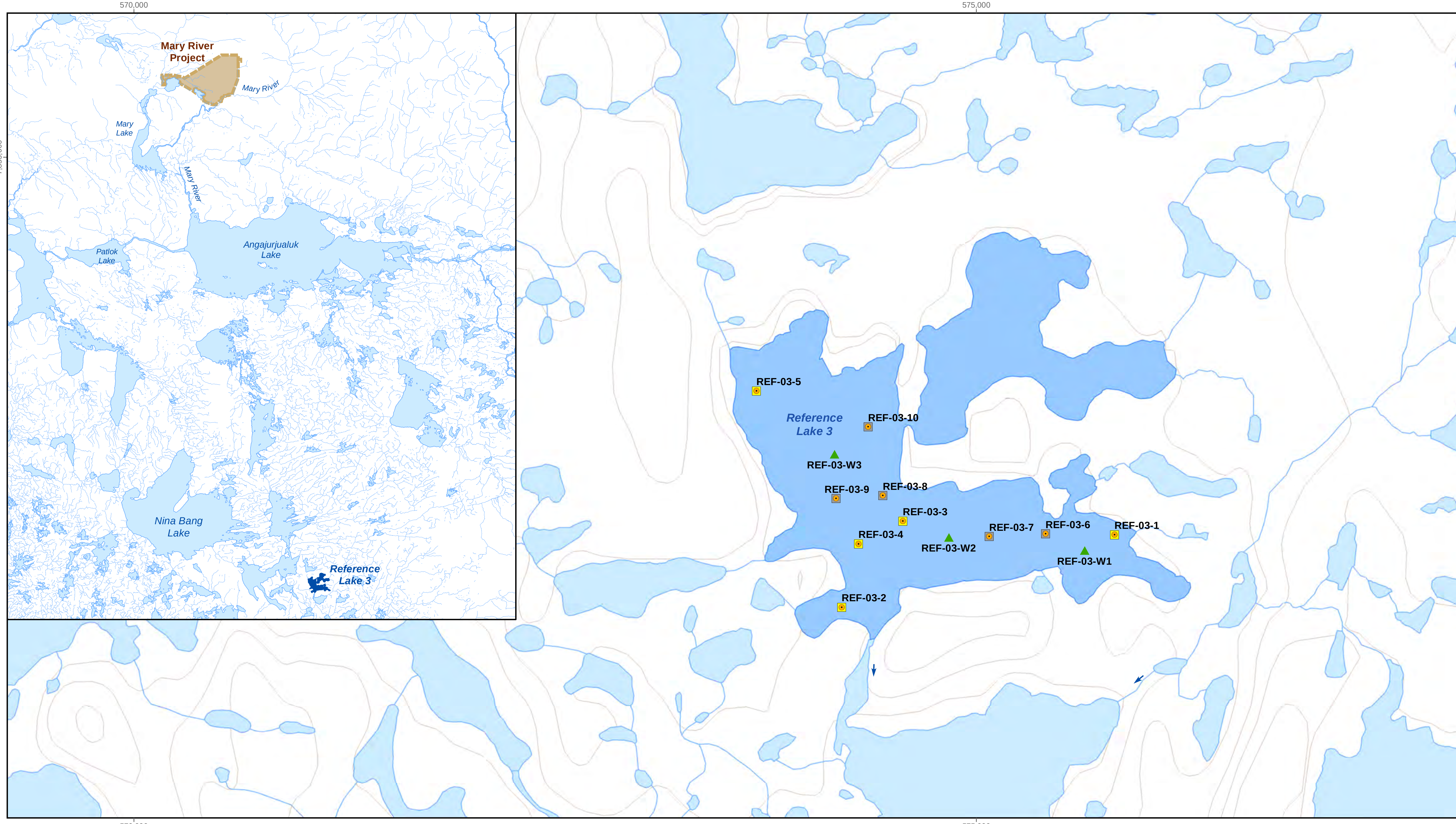
Figure 2.2

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 217202.0075 - AEMP 2021\D - GIS\AEMP 2021\21-75 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 0.25 0.5 1
Kilometers

N W E S

Map Projection: UTM Zone 17N NAD 1983
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Mary River Project CREMP Reference Lake 3 Monitoring Station Locations

Date: February 2022
Project 217202.0075

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Figure 2.3

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 217202.0075 - AEMP 2021\D - GIS\AEMP 2021\21-75 Figure 2.3 Reference Lake 3.mxd water-sediment interface) at all lake benthic invertebrate community (benthic) stations during biological sampling conducted in August by Minnow 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 *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).

2.2.2.2 Data Analysis

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[3]; 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) and between littoral and profundal habitats 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

[3] Canadian Environmental Water Quality Guidelines (CCME 1999, 2021) were used as the primary source for WQG, including those for pH and dissolved oxygen concentrations.

distributed data were subject to Tukey's Honestly Significant Difference (HSD) and non-parametric post-hoc test were compared using Dunn's Kruskal-Wallis Multiple Comparisons test (Dunn 1964). 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[4]. 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 evaluated to provide a better understanding of natural conditions and/or mine-related influences on within-lake water quality.

2.2.3 Water Chemistry

2.2.3.1 Sample Collection and Laboratory Analysis

Surface water chemistry samples were collected from both lotic and lentic environments (Table 2.1 and 2.2). 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).

[4] 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.

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.

2.2.3.2 Data Analysis

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 and 2.2); ii) spatially and seasonally at each mine-exposed waterbody; iii) to applicable WQG for the protection of aquatic life (Table 2.3) 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 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 2021 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 2.4.

Applicable WQG included the Canadian Water Quality Guidelines (CWQG; CCME 1999, 2021) 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 2.3). 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

Table 2.3: Water Quality Guidelines Used for the Mary River Project 2015 to 2021 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	-

Note: "-" indicates not applicable.

[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.

Table 2.4: 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.

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 2021) years.

2.3 Sediment Quality

2.3.1 General Design

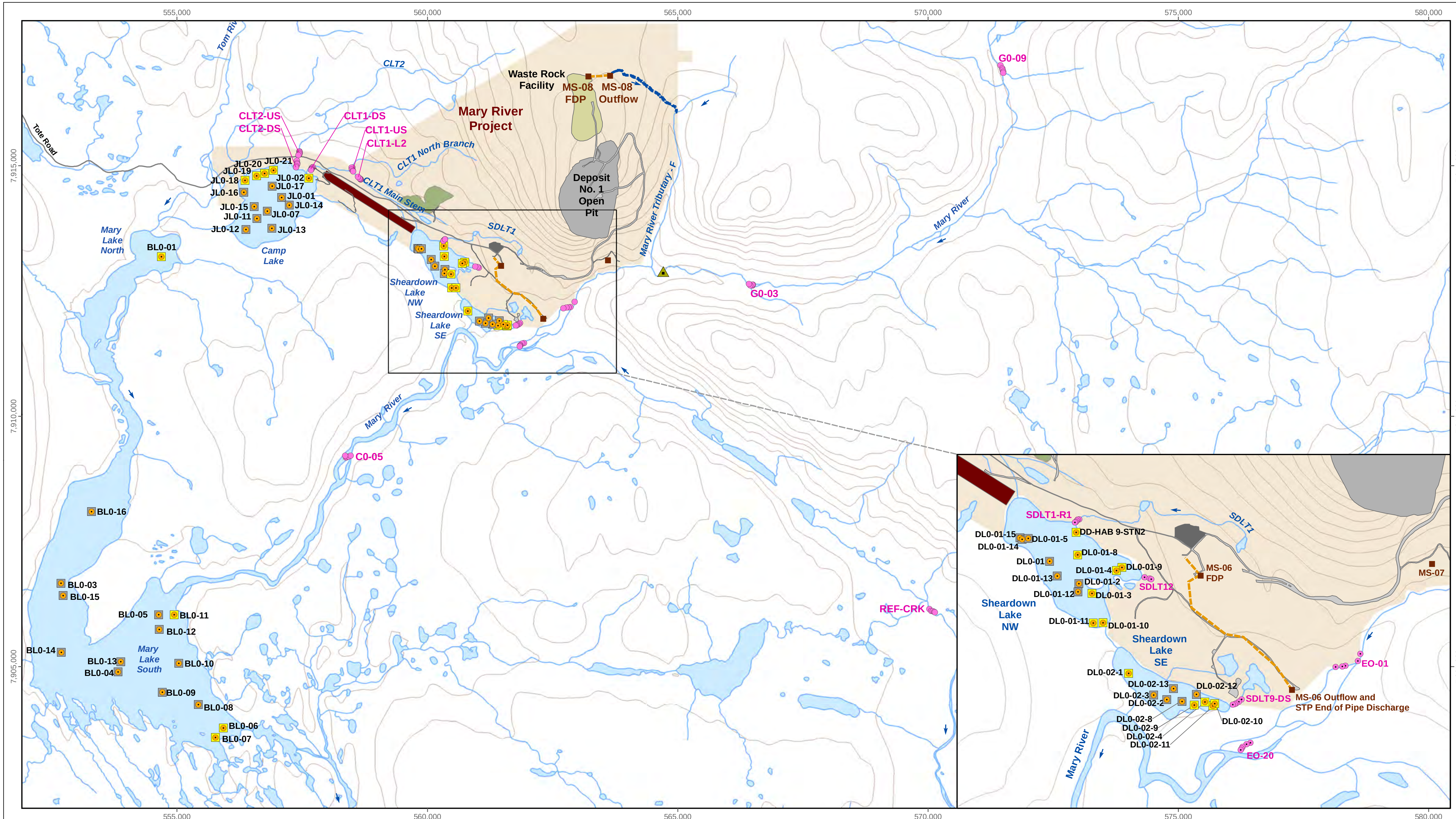
Sediment quality monitoring for the CREMP was designed to assess potential mine-related effects to the sediment of lake environments using a gradient-based approach (Baffinland 2015). Sediment quality sampling was conducted at five to ten stations per study lake for physical and chemical characterization 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 2.5; 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 of lotic (stream and river) habitats is conducted once every three years under the CREMP,[5] and because sediment quality sampling of lotic habitat was last conducted in 2020, no sediment was collected at stream and river habitats in 2021. As with water quality, the

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

Table 2.5: Lake Sediment Quality and Benthic Invertebrate Community Monitoring Station Coordinates Used for the Mary River Project 2021 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	575820	7852759	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	JL0-02	557629	7914751	littoral	✓	-	✓
	JL0-01	557091	7914371	profundal	✓	-	✓
	JL0-14	557244	7914215	profundal	✓	-	-
	JL0-17	556900	7914595	profundal	✓	-	-
	JL0-21	556926	7914912	littoral	-	✓	✓
	JL0-20	556750	7914850	littoral	-	✓	✓
	JL0-19	556587	7914802	littoral	-	✓	✓
	JL0-07	556803	7914096	profundal	✓	-	✓
	JL0-18	556356	7914707	littoral	-	✓	✓
	JL0-16	556335	7914471	profundal	✓	-	✓
	JL0-15	556542	7914185	profundal	✓	-	-
	JL0-11	556594	7913947	profundal	✓	-	✓
	JL0-13	556896	7913752	profundal	✓	-	-
	JL0-12	556378	7913729	profundal	✓	-	✓
Sheardown Lake Northwest (NW)	DL0-01-5	559805	7913349	profundal	✓	-	✓
	DL0-01-14	559824	7913337	profundal	-	✓	✓
	DL0-01-15	559882	7913344	profundal	-	✓	✓
	DD-HAB 9-STN2	560324	7913401	littoral	✓	-	-
	DL0-01-8	560338	7913193	littoral	✓	-	-
	DL0-01	560078	7913132	profundal	✓	-	-
	DL0-01-13	560150	7912998	profundal	✓	-	-
	DL0-01-2	560350	7912928	profundal	✓	-	✓
	DL0-01-12	560339	7912852	profundal	-	✓	✓
	DL0-01-9	560745	7913077	littoral	✓	-	✓
	DL0-01-4	560696	7913049	littoral	-	✓	✓
	DL0-01-3	560471	7912838	littoral	-	✓	✓
	DL0-01-11	560482	7912563	littoral	-	✓	✓
	DL0-01-10	560570	7912567	littoral	✓	-	✓
Sheardown Lake Southeast (SE)	DL0-02-1	560808	7912101	littoral	✓	-	✓
	DL0-02-11	561585	7911800	littoral	✓	-	✓
	DL0-02-10	561602	7911822	littoral	-	✓	✓
	DL0-02-4	561512	7911834	littoral	✓	-	✓
	DL0-02-12	561433	7911906	profundal	-	✓	✓
	DL0-02-9	561413	7911807	littoral	-	✓	✓
	DL0-02-8	561300	7911840	profundal	-	✓	✓
	DL0-02-13	561222	7911958	profundal	-	✓	✓
	DL0-02-2	561161	7911859	profundal	✓	-	✓
	DL0-02-3	561038	7911899	profundal	✓	-	✓
Mary Lake	BL0-01	554690	7913187	littoral	✓	-	✓
	BL0-16	553289	7908093	profundal	✓	-	-
	BL0-03	552679	7906661	profundal	✓	-	✓
	BL0-15	552723	7906419	profundal	-	✓	✓
	BL0-14	552688	7905283	profundal	✓	-	✓
	BL0-05	554635	7906033	profundal	-	✓	✓
	BL0-11	554942	7906033	littoral	-	✓	✓
	BL0-12	554643	7905743	profundal	✓	-	-
	BL0-13	553879	7905094	profundal	-	✓	✓
	BL0-04	553820	7904894	profundal	✓	-	✓
	BL0-10	555033	7905066	profundal	✓	-	-
	BL0-09	554707	7904487	profundal	✓	-	-
	BL0-08	555424	7904240	profundal	✓	-	-
	BL0-07	555767	7903583	littoral	-	✓	✓
	BL0-06	555925	7903772	littoral	✓	-	✓

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



LEGEND

- Sediment and Benthic Monitoring Location
- Stream - Sediment and Benthic Sampling Location
- Littoral Sampling Depth
- Profundal Sampling Depth
- Mary River Cascade Barrier
- Final Discharge Point (FDP)
- Discharge Line
- Overland Effluent Channel
- Contours (20 m)
- Ultimate Deposit No. 1 Pit Limits
- Mine Site Complex Pad
- Airstrip
- Waste Rock Facility
- QMR2 Quarry
- Mary River Project
- Road

0 1 2 4
km

N W E S

Map Projection: UTM Zone 17N NAD 1983
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Mary River Project 2021 CREMP Mine Area Sediment Quality and Benthic Station Locations

Date: March 2022
Project 217202.0075

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Figure 2.4

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 217202.0075 - AEMP 2021\D - GIS\AEMP 2021\21-75 Figure 2.4 Sediment and Benthic.mxd 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 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 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 2.2.2.2).

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, 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 2.2.3.2; Table 2.4). Sediment chemistry data collected at lake environments were compared to applicable Canadian Sediment Quality Guidelines (CSQG; CCME 1999, 2021) 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 2021 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 2021 relative to baseline (2005 to 2013) and earlier in the period of mine operation (2015 to 2020). In addition, as described previously, enrichment factors were calculated between the 2021 and baseline data for each individual study lake using the same calculation (and categorization description) as described previously (Section 2.2.3.2; Table 2.4).

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 2.2.3.1). 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 2.2.3.2).

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 2020). In 2021, 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 2.2.2.2). An AEMP benchmark chlorophyll-a concentration of 3.7 µg/L was established for the Mary River Project (Baffinland 2015). The 2021 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 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 2.6; Figure 2.4)[6]. 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 2021 CREMP to augment the original study design (Table 2.6; 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). Once every three years, benthic invertebrate community data is collected under the Environmental Effects Monitoring (EEM) program 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. The EEM sampling was last conducted in 2020 (Minnow 2021a), therefore benthic invertebrate community samples were not collected at these areas in 2021. 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 2020, the level of replication used for lotic benthic sampling in 2021 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 2021 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 2.5; Figures 2.3 and 2.4).

[6] In 2021, benthic invertebrate community sampling area CLT1-L2 was included in the Camp lake Tributaries 1 assessment. Aqueous total aluminum and total iron AEMP benchmarks were exceeded at this location in past studies (Minnow 2021b), and therefore benthic invertebrate community monitoring was added at this location to evaluate possible effects on biota in this portion of the CLT1 system.

Table 2.6: Stream and River Benthic Invertebrate Community Monitoring Station Identifiers and Coordinates Used for the Mary River Project CREMP 2021 Study

Lake System	Waterbody	Station Code	Station Type	UTM Zone 17W, NAD83	
				Easting	Northing
Angajurjualuk Lake	Unnamed Tributary	REF-CRK-B1	Reference	570025	7906148
		REF-CRK-B2	Reference	570060	7906115
		REF-CRK-B3	Reference	570093	7906110
		REF-CRK-B4	Reference	570121	7906099
		REF-CRK-B5	Reference	570137	7906086
Camp Lake	Camp Lake Tributary 1	CLT1-US-B1	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-L2-B1	Mine-Exposed	558670	7914727
		CLT1-L2-B2	Mine-Exposed	558663	7914736
		CLT1-L2-B3	Mine-Exposed	558658	7914741
		CLT1-L2-B4	Mine-Exposed	558642	7914752
		CLT1-L2-B5	Mine-Exposed	558613	7914781
		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	557383	7914994
		CLT2-DS-B5	Mine-Exposed	557377	7914971
Sheardown Lake Northwest (NW)	Sheardown Lake Tributary 1 (Reach 1)	SDLT1-R1-B1	Mine-Exposed	560350	7913536
		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	560350	7913536
	Sheardown Lake Tributary 12	SDLT12-B1	Mine-Exposed	561026	7912969
		SDLT12-B2	Mine-Exposed	561002	7912976
		SDLT12-B3	Mine-Exposed	560953	7912989
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	G0-09-B1	Reference	571447	7917011
		G0-09-B2	Reference	571479	7916946
		G0-09-B3	Reference	571489	7916919
		G0-09-B4	Reference	571499	7916883
		G0-09-B5	Reference	571503	7916858
		G0-03-B1	Mine-Exposed	566491	7912606
		G0-03-B2	Mine-Exposed	566499	7912622
		G0-03-B3	Mine-Exposed	566489	7912627
		G0-03-B4	Mine-Exposed	566444	7912612
		G0-03-B5	Mine-Exposed	566425	7912643
		E0-01-B1	Mine-Exposed	562944	7912282
		E0-01-B2	Mine-Exposed	562855	7912177
		E0-01-B3	Mine-Exposed	562820	7912173
		E0-01-B4	Mine-Exposed	562778	7912165
		E0-01-B5	Mine-Exposed	562717	7912158
		E0-20-B1	Mine-Exposed	561930	7911460
		E0-20-B2	Mine-Exposed	561895	7911447
		E0-20-B3	Mine-Exposed	561858	7911420
		E0-20-B4	Mine-Exposed	561848	7911408
		E0-20-B5	Mine-Exposed	561841	7911393
		C0-05-B1	Mine-Exposed	558373	7909176
		C0-05-B2	Mine-Exposed	558387	7909184
		C0-05-B3	Mine-Exposed	558428	7909192
		C0-05-B4	Mine-Exposed	558464	7909209
		C0-05-B5	Mine-Exposed	558358	7909210

Analysis of benthic data collected at Reference Lake 3 from 2015 to 2020 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, 2021b). Analysis of benthic data collected from Reference Lake 3 in 2021 provided on-going confirmation of the occurrence of natural depth-related influences on benthic invertebrate community structure in area lakes (Appendix B). Because of the occurrence of natural depth-related differences in benthic invertebrate communities, the benthic stations at each mine-exposed and reference lake were categorized as littoral zone (2 to 12 m depth) or profundal zone (>12 m depth) stations based on station depth (Table 2.5). To the extent possible, five littoral and five profundal stations were designated for each study lake based on the previously established suite of CREMP lentic benthic stations[7] to provide temporal continuity with the baseline studies and the original CREMP design (Table 2.5; 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 2021 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 lotic mine-exposed and reference study area stations to minimize natural influences on community variability. Once all three sub-samples were collected at each respective station, all material gathered in the Surber sampler net was transferred to a plastic sampling jar 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

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

grab sampler (15.24 x 15.24 cm; 0.023 m^2 sampling area). A single sample, consisting of a composite of five grabs (i.e., 0.115 m^2 sampling area) was collected at each station with care taken to ensure that each grab was acceptable (i.e., that the grab captured sufficient surface material and was full to each edge). Any incomplete grabs were discarded. For each acceptable grab, the petite-Ponar was thoroughly rinsed and the material then field-sieved through 500-µm mesh. Following sieving of all five grabs, the retained material was carefully transferred into a plastic sampling jar which was labelled with both an external and internal station identifier.

Following collection, the benthic samples were preserved to a level of 10% buffered formalin in ambient water. Supporting measurements and information collected at each replicate grab location for lotic stations included sampling depth, water velocity, and description of aquatic vegetation/algae presence. In addition, *in situ* water quality at the bottom of the water column and 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 then examined under a stereomicroscope at a magnification of at least ten times by a technician. All benthic invertebrates were removed from the sample debris and placed into vials containing 70% ethanol according to major taxonomic groups (i.e., order or family levels). A senior taxonomist later enumerated and identified the benthic organisms to the lowest practical level (typically genus or species) utilizing up-to-date taxonomic keys. Quality assurance/quality control (QA/QC) conducted during the laboratory processing of benthic samples included organism recovery and sub-sampling checks on as many as 10% of the total samples collected for the 2021 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), and Simpson's Evenness Index. 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, were conducted using the same tests described for the *in situ* water quality comparisons (see Section 2.2.2.2). 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 the highest p-value from a Shapiro-wilks normality test was used. 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).

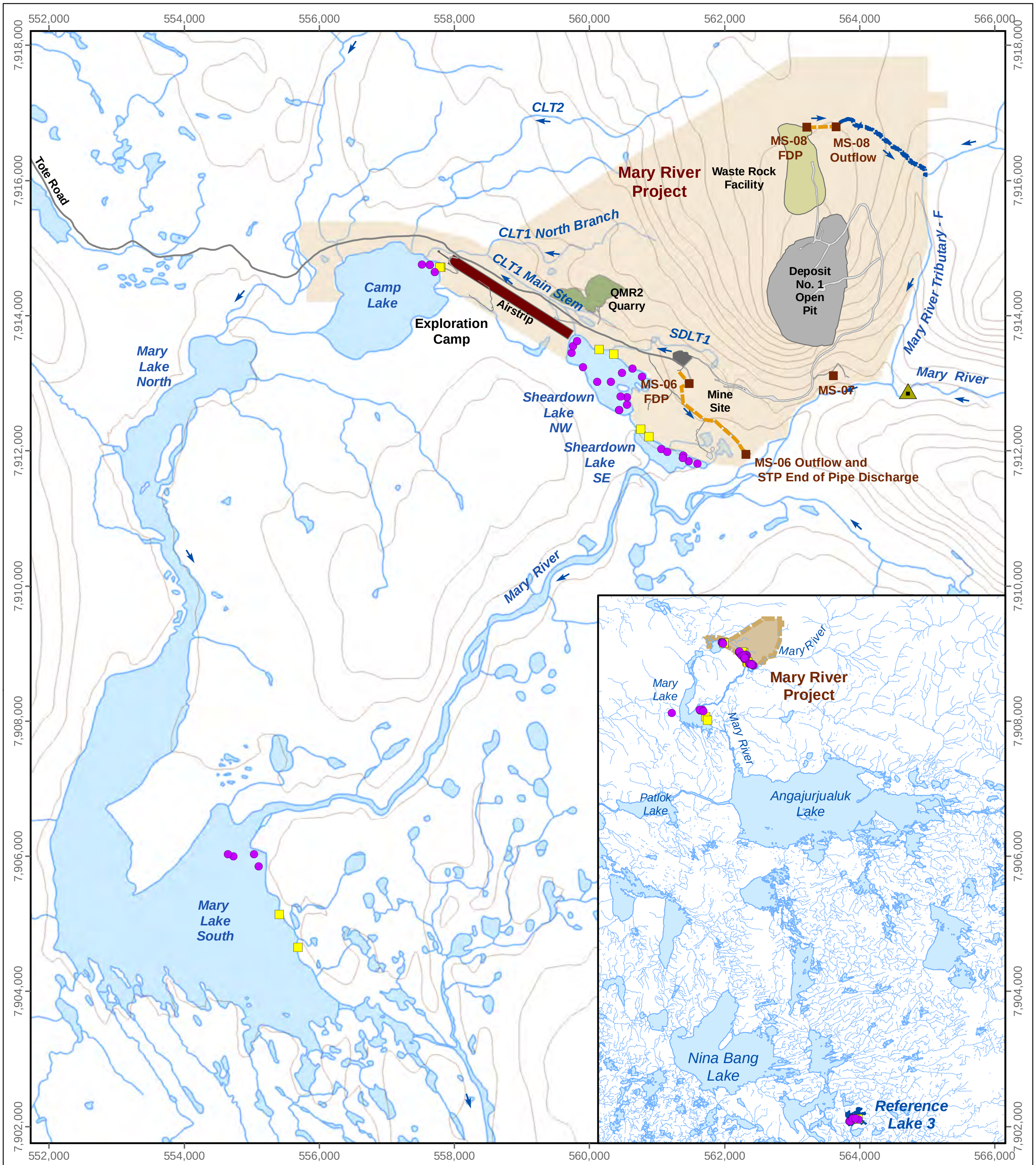
The Bray-Curtis Index was used to evaluate community level differences between study areas, and was computed and assessed statistically using procedures recommended for federal EEM studies (i.e., Borcard and Legendre 2013). Specifically, community level differences between study areas were assessed in 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 2021 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 where appropriate. 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 2.2.2.2). Similar to the 2021 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 2.5). 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 2021 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. Once every three years under the EEM program, fish population survey and fish tissue data are collected from Mary River near- and far-field mine-exposed areas, as well as an unnamed tributary to Angajurjualuk Lake. The EEM sampling was last conducted in 2020



LEGEND

- Final Discharge Point (FDP)
- Gill Netting
- Electrofishing
- Mary River Cascade Barrier
- Discharge Line
- Road
- Waste Rock Facility
- Ultimate Deposit No. 1 Pit Limits
- Mine Site Complex Pad
- Airstrip

Mary River Project 2021 CREMP Fish Survey Sampling Locations

0 1.25 2.5 5 km

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

N W E S

Date: March 2022
Project 217202.0075

minnow environmental inc.

Figure 2.5

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 217202.0075 - AEMP 2021\D - GIS\AEMP 2021\21-75 Figure 2.5 Fish Survey Sampling Locations.mxd

(Minnow 2021a), therefore fish population survey and fish tissue data were not collected at these areas in 2021.

2.4.3.2 Sample Collection

Nearshore areas of study lakes were sampled for arctic charr using a battery powered backpack electrofishing unit (Model LR-24, Smith-Root Inc., Vancouver, WA). An electrofishing team, consisting of the backpack electrofisher operator and a single netter, conducted a single fishing pass at up to two shoreline reaches of each study lake (Figure 2.5). The number of passes conducted at each lake was dependent upon catch success, with an additional pass required in instances in which 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 below (Section 2.4.3.3). Additional supporting information collected for each electrofishing pass included recording the GPS coordinates at boundaries of each electrofishing reach and a description of the habitat within the 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 0.02 to 3.0 hours per set; average of 1.48 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, sampling depth range, GPS coordinates, and habitat descriptions were recorded.

2.4.3.3 Field and Laboratory Processing

Following completion of each electrofishing pass and retrieval of each individual gill net panel, all captured arctic charr were subject to processing in the field. For all live captures, the external condition of each individual was assessed visually for the presence of any deformities, erosions, lesions, and tumors (DELT), in addition to evidence of external and/or internal parasites. All observations were recorded on field sheets, with supporting photographs taken as appropriate. Each fish was then subject to measurement of fork and total length to the nearest millimetre using a standard measuring board. Following length measurements, fish captured 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 captured by electrofishing and gill netting that were not selected for the collection of aging structures were released near the location of capture following these individual measurements of length and weight.

As specified for EEM non-lethal fish population surveys (see Environment Canada 2012), approximately 10% of the targeted number of arctic charr captured using electrofishing methods were sacrificed for collection of aging structures. Otoliths were removed from all sacrificed individuals and incidental mortalities for age determination. Upon removal, these aging structures were wrapped separately in wax paper, placed inside envelopes labelled with the fish identification, and then dried for storage. Age structures (otoliths) were shipped to North Shore Environmental Services (NSES; Thunder Bay, ON) for age determination. At the laboratory, otoliths were prepared for aging using a "crack and burn" method. The prepared otolith samples were mounted on a glass slide using a mounting medium and examined under a compound microscope using transmitted light to determine fish age. For each structure, the age and edge condition 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 2021.

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),. 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 2.7) 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 collected in 2021, and between the combined baseline period and 2021 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 2021, and between the mine baseline period and 2021. 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 2021 mine-exposed and reference area data, or between the 2021 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 2021 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

Table 2.7: Fish Population Survey Endpoints Examined for the Mary River Project CREMP 2021 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).

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 parallel-regression 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.

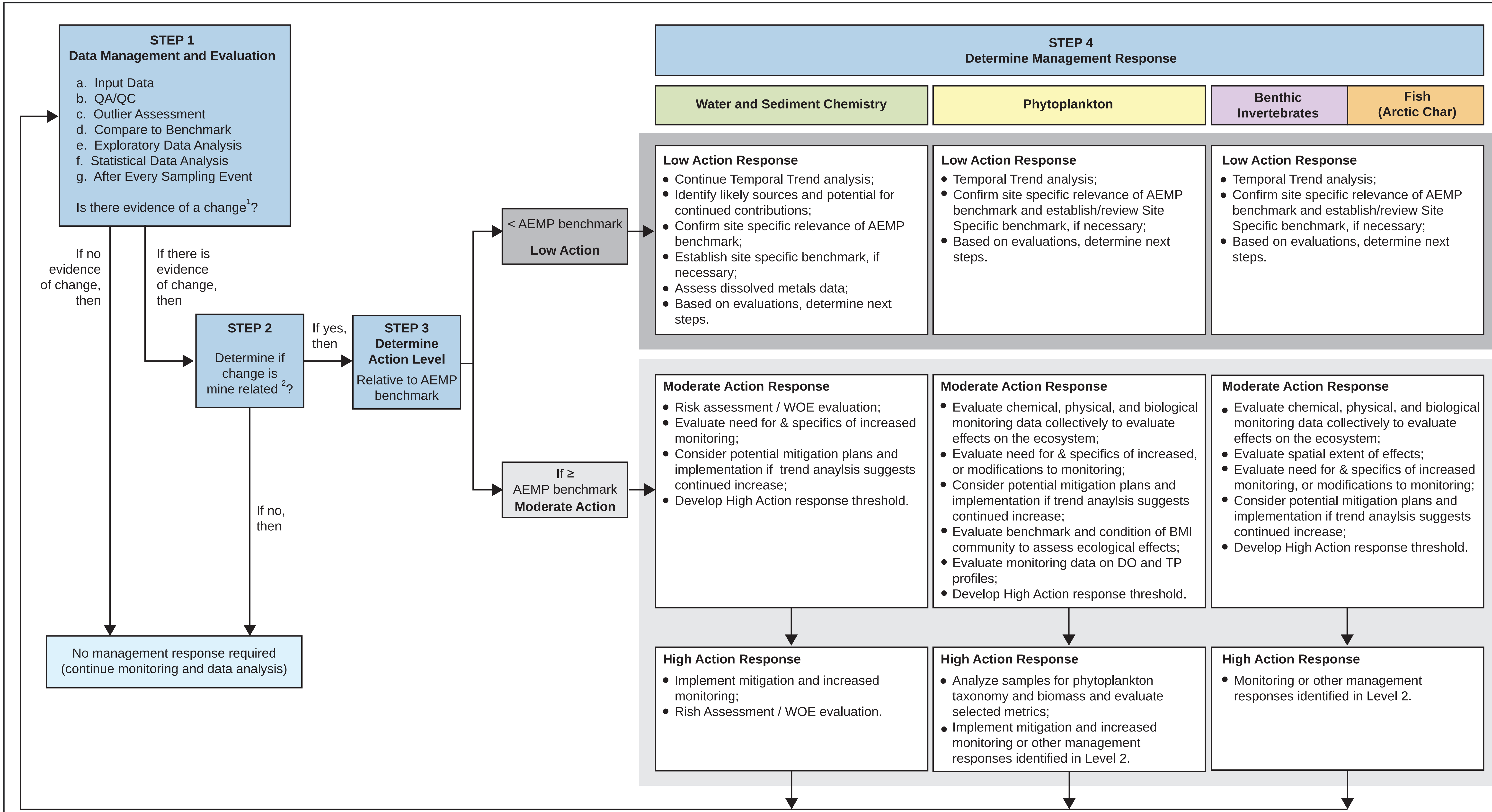
For endpoints showing significant differences, the magnitude of difference between 2021 mine-exposed and reference data or between 2021 and baseline data was calculated as described by Environment Canada (2012) using mean (ANOVA), adjusted mean (ANCOVA with no significant interaction), or predicted values (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 2.7; 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 2021, or at mine-exposed areas in 2021 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

2.5.1 2021 CREMP Objective and Approach

The objective of the Mary River Project 2021 CREMP was to evaluate potential mine-related influences on chemical and biological conditions at aquatic environments located near the mine following the seventh full year of mine operation. The 2021 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 2021 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 2.6; 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 2021 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 trents
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: February 2022
Project 217202.0075

minnow environmental inc.

Figure 2.6

217202.0075/Baffinland - 217202.0075 - AEMP 2021//D-GIS/AEMP 2021/Graphics/21-75 Figure 2.6 Schematic

2.5.2 Implementation of 2020 Effects Assessment Recommendations

The effects assessment approach described above was applied to data collected in 2020 as part of the Mary River Project 2020 CREMP resulting in the provision of recommendations to address instances in which AEMP benchmarks for water quality and sediment quality were exceeded, or to address data gaps under the current Revision 1 of the Baffinland AEMP (Minnow 2021b). In response, Baffinland implemented the following actions in 2021:

- At the CLT1 north branch, the aqueous total copper concentration was greater than the AEMP benchmark in spring and summer of 2020 at Station L1-08. Based on uncertainty in whether a change in copper concentrations had occurred at the CLT1 north branch between the period of commercial mine production and baseline, a low action response was recommended to explore sources of copper to this portion of the CLT1 system (Minnow 2021b). In 2021, an expanded spatial water quality sampling program was implemented at the CLT1 north branch as a special investigation to identify distinct source(s) of copper to the watercourse. For this special investigation, three additional water quality monitoring stations (L1-CUSI-T1, L1-CUSI-T2, and L1-CUSI-A) to those currently sampled in the CLT1 north branch (L1-02, L1-08) were situated above and below gulches deemed to potentially act as potential sources of copper to the CLT1 north branch system (Figure 2.2). Water quality sampling was conducted at these additional stations using methods outlined herein (see Section 2.2.1) in fall 2021. Analysis of data from these additional stations included review of turbidity and changes in total and dissolved concentrations of copper with progression from upstream to downstream in the system to identify whether a distinct source of copper existed.
- At the CLT1 upper main stem Station L2-03, aqueous total aluminum and iron concentrations were greater than the AEMP benchmark in spring and throughout the year, respectively, in 2020 that, for iron, appeared to be mine-related (Minnow 2021b). The mine-related elevation in iron concentration at the CLT1 upper main stem was spatially limited and did not result in effects to phytoplankton or benthic invertebrates in the CLT1 lower main steam, and thus a low action response was recommended (Minnow 2021b). In response, a benthic invertebrate community sampling area was established at the CLT1 upper main stem in fall 2021, close to water quality Station L2-03, to provide information for evaluating possible biological effects related to iron concentrations in this portion of the CLT1 system (Figure 2.4). Benthic invertebrate community sampling and data analysis applied in 2021 for this assessment were identical to those described herein for other lotic areas within the CLT1/CLT2 system (see Section 2.4.2).

- At SDLT1, aqueous total copper concentrations were greater than the AEMP benchmark in spring, summer, and fall monitoring events in 2020, as well as during baseline (Minnow 2021b). Because concentrations of copper were greater than the AEMP benchmark during baseline and no adverse effects to biota were associated with copper concentrations above the AEMP benchmark in 2021 at SDLT1, a low action response to identify the likely source(s) of copper to the SDLT1 system was recommended to meet obligations under the AEMP Management Response Framework (Minnow 2021b). In response, an expanded spatial water quality sampling program was implemented at SDLT1 in fall 2021 as a special investigation to identify distinct source(s) of copper to the watercourse. For this special investigation, six additional water quality monitoring stations (from upstream to downstream, D1-CUSI-A, D1-CUSI-B, D1-CUSI-C, MS-C-F, MS-C-B, and MS-C-A) to those currently sampled at SDLT1 (D1-05, D1-00) were sampled (Figure 2.2). Water quality sampling was conducted at these additional stations using methods outlined herein (see Section 2.2.1) in fall 2021. Analysis of data from these additional stations included review of turbidity and changes in total and dissolved concentrations of copper with progression from upstream to downstream in the system to identify whether a distinct source of copper existed.

- The previous CREMP study identified the lack of water quality monitoring at SDLT12 and SDLT9 as a data gap for understanding potential mine-related effects on benthic invertebrate communities sampled in these tributaries under the CREMP (Minnow 2021b). In response, water quality monitoring stations were established at SDLT12 and SDLT9 (Stations LDFG-OUT and MS-C-G, respectively; Figure 2.2) in fall 2021 to provide water chemistry data to support the interpretation of biological data. Water chemistry sampling and data analysis applied at these new stations were identical to those described herein for other lotic areas including SDLT1 (see Section 2.2.1).

- At Camp Lake, arsenic, iron, and nickel concentrations in sediment were above applicable lake-specific AEMP benchmarks in 2020 which resulted in a low action response (Minnow 2021b). The recommendation to address concentrations of these metals above AEMP benchmarks included relocating and harmonizing sediment quality monitoring stations with benthic invertebrate monitoring stations. This recommendation intended to improve the ability of the CREMP to evaluate mine related effects to biota by potentially establishing linkages between metal concentrations in sediment and benthic invertebrate responses at Camp Lake (Minnow 2021b). In response, updates to the design of the CREMP that reflected this recommendation were incorporated into the draft Revision 2 AEMP for intervenor review and the Nunavut Water Board approval.

- At Sheardown Lake, lake-specific AEMP benchmarks for sediment quality were exceeded for iron, manganese, and nickel concentrations at both the Northwest (NW) and Southeast (SE) basins, as well as for arsenic at Sheardown Lake NW and chromium at Sheardown Lake SE, in 2020 (Minnow 2021b). None of these metals were elevated in the sediment of either the Sheardown lakes compared to the reference lake as well as compared to concentrations at the Sheardown lakes during baseline (Minnow 2021b). 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). Because no mine-related changes in metal concentrations occurred in sediment at Sheardown Lake NW or Sheardown Lake SE in 2020, and also no adverse effects to biota were associated with concentrations of metals above AEMP benchmarks for sediment quality at these lakes in 2020, a low action response was recommended (Minnow 2021b). Specifically, it was recommended that because concentrations of metals in Sheardown Lake NW and Sheardown Lake SE sediment were similar to those shown at the reference lake, updates to the AEMP sediment quality benchmarks for these lakes to reflect not only baseline data, but also reference lake data and/or applicable SQG. In response, updates to the design of the CREMP that reflected this recommendation were incorporated into the draft Revision 2 AEMP for intervenor review and the Nunavut Water Board approval.

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, upper main stem, 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 2021 (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 main stem stations compared to Unnamed Reference Creek during the fall sampling event in August 2021, the pH at all CLT1 stations was consistently within WQG limits in 2021 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 upper and lower main stem in August 2021, indicating no substantial influence of the Tote Road on in-stream pH (Figure 3.1; Appendix Table C.13).

Specific conductance at CLT1 was generally highest in the upper main stem (Station L2-03) and lowest in the north branch (Stations CLT1-US, 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 (Figure 3.1, Appendix Figure C.1; Appendix Tables C.1 to C.3, and C.12). Specific conductance was consistently higher at CLT1 upper and lower main stem compared to the reference creek stations over the spring, summer, and fall sampling events in 2021 (Appendix Tables C.1 to C.3, and C.13), and was also significantly higher compared to Unnamed Reference Creek during the August 2021 biological study (Figure 3.1). No significant difference in specific conductance was indicated between the CLT1 lower main stem and the upper main stem or between the CLT1 lower main stem and the upstream north branch in August 2021 (Appendix Table C.13), suggesting that the source of elevated specific conductance was unrelated to the Tote Road but rather was associated with the upper main stem portion of the CLT1 system.

At the CLT1 north branch stations (L1-08 and L1-02), water chemistry met AEMP benchmarks and WQG in 2021 except for copper concentrations, which were elevated relative to one or both

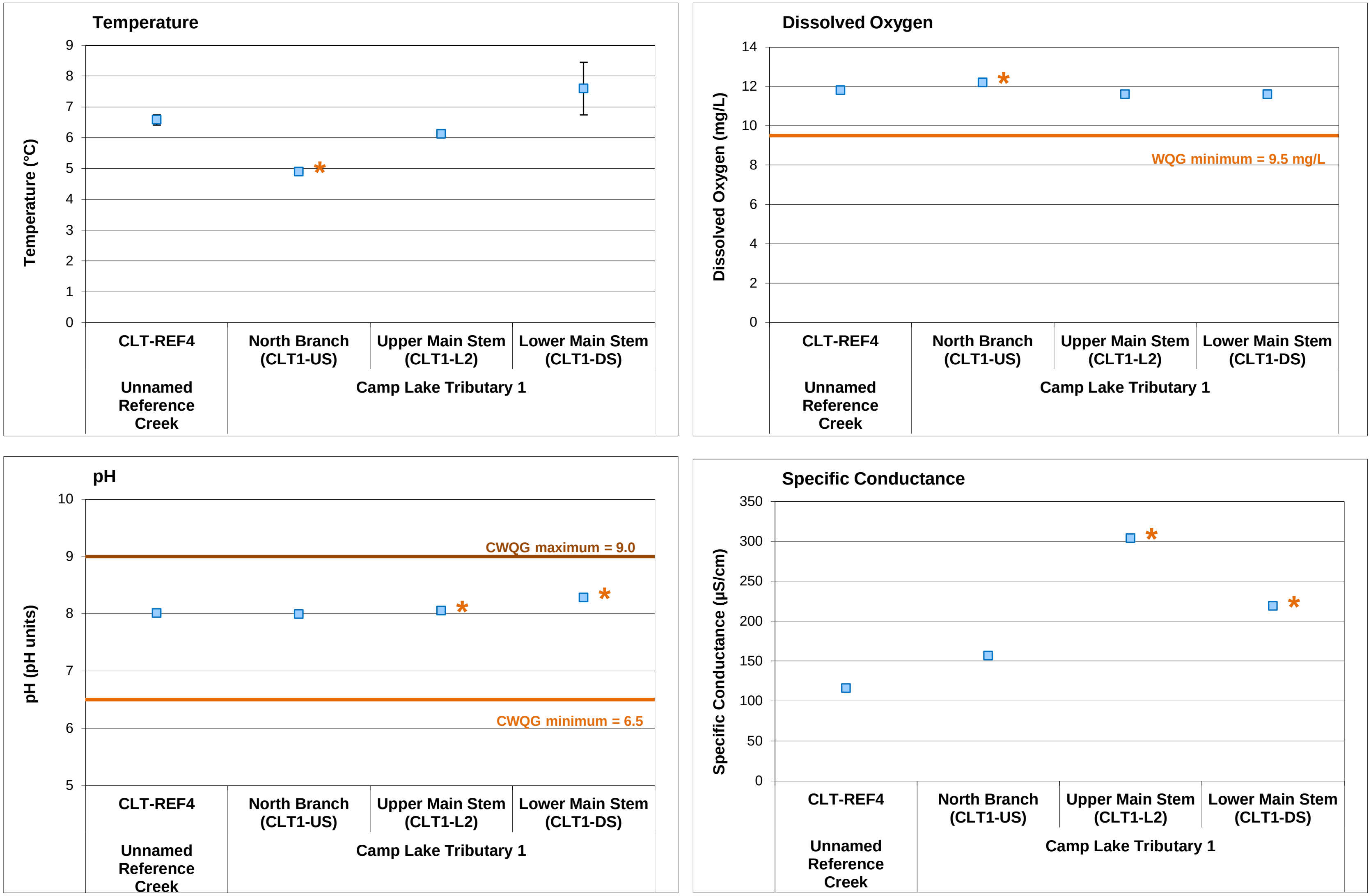


Figure 3.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 1 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2021

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

criteria during the summer and fall sampling events (Table 3.1; Appendix Table C.14). However, most parameters, including copper concentrations at the CLT1 north branch, were not particularly elevated compared to the reference creek stations, with only total and dissolved potassium concentrations showing slight elevation (i.e., 3- to 5-times higher) at the CLT1 north branch during the spring sampling event in 2021 (Table 3.1; Appendix Tables C.14, C.15, and C.17). Ranges of total copper concentrations at the CLT1 north branch were comparable between years of mine production from 2015 to 2021 and baseline for spring, summer, and fall (Appendix Figure C.2). Water quality sampling at CLT1 north branch was spatially expanded in fall 2021 as a special investigation into elevated copper concentrations (see Section 2.5.2). Water quality monitoring samples collected downstream of Station L1-08 (Station L1-CUSI-T1 and Station L1-CUSI-T2) met the AEMP benchmark and WQG for copper (Appendix Table H.1), suggesting elevated copper concentrations had a limited spatial extent within the CLT1 north branch. Conversely, the concentration of copper at the station upstream of L1-08 (Station L1-CUSI-A) was above both criteria (Appendix Table H.1), suggesting an upstream source of copper in the CLT1 system. Therefore, except for naturally elevated concentrations of copper above the AEMP benchmark within a limited portion of the CLT1 system, water quality has generally met applicable benchmarks at the CLT1 north branch since commercial mine production commenced at the project in 2015.

At the CLT1 upper main stem (Station L2-03), concentrations of aluminum and iron were above their respective AEMP benchmarks and WQG in spring, summer, and fall sampling events in 2021 (Table 3.1). The total concentration of uranium was also elevated above WQG at the upper main stem during spring, summer, and fall 2021 (Table 3.1). In addition to aluminum, iron, and uranium concentrations, of the parameters with AEMP benchmarks, total ammonia, chloride, sulphate, total lead, total nickel, and nitrate were slightly (i.e., 3-times to 5-times higher) to highly (i.e., ≥10-times higher) elevated at the CLT1 upper main stem compared to average concentrations at the reference creeks in two or more seasonal sampling events in 2021 (Table 3.1; Appendix Table C.15). Although total concentrations of aluminum, iron, and several other parameters were elevated at the CLT1 upper main stem during the spring, summer, and fall sampling events 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).[8]

[8] Total aluminum concentrations were also above AEMP benchmarks and/or WQG at the MRY-REF3 lotic reference station in 2021, where higher turbidity typically occurs compared to the other reference creek stations (Appendix Table B.2). This suggested natural elevation of aluminum 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

Table 3.1: Mean Water Chemistry at Camp Lake Tributary 1 (CLT1) Monitoring Stations During Spring, Summer, and Fall, Mary River Project CREMP, 2021

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creeks (n=4)			North Branch (n=2)			Upper Main Stem L2-03 (n=1)			Lower Main Stem (n=3)		
					Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	45.2	97.8	149	97.95	150	181.5	116	314	356	115.0	197.3	262
	pH (lab)	pH	6.5 - 9.0	-	7.58	7.75	7.98	7.8	8.07	8.09	6.01	8.15	8.05	7.21	8.18	8.12
	Hardness (as $CaCO_3$)	mg/L	-	-	19.7	43.6	68.8	45.2	72.15	89.15	51.3	120	140	50.7	88.9	113
	Total Suspended Solids (TSS)	mg/L	-	-	3.12	2.05	2.22	2	2	2	6.6	8.7	4.5	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	23.0	50.8	87.2	30.5	62.5	94.5	146	158	180	68	102	133
	Turbidity	NTU	-	-	2.05	3.11	2.93	1.92	0.46	0.325	19.9	16.8	18.6	3.43	2.77	2.55
	Alkalinity (as $CaCO_3$)	mg/L	-	-	20.4	40.3	68.1	46	68.2	93.7	88.6	119	140	50.4	90.2	117
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	<0.01	<0.01	<0.01	0.010	0.012	0.010	0.027	0.263	0.034	<0.010	0.0295	0.0102
	Nitrate	mg/L	3	3	<0.02	<0.02	0.0363	0.0245	0.0305	0.0715	0.506	0.969	0.935	0.066	0.144	0.302
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.005	0.005	0.005	0.005	0.005	0.0056	0.0044	<0.0050	<0.0010	0.0015
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.125	0.100	0.105	0.18	0.11	0.1195	0.27	0.724	0.349	0.113	0.167	0.157
	Dissolved Organic Carbon	mg/L	-	-	1.03	2.75	2.65	1.555	2.585	2.965	4.41	4.28	4.02	1.98	2.83	2.64
	Total Organic Carbon	mg/L	-	-	1.66	1.76	2.10	2.335	2.445	2.62	4.67	4.79	3.84	2.36	2.99	2.40
	Total Phosphorus	mg/L	0.030[α]	-	0.00475	0.00470	0.00462	0.00375	0.00425	0.0112	0.0208	0.0047	0.0098	0.0034	0.002	0.0029
	Phenols	mg/L	0.004[α]	-	<0.001	0.00103	<0.001	0.001	0.001	0.001	0.001	0.001	0.001	<0.0010	<0.0010	<0.0010
Anions	Bromide (Br)	`	-	-	<0.1	<0.1	<0.1	0.10	0.10	0.10	0.10	0.066	0.086	<0.10	<0.050	0.0372
	Chloride (Cl)	mg/L	120	120	1.19	2.66	4.61	1.035	1.465	2.605	17.4	19.8	19.6	3.48	5.28	9.00
	Sulphate (SO_4)	mg/L	218[β]	218	1.26	3.34	5.78	2.38	2.57	4.765	10.4	13.5	14.1	2.71	4.75	7.6
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0469	0.0822	0.0476	0.0376	0.01345	0.01145	**0.234**	**0.335**	**0.391**	0.0442	0.0610	0.0455
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	0.0001	0.0001	0.0001	0.0001	0.00006	0.000065	<0.00010	<0.000020	<0.000030
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	0.000102	<0.0001	0.0001	0.0001	0.0001	0.00018	0.00016	0.000142	<0.00010	0.000071	0.000079
	Barium (Ba)	mg/L	-	-	0.00292	0.00592	0.00814	0.006525	0.009645	0.01115	0.0123	0.0143	0.0147	0.00677	0.01073	0.0121
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.0005	0.0005	0.0005	0.0005	0.0005	0.0000215	0.000021	<0.00050	<0.0000050	<0.0000050
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.0005	0.0005	0.0005	0.0005	0.0005	0.0000154	0.00005	<0.00050	<0.0000050	<0.000050
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	0.01	0.01	0.01	0.017	0.021	0.019	<0.010	0.0074	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.00001	<0.00001	<0.00001	0.00001	0.00001	0.00001	0.00001	0.0000074	0.0000077	<0.000010	<0.0000050	<0.0000050
	Calcium (Ca)	mg/L	-	-	4.09	9.02	13.2	8.955	14.05	17.1	19.1	24.4	0	10.2	17.6	-
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	<0.0005	0.0005	0.0005	0.0005	0.0005	0.00063	0.00075	<0.00050	0.00030	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.0040	<0.0001	<0.0001	<0.0001	0.0001	0.0001	0.0001	0.00028	0.000288	0.000348	<0.00010	0.0000695	0.000072
	Copper (Cu)	mg/L	0.002	0.0022	0.000645	0.000852	0.000888	0.00177	**0.00226**	0.00205	0.0017	0.00183	0.00182	0.00144	0.00195	0.00192
	Iron (Fe)	mg/L	0.30	0.326	0.0582	0.0672	0.0570	0.0535	0.03	0.03	**0.444**	**0.530**	**0.824**	0.075	0.123	0.140
	Lead (Pb)	mg/L	0.001	0.001	0.0000950	0.000110	0.0000890	0.000079	0.00005	0.00005	0.000855	0.00077	0.000547	0.000109	0.000119	0.000076
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	0.001	0.001	0.001	0.0045	0.00427	0.00478	0.0013	0.0018	0.00216
	Magnesium (Mg)	mg/L	-	-	2.36	5.10	8.50	5.585	8.945	11.6	12.1	15.6	18.2	6.22	11.07	14.5
	Manganese (Mn)	mg/L	0.935[β]	-	0.00104	0.00100	0.000942	0.001173	0.0005375	0.0006095	0.0287	0.0247	0.0342	0.00337	0.00556	0.00763
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	0.000005	0.000005	0.000005	0.000005	0.000005	0.000005	<0.0000050	<0.0000050	<0.0000050
	Molybdenum (Mo)	mg/L	0.073	-	0.000119	0.000271	0.000399	0.000308	0.0008005	0.0011065	0.00248	0.00382	0.00326	0.000614	0.001120	0.00150
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	0.000512	0.000532	0.0005	0.000575	0.00053	0.00162	0.00188	0.00183	0.00065	0.00097	0.00102
	Potassium (K)	mg/L	-	-	0.358	0.675	0.883	1.13	1.865	2.14	3.42	4.35	4.12	1.38	2.38	2.58
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	<0.001	0.001	0.001	0.001	0.001	0.000128	0.0002	<0.0010	0.00005	<0.00020
	Silicon (Si)	mg/L	-	-	0.528	0.925	0.948	0.71	0.8	0.92	1.06	1.33	1.77	0.59	0.90	1.16
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	0.00001	0.00001	0.00001	0.00001	0.00001	0.000005	<0.000010	<0.0000050	<0.0000050
	Sodium (Na)	mg/L	-	-	0.674	1.66	2.62	0.536	1.0025	1.465	9.09	13.5	11.7	2.02	3.74	5.48
	Strontium (Sr)	mg/L	-	-	0.00406	0.00962	0.0149	0.005175	0.008965	0.01185	0.0245	0.0274	0.0373	0.00791	0.0155	0.0211
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.0001	0.0001	0.0001	0.0001	0.0001	0.0000148	0.0000141	<0.00010	0.0000090	0.0000079
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	0.0001	0.0001	0.0001	0.0001	0.00002	0.0002	<0.00010	<0.000020	<0.00020
	Titanium (Ti)	mg/L	-	-	<0.01	0.0102	<0.01	0.01	0.01	0.01	0.01	0.0149	0.0182	<0.010	0.0030	0.00188
	Uranium (U)	mg/L	0.015	-	0.000380	0.00167	0.00527	0.0008225	0.002585	0.00557	0.0154	0.0284	0.0282	0.00256	0.00526	0.00919
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	<0.001	0.001	0.001	0.001	0.001	0.000719	0.00076	<0.0010	0.00029	0.00030
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	<0.003	0.003	0.003	0.003	0.003	0.00231	0.0032	<0.0030	0.0010	0.0031

[shaded] 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.

In contrast, dissolved concentrations of lithium, manganese, molybdenum, potassium, sodium, and uranium were slightly to highly elevated at the upper main stem in all sampling events in 2021, and thus elevations of these metals (and chloride, nitrate, and sulphate) 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 2021 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 benchmarks, also showed elevated concentrations in water at the CLT1 upper main stem in2021 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 2021, except for copper concentration elevated above WQG, but not the AEMP benchmark, in one sample collected in the fall (Table 3.1; Appendix Table C.14). Nevertheless, manganese, molybdenum, nickel, nitrate, potassium, and uranium concentrations were slightly or moderately elevated at the CLT1 lower main stem compared to the reference creeks in at least one of the three seasonal sampling events in 2021 (Appendix Table C.15). Of those parameters with AEMP benchmarks, only copper, iron, nitrate, and sulphate showed elevated concentrations in 2021 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 2021 compared to baseline (Appendix Figure C.2).

Higher copper, 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[9], and/or fugitive dust generation from increased truck usage on the 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. 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 upper main stem stations from 2019 to 2021 compared to all previous years of mine operation, and exceeded WQG, suggesting some potential for biological effects (Appendix Figure C.2). Overall, mine-related influences on water quality of

[9] The QMR2 quarry is used to provide material for mine infrastructure projects (e.g., road construction). No blasting occurred at QMR2 in 2021.

the CLT1 were primarily reflected as elevated conductivity and concentrations of copper and potassium at the north branch, and elevated concentrations of nitrate, chloride, sulphate, and total metals including iron, manganese, molybdenum, potassium, 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 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 2021 (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 2021 (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, summer, and fall sampling events (Appendix Table E.2). Relatively high chlorophyll-a concentrations at Station L2-03 and in the CLT1 lower main stem during spring, summer, and fall sampling events potentially reflected higher nutrient (e.g., nitrate) concentrations compared to the reference creeks (Appendix Tables C.14 and C.15). Nevertheless, chlorophyll-a concentrations at all CLT1 north branch and main stem monitoring stations were well below the AEMP benchmark of 3.7 µg/L for all seasonal sampling events in 2021 (Figure 3.2). Similar to the reference creek stations, chlorophyll-a concentrations at all CLT1 stations in 2021 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 2021) 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 2021 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 2021 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

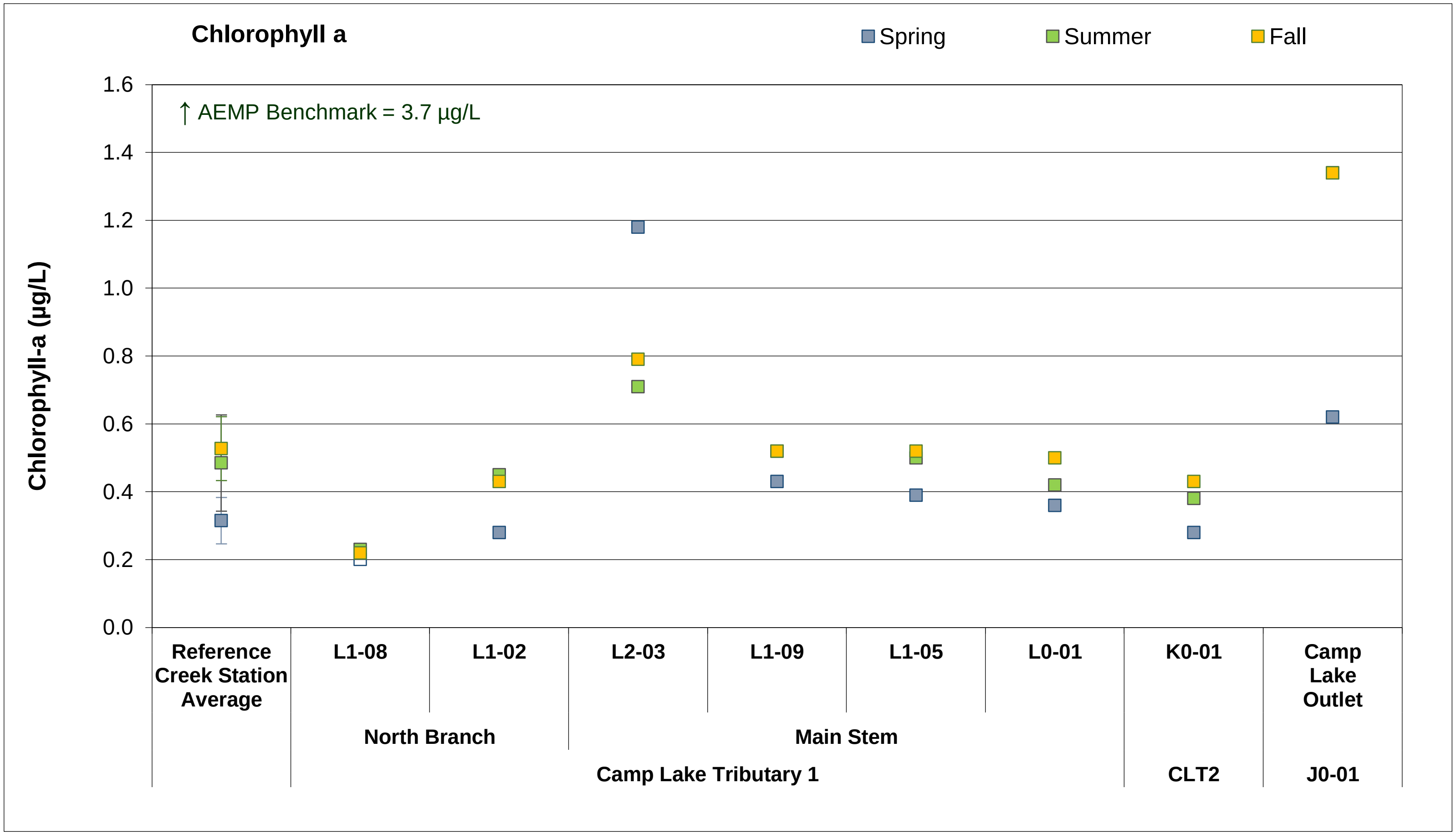


Figure 3.2: Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT1) and Tributary 2 (CLT2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2021

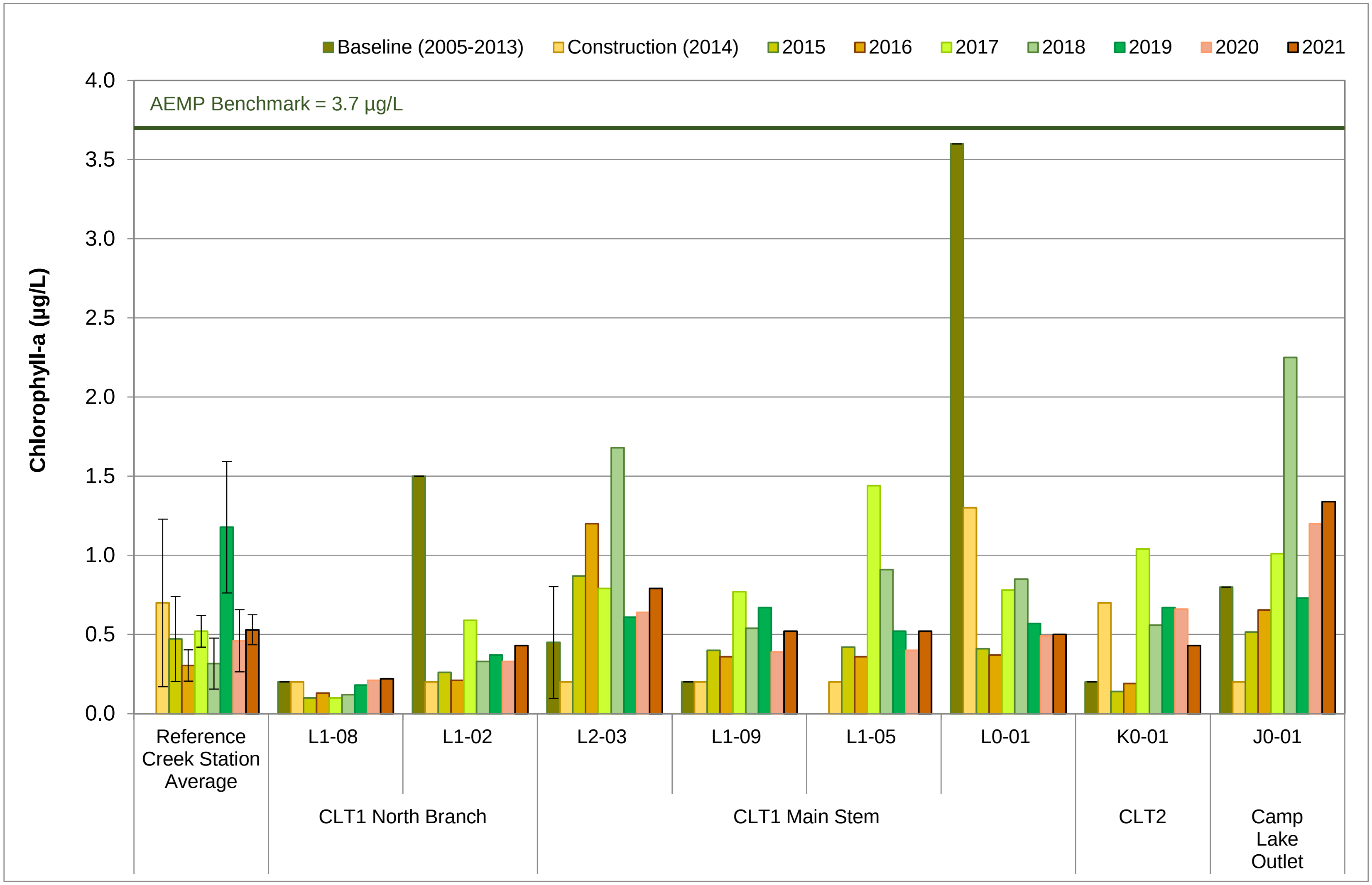


Figure 3.3: Temporal Comparison of Chlorophyll-a Concentrations at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2021) Periods during Fall

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

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 2021 (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 the CLT1 upper main stem station during spring, summer, 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 upper 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 upper 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 upper main stem stations than at the reference creeks in spring, summer, and fall of 2021, the CLT1 north branch and lower main stem have remained 'oligotrophic' since the commencement of commercial mine operation in 2015.

3.1.3 Benthic Invertebrate Community

3.1.3.1 Upstream North Branch (CLT1-US)

The benthic invertebrate community at the CLT1 upstream (north branch) had no significant differences in density, richness, and Simpson's Evenness compared to the Unnamed Reference Creek in 2021 (Table 3.2; Appendix Figure F.1). The benthic invertebrate community assemblage differed between the CLT1 north branch and Unnamed Reference Creek as indicated by significantly differing Bray-Curtis Index (Appendix Table F.7), which included ecologically significant[10] greater relative abundance of Chironomidae and Tipulidae dominant groups, and significantly lower relative abundance of Simuliidae, at the CLT1 north branch (Table 3.2). Within the Chironomidae, no significant difference in relative abundance of metal-sensitive taxa was indicated at the CLT1 north branch compared to at the reference creek, indicating no adverse influences on biota related to metal concentrations within the watercourse. Key differences in FFGs that included significantly higher and lower relative abundance of shredders and filterers, respectively, at the CLT1 north branch compared to the reference creek (Table 3.2) suggested a potential difference in types of food resources available to benthic invertebrates between these areas. No significant differences in relative abundance of individual HPG were shown

[10] 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.

Table 3.2: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 1 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2021

Metric	Overall 4-Area Comparison[a]				Pair-wise, *post hoc* comparisons				
	Statistical Test	Transform-ation	Significant Difference between Areas?	p-value	Study Area	Mean	Standard Deviation	Magnitude of Difference[b]	Pairwise Comparison
Density (No. per m²)	ANOVA	log10	YES	0.002	Reference Creek	1,129	820	nc	B
					CLT1 North Branch	1,087	377	-0.1	B
					CLT1 Upper Main Stem	3,128	1,579	2.4	A
					CLT1 Lower Main Stem	675	344	-0.6	B
Richness (No. of Taxa)	ANOVA	none	NO	0.246	Reference Creek	20.0	3.3	nc	A
					CLT1 North Branch	17.6	3.9	-0.7	A
					CLT1 Upper Main Stem	17.6	2.4	-0.7	A
					CLT1 Lower Main Stem	16.0	2.0	-1.2	A
Simpson's Evenness	ANOVA	none	NO	0.120	Reference Creek	0.924	0.052	nc	A
					CLT1 North Branch	0.894	0.051	-0.6	A
					CLT1 Upper Main Stem	0.838	0.073	-1.7	A
					CLT1 Lower Main Stem	0.891	0.024	-0.6	A
Nemata (% of community)	ANOVA	log10	NO	0.355	Reference Creek	2.3	2.0	nc	A
					CLT1 North Branch	3.7	1.1	0.7	A
					CLT1 Upper Main Stem	2.0	1.5	-0.2	A
					CLT1 Lower Main Stem	3.5	2.9	0.6	A
Oligochaeta (% of community)	K-W	rank	YES	0.012	Reference Creek	0.2	0.3	nc	B
					CLT1 North Branch	0.4	0.4	0.7	B
					CLT1 Upper Main Stem	13.4	9.9	48.4	A
					CLT1 Lower Main Stem	1.0	1.7	2.8	B
Hydracarina (% of community)	ANOVA	none	YES	0.097	Reference Creek	5.0	3.0	nc	A
					CLT1 North Branch	2.9	1.7	-0.7	AB
					CLT1 Upper Main Stem	2.1	2.0	-0.9	AB
					CLT1 Lower Main Stem	1.7	0.9	-1.1	B
Chironomidae (% of community)	ANOVA	none	YES	<0.001	Reference Creek	40.2	7.5	nc	B
					CLT1 North Branch	70.6	7.3	4.0	A
					CLT1 Upper Main Stem	78.2	10.8	5.0	A
					CLT1 Lower Main Stem	80.8	3.0	5.4	A
Metal Sensitive Chironomidae (% of community)	ANOVA	log10	NO	0.811	Reference Creek	6.4	2.0	nc	A
					CLT1 North Branch	7.2	5.3	0.4	A
					CLT1 Upper Main Stem	10.6	8.0	2.1	A
					CLT1 Lower Main Stem	11.1	9.8	2.4	A
Simuliidae (% of community)	K-W	rank	YES	0.002	Reference Creek	34.6	11.4	nc	A
					CLT1 North Branch	2.4	1.8	-2.8	B
					CLT1 Upper Main Stem	0.0	0.0	-3.0	C
					CLT1 Lower Main Stem	1.7	1.3	-2.9	BC
Tipulidae (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	1.2	1.2	nc	B
					CLT1 North Branch	12.8	5.2	9.8	A
					CLT1 Upper Main Stem	1.9	1.4	0.6	B
					CLT1 Lower Main Stem	4.4	1.7	2.7	B
Collector Gatherers FFG (% of community)	ANOVA	log10	YES	0.012	Reference Creek	43.0	9.1	nc	B
					CLT1 North Branch	50.1	10.9	0.8	AB
					CLT1 Upper Main Stem	62.2	10.7	2.1	A
					CLT1 Lower Main Stem	61.8	5.8	2.1	A
Filterers FFG (% of community)	K-W	rank	YES	0.002	Reference Creek	33.9	12.6	nc	A
					CLT1 North Branch	0.4	0.5	-2.7	C
					CLT1 Upper Main Stem	9.8	8.4	-1.9	AB
					CLT1 Lower Main Stem	1.6	1.3	-2.6	BC
Shredders FFG (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	3.5	3.0	nc	C
					CLT1 North Branch	41.1	12.1	12.6	A
					CLT1 Upper Main Stem	23.9	3.7	6.8	B
					CLT1 Lower Main Stem	29.3	6.8	8.6	AB
Clingers HPG (% of community)	ANOVA	log10	YES	0.033	Reference Creek	42.2	13.2	nc	A
					CLT1 North Branch	33.5	8.1	-0.7	AB
					CLT1 Upper Main Stem	24.1	4.2	-1.4	B
					CLT1 Lower Main Stem	28.3	7.7	-1.1	AB
Sprawlers HPG (% of community)	ANOVA	none	YES	0.076	Reference Creek	44.3	11.7	nc	A
					CLT1 North Branch	48.9	12.5	0.4	A
					CLT1 Upper Main Stem	58.5	8.1	1.2	A
					CLT1 Lower Main Stem	60.3	8.0	1.4	A
Burrowers HPG (% of community)	ANOVA	log10	NO	0.480	Reference Creek	13.5	5.8	nc	A
					CLT1 North Branch	17.4	6.3	0.7	A
					CLT1 Upper Main Stem	17.4	10.4	0.7	A
					CLT1 Lower Main Stem	11.4	4.2	-0.4	A

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

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

Notes: nc= no comparison; nm=MOD could not be calculated due to SD = 0.

[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).

[b] Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref}$. MCT = Measure of Central Tendency; MCT reported as geometric mean for log10-transformed data, median for rank-transformed data, back-transformed means for untransformed data.

between CLT1 north branch and Unnamed Reference Creek in 2021 (Table 3.2). No consistent ecologically significant differences in density, richness, Simpson's Evenness, dominant taxonomic groups, or FFGs were indicated at the CLT1 north branch from 2015 to 2021 compared to baseline studies conducted in both 2007 and 2011 (Appendix Table F.8; Appendix Figure F.2). Collectively, the 2021 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.

3.1.3.2 CLT1 Upper Main Stem (CLT1-L2)

Benthic invertebrate density at the CLT1 upper main stem (CLT1-L2), upstream of the confluence with the north branch, was significantly greater than at the reference creek, but no significant differences in richness and Simpson's Evenness were indicated between the CLT1 upper main stem and the Unnamed Reference Creek study areas in 2021 (Table 3.2; Appendix Figure F.1). Differences in benthic invertebrate community assemblage between CLT1-L2 and the reference creek, as indicated by significantly differing Bray-Curtis Index (Appendix Table F.7), included significant greater relative abundance of Oligochaeta and Chironomidae, and significantly lower relative abundance of Simuliidae, at CLT1-L2 (Table 3.2). Within Chironomidae, no significant difference in the relative abundance of metal-sensitive taxa was indicated at the CLT1 upper main stem compared to at the reference creek, indicating no adverse influences on biota related to metal concentrations within the watercourse. Key differences in FFGs that included significantly greater relative abundance of collector gatherers and shredders at the CLT1 upper main than at the reference creek, were consistent with greater amounts of in-stream vegetation at CLT1-L2 (Appendix Table F.1). No significant differences in relative abundance of individual HPG were shown between CLT1-L2 and Unnamed Reference Creek in 2021 (Table 3.2).

No consistent ecologically significant differences in density, richness, Simpson's Evenness, and relative abundance of dominant taxonomic groups and FFGs were indicated at the CLT1 upper main stem from 2016 to 2021 compared to the baseline study conducted in 2011 (Appendix Table F.9; Appendix Figure F.2). The benthic invertebrate community at CLT1-L2 in 2021 had greater relative abundance of Tipulidae and Collector Gatherers FFG at the CLT1 upper main stem compared to baseline (2011) and 2016 (Appendix Table F.9; Appendix Figure F.2), suggesting organic matter may have been more available as a food source in 2021. Notably, no significant difference in the relative abundance of metal-sensitive taxa were shown between years of mine operation and baseline at CLT1-L2, indicating no adverse influences on biota related to metal concentrations within the watercourse. Collectively, the 2021 data suggested that differences in benthic invertebrate community assemblage between the CLT1 upper main stem 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 few ecologically significant changes in benthic invertebrate community metrics at the CLT1 upper main stem since the commencement of commercial mine operations in 2015. Therefore, elevation of aqueous iron concentrations above AEMP benchmarks at CLT1-L2 upper main in 2020 and 2021 (Sections 2.5.2 and 3.1.1) did not appear to be biologically available, resulting in having no effects on the benthic invertebrate community of this watercourse.

3.1.3.3 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 Tote Road crossing, did not differ significantly in density, richness, or Simpson's Evenness compared to Unnamed Reference Creek in 2021 (Table 3.2; 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 significantly greater relative abundance of Chironomidae and Tipulidae, and significantly lower relative abundance of Simuliidae, at CLT1-DS (Table 3.2). There was no significant difference in the relative abundance of metal-sensitive Chironomidae or HPGs, but there was significantly higher relative abundance of the collector gatherer shredder FFGs and significantly lower relative abundance of filterer FFG at CLT1-DS compared to the reference creek in 2021 (Table 3.2). This indicated that the differences in community features between CLT1-DS and Unnamed Reference Creek were unlikely associated with metal concentrations or substrate characteristics, but rather due to naturally differing habitat (e.g., food resources, potentially including in-stream vegetation) between study areas. No consistent ecologically significant differences in density, richness, Simpson's Evenness, or dominant taxonomic groups, including the relative abundance of metal-sensitive chironomids, were indicated at the CLT1 lower main stem from 2015 to 2021 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.10). 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 metal concentrations.

Between the CLT1 study areas in 2021, no significant differences in richness, evenness, or dominant taxonomic groups were indicated between CLT1 areas downstream (CLT1-DS) and upstream (CLT1-US) of the Tote Road crossing, except for lower relative abundance of Tipulidae downstream (Table 3.2; Appendix Figure F.1). Furthermore, differences in community features between the upper main stem (CLT1-L2) and upstream north branch (CLT1-US; i.e., greater density and relative abundance of filterers FFG, as well as lower relative abundance of Oligochaeta, Simuliidae, Tipulidae, and shedders FFG) did not extend to the downstream lower mainstem (CLT1-DS), indicating no substantial influence of the Tote Road or CLT1 upper main stem on the lower main stem CLT1.

3.1.4 Effects Assessment and Recommendations

3.1.4.1 Upstream North Branch (CLT1-US)

At the CLT1 north branch, the following AEMP benchmarks were exceeded in 2021:

- Aqueous total copper concentration greater than the benchmark of 0.0022 mg/L in spring (0.00245 mg/L) at Station L1-08.

Copper concentrations at the CLT1 north branch during spring, summer, and fall sampling events in 2021 were comparable to baseline (Appendix Figure C.2). 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. Because copper concentrations elevated above the AEMP benchmark in 2021 were comparable to concentrations at the time of baseline, the source of copper to the CLT1 north branch likely reflected natural minerology of the bedrock/overburden within the upper north branch watershed. No adverse effects on phytoplankton (chlorophyll-a) or benthic invertebrates of the CLT1 north branch were indicated in 2021, 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. Under the Mary River Project AEMP Management Response Framework, because no mine-related changes in concentrations of AEMP benchmark parameters occurred relative to background and no adverse biological effects were indicated in 2021, no adjustment to the existing AEMP need be applied at CLT1 north branch for the 2022 CREMP. Temporal trend analysis of copper concentrations at the CLT1 north branch will be considered in the 2022 CREMP to confirm that copper concentrations continue to show no increasing trends over time.

3.1.4.2 CLT1 Main Stem (CLT1-US and CLT1-DS)

At the CLT1 main stem, the following AEMP benchmarks were exceeded in 2021:

- Aqueous total aluminum concentration was greater than the benchmark of 0.179 mg/L in spring, summer, and fall at the upper main stem Station L2-03 (0.234 mg/L, 0.335 mg/L, and 0.391 mg/L respectively); 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 (0.444 mg/L, 0.530 mg/L, and 0.824 mg/L, respectively).

Concentrations of all parameters were below AEMP water quality benchmarks at all stations within the lower main stem in 2021 (i.e., Stations L1-09, L0-05, and L0-01). Elevation of total aluminum concentrations above the AEMP water quality benchmark at the upper main stem in spring 2021 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 2021 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 2021 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 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 upstream or downstream areas in 2021, 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. Lack of effects on the benthic invertebrate community of this portion of the watercourse also suggested that elevated aqueous iron concentrations at the CLT1-L2 upper main stem were not biologically available.

Under the Mary River Project AEMP Data Management Response Framework (Figure 2.6), the determination of a mine-related change to a parameter concentration above the AEMP benchmark necessitates a management response (Steps 2 and 3). As a result of similar elevation of iron concentrations above the AEMP benchmark at CLT1-L2 in 2020, benthic invertebrate community monitoring within the upper main stem (i.e., CLT1-L2) was implemented as a management response to determine biological implications associated with high iron concentrations at this location. Because a mine-related elevation in iron concentrations occurred at the CLT1 upper main stem in 2021, but the spatial extent was limited and no biological effects were observed, continued monitoring of the benthic invertebrate community at CLT1-L2 is recommended in 2022 (and future CREMP studies) to monitor and track changes in potential effects to biota in the CLT1 upper main stem over time.

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; Appendix Figure C.1). 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 2021 (Figure 3.4; Appendix Table C.12). Aqueous pH at the CLT2 upstream and downstream study area 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 2021 (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 Tote Road suggesting that this road crossing did not markedly influence the pH of CLT2 (Appendix Table C.19). Although *in situ* specific conductance was consistently higher at CLT2 compared to the reference creeks and also significantly higher downstream compared to upstream of the Tote Road in August 2021, the mean incremental difference in specific conductance between CLT2 areas was small (i.e., 2 µS/cm) and thus indicated only a slight influence, if any, of the road crossing on water quality at the downstream CLT2 area (Figure 3.4; Appendix Tables C.12 and C.19).

Water chemistry at CLT2 (Station K0-01) met all AEMP benchmarks and WQG in spring, summer, and fall sampling events of 2021 (Table 3.3; Appendix Table C.14). Among those parameters with established AEMP benchmarks, concentrations at CLT2 were similar to those at the reference creeks in 2021 except for moderately elevated (i.e., 5- to 10-times higher) sulphate concentrations at CLT2 compared to the references creeks during the spring sampling event and slightly elevated (i.e., 3- to 15-times higher) concentrations of chloride, nickel, and sulphate at CLT2 compared to at reference creeks during the summer sampling event (Appendix Table C.14). Alkalinity, conductivity, hardness, and concentrations of dissolved nickel, as well as total and dissolved concentrations of potassium, magnesium, manganese, and sodium, which do not have AEMP benchmarks, were also elevated at CLT2 compared to the reference creeks in 2021 (Appendix Table C.15 and C.17).

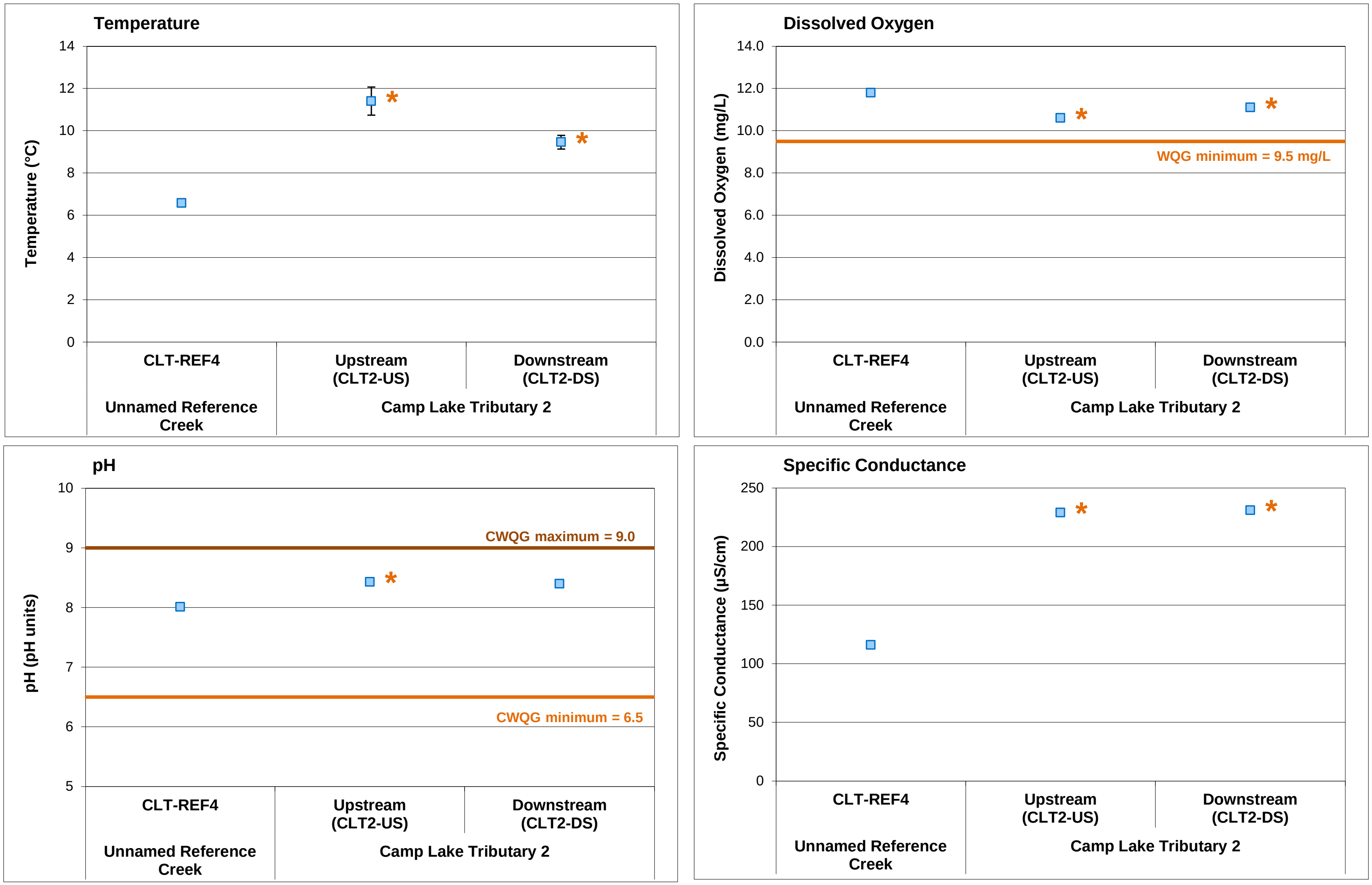


Figure 3.4: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake Tributary 2 Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2021

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

Table 3.3: Mean Water Chemistry at Camp Lake Tributary 2 (CLT2) Monitoring Stations During Spring, Summer, and Fall, Mary River Project CREMP, 2021

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creeks (n=4)			Camp Lake Tributary 2 (K0-01, n = 1)		
					Spring	Summer	Fall	Spring	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	45.2	97.8	149	103	303.0	265
	pH (lab)	pH	6.5 - 9.0	-	7.58	7.75	7.98	7.75	8.18	8.19
	Hardness (as $CaCO_3$)	mg/L	-	-	19.7	43.6	68.8	46	139	120
	Total Suspended Solids (TSS)	mg/L	-	-	3.12	2.05	2.22	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	23.0	50.8	87.2	63	177	119
	Turbidity	NTU	-	-	2.05	3.11	2.93	1.09	0.50	0.23
	Alkalinity (as $CaCO_3$)	mg/L	-	-	20.4	40.3	68.1	41	150.0	122
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	<0.01	<0.01	<0.01	<0.010	<0.0050	<0.0050
	Nitrate	mg/L	3	3	<0.02	<0.02	0.0363	0.035	0.0417	0.103
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.005	<0.0050	<0.0010	<0.0010
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.125	0.100	0.105	0.07	0.126	0.123
	Dissolved Organic Carbon	mg/L	-	-	1.03	2.75	2.65	0.90	2.27	1.82
	Total Organic Carbon	mg/L	-	-	1.66	1.76	2.10	1.50	2.32	2.05
	Total Phosphorus	mg/L	0.030[α]	-	0.00475	0.00470	0.00462	<0.0030	<0.0020	<0.0020
	Phenols	mg/L	0.004[α]	-	<0.001	0.00103	<0.001	<0.0010	<0.0010	<0.0010
Anions	Bromide (Br)	`	-	-	<0.1	<0.1	<0.1	<0.10	<0.050	<0.050
	Chloride (Cl)	mg/L	120	120	1.19	2.66	4.61	1.7	8.46	5.81
	Sulphate (SO_4)	mg/L	218[β]	218	1.26	3.34	5.78	7.93	12.3	11.0
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0469	0.0822	0.0476	0.0126	0.0091	0.0052
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.00010	<0.000020	<0.000020
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	0.000102	<0.0001	<0.00010	0.000072	0.000055
	Barium (Ba)	mg/L	-	-	0.00292	0.00592	0.00814	0.00585	0.0154	0.0129
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.0005	<0.00050	<0.0000050	<0.0000050
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.0005	<0.00050	<0.0000050	<0.0000050
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.010	0.0064	<0.0050
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.00001	<0.00001	<0.00001	<0.000010	<0.0000050	<0.0000050
	Calcium (Ca)	mg/L	-	-	4.09	9.02	13.2	8.9	26.7	23.3
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	<0.0005	<0.00050	0.00021	0.00011
	Cobalt (Co)	mg/L	0.0009[α]	0.0040	<0.0001	<0.0001	<0.0001	<0.00010	0.0000658	0.0000364
	Copper (Cu)	mg/L	0.002	0.0022	0.000645	0.000852	0.000888	0.00078	0.00145	0.00156
	Iron (Fe)	mg/L	0.30	0.326	0.0582	0.0672	0.0570	<0.030	0.109	0.0134
	Lead (Pb)	mg/L	0.001	0.001	0.0000950	0.000110	0.0000890	<0.000050	0.000017	<0.000010
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.0010	0.00173	0.00156
	Magnesium (Mg)	mg/L	-	-	2.36	5.10	8.50	5.7	17.4	16.3
	Manganese (Mn)	mg/L	0.935[β]	-	0.00104	0.00100	0.000942	0.00073	0.0128	0.000638
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.0000050	<0.0000050	<0.0000050
	Molybdenum (Mo)	mg/L	0.073	-	0.000119	0.000271	0.000399	0.00021	0.000433	0.000579
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	0.000512	0.000532	<0.00050	0.00166	0.000625
	Potassium (K)	mg/L	-	-	0.358	0.675	0.883	0.89	2.08	1.99
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	<0.001	<0.0010	<0.000040	0.000042
	Silicon (Si)	mg/L	-	-	0.528	0.925	0.948	0.40	0.923	0.853
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.000010	<0.0000050	<0.0000050
	Sodium (Na)	mg/L	-	-	0.674	1.66	2.62	1.31	5.64	3.97
	Strontium (Sr)	mg/L	-	-	0.00406	0.00962	0.0149	0.00598	0.0176	0.0173
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.0001	<0.00010	0.0000098	0.0000050
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.00010	<0.000020	<0.000020
	Titanium (Ti)	mg/L	-	-	<0.01	0.0102	<0.01	<0.010	0.000476	0.000270
	Uranium (U)	mg/L	0.015	-	0.000380	0.00167	0.00527	0.00035	0.00227	0.00295
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	<0.001	<0.0010	0.000169	0.000146
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	<0.003	<0.0030	<0.00050	0.00063

(shaded box) 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.

Chloride and sulphate concentrations were the only parameters with established AEMP benchmarks that were higher at CLT2 in 2021 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). Concentrations of chloride and sulphate at CLT2 were similar to those observed at the reference creeks in 2021, suggesting a natural factor may have accounted for higher concentrations of these parameters at CLT2 since baseline (Appendix Figure C.3). In consideration of all spatial and temporal (baseline) comparisons, no marked mine-related influence on water quality was indicated within the CLT2 system in 2021.

3.2.2 Phytoplankton

Chlorophyll-a concentrations at CLT2 (Station K0-01) were within the range observed at the reference creeks during spring, summer, and fall sampling events in 2021 (Figure 3.2). Chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L for all sampling events in 2021 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 (2021) WQG categorization of oligotrophic based on aqueous total phosphorus concentrations below 10 µg/L at CLT2 during all spring, summer, and fall sampling events (Table 3.3; Appendix Table C.14). Higher chlorophyll-a concentrations occurred at CLT2 from 2017 to 2021 compared to the mine baseline period for the fall sampling event, but no increasing trend over time was suggested (Figure 3.3). Overall, chlorophyll-a concentrations within the range observed at reference creeks and well below the AEMP benchmark suggested no adverse mine-related impact on phytoplankton productivity at CLT2 in 2021.

3.2.3 Benthic Invertebrate Community

Benthic invertebrate density at the upstream study area of CLT2 was lower than at the Unnamed Reference Creek; however, the magnitude of difference was within the CES_{BIC} of ±2 SD_{REF}, indicating that this difference was not ecologically significant (Table 3.4; Appendix Figure F.3). Benthic invertebrate richness and Simpson's Evenness at both the upstream and downstream study areas of CLT2, as well as density at the downstream study area, did not differ significantly from Unnamed Reference Creek (Table 3.4; Appendix Figure F.3). 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 Simuliidae dominant groups, respectively, at both CLT2 study areas compared to the reference creek

Table 3.4: Benthic Invertebrate Community Metric Statistical Comparison Results among Camp Lake Tributary 2 and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2021

Metric	Overall 3-Area Comparison[a]				Pair-wise, *post hoc* comparisons				
	Statistical Test	Transform-ation	Significant Difference between Areas?	p-value	Study Area	Mean	Standard Deviation	Magnitude of Difference[b]	Pairwise Comparison
Density (No. per m^2)	ANOVA	log10	YES	0.076	Reference Creek	1,129	820	nc	A
					CLT2 Upstream	392	318	-0.9	B
					CLT2 Downstream	495	279	-0.8	AB
Richness (No. of Taxa)	ANOVA	none	NO	0.377	Reference Creek	20.0	3.3	nc	A
					CLT2 Upstream	16.6	4.0	-1.0	A
					CLT2 Downstream	18.0	3.7	-0.6	A
Simpson's Evenness	ANOVA	none	NO	0.235	Reference Creek	0.924	0.052	nc	A
					CLT2 Upstream	0.870	0.078	-1.0	A
					CLT2 Downstream	0.932	0.040	0.2	A
Nemata (% of community)	ANOVA	log10(x+1)	NO	0.261	Reference Creek	2.3	2.0	nc	A
					CLT2 Upstream	1.6	1.7	-0.4	A
					CLT2 Downstream	3.6	1.8	0.6	A
Oligochaeta (% of community)	K-W	rank	YES	0.068	Reference Creek	0.2	0.3	nc	B
					CLT2 Upstream	11.0	20.6	39.6	A
					CLT2 Downstream	3.2	2.2	11.0	A
Hydracarina (% of community)	ANOVA	log10	NO	0.167	Reference Creek	5.0	3.0	nc	A
					CLT2 Upstream	5.5	1.8	0.2	A
					CLT2 Downstream	3.2	3.2	-0.6	A
Ostracoda (% of community)	K-W	rank	YES	0.043	Reference Creek	0.9	0.8	nc	A
					CLT2 Upstream	0.3	0.7	-0.7	B
					CLT2 Downstream	0.0	0.0	-1.2	B
Chironomidae (% of community)	ANOVA	none	YES	0.004	Reference Creek	40.2	7.5	nc	B
					CLT2 Upstream	62.2	14.9	2.9	A
					CLT2 Downstream	69.7	10.0	3.9	A
Metal Sensitive Chironomidae (% of community)	ANOVA	log10	YES	0.026	Reference Creek	6.4	2.0	nc	B
					CLT2 Upstream	13.5	7.3	3.6	AB
					CLT2 Downstream	17.3	7.9	5.5	A
Simuliidae (% of community)	ANOVA	log10	YES	0.002	Reference Creek	34.6	11.4	nc	A
					CLT2 Upstream	3.7	3.6	-2.7	B
					CLT2 Downstream	8.1	4.4	-2.3	B
Tipulidae (% of community)	ANOVA	log10(x+1)	NO	0.276	Reference Creek	1.2	1.2	nc	A
					CLT2 Upstream	2.1	2.2	0.7	A
					CLT2 Downstream	3.4	2.5	1.9	A
Collector Gatherers FFG (% of community)	ANOVA	none	YES	<0.001	Reference Creek	43.0	9.1	nc	B
					CLT2 Upstream	74.3	4.3	3.4	A
					CLT2 Downstream	75.3	9.2	3.5	A
Filterers FFG (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	33.9	12.6	nc	A
					CLT2 Upstream	1.5	1.7	-2.6	B
					CLT2 Downstream	4.3	4.6	-2.3	B
Shredders FFG (% of community)	ANOVA	none	NO	0.401	Reference Creek	3.5	3.0	nc	A
					CLT2 Upstream	4.7	2.9	0.4	A
					CLT2 Downstream	6.1	3.0	0.9	A
Clingers HPG (% of community)	ANOVA	none	YES	<0.001	Reference Creek	42.2	13.2	nc	A
					CLT2 Upstream	11.9	4.4	-2.3	B
					CLT2 Downstream	13.9	6.9	-2.1	B
Sprawlers HPG (% of community)	K-W	rank	YES	0.036	Reference Creek	44.3	11.7	nc	B
					CLT2 Upstream	72.3	21.9	2.4	A
					CLT2 Downstream	74.1	10.1	2.5	A
Burrowers HPG (% of community)	ANOVA	log10	NO	0.816	Reference Creek	13.5	5.8	nc	A
					CLT2 Upstream	15.8	19.1	0.4	A
					CLT2 Downstream	12.0	6.3	-0.3	A

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

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

Notes: nc= no comparison; nm=MOD could not be calculated due to SD = 0.

[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).

[b] Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; MCT reported as geometric mean for log10-transformed data, median for rank-transformed data, back-transformed means for untransformed data.

(Table 3.4; Appendix Figure F.3). Ecologically significant higher relative abundance of metal-sensitive chironomids was indicated at the downstream CLT2 study area compared to the reference creek and although higher, there was no significant difference in relative abundance of this group at the CLT2 upstream area compared to the reference creek (Table 3.4). This suggested that the community composition differences between watercourses were not likely related to metal concentrations. Ecologically significant differences in benthic invertebrate FFG and HPG were shown at both CLT2 study areas compared to the reference creek (Table 3.4), including significantly greater relative abundance of collector gatherers FFG and the sprawler HPG but significantly lower relative abundance of filterers FFG and the clinger HPG at the CLT2 study areas compared to the reference creek. This suggested potential differences in key food resources and habitat for 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 2021) compared to 2007 baseline data except for routinely higher evenness, lower relative abundance of Chironomidae and shredders at upstream CLT2 in years of mine operation(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 2021 compared to baseline was not consistent with an adverse influence related to recent mine operations. Overall, greater evenness and significantly greater or no difference in relative abundance of metal-sensitive taxa at CLT2 compared to the reference creek in 2021, 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 all dominant taxonomic groups, FFGs, and HPGs were indicated between study areas located downstream and upstream of the Tote Road crossing in 2021 (Table 3.4; Appendix Figure F.3). Therefore, no effects to benthic invertebrates were evident at CLT2 in 2021 as a result of potential influences associated with the Tote Road.

3.2.4 Effects Assessment and Recommendations

Water chemistry at CLT2 met all AEMP benchmarks in 2021. In addition, no adverse effects on phytoplankton or benthic invertebrates were indicated at CLT2 in 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 2.6). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2021, no adjustment to the existing AEMP need be applied at CLT2 as part of the 2022 CREMP.

3.3 Camp Lake (JL0)

3.3.1 Water Quality

In situ water quality profiles measured 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 2021 (Appendix Figures C.4 to C.7). The 2021 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, but was generally consistent from surface to bottom during summer and fall sampling events (Figure 3.5). The average temperature profiles at Camp Lake in summer differed slightly from the reference lake where a distinctly warmer surface layer extended to a depth of approximately 6 metres, but the average profile from the fall sampling event at Camp Lake roughly mirrored that of Reference Lake 3 in 2021 (Figure 3.5). Water temperature near the bottom of the water column at littoral stations did not differ significantly between lakes, but was significantly (albeit slightly) warmer at profundal stations of Camp Lake than at Reference Lake 3 during August 2021 biological monitoring (Figure 3.6; Appendix Tables C.23 and C.25).

Dissolved oxygen profiles conducted at Camp Lake in 2021 showed declining saturation levels with increased depth beginning at approximately 15 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). Dissolved oxygen at the bottom of the water column was near full saturation at littoral and profundal sampling depths of Camp Lake, and did not differ significantly from Reference Lake 3 at the time of biological sampling in August 2021 (Figure 3.6; Appendix Tables C.23 and 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 2021 except at water depths greater than approximately 27 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 27 m deep during the winter.

In situ profiles showed decreasing pH with increased depth at Camp Lake in winter and summer, and similarly at Reference Lake 3 in summer and fall, but unlike at the reference lake, higher pH occurred with increased depth at Camp Lake in the fall (Figure 3.5). Although pH near the bottom

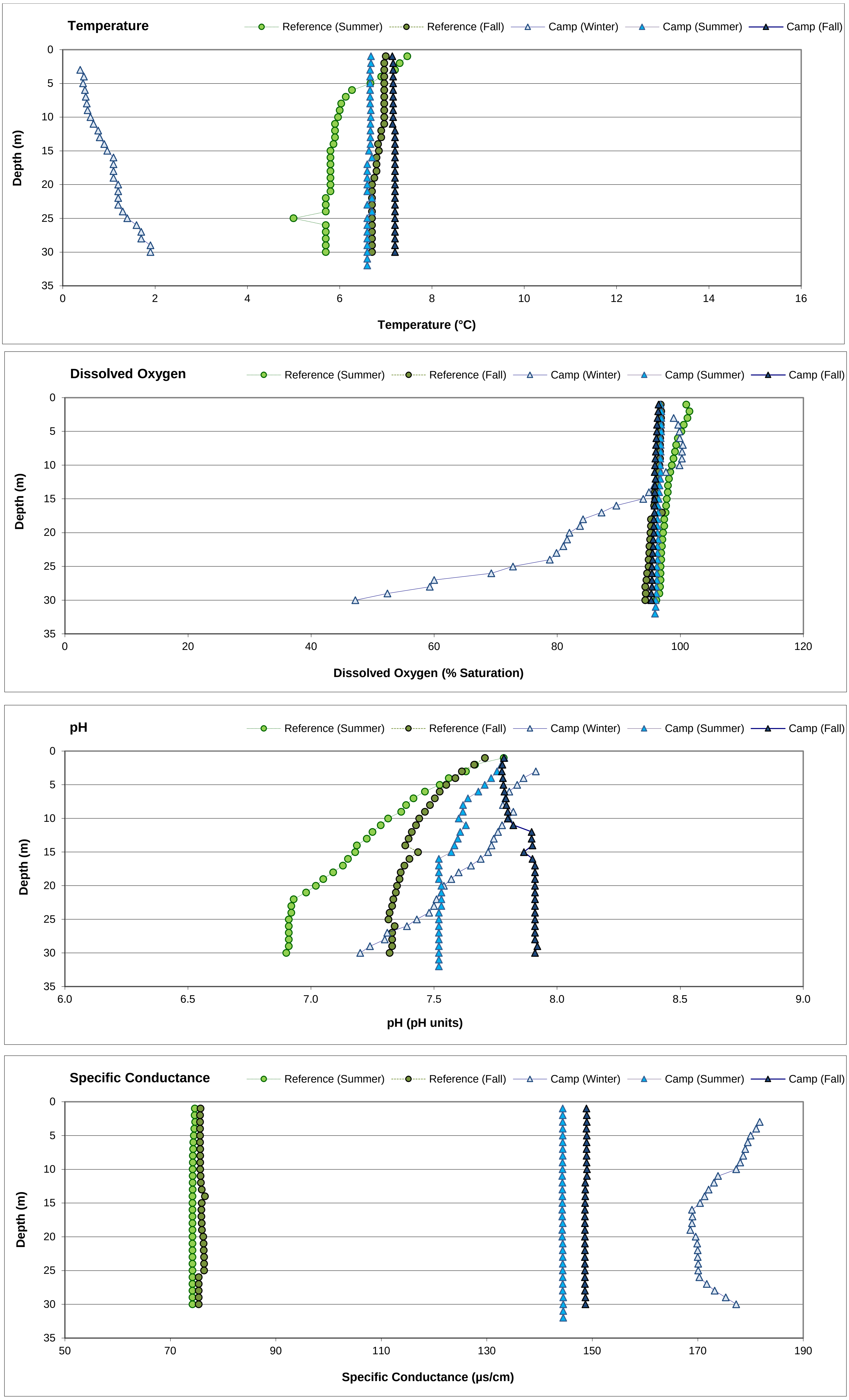


Figure 3.5: Average *In Situ* Water Quality with Depth from Surface at Camp Lake (JL0) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

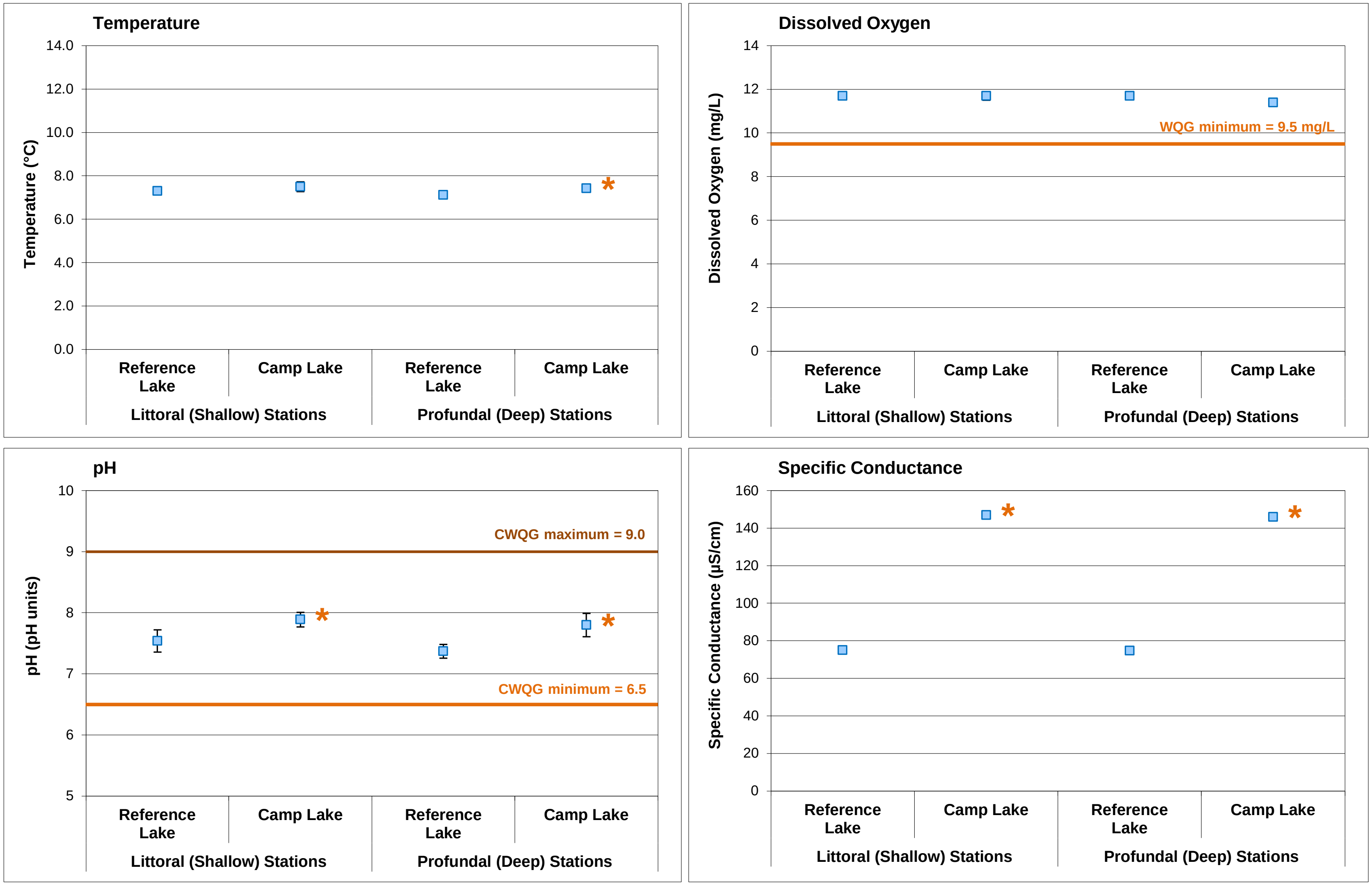


Figure 3.6: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Camp Lake (JL0) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2021

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.

at littoral and profundal stations of Camp Lake were significantly higher than at the reference lake during the August 2021 biological study, the mean incremental difference in pH between lakes was small (i.e., approximately 0.4 pH units) and all pH values were consistently within WQG limits (Figure 3.6, Appendix Table C.26). This suggested 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 2021 (Figure 3.5), the difference of which was shown to be significant during the August 2021 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 2021 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 2021 (Table 3.5; Appendix Table C.26).[11] Among those parameters with established AEMP benchmarks, aluminum, and chloride concentrations were slightly elevated (i.e., 3- to 5-times higher) in fall and summer at Camp Lake compared to the reference lake (Table 3.5; Appendix Table C.27). Of those parameters without AEMP benchmarks, only turbidity and concentrations of TDS and (total and dissolved) uranium were slightly elevated at Camp Lake compared to the reference lake during summer and/or fall sampling events in 2021 (Appendix Tables C.27 and C.29). Average concentrations of total aluminum, chloride, total manganese, sulphate, and total uranium were elevated at Camp Lake in at least one of the three seasons of 2021 compared to respective seasons during baseline, but concentrations of parameters with AEMP benchmarks consistently well below AEMP benchmarks since commercial mine operations commenced in 2015 (Appendix Figure C.9; Appendix Tables C.27 and C.29). Overall, comparisons to Reference Lake 3 water chemistry in 2021 and to Camp Lake baseline water chemistry suggested slightly elevated turbidity and concentrations of aluminum, chloride, manganese, TDS, and uranium at Camp Lake in 2021 which reflected a slight mine-related influence on water quality of the lake. However, because mean concentrations of all parameters remained well below AEMP benchmarks and WQG since commercial mine operations commenced in 2015, including in 2021, no adverse effects on biota were expected at Camp Lake.

[11] The reported concentration of iron at the Station JL0-10 (bottom of the water column) was above the AEMP benchmark during the winter 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 winter 2021 (Appendix Table C.26).

Table 3.5: Mean Water Chemistry at Camp Lake (JL0) and Reference Lake 3 (REF3) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

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	-	-	79.1	83.3	188	156	155
	pH (lab)	pH	6.5 - 9.0	-	7.71	7.68	7.77	8.05	8.05
	Hardness (as $CaCO_3$)	mg/L	-	-	36.8	37.7	86.1	72.0	70.1
	Total Suspended Solids (TSS)	mg/L	-	-	<2	<2	3.24	<2	<2
	Total Dissolved Solids (TDS)	mg/L	-	-	29.0	41.0	98.6	83.4	55.6
	Turbidity	NTU	-	-	0.240	0.183	0.110	0.986	0.676
	Alkalinity (as $CaCO_3$)	mg/L	-	-	37.3	46.3	88.4	58.1	67.6
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.0140	<0.01	0.0172	<0.01	<0.005
	Nitrate	mg/L	3	3	<0.02	<0.02	0.0416	0.0376	0.0310
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.005	<0.005	<0.001
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.177	0.200	0.102	0.116	0.104
	Dissolved Organic Carbon	mg/L	-	-	3.00	4.33	2.54	2.58	1.96
	Total Organic Carbon	mg/L	-	-	3.46	3.49	2.68	2.18	2.00
	Total Phosphorus	mg/L	0.020[α]	-	0.00443	0.00303	0.00416	0.00304	-
	Phenols	mg/L	0.004[α]	-	0.00113	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.05
	Chloride (Cl)	mg/L	120	120	1.32	1.35	5.79	4.62	4.57
	Sulphate (SO_4)	mg/L	218[β]	218	3.68	3.69	5.75	4.72	4.71
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.00373	0.00370	0.00318	0.0174	0.0192
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.00002
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.000104	<0.0001	0.0000590
	Barium (Ba)	mg/L	-	-	0.00676	0.00651	0.00938	0.00790	0.00745
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.000005
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.000005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.01	<0.005
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.00001	<0.00001	<0.00001	<0.00001	<0.000005
	Calcium (Ca)	mg/L	-	-	7.40	7.43	16.2	14.4	13.2
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	<0.0005	<0.0005	0.000106
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.0001	<0.0001	<0.0001	<0.0001	0.0000261
	Copper (Cu)	mg/L	0.002	0.0022	0.000810	0.000750	0.00105	0.000900	0.000936
	Iron (Fe)	mg/L	0.30	0.326	<0.03	<0.03	<0.03	<0.03	0.0229
	Lead (Pb)	mg/L	0.001	0.001	<0.00005	<0.00005	<0.00005	<0.00005	0.0000286
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	0.00138	0.00118
	Magnesium (Mg)	mg/L	-	-	4.67	4.77	10.9	8.87	8.98
	Manganese (Mn)	mg/L	0.935[β]	-	0.000674	0.000591	0.000561	0.00196	0.00118
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	0.00000500	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000155	0.000157	0.000467	0.000413	0.000415
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000748	0.000660	0.000681
	Potassium (K)	mg/L	-	-	0.960	0.910	1.62	1.39	1.41
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	<0.001	<0.001	0.0000406
	Silicon (Si)	mg/L	-	-	0.507	0.490	0.424	0.452	0.377
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.000005
	Sodium (Na)	mg/L	-	-	0.994	0.929	2.51	2.09	2.14
	Strontium (Sr)	mg/L	-	-	0.00877	0.00856	0.0140	0.0116	0.0115
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.0001	<0.0001	<0.000005
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.00002
	Titanium (Ti)	mg/L	-	-	<0.01	<0.01	<0.01	<0.01	0.00103
	Uranium (U)	mg/L	0.015	-	0.000328	0.000342	0.00166	0.00136	0.00139
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	<0.001	<0.001	0.0000812
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	<0.003	<0.003	0.000500

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.

3.3.2 Sediment Quality

Surficial sediment (i.e., top 2 cm) collected at the Camp Lake coring stations in 2021 was primarily composed of silt and sand with low TOC content (Figure 3.7; Appendix Table D.7). The proportion of clay in surficial sediment at littoral stations of Camp Lake was significantly higher than at Reference Lake 3, whereas all particle size fractions of sediment from profundal areas at Camp Lake did not differ significantly with those at the reference lake (Appendix Table D.8). While the TOC content in sediment at littoral and profundal stations at Camp Lake was significantly lower than at the reference lake, moisture content did not significantly differ (Figure 3.7; Appendix Table D.8). A distinctly visible reddish-orange to orange-brown layer of oxidized material (likely iron oxyhydroxides) was observed in the surficial and/or sub-surficial sediment sections at some stations at Camp Lake (Appendix Table D.6) which was similar to the layer of oxidized material observed in sediment at Reference Lake 3 (Appendix Table D.2). This suggested the presence of iron (oxy)hydroxides was a natural phenomenon, likely indicating oxic conditions at the surficial sediment layer (Rozan et al., 2002). The substrate from Camp Lake sediment coring stations in 2021 did not exhibit a detectable hydrogen sulphide odour or visible redox boundaries (Appendix Table D.5). In contrast, hydrogen sulphide odour was detected in two (REF-03-4; REF-03-5) of the 10 stations at the reference lake (Appendix Table D.1). Overall, sediment core observations suggested that the redox conditions in sediment at Camp Lake and Reference Lake 3 in 2021 were comparable (Appendix Tables D.1, D.2, D.5, and D.6).

Mean concentrations of metals including arsenic, copper, iron, manganese, molybdenum, nickel, and phosphorus were slightly higher in sediment at stations near the CLT1 inlet (e.g., JL0-14 and JL0-17) than in sediment at stations near the outlet of Camp Lake (Appendix Table D.7). No spatial changes in water chemistry were indicated within Camp Lake between the CLT1 inlet and the lake outlet (Section 3.3.1). Thus, the spatial differences in metal concentrations in sediment were potentially attributable to spatial differences in the particle size which included higher proportion of silt-sized material in sediment at stations closer to the CLT1 inlet (Appendix Table D.7). Metal concentrations in littoral and profundal sediments at Camp Lake were comparable (i.e., less than 3-times higher) to those at Reference Lake 3 in 2021 (Table 3.6; Appendix Table D.9). Mean metal concentrations in sediment of littoral and profundal sediment of Camp Lake were below SQG and AEMP benchmarks, except for average concentrations of manganese at profundal stations, which was above SQG but not the AEMP benchmark (Figure 3.8, Appendix Table D.7). Manganese concentrations in sediment may be naturally elevated in the study area, as indicated by manganese concentrations at the reference lake that were also frequently greater than the SQG (Appendix Table D.4). Concentrations of arsenic, copper, iron, nickel, and phosphorus were above respective Camp Lake AEMP benchmarks in sediment at some stations of Camp Lake, but on average, were below

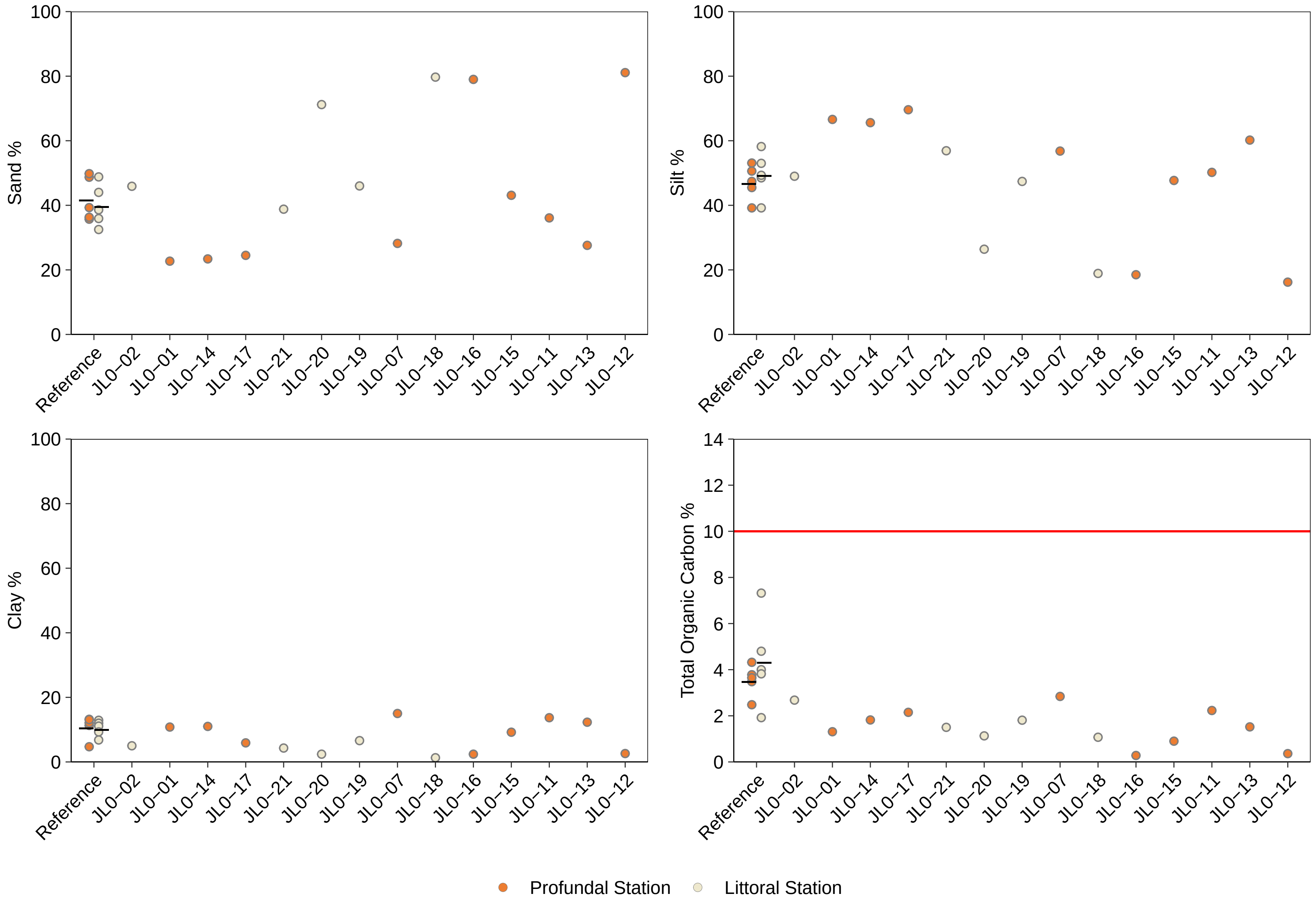


Figure 3.7: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Camp Lake (JL0) Sediment Monitoring Stations and to Reference Lake 3, Mary River Project CREMP, August 2021

Note: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

Table 3.6: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Camp Lake (JL0) and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2021

	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]	-	3.70 ± 1.09	2.68	4.21 ± 1.83	1.49 ± 0.87
Metals	Aluminum (Al)	mg/kg	-	-	21,120 ± 4,592	17,700	20,520 ± 4,129	15,138 ± 5,591
Metals	Antimony (Sb)	mg/kg	-	-	0.10 ± 0	0.100	0.10 ± 0	0.10 ± 0.01
Metals	Arsenic (As)	mg/kg	17	5.9	5.54 ± 1.03	4.45	5.13 ± 0.98	5.03 ± 3.24
Metals	Barium (Ba)	mg/kg	-	-	123 ± 29.4	94.0	129 ± 13.4	94.1 ± 69.5
Metals	Beryllium (Be)	mg/kg	-	-	0.812 ± 0.176	0.830	0.782 ± 0.122	0.800 ± 0.307
Metals	Bismuth (Bi)	mg/kg	-	-	0.20 ± 0	0.320	0.20 ± 0	0.27 ± 0.05
Metals	Boron (B)	mg/kg	-	-	14.9 ± 2.68	17.7	15.2 ± 2.23	19.5 ± 7.17
Metals	Cadmium (Cd)	mg/kg	3.5	1.5	0.144 ± 0.046	0.178	0.167 ± 0.048	0.168 ± 0.083
Metals	Calcium (Ca)	mg/kg	-	-	5,026 ± 849	4,540	5,186 ± 660	5,538 ± 4,085
Metals	Chromium (Cr)	mg/kg	90	98	67.2 ± 15.0	74.8	64.6 ± 10.7	65.9 ± 19.4
Metals	Cobalt (Co)	mg/kg	-	-	14.8 ± 3.35	16.4	13.8 ± 3.48	15.8 ± 6.15
Metals	Copper (Cu)	mg/kg	110[α]	50	***81.2*** ± 27.5	44.7	***85.0*** ± 10.5	40.5 ± 16.5
Metals	Iron (Fe)	mg/kg	40,000[α]	52,400	44,360 ± 8,848	39,800	40,900 ± 11,600	35,078 ± 15,580
Metals	Lead (Pb)	mg/kg	91	35	17.3 ± 2.81	18.8	16.2 ± 2.46	18.0 ± 7.48
Metals	Lithium (Li)	mg/kg	-	-	34.7 ± 5.36	28.0	33.3 ± 5.37	26.2 ± 9.91
Metals	Magnesium (Mg)	mg/kg	-	-	13,760 ± 2,748	15,300	12,918 ± 2,534	12,921 ± 3,547
Metals	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	958 ± 402	697	882 ± 346	2,358 ± 2,765
Metals	Mercury (Hg)	mg/kg	0.486	0.17	0.052 ± 0.024	0.035	0.045 ± 0.010	0.032 ± 0.020
Metals	Molybdenum (Mo)	mg/kg	-	-	3.23 ± 0.556	1.16	2.69 ± 0.733	1.37 ± 1.32
Metals	Nickel (Ni)	mg/kg	75[α,β]	72	46.6 ± 10.1	66.8	45.9 ± 6.89	61.2 ± 18.0
Metals	Phosphorus (P)	mg/kg	2,000[α]	1,580	955 ± 154	930	902 ± 111	1,080 ± 588
Metals	Potassium (K)	mg/kg	-	-	5,140 ± 946	4,250	4,862 ± 765	4,012 ± 1,495
Metals	Selenium (Se)	mg/kg	-	-	0.628 ± 0.252	0.370	0.554 ± 0.140	0.352 ± 0.151
Metals	Silver (Ag)	mg/kg	-	-	0.202 ± 0.077	0.110	0.172 ± 0.050	0.133 ± 0.042
Metals	Sodium (Na)	mg/kg	-	-	394 ± 80	188	362 ± 59	214 ± 112
Metals	Strontium (Sr)	mg/kg	-	-	13.5 ± 2.11	9.26	13.2 ± 1.41	12.6 ± 4.97
Metals	Sulphur (S)	mg/kg	-	-	1,340 ± 297	1,000	1,380 ± 444	1,311 ± 933
Metals	Thallium (Tl)	mg/kg	-	-	0.594 ± 0.205	0.457	0.556 ± 0.129	0.443 ± 0.204
Metals	Tin (Sn)	mg/kg	-	-	2.0 ± 0	2.0	2.0 ± 0	2.0 ± 0
Metals	Titanium (Ti)	mg/kg	-	-	1,204 ± 112	1,070	1,096 ± 158	824 ± 237
Metals	Uranium (U)	mg/kg	-	-	22.0 ± 8.39	5.69	20.3 ± 7.92	5.20 ± 2.53
Metals	Vanadium (V)	mg/kg	-	-	63.2 ± 11.0	62.3	60.7 ± 9.2	52.5 ± 17.6
Metals	Zinc (Zn)	mg/kg	315	135	88.0 ± 18.2	60.4	87.5 ± 11.8	50.0 ± 18.4
Metals	Zirconium (Zr)	mg/kg	-	-	5.16 ± 1.07	8.90	4.66 ± 1.28	5.92 ± 3.07

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.

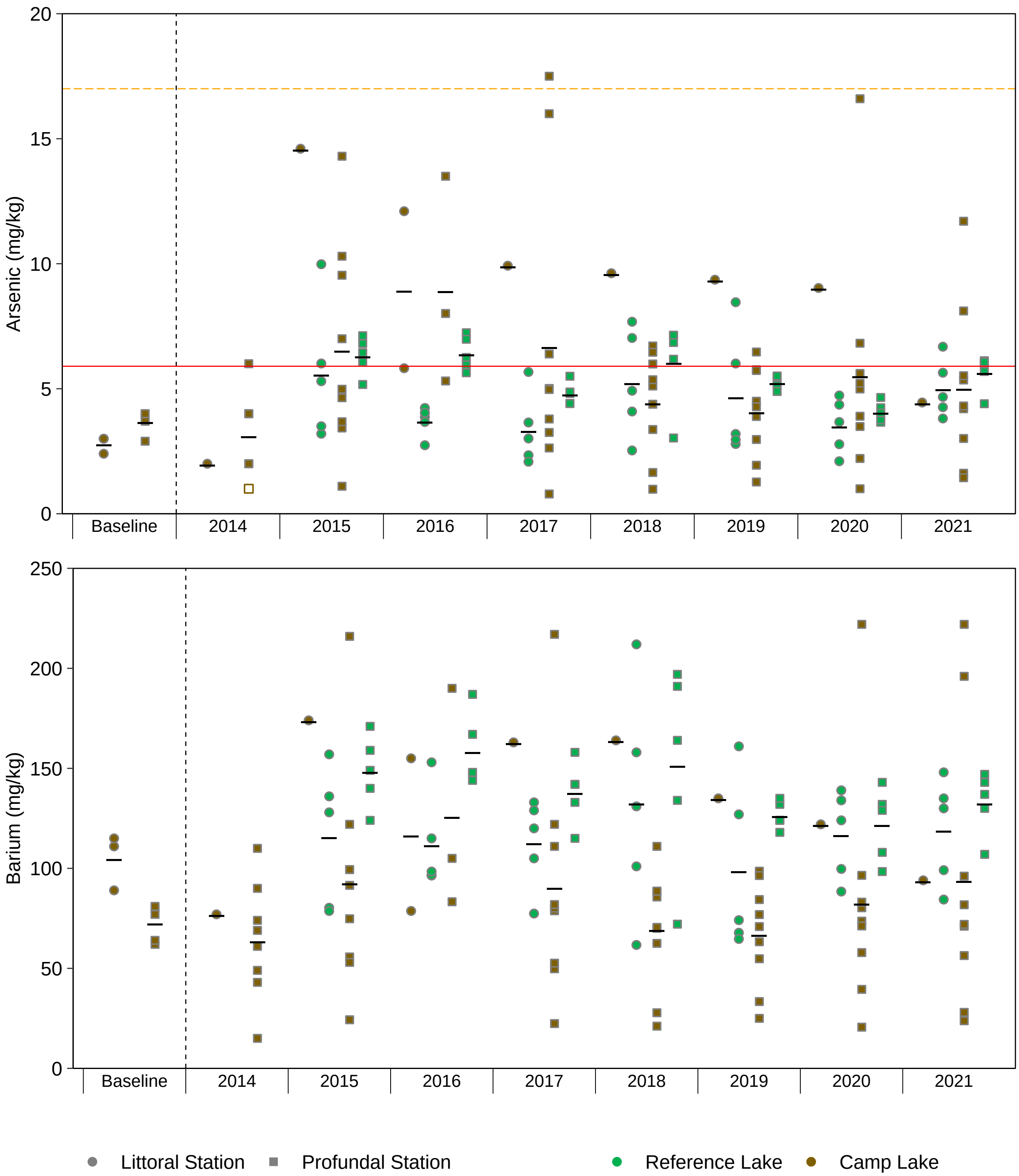
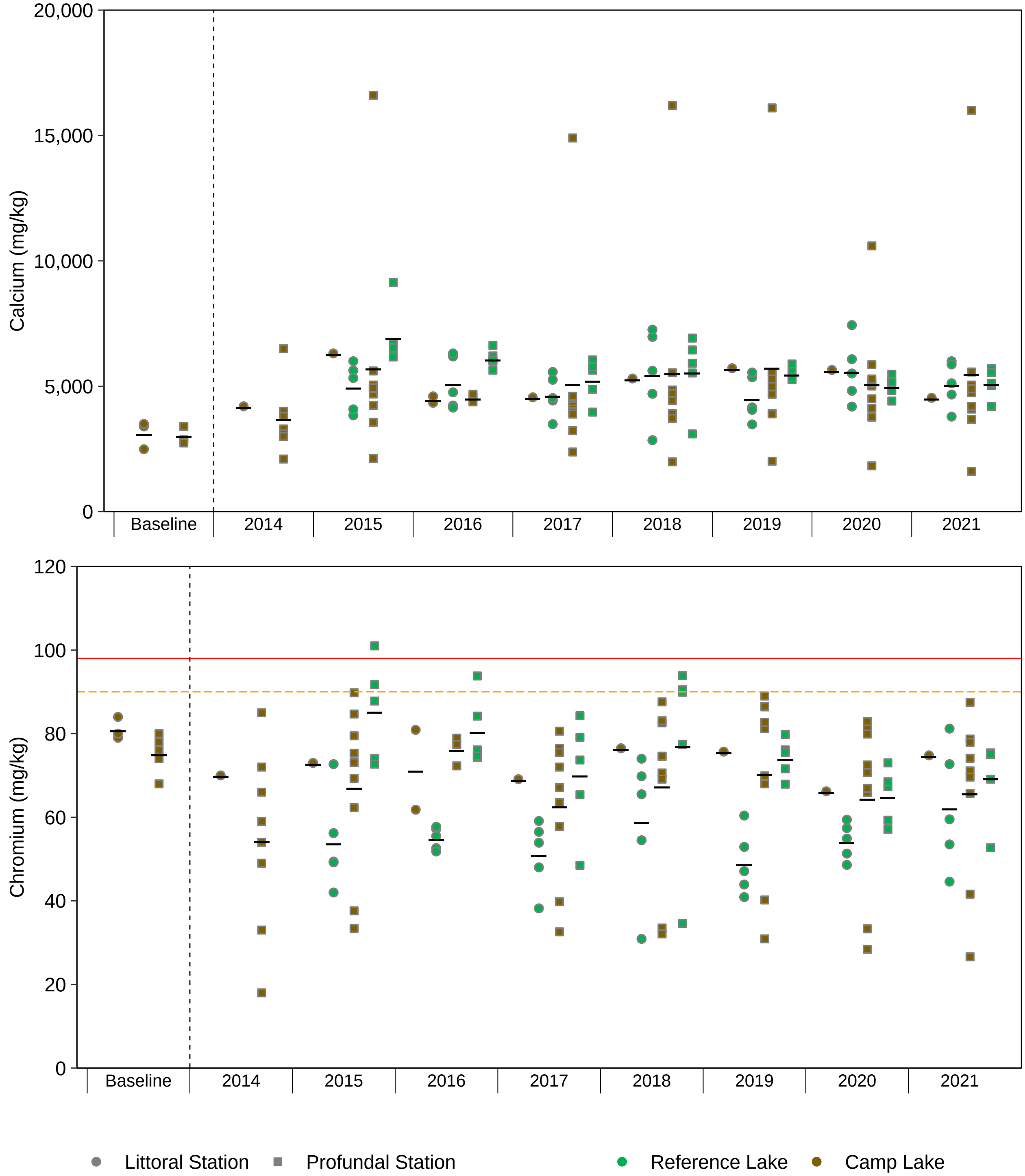


Figure 3.8: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Camp Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.



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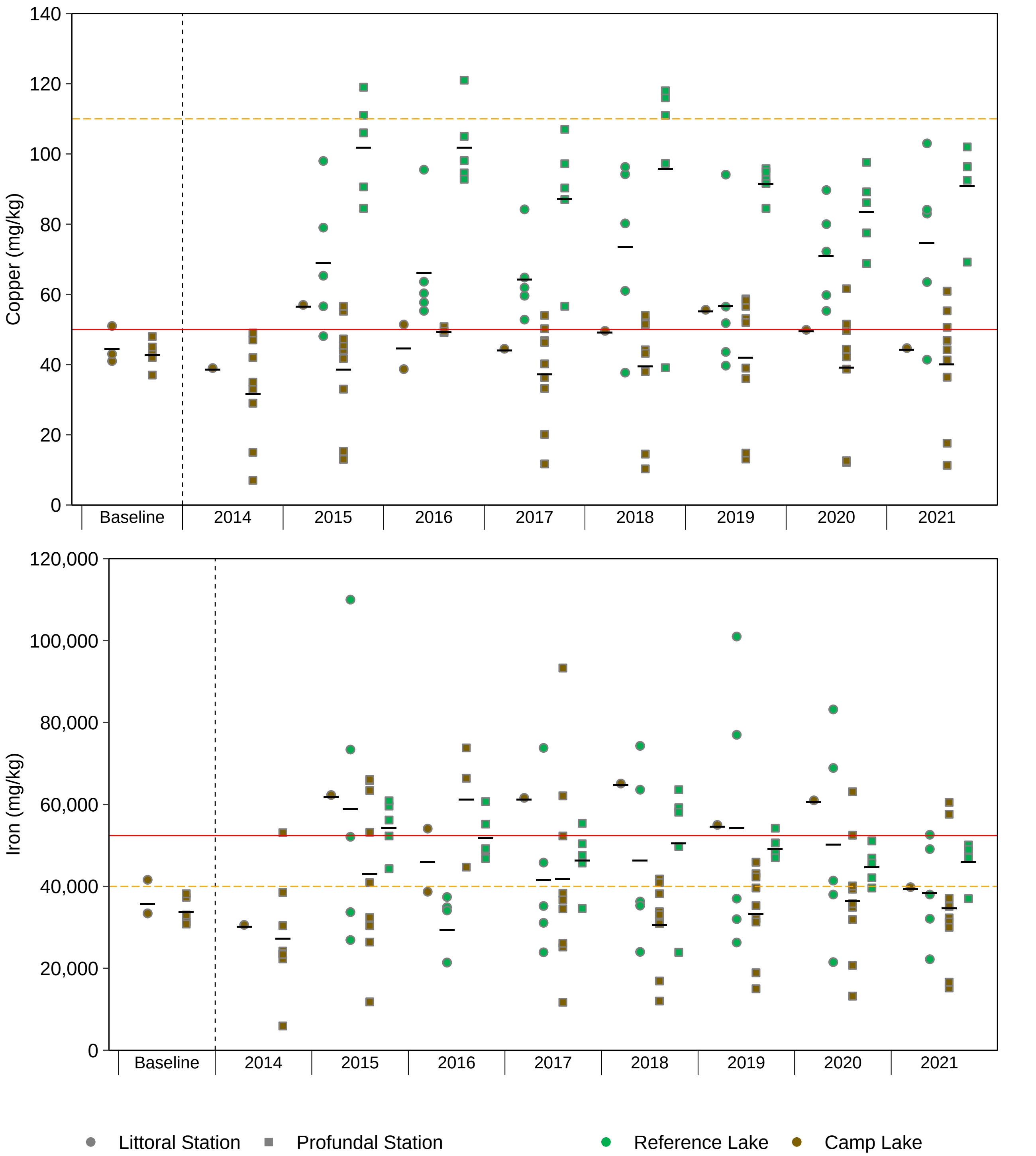

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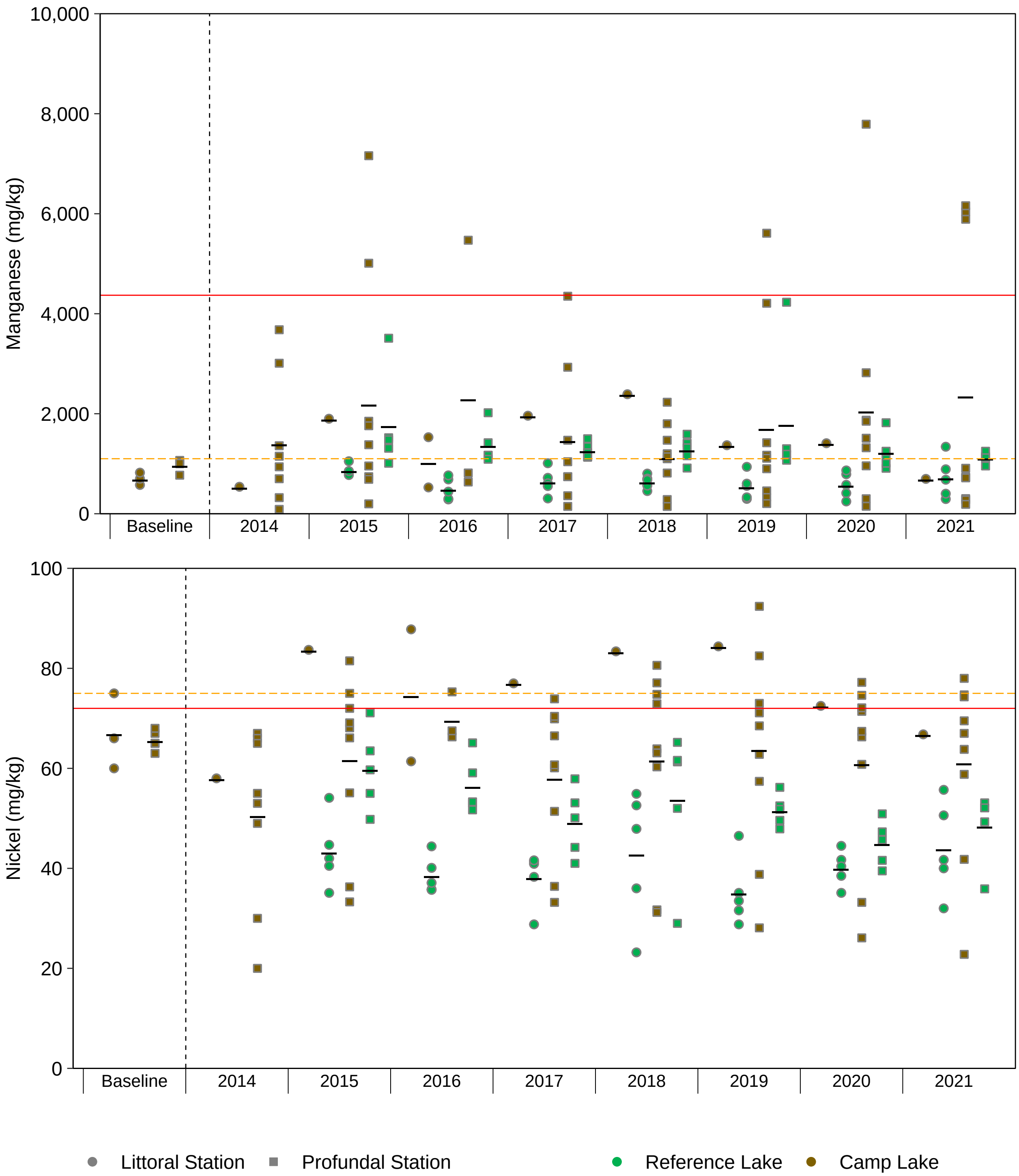

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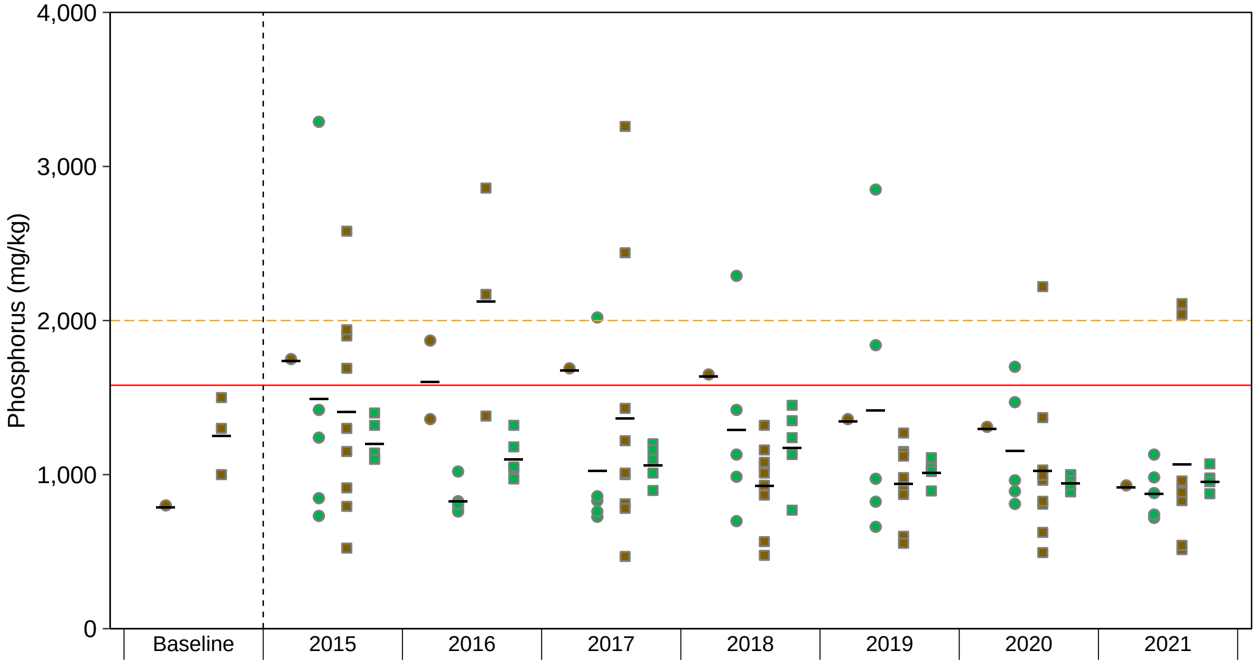


Littoral Station Profundal Station Reference Lake Camp Lake

Figure 3.8: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Camp Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

the applicable benchmarks (Table 3.6; Appendix Table D.7). Of these metals, the average concentrations of copper and iron were also above the Camp Lake AEMP benchmark in sediment at Reference Lake 3 (Table 3.6), providing further support for naturally elevated concentrations of iron and copper in sediment.

Mean metal concentrations in sediment collected from Camp Lake littoral and profundal stations in 2021 were comparable to concentrations measured during the baseline period (2005 to 2013; Figure 3.8, Appendix Table D.9).[12] Metal concentrations in sediment from Camp Lake littoral and profundal stations in 2021 were typically within the ranges observed from 2015 to 2020 (Figure 3.8). In addition, except for slightly higher mean concentrations of calcium, manganese, nickel, and phosphorus compared to baseline, there was no evidence showing an increasing trend in concentrations of metals in Camp Lake sediments over the mine operation period (2015 to 2021) in comparison to the baseline period (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 2021 (Figure 3.9). Chlorophyll-a concentrations were significantly lower in winter compared to summer and fall at Camp Lake in 2021 (Figure 3.9; Appendix Table E.6). Chlorophyll-a concentrations at Camp Lake were significantly higher compared to those at Reference Lake 3 in both the summer and fall sampling events (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 2021 (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 2021) categorization for Camp Lake based on mean aqueous total phosphorus concentrations below 10 µg/L for all seasonal sampling events (Table 3.5; Appendix Table C.26).

Temporal comparisons of the Camp Lake chlorophyll-a data did not indicate any consistent significant differences among years of mine construction (2014) and mine operation

[12] Boron concentrations in sediment from 2015 to 2021 were considerably higher (i.e., 10- to 70-times) 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.

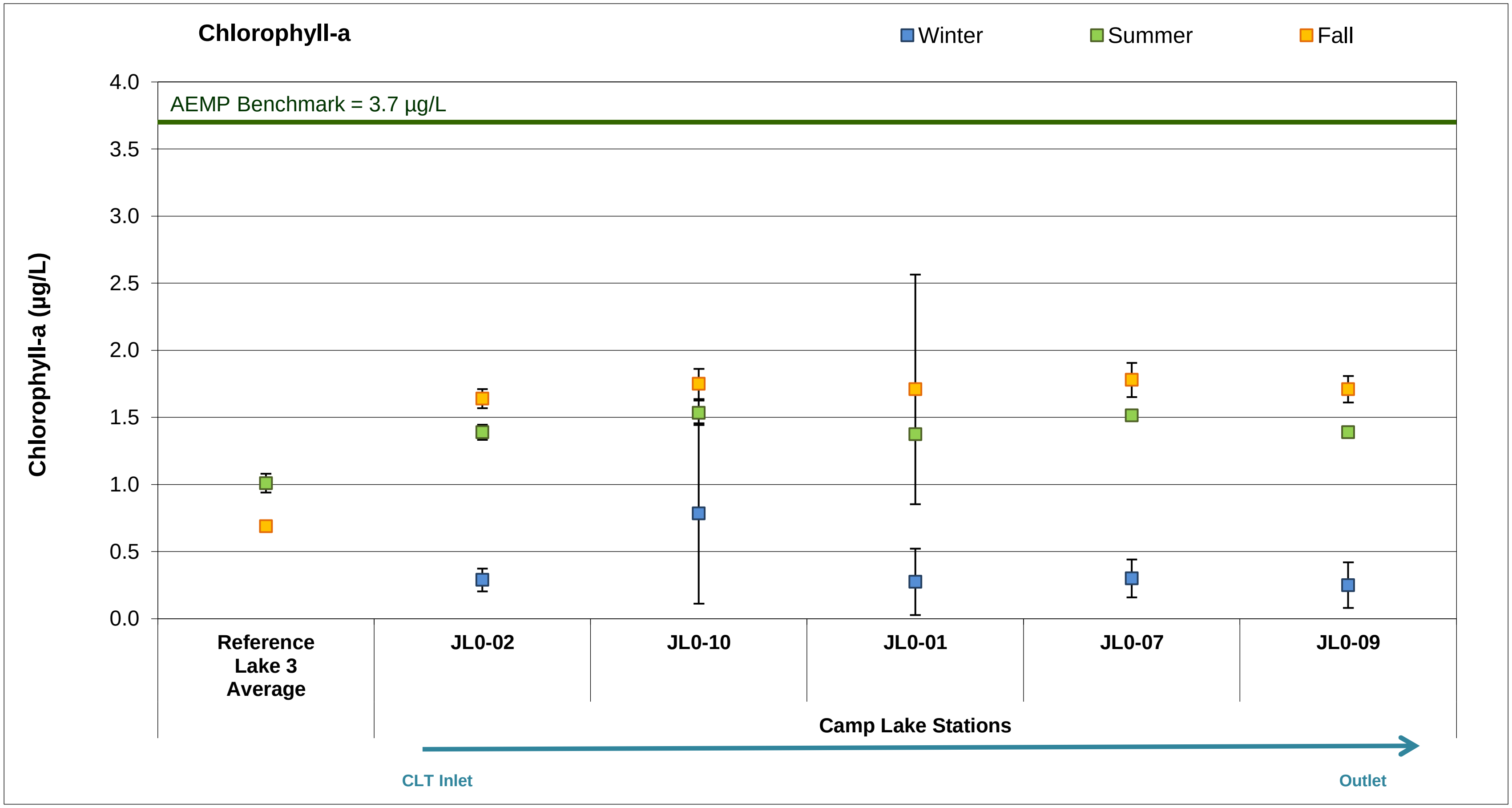


Figure 3.9: Chlorophyll a Concentrations at Camp Lake (JL0) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2021

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 = 6). Reference Lake 3 was not sampled in winter 2021.

(2015 to 2021) 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 seven 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 habitats of Camp Lake compared to like-habitat stations at Reference Lake 3 (Tables 3.7 and 3.8). For both habitat types, the difference was ecologically significant based on the magnitude being outside of the CES_{BIC} of ±2 SD_{REF}. Significantly higher richness was indicated at profundal habitat of Camp Lake compared to the reference lake; however, the absolute magnitude of this difference was <2 SD (i.e., not ecologically significant), and there was no difference at littoral habitat (Table 3.8). No significant differences in evenness were identified between Camp Lake and the reference lake for littoral habitat and profundal habitat (Tables 3.7 and 3.8). 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 identified between Camp Lake and Reference Lake 3 (Tables 3.7 and 3.8), which was consistent with metal concentrations in water and sediment of Camp Lake generally below applicable guidelines (Tables 3.5 and 3.6).[13] 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 (Table 3.7). The only ecologically significant difference in FFGs identified between Camp Lake and the reference lake was lower relative abundance of collector-gathers in littoral habitats at Camp Lake (Tables 3.7 and 3.8), which suggested a similar food resource base for benthic invertebrates between lakes. There was no ecologically significant difference in HPG at littoral stations of Camp Lake compared to Reference Lake 3 (Table 3.7), whereas the relative abundance of sprawler and burrower HPG at profundal stations of Camp Lake were significantly less than and greater than those at Reference Lake 3, respectively (Table 3.8). Overall, significantly higher benthic

[13] Although mean concentrations of manganese in sediment were above SQG at Camp Lake, the concentrations were variable among stations, with concentrations at three stations above the SQG and concentrations at the other seven stations an order of magnitude lower than the SQG (Appendix Table D.7).

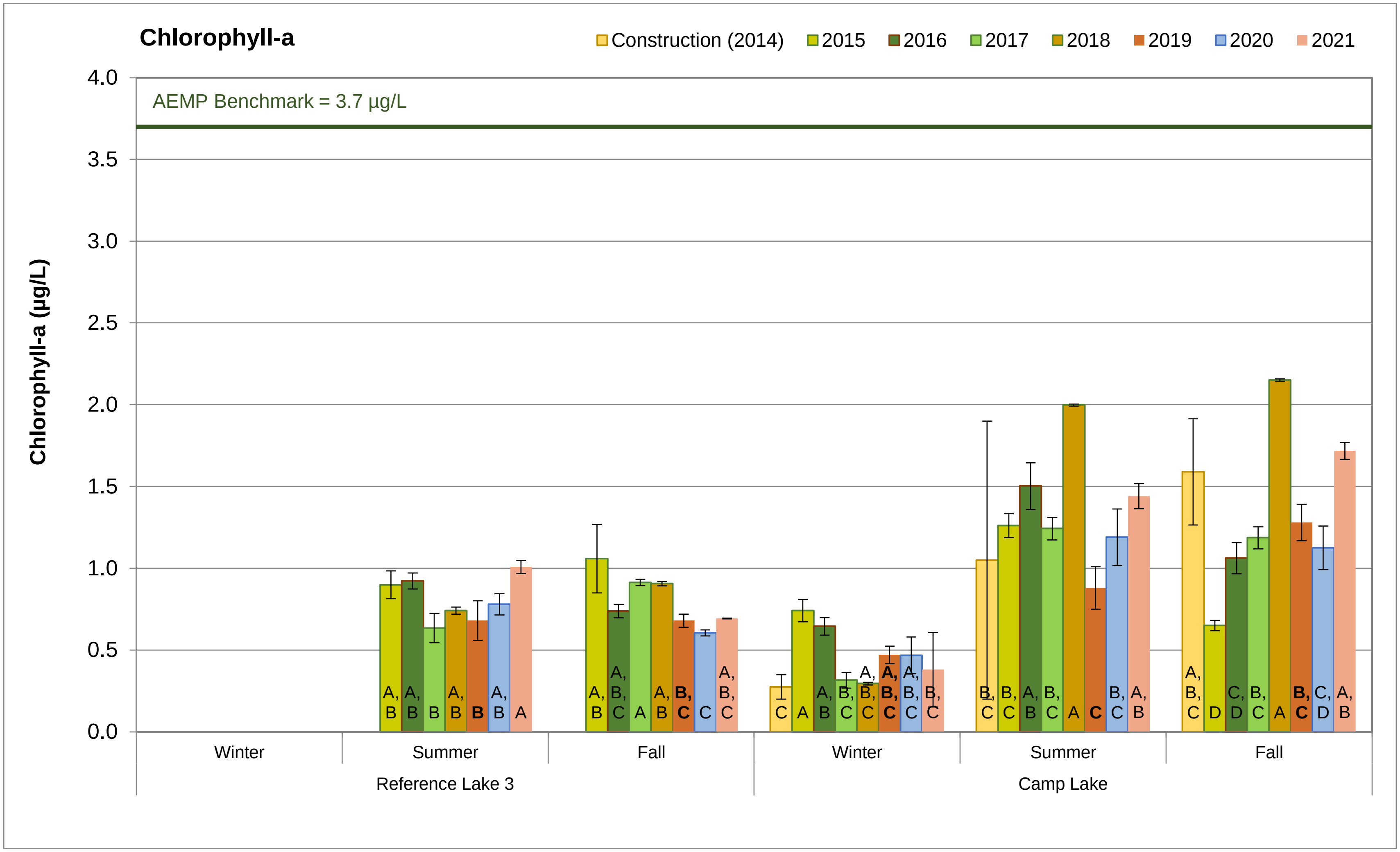


Figure 3.10: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Camp Lake and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2021) Periods (mean ± SE)

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

Table 3.7: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JL0) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	log10	YES	<0.001	7.4	Reference Lake 3	1,064	448	200	508	1,008	1,516
						Camp Lake Littoral	4,373	994	444	3,134	4,891	5,330
Richness (Number of Taxa)	tequal	log10	NO	0.221	0.9	Reference Lake 3	10.8	2.17	0.97	8	11	14
						Camp Lake Littoral	12.8	2.68	1.2	10	13	17
Simpson's Evenness (E)	tequal	none	NO	0.377	0.6	Reference Lake 3	0.703	0.14	0.0626	0.502	0.775	0.838
						Camp Lake Littoral	0.786	0.143	0.0638	0.55	0.816	0.91
Shannon Diversity	tequal	none	NO	0.195	1.0	Reference Lake 3	2.13	0.409	0.183	1.58	2.38	2.46
						Camp Lake Littoral	2.55	0.521	0.233	1.74	2.57	3.03
Hydracarina (%)	tequal	none	NO	0.161	-0.8	Reference Lake 3	3.12	2.15	0.96	0	3.39	5.68
						Camp Lake Littoral	1.32	1.48	0.663	0	0.99	3.42
Ostracoda (%)	M-W	rank	YES	0.056	-1.6	Reference Lake 3	46.6	27.9	12.5	1.69	47.7	72.7
						Camp Lake Littoral	3.24	4.02	1.8	0	2.2	10.2
Chironomidae (%)	tequal	none	YES	<0.001	4.2	Reference Lake 3	38.3	13.2	5.89	20.4	39.3	54.2
						Camp Lake Littoral	93.2	3.76	1.68	86.6	95	95.6
Metal-Sensitive Chironomidae (%)	tequal	log10	NO	0.857	0.6	Reference Lake 3	17.4	10.6	4.73	8.36	11.1	29.1
						Camp Lake Littoral	23.5	22.4	10	2.33	15.6	51.1
Collector-Gatherers (%)	tequal	log10	YES	0.098	-2.1	Reference Lake 3	82	9.73	4.35	66.7	87.2	90
						Camp Lake Littoral	62	21.1	9.43	35.5	66.8	88.5
Filterers (%)	tequal	none	NO	0.334	1.2	Reference Lake 3	11.8	9.47	4.23	2.22	7.95	25.6
						Camp Lake Littoral	22.7	21.8	9.74	2.33	15.6	50.4
Shredders (%)	M-W	rank	NO	0.116	-0.7	Reference Lake 3	2.8	3.0	1.4	0.0	1.7	7.8
						Camp Lake Littoral	0.5	0.3	0.1	0.0	0.7	0.8
Clingers (%)	tequal	log10	NO	0.631	-0.1	Reference Lake 3	14.9	10.6	4.73	2.22	11.7	29.9
						Camp Lake Littoral	13.6	12.4	5.57	0.704	9.58	29.4
Sprawlers (%)	tequal	none	YES	0.098	-1.0	Reference Lake 3	64.8	22.8	10.2	30.5	71.6	85.6
						Camp Lake Littoral	42.4	13.8	6.16	23.2	40.1	60.9
Burrowers (%)	tequal	log10	YES	0.072	1.4	Reference Lake 3	20.4	16.5	7.4	7.39	16.2	49.1
						Camp Lake Littoral	44	21.8	9.77	15.2	50.7	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.

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases.

Table 3.8: Benthic Invertebrate Community Statistical Comparison Results between Camp Lake (JL0) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test[a]	Data Transformation	Significant Difference Between	p-value	Magnitude of Difference (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	YES	0.005	5.0	Reference Lake 3	327	151	67.4	60.3	396	422
						Camp Lake Profundal	1,083	416	186	646	1,231	1,576
Richness (Number of Taxa)	tequal	none	YES	0.047	1.7	Reference Lake 3	6.4	2.6	1.2	2.0	8.0	8.0
						Camp Lake Profundal	10.8	3.3	1.5	7.0	10.0	16.0
Simpson's Evenness (E)	tequal	log10	NO	0.144	1.3	Reference Lake 3	0.545	0.113	0.050	0.437	0.510	0.731
						Camp Lake Profundal	0.695	0.180	0.080	0.521	0.608	0.911
Shannon Diversity	tequal	log10	NO	0.101	1.6	Reference Lake 3	1.38	0.51	0.23	0.59	1.46	2.00
						Camp Lake Profundal	2.17	0.78	0.35	1.50	1.70	3.16
Hydracarina (%)	tequal	log10(x+1)	NO	0.254	-0.6	Reference Lake 3	5.3	4.3	1.9	0.0	4.4	11.9
						Camp Lake Profundal	2.9	1.2	0.5	1.3	2.7	4.2
Ostracoda (%)	M-W	rank	NO	0.753	-0.2	Reference Lake 3	8.2	8.7	3.9	0.0	4.4	20.4
						Camp Lake Profundal	6.5	8.0	3.6	0.0	1.3	15.8
Chironomidae (%)	tequal	none	NO	0.904	0.1	Reference Lake 3	86.0	8.7	3.9	71.4	88.1	93.5
						Camp Lake Profundal	86.7	8.4	3.7	76.5	86.8	95.1
Metal-Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.283	2.4	Reference Lake 3	7.8	6.2	2.8	0.0	10.9	14.3
						Camp Lake Profundal	22.4	27.8	12.4	2.2	14.5	70.2
Collector-Gatherers (%)	M-W	rank	NO	0.421	-1.4	Reference Lake 3	86.3	7.8	3.5	80.8	83.7	100.0
						Camp Lake Profundal	75.1	28.2	12.6	26.3	80.6	96.7
Filterers (%)	M-W	rank	NO	0.833	2.4	Reference Lake 3	6.5	4.9	2.2	0.0	8.9	10.9
						Camp Lake Profundal	18.1	28.5	12.7	0.0	10.1	68.0
Shredders (%)	M-W	rank	NO	1.0	-0.2	Reference Lake 3	0.9	1.2	0.5	0.0	0.0	2.2
						Camp Lake Profundal	0.7	0.7	0.3	0.0	0.6	1.5
Clingers (%)	tequal	log10(x+1)	NO	0.318	2.2	Reference Lake 3	12.3	7.0	3.1	0.0	15.2	16.8
						Camp Lake Profundal	28.0	29.9	13.4	2.2	17.2	72.2
Sprawlers (%)	tequal	none	YES	0.061	-4.1	Reference Lake 3	86.0	8.4	3.7	77.5	84.5	100.0
						Camp Lake Profundal	51.5	34.4	15.4	18.3	36.1	89.0
Burrowers (%)	M-W	rank	YES	0.011	7.1	Reference Lake 3	1.7	2.7	1.2	0.0	0.0	6.1
						Camp Lake Profundal	20.5	20.8	9.3	8.0	9.5	56.7

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.

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - $MCT_{Ref)}$/SD_{Ref}. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD for rank-transformed data and as back-transformed mean and SD for all other cases.

[a] Contrast MODs could not be calculated because the MAD (Median Absolute Deviation) =0.

invertebrate density without accompanying ecologically significant differences in richness, evenness, and relative abundance of most FFGs at Camp Lake compared to Reference Lake 3 suggested that Camp Lake was more biologically productive.

No consistent ecologically 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 2021) compared to one or both years of baseline (2007, 2013; Appendix Tables F.22 and F.23; Appendix Figures F.5 and F.6). Similarly, the relative abundance of benthic invertebrate dominant taxonomic groups and FFGs over years of mine operation from 2015 to 2021 did not differ significantly from baseline for littoral habitat at Camp Lake (Appendix Table F.22). At profundal habitat of Camp Lake, the relative abundance of metal-sensitive chironomids, the collector gatherer FFG, 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; however, differences from the 2013 baseline data were within the CES_{BIC} and therefore likely not ecologically significant (Appendix Table F.23).

3.3.5 Fish Population

3.3.5.1 Camp Lake Fish Community

Arctic charr (*Salvelinus alpinus*) was the only fish species captured at Camp Lake in 2021 (Table 3.9). Ninespine stickleback (*Pungitius pungitius*) have been captured previously in low abundance at Camp Lake (Minnow 2020) but were not encountered during shoreline electrofishing in 2021. Higher CPUE for arctic charr occurred at Camp Lake compared to Reference Lake 3 suggesting greater density of this species at Camp Lake (Table 3.9). 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 2021 was within the range observed during baseline studies (2007 to 2013) and higher than the range observed over the six previous years (2015 to 2020) since mine operation commenced (Figure 3.11). The arctic charr CPUE associated with gill netting at Camp Lake in 2021 was substantially greater than baseline and mine operation years from 2015 to 2019, but comparable to 2020 (Figure 3.12). An increase in the gill netting CPUE (two- to four-times greater than 2019) was also observed at Reference Lake 3 in 2020 and 2021 (Figure 3.11). Higher gill netting CPUE in 2020 and 2021 at both Camp Lake and Reference Lake 3 compared to previous studies likely reflected sampling having been conducted slightly earlier in 2020 and 2021 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 2021 and baseline, and between 2021 and

Table 3.9: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Camp Lake (JL0) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	115	4	119	2
		CPUE	1.07	0.04	1.11	
	Gill netting	No. Caught	56	-	56	
		CPUE	1.65	-	1.65	
Camp Lake	Electrofishing	No. Caught	117	-	117	2
		CPUE	10.99	-	10.99	
	Gill netting	No. Caught	117	-	117	
		CPUE	9.68	-	9.68	

Note: "-" indicates not applicable.

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

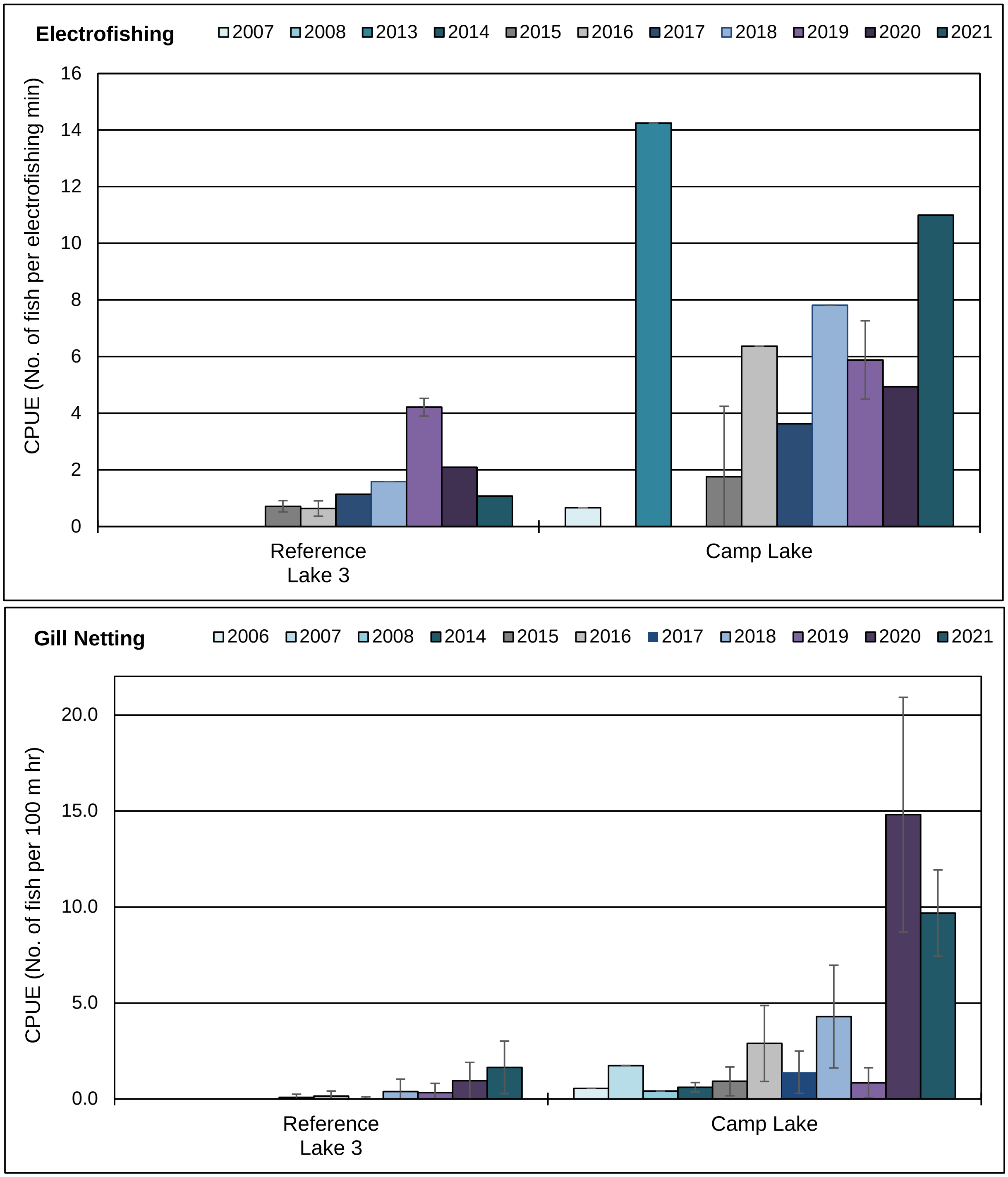


Figure 3.11: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JL0) and Reference Lake 3 (REF3), Mary River Project CREMP, 2006 to 2021

Note: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015 to 2021) mine phases.

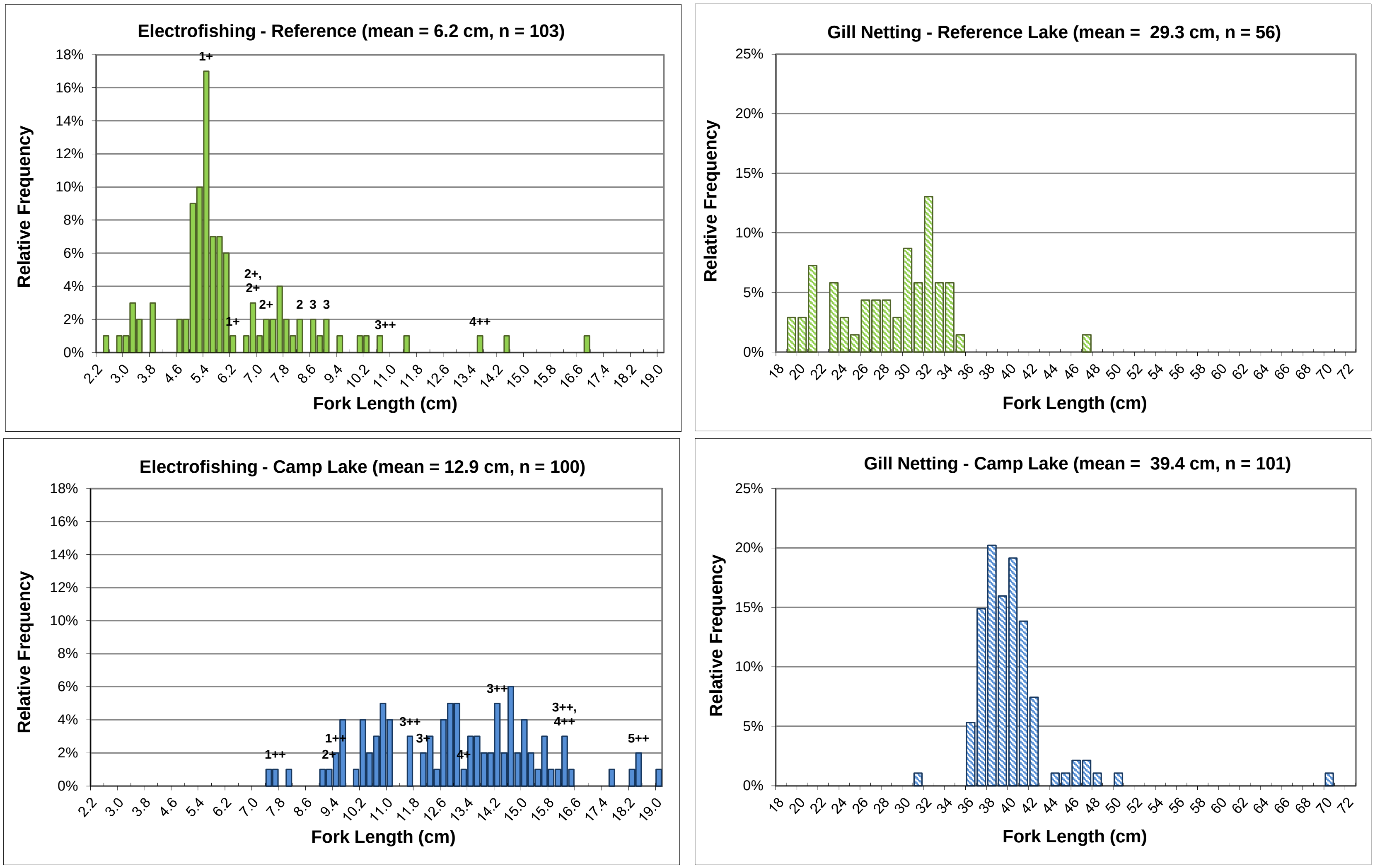


Figure 3.12: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JL0) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

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

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 2021.

3.3.5.2 Camp Lake Fish Health Assessment

Nearshore Arctic Charr

A total of 117 and 115 arctic charr were sampled from the nearshore habitat of Camp Lake and Reference Lake 3, respectively, in August 2021 (Table 3.9). Arctic charr YOY were distinguished from older (non-YOY) age classes at both lakes using a fork length of 4.0 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 2021b). As no YOY were captured in Camp Lake, 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 3.10; Appendix Table G.6) based on no YOY and fewer smaller-sized individuals captured at Camp Lake (Figure 3.12). Non-YOY arctic charr from Camp Lake were significantly longer (134%) and heavier (1,103%) than those from Reference Lake 3 (Table 3.10; 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 less than the CES (referred to herein as CES_C), suggesting that this difference was not ecologically significant (Table 3.10; 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 2021, indicating consistent presence of larger juveniles at Camp Lake (Table 3.10). 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 2021 (Table 3.10) 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 3.10). Like most previous years of mine operation, non-YOY arctic charr from Camp Lake were significantly shorter, lighter, and had lower condition in 2021 than during baseline (Table 3.10; 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 below 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.

Table 3.10: Summary of Statistical Results for Arctic Charr Population Comparisons between Camp Lake and Reference Lake 3 from 2015 to 2021, 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	2021	2015	2016	2017	2018	2019	2020	2021
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	**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 (+134%)**	**Yes (-15%)**	**Yes (-32%)**	**Yes (-35%)**	**Yes (-28%)**	No	**Yes (-22%)**	**Yes (+9.6%)**
		Size (mean weight)	**Yes (+176%)**	No	**Yes (+51%)**	**Yes (+135%)**	**Yes (+29%)**	**Yes (+44%)**	**Yes (+1,103%)**	**Yes (-42%)**	**Yes (-71%)**	**Yes (-74%)**	**Yes (-56%)**	No	**Yes (-52%)**	**Yes (+38%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	No	**Yes (-6%)**	No	**Yes (-14%)**	**Yes (-7%)**	**Yes (+7%)**	**Yes (-7.4%)**	**Yes (-6%)**	**Yes (-10%)**	**Yes (-10%)**	**Yes (-9%)**	**Yes (-11%)**	No	**Yes (-3.5%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**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 (+33%)**	**Yes (+6%)**	No	**Yes (+12%)**	**Yes (+15%)**	**Yes (+17%)**	**Yes (+19%)**	**Yes (+21%)**
		Size (mean weight)	-	-	-	**Yes (+46%)**	**Yes (+130%)**	**Yes (+129%)**	**Yes (+180%)**	No	No	**Yes (+37%)**	**Yes (+46%)**	**Yes (+44%)**	**Yes (+47%)**	**Yes (+52%)**
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+12%)**	**Yes (+6%)**	**Yes (+18%)**	**Yes (+34%)**	No	**Yes (-3%)**	No	No	No	No	**Yes (-4.4%)**

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.

Littoral/Profundal Arctic Charr

A total of 117 and 56 arctic charr were sampled from littoral/profundal habitat of Camp Lake and Reference Lake 3, respectively, in August 2021 (Table 3.9). 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 3.10; Figure 3.12). Littoral/profundal arctic charr from Camp Lake were significantly longer (33%) and heavier (180%) and had greater body condition (34%) than those captured at the reference lake (Table 3.10; 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 three previous years (Table 3.10), 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 2021 and the combined baseline data set (i.e., 2006, 2007, and 2008 studies; Table 3.10). Although fork length, body weight, and condition were significantly greater for littoral/profundal arctic charr captured at Camp Lake in 2021 compared to the baseline period, the absolute magnitude of difference in condition was less than the CES_C of 10%, suggesting the difference was not ecologically significant (Table 3.10), and other than 2021, few significant differences in body condition have been indicated since 2015 (Table 3.10). Overall, littoral/profundal arctic charr at Camp Lake were consistently larger with greater condition during mine operational years compared to the reference lake, and were consistently larger but showed no ecologically significant difference in condition compared to baseline, suggesting 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 2021:

- Arsenic, copper, iron, manganese, nickel, and phosphorus concentrations in sediment were above respective benchmarks at individual stations (two or three stations per parameter), but on average were below these benchmarks among the Camp Lake profundal stations.

At profundal habitat of Camp Lake in 2021, 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. No AEMP water quality benchmarks were exceeded at Camp Lake during spring, summer, or fall sampling events in 2021.[14] In addition, no adverse effects on phytoplankton, benthic invertebrates, nor on fish (arctic charr) health were indicated at Camp Lake in 2021 based on comparisons to reference lake conditions and to Camp Lake baseline data. Because no mine-related changes in concentrations of parameters with established AEMP benchmarks occurred relative to background or baseline and no adverse biological effects were indicated at Camp Lake in 2021, no adjustment to the existing AEMP for Camp Lake sampling is required for the 2022 CREMP. Nevertheless, 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 the ability to draw linkages 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 CREMP to evaluate mine-related effects to biota and potentially allow linkages to be determined between metal concentrations in sediment and benthic invertebrate responses.

[14] The reported concentration of iron at the Station JL0-10 (bottom of the water column) was above the AEMP benchmark during the winter 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 winter 2021 (Appendix Table C.26).

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 2021 (Appendix Tables C.1 to C.3). Dissolved oxygen concentrations at Sheardown Lake Tributary 1 (SDLT1) and Sheardown Lake Tributary 12 (SDLT12) did not differ significantly from those at Unnamed Reference Creek during the August 2021 biological study (Figure 4.1). Although dissolved oxygen concentrations were significantly lower at Sheardown Lake Tributary 9 (SDLT9) than at Unnamed Reference Creek, the dissolved oxygen concentrations at SDLT9, 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 2021 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 2021 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 2021 biological study (Figure 4.1; Appendix Tables C.30 to 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 under AEMP Revision 1 (Baffinland 2015), 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). As a special investigation into elevated copper concentrations at SDLT1, water quality sampling for SDLT1 was spatially expanded in fall 2021 (see Section 2.5.2). Also in fall 2021, additional routine water quality monitoring was added at SDLT12 and SDLT9 to provide water chemistry data to support the interpretation of biological data (see Section 2.5.2, Figure 2.2). Water chemistry of SDLT1 met AEMP benchmarks and WQG in spring, summer, and fall sampling events of 2021 except for 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,

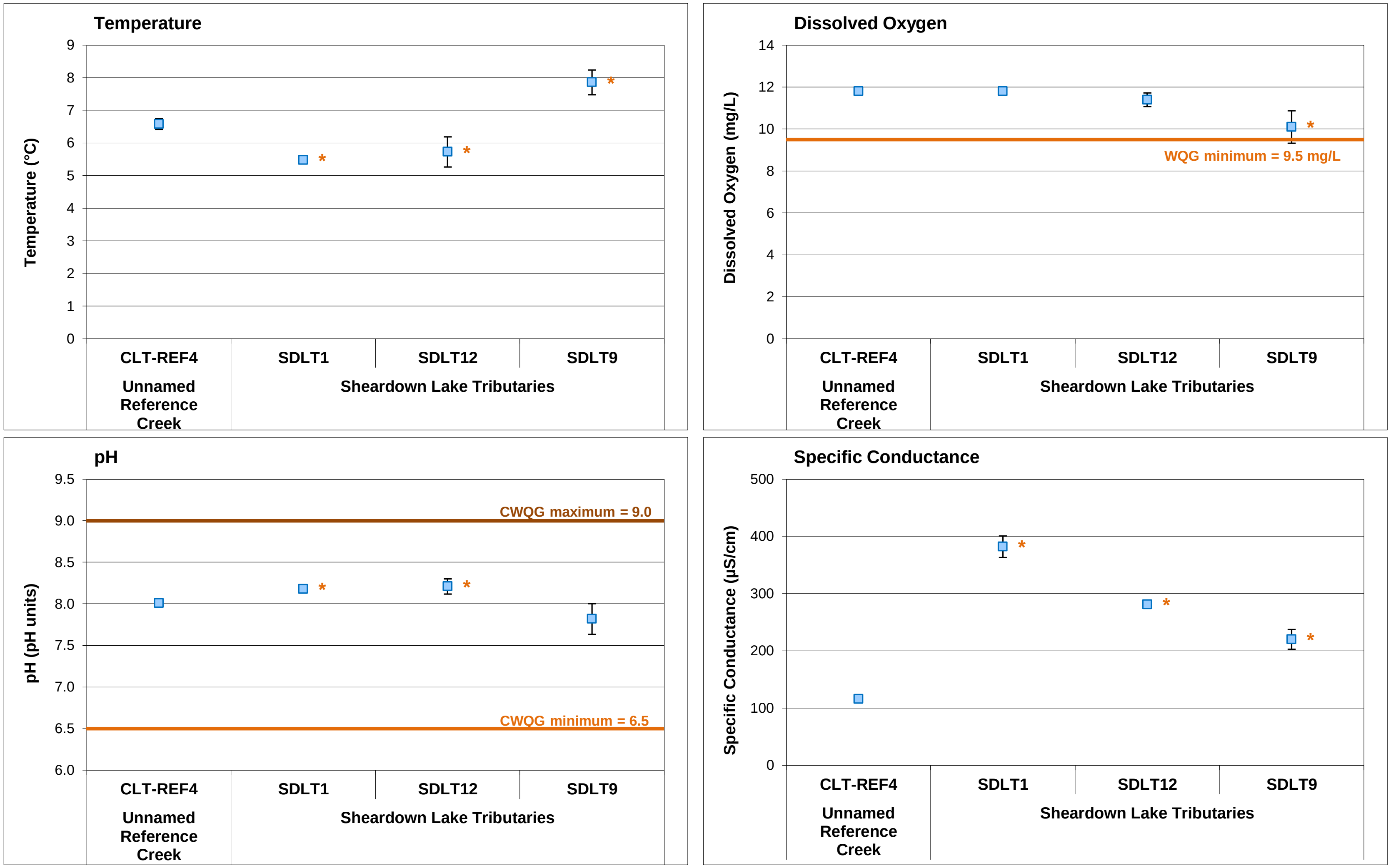


Figure 4.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Tributaries (SDLT) and Unnamed Reference Creek Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2021

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

Table 4.1: Mean Water Chemistry at Sheardown Lake Tributaries (SDLT1, SDLT12, and SDLT9) Monitoring Stations in Spring, Summer, and Fall, Mary River Project CREMP, 2021

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Bench-mark[b]	Reference Creek (n = 4)			Sheardown Lake Tributary 1 (n = 2)			Sheardown Lake Tributary 12[c] (LDFG-OUT; n = 1)	Sheardown Lake Tributary 9[c] (MS-C-G; n = 1)
					Spring	Summer	Fall	Spring	Summer	Fall	Fall	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	45.2	97.8	149	191	288	314	276	341
	pH (lab)	pH	6.5 - 9.0	-	7.58	7.75	7.98	7.85	7.99	7.98	7.81	7.84
	Hardness (as $CaCO_3$)	mg/L	-	-	19.7	43.6	68.8	86.6	129	140	131	174
	Total Suspended Solids	mg/L	-	-	3.12	2.05	2.22	<2	2.05	<2	<2.0	<2.0
	Total Dissolved Solids	mg/L	-	-	23.0	50.8	87.2	102	154	165	173	197
	Turbidity	NTU	-	-	2.05	3.11	2.93	2.16	2.16	4.64	<0.10	0.17
	Alkalinity (as $CaCO_3$)	mg/L	-	-	20.4	40.3	68.1	69.0	82.8	101	118	157
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	<0.01	<0.01	<0.01	<0.01	<0.005	<0.005	***1.82***	<0.010
	Nitrate	mg/L	3	3	<0.02	<0.02	0.0363	0.291	0.641	0.888	***7.330***	0.497
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.005	<0.005	<0.001	0.00125	0.043	<0.0050
	Total Kjeldahl Nitrogen	mg/L	-	-	0.125	0.100	0.105	0.145	0.188	0.217	1.710	0.223
	Dissolved Organic Carbon	mg/L	-	-	1.03	2.75	2.65	2.00	2.86	2.14	3.40	3.81
	Total Organic Carbon	mg/L	-	-	1.66	1.76	2.10	2.71	3.10	2.08	3.43	3.58
	Total Phosphorus	mg/L	0.030[α]	-	0.00475	0.00470	0.00462	0.00315	<0.003	-	<0.0030	0.0032
	Phenols	mg/L	0.004[α]	-	<0.001	0.00103	<0.001	<0.001	<0.001	<0.001	<0.0010	<0.0010
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.05	0.0570	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.19	2.66	4.61	5.92	7.88	8.26	1.64	6.83
	Sulphate (SO_4)	mg/L	218[β]	218	1.26	3.34	5.78	16.9	39.4	42.4	3.4	25.2
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0469	0.0822	0.0476	0.0332	0.0438	0.0942	<0.0030	0.0043
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.00002	<0.00003	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	0.000102	<0.0001	<0.0001	0.0000480	0.0000845	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00292	0.00592	0.00814	0.0103	0.0135	0.0132	0.0171	0.0176
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.000005	0.00000980	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.000005	<0.00005	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	0.0120	0.0125	0.0135	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.00001	<0.00001	<0.00001	0.0000220	0.0000238	0.0000268	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	4.09	9.02	13.2	16.4	23.8	-	25.4	30.9
	Chromium (Cr)	mg/L	0.0089	0.00856	<0.0005	<0.0005	<0.0005	<0.0005	0.000210	0.000660	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.0001	<0.0001	<0.0001	<0.0001	0.0000954	0.000143	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.000645	0.000852	0.000888	***0.00221***	***0.00244***	***0.00253***	0.00104	0.00169
	Iron (Fe)	mg/L	0.30	0.326	0.0582	0.0672	0.0570	0.100	0.0840	0.159	<0.030	<0.030
	Lead (Pb)	mg/L	0.001	0.001	0.0000950	0.000110	0.0000890	0.0000980	0.0000850	0.000188	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	0.00225	0.00233	0.00235	0.0014	0.0020
	Magnesium (Mg)	mg/L	-	-	2.36	5.10	8.50	11.1	17.4	18.6	16.5	22.6
	Manganese (Mn)	mg/L	0.935[β]	-	0.00104	0.00100	0.000942	0.00356	0.00334	0.00411	0.00044	0.00015
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.0000050	<0.0000050
	Molybdenum (Mo)	mg/L	0.073	-	0.000119	0.000271	0.000399	0.00282	0.00573	0.00561	0.00063	0.00222
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	0.000512	0.000532	0.00110	0.00126	0.00139	0.00125	0.00107
	Potassium (K)	mg/L	-	-	0.358	0.675	0.883	2.22	3.20	3.18	1.77	2.05
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	<0.001	<0.001	0.0000430	<0.0002	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.528	0.925	0.948	1.06	1.52	1.56	1.08	1.45
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.000005	<0.000005	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.674	1.66	2.62	1.83	2.98	3.26	1.88	2.94
	Strontium (Sr)	mg/L	-	-	0.00406	0.00962	0.0149	0.0207	0.0229	0.0258	0.0152	0.0185
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.0001	<0.0001	0.0000118	0.0000114	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.00002	<0.0002	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<0.01	0.0102	<0.01	<0.01	<0.0021	0.00401	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.000380	0.00167	0.00527	0.00326	0.00694	0.00946	0.00092	0.00418
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	<0.001	<0.001	0.000244	0.000355	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	<0.003	0.00390	0.00274	0.00350	<0.0030	<0.0030

[shaded] 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.

[c] Stations LDFG-OUT and MS-C-G were added to the CREMP in fall 2021, and therefore were not sampled in earlier seasons. There stations will be sampled in spring, summer, and fall in future years.

copper, nitrate, and sulphate concentrations were elevated at SDLT1 compared to the reference creeks during at least one sampling event in 2021, 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 were moderately (i.e., 5- to 10-times higher) to highly (i.e., ≥10-times higher) elevated at SDLT1 compared to the reference creeks in at least one of the spring, summer, or fall 2021 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 former main mine camp (Appendix Table C.34).

Special investigation into sources of copper to SDLT1, conducted by sampling additional locations upstream of the existing CREMP stations (Stations D1-CUSI-A, D1-CUSI-B, D1-CUSI-C, MS-C-F, MS-C-B) and downstream (Station MS-C-A) of Station D1-05 (Figure 2.2) in fall 2021 indicated concentrations of total copper above the WQG at all stations at SDLT1 and above the AEMP Benchmark at all but one station at SDLT1 (Appendix Table H.2). Highest total concentrations of these metals occurred in those samples with highest turbidity (Appendix Table H.2), suggesting that these metals were likely bound to suspended mineral material and not bioavailable. Review of dissolved copper concentrations within SDLT1 indicated no upstream to downstream spatial changes that would suggest a distinct source of copper to the SDLT1 system. The intent of this special investigation was to examine whether the source of elevated copper concentrations at SDLT1 during baseline was related to an isolated source (e.g., key tributary, groundwater upwelling, etc.) within the system. Recognizing that current mine operations may have obscured historical spatial patterns, the spatial examination of dissolved copper concentrations within SDLT1 did not indicate any distinct source of copper to the system, suggesting that elevated concentrations of copper at SDLT1 during baseline were related to natural minerology of the bedrock/overburden in the SDLT1 catchment.

Despite total copper concentrations above the AEMP benchmark and WQG at SDLT1 in 2021, the concentrations of copper during each seasonal sampling event were comparable to those reported at SDLT1 stations D1-05 and D1-00 during baseline (Appendix Figure C.11; Appendix Table C.34). This suggested that copper concentrations were naturally high within SDLT1 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 2021 (and other years of mine operation) compared to baseline (Appendix Figure C.11; Appendix Table C.34). For parameters without AEMP benchmarks, concentrations in at least two of the three seasonal sampling events in 2021 were comparable to baseline at SDLT1 (Appendix Table C.34; 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.

Water chemistry of SDLT12 and SDLT9 met AEMP benchmarks and WQG in the fall 2021 sampling event except for total ammonia and nitrate concentrations above respective AEMP benchmarks at SDLT12 (Table 4.1; Appendix Table C.33). Concentrations of total ammonia, nitrate, nitrite, and TKN were moderately or highly elevated at SDLT12 compared to the reference creeks the during fall 2021 sampling event (Appendix Table C.34). Concentrations of nitrate, molybdenum, and sulphate were elevated at SDLT9 compared to the reference creeks the during fall 2021 sampling event (Appendix Table C.34). The occurrence of elevated concentrations of these parameters, particularly elevated nitrogen compounds, suggested possible mine-related influences on water quality at SDLT12 and SDLT9 in 2021.

4.1.2 Phytoplankton

Phytoplankton (chlorophyll-a) monitoring was conducted at SDLT1 in spring, summer, and fall, whereas stations at SDLT12 and SDTL9 were added in fall 2021 and therefore not monitored in spring and summer (Figure 4.2). 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 2021 (Figure 4.2). Nitrate, phosphorus, and TKN concentrations were consistently the same or higher near the mouth of SDLT1 in 2021 (Appendix Table C.33), and thus higher chlorophyll-a concentrations near the mouth were in line with typical responses of phytoplankton to higher nutrient concentrations. Chlorophyll-a concentrations at SDLT1 and SDLT9 were within the range of variability observed among reference creeks in spring, summer, and fall sampling events (Figure 4.2). Chlorophyll-a concentrations were lower at SDLT12 compared to at the reference creeks and to those at SDLT1 and SDLT9 in fall 2021 (Figure 4.2), despite substantially higher concentrations of nitrate and TKN at SDLT12 (Appendix Table C.33). For all sampling events in 2021, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L at SDLT1, SDLT12, and SDLT9 (Figure 4.2). Similar to the reference creeks and Camp Lake tributaries, chlorophyll-a concentrations at the Sheardown Lake tributaries were suggestive of oligotrophic, low productivity conditions based on Dodds et al (1998) trophic status classification for stream environments (i.e., chlorophyll-a concentration <10 µg/L). Relatively low chlorophyll-a concentrations at the Sheardown Lake tributaries in 2021 were also consistent with an

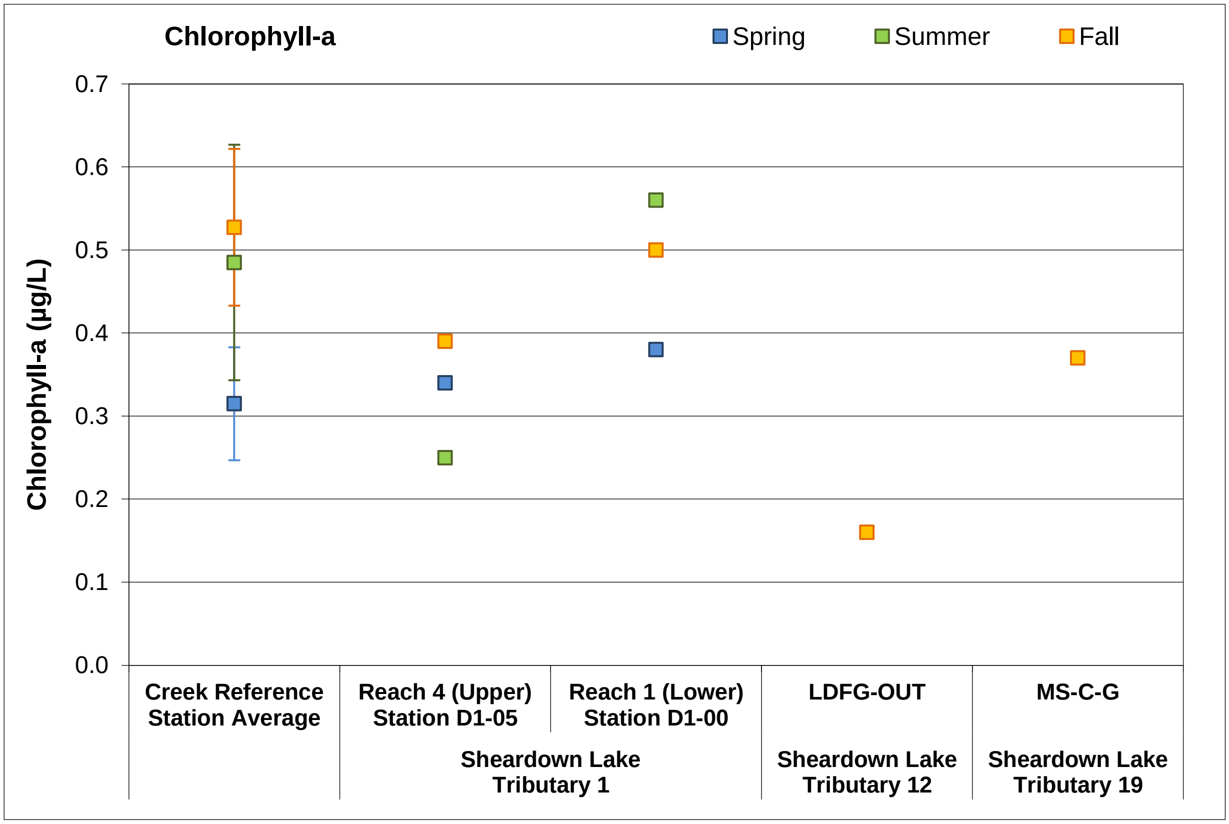


Figure 4.2: Chlorophyll-a Concentrations at Phytoplankton Monitoring Stations of the Sheardown Lake Tributaries (SDLT1, SDLT12, SDLT9), Mary River Project CREMP, 2021

Note: Reference creek data represented by average (± SD; n = 4) calculated from CLT-REF and MRY-REF stations. Stations LDFG-OUT and MS-C-G were added to the CREMP in fall 2021, and therefore were not sampled in earlier seasons. There stations will be sampled in spring, summer, and fall in future years.

oligotrophic categorization using CWQG (CCME 2021 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 2021 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 2021) periods (Figure 4.3). These results suggested no adverse mine-related influences on phytoplankton productivity at SDLT1 over the past seven years of mine operation. No chlorophyll-a data were available for SDLT12 and SDLT9 over the mine baseline period (2005 to 2013), precluding comparisons of SDLT12 and SDLT9 chlorophyll-a data to the period prior to mine construction.

4.1.3 Benthic Invertebrate Community

4.1.3.1 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 richness and significant differences in composition (as indicated by Bray-Curtis Index) compared to Unnamed Reference Creek in 2021 (Table 4.2; Appendix Table F.29). Marked differences in community composition between SDLT1 and the reference creek included significantly higher relative abundance of Oligochaeta and Chironomidae, and significantly lower relative abundance of Hydracarina and Simuliidae at SDLT1 (Table 4.2). However, no significant difference in the relative abundance of metal-sensitive Chironomidae occurred at SDLT1 compared to Unnamed Reference Creek (Table 4.2), 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 2021 (Table 4.1). Ecologically significant higher relative abundance of the collector gatherer and shredder FFG, as well as ecologically significant lower relative abundance of the filterer 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, significantly higher relative abundance of the sprawler HPG, and significantly lower relative abundance of the clinger HPG was shown at SDLT1 compared to the reference creek (Table 4.2), 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 2021) compared to baseline

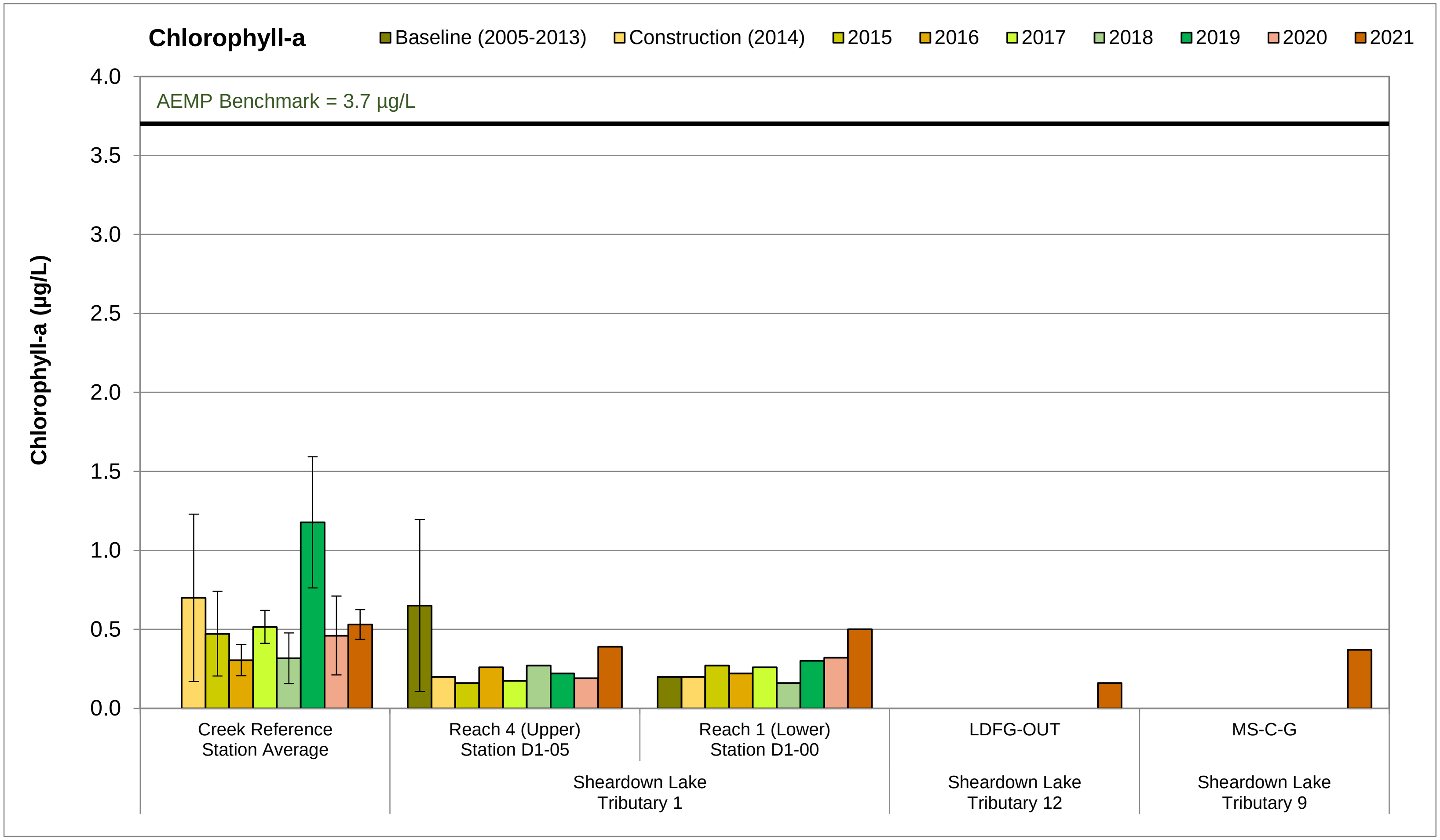


Figure 4.3: Temporal Comparison of Chlorophyll-a Concentrations at the Sheardown Lake Tributaries (SDLT1, SDLT12, and SDLT9) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2021) Periods in the Fall, Mary River Project CREMP

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

Table 4.2: Benthic Invertebrate Community Metric Statistical Comparison Results Among the Sheardown Lake Tributaries and Unnamed Reference Creek Study Areas, Mary River Project CREMP, August 2021

Metric	Overall 4-Area Comparison[a]				Pair-wise, *post hoc* comparisons				
	Statistical Test	Transform-ation	Significant Difference between Areas?	p-value	Study Area	Mean	Standard Deviation	Magnitude of Difference[b]	Pairwise Comparison
Density (No. per m^2)	ANOVA	log10	YES	0.043	Reference Creek	1,129	820	nc	AB
					SDLT1	627	345	-0.6	AB
					SDLT12	693	964	-0.5	B
					SDLT9	2,661	1,929	1.9	A
Richness (No. of Taxa)	ANOVA	log10	YES	0.004	Reference Creek	20	3.32	nc	A
					SDLT1	13	2.24	-2.1	B
					SDLT12	12.3	4.16	-2.3	B
					SDLT9	18.2	2.59	-0.5	A
Simpson's Evenness	ANOVA	none	YES	0.024	Reference Creek	0.924	0.052	nc	A
					SDLT1	0.885	0.033	-0.7	AB
					SDLT12	0.903	0.046	-0.4	AB
					SDLT9	0.821	0.054	-2.0	B
Nemata (% of community)	ANOVA	none	YES	<0.001	Reference Creek	2.3	2.0	nc	B
					SDLT1	5.5	1.7	1.5	B
					SDLT12	12.2	2.6	4.9	A
					SDLT9	3.5	1.9	0.6	B
Hydracarina (% of community)	ANOVA	log10(x+1)	YES	0.004	Reference Creek	5.0	3.0	nc	A
					SDLT1	0.8	0.7	-1.4	B
					SDLT12	0.0	0.0	-1.6	B
					SDLT9	2.2	0.9	-0.9	B
Ostracoda (% of community)	K-W	rank	YES	0.003	Reference Creek	0.9	0.8	nc	B
					SDLT1	0.0	0.0	-1.2	B
					SDLT12	0.8	0.8	-0.1	B
					SDLT9	4.8	3.4	5.2	A
Oligochaeta (% of community)	K-W	rank	YES	0.012	Reference Creek	0.2	0.3	nc	C
					SDLT1	4.3	2.2	14.9	AB
					SDLT12	17.8	11.5	64.5	A
					SDLT9	3.0	4.0	10.3	BC
Chironomidae (% of community)	ANOVA	log10	YES	<0.001	Reference Creek	40.2	7.5	nc	B
					SDLT1	78.4	6.2	5.1	A
					SDLT12	65.5	14.1	3.4	A
					SDLT9	72.0	9.4	4.2	A
Metal Sensitive Chironomidae (% of community)	ANOVA	log10(x+1)	YES	0.077	Reference Creek	6.4	2.0	nc	AB
					SDLT1	13.2	5.1	3.4	A
					SDLT12	4.7	5.8	-0.9	B
					SDLT9	8.5	5.2	1.1	AB
Simuliidae (% of community)	K-W	rank	YES	0.002	Reference Creek	34.6	11.4	nc	A
					SDLT1	0.4	0.9	-3.0	B
					SDLT12	0.0	0.0	-3.0	B
					SDLT9	7.2	5.5	-2.4	A
Tipulidae (% of community)	ANOVA	none	YES	0.050	Reference Creek	1.2	1.2	nc	A
					SDLT1	3.2	1.5	1.7	AB
					SDLT12	3.6	1.8	2.1	B
					SDLT9	1.7	0.9	0.5	AB
Collector Gatherers FFG (% of community)	ANOVA	none	YES	<0.001	Reference Creek	43.0	9.1	nc	B
					SDLT1	67.0	3.7	2.6	A
					SDLT12	77.8	5.3	3.8	A
					SDLT9	51.3	9.8	0.9	B
Filterers FFG (% of community)	K-W	rank	YES	0.002	Reference Creek	33.9	12.6	nc	A
					SDLT1	0.6	0.4	-2.6	B
					SDLT12	0.9	1.5	-2.6	B
					SDLT9	6.7	4.3	-2.2	A
Shredders FFG (% of community)	ANOVA	log10(x+1)	YES	<0.001	Reference Creek	3.5	3.0	nc	C
					SDLT1	29.4	4.6	8.7	AB
					SDLT12	21.5	4.3	6.0	B
					SDLT9	37.2	12.5	11.3	A
Clingers HPG (% of community)	ANOVA	log10	YES	<0.001	Reference Creek	45.9	11.4	nc	A
					SDLT1	29.2	4.5	-1.5	B
					SDLT12	18.5	6.6	-2.4	C
					SDLT9	46.8	9.6	0.1	A
Sprawlers HPG (% of community)	ANOVA	none	YES	0.060	Reference Creek	40.6	9.3	nc	B
					SDLT1	57.8	6.0	1.9	A
					SDLT12	47.8	10.0	0.8	AB
					SDLT9	44.5	11.5	0.4	AB
Burrowers HPG (% of community)	ANOVA	none	YES	0.001	Reference Creek	13.5	5.8	nc	B
					SDLT1	12.9	2.1	-0.1	B
					SDLT12	33.8	14.0	3.5	A
					SDLT9	8.2	6.0	-0.9	B

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

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

Notes: nc= no comparison; nm=MOD could not be calculated due to SD = 0.

[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

[b] Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; MCT reported as geometric mean for log10-transformed data, median for rank-transformed data, back-transformed means for untransformed data.

(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 of mine operation compared to both years of baseline at SDLT1 (Appendix Table F.30).[15] However, consistent differences in FFG composition that included significantly higher relative abundance of collector-gatherers and significantly lower relative abundance of filterers at SDLT1 beginning in 2018 and 2019, respectively, compared to at least one year of baseline. The 2021 results supported previous observations in 2020 that suggested a recent shift in the benthic food base has occurred such that the relative abundance of collector-gatherer and filterer FFG at SDLT1 since 2018 has more closely reflected the FFG composition at the reference creek than was observed during baseline.

4.1.3.2 Sheardown Lake Tributary 12 (SDLT12)

Benthic invertebrate density and evenness at SDLT12 did not differ significantly compared to Unnamed Reference Creek, but benthic invertebrate richness was significant lower compared to Unnamed Reference Creek in 2021 (Table 4.2). Benthic invertebrate community compositional differences were indicated between SDLT12 and the reference creek in 2021 based on significantly differing Bray-Curtis Index (Appendix Table F.29). The differences in community composition included significantly higher relative abundance of Nemata, Oligochaeta, Chironomidae, Tipulidae, collector gatherer FFG, shredder FFG, and burrower HPG, and significantly lower relative abundance of Simuliidae, filterer FFG, and the clinger HPG at SDLT12 compared to Unnamed Reference Creek (Table 4.2). Significantly lower relative abundance of Hydracarina at SDLT12 compared to Unnamed Reference Creek was also observed, but not considered ecologically significant based on a CES within ±2 SD_{REF} (Table 4.2). However, no significant differences in the relative abundance of metal-sensitive Chironomidae were indicated at SDLT12 compared to the reference creek in 2021 (Table 4.2).

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 occurrence of significantly higher and lower relative abundance of burrowers such as Tipulidae and filterers such as Simuliidae, respectively, at SDLT12 compared to the reference creek in 2021 without an accompanying consistent difference in the relative abundance of Tipulidae and the filterer FFG between 2021 and baseline at SDLT12 suggested

[15] 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 2021 was comparable to that shown at the reference creek in 2021, suggesting that the relative abundance of this group during baseline in 2008 was unusually high.

that significantly lower flow at SDLT12 compared to Unnamed Reference Creek 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 2021 and since the commencement of commercial mine operations in 2015.

4.1.3.3 Sheardown Lake Tributary 9 (SDLT9)

Benthic invertebrate density and richness at SDLT9 did not differ significantly compared to Unnamed Reference Creek, but evenness was significantly lower at SDLT9 than at the reference creek in 2021 (Table 4.2). Benthic invertebrate community compositional differences were indicated between SDLT9 and the reference creek in 2021 based on significantly differing Bray-Curtis Index (Appendix Table F.29). Relative abundances of Ostracoda and Chironomidae were significantly higher at SDLT9 compared to Unnamed Reference Creek. Although the relative abundance of Hydracarina was significantly lower at SDLT9 compared to the reference creek, this difference was not ecologically meaningful (i.e., CES within ±2 SD_{REF}; Table 4.2). No significant differences in the relative abundance of metal-sensitive Chironomidae or any HPG were indicated between SDLT9 and the reference creek (Table 4.2). However, significantly higher relative abundance of the shredder FFG occurred at SDLT9 compared to the reference creek in 2021 (Table 4.2), which was consistent with field observations of greater amounts of rooted in-stream vegetation and organic debris, the primary food source for shredders, at SDLT9 (Appendix Table F.24). In turn, this suggested natural differences in habitat accounted for the differences in benthic invertebrate community structure between SDLT9 and the reference creek in 2021.

Temporally, benthic invertebrate density, richness, evenness, and the relative abundance of all dominant taxonomic groups and FFGs showed no consistent differences over years of mine operation from 2015 to 2021 compared to baseline at SDLT9 (Appendix Table F.34). Overall, despite some difference in benthic invertebrate community composition between SDLT9 and Unnamed Reference Creek in 2021, these differences appeared to be related to naturally greater amount of in-stream vegetation at SDLT9 and did not include differences in relative abundance of metal-sensitive taxa. In addition, no significant differences in benthic invertebrate community metrics were indicated for SDLT9 in 2021 relative to baseline and previous years of mine operation from 2015 to 2020. Collectively, these findings indicated no adverse influences of the mine on the benthic invertebrate community structure of SDLT9 since the commencement of commercial mine operations in 2015.

4.1.4 Effects Assessment and Recommendations

4.1.4.1 Sheardown Lake Tributary 1

At SDLT1, the following AEMP benchmarks were exceeded in 2021:

- Aqueous total copper concentration greater than the benchmark of 0.0022 mg/L in spring, summer, and fall monitoring events (i.e., 0.00221 mg/L, 0.00244 mg/L, and 0.00253 mg/L, respectively).

Although copper concentrations at SDLT1 were, on average, slightly higher than at the reference creeks in 2021, the concentration of copper in 2021 was closely comparable to those reported during baseline suggesting a natural source contributed to elevation of copper at SDLT1 in 2021 (and historically). 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 2021 compared to baseline conditions, copper concentrations at SDLT1 may just naturally be similar to the AEMP benchmark. In response to elevated copper concentrations measured in 2020, a spatially expanded water quality monitoring special investigation was implemented in 2021, the results of which generally supported the notion stated above that copper concentrations in water of SDLT1 naturally are near the AEMP benchmark. Biological monitoring conducted at SDLT1 in 2021 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, no adjustment to the existing AEMP need be applied at SDLT1 under the AEMP Management Response Framework. Although no action is required by the AEMP Management Response Framework, Baffinland is currently considering alterations to the SDLT1 system, which may address elevated aqueous copper concentrations.

4.1.4.2 Sheardown Lake Tributary 12

At the SDLT12, the following AEMP benchmarks were exceeded in 2021:

- Aqueous total ammonia concentration greater than the benchmark of 0.855 mg/L during the fall monitoring event; and
- Aqueous nitrate concentration greater than the benchmark of 3 mg/L during the fall monitoring event.

Phytoplankton and benthic invertebrate community monitoring at SDLT12 indicated no adverse influences of the mine on the aquatic biota of this tributary since the commencement of commercial mine operations in 2015, including in 2021. Because routine water quality monitoring at SDLT12 was added during fall 2021 and only a single water quality data point exists, and considering that no adverse effects to biota were associated with total ammonia and nitrate concentrations above respective AEMP benchmarks at SDLT12, a low action response to continue collecting water quality data at SDLT12 is recommended to meet obligations under the AEMP Management Response Framework.

4.1.4.3 Sheardown Lake Tributary 9

Water chemistry at SDLT9 met all AEMP benchmarks in 2021. In addition, phytoplankton and benthic invertebrate community monitoring conducted at SDLT9 indicated no adverse influences of the mine on aquatic biota of this tributary since the commencement of commercial mine operations in 2015, including in 2021. Under the Mary River Project AEMP Management Response Framework, because no changes in concentrations of parameters with AEMP benchmarks occurred relative to background and no adverse biological effects were indicated in 2021, no adjustment to the existing AEMP need be applied at SDLT9 as part of the 2022 CREMP. Continued water quality monitoring at SDLT9 is recommended for 2022.

4.2 Sheardown Lake Northwest (DL0-1)

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 2021 showed no substantial station-to-station differences during any of the winter, summer, or fall sampling events (Appendix Figures C.12 to C.15). No thermal stratification developed at Sheardown Lake NW during the winter, summer, or fall 2021 sampling events, whereas Reference Lake 3 had a warmer surface layer that extended to a depth of approximately 6 m during the summer sampling event (Figure 4.4). The average water temperature at the bottom of the water column at Sheardown Lake NW littoral and profundal stations were significantly warmer than at Reference Lake 3 for respective habitats during the August 2021 biological study, but the incremental differences in bottom water temperature between lakes were minor (i.e., up to 0.7°C) and thus unlikely to be ecologically important (Figure 4.5; Appendix Table C.41).

Dissolved oxygen profiles measured at Sheardown Lake NW in 2021 showed declining saturation levels with increased depth 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 4.4). Dissolved oxygen concentrations near the bottom of the water column were significantly lower at Sheardown Lake NW littoral and profundal stations than like habitat stations at Reference Lake 3 during the August 2021 biological study (Figure 4.5; Appendix Tables C.39 and C.41). However, mean dissolved oxygen concentrations were above

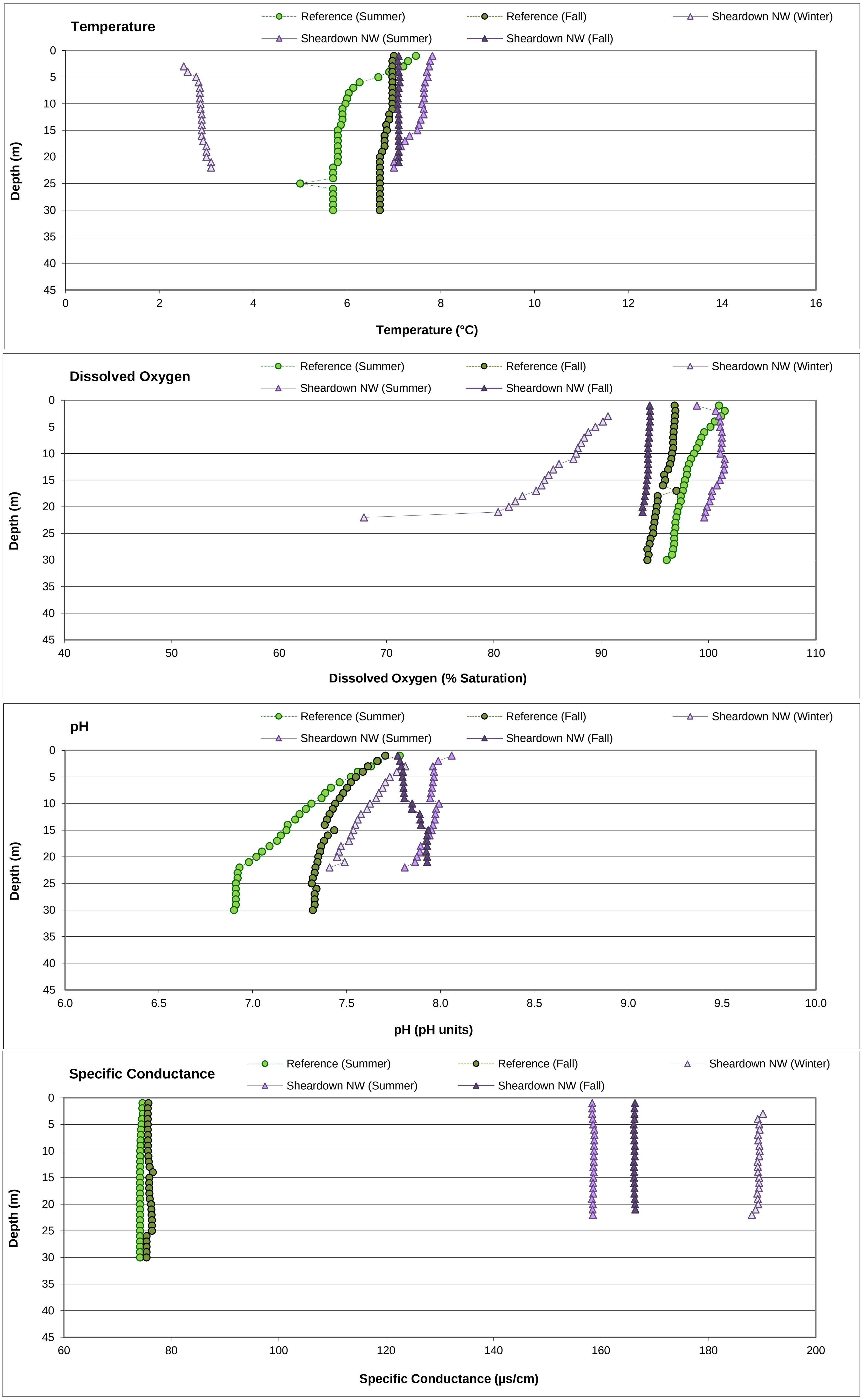


Figure 4.4: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake NW (DL0-01) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

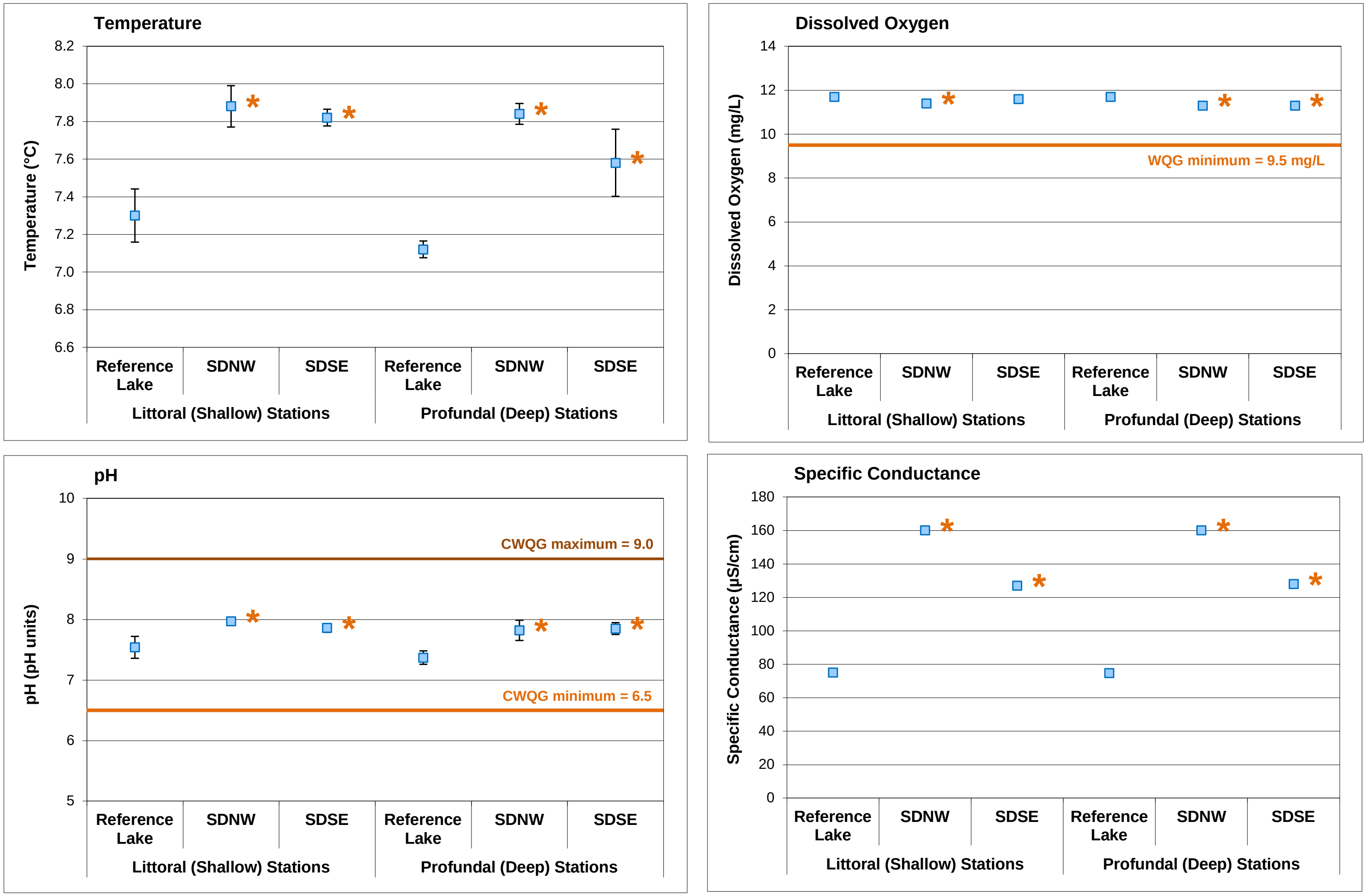


Figure 4.5: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Basins (SDNW and SDSE) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2021

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.

the WQG of 9.5 mg/L near the bottom at littoral and profundal stations of Sheardown Lake NW during biological monitoring in August 2021 (Figure 4.5; Appendix Table C.41).

Water column profiles showed decreasing pH with increased depth at Sheardown Lake NW and Reference Lake 3 in 2021 in all seasons except for at Sheardown Lake NW in fall where pH increased slightly with depth (Figure 4.4). The pH near the bottom at littoral and profundal stations of Sheardown Lake NW were significantly higher (i.e., more alkaline) than at respective habitats at the reference lake during the August 2021 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 Tables C.39 and C.41), 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 2021, and exhibited similar patterns to those observed at Reference Lake 3 (Figure 4.4). Specific conductance near the bottom of the water column at littoral and profundal stations were significantly higher at Sheardown Lake NW than at like-habitat at the reference lake (Figure 4.5; Appendix Table C.41). Water clarity, as determined through evaluation of Secchi depth, was significantly lower at Sheardown Lake NW than at the reference lake, indicating less water clarity, at the time of the August 2021 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 2021 (Table 4.3; Appendix Table C.42).[16] 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 4.3; Appendix Table C.43). Of those parameters without AEMP benchmarks, turbidity, total concentrations of manganese, total and dissolved concentrations of molybdenum and uranium, and TDS concentrations were elevated at Sheardown Lake NW compared to the reference lake during summer and/or fall sampling events in 2021 (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 2021 were associated with suspended material that contributed to elevated turbidity at Sheardown Lake NW (Table 4.3; Appendix Table C.42).

[16] Total phosphorous and total aluminum concentrations were higher than the applicable WQGs in one sample each, however these concentrations appeared to be anomalies based on an order of magnitude difference in concentration between these samples and data reported for all other Sheardown Lake NW samples during like-seasons (Appendix Table C.42).

Table 4.3: Mean Water Chemistry at Sheardown Lake NW (DL0-01) and Reference Lake 3 (REF3) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 (n = 3)		Sheardown Lake NW Stations (n = 6)		
					Summer	Fall	Winter	Summer	Fall
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	79.1	83.3	193	162	174
	pH (lab)	pH	6.5 - 9.0	-	7.71	7.68	7.68	8.00	8.04
	Hardness (as $CaCO_3$)	mg/L	-	-	36.8	37.7	94.3	75.9	77.0
	Total Suspended Solids (TSS)	mg/L	-	-	<2	<2	<2	<2	<2
	Total Dissolved Solids (TDS)	mg/L	-	-	29.0	41.0	115	96.0	90.6
	Turbidity	NTU	-	-	0.240	0.183	0.136	1.58	0.820
	Alkalinity (as $CaCO_3$)	mg/L	-	-	37.3	46.3	75.4	58.7	62.1
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.0140	<0.01	<0.005	<0.005	<0.005
	Nitrate	mg/L	3	3	<0.02	<0.02	0.303	0.228	0.257
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.001	<0.001	0.00116
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.177	0.200	0.113	0.106	0.119
	Dissolved Organic Carbon	mg/L	-	-	3.00	4.33	1.51	1.43	2.58
	Total Organic Carbon	mg/L	-	-	3.46	3.49	1.54	1.31	1.95
	Total Phosphorus	mg/L	0.020[α]	-	0.00443	0.00303	-	-	-
	Phenols	mg/L	0.004[α]	-	0.00113	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.05	<0.05	<0.05
	Chloride (Cl)	mg/L	120	120	1.32	1.35	5.29	4.14	4.61
	Sulphate (SO_4)	mg/L	218[β]	218	3.68	3.69	18.5	14.4	16.5
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.00373	0.00370	0.00521	0.0204	0.0149
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.00002	0.0000235	0.0000207
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.0000589	0.0000894	0.0000549
	Barium (Ba)	mg/L	-	-	0.00676	0.00651	0.00886	0.00756	0.00787
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.000005	<0.000005	<0.000005
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.000005	<0.000005	<0.000005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	0.0185	0.0162	0.0165
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.00001	<0.00001	<0.000005	0.00000504	<0.000005
	Calcium (Ca)	mg/L	-	-	7.40	7.43	18.1	14.2	14.0
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	0.000104	0.000224	<0.0001
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.0001	<0.0001	0.0000239	0.0000371	0.0000315
	Copper (Cu)	mg/L	0.002	0.0024	0.000810	0.000750	0.000878	0.000930	0.000856
	Iron (Fe)	mg/L	0.30	0.300	<0.03	<0.03	0.00559	0.0300	0.0188
	Lead (Pb)	mg/L	0.001	0.001	<0.00005	<0.00005	0.0000113	0.0000374	0.0000220
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	0.00173	0.00164	0.00158
	Magnesium (Mg)	mg/L	-	-	4.67	4.77	12.4	9.66	9.95
	Manganese (Mn)	mg/L	0.935[β]	-	0.000674	0.000591	0.000582	0.00218	0.00108
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000155	0.000157	0.00114	0.00101	0.00121
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000741	0.000672	0.000656
	Potassium (K)	mg/L	-	-	0.960	0.910	1.64	1.45	1.55
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	0.0000443	0.0000405	0.0000401
	Silicon (Si)	mg/L	-	-	0.507	0.490	0.757	0.683	0.563
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.000005	<0.000005	<0.000005
	Sodium (Na)	mg/L	-	-	0.994	0.929	2.43	2.00	2.07
	Strontium (Sr)	mg/L	-	-	0.00877	0.00856	0.0122	0.0112	0.0119
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.000005	<0.000005	<0.000005
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.00002	0.0000213	<0.00002
	Titanium (Ti)	mg/L	-	-	<0.01	<0.01	0.000304	0.00114	0.000734
	Uranium (U)	mg/L	0.015	-	0.000328	0.000342	0.00160	0.00139	0.00182
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	0.0000666	0.0000818	<0.00005
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	0.000679	0.000861	0.000533

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).

Naturally high turbidity[17] 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 aluminum and manganese between Sheardown Lake NW and Reference Lake 3 in 2021 (and historically) suggested that the mine was unlikely to be the source of these metals (Appendix Tables C.45). Nitrate and sulphate were the only parameters among those with established AEMP benchmarks that were elevated in at least one season in 2021 compared to baseline at Sheardown Lake NW (Appendix Figure C.16), as were dissolved molybdenum and dissolved uranium concentrations among those parameters without AEMP benchmarks (Appendix Tables C.43 and C.45). Of the parameters elevated at Sheardown Lake NW in 2021 compared to baseline and the reference lake, sulphate and dissolved uranium concentrations have recently shown increasing trends in groundwater adjacent to Sheardown Lake NW (Tetra Tech 2022), suggesting that a nearby landfill was a possible source of these parameters. However, no other parameters that exhibited increasing concentrations in groundwater over time at wells located adjacent to Sheardown Lake NW (i.e., dissolved iron and nickel) showed elevation and/or increasing trends in surface water of Sheardown Lake NW in 2021.

Overall, a slight mine-related influence on water quality of Sheardown Lake NW was suggested in 2021 as reflected by elevated total and/or dissolved 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 in 2021 was primarily composed of silt, except at a littoral station (DL0-01-10), which contained 89% sand (Figure 4.6; Appendix Table D.12). No significant differences in sediment particle size were indicated between Sheardown Lake NW and Reference Lake 3 for respective littoral and profundal stations (Appendix Table D.13). Key differences in physical sediment properties between Sheardown Lake NW and Reference Lake 3 included significantly lower TOC content in sediment at profundal stations and significantly less compact sediment (i.e., higher moisture content) at littoral stations of Sheardown Lake NW (Figure 4.6; Appendix Table D.13). A reddish-brown oxidized layer of material was observed on the surface of Sheardown Lake NW sediments at all littoral and profundal stations, consistent with

[17] Turbidity at Sheardown Lake NW in 2021 was comparable to turbidity shown at the lake during baseline (Appendix Table C.43), suggesting that greater turbidity at this lake compared to Reference Lake 3 reflects a natural phenomenon.

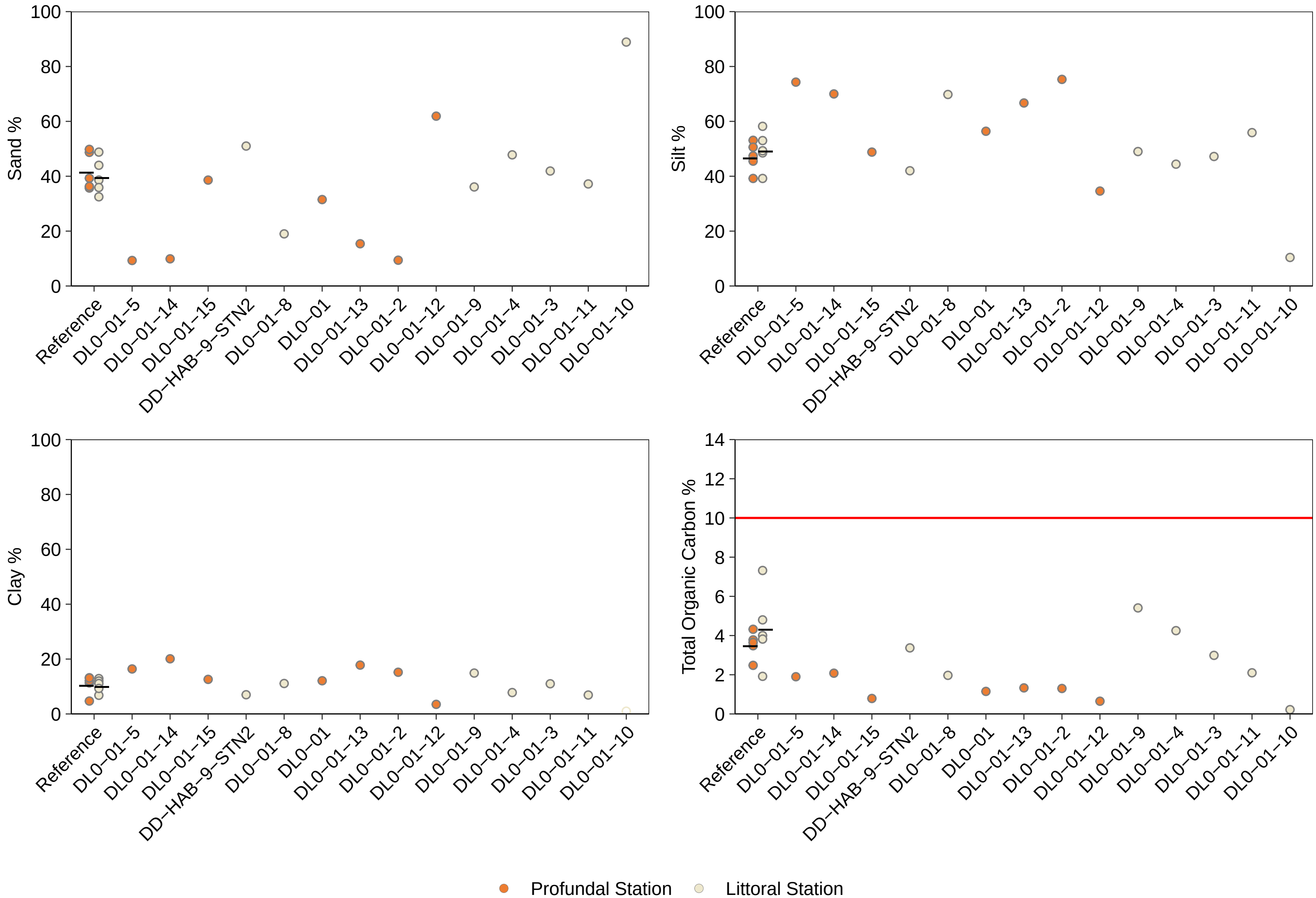


Figure 4.6: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake NW (DL0−01) Sediment Monitoring Stations and Reference Lake 3, Mary River Project CREMP, August 2021

Note: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

the substrate observed at Camp Lake (Appendix Tables D.10 and D.11). This material occasionally occurred as a thin, distinct layer that was likely composed mainly 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 sampling in August 2021 (Appendix Table D.10), suggesting the occurrence of reducing conditions within the sediment.

No spatial differences in sediment metal concentrations were indicated between stations located nearest to key tributary inlets (e.g., SDLT1 and SDLT12) and those located near the Sheardown Lake NW outlet in 2021 (Appendix Table D.12). However, arsenic, cadmium, iron, manganese, and uranium concentrations were highest in the sediment at the Sheardown Lake NW station located closest to the outlet of SDLT1 (i.e., Station DD-HAB-9-STN2). Elevated concentrations of these metals may be related to high TOC content in sediment at this location (Appendix Table D.12). 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 4.4). Mean metal concentrations in littoral and profundal sediments at Sheardown Lake NW were very similar (within 3-times) to mean concentrations observed at Reference Lake 3 (Table 4.4; Appendix Table D.12).

Mean metal concentrations in littoral and profundal sediments from Sheardown Lake NW in 2021 were similar to concentrations measured during the mine baseline period (2005 to 2013; Figure 4.7; Appendix Table D.14).[18] Metal concentrations in the sediment of Sheardown Lake NW in 2021 were within respective ranges observed from 2015 to 2020 (Figure 4.7). There were no increasing trends in concentrations of metals in Sheardown Lake NW sediments over the mine operation period (2015 to 2021) in comparison to the baseline period (Figure 4.7). Overall, no substantial changes in sediment chemistry have been observed at Sheardown Lake NW following 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 2021 (Figure 4.8). Chlorophyll-a concentrations differed significantly among seasons at Sheardown Lake NW, with highest and lowest concentrations observed in fall and winter, respectively (Appendix Tables E.5 and E.6), which was contrary to highest chlorophyll-a concentrations

[18] See footnote 12 regarding differences in the concentration of boron in sediment between baseline and recent CREMP studies.

Table 4.4: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Sheardown Lake NW (DL0-01) and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2021

	Analyte	Units	Canadian or Provincial SQ Criteria[a]	AEMP Benchmark[b]	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	-	3.70 ± 1.09	2.74 ± 2.20	4.21 ± 1.83	1.42 ± 0.330
Metals	Aluminum (Al)	mg/kg	-	-	21,120 ± 4,592	15,168 ± 8,550	20,520 ± 4,129	23,000 ± 2,699
Metals	Antimony (Sb)	mg/kg	-	-	<0.100 ± 0.000	<0.100 ± 0	<0.100 ± 0	0.100 ± 0
Metals	Arsenic (As)	mg/kg	17	6.2	5.54 ± 1.03	5.54 ± 3.90	5.13 ± 0.978	4.50 ± 0.484
Metals	Barium (Ba)	mg/kg	-	-	123 ± 29.4	106 ± 79.5	129 ± 13.4	93.1 ± 5.91
Metals	Beryllium (Be)	mg/kg	-	-	0.812 ± 0.176	0.725 ± 0.411	0.782 ± 0.122	1.09 ± 0.064
Metals	Bismuth (Bi)	mg/kg	-	-	<0.200 ± 0	0.260 ± 0.055	<0.200 ± 0	0.268 ± 0.036
Metals	Boron (B)	mg/kg	-	-	14.9 ± 2.68	22.6 ± 12.6	15.2 ± 2.23	30.3 ± 3.40
Metals	Cadmium (Cd)	mg/kg	3.5	1.5	0.144 ± 0.046	0.302 ± 0.198	0.167 ± 0.048	0.254 ± 0.032
Metals	Calcium (Ca)	mg/kg	-	-	5,026 ± 849	3,855 ± 2,041	5,186 ± 660	4,348 ± 170
Metals	Chromium (Cr)	mg/kg	90	97	67.2 ± 15.0	56.6 ± 30.0	64.6 ± 10.7	82.1 ± 7.53
Metals	Cobalt (Co)	mg/kg	-	-	14.8 ± 3.35	12.5 ± 6.98	13.8 ± 3.48	16.3 ± 0.772
Metals	Copper (Cu)	mg/kg	110	58	***81.2*** ± 27.5	41.7 ± 26.2	***85.0*** ± 10.5	50.2 ± 3.42
Metals	Iron (Fe)	mg/kg	40,000	52,200	44,360 ± 8,848	***52,275*** ± 32,994	40,900 ± 11,600	40,700 ± 3,069
Metals	Lead (Pb)	mg/kg	91.3	35	17.3 ± 2.81	16.1 ± 8.68	16.2 ± 2.46	22.6 ± 1.55
Metals	Lithium (Li)	mg/kg	-	-	34.7 ± 5.36	24.3 ± 14.3	33.3 ± 5.37	38.5 ± 3.03
Metals	Magnesium (Mg)	mg/kg	-	-	13,760 ± 2,748	10,065 ± 5,351	12,918 ± 2,534	14,400 ± 1,924
Metals	Manganese (Mn)	mg/kg	1,100	4,530	958 ± 402	913 ± 1,034	882 ± 346	937 ± 362
Metals	Mercury (Hg)	mg/kg	0.486	0.17	0.052 ± 0.024	0.035 ± 0.022	0.045 ± 0.010	0.037 ± 0.013
Metals	Molybdenum (Mo)	mg/kg	-	-	3.23 ± 0.556	5.88 ± 3.88	2.69 ± 0.733	1.81 ± 0.106
Metals	Nickel (Ni)	mg/kg	75	77	46.6 ± 10.1	63.0 ± 35.8	45.9 ± 6.89	69.0 ± 5.87
Metals	Phosphorus (P)	mg/kg	2,000	1,958	955 ± 154	843 ± 410	902 ± 111	812.3 ± 61.6
Metals	Potassium (K)	mg/kg	-	-	5,140 ± 946	3,870 ± 2,199	4,862 ± 765	5,660 ± 715
Metals	Selenium (Se)	mg/kg	-	-	0.628 ± 0.252	0.458 ± 0.198	0.554 ± 0.140	0.318 ± 0.086
Metals	Silver (Ag)	mg/kg	-	-	0.202 ± 0.077	0.158 ± 0.042	0.172 ± 0.050	0.185 ± 0.025
Metals	Sodium (Na)	mg/kg	-	-	394 ± 80.1	235 ± 119	362 ± 58.6	338 ± 65.8
Metals	Strontium (Sr)	mg/kg	-	-	13.5 ± 2.11	9.22 ± 3.71	13.2 ± 1.41	12.3 ± 0.695
Metals	Sulphur (S)	mg/kg	-	-	1,340 ± 297	1,300 ± 424	1,380 ± 444	1,000 ± 0
Metals	Thallium (Tl)	mg/kg	-	-	0.594 ± 0.205	0.425 ± 0.247	0.556 ± 0.129	0.564 ± 0.025
Metals	Tin (Sn)	mg/kg	-	-	<2.000 ± 0	<2.000 ± 0	<2.000 ± 0	2.000 ± 0
Metals	Titanium (Ti)	mg/kg	-	-	1,204 ± 112	911 ± 466	1,096 ± 158	1,293 ± 198
Metals	Uranium (U)	mg/kg	-	-	22.0 ± 8.39	7.32 ± 5.04	20.3 ± 7.92	8.81 ± 0.600
Metals	Vanadium (V)	mg/kg	-	-	63.2 ± 11.0	43.7 ± 24.0	60.7 ± 9.23	62.8 ± 5.74
Metals	Zinc (Zn)	mg/kg	315	135	88.0 ± 18.2	55.6 ± 31.0	87.5 ± 11.8	76.0 ± 7.22
Metals	Zirconium (Zr)	mg/kg	-	-	5.16 ± 1.07	10.8 ± 7.02	4.66 ± 1.28	11.1 ± 4.34

(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 NW

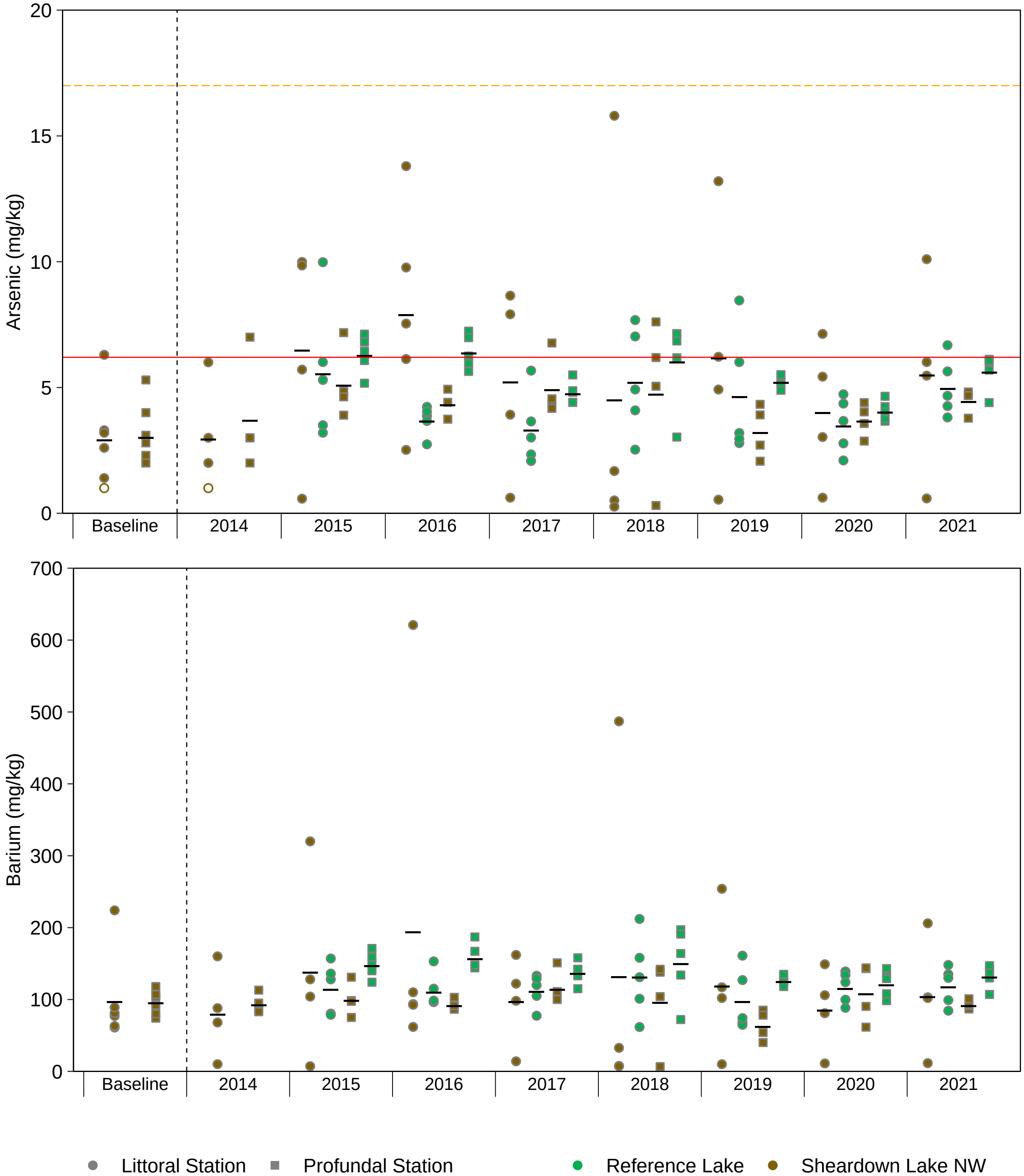
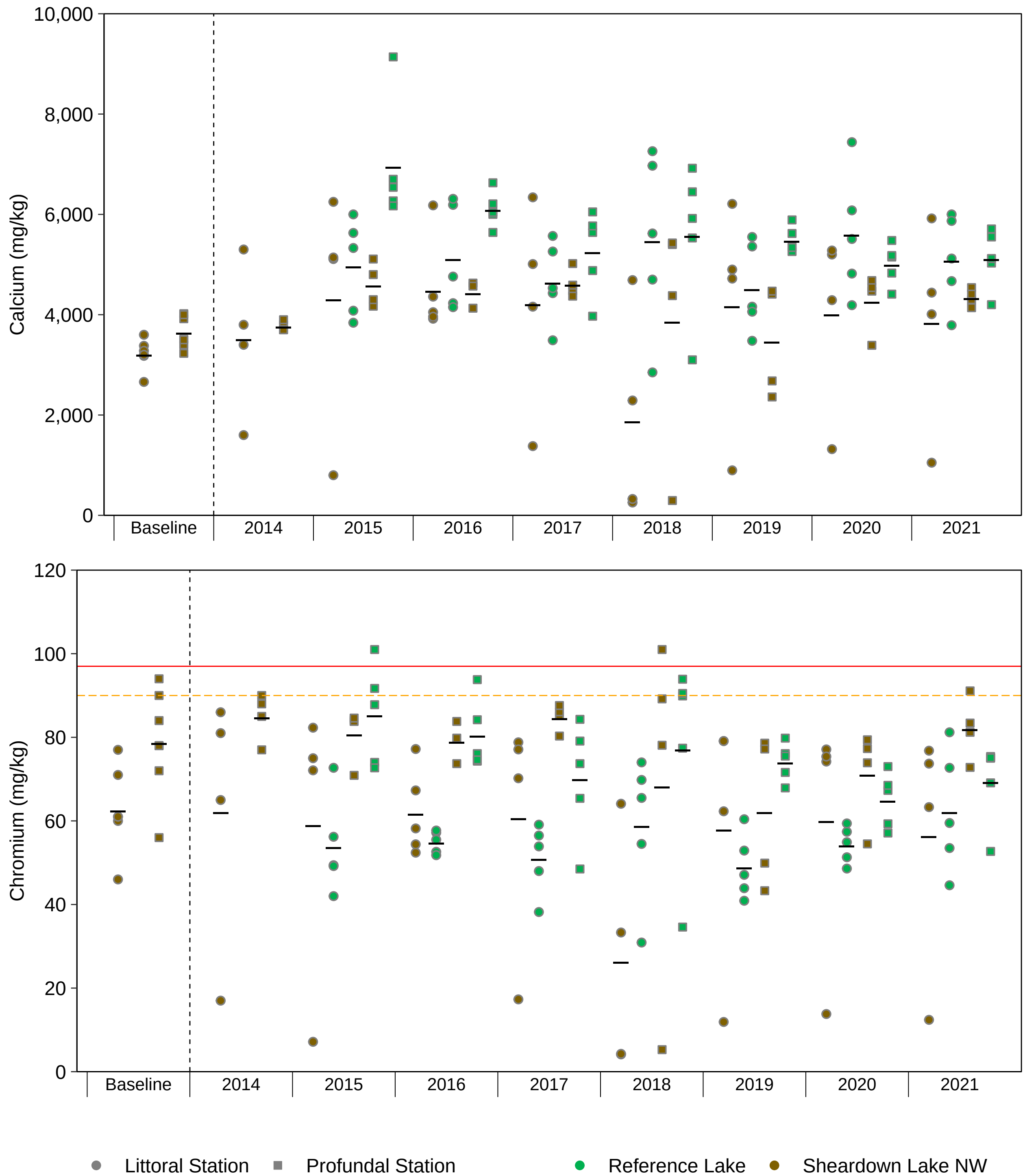


Figure 4.7: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake NW (SDNW) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.



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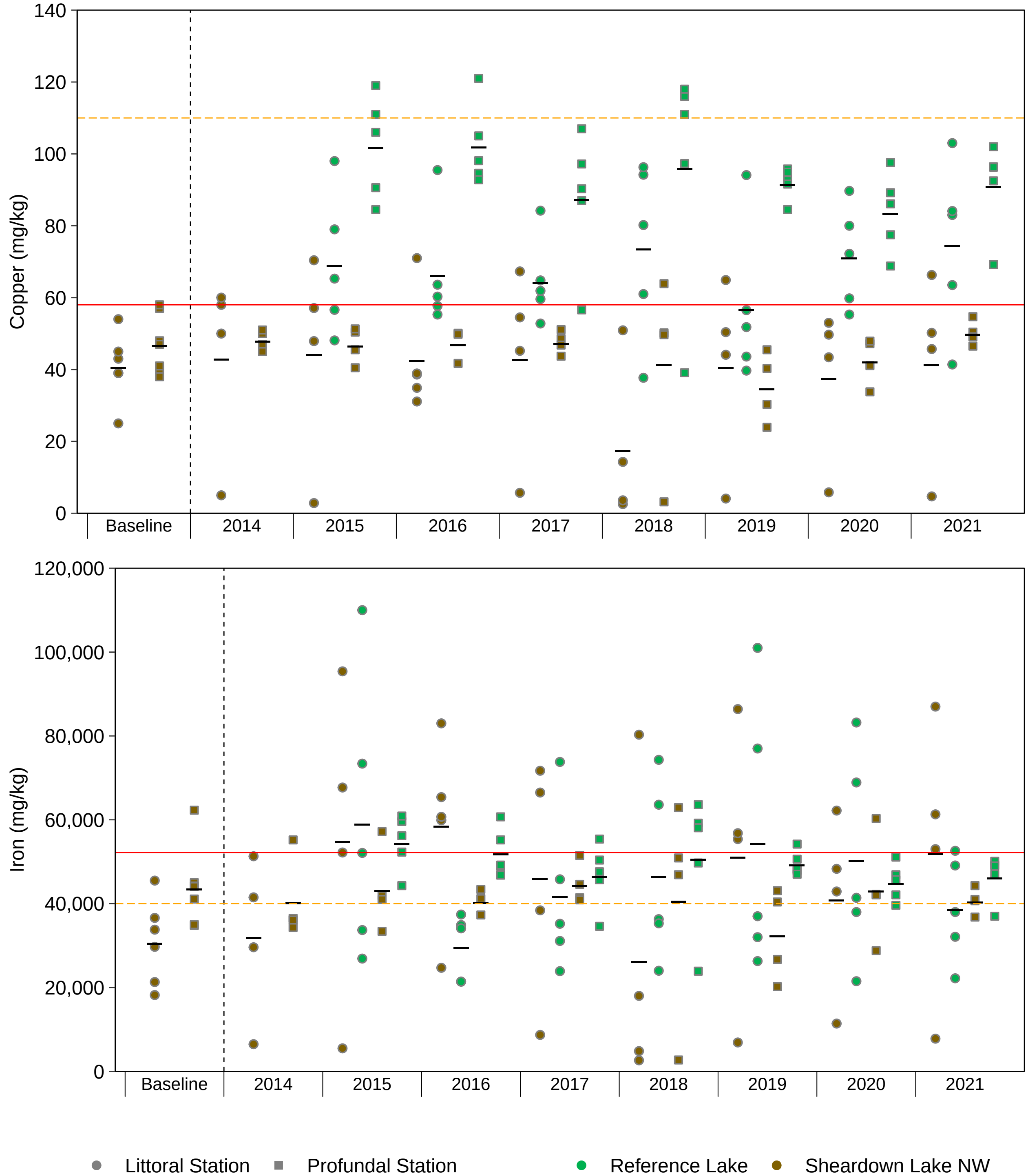

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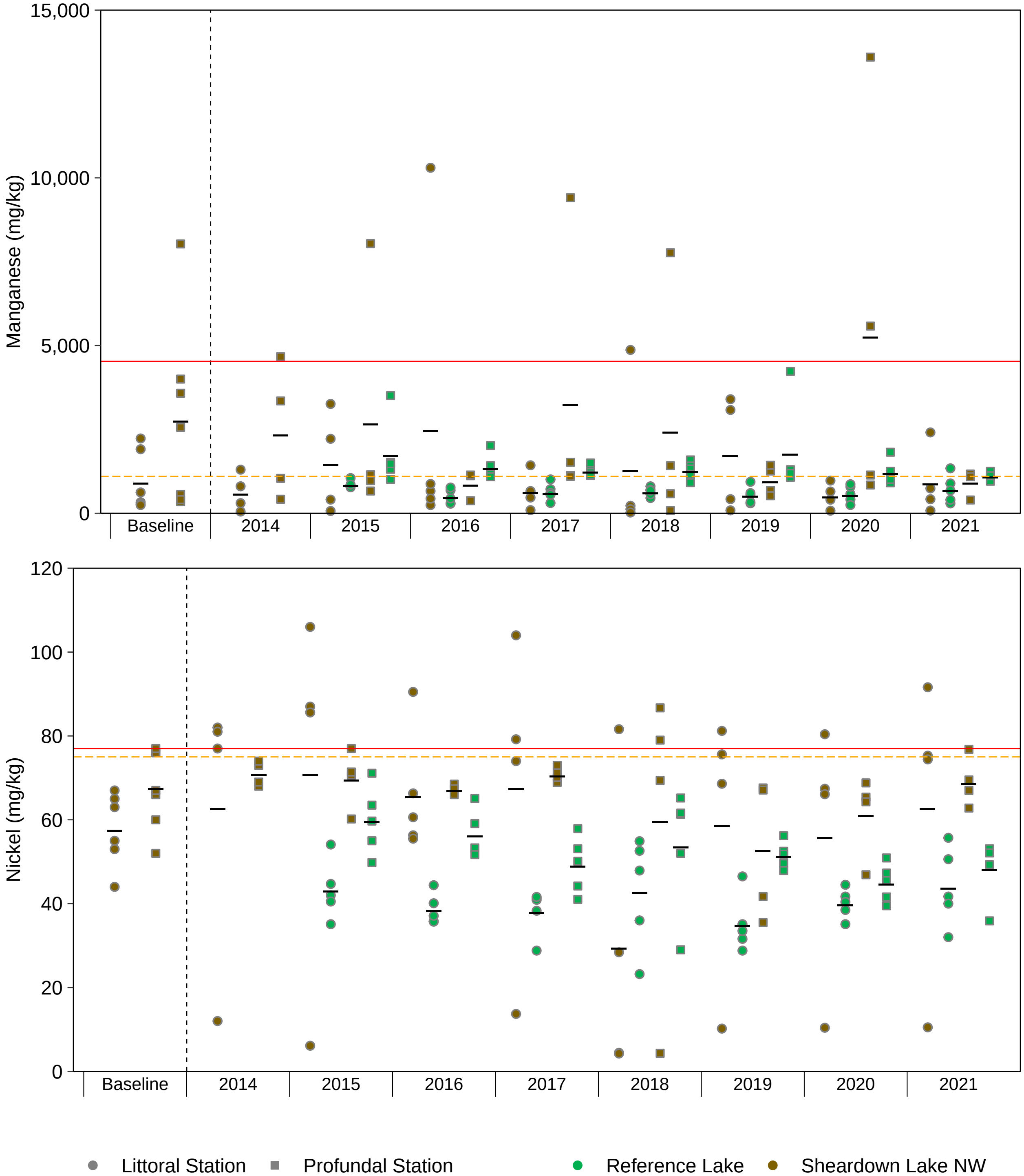

Figure 4.7: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake NW (SDNW) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

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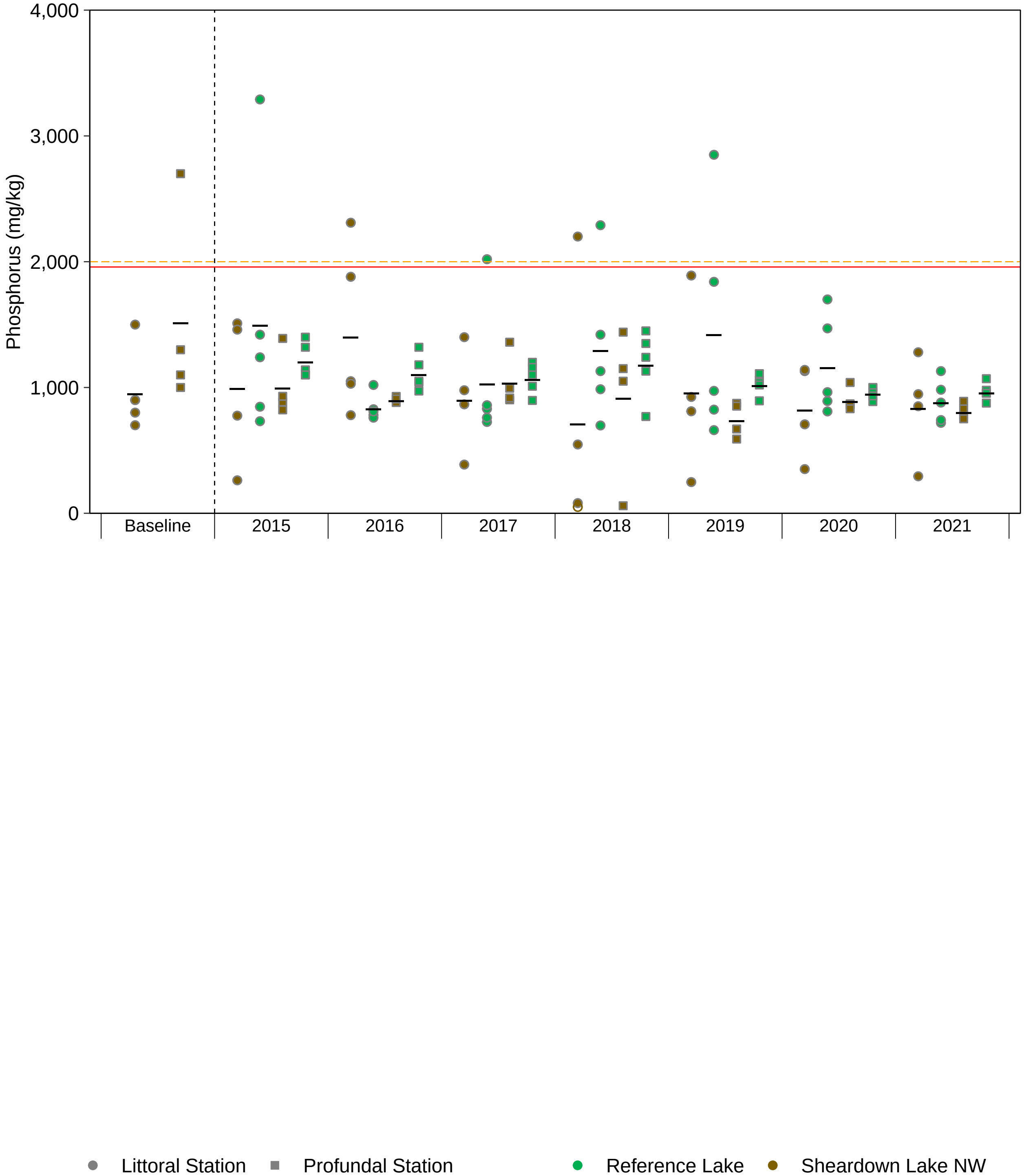


Figure 4.7: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake NW (SDNW) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

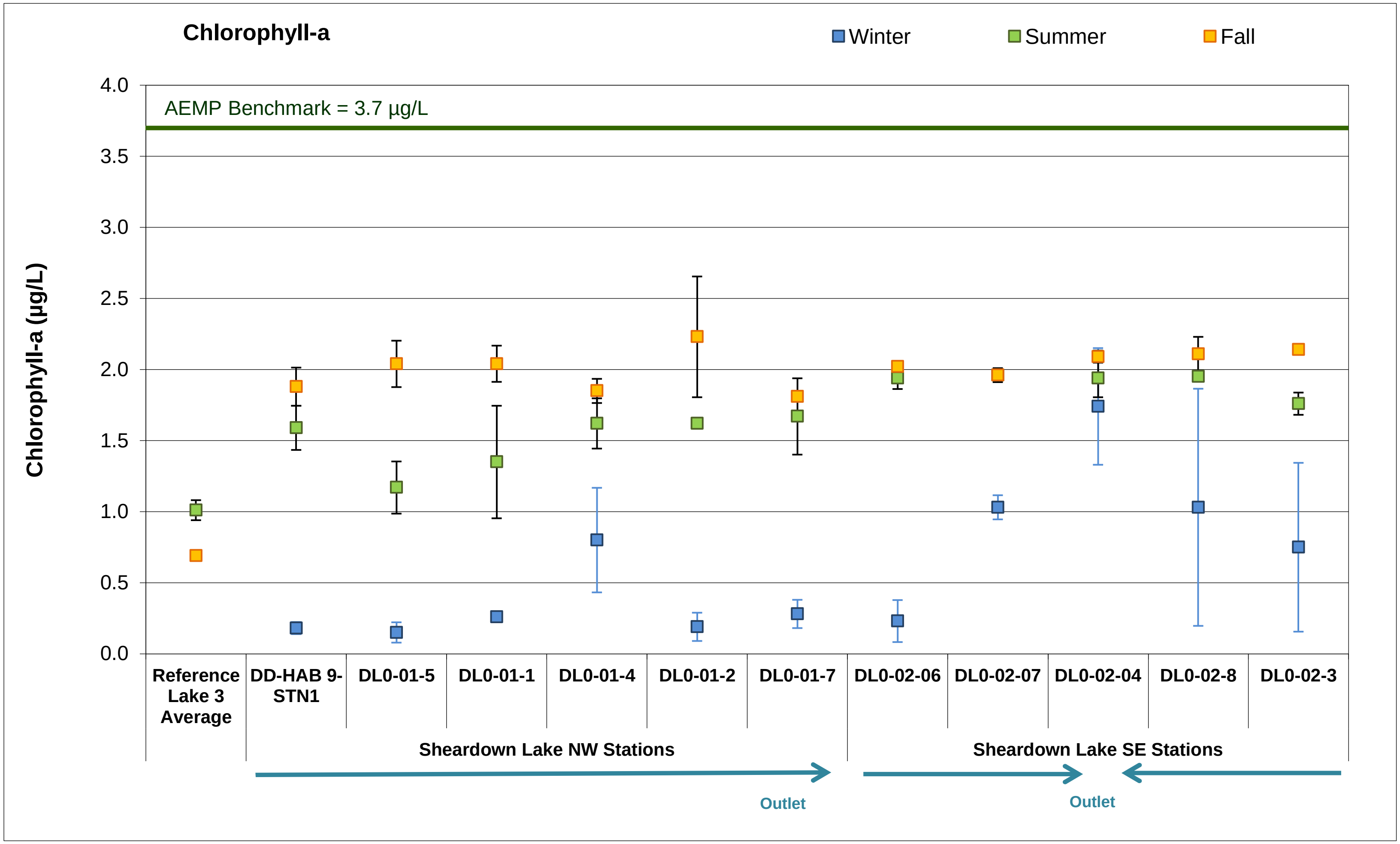


Figure 4.8: Chlorophyll-a Concentrations at Sheardown Lake NW (DL0-1) and Sheardown Lake SE (DL0-2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2021

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 = 6). Reference Lake 3 was not sampled in winter 2021.

occurring during the summer sampling event at the reference lake (Appendix Figure B.7). 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 2021 (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 in all seasons (Figure 4.8). 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 2021) classifications based on aqueous total phosphorus concentrations near the surface (i.e., concentrations below 10 µg/L; Table 4.3; Appendix Table C.42).

Chlorophyll-a concentrations at Sheardown Lake NW in 2021 were within the seasonal ranges shown among years of mine construction (2014) and previous mine operation (2015 to 2020), 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 4.5 and 4.6). Although no significant differences in richness or evenness were identified between Sheardown Lake NW and the reference lake for either littoral or profundal habitat (Tables 4.5 and 4.6), Bray-Curtis Index differed significantly between Sheardown Lake NW and Reference Lake 3 for both habitat types (Appendix Table F.21). No significant differences in the relative abundance of individual dominant taxonomic groups were indicated between Sheardown Lake NW and Reference Lake 3 littoral and profundal habitats except for significantly higher relative abundance of chironomids at littoral habitat of Sheardown Lake NW (Tables 4.5 and 4.6). Therefore, 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 suggested that Sheardown Lake NW was more productive than Reference Lake 3. This was supported by no significant differences in the relative abundance of metal-sensitive chironomids and FFG between lakes, and greater relative abundance of burrowers HPG at Sheardown Lake NW compared to the reference lake (Tables 4.5 and 4.6), which indicated no sediment metal-related influences or effects to available

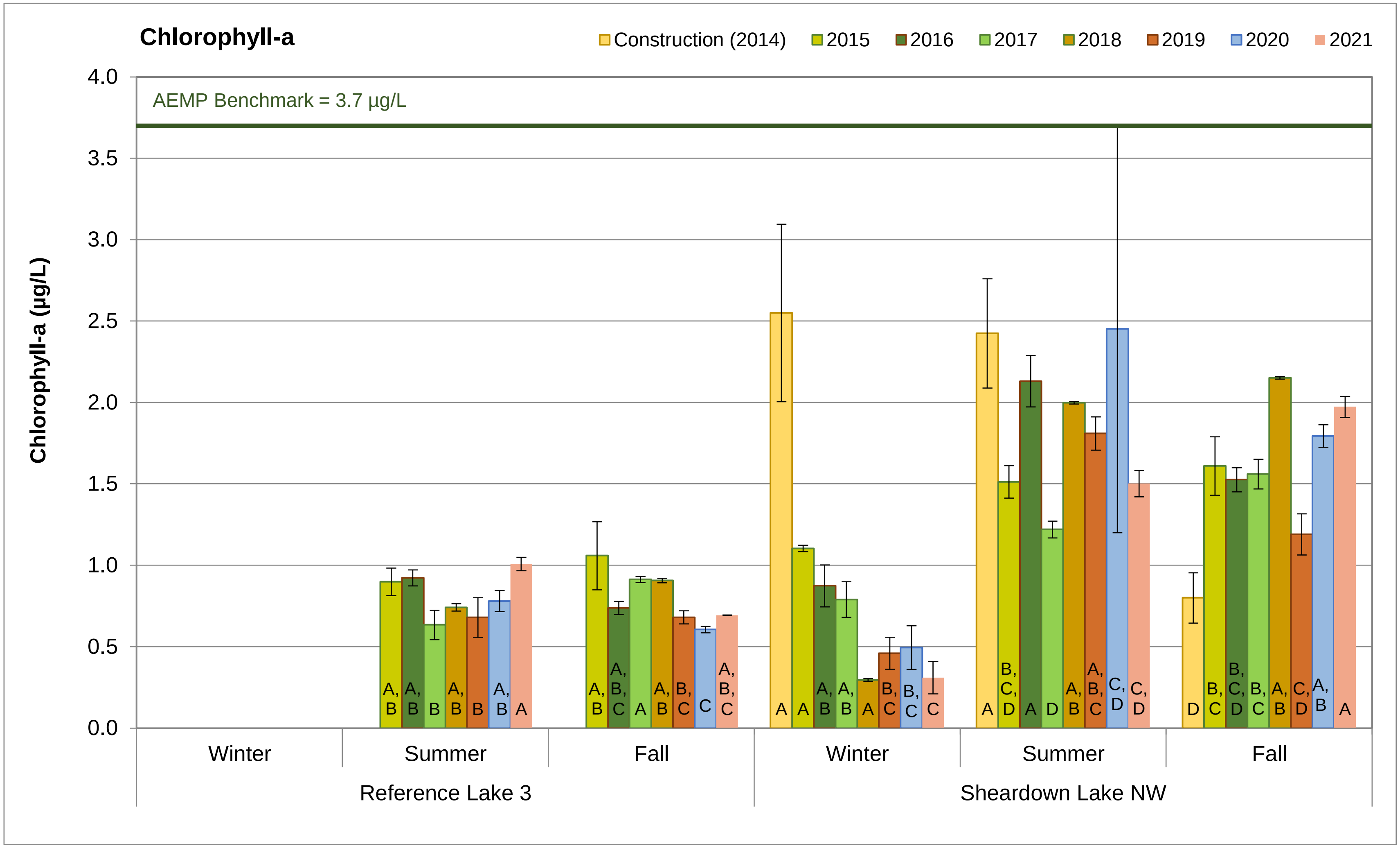


Figure 4.9: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Sheardown Lake NW and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2021) Periods (mean ± SE)

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

Table 4.5: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake NW (DL0-01) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	YES	0.013	9.7	Reference Lake 3	1,064	448	200	508	1,008	1,516
						Sheardown NW Littoral	5,399	3,000	1,342	1,894	6,510	8,301
Richness (Number of Taxa)	tequal	none	NO	0.449	0.5	Reference Lake 3	10.8	2.17	0.97	8	11	14
						Sheardown NW Littoral	11.8	1.79	0.8	9	13	13
Simpson's Evenness (E)	tequal	none	NO	0.556	0.3	Reference Lake 3	0.703	0.14	0.0626	0.502	0.775	0.838
						Sheardown NW Littoral	0.75	0.101	0.045	0.628	0.732	0.894
Shannon Diversity	tequal	log10	NO	0.384	0.5	Reference Lake 3	2.13	0.409	0.183	1.58	2.38	2.46
						Sheardown NW Littoral	2.32	0.234	0.105	1.97	2.41	2.52
Hydracarina (%)	tequal	log10(x+1)	NO	0.771	0.2	Reference Lake 3	3.1	2.2	1.0	0.0	3.4	5.7
						Sheardown NW Littoral	3.5	2.2	1.0	1.6	2.5	6.4
Ostracoda (%)	tequal	none	NO	0.174	-0.8	Reference Lake 3	46.6	27.9	12.5	1.7	47.7	72.7
						Sheardown NW Littoral	25.0	16.5	7.4	12.7	20.6	53.9
Chironomidae (%)	tequal	none	YES	0.009	2.4	Reference Lake 3	38.3	13.2	5.9	20.4	39.3	54.2
						Sheardown NW Littoral	70.1	16.0	7.2	41.5	77.0	79.1
Metal-Sensitive Chironomidae (%)	tequal	log10	NO	0.298	-0.5	Reference Lake 3	17.4	10.6	4.7	8.4	11.1	29.1
						Sheardown NW Littoral	11.8	10.0	4.5	4.1	6.8	28.6
Collector-Gatherers (%)	tequal	none	NO	0.546	-0.4	Reference Lake 3	82.0	9.7	4.4	66.7	87.2	90.0
						Sheardown NW Littoral	78.1	9.8	4.4	64.0	78.4	88.8
Filterers (%)	tequal	log10	NO	0.834	-0.1	Reference Lake 3	11.8	9.5	4.2	2.2	8.0	25.6
						Sheardown NW Littoral	11.2	10.5	4.7	1.4	6.8	28.6
Shredders (%)	tequal	log10(x+1)	NO	0.125	-0.8	Reference Lake 3	2.8	3.0	1.4	0.0	1.7	7.8
						Sheardown NW Littoral	0.5	0.7	0.3	0.0	0.0	1.4
Clingers (%)	tequal	none	NO	0.604	-0.3	Reference Lake 3	14.9	10.6	4.7	2.2	11.7	29.9
						Sheardown NW Littoral	11.9	6.2	2.8	5.5	12.9	21.0
Sprawlers (%)	tequal	log10	NO	0.15	-1.0	Reference Lake 3	64.8	22.8	10.2	30.5	71.6	85.6
						Sheardown NW Littoral	43.1	17.9	8.0	28.2	36.9	72.4
Burrowers (%)	tequal	log10	YES	0.072	1.5	Reference Lake 3	20.4	16.5	7.4	7.4	16.2	49.1
						Sheardown NW Littoral	45.0	21.7	9.7	14.3	50.2	64.9

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.

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - $MCT_{Ref)}$/SD_{Ref}. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases.

Table 4.6: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake NW (DL0-01) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	YES	0.068	2.1	Reference Lake 3	327	151	67.4	60.3	396	422
						Sheardown NW Profundal	648	304	136	198	732	964
Richness (Number of Taxa)	tequal	none	NO	0.269	0.7	Reference Lake 3	6.4	2.61	1.17	2	8	8
						Sheardown NW Profundal	8.2	2.17	0.97	5	8	11
Simpson's Evenness (E)	tequal	log10	NO	0.449	0.9	Reference Lake 3	0.545	0.113	0.0504	0.437	0.51	0.731
						Sheardown NW Profundal	0.647	0.218	0.0975	0.424	0.648	0.94
Shannon Diversity	tequal	none	NO	0.189	0.8	Reference Lake 3	1.38	0.508	0.227	0.592	1.46	2
						Sheardown NW Profundal	1.8	0.431	0.193	1.31	2.1	2.14
Hydracarina (%)	tequal	log10(x+1)	NO	0.242	0.9	Reference Lake 3	5.3	4.3	1.9	0.0	4.4	11.9
						Sheardown NW Profundal	9.1	5.2	2.3	4.4	8.2	17.2
Ostracoda (%)	M-W	rank	NO	0.548	0.3	Reference Lake 3	8.2	8.7	3.9	0.0	4.4	20.4
						Sheardown NW Profundal	10.5	9.1	4.1	3.1	5.4	21.7
Chironomidae (%)	tequal	none	NO	0.362	-0.8	Reference Lake 3	86.0	8.7	3.9	71.4	88.1	93.5
						Sheardown NW Profundal	79.4	12.5	5.6	60.3	83.9	91.8
Metal-Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.589	-0.4	Reference Lake 3	7.8	6.2	2.8	0.0	10.9	14.3
						Sheardown NW Profundal	5.5	6.8	3.1	0.0	3.5	17.4
Collector-Gatherers (%)	tequal	log10	NO	0.291	-0.6	Reference Lake 3	86.3	7.8	3.5	80.8	83.7	100.0
						Sheardown NW Profundal	81.6	5.5	2.5	73.9	84.7	85.9
Filterers (%)	tequal	none	YES	0.036	1.7	Reference Lake 3	6.48	4.93	2.21	0	8.90	10.9
						Sheardown NW Profundal	14.9	5.59	2.50	7.07	15.3	21.1
Shedders (%)	tequal	none	NO	0.210	-0.6	Reference Lake 3	0.880	1.21	0.539	0	0	2.23
						Sheardown NW Profundal	0.126	0.281	0.126	0	0	0.629
Clingers (%)	tequal	none	NO	0.557	-0.4	Reference Lake 3	12.3	7.0	3.1	0.0	15.2	16.8
						Sheardown NW Profundal	9.8	5.7	2.6	4.4	8.2	17.2
Sprawlers (%)	tequal	none	NO	0.109	-3.7	Reference Lake 3	86.0	8.4	3.7	77.5	84.5	100.0
						Sheardown NW Profundal	55.5	36.9	16.5	11.8	75.9	91.8
Burrowers (%)	tequal	log10(x+1)	YES	0.06	12.4	Reference Lake 3	1.7	2.7	1.2	0.0	0.0	6.1
						Sheardown NW Profundal	34.7	35.7	16.0	3.1	17.4	80.0

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.

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases. Endpoints that did not have data in both lakes were excluded from the analyses.

food resources, respectively, on the benthic invertebrate community of Sheardown Lake NW in 2021.

Significant differences in benthic invertebrate richness and evenness were identified between Sheardown Lake NW and reference lake littoral stations over years of mine operation (2015 to 2021) compared to baseline studies conducted in 2007, 2008, and 2013 (Appendix Figure F.10; Appendix Table F.37), however these differences were within the CES_{BIC} and therefore not ecologically meaningful. At profundal stations of Sheardown Lake NW, significant differences among years were identified for benthic invertebrate density and the relative community contributions of nematodes, Ostracods, chironomids, metal sensitive chironomids, and collector gatherers. Only benthic invertebrate densities and the relative abundances of metal-sensitive chironomids routinely differed by more than the CES_{BIC} between the mine operational period and both years of mine baseline (densities were significantly lower and metal sensitive chironomids were significantly higher relative to 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, minimal ecologically 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 2021 (Table 4.7), reflecting the same fish species that were observed historically (Minnow 2021b). Arctic charr and ninespine stickleback CPUE were higher at Sheardown Lake NW than at the reference lake in 2021 (Table 4.7), 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 2021 was within the range observed over the mine baseline period (2007 to 2013) and previous years of mine operation (2015 to 2020; Figure 4.10). Gill netting CPUE for arctic charr in 2021 was also within the range observed during baseline, but slightly greater than the previous six years of mine operation (Figure 4.10). The similarities in CPUE among study years suggested that the relative abundance of arctic charr

Table 4.7: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Sheardown Lake NW (DL0-01), Sheardown Lake SE (DL0-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	115	4	119	2
		CPUE	1.07	0.040	1.11	
	Gill netting	No. Caught	56	-	56	
		CPUE	1.65	-	1.65	
Sheardown Lake Northwest	Electrofishing	No. Caught	116	3	119	2
		CPUE	2.36	0.060	2.42	
	Gill netting	No. Caught	131	-	131	
		CPUE	7.82	-	7.82	
Sheardown Lake Southeast	Electrofishing	No. Caught	108	77	185	2
		CPUE	1.09	0.78	1.87	
	Gill netting	No. Caught	148	-	148	
		CPUE	10.35	-	10.35	

Note: "-" indicates not applicable.

[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.

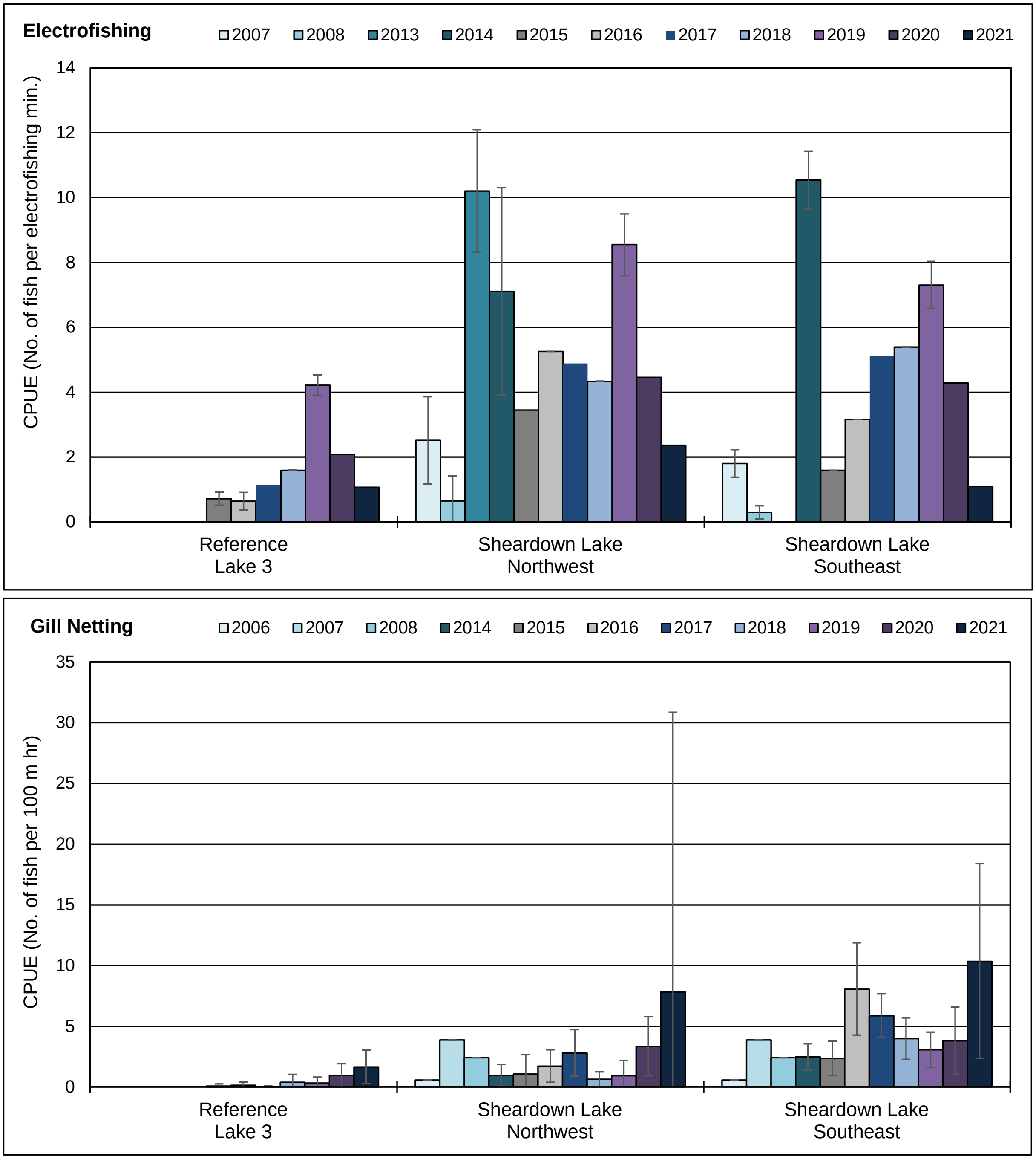


Figure 4.10: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Back-pack Electrofishing and Gill Netting at Sheardown Lake NW (DL0-01) and Sheardown Lake SE (DL0-02), Mary River Project CREMP, 2006 to 2021

Notes: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015 to 2021) 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 Health Assessment

Nearshore Arctic Charr

A total of 116 and 115 arctic charr were captured from nearshore habitat of each of Sheardown Lake NW and Reference Lake 3, respectively, in August 2021 (Table 4.8). Distinguishing arctic charr YOY from the older, non-YOY age class was possible using a fork length cut-off of 4.8 cm and 4.0 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 captured 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 4.8; Appendix Table G.14). 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 than those from the reference lake (Table 4.8; Appendix Table G.14). Additionally, non-YOY arctic charr from nearshore habitats in Sheardown Lake NW had significantly greater condition than those from the reference lake, although the magnitudes of the differences in condition between lake populations were not considered ecologically significant because they were within CES_C (i.e., ±10%; Table 4.8; 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 4.8; Appendix Table G.14). Although the lengths and weights of non-YOY arctic charr in years of mine operation (i.e., 2015 to 2021) 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 the CES_C of ±10% (Table 4.8). 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, as well as the likelihood of natural variability in fish population dynamics, resulted in uncertainty as to whether current mine operations have affected non-YOY arctic charr health at Sheardown Lake NW.

Table 4.8: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake NW and Reference Lake 3 from 2015 to 2021, 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	2021	2015	2016	2017	2018	2019	2020	2021
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**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%)**	**Yes (+48%)**	No	No	No	**Yes (-12%)**	No	**Yes (+13%)**	No
		Size (mean weight)	**Yes (+121%)**	**Yes (+60%)**	No	**Yes (+83%)**	**Yes (-24%)**	**Yes (+99%)**	**Yes (+234%)**	No	**Yes (-29%)**	No	**Yes (-50%)**	No	No	No
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	**Yes (+7%)**	**Yes (-5%)**	**Yes (+4%)**	**Yes (+10%)**	**Yes (+4.6%)**	**Yes (-13%)**	**Yes (-12%)**	**Yes (-9%)**	**Yes (-10%)**	**Yes (-13%)**	**Yes (-9%)**	**Yes (-7.5%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	No	**Yes**
		Age	-	-	-	-	-	-	-	**Yes (-35%)**	**Yes (-28%)**	**Yes (-26%)**	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (+22%)**	**Yes (+18%)**	**Yes (+13%)**	**Yes (-21%)**	**Yes (-14%)**	**Yes (-6%)**	No	No	No	**Yes (-7.7%)**
		Size (mean weight)	-	-	-	No	**Yes (+92%)**	**Yes (+94%)**	**Yes (+68%)**	**Yes (-47%)**	**Yes (-31%)**	**Yes (-9%)**	No	No	No	**Yes (-20%)**
		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 (+20%)**	**Yes (+8%)**	**Yes (+11%)**	**Yes (+6%)**	No	No	No	**Yes (+6.0%)**

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 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.

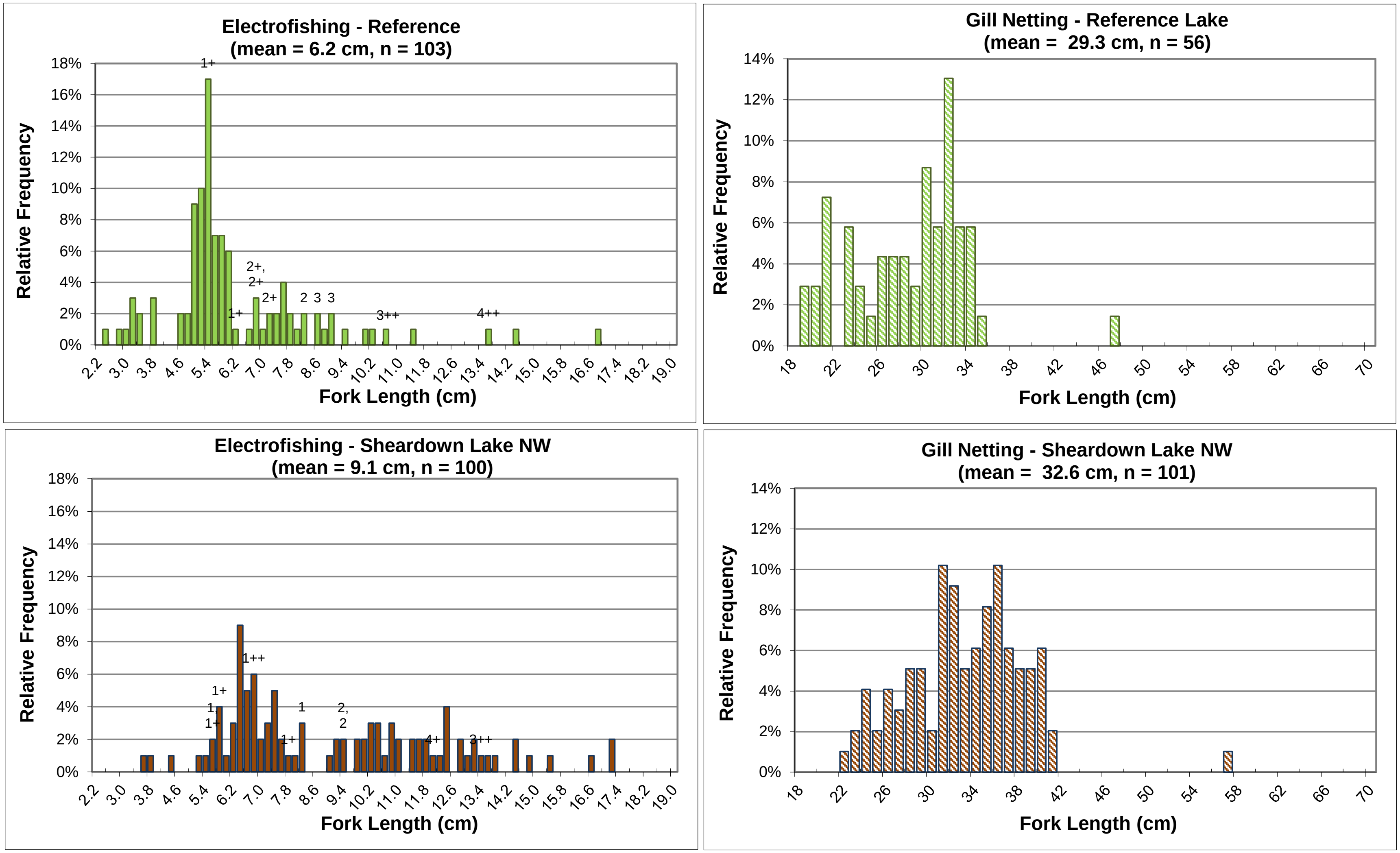


Figure 4.11: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake NW (DL0-01) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

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

Littoral/Profundal Arctic Charr

A total of 101 and 56 arctic charr were sampled from littoral/profundal habitat of Sheardown Lake NW and Reference Lake 3, respectively, in August 2021 (Table 4.7). 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 4.8; Figure 4.11). Arctic charr captured by gill net at Sheardown Lake NW in 2021 were significantly larger and had greater condition than those captured at Reference Lake 3 (Table 4.8; Appendix Table G.18). The magnitude of difference in condition (20%) was outside of the CES_C of ±10%, suggesting an 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 2021 were similar to the differences shown in 2018, 2019, and/or 2020, suggesting no appreciable changes in health of littoral/profundal arctic charr at Sheardown Lake NW over time. Body size (mean fork length and weight) was significantly lower for fish captured at Sheardown Lake NW in 2021 compared to baseline, whereas condition of arctic charr captured from littoral/profundal habitat of Sheardown Lake NW was significantly greater in 2021 relative to the baseline period (Table 4.8; Appendix Figure G.10; Appendix Table G.18). Similarly, 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 4.8). There were no differences in size and condition of arctic charr from Sheardown Lake NW over the 2018 to 2020 period compared to 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 2021 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 2021:

- 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;

- Copper concentration in sediment was greater than the benchmark of 58 mg/kg at one littoral monitoring station (DL0-01-9), although the average concentration of copper in sediment at littoral stations was below this benchmark;
- Iron concentrations in sediment were greater than the benchmark of 52,200 mg/kg at three littoral stations (DD-HAB 9-STN2, DL0-01-8, and DL0-01-9), although average concentrations of iron in sediment at littoral stations were below this benchmark;
- Nickel concentration in sediment was greater than the benchmark of 77 mg/kg at one littoral monitoring station (DL0-01-9), 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 2021 at Sheardown Lake NW. Lake-specific AEMP benchmarks for sediment quality were exceeded for arsenic, copper, iron, and nickel at as many as three littoral stations in 2021, 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 2021 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 2021, 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 (DL0-2)

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 2021 (Appendix Figures C.17 to C.20). The 2021 Sheardown SE water column profiles indicated a slight increase in temperature from surface to bottom (i.e., approximately 0.5˚C) during the winter sampling event (Figure 4.12). No thermal stratification developed at Sheardown Lake SE during the summer or fall 2021 sampling events, whereas Reference Lake 3 had a warmer surface layer

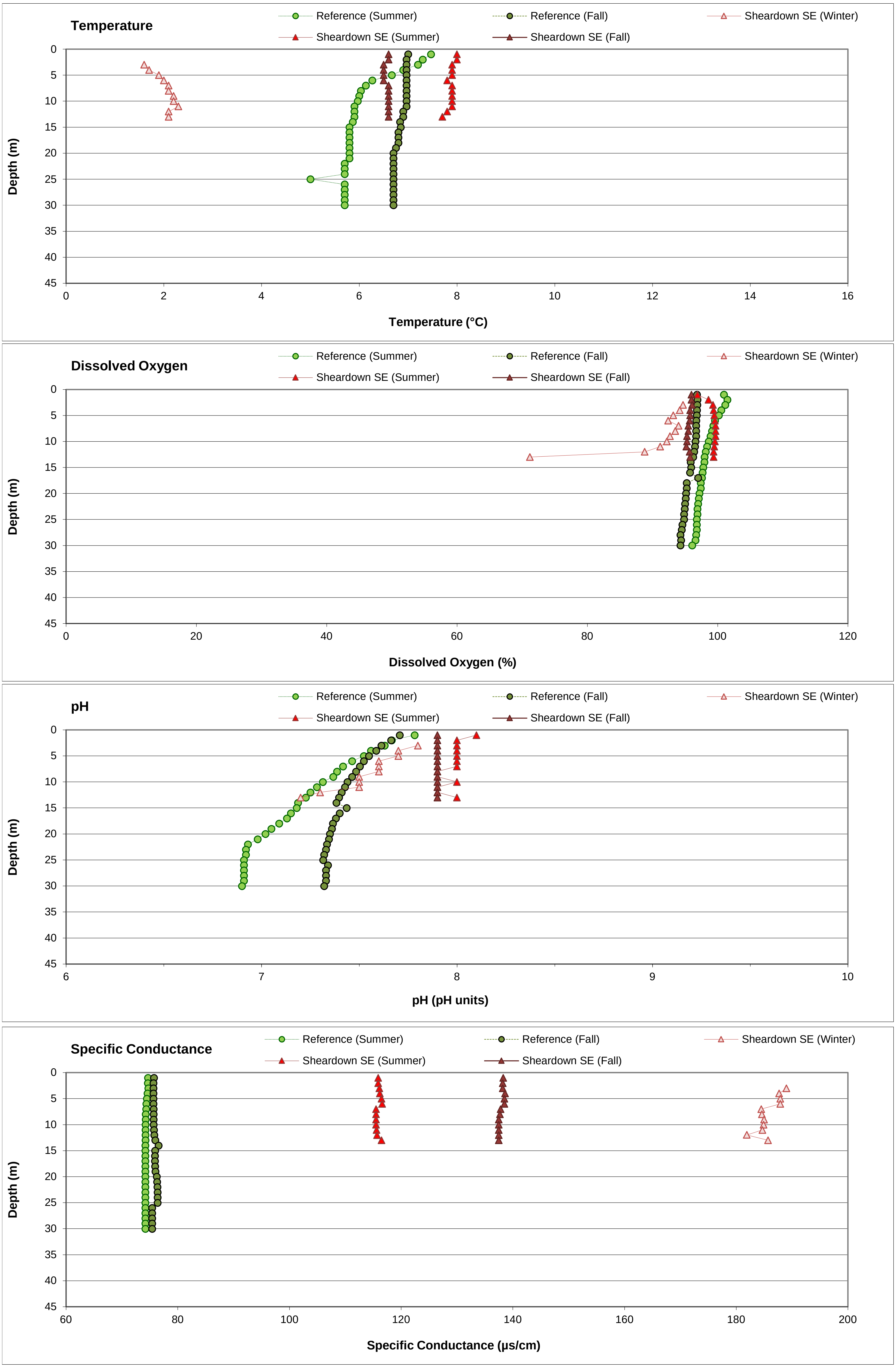


Figure 4.12: Average In Situ Water Quality with Depth from Surface at Sheardown Lake SE (DL0-02) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

that extended to a depth of approximately 6 m during summer sampling (Figure 4.12). The average water temperature at the bottom of the water column at Sheardown Lake SE littoral and profundal stations was significantly warmer than at Reference Lake 3 during the August 2021 biological study (Figure 4.5; Appendix Tables C.49 and 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 2021 showed a gradient of decreasing saturation levels with increased depth in winter, but not in the summer or fall sampling events when dissolved oxygen saturation profiles showed well oxygenated water extending from surface to bottom similar to profiling conducted at the reference lake (Figure 4.12). Dissolved oxygen concentrations near the bottom of the water column at Sheardown Lake SE did not differ significantly from the reference lake for littoral stations, but were significantly lower than at Reference Lake 3 for profundal stations during the August 2021 biological study (Figure 4.5; Appendix Tables C.49 and C.51). 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).

In 2021, water column profiles showed decreasing pH with increased depth at Sheardown Lake SE during the winter sampling event, but not in the summer or fall sampling events when no substantial change in pH occurred from surface to bottom, which differed from the pH profiles conducted at Reference Lake 3 (Figure 4.12). The changes in pH through the water column at both lakes appeared to correlate 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 station types at the reference lake during the August 2021 biological study (Figure 4.5). However, the mean incremental difference in bottom pH between lakes was less than a half 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 and Reference Lake 3 did not differ 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 2021 biological study (Figure 4.5). Secchi depth at Sheardown Lake SE was significantly lower, indicating less water clarity, than at the reference lake during the August 2021 biological study (Appendix Figure C.8). Indeed, water clarity at Sheardown Lake SE was the lowest among all CREMP study lakes in 2021 and historically, which reflects backflow containing naturally high suspended sediment entering the lake from Mary River during high flow periods.

Water chemistry at Sheardown Lake SE met all AEMP benchmarks and WQG over the duration of spring, summer, and fall sampling events in 2021 (Table 4.9; Appendix Table C.52). 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 4.9; Appendix Table C.43). Of those parameters without AEMP benchmarks, turbidity and concentrations of total and dissolved manganese, molybdenum, and uranium were elevated at Sheardown Lake SE compared to the reference lake during summer and/or fall sampling events in 2021 (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 2021 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 4.9; Appendix Table C.52).[19] Similar dissolved aluminum concentrations between Sheardown Lake SE and the reference lake in 2021 (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 2021 compared to baseline at Sheardown Lake SE (Appendix Figure C.21), as were dissolved concentrations of aluminum, manganese, and molybdenum in at least one seasonal sampling event among those parameters without AEMP benchmarks (Appendix Tables C.43 and C.54). Of the parameters elevated at Sheardown Lake SE in 2021 compared to baseline and the reference lake, sulphate concentrations have recently shown increasing trends in groundwater adjacent to Sheardown Lake SE (Tetra Tech 2022), suggesting that a nearby landfill was a possible source of sulphate. However, no other parameters that exhibited increasing concentrations in groundwater over time at wells located adjacent to Sheardown Lake NW (i.e., dissolved iron, nickel, and uranium) showed elevation and/or increasing trends in surface water of Sheardown Lake SE in 2021.

Overall, a slight mine-related influence on water quality of Sheardown Lake SE was indicated in 2021 as reflected by elevated concentrations of manganese, molybdenum, nitrate, sulphate, and uranium. 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.

[19] Turbidity at Sheardown Lake SE in 2021 was comparable to turbidity shown at the lake during baseline (Appendix Table C.43), suggesting that greater turbidity at this lake compared to Reference Lake 3 reflects a natural phenomenon.

Table 4.9: Mean Water Chemistry at Sheardown Lake SE (DL0-02) and Reference Lake 3 (REF3) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

	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	-	-	79.1	83.3	196	118	145
	pH (lab)	pH	6.5 - 9.0	-	7.71	7.68	7.54	7.91	8.01
	Hardness (as $CaCO_3$)	mg/L	-	-	36.8	37.7	94.3	51.4	64.7
	Total Suspended Solids (TSS)	mg/L	-	-	<2	<2	<2	<2	<2
	Total Dissolved Solids (TDS)	mg/L	-	-	29.0	41.0	103	70.0	72.4
	Turbidity	NTU	-	-	0.240	0.183	0.122	2.21	0.894
	Alkalinity (as $CaCO_3$)	mg/L	-	-	37.3	46.3	82.5	45.3	55.4
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.0140	<0.01	0.0135	<0.005	<0.005
	Nitrate	mg/L	3	3	<0.02	<0.02	0.222	0.117	0.173
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	0.00128	<0.001	<0.001
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.177	0.200	0.128	0.123	0.113
	Dissolved Organic Carbon	mg/L	-	-	3.00	4.33	1.64	1.34	3.35
	Total Organic Carbon	mg/L	-	-	3.46	3.49	1.55	1.13	1.77
	Total Phosphorus	mg/L	0.020[α]	-	0.00443	0.00303	0.0022	0.0025	0.0025
	Phenols	mg/L	0.004[α]	-	0.00113	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.05	<0.05	<0.05
	Chloride (Cl)	mg/L	120	120	1.32	1.35	5.21	2.84	3.69
	Sulphate (SO_4)	mg/L	218[β]	218	3.68	3.69	13.1	8.36	10.7
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.00373	0.00370	0.00676	0.0499	0.0269
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	0.0000200	<0.00002	<0.00002
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.0000752	0.0000500	0.0000600
	Barium (Ba)	mg/L	-	-	0.00676	0.00651	0.00938	0.00605	0.00715
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.000005	<0.000005	<0.000005
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.000005	<0.000005	<0.000005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	0.0176	0.0104	0.0133
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.00001	<0.00001	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	7.40	7.43	18.6	10.0	12.0
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	0.000102	0.000140	0.000160
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.0001	<0.0001	0.0000199	0.0000418	0.0000283
	Copper (Cu)	mg/L	0.002	0.0024	0.000810	0.000750	0.000861	0.000715	0.000860
	Iron (Fe)	mg/L	0.300	0.300	<0.03	<0.03	0.0161	0.0601	0.0315
	Lead (Pb)	mg/L	0.001	0.001	<0.00005	<0.00005	<0.00001	0.0000524	0.0000270
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	0.00159	0.00104	0.00125
	Magnesium (Mg)	mg/L	-	-	4.67	4.77	12.3	6.69	8.32
	Manganese (Mn)	mg/L	0.935[β]	-	0.000674	0.000591	0.00194	0.00311	0.00196
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000155	0.000157	0.000797	0.000546	0.000685
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000697	0.000524	0.000569
	Potassium (K)	mg/L	-	-	0.960	0.910	1.61	1.08	1.20
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	0.0000416	<0.00004	0.0000404
	Silicon (Si)	mg/L	-	-	0.507	0.490	0.547	0.551	0.540
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.000005	<0.000005	<0.000005
	Sodium (Na)	mg/L	-	-	0.994	0.929	2.50	1.50	1.79
	Strontium (Sr)	mg/L	-	-	0.00877	0.00856	0.0131	0.00827	0.0104
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.000005	<0.000005	<0.000005
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	0.0000214	<0.00002	<0.00002
	Titanium (Ti)	mg/L	-	-	<0.01	<0.01	0.000358	0.00303	0.00142
	Uranium (U)	mg/L	0.015	-	0.000328	0.000342	0.00127	0.000848	0.00125
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	0.0000652	0.000159	0.0000852
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	0.000606	0.000616	<0.0005

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).

4.3.2 Sediment Quality

Surficial sediment collected from Sheardown Lake SE in 2021 was primarily composed of silt (Figure 4.13; Appendix Table D.17). Sediment particle size at littoral and profundal stations of Sheardown Lake SE differed significantly from like-habitat at Reference Lake 3 except for the clay-sized fraction of littoral sediments (Appendix Table D.18). The TOC contents in both littoral and profundal sediments at Sheardown Lake SE were significantly lower than the respective TOC contents in sediment at Reference Lake 3, but no significant difference in sediment moisture content was indicated between lakes for either habitat (Appendix Table D.18). 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 substantial quantities of naturally suspended, fine-grained material. An orange-brown layer of oxidized material, likely composed of primarily iron (oxy)hydroxide precipitate, was observed on the surface of Sheardown Lake SE sediments at all five littoral and profundal stations, consistent with the substrate characteristics observed at Reference Lake 3 (Appendix Tables D.16).

Metal concentrations in sediment at Sheardown Lake SE showed no clear spatial gradients with progression towards the lake outlet in 2021, suggesting no point sources of metals to the lake (Appendix Table D.17). Metal concentrations in sediment at Sheardown Lake SE were comparable to those observed at Reference Lake 3 (Table 4.10; Appendix Table D.19). Mean concentrations of iron in both littoral and profundal sediments at Sheardown Lake SE were higher than applicable AEMP benchmarks and SQG; however, concentrations were lower than mean concentrations of iron in sediment at Reference Lake 3, indicating iron is naturally elevated in the study area (Table 4.10; Appendix Table D.17). Mean concentrations of chromium and manganese in both littoral and profundal sediments of Sheardown Lake SE were above AEMP benchmarks, but were lower than respective SQG (Table 4.10; Appendix Table D.17). Notably, the mean concentration of manganese in sediment at Reference Lake 3 was above the AEMP benchmark, half of the samples had concentrations of manganese above the SQG, and one sample had a concentration of chromium above the AEMP benchmark (Table 4.11; Appendix Table D.4). This suggested that the elevation of manganese and chromium 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.

Mean metal concentrations in sediments at both littoral and profundal stations at Sheardown Lake SE in 2021 were comparable (i.e., less than 3-times greater) to those reported for the baseline period (2005 to 2013) and were also within respective ranges observed in previous years

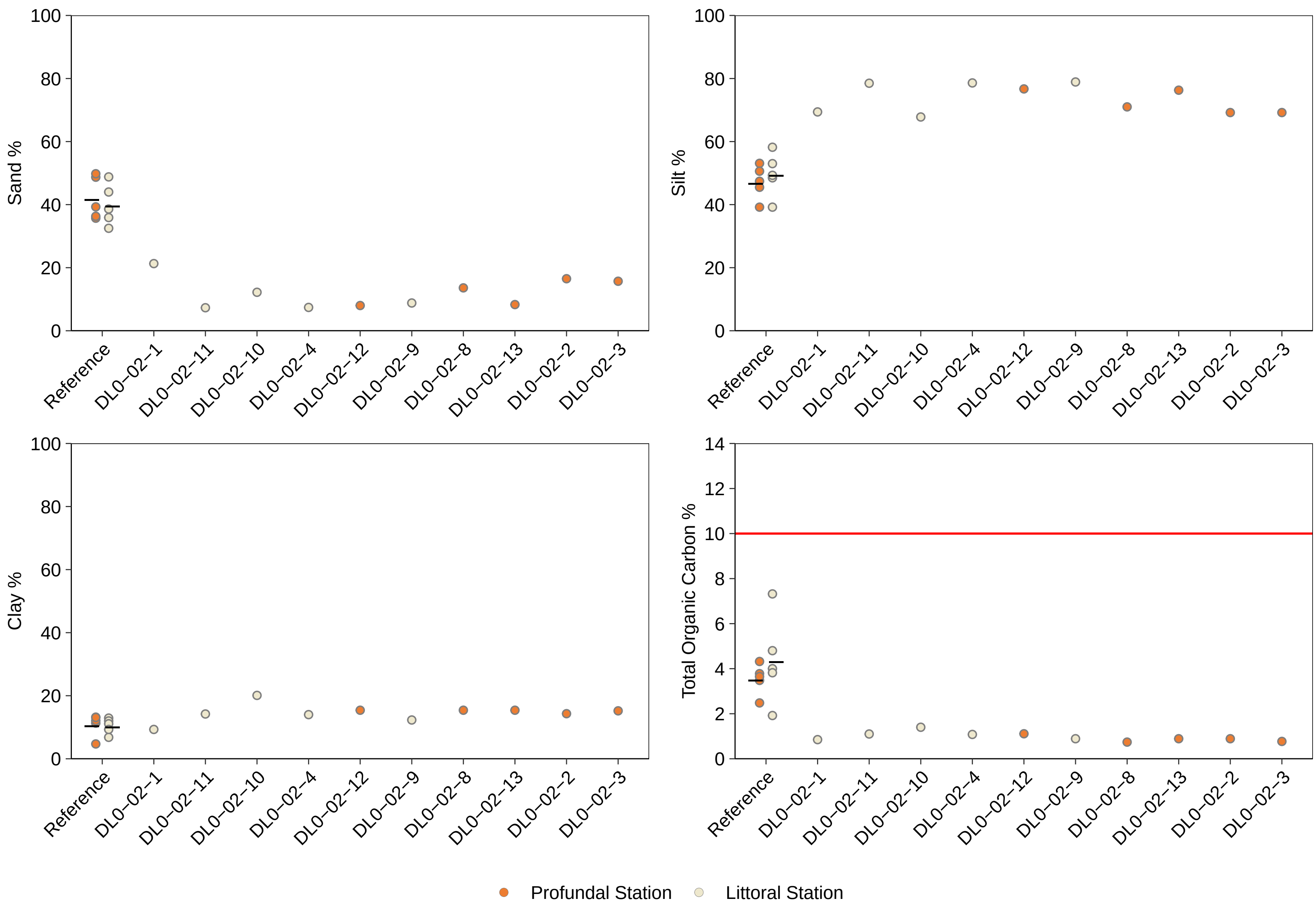


Figure 4.13: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake SE (DL0−02) Sediment Monitoring Stations and Reference Lake 3, Mary River Project CREMP, August 2021

Note: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

Table 4.10: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Sheardown Lake SE (DL0-02), and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2021

	Analyte	Units	SQG[a]	AEMP Benchmark[b]	Littoral: Reference Lake (n = 5) Average ± SD	Littoral: Sheardown Lake SE (n = 3) Average ± SD	Profundal: Reference Lake (n = 5) Average ± SD	Profundal: Sheardown Lake SE (n = 2) Average ± SD
TOC		%	10[α]	-	3.70 ± 1.09	1.01 ± 0.139	4.21 ± 1.83	0.83 ± 0.085
Metals	Aluminum (Al)	mg/kg	-	-	21,120 ± 4,592	18,767 ± 3,465	20,520 ± 4,129	19,600 ± 141
	Antimony (Sb)	mg/kg	-	-	<0.100 ± 0	<0.100 ± 0	<0.100 ± 0	<0.100 ± 0
	Arsenic (As)	mg/kg	17	5.9	5.54 ± 1.03	4.10 ± 1.26	5.13 ± 0.978	3.78 ± 0.417
	Barium (Ba)	mg/kg	-	-	123 ± 29.4	108 ± 31.6	129 ± 13.4	92.9 ± 4.53
	Beryllium (Be)	mg/kg	-	-	0.812 ± 0.176	0.837 ± 0.110	0.782 ± 0.122	0.795 ± 0.007
	Bismuth (Bi)	mg/kg	-	-	<0.200 ± 0	0.223 ± 0.025	<0.200 ± 0	0.215 ± 0.021
	Boron (B)	mg/kg	-	-	14.9 ± 2.68	22.8 ± 4.17	15.2 ± 2.23	22.9 ± 0.778
	Cadmium (Cd)	mg/kg	3.5	1.5	0.144 ± 0.046	0.109 ± 0.018	0.167 ± 0.048	0.112 ± 0.006
	Calcium (Ca)	mg/kg	-	-	5,026 ± 849	6,287 ± 1,739	5,186 ± 660	6,420 ± 354
	Chromium (Cr)	mg/kg	90	79	67.2 ± 15.0	***82.2*** ± 5.10	64.6 ± 10.7	***81.7*** ± 1.34
	Cobalt (Co)	mg/kg	-	-	14.8 ± 3.35	14.4 ± 1.69	13.8 ± 3.48	15.6 ± 0.141
	Copper (Cu)	mg/kg	110	56	***81.2*** ± 27.5	30.4 ± 4.46	***85.0*** ± 10.5	30.5 ± 0
	Iron (Fe)	mg/kg	40,000[α]	34,400	***44,360*** ± 8,848	***40,733*** ± 8,501	***40,900*** ± 11,600	***41,200*** ± 5,091
	Lead (Pb)	mg/kg	91.3	35	17.3 ± 2.81	17.8 ± 2.78	16.2 ± 2.46	17.0 ± 0.141
	Lithium (Li)	mg/kg	-	-	34.7 ± 5.36	32.5 ± 4.36	33.3 ± 5.37	30.9 ± 0
	Magnesium (Mg)	mg/kg	-	-	13,760 ± 2,748	14,900 ± 529	12,918 ± 2,534	15,700 ± 141
	Manganese (Mn)	mg/kg	1,100[α,β]	657	***958*** ± 402	***1,004*** ± 538	***882*** ± 346	***981*** ± 380
	Mercury (Hg)	mg/kg	0.486	0.17	0.052 ± 0.024	0.022 ± 0.004	0.045 ± 0.010	0.023 ± 0
	Molybdenum (Mo)	mg/kg	-	-	3.23 ± 0.556	1.58 ± 0.518	2.69 ± 0.733	1.43 ± 0.424
	Nickel (Ni)	mg/kg	75[α,β]	66	46.6 ± 10.1	65.8 ± 4.55	45.9 ± 6.89	65.7 ± 2.83
	Phosphorus (P)	mg/kg	2,000[α]	1,278	955 ± 154	1,017 ± 146	902 ± 111	1,040 ± 28.3
	Potassium (K)	mg/kg	-	-	5,140 ± 946	4,637 ± 866	4,862 ± 765	4,625 ± 35.4
	Selenium (Se)	mg/kg	-	-	0.628 ± 0.252	0.213 ± 0.023	0.554 ± 0.140	0.200 ± 0
	Silver (Ag)	mg/kg	-	-	0.202 ± 0.077	0.120 ± 0.010	0.172 ± 0.050	0.115 ± 0.007
	Sodium (Na)	mg/kg	-	-	394 ± 80.1	349 ± 25.9	362 ± 58.6	321 ± 12.7
	Strontium (Sr)	mg/kg	-	-	13.5 ± 2.11	12.6 ± 1.00	13.2 ± 1.41	12.5 ± 0.424
	Sulphur (S)	mg/kg	-	-	1,340 ± 297	<1000.000 ± 0	1,380 ± 444	<1,000.000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.594 ± 0.205	0.432 ± 0.078	0.556 ± 0.129	0.400 ± 0.003
	Tin (Sn)	mg/kg	-	-	<2.000 ± 0	<2.000 ± 0	<2.000 ± 0	<2.000 ± 0
	Titanium (Ti)	mg/kg	-	-	1,204 ± 112	1,387 ± 155	1,096 ± 158	1,520 ± 42.4
	Uranium (U)	mg/kg	-	-	22.0 ± 8.39	5.40 ± 0.806	20.3 ± 7.92	5.50 ± 0
	Vanadium (V)	mg/kg	-	-	63.2 ± 11.0	55.1 ± 7.53	60.7 ± 9.23	56.5 ± 1.91
	Zinc (Zn)	mg/kg	315	135	88.0 ± 18.2	61.9 ± 9.60	87.5 ± 11.8	63.0 ± 0.636
	Zirconium (Zr)	mg/kg	-	-	5.16 ± 1.07	20.3 ± 1.36	4.66 ± 1.28	22.2 ± 2.05

(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

[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 SI

Table 4.11: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake SE (DL0-02) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	YES	<0.001	9.4	Reference Lake 3	1,064	448	200	508	1,008	1,516
						Sheardown SE Littoral	5,253	858	384	3,987	5,304	6,355
Richness (Number of Taxa)	tequal	none	NO	0.376	0.6	Reference Lake 3	10.8	2.2	1.0	8.0	11.0	14.0
						Sheardown SE Littoral	12.0	1.9	0.8	9.0	12.0	14.0
Simpson's Evenness (E)	tequal	none	NO	0.142	0.8	Reference Lake 3	0.703	0.14	0.0626	0.502	0.775	0.838
						Sheardown SE Littoral	0.811	0.0502	0.0224	0.748	0.816	0.878
Shannon Diversity	tequal	none	NO	0.231	0.7	Reference Lake 3	2.13	0.41	0.18	1.58	2.38	2.46
						Sheardown SE Littoral	2.42	0.29	0.13	2.13	2.38	2.90
Hydracarina (%)	tequal	log10(x+1)	NO	0.614	0.3	Reference Lake 3	3.1	2.2	1.0	0.0	3.4	5.7
						Sheardown SE Littoral	3.7	1.5	0.7	1.9	3.6	6.1
Ostracoda (%)	M-W	rank	NO	0.151	-1.5	Reference Lake 3	46.6	27.9	12.5	1.7	47.7	72.7
						Sheardown SE Littoral	5.6	4.7	2.1	2.1	4.5	13.6
Chironomidae (%)	tequal	none	YES	<0.001	3.9	Reference Lake 3	38.3	13.2	5.9	20.4	39.3	54.2
						Sheardown SE Littoral	89.7	5.2	2.3	81.3	91.0	94.2
Metal-Sensitive Chironomidae (%)	tequal	none	NO	0.466	-0.4	Reference Lake 3	17.4	10.6	4.7	8.4	11.1	29.1
						Sheardown SE Littoral	13.1	6.9	3.1	2.6	14.5	19.1
Collector-Gatherers (%)	tequal	log10	YES	<0.001	-2.8	Reference Lake 3	82.0	9.7	4.4	66.7	87.2	90.0
						Sheardown SE Littoral	54.9	5.5	2.5	48.5	54.9	63.1
Filterers (%)	tequal	none	NO	0.818	0.1	Reference Lake 3	11.8	9.5	4.2	2.2	8.0	25.6
						Sheardown SE Littoral	13.0	7.0	3.1	2.2	14.5	19.1
Clingers (%)	tequal	none	NO	0.897	0.1	Reference Lake 3	14.9	10.6	4.7	2.2	11.7	29.9
						Sheardown SE Littoral	15.6	4.9	2.2	8.2	16.0	21.3
Sprawlers (%)	tequal	none	YES	0.08	-0.9	Reference Lake 3	64.8	22.8	10.2	30.5	71.6	85.6
						Sheardown SE Littoral	44.0	4.5	2.0	37.7	46.0	48.9
Burrowers (%)	M-W	rank	NO	0.151	1.2	Reference Lake 3	20.4	16.5	7.4	7.4	16.2	49.1
						Sheardown SE Littoral	40.5	4.3	1.9	35.1	39.8	45.6

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.

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases.

of mine operation (2015 to 2020; Figure 4.14; Appendix Table D.32).[20] 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 2021 period of mine operation relative to baseline (Figure 3.8). Overall, there were no substantial changes to metal concentrations in sediments at Sheardown Lake SE 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 summer and fall sampling events in 2021, whereas during the winter sampling event, chlorophyll-a concentrations appeared to increase with proximity to the outlet (Figure 4.8). Chlorophyll-a concentrations at Sheardown Lake SE in 2021 did not differ significantly between the summer and fall sampling events, or between the summer and winter sampling events, but concentrations in winter were significantly lower than in fall (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 2021 (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 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 2021) trophic classifications defined by total phosphorus concentrations (i.e., average concentrations below 10 µg/L; Table 4.9; Appendix Table C.52).

Chlorophyll-a concentrations showed no consistent direction of significant differences between 2021 and the mine construction (2014) period and relative to previous years of mine operation (2015 to 2020) for winter, summer, and fall seasons (Figure 4.15; Appendix Table E.13). The variability in chlorophyll-a concentrations among years at Sheardown Lake SE may reflect the 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.

[20] See footnote 12 regarding differences in the concentration of boron in sediment between baseline and recent CREMP studies.

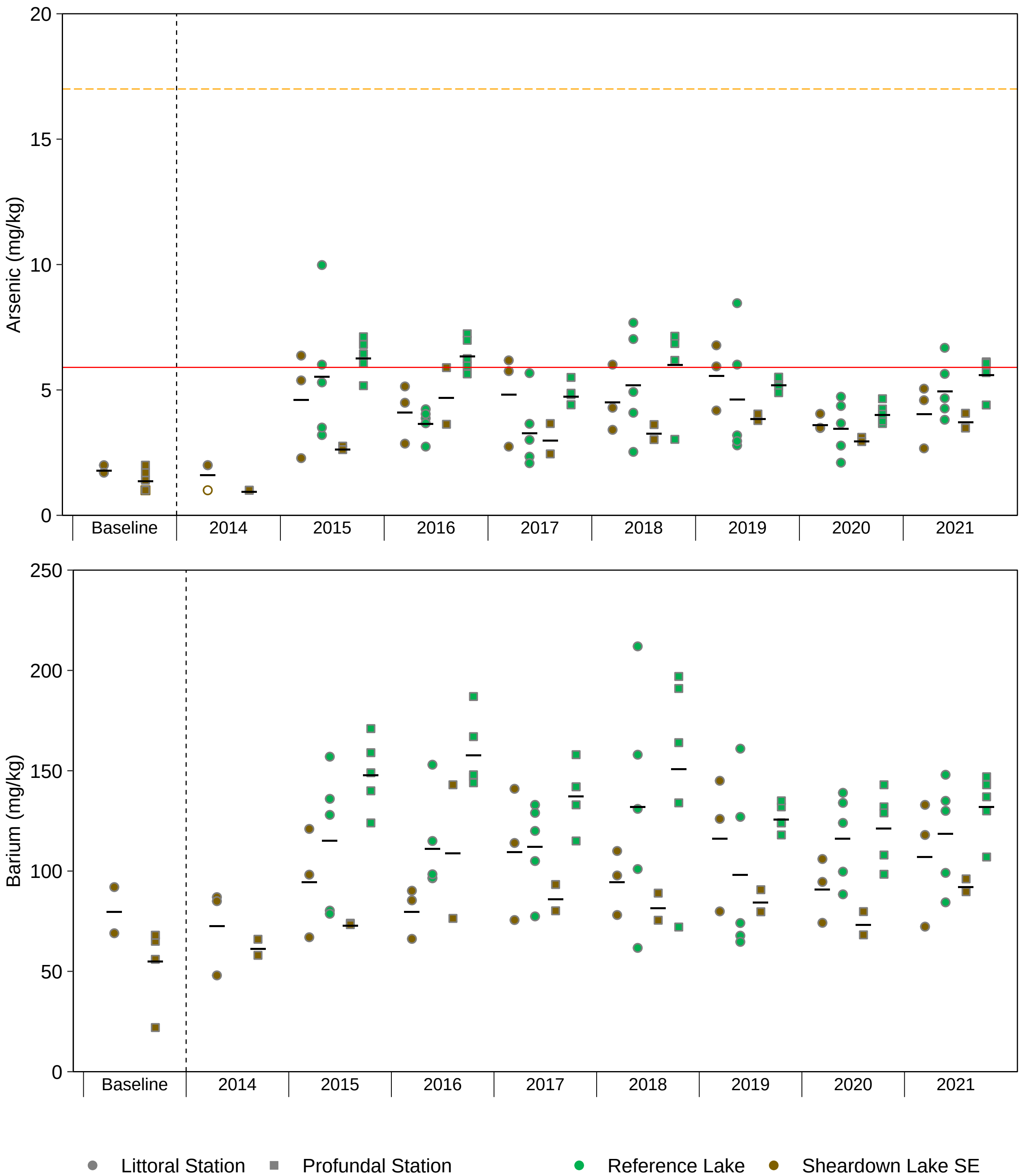


Figure 4.14: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake SE (SDSE) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

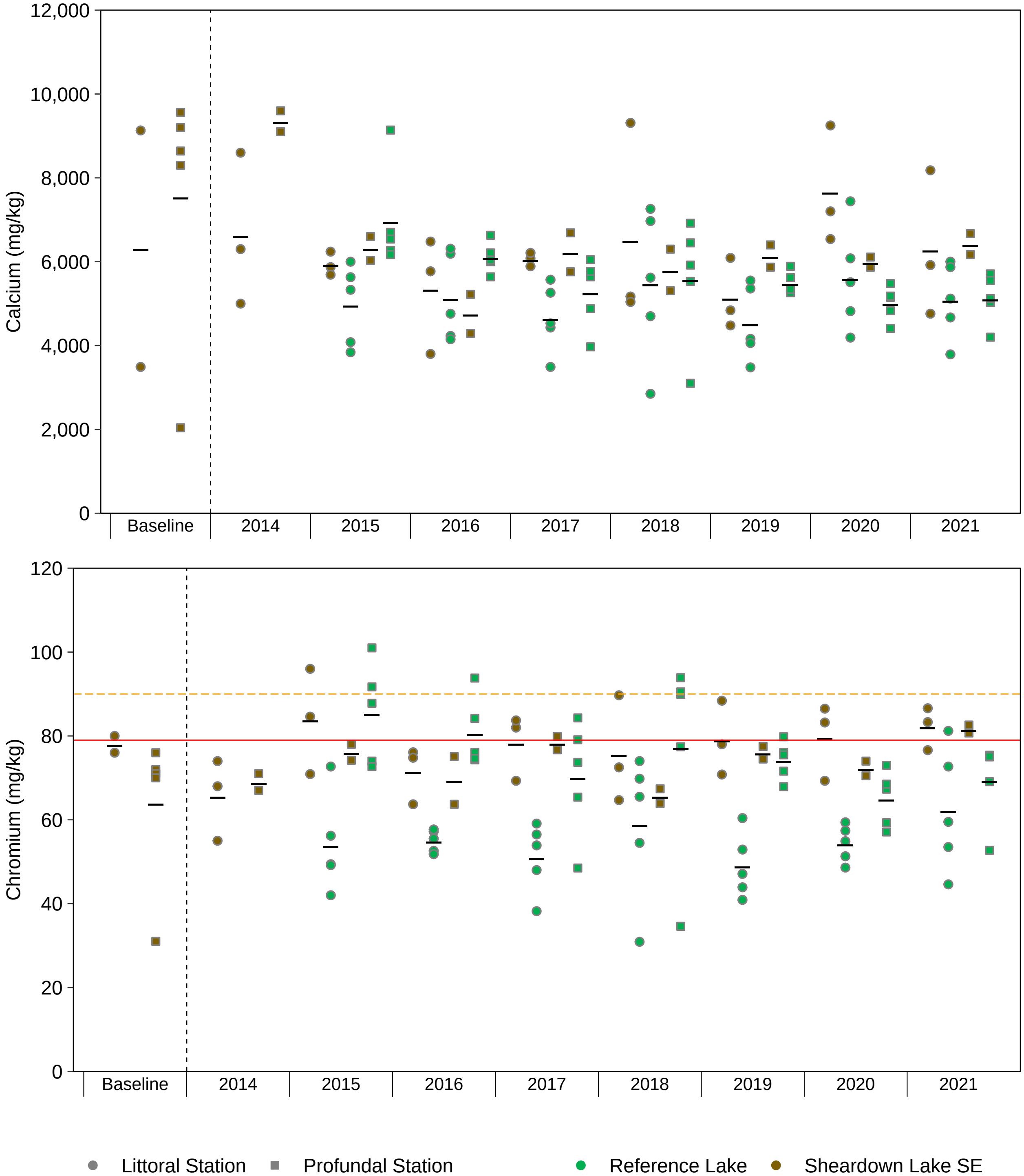


Figure 4.14: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake SE (SDSE) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

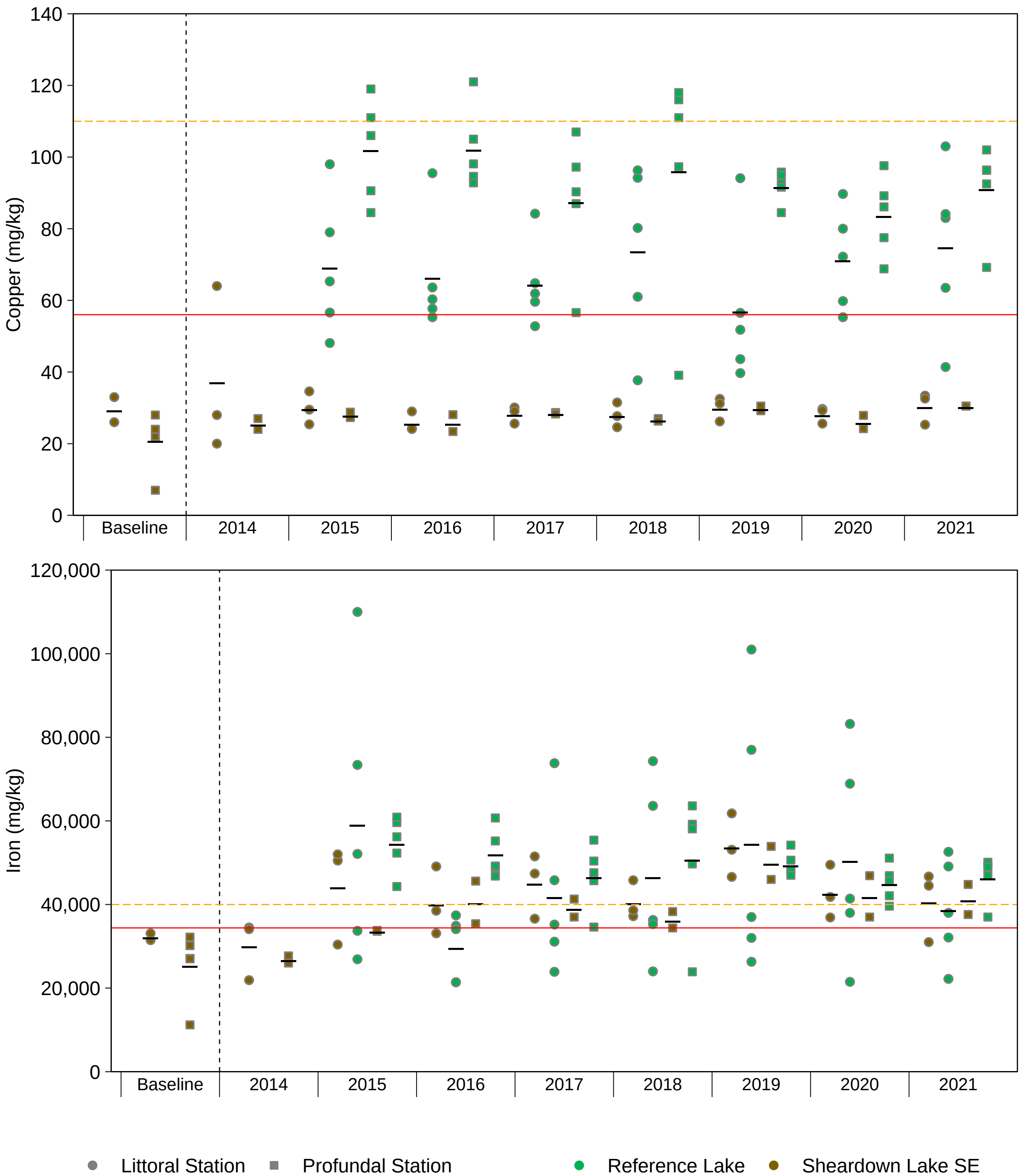


Figure 4.14: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake SE (SDSE) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

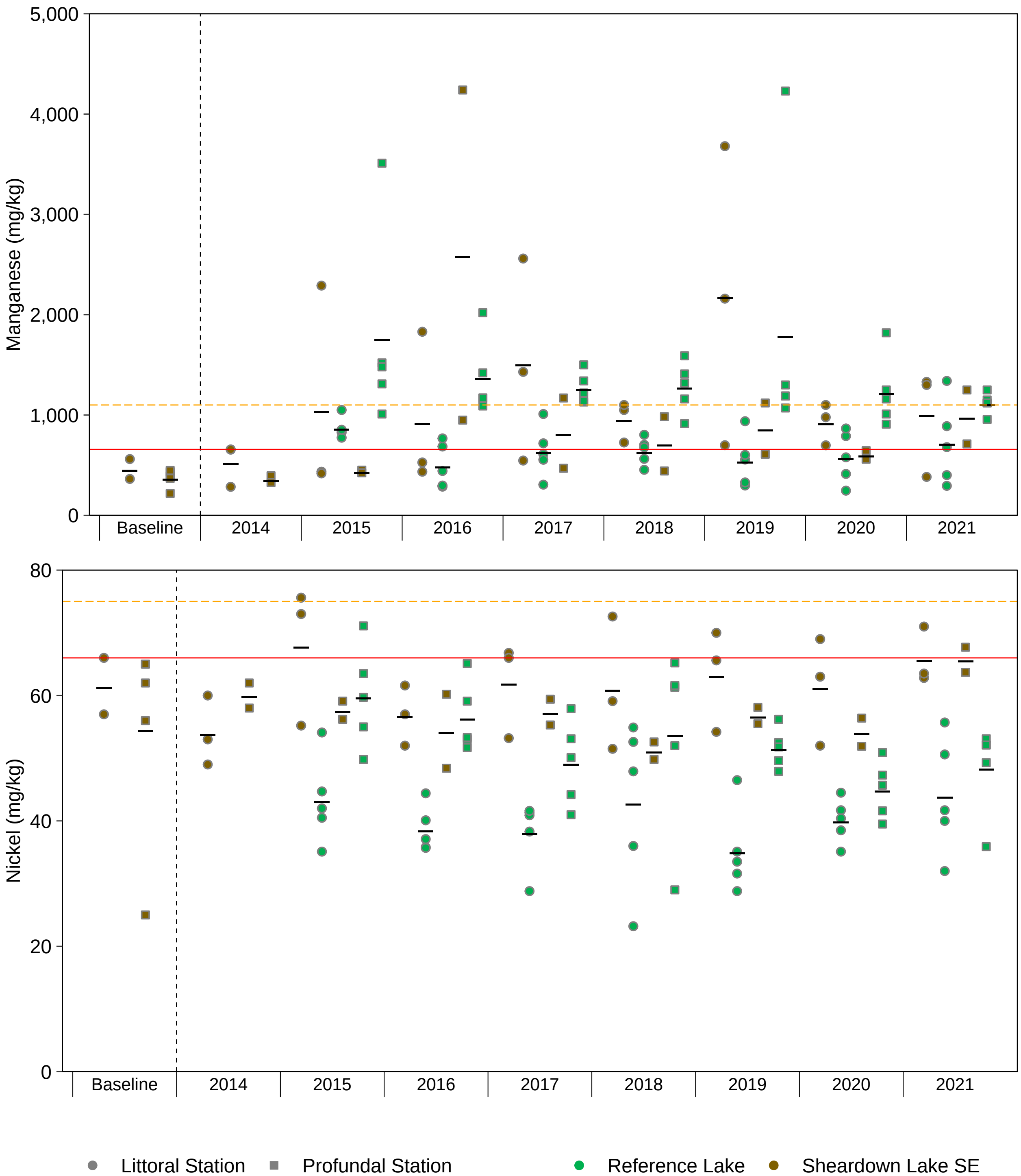


Figure 4.14: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake SE (SDSE) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

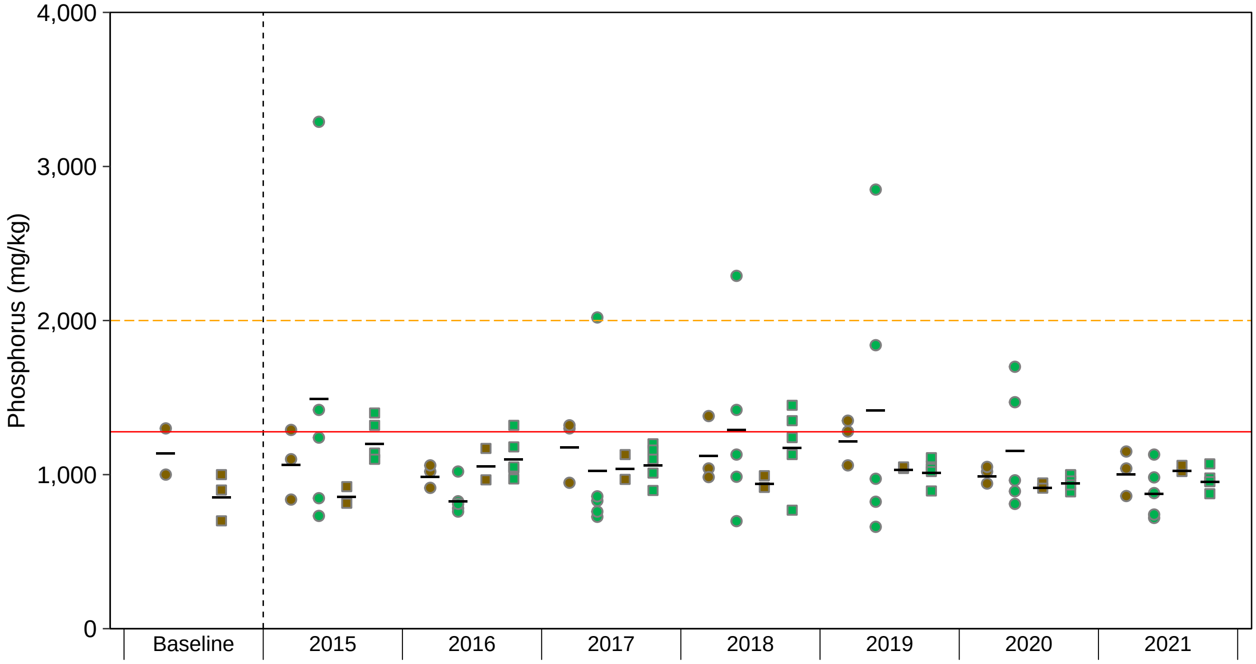


Figure 4.14: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Sheardown Lake SE (SDSE) and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

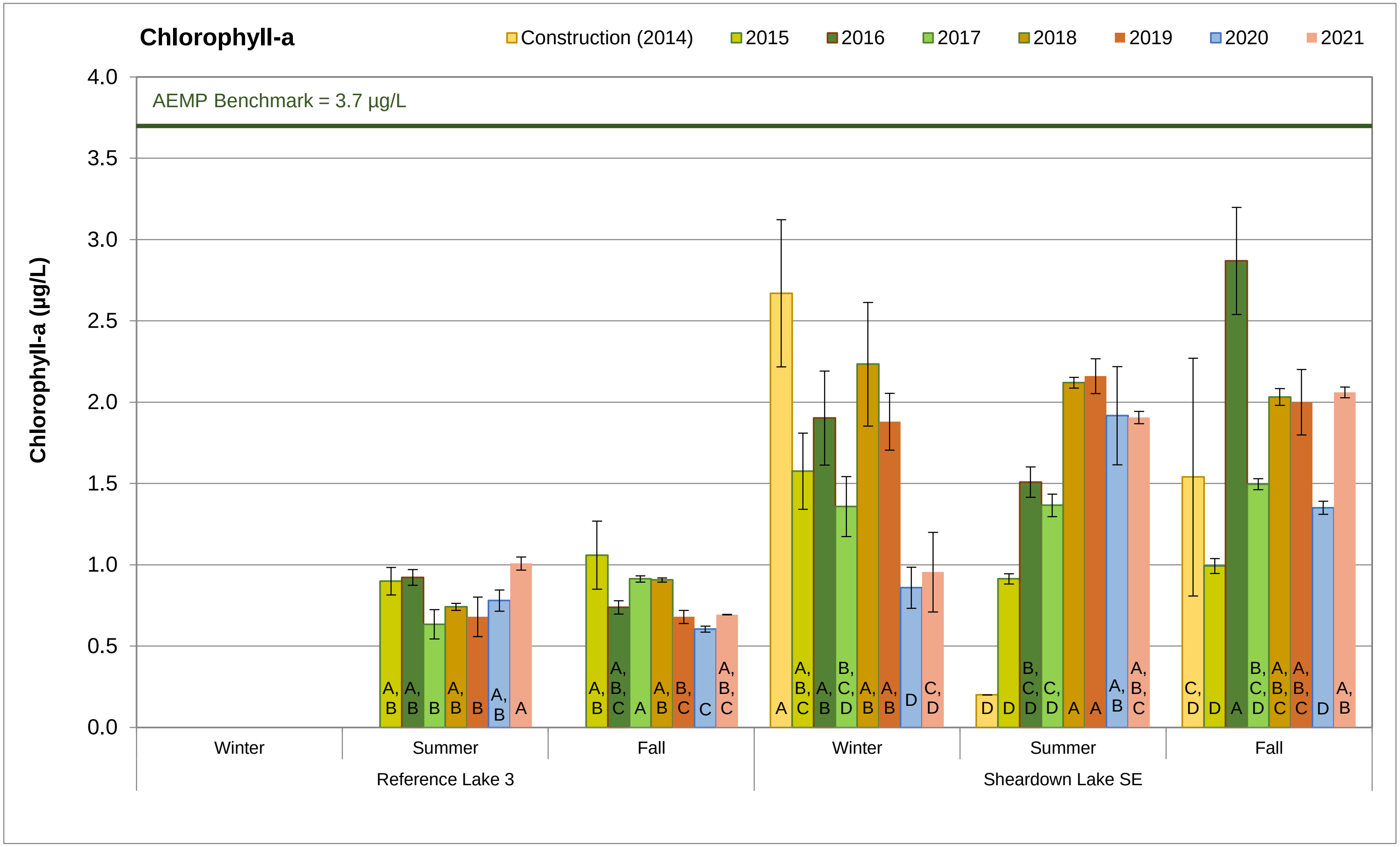


Figure 4.15: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Sheardown Lake SE and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2021) Periods (mean ± SE)

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

4.3.4 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral and profundal habitats, and richness, evenness, and Shannon Diversity were significantly higher at profundal habitats, of Sheardown Lake SE compared to like-habitat stations at Reference Lake 3, but only the differences densities and Shannon Diversity occurred at magnitudes outside of the CES_{BIC} of ±2 SD_{REF} (Tables 4.11 and 4.12). 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). The only ecologically significant difference in dominant taxonomic groups between lakes included higher relative abundance of Chironomidae at littoral stations of Sheardown Lake SE compared to Reference Lake 3 (Tables 4.11 and 4.12). Among FFG metrics, littoral and profundal habitats of Sheardown Lake SE each had significantly lower relative abundance of collector-gatherers relative to like-habitat at Reference Lake 3, the magnitudes of difference of which were outside the CES_{BIC} of ±2 SD_{REF} (Tables 4.11 and 4.12). This may reflect low TOC at Sheardown Lake SE relative to Reference Lake 3 (Figure 4.13). Similar to Sheardown Lake NW, the occurrence of higher benthic invertebrate density without ecologically significant differences in evenness or significantly lower relative abundance of metal-sensitive taxa suggested that Sheardown Lake SE was simply more productive than Reference Lake 3, and not adversely influenced by mine operations in 2021.

Significantly higher relative abundance of the burrower HPG was shown at Sheardown Lake SE relative to Reference Lake 3 in 2021 (Tables 4.11 and 4.12; Appendix Table F.39), possibly due to the higher proportion of silt at Sheardown Lake SE compared to Reference Lake 3 (Figure 4.13; Appendix Table D.18). In addition to 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 2021 (Appendix Table B.9; 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 2021 likely reflected a combination

Table 4.12: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake SE (DL0-02) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	YES	<0.001	21.3	Reference Lake 3	327	151	67.4	60.3	396	422
						Sheardown SE Profundal	3,546	653	292	2,738	3,462	4,443
Richness (Number of Taxa)	tequal	none	YES	0.046	1.5	Reference Lake 3	6.4	2.6	1.2	2.0	8.0	8.0
						Sheardown SE Profundal	10.2	2.5	1.1	7.0	10.0	14.0
Simpson's Evenness (E)	tequal	none	YES	0.026	1.9	Reference Lake 3	0.545	0.113	0.050	0.437	0.510	0.731
						Sheardown SE Profundal	0.759	0.135	0.060	0.527	0.810	0.858
Shannon Diversity	M-W	rank	YES	0.056	1.5	Reference Lake 3	1.38	0.508	0.227	0.592	1.46	2
						Sheardown SE Profundal	2.14	0.449	0.201	1.35	2.33	2.47
Hydracarina (%)	tequal	log10(x+1)	NO	0.159	-0.7	Reference Lake 3	5.3	4.3	1.9	0.0	4.4	11.9
						Sheardown SE Profundal	2.2	1.4	0.6	0.3	2.0	4.0
Ostracoda (%)	tequal	log10(x+1)	NO	0.278	-0.6	Reference Lake 3	8.2	8.7	3.9	0.0	4.4	20.4
						Sheardown SE Profundal	3.3	3.1	1.4	0.4	3.2	8.2
Chironomidae (%)	tequal	none	YES	0.092	1.0	Reference Lake 3	86.0	8.7	3.9	71.4	88.1	93.5
						Sheardown SE Profundal	94.3	4.4	2.0	87.8	94.0	98.9
Metal-Sensitive Chironomidae (%)	tequal	none	YES	0.089	1.2	Reference Lake 3	7.8	6.2	2.8	0.0	10.9	14.3
						Sheardown SE Profundal	15.0	5.7	2.6	7.1	15.3	21.1
Collector-Gatherers (%)	tequal	log10	YES	0.005	-3.5	Reference Lake 3	86.3	7.8	3.5	80.8	83.7	100.0
						Sheardown SE Profundal	59.2	13.2	5.9	46.3	52.8	79.6
Filterers (%)	tequal	none	YES	0.036	1.7	Reference Lake 3	6.5	4.9	2.2	0.0	8.9	10.9
						Sheardown SE Profundal	14.9	5.6	2.5	7.1	15.3	21.1
Shredders (%)	tequal	none	NO	0.21	-0.6	Reference Lake 3	0.9	1.2	0.5	0.0	0.0	2.2
						Sheardown SE Profundal	0.1	0.3	0.1	0.0	0.0	0.6
Clingers (%)	tequal	none	NO	0.345	0.6	Reference Lake 3	12.3	7.0	3.1	0.0	15.2	16.8
						Sheardown SE Profundal	16.5	6.1	2.7	7.3	17.5	23.1
Sprawlers (%)	tunequal	log10	YES	0.019	-6.1	Reference Lake 3	86.0	8.4	3.7	77.5	84.5	100.0
						Sheardown SE Profundal	34.7	19.2	8.6	14.4	33.6	55.4
Burrowers (%)	tunequal	log10(x+1)	YES	0.003	17.7	Reference Lake 3	1.7	2.7	1.2	0.0	0.0	6.1
						Sheardown SE Profundal	48.8	20.7	9.3	27.1	46.0	78.3

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.

Notes: MOD = Magnitude of Difference = (MCTExp - MCTRef)/SDRef. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases. Endpoints that did not have data in both lakes were excluded from the analyses.

of naturally greater productivity 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 2021, 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, although the magnitude of the differences in density relative to the baseline year of 2013 were within CES_{BIC} of ±2 $SD_{Baseline}$ for littoral stations indicating that these differences were not ecologically meaningful (Appendix Tables F.41 and F.42). No ecologically significant differences in richness, evenness, and relative abundance of dominant taxonomic groups or FFG were consistently indicated for littoral or profundal habitat at Sheardown Lake SE over the 2015 to 2021 period of mine operation compared to baseline (Appendix Figure F.12; 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 in both habitat types, 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 Community

The Sheardown Lake SE fish community was composed of arctic charr and ninespine stickleback in 2021 (Table 4.7), the same fish species as observed in previous years (Minnow 2021b). 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 4.7). 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 2021 were both within respective ranges observed during the previous five years of mine operation (i.e., 2015 to 2020), 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 Health Assessment

Nearshore Arctic Charr

A total of 108 and 115 arctic charr were captured from nearshore habitat at each of Sheardown Lake SE and Reference Lake 3, respectively, in August 2021 (Table 4.7). Arctic charr YOY were distinguished from non-YOY using fork length cut-offs of 4.8 cm and 4.0 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 4.16; 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 differed significantly between Sheardown Lake SE and Reference Lake 3 for both the whole population of nearshore arctic charr and for non-YOY (Appendix Table G.20). This difference reflected slightly larger fish captured at Sheardown Lake SE compared to Reference Lake 3 (Figure 4.16). 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 4.13; Appendix Table G.20). The absolute magnitudes of difference in condition were greater than the CES_C of 10% and 5% for YOY and non-YOY, respectively, at Sheardown Lake SE, suggesting that the differences were ecologically significant (Table 4.13; Appendix Table G.20).

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 2021, although most often larger fish of slightly greater condition occurred at Sheardown Lake SE over this period (Table 4.13). Although before-after analysis of data collected at Sheardown Lake SE in 2021 (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 56 arctic charr were sampled from littoral/profundal habitat of Sheardown Lake SE and Reference Lake 3, respectively, in August 2021. 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 4.13; Figure 4.16). Littoral/profundal arctic charr from Sheardown Lake SE were significantly longer, heavier, and had greater condition than those from

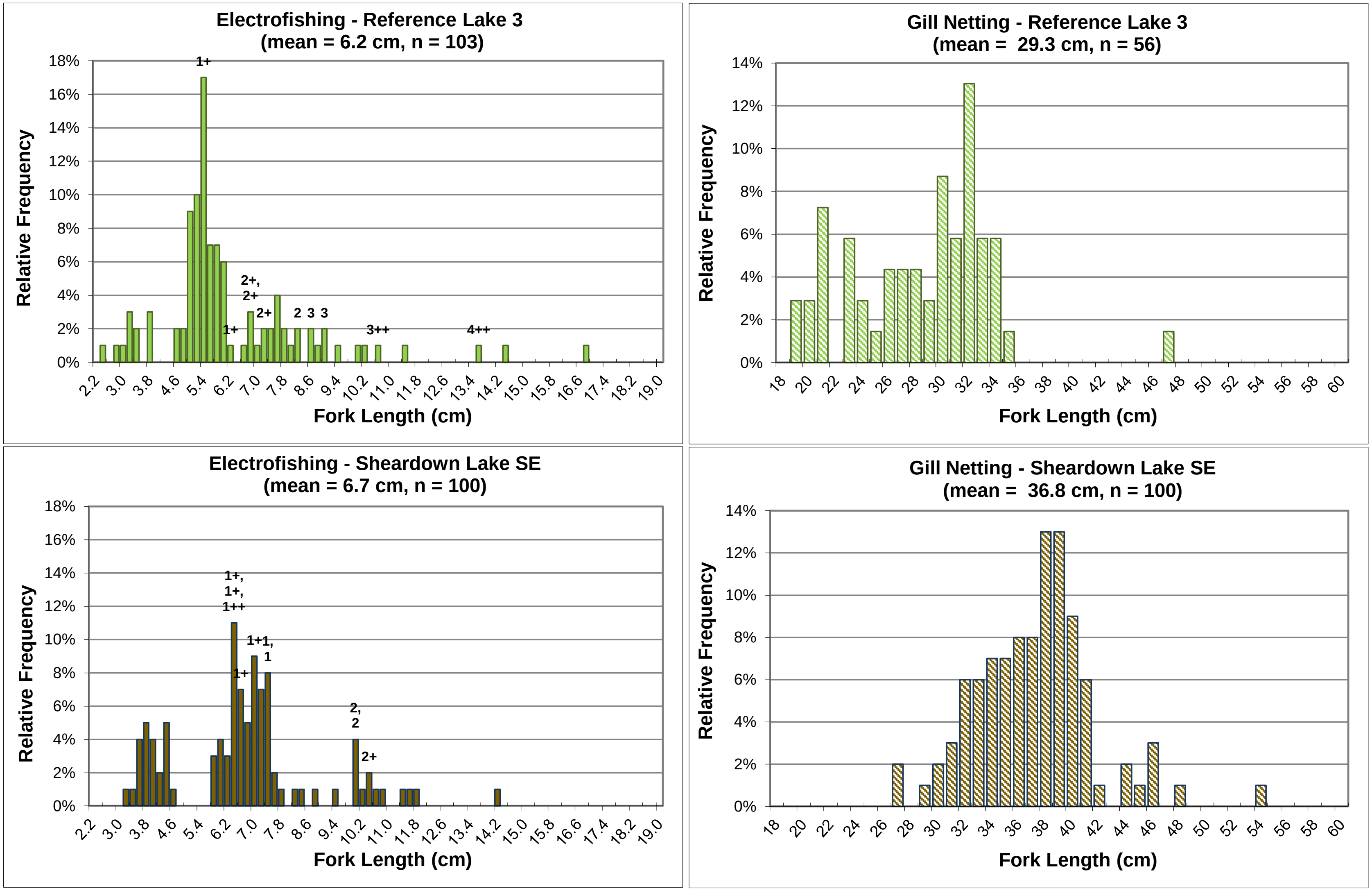


Figure 4.16: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake SE (DL0-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

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

Table 4.13: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake SE and Reference Lake 3 from 2015 to 2021, 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	2021	2015	2016	2017	2018	2019	2020	2021
Nearshore Electrofishing	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**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 (+26%)**	**Yes (+7%)**	**Yes (-15%)**	**Yes (+19%)**	**Yes (-47%)**	No	**Yes (+30%)**	**Yes (+11%)**
		Size (mean weight)	No	No	**Yes (+55%)**	**Yes (+59%)**	**Yes (-59%)**	**Yes (+53%)**	**Yes (+124%)**	No	**Yes (-43%)**	**Yes (+54%)**	No	No	**Yes (+117%)**	**Yes (+32%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+4%)**	No	**Yes (+9%)**	**Yes (-13%)**	**Yes (+4%)**	**Yes (+14%)**	**Yes (+10%)**	**Yes (-14%)**	**Yes (-16%)**	No	**Yes (-15%)**	**Yes (-13%)**	No	No
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	No	No
		Age	-	-	-	-	-	-	-	**Yes (-13%)**	No	No	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (+23%)**	**Yes (+21%)**	**Yes (+27%)**	**Yes (-9%)**	**Yes (-7%)**	**Yes (-5%)**	**Yes (-4%)**	**Yes (-2%)**	No	No
		Size (mean weight)	-	-	-	No	**Yes (+102%)**	**Yes (+107%)**	**Yes (+158%)**	**Yes (-26%)**	**Yes (-20%)**	**Yes (-16%)**	**Yes (-16%)**	**Yes (-11%)**	**Yes (-7.0%)**	No
		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%)**	**Yes (+24%)**	No	No	**Yes (-6%)**	**Yes (-7%)**	**Yes (-6%)**	**Yes (-5.0%)**	**Yes (-3.6%)**

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.

Reference Lake 3 (Table 4.13; Appendix Table G.24). The absolute magnitude of difference in condition was also above the CES_C of 10%, suggesting the difference was ecologically significant (Table 4.13; 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 2021 were similar to the differences shown in 2018, 2019, and/or 2020, 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 2021 and baseline, although differences were reported historically (Table 4.13). Arctic charr sampled from littoral/profundal habitat of Sheardown Lake SE in years of mine operation (i.e., 2015 to 2021) 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 4.13). Differences in arctic charr condition from 2015 to 2021 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 4.13). In turn, this suggested no adverse influences on adult arctic charr at Sheardown Lake SE through the first seven years of mine operation.

4.3.6 Effects Assessment and Recommendations

At Sheardown Lake SE, the following AEMP benchmarks were exceeded in 2021:

- Chromium concentration in sediment, on average, was greater than the benchmark of 79 mg/kg at littoral and profundal 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, on average, was greater than the benchmark of 657 mg/kg at littoral and profundal stations; and,
- Nickel concentration in sediment was greater than the benchmark of 66 mg/kg at one littoral monitoring station (DL0-02-11) and one profundal monitoring station (DL0-02-2), although the average concentrations of nickel in sediment at littoral and profundal stations was below this benchmark.

No AEMP benchmarks for water quality were exceeded over the duration of spring, summer, and fall sampling events in 2021 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 2021. 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 2021 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 2021 and no adverse effects to biota were associated with concentrations of metals above lake-specific 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

5.1.1 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 2021 (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 2021, suggesting no increased oxygen demand associated with mine operations (Appendix Figure C.22; Appendix Tables C.56 and C.57). 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 2021 (Figure 5.1; Appendix Figure C.18; Appendix Table C.55), indicating that the 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 2021 (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 less than a quarter pH unit, and pH at all Mary River areas were consistently within WQG limits (Figure 5.1; Appendix Tables C.56 and 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; Figures 2.2 and 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 location, compared to each of the upstream G0 series stations, at the time of biological monitoring in August 2021 (Appendix Figure C.22; Appendix Table C.57) suggesting that mine-related influences on Mary River water quality were of limited spatial scale.

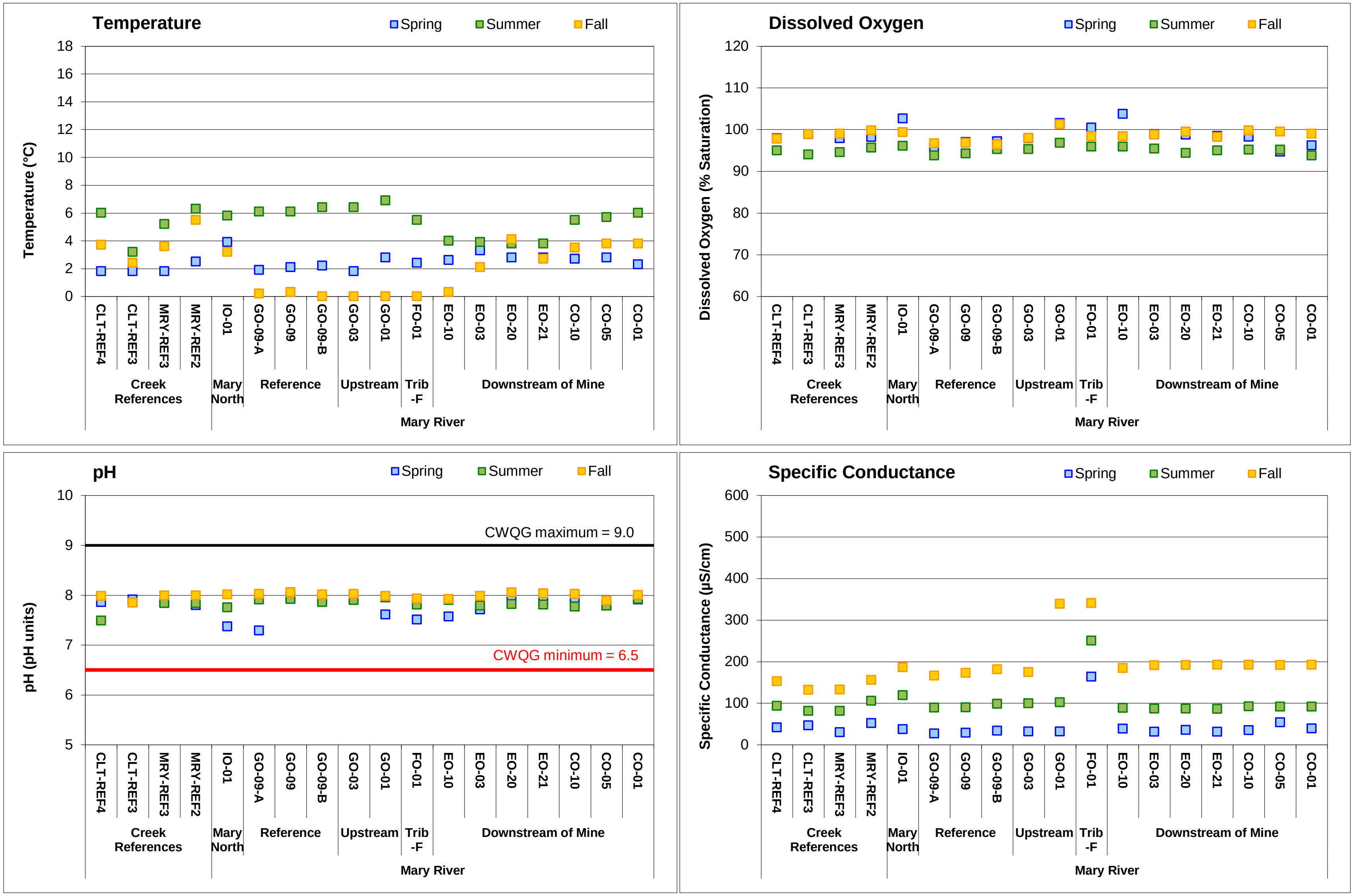


Figure 5.1: Comparison of *In Situ* Water Quality Variables Measured at Mary River Water Quality Monitoring Stations in Spring, Summer, and Fall 2021, Mary River Project CREMP

Within Mary River, mean concentrations of aluminum were below the AEMP benchmark but above WQG at stations located adjacent to (E0 series) and downstream of (C0 series) the mine during spring, summer and fall sampling events in 2021 (Table 5.1; Appendix Table C.58). In addition, the mean concentration of iron was above applicable WQG, but not the AEMP benchmark, during the fall 2021 sampling event for the C0 series stations (Table 5.1). However, when aluminum concentrations were above WQG at areas adjacent to and downstream of the mine (E0 series and C0 series), the mean concentration of aluminum was comparable to or higher and were similarly above the applicable WQG value than at the Mary River reference stations (G0-09 series) and/or other upstream stations (G0 series) for the spring and summer sampling events (Table 5.1).[21] Relatively high concentrations of aluminum within Mary River at the time of the spring, summer, and fall sampling events appeared to be associated with highly turbid sampling conditions (Table 5.1).

Water quality monitoring similarly indicated that elevated concentrations of aluminum, copper, and iron within Mary River in 2019 and 2020 were associated with naturally high turbidity in Mary River (Minnow 2021b). As part of the review of the 2020 CREMP report, Environment and Climate Change Canada (ECCC) requested additional discussion of 2019 and 2020 results for aluminum, copper, and iron concentrations in Mary River samples (ECCC 2021). Correlation analyses indicated that while total aluminum and iron concentrations were significantly and strongly positively correlated with turbidity, dissolved concentrations of these metals showed only weak positive correlation with turbidity for Mary River stations over the mine operational period (2015 to 2021) and for the combined baseline/operational data set (2005 to 2021; Table 5.2). This indicated that aluminum and iron were largely bound to/composed the inorganic suspended material which suggested that natural surface runoff/bedload transport accounted for total concentrations of these metals being routinely above respective AEMP benchmarks at mine-exposed stations of Mary River as opposed to a mine-related source of these metals. High turbidity at the mine-exposed stations of Mary River reflected natural phenomena unrelated to the Mary River Project operations based on similar turbidity between areas located upstream (G0 series) and adjacent to or downstream of the mine. Total and dissolved concentrations of copper showed a similar but weaker positive correlation with turbidity for both sets of data (Table 5.2), suggesting that changes in copper concentrations were less strongly tied to high turbidity events.

[21] 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.

Table 5.1: Mean Water Chemistry at Mary River Monitoring Stations During Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

	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	-	-	31.6	96.7	178	33.5	106	260	170	263	348	36.4	93.8	194	45.6	101	198
	pH (lab)	pH	6.5 - 9.0	-	7.43	7.92	8.09	7.52	7.95	8.15	7.89	8.16	8.19	7.44	7.82	8.11	7.49	7.82	8.13
	Hardness (as $CaCO_3$)	mg/L	-	-	13.5	42.6	79.1	14.6	47.5	128	77.2	129	172	16.0	41.5	90.0	20.1	43.3	90.8
	Total Suspended Solids (TSS)	mg/L	-	-	4.37	2.27	2.50	<2	2.05	<2	<2	<2	<2	5.12	9.12	4.97	9.13	3.70	6.77
	Total Dissolved Solids (TDS)	mg/L	-	-	21.3	43.3	97.7	19.5	42.5	138	100	113	189	70.8	53.2	104	32.3	56.7	119
	Turbidity	NTU	-	-	8.25	8.31	3.01	4.69	5.39	1.21	1.10	1.52	0.580	5.16	10.9	4.69	5.83	6.81	7.78
	Alkalinity (as $CaCO_3$)	mg/L	-	-	12.9	44.7	81.3	14.2	49.8	101	43.8	94.7	116	14.4	37.1	88.1	19.6	36.1	86.7
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	<0.01	0.0127	<0.01	<0.01	0.0110	<0.01	0.0190	0.0110	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Nitrate	mg/L	3	3	<0.02	0.0207	0.0693	<0.02	<0.02	0.528	0.590	0.510	1.03	0.0235	0.0335	0.138	0.0270	0.101	0.201
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.0900	0.0700	0.110	0.0900	0.0500	0.137	0.120	0.130	0.144	0.105	0.108	0.161	0.0833	0.0700	0.128
	Dissolved Organic Carbon	mg/L	-	-	1.17	1.38	1.82	1.20	1.67	1.65	0.770	1.71	1.88	0.990	1.82	3.04	1.04	1.34	2.79
	Total Organic Carbon	mg/L	-	-	1.35	1.31	2.59	1.43	1.48	1.69	0.960	1.42	1.63	1.55	1.66	1.86	1.80	1.67	2.36
	Total Phosphorus	mg/L	0.020^{α}	-	0.0109	0.00837	0.00653	0.00530	0.00590	<0.003	<0.003	0.00340	<0.003	0.0115	0.0118	0.00600	0.0116	0.00507	0.00830
	Phenols	mg/L	0.004^{α}	-	<0.001	<0.001	<0.001	<0.001	0.00100	<0.001	<0.001	<0.001	<0.001	<0.001	0.00100	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	0.940	3.58	7.36	0.905	3.77	5.57	1.33	2.92	4.69	0.792	2.70	6.78	1.08	2.62	6.28
	Sulphate (SO_4)	mg/L	218^{β}	218	0.523	1.98	4.71	0.475	1.94	31.4	38.9	37.8	60.3	1.41	3.09	9.30	1.73	6.72	12.0
Total Metals	Aluminum (Al)	mg/L	0.100	0.966	0.120	0.137	0.0786	0.0977	0.154	0.0398	0.0410	0.0450	0.0186	0.104	0.217	0.144	0.143	0.159	0.246
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<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	<0.0001	<0.0001	0.000102	<0.0001	<0.0001	<0.0001	<0.0001
	Barium (Ba)	mg/L	-	-	0.00343	0.00725	0.0110	0.00312	0.00789	0.0119	0.00618	0.0118	0.0135	0.00335	0.00764	0.0124	0.00448	0.00726	0.0139
	Beryllium (Be)	mg/L	0.011^{α}	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<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	<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	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Calcium (Ca)	mg/L	-	-	2.85	8.77	16.0	3.07	9.67	20.8	13.0	21.0	25.5	3.29	8.55	16.8	4.18	8.54	17.0
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	0.000528	0.000505	<0.0005	<0.0005	0.000583
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.000120	0.000150	<0.0001	0.000140	0.000115	0.000125	0.000142	0.000120	<0.0001	0.000177
	Copper (Cu)	mg/L	0.002	0.0024	0.000687	0.000953	0.00100	0.000560	0.000930	0.000805	<0.0005	0.000860	0.000680	0.000588	0.000932	0.000985	0.000677	0.000807	0.00107
	Iron (Fe)	mg/L	0.30	0.874	0.145	0.111	0.0620	0.104	0.103	0.0350	0.0450	0.0500	<0.03	0.138	0.225	0.158	0.188	0.141	0.314
	Lead (Pb)	mg/L	0.001	0.001	0.000177	0.000150	0.0000730	0.000112	0.000116	<0.00005	<0.00005	<0.00005	<0.00005	0.000180	0.000265	0.000114	0.000232	0.000159	0.000220
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	0.00125	0.00130	0.00200	0.00160	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Magnesium (Mg)	mg/L	-	-	1.70	5.11	9.90	1.76	5.80	18.4	11.0	18.6	26.7	1.99	5.10	11.7	2.59	5.40	12.2
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00344	0.00144	0.00108	0.00210	0.00124	0.00124	0.0104	0.00134	0.00165	0.00414	0.00463	0.00332	0.00613	0.00298	0.00620
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.0000530	0.000240	0.000423	0.0000510	0.000240	0.000302	0.000109	0.000262	0.000294	0.0000625	0.000157	0.000469	0.0000833	0.000246	0.000426
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	0.000558	0.000588	0.000567	0.000507	0.000800
	Potassium (K)	mg/L	-	-	0.387	0.917	1.22	0.355	0.935	1.24	0.710	1.29	1.36	0.350	0.762	1.24	0.457	0.817	1.31
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Silicon (Si)	mg/L	-	-	0.563	0.913	1.05	0.545	1.00	0.915	0.520	0.680	0.810	0.508	1.09	1.08	0.600	0.917	1.18
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.538	2.20	4.12	0.490	2.09	2.67	0.565	1.60	2.14	0.426	1.43	3.35	0.556	1.56	3.21
	Strontium (Sr)	mg/L	-	-	0.00329	0.0107	0.0197	0.00316	0.0107	0.0224	0.0124	0.0217	0.0270	0.00321	0.00897	0.0190	0.00388	0.00940	0.0187
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<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	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.0100	0.0140	0.0148	0.0110	<0.01	0.0180
	Uranium (U)	mg/L	0.015	-	0.000226	0.00159	0.00522	0.000187	0.00142	0.00356	0.000549	0.00211	0.00297	0.000216	0.00104	0.00409	0.000280	0.000823	0.00368
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003

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 Mary River

Table 5.2: Kendall Correlations between Parameters of Interest and Turbidity (NTU) at Mary River Stations

Parameters	Operational (2015 to 2021)		All Years (2005 to 2021)	
	Tau	P-value	Tau	P-value
Total Aluminum (mg/L)	0.664	< 0.001	0.668	< 0.001
Dissolved Aluminum (mg/L)	0.159	< 0.001	0.285	< 0.001
Total Copper (mg/L)	0.445	< 0.001	0.414	< 0.001
Dissolved Copper (mg/L)	0.244	< 0.001	0.207	< 0.001
Total Iron (mg/L)	0.695	< 0.001	0.658	< 0.001
Dissolved Iron (mg/L)	0.0718	0.061	0.134	< 0.001

P-value < 0.05

Indicates significant and strong correlation (Tau > 0.6 or Tau < -0.6)

Among those parameters with established AEMP benchmarks, concentrations of aluminum, cobalt, iron, lead, nitrate, and sulphate were elevated by factors greater than three at one or more stations located adjacent to or downstream of the mine during one or more sampling events in 2021 compared to the G0-09 reference area in Mary River (Appendix Table C.59; Appendix Figure C.23). Concentrations of aluminum, iron, and lead were lower at MRTF (Station F0-01) than at the Mary River reference and mine-exposed stations (Table 5.1), suggesting that this tributary was not a substantial source of these parameters to Mary River in 2021. Elevation in concentrations of nitrate and sulphate in Mary River appeared to be associated with mine deposits to MRTF (e.g., MS-08 effluent). Concentrations of few parameters were elevated in water at Mary River stations located adjacent to or downstream of the mine in 2021, and none were consistently elevated across all sampling events, compared to concentrations shown at respective stations during baseline (Appendix Table C.59; Appendix Figure C.23), indicating no substantial mine-related influences on water quality of Mary River in 2021. Overall, no marked influences on water quality of Mary River were indicated in 2021 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.

5.1.2 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 2021 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 2021 (Figure 5.2), 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 2021. Low to moderate phytoplankton productivity was expected for Mary River reference and mine-exposed stations in 2021 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 2021 were generally similar to those shown at the time of baseline and previous years of mine operation (Figure 5.3). Chlorophyll-a concentrations at the mine-exposed stations of Mary River in fall 2021 were not disproportionately higher or lower compared to baseline or to concentrations at upstream reference stations in 2021, suggesting no adverse change/differences in phytoplankton abundance from mine operations since the commencement of commercial production in 2015.

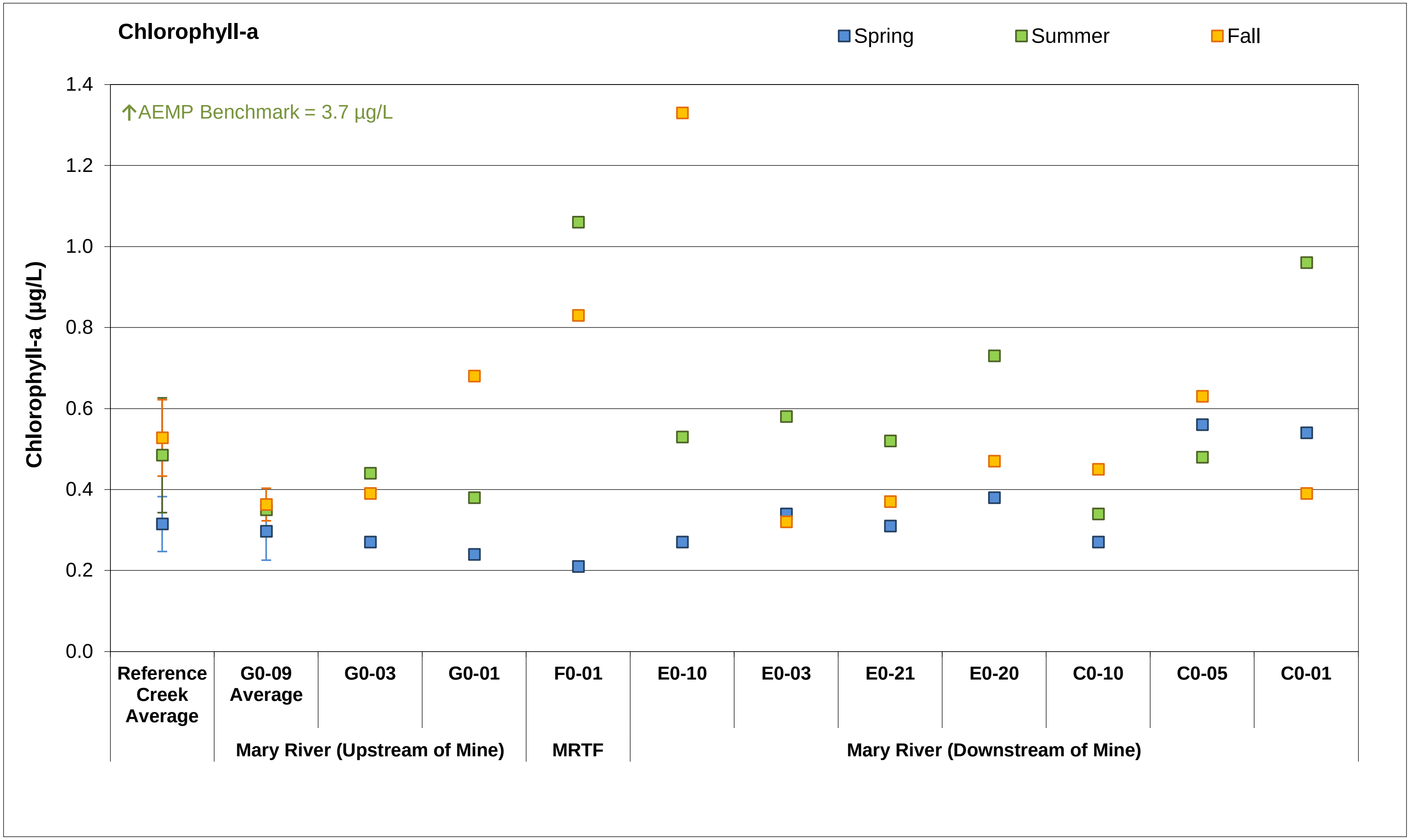


Figure 5.2: Chlorophyll-a Concentrations at Mary River Phytoplankton Monitoring Stations Located Upstream and Downstream of the Mine, Mary River Project CREMP, 2021

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

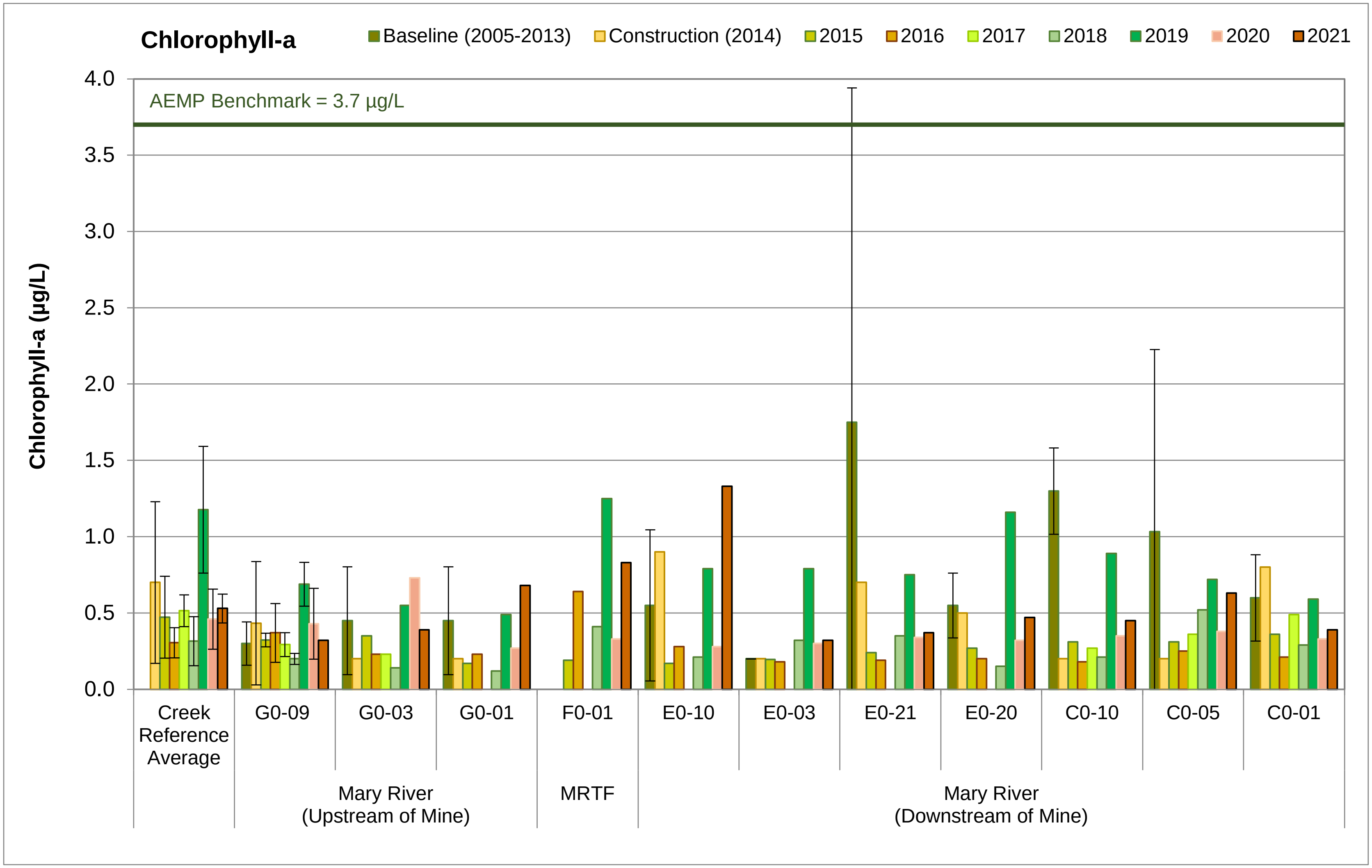


Figure 5.3: Temporal Comparison of Chlorophyll-a Concentrations at Mary River Stations for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2021) Periods during the Fall

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

5.1.3 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 2.6). At the Mary River G0-03 study area, no ecologically significant differences in benthic invertebrate density, richness, evenness, and relative abundance of any dominant taxonomic groups, including metal-sensitive Chironomidae, were indicated compared to the G0-09 reference area, suggesting no marked influences of the mine operation on the benthic invertebrate community at G0-03 (Table 5.3). Significant differences in community assemblage were suggested between G0-03 and G0-09 study areas by differing Bray-Curtis Index (Appendix Table F.51), which was reflected as significantly higher relative abundances of the filterer FFG and the burrower HPG at G0-03 in (Appendix Table F.50). Benthic invertebrate density, richness, and evenness at G0-03 differed significantly between 2021 and baseline, but because these metrics were higher in 2021 and no significantly differences in relative abundance of most dominant taxonomic groups, including metal-sensitive taxa, were indicated for years of mine operation compared to baseline (Appendix Table F.52), no adverse mine-related influences were suggested at G0-03 over the duration of mine operation.

At the near-field mine-exposed study areas (i.e., E0-01 and E0-20), no significant differences in benthic invertebrate density, richness, evenness, and the relative abundance of all dominant taxonomic groups except Chironomidae were indicated relative to the G0-09 reference area (Table 5.3; Appendix Table F.50). Differing Bray-Curtis Index suggested differing community composition between the E0 and G0-09 study areas, which was likely driven by significantly lower relative abundance of Chironomidae and significantly higher relative abundance of the burrower HPG at both E0-01 and E0-20, as well as ecologically significant lower relative abundance of collector-gatherer FFG at E0-20, compared to the G0-09 reference area in 2021 (Table 5.3; Appendix Table F.50). These differences potentially indicated habitat differences between the E0 and G0-09 study areas. Similar to previous years of mine operation, no consistent ecologically significant differences in density, richness, and relative abundance of dominant taxonomic groups (including metal-sensitive Chironomids) were indicated for mine operational years (2015 to 2021) compared to baseline (2007) at either of the E0 near-field study areas (Appendix Tables F.54 and F.56). Although evenness was consistently significantly higher at E0-01 and E0-20 than during the 2007 baseline since 2015 and 2017, respectively, higher evenness is not associated with an adverse influence and thus was not indicative of effects to the Mary River near-field area benthic invertebrate community related to mine operations.

Table 5.3: Benthic Invertebrate Community Metric Statistical Comparison Results among Mary River Reference (GO-09), Upstream (G0-03), and Mine-Exposed (E0-01, E0-20, C0-05) Study Areas, Mary River Project CREMP, August 2021

Metric	Overall 5-Area Comparison				Pair-wise, post-hoc Comparisons				
	Statistical Test[a]	Data Transform-ation	Significant Difference Among Areas?	P-value	Area	Mean	Standard Deviation	Magnitude of Difference[b] (G0-09 Reference SD)	Pairwise Comparison
Density (No. per m^2)	ANOVA	log10	YES	0.001	G0-09 Ref	471	219	nc	AB
					G0-03	468	275	0.0	AB
					E0-01	173	140	-1.4	B
					E0-20	304	158	-0.8	B
					C0-05	1,458	885	4.5	A
Richness (No. of Taxa)	ANOVA	none	NO	0.234	G0-09 Ref	15.0	2.6	nc	A
					G0-03	18.0	5.5	1.2	A
					E0-01	13.4	4.3	-0.6	A
					E0-20	13.6	3.4	-0.5	A
					C0-05	17.8	3.8	1.1	A
Simpson's Evenness	ANOVA	none	NO	0.258	G0-09 Ref	0.860	0.108	nc	A
					G0-03	0.917	0.059	0.5	A
					E0-01	0.948	0.018	0.8	A
					E0-20	0.915	0.013	0.5	A
					C0-05	0.886	0.062	0.2	A
% Nemata (% of community)	K-W	rank	YES	0.019	G0-09 Ref	1.2	1.7	nc	B
					G0-03	2.6	2.2	0.8	B
					E0-01	1.8	1.9	0.3	B
					E0-20	1.3	1.0	0.1	B
					C0-05	15.3	12.2	8.1	A
Hydracarina (% of community)	ANOVA	log10(x+1)	YES	0.094	G0-09 Ref	1.1	0.9	nc	AB
					G0-03	2.2	1.1	1.3	A
					E0-01	1.2	1.3	0.1	AB
					E0-20	1.3	1.6	0.2	AB
					C0-05	0.1	0.2	-1.2	B
Chironomidae (% of community)	K-W	rank	YES	0.052	G0-09 Ref	85.6	4.4	nc	A
					G0-03	79.8	9.6	-1.3	AB
					E0-01	72.0	4.7	-3.1	B
					E0-20	71.9	3.6	-3.1	B
					C0-05	63.9	22.9	-5.0	B
Tipulidae (% of community)	ANOVA	log10(x+1)	NO	0.104	G0-09 Ref	1.1	1.2	nc	A
					G0-03	1.6	1.0	0.4	A
					E0-01	3.0	2.8	1.6	A
					E0-20	3.4	3.0	1.9	A
					C0-05	0.3	0.4	-0.7	A
Metal Sensitive Chironomidae (% of community)	K-W	rank	YES	0.034	G0-09 Ref	23.7	22.3	nc	A
					G0-03	26.7	14.6	0.1	A
					E0-01	18.2	7.4	-0.2	A
					E0-20	22.6	5.7	0.0	A
					C0-05	7.0	5.1	-0.7	B

Indicates a significant difference for respective comparison (p-value ≤ 0.05).

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.

Notes: nc= no comparison; nm=MOD could not be calculated due to SD = 0.

[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).

[b] Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; MCT reported as geometric mean for log10-transformed data, median for rank-transformed data, means for untransformed data.

At far-field mine-exposed area C0-05, no ecologically significant differences in benthic invertebrate density, richness, evenness, and relative abundance of metal-sensitive taxa and FFG were indicated compared to the G0-09 reference area, suggesting no marked influences of the mine operation on the benthic invertebrate community in 2021 (Table 5.3; Appendix Table F.50). Differences in community assemblage were indicated between the C0-05 and G0-09 study areas based on differing Bray-Curtis Index (Appendix Table F.51) that were reflected by a significantly higher and lower relative abundance of Nemata and Chironomidae, respectively, between study areas in 2021 (Table 5.3). Although differences in community assemblage between C0-05 and G0-09 reference area were identified in 2021, no consistent ecologically significant differences in density, richness, evenness, and dominant taxonomic groups were indicated at C0-05 for all years of mine operation (2015 to 2021) 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.

5.1.4 Effects Assessment and Recommendations

Water chemistry at Mary River met all AEMP benchmarks in 2021. In addition, no adverse effects on phytoplankton or benthic invertebrates were indicated at all Mary River mine-exposed areas in 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 2.6). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2021, no management response (i.e., alteration of existing AEMP) is required for Mary River as part of the 2022 CREMP.

5.2 Mary Lake (BL0)

5.2.1 Water Quality

Vertical profiles of *in situ* water temperature, dissolved oxygen, pH, and specific conductance conducted at Mary Lake showed no substantial within-season station-to-station differences for each of the north and south basins during any of the winter, summer, or fall sampling events in 2021 (Appendix Figures C.25 to C.28). Between basins, the north basin water temperature was warmer and cooler in winter and fall, respectively, dissolved oxygen saturation was lower in the winter, and specific conductance was higher in all seasons, compared to the south basin in 2021, the latter of which reflecting the predominant influence from the Tom River in the north basin versus the Mary River at the south basin (Appendix Figures C.25 to C.28).

At the north basin, there was little change in temperature from surface to bottom of the water column in the summer, whereas the reference lake had warmer temperatures in the top 6 m (Figure 5.4). In the fall, the profile patterns between lakes were reversed, with warmer water temperatures in the top 3 m shown at the Mary Lake north basin contrasting with no change in temperature from surface to bottom at the reference lake (Figure 5.4). Water quality profiles measured at the south basin 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 2021 (Figure 5.5). During winter, a clear gradient of increasing temperature with depth occurred at both the north and south basins of Mary Lake (Figures 5.4 and 5.5). The average water temperature at the bottom of the water column at Mary Lake littoral stations was significantly cooler than at Reference Lake 3, whereas profundal stations did not differ significantly from temperatures at the reference lake during the August 2021 biological study (Figure 5.6; Appendix Table C.65). The mean incremental difference in temperature between littoral stations of Mary Lake and the reference lake was only 0.45°C (Appendix Tables C.65 and C.67), suggesting that the difference in temperature between lakes was not likely to be ecologically meaningful.

Dissolved oxygen profiles showed the development of strong oxyclines extending through the entire water column at both the Mary Lake north and south basins during winter 2021 (Figures 5.4 and 5.5). The dissolved oxygen profiles conducted during summer and fall at Mary Lake mirrored similar profiles at the reference lake. Dissolved oxygen concentrations near the bottom of the water column did not differ significantly between Mary Lake and the reference lake for respective littoral and profundal stations during the August 2021 biological study (Figure 5.6; Appendix Table C.65 and C.67). In addition, 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 stations of Mary Lake during the August 2021 study (Figure 5.6; 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, summer, and fall sampling events in 2021 (Figures 5.4 and 5.5). The pH near the bottom of the water column at littoral and profundal stations of Mary Lake did not differ significantly from the reference lake and was consistently with WQG during the August 2021 biological study (Figure 5.6). Specific conductance was substantially higher at the north basin compared to the south basin of Mary Lake (Figures 5.4 and 5.5; 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 were relatively uniform from the surface to bottom of the water column at the north and south basins over winter, summer, and fall sampling events in 2021 (Figures 5.4 and 5.5; Appendix Figure C.28). Specific conductance near the bottom of the water column at

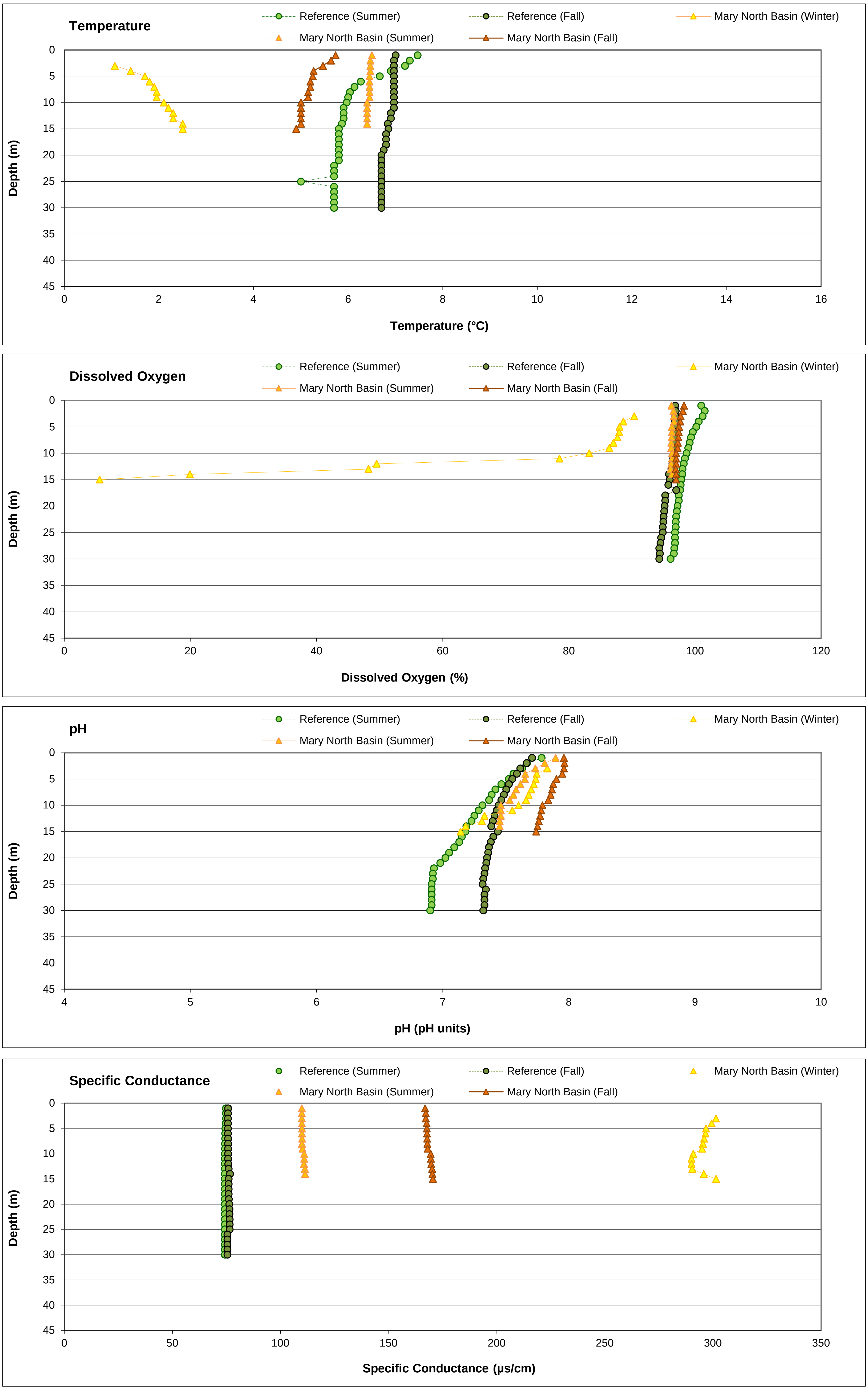


Figure 5.4: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake North Basin (BL0) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

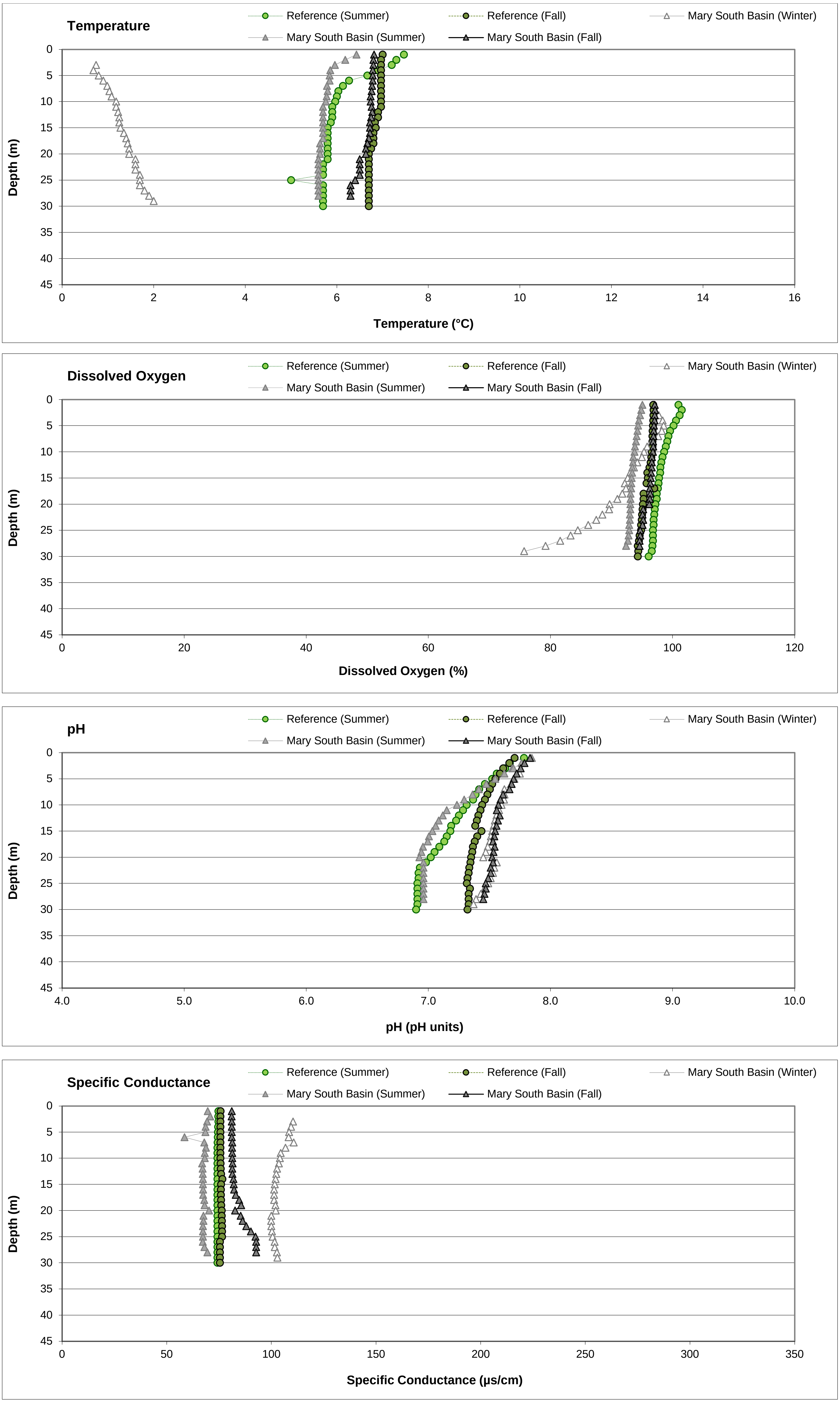


Figure 5.5: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake South Basin (BL0) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2021

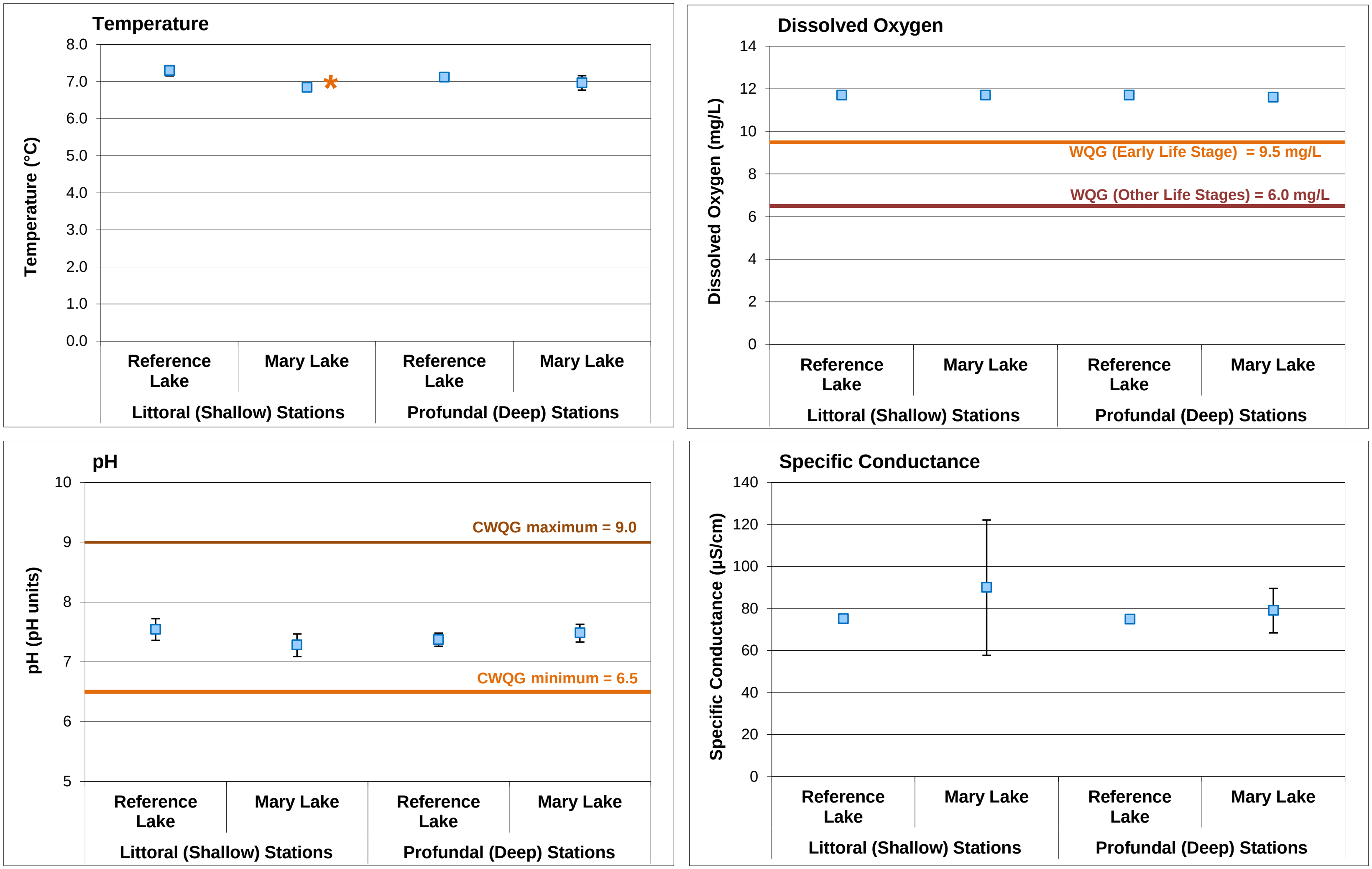


Figure 5.6: Comparison of *In Situ* Water Quality Variables (mean ± SD) Measured at Mary Lake (BL0) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2021

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.

littoral and profundal stations of Mary Lake did not differ significantly from like-habitat stations at Reference Lake 3 during the August 2021 biological study (Figure 5.6). Water clarity, as determined using Secchi depth readings, was significantly lower at Mary Lake compared to Reference Lake 3 in August 2021 (Appendix Table C.64; Appendix Figure C.8), indicating less water clarity at Mary Lake.

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 2021 except for the total mercury concentration in one surface sample collected during winter at the south basin not meeting the respective WQG (Table 5.4; Appendix Table C.72). Among those parameters with established AEMP benchmarks, chloride concentrations were elevated at the north basin, and total aluminum concentrations were elevated at both the north and south basins of Mary Lake, compared to Reference Lake 3 in the summer and/or fall sampling events (Table 5.4; Appendix Table C.69). Of those parameters without AEMP benchmarks, turbidity and concentrations of total and dissolved manganese concentrations were elevated at the north and south basins of Mary Lake, and total and dissolved sodium 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 2021 (Appendix Tables C.69 and C.71). Similar to the Sheardown Lake system, elevated total aluminum concentrations at Mary Lake compared to the reference lake in 2021 were connected to naturally higher turbidity at Mary Lake and thus were unrelated to the mine operations (Appendix Tables C.68 and C.72).[22] 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 2021 and baseline for the Mary River north basin and south basin, except for total and dissolved manganese concentrations and dissolved sodium concentrations at the north basin in winter 2021 (Appendix Figure C.29; Appendix Tables C.68 to C.73). Overall, mine-related influences on water quality of Mary Lake in 2021 included a slight elevation in chloride, manganese, sodium, and uranium concentrations compared to the reference lake. However, the occurrence of water quality below AEMP benchmarks and lack of consistent 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.

[22] Turbidity at Mary Lake in 2021 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 2021 (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 5.4: 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, 2021

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	-	-	79.1	83.3	300	118	182	110	73.6	89.0
	pH (lab)	pH	6.5 - 9.0	-	7.71	7.68	7.73	7.97	8.12	7.59	7.64	7.80
	Hardness (as $CaCO_3$)	mg/L	-	-	36.8	37.7	142	55.6	81.3	52.1	32.8	39.4
	Total Suspended Solids (TSS)	mg/L	-	-	<2	<2	<2	2.17	<2	2.14	<2	<2
	Total Dissolved Solids (TDS)	mg/L	-	-	29.0	41.0	140	63.7	93.3	61.9	42.0	43.7
	Turbidity	NTU	-	-	0.240	0.183	<0.1	0.953	0.767	<0.1	1.90	0.799
	Alkalinity (as $CaCO_3$)	mg/L	-	-	37.3	46.3	138	49.4	88.4	53.7	34.4	49.4
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.0140	<0.01	0.0110	<0.01	<0.01	<0.01	0.0110	<0.01
	Nitrate	mg/L	3	3	<0.02	<0.02	0.123	<0.02	<0.02	0.0653	0.0364	0.0364
	Nitrite	mg/L	0.06	0.06	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.177	0.200	0.100	0.143	0.160	0.0743	0.0843	0.164
	Dissolved Organic Carbon	mg/L	-	-	3.00	4.33	3.00	2.37	3.27	2.01	1.57	2.05
	Total Organic Carbon	mg/L	-	-	3.46	3.49	2.84	1.72	2.15	1.85	2.00	1.85
	Total Phosphorus	mg/L	0.020[α]	-	0.00443	0.00303	0.00377	0.00327	0.00547	0.00316	0.00561	0.00501
	Phenols	mg/L	0.004[α]	-	0.00113	<0.001	<0.001	<0.001	<0.001	<0.001	0.00109	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	1.32	1.35	15.2	2.94	5.41	4.37	2.32	2.66
	Sulphate (SO_4)	mg/L	218[β]	218	3.68	3.69	6.11	1.40	2.55	3.91	2.10	2.66
Total Metals	Aluminum (Al)	mg/L	0.100	0.130	0.00373	0.00370	0.00630	0.0338	0.0232	0.00444	0.0617	0.0220
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.000103	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Barium (Ba)	mg/L	-	-	0.00676	0.00651	0.0149	0.00591	0.00827	0.00628	0.00453	0.00471
	Beryllium (Be)	mg/L	0.011[α]	-	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Bismuth (Bi)	mg/L	-	-	<0.0005	<0.0005	<0.0005	<0.0005	<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	<0.00001	<0.00001	<0.00001	0.0000101	<0.00001
	Calcium (Ca)	mg/L	-	-	7.40	7.43	27.7	11.1	16.6	10.6	6.84	7.76
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<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	0.000810	0.000750	0.00116	0.000767	0.000900	0.000663	0.000641	0.000607
	Iron (Fe)	mg/L	0.30	0.326	<0.03	<0.03	<0.03	0.0383	<0.03	<0.03	0.0573	<0.03
	Lead (Pb)	mg/L	0.001	0.001	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	0.0000619	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Magnesium (Mg)	mg/L	-	-	4.67	4.77	17.7	6.87	10.4	6.25	4.10	5.02
	Manganese (Mn)	mg/L	0.935[β]	-	0.000674	0.000591	0.00110	0.00202	0.00198	0.000363	0.00213	0.000915
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	0.0000271	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000155	0.000157	0.000414	0.000152	0.000242	0.000222	0.000132	0.000153
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000693	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Potassium (K)	mg/L	-	-	0.960	0.910	1.57	0.713	0.947	0.804	0.590	0.610
	Selenium (Se)	mg/L	0.001	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Silicon (Si)	mg/L	-	-	0.507	0.490	1.16	0.673	0.737	0.453	0.519	0.401
	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	-	-	0.994	0.929	6.75	1.76	3.01	2.12	1.25	1.43
	Strontium (Sr)	mg/L	-	-	0.00877	0.00856	0.0226	0.00812	0.0120	0.00979	0.00601	0.00667
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0001	<0.0001	<0.0001	<0.0001	<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	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Uranium (U)	mg/L	0.015	-	0.000328	0.000342	0.00521	0.000985	0.00224	0.00131	0.000573	0.000771
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Zinc (Zn)	mg/L	0.030	0.030	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	0.00343	<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.

5.2.2 Sediment Quality

Surficial sediment of the Mary Lake north basin (BL0-01) was mainly composed of silt with low TOC content (Figure 5.7; Appendix Table D.22). Sediments from the Mary Lake south basin also had low TOC content (i.e., <1.5%) and were predominantly composed of silt except at Station BL0-03 (profundal) which contained 92% sand (Figure 5.7; Appendix Table D.22). Particle size of littoral sediments at Mary Lake did not differ significantly with those at the reference lake (Appendix Table D.23). However, the sand-sized and clay-sized fractions of profundal sediment at Mary Lake were significantly lower and higher, respectively, than those at the reference lake. In addition, the TOC content of both littoral and profundal sediment at Mary Lake were significantly lower than that of reference lake sediment for respective habitats (Appendix Table D.23). Orange-brown coloured iron (oxy)hydroxide material was observed at various sediment sampling locations throughout Mary Lake (Appendix Tables D.20 and D.21), mirroring similar observations at Reference Lake 3. This material was also commonly observed in sediment at other mine-exposed lakes as a thin, distinct layer or floc on or within surficial sediment. No distinct redox boundaries were observed and no sulphidic odour was detected in sediment collected at Mary Lake in 2021 (Appendix Tables D.20 and D.21).

Metal mean concentrations in sediment at Mary Lake were comparable to those at Reference Lake 3 in 2021 except for slightly (i.e., 3- to 5-times) higher concentrations of zirconium at both littoral and profundal habitat (Appendix Table D.24). At the lone Mary Lake north basin station (i.e., BL0-01), concentrations of all metals in sediment were below applicable SQG and lake-specific AEMP benchmarks except for the concentration of arsenic (which was above the AEMP benchmark), and iron and manganese concentrations (which were above SQG; Table 5.5). In the south basin, concentrations of 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 5.5). 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.22).[23] As indicated previously, mean concentration of iron in sediment at the reference lake was elevated above SQG (Table 5.5), suggesting that elevated concentration of iron 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 2021 have not changed substantially from those observed during the baseline period (2005 to 2013; Figure 5.8;

[23] Spatially, the order of sediment quality from closest to Mary River to the lake outlet were as follows: BL0-12, BL0-10, BL0-09, BL0-08, and BL0-06 (Figure 2.4). All these stations, except BL0-06, were profundal.

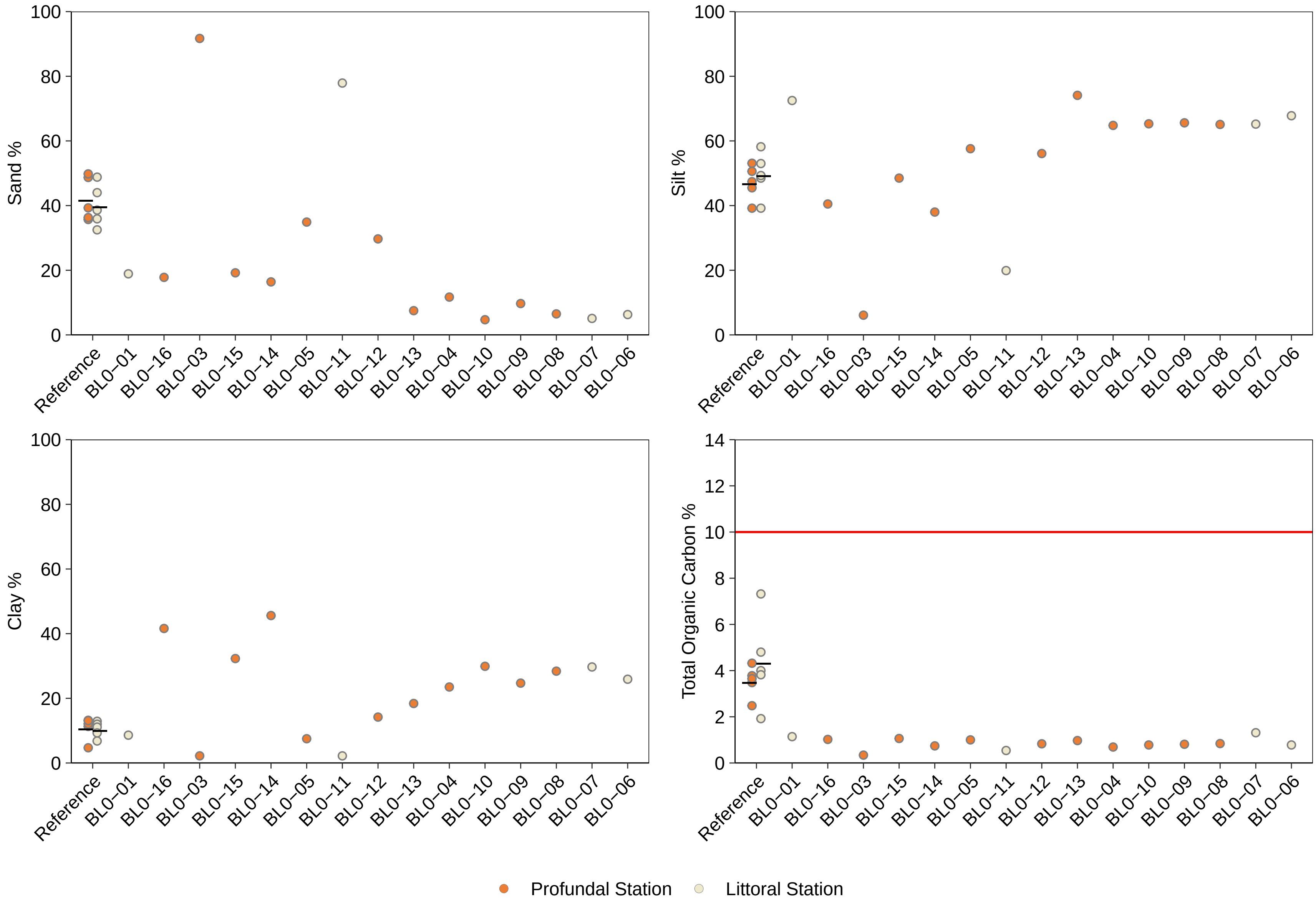


Figure 5.7: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Mary Lake (BL0) Sediment Monitoring Stations and to Reference Lake 3, Mary River Project CREMP, August 2021

Note: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

Table 5.5: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Mary Lake North Basin (BL0-01) and South Basin (BL0-06), and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2021

	Analyte	Units	SQG[a]	AEMP Benchmark[b]	Littoral			Profundal	
					Reference Lake (n = 5) Average ± SD	Mary Lake North (n = 1)	Mary Lake South (n = 1)	Reference Lake (n = 5) Average ± SD	Mary Lake (South Basin) (n = 8) Average ± SD
TOC		%	10[α]	-	3.70 ± 1.09	1.14	0.780	4.21 ± 1.83	0.756 ± 0.194
Metals	Aluminum (Al)	mg/kg	-	-	21,120 ± 4,592	19,500	26,800	20,520 ± 4,129	22,436 ± 2,768
Metals	Antimony (Sb)	mg/kg	-	-	<0.100 ± 0	<0.10	<0.10	<0.100 ± 0	<0.100 ± 0
Metals	Arsenic (As)	mg/kg	17	5.9	5.54 ± 1.03	***7.16***	3.82	5.13 ± 0.978	4.23 ± 0.653
Metals	Barium (Ba)	mg/kg	-	-	123 ± 29.4	102	106	129 ± 13.4	95.3 ± 12.2
Metals	Beryllium (Be)	mg/kg	-	-	0.812 ± 0.176	0.800	1.24	0.782 ± 0.122	1.04 ± 0.129
Metals	Bismuth (Bi)	mg/kg	-	-	<0.200 ± 0	0.200	0.220	<0.200 ± 0	0.236 ± 0.007
Metals	Boron (B)	mg/kg	-	-	14.9 ± 2.68	21.7	36.2	15.2 ± 2.23	28.9 ± 3.28
Metals	Cadmium (Cd)	mg/kg	3.5	1.5	0.144 ± 0.046	0.121	0.092	0.167 ± 0.048	0.123 ± 0.015
Metals	Calcium (Ca)	mg/kg	-	-	5,026 ± 849	8,240	4,560	5,186 ± 660	4,068 ± 822
Metals	Chromium (Cr)	mg/kg	90	98	67.2 ± 15.0	84.4	84.8	64.6 ± 10.7	81.4 ± 9.75
Metals	Cobalt (Co)	mg/kg	-	-	14.8 ± 3.35	18.0	16.5	13.8 ± 3.48	15.6 ± 1.88
Metals	Copper (Cu)	mg/kg	110	50	***81.2*** ± 27.5	36.3	33.2	***85.0*** ± 10.5	31.0 ± 4.09
Metals	Iron (Fe)	mg/kg	40,000[α]	52,400	44,360 ± 8,848	42,800	40,500	40,900 ± 11,600	39,888 ± 4,172
Metals	Lead (Pb)	mg/kg	91.3	35	17.3 ± 2.81	17.1	22.5	16.2 ± 2.46	20.8 ± 2.56
Metals	Lithium (Li)	mg/kg	-	-	34.7 ± 5.36	32.0	48.5	33.3 ± 5.37	39.9 ± 4.92
Metals	Magnesium (Mg)	mg/kg	-	-	13,760 ± 2,748	18,000	16,000	12,918 ± 2,534	14,703 ± 1,915
Metals	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	958 ± 402	1,670	645	882 ± 346	1,683 ± 227
Metals	Mercury (Hg)	mg/kg	0.486	0.170	0.052 ± 0.024	0.022	0.022	0.045 ± 0.010	0.042 ± 0.007
Metals	Molybdenum (Mo)	mg/kg	-	-	3.23 ± 0.556	0.620	0.680	2.69 ± 0.733	0.876 ± 0.086
Metals	Nickel (Ni)	mg/kg	75[α,β]	72	46.6 ± 10.1	68.9	57.3	45.9 ± 6.89	57.2 ± 7.00
Metals	Phosphorus (P)	mg/kg	2,000[α]	1,580	955 ± 154	1,380	765	902 ± 111	885 ± 129
Metals	Potassium (K)	mg/kg	-	-	5,140 ± 946	4,630	6,510	4,862 ± 765	5,810 ± 729
Metals	Selenium (Se)	mg/kg	-	-	0.628 ± 0.252	0.210	<0.20	0.554 ± 0.140	0.233 ± 0.007
Metals	Silver (Ag)	mg/kg	-	-	0.202 ± 0.077	<0.10	0.110	0.172 ± 0.050	0.144 ± 0.010
Metals	Sodium (Na)	mg/kg	-	-	394 ± 80.1	340	411	362 ± 58.6	380 ± 48.2
Metals	Strontium (Sr)	mg/kg	-	-	13.5 ± 2.11	14.3	15.6	13.2 ± 1.41	14.8 ± 1.67
Metals	Sulphur (S)	mg/kg	-	-	1,340 ± 297	<1000	<1000	1,380 ± 444	1,000 ± 0
Metals	Thallium (Tl)	mg/kg	-	-	0.594 ± 0.205	0.366	0.585	0.556 ± 0.129	0.496 ± 0.060
Metals	Tin (Sn)	mg/kg	-	-	<2.000 ± 0	<2.0	<2.0	<2.000 ± 0	<2.000 ± 0
Metals	Titanium (Ti)	mg/kg	-	-	1,204 ± 112	1,330	1,690	1,096 ± 158	1,508 ± 181
Metals	Uranium (U)	mg/kg	-	-	22.0 ± 8.39	4.46	6.20	20.3 ± 7.92	7.92 ± 1.01
Metals	Vanadium (V)	mg/kg	-	-	63.2 ± 11.0	64.2	70.5	60.7 ± 9.23	62.6 ± 7.56
Metals	Zinc (Zn)	mg/kg	315	135	88.0 ± 18.2	63.8	80.4	87.5 ± 11.8	71.3 ± 8.61
Metals	Zirconium (Zr)	mg/kg	-	-	5.16 ± 1.07	10.8	21.9	4.66 ± 1.28	22.4 ± 3.48

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.

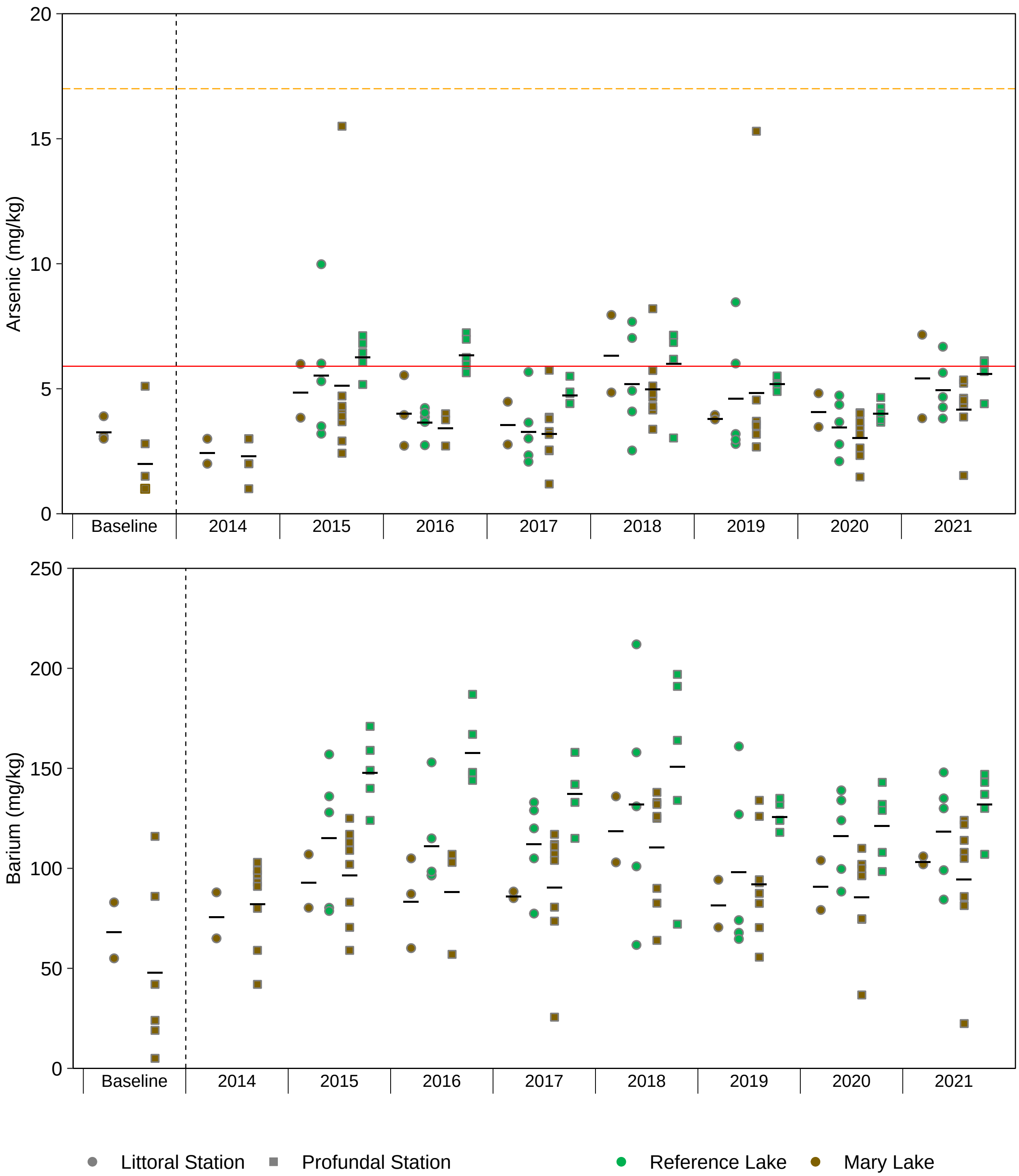
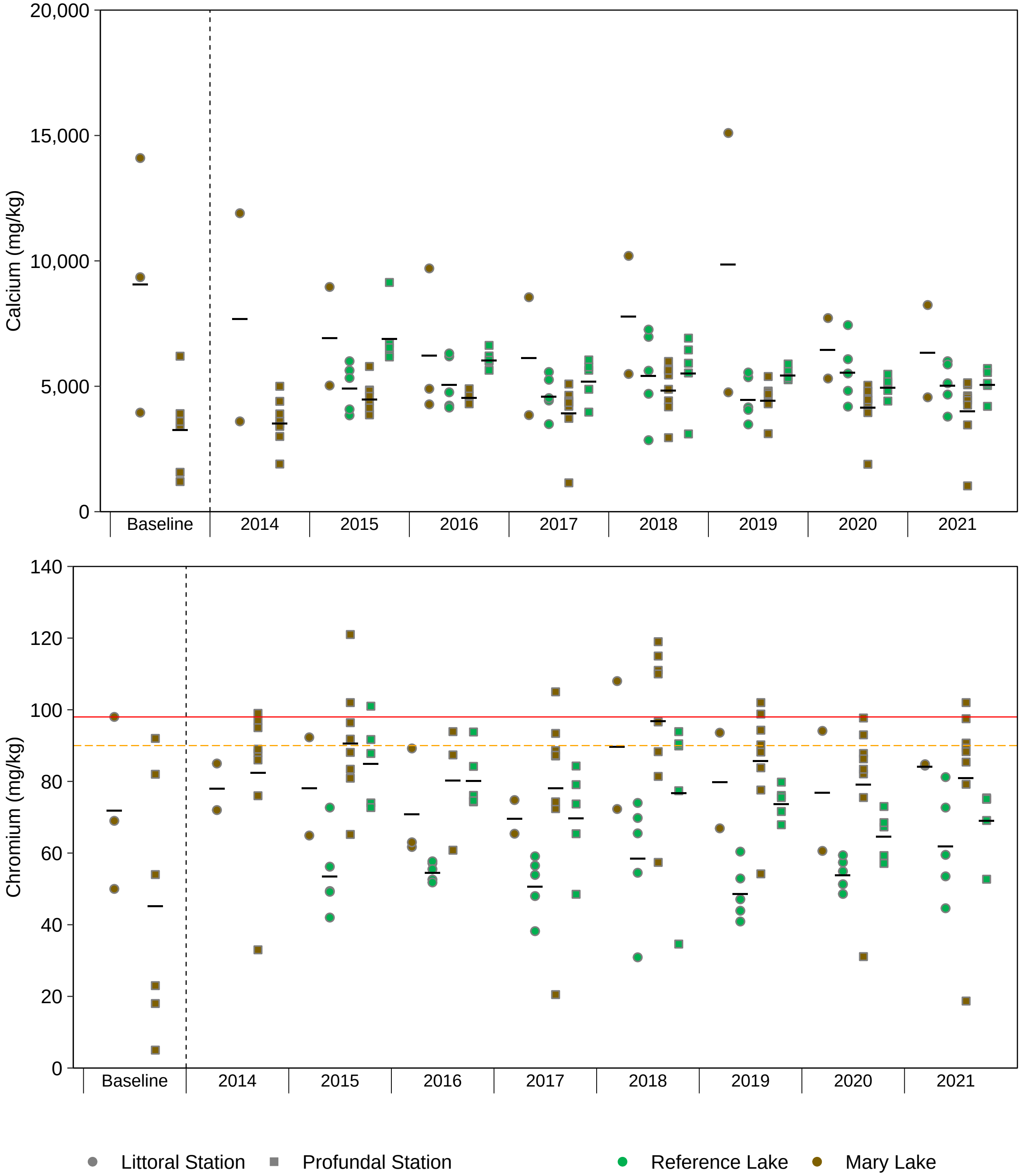


Figure 5.8: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Mary Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.



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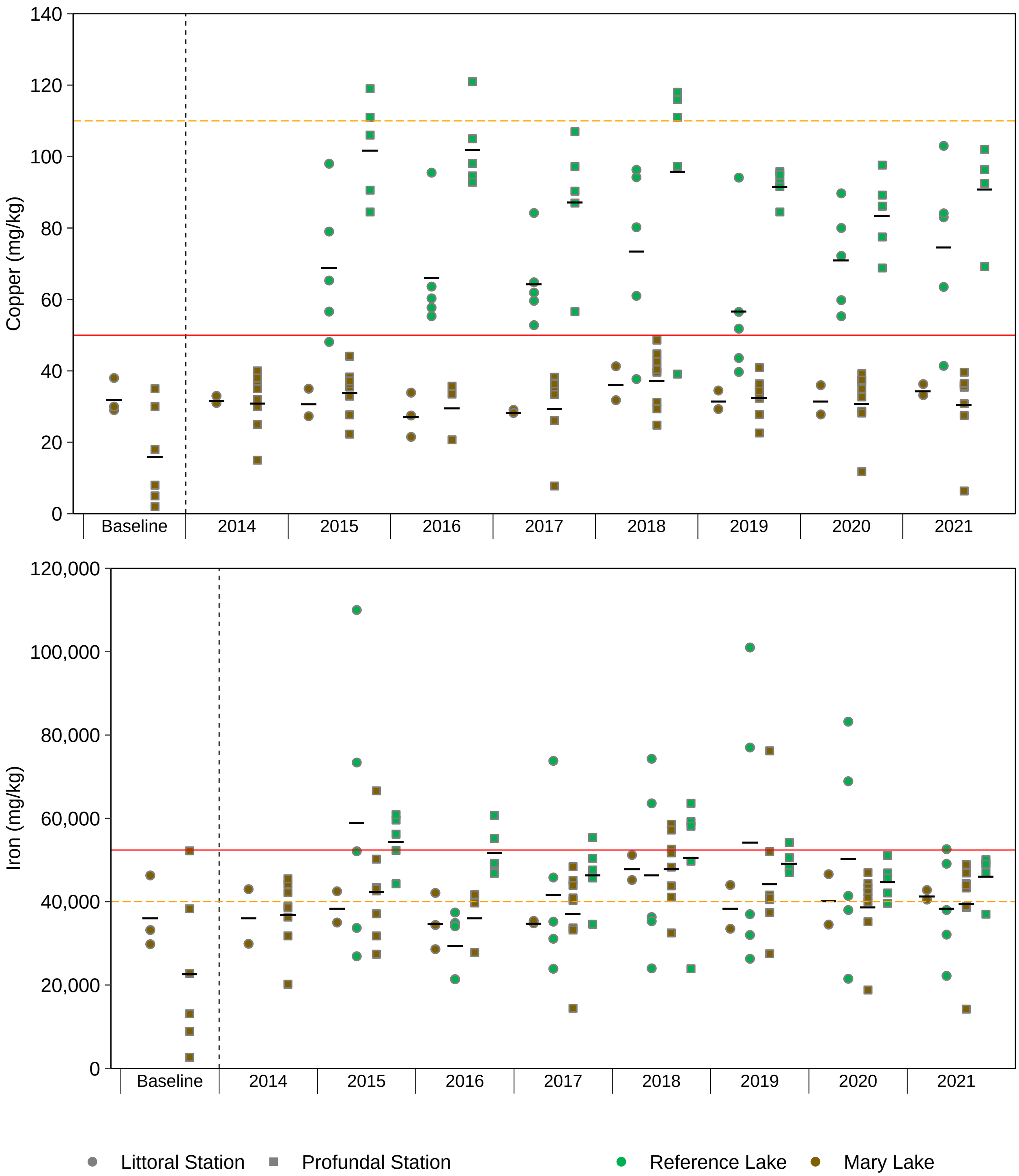

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Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

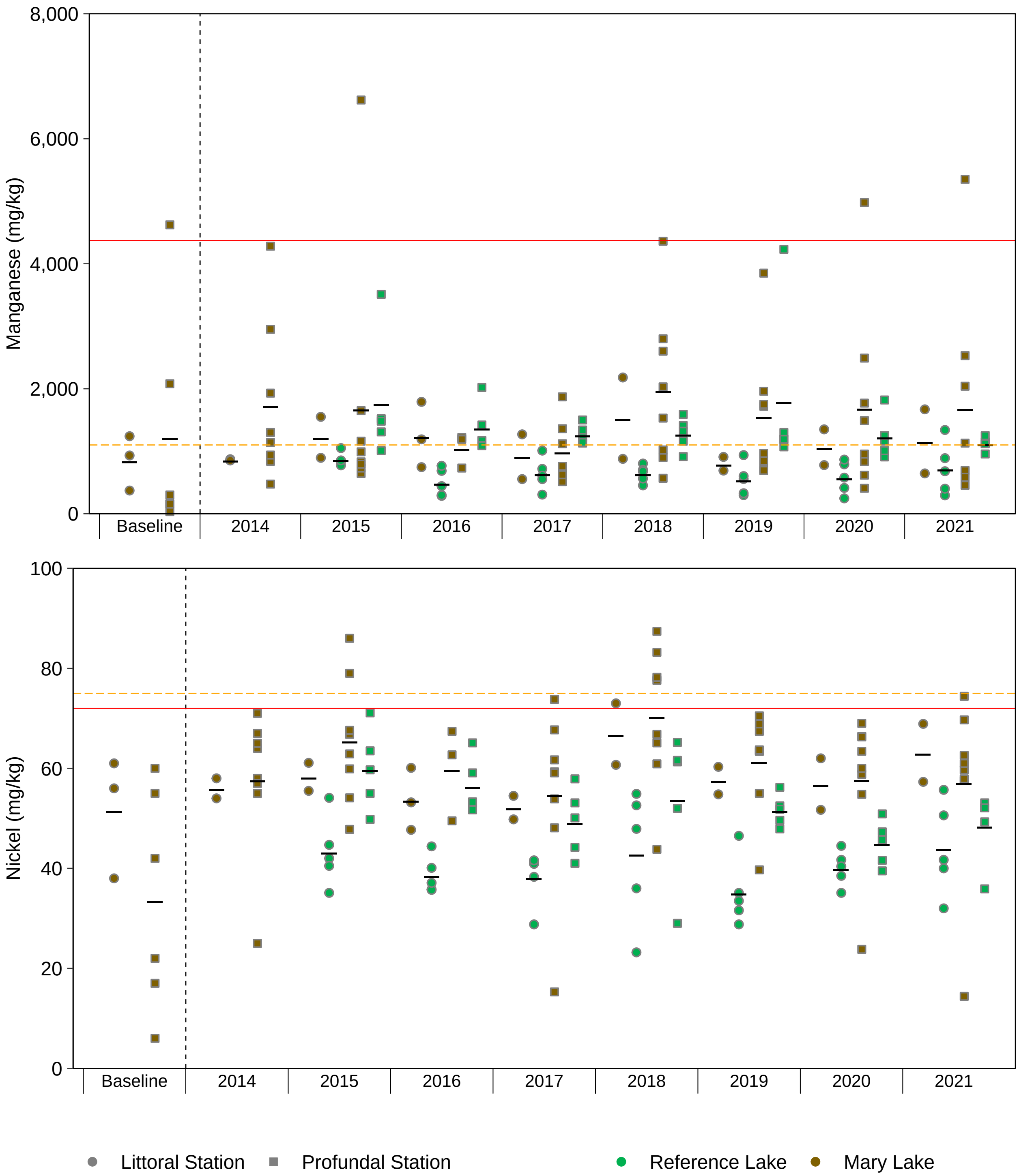

Figure 5.8: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Mary Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

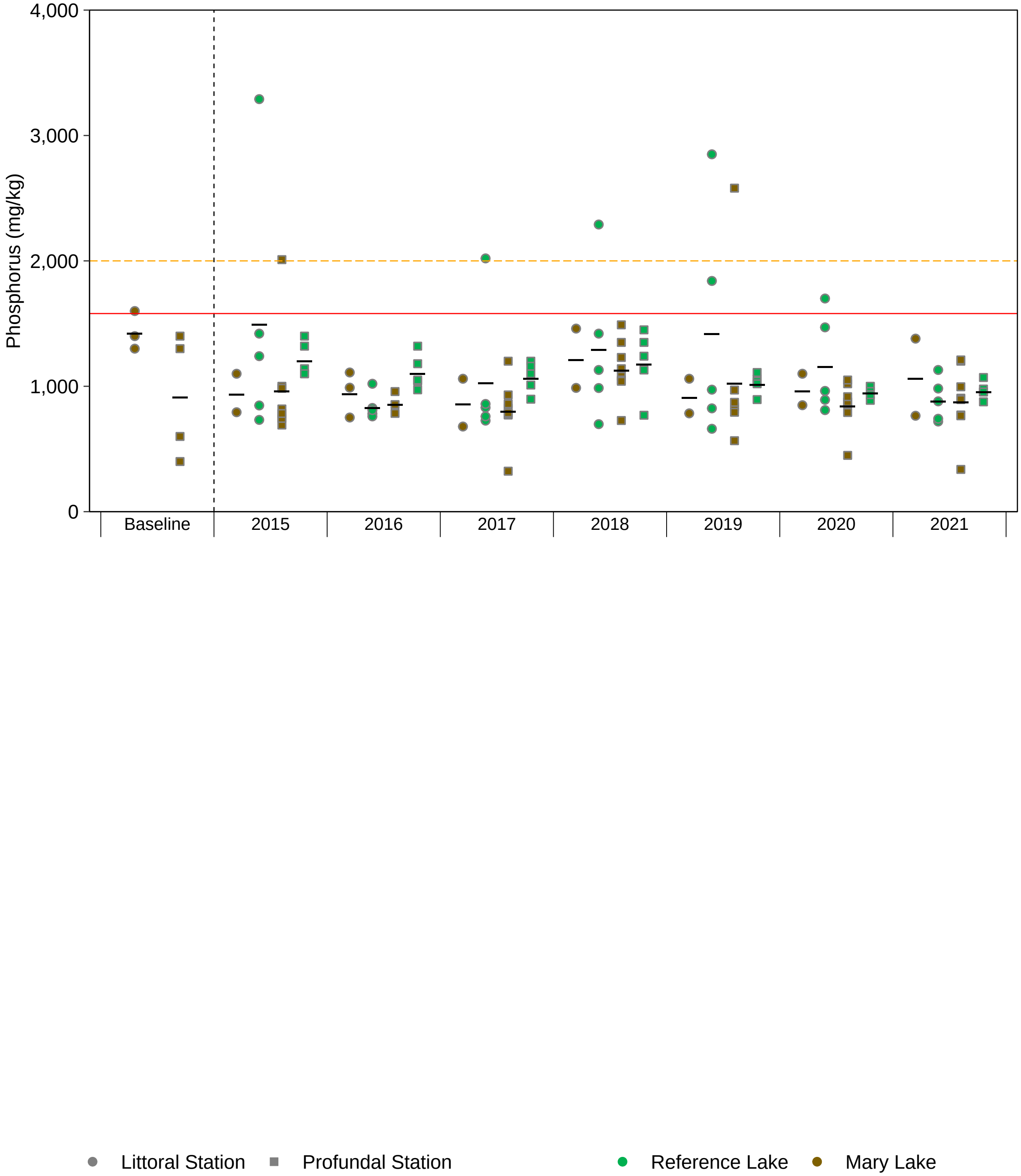

Figure 5.8: Temporal Comparison of Sediment Metal Concentrations at Littoral and Profundal Stations of Mary Lake and Reference Lake 3 for Mine Baseline (2005 to 2013) and Operational (2015 to 2021) Periods

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline, Probable Effect Level or Ontario Provincial Sediment Quality Guideline, Severe Effect Level. Black bars indicate average of samples.

Appendix Table D.24).[24] On average, metal concentrations in sediment from Mary Lake were within the range of those observed from 2015 to 2021 and there was no evidence of an increasing trend over time for any metals (Figure 5.8). Overall, no changes in metal concentrations in sediment were apparent in sediments at Mary Lake since the initiation of commercial mine operations in 2015.

5.2.3 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 2021 (Figure 5.9). Chlorophyll-a concentrations were typically lowest in winter and highest in fall at both the north and south basins of Mary Lake (Figure 5.9). Chlorophyll-a concentrations at the Mary Lake north and south basins were significantly lower than at Reference Lake 3 in the summer, but were significantly higher at the Mary Lake north basin than at the reference lake in the fall of 2021 (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 2021 (Figure 5.9) and reflected an 'oligotrophic' primary productivity categorization based on Wetzel (2001) classification. This oligotrophic categorization agreed with CWQG trophic status classification based on average aqueous total phosphorus concentrations below 10 µg/L (Table 5.4; 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 2021 data and data from the mine construction (2014) period or previous years of mine operation (2015 to 2020) during any of the winter, summer, or fall seasons (Figure 5.10; 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 at either basin of Mary Lake since the mine was constructed in 2014 (Figure 5.10; 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.

5.2.4 Benthic Invertebrate Community

Benthic invertebrate density, richness, and evenness at littoral habitat of Mary Lake did not differ significantly from like-habitat at Reference Lake 3 in 2021 (Table 5.6). Within profundal habitat,

[24] See footnote 12 regarding differences in the concentration of boron in sediment between baseline and recent CREMP studies.

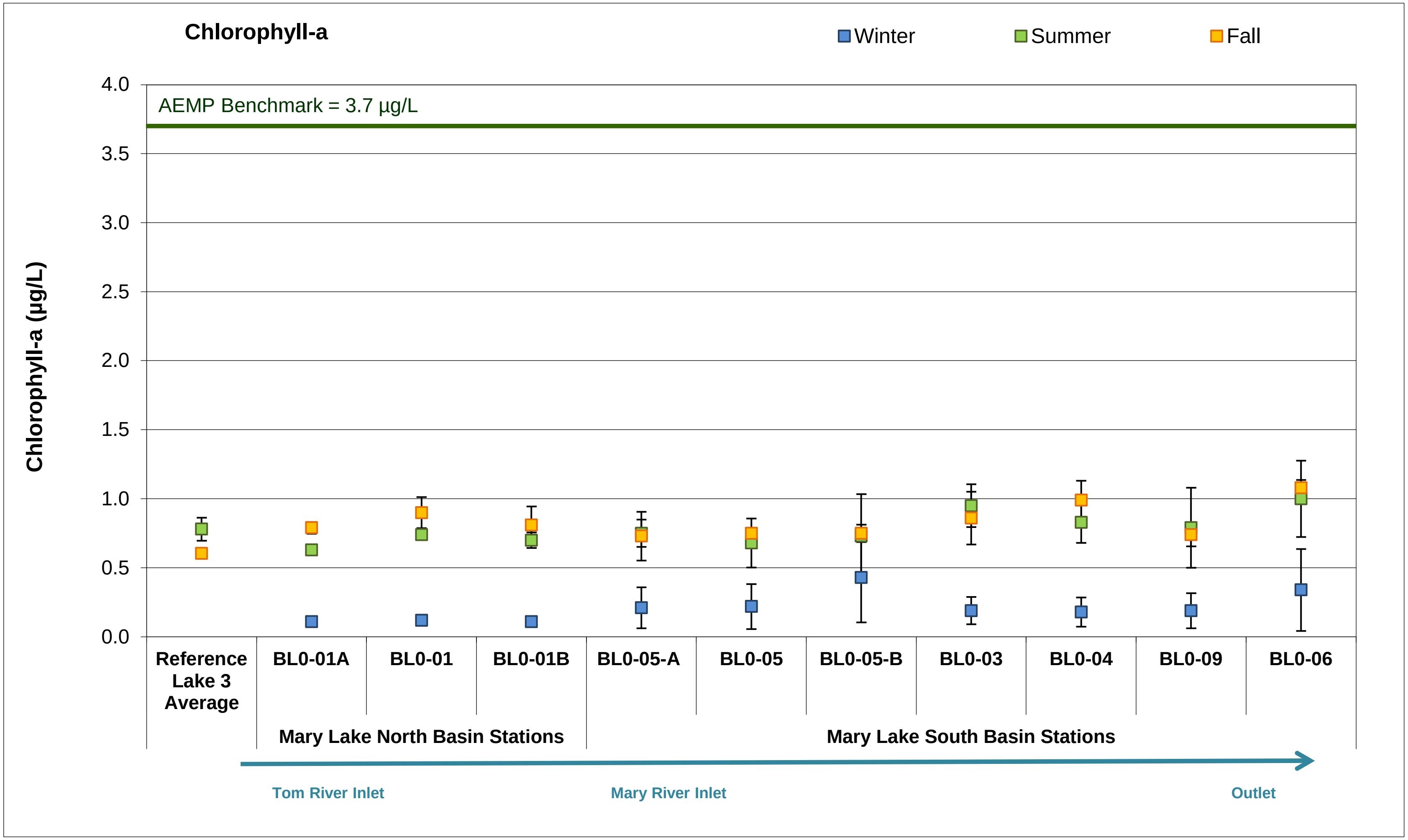

Figure 5.9: Chlorophyll-a Concentrations at Mary Lake (BL0) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2021

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 2021.

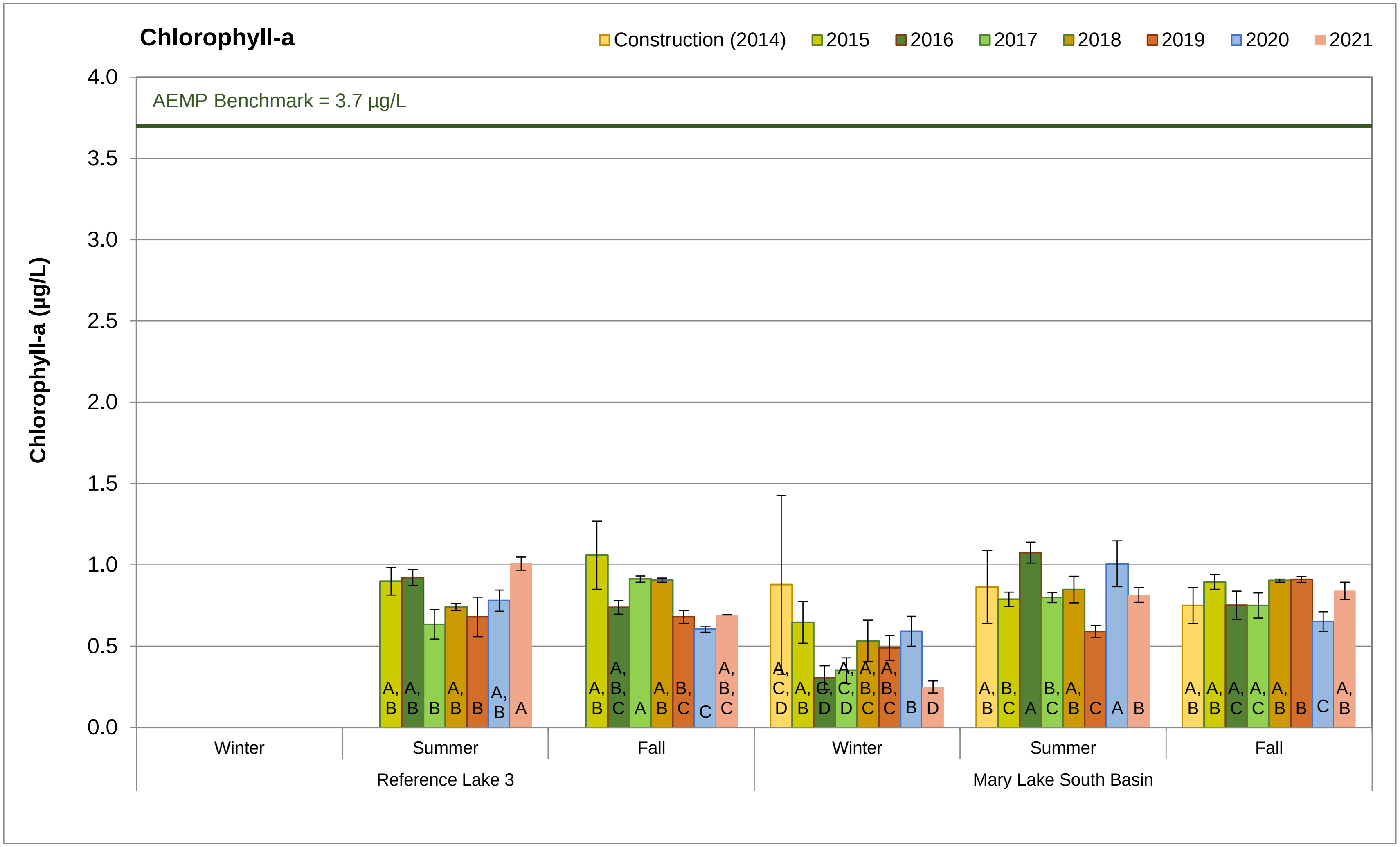

Figure 5.10: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between the Mary Lake South Basin and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2021) Periods (mean ± SE)

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

Table 5.6: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BL0) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Littoral Habitat	Mean	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tunequal	none	NO	0.161	3.6	Reference Lake 3	1,064	448	200	508	1,008	1,516
						Mary Lake Littoral	2,685	1,758	879	456	2,954	4,374
Richness (Number of Taxa)	tequal	none	NO	0.36	0.9	Reference Lake 3	10.8	2.2	1.0	8.0	11.0	14.0
						Mary Lake Littoral	12.8	3.8	1.9	8.0	13.0	17.0
Simpson's Evenness (E)	tequal	none	NO	0.335	0.6	Reference Lake 3	0.703	0.140	0.063	0.502	0.775	0.838
						Mary Lake Littoral	0.783	0.071	0.036	0.695	0.787	0.862
Hydracarina (%)	tequal	none	NO	0.941	0.0	Reference Lake 3	3.1	2.2	1.0	0.0	3.4	5.7
						Mary Lake Littoral	3.0	1.1	0.6	1.9	3.1	4.0
Ostracoda (%)	tequal	none	YES	0.042	-1.3	Reference Lake 3	46.6	27.9	12.5	1.7	47.7	72.7
						Mary Lake Littoral	10.8	6.8	3.4	0.7	13.7	15.1
Chironomidae (%)	tequal	none	YES	<0.001	3.5	Reference Lake 3	38.3	13.2	5.9	20.4	39.3	54.2
						Mary Lake Littoral	84.1	9.2	4.6	74.7	82.5	96.8
Metal-Sensitive Chironomidae (%)	M-W	rank	NO	0.111	-0.6	Reference Lake 3	17.4	10.6	4.7	8.4	11.1	29.1
						Mary Lake Littoral	10.8	9.1	4.5	5.6	6.7	24.4
Collector-Gatherers (%)	tunequal	none	NO	0.319	-2.2	Reference Lake 3	82.0	9.7	4.4	66.7	87.2	90.0
						Mary Lake Littoral	60.7	35.4	17.7	16.5	67.2	92.0
Filterers (%)	tequal	log10(x+1)	NO	0.545	-0.4	Reference Lake 3	11.8	9.5	4.2	2.2	8.0	25.6
						Mary Lake Littoral	7.6	11.4	5.7	0.0	3.0	24.4
Shredders (%)	tequal	log10(x+1)	NO	0.267	-0.6	Reference Lake 3	2.8	3.0	1.4	0.0	1.7	7.8
						Mary Lake Littoral	0.9	1.0	0.5	0.0	0.8	1.9
Clingers (%)	tequal	none	NO	0.998	0.0	Reference Lake 3	14.9	10.6	4.7	2.2	11.7	29.9
						Mary Lake Littoral	14.9	12.2	6.1	1.9	15.6	26.4
Sprawlers (%)	tequal	none	NO	0.839	0.1	Reference Lake 3	64.8	22.8	10.2	30.5	71.6	85.6
						Mary Lake Littoral	67.9	21.3	10.7	37.5	75.0	84.3
Burrowers (%)	tequal	log10	NO	0.633	-0.2	Reference Lake 3	20.4	16.5	7.4	7.4	16.2	49.1
						Mary Lake Littoral	17.2	15.2	7.6	4.4	12.9	38.5

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.

Notes: MOD = Magnitude of Difference = (MCTExp - MCTRef)/SDRef. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases.

benthic invertebrate densities at Mary Lake were significantly greater than at the reference lake in 2021, the magnitude of which was considerably outside of the CES_{BIC} of ±2 SD_{REF}; Table 5.7). No significant differences in general community effect indicators of richness and evenness for profundal habitat between Mary Lake and Reference Lake 3 (Table 5.7). 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). However, the only ecologically significant difference in community composition between Mary Lake and Reference Lake 3 based on evaluation of dominant taxonomic group, FFG, or HPG was significantly higher relative abundance of Chironomidae at Mary Lake (Tables 5.6 and 5.7). Higher benthic invertebrate density with no significant difference in richness, evenness, and relative abundance of metal-sensitive Chironomidae at Mary Lake compared to Reference Lake 3 indicated that Mary Lake was more productive than Reference Lake 3, suggesting that differences in benthic invertebrate community features between lakes were not related to metal concentrations.

No ecologically 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 habitats over years of mine operation (2015 to 2021) compared to baseline, except for significantly higher relative abundance of Ostracods at littoral habitat of Mary Lake (Appendix Figures F.15 and F.16; Appendix Tables F.61 and F.62). No significant differences in the relative abundance of metal-sensitive Chironomidae were indicated for years of mine-operation relative to baseline at Mary Lake (Appendix Tables F.61 and F.62), indicating that the differences in the relative abundance of Ostracoda between the mine operational and baseline periods likely reflected natural variability. Therefore, consistent with no substantial changes in water and sediment quality since the mine baseline period, minimal 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.

5.2.5 Fish Population

5.2.5.1 Mary Lake (South Basin) Fish Community

Arctic charr and ninespine stickleback were captured at Mary Lake in 2021 (Table 5.8), consistent with the species captured at this lake over the previous six years of sampling (Minnow 2021b). Electrofishing and gill netting CPUE were each higher at Mary Lake than at Reference Lake 3 (Table 5.8), 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.

Table 5.7: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BL0) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2021

Metric	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transformation	Significant Difference Between Areas?	p-value	Magnitude of Difference (No. of SD)	Study Lake Profundal Habitat	Mean	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	YES	0.01	6.6	Reference Lake 3	327	151	67.4	60.3	396	422
						Mary Lake Profundal	1,318	666	272	344	1,516	2,101
Richness (Number of Taxa)	tequal	log10	NO	0.412	0.9	Reference Lake 3	6.4	2.6	1.2	2.0	8.0	8.0
						Mary Lake Profundal	8.7	5.4	2.2	5.0	6.5	19.0
Simpson's Evenness (E)	tequal	log10	NO	0.422	-0.4	Reference Lake 3	0.545	0.113	0.050	0.437	0.510	0.731
						Mary Lake Profundal	0.497	0.334	0.136	0.113	0.440	0.949
Hydracarina (%)	tequal	none	NO	0.738	0.3	Reference Lake 3	5.34	4.31	1.93	0.00	4.35	11.90
						Mary Lake Profundal	6.51	6.41	2.62	0.00	5.74	15.00
Ostracoda (%)	tequal	log10(x+1)	NO	0.713	-0.2	Reference Lake 3	8.2	8.7	3.9	0.0	4.4	20.4
						Mary Lake Profundal	6.4	6.4	2.6	0.0	4.3	15.8
Chironomidae (%)	tequal	none	NO	0.95	0.0	Reference Lake 3	86.0	8.7	3.9	71.4	88.1	93.5
						Mary Lake Profundal	86.4	10.1	4.1	70.0	86.4	100.0
Metal-Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.939	0.1	Reference Lake 3	7.8	6.2	2.8	0.0	10.9	14.3
						Mary Lake Profundal	8.3	9.7	4.0	0.6	3.4	22.5
Collector-Gatherers (%)	tequal	log10	NO	0.675	-0.3	Reference Lake 3	86.3	7.8	3.5	80.8	83.7	100.0
						Mary Lake Profundal	83.8	16.4	6.7	62.5	90.5	97.8
Filterers (%)	M-W	rank	NO	0.632	-0.1	Reference Lake 3	6.5	4.9	2.2	0.0	8.9	10.9
						Mary Lake Profundal	5.9	9.2	3.8	0.0	0.0	20.0
Shredders (%)	M-W	rank	NO	0.916	0.1	Reference Lake 3	0.9	1.2	0.5	0.0	0.0	2.2
						Mary Lake Profundal	1.0	2.1	0.8	0.0	0.0	5.1
Clingers (%)	tequal	log10(x+1)	NO	0.979	0.1	Reference Lake 3	12.3	7.0	3.1	0.0	15.2	16.8
						Mary Lake Profundal	13.2	16.1	6.6	0.0	5.7	35.0
Sprawlers (%)	tequal	none	NO	0.598	-0.9	Reference Lake 3	86.0	8.4	3.7	77.5	84.5	100.0
						Mary Lake Profundal	78.9	27.9	11.4	30.6	92.8	98.8
Burrowers (%)	M-W	rank	NO	0.231	2.3	Reference Lake 3	1.7	2.7	1.2	0.0	0.0	6.1
						Mary Lake Profundal	7.9	14.6	6.0	0.5	1.8	37.4

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.

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/SD_{Ref}. MCT = Measure of Central Tendency; SD = Standard Deviation. MCT and SD reported as median and MAD (Median Absolute Deviation) for rank-transformed data and as back-transformed mean and SD for all other cases.

Table 5.8: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Mary Lake (BL0) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	115	4	119	2
		CPUE	1.07	0.040	1.11	
	Gill netting	No. Caught	56	-	56	
		CPUE	1.65	-	1.65	
Mary Lake	Electrofishing	No. Caught	102	31	133	2
		CPUE	1.62	0.490	2.11	
	Gill netting	No. Caught	88	-	88	
		CPUE	3.37	-	3.37	

Note: "-" indicates not applicable.

[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.

Arctic charr CPUE associated with electrofishing in 2021 at Mary Lake was comparable to CPUE from other years of mine operation and substantially greater than baseline monitoring conducted in 2008 (Figure 5.11). Gill netting CPUE at Mary Lake in 2021 was within the range observed during previous years of mine operation (2015 to 2020), and also greater than CPUE during baseline (2006 and 2007; Figure 5.11). 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.

5.2.5.2 Mary Lake (South Basin) Fish Health Assessment

Nearshore Arctic Charr

A total of 102 and 115 arctic charr were captured from nearshore habitats at Mary Lake and Reference Lake 3, respectively, in August 2021 (Table 5.8). Arctic charr YOY were distinguished from non-YOY using fork length cut-offs of 4.2 cm and 4.0 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 5.12; Appendix Tables G.4 and G.25). Because greater than ten YOY arctic charr were identified from the Mary Lake 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 differed significantly between Mary Lake and Reference Lake 3 for both the whole population of nearshore arctic charr and for non-YOY (Appendix Table G.25). This difference reflected slightly larger fish captured at Mary Lake compared to Reference Lake 3 (Figure 5.12). Arctic charr YOY and non-YOY from nearshore areas of Mary Lake were significantly larger and had greater condition than those from Reference Lake 3 (Table 5.9; Appendix Table G.26). The absolute magnitudes of difference in condition for non-YOY were less than the CES_C of 10%; conversely, absolute magnitudes of difference in condition for YOY were greater than the CES_C at Mary Lake compared to the reference lake, suggesting that the differences may be ecologically significant (Table 5.9; 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 2021, suggesting that differences between lakes over time reflected natural variability (Table 5.9). 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.

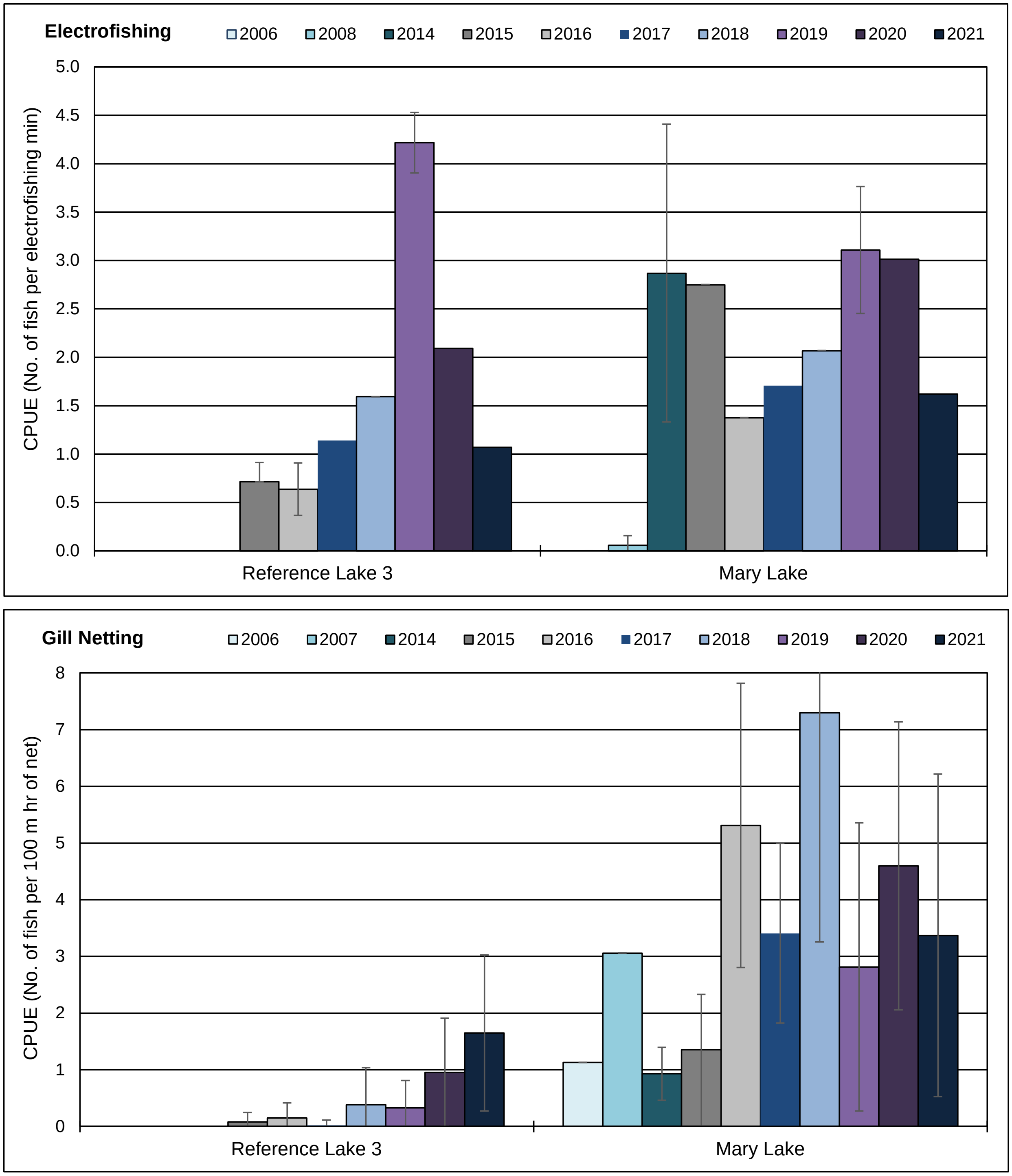

Figure 5.11: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Back-pack Electrofishing and Gill Netting at Mary Lake (BL0), Mary River Project CREMP, 2006 to 2021

Note: Data presented for fish sampling conducted in fall during baseline (2006, 2007), construction (2014), and operational (2015 to 2021) mine phases.

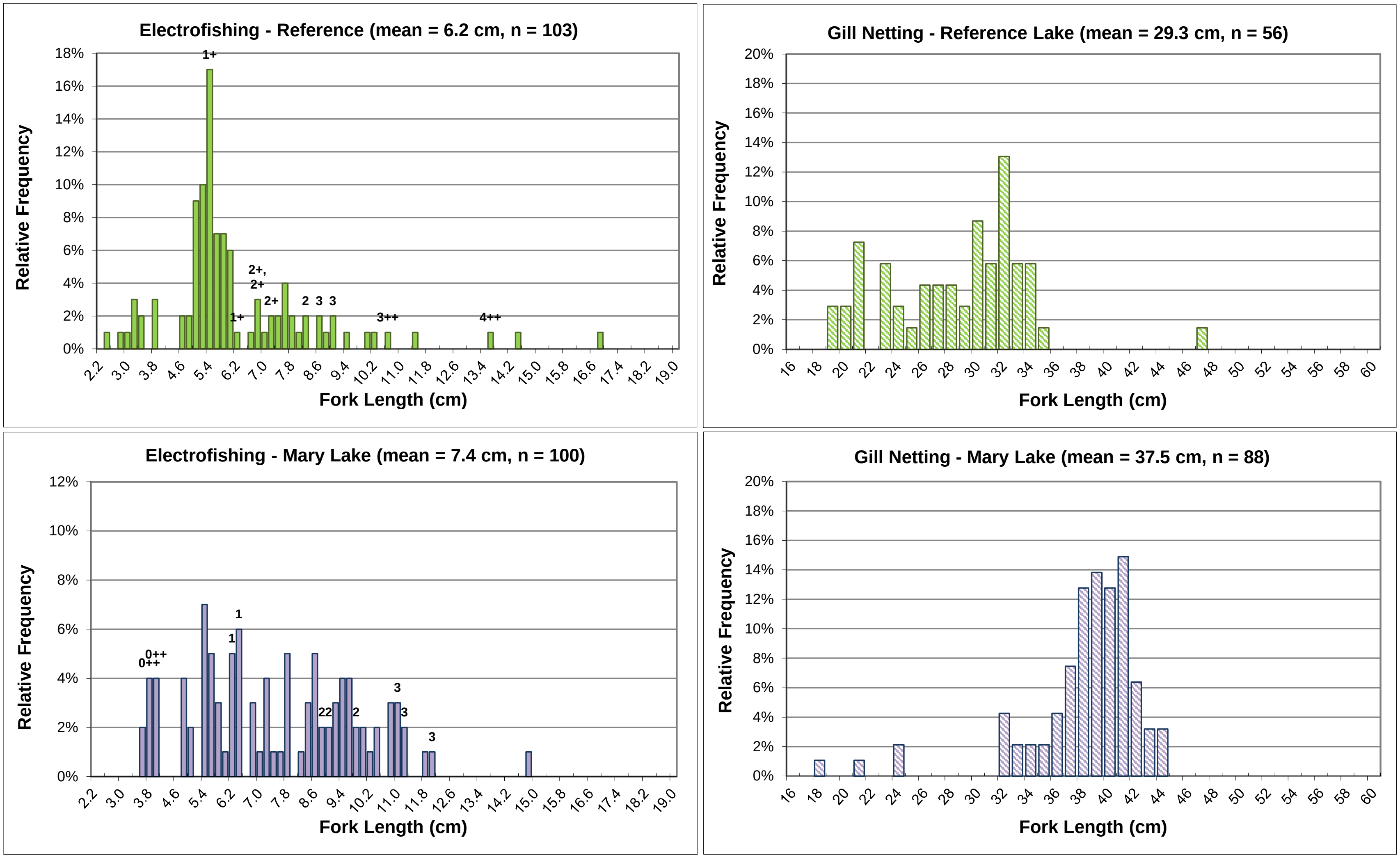

Figure 5.12: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Mary Lake (BL0) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2021

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

Table 5.9: Summary of Statistical Results for Arctic Charr Population Comparisons between Mary Lake and Reference Lake 3 from 2015 to 2021, 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	2021	2015	2016	2017	2018	2019	2020	2021
Electrofishing Samples	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes (-27%)**	**Yes**	-	-	-	-	-	-	-
		Age	**Yes (-43%)**	No	No	-	-	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	No	No	**Yes (+17%)**	**Yes (+10%)**	**Yes (-27%)**	No	**Yes (+39%)**	-	-	-	-	-	-	-
		Size (mean weight)	No	No	**Yes (+51%)**	No	**Yes (-61%)**	No	**Yes (+185%)**	-	-	-	-	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	No	**Yes (-8%)**	**Yes (+4%)**	**Yes (+2.6%)**	**Yes (+5.1%)**	-	-	-	-	-	-	-
Gill Netting Samples [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes (-64%)**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes (+21%)**	No
		Age	-	-	-	-	-	-	-	No	**Yes (-14%)**	No	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	**Yes (+12%)**	**Yes (+24%)**	**Yes (+23%)**	**Yes (+32%)**	**Yes (+6%)**	No	**Yes (-5%)**	No	**Yes (-4%)**	No	No
		Size (mean weight)	-	-	-	**Yes (+51%)**	**Yes (+96%)**	**Yes (+118%)**	**Yes (+186%)**	**Yes (+19%)**	No	**Yes (-9%)**	No	**Yes (-14%)**	No	**Yes (+8.5%)**
		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%)**	**Yes (+28%)**	No	**Yes (+3%)**	**Yes (+5%)**	**Yes (-3%)**	**Yes (-5%)**	No	**Yes (+6.0%)**

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.

Littoral/Profundal Arctic Charr

A total of 88 and 56 arctic charr were sampled from littoral/profundal habitat of Mary Lake and Reference Lake 3, respectively, in August 2021. 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 5.9; Figure 5.12; 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 2021 (Table 5.9; Appendix Table G.30). The absolute magnitude of difference in body condition was greater than the CES_C of 10%, suggesting that this difference was ecologically significant (Table 5.9; 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, but length-frequency distribution did not differ in 2021 (Table 5.9; Appendix Table G.30). No consistent differences in arctic charr size and condition endpoints for fish captured at littoral/profundal habitat of Mary Lake have occurred from 2015 to 2021 compared to baseline (Table 5.9). 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.

5.2.6 Effects Assessment and Recommendations

At Mary Lake, the following AEMP benchmarks were exceeded in 2021:

- Arsenic concentration in sediment was greater than the benchmark of 5.9 mg/kg at one littoral monitoring station (BL0-01; 5.9 mg/kg), although the average concentration of arsenic in sediment at littoral stations was below this benchmark;
- Chromium concentration in sediment was greater than the benchmark of 98 mg/kg at one profundal monitoring station (BL0-08; 102 mg/kg), although the average concentration of chromium in sediment at profundal stations was below this benchmark;
- Manganese concentration in sediment was greater than the benchmark of 4,370 mg/kg at one profundal monitoring station (BL0-09; 5,350 mg/kg), 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 72 mg/kg at one profundal monitoring station (BL0-08; 74.4 mg/kg), although the average concentration of nickel in sediment at profundal stations was below this benchmark.

The AEMP benchmarks for sediment quality were exceeded for four parameters at only a single station for each parameter, across three different sampling stations. Considering the isolated occurrence of these exceedances and that average concentrations of these parameters in sediment at Mary Lake were not elevated compared to concentrations at the reference lake or to those at Mary Lake during baseline, no mine-related changes in arsenic, chromium, manganese, and nickel concentrations were suggested 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 2021 at Mary Lake. In addition, no adverse effects on phytoplankton, benthic invertebrates, nor on fish (arctic charr) health were indicated at Mary Lake in 2021 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 2.6). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2021, 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 2021 CREMP was to evaluate potential mine-related influences on chemical and biological conditions at aquatic environments located near the mine following the seventh full year of mine operation. The CREMP employs an effects-based approach that includes standard 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, and Mary River and Mary Lake systems. The evaluation of potential mine-related effects within these systems was based upon comparisons of the 2021 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 2021 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 2021 for copper at the north branch, and for aluminum and iron at the upper main stem portion 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 2021 were elevated compared to those at reference creeks and to baseline suggesting a potential mine-related influence on CLT1 water quality. However, no adverse effects on phytoplankton (chlorophyll-a) or benthic invertebrates were indicated at CLT1 in 2021 compared to reference creek and CLT1 baseline conditions. Because no mine related changes in concentrations of

Table 6.1: Summary of AEMP Benchmark Exceedances and Effects Determination for the Mary River Project 2021 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 at the north branch (0.00245 mg/L).	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.	No changes recommended to monitoring program for CLT1 north branch. Temporal trend analysis of copper concentrations at the CLT1 north branch will be considered in the 2022 CREMP to confirm that copper concentrations continue to show no increasing trends over time.
Camp Lake Tributary 1 (Main Stem)	Aqueous total aluminum concentration greater than 0.179 mg/L benchmark in spring, summer, and fall in the upper main stem (0.234 mg/L, 0.335 mg/L, and 0.391 mg/L respectively). Aqueous total iron concentration greater than 0.326 mg/L benchmark in spring, summer, and fall at upper main stem (0.444 mg/L, 0.530 mg/L, and 0.824 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 CLT1.	Because a mine-related elevation in iron concentrations occurred at the CLT1 upper main stem in 2021, but the spatial extent was limited and no biological effects were observed, continued monitoring of the benthic invertebrate community at CLT1-L2 is recommended in 2022 (and future CREMP studies) to monitor and track changes in potential effects to biota in the CLT1 upper main stem over time.
Camp Lake Tributary 2	Water quality met all AEMP benchmarks in 2021.	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.
Camp Lake	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 2021. For all parameters, no change in parameter concentration in sediment was shown compared to background and/or baseline, indicating the change was not mine-related. 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 (0.00221 mg/L, 0.00244 mg/L, and 0.00253 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.	No changes recommended to monitoring program for SDLT1. Although no action is required by the AEMP Management Response Framework, Baffinland is currently considering alterations to the SDLT1 system, which may address elevated aqueous copper concentrations.
Sheardown Lake Tributary 12	Aqueous total ammonia concentration greater than the benchmark of 0.855 mg/L during the fall monitoring event. Aqueous nitrate concentration greater than the benchmark of 3 mg/L during the fall monitoring event.	No ecologically significant and/or adverse effects on phytoplankton or benthic invertebrate community endpoints based on comparisons to reference data and to baseline data.	Because routine water quality monitoring at SDLT12 was added during fall 2021 and only a single water quality data point exists, and considering that no adverse effects to biota were indicated, a low action response to continue collecting water quality data at SDLT12 is recommended.
Sheardown Lake Tributary 9	Water quality met all AEMP benchmarks in 2021.	No ecologically significant and/or adverse effects on phytoplankton or benthic invertebrate community endpoints based on comparisons to reference data and to baseline data.	Continued water quality monitoring at SDLT9 is recommended for 2022.
Sheardown Lake Northwest and Southeast basins	Arsenic concentration in sediment greater than AEMP benchmark. Chromium concentration in sediment greater than AEMP benchmark. Copper 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 2021. 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 is recommended 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 and Mary River Tributary-F	Water quality met all AEMP benchmarks in 2021.	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 Mary River and Mary River Tributary-F.
Mary Lake	Arsenic concentration in sediment was greater than AEMP benchmark at one station. Chromium concentration in sediment was greater than AEMP benchmark at one station. Manganese concentration in sediment was greater than AEMP benchmark at one station. Nickel concentration in sediment was greater than AEMP benchmark at one station. The AEMP benchmark was exceeded at one station for each paramter, but the average concentrations of each parameter in sediment were below the respective AEMP benchmarks.	No AEMP water quality benchmarks were exceeded at Mary Lake in 2021. Isolated occurances of AEMP benchmark exceedances in sediment, and the fact that average concentrations of these paramters 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.

AEMP benchmark parameters occurred relative to background and no adverse biological effects were indicated in 2021 at CLT1, in accordance with the Mary River Project AEMP Management Response Framework, no adjustment to the existing AEMP need be applied at CLT1 for the 2022 CREMP. Temporal trend analysis of copper concentrations at the CLT1 north branch will be considered in the 2022 CREMP to confirm no increases in copper concentrations within this portion of the CLT1 system over time. Because a mine-related elevation in iron concentrations occurred at the CLT1 upper main stem in 2021, but the spatial extent was limited and no biological effects were observed, continued monitoring of the benthic invertebrate community at CLT1-L2 is recommended in 2022 (and future CREMP studies) to monitor and track changes in potential effects to biota in the CLT1 upper main stem over time.

At CLT2, water chemistry met all AEMP benchmarks and no adverse effects on phytoplankton or benthic invertebrates were indicated relative to reference creek conditions and CLT2 baseline data in 2021. Because no changes in concentrations of parameters with AEMP benchmarks occurred relative to background and baseline and no adverse biological effects were indicated in 2021, no adjustments to the existing AEMP are recommended for future monitoring at CLT2.

At Camp Lake, no AEMP water quality benchmarks were exceeded (Table 6.1)., Arsenic, copper, iron, manganese, nickel, and phosphorus concentrations in sediment were above the AEMP sediment quality benchmarks at individual stations, although average concentrations of these metals were below benchmarks and comparable to background and/or baseline concentrations (Table 6.1). Under the AEMP Management Response Framework, because no changes in concentrations of parameters with AEMP benchmarks occurred relative to background and baseline, and no adverse biological effects were indicated at Camp Lake in 2021, no adjustments to the existing AEMP are required. However, 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 draw linkages 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 SDLT1, AEMP water quality benchmarks for copper were exceeded in spring, summer, and fall 2021 (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 in 2021 based on comparison to reference creek and baseline data. Because no adverse effects to biota were associated with copper concentrations above the AEMP benchmark at SDLT1, no adjustment to the existing AEMP need be applied at SDLT1 under the AEMP Management Response Framework. Although no action is required by the AEMP Management Response Framework, Baffinland is currently considering alterations to the SDLT1 system, which may address elevated aqueous copper concentrations.

At SDLT12, the AEMP water quality benchmarks for total ammonia and nitrate were exceeded in the single sample collected in fall 2021 (Table 6.1). Phytoplankton and benthic invertebrate community monitoring at SDLT12 indicated no adverse influences of the mine on the aquatic biota of this tributary since the commencement of commercial mine operations in 2015, including in 2021. Because routine water quality monitoring at SDLT12 was added during fall 2021 and only a single water quality data point exists, and considering that no adverse effects to biota were indicated at SDLT12, a low action response to continue collecting water quality data at SDLT12 is recommended to meet obligations under the AEMP Management Response Framework.

Water chemistry at SDLT9 met all AEMP benchmarks in 2021 (Table 6.1). In addition, phytoplankton and benthic invertebrate community monitoring conducted at SDLT9 indicated no adverse influences of the mine on aquatic biota of this tributary since the commencement of commercial mine operations in 2015, including in 2021. Under the Mary River Project AEMP Management Response Framework, because no changes in concentrations of parameters with AEMP benchmarks occurred relative to background and no adverse biological effects were indicated in 2021, no adjustment to the existing AEMP need be applied at SDLT9 as part of the 2022 CREMP. Continued water quality monitoring at SDLT9 is recommended for 2022.

At Sheardown Lake NW, no AEMP benchmarks for water quality were exceeded in 2021. Lake specific AEMP benchmarks for sediment quality were exceeded for arsenic, copper, iron, and nickel in 2021, 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 2021 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 2021, 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 2021. Lake-specific AEMP benchmarks for sediment quality were exceeded for chromium, iron, manganese, and nickel concentrations at Sheardown Lake SE in 2021 (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 2021 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 2021 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 2021 (Table 6.1). No adverse effects on phytoplankton were indicated at MRTF in 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 2021, no changes to the existing sampling program at MRTF are recommended.

Water chemistry Mary River met all AEMP benchmarks in 2021 (Table 6.1). In addition, no adverse effects on phytoplankton or benthic invertebrates were indicated at all Mary River mine-exposed areas in 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 2.6). Because no changes in concentrations of AEMP benchmark parameters occurred relative to background and baseline and no adverse biological effects were indicated in 2021, no management response (i.e., alteration of existing AEMP) is required for Mary River as part of the 2022 CREMP as per the AEMP Management Response Framework.

At Mary Lake, no AEMP benchmarks for water quality were exceeded in 2021. Lake-specific AEMP benchmarks for sediment quality were exceeded for arsenic, chromium, manganese, and nickel concentrations at a single station for each parameter in 2021 (Table 6.1). The isolated occurrence of these exceedances, and the fact that average concentrations of each parameter 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 arsenic, chromium, manganese, and nickel 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 2021 based on comparisons to reference lake conditions and to Mary Lake baseline data. Because no changes in concentrations of parameters with AEMP benchmarks occurred relative to background and baseline and no adverse biological effects were indicated in 2021, no changes to AEMP monitoring at Mary Lake are recommended as per the AEMP Management Response Framework.

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