APPENDIX E.9.1

2017 CREMP MONITORING REPORT

(Part 2)

Mary River Project 2017
Core Receiving Environment Monitoring Program Report

Part 2 of 3
(Chapters 4 - 7)

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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 2017 (Figure 4.1; Appendix Tables C.1 – C.3). Although DO saturation was significantly lower at Sheardown Lake Tributary 9 (SDLT9) than at Unnamed Reference Creek and the other Sheardown Lake tributaries during August 2017 sampling, DO saturation was well above the WQG minimum limit for cold-water biota (i.e., 54%) at all Sheardown Lake tributaries (Figure 4.1; Appendix Tables C.1 – C.3). *In situ* pH was significantly higher at Sheardown Lake Tributary 1 and 12 (SDLT1 and SDLT12, respectively) compared to Unnamed Reference Creek, whereas pH at SDLT9 did not differ significantly from reference conditions during the fall sampling event in 2017. Despite minor differences in pH among the Sheardown Lake tributaries, pH was consistently within WQG limits at each mine-exposed tributary and thus the 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 2017 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). Nitrate, sulphate, and molybdenum concentrations were moderately to highly elevated (i.e., 5- to 10-fold, and ≥10-fold, respectively) at both SDLT1 stations compared to reference creek station mean concentrations at the time of fall sampling (Table 4.1). In addition, slightly elevated (i.e., 3- to 5-fold higher) total concentrations of cadmium were observed at upper SDLT1, and slightly elevated hardness and concentrations of chloride and potassium were observed at lower SDLT1, compared to reference creek stations at the time of fall sampling in 2017 (Table 4.1). Along with the aforementioned parameters, hardness, alkalinity, TDS, and total concentrations of copper, sodium, strontium, and uranium were generally elevated (i.e., ≥3-fold higher) in spring and/or summer at one or both SDLT1 monitoring stations compared to reference creek station mean values during each seasonal sampling event (Appendix Table C.35). Average dissolved copper, manganese, molybdenum, and uranium concentrations were also elevated at SDLT1 compared

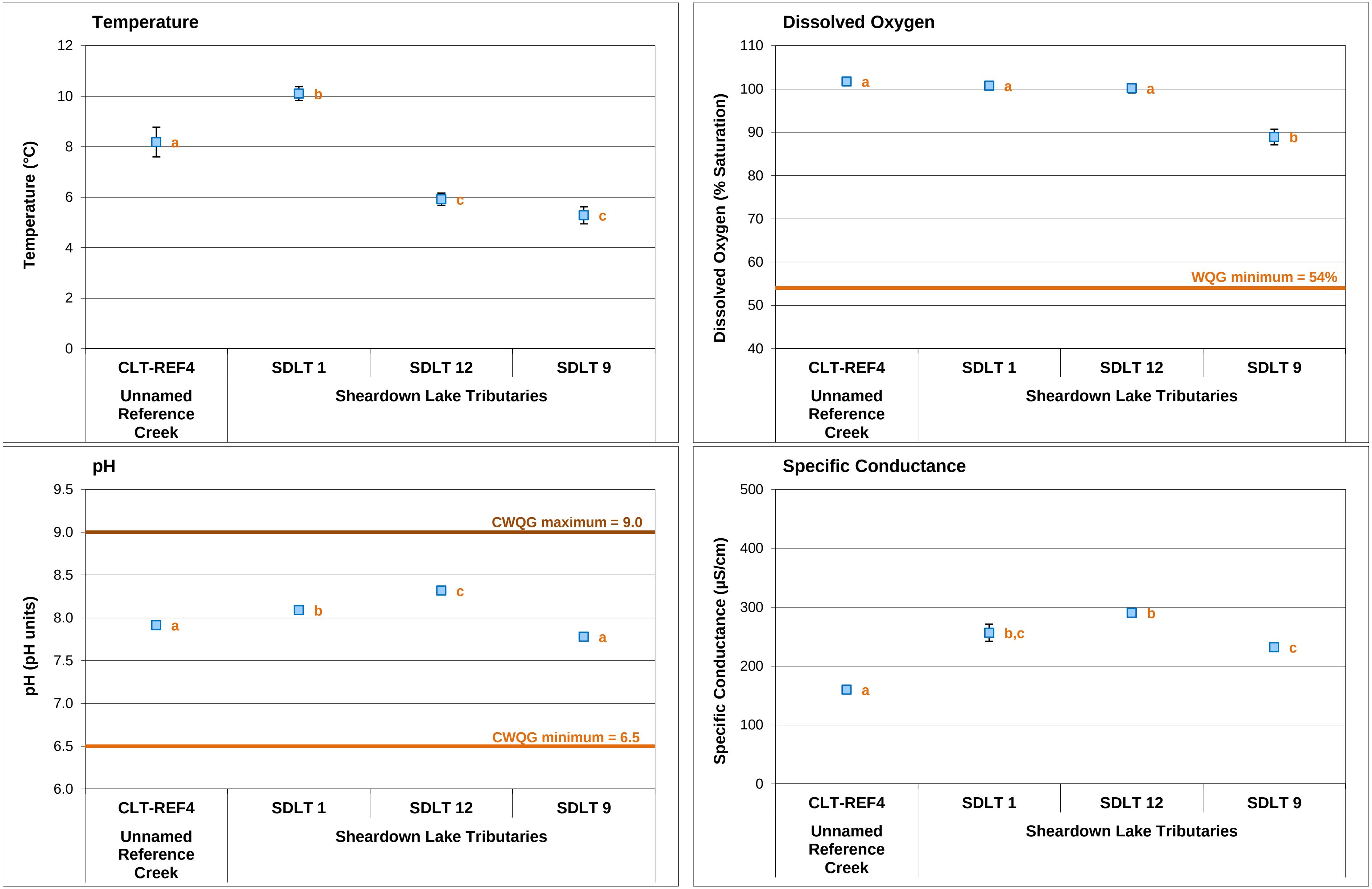


Figure 4.1: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Tributaries (SDLT) and Unnamed Reference Creek Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

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

Figure 4.2: Temporal Comparison of Water Chemistry at Sheardown Lake Tributaries (SDLT) for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015, 2016, and 2017) Periods during Fall

Notes: Values represent mean ± SD. Lotic reference stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4). Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to the Sheardown Lake Tributaries.

Table 4.1: Water Chemistry at Sheardown Lake Tributary 1 (SDLT1) Monitoring Stations, Mary River Project CREMP, Fall 2017

Parameters		Units	Water Quality Guideline (WQG)[a]	AEMP Benchmark[b]	Reference Creek Average (n = 4) Fall 2017	Sheardown Lake Tributary 1			
						D1-05 (Upper) 28-Aug-2017	D1-00 (Lower) 28-Aug-2017	D1-05 (Upper) 1-Sep-2017	D1-00 (Lower) 1-Sep-2017
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	116	215	396	200	306
	pH (lab)	pH	6.5 - 9.0	-	7.90	8.05	8.20	8.06	8.17
	Hardness (as $CaCO_3$)	mg/L	-	-	55	104	195	95	145
	Total Suspended Solids (TSS)	mg/L	-	-	4.1	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	60	80	241	111	159
	Turbidity	NTU	-	-	6.08	1.17	2.56	0.43	0.60
	Alkalinity (as $CaCO_3$)	mg/L	-	-	52	76	116	77	104
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.027	<0.020	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	0.063	0.428	1.330	0.319	0.862
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.15	<0.15	0.18	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	1.6	2.5	2.9	2.7	2.7
	Total Organic Carbon	mg/L	-	-	1.7	2.7	3.2	2.8	2.9
	Total Phosphorus	mg/L	0.030[α]	-	0.0078	0.0047	0.0031	0.0039	0.0046
	Phenols	mg/L	0.004[α]	-	<0.0010	0.0016	<0.0010	<0.0010	<0.0010
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	2.5	4.2	10.9	4.7	8.1
	Sulphate (SO_4)	mg/L	218[β]	218	4.1	18.3	61.7	15.9	37.6
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	***0.208***	0.017	0.075	0.009	0.016
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	0.00012	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0076	0.0101	0.0206	0.0091	0.0150
	Boron (B)	mg/L	1.5	-	<0.010	0.013	0.014	<0.010	0.011
	Cadmium (Cd)	mg/L	0.00012	0.00008	<0.000010	0.000034	0.000010	0.000037	0.000015
	Chromium (Cr)	mg/L	0.0089	0.00856	0.00074	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	0.00015	<0.00010	0.00015	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0022	0.0013	***0.0032***	0.0021	***0.0031***	***0.0024***
	Iron (Fe)	mg/L	0.30	0.326	0.222	<0.050	0.154	<0.030	0.045
	Lead (Pb)	mg/L	0.001	0.001	0.00021	<0.000050	0.00011	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	0.0014	0.0011	0.0016	0.0012	0.0018
	Manganese (Mn)	mg/L	0.935[β]	-	0.00332	<0.00050	0.00697	0.00027	0.00254
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00034	0.00403	0.00267	0.00343	0.00282
	Nickel (Ni)	mg/L	0.025	0.025	0.0007	0.0011	0.0016	0.0012	0.0013
	Potassium (K)	mg/L	-	-	0.82	2.26	2.73	2.13	2.34
	Selenium (Se)	mg/L	0.001	-	<0.000050	0.000111	0.00011	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	1.26	1.28	1.63	1.36	1.55
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000050	<0.000050	<0.000050	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	1.75	2.57	5.83	2.23	3.98
	Strontium (Sr)	mg/L	-	-	0.0113	0.0116	0.0216	0.0104	0.0165
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00001	0.00001	0.00001	<0.00010	<0.00010
	Titanium (Ti)	mg/L	-	-	0.0123	0.0007	<0.0050	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00214	0.00616	0.00567	0.00554	0.00629
	Vanadium (V)	mg/L	0.006[α]	0.006	0.00075	<0.00050	<0.00050	<0.0010	<0.0010
	Zinc (Zn)	mg/L	0.030	0.030	<0.0030	<0.0030	0.0047	<0.0030	<0.0030

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

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

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

BOLD Indicates parameter concentration above the AEMP benchmark.

to average reference creek conditions during all seasonal sampling events in 2017, strongly suggesting a mine-related source for these parameters. Despite elevation of these parameters at the SDLT1 stations compared to reference conditions, copper was the only parameter present at concentrations greater than respective WQG or AEMP benchmarks at either of the SDLT1 monitoring stations in 2017 (Table 4.1; Appendix Table C.34).

Temporal comparisons of SDLT1 water chemistry data indicated that, of the parameters shown to be elevated above average reference conditions, nitrate, sodium, and sulphate concentrations were moderately to highly elevated (i.e., ≥5-fold higher) at upper and/or lower SDLT1 in 2017 compared to respective baseline period conditions in at least one sampling season (Figure 4.2; Appendix Table C.35 and Figure C.9). The SDLT1 concentrations of these parameters, and uranium, were elevated compared to baseline conditions in 2015 and 2016 as well, suggesting a mine-related source of these metals since the initiation of mine operations at the Mary River Project. Notably, total copper concentrations at SDLT1 in 2017 were comparable to those during the baseline period, suggesting naturally high concentrations of this metal within this tributary (Appendix Figure C.9).

4.1.2 Sediment Quality

Deposited sediment at SDLT1 and SDLT12 study areas was visually characterized as reddish brown silt, whereas deposited sediment at SDLT9 was described as medium to coarse sand (Appendix Table D.19). Although natural in-stream substrate at tributaries SDLT1 and SDLT12 is composed almost entirely of cobble and boulder material, fine reddish-brown silt occurred as a precipitate on the natural substrate and formed thicker deposits interstitially, in slow flowing areas, and along the shoreline, at both tributaries (Appendix Table F.1). In contrast, cobble composes the primary substrate type at SDLT9, but natural sand can constitute as much as 5 – 10% of the surficial bed material observed at this tributary to Sheardown Lake SE (Appendix Table F.1). No silt precipitate or deposits were observed at SDLT9 during the August 2017 sampling event. Deposited sediment was collected mainly from shoreline/streambank areas at SDLT1, and predominantly in-stream from between and behind boulders at SDLT12 and SDLT9 study areas (Appendix Table D.19). Sediment TOC content was low (i.e., <1%) at all of the Sheardown Lake Tributary study areas, but was slightly (i.e., 3- to 5-fold higher) to moderately (i.e., 5-fold to 10-fold higher) elevated compared to average lotic reference area TOC content (Table 4.2; Appendix Table D.21). This suggested a more depositional environment and/or greater suspended sediment loads at the three Sheardown Lake tributaries compared to reference conditions.

Metal concentrations in deposited sediment at both SDLT1 and SDLT12 were generally elevated compared to average lotic reference area metal concentrations (Table 4.2; Appendix Table D.21). In particular, concentrations of aluminum, arsenic, barium, chromium, cobalt, copper, iron, lead,

Table 4.2: Sediment Total Organic Carbon and Metal Concentrations at Sheardown Lake Tributaries (SDLT1, 12, and 9) and Applicable Reference Creek and River Sediment Monitoring Stations, Mary River Project CREMP, August 2017

Parameter	Units	Sediment Quality Guideline (SQG)[a]	Lotic Reference Stations		Sheardown Lake Tributaries		
			Unnamed Reference Creek (REFCRK; n = 5)	Mary River Reference (GO-09; n = 5)	Sheardown Trib 1 SDLT1 (n = 5)	Sheardown Trib 12 SDLT12 (n = 3)	Sheardown Trib 9 SDLT9 (n = 5)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD	Average ± SD
Total Organic Carbon	%	10[α]	<0.10 ± 0.00	<0.10 ± 0.00	0.64 ± 0.07	0.46 ± 0.15	0.63 ± 0.14
Aluminum (Al)	mg/kg	-	418 ± 126	763 ± 271	15,366 ± 4,088	9,450 ± 1,631	2,538 ± 722
Antimony (Sb)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	0.13 ± 0.03	0.19 ± 0.02	<0.10 ± 0
Arsenic (As)	mg/kg	17	0.12 ± 0.02	0.16 ± 0.02	2.74 ± 0.90	2.88 ± 0.90	0.64 ± 0.08
Barium (Ba)	mg/kg	-	2 ± 0.3	4 ± 1.2	81 ± 24.0	33 ± 7.7	11 ± 3
Beryllium (Be)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	0.70 ± 0.18	0.72 ± 0.14	0.12 ± 0.02
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	0.71 ± 0.23	0.55 ± 0.19	0.20 ± 0.00
Boron (B)	mg/kg	-	<5.0 ± 0	<5.0 ± 0	7.1 ± 0.8	5.2 ± 0.4	5.1 ± 0.2
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	<0.020 ± 0	0.280 ± 0.051	0.095 ± 0.040	0.036 ± 0.010
Calcium (Ca)	mg/kg	-	214 ± 44	842 ± 508	4,254 ± 2,051	1,433 ± 425	1,660 ± 422
Chromium (Cr)	mg/kg	90	1.4 ± 0.4	3.4 ± 1.0	33.5 ± 5.6	33.2 ± 4.0	13.5 ± 0.6
Cobalt (Co)	mg/kg	-	0.32 ± 0.09	0.67 ± 0.19	13.79 ± 3.68	15.63 ± 1.50	2.36 ± 0.62
Copper (Cu)	mg/kg	110[α]	0.8 ± 0.5	1.3 ± 0.5	41 ± 8.2	36 ± 10.7	7.2 ± 2.2
Iron (Fe)	mg/kg	40,000[α]	1,240 ± 475	2,826 ± 1,271	155,000 ± 65,662	272,667 ± 69,759	15,000 ± 5,251
Lead (Pb)	mg/kg	91	0.7 ± 0.2	1.1 ± 0.1	23.6 ± 9.0	14.2 ± 5.6	3.6 ± 0.8
Lithium (Li)	mg/kg	-	<2.0 ± 0.0	2.2 ± 0.5	15.8 ± 3.3	10.2 ± 1.3	3.9 ± 0.7
Magnesium (Mg)	mg/kg	-	333 ± 116	826 ± 521	13,540 ± 2,415	7,237 ± 1,331	2,572 ± 895
Manganese (Mn)	mg/kg	1,100[α,β]	10 ± 2.4	22 ± 7.1	780 ± 160	1,056 ± 159	121 ± 52
Mercury (Hg)	mg/kg	0.486	0.0050 ± 0	<0.0050 ± 0	0.0104 ± 0.0019	0.0053 ± 0.0006	0.0103 ± 0.0087
Molybdenum (Mo)	mg/kg	-	<0.10 ± 0.00	0.11 ± 0.01	5.38 ± 2.04	5.28 ± 2.06	0.43 ± 0.27
Nickel (Ni)	mg/kg	75[α,β]	0.8 ± 0.2	1.9 ± 0.7	34.8 ± 5.6	34.9 ± 5.3	10.7 ± 2.4
Phosphorus (P)	mg/kg	2,000[α]	61 ± 16	113 ± 49	410 ± 73	388 ± 28	321 ± 83
Potassium (K)	mg/kg	-	106 ± 13	168 ± 77	5,808 ± 2,051	2,223 ± 297	582 ± 199
Selenium (Se)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	0.30 ± 0.04	0.24 ± 0.05	0.21 ± 0.03
Silver (Ag)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	0.26 ± 0.14	0.19 ± 0.08	<0.10 ± 0.00
Sodium (Na)	mg/kg	-	<50 ± 0	<50 ± 0	117 ± 32	50 ± 0	51 ± 3
Strontium (Sr)	mg/kg	-	1.2 ± 0.2	1.9 ± 0.3	3.9 ± 0.7	2.3 ± 0.4	2.6 ± 0.4
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	<0.050 ± 0.000	<0.050 ± 0	0.302 ± 0.074	0.136 ± 0.032	0.064 ± 0.009
Tin (Sn)	mg/kg	-	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0
Titanium (Ti)	mg/kg	-	36 ± 16	109 ± 26	877 ± 258	382 ± 47	247 ± 38
Uranium (U)	mg/kg	-	0.2 ± 0.1	0.30 ± 0.0	3.2 ± 0.8	2.8 ± 0.2	0.60 ± 0.1
Vanadium (V)	mg/kg	-	1.9 ± 0.7	5.0 ± 2.2	28.5 ± 4.6	18.0 ± 2.5	12.9 ± 1.6
Zinc (Zn)	mg/kg	315	2.0 ± 0.0	3.7 ± 1.3	87 ± 18.3	44 ± 10.0	11 ± 2.9
Zirconium (Zr)	mg/kg	-	1.1 ± 0.1	1.7 ± 0.3	6.6 ± 1.7	3.9 ± 0.5	1.0 ± 0.0

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

Indicates parameter concentration above Sediment Quality Guideline (SQG).

magnesium, manganese, molybdenum, nickel, uranium, and zinc were highly elevated (i.e., ≥10-fold higher) in deposited sediment at both of these tributaries compared to average lotic reference area concentrations (Appendix Table D.21). In part, elevated metal concentrations in the deposited sediment collected at SDLT1 and SDLT12 may reflect finer substrate size and more depositional features of these tributaries compared to average lotic reference area conditions. On average, metal concentrations in deposited sediment at SDLT1 and SDLT12 were below applicable SQG with the exception of iron, which occurred at mean concentrations approximately four-times and seven-times higher than the SQG, respectively, at these tributaries (Table 4.2; Appendix Tables D.20 and D.22). The concentration of manganese was also slightly above SQG in deposited sediment at one of the three SDLT12 replicate stations (Appendix Table D.22). Deposited sediment at SDLT9 showed slightly (i.e., 3- to 5-fold higher) to moderately (i.e., 5-fold to 10-fold higher) elevated concentrations of most of the metals indicated above compared to the lotic reference areas, but concentrations of all metals including iron were well below SQG (Table 4.2; Appendix Table D.21). Limited baseline sediment metal concentration data were collected at the Sheardown Lake tributaries, and thus no evaluation of potential mine-related influences on sediment quality following commencement of commercial mine operations could be conducted.

4.1.3 Phytoplankton

Phytoplankton (chlorophyll-a) monitoring is conducted only at SDLT1 within the Sheardown Lake system as part of the Mary River Project CREMP. Chlorophyll-a concentrations were lower at upper SDLT1 (Station D1-05) compared to near the creek mouth (Station D1- 00) during each of the spring, summer, and fall sampling events in 2017 (Figure 4.3). Higher chlorophyll-a concentrations observed near the mouth of SDLT1 may have reflected the occurrence of elevated nutrient concentrations and specifically, higher aqueous nitrate concentrations, compared to the upper SDLT1 station (Section 4.1.1). Chlorophyll-a concentrations at SDLT1 were higher than average reference creek concentrations in both spring and summer, but were substantially lower than the average reference creek concentration in the fall (Figure 4.3). For all sampling events in 2017, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L at both SDLT1 stations (Figure 4.3). Similar to the reference creek stations and Camp Lake tributary systems, 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 2017 were also consistent with an oligotrophic WQG categorization based on aqueous phosphorus concentrations near or below 10 µg/L (Table 4.1; Appendix Table C.34).

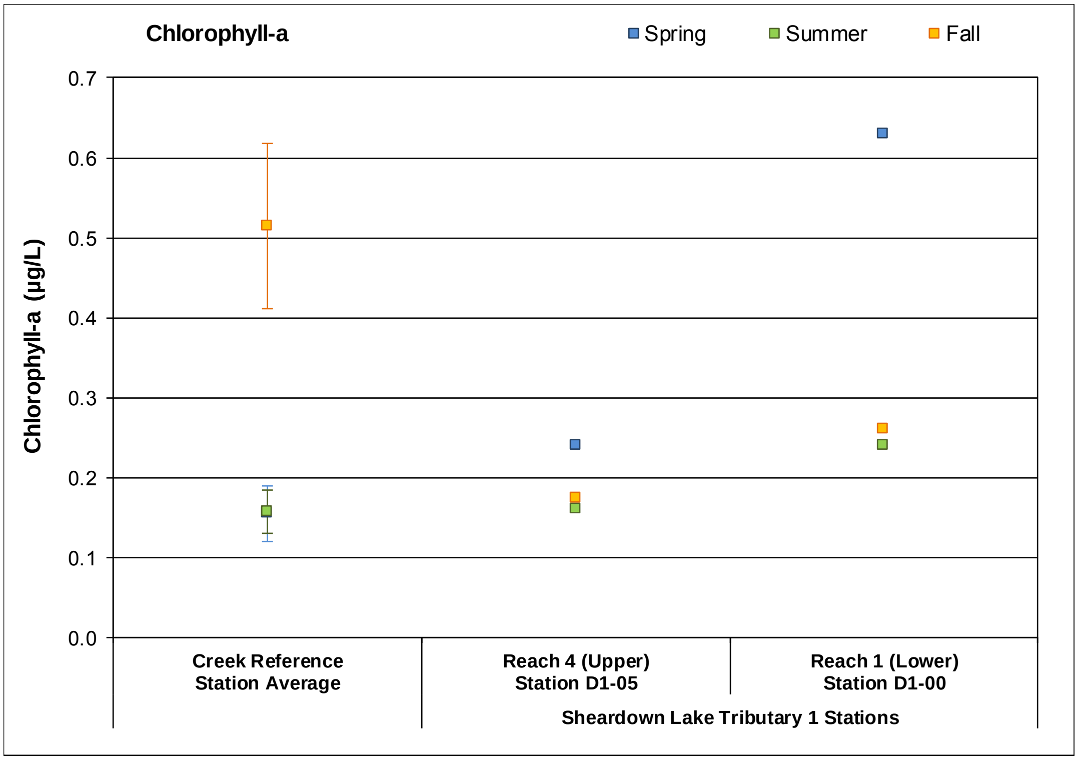


Figure 4.3: Chlorophyll-a Concentrations at Sheardown Lake Tributary 1 Phytoplankton Monitoring Stations, Mary River Project CREMP, 2017

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

Temporal comparisons indicated chlorophyll-a concentrations at SDLT1 stations in 2017 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 over the mine baseline (2005 – 2013), construction (2014), and operational (2015, 2016, 2017) periods (Figure 4.4). These data suggested no adverse mine-related influences to phytoplankton productivity at SDLT1 over the initial three years of mine operation.

4.1.4 Benthic Invertebrate Community

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 composition (as indicated by Bray-Curtis Index) compared to Unnnamed Reference Creek in 2017 (Figure 4.5; Appendix Table F.32). The key differences in relative abundance of dominant taxonomic groups included ecologically significant greater proportions of Oligochaeta (aquatic worms) and metal-sensitive Chironomidae (non-biting midges), and conversely, an absence of Ephemeroptera (mayflies) and significantly lower proportion of Simuliidae (blackflies)

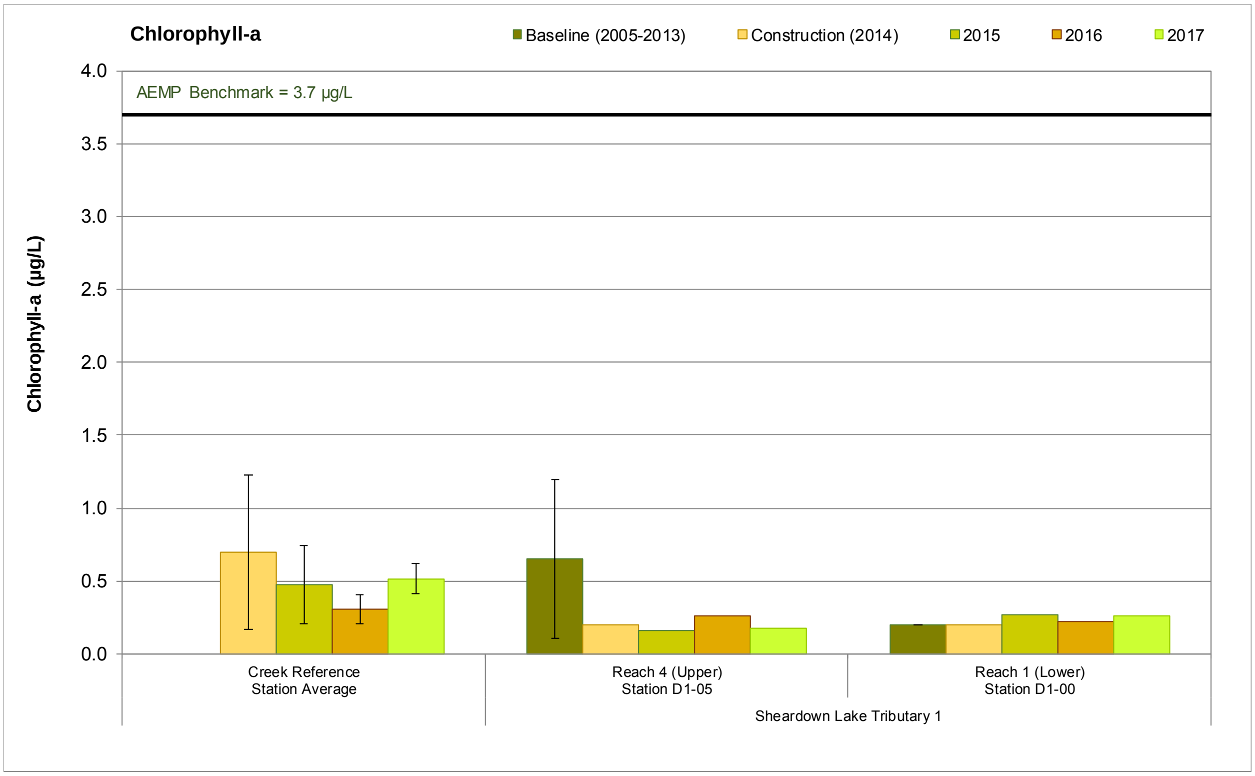


Figure 4.4: Temporal Comparison of Chlorophyll-a Concentrations at Sheardown Lake Tributary 1 for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015 - 2017) Periods in the Fall, Mary River Project CREMP

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

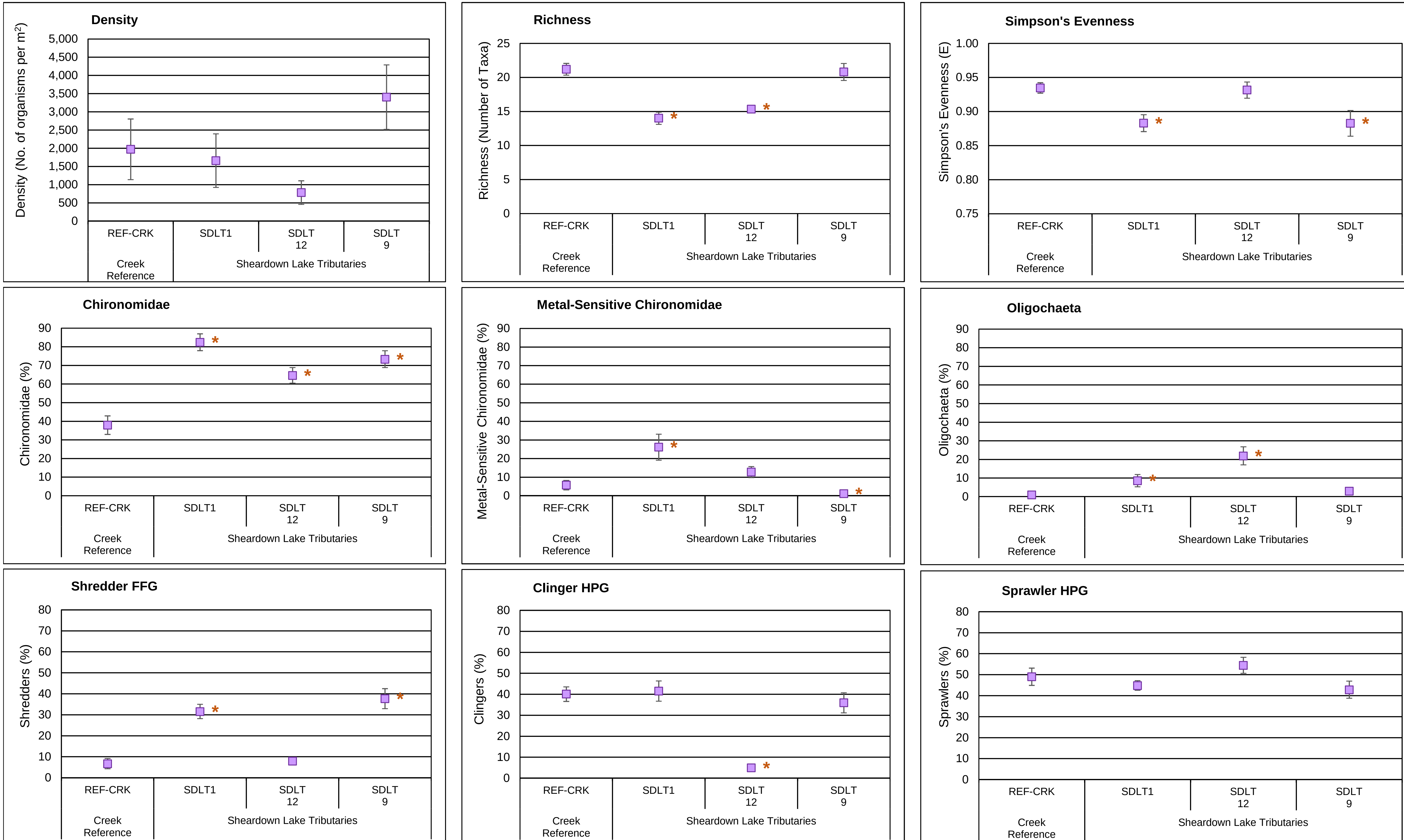


Figure 4.5: Comparison of Benthic Invertebrate Community Metrics between Sheardown Lake Tributary and Unnamed Reference Creek Study Areas (mean ± SE), Mary River Project CREMP, August 2017

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

at SDLT1 compared to Unnamed Reference Creek (Appendix Table F.32). However, of these, only absolute densities of mayflies differed significantly at magnitudes outside of CES_{BIC} between SDLT1 and the reference creek (Appendix Table F.33). A higher relative abundance of metal-sensitive chironomids at lower SDLT1 suggested that metal concentrations were not biologically available and/or did not account for differences in community composition compared to Unnamed Reference Creek, which was consistent with concentrations of all metals but copper and iron below WQG and SQG, respectively, at SDLT1 in 2017 (see Appendix Table C.34). A significantly higher relative abundance of FFG shredders (Appendix Table F.32), 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.25). In turn, this suggested that differences in in-stream vegetation likely contributed to differing benthic invertebrate community composition between SDLT1 and Unnamed Reference Creek. Notably, no significant differences in relative abundance of HPG were indicated between SDLT1 and the reference creek (Appendix Table F.32), 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, no definitive mine-related influences on the benthic invertebrate community of SDLT1 were indicated by the 2017 data.

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, 2016, 2017) compared to baseline studies conducted in 2008 and 2013 (Figure 4.6; Appendix Table F.34). 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 Table F.34). The absence of any consistent, ecologically significant changes in benthic invertebrate community density, richness, Simpson's Evenness, and composition at lower 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.

Sheardown Lake Tributary 12 (SDLT12)

The benthic invertebrate community of Sheardown Lake Tributary 12 (SDLT12) did not differ significantly from Unnamed Reference Creek for primary endpoints of density and Simpson's Evenness, but richness at SDLT12 was significantly lower at a magnitude outside of CES_{BIC}, in 2017 (Figure 4.5; Appendix Table F.32). In addition, marked differences in community composition were indicated between SDLT12 and the reference creek based on significant differences in Bray-Curtis Index and several dominant invertebrate, functional feeding, and habitat preference groups (Figure 4.5; Appendix Table F.32). Because the relative abundance of metal-

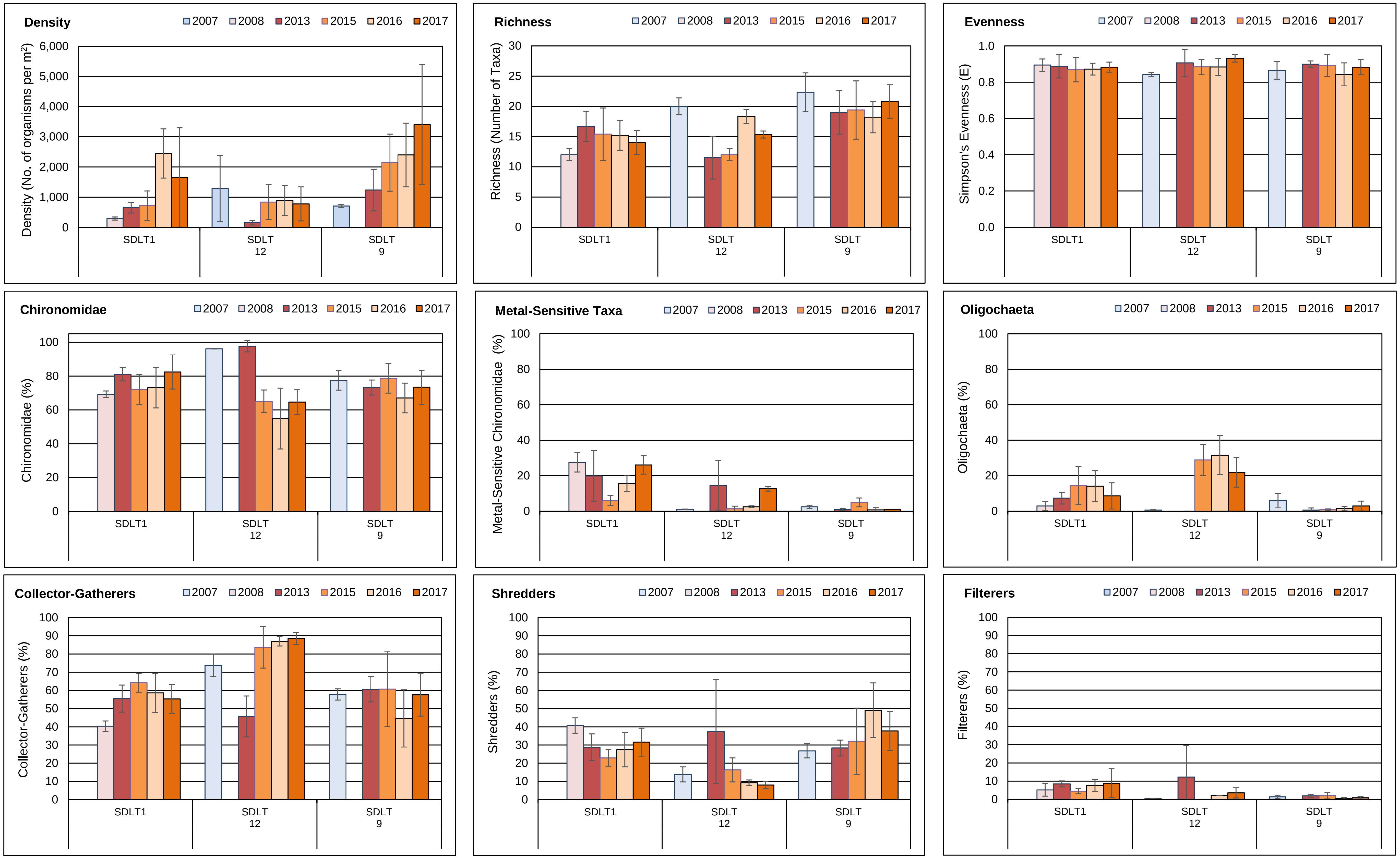


Figure 4.6: Comparison of Benthic Invertebrate Community Metrics (mean ± SD) at Sheardown Lake Tributaries 1, 12, and 9 among Operational (2015, 2016, 2017) and Baseline (2007, 2008, 2011, 2013) Studies for the Mary River Project CREMP

sensitive chironomids did not differ significantly between areas (Figure 4.5; Appendix Table F.32), the differences in community composition between SDLT12 and Unnamed Reference Creek were not likely directly related to metal concentrations. Rather, significantly higher relative abundance of the burrower HPG, which include Oligochaeta (aquatic worms) and Tipulidae (crane flies) taxonomic groups, as well as significantly greater relative abundance of FFG collector-gatherer deposit feeders, was consistent with the occurrence of significantly slower water velocity (i.e., more depositional habitat) at SDLT12 than at Unnamed Reference Creek (Appendix Tables F.28 and F.32). Therefore, differing habitat features between SDLT12 and Unnamed Reference Creek potentially accounted for the differences in benthic invertebrate community composition between watercourses.

Temporal comparison of the SDLT12 benthic invertebrate community data did not indicate any on-going significant differences in density, richness, or Simpson's Evenness between mine operational years and baseline data collected in 2007 (Figure 4.6; Appendix Table F.35). However, the consistent occurrence of significantly higher relative abundance of burrowing invertebrates, including aquatic worm and crane fly taxonomic groups, and the collector-gatherer FFG in 2015, 2016, and 2017 compared to the 2007 baseline data suggested changes in habitat conditions with the commencement of mine operations. Although such temporal changes potentially reflected slight differences in sampling location between the mine operational and baseline periods, field observations from the 2016 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 that included a shift to higher abundance of deposit feeding, burrowing benthic invertebrates. Notably, the relative abundance of metal-sensitive chironomids was significantly higher, or showed no difference, between the mine operational and baseline period at SDLT12, suggesting that metals were largely biologically unavailable and/or did not account for the differences in benthic invertebrate community endpoints among these mine periods.

Sheardown Lake Tributary 9 (SDLT9)

The benthic invertebrate community of Sheardown Lake Tributary 9 (SDLT9) did not differ significantly from Unnamed Reference Creek for primary endpoints of density or richness in 2017 (Figure 4.5; Appendix Table F.32). However, significantly lower Simpson's Evenness was indicated at SDLT9 than at the reference creek. In addition, marked differences in community composition were indicated between SDLT9 and Unnamed Reference Creek based on significant differences in Bray-Curtis Index and several groups of dominant taxa and FFG (Figure 4.5;

Appendix Table F.32). However, the magnitude of difference in the relative abundance of metal-sensitive chironomids between SDLT9 and the reference creek was within the CES_{BIC} of ±2 SD_{REF} (Figure 4.5; Appendix Table F.32), suggesting that differences in community composition between watercourses were unlikely to be related to differing metal concentrations. A significantly higher relative abundance of the shredder FFG, which included taxa represented by Tipulidae (crane flies), was consistent with field observations of greater amounts of rooted in-stream vegetation at SDLT9 compared to the reference creek (Appendix Tables F.25 and F.32) given that plants serve as a food source for the shredder FFG. 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. Notably, no significant differences in the relative abundance of HPG were indicated between SDLT9 and the reference creek (Appendix Table F.32), suggesting that a mine-related factor such as increased sedimentation had not substantially altered benthic invertebrate community composition at SDLT9 compared to reference conditions.

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 mine operational period data collected in 2015, 2016, and 2017 compared to baseline period data collected in 2007 and 2013 (Figure 4.6; Appendix Table F.36). In turn, this suggested that the differences in benthic invertebrate community composition between SDLT9 and Unnamed Reference Creek in 2017 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.

4.1.5 Integrated Effects Evaluation

At Sheardown Lake Tributary 1 (SDLT1), aqueous concentrations of several parameters were elevated compared to average concentrations observed at the reference creek stations in 2017. However, similar to previous CREMP studies, only nitrate, sodium, and sulphate concentrations were elevated at SDLT1 in 2017 compared to the baseline period and, with the exception of copper, no parameters were present at concentrations above WQG or AEMP benchmarks in 2017. Mine-related sedimentation was evident at SDLT1 by the presence of reddish brown silt precipitate/deposits containing highly elevated concentrations of several metals compared to the reference creek. Although the accumulation of deposited sediment affected less than 1% of surficial bed material at SDLT1, iron concentrations of the deposited sediment were considerably higher than SQG. Chlorophyll-a concentrations at SDLT1 were higher in spring and summer, but lower in fall, compared to reference creek stations in 2017, suggesting that elevated nitrate concentrations may contribute variably to biological enrichment at SDLT1. However, similar chlorophyll-a concentrations between 2017 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 2017 compared to Unnamed Reference Creek. However, no significant difference in the relative abundance of metal-sensitive chironomids were indicated between SDLT1 and the reference creek in 2017. In addition, no consistent ecologically significant differences in any primary benthic invertebrate community metrics, dominant taxonomic groups or FFG were indicated for individual years of mine operation (2015, 2016, 2017) compared to baseline studies conducted in 2008 and 2013 at SDLT1. In turn, this suggested that metals in water and sediment at SDLT1 were not highly bio-available, and that differences in benthic invertebrate community composition between SDLT1 and the reference creek reflected natural differences in the amount and/or types of in-stream vegetation between watercourses. Overall, similar to the findings of the two previous CREMP studies, no adverse mine-related effects to biota of SDLT1 were indicated in 2017 based on the chlorophyll-a and benthic invertebrate community data analysis.

At Sheardown Lake Tributary 12 (SDLT12), mine-related sedimentation was evident as deposits of reddish brown silt which was shown to contain high concentrations of several metals compared to deposited sediment of the reference creek. Although the accumulation of deposited sediment affected less than 5% of surficial bed material at SDLT12, iron concentrations of the deposited sediment highly exceeded the applicable SQG. The SDLT12 benthic invertebrate community showed a significantly higher relative abundance of collector-gatherers and burrowers relative to Unnamed Reference Creek in 2017, and at SDLT12 between studies conducted in 2015, 2016, and 2017 relative to 2007 baseline data. The temporal changes in benthic invertebrate community composition at SDLT12 are hypothesized to reflect a mine-related reduction in flow and/or increased particle loadings (e.g., through dust and/or erosional deposition) over time at SDLT12, constituting a potential mine-related effect at this tributary.

At Sheardown Lake Tributary 9 (SLDT9), deposited sediment contained slight to moderate elevation in metal concentrations compared to the reference creek, but concentrations of all metals, including iron, were well below SQG in the SDLT9 deposited sediment. Significantly lower benthic invertebrate evenness, as well as significant differences in community structure, were indicated at SDLT9 in 2017 compared to the reference creek. However, no significant difference in the relative abundance of metal-sensitive chironomids were indicated between SDLT9 and the reference creek in 2017, and no consistent ecologically significant differences in any primary benthic invertebrate community metrics were indicated for years of mine operation (2015, 2016, 2017) compared to the baseline studies. Examination of FFG and HPG differences between SDLT9 and the reference creek indicated that naturally differing amounts and/or types of in-stream vegetation likely accounted for the differing benthic invertebrate community structure between these watercourses. Overall, no adverse mine-related effects to biota of SDLT9 were indicated in 2017 based on the benthic invertebrate community data analysis.

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 using mean annual watershed runoff (2007 – 2016 data) 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 (from NSC 2015b).

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 2017 showed no substantial station-to-station differences during any of the winter, summer, or fall sampling events (Appendix Figures C.11 – C.14). No thermal stratification was indicated at Sheardown Lake NW during winter or summer, but a weak thermocline developed at depths extending from 14 m to lake bottom during the fall (Figure 4.7). Average water temperature at the bottom of the water column at Sheardown Lake NW littoral and profundal stations was slightly warmer than at Reference Lake 3 at the time of fall sampling in 2017, the difference of which was statistically significant only for the littoral stations (Figure 4.8). However, the incremental difference in average bottom water temperature between lakes at each respective depth was small (i.e., ≤0.5˚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 and fall, but no appreciable change in dissolved oxygen saturation from surface to bottom in the summer of 2017 (Figure 4.7; Appendix Figure C.12). No substantial oxycline was observed at Reference Lake 3 during the summer or fall sampling events in 2017 (Figure 4.7). Despite the slight difference in dissolved oxygen profiles between Sheardown Lake NW and Reference Lake 3 during the fall sampling event, dissolved oxygen saturation levels at the bottom of the water column did not differ significantly between these study lakes at littoral and profundal stations in 2017 (Figure 4.8; Appendix Table C.37). In addition, dissolved oxygen saturation levels were consistently well above the WQG of 54% at Sheardown Lake NW during all winter, summer, and fall sampling events in 2017 (Figures 4.7 and 4.8). This indicated that dissolved oxygen was not limiting for pelagic or bottom-dwelling biota within Sheardown Lake NW in 2017.

In situ profiles of pH and specific conductance showed no substantial change from the surface to bottom of the Sheardown Lake NW water column during any of the three sampling seasons in

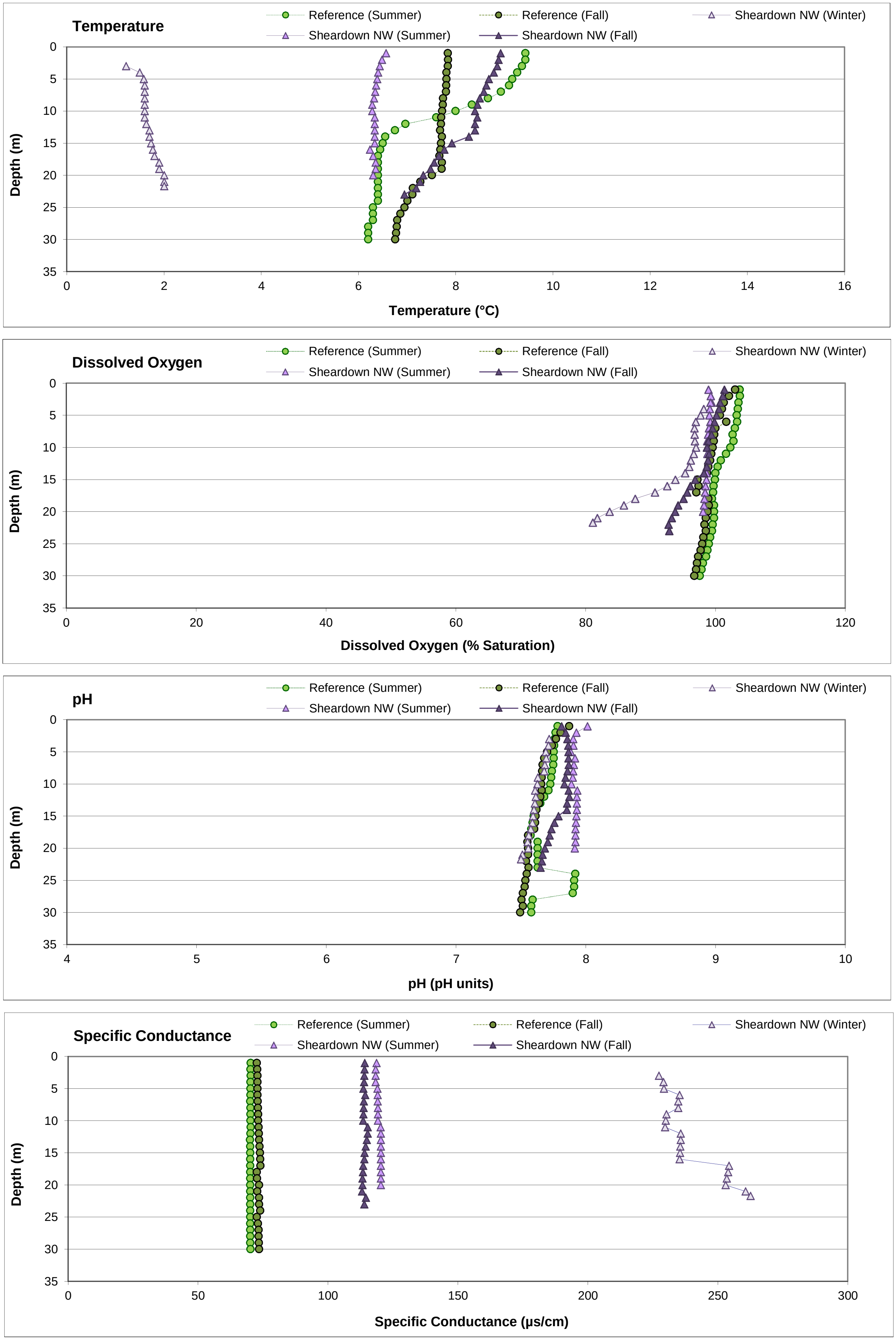


Figure 4.7: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake NW (DLO-01) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2017

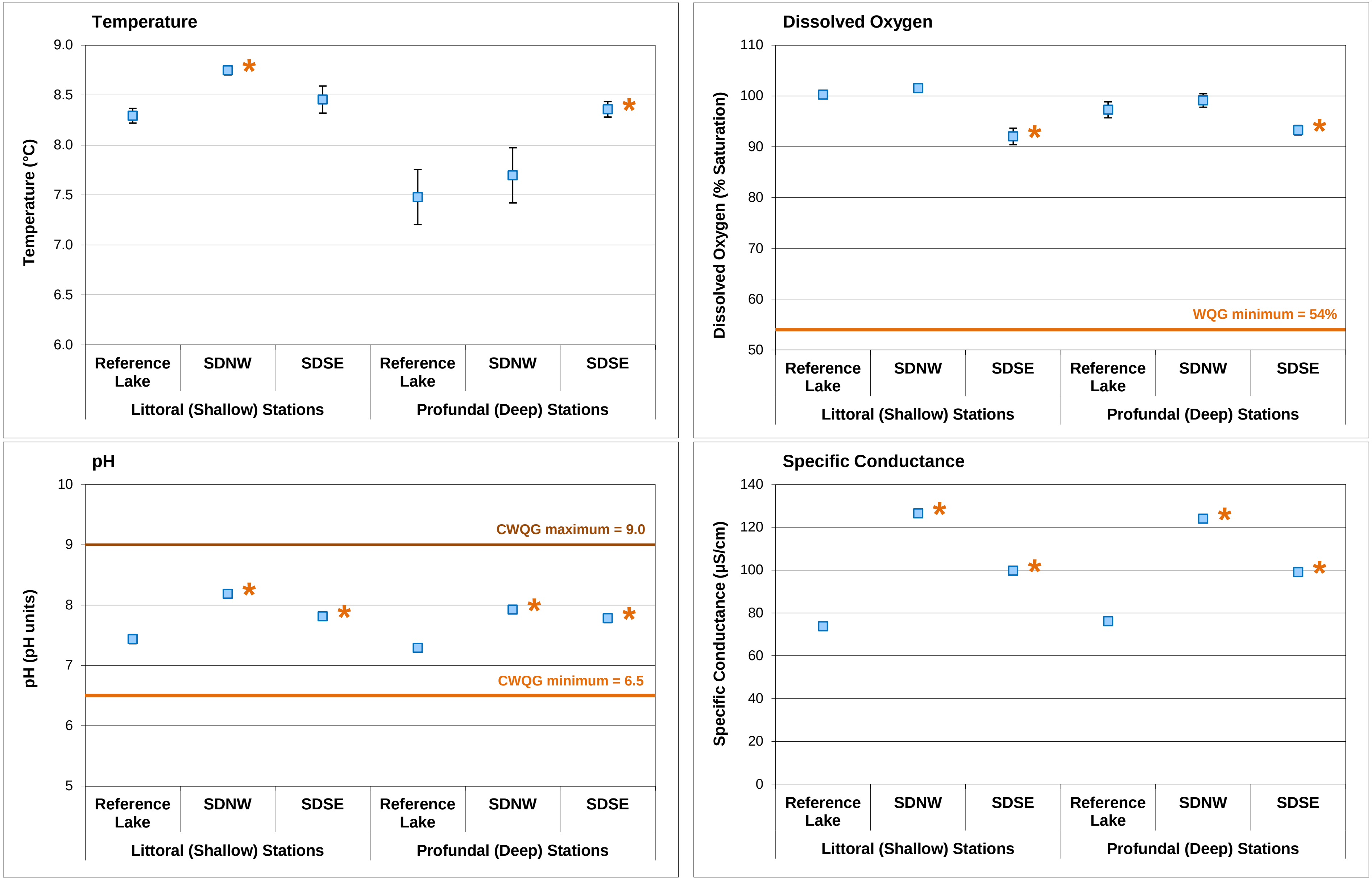


Figure 4.8: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Sheardown Lake Basins (SDNW and SDSE) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

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

2017, 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 2017 (Figure 4.8; Appendix Table C.37). However, pH values were consistently within WQG limits of 6.5 – 9.0 through the entire water column during all 2017 sampling events conducted at Sheardown Lake NW (Figures 4.7 and 4.8; Appendix Tables C.33 – C.36). Specific conductance was significantly higher at Sheardown Lake NW compared to the reference lake during fall sampling (Figure 4.8; Appendix Table C.42). However, similar to observations at Camp Lake (Section 4.2.1), specific conductance at Sheardown Lake NW was intermediate to that of reference creek and river stations in fall 2017 (i.e., range from 55 – 168 µS/cm). 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 2017 fall sampling event (Appendix Table C.42; Appendix Figure C.7). Secchi depth readings showed relatively low variability among stations at Sheardown Lake NW in the fall of 2017, suggesting no spatial differences in water clarity throughout the lake (Appendix Table C.40).

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 2017 (Table 4.3; Appendix Table C.43), suggesting that the lake waters were continually well mixed both laterally and vertically. Turbidity and total concentrations of aluminum, manganese, molybdenum, tin, and uranium were slightly (3- to 5-fold higher) to moderately (5- to 10 fold higher) elevated at Sheardown Lake NW compared to Reference Lake 3 during the summer and/or fall sampling events (Table 4.3; Appendix Tables C.43 and C.44). Similar to the previous studies, total aluminum and manganese concentrations showed a strong positive correlation with turbidity at Sheardown Lake NW in 2017 (r_s = 0.79 and 0.85, respectively; Appendix Table C.47). This suggested that elevated total aluminum and manganese concentrations at Sheardown Lake NW reflected influences associated with surface runoff or backflow received from Mary River that contained naturally high concentrations of aluminum-based, manganese bearing, particulate minerals. This was supported through evaluation of dissolved metal concentrations, which indicated similar dissolved aluminum and manganese concentrations between Sheardown Lake NW and the reference lake (Appendix Table C.46), and the absence of a strong correlation between dissolved concentrations of these metals and turbidity. In addition, the ratio of dissolved to total concentrations of aluminum and manganese indicated that the majority (i.e., 76% and 89%, respectively) of these metals was in the total fraction at Sheardown Lake NW based on the 2017 data. Total and dissolved concentrations of molybdenum and uranium were each elevated at Sheardown Lake NW compared to the reference

Table 4.3: Water Chemistry at Sheardown Lake NW (DLO-01) and Reference Lake 3 (REF3) Monitoring Stations[a], Mary River Project CREMP, August 2017

Parameters		Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3)	Sheardown Lake NW Station					
						DD-HAB9 STN1	DL0-01-5	DL0-01-1	DL0-01-4	DL0-01-2	DL0-01-7
					Fall 2017	21-Aug-2017	21-Aug-2017	26-Aug-2017	20-Aug-2017	20-Aug-2017	20-Aug-2017
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	76	132	130	132	131	130	130
	pH (lab)	pH	6.5 - 9.0	-	7.76	8.06	7.86	7.99	8.01	8.04	8.01
	Hardness (as $CaCO_3$)	mg/L	-	-	35	62	61	61	62	62	62
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	2.85	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	33	54	52	70	49	63	53
	Turbidity	NTU	-	-	0.69	1.09	0.97	0.66	0.91	1.06	0.99
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31	54	56	58	62	52	52
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.020	<0.020	<0.020	0.038	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	0.030	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	0.465	0.1725	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.7	1.6	1.6	1.5	1.7	1.5	1.6
	Total Organic Carbon	mg/L	-	-	2.8	1.7	1.6	1.6	1.7	1.6	1.7
	Total Phosphorus	mg/L	0.020[α]	-	0.004	0.005	0.004	0.005	0.004	0.006	0.005
	Phenols	mg/L	0.004[α]	-	0.002	<0.0010	<0.0010	0.002	<0.0010	<0.0010	<0.0010
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	0.1	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.3	2.8	3.0	3.0	3.0	2.9	3.0
	Sulphate (SO_4)	mg/L	218[β]	218	4.0	4.9	4.9	5.0	5.1	4.9	5.1
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.005	0.012	0.011	0.010	0.013	0.011	0.013
	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	-	-	0.00642	0.00643	0.00609	0.00624	0.00623	0.00619	0.00616
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	6.9	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	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.0008	0.0008	0.0009	0.0008	0.0012	0.0008	0.0008
	Iron (Fe)	mg/L	0.30	0.300	<0.030	<0.030	<0.030	<0.030	<0.030	<0.030	<0.030
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	<0.0010	0.0010	0.0012	<0.0010	<0.0010	<0.0010
	Magnesium (Mg)	mg/L	-	-	4.4	7.9	7.7	7.9	7.7	7.5	7.5
	Manganese (Mn)	mg/L	0.935[β]	-	0.00063	0.00179	0.00173	0.00160	0.00175	0.00183	0.00178
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.00012	0.00075	0.00075	0.00074	0.00075	0.00074	0.00075
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	0.00064	0.00061	0.00058	0.00064	0.00062	0.00059
	Potassium (K)	mg/L	-	-	0.88	1.18	1.13	1.14	1.15	1.14	1.14
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.41	0.41	0.47	0.46	0.42	0.41	0.42
	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	-	-	0.85	1.46	1.40	1.32	1.42	1.39	1.43
	Strontium (Sr)	mg/L	-	-	0.0078	0.0079	0.0078	0.0080	0.0079	0.0078	0.0079
	Tin (Sn)	mg/L	-	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Uranium (U)	mg/L	0.015	-	0.00025	0.00084	0.00087	0.00080	0.00089	0.00088	0.00088
	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.

lake, but concentrations of these metals at Sheardown Lake NW were not strongly correlated with turbidity, suggesting a potential mine-related source. Despite elevation of total aluminum, manganese, molybdenum, and uranium metals at Sheardown Lake NW compared to Reference Lake 3, concentrations of each of these metals, as well as all other parameters, were well below applicable WQG and AEMP benchmarks at Sheardown Lake NW during all sampling events in 2017 (Table 4.3; Appendix Table C.43).

Temporal comparisons of the Sheardown Lake NW water chemistry data suggested that 2017 seasonal average total and dissolved concentrations of most parameters were within each respective range of baseline concentrations (2005 – 2013; Figure 4.9; Appendix Figure C.19). However, a key exception was dissolved molybdenum, which showed slight elevation (i.e., 3- to 5-fold higher) in 2017 compared to the baseline data based on fall sampling results (Appendix Table C.44). In addition, turbidity and concentrations of molybdenum, sodium, and sulphate showed successively higher concentrations over years of mine-construction (2014) through mine operation (2015 – 2017; Figure 4.9; Appendix Figure C.19; Appendix Table C.44). Overall, the magnitude of these changes over time were relatively minor and unlikely to be ecologically meaningful given concentrations remained well below WQG, but nevertheless the sequential increases were consistent with greater mine-related influence on water quality over time at Sheardown Lake NW.

4.2.3 Sediment Quality

Surficial sediment at Sheardown Lake NW showed substrate properties varying from silt loam to sand at littoral areas, and substrate consistently represented by silt loam at profundal areas (Figure 4.10; Appendix Table D.25). Although no significant differences in sediment particle size were indicated between Sheardown Lake NW and the reference lake for littoral stations, sediment at profundal stations of Sheardown Lake NW was composed of significantly less sand and significantly more silt than at Reference Lake 3 (Figure 4.10; Appendix Table D.5). In addition, sediment at littoral and profundal stations of Sheardown Lake NW contained a significantly lower proportion of TOC than at the reference lake (Figure 4.10; Appendix Table D.5). 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 Table D.24). 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 did not exhibit any substantial blackening (or unusually dark colouration) or noticeable sulphidic odour at the time of the August 2017 sampling event, suggesting the absence of strongly reducing conditions in the sediment (Appendix Table D.24).

Figure 4.9: Temporal Comparison of Water Chemistry at Sheardown Lake Northwest (DLO-01) and Sheardown Lake Southeast (DLO-02) for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015, 2016, 2017) Periods during Fall

Notes: Values represent mean ± SD. Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Sheardown Lake (northwest and southeast).

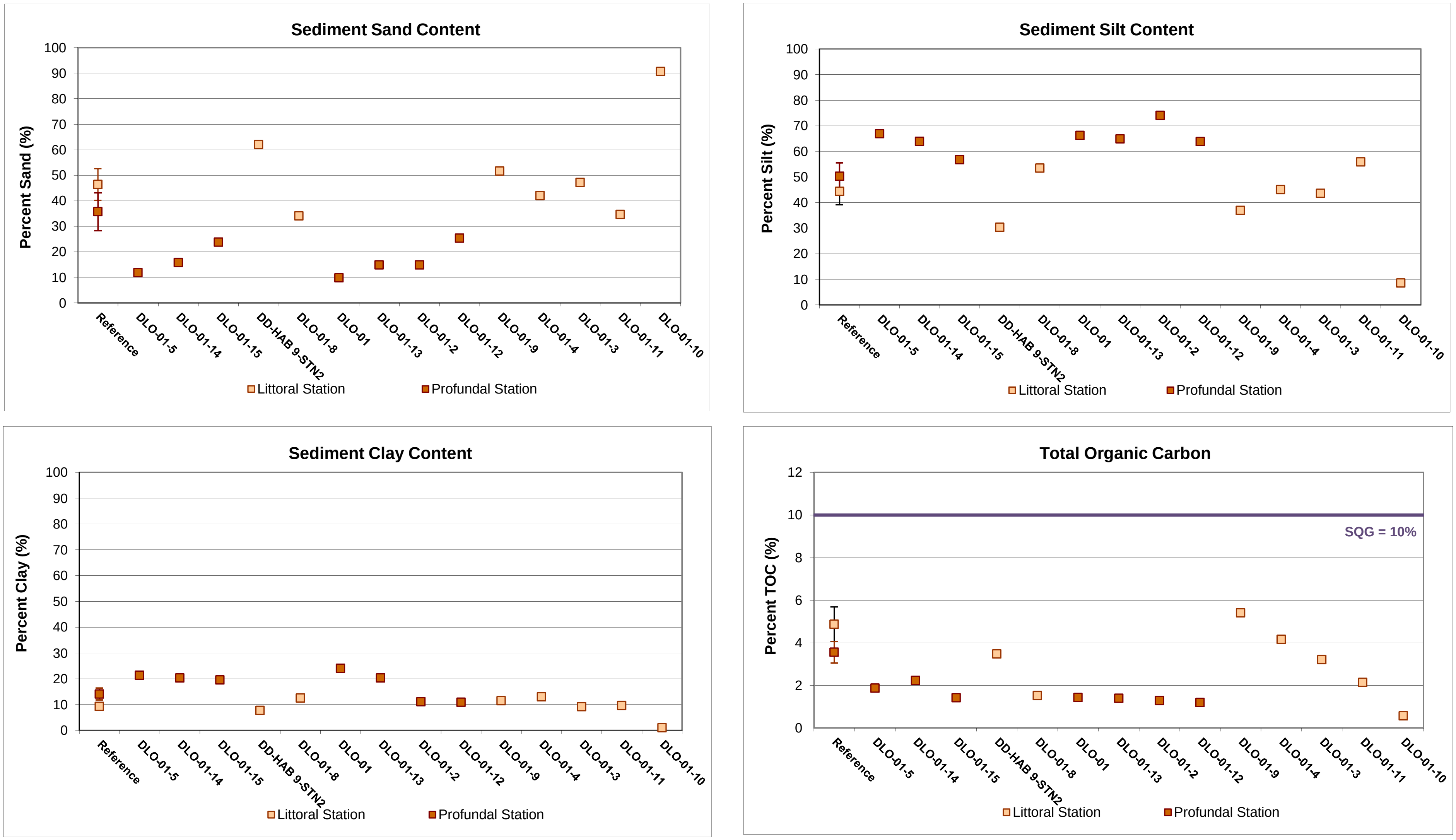


Figure 4.10: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake NW (DLO-01) Sediment Monitoring Stations and Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2017

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 2017 (Appendix Table D.26). However, sediment iron concentrations appeared to be highest at Sheardown Lake NW stations situated closest to the outlets of SDLT1 and SDLT12 (Stations DD-HAB 0-STN2 and DLO-01-9, respectively; Appendix Table D.26). Iron concentrations in deposited sediment at SDLT1 and SDLT12 were considerably higher than sediment of Sheardown Lake NW (Appendix Table D.29), indicating that these tributaries were a source of iron loadings to the lake. Concentrations of other metals were generally similar to or higher in sediment of Sheardown Lake NW compared to deposited sediment of SDLT1 and SDLT12 (Appendix Table D.29), reflecting the depositional nature of the lake versus erosional characteristics of the tributaries. Sediment metal concentrations at littoral and profundal stations of Sheardown Lake NW were very similar to averages observed for the same respective station types at Reference Lake 3 in 2017 (Table 4.4; Appendix Table D.27), suggesting no marked mine-related influences on sediment metal concentrations in Sheardown Lake NW. Although mean concentrations of iron were above SQG in sediment at littoral and profundal stations of Sheardown Lake NW, mean concentrations of iron were also above SQG at both station types of Reference Lake 3 in 2017 (Table 4.4). On average, manganese concentrations were above SQG in sediment at profundal stations, as were concentrations of manganese and nickel in sediment at individual littoral stations, of Sheardown Lake NW in 2017 (Table 4.4; Appendix Table D.26). However, iron and manganese concentrations were elevated above SQG in sediment at littoral and profundal stations of Reference Lake 3 (Table 4.4; Appendix Table D.6), indicating naturally elevated concentrations of these metals in sediment of local study area lakes. Although iron, manganese, and nickel concentrations were above AEMP benchmarks at individual littoral and profundal stations, on average, concentrations of these and all other metals were below their respective AEMP benchmarks at Sheardown Lake NW in 2017 (Table 4.4; Appendix Table D.26).

Temporal comparisons indicated that metal concentrations in sediment at littoral and profundal stations of Sheardown Lake NW in 2017 were comparable to those observed during the mine baseline (2005 – 2013) period (Figure 4.11; Appendix Table D.27). On average, metal concentrations in sediment at respective Sheardown Lake NW littoral and profundal stations in 2017 were typically within the range of those observed from 2015 and 2016, with no consistently higher concentrations occurring that would suggest an increasing trend over time (Figure 4.11; Appendix Table D.27). This contrasted with the results of the previous CREMP study, which suggested progressively higher concentrations of arsenic, barium, iron, manganese, and molybdenum at littoral stations of Sheardown Lake NW from baseline, to mine construction, to 2015 and 2016 mine operational years (Figure 4.11; Minnow 2017). Changes in station replication and location among studies likely contributed to the appearance of greater mean

Table 4.4: 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 2017

	Parameter	Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b] (NW, SE)	Littoral: Reference Lake (n = 5)	Littoral: Sheardown Lake NW (n=4)	Littoral: Sheardown Lake SE (n=3)	Profundal: Reference Lake (n = 5)	Profundal: Sheardown Lake NW (n=4)	Profundal: Sheardown Lake SE (n=2)
					Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error	Average ± Std. Error
Non-metals	Sand	%	-	-	46 ± 6	60 ± 12	9 ± 2	36 ± 7	13 ± 1	15 ± 2
	Silt	%	-	-	44 ± 5	32 ± 9	76 ± 8	50 ± 5	68 ± 2	70 ± 2
	Clay	%	-	-	9 ± 1	8 ± 3	14 ± 7	14 ± 2	19 ± 3	15 ± 0.4
	Moisture	%	-	-	81 ± 5	57 ± 12	51 ± 4	76 ± 6	69 ± 1	46 ± 4
	Total Organic Carbon	%	10[α]	-	4.9 ± 0.8	2.7 ± 1.1	1.2 ± 0.1	3.6 ± 0.5	1.5 ± 0.13	1.1 ± 0.05
Metals	Aluminum (Al)	mg/kg	-	-	14,720 ± 736	15,683 ± 4,186	16,917 ± 705	22,140 ± 1,652	22,200 ± 883	18,450 ± 150
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	0.11 ± 0.005	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0.00	<0.10 ± 0
	Arsenic (As)	mg/kg	17	6.2, 5.9	3.35 ± 0.64	5.28 ± 1.87	4.83 ± 1.14	4.80 ± 0.20	4.88 ± 0.63	3.06 ± 0.61
	Barium (Ba)	mg/kg	-	-	113 ± 10	99 ± 31	110 ± 19	138 ± 7.0	113 ± 9	87 ± 6.6
	Beryllium (Be)	mg/kg	-	-	0.57 ± 0.02	0.77 ± 0.21	0.79 ± 0.043	0.85 ± 0.0573	1.09 ± 0.057	0.85 ± 0.000
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	0.30 ± 0.063	0.22 ± 0.009	<0.2 ± 0.0	0.25 ± 0.016	0.20 ± 0.0000
	Boron (B)	mg/kg	-	-	12.1 ± 0.80	26.4 ± 6.96	21.4 ± 1.41	16.6 ± 1.487	33.7 ± 2.58	27.3 ± 0.15
	Cadmium (Cd)	mg/kg	3.5	1.5, 1.5	0.182 ± 0.032	0.355 ± 0.114	0.122 ± 0.00753	0.179 ± 0.024	0.259 ± 0.008	0.119 ± 0.0005
	Calcium (Ca)	mg/kg	-	-	4,656 ± 362	4,223 ± 1,048	6,072 ± 95	5,262 ± 377	4,598 ± 126	6,225 ± 465
	Chromium (Cr)	mg/kg	90	97, 79	51.1 ± 3.7	61 ± 15	79 ± 4.3	70.2 ± 6.3	84 ± 1.9	78 ± 1.60
	Cobalt (Co)	mg/kg	-	-	10.6 ± 0.794	12.9 ± 3.49	14.0 ± 0.6	16.0 ± 1.1	17.4 ± 0.4	14.0 ± 0.450
	Copper (Cu)	mg/kg	110	58, 56	***64.7*** ± 5.27	43 ± 13	28 ± 1.3	***87.6*** ± 8.5	47 ± 1.8	29 ± 0.20
	Iron (Fe)	mg/kg	40,000[α]	52,200, 34,400	***41,960*** ± 8,713	46,323 ± 14,520	***44,867*** ± 4,734	***46,740*** ± 3,447	43,938 ± 2,530	***39,150*** ± 2,150
	Lead (Pb)	mg/kg	91.3	35	12.4 ± 0.6	16.2 ± 4.13	16.2 ± 1.04	16.9 ± 1.0	21.9 ± 0.8	15.9 ± 0.050
	Lithium (Li)	mg/kg	-	-	24.0 ± 0.9	25.5 ± 6.75	30.1 ± 1.41	35.0 ± 2.1	37.7 ± 1.6	31.8 ± 0.15
	Magnesium (Mg)	mg/kg	-	-	10,256 ± 714	10,503 ± 2,619	14,167 ± 338	14,660 ± 1,126	14,488 ± 315	14,650 ± 250.0
	Manganese (Mn)	mg/kg	1,100[α,β]	4,530, 657	639 ± 115	664 ± 281	***1,506*** ± 588	***1,266*** ± 70	3,389 ± 2,141	***820*** ± 350.5
	Mercury (Hg)	mg/kg	0.486	0.17	0.0440 ± 0.0081	0.0355 ± 0.0118	0.0262 ± 0.00186	0.0528 ± 0.0089	0.0388 ± 0.00440	0.0299 ± 0.0008
	Molybdenum (Mo)	mg/kg	-	-	3.50 ± 0.97	4.40 ± 1.70	1.77 ± 0.469	2.90 ± 0.20	3.63 ± 1.69	1.10 ± 0.270
	Nickel (Ni)	mg/kg	75[α,β]	77, 66	38.2 ± 2.4	67.7 ± 19.2	62.4 ± 4.06	49.3 ± 3.0	70.3 ± 0.9	57.4 ± 2.050
	Phosphorus (P)	mg/kg	2,000[α]	1,958, 1,278	1,039 ± 246	908 ± 208	1,188 ± 122	1,073 ± 54	1,028 ± 111	1,050 ± 80.5
	Potassium (K)	mg/kg	-	-	3,754 ± 212	4,240 ± 1,142	4,315 ± 214	5,694 ± 366	5,896 ± 265	4,780 ± 30
	Selenium (Se)	mg/kg	-	-	0.61 ± 0.12	0.46 ± 0.12	0.20 ± 0.00	0.59 ± 0.120	0.33 ± 0.035	0.23 ± 0.0250
	Silver (Ag)	mg/kg	-	-	0.13 ± 0.01	0.15 ± 0.020	0.11 ± 0.000	0.20 ± 0.025	0.17 ± 0.011	0.12 ± 0.0100
	Sodium (Na)	mg/kg	-	-	284 ± 21	237 ± 60	274 ± 14	410 ± 36	323 ± 15	313 ± 6.5
	Strontium (Sr)	mg/kg	-	-	10.0 ± 0.461	9.5 ± 1.71	10.7 ± 0.450	12.7 ± 0.80	12.0 ± 0.34	11.8 ± 0.200
	Sulphur (S)	mg/kg	-	-	1,120 ± 96.9536	1,175 ± 175	<1,000 ± 0	1,140 ± 74.8	<1,000 ± 0	<1,000 ± 0
	Thallium (Tl)	mg/kg	-	-	0.35 ± 0.036	0.44 ± 0.13	0.38 ± 0.026	0.657 ± 0.038	0.57 ± 0.01	0.37 ± 0.0020
	Tin (Sn)	mg/kg	-	-	2.12 ± 0.12	<2.0 ± 0	<2.0 ± 0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0
	Titanium (Ti)	mg/kg	-	-	997 ± 51	984 ± 229	1,240 ± 15	1,288 ± 56	1,313 ± 35.0	1,455 ± 45.0
	Uranium (U)	mg/kg	-	-	11.0 ± 1.04	7.40 ± 2.69	4.71 ± 0.207	23.5 ± 2.26	7.64 ± 0.44	5.04 ± 0.015
	Vanadium (V)	mg/kg	-	-	47.8 ± 2.04	47.3 ± 12.0	51.1 ± 1.91	67.2 ± 3.86	63.8 ± 1.62	54.0 ± 1.050
	Zinc (Zn)	mg/kg	315	135	67.3 ± 2.51	55.5 ± 13.8	57.3 ± 2.08	91.1 ± 6.76	72.3 ± 1.68	59.6 ± 0.500
	Zirconium (Zr)	mg/kg	-	-	3.9 ± 0.4	10.0 ± 3.66	15.3 ± 0.89	3.38 ± 0.150	8.06 ± 0.69	19.9 ± 1.45

[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.

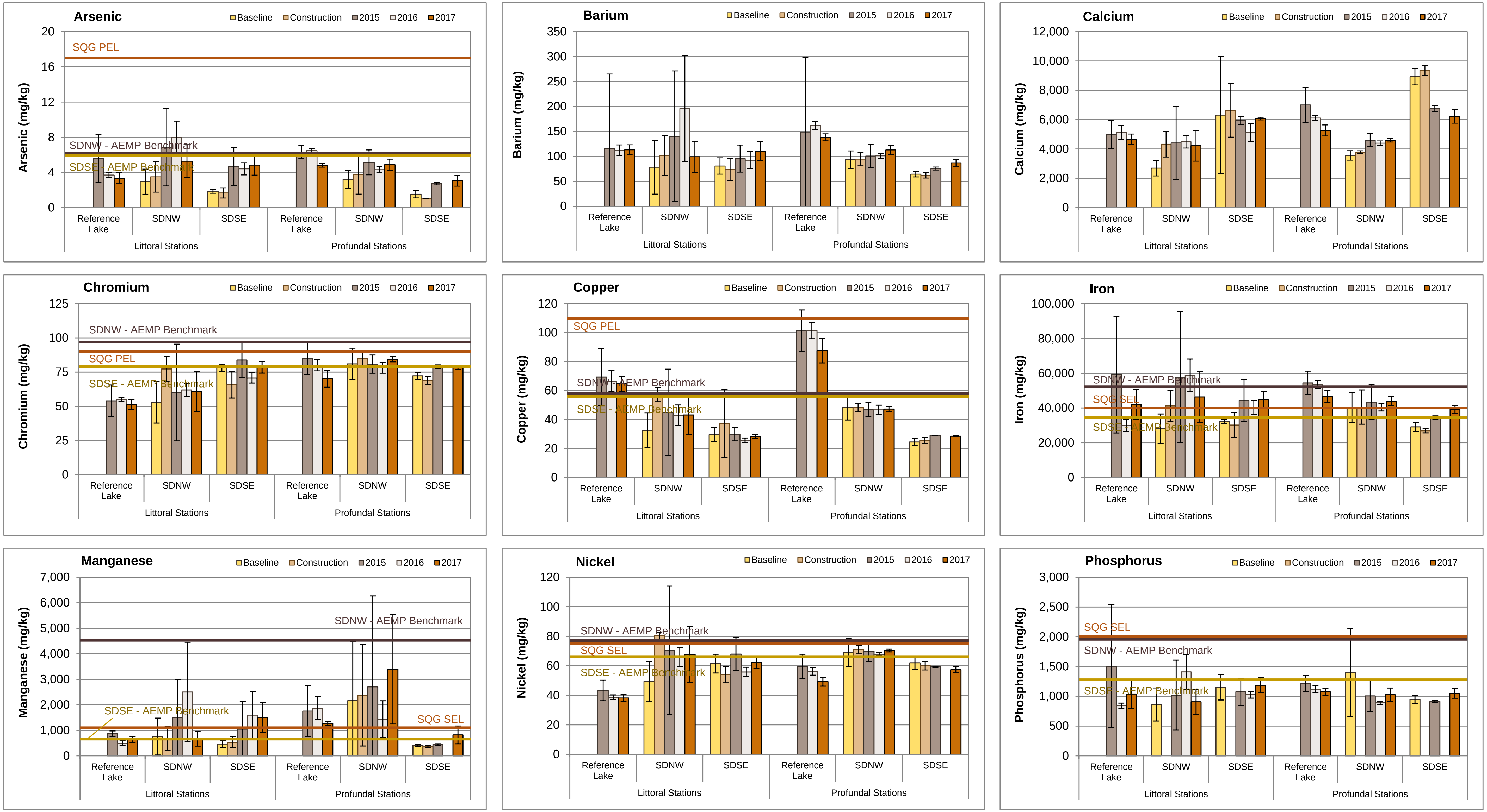


Figure 4.11: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Sheardown Lake NW (SDNW), Sheardown Lake SE (SDSE), and Reference Lake 3 for Mine Baseline (2005 - 2013), Construction (2014) and Individual Operational Year (2015, 2016, and 2017) Periods

concentrations of these parameters in sediment at the Sheardown Lake NW littoral stations leading into 2016. However, based on evaluation of data current to 2017, no substantial changes in sediment metal concentrations were indicated at Sheardown Lake NW littoral and profundal stations following the commencement of Baffinland 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 2017 (Figure 4.12). Chlorophyll-a concentrations differed significantly among seasons at Sheardown Lake NW in 2017, with highest and lowest concentrations observed in fall and winter, respectively (Appendix Table E.6), and reflecting similar seasonal differences in chlorophyll-a concentrations at the reference lake (Appendix Table B.8). Although chlorophyll-a concentrations were significantly higher at Sheardown Lake NW compared to Reference Lake 3 for both the summer and fall sampling events in 2017 (Appendix Tables E.7 – 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 a CWQG oligotrophic categorization for Sheardown Lake NW based on mean aqueous total phosphorus concentrations below 10 µg/L during all sampling events (Table 4.3; Appendix Table C.43).

Temporally, Sheardown Lake NW chlorophyll-a concentrations in 2017 did not differ significantly from concentrations during the mine construction (2014) and early operational (2015, 2016) periods in any consistent direction among the winter, summer, and fall seasons (Figure 4.13). In addition, annual average chlorophyll-a concentrations did not differ significantly among years from 2014 to 2017 (Appendix Table E.11), suggesting no ecologically meaningful changes in the trophic status of Sheardown Lake NW since the onset of mine operations at the Mary River Project. No chlorophyll-a data are available for the baseline (2005 – 2013) period for Sheardown Lake NW, precluding comparisons of chlorophyll-a data to the period prior to mine construction.

4.2.5 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral and profundal habitats of Sheardown Lake NW compared to like-habitat stations at Reference Lake 3 at magnitudes outside of the CES_{BIC} of ±2 SD_{REF} (Tables 4.5 and 4.6). Richness did not differ significantly between Sheardown Lake NW and the reference lake for littoral stations, but was significantly higher at Sheardown Lake NW compared to Reference Lake 3 for profundal stations (Table 4.5 and 4.6). In addition to these differences, benthic invertebrate community structure differences

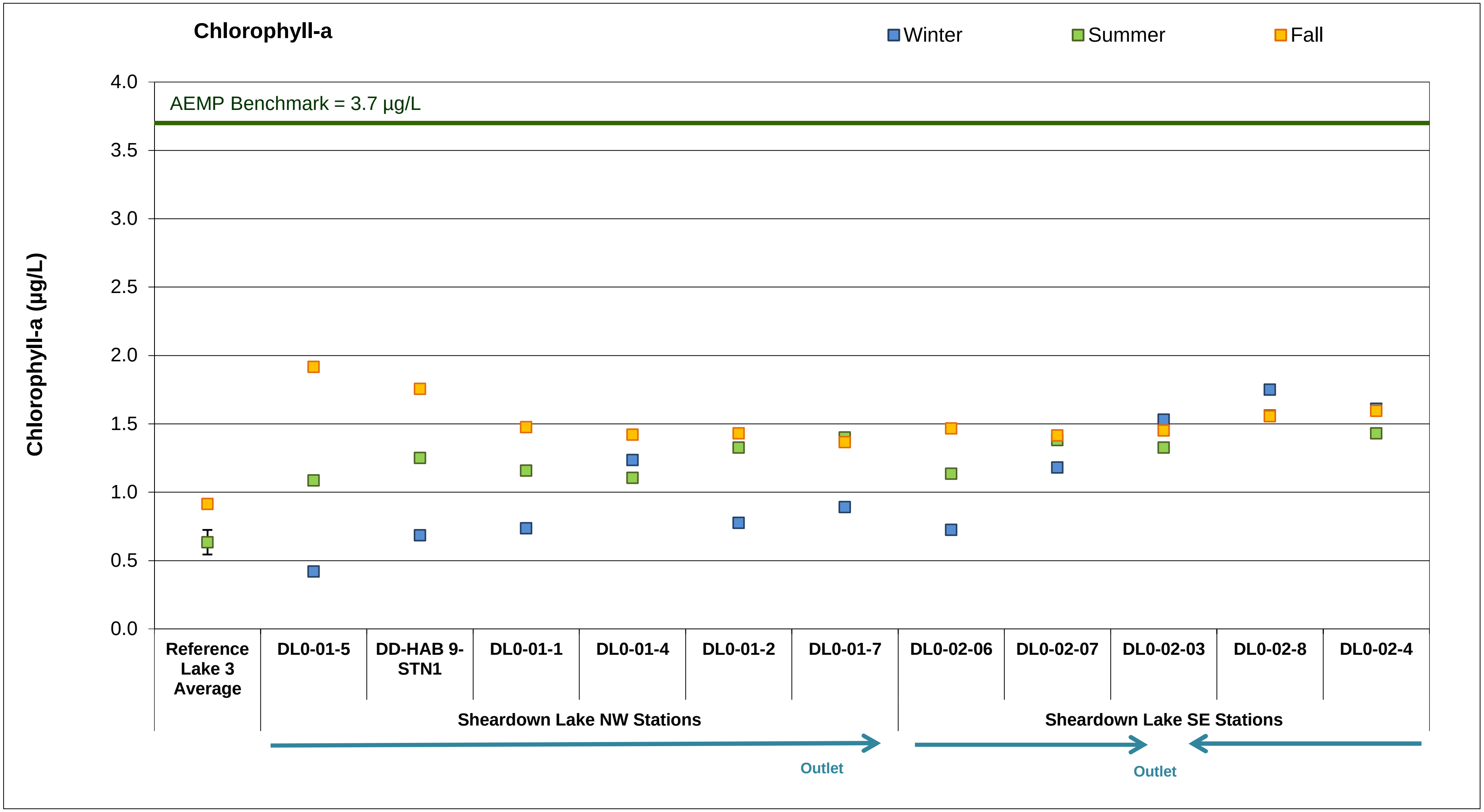


Figure 4.12: Chlorophyll-a Concentrations at Sheardown Lake NW (DLO-1) and Sheardown Lake SE (DLO-2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2017

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 2017.

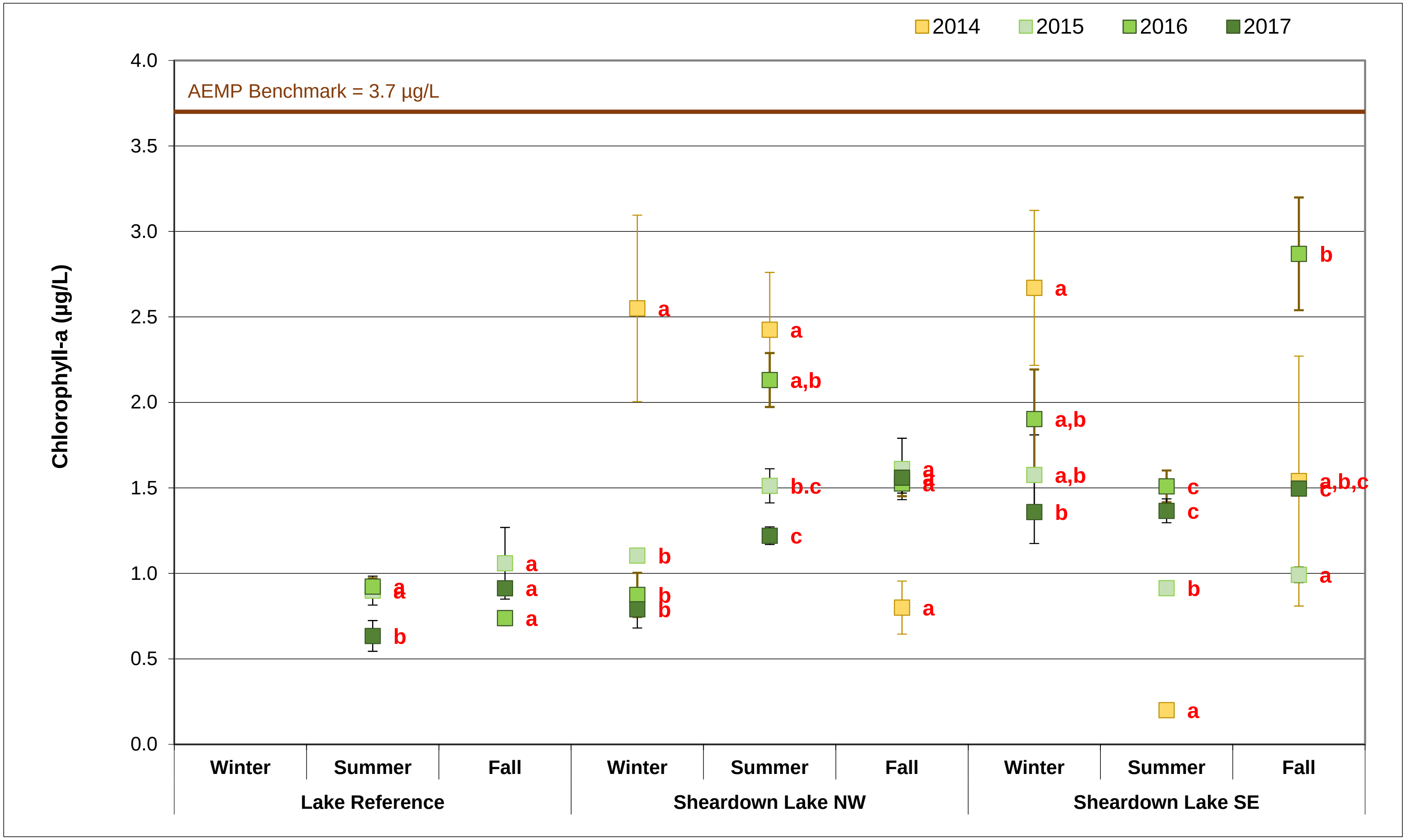


Figure 4.13: Chlorophyll-a Concentration Seasonal Comparison among 2014, 2015, 2016, and 2017 (mean ± SE) at Sheardown Lake Phytoplankton Monitoring Stations, Mary River Project CREMP

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

Table 4.5: 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 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	fourth root	YES	0.005	ANOVA	4.4	Reference Lake 3	1,489	850	380	372	2,618
						Sheardown NW Littoral	5,216	2,398	1,072	3,222	9,344
Richness (Number of Taxa)	square root	NO	0.398	ANOVA	0.6	Reference Lake 3	12.4	2.5	1.1	10.0	15.0
						Sheardown NW Littoral	14.0	3.2	1.4	11.0	19.0
Simpson's Evenness (E)	none	NO	0.696	ANOVA	0.2	Reference Lake 3	0.807	0.142	0.063	0.605	0.934
						Sheardown NW Littoral	0.842	0.048	0.022	0.790	0.911
Bray-Curtis Index	none	YES	< 0.001	ANOVA	3.2	Reference Lake 3	0.384	0.120	0.054	0.203	0.536
						Sheardown NW Littoral	0.762	0.066	0.030	0.695	0.859
Nemata (%)	none	NO	0.146	ANOVA	-0.8	Reference Lake 3	3.9	3.3	1.5	0.8	9.1
						Sheardown NW Littoral	1.3	1.5	0.7	0.0	3.4
Hydracarina (%)	fourth root	NO	0.192	ANOVA	-0.2	Reference Lake 3	5.3	3.0	1.4	1.8	8.1
						Sheardown NW Littoral	4.6	7.4	3.3	0.0	17.2
Ostracoda (%)	none	YES	0.078	ANOVA	-1.1	Reference Lake 3	38.8	18.4	8.2	22.3	63.9
						Sheardown NW Littoral	19.5	11.1	5.0	10.2	37.0
Chironomidae (%)	log	YES	0.065	ANOVA	1.2	Reference Lake 3	51.8	17.9	8.0	29.5	67.9
						Sheardown NW Littoral	73.5	11.2	5.0	58.9	85.8
Metal-Sensitive Chironomidae (%)	square root	NO	0.649	ANOVA	0.1	Reference Lake 3	15.5	13.4	6.0	0.5	37.0
						Sheardown NW Littoral	16.6	7.9	3.5	9.7	29.8
Collector-Gatherers (%)	none	NO	0.600	ANOVA	-0.3	Reference Lake 3	73.9	16.0	7.2	56.0	95.5
						Sheardown NW Littoral	69.4	9.2	4.1	59.4	80.8
Filterers (%)	square root	NO	0.576	ANOVA	0.1	Reference Lake 3	14.7	13.3	5.9	0.5	36.4
						Sheardown NW Littoral	16.5	8.0	3.6	9.7	29.8
Shredders (%)	fourth root	NO	0.263	ANOVA	-0.5	Reference Lake 3	4.2	6.6	3.0	0.0	15.9
						Sheardown NW Littoral	0.7	0.9	0.4	0.0	2.1
Clingers (%)	none	NO	0.906	ANOVA	-0.1	Reference Lake 3	19.8	12.8	5.7	2.7	38.1
						Sheardown NW Littoral	19.1	4.7	2.1	15.0	25.7
Sprawlers (%)	fourth root	YES	0.009	ANOVA	-2.3	Reference Lake 3	72.6	13.9	6.2	55.7	92.9
						Sheardown NW Littoral	41.0	14.0	6.2	22.9	59.4
Burrowers (%)	square root	YES	< 0.001	ANOVA	10.6	Reference Lake 3	7.5	3.0	1.4	4.5	12.0
						Sheardown NW Littoral	39.8	14.8	6.6	24.2	61.2

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

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

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

Table 4.6: 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 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	log	YES	< 0.001	ANOVA	22.4	Reference Lake 3	149	32	14	113	190
						Sheardown NW Profundal	861	391	175	465	1,457
Richness (Number of Taxa)	none	YES	0.019	ANOVA	3.4	Reference Lake 3	4.2	1.5	0.7	2.0	6.0
						Sheardown NW Profundal	9.2	3.5	1.6	5.0	13.0
Simpson's Evenness (E)	none	NO	0.856	ANOVA	0.1	Reference Lake 3	0.704	0.105	0.047	0.597	0.843
						Sheardown NW Profundal	0.717	0.113	0.051	0.618	0.899
Bray-Curtis Index	square root	YES	< 0.001	ANOVA	4.7	Reference Lake 3	0.239	0.114	0.051	0.111	0.376
						Sheardown NW Profundal	0.779	0.112	0.050	0.645	0.946
Hydracarina (%)	none	NO	0.656	ANOVA	-0.3	Reference Lake 3	8.2	5.5	2.5	0.0	15.0
						Sheardown NW Profundal	6.7	4.3	1.9	0.0	10.3
Ostracoda (%)	log(X+1)	NO	0.209	ANOVA	1.0	Reference Lake 3	2.8	4.1	1.8	0.0	8.9
						Sheardown NW Profundal	6.8	4.6	2.0	0.0	10.4
Chironomidae (%)	log	NO	0.455	ANOVA	-0.5	Reference Lake 3	89.0	7.6	3.4	81.6	100.0
						Sheardown NW Profundal	85.1	8.5	3.8	74.6	95.8
Metal-Sensitive Chironomidae (%)	none	NO	0.237	ANOVA	-0.6	Reference Lake 3	12.0	8.9	4.0	0.0	20.2
						Sheardown NW Profundal	6.3	4.4	2.0	2.8	13.7
Collector-Gatherers (%)	log	YES	0.044	ANOVA	-2.1	Reference Lake 3	84.3	4.1	1.9	79.8	90.5
						Sheardown NW Profundal	75.5	7.3	3.3	70.3	87.5
Filterers (%)	fourth root	NO	0.638	ANOVA	-0.4	Reference Lake 3	6.5	9.3	4.2	0.0	20.2
						Sheardown NW Profundal	2.9	2.5	1.1	0.0	6.0
Clingers (%)	none	NO	0.155	ANOVA	-1.1	Reference Lake 3	14.6	4.9	2.2	9.5	20.2
						Sheardown NW Profundal	9.4	5.6	2.5	0.0	15.2
Sprawlers (%)	none	NO	0.343	ANOVA	-2.1	Reference Lake 3	81.4	6.3	2.8	74.8	90.5
						Sheardown NW Profundal	68.2	28.6	12.8	28.0	97.2
Burrowers (%)	square root	NO	0.107	ANOVA	5.0	Reference Lake 3	3.9	3.7	1.7	0.0	8.0
						Sheardown NW Profundal	22.4	26.1	11.7	2.8	62.5

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

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

were indicated between Sheardown Lake NW and Reference Lake 3 by significantly differing Bray-Curtis Index for both littoral and profundal habitat types (Tables 4.5 and 4.6). However, because no ecologically significant differences (i.e., CES_{BIC} outside of ±2 SD_{REF}) in relative abundance of any dominant taxonomic groups was indicated between Sheardown NW Lake and the reference lake for either habitat type, the difference in Bray-Curtis Index between lakes mostly reflected substantially higher benthic invertebrate density 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 2017. The latter point was supported by similar relative abundance of metal-sensitive chironomids between lakes, as well as the occurrence of a higher proportion of burrowing taxa (significantly so, for profundal stations) at Sheardown Lake NW compared to Reference Lake 3 (Tables 4.5 and 4.6), which collectively indicated no sediment metal-related influences on the benthic invertebrate community of Sheardown Lake NW. Benthic invertebrate FFG composition differed between lakes as indicated by the occurrence of significantly lower relative abundance of collector-gatherers at profundal stations of Sheardown Lake NW compared to Reference Lake 3 (Table 4.6), but this difference likely reflected naturally higher sediment TOC content, a key food source for the collector-gatherer FFG, at the reference lake (Appendix Table F.37). Overall, no adverse mine-related influences to the benthic invertebrate community of Sheardown Lake NW were indicated in 2017 based on comparisons to reference lake conditions.

Temporal comparisons did not indicate any consistent ecologically significant differences in general community descriptors of density, richness, and Simpson's Evenness at littoral and profundal habitats of Sheardown Lake NW between the mine baseline (2007, 2008, 2013) period and individual years since the commencement of commercial mine operation (2015, 2016, 2017; Figure 4.14; Appendix Tables F.39 and F.40). Although significantly lower density was indicated at profundal habitat of Sheardown Lake NW in both 2015 and 2017 compared to the 2013 baseline study, a significant difference in density was indicated between the 2007 and 2013 baseline studies, and no ecologically significant difference in density was indicated in either 2015 or 2017 during mine operation compared to the 2007 baseline study (Figure 4.14). This indicated that the differences in density between mine-operational and baseline studies at profundal stations of Sheardown Lake NW were likely the result of sampling artifacts (e.g., differences in sampling station locations and/or replication among studies) or natural temporal variability among studies unrelated to potential influences from commercial mine operation. Notably, no significant differences in benthic invertebrate dominant taxonomic groups or FFG were consistently indicated between baseline and mine operational years for littoral or profundal habitats of Sheardown Lake

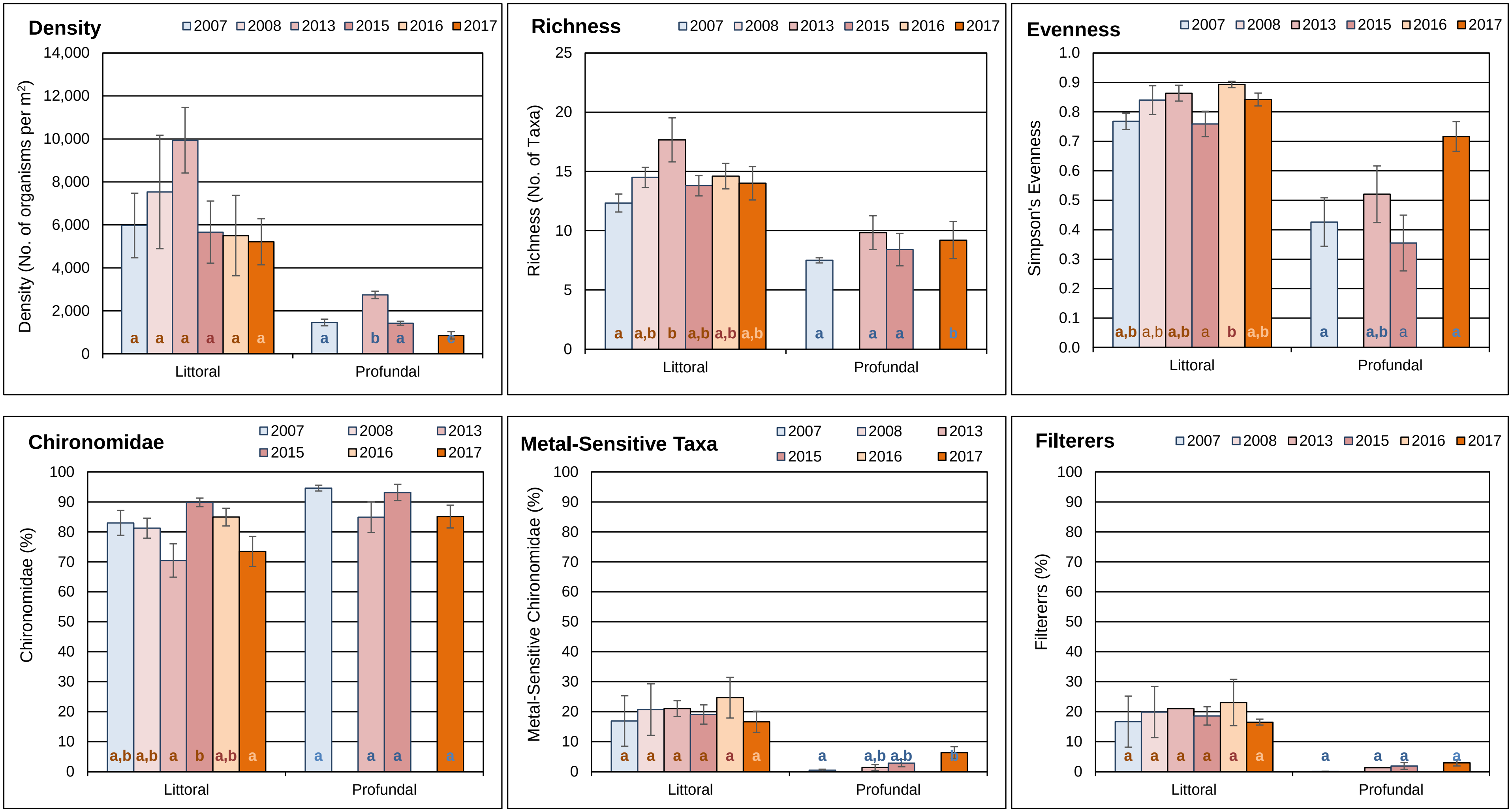


Figure 4.14: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Sheardown Lake NW Littoral and Profundal Study Areas among Mine Baseline (2007, 2008, 2013) and Operational (2015, 2016, 2017) Periods

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

NW (Figure 4.14; Appendix Tables F.39 and F.40). 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.

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 2017, 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 4.7). Total fish CPUE was much higher at Sheardown Lake NW than at the reference lake for nearshore electrofishing and for littoral/profundal gill net sampling (Table 4.7), suggesting higher densities and/or productivity of arctic charr at the Sheardown Lake northwest basin. 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.

Temporal comparison of the Sheardown Lake NW electrofishing catch data indicated that arctic charr CPUE in 2017 was within the range shown over the mine baseline period (2006-2013), and was also comparable to CPUE during mine construction (2014) and previous studies in the mine operation phase (2015, 2016), at nearshore rocky habitat of the lake (Figure 4.15). The 2017 gill netting CPUE for arctic charr was also within the range shown during the baseline period (Figure 4.15). These results suggested that the relative abundance of arctic charr at the nearshore and littoral/profundal habitats of Sheardown Lake NW in 2017 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 using a control-impact analysis using data collected from Sheardown Lake NW and Reference Lake 3 in 2017, as well as a before-after analysis using data collected from Sheardown Lake NW in 2017 and during 2013 baseline characterization. A total of 101 and 100 arctic charr were captured at nearshore habitat of Sheardown Lake NW and Reference Lake 3, respectively, in August 2017 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 5.0 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

Table 4.7: 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 2017

Lake	Method [a]		Arctic Charr	Nine-spine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	100	11	111	2
		CPUE	1.03	0.11	1.14	
	Gill netting	No. Caught	1	0	1	
		CPUE	0.02	0	0.02	
Sheardown Lake Northwest	Electrofishing	No. Caught	101	0	101	1
		CPUE	4.89	0	4.89	
	Gill netting	No. Caught	91	0	91	
		CPUE	2.81	0	2.81	
Sheardown Lake Southeast	Electrofishing	No. Caught	100	15	115	2
		CPUE	4.45	0.67	5.11	
	Gill netting	No. Caught	104	0	104	
		CPUE	5.88	0	5.88	

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

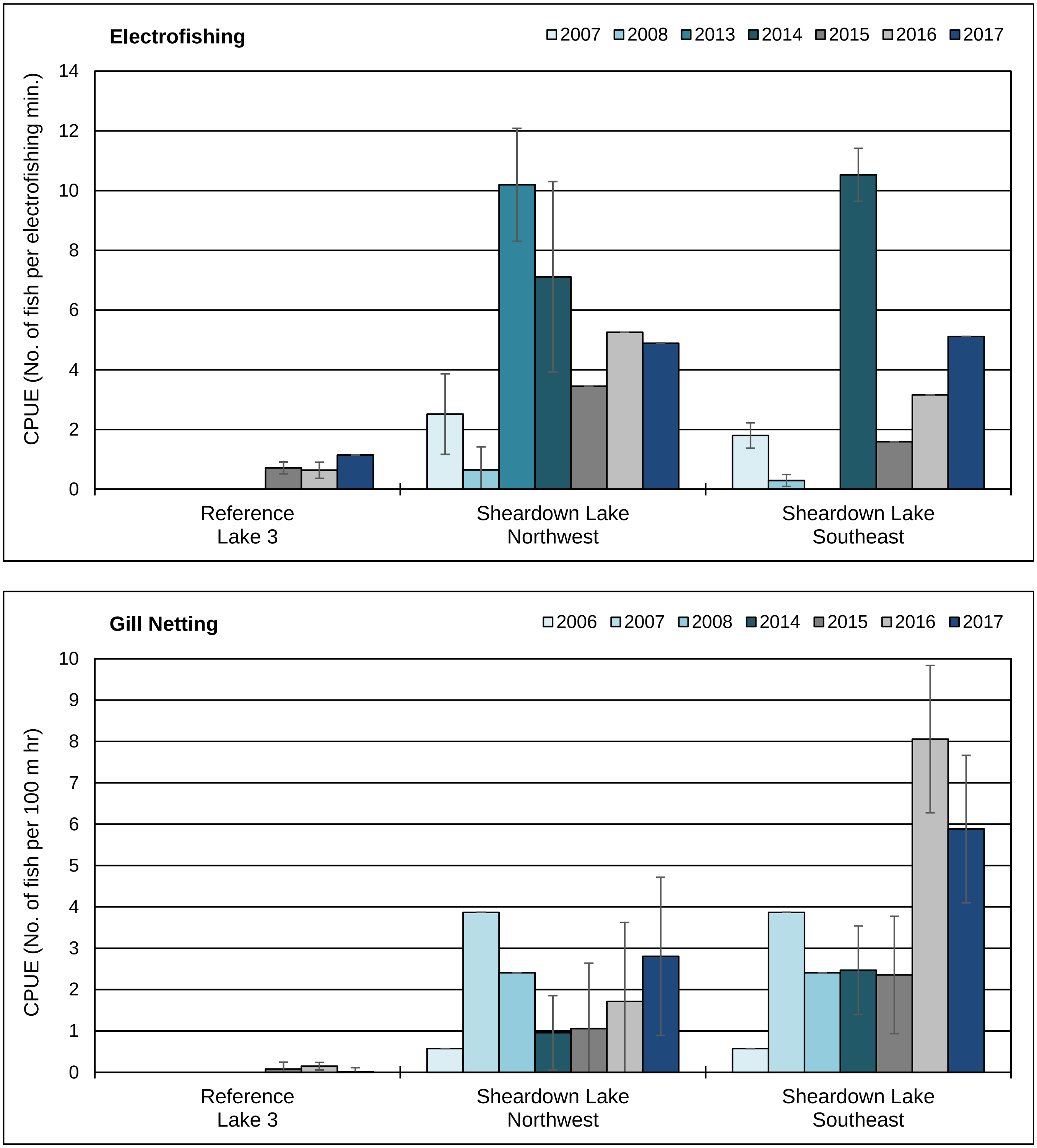
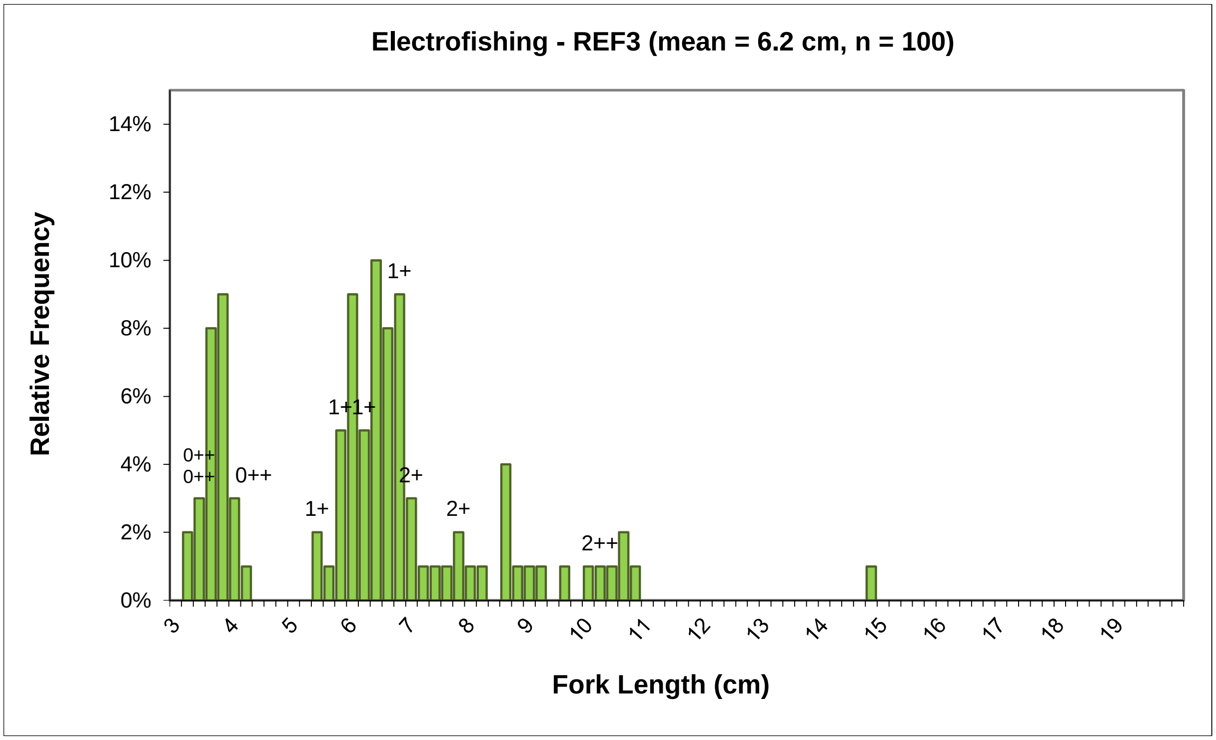


Figure 4.15: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake NW (DLO-01) and Sheardown Lake SE (DLO-02), Mary River Project CREMP, 2006 - 2017

Notes: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014) and operational (2015, 2016, 2017) mine phases. Lake basins (i.e., NW or SE) were not differentiated historically for baseline gill netting catches.



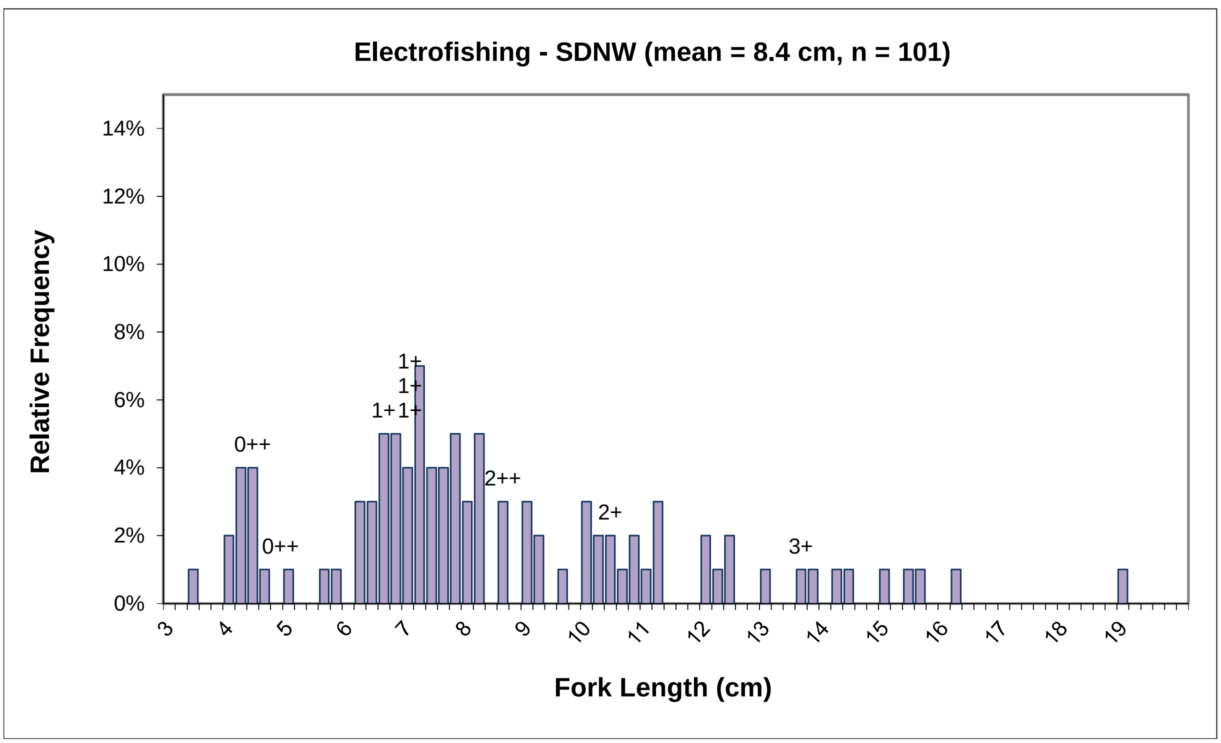


Figure 4.16: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing at Sheardown Lake NW (SDNW) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2017

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

comparisons involved separate assessment of the YOY and non-YOY data sets to account for naturally differing weight-at-length relationships that occur between these life stages.

Length-frequency distributions for the nearshore arctic charr differed significantly between Sheardown Lake NW and Reference Lake 3 (Table 4.8), potentially reflecting a lower proportion of YOY and larger mean size of individuals captured at Sheardown Lake NW. Arctic charr YOY and non-YOY were longer and heavier at the Sheardown Lake NW nearshore than at the reference lake nearshore, with only fresh body weight of non-YOY not differing significantly between lakes (Table 4.8; Appendix Table G.15). No significant differences in nearshore arctic charr YOY condition (i.e., weight-at-length relationship) were indicated between Sheardown Lake NW and Reference Lake 3, and although condition of non-YOY was significantly greater 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 4.8; Appendix Table G.15). Overall, these results indicated no substantial differences in the health of nearshore arctic charr between Sheardown Lake NW and reference lake conditions in 2017.

Temporal comparisons of the Sheardown Lake NW nearshore arctic charr data indicated a significantly different length-frequency distribution between 2017 and the combined 2007 and 2013 baseline data (Table 4.8; Appendix Table G.7). Lengths and weights of arctic charr non-YOY captured at the nearshore of Sheardown Lake NW in 2017 did not differ significantly from those fish captured during the mine baseline characterization (Table 4.8). However, as in the two previous CREMP studies, condition of arctic charr non-YOY was significantly lower in 2017 than during baseline studies conducted at Sheardown Lake NW (Table 4.8). Notably, whereas the magnitude of difference in non-YOY condition between 2017 and baseline (i.e., -9%) was just within the CES_C of ±10%, the magnitude of difference in both previous studies was just outside the CES_C compared to the baseline period (Table 4.8). This suggested on-going, lower condition of arctic charr non-YOY at Sheardown Lake NW nearshore habitat following the commencement of commercial mine operations.

Littoral/Profundal Arctic Charr

Mine-related influences on the Sheardown Lake NW littoral/profundal Arctic charr population were assessed using a before-after analysis between data collected in 2017 and the baseline characterization (combined 2006, 2007, 2008, and 2013) studies. Similar to the two previous CREMP studies, a small sample size from Reference Lake 3 (i.e., n = 1) precluded implementing a control-impact statistical analysis using data collected in 2017. Biological information collected from arctic charr mortalities indicated that non-spawners of reproductive age accounted for approximately 70% of the Sheardown Lake NW Arctic charr population at the time of sampling in August 2017 (Appendix Table G.19). No significant difference in LSI was indicated between

Table 4.8: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake NW and Reference Lake 3 in 2015, 2016, and 2017, 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	2015	2016	2017
Nearshore Electrofishing	Survival	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	No	-	-
	Energy Use (non-YOY)	Size (mean weight)	**Yes (+121%)**	**Yes (+60%)**	No	No	**Yes (-29%)**	No
		Size (mean fork length)	**Yes (+29%)**	**Yes (+17%)**	**Yes (+20%)**	No	No	No
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	**Yes (+7%)**	**Yes (-13%)**	**Yes (-12%)**	**Yes (-9%)**
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**
		Age	-	-	-	**Yes (-35%)**	**Yes (-28%)**	**Yes (-26%)**
	Energy Use	Size (mean weight)	-	-	-	**Yes (-47%)**	**Yes (-31%)**	**Yes (-9%)**
		Size (mean fork length)	-	-	-	**Yes (-21%)**	**Yes (-14%)**	**Yes (-6%)**
		Growth (weight-at-age)	-	-	-	No	No	**Yes (+24%)**
		Growth (fork length-at-age)	-	-	-	No	No	No
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+8%)**	**Yes (+11%)**	**Yes (+6%)**

[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.

non-spawners and female spawners at Sheardown Lake NW (ANOVA; p = 0.12), suggesting similar energy reserves available for gamete development between these groups. The incidence of body cavity parasites was very high in arctic charr mortalities (i.e., 85%), with sparse to very abundant occurrence of encysted worms observed in affected individuals (Appendix Table G.19). High incidence of internal parasites in arctic charr were noted at Camp Lake in 2017, at all mine-exposed lakes in 2015 and 2016 (Minnow 2016a, 2017), and at the various Mary River Project mine area lakes in baseline studies (NSC 2014, 2015a).

The length-frequency distribution for arctic charr captured at littoral/profundal areas of Sheardown Lake NW in 2017 differed significantly from those captured during baseline monitoring (Table 4.8). The differences in length-frequency distribution may have reflected significantly younger and smaller individuals captured in 2017 compared to the baseline period (Table 4.8). No significant difference in length-related growth was indicated in arctic charr captured at Sheardown Lake NW between 2017 and the baseline period, and although arctic charr captured in 2017 showed significantly greater weight-related growth, the magnitude of this difference was within an ecologically meaningful growth CES of ±25% (referred to herein as CES_G; Table 4.6). Arctic charr captured at littoral/profundal areas of Sheardown Lake NW exhibited significantly greater condition in 2017 than during baseline monitoring, but at a magnitude of the difference within the ecologically relevant CES_C of ±10% (Table 4.8; Appendix Table G.20). Notably, the same type and direction of differences in length-frequency distribution, age, mean size, and condition for arctic charr captured at littoral/profundal areas of Sheardown Lake NW were consistently demonstrated from 2015 - 2017 relative to the mine baseline data (Table 4.8). The lack of significant, ecologically meaningful differences in growth combined with significantly greater condition of arctic charr captured at littoral/profundal areas of Sheardown Lake NW in 2017 versus the baseline period suggested no adverse mine-related influences on the adult Arctic charr population of the lake as a result of on-going mine operation.

4.2.7 Integrated Effects Evaluation

At Sheardown Lake NW, aqueous total concentrations of aluminum, manganese, molybdenum, and uranium were elevated compared to the reference lake in 2015, 2016, and 2017, but none of these metals, or any other parameters, showed substantial elevation from concentrations observed during the baseline period, and none were above WQG or AEMP benchmarks. As during the previous CREMP studies, total aluminum and manganese concentrations showed strong positive correlations with turbidity in 2017 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 stations of Sheardown Lake NW were very similar to averages observed for the same respective station types at Reference Lake 3 in 2017, suggesting no marked mine-related influences on sediment metal concentrations in Sheardown Lake NW. Mean concentrations of iron and manganese were above SQG in sediment of Sheardown Lake NW, but concentrations of these metals were also above SQG at the reference lake. Although iron, manganese, and nickel concentrations were above AEMP benchmarks at individual littoral and profundal stations, on average, concentrations of these and all other metals were below their respective AEMP benchmarks at Sheardown Lake NW in 2017. Temporal comparisons indicated that metal concentrations in sediment of Sheardown Lake NW in 2017 were within baseline ranges, indicating no substantial mine-related influences on sediment quality over time at the northwest lake basin.

Chlorophyll-a concentrations at Sheardown Lake NW were significantly higher than at the reference lake in 2017 suggesting greater primary production at Sheardown Lake. However, chlorophyll-a concentrations remained well below the AEMP benchmark during all seasonal sampling events in 2017 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 but no ecologically significant differences in Simpson's Evenness and relative abundance of dominant groups including metal-sensitive chironomids compared to the reference lake in 2017. 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 2017. No ecologically significant differences in primary benthic invertebrate community metrics, dominant taxonomic groups, or FFG were consistently indicated between the mine baseline (2007, 2013) period and individual years since the commencement of commercial mine operation (2015, 2016, 2017) at Sheardown Lake NW. Analysis of arctic charr populations suggested greater fish abundance at Sheardown Lake NW compared to the reference lake in 2017, but similar numbers of arctic charr in 2017 compared to Sheardown Lake 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 the reference lake in 2017, and to those captured from the northwest basin during baseline studies. In addition, arctic charr captured at Sheardown Lake NW in 2017 showed slightly faster growth and greater condition compared to those captured during baseline studies, but the magnitude of these differences were not ecologically meaningful. 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 third 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 using mean annual watershed runoff (2007 – 2016 data) 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 (from NSC 2015b).

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 2017 (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 summer water temperature profile at Sheardown Lake SE did not show the gradual decrease in temperature with depth that was shown at the reference lake through the upper 14 m of the water column, but a similar profile structure was indicated between these lakes during the fall (Figure 4.17). Mean temperature near the bottom of the water column at littoral stations did not differ significantly between Sheardown Lake SE and Reference Lake 3 in fall 2017 (Figure 4.8; Appendix Table C.53). However, mean water temperature near the bottom at profundal stations was significantly warmer at Sheardown Lake SE than at the reference lake, reflecting significantly shallower profundal station depth at the former (Figure 4.8; Appendix Table C.53). 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 at Sheardown Lake SE in 2017 showed no substantial change in saturation with depth during summer, but oxycline development characterized by decreasing saturation levels with increasing depth occurring at depths greater than 11 m during the winter and fall sampling events (Figure 4.17). No oxycline had developed in summer or fall at Reference Lake 3 in 2017 (Figure 4.17). Dissolved oxygen saturation levels at the bottom of the water column at littoral and profundal stations of Sheardown Lake SE were significantly lower than at Reference Lake 3 during fall 2017 sampling (Figure 4.8; Appendix Table C.53). However, in all cases, 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, and fall sampling events in 2017

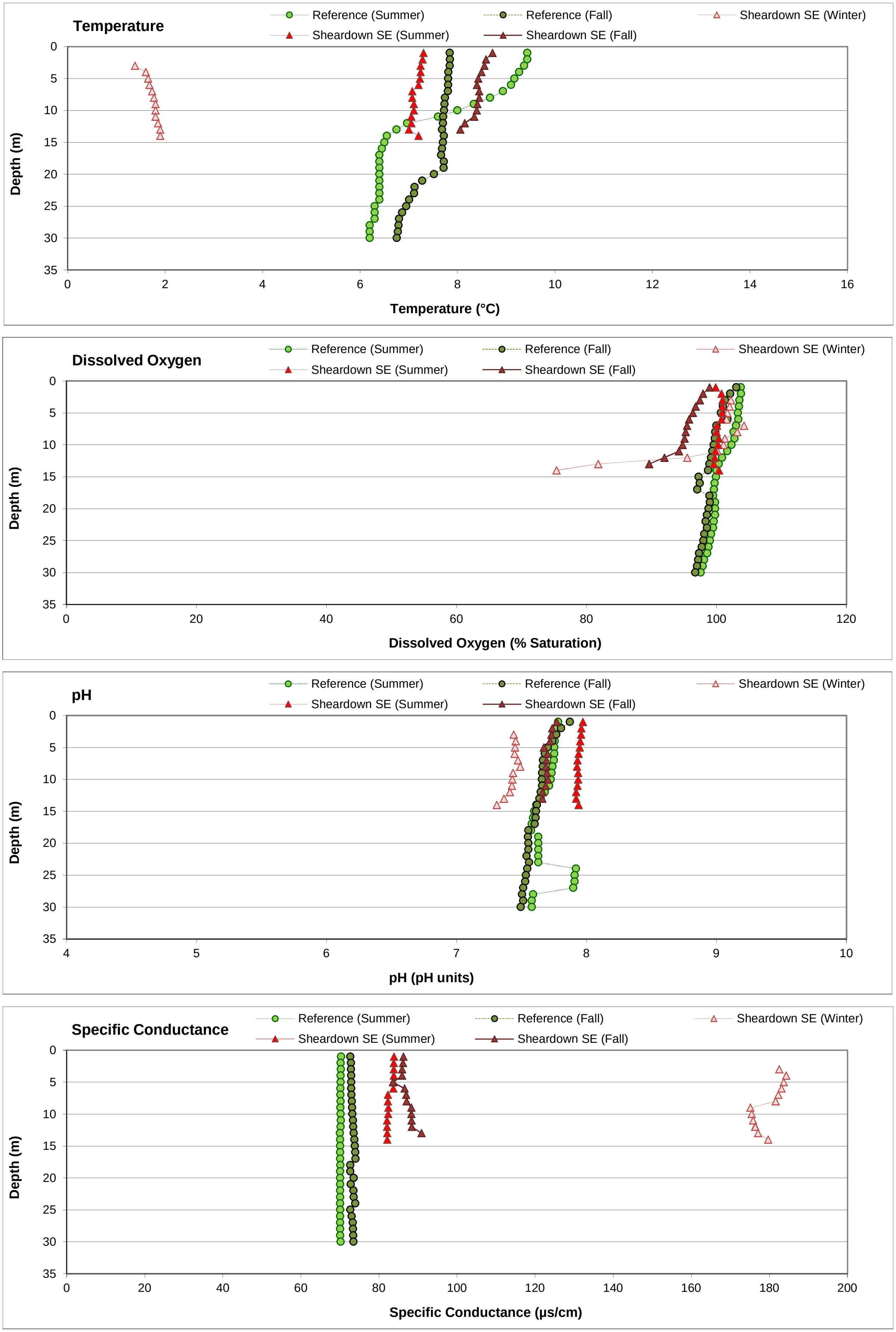


Figure 4.17: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake SE (DLO-02) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2017

(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, pH was significantly higher (i.e., more alkaline) at the bottom of the water column at both littoral and profundal stations of Sheardown Lake SE compared to the reference lake during the 2017 fall sampling event, and was consistently within WQG limits in 2017 at Sheardown Lake SE (Figure 4.8; Appendix Table C.53; Figure 4.17). Specific conductance was significantly higher at Sheardown Lake SE compared to the reference lake during 2017 fall sampling (Figure 4.8). However, mean specific conductance at Sheardown Lake SE (i.e., 99 µS/cm) was intermediate to that of the reference creek and river areas (i.e., range from 55 – 168 µS/cm) in fall 2017. 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 the southeast basin of Sheardown Lake 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 2017 fall sampling event, 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 2017 (Table 4.9; Appendix Table C.54), suggesting that the lake waters were generally well mixed both laterally and vertically. Total aluminum concentrations were highly elevated (i.e., ≥10-fold), turbidity and concentrations of total manganese and tin were moderately elevated (i.e., 5- to 10-fold), and concentrations of total molybdenum were slightly elevated (i.e., 3- to 5-fold), at Sheardown Lake SE compared to Reference Lake 3 during the 2017 summer and/or fall sampling events (Table 4.9; Appendix Tables C.44 and C.54). Dissolved aluminum and molybdenum concentrations were also slightly elevated at Sheardown Lake SE compared to the reference lake, but only during the fall sampling event (Appendix Table C.56). Similar to the northwest basin, total aluminum concentrations showed strong positive correlations with turbidity for the Sheardown Lake SE combined data set (i.e., winter, summer, and fall data; r_s = 0.92), suggesting that much of the aqueous aluminum was associated with suspended particles (Appendix Table C.57). This was corroborated by comparison of total and dissolved fractions of aluminum, which indicated that on average, most (i.e., 78%) 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

Table 4.9: Water Chemistry at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3) Monitoring Stations[a], Mary River Project CREMP, August 2017

	Parameters	Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3)	Sheardown Lake Southeast (SDSE) Station				
						DL0-02-6	DL0-02-7	DL0-02-4	DL0-02-8	DL0-02-3
					Fall 2017	21-Aug-17	21-Aug-17	21-Aug-17	21-Aug-17	21-Aug-17
Conventionals[b]	Conductivity (lab)	umho/cm	-	-	76	104	102	102	102	102
	pH (lab)	pH	6.5 - 9.0	-	7.76	7.91	7.92	7.94	7.89	7.92
	Hardness (as $CaCO_3$)	mg/L	-	-	35	49	48	49	48	49
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	2.1	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	33	54	46	46	38	45
	Turbidity	NTU	-	-	0.7	4.0	3.6	3.8	4.0	4.2
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31	44	46	45	44	44
Nutrients and Organics	Total Ammonia	mg/L	variable[c]	0.855	0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.65	1.40	1.50	1.25	1.25	1.40
	Total Organic Carbon	mg/L	-	-	2.83	1.50	1.45	1.40	1.45	1.45
	Total Phosphorus	mg/L	0.020[α]	-	0.0040	0.0068	0.0059	0.0062	0.0070	0.0076
	Phenols	mg/L	0.004[α]	-	0.0020	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	1.28	2.17	2.11	2.11	2.11	2.10
	Sulphate (SO_4)	mg/L	218[β]	218	3.97	3.09	2.97	2.96	2.95	2.93
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[d]	0.005	0.086	0.089	0.083	0.076	0.080
	Antimony (Sb)	mg/L	0.020[α]	-	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Arsenic (As)	mg/L	0.005	0.005	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Barium (Ba)	mg/L	-	-	0.0064	0.0061	0.0061	0.0061	0.0061	0.0061
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Bismuth (Bi)	mg/L	-	-	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Calcium (Ca)	mg/L	-	-	6.93	9.60	9.19	9.47	9.43	9.57
	Chromium (Cr)	mg/L	0.0089	0.0089	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
	Copper (Cu)	mg/L	0.002	0.0024	0.0008	0.0008	0.0009	0.0008	0.0008	0.0008
	Iron (Fe)	mg/L	0.30	0.300	<0.030	0.067	0.065	0.065	0.065	0.060
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	0.0000925	0.000091	0.0000915	0.0000905	0.0000895
	Lithium (Li)	mg/L	-	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Magnesium (Mg)	mg/L	-	-	4.44	5.96	5.88	5.94	5.95	5.91
	Manganese (Mn)	mg/L	0.935[β]	-	0.00063	0.00392	0.00363	0.00373	0.00376	0.00342
	Mercury (Hg)	mg/L	0.000026	-	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Molybdenum (Mo)	mg/L	0.073	-	0.000124	0.0004435	0.000423	0.0004375	0.00043	0.000434
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	0.000605	0.00058	0.000595	0.0006	0.000565
	Potassium (K)	mg/L	-	-	0.88	0.94	0.92	0.92	0.92	0.91
	Selenium (Se)	mg/L	0.001	-	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Silicon (Si)	mg/L	-	-	0.408	0.595	0.625	0.610	0.615	0.600
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010
	Sodium (Na)	mg/L	-	-	0.85	1.11	1.11	1.10	1.09	1.07
	Strontium (Sr)	mg/L	-	-	0.0078	0.0068	0.0066	0.0068	0.0067	0.0067
	Thallium (Tl)	mg/L	0.0008	0.0008	<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
	Titanium (Ti)	mg/L	-	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00025	0.00063	0.00060	0.00061	0.00061	0.00060
	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
	Zirconium (Zr)	mg/L	-	-	-	<0.00030	<0.00030	<0.00030	<0.00030	<0.00030

[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).

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

basin during moderate to high flow periods. In contrast with aluminum, molybdenum concentrations at Sheardown Lake SE were not strongly correlated with turbidity, suggesting that slight elevation in molybdenum compared to Reference Lake 3 was related to mine operation and/or natural geochemical differences between these lakes. Despite elevation of the metals at Sheardown Lake SE, parameter concentrations were typically well below established WQG and AEMP benchmarks during the winter, summer, and fall sampling events in 2017 (Table 4.9; Appendix Table C.54).

Temporal comparisons of the Sheardown Lake SE water chemistry data indicated no appreciable changes in average parameter concentrations between the 2017 study and mine baseline (2005 – 2013) period, the only exception of which was a slightly elevated average dissolved aluminum concentration in fall 2017 (Figure 4.9; Appendix Figure C.19). As indicated above, because aluminum concentrations were strongly correlated with turbidity, higher dissolved aluminum concentrations in fall 2017 compared to baseline at Sheardown Lake SE likely reflected natural phenomena (e.g., surface runoff events). No parameters showed consistently higher concentrations annually over the mine construction (2014) and 2015 – 2017 mine operational stages with the exception of sulphate (Figure 4.9; Appendix Figure C.19), suggesting a potential mine-related source of this constituent. However, average sulphate concentrations at Sheardown Lake SE of approximately 3 mg/L in 2017 remained 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 represented predominantly by silt loam material containing low TOC content at both littoral and profundal habitats (Figure 4.18; Appendix Table D.31). 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.5). 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 the reference lake, iron (oxy)hydroxide material was visible in surficial and/or sub-surface substrate of Sheardown Lake SE, in some cases occurring as a thin, distinct layer or floc (Appendix Table D.30). Below the surficial layer, substrates at Sheardown Lake SE exhibited some sporadic blackening and, at one station, had a slight sulphidic odour, suggesting development of reducing conditions. However, no distinct redox boundary was observed in the littoral station sediments (Appendix Table D.30). Observations regarding reducing sediment conditions at Sheardown Lake SE were similar to

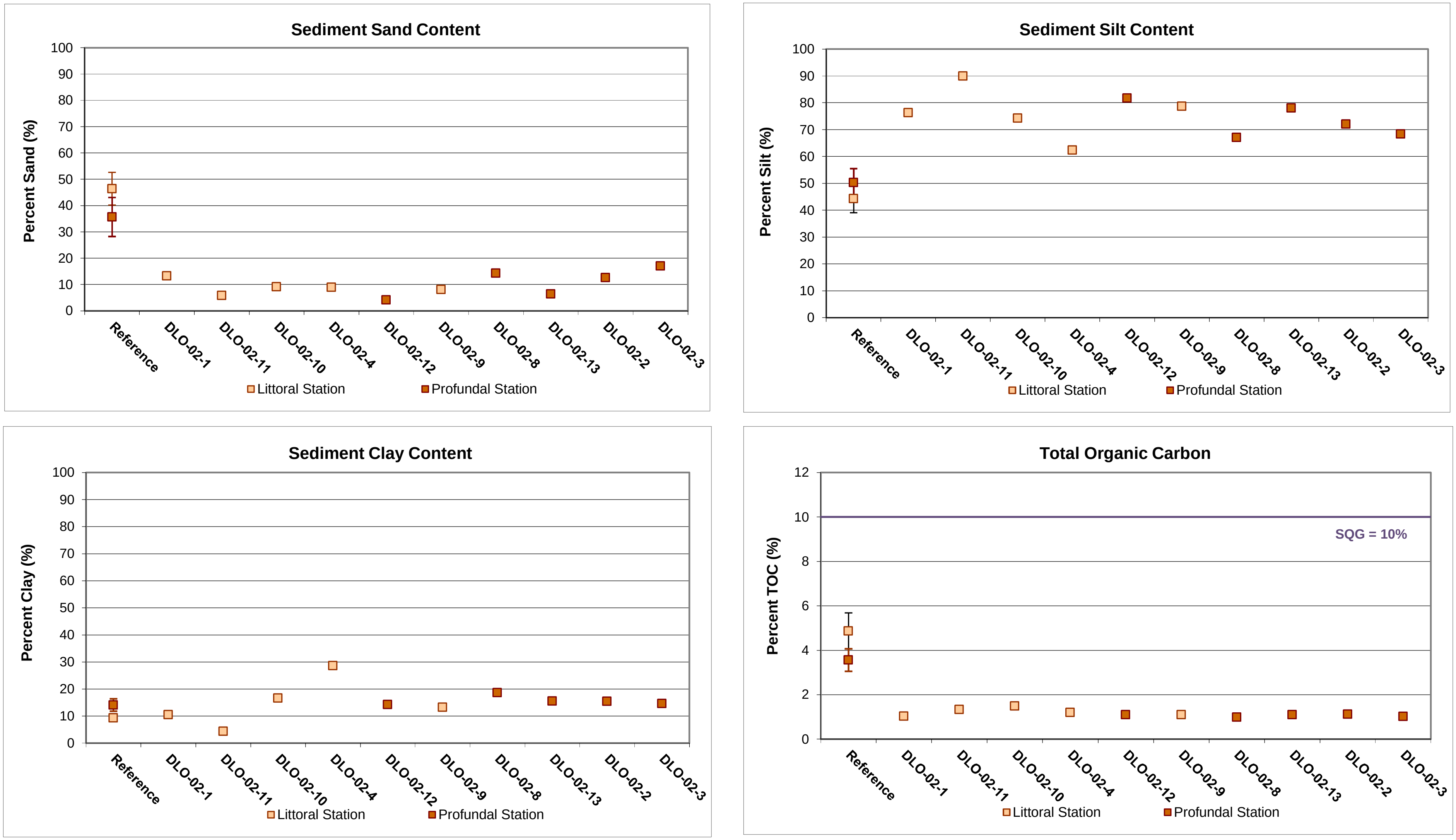


Figure 4.18: **Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sheardown Lake SE (DLO-02) Sediment Monitoring Stations and Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2017**

those made at Reference Lake 3 (Appendix Tables D.3 and D.30), suggesting that factors leading to reduced sediment conditions were comparable between lakes.

Sediment metal concentrations at Sheardown Lake SE showed no clear spatial gradients with progression towards the lake outlet in 2017, suggesting no clear point sources of metals to the lake (Appendix Table D.32). Metal concentrations were considerably higher in sediment of Sheardown Lake SE than in deposited sediment at SDLT9 and Mary River (EO-20) lotic environments that are connected to the southeast basin (Appendix Table D.35), reflecting the depositional nature of lake habitat versus erosional characteristics of the lotic systems. With the exception of very slightly elevated manganese concentrations, 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 in 2017 (Table 4.4; Appendix Table D.33) suggesting no marked mine-related influences on sediment metal concentrations at the southeast lake basin. Mean iron and manganese concentrations were above respective SQG and AEMP benchmarks at the Sheardown Lake SE littoral stations (Table 4.4; Appendix Table D.32). As indicated previously, average concentrations of iron and manganese were also above respective SQG at littoral and/or profundal stations of Reference Lake 3 (Table 4.4). In turn, this suggested that the elevation of iron and manganese concentrations in sediment of Sheardown Lake SE relative to SQG reflected natural lake conditions in the mine local study area. Arsenic, chromium, nickel, and phosphorus concentrations were above site-specific AEMP benchmarks at individual littoral and profundal stations of the southeast basin, but on average, concentrations of these metals were below their respective AEMP benchmarks at Sheardown Lake SE in 2017 (Table 4.4; Appendix Table D.32).

Temporal comparisons indicated that metal concentrations in sediment at littoral and profundal stations of Sheardown Lake SE in 2017 were comparable to those observed during the mine baseline (2005 – 2013) period. The only exception to this was slight elevation (i.e., 3- to 5-fold higher) in the average manganese concentration at littoral stations of the lake in 2017 (Figure 4.11; Appendix Table D.33). On average, metal concentrations in sediment at respective Sheardown Lake SE littoral and profundal stations in 2017 were also within the range of those observed from 2015 and 2016, with no consistently higher concentrations occurring that would suggest an increasing trend over time (Figure 4.11; Appendix Table D.34). This contrasted with the results of the previous CREMP study, which suggested progressively higher concentrations of arsenic and manganese at littoral stations of Sheardown Lake SE from baseline, to mine construction, to 2015 and 2016 mine operational years (Figure 4.11; Minnow 2017). Changes in station replication and location among studies likely contributed to the appearance of greater mean concentrations of these parameters in sediment at the Sheardown Lake SE littoral stations leading into 2016. However, based on evaluation of data current to 2017, perhaps with the exception of a step-increase in concentrations of manganese at littoral stations, no substantial changes in sediment metal concentrations were indicated at Sheardown Lake SE littoral and profundal stations following the commencement of Baffinland 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 2017 (Figure 4.12). Chlorophyll-a concentrations did not differ significantly among the winter, summer, and fall sampling events in 2017, reflecting comparable concentrations among seasons (Appendix Table E.6). Similar to Camp Lake and Sheardown Lake NW, chlorophyll-a concentrations at the Sheardown Lake SE were significantly higher than at the reference lake for both the summer and fall sampling events in 2017 (Appendix Table E.7 and E.8), but concentrations were well below the AEMP benchmark of 3.7 µg/L at all stations and for all sampling events in 2017 (Figure 4.12). On average, chlorophyll-a concentrations at Sheardown Lake SE indicated an 'oligotrophic' trophic status as defined by Wetzel (2001). This trophic status classification was consistent with a CWQG oligotrophic categorization for Sheardown Lake SE based on mean total phosphorus concentrations below 10 µg/L during all sampling events (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 2017 data and data from the mine construction (2014) period or previous years of mine operation (2015, 2016) among the winter, summer, or fall seasons (Figure 4.13). Annual average chlorophyll-a concentrations did not differ significantly among 2014, 2015, and 2017, but chlorophyll-a concentrations from each of these years were significantly lower than in 2016 (Appendix Table E.13). The variable differences in chlorophyll-a concentrations among years at Sheardown Lake SE may reflect the combination of mine-related influences (as suggested in 2016; Minnow 2017) and variable influence of Mary River on Sheardown Lake SE water levels, hydraulic retention time, and/or chemistry among years/seasons. Depending on flow conditions, Mary River discharges into (high flow periods) or drains (low flow periods) Sheardown Lake SE, respectively. No chlorophyll-a baseline (2005 – 2013) data are available for Sheardown Lake SE, precluding comparisons to conditions prior to the mine construction period.

4.3.5 Benthic Invertebrate Community

Benthic invertebrate density was significantly higher at littoral habitat of Sheardown Lake SE than at like-habitat stations of Reference Lake 3, the magnitude of which was outside of the CES_{BIC} of ±2 SD_{REF} (Table 4.10). However, no ecologically significant differences in richness or Simpson's

Evenness were indicated between study lakes at littoral sampling depths. Similar to the northwest basin, differences in littoral habitat benthic invertebrate community structure were indicated between Sheardown Lake SE and Reference Lake 3 based on significantly differing Bray-Curtis Index (Table 4.10). In addition to the substantial difference in density between lakes, the differences in Bray-Curtis Index likely reflected significantly lower and higher relative abundance of Ostracoda (seed shrimp) and Chironomidae (non-biting midges), respectively, at littoral habitat of Sheardown Lake SE compared to like-habitat stations at the reference lake (Table 4.10). The differences in littoral habitat benthic invertebrate community structure between Sheardown Lake SE and Reference Lake 3 almost certainly reflected marked differences in physical sediment properties between lakes, which included significantly lower TOC content and greater proportion of silt at Sheardown Lake SE littoral stations (Appendix Table F.41). Ostracoda are categorized within the collector-gatherer FFG, and because TOC serves as a key food source for this FFG, lower relative abundance of this taxonomic group was consistent with lower sediment TOC content at Sheardown Lake SE compared to the reference lake (Appendix Table F.41). Similarly, Ostracoda are categorized within the sprawler HPG, and because more compact sediments support fewer numbers of invertebrates exhibiting this mode of existence in lake environments, lower relative abundance of Ostracoda was consistent with the occurrence of finer-grained, compact sediment at Sheardown Lake SE compared to the reference lake (Appendix Table F.41). Therefore, the differences in littoral habitat benthic invertebrate community structure between Sheardown Lake SE and the reference lake likely reflected natural differences in substrate properties that included lower sediment TOC and greater compactness at Sheardown Lake SE. Notably, the relative abundance of metal-sensitive chironomids did not differ significantly between Sheardown Lake SE and Reference Lake 3 (Table 4.10), indicating no sediment metal-related influences on the benthic invertebrate community of Sheardown Lake SE. Overall, no adverse mine-related influence to the littoral benthic invertebrate community of Sheardown Lake SE was indicated in 2017 based on comparisons to reference lake conditions. In addition, as observed at the other mine-exposed lakes, higher benthic invertebrate density also suggested naturally higher secondary productivity at Sheardown Lake SE compared to reference lake conditions.

Marked, ecologically significant, differences in density, richness, Bray-Curtis Index, and relative abundance of collector-gatherer FFG and sprawler and burrower HPG were indicated between Sheardown Lake SE and Reference Lake 3 for profundal habitat (Table 4.11). However, the differences in benthic invertebrate community endpoints between lakes likely reflected significantly shallower ‘profundal’ sampling depths at Sheardown Lake SE compared to the reference lake, which on average differed by 8.1 m (Appendix Table F.41). 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

Table 4.10: 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 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	none	YES	0.003	ANOVA	3.4	Reference Lake 3	1,489	850	380	372	2,618
						Sheardown SE Littoral	4,417	1,317	589	2,259	5,671
Richness (Number of Taxa)	square root	YES	0.030	t-test (unequal variance)	-1.4	Reference Lake 3	12.4	2.5	1.1	10.0	15.0
						Sheardown SE Littoral	9.0	0.7	0.3	8.0	10.0
Simpson's Evenness (E)	none	NO	0.197	ANOVA	-0.7	Reference Lake 3	0.807	0.142	0.063	0.605	0.934
						Sheardown SE Littoral	0.712	0.055	0.024	0.634	0.771
Bray-Curtis Index	rank	YES	0.009	Mann-Whitney	4.4	Reference Lake 3	0.384	0.120	0.054	0.203	0.536
						Sheardown SE Littoral	0.916	0.006	0.003	0.908	0.924
Nemata (%)	log(X+1)	YES	0.017	ANOVA	-1.0	Reference Lake 3	3.9	3.3	1.5	0.8	9.1
						Sheardown SE Littoral	0.5	0.6	0.3	0.0	1.5
Hydracarina (%)	none	NO	0.182	ANOVA	-0.7	Reference Lake 3	5.3	3.0	1.4	1.8	8.1
						Sheardown SE Littoral	3.1	1.5	0.7	1.5	5.3
Ostracoda (%)	square root	YES	< 0.001	ANOVA	-2.1	Reference Lake 3	38.8	18.4	8.2	22.3	63.9
						Sheardown SE Littoral	0.8	0.8	0.4	0.2	2.1
Chironomidae (%)	none	YES	0.005	t-test (unequal variance)	2.4	Reference Lake 3	51.8	17.9	8.0	29.5	67.9
						Sheardown SE Littoral	95.6	1.8	0.8	92.8	97.3
Metal-Sensitive Chironomidae (%)	square root	NO	0.903	ANOVA	-0.3	Reference Lake 3	15.5	13.4	6.0	0.5	37.0
						Sheardown SE Littoral	12.1	4.2	1.9	6.2	16.2
Collector-Gatherers (%)	none	YES	0.050	ANOVA	-1.6	Reference Lake 3	73.9	16.0	7.2	56.0	95.5
						Sheardown SE Littoral	48.4	18.8	8.4	34.7	80.1
Filterers (%)	square root	NO	0.999	ANOVA	-0.2	Reference Lake 3	14.7	13.3	5.9	0.5	36.4
						Sheardown SE Littoral	12.1	4.2	1.9	6.2	16.2
Shredders (%)	rank	YES	0.037	Mann-Whitney	-0.6	Reference Lake 3	4.2	6.6	3.0	0.0	15.9
						Sheardown SE Littoral	0.0	0.0	0.0	0.0	0.0
Clingers (%)	none	NO	0.474	ANOVA	-0.4	Reference Lake 3	19.8	12.8	5.7	2.7	38.1
						Sheardown SE Littoral	15.2	5.6	2.5	7.7	21.5
Sprawlers (%)	none	YES	0.004	ANOVA	-2.5	Reference Lake 3	72.6	13.9	6.2	55.7	92.9
						Sheardown SE Littoral	37.7	14.2	6.4	14.2	51.5
Burrowers (%)	log	YES	< 0.001	ANOVA	13.0	Reference Lake 3	7.5	3.0	1.4	4.5	12.0
						Sheardown SE Littoral	47.1	18.2	8.1	34.3	78.2

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

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

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

Table 4.11: 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 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean (n = 5)	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	fourth root	YES	< 0.001	ANOVA	96.9	Reference Lake 3	149	32	14	113	190
						Sheardown SE Profundal	3,234	880	394	2,277	3,999
Richness (Number of Taxa)	square root	YES	0.002	ANOVA	3.1	Reference Lake 3	4.2	1.5	0.7	2.0	6.0
						Sheardown SE Profundal	8.8	1.6	0.7	7.0	11.0
Simpson's Evenness (E)	none	NO	0.409	ANOVA	-0.5	Reference Lake 3	0.704	0.105	0.047	0.597	0.843
						Sheardown SE Profundal	0.651	0.086	0.039	0.547	0.768
Bray-Curtis Index	none	YES	< 0.001	t-test (unequal variance)	6.4	Reference Lake 3	0.239	0.114	0.051	0.111	0.376
						Sheardown SE Profundal	0.975	0.008	0.004	0.964	0.983
Hydracarina (%)	none	YES	0.035	ANOVA	-1.1	Reference Lake 3	8.2	5.5	2.5	0.0	15.0
						Sheardown SE Profundal	1.9	0.7	0.3	0.7	2.6
Ostracoda (%)	fourth root	NO	0.779	ANOVA	-0.4	Reference Lake 3	2.8	4.1	1.8	0.0	8.9
						Sheardown SE Profundal	1.0	1.4	0.6	0.0	3.6
Chironomidae (%)	log	YES	0.046	ANOVA	1.1	Reference Lake 3	89.0	7.6	3.4	81.6	100.0
						Sheardown SE Profundal	97.1	1.6	0.7	94.5	98.9
Metal-Sensitive Chironomidae (%)	none	NO	0.960	ANOVA	0.0	Reference Lake 3	12.0	8.9	4.0	0.0	20.2
						Sheardown SE Profundal	12.3	9.5	4.2	3.0	28.0
Collector-Gatherers (%)	none	YES	0.001	ANOVA	-9.5	Reference Lake 3	84.3	4.1	1.9	79.8	90.5
						Sheardown SE Profundal	45.1	17.4	7.8	18.5	64.8
Filterers (%)	fourth root	YES	0.092	ANOVA	0.6	Reference Lake 3	6.5	9.3	4.2	0.0	20.2
						Sheardown SE Profundal	12.2	9.6	4.3	3.0	28.0
Clingers (%)	log	NO	0.642	ANOVA	-0.1	Reference Lake 3	14.6	4.9	2.2	9.5	20.2
						Sheardown SE Profundal	14.1	9.5	4.3	5.2	29.9
Sprawlers (%)	square root	YES	0.005	ANOVA	-6.1	Reference Lake 3	81.4	6.3	2.8	74.8	90.5
						Sheardown SE Profundal	42.8	20.1	9.0	24.6	70.7
Burrowers (%)	none	YES	0.001	ANOVA	10.5	Reference Lake 3	3.9	3.7	1.7	0.0	8.0
						Sheardown SE Profundal	43.1	17.0	7.6	17.5	62.5

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

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

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

were consistently evident at Reference Lake 3 from 2015 to 2017 (Appendix B) indicating similar influences in pristine lakes of the Mary River Project region. Notably, the maximum depth of Sheardown Lake SE is approximately 14 m (Appendix Table B.1). Because littoral habitat for the Mary River Project CREMP is defined as depths ≤12 m, benthic invertebrate community data collected from profundal depths of Sheardown Lake SE are not directly comparable to those collected at the other mine-exposed lakes nor to Reference Lake 3, all of which attain maximum depths ranging from 30 – 40 m (Appendix Table B.1). Nevertheless, the relative abundance of metal-sensitive chironomids at 'profundal' depths of Sheardown Lake SE were comparable to those observed at littoral habitat as well as to those observed at Reference Lake 3 (Tables 4.10 and 4.11). In turn, this suggested no sediment metal-related influences on the profundal benthic invertebrate community of Sheardown Lake SE in 2017.

Temporal comparisons did not indicate any consistent ecologically significant differences in general community descriptors of richness and Simpson's Evenness at littoral and profundal habitats of Sheardown Lake SE between mine baseline studies (2007, 2013) and individual years since the commencement of commercial mine operation (2015, 2016, 2017; Figure 4.19; Appendix Tables F.43 and F.44). Although significantly lower density was indicated at littoral and profundal habitat of Sheardown Lake SE in at least one individual year of mine operation compared to the 2007 and/or 2013 baseline study data, the differences were not consistent among all years for both habitat types, nor for comparisons against both years of baseline studies (Appendix Tables F.43 and F.44). No significant differences in benthic invertebrate dominant taxonomic groups or FFG were indicated between the baseline and mine operational years for littoral or profundal habitats of Sheardown Lake SE (Figure 4.19; Appendix Tables F.43 and F.44). Because density was the only benthic invertebrate community metric that differed significantly between mine-operational and baseline studies at Sheardown Lake SE, natural temporal variability among studies most likely accounted for the indicated differences. 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 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 the reference lake, in 2017 (Table 4.7). 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

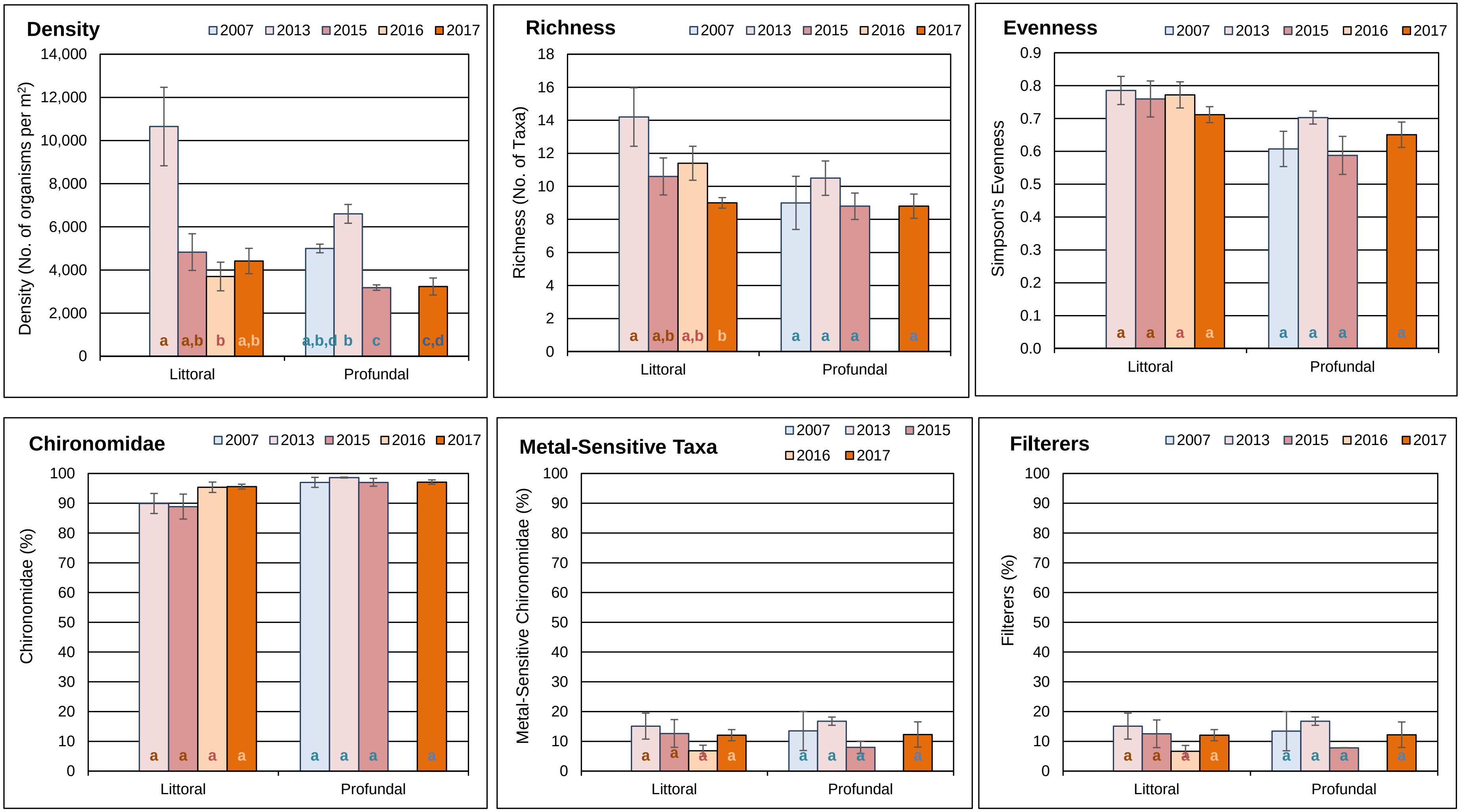


Figure 4.19: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Sheardown Lake SE Littoral and Profundal Study Areas among Mine Baseline (2007, 2013) and Operational (2015, 2016, 2017) Periods

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

of both arctic charr and ninespine stickleback at Sheardown Lake SE (Table 4.7). 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.

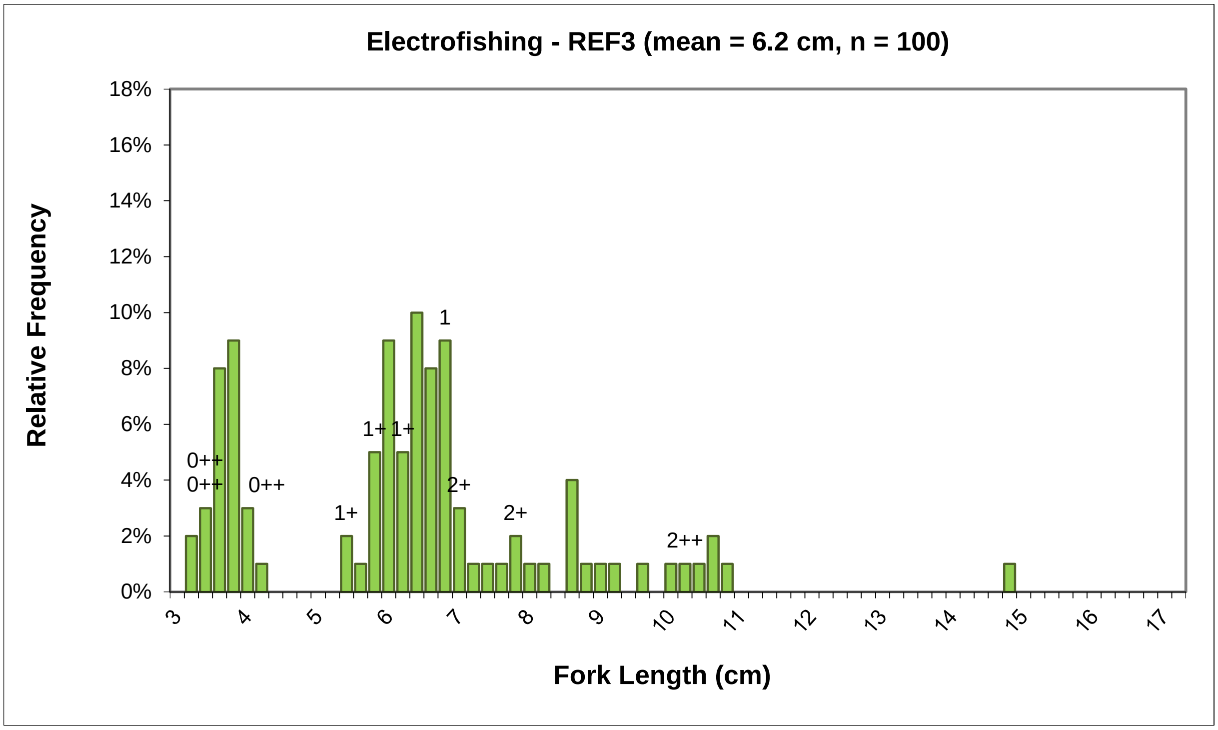
Temporal comparison of the Sheardown Lake SE electrofishing catch data indicated higher fish CPUE in 2017 and the two previous mine operational years compared to during mine baseline studies (2007, 2008; Figure 4.15). Gill netting CPUE for arctic charr was markedly higher in 2016 and 2017 compared to all previous baseline (2006 – 2008), mine construction (2014) and mine operational (2015) studies (Figure 4.15). In part, higher fish CPUE at Sheardown Lake SE during both the 2016 and 2017 studies potentially reflected improvements in sampling efficiency gained through experience from previous studies (see Minnow 2016b, 2017). 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 with a control-impact analysis using data collected from Sheardown Lake SE and Reference Lake 3 in 2017. Although before-after analysis of data collected from Sheardown Lake SE in 2017 (mine operation) and 2007 (baseline) was conducted, 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 100 arctic charr were captured at nearshore habitat of each of Sheardown Lake SE and Reference Lake 3 in August 2017 for the control-impact analysis. Distinguishing of arctic charr YOY from the older, non-YOY age category was possible using a fork length cut-off of 5.0 cm based on evaluation of length-frequency distributions coupled with supporting age determinations for the Sheardown Lake SE and Reference Lake 3 data sets, respectively (Figure 4.20). Nearshore arctic charr health comparisons were conducted separately for the YOY and non-YOY data sets to account for naturally differing weight-at-length relationships that occur between these age categories.

Length-frequency distributions for the nearshore arctic charr differed significantly between Sheardown Lake SE and Reference Lake 3 (Table 4.12), 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 4.20). 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



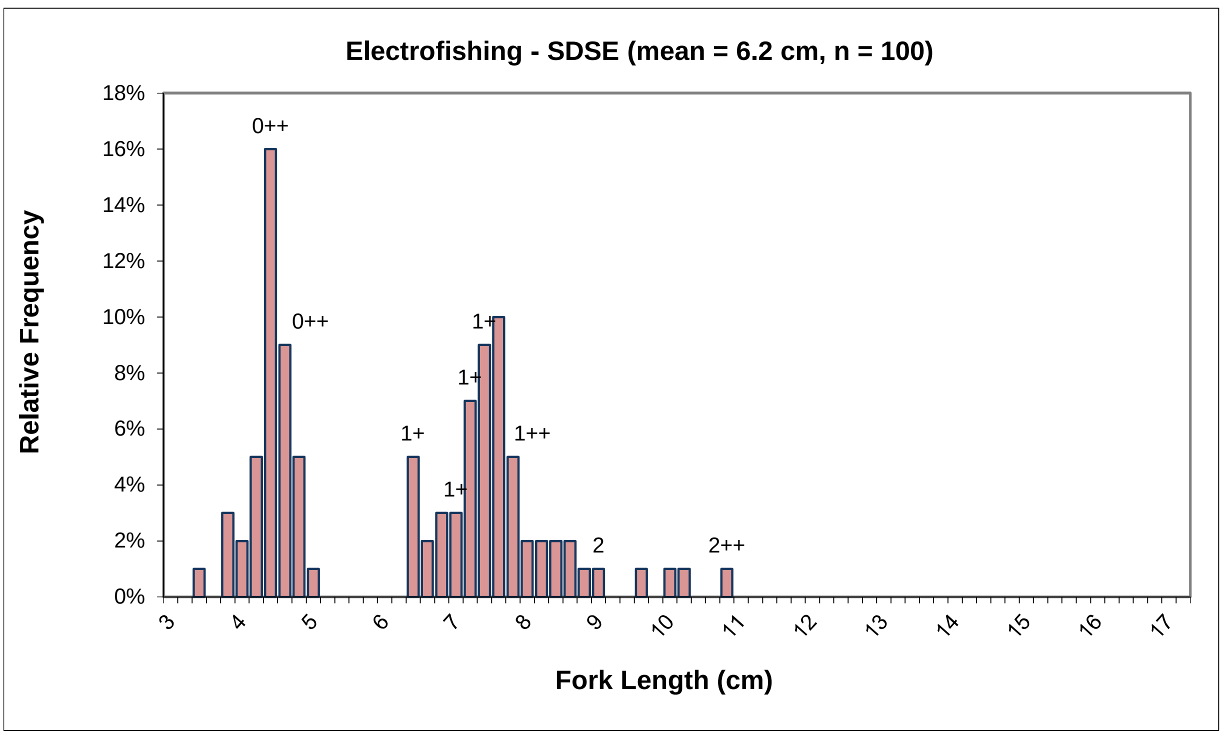


Figure 4.20: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing at Sheardown Lake SE (DLO-02) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2017

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

nearshore (Table 4.12; Appendix Table G.22). The occurrence of significantly larger YOY suggested faster arctic charr growth at Sheardown Lake SE than at the reference lake in 2017. Although condition of nearshore arctic charr YOY did not differ significantly between Sheardown Lake SE and the reference lake, the condition of arctic charr non-YOY was significantly greater at Sheardown Lake SE in 2017 (Table 4.12; Appendix Table G.22). Notably, the magnitude of this difference was just within the CES_C of ±10% suggesting that this difference was not ecologically meaningful (Table 4.12). Similar differences in size and condition of arctic charr YOY were shown between Sheardown Lake SE and the reference lake in studies conducted in 2016 and 2017, but differences in arctic charr non-YOY size and condition were greater between Sheardown Lake SE and the reference lake in 2017 relative to either of the two previous CREMP studies (Table 4.12). Because the direction of difference was positive, the occurrence and/or larger magnitude of differences in size and condition of arctic charr non-YOY between Sheardown Lake SE and the reference lake in 2017 compared to the previous studies was not suggestive of an adverse response related to changes in mine operations over time.

Littoral/Profundal Arctic Charr

Mine-related influences on the Sheardown Lake SE littoral/profundal arctic charr population were assessed using a before-after analysis between data collected in 2017 and the baseline characterization (combined 2007/2008) studies. Similar to the two previous CREMP studies, a small sample size from Reference Lake 3 (i.e., n = 1) precluded implementation of a control-impact statistical analysis using data collected in 2017. Biological information collected from arctic charr mortalities indicated that non-spawners of reproductive age constituted approximately 60% of the Sheardown Lake SE arctic charr population during the August 2017 field study (Appendix Table G.26). No significant difference in LSI was indicated between non-spawners and female spawners at Sheardown Lake SE in 2017 (ANOVA; $p = 0.33$), suggesting similar energy reserves available for gamete development between these two groups. A high proportion of individuals (i.e., 90%) contained body cavity parasites (Appendix Table G.26), the incidence of which was comparable to that observed at other mine-exposed lakes in 2015, 2016, and 2017, as well as during baseline studies (NSC 2014, 2015a). An arctic charr that had been tagged and released previously at Sheardown Lake SE was re-captured in 2017, and showed a 2.1 mm/yr average increase in fork length over the past 11 years (Table 4.13). This growth rate was considerably lower than the incremental change in growth rate for recaptured tagged individuals from the lake in 2015 (Table 4.13), but was not unlike rates shown in resident populations at other arctic lakes that are available in published literature. The growth rate of tagged arctic charr captured at Sheardown Lake SE appeared to be lower than at those captured at the northwest basin of the lake (9.8 mm/yr; n = 1) and Mary Lake (24.4 mm/yr; n = 2) in 2015 and 2016 (Minnow 2017). The

Table 4.12: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake SE and Reference Lake 3 in 2015, 2016, and 2017, 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	2015	2016	2017
Nearshore Electrofishing	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	**Yes (+273%)**	-	-
	Energy Use (non-YOY)	Size (mean weight)	No	No	**Yes (+55%)**	No	**Yes (-43%)**	**Yes (+54%)**
		Size (mean fork length)	No	No	**Yes (+12%)**	**Yes (+7%)**	**Yes (-15%)**	**Yes (+19%)**
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+4%)**	No	**Yes (+9%)**	**Yes (-14%)**	**Yes (-16%)**	No
Littoral/Profundal Gill Netting [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**
		Age	-	-	-	**Yes (-13%)**	No	No
	Energy Use	Size (mean weight)	-	-	-	**Yes (-26%)**	**Yes (-20%)**	**Yes (-16%)**
		Size (mean fork length)	-	-	-	**Yes (-9%)**	**Yes (-7%)**	**Yes (-5%)**
		Growth (weight-at-age)	-	-	-	**Yes (+18%)**	**Yes (+24%)**	No
		Growth (fork length-at-age)	-	-	-	No	No	No
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	No	No	**Yes (-6%)**

[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.

tagging information suggested that arctic charr can reside in the same lake for a prolonged period, and that faster growth rates in arctic charr may be associated with larger lake size.

Table 4.13: Fork Length and Weight Measurement Data for Tagged Arctic Charr Captured at Sheardown Lake SE in August 2015 and 2017, Mary River Project CREMP

Fish Tag Number	Capture Information			Re-Capture Information			Growth Rate
	Date of Capture	Length (mm)	Weight (g)	Date of Capture	Length (mm)	Weight (g)	Δ Length (mm/yr)
85944	03-Aug-2007	363	375	17-Aug-2015	407	500	5.5
89525	21-Aug-2014	368	500	17-Aug-2015	375	505	7.0
86480	03-Aug-2007	338	375	18-Aug-2017	361	430	2.1

Length-frequency distributions of arctic charr captured at littoral/profundal areas of Sheardown Lake SE in 2017 differed significantly than those captured during the baseline period (Table 4.12). In part, the differences in length-frequency distribution may have reflected significantly smaller size (i.e., weight and length) of individuals captured in 2017 compared to the baseline period (Table 4.12; Appendix Table G.27). No significant differences in length- or weight-related growth were indicated for arctic charr captured by gill netting at Sheardown Lake SE between 2017 and the baseline period (Table 4.12; Appendix Table G.27). Although condition of arctic charr from littoral/profundal areas of Sheardown Lake SE was significantly lower in 2017 compared to the baseline period, the magnitude of this difference was within the ecologically relevant CES_C of ±10% (Table 4.12). The arctic charr data collected from littoral/profundal areas of Sheardown Lake SE between 2017 and the baseline period showed slightly different types, direction, and/or magnitude of differences compared to those that were shown during the two previous CREMP studies (Table 4.12). However, the absence of any ecologically significant differences in growth and condition for arctic charr captured at littoral/profundal areas of Sheardown Lake SE in 2017 compared to the baseline period suggested no adverse influences on adult arctic charr following the initial three years of mine operation.

4.3.7 Integrated Effects Evaluation

At Sheardown Lake SE, aqueous total concentrations of aluminum, manganese, and molybdenum were elevated compared to the reference lake in 2015, 2016, and 2017, but none of these metals, or any other parameters, showed substantial elevation from concentrations observed during the baseline period, and none were above WQG or AEMP benchmarks. Similar to the northwest basin, total aluminum and manganese concentrations showed strong positive correlations with turbidity at Sheardown Lake SE in 2017 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 stations of Sheardown Lake SE were very similar to averages observed for the same respective station types at the reference lake in 2017, with the exception of slightly higher manganese at the southeast basin. 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 the reference lake. Although arsenic, chromium, nickel, and phosphorus concentrations were above AEMP benchmarks at individual littoral profundal stations, on average, concentrations of these and all other metals were below their respective AEMP benchmarks at Sheardown Lake SE in 2017. Temporal comparisons indicated that metal concentrations in sediment of Sheardown Lake SE in 2017 were within ranges shown during baseline studies, perhaps with the exception of slightly higher manganese concentrations in sediment of littoral habitat, generally 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 the reference lake in 2017 suggesting greater primary production at Sheardown Lake. However, chlorophyll-a concentrations remained well below the AEMP benchmark during all seasonal sampling events in 2017 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. Benthic invertebrate community differences between Sheardown Lake SE and the reference lake closely mirrored those shown for the northwest basin in 2017, indicating that differences in benthic invertebrate community features between Sheardown Lake SE and the reference lake were more consistent with natural physical substrate differences than any metal-related influences. In addition, no ecologically significant differences in primary benthic invertebrate community metrics, dominant taxonomic groups, or FFG were consistently indicated between the mine baseline period and individual years since the commencement of commercial mine operation (2015, 2016, 2017) at Sheardown Lake SE. Arctic charr population size was greater at Sheardown Lake SE compared to the reference lake in 2017, but similar numbers of arctic charr were present at the Sheardown Lake southeast basin in 2017 compared to the baseline period. Arctic charr captured at nearshore habitat of Sheardown Lake SE were significantly larger and had similar condition than those captured at the reference lake in 2017, but did not differ significantly in size and were of slightly lower condition in 2017 compared to those captured at Sheardown Lake SE during baseline studies. Arctic charr captured at Sheardown Lake SE littoral/profundal habitat in 2017 showed slightly faster growth and were of lower condition than those captured during baseline studies, but the magnitude of these differences were not ecologically meaningful. 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 third year of mine operation at the Mary River Project.

5 MARY RIVER AND MARY LAKE SYSTEM

5.1 Mary River Tributary-F

5.1.1 Water Quality

Water quality monitoring was conducted at Station FO-01 of Mary River Tributary-F (MRTF) in spring, summer, and fall 2017 to meet CREMP requirements, as well as upstream and downstream of the MS-08 effluent discharge channel in fall 2017 to meet EEM requirements. Dissolved oxygen (DO) concentrations did not differ significantly between the MRTF effluent-exposed and reference study areas, and were well above the WQG lowest acceptable concentration for sensitive, early life stages of cold water biota (i.e., 9.5 mg/L) at both study areas (Figure 5.1). Although pH and specific conductance were each significantly higher at the effluent-exposed area than at the reference area of Mary River Tributary-F, the mean incremental difference between areas for each of these parameters was very small, and pH values were well within the WQG acceptable range for the protection of aquatic life (Figure 5.1). As a result, the small differences in pH and specific conductance between the MRTF effluent-exposed and reference areas was not likely to be ecologically meaningful.

Based on extrapolation of specific conductance measures, the proportion of MS-08 mine effluent at the MRTF effluent-exposed area immediately below the effluent channel confluence was estimated as 0.17% at the time of the August 2017 field study, with a maximum of 0.85% estimated for 2017 during periods of effluent discharge (Minnow 2018). Notably, a substantial step increase in specific conductance was observed approximately 1.9 km downstream of the MS-08 effluent channel confluence on MRTF at the time of the August 2017 field study (Appendix Figure C.20). Higher specific conductance at this location and farther downstream in Mary River Tributary-F was attributed to the receipt of surface runoff from areas at which chloride salts (e.g., $CaCl_2$) were used to assist with exploratory/operational drilling through material exhibiting subsurface permafrost, as well as potential natural variation in geological properties with progression through the MRTF system.

Water chemistry monitoring at MRTF indicated that, on average, only ammonia, nitrate, and/or sulphate concentrations were slightly elevated (i.e., three- to five-fold higher) at Mary River Tributary-F (Stations MRTF-1 and F0-01) compared to Mary River upstream reference conditions during periods of effluent discharge in 2017 (Appendix Table C.56). However, concentrations of these parameters were consistently well below applicable WQG at MRTF (Appendix Table C.56). Although total concentrations of aluminum and iron were occasionally above respective WQG at MRTF in 2017, similar or higher concentrations of these metals were observed at the Mary River upstream reference stations during any given sampling event (Appendix Table C.56), indicating

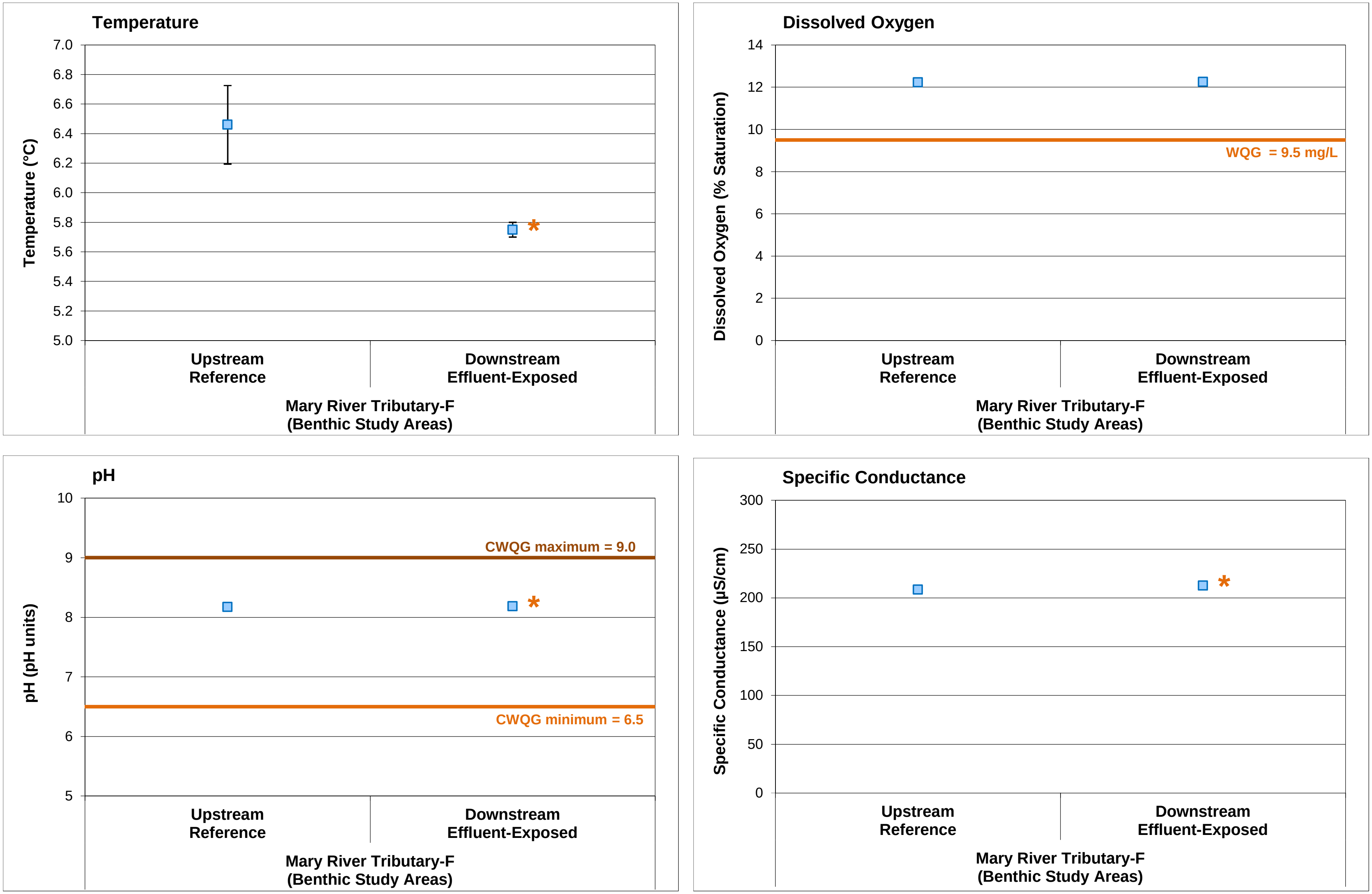


Figure 5.1: Comparison of *In Situ* Water Quality Variables (mean ± SE; n = 5) Measured at Mary River Tributary-F Benthic Stations During Mary River Project Phase 1 EEM, August 2017

Note: An asterisk (*) next to effluent-exposed area data point indicates that the mean value differed significantly from that of the applicable reference area.

natural elevation of total aluminum and iron concentrations in regional watercourses. Overall, the MS-08 effluent discharge resulted in only a marginal elevation in ammonia, nitrate, and/or sulphate concentrations at MRTF.

5.1.2 Phytoplankton

Chlorophyll-a concentrations at MRTF (Station FO-01) were lower than those observed among the reference stations during individual spring and summer sampling events in 2017 (Appendix Table E.14), and were well below the AEMP benchmark of 3.7 µg/L for each of these sampling events. Low phytoplankton productivity, indicative of oligotrophic conditions, was suggested at MRTF based on comparison of chlorophyll-a concentrations to Dodds et al (1998) trophic status classification for creek environments. This productivity classification was supported by a WQG categorization of ultra-oligotrophic to oligotrophic based on mean aqueous phosphorus concentrations near or below 10 µg/L at MRTF during all spring, summer, and fall sampling events (Table 3.1; Appendix Tables C.58 and C.62). Overall, no mine-related influences to MRTF phytoplankton density were suggested by the 2017 chlorophyll-a concentration data.

5.1.3 Benthic Invertebrate Community

Benthic invertebrate density, richness, Simpson's Evenness, and Bray-Curtis Index did not differ significantly between the MRTF effluent-exposed and reference study areas during the August 2017 EEM survey (Table 5.1)[7]. Direct comparison of dominant benthic invertebrate community groups indicated a subtle difference in community composition between the effluent-exposed and reference areas of MRTF that was driven entirely by significantly greater density of Simuliidae (blackflies) at the effluent-exposed study area (Table 5.1). Because blackflies exhibit a filter-feeding, clinging mode of existence in aquatic habitats (Merritt et al. 2008), differences in filterer FFG and clinger HPG densities between the MRTF effluent-exposed and reference study areas (Table 5.1) reflected the difference in blackfly densities shown between areas. No significant differences in any individual dominant taxonomic group, FFG or HPG were indicated between the MRTF effluent-exposed and reference study areas with the removal of Simuliidae from the data set metrics except for the proportion of collector-gatherer FFG (Minnow 2018).

Higher densities of blackflies generally occur at the outlets of tributaries and in larger-sized streams (Carlsson 1967; Grillet and Barrera 1997; Pramul and Wongpakum 2010), possibly due to greater inputs of suspended organic matter, the predominant food source for blackflies, at these habitats (Carlsson et al. 1977). Therefore, a greater density of blackflies downstream of the

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

Table 5.1: Benthic Invertebrate Community Statistical Comparison Results between Mary River Tributary-F Effluent-Exposed and Reference Study Areas, Calculated Using Family Level Taxonomy, for the Mary River Project EEM, August 2017

Metric	Two-Sample Comparison					Summary Statistics						
	Significant Difference Between Areas?	Trans-formation	Test	p-value	Magnitude of Difference [a] (No. of SD)	Area	Median	Mean	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	NO	fourth root	ANOVA	0.1238	~	Reference	474	533	334	149	188	1,058
						Effluent-Exposed	855	849	276	123	448	1,175
Richness (Number of Taxa)	NO	fourth root	ANOVA	0.9727	~	Reference	4.0	4.6	1.3	0.6	3.0	6.0
						Effluent-Exposed	5.0	4.6	1.1	0.5	3.0	6.0
Simpson's Evenness	NO	log_{10}	ANOVA	0.7872	~	Reference	0.430	0.461	0.154	0.069	0.297	0.689
						Effluent-Exposed	0.379	0.430	0.120	0.054	0.338	0.637
Bray-Curtis Index	NO	none	ANOVA	0.1006	~	Reference	0.204	0.242	0.161	0.072	0.069	0.439
						Effluent-Exposed	0.423	0.398	0.096	0.043	0.291	0.491
Chironomidae (No. per m^2)	NO	none	ANOVA	0.8030	~	Reference	241	309	170	76	102	531
						Effluent-Exposed	284	283	139	62	133	426
Metal Sensitive Chironomidae	NO	none	ANOVA	0.8397	~	Reference	107	121	59	27	40	199
						Effluent-Exposed	112	114	34	15	70	155
Simuliidae (No. per m^2)	YES	none	ANOVA	0.0137	2.0	Reference	161	205	169	75	75	487
						Effluent-Exposed	552	540	169	75	297	706
Collector-gatherers (No. per m^2)	NO	none	ANOVA	0.7417	~	Reference	240	310	173	77	102	532
						Effluent-Exposed	277	277	132	59	133	416
Filterers (No. per m^2)	YES	none	ANOVA	0.0137	2.0	Reference	161	205	169	75	75	487
						Effluent-Exposed	552	540	169	75	297	706
Clingers (No. per m^2)	YES	none	ANOVA	0.0151	2.0	Reference	165	212	175	78	79	505
						Effluent-Exposed	563	558	179	80	308	763
Sprawlers (No. per m^2)	NO	none	ANOVA	0.7510	~	Reference	240	305	166	74	102	517
						Effluent-Exposed	277	274	130	58	133	412

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

Highlighted values indicates significant difference between study areas based on a p-value less than 0.10.

effluent channel confluence on MRTF may have reflected increased food resources originating from the effluent-channel. Notably, blackfly larval densities do not appear to be strongly influenced by plankton abundance (Carlsson 1967), suggesting that non-living organic matter received from runoff potentially accounted for higher densities of blackflies at the effluent-exposed area. No significant differences in densities of metal-sensitive chironomids were indicated between the MRTF effluent-exposed and reference study areas, suggesting that between-area differences in metal concentrations did not affect the composition of the benthic invertebrate community at the effluent-exposed area. In addition, no significant differences in sample replicate water velocity, substrate size, or substrate embeddedness were indicated between the MRTF effluent-exposed and reference study areas (Minnow 2018), suggesting that the difference in blackfly density between these areas was unrelated to these variables.

Overall, statistical similarity in benthic invertebrate density, richness, Simpson's Evenness, and Bray-Curtis Index between effluent-exposed and reference areas of MRTF indicated no effluent-related effects on the benthic invertebrate community in the receiving environment downstream of the MS-08 effluent discharge

5.1.4 Fish Population

No fish were captured within MRTF either downstream or upstream of the MS-08 effluent discharge channel during the August 2017 fish population survey conducted for EEM (Table 5.2; Appendix Table G.28). Fish sampling was conducted at reaches extending from the MRTF outlet at Mary River to just of upstream of the effluent channel confluence approximately 3 km from Mary River (Figure 2.3), and therefore the lack of fish captures indicated that fish were naturally absent through the entire MRTF system in August 2017. The natural absence of fish from MRTF presumably reflects the combination of complete freezing overwinter and an inability of fish to colonize the tributary due to relatively high stream gradient and the presence of natural in-stream barriers. An average gradient of 12% was documented through the lower 750 m of MRTF during the fish population survey (Minnow 2018). In addition, an approximately 1.75 m high step-drop over large boulder habitat occurred on MRTF approximately 50 m upstream of the outlet to Mary River (Appendix Photo Plate G.1), representing an impassable barrier for upstream migration by fish under the flow conditions observed at the time of the fish population survey. The absence of fish precluded the implementation of a fish population survey to assess fish health at MRTF in 2017 as part of the EEM biological study (Minnow 2018).

5.1.5 Integrated Effects Evaluation

Potential mine-related effects on water quality of MRTF in 2017 included slightly elevated concentrations of ammonia, nitrate, and/or sulphate compared to reference conditions during

Table 5.2: Summary of Fish Catches at Mary River Project EEM Fish Population Survey Study Areas, August 2017

Study Area	Total Effort		Summary Statistic Endpoint	Fish Species			Catch Summary	
	Distance Sampled (m)	Electrofishing Seconds		Arctic Charr		Ninespine Stickleback	Totals	Total No. Species
				YOY [b]	Non-YOY [b]			
Mary River Tributary-F	678	4,157	Total No. Caught	0	0	0	**0**	**0**
			CPUE[a]	0.0	0.0	0.0	**0.0**	
Mary River Near-Field	388	4,587	Total No. Caught	0	100	0	**100**	**1**
			CPUE[a]	0	1.30	0	**1.30**	
Mary River Far-Field	708	8,340	Total No. Caught	2	103	3	**108**	**2**
			CPUE[a]	0.01	0.75	0.02	**0.78**	

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

periods of effluent discharge in 2016 and 2017. However, concentrations of these parameters were consistently well below applicable WQG at MRTF. Despite slightly higher nutrient concentrations (e.g., nitrate) at MRTF, chlorophyll-a concentrations were lower than those observed at the reference stations in 2017 and were also well below the applicable AEMP benchmark. The benthic invertebrate community survey indicated no significant differences in primary endpoints of density, richness, Simpson's Evenness, and Bray-Curtis Index between effluent-exposed and reference areas of MRTF. Fish were determined to be naturally absent from MRTF as a result of the combination of complete freezing of the watercourse overwinter and an inability of fish to colonize the tributary due to relatively high stream gradient and the presence of natural in-stream barriers. Overall, no adverse influences to the phytoplankton and benthic invertebrate community of MRTF were indicated in 2017 as a result of the receiving of intermittent discharge of mine effluent to this watercourse.

5.2 Mary River

5.2.1 Water Quality

Dissolved oxygen (DO) levels 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 stations for each respective seasonal sampling event (Figure 5.2; Appendix Tables C.1 - C.3). Although DO saturation levels differed significantly among the Mary River benthic study areas, no gradient in DO saturation levels was shown from upstream to downstream of the mine at the time of biological sampling in August 2017. In addition, DO saturation was consistently well above the WQG minimum limit for cold-water biota (i.e., 54%) at all times (Figure 5.2; Appendix Figure C.21 and Table C.61). This suggested that slight differences in DO concentrations/saturation among Mary River study areas were not ecologically meaningful and were unrelated to potential mine influences.

In situ pH at all Mary River stations was similar to pH at the GO-09 series reference stations for each respective seasonal sampling event (Appendix Table C.1 – C.3 and Figure C.21). Although pH differed significantly among the Mary River benthic study areas, no gradient in pH was shown from upstream to downstream of the mine at the time of biological sampling in August 2017 and pH at all Mary River stations was consistently within WQG limits during all spring, summer and fall sampling events (Figure 5.2; Appendix Table C.61). Conductivity at Mary River stations showed no distinct spatial changes with progression from upstream to downstream of the mine that were consistent over the spring, summer, or fall sampling events, suggesting no mine-related influences on Mary River conductivity (Appendix Figure C.21). Notably, conductivity was consistently lowest in spring and highest in fall at all stations, reflecting natural seasonal differences related to proportion of flow from surface runoff (e.g., spring snowmelt) and baseflow/ groundwater sources. Although specific conductance differed significantly among Mary River benthic study areas during fall biological monitoring in 2017, specific conductance upstream of the mine at GO-03 did not differ significantly from the mine-exposed areas (Figure 5.2). Therefore, rather than being indicative of a potential mine-related influence, the differences in specific conductance among the Mary River study areas likely reflected differences in the natural proportion of flow contributed by various tributaries to the river, as well as differences in the geology of base material between Mary River and these tributaries.

Water chemistry within Mary River showed no distinct and/or consistent spatial differences with progression downstream from the GO-09 series river reference stations during any of the spring, summer, or fall sampling events in 2017 (Table 5.3; 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, and often lower than, concentrations observed at the upstream reference stations (GO-09 series stations) during each respective sampling event (Table 5.3; Appendix Table C.62). The exception to the above statement related to total and dissolved concentrations of manganese, which were elevated at the EO and CO stations compared to the GO-09 reference stations during the summer and/or fall sampling events (Appendix Tables C.63 and C.65). Total concentrations of several other metals, including aluminum, chromium, cobalt, iron, lead, nickel, and phosphorus, were elevated at Station CO-05 compared to the GO-09 reference stations during the fall monitoring event (Table 5.3). However, relatively high total

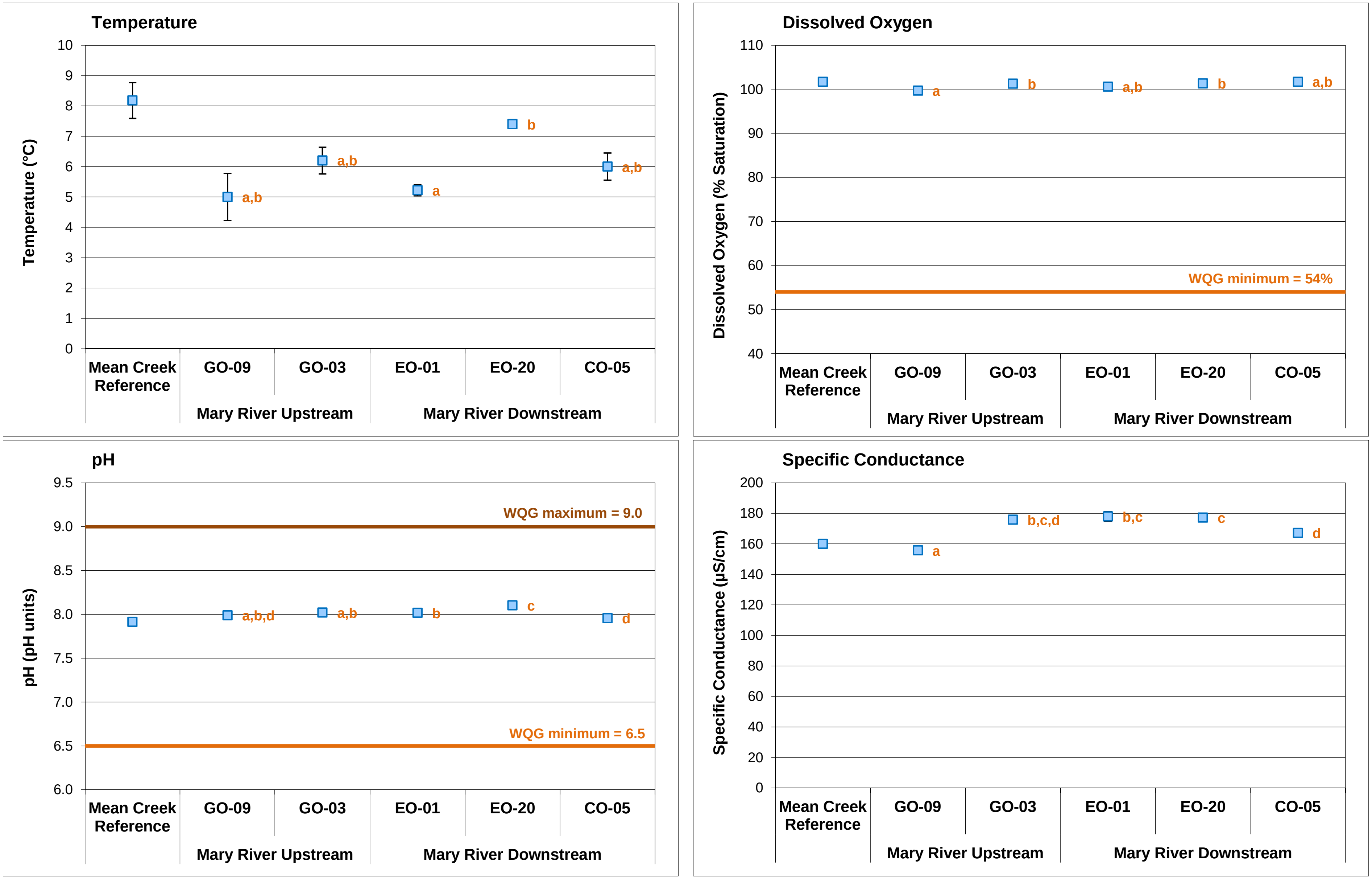


Figure 5.2: Comparison of *In Situ* Water Quality Variables (mean ± SD; n = 5) Measured at Mary River Mine-Exposed and Reference (GO-09) Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

Note: The same letters next to Mary River study area data points indicates no significant difference between areas.

Table 5.3: Water Chemistry at Mary River Monitoring Stations, Mary River Project CREMP, August 2017

	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 2017	29-Aug-2017	29-Aug-2017	29-Aug-2017	29-Aug-2017	1-Sep-2017	1-Sep-2017	1-Sep-2017	1-Sep-2017	1-Sep-2017	1-Sep-2017	29-Aug-2017	27-Aug-2017	27-Aug-2017
Conventionals	Conductivity (lab)	umho/cm	-	-	116	120	158	138	132	151	266	158	164	164	163	146	151	143
	pH (lab)	pH	6.5 - 9.0	-	7.90	7.93	8.09	8.01	7.99	8.08	8.22	8.10	8.06	8.04	7.99	7.98	8.01	8.01
	Hardness (as $CaCO_3$)	mg/L	-	-	55	52	76	62	61	70	134	74	77	78	80	66	71	72
	Total Suspended Solids (TSS)	mg/L	-	-	4.1	2.3	<2.0	<2.0	<2.0	<2.0	5.2	<2.0	<2.0	<2.0	9.5	<2.0	20.2	3.3
	Total Dissolved Solids (TDS)	mg/L	-	-	60	68	87	59	61	74	136	76	71	79	82	78	70	71
	Turbidity	NTU	-	-	6.1	8.3	2.8	5.8	5.3	1.8	6.3	1.9	2.3	2.2	4.5	7.4	45.3	7.6
	Alkalinity (as $CaCO_3$)	mg/L	-	-	52	48	77	58	59	66	107	69	73	69	69	62	63	63
Nutrients and Organics	Total Ammonia	mg/L	variable	0.855	0.027	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	0.063	0.057	<0.020	0.029	0.028	<0.020	0.134	0.035	0.062	0.058	0.057	0.053	0.076	0.070
	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	0.22	0.15
	Dissolved Organic Carbon	mg/L	-	-	1.6	1.0	1.2	1.2	1.2	1.2	<1.0	1.1	1.0	1.1	1.3	1.3	1.8	1.2
	Total Organic Carbon	mg/L	-	-	1.7	<1.0	1.2	1.1	1.2	1.1	1.0	1.2	1.1	1.1	1.3	1.3	1.6	1.3
	Total Phosphorus	mg/L	0.020[α]	-	0.0078	0.0097	0.0045	0.0053	0.0060	0.0036	0.0067	0.0046	0.0196	0.0037	0.0060	0.0078	0.0250	0.0066
	Phenols	mg/L	0.004[α]	-	<0.0010	0.0015	<0.0010	0.0012	<0.0010	<0.0010	0.0012	0.0013	0.0012	<0.0010	<0.0010	0.0023	<0.0010	<0.0010
Anions	Bromide (Br)	mg/L	-	-	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10	<0.10
	Chloride (Cl)	mg/L	120	120	2.5	5.2	2.9	4.6	4.1	4.6	5.4	4.7	4.7	4.7	4.2	3.9	5.2	4.1
	Sulphate (SO_4)	mg/L	218[β]	218	4.1	3.2	2.5	2.8	2.7	2.9	25.3	4.3	7.5	7.5	6.7	5.5	4.8	3.8
Total Metals	Aluminum (Al)	mg/L	0.100	0.966	0.208	0.186	0.087	0.168	0.167	0.059	0.187	0.071	0.108	0.070	0.072	0.178	***1.480***	0.219
	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	0.00012	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	0.00016	<0.00010
	Barium (Ba)	mg/L	-	-	0.0076	0.0090	0.0087	0.0090	0.0089	0.0090	0.0138	0.0093	0.0102	0.0097	0.0101	0.0095	0.0179	0.0101
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00010	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00010	<0.00010
	Bismuth (Bi)	mg/L	-	-	<0.000050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.000050	<0.000050
	Boron (B)	mg/L	1.5	-	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<0.010	<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	-	-	11.2	10.9	16.2	13.3	12.6	13.7	26.5	14.9	15.2	15.7	15.1	14.0	14.8	13.9
	Chromium (Cr)	mg/L	0.0089	0.0089	0.00074	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	0.00306	<0.00050
	Cobalt (Co)	mg/L	0.0009[α]	0.004	0.00015	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	0.00017	<0.00010	<0.00010	<0.00010	<0.00010	0.00011	0.00091	0.00013
	Copper (Cu)	mg/L	0.002	0.0024	0.0013	0.0014	0.0007	0.0009	0.0009	0.0008	0.0010	0.0008	0.0009	0.0009	0.0009	0.0009	***0.0025***	0.0011
	Iron (Fe)	mg/L	0.30	0.874	0.222	0.146	0.052	0.103	0.105	0.043	0.237	0.053	0.098	0.053	0.044	0.159	***2.120***	0.237
	Lead (Pb)	mg/L	0.001	0.001	0.00021	0.00020	0.00006	0.00012	0.00011	<0.000050	0.00025	0.00006	0.00010	0.00007	<0.000050	0.00016	0.00086	0.00018
	Lithium (Li)	mg/L	-	-	0.0014	0.0011	<0.0010	0.0011	<0.0010	<0.0010	0.0015	<0.0010	<0.0010	<0.0010	<0.0010	0.0011	0.0023	<0.0010
	Magnesium (Mg)	mg/L	-	-	6.5	6.1	9.0	7.6	7.2	8.0	16.4	8.9	9.3	8.9	9.6	8.3	9.2	8.2
	Manganese (Mn)	mg/L	0.935[β]	-	0.0033	0.0026	0.0007	0.0015	0.0013	0.0006	0.0068	0.0010	0.0068	0.0051	0.0033	0.0064	0.0360	0.0054
	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	-	0.00034	0.00029	0.00018	0.00024	0.00023	0.00027	0.00026	0.00026	0.00050	0.00056	0.00055	0.00034	0.00034	0.00032
	Nickel (Ni)	mg/L	0.025	0.025	0.00068	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	0.00068	<0.00050	<0.00050	0.00050	0.00061	0.00059	0.00290	0.00078
	Potassium (K)	mg/L	-	-	0.82	1.07	0.94	1.05	0.98	0.92	1.38	0.97	1.03	0.98	1.12	1.04	1.60	1.11
	Selenium (Se)	mg/L	0.001	-	<0.000050	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.000050	<0.000050
	Silicon (Si)	mg/L	-	-	1.26	1.10	1.19	1.27	1.28	0.99	1.23	1.02	1.08	1.01	1.05	1.26	3.09	1.11
	Silver (Ag)	mg/L	0.00025	0.0001	<0.000050	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000010	<0.000050	<0.000050
	Sodium (Na)	mg/L	-	-	1.8	2.6	1.9	2.5	2.0	2.3	1.8	2.3	2.3	2.3	2.2	1.9	2.5	2.4
	Strontium (Sr)	mg/L	-	-	0.0113	0.0136	0.0131	0.0137	0.0123	0.0132	0.0191	0.0134	0.0157	0.0156	0.0144	0.0135	0.0138	0.0129
	Thallium (Tl)	mg/L	0.0008	0.0008	0.00001	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010	0.00003	<0.000010
	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	-	-	0.012	<0.010	<0.010	<0.010	<0.010	<0.010	0.014	<0.010	<0.010	<0.010	<0.010	<0.010	0.073	0.011
	Uranium (U)	mg/L	0.015	-	0.0021	0.0026	0.0027	0.0027	0.0022	0.0028	0.0026	0.0028	0.0027	0.0027	0.0016	0.0021	0.0021	0.0021
	Vanadium (V)	mg/L	0.006[α]	0.006	0.0007	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	0.00231	0.00055
	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	0.0038	<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.

concentrations of these metals at this station appeared to be associated with elevated turbidity at the time of the fall sampling event (Table 5.3). This was supported by no elevation in the corresponding dissolved concentrations for these metals between Station CO-05 and the reference stations, nor between the other mine-exposed stations and the reference stations, in each of the spring, summer, and fall sampling events (Appendix Table C.64).

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 2017 (Table 5.3; 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 summer and fall monitoring events in 2017 (and during previous study years), suggesting naturally high concentrations of aluminum in the Mary River system. Although a number of other metals were also above WQG and/or AEMP benchmarks at Mary River Station CO-05 during fall monitoring in 2017, as discussed above, high concentrations appeared to be associated with elevated turbidity at the time of sampling (Appendix Table C.62). Notably, a high proportion (i.e., ≥65%) of aluminum, iron, and manganese were in the 'total' concentration form, suggesting that these metals were largely associated with suspended particulate matter and were unlikely to be bioavailable. Of these metals, total aluminum and iron concentrations showed a strong positive correlation with turbidity (i.e., r_s of 0.71 and 0.74, respectively; Appendix Table C.66). High turbidity was observed at the river reference (i.e., GO series) stations indicating that elevated turbidity in the Mary River was a natural phenomenon unrelated to the Mary River Project operations.

Temporal evaluation of Mary River water chemistry data indicated that parameter concentrations in 2017 were almost all within respective parameter concentration ranges measured at each station over the mine baseline period (2005-2013; Figure 5.3; Appendix Figure C.22). The only exception occurred at Station CO-05, where in fall 2017 higher total concentrations of several metals, including iron and manganese, was observed compared to baseline and previous years of commercial mine operation (Figure 5.3; Appendix Figure C.22). However, as discussed previously, higher total concentrations of these metals in fall 2017 reflected much greater amounts of material suspended in the water column than during the baseline period (e.g., on average, turbidity at Mary River Station CO-05 was 11.5 times higher in 2017 than during the baseline sampling in fall; Appendix Figure C.22). Concentrations of more conservative parameters commonly used as indicators of anthropogenic influences in aquatic environments (e.g., chloride, conductivity, nitrate, sulphate, hardness) indicated no substantial water quality changes between 2017 and the baseline period at the Mary River mine-exposed stations during all spring, summer, and fall sampling events (Figure 5.3; Appendix Figure C.22). Overall, no marked mine-related

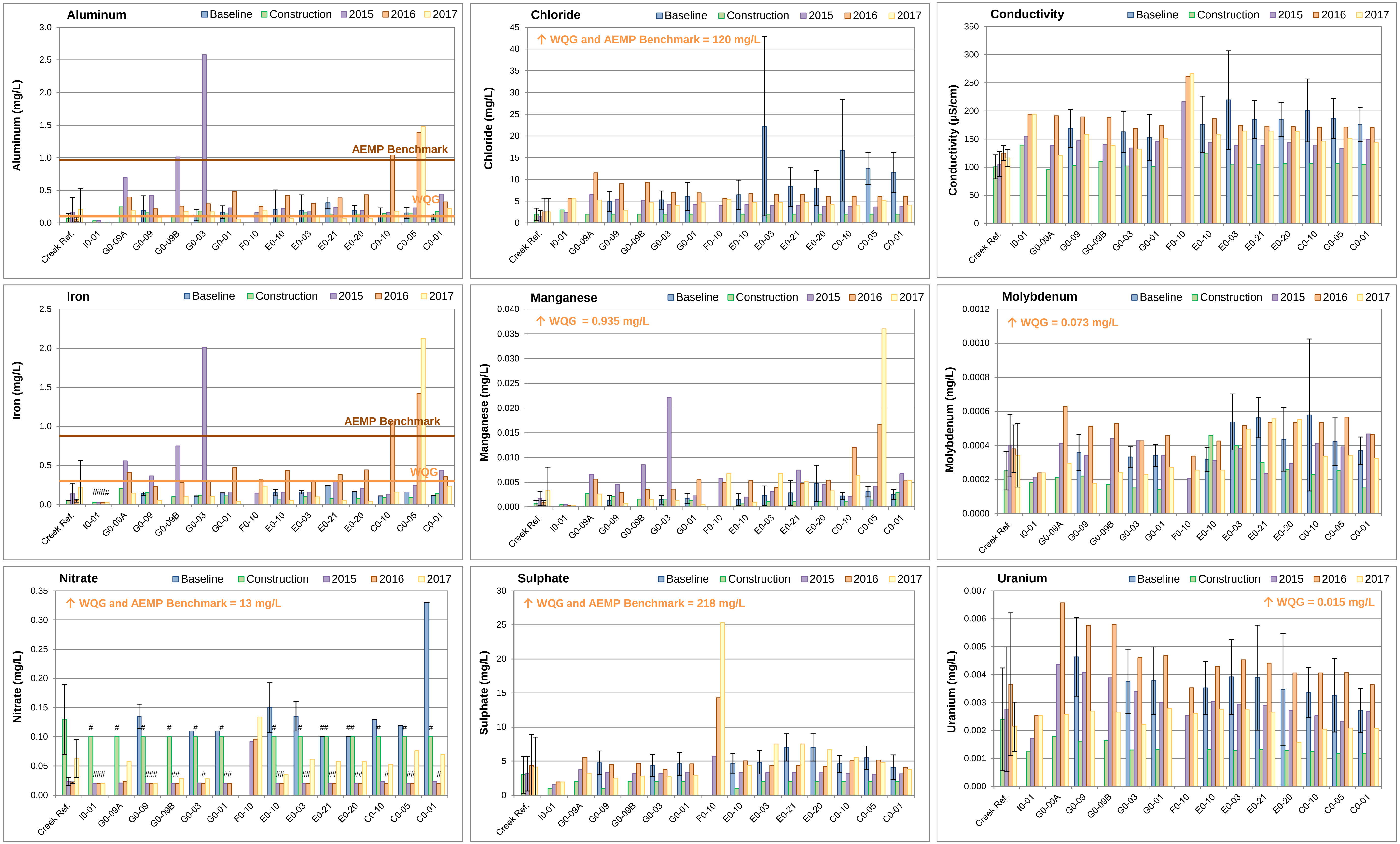


Figure 5.3: Temporal Comparison of Water Chemistry at Mary River Stations for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015 - 2017) Periods during Fall

Notes: Values represent mean ± SD. Lotic reference stations include the CLT-REF and MRY-REF series (mean ± SD; n = 4). Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to the Camp Lake Tributaries.

influences to water quality of the Mary River were indicated in 2017 based on comparisons to reference conditions and to mine baseline data.

5.2.2 Sediment Quality

Deposited sediment at Mary River study areas was variable, visually characterized as ranging from predominantly medium sand (CO-05), medium to coarse sand (GO-03, EO-20), coarse sand (GO-09), and very coarse sand (EO-01) among the five study areas (Appendix Table D.36). Substrate among the Mary River study areas was largely composed of cobble and boulder material with minimal amounts of sand and finer material except at the downstream-most study area CO-05, where medium sand composed approximately 65% of the surficial in-stream substrate (Appendix Table F.1). Silt precipitate and/or deposits were generally absent at all Mary River study areas during the August 2017 sampling event. Deposited sediment for TOC and metal determination was collected mainly from shoreline/streambank areas at most of the Mary River study areas, although in-stream material was occasionally collected from depositional areas behind boulders or, for study area CO-05, directly from the channel (Appendix Table D.36). Sediment TOC content was low (i.e., <0.1%) at all of the Mary River study areas, and showed no detectable change between the mine-exposed and GO-09 upstream reference study areas which suggested similar depositional characteristics among the Mary River study areas (Table 5.2; Appendix Table D.39).

Metal concentrations in deposited sediment at Mary River study areas located upstream of the mine were highly comparable between the GO-09 and GO-03 study areas (Table 5.4; Appendix Table D.39). Downstream of the MRTF confluence with Mary River at the EO-01 mine-exposed area, deposited sediment contained slightly elevated (i.e., 3- to 5-fold higher) concentrations of arsenic, chromium, iron, and nickel compared to average concentrations at the upstream GO-09 reference area (Table 5.4). With the exception of arsenic, concentrations of these metals were moderately elevated (i.e., 5-fold to 10-fold higher), and concentrations of calcium, cobalt, copper, magnesium, manganese, and vanadium were slightly elevated, in deposited sediment further downstream at the EO-20 mine-exposed area compared to respective average concentrations at the GO-09 reference area (Table 5.4; Appendix Table D.39). However, deposited sediment collected downstream of the mine at the CO-05 mine-exposed area showed only slightly elevated concentrations of chromium and copper, and moderately elevated concentrations of nickel, compared to deposited sediment at the GO-09 reference area (Table 5.4), indicating that mine-related influences on deposited sediment metal concentrations were greatest adjacent to the mine. Notably, concentrations of metals were all well below applicable SQG in deposited sediment collected at each of the five Mary River study areas (Table 5.4; Appendix Tables D.37 – D.38 and D.40 – D.42). No baseline sediment metal concentration data were collected at Mary

Table 5.4: Sediment Total Organic Carbon and Metal Concentrations at Mary River Mine-Exposed and Reference (GO-09) Sediment Monitoring Stations, Mary River Project CREMP, August 2017

Parameter	Units	Sediment Quality Guideline (SQG)[a]	Mary River Reference (GO-09; n = 5)	Mary River Mine-Exposed Areas			
				Upstream GO-03 (n = 5)	Adjacent EO-01 (n = 5)	Adjacent EO-20 (n = 5)	Downstream CO-05 (n = 5)
			Average ± SD	Average ± SD	Average ± SD	Average ± SD	Average ± SD
Total Organic Carbon	%	10[α]	<0.10 ± 0.00	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Aluminum (Al)	mg/kg	-	763 ± 271	790 ± 96	1,791 ± 875	2,034 ± 1,573	1,302 ± 719
Antimony (Sb)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0.00	<0.10 ± 0.00	<0.10 ± 0
Arsenic (As)	mg/kg	17	0.16 ± 0.02	0.15 ± 0.02	0.51 ± 0.51	0.46 ± 0.18	0.31 ± 0.08
Barium (Ba)	mg/kg	-	4 ± 1.2	4 ± 0.4	9 ± 4.8	9 ± 6.4	5 ± 2
Beryllium (Be)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	0.12 ± 0.04	<0.10 ± 0
Bismuth (Bi)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0
Boron (B)	mg/kg	-	<5.0 ± 0	<5.0 ± 0	<5.0 ± 0	5.2 ± 0.4	<5.0 ± 0
Cadmium (Cd)	mg/kg	3.5	<0.020 ± 0	<0.020 ± 0	<0.020 ± 0	0.023 ± 0.005	<0.020 ± 0
Calcium (Ca)	mg/kg	-	842 ± 508	619 ± 83	1,750 ± 924	3,094 ± 2,706	1,056 ± 593
Chromium (Cr)	mg/kg	90	3.4 ± 1.0	3.4 ± 1.3	13.1 ± 4.3	23.1 ± 9.9	13.7 ± 8.9
Cobalt (Co)	mg/kg	-	0.67 ± 0.19	0.69 ± 0.14	1.85 ± 0.69	2.89 ± 1.34	1.68 ± 0.80
Copper (Cu)	mg/kg	110[α]	1.3 ± 0.5	1.4 ± 0.4	3 ± 0.9	6 ± 6.1	3.9 ± 4.7
Iron (Fe)	mg/kg	40,000[α]	2,826 ± 1,271	2,916 ± 1,532	8,534 ± 3,433	15,518 ± 8,546	7,726 ± 8,774
Lead (Pb)	mg/kg	91	1.1 ± 0.1	1.2 ± 0.2	1.7 ± 0.5	2.3 ± 0.7	1.6 ± 0.6
Lithium (Li)	mg/kg	-	2.2 ± 0.5	<2.0 ± 0.0	3.5 ± 1.3	4.9 ± 4.0	3.5 ± 1.8
Magnesium (Mg)	mg/kg	-	826 ± 521	682 ± 94	2,252 ± 1,276	3,617 ± 3,678	1,894 ± 1,393
Manganese (Mn)	mg/kg	1,100[α,β]	22 ± 7.1	21 ± 3.3	59 ± 22	85 ± 48	49 ± 18
Mercury (Hg)	mg/kg	0.486	<0.0050 ± 0	<0.0050 ± 0	<0.0050 ± 0.0000	<0.0050 ± 0.0000	<0.0050 ± 0.0000
Molybdenum (Mo)	mg/kg	-	0.11 ± 0.01	<0.10 ± 0	0.12 ± 0.03	0.16 ± 0.08	<0.10 ± 0
Nickel (Ni)	mg/kg	75[α,β]	1.9 ± 0.7	1.8 ± 0.3	7.8 ± 3.8	15.3 ± 12.9	10.3 ± 6.1
Phosphorus (P)	mg/kg	2,000[α]	113 ± 49	121 ± 44	236 ± 94	316 ± 142	213 ± 110
Potassium (K)	mg/kg	-	168 ± 77	166 ± 13	480 ± 302	466 ± 407	236 ± 138
Selenium (Se)	mg/kg	-	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0	<0.20 ± 0
Silver (Ag)	mg/kg	-	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0	<0.10 ± 0
Sodium (Na)	mg/kg	-	<50 ± 0	<50 ± 0	<50 ± 0	59 ± 13	<50 ± 0
Strontium (Sr)	mg/kg	-	1.9 ± 0.3	1.9 ± 0.2	3.1 ± 1.2	3.6 ± 1.6	2.5 ± 0.6
Sulphur (S)	mg/kg	-	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0	<1,000 ± 0
Thallium (Tl)	mg/kg	-	<0.050 ± 0	<0.050 ± 0	0.051 ± 0.001	0.055 ± 0.012	<0.050 ± 0
Tin (Sn)	mg/kg	-	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0	<2.0 ± 0.0
Titanium (Ti)	mg/kg	-	109 ± 26	114 ± 21	201 ± 72	304 ± 171	194 ± 79
Uranium (U)	mg/kg	-	0.30 ± 0.0	0.31 ± 0.0	0.4 ± 0.1	0.8 ± 0.2	0.48 ± 0.2
Vanadium (V)	mg/kg	-	5.0 ± 2.2	5.2 ± 3.1	12.7 ± 4.7	25.0 ± 14.1	12.9 ± 14.7
Zinc (Zn)	mg/kg	315	3.7 ± 1.3	3.1 ± 0.6	6 ± 2.8	9 ± 6.8	5 ± 2.6
Zirconium (Zr)	mg/kg	-	1.7 ± 0.3	1.7 ± 0.2	2.2 ± 0.5	2.8 ± 1.2	2.0 ± 0.5

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

Indicates parameter concentration above Sediment Quality Guideline (SQG).

River study areas, and thus no evaluation of potential mine-related influences on sediment quality following commencement of commercial mine operations could be conducted.

5.2.3 Phytoplankton

Mary River chlorophyll-a concentrations at stations downstream of the mine were generally within the range of the GO series river reference stations and/or creek reference stations during the 2017 spring, summer, and/or fall sampling events (Figure 5.4). In addition, chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events at all Mary River sampling stations in 2017, 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 in 2017. Low to moderate phytoplankton productivity was predicted for Mary River reference and mine-exposed stations given 'oligotrophic' to 'mesotrophic' CWQG categorization based on total phosphorus concentrations of up to 25 µg/L in 2017 (Table 5.3; Appendix Table C.62).

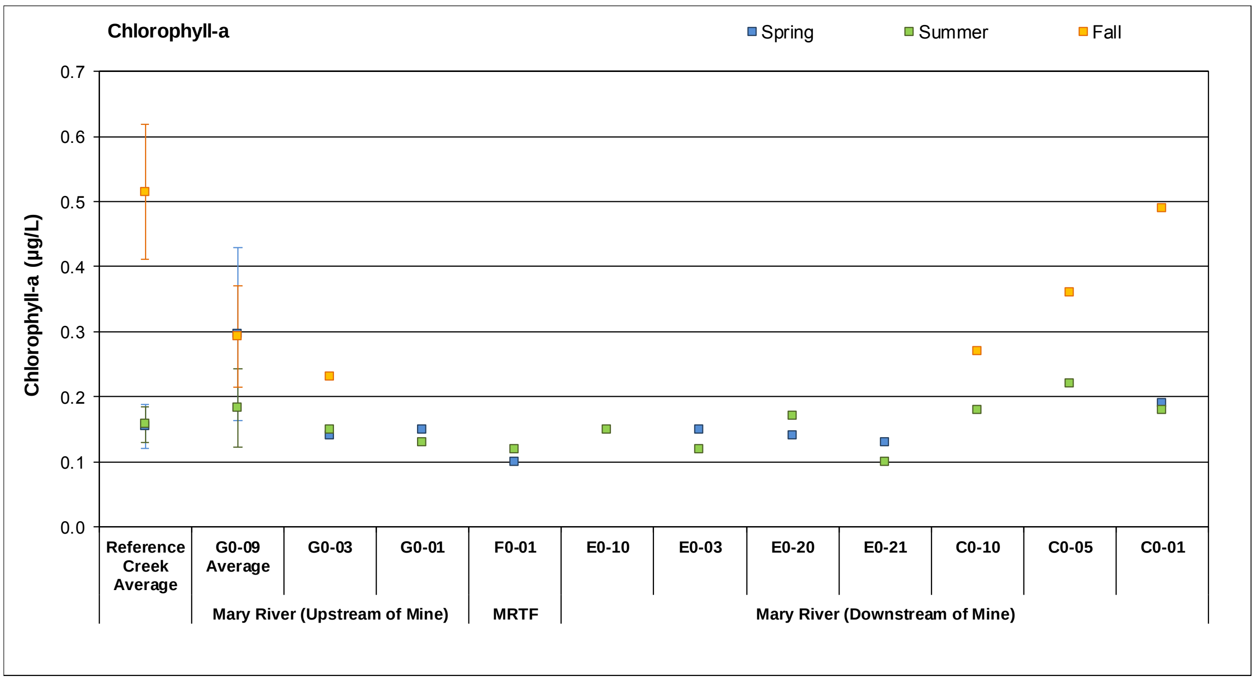


Figure 5.4: Chlorophyll-a Concentrations at Mary River Phytoplankton Monitoring Stations Located Upstream and Downstream of the Mine, Mary River Project CREMP, 2017

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

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 2017 and in each of the two previous years of mine operation (2015, 2016) than those observed during the baseline period (Figure 5.5). 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 5.5). 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 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 2015, 2016, 2017, and the baseline period were consistent with effects associated with natural differences in turbidity (i.e., originating from sources upstream of the mine), but may also reflect influences from sources currently unidentifiable.

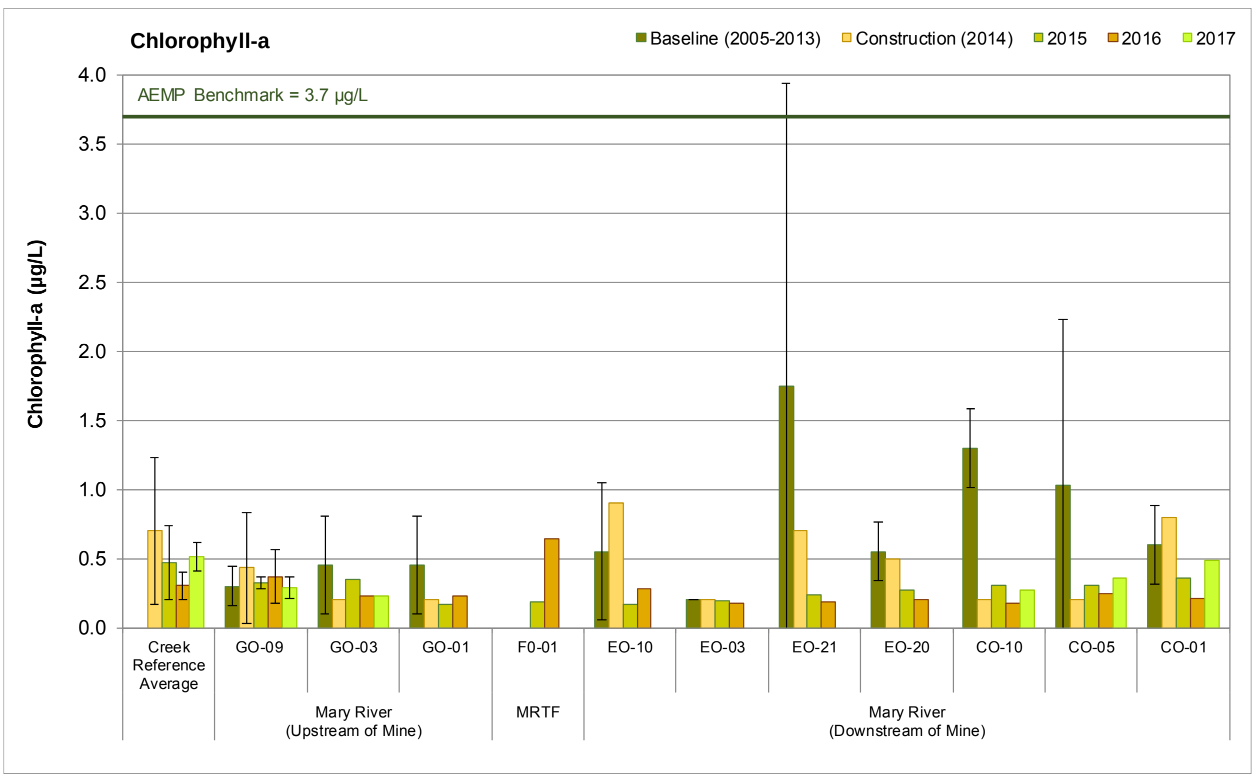


Figure 5.5: Temporal Comparison of Chlorophyll-a Concentrations at Mary River Stations for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015 - 2017) Periods during the Fall

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

5.2.4 Benthic Invertebrate Community

The Mary River benthic invertebrate community assessment included a spatial statistical analysis of key benthic endpoints among upstream reference areas (GO-09, GO-03), near-field mine-exposed areas located adjacent to the mine (EO-01, EO-20) and a far-field, cumulative effects mine-exposed area located downstream of the mine (CO-05; see Table 2.5, Figure 2.4). Benthic invertebrate density, richness, and Simpson's Evenness at Mary River near-field mine-exposed study areas EO-01 and EO-20 did not differ significantly from the GO-09 reference area in 2017 (Figure 5.6; Appendix Table F.58). Benthic invertebrate density and richness were significantly higher at the Mary River CO-05 far-field mine-exposed area compared to the GO-09 reference area (Figure 5.6), suggesting slight nutrient enrichment downstream of the mine. In addition to these differences, benthic invertebrate community structure at the near- and far-field mine-exposed areas of Mary River differed significantly from the upstream GO-09 reference area based on evaluation of Bray-Curtis Index (Appendix Table F.58).

Key differences in dominant benthic invertebrate taxonomic groups at Mary River near- and far-field mine-exposed areas compared to the upstream GO-09 reference area included the presence of Ephemeroptera (mayflies) at the mine-exposed areas, as well as a significantly lower relative abundance of metal-sensitive chironomids at near-field study area EO-01 (Figure 5.6; Appendix Table F.58). Because the relative abundance of metal-sensitive chironomids did not differ significantly between the near-field mine-exposed area EO-01 and the upstream reference area GO-03, and because mayflies, which are relatively metal-sensitive, were also present at near-field mine-exposed area EO-01, differences in benthic invertebrate community structure at EO-01 were not considered mine-related. No ecologically significant differences in the relative abundance of individual FFG were indicated between any of the Mary River mine-exposed areas compared to the GO-09 reference area (Figure 5.6; Appendix Table F.58), suggesting no mine-related influences to aquatic food resources available to benthic invertebrates. In addition, no ecologically significant differences in the relative abundance of individual HPG were indicated between any of the Mary River mine-exposed areas compared to the GO-09 reference area with the exception of at far-field mine-exposed area CO-05, where significantly higher proportion of burrowers was indicated compared to the reference area (Figure 5.6). A greater proportion of the burrower HPG at mine-exposed area CO-05 was consistent with significantly greater substrate embeddedness and greater proportion of sand substrate at this area compared to the GO-09 upstream reference area (Appendix Tables F.50 and F.53). Because the relative abundance of metal-sensitive taxa was similar between the mine-exposed and reference areas of Mary River, the differences in benthic invertebrate community structure suggested by differing Bray-Curtis Index between Mary River mine-exposed and reference areas in 2017 likely reflected natural differences in physical habitat features, including the amount of substrate embeddedness.

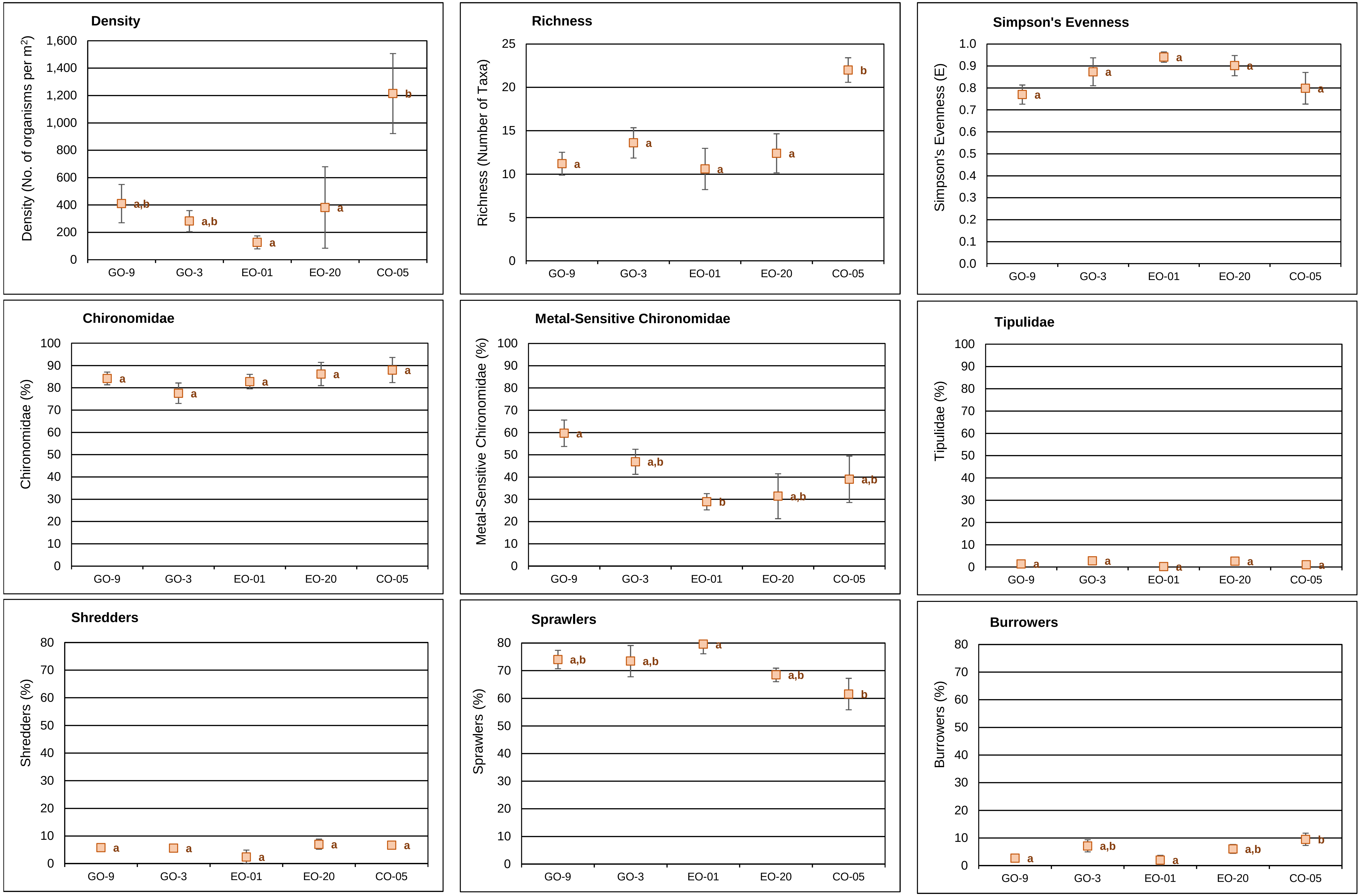


Figure 5.6: Comparison of Benthic Invertebrate Community Metrics among Mary River Study Areas (mean ± SE), Mary River Project CREMP, August 2017

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

Temporal comparison of the Mary River benthic invertebrate community data indicated no ecologically significant differences in density between mine operational (2015, 2016, 2017) and baseline (2006 – 2011 data) periods at any of the mine-exposed study areas (i.e., EO-01, EO-20, or CO-05; Figure 5.7; Appendix Tables F.61 – F.63). Simpson's Evenness and chironomid relative abundance were often significantly higher and lower, respectively, at the Mary River mine-exposed areas during studies conducted at the time of mine operation compared to those conducted during mine baseline. However, these same benthic metrics showed the same direction of significant differences at Mary River upstream reference areas (Appendix Tables F.59 – F.63), suggesting that the differences in these metrics at all Mary River areas over time reflected natural temporal variability and/or represented sampling artifacts of the CREMP (e.g., changes in sampling location, personnel collecting samples, etc.). Although the relative abundance of the collector-gatherer FFG was significantly higher at near-field mine-exposed area EO-20 in each of 2015, 2016, and 2017 than during the 2011 baseline study, the proportion of collector-gatherers at this area became more similar to the reference condition in the mine operational years suggesting that the temporal changes were not mine-related (Appendix Tables F.59 and F.62). Notably, the types, direction, and magnitude of difference for endpoints that differed significantly between the mine operational and baseline periods at the Mary River mine-exposed areas were similar among the 2015 - 2017 CREMP studies (Figure 5.7), suggesting no cumulative temporal influences on benthic invertebrates of the Mary River since the commencement of commercial mine operations in 2015.

5.2.5 Fish Population

5.2.5.1 Mary River Fish Community

The fish community at the near-field area of Mary River, adjacent to the mine, was composed only of arctic charr, which differed slightly from that of the far-field area where low numbers of ninespine stickleback were captured in addition to arctic charr (Table 5.2; Appendix Table G.28). Arctic charr CPUE was substantially higher at the near-field area than further downstream near the mouth of the river at Mary Lake (Table 5.2), suggesting greater abundance of arctic charr closer to the mine. The between-area difference in arctic charr abundance may have reflected natural differences in the type of habitat sampled between the Mary River near- and far-field study areas. At the near-field study area, the predominant habitat consists of side and braided channels characterized by variable water velocity and large, loosely embedded cobble substrate, whereas at the far-field study area, habitat is dominated by a single main channel characterized by relatively deep, fast flowing water over highly embedded boulder substrate (Appendix Photo Plate G.2). These habitat features allowed fish sampling to be conducted throughout side-channels at the near-field study area, but limited the sampling to shoreline areas at the far-field

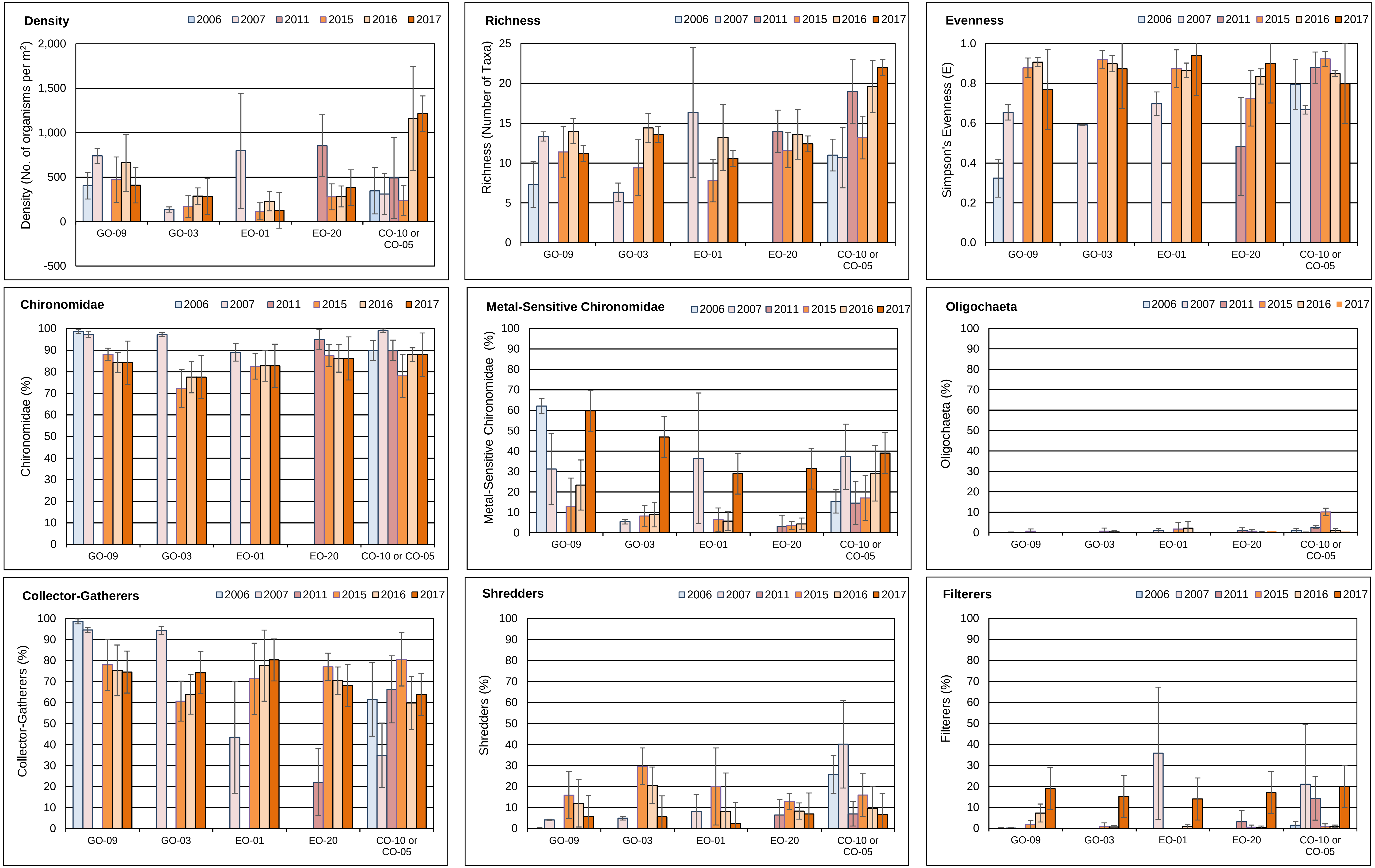


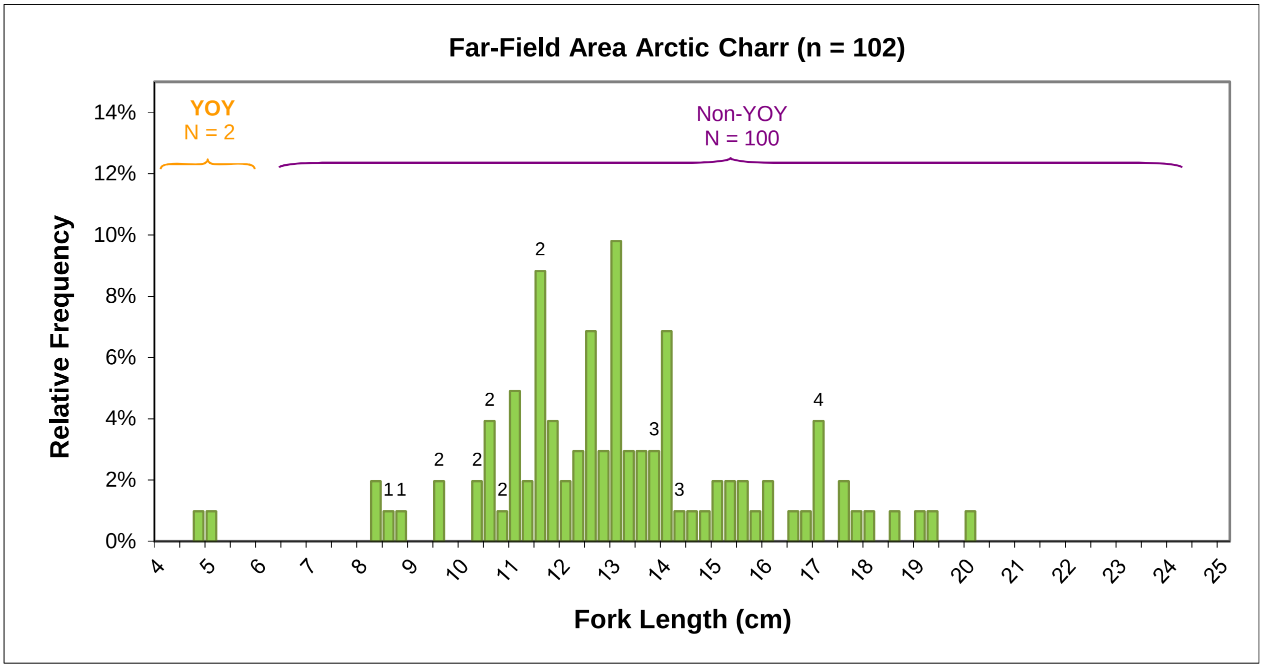
Figure 5.7: **Comparison of Benthic Invertebrate Community Metrics (mean ± SD) at Mary River Study Areas among Baseline (2006, 2007, 2011) and Operational (2015, 2016, 2017) Years for the Mary River Project CREMP**

study area. In addition, deeper, faster flowing water at the far-field study area likely reduced fish catch efficiency by limiting field study team mobility. Overall, the determination of mine-related influences on fish community composition and arctic charr abundance between Mary River near- and far-field study areas was partly confounded by differing habitat features and the commensurate influences on sampler efficiency.

5.2.5.2 Mary River Fish Population Assessment

Non-lethal measurements of length and weight were collected from 102 and 100 arctic charr at Mary River near- and far-field study areas, respectively, for the assessment of fish population endpoints (Appendix Tables G.29 and G.30). Arctic charr YOY were distinguishable from non-YOY individuals at a fork length of 5.0 cm based on evaluation of length-frequency distributions coupled with supporting age determinations (Figure 5.8). Based on this cut-off value, no YOY were captured at the near-field area, and only two YOY were captured at the far-field area (i.e., approximately 2% of arctic charr population). As a result, the Mary River arctic charr population assessment focused on non-YOY individuals.

Arctic charr length-frequency distributions did not differ significantly between the Mary River near- and far-field study areas, regardless of whether YOY were included or excluded from the data set (Table 5.5; Appendix Figure G.22). Because the inclusion of YOY did not change the outcome of the length-frequency distribution statistical comparison, no difference in the proportion of YOY was indicated between the Mary River study areas (Table 5.5). Among non-YOY arctic charr, no separation of age (i.e., cohorts) was possible at either study area using the length-frequency distribution and confirmatory aging results (Figure 5.8). Nevertheless, visual evaluation of the plotted data suggested a similar arctic charr length-at-age relationship between the Mary River near- and far-field areas (Figure 5.8). Fork length and body weight of non-YOY arctic charr captured at the near-field area did not differ significantly from those captured at the far-field area (Table 5.5). Although condition (i.e., weight-at-length relationship) of non-YOY individuals was significantly lower at the Mary River near-field area than at the far-field area, the magnitude of this difference was within applicable CES (i.e., ±10%; Table 5.5; Appendix Table G.31) suggesting that this difference was not ecologically meaningful. No externally visible abnormalities or parasitic infections were observed on any arctic charr captured at the Mary River study areas. Overall, no significant, ecologically meaningful differences in arctic charr non-YOY health endpoints were indicated between the near- and far-field study areas, suggesting limited influence of the mine on the health of arctic charr within Mary River in 2017.



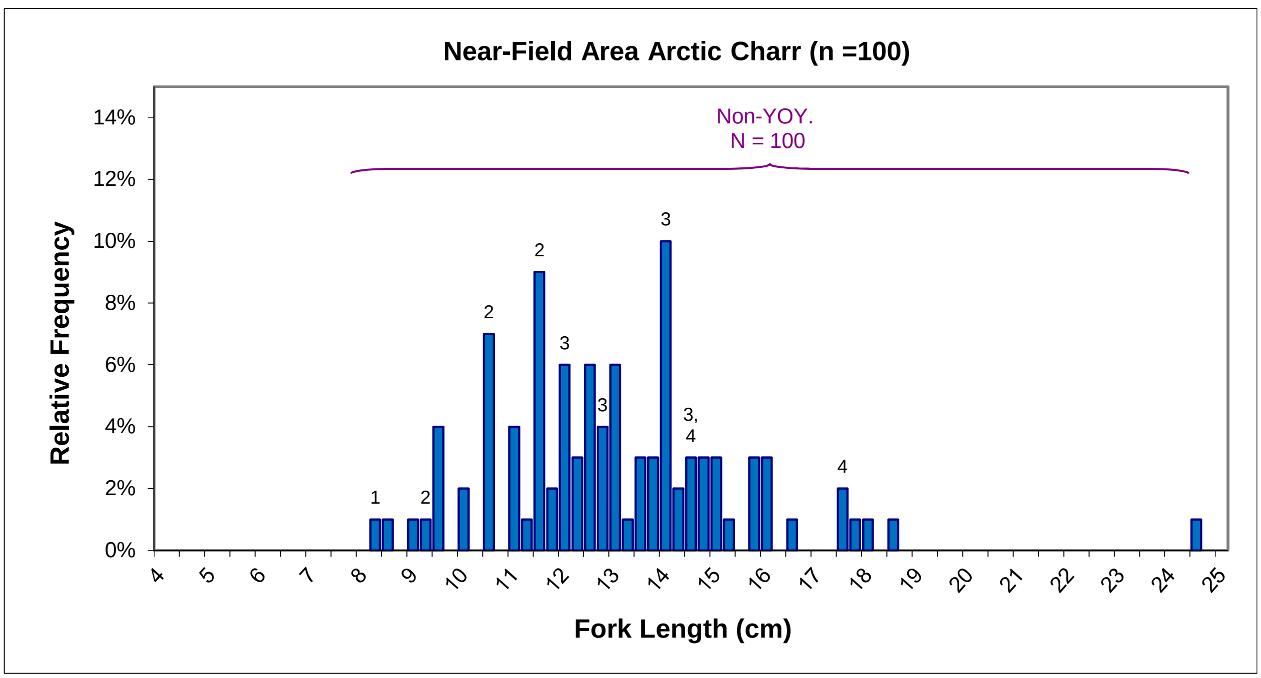


Figure 5.8: Length-frequency Distributions for Arctic Charr Collected at Mary River Project Phase 1 EEM Effluent-Exposed and Reference Study Areas, August 2017

Note: Numbers above bars represent individual fish ages, where available.

Table 5.5: Summary of Arctic Charr Population Statistical Comparison Results between Near-Field and Far-Field Areas of Mary River, August 2017

Endpoint		Significant Difference		Magnitude of Difference (%)
		Yes/No	p-value	
Survival – Length Frequency Distribution	All Fish	No	0.936	-
	Non-YOY only	No	0.906	-
Growth	Non-YOY length	No	0.523	-
	Non-YOY weight	No	0.200	-
Energy Storage	Non-YOY condition	Yes	<0.001	-4.5
Reproduction	YOY Proportion	No		

5.2.6 Integrated Effects Evaluation

Mine-related influences on water quality of Mary River in 2017 were limited to slight elevation of manganese concentrations at mine-exposed areas compared to the upstream reference area in summer and fall sampling events. Although total concentrations of a number of metals, including aluminum, chromium, cobalt, iron, lead, nickel, and phosphorus, were elevated at one or more mine-exposed areas of the Mary River in 2017 compared to reference and baseline data, naturally high turbidity in 2017 likely accounted for these spatial and temporal differences. This was supported by the occurrence of similar dissolved metal concentrations at Mary River mine-exposed areas in 2017 compared to Mary River reference and baseline data, by significant positive correlations between total concentrations of key metals (e.g., aluminum, manganese) and turbidity, and by observations of high ratios of total to dissolved metal concentrations for the Mary River water quality data. Notably, turbidity within Mary River was often highest upstream of the mine (i.e., the GO series stations) during all mine baseline (2005 – 2013) and operational (2015, 2016, 2017) periods, indicating that the dominant source of turbidity at mine-exposed areas of the Mary River reflected natural (runoff) inputs unrelated to the mine operation. Although total aluminum concentrations were above WQG and/or AEMP benchmarks at one or more Mary River mine-exposed stations in 2017, as discussed above, the elevation in these metals compared to water quality criteria appeared to be associated with naturally high turbidity. Deposited sediment at Mary River mine-exposed areas most frequently contained elevated concentrations of chromium, copper, iron, and nickel compared to the reference area, and were generally highest adjacent to the mine. However, concentrations of all metals were well below SQG and deposited sediment material composed less than 1% of surficial bed material at all but the downstream-most station of Mary River.

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 2017 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. No adverse or ecologically meaningful significant differences in benthic invertebrate density, richness, or relative abundance of metal-sensitive taxa were shown between Mary River mine-exposed areas compared to the upstream reference area (i.e., GO- 09) in 2017. Although some differences in community composition were indicated between the Mary River mine-exposed and reference areas in 2017, these differences appeared to be related to naturally greater substrate embeddedness at the mine-exposed areas rather than a mine-related influence. Temporal comparisons indicated significantly higher Simpson's Evenness and significantly lower relative abundance of chironomid midges at Mary River mine-exposed areas compared to the reference area between the 2017 and baseline studies. However, because the direction of these responses was opposite to those typically related to adverse mine-related effects, natural temporal variability and/or sampling artifacts of the CREMP likely accounted for the temporal differences in these endpoints. No ecologically significant differences in arctic charr size and condition were indicated between Mary River near- and far-field study areas in 2017, suggesting no mine-related influences on fish populations of Mary River with closer proximity to the mine. Overall, the chlorophyll-a, benthic invertebrate community, and arctic charr fish population data suggested no adverse mine-related influences to Mary River biota since the commencement of commercial mine operations.

5.3 Mary Lake

5.3.1 Hydraulic Retention Time

A hydraulic retention time of 75 ± 29 days was estimated for Mary Lake using mean annual watershed runoff (2007 – 2016 data) extrapolated from Baffinland flow monitoring stations installed in the primary tributaries of Mary Lake (Tom and Mary rivers) and at small watercourses (i.e., watershed ≤15 km^2) located on the mine property. The volume of Mary Lake used for the calculation of hydraulic retention time was 156.35 million cubic metres.

5.3.2 Water Quality

Water quality profiles taken at Mary Lake in 2017 showed similar *in situ* water temperature, dissolved oxygen saturation and pH values, but consistently higher specific conductance, at the north basin compared to the south basin throughout the year (Figures 5.9 and 5.10). Water

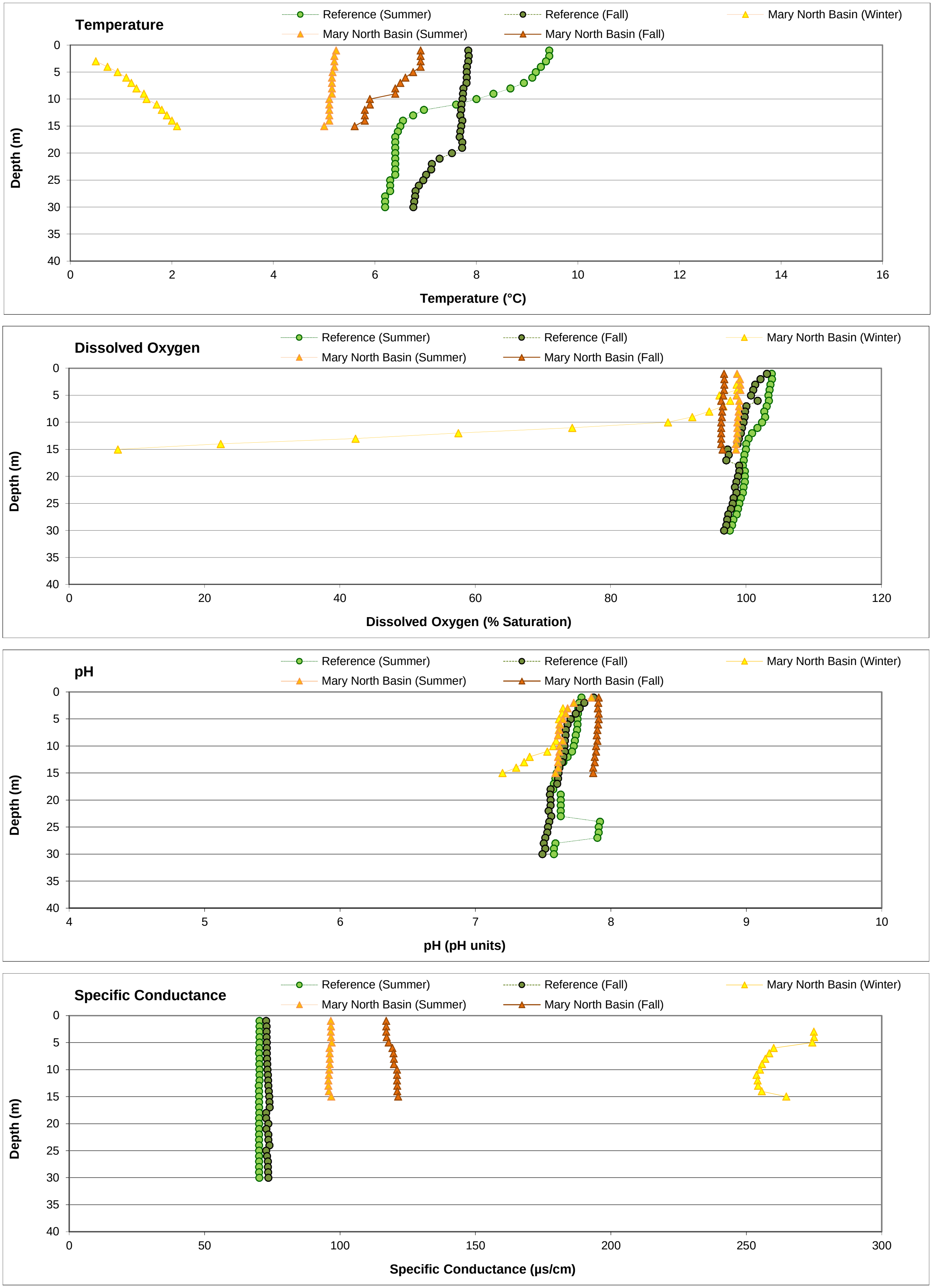


Figure 5.9: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake North Basin (BLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2017

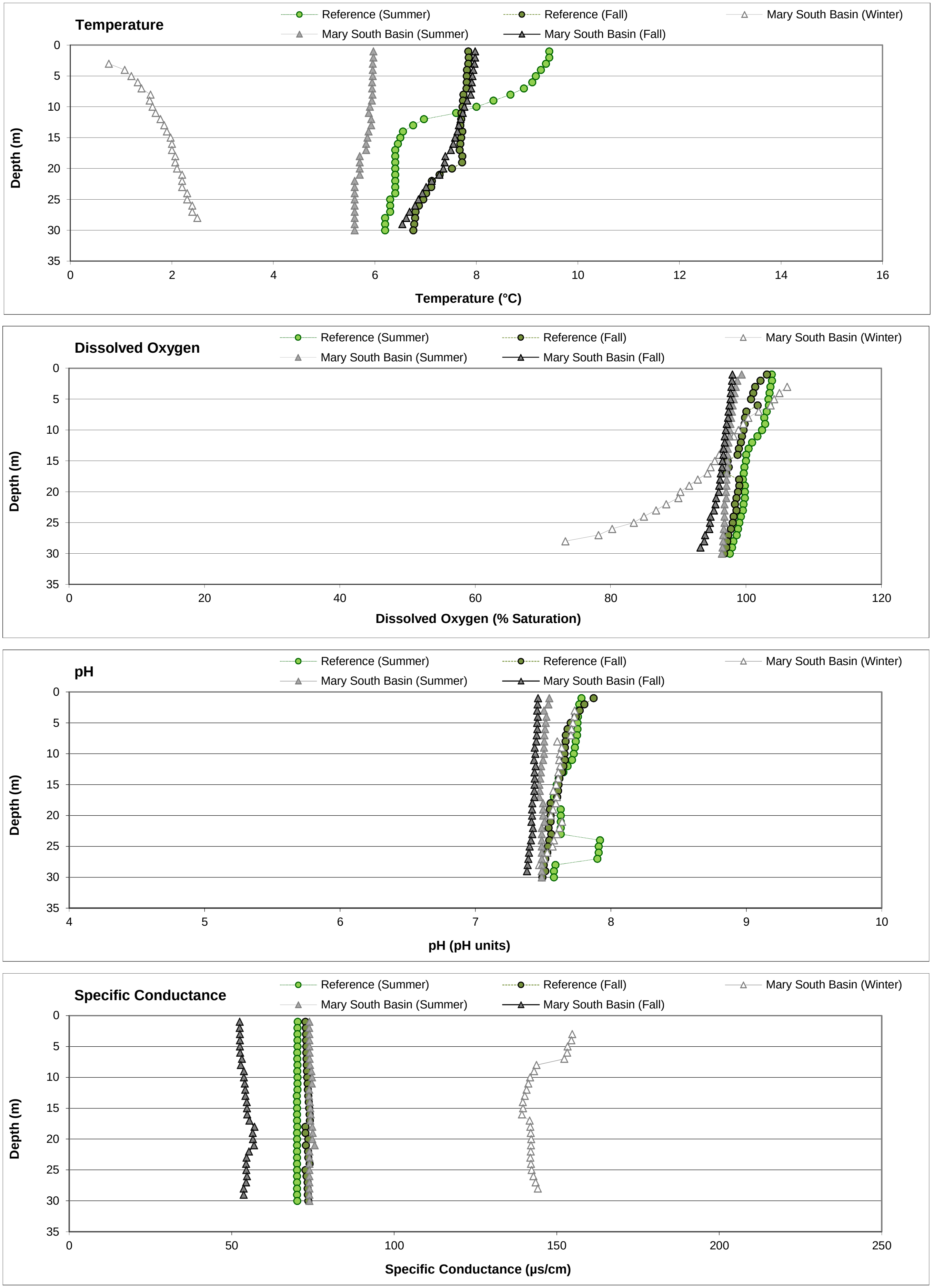


Figure 5.10: Average *In Situ* Water Quality with Depth from Surface at the Mary Lake South Basin (BLO) Compared to Reference Lake 3 during Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2017

temperatures showed a gradient from surface to bottom during the winter and fall at the Mary Lake north and south basins. However, the temperature profile did not suggest any establshied thermal layers at either basin during the winter and fall sampling events in 2017 (Figures 5.9 and 5.10). The Mary Lake temperature profiles contrasted with those of the reference lake, where weak thermal stratification occurred during the summer and fall sampling events (Figures 5.9 and 5.10). Nevertheless, mean water temperature at the bottom of water column at Mary Lake littoral and profundal stations did not differ significantly from the reference lake at comparable station depths in fall 2017 (Figure 5.11; Appendix Table C.72).

Dissolved oxygen profiles conducted at Mary Lake in 2017 indicated the development of a strong oxycline at the north basin in winter beginning at a depth of approximately 7 m, and a weak oxycline at the south basin in winter through the entire water column (Figures 5.9 and 5.10). However, similar to Reference Lake 3, no oxycline development was apparent in the summer or fall of 2017 at either Mary Lake basin. Dissolved oxygen saturation levels at Mary Lake remained above WQG (i.e., 54%) through the entire water column at the south basin in all seasons, and at the north basin in summer and fall seasons (Figures 5.9 and 5.10). However, dissolved oxygen saturation levels below the WQG of 54% occurred at depths greater than approximately 12.5 m at the Mary Lake north basin in the winter (Figure 5.9). During the 2017 fall sampling event, dissolved oxygen saturation levels at Mary Lake littoral and profundal stations were well above the applicable WQG at the bottom of the water column, but were significantly lower than those at Reference Lake 3 (Figure 5.11; 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 winter, summer, or fall sampling in 2017, and were also comparable to pH profiles at Reference Lake 3 (Figures 5.9 and 5.10). No significant differences in bottom pH were indicated between Mary Lake and Reference Lake 3 at littoral stations sampled in fall 2017, but significantly lower pH was indicated at profundal stations of Mary Lake compared to the reference lake (Figure 5.11; Appendix Table F.72). However, pH values at Mary Lake water quality and benthic littoral stations were consistently within WQG limits (Figure 5.10). Specific conductance was substantially higher at the north basin compared to the south basin of Mary Lake (Figures 5.9 and 5.10; 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 2017, reflecting similar profile structure at Reference Lake 3 (Figures 5.9 and 5.10). Although specific conductance at the bottom of the water column of littoral and profundal stations was

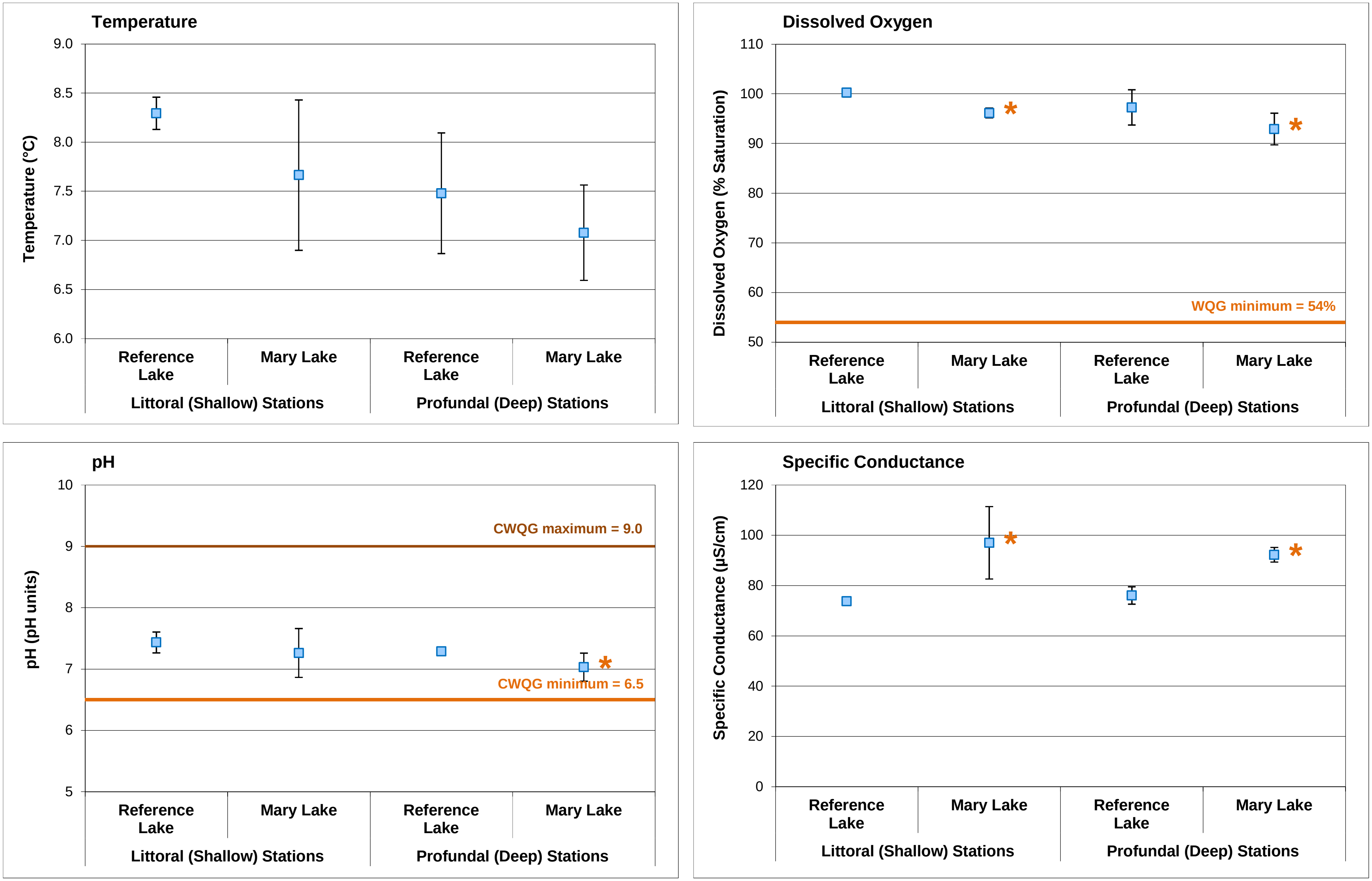


Figure 5.11: Comparison of *In Situ* Water Quality Variables (mean ± SD) Measured at Mary Lake (BLO) and Reference Lake 3 (REF3) Littoral and Profundal Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2017

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

significantly higher at Mary Lake than at the reference lake, mean values of 97 and 92 µS/cm at littoral and profundal stations, respectively, of Mary Lake were intermediate to that of the reference creek and river areas (i.e., range from 55 – 168 µS/cm) in fall 2017. Similar to the other mine-exposed lakes, the extent to which higher specific conductance at Mary Lake was related to natural regional variability or a mine-related influence was unclear.

Water clarity, as determined using Secchi depth readings, was significantly lower at Mary Lake compared to Reference Lake 3 in fall 2017 (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 moderately (i.e., 5- to 10-fold higher) to highly elevated (i.e., ≥10-fold higher) turbidity and concentrations of total aluminum, manganese, silver, and/or uranium compared to Reference Lake 3 during summer and/or fall sampling in 2017 (Table 5.6; Appendix Tables C.73 and C.74). However, no parameters were above WQG and, with the exception of total silver concentrations during the summer sampling event, concentrations of all parameters were below AEMP benchmarks at the Mary Lake north basin during the winter, summer, and fall monitoring events in 2017 (Table 5.6; Appendix Table C.73). Total silver concentrations were below laboratory reportable detection limits (RDL) at the Mary Lake north basin during winter and fall sampling events (Appendix Table C.73), as well as further upstream closer to the mine at Camp Lake during all winter, summer, and fall sampling events (Appendix Table C.26). In turn, this suggested that elevated total silver concentrations at the Mary Lake north basin in summer 2017 were likely a laboratory analysis anomaly. 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 in 2017, 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 2017 were mostly within respective parameter concentration ranges measured over the mine baseline period (2005-2013; Appendix Table C.74), although annual concentrations of some parameters, including chloride, sodium, and sulphate, showed consistently higher concentrations since the onset of commercial mine operations in 2015 (Figure 5.12; 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 2017 (Table 5.6; Appendix Table C.78), suggesting that the south basin waters were generally well mixed both laterally and vertically. On average, turbidity and total aluminum concentrations were highly elevated, total manganese concentrations moderately elevated, and total iron and lead concentrations slightly elevated (i.e.,

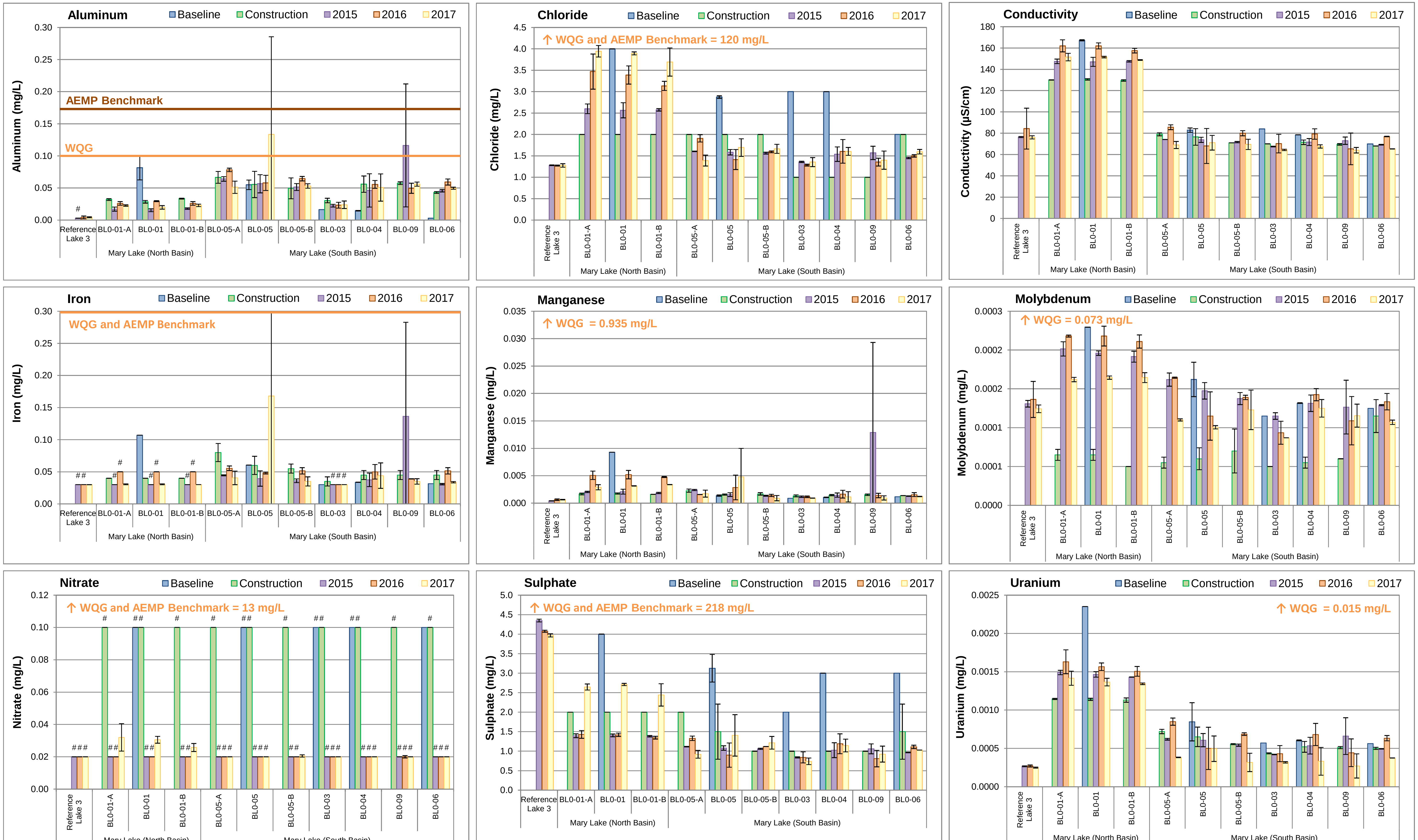


Figure 5.12: Temporal Comparison of Water Chemistry at Mary Lake (BLO) for Mine Baseline (2005 - 2013), Construction (2014), and Operational (2015-2017) Periods during Fall

Note: Values represent mean ± SD. Pound symbol (#) indicates parameter concentration is below the laboratory method detection limit. See Table 2.3 for information regarding Water Quality Guideline (WQG) criteria. AEMP Benchmarks are specific to Mary Lake.

Table 5.6: Water Chemistry at Mary Lake North Basin (BLO-01) and South Basin (BLO) Monitoring Stations[a], Mary River Project CREMP, 2017

	Parameters	Units	Water Quality Guideline (WQG)[b]	AEMP Benchmark[c]	Reference Lake 3 Average (n = 3)	North Basin (Mine-exposed)			South Basin (Mine-exposed)						
						BL0-01-A	BL0-01	BL0-01-B	BL0-05-A	BL0-05	BL0-05-B	BL0-03	BL0-04	BL0-09	BL0-06
					Fall 2017	30-Aug-2017	30-Aug-2017	30-Aug-2017	28-Aug-2017	29-Aug-2017	29-Aug-2017	29-Aug-2017	29-Aug-2017	29-Aug-2017	29-Aug-2017
Conventionals	Conductivity (lab)	umho/cm	-	-	76	152	152	149	69	71	70	64	68	64	65
	pH (lab)	pH	6.5 - 9.0	-	7.76	8.05	8.05	8.06	7.55	7.75	7.74	7.71	7.65	7.67	7.72
	Hardness (as $CaCO_3$)	mg/L	-	-	35	74	74	72	31	32	31	30	31	30	30
	Total Suspended Solids (TSS)	mg/L	-	-	<2.0	<2.0	<2.0	<2.0	<2.0	4.5	<2.0	<2.0	<2.0	<2.0	<2.0
	Total Dissolved Solids (TDS)	mg/L	-	-	33	75	89	70	34	38	35	36	36	39	28
	Turbidity	NTU	-	-	0.7	0.9	0.9	0.9	1.9	6.5	1.6	0.7	1.7	2.1	1.3
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31	71	66	66	31	33	28	29	31	25	28
Nutrients and Organics	Total Ammonia	mg/L	variable	0.855	0.020	<0.020	0.024	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020	<0.020
	Nitrate	mg/L	13	13	<0.020	0.032	0.0305	0.02575	<0.020	<0.020	0.0205	<0.020	<0.020	<0.020	<0.020
	Nitrite	mg/L	0.06	0.06	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.15	<0.15	<0.15	<0.15	0.15	<0.15	0.17	<0.15	<0.15	<0.15	<0.15
	Dissolved Organic Carbon	mg/L	-	-	2.7	1.5	1.7	1.6	1.1	1.2	1.2	1.2	1.2	1.2	1.3
	Total Organic Carbon	mg/L	-	-	2.8	1.7	1.8	1.7	1.2	1.3	1.2	1.3	1.2	1.2	1.2
	Total Phosphorus	mg/L	0.020$^{\alpha}$	-	0.004	0.014	0.004	0.004	0.011	0.008	0.003	0.003	0.005	0.004	0.004
	Phenols	mg/L	0.004$^{\alpha}$	-	0.002	<0.0010	<0.0010	0.001	0.004	<0.0010	<0.0010	0.001	0.001	0.001	<0.0010
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	1.28	3.95	3.90	3.69	1.39	1.70	1.67	1.36	1.61	1.40	1.60
	Sulphate (SO_4)	mg/L	218$^{\beta}$	218	3.97	2.65	2.71	2.44	0.92	1.41	1.22	0.74	1.15	0.93	1.03
Total Metals	Aluminum (Al)	mg/L	0.100	0.13	0.005	0.023	0.020	0.023	0.051	***0.134***	0.053	0.024	0.051	0.056	0.050
	Antimony (Sb)	mg/L	0.020$^{\alpha}$	-	<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	0.0073	0.0073	0.0073	0.0041	0.0055	0.0046	0.0036	0.0045	0.0043	0.0041
	Beryllium (Be)	mg/L	0.011$^{\alpha}$	-	<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	-	-	6.9	15.0	14.5	14.5	6.3	6.6	6.1	6.1	6.2	5.9	6.1
	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$^{\alpha}$	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	0.00079	0.00084	0.00082	0.00085	0.00060	0.00074	0.00054	0.00051	0.00056	0.00053	0.00056
	Iron (Fe)	mg/L	0.30	0.326	<0.030	0.031	0.031	<0.030	0.041	0.168	0.035	<0.030	0.044	0.035	0.034
	Lead (Pb)	mg/L	0.001	0.001	<0.000050	<0.000050	<0.000050	<0.000050	<0.000050	0.000143	<0.000050	<0.000050	0.000055	<0.000050	<0.000050
	Lithium (Li)	mg/L	-	-	<0.0010	0.0011	<0.0010	0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010	<0.0010
	Magnesium (Mg)	mg/L	-	-	4.4	8.9	8.9	8.6	3.8	4.2	4.1	3.7	4.1	3.9	3.7
	Manganese (Mn)	mg/L	0.935$^{\beta}$	-	0.00063	0.00289	0.00313	0.00338	0.00173	0.00478	0.00090	0.00091	0.00115	0.00095	0.00122
	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	-	0.00012	0.00016	0.00016	0.00016	0.00011	0.00010	0.00012	0.00009	0.00013	0.00012	0.00011
	Nickel (Ni)	mg/L	0.025	0.025	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050	0.00057	<0.00050	<0.00050	<0.00050	<0.00050	<0.00050
	Potassium (K)	mg/L	-	-	0.88	0.82	0.81	0.82	0.55	0.63	0.58	0.48	0.58	0.57	0.53
	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	-	-	0.41	0.80	0.80	0.75	0.51	0.60	0.52	0.43	0.53	0.56	0.50
	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	-	-	0.85	1.98	2.02	2.01	0.85	0.90	0.91	0.78	0.90	0.87	0.83
	Strontium (Sr)	mg/L	-	-	0.0078	0.0097	0.0096	0.0098	0.0050	0.0056	0.0051	0.0046	0.0051	0.0049	0.0050
	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	0.0135	<0.010	<0.010	<0.010	<0.010	<0.010
	Uranium (U)	mg/L	0.015	-	0.00025	0.00142	0.00137	0.00134	0.00038	0.00050	0.00032	0.00032	0.00033	0.00027	0.00037
	Vanadium (V)	mg/L	0.006$^{\alpha}$	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.

3- to 5-fold higher) at the Mary Lake south basin compared to Reference Lake 3 during the 2017 summer and/or fall sampling events (Table 5.6; Appendix Tables C.74 and C.78). Of the metals indicated above, total aluminum, manganese, and iron concentrations showed a very strong positive correlation with turbidity for the Mary Lake south basin combined data set (i.e., winter, summer and fall data; $r_s \geq 0.90$; Appendix Table C.80), suggesting that these metals were associated with suspended particles (e.g., aluminosilicates). Indeed, evaluation of the ratios of dissolved to total concentrations of aluminum, manganese, and iron indicated that on average, 85%, 56%, and 73% of these metals were associated with the total fraction, respectively, during summer and fall sampling events. As indicated previously, 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. Aluminum concentrations were occasionally above applicable WQG and/or AEMP benchmarks, and copper, iron, and mercury concentrations were above respective WQG on a single occasion, at the south basin of Mary Lake over the course of the 2017 winter, summer, and fall sampling events (Table 5.6; Appendix Table C.78).

Temporal comparisons of the Mary Lake south basin water chemistry data suggested no changes in average concentrations of mine-related parameters in 2017 compared to the baseline (2005 – 2013) period except for slightly higher turbidity during the summer 2017 sampling event (Figure 5.12; Appendix Figure C.28). Highest turbidity was observed at stations most distant to the inlets of the Tom and Mary rivers to Mary Lake during the summer 2017 sampling event, and therefore the source of turbidity to the Mary Lake south basin was unclear, but did not appear to be related to discharge from the Tom or Mary rivers. Parameter concentrations at the Mary River south basin in fall 2017 did not show any consistent increase compared to the year of mine construction (2014) and the two previous years of mine operation (2015 and 2016; Figure 5.12; Appendix Figure C.28). The absence of any 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.

5.3.3 Sediment Quality

Surficial sediment of the Mary Lake north basin (BLO-01) was composed of silt loam material with low TOC content (Figure 5.13). At the Mary Lake south basin littoral stations, surficial sediment varied from loamy sand to silty clay loam (Figure 5.13; Appendix Table D.45), whereas at the south basin profundal stations, surficial sediment was predominantly silt and clay loams except at two stations where sand was more prevalent (Figure 5.13). Sediment TOC content was significantly lower at littoral and profundal stations of Mary Lake compared to respective habitat types at the reference lake (Appendix Table D.5). Substrate containing visible iron (oxy)hydroxide

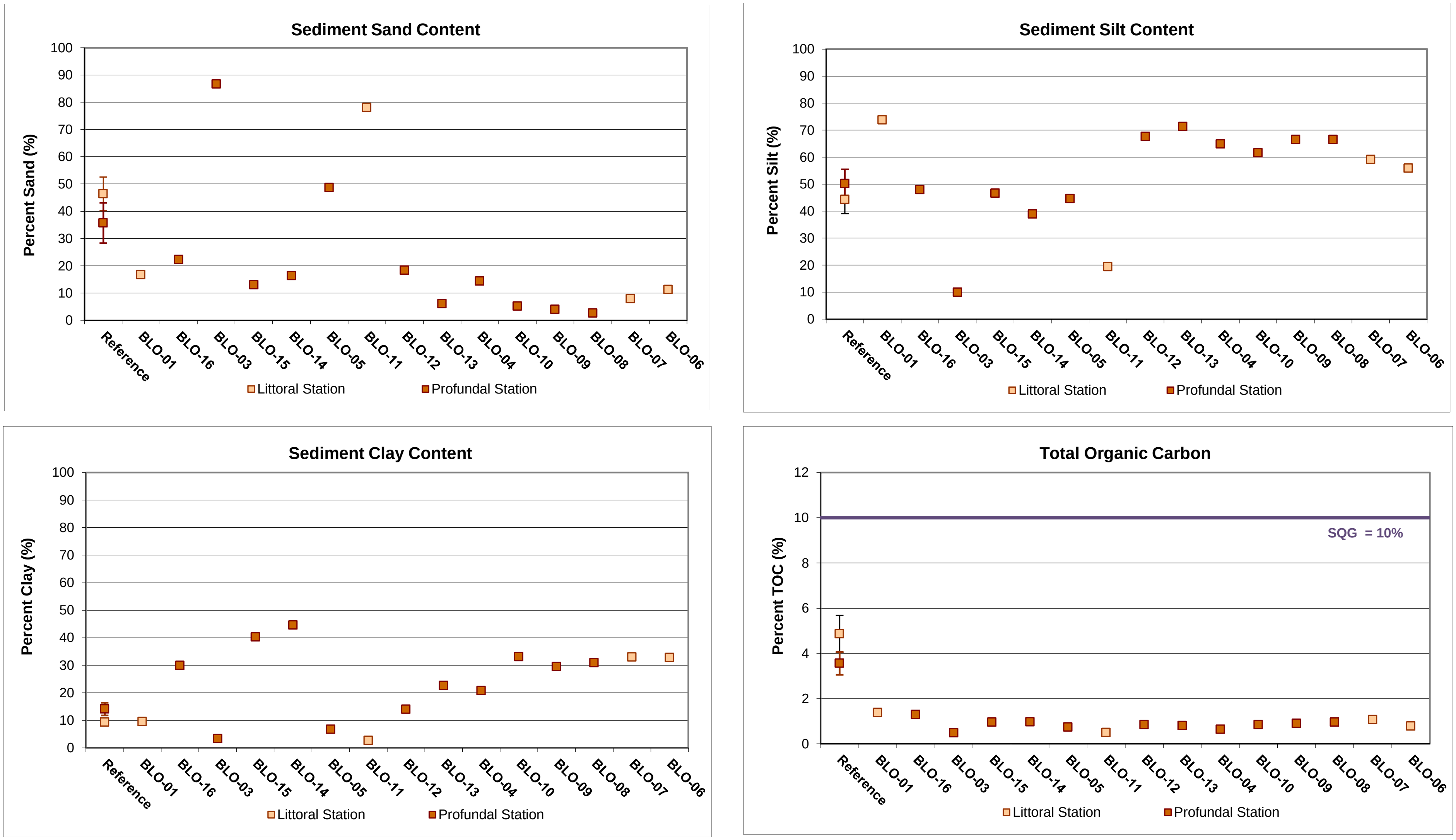


Figure 5.13: **Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Mary Lake (BLO) North and South Basin Sediment Monitoring Stations and to Reference Lake 3 Averages (mean ± SE), Mary River Project CREMP, August 2017**

material was not observed at the Mary Lake north basin, but was present at some south basin stations in 2017 (Appendix Table D.43), 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 occasionally contained sub-surface blackening/dark colouration which appeared as bands/layers indicating the presence of reduced sediment demarcated by distinct redox boundaries in some cases (Appendix Table D.43). Similar sub-surface reducing conditions were observed in sediment of the reference lake, though no distinct redox boundaries were visible (Appendix Table D.3).

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 (Table 5.7; Appendix Table D.46). 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, suggesting that the Mary River was not contributing disproportionate concentrations of metals (Appendix Table D.45). In addition, sediment metal concentrations at the Mary Lake south basin profundal stations were similar to average metal concentrations at like-depth stations of the reference lake (Table 5.7; Appendix Table D.46). Although manganese concentrations were above SQG at the Mary Lake north basin littoral station, and chromium, iron, and manganese concentrations were above respective SQG at some individual profundal stations of the Mary Lake south basin, average concentrations of these metals were below the applicable guidelines at Mary Lake (Table 5.7; Appendix Table D.45). As indicated previously, concentrations of iron and manganese were elevated above SQG in sediment at Reference Lake 3 littoral and/or profundal stations, suggesting that concentrations of these metals above SQG at Mary Lake may reflect natural conditions un-related to mine activity. On average, no metals were observed at concentrations above the sediment AEMP benchmarks at littoral and profundal stations of the Mary Lake north or south basins (Table 5.7; Appendix Table D.45).

Temporal comparisons indicated that metal concentrations in sediment at littoral and profundal stations of Mary Lake in 2017 closely mirrored those observed during the mine baseline (2005 – 2013) period (Figure 5.14; Appendix Table D.46). On average, metal concentrations in sediment at Mary Lake littoral and profundal stations in 2017 were also within the range of those observed from 2015 and 2016, with no consistently higher concentrations indicated that would suggest an increasing trend over time (Figure 5.14; Appendix Table D.47). Accordingly, evaluation of data current to 2017 indicated no substantial changes in sediment metal concentrations at Mary Lake littoral and profundal stations following the commencement of Baffinland commercial mine operations in 2015.

Table 5.7: 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 2017

Parameter		Units	Sediment Quality Guideline (SQG)[a]	AEMP Benchmark[b]	Littoral			Profundal	
					Reference Lake (n = 5)	Mary Lake (North Basin) (n = 1)	Mary Lake (South Basin) (n = 1)	Reference Lake (n = 5)	Mary Lake (South Basin) (n = 8)
					Average ± Std. Error			Average ± Std. Error	Average ± Std. Error
Total Organic Carbon		%	10[α]	-	4.87 ± 0.81	1.38	0.78	3.60 ± 0.52	0.86 ± 0.06
Metals	Aluminum (Al)	mg/kg	-	-	14,720 ± 736	14,900	21,200	22,140 ± 1,652	21,334 ± 2,557
	Antimony (Sb)	mg/kg	-	-	<0.10 ± 0	<0.10	<0.10	<0.10 ± 0.00	<0.10 ± 0.00
	Arsenic (As)	mg/kg	17	5.9	3.35 ± 0.64	4.48	2.77	4.80 ± 0.20	3.26 ± 0.47
	Barium (Ba)	mg/kg	-	-	113 ± 10	85	88	138 ± 7	91 ± 11
	Beryllium (Be)	mg/kg	-	-	0.57 ± 0.02	0.77	1.01	0.85 ± 0.06	0.99 ± 0.12
	Bismuth (Bi)	mg/kg	-	-	<0.20 ± 0	<0.20	<0.20	0.20 ± 0.000	0.23 ± 0.01
	Boron (B)	mg/kg	-	-	12.1 ± 0.8	19.4	27.9	16.6 ± 1.5	30.6 ± 3.5
	Cadmium (Cd)	mg/kg	3.5	1.5	0.182 ± 0.032	0.112	0.098	0.179 ± 0.024	0.142 ± 0.017
	Calcium (Ca)	mg/kg	-	-	4,656 ± 362	8,550	3,850	5,262 ± 377	3,987 ± 428
	Chromium (Cr)	mg/kg	90	98	51.1 ± 3.7	65.4	74.8	70.2 ± 6.3	78.4 ± 9.1
	Cobalt (Co)	mg/kg	-	-	10.63 ± 0.79	14.20	14.50	15.98 ± 1.14	14.92 ± 1.65
	Copper (Cu)	mg/kg	110	50	***64.7*** ± 5.3	29.1	28.2	***87.6*** ± 8.5	29.8 ± 3.6
	Iron (Fe)	mg/kg	40,000[α]	52,400	41,960 ± 8,713	34,800	35,400	46,740 ± 3,447	37,400 ± 3,801
	Lead (Pb)	mg/kg	91.3	35	12.4 ± 0.6	14.6	18.3	16.9 ± 1.0	19.6 ± 2.4
	Lithium (Li)	mg/kg	-	-	24.0 ± 0.9	29.2	37.9	35.0 ± 2.1	37.9 ± 4.5
	Magnesium (Mg)	mg/kg	-	-	10,256 ± 714	15,000	13,700	14,660 ± 1,126	14,580 ± 1,687
	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	639 ± 115	1,270	554	1,266 ± 70	996 ± 171
	Mercury (Hg)	mg/kg	0.486	0.17	0.0440 ± 0.0081	0.0328	0.0218	0.0528 ± 0.0089	0.0541 ± 0.0070
	Molybdenum (Mo)	mg/kg	-	-	3.50 ± 0.97	0.71	0.55	2.90 ± 0.20	0.78 ± 0.10
	Nickel (Ni)	mg/kg	75[α,β]	72	38.2 ± 2.4	54.5	49.8	49.3 ± 3.0	54.8 ± 6.3
	Phosphorus (P)	mg/kg	2,000[α]	1,580	1,039 ± 246	1,060	679	1,073 ± 54	808 ± 86
	Potassium (K)	mg/kg	-	-	3,754 ± 212	3,640	5,530	5,694 ± 366	5,632 ± 678
	Selenium (Se)	mg/kg	-	-	0.61 ± 0.12	<0.20	0.20	0.59 ± 0.12	0.24 ± 0.02
	Silver (Ag)	mg/kg	-	-	0.13 ± 0.01	<0.10	0.10	0.20 ± 0.03	0.14 ± 0.01
	Sodium (Na)	mg/kg	-	-	284.2 ± 21	258	323	410 ± 36	360 ± 43
	Strontium (Sr)	mg/kg	-	-	10.0 ± 0.5	11.5	11.9	12.7 ± 0.8	13.2 ± 1.5
	Thallium (Tl)	mg/kg	-	-	0.346 ± 0.036	0.325	0.470	0.657 ± 0.038	0.459 ± 0.054
	Uranium (U)	mg/kg	-	-	11.0 ± 1.04	3.9	5.10	23.5 ± 2.26	7.27 ± 0.80
	Vanadium (V)	mg/kg	-	-	47.8 ± 2.0	50.1	60.2	67.2 ± 3.9	60.2 ± 6.8
	Zinc (Zn)	mg/kg	315	135	67.3 ± 2.5	52.1	66.4	91 ± 6.8	68.0 ± 7.8

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

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

Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

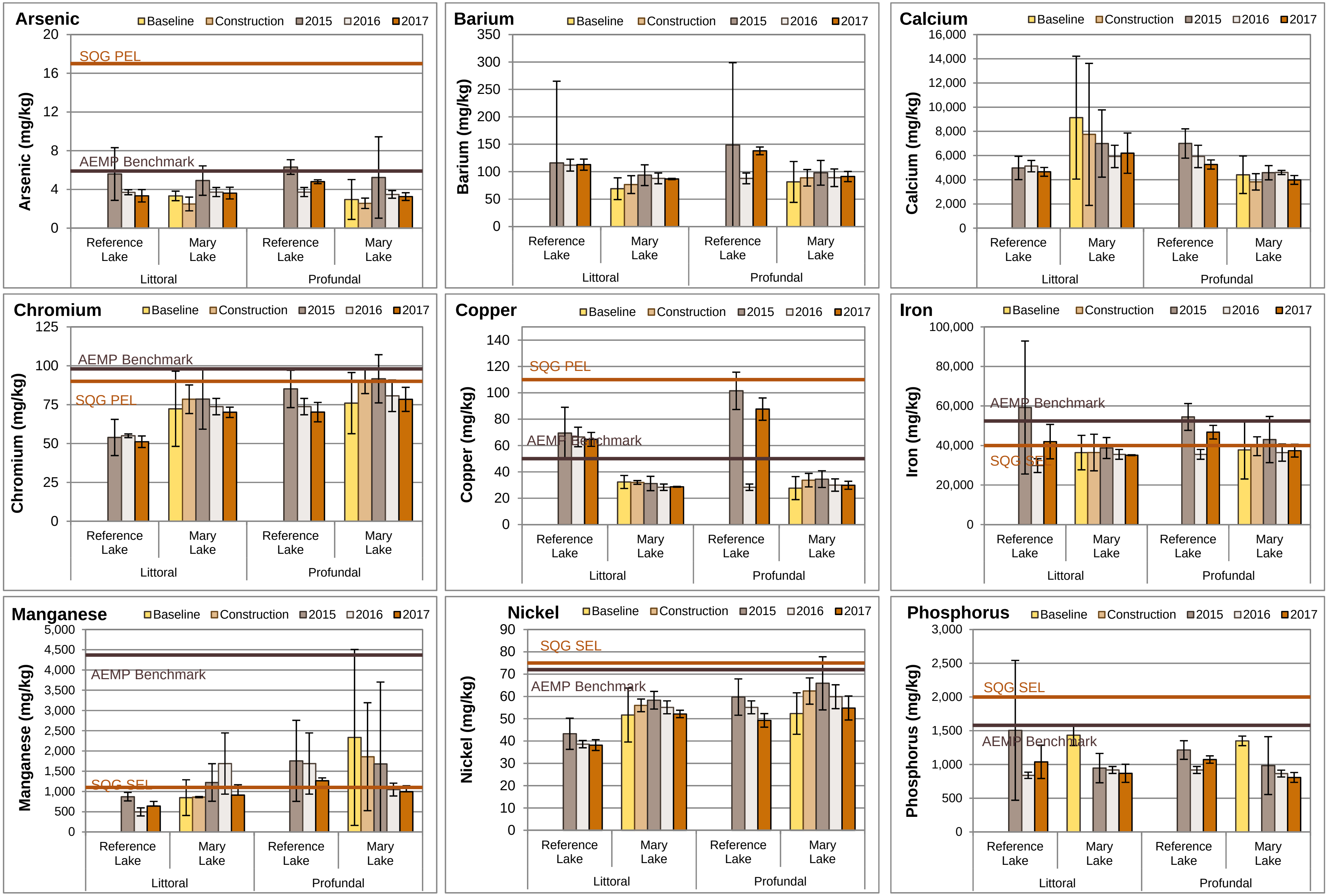


Figure 5.14: Temporal Comparison of Sediment Metal Concentrations (mean ± SD) at Littoral and Profundal Stations of Mary Lake and Reference Lake 3 for Mine Baseline (2005 - 2013), Construction (2014) and Operational (2015, 2016, and 2017) Periods, Mary River Project CREMP, 2017

5.3.4 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 2017 (Figure 5.15). Chlorophyll-a concentrations were consistently lowest in winter and highest in fall at both the north and south basins of Mary Lake in 2017 (Appendix Table E.6), and mirrored similar relative differences in chlorophyll-a concentrations between summer and fall sampling events at the reference lake (Appendix Table B.8). Similar to the other mine-exposed lakes, chlorophyll-a concentrations were significantly higher at the Mary Lake north and south basins than at the reference lake in summer, as well as at the south basin during the fall sampling event (Appendix Tables E.7 and E.8). However, Mary Lake chlorophyll-a concentrations were well below the AEMP benchmark of 3.7 µg/L during all winter, summer, and fall sampling events in 2017 (Figure 5.15). Chlorophyll-a concentrations at Mary Lake reflected an 'oligotrophic' primary productivity categorization (sensu Wetzel 2001), which agreed closely with an 'oligotrophic' CWQG categorization based on mean aqueous total phosphorus concentrations generally between 4 – 10 µg/L during the 2017 Mary Lake winter, summer, and fall sampling events (Table 5.6; Appendix Tables C.73 and C.78).

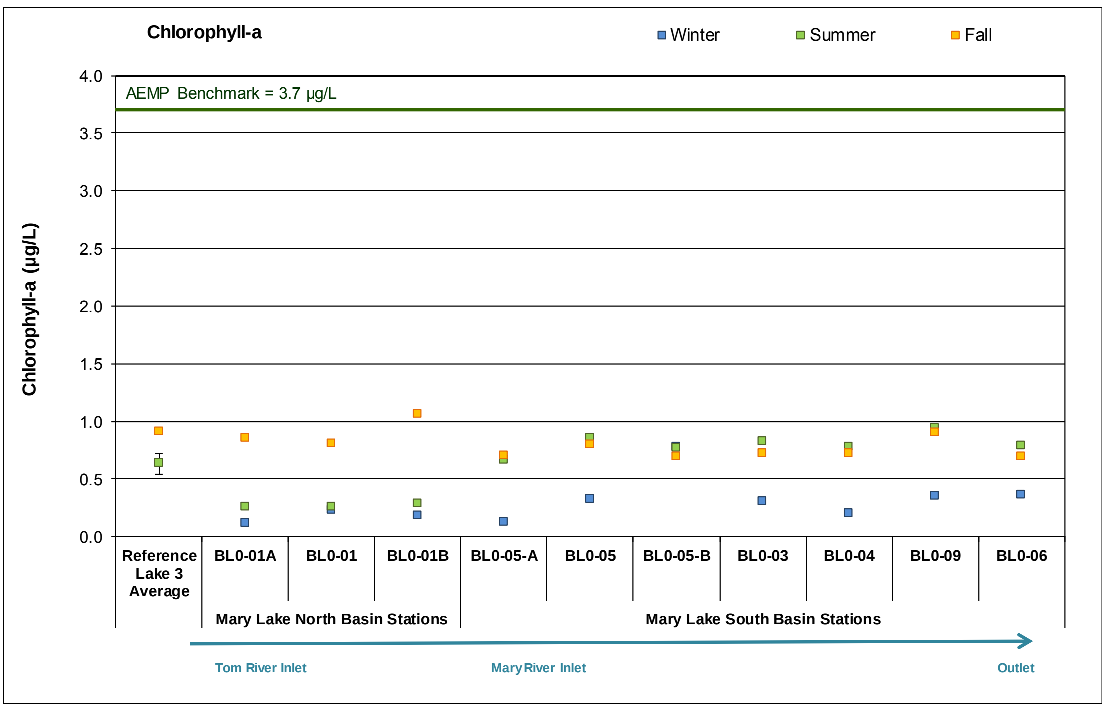


Figure 5.15: Chlorophyll-a Concentrations at Mary Lake (BLO) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2017

Notes: Values presented are averages of samples taken from the surface and the bottom of the water column at each station. Reference values represent mean ± standard deviation (n = 3). Reference Lake 3 was not sampled in winter 2017.

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 2017 data and data from the mine construction (2014) period or previous years of mine operation (2015, 2016) among the winter, summer, or fall sampling events (Figure 5.16; Appendix Tables E.16 and E.17). In addition, annual average chlorophyll-a concentrations did not differ significantly among years from 2014 – 2017 at either basin of Mary Lake (Appendix Tables E.16 and E.17), suggesting no substantial changes in the trophic status since mine operations commenced at the Mary River Project. No chlorophyll-a baseline (2005 – 2013) data are available for Mary Lake, precluding comparisons to conditions prior to the mine construction period.

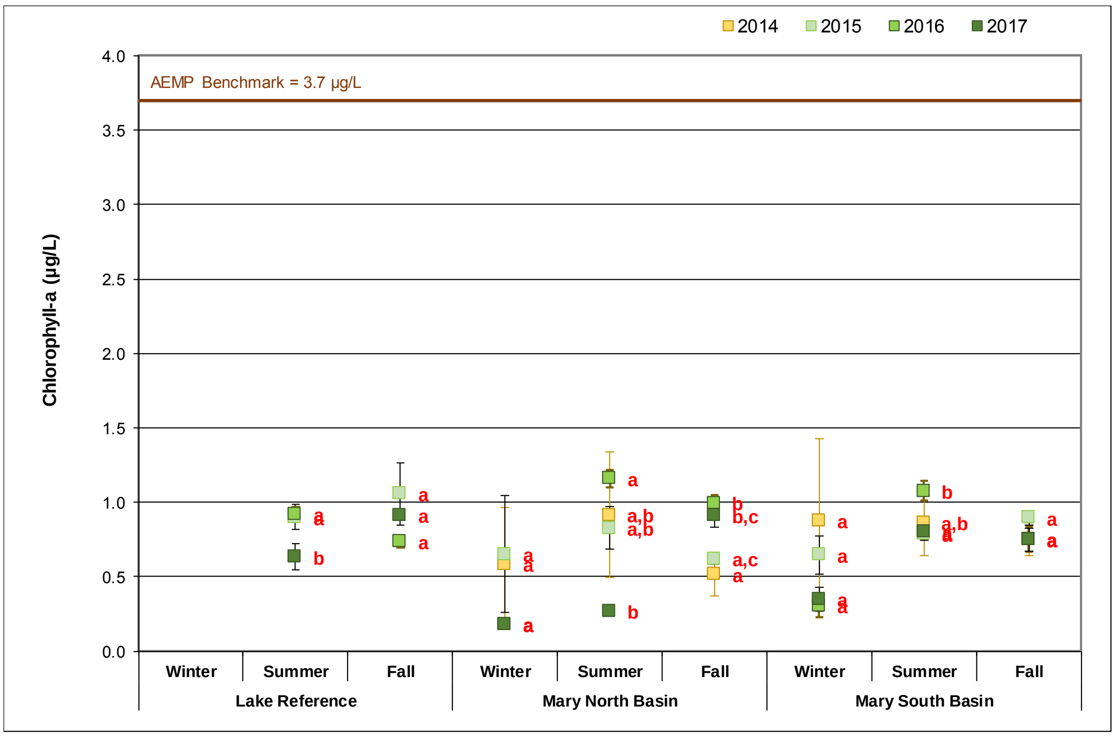


Figure 5.16: Chlorophyll-a Concentration Seasonal Comparison among 2014, 2015, 2016, and 2017 years (mean ± SE) at Mary Lake Phytoplankton Monitoring Stations, Mary River Project CREMP

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

5.3.5 Benthic Invertebrate Community

Benthic invertebrate density, richness, and Simpson's Evenness did not differ significantly between Mary Lake and Reference Lake 3 at littoral stations, but at profundal stations, both density and richness were significantly higher at Mary Lake in 2017 (Tables 5.8 and 5.9). Benthic invertebrate community structure differed between Mary Lake and Reference Lake 3 based on significant differences in Bray-Curtis Index for both littoral and profundal habitat types (Tables 5.8

Table 5.8: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BLO) and Reference Lake 3 for Littoral Habitat Stations, Mary River Project CREMP, August 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Littoral Habitat	Mean	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	none	NO	0.745	t-test (unequal variance)	0.4	Reference Lake 3	1,489	850	380	372	2,618
						Mary Lake Littoral	1,839	1,853	926	216	3,911
Richness (Number of Taxa)	log	NO	0.102	ANOVA	-1.2	Reference Lake 3	12.4	2.5	1.1	10.0	15.0
						Mary Lake Littoral	9.5	2.1	1.0	7.0	12.0
Simpson's Evenness (E)	none	NO	0.905	ANOVA	0.1	Reference Lake 3	0.807	0.142	0.063	0.605	0.934
						Mary Lake Littoral	0.818	0.110	0.055	0.717	0.934
Bray-Curtis Index	none	YES	< 0.001	ANOVA	3.7	Reference Lake 3	0.384	0.120	0.054	0.203	0.536
						Mary Lake Littoral	0.828	0.040	0.020	0.768	0.853
Nemata (%)	fourth root	NO	0.219	ANOVA	-0.1	Reference Lake 3	3.9	3.3	1.5	0.8	9.1
						Mary Lake Littoral	3.5	6.2	3.1	0.0	12.8
Hydracarina (%)	none	NO	0.509	ANOVA	0.8	Reference Lake 3	5.3	3.0	1.4	1.8	8.1
						Mary Lake Littoral	7.8	7.2	3.6	0.0	15.5
Ostracoda (%)	log(X+1)	YES	< 0.001	ANOVA	-2.0	Reference Lake 3	38.8	18.4	8.2	22.3	63.9
						Mary Lake Littoral	2.1	2.2	1.1	0.0	5.1
Chironomidae (%)	log	YES	0.033	ANOVA	1.9	Reference Lake 3	51.8	17.9	8.0	29.5	67.9
						Mary Lake Littoral	85.7	13.1	6.6	66.6	94.3
Metal-Sensitive Chironomidae (%)	square root	NO	0.361	ANOVA	0.4	Reference Lake 3	15.5	13.4	6.0	0.5	37.0
						Mary Lake Littoral	21.3	7.7	3.8	15.5	32.2
Collector-Gatherers (%)	none	NO	0.143	ANOVA	-1.4	Reference Lake 3	73.9	16.0	7.2	56.0	95.5
						Mary Lake Littoral	52.2	23.6	11.8	20.7	74.3
Filterers (%)	fourth root	NO	0.415	t-test (unequal variance)	-0.1	Reference Lake 3	14.7	13.3	5.9	0.5	36.4
						Mary Lake Littoral	13.3	16.0	8.0	0.0	32.2
Shredders (%)	fourth root	NO	0.197	ANOVA	-0.6	Reference Lake 3	4.2	6.6	3.0	0.0	15.9
						Mary Lake Littoral	0.4	0.6	0.3	0.0	1.2
Clingers (%)	square root	NO	0.636	ANOVA	0.3	Reference Lake 3	19.8	12.8	5.7	2.7	38.1
						Mary Lake Littoral	23.5	13.3	6.6	12.0	41.7
Sprawlers (%)	none	NO	0.231	ANOVA	-1.5	Reference Lake 3	72.6	13.9	6.2	55.7	92.9
						Mary Lake Littoral	52.3	31.4	15.7	5.9	71.8
Burrowers (%)	log	YES	0.019	ANOVA	5.5	Reference Lake 3	7.5	3.0	1.4	4.5	12.0
						Mary Lake Littoral	24.2	18.8	9.4	12.8	52.4

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

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

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

Table 5.9: Benthic Invertebrate Community Statistical Comparison Results between Mary Lake (BLO) and Reference Lake 3 for Profundal Habitat Stations, Mary River Project CREMP, August 2017

Metric	Statistical Test Results					Summary Statistics					
	Data Transform-ation	Significant Difference Between Areas?	p-value	Statistical Analysis[a]	Magnitude of Difference [a] (No. of SD)	Study Lake Profundal Habitat	Mean	Standard Deviation	Standard Error	Minimum	Maximum
Density (Individuals/m^2)	log	YES	0.020	ANOVA	12.2	Reference Lake 3	149	32	14	113	190
						Mary Lake Profundal	536	497	203	156	1,515
Richness (Number of Taxa)	fourth root	YES	0.017	ANOVA	1.9	Reference Lake 3	4.2	1.5	0.7	2.0	6.0
						Mary Lake Profundal	7.0	1.5	0.6	5.0	9.0
Simpson's Evenness (E)	none	NO	0.245	t-test (unequal variance)	-1.0	Reference Lake 3	0.704	0.105	0.047	0.597	0.843
						Mary Lake Profundal	0.604	0.236	0.096	0.394	0.962
Bray-Curtis Index	log	YES	0.001	ANOVA	3.7	Reference Lake 3	0.239	0.114	0.051	0.111	0.376
						Mary Lake Profundal	0.665	0.165	0.067	0.471	0.958
Nemata (%)	rank	YES	0.022	Mann-Whitney	nc	Reference Lake 3	0.0	0.0	0.0	0.0	0.0
						Mary Lake Profundal	2.4	1.9	0.8	0.0	5.8
Hydracarina (%)	log(X+1)	NO	0.860	ANOVA	0.2	Reference Lake 3	8.2	5.5	2.5	0.0	15.0
						Mary Lake Profundal	9.5	11.8	4.8	2.2	33.3
Ostracoda (%)	rank	NO	0.855	Mann-Whitney	0.1	Reference Lake 3	2.8	4.1	1.8	0.0	8.9
						Mary Lake Profundal	3.2	6.2	2.5	0.0	15.4
Chironomidae (%)	none	NO	0.568	ANOVA	-0.5	Reference Lake 3	89.0	7.6	3.4	81.6	100.0
						Mary Lake Profundal	84.9	13.8	5.7	60.9	96.3
Metal-Sensitive Chironomidae (%)	none	NO	0.131	ANOVA	-0.7	Reference Lake 3	12.0	8.9	4.0	0.0	20.2
						Mary Lake Profundal	5.6	3.2	1.3	1.8	10.7
Collector-Gatherers (%)	rank	NO	0.584	Mann-Whitney	-0.9	Reference Lake 3	84.3	4.1	1.9	79.8	90.5
						Mary Lake Profundal	80.7	18.2	7.4	44.2	92.0
Filterers (%)	fourth root	NO	0.471	ANOVA	-0.3	Reference Lake 3	6.5	9.3	4.2	0.0	20.2
						Mary Lake Profundal	3.8	2.7	1.1	0.0	6.8
Clingers (%)	fourth root	NO	0.870	ANOVA	0.4	Reference Lake 3	14.6	4.9	2.2	9.5	20.2
						Mary Lake Profundal	16.7	14.3	5.8	3.7	39.1
Sprawlers (%)	rank	NO	0.855	Mann-Whitney	-1.9	Reference Lake 3	81.4	6.3	2.8	74.8	90.5
						Mary Lake Profundal	69.1	34.4	14.1	4.5	91.4
Burrowers (%)	fourth root	NO	0.188	ANOVA	2.8	Reference Lake 3	3.9	3.7	1.7	0.0	8.0
						Mary Lake Profundal	14.2	25.5	10.4	1.8	66.0

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

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

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

and 5.9). However, Ostracoda (seed shrimp) was the only dominant taxonomic group that differed significantly in relative abundance between Mary Lake and the reference lake at a magnitude outside of the CES_{BIC} of ±2 SD_{REF}, but only at the littoral sampling depth. As indicated previously, because TOC serves as a key food source for Ostracoda, lower relative abundance of this taxonomic group at Mary Lake littoral habitat was consistent with significantly lower sediment TOC at Mary Lake compared to the reference lake (Appendix Table F.64). No 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 stations (Tables 5.7 and 5.8), suggesting no sediment metal-related influences on the benthic invertebrate community of Mary Lake. In addition, Mary Lake profundal stations exhibited no significant differences in the relative abundance of dominant taxonomic groups, FFG, or HPG compared to like-habitat at Reference Lake 3 (Table 5.8). Overall, no adverse mine-related influence to the littoral or profundal benthic invertebrate community of Mary Lake was indicated relative to reference lake conditions in 2017.

Temporal comparisons did not indicate any consistent ecologically significant differences in benthic invertebrate community density, richness, or Simpson's Evenness at littoral and profundal habitats of Mary Lake between the mine 2007 baseline study and individual years since the commencement of commercial mine operation (2015, 2016, 2017; Figure 5.17; Appendix Tables F.66 and F.67). 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 with the exception of consistently lower relative abundance of Chironomidae (non-biting midges) at profundal stations in 2015 and 2017 compared to the 2007 baseline study (Appendix Table F.67). Adverse responses of benthic invertebrate communities to anthropogenic industrial inputs normally result in increased relative abundance of chironomids (Taylor and Bailey 1997; Barbour et al. 1999). Therefore, a lower relative abundance of this group at Mary Lake profundal stations in 2015 and 2017 was not consistent with a biological response typical of exposure to mine-related inputs to aquatic systems. Moreover, no significant differences in the relative abundance of metal-sensitive chironomids were indicated between the mine baseline and mine operational years at Mary Lake (Figure 5.17). 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 Mary Lake following the commencement of commercial mine operation in 2015.

5.3.6 Fish Population

5.3.6.1 Mary Lake (South) Fish Community

Arctic charr and ninespine stickleback composed the fish community of Mary Lake in 2017, reflecting the same fish species composition as Reference Lake 3 (Table 5.10). Similar to the

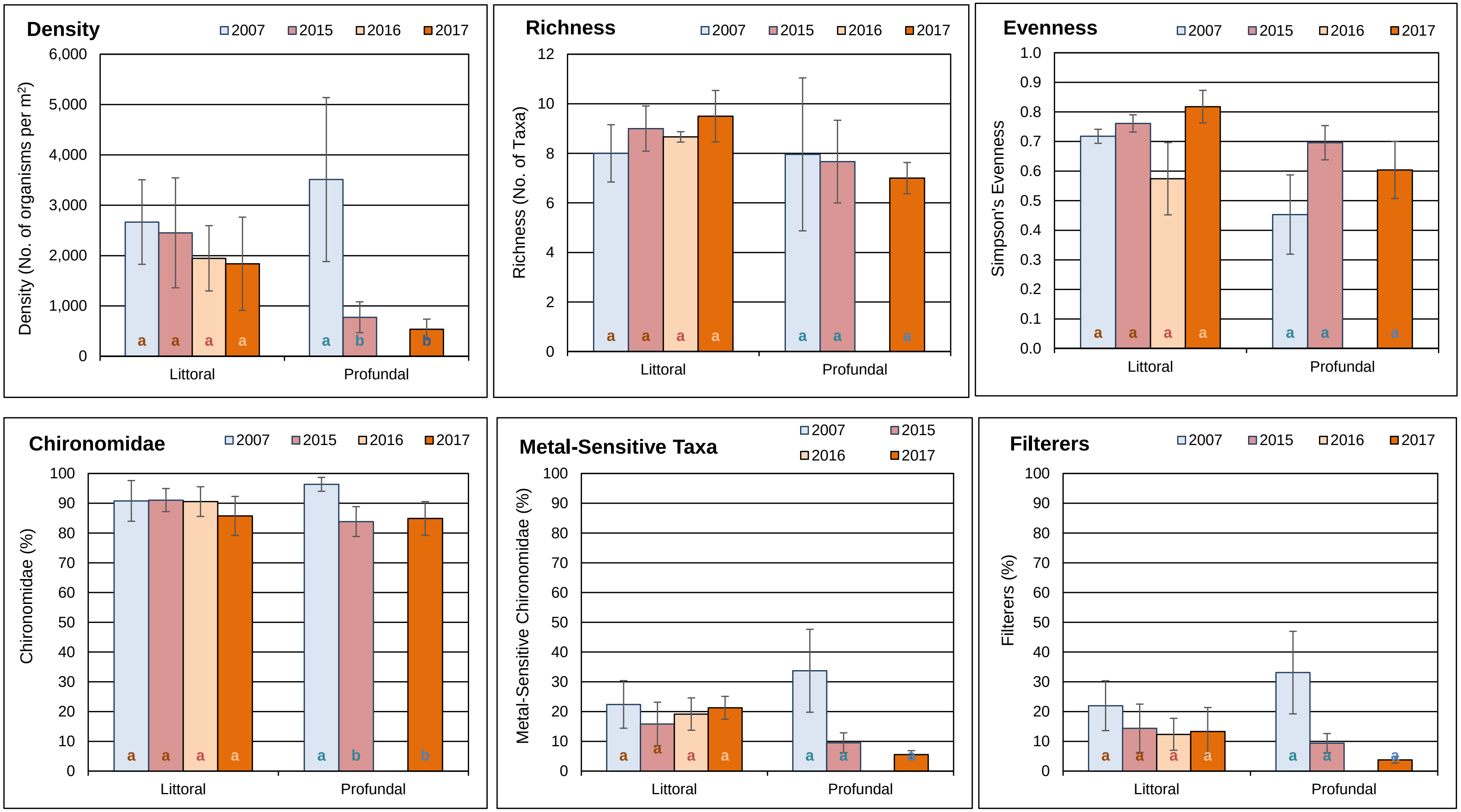


Figure 5.17: Comparison of Key Benthic Invertebrate Community Metrics (mean ± SE) at Mary Lake Littoral and Profundal Study Areas among Mine Baseline (2007) and Operational (2015, 2016, 2017) Periods

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

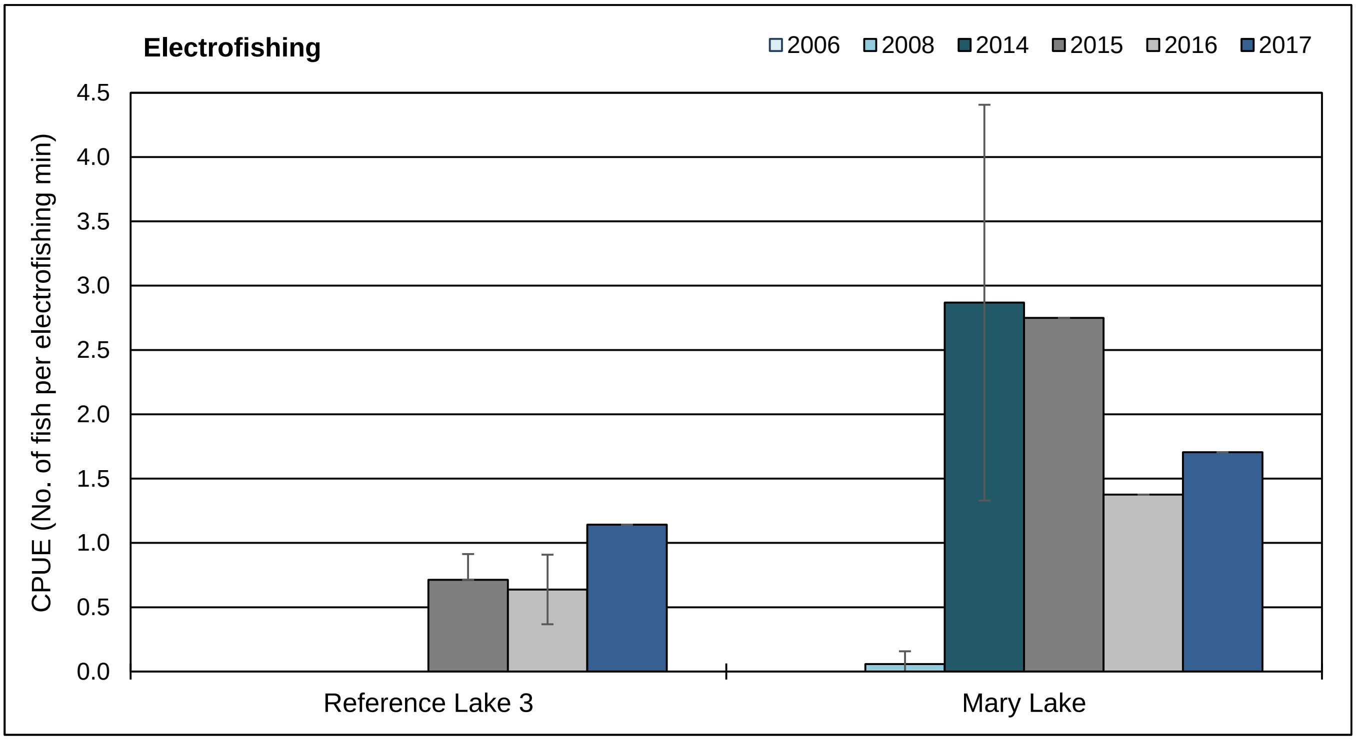
Table 5.10: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Mary Lake (BLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2017

Lake	Method [a]		Arctic Charr	Nine-spine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	100	11	111	2
		CPUE	1.03	0.11	1.14	
	Gill netting	No. Caught	1	0	1	
		CPUE	0.02	0	0.02	
Mary Lake	Electrofishing	No. Caught	98	20	118	2
		CPUE	1.42	0.29	1.71	
	Gill netting	No. Caught	93	0	93	
		CPUE	3.41	0	3.41	

[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.

other mine-exposed lakes, arctic charr CPUE was much higher at Mary Lake than at the reference lake for electrofishing and gill netting collection methods in 2017, 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 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 2017, as well as in other mine construction/operation years, than during baseline monitoring conducted in 2008 (Figure 5.18). Similar to other mine-exposed lakes, gill netting CPUE for arctic charr was higher in 2016 and 2017 compared to all previous baseline (2007 – 2008), mine construction (2014) and mine operational (2015) studies (Figure 5.18), likely reflecting greater sampling efficiencies in 2016 and 2017 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.



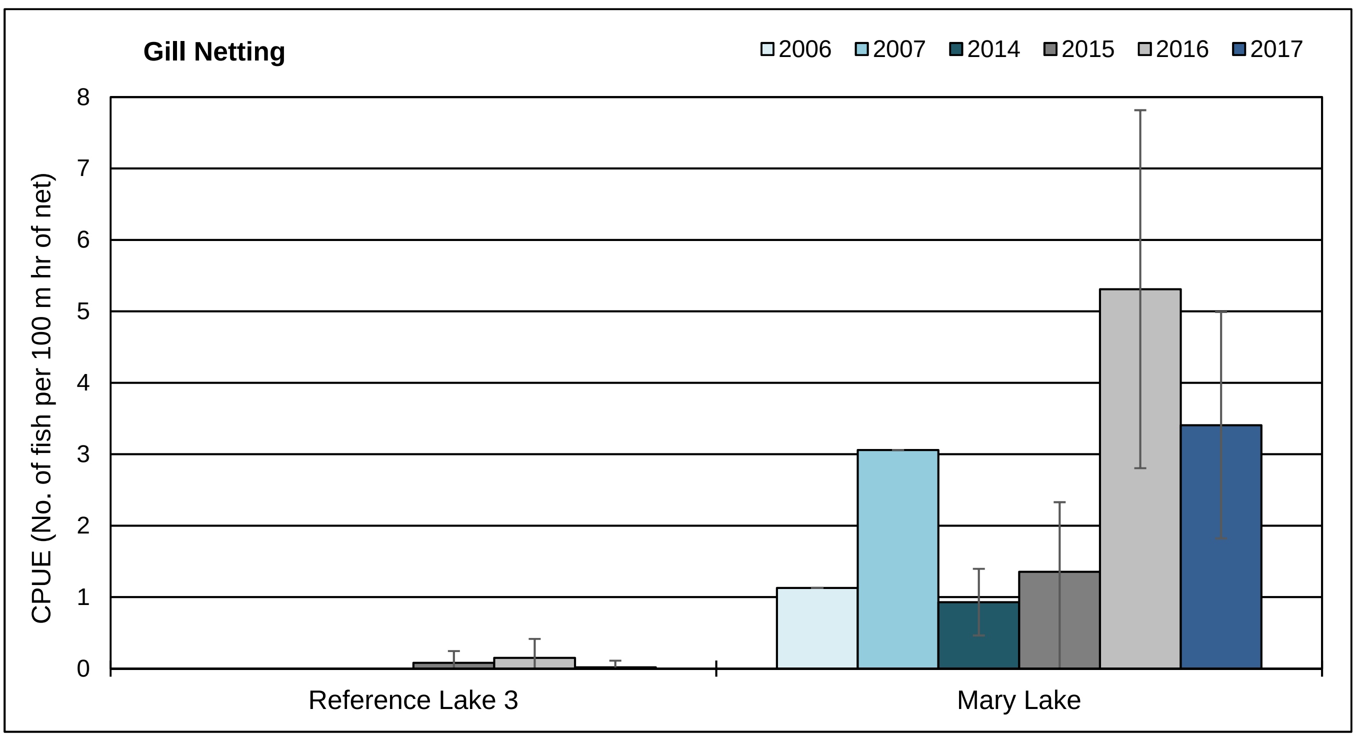


Figure 5.18: Catch-per-unit-effort (CPUE; mean ± SD) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Mary Lake (BLO), Mary River Project CREMP, 2006 - 2017

5.3.6.2 Mary Lake (South) Fish Population Assessment

Nearshore Arctic Charr

Mine-related influences on the Mary Lake nearshore arctic charr population were assessed with a control-impact analysis using data collected from Mary Lake and Reference Lake 3 in 2017. No nearshore arctic charr baseline data were collected at Mary Lake, precluding data analysis using a before-after design. A total of 98 and 100 arctic charr were captured at nearshore habitat of Mary Lake and Reference Lake 3, respectively, in August 2017, for the control-impact analysis. Arctic charr YOY were distinguished from the older, non-YOY age class using a fork length cut-off of 5.0 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 5.19). Nearshore arctic charr health comparisons were conducted separately for the YOY and non-YOY data sets to account for naturally differing weight-at-length relationships that occur between these age categories.

Nearshore arctic charr length-frequency distributions differed significantly between Mary Lake and Reference Lake 3, reflecting the occurrence of larger YOY and greater numbers of larger non-YOY individuals at Mary Lake (Table 5.11; Figure 5.19; Appendix Table G.34). Arctic charr in YOY and non-YOY age classes were significantly longer and heavier at Mary Lake than at the reference lake (Table 5.11; Appendix Table G.34). The occurrence of significantly larger YOY suggested faster arctic charr growth at Mary Lake than at the reference lake in 2017. Although condition of nearshore arctic charr YOY was significantly lower at Mary Lake than at the reference lake, the condition of arctic charr non-YOY did not differ significantly between lakes in 2017 (Table 5.11; Appendix Table G.34). Notably, the magnitude of difference in YOY condition between lakes was within the CES_C of ±10% suggesting that this difference was not ecologically meaningful (Table 5.11). Arctic charr non-YOY at Mary Lake showed no substantial, ecologically meaningful, changes in condition relative to those at Reference Lake 3 over the past three years of mine operation (Table 5.11), indicating no adverse response to fish at Mary Lake nearshore areas since the commencement of commercial mine operations.

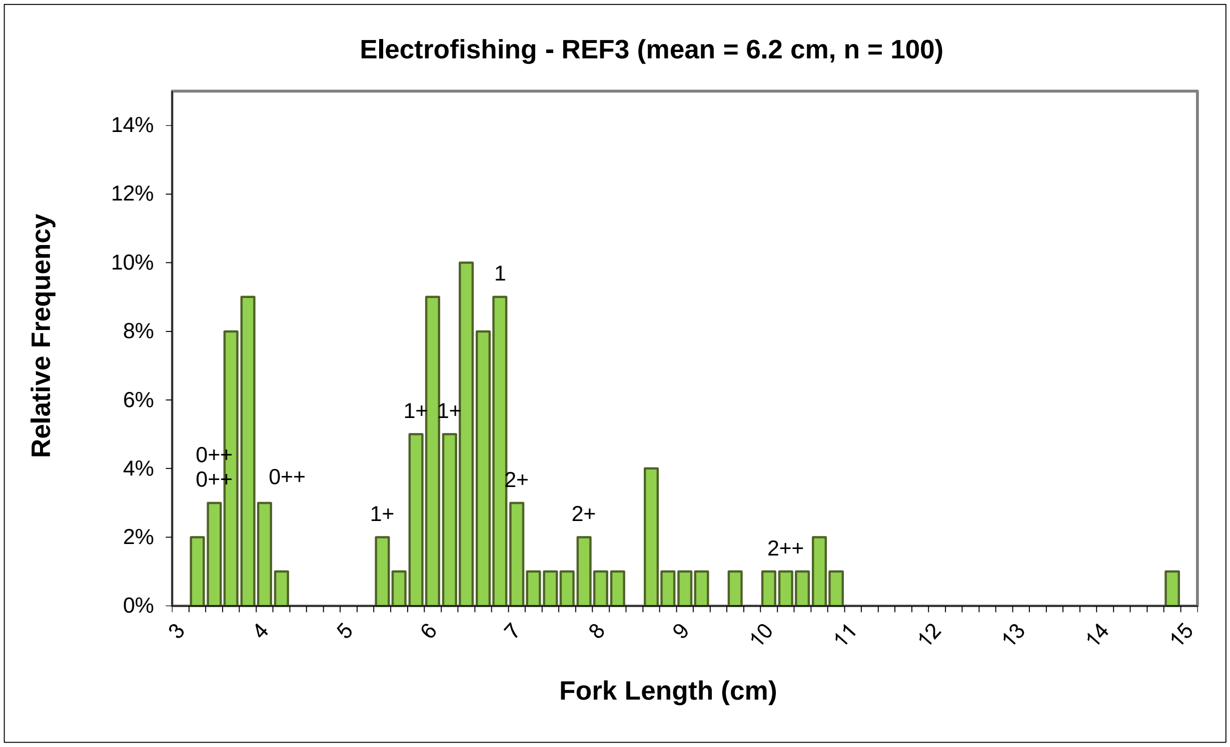
Littoral/Profundal Arctic Charr

Mine-related influences on the Mary Lake littoral/profundal arctic charr population were assessed with a before-after analysis using data collected from Mary Lake in 2017 and during 2006-2007 baseline monitoring. Similar to the two previous CREMP studies, a small sample size from Reference Lake 3 (i.e., n = 1) precluded conducting a control-impact statistical analysis using the 2017 data for arctic charr of spawning size. Biological information collected from arctic charr mortalities indicated that non-spawners of reproductive age constituted approximately 62% of the Mary Lake sampled arctic charr population during the August 2017 field study (Appendix

Table 5.11: Summary of Statistical Results for Arctic Charr Population Comparisons between Mary Lake and Reference Lake 3 in 2015, 2016, and 2017, 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	2015	2016	2017
Electrofishing Samples	Survival	Length-Frequency Distribution	No	**Yes**	**Yes**	-	-	-
		Age	**Yes (-43%)**	No	No	-	-	-
	Energy Use (non-YOY)	Size (mean weight)	No	No	**Yes (+51%)**	-	-	-
		Size (mean fork length)	No	No	**Yes (+17%)**	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	No	-	-	-
Gill Netting Samples [c]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**
		Age	-	-	-	No	**Yes (-14%)**	No
	Energy Use	Size (mean weight)	-	-	-	**Yes (+19%)**	No	**Yes (-9%)**
		Size (mean fork length)	-	-	-	**Yes (+6%)**	No	**Yes (-5%)**
		Growth (weight-at-age)	-	-	-	No	**Yes (nc)**	No
		Growth (fork length-at-age)	-	-	-	No	**Yes (nc)**	No
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	No	**Yes (+3%)**	**Yes (+5%)**

[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.



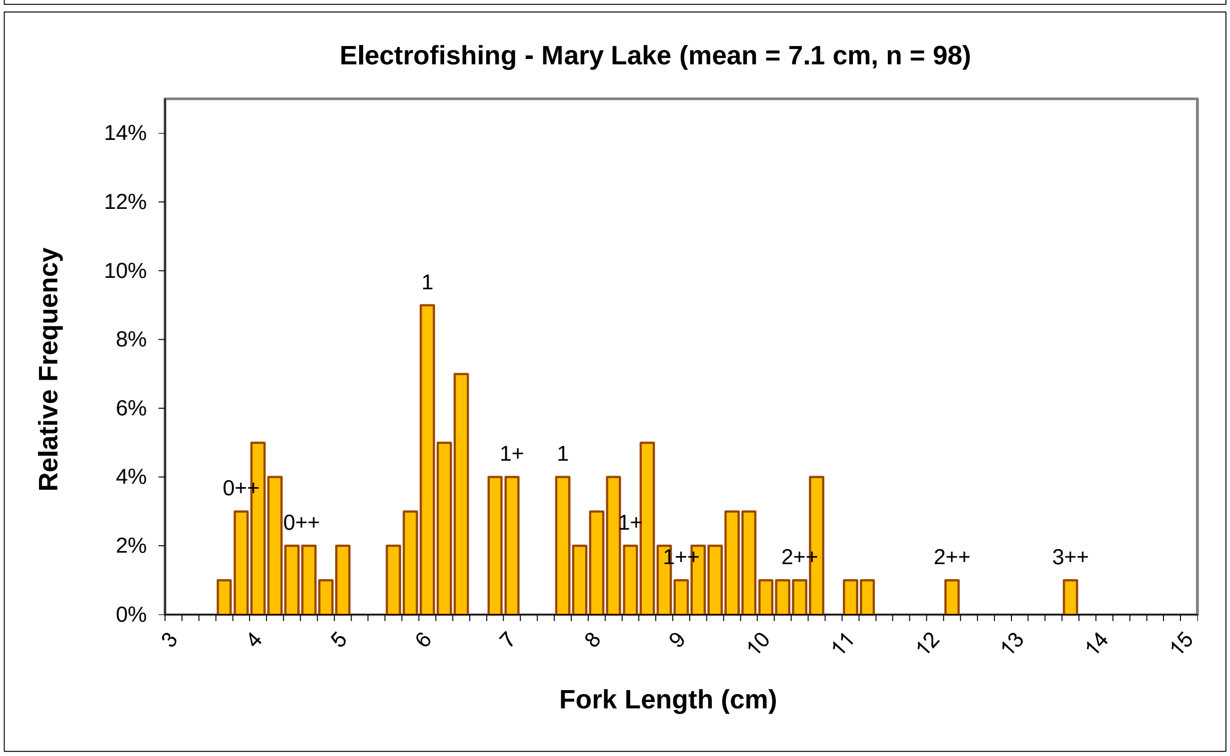


Figure 5.19: Length-Frequency Distributions for Arctic Charr Captured by Backpack Electrofishing at Mary Lake (BLO) and Reference Lake 3 (REF3), Mary River Project CREMP, August 2017

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

Table G.38). On average, arctic charr non-spawners exhibited age, size (length and weight), and LSI that were similar to male and female individuals with developing gonads (Appendix Table G.38). All captured arctic charr of spawning size contained body cavity parasites (Appendix Table G.38), the incidence rate of which was comparable to that observed previously at Mary Lake, at other mine-exposed lakes, and during historical studies at mine local study area lakes (NSC 2014, 2015a).

The length-frequency distribution of arctic charr captured at littoral/profundal areas of Mary Lake in 2017 differed significantly from that of fish captured during the baseline period (Table 5.11; Appendix Table G.39). On average, arctic charr captured at littoral/profundal areas of Mary Lake in 2017 were significantly shorter and lighter than those captured during the baseline period, but no significant differences in growth were shown between these periods (Table 5.11). Although condition of adult arctic charr captured at Mary Lake differed significantly between 2017 and the baseline period, the magnitude of this difference was within the CES_C of ±10% (Table 5.11) suggesting that this difference was not ecologically meaningful. No consistent differences in adult arctic charr health endpoints of size, growth, or condition were indicated at Mary Lake for individual years of mine operation from 2015 to 2017 compared to baseline data (Table 5.11). 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.

5.3.7 Integrated Effects Evaluation

At Mary Lake, turbidity and aqueous concentrations of total aluminum, iron, manganese, and uranium were elevated compared to the reference lake in 2017, 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 the reference lake as a result of receiving flow from relatively large river systems (i.e., Tom River and Mary River inflows to the Mary Lake north and south basins, respectively). Aluminum, iron, 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 the reference lake in 2017 and, with the exception of slightly elevated sediment manganese concentrations at littoral stations, were similar to concentrations observed during the baseline period. Although sediment chromium, iron, and manganese concentrations were above SQG at Mary Lake in 2017, with the exception of chromium, these metals were also above SQG at the reference lake suggesting low potential for any adverse effects to biota associated with these metals. No metals were observed at concentrations above the sediment AEMP benchmarks at littoral and profundal stations of Mary Lake in 2017.

Mary Lake chlorophyll-a concentrations were significantly higher than at the reference lake in 2017, suggesting greater primary production at Mary Lake. However, Mary Lake chlorophyll-a concentrations were continuously well below the AEMP benchmark during all seasonal sampling events in 2017, 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. Benthic invertebrate community data collected at Mary Lake in 2017 indicated no significant differences in primary community endpoints at littoral habitat, but significantly higher density and richness at profundal habitat, compared to the reference lake. Similar to Sheardown Lake, the differences in community composition appeared to reflect naturally differing sediment TOC and/or particle size between Mary Lake and the reference lake in 2017. No ecologically significant differences in primary benthic invertebrate community metrics, dominant taxonomic groups, or FFG were consistently indicated between the mine baseline period and individual years since the commencement of commercial mine operation (2015, 2016, 2017) at Mary Lake that could be linked to an adverse response to mine operations. Analysis of Mary Lake arctic charr populations suggested greater fish abundance compared to the reference lake in 2017, but no definitive changes in numbers of arctic charr in 2017 relative to baseline data. No significant or ecologically meaningful differences in growth and condition of nearshore captured arctic charr occurred between Mary Lake and the reference lake in 2017, nor between arctic charr collected in 2017 compared to the baseline period for nearshore and littoral/profundal arctic charr populations at Mary Lake. Collectively, the chlorophyll-a, benthic invertebrate community and arctic charr fish population data all suggested no adverse mine-related influences to the biota of Mary Lake in the third year of mine operation at the Mary River Project.

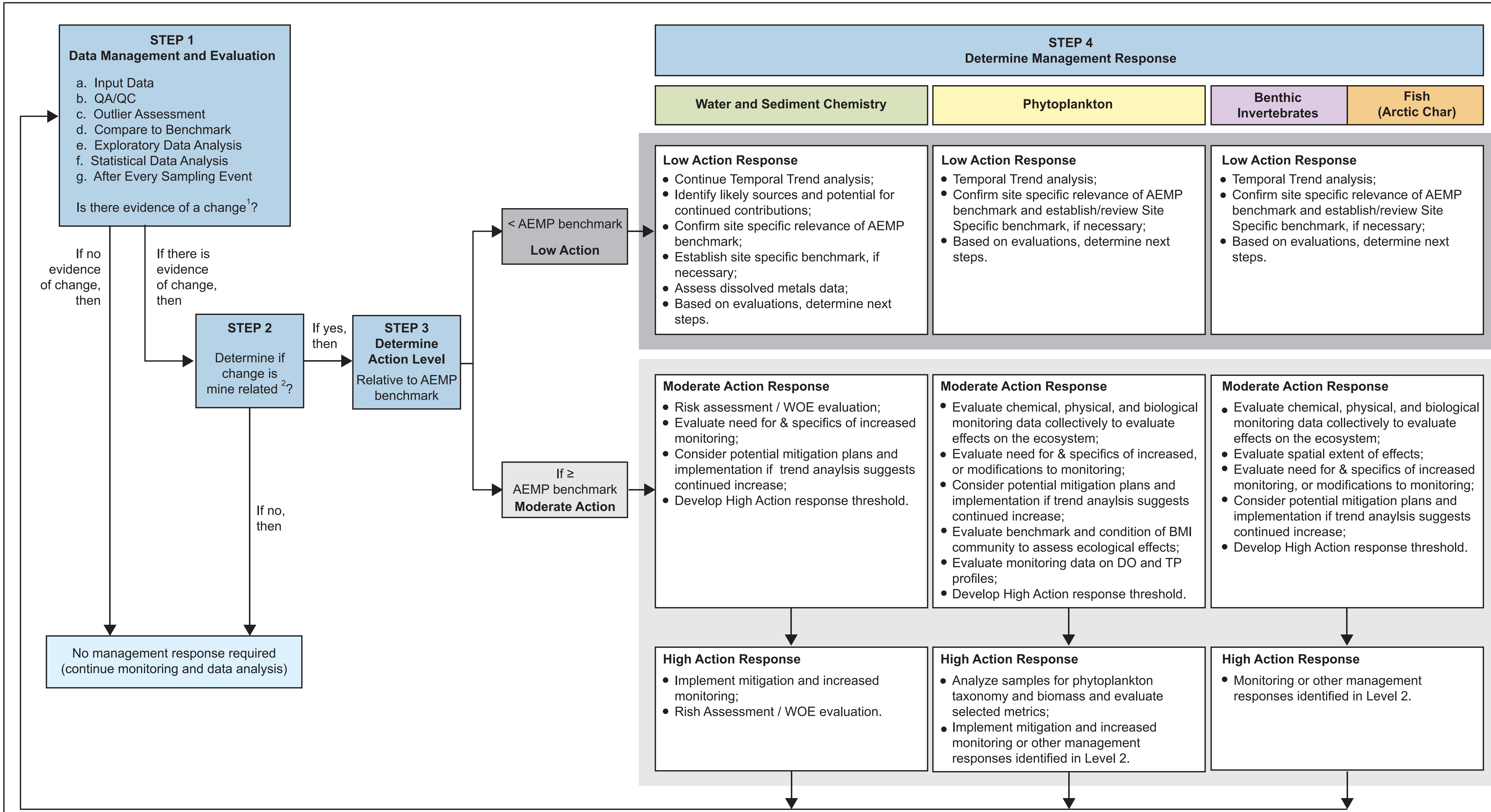
6 EFFECTS DETERMINATION AND RECOMMENDATIONS

6.1 Contextual Overview

The objective of the 2017 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 third full year of mine operation. The 2017 CREMP utilized an effects-based approach that included standard environmental effects monitoring techniques to provide rigorous analysis of potential mine-related effects at key receiving waterbodies. Under this approach, water quality and sediment quality data were used to support the interpretation of phytoplankton, benthic invertebrate community, and fish population survey data collected at mine-exposed areas of the Camp Lake, Sheardown Lake, Mary River, and Mary Lake systems. The evaluation of potential mine-related effects within these systems was based on comparisons of the 2017 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 2014). This effects assessment summarizes instances in which the Mary River Project AEMP benchmarks for water and sediment quality were exceeded at waterbodies examined under the CREMP, as well as relevant reference area information and evidence of biological effects at these waterbodies to provide additional perspective and 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 north branch and main stem channels of Camp Lake Tributary 1 (CLT1), and AEMP benchmarks for sediment quality were exceeded at Camp Lake (Table 6.1). At the CLT1 north branch, aqueous copper concentrations were elevated compared to the average concentration from comparable reference creek stations in 2017, as well as to concentrations observed at the north branch during baseline studies (2005 – 2013 data; Appendix Figure C.2). In turn, this suggested a mine-related source of copper to the CLT1 north branch. Biological monitoring conducted at the CLT1 north branch in 2017 indicated no adverse effects to phytoplankton (chlorophyll-a) 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 the CLT1 north branch, a low action response to identify the likely source(s) of copper to the system is recommended to meet obligations under the AEMP Management Response Framework.



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 2017 Mary River Project CREMP and Supporting Reference Area and Biological Effects Summary Information

Waterbody	AEMP Benchmark Exceedance	Reference Area Information	Evidence of Biological Effects
Camp Lake Tributary 1 (North Branch)	Aqueous copper concentration greater than 0.0022 mg/L benchmark in summer (mean = 0.022 mg/L; max = 0.0024 mg/L) & fall (mean = 0.0023; max = 0.0025 mg/L)	Mean copper concentration (summer) = 0.0009 mg/L (max = 0.0012 mg/L) Mean copper concentration (fall) = 0.0013 mg/L (max = 0.0019 mg/L)	No adverse effects on phytoplankton or benthic invertebrate community endpoints.
Camp Lake Tributary 1 (Main Stem)	Aqueous aluminum concentration greater than 0.179 mg/L benchmark in fall at upper main stem (0.395 mg/L) and lower main stem (mean = 0.300 mg/L; max = 0.620 mg/L) Aqueous iron concentration greater than 0.326 mg/L benchmark in fall at upper main stem (0.705 mg/L) and lower main stem (mean = 0.449 mg/L; max = 0.821 mg/L)	Mean aluminum concentration (fall) = 0.208 mg/L (max = 0.693 mg/L) Mean iron concentration (fall) = 0.222 mg/L (max = 0.739 mg/L)	No adverse, metal related effects on phytoplankton or benthic invertebrate community endpoints.
Camp Lake	Sediment arsenic concentration > 5.9 mg/kg benchmark at single littoral monitoring station (9.9 mg/kg). Sediment iron concentration > 52,400 mg/kg benchmark at single littoral monitoring station (61,600 mg/kg). Sediment nickel concentration > 72 mg/kg benchmark at single littoral monitoring station (77 mg/kg). Sediment phosphorus concentration > 1,580 mg/kg benchmark at single littoral monitoring station (1,690 mg/kg). Sediment arsenic concentration > 5.9 mg/kg benchmark, on average, at profundal stations (mean = 6.68 mg/kg; max = 17.5 mg/kg).	Reference lake littoral sediment mean arsenic concentration = 3.35 mg/kg (max = 5.67 mg/kg) Reference lake littoral sediment mean iron concentration = 41,960 mg/kg (max = 73,400 mg/kg). Reference lake littoral sediment mean nickel concentration = 38 mg/kg (max = 58 mg/kg). Reference lake littoral sediment mean phosphorus concentration = 1,039 mg/kg (max = 2,020 mg/kg). Reference lake profundal sediment mean arsenic concentration = 4.78 mg/kg (max = 5.50 mg/kg).	No adverse, ecologically significant difference in phytoplankton, benthic invertebrate community or fish population endpoints compared to reference and baseline conditions.
Sheardown Lake Tributary 1	Aqueous copper concentration greater than 0.0022 mg/L benchmark in spring, summer and fall (annual mean = 0.0027 mg/L; max = 0.0034 mg/L)	Mean copper concentration (annual) = 0.0009 mg/L (max = 0.0012 mg/L)	No adverse, metal related effects on phytoplankton or benthic invertebrate community endpoints.
Sheardown Lake SE	Littoral sediment iron concentration > 34,400 mg/kg benchmark (mean = 44,867 mg/kg; max = 51,500 mg/kg). Littoral sediment manganese concentration > 657 mg/kg benchmark (mean = 1,506 mg/kg; max = 2,560 mg/kg). Profundal sediment iron concentration > 34,400 mg/kg benchmark (mean = 39,150 mg/kg; max = 41,300 mg/kg). Profundal sediment manganese concentration > 657 mg/kg benchmark (mean = 820 mg/kg; max = 1,170 mg/kg).	Reference lake littoral sediment mean iron concentration = 41,960 mg/kg (max = 73,800 mg/kg) Reference lake littoral sediment mean manganese concentration = 639 mg/kg (max = 1,010 mg/kg). Reference lake profundal sediment mean iron concentration = 46,740 mg/kg (max = 55,400 mg/kg). Reference lake profundal sediment mean manganese concentration = 1,266 mg/kg (max = 1,500 mg/kg).	No adverse, ecologically significant difference in phytoplankton, benthic invertebrate community, or fish population endpoints compared to reference and baseline conditions.
Mary River	Aqueous aluminum concentration greater than 0.966 mg/L benchmark in fall at Station CO-05 (1.48 mg/L) Aqueous copper concentration greater than 0.024 mg/L benchmark in fall at Station CO-05 (0.025 mg/L) Aqueous iron concentration greater than 0.874 mg/L benchmark in fall at Station CO-05 (2.12 mg/L)	Mean aluminum concentration (fall) = 0.147 mg/L (max = 0.186 mg/L) Mean copper concentration (fall) = 0.0010 mg/L (max = 0.0014 mg/L) Mean iron concentration (fall) = 0.100 mg/L (max = 0.146 mg/L)	No adverse, metal related effects on phytoplankton or benthic invertebrate community endpoints.
Mary Lake	Aqueous silver concentration > 0.00010 mg/L benchmark at north basin in summer (mean = 0.000103 mg/L; max = 0.00017 mg/L).	Mean silver concentration (summer) = < 0.00001 mg/L	No adverse, ecologically significant difference in phytoplankton, benthic invertebrate community, or fish population endpoints compared to reference and baseline conditions.

At the CLT1 main stem, aqueous concentrations of aluminum and iron were elevated above their respective AEMP benchmarks during the fall sampling event in 2017 (Table 6.1). However, similar concentrations of these metals were also indicated at one of the four reference creek stations (Station MRY-REF-3) in fall 2017 (Table 6.1; Appendix Table B.2). At CLT1 main stem stations L2-03 and L1-09, and reference creek station MRY-REF-3, total aluminum and iron concentrations were elevated above AEMP benchmarks, and turbidity was also considerably elevated compared to other CLT1 main stem or reference creek stations on the same collection date (i.e., 4.3 times and 14 times higher, respectively, on average; see Appendix Tables B.2 and C.14). Because similar turbidity was not observed among all CLT1 main stem and reference creek stations on the collection date, higher turbidity in these samples, and aluminum and iron metal concentrations associated with this turbidity, most likely reflected compromised samples at the time of collection[8]. The occurrence of aluminum and iron concentrations above AEMP benchmarks at the CLT1 main stem therefore was unlikely related to mine activity, and thus a low action response is recommended to meet obligations under the AEMP Management Response Framework. In this case, it is recommended that in future sampling, all personnel conducting water sampling should be made aware of, and adhere to, Standard Operating Procedures (SOP) developed by Baffinland (2014) for AEMP field water sample collection to ensure that samples do not become compromised.

At Camp Lake, AEMP benchmarks for water quality were consistently met in 2017, but benchmarks for sediment quality were exceeded for four metals (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, and the average arsenic concentration from profundal stations was also above the respective AEMP benchmark. Because the lone littoral sediment chemistry monitoring station is located near the inlet from CLT1, mine-influenced flow from this tributary likely contributed to elevation of the metals indicated above in sediment at this location. Notably, iron concentrations in sediment of Reference Lake 3 were above SQG, indicating natural elevation within the region of the mine (Appendix Table D.6). Sediment arsenic concentrations at the Camp Lake littoral sediment chemistry station in 2017 were slightly elevated compared to concentrations at the reference lake, and to Camp Lake average littoral station concentrations recorded in the mine baseline period. 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

[8] Within the CLT1 main stem, no gradient in turbidity was indicated with progression downstream from any of the four sampling stations that suggested a definitive point source of suspended material (e.g., the Tote Road), indicating that elevated turbidity in the water samples most likely reflected sampler error on August 27th, 2017 (see Appendix Table C.14).

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[9]. 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 to meet obligations 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 - 17 provided by Minnow (2016b) 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 only at Sheardown Lake SE (Table 6.1). Sheardown Lake NW was the only waterbody at which no water or sediment quality AEMP benchmarks were exceeded during sampling completed in 2017. At SDLT1, aqueous copper concentrations were elevated compared to the average concentration from reference creek stations in 2017, but not to concentrations observed at SDLT1 during baseline studies (2005 – 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 2017 compared to baseline conditions, copper concentrations at SDLT1 may naturally be similar to the AEMP benchmark. Biological monitoring conducted at SDLT1 in 2017 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

[9] Notably, in addition to benthic biota, no adverse effects to biota typically associated with pelagic habitat (i.e., phytoplankton or arctic charr population and health) were indicated at Camp Lake in the 2017 CREMP.

associated with copper concentrations above the AEMP benchmark at SDLT1, a low action response to identify the likely source(s) of copper to the system is recommended to meet obligations under the AEMP Management Response Framework.

At Sheardown Lake SE, AEMP benchmarks for water quality were consistently met, but AEMP benchmarks for sediment quality were exceeded for iron and manganese for both littoral and profundal habitat stations in 2017 (Table 6.1). Sediment iron concentrations at Sheardown Lake SE in 2017 were similar to concentrations at the reference lake, as well as to concentrations documented at Sheardown Lake SE in baseline studies. However, sediment manganese concentrations were slightly greater at Sheardown Lake SE than at the reference lake and compared to baseline studies (littoral habitat only) at the southeast basin. 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 the reference lake (Appendix Table D.6). 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. No adverse effects to benthic invertebrates and other biota were indicated at Sheardown Lake SE in 2017 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 to meet obligations under the AEMP Management Response Framework. Specifically, it is recommended that the relevance of site-specific sediment quality AEMP benchmarks for Sheardown Lake SE be assessed and, if necessary, determined anew taking into consideration data from the reference lake and applicable SQG.

6.4 Mary River and Mary Lake System

Within the Mary River and Mary Lake systems, AEMP benchmarks for water quality were exceeded at only one of five Mary River water quality monitoring stations, and at the Mary Lake north basin in 2017 (Table 6.1). Notably, no AEMP benchmarks for sediment quality were exceeded at any of the Mary Lake north or south basin stations. AEMP benchmarks for aqueous concentrations of aluminum, copper, and iron were exceeded at Mary River far-field mine-exposed Station CO-05 in 2017, but only during the fall sampling event (Table 6.1). Similar to observations at the CLT1 main stem, very high turbidity in the Station CO-05 water sample relative to samples collected at other Mary River stations during the fall sampling event suggested that elevated concentrations of aluminum, copper and iron in the August 2017 CO-05 sample reflected a compromised sample rather than a mine-related influence on Mary River water quality. Therefore, a low action response requiring that all personnel conducting water sampling be made aware of, and adhere to, Baffinland (2014) SOP for AEMP field water sample collection is recommended to ensure that water samples do not become compromised in the future.

At Mary Lake, aqueous silver concentrations greater than the applicable AEMP benchmark were indicated at the north basin during the summer sampling event in 2017 (Table 6.1; Appendix Table C.73). Silver concentrations were consistently below laboratory reportable detection limits (RDL) at all other CREMP stream and lake water quality monitoring stations in 2017, including those stations located in Camp Lake and Tom River upstream of the Mary Lake north basin, for all winter, spring, summer, and fall sampling events (Appendix Table C.26). In addition, silver concentrations had consistently been below laboratory RDL during all winter, spring, summer, and fall sampling events in both previous years at all CREMP stations, including those at the Mary Lake north basin (see Minnow 2016a, 2017). The absence of any spatial gradient in silver concentrations with distance from the mine, and the extremely rare occurrence of silver concentrations above laboratory RDL, suggested that concentrations of silver greater than the applicable AEMP benchmark at the Mary Lake north basin in summer 2017 likely reflected a laboratory related anomaly and was unlikely mine-related. Therefore, a low action response that includes on-going monitoring to re-evaluate silver concentrations above laboratory RDL at this study area, and more detailed scrutiny of laboratory data to ensure accuracy, is recommended in future CREMP to meet obligations under the AEMP Management Response Framework.

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