APPENDIX D.9

AQUATIC EFFECTS MONITORING REPORTS

APPENDIX D.9.1

2016 CORE RECEIVING ENVIRONMENT MONITORING PLAN

Mary River Project 2016
Core Receiving Environment
Monitoring Program Report

Report Prepared For:
Baffinland Iron Mines Corporation
Mary River Project
Oakville, ON

Prepared By:
Minnow Environmental Inc.
Georgetown, ON

March 2017

Mary River Project 2016 Core Receiving Environment Monitoring Program Report

Prepared for:

Baffinland Iron Mines Corp.
Mary River Project

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EXECUTIVE SUMMARY

The Mary River Project is a high-grade iron ore mining operation located in the Qikiqtani Region of northern Baffin Island, Nunavut. Construction of mine infrastructure for the initial mining stages at the Mary River Project, which is owned and operated by Baffinland Iron Mines Corporation (Baffinland), occurred from mid-2013 through 2014. Surface mining commenced in mid-September 2014, and has since included pit bench development, ore haulage and stockpiling, and the crushing and screening of high-grade iron ore at the mine site. Crushed/screened ore is transported by truck to Milne Port, located approximately 100 km north of the mine site, where it is stockpiled before being loaded onto bulk carrier ships for transport to European markets during the summer ice-free period. Because no tailings are produced during the processing of the ore, the only mine waste management facility at the Mary River Project is a waste rock pad and disposal area, which has been established to the east of the current pit bench/mining operation. In addition to periodic discharge of treated effluent from the mine waste rock disposal area to the Mary River system, other potential mine inputs to aquatic systems located adjacent to the mine include runoff and dust from ore (crusher) stockpiles located on the mine site within the Sheardown Lake catchment, treated sewage effluent discharge to Mary River, runoff and explosives residue from quarry operations to the Camp Lake catchment, deposition of fugitive dust generated by mine activities, and general mine site runoff.

Under terms and conditions of a Type A water licence issued by the Nunavut Water Board, Baffinland was required to develop and implement an Aquatic Effects Monitoring Program (AEMP) at the Mary River Project. In order to meet the AEMP objectives for the Mary River Project, Baffinland developed a Core Receiving Environment Monitoring Program (CREMP) to provide a basis for the evaluation of potential mine-related influences on water quality, sediment quality and/or biota (including phytoplankton, benthic invertebrates and/or fish) within aquatic environments near the mine (Baffinland 2014; KP 2014a; NSC 2014). This report presents the results of the 2016 CREMP, including the evaluation of potential mine-related influences on chemical and biological conditions at mine-exposed water bodies following the first full year of mine operation.

The 2016 Mary River Project CREMP included water quality monitoring, sediment quality monitoring, phytoplankton (chlorophyll a) monitoring, benthic invertebrate community assessment and an Arctic charr (*Salvelinus alpinus*) fish population survey. The 2016 CREMP used an effects-based approach that incorporated standard environmental effects monitoring techniques as the basis for the evaluation of potential mine-related effects within the mine aquatic receivers. Additional evaluation of sedimentation-related effects was conducted as

part of the 2016 CREMP in consideration of an Environment and Climate Change Canada *Fisheries Act* Direction (FAD) and an Indigenous and Northern Affairs Canada Letter of Non-Compliance (LNC) related to unauthorized sediment releases in 2016. The primary receiving systems that serve as the focus for the CREMP include the Camp Lake system (i.e., Camp Lake tributaries 1 and 2, Camp Lake), the Sheardown Lake system (i.e., Sheardown Lake tributaries 1, 9 and 12; Sheardown Lake NW and Sheardown Lake SE), and the Mary River and Mary Lake system. The evaluation of potential mine-related effects within these systems was based on comparisons of data collected in 2016 to applicable reference data and to available baseline data. The principal conclusions of the 2016 CREMP for each of these aquatic systems are discussed separately below.

Camp Lake System

Within the Camp Lake system, mine-related effects on water quality were apparent mainly within the main stem channel of Camp Lake Tributary 1 (CLT1) and at Camp Lake. Conductivity and concentrations of mine parameters including chloride, nitrate, sulphate and certain metals (e.g., iron, manganese, molybdenum, sodium, strontium and uranium) were the primary constituents reflecting a mine-related influence within CLT1 and Camp Lake in 2016 based on elevation relative to reference conditions and/or to the baseline (2005 – 2013) period. Of these parameters, only iron and uranium concentrations were above applicable water quality guideline (WQG) and/or AEMP benchmarks, but only at the upper-most monitoring station on the CLT1 main stem. Active quarrying at the QMR2 pit in 2016 likely served as the key source for these parameters at CLT1. Water chemistry at Camp Lake Tributary 2 (CLT2) was similar to applicable reference stations and to baseline water quality, with all parameters consistently observed at concentrations below applicable WQG and AEMP benchmarks. Overall, mine-related effects to water quality of the Camp Lake system were evident at the upper main stem of CLT1 and Camp Lake, with minimal effects suggested at CLT2, following the second year of mine operation. Sediment arsenic and manganese concentrations were slightly elevated at Camp Lake littoral stations compared to mean reference lake concentrations in 2016, and together with molybdenum, were also elevated compared to concentrations during the baseline period, suggesting a mine-related influence on sediment quality of Camp Lake. No metals were elevated in sediment of the profundal stations compared to the reference lake in 2016. Phosphorus was the only parameter observed at concentrations above sediment quality guidelines (SQG) in littoral and profundal sediment of Camp Lake that was not also above applicable SQG at the reference lake in 2016.

Chlorophyll a concentrations were elevated at the upper main stem of CLT1 and within Camp Lake compared to respective reference areas and to baseline data, suggesting slight enrichment possibly related to higher aqueous nitrate and/or micro-nutrient concentrations from Mary River Project mine activities. However, chlorophyll a concentrations at CLT1 north branch and lower main stem areas, and at CLT2 in 2016, were comparable to applicable reference and baseline concentrations. In addition, chlorophyll a concentrations were consistently well below the AEMP benchmark at all Camp Lake system receivers in 2016 indicating no adverse mine influence to phytoplankton. No adverse mine-related influences on the benthic invertebrate community of the Camp Lake system, including CLT1, CLT2 and Camp Lake, were indicated in 2016 based on comparisons to respective reference areas and to baseline data. Consistent with the chlorophyll a data, benthic invertebrate community data collected at the upper main stem of CLT1 suggested a slight enrichment-related influence based on higher invertebrate density, richness and proportion of Functional Feeding Group (FFG) filterers compared to an unnamed reference creek. The fish population survey suggested greater fish abundance compared to the reference lake in 2016, but similar numbers of Arctic charr in 2016 relative to the Camp Lake baseline studies. No significant, ecologically meaningful, differences in Arctic charr condition were indicated between Camp Lake and the reference lake in 2016, nor between Camp Lake Arctic charr collected in 2016 and the baseline period, for nearshore and littoral/profundal Arctic charr populations. Overall, consistent with the water chemistry and sediment chemistry generally meeting respective environmental quality guidelines and AEMP benchmarks, the phytoplankton, benthic invertebrate community and fish population survey data collectively suggested no adverse mine-related influences to the biota of the Camp Lake system in the second year of mine operation at the Mary River Project.

Sheardown Lake System

At Sheardown Lake Tributary 1 (SDLT1), aqueous concentrations of several parameters were elevated compared to average concentrations observed at the reference creek stations in 2016. However, similar to the 2015 CREMP, only nitrate and sulphate concentrations were elevated at SDLT1 in 2016 compared to the baseline period and, with the exception of copper, no parameters were present at concentrations above WQG or AEMP benchmarks in 2016. Within Sheardown Lake, aqueous total concentrations of aluminum, manganese, molybdenum and/or uranium were elevated compared to the reference lake in both 2015 and 2016, but none of these metals, or any other parameters, were elevated compared to concentrations observed during the baseline period, and none were above WQG or AEMP benchmarks. Similar to findings of the 2015 CREMP, elevated total aluminum and manganese concentrations were correlated with greater turbidity in 2016 suggesting that these metals were largely bound to/composed the suspended particulate matter and were not likely biologically available.

Sediment metal concentrations at Sheardown Lake littoral stations in 2016 were similar to those at the reference lake and compared to baseline data with the exception of slightly elevated arsenic, manganese and/or molybdenum concentrations, suggesting some mine-related influences on Sheardown Lake sediment quality. However, sediment metal concentrations at Sheardown Lake profundal stations in 2016 were similar to the reference lake and baseline data, indicating that mine-related influences on sediment quality were confined to littoral habitats. Notably, no metals were present in sediment of Sheardown Lake at concentrations above SQG or AEMP benchmarks that were not also above these criteria at the reference lake, suggesting the natural occurrence of elevated concentrations of some metals (e.g., iron, manganese) in sediment of lakes in the Mary River Project region.

Chlorophyll a concentrations at SDLT1 and Sheardown Lake were greater than concentrations observed at respective reference areas, but were similar to chlorophyll a concentrations reported during mine baseline and construction periods, respectively. In all cases, chlorophyll a concentrations were well below the AEMP benchmark at all Sheardown Lake system monitoring stations, suggesting no adverse mine-related effects to phytoplankton within the system. Consistent with higher chlorophyll a concentrations, greater relative abundance of FFG filterers and organism density at SDLT1 in 2016 compared to an unnamed reference creek and the baseline period, respectively, suggested a slight enrichment influence. However, a greater relative abundance of Habitat Preference Group (HPG) burrowers at SDLT1 and Sheardown Lake Tributary 12 (SDLT12) compared to an unnamed reference creek and to baseline data (SDLT12 only) was potentially indicative of sedimentation influences at these tributaries in 2016. No adverse mine-related influences to benthic invertebrate communities at Sheardown Lake Tributary 9 (SDLT9) and the Sheardown Lake littoral benthic invertebrate community were apparent in 2016 based on comparisons to respective reference areas and/or to baseline data. Greater Arctic charr abundance was suggested at the Sheardown Lake NW and SE basins compared to the reference lake in 2016, but similar abundance was suggested between the 2016 and baseline studies for both lake basins. The Arctic charr population exhibited different direction of significant responses in growth and condition between Sheardown Lake and the reference lake in 2016, and between Arctic charr collected at nearshore and littoral/profundal habitats for Sheardown Lake in 2016 compared to baseline studies. The differential responses in Arctic charr population endpoints suggested that the various differences between the mine-exposed and reference areas, or between studies at Sheardown Lake, reflected natural variability in the resident fish population. Overall, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Sheardown Lake in the second year of mine operation at the Mary River Project.

Mary River and Mary Lake System

At Mary River, no adverse mine-related influences on water chemistry were apparent at the mine-exposed areas in 2016 based on comparisons to the Mary River upstream reference area and to baseline water chemistry taking influences of naturally high turbidity into account. At Mary Lake, aqueous total aluminum, manganese and uranium concentrations were elevated compared to the reference lake in 2016, but concentrations of these metals and all other parameters were comparable to concentrations during the baseline period, and none were above WQG or AEMP benchmarks. Similar to Sheardown Lake and Mary River, aluminum and manganese concentrations were correlated with turbidity at Mary Lake, which suggested that these metals were largely bound to/composed the suspended particulate matter and were thus unlikely to be biologically available. Sediment metal concentrations at Mary Lake littoral and profundal stations were similar to those at the reference lake in 2016 and, with the exception of slightly elevated sediment manganese concentrations at littoral stations, were similar to concentrations observed during the baseline period. Although sediment chromium, iron and manganese concentrations were above SQG at Mary Lake in 2016, with the exception of chromium, these metals were also above respective criteria at the reference lake indicating natural elevation and suggesting low potential for any adverse effects to biota associated with these metals. No metals were observed at concentrations above the sediment AEMP benchmarks at littoral and profundal stations of Mary Lake in 2016.

Chlorophyll a concentrations at Mary River and Mary Lake were, on average, similar to or slightly higher than concentrations at respective reference areas in 2016. Although relatively low chlorophyll a concentrations were observed at individual Mary River stations in 2015 and 2016 compared to the baseline period, these differences likely reflected naturally high turbidity in both 2015 and 2016, which would be expected to affect phytoplankton productivity by limiting the amount of light available for photosynthesis. In all cases, chlorophyll a concentrations were well below the AEMP benchmark, indicating no adverse mine-related influences to phytoplankton of the Mary River/Mary Lake system. The benthic invertebrate community of the Mary River exhibited few differences between mine-exposed and reference areas in 2016, and compared to respective areas during the baseline period, with the direction of the few differences in community composition between areas/studies opposite those normally reflective of an adverse mine-related effect. Benthic invertebrate community data collected at littoral habitat of Mary Lake in 2016 indicated significantly lower richness and differences in community composition compared to the reference lake that appeared to reflect natural differences in sediment physical properties between lakes. In part, this was supported by no significant differences in benthic metrics between 2016 and the baseline data for Mary Lake littoral stations. The fish population survey suggested greater fish abundance at Mary Lake compared to the reference lake in 2016. No significant or ecologically meaningful differences in growth and condition of nearshore captured Arctic charr occurred between Mary Lake and the reference lake in 2016, nor between Arctic charr collected in 2016 and the baseline period for nearshore and littoral/profundal Arctic charr populations at Mary Lake. Overall, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Mary Lake in the second year of mine operation at the Mary River Project.

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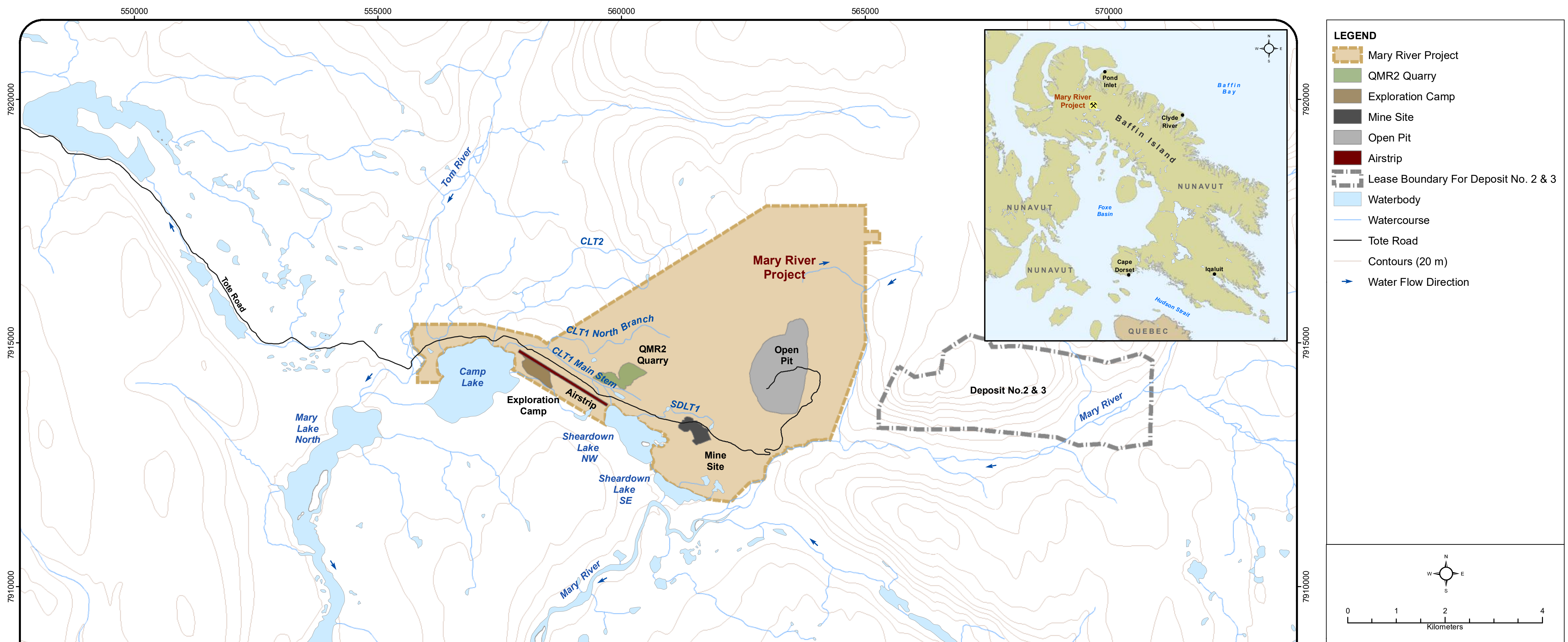
1.0 INTRODUCTION

1.1 Background

The Mary River Project, owned and operated by Baffinland Iron Mines Corporation (Baffinland), is a high-grade iron ore mining operation located in the Qikiqtani Region of northern Baffin Island, Nunavut (Figure 1.1). Construction of mine infrastructure for the initial mining stages at the Mary River Project, referred to as the Early Revenue Phase (ERP), commenced in mid-2013 and is currently on-going. Surface (contour strip) mining for the ERP commenced in mid-September 2014, and has since included pit bench development, ore haulage and stockpiling, and the crushing and screening of high-grade iron ore at the mine site. No milling or additional processing of the ore is conducted on-site. Baffinland has received approval to transport 3.5 million tonnes (Mt) of crushed/screened ore annually by truck to Milne Port, which is located approximately 100 km north of the mine site, for the ERP. At Milne Port, the ore is stockpiled before being loaded onto bulk carrier ships for transport to European markets during the summer ice-free period. No tailings are produced during ore processing, and therefore the only mine waste management facility at the Mary River Project is a waste rock pad and disposal area, which has been established to the east of the current pit bench/mining operation. In addition to periodic discharge of treated effluent from the mine waste rock disposal area to the Mary River system, other potential mine inputs to aquatic systems located adjacent to the mine include runoff and dust from ore (crusher) stockpiles located on the mine site within the Sheardown Lake catchment, treated mine camp sewage effluent discharge to Mary River, runoff and explosives residue from quarry operations to the Camp Lake catchment, deposition of fugitive dust generated by mine activities, and general mine site runoff.

Under terms and conditions of a Type A water licence issued by the Nunavut Water Board (No. 2AM-MRY1325 Amendment No. 1), Baffinland developed an Aquatic Effects Monitoring Program (AEMP) for the Mary River Project. A key objective of the AEMP was to provide data and information to allow the evaluation of short- and long-term effects of the project on aquatic ecosystems. To meet this objective, Baffinland developed a Core Receiving Environment Monitoring Program (CREMP) to assess potential mine-related influences on water quality, sediment quality and biota (including phytoplankton, benthic invertebrates and fish) at aquatic environments located near the mine (Baffinland 2014; KP 2014a; NSC 2014). In 2015, the CREMP approach transitioned from a characterization-based design to an effects-based approach that incorporated standard environmental effects monitoring techniques to allow the evaluation of mine-related effects within the mine aquatic receivers. Briefly, the 2015 study suggested some effects of the Baffinland mine operations on water quality and sediment



MAP INFORMATION
Map Projection: UTM Zone 17N NAD 1983
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Figure 1.1 : Baffinland Iron Mines Corporation, Mary River Project Location

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quality, but these effects were confined to single tributaries feeding into each of Camp and Sheardown lakes, as well as near the immediate outlets of these tributaries to each respective lake (Minnow 2016a). No adverse mine-related effects to phytoplankton, benthic invertebrate, or fish were suggested at any of the Camp, Sheardown or Mary lake systems in 2015 based on comparisons to representative reference waterbodies and to available pre-mine baseline data for each lake system (Minnow 2016a).

The CREMP was designed as an iterative series of monitoring and interpretative phases, with the results of previous studies used to inform the direction of future monitoring. Following the initial 2015 study, some minor adjustments were made to the 2016 CREMP to improve the ability of the program to meet overall objectives and provide greater efficiencies (Baffinland 2016a; Minnow 2016b). The key changes to the CREMP in 2016 included the addition of reference and mine-exposed creek benthic invertebrate community study areas and modification of the lake sediment/benthic invertebrate community survey to improve the ability of the program to assess mine-related influences. The 2016 CREMP also applied additional effort in examination of potential sedimentation-related effects during data evaluation in consideration of an Environment and Climate Change Canada (ECCC) *Fisheries Act* Direction (FAD) and an Indigenous and Northern Affairs Canada (INAC) Letter of Non-Compliance (LNC) issued to Baffinland in June 2016. The FAD and LNC were issued in response to unauthorized sediment releases, and specifically, aqueous Total Suspended Solids (TSS) concentrations above applicable discharge criteria, at several creeks on/adjacent to the mine property, mine tote road and/or mine haul road during May 2016 freshet (Baffinland 2016b).

The 2016 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. This report presents the results of the 2016 CREMP, including the evaluation of potential Mary Lake Project-related influences on chemical and biological conditions at mine-exposed waterbodies following the initial two years of mine operation.

1.2 Report Organization

The content of this report reflects the requirements outlined within the CREMP study design (Baffinland 2014; KP 2014a; NSC 2014) and adjustments to the original program in consideration of the results from the 2015 CREMP (Baffinland 2016a; Minnow 2016b). A description of the aquatic environments that serve as the focus for the CREMP, as well as detailed methods used for evaluation of water quality, sediment quality and biological components (i.e., phytoplankton, benthic invertebrate communities and fish populations) for the 2016 study are provided in Section 2.0. Because of the relatively large geographic scope and multi-component sampling approach used for the Mary River Project CREMP, study results are presented in separate sections according to lake catchment (or sub-catchment, as applicable). Accordingly, water quality, sediment quality and biological effects assessment data and analysis for the Camp Lake system, the Sheardown Lake system (including separate evaluation for the northwest and southeast segments of the lake), and the Mary River/Mary Lake waterbodies are presented in Sections 3.0, 4.0 and 5.0, respectively. The conclusions of the 2016 CREMP are presented in Section 6.0. All references cited within this document are listed in Section 7.0.

Supporting information for the 2016 CREMP is provided in seven appendices. An assessment of the quality of data used for the 2016 study is provided as a Data Quality Review in Appendix A. Natural physico-chemical and biological characteristics important to the assessment of potential mine-related effects at the aquatic mine receiving environments were identified at the study reference areas, and therefore reference conditions are described more fully in Appendix B to provide context and perspective for the CREMP. In addition to all raw water quality data, the results of supplementary baseline lake water quality power analysis conducted to evaluate suitable sample sizes for lake water quality monitoring is presented in Appendix C. Supporting sediment quality information is provided in Appendix D. Finally, supporting biological data from the phytoplankton, benthic invertebrate community and fish population surveys are provided in Appendices E, F and G, respectively.

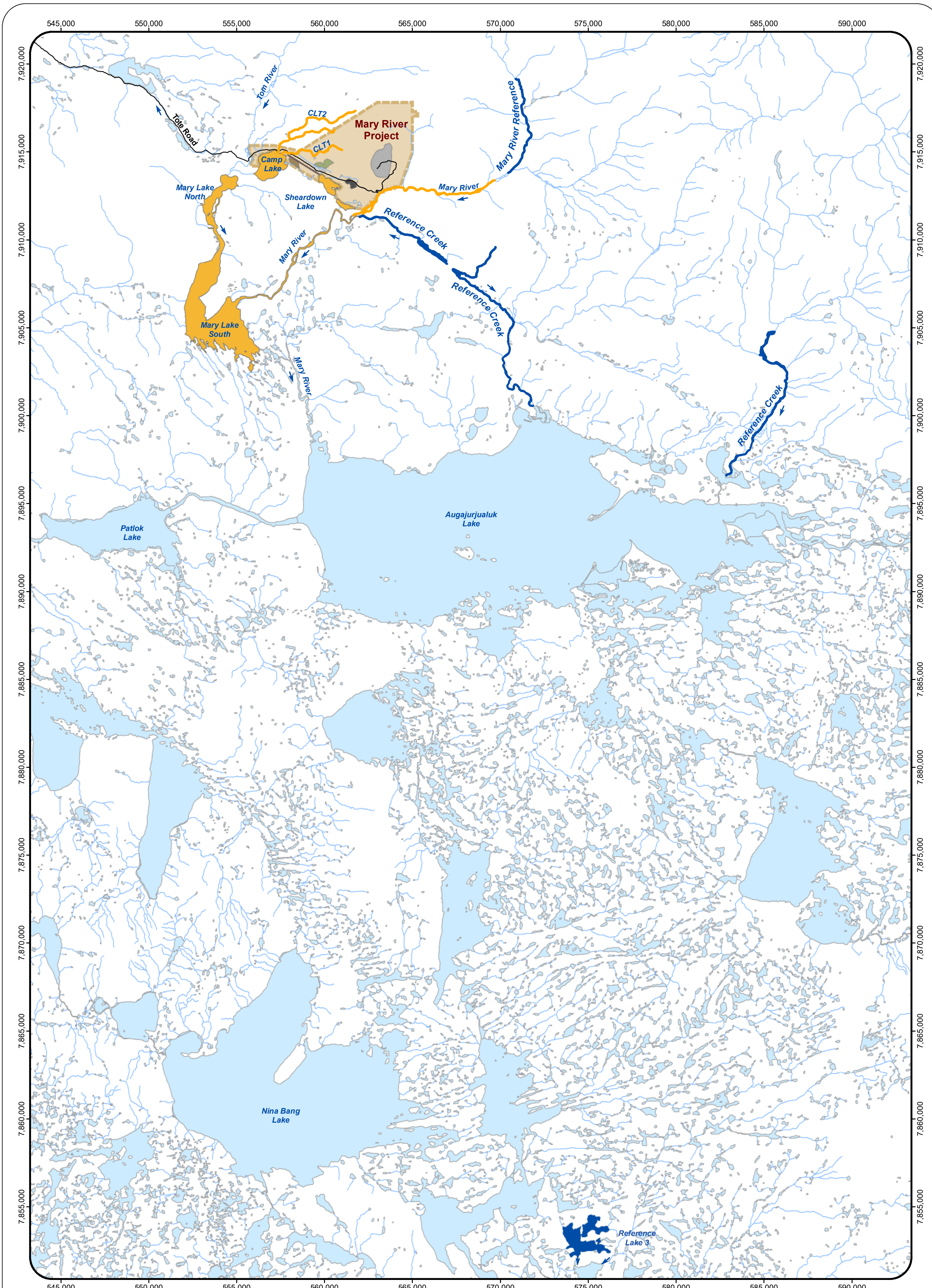
2.0 METHODS

The Mary River Project CREMP includes water quality monitoring, sediment quality monitoring, phytoplankton (chlorophyll a) monitoring, benthic invertebrate community assessment and a fish population assessment. In 2016, water quality and phytoplankton monitoring was conducted by Baffinland personnel over four separate sampling events, including an ice-cover event (April 23rd – May 7th) and open-water season events corresponding to Arctic spring (freshet), summer and autumn (June 25th – 27th, July 18th – 29th, and August 19th – 24th, respectively). Sediment quality, benthic invertebrate community and fish population sampling was conducted by Minnow Environmental Inc. (Minnow) personnel with assistance from Baffinland environment department staff from August 11th – 19th 2016, the seasonal timing of which was consistent with monitoring for previous baseline (2005 – 2013), mine construction (2014), and mine operational (2015) periods at the Mary River Project mine site. Similar to the 2015 CREMP, the 2016 program included field sampling and standard laboratory quality assurance/quality control (QA/QC) for individual water quality, sediment quality and benthic invertebrate community study components to allow for an assessment of the overall quality of each respective data set (Appendix A).

The 2016 CREMP study areas included the same mine-exposed and reference waterbodies established in the original design documents (Baffinland 2014; KP 2014a; NSC 2014) and the same reference lake that was added to the program in 2015 (Figure 2.1). To simplify the discussion of results, the mine-exposed study areas were separated by lake catchment as follows:

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

Reference Lake 3, which served as a reference waterbody for lentic (lake) environments beginning in the previous 2015 CREMP study, was again used as the reference lake for the 2016 study. Reference Lake 3 is located approximately 62 km south of the Mary River Project (Figure 2.1), and is well outside the area of any potential mine influence. Streams used as reference areas in the current and previous CREMP included an unnamed tributary to the Mary River and two unnamed tributaries to Angajurjuatuk Lake, all of which are located southeast of the mine (Figure 2.1). As in the previous CREMP studies, an area of Mary River located well upstream of current Baffinland mine activity (i.e., GO-09) served as a reference area for the mine-exposed portion of Mary River in the 2016 study (Figure 2.1).



LEGEND

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

N W E S

0 2.75 5.5 11
Kilometers

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Figure 2.1: Mary River Project CREMP Study Water Bodies.

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2.1 Water Quality

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

2.1.1 In-situ Water Quality Measurement Data Collection and Analysis

In-situ measurements of water temperature, dissolved oxygen, pH, specific conductance (i.e., temperature standardized measurement of conductivity) and turbidity were taken at the bottom of the water column at all lotic (i.e., creek, river) stations and as a vertical profile at one-meter intervals at each lentic (i.e., lake) water quality monitoring station during routine monitoring conducted by Baffinland. These *in-situ* measurements were also collected at the surface and bottom (i.e., approximately 30 cm above the water-sediment interface) at all lake benthic invertebrate community (benthic) stations during the fall biological sampling completed by Minnow, with the exception of turbidity measurements. The *in-situ* measurements were collected using YSI 556 MDS (Multiparameter Display System) or Pro DSS meters equipped with YSI 6820 or YSI 600Q sondes, respectively (YSI Inc., Yellow Springs, OH). Meter readings for pH, specific conductance and turbidity were checked against standard solutions and calibrated as necessary on the day of field sampling. Dissolved oxygen concentration readings were checked and calibrated at greater frequency through each sampling day in response to changing sampling conditions (e.g., changes in elevation, barometric pressure and/or ambient temperature). During the April-May under-ice sampling event, a gas-powered, 15 centimeter (6-inch) diameter ice auger was used to access the water column at all lake water quality monitoring stations. All ice shavings were removed from the auger hole prior to the collection of *in-situ* measures. To avoid confounding influences associated with snow/ice melt in the auger hole, the *in-situ* measurements were collected beginning just below the ice layer. Additional supporting observations of water colour and clarity were recorded at the time of water quality and biological sampling at all benthic stations, and Secchi depth was measured at all lake stations using the methods outlined in Wetzel and Likens (2000).

In-situ water quality data collected at the mine-exposed study streams, rivers and lakes were compared to respective reference area data, to applicable water quality guidelines (i.e., the Canadian Water Quality Guidelines [WQG; CCME 1999, 2016]) and, for pH and conductivity, to baseline data. The evaluation of the *in-situ* dissolved oxygen concentration and pH data included comparisons to WQG. *In-situ* water quality data were compared spatially within each system (i.e., from upstream- to downstream-most stations) using both qualitative and statistical

Table 2.1: Mary River Project CREMP water quality and phytoplankton monitoring station coordinates and annual sampling schedule.

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

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

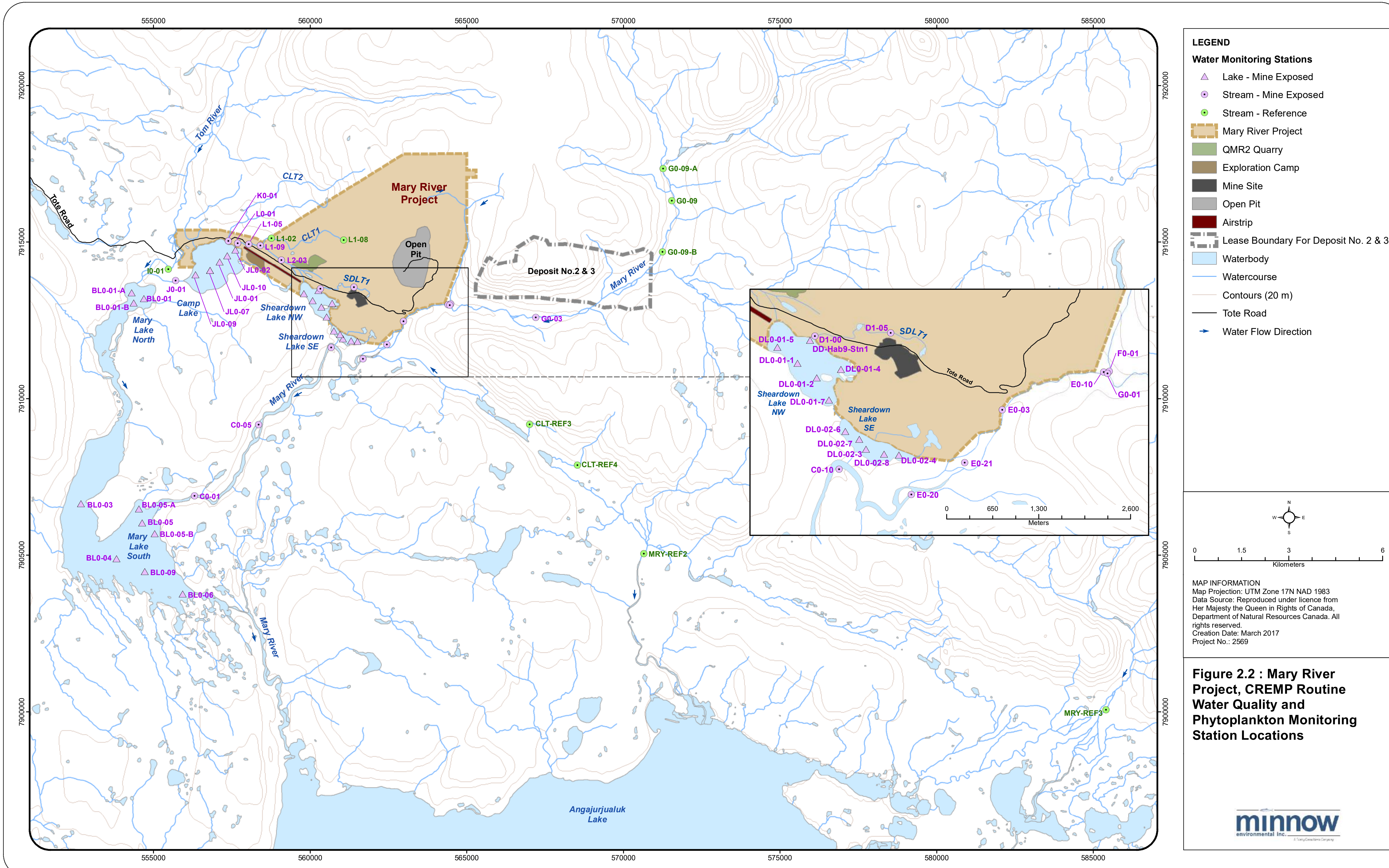
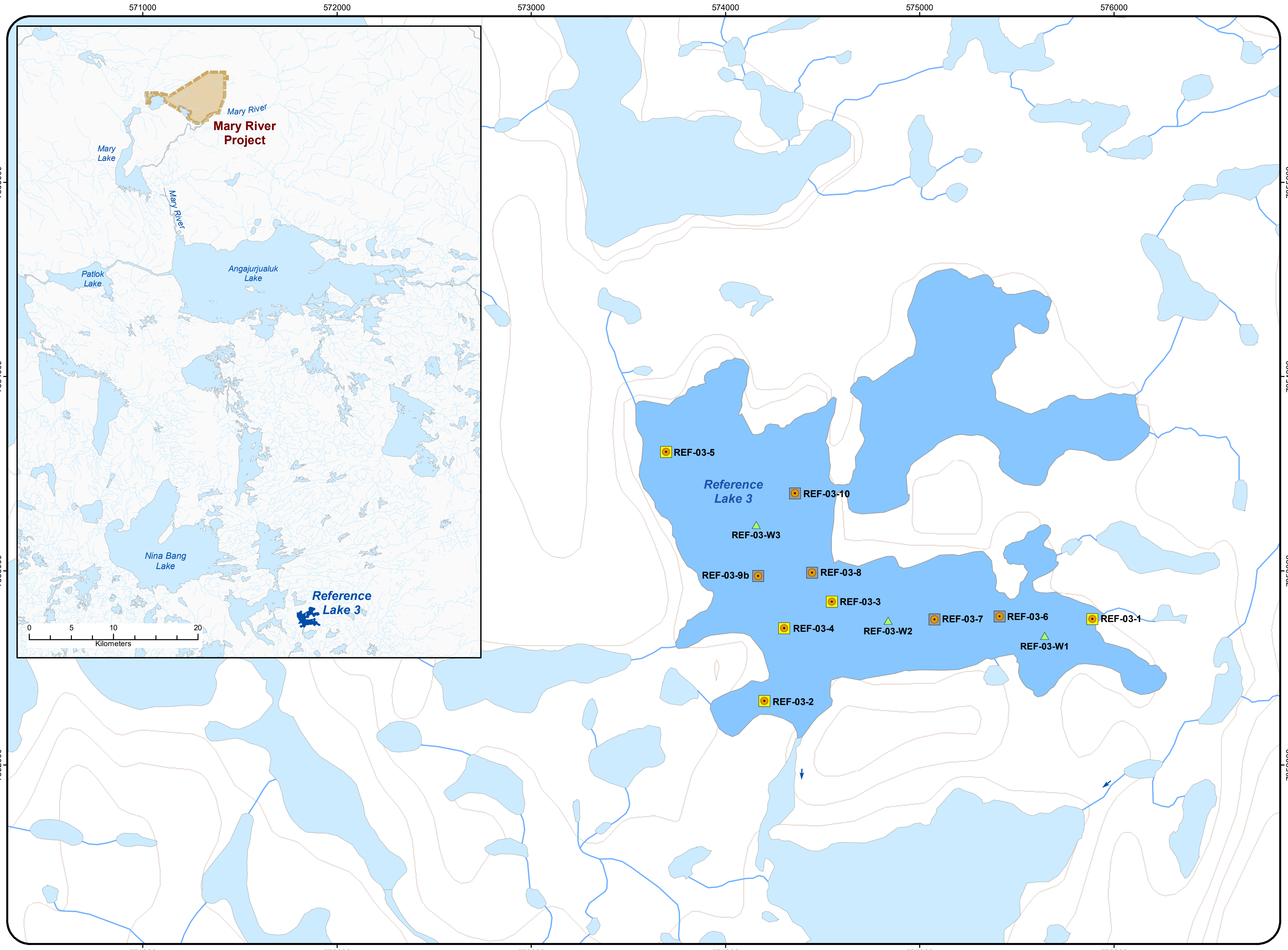


Figure 2.2 : Mary River Project, CREMP Routine Water Quality and Phytoplankton Monitoring Station Locations



LEGEND

- Sediment and Benthic Monitoring Location
- Water Quality and Phytoplankton Monitoring Station
- Littoral Sampling Depth
- Profundal Sampling Depth
- Reference Lake
- Waterbody
- Watercourse
- Tote Road
- Contours (20 m)
- Water Flow Direction

MAP INFORMATION
Map Projection: UTM Zone 17N NAD 1983
Data Source: Reproduced under licence from Her Majesty the Queen in Rights of Canada, Department of Natural Resources Canada. All rights reserved.
Creation Date: March 2017
Project No.: 2569

Figure 2.3: Mary River Project CREMP Reference Lake 3 Monitoring Station Locations

minnow environmental inc.

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 areas using Analysis-of-Variance (ANOVA). The selection of whether untransformed or log-transformed data were used for the ANOVA tests was determined based on which data best met the assumptions of ANOVA. In instances where normality could not be achieved through data transformation, non-parametric Mann-Whitney U-test and Kruskal-Wallis H-test statistics were applied using the raw data to validate the pair-wise and multiple-group ANOVA statistical results, respectively. Similarly, in instances in which variances of normal data could not be homogenized by transformation, Student's t-tests assuming unequal variance were applied using either raw or log-transformed data to validate the pair-wise ANOVA statistical results. In cases in which multiple-group comparisons were conducted, Tukey's Honestly Significant Difference (HSD) or Tamhane's pair-wise *post-hoc* tests were implemented for homogenous and non-homogenous data, respectively All statistical comparisons were conducted using SPSS Version 12.0 software (SPSS Inc., Chicago, IL).

Vertical profiles of the *in-situ* measurements taken from lake stations were plotted and visually assessed to evaluate potential thermal, dissolved oxygen or chemical (i.e., pH and/or specific conductance) stratification and the corresponding depths associated with any distinct layering. The occurrence of a thermocline was assessed as a ≥1˚C change in temperature per 1 m incremental change in depth (Wetzel 2001). The vertical profile data collected at the mine-exposed study lakes were compared to that of the reference lake for each seasonal monitoring event using profile data averaged for each incremental depth below the water surface among lake stations by season. At each study lake, spatial and seasonal differences in the vertical profile plots were evaluated to provide better understanding of natural conditions and/or mine-related influences on within-lake water quality. Additional evaluation of the *in-situ* dissolved oxygen concentration and pH data included comparisons to WQG (CCME 1999, 2016).

2.1.2 Water Chemistry Sampling and Data Analysis

Surface water chemistry samples were collected from both lotic and lentic environments (Table 2.1). At lotic stations, the water chemistry samples were collected from approximately mid-water column by hand directly into pre-labeled sample bottles which, for those requiring preservation, were pre-dosed with required chemical preservatives. At lentic stations, two water chemistry samples were collected, one approximately 1 m below the surface (or just below the ice layer for the winter sampling event) and the other from approximately 1 m above the bottom, using a non-metallic beta-bottle, vertically-oriented 2.2 L TT Silicon Kemmerer bottle (Wildco Supply Co., Yulee, FL) or, for winter sampling only, a stainless steel Kemmerer bottle. During the winter sampling event, the water column was accessed at the same time and using the same methods as described above for the *in-situ* measurements. Lake water collected using the beta-bottle/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 protocols (Baffinland 2014).

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

The water chemistry data were compared: i) among mine-exposed and reference areas for each study lake catchment (Table 2.1); ii) spatially and seasonally at each mine-exposed waterbody; iii) to applicable water quality guidelines/objectives for the protection of aquatic life (Table 2.2); iv) to site specific water quality benchmarks developed for the Mary River Project AEMP (Intrinsik 2014); and, v) to baseline water quality data. For data screening, and to simplify discussion of results, the magnitude of difference in parameter concentrations was calculated as the mine-exposed area mean concentration divided by the respective reference station/area mean concentration using the 2016 data. Similarly, for temporal comparisons, the magnitude of difference in parameter concentrations was calculated by dividing the individual mine-exposed station/area 2016 mean concentrations by the baseline (2005 - 2013) mean concentration for each parameter. The resulting magnitude of differences in parameter concentrations were qualitatively assigned as slightly, moderately or highly elevated compared to reference and/or baseline conditions using the categorization described in Table 2.3.

Applicable water quality guidelines/objectives included CWQG (CCME 1999, 2016) 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, 2016). The water quality

Table 2.2: Water quality guidelines used for the Mary River Project 2015 and 2016 CREMP.

Parameters		Units	Water Quality Guideline (WQG)[a]	Criteria Source[a]	Supporting Information and/or Calculations Used to Derive Hardness Dependent Criteria
Conventionals	pH (lab)	pH	6.5 - 9.0	CWQG	
Nutrients and Organics	Nitrate	mg/L	13	CWQG	
	Nitrite	mg/L	0.06	CWQG	
	Total Phosphorus	mg/L	0.020	PWQO	Total phosphorus objective is 0.020 mg/L for lotic (rivers, streams) environments, and 0.030 mg/L for lentic (lake) environments.
	Phenols	mg/L	0.001	PWQO	
Anions	Chloride (Cl)	mg/L	120	CWQG	
	Sulphate (SO_4)	mg/L	218	BCWQG	Sulphate guideline is hardness (mg/L $CaCO_3$) dependent as follows: 128 mg/L at 0 - 30 hardness, 218 mg/L at 31 - 75 hardness, 309 mg/L at 76 - 180 hardness, and 429 mg/L at 181 - 250 hardness. Sample-specific (mean) hardness was used for screening purposes. Value presented applicable to water with 75 mg/L hardness.
Total Metals	Aluminum (Al)	mg/L	0.100	CWQG	
	Antimony (Sb)	mg/L	0.020	PWQO	
	Arsenic (As)	mg/L	0.005	CWQG	
	Beryllium (Be)	mg/L	0.011	PWQO	
	Boron (B)	mg/L	1.5	CWQG	
	Cadmium (Cd)	mg/L	0.00012	CWQG	Cadmium guideline is hardness (mg/L $CaCO_3$) dependent. For hardness between 17 and 280 mg/L, the cadmium guideline is calculated using the equation Cd (ug/L) = $10^{(0.83[\log(\text{hardness}] -2.46)}$. 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 - 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 - 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 - 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	
	Tin (Sn)	mg/L	-	-	
	Titanium (Ti)	mg/L	-	-	
	Tungsten	mg/L	0.030	PWQO	
	Uranium (U)	mg/L	0.015	CWQG	
	Vanadium (V)	mg/L	0.006	PWQO	
	Zinc (Zn)	mg/L	0.030	CWQG	

[a] Canadian Environment Water Quality Guideline for the protection of aquatic life (CCME1999, 2016) 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; MOE 1994) or the British Columbia Water Quality Guideline (BCWQG; BCMOE 2013), as available.

Table 2.3: Categorization of magnitudes of difference used for screening parameter concentrations between mine-exposed areas and reference areas, and between 2016 and baseline data for individual mine-exposed stations/areas, Mary River Project CREMP, 2016.

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

guidelines used in this 2016 CREMP were abbreviated simply as 'WQG', although it is recognized that in certain cases the values presented may represent water quality 'objectives'. For those water quality guidelines that are hardness dependent, the hardness of the individual sample was used to calculate the water quality guideline for the specific parameter according to established formulae (Table 2.2). The 2016 water chemistry data were also compared to site specific water quality benchmarks developed for the Mary River Project AEMP (Intrinsik 2014). The Mary River Project AEMP water chemistry benchmarks were derived using an evaluation of background (i.e., baseline) water chemistry data together with existing generic water quality guidelines that consider aquatic toxicity thresholds. The AEMP benchmarks were developed to inform management decisions under the AEMP assessment approach and management response framework (Baffinland 2014). An elevation in parameter concentration above the respective AEMP benchmark may trigger various actions (e.g., sampling design modifications, additional statistical assessment, considerations for mitigation, etc.) to better understand and potentially mitigate effects resulting from elevated concentrations of the parameter of concern (Baffinland 2014). Water chemistry data for key parameters (i.e., parameters with concentrations that were notably higher at mine-exposed areas compared to reference areas, that were historically identified as site-specific parameters of concern, and/or that were above WQG and/or AEMP benchmarks) were plotted to evaluate changes in concentrations in 2016 compared to baseline (2005 – 2013) and previous mine construction (2014) and operational (2015) periods.

2.2 Sediment Quality

The objective of the sediment quality monitoring component of the original Mary River Project CREMP was to assess the potential effects of mine operation on sediment quality of lake environments based on a gradient design (Baffinland 2014; KP 2014a, 2015). In 2016, the lake sediment quality monitoring approach was modified to an effects-based design that included both sediment quality and benthic invertebrate community sampling at littoral stations while maintaining key profundal stations for the long-term monitoring of changes in lake sediment chemistry. Under the modified 2016 design, sediment quality sampling was conducted at five littoral stations (i.e., water depths approximately between 7 m and 12 m) and three profundal stations (i.e., water depths greater than approximately 18 m) at each study lake except Sheardown Lake SE (Table 2.4; Figure 2.4). Because the maximum depth of Sheardown Lake SE reaches approximately 14 m, only 'littoral' depth samples were collected at this lake. Although the CREMP also proposed sediment sampling within Camp Lake tributaries (three stations), Sheardown Lake tributaries (six stations) and within the Mary River (four stations), as in previous studies conducted in 2014 and 2015, these watercourses were found to contain limited depositional habitat during the 2016 field survey. The general absence of any substantial accumulation of fine sediments within these watercourses precluded any meaningful assessment of potential mine-related influences on sediment chemistry within, along and/or between watercourses, and therefore no sediment sampling was conducted at lotic environments as part of the 2016 CREMP.

2.2.1 Sample Collection and Laboratory Analysis

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

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

Table 2.4: Mary River Project CREMP lake sediment quality and benthic invertebrate community monitoring station coordinates, 2016.

Waterbody	Station Code	UTM Zone 17W, NAD83		New (2016) or Existing Station	Sampling Habitat	Sample Type	
		Easting	Northing			Sediment Sampling [a]	Benthic Invertebrate Community
Reference Lake	REF-03-1	575889	7852752	Existing	littoral	✓	✓
	REF-03-2	574200	7852330	Existing	littoral	✓	✓
	REF-03-3	574548	7852842	Existing	littoral	✓	✓
	REF-03-4	574301	7852705	Existing	littoral	✓	✓
	REF-03-5	573694	7853613	Existing	littoral	✓	✓
	REF-03-6	575411	7852766	Existing	profundal	✓	✓
	REF-03-7	575076	7852750	Existing	profundal	✓	✓
	REF-03-8	574445	7852992	Existing	profundal	✓	✓
	REF-03-9[b]	574168	7852975	Existing	profundal	✓	✓
	REF-03-10	574358	7853400	Existing	profundal	✓	✓
Camp Lake	JLO-02[b]	557619	7914753	Existing	littoral	✓	✓
	JLO-21	556926	7914911	Existing	littoral	✓	✓
	JLO-32	557590	7914174	New	littoral	✓	✓
	JLO-14	557246	7914224	Existing	profundal	✓	-
	JLO-31	557213	7913826	New	littoral	✓	✓
	JLO-07	556803	7914095	Existing	profundal	✓	-
	JLO-11	556594	7913929	Existing	profundal	✓	-
	JLO-30	556446	7913562	New	littoral	✓	✓
Sheardown Lake Northwest (NW)	DLO-01-5	559806	7913348	Existing	profundal	✓	-
	DD-HAB 9-STN2	560315	7913398	Existing	littoral	✓	✓
	DLO-01-8	560338	7913192	Existing	littoral	✓	✓
	DLO-01	560079	7913132	Existing	profundal	✓	-
	DLO-01-9	560740	7913073	Existing	littoral	✓	✓
	DLO-01-2[b]	560350	7912927	Existing	profundal	✓	-
	DLO-01-3	560478	7912827	Existing	littoral	✓	✓
	DLO-01-10	560471	7912574	New	littoral	✓	✓
Sheardown Lake Southwest (SE)	DLO-02-1	560813	7912114	Existing	littoral	✓	✓
	DLO-02-11	561680	7911809	Existing	littoral	✓	✓
	DLO-02-9	561419	7911808	Existing	littoral	✓	✓
	DLO-02-13	561245	7911947	Existing	profundal	✓	✓
	DLO-02-3	561043	7911919	Existing	profundal	✓	✓
Mary Lake	BLO-01	554690	7913194	Existing	littoral	✓	✓
	BLO-20	554382	7906326	New	littoral	✓	✓
	BLO-11	554872	7905869	Existing	littoral	✓	✓
	BLO-12	554644	7905742	Existing	profundal	✓	-
	BLO-21	554966	7905443	New	littoral	✓	✓
	BLO-10	555033	7905065	Existing	profundal	✓	-
	BLO-08	555424	7904239	Existing	profundal	✓	-
	BLO-22	555607	7904040	New	littoral	✓	✓
	BLO-06[b]	555925	7903771	Existing	littoral	✓	✓

[a] Sediment core samples analyzed for particle size, TOC and total metals. Composite of three cores, using top 3 cm of sediment.

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

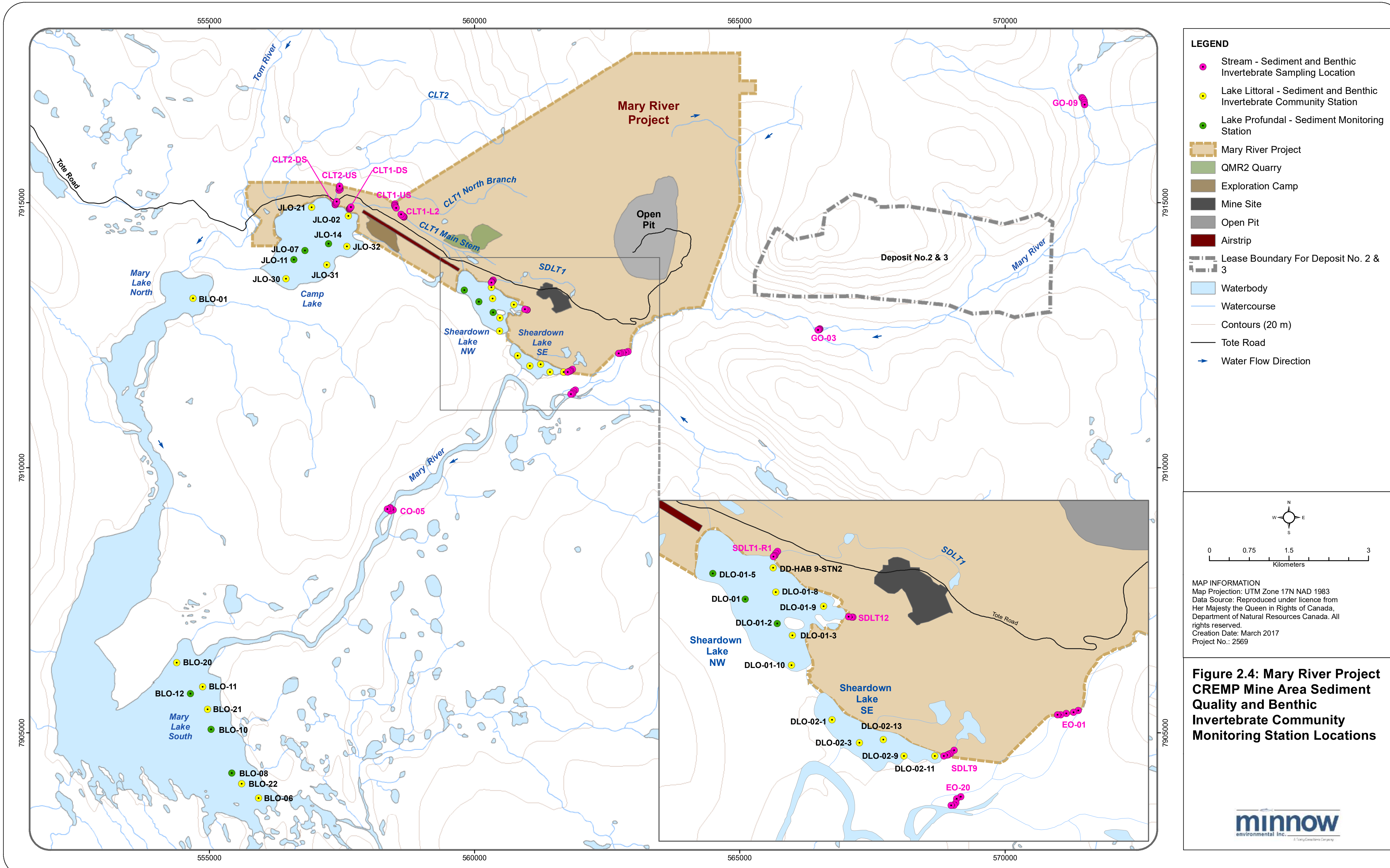


Figure 2.4: Mary River Project CREMP Mine Area Sediment Quality and Benthic Invertebrate Community Monitoring Station Locations

particle size analyses, and chemical characterization included analyses of total organic carbon (TOC) and total metals including mercury. Standard laboratory methods were used for all physical and chemical sediment analyses.

2.2.2 Data Analysis

Sediment quality data from the mine-exposed areas were compared to reference area data, to applicable sediment quality guidelines/AEMP benchmarks and, where applicable, to baseline sediment quality data. Sediment physical characteristics (i.e., moisture, particle size) and TOC were statistically summarized based on separate calculation of mean, standard deviation, standard error, minimum and maximum for littoral and profundal habitat at each study lake. These data were compared statistically between mine-exposed and reference study areas using the same tests, transformations (with the exception that logit transformations were conducted for dependent proportional data rather than log transformations), assumptions and software described previously for the statistical evaluation of *in-situ* water quality (see Section 2.1.1).

The sediment chemistry data from the mine-exposed lakes were initially assessed to identify potential gradients in sediment metal concentrations with distance from known or suspected sources of mine-related deposits to the lake. Sediment chemistry data were then averaged by study lake and compared between mine-exposed and reference areas. For each sediment chemistry parameter, the data from each study lake were separately averaged for littoral and profundal habitat and then compared between each respective mine-exposed and reference lake based on the magnitude of difference in parameter concentrations. The magnitude of difference between the mine-exposed and reference lakes was calculated and compared as described previously (Section 2.1.2; Table 2.3).

Sediment chemistry data were compared to applicable Canadian Sediment Quality Guidelines (CSQG; CCME 1999, 2015) probable effect levels (PEL) or, for parameters with no CSQG, to Ontario Provincial Sediment Quality Guidelines (PSQG; OMOE 1993) severe effect levels (SEL). The sediment quality guidelines used for the 2016 CREMP were abbreviated simply as 'SQG', although it is recognized that the values presented may represent either national PEL or Ontario provincial SEL guidelines. The 2016 sediment chemistry data analyses also included comparisons to Mary River Project AEMP sediment quality benchmarks that were derived using baseline sediment chemistry data for each mine-exposed lake and existing generic CSQG interim or PSQG lowest effect level sediment quality guidelines (Intrinsik 2014, 2015). As indicated previously, the AEMP benchmarks were developed to inform management decisions under the AEMP assessment approach and management response framework (Baffinland 2014). An elevation in parameter concentration above the AEMP benchmark may trigger various actions to better understand and potentially mitigate effects resulting from elevated concentrations of the parameter of concern (Baffinland 2014).

Sediment chemistry data for key parameters (i.e., parameters with concentrations that were notably higher at mine-exposed areas compared to the reference area, that have been identified as site-specific parameters of concern in previous studies, and/or those with concentrations above SQG and/or AEMP benchmarks) were plotted to evaluate potential changes in parameter concentrations among 2016 data, baseline (2005 – 2013) data, and previous 2015 mine operation period data. In addition, as described previously, the magnitude of difference was calculated for all parameters between 2016 and baseline data for each individual study lake using the same calculation (and categorization description) as described previously (Section 2.1.2; Table 2.3).

2.3 Biological Assessment

2.3.1 Phytoplankton

The Mary River Project CREMP uses measures of aqueous chlorophyll *a* concentrations to assess potential mine-related influences to phytoplankton. Because chlorophyll *a* is the primary pigment of phytoplankton (i.e., algae and other photosynthetic microbiota suspended in the water column), aqueous chlorophyll *a* concentrations are often used as a surrogate for evaluating the amount of photosynthetic microbiota in aquatic environments (Wetzel 2001). Chlorophyll *a* samples were collected at the same stations and same time as the collection of water chemistry samples by Baffinland environmental department staff (Table 2.1; Figures 2.2 and 2.3). Water samples for chlorophyll *a* analyses were collected using the same methods and equipment, and at the same locations, as described for water chemistry samples (Section 2.1.2). The chlorophyll *a* samples were collected into 1 L glass amber bottles and maintained in a cool and dark environment prior to submission to ALS (Mary River On-Site Laboratory, NU). On the same day of collection, the laboratory filtered the samples through a 0.45 micron cellulose acetate membrane filter assisted by vacuum pump. Following filtration, the membrane filter was wrapped in aluminum foil, inserted into a labelled envelope, and then frozen. At the completion of field collections for the seasonal sampling event, the filters were shipped frozen to the ALS Waterloo, ON laboratory for chlorophyll *a* analysis using standard methods. The field QA/QC applied during chlorophyll *a* sampling was similar to that described for water chemistry sampling (see Section 2.1.2).

The CREMP study design also stipulates the collection of phytoplankton community samples for archiving (NSC 2014, 2015a). In the event that water quality, chlorophyll *a* and/or other biological components indicate potential mine-related effects to primary productivity at any of the mine-exposed water bodies, these phytoplankton community samples may be processed to further investigate the nature of mine-related effects to phytoplankton biomass and community structure (i.e., taxonomic composition, richness, density). To date, none of the archived phytoplankton community samples have been processed (2006 – 2015). In 2016, phytoplankton community samples were collected using the same methods described in the CREMP (NSC 2014). As in the past, these samples were not processed, but were archived for potential future usage.

The analysis of aqueous chlorophyll *a* concentrations closely mirrored the approach used to evaluate the water quality data. Briefly, chlorophyll *a* concentrations were compared: i) between respective mine-exposed and reference areas; ii) spatially and seasonally at each mine-exposed waterbody; iii) to AEMP benchmarks; and, iv) to baseline data. Comparisons of chlorophyll *a* concentrations between the mine-exposed and reference areas were based on both qualitative and statistical approaches, the latter of which used the same parametric and/or non-parametric statistics, as appropriate, as described previously (Section 2.1). An AEMP benchmark chlorophyll *a* concentration of 3.7 µg/L was established for the Mary River Project (NSC 2014), and therefore the 2016 chlorophyll *a* concentration data were compared to this benchmark to assist with the determination of potential mine-related enrichment effects at water bodies influenced by mine operations. A mine-related effect on the productivity of a waterbody of interest was assessed as a chlorophyll a concentration above the AEMP benchmark, the representative reference area, and/or the respective waterbody baseline condition.

2.3.2 Benthic Invertebrate Community

The Mary River Project CREMP benthic invertebrate community (benthic) survey outlines a habitat-based approach for characterizing potential mine-related effects to benthic biota of lotic (river/stream) and lentic (lake) environments (NSC 2014). Lotic areas sampled for benthic invertebrates in 2016 included Camp Lake Tributaries 1 and 2 at historically established areas located upstream and downstream of the mine tote road, Sheardown Lake Tributaries 1, 9 and 12 near their respective outlets, and the Mary River upstream (two areas) and downstream (three areas) of the mine site (Table 2.5; Figure 2.4), all of which had been sampled as part of the 2015 CREMP. In addition to these mine-exposed areas, a benthic area was established at upper Camp Lake Tributary 1 in 2016 (CLT1-L2; Table 2.5) to evaluate potential effects of elevated concentrations of mine-related parameters of concern that were shown within this portion of the tributary in the previous 2015 study (Minnow 2016). As well, a reference creek benthic study area located within at the same unnamed tributary to Angajurjualuk Lake that is used for reference water quality sampling (Stations CLT-REF4 and MRY-REF2) was added to

Table 2.5: Mary River Project CREMP stream benthic invertebrate community monitoring station coordinates for the 2016 study.

Lake System	Waterbody	Station Code	Station Type	UTM Zone 17W, NAD83	
				Easting	Northing
Angajurjualuk Lake	Unnamed Tributary	REF-CRK-B1	Reference	570069	7906132
		REF-CRK-B2	Reference	570107	7906119
		REF-CRK-B3	Reference	570135	7906103
		REF-CRK-B4	Reference	570145	7906088
		REF-CRK-B5	Reference	570148	7906078
Camp Lake	Camp Lake Tributary 1	CLT1-US-B1	Reference	558500	7914976
		CLT1-US-B2	Reference	558492	7914947
		CLT1-US-B3	Reference	558497	7914935
		CLT1-US-B4	Reference	558508	7914918
		CLT1-US-B5	Reference	558518	7914901
		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	558612	7914777
		CLT1-DS-B1	Mine-Exposed	557643	7914882
		CLT1-DS-B2	Mine-Exposed	557646	7914890
		CLT1-DS-B3	Mine-Exposed	557653	7914896
		CLT1-DS-B4	Mine-Exposed	557656	7914907
		CLT1-DS-B5	Mine-Exposed	557670	7914913
	Camp Lake Tributary 2	CLT2-US-B1	Reference	557444	7915234
		CLT2-US-B2	Reference	557464	7915253
		CLT2-US-B3	Reference	557454	7915278
		CLT2-US-B4	Reference	557449	7915290
		CLT2-US-B5	Reference	557453	7915313
		CLT2-DS-B1	Mine-Exposed	557372	7914958
		CLT2-DS-B2	Mine-Exposed	557374	7914970
		CLT2-DS-B3	Mine-Exposed	557381	7914990
		CLT2-DS-B4	Mine-Exposed	557395	7914999
		CLT2-DS-B5	Mine-Exposed	557402	7915018
Sheardown Lake Northwest (NW)	Sheardown Lake Tributary 1 (Reach 1)	SDLT1-R1-B1	Mine-Exposed	560352	7913537
		SDLT1-R1-B2	Mine-Exposed	560337	7913518
		SDLT1-R1-B3	Mine-Exposed	560330	7913502
		SDLT1-R1-B4	Mine-Exposed	560322	7913497
		SDLT1-R1-B5	Mine-Exposed	560318	7913493
	Sheardown Lake Tributary 12	SDLT12-B1	Mine-Exposed	560990	7912979
		SDLT12-B2	Mine-Exposed	560980	7912981
		SDLT12-B3	Mine-Exposed	560951	7912986
Sheardown Lake Southwest (SE)	Sheardown Lake Tributary 9	SDLT9-DS-B1	Mine-Exposed	561842	7911855
		SDLT9-DS-B2	Mine-Exposed	561813	7911827
		SDLT9-DS-B3	Mine-Exposed	561798	7911824
		SDLT9-DS-B4	Mine-Exposed	561785	7911816
		SDLT9-DS-B5	Mine-Exposed	561756	7911809
Mary Lake	Mary River	GO-09-B1	Reference	571450	7916984
		GO-09-B2	Reference	571468	7916966
		GO-09-B3	Reference	571491	7916924
		GO-09-B4	Reference	571499	7916889
		GO-09-B5	Reference	571502	7916847
		GO-03-B1	Mine-Exposed	566506	7912613
		GO-03-B2	Mine-Exposed	566508	7912617
		GO-03-B3	Mine-Exposed	566501	7912610
		GO-03-B4	Mine-Exposed	566490	7912603
		GO-03-B5	Mine-Exposed	566477	7912600
		EO-01-B1	Mine-Exposed	562891	7912193
		EO-01-B2	Mine-Exposed	562851	7912177
		EO-01-B3	Mine-Exposed	562791	7912169
		EO-01-B4	Mine-Exposed	562743	7912156
		EO-01-B5	Mine-Exposed	562718	7912156
		EO-20-B1	Mine-Exposed	561900	7911465
		EO-20-B2	Mine-Exposed	561866	7911446
		EO-20-B3	Mine-Exposed	561857	7911413
		EO-20-B4	Mine-Exposed	561844	7911393
		EO-20-B5	Mine-Exposed	561819	7911391
		CO-05-B1	Mine-Exposed	558466	7909205
		CO-05-B2	Mine-Exposed	558412	7909185
		CO-05-B3	Mine-Exposed	558410	7909248
		CO-05-B4	Mine-Exposed	558397	7909234
		CO-05-B5	Mine-Exposed	558357	7909220

the CREMP in 2016 (Table 2.5; Figure 2.4). This reference creek is referred to as Unnamed Reference Creek herein for the purposes of the 2016 CREMP. Consistent with the federal Environmental Effects Monitoring (EEM) program (Environment Canada 2012), five stations were sampled at each lotic study area with the exception of Sheardown Lake Tributary 12, where only three stations were sampled due to limited habitat available for sampling using conventional gear suitable for erosional habitat. As in 2015, the level of replication used for lotic benthic sampling in 2016 was greater than specified under the original CREMP design in order to provide consistency with EEM standards (Minnow 2016a). To the extent possible, previously established lotic benthic stations were incorporated into the 2016 sampling program to provide comparability to historical baseline information.

The lake benthic study approach outlined in the original Mary River Project CREMP focussed on habitat-based characterization of the community at each mine-exposed lake (Baffinland 2014; NSC 2014, 2015a). In 2016, the lake benthic monitoring approach was modified to reflect an effects-based design consistent with that recommended for mines under the national EEM program (Environment Canada 2012). In addition, the 2016 study instituted harmonized sediment quality and benthic sampling at each lake benthic station to potentially improve the ability of the study to evaluate sediment physical feature and/or metal concentration influences on the benthic invertebrate community. Under the modified 2016 design, lake benthic sampling targeted littoral habitat (i.e., water depths ranging from approximately 7 m to 12 m) with substrate composed predominantly of fine sand- to silt-sized particles at each mine-exposed and reference study lake. Analysis of benthic data collected at Reference Lake 3 in 2015 indicated that, similar to temperate lakes (Ward 1992), depth-related influences on benthic invertebrate community structure (e.g., density and richness) occurs naturally in lakes of the Baffinland region (Minnow 2016a). Additional sampling conducted at Reference Lake 3 in 2016 confirmed the occurrence of natural depth-related influences on benthic invertebrate community structure in area lakes (Appendix B). Because the occurrence of naturally lower density and richness with greater depth (i.e., profundal habitat) potentially limits the ability of the AEMP study to identify mine-related effects at area lakes, littoral habitat was preferred for CREMP lake benthic sampling. Five littoral stations were sampled at each study lake which, to the extent possible, included previously established CREMP benthic stations to provide temporal continuity (Table 2.4; Figure 2.4).

2.3.2.1 Sample Collection and Laboratory Analysis

Two types of sampling equipment and methods were employed during the 2016 CREMP benthic survey to reflect different habitat types as follows:

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

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

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

Benthic samples were submitted to and processed by Zeas Inc. (Nobleton, ON) using standard sorting methods. Upon arrival at the laboratory, a biological stain was added to each benthic sample to facilitate greater sorting accuracy. The samples were washed free of formalin in a

500 µm sieve and the remaining sample material was then examined under a stereomicroscope at a magnification of at least ten times by a technician. All benthic invertebrates were removed from the sample debris and placed into vials containing 70% ethanol according to major taxonomic groups (i.e., order or family levels). A senior taxonomist later enumerated and identified the benthic organisms to the lowest practical level (typically genus or species) utilizing up-to-date taxonomic keys. Quality assurance/quality control (QA/QC) conducted during the laboratory processing of benthic samples included organism recovery and sub-sampling checks on as many as 10% of the total samples collected for the 2016 CREMP (Appendix A).

2.3.2.2 Data Analysis

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

Statistical comparisons of all applicable benthic invertebrate community indices and community composition endpoints were conducted using the same tests, transformations[1], assumptions and software described for the *in-situ* water quality comparisons (see Section 2.1.2). An effect on benthic invertebrate communities was defined as a statistically significant difference between any paired mine-exposed and reference areas at a p-value of 0.10. For each endpoint showing a significant difference, the magnitude of difference was calculated between study area means. Because the benthic survey was designed to have sufficient power to detect a difference (effect size) of ± two standard deviations (SD), the

[1] Rather than log-transformations like those conducted for non-normal *in-situ* water quality data, non-normal dependent proportional benthic data were subject to a modified probit transformation that better accounted for nil (or near-zero) values in the statistical analysis.

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 $\pm$ 2 SD_{REF} was used to define any ecologically relevant 'effects', which is analogous to differences beyond those expected to occur naturally between two areas that are uninfluenced by anthropogenic inputs (i.e., between pristine reference areas; see Munkittrick et al. 2009, Environment Canada 2012).

Temporal comparisons included statistical evaluations among the baseline, 2015 and 2016 data for primary benthic metrics (i.e., density, richness, Simpson's Evenness) and dominant invertebrate groups and FFG using uni-variate tests (e.g., ANOVA) and pair-wise *post-hoc* tests. The temporal statistical comparisons were conducted using the same tests, transformations, assumptions and software described above for the *in-situ* water quality comparisons (see Section 2.1.1). For study areas that contained data for multiple years (i.e., 3 or more), Tukey's HSD *post-hoc* tests were used in instances in which normal data showed equal variance, and Tamhane's *post-hoc* tests were used in instances in which normal data showed unequal variance. Similar to the 2016 within-year statistical analyses, the magnitude of difference was calculated for endpoints that differed significantly between years in the *post-hoc* tests and compared to the benthic survey CES_{BIC} of within two standard deviations of the baseline year mean (abbreviated as ±2 $SD_{BL\text{-}year}$).

2.3.3 Fish Population

The Mary River Project CREMP fish population survey outlines a non-lethal sampling design to evaluate potential mine-related effects to the fish population (e.g., age structure, growth, condition) at the mine-exposed lakes (NSC 2014, 2015a). The fish population survey targeted Arctic charr (*Salvelinus alpinus*) primarily because this species is the only abundant fish common to the mine's regional lakes, sufficient baseline catch and measurement data is available for this species to allow application of a before-after statistical evaluation, and because of this species importance as an Inuit subsistence food source. The approach employed for the CREMP fish population survey closely mirrored the recommended EEM approach for non-lethal sampling (Environment Canada 2012). Specifically, the 2016 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 2.1). Although the 2016 study also targeted Arctic charr from Reference Lake 3 as a basis for the evaluation of potential mine-related influences to the fish population, similar to the 2015 CREMP study, low numbers of Arctic charr were captured from the littoral/profundal zone of the reference lake in 2016. Thus, the 2016 fish population survey focussed on comparisons of fish collected at the nearshore of the mine-exposed and reference lakes, as well as on comparisons of fish captured at nearshore and littoral/profundal zones of individual mine-exposed lakes before-and-after the commencement of the Mary River Project ERP mine operations.

2.3.3.1 Sample Collection

Nearshore areas of the 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 one to three shoreline reaches of each study lake. The number of passes conducted at each study lake was dependent upon catch success, with more passes required in instances in which target numbers were not cumulatively attained. All fish captured during each pass were retained in buckets of aerated water. At the conclusion of each pass, total fishing effort (i.e., electrofishing seconds) was recorded to allow calculation of time-standardized catch. All captured fish were identified to species and enumerated, with any non-target species subsequently released alive at the area of capture. All captured Arctic charr were temporarily retained for processing using methods described below (Section 2.3.3.2). Additional supporting information collected for each electrofishing pass included recording the GPS coordinates at the points of commencement and completion of electrofishing activities, and a description of the sampled habitat.

Littoral/profundal areas of the study lakes were sampled for Arctic charr using experimental (gang index) gill nets. Multiple-panel, 2 m high gill nets with total lengths ranging from 61 – 91 m (200’ – 300’) and bar mesh sizes ranging from 38 – 76 mm (1.5” – 3”) were set on the bottom for short durations (approximately 0.6 – 5.7 hours per set; mean 2.5 hours) during daylight hours only. Upon retrieval of each net, all captured fish were identified to species, enumerated and processed (see below) separately for each individual gill net panel mesh size. For each gill net set, information including mesh size, duration of sampling, sampling depth range, GPS coordinates and habitat descriptions were recorded.

2.3.3.2 Field and Laboratory Processing

Following completion of each electrofishing pass and retrieval of each individual gill net panel, all captured Arctic charr were subject to processing in the field. For all live captures, the external condition of each individual was assessed visually for the presence of any deformities, erosions, lesions and tumors (DELT) or evidence of external and/or internal parasites. All observations were recorded on field sheets, with supporting photographs taken as appropriate.

Each fish was then subject to measurement of fork and total length to the nearest millimetre using a standard measuring board. Following length measurements, fish captured using the electrofishing unit were individually weighed to the nearest milligram using an Ohaus Model 123 Scout-Pro analytical balance (Ohaus Corp., Pine Brook, NJ) with a surrounding draft shield. For Arctic charr captured in gill nets, individuals were weighed using Pesola™ spring scales (Pesola AG, Baar Switzerland) demarcated at intervals of 1-2% of the total scale range and with precision of ± 0.3%. The Pesola™ spring scale for individual weight measurement of gill-net captured fish was selected so that the fish weight was near the top of the scale's range to ensure that measurements achieved a resolution near 1%. All live Arctic charr captured by electrofishing and gill netting methods that were not selected for the collection of aging structures were released near the location of capture following these individual measurements of length and weight.

As specified for EEM non-lethal fish population surveys (see Environment Canada 2012), approximately 10% of the targeted number of Arctic charr captured using electrofishing methods were sacrificed for collection of aging structures. Arctic charr mortalities from experimental gill netting were approximately 20% of targeted catch numbers, and therefore aging structures were removed from each incidental mortality. Otoliths and pectoral fin rays were removed from all sacrificed individuals and incidental mortalities. Upon removal, these aging structures were wrapped separately in wax paper, placed inside envelopes labelled with the fish identification, and then dried for storage. For all incidental mortalities, in addition to removal of aging structures, fish were dissected to determine sex and for removal of the liver and whole gonads for weight measurement. These organs were weighed to the nearest milligram using an Ohaus Model 123 Scout-Pro balance outfitted with a surrounding draft shield. During processing, fish were also inspected for any internal abnormalities (e.g., parasites, lesions, tumours, etc.) with descriptions recorded accordingly.

Age structures (otoliths and pectoral fin rays) 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. Pectoral fin rays were cleaned, embedded in epoxy resin and, after the epoxy hardened, sectioned transversely using a Buehler Isomet (Lake Bluff, IL) low-speed diamond saw. The prepared otolith and pectoral fin ray samples were later mounted on a glass slide using a mounting medium and examined under a compound microscope using transmitted light to determine fish age. For each structure, the age and edge condition was recorded along with a confidence rating for the age determination.

2.3.3.3 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, and gill netting CPUE was calculated as the number of fish captured per 100 meter·hours of net used for each study lake. Temporal comparison of fish community assemblage was conducted using electrofishing CPUE and gill netting CPUE to evaluate relative changes in fish catches at mine area lakes between mine baseline and the 2016 year of mine operation.

Arctic charr population health was assessed separately for electrofishing and experimental gill netting data sets. Initial data analysis for the non-lethal survey included the plotting of length frequency distributions as described by Bonar (2002) and Gray et al. (2002), so that, together with appropriate aging data, YOY individuals could be distinguished from the juvenile/adult life stages (electrofishing data set), or various size/age classes could be distinguished from one another (gill netting data set). Where relevant, the YOY age class was assessed separately from the juvenile/adult age classes for fish survey endpoints between the individual mine-exposed lakes and the reference lake. Fish size endpoints of fork length and fresh body weight were summarized by separately reporting mean, median, minimum, maximum, standard deviation, standard error and sample size by size class (if possible) for each study area. The recorded measurement endpoints were used as the basis for evaluating four response categories (survival, growth, reproduction and energy storage; Table 2.6) according to the procedures outlined by Environment Canada (2012) for environmental effects monitoring. Length-frequency distribution was compared between mine-exposed and reference areas, for data collected in 2016, and for before-after analysis using data collected in 2016 and during the combined baseline period, using a non-parametric two-sample Kolmogorov-Smirnov (KS) test. Mean fork length and body weight were compared between mine-exposed and reference study areas in 2016, and between 2016 and the mine baseline period, using ANOVA, with data inspected for normality and homogeneity of variance before applying parametric statistical procedures. In cases where data did not meet the assumptions of ANOVA despite log-transformation, a non-parametric Mann-Whitney U-test was also performed to test for/validate significant differences between study areas or study periods, as appropriate, indicated by the ANOVA test.

Body weight at fork length (condition) was compared using Analysis-of-Covariance (ANCOVA). Prior to conducting the ANCOVA tests, scatter plots of all variable and covariate combinations were examined to identify outliers, leverage values or other unusual data. The scatter plots were also examined to ensure there was adequate overlap between the 2016 mine-exposed

Table 2.6: Fish population survey endpoints examined for the Mary River Project CREMP, August 2016.

Response Category	Endpoint	Statistical Procedure[c,d,e]	Critical Effect Size
Survival	Length-frequency distribution[a]	K-S Test	not applicable
	Age[a,f]	ANOVA	not applicable
Energy Use (size)	Size (fresh body weight)[b]	ANOVA	25%
	Size (fork length)[b]	ANOVA	25%
Energy Use (growth)	Size-at-age (body weight against age)[a,f]	ANCOVA	25%
	Size-at-age (fork length against age)[b,f]	ANCOVA	25%
Energy Use (reproduction)	Relative abundance of YOY (% composition)[b]	None	not applicable
Energy Storage	Condition (body weight against length)[a]	ANCOVA	10%

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

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

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

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

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

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

and reference/mine-exposed baseline data sets, and that there was a linear relationship between the variable and the covariate. In order to verify the existence of a linear relationship, each relationship was tested using linear regression analysis by area and evaluated at an alpha level of 0.05. If it was determined that there was no significant linear regression relationship between the variable and covariate for the 2016 mine-exposed and/or reference/ mine-exposed baseline data sets, then the ANCOVA was not performed. Once it was determined that ANCOVA could be used for statistical analysis of the data, the first step in the ANCOVA analysis was to test whether the slopes of the regression lines for the 2016 mine-exposed and reference/baseline data sets were equal. This was accomplished by including an interaction term (dependent × covariate) in the ANCOVA model and evaluating if the interaction term was significantly different, in which case the regression slopes would not be equal between data sets and the resulting ANCOVA would provide spurious results. In such cases, two methodologies were employed to assess whether a full ANCOVA could proceed. In order of preference these were: 1) removal of influential points using Cook's distance and re-assessment of equality of slopes; and, 2) Coefficients of Determination that considered slopes equal regardless of an interaction effect (Environment Canada 2012). For the Coefficients of Determination, the full ANCOVA was completed to test for main effects, and if the r^2 value of both the parallel regression model (interaction term) and full regression model were greater than 0.8 and within 0.02 units in value, the full ANCOVA model was considered valid (Environment Canada 2012). If both methods proved unacceptable, the magnitude of effect was estimated at both the minimum and maximum overlap of covariate variables between areas (Environment Canada 2012). This results in a statistically significant interaction effect (slopes are not equal), but the calculation of the magnitude of difference at the minimum and maximum values of covariate overlap is not assigned statistical difference as it would for a full ANCOVA model. If the interaction term was not significant (i.e., homogeneous slopes between the two populations), then the full ANCOVA model was run without the interaction term to test for differences in adjusted means between the two data sets. The adjusted mean was then used as an estimate of the population mean based on the value of the covariate in the ANCOVA model.

For endpoints showing significant data set differences, the magnitude of difference between 2016 mine-exposed and reference data or the baseline data was calculated as described by Environment Canada (2012) using mean (ANOVA), adjusted mean (ANCOVA with no significant interaction) or predicted values (ANCOVA with significant interaction). The anti-log of the mean, adjusted mean, or predicted value was used in the equations for endpoints that were $\log_{10}$-transformed. In addition, the magnitude of difference for ANCOVA with a significant interaction was calculated for each of the minimum and maximum values of the covariate.

If there was no significant difference indicated between data sets, the minimum detectable effect size was calculated as a percent difference from the reference mean/mine-exposed baseline mean for ANOVA or adjusted reference mean/mine-exposed baseline mean for ANCOVA at alpha = beta = 0.10 using the square root of the mean square error (generated during either the ANOVA or ANCOVA procedures) as a measure of variability in the sample population based on formula provided by Environment Canada (2012). Finally, if outliers or leverage values were observed in a data set (or sets) upon examination of scatter plots and residuals, then the values were removed and ANOVA or ANCOVA tests were repeated and presented only for the reduced data sets.

3.0 CAMP LAKE SYSTEM

3.1 Camp Lake Tributaries (CLT)

3.1.1 Water Quality

3.1.1.1 Camp Lake Tributary 1

Camp Lake Tributary 1 (CLT1) dissolved oxygen (DO) concentrations were consistently at or above saturation at all north branch and main stem stations during all spring, summer and fall monitoring events (Appendix Tables C.1 – C.3). Dissolved oxygen concentrations and percent saturation at the CLT1 north branch and upper and lower main stem stations (downstream of QMR2 Quarry and mine-tote road, respectively) differed significantly among each other and compared to the reference creek at the time of biological sampling in August 2016 (Figure 3.1; Appendix Table C.13). However, DO saturation was well above the WQG minimum limit for cold-water biota (i.e., 54%) at all stations (Figure 3.1), suggesting that these differences were not likely to be ecologically meaningful, and that mine activity had not adversely affected DO concentrations at CLT1. No consistent spatial patterns in *in-situ* pH were shown with distance from the mine during all spring, summer and fall monitoring events within the CLT1 system (Appendix Tables C.1 – C.3). Although pH was significantly higher at all CLT1 stations compared to Unnamed Reference Creek, no significant differences in pH were indicated among the north branch and main stem study areas in August 2016 (Figure 3.1). In addition, pH at CLT1 was similar to other lotic reference stations and was consistently within WQG limits, suggesting that pH differences at CLT1 compared to Unnamed Reference Creek reflected natural variation in pH among regional creeks, and that mine activity had not adversely affected pH within the CLT1 system.

Water chemistry of the CLT1 north branch was similar to the reference creek stations with the exception of a slightly higher (i.e., 3- to 5-fold) nitrate concentration during the summer sampling event in 2016 (Table 3.1; Appendix Table C.14). *In-situ* specific conductance was significantly higher at the CLT1 stations compared to Unnamed Reference Creek, and differed significantly among the north branch and upper and lower main stem study areas during the August 2016 sampling event (Figure 3.1) suggesting a mine-related influence on water quality of the CLT1 system. In addition to conductivity and nitrate concentrations, hardness, alkalinity and concentrations of total dissolved solids (TDS), ammonia, total Kjeldahl nitrogen (TKN), organic carbon, chloride, sulphate and several metals, including cobalt, iron, manganese, molybdenum, potassium, sodium, strontium and uranium, were slightly to highly elevated (i.e., 3-fold to ≥10-fold higher, respectively) at the upstream-most CLT1 main stem station (L2-03)

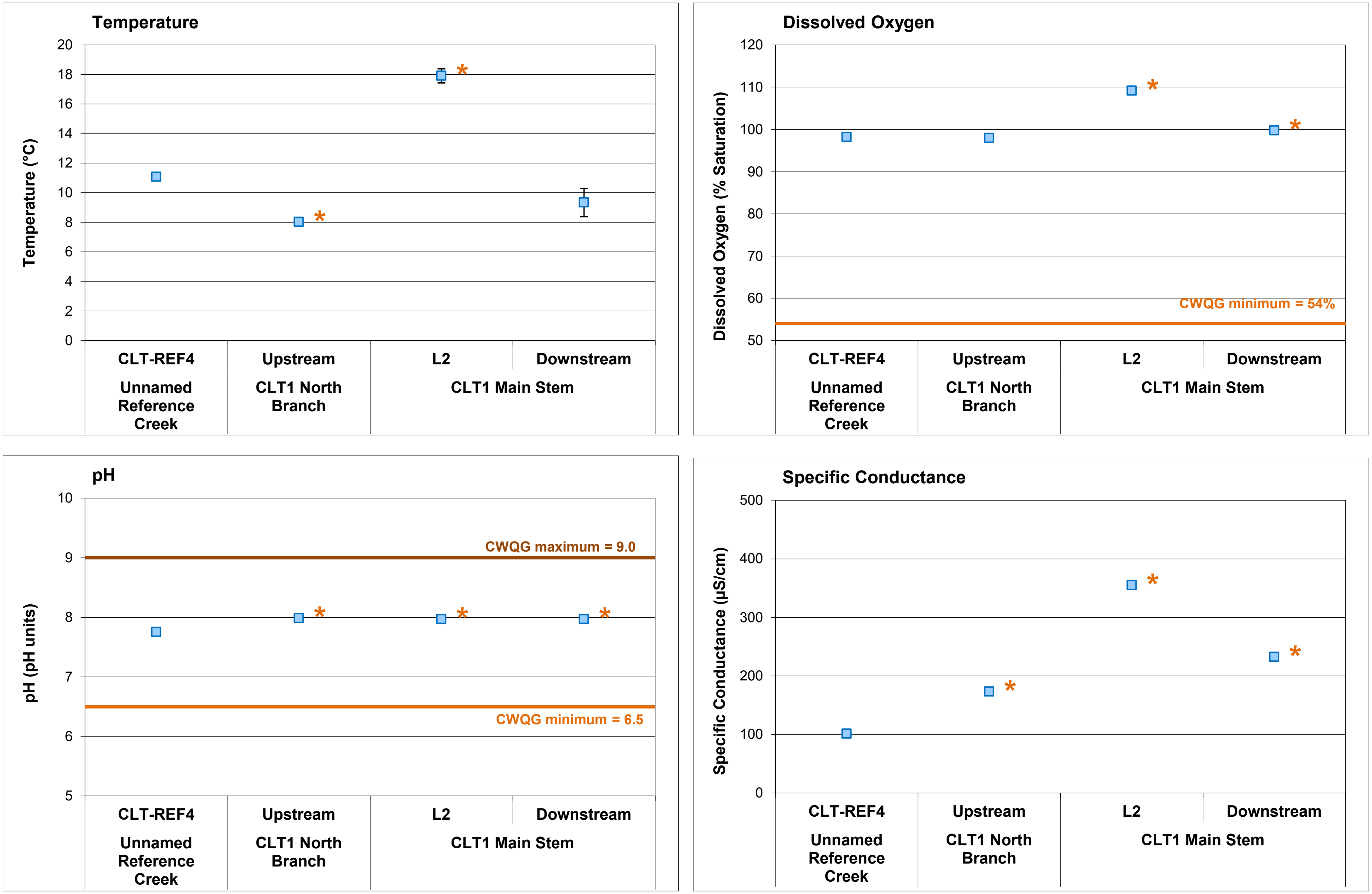


Figure 3.1: Comparison of *in-situ* water quality variables (mean ± SD; n = 5) measured at Camp Lake Tributary 1 benthic invertebrate community stations, Mary River Project CREMP, August 2016. An asterisk (*) next to data point indicates mean value differs significantly from the Unnamed Reference Creek mean.

Table 3.1: Water chemistry at Camp Lake Tributary (CLT) monitoring stations during fall (August) sampling, Mary River Project CREMP, 2016.

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creek Average (n=4)	North Branch CLT1		Main Stem CLT1				CLT-2
						L1-08	L1-02	L2-03	L1-09	L1-05	L0-01	K0-01
					Fall 2016	20-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	125	147	209	431	293	298	296	255
	pH (lab)	pH	6.5 - 9.0	-	7.99	7.97	8.21	7.99	8.16	8.11	8.17	8.27
	Hardness (as $CaCO_3$)	mg/L	-	-	57.75	72	105	176	136	138	138	130
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	65	77	94	230	156	159	143	123
	Turbidity	NTU	-	-	1.10	0.34	0.26	2.88	0.97	0.93	0.95	0.29
	Alkalinity (as $CaCO_3$)	mg/L	-	-	57	72	104	140	119	116	116	125
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	<0.020	<0.020	<0.020	0.237	0.048	0.047	0.042	0.031
	Nitrate	mg/L	13	13	0.021	0.079	<0.020	1.67	0.353	0.411	0.380	0.048
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	0.0203	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	0.56	0.20	0.24	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.3	1.8	2.4	4.6	3.0	3.0	2.9	3.0
	Total Organic Carbon	mg/L	-	-	1.5	1.9	2.6	4.6	3.2	3.4	3.1	3.2
	Total Phosphorus	mg/L	0.020[α]	-	0.0059	0.0087	<0.0030	0.0096	0.0033	0.0059	0.0031	0.0108
	Phenols	mg/L	0.004[α]	-	0.0055	0.0070	0.0067	0.0076	0.0041	0.0038	0.0025	0.0067
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	2.4975	1.91	2.13	36.7	18.4	18.9	17.2	5.11
	Sulphate (SO_4)	mg/L	218[β]	218	4.39	2.98	4.83	18.4	7.84	8.25	7.70	5.29
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0578	0.0137	0.0071	0.031	0.0098	0.0110	0.0154	0.0080
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	0.00014	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00779	0.0109	0.0128	0.0168	0.0163	0.0155	0.0157	0.0142
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00040	<0.00050	<0.00050	<0.00010	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.0003875	<0.00050	<0.00050	<0.000050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	0.020	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	12.3	14.2	20.3	34.3	28.3	27.9	28.8	25.2
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	0.00034	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.0010	***0.00228***	***0.00226***	0.0013	0.00194	0.00191	0.00183	0.00156
	Iron (Fe)	mg/L	0.30	0.326	0.051	<0.030	<0.030	***0.459***	0.120	0.112	0.094	<0.030
	Lead (Pb)	mg/L	0.001	0.001	0.000096	<0.000050	<0.000050	<0.00010	<0.000050	<0.000050	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	<0.0010	0.0013	0.0031	0.0037	0.0036	0.0034	0.0016
	Magnesium (Mg)	mg/L	-	-	6.77	8.69	12.9	21.0	15.7	15.9	15.8	15.7
	Manganese (Mn)	mg/L	0.935[β]	-	0.00086	0.000651	0.000694	0.0511	0.0108	0.00822	0.00535	0.00104
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.000380	0.000851	0.000647	0.00353	0.00120	0.00115	0.000988	0.000436
	Nickel (Ni)	mg/L	0.025	0.025	0.00056	<0.00050	0.00071	0.00146	0.00103	0.00102	0.00101	0.00066
	Potassium (K)	mg/L	-	-	0.84	2.15	2.05	3.30	2.41	2.35	2.28	1.79
	Selenium (Se)	mg/L	0.001	-	<0.0007625	<0.0010	<0.0010	0.000118	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.95	0.83	1.10	1.22	1.21	1.26	1.30	1.05
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000020	<0.000010	<0.000010	<0.000050	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	1.830	0.584	1.55	16.3	5.32	5.37	5.10	2.80
	Strontium (Sr)	mg/L	-	-	0.01240	0.00826	0.0106	0.0415	0.0487	0.0460	0.0401	0.0151
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0000775	<0.00010	<0.00010	0.000010	<0.00010	<0.00010	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	0.00799	<0.010	<0.010	0.00115	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00366	0.00399	0.00277	0.0172	0.00580	0.00571	0.00501	0.00236
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.000875	<0.0010	<0.0010	<0.00050	<0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	0.0082

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). See Table 2.2 for information regarding WQG criteria.

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

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

compared to average reference creek station water chemistry at the time of the August 2016 sampling event (Table 3.1; Appendix Tables C.14 and C.16). However, on average, only concentrations of nitrate, chloride, manganese and strontium were elevated at the CLT1 lower main stem (i.e., stations L1-09, L1-05 and L0-01) compared to respective reference creek station average concentrations during the fall sampling event (Appendix Table C.14), reflecting natural dilution of the main stem from the north branch. Similar to the 2015 data, the spatial patterns in the 2016 water quality data suggested a mine-related influence within the CLT1 main stem, whereas at the north branch, only a slight mine-related influence on water quality was evident. Despite evidence of continued mine-related influence on water quality of the CLT1 system, concentrations of all parameters were below applicable WQG and watercourse-specific AEMP benchmarks at CLT1 with the exception of copper concentrations at the north branch, and iron and uranium concentrations at upstream-most Station L2-03 of the main stem[2] (Table 3.1).

Temporal comparisons of the CLT1 north branch water chemistry data indicated that parameter concentrations in fall 2016 were generally within the range of those measured during the mine baseline (2005 – 2013) period with the exception of higher copper concentrations in both 2015 and 2016 (Figure 3.2; Appendix Figure C.2). Temporal comparisons of CLT1 main stem water chemistry data indicated that, of the parameters shown to have elevated concentrations relative to the reference creek stations, hardness and concentrations of TDS, chloride and strontium in 2016 were comparable to or only slightly higher than concentrations during the mine baseline period (Figure 3.2; Appendix Figure C.2). However, conductivity, nitrate, sulphate, iron, manganese, molybdenum, sodium and uranium showed progressively higher concentrations from mine baseline, to construction, to 2015 and/or 2016 mine operational years at all four CLT1 main stem stations (Figure 3.2; Appendix Figure C.2). Higher concentrations of these parameters at the main stem CLT1 stations over time likely reflected greater blasting/excavating activity (including associated dust generation) at mine quarry QMR2, and potentially greater fugitive dust generation from increased truck usage on the mine tote road during mine activities from 2014 - 2016 compared to the baseline period. The QMR2 quarry is used to provide material for mine infrastructure projects (e.g., road construction).

[2] Although phenol concentrations were above WQG at the CLT1 tributaries, all mine lakes (including Camp, Sheardown NW, Sheardown SE and Mary) and Mary River, phenol concentrations were also above WQG at the reference creek stations, Mary River reference stations (i.e., GO-09 series stations) and Reference Lake 3, indicating natural elevation of phenol concentrations in regional water bodies unrelated to mine operations (see Appendix B for additional discussion). Because elevated aqueous phenol concentrations appeared to be a natural phenomenon, no discussion of phenol concentrations was included in comparisons to WQG for the mine-exposed waterbodies in the 2016 CREMP.

Figure 3.2: Temporal comparison of water chemistry at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during fall. Values represent mean ± SD. Reference creek stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4). Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to the Camp Lake Tributaries.

3.1.1.2 Camp Lake Tributary 2 (CLT2)

Camp Lake Tributary 2 (CLT2) dissolved oxygen saturation levels were consistently high at Station KO-01 in 2016, and were similar to mean DO saturation observed among the reference creek stations (Appendix Tables C.1 – C.3). However, *in-situ* DO concentrations/saturation and pH at CLT2 differed significantly upstream and downstream of the mine tote road, and compared to Unnamed Reference Creek, at the time of biological sampling in August 2016 (Figure 3.3; Appendix Tables C.17). Despite these differences, DO saturation was well above the WQG minimum limit for cold-water biota (i.e., 54%) and pH was consistently within WQG limits at all CLT2 stations during all 2016 sampling events (Figure 3.3; Appendix Tables C.1 to C.3). Therefore, the differences in DO concentrations/saturation and pH between areas within the CLT2 system and at CLT2 compared to Unnamed Reference Creek were not likely to be ecologically meaningful, nor indicate an adverse mine-related influence.

Water chemistry at CLT2 (Station KO-01) was similar to the reference creek stations with the exceptions of slightly higher (i.e., 3- to 5-fold) sulphate and zinc concentrations during the spring and/or summer sampling events in 2016 (Table 3.1; Appendix Table C.14). *In-situ* specific conductance was significantly higher at CLT2 compared to the reference creek, but did not differ significantly upstream and downstream of the mine tote road during the August 2016 sampling event (Figure 3.3). However, aqueous concentrations of all parameters were consistently well below established WQG and AEMP benchmarks at the CLT2 monitoring station in 2016[3] (Table 3.1; Appendix Table C.14). Temporal comparisons of CLT2 water chemistry data indicated that parameter concentrations in fall 2016 were generally within the range of those measured during the mine baseline (2005 – 2013) period and not unlike those observed during the 2014 mine construction and 2015 mine operation periods (Figure 3.2; Appendix Figure C.2). Collectively, the 2016 water chemistry data suggested only minor mine influence on aqueous conductivity, sulphate and/or zinc concentrations within the CLT2 system in 2016.

3.1.2 Phytoplankton

3.1.2.1 Camp Lake Tributary 1 (CLT1)

Camp Lake Tributary 1 (CLT1) north branch chlorophyll a concentrations were lower than the average concentration among reference creek stations for spring, summer and fall seasons in 2016, but were within the overall range of reference creek chlorophyll a concentrations suggesting no marked differences in phytoplankton productivity between the CLT1 north

[3] Refer to Footnote 2 (page 23) and Appendix B regarding phenol concentrations above WQG at the mine-exposed and reference areas of the Mary River Project LSA waterbodies.

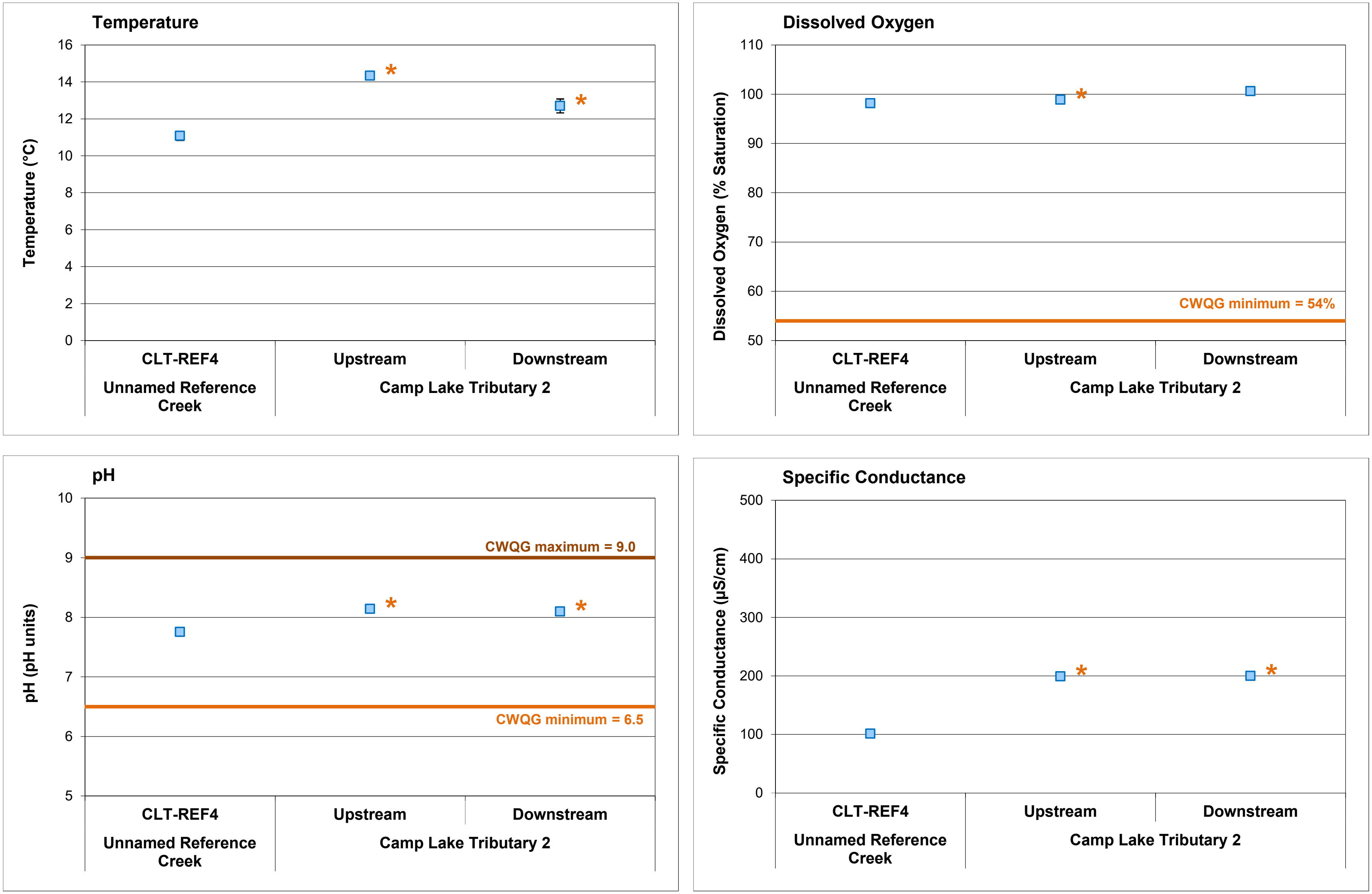


Figure 3.3: Comparison of *in-situ* water quality variables (mean ± SD; n = 5) measured at Camp Lake Tributary 2 benthic invertebrate community stations, Mary River Project CREMP, August 2016. An asterisk (*) next to data point indicates mean value differs significantly from the Unnamed Reference Creek mean.

branch and the reference creek stations (Figure 3.4). Within the CLT1 main stem, chlorophyll a concentrations were consistently highest at upstream-most Station L2-03, with concentrations at this station also consistently greater than at the reference creek stations in 2016. Downstream of the north branch confluence, beginning at Station L1-09, chlorophyll a concentrations were comparable to, or slightly greater than, those at the reference creek stations (Figure 3.4). 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 2016 (Figure 3.4). Similar to the reference creek stations, chlorophyll a concentrations observed at all CLT1 stations in 2016 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 for CLT1 based on mean aqueous total phosphorus concentrations less than 10 µg/L during all spring, summer and fall sampling events (Table 3.1; Appendix Table C.14).

Temporal comparisons of the CLT1 chlorophyll a data indicated that concentrations at the north branch in 2015 and 2016 mine operation years were similar to, or lower than, those observed during the baseline (2005 – 2013) period (Figure 3.5). However, at the CLT1 main stem, chlorophyll a concentrations were generally higher in 2015/2016 than during the mine baseline period with the exception of at the CLT1 mouth (Station L0-01; Figure 3.5). The spatial and temporal analyses of chlorophyll a concentrations at CLT1 suggested that mine operation may have contributed to slightly higher phytoplankton productivity within the upper main stem (i.e., Station L2-03), but not at the north branch or at the lower main stem stations. As described in the 2015 CREMP, higher phytoplankton productivity within the CLT1 upper main stem was consistent with the occurrence of elevated aqueous nutrient (e.g., ammonia, nitrate) concentrations in the 2015/2016 (see Section 3.1.1). This suggested that slightly greater phytoplankton productivity at Station L2-03 in 2016 was the result of current mine operations and specifically, the introduction of nutrients to the CLT1 system as a result of active quarrying at the QMR2 pit.

3.1.2.2 Camp Lake Tributary 2 (CLT2)

Camp Lake Tributary 2 (CLT2; Station KO-01) chlorophyll a concentrations were consistently low, but within the range observed among the reference creek stations during individual spring, summer and fall seasonal sampling events in 2016 (Figure 3.4). The CLT2 chlorophyll a concentrations also met the AEMP benchmark of less than 3.7 µg/L for all 2016 sampling events. Low phytoplankton productivity, indicative of oligotrophic conditions, was suggested at CLT2 based on comparison of chlorophyll a concentrations to Dodds et al (1998) trophic

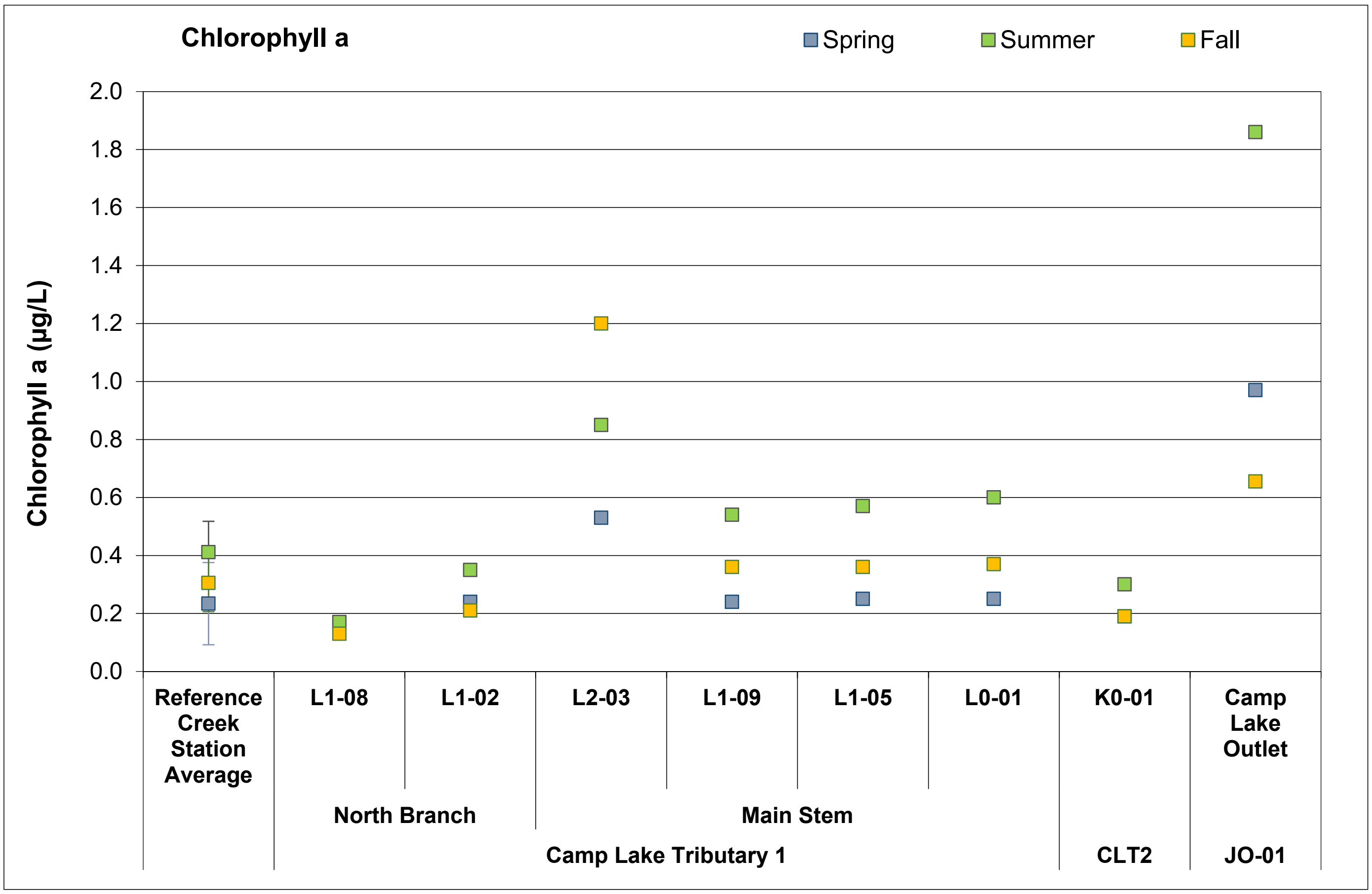


Figure 3.4: Chlorophyll a concentrations at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) phytoplankton monitoring stations, Mary River Project CREMP, 2016. Reference creek stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4).

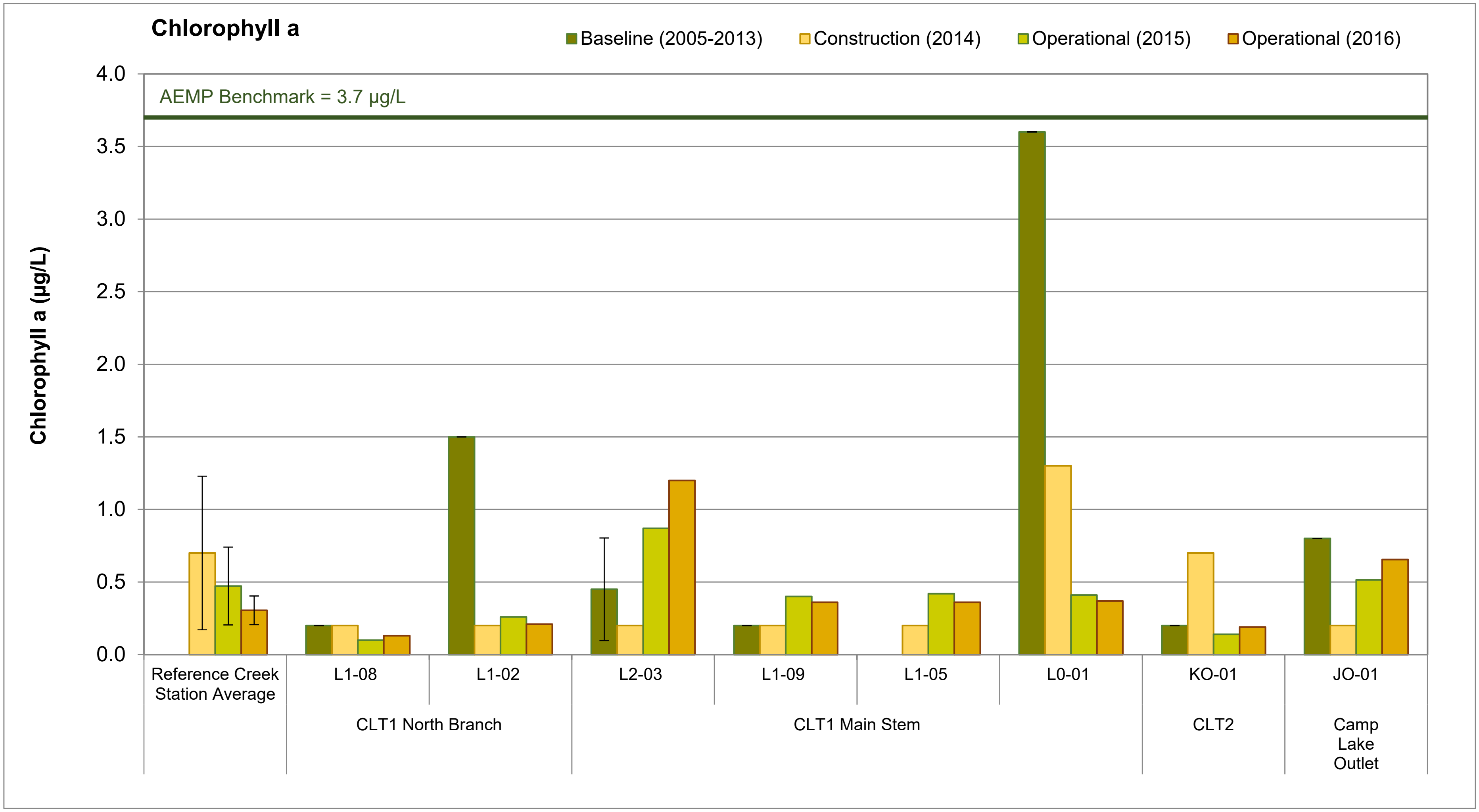


Figure 3.5: Temporal comparison of chlorophyll a concentrations at Camp Lake Tributary 1 (CLT-1) and Tributary 2 (CLT-2) for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during fall, Mary River Project CREMP. The reference creek stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4).

status classification for creek environments. This productivity classification was supported by a WQG categorization of ultra-oligotrophic to oligotrophic based on mean aqueous phosphorus concentrations below 10 µg/L at CLT2 during all spring, summer and fall sampling events (Table 3.1; Appendix Table C.14). Temporal comparisons of the CLT2 chlorophyll a data indicated that the 2015 and 2016 chlorophyll a concentrations were similar to those during the mine baseline period (Figure 3.5). Overall, no mine-related influences to phytoplankton density at CLT2 were suggested by the 2016 chlorophyll a concentration data.

3.1.3 Benthic Invertebrate Community

3.1.3.1 Camp Lake Tributary 1 (CLT1)

North Branch (CLT1 US)

Benthic invertebrate density and Simpson's Evenness did not differ significantly between the CLT1 north branch and Unnamed Reference Creek (Table 3.2). However, in addition to significantly lower richness at the CLT1 north branch compared to Unnamed Reference Creek, differences in community assemblage were suggested between watercourses based on significant differences in Bray-Curtis Index (Table 3.2). Notably, the relative abundance of metal-sensitive chironomids did not differ significantly between the CLT1 north branch and Unnamed Reference Creek, suggesting that the community composition differences between watercourses was unrelated to metal concentrations. Rather, a significantly higher proportion of the shredder functional feeding group (FFG) at the CLT1 north branch suggested the presence of greater amounts of living and/or decomposing large leafy/woody vegetation compared to Unnamed Reference Creek, which was consistent with field observations of bryophyte abundance between watercourses in 2016 (Appendix Tables F.1 and F.7). Temporal comparisons of the CLT1 north branch benthic invertebrate community data indicated that density, richness, Simpson's Evenness and relative abundance of key dominant groups and FFG in 2016 did not show any consistent type and/or direction of significant differences compared to baseline data collected in 2007 and 2011 (Figure 3.6; Appendix Table F.8). Overall, no adverse mine-related influences on benthic invertebrate community features were indicated at the CLT1 north branch in 2016 based on comparisons to 2016 reference creek data and to historical 2007 and 2011 baseline data.

Upper Main Stem (CLT1 L2)

The benthic invertebrate community of upper main stem of Camp Lake Tributary (CLT1 L2), which is located near the QMR2 mine quarry, showed significantly higher benthic invertebrate density and significant differences in community composition (as indicated by Bray-Curtis

Table 3.2: Benthic invertebrate community statistical comparison results among Camp Lake Tributary 1 and Unnamed Reference Creek study areas, Mary River Project CREMP, August 2016.

Metric	Overall four-group ANOVA[a]			ANOVA Comparison to Reference				
	Significant Difference Among Areas?	p-value	Statistical Test	CLT1 Study Area	Significantly Different from Reference?	p-value	Magnitude of Difference (no. of SD)[b]	*Post-hoc* Statistical Test
Density (No. organisms/ m^2)	YES	0.0000	α , δ	Upstream (North Branch)	NO	1.0000	-	Tamhane's
				L2 (Upper Main Stem)	YES	0.0025	9.8	
				Downstream (Lower Main Stem)	NO	0.7027	-	
Richness (Number of Taxa)	YES	0.0005	α , δ	Upstream (North Branch)	YES	0.0045	-5.1	Tukey's HSD
				L2 (Upper Main Stem)	NO	0.6133	-	
				Downstream (Lower Main Stem)	NO	0.5090	-	
Simpson's Evenness	NO	0.6326	α , δ	Upstream (North Branch)	NO	0.8334	-	Tukey's HSD
				L2 (Upper Main Stem)	NO	0.9819	-	
				Downstream (Lower Main Stem)	NO	0.9962	-	
Bray-Curtis Index	YES	0.0000	α , δ	Upstream (North Branch)	YES	0.0000	2.6	Tukey's HSD
				L2 (Upper Main Stem)	YES	0.0000	4.6	
				Downstream (Lower Main Stem)	YES	0.0000	3.6	
Oligochaeta (% of Community)	YES	0.0001	β , δ	Upstream (North Branch)	NO	0.1554	-	Tamhane's
				L2 (Upper Main Stem)	NO	0.1762	-	
				Downstream (Lower Main Stem)	YES	0.0099	14.0	
Hydracarina (% of Community)	YES	0.0000	β , δ	Upstream (North Branch)	NO	0.7896	-	Tukey's HSD
				L2 (Upper Main Stem)	YES	0.0114	3.2	
				Downstream (Lower Main Stem)	***YES***	***0.0027***	***-1.9***	
Chironomidae (% of Community)	NO	0.3439	β , δ	Upstream (North Branch)	NO	0.9884	-	Tukey's HSD
				L2 (Upper Main Stem)	NO	0.5414	-	
				Downstream (Lower Main Stem)	NO	0.9665	-	
Metal-Sensitive Chironomidae (%)	YES	0.0011	β , δ	Upstream (North Branch)	NO	0.9572	-	Tukey's HSD
				L2 (Upper Main Stem)	YES	0.0322	3.6	
				Downstream (Lower Main Stem)	NO	0.2631	-	
Tipulidae (% of Community)	YES	0.0002	β , δ	Upstream (North Branch)	NO	0.2555	-	Tukey's HSD
				L2 (Upper Main Stem)	***YES***	***0.0053***	***-1.5***	
				Downstream (Lower Main Stem)	NO	0.9621	-	

[a] Data analysis included: α - data untransformed; β - data logit transformed; ε - data log transformed; δ - single factor ANOVA test; γ - ANOVA test validated using Kruskal-Wallis H- or Mann Whitney U-test.

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

Highlighted values indicate significant difference between study areas based on ANOVA p-value less than 0.10 that were also outside of a CES of ±2 SD, suggesting an ecologically meaningful difference.

BOLD Bold text values indicate significant differences between study areas based on ANOVA p-value less than 0.10, but a Critical Effect Size within ±2 SD, suggesting the difference is not ecologically meaningful.

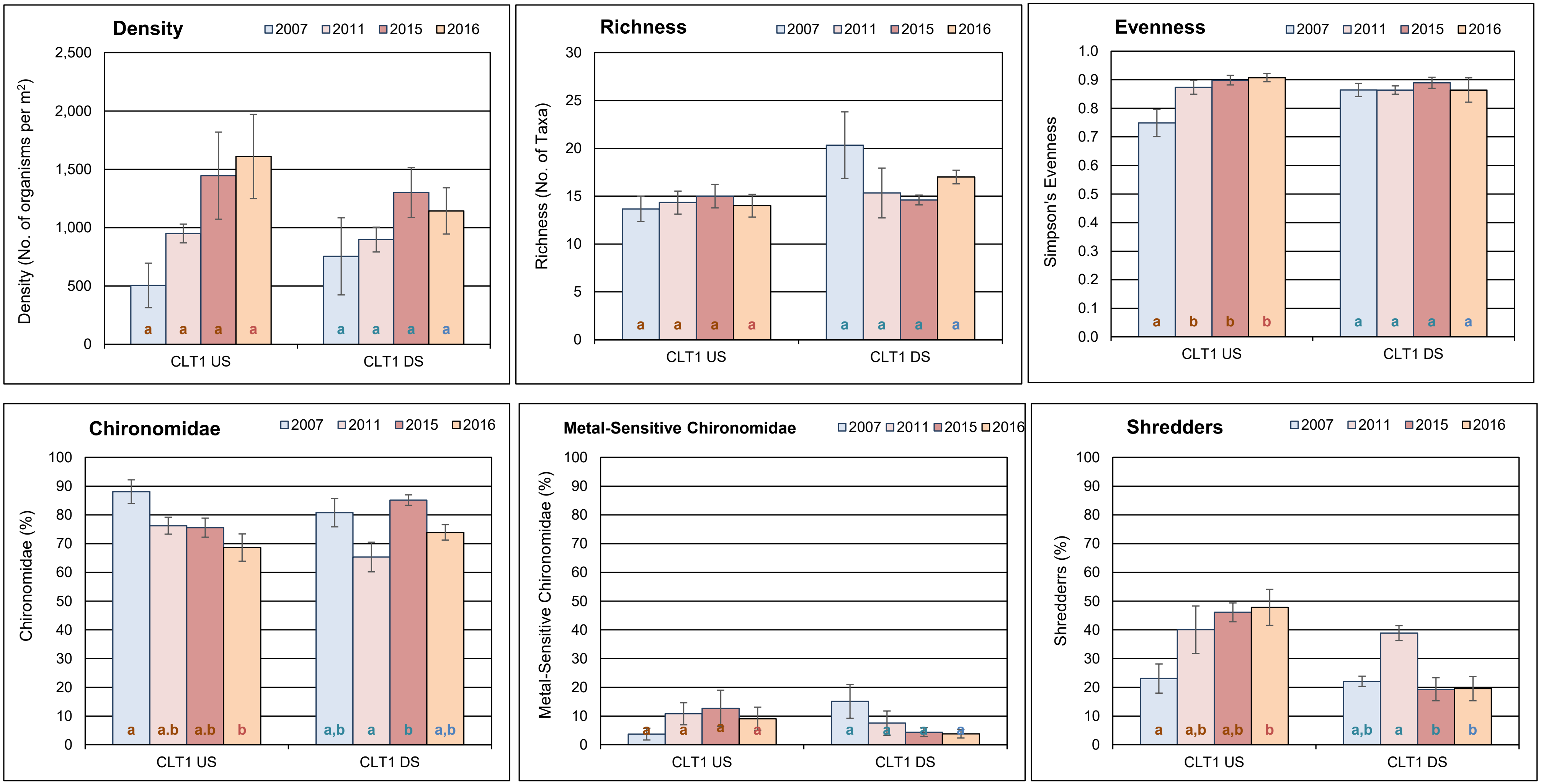


Figure 3.6: Comparison of key benthic invertebrate metrics (mean ± SE) at Camp Lake Tributary 1 stations among mine baseline (2007, 2011) and operational (2015, 2016) periods, Mary River Project CREMP, 2016. The same like-coloured letter inside bars indicate no significant difference between study years.

Index) compared to Unnnamed Reference Creek in 2016 (Table 3.2; Appendix Table F.7). Compositionally, the relative abundances of Hydracarina (water mites) and metal-sensitive chironomids were significantly higher at the CLT1 upper main stem than at Unnamed Reference Creek (Table 3.2; Appendix Table F.7). High relative abundance of metal-sensitive chironomids at the CLT1 upper main stem area, despite highest aqueous concentrations of metals within the Camp Lake system (Figure 3.2; Appendix Figure C.2), was consistent with concentrations of most metals below WQG at this area (see Appendix Table C.14). In addition, high relative abundance of metal-sensitive chironomids at the CLT1 upper main stem suggested that iron and uranium, which were observed at concentrations above WQG at this area (see Appendix Table C.14), were in forms that were not highly bioavailable. Other notable community compositional differences, including significantly higher and lower relative abundance of filterer and shredder FFG, respectively, at the CLT1 upper main stem compared to Unnamed Reference Creek (Appendix Table F.7), suggested a shift in dominant food resource at the CLT1 upper main stem. Specifically, a relatively high abundance of filterers at the CLT1 upper main stem suggested a greater reliance upon food resources suspended in the water column, including phytoplankton and fine particulate organic matter, than at Unnamed Reference Creek. These results were consistent with occurrence of relatively high chlorophyll a concentrations at the CLT1 upper main stem compared to the other CLT1 stations and the reference creeks (see Section 3.1.1.1). Collectively, the combination of relatively high benthic invertebrate density, richness (compared to the CLT1 north branch; Table 3.2; Appendix Table F.7) and proportion of the filterer FFG, together with relatively high chlorophyll a and aqueous nitrate concentrations, was consistent with a slight, mine-related enrichment effect on the benthic invertebrate community at the CLT1 upper main stem in 2016.

Despite suggestion of a mine-related enrichment influence at the CLT1 upper main stem, temporal comparisons did not indicate significant differences in benthic invertebrate density, richness, Simpson’s Evenness and relative abundance of key dominant groups and FFG in 2016 compared to baseline data collected in 2007 (Figure 3.6; Appendix Table F.9). In turn, this suggested that benthic invertebrate community features at the CLT1 upper main stem in 2016 had not changed appreciably from the pre-mine operation period, and that differences in community composition relative to reference conditions may reflect natural phenomena.

Lower Main Stem (CLT1 DS)

The benthic invertebrate community at the lower main stem of Camp Lake Tributary (CLT1 DS), just downstream of the mine tote road, showed no significant, ecologically meaningful, differences in density, richness and Simpson’s Evenness compared to Unnamed Reference Creek (Table 3.2; Appendix Table F.7). Nevertheless, the benthic invertebrate community assemblage at the CLT1 lower main stem differed from the reference areas based on significant differences in Bray-Curtis Index and composition of dominant invertebrate groups, FFG and habit preference groups (HPG; Table 3.2). Because no significant difference in the relative abundance of metal-sensitive chironomids was indicated between the CLT1 lower main stem and reference area (Table 3.2), the community composition differences between the mine-exposed and reference areas appeared to be unrelated to metal concentrations. Rather, the key differences in benthic invertebrate composition between areas, which included a significantly lower proportion of the collector-gatherer FFG and the clinger HPG at the CLT1 lower main stem, may have reflected greater reliance on interstitially deposited particulate organic matter food resources compared to a heavier reliance on in-stream vegetation as a food source at the reference area. Because substrate with significantly smaller diameter was sampled at the CLT1 lower main stem compared to Unnamed Reference Creek (Appendix Tables F.3 and F.4), differences in habitat may have also contributed to the indicated differences in benthic invertebrate community compositional features between areas.

Temporal comparison of the CLT1 lower main stem data indicated no significant differences in benthic invertebrate density, richness, Simpson's Evenness or the proportion of metal-sensitive chironomids between individual years of mine operation (2015, 2016) and the mine baseline (2007, 2011 data) period (Figure 3.6; Appendix Table F.10). In addition, no consistent types and/or direction of differences in the relative abundance of dominant groups or FFG were indicated between 2016 and years in which baseline data were collected at the CLT1 lower main stem (Figure 3.6; Appendix Table F.10). Overall, these results suggested no substantial changes in benthic invertebrate community features between the mine operational and mine baseline periods at the CLT1 lower main stem.

3.1.3.2 Camp Lake Tributary 2

At Camp Lake Tributary 2 (CLT2), sampling was conducted upstream and downstream of the mine tote road (areas CLT2 US and CLT2 DS, respectively) to assess for potential mine-related influences to the benthic invertebrate community. Benthic invertebrate density was significantly lower at both CLT2 study areas compared to Unnamed Reference Creek (Table 3.3). In addition, differences in community composition were indicated by significantly higher Bray-Curtis Index at both CLT2 study areas compared to the Unnamed Reference Creek. A significantly lower relative abundance of Hydracarina (water mites) and HPG clingers occurred at both CLT2 study areas compared to Unnamed Reference Creek (Table 3.3). Significantly lower relative abundance of chironomids and significantly higher relative abundance of FFG collector-gatherers and HPG sprawlers was also indicated at the CLT2 downstream area compared to Unnamed Reference Creek (Table 3.3; Appendix Table F.14).

Table 3.3: Benthic invertebrate community statistical comparison results among Camp Lake Tributary 2 and Unnamed Reference Creek study areas, Mary River Project CREMP, August 2016.

Season	Overall 3-group Comparison			Summary		Pair-wise, post-hoc comparisons[a]					
	Significant Difference Among Areas?	p-value	Statistical Test[b]	Area	Mean Value	(I) Area	(J) Area	Significant Difference Between Areas?	p-value	Magnitude of Difference	Statistical Test
Density (No. organisms/ m^2)	YES	0.00008	α	Reference	1,645	Reference	CLT2 US	YES	0.0311	-2.0	Tamhane's (α)
				CLT2 Upstream	412	Reference	CLT2 DS	YES	0.0188	-2.3	
				CLT2 Downstream	205	CLT2 US	CLT2 DS	YES	0.0187	-2.1	
Richness (Number of Taxa)	NO	0.10651	α , γ	Reference	18.6	Reference	CLT2 US	NO	0.7365	-	Tamhane's (α)
				CLT2 Upstream	17.2	Reference	CLT2 DS	NO	0.1708	-	
				CLT2 Downstream	14.0	CLT2 US	CLT2 DS	NO	0.4707	-	
Simpson's Evenness	NO	0.35742	α	Reference	0.873	Reference	CLT2 US	NO	0.8111	-	Tukey's (α)
				CLT2 Upstream	0.898	Reference	CLT2 DS	NO	0.6688	-	
				CLT2 Downstream	0.838	CLT2 US	CLT2 DS	NO	0.3291	-	
Bray-Curtis Index	YES	0.00000	α	Reference	0.237	Reference	CLT2 US	YES	0.0000	3.8	Tukey's (α)
				CLT2 Upstream	0.726	Reference	CLT2 DS	YES	0.0000	4.7	
				CLT2 Downstream	0.844	CLT2 US	CLT2 DS	NO	0.1376	-	
Oligochaeta (% of Community)	YES	0.08905	β	Reference	2.5%	Reference	CLT2 US	NO	0.6686	-	Tamhane's (β)
				CLT2 Upstream	4.9%	Reference	CLT2 DS	NO	0.4703	-	
				CLT2 Downstream	1.9%	CLT2 US	CLT2 DS	NO	0.2621	-	
Hydracarina (% of Community)	YES	0.00630	β	Reference	11.7%	Reference	CLT2 US	***YES***	***0.0220***	***-1.7***	Tukey's (β)
				CLT2 Upstream	5.5%	Reference	CLT2 DS	***YES***	***0.0078***	***-1.9***	
				CLT2 Downstream	4.5%	CLT2 US	CLT2 DS	NO	0.8324	-	
Chironomidae (% of Community)	YES	0.09836	β	Reference	70.8%	Reference	CLT2 US	NO	0.2460	-	Tukey's (β)
				CLT2 Upstream	79.5%	Reference	CLT2 DS	***YES***	***0.0955***	***1.3***	
				CLT2 Downstream	82.4%	CLT2 US	CLT2 DS	NO	0.8252	-	
Metal-Sensitive Chironomidae (%)	NO	0.30569	β	Reference	8.9%	Reference	CLT2 US	NO	0.2847	-	Tukey's (β)
				CLT2 Upstream	5.3%	Reference	CLT2 DS	NO	0.5718	-	
				CLT2 Downstream	5.4%	CLT2 US	CLT2 DS	NO	0.8413	-	
Tipulidae (% of Community)	NO	0.20459	β	Reference	4.3%	Reference	CLT2 US	NO	0.9992	-	Tukey's (β)
				CLT2 Upstream	4.0%	Reference	CLT2 DS	NO	0.2706	-	
				CLT2 Downstream	2.2%	CLT2 US	CLT2 DS	NO	0.2564	-	

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

[b] Data analysis included: α - data untransformed, single factor ANOVA test conducted; β - data logit transformed, single factor ANOVA test conducted; γ - data non-normal, test results validated using Kruskal-Wallis H-test (multiple group comparison) or Mann-Whitney U-test (pair-wise comparison), as appropriate.

Highlighted values indicate significant difference between study areas based on ANOVA p-value less than 0.10 that were also outside of a CES of ±2 SD, suggesting an ecologically meaningful difference.

BOLD Bold text values indicate significant differences between study areas based on ANOVA p-value less than 0.10, but a Critical Effect Size within ±2 SD, suggesting the difference is not ecologically meaningful.

In addition to a greater number of differences, the magnitude of these differences (compared to Unnamed Reference Creek) was greater at the CLT2 downstream area than at the upstream area, potentially indicating that the mine tote road had a greater influence on benthic invertebrates within CLT2 (Table 3.3; Appendix Table F.14). However, differences in habitat features that included significantly greater water velocity and less in-stream vegetation (Appendix Tables F.1, F.3 and F.4) potentially accounted for lower benthic invertebrate density and relative abundance of water mites and other HPG clinger taxa at the CLT2 study areas compared to the Unnamed Reference Creek. In part, this was supported by the lack of significant differences in richness, Simpson's Evenness, and relative abundance of all dominant invertebrate groups, FFG and HPG between the CLT2 upstream and downstream areas (Table 3.3; Appendix Table F.14).

Temporal comparisons indicated no significant differences in any benthic invertebrate community endpoints, including the relative abundance of all dominant invertebrate groups and FFG, at both CLT2 study areas in 2016 compared to 2007 baseline data with the exception of Simpson's Evenness (Figure 3.7; Appendix Tables F.15 and F.16.). Because high Simpson's Evenness is normally associated with a diverse, healthy benthic invertebrate community, the occurrence of significantly higher Simpson's Evenness at CLT2 in 2016 compared to 2007 was not consistent with an adverse influence related to recent mine operations. These results suggested that differences in benthic invertebrate community features between CLT2 and Unnamed Reference Creek in 2016 were most likely related to natural differences in habitat between watercourses, and that no appreciable changes to the benthic invertebrate community of CLT2 have occurred since commercial mine operations commenced in 2014.

3.2 Camp Lake (JLO)

3.2.1 Water Quality

In-situ water quality profiles conducted at Camp Lake showed no substantial spatial differences in water temperature, dissolved oxygen, pH or specific conductance with progression from the CLT1 inlet to the lake outlet during any of the winter, summer or fall seasonal sampling events in 2016[4] (Appendix Figures C.3 - C.6). Camp Lake water temperature profiles in 2016 suggested no thermal stratification during the winter and summer sampling events, but weak stratification during fall sampling that mirrored the fall temperature profile pattern at Reference Lake 3 (Figure 3.8). On average, water temperature near the bottom of the water column at

[4] The summer 2016 data suggested considerable variation among Camp Lake stations, but review of field collection notes suggested that this variation likely reflected meter calibration-related differences between sampling dates.

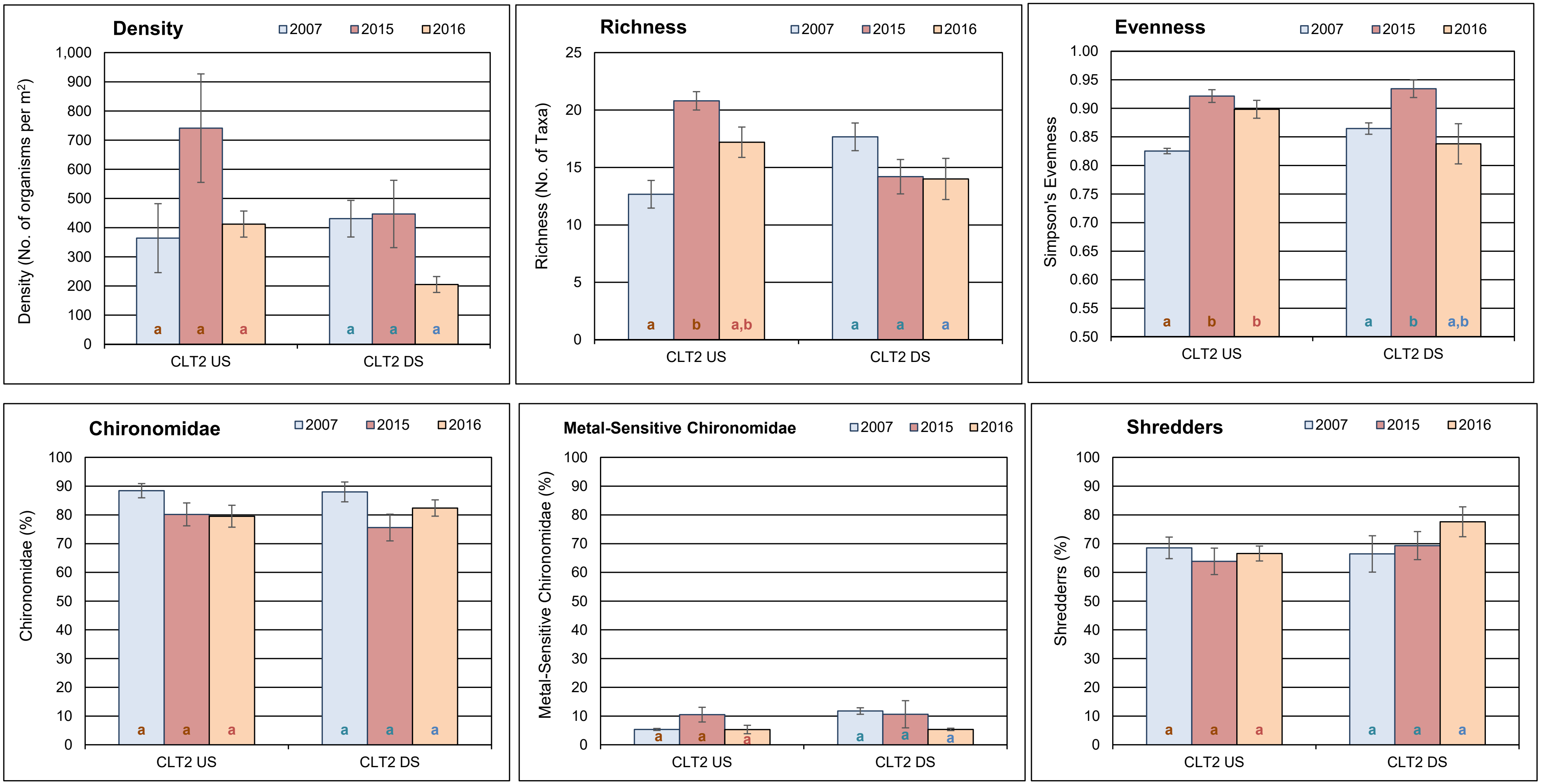


Figure 3.7: Comparison of key benthic invertebrate metrics (mean ± SE) at Camp Lake Tributary 2 stations among mine baseline (2007, 2011) and operational (2015, 2016) periods, Mary River Project CREMP, 2016. The same like-coloured letter inside bars indicate no significant difference between study years.

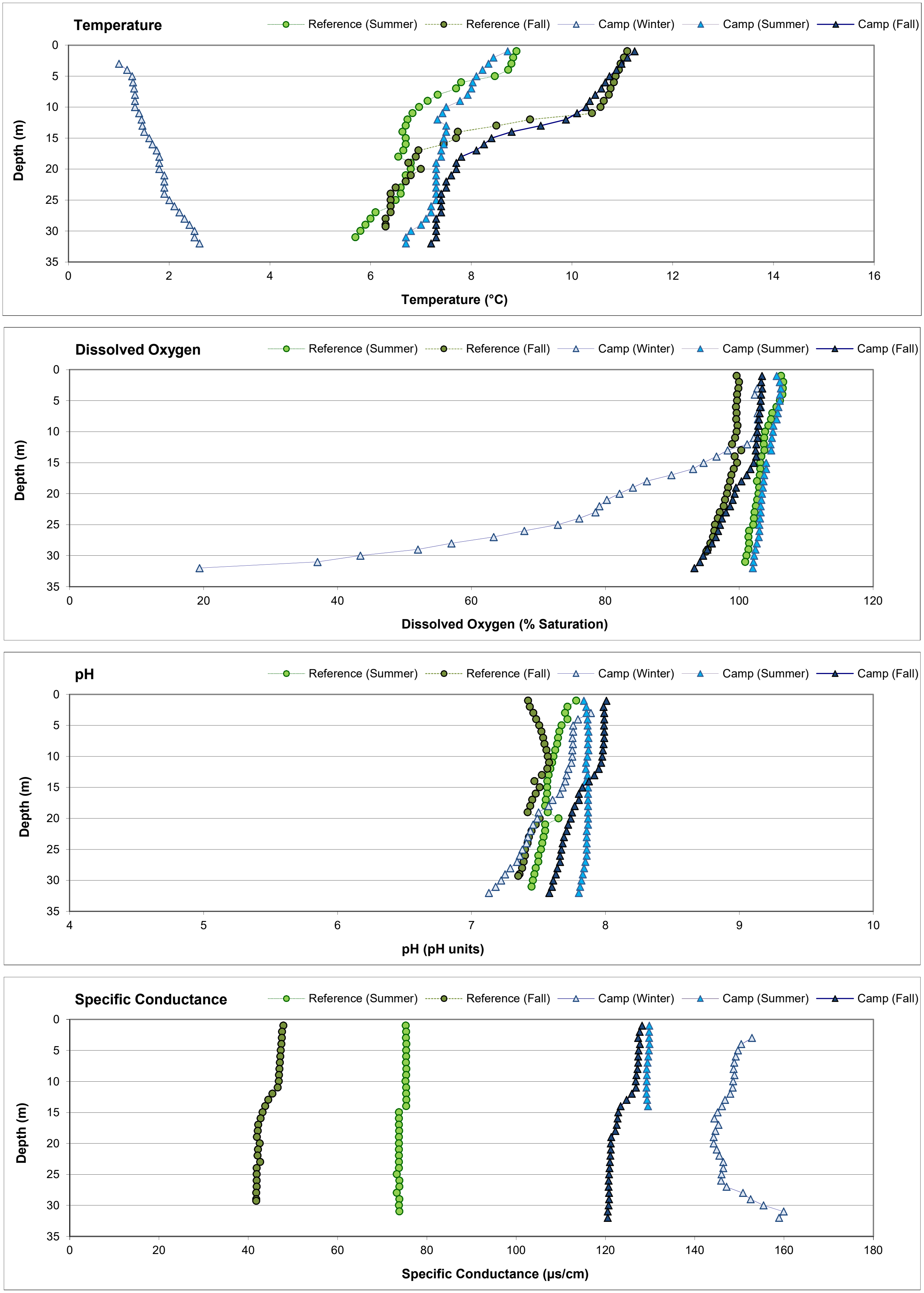


Figure 3.8: Average *in-situ* water quality with depth from surface at Camp Lake (mine-exposed area) compared to Reference Lake 3 during winter, summer, and fall sampling events, Mary River Project CREMP, 2016.

littoral stations of Camp Lake was significantly cooler than at Reference Lake 3 (Figure 3.9; Appendix Tables C.22 – C.23). Although cooler bottom water temperatures at Camp Lake littoral stations may have reflected greater station depth compared to the reference lake, the small incremental difference in water temperature (i.e., 0.7˚C) was unlikely to result in meaningful ecological differences between lakes. Dissolved oxygen profiles conducted at Camp Lake in 2016 showed declining saturation levels with increased depth beginning at approximately 12 m below surface in the winter, but otherwise showed no appreciable changes from surface to bottom during summer or fall 2016, mirroring the dissolved oxygen profiles at Reference Lake 3 (Figure 3.8) and observations from Camp Lake in 2015. Dissolved oxygen conditions near the bottom of the water column at littoral sampling depths of Camp Lake were fully saturated, and significantly higher than at Reference Lake 3 during fall sampling in 2016 (Figure 3.9; Appendix Table C.23). In addition, dissolved oxygen saturation at Camp Lake was typically well above the WQG minimum for the protection of cold water biota (i.e., 54%) during all seasonal sampling events in 2016 except at water depths greater than approximately 30 m in winter (Figures 3.8 and 3.9). This suggested that dissolved oxygen concentrations were not likely to be limiting to biota at Camp Lake for the entire lake volume for the majority of the year.

In-situ profiles of pH and specific conductance showed no substantial change from the surface to bottom of the Camp Lake water column, indicating the absence of any chemical stratification (Figure 3.8). Although the bottom pH at littoral stations of Camp Lake was significantly higher than at the reference lake during the fall sampling event (Appendix Tables C.22 – C.23), the mean incremental difference between lakes was very small (i.e., 0.3 pH units) and all pH values were consistently within WQG limits (Figure 3.9), suggesting that the pH difference between lakes was not ecologically meaningful. Specific conductance was significantly higher at Camp Lake compared to the reference lake during fall sampling in 2016 (Figure 3.9). However, because mean specific conductance at Camp Lake was intermediate to that of the reference creek and river stations, the occurrence of higher specific conductance at Camp Lake compared to the reference lake likely reflected natural phenomena. Secchi depth readings, which served as a proxy for water clarity, were significantly lower (i.e., shallower) at Camp Lake compared to Reference Lake 3 during the 2016 fall sampling event (Appendix Tables C.22 – C.23). No spatial gradient in Secchi depth readings was apparent with progression from the CLT inlet to the lake outlet stations in fall 2016 at Camp Lake (Appendix Table C.21).

Water chemistry data collected at Camp Lake in 2016 showed no distinct spatial differences with progression from the CLT inlets to the lake outlet during any of the winter, summer or fall sampling events in 2016 (Table 3.4; Appendix Table C.24), suggesting that the lake waters

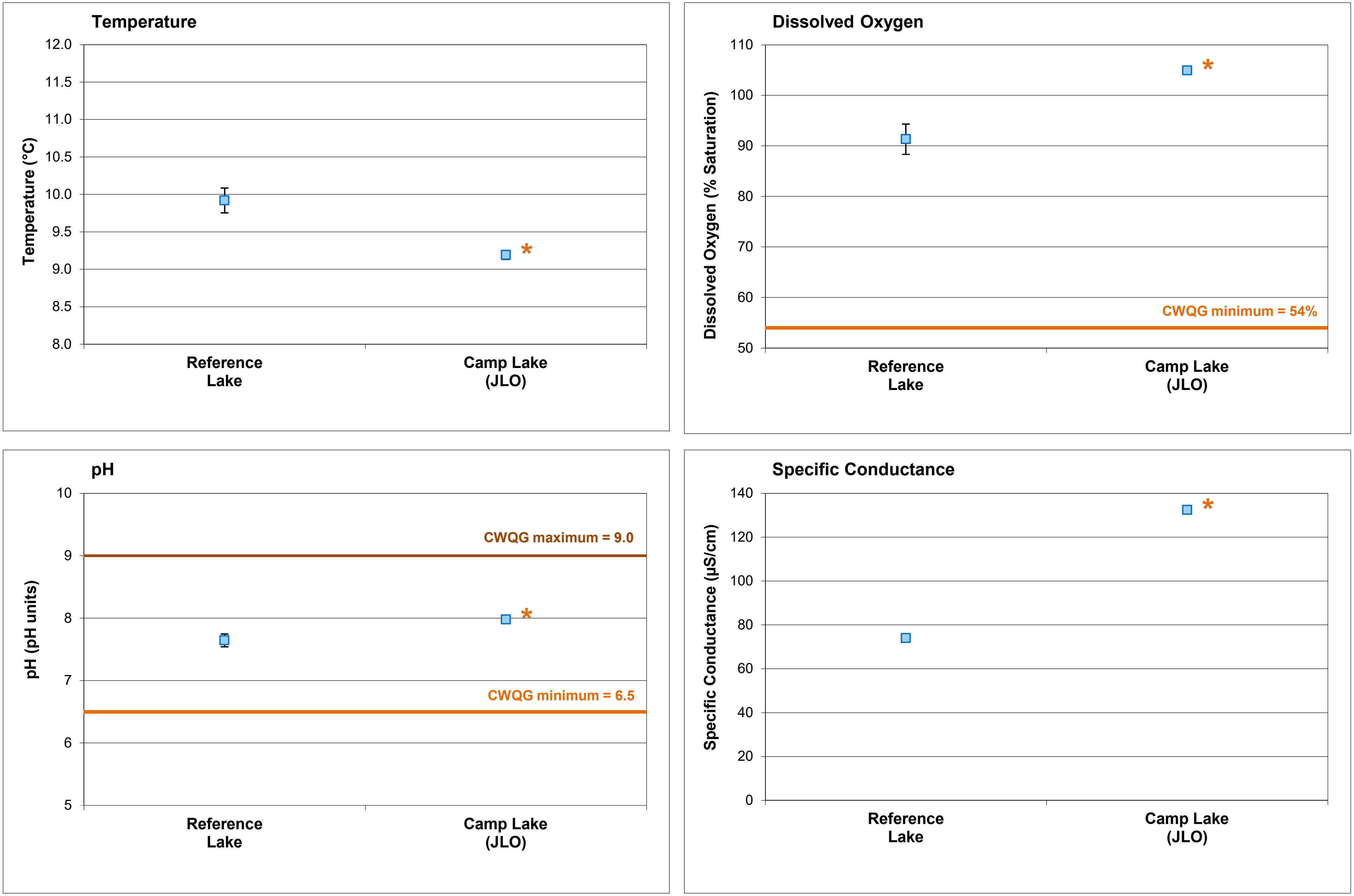


Figure 3.9: Comparison of in-situ water quality variables (mean ± SD; n = 5) measured near the bottom of the water column at Camp Lake (JLO) and Reference Lake 3 (REF3) littoral benthic invertebrate community stations, Mary River Project CREMP, August 2016. An asterisk (*) next to the Camp Lake data point indicates a significant difference compared to the reference lake measure.

Table 3.4: Water chemistry at Camp Lake (JLO) and Reference Lake 3 (REF3) monitoring stations, Mary River Project CREMP, August 2016. Values are averages of samples taken from the surface and the bottom of the water column at each station.

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Lake 3 Average (n = 3) Fall 2016	Camp Lake Stations JL0-02 22-Aug-16	JL0-10 22-Aug-16	JL0-01 22-Aug-16	JL0-07 22-Aug-16	JL0-09 22-Aug-16	J0-01 Camp Lake Outlet 20-Aug-2016
Conventionals	Conductivity (lab)	umho/cm	-	-	84	139	139	135	136	136	137
	pH (lab)	pH	6.5 - 9.0	-	7.68	8.11	8.11	8.04	8.00	8.07	8.01
	Hardness (as $CaCO_3$)	mg/L	-	-	35	65	67	64	65	66	67
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	2.45	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	39	76	73	64	71	74	67
	Turbidity	NTU	-	-	0.33	0.47	0.43	0.72	0.47	0.47	0.40
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	65	61	65	65	67	64
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.040	<0.020	<0.020	0.042	0.030	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.7	1.7	1.65	1.55	1.8	1.7	1.7
	Total Organic Carbon	mg/L	-	-	2.8	2.4	1.8	1.725	2.1	2.4	2.0
	Total Phosphorus	mg/L	0.020[α]	-	0.0099	0.0037	0.0059	0.0036	0.0045	0.0069	0.0039
	Phenols	mg/L	0.004[α]	-	0.0031	0.0015	0.0011	0.0017	0.0012	0.0061	0.0038
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.27	3.63	3.49	3.39	3.50	3.42	3.47
	Sulphate (SO_4)	mg/L	218[β]	218	4.1	2.3	2.2	2.1	2.1	2.1	2.2
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0042	0.0062	0.0050	0.0042	0.0052	0.0050	0.0047
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00653	0.00678	0.00644	0.00616	0.00657	0.00634	0.00663
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	7.0	13.7	13.5	13.3	13.2	13.2	13.3
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.00082	0.00101	0.00084	0.00080	0.00093	0.00084	0.00082
	Iron (Fe)	mg/L	0.30	0.326	<0.030	<0.030	<0.030	<0.030	<0.030	<0.030	0.094
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	0.0014	0.0013	0.0013	0.0013	0.0012	0.0011
	Magnesium (Mg)	mg/L	-	-	4.3	8.2	8.0	7.6	7.8	8.0	8.2
	Manganese (Mn)	mg/L	0.935[β]	-	0.00062	0.00146	0.00138	0.00153	0.00171	0.00154	0.00277
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00014	0.00026	0.00026	0.00025	0.00025	0.00026	0.00027
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	0.00059	0.00061	0.00058	0.00060	0.00060	0.00073
	Potassium (K)	mg/L	-	-	0.9	1.1	1.1	1.0	1.1	1.1	1.0
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.42	0.36	0.34	0.35	0.41	0.36	0.38
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.84	1.44	1.36	1.33	1.45	1.40	1.36
	Strontium (Sr)	mg/L	-	-	0.0081	0.0106	0.0103	0.0100	0.0100	0.0100	0.0098
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.000270	0.0008175	0.0007745	0.00073275	0.000711	0.00075	0.000782
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030

[a] Canadian Water Quality Guideline (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). See Table 2.3 for information regarding WQG criteria.

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

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

BOLD Indicates parameter concentration above the applicable AEMP benchmark.

were well mixed laterally. Only a slight elevation (i.e., 3- to 5-fold higher) in manganese concentrations was evident at Camp Lake compared to the reference lake during the summer 2016 sampling event (Table 3.4; Appendix Table C.26). Concentrations of manganese, together with aluminum, showed a significant positive correlation with turbidity at Camp Lake using all 2016 data ($r = 0.52$ and 0.65, respectively), suggesting that these metals were largely associated with suspended particulate material in Camp Lake and thus were unlikely to be bioavailable. Notably, concentrations of all parameters were well below established WQG and AEMP benchmarks at Camp Lake during all sampling events in 2016[5] (Table 3.4; Appendix Table C.24), further indicating that parameter concentrations at Camp Lake were unlikely to adversely affect biota.

Temporal comparisons of Camp Lake water chemistry data indicated that, of the parameters shown to be elevated at CLT1 in 2016, only conductivity and concentrations of chloride, molybdenum, sodium, strontium and uranium showed continuous increases over the mine baseline, construction and operational periods (Figure 3.10; Appendix Figure C.7). Other parameters, including hardness, iron, manganese, nitrate and sulphate, showed no consistent direction of change between the mine baseline and operational periods. Notably, parameter concentrations were consistently well below WQG and AEMP benchmarks through all years of mine construction and operation at Camp Lake (e.g., Appendix Table C.24) and thus, no adverse mine-related influences on lake water quality were suggested at Camp Lake since commercial mine operations commenced in 2014.

3.2.2 Sediment Quality

Surficial sediment (i.e., top 2 cm) collected at the Camp Lake coring stations was composed mainly of silty loam and sandy loam with low total organic carbon (TOC) content, except at the outlet littoral station (JLO-30) where sand constituted the predominant substrate material (Figure 3.11). A surficial and/or sub-surface layer of oxidized material (likely iron hydroxide or oxy-hydroxides), visible as reddish-orange to orange-brown substrate, was commonly observed in sediments of Camp Lake (Appendix Tables D.5 – D.7). However, similar substrate was observed at Reference Lake 3 (Appendix Tables D.1 – D.3), suggesting the natural occurrence of iron (oxy)hydroxides in the sediment of lakes within the mine LSA. Substrates of Camp Lake exhibited minor, sporadic blackening at sediment depths greater than 2 cm at some stations, suggesting occasional incidence of reducing conditions within substrates of the lake. However, no strongly defined redox boundaries were identified visually, and no noticeable sulphidic odours potentially associated with reducing sediment conditions were

[5] Refer to footnote 2 (page 23) and Appendix B regarding phenol concentrations above WQG at the mine-exposed and reference areas of the Mary River Project LSA waterbodies.

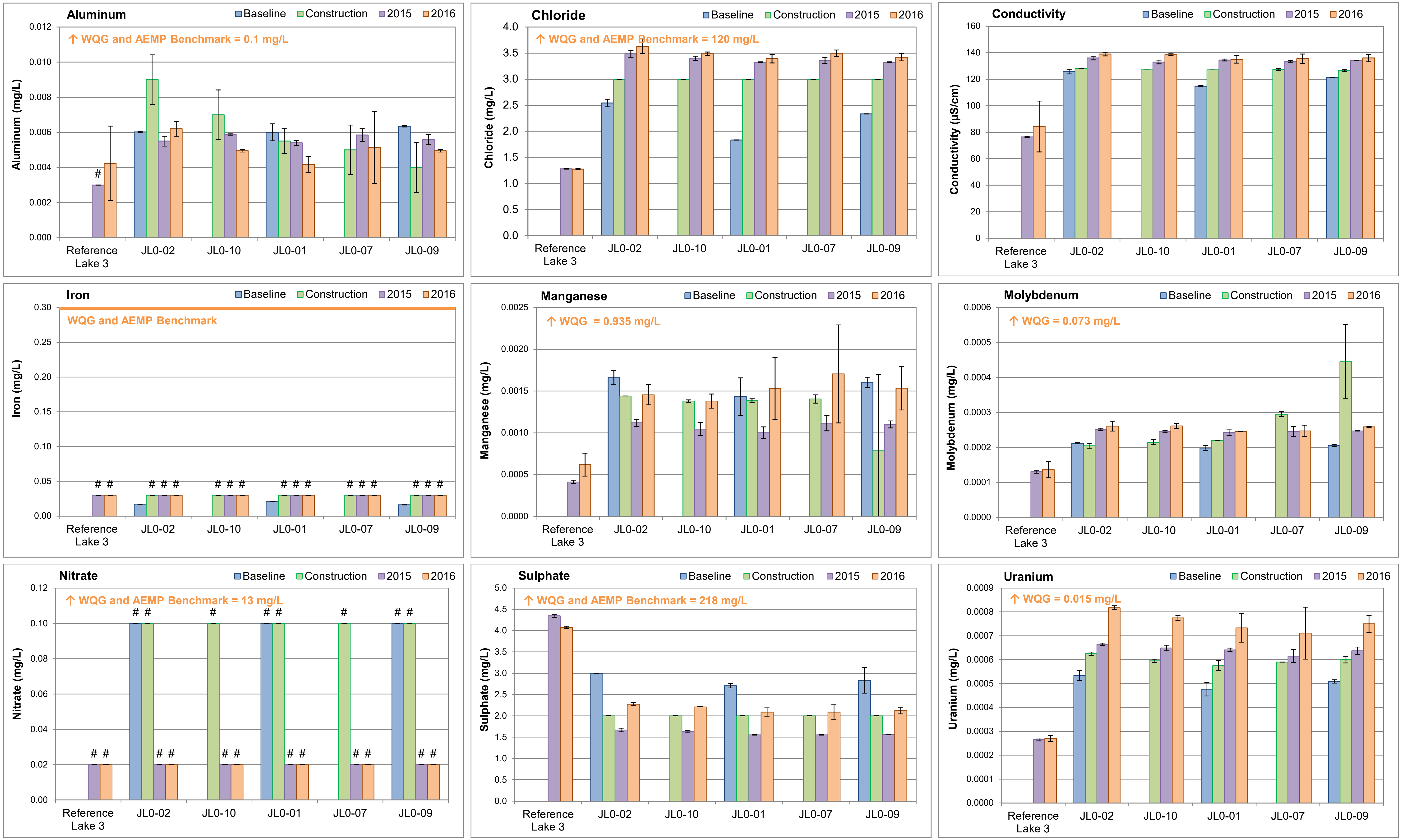


Figure 3.10: Temporal comparison of water chemistry at Camp Lake (JLO) for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during fall. Values represent mean ± SD. Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Camp Lake.

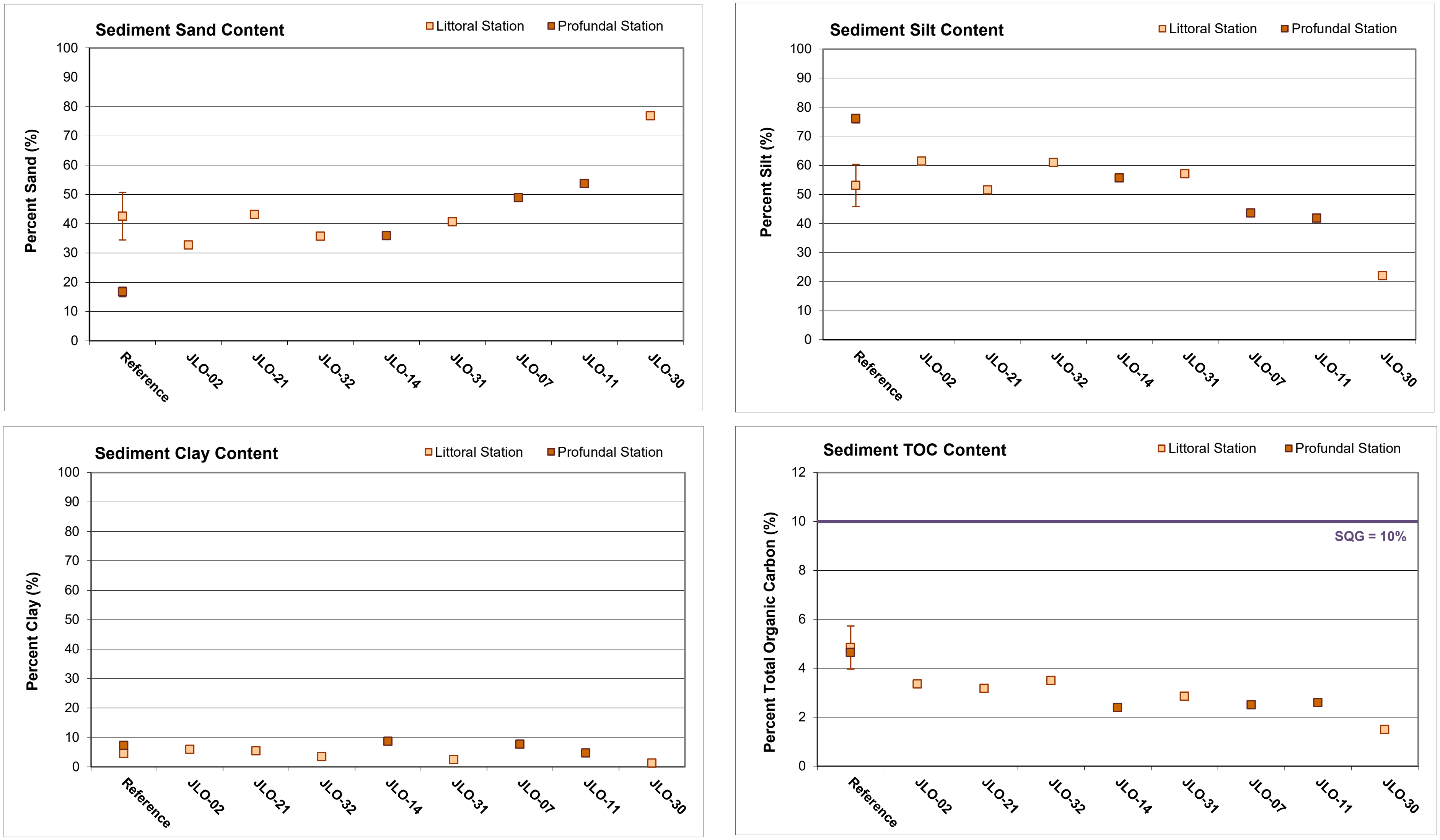


Figure 3.11: Sediment particle size and total organic carbon (TOC) content comparisons among Camp Lake (JLO) sediment monitoring stations and to Reference Lake 3 averages (mean ± SE), Mary River Project CREMP, August 2016.

detected at Camp Lake littoral and profundal stations to sediment depths as great as approximately 20 cm during the 2016 fall sampling event (Appendix Tables D.5 – D.7). Qualitative observations suggestive of reducing sediment conditions were similar between Camp Lake and Reference Lake 3 in 2016 (Appendix Tables D.1 – D.3 and D.5 – D.7), which indicated that factors leading to reduced sediment conditions were comparable between lakes.

No spatial gradients in sediment metal concentrations were evident with progression from stations located nearest to the CLT1 inlet to those located near the lake outlet of Camp Lake in 2016 (Appendix Table D.9). Sediment metal concentrations were generally lower at littoral stations than at profundal stations of Camp Lake (Table 3.5; Appendix Table D.9), mirroring similar patterns at the reference lake. On average, sediment arsenic and manganese concentrations were slightly elevated (i.e., 2- to 5-fold higher) at Camp Lake littoral stations compared to sediment at Reference Lake 3 littoral stations (Table 3.5; Appendix Table D.10). However, metal concentrations in the profundal sediment of Camp Lake were comparable to those of the reference lake in 2016 (Table 3.5; Appendix Table D.10). Although mean iron, manganese and phosphorus concentrations were above respective SQG at Camp Lake littoral and/or profundal stations, mean concentrations of iron and manganese were also above SQG in the Reference Lake 3 profundal sediments in 2016 (Table 3.5). Similarly, although mean arsenic concentrations in littoral and profundal sediments, and mean iron and phosphorus concentrations in profundal sediments, were above respective AEMP benchmarks at Camp Lake, mean arsenic and iron concentrations were also above AEMP benchmarks in profundal sediment of Reference Lake 3 (Table 3.5). These data suggested natural elevation of arsenic, iron and manganese in sediments of LSA lakes relative to applicable SQG and/or AEMP benchmarks.

Temporal comparisons of the sediment chemistry data indicated slightly higher (2- to 5-fold greater) arsenic, manganese and molybdenum concentrations in littoral and/or profundal sediment of Camp Lake in 2016 compared to the baseline period[6] (Figure 3.12; Appendix Table D.10). Of these metals, only manganese showed progressively higher concentrations over baseline, mine construction and 2015 and 2016 mine operation periods at littoral stations of Camp Lake (Figure 3.12). Similarly, arsenic and other metals including barium, iron, magnesium and phosphorus, showed continuously higher concentrations between mine baseline and 2016 periods at profundal stations of Camp Lake (Figure 3.12; Appendix

[6] Reported sediment boron concentrations in 2015 and 2016 were considerably higher (i.e., 10- to 70-fold) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences.

Table 3.5: Sediment particle size, total organic carbon, and metal concentrations at Camp Lake (JLO) and Reference Lake 3 (REF3) sediment monitoring stations, Mary River Project CREMP, August 2016.

	Analyte	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b]	Littoral Stations		Profundal Stations	
					Reference Lake (n = 5)	Camp Lake (n = 5)	Reference Lake (n = 5)	Camp Lake (n = 3)
					Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error
Non-metals	Sand	%	-	-	42.5 ± 8.1	45.8 ± 8.0	16.7 ± 1.5	46.1 ± 5.3
	Silt	%	-	-	53.1 ± 7.3	50.6 ± 7.4	76.1 ± 1.4	47.0 ± 4.3
	Clay	%	-	-	4.4 ± 1.0	3.7 ± 0.9	7.2 ± 0.4	6.9 ± 1.2
	Moisture	%	-	-	89.7 ± 6.0	73.5 ± 4.2	83.5 ± 5.4	68.8 ± 4.0
	Total Organic Carbon	%	10^{α}	-	4.85 ± 0.88	2.87 ± 0.36	4.64 ± 0.13	2.49 ± 0.06
Metals	Aluminum (Al)	mg/kg	-	-	16,480 ± 397	13,460 ± 1,760	25,150 ± 1,418	18,900 ± 702
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.11 ± 0.006	0.12 ± 0.02	<0.10 ± 0
	Arsenic (As)	mg/kg	17	5.9	3.71 ± 0.26	***8.70*** ± 1.96	***6.47*** ± 0.27	***8.94*** ± 2.41
	Barium (Ba)	mg/kg	-	-	112 ± 11	128 ± 28	162 ± 8	126 ± 33
	Beryllium (Be)	mg/kg	-	-	0.67 ± 0.02	0.77 ± 0.10	1.02 ± 0.05	1.04 ± 0.04
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.30 ± 0.04	0.21 ± 0.00	0.32 ± 0.02
	Boron (B)	mg/kg	-	-	13.0 ± 0.9	20.9 ± 2.6	19.2 ± 1.0	27.9 ± 1.5
	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.035	0.201 ± 0.044	0.180 ± 0.010	0.176 ± 0.027
	Calcium (Ca)	mg/kg	-	-	5,128 ± 470	4,404 ± 611	6,111 ± 156	4,540 ± 87
	Chromium (Cr)	mg/kg	90	98	55.0 ± 1.2	57.5 ± 6.9	80.0 ± 4.1	76.2 ± 2.0
	Cobalt (Co)	mg/kg	-	-	10.15 ± 0.57	17.00 ± 2.44	18.15 ± 0.75	20.30 ± 2.21
	Copper (Cu)	mg/kg	110^{α}	50	***66.5*** ± 7.4	38.2 ± 6.1	***101*** ± 5.6	49.8 ± 0.5
	Iron (Fe)	mg/kg	$40,000^{\alpha}$	52,400	29,840 ± 3,488	48,150 ± 8,692	***53,580*** ± 2,174	***61,633*** ± 8,732
	Lead (Pb)	mg/kg	91	35	***46.0*** ± 17.4	20.0 ± 1.8	29.5 ± 5.0	24.1 ± 1.0
	Lithium (Li)	mg/kg	-	-	27.3 ± 0.4	25.5 ± 3.3	41.7 ± 2.1	34.6 ± 1.7
	Magnesium (Mg)	mg/kg	-	-	10,852 ± 274	10,792 ± 1,375	16,160 ± 814	13,567 ± 240
	Manganese (Mn)	mg/kg	$1,100^{\alpha,\beta}$	4,370	496 ± 99	2,583 ± 758	1,866 ± 449	2,307 ± 1,583
	Mercury (Hg)	mg/kg	0.486	0.17	0.0355 ± 0.0063	0.0368 ± 0.0064	0.0699 ± 0.0019	0.0555 ± 0.0032
	Molybdenum (Mo)	mg/kg	-	-	2.19 ± 0.49	2.64 ± 0.83	3.27 ± 0.34	1.78 ± 0.62
	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	72	38.6 ± 1.6	64.7 ± 9.0	56.3 ± 2.6	69.7 ± 2.8
	Phosphorus (P)	mg/kg	$2,000^{\alpha}$	1,580	840 ± 47	1,521 ± 256	1,121 ± 57	***2,137*** ± 428
	Potassium (K)	mg/kg	-	-	3,894 ± 172	3,383 ± 428	5,891 ± 281	4,773 ± 205
	Selenium (Se)	mg/kg	-	-	0.49 ± 0.06	0.39 ± 0.05	0.85 ± 0.06	0.54 ± 0.04
	Silver (Ag)	mg/kg	-	-	0.12 ± 0.01	0.11 ± 0.00	0.27 ± 0.01	0.15 ± 0.01
	Sodium (Na)	mg/kg	-	-	296 ± 29	152 ± 19	455 ± 24	274 ± 23
	Strontium (Sr)	mg/kg	-	-	11.4 ± 0.5	8.9 ± 1.0	15.8 ± 0.6	15.4 ± 2.3
	Sulphur (S)	mg/kg	-	-	<5,000 ± 0	<5,000 ± 0	<5,000 ± 0	<5,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.388 ± 0.021	0.475 ± 0.075	0.801 ± 0.035	0.504 ± 0.069
	Tin (Sn)	mg/kg	-	-	56.3 ± 28.9	5.7 ± 1.4	16.3 ± 7.8	3.3 ± 0.9
	Titanium (Ti)	mg/kg	-	-	1,072 ± 36	733 ± 89	1,331 ± 69	877 ± 53
	Uranium (U)	mg/kg	-	-	11.9 ± 1.5	5.05 ± 1.0	27.3 ± 1.5	7.20 ± 0.1
	Vanadium (V)	mg/kg	-	-	50.0 ± 1.3	47.9 ± 6.1	72.0 ± 3.6	62.6 ± 1.0
	Zinc (Zn)	mg/kg	315	135	73.7 ± 2.7	47.4 ± 6.3	105 ± 5.1	61.9 ± 1.8
	Zirconium (Zr)	mg/kg	-	-	4.3 ± 0.6	4.1 ± 1.0	4.0 ± 0.2	5.2 ± 0.8

[a] Canadian Sediment Quality Guideline for the protection of aquatic life, probable effects level (PEL; CCME 2016) except those indicated by α (Ontario Provincial Sediment Quality Objective [PSQO], severe effect level (SEL); OMOE 1993) and β (British Columbia Working Sediment Quality Guideline [BCSQG], probable effects level (PEL; BCMOE 2016)).

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

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

BOLD Indicates parameter concentration above the AEMP Benchmark.

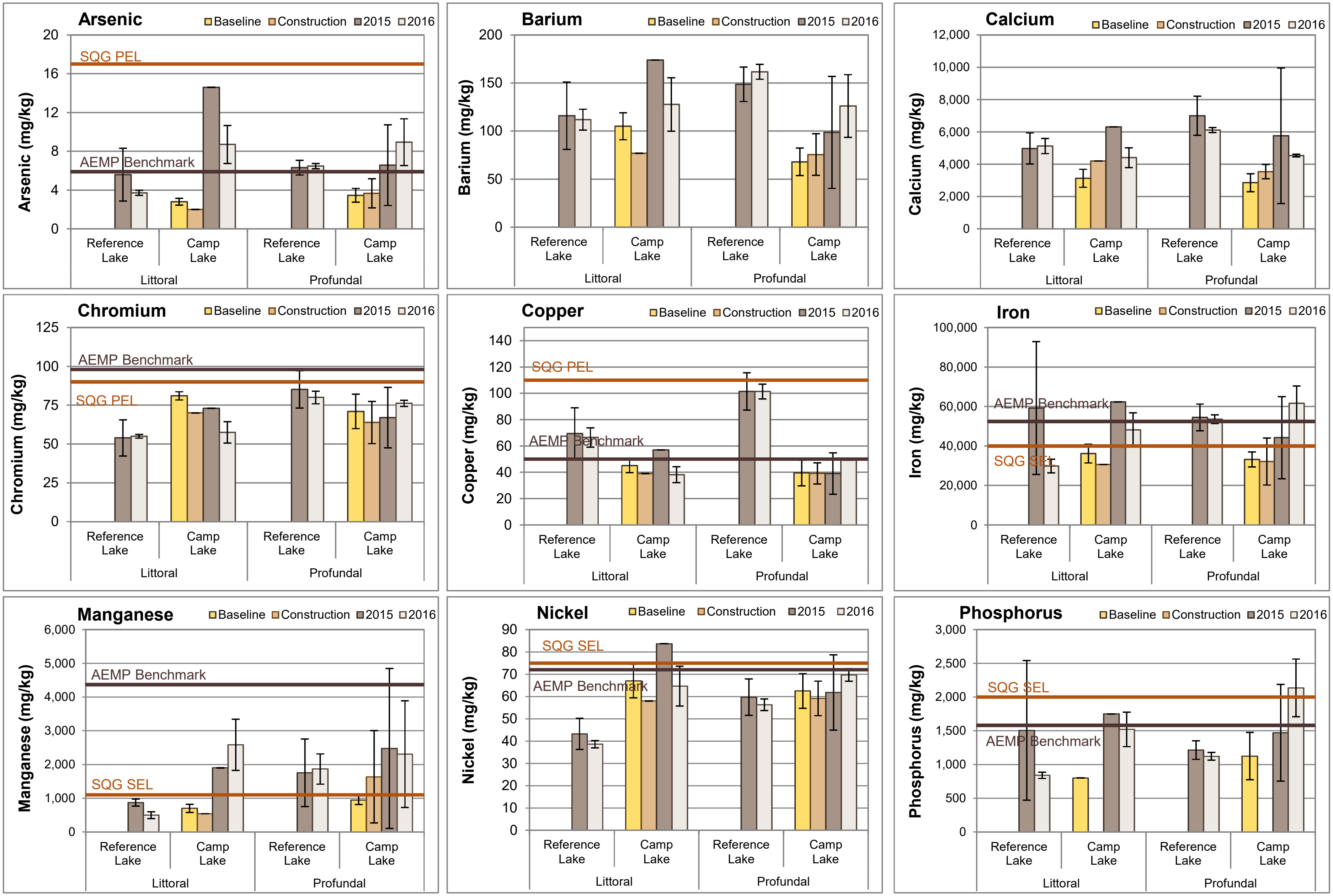


Figure 3.12: Temporal comparison of sediment metal concentrations (mean ± SD) at littoral and profundal stations of Camp Lake and Reference Lake 3 for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods, Mary River Project CREMP.

Table D.10). In part, the changes in sediment metal concentrations may have reflected changes in the number and/or location of littoral and profundal sediment quality monitoring stations at Camp Lake among studies. For instance, Station JLO-2 represented the only littoral station in Camp Lake during the baseline, 2014 and 2015 studies, and only the three deepest profundal stations were maintained in the 2016 study compared to previous studies that included up to nine profundal stations. The occurrence of Camp Lake sediment metal concentrations more closely reflecting those of the reference lake during mine operation (i.e., 2015, 2016) than during the mine baseline period was consistent with changes that may be expected from increased/decreased sampling replication at Camp Lake. Notwithstanding uncertainty related to changes in station replication among studies at Camp Lake, and taking reference lake sediment metal concentrations into account, higher concentrations of arsenic and manganese in littoral sediments of Camp Lake since the baseline period potentially reflected recent mine construction and/or operation influences to the lake shallows. In contrast, metals in Camp Lake profundal sediments showed no definitive changes in concentrations since the mine baseline period.

3.2.3 Phytoplankton

Camp Lake chlorophyll a concentrations showed no distinct gradients with distance from the CLT inlet to the lake outlet stations during any of the winter, summer or fall sampling events in 2016, although concentrations were somewhat lower at stations near the lake outlet during the summer and winter sampling events (Figure 3.13). Chlorophyll a concentrations differed significantly among all seasons at Camp Lake in 2016, with highest and lowest concentrations observed in summer and winter, respectively (Appendix Table E.4), and mirroring seasonal differences observed at the reference lake (Appendix Table B.8). On average, chlorophyll a concentrations at Camp Lake were significantly higher than at Reference Lake 3 during the summer and fall sampling events (Appendix Tables E.5 and E.6), suggesting greater phytoplankton density at Camp Lake. However, chlorophyll a concentrations were well below the AEMP benchmark of 3.7 µg/L during all winter, summer and fall sampling events in 2016 (Figure 3.13). Camp Lake mean chlorophyll a concentrations in 2016 suggested low phytoplankton productivity and an 'oligotrophic' trophic status based on Wetzel (2001) lake classification. This trophic status classification was also consistent with an ultra-oligotrophic to oligotrophic CWQG categorization for Camp Lake based on mean aqueous total phosphorus concentrations below 10 µg/L during all 2016 lake sampling events (Table 3.4; Appendix Table C.24).

Temporal comparisons of the Camp Lake chlorophyll a data did not indicate any consistent significant differences among the mine construction (2014) and operational (2015, 2016) years

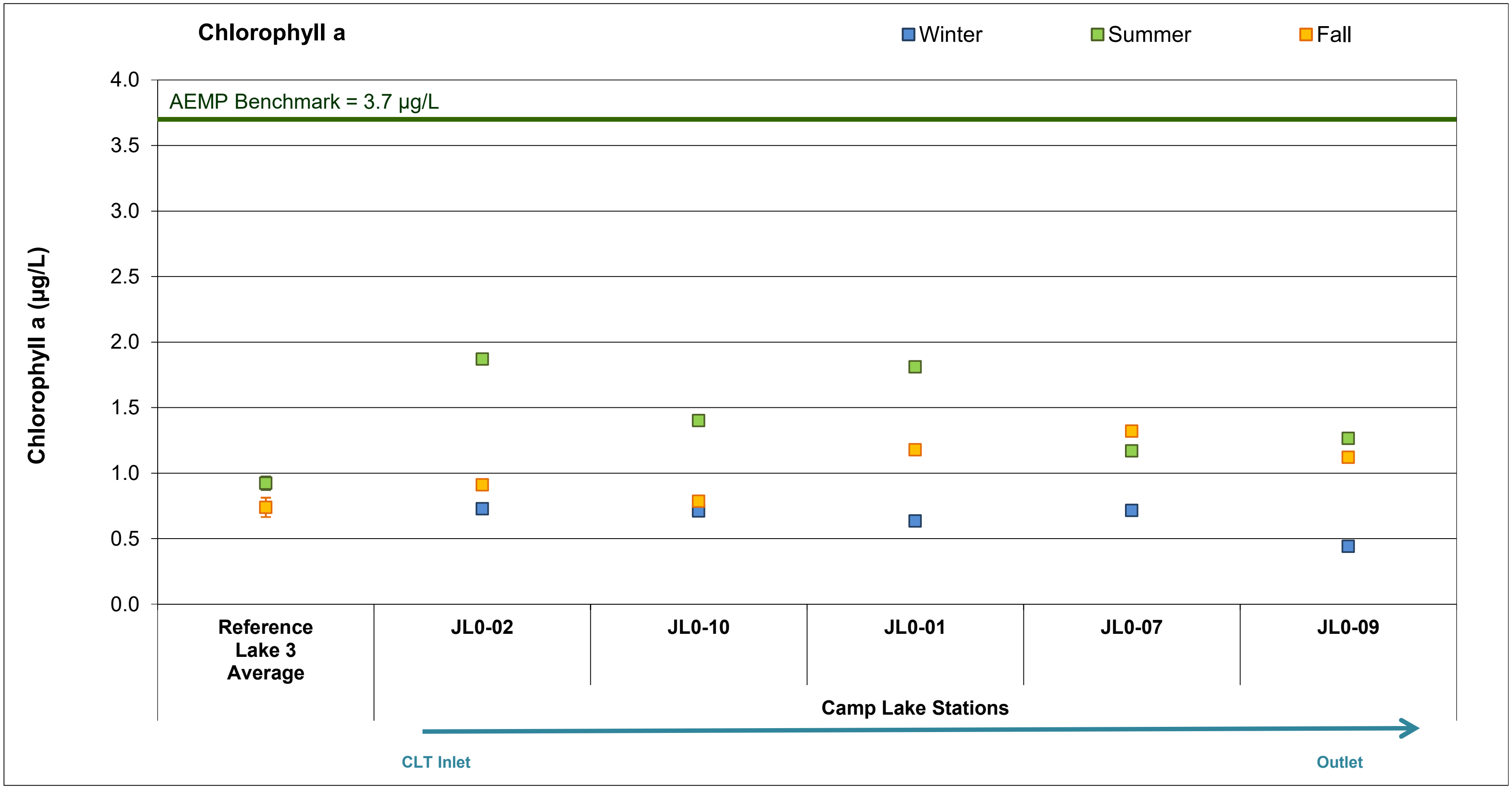


Figure 3.13: Chlorophyll a concentrations at Camp Lake (JLO) phytoplankton monitoring stations, Mary River Project CREMP, 2016. Values are averages of samples taken from the surface and the bottom of the water column at each station. Reference values represent mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2016.

for seasonal data collected in winter, summer and fall (Figure 3.14). In addition, annual average chlorophyll a concentrations did not differ significantly among the most recent three years (Appendix Table E.7), suggesting no changes in the trophic status of Camp Lake since mine operations commenced at the Mary River Project. No chlorophyll a baseline (2005 – 2013) data are available for Camp Lake, precluding comparisons to conditions prior to the mine construction period.

3.2.4 Benthic Invertebrate Community

Benthic invertebrate community density and richness at littoral habitat of Camp Lake did not differ significantly from Reference Lake 3 in 2016 (Table 3.6). Simpson's Evenness was significantly higher at Camp Lake than at the reference lake in 2016, indicating that organism numbers were more uniformly distributed across a diversity of taxa at Camp Lake. Although a high Simpson's Evenness is generally indicative of healthy benthic invertebrate community conditions, the magnitude of difference in Simpson's Evenness between lakes was within a critical effect size (CES_{BIC}) of ±2 reference area standard deviations (SD_{REF}), suggesting that this difference was not ecologically meaningful. Benthic invertebrate community composition differences were evident between Camp Lake and Reference Lake 3 littoral habitat based on significantly higher Bray-Curtis Index at Camp Lake, and by significant differences in the relative abundance of dominant taxonomic groups and HPG between lakes (Table 3.6). The key differences in community structure included significantly lower relative abundance of Ostracoda (seed shrimp) and significantly higher relative abundance of Chironomidae (non-biting midges) at Camp Lake compared to the reference lake. However, because the relative abundance of metal-sensitive Chironomidae did not differ significantly between Camp Lake and Reference Lake 3 (Table 3.6), the difference in benthic invertebrate community structure between lakes was not suggestive of adverse metal-related influences at Camp Lake. This was supported by water quality monitoring data that showed aqueous metal concentrations were below WQG and AEMP benchmarks at Camp Lake, and by sediment quality monitoring data that showed sediment metal concentrations were below SQG at Camp Lake with the exception of iron and manganese, which were also above SQG at Reference Lake 3.

Benthic invertebrate community compositional differences between the Camp Lake and Reference Lake 3 littoral stations did not appear to reflect differing food resources between lakes given an absence of significant differences in FFG (Table 3.6). Although the relative abundance of benthic invertebrate HPG differed significantly between Camp Lake and the reference lake, the magnitude of these differences were within a CES_{BIC} of ±2 SD_{REF} (Table 3.6) suggesting that the dissimilarity in the benthic invertebrate HPG proportions between lakes was within natural ranges of ecological variability. Notably, sediment particle

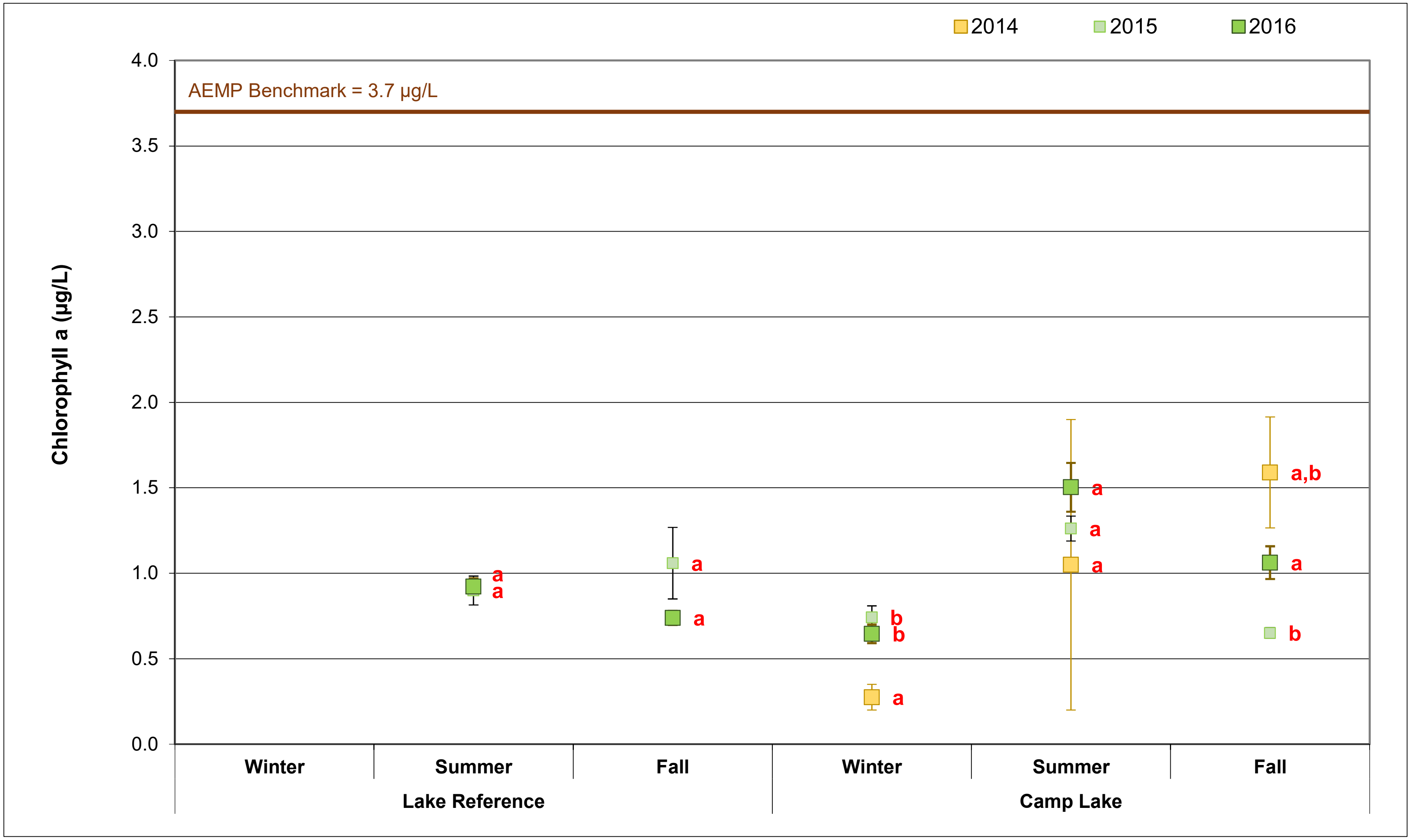


Figure 3.14: Chlorophyll a concentration seasonal comparison among 2014, 2015 and 2016 years (mean ± SE) at Camp Lake phytoplankton monitoring stations, Mary River Project CREMP. Data points with the same letter on the right do not differ significantly between years for the applicable season.

Table 3.6: Benthic invertebrate community statistical comparison results between Camp Lake (JLO) and Reference Lake 3 littoral stations, Mary River Project CREMP, August 2016.

Metric	Statistical Test Results					Summary Statistics					
	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Power	Magnitude of Difference [b] (No. of SD)	Area	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	No	0.728	α, δ, γ	-	-	Reference Lake 3	2,390	1,396	624	897	4,240
						Camp Lake Littoral	2,639	668	299	1,825	3,343
Richness (Number of Taxa)	No	0.151	γ	-	-	Reference Lake 3	12.2	1.1	0.5	11.0	14.0
						Camp Lake Littoral	15.8	3.3	1.5	10.0	18.0
Simpson's Evenness (E)	**Yes**	**0.008**	γ	**-**	**0.8**	Reference Lake 3	0.758	0.189	0.084	0.420	0.849
						Camp Lake Littoral	0.917	0.034	0.015	0.869	0.951
Bray-Curtis Index	Yes	0.001	α, δ, γ	1.000	2.8	Reference Lake 3	0.334	0.122	0.054	0.245	0.527
						Camp Lake Littoral	0.677	0.069	0.031	0.576	0.744
Nemata (%)	No	0.436	β, δ, γ	-	-	Reference Lake 3	4.0%	5.6%	2.5%	0.0%	13.5%
						Camp Lake Littoral	4.4%	4.8%	2.2%	1.0%	11.9%
Hydracarina (%)	No	0.182	β, ε	-	-	Reference Lake 3	3.6%	2.0%	0.9%	1.8%	6.7%
						Camp Lake Littoral	2.4%	3.1%	1.4%	0.0%	7.1%
Ostracoda (%)	Yes	0.008	γ	-	-2.6	Reference Lake 3	46.9%	17.5%	7.8%	37.8%	78.0%
						Camp Lake Littoral	1.8%	1.1%	0.5%	0.0%	2.8%
Chironomidae (%)	Yes	0.002	β, δ, γ	0.993	2.2	Reference Lake 3	45.4%	18.8%	8.4%	15.4%	59.2%
						Camp Lake Littoral	87.4%	7.0%	3.1%	78.6%	95.9%
Metal-Sensitive Chironomidae (%)	No	0.149	β, δ, γ	-	-	Reference Lake 3	19.3%	8.3%	3.7%	7.7%	28.1%
						Camp Lake Littoral	29.7%	11.8%	5.3%	16.2%	46.8%
Collector-Gatherers (%)	No	0.155	β, δ, γ	-	-	Reference Lake 3	75.0%	11.4%	5.1%	61.1%	89.7%
						Camp Lake Littoral	65.7%	7.8%	3.5%	52.4%	71.5%
Filterers (%)	No	0.103	β, δ, γ	-	-	Reference Lake 3	16.1%	8.4%	3.8%	7.0%	26.4%
						Camp Lake Littoral	25.0%	7.5%	3.3%	16.2%	36.5%
Clingers (%)	**Yes**	**0.093**	β, δ	**0.539**	**1.6**	Reference Lake 3	19.2%	7.6%	3.4%	8.8%	28.3%
						Camp Lake Littoral	31.5%	12.2%	5.4%	17.5%	45.3%
Sprawlers (%)	**Yes**	**0.026**	β, δ, γ	**0.803**	**-1.9**	Reference Lake 3	65.7%	12.1%	5.4%	57.2%	85.7%
						Camp Lake Littoral	42.7%	12.5%	5.6%	23.1%	54.8%
Burrowers (%)	**Yes**	**0.053**	β, δ, γ	**0.667**	**1.7**	Reference Lake 3	15.1%	6.2%	2.8%	5.5%	22.2%
						Camp Lake Littoral	25.6%	7.2%	3.2%	19.1%	35.1%

[a] Data analysis included: α - data untransformed; β - data logit transformed; ι - $\log_{10}$ transformed; δ - single factor ANOVA test conducted; ε - t-test assuming unequal variance; γ - ANOVA test validated using Mann Whitney U-test.

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

[c] Estimated minimum effect size detectable (±) calculated using variance as square root of MSE from ANOVA and alpha = beta = 0.10.

Highlighted values indicate significant differences between study areas based on ANOVA p-value less than 0.10 that were also outside of a Critical Effect Size of ±2 SD, suggesting an ecologically meaningful difference.

BOLD Bold text values indicate significant differences between study areas based on ANOVA p-value less than 0.10, but a Critical Effect Size within ±2 SD, suggesting the difference is not ecologically meaningful.

size did not differ significantly between the Camp Lake and reference lake littoral stations (Appendix Table D.8), suggesting that the differences in benthic invertebrate HPG between lakes were also not related to differing substrate texture as an artifact of the sampling program. Collectively, the lack of significant differences in FFG and ecologically meaningful differences in HPG suggested that benthic invertebrate community structural differences between Camp Lake and Reference Lake 3 littoral stations may have simply reflected natural variability between these lakes.

Temporal comparisons of the Camp Lake littoral habitat benthic invertebrate community indicated no significant differences in density, richness, dominant taxonomic group composition or FFG composition between the mine baseline (2013) and operational (2015, 2016) periods (Figure 3.15; Appendix Table F.19). Although Simpson's Evenness was significantly lower at Camp Lake littoral stations in 2015 than during either of the 2013 and 2016 studies (Figure 3.15), high Simpson's Evenness in 2016 and the absence of differences in any of the remaining key indices suggested that low evenness in 2015 did not reflect a mine-related influence. Thus, the study-to-study differences in Simpson's Evenness most likely reflected natural year-to-year variability in benthic invertebrate community features at Camp Lake. No consistent differences in benthic invertebrate community density, richness, Simpson's Evenness, FFG or HPG were indicated between Camp Lake and Reference Lake 3 littoral stations over the 2015 and 2016 studies (Figure 3.15; Appendix Table F.19). This supported the baseline data analyses in suggesting that the indicated differences for select metrics in 2015 and 2016 between the Camp Lake and reference lake benthic invertebrate communities were related to natural ecological variability rather than a mine-related influence.

3.2.5 Fish Population

3.2.5.1 Camp Lake Fish Community

The Camp Lake fish community included Arctic charr (*Salvelinus alpinus*) and ninespine stickleback (*Pungitius pungitius*), which mirrored the fish species composition observed at Reference Lake 3 in 2016 (Table 3.7). A higher density of Arctic charr was suggested at Camp Lake compared to Reference Lake 3 based on greater electrofishing total catch-per-unit-effort (CPUE) from shallow rocky nearshore habitat, and on greater gill netting CPUE from deeper littoral/profundal habitat at Camp Lake in 2016 (Table 3.7). In turn, this suggested higher fish productivity at Camp Lake compared to Reference Lake 3, corroborating the chlorophyll a results which indicated higher phytoplankton productivity at Camp Lake. Notably, although ninespine stickleback have been presumed to reside in low abundance at most lakes within the mine LSA (NSC 2014), the occurrence of ninespine stickleback at Camp Lake in 2016

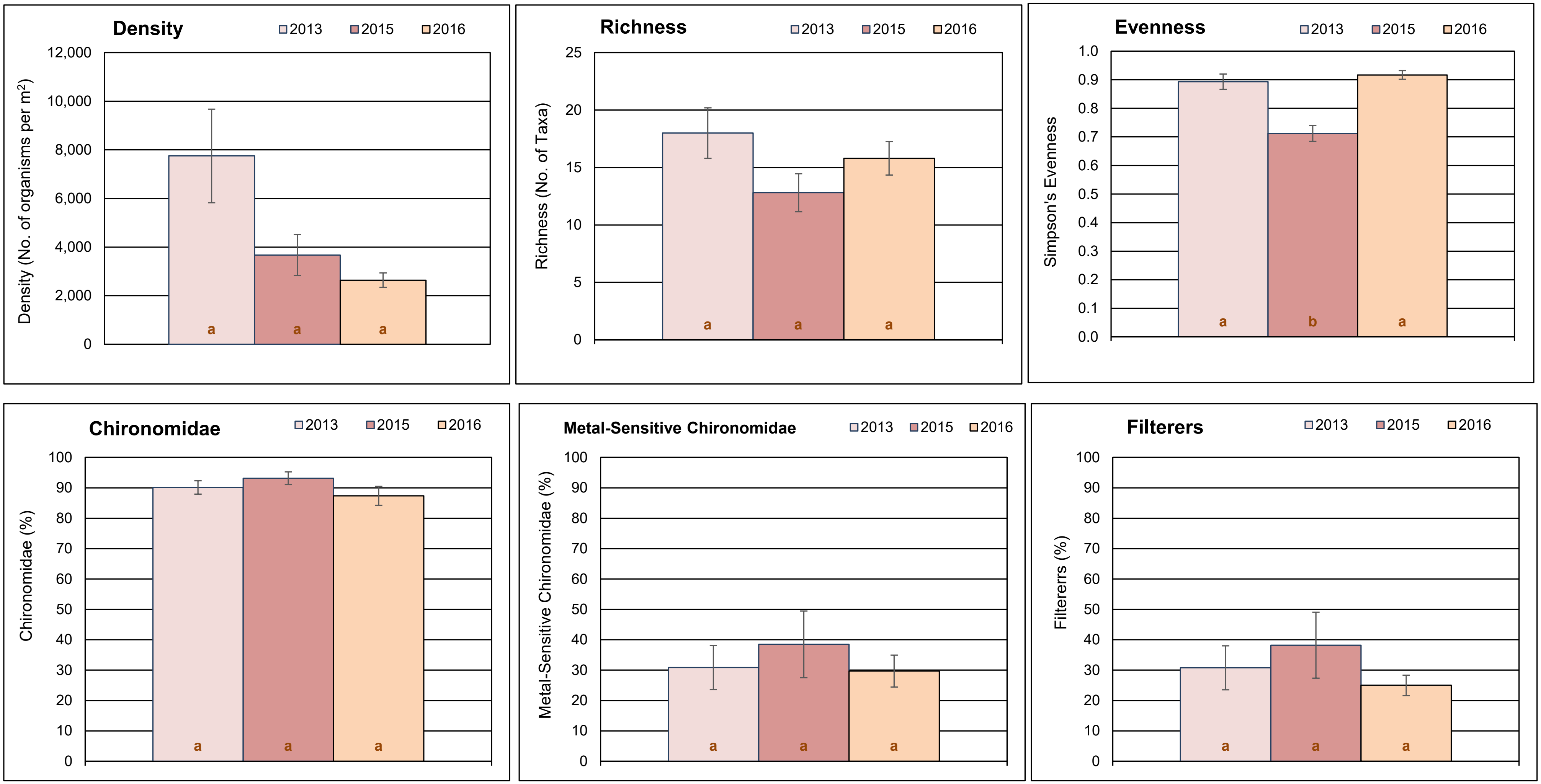


Figure 3.15: Comparison of key benthic invertebrate metrics (mean ± SE) at Camp Lake littoral stations between mine baseline (2007, 2013) and operational (2015, 2016) periods, Mary River Project CREMP, 2016. The same like-coloured letter inside bars indicate no significant difference between areas.

Table 3.7: Fish catch and community summary from backpack electrofishing and gill netting conducted at Camp Lake (JLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2016.

Lake	Method [a]		Arctic Charr	Nine-spine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	101	28	129	2
		CPUE	0.48	0.16	0.64	
	Gill netting	No. Caught	14	0	14	
		CPUE	0.15	0	0.15	
Camp Lake	Electrofishing	No. Caught	98	2	100	2
		CPUE	6.24	0.13	6.37	
	Gill netting	No. Caught	89	0	89	
		CPUE	5.43	0	5.43	

[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.

marks the first record of this species in the lake since the implementation of the Mary River Project AEMP studies. Similar abundance of ninespine stickleback along rocky nearshore habitat was suggested at both lakes based on comparable electrofishing CPUE for this species in 2016 (Table 3.7).

The Camp Lake 2016 electrofishing CPUE for Arctic charr was within the range of that observed during baseline (2005 - 2013) studies (Figure 3.16). This suggested that the abundance of Arctic charr at nearshore habitat of Camp Lake in 2016 was comparable to abundance observed prior to mine start-up. The Arctic charr CPUE for gill net collections was markedly higher in the 2016 study than in all previous baseline (2006 – 2008), mine construction (2014) and mine operational (2015) studies (Figure 3.16). Higher Arctic charr CPUE in 2016 may have reflected a combination of greater sampling efficiency due to experience gained from previous studies (e.g., selection of netting locations), changes in sampling gear dimensions relative to previous studies (i.e., focus on most efficient net mesh sizes as per Minnow 2016b), differences in the amount of gill netting effort applied during each study (see Minnow 2016a) and/or natural factors (e.g., weather conditions). Nevertheless, CPUE comparisons among studies suggested that the relative abundance of Arctic charr in Camp Lake had not likely changed substantially, and was not lower, in 2016 compared to the baseline and mine-construction periods.

3.2.5.2 Camp Lake Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Camp Lake nearshore Arctic charr population (i.e., fish captured by electrofishing) were assessed based on a control-impact analysis using 2016 data from Camp Lake and Reference Lake 3, as well as a before-after analysis using Camp Lake 2016 and baseline (2013) data. A total of 98 and 100 Arctic charr were captured at nearshore habitat of Camp Lake and Reference Lake 3, respectively, in August 2016, for the control-impact analysis. Young-of-the-year (YOY) were distinguished from older (non-YOY) age classes at a fork length cut-off of 3.9 and 5.1 cm for the Camp Lake and Reference Lake 3 data sets, respectively, based on the evaluation of length-frequency distributions coupled with supporting age determinations (Figure 3.17). Due to a low number of Arctic charr YOY captured at Camp Lake (i.e., 4), fish population comparisons were conducted using only non-YOY individuals, where applicable, to limit confounding influences of naturally differing weight-at-length relationships between YOY and non-YOY individuals on data interpretation.

The length-frequency distribution for the nearshore Arctic charr differed significantly between Camp Lake and Reference Lake 3 (Table 3.8), reflecting the occurrence of very few YOY and

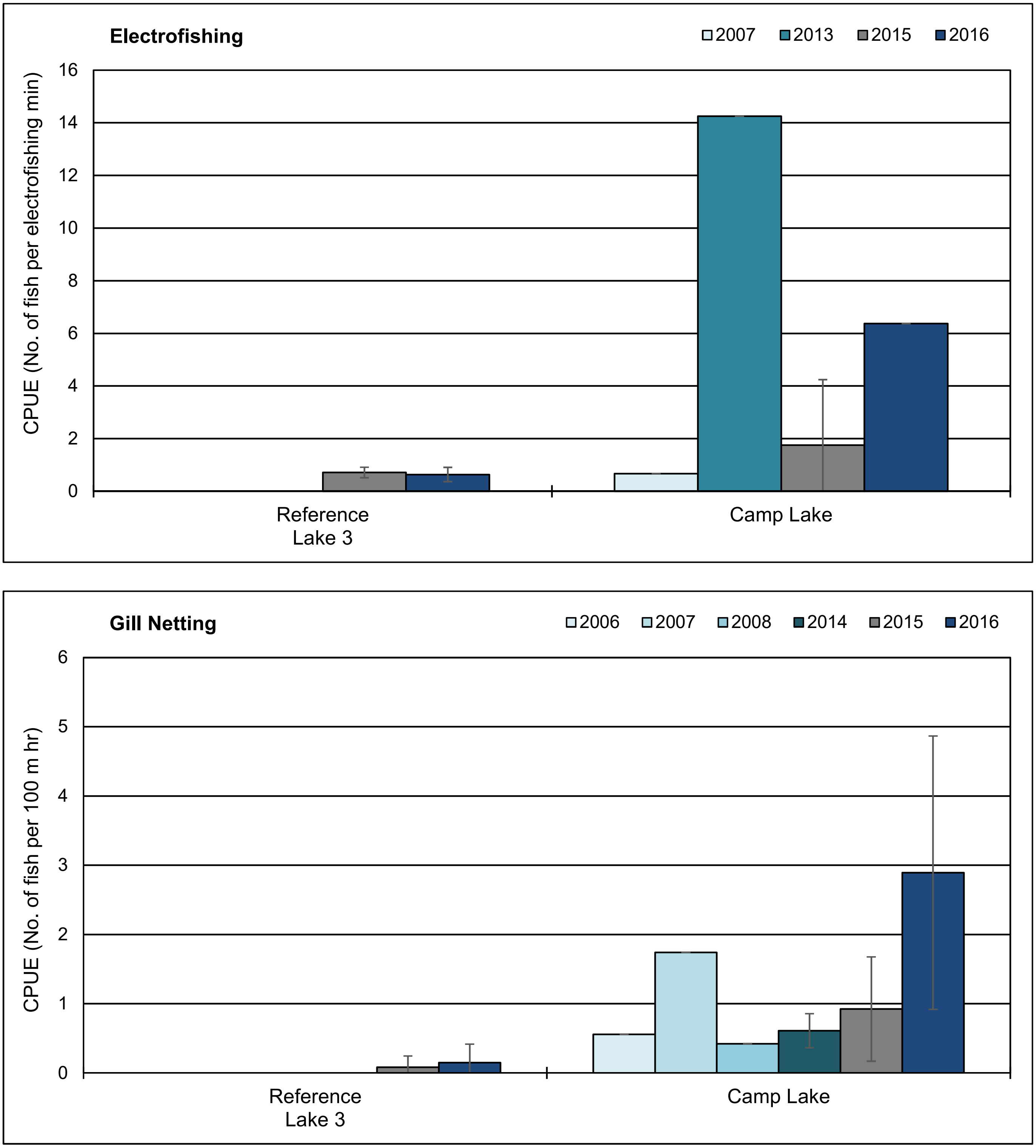


Figure 3.16: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic charr captured by backpack electrofishing and gill netting at Camp Lake (JLO) for baseline (2006, 2007, 2008, 2013), mine construction (2014) and operational (2015, 2016) periods during fall, Mary River Project CREMP.

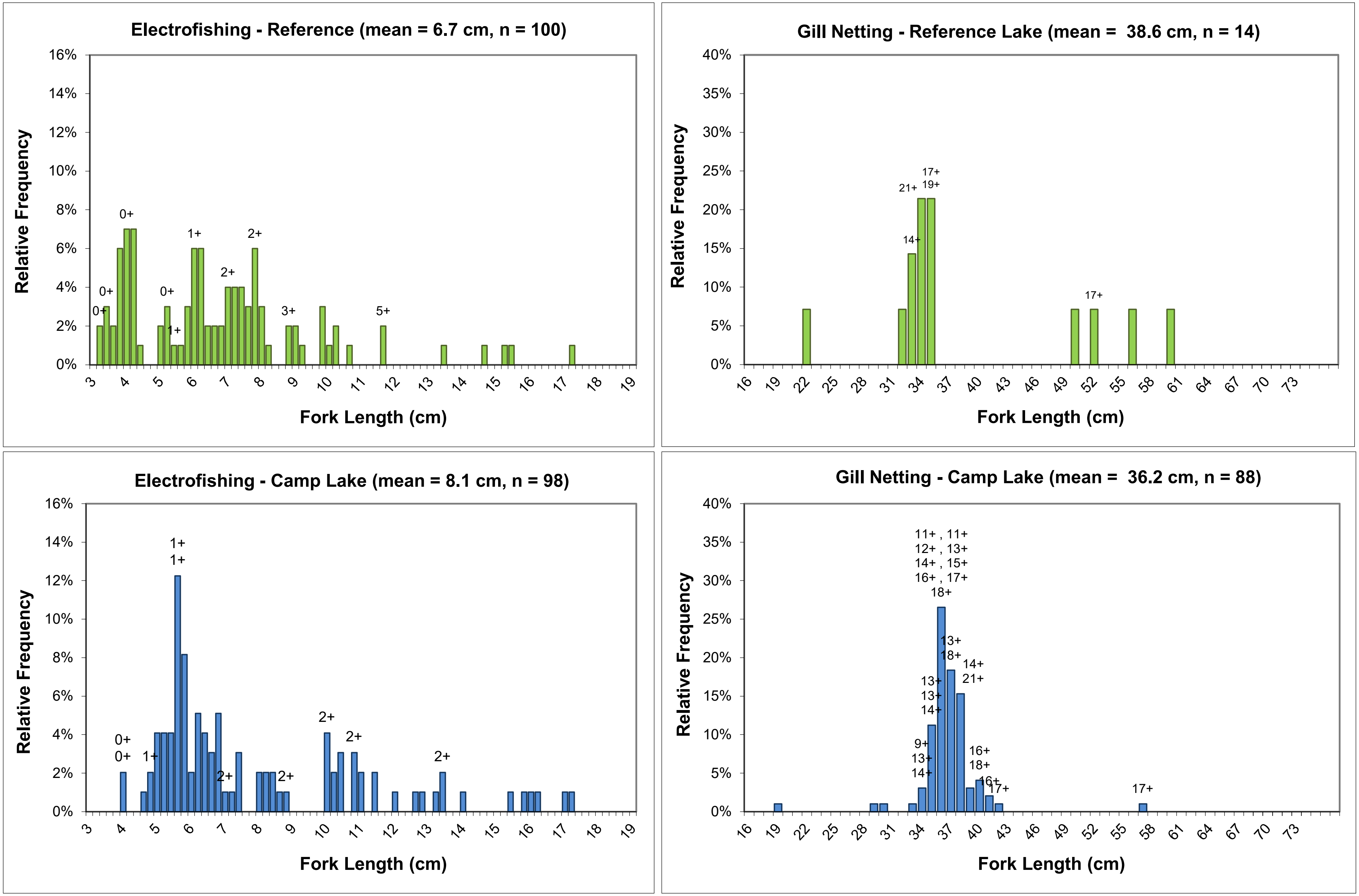


Figure 3.17: Length-frequency distributions for Arctic charr captured by backpack electrofishing and gill netting at Camp Lake (JLO) and Reference Lake 3 (REF3), August 2016, Mary River Project CREMP. Fish ages are shown above the bars, where available.

Table 3.8: Summary of statistical results for Arctic charr population comparisons between Camp Lake and Reference Lake 3 for the mine operational period (2015, 2016) and between Camp Lake mine-operational and baseline period data for fish captured by electrofishing and gill netting methods, Mary River Project CREMP, August 2016. Values in parentheses indicate direction and magnitude of any significant differences.

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed?			
			versus Reference Lake 3		versus Camp Lake baseline period data [b]	
			2015	2016	2015	2016
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	-	-
	Energy Use	Size (mean weight)	**Yes (+176%)**	No	**Yes (-42%)**	**Yes (-71%)**
		Size (mean fork length)	**Yes (+41%)**	No	**Yes (-15%)**	**Yes (-32%)**
		Growth (weight-at-age)	**Yes (+154%)**	No	-	-
		Growth (fork length-at-age)	**Yes (+36%)**	**Yes (+18%)**	-	-
	Energy Storage	Condition (body weight-at-fork length)	No	**Yes (-6%)**	**Yes (-6%)**	**Yes (-10%)**
Littoral/Profundal Gill Netting [a]	Survival	Length Frequency Distribution	-	-	**Yes**	**Yes**
		Age	-	-	**Yes (+48%)**	**Yes (+58%)**
	Energy Use	Size (mean weight)	-	-	No	No
		Size (mean fork length)	-	-	**Yes (+6%)**	No
		Growth (weight-at-age)	-	-	No	**Yes (nc)**
		Growth (fork length-at-age)	-	-	No	**Yes (nc)**
	Energy Storage	Condition (body weight-at-fork length)	-	-	No	**Yes (-3%)**

[a] Due to low catches of Arctic charr at Reference Lake 3 in 2015 and 2016, no comparison of fish health was possible for gill netted fish.
[b] Baseline period data included 2013 nearshore electrofishing data and 2006, 2008 and 2013 littoral/profundal gill netting data. nc = non-calculable magnitude.

greater numbers of larger individuals captured at Camp Lake. Mean fresh body weight and fork length of non-YOY Arctic charr captured at the Camp Lake nearshore did not differ significantly from those captured at the reference lake nearshore (Table 3.8; Appendix Table G.11). Non-YOY Arctic charr captured at the Camp Lake nearshore exhibited significantly faster length-based growth (i.e., length-at-age) compared to non-YOY captured at Reference Lake 3 (Table 3.8; Figure 3.17; Appendix Table G.11). However, the magnitude of difference in growth was within an ecologically meaningful Critical Effect Size (CES) of ±25% (referred to herein as CES_G; Table 3.8), suggesting that the differences in non-YOY Arctic charr energy use between lakes was within the range of variability expected to occur naturally between waterbodies uninfluenced by human activity. Notably, sample sizes used for growth comparisons were small (i.e., ten for each study area; Appendix Table G.11), resulting in some uncertainty regarding the strength of the indicated growth relationships. Non-YOY Arctic charr condition (i.e., weight-at-length relationship) was significantly lower at Camp Lake than at the reference lake (Table 3.8; Appendix Table G.11). Similar to the growth analysis, the magnitude of difference in condition of non-YOY Arctic charr between lakes was within a CES of ±10% (referred to herein as CES_C; Table 3.8), suggesting that the difference in non-YOY Arctic charr energy storage between lakes was not ecologically meaningful. Collectively, the 2016 fish health assessment results suggested only minor differences in nearshore Arctic charr energy use and storage between Camp Lake and Reference Lake 3 populations, the implications of which were not likely to be ecologically meaningful.

Temporal comparisons of the Camp Lake nearshore Arctic charr data indicated significantly different length-frequency distribution between the 2016 mine operational study and the 2013 baseline study (Table 3.8). In addition, Arctic charr captured at the nearshore of Camp Lake in 2016 were significantly lighter, shorter and of lower condition than those captured during the 2013 baseline study (Table 3.8; Appendix Table G.12). Similar differences in nearshore Arctic charr size and condition were demonstrated between the 2015 mine operational study and the 2013 baseline data (Table 3.8). However, the magnitude of difference in condition between the individual mine operational studies (i.e., 2015 and 2016) and the 2013 baseline data was within a CES_C of ±10% (Table 3.8; Appendix Table G.12), suggesting that the differences were within the natural range of variability expected between lakes uninfluenced by human activity.

Littoral/Profundal Arctic Charr

Mine-related influences on the Camp Lake littoral/profundal Arctic charr population (i.e., fish captured by gill netting) was assessed using a before-after analysis of Camp Lake 2016 versus baseline (combined 2006, 2007, 2008 and 2013) data. Similar to the 2015 CREMP, despite a total of 87 Arctic charr captured at littoral/profundal areas of Camp Lake and application of similar fishing effort, the Arctic charr sample size was small (i.e., 14) at Reference Lake 3 in August 2016, precluding a control-impact analysis for the determination of mine-related effects. Biological information collected from Arctic charr mortalities encountered during the 2016 Camp Lake littoral/profundal sampling suggested that 67% of the population was represented by non-spawners of reproductive age (referred to simply as non-spawners herein; Appendix Table G.15). The average age, length and weight of non-spawners was comparable to that of female spawners (Appendix Table G.15) indicating that, typical of high Arctic systems, individual Arctic charr do not spawn yearly at Camp Lake. Liver somatic index (LSI) was significantly lower in non-spawners than female spawners (ANOVA; $p = 0.004$), suggesting that lower energy was available for gamete development in the non-spawners. Internal body cavity parasites were present in almost all of the Arctic charr incidental mortalities (Appendix Table G.15), potentially contributing to biennial or longer frequency between spawning events for Arctic charr in the mine LSA lakes as a result of lower energy applied towards gamete production stemming from the parasitic infection. High incidence rates of internal parasites in Arctic charr of the Mary River Project mine area lakes was noted in baseline studies (NSC 2014, 2015a) and the 2015 CREMP (Minnow 2016a).

Temporal comparisons of Arctic charr data collected from Camp Lake littoral/profundal areas indicated significantly different length-frequency distribution of Arctic charr in 2016 compared to the combined baseline data set (i.e., 2006, 2007, 2008 and 2013 studies; Table 3.8). The differences in length-frequency distributions were consistent with the capture of significantly older Arctic charr at Camp Lake in 2016 compared to the baseline period (Table 3.8; Appendix Table G.16). No significant differences in Arctic charr fresh body weight or fork length were demonstrated between the 2016 and the baseline period. Arctic charr of spawning size showed significant differences in growth between 2016 and the baseline period, although the magnitude and direction of difference was non-calculable due to a significant interaction result (Table 3.8). Finally, significantly lower condition was indicated for Arctic charr of spawning size at Camp Lake between 2016 and the baseline period, but the magnitude of this difference was very small and within the CES_C of ±10% (Table 3.8; Appendix Table G.16), suggesting that this difference was not ecologically meaningful. Although length frequency distribution and average age of Arctic charr captured at Camp Lake in 2015 and 2016 consistently differed from those of the baseline period, no consistent differences in size, growth or condition were demonstrated between individual mine operational years and the baseline period.

3.3 Synthesis of Mine-Related Influences within the Camp Lake System

3.3.1 Camp Lake Tributaries

3.3.1.1 Camp Lake Tributary 1

Mine-related effects on water quality of the CLT1 north branch in 2016 included slightly elevated nitrate and copper concentrations compared to 2016 reference creek data and to 2005 - 2013 baseline data. Despite copper concentrations above WQG, chlorophyll a concentrations (a surrogate for phytoplankton abundance) at the CLT1 north branch were comparable to those of the reference creek stations in 2016, and to those during the baseline period, all of which were well below the AEMP benchmark and suggested oligotrophic conditions typical of Arctic watercourses. In addition, despite some differences in benthic invertebrate community composition between the CLT1 north branch and the reference creek in 2016, these differences appeared to be related to naturally differing amounts of in-stream vegetation between watercourses. This was supported by the absence of differences in relative abundance of metal-sensitive taxa between the CLT1 north branch and Unnamed Reference Creek in 2016, and for CLT1 north branch data collected in 2016 compared to 2005 - 2013 baseline data. Moreover, temporal comparisons that indicated no consistent differences in primary benthic invertebrate community endpoints (i.e., density, richness, Simpson's Evenness) or relative abundance of dominant invertebrate groups and FFG in 2016 compared to baseline data. Therefore, similar to the findings of the 2015 CREMP, no adverse effects to biota of the CLT1 north branch were suggested by the 2016 study.

At the CLT1 upper main stem (Station L2-03), mine-related influences on water quality were evident as elevated conductivity, hardness and concentrations of nitrate, sulphate and several metals including iron, manganese, molybdenum, sodium, strontium and uranium in 2016 compared to 2016 reference creek station data and to 2005 - 2013 baseline data. As identified during the 2015 CREMP, quarrying activity at the QMR2 pit was likely a key source for parameters elevated at the CLT1 main stem stations in 2016. Despite evidence of continued mine-related influence on water quality of the CLT1 upper main stem in 2016, parameter concentrations were below applicable WQG and site-specific AEMP benchmarks with the exception of iron and uranium at the upper main stem. However, elevated chlorophyll a concentrations and significantly higher benthic invertebrate density, richness and relative abundance of metal-sensitive taxa at the CLT1 upper main stem compared to Unnamed Reference Creek in 2016 suggested that concentrations of iron, uranium and other metals were not highly bioavailable at the CLT1 upper main stem. In fact, biological data collected at the CLT1 upper main stem in 2016 suggested a biological enrichment effect related to elevated nutrient concentrations. Temporal comparisons suggested that chlorophyll a concentrations at the CLT1 upper main stem were higher following commencement of mine operations than during the baseline period, but no significant differences in benthic invertebrate community primary endpoints, key dominant invertebrate groups, or FFG were evident between 2016 and baseline data collected in 2007. In turn, this suggested that mine-related enrichment effects at the CLT1 upper main stem, if any, were relatively minor.

At the CLT1 lower main stem (i.e., stations L1-01, L1-05 and L1-09), natural dilution of the main stem from the north branch resulted in only conductivity and aqueous concentrations of nitrate, chloride, manganese and strontium being elevated compared to concentrations observed at reference creek stations in 2016. Concentrations of all parameters were below applicable WQG and AEMP benchmarks at the CLT1 lower main stem in 2016. However, temporal comparisons suggested increased conductivity, hardness and concentrations of nitrate, sulphate and metals including iron, manganese, molybdenum, sodium, strontium and uranium during the 2015/2016 mine operation period compared to the 2005 - 2013 baseline period. Chlorophyll a concentrations at the CLT1 lower main stem in 2016 were comparable to those of the reference creek stations in 2016, and those observed during the baseline period. In all cases, chlorophyll a concentrations were well below the AEMP benchmark and suggested oligotrophic conditions typical of Arctic watercourses. No significant, ecologically meaningful, differences in benthic invertebrate community primary endpoints or relative abundance of metal-sensitive taxa were indicated at the CLT1 lower main stem between mine operation (2015, 2016) and baseline (2007, 2011) studies. Although benthic invertebrate community composition differed significantly between the CLT1 lower main stem and Unnamed Reference Creek communities in 2016, similar to the results of the 2015 CREMP, this appeared to be related to natural differences in dominant food source between the mine-exposed and reference study areas. No consistent types and/or direction of differences in the relative abundance of dominant groups or FFG were indicated between 2016 and the baseline data at the CLT1 lower main stem. Overall, no adverse mine-related effects to biota of the CLT1 lower main stem were suggested in 2016 based on comparison to Unnamed Reference Creek and baseline data.

3.3.1.2 Camp Lake Tributary 2

Mine-related effects on water quality of CLT2 in 2016 potentially included slightly elevated conductivity, sulphate and zinc concentrations based on comparisons to 2016 reference creek station data. However, water chemistry at CLT2 in 2016 was comparable to the 2005 - 2013 baseline data, suggesting that natural regional variability in water chemistry among lotic environments may have accounted for seemingly elevated concentrations of the aforementioned parameters at CLT2 in 2016 compared to the reference creek stations. Aqueous concentrations of all parameters were consistently well below established WQG and AEMP benchmarks at CLT2 during the 2015 and 2016 mine operation period. Chlorophyll a concentrations at CLT2 were consistently within the range observed among the reference creek stations in 2016 and, in addition to being well below the AEMP benchmark, were also within the range observed at CLT2 during baseline studies. Although the benthic invertebrate community of CLT2 exhibited significantly lower density and significantly different composition than Unnamed Reference Creek in 2016, these differences appeared to be related to natural habitat differences between watercourses. This was supported by no significant differences in richness, Simpson's Evenness and relative abundance of dominant invertebrate groups, FFG and HPG between areas located upstream and downstream of the mine tote road. In addition, no significant differences in benthic invertebrate community endpoints occurred between 2016 and the 2007 baseline data at either CLT2 study area with the exception of Simpson's Evenness, which was higher in 2016 and thus not consistent with a typical adverse mine-related response. Similar to the findings of the 2015 CREMP, the occurrence of few significant differences in benthic invertebrate community endpoints upstream and downstream of the mine tote road in 2016, and between the 2016 mine operational and 2007 baseline data, suggested no adverse mine-related influences to the benthic invertebrate community of CLT2.

3.3.2 Camp Lake

Mine-related influences on water quality of Camp Lake in 2016 included slightly elevated manganese concentrations compared to the reference lake, as well as slightly higher conductivity and concentrations of chloride, molybdenum, sodium, strontium and uranium compared to 2005 - 2013 baseline data. However, in all cases, parameter concentrations at Camp Lake were consistently well below WQG and AEMP benchmarks in 2015 and 2016. Sediment arsenic and manganese concentrations were elevated at Camp Lake littoral stations compared to the reference lake in 2016 and, together with molybdenum, were also elevated compared to concentrations during the baseline period. However, no metals were elevated in sediment at Camp Lake profundal stations compared to the reference lake in 2016. Although some changes in average sediment metal concentrations were suggested between 2016 and the baseline period at profundal stations, these changes may have reflected changes to the number of profundal sediment quality monitoring stations sampled between 2016 and the previous studies (i.e., three versus nine, respectively). Phosphorus was the only parameter observed at concentrations above SQG in littoral and profundal sediment of Camp Lake that was not also above applicable SQG at the reference lake. Overall, recent mine operations appeared to contribute to higher manganese and molybdenum concentrations in water and littoral sediment of Camp Lake, as well as higher chloride, sodium, strontium and uranium in water and potentially higher arsenic in littoral sediment, but concentrations of these parameters remained below applicable guidelines and AEMP benchmarks. In turn, this suggested a low potential for adverse effects to biota of Camp Lake.

Camp Lake chlorophyll a concentrations were significantly higher than at the reference lake in 2016 suggesting greater primary production at Camp Lake. However, Camp Lake chlorophyll a concentrations remained well below the AEMP benchmark during all seasonal sampling events in 2016, and suggested oligotrophic conditions typical of Arctic waterbodies. No significant differences in chlorophyll a concentrations were indicated among the mine construction (2014) and operational (2015, 2016) periods, suggesting no changes in the trophic status of Camp Lake since mine operations commenced at the Mary River Project. Benthic invertebrate community data collected at littoral habitat of Camp Lake in 2016 indicated significantly greater evenness and similar density, richness and relative abundance of metal sensitive taxa, FFG and HPG compared to the reference lake. In addition, no significant differences in benthic invertebrate community primary and FFG metrics were observed between 2016 and the 2013 baseline data for Camp Lake littoral stations. Analysis of Camp Lake Arctic charr populations suggested greater fish abundance compared to the reference lake in 2016, but similar numbers of Arctic charr in 2016 relative to the Camp Lake baseline studies. No significant, ecologically meaningful, differences in Arctic charr condition were indicated between Camp Lake and the reference lake in 2016, nor between Camp Lake Arctic charr collected in 2016 compared to the baseline period, for nearshore and littoral/profundal Arctic charr populations. Collectively, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Camp Lake in the second year of mine operation at the Mary River Project.

4.0 SHEARDOWN LAKE SYSTEM

4.1 Sheardown Lake Tributaries (SDLT1, 9 and 12)

4.1.1 Water Quality

Sheardown Lake Tributary 1 (SDLT1) dissolved oxygen (DO) concentrations were consistently at or above saturation in spring, summer and fall monitoring events in 2016, and did not differ significantly from Unnamed Reference Creek at the time of biological sampling in August 2016 (Figure 4.1; Appendix Tables C.1 – C.3). Although DO saturation was slightly lower at Sheardown Lake Tributary 9 and 12 (SDLT9 and SDLT12, respectively) than at SDLT1 and Unnamed Reference Creek during August 2016 sampling, DO saturation at all of the Sheardown Lake tributaries was well above the WQG minimum limit for cold-water biota (i.e., 54%) during all seasonal sampling events (Figure 4.1; Appendix Tables C.1 – C.3). *In-situ* pH was significantly higher at SDLT1 compared to Unnamed Reference Creek, whereas pH at SDLT9 and SDLT12 did not differ significantly from reference conditions during the fall sampling event in 2016. Despite minor differences in pH among the Sheardown Lake tributaries, pH was consistently within WQG limits at each mine-exposed tributary and thus slight dissimilarity in pH among areas was unlikely to be ecologically meaningful. Conductivity at each of the Sheardown Lake tributaries was significantly higher than at Unnamed Reference Creek during the August 2016 biological sampling (Figure 4.1; Appendix Table C.29). Because conductivity often serves as an indication of mine-associated influences on water quality (e.g., Environment Canada 2012), these observations suggested a mine-related influence on water quality of the SDLT1, SDLT9 and SDLT12 watercourses.

Sheardown Lake Tributary 1 is the only tributary of the Sheardown Lake system at which routine water quality monitoring is conducted, with one monitoring station established in each of the upper and lower reaches of the tributary (i.e., Stations D1-05 and D1-00, respectively; Figure 2.2). Nitrate, sulphate and molybdenum concentrations were moderately to highly elevated (i.e., 5- to 10-fold, and ≥10-fold, respectively) at both SDLT1 stations compared to reference creek station mean concentrations at the time of fall sampling (Table 4.1). In addition, slightly elevated (i.e., 3- to 5-fold higher) concentrations of cadmium and copper were observed at upper SDLT1, and slightly elevated concentrations of chloride and manganese were observed at lower SDLT1, compared to reference creek stations at the time of fall sampling in 2016 (Table 4.1). Along with the aforementioned parameters, hardness, alkalinity and concentrations of TDS, potassium, sodium, strontium and uranium were generally elevated (i.e., ≥3-fold higher) in spring and/or summer at one or both SDLT1 monitoring stations compared to reference creek station mean values for each respective seasonal

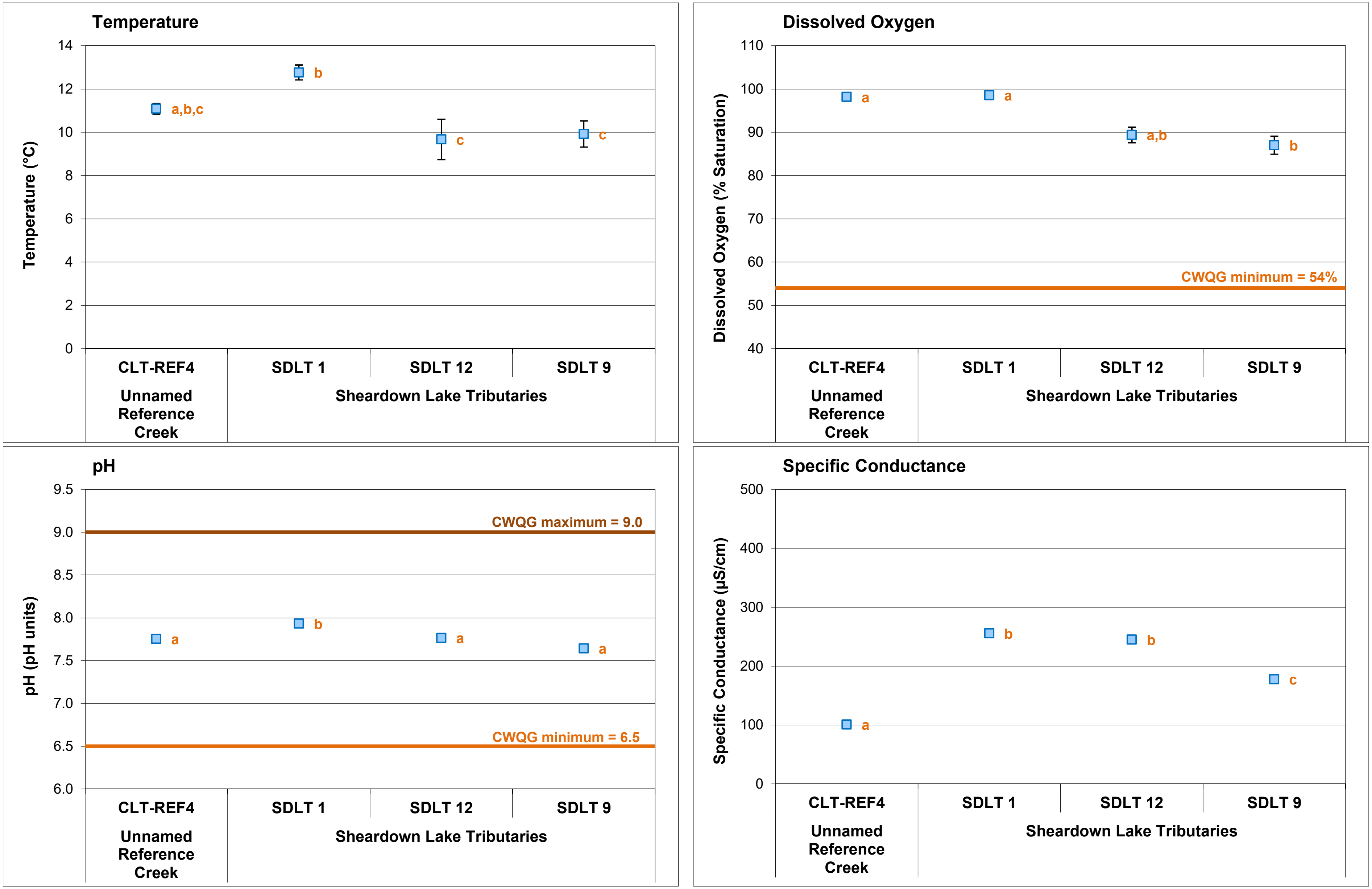


Figure 4.1: Comparison of *in-situ* water quality variables (mean ± SE; n = 5 except for SDLT 12, where n = 3) measured at the Sheardown Lake Tributaries (SDLT) and creek reference stations, Mary River Project CREMP, August 2016. The same letters next to data points indicate study area values do not differ significantly.

Table 4.1: Water chemistry at Sheardown Lake Tributary 1 (SDLT1) monitoring stations, Mary River Project CREMP, August 2016.

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Lotic Reference Average (n = 4)	Sheardown Lake Tributary 1	
						D1-05 (Upper)	D1-00 (Lower)
					Fall 2016	19-Aug-2016	19-Aug-2016
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	125	232	308
	pH (lab)	pH	6.5 - 9.0	-	7.99	7.85	8.08
	Hardness (as $CaCO_3$)	mg/L	-	-	57.75	108	144
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	65	118	166
	Turbidity	NTU	-	-	1.10	0.27	0.65
	Alkalinity (as $CaCO_3$)	mg/L	-	-	57	83	114
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	<0.020	0.030	<0.020
	Nitrate	mg/L	13	13	0.021	0.733	0.946
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.3	2.7	3.1
	Total Organic Carbon	mg/L	-	-	1.5	2.8	3.2
	Total Phosphorus	mg/L	0.020$^{\alpha}$	-	0.0059	0.0110	0.0032
	Phenols	mg/L	0.004$^{\alpha}$	-	0.0055	0.0110	0.0042
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	2.4975	6.41	9.47
	Sulphate (SO_4)	mg/L	218$^{\beta}$	218	4.39	22.6	26.8
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0578	0.0082	0.0138
	Antimony (Sb)	mg/L	0.020$^{\alpha}$	-	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00779	0.0115	0.0170
	Beryllium (Be)	mg/L	0.011$^{\alpha}$	-	<0.00040	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.0003875	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	0.012	0.012
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	0.000037	0.000011
	Calcium (Ca)	mg/L	-	-	12.3	19.5	27.9
	Chromium (Cr)	mg/L	0.0089	0.00856	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009$^{\alpha}$	0.004	<0.00010	<0.00010	0.00011
	Copper (Cu)	mg/L	0.002	0.0022	0.0010	***0.00310***	***0.00222***
	Iron (Fe)	mg/L	0.30	0.326	0.051	<0.030	0.098
	Lead (Pb)	mg/L	0.001	0.001	0.000096	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	0.0013	0.0018
	Magnesium (Mg)	mg/L	-	-	6.77	14.1	18.9
	Manganese (Mn)	mg/L	0.935$^{\beta}$	-	0.00086	0.000436	0.00559
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.000380	0.00325	0.00243
	Nickel (Ni)	mg/L	0.025	0.025	0.00056	0.00114	0.00146
	Potassium (K)	mg/L	-	-	0.84	2.33	2.41
	Selenium (Se)	mg/L	0.001	-	<0.0007625	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.95	1.36	1.59
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000020	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	1.830	2.98	3.88
	Strontium (Sr)	mg/L	-	-	0.01240	0.0130	0.0169
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0000775	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	0.00799	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00366	0.00654	0.00532
	Vanadium (V)	mg/L	0.006$^{\alpha}$	0.006	<0.000875	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030

[a] Canadian Water Quality Guideline (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). 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 the Camp Lake Tributaries.

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

BOLD Indicates parameter concentration above the AEMP benchmark.

sampling event (Appendix Table C.32). Despite elevation of these parameters at the SDLT1 stations compared to reference conditions, copper was the only parameter present at concentrations greater than respective WQG or AEMP benchmarks at either of the SDLT1 monitoring stations in 2016[7] (Table 4.1; Appendix Table C.30).

Temporal comparisons of SDLT1 water chemistry data indicated that, of the parameters shown to be elevated above average reference conditions, only nitrate and sulphate concentrations were slightly elevated (i.e., 3- to 5-fold higher) at upper and lower SDLT1 in 2016 compared to respective baseline period conditions (Figure 4.2; Appendix Table C.32 and Figure C.9). The SDLT1 concentrations of these parameters, and uranium, were elevated compared to baseline conditions in 2015 as well, suggesting a mine-related source of these metals since the initiation of mine operations at the Mary River Project.

4.1.2 Phytoplankton

Phytoplankton (chlorophyll a) monitoring is conducted only at SDLT1 within the Sheardown Lake system as part of the Mary River Project CREMP. Chlorophyll a concentrations at SDLT1 were lower at upstream-most Station D1-05 compared to near the creek mouth (Station D1- 00), during the spring and summer 2016 sampling events, but not during the fall (Figure 4.3). With the exception of markedly higher chlorophyll a concentrations near the SDLT1 creek mouth compared to reference conditions in summer, chlorophyll a concentrations were generally within the range shown among the reference creek stations and were well below the AEMP benchmark of 3.7 μg/L during all 2016 seasonal sampling events. Higher chlorophyll a concentrations observed near the mouth of SDLT1 may have reflected the occurrence of elevated nutrient concentrations, and aqueous nitrate concentrations specifically, shown at SDLT1 in 2016 (Section 4.1.1). Similar to the reference creek stations and Camp Lake tributary systems, chlorophyll a concentrations at SDLT1 were suggestive of 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). Relatively low chlorophyll a concentrations at SDLT1 stations in 2016 were consistent with an oligotrophic WQG categorization based on aqueous phosphorus concentrations near or below 10 μg/L (Table 4.1; Appendix Table C.30).

Temporal comparisons indicated that chlorophyll a concentrations at SDLT1 stations in 2016 were comparable to concentrations measured during the baseline period (Figure 4.4). In addition, no consistent directional changes in chlorophyll a concentrations were shown at the

[7] Refer to footnote 2 (page 23) and Appendix B regarding phenol concentrations above WQG at the mine-exposed and reference areas of the Mary River Project LSA waterbodies.

Figure 4.2: Temporal comparison of water chemistry at Sheardown Lake Tributaries (SDLT) for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during fall. Values represent mean ± SD. Creek reference stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4). Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are adopted from the Camp Lake Tributaries.

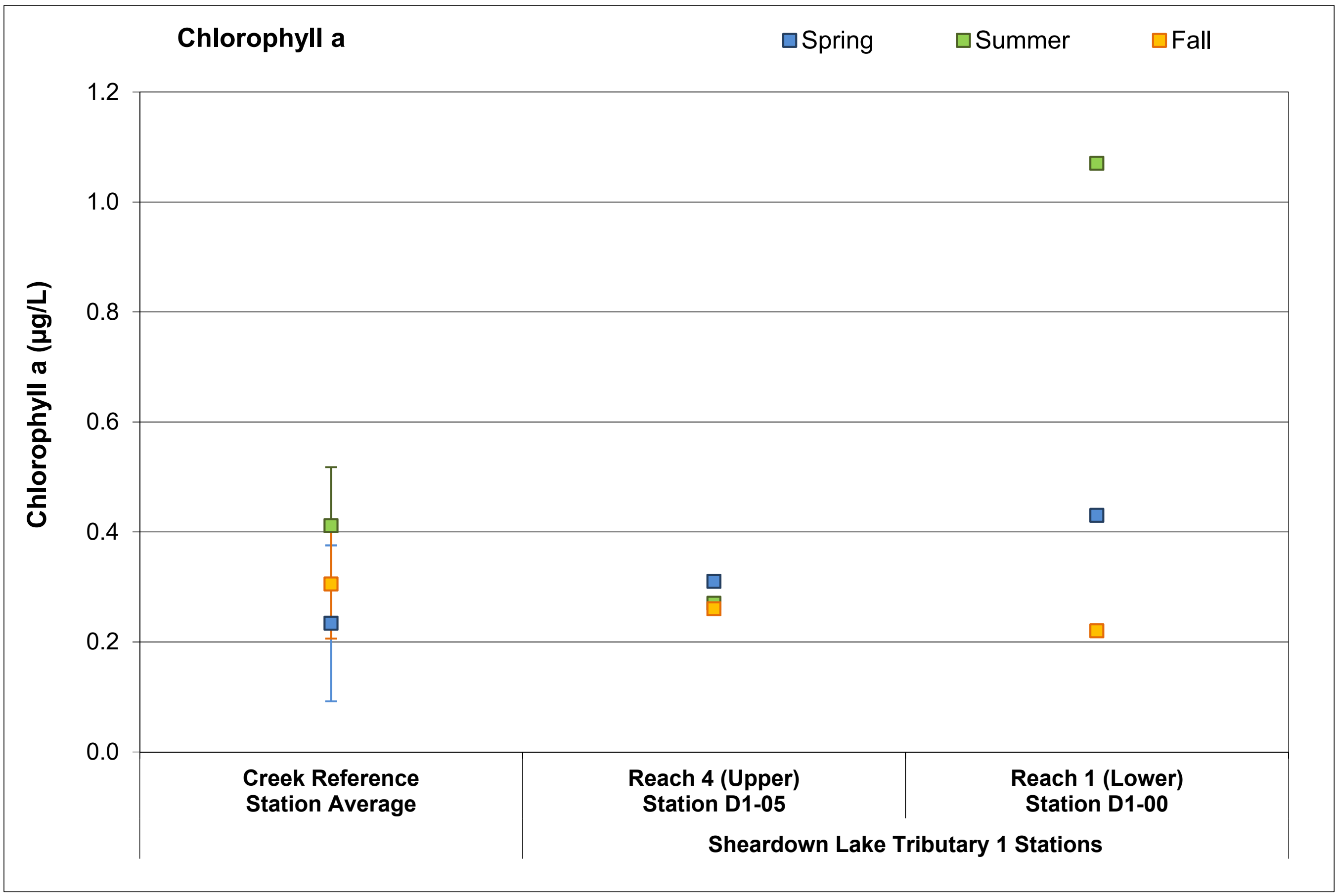


Figure 4.3: Chlorophyll a concentrations at Sheardown Lake Tributary 1 phytoplankton monitoring stations, Mary River Project CREMP, 2016. Creek reference includes the CLT-REF and MRY-REF series stations (mean ± SD; n = 4).

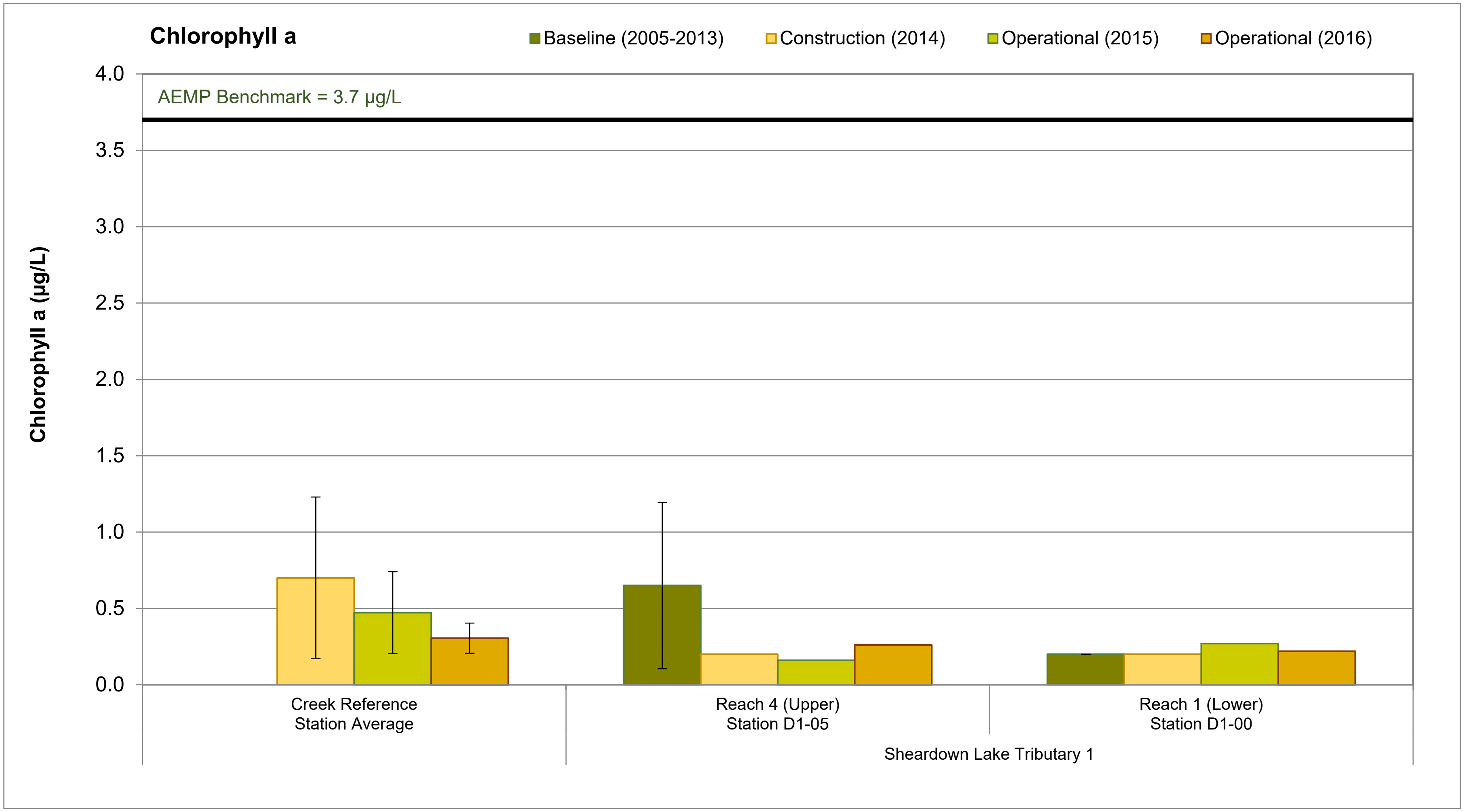


Figure 4.4: Temporal comparison of chlorophyll a concentrations at Sheardown Lake Tributary 1 for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods in the fall, Mary River Project CREMP. The creek reference includes the CLT-REF and MRY-REF series stations (mean ± SD; n = 4).

SDLT1 stations over the mine baseline (2005 – 2013), construction (2014), and operational (2015, 2016) periods (Figure 4.4). These data suggested no adverse mine-related influences to phytoplankton productivity at SDLT1 over the initial two years of mine operation.

4.1.3 Benthic Invertebrate Community

Sheardown Lake Tributary 1 (SDLT1)

The benthic invertebrate community at the lower reach of Sheardown Lake Tributary 1 (SDLT1 R1), near the outlet to Sheardown Lake NW, exhibited significantly lower richness and significant differences in composition (as indicated by Bray-Curtis Index) compared to Unnnamed Reference Creek in 2016 (Figure 4.5; Appendix Table F.25). Although the relative abundances of Hydracarina (water mites) and metal-sensitive chironomids were significantly lower and higher, respectively, at lower SDLT1 than at Unnamed Reference Creek, the magnitude of these differences was within a CES_{BIC} of ± 2 SD_{REF} (Figure 4.5; Appendix Table F.25), suggesting that these differences were not ecologically meaningful. A higher relative abundance of metal-sensitive chironomids at lower SDLT1 also suggested that the differences in community composition compared to Unnamed Reference Creek were unrelated to metal concentrations, which was consistent with concentrations of most metals below WQG at SDLT1 in 2016 (see Appendix Table C.30). A significantly higher relative abundance of FFG filterers (Appendix Table F.25), which were represented predominantly by metal-sensitive chironomids, suggested that higher nitrate (i.e., nutrient) concentrations contributed to higher abundance of phytoplankton (i.e., chlorophyll a) and a consequent shift in benthic food resources at SDLT1 compared to reference conditions. Notably, the occurrence of significantly higher relative abundance of HPG burrowers was consistent with significantly greater substrate embeddedness at SDLT1 than at Unnamed Reference Creek (Appendix Tables F.22 and F.25). Greater substrate embeddedness at SDLT1 may reflect a natural phenomenon, but could also be the result of mine-related sedimentation events in 2016 (Baffinland 2016b). Therefore, the slight shift towards a greater proportion of HPG burrowers in the benthic invertebrate community may have reflected a sedimentation influence at lower SDLT1 in 2016.

Temporal comparison of the lower SDLT1 benthic invertebrate community data indicated significantly higher invertebrate density in 2016 compared to baseline data collected in 2008 and 2013 (Figure 4.6; Appendix Table F.26). However, no significant differences in richness, Simpson's Evenness or any community compositional features occurred consistently between the 2016 data and both respective baseline data sets. Increased benthic invertebrate density can often occur as an outcome of slight nutrient enrichment of aquatic systems (Ward 1992; Taylor and Bailey 1997). However, temporal comparisons indicated similar chlorophyll a concentrations between 2016 and the baseline period at SDLT1 (Figure 4.4), suggesting that

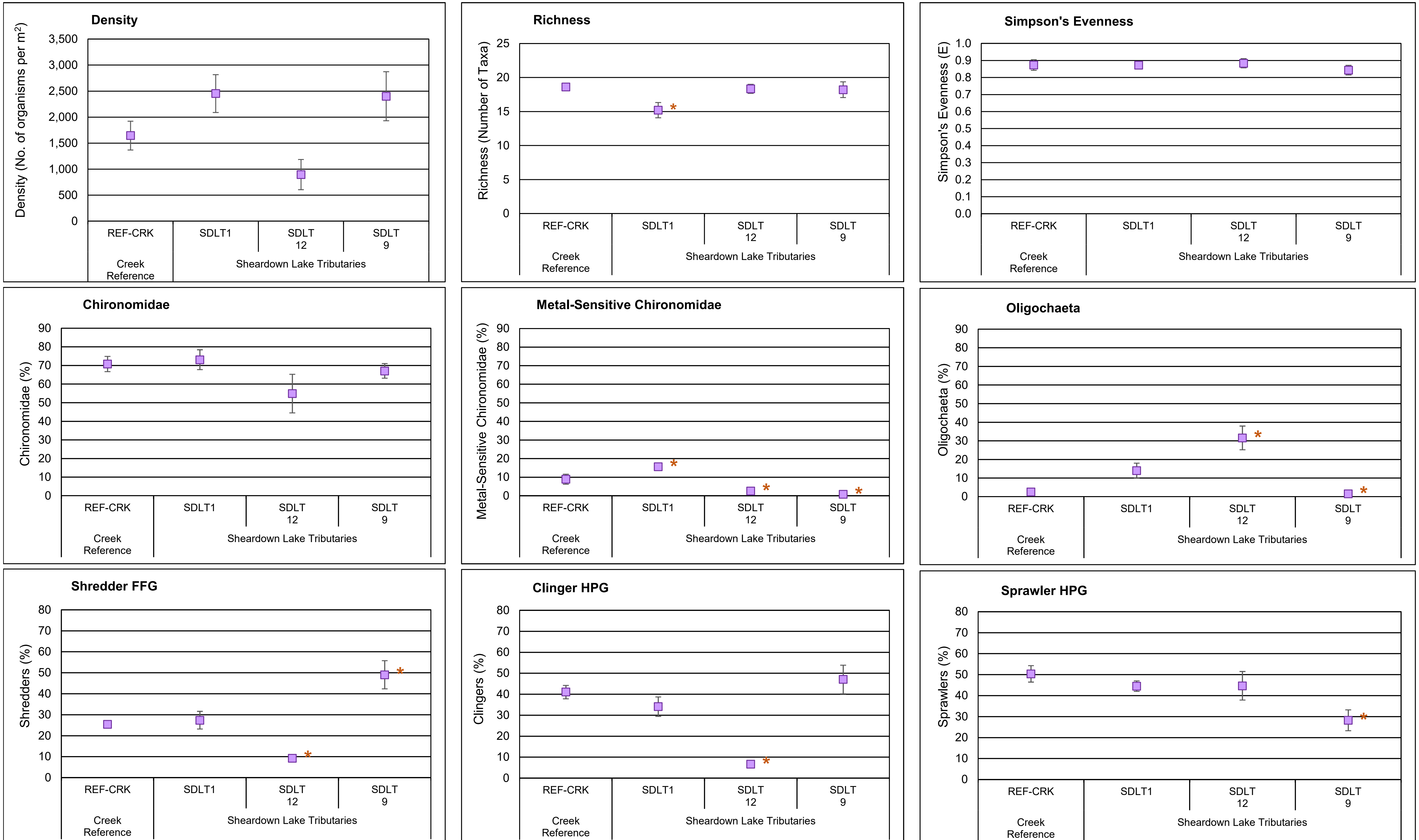


Figure 4.5: Comparison of benthic invertebrate community metrics between Sheardown Lake Tributary and creek reference study areas (mean ± SE), Mary River Project CREMP, August 2016. Asterisk (*) next to SDLT data points indicates significant difference from Unnamed Reference Creek.

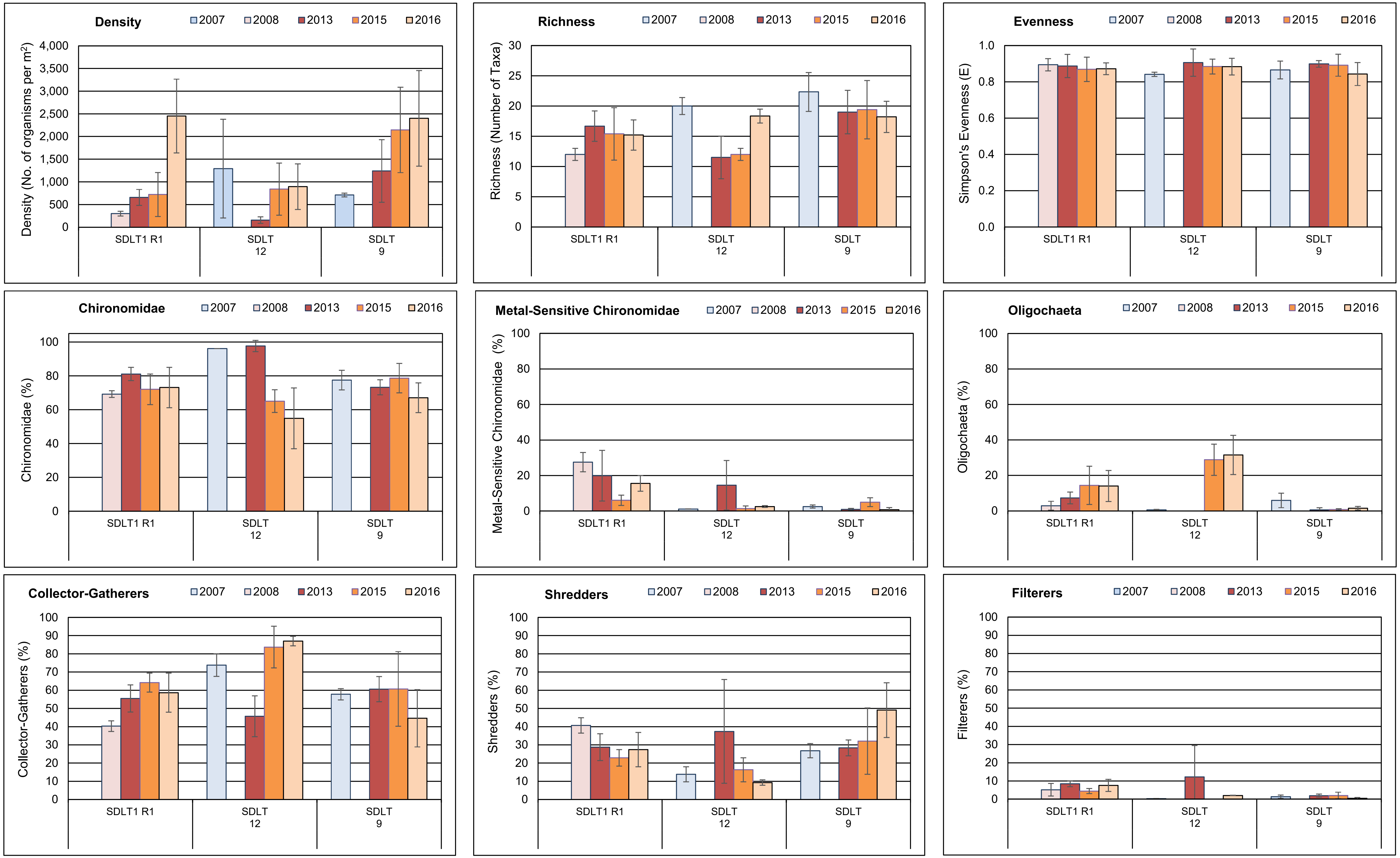


Figure 4.6: Comparison of benthic invertebrate community metrics (mean ± SD) at Sheardown Lake Tributaries 1, 12 and 9 among operational (2015, 2016) and baseline (2007, 2008, 2011, 2013) studies, Mary River Project CREMP.

higher benthic invertebrate density in 2016 was not likely related to a mine-associated change in trophic status of the SDLT1 system. Given the occurrence of few differences in benthic invertebrate community endpoints between 2016 and the baseline period, and the fact that the few differences were not consistently observed in 2015 and 2016 compared to the baseline period, higher density in 2016 potentially reflected natural year-to-year variability within the SDLT1 system. Baseline studies did not include HPG analysis precluding temporal evaluation of benthic endpoints important to assessment of sedimentation influences on in-stream biota.

Sheardown Lake Tributary 12 (SDLT12)

The benthic invertebrate community of Sheardown Lake Tributary 12 (SDLT12) did not differ significantly from Unnamed Reference Creek for primary endpoints of density, richness or Simpson's Evenness in 2016 (Figure 4.5; Appendix Table F.25). However, marked differences in community composition were indicated between these watercourses based on significant differences in Bray-Curtis Index and several key dominant invertebrate, functional feeding and habitat preference groups (Figure 4.5; Appendix Table F.25). Because the magnitude of difference in the relative abundance of metal-sensitive chironomids was within a CES_{BIC} of ± 2 SD_{REF} (Figure 4.5; Appendix Table F.25), the differences in community composition between SDLT12 and Unnamed Reference Creek were not likely related to metal concentrations. Rather, significantly higher relative abundance of HPG burrowers including Nemata (roundworms) and Oligochaeta (aquatic worms) and FFG collector-gatherer deposit feeders was consistent with the occurrence of significantly slower water velocity and greater substrate embeddedness (i.e., more depositional habitat) at SDLT12 than at Unnamed Reference Creek (Appendix Tables F.22 and F.25). Therefore, a natural difference in habitat features between SDLT12 and Unnamed Reference Creek potentially accounted for differences in benthic invertebrate community compositional features between watercourses. However, similar to SDLT1, a higher relative abundance of HPG burrowers at SDLT12 was also consistent with greater substrate embeddedness that may have resulted from sedimentation events in 2016.

Temporal comparison of the SDLT12 benthic invertebrate community data did not indicate any significant differences in density, richness and Simpson's Evenness between 2016 and baseline data collected in 2007 (Figure 4.6; Appendix Table F.27). However, significantly higher relative abundance of burrowing invertebrates including aquatic worms and Tipulidae (crane flies) together with significantly greater relative abundance of FFG collector-gatherers in both 2015 and 2016 compared to the 2007 baseline study suggested changes in habitat conditions at SDLT12 with the commencement of mine operations. Although such temporal changes potentially reflected slight differences in sampling location between the mine operational and baseline periods, field observations from the 2016 study included the occurrence of silt deposits on in-stream substrate of SDLT12. Therefore, a mine-related reduction in flow and/or increased particle loadings (e.g., through dust and/or erosional deposition) over time may have accounted for subtle temporal changes in the benthic invertebrate community between the 2015/2016 mine operational and 2007 baseline studies. Overall, it was uncertain as to whether changes in benthic invertebrate compositional features over time at SDLT12 reflected natural variability in habitat or a mine-related influence that potentially included greater sedimentation in 2016.

Sheardown Lake Tributary 9 (SDLT9)

The benthic invertebrate community of Sheardown Lake Tributary 9 (SDLT9) did not differ significantly from Unnamed Reference Creek for primary endpoints of density, richness or Simpson's Evenness in 2016 (Figure 4.5; Appendix Table F.25). However, similar to SDLT12, marked differences in community composition were indicated between SDLT9 and Unnamed Reference Creek based on significant differences in Bray-Curtis Index and several groups of dominant taxa, FFG and HPG (Figure 4.5; Appendix Table F.25). Notably, the magnitude of difference in the relative abundance of metal-sensitive chironomids between SDLT9 and the reference creek was within a CES_{BIC} of ± 2 SD_{REF} (Figure 4.5; Appendix Table F.25), suggesting that differences in community composition between watercourses were not likely related to metal concentrations. Rather, a significantly higher relative abundance of HPG burrowers including nemata (roundworms) and Tipulidae (crane flies) combined with a significantly greater relative abundance of FFG shredders was consistent with field observations of greater amounts of rooted in-stream vegetation at SDLT9 compared to the reference creek (Appendix Tables F.1 and F.25). Temporal comparisons indicated no significant differences in benthic invertebrate density, richness, Simpson's Evenness or any dominant invertebrate groups, FFG and HPG at SDLT9 between mine operational period data collected in 2015/2016 and baseline period data collected in 2007 and 2013 (Figure 4.6; Appendix Table F.28). In turn, this suggested that the differences in benthic invertebrate community composition (and amount of in-stream vegetation) between SDLT9 and Unnamed Reference Creek in 2016 likely reflected a natural difference in habitat features between watercourses.

4.2 Sheardown Lake NW (DLO-1)

4.2.1 Water Quality

Water quality profiles of *in-situ* water temperature, dissolved oxygen, pH and specific conductance conducted at Sheardown Lake NW in 2016 showed no substantial station-to-station differences during any of the winter, summer or fall sampling events (Appendix Figures C.10 – C.13). On average, water temperature profiles suggested weak stratification during the summer sampling event, but more strongly established stratification during the fall sampling event at Sheardown Lake NW in 2016 (Figure 4.7). In both seasons, the greatest change in temperature occurred between lake depths of approximately 10 and 15 m, which was comparable to the thermocline depth range observed at Reference Lake 3 (Figure 4.7). Average water temperature at the bottom of the water column at Sheardown Lake NW littoral stations was slightly warmer than at Reference Lake 3 at the time of fall sampling in 2016, the difference of which was statistically significant (Figure 4.8). However, the incremental difference in average bottom water temperature between lakes was small (i.e., 0.6˚C) and thus was unlikely to be ecologically meaningful. Dissolved oxygen profiles at Sheardown Lake NW showed an oxycline at depths greater than approximately 16 m and 10 m during the winter and fall, respectively, but no appreciable change in dissolved oxygen saturation from surface to bottom in the summer of 2016 (Figure 4.7; Appendix Figure C.11). No oxycline was observed at Reference Lake 3 in 2016 during the summer or fall sampling events (Appendix Figure B.3). Dissolved oxygen saturation levels at the bottom of the water column at littoral stations (i.e., approximately 10 m deep) of Sheardown Lake NW were significantly higher than those at Reference Lake 3 during fall 2016 sampling (Figure 4.8; Appendix Table C.37). In addition, dissolved oxygen saturation levels were well above the WQG of 54% at all littoral stations of Sheardown Lake NW in fall 2016 (Figure 4.8) and, with the exception of depths greater than approximately 22 m in winter, through the majority of the water column during winter, summer and fall sampling events (Figure 4.7). This suggested that dissolved oxygen was not limiting for pelagic or bottom-dwelling biota within Sheardown Lake NW for the majority of the year in 2016.

In-situ profiles of pH and specific conductance showed no substantial change from the surface to bottom of the Sheardown Lake NW water column, indicating no chemical stratification (Figure 4.7). Mean pH at the bottom of the water column at littoral stations of Sheardown Lake NW did not differ significantly from that of Reference Lake 3 during fall sampling in 2016 (Figure 4.8; Appendix Table C.37). In addition, pH values were consistently within WQG limits of 6.5 – 9.0 through the entire water column during all 2016 sampling events conducted at Sheardown Lake NW (Appendix Tables C.33 – C.36). Specific conductance was significantly higher at Sheardown Lake NW compared to the reference lake during fall sampling (Figure 4.8; Appendix Table C.37). However, similar to observations at Camp Lake (Section 4.2.1), specific conductance at Sheardown Lake NW was intermediate to that of reference creek and river stations in fall 2016, and therefore it was unclear whether higher specific conductance at Sheardown Lake NW than at Reference Lake 3 was related to natural regional variability in

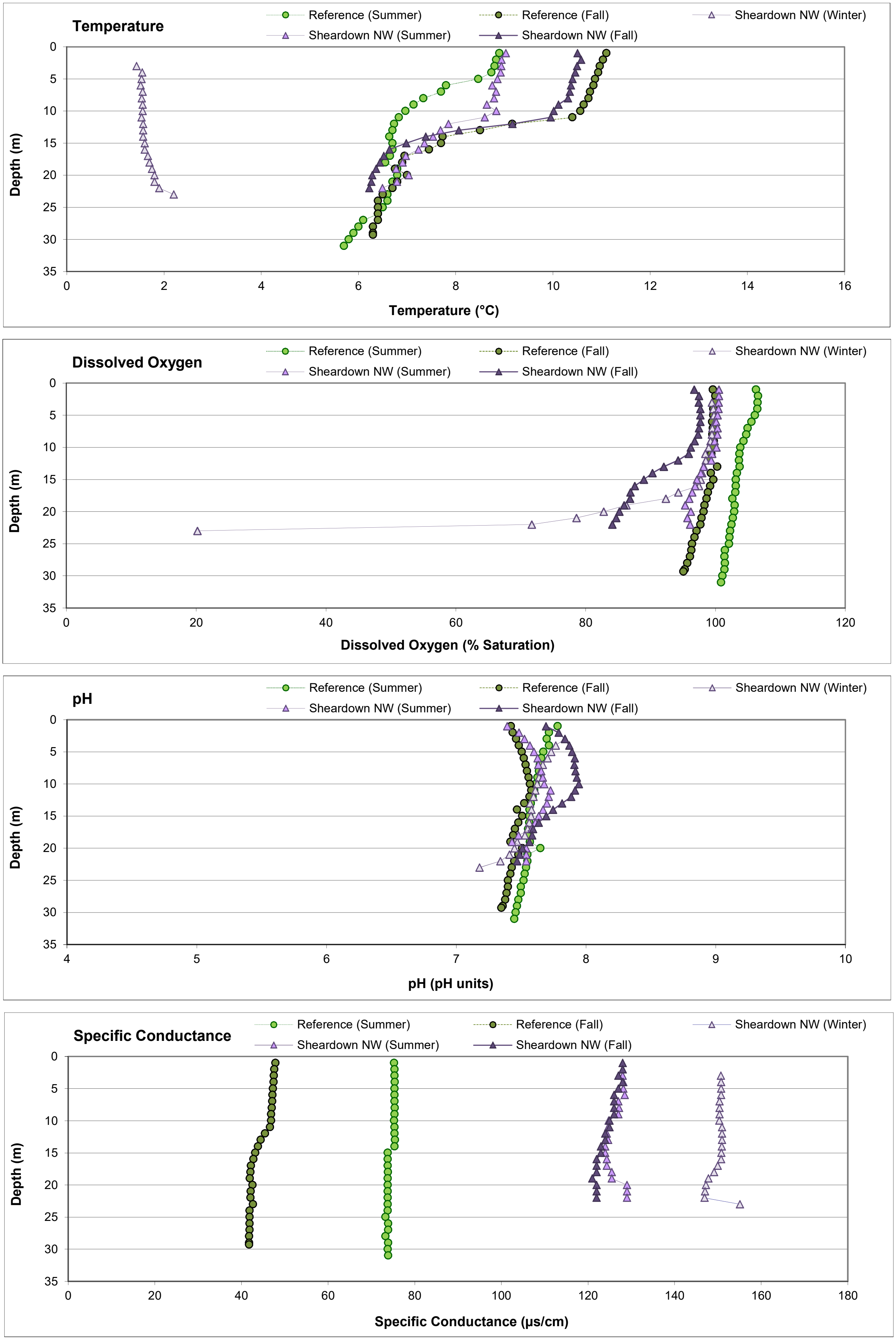


Figure 4.7: Average *in-situ* water quality with depth from surface at Sheardown Lake NW (mine-exposed area) compared to Reference Lake 3 during winter, summer, and fall sampling events, Mary River Project CREMP, 2016.

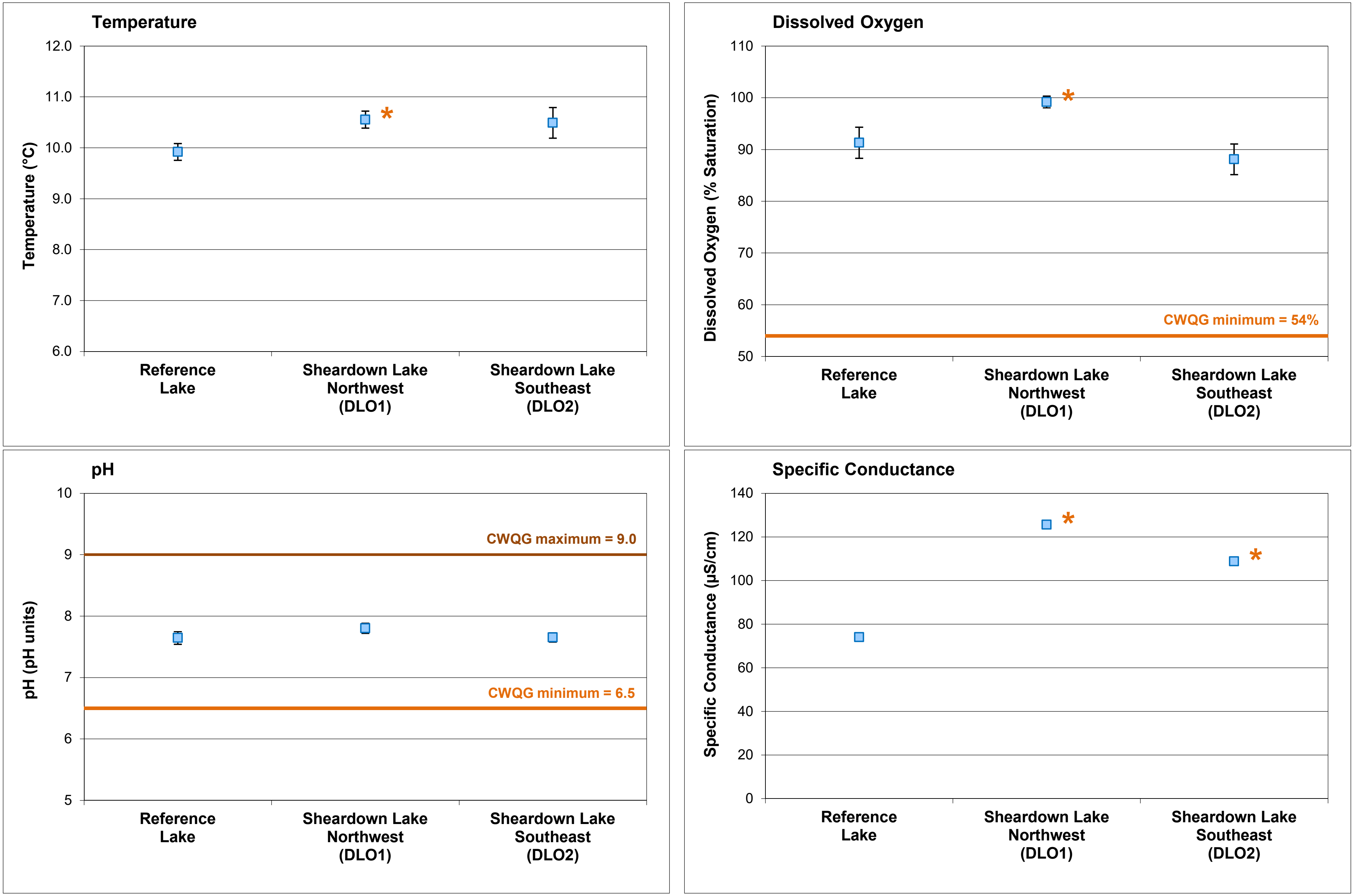


Figure 4.8: Comparison of in-situ water quality (mean ± SD; n = 5) measured near the bottom of the water column at the Sheardown Lake basins and Reference Lake 3 (REF3) littoral benthic invertebrate community stations, Mary River Project CREMP, August 2016. An asterisk (*) next to the Sheardown Lake data point indicates a significant difference compared to the reference lake measure.

surface waters or a mine-related influence. Water clarity, as determined through evaluation of Secchi depth, was significantly lower at Sheardown Lake NW than at Reference Lake 3 during the 2016 fall sampling event (Appendix Tables C.36 – C.37). Secchi depth readings showed relatively low variability among stations at Sheardown Lake NW in the fall of 2016, suggesting no spatial differences in water clarity throughout the lake (Appendix Table C.36).

Water chemistry within Sheardown Lake NW showed no distinct spatial differences in parameter concentrations among the six sampling stations during any of the winter, summer or fall sampling events in 2016 (Table 4.2; Appendix Table C.38), suggesting that the lake waters were continually well mixed both laterally and vertically. Turbidity and total concentrations of aluminum, manganese, molybdenum and uranium were slightly (3- to 5-fold higher) to moderately (5- to 10-fold higher) elevated at Sheardown Lake NW compared to Reference Lake 3 during the summer and/or fall sampling events (Table 4.2; Appendix Table C.38). Similar to the 2015 study, total aluminum and manganese concentrations showed a significant positive correlation with turbidity at Sheardown Lake NW in 2016 ($r = 0.54$ and 0.49, respectively). This suggested that elevated total aluminum and manganese concentrations at Sheardown Lake NW reflected influences associated with surface runoff or backflow received from Mary River that contained naturally high concentrations of aluminum-based, manganese bearing, particulate minerals. This was supported through comparisons of dissolved metal concentrations, which indicated that only dissolved molybdenum and uranium concentrations (and not aluminum or manganese) were elevated at Sheardown Lake NW compared to Reference Lake 3 (Appendix Table C.39). In addition, the ratio of dissolved to total concentrations of aluminum and manganese indicated that the majority (i.e., >65%) of each of these metals was in the dissolved fraction at Sheardown Lake NW based on the 2016 data. Although total molybdenum and uranium concentrations were not correlated with turbidity, similar concentrations of these metals were observed between Sheardown lake NW and the reference creek and river stations during summer and fall 2016 monitoring. In turn, this suggested that higher molybdenum and uranium concentrations at Sheardown Lake NW compared to Reference Lake 3 may have also reflected natural geochemical differences between these lakes. Despite elevation of total aluminum, manganese, molybdenum and uranium metals at Sheardown Lake NW compared to Reference Lake 3, concentrations of all parameters were well below established WQG and AEMP benchmarks at Sheardown Lake NW during all sampling events in 2016[8] (Table 4.2; Appendix Table C.38).

[8] Refer to footnote 2 (page 23) and Appendix B regarding phenol concentrations above WQG at the mine-exposed and reference areas of the Mary River Project LSA waterbodies.

Table 4.2: Water chemistry at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3) monitoring stations, Mary River Project CREMP, August 2016. Values presented are averages of samples taken from the surface and the bottom of the water column at each station. * Copper data confounded by sampling equipment.

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Lake 3 Average (n = 3) Fall 2016	Sheardown Lake NW Station DD-HAB9 STN1 21-Aug-2016	DL0-01-5 21-Aug-2016	DL0-01-1 21-Aug-2016	DL0-01-4 22-Aug-2016	DL0-01-2 22-Aug-2016	DL0-01-7 22-Aug-2016
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	84	134	130	130	133	129	133
	pH (lab)	pH	6.5 - 9.0	-	7.68	8.14	7.89	7.98	8.12	7.93	8.12
	Hardness (as $CaCO_3$)	mg/L	-	-	35	64	63	63	63	62	62
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	39	65	62	67	67	59	63
	Turbidity	NTU	-	-	0.33	0.91	0.81	0.82	0.84	0.88	0.79
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	61	61	59	60	59	59
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.040	0.027	0.026	0.040	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	0.028	0.023	0.022	0.027	0.025	0.023
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.7	1.6	1.8	1.9	1.8	1.7	1.7
	Total Organic Carbon	mg/L	-	-	2.8	2.0	1.9	2.0	1.9	1.8	1.8
	Total Phosphorus	mg/L	0.020[α]	-	0.010	0.005	0.015	0.007	0.004	0.006	0.010
	Phenols	mg/L	0.004[α]	-	0.003	0.002	0.016	0.009	0.004	0.002	0.004
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.3	3.0	3.0	3.0	3.1	3.0	3.1
	Sulphate (SO_4)	mg/L	218[β]	218	4.1	4.3	3.6	3.8	4.2	3.8	4.1
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[c]	0.004	0.013	0.011	0.013	0.017	0.011	0.017
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00653	0.00618	0.00607	0.00615	0.00617	0.00601	0.00643
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	7.0	12.2	12.3	12.7	12.6	12.2	12.8
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.0008	0.0010	0.0009	0.0009	*	*	*
	Iron (Fe)	mg/L	0.30	0.300	<0.030	0.03	0.03	0.03	0.03	0.03	0.03
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	0.00005	0.00005	0.00005	0.00005	0.00005	0.00005
	Lithium (Li)	mg/L	-	-	<0.0010	0.001	0.001	0.001	0.0012	0.0011	0.0013
	Magnesium (Mg)	mg/L	-	-	4.3	7.9	7.5	7.6	7.8	7.6	7.9
	Manganese (Mn)	mg/L	0.935[β]	-	0.00062	0.00201	0.00240	0.00207	0.00201	0.00214	0.00217
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00014	0.00075	0.00076	0.00077	0.00076	0.00072	0.00077
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	0.00065	0.00061	0.00064	0.00061	0.00063	0.00065
	Potassium (K)	mg/L	-	-	0.89	1.09	1.07	1.06	1.08	1.06	1.10
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.42	0.46	0.51	0.48	0.48	0.51	0.48
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.84	1.40	1.35	1.33	1.40	1.35	1.39
	Strontium (Sr)	mg/L	-	-	0.0081	0.0082	0.0083	0.0085	0.0084	0.0081	0.0084
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00027	0.00103	0.00094	0.00094	0.00102	0.00094	0.00098
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030

[a] Canadian Water Quality Guideline (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). 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 Sheardown Lake NW.

[c] Benchmark is 0.179 mg/L and 0.173 mg/L for shallow and deep stations, respectively.

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

Temporal comparisons of the Sheardown Lake NW water chemistry data suggested that average (total) concentrations of the majority of parameters in 2016 were within the range of baseline concentrations (2005 – 2013; Figure 4.9; Appendix Figure C.18). Only phenol concentrations showed moderate elevation (i.e., 5- to 10-fold higher) in 2016 compared to the baseline data based on fall sampling results (Appendix Table C.40). A number of parameters, including conductivity, molybdenum, sodium and strontium, showed successively higher concentrations over years of mine-construction (2014), initial mine operation (2015) and 2016 (Figure 4.9; Appendix Figure C.18; Appendix Table C.40). Although the magnitude of these changes were relatively minor and, because concentrations in 2016 remained well below WQG, were unlikely to be ecologically meaningful, the sequential increases were consistent with greater mine-related influence on water quality over time at Sheardown Lake NW.

4.2.2 Sediment Quality

Surficial sediment collected at the Sheardown Lake NW coring stations was characterized by silt to sandy loam material with low TOC content (Figure 4.10). Although littoral station co-dominant sand and silt sediment particle sizes did not differ significantly between Sheardown Lake NW and the reference lake, sediment TOC content was significantly lower at Sheardown Lake NW (Appendix Table D.14). Similar to observations at Reference Lake 3 and Camp Lake, reddish- to orange-brown oxidized material was commonly observed on the surface of Sheardown Lake NW littoral and profundal sediments (Appendix Tables D.11 – D.13). In Sheardown Lake NW, this material occasionally occurred as a thin, distinct layer that was likely composed principally of iron (oxy)hydroxide precipitate. No visible evidence of excessive sedimentation was observed at Sheardown Lake NW in 2016 (Appendix Tables D.11 – D.13). Below the surficial layer, substrates at some Sheardown Lake NW littoral and profundal stations exhibited blackening and/or darkening and possessed a slight sulphidic odour suggesting the occurrence of reducing conditions and, in some cases, a distinct redox boundary was observed in sediments of the lake (Appendix Tables D.11 to D.13). The occurrence of reducing sediment conditions in 2016 appeared to be more pronounced at Sheardown Lake NW than at the reference lake, where reducing sediment conditions occurred sporadically within the sediment (Appendix Tables D.1 – D.3 and D.11 – D.13).

Sediment metal concentrations at Sheardown Lake NW showed no spatial differences among stations in 2016 with the possible exception of at the littoral station located nearest the SDLT1 lake inlet (i.e., Station DD-HAB9-Stn2; Appendix Table D.15). At this station, sediment barium, iron, manganese, molybdenum and phosphorus concentrations were noticeably higher than at other littoral stations, and compared to profundal stations, suggesting that these metals originated from the SDLT1 watercourse. Erosion events that resulted in elevated total

Figure 4.9: Temporal comparison of water chemistry at Sheardown Lake Northwest (DLO-01) and Sheardown Lake Southeast (DLO-02) for mine baseline (2005 - 2013), construction (2014), and operational (2015, 2016) periods during fall. Values represent mean ± SD. Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Sheardown Lake (northwest and southeast).

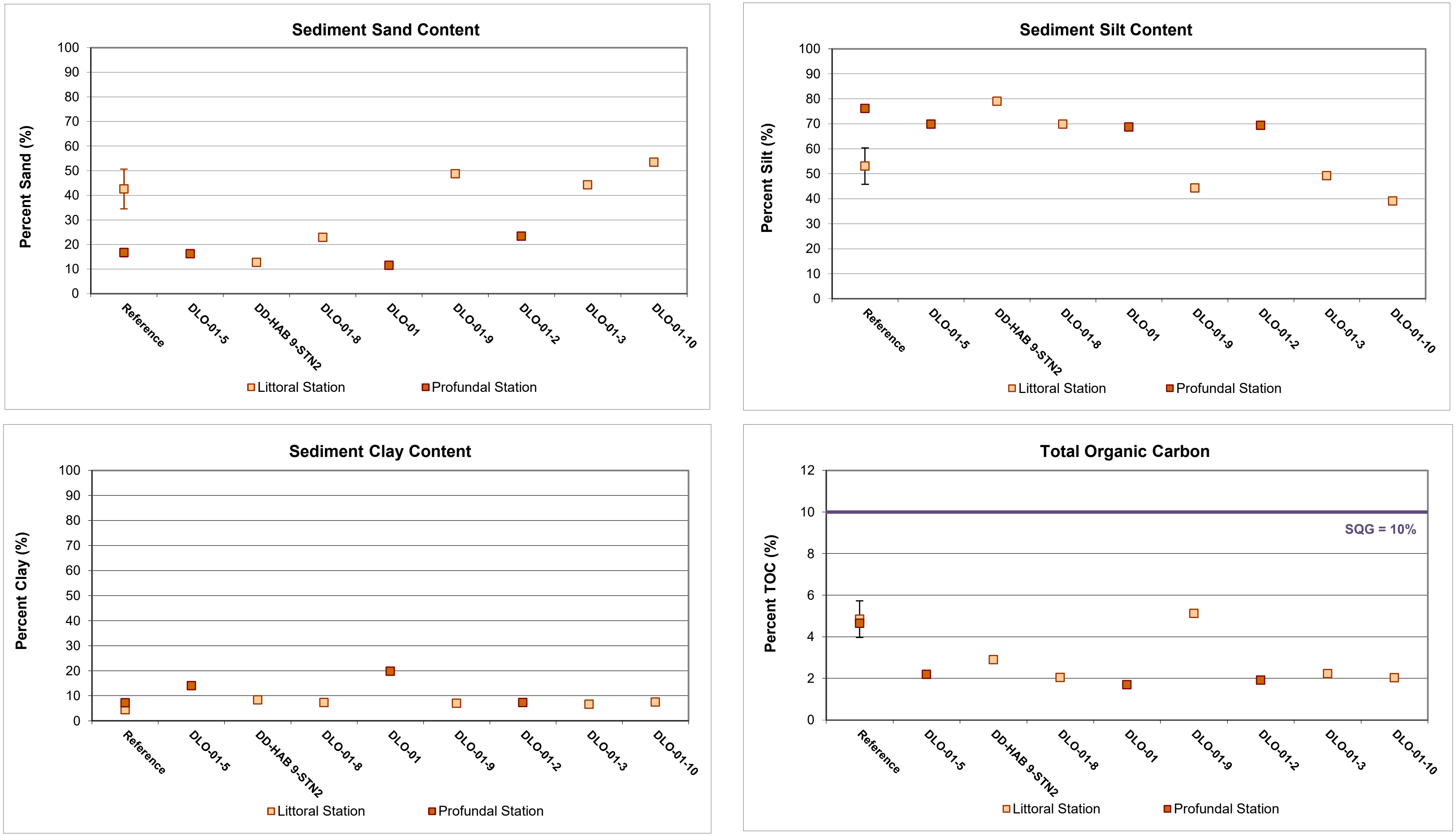


Figure 4.10: Sediment particle size and total organic carbon (TOC) content comparisons among Sheardown Lake NW (DLO-01) sediment monitoring stations and Reference Lake 3 averages (mean ± SE), Mary River Project CREMP, August 2016.

suspended solids (TSS) concentrations at SDLT1 during spring freshet potentially contributed to higher concentrations of these metals in lake sediments near the watercourse outlet to Sheardown Lake NW in 2016 (see Baffinland 2016). On average, concentrations of arsenic, manganese and molybdendum were slightly elevated (i.e., 2- to 5-fold higher) in sediment at littoral stations of Sheardown Lake NW compared to the reference lake littoral stations (Table 4.3). However, average metal concentrations in sediment at profundal stations were similar between lakes (Table 4.3). Although mean iron and manganese concentrations were above applicable SQG at littoral and profundal stations of Sheardown Lake NW, mean concentrations of these metals were also above SQG at profundal stations of Reference Lake 3 (Table 4.3). Similarly, despite mean arsenic and iron concentrations above respective AEMP benchmarks in sediment at profundal stations of Sheardown Lake NW, mean concentrations of these metals, together with chromium and copper, were above applicable AEMP benchmarks in sediment at profundal stations of Reference Lake 3 (Table 4.3). This suggested that, in part, elevated arsenic, iron, manganese concentrations at Sheardown Lake NW compared to sediment quality guidelines/benchmarks reflected at natural phenomenon. Lastly, sediment nickel and phosphorus concentrations were above SQG and the AEMP benchmark at individual stations in Sheardown Lake NW, but on average, were below the applicable guidelines/benchmarks (Table 4.3; Appendix Table D.15).

Temporal comparisons of the sediment metals data indicated slightly elevated (i.e., 2- to 5-fold higher) average concentrations of arsenic, barium, iron, manganese and molybdenum at littoral stations of Sheardown Lake NW in 2016 compared to the baseline (2005 – 2013) period[9] (Figure 4.11). No substantial changes in metal concentrations occurred at profundal stations between 2016 and the baseline period (Figure 4.11; Appendix Table D.16). The parameters listed above showed progressively higher mean concentrations from baseline, to mine construction, to 2015 and 2016 mine operational years in sediment at littoral stations of Sheardown Lake NW. However, variability in parameter concentrations was high, and none of the above listed parameters exhibited concentrations greater than at the reference lake littoral and profundal stations (Figure 4.11; Appendix Table D.16). Similar to the analysis of Camp Lake sediment quality data, this suggested that changes in station replication and location among studies likely contributed to the appearance of greater mean concentrations of select parameters in sediment over time at the Sheardown Lake NW littoral stations. Nevertheless, because arsenic, barium, iron, manganese and molybdenum have shown progressively higher mean concentrations in littoral and/or profundal sediment of both

[9] Refer to footnote 6 (page 32) regarding temporal differences in sediment boron concentrations at Mary River Project LSA waterbodies.

Table 4.3: Sediment particle size, total organic carbon, and metal concentrations at Sheardown Lake NW (DLO-01), Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3) sediment monitoring stations, Mary River Project CREMP, August 2016.

	Analyte	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b] (NW ; SE)	Littoral			Profundal	
					Reference Lake (n = 5)	Sheardown Lake NW (n=5)	Sheardown Lake SE (n=5)	Reference Lake (n = 5)	Sheardown Lake NW (n=3)
					Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error
Non-metals	Sand	%	-	-	42.5 ± 8.1	36.4 ± 7.9	12.0 ± 2.5	16.7 ± 1.5	17.0 ± 3.5
	Silt	%	-	-	53.1 ± 7.3	56.3 ± 7.7	73.0 ± 1.8	76.1 ± 1.4	69.3 ± 0.3
	Clay	%	-	-	4.4 ± 1.0	7.3 ± 0.3	14.9 ± 1.8	7.2 ± 0.4	13.7 ± 3.6
	Moisture	%	-	-	89.7 ± 6.0	70.4 ± 5.0	41.4 ± 3.3	83.5 ± 5.4	58.4 ± 2.7
	Total Organic Carbon	%	10^{α}	-	4.85 ± 0.88	2.86 ± 0.59	1.30 ± 0.19	4.64 ± 0.13	1.94 ± 0.14
Metals	Aluminum (Al)	mg/kg	-	-	16,480 ± 397	15,620 ± 1,329	16,440 ± 1,127	25,150 ± 1,418	21,217 ± 1,516
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	0.12 ± 0.02	<0.10 ± 0
	Arsenic (As)	mg/kg	17	6.2 ; 5.9	3.71 ± 0.26	***7.95 ± 1.88***	4.40 ± 0.69	***6.47 ± 0.27***	4.30 ± 0.35
	Barium (Ba)	mg/kg	-	-	112 ± 11	196 ± 107	92 ± 17	162 ± 8	101 ± 5
	Beryllium (Be)	mg/kg	-	-	0.67 ± 0.02	0.82 ± 0.07	0.76 ± 0.05	1.02 ± 0.05	1.11 ± 0.09
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0.0	0.23 ± 0.02	0.29 ± 0.10	0.21 ± 0.004	0.27 ± 0.03
	Boron (B)	mg/kg	-	-	13.0 ± 0.9	24.0 ± 2.6	21.5 ± 1.7	19.2 ± 1.0	30.7 ± 2.0
	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.035	0.267 ± 0.059	0.103 ± 0.008	0.180 ± 0.010	0.257 ± 0.009
	Calcium (Ca)	mg/kg	-	-	5,128 ± 470	4,494 ± 429	5,112 ± 627	6,111 ± 156	4,402 ± 146
	Chromium (Cr)	mg/kg	90	97 ; 79	55.0 ± 1.2	61.9 ± 4.6	70.7 ± 3.7	***80.0 ± 4.1***	78.1 ± 3.9
	Cobalt (Co)	mg/kg	-	-	10.15 ± 0.57	12.70 ± 0.83	13.08 ± 0.77	18.15 ± 0.75	15.83 ± 0.70
	Copper (Cu)	mg/kg	110	58 ; 56	***66.5 ± 7.4***	42.9 ± 7.2	25.8 ± 1.5	***101.4 ± 5.6***	46.7 ± 3.3
	Iron (Fe)	mg/kg	$40,000^{\alpha}$	52,200 ; 34,400	29,840 ± 3,488	***58,740 ± 9,478***	***40,340 ± 3,922***	***53,580 ± 2,174***	40,333 ± 2,067
	Lead (Pb)	mg/kg	91.3	35	***46.0 ± 17.4***	19.8 ± 1.6	21.7 ± 4.3	29.5 ± 5.0	31.0 ± 3.6
	Lithium (Li)	mg/kg	-	-	27.3 ± 0.4	27.5 ± 2.4	30.4 ± 2.3	41.7 ± 2.1	39.1 ± 2.5
	Magnesium (Mg)	mg/kg	-	-	10,852 ± 274	10,896 ± 780	12,720 ± 357	16,160 ± 814	13,517 ± 738
	Manganese (Mn)	mg/kg	$1,100^{\alpha,\beta}$	4,530 ; 657	496 ± 99	2,503 ± 1,952	***1,596 ± 911***	***1,866 ± 449***	1,435 ± 720
	Mercury (Hg)	mg/kg	0.486	0.17	0.0355 ± 0.0063	0.0385 ± 0.0057	0.0252 ± 0.0028	0.0699 ± 0.0019	0.0432 ± 0.0078
	Molybdenum (Mo)	mg/kg	-	-	2.19 ± 0.49	8.80 ± 2.94	1.65 ± 0.45	3.27 ± 0.34	2.99 ± 1.39
	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	77 ; 66	38.6 ± 1.6	65.8 ± 6.5	55.8 ± 3.2	56.3 ± 2.6	67.8 ± 0.9
	Phosphorus (P)	mg/kg	$2,000^{\alpha}$	1,958 ; 1,278	840 ± 47	1,410 ± 292	1,026 ± 56	1,121 ± 57	891 ± 29
	Potassium (K)	mg/kg	-	-	3,894 ± 172	3,806 ± 311	3,908 ± 319	5,891 ± 281	5,255 ± 328
	Selenium (Se)	mg/kg	-	-	0.49 ± 0.06	0.42 ± 0.07	0.20 ± 0	0.85 ± 0.06	0.40 ± 0.05
	Silver (Ag)	mg/kg	-	-	0.12 ± 0.01	0.12 ± 0.01	0.11 ± 0.01	0.27 ± 0.01	0.18 ± 0.03
	Sodium (Na)	mg/kg	-	-	296 ± 29	231 ± 20	267 ± 22	455 ± 24	301 ± 15
	Strontium (Sr)	mg/kg	-	-	11.4 ± 0.5	10.0 ± 0.5	10.5 ± 0.4	15.8 ± 0.6	12.2 ± 0.6
	Thallium (Tl)	mg/kg	-	-	0.388 ± 0.021	0.448 ± 0.045	0.377 ± 0.027	0.801 ± 0.035	0.583 ± 0.026
	Tin (Sn)	mg/kg	-	-	56.3 ± 28.9	4.6 ± 1.3	10.6 ± 6.3	16.3 ± 7.8	13.1 ± 7.0
	Titanium (Ti)	mg/kg	-	-	1,072 ± 36	968 ± 67	1,188 ± 42	1,331 ± 69	1,257 ± 62
	Uranium (U)	mg/kg	-	-	11.9 ± 1.5	8.16 ± 1.8	5.17 ± 0.5	27.3 ± 1.5	8.29 ± 1.0
	Vanadium (V)	mg/kg	-	-	50.0 ± 1.3	46.5 ± 3.9	47.6 ± 2.5	72.0 ± 3.6	59.6 ± 3.2
	Zinc (Zn)	mg/kg	315	135	73.7 ± 2.7	56.6 ± 4.7	51.5 ± 2.7	105 ± 5.1	73.0 ± 4.1
	Zirconium (Zr)	mg/kg	-	-	4.3 ± 0.6	9.7 ± 2.8	15.2 ± 1.5	4.0 ± 0.2	9.4 ± 2.9

[a] Canadian Sediment Quality Guideline for the protection of aquatic life, probable effects level (PEL; CCME 2015) except those indicated by α (Ontario Provincial Sediment Quality Objective [PSQO], severe effect level (SEL); OMOE 1993) and β (British Columbia Working Sediment Quality Guideline [BCSQG], probable effects level (PEL; BCMOE 2015)).

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

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

BOLD Indicates parameter concentration above the AEMP Benchmark.

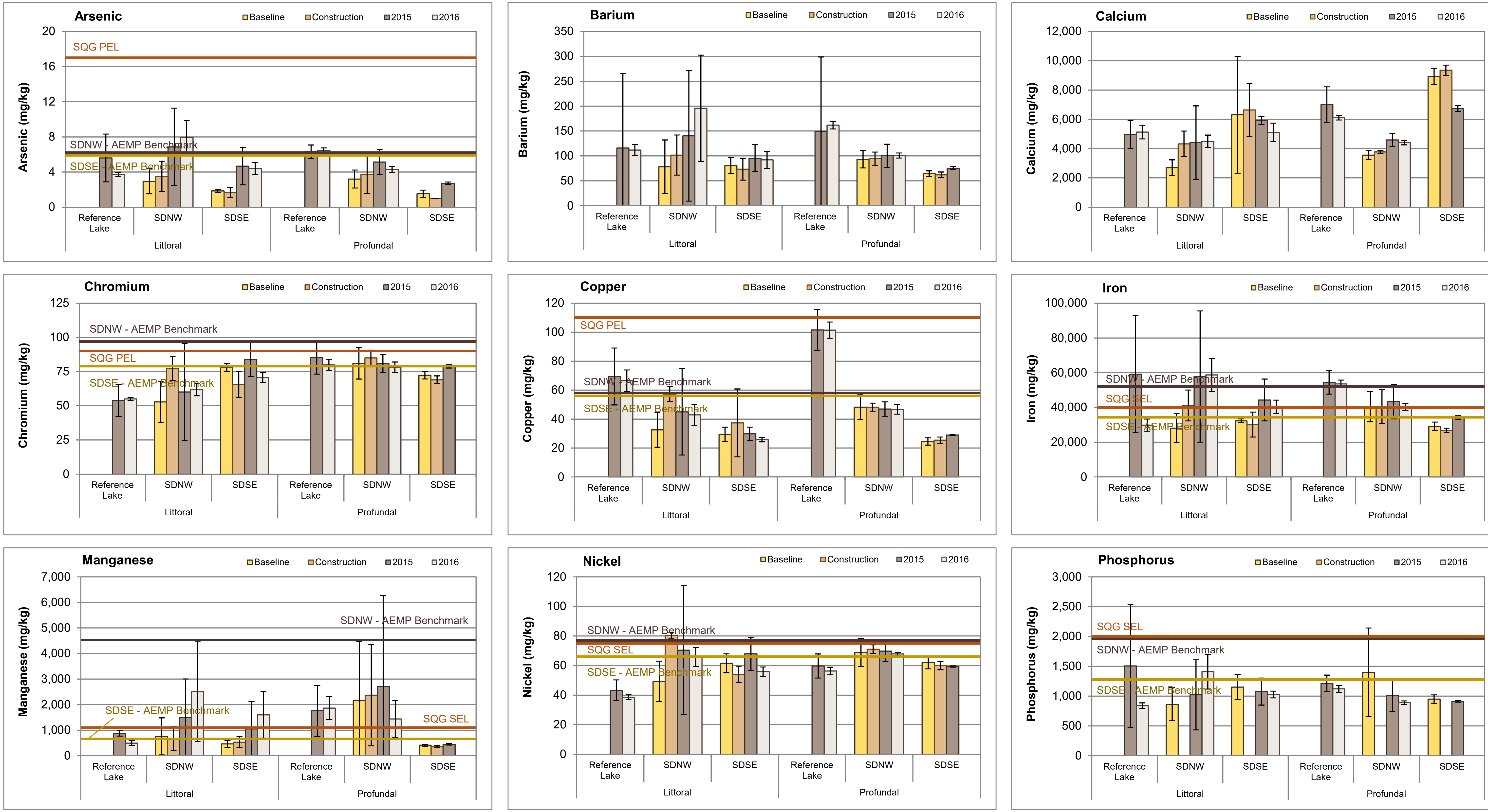


Figure 4.11: Temporal comparison of sediment metal concentrations (mean ± SD) at littoral and profundal stations of Sheardown Lake NW (SDNW), Sheardown Lake SE (SDSE), and Reference Lake 3 for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during fall, Mary River Project CREMP, 2016.

Sheardown Lake NW and Camp Lake, these parameters also reflect a potential mine-related influence on sediment quality at these mine-exposed lakes.

4.2.3 Phytoplankton

Chlorophyll a concentrations at Sheardown Lake NW showed no distinct spatial gradients among stations during the winter and fall sampling events, but higher concentrations were apparent with closer proximity to the lake outlet during the summer sampling event in 2016 (Figure 4.12). Chlorophyll a concentrations differed significantly among seasons at Sheardown Lake NW in 2016, with highest and lowest concentrations observed in summer and winter, respectively (Appendix Table E.9), reflecting similar seasonal differences in chlorophyll a concentrations at the reference lake (Appendix Table B.8). 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 2016 (Appendix Tables E.5 – E.6), chlorophyll a concentrations during each of the winter, summer and fall sampling events were well below the AEMP benchmark of 3.7 µg/L (Figure 4.12). 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 a CWQG oligotrophic categorization of Sheardown Lake NW based on mean aqueous total phosphorus concentrations below 10 µg/L during all sampling events (Table 4.2; Appendix Table C.38).

Temporally, the 2016 Sheardown Lake NW chlorophyll a concentrations did not differ significantly from concentrations during the mine construction (2014) and 2015 early-operational periods in any consistent direction among the winter, summer or fall seasons (Figure 4.13). In addition, annual average chlorophyll a concentrations did not differ significantly among 2014, 2015 and 2016 (Appendix Table E.9), suggesting no ecologically meaningful changes in the trophic status of Sheardown Lake NW since the onset of mine operations at the Mary River Project. No chlorophyll a data are available for the baseline (2005 – 2013) period for Sheardown Lake NW, precluding comparisons of chlorophyll a data to the period prior to mine construction.

4.2.4 Benthic Invertebrate Community

The benthic invertebrate community at Sheardown Lake NW littoral stations exhibited significantly higher richness, but no significant differences in density or Simpson's Evenness, compared to Reference Lake 3 littoral stations in 2016 (Table 4.4). The occurrence of a higher taxonomic richness at Sheardown Lake NW was not consistent with effects that would be expected as a result of exposure to elevated metal concentrations. Moderate Simpson's Evenness at Sheardown Lake NW indicated that the distribution of benthic invertebrates in the

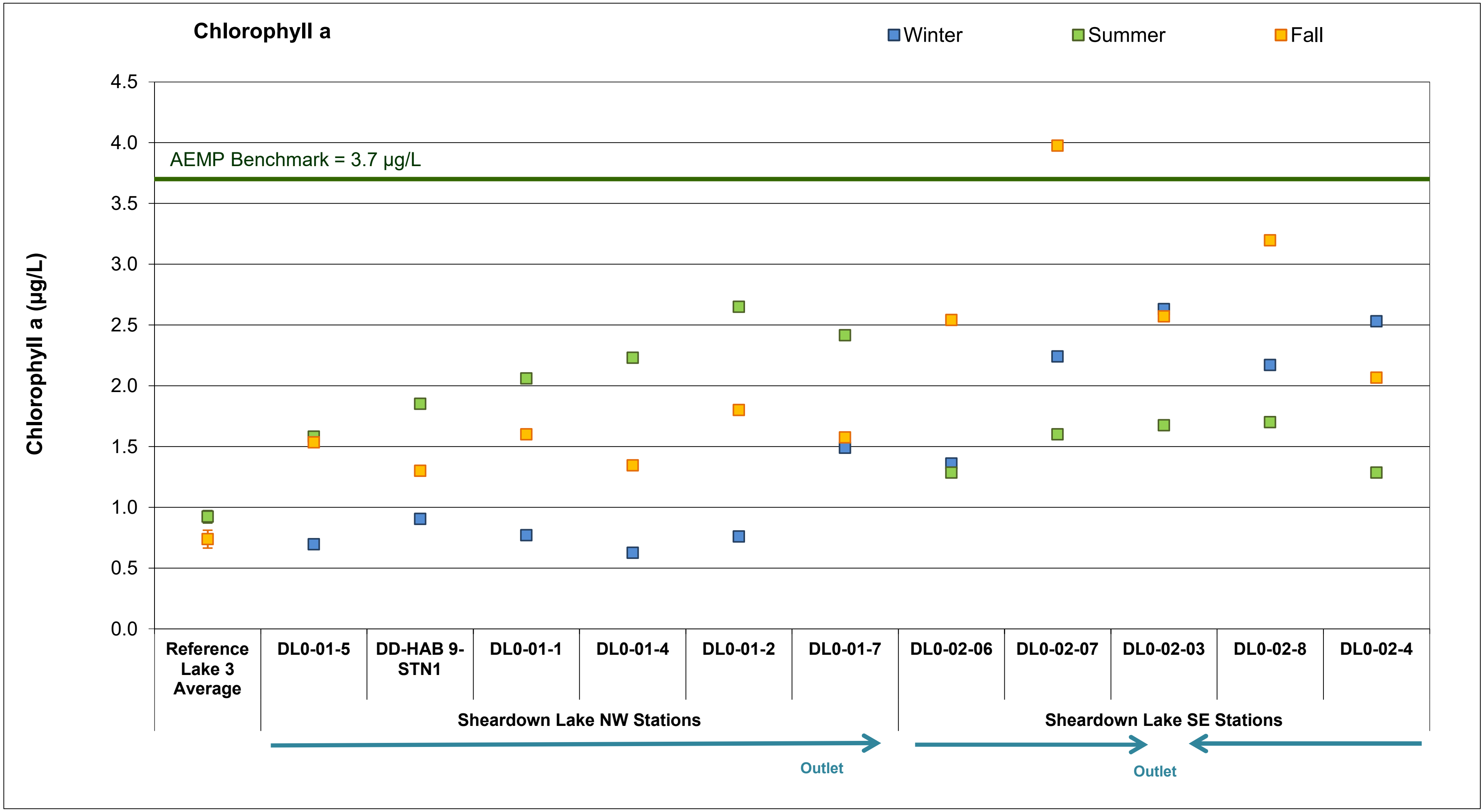


Figure 4.12: **Chlorophyll a concentrations at Sheardown Lake NW (DLO-1) and Sheardown Lake SE (DLO-2) phytoplankton monitoring stations, Mary River Project CREMP, 2016. Values are averages of samples taken from the surface and the bottom of the water column at each station. Reference values are expressed as mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2016.**

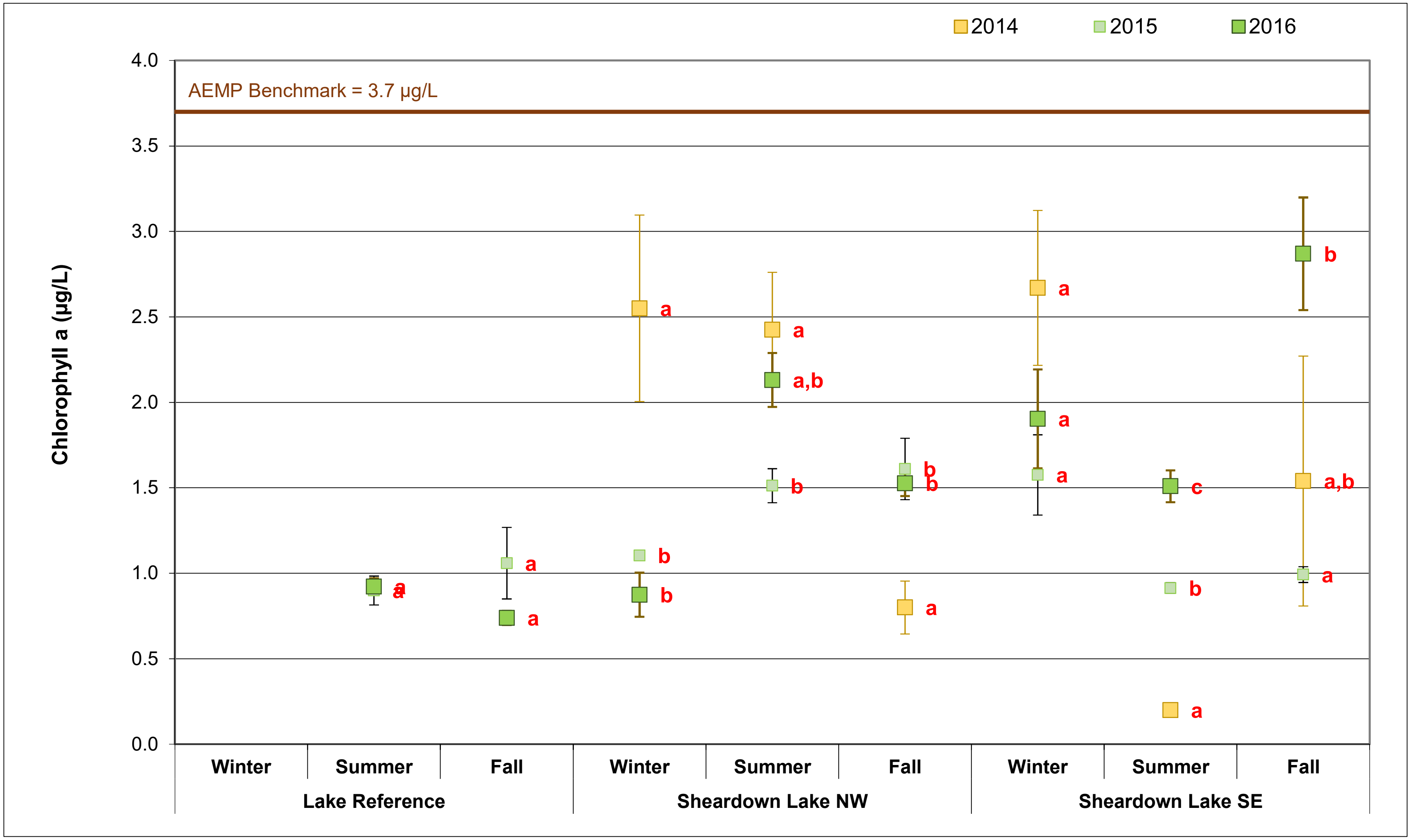


Figure 4.13: Chlorophyll a concentration seasonal comparison among 2014, 2015 and 2016 years (mean ± SE) at Sheardown Lake phytoplankton monitoring stations, Mary River Project CREMP. Data points with the same letter on the right do not differ significantly between years for the applicable season.

Table 4.4: Benthic invertebrate community statistical comparison results between the Sheardown Lake Nowrthwest basin (DLO1) and Reference Lake 3 littoral stations, Mary River Project CREMP, August 2016.

Metric	Statistical Test Results					Summary Statistics					
	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Power	Magnitude of Difference [b] (No. of SD)	Area	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	No	0.236	ι, δ, γ	-	-	Reference Lake 3	2,390	1,396	624	897	4,240
						SDNW Lake Littoral	5,503	4,184	1,871	1,415	10,484
Richness (Number of Taxa)	Yes	0.077	α, δ, γ	0.583	2.2	Reference Lake 3	12.2	1.1	0.5	11.0	14.0
						SDNW Lake Littoral	14.6	2.4	1.1	12.0	18.0
Simpson's (E) Krebs	No	0.008	γ	-	-	Reference Lake 3	0.758	0.189	0.084	0.420	0.849
						SDNW Lake Littoral	0.893	0.024	0.011	0.860	0.918
Bray-Curtis Index	Yes	0.002	α, ε, γ	1.000	2.8	Reference Lake 3	0.334	0.122	0.054	0.245	0.527
						SDNW Lake Littoral	0.669	0.037	0.016	0.628	0.711
Nemata (%)	No	1.000	γ	-	-	Reference Lake 3	4.0%	5.6%	2.5%	0.0%	13.5%
						SDNW Lake Littoral	1.1%	0.7%	0.3%	0.0%	2.0%
Hydracarina (%)	No	0.345	β, δ, γ	-	-	Reference Lake 3	3.6%	2.0%	0.9%	1.8%	6.7%
						SDNW Lake Littoral	4.2%	5.1%	2.3%	0.0%	11.3%
Ostracoda (%)	Yes	0.001	β, δ, γ	0.998	-2.2	Reference Lake 3	46.9%	17.5%	7.8%	37.8%	78.0%
						SDNW Lake Littoral	9.2%	6.1%	2.7%	3.7%	19.1%
Chironomidae (%)	Yes	0.002	β, δ, γ	0.992	2.1	Reference Lake 3	45.4%	18.8%	8.4%	15.4%	59.2%
						SDNW Lake Littoral	85.0%	6.6%	2.9%	76.3%	92.7%
Metal-Sensitive Chironomidae (%)	No	0.713	β, δ, γ	-	-	Reference Lake 3	19.3%	8.3%	3.7%	7.7%	28.1%
						SDNW Lake Littoral	24.6%	15.2%	6.8%	6.6%	41.0%
Scrapers (%)	No	0.571	β, δ, γ	-	-	Reference Lake 3	0.2%	0.4%	0.2%	0.0%	0.8%
						SDNW Lake Littoral	0.1%	0.3%	0.1%	0.0%	0.7%
Collector-Gatherers (%)	**Yes**	**0.025**	**β, δ, γ**	**0.805**	**-1.6**	Reference Lake 3	75.0%	11.4%	5.1%	61.1%	89.7%
						SDNW Lake Littoral	56.8%	7.7%	3.4%	47.6%	66.9%
Filterers (%)	No	0.803	β, ε, γ	-	-	Reference Lake 3	16.1%	8.4%	3.8%	7.0%	26.4%
						SDNW Lake Littoral	23.0%	17.3%	7.7%	3.7%	41.0%
Clingers (%)	No	0.922	β, δ, γ	-	-	Reference Lake 3	19.2%	7.6%	3.4%	8.8%	28.3%
						SDNW Lake Littoral	19.2%	5.8%	2.6%	11.0%	26.4%
Sprawlers (%)	No	0.095	γ	-	-	Reference Lake 3	65.7%	12.1%	5.4%	57.2%	85.7%
						SDNW Lake Littoral	53.0%	13.3%	6.0%	44.6%	75.6%
Burrowers (%)	No	0.156	β, δ	-	-	Reference Lake 3	15.1%	6.2%	2.8%	5.5%	22.2%
						SDNW Lake Littoral	27.8%	13.5%	6.0%	7.7%	44.2%

[a] Data analysis included: α - data untransformed; β - data logit transformed; ι - $\log_{10}$ transformed; δ - single factor ANOVA test conducted; ε - t-test assuming unequal variance; γ - ANOVA test validated using Mann Whitney U-test.

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

Highlighted values indicate significant differences between study areas based on ANOVA p-value less than 0.10 that were also outside of a Critical Effect Size of ±2 SD, suggesting an ecologically meaningful difference.

BOLD Bold text values indicate significant differences between study areas based on ANOVA p-value less than 0.10, but a Critical Effect Size within ±2 SD, suggesting the difference is not ecologically meaningful.

community was not unusually skewed towards relatively few taxa and thus, was not adversely altered.

Benthic invertebrate community structural differences were suggested between Sheardown Lake NW and Reference Lake 3 littoral habitats based on significantly higher Bray-Curtis Index at Sheardown Lake NW, and by significant differences in the relative abundance of dominant taxonomic groups and FFG between lakes (Table 4.4). Similar to Camp Lake, a significantly lower and higher relative abundance of Ostracoda (seed shrimp) and Chironomidae (non-biting midges) occurred, respectively, at Sheardown Lake NW compared to the reference lake. However, the relative abundance of metal-sensitive Chironomidae did not differ significantly between Sheardown Lake NW and Reference Lake 3 (Table 4.4), and therefore the difference in benthic invertebrate community structure between lakes did not appear to be associated with an ecological response to aqueous and/or sediment metals exposure. Rather, a significantly lower relative abundance of FFG collector-gatherers (which include seed shrimp) at Sheardown Lake NW compared to the reference lake (Table 4.4) suggested that the difference in benthic invertebrate community structure between lakes was related to differences in food resources. Because collector-gatherers are deposit feeders of coarse organic matter, the occurrence of significantly lower proportion of FFG collector-gatherers was consistent with significantly lower sediment TOC content at littoral stations of Sheardown Lake NW compared to Reference Lake 3 (Table 4.4). Benthic invertebrate community structural differences between Sheardown Lake NW and Reference Lake 3 did not appear to reflect different habitat conditions between littoral areas of these lakes given the lack of significant differences in HPG (Table 4.4). This was supported by sediment particle size analysis, which indicated that the proportion of dominant sand and silt particle sizes in sediment did not differ significantly between lakes (Appendix Table D.14).

Temporal comparisons of the Sheardown Lake NW benthic invertebrate community data indicated no significant differences in density, richness or Simpson's Evenness in 2016 compared to the 2007 and 2013 baseline studies (Figure 4.14; Appendix Table F.30). In addition, among the three dominant taxonomic groups and two FFG examined, only the relative abundance of Chironomidae differed significantly between the mine-operational and baseline periods at Sheardown Lake NW (Figure 4.14). However, this difference only occurred for data collected between 2015 and the baseline studies, and because there was no significant difference in the relative abundance of metal-sensitive Chironomidae in 2016 versus the baseline studies (Figure 4.14; Appendix Table F.30), no adverse mine-related response was suggested. Moreover, no consistent differences in benthic invertebrate community density, richness, Simpson's Evenness, FFG or HPG were indicated between Sheardown Lake NW

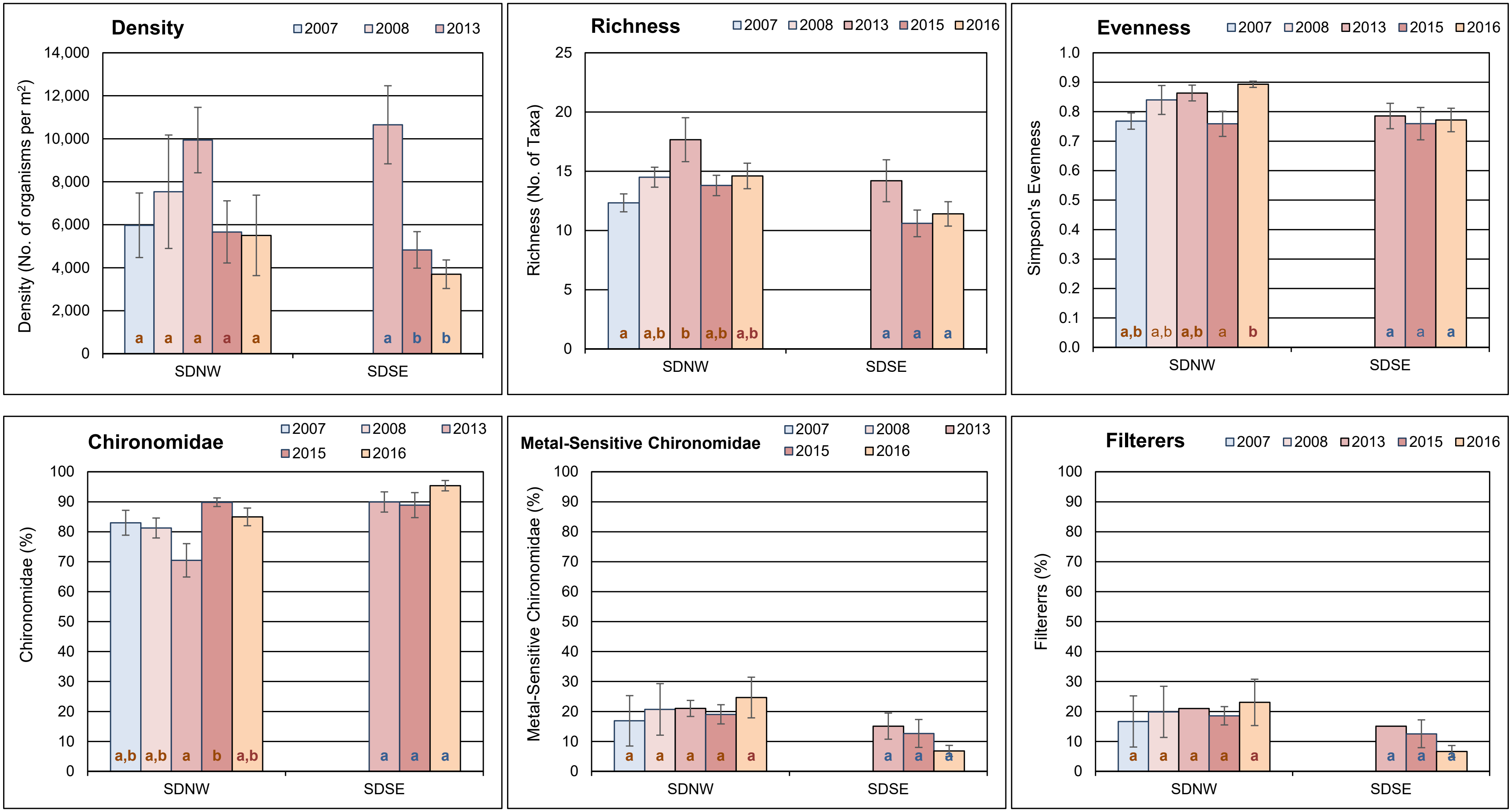


Figure 4.14: Comparison of key benthic invertebrate metrics (mean ± SE) at Sheardown Lake northwest (SDNW) and southeast (SDSE) basin littoral stations between mine baseline (2007, 2008 and 2013) and operational (2015, 2016) periods, Mary River Project CREMP, 2016. The same like-coloured letter inside bars indicate no significant difference among study years for respective lake basins.

and Reference Lake 3 littoral stations in both the 2015 and 2016 studies (Figure 4.14; Appendix Table F.30). Collectively, these results suggested no clear changes in benthic invertebrate community features in 2015/2016 compared to the baseline period, and specifically, no adverse influences associated with the recent initiation of mine operations at the Mary River Project.

4.2.5 Fish Population

4.2.5.1 Sheardown Lake NW Fish Community

Arctic charr was the only fish species captured at the northwest basin of Sheardown Lake in 2016, which differed slightly from that of Reference Lake 3 where low numbers of nine-spine stickleback were captured in nearshore rocky areas in addition to Arctic charr (Table 4.5). Total fish CPUE was much higher at Sheardown Lake NW than at the reference lake for nearshore electrofishing and for littoral/profundal gill net sampling (Table 4.5), suggesting higher densities and/or productivity of Arctic charr at Sheardown Lake. Greater numbers 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.

Temporal comparison of the Sheardown Lake NW electrofishing catch data indicated similar Arctic charr CPUE over the mine baseline (2006-2013), construction (2014) and operational (2015, 2016) periods at nearshore rocky habitat of the lake (Figure 4.15). In addition, the 2016 Arctic charr CPUE for gill net sampling was within the range shown during the baseline period (Figure 4.15). These results suggested that the relative abundance of Arctic charr at the nearshore and littoral/profundal areas of Sheardown Lake NW remained similar between the 2016 mine operational and baseline studies, which in turn, suggested no mine-related influences to Arctic charr numbers in the lake.

4.2.5.2 Sheardown Lake NW Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Sheardown Lake NW nearshore Arctic charr population were assessed using a control-impact analysis using data collected from Sheardown Lake NW and Reference Lake 3 in 2016, as well as a before-after analysis using data collected from Sheardown Lake NW in 2016 and during 2013 baseline characterization. A total of 100 Arctic charr were captured at nearshore habitat of each of Sheardown Lake NW and Reference Lake 3 in August 2016 for the control-impact analysis. Distinction of Arctic charr YOY from the older, non-YOY age class was possible using a fork length cut-off of 5.0 and 5.1 cm based on evaluation of length-frequency distributions coupled with supporting age determinations for the

Table 4.5: Fish catch and community summary from backpack electrofishing and gill netting conducted at Sheardown Lake NW (DLO-01), Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2016.

Lake	Method [a]		Arctic Charr	Nine-spine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	101	28	129	2
		CPUE	0.48	0.16	0.64	
	Gill netting	No. Caught	14	0	14	
		CPUE	0.15	0	0.15	
Sheardown Lake Northwest	Electrofishing	No. Caught	106	0	106	1
		CPUE	5.26	0	5.26	
	Gill netting	No. Caught	93	0	93	
		CPUE	1.71	0	1.71	
Sheardown Lake Southeast	Electrofishing	No. Caught	109	19	128	2
		CPUE	2.69	0.47	3.16	
	Gill netting	No. Caught	83	0	83	
		CPUE	8.06	0	8.06	

[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.

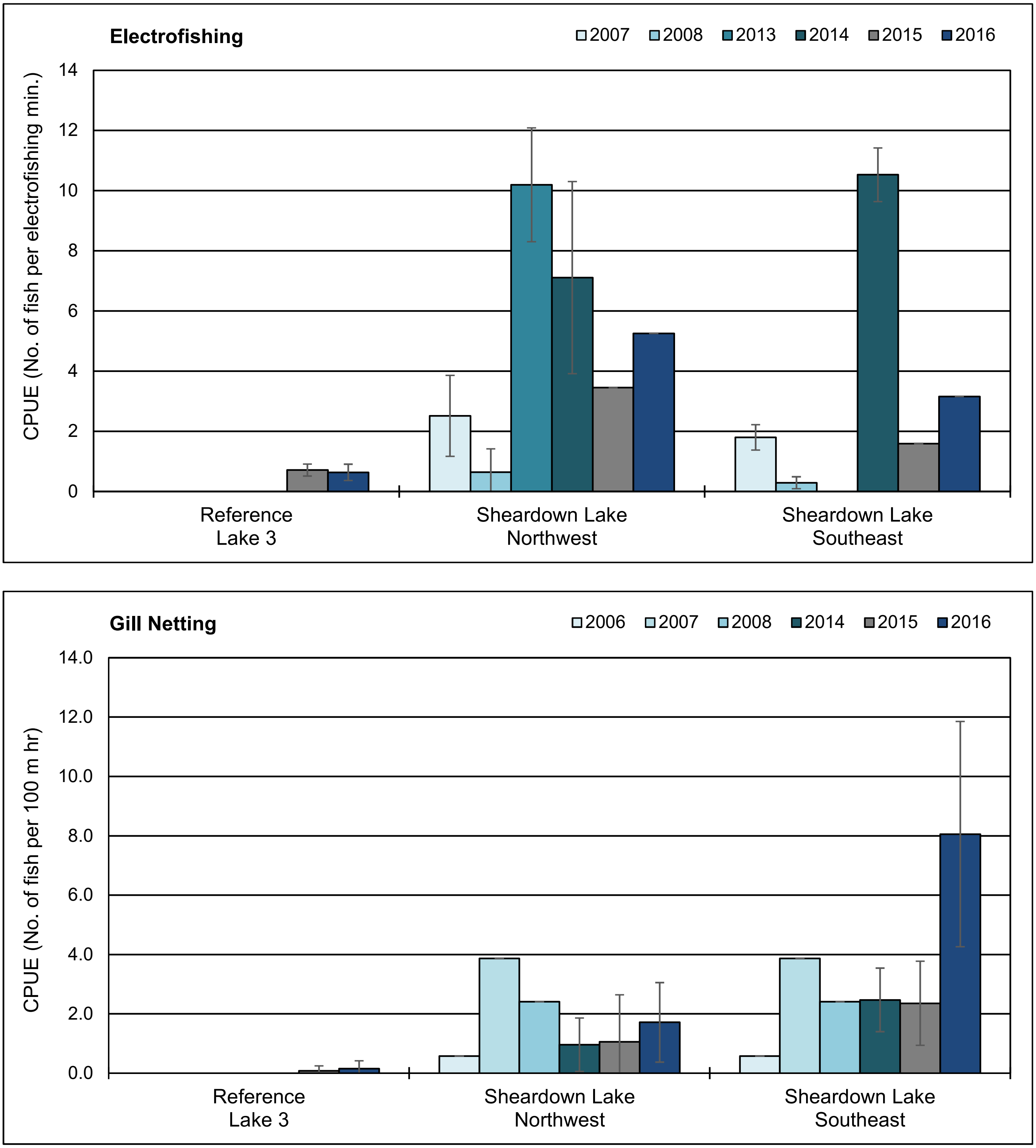


Figure 4.15: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic charr captured by backpack electrofishing and gill netting at Sheardown Lake NW (DLO-01) and Sheardown Lake SE (DLO-02) for baseline (2006, 2007, 2008, 2013), mine construction (2014) and operational (2015, 2016) periods in the fall, Mary River Project CREMP. Lake basins (i.e., NW or SE) were not differentiated historically for baseline gill netting catches.

Sheardown Lake NW and Reference Lake 3 data sets, respectively (Figure 4.16). The nearshore Arctic charr health comparisons involved separate assessment of the YOY and non-YOY data sets to account for naturally differing weight-at-length relationships that occur between these life stages.

Length-frequency distributions for the nearshore Arctic charr differed significantly between Sheardown Lake NW and Reference Lake 3 (Table 4.6), potentially reflecting a lower proportion of YOY and larger mean size of individuals captured at Sheardown Lake NW. Arctic charr YOY and non-YOY were significantly heavier and longer at the Sheardown Lake NW nearshore than at the reference lake nearshore (Table 4.6; Appendix Tables G.18 and G.19). In addition, Arctic charr captured at the Sheardown Lake NW nearshore grew significantly faster than those collected from the reference lake nearshore (Table 4.6; Appendix Tables G.18 and G.19). The magnitude of the differences in weight-based size and growth were outside of the ±25% CES_G, suggesting an ecologically meaningful difference in energy use between nearshore Arctic charr populations of Sheardown Lake NW and Reference Lake 3 for both the YOY and non-YOY size categories. However, no significant differences in condition (i.e., weight-at-length relationship) were indicated between nearshore Arctic charr populations of Sheardown Lake NW and Reference Lake 3 for both the YOY and non-YOY size categories in 2016 (Table 4.6; Appendix Tables G.18 and G.19). Overall, Arctic charr of the Sheardown Lake NW nearshore were significantly larger and grew significantly faster, but exhibited similar condition, compared to those of the reference lake. Similar to the fish population results at Camp Lake, the occurrence of significantly faster growing Arctic charr with similar condition at Sheardown Lake NW compared to the reference area suggested no adverse mine-related influences on Arctic charr health for juveniles residing within Sheardown Lake NW in 2016.

Temporal comparisons of the Sheardown Lake NW nearshore Arctic charr data indicated significantly different length-frequency distribution between the 2016 mine operational study and 2007/2013 baseline study data (Table 4.6; Appendix Table G.20). In addition, Arctic charr captured at the nearshore of Sheardown Lake NW in 2016 were significantly lighter and of significantly lower condition than those captured during mine baseline characterization (Table 4.6). For each of the significantly differing nearshore Arctic charr endpoints between 2016 and the baseline data, the magnitude of difference was outside of respective CES, suggesting that the differences were ecologically meaningful (Table 4.6; Appendix Table G.20). Although no differences in size were indicated, similar differences in nearshore Arctic charr condition were demonstrated between the previous 2015 mine operational study and the 2013 baseline study data (Table 4.6). Because a similar direction and magnitude of difference in juvenile Arctic charr condition was observed temporally at both Camp and

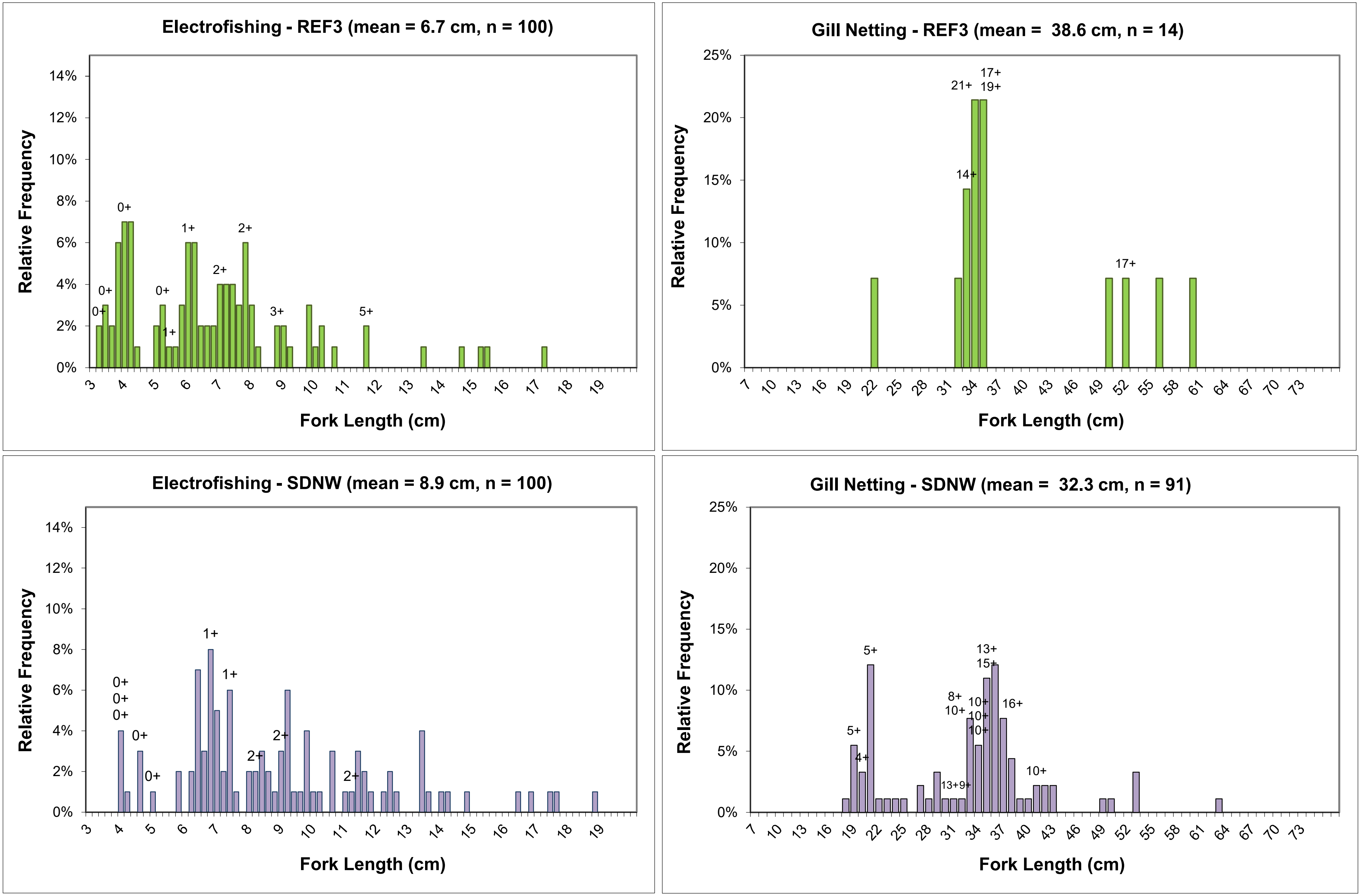


Figure 4.16: Length-frequency distributions for Arctic charr captured by backpack electrofishing and gill netting at Sheardown Lake NW (SDNW) and Reference Lake 3 (REF3), August 2016, Mary River Project CREMP. Fish ages are shown above the bars, where available.

Table 4.6: Summary of statistical results for Arctic charr population comparisons between Sheardown Lake NW and Reference Lake 3 for the mine operational period (2015, 2016) and between Sheardown Lake NW mine-operational and baseline period data for fish captured by electrofishing and gill netting methods, Mary River Project CREMP, August 2016. Values in parentheses indicate direction and magnitude of any significant differences.

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed?			
			versus Reference Lake 3		versus Sheardown Lake NW baseline period data [b]	
			2015	2016	2015	2016
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-
	Energy Use	Size (mean weight)	**Yes (+121%)**	**Yes (+60%)**	No	**Yes (-29%)**
		Size (mean fork length)	**Yes (+29%)**	**Yes (+17%)**	No	No
		Growth (weight-at-age)	**Yes (+156%)**	**Yes (+66%)**	No	-
		Growth (fork length-at-age)	**Yes (+38%)**	**Yes (+24%)**	No	-
	Energy Storage	Condition (body weight-at-fork length)	**Yes (+3%)**	No	**Yes (-13%)**	**Yes (-12%)**
Littoral/Profundal Gill Netting [a]	Survival	Length Frequency Distribution	-	-	**Yes**	**Yes**
		Age	-	-	**Yes (-35%)**	**Yes (-28%)**
	Energy Use	Size (mean weight)	-	-	**Yes (-47%)**	**Yes (-31%)**
		Size (mean fork length)	-	-	**Yes (-21%)**	**Yes (-14%)**
		Growth (weight-at-age)	-	-	No	No
		Growth (fork length-at-age)	-	-	No	No
	Energy Storage	Condition (body weight-at-fork length)	-	-	**Yes (+8%)**	**Yes (+11%)**

[a] Due to low catches of Arctic charr at Reference Lake 3 in 2015 and 2016, no comparison of fish health was possible for gill netted fish.

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

Sheardown Lake NW, this suggested that differences in condition between 2016 and the mine baseline periods likely reflected natural temporal variability.

Littoral/Profundal Arctic Charr

Mine-related influences on the Sheardown Lake NW littoral/profundal Arctic charr population were assessed using a before-after analysis between data collected in 2016 and the baseline characterization (combined 2006, 2007, 2008 and 2013) studies. Similar to the 2015 CREMP, a small sample size from Reference Lake 3 (i.e., n = 14) precluded meaningful control-impact statistical analysis using data collected in 2016. Biological information collected from Arctic charr mortalities indicated that non-spawners of reproductive age accounted for approximately 92% of the Sheardown Lake NW Arctic charr population at the time of sampling in August 2016 (Appendix Table G.23). The incidence rate for body cavity parasites was very high in the incidental Arctic charr mortalities (i.e., 86%), with sparse to very abundant occurrence of encysted worms and/or tapeworms observed in affected individuals (Appendix Table G.23). High incidence rates of internal parasites in Arctic charr were noted at Camp Lake in 2016, at all mine-exposed lakes in 2015 (Minnow 2016a), and at the various Mary River Project mine area lakes in baseline studies (NSC 2014, 2015a). One Arctic charr that had been tagged and released previously at Sheardown Lake NW was re-captured in 2016. This fish showed a 9.8 mm/year mean annual incremental increase in fork length over the approximately three years since being tagged (Table 4.7).

Table 4.7: Fork length and weight measurement data for tagged Arctic charr captured at Sheardown Lake NW in August 2016, Mary River Project CREMP.

Fish Tag Number	Capture Information			Re-Capture Information			Growth Rate
	Date of Capture	Length (mm)	Weight (g)	Date of Capture	Length (mm)	Weight (g)	Δ Length (mm/yr)
77647	30-Aug-2013	330	400	12-Aug-2016	359	470	9.8

The length-frequency distribution for Arctic charr captured at littoral/profundal areas of Sheardown Lake NW in 2016 differed significantly from those captured during baseline monitoring (Table 4.6; Figure 4.16). In part, the differences in length-frequency distribution may have reflected significantly younger and smaller individuals captured in 2016 compared to the baseline period (Table 4.6). Arctic charr growth did not differ significantly between 2016 and the baseline period at Sheardown Lake NW (Table 4.6; Appendix Table G.24). However, Arctic charr captured at littoral/profundal areas of Sheardown Lake NW exhibited significantly greater condition in 2016 than during baseline monitoring, at a magnitude of the difference slightly outside of the ecologically relevant CES_C of ±10% (Table 4.6; Appendix Table G.24). Notably, the same type and direction of differences in length-frequency distribution, age, mean size and condition for Arctic charr captured at littoral/profundal areas of Sheardown Lake NW were consistently demonstrated in 2015 and 2016 relative to the mine baseline data (Table 4.6). Overall, the lack of significant differences in growth combined with significantly greater condition of Arctic charr captured at littoral/profundal areas of Sheardown Lake NW in 2016 versus the baseline period suggested no adverse mine-related influences on the adult Arctic charr population of the lake as a result of on-going mine operation.

4.3 Sheardown Lake SE (DLO-2)

4.3.1 Water Quality

Vertical water quality profiles of *in-situ* water temperature, dissolved oxygen, pH and specific conductance conducted at Sheardown Lake SE showed no substantial station-to-station differences during any of the winter, summer or fall sampling events in 2016 (Appendix Figures C.14 to C.17). No thermal stratification was evident at the Sheardown Lake SE basin during any of the winter, summer or fall sampling events, and although gradually cooler water was observed with increased depth during summer and fall, no distinct layers had developed (Figure 4.17). The summer and fall water temperature profiles at Sheardown Lake SE were similar to those from the reference lake, with highest gradients in temperature with depth occurring between 5 - 10 m in summer and 10 – 17 m in fall (Figure 4.17). Mean water temperature near the bottom of the water column at littoral stations in fall 2016 did not differ significantly between Sheardown Lake SE and Reference Lake 3 (Figure 4.8; Appendix Table C.45). Notably, Sheardown Lake SE is a much 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 2016 showed no change in dissolved oxygen saturation with depth during summer, but oxycline development characterized by decreasing saturation levels with increasing depth occurring at depths greater than 10 m during the winter and fall sampling events (Figure 4.17). No oxycline had developed in summer and fall at Reference Lake 3 in 2016 (Figure 4.17). Despite the differences in dissolved oxygen profiles, saturation levels at the bottom of the water column at littoral stations (i.e., approximately 10 m depth) did not differ significantly between the Sheardown Lake southeast basin and Reference Lake 3 during fall 2016 sampling (Figure 4.8; Appendix Tables C.44 – C.45). Dissolved oxygen saturation levels were generally well above the WQG of 54% at Sheardown Lake SE at all depths during the summer and fall sampling events in 2016

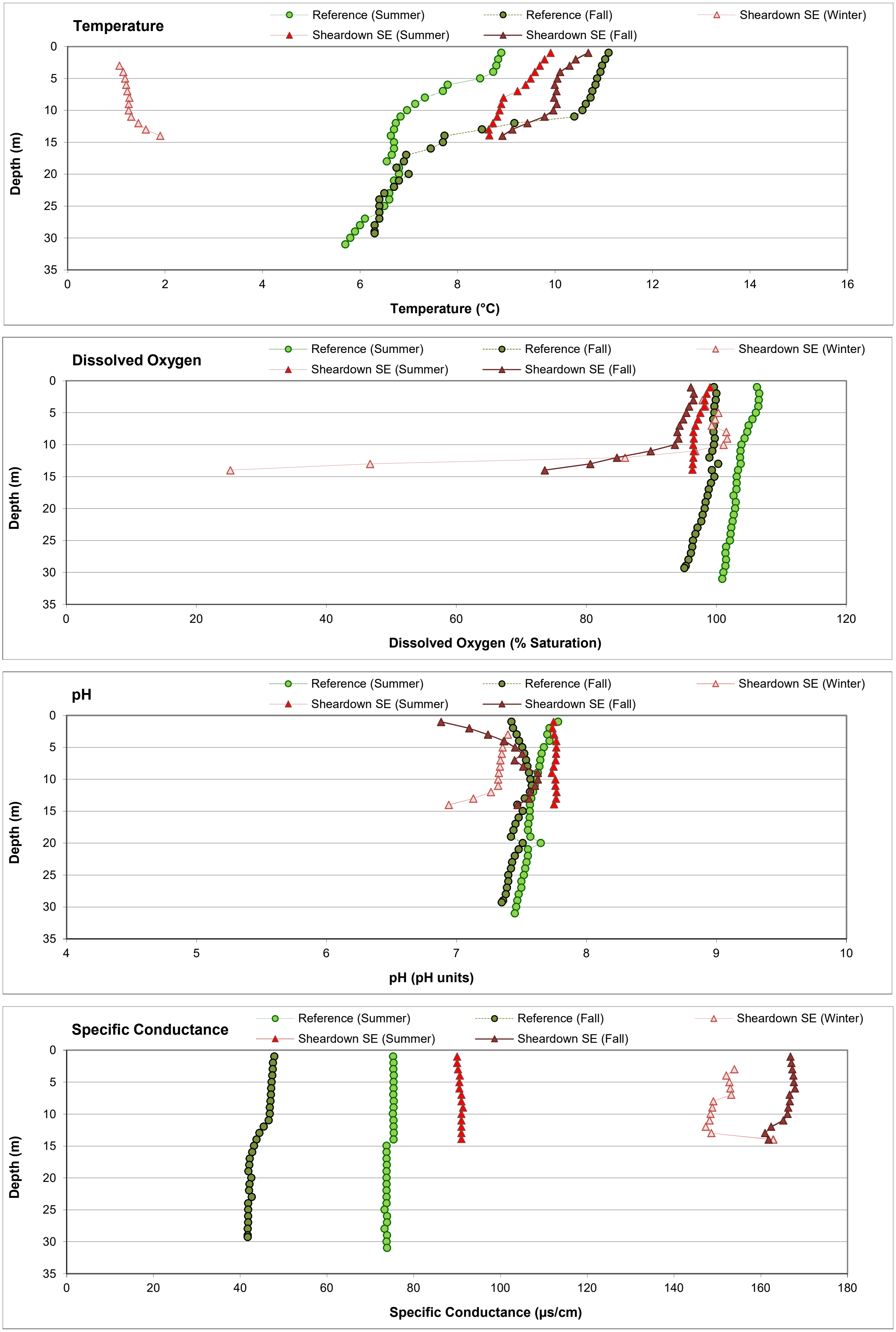


Figure 4.17: Average *in-situ* water quality with depth from surface at Sheardown Lake SE (mine-exposed area) compared to Reference Lake 3 during winter, summer, and fall sampling events, Mary River Project CREMP, 2016.

(Figure 4.8), indicating that dissolved oxygen was not likely to be limiting to pelagic or bottom-dwelling biota within the lake. However, dissolved oxygen saturation levels below 54% were observed at depths greater than 13 m during the winter at Sheardown Lake SE, the cause of which may be related to natural (e.g., sediment TOC content) or mine-related (e.g., current/historical STP inputs) influences to lake dissolved oxygen levels.

In-situ profiles of pH and specific conductance showed no substantial change from the surface to the bottom of the Sheardown Lake SE water column, indicating no chemical stratification (Figure 4.17). Similar to the northwest basin, no significant differences in bottom pH at littoral stations were indicated between Sheardown Lake SE and Reference Lake 3 during the 2016 fall sampling, with pH at the southeast basin of Sheardown Lake also consistently within WQG limits in 2016 (Figure 4.8; Appendix Tables C.44; Figure 4.17). Specific conductance was significantly higher at Sheardown Lake SE compared to the reference lake during 2016 fall sampling (Figure 4.8). However, specific conductance at Sheardown Lake SE was intermediate to that of the reference creek and river areas (i.e., mean of 101 and 133 µS/cm, respectively) in fall 2016, and therefore the extent to which higher specific conductance at Sheardown Lake SE was related to natural regional variability or a mine-related influence was unclear. Water clarity at the southeast basin of Sheardown Lake was the lowest among the mine-exposed lakes. Secchi depth readings from Sheardown Lake SE were significantly lower (shallower) than at Reference Lake 3 during the 2016 fall sampling event, but were relatively consistent among stations, suggesting no spatial differences in water clarity of the lake (Appendix Tables C.44 – C.45).

Water chemistry at Sheardown Lake SE showed no consistent spatial differences in parameter concentrations among the five sampling stations during any of the winter, summer or fall sampling events in 2016 (Table 4.8; Appendix Table C.46), suggesting that the lake waters were generally well mixed both laterally and vertically. Total aluminum concentrations were highly elevated (i.e., ≥10-fold), turbidity and concentrations of total manganese moderately elevated (i.e., 5- to 10-fold), and concentrations of phenols and total molybdenum slightly elevated (i.e., 3- to 5-fold), at Sheardown Lake SE compared to Reference Lake 3 during the 2016 summer and fall sampling events (Table 4.8; Appendix Tables C.40 and C.46). Similar to the northwest basin, aluminum and manganese concentrations showed strong and modest positive correlations with turbidity, respectively, for the Sheardown Lake SE combined data set (i.e., winter, summer and fall data; r^2 = 0.90 and 0.60, respectively), suggesting that much of the aqueous aluminum and manganese was associated with suspended particles. This was corroborated by comparison of total and dissolved fractions for these metals, which indicated that most (i.e., ≥75%) was in particulate form at Sheardown Lake SE (compare Appendix

Table 4.8: Water chemistry at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3) monitoring stations, Mary River Project CREMP, August 2016. Values presented are averages of samples taken from the surface and the bottom of the water column at each station.

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Lake 3 Average (n = 3)	Sheardown Lake Southeast (SDSE) Station				
						DL0-02-6	DL0-02-7	DL0-02-4	DL0-02-8	DL0-02-3
					Fall 2016	21-Aug-16	21-Aug-16	21-Aug-16	21-Aug-16	21-Aug-16
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	84.3	118	116	116	115	113
	pH (lab)	pH	6.5 - 9.0	-	7.68	8.10	8.10	8.01	8.05	7.91
	Hardness (as $CaCO_3$)	mg/L	-	-	35	56	55	54	54	54
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	2.7	2.1	2.2	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	39	57	60	62	62	63
	Turbidity	NTU	-	-	0.33	2.05	2.21	2.45	2.40	2.62
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	52	52	52	53	52
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.040	<0.020	0.025	<0.020	0.032	0.036
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	0.022
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	0.16
	Dissolved Organic Carbon	mg/L	-	-	2.68	1.45	1.50	1.70	1.55	1.30
	Total Organic Carbon	mg/L	-	-	2.78	1.90	1.78	1.55	1.65	1.55
	Total Phosphorus	mg/L	0.020[α]	-	0.0099	0.0064	0.0115	0.0060	0.0119	0.0138
	Phenols	mg/L	0.004[α]	-	0.0031	0.0022	0.0025	0.0020	0.0299	0.0151
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.27	2.29	2.27	2.23	2.22	2.25
	Sulphate (SO_4)	mg/L	218[β]	218	4.07	2.71	2.65	2.59	2.62	2.57
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[c]	0.0042	0.052	0.051	0.070	0.052	0.059
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.00653	0.00618	0.00637	0.00659	0.00630	0.00622
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	6.99	11.05	11.15	10.95	11.1	10.65
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.00082	0.001	0.00115	0.00105	0.0011	0.001
	Iron (Fe)	mg/L	0.30	0.300	<0.030	0.0575	0.0525	0.0765	0.0565	0.066
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.00010	<0.00010	<0.00010	<0.00010	0.000105
	Lithium (Li)	mg/L	-	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Magnesium (Mg)	mg/L	-	-	4.26	6.45	6.39	6.31	6.34	6.11
	Manganese (Mn)	mg/L	0.935[β]	-	0.00062	0.00373	0.00365	0.00451	0.00386	0.00561
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.000136	0.000497	0.00050625	0.0004925	0.0004955	0.000459
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	0.000665	0.0007	0.000705	0.0007	0.00069
	Potassium (K)	mg/L	-	-	0.89	0.93	0.92	0.93	0.92	0.90
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Silicon (Si)	mg/L	-	-	0.420	0.687	0.615	0.667	0.617	0.685
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Sodium (Na)	mg/L	-	-	0.836	1.140	1.135	1.120	1.125	1.110
	Strontium (Sr)	mg/L	-	-	0.0081	0.0083	0.0085	0.0084	0.0085	0.0082
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<0.010	0.00234	0.00204	0.00327	0.00220	0.00275
	Uranium (U)	mg/L	0.015	-	0.000270	0.000798	0.000811	0.0007975	0.0007975	0.000748
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0010	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030
	Zirconium (Zr)	mg/L	-	-		<0.00030	<0.00030	<0.00030	<0.00030	<0.00030

[a] Canadian Water Quality Guideline (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). 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 Sheardown Lake SE.

[c] Benchmark is 0.179 mg/L and 0.173 mg/L for shallow and deep stations, respectively.

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

BOLD Indicates parameter concentration above the AEMP benchmark.

Tables C.46 and C.47). Higher turbidity at Sheardown Lake SE, and lower water clarity (Secchi depth) associated with this turbidity, likely reflected backflow received from the Mary River, which directly affects water levels and chemistry of the southeast basin during moderate to high flow periods. In contrast to aluminum and manganese, molybdenum concentrations at Sheardown Lake SE were not associated with greater turbidity, suggesting that slight elevation molybdenum compared to Reference Lake 3 was related to mine operation and/or natural geochemical differences between these lakes. Despite elevation of these metals at Sheardown Lake SE, concentrations of most parameters were well below established WQG and AEMP benchmarks during the winter, summer and fall sampling events in 2016[10] (Table 4.8; Appendix Table C.46), suggesting no adverse influences of water quality on biota of Sheardown Lake SE in 2016.

Temporal comparisons of the Sheardown Lake SE water chemistry data indicated no appreciable changes in average concentrations of parameters between the 2016 study and mine baseline (2005 – 2013) period (Figure 4.9; Appendix Figure C.18). This suggested that the differences in water chemistry between Sheardown Lake SE and Reference Lake 3 in 2016 likely reflected natural differences in mineralogy/geochemical conditions between lakes. Nevertheless, conductivity, hardness and concentrations of chloride, manganese, nickel, sodium, strontium, sulphate and uranium were consistently greater at all Sheardown Lake SE stations in 2016 compared to the previous years of mine construction (2014) and initial operation (2015; Figure 4.9; Appendix Figure C.18). Higher concentrations of these parameters in 2016 may have reflected natural temporal variability in water chemistry, but may also indicate a more recent, slight mine-related influence on water quality of Sheardown Lake SE.

4.3.2 Sediment Quality

Surficial sediment at Sheardown Lake SE littoral stations was uniformly composed of compact silty loam material with low TOC content (Figure 4.18). Substrate at littoral stations of Sheardown Lake SE contained significantly lower sand and TOC content, but significantly greater silt and clay content, than at Reference Lake 3 (Appendix Table D.19). The high proportion of fines in substrate of Sheardown Lake SE potentially reflects the receipt of Mary River backflow during high flow periods, which can be expected to result in the deposition of high quantities of naturally suspended, fine-grained material. Similar to observations at the other mine-exposed lakes and the reference lake, iron (oxy)hydroxide material was visible in surficial and/or sub-surface substrate of Sheardown Lake SE, in some cases occurring as a

[10] Refer to footnote 2 (page 23) and Appendix B regarding phenol concentrations above WQG at the mine-exposed and reference areas of the Mary River Project LSA waterbodies.

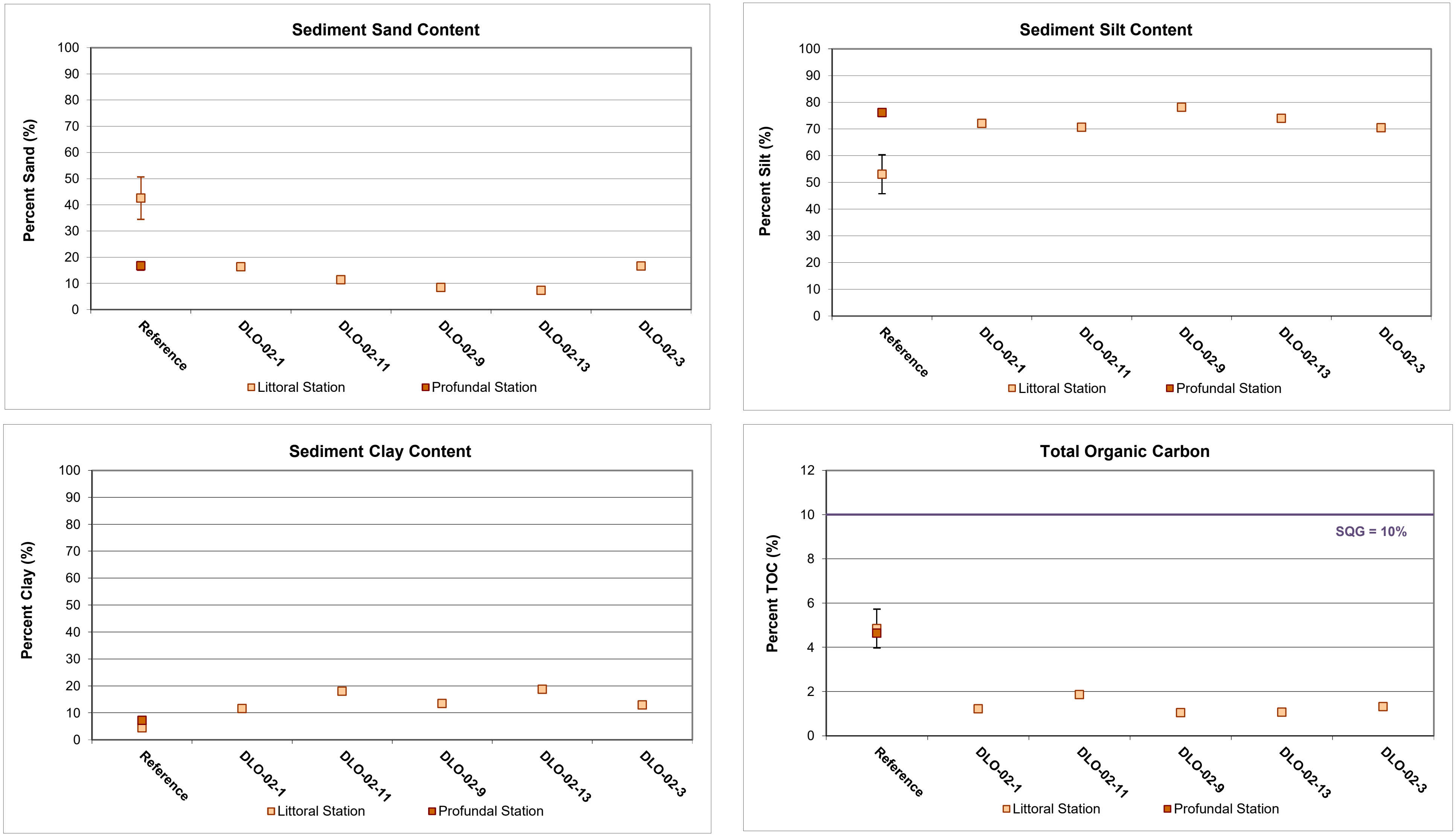


Figure 4.18: Sediment particle size and total organic carbon (TOC) content comparisons among Sheardown Lake SE (DLO-02) sediment monitoring stations and Reference Lake 3 averages (mean ± SE), Mary River Project CREMP, August 2016.

thin, distinct layer (Appendix Tables D.17 – D.18). Below the surficial layer, substrates at Sheardown Lake SE littoral stations exhibited some sporadic blackening and, at one station, had a slight sulphidic odour, suggesting development of reducing conditions. However, no distinct redox boundary was observed in the littoral station sediments (Appendix Tables D.17 - D.18). Observations regarding reducing sediment conditions at Sheardown Lake SE were similar to those made at Reference Lake 3 in 2016 (Appendix Tables D.1 – D.3 and D.17 – D.18), suggesting that factors leading to reduced sediment conditions were comparable between lakes.

Sediment metal concentrations at Sheardown Lake SE showed no spatial gradients with progression towards the lake outlet in 2016, suggesting no clear point sources of metals to the lake (Appendix Table D.20). With the exception of slightly elevated manganese concentrations, sediment metal concentrations at littoral stations of Sheardown Lake SE were, on average, similar to those observed at the reference lake littoral stations (Table 4.3; Appendix Tables D.20 – D.21). Mean iron and manganese concentrations were above respective SQG and AEMP benchmarks at the Sheardown Lake SE littoral stations, although concentrations of these metals were above SQG at only two of the five stations sampled (Table 4.3; Appendix Table D.20). As indicated previously, average concentrations of iron and manganese were also above respective SQG at profundal stations of Reference Lake 3 (Table 4.3). These results suggested that the elevation of iron and manganese concentrations in sediment of Sheardown Lake SE relative to SQG potentially reflected a natural phenomenon at lakes in the mine LSA.

Temporal comparisons of the sediment metals data suggested slightly elevated (i.e., 2- to 5-fold higher) average concentrations of arsenic and manganese at littoral stations of Sheardown Lake SE in 2016 compared to the baseline (2005 – 2013) period[11] (Figure 4.11; Appendix Table D.21). Arsenic and manganese showed progressively higher mean concentrations in 2015 and 2016 compared to the baseline/mine construction periods at littoral stations of Sheardown Lake SE (Figure 4.11). However, as at the other mine-exposed lakes, variability in parameter concentrations was high and neither parameter occurred at concentrations greater than at the reference lake littoral and/or profundal stations (Figure 4.11). This suggested that, similar to the other mine-exposed lakes, slight variation in station locations and/or data treatment among studies likely contributed to the appearance of higher mean concentrations of arsenic and manganese in sediment at the Sheardown Lake SE littoral stations in 2015 and 2016 compared to the baseline period. Nevertheless, because arsenic

[11] Refer to footnote 6 (page 32) regarding temporal differences in sediment boron concentrations at Mary River Project LSA waterbodies.

and/or manganese showed progressively higher mean concentrations in sediment at all the other mine-exposed lakes, and despite similarity to the reference lake sediment metal concentrations, greater concentrations of these parameters in sediment at the Sheardown Lake SE littoral stations over time potentially reflected a mine-related influence.

4.3.3 Phytoplankton

Chlorophyll a concentrations at Sheardown Lake SE showed no spatial gradients with closer proximity to the lake outlet during any of the winter, summer or fall sampling events in 2016 (Figure 4.12). Chlorophyll a concentrations were significantly higher in the fall than during the either the winter or summer seasons in 2016, with comparable concentrations shown between the latter (Appendix Table E.10). Similar to Camp Lake and Sheardown Lake NW, chlorophyll a concentrations at the Sheardown Lake SE were significantly higher than at the reference lake for both the summer and fall sampling events in 2016 (Appendix Table E.5 and E.6). Moreover, chlorophyll a concentrations were well below the AEMP benchmark of 3.7 µg/L at all but one of the Sheardown Lake SE stations during the winter, summer and fall sampling events in 2016 (Figure 4.12). On average, chlorophyll a concentrations at Sheardown Lake SE fell within an 'oligotrophic' trophic status as defined by Wetzel (2001), although chlorophyll a concentrations at individual stations fell near the maxima for this designation during the fall 2016 sampling event. Mean aqueous total phosphorus concentrations at Sheardown Lake SE were also near the oligotrophic-mesotrophic boundary designation of 10 µg/L during the 2016 summer and fall sampling events (Table 4.8; Appendix Table C.46).

Chlorophyll a concentrations were significantly higher in the summer and fall of 2016 than during the same seasons in 2014 and/or 2015 at Sheardown Lake SE, but no significant differences in chlorophyll a concentrations were shown among years for data collected in the winter (Appendix Table E.11). Annual average chlorophyll a concentrations were significantly higher in 2016 than 2015, and although concentrations did not differ significantly between 2014 and 2016, higher absolute concentrations in 2016 were suggestive of slightly increased primary productivity over time at Sheardown Lake SE, particularly during the ice-free period. This suggested that the trophic status may have increased at Sheardown Lake SE since the mine-construction period, potentially representing a mine-related influence to the lake. No chlorophyll a baseline (2005 – 2013) data are available for Sheardown Lake SE, precluding comparisons to conditions prior to the mine construction period.

4.3.4 Benthic Invertebrate Community

Benthic invertebrate density, richness and Simpson's Evenness at littoral stations did not differ significantly between Sheardown Lake SE and Reference Lake 3 in 2016 (Table 4.9).

Table 4.9: Benthic invertebrate community statistical comparison results between Southeast Sheardown Lake (DLO2) and Reference Lake 3 littoral stations, Mary River Project CREMP, August 2016.

Metric	Statistical Test Results					Summary Statistics					
	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Power	Magnitude of Difference [b] (No. of SD)	Area	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	No	0.189	α, δ, γ	-	-	Reference Lake 3	2,390	1,396	624	897	4,240
						SDSE Lake Littoral	3,700	1,485	664	2,343	6,225
Richness (Number of Taxa)	No	0.510	α, ε, γ	-	-	Reference Lake 3	12.2	1.1	0.5	11.0	14.0
						SDSE Lake Littoral	11.4	2.3	1.0	9.0	14.0
Simpson's (E) Krebs	No	0.548	γ	-	-	Reference Lake 3	0.758	0.189	0.084	0.420	0.849
						SDSE Lake Littoral	0.772	0.089	0.040	0.696	0.900
Bray-Curtis Index	Yes	<0.001	α, δ, γ	1.000	3.5	Reference Lake 3	0.334	0.122	0.054	0.245	0.527
						SDSE Lake Littoral	0.761	0.082	0.037	0.669	0.886
Nemata (%)	No	0.445	β, δ, γ	-	-	Reference Lake 3	4.0%	5.6%	2.5%	0.0%	13.5%
						SDSE Lake Littoral	1.1%	1.3%	0.6%	0.0%	2.9%
Ostracoda (%)	Yes	0.001	β, δ, γ	0.997	-2.6	Reference Lake 3	46.9%	17.5%	7.8%	37.8%	78.0%
						SDSE Lake Littoral	1.7%	2.5%	1.1%	0.0%	5.9%
Chironomidae (%)	Yes	0.000	β, δ, γ	1.000	2.7	Reference Lake 3	45.4%	18.8%	8.4%	15.4%	59.2%
						SDSE Lake Littoral	95.4%	3.9%	1.7%	89.7%	98.9%
Metal-Sensitive Chironomidae (%)	**Yes**	**0.020**	**β, δ, γ**	**0.838**	**-1.5**	Reference Lake 3	19.3%	8.3%	3.7%	7.7%	28.1%
						SDSE Lake Littoral	6.8%	4.2%	1.9%	1.9%	12.2%
Collector-Gatherers (%)	**Yes**	**0.045**	**β, δ, γ**	**0.697**	**-1.6**	Reference Lake 3	75.0%	11.4%	5.1%	61.1%	89.7%
						SDSE Lake Littoral	56.5%	12.8%	5.7%	42.6%	71.0%
Filterers (%)	**Yes**	**0.063**	**β, δ, γ**	**0.627**	**-1.1**	Reference Lake 3	16.1%	8.4%	3.8%	7.0%	26.4%
						SDSE Lake Littoral	6.7%	4.4%	2.0%	1.1%	12.2%
Clingers (%)	**Yes**	**0.056**	**γ**	**-**	**-1.5**	Reference Lake 3	19.2%	7.6%	3.4%	8.8%	28.3%
						SDSE Lake Littoral	8.1%	3.9%	1.8%	1.7%	12.0%
Sprawlers (%)	**Yes**	**0.054**	**β, δ, γ**	**0.661**	**-1.6**	Reference Lake 3	65.7%	12.1%	5.4%	57.2%	85.7%
						SDSE Lake Littoral	46.2%	14.0%	6.3%	28.0%	63.1%
Burrowers (%)	Yes	0.002	β, δ, γ	0.994	4.9	Reference Lake 3	15.1%	6.2%	2.8%	5.5%	22.2%
						SDSE Lake Littoral	45.6%	12.5%	5.6%	33.3%	64.6%

[a] Data analysis included: α - data untransformed; β - data logit transformed; ι - log_{10} transformed; δ - single factor ANOVA test conducted; ε - t-test assuming unequal variance; γ - ANOVA test validated using Mann Whitney U-test.

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

Highlighted values indicate significant differences between study areas based on ANOVA p-value less than 0.10 that were also outside of a Critical Effect Size of ±2 SD, suggesting an ecologically meaningful difference.

BOLD Bold text values indicate significant differences between study areas based on ANOVA p-value less than 0.10, but a Critical Effect Size within ±2 SD, suggesting the difference is not ecologically meaningful.

However, benthic invertebrate community structural differences were indicated between Sheardown Lake SE and reference lake littoral habitats based on significantly differing Bray-Curtis Index and by significant differences in the relative abundance of dominant taxonomic groups, FFG and HPG between lakes (Table 4.9). Similar to the northwest basin of Sheardown Lake, significant differences in the relative abundance of dominant taxonomic groups (i.e., seed shrimp and chironomids) and FFG between the Sheardown Lake SE and reference lake littoral stations were potentially linked to differing food resources between lakes. Specifically, a significantly lower sediment TOC content potentially accounted for lower relative abundance of seed shrimp and the collector-gatherer FFG at Sheardown Lake SE than at the reference lake. The analysis of HPG suggested that differences in habitat also could have accounted for benthic invertebrate community structural differences between Sheardown Lake SE and Reference Lake 3 littoral areas. For instance, a significantly higher relative abundance of burrowing benthic invertebrates was consistent with the occurrence of significantly higher proportion of fines (i.e., silt and clay) in substrate of Sheardown Lake SE compared to the reference lake (Appendix Table D.19). Finer substrate composition at Sheardown Lake SE would presumably provide more suitable habitat quality for burrowing invertebrates, thus accounting for some of the differences in community structure between Sheardown Lake SE and Reference Lake 3. Lower sediment TOC and differences in sediment particle size largely reflect natural differences in habitat features between Sheardown Lake SE and the reference lake, including potential influences of backflow from the Mary River to Sheardown Lake SE during periods of high flow that would result in the deposition of fines to the lake.

Temporal comparisons of the Sheardown Lake SE benthic invertebrate community data indicated significantly lower density in 2015 and 2016 mine operational years compared to 2013 baseline period data, but no significant differences in density between 2015 and 2016 (Figure 4.14; Appendix Table F.31). In addition, richness, Simpson's Evenness, and the relative abundance of dominant taxonomic groups and FFG did not differ significantly among the mine operational and mine baseline studies (Figure 4.14; Appendix Table F.31). Because density was the only benthic invertebrate community metric that differed among periods, natural variability in density among studies most likely accounted for this difference. This was supported by the facts that no significant difference in the proportion of metal-sensitive taxa was indicated among years (Figure 4.14) and parameter concentrations in water and sediment were below applicable WQG/SQG and AEMP benchmarks at Sheardown Lake SE in 2016[12]. Consistent differences in benthic invertebrate community dominant taxonomic groups, FFG

[12] Although sediment iron and manganese concentrations were above SQG at littoral stations of Sheardown Lake SE in 2016, concentrations of these metals were also above SQG at profundal stations of Reference Lake 3, suggesting iron concentrations were naturally high within the mine LSA lakes.

and HPG were indicated between Sheardown Lake SE and Reference Lake 3 littoral stations in both the 2015 and 2016 studies, in addition to an overall greater number of significantly differing endpoints in 2016 compared to 2015 (Table 4.9; Appendix Table F.31). This suggested that factors contributing to differences in benthic invertebrate community structure between Sheardown Lake SE and Reference Lake 3 in both 2015 and 2016 had remained relatively unchanged between years.

4.3.5 Fish Population

4.3.5.1 Sheardown Lake SE Fish Community

The Sheardown Lake SE fish community was composed of Arctic charr and ninespine stickleback, reflecting the same fish species composition as the reference lake in 2016 (Table 4.6). However, total fish CPUE was much higher at Sheardown Lake SE than at Reference Lake 3 for both electrofishing and gill netting collection methods, suggesting higher densities and/or productivity of both Arctic charr and ninespine stickleback at Sheardown Lake SE (Table 4.6). Consistent with the other mine 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.

Temporal comparison of the Sheardown Lake SE electrofishing catch data indicated that fish CPUE was highly variable among the mine baseline (2007 - 2008), construction (2014) and operational (2015, 2016) studies (Figure 4.15). Nevertheless, the abundance of Arctic charr at nearshore habitat of Sheardown Lake SE following the initial two years of mine operation (i.e., 2015 – 2016) was within the range observed prior to mine start-up. Arctic charr CPUE for gill net collections was markedly higher in 2016 compared to all previous baseline (2006 – 2008), mine construction (2014) and mine operational (2015) studies (Figure 4.15). However, similar to 2016 results at Camp Lake, the higher CPUE at Sheardown Lake SE in 2016 likely reflected improvements in sampling efficiency from experienced gained through previous studies (see Minnow 2016b) rather than higher fish densities/productivity at the lake in 2016. Nevertheless, CPUE comparisons between studies suggested that the relative abundance of Arctic charr in Sheardown Lake SE had not been reduced in 2016 compared to baseline conditions or to the previous mine construction and mine operation years.

4.3.5.2 Sheardown Lake SE Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Sheardown Lake SE nearshore Arctic charr population were assessed with a control-impact analysis using data collected from Sheardown Lake SE and

Reference Lake 3 in 2016. Although before-after analysis of data collected from Sheardown Lake SE in 2016 (mine operation) and 2007 (baseline) was conducted, poor accuracy in fresh body weight measures during the baseline sampling precluded meaningful data interpretation and therefore these results were not discussed further herein. A total of 100 Arctic charr were captured at nearshore habitat of each of Sheardown Lake SE and Reference Lake 3 in August 2016 for the control-impact analysis. Distinction of Arctic charr YOY from the older, non-YOY age category was possible using a fork length cut-off of 5.0 and 5.1 cm based on evaluation of length-frequency distributions coupled with supporting age determinations for the Sheardown Lake SE and Reference Lake 3 data sets, respectively (Figure 4.19). Nearshore Arctic charr health comparisons were conducted separately for the YOY and non-YOY data sets to account for naturally differing weight-at-length relationships that occur between these age categories.

Length-frequency distributions for the nearshore Arctic charr differed significantly between Sheardown Lake SE and Reference Lake 3 (Table 4.10), potentially reflecting a greater prevalence of YOY and smaller mean size of individuals captured at Sheardown Lake SE (Figure 4.19). Although Arctic charr YOY were significantly heavier and longer at the Sheardown Lake SE nearshore than at the reference lake nearshore, the size of non-YOY did not differ significantly between lakes in 2016 (Table 4.10; Appendix Tables G.26 – G.27). Similar to the northwest basin, Arctic charr captured at the Sheardown Lake SE nearshore grew significantly faster than those collected from the reference lake nearshore (Table 4.10). The magnitude of the differences in weight- and length-based growth were well outside of the ecologically meaningful CES_G of ±25% between Sheardown Lake SE and the reference lake (Table 4.10). However, as at the northwest basin, no significant differences in condition of nearshore Arctic charr were indicated between Sheardown Lake SE and the reference lake for YOY and non-YOY individuals in 2016 (Appendix Tables G.26 – G.27). Similar to the other mine-exposed lakes, the occurrence of faster growing Arctic charr with similar condition to those of the reference lake suggested no adverse mine-related influences on fish energy use and storage at Sheardown Lake SE in 2016.

Littoral/Profundal Arctic Charr

Mine-related influences on the Sheardown Lake SE littoral/profundal Arctic charr population was assessed using a before-after analysis between data collected in 2016 and the baseline characterization (combined 2007/2008) studies. Similar to the 2015 CREMP, a small sample size from Reference Lake 3 (i.e., n = 14) precluded meaningful control-impact statistical analysis using data collected in 2016. Biological information collected from Arctic charr mortalities indicated that non-spawners of reproductive age constituted approximately 57% of

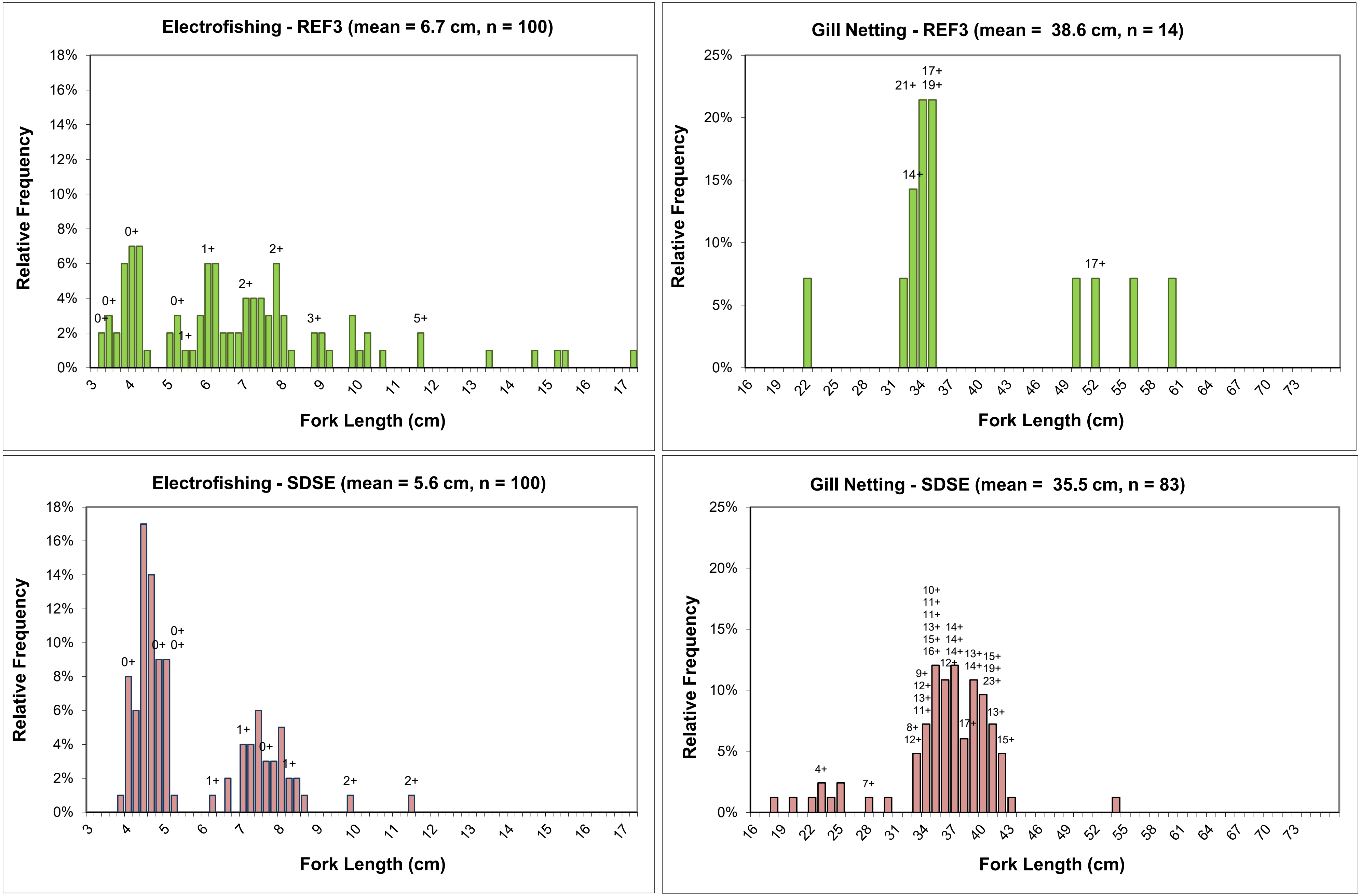


Figure 4.19: Length-frequency distributions for Arctic charr captured by backpack electrofishing and gill netting at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2016. Fish ages are shown above the bars, where available.

Table 4.10: Summary of statistical results for Arctic charr population comparisons between Sheardown Lake SE and Reference Lake 3 for the mine operational period (2015, 2016) and between Sheardown Lake SE mine-operational and baseline period data for fish captured by electrofishing and gill netting methods, Mary River Project CREMP, August 2016. Values in parentheses indicate direction and magnitude of any significant differences.

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed?			
			versus Reference Lake 3		versus Sheardown Lake SE baseline period data [b]	
			2015	2016	2015	2016
Nearshore Electrofishing	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**
		Age	No	No	**Yes (+273%)**	-
	Energy Use	Size (mean weight)	No	No	No	**Yes (-43%)**
		Size (mean fork length)	No	No	**Yes (+7%)**	**Yes (-15%)**
		Growth (weight-at-age)	**Yes (+85%)**	**Yes (+120%)**	No	-
		Growth (fork length-at-age)	**Yes (+21%)**	**Yes (+34%)**	No	-
	Energy Storage	Condition (body weight-at-fork length)	**Yes (+4%)**	No	**Yes (-14%)**	**Yes (-16%)**
Littoral/Profundal Gill Netting [a]	Survival	Length Frequency Distribution	-	-	**Yes**	**Yes**
		Age	-	-	**Yes (-13%)**	No
	Energy Use	Size (mean weight)	-	-	**Yes (-26%)**	**Yes (-20%)**
		Size (mean fork length)	-	-	**Yes (-9%)**	**Yes (-7%)**
		Growth (weight-at-age)	-	-	**Yes (+18%)**	**Yes (+24%)**
		Growth (fork length-at-age)	-	-	No	No
	Energy Storage	Condition (body weight-at-fork length)	-	-	No	No

[a] Due to low catches of Arctic charr at Reference Lake 3 in 2015 and 2016, no comparison of fish health was possible for gill netted fish.
[b] Baseline period data included 2007 nearshore electrofishing data and 2007 and 2008 littoral/profundal gill netting data.

the Sheardown Lake SE Arctic charr population during the August 2016 field study (Appendix Table G.32). On average, Arctic charr non-spawners were younger and were slightly smaller, but showed no significant difference in LSI compared to those fish with developing gonads (Appendix Table G.32; ANOVA p = 0.464). A high proportion of individuals (i.e., 96%) contained body cavity parasites (Appendix Table G.32), the incidence rate of which was comparable to that observed at other mine-exposed lakes in 2015 and 2016, as well as during baseline studies.

Length-frequency distributions of Arctic charr captured at littoral/profundal areas of Sheardown Lake SE in 2016 differed significantly from those captured during the baseline period (Table 4.10). In part, the differences in length-frequency distribution may have reflected significantly smaller size (i.e., weight and length) of individuals captured in 2016 compared to the baseline period (Table 4.10; Appendix Table G.31). Significantly greater weight-related growth was indicated in 2016 compared to the baseline period for Arctic charr captured at littoral/profundal areas of Sheardown Lake SE, but the difference was within the ecologically meaningful CES_G of ±25% (Table 4.10; Appendix Table G.31). However, condition of Arctic charr from littoral/profundal areas of Sheardown Lake SE did not differ significantly between 2016 and the baseline period (Table 4.10). The Arctic charr data collected from littoral/profundal areas of Sheardown Lake SE between 2016 and the baseline periods generally showed the same type, direction and magnitude of differences that were shown during the 2015 CREMP (Table 4.10), suggesting no substantial changes in conditions between 2015 and 2016. Overall, the absence of any ecologically significant differences in growth and condition for Arctic charr captured at littoral/profundal areas of Sheardown Lake SE in 2016 compared to the baseline period suggested no adverse influences on adult Arctic charr following the initial two years of mine operation.

4.4 Synthesis of Mine-Related Influences within the Sheardown Lake System

4.4.1 Sheardown Lake Tributaries

At Sheardown Lake Tributary 1 (SDLT1), aqueous concentrations of several parameters were elevated compared to average concentrations observed at the reference creek stations in 2016. However, similar to the 2015 CREMP, only nitrate and sulphate concentrations were elevated at SDLT1 in 2016 compared to the baseline period and, with the exception of copper, no parameters were present at concentrations above WQG or AEMP benchmarks in 2016. Chlorophyll a concentrations were elevated at lower SDLT1 compared to reference creek stations in 2016, suggesting that elevated nitrate concentrations may have contributed to biological enrichment at SDLT1. However, similar chlorophyll a concentrations between 2016 and the baseline period indicated that SDLT1 may naturally exhibit greater phytoplankton growth than at the reference creek stations. The key differences in benthic invertebrate community metrics between SDLT1 and Unnamed Reference Creek in 2016 included lower richness and greater relative abundance of filterer FFG and burrower HPG at SDLT1. Because a higher proportion of filterers may signify greater reliance upon phytoplankton as a food source within the benthic invertebrate community, these results were consistent with greater phytoplankton abundance (as indicated by chlorophyll a concentrations) at SDLT1 in 2016, and potentially indicated a slight enrichment influence at SDLT1 due to elevated nitrate concentrations. The occurrence of significantly greater relative abundance of HPG burrowers at SDLT1 compared to Unnamed Reference Creek was consistent with influences due to sedimentation, but may have also reflected naturally greater substrate embeddedness at lower SDLT1. Comparisons to baseline indicated significantly higher density at SDLT1 in 2016, which was consistent with a slight mine-related enrichment influence at SDLT1 and similar to findings of the 2015 CREMP. No other differences in benthic endpoints were observed between 2016 and baseline studies, suggesting that any enrichment influences were minor.

At Sheardown Lake Tributary 12 (SDLT12), a significantly higher relative abundance of benthic invertebrate collector-gatherers and burrowers occurred relative to Unnamed Reference Creek in 2016, as well as during the 2015/2016 mine-operational period compared to 2007 baseline data. The temporal changes in benthic invertebrate community composition at SDLT12 are hypothesized to reflect a mine-related reduction in flow and/or increased particle loadings (e.g., through dust and/or erosional deposition) over time. At Sheardown Lake Tributary 9 (SLDT9), the relative abundance of benthic invertebrate HPG burrowers and FFG shredders was significantly higher than at Unnamed Reference Creek in 2016. However, because similar differences in composition were not indicated at SDLT9 between 2016 and baseline studies conducted in 2007 and 2013, the differences in community composition between SDLT9 and Unnamed Reference Creek in 2016 potentially reflected naturally greater amounts of in-stream vegetation at SDLT9. Notably, primary benthic invertebrate community endpoints of density, richness and Simpson's Evenness, as well as the relative abundance of metal-sensitive chironomids, showed no significant, ecologically meaningful, differences at SDLT12 or SDLT9 compared to Unnamed Reference Creek in 2016, nor compared to baseline data. This suggested that benthic invertebrate community differences at these tributaries compared to Unnamed Reference Creek in 2016 and to the baseline studies were subtle.

4.4.2 Sheardown Lake (NW and SE Basins)

At the Sheardown Lake NW and SE basins, aqueous concentrations of aluminum, manganese, molybdenum and/or uranium were elevated compared to the reference lake in both 2015 and 2016, but none of these metals, or any other parameters, were elevated compared to concentrations observed during the baseline period, and none were above WQG or AEMP benchmarks. Similar to findings of the 2015 CREMP, total aluminum and manganese concentrations showed strong positive correlations with turbidity in 2016 that, in turn, suggested that these metals were largely bound to/composed suspended particulate matter and were not likely biologically available. High turbidity in Sheardown Lake is hypothesized to reflect natural sources of suspended particulates originating from Mary River, upstream of the mine. Sediment metal concentrations at littoral stations of the Sheardown Lake basins in 2016 were similar to those at the reference lake and compared to baseline data with the exception of slightly elevated arsenic, manganese and/or molybdenum concentrations at littoral stations, suggesting some mine-related influence on sediment quality of the shallow lake zone in Sheardown Lake. However, sediment metal concentrations at profundal stations of the Sheardown Lake basins in 2016 were similar to the reference lake and baseline data, indicating that mine-related influences on sediment quality were confined to littoral habitats. Notably, no metals were present in sediment of Sheardown Lake at concentrations above SQG or AEMP benchmarks that were not also above these criteria at the reference lake, suggesting the natural occurrence of elevated concentrations of some metals (e.g., iron, manganese) in sediment of lakes in the Mary River Project LSA.

Chlorophyll a concentrations at both of the Sheardown Lake basins were significantly higher than at the reference lake in 2016 suggesting greater primary production within the Sheardown Lake system. However, chlorophyll a concentrations within the Sheardown Lake basins remained well below the AEMP benchmark during all seasonal sampling events in 2016, and were consistent with oligotrophic conditions typical of Arctic waterbodies. No significant differences in annual average chlorophyll a concentrations were indicated among the mine construction (2014) and operational (2015, 2016) periods, suggesting no changes in the trophic status of either Sheardown Lake basin since mine operations commenced at the Mary River Project. Benthic invertebrate community data collected at littoral habitat of the Sheardown Lake basins in 2016 indicated no adverse significant differences in primary endpoints (density, richness and Simpson’s Evenness) compared to the reference lake. Although significant differences in relative abundance of dominant invertebrate groups, FFG and HPG were observed between the Sheardown Lake basins and the reference lake in 2016, these differences appeared to reflect naturally differing sediment TOC and/or particle size between the mine-exposed and reference lakes. No consistent types and/or direction of differences in benthic invertebrate community endpoints were observed between 2016 and 2007/2013 baseline data for littoral stations of either Sheardown Lake basin, suggesting no adverse influences to benthic invertebrates associated with the Mary River Project mine operations.

Analysis of Arctic charr populations at the Sheardown Lake basins suggested greater fish abundance compared to the reference lake in 2016, but similar numbers of Arctic charr in 2016 compared to Sheardown Lake baseline studies. Arctic charr captured at nearshore habitat of the Sheardown Lake basins were significantly larger and grew significantly faster, but exhibited similar condition, than those captured at the reference lake in 2016. Arctic charr captured at nearshore habitat of Sheardown Lake NW in 2016 exhibited significantly lower condition than those captured during the baseline period. However, no significant, ecologically meaningful differences in growth and significantly greater condition was indicated for Arctic charr captured at littoral/profundal habitat in 2016 compared to the baseline period. The differential responses in Arctic charr endpoints between Sheardown Lake and the reference lake in 2016, and between Arctic charr collected at nearshore and littoral/profundal habitats for Sheardown Lake studies in 2016 compared to baseline, were not consistent with an adverse mine-related effect on Arctic charr populations at Sheardown Lake. Collectively, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Sheardown Lake in the second year of mine operation at the Mary River Project.

5.0 MARY RIVER AND MARY LAKE SYSTEM

5.1 Mary River

5.1.1 Water Quality

Dissolved oxygen (DO) concentrations at Mary River stations were consistently at or above saturation during all spring, summer and fall monitoring events, and were comparable to DO saturation levels observed among the GO-09 series reference river stations for each respective seasonal sampling event (Figure 5.1; Appendix Tables C.1 - C.3). Although DO saturation levels differed significantly among the Mary River benthic study areas, no gradient in DO saturation levels was shown from upstream to downstream of the mine at the time of biological sampling in August 2016 and DO saturation was consistently well above the WQG minimum limit for cold-water biota (i.e., 54%) at all times (Figure 5.1; Appendix Figure C.19 and Table C.50). This suggested that slight differences in DO concentrations/saturation among Mary River study areas were not ecologically meaningful and were unrelated to potential mine influences.

In-situ pH at all Mary River stations was similar to pH at the GO-09 series river reference stations for each respective seasonal sampling event (Appendix Table C.1 – C.3 and Figure C.19). Although pH at Mary River Station CO-05, well downstream of the mine, was significantly lower than at all other Mary River study areas, including the GO-09 river reference area, during the 2016 fall sampling event, pH at all Mary River stations was consistently within WQG limits during all spring, summer and fall sampling events (Figure 5.1; Appendix Table C.50). Aqueous conductivity at Mary River stations showed no distinct spatial changes with progression from upstream to downstream of the mine during the spring, summer or fall sampling events, suggesting no mine-related influences on Mary River conductivity (Appendix Figure C.19). Notably, conductivity varied considerably among spring, summer and fall at all stations, reflecting natural seasonal differences in conductivity of surface runoff related to dilution sources (e.g., spring snowmelt). Similar to comparisons of pH, conductivity differed significantly among Mary River benthic study areas during fall biological monitoring in 2016. However, the incremental differences in conductivity among reference and mine-exposed areas of the Mary River were small and unlikely to be ecologically meaningful. Moreover, rather than being indicative of potential mine-related influences, the differences in conductivity among Mary River study areas likely reflected the natural proportion of flow contributed by various tributaries to the river, as well as differences in the geology of base material between Mary River and these tributaries.

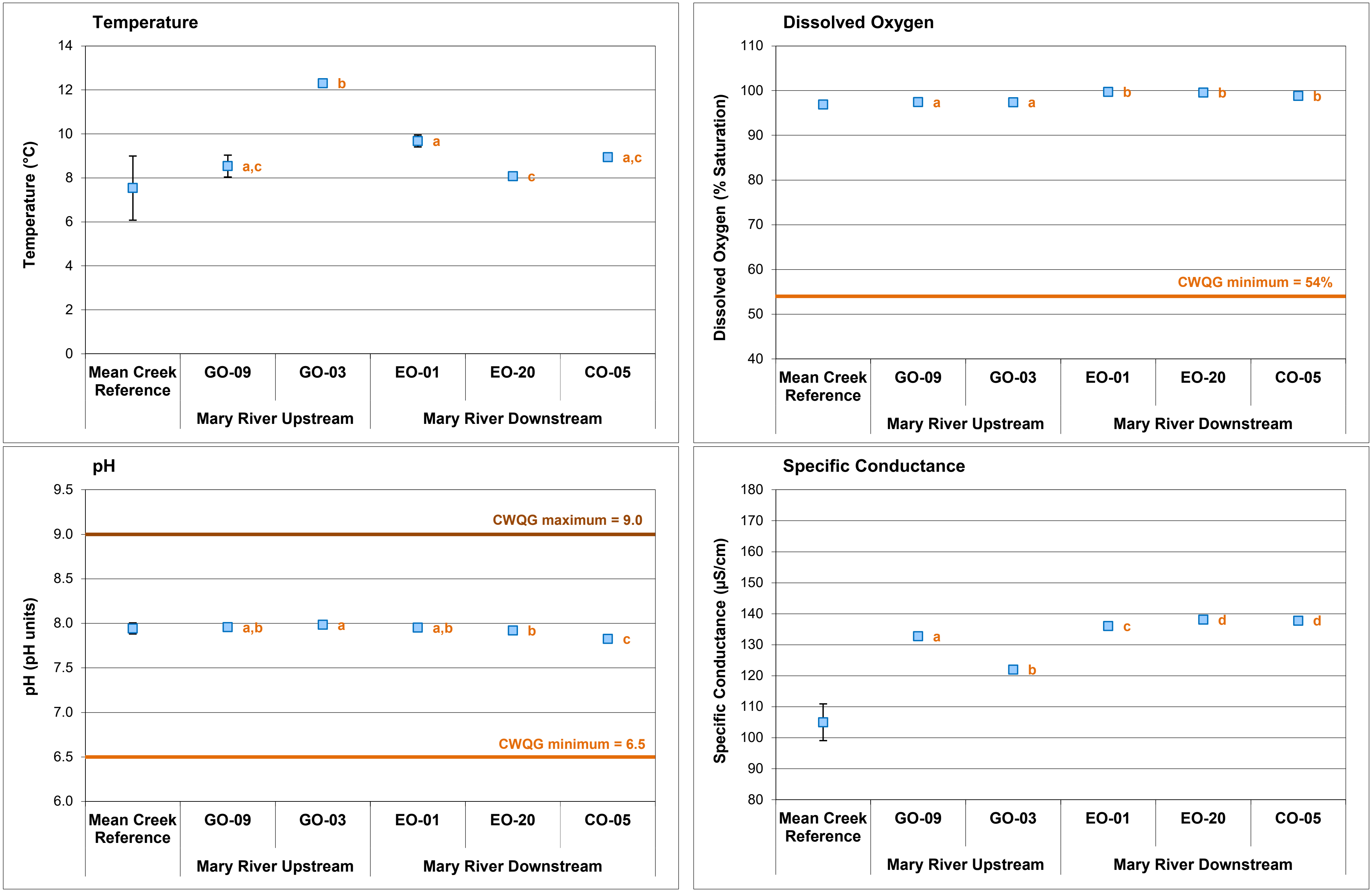


Figure 5.1: Comparison of *in-situ* water quality variables (mean ± SE) measured at the Mary River benthic invertebrate community stations (n = 5) and lotic reference stations (n = 4), Mary River Project CREMP, August 2016. The same letters next to Mary River study area data points indicate no significant difference between study areas.

Water chemistry within Mary River showed no distinct and/or consistent spatial differences with progression downstream from the GO-09 series river reference stations during any of the winter, summer or fall sampling events in 2016 (Table 5.1; Appendix Table C.51). In general, parameter concentrations at Mary River stations located adjacent to or downstream of the mine (EO and CO series stations) were similar to, and often lower than, concentrations observed at the upstream river reference stations (GO-09 series stations) during each respective spring and summer sampling event, as well as at EO series stations during the fall sampling event (Table 5.1; Appendix Table C.51). Total concentrations of several metals, including phosphorus, aluminum, chromium, iron and lead, were slightly elevated (i.e., 3- to 5-fold) at CO stations located immediately downstream of the mine compared to the GO-09 reference stations during the fall monitoring event (Table 5.1). Relatively high total concentrations of these metals at the Mary River CO stations appeared to be associated with elevated turbidity at the time of the fall sampling event (Table 5.1). Despite elevation of total concentrations of these metals, dissolved metal concentrations were consistently similar among Mary River reference and mine-exposed stations for each of the spring, summer and fall sampling events (Appendix Table C.52).

Total aluminum concentrations were above WQG and AEMP benchmarks at all Mary River mine-exposed stations during the summer and fall monitoring events, and total iron concentrations were also above WQG and/or AEMP benchmarks at all Mary River mine-exposed stations during the fall monitoring event, in 2016 (Table 5.1; Appendix Table C.51). However, concentrations of both of these metals were elevated above applicable WQG at one or more of the Mary River GO series reference stations during the spring, summer and fall monitoring events, suggesting naturally high concentrations of aluminum and iron in the Mary River system. Total phosphorus, copper and lead concentrations were also above WQG and/or AEMP benchmarks at one or more Mary River CO stations during fall monitoring in 2016 which, as discussed above, appeared to be associated with elevated turbidity at the time of sampling (Appendix Table C.51). Notably, a very high proportion (i.e., ≥80%) of aluminum, iron, lead and other metals (e.g., manganese, silicon) were in the ‘total’ concentration form, suggesting that these metals were largely associated with suspended particulate matter and were unlikely to be bioavailable. High turbidity was observed at reference (i.e., GO series) stations indicating that elevated turbidity in the Mary River was a natural phenomenon unrelated to the Mary River Project operations. Dissolved metal concentrations at all Mary River stations were well below WQG and AEMP benchmarks.

Temporal evaluation of Mary River water chemistry data suggested higher total concentrations of several metals, including aluminum, copper, iron, lead, manganese and nickel, at one or

Table 5.1: Water chemistry at Mary River monitoring stations, Mary River Project CREMP, August 2016.

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creek Average (n = 4)	Mary River Reference Station			Mary River Upstream		Tributary	Mary River Downstream of Mine					
						G0-09-A	G0-09	G0-09-B	G0-03	G0-01	F0-01	E0-10	E0-03	E0-21	E0-20	C0-10	C0-05
					Fall 2016	20-Aug-2016	20-Aug-2016	20-Aug-2016	20-Aug-2016	20-Aug-2016	20-Aug-2016	20-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016	19-Aug-2016
Conventionals	Conductivity (lab)	umho/cm	-	-	125	191	189	188	169	174	261	186	174	173	172	170	171
	pH (lab)	pH	6.5 - 9.0	-	7.99	8.23	8.24	8.21	8.14	8.14	8.28	8.14	8.12	8.16	8.17	8.13	8.15
	Hardness (as $CaCO_3$)	mg/L	-	-	58	80	84	82	76	79	131	84	80	79	80	79	79
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	5.4	2.9	2.9	2.95	2.5	3.0	2.9	<2.0	3.5	3.4	5.6	6.9
	Total Dissolved Solids (TDS)	mg/L	-	-	65	107	98	98	95	69	141	102	90	90	86	94	99
	Turbidity	NTU	-	-	1.1	16.3	9.7	11.0	12.3	16.1	11.0	16.5	12.9	14.6	16.0	32.7	41.5
	Alkalinity (as $CaCO_3$)	mg/L	-	-	57	73	79	79	74	75	118	82	70	72	68	76	76
Nutrients and Organics	Total Ammonia	mg/L	variable	0.855	<0.020	0.032	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	0.039	0.026	0.065	0.022
	Nitrate	mg/L	13	13	0.021	0.023	<0.020	<0.020	<0.020	<0.020	0.096	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	0.15	<0.15	0.16	0.25	<0.15	0.15	0.21	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.3	1.2	1.3	1.1	1.3	1.3	1.2	1.2	1.2	1.2	1.8	1.6	1.9
	Total Organic Carbon	mg/L	-	-	1.5	1.5	1.4	1.4	1.5	1.5	1.4	2.3	1.3	1.4	1.5	1.6	1.8
	Total Phosphorus	mg/L	0.020[α]	-	0.0059	0.0125	0.0090	0.0107	0.0089	0.0098	0.0112	0.0117	0.0097	0.0097	0.0157	0.0358	0.0206
	Phenols	mg/L	0.004[α]	-	0.0055	0.0056	0.0063	0.0086	0.0048	0.0037	0.0057	0.0058	0.0039	0.0042	0.0160	0.0552	0.0039
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	2.5	11.5	9.0	9.3	7.0	6.9	5.6	6.7	6.6	6.6	6.1	6.1	6.1
	Sulphate (SO_4)	mg/L	218[β]	218	4.4	5.6	4.5	4.6	3.8	4.6	14.3	5.0	4.4	4.4	4.2	5.0	5.2
Total Metals	Aluminum (Al)	mg/L	0.100	0.966	0.058	0.395	0.217	0.258	0.291	0.484	0.251	0.418	0.301	0.382	0.431	***1.040***	***1.390***
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	0.00013	<0.00010	<0.00010	0.00011	0.00013	0.00015	0.00014	0.00012	0.00012	0.00014	0.00020	0.00026
	Barium (Ba)	mg/L	-	-	0.0078	0.0147	0.0130	0.0131	0.0126	0.0142	0.0148	0.0143	0.0133	0.0133	0.0143	0.0174	0.0196
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00040	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Bismuth (Bi)	mg/L	-	-	<0.0003875	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00006	<0.000010	<0.000010	0.000011	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	12.3	17.0	17.8	18.0	15.7	16.9	27.0	17.5	17.1	16.7	16.9	16.5	16.5
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	0.00096	0.00060	0.00065	0.00073	0.00110	0.00108	0.00112	0.00076	0.00096	0.00108	0.00237	0.00319
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	0.00020	0.00012	0.00014	0.00015	0.00023	0.00024	0.00022	0.00016	0.00018	0.00022	0.00048	0.00065
	Copper (Cu)	mg/L	0.002	0.0024	0.0010	0.0017	0.0017	0.0015	0.00150	0.0016	0.0019	0.0016	0.0016	0.0015	0.0016	0.0024	***0.0027***
	Iron (Fe)	mg/L	0.30	0.874	0.051	0.410	0.227	0.278	0.298	0.471	0.325	0.437	0.308	0.383	0.442	***1.07***	***1.42***
	Lead (Pb)	mg/L	0.001	0.001	0.000096	0.00036	0.00025	0.00024	0.00029	0.00041	0.00042	0.00040	0.00031	0.00034	0.00039	0.00083	***0.00108***
	Lithium (Li)	mg/L	-	-	<0.0010	0.0010	<0.0010	<0.0010	<0.0010	0.0011	0.0011	<0.0010	<0.0010	<0.0010	<0.0010	0.0017	0.0024
	Magnesium (Mg)	mg/L	-	-	6.77	9.32	9.46	9.21	8.86	9.18	15.9	10.2	9.58	9.18	9.40	9.66	9.72
	Manganese (Mn)	mg/L	0.935[β]	-	0.00086	0.00562	0.00297	0.00359	0.00365	0.00547	0.00498	0.00531	0.00400	0.00471	0.00541	0.0121	0.0167
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.000380	0.000628	0.000510	0.000529	0.000426	0.000457	0.000337	0.000425	0.000515	0.000532	0.000534	0.000533	0.000566
	Nickel (Ni)	mg/L	0.025	0.025	0.00056	0.00079	0.00067	0.00059	0.00073	0.00102	0.00148	0.00111	0.00092	0.00103	0.00117	0.00241	0.00259
	Potassium (K)	mg/L	-	-	0.84	1.68	1.45	1.45	1.34	1.42	1.46	1.44	1.43	1.41	1.40	1.70	1.88
	Selenium (Se)	mg/L	0.001	-	<0.0007625	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	0.000052	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Silicon (Si)	mg/L	-	-	0.95	1.55	1.27	1.34	1.33	1.73	1.25	1.56	1.35	1.50	1.66	2.74	3.62
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000020	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Sodium (Na)	mg/L	-	-	1.830	5.67	4.57	4.56	3.71	3.69	2.20	3.54	3.59	3.46	3.35	3.42	3.40
	Strontium (Sr)	mg/L	-	-	0.01240	0.0234	0.0211	0.0216	0.0179	0.0184	0.0197	0.0188	0.0184	0.0182	0.0179	0.0187	0.0188
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.0000775	0.000014	0.000011	0.000012	0.0000105	0.000014	0.000013	0.000015	0.000011	0.000014	0.000015	0.000027	0.000036
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	0.00015	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	0.00799	0.0233	0.0124	0.0156	0.01615	0.0271	0.0156	0.0245	0.0164	0.0212	0.0248	0.0611	0.0822
	Uranium (U)	mg/L	0.015	-	0.00366	0.00657	0.00577	0.00580	0.004605	0.00468	0.00353	0.00430	0.00453	0.00441	0.00406	0.00406	0.00407
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.000875	0.00099	0.00063	0.00074	0.000735	0.00104	0.00078	0.00101	0.00075	0.00092	0.00098	0.00208	0.00274
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	0.0031	0.0050

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). 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 Mary River.

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

more Mary River mine-exposed stations in 2016 compared to mine baseline period (2005-2013; Figure 5.2; Appendix Figure C.20). However, as in 2015, higher total concentrations of these metals in 2016 almost certainly reflected much greater amounts of suspended matter during the fall sampling event than during the baseline period (e.g., on average, Mary River turbidity was 4.6 times higher in 2016 than during the baseline sampling in fall; Appendix Figure C.20). Turbidity of the Mary River was generally similar among reference and mine-exposed stations, suggesting naturally high suspended matter in the river that were unrelated to mine activity (Appendix Figure C.20). Comparisons of more conservative parameters commonly used as indicators of anthropogenic influences in aquatic environments (e.g., chloride, conductivity, nitrate, sulphate, hardness) indicated no substantial changes in concentrations between 2016 and the baseline period at the Mary River mine-exposed stations during fall sampling events (Figure 5.2; Appendix Figure C.20). In addition, no substantial changes in concentrations of any parameters were observed between 2016 and the mine baseline period for sampling conducted during spring and summer at the Mary River mine-exposed stations. Overall, these results suggested that mine-related influences to water quality of the Mary River, if any, were minor in 2016 based on comparisons to reference conditions and to mine baseline data.

5.1.2 Phytoplankton

Mary River chlorophyll a concentrations at stations downstream of the mine were generally within the range of the GO series river reference stations and/or stream reference stations during the 2016 spring, summer and fall sampling events (Figure 5.3). No significant differences in annual average chlorophyll a concentrations were indicated among the ten Mary River monitoring stations in 2016 (Appendix Table E.13). Chlorophyll a concentrations were well below the AEMP benchmark of 3.7 µg/L during all winter, summer and fall sampling events in 2016 at all Mary River sampling stations, and were suggestive of low (i.e., oligotrophic) phytoplankton productivity based on Dodds et al (1998) trophic status classification for stream environments (Figure 5.3). These results suggested no adverse mine-related influences on phytoplankton density at Mary River in 2016. Low to moderate phytoplankton productivity was predicted for the Mary River given ‘oligotrophic’ to ‘mesotrophic’ CWQG categorization derived from evaluation of total phosphorus concentrations of up to 36 µg/L in 2016 (Table 5.1; Appendix Table C.51). Notably, total phosphorus concentrations were not significantly correlated with chlorophyll a concentrations, and strong correlations between turbidity and total phosphorus suggested that phosphorus was bound to suspended particulates. As such, the availability of phosphorus for phytoplankton productivity at Mary River stations may be more

Figure 5.2: Temporal comparison of water chemistry at Mary River stations for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during fall. Values represent mean ± SD. Creek reference includes CLT-REF and MRY-REF series stations (mean ± SD; n = 4). Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.2 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Mary River.

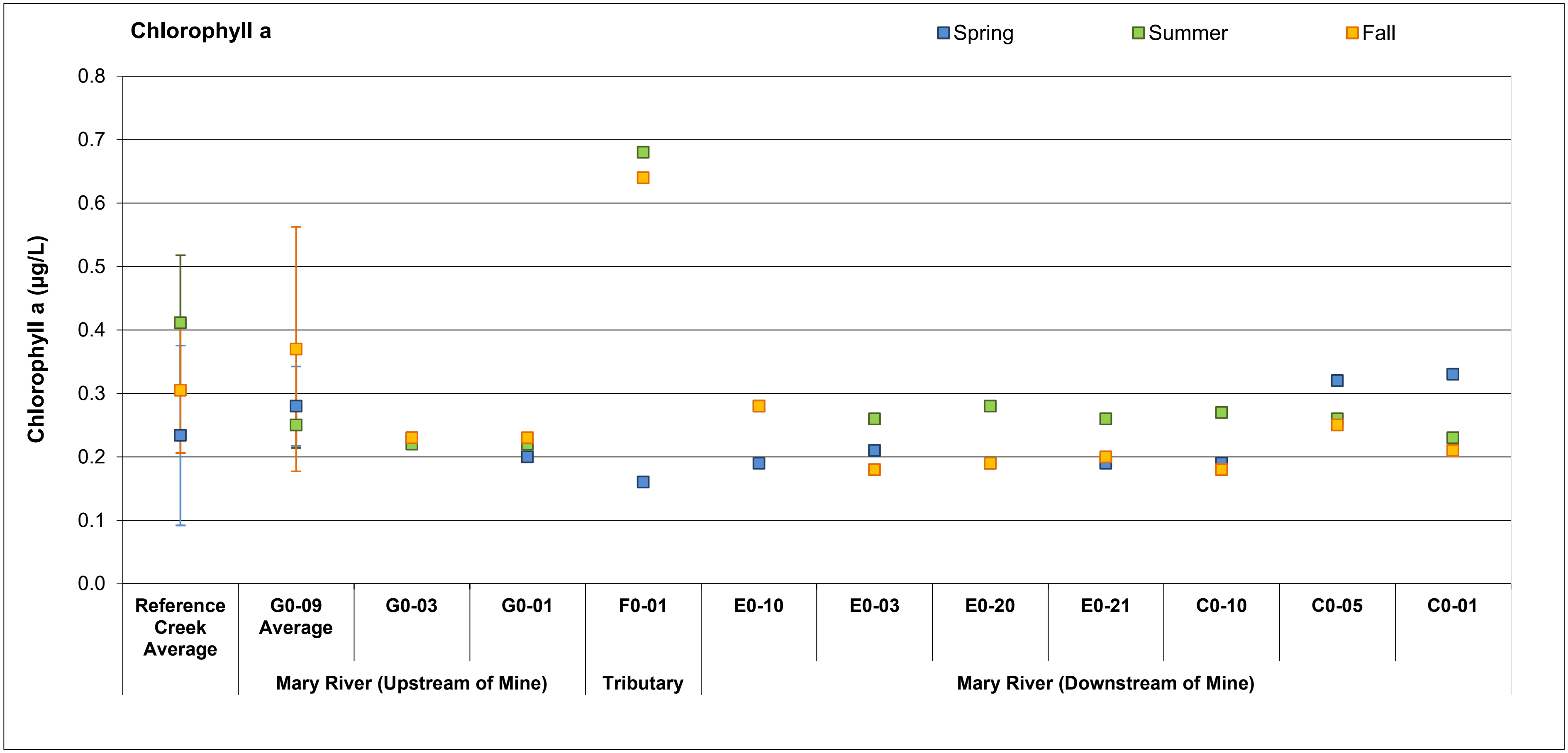


Figure 5.3: Chlorophyll a concentrations at Mary River phytoplankton monitoring stations located upstream and downstream of the mine, Mary River Project CREMP, 2016. Creek reference includes the CLT-REF and MRY-REF series stations (mean ± SD; n = 4).

limited than that suggested by the trophic categorization for the watercourse based on CWQG definitions.

Temporal comparisons of the Mary River chlorophyll a data suggested that concentrations were generally lower at stations downstream of the mine sewage treatment plant outfall (i.e., Station EO-21, -20 and CO series stations) in 2015 and 2016 compared to the baseline period (Figure 5.4). Notably, baseline period chlorophyll a concentrations at these stations were considerably higher than at reference and mine-exposed stations located upstream (Figure 5.4). Chlorophyll a concentrations at EO and CO stations located downstream of the mine sewage treatment plant outfall in 2015/2016 were comparable to concentrations at reference stations (GO) and EO stations located upstream of the mine sewage treatment plant discharge (i.e., Stations EO-10 and -03) during the baseline period (Figure 5.4). Similar to the water chemistry data for Mary River CREMP stations, variability in chlorophyll a concentrations at the Mary River stations among mine periods may have reflected natural differences in turbidity affecting the amount of light energy available to phytoplankton as opposed to adverse response to metals, nutrient enrichment or other potential mine-related influences on phytoplankton productivity. Accordingly, lower chlorophyll a concentrations in 2015 and 2016 at Mary River stations downstream of the mine sewage treatment plant discharge may have been due to naturally higher turbidity (i.e., originating from sources upstream of the mine) rather than an adverse response to mine operations.

5.1.3 Benthic Invertebrate Community

The Mary River benthic invertebrate community assessment included a spatial statistical analysis of key benthic endpoints among upstream reference areas (GO-09, GO-03), near-field mine-exposed areas located adjacent to the mine (EO-01, EO-20) and a far-field, cumulative effects mine-exposed area located downstream of the mine (CO-05; see Table 2.6, Figure 2.4). Benthic invertebrate density did not differ significantly at the three mine-exposed Mary River study areas from the GO-09 reference area in 2016 (Figure 5.5; Appendix Table F.37). Among Mary River mine-exposed areas, richness differed significantly from reference conditions only at the lower CO-05 (cumulative effects) study area. However, the occurrence of significantly higher richness downstream of the mine at CO-05 was not consistent with an adverse mine-related influence (Figure 5.5). Simpson’s Evenness at Mary River mine-exposed areas EO-20 and CO-05 was significantly lower than at the GO-09 reference area in 2016 (Figure 5.5; Appendix Table F.37). Lower Simpson’s Evenness at these two mine-exposed areas reflected dominance of the benthic invertebrate community by relatively few taxa, of which *Tokungaia* midges were the most numerous (Appendix Table F.35).

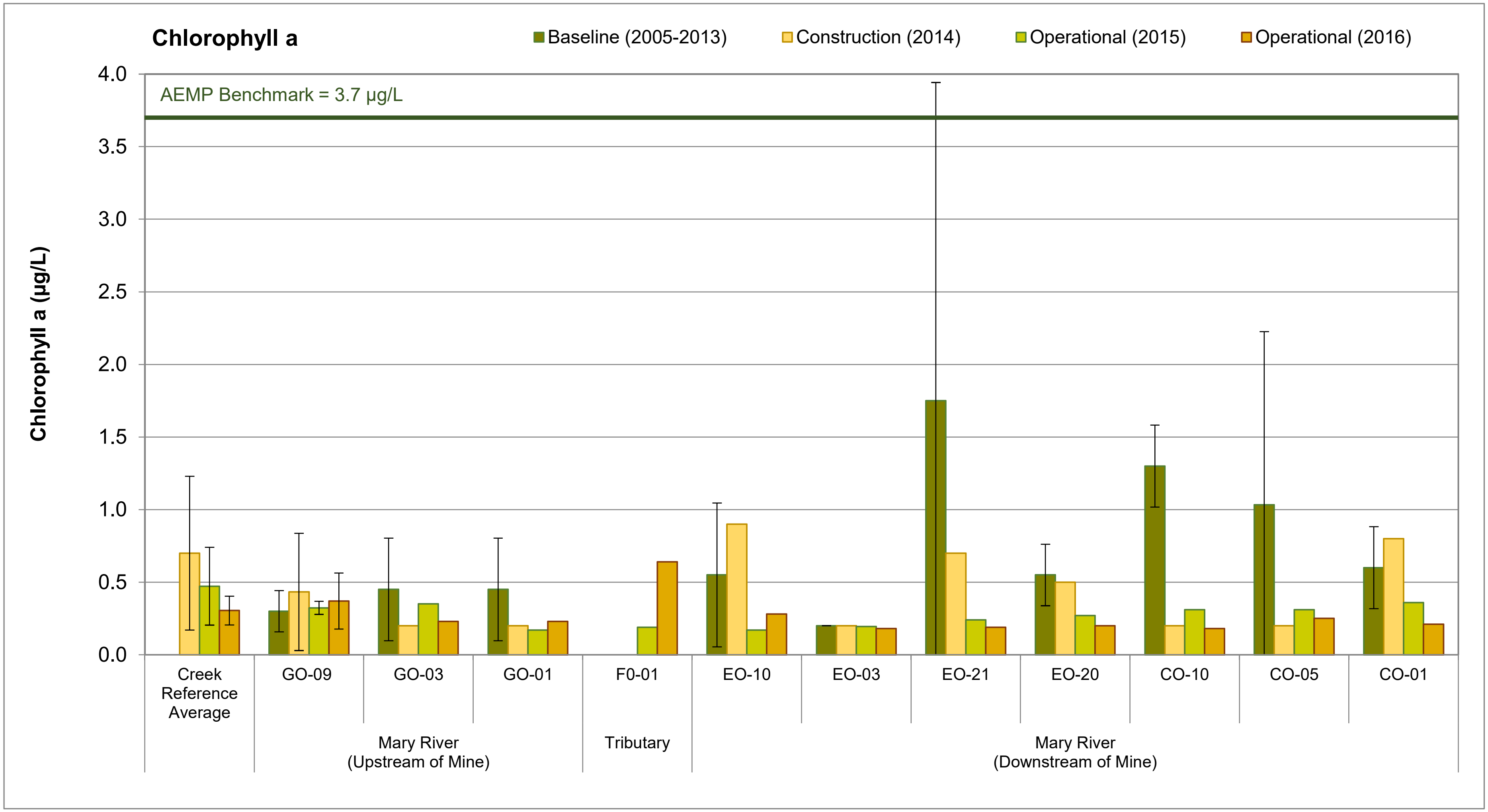


Figure 5.4: Temporal comparison of chlorophyll a concentrations at Mary River stations for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods during the fall, Mary River Project CREMP. The creek reference stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4).

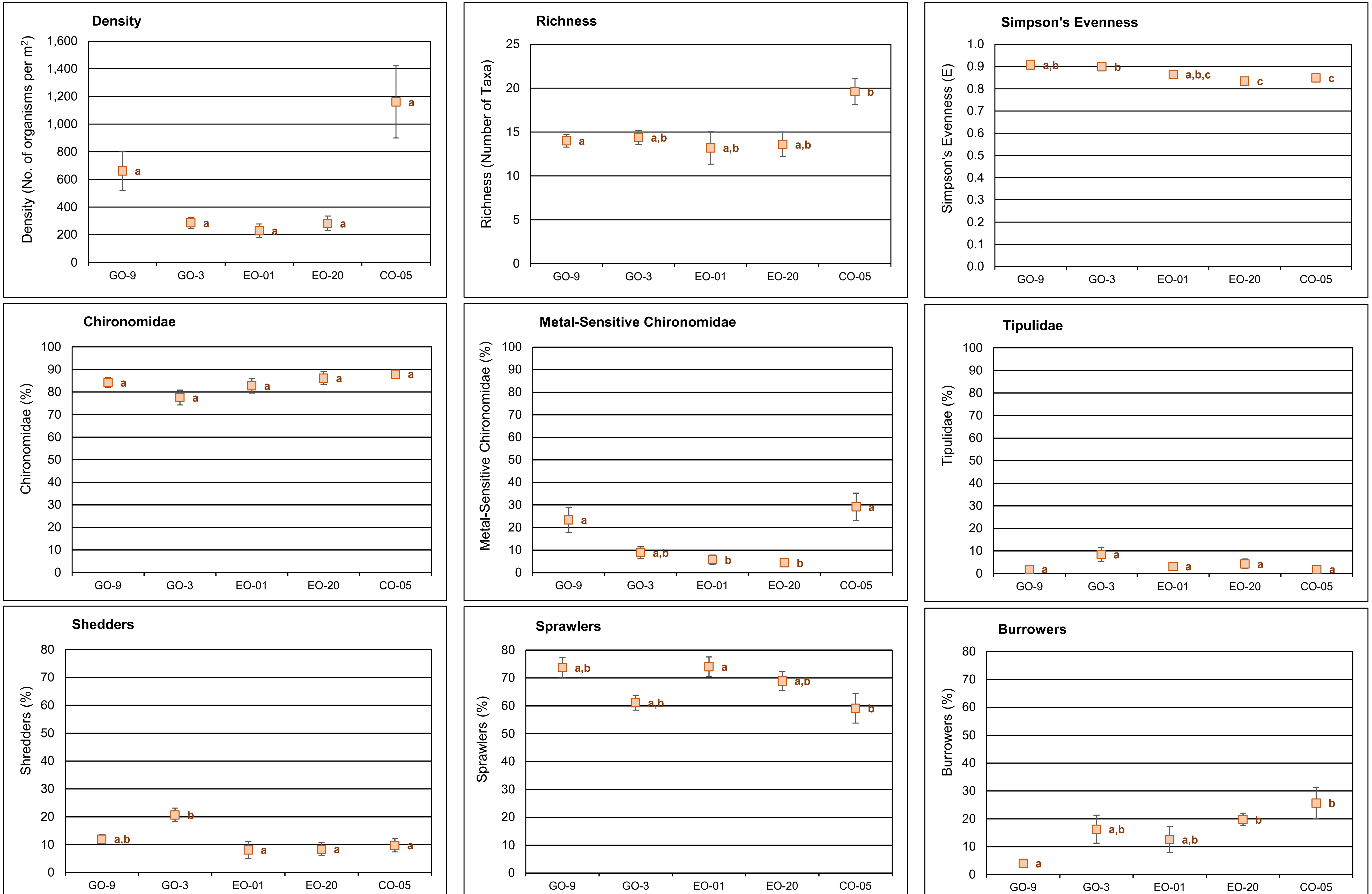


Figure 5.5: Comparison of benthic invertebrate community metrics among Mary River areas (mean ± SE), Mary River Project CREMP, August 2016. The same letters next to data points indicates no significant difference between/among study areas.

Some differences in benthic invertebrate community composition were suggested between the mine-exposed and reference areas of Mary River based on significant differences in Bray-Curtis Index (Figure 5.5; Appendix Table F.37). However, the relative abundance of dominant invertebrate groups did not differ significantly among the Mary River mine-exposed and reference areas (Figure 5.5). Despite the occurrence of significantly lower relative abundance of metal-sensitive chironomids at near-field mine-exposed areas EO-01 and EO-20 compared to the reference area, the magnitude of these differences was within a CES_{BIC} of ± 2 SD_{REF} (Figure 5.5; Appendix Table F.37). This suggested that lower relative abundance of metal-sensitive chironomids at the Mary River near-field mine-exposed areas compared to the reference area was not ecologically meaningful. No significant, ecologically meaningful, differences in the relative abundance of major FFG were shown among the Mary River study areas (Figure 5.5), suggesting no mine-related influences on food resources available for benthic invertebrates in the Mary River. A significantly higher relative abundance of HPG burrowers at Mary River mine-exposed areas EO-20 and CO-05 compared to the GO-09 reference area (Appendix Table F.37) suggested that natural differences in habitat accounted for the differences in Bray-Curtis Index between the mine-exposed and reference areas. Substrate embeddedness was significantly higher at mine-exposed CO-05 than at the reference area, which could partially explain mine-exposed and reference area differences in Bray-Curtis Index (Appendix Table F.34). Higher substrate embeddedness potentially contributed to relatively high abundance of *Tokungaia* midges at the EO-20 and CO-05 mine-exposed areas given that this genus of midges prefers more stable, depositional zones of cold water lotic environments (Oliver and Dillon 1997; Lods-Crozet et al 2012). Therefore, the differences in benthic invertebrate community composition between mine-exposed and reference areas of the Mary River suggested by significantly differing Bray Curtis Index likely reflected natural habitat factors such as substrate embeddedness.

Temporal comparison of the Mary River benthic invertebrate community data indicated no consistent significant differences in density or richness between mine operational (2015, 2016) and baseline (2006 – 2011 data) periods at any of the mine-exposed study areas (i.e., EO-01, EO-20 or CO-05; Figure 5.6; Appendix Tables F.40 – F.42). Simpson’s Evenness and chironomid relative abundance was generally significantly higher and lower, respectively, at the mine-exposed areas at the time of mine operational studies compared to the mine baseline studies. However, the same type and direction of significant differences were observed at Mary River areas located upstream of the mine (Appendix Tables F.38 – F.42), suggesting that the differences in these metrics at all Mary River areas over time reflected natural temporal variability and/or represented sampling artifacts of the CREMP (e.g., changes in sampling location, personnel). Although the relative abundance of FFG collector-gatherers was

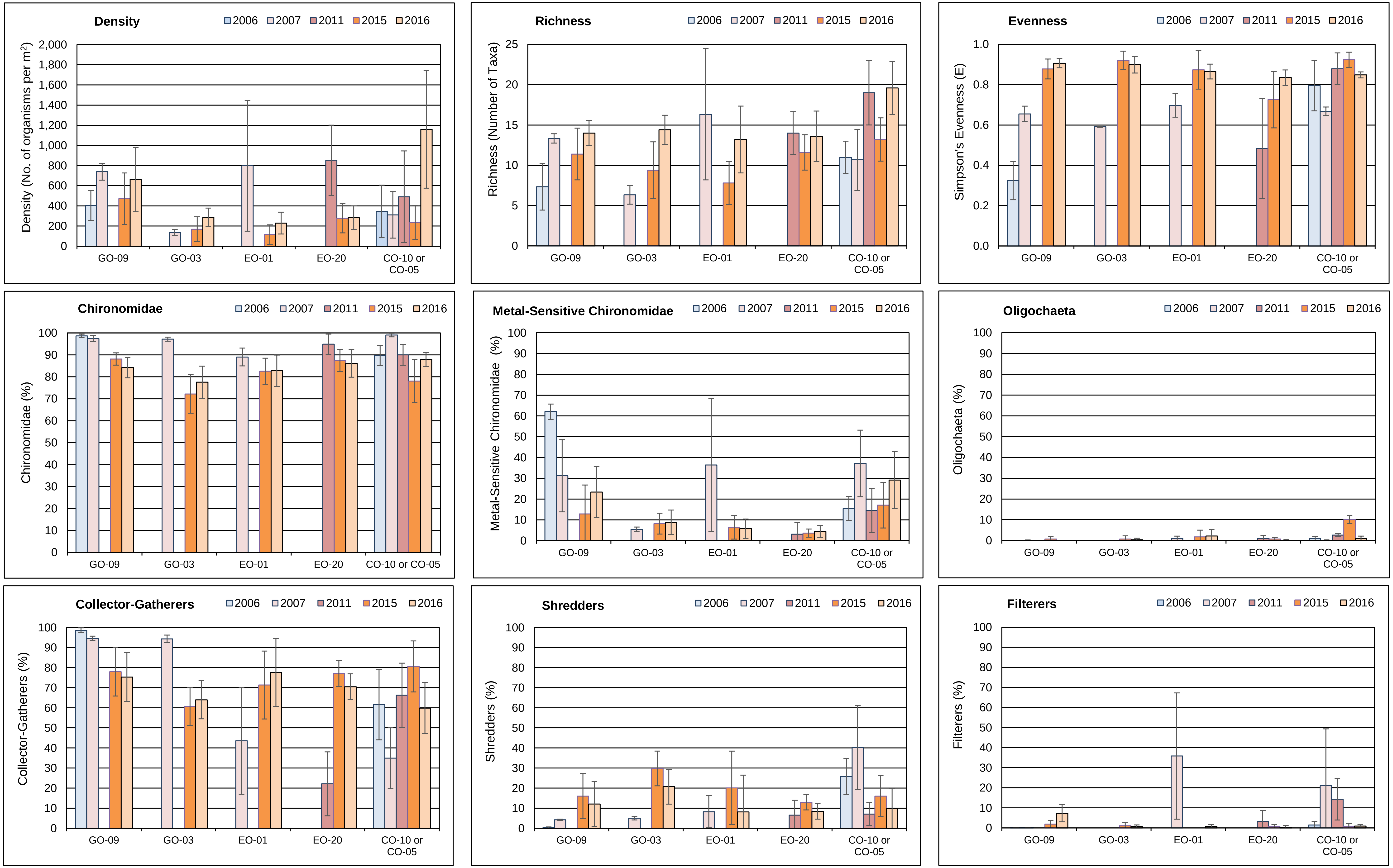


Figure 5.6: Comparison of benthic invertebrate community metrics (mean ± SD) at Mary River stations among baseline (2006, 2007), construction (2014) and operational (2015, 2016) years, Mary River Project CREMP.

significantly higher at upper mine-exposed area EO-20 following initiation of mine-operation than during the baseline period, the proportion of collector-gatherers at this area became more similar to the reference condition in the mine operational period, suggesting that the temporal changes were not mine-related (Appendix Tables F.38 and F.41). Notably, the types, direction, and magnitude of difference for endpoints that differed significantly between the mine operational and baseline periods at the Mary River mine-exposed areas were similar between the 2015 and 2016 CREMP studies (Figure 5.6), suggesting no cumulative temporal influences on benthic invertebrates of the Mary River since mine operations commenced.

5.2 Mary Lake (BLO)

5.2.1 Water Quality

Water quality profiles conducted at Mary Lake in 2016 showed similar *in-situ* water temperature, dissolved oxygen saturation and pH values, but consistently higher specific conductance, at the north basin compared to the south basin throughout the year (Figures 5.7 and 5.8). Water temperatures typically showed a gradient from surface to bottom during the winter, summer and fall at the Mary Lake north and south basins. However, the temperature profile suggested only weak thermal stratification at the south basin water column during the summer and fall sampling events in 2016, with the greatest change in temperature occurring between lake depths of approximately 10 and 20 m in both seasons (Figures 5.7 and 5.8). Weak to more strongly established thermal stratification occurred at Reference Lake 3 during the summer and fall sampling events, respectively (Figures 5.7 and 5.8). The mean water temperature at the bottom of water column at Mary Lake littoral stations did not differ significantly from that of Reference Lake 3 littoral stations in fall 2016 (Figure 5.9; Appendix Tables C.22 and C.57).

Dissolved oxygen profiles conducted at Mary Lake in 2016 indicated the development of a strong oxycline at the north basin in winter beginning at a depth of approximately 5 m, and a weak oxycline at the south basin in winter, summer and fall at depths greater than approximately 10 m (Figures 5.7 and 5.8). This differed from Reference Lake 3, where no oxycline development was apparent in the summer or fall of 2016. Nevertheless, dissolved oxygen saturation levels at Mary Lake remained above the WQG of 54% through the entire water column at the south basin in all seasons, and at the north basin in summer and fall seasons (Figures 5.7 and 5.8). Dissolved oxygen saturation levels below the WQG of 54% occurred at depths greater than approximately 11.5 m at the Mary Lake north basin in the winter (Figure 5.7). Dissolved oxygen saturation levels at Mary Lake littoral stations were well above the respective WQG at the bottom of the water column, and did not differ significantly

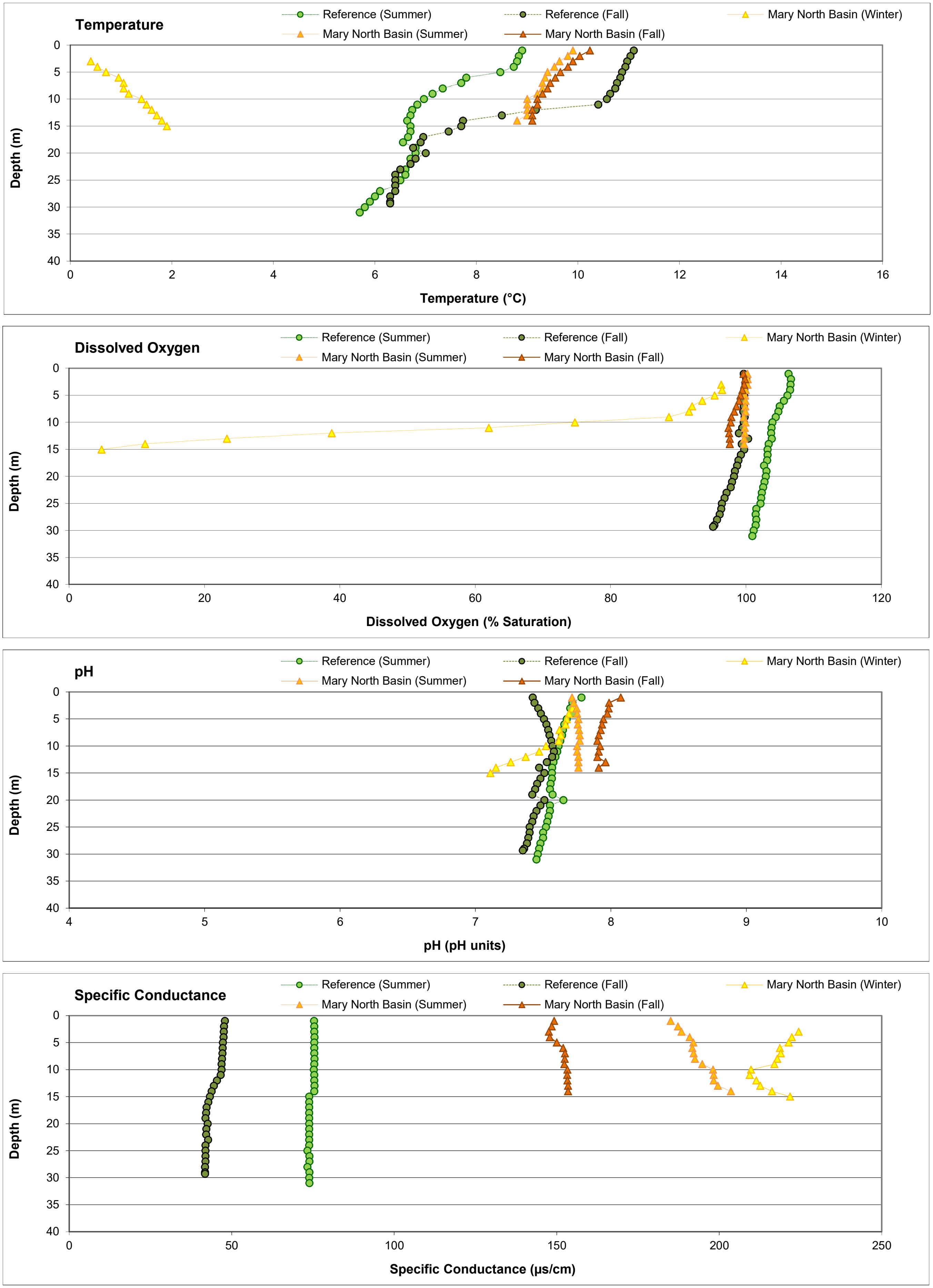
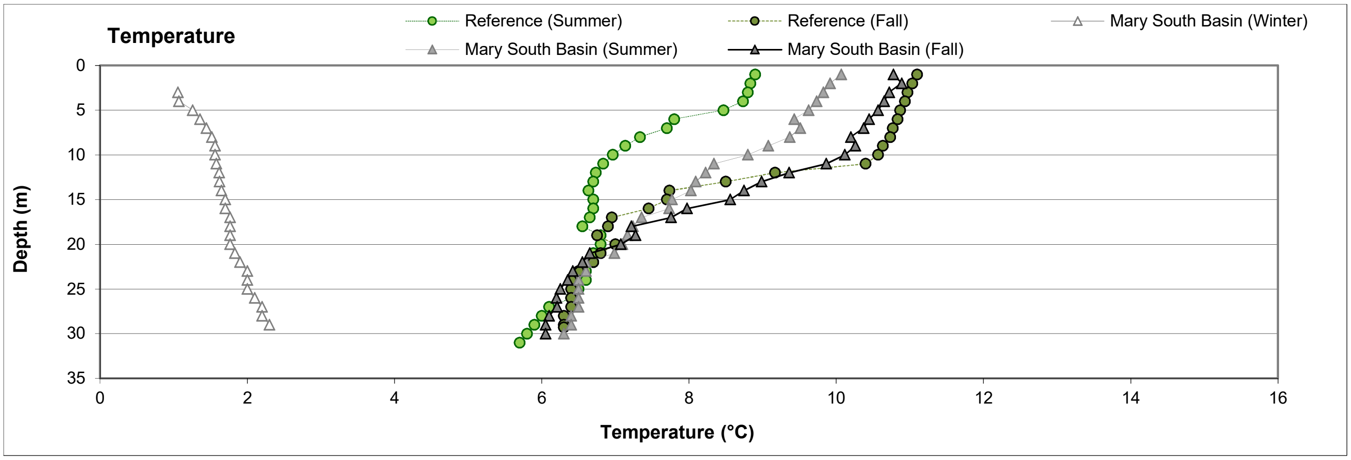
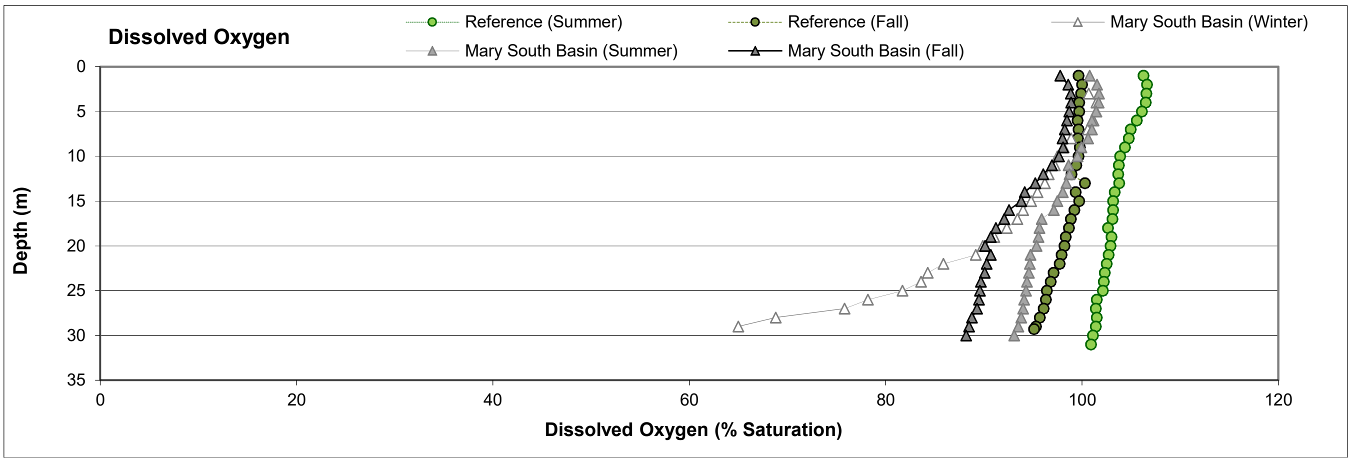
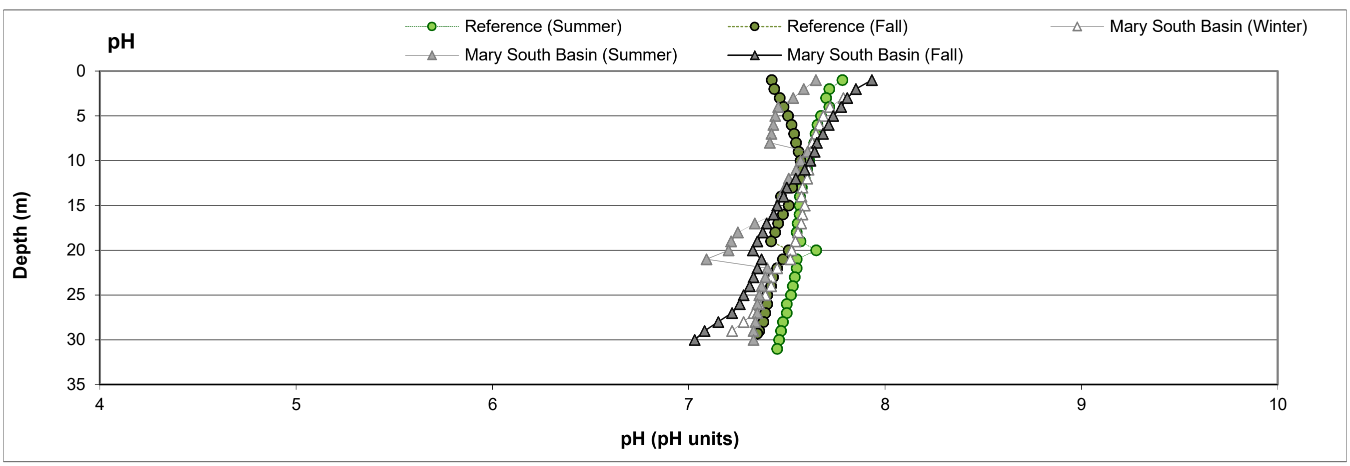


Figure 5.7: Average *in-situ* water quality with depth from surface at the Mary Lake (mine-exposed area) north basin compared to Reference Lake 3 during winter, summer, and fall sampling events, Mary River Project CREMP, 2016.







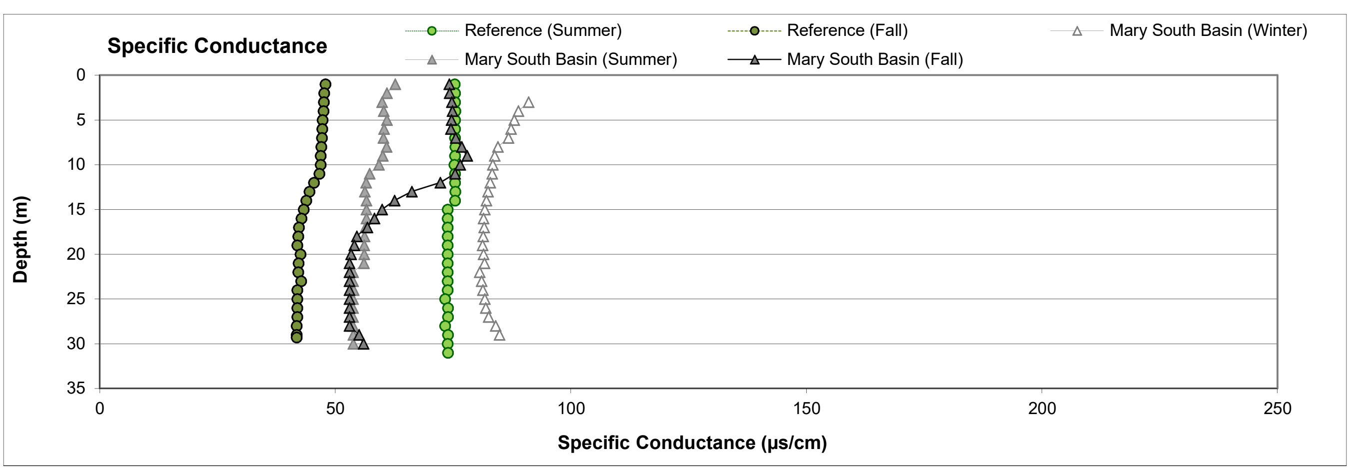


Figure 5.8: Average *in-situ* water quality with depth from surface at the Mary Lake (mine-exposed area) south basin compared to Reference Lake 3 during winter, summer, and fall sampling events, Mary River Project CREMP, 2016.

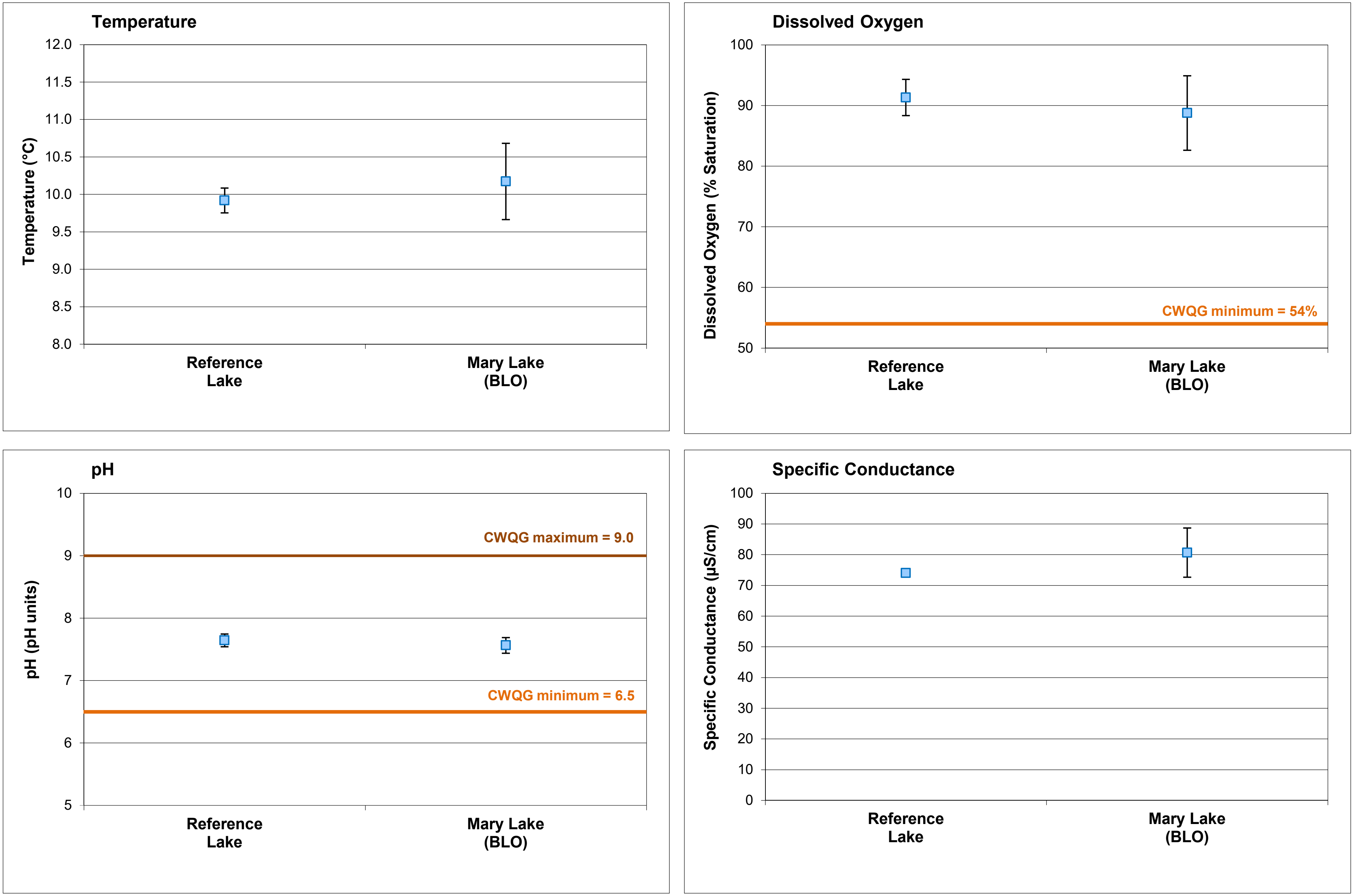


Figure 5.9: Comparison of in-situ water quality variables (mean ± SD; n = 5) measured near the bottom of the water column at Mary Lake (BLO) and Reference Lake 3 (REF3) littoral benthic invertebrate community stations, Mary River Project CREMP, August 2016. An asterisk (*) next to the Mary Lake data point indicates a significant difference compared to the reference lake measure.

from those at Reference Lake 3, during the 2016 fall sampling event (Figure 5.9; Appendix Tables C.22 and C.57).

In-situ profiles of pH showed no substantial change from the surface to bottom of the water column at either the north or south basins of Mary Lake during winter, summer or fall sampling in 2016, and were also comparable to pH profiles at Reference Lake 3 (Figures 5.7 and 5.8). No significant differences in bottom pH were indicated between Mary Lake and Reference Lake 3 at littoral stations sampled in fall 2016 (Figure 5.9; Appendix Table F.57). In addition, pH values at Mary Lake water quality and benthic littoral stations were consistently within WQG limits (Figure 5.9). Specific conductance was substantially higher at the north basin compared to the south basin of Mary Lake (Figures 5.7 and 5.8; Appendix Figure C.25). The differences in specific conductance between lake basins likely reflected 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 of the Mary Lake north basin was higher than at Reference Lake 3, but comparable to that of the reference creek stations. However, specific conductance measured at the water column bottom did not differ significantly between Mary Lake and Reference Lake 3 at littoral stations (Figure 5.9; Appendix Table C.57), reflecting the fact that specific conductance at the south basin of Mary Lake was comparable to that of Reference Lake 3 (Figures 5.7 and 5.8). Only minor changes in specific conductance were observed with depth (i.e., ≤20 µS/cm) during the winter, summer and fall sampling events in 2016 at the Mary Lake north and south basins (Figures 5.7 and 5.8). Water clarity, as determined using Secchi depth readings, was significantly lower at Mary Lake compared to Reference Lake 3 in fall 2016 (Appendix Table C.22 and C.57). In general, Secchi depth readings were similar among the Mary Lake stations, suggesting no spatial differences in water clarity throughout the lake (Appendix Table C.56).

Water chemistry of the Mary Lake north basin showed slightly (i.e., 3- to 5-fold higher) to moderately elevated (i.e., 5- to 10-fold higher) turbidity and concentrations of total aluminum, total manganese and/or total uranium compared to Reference Lake 3 at the time of summer and fall sampling in 2016 (Table 5.2; Appendix Tables C.58 and C.62). However, of these parameters, only manganese was moderately elevated at the Mary Lake north basin compared to respective mean values for the lotic reference stations, and only during the fall sampling event. In addition, no parameters were above WQG and AEMP benchmarks at the Mary Lake north basin during any of the winter, summer or fall monitoring events in 2016 (Table 5.2; Appendix Table C.58). Furthermore, despite continuously higher concentrations since mine construction (2014) and initial mine operation (2015) periods, average concentrations of the

Table 5.2: Water chemistry at Mary Lake north basin (BLO-01) and south basin (BLO) monitoring stations, Mary River Project CREMP, 2016. Values presented are averages of samples taken from the surface and the bottom of the water column at each station. * Copper data confounded by field sampling equipment.

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Lake 3 Average (n = 3)	Tom River I0-01	North Basin (Mine-exposed)			South Basin (Mine-exposed)						
							BL0-01-A	BL0-01	BL0-01-B	BL0-05-A	BL0-05	BL0-05-B	BL0-03	BL0-04	BL0-09	BL0-06
					Fall 2016	19-Aug-2016	21-Aug-2016	21-Aug-2016	21-Aug-2016	23-Aug-2016	23-Aug-2016	23-Aug-2016	24-Aug-2016	23-Aug-2016	23-Aug-2016	23-Aug-2016
Conventionals	Conductivity (lab)	umho/cm	-	-	84	194	162	162	158	86	68	80	70	79	65	77
	pH (lab)	pH	6.5 - 9.0	-	7.68	8.24	8.15	8.13	8.16	7.82	7.77	7.71	7.78	7.85	7.72	7.83
	Hardness (as $CaCO_3$)	mg/L	-	-	35	95	79	78	75	40	32	37	33	36	31	36
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	2.25	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	39	98	90	84	81	45	33	40	39	42	31	38
	Turbidity	NTU	-	-	0.3	0.4	1.3	1.2	1.5	1.7	1.6	1.8	0.7	2.4	1.3	2.0
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	94	78	80	79	38	29	38	33	35	28	34
Nutrients and Organics	Total Ammonia	mg/L	variable	0.855	0.0398	<0.020	0.021	0.057	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.7	1.7	1.8	1.8	1.9	1.2	1.2	1.2	1.3	1.2	1.2	1.2
	Total Organic Carbon	mg/L	-	-	2.8	1.8	2.0	1.8	1.9	1.4	1.3	1.3	1.4	1.5	1.3	1.4
	Total Phosphorus	mg/L	0.020[α]	-	0.010	<0.0030	0.003	0.004	0.003	0.007	0.007	0.010	0.006	0.006	0.008	0.005
	Phenols	mg/L	0.004[α]	-	0.003	0.004	0.003	0.002	0.002	0.011	0.004	0.008	0.003	0.006	0.009	0.008
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.27	5.51	3.47	3.39	3.14	1.91	1.42	1.60	1.29	1.61	1.36	1.50
	Sulphate (SO_4)	mg/L	218[β]	218	4.07	1.95	1.43	1.42	1.35	1.33	0.90	1.12	0.84	1.19	0.81	1.11
Total Metals	Aluminum (Al)	mg/L	0.100	0.13	0.004	0.010	0.026	0.030	0.026	0.078	0.058	0.065	0.023	0.056	0.050	0.059
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	0.00015	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0065	0.0098	0.0079	0.0078	0.0082	0.0056	0.0045	0.0053	0.0038	0.0049	0.0040	0.0048
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00010	<0.00010	<0.00010	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.000050	<0.000050	<0.000050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00006	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	6.99	18.90	15.55	15.40	15.45	8.24	6.48	7.63	6.68	7.52	6.26	7.26
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.00082	0.00099	0.001	0.0011	0.0011	0.00068	0.00064	0.00071	0.00053	*	*	*
	Iron (Fe)	mg/L	0.30	0.326	<0.030	<0.030	<0.050	<0.050	<0.050	0.056	0.048	0.052	<0.030	0.050	0.039	0.052
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.00010	<0.00010	<0.00010	0.000067	0.000060	0.000062	<0.000050	0.000067	0.0000555	0.000068
	Lithium (Li)	mg/L	-	-	<0.0010	0.0011	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Magnesium (Mg)	mg/L	-	-	4.3	11.4	8.7	8.7	8.5	4.8	3.8	4.4	4.0	4.4	3.7	4.3
	Manganese (Mn)	mg/L	0.935[β]	-	0.00062	0.00032	0.00507	0.00520	0.00476	0.00158	0.00286	0.00140	0.00115	0.00163	0.00139	0.00155
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00014	0.00024	0.00022	0.00022	0.00021	0.00016	0.00012	0.00014	0.00009	0.00014	0.00011	0.00013
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	<0.00050	0.00052	0.00054	0.00055	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Potassium (K)	mg/L	-	-	0.89	1.04	0.88	0.88	0.87	0.68	0.57	0.63	0.51	0.63	0.54	0.62
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.000050	<0.000050	<0.000050	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.42	0.86	0.81	0.81	0.84	0.68	0.57	0.64	0.54	0.63	0.53	0.62
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000050	<0.000050	<0.000050	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.84	3.20	2.03	2.00	1.94	1.24	0.96	1.11	0.89	1.09	0.90	1.05
	Strontium (Sr)	mg/L	-	-	0.0081	0.0130	0.0111	0.0111	0.0110	0.0071	0.0054	0.0064	0.0050	0.0063	0.0051	0.0061
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.000010	<0.000010	<0.000010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<0.010	<0.010	0.0009	0.0011	0.0009	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00027	0.00253	0.00163	0.00157	0.00151	0.00085	0.00050	0.00069	0.00043	0.00068	0.00044	0.00063
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0010	<0.0010	<0.00050	<0.00050	<0.00050	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME 1987, 1999) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2013). 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 Lake.

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

majority of parameters at the Mary Lake north basin in 2016 were comparable to, and often lower than, concentrations observed during the mine baseline (2005 – 2013) period (Figure 5.10; Appendix Table C.62 and Figure C.26). This suggested that, similar to Mary River, elevated aluminum, manganese and uranium concentrations at the Mary Lake north basin compared to Reference Lake 3 most likely reflected naturally high turbidity and specifically, particulate-bound metals, as opposed to potential mine-related influences on water chemistry.

Water chemistry at the Mary Lake south basin showed no consistent spatial differences in parameter concentrations with progression from the Mary River inlet to the lake outlet during any of the winter, summer or fall sampling events in 2016 (Table 5.2; Appendix Table C.59), suggesting that the south basin waters were generally well mixed both laterally and vertically. On average, turbidity was moderately elevated (i.e., 5- to 10-fold higher), and aluminum, copper and manganese concentrations moderately to highly elevated (i.e., ≥10-fold higher), at the Mary Lake south basin compared to Reference Lake 3 during the 2016 summer and/or fall sampling events (Table 5.2; Appendix Tables C.59 and C.62). Similar to water chemistry of the Mary River and Sheardown Lake SE water bodies, aluminum, manganese and iron concentrations showed a strong positive correlation with turbidity for the Mary Lake south basin combined data set (i.e., winter, summer and fall data; $r^2 \geq 0.70$), suggesting that much of the aqueous aluminum and manganese was associated with suspended particles (e.g., aluminosilicates). As indicated previously, high turbidity in the Mary River originated from natural sources upstream of the mine and accordingly, relatively high turbidity at Mary Lake was therefore not associated with the mine operations. Despite elevation of these metals at the south basin of Mary Lake relative to Reference Lake 3, concentrations of all parameters were generally well below established WQG and AEMP benchmarks during all 2016 sampling events[13] at the time of the fall sampling event (Table 5.2; Appendix Table C.59).

Temporal comparisons of the Mary Lake south basin water chemistry data suggested no changes in average concentrations of mine-related parameters in 2016 compared to the baseline (2005 – 2013) period except for aluminum and turbidity (Figure 5.10; Appendix Figure C.26). Although higher turbidity and concentrations of aluminum were observed at stations most distant to the Tom and Mary rivers inlets to Mary Lake in 2016 compared to baseline conditions, parameter levels closer to these river inlets (i.e., BLO-01 and BLO-05/-03, respectively) were comparable between 2016 and the baseline period (Figure 5.10; Appendix Figure C.26). Therefore, the source of turbidity and aluminum to the Mary Lake south basin in

[13] Refer to footnote 2 (page 23) and Appendix B regarding phenol concentrations above WQG at the mine-exposed and reference areas of the Mary River Project LSA waterbodies.

Figure 5.10: Temporal comparison of water chemistry at Mary Lake (BLO) for mine baseline (2005 - 2013), construction (2014), and operational (2015, 2016) periods during fall. Values represent mean ± SD. Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.2 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Mary Lake.

fall 2016 was unclear, but did not appear to be related to discharge from the Tom or Mary rivers. Parameter concentrations at the Mary River south basin in 2016 were similar to those in years of mine construction (2014) and initial mine operation (2015; Figure 5.10; Appendix Figure C.26). The general lack of temporal differences in water quality of the Mary Lake south basin over time provided additional support that elevated aluminum concentrations at the south basin relative to Reference Lake 3 were related to naturally higher turbidity at Mary Lake rather than a mine influence on lake water quality.

5.2.2 Sediment Quality

Surficial sediment of the Mary Lake north basin (BLO-01) was composed of silt loam material with low TOC content (Figure 5.11). At the Mary Lake south basin, sediment of the littoral and profundal stations was characterized by silt loam and silty clay loam material with low TOC content (Figure 5.11). Silt was the predominant particle size among littoral stations of both Mary Lake and Reference Lake 3, with no significant difference in silt content indicated between lakes (Appendix Table D.25). However, sediment sand and TOC content was significantly lower at littoral stations of Mary Lake compared to the reference lake. Substrate containing visible iron (oxy)hydroxide material was not observed at the Mary Lake north or south basins in 2016 (Appendix Tables D.22 – D.24), which contrasted with that of Reference Lake 3 and the other mine-exposed lakes where such material was commonly visible as a thin, distinct layer or floc on or within surficial sediment. Substrate of Mary Lake often contained sub-surface blackening/dark colouration which occasionally occurred as bands/layers indicating the presence of reduced sediment demarcated by distinct redox boundaries in some cases (Appendix Tables D.22 – D.24). Similar sub-surface reducing conditions were observed in sediment of the reference lake, though no distinct redox boundaries were visible (Appendix Tables D.22 – D.24).

Sediment metal concentrations at the Mary Lake north basin were similar to those observed at littoral stations of Reference Lake 3, with only manganese showing slight elevation in concentration at the Mary Lake north basin station (Table 5.3; Appendix Table D.26). Sediment metal concentrations at the Mary Lake south basin showed no spatial gradients with progression from the Mary River inlet to the lake outlet for either the littoral or profundal stations, suggesting that the Mary River was not contributing disproportionate concentrations of metals (Appendix Table D.26). Sediment metal concentrations at the Mary Lake south basin littoral and profundal sediment monitoring stations were comparable to average metal concentrations at like-depth stations of the reference lake (Table 5.3; Appendix Table D.27). Of those metals with established SQG, only manganese was above the applicable guidelines at the north basin littoral station, and on average, at the south basin littoral stations of Mary

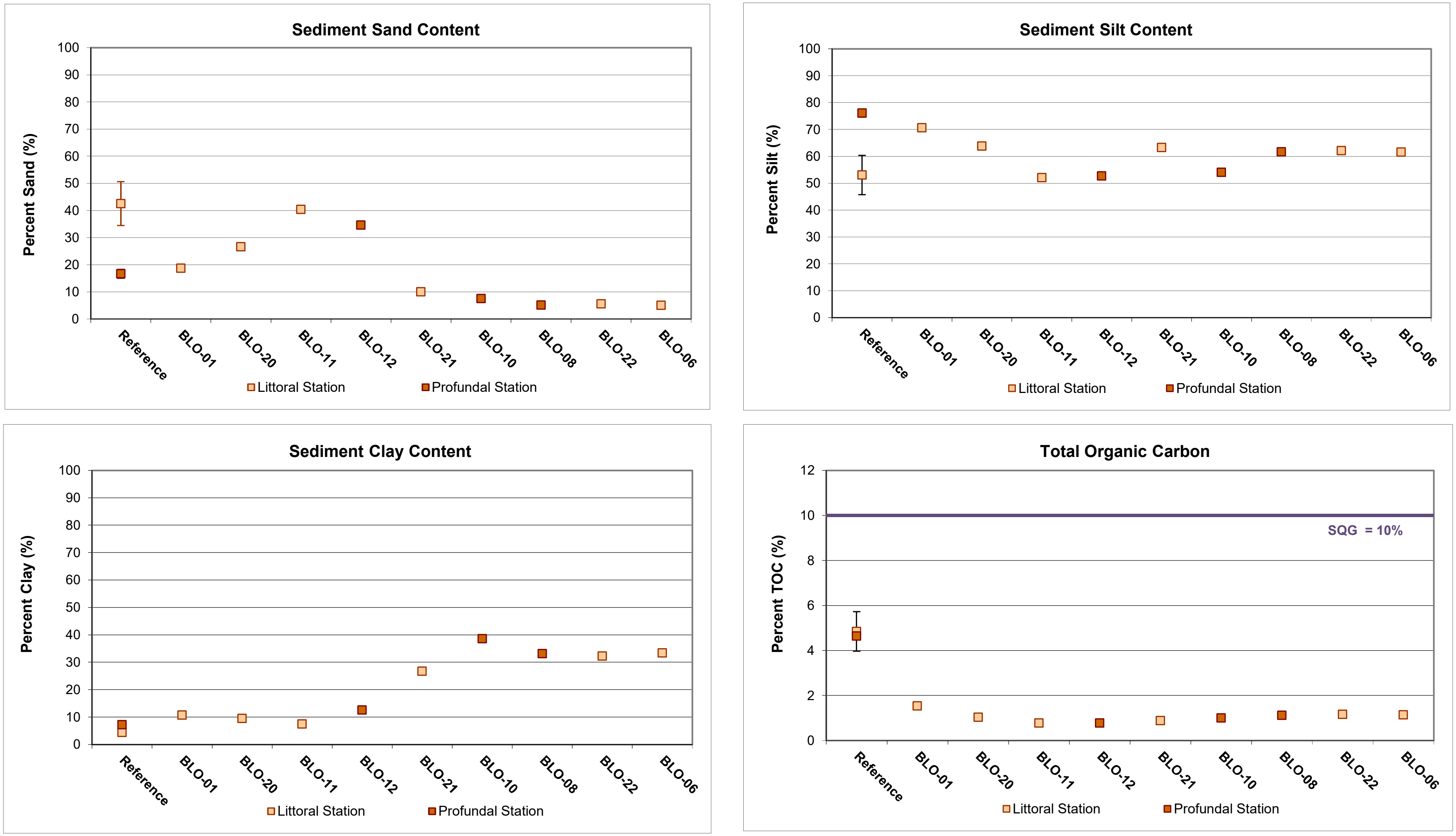


Figure 5.11: Sediment particle size and total organic carbon (TOC) content comparisons among Mary Lake (BLO) north and south basin sediment monitoring stations and to Reference Lake 3 averages (mean ± SE), Mary River Project CREMP, August 2016.

Table 5.3: Sediment particle size, total organic carbon, and metal concentrations at Mary Lake north basin (BLO-01), Mary Lake south basin (BLO), and Reference Lake 3 (REF3) sediment monitoring stations, Mary River Project CREMP, August 2016.

	Analyte	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b]	Littoral			Profundal	
					Reference Lake (n = 5)	Mary Lake (North Basin) (n = 1)	Mary Lake (South Basin) (n = 5)	Reference Lake (n = 5)	Mary Lake (South Basin) (n = 3)
					Average ± Std. Error	Average	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error
Non-metals	Sand	%	-	-	42.5 ± 8.1	18.7	17.5 ± 6.3	16.7 ± 1.5	15.7 ± 9.5
	Silt	%	-	-	53.1 ± 7.3	70.6	60.6 ± 2.0	76.1 ± 1.4	56.1 ± 2.8
	Clay	%	-	-	4.4 ± 1.0	10.7	21.9 ± 5.1	7.2 ± 0.4	28.1 ± 7.9
	Moisture	%	-	-	89.7 ± 6.0	55.9	50.0 ± 3.4	83.5 ± 5.4	54.3 ± 7.7
	Total Organic Carbon	%	10^{α}	-	4.85 ± 0.88	1.54	1.00 ± 0.07	4.64 ± 0.13	0.97 ± 0.10
Metals	Aluminum (Al)	mg/kg	-	-	16,480 ± 397	14,700	20,260 ± 2,189	25,150 ± 1,418	21,533 ± 3,481
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10	<0.10 ± 0	0.12 ± 0.02	<0.10 ± 0
	Arsenic (As)	mg/kg	17	5.9	3.71 ± 0.26	5.54	3.37 ± 0.34	***6.47*** **± 0.27**	3.49 ± 0.40
	Barium (Ba)	mg/kg	-	-	112 ± 11	87	88 ± 11	162 ± 8	89 ± 16
	Beryllium (Be)	mg/kg	-	-	0.67 ± 0.02	0.76	1.02 ± 0.13	1.02 ± 0.05	1.07 ± 0.19
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	<0.20	0.23 ± 0.01	0.21 ± 0.004	0.24 ± 0.02
	Boron (B)	mg/kg	-	-	13.0 ± 0.9	21.6	27.9 ± 3.2	19.2 ± 1.0	29.5 ± 5.4
	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.035	0.100	0.120 ± 0.017	0.180 ± 0.010	0.128 ± 0.023
	Calcium (Ca)	mg/kg	-	-	5,128 ± 470	9,700	5,176 ± 593	6,111 ± 156	4,603 ± 173
	Chromium (Cr)	mg/kg	90	98	55.0 ± 1.2	61.7	76.2 ± 5.3	80.0 ± 4.1	80.7 ± 10.1
	Cobalt (Co)	mg/kg	-	-	10.15 ± 0.57	13.90	14.76 ± 1.32	18.15 ± 0.75	15.33 ± 1.93
	Copper (Cu)	mg/kg	110	50	***66.5*** **± 7.4**	27.5	28.5 ± 2.7	***101.4*** **± 5.6**	30.0 ± 4.7
	Iron (Fe)	mg/kg	$40,000^{\alpha}$	52,400	29,840 ± 3,488	34,400	35,750 ± 2,763	***53,580*** **± 2,174**	36,400 ± 4,339
	Lead (Pb)	mg/kg	91.3	35	***46.0*** **± 17.4**	16.3	23.1 ± 3.6	29.5 ± 5.0	24.7 ± 3.7
	Lithium (Li)	mg/kg	-	-	27.3 ± 0.4	29.9	39.2 ± 4.5	41.7 ± 2.1	39.6 ± 6.2
	Magnesium (Mg)	mg/kg	-	-	10,852 ± 274	14,600	14,500 ± 1,000	16,160 ± 814	14,633 ± 1,822
	Manganese (Mn)	mg/kg	$1,100^{\alpha,\beta}$	4,370	496 ± 99	1,790	1,670 ± 845	1,866 ± 449	1,047 ± 158
	Mercury (Hg)	mg/kg	0.486	0.17	0.0355 ± 0.0063	0.0275	0.0403 ± 0.0071	0.0699 ± 0.0019	0.0516 ± 0.0126
	Molybdenum (Mo)	mg/kg	-	-	2.19 ± 0.49	0.58	0.87 ± 0.16	3.27 ± 0.34	0.94 ± 0.11
	Nickel (Ni)	mg/kg	$75^{\alpha,\beta}$	72	38.6 ± 1.6	53.2	55.5 ± 3.2	56.3 ± 2.6	59.9 ± 5.4
	Phosphorus (P)	mg/kg	$2,000^{\alpha}$	1,580	840 ± 47	1,110	881 ± 38	1,121 ± 57	865 ± 51
	Potassium (K)	mg/kg	-	-	3,894 ± 172	3,400	4,921 ± 607	5,891 ± 281	5,210 ± 924
	Selenium (Se)	mg/kg	-	-	0.49 ± 0.06	<0.20	0.21 ± 0.01	0.85 ± 0.06	0.23 ± 0.01
	Silver (Ag)	mg/kg	-	-	0.12 ± 0.01	<0.10	0.12 ± 0.01	0.27 ± 0.01	0.13 ± 0.02
	Sodium (Na)	mg/kg	-	-	296 ± 29	239	310 ± 33	455 ± 24	331 ± 53
	Strontium (Sr)	mg/kg	-	-	11.4 ± 0.5	13.8	13.0 ± 1.0	15.8 ± 0.6	13.5 ± 1.8
	Sulphur (S)	mg/kg	-	-	<5,000 ± 0	<5,000	<5,000 ± 0	<5,000 ± 0	<5,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.388 ± 0.021	0.331	0.491 ± 0.063	0.801 ± 0.035	0.504 ± 0.088
	Tin (Sn)	mg/kg	-	-	56.3 ± 28.9	4.1	6.9 ± 3.1	16.3 ± 7.8	8.3 ± 1.1
	Titanium (Ti)	mg/kg	-	-	1072 ± 36	965	1414 ± 94	1331 ± 69	1407 ± 159
	Uranium (U)	mg/kg	-	-	11.9 ± 1.52	3.78	7.63 ± 1.00	27.3 ± 1.52	8.58 ± 1.77
	Vanadium (V)	mg/kg	-	-	50.0 ± 1.3	46.8	57.0 ± 5.3	72.0 ± 3.6	58.8 ± 8.3
	Zinc (Zn)	mg/kg	315	135	73.7 ± 2.7	49.8	68.6 ± 7.0	105 ± 5.1	70.0 ± 10.6
	Zirconium (Zr)	mg/kg	-	-	4.3 ± 0.6	9.3	19.4 ± 1.9	4.0 ± 0.2	20.2 ± 3.2

[a] Canadian Sediment Quality Guideline for the protection of aquatic life, probable effects level (PEL; CCME 2015) except those indicated by α (Ontario Provincial Sediment Quality Objective [PSQO], severe effect level (SEL); OMOE 1993) and β (British Columbia Working Sediment Quality Guideline [BCSQG], probable effects level (PEL; BCMOE 2015)).

[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 Mary Lake.

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

BOLD Indicates parameter concentration above the AEMP Benchmark.

Lake in 2016 (Table 5.3; Appendix Table D.26). Although sediment chromium and iron concentrations were above respective SQG at some individual littoral and profundal stations of the Mary Lake south basin, average concentrations of these metals were below the applicable guidelines (Table 5.3; Appendix Table D.26). Notably, concentrations of manganese (and iron) were elevated above SQG in sediment at the reference lake profundal stations, suggesting that concentrations of manganese above guidelines at Mary Lake may reflect natural conditions un-related to mine activity. No metals were observed at concentrations above the sediment AEMP benchmarks at littoral and profundal stations of the Mary Lake north or south basins (Table 5.3; Appendix Table D.26).

Temporal comparisons of the sediment metals data suggested only a slight elevation (i.e., 2- to 5-fold higher) in manganese concentrations at Mary Lake littoral stations, but similar metal concentrations at Mary Lake profundal stations, between 2016 and the baseline period[14] (Figure 5.12; Appendix Table D.27). With the exception of sediment manganese concentrations at littoral stations of Mary Lake, no metals showed progressively higher concentrations from mine baseline, to mine construction, to 2015 and 2016 mine operational years in sediment of Mary Lake littoral or profundal stations (Figure 5.12). Similar to the other mine-exposed lakes, slight variation in station locations and/or data treatment among studies likely contributed to the appearance of higher average manganese concentrations in sediment at the Mary Lake littoral stations in 2015 and 2016 compared to the mine baseline/construction periods. In addition, concentrations of all metals at Mary Lake sediment stations, including manganese, were comparable to those of the reference lake littoral and/or profundal sediment stations (Figure 5.12), suggesting no mine influence on sediment metal chemistry of Mary Lake since the onset of Mary River Project mine operations.

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 2016 (Figure 5.13). Similar to the other mine-exposed lakes, chlorophyll a concentrations generally showed significant differences among winter, summer and fall sampling events at both the north and south basins of Mary Lake in 2016 (Appendix Table E.4). Highest and lowest concentrations of chlorophyll a were observed in summer and winter, respectively, at both Mary Lake basins (Appendix Table E.14), and mirrored the summer and fall seasonal differences observed at the reference lake (Appendix Table B.8). Although chlorophyll a concentrations at the Mary Lake north basin were

[14] Refer to footnote 6 (page 32) regarding temporal differences in sediment boron concentrations at Mary River Project LSA waterbodies.

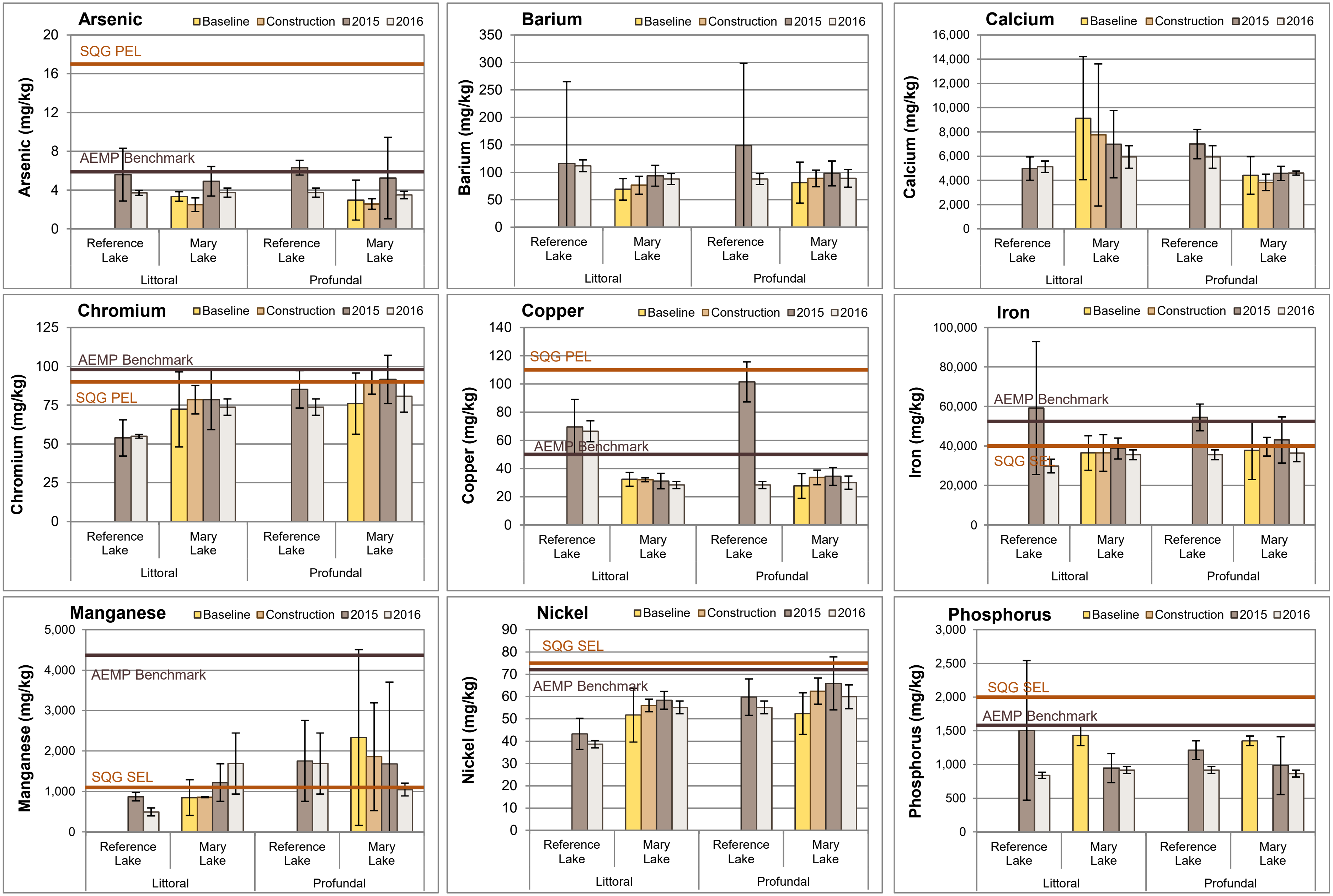


Figure 5.12: Temporal comparison of sediment metal concentrations (mean ± SD) at littoral and profundal stations of Mary Lake and Reference Lake 3 for mine baseline (2005 - 2013), construction (2014) and operational (2015, 2016) periods, Mary River Project CREMP.

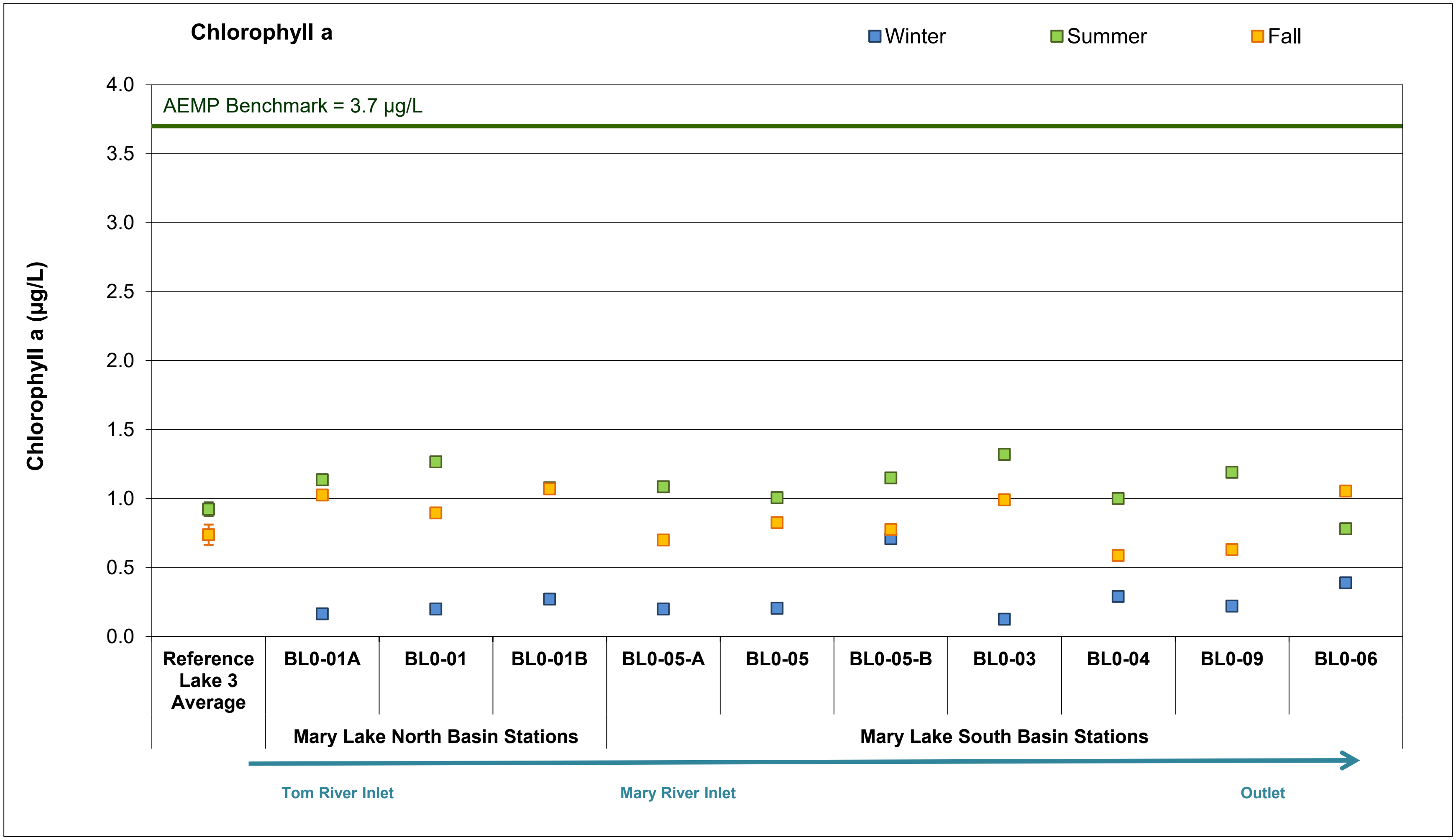


Figure 5.13: Chlorophyll a concentrations at Mary Lake (BLO) phytoplankton monitoring stations, Mary River Project CREMP, 2016. Values presented are averages of samples taken from the surface and the bottom of the water column at each station. Reference values represent mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2016.

significantly higher than at the reference lake, concentrations did not differ significantly between the Mary Lake south basin and Reference Lake 3 for both the summer and fall sampling events in 2016 (Appendix Tables E.5 and E.6). The Mary Lake chlorophyll a concentrations were well below the AEMP benchmark of 3.7 µg/L during all winter, summer and fall sampling events in 2016 (Figure 5.13). Chlorophyll a concentrations at Mary Lake reflected an 'oligotrophic' primary productivity categorization (sensu Wetzel 2001), which agreed closely with an 'oligotrophic' CWQG categorization based on mean aqueous total phosphorus concentrations between 4 – 10 µg/L for the Mary Lake winter, summer and fall sampling events in 2016 (Table 5.2; Appendix Tables C.58 – C.59).

Temporal comparisons of the Mary Lake chlorophyll a data did not indicate any significant differences among the mine construction (2014) and operational (2015, 2016) yearly data that were consistent over the winter, summer or fall seasons with the exception of significantly higher concentrations in fall 2016 than in fall 2014 (Figure 5.14; Appendix Tables E.14 and E.15). In addition, annual average chlorophyll a concentrations did not differ significantly among 2014, 2015 and 2016 (Appendix Tables E.15 and E.16), suggesting no changes in the trophic status of Mary Lake since mine operations commenced at the Mary River Project. No chlorophyll a baseline (2005 – 2013) data are available for Mary Lake, precluding comparisons to conditions prior to the mine construction period.

5.2.4 Benthic Invertebrate Community

Benthic invertebrate community richness was significantly lower at Mary Lake compared to Reference Lake 3, but density and Simpson's Evenness did not differ significantly between lakes for littoral station samples collected in 2016 (Table 5.4). Although differences in benthic invertebrate community structure were indicated between Mary Lake and Reference Lake 3 based on significantly differing Bray-Curtis Index, only the relative abundance of dominant taxonomic groups differed significantly between lakes and not the proportion of key FFG and HPG (Table 5.4). Similar to the other mine-exposed lakes, significantly lower and higher relative abundance of seed shrimp and chironomids, respectively, at Mary Lake compared to the reference lake potentially reflected lower sediment TOC content, higher proportion of fine-grained sediments and/or more compact sediment (i.e., lower moisture content) at the Mary Lake littoral stations (Appendix Table D.25). Because the relative abundance of metal-sensitive Chironomidae did not differ significantly between Mary Lake and Reference Lake 3 (Table 5.4), the difference in benthic invertebrate community structure between lakes did not appear to be associated with an ecological response to aqueous and/or sediment metal concentrations.

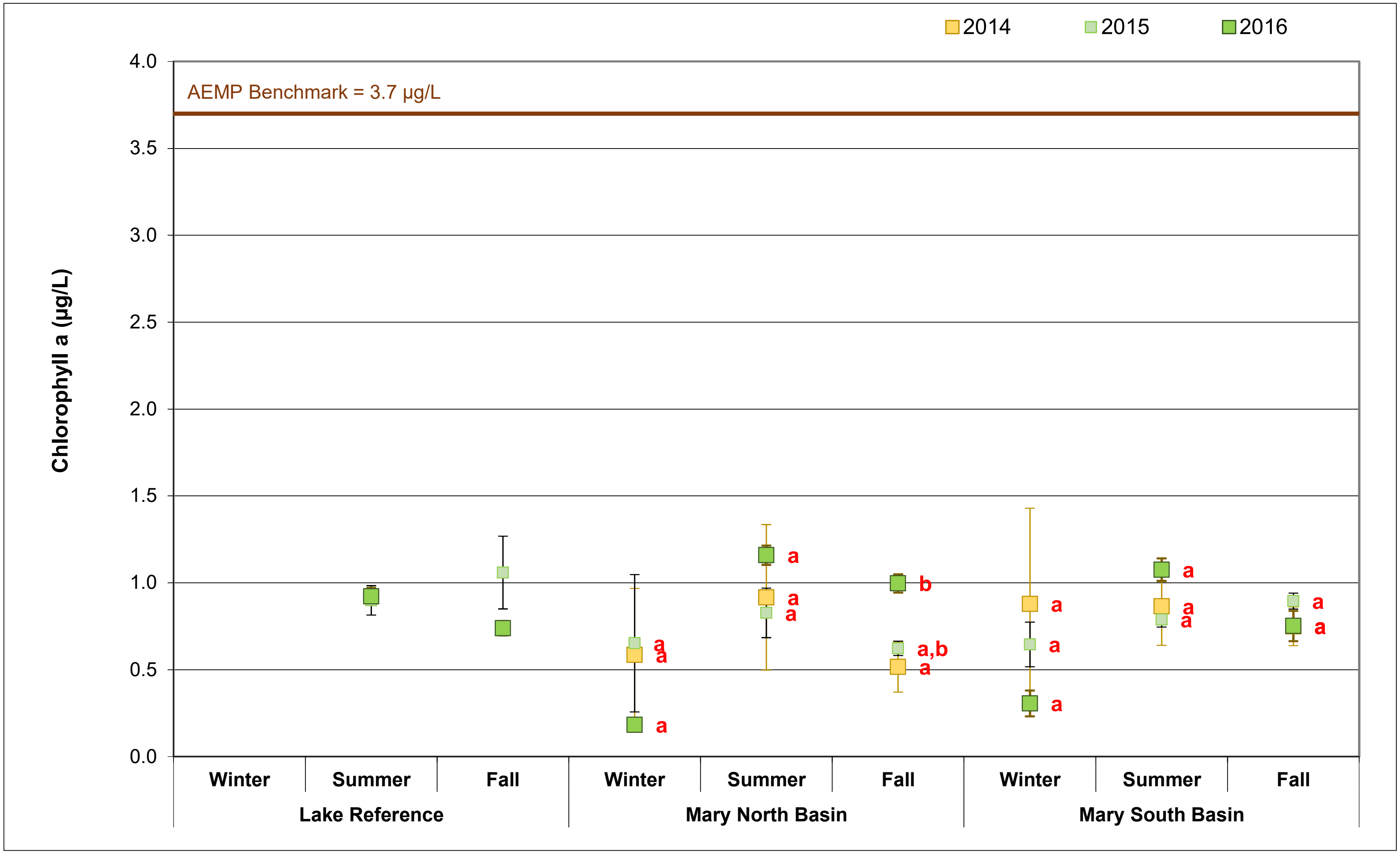


Figure 5.14: Chlorophyll a concentration seasonal comparison among 2014, 2015 and 2016 years (mean ± SE) at Mary Lake phytoplankton monitoring stations, Mary River Project CREMP. Data points with the same letter on the right do not differ significantly between years for the applicable season.

Table 5.4: Benthic invertebrate community statistical comparison results between Mary Lake (BLO) and Reference Lake 3 littoral stations, Mary River Project CREMP, August 2016.

Metric	Statistical Test Results					Summary Statistics					
	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Power	Magnitude of Difference [b] (No. of SD)	Area	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	No	0.483	ι, δ, γ	-	-	Reference Lake 3	2,390	1,396	624	897	4,240
						Mary Lake Littoral	1,947	1,591	649	457	4,036
Richness (Number of Taxa)	Yes	<0.001	ι, δ, γ	1.000	-3.2	Reference Lake 3	12.2	1.1	0.5	11.0	14.0
						Mary Lake Littoral	8.7	0.5	0.2	8.0	9.0
Simpson's (E) Krebs	No	0.249	γ	-	-	Reference Lake 3	0.758	0.189	0.084	0.420	0.849
						Mary Lake Littoral	0.574	0.299	0.122	0.249	0.958
Bray-Curtis Index	Yes	0.000	α, δ, γ	1.000	4.0	Reference Lake 3	0.334	0.122	0.054	0.245	0.527
						Mary Lake Littoral	0.820	0.093	0.038	0.642	0.902
Nemata (%)	No	0.670	β, δ, γ	-	-	Reference Lake 3	4.0%	5.6%	2.5%	0.0%	13.5%
						Mary Lake Littoral	3.6%	7.5%	3.1%	0.0%	18.8%
Hydracarina (%)	No	0.382	β, δ, γ	-	-	Reference Lake 3	3.6%	2.0%	0.9%	1.8%	6.7%
						Mary Lake Littoral	3.3%	4.2%	1.7%	0.7%	11.4%
Ostracoda (%)	Yes	0.004	β, ε, γ	0.982	-2.6	Reference Lake 3	46.9%	17.5%	7.8%	37.8%	78.0%
						Mary Lake Littoral	2.3%	2.2%	0.9%	0.0%	5.0%
Chironomidae (%)	Yes	0.002	β, δ, γ	0.992	2.4	Reference Lake 3	45.4%	18.8%	8.4%	15.4%	59.2%
						Mary Lake Littoral	90.6%	12.2%	5.0%	66.1%	99.1%
Metal-Sensitive Chironomidae (%)	No	1.000	γ	-	-	Reference Lake 3	19.3%	8.3%	3.7%	7.7%	28.1%
						Mary Lake Littoral	19.2%	13.3%	5.4%	1.7%	33.9%
Collector-Gatherers (%)	No	0.865	β, δ, γ	-	-	Reference Lake 3	75.0%	11.4%	5.1%	61.1%	89.7%
						Mary Lake Littoral	73.5%	24.7%	10.1%	28.2%	94.9%
Filterers (%)	No	0.246	β, ε, γ	-	-	Reference Lake 3	16.1%	8.4%	3.8%	7.0%	26.4%
						Mary Lake Littoral	12.4%	13.2%	5.4%	0.0%	31.6%
Clingers (%)	No	0.457	β, δ, γ	-	-	Reference Lake 3	19.2%	7.6%	3.4%	8.8%	28.3%
						Mary Lake Littoral	16.5%	13.1%	5.3%	1.7%	37.4%
Sprawlers (%)	No	0.855	β, δ, γ	-	-	Reference Lake 3	65.7%	12.1%	5.4%	57.2%	85.7%
						Mary Lake Littoral	64.8%	32.9%	13.4%	8.0%	97.4%
Burrowers (%)	No	0.581	β, δ, γ	-	-	Reference Lake 3	15.1%	6.2%	2.8%	5.5%	22.2%
						Mary Lake Littoral	18.7%	21.4%	8.7%	0.9%	54.6%

[a] Data analysis included: α - data untransformed; β - data logit transformed; ι - $\log_{10}$ transformed; δ - single factor ANOVA test conducted; ε - t-test assuming unequal variance; γ - ANOVA test validated using Mann Whitney U-test.

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

Highlighted values indicate significant differences between study areas based on ANOVA p-value less than 0.10 that were also outside of a Critical Effect Size of ±2 SD, suggesting an ecologically meaningful difference.

BOLD Bold text values indicate significant differences between study areas based on ANOVA p-value less than 0.10, but a Critical Effect Size within ±2 SD, suggesting the difference is not ecologically meaningful.

Temporal comparisons of the Mary Lake benthic invertebrate community data did not indicate any significant differences in density, richness, Simpson's Evenness and the relative abundance of dominant taxonomic groups and FFG among data collected in 2015 and 2016 mine operational years and in 2007 prior to mine operation (i.e., baseline; Figure 5.15; Appendix Table F.44). The close similarity in benthic invertebrate community endpoints among years was consistent with the relatively minor changes in water and sediment chemistry observed at Mary Lake between the mine operational and baseline periods (Sections 5.2.1 and 5.2.2). Moreover, no mine-related influence on lotic benthic invertebrate communities was apparent within the Mary River downstream of the mine, suggesting a low potential for mine-related effects to biota of Mary Lake. The benthic invertebrate community at littoral stations of Mary Lake showed consistently lower and higher relative abundance of seed shrimp and chironomids, respectively, compared to the reference lake in both 2015 and 2016, but no consistent differences in richness and Simpson's Evenness, and no differences entirely for density, FFG and HPG endpoints (Appendix Table F.44). This suggested that factors contributing to differences in benthic invertebrate community structure between Mary Lake and Reference Lake 3 remained relatively unchanged over the 2015 to 2016 studies.

5.2.5 Fish Population

5.2.5.1 Mary Lake (South) Fish Community

Arctic charr and ninespine stickleback comprised the fish community of Mary Lake, mirroring the fish species composition observed at Reference Lake 3 (Table 5.5). Similar to the other mine-exposed lakes, Arctic charr CPUE was much higher at Mary Lake than at the reference lake for electrofishing and gill netting collection methods, suggesting higher densities and/or productivity of Arctic charr at Mary Lake. Consistent with the other mine-exposed lakes, greater numbers of Arctic charr, together with greater density of benthic invertebrates, suggested that secondary productivity was higher at Mary Lake than at Reference Lake 3.

Temporal comparison of the Mary Lake electrofishing catch data indicated that Arctic charr CPUE was much higher in 2016 and other years of mine construction/operation than during baseline monitoring conducted in 2008 (Figure 5.16). Similar to other mine-exposed lakes, Arctic charr CPUE for gill net collections was markedly higher in 2016 compared to all previous baseline (2007 – 2008), mine construction (2014) and mine operational (2015) studies (Figure 5.15), reflecting efficiencies in sampling relative to previous studies. Overall, the CPUE data were not indicative of temporal changes in the relative abundance of Arctic charr at the nearshore or littoral/profundal areas of Mary Lake.

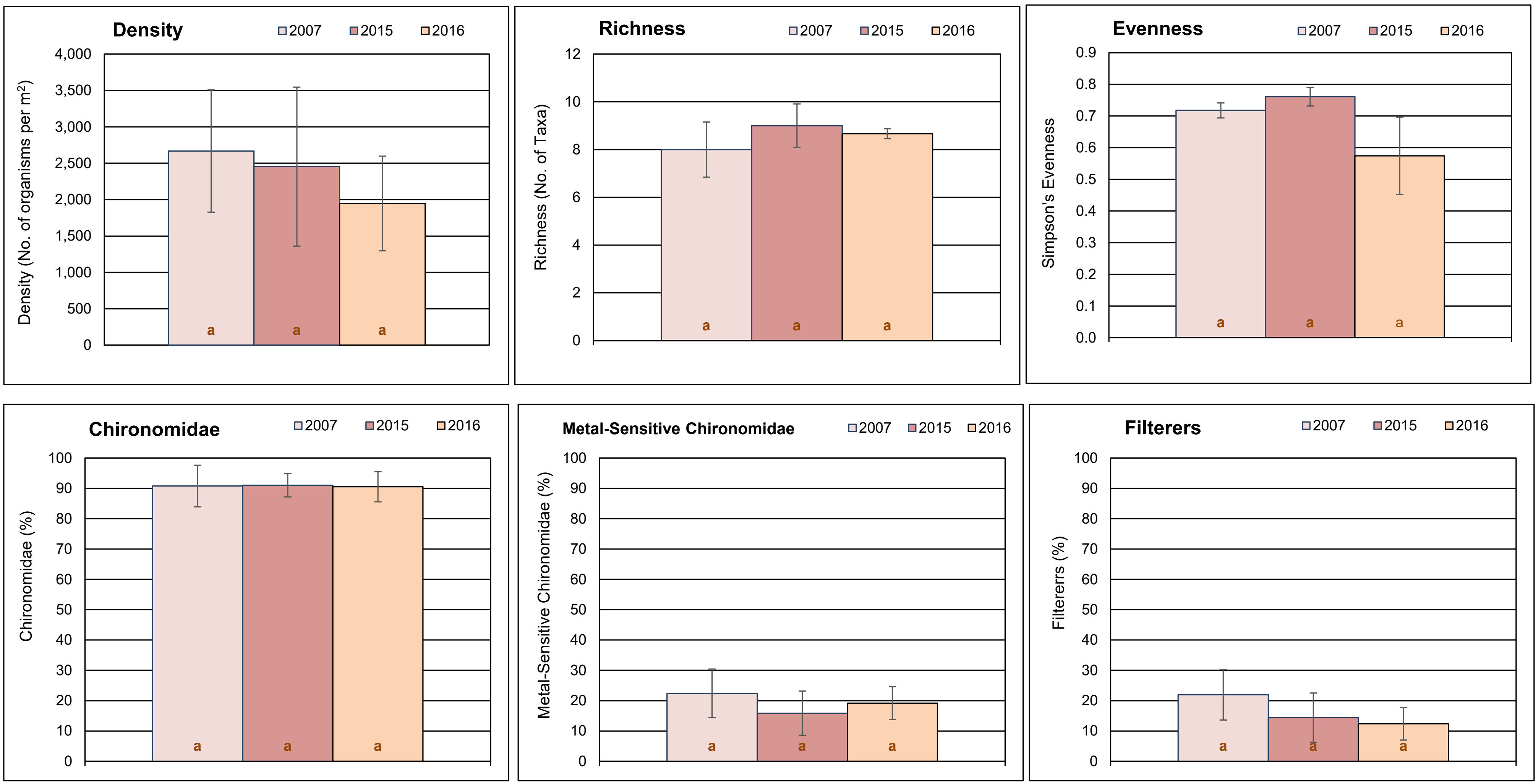


Figure 5.15: Comparison of key benthic invertebrate metrics (mean ± SE) at Mary Lake littoral stations between mine baseline (2007) and operational (2015, 2016) periods, Mary River Project CREMP, 2016. The same like-coloured letter inside bars indicate no significant difference between areas.

Table 5.5: Fish catch and community summary from backpack electrofishing and gill netting conducted at Mary Lake (BLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2016.

Lake	Method [a]		Arctic Charr	Nine-spine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	101	28	129	2
		CPUE	0.48	0.16	0.64	
	Gill netting	No. Caught	14	0	14	
		CPUE	0.15	0	0.15	
Mary Lake	Electrofishing	No. Caught	107	1	108	2
		CPUE	1.36	0.01	1.38	
	Gill netting	No. Caught	97	0	97	
		CPUE	5.31	0	5.31	

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

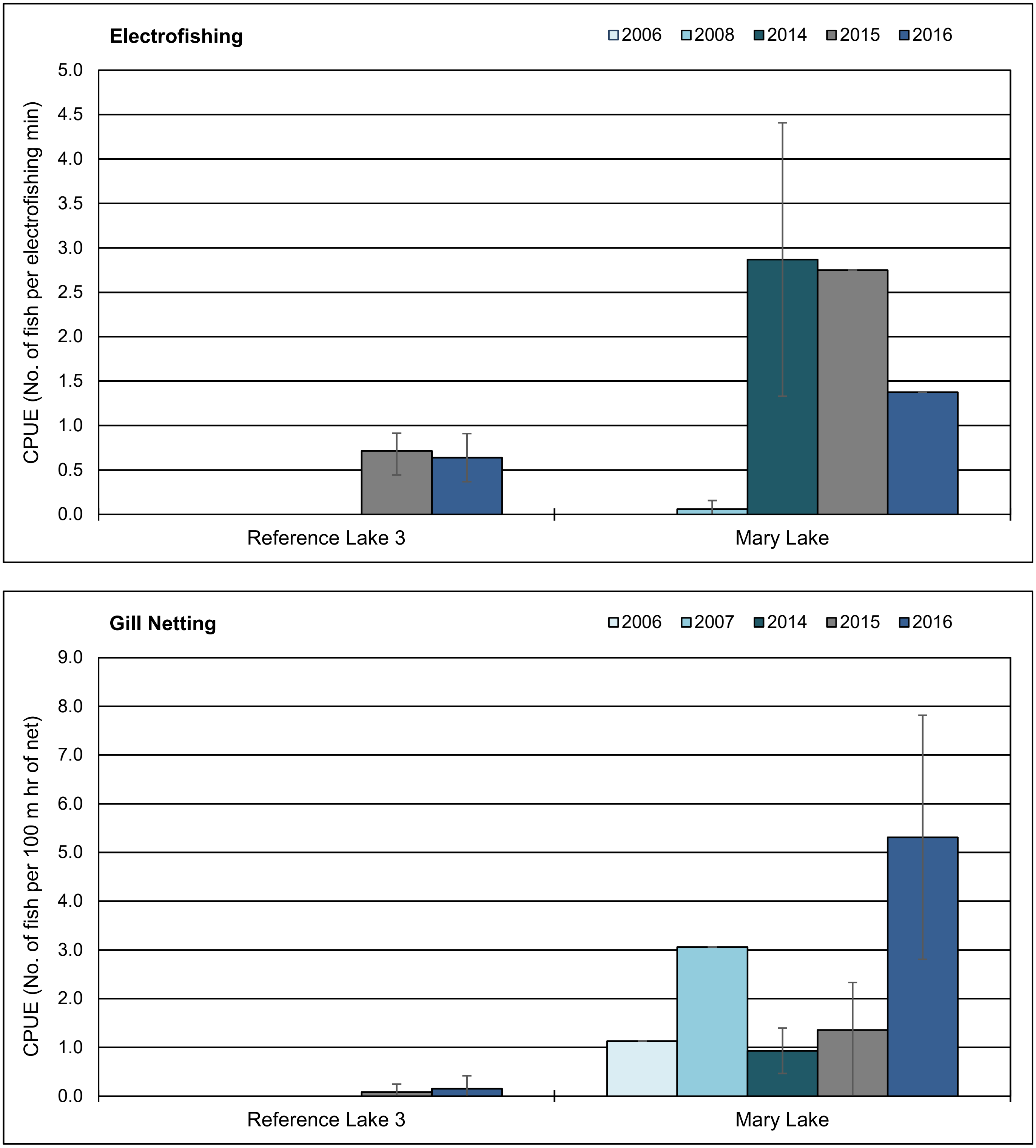


Figure 5.16: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic charr captured by backpack electrofishing and gill netting at Mary Lake (BLO) for baseline (2006, 2007, 2008), mine construction (2014) and operational (2015, 2016) periods during the fall, Mary River Project CREMP.

5.2.5.2 Mary Lake (South) Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Mary Lake nearshore Arctic charr population were assessed with a control-impact analysis using data collected from Mary Lake and Reference Lake 3 in 2016. No nearshore Arctic charr baseline data were collected at Mary Lake, precluding data analysis using a before-after design. A total of 100 Arctic charr captured at nearshore habitat at each of Mary Lake and Reference Lake 3 in August 2016 were used for the control-impact analysis. Distinction of Arctic charr YOY from the older, non-YOY age class was possible using a fork length cut-off of 4.9 and 5.1 cm based on the evaluation of length-frequency distributions coupled with supporting age determinations for the Mary Lake and Reference Lake 3 data sets, respectively (Figure 5.17). Due to a low number of Arctic charr YOY captured at the Mary Lake nearshore (i.e., 5), fish health comparisons were conducted using only non-YOY individuals, where applicable, to limit confounding influences of naturally differing weight-at-length relationships between YOY and non-YOY individuals on the data interpretation.

Nearshore Arctic charr length-frequency distributions differed significantly between Mary Lake and Reference Lake 3, reflecting the occurrence of very few YOY and greater numbers of larger individuals at Mary Lake (Table 5.6; Figure 5.17; Appendix Table G.34). However, nearshore Arctic charr non-YOY size, growth and condition did not differ significantly between Mary Lake and Reference Lake 3 in 2016 (Table 5.6; Appendix Table G.34). Fewer significant differences between nearshore Arctic charr populations of Mary Lake and Reference Lake 3 were evident in 2016 than in 2015 (Table 5.6). The dissimilarity in endpoints that differed between studies may have reflected small samples sizes of approximately ten individuals used for the age and growth endpoint comparisons during each study. Nevertheless, similar to the other mine-exposed lakes, no adverse mine-related influences on nearshore Arctic charr energy use and storage were suggested at Mary Lake for either of the 2015 and 2016 studies.

Littoral/Profundal Arctic Charr

Mine-related influences on the Mary Lake littoral/profundal Arctic charr population were assessed with a before-after analysis using data collected from Mary Lake in 2016 and during 2006-2007 baseline monitoring. Similar to the 2015 CREMP, a small sample size from Reference Lake 3 (i.e., n = 14) precluded meaningful control-impact statistical analysis using data collected in 2016. Biological information collected from Arctic charr mortalities indicated that non-spawners of reproductive age constituted approximately 63% of the Mary Lake Arctic charr population during the August 2016 field study (Appendix Table G.38). On average, Arctic charr non-spawners exhibited similar age, size (length and weight) and LSI than females with

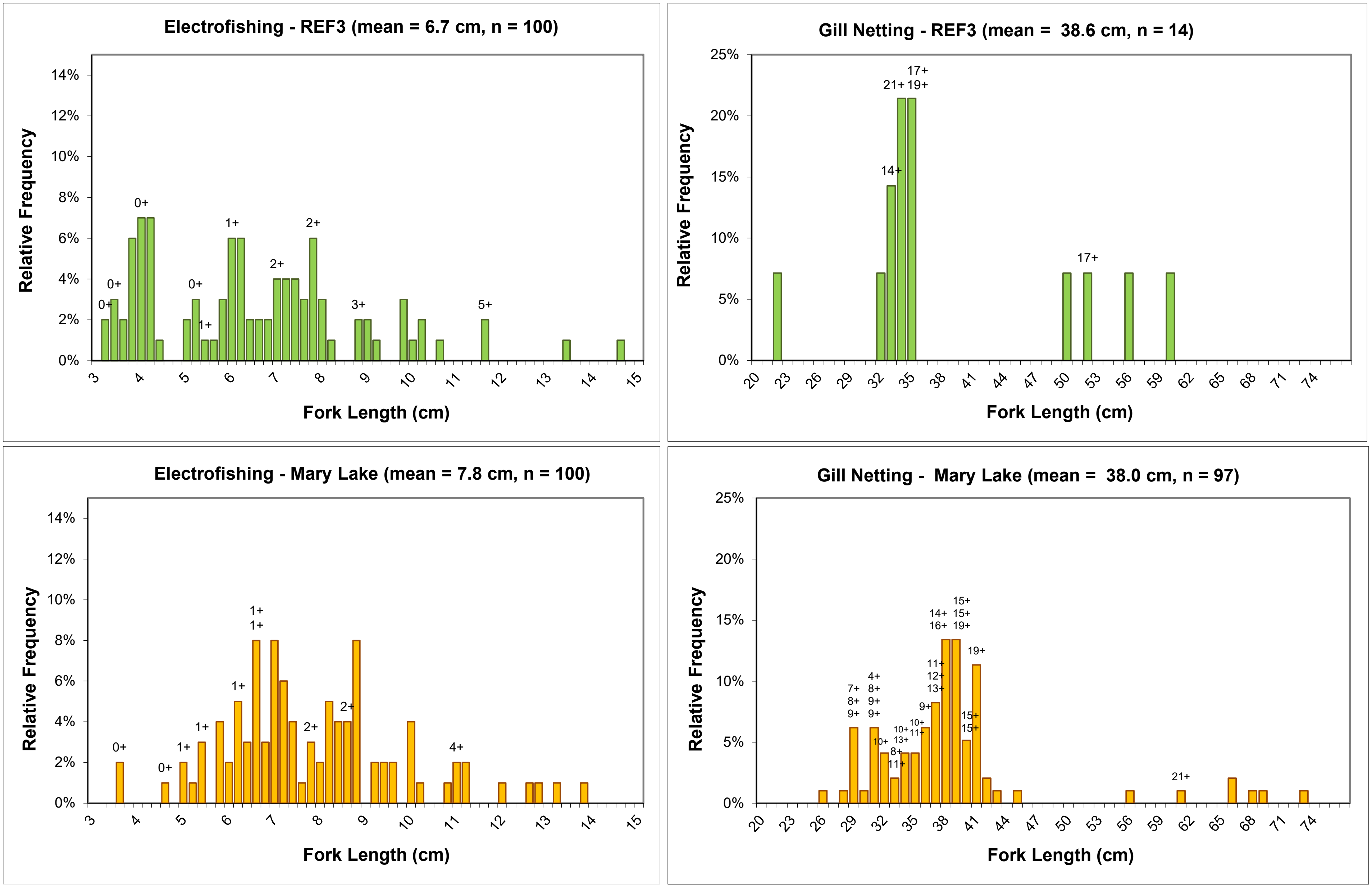


Figure 5.17: Length-frequency distributions for Arctic charr captured by backpack electrofishing and gill netting at Mary Lake and Reference Lake 3 (REF3), August 2016, Mary River Project CREMP. Fish ages are shown above the bars, where available.

Table 5.6: Summary of statistical results for Arctic charr population comparisons between Mary Lake and Reference Lake 3 for the mine operational period (2015, 2016) and between Mary Lake mine-operational and baseline period data for fish captured by electrofishing and gill netting methods, Mary River Project CREMP, August 2016. Values in parentheses indicate direction and magnitude of any signficant differences.

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed?			
			versus Reference Lake 3		versus Mary Lake baseline period data [b]	
			2015	2016	2015	2016
Electrofishing Samples	Survival	Length-Frequency Distribution	No	**Yes**	-	-
		Age	**Yes (-43%)**	No	-	-
	Energy Use	Size (mean weight)	No	No	-	-
		Size (mean fork length)	No	No	-	-
		Growth (weight-at-age)	**Yes (+99%)**	No	-	-
		Growth (fork length-at-age)	**Yes (+23%)**	No	-	-
	Energy Storage	Condition (body weight-at-fork length)	**Yes (+3%)**	No	-	-
Gill Netting Samples [a]	Survival	Length Frequency Distribution	-	-	**Yes**	**Yes**
		Age	-	-	No	**Yes (-14%)**
	Energy Use	Size (mean weight)	-	-	**Yes (+19%)**	No
		Size (mean fork length)	-	-	**Yes (+6%)**	No
		Growth (weight-at-age)	-	-	No	**Yes (nc)**
		Growth (fork length-at-age)	-	-	No	**Yes (nc)**
	Energy Storage	Condition (body weight-at-fork length)	-	-	No	**Yes (+3%)**

[a] Due to low catches of Arctic charr at Reference Lake 3 in 2015 and 2016, no comparison of fish health was possilbe for gill netted fish.

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

developing gonads (Appendix Table G.38). A high proportion of individuals (i.e., 85%) also contained body cavity parasites (Appendix Table G.38), the incidence rate of which was comparable to that observed at other mine-exposed lakes and during historical studies at mine LSA lakes. One Arctic charr that had been tagged and released previously at Mary Lake was re-captured in 2016, and showed a 26.3 mm/yr average increase in fork length over the past 9 years (Table 5.7). This growth rate showed close agreement with the incremental change in growth rate for a recaptured tagged individual from Mary Lake in 2015 (Table 5.7), as well as resident populations in other Arctic lakes available in published literature. Growth of tagged Arctic charr appeared to be considerably higher at Mary Lake than at the northwest and southeast basins of Sheardown Lake, where tagged Arctic charr showed a mean annual incremental increase in fork length of 7.5 mm/yr (Tables 4.7 and 5.7; Minnow 2016a). The tagging information suggested that Arctic charr may reside in a same lake for a prolonged period, and that faster growth rates in Arctic charr may be associated with larger lake size.

Table 5.7: Length and weight measurement data for tagged Arctic charr captured at the Mary Lake south basin in August 2015 and 2016, Mary River Project CREMP.

Fish Tag Number	Capture Information			Re-Capture Information			Growth Rate
	Date of Capture	Length (mm)	Weight (g)	Date of Capture	Length (mm)	Weight (g)	Δ Length (mm/yr)
83214	30-Jul-2007	392	500	19-Aug-2015	587	2,250	24.4
85533	29-Jul-2007	422	725	19-Aug-2016	660	>2,500	24.4

Length-frequency distributions of Arctic charr captured at littoral/profundal areas of Mary Lake in 2016 differed significantly from those captured during the baseline period (Table 5.6; Appendix Table G.37). On average, Arctic charr captured at littoral/profundal areas of Mary Lake in 2016 were significantly younger than those captured during the baseline period, but no significant differences in mean size were shown between these mine periods (Table 5.6). No definitive differences in adult Arctic charr growth were indicated between 2016 and the baseline data at Mary Lake based on considerable overlap of data (Appendix Figure G.23). Similarly, although adult Arctic charr condition differed significantly between 2016 and the baseline period, the magnitude of this difference was within ±10% CES_C (Table 5.6) suggesting that this difference was not ecologically meaningful. The responses of Arctic charr captured at littoral/profundal areas of Mary Lake in 2016 were generally not consistent with those documented in 2015 for comparisons to baseline, which potentially reflected natural sampling variability between years.

5.3 Synthesis of Mine-Related Influences at the Mary River and Mary Lake System

5.3.1 Mary River

No mine-related influences on water quality were apparent at Mary River in 2016. Although total concentrations of a number of metals, including aluminum, chromium, copper, iron, lead, manganese, nickel and phosphorus, were elevated at one or more mine-exposed areas of the Mary River in 2016 compared to reference and baseline data, naturally high turbidity in 2016 likely accounted for these spatial and temporal differences. This was supported by the occurrence of similar dissolved metal concentrations in 2016 compared to Mary River reference and baseline data, by significant positive correlations between total concentrations of key metals (e.g., aluminum, manganese) and turbidity, and by observations of high ratios of total to dissolved metal concentrations for the Mary River water quality data. Notably, turbidity within Mary River was consistently highest upstream of the mine (i.e., the GO series stations) during all mine baseline (2005 – 2013), construction (2014) and operational (2015, 2016) periods, indicating that the dominant source of turbidity at mine-exposed areas of the Mary River reflected natural (runoff) inputs unrelated to the mine operation. Although total aluminum, copper, iron, lead and phosphorus concentrations were above WQG and/or AEMP benchmarks at one or more Mary River mine-exposed stations in 2016, as discussed above, the elevation in these metals compared to water quality criteria appeared to be associated with naturally high turbidity.

Chlorophyll a concentrations were similar among the ten Mary River phytoplankton monitoring stations, with no significant differences in annual chlorophyll a concentrations indicated between Mary River mine-exposed and reference stations. Although lower chlorophyll a concentrations were indicated at individual Mary River stations in 2016 compared to the baseline period, these differences likely reflected higher natural turbidity in 2016, which would be expected to affect phytoplankton productivity by limiting the amount of light available for photosynthesis. No adverse or ecologically meaningful significant differences in benthic invertebrate density, richness or relative abundance of metal-sensitive taxa were shown between Mary River mine-exposed areas compared to an upstream reference area (i.e., GO- 09) in 2016. Although some differences in community composition were indicated between the Mary River mine-exposed and reference areas in 2016, these differences appeared to be related to naturally greater substrate embeddedness at the mine-exposed areas rather than a mine-related influence. Temporal comparisons indicated significantly higher Simpson’s Evenness and significantly lower relative abundance of chironomid midges at Mary River mine-exposed areas compared to the reference area between the 2016 and baseline studies. However, because the direction of these responses was opposite to those typically related to adverse mine-related effects, natural temporal variability and/or sampling artifacts of the CREMP likely accounted for the temporal differences in these endpoints.

5.3.2 Mary Lake

At Mary Lake, turbidity and aqueous concentrations of total aluminum, manganese and uranium were elevated (i.e., ≥3-fold higher) compared to the reference lake in 2016, but none of these metals, or any other parameters, were consistently elevated compared to concentrations observed during the baseline period, and none were above WQG or AEMP benchmarks. Similar to Sheardown Lake, turbidity at Mary Lake was naturally higher than the reference lake as a result of receiving flow from relatively large river systems (i.e., Tom River and Mary River inflows to the Mary Lake north and south basins, respectively). Aluminum and manganese were consistently shown to be associated with turbidity at all mine lakes, including Mary Lake, which suggested that these metals were largely bound to/comprised the suspended particulate matter and were thus unlikely to be biologically available. Sediment metal concentrations at Mary Lake littoral and profundal stations were similar to those at the reference lake in 2016 and, with the exception of slightly elevated sediment manganese concentrations at littoral stations, were similar to concentrations observed during the baseline period. Although sediment chromium, iron and manganese concentrations were above SQG at Mary Lake in 2016, with the exception of chromium, these metals were also above SQG at the reference lake suggesting low potential for any adverse effects to biota associated with these metals. No metals were observed at concentrations above the sediment AEMP benchmarks at littoral and profundal stations of Mary Lake in 2016.

Mary Lake chlorophyll a concentrations were significantly higher than at the reference lake in 2016, but only at the north basin. However, Mary Lake chlorophyll a concentrations were continuously well below the AEMP benchmark during all seasonal sampling events in 2016, and were indicative of oligotrophic conditions normally encountered in Arctic waterbodies. No significant differences in annual chlorophyll a concentrations were indicated among the mine construction (2014) and operational (2015, 2016) periods, suggesting no changes in the trophic status of Mary Lake since commencement of mine operations. Benthic invertebrate community data collected at littoral habitat of Mary Lake in 2016 indicated significantly lower richness and relative abundance of ostracods, and significantly higher relative abundance of chironomids, but no differences in density, evenness and relative abundance of metal sensitive taxa, FFG and HPG compared to the reference lake. Similar to Sheardown Lake, the differences in community composition appeared to reflect naturally differing sediment TOC and/or particle size between Mary Lake and the reference lake in 2016. No significant differences in any primary and FFG benthic metrics were indicated between 2016 and 2007 baseline data for Mary Lake littoral habitat. Analysis of Mary Lake Arctic charr populations suggested greater fish abundance compared to the reference lake in 2016, but no definitive changes in numbers of Arctic charr in 2016 relative to baseline data. No significant or ecologically meaningful differences in growth and condition of nearshore captured Arctic charr occurred between Mary Lake and the reference lake in 2016, nor between Arctic charr collected in 2016 compared to the baseline period for nearshore and littoral/profundal Arctic charr populations at Mary Lake. Collectively, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Mary Lake in the second year of mine operation at the Mary River Project.

6.0 CONCLUSIONS

The objective of the 2016 Mary River Project CREMP was to evaluate potential mine-related influences on chemical and biological conditions at aquatic environments located near the mine following the second full year of ERP mine operation. Additional attention towards the evaluation of sedimentation-related effects was conducted as part of the 2016 CREMP assessment in consideration of an Environment and Climate Change Canada FAD and an Indigenous and Northern Affairs Canada LNC related to unauthorized sediment releases in 2016. The 2016 CREMP utilized an effects-based approach that included standard environmental effects monitoring techniques to provide rigorous analysis of potential mine-related effects at key receiving water bodies. 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 on comparisons of the 2016 data to applicable reference data and to available baseline data. Potential mine-related effects identified in the 2016 CREMP are provided separately below for the Camp, Sheardown and Mary River/Lake systems.

6.1 Camp Lake System

Within the Camp Lake system, mine-related effects on water quality were apparent mainly within the main stem channel of Camp Lake Tributary 1 (CLT1) and at Camp Lake. Conductivity and concentrations of mine parameters including chloride, nitrate, sulphate and metals including iron, manganese, molybdenum, sodium, strontium and uranium were the primary constituents reflecting a mine-related influence within CLT1 and Camp Lake in 2016 based on elevation (i.e., ≥3-fold higher) relative to reference conditions and/or to the baseline (2005 – 2013) period. Of these parameters, only iron and uranium concentrations were above applicable water quality guideline (WQG) and/or AEMP benchmarks, but only at the upper-most monitoring station on the CLT1 main stem. Active quarrying at the QMR2 pit in 2016 likely served as the key source for these parameters at CLT1. Water chemistry at Camp Lake Tributary 2 (CLT2) was similar to applicable reference stations and to baseline water quality, with all parameters consistently observed at concentrations below applicable WQG and AEMP benchmarks. Overall, mine-related effects to water quality of the Camp Lake system were evident at the upper main stem of CLT1 and Camp Lake, with minimal effects suggested at CLT2, following the second year of mine operation. Sediment arsenic and manganese concentrations were slightly elevated (i.e., 2- to 5-fold higher) at Camp Lake littoral stations compared to mean reference lake concentrations in 2016, and together with molybdenum, were also elevated compared to concentrations during the baseline period, suggesting a mine-related influence on sediment quality of Camp Lake. No metals were elevated in sediment of the profundal stations compared to the reference lake in 2016. Phosphorus was the only parameter observed at concentrations above SQG in littoral and profundal sediment of Camp Lake that was not also above applicable SQG at the reference lake in 2016.

Chlorophyll a concentrations were elevated at the upper main stem of CLT1 (Station L2-03) and within Camp Lake compared to respective reference areas and to baseline data, suggesting slight enrichment possibly related to higher aqueous nitrate and/or micro-nutrient concentrations from Mary River Project mine activities. However, chlorophyll a concentrations at CLT1 north branch and lower main stem areas, and at CLT2 in 2016, were comparable to applicable reference and baseline concentrations. In addition, chlorophyll a concentrations were consistently well below the AEMP benchmark at all Camp Lake system receivers in 2016 indicating no adverse mine influence to phytoplankton. No adverse mine-related influences on the benthic invertebrate community of the Camp Lake system, including CLT1, CLT2 and Camp Lake, were indicated in 2015 based on comparisons to respective reference areas and to baseline studies. In fact, consistent with the chlorophyll a data, benthic data collected at the upper main stem of CLT1 suggested a slight enrichment-related influence based on higher invertebrate density, richness and proportion of FFG filter feeders compared to Unnamed Reference Creek. The fish population survey suggested greater fish abundance compared to the reference lake in 2016, but similar numbers of Arctic charr in 2016 relative to the Camp Lake baseline studies. No significant, ecologically meaningful, differences in Arctic charr condition were indicated between Camp Lake and the reference lake in 2016, nor between Camp Lake Arctic charr collected in 2016 compared to the baseline period, for nearshore and littoral/profundal Arctic charr populations. Overall, consistent with the water chemistry and sediment chemistry generally meeting respective environmental quality guidelines and AEMP benchmarks, the phytoplankton, benthic invertebrate community and fish population survey data collectively suggested no adverse mine-related influences to the biota of the Camp Lake system in the second year of mine operation at the Mary River Project.

6.2 Sheardown Lake System

At Sheardown Lake Tributary 1 (SDLT1), aqueous concentrations of several parameters were elevated compared to average concentrations observed at the reference creek stations in 2016. However, similar to the 2015 CREMP, only nitrate and sulphate concentrations were elevated at SDLT1 in 2016 compared to the baseline period and, with the exception of copper, no parameters were present at concentrations above WQG or AEMP benchmarks in 2016. Within Sheardown Lake, aqueous total concentrations of aluminum, manganese, molybdenum and/or uranium were elevated compared to the reference lake in both 2015 and 2016, but none of these metals, or any other parameters, were elevated compared to concentrations observed during the baseline period, and none were above WQG or AEMP benchmarks. Similar to findings of the 2015 CREMP, elevated total aluminum and manganese concentrations were correlated with greater turbidity in 2016 suggesting that these metals were largely bound to/composed the suspended particulate matter and were not likely biologically available. Sediment metal concentrations at Sheardown Lake littoral stations in 2016 were similar to those at the reference lake and comparable to baseline with the exception of slightly elevated arsenic, manganese and/or molybdenum concentrations, suggesting some mine-related influences on Sheardown Lake sediment quality. However, sediment metal concentrations at Sheardown Lake profundal stations in 2016 were similar to the reference lake and baseline data, indicating that mine-related influences on sediment quality were confined to littoral habitats. Notably, no metals were present in sediment of Sheardown Lake at concentrations above SQG or AEMP benchmarks that were not above these criteria at the reference lake, suggesting the natural occurrence of elevated concentrations of some metals (e.g., iron, manganese) in sediment of lakes in the Mary River Project LSA.

Chlorophyll a concentrations at SDLT1 and Sheardown Lake were greater than concentrations observed at respective reference areas, but were similar to chlorophyll a concentrations reported during mine baseline and construction periods, respectively. In all cases, chlorophyll a concentrations were well below the AEMP benchmark at all Sheardown Lake system monitoring stations, suggesting no adverse mine-related effects to phytoplankton within the system. Consistent with higher chlorophyll a concentrations, greater relative abundance of FFG filterers and organism density at SDLT1 in 2016 compared to Unnamed Reference Creek and the baseline period, respectively, suggested a slight enrichment influence. However, a greater relative abundance of HPG burrowers at SDLT1 and SDLT12 compared to the Unnamed Reference Creek and to baseline data (SDLT12 only) was potentially indicative of sedimentation influences at these tributaries in 2016. No adverse mine-related influences to benthic invertebrate communities of SDLT9 and the Sheardown Lake littoral benthic invertebrate community were apparent in 2016 based on comparisons to respective reference areas and/or to baseline data. Greater Arctic charr abundance was suggested at the Sheardown Lake NW and SE basins compared to the reference lake in 2016, but similar relative numbers of Arctic charr were indicated between 2016 and baseline studies for both basins. The Arctic charr population exhibited different direction of significant responses in growth and condition between Sheardown Lake and the reference lake in 2016, and between Arctic charr collected at nearshore and littoral/profundal habitats for Sheardown Lake in 2016 compared to baseline studies. The differential responses in Arctic charr population endpoints suggested that the various differences between the mine-exposed and reference areas, or between studies at Sheardown Lake, reflected natural variability in the resident fish population. Overall, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Sheardown Lake in the second year of mine operation at the Mary River Project.

6.3 Mary River and Mary Lake System

At Mary River, no adverse mine-related influences on water chemistry were apparent at the mine-exposed areas in 2016 based on comparisons to the Mary River upstream reference area and to baseline period water chemistry taking influences of naturally high turbidity into account. At Mary Lake, aqueous total aluminum, manganese and uranium concentrations were elevated compared to the reference lake in 2016, but concentrations of these metals and all other parameters were comparable to concentrations during the baseline period, and none were above WQG or AEMP benchmarks. Similar to Sheardown Lake and Mary River, aluminum and manganese concentrations were correlated with turbidity at Mary Lake, which suggested that these metals were largely bound to/composed the suspended particulate matter and were thus unlikely to be biologically available. Sediment metal concentrations at Mary Lake littoral and profundal stations were similar to those at the reference lake in 2016 and, with the exception of slightly elevated sediment manganese concentrations at littoral stations, were similar to concentrations observed during the baseline period. Although sediment chromium, iron and manganese concentrations were above SQG at Mary Lake in 2016, with the exception of chromium, these metals were also above respective criteria at the reference lake suggesting low potential for any adverse effects to biota associated with these metals. No metals were observed at concentrations above the sediment AEMP benchmarks at littoral and profundal stations of Mary Lake in 2016.

Chlorophyll a concentrations at Mary River and Mary Lake were generally similar to, or slightly higher than, respective reference areas in 2016. Although lower chlorophyll a concentrations were indicated at individual Mary River stations in 2015 and 2016 compared to the baseline period, these differences likely reflected naturally turbidity in both 2015 and 2016, which would be expected to affect phytoplankton productivity by limiting the amount of light available for photosynthesis. In all cases, chlorophyll a concentrations were well below the AEMP benchmark, indicating no adverse mine-related influences to phytoplankton of the Mary River/Mary Lake system. The benthic invertebrate community of the Mary River exhibited few differences between mine-exposed and reference areas in 2016, and compared to respective areas during the baseline period, with the direction of the few indicated differences in community composition between areas/studies opposite those responses normally reflective of an adverse mine-related effect. Benthic invertebrate community data collected at littoral habitat of Mary Lake in 2016 indicated significantly lower richness and differences in community composition compared to the reference lake that appeared to reflect natural differences in sediment physical properties between lakes. In part, this was supported by no significant differences in benthic metrics between 2016 and 2007 baseline data for Mary Lake littoral stations. The fish population survey suggested greater fish abundance at Mary Lake compared to the reference lake in 2016. No significant or ecologically meaningful differences in growth and condition of nearshore captured Arctic charr occurred between Mary Lake and the reference lake in 2016, nor between Arctic charr collected in 2016 compared to the baseline period for nearshore and littoral/profundal Arctic charr populations at Mary Lake. Overall, the chlorophyll a, benthic invertebrate community and Arctic charr fish population data all suggested no adverse mine-related influences to the biota of Mary Lake in the second year of mine operation at the Mary River Project.

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