APPENDIX E.9.1

2018 CREMP MONITORING REPORT

(PART 2 OF 3)

Mary River Project 2018
Core Receiving Environment Monitoring Program Report

Part 2 of 3
(Sections 4 to 7)

Prepared for:
Baffinland Iron Mines Corporation
Oakville, Ontario

Prepared by:
Minnow Environmental Inc.
Georgetown, Ontario

March 2019

4 SHEARDOWN LAKE SYSTEM

4.1 Sheardown Lake Tributaries (SDLT1, SDLT12 and SDLT9)

4.1.1 Water Quality

Sheardown Lake Tributary 1 (SDLT1) dissolved oxygen (DO) concentrations were consistently near or slightly above saturation in spring, summer, and fall monitoring events in 2018 (Appendix Tables C.1 to C.3; Figure 4.1). Dissolved oxygen concentrations at SDLT1 and Sheardown Lake Tributary 12 (SDLT12) did not differ significantly from those at Unnamed Reference Creek during the August 2018 biological study (Figure 4.1). Although DO concentrations were significantly lower at Sheardown Lake Tributary 9 (SDLT9) than at Unnamed Reference Creek and the other Sheardown Lake tributaries, the DO concentrations were well above the WQG minimum for supporting sensitive life stages of cold-water biota (i.e., 9.5 mg/L) at all Sheardown Lake tributaries (Figure 4.1; Appendix Table C.31). *In situ* pH was significantly lower at SDLT1 and SDLT12 compared to Unnamed Reference Creek, whereas pH at SDLT9 did not differ significantly from reference conditions during the August 2018 biological study (Figure 4.1). 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. Specific conductance at each of the Sheardown Lake tributaries was significantly higher than at Unnamed Reference Creek during August 2018 biological sampling (Figure 4.1; Appendix Table C.33). Because specific conductance 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). Several parameters, including hardness, TDS, alkalinity, and concentrations of barium, chloride, nitrate, sulphate, total copper, total molybdenum, potassium, sodium, total strontium, and total uranium were elevated (i.e., ≥3-fold) at the SDLT1 stations compared to respective mean concentrations from the reference creek stations. Highest elevation of these parameters typically occurred during the spring sampling event, followed by the summer and fall sampling events (Appendix Tables C.34 and C.35). In addition to the parameters listed above, total manganese and total nickel concentrations were also elevated at the lower SDLT1 station compared to respective mean concentrations from the reference creek stations, but unlike the parameters listed above, the magnitude of elevation for total manganese and nickel concentrations (and

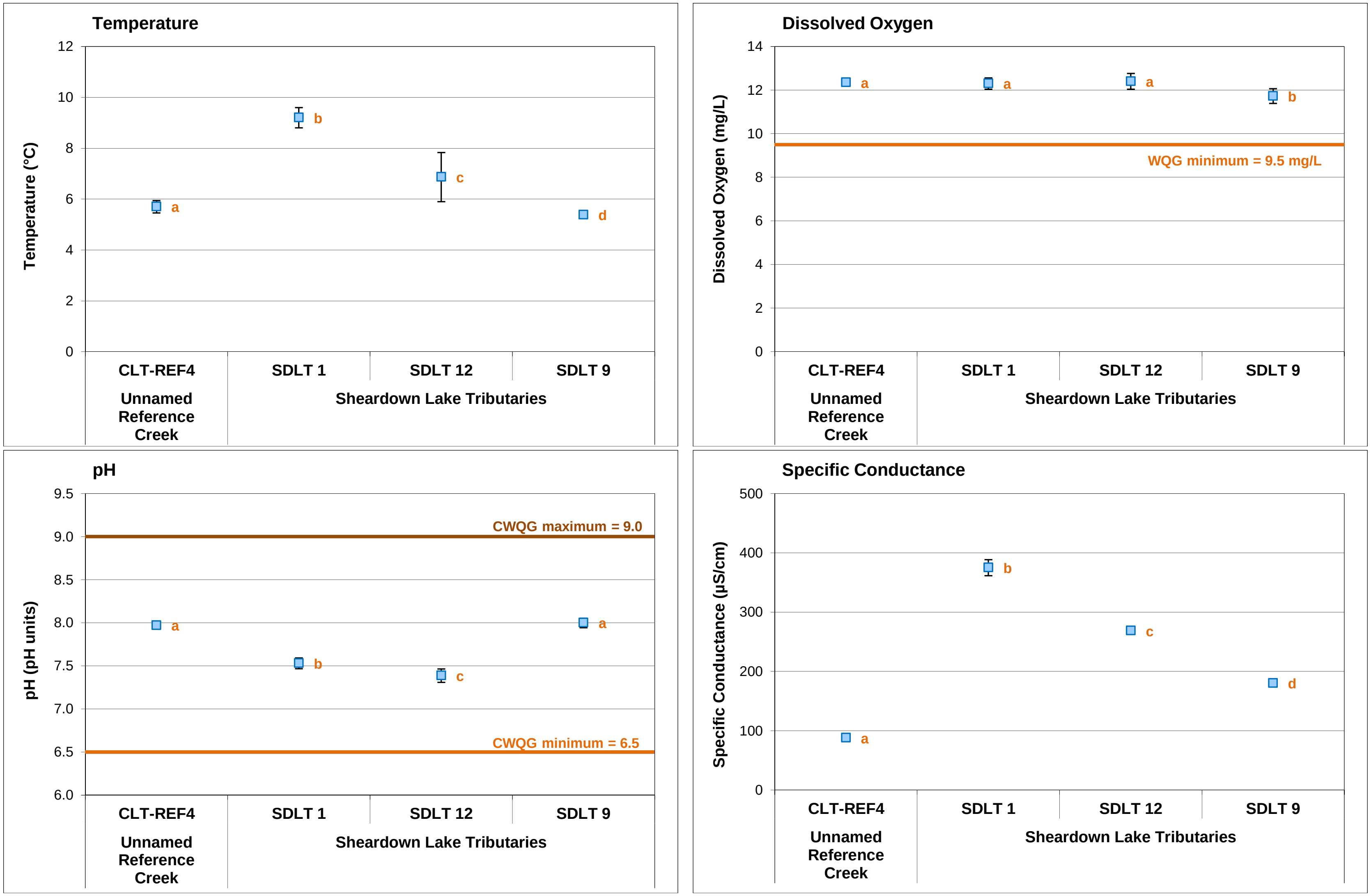


Figure 4.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Tributaries (SDLT) and Unnamed Reference Creek Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2018

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

nitrate) was smallest in spring and largest in fall (Appendix Tables C.34 and C.35). In most cases, higher parameter concentrations were observed at lower SDLT1 compared to upper SDLT1, suggesting that additional inputs of metals to SDLT1 occurred with distance downstream of the headwaters at the main mine camp (Table 4.1). On average, dissolved concentrations of copper, manganese, molybdenum, potassium, and uranium concentrations were elevated at SDLT1 compared to respective average concentrations from the reference creek stations during all seasonal sampling events in 2018, strongly suggesting a mine-related source for these parameters. Despite elevation of the aforementioned 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 2018 (Table 4.1; Appendix Table C.34).

Temporal comparisons of SDLT1 water chemistry data generally indicated the same parameters observed at elevated concentrations compared to the reference creek stations were also elevated compared to concentrations at the time of mine baseline at the lower SDLT1 monitoring station. In particular, conductivity, hardness, and concentrations of manganese, nickel, nitrate, sodium, strontium, sulphate, TDS, and uranium were elevated at lower SDLT1 in 2018 compared to respective concentrations during the mine baseline in at least one sampling season (Appendix Table C.35; Appendix Figure C.9; Figure 4.2). At upper SDLT1, concentrations of molybdenum, sodium, strontium, sulphate, and uranium were elevated in 2018 compared to baseline conditions only during the spring sampling event (Appendix Table C.35). Notably, total copper concentrations at SDLT1 in 2018 were generally comparable to those during the baseline period (Appendix Table C.35; Appendix Figure C.10), suggesting that concentrations of this metal were naturally high within this tributary prior to commencement of mine operations in 2015.

4.1.2 Phytoplankton

Among the Sheardown Lake tributaries, phytoplankton (chlorophyll-a) monitoring is conducted only at SDLT1 as part of the Mary River Project CREMP (Table 2.1). Chlorophyll-a concentrations were higher at upper SDLT1 (Station D1-05) compared to near the creek mouth (Station D1-00) during each of the spring, summer, and fall sampling events in 2018 (Figure 4.3). Nitrate concentrations were higher near the mouth of SDLT1 (Appendix Table C.34), and therefore lower chlorophyll-a concentrations near the mouth were contrary to typical responses of phytoplankton to higher nutrient concentrations.[10] Therefore, a factor (or factors) other than differing nutrient

[10] Concentrations of total ammonia, TKN, and total phosphorus were comparable between the upper and lower stations of SDLT1 during each of the spring, summer, and fall sampling events (Appendix Table C.34). Because total phosphorus concentrations were similar between the SDLT1 upper and lower stations, the differences in chlorophyll a between stations did not appear to be related to phosphorus limitation at the lower station.

Table 4.1: Water Chemistry at Sheardown Lake Tributary 1 (SDLT1) Monitoring Stations, Mary River Project CREMP, Fall 2018

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Bench-mark[b]	Reference Creek Average (n = 4) Fall 2018	Sheardown Lake Tributary 1 D1-05 (Upper) 26-Aug-2018	Sheardown Lake Tributary 1 D1-00 (Lower) 26-Aug-2018
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	97	190	441
	pH (lab)	pH	6.5 - 9.0	-	7.88	7.72	8.05
	Hardness (as $CaCO_3$)	mg/L	-	-	47	99	205
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	53	95	240
	Turbidity	NTU	-	-	2.44	0.28	1.01
	Alkalinity (as $CaCO_3$)	mg/L	-	-	43	84	92
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.021	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	1.280
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	0.20
	Dissolved Organic Carbon	mg/L	-	-	1.5	3.4	4.1
	Total Organic Carbon	mg/L	-	-	2.0	3.8	4.3
	Total Phosphorus	mg/L	0.030^{α}	-	0.0035	<0.0030	<0.0030
	Phenols	mg/L	0.004^{α}	-	<0.0010	0.0011	0.0011
Anions	Bromide (Br)	mg/L	-	-	0.1	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.6	0.6	8.8
	Sulphate (SO_4)	mg/L	218^{β}	218	2.7	7.8	110.0
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.054	0.012	0.026
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0059	0.0093	0.0174
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	<0.000010	0.000023
	Calcium (Ca)	mg/L	-	-	9.8	19.8	37.6
	Chromium (Cr)	mg/L	0.0089	0.00856	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	<0.00010	0.00057	0.00064
	Copper (Cu)	mg/L	0.002	0.0022	0.0009	***0.0036***	0.0021
	Iron (Fe)	mg/L	0.30	0.326	0.046	<0.030	0.072
	Lead (Pb)	mg/L	0.001	0.001	0.00008	0.00006	0.00005
	Magnesium (Mg)	mg/L	-	-	5.31	12.7	29.2
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00065	0.00019	0.10000
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00029	0.00091	0.00322
	Nickel (Ni)	mg/L	0.025	0.025	0.0005	0.0019	0.0044
	Potassium (K)	mg/L	-	-	0.65	1.49	2.81
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.83	1.43	1.38
	Sodium (Na)	mg/L	-	-	1.24	0.39	4.08
	Strontium (Sr)	mg/L	-	-	0.0094	0.0089	0.0240
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010
	Uranium (U)	mg/L	0.015	-	0.00186	0.00337	0.00488
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	0.0075

[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.

Figure 4.2: Temporal Comparison of Water Chemistry at Sheardown Lake Tributaries (SDLT) for Mine Baseline (2005 to 2013), Construction (2014) and Operational (2015 to 2018) Periods during Fall

Notes: Values represent mean ± SD. Creek reference includes the 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 the Sheardown Lake Tributaries.

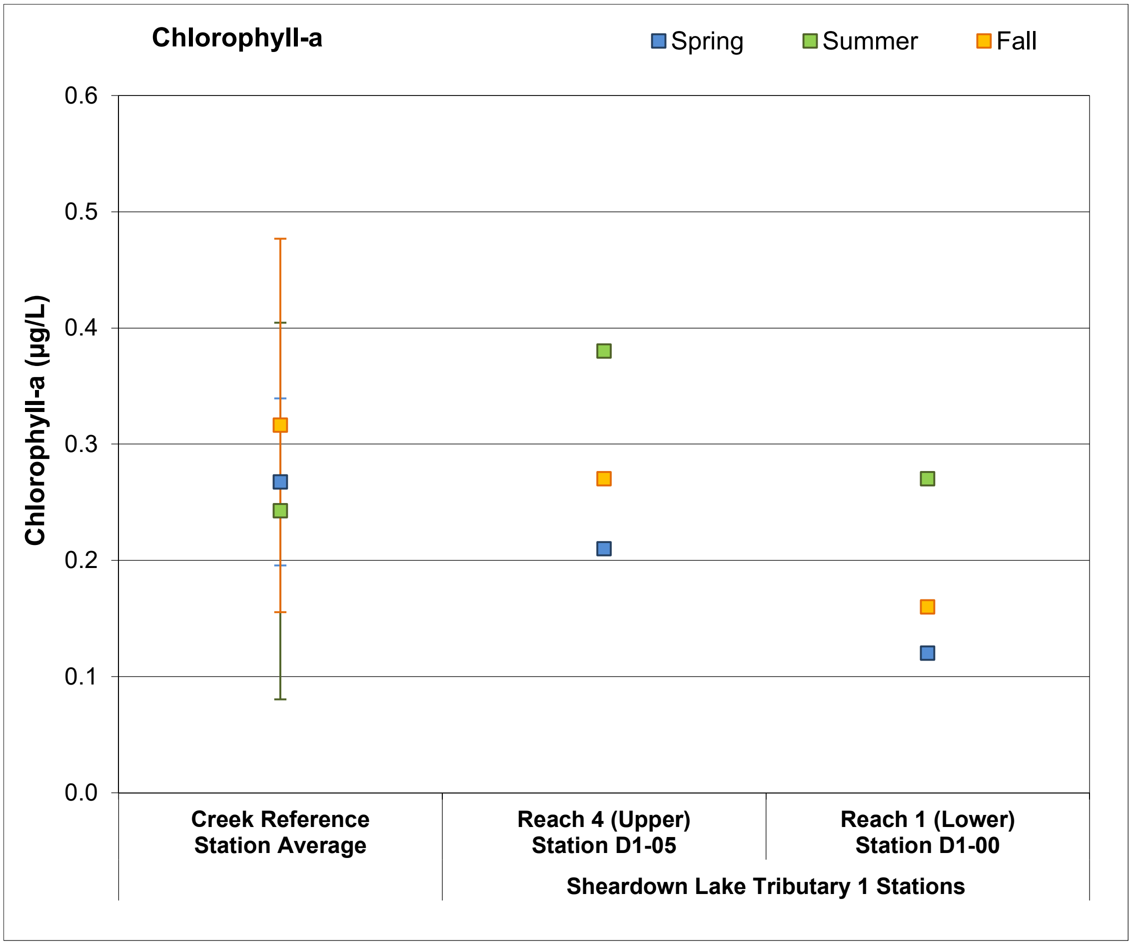


Figure 4.3: Chlorophyll-a Concentrations at Sheardown Lake Tributary 1 Phytoplankton Monitoring Stations, Mary River Project CREMP, 2018

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

concentrations appeared to account for the occurrence of lower chlorophyll-a concentrations at the lower versus upper monitoring station at SDLT1. Chlorophyll-a concentrations at SDLT1 were within the range of variability observed among reference creeks during spring, summer, and fall sampling events at the upper station (D1-05), and during the summer sampling event at the lower station (D1-00), but not during the spring and fall sampling events at the lower station (Figure 4.3). The latter suggested that a mine-related influence on phytoplankton abundance may have occurred seasonally at lower SDLT1. For all sampling events in 2018, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L at both of the SDLT1 monitoring stations (Figure 4.3). Similar to the reference creeks and Camp Lake tributaries, chlorophyll-a concentrations at SDLT1 were suggestive of oligotrophic, low productivity conditions based on Dodds et al (1998) trophic status classification for stream environments (i.e., chlorophyll-a concentration <10 µg/L). Relatively low chlorophyll-a concentrations at SDLT1 stations in 2018 were also consistent with an oligotrophic categorization using CWQG classifications based on aqueous phosphorus concentrations (i.e., concentrations below 10 µg/L; Table 4.1; Appendix Table C.34).

Temporal comparisons indicated that chlorophyll-a concentrations at SDLT1 stations in fall 2018 were similar to those during the baseline period (Figure 4.4). In addition, no consistent directional changes in chlorophyll-a concentrations were shown at the SDLT1 stations during fall sampling events over the mine baseline (2005 to 2013), construction (2014), and operational (2015 to 2018) periods (Figure 4.4). These data suggested no adverse mine-related influences to phytoplankton productivity at SDLT1 over the initial four years of mine operation.

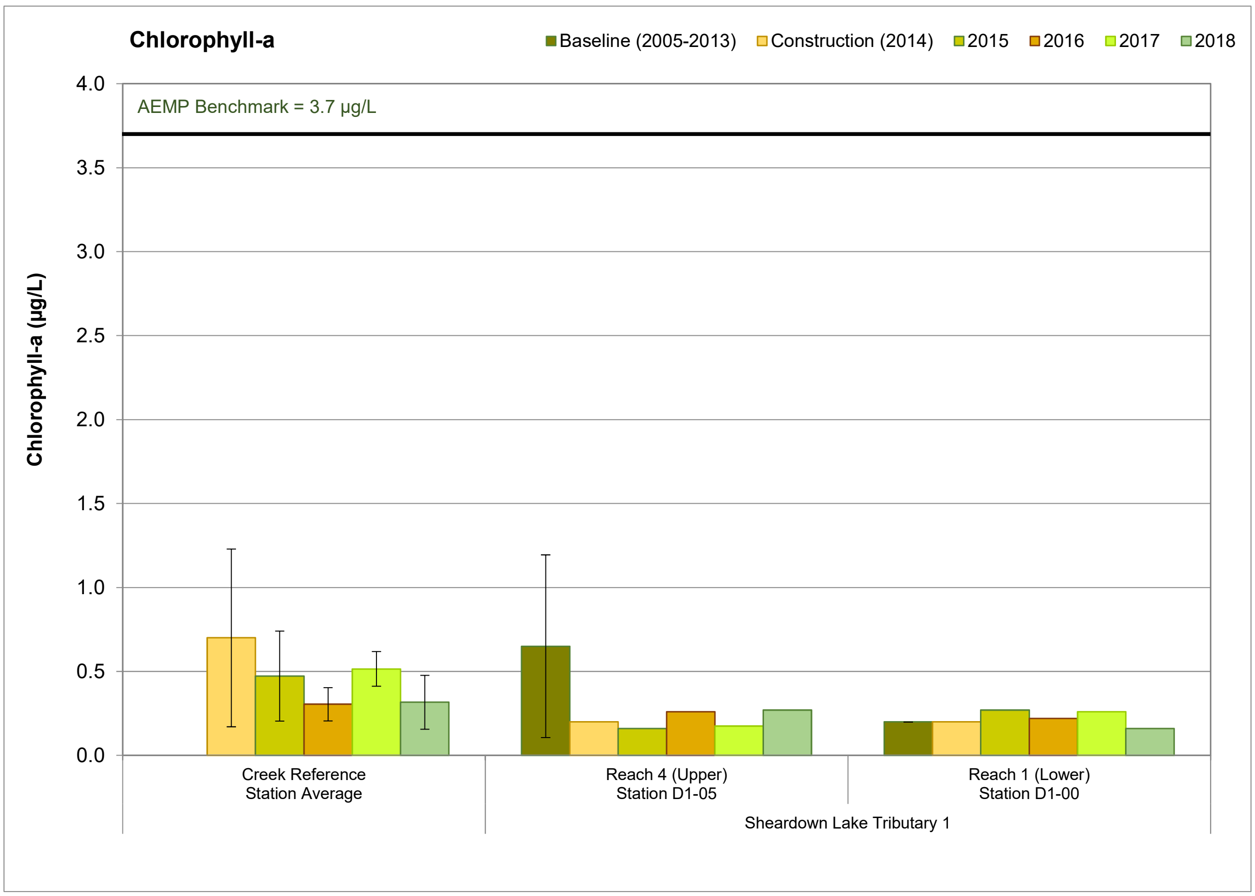


Figure 4.4: Temporal Comparison of Chlorophyll-a Concentrations at Sheardown Lake Tributary 1 for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015 to 2018) Periods in the Fall, Mary River Project CREMP

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

4.1.3 Benthic Invertebrate Community

4.1.3.1 Sheardown Lake Tributary 1 (SDLT1)

The benthic invertebrate community at the lower reach of SDLT1, near the outlet to Sheardown Lake NW, exhibited significantly lower richness and Simpson's Evenness, and significant differences in assemblage composition (as indicated by Bray-Curtis Index) compared to Unnnamed Reference Creek in 2018 (Figure 4.5; Appendix Table F.30). The key differences in relative abundance of dominant taxonomic groups included ecologically significant greater proportions of Oligochaeta (aquatic worms) and Chironomidae (non-biting midges), and conversely, significantly lower proportion of Nemata (roundworms), Ephemeroptera (mayflies), and Simuliidae (blackflies), at SDLT1 compared to Unnamed Reference Creek (Figure 4.5; Appendix Table F.30). A higher relative abundance of metal-sensitive chironomids at SDLT1 suggested that metal concentrations were not biologically available and/or were not a large contributor to differences in community composition compared to Unnamed Reference Creek, which was consistent with concentrations of all metals but copper below WQG at SDLT1 in 2018 (see Appendix Table C.34).

A significantly higher relative abundance of FFG shredders (Appendix Table F.30), which rely upon plants as an important food source, was consistent with greater density of attached bryophytes (mosses) at SDLT1 compared to the reference creek (Appendix Table F.24). In turn, this suggested that differences in in-stream vegetation likely contributed to differing benthic invertebrate community composition between SDLT1 and Unnamed Reference Creek. A significantly lower relative abundance of FFG filterers was also indicated at SDLT1 compared to the reference creek in 2018. This suggested that benthic invertebrate food sources differed between these watercourses and/or factors contributing to lower filter feeding efficiency were potentially associated with the mine operation. Notably, no significant differences in relative abundance of predominant HPG (i.e., clingers and sprawlers) were indicated between SDLT1 and the reference creek (Figure 4.5; Appendix Table F.30), suggesting that physical habitat alteration from factors such as sedimentation had not substantially affected benthic invertebrate community composition at SDLT1 relative to reference conditions. Overall, the differences in the benthic invertebrate community between SDLT1 and the reference creek in 2018 may have reflected natural differences in the types/amount of in-stream vegetation between watercourses, and mine-related influences on invertebrate filter-feeding efficiency at SDLT1, but did not appear to be related to metal concentrations.

Temporal comparison of the lower SDLT1 benthic invertebrate community data did not indicate any consistent ecologically significant differences in density, richness, or Simpson's Evenness for individual years of mine operation (2015 to 2018) compared to baseline studies conducted in 2008

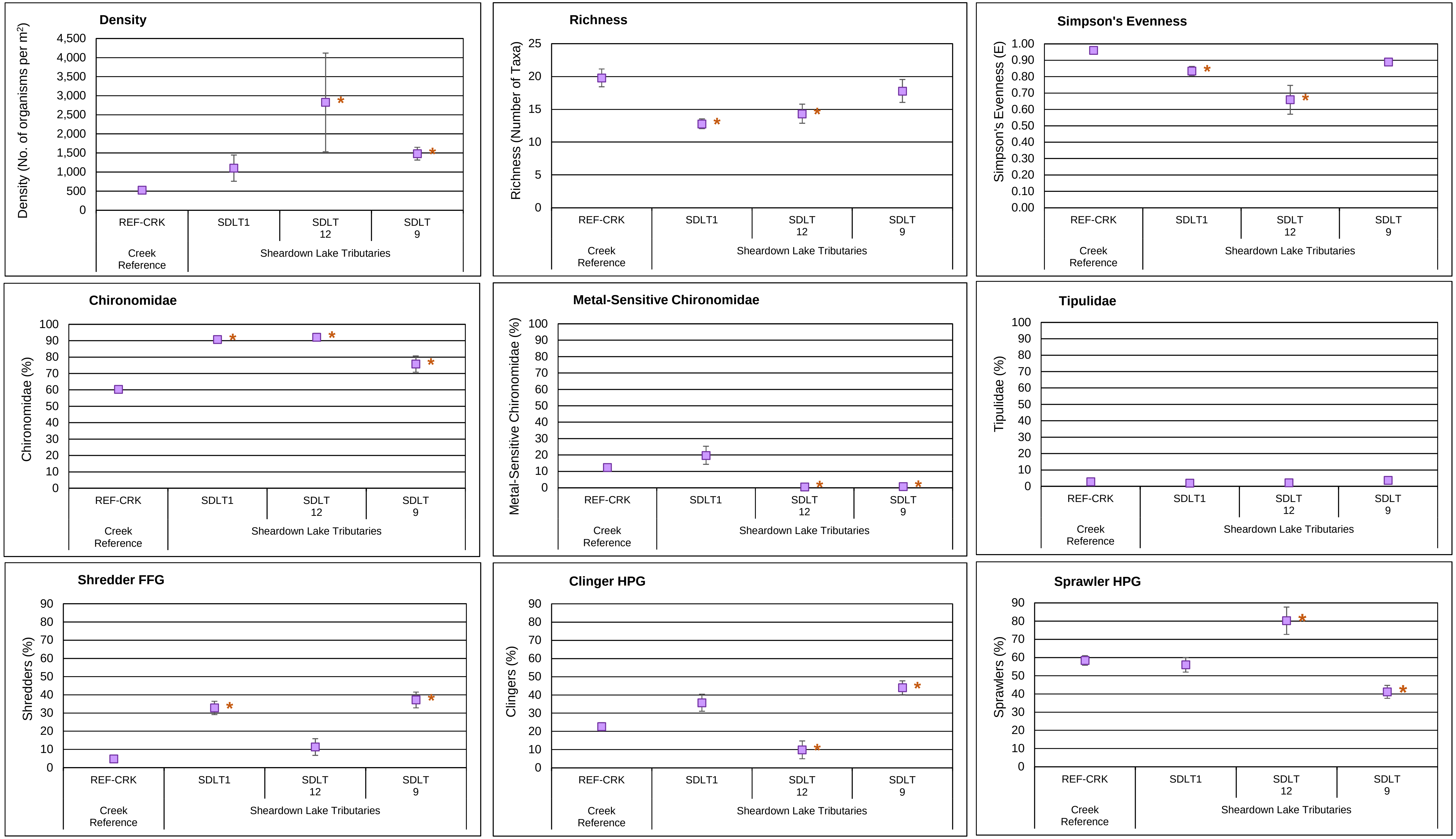


Figure 4.5: Comparison of Benthic Invertebrate Community Metrics between Sheardown Lake Tributary and Unnamed Reference Creek Study Areas (mean ± SE), Mary River Project CREMP, August 2018

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

and 2013 (Figure 4.6; Appendix Table F.31). Similarly, no ecologically significant differences in the relative abundance of any dominant taxonomic groups or FFG were consistently indicated among years of mine operation and baseline studies at SDLT1 (Appendix Tables F.31 and F.32). The absence of any consistent, ecologically significant differences in benthic invertebrate community density, richness, Simpson's Evenness, and composition at SDLT1 between the mine operational and baseline periods indicated no ecologically meaningful influences on benthic biota since the commencement of commercial mine operations in 2015.

4.1.3.2 Sheardown Lake Tributary 12 (SDLT12)

The benthic invertebrate community at SDLT12 exhibited significantly greater density and significantly lower richness and Simpson's Evenness compared to Unnnamed Reference Creek in 2018 (Figure 4.5; Appendix Table F.30). These differences reflected the occurrence of high densities of *Diplocladius* midges at SDLT12, which are characteristic of small, cool, slow-flowing or still streams (compare Appendix Tables F.4 and F.27; Armitage et al. 1995; Namayandeh et al. 2016). The existence of significantly slower water velocity at SDLT12 compared to Unnamed Reference Creek (Appendix Table F.28) thus likely accounted for the differences in benthic invertebrate density and Simpson's Evenness indicated above. Marked differences in community composition indicated between SDLT12 and the reference creek based on significant differences in Bray-Curtis Index, reflected significantly lower relative abundance of roundworms, mayflies, and blackflies, and significantly higher relative abundance of Chironomidae (Figure 4.5; Appendix Table F.30) at SDLT12. In addition, ecologically significant lower relative abundance of FFG filterers and HPG clingers, as well as ecologically significant higher relative abundance of HPG sprawlers, occurred at SDLT12 compared to Unnamed Reference Creek in 2018 (Figure 4.5; Appendix Table F.30). Because mayflies, blackflies, filterers, and clingers generally occur in moderate to swiftly flowing watercourses, the differences in benthic invertebrate community assemblage between SDLT12 and the reference creek were consistent with markedly slower water velocity at SDLT12. Therefore, differing habitat features between SDLT12 and Unnamed Reference Creek appeared to account for the differences in benthic invertebrate community composition between watercourses.

Temporal comparison of the SDLT12 benthic invertebrate community data indicated no on-going unidirectional significant differences in density and Simpson's Evenness, but significantly lower richness on a routine basis, between years of mine operation and baseline (Figure 4.6; Appendix Table F.33). A consistent occurrence of significantly higher relative abundance of burrowing invertebrates, including Oligochaeta and crane fly taxonomic groups, and the collector-gatherer FFG from 2015 to 2018 compared to the 2007 baseline data suggested changes in habitat conditions with the commencement of mine operations. Although such temporal changes

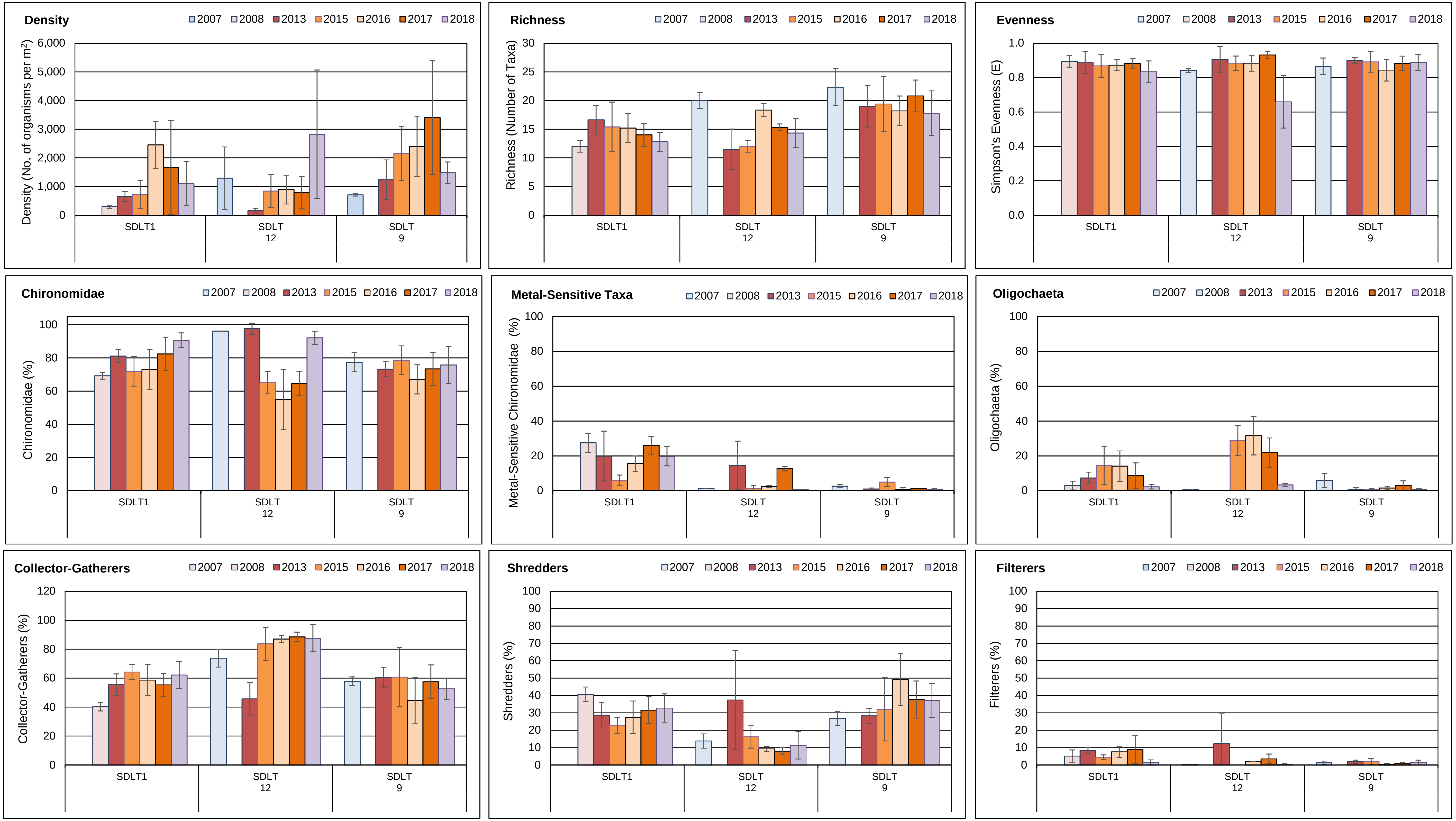


Figure 4.6: Comparison of Benthic Invertebrate Community Metrics (mean ± SD) at Sheardown Lake Tributaries 1, 12, and 9 among Mine Operational (2015 to 2018) and Baseline (2007, 2008, 2011, 2013) Studies for the Mary River Project CREMP

potentially reflected slight differences in sampling location between the mine operational and baseline periods, field observations from the 2016 and 2017 studies documented the occurrence of silt deposits on in-stream substrate of SDLT12 suggesting sedimentation within this watercourse. Therefore, a mine-related reduction in flow and/or increased particle loadings (e.g., through dust and/or erosional deposition) may have accounted for temporal changes in the benthic invertebrate community between the mine operational and baseline periods at SDLT12 that included a shift to higher abundance of deposit feeding, burrowing benthic invertebrates. Notably, the relative abundance of metal-sensitive chironomids did not differ significantly among years of mine operation and baseline at SDLT12, suggesting that metals were largely biologically unavailable and/or did not account for the differences in benthic invertebrate community endpoints shown between the mine operational and baseline studies.

4.1.3.3 Sheardown Lake Tributary 9 (SDLT9)

The benthic invertebrate community of Sheardown Lake Tributary 9 (SDLT9) exhibited significantly greater density but no significant differences in richness or Simpson's Evenness compared to Unnnamed Reference Creek in 2018 (Figure 4.5; Appendix Table F.30). Benthic invertebrate community composition differences were indicated between SDLT9 and Unnamed Reference Creek based on significant differences in Bray-Curtis Index. The key differences in dominant taxonomic groups included significantly lower relative abundance of mayflies and blackflies, and significantly higher relative abundance of Hydracarina (aquatic mites) and Chironomidae, at SDLT9 compared to the reference creek (Figure 4.5; Appendix Table F.30). Notably, the relative abundance of metal-sensitive chironomids was significantly lower at SDLT9 than at the reference creek, the magnitude of difference of which was ecologically meaningful (i.e., outside the CES_{BIC} of ±2 SD_{REF}; Figure 4.5; Appendix Table F.30). This suggested that differences in community composition between watercourses were possibly related to differing metal concentrations. However, differing food resources could also have accounted for the differing benthic invertebrate community composition between watercourses as indicated by significant differences in FFG composition between SDLT9 and the reference creek. For instance, the relative abundance of FFG shredders was significantly higher at SDLT9 compared to the reference creek, and was consistent with field observations of greater amounts of rooted in-stream vegetation at SDLT9 compared to the reference creek (Appendix Tables F.24 and F.30) given that vegetation is an important for source for shredders. In addition, because invertebrates within the clinger HPG are often associated with in-stream vegetation, significantly greater relative abundance of clinging aquatic mites at SDLT9 compared to Unnamed Reference Creek likely reflected greater amounts of in-stream vegetation at SDLT9 (Appendix Table F.30). In turn, this suggested that differing amounts and/or types of in-stream vegetation accounted for the differences in benthic invertebrate community composition between SDLT9 and the reference creek.

Temporal comparisons indicated no consistent ecologically significant differences in benthic invertebrate density, richness, Simpson's Evenness, or any dominant taxonomic groups and FFG at SDLT9 between data collected from the 2015 to 2018 mine operational years and baseline period data collected in 2007 and 2013 (Figure 4.6; Appendix Tables F.34 and F.35). Overall, this suggested that the differences in benthic invertebrate community composition between SDLT9 and Unnamed Reference Creek in 2018 likely reflected a natural difference in the amount of in-stream vegetation between watercourses and the associated influences of this vegetation on benthic invertebrate community composition.

4.1.4 Integrated Summary

At Sheardown Lake Tributary 1 (SDLT1), aqueous concentrations of several parameters were elevated compared to average concentrations observed at the reference creek stations in 2018. Of those parameters that were elevated compared to reference conditions, concentrations of manganese, nickel, nitrate, sodium, strontium, sulphate, TDS, and uranium were also elevated at SDLT1 in 2018 compared to the baseline period, suggesting a mine-related influence on water quality of SDLT1. However, with the exception of copper, no parameters were present at concentrations above WQG or AEMP benchmarks in 2018. Chlorophyll-a concentrations outside of the range of variability observed among the reference creeks during spring and fall sampling events only at the lower SDLT1 station, suggesting that a mine-related influence on phytoplankton abundance may occur seasonally at SDLT1. However, chlorophyll-a concentrations were similar between 2018 and the baseline period indicating no clear change to the trophic status of SDLT1 since commercial mine operation commenced. Significantly lower benthic invertebrate richness and Simpson's Evenness, as well as significant differences in community structure, were indicated at SDLT1 in 2018 compared to Unnamed Reference Creek. However, the occurrence of significantly greater relative abundance of shredders, as well as no significant difference in the relative abundance of metal-sensitive chironomids, at SDLT1 compared to the reference creek suggested that differing benthic invertebrate communities likely reflected differences in the types and/or amounts of in-stream vegetation between watercourses rather than influences associated with differing metal concentrations. In addition, studies conducted during years of mine operation from 2015 to 2018 showed no consistent ecologically significant differences in any primary benthic invertebrate community metrics, dominant taxonomic groups, or FFG compared to studies conducted during the mine baseline. In turn, this suggested that metals were not highly bioavailable in water of SDLT1, and that differences in benthic invertebrate community composition between SDLT1 and the reference creek likely reflected natural differences in the amount and/or types of in-stream vegetation between watercourses. Overall, similar to the findings of the three previous CREMP studies, no adverse mine-related effects to biota of SDLT1 were indicated in 2018 based on the chlorophyll-a and benthic invertebrate community data analyses.

At Sheardown Lake Tributary 12 (SDLT12), significantly higher benthic invertebrate density but significantly lower relative abundance of mayflies, blackflies, filterers, and clingers compared to the reference creek were consistent with a difference in habitat between watercourses that most notably included slower water velocities at SDLT12. However, temporal changes in the benthic invertebrate community of SDLT12 that included significantly lower richness and significantly higher relative abundance of collector-gatherers and burrowers following commencement of mine operations potentially indicated a mine-related reduction in flow and/or increased particle loadings (e.g., through dust and/or erosional deposition) over time at SDLT12. Therefore, some mine-related effects to biota of SDLT12 may have occurred since commercial mine operations commenced in 2015.

At Sheardown Lake Tributary 9 (SLDT9), significantly higher benthic invertebrate density and significant differences in community structure were indicated in 2018 compared to the reference creek. However, a significantly greater relative abundance of shredders and clingers at SDLT9 compared to the reference creek suggested that naturally differing amounts and/or types of in-stream vegetation accounted for the differing benthic invertebrate community structure between watercourses. Sampling conducted at SDLT9 during years of mine operation from 2015 to 2018 showed no consistent ecologically significant differences in benthic invertebrate density, richness, Simpson's Evenness, or relative abundance of dominant taxonomic groups and FFG compared to data collected from the mine baseline period. Overall, no adverse mine-related effects to biota were indicated at SDLT9 following commencement of commercial mine operation in 2015.

4.2 Sheardown Lake Northwest (DLO-1)

4.2.1 Hydraulic Retention Time

A hydraulic retention time of 511 ± 213 days was estimated for Sheardown Lake NW by Minnow (2018) using mean annual watershed runoff extrapolated from Baffinland flow monitoring stations installed in small watershed watercourses (i.e., ≤15 km^2) located on the mine property and a lake volume of 8.18 million cubic metres.

4.2.2 Water Quality

Water quality profiles of *in situ* water temperature, dissolved oxygen, pH, and specific conductance conducted at Sheardown Lake NW in 2018 showed no substantial station-to-station differences during any of the winter, summer, or fall sampling events (Appendix Figures C.11 to C.14). No thermal stratification was indicated at Sheardown Lake NW during the winter, summer, or fall sampling events in 2018 (Figure 4.7). The average water temperature at the bottom of the water column at Sheardown Lake NW littoral and profundal stations was significantly warmer than at Reference Lake 3 during the August 2018 biological sampling (Figure 4.8). However, the incremental difference in average bottom water temperature between lakes at each respective depth was small (i.e., ≤0.6˚C) and thus was unlikely to be ecologically meaningful. Dissolved oxygen profiles at Sheardown Lake NW showed a slight oxycline at depths greater than approximately 14 m during the winter, but no appreciable change in dissolved oxygen saturation from surface to bottom in the summer and fall of 2018 (Figure 4.7; Appendix Figure C.12). Similar dissolved oxygen profiles were observed between Sheardown Lake NW and Reference Lake 3 at like depths during the summer and fall sampling events (Figure 4.7). On average, dissolved oxygen concentrations near the bottom of the water column were slightly greater at Sheardown Lake NW littoral and profundal stations than at like stations in Reference Lake 3 during the August 2018 biological sampling, the difference of which was significant only for the littoral stations (Figure 4.8). Notably, dissolved oxygen concentrations were well above the WQG of 9.5 mg/L at Sheardown Lake NW during the fall sampling events in 2018 (Figure 4.8; Appendix Table C.40).

In situ profiles of pH and specific conductance showed no substantial step changes from the surface to bottom of the Sheardown Lake NW water column during any of the three sampling seasons in 2018, indicating no chemical stratification (Figure 4.7). Mean pH at the bottom of the water column at littoral and profundal stations of Sheardown Lake NW was significantly higher than at Reference Lake 3 during fall sampling in 2018 (Figure 4.8; Appendix Table C.37). However, pH values were consistently within WQG limits of 6.5 to 9.0 through the entire water column during all 2018 sampling events conducted at Sheardown Lake NW (Figures 4.7 and 4.8; Appendix Tables C.33 to C.36). Specific conductance was significantly higher at Sheardown Lake NW compared to Reference Lake 3 during fall sampling (Figure 4.8; Appendix Table C.42). However, specific conductance at Sheardown Lake NW was only slightly higher than that of reference creek and river stations in fall 2018 (i.e., range from 67 to 107 µS/cm), 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 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 August 2018 biological sampling (Appendix Table C.42; Appendix Figure C.7). Secchi depth readings showed relatively low variability among stations at Sheardown Lake NW, suggesting no spatial differences in water clarity across the lake (Appendix Table C.40).

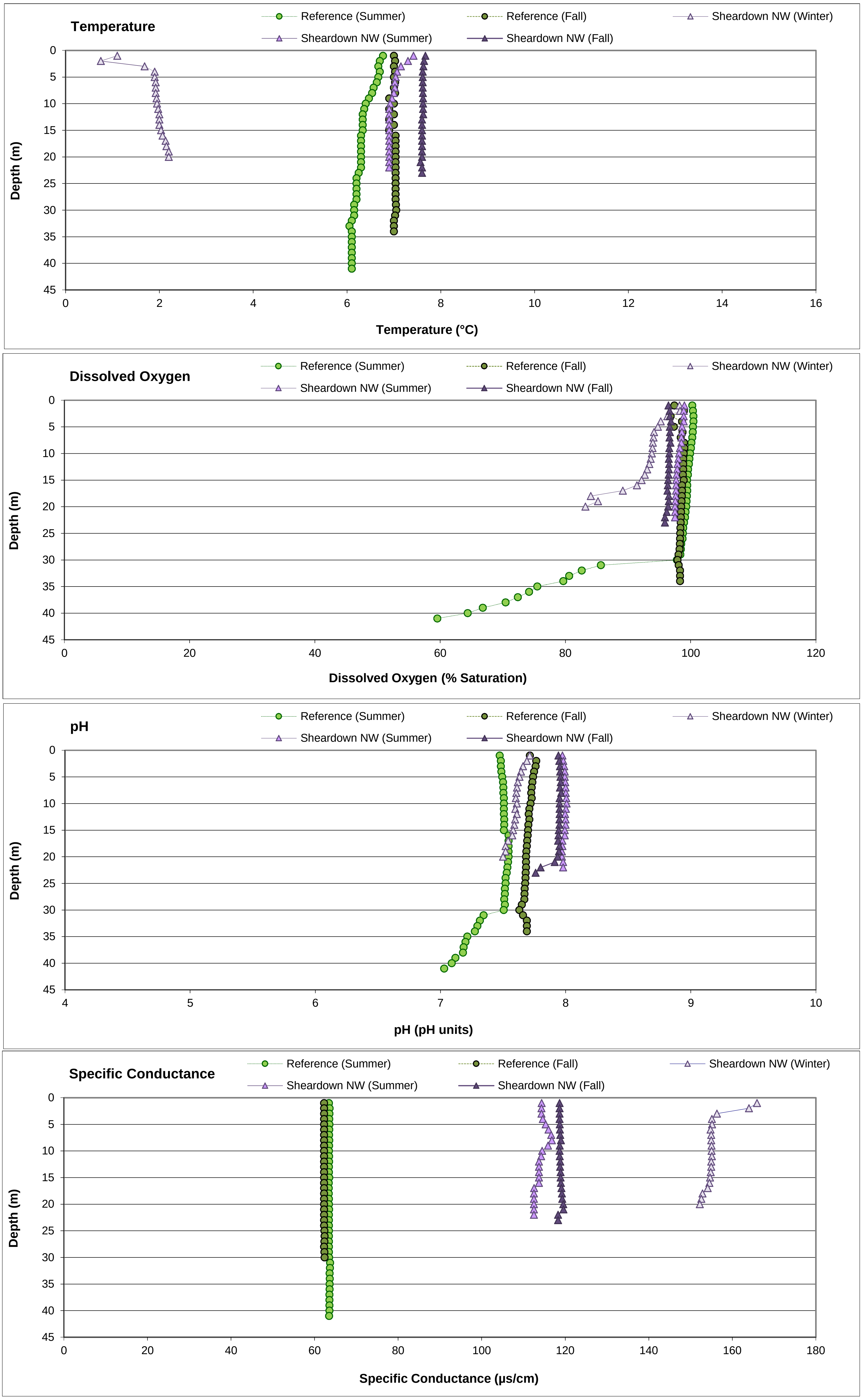


Figure 4.7: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake NW (DLO-01) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2018

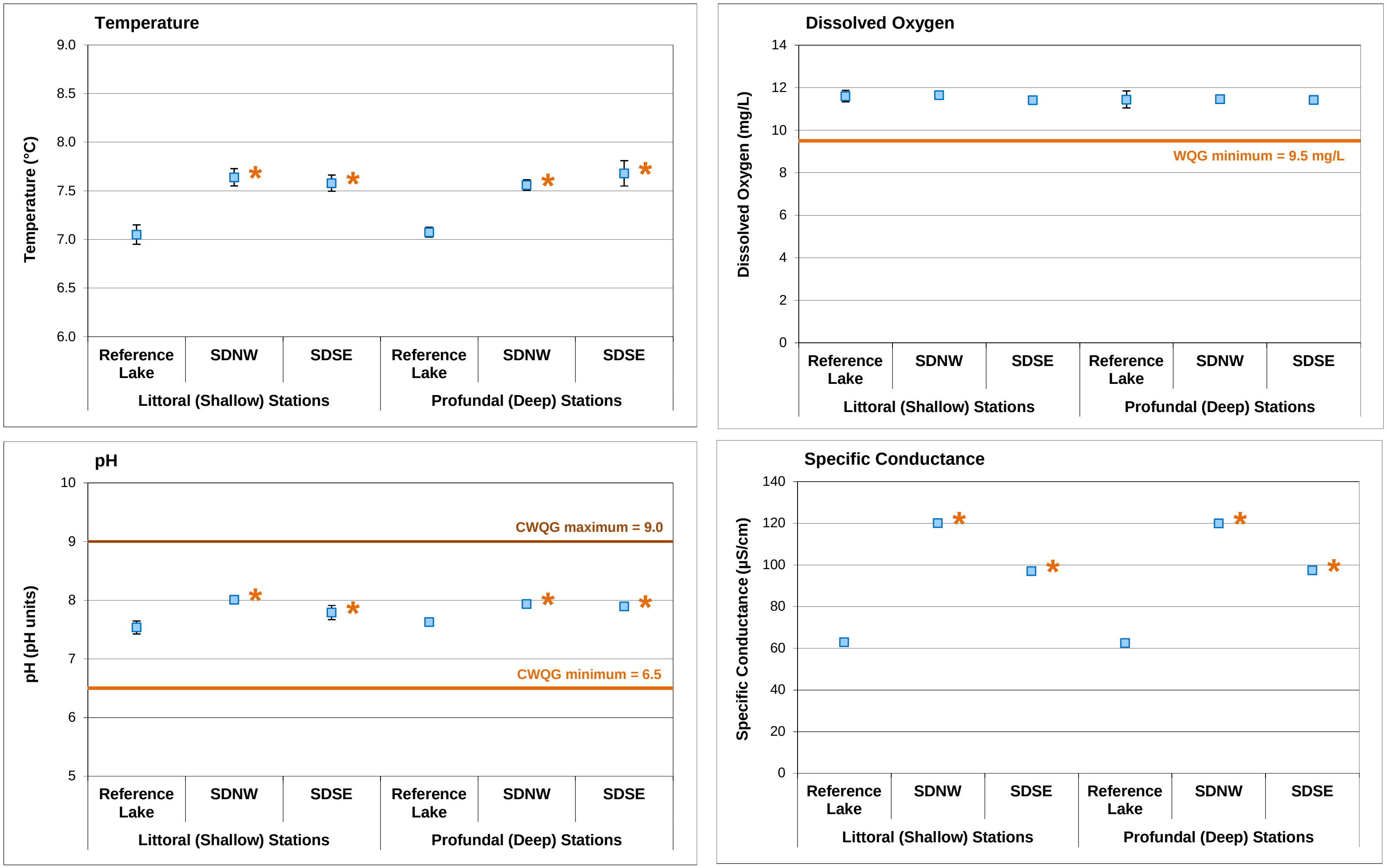


Figure 4.8: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Basins (SDNW and SDSE) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2018

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

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 2018 (Table 4.2; Appendix Table C.43), suggesting that the lake waters were continually well mixed both laterally and vertically. Turbidity, chloride concentrations, and total concentrations of aluminum, manganese, molybdenum, and uranium were elevated (i.e., ≥3-fold higher) at Sheardown Lake NW compared to Reference Lake 3 during the summer and/or fall sampling events (Table 4.2; Appendix Tables C.43 and C.44). Similar to previous studies, total aluminum and manganese concentrations showed a moderately strong positive correlation with turbidity at Sheardown Lake NW in 2018 (r_s = 0.56 and 0.57, respectively; Appendix Table C.47). This suggested that elevated total aluminum and manganese concentrations at Sheardown Lake NW may have reflected influences associated with surface runoff and/or backflow received from Mary River that contained naturally high concentrations of aluminum and manganese bearing particulate minerals. This was supported by an evaluation of dissolved metal concentrations, which indicated similar dissolved aluminum concentrations between Sheardown Lake NW and Reference Lake 3 (Appendix Table C.46), and the lack of a strong correlation between dissolved concentrations of these metals and turbidity (Appendix Table C.47). In addition, the ratio of dissolved to total concentrations of aluminum and manganese indicated that the majority (i.e., approximately 80%) of each of these metals was in the particulate fraction at Sheardown Lake NW. Total and dissolved concentrations of molybdenum and uranium were not positively correlated with turbidity (Appendix Table C.47), suggesting that these metals were not associated with suspended particulate matter. Despite elevation of total aluminum, manganese, molybdenum, and uranium concentrations at Sheardown Lake NW compared to Reference Lake 3, concentrations of each of these metals were well below applicable WQG and AEMP benchmarks at Sheardown Lake NW during all sampling events in 2018 (Table 4.2; Appendix Table C.43).

Temporal comparisons of the Sheardown Lake NW water chemistry data suggested that 2018 seasonal average total and dissolved concentrations of most parameters were within their respective range of baseline concentrations (Figure 4.9; Appendix Figure C.19). Key exceptions included concentrations of chloride, sulphate, total and dissolved manganese, and dissolved molybdenum, which showed slight elevation (i.e., 3- to 5-fold higher) in 2018 compared to the baseline data in at least one seasonal period (Appendix Tables C.44 and C.46). Conductivity, hardness, and concentrations of chloride, molybdenum, sodium, strontium, and sulphate showed successively higher concentrations over years of mine-construction (2014) through mine operation (2015 to 2018) during fall sampling events at Sheardown Lake NW (Figure 4.9; Appendix Figure C.19). The magnitude of these year-to-year changes were relatively minor and unlikely to be ecologically meaningful given parameter concentrations remained well below WQG.

Table 4.2: Water Chemistry at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3) Monitoring Stations[a], Mary River Project CREMP, August 2018

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3)	Sheardown Lake NW Station					
						DD-HAB9 STN1	DL0-01-5	DL0-01-1	DL0-01-4	DL0-01-2	DL0-01-7
					Fall 2018	21-Aug-2018	22-Aug-2018	21-Aug-2018	22-Aug-2018	22-Aug-2018	22-Aug-2018
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	75	139	137	133	140	140	141
	pH (lab)	pH	6.5 - 9.0	-	7.65	7.98	7.99	7.96	7.99	8.00	8.02
	Hardness (as $CaCO_3$)	mg/L	-	-	35	69	68	70	67	69	68
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	2.2	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	46	63	83	57	73	73	80
	Turbidity	NTU	-	-	0.51	0.96	0.92	0.96	0.92	0.96	0.95
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	52	51	52	52	49	52
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.044	0.039	0.023	0.033	0.02	0.021	0.02
	Nitrate	mg/L	13	13	<0.020	0.096	0.077	0.086	0.081	0.079	0.078
	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	-	-	0.16	<0.15	<0.15	0.155	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.9	2.3	2.5	2.2	2.6	2.5	2.5
	Total Organic Carbon	mg/L	-	-	3.8	2.3	2.6	2.6	2.7	2.7	2.6
	Total Phosphorus	mg/L	0.020^{α}	-	0.005	0.005	0.004	0.011	0.005	0.004	0.004
	Phenols	mg/L	0.004^{α}	-	0.001	0.0012	<0.0010	0.0022	0.0011	0.0011	<0.0010
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	3.5	3.5	3.5	3.5	3.5	3.5
	Sulphate (SO_4)	mg/L	218^{β}	218	3.7	9.5	9.3	9.3	9.3	9.5	9.4
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.004	0.022	0.018	0.040	0.018	0.014	0.021
	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	-	-	0.00644	0.00689	0.00694	0.00700	0.00709	0.00592	0.00692
	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	-	-	7.2	12.0	11.7	12.3	12.1	11.7	12.1
	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	0.000115	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.0008	0.0009	0.0012	0.0010	0.0009	0.0007	0.0009
	Iron (Fe)	mg/L	0.30	0.300	<0.030	0.0325	<0.030	0.0645	<0.030	<0.030	<0.030
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	0.000107	<0.000050	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	0.001	0.00105	0.0011	0.0011	0.0011	0.0011	<0.0010
	Magnesium (Mg)	mg/L	-	-	4.3	8.6	8.5	8.6	8.5	7.3	8.5
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00064	0.01115	0.01004	0.01345	0.00974	0.00905	0.01035
	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	0.00081	0.00079	0.00074	0.00082	0.00079	0.00081
	Nickel (Ni)	mg/L	0.025	0.025	0.0005	0.00077	0.00075	0.00082	0.00076	0.00064	0.00102
	Potassium (K)	mg/L	-	-	0.86	1.31	1.28	1.30	1.29	1.12	1.29
	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	0.44	0.43	0.45	0.44	0.41	0.44
	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	-	-	0.86	1.68	1.65	1.66	1.62	1.57	1.64
	Strontium (Sr)	mg/L	-	-	0.0081	0.0089	0.0091	0.0088	0.0088	0.0086	0.0088
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Uranium (U)	mg/L	0.015	-	0.00026	0.00095	0.00099	0.00101	0.00096	0.00095	0.00095
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030

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

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

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

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

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

Figure 4.9: Temporal Comparison of Water Chemistry at Sheardown Lake Northwest (DLO-01) and Sheardown Lake Southeast (DLO-02) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2018) Periods during Fall

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

4.2.3 Sediment Quality

Surficial sediment at Sheardown Lake NW varied from silt and sandy loam to loam at littoral areas, to primarily silt loam at profundal areas (Figure 4.10; Appendix Table D.11). Surficial sediment at littoral and profundal areas of Sheardown Lake NW had significantly less sand and significantly more silt than at Reference Lake 3 (Appendix Table D.12). In addition, the TOC content of profundal sediment at Sheardown Lake NW was significantly lower than at Reference Lake 3 (Figure 4.10; Appendix Table D.12). 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.10 and D.11). In Sheardown Lake NW, this material occasionally occurred as a thin, distinct layer that was likely composed principally of iron (oxy)hydroxide precipitate. Substrate of Sheardown Lake NW exhibited some blackening (or unusually dark colouration), but no noticeable sulphidic odour, at the time of the August 2018 sampling event (Appendix Tables D.10 and D.11), suggesting the occurrence of reducing conditions in the sediment similar to that observed at Reference Lake 3 (Appendix Tables D.1 and D.2).

Sediment metal concentrations at Sheardown Lake NW showed no consistent spatial differences from stations located nearest to key tributary inlets (e.g., SDLT1 and SDLT12) to those located near the lake outlet in 2018 (Appendix Table D.13). However, iron and phosphorus concentrations in sediment appeared to be highest at Sheardown Lake NW stations situated closest to the outlet of SDLT1 (Stations DD-HAB 9-STN2; Appendix Table D.13). This was consistent with the previous CREMP study, which indicated that SDLT1 was a source of iron loadings to the lake (Minnow 2018). Metal concentrations in littoral and profundal sediment of Sheardown Lake NW were very similar to averages observed for like sampling depths at Reference Lake 3 in 2018 (Table 4.3; Appendix Table D.14), suggesting no marked mine-related influences on sediment metal concentrations at Sheardown Lake NW. Although mean concentrations of iron and manganese were above SQG in sediment at littoral and/or profundal stations of Sheardown Lake NW, the mean concentration of these metals was also above SQG in sediment at Reference Lake 3 (Table 4.3). On average, nickel concentrations were above SQG in sediment at profundal stations, as were concentrations of phosphorus and chromium in sediment at individual littoral and profundal stations, respectively, at Sheardown Lake NW in 2018 (Table 4.3; Appendix Table D.13). However, phosphorus and chromium concentrations were also elevated above SQG in sediment at individual littoral and profundal stations, respectively, at

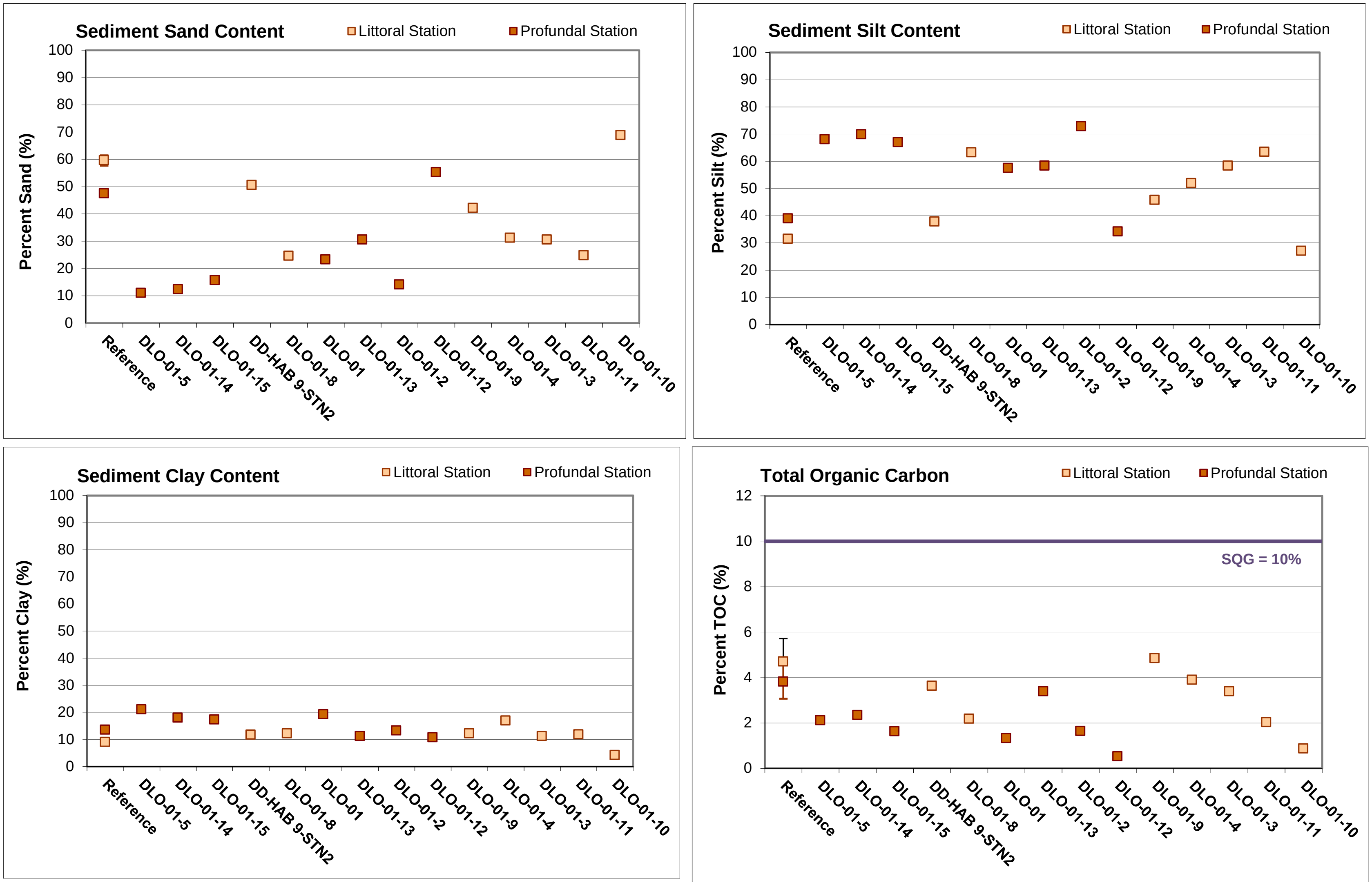


Figure 4.10: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake NW (DLO-01) Sediment Monitoring Stations and Reference Lake 3 (mean ± SE), Mary River Project CREMP, August 2018

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 2018

	Parameter	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b] (NW, SE)	Littoral			Profundal		
					Reference Lake (n = 5)	Sheardown Lake NW (n=4)	Sheardown Lake SE (n=3)	Reference Lake (n = 5)	Sheardown Lake NW (n=3)	Sheardown Lake SE (n=2)
					Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error
	Total Organic Carbon	%	10[α]	-	4.7 ± 1.0	3.0 ± 0.7	1.3 ± 0.2	3.8 ± 0.8	2.1 ± 0.52	1.2 ± 0.02
Metals	Aluminum (Al)	mg/kg	-	-	17,880 ± 1,993	6,640 ± 3,767	16,800 ± 866	24,420 ± 3,494	24,067 ± 1,934	16,650 ± 50
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	0.11 ± 0.01	<0.10 ± 9.813E-18	<0.10 ± 0	<0.10 ± 0.00	<0.10 ± 0
	Arsenic (As)	mg/kg	17	6.2, 5.9	5.25 ± 0.95	4.56 ± 3.76	4.57 ± 0.76	***6.07*** ± 0.78	***6.28*** ± 0.74	3.32 ± 0.30
	Barium (Ba)	mg/kg	-	-	133 ± 25	134 ± 118	95 ± 9	152 ± 22.8	128 ± 12	82 ± 6.8
	Beryllium (Be)	mg/kg	-	-	0.68 ± 0.08	0.33 ± 0.16	0.76 ± 0.044	0.87 ± 0.1218	1.19 ± 0.063	0.78 ± 0.015
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.22 ± 0.018	0.23 ± 0.028	<0.2 ± 0.0	0.31 ± 0.018	0.20 ± 0.0000
	Boron (B)	mg/kg	-	-	13.9 ± 1.59	9.9 ± 3.69	17.6 ± 0.62	15.6 ± 2.208	29.3 ± 2.70	17.5 ± 1.55
	Cadmium (Cd)	mg/kg	3.5	1.5, 1.5	0.195 ± 0.044	0.143 ± 0.104	0.108 ± 0.00551	0.197 ± 0.005	0.318 ± 0.023	0.088 ± 0.0015
	Calcium (Ca)	mg/kg	-	-	5,480 ± 804	1,891 ± 1,045	6,507 ± 1,402	5,584 ± 664	5,070 ± 345	5,805 ± 495
	Chromium (Cr)	mg/kg	90	97, 79	58.9 ± 7.7	26 ± 14	76 ± 7.4	77.3 ± 11.0	89 ± 6.6	66 ± 1.75
	Cobalt (Co)	mg/kg	-	-	11.7 ± 1.401	5.9 ± 3.50	13.5 ± 0.7	17.4 ± 2.4	19.1 ± 1.1	13.1 ± 0.400
	Copper (Cu)	mg/kg	110	58, 56	***73.9*** ± 11.02	18 ± 11	28 ± 2.0	***96.3*** ± 14.7	55 ± 4.7	27 ± 0.35
	Iron (Fe)	mg/kg	40,000[α]	52,200, 34,400	***46,700*** ± 9,489	26,440 ± 18,271	***40,533*** ± 2,664	***50,900*** ± 7,115	***53,567*** ± 4,807	***36,350*** ± 1,950
	Lead (Pb)	mg/kg	91.3	35	16.4 ± 2.1	7.8 ± 4.57	16.9 ± 1.20	19.5 ± 2.8	25.8 ± 1.6	15.8 ± 0.200
	Lithium (Li)	mg/kg	-	-	26.0 ± 2.7	11.1 ± 6.18	30.0 ± 0.96	36.1 ± 5.0	39.7 ± 1.7	35.0 ± 2.20
	Magnesium (Mg)	mg/kg	-	-	11,104 ± 1,352	4,625 ± 2,530	13,900 ± 1,375	15,394 ± 2,199	15,667 ± 1,065	12,950 ± 50.0
	Manganese (Mn)	mg/kg	1,100[α,β]	4,530, 657	640 ± 60	1,309 ± 1,188	***959*** ± 117	***1,279*** ± 115	3,258 ± 2,269	***713*** ± 270.5
	Mercury (Hg)	mg/kg	0.486	0.17	0.0433 ± 0.0111	0.0371 ± 0.0084	0.0218 ± 0.00139	0.0650 ± 0.0121	0.0415 ± 0.00942	0.0263 ± 0.0001
	Molybdenum (Mo)	mg/kg	-	-	3.84 ± 0.86	3.66 ± 2.92	1.43 ± 0.040	2.57 ± 0.27	4.33 ± 1.74	1.17 ± 0.300
	Nickel (Ni)	mg/kg	75[α,β]	77, 66	42.9 ± 5.9	29.7 ± 18.2	61.1 ± 6.17	53.8 ± 6.6	***78.4*** ± 5.0	51.2 ± 1.400
	Phosphorus (P)	mg/kg	2,000[α]	1,958, 1,278	***1,305*** ± 272	720 ± 506	1,135 ± 124	1,188 ± 118	1,213 ± 117	955 ± 38.0
	Potassium (K)	mg/kg	-	-	4,134 ± 469	1,638 ± 906	3,970 ± 197	5,660 ± 796	6,023 ± 540	4,055 ± 135
	Selenium (Se)	mg/kg	-	-	0.66 ± 0.14	0.29 ± 0.09	0.20 ± 0.00	0.81 ± 0.153	0.44 ± 0.061	0.20 ± 0.0000
	Silver (Ag)	mg/kg	-	-	0.15 ± 0.02	0.11 ± 0.013	0.11 ± 0.006	0.26 ± 0.042	0.20 ± 0.018	0.11 ± 0.0000
	Sodium (Na)	mg/kg	-	-	320 ± 43	108 ± 42	245 ± 12	433 ± 62	331 ± 20	249 ± 2.5
	Strontium (Sr)	mg/kg	-	-	12.2 ± 1.476	4.2 ± 2.19	10.5 ± 0.874	13.8 ± 1.62	12.9 ± 0.76	9.5 ± 0.090
	Sulphur (S)	mg/kg	-	-	1,780 ± 349.85711	1,025 ± 25	<1,000 ± 0	1,400 ± 130.4	<1,000 ± 0	<1,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.45 ± 0.063	0.21 ± 0.12	0.39 ± 0.031	0.754 ± 0.091	0.64 ± 0.07	0.37 ± 0.0005
	Tin (Sn)	mg/kg	-	-	2.00 ± 0	<2.0 ± 0	<2.0 ± 0	<2.1 ± 0.0	<2.0 ± 0.0	<2.0 ± 0
	Titanium (Ti)	mg/kg	-	-	1,155 ± 132	431 ± 231	1,247 ± 47	1,388 ± 163	1,483 ± 123.5	1,225 ± 15.0
	Uranium (U)	mg/kg	-	-	13.4 ± 2.32	3.47 ± 2.18	4.86 ± 0.026	24.5 ± 3.90	9.56 ± 0.85	5.34 ± 0.095
	Vanadium (V)	mg/kg	-	-	58.3 ± 6.90	20.1 ± 11.3	50.3 ± 2.75	72.7 ± 9.36	70.5 ± 6.18	48.2 ± 0.400
	Zinc (Zn)	mg/kg	315	135	81.4 ± 10.17	24.5 ± 14.3	55.2 ± 2.20	99.2 ± 14.16	83.0 ± 5.65	55.5 ± 0.050
	Zirconium (Zr)	mg/kg	-	-	4.1 ± 0.8	3.5 ± 1.50	14.1 ± 0.85	3.88 ± 0.483	9.13 ± 1.42	16.7 ± 0.00

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

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

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

BOLD Indicates parameter concentration above the AEMP Benchmark.

Reference Lake 3 (Table 4.3; Appendix Table D.13), indicating naturally elevated concentrations of these metals, in addition to iron and manganese, in sediment of local study area lakes. Average concentrations of iron and nickel were above Sheardown Lake NW AEMP benchmarks in profundal sediment, however concentrations of these metals were not unlike those observed in profundal sediment at Reference Lake 3 (Table 4.3; Appendix Tables D.13 and D.14).

Temporal comparisons indicated that metal concentrations in sediment at littoral and profundal stations of Sheardown Lake NW in 2018 were comparable to those observed during the mine baseline (2005 to 2013) period (Figure 4.11; Appendix Table D.14). On average, the 2018 metal concentrations in sediment at Sheardown Lake NW littoral stations were in the lower range, and at profundal stations were in the upper range, of those observed at respective station types from 2015 to 2017 (Figure 4.11). No consistent increase in average metal concentrations appeared to occur from 2015 to 2018 at the Sheardown Lake NW littoral stations, but at profundal stations, visual evaluation of plotted data suggested very slightly increasing concentrations of a number of metals occurring from the onset of commercial mine operations in 2015 (Figure 4.11). Nevertheless, based on evaluation of data current to 2018, only minor changes in sediment metal concentrations were indicated at Sheardown Lake NW littoral and profundal stations following the commencement of commercial mine operations in 2015.

4.2.4 Phytoplankton

Chlorophyll-a concentrations at Sheardown Lake NW showed no consistent spatial gradients with progression towards the lake outlet among the winter, summer, and fall sampling events in 2018 (Figure 4.12). Chlorophyll-a concentrations differed significantly among seasons at Sheardown Lake NW in 2018, with highest and lowest concentrations observed in summer and winter, respectively (Appendix Table E.6). The direction of seasonal differences in chlorophyll-a concentrations at Sheardown Lake NW contrasted with those at Reference Lake 3, where highest chlorophyll-a concentrations occurred during the fall sampling event (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 2018 (Appendix Tables E.7 and E.8), 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 an oligotrophic categorization for Sheardown Lake NW using CWQG classifications based on aqueous total phosphorus concentrations (i.e., concentrations below 10 µg/L; Table 4.2; Appendix Table C.43).

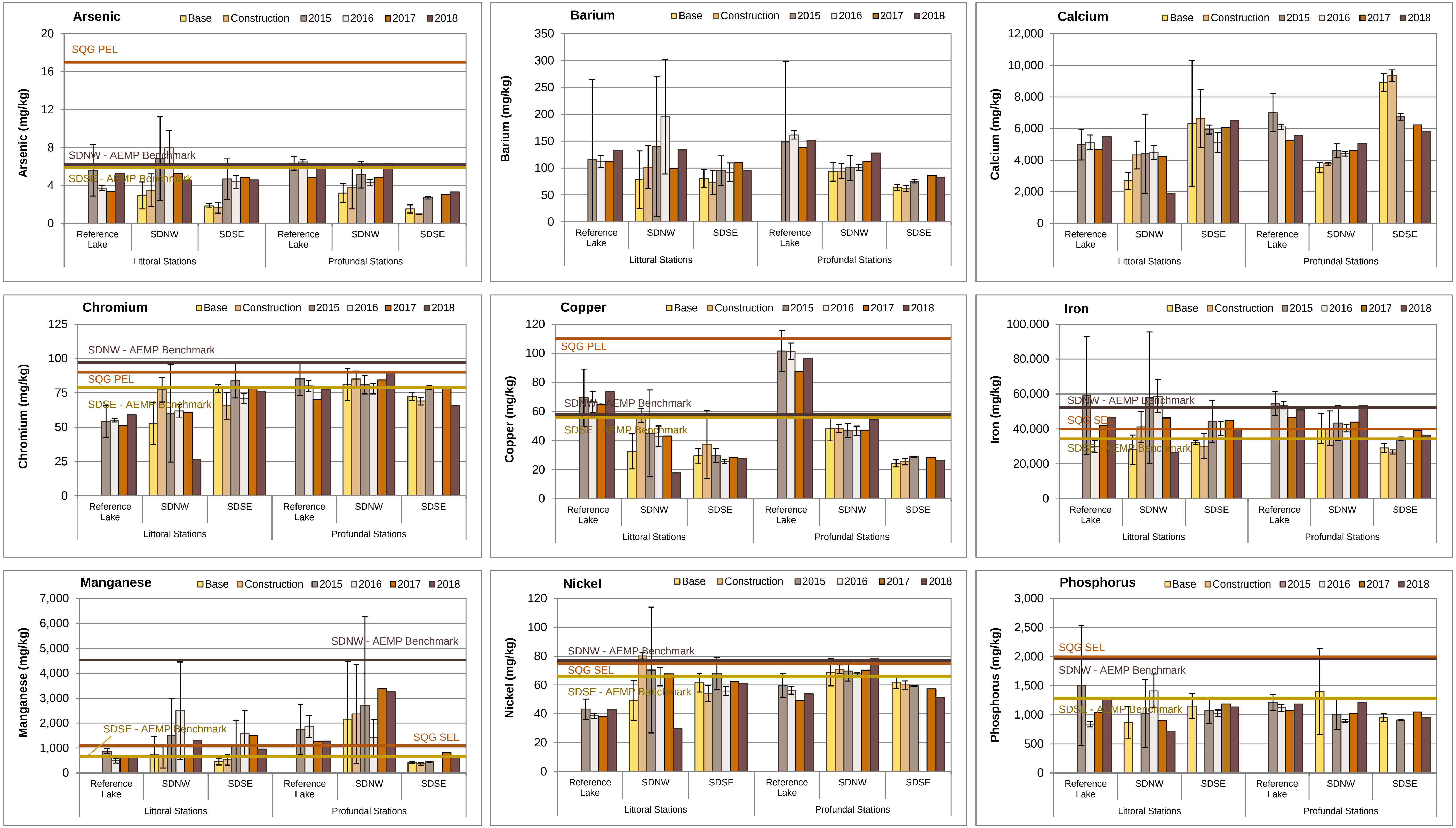


Figure 4.11: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Sheardown Lake NW (SDNW), Sheardown Lake SE (SDSE), and Reference Lake 3 for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2018) Periods

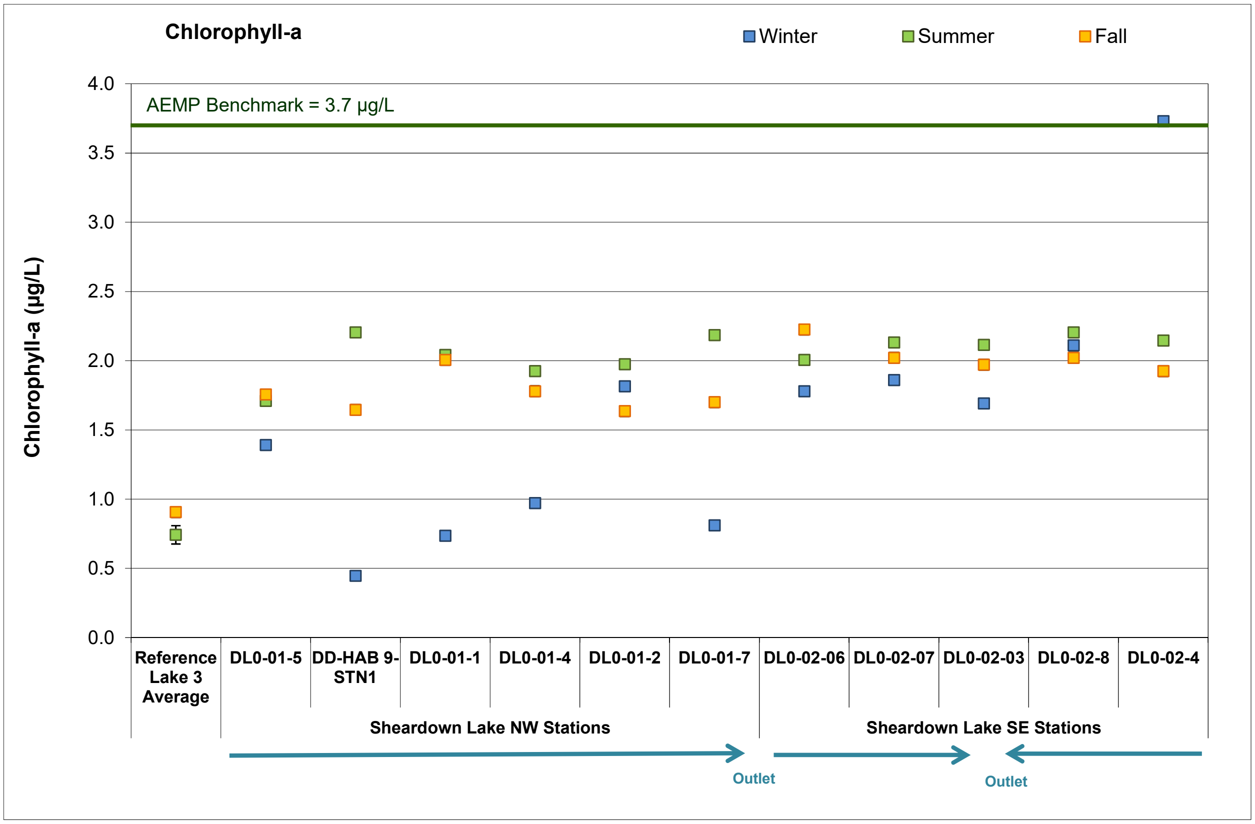


Figure 4.12: Chlorophyll-a Concentrations at Sheardown Lake NW (DLO-1) and Sheardown Lake SE (DLO-2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2018

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

Temporally, Sheardown Lake NW chlorophyll-a concentrations did not differ significantly between 2018 and years of mine construction (2014) or previous mine operation (2015 to 2016) in any consistent direction for the winter, summer, or fall seasons (Figure 4.13; Appendix Table E.11). This suggested 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 Sheardown Lake NW over the mine baseline period (2005 to 2013), precluding comparisons of Sheardown Lake NW chlorophyll-a data to the period prior to mine construction.

4.2.5 Benthic Invertebrate Community

Benthic invertebrate density and richness were significantly higher at littoral and profundal habitats of Sheardown Lake NW compared to like-habitat stations at Reference Lake 3. With the exception of richness at littoral habitat, these differences were at magnitudes outside of the CES_{BIC} of ±2 SD_{REF} indicating they were ecologically meaningful (Tables 4.4 and 4.5). In addition

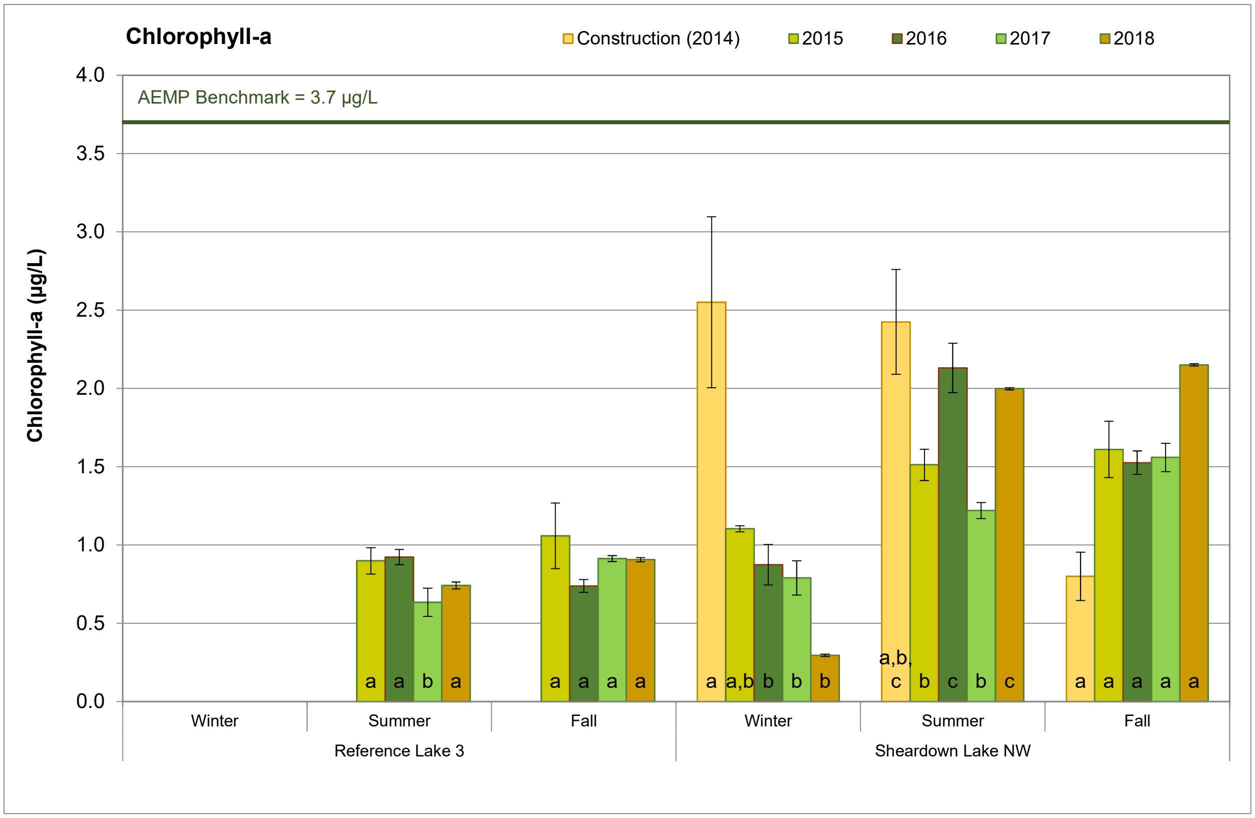


Figure 4.13: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Sheardown Lake NW and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2018) Periods (mean ± SE)

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

Table 4.4: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake NW (DLO-01) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	log	YES	< 0.001	ANOVA	20.7	Reference Lake 3	1,045	258	116	696	1,000	1,391
						Sheardown NW Littoral	6,390	3,751	1,677	3,087	5,530	11,861
Richness (Number of Taxa)	none	YES	0.007	ANOVA	1.8	Reference Lake 3	10.8	2.3	1.0	7.0	11.0	13.0
						Sheardown NW Littoral	15.0	1.2	0.5	14.0	15.0	17.0
Simpson's Evenness (E)	square root	NO	0.949	ANOVA	-0.5	Reference Lake 3	0.825	0.103	0.046	0.720	0.816	0.939
						Sheardown NW Littoral	0.769	0.163	0.073	0.558	0.827	0.914
Bray-Curtis Index	square root	YES	< 0.001	ANOVA	5.5	Reference Lake 3	0.313	0.092	0.041	0.178	0.358	0.394
						Sheardown NW Littoral	0.823	0.059	0.027	0.754	0.827	0.900
Nemata (%)	square root	NO	0.254	ANOVA	-0.7	Reference Lake 3	7.1	8.8	3.9	0.0	3.4	21.3
						Sheardown NW Littoral	1.3	1.0	0.5	0.3	1.2	2.8
Ostracoda (%)	fourth root	NO	0.381	ANOVA	-0.6	Reference Lake 3	23.9	18.3	8.2	3.4	20.6	53.3
						Sheardown NW Littoral	13.0	4.1	1.8	9.2	11.6	19.1
Chironomidae (%)	none	NO	0.133	ANOVA	0.8	Reference Lake 3	66.9	22.2	10.0	35.5	73.8	91.4
						Sheardown NW Littoral	83.7	2.9	1.3	78.8	84.8	85.7
Metal-Sensitive Chironomidae (%)	log	NO	0.135	ANOVA	-0.9	Reference Lake 3	36.5	19.6	8.8	17.8	27.5	60.1
						Sheardown NW Littoral	18.3	15.0	6.7	1.4	20.9	35.5
Collector-Gatherers (%)	log	YES	0.084	ANOVA	1.1	Reference Lake 3	55.6	19.0	8.5	33.0	57.5	79.2
						Sheardown NW Littoral	76.2	13.1	5.9	60.7	77.8	90.7
Filterers (%)	fourth root	NO	0.148	ANOVA	-0.9	Reference Lake 3	33.9	18.7	8.4	15.5	24.9	56.6
						Sheardown NW Littoral	17.5	15.7	7.0	0.3	20.6	35.2
Shredders (%)	square root	YES	< 0.001	ANOVA	-2.6	Reference Lake 3	7.0	2.6	1.1	2.9	7.5	9.4
						Sheardown NW Littoral	0.4	0.4	0.2	0.0	0.3	1.1
Clingers (%)	none	YES	0.014	ANOVA	-1.4	Reference Lake 3	36.1	18.4	8.2	17.1	26.9	58.3
						Sheardown NW Littoral	9.6	4.7	2.1	4.5	7.9	15.5
Sprawlers (%)	log	NO	0.409	ANOVA	-0.5	Reference Lake 3	51.9	17.7	7.9	29.5	52.5	71.8
						Sheardown NW Littoral	42.5	17.0	7.6	25.9	36.3	65.0
Burrowers (%)	log	YES	< 0.002	ANOVA	5.6	Reference Lake 3	12.0	6.4	2.8	6.9	11.1	22.6
						Sheardown NW Littoral	47.8	19.6	8.8	21.5	48.1	69.6

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

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

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

Table 4.5: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake NW (DLO-01) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	none	YES	< 0.001	ANOVA	5.1	Reference Lake 3	377	155	69	104	452	470
						Sheardown NW Profundal	1,163	242	108	930	1,052	1,513
Richness (Number of Taxa)	none	YES	0.044	ANOVA	3.0	Reference Lake 3	5.4	1.3	0.6	4.0	6.0	7.0
						Sheardown NW Profundal	9.4	3.5	1.6	5.0	10.0	14.0
Simpson's Evenness (E)	log	NO	0.294	ANOVA	0.1	Reference Lake 3	0.455	0.296	0.132	0.218	0.296	0.933
						Sheardown NW Profundal	0.491	0.132	0.059	0.331	0.545	0.632
Bray-Curtis Index	log	YES	0.014	ANOVA	1.5	Reference Lake 3	0.224	0.304	0.136	0.0505	0.109	0.763
						Sheardown NW Profundal	0.686	0.268	0.120	0.429	0.584	0.976
Hydracarina (%)	square root	NO	0.486	ANOVA	0.6	Reference Lake 3	3.7	3.8	1.7	0.0	3.9	8.7
						Sheardown NW Profundal	5.9	1.7	0.8	3.4	5.8	7.9
Ostracoda (%)	none	NO	0.873	ANOVA	-0.1	Reference Lake 3	3.1	2.9	1.3	0.0	2.0	7.5
						Sheardown NW Profundal	2.9	2.4	1.1	0.0	2.8	6.6
Chironomidae (%)	none	NO	0.892	ANOVA	-0.1	Reference Lake 3	90.8	4.9	2.2	82.7	92.2	95.7
						Sheardown NW Profundal	90.4	3.8	1.7	87.5	88.1	96.6
Metal-Sensitive Chironomidae (%)	log	NO	0.222	ANOVA	-0.5	Reference Lake 3	11.4	16.8	7.5	2.3	3.9	41.4
						Sheardown NW Profundal	3.3	2.0	0.9	0.9	3.5	6.0
Collector-Gatherers (%)	rank	NO	0.222	ANOVA	-0.3	Reference Lake 3	89.8	13.6	6.1	66.3	96.2	100.0
						Sheardown NW Profundal	85.2	7.7	3.5	71.9	88.3	91.4
Filterers (%)	fourth root	NO	0.503	ANOVA	-0.5	Reference Lake 3	6.5	10.5	4.7	0.0	3.7	25.0
						Sheardown NW Profundal	1.0	1.4	0.6	0.0	0.6	3.5
Clingers (%)	square root	NO	0.958	ANOVA	-0.2	Reference Lake 3	10.2	13.6	6.1	0.0	3.9	33.6
						Sheardown NW Profundal	7.1	2.9	1.3	3.4	7.5	11.4
Sprawlers (%)	log	NO	0.286	ANOVA	-0.8	Reference Lake 3	79.3	26.8	12.0	32.7	90.4	100.0
						Sheardown NW Profundal	58.8	43.5	19.4	8.3	88.5	92.5
Burrowers (%)	none	NO	0.262	ANOVA	1.7	Reference Lake 3	10.6	14.1	6.3	0.0	5.6	33.6
						Sheardown NW Profundal	34.1	41.3	18.5	1.8	6.0	80.3

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

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

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

to these differences, benthic invertebrate community structure differed significantly between Sheardown Lake NW and Reference Lake 3 at both littoral and profundal habitat types based on Bray-Curtis Index (Tables 4.4 and 4.5). However, because no ecologically significant differences (i.e., CES_{BIC} outside of ±2 SD_{REF}) in the relative abundance of any dominant taxonomic groups were indicated between Sheardown Lake NW and Reference Lake 3 for either habitat type, the difference in Bray-Curtis Index between lakes mostly reflected substantially higher benthic invertebrate density and richness at Sheardown Lake NW. The occurrence of higher benthic invertebrate density without an accompanying difference in Simpson's Evenness or compositional change in dominant taxonomic groups suggested that Sheardown Lake NW was simply more productive than Reference Lake 3, and was not adversely influenced by mine operations in 2018. This was supported by no significant differences in the relative abundance of metal-sensitive chironomids between lakes, as well as by the occurrence of a higher proportion of burrowing taxa (significantly so, for littoral stations) at Sheardown Lake NW compared to Reference Lake 3 (Tables 4.4 and 4.5), which indicated no sediment metal-related influences on the benthic invertebrate community of Sheardown Lake NW. The only ecologically meaningful difference in benthic invertebrate FFG composition between lakes was significantly greater relative abundance of shredders at littoral stations of Sheardown Lake NW compared to Reference Lake 3 (Tables 4.4 and 4.5). However, vegetation and coarse particulate organic matter, which are key food sources for the shredder FFG (Merritt et al. 2008), were observed between the Sheardown Lake NW and reference lake littoral stations suggesting similar food resources for shredders between lakes (Appendix Tables D.1 and D.10). Therefore, the reason for the difference in relative abundance of shredders between Sheardown Lake NW and Reference Lake 3 in 2018 were uncertain. Overall, no adverse mine-related influences to the benthic invertebrate community of Sheardown Lake NW were indicated in 2018 based on comparisons to reference lake conditions.

Temporal comparisons did not indicate any consistent ecologically significant differences in density, richness, and Simpson's Evenness at littoral and profundal habitats of Sheardown Lake NW between the mine baseline (2007, 2008, 2013) period and individual years since the commencement of commercial mine operation (2015 to 2018; Figure 4.14; Appendix Tables F.38 and F.39). In addition, no significant differences in benthic invertebrate dominant taxonomic groups or FFG were uniformly indicated between baseline and mine operational years for littoral or profundal habitats at Sheardown Lake NW (Figure 4.14; Appendix Tables F.38 and F.39). Overall, consistent with no substantial changes in water and sediment quality since the mine baseline period, no significant changes in benthic invertebrate community features were indicated at littoral and profundal habitat of Sheardown Lake NW following the commencement of commercial mine operation in 2015.

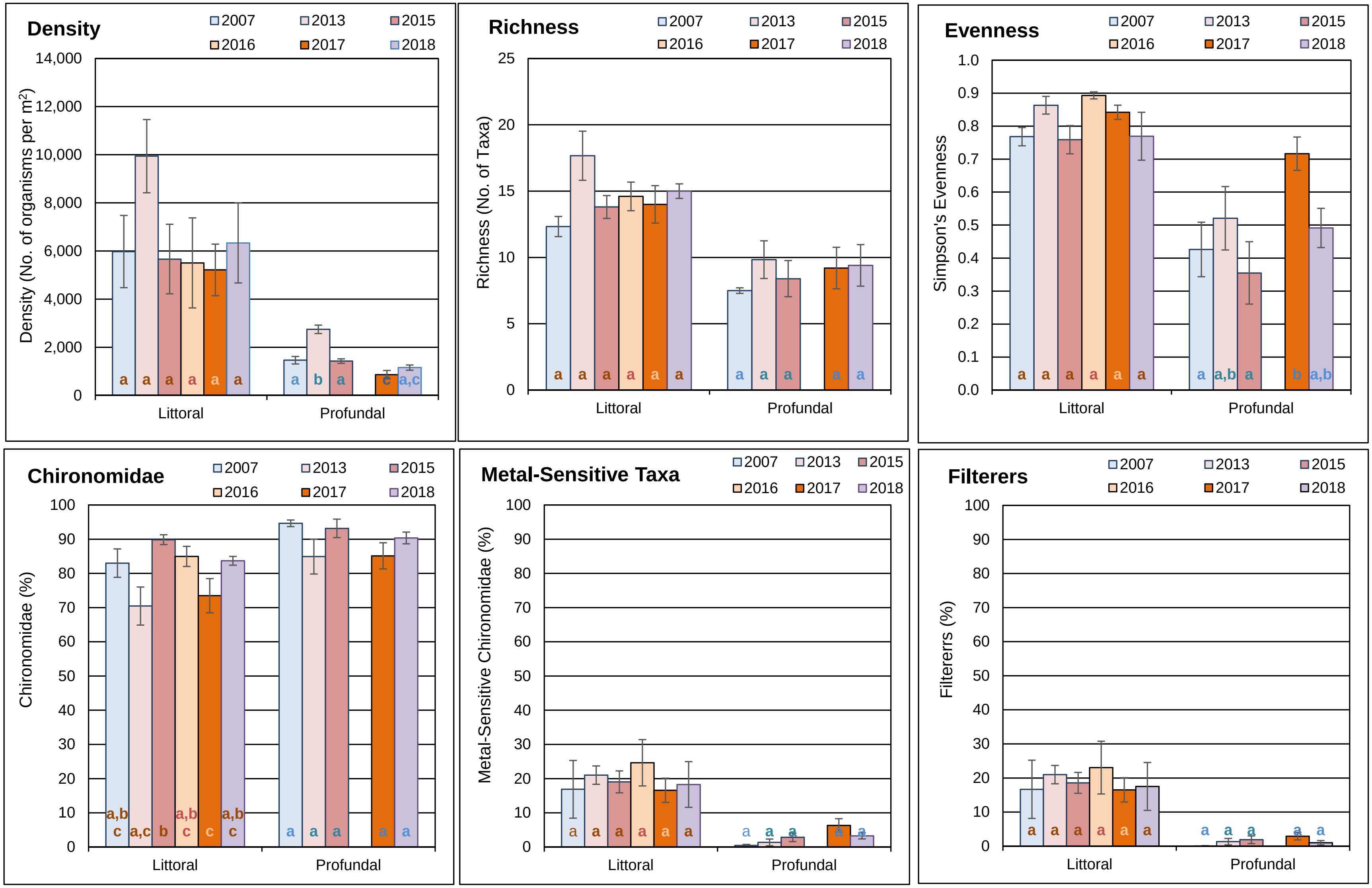


Figure 4.14: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Sheardown Lake NW Littoral and Profundal Study Areas among Mine Baseline (2007, 2013) and Operational (2015 to 2018) Periods

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

4.2.6 Fish Population

4.2.6.1 Sheardown Lake NW Fish Community

Arctic charr was the only fish species captured at the northwest basin of Sheardown Lake in 2018, which differed slightly from that of Reference Lake 3 where low numbers of ninespine stickleback were captured at nearshore rocky habitat in addition to arctic charr (Table 4.6). Total fish CPUE was higher at Sheardown Lake NW than at Reference Lake 3 for nearshore electrofishing and for littoral/profundal gill net sampling (Table 4.6), suggesting higher densities, and/or productivity of arctic charr at the Sheardown Lake northwest basin. A greater relative abundance of fish, together with higher chlorophyll-a concentrations and greater benthic invertebrate density, suggested that overall biological productivity was higher at Sheardown Lake NW than at Reference Lake 3.

Table 4.6: 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 2018

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	101	2	103	2
		CPUE	1.59	0.02	1.61	
	Gill netting	No. Caught	34	0	34	
		CPUE	0.38	0	0.38	
Sheardown Lake Northwest	Electrofishing	No. Caught	98	0	98	1
		CPUE	4.33	0	4.33	
	Gill netting	No. Caught	71	0	71	
		CPUE	0.63	0	0.63	
Sheardown Lake Southeast	Electrofishing	No. Caught	99	1	100	2
		CPUE	5.39	0.05	5.44	
	Gill netting	No. Caught	85	0	85	
		CPUE	3.98	0	3.98	

[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.

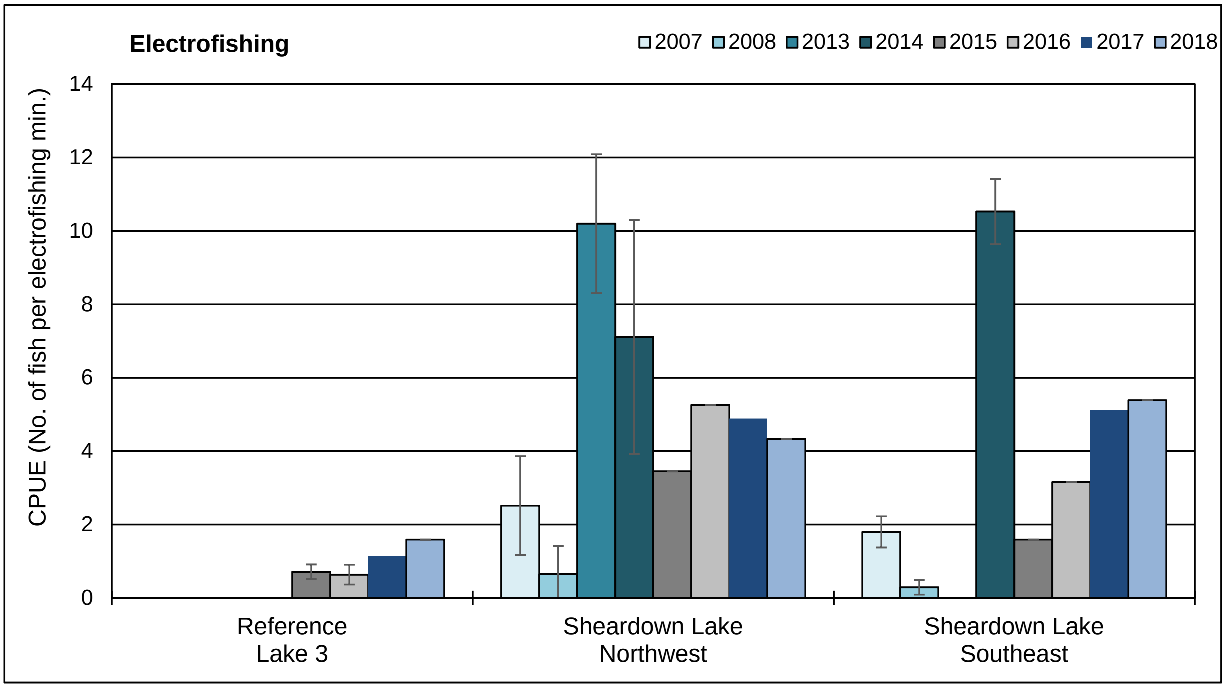
Temporal comparison of the Sheardown Lake NW electrofishing catch data indicated that arctic charr CPUE in 2018 was within the range shown over the mine baseline period (2006 to 2013), and was also comparable to CPUE during mine construction (2014) and previous studies conducted during mine operation (2015 to 2017), at nearshore rocky habitat of the lake (Figure 4.15). Gill netting CPUE for arctic charr in 2018 was also within the range shown during the baseline period, but was somewhat lower than in each of the three previous years of mine operation (Figure 4.15). These results suggested that the relative abundance of arctic charr at the nearshore and littoral/profundal habitats of Sheardown Lake NW in 2018 was similar to baseline studies, in turn suggesting no mine-related influences to arctic charr numbers in the lake.

4.2.6.2 Sheardown Lake NW Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Sheardown Lake NW nearshore arctic charr population were assessed based on a control-impact analysis using data collected from Sheardown Lake NW and Reference Lake 3 in 2018, as well as a before-after analysis using data collected from Sheardown Lake NW in 2018 and during 2013 baseline characterization. A total of 98 and 100 arctic charr were captured at nearshore habitat of Sheardown Lake NW and Reference Lake 3, respectively, in August 2018 for the control-impact analysis. Distinguishing arctic charr YOY from the older, non-YOY age class was possible using a fork length cut-off of 4.5 cm based on evaluation of length-frequency distributions coupled with supporting age determinations for the Sheardown Lake NW and Reference Lake 3 data sets (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 can occur between these life stages. However, because the YOY data set was small (i.e., 10 and 8 YOY for Sheardown Lake NW and Reference Lake 3, respectively), a greater degree of caution is warranted around conclusions drawn from the analysis of YOY endpoints.

Length-frequency distributions for the nearshore arctic charr differed significantly between Sheardown Lake NW and Reference Lake 3 (Table 4.7), potentially reflecting a larger mean size of both YOY and non-YOY individuals captured at Sheardown Lake NW. Arctic charr YOY and non-YOY were significantly longer and heavier at the Sheardown Lake NW nearshore than at the Reference Lake 3 nearshore (Table 4.7; Appendix Table G.14). No significant difference in the condition of arctic charr YOY was indicated between Sheardown Lake NW and Reference Lake 3 nearshore habitats, and although condition of non-YOY was significantly lower at Sheardown Lake NW, the magnitude of this difference was within the CES_C of ±10% suggesting that this difference was not ecologically meaningful (Table 4.7; Appendix Table G.14). Overall, these



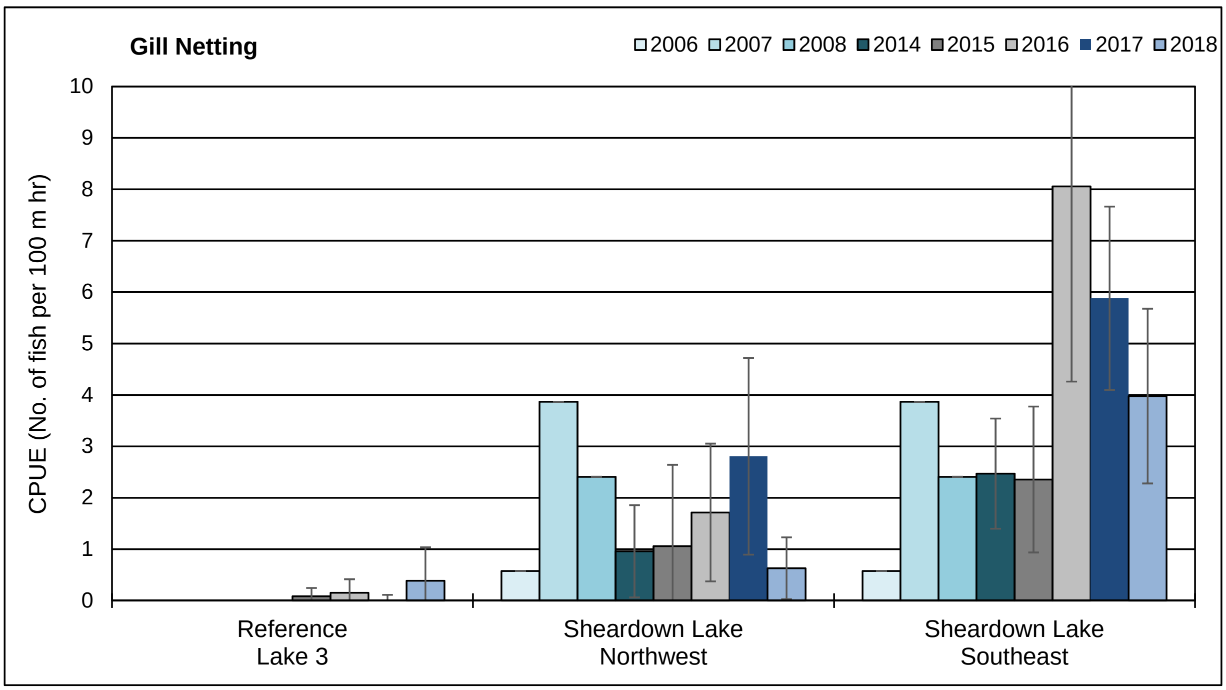


Figure 4.15: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake NW (DLO-01) and Sheardown Lake SE

Notes: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015 to 2018) mine phases. Lake basins (i.e., NW or SE) were not differentiated historically for baseline gill netting catches.

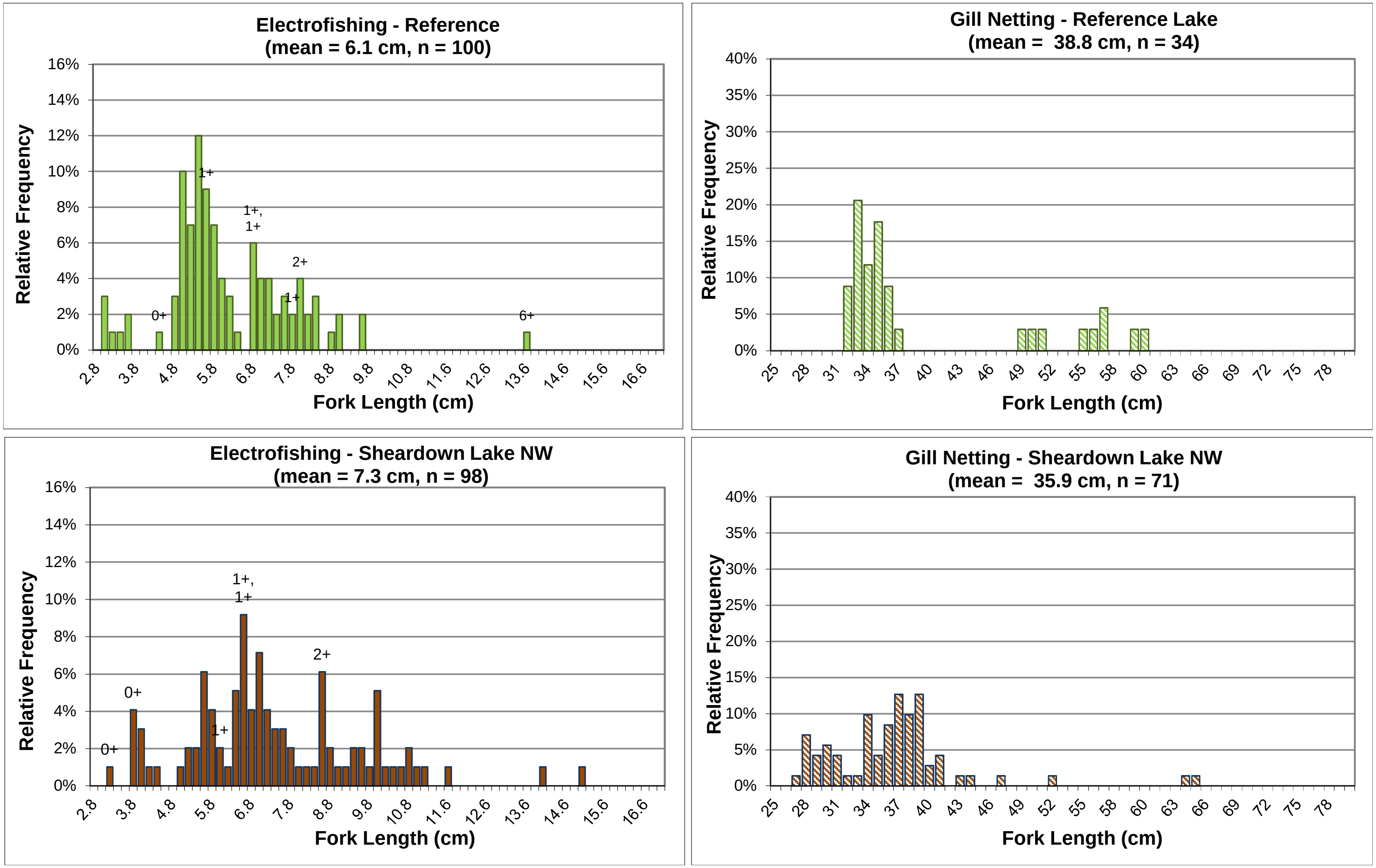


Figure 4.16: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2018

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

Table 4.7: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake NW and Reference Lake 3 from 2015 to 2018, and between Sheardown Lake NW Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]							
			versus Reference Lake 3				versus Sheardown Lake NW baseline period data [b]			
			2015	2016	2017	2018	2015	2016	2017	2018
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	No	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	**Yes (+29%)**	**Yes (+17%)**	**Yes (+20%)**	**Yes (+24%)**	No	No	No	**Yes (-12%)**
		Size (mean weight)	**Yes (+121%)**	**Yes (+60%)**	No	**Yes (+83%)**	No	**Yes (-29%)**	No	**Yes (-50%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	**Yes (+7%)**	**Yes (-5%)**	**Yes (-13%)**	**Yes (-12%)**	**Yes (-9%)**	**Yes (-10%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	No	**Yes**	**Yes**	**Yes**	No
		Age	-	-	-	-	**Yes (-35%)**	**Yes (-28%)**	**Yes (-26%)**	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (-21%)**	**Yes (-14%)**	**Yes (-6%)**	No
		Size (mean weight)	-	-	-	No	**Yes (-47%)**	**Yes (-31%)**	**Yes (-9%)**	No
		Growth (fork length-at-age)	-	-	-	-	No	No	No	-
		Growth (weight-at-age)	-	-	-	-	No	No	**Yes (+24%)**	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+4%)**	**Yes (+8%)**	**Yes (+11%)**	**Yes (+6%)**	No

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

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

[c] Due to low catches of arctic charr in gill nets at Reference Lake 3 in 2015, 2016, and 2017, no comparison of fish health was conducted for gill netted fish.

results indicated no substantial differences in the health of nearshore arctic charr between Sheardown Lake NW and reference lake conditions in 2018.

Temporal comparisons of the Sheardown Lake NW nearshore arctic charr data indicated a significantly different length-frequency distribution between 2018 and the combined 2007 and 2013 baseline data (Table 4.7; Appendix Table G.7). Lengths and weights of arctic charr non-YOY captured at the nearshore of Sheardown Lake NW in 2018 were significantly lower than non-YOY captured during the mine baseline (Table 4.7). In addition, the condition of arctic charr non-YOY was significantly lower in 2018 than during baseline studies conducted at Sheardown Lake NW (Table 4.7). Although the length and weight of non-YOY arctic charr in years of mine operation (i.e., 2015 to 2018) has not shown consistent differences from the baseline period, the condition of non-YOY arctic charr has consistently been significantly lower, at magnitude near the CES_C of ±10%, during all years of mine operation compared to the baseline period (Table 4.7). This suggested on-going, lower condition of arctic charr non-YOY at Sheardown Lake NW nearshore habitat following the commencement of commercial mine operations compared to the baseline period. Temporal comparisons of nearshore arctic charr populations between Sheardown Lake NW and Reference Lake 3 since 2015 generally indicated the continual presence of significantly larger non-YOY at Sheardown Lake NW, but no consistent differences in nearshore arctic charr condition (Table 4.7).

Littoral/Profundal Arctic Charr

Mine-related influences on the Sheardown Lake NW littoral/profundal Arctic charr population were assessed based on a control-impact analysis using 2018 data from Sheardown Lake NW and Reference Lake 3, as well as using a before-after analysis between data collected in 2018 and the baseline characterization studies (combined 2006, 2007, 2008, and 2013). A total of 71 and 34 arctic charr were sampled from littoral/profundal habitat of Sheardown Lake NW and Reference Lake 3, respectively, in August 2018, for the control-impact analysis. The length-frequency distribution for littoral/profundal arctic charr did not differ significantly between lakes (Table 4.7; Figure 4.16). In addition, no significant differences in mean length or weight of littoral/ profundal arctic charr were indicated between Sheardown Lake NW and Reference Lake 3 (Table 4.7; Appendix Table G.18). Although condition of arctic charr captured at littoral/profundal areas of Sheardown Lake NW was significantly higher than at Reference Lake 3, the absolute magnitude of this difference was less than the CES_C of 10% suggesting that the difference in arctic charr condition between lakes was not ecologically meaningful (Table 4.7; Appendix Table G.18).

The length-frequency distribution for arctic charr captured at littoral/profundal habitat of Sheardown Lake NW did not differ significantly between 2018 and the baseline period studies

(Table 4.7). In addition, no significant differences in length, weight, or condition of arctic charr captured at littoral/profundal habitat were indicated between 2018 and the baseline period (Table 4.7; Appendix Table G.18). In all previous years of mine operation (i.e., 2015 to 2017), arctic charr sampled from littoral/profundal habitat of Sheardown Lake NW were significantly shorter, lighter, and of greater condition than during the baseline period (Table 4.7). The absence of size and condition differences in 2018 compared to the baseline period appeared to reflect closer comparability in fish size between these study periods than those sampled over the period from 2015 to 2017 when evaluating the data relative to the baseline period.[11] This suggested that arctic charr condition is strongly size dependent and as such, will vary among different age classes of fish. Overall, the general absence of significant, ecologically meaningful differences in condition of arctic charr captured at littoral/ profundal areas of Sheardown Lake NW from 2015 to 2018 compared to 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.

4.2.7 Integrated Summary

At Sheardown Lake NW, aqueous concentrations of chloride, molybdenum, and uranium were elevated compared to Reference Lake 3 in 2018, and chloride, manganese, molybdenum, and sulphate concentrations were elevated compared to the baseline period, suggesting a mine-related source of these parameters to the lake. As during the previous CREMP studies, total aluminum and manganese concentrations showed strong positive correlations with turbidity that, in turn, suggested that these metals were largely bound to/contained in suspended particulate matter and were not likely biologically available. The occurrence of relative high turbidity in Sheardown Lake is hypothesized to reflect natural sources of suspended particulates originating from Mary River, upstream of the mine. Notably, no parameters were elevated above WQG or AEMP benchmarks at Sheardown Lake NW in 2018. Metal concentrations in sediment at littoral and profundal habitats of Sheardown Lake NW were very similar to concentrations observed for the same respective habitat types at Reference Lake 3 in 2018, suggesting no marked mine-related influences on sediment metal concentrations in Sheardown Lake NW. Concentrations of chromium, iron, manganese, nickel, and phosphorus were above SQG in sediment at littoral and/or profundal habitats, and concentrations of manganese and nickel were above site-specific AEMP benchmarks in sediment at profundal habitat of Sheardown Lake NW in 2018. However, with the exception of nickel, concentrations of these metals were also above respective SQG and Sheardown Lake NW AEMP benchmarks at Reference Lake 3, suggesting natural elevation of some metals in sediment of local study area lakes. Overall, some mine-related effects on water

[11] Average fork length of arctic charr sampled for CREMP studies was 37.2 cm during baseline, 29.9 cm in 2015, 32.3 cm in 2016, 32.9 cm in 2017, and 35.9 cm in 2018.

quality and sediment quality were evident at Sheardown Lake NW in 2018, but the effects were minor and did not result in parameter concentrations substantially exceeding applicable guidelines.

Chlorophyll-a concentrations at Sheardown Lake NW were significantly higher than at Reference Lake 3 in 2018 suggesting greater primary production at Sheardown Lake. However, chlorophyll-a concentrations remained well below the AEMP benchmark during all seasonal sampling events in 2018 at Sheardown Lake NW, and suggested oligotrophic conditions typical of Arctic waterbodies. Temporal evaluation of the chlorophyll-a data indicated no changes to the trophic status of Sheardown Lake NW since commencement of commercial mine operations. The benthic invertebrate community of Sheardown Lake NW showed significantly higher density and richness, but no ecologically significant differences in Simpson's Evenness and relative abundance of dominant groups including metal-sensitive chironomids, compared to Reference Lake 3 in 2018. The occurrence of higher benthic invertebrate density without an accompanying difference in Simpson's Evenness or compositional change in dominant taxonomic groups suggested that Sheardown Lake NW was simply more productive than Reference Lake 3, and was not adversely influenced by mine operations. No ecologically significant differences in benthic invertebrate density, richness, Simpson's Evenness, and relative abundance of dominant taxonomic groups or FFG were consistently shown from 2015 to 2018 compared to years in which mine baseline data were collected. Analysis of arctic charr populations suggested greater fish abundance at Sheardown Lake NW compared to Reference Lake 3 in 2018, and similar abundance of arctic charr at Sheardown Lake NW in 2018 compared to the mine baseline studies. Arctic charr captured at nearshore habitat of Sheardown Lake NW showed no ecologically significant differences in size and condition compared to those captured at Reference Lake 3 in 2018. Although non-YOY arctic charr captured at nearshore habitat were of significantly lower condition in 2018 compared to those captured during mine baseline studies, condition has not differed consistently in all years at Sheardown Lake NW since commercial mine operation commenced in 2015. Arctic charr captured at littoral/profundal habitat of Sheardown Lake NW showed no ecologically significant differences in condition compared to Reference Lake 3 in 2018, nor any ecologically meaningful difference in condition compared to those captured during baseline studies. 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 NW in the fourth year of mine operation at the Mary River Project.

4.3 Sheardown Lake Southeast (DLO-2)

4.3.1 Hydraulic Retention Time

A hydraulic retention time of 83 ± 35 days was estimated for Sheardown Lake SE by Minnow (2018) using mean annual watershed runoff extrapolated from Baffinland flow monitoring stations installed in small watershed watercourses (i.e., ≤15 km^2) located on the mine property and a lake volume of 1.80 million cubic metres.

4.3.2 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 2018 (Appendix Figures C.15 to C.18). No thermal stratification was evident at the Sheardown Lake SE basin during any of the winter, summer, or fall sampling events (Figure 4.17). The average water temperature at the bottom of the water column at Sheardown Lake SE littoral and profundal stations was significantly warmer than at Reference Lake 3 during the August 2018 biological sampling (Figure 4.8; Appendix Table C.53). However, the incremental difference in average bottom water temperature between lakes at each respective depth was small (i.e., ≤0.6˚C) and thus was unlikely to be ecologically meaningful. 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 2018 showed no substantial change in dissolved oxygen saturation with depth during summer and fall, but oxycline development characterized by decreasing saturation levels with increasing depth occurring at depths greater than 9 m during the winter sampling event (Figure 4.17). Dissolved oxygen saturation levels at the bottom of the water column at littoral and profundal stations of Sheardown Lake SE did not differ significantly than those at Reference Lake 3 during the August 2018 biological sampling (Figure 4.8; Appendix Table C.53). Dissolved oxygen saturation levels were well above WQG (54% saturation, or 9.5 mg/L) at Sheardown Lake SE at all depths during the winter, summer, fall sampling events in 2018 (Figures 4.8 and 4.17), indicating that dissolved oxygen was not likely to be limiting to pelagic or bottom-dwelling biota within the lake.

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, despite pH being significantly higher (i.e., more

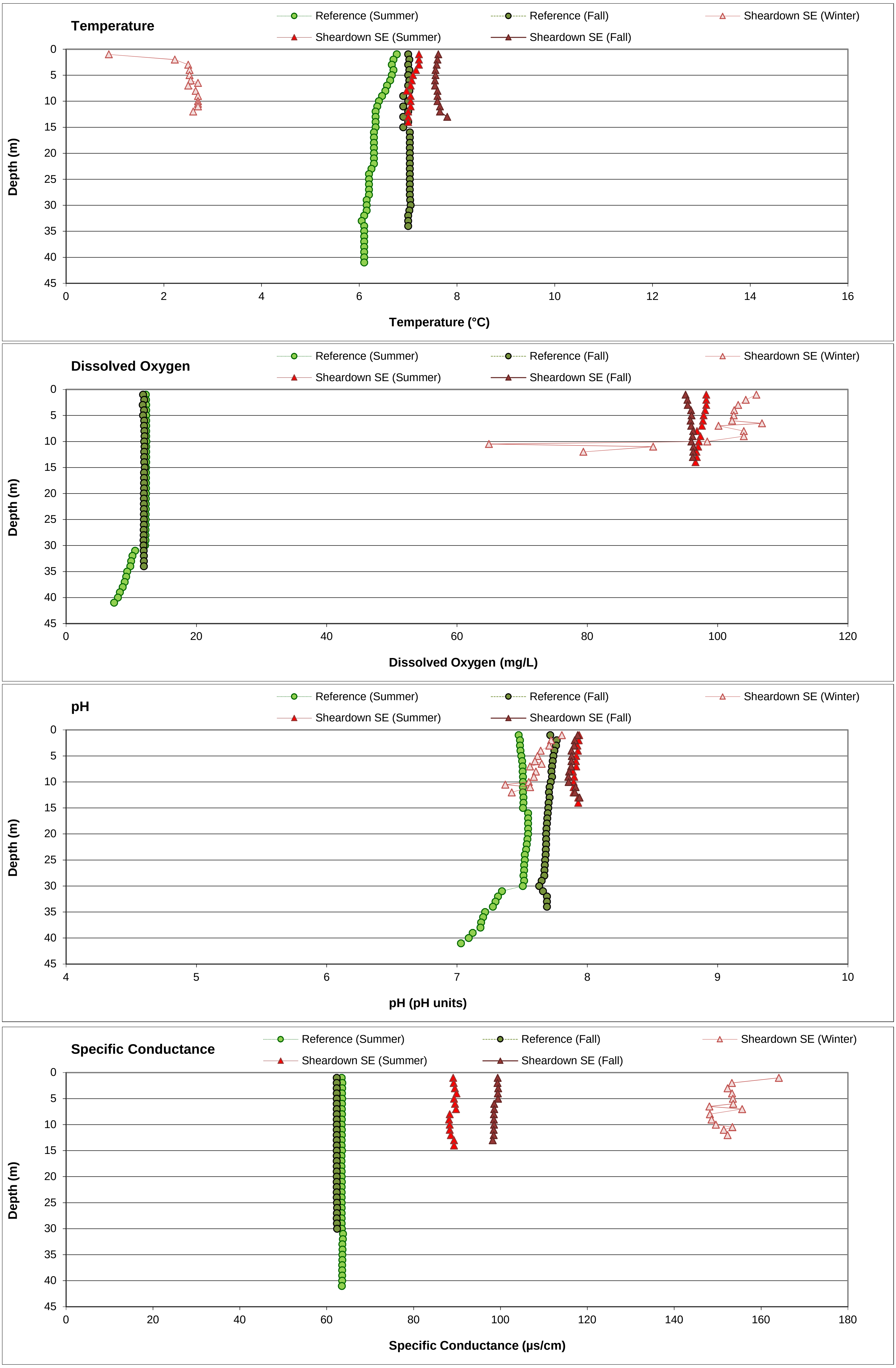


Figure 4.17: **Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake SE (DLO-02) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2018**

alkaline) at Sheardown Lake SE compared to Reference Lake 3 during August 2018 sampling, pH was consistently within WQG limits at Sheardown Lake SE in 2018 (Figure 4.8; Appendix Table C.53; Figure 4.17). Specific conductance was also significantly higher at Sheardown Lake SE compared to Reference Lake 3 during the August 2018 biological study (Figure 4.8). However, mean specific conductance at Sheardown Lake SE (i.e., 97 µS/cm) was within the range observed among the reference creek and river stations in fall 2018 (i.e., 67 to 107 µS/cm). Therefore, similar to previous CREMP studies, 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 Sheardown Lake SE was the lowest among the mine-exposed lakes (Appendix Figure C.7). Secchi depth readings from Sheardown Lake SE were significantly lower (shallower) than at Reference Lake 3 during the August 2018 biological study, but were relatively consistent among stations, suggesting no spatial differences in water clarity of the lake (Appendix Tables C.51 and C.53).

Water chemistry at Sheardown Lake SE showed no consistent spatial changes in parameter concentrations among the five lake sampling stations during any of the winter, summer or fall sampling events in 2018 (Table 4.8; Appendix Table C.54), suggesting that the lake waters were well mixed both laterally and vertically. Total aluminum and manganese concentrations were highly elevated (i.e., ≥10-fold), turbidity was moderately elevated (i.e., 5- to 10-fold), and concentrations of total copper and molybdenum were slightly elevated (i.e., 3- to 5-fold), at Sheardown Lake SE compared to Reference Lake 3 during the 2018 summer and/or fall sampling events (Table 4.8; Appendix Tables C.44 and C.54). Dissolved aluminum and molybdenum concentrations were also slightly elevated at Sheardown Lake SE compared to Reference Lake 3 in one or both of the summer and fall sampling events (Appendix Table C.56). Similar to the northwest basin, total aluminum and manganese concentrations showed highly to moderately strong positive correlations with turbidity for the Sheardown Lake SE combined data set (i.e., winter, summer and fall data; r_s = 0.77 and 0.48, respectively), suggesting that much of the total aluminum and manganese was associated with suspended particles (Appendix Table C.57). This was corroborated by comparison of total and dissolved fractions, which indicated that on average, most aluminum and manganese (i.e., 90% and 83%, respectively) was in particulate form at Sheardown Lake SE (compare Appendix Tables C.54 and C.55). 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 with aluminum and manganese, total copper and molybdenum concentrations at Sheardown Lake SE were not positively correlated with turbidity, suggesting that slight elevation in these parameters compared to Reference Lake 3 was related to mine operation and/or natural geochemical differences between lakes. Despite elevation of some metals at Sheardown Lake SE, on average, parameter concentrations were all well below established WQG and AEMP benchmarks during the winter, summer and fall sampling events in 2018 (Table 4.8; Appendix Table C.54).

Temporal comparisons of the Sheardown Lake SE water chemistry data indicated no appreciable changes in average parameter concentrations between the 2018 study and mine baseline period (2005 to 2013), the only exception of which was a slightly elevated average dissolved aluminum concentration in fall 2018 (Figure 4.9; Appendix Tables C.44 and C.56; Appendix Figure C.19). As indicated above, because aluminum concentrations were strongly correlated with turbidity, higher dissolved aluminum concentrations in fall 2018 compared to baseline at Sheardown Lake SE likely reflected natural phenomena. No parameters showed consistently higher concentrations annually over the mine construction (2014) and 2015 to 2018 mine operational periods with the exceptions of sodium and sulphate (Figure 4.9; Appendix Figure C.19), suggesting a potential mine-related source of these constituents. However, an average sulphate concentration of approximately 5.5 mg/L at Sheardown Lake SE in 2018 was well below the WQG of 218 mg/L, indicating adverse effects associated with sulphate concentrations were highly unlikely.

4.3.3 Sediment Quality

Surficial sediment at Sheardown Lake SE was composed of silt loam material containing low TOC content throughout the lake (Figure 4.18; Appendix Tables D.15 and D.16). Substrate at littoral stations of Sheardown Lake SE contained significantly lower sand and TOC content, and significantly greater silt and clay content, than at Reference Lake 3 (Appendix Table D.17). Similarly, sediment at profundal stations of Sheardown Lake SE showed significantly lower sand content and significantly higher silt content than at Reference Lake 3 (Appendix Table D.17). The relatively 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 Reference Lake 3, iron (oxy)hydroxide material was visible in surficial and/or sub-surface substrate of Sheardown Lake SE, in some cases occurring as a thin, distinct layer or floc (Appendix Tables D.15 and D.16). Below the surficial layer, substrates at Sheardown Lake SE exhibited some sporadic blackening suggesting development of reducing conditions. However, no distinct redox boundary was generally observed in sediment at the Sheardown Lake SE stations (Appendix Tables D.15 and D.16). Observations regarding reducing sediment conditions at Sheardown Lake SE were similar to those made at Reference Lake 3 (Appendix Tables D.1, D.2, D.15 and D.16), suggesting that factors leading to reduced sediment conditions were comparable between lakes.

Table 4.8: Water Chemistry at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3) Monitoring Stations[a], Mary River Project CREMP, August 2018

	Parameters	Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3)	Sheardown Lake Southeast (SDSE) Station				
						DL0-02-6	DL0-02-7	DL0-02-4	DL0-02-8	DL0-02-3
					Fall 2018	23-Aug-18	23-Aug-18	23-Aug-18	23-Aug-18	23-Aug-18
Conventionals	Conductivity (lab)	umho/cm	-	-	75	121	120	119.5	118	117
	pH (lab)	pH	6.5 - 9.0	-	7.65	8.02	7.92	8.00	7.97	7.95
	Hardness (as $CaCO_3$)	mg/L	-	-	35	60	57	57	58	58
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	2.1	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	46	43	58	37	39	53
	Turbidity	NTU	-	-	0.5	2.1	2.3	2.3	2.2	2.3
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	48	46	45	47	43
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.044	<0.020	0.067	0.035	0.027	0.021
	Nitrate	mg/L	13	13	<0.020	0.0435	0.04225	0.0435	0.041	0.0415
	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	-	-	0.16	<0.15	<0.15	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.94	1.65	1.68	1.65	1.88	1.61
	Total Organic Carbon	mg/L	-	-	3.84	2.60	2.60	2.57	2.79	2.51
	Total Phosphorus	mg/L	0.020[α]	-	0.0049	0.0077	0.0040	0.0036	0.0052	0.0050
	Phenols	mg/L	0.004[α]	-	0.0011	0.0012	0.0014	<0.0010	<0.0010	0.0011
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	2.87	2.79	2.73	2.76	2.75
	Sulphate (SO_4)	mg/L	218[β]	218	3.74	6.00	5.69	5.56	5.58	5.58
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.004	0.099	0.062	0.065	0.048	0.061
	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	0.000105	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0064	0.0070	0.0064	0.0059	0.0067	0.0061
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	7.16	11.75	11.325	11.15	10.85	11.10
	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	0.00015	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.0008	0.0010	0.0009	0.0007	0.0007	0.0008
	Iron (Fe)	mg/L	0.30	0.300	<0.030	0.258	0.061	0.064	0.046	0.062
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	0.0001625	0.00007425	0.0000735	0.0000625	0.000074
	Magnesium (Mg)	mg/L	-	-	4.26	7.41	7.03	6.58	7.54	6.78
	Manganese (Mn)	mg/L	0.935[β]	-	0.00064	0.02290	0.00555	0.00499	0.00319	0.00500
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.000138	0.000457	0.0004715	0.00045	0.000514	0.00046
	Nickel (Ni)	mg/L	0.025	0.025	0.0005	0.00081	0.000625	0.000595	0.00058	0.000585
	Potassium (K)	mg/L	-	-	0.86	1.08	1.05	0.98	1.12	1.00
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.417	0.605	0.568	0.575	0.565	0.550
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.86	1.30	1.28	1.18	1.34	1.21
	Strontium (Sr)	mg/L	-	-	0.0081	0.0084	0.0081	0.0081	0.0079	0.0081
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Uranium (U)	mg/L	0.015	-	0.00026	0.00079	0.00071	0.00067	0.00058	0.00068
	Vanadium (V)	mg/L	0.006[α]	0.006	<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

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

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

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

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

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

BOLD Indicates parameter concentration above the AEMP benchmark.

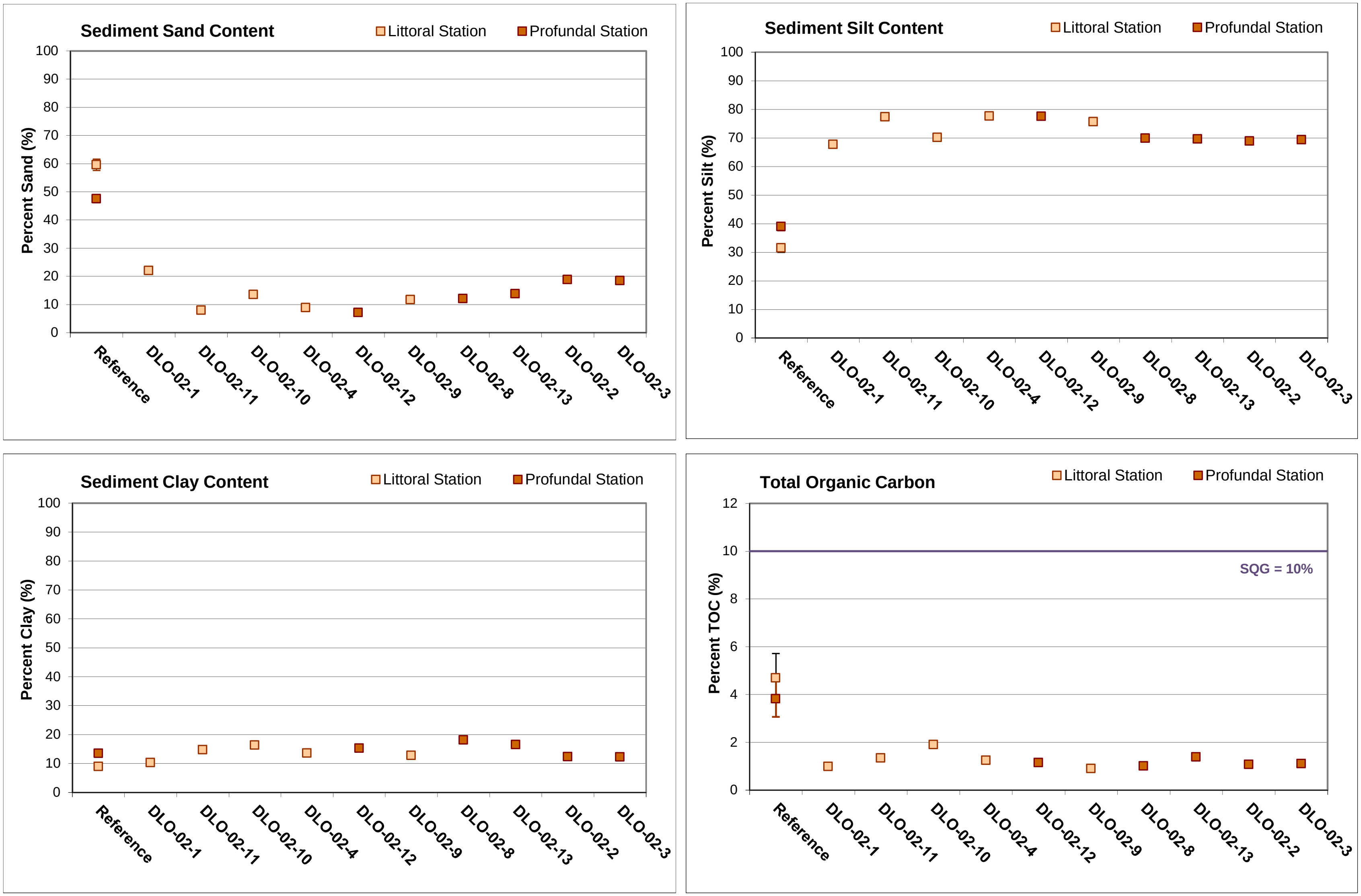


Figure 4.18: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake SE (DLO-02) Sediment Monitoring Stations and Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2018

Sediment metal concentrations at Sheardown Lake SE showed no clear spatial gradients with progression towards the lake outlet in 2018, suggesting no clear point sources of metals to the lake (Appendix Table D.18). Sediment metal concentrations at littoral and profundal stations of Sheardown Lake SE were, on average, similar to those observed for the same respective station types at Reference Lake 3 (Table 4.3; Appendix Table D.19) suggesting no marked mine-related influences on sediment metal concentrations at the southeast lake basin. The average concentration of iron in littoral sediment was above SQG, and average iron and manganese concentrations in littoral and profundal sediment were above AEMP benchmarks, at Sheardown Lake SE (Table 4.3; Appendix Table D.18). However, as indicated previously, average concentrations of iron and manganese were also above respective SQG and AEMP benchmarks at littoral and/or profundal stations of Reference Lake 3 (Table 4.3). This suggested that the elevation of iron and manganese concentrations in sediment of Sheardown Lake SE relative to SQG and AEMP benchmarks may be a natural phenomenon in lakes within the local study area of the mine. Arsenic, chromium, nickel, and phosphorus concentrations were also above lake-specific AEMP benchmarks at littoral station DLO-02-4, but on average, concentrations of these metals were below their respective AEMP benchmarks at Sheardown Lake SE, and were not unlike concentrations observed at individual stations at Reference Lake 3 (Table 4.3; Appendix Tables D.4 and D.18).

Temporal comparisons indicated that metal concentrations in sediment at littoral and profundal stations of Sheardown Lake SE in 2018 were comparable to those observed during the mine baseline (2005 to 2013) period (Figure 4.11; Appendix Table D.19). On average, metal concentrations in sediment at littoral and profundal stations in 2018 were also within the range of those observed from 2015 to 2017, with no occurrence of consistently higher metal concentrations that would suggest an increasing trend over time (Figure 4.11). Overall, no substantial changes in metal concentrations were indicated in sediment at Sheardown Lake SE since the commencement of commercial mine operations in 2015.

4.3.4 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 2018 (Figure 4.12). Chlorophyll-a concentrations did not differ significantly among the winter, summer, and fall sampling events in 2018, indicating relatively uniform phytoplankton abundance among seasons (Appendix Table E.6). Similar to Camp Lake and Sheardown Lake NW, chlorophyll-a concentrations at Sheardown Lake SE were significantly greater than at Reference Lake 3 for both the summer and fall sampling events in 2018 (Appendix Table E.7 and E.8), but concentrations were generally well below the AEMP benchmark of 3.7 µg/L at all stations and for all sampling events (Figure 4.12). On average, chlorophyll-a concentrations at Sheardown Lake SE indicated an oligotrophic status as defined by Wetzel (2001). This trophic status classification was consistent with an oligotrophic categorization for Sheardown Lake SE based on CWQG trophic classifications as defined by total phosphorus concentrations (i.e., average concentrations below 10 µg/L; Table 4.8; Appendix Table C.54).

Temporal comparison of Sheardown Lake SE chlorophyll-a concentrations did not indicate any consistent direction of significant differences between the 2018 data and data from the mine construction (2014) period or previous years of mine operation (2015 to 2017) among the winter, summer, or fall seasons (Figure 4.19; Appendix Table E.13). The variability in chlorophyll-a concentrations among years at Sheardown Lake SE may reflect the combination of mine-related influences and variable influence of Mary River on Sheardown Lake SE water levels, hydraulic retention time, and/or chemistry among years/seasons. For instance, Mary River discharges into or drains Sheardown Lake SE during high and low flow periods, respectively, the nature of which may affect phytoplankton abundance and/or community structure. No chlorophyll-a baseline (2005 to 2013) data are available for Sheardown Lake SE, precluding comparisons to conditions prior to the mine construction period.

4.3.5 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral and profundal habitats of Sheardown Lake SE compared to like-habitat stations at Reference Lake 3, the differences of which were at magnitudes well outside of the CES_{BIC} of ±2 SD_{REF} (Tables 4.9 and 4.10). An ecologically meaningful difference in richness was also indicated between Sheardown Lake SE and Reference Lake 3, but only for profundal habitat. In addition to these differences, benthic invertebrate community structure differences were indicated between Sheardown Lake SE and Reference Lake 3 based on significantly differing Bray-Curtis Index for both littoral and profundal habitat types (Tables 4.9 and 4.10). However, similar to Sheardown Lake NW, no ecologically significant differences in the relative abundance of any dominant taxonomic groups were shown between Sheardown Lake SE and Reference Lake 3 for each habitat type, and therefore the difference in Bray-Curtis Index between lakes most likely reflected substantially higher benthic invertebrate density at Sheardown Lake SE. As at Sheardown Lake NW, the occurrence of higher benthic invertebrate density without an accompanying difference in Simpson's Evenness or change in dominant taxonomic group composition suggested that Sheardown Lake SE was simply more productive than Reference Lake 3, and was not adversely influenced by mine operations in 2018. This was supported by the occurrence of no ecologically significant differences in the relative abundance of metal-sensitive chironomids between lakes (Tables 4.9 and 4.10).

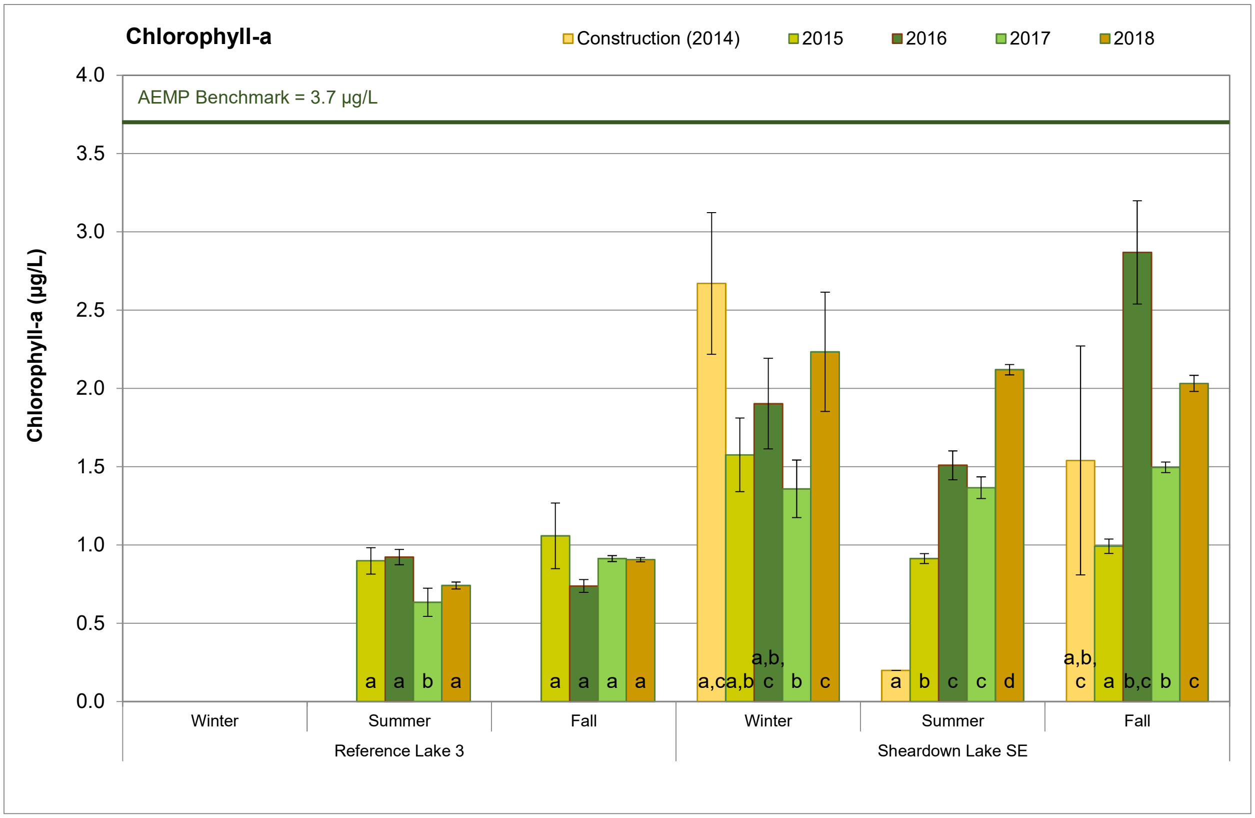


Figure 4.19: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between Sheardown Lake SE and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2018) Periods (mean ± SE)

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

Table 4.9: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake SE (DLO-02) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	square root	YES	0.002	t-test (unequal)	12.5	Reference Lake 3	1,045	258	116	696	1,000	1,391
						Sheardown SE Littoral	4,277	1,533	686	2,687	4,287	6,000
Richness (Number of Taxa)	none	NO	0.707	ANOVA	-0.3	Reference Lake 3	10.8	2.3	1.0	7.0	11.0	13.0
						Sheardown SE Littoral	10.2	2.6	1.2	7.0	10.0	14.0
Simpson's Evenness (E)	log	NO	0.125	ANOVA	-1.2	Reference Lake 3	0.825	0.103	0.046	0.720	0.816	0.939
						Sheardown SE Littoral	0.704	0.131	0.059	0.582	0.695	0.922
Bray-Curtis Index	none	YES	< 0.001	ANOVA	6.1	Reference Lake 3	0.313	0.092	0.041	0.178	0.358	0.394
						Sheardown SE Littoral	0.871	0.034	0.015	0.833	0.862	0.908
Nemata (%)	square root	NO	0.116	ANOVA	-0.7	Reference Lake 3	7.1	8.8	3.9	0.0	3.4	21.3
						Sheardown SE Littoral	0.6	0.5	0.2	0.0	0.6	1.3
Ostracoda (%)	fourth-root	YES	0.042	ANOVA	-1.0	Reference Lake 3	23.9	18.3	8.2	3.4	20.6	53.3
						Sheardown SE Littoral	6.1	9.9	4.4	0.3	2.0	23.6
Chironomidae (%)	square root	YES	0.058	ANOVA	1.1	Reference Lake 3	66.9	22.2	10.0	35.5	73.8	91.4
						Sheardown SE Littoral	92.4	10.0	4.5	74.5	96.5	97.8
Metal-Sensitive Chironomidae (%)	log	YES	0.017	ANOVA	-1.2	Reference Lake 3	36.5	19.6	8.8	17.8	27.5	60.1
						Sheardown SE Littoral	12.1	8.9	4.0	5.0	10.7	26.7
Collector-Gatherers (%)	none	NO	0.760	ANOVA	-0.2	Reference Lake 3	55.6	19.0	8.5	33.0	57.5	79.2
						Sheardown SE Littoral	52.5	11.1	5.0	37.3	50.1	65.3
Filterers (%)	log	YES	0.025	ANOVA	-1.2	Reference Lake 3	33.9	18.7	8.4	15.5	24.9	56.6
						Sheardown SE Littoral	12.1	8.9	4.0	5.0	10.7	26.7
Shredders (%)	square root	YES	< 0.001	ANOVA	-2.7	Reference Lake 3	7.0	2.6	1.1	2.9	7.5	9.4
						Sheardown SE Littoral	0.1	0.2	0.1	0.0	0.0	0.3
Clingers (%)	none	YES	0.017	ANOVA	-1.4	Reference Lake 3	36.1	18.4	8.2	17.1	26.9	58.3
						Sheardown SE Littoral	10.6	4.8	2.1	5.8	11.4	16.8
Sprawlers (%)	log	NO	0.476	ANOVA	-0.5	Reference Lake 3	51.9	17.7	7.9	29.5	52.5	71.8
						Sheardown SE Littoral	43.9	16.9	7.6	30.3	39.8	72.7
Burrowers (%)	log	YES	0.012	ANOVA	5.3	Reference Lake 3	12.0	6.4	2.8	6.9	11.1	22.6
						Sheardown SE Littoral	45.5	21.2	9.5	10.6	47.0	63.8

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

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

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

Table 4.10: Benthic Invertebrate Community Statistical Comparison Results between Sheardown Lake SE (DLO-02) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	log	YES	0.002	ANOVA	18.5	Reference Lake 3	377	155	69	104	452	470
						Sheardown SE Profundal	3,237	2,771	1,239	1,296	1,757	7,896
Richness (Number of Taxa)	rank	YES	0.080	Mann-Whitney U	2.2	Reference Lake 3	5.4	1.3	0.6	4.0	6.0	7.0
						Sheardown SE Profundal	8.4	2.2	1.0	6.0	10.0	10.0
Simpson's Evenness (E)	log	NO	0.206	ANOVA	0.4	Reference Lake 3	0.455	0.296	0.132	0.218	0.296	0.933
						Sheardown SE Profundal	0.568	0.050	0.022	0.516	0.556	0.643
Bray-Curtis Index	rank	YES	0.008	Mann-Whitney U	2.5	Reference Lake 3	0.224	0.304	0.136	0.051	0.109	0.763
						Sheardown SE Profundal	0.981	0.009	0.004	0.968	0.980	0.991
Ostracoda (%)	square root	NO	0.145	ANOVA	-0.8	Reference Lake 3	3.1	2.9	1.3	0.0	2.0	7.5
						Sheardown SE Profundal	0.8	1.3	0.6	0.0	0.4	3.1
Chironomidae (%)	none	YES	0.021	ANOVA	1.4	Reference Lake 3	90.8	4.9	2.2	82.7	92.2	95.7
						Sheardown SE Profundal	97.6	2.1	0.9	95.5	97.0	100.0
Metal-Sensitive Chironomidae (%)	rank	NO	1.000	Mann-Whitney U	-0.3	Reference Lake 3	11.4	16.8	7.5	2.3	3.9	41.4
						Sheardown SE Profundal	5.9	3.5	1.6	3.4	5.1	11.9
Collector-Gatherers (%)	rank	YES	0.056	Mann-Whitney U	-1.9	Reference Lake 3	89.8	13.6	6.1	66.3	96.2	100.0
						Sheardown SE Profundal	63.8	22.4	10.0	26.2	70.5	83.9
Filterers (%)	fourth root	NO	0.278	ANOVA	-0.1	Reference Lake 3	6.5	10.5	4.7	0.0	3.7	25.0
						Sheardown SE Profundal	5.9	3.5	1.6	3.4	5.1	11.9
Clingers (%)	square root	NO	0.974	ANOVA	-0.3	Reference Lake 3	10.2	13.6	6.1	0.0	3.9	33.6
						Sheardown SE Profundal	6.7	3.2	1.4	3.4	5.7	11.9
Sprawlers (%)	none	YES	0.016	ANOVA	-1.8	Reference Lake 3	79.3	26.8	12.0	32.7	90.4	100.0
						Sheardown SE Profundal	31.2	23.2	10.4	14.1	19.4	70.8
Burrowers (%)	none	YES	0.003	ANOVA	3.7	Reference Lake 3	10.6	14.1	6.3	0.0	5.6	33.6
						Sheardown SE Profundal	62.2	23.6	10.6	22.4	68.8	82.5

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

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

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

The subtle differences in benthic invertebrate community structure between Sheardown Lake SE and Reference Lake 3 likely reflected marked differences in physical sediment properties between lakes. The key differences in sediment properties between lakes included significantly lower TOC content, significantly greater proportion of silt, and significantly greater sediment compactness (as indicated by lower proportion of moisture) at Sheardown Lake SE compared to Reference Lake 3 (Appendix Table F.40). The occurrence of more stable, compact sediment likely accounted for significantly higher relative abundance of HPG burrowers at Sheardown Lake SE compared to Reference Lake 3 (Tables 4.9 and 4.10). In addition to differences in sediment properties between lakes, significantly shallower 'profundal' sampling depths at Sheardown Lake SE also likely contributed to the differences in benthic invertebrate community features compared to Reference Lake 3 (Appendix Table F.40). Natural depth-related influences on benthic invertebrate community structure that include lower density and richness at greater depth in lake environments are well documented (Ward 1992; Armitage et al. 1995), and were consistently evident at Reference Lake 3 from 2015 to 2018 (Appendix B) suggesting similar patterns in pristine lakes of the Mary River Project region. Notably, the maximum depth of Sheardown Lake SE is approximately 14 m (Minnow 2018). Because profundal habitat for the Mary River Project CREMP is defined as water depths ≥12 m, benthic invertebrate community data collected from profundal depths of Sheardown Lake SE (average station depth of 12.4 m; Appendix Table F.40) are not directly comparable to those collected at the other mine-exposed lakes nor to Reference Lake 3, at which the average profundal sampling depth is ≥ 20 m. Overall, the differences in benthic invertebrate community endpoints between Sheardown Lake SE and Reference Lake 3 likely reflected a combination of naturally greater productivity, naturally more compact sediment with low TOC content, and naturally shallower 'profundal' sampling depths at Sheardown Lake SE. Moreover, no evidence of metal-related influences on the benthic invertebrate community of Sheardown Lake SE were indicated in 2018.

Temporal comparisons indicated no consistent, ecologically significant, differences in general community effect indicators of richness and Simpson's Evenness at littoral or profundal habitats of Sheardown Lake SE between the mine baseline (2007, 2013) and individual years since the commencement of commercial mine operation (2015 to 2018; Figure 4.20; Appendix Tables F.42 and F.43). In addition, no significant differences in benthic invertebrate dominant taxonomic groups or FFG were indicated between mine baseline and mine operational years at littoral or profundal habitats of Sheardown Lake SE (Figure 4.20; Appendix Tables F.42 and F.43). In contrast, significantly lower density has generally occurred at both littoral and profundal habitats of Sheardown Lake SE during individual years of commercial mine operation from 2015 to 2018 compared to mine baseline data collected in 2007 and/or 2013 (Figure 4.20; Appendix Tables F.42 and F.43). Because density was the only benthic invertebrate community metric that differed

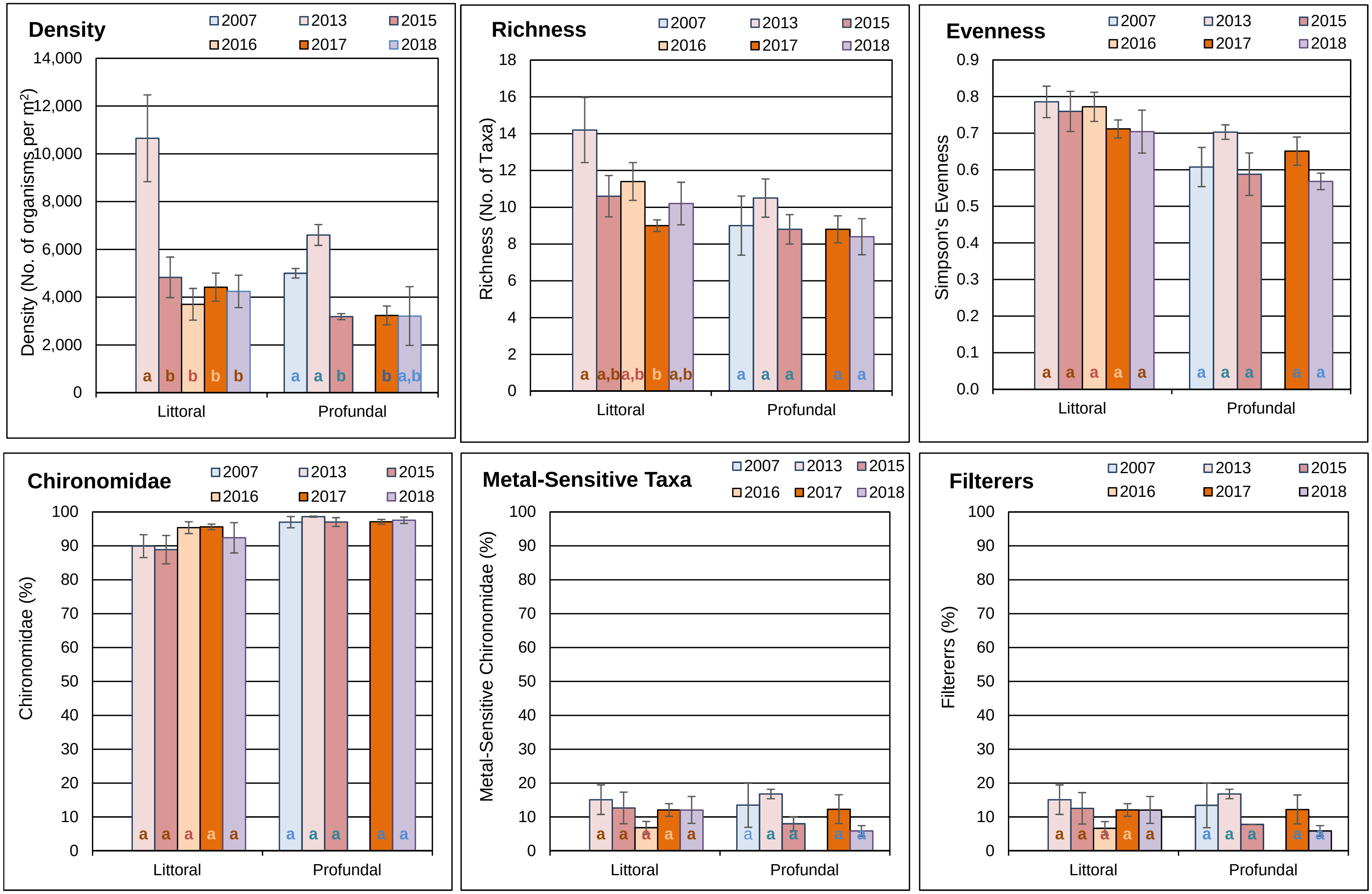


Figure 4.20: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Sheardown Lake SE Littoral and Profundal Study Areas among Mine Baseline (2007, 2013) and Operational (2015 to 2018) Periods

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

significantly between mine-operational and baseline studies at Sheardown Lake SE, natural temporal variability among studies (and in particular, high density during the 2007 baseline study) most likely accounted for the temporal differences in benthic invertebrate density. Overall, consistent with no substantial changes in water and sediment quality since the mine baseline period, no ecologically meaningful changes in benthic invertebrate community features were indicated at littoral and profundal habitat of Sheardown Lake SE following the commencement of commercial mine operation in 2015.

4.3.6 Fish Population

4.3.6.1 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 Reference Lake 3, in 2018 (Table 4.6). However, total fish CPUE was much higher at Sheardown Lake SE than at Reference Lake 3 for 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 higher fish CPUE in 2018 and the three previous mine operational years (i.e., 2015 to 2017) than during the mine baseline studies (2007 and 2008; Figure 4.15). Gill netting CPUE for arctic charr was also higher from 2016 to 2018 compared to the baseline (2006 to 2008), mine construction (2014) and mine operational (2015) studies (Figure 4.15). In part, higher fish CPUE at Sheardown Lake SE in studies conducted from 2016 to 2018 potentially reflected improvements in sampling efficiency gained through experience from previous studies (see Minnow 2016b, 2017, 2018). Nevertheless, the CPUE data suggested that arctic charr abundance at nearshore and littoral/profundal habitats was likely comparable to, or greater than, the abundance of this species during the baseline period at Sheardown Lake SE, indicating no mine-related influences to arctic charr numbers in the lake following the commencement of commercial mine operation in 2015.

4.3.6.2 Sheardown Lake SE Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Sheardown Lake SE nearshore Arctic charr population were assessed based on a control-impact analysis using data collected from Sheardown Lake SE and Reference Lake 3 in 2018. Although before-after analysis of data collected from Sheardown Lake SE in 2018 (mine operation) and 2007 (baseline) was conducted (Appendix Table G.7), poor accuracy in fresh body weight measures during baseline sampling precluded meaningful data interpretation, and therefore these results were not discussed further herein. A total of 99 and 100 arctic charr were captured at nearshore habitat of Sheardown Lake SE and Reference Lake 3, respectively, in August 2018 for the control-impact analysis. Distinguishing arctic charr YOY from the older, non-YOY age category was possible using a fork length cut-off of 4.7 cm and 4.5 cm for Sheardown Lake SE and Reference Lake 3 data sets, respectively, based on evaluation of length-frequency distributions coupled with supporting age determinations (Figure 4.21). 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. However, because the YOY data set for Reference Lake 3 was small (i.e., 8 YOY individuals), a greater degree of caution is warranted around conclusions drawn from the analysis of YOY endpoints.

Length-frequency distributions for the nearshore arctic charr differed significantly between Sheardown Lake SE and Reference Lake 3 (Table 4.11), potentially reflecting the combination of greater prevalence of YOY and larger size of individuals within YOY and non-YOY age classes at Sheardown Lake SE (Figure 4.21). Arctic charr in YOY and non-YOY age classes were significantly longer and heavier at the Sheardown Lake SE nearshore than at the Reference Lake 3 nearshore (Table 4.11; Appendix Table G.20). The occurrence of significantly larger YOY suggested faster arctic charr growth at Sheardown Lake SE than at Reference Lake 3 in 2018. Although condition of nearshore arctic charr YOY was significantly greater at Sheardown Lake SE compared to Reference Lake 3, the condition of arctic charr non-YOY was significantly lower at Sheardown Lake SE (Table 4.11; Appendix Table G.20). For each age class, the magnitude of difference in condition was just outside of the CES_C of ±10% suggesting that these differences were ecologically meaningful (Table 4.11; Appendix Table G.20). Temporal comparisons indicated no consistent differences in nearshore non-YOY arctic charr size or condition between Sheardown Lake SE and Reference Lake 3 from 2015 to 2018 (Table 4.11). In turn, this suggested that the differences in nearshore non-YOY arctic charr size and condition between Sheardown Lake SE and Reference Lake 3 reflected natural variability between study lakes over time. Overall, no adverse effects on the health of arctic charr fish collected at the Sheardown Lake SE nearshore were indicated since commercial mine operations commenced in 2015.

Littoral/Profundal Arctic Charr

Mine-related influences on the Sheardown Lake SE littoral/profundal arctic charr population were assessed based on a control-impact analysis using 2018 data collected at Sheardown Lake SE and Reference Lake 3, and based on a before-after analysis using data collected at Sheardown Lake SE in 2018 and during baseline characterization studies (2006 and 2008 combined data).

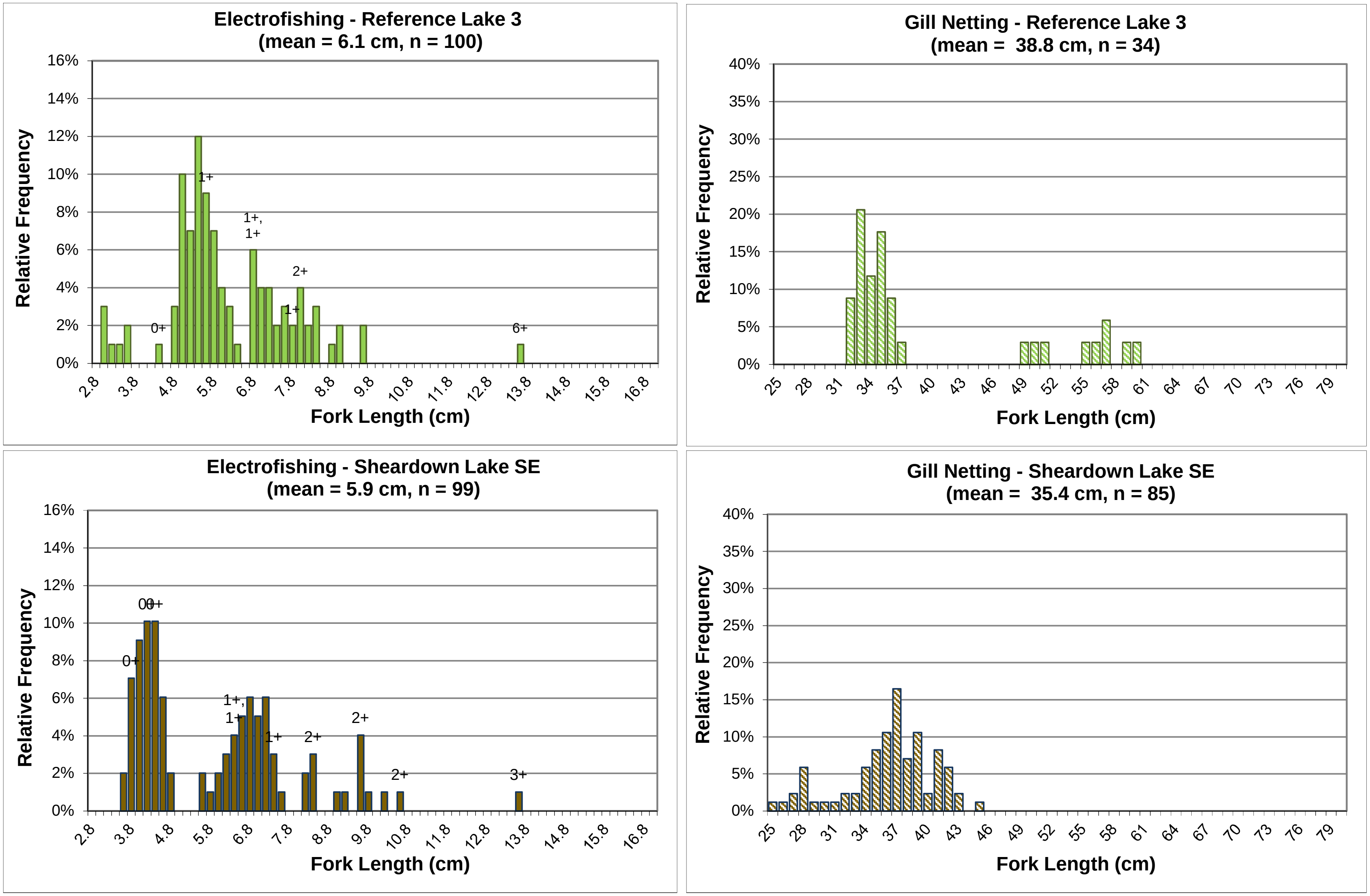


Figure 4.21: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2018

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

Table 4.11: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake SE and Reference Lake 3 from 2015 to 2018, and between Sheardown Lake SE Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a] versus Reference Lake 3				versus Sheardown Lake SE baseline period data [b]			
			2015	2016	2017	2018	2015	2016	2017	2018
Nearshore Electrofishing	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	**Yes (+273%)**	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	No	No	**Yes (+12%)**	**Yes (+21%)**	**Yes (+7%)**	**Yes (-15%)**	**Yes (+19%)**	**Yes (-47%)**
		Size (mean weight)	No	No	**Yes (+55%)**	**Yes (+59%)**	No	**Yes (-43%)**	**Yes (+54%)**	No
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+4%)**	No	**Yes (+9%)**	**Yes (-13%)**	**Yes (-14%)**	**Yes (-16%)**	No	**Yes (-15%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	-	-	-	-	**Yes (-13%)**	No	No	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (-9%)**	**Yes (-7%)**	**Yes (-5%)**	**Yes (-4%)**
		Size (mean weight)	-	-	-	No	**Yes (-26%)**	**Yes (-20%)**	**Yes (-16%)**	**Yes (-16%)**
		Growth (fork length-at-age)	-	-	-	-	No	No	No	-
		Growth (weight-at-age)	-	-	-	-	**Yes (+18%)**	**Yes (+24%)**	No	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+7%)**	No	No	**Yes (-6%)**	**Yes (-7%)**

[a] Values in parentheses indicate direction and magnitude of any significant differences.
[b] Baseline period data included 2007 nearshore electrofishing data and 2007 and 2008 littoral/profundal gill netting data.
[c] Due to low catches of arctic charr in gill nets at Reference Lake 3 in 2015, 2016, and 2017, no comparison of fish health was conducted for gill netted fish.

A total of 85 and 34 arctic charr were sampled from littoral/profundal habitat of Sheardown Lake SE and Reference Lake 3, respectively, in August 2018, for the control-impact analysis. The length-frequency distribution for littoral/profundal arctic charr differed significantly between lakes (Table 4.11; Figure 4.20). However, no significant differences in mean length or weight of littoral/ profundal arctic charr were indicated between Sheardown Lake SE and Reference Lake 3 in 2018 (Table 4.11; Appendix Table G.24). In addition, although condition of arctic charr captured at littoral/profundal areas of Sheardown Lake SE was significantly greater than at Reference Lake 3, the absolute magnitude of this difference was less than the CES_C of 10% suggesting that the difference in arctic charr condition between lakes was not ecologically meaningful (Table 4.11; Appendix Table G.24).

The length-frequency distribution of arctic charr captured at littoral/profundal habitat of Sheardown Lake SE differed significantly between 2018 and the baseline period (Table 4.11). In part, the difference in length-frequency distributions may have reflected significantly smaller size (i.e., weight and length) of individuals captured in 2018 compared to the baseline period (Table 4.11; Appendix Table G.24). Although the condition of arctic charr sampled from littoral/profundal habitat of Sheardown Lake SE was significantly lower in 2018 compared to the baseline period, the magnitude of this difference was within the ecologically meaningful CES_C of ±10% (Table 4.11). Temporal comparisons indicated that arctic charr sampled at littoral/profundal habitat of Sheardown Lake SE have consistently been significantly shorter and lighter during years of mine operation from 2015 to 2018 compared to the mine baseline period, but significant differences in condition only occurred in 2017 and 2018 compared to the mine baseline studies (Table 4.11). Notably, the difference in arctic charr condition in both 2017 and 2018 compared to the baseline period was not ecologically meaningful based on the magnitude of difference within the CES_C of ±10% (Table 4.11). In turn, this suggested no adverse influences on adult arctic charr at Sheardown Lake SE through the initial four years of mine operation.

4.3.7 Integrated Summary

At Sheardown Lake SE, aqueous concentrations of copper and molybdenum were elevated compared to Reference Lake 3 in 2018, and molybdenum was elevated compared to the baseline period. However, all water quality parameters were observed at concentrations below applicable WQG and AEMP benchmarks in 2018. Similar to the northwest basin, total aluminum and manganese concentrations showed strong positive correlations with turbidity at Sheardown Lake SE in 2018 that, in turn, suggested that these metals were largely bound to/contained in 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 and profundal habitats of Sheardown Lake SE were very similar to average concentrations observed for respective station habitats at Reference Lake 3 in 2018. Mean concentrations of iron and manganese were above SQG and AEMP benchmarks in sediment of Sheardown Lake SE, but concentrations of these metals were also above SQG and/or AEMP benchmarks at Reference Lake 3. Although arsenic, chromium, nickel, and phosphorus concentrations were above AEMP benchmarks at individual littoral and profundal stations, concentrations of these metals were also above AEMP benchmarks at Reference Lake 3. Temporal comparisons indicated that metal concentrations in sediment of Sheardown Lake SE in 2018 were within ranges shown during baseline studies, indicating no substantial mine-related influences on sediment quality over time at Sheardown Lake SE.

Chlorophyll-a concentrations at Sheardown Lake SE were significantly higher than at Reference Lake 3 in 2018 suggesting greater primary production at Sheardown Lake. However, chlorophyll-a concentrations remained well below the AEMP benchmark during all seasonal sampling events in 2018 at Sheardown Lake SE, and suggested oligotrophic conditions typical of Arctic waterbodies. Temporal evaluation of the chlorophyll-a data indicated no changes to the trophic status of Sheardown Lake SE since commencement of commercial mine operations. The benthic invertebrate community of Sheardown Lake SE showed significantly higher density and richness, but no ecologically significant differences in Simpson's Evenness and relative abundance of dominant groups including metal-sensitive chironomids, compared to Reference Lake 3 in 2018. In addition, no ecologically significant differences in benthic invertebrate density, richness, Simpson's Evenness, and relative abundance of dominant taxonomic groups or FFG were consistently shown from 2015 to 2018 compared to years in which mine baseline data were collected at Sheardown Lake SE. The size of the arctic charr population was greater at Sheardown Lake SE compared to Reference Lake 3 in 2018, but similar numbers of arctic charr were present at Sheardown Lake SE in 2018 compared to the baseline period. Arctic charr YOY and non-YOY captured at nearshore habitat of Sheardown Lake SE showed significantly higher and lower condition, respectively, than those captured at Reference Lake 3 in 2018. However, no consistent differences in nearshore non-YOY arctic charr condition was indicated between Sheardown Lake SE and Reference Lake 3 from 2015 to 2018, suggesting that the differences in nearshore non-YOY arctic charr condition reflected natural variability between study lakes over time. No ecologically significant differences in the condition of arctic charr captured at littoral/ profundal habitat were indicated between Sheardown Lake SE and Reference Lake 3 in 2018, nor at Sheardown Lake SE between 2018 and the mine baseline period, indicating no adverse effects on the health of arctic charr at Sheardown Lake SE. 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 SE in the fourth year of mine operation at the Mary River Project.

5 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 to C.3). Although DO concentrations differed significantly among the Mary River benthic study areas, higher DO concentrations were shown downstream compared to upstream of the mine at the time of biological sampling in August 2018, and concentrations were consistently well above WQG acceptable levels for sensitive life stages of cold-water biota (i.e., 9.5 mg/L) at all times (Figure 5.1; Appendix Figure C.21; Appendix Table C.61). This suggested that slight differences in DO concentrations among the 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 during the summer and fall sampling events, but were higher (more alkaline) at and downstream of the mine compared to the GO-09 series river reference stations during the spring sampling event in 2018 (Figure 5.1; Appendix Tables C.1 to C.3). Highest pH was generally observed at Mary River Tributary-F (i.e., Station FO-01) in each season (Figure 5.1). Because Mary River Tributary-F runs adjacent to the pit mine haul road, the occurrence of highest pH suggested a mine-related influence on this tributary. Nevertheless, 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.61). Specific conductance was consistently lowest in spring and highest in fall at all stations, which likely was a reflection of natural seasonal differences related to the relative proportion of flow from surface runoff (e.g., spring snowmelt). Spatially, specific conductance was slightly higher at Mary River water quality stations located downstream than upstream of the Mary River Tributary-F confluence during summer and fall sampling events, but not during the spring sampling event in 2018 (Figure 5.1). Similar to patterns in pH, highest specific conductance was consistently observed at the Mary River Tributary-F water monitoring station, suggesting that this tributary may be the primary receiver for mine-related inputs within the Mary River system. However, natural differences in base material geology of the Mary River Tributary-F watershed compared to that of Mary River may also contribute to the differences in specific conductance observed between these watercourses.

Water chemistry within Mary River showed no distinct and/or consistent spatial gradients with progression downstream from the GO-09 series river reference stations during any of the spring,

Figure 5.1: **Comparison of *In Situ* Water Quality Variables Measured at Mary River Water Quality Monitoring Stations in Spring, Summer, and Fall 2018, Mary River Project CREMP**

summer, or fall sampling events in 2018 with the exception of concentrations of manganese and sulphate (Table 5.1; Appendix Table C.62). In general, parameter concentrations at Mary River stations located adjacent to or downstream of the mine (EO and CO series stations) were similar to concentrations observed at the upstream river reference stations (GO-09 series stations) during each respective sampling event (Table 5.1; Appendix Tables C.62 to C.65). However, total and dissolved concentrations of manganese were distinctly elevated at Station EO-03 compared to stations located upstream in the Mary River during the summer and fall sampling events (Table 5.1; Appendix Tables C.62 to C.65). In addition, a generally decreasing gradient in manganese concentrations in Mary River was observed with distance downstream from Station EO-03, indicating a clear source of manganese to Mary River between Stations EO-10 and EO-03 (Table 5.1; Appendix Tables C.62 to C.65). Sulphate concentrations were also distinctly elevated at Mary River stations located downstream of the Mary River Tributary-F confluence during the summer and fall sampling events in 2018 (Table 5.1; Appendix Tables C.62 and C.63). Because highest sulphate concentrations were consistently observed at Mary River Tributary-F in 2018 (Appendix Tables C.62 and C.63), this tributary was clearly an important source of sulphate to Mary River.

Total aluminum concentrations were above WQG at a number of Mary River mine-exposed stations, but were typically below the applicable AEMP benchmark, during the spring, summer and fall monitoring events in 2018 (Table 5.1; Appendix Table C.62). However, total concentrations of aluminum were also elevated above applicable WQG at one or more of the Mary River GO series reference stations during the spring, summer, and fall monitoring events in 2018 (Appendix Table C.62), suggesting naturally high concentrations of aluminum in the Mary River system.[12] Phenol concentrations were above WQG at Mary River mine-exposed stations EO-03 and CO-01 during the spring sampling event, but because phenol concentrations were also above WQG at the upstream-most reference station GO-09A in spring, phenol concentrations above WQG at Mary River were not likely attributable to mine operations (Appendix Table C.62).

Temporal evaluation of Mary River water chemistry data indicated that parameter concentrations during the fall sampling event in 2018 were generally within respective parameter concentration ranges measured at each station during the mine baseline period (2005 to 2013; Figure 5.2; Appendix Figure C.22). Only manganese and sulphate showed higher concentrations in 2018 than during the mine baseline period, as well as a generally increasing trend from mine

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

Table 5.1: Water Chemistry at Mary River Monitoring Stations, Mary River Project CREMP, August 2018

	Parameters	Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creek Average (n = 4)	Mary River Reference Station			Mary River Upstream		MRTF	Mary River Downstream of Mine						
						G0-09-A	G0-09	G0-09-B	G0-03	GO-01	F0-01	E0-10	EO-03	EO-21	EO-20	C0-10	C0-05	CO-01
					Fall 2018	27-Aug-2018	25-Aug-2018	25-Aug-2018	25-Aug-2018	27-Aug-2018	25-Aug-2018	27-Aug-2018	26-Aug-2018	27-Aug-2018	27-Aug-2018	27-Aug-2018	27-Aug-2018	27-Aug-2018
Conventionals	Conductivity (lab)	umho/cm	-	-	97	134	113	112	121	134	283	177	153	164	163	165	160	149
	pH (lab)	pH	6.5 - 9.0	-	7.88	7.99	8.07	8.14	8.02	8.00	8.29	8.03	8.03	8.05	8.04	8.07	8.01	8.11
	Hardness (as $CaCO_3$)	mg/L	-	-	47	66	55	55	57	65	143	84	72	80	79	81	78	70
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	3.2	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	53	55	70	70	60	80	160	81	115	75	80	90	95	75
	Turbidity	NTU	-	-	2.4	2.7	4.9	3.8	3.2	2.7	1.7	2.7	3.0	2.6	2.3	2.1	2.6	2.2
	Alkalinity (as $CaCO_3$)	mg/L	-	-	43	65	50	52	50	59	95	61	58	59	62	65	64	60
Nutrients and Organics	Total Ammonia	mg/L	variable	0.855	0.021	0.037	0.069	<0.020	<0.020	<0.020	<0.020	0.022	<0.020	<0.020	<0.020	<0.020	0.032	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	0.130	0.049	0.021	0.034	0.047	0.044	0.041	0.026
	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	<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	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.5	1.3	2.0	1.9	1.4	1.4	2.2	1.3	2.0	1.4	1.5	1.5	1.7	1.7
	Total Organic Carbon	mg/L	-	-	2.0	2.1	2.1	2.0	1.9	1.9	2.3	1.8	2.1	1.9	2.0	2.3	2.4	2.6
	Total Phosphorus	mg/L	0.020[α]	-	0.0035	0.0034	0.0073	0.0034	<0.0030	<0.0030	0.0036	<0.0030	<0.0030	<0.0030	<0.0030	<0.0030	0.0030	<0.0030
	Phenols	mg/L	0.004[α]	-	<0.0010	<0.0010	0.0027	<0.0010	0.0011	<0.0010	0.0011	<0.0010	0.0010	<0.0010	<0.0010	<0.0010	0.0012	<0.0010
Anions	Bromide (Br)	mg/L	-	-	0.1	<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	1.6	2.8	2.9	2.8	5.4	5.5	2.5	5.3	5.5	5.4	5.0	5.0	4.9	4.7
	Sulphate (SO_4)	mg/L	218[β]	218	2.7	1.6	1.8	1.7	1.6	1.7	49.1	17.9	8.2	12.7	11.8	11.9	10.6	7.3
Total Metals	Aluminum (Al)	mg/L	0.100	0.966	0.054	0.126	0.155	0.105	0.128	0.139	0.034	0.135	0.123	0.125	0.123	0.093	0.102	0.097
	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	<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	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0059	0.0085	0.0082	0.0080	0.0086	0.0089	0.0132	0.0107	0.0096	0.0101	0.0105	0.0103	0.0099	0.0093
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<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	<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	<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	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	9.8	13.9	11.8	11.7	12.1	14.0	25.8	16.5	15.1	16.2	16.2	15.9	15.0	16.0
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<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	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.0009	0.0008	0.0009	0.0009	0.0008	0.0008	0.0007	0.0008	0.0008	0.0009	0.0009	0.0010	0.0009	0.0008
	Iron (Fe)	mg/L	0.30	0.874	0.046	0.071	0.102	0.082	0.079	0.074	0.042	0.075	0.075	0.075	0.092	0.059	0.085	0.072
	Lead (Pb)	mg/L	0.001	0.001	0.00008	0.00008	0.00011	0.00009	0.00009	0.00007	<0.000050	0.00007	0.00008	0.00008	0.00009	0.00006	0.00007	0.00006
	Lithium (Li)	mg/L	-	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	0.0012	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Magnesium (Mg)	mg/L	-	-	5.3	7.4	6.1	6.5	6.3	7.2	20.4	10.4	8.3	9.2	9.5	9.7	9.3	8.4
	Manganese (Mn)	mg/L	0.935[β]	-	0.0007	0.0009	0.0012	0.0011	0.0010	0.0008	0.0011	0.0009	0.0094	0.0072	0.0074	0.0057	0.0064	0.0040
	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	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00029	0.00025	0.00029	0.00026	0.00024	0.00030	0.00032	0.00029	0.00045	0.00046	0.00039	0.00044	0.00042	0.00045
	Nickel (Ni)	mg/L	0.025	0.025	0.00053	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	0.00051	0.00054	0.00068	0.00092	0.00078	0.00072
	Potassium (K)	mg/L	-	-	0.65	0.91	0.95	0.97	0.87	0.92	1.26	0.99	0.98	0.99	1.00	0.97	0.99	0.96
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.83	0.96	0.94	0.85	0.96	0.97	0.70	1.01	0.97	0.94	1.00	0.93	0.94	0.93
	Silver (Ag)	mg/L	0.00025	0.0001	<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	<0.000010
	Sodium (Na)	mg/L	-	-	1.2	2.0	2.0	2.0	1.7	1.9	1.5	1.8	1.7	1.9	1.9	1.7	1.8	1.7
	Strontium (Sr)	mg/L	-	-	0.0094	0.0128	0.0127	0.0129	0.0108	0.0127	0.0199	0.0148	0.0158	0.0159	0.0154	0.0152	0.0147	0.0141
	Thallium (Tl)	mg/L	0.0008	0.0008	<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	<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	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	<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	<0.010
	Uranium (U)	mg/L	0.015	-	0.0019	0.0026	0.0025	0.0027	0.0020	0.0021	0.0023	0.0023	0.0020	0.0022	0.0019	0.0019	0.0017	0.0017
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<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	<0.0030	<0.0030

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

[b] AEMP Water Quality Benchmarks developed by Intrinsik (2013) using baseline water quality data specific to Mary River

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

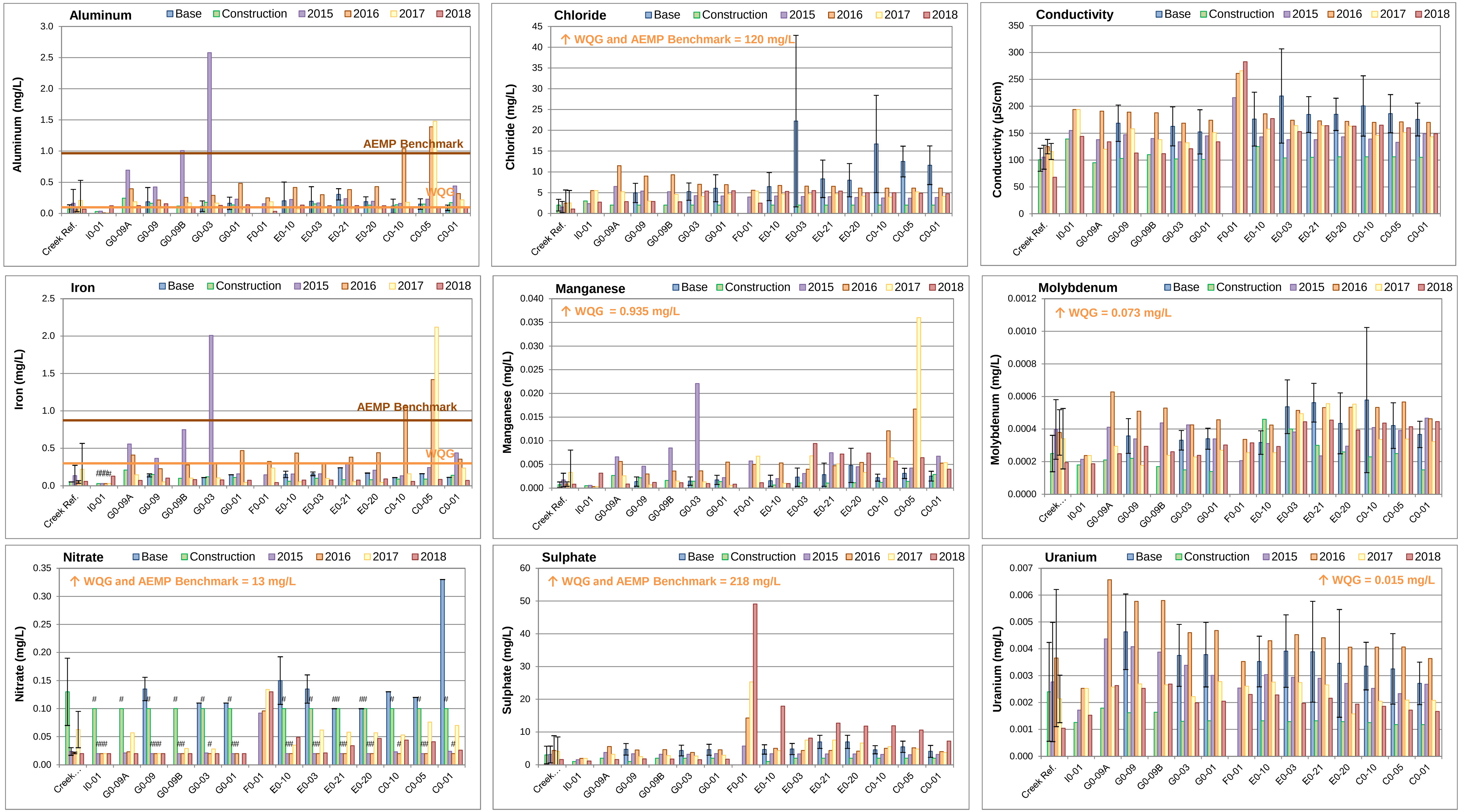


Figure 5.2: Temporal Comparison of Water Chemistry at Mary River Stations for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2018) Periods in the Fal

Notes: Values represent mean ± SD. Creek reference includes the 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 Guidelines (WQG) AEMP Benchmarks are specific to Mary River.

construction (2014) through operational phases (i.e., 2015 to 2018), suggesting that mine operations have contributed to elevated concentrations of these parameters in Mary River waters (Figure 5.2; Appendix Figure C.22). Despite higher concentrations of manganese and sulphate over time at Mary River water quality stations located downstream of the mine, concentrations of both parameters remained well below applicable WQG and AEMP benchmarks in and prior to 2018 (Figure 5.2).

5.1.2 Phytoplankton

Chlorophyll-a concentrations at Mary River stations located downstream of the mine were generally within the range of, or slightly higher, than the GO series river reference stations and/or creek reference stations during the 2018 spring, summer, and/or fall sampling events (Figure 5.3). Chlorophyll-a concentrations at Mary River Tributary-F (MRTF; Station FO-01), which receives treated effluent discharge from the mine, were also comparable to seasonal average concentrations observed at the reference stations (Figure 5.3). Chlorophyll-a concentrations were consistently well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events at all Mary River and MRTF sampling stations in 2018, and were suggestive of low (i.e., oligotrophic) phytoplankton productivity based on Dodds et al (1998) trophic status classification for stream environments. These results suggested no adverse mine-related influences on phytoplankton abundance at Mary River or MRTF in 2018. Low to moderate phytoplankton productivity was expected for Mary River reference and mine-exposed stations in 2018 given oligotrophic to mesotrophic productivity categorizations based on CWQG classifications that use total phosphorus concentrations to define trophic status (Table 5.1; Appendix Table C.62).

Temporal comparisons of the Mary River chlorophyll-a data suggested that concentrations were generally lower at stations located downstream of the mine sewage treatment plant outfall (i.e., EO-21, EO-20, and CO series stations) in 2018 and during each of the three previous years of mine operation (2015 to 2017) than those observed during the baseline period (Figure 5.4). Notably, baseline period chlorophyll-a concentrations at these same stations were considerably higher than at the reference and mine-exposed stations located upstream for this same period (Figure 5.4). Some of the variability in chlorophyll-a concentrations at Mary River EO-21, EO-20 and CO series stations among baseline and commercial mine operation years may have reflected natural differences in turbidity affecting the amount of light energy available to phytoplankton as opposed to responses related to exposure to metals, nutrient enrichment, or other potential mine-related influences on phytoplankton productivity (Minnow 2017). Changes in chlorophyll-a concentrations at Mary River stations located downstream of the mine among individual years of commercial mine operation (2015 to 2018) and the baseline period were consistent with natural

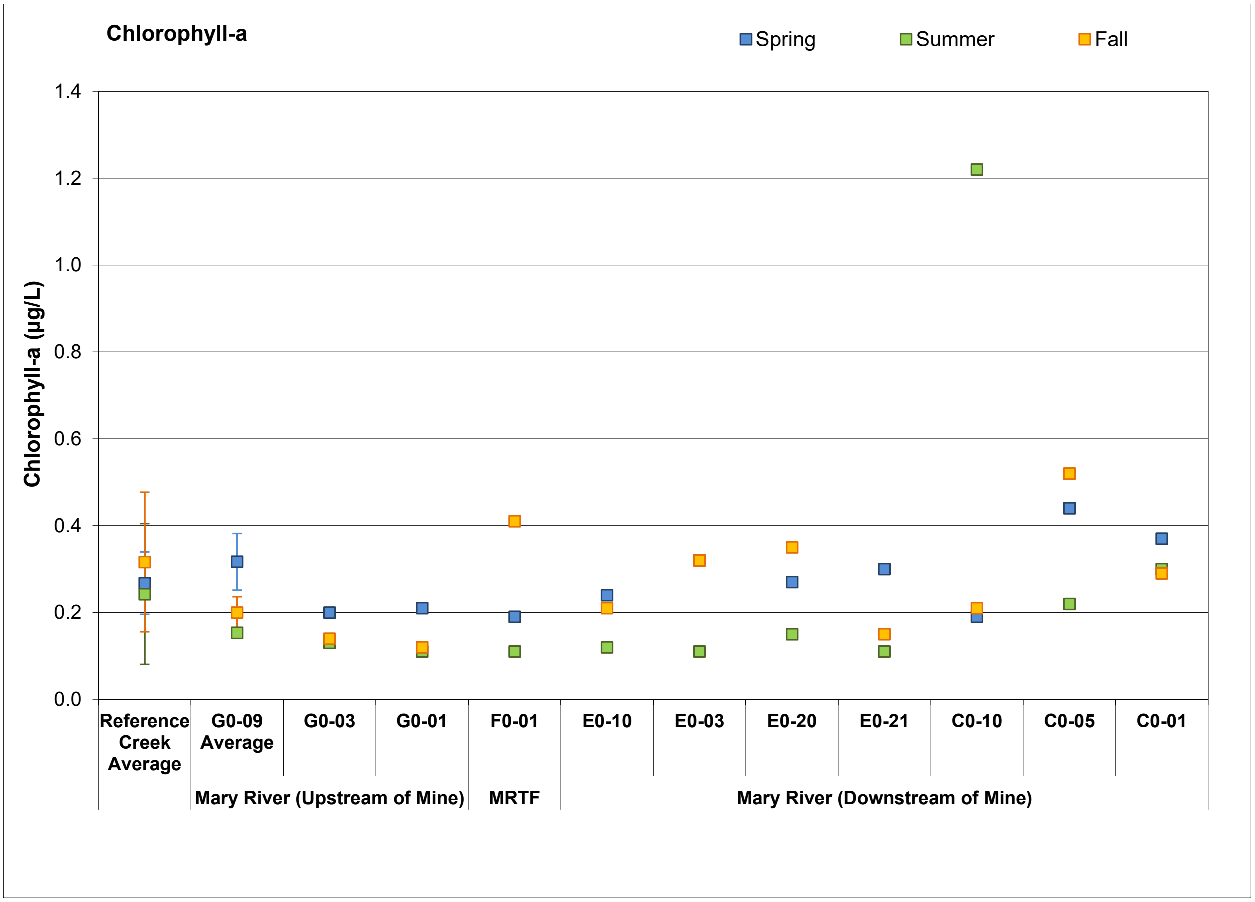


Figure 5.3: Chlorophyll-a Concentrations at Mary River Phytoplankton Monitoring Stations Located Upstream and Downstream of the Mine, Mary River Project CREMP, 2018

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

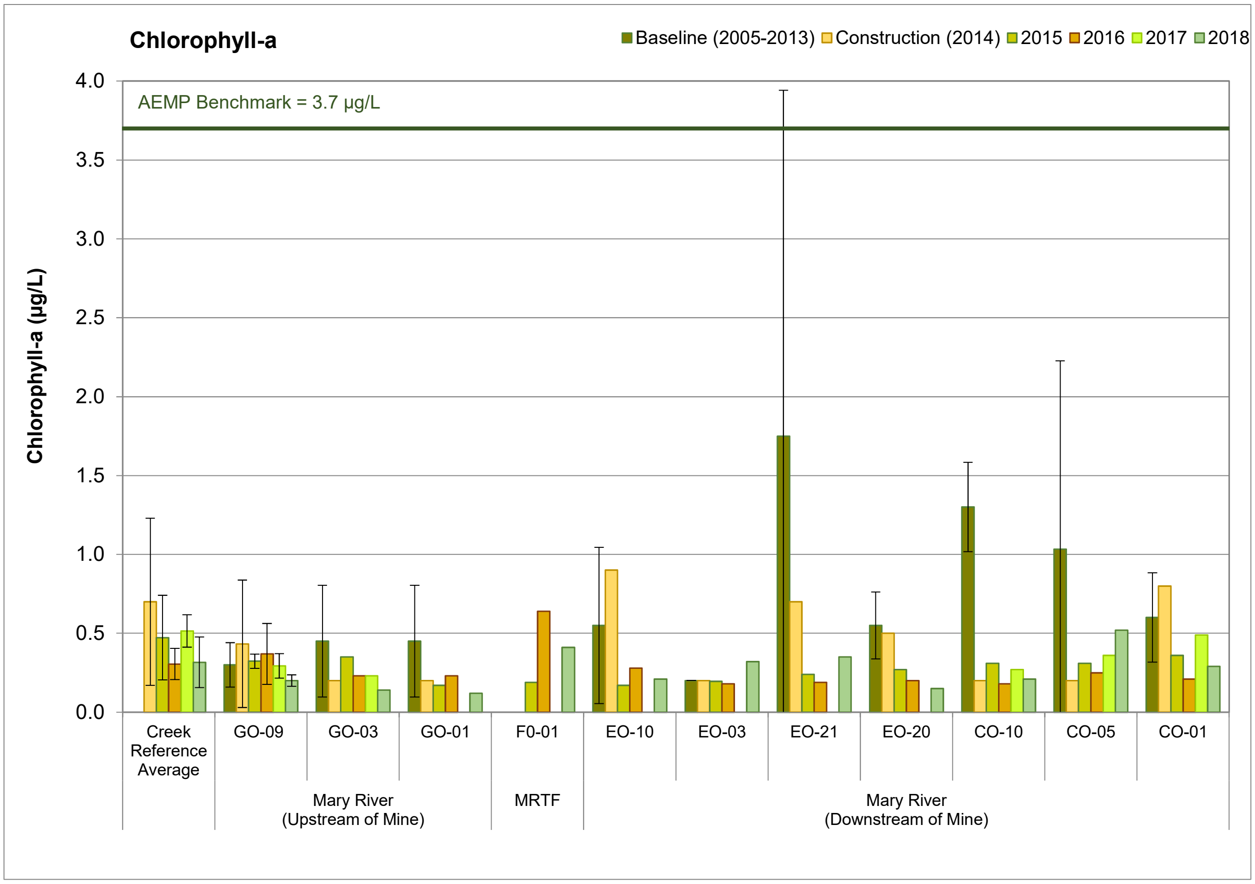


Figure 5.4: Temporal Comparison of Chlorophyll-a Concentrations at Mary River Stations for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2018) Periods during the Fall

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

differences in turbidity (i.e., originating from sources upstream of the mine), but may also have reflected influences from sources currently unidentified.

5.1.3 Benthic Invertebrate Community

The Mary River benthic invertebrate community assessment included a spatial statistical analysis of endpoints among two upstream reference areas (GO-09, GO-03), two near-field mine-exposed areas located in close proximity to the mine (EO-01, EO-20), and a far-field cumulative effects mine-exposed area located well downstream of the mine (CO-05; see Table 2.5, Figure 2.4). Benthic invertebrate density, richness, and Simpson's Evenness at the Mary River upper mine-exposed study area EO-01 did not differ significantly from the GO-09 reference area in 2018 (Figure 5.5). Of these endpoints, only richness was significantly lower at the Mary River middle mine-exposed study area EO-20 compared to the GO-09 reference area (Figure 5.5). In contrast, significantly higher density and richness, and significantly lower Simpson's Evenness was

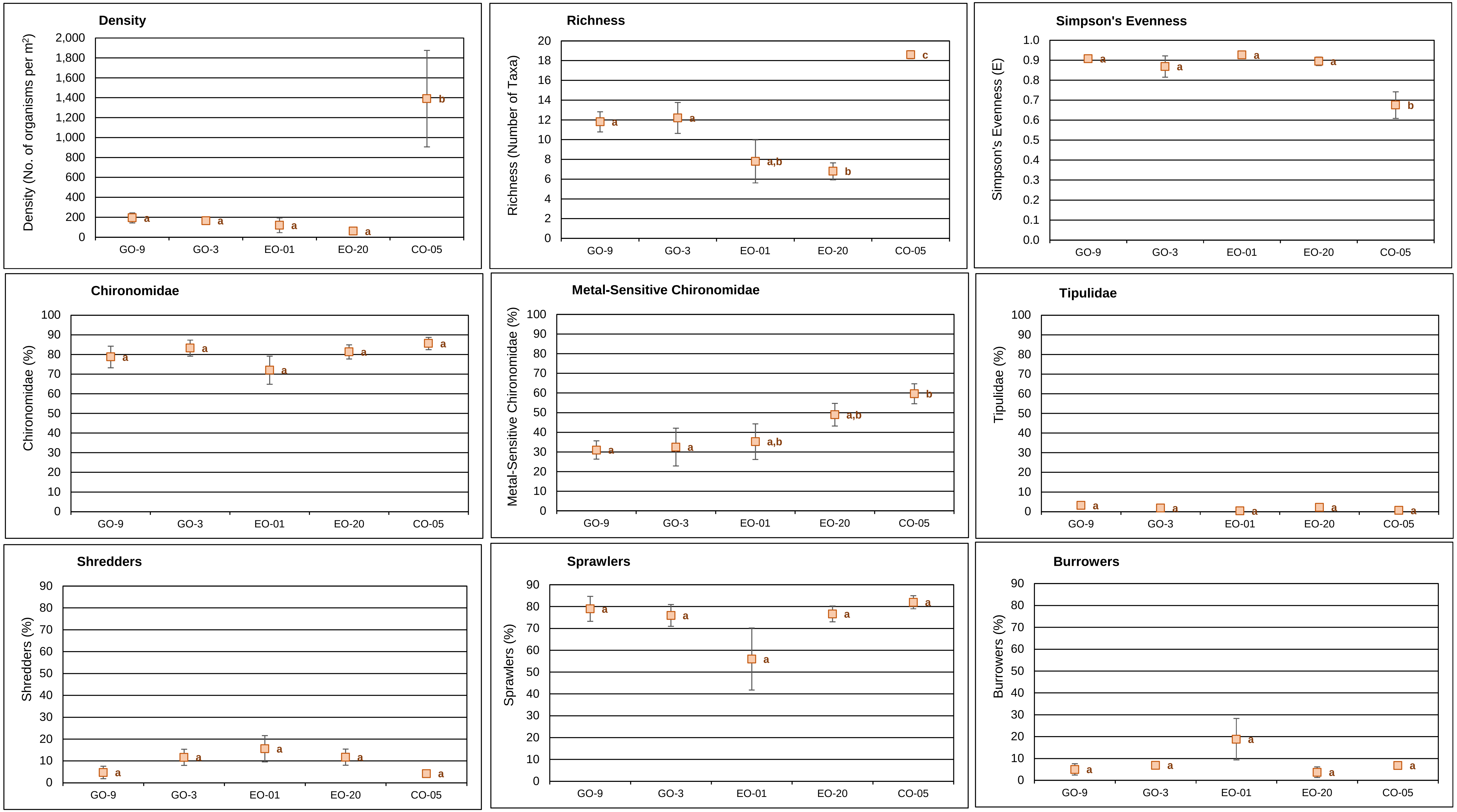


Figure 5.5: **Comparison of Benthic Invertebrate Community Metrics among Mary River Study Areas (mean ± SE), Mary River Project CREMP, August 2018**

Notes: The same letter(s) next to data points indicates no significant difference between/among study areas.

indicated at the Mary River far-field mine-exposed study area CO-05 compared to the reference area (Figure 5.5). High numbers of the midge genus *Pseudokiefferiella*, which characteristically inhabits clean, cool, arctic-alpine lotic environments (Doughman 1983), were present at CO-05 (Appendix Table F.51). The disproportionately high numbers of this midge resulted in higher density and lower Simpson's Evenness of benthic invertebrates at this study area compared to the upstream reference area and other Mary River mine-exposed study areas (Appendix Table F.53). Notably, the occurrence of higher benthic invertebrate density and greater richness at CO-05 potentially reflected a slight nutrient enrichment-related effect associated with the mine, which discharges treated sewage effluent to the Mary River near the confluence with the Sheardown Lake SE outlet.

In addition to the differences indicated above, Bray-Curtis Index at the near- and far-field mine-exposed areas of Mary River differed significantly from the upstream GO-09 reference area (Appendix Table F.53). However, no significant differences in dominant taxonomic groups, FFG, or HPG were indicated between the Mary River mine-exposed and reference study areas (Figure 5.5; Appendix Table F.53). The lack of differences in FFG among Mary River study areas suggested no mine-related influences to aquatic food resources available to benthic invertebrates. Similarly, the absence of differences in HPG among Mary River study areas suggested no adverse mine-related influences to physical habitat features (e.g., sedimentation) adjacent to or downstream of the mine. In addition, no adverse significant differences in the relative abundance of metal-sensitive taxa were indicated between Mary River mine-exposed and reference areas (Figure 5.5), suggesting no adverse mine-related influences to benthic invertebrates associated with metal concentrations. Therefore, differences in benthic invertebrate community structure that included differing density, richness, Simpson's Evenness, and Bray-Curtis Index between Mary River mine-exposed and reference areas in 2018 did not appear to be mine-related but rather, likely reflected natural variability among study areas.

Temporal comparison of the Mary River benthic invertebrate community data indicated no consistent ecologically significant differences in density and richness between mine operational (2015 to 2018) and baseline (2006 to 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.56 to F.58). Simpson's Evenness at the Mary River upper mine-exposed study area EO-01 was continually significantly higher during years of mine operation than during the 2007 mine baseline study, but all other indices did not differ significantly among the years of mine operation and mine baseline at this study area (Appendix Table F.56). At middle mine-exposed study area EO-20, significantly lower and higher relative abundance of chironomids and the collector-gatherer FFG, respectively, generally occurred during years of mine operation compared to mine baseline data collected in 2011 (Appendix Table F.57). At far-field mine-exposed area CO-05, despite several endpoints differing

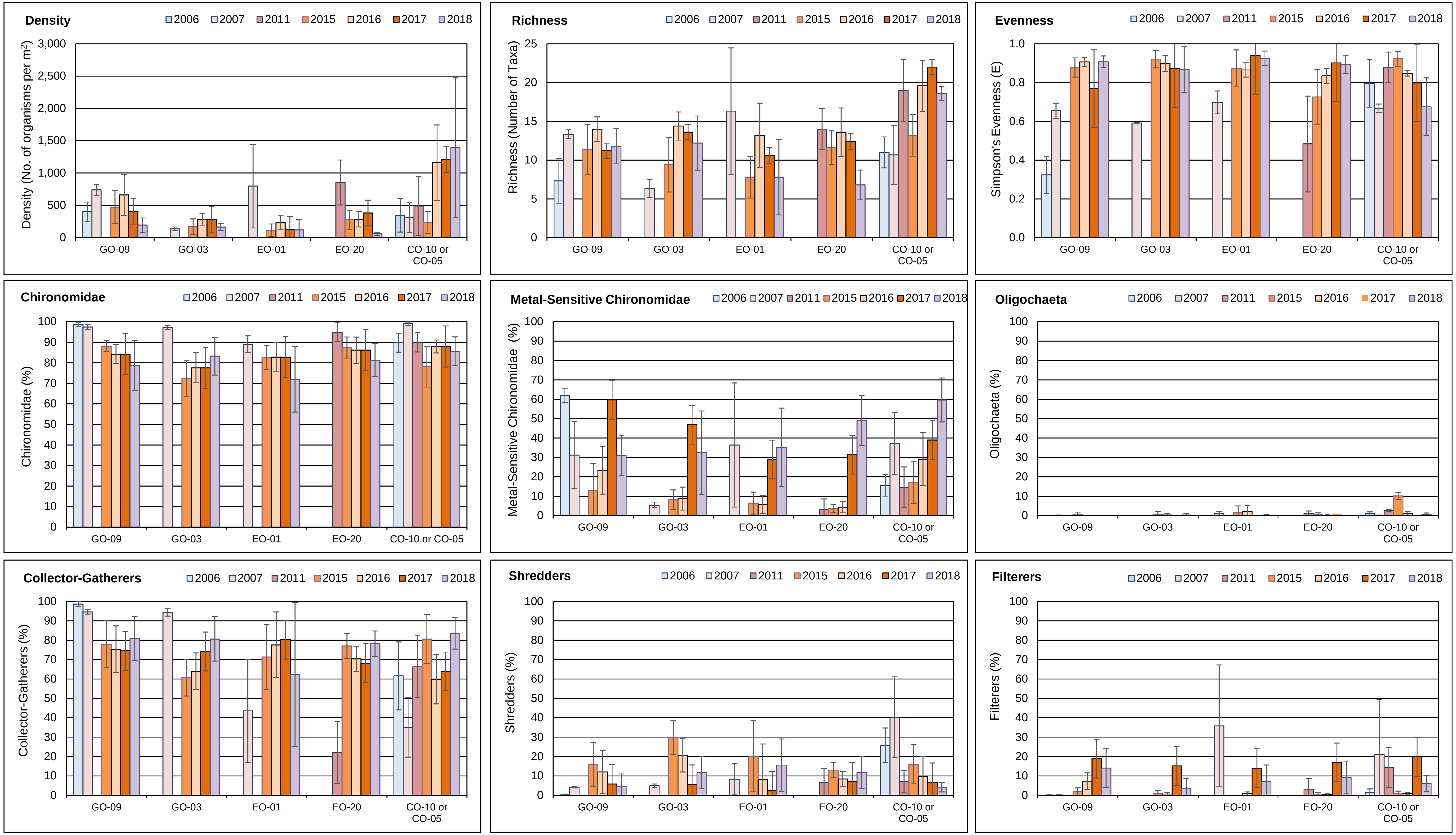


Figure 5.6: Comparison of Benthic Invertebrate Community Metrics (mean ± SD) at Mary River Study Areas among Mine Baseline (2006, 2007, 2011) and Operational (2015 to 2018) Years for the Mary River Project CREMP

significantly among the four mine-operational years and two mine baseline years, none of the endpoints showed consistent significant differences between the mine-operational and baseline periods (Appendix Table F.58). Notably, for those benthic invertebrate community metrics that differed significantly between years of mine-operation and baseline at the Mary River mine-exposed study areas, similar types, direction, and magnitude of these differences were generally observed at the Mary River upstream GO-09 and GO-03 reference areas between years of mine operation and baseline (Appendix Tables F.54 and F.55). In turn, this suggested that the differences in these metrics at Mary River areas over time reflected natural temporal variability and/or sampling artifacts of the CREMP (e.g., changes in sampling location, personnel collecting samples, etc.). In addition, temporal comparison of the data at each individual mine-exposed area indicated no cumulative temporal influences on benthic invertebrates of the Mary River since the commencement of commercial mine operations in 2015.

5.1.4 Integrated Summary

Mine-related influences on water quality of Mary River in 2018 included slight elevation of manganese and sulphate concentrations at mine-exposed areas compared to the upstream reference area, as well as to baseline data. Although total aluminum concentrations were above WQG at one or more Mary River mine-exposed stations in 2018, the elevation in aluminum appeared to be associated with naturally high turbidity within Mary River. Aqueous concentrations of all other parameters were well below WQG and AEMP benchmarks at the Mary River mine-exposed stations in 2018, with no indication of increasing concentrations over time. Chlorophyll-a concentrations were similar among the ten Mary River phytoplankton monitoring stations, with no significant differences in annual chlorophyll-a concentrations indicated between the Mary River mine-exposed and reference stations. Although lower chlorophyll-a concentrations were indicated at individual Mary River stations in 2018 compared to the baseline period, these differences likely reflected natural differences in turbidity among years, which would be expected to affect phytoplankton productivity by affecting the amount of light available for photosynthesis. The most notable differences in benthic invertebrate community endpoints among Mary River mine-exposed and reference areas included significantly higher density and richness, and significantly lower Simpson's Evenness, at the far-field (CO-05) study area. No significant differences in the relative abundance of dominant taxonomic groups, FFG, HPG, or metal-sensitive chironomids were indicated at any of the three Mary River mine-exposed study areas compared to the upstream reference area. In addition, no benthic invertebrate community endpoints differed significantly on a continual basis at ecologically meaningful magnitudes over the four years of mine operation compared to baseline data sets at any of the Mary River mine-exposed areas. Moreover, for those metrics that differed significantly between years of mine operation and baseline at mine-exposed areas, similar differences were observed among these years at the Mary River upstream reference area. Therefore, although the occurrence of greater benthic invertebrate density and richness and lower Simpson's Evenness at Mary River mine-exposed study area CO-05 potentially reflected a slight nutrient enrichment-related effect associated with the mine, it is more likely that natural habitat variability and/or sampling artifacts of the CREMP accounted for these differences in 2018. Overall, the chlorophyll-a and benthic invertebrate community data suggested no adverse mine-related influences to Mary River biota since commercial mine operations commenced in 2015.

5.2 Mary Lake

5.2.1 Hydraulic Retention Time

A hydraulic retention time of 75 ± 29 days was estimated for Mary Lake by Minnow (2018) using mean annual watershed runoff extrapolated from Baffinland flow monitoring stations installed in the primary tributaries of Mary Lake (Tom and Mary rivers) and at small watershed watercourses (i.e., ≤15 km^2) located on the mine property, and a lake volume of 156.35 million cubic metres.

5.2.2 Water Quality

Water quality profiles conducted at Mary Lake in 2018 showed similar values and patterns with depth for *in situ* water temperature, DO concentration, and pH measures at the Mary Lake north and south basins, but higher specific conductance was observed at the north basin throughout the year (Figures 5.7 and 5.8). Water temperatures increased from surface to bottom during the winter, and decreased from surface to bottom during the summer, at each of the north and south basins of Mary Lake in 2018, but in all cases, the water temperature difference between the surface and bottom was insufficient to result in thermal stratification of the water column (Figures 5.7 and 5.8). Mary Lake temperature profiles conducted during the summer monitoring event differed slightly from that at Reference Lake 3, where the trend of decreasing water temperature with increased depth was not as pronounced (Figures 5.7 and 5.8). Although similar water temperature profiles were observed between Mary Lake and Reference Lake 3 during the fall monitoring event (Figures 5.7 and 5.8), the average water temperature at the bottom of the water column at Mary Lake littoral stations was significantly cooler than at Reference Lake 3 (Figure 5.9; Appendix Table C.72). Nevertheless, the incremental difference in average bottom water temperature between lakes at littoral sampling depths was small (i.e., ≤0.5˚C), and thus was unlikely to be ecologically meaningful.

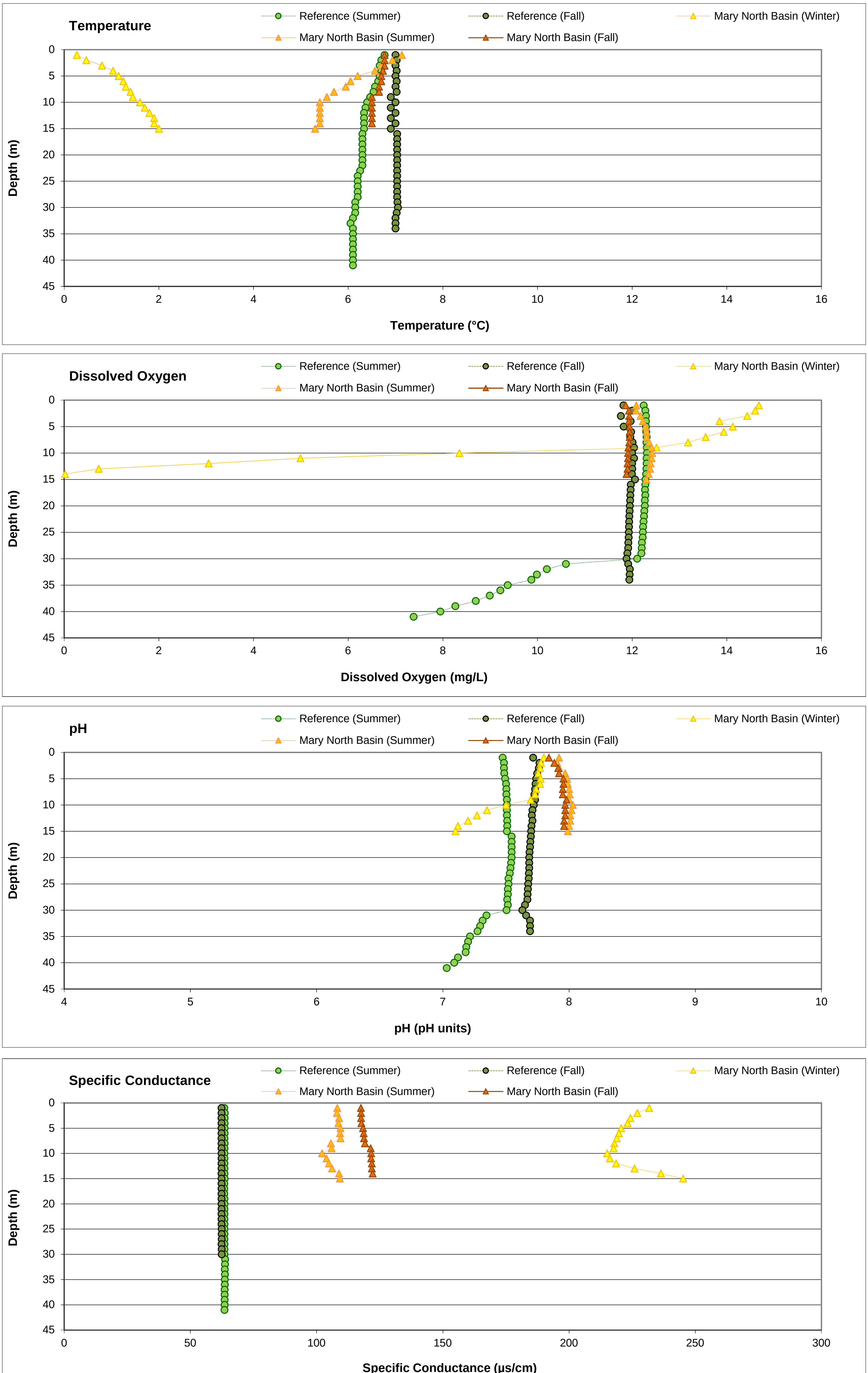


Figure 5.7: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake North Basin (BLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2018

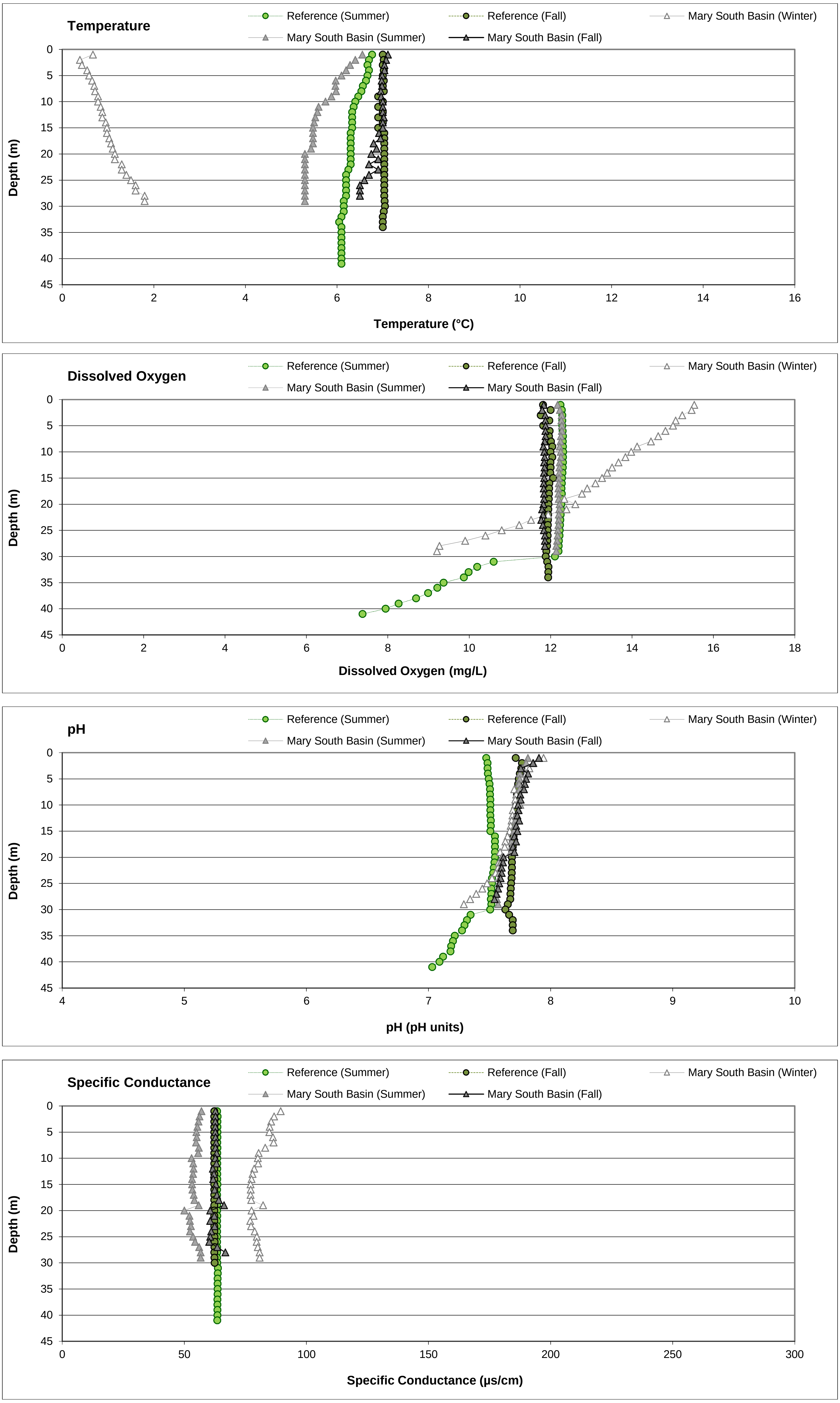


Figure 5.8: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake South Basin (BLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2018

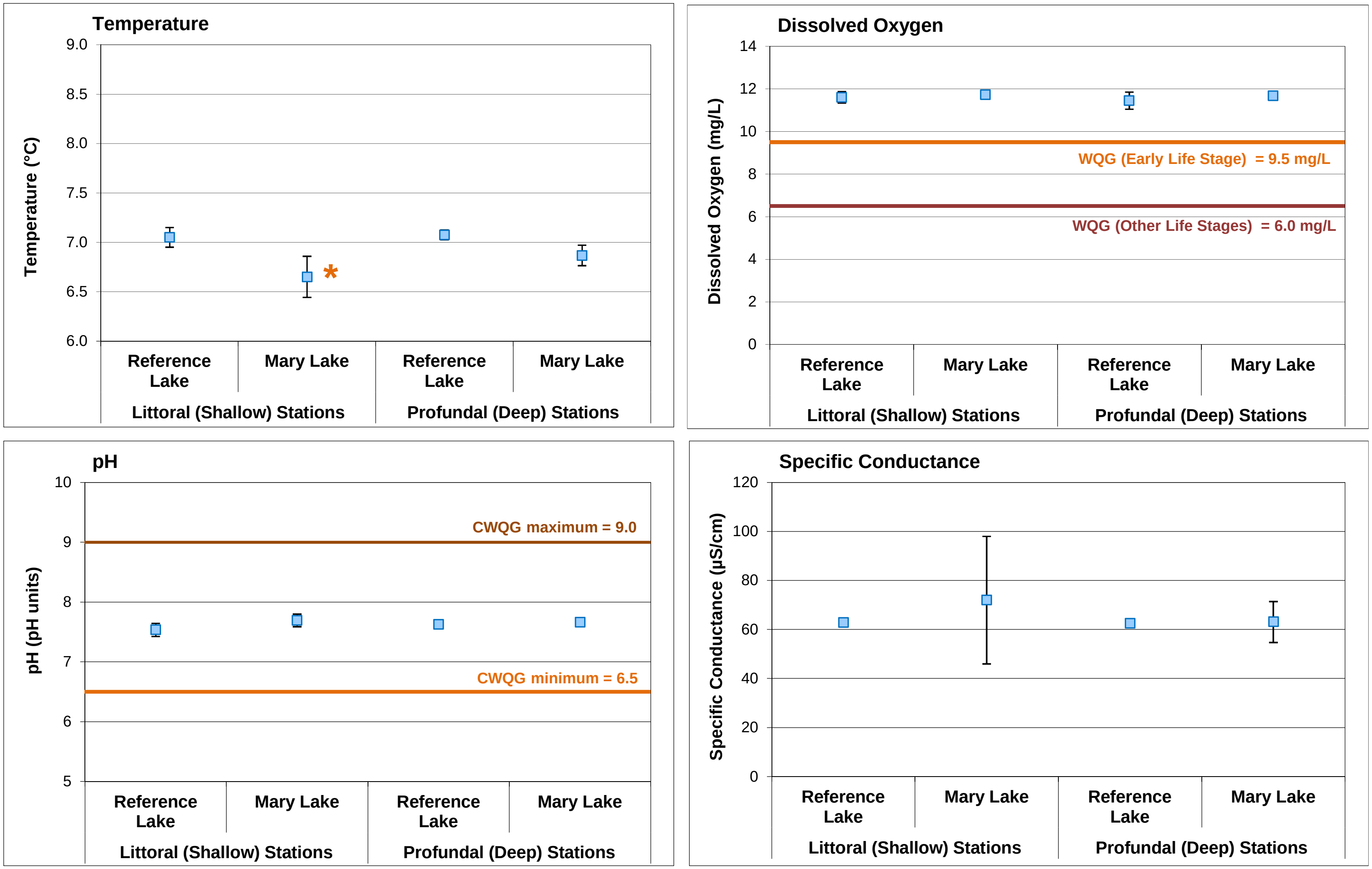


Figure 5.9: Comparison of *In Situ* Water Quality Variables (mean ± SD) Measured at Mary Lake (BLO) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2018

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

Dissolved oxygen profiles conducted at Mary Lake in 2018 indicated the development of a strong oxycline extending through the entire water column at both the north and south basins in winter (Figures 5.7 and 5.8). However, similar to Reference Lake 3, no oxycline development was apparent in the summer or fall of 2018 at either Mary Lake basin. Dissolved oxygen concentrations at Mary Lake were above WQG acceptable levels for early life stages of cold water biota (i.e., 9.5 mg/L) 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). However, DO concentrations below this WQG occurred at depths between approximately 10 m and bottom (i.e., 15 m) at the Mary Lake north basin in the winter (Figure 5.7). Dissolved oxygen concentrations near the bottom of the water column at littoral and profundal stations of Mary Lake were well above the WQG, and did not differ significantly from those at respective station types in Reference Lake 3 during August 2018 biological sampling (Figure 5.9; Appendix Table C.72).

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 summer and fall sampling events in 2018, and were also comparable to pH profiles at Reference Lake 3 (Figures 5.7 and 5.8). During the winter sampling event, the pH at the north and south basins of Mary Lake generally decreased (i.e., became more neutral) with increased depth, and appeared to mirror the pattern for DO concentration profiles at each basin in winter (Figures 5.7 and 5.8). Therefore, the winter pH profiles at Mary Lake were likely the result of slight changes in redox conditions with depth. No significant differences in pH near the bottom of the water column were indicated between Mary Lake and Reference Lake 3 at littoral or profundal stations during August 2018 biological sampling (Figure 5.9; Appendix Table F.72). In addition, pH values at Mary Lake water quality and benthic stations were consistently within WQG limits (Figures 5.7 to 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.27). 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 profiles 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 2018, and also were similar in profile structure to those at Reference Lake 3 during the summer and fall sampling events (Figures 5.7 and 5.8). Specific conductance at the bottom of the water column of littoral and profundal stations of Mary Lake did not differ significantly from those at like-stations of Reference Lake 3 during the August 2018 biological sampling (Figure 5.9). Water clarity, as determined using Secchi depth readings, was significantly lower at Mary Lake compared to Reference Lake 3 in fall 2018 (Appendix Table C.72; Appendix Figure C.7). 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.70).

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 nitrate, total aluminum, total and dissolved manganese, and total and dissolved uranium compared to Reference Lake 3 at the time of summer and/or fall sampling in 2018 (Table 5.2; Appendix Tables C.74 and C.76). However, on average, concentrations of all parameters were below applicable WQG and AEMP benchmarks at the Mary Lake north basin during the winter, summer, and fall monitoring events in 2018 (Table 5.2; Appendix Table C.73). As in previous studies and other mine-exposed areas, aluminum concentrations showed a strong positive correlation with turbidity at the Mary Lake north basin stations using data collected in 2018, suggesting that much of the aqueous aluminum was associated with suspended particles (e.g., aluminosilicates; Appendix Table C.77). Temporal evaluation of the data indicated that parameter concentrations in 2018 were comparable to respective parameter concentration ranges shown during the mine baseline period (2005 to 2013; Figure 5.10; Appendix Table C.74; Appendix Figure C.28).

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 2018 (Table 5.2; Appendix Table C.78), suggesting that the south basin waters were well mixed. On average, turbidity, total aluminum, and dissolved manganese concentrations were moderately to highly elevated, and dissolved aluminum and total manganese slightly elevated (i.e., 3- to 5-fold higher) at the Mary Lake south basin compared to Reference Lake 3 during the 2018 summer and/or fall sampling events (Table 5.2; Appendix Tables C.74 and C.76). The total and dissolved concentrations of aluminum and manganese showed strong to very strong positive correlation with turbidity for the Mary Lake south basin 2018 data (i.e., r_s range from 0.67 to 0.91; Appendix Table C.80), suggesting that these metals were associated with suspended particles (e.g., aluminosilicates). In addition, ratios of dissolved to total concentrations of aluminum and manganese indicated that on average, approximately 69% of these metals were associated with the particulate fraction during summer and fall sampling events. As indicated in previous CREMP, high turbidity in the Mary River originates from natural sources upstream of the mine which, in turn, contributes to high turbidity and elevated concentrations of metals such as aluminum, iron, and manganese at Mary Lake. Concentrations of all parameters were below applicable WQG and AEMP benchmarks at the Mary Lake south basin during the winter, summer, and fall monitoring events in 2018, the lone exception being phenol concentrations above the applicable WQG during the summer monitoring event (Appendix Table C.78). Temporal comparisons of the Mary Lake south basin water chemistry

Table 5.2: Water Chemistry at Mary Lake North Basin (BLO-01) and South Basin (BLO) Monitoring Stations[a], Mary River Project CREMP, 2018

	Parameters	Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3) Fall 2018	North Basin (Mine-exposed)			South Basin (Mine-exposed)						
						BL0-01-A 26-Aug-2018	BL0-01 26-Aug-2018	BL0-01-B 26-Aug-2018	BL0-05-A 25-Aug-2018	BL0-05 24-Aug-2018	BL0-05-B 24-Aug-2018	BL0-03 25-Aug-2018	BL0-04 25-Aug-2018	BL0-09 25-Aug-2018	BL0-06 25-Aug-2018
Conventionals	Conductivity (lab)	umho/cm	-	-	75	146	137	139	82	76	74	74	75	75	76
	pH (lab)	pH	6.5 - 9.0	-	7.65	8.08	8.15	8.08	7.73	7.86	7.88	7.76	7.72	7.71	7.66
	Hardness (as $CaCO_3$)	mg/L	-	-	35	73	70	71	38	38	36	36	35	34	35
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	46	70	68	58	48	114	41	55	40	108	45
	Turbidity	NTU	-	-	0.5	0.7	0.9	0.8	1.4	1.4	1.5	0.8	1.3	1.4	1.5
	Alkalinity (as $CaCO_3$)	mg/L	-	-	33	68	65	67	37	35	34	32	76	33	34
Nutrients and Organics	Total Ammonia	mg/L	variable	0.855	0.044	<0.020	<0.020	0.021	<0.020	<0.020	0.021	0.0415	0.0325	0.0265	0.034
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	0.0205	<0.020	0.070	<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
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.16	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	-	-	2.9	2.6	2.6	2.7	1.8	1.5	1.4	1.9	1.7	1.8	1.7
	Total Organic Carbon	mg/L	-	-	3.8	2.8	2.8	2.8	2.1	1.6	1.5	2.4	2.3	2.3	2.5
	Total Phosphorus	mg/L	0.020^{α}	-	0.005	0.003	0.006	0.003	0.005	0.007	0.005	0.005	0.005	0.004	0.005
	Phenols	mg/L	0.004^{α}	-	0.001	<0.0010	<0.0010	<0.0010	<0.0010	0.0012	0.0011	0.0017	0.0012	<0.0010	0.0012
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
	Chloride (Cl)	mg/L	120	120	1.27	2.43	2.24	2.29	1.91	1.64	1.66	1.38	1.62	1.61	1.70
	Sulphate (SO_4)	mg/L	218^{β}	218	3.74	1.39	1.33	1.34	3.10	2.54	2.61	1.34	2.50	2.53	2.84
Total Metals	Aluminum (Al)	mg/L	0.100	0.13	0.004	0.022	0.028	0.026	0.041	0.046	0.039	0.022	0.038	0.039	0.048
	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
	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
	Barium (Ba)	mg/L	-	-	0.0064	0.0074	0.0071	0.0074	0.0046	0.0048	0.0044	0.0043	0.0043	0.0043	0.0045
	Beryllium (Be)	mg/L	0.011^{α}	-	<0.00050	<0.00050	<0.00050	<0.00050	<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	<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
	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
	Calcium (Ca)	mg/L	-	-	7.2	15.0	14.7	14.8	7.8	7.6	7.2	7.3	7.2	7.1	7.1
	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
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	0.00015	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.00076	0.00086	0.00087	0.00090	0.00060	0.00063	0.00056	0.00058	0.00062	0.00057	0.00059
	Iron (Fe)	mg/L	0.30	0.326	<0.030	<0.030	<0.030	<0.030	0.041	0.042	0.038	<0.030	0.035	0.034	0.044
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	0.000054	0.000062	<0.000050	<0.000050	<0.000050	0.000053
	Lithium (Li)	mg/L	-	-	0.001	<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	8.6	8.1	8.3	4.6	4.6	4.3	4.2	4.3	4.2	4.4
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00064	0.00240	0.00226	0.00225	0.00347	0.00304	0.00275	0.00156	0.00255	0.00249	0.00299
	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
	Molybdenum (Mo)	mg/L	0.073	-	0.00014	0.00019	0.00019	0.00021	0.00015	0.00014	0.00012	0.00011	0.00012	0.00012	0.00013
	Nickel (Ni)	mg/L	0.025	0.025	0.0005	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Potassium (K)	mg/L	-	-	0.86	0.82	0.81	0.82	0.57	0.58	0.54	0.50	0.54	0.53	0.55
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.42	0.69	0.68	0.73	0.52	0.53	0.50	0.45	0.49	0.50	0.52
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.86	1.73	1.59	1.59	0.93	0.93	0.87	0.85	0.88	0.87	0.86
	Strontium (Sr)	mg/L	-	-	0.0081	0.0094	0.0097	0.0103	0.0066	0.0065	0.0059	0.0055	0.0058	0.0059	0.0058
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<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	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00026	0.00135	0.00123	0.00124	0.00053	0.00056	0.00045	0.00041	0.00044	0.00043	0.00046
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<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

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

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

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

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

data did not indicate any substantial changes in average concentrations of mine-related parameters in 2018 compared to the baseline period (2005 to 2013; Figure 5.10; Appendix Figure C.28). In addition, parameter concentrations at the Mary River south basin in fall 2018 did not show any consistent increase compared to the year of mine construction (2014) and the three previous years of mine operation (2015 to 2017; Figure 5.10; Appendix Figure C.28). The absence of temporal changes in water quality suggested no adverse mine-related influences on water chemistry of the Mary Lake south basin since the onset of commercial mine operations.

5.2.3 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 littoral stations, surficial sediment varied from silt loam to silty clay loam (Figure 5.11; Appendix Table D.21), whereas at the south basin profundal stations, surficial sediment was predominantly variations of silt loam, clay loam, and silty clay loam except at Station BLO-03 where sand was more prevalent (Figure 5.11; Appendix Table D.21). Substrate at littoral and profundal stations of Mary Lake contained significantly lower sand and TOC, and significantly greater silt, than at Reference Lake 3 (Appendix Table D.22). Reddish-brown coloured iron (oxy)hydroxide material was not observed in substrate at the Mary Lake north basin, but was present at some south basin stations, (Appendix Tables D.20 and D.21), mirroring similar observations at Reference Lake 3 and the other mine-exposed lakes where such material was commonly visible as a thin, distinct layer or floc on or within surficial sediment. Substrate of Mary Lake commonly contained sub-surface blackening/dark colouration indicating the presence of reduced sediment conditions, but no distinct redox boundaries were observed (Appendix Table D.21). Similar sub-surface reducing conditions were observed in sediment of Reference Lake 3, including the absence of distinct redox boundaries (Appendix Tables D.1 and D.2), suggesting that factors leading to reduced sediment conditions were comparable between lakes.

Sediment metal concentrations at littoral stations of the Mary Lake north and south basins were comparable to those observed at littoral stations of Reference Lake 3, the only notable exception being manganese, the concentration of which was slightly elevated (i.e., 3- to 5-fold higher) at the north basin (Table 5.3; Appendix Table D.24). Concentrations of iron and manganese were above applicable SQG, and the concentration of arsenic was above the Mary Lake AEMP benchmark, in sediment at the lone Mary Lake north basin station (i.e., BLO-01; Table 5.3). Sediment metal concentrations at the Mary Lake south basin showed no spatial gradients with progression from the Mary River inlet to the lake outlet among the profundal stations (Appendix Table D.23),[13]

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

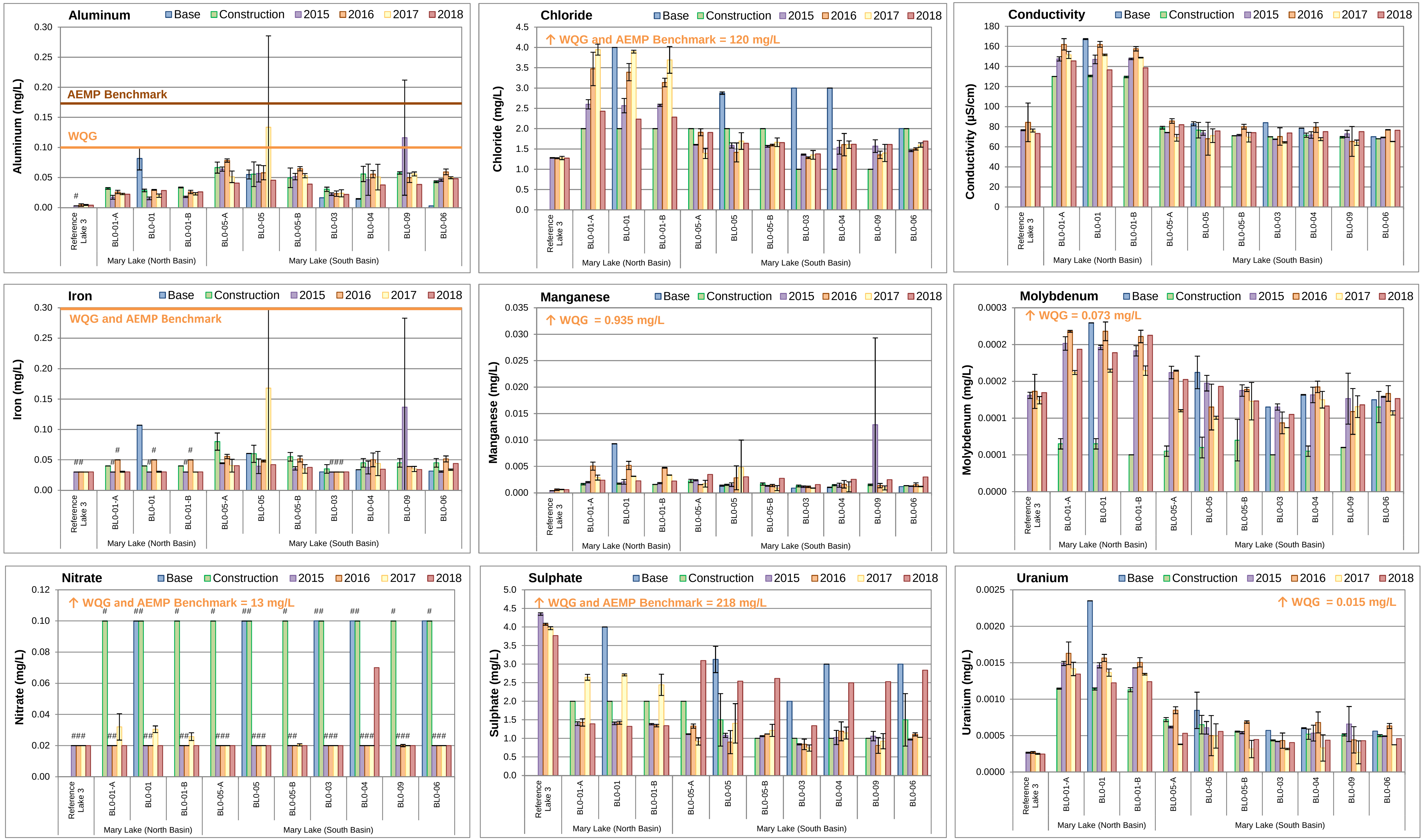


Figure 5.10: Temporal Comparison of Water Chemistry at Mary Lake (BLO) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2018) Periods during Fall

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

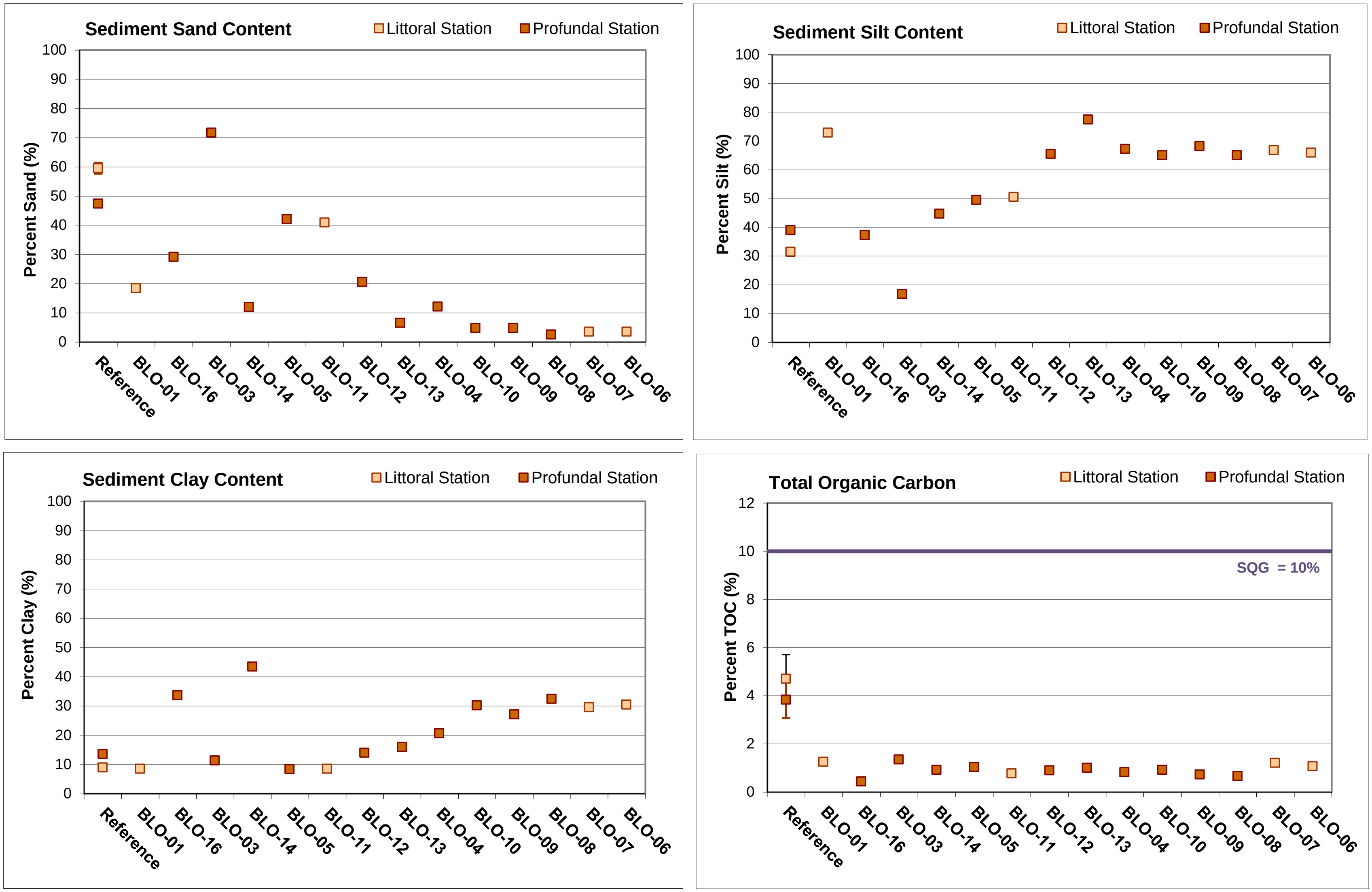


Figure 5.11: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Mary Lake (BLO) North and South Basin Sediment Monitoring Stations and to Reference Lake 3 (mean ± SE), Mary River Project CREMP, August 2018

Table 5.3: Sediment Total Organic Carbon and Metal Concentrations at Mary Lake North Basin (BLO-01) and South Basin (BLO), and Reference Lake 3 (REF3) Sediment Monitoring Stations, Mary River Project CREMP, August 2018

	Parameter	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b]	Littoral: Reference Lake (n = 5) Average ± Std. Error	Littoral: Mary Lake (North Basin) (n = 1)	Littoral: Mary Lake (South Basin) (n = 1)	Profundal: Reference Lake (n = 5) Average ± Std. Error	Profundal: Mary Lake (South Basin) (n = 8) Average ± Std. Error
Total Organic Carbon		%	10^{α}	-	4.70 ± 1.01	1.25	1.07	3.82 ± 0.75	0.87 ± 0.09
Metals	Aluminum (Al)	mg/kg	-	-	17,880 ± 1,993	17,600	31,200	24,420 ± 3,494	26,813 ± 2,324
Metals	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10	<0.10	<0.10 ± 0.00	<0.10 ± 0.00
Metals	Arsenic (As)	mg/kg	17	5.9	5.25 ± 0.95	***7.95***	4.85	***6.07*** ± 0.78	5.04 ± 0.51
Metals	Barium (Ba)	mg/kg	-	-	133 ± 25	103	136	152 ± 23	111 ± 10
Metals	Boron (B)	mg/kg	-	-	13.9 ± 1.6	23.4	41.0	15.6 ± 2.2	34.4 ± 3.0
Metals	Cadmium (Cd)	mg/kg	3.5	1.5	0.195 ± 0.044	0.106	0.166	0.197 ± 0.005	0.157 ± 0.011
Metals	Calcium (Ca)	mg/kg	-	-	5,480 ± 804	10,200	5,490	5,584 ± 664	4,899 ± 359
Metals	Chromium (Cr)	mg/kg	90	98	58.9 ± 7.7	72.3	***108.0***	77.3 ± 11.0	97.3 ± 7.4
Metals	Cobalt (Co)	mg/kg	-	-	11.70 ± 1.40	16.90	21.00	17.42 ± 2.37	19.23 ± 1.25
Metals	Copper (Cu)	mg/kg	110	50	***73.9*** ± 11.0	31.8	41.3	***96.3*** ± 14.7	37.7 ± 2.9
Metals	Iron (Fe)	mg/kg	40,000$^{\alpha}$	52,400	46,700 ± 9,489	45,200	51,200	50,900 ± 7,115	48,225 ± 3,097
Metals	Lead (Pb)	mg/kg	91.3	35	16.4 ± 2.1	17.0	30.1	19.5 ± 2.8	25.9 ± 2.2
Metals	Magnesium (Mg)	mg/kg	-	-	11,104 ± 1,352	16,200	20,100	15,394 ± 2,199	17,638 ± 1,362
Metals	Manganese (Mn)	mg/kg	1,100$^{\alpha,\beta}$	4,370	640 ± 60	2,180	878	1,279 ± 115	1,976 ± 444
Metals	Mercury (Hg)	mg/kg	0.486	0.17	0.0433 ± 0.0111	0.0262	0.0560	0.0650 ± 0.0121	0.0513 ± 0.0044
Metals	Molybdenum (Mo)	mg/kg	-	-	3.84 ± 0.86	0.78	0.95	2.57 ± 0.27	1.03 ± 0.07
Metals	Nickel (Ni)	mg/kg	75$^{\alpha,\beta}$	72	42.9 ± 5.9	60.7	***73.0***	53.8 ± 6.6	70.4 ± 5.0
Metals	Phosphorus (P)	mg/kg	2,000$^{\alpha}$	1,580	1,305 ± 272	1,460	987	1,188 ± 118	1,138 ± 81
Metals	Potassium (K)	mg/kg	-	-	4,134 ± 469	3,900	7,870	5,660 ± 796	6,589 ± 585
Metals	Selenium (Se)	mg/kg	-	-	0.66 ± 0.14	0.24	0.31	0.81 ± 0.15	0.26 ± 0.02
Metals	Silver (Ag)	mg/kg	-	-	0.15 ± 0.02	0.10	0.17	0.26 ± 0.04	0.16 ± 0.01
Metals	Sodium (Na)	mg/kg	-	-	320 ± 43	262	483	433 ± 62	430 ± 37
Metals	Strontium (Sr)	mg/kg	-	-	12.2 ± 1.5	14.3	17.2	13.8 ± 1.6	16.6 ± 1.5
Metals	Thallium (Tl)	mg/kg	-	-	0.450 ± 0.063	0.366	0.729	0.754 ± 0.091	0.598 ± 0.050
Metals	Uranium (U)	mg/kg	-	-	13.4 ± 2.32	4.2	10.8	24.5 ± 3.90	9.39 ± 0.82
Metals	Vanadium (V)	mg/kg	-	-	58.3 ± 6.9	56.9	87.8	72.7 ± 9.4	75.9 ± 6.0
Metals	Zinc (Zn)	mg/kg	315	135	81.4 ± 10.2	57.8	103.0	99.2 ± 14.2	80.1 ± 7.5

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

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

Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

suggesting that the Mary River was not a disproportionate source of metals. Sediment metal concentrations at the Mary Lake south basin profundal stations were similar to average metal concentrations at like-depth stations of Reference Lake 3 (Table 5.3; Appendix Table D.24). On average, concentrations of chromium and iron were above SQG at the Mary Lake south basin littoral station and profundal stations, as was the concentration of manganese at the south basin (Table 5.3). In addition, concentrations of nickel were above SQG, and concentrations of arsenic, chromium, iron, and nickel were above applicable AEMP benchmarks, at individual stations in the Mary Lake south basin (Appendix Table D.23). However, as indicated previously, average concentrations of iron and manganese were elevated above SQG in sediment at Reference Lake 3, as were average concentrations of chromium at individual stations at Reference Lake 3 (Appendix Table D.4). Arsenic, iron, and phosphorus concentrations were also above the Mary Lake AEMP benchmarks at individual stations of Reference Lake 3. In turn, this suggested that concentrations of chromium, iron, and manganese above SQG, and concentrations of arsenic and iron above AEMP benchmarks, at Mary Lake likely reflect natural conditions un-related to mine activity.

Temporal comparisons indicated that metal concentrations in sediment at littoral and profundal stations of Mary Lake in 2018 did not change substantially from those observed during the mine baseline (2005 to 2013) period (Figure 5.12; Appendix Table D.24). On average, metal concentrations in sediment at Mary Lake littoral and profundal stations in 2018 were slightly higher, or were in the upper range, of those observed from 2015 to 2017 (Figure 5.12). However, no occurrence of continual, year-to-year increases in metal concentrations were indicated that would suggest an increasing trend over time (Figure 5.12). Overall, no substantial changes in sediment metal concentrations have been observed at Mary Lake littoral and profundal habitats following the commencement of commercial mine operations in 2015.

5.2.4 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 2018 (Figure 5.13). Chlorophyll-a concentrations were typically lowest in winter and highest in fall at both the north and south basins of Mary Lake (Figure 5.13), and mirrored similar relative differences in chlorophyll-a concentrations between summer and fall sampling events at Reference Lake 3 (Appendix Table B.8). Chlorophyll-a concentrations were significantly lower at the Mary Lake north basin than at Reference Lake 3 in summer, but no significant differences in chlorophyll-a concentrations were indicated between Mary Lake north basin and Reference Lake 3 during the fall sampling event (Appendix Tables E.7 and E.8). At the Mary Lake south basin, chlorophyll-a concentrations did not differ significantly

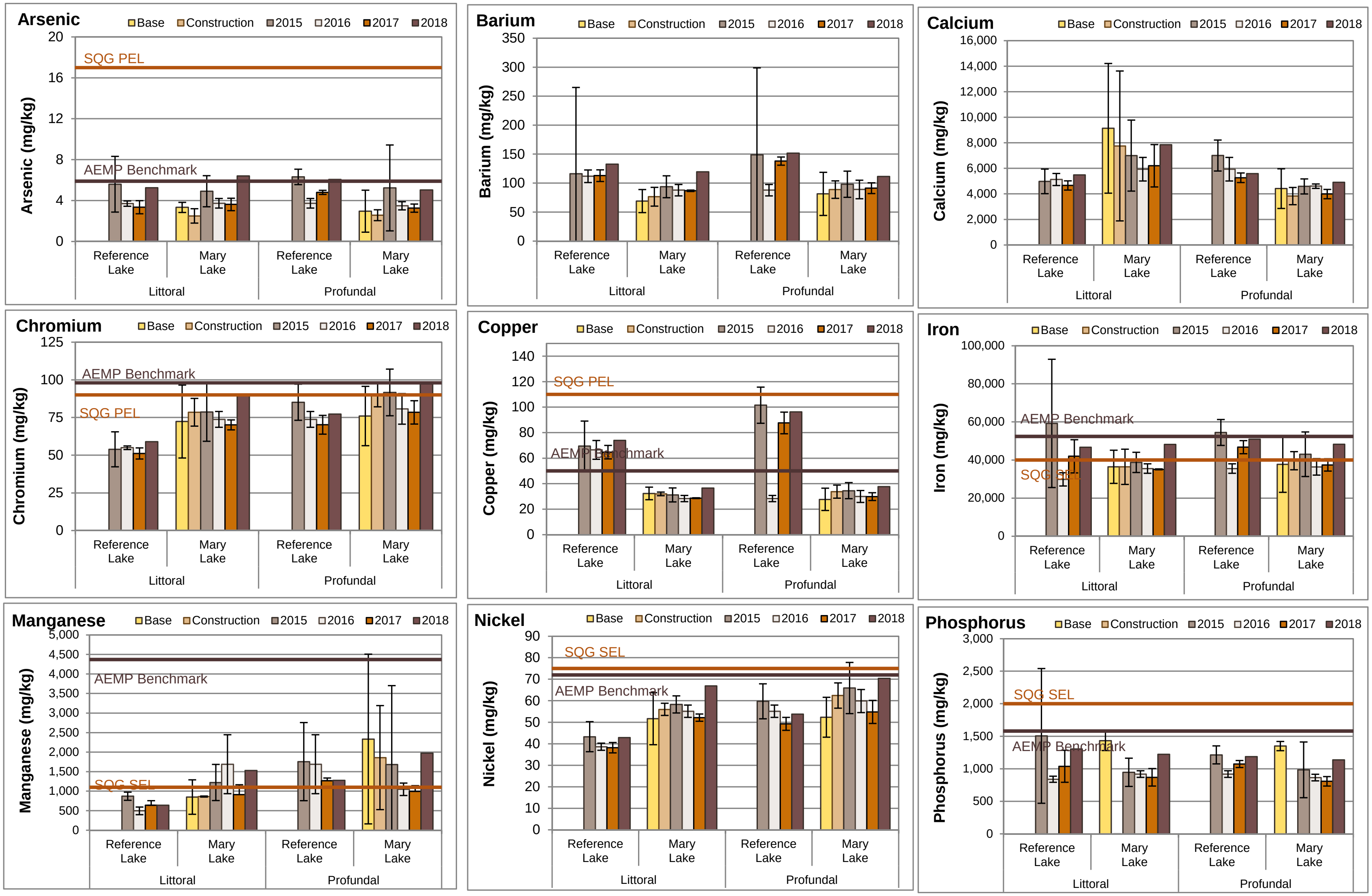


Figure 5.12: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Mary Lake and Reference Lake 3 for Mine Baseline (2005 to 2013), Construction (2014) and Operational (2015 to 2018) Periods, Mary River Project CREMP, 2018

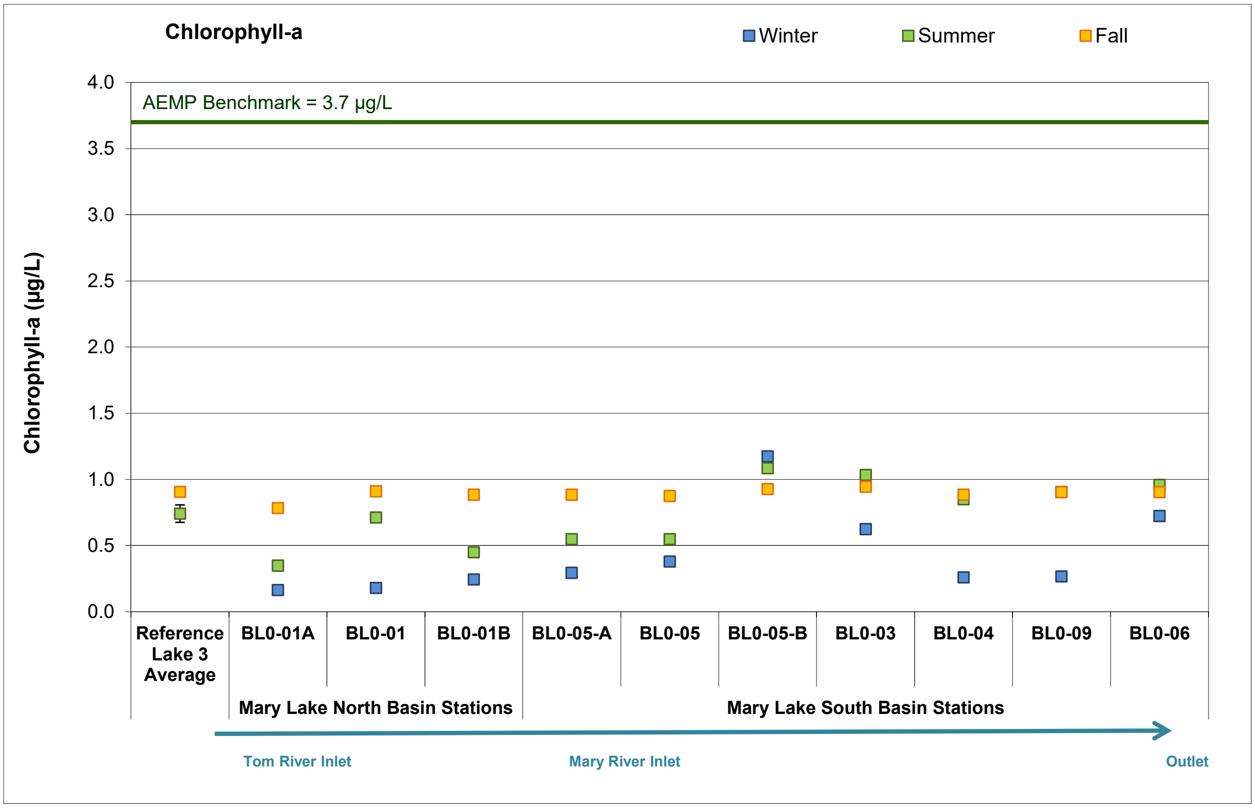


Figure 5.13: Chlorophyll-a Concentrations at Mary Lake (BLO) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2018

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

from those at Reference Lake 3 during either of the summer or fall sampling events (Appendix Tables E.7 and E.8). Chlorophyll-a concentrations at the Mary Lake north and south basins were well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events in 2018 (Figure 5.13) and reflected an oligotrophic primary productivity categorization based on Wetzel (2001) classification. This oligotrophic categorization was in agreement with CWQG trophic status classification that is based on average aqueous total phosphorus concentrations below 10 µg/L (Table 5.2; Appendix Tables C.73 and C.78).

Temporal comparison of Mary Lake chlorophyll-a concentrations, conducted separately for the north and south basins, did not indicate any consistent direction of significant differences between the 2018 data and data from the mine construction (2014) period or previous years of mine operation (2015 to 2017) during any of the winter, summer, or fall seasons (Figure 5.14; Appendix Figure E.1). In addition, annual average chlorophyll-a concentrations have not shown any consistent direction of change (i.e., increase or decrease) over time since the mine was constructed in 2014 (Figure 5.14; Appendix Figure E.1) suggesting no substantial changes in the

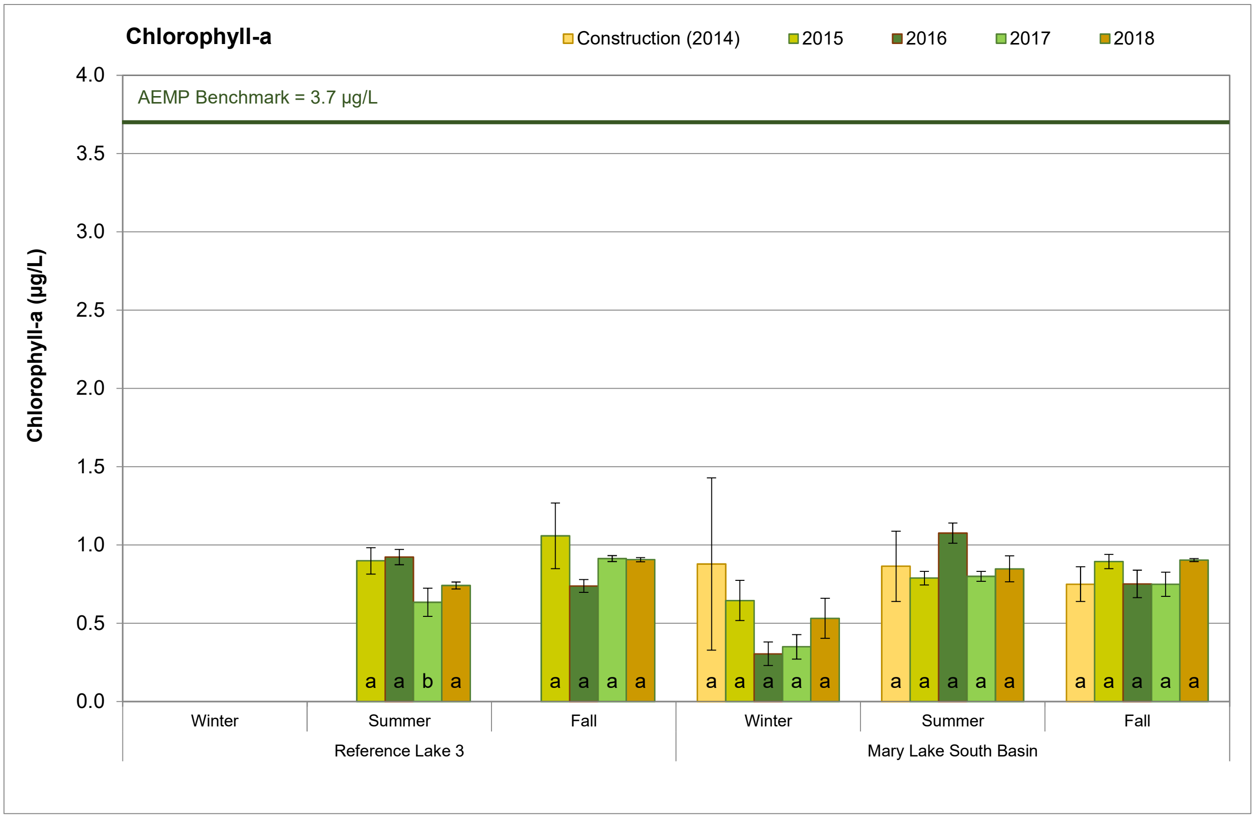


Figure 5.14: Temporal Comparison of Chlorophyll-a Concentrations Among Seasons between the Mary Lake South Basin and Reference Lake 3 for Mine Construction (2014) and Operational (2015 to 2018) Periods (mean ± SE)

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

trophic status of the lake since mine operations commenced at the Mary River Project. No chlorophyll-a baseline (2005 to 2013) data are available for Mary Lake, precluding comparisons to conditions prior to mine construction.

5.2.5 Benthic Invertebrate Community

Benthic invertebrate density at littoral habitat did not differ significantly between Mary Lake and Reference Lake 3, but for profundal habitat, significantly higher density was observed at Mary Lake in 2018 (Tables 5.4 and 5.5). *Heterotrissocladius* midges, which are characteristic of ultra-oligotrophic to oligotrophic habitats, were the dominant benthic invertebrates observed at both Mary Lake and Reference Lake 3 (Appendix Tables F.17 and F.60), suggesting similar trophic status between these lakes. No significant differences in richness or Simpson's Evenness were indicated between Mary Lake and Reference Lake 3 for either habitat type (Tables 5.4 and 5.5). Although Bray-Curtis Index at the littoral habitat benthic invertebrate community differed significantly between Mary Lake and Reference Lake 3, no ecologically significant differences in

Table 5.4: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BLO) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	square root	NO	0.577	t-test (unequal)	2.7	Reference Lake 3	1,045	258	116	696	1,000	1,391
						Mary Lake Littoral	1,733	1,431	715	287	1,765	3,113
Richness (Number of Taxa)	none	NO	0.339	ANOVA	-0.7	Reference Lake 3	10.8	2.3	1.0	7.0	11.0	13.0
						Mary Lake Littoral	9.3	2.2	1.1	7.0	9.0	12.0
Simpson's Evenness (E)	none	NO	0.114	ANOVA	-2.4	Reference Lake 3	0.825	0.103	0.046	0.720	0.816	0.939
						Mary Lake Littoral	0.575	0.293	0.146	0.141	0.695	0.767
Bray-Curtis Index	none	YES	0.001	ANOVA	4.9	Reference Lake 3	0.313	0.092	0.041	0.178	0.358	0.394
						Mary Lake Littoral	0.768	0.163	0.082	0.593	0.774	0.932
Nemata (%)	fourth root	NO	0.515	ANOVA	-0.4	Reference Lake 3	7.1	8.8	3.9	0.0	3.4	21.3
						Mary Lake Littoral	3.5	6.4	3.2	0.0	0.5	13.1
Ostracoda (%)	log	NO	0.137	ANOVA	-0.8	Reference Lake 3	23.9	18.3	8.2	3.4	20.6	53.3
						Mary Lake Littoral	8.9	10.9	5.5	1.4	4.5	25.0
Chironomidae (%)	log	NO	0.239	ANOVA	0.9	Reference Lake 3	66.9	22.2	10.0	35.5	73.8	91.4
						Mary Lake Littoral	86.2	18.7	9.3	58.3	94.5	97.2
Metal-Sensitive Chironomidae (%)	log	YES	0.035	ANOVA	-1.3	Reference Lake 3	36.5	19.6	8.8	17.8	27.5	60.1
						Mary Lake Littoral	11.6	9.8	4.9	2.5	9.9	23.9
Collector-Gatherers (%)	none	NO	0.308	ANOVA	1.1	Reference Lake 3	55.6	19.0	8.5	33.0	57.5	79.2
						Mary Lake Littoral	76.1	36.4	18.2	21.8	91.8	98.9
Filterers (%)	fourth root	YES	0.009	ANOVA	-1.6	Reference Lake 3	33.9	18.7	8.4	15.5	24.9	56.6
						Mary Lake Littoral	4.1	7.3	3.7	0.0	0.6	15.0
Shredders (%)	fourth root	YES	0.045	t-test (unequal)	-2.4	Reference Lake 3	7.0	2.6	1.1	2.9	7.5	9.4
						Mary Lake Littoral	0.8	1.1	0.6	0.0	0.5	2.3
Clingers (%)	fourth root	YES	0.017	ANOVA	-1.7	Reference Lake 3	36.1	18.4	8.2	17.1	26.9	58.3
						Mary Lake Littoral	5.5	7.4	3.7	0.0	3.0	16.2
Sprawlers (%)	fourth root	YES	0.028	ANOVA	1.8	Reference Lake 3	51.9	17.7	7.9	29.5	52.5	71.8
						Mary Lake Littoral	83.0	10.3	5.1	72.3	81.7	96.3
Burrowers (%)	square root	NO	0.837	ANOVA	-0.1	Reference Lake 3	12.0	6.4	2.8	6.9	11.1	22.6
						Mary Lake Littoral	11.5	6.3	3.2	2.5	13.3	16.7

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

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

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

Table 5.5: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BLO) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2018

Metric	Statistical Test Results					Summary Statistics						
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	none	YES	0.005	t-test (unequal)	7.5	Reference Lake 3	377	155	69	104	452	470
						Mary Lake Profundal	1,533	606	247	991	1,439	2,487
Richness (Number of Taxa)	log	NO	0.265	ANOVA	2.1	Reference Lake 3	5.4	1.3	0.6	4.0	6.0	7.0
						Mary Lake Profundal	8.2	4.4	1.8	4.0	7.0	14.0
Simpson's Evenness (E)	log	NO	0.418	ANOVA	-0.2	Reference Lake 3	0.455	0.296	0.132	0.218	0.296	0.933
						Mary Lake Profundal	0.386	0.360	0.147	0.062	0.248	0.867
Bray-Curtis Index	rank	NO	0.126	Mann-Whitney	1.3	Reference Lake 3	0.224	0.304	0.136	0.051	0.109	0.763
						Mary Lake Profundal	0.611	0.128	0.052	0.441	0.656	0.727
Nemata (%)	fourth root	NO	0.692	ANOVA	-0.2	Reference Lake 3	2.5	3.8	1.7	0.0	0.0	8.7
						Mary Lake Profundal	1.7	1.9	0.8	0.0	1.2	5.2
Hydracarina (%)	none	NO	0.183	t-test (unequal)	-0.7	Reference Lake 3	3.7	3.8	1.7	0.0	3.9	8.7
						Mary Lake Profundal	1.0	0.9	0.4	0.0	0.9	2.1
Ostracoda (%)	square-root	NO	0.882	ANOVA	0.1	Reference Lake 3	3.1	2.9	1.3	0.0	2.0	7.5
						Mary Lake Profundal	3.5	3.2	1.3	0.0	2.9	8.8
Chironomidae (%)	none	NO	0.294	ANOVA	0.6	Reference Lake 3	90.8	4.9	2.2	82.7	92.2	95.7
						Mary Lake Profundal	93.8	4.1	1.7	89.7	92.8	100.0
Metal-Sensitive Chironomidae (%)	log	NO	0.545	ANOVA	-0.2	Reference Lake 3	11.4	16.8	7.5	2.3	3.9	41.4
						Mary Lake Profundal	8.6	11.2	4.6	0.5	2.2	26.9
Collector-Gatherers (%)	rank	NO	0.855	Mann-Whitney	0.0	Reference Lake 3	89.8	13.6	6.1	66.3	96.2	100.0
						Mary Lake Profundal	90.0	13.2	5.4	68.6	97.0	100.0
Filterers (%)	fourth root	NO	0.912	ANOVA	0.1	Reference Lake 3	6.5	10.5	4.7	0.0	3.7	25.0
						Mary Lake Profundal	7.8	11.4	4.7	0.0	1.3	26.0
Clingers (%)	fourth root	NO	0.908	ANOVA	0.0	Reference Lake 3	10.2	13.6	6.1	0.0	3.9	33.6
						Mary Lake Profundal	9.8	13.3	5.4	0.5	1.7	28.7
Sprawlers (%)	rank	NO	0.792	Mann-Whitney	-0.2	Reference Lake 3	79.3	26.8	12.0	32.7	90.4	100.0
						Mary Lake Profundal	75.0	35.3	14.4	24.4	96.5	99.0
Burrowers (%)	fourth root	NO	0.750	ANOVA	0.3	Reference Lake 3	10.6	14.1	6.3	0.0	5.6	33.6
						Mary Lake Profundal	15.2	21.9	9.0	0.0	2.1	46.9

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

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

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

any of the dominant taxonomic groups was indicated between lakes (Table 5.4). Similarly, although the relative abundance of certain FFG and HPG differed significantly between Mary Lake and Reference Lake 3 for littoral habitat, the magnitudes of these differences were generally below ecologically meaningful thresholds (i.e., within the CES_{BIC} of ±2 SD_{REF}; Table 5.4). At profundal habitat, no significant differences in the relative abundance of dominant taxonomic groups, FFG, or HPG were indicated at Mary Lake compared to Reference Lake 3 (Table 5.5). In addition, no ecologically significant difference in the relative abundance of metal-sensitive chironomids was indicated between Mary Lake and Reference Lake 3 for either littoral or profundal habitat types (Tables 5.4 and 5.5), suggesting no metal-related influences on the benthic invertebrate community of Mary Lake. Overall, no adverse mine-related influences to the littoral or profundal benthic invertebrate community were indicated at Mary Lake relative to reference lake conditions in 2018.

Temporal comparisons indicated no ecologically significant differences in benthic invertebrate community density, richness, or Simpson's Evenness at littoral or profundal habitats of Mary Lake between years of mine operation and mine baseline studies (Figure 5.15; Appendix Tables F.61 and F.62). In addition, no significant differences in the relative abundance of dominant taxonomic groups and FFG were indicated between baseline and mine operational years at Mary Lake (Appendix Tables F.61 and F.62). Therefore, consistent with no substantial changes in water and sediment quality since the mine baseline period, no significant changes in benthic invertebrate community features were indicated at littoral and profundal habitat of Mary Lake following the commencement of commercial mine operations in 2015.

5.2.6 Fish Population

5.2.6.1 Mary Lake (South) Fish Community

Arctic charr and ninespine stickleback composed the fish community of Mary Lake in 2018, reflecting the same fish species composition as Reference Lake 3 (Table 5.6). Similar to the other mine-exposed lakes, arctic charr CPUE was higher at Mary Lake than at Reference Lake 3 for electrofishing and gill netting collection methods in 2018, suggesting higher densities and/or productivity of arctic charr at Mary Lake. Also consistent with the other mine-exposed lakes, greater numbers of arctic charr together with greater density of benthic invertebrates suggested that overall biological productivity was higher at Mary Lake than at Reference Lake 3.

Temporal comparison of the Mary Lake electrofishing catch data indicated substantially higher arctic charr CPUE in 2018, as well as in other mine construction/operation years, than during baseline monitoring conducted in 2008 (Figure 5.16). Gill netting CPUE for arctic charr was higher in 2018 compared to all previous baseline (2006 and 2007), mine construction (2014), and mine

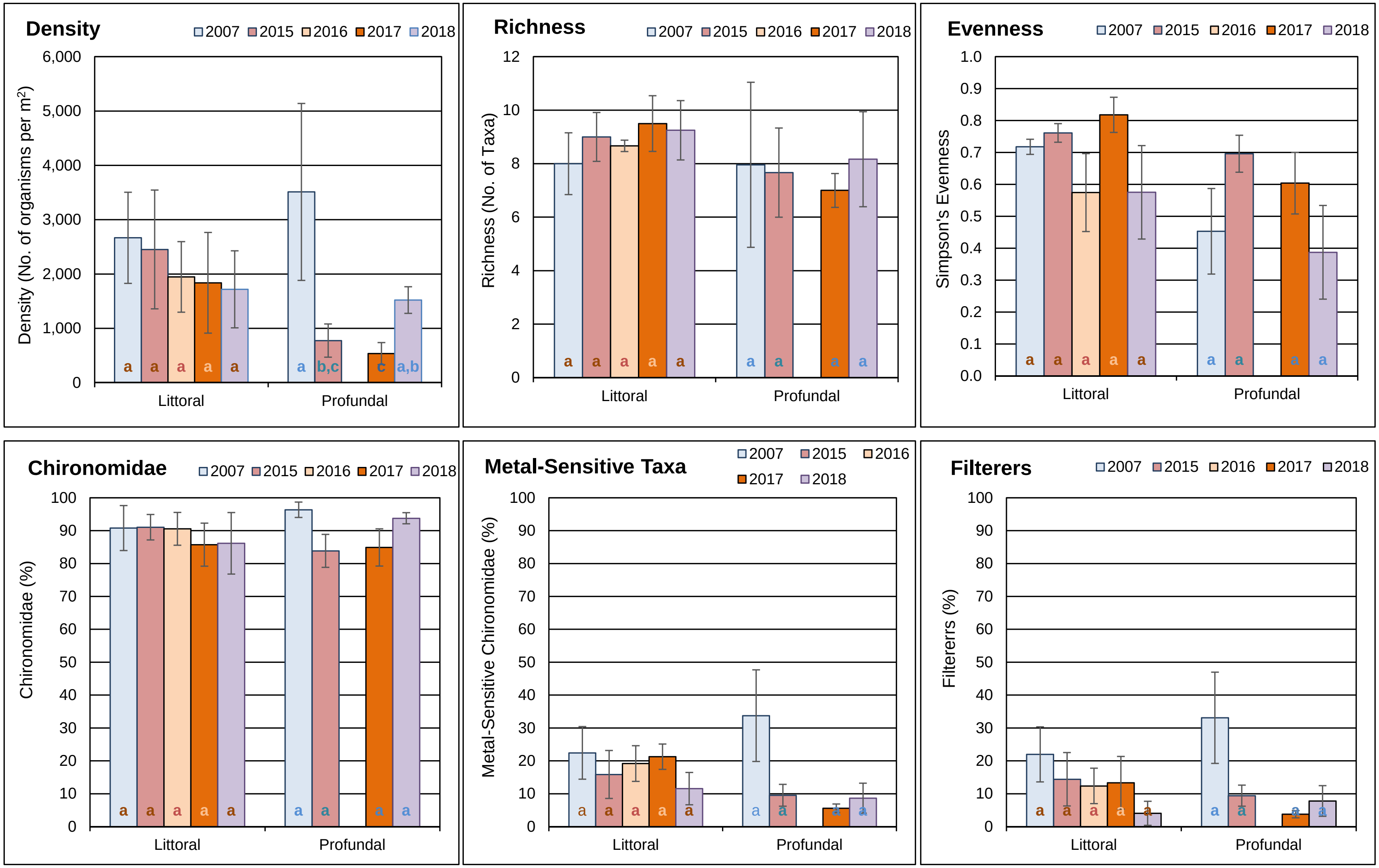


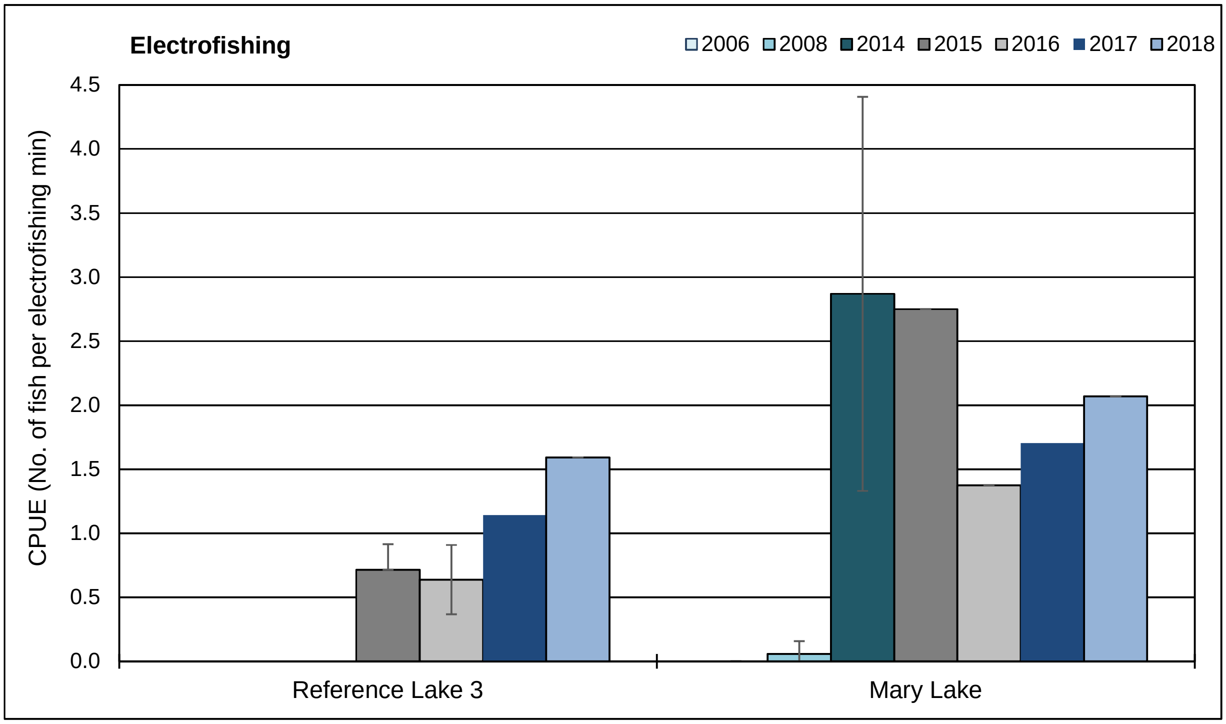
Figure 5.15: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Mary Lake Littoral and Profundal Study Areas among Mine Baseline (2007) and Operational (2015 to 2018) Periods

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

Table 5.6: 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 2018

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	101	2	103	2
		CPUE	1.59	0.02	1.61	
	Gill netting	No. Caught	34	0	34	
		CPUE	0.38	0	0.38	
Mary Lake	Electrofishing	No. Caught	103	4	107	2
		CPUE	2.07	0.08	2.15	
	Gill netting	No. Caught	129	0	129	
		CPUE	7.30	0	7.30	

[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.



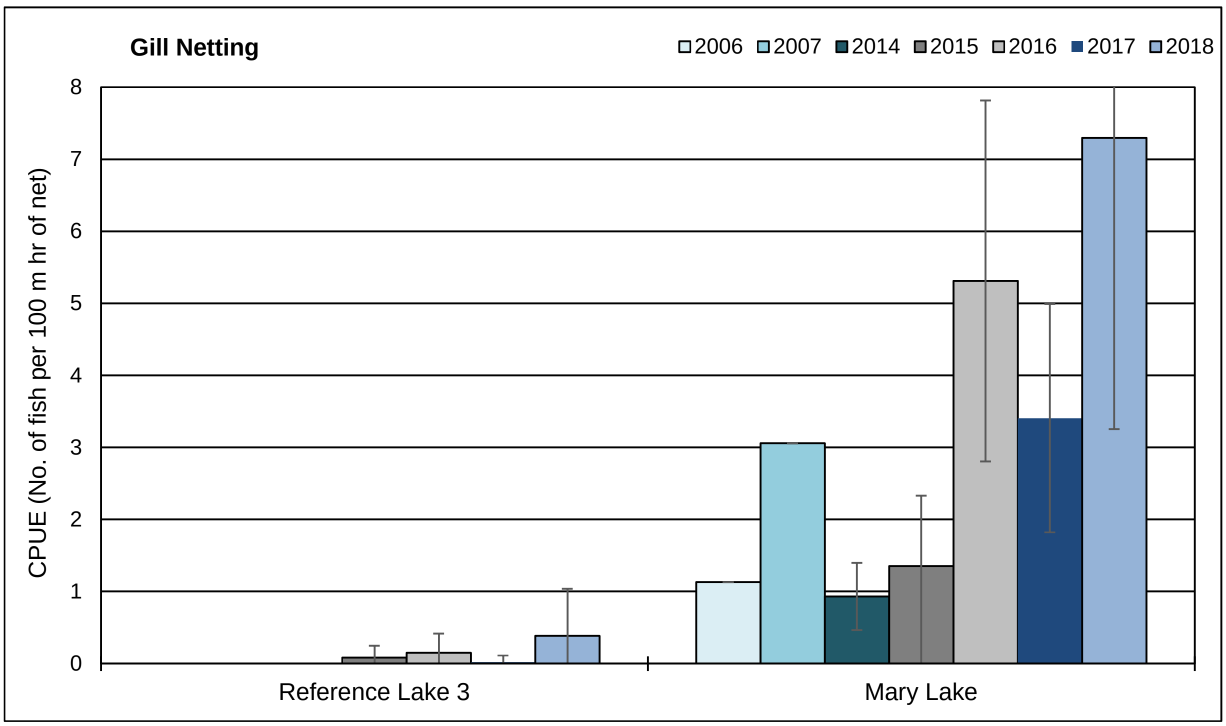


Figure 5.16: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Mary Lake (BLO), Mary River Project CREMP, 2006 to 2018

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

operational (2015 to 2017) studies (Figure 5.16), most likely reflecting improved sampling efficiencies in 2018 relative to the previous studies. Nevertheless, the CPUE data suggested that arctic charr abundance at nearshore and littoral/profundal habitats was likely comparable to, or greater than, the abundance of this species during the baseline period at Mary Lake, indicating no mine-related influences to arctic charr numbers in the lake following commercial mine operation start-up in 2015.

5.2.6.2 Mary Lake (South) Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Mary Lake nearshore arctic charr population were assessed based on a control-impact analysis using data collected from Mary Lake and Reference Lake 3 in 2018. No nearshore arctic charr baseline data were collected at Mary Lake, precluding data analysis using a before-after design. A total of 103 and 100 arctic charr were captured at nearshore habitat of Mary Lake and Reference Lake 3, respectively, in August 2018, for the control-impact analysis. Arctic charr YOY were distinguished from the older, non-YOY age class using a fork length cut-off of 4.5 cm based on the evaluation of length-frequency distributions coupled with supporting age determinations for the Mary Lake and Reference Lake 3 data sets (Figure 5.17). 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. However, because the YOY data set was small (i.e., 14 and 8 YOY for Mary Lake and Reference Lake 3, respectively), caution is warranted around conclusions drawn from the analysis of YOY endpoints.

Nearshore arctic charr length-frequency distributions differed significantly between Mary Lake and Reference Lake 3, potentially reflecting the occurrence of more YOY and greater numbers of larger non-YOY individuals at Mary Lake (Table 5.7; Figure 5.17; Appendix Table G.26). Arctic charr in YOY and non-YOY age classes were significantly heavier and longer, respectively, at Mary Lake compared to Reference Lake 3 (Table 5.7; Appendix Table G.26). However, YOY condition did not differ significantly between Mary Lake and Reference Lake 3, and although the condition of non-YOY was significantly lower at Mary Lake, the magnitude of this difference was within the CES_C of ±10%, indicating that this difference was not ecologically meaningful (Table 5.7; Appendix Table G.26). Temporal comparisons indicated no consistent differences in size or condition of arctic charr non-YOY at Mary Lake compared to Reference Lake 3 from 2015 to 2018 (Table 5.7). Collectively, the data indicated no adverse response to arctic charr at Mary Lake nearshore areas since the commencement of commercial mine operations in 2015.

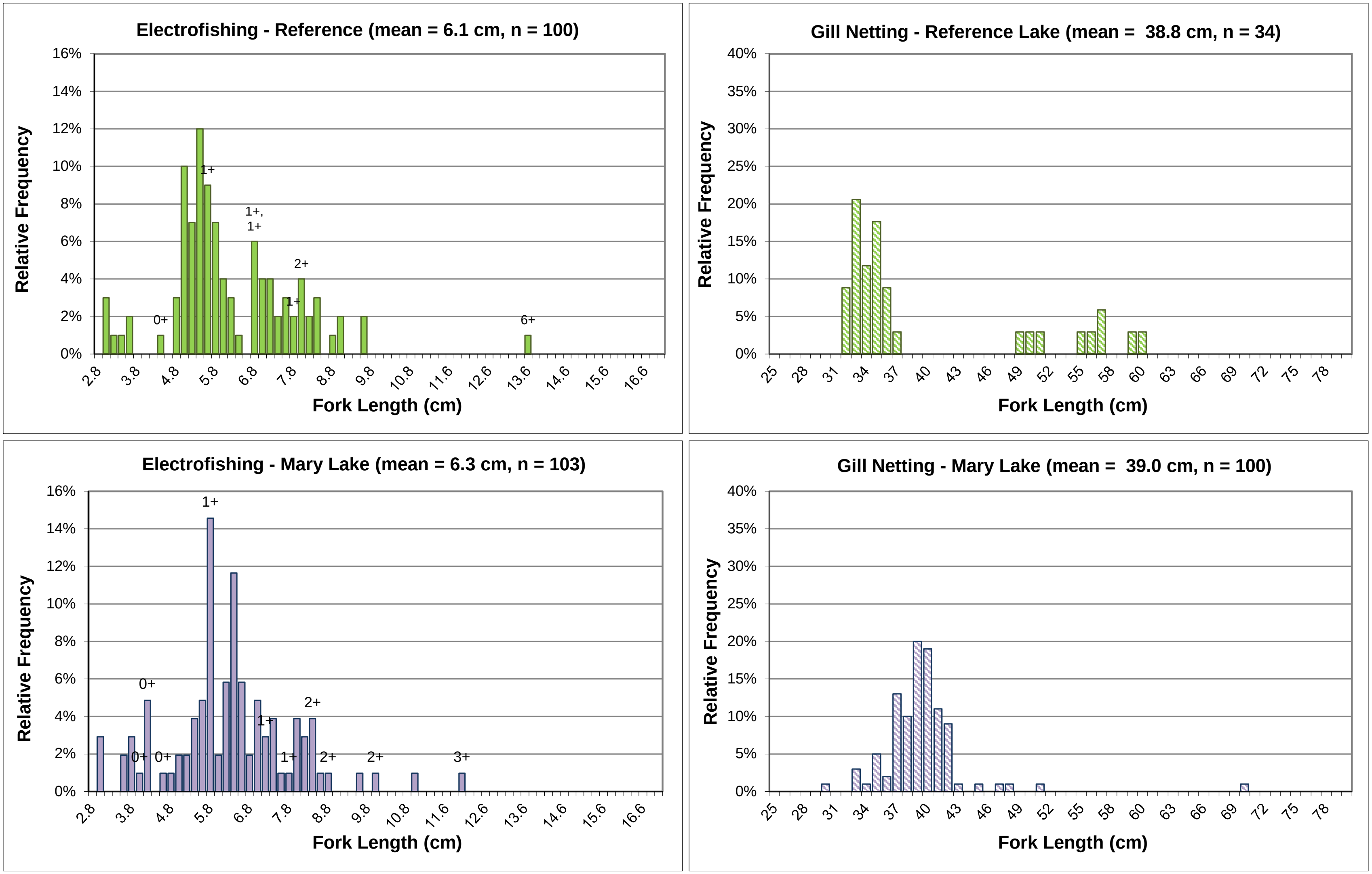


Figure 5.17: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Mary Lake (BLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2018

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

Table 5.7: Summary of Statistical Results for Arctic Charr Population Comparisons between Mary Lake and Reference Lake 3 from 2015 to 2018, and between Mary Lake Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP

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

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

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

[c] Due to low catches of arctic charr in gill nets at Reference Lake 3 in 2015, 2016, and 2017, no comparison of fish health was conducted for gill netted fish.

Littoral/Profundal Arctic Charr

Mine-related influences on the littoral/profundal arctic charr population were evaluated based on a control-impact analysis using 2018 data collected at Mary Lake and Reference Lake 3, and based on a before-after analysis using data collected from Mary Lake in 2018 and during 2006 to 2007 baseline studies. A total of 100 and 34 arctic charr were sampled from littoral/profundal habitat of Mary Lake and Reference Lake 3, respectively, in August 2018, for the control-impact analysis. The length-frequency distribution for littoral/profundal arctic charr differed significantly between lakes (Table 5.7; Figure 5.17). In addition, arctic charr sampled from littoral/profundal habitat of Mary Lake were significantly longer, heavier, and of greater condition than at Reference Lake 3 (Table 5.7). However, the magnitude of difference in condition was within the CES_C of ±10%, indicating that this difference was not ecologically meaningful (Table 5.7; Appendix Table G.30).

The length-frequency distribution of arctic charr captured at littoral/profundal habitat of Mary Lake differed significantly between 2018 and the baseline period (Table 5.7; Appendix Table G.30). However, no significant difference in arctic charr length or weight were indicated between 2018 and the baseline period in fish sampled from littoral/profundal habitat (Table 5.7). In addition, although condition of arctic charr sampled from Mary Lake littoral/profundal habitat was significantly lower in 2018 than during the baseline studies, the magnitude of this difference was within the CES_C of ±10% (Table 5.7) suggesting that this difference was not ecologically meaningful. No consistent differences in adult arctic charr health endpoints of size and condition were indicated at Mary Lake for individual years of mine operation from 2015 to 2018 compared to baseline data (Table 5.7). In turn, this suggested that natural and/or sampling variability accounted for slight differences in the arctic charr health endpoints shown during years of mine operation relative to baseline conditions at Mary Lake.

5.2.7 Integrated Summary

Turbidity and aqueous concentrations of total aluminum, manganese, nitrate, and uranium were elevated compared to Reference Lake 3 in 2018, but none of these metals, or any other parameters, were consistently elevated compared to concentrations observed during the baseline period, and none were consistently above WQG or AEMP benchmarks. Similar to Sheardown Lake, turbidity at Mary Lake was naturally higher than at Reference Lake 3 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 generally 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 Reference Lake 3 in 2018 and, with the exception of slightly elevated sediment manganese concentrations at littoral stations, were similar to concentrations observed during the baseline period. Although chromium, iron, manganese, and nickel concentrations were above SQG at Mary Lake in 2018, with the exception of nickel, these metals were also above SQG at Reference Lake 3 suggesting low potential for any adverse effects to biota associated with these metals. Similarly, although arsenic, chromium, iron, and nickel concentrations in sediment of Mary Lake were above AEMP benchmarks at Mary Lake, arsenic, iron, and phosphorus concentrations were above respective AEMP benchmarks at Reference Lake 3 as well, suggesting naturally high concentrations of metals in local study area lakes.

Mary Lake chlorophyll-a concentrations did not consistently differ significantly from those at Reference Lake 3 over all three seasons in 2018, suggesting similar primary production between lakes. Mary Lake chlorophyll-a concentrations were continuously well below the AEMP benchmark during all seasonal sampling events in 2018, and were indicative of oligotrophic conditions normally encountered in Arctic waterbodies. Temporal evaluation of the chlorophyll-a data indicated no changes to the trophic status of Mary Lake since commencement of commercial mine operations. No significant differences in benthic invertebrate density, richness, Simpson's Evenness, and relative abundance of dominant taxonomic groups, FFG, or metal-sensitive chironomids were indicated at littoral and profundal habitat of Mary Lake compared to Reference Lake 3 in 2018, the lone exception being significantly higher density at profundal habitat of Mary Lake. In addition, no ecologically significant differences in any of the above benthic invertebrate community endpoints occurred continually between years of mine operation and the mine baseline at Mary Lake. Analysis of Mary Lake arctic charr populations suggested greater fish abundance compared to Reference Lake 3 in 2018, and suggested no substantial changes in numbers of arctic charr at Mary Lake in 2018 relative to baseline data. No ecologically significant differences in condition of YOY and non-YOY arctic charr captured at nearshore habitat occurred between Mary Lake and Reference Lake 3 in 2018. In addition, no ecologically significant difference in condition of arctic charr captured at littoral/profundal habitat occurred between Mary Lake and Reference Lake 3 in 2018, nor between 2018 and baseline studies conducted 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 since commercial mine operations commenced in 2015.

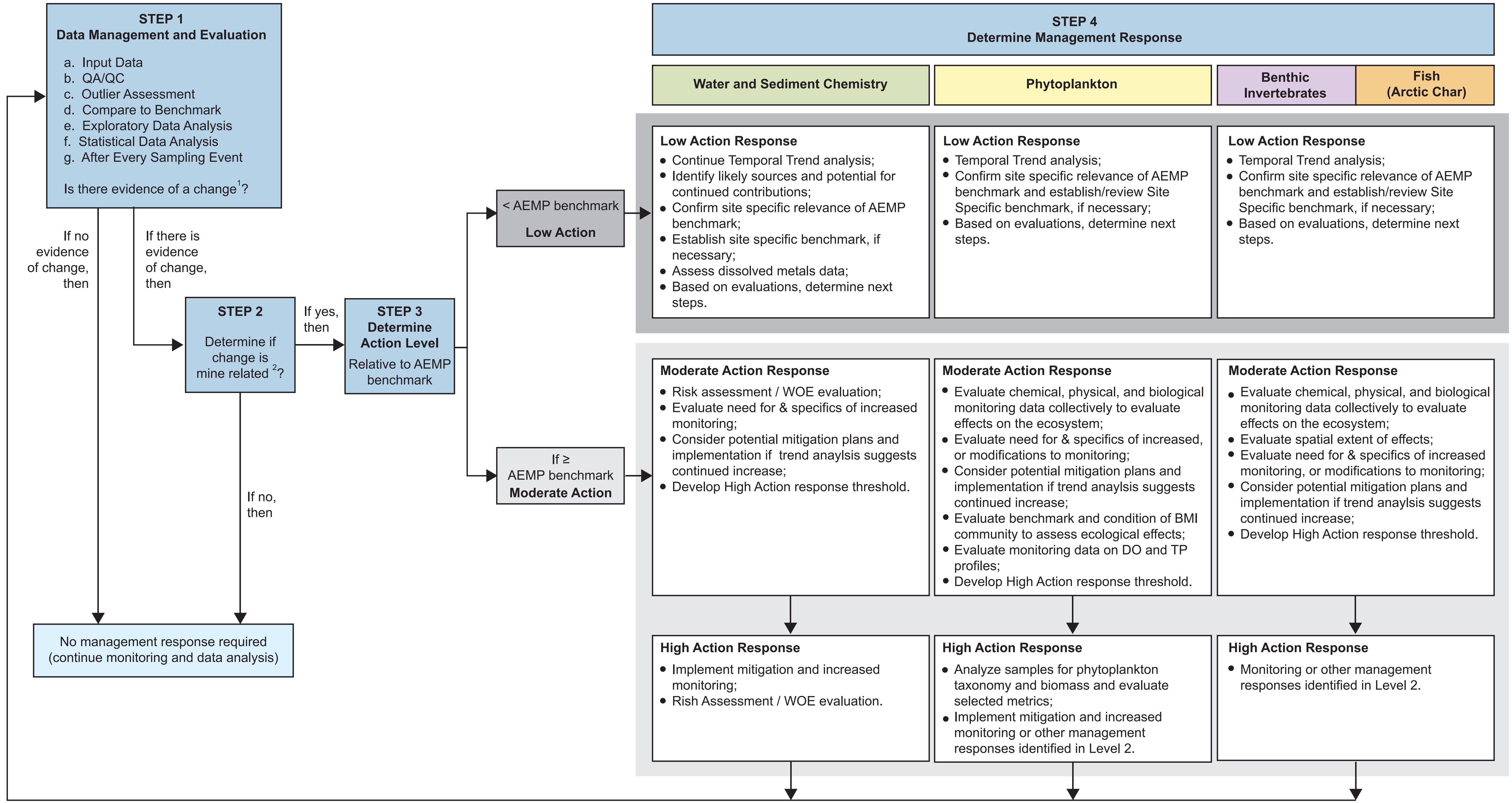
6 EFFECTS DETERMINATION AND RECOMMENDATIONS

6.1 Effects Determination Context

The objective of the 2018 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 fourth full year of mine operation. The 2018 CREMP utilized an effects-based approach that included standard environmental effects monitoring techniques to provide rigorous evaluation of potential mine-related effects at key waterbodies that receive mine-related deposits from various mine effluents, surface runoff, and among other routes, aerial deposition of dust originating from mine operations. Under this approach, water quality and sediment quality data were used to support the interpretation of phytoplankton, benthic invertebrate community, and fish population survey data collected at mine-exposed areas of the Camp Lake, Sheardown Lake, Mary River, and Mary Lake systems. The evaluation of potential mine-related effects within these systems was based upon comparisons of the 2018 data to applicable reference data, to available baseline data, and to guidelines that included site-specific AEMP benchmarks. The latter were developed to guide management response decisions within a four-step Assessment Approach and Management Response Framework as outlined in the Mary River Project AEMP (Figure 6.1; Baffinland 2015). This effects determination summarizes instances in which the Mary River Project AEMP benchmarks for water and sediment quality were exceeded at waterbodies examined under the CREMP and, based on weight-of-evidence, determines if there have been biological effects at these waterbodies to assist Baffinland with decisions regarding appropriate management actions.

6.2 Camp Lake System

Within the Camp Lake system, AEMP benchmarks for water quality were exceeded only at the main stem channel of Camp Lake Tributary 1 (CLT1), and AEMP benchmarks for sediment quality were exceeded at Camp Lake (Table 6.1). At the CLT1 main stem, aqueous concentrations of aluminum and iron were elevated above their respective AEMP benchmarks during the spring sampling event in 2018, but only at the upstream-most station (i.e., Station L2-03; Table 6.1). Total aluminum concentrations were also above the Camp Lake Tributary AEMP benchmarks at the MRY-REF3 lotic reference station during the summer sampling events in 2018. Notably, higher turbidity was evident at the CLT1 main stem and MRY-REF 3 lotic reference stations than at the other mine-exposed and reference creek stations, which suggested that elevation in total aluminum and iron concentrations compared to AEMP benchmarks reflected association of these metals with suspended particulate matter. This was corroborated by evaluation of the dissolved concentrations of aluminum and iron, which showed similar average concentrations between



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

Baffinland Mary River Project AEMP Data Assessment Approach and Response Framework

Date: March 2018
Project 177202.0033

minnow environmental inc.

Figure 6.1

Table 6.1: Summary of AEMP Benchmark Exceedances for the Mary River Project 2018 CREMP and Supporting Reference Area and Biological Effects Summary Information

Waterbody	AEMP Benchmark Exceedance	Reference Area Information	Evidence of Biological Effects at Mine-Exposed Area
Camp Lake Tributary 1 (Main Stem)	Aqueous total aluminum concentration greater than 0.179 mg/L benchmark in spring at upper main stem (0.280 mg/L). Aqueous total iron concentration greater than 0.326 mg/L benchmark in spring at upper main stem (0.439 mg/L).	Mean aluminum concentration (spring) = 0.104 mg/L (max = 0.122 mg/L) Mean iron concentration (spring) = 0.077 mg/L (max = 0.106 mg/L)	No ecologically significant and/or adverse effects on phytoplankton or benthic invertebrate community endpoints based on comparisons to reference data and to baseline data.
Camp Lake	No AEMP water quality benchmarks were exceeded at Camp Lake during spring, summer, or fall sampling events in 2018. Sediment arsenic concentration > 5.9 mg/kg benchmark at single littoral monitoring station (9.6 mg/kg). Sediment iron concentration > 52,400 mg/kg benchmark at single littoral monitoring station (65,100 mg/kg). Sediment nickel concentration > 72 mg/kg benchmark at single littoral monitoring station (83 mg/kg). Sediment phosphorus concentration > 1,580 mg/kg benchmark at single littoral monitoring station (1,650 mg/kg). Sediment arsenic, copper, and nickel concentrations above respective benchmarks at individual stations, but below benchmarks on average, at profundal stations.	Aqueous concentrations of all parameters were below applicable Water Quality Guidelines (WQG). Reference lake littoral sediment mean arsenic concentration = 5.25 mg/kg (max = 7.7 mg/kg) Reference lake littoral sediment mean iron concentration = 46,700 mg/kg (max = 74,300 mg/kg). Reference lake littoral sediment mean nickel concentration = 43 mg/kg (max = 55 mg/kg). Reference lake littoral sediment mean phosphorus concentration = 1,305 mg/kg (max = 2,290 mg/kg).	No ecologically significant and/or adverse effects on phytoplankton, benthic invertebrate community, or fish population endpoints compared to reference data and to baseline conditions.
Sheardown Lake Tributary 1	Aqueous copper concentration greater than 0.0022 mg/L benchmark in spring, summer and fall (annual mean = 0.0028 mg/L; max = 0.0036 mg/L)	Mean copper concentration (annual) = 0.0008 mg/L (max = 0.0014 mg/L)	No ecologically significant and/or adverse effects on phytoplankton or benthic invertebrate community endpoints based on comparisons to reference data and to baseline data.
Sheardown Lake NW	No AEMP water quality benchmarks were exceeded at Sheardown Lake NW during spring, summer, or fall sampling events in 2018. Profundal sediment arsenic concentration > 6.2 mg/kg benchmark (mean = 6.3 mg/kg; max = 7.6 mg/kg). Profundal sediment iron concentration > 34,400 mg/kg benchmark (mean = 53,567 mg/kg; max = 62,900 mg/kg). Profundal sediment nickel concentration > 77 mg/kg benchmark (mean = 78 mg/kg; max = 87 mg/kg).	Reference lake profundal sediment mean arsenic concentration = 6.1 mg/kg (max = 7.1 mg/kg) Reference lake profundal sediment mean iron concentration = 50,900 mg/kg (max = 63,600 mg/kg). Reference lake profundal sediment mean nickel concentration = 54 mg/kg (max = 65 mg/kg).	No ecologically significant and/or adverse effects on phytoplankton, benthic invertebrate community, or fish population endpoints compared to reference data and to baseline conditions.
Sheardown Lake SE	No AEMP water quality benchmarks were exceeded at Sheardown Lake SE during spring, summer, or fall sampling events in 2018. Mean sediment iron concentration for lake > 34,400 mg/kg benchmark (mean = 38,860 mg/kg; max = 45,800 mg/kg). Mean sediment manganese concentration for lake > 657 mg/kg benchmark (mean = 860 mg/kg; max = 1,100 mg/kg).	Reference lake mean sediment iron concentration = 48,800 mg/kg (max = 74,300 mg/kg). Reference lake mean sediment manganese concentration = 960 mg/kg (max = 1,590 mg/kg).	No ecologically significant and/or adverse effects on phytoplankton, benthic invertebrate community, or fish population endpoints compared to reference data and to baseline conditions.
Mary Lake	No AEMP water quality benchmarks were exceeded at Mary Lake during spring, summer, or fall sampling events in 2018. Sediment chromium concentration > 98 mg/kg benchmark at single littoral monitoring station (108 mg/kg) of south basin. Sediment nickel concentration > 72 mg/kg benchmark at single littoral monitoring station (73 mg/kg) of south basin. Sediment chromium, iron, and nickel concentrations above respective benchmarks at individual stations, but below benchmarks on average, at profundal stations.	Aqueous concentrations of all parameters were below applicable Water Quality Guidelines (WQG). Reference lake littoral sediment mean chromium concentration = 59 mg/kg (max = 74 mg/kg) Reference lake littoral sediment mean nickel concentration = 43 mg/kg (max = 55 mg/kg).	No ecologically significant and/or adverse effects on phytoplankton, benthic invertebrate community, or fish population endpoints compared to reference data and to baseline conditions.

CLT1 stations and the reference creek stations and suggested that aluminum and iron were not likely to be bioavailable. This was supported by the absence of any ecologically significant, adverse, effects on phytoplankton and benthic invertebrates within the CLT1 main stem in 2018 compared to both reference and baseline conditions. Based on these empirical results, a low action response to isolate the likely source(s) of aluminum and iron to the CLT1 main stem is recommended under the AEMP Management Response Framework.

At Camp Lake, AEMP benchmarks for water quality were consistently met, but benchmarks for sediment quality were exceeded for five parameters, in 2018 (Table 6.1). Arsenic, iron, nickel, and phosphorus concentrations were elevated above AEMP benchmarks in sediment at the single Camp Lake littoral sediment chemistry monitoring station. Arsenic, copper, and nickel concentrations were above AEMP benchmarks in sediment at individual profundal stations, but on average, were below benchmarks within the profundal sediments. Notably, arsenic, iron, and phosphorus concentrations in sediment of Reference Lake 3 were also above the Camp Lake AEMP benchmarks at individual littoral and profundal stations, indicating natural elevation of these parameters in lake sediments of the region (Appendix Table D.4). Sediment arsenic concentrations at the Camp Lake littoral sediment chemistry station in 2018 were slightly elevated compared to concentrations at Reference Lake 3, and concentrations at littoral habitat during baseline at Camp Lake. However, iron, nickel, and phosphorus concentrations in sediment at Camp Lake did not show similar elevation compared to reference lake conditions or Camp Lake baseline conditions for littoral or profundal stations. In addition, no adverse effects to biota in direct contact with sediment (i.e., benthic invertebrates) were indicated at Camp Lake relative to reference conditions and Camp Lake baseline conditions for both littoral and profundal habitats. Because no adverse effects to biota were associated with concentrations of metals above the AEMP benchmarks for sediment quality at Camp Lake, a moderate action response is recommended under the AEMP Management Response Framework. Notably, sediment metal concentrations were elevated above AEMP benchmarks at Reference Lake 3, sediment quality monitoring is conducted only at a single littoral station within Camp Lake, and sediment chemistry data is not always collected at the same locations as benthic invertebrate community samples, under the CREMP. Therefore, as per recommendations 14 to 17 provided by Minnow (2016b; Appendix H) following the 2015 CREMP, the following changes to the existing CREMP lake sediment quality and benthic invertebrate community survey study component designs (including Camp Lake) are recommended:

- Consider updating the AEMP sediment quality benchmarks to reflect not only baseline data, but also reference lake data; and,

- Harmonize the lake sediment quality and benthic invertebrate monitoring stations, focusing only on littoral habitat, to improve the ability of the program to evaluate mine-related effects to biota and potentially allow linkages to be assessed between sediment metal concentrations and benthic endpoints.

6.3 Sheardown Lake System

Within the Sheardown Lake system, AEMP benchmarks for water quality were exceeded only at Sheardown Lake Tributary 1 (SDLT1), and AEMP benchmarks for sediment quality were exceeded at both Sheardown Lake NW and Sheardown Lake SE, in 2018 (Table 6.1). At SDLT1, aqueous copper concentrations were elevated compared to the average concentration from reference creek stations in 2018, but not to concentrations observed at SDLT1 during baseline studies (2005 to 2013 data; Appendix Table C.35). Given close proximity to mine operations and evidence of sedimentation, a mine-related source of copper to SDLT1 seems likely, but because no elevation in copper concentrations was indicated at SDLT1 in 2018 compared to baseline conditions, copper concentrations at SDLT1 may naturally be similar to the AEMP benchmark. Biological monitoring conducted at SDLT1 in 2018 indicated no adverse effects to phytoplankton or benthic invertebrates, potentially reflecting copper concentrations at, or just marginally above, the WQG. Because no adverse effects to biota were associated with copper concentrations above the AEMP benchmark at SDLT1, a low action response to identify the likely source(s) of copper to the system is recommended under the AEMP Management Response Framework.

At Sheardown Lake NW, AEMP benchmarks for water quality were consistently met, but AEMP benchmarks for sediment quality were exceeded for arsenic, iron, and nickel at profundal habitat stations in 2018 (Table 6.1). Concentrations of these metals in profundal sediment of Sheardown Lake NW were similar to concentrations in sediment of like-habitat at Reference Lake 3, as well as to concentrations documented at Sheardown Lake NW in baseline studies (Appendix Table D.14). No adverse effects to benthic invertebrates and other biota were indicated at Sheardown Lake NW in 2018 based on comparisons to reference conditions and to Sheardown Lake NW baseline conditions. Because no adverse effects to biota were associated with concentrations of these metals above AEMP benchmarks, a low action response is recommended under the AEMP Management Response Framework for Sheardown Lake NW. Specifically, it is recommended that, because concentrations of metals in Sheardown Lake NW sediment are similar to Reference Lake 3, consideration should be given to updating the AEMP sediment quality benchmarks for Sheardown Lake NW to reflect not only baseline data, but also reference lake data.

At Sheardown Lake SE, AEMP benchmarks for water quality were consistently met, but AEMP benchmarks for sediment quality were exceeded for iron and manganese at both littoral and profundal habitat stations in 2018 (Table 6.1). Iron and manganese concentrations in sediment of Sheardown Lake SE in 2018 were similar to respective concentrations observed at Reference Lake 3, as well as to concentrations documented at Sheardown Lake SE in baseline studies. Although mean concentrations of iron and manganese were above AEMP benchmarks in sediment of Sheardown Lake SE, concentrations of these metals were also well above these AEMP benchmarks at Reference Lake 3. Notably, AEMP benchmarks established for sediment quality at Sheardown Lake SE tend to be lower than SQG, and are generally lower than AEMP benchmarks established for the other mine-exposed lakes (Baffinland 2015). No adverse effects to benthic invertebrates and other biota were indicated at Sheardown Lake SE in 2018 based on comparisons to reference conditions and to Sheardown Lake SE baseline conditions. Because no adverse effects to biota were associated with sediment iron and manganese concentrations above AEMP benchmarks at Sheardown Lake SE, a low action response is recommended under the AEMP Management Response Framework. Specifically, it is recommended that the relevance of site-specific sediment quality AEMP benchmarks for Sheardown Lake SE be assessed and, if necessary, determined anew taking into consideration data from Reference Lake 3 and applicable SQG.

6.4 Mary River and Mary Lake Systems

Within the Mary River and Mary Lake systems, AEMP benchmarks for water quality were consistently met at all Mary River stations (i.e., 13 in total) and at Mary Lake north and south basins, but benchmarks for sediment quality were exceeded for three metals at the Mary Lake south basin, in 2018 (Table 6.1). At the Mary Lake south basin, AEMP benchmarks for sediment quality were exceeded for chromium and nickel at the single littoral habitat station (Table 6.1). Chromium, iron, and nickel concentrations were also above AEMP benchmarks in sediment at individual profundal stations, but on average, were below benchmarks within sediment at profundal habitat. However, for each of these metals, similar concentrations were observed in like-habitat between Mary Lake and Reference Lake 3 in 2018, and concentrations were similar to those documented at Mary Lake in baseline studies (Appendix Table D.14). In addition, no adverse effects to benthic invertebrates and other biota were indicated at Mary Lake in 2018 based on comparisons to reference conditions and to Mary Lake baseline data. Because no adverse effects to biota were associated with concentrations of these metals above AEMP benchmarks, a low action response is recommended under the AEMP Management Response Framework for Mary Lake. Recommended changes to the existing CREMP lake sediment quality and benthic invertebrate community survey study component designs are the same as those provided previously for Camp Lake, which include:

- Consider updating the AEMP sediment quality benchmarks to reflect not only baseline data, but also reference lake data; and,
- Harmonize the lake sediment quality and benthic invertebrate monitoring stations, focusing only on littoral habitat, to improve the ability of the program to evaluate mine-related effects to biota and potentially allow linkages to be assessed between sediment metal concentrations and benthic endpoints.

7 REFERENCES

Armitage, P., P.S. Cranston, and L.C.V. Pinder. 1995. The Chironomidae: The Biology and Ecology of Non-Biting Midges. Chapman and Hall, London, U.K. 572 pp.

Baffinland (Baffinland Iron Mines Corporation). 2015. Mary River Project Aquatic Effects Monitoring Plan. Document No. BAF-PH1-830-P16-0039. Rev 0. June 27, 2014.

BCMOE (British Columbia Ministry of the Environment). 2006. A Compendium of Working Water Quality Guidelines for British Columbia. Environmental Protection Division, Victoria, British Columbia.

BCMOE. 2017. British Columbia Approved Water Quality Guidelines. http://www2.gov.bc.ca/gov/content/environment/air-land-water/water/water-quality/water-quality-guidelines/approved-water-quality-guidelines Accessed December 2017.

CCME (Canadian Council of Ministers of the Environment). 1999. Canadian Environmental Quality Guidelines. Canadian Council of Ministers of the Environment. Winnipeg, MB. With Updates to 2017.

CCME. 2017. Canadian Environmental Quality Guidelines. Canadian Council of Ministers of the Environment. Winnipeg. http://www.ccme.ca/publications/ceqg_rcqe.html Accessed December 2017.

Dodds, W.K., J.R. Jones and E.B. Welch. 1998. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Resources 12 : 1455 – 1462.

Doughman, J. 1983. A guide to the larvae of the Nearctic Diamesinae (Diptera; Chironomidae), the genera *Boreoheptagyia*, *Protanypus*, *Diamesa*, and *Pseudokiefferiella*. Water-Resources Investigations Report 83-4006

Environment Canada. 2012. Metal Mining Technical Guidance for Environmental Effects Monitoring. ISBN 978-1-100-20496-3.

Intrinsik (Intrinsik Environmental Sciences Inc.). 2014. Development of Water and Sediment Quality Benchmarks for Application in Aquatic Effects Monitoring at the Mary River Project. Report Prepared for Baffinland Iron Ore. June 26 2014.

Intrinsik. 2015. Establishment of Final Sediment Quality Aquatic Effects Monitoring Program Benchmarks. Report Prepared for Baffinland Iron Ore. March 2015.

KP (Knight Piesold Ltd.). 2014. Baffinland Iron Mines Corporation – Mary River Project – Water and Sediment Quality Review and CREMP Monitoring Report. KP Ref. No. NB102-181/33-1. June 25, 2014.

KP. 2015. Baffinland Iron Mines Corporation – Mary River Project – 2014 Water and Sediment CREMP Monitoring Report. KP Ref. No. NB102-181/34-6. March 19, 2015.

Mandaville, S.M. 2002. Benthic Macroinvertebrates in Freshwaters – Taxa Tolerance Values, Metrics and Protocols. Project H-1. Soil and Water Conservation Society of Metro Halifax.

Merritt, R.L., K.M. Cummins, and M.B. Berg. 2008. An Introduction to the Aquatic Insects of North America. 4th Ed. Kendall/Hunt Publishing, Dubuque. 1214 pp.

Minnow (Minnow Environmental Inc.). 2016a. Mary River Project 2015 Core Receiving Environment Monitoring Program Report. Prepared for Baffinland Iron Mines Corp. March 2016.

Minnow. 2016b. Mary River Project CREMP Recommendations for Future Monitoring. Letter report to Jim Millard, Baffinland Iron Mines Corp. March 17, 2016.

Minnow. 2017. Mary River Project 2016 Core Receiving Environment Monitoring Program Report. Prepared for Baffinland Iron Mines Corp. March 2017.

Minnow. 2018. Mary River Project 2017 Core Receiving Environment Monitoring Program Report. Prepared for Baffinland Iron Mines Corp. March 2018. Project 177202.0033.

Munkittrick, K.R., C.J. Arens, R.B. Lowell, and G.P. Kaminski. 2009. A review of potential methods of determining critical effect size for designing environmental monitoring programs. Environmental Toxicology and Chemistry 8: 1361 – 1371.

Namayandeh, A., K.S. Heard, E.A. Luiker, and J.M.Culp. 2016. Chironomidae (Insecta: Diptera) from the Eastern Canadian Arctic and Subarctic with descriptions of new life stages, a possible new genus, and new geographical records. Journal of Entomological and Acarological Research Vol. 48, No. 2 (2016): Special Issue on Chironomidae.

NSC (North/South Consultants Inc.). 2014. Mary River Project – Core Receiving Environment Monitoring Program: Freshwater Biota. Report prepared for Baffinland Iron Mines Corporation. June 2014.

NSC . 2015. Mary River Project - Description of Biological Sampling Completed in the Mine Area: 2014. Report prepared for Baffinland Iron Mines Corporation. March 2015.

OMOE (Ontario Ministry of Environment). 1993. Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario. August 1993, Reprinted October, 1996.

OMOEE (Ontario Ministry of Environment and Energy). 1994. Water Management: Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of Environment and Energy. July, 1994. Reprinted February 1999.

Pennak, R.W. 1989. Freshwater invertebrates of the United States. Third edition. John Wiley and Sons, Inc., New. York. 628 pp.

Smith, B. and J.B. Wilson. 1996. A consumer's guide to evenness indices. Oikos 76: 70 – 82.

Ward, J.V. 1992. Aquatic Insect Ecology, Part 1, Biology and Habitat. John Wiley and Sons, Inc. New York. 456 pp.

Wetzel, R.G. 2001. Limnology: Lake and River Ecosystems. Third Edition. Academic Press. San Diego, CA, USA. 1006 pp.

Wetzel, R.G., and G. E. Likens. 2000. Limnological Analysis. Third Edition. Springer Science + Business Media Inc. New York, NY, USA. 429 pp.