Mary River Project 2024 Core Receiving Environment Monitoring Program Report

Part 1 of 3
(Main Report)

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Program Report

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EXECUTIVE SUMMARY

The Mary River Project (the Project) is an operating high-grade iron mine located in the Qikiqtani Region of northern Baffin Island, Nunavut. Owned and operated by Baffinland Iron Mines Corporation (Baffinland), the mine began commercial operation in 2015. Mining activities at the mine site in 2024 included open pit ore extraction, ore haulage, stockpiling, crushing, and screening, followed by transport by truck to Milne Port for subsequent seasonal loading onto bulk carrier ships for transfer to international markets. No milling or additional processing of the ore is conducted on-site and therefore no tailings are produced at the Project. Mine waste management facilities at the mine site include a mine waste rock stockpile and surface runoff collection/containment ponds situated near the mine waste rock stockpile and ore stockpile areas. In addition to periodic discharge of treated effluent from these facilities to the Mary River system, other potential mine inputs to aquatic systems located adjacent to the mine site include runoff and dust from ore (crusher) stockpiles located within the Sheardown Lake catchment, treated sewage discharge to the Mary River, deposition of fugitive dust generated by mine activities, and general mine site runoff.

Under the terms and conditions of the Mine's Type 'A' Water Licence from the Nunavut Water Board (NWB), Baffinland was required to develop and implement an Aquatic Effects Monitoring Plan (AEMP) for the mine site. To meet AEMP objectives, Baffinland established the Core Receiving Environment Monitoring Program (CREMP) to assess mine-related impacts on water quality, sediment quality, and aquatic biota (phytoplankton, benthic invertebrates, and fish). The CREMP focuses on primary receiving systems, including the Camp Lake system (Camp Lake and Camp Lake Tributaries 1 and 2), the Sheardown Lake system (Sheardown Lake Northwest, Sheardown Lake Southeast, and Sheardown Lake Tributaries 1, 9, and 12), the Mary River (including Mary River Tributary-F), and the Mary Lake system. Since the mine's commercial operation began, annual assessments under CREMP have used site-specific benchmarks for water and sediment quality developed for the AEMP, along with standard Environmental Effects Monitoring (EEM) techniques. Annual results are applied within a four-step Assessment Approach and Management Response Framework k designed for the Mary River Project AEMP to guide management response decisions related to changes in parameter concentrations and/or aquatic biota attributable to mine operations.

In 2024, the Mary River Project CREMP identified potential mine-related effects on abiotic and biotic factors within the Camp Lake system. Mine-related influences on water quality were observed in the Camp Lake Tributary 1 (CLT1) Main Stem, particularly in the Upper Main Stem (Station L2-03), where concentrations of aluminum, iron, uranium, sulphate, sodium, and molybdenum indicated a potential mine-related influence based on concentrations that were elevated compared to baseline and reference in 2024 and increasing trends/patterns since the baseline period. However, there were no similar increasing trends/patterns for these parameters over the mine operations period suggesting that potential mine-related influence has not been intensifying with ongoing mine operations. Of these parameters, only aqueous concentrations of total aluminum and iron exceeded their respective AEMP benchmarks (in summer and summer and fall, respectively). Increasing concentrations, since the baseline period and over the period of mine operations, were observed for total and dissolved uranium and indicated a mine-related influence. No corresponding adverse effects on phytoplankton or the benthic invertebrate community (BIC) in the CLT1 Main Stem were noted. In Camp Lake, uranium was the only water quality parameter for which elevated concentrations relative to baseline and reference and increasing temporal patterns indicated a mine-related influence, concentrations remained below the WQG. No other mining-related effects were identified within the Camp Lake system.

Results of 2024 CREMP monitoring in the Camp Lake system require response actions under the AEMP Management Response Framework. Recommendations were made for continued monitoring of the BIC in the CLT1 mainstem to monitor for potential effects to biota resulting from mine-related influences on water quality parameters. Additionally, in 2025, for water quality parameters for which there was a determination of mine-related or potential mine-related influence, temporal trend analyses will be conducted to further investigate temporal trends/patterns, total compared to dissolved concentrations of metals will be investigated to assess biological availability and potential effects on aquatic biota, and potential sources to affected waterbodies/watercourses in the Camp Lake system will be investigated to better define mine-related influence and the potential for continued contributions. Finally, development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations.

In 2024, within the Sheardown Lake system, Sheardown Lake Tributary 1 (SDLT1) exhibited the most pronounced mine-related influences on water quality across the entire CREMP monitoring area. Mine-related influence was determined for several water quality parameters including barium, cadmium, calcium, chloride, cobalt, conductivity, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, total dissolved solids (TDS), total Kjeldahl nitrogen (TKN), and uranium based on concentrations that were elevated compared to baseline and reference and increasing trends/patterns, particularly from 2022 to 2024. Of these parameters, only aqueous concentrations of total cadmium exceeded the AEMP benchmark (in summer and fall). Mine-related influence at SDLT1 is likely linked to the extensive mine site infrastructure within the SDLT1 catchment area, particularly site water management through the KM 105 Surface Water Management Pond (KM 105 Pond).

Since its commissioning in 2022, the pond has not performed as expected, leading to persistent seepage and water quality challenges and multiple remediation efforts. Mine-related influences on the BIC at SDLT 1 were also detected, with results suggesting they were likely driven by organic matter enrichment and differences in physical habitat conditions rather than metal contamination as a primary stressor. Influences associated with site water management and remediation efforts at the KM 105 Pond are consistent with the factors that may have resulted in shifts to the SDLT1 BIC in 2024.

Water quality at Sheardown Lake Tributary 9 (SDLT9) has also been influenced by mining activities, resulting in elevated nitrogen-related compounds (ammonia, nitrate, nitrite, and TKN) as identified in the 2023 CREMP (Minnow 2024a) and again in 2024. A special investigation, involving expanded spatial sampling and completed in the fall of 2024 identified activities occurring at the Dyno Nobel Emulsion Plant (Dyno facility), which stores ammonium nitrate on-site and is located adjacent to SDLT9, as the primary source of these compounds. No adverse mine-related influences on phytoplankton were determined but there were ecologically meaningful differences in BIC structure at SDLT9 in 2024 compared to baseline (however the BIC was comparable to the reference creek). Though localized natural inter-annual variability in habitat conditions may account for changes in the BIC relative to baseline, mine-related influences on water quality at SDLT9 in 2024 also suggest the potential for a mine-related.

At Sheardown Lake Tributary 12 (SDLT12) in 2024, potential mine-related influence was determined on water quality parameters including alkalinity, barium, calcium, chloride, conductivity, hardness, magnesium, molybdenum, potassium, sodium, strontium, TDS, and uranium based on spring concentrations that were elevated compared to baseline and reference and increasing trends/patterns since the initiation of sampling at this location in 2021. Mine-related influence on water quality at SDLT12 is likely linked to snow stockpiling activities and inputs from dust deposition (mostly originating from the mine site crusher facility) in the catchment area upstream of the SDLT12 monitoring location.

At both Sheardown Lakes (Northwest [NW] and Southeast [SE]) in 2024, mine-related influences on water quality were determined for nitrate, sulphate, molybdenum, and uranium, as well as for chloride in Sheardown Lake NW only. Determinations were based on elevated aqueous concentrations relative to baseline and/or reference in 2024 as well as evidence of increasing trends/patterns in parameter concentrations, generally since 2018/2019 and persisting in 2024. These trends suggest potential influences from activities occurring at the Dyno Facility (in Sheardown Lake SE only), site water management through the KM 105 Pond, and the broader mine site infrastructure within the catchments of the Sheardown Lakes.

At Sheardown Lake NW, mean iron concentrations in littoral and profundal sediments exceeded the AEMP benchmark, with statistically significant increasing trends over both baseline and mine operation periods. Spatial patterns in iron concentration within the lake were also observed suggesting the emergence of a mine-related influence on sediment quality, that may be linked to contributions of sediment from tributaries. To date, no adverse-mine related biological effects have been identified in either of the Sheardown Lakes.

Results of 2024 CREMP monitoring in the Sheardown Lake system require response actions under the AEMP Management Response Framework. Recommendations were made for continued monitoring of the BIC in at SDLT1 to monitor for potential effects to biota resulting from mine-related influences on water quality parameters. Additionally, in 2025, for water quality parameters for which there was a determination of mine-related or potential mine-related influence at SDLT1, SDLT12, Sheardown Lake NW and/or Sheardown Lake SE, temporal trend analyses will be conducted to further investigate temporal trends/patterns, total compared to dissolved concentrations of metals will be investigated to assess biological availability and potential effects on aquatic biota, and potential sources to affected waterbodies/watercourses in the Sheardown Lake system will be investigated to better define mine-related influence and the potential for continued contributions. Development of an AEMP benchmark for uranium will also be considered to support evaluation of the potential biological effects of observed concentrations.

Mitigation efforts will be implemented to improve water quality in the Sheardown Lake Tributaries and Sheardown Lakes NW and SE. At the KM 105 Pond, the focus for remediation efforts in 2025 will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. The installation of a filter berm upstream of the water license Surveillance Network Program (SNP) monitoring location Station MS-C-D (which is located on a tributary to SDTL1 that originates from the southeast and flows into SDLT1 between Stations D1-05 and D1-00) is also planned in 2025. The purpose of this additional infrastructure is to further mitigate for mine-related contributions of total suspended solids (TSS) to SDLT1 associated with dust and other sources of TSS within the upstream catchment area. Water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT1 and Sheardown Lakes NW and SE as a basis for informing the potential need for further investigations and mitigation.

An activity audit concerning the transportation, storage, and handling of ammonium nitrate on the activities occurring at the Dyno facility is being implemented, along with potential additional water sampling during the open water season in 2025, as needed, to help identify point source(s) of aqueous nitrogen compounds. Mitigation measures will be developed based on the findings. Water quality monitoring at SDLT9 will continue in the 2025 CREMP to assess the effectiveness of mitigation efforts at the Dyno facility in reducing the concentrations of aqueous nitrogen compounds. This monitoring may be supplemented by expanded spatial sampling in the fall of 2025, if necessary to fully evaluate mitigation effectiveness.

Finally, temporal trend analysis of iron concentrations in littoral and profundal sediment in Sheardown Lake NW will be repeated with the inclusion of new monitoring data to evaluate whether an increasing trend continues to be identified and to contribute to determination of mine related influences despite iron sediment concentrations that were similar to reference and baseline conditions in 2024. Further, spatial comparisons between iron concentrations in sediment within the lake will be completed to support determination of the influence of key lake tributaries on the influx of sediment iron into Sheardown Lake NW.

Within the Mary River and Mary Lake System, mine-related influences in 2024 were limited to a small number of water quality parameters, including nitrate, sulphate, and selenium in Mary River Tributary-F (MRTF). At MRTF, aqueous concentrations of nitrate and sulphate were elevated compared to baseline and reference in at least one season in 2024 and concentrations of these parameters have shown increasing trends/patterns that started in 2019 and 2017 (for nitrate and sulphate, respectively) but have not be consistent over time, suggesting they are not intensifying with ongoing mine operations. Selenium concentrations at MRTF have frequently been below the laboratory reporting limits (LRL) and evaluation for a temporal pattern is confounded by changing LRLs, therefore evidence does not suggest a potential mine-related effect on selenium at MRTF. Despite potential influences on water quality, no effects on phytoplankton at MRTF were determined. Mine-related influences on nitrate and sulphate concentrations at MRTF may be associated with effluent discharge (i.e., from the MS-08 final discharge point [FDP] into MRTF). No mine-related influences on water quality, sediment quality, or biota were identified elsewhere in the Mary River or in Mary Lake.

Results of 2024 CREMP monitoring in MRTF require response actions under the AEMP Management Response Framework. Recommendations were made including temporal trend analysis of aqueous concentrations of nitrate and sulphate to be conducted in 2025 to further investigate temporal trends/patterns. Further, in 2025, a special investigation will be conducted evaluating effluent and receiving water quality data that are routinely collected as part of MDMER requirements for the MS-08 FDP to evaluate influence of the MS-08 FDP as a potential source of nitrate and sulphate to MRTF.

Results of the 2024 CREMP were compared to predictions for magnitude of effects made in the Final Environmental Impact Statement (FEIS) for the Project (Baffinland 2012). Overall comparisons of water quality and sediment quality data within the Camp Lake, Sheardown Lake, and Mary Lake systems in 2024 to FEIS predictions indicated all parameter concentrations were within applicable significance ratings for magnitude. This also meant that FEIS predictions for (absence of) effects on arctic charr health and condition were met. Therefore arctic charr health and condition at Camp Lake, Sheardown Lake, and Mary River and Lake conformed with predictions made in the FEIS (Baffinland 2012). Project-related sedimentation accumulation thickness of less than 1 mm/year was predicted in the FEIS to result in negligible effects on direct mortality of arctic charr. Because the sediment accumulation rate over the 2023 to 2024 arctic charr egg incubation period was well below 1 mm/y at Sheardown Lake NW, FEIS predictions for (absence of) direct mortality of arctic charr were met (Minnow 2025). Therefore, direct fish mortality effects at Sheardown Lake NW were in conformance with predictions made in the FEIS (Baffinland 2012).

Overall, the most significant mine-related influences have been observed within the Sheardown Lake System, where most watercourses/waterbodies assessed in the CREMP have shown some degree of mine-related influence, with effects extending to the BIC in tributaries of Sheardown Lake NW (i.e., SDLT1) and Sheardown Lake SE (i.e., SDLT9). Links between mining activities within the Sheardown Lake System and the observed changes have been identified, and corresponding mitigation measures and recommendations have been provided. While some mine-related influences were noted in the Camp Lake and Mary River/Lake Systems, these effects appear to be more localized and, in the Camp Lake system, may be influenced by natural variation. Ongoing implementation of the annual CREMP will continue to assess potential mine-related influences and management actions will be applied as required according to the AEMP Management Response Framework.

EXECUTIVE SUMMARY ᓯᕗᓕᖅᑎᓃᖅᑐᖅ ᓇᐃᓈᖅᓯᒪᔪᖅ

ᓄᓘᔾᔨ ᐱᓕᕆᐊᖅ (ᐱᓕᕆᐊᖅ) ᐊᐅᓚᓂᖃᖅᑎᑦᑎᔪᖅ ᓴᕕᒃᓴᑎᐊᕙᐅᓃᓂᒃ ᐅᔭᕋᖕᓂᐊᖅᑐᑎᒃ ᕿᑭᖅᑕᓂ ᓄᓇᖁᑖᓂ ᕿᑭᖅᑖᓗᑉ ᐅᐊᖕᓇᖕᒐᓂ, ᓄᓇᕗᒻᒥ. ᓇᖕᒥᓂᕆᔭᐅᓪᓗᓂ ᐊᐅᓚᑕᐅᓪᓗᓂᓗ ᐹᕙᓛᓐ ᓴᕕᒃᓴᓂᒃ ᐅᔭᕆᖕᓂᐊᖅᑏᑦ ᑯᐊᐳᕇᓴᓐ (ᐹᕙᓛᓐ), ᐅᔭᕋᖕᓂᐊᕐᕕᒃ ᐱᒋᐊᓚᐅᖅᑐᖅ ᓂᐅᕈᑎᖃᖃᑦᑕᓕᖅᑐᑎᒃ 2015-ᒥ. ᐅᔭᕋᖕᓂᐊᕐᓃᑦ ᐱᓕᕆᓂᐅᖃᑦᑕᖅᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ 2024-ᒥ ᐱᖃᓯᐅᔾᔨᓯᒪᔪᖅ ᐃᑎᖅᓴᓕᐅᖅᑐᑎᒃ ᐅᔭᕋᖕᓂᐊᖅᑐᑎᒃ, ᓴᕕᒃᓴᒥᒃ ᐅᓯᑲᖅᑕᕐᓂᖅ, ᑲᑎᖅᓱᐃᓂᖅ, ᓯᖃᓕᑦᑎᕆᓂᖅ, ᐊᒻᒪᓗ ᐊᕕᒃᑐᖅᑐᒋᑦ, ᐅᓯᑲᖅᑕᖅᑕᐅᕚᒃᑐᑎᑦ ᑮᖑᓂᐊᒎᑦ ᓄᓇᓯᐅᑎᒃᑯᑦ ᕿᖑᐊᓄᑦ ᐅᒥᐊᕐᔪᐊᓄᑦ ᐊᕐᕌᒎᑉ ᐃᓚᐃᓐᓇᖕᒪᓄᑦ ᐅᓯᓕᖅᓱᖅᑕᐅᓪᓗᑎᒃ ᓇᒡᓯᐅᑎᔭᐅᓪᓗᑎᑦ ᓄᓇᕐᔪᐊᓕᒫᒧᑦ ᓂᐅᕐᕈᓯᕐᕕᐅᕚᒃᑐᓄᑦ. ᐃᒪᕐᒥᒃ ᐊᑐᕐᓗᑎᒃ ᓂᐅᖅᑐᐃᓂᖅᑕᖄᖏᑦᑐᖅ ᐅᕝᕙᓗᓐᓃᑦ ᐃᓚᐅᖅᑐᕆᐊᖃᖏᑦᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐊᒻᒪᓗ ᑕᓯᕋᓕᐊᕐᓂᖅᑕᖄᖏᑦᑐᖅ ᐱᓕᕆᐊᒥ. ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐊᑐᖅᑕᐅᔾᔮᖏᑦᑐᓄᑦ ᐱᓕᕆᕝᕕᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐱᖃᓯᐅᔾᔨᔪᑦ ᐅᔭᕋᖕᓂᐊᑕᐅᔪᓂ ᐊᑐᖅᑕᐅᔾᔫᖏᑦᑐᓂᒃ ᑲᑎᖅᓱᐃᕝᕕᒃ ᐊᒻᒪᓗ ᑕᓯᕋᕐᓂᒃ ᑰᒃᑐᓂᒃ ᑲᑎᖅᓱᐃᕕᖕᒥ/ᐃᒻᒥᖅᓱᐃᕝ ᐃᓂᒋᔭᐅᔪᕕᒃ ᑕᓯᕋᖅ ᐅᔭᕋᖕᓂᐊᕐᕕᐅᑉ ᖃᓂᒋᔭᖕᒐᓂ ᐅᔭᕋᖕᓂᐊᖅᑕᐅᔪᓂ ᐊᑐᖅᑕᐅᔾᔫᖏᑦᑐᓂ ᐅᔭᖅᑲᓂᒃ ᑲᑎᖅᓱᐃᕝᕕᐅᔪᓂ ᐊᒻᒪᓗ ᓴᕕᒃᓴᓂᒃ ᑲᑎᖅᓱᐃᕖᑦ ᖃᓂᒋᔭᖕᒐᓂ. ᖁᒃᑲᓐᓂᐊᒍᑦ ᐃᓖᓐᓂᒃᑯᑦ ᐃᒪᖅ ᑯᕕᔭᐅᖃᑦᑕᖅᑐᖅ ᓴᓗᒪᖅᓴᖅᓯᒪᔪᖅ ᑖᒃᑯᓇᙱᑦ ᓄᓘᔾᔨ ᐱᓕᕆᕕᖕᓂ. ᐊᓯᖏᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐃᒻᒥᑐᐃᓐᓇᕆᐊᓖᑦ ᑕᓯᕐᓄᑦ ᖃᓂᒋᔭᖏᓐᓂ ᐅᔭᕋᖕᓂᐊᕐᕕᐅᔩᑉ ᐱᖃᓯᐅᔾᔨᔪᑦ ᑰᒃᑐᓂᒃ ᐊᒻᒪ ᓴᕕᒃᓴᓐᓃᖅᑐᑦ ᐳᔪᕋᑦ (ᓯᖃᓕᑦᑎᕕᖕᒥ) ᑲᑎᑕᐅᔪᑦ ᐃᓂᖃᖅᑐᑦ ᑕᐅᕙᓂ ᓄᓘᔾᔨᑉ ᑕᓯᑯᓗᐊᓂ ᐃᒻᒥᖅᓱᐃᕝᕕᓕᐊᖑᓯᒪᔪᒥ, ᓴᓗᒻᒪᖅᓴᖅᓯᒪᔪᑦ ᑭᓈᓗᐃᑦ ᑯᕕᓯᕝᕕᐅᕙᒃᑐᑦ ᓄᓘᔾᔨᓂ ᐊᒻᒪ ᓯᐊᒻᒪᒃᐸᓪᓕᐊᓂᖏᑦ ᑎᒃᑕᐅᕙᒃᑐᑦ ᐅᔪᕋᐃᑦ ᓴᖅᑭᑦᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒥ ᐱᓕᕆᓂᕐᓂ, ᐊᒻᒪ ᐅᔭᕋᖕᓂᐊᕐᕕᒥᖕᒫᑐᐃᓐᓇᖅᑐᑦ ᐊᐅᒃᐸᒃᑐᑦ ᑰᓕᓲᑦ.

ᐊᑐᖅᓱᒋᑦ ᑕᐃᒨᓲᑦ ᐊᒻᒪ ᒪᓕᒋᐊᖁᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᖃᓄᐃᑦᑑᓂᖓ 'ᐃ' ᐃᒪᕐᒧᑦ ᓚᐃᓴᓐᓯ ᓄᓇᕗᒻᒥ ᐃᒪᓕᕆᔨᒃᑯᑦ ᑲᑎᒪᔨᖏᓐᓂ (NWB), ᐹᕙᓛᓐᑯᑦ ᓴᖅᑭᑎᔭᕆᐊᖃᓚᐅᖅᑐᐃᑦ ᐊᒻᒪᓗ ᐊᑐᓕᖅᑎᓗᒋᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᓂᒃ (AEMP) ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ. ᑎᑭᐅᒪᕐᑯᓪᓗᒋᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᑦ (AEMP) ᑎᑭᑕᐅᓇᓱᒃᑐᓄᑦ, ᐹᕙᓛᓐᑯᑦ ᓴᖅᑭᑦᑎᓚᐅᖅᑐᐃ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᓂᒃ (CREMP) ᖃᐅᔨᓴᕈᒪᓪᓗᑎᒃ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᓂᒃ ᐊᒃᑐᐃᓂᕐᓂᒃ ᐃᒪᐅᒃ ᖃᓄᐃᓐᓂᖓᓄᑦ, ᐃᓱᕐᓂᖓᑕ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ, ᐊᒻᒪᓗ ᐃᒪᕐᒥᐅᑕᑦ ᐆᒪᔪᑦ (ᖁᐱᕐᕈᕋᕐᖠᑦ, ᐃᖅᑲᖓᓂᕐᒥᐅᑕᑦ ᕿᒥᕐᕈᒃᑲᖏᑦᑐᑦ ᐊᒻᒪᓗ ᐃᖅᑲᓗᐃᑦ). ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᑐᕋᖓᖃᖅᑐᑦ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂᒃ, ᐃᓚᐅᓪᓗᑎᒃ ᓇᔪᒐᕐᒥ ᑕᓯᕐᓂ (ᓇᔪᒐᕐᒥ ᑕᓯᖅ ᐊᒻᒪᓗ ᓇᔪᒐᕐᒥ ᑕᓯᖅ ᑰᑕᐅᔪᑦ 1 ᐊᒻᒪᓗ 2), ᓄᓘᔾᔨᒃ ᑕᓯᑯᓗᐊ (ᓄᓘᔾᔨᑉ ᑕᓯᑯᓗᐊ ᐅᐊᖕᓇᖓᔾᔨᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ, ᓄᓘᔾᔨᑉ ᑕᓯᑯᓗᐊ ᓂᒋᖅᐸᓯᖕᒥ ᑲᓇᖕᓇᖓᔾᔨᒥ ᐊᒻᒪᓗ ᓄᓘᔾᔨ ᑕᓯᑯᓗᐊ ᑰᑕᐅᔪᖅ 1, 9, ᐊᒻᒪᓗ 12), ᓄᓘᔾᔨᒃ ᑰᖓ (ᐃᓚᐅᓪᓗᓂ ᓄᓘᔾᔨᒃ ᑰᖓᓂ ᑰᑐᖅ ᒥ), ᐊᒻᒪᓗ ᓄᓘᔾᔨᖕᓂ ᑕᓯᖅ. ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᓂᐅᕈᑎᖃᕐᓂᕐᒧᑦ ᐊᐅᓚᓂᖓ ᐱᒋᐊᓚᐅᖅᓯᒪᑎᓪᓗᒍ, ᐊᕐᕌᒍᑕᒫᖅ ᖃᐅᔨᓴᖅᑕᐅᓂᖅ ᐊᑐᖅᓱᒋᑦ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᐊᑐᖃᑦᑕᖅᓯᒪᔪᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᓪᓚᑦᑖᖅ ᓄᖅᑲᖓᕝᕕᐅᕚᒃᑐᑦ ᐃᒪᐅᑉ ᐊᒻᒪᓗ ᐃᓱᕐᓂᖓᑕ ᖃᓄᐃᑦᑑᓂᖓ ᐱᕚᓕᐊᑎᑕᐅᓯᒪᔪᖅ ᑕᒪᑐᒧᖓ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᒧᑦ (AEMP), ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᐊᑐᖅᑕᐅᓚᔪᒃᑐᑦ ᐊᕙᑕᒥᒃ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ (EEM) ᐱᓕᕆᔾᔪᓯᕐᓄᑦ. ᐊᕐᕌᒍᑕᒫᖅ ᖃᐅᔨᓴᐅᔪᑦ ᐊᑐᑕᐅᓯᒪᔪᑦ ᐃᓗᐊᓂ ᑎᓴᒪᐃᓕᖅᑲᖓᔪᑦ-ᐱᓕᕆᕙᓪᓕᐊᓴᐅᔪᑦ ᕿᒥᕐᕈᓂᕐᒧᑦ ᐊᑐᖅᑕᐅᓇᓱᐊᖅᑐᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᒪᔾᔪᑎᓄᑦ ᐋᖅᑭᒃᓯᒪᓂᑦ ᓴᓇᔭᐅᓯᒪᔪᑦ ᓄᓘᔾᔨᖕᓂ ᐱᓕᕆᐊᕐᒥ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᓄᑦ (AEMP) ᖃᐅᔨᒪᔾᔪᑕᐅᓗᑎᒃ ᐊᐅᓚᑦᑎᔨᓄᑦ

ᑲᒪᓇᓱᐊᖅᑎᓪᓗᒋᑦ ᐃᓱᒪᓕᐅᕈᑎᓄᑦ ᐊᒃᑐᐊᔪᓂ ᐊᓯᔾᔨᐅᑎᓄᓪᓗ ᑭᒡᓕᐅᔪᓄᑦ ᐱᑕᖃᕐᓂᐅᔪᓂ ᐊᒻᒪᓗ/ᐅᕝᕙᓘᓐᓃᑦ ᐃᒫᓂ ᐆᒪᔪᑦ ᐱᔾᔪᑕᐅᑐᐃᓐᓇᕆᐊᓖᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐊᐅᓚᓂᕐᓂ.

2024-ᖑᑎᓪᓗᒍ, ᓄᓘᔮᖕᓂ ᐱᓕᕆᐊᒥ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᓇᓗᓇᐃᔭᐃᔪᖅ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᓕᖕᓂᒃ ᐱᕈᖅᑐᓄᑦ ᐊᒻᒪᓗ ᐅᒪᔪᕋᓛᕐᓄᑦ ᓇᔪᒐᕐᒥ ᑕᓯᕐᒥ. ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᐅᔾᔨᕆᔭᐅᔪᑦ ᓇᔪᒐᕐᒥ ᑕᓯᖅ 1 (CLT1) ᑰᓪᕕᐅᓪᓗᐊᑕᖅᑐᓪᓗᐊᑖᑦ, ᐱᓗᐊᖅᑐᒥᒃ ᑕᑉᐸᓃᓐᓂᖅᓴᑦ ᑰᓪᕕᐅᓪᓗᐊᑕᖅᑐᓪᓗᐊᑖᑦ (ᐱᓕᕆᕕᒃ L2-03), ᐱᑕᖃᕐᕕᐅᔪᑦ ᐊᓘᒥᓇᒻ, ᓴᕙᒃᓴᖅ, ᓄᖑᓯᐃᑦᑐᖅ, ᐆᓇᖅᑐᑦ, ᑕᕆᐅᖅ, ᐊᒻᒪᓗ ᓴᕙᕋᔭᐅᔭᖅ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᓂᓖᑦ ᑐᙵᕙᖃᖅᑐᑦ ᐱᑕᖃᕐᕕᐅᔪᓂ ᐱᕙᓪᓕᐊᓯᒪᔪᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᑕᑯᓪᓗᒋᑦ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᐊᒻᒪ ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ 2024-ᒥ ᐱᕙᓪᓕᐊᓯᒪᓪᓗᑎᓪᓗ ᖃᓄᐃᓕᖓᓕᕐᓂᐅᕚᑦ / ᐃᓕᖅᑯᓯᐅᕚᑦ ᑕᐃᒪᙵᓂᑦ ᐱᒋᐊᕈᑕᐅᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᕈᑎᓕᕆᓐᓇᐅᓚᐅᖅᑐᒥ. ᑭᓯᐊᓂᓕ, ᐱᑕᖃᙱᑦᑐᑦ ᐱᕙᓪᓕᐊᓂᐅᔪᓂ ᖃᓄᐃᓕᖓᓕᕐᓂᐅᕙᒃᑐᑦ / ᐃᓕᖅᑯᓯᐅᕙᒃᑐᑦ ᑕᒪᒃᑯᓄᖓ ᑭᒡᓕᐅᔪᓄᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐊᐅᓚᓂᖃᖅᑎᓪᓗᖏᑦ ᐃᒪᐃᓐᓇᓱᒋᓐᓈᔪᖅ ᑕᒪᓐᓇ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᓖᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᒧᑦ ᐊᒃᑐᐃᓂᖃᖅᓯᒪᔪᑦ ᐱᕙᓪᓕᐊᕐᔪᐊᖅᓯᒪᙱᑦᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᓂᖅ ᐊᐅᓚᓂᖃᕈᓐᓇᖅᑎᓪᓗᒍ. ᑖᒃᑯᓇᓂ ᑭᒡᓕᐅᔪᓂ, ᐃᒫᓂ ᐱᑕᖃᕐᕕᐅᓪᓚᕆᒃᑐᑦ ᑲᑎᒍᒋᑦ ᐊᓘᒥᓇᒻ ᐊᒻᒪ ᓴᕙᒃᓴᖅ ᐅᖓᑕᐅᔾᔨᓯᒪᔪᑦ ᐃᓱᒪᒋᓪᓗᒋᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᑦ (AEMP) ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᓐᓄᑦ ᕿᒥᕐᕈᔾᔭᑕᐅᔪᓐᓇᖅᑐᑦ (ᐊᐅ.ᒃᑯᑦ ᐊᒻᒪ ᐅᑭᐊᒃᓵᒃᑯᑦ, ᐃᓱᒪᒋᓪᓗᒋᑦ). ᐱᑕᖃᕐᕕᐅᓪᓚᕆᒃᐸᓪᓕᐊᓂᖏᑦ, ᑕᐃᒪᙵᓂᑦ ᐱᒋᐊᕈᑕᐅᔪᑦ ᖃᐅᔨᓴᕈᑏᑦ ᐱᓕᕆᐊᖑᓂᕆᓚᐅᖅᑕᖏᓐᓂᒃ ᐊᒻᒪ ᐱᓕᕆᐊᖑᓕᖅᑎᓪᓗᒍ ᐅᔭᕋᖕᓂᐊᕐᕕᐅᑉ ᐊᐅᓚᓂᖓ, ᐅᔾᔨᕆᔭᐅᓯᒪᔪᑦ ᑲᑎᒍᒋᑦ ᐊᒻᒪ ᑕᖏᖅᑲᐳᓐᓂᖅᑐᑦ ᓄᖑᔪᐃᑦᑐᑦ ᐊᒻᒪ ᓇᓗᓇᐃᖅᓯᓪᓗᑎᒃ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ - ᐊᒃᑐᐊᓂᓖᑦ ᐊᒃᑐᐃᓃᑦ. ᒪᓕᒃᑐᓂᒃ ᐊᒃᑐᐃᓂᕐᓗᖕᓂᕐᓂᒃ ᑯᒪᕈᕐᓄᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐃᖅᑲᕐᒥᐅᑕᓄᑦ ᕿᒥᕐᓗᖅᑲᙱᑦᑐᓄᑦ ᐱᑕᖃᕐᕕᐅᔪᑦ (BIC) ᑕᐃᑲᓂ CLT1 ᑰᒃᑐᓪᓗᐊᑕᓂ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ. ᓇᔪᒐᕐᒥ ᑕᓯᖅ, ᓄᖑᔪᐃᑦᑐᑦ ᑖᓇᑐᐊᖑᓪᓗᑎᒃ ᐃᒫᓂ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑭᒡᓕᐅᔪᓂ ᐱᕙᓕᐊᕝᕕᐅᓯᒪᓪᓗᓂ ᐱᑕᖃᕈᓐᓂᖏᑦ ᐊᒃᑐᐊᓂᓖᑦ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᐊᒻᒪ ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ ᐱᕙᓪᓕᐊᓪᓚᕆᒃᑐᑦ ᒫᓐᓇᓚᐅᑲᒃ ᐃᓕᖅᑯᓯᐅᕚᑦ ᓇᓗᓇᐃᖅᓯᔪᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᓂᓖᑦ ᐊᒃᑐᐃᓃᑦ, ᐱᑕᖃᕐᓂᐅᒧᑦ ᓱᓕ ᑐᙳᓃᑦᑐᑦ ᑕᐃᑲᓂ WQG-ᒥ. ᐊᓯᖏᓐᓂᒃ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᓇᓗᓇᐃᖅᑕᐅᔪᖃᓚᐅᙱᑦᑐᑦ ᑕᐅᕙᓂ ᓇᔪᒐᕐᒥ ᑕᓯᕐᓂ.

ᖃᐅᔨᓴᐅᔪᑦ 2024-ᖑᑎᓪᓗᒍ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓃᑦ ᓇᔪᒐᐅᑉ ᑕᓯᖏᓐᓂ ᐱᑕᖃᕐᑲᐊᖅᑕᑎᑎᐅᖅ ᑲᒪᒃᓯᑎᓂᒃ ᐱᓕᕆᐊᒃᓴᓂᒃ ᐊᑐᖅᓯᒪᓂᑦ ᐃᒪᕐᒥᒃ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᓂᒃ (AEMP) ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᒪᔾᔨᕖᑦ ᐊᖅᑭᒃᓯᒪᓂᖏᑦ. ᐊᑐᓕᖅᑯᔾᔨᕖᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᑯᔨᓂᕐᒧᑦ BIC-ᓂᒃ ᑕᐃᑲᓂ CLT1-ᒥ ᑰᓕᓪᓗᐊᑕᖏᓐᓂ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᑯᓪᓗᒋᑦ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᓕᖕᓂᒃ ᐆᒪᔪᕋᓛᓂᒃ ᓴᖅᑭᓯᒪᔪᓂᒃ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᓂ ᐊᒃᑐᐃᔾᔪᑎᓂᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖏᓐᓄᑦ ᑭᒡᓕᐅᔪᓄᑦ. ᒃᑲᓐᓂᐊᒍᑦ, 2025-ᒥ, ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑭᒡᓕᐅᔪᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐅᕝᕙᓘᓐᓃᑦ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᓖᑦ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ, ᒫᓐᓇᓚᐅᑲᒃ ᖃᓄᐃᓕᖓᓂᕐᓂᐅᔪᒃ ᕿᒥᕐᕈᓂᐊᖅᑕᖏᑦ ᖃᖅᑲᐅᔨᓴᖅᑕᐅᒋᐊᓕᒃᑐᑦ ᒫᓐᓇᓚᐅᑲᒃ ᐃᓕᖅᑯᓯᐅᕚᑦ / ᐱᐅᓯᐅᕚᑦ, ᑲᑎᒍᒋᑦ ᖃᓄᐃᓕᖓᓂᕐᓂᐅᔪᓂᒃ ᖃᐅᔨᓴᕐᓗᒋᑦ ᑕᖏᖅᑲᐳᓐᓂᖅᐸᒃᑐᑦ ᐱᑕᖃᕐᓂᖏᑦ ᓴᕙᒃᓴᐅᔪᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᓂᐊᕐᒪᑕ ᖃᐅᔨᓴᕐᓗᑎᒃ ᐅᒪᔪᕕᐊᓂᒃ ᐊᐅᐃᓇᐅᔪᓂᒃ ᐊᒻᒪ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᓕᖕᓂᒃ ᐃᒫᓂ ᐆᒪᔪᕋᓛᓂᒃ,

ᐱᑕᖃᓕᕐᕕᐅᑐᐃᓐᓇᕆᐊᓕᖕᓂᓪᓗ ᐊᒃᑐᐃᓯᒪᔪᓂᒃ ᑕᓯᕐᓄᑦ / ᐃᒪᕐᓄᑦ ᓇᔪᒐᐅᑉ ᑕᓯᖃᕐᕕᖕᒋᓐᓂ ᖃᐅᔨᓴᖅᑕᐅᓗᑎᒃ ᑐᑭᓕᐅᖅᑕᐅᑦᑎᐊᒃᑯᓐᓂᖁᓪᓗᒋᑦ ᐅᓭᕆᖕᓂᐊᕐᕕᖕᒍᑦ-ᐊᒃᑐᐊᓂᓛᑦ ᐊᒃᑐᐃᓃᑦ ᐊᒻᒪ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᓛᑦ ᐱᕙᓪᓕᐊᑎᑦᑏᓐᓇᖅᑐᑦ. ᑭᖑᓪᓕᖅᐹᒥᒃ, ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖏᑦ ᐃᒪᕐᒍᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒍᑦ ᐱᓕᕆᐊᑦ (AEMP) ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ ᓄᖑᔪᐃᑦᑐᓄᑦ ᐃᓱᒪᒋᖃᓯᐅᔾᔭᐅᓂᐊᖅᑐᑦ ᐃᑲᔪᖅᑕᐅᕐᑯᓪᓗᒋᑦ ᖃᐅᔨᓴᕐᓃᑦ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᓕᖏᓄᑦ ᐆᒪᔪᕈᔪᖕᓄᑦ ᐅᔾᔨᕆᔭᐅᔪᓂᑦ ᐱᑕᖃᕐᓂᓪᓚᕆᐅᔪᓂᑦ.

2024-ᒥ, ᑕᐅᕙᓂ ᓄᓘᔾᔮᑉ ᑕᓯᑯᓗᖃᕐᕕᐊᓂ, ᓄᓘᔾᔮᑉ ᑕᓯᑯᓗᐊᑕ ᑰᒪᓚᖓᓂ 1 (SDLT1) ᓴᖅᑮᔾᔭᕐᕕᐅᓚᐅᖅᑐᑦ ᓇᓗᓇᖅᑭᓪᓚᑦᑖᖅᑐᓂᒃ ᐅᔾᔪᕐᓂᐊᕐᕕᖕᒍᑦ-ᐊᒃᑐᐊᔪᓂᒃ ᐊᒃᑐᐃᓂᕐᓂᒃ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑕᐅᕗᓂᓗᒃᑖᖅ ᑯᕕᓯᕙᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒍᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᕕᐅᔪᒥ. ᐅᔾᔪᕐᓂᐊᕐᓂᕐᒍᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᐊᒥᓱᓚᖓᓄᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖏᓐᓄᑦ ᑮᓕᐅᔪᓄᑦ ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᓴᕕᕋᔭᐅᔭᖅ, ᓴᕕᕋᔭᒃ, ᓴᐅᓂᒃᓴᖅ, ᑕᕆᐅᓕᒃ, ᓴᕕᕋᔭᒃᓴᖅ, ᐆᒻᒪᖅᑯᑎᖃᕐᑐᑦ, ᑕᕆᐅᒃᓭᑦ, ᒪᒃᓃᓯᐊᒻ, ᒪᒐᓃᔅ, ᓴᕕᕋᔭᐅᔭᖅ, ᑕᕆᐅᓕᒃ, ᐳᑖᓯᐊᒻ, ᓯᓕᓂᐊᒻ, ᑕᕆᐅᖅ, ᓴᕕᕋᔭᐅᔭᖅ, ᑎᒃᔭᐅᖅᑐᓗᒃᑖᑦ ᑕᖕᒋᓛᑦ (TDS), ᑲᑎᕝᓱᒋᑦ ᐆᓇᖅᑐᖃᖅᑐᓂᒃ ᐆᒃᑐᕋᐅᑎ ᖃᐅᔨᓴᒐᒃᓴᓂ (TKN), ᐊᒻᒪ ᓄᖑᓯᐃᑦᑐᖃᖅᑐᑦ ᐱᑕᖃᕐᓂᐅᔪᑦ ᐱᕙᓪᓕᐊᓪᓚᓇᒃᓯᒪᔪᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖕᒋᑦ ᑕᑯᓪᓗᒋᑦ ᐱᒋᐊᕈᑕᐅᓯᒪᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓂᒃ ᐊᒻᒪ ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ ᐊᒻᒪᓗ ᐱᕙᓪᓕᐊᓪᓚᓇᒃᓯᒪᔪᑦ ᐃᓕᖅᑯᓯᐅᕙᒃᑐᑦ / ᐱᐅᓯᐅᕙᒃᑐᑦ, ᐱᓗᐊᖅᑐᒥᒃ 2022-ᒥᑦ 2024-ᒧᑦ. ᑖᒃᑯᓇᓂ ᑭᓪᓕᐅᔪᑦ, ᐊᑕᐅᓯᑐᐊᖅ ᐃᒪᕐᓂ ᐱᑕᖃᕐᕕᐅᓪᓚᓇᒃᑐᑦ ᑲᑎᕝᓱᒋᑦ ᓴᕕᕋᔭᒃᓴᖅ ᐅᖓᑕᐅᔾᔨᓯᒪᔪᖅ ᑖᒃᑯᓇᓂ ᐃᒪᕐᒍᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒍᑦ ᐱᓕᕆᐊᑦ (AEMP) ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᓂ (ᐊᐅᔭᒃᑯᑦ ᐅᑭᐅᒃᓵᒃᑯᓪᓗ). ᐅᔾᔪᕐᓂᐊᕐᕕᖕᒍᑦ-ᐊᒃᑐᐊᓂᓛᑦ ᐊᒃᑐᐃᓃᑦ ᑕᐃᑲᓂ SDLT1-ᒥ ᐊᒃᑐᐊᔪᒃᓴᐅᔪᑦ ᐱᓕᕆᕝᕕᐅᕝᔪᐊᕐᓂᖓᓄᑦ ᐅᔾᔪᕐᓂᐊᕐᕕᖕᒥ ᐱᕐᑯᑎᕝᔪᐊᓂ ᖃᓂᒋᔭᖓᓂ SDLT1 ᐃᒻᒥᖕᐸᒃᑐᓂ ᐃᓂᓂ, ᐱᓗᐊᖅᑐᒥᒃ ᐅᔾᔪᕐᓂᐊᕐᕕᖕᒥ ᐃᒪᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑕᐅᕚᓇ ᑭᓵᒥᑕ 105-ᒥ ᑕᓯᕐᓂᒃ ᐃᒪᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑕᓯᖁᐅᔪᒥ (ᑭᓵᒥᑕ 105 ᑕᓯᖁᖅ).

ᑕᐃᒪᖓᓂ ᐊᖏᖅᑕᐅᓯᒪᔪᓂᑦ 2022ᒥᓂ, ᑕᓯᖅ ᐱᓕᕆᔭᐅᓚᐅᖏᑦᑐᖅ ᐃᓱᓕᓚᐅᕐᒪᑦ, ᑯᑐᖕᓂᐅᔪᒥᑦ ᐊᒻᒪᓗ ᐃᒪᐅᑦ ᖃᓄᐃᓐᓂᖓᓂ ᐊᒻᒪᓗ ᐊᒥᓱᑦ ᐊᒃᑐᐃᓂᐅᔪᑦ. ᐅᔾᔪᕐᓂᐊᕐᓂᕐᒍᑦ ᐊᒃᑐᖅᑕᐅᔪᑦ ᐅᑯᓂᖓ BIC ᑕᐃᑲᓂ SDLT1 ᖃᐅᔨᔭᐅᓚᐅᖅᑐᑦ. ᖃᐅᔨᔭᐅᓯᒪᔪᓂᑦ ᖃᓄᐃᓕᐅᕆᐊᖃᕐᓂᖏᑦ ᓄᓇᒥᐅᑖᓪᓚᑦᑕ ᐊᒻᒪᓗ ᐊᔾᔨᒌᖏᑦᑐᑦ ᖃᓄᐃᓕᐅᕐᓂᐅᔪᑦ ᖃᓄᐃᓐᓂᖏᑦ ᓴᕕᒃᓴᒧᑦ ᓱᕈᖅᑕᐅᖏᓐᓗᑎ ᖃᐅᔨᓴᐅᑎᓪᓗᐊᑕᕐᒧᑦ. ᐊᒃᑐᐃᓂᐅᔪᑦ ᐊᒃᑐᐊᔪᑦ ᐃᒪᕐᒧᑦ ᐊᐅᓚᑕᐅᔪᒥᑦ ᐊᒻᒪᓗ ᐊᑐᓕᕐᑯᔭᐅᔪᑦ ᐊᒃᑐᐃᓂᐅᔪᑦ ᑕᐃᑲᓂ ᑭᓵᒥᑐ 105 ᑕᓯᒃᒥ ᐱᓕᕆᓂᐅᖏᓐᓇᖅᑐᑦ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ ᐊᕕᒃᑐᖅᓯᒪᔪᓂᑦ SDLT1 BIC 2024ᒥᑦ. ᑕᐃᒪᙵᓂᑦ ᐅᔾᔪᕐᓂᐊᕈᓐᓇᖅᓯᓚᐅᕐᓂᖏᓐᓂᑦ 2022-ᒥ, ᑕᓯᖁᖅ ᑕᐃᒪᐃᖃᑦᑕᖅᓯᒪᙱᑦᑐᖅ ᓂᕆᐅᒋᔭᐅᔪᑦ, ᑎᑭᐅᑎᓪᓗᓂ ᖁᕐᑳᖏᓐᓇᖃᑦᑕᖅᓱᓂ ᐊᒻᒪ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᐊᔪᕐᖁᑎᖃᕐᐸᒃᓱᓂ ᐊᒻᒪ ᐊᒥᓱᐊᖅᑎᑦ ᓄᓇᐅᑉ ᑕᐅᑦᑐᓇᓚᐅᖅᑕᖓᓂ ᐅᑎᖅᑎᑦᑎᓇᓱᖃᑦᑕᖁᓱᐊᖅᑎᓪᓗᒋᑦ. ᐅᔾᔪᕐᓂᐊᕐᕕᖕᒍᑦ-ᐊᒃᑐᐊᓂᓛᑦ ᐊᒃᑐᐃᓃᑦ BIC-ᒥ ᑕᐃᑲᓂ SDLT 1-ᒥ ᖃᐅᔨᔭᐅᓯᒪᔪᑦ, ᖃᐅᔨᔭᐅᔪᓪᓗ ᐃᒪᓐᓇᐅᓇᓱᒋᔭᐅᓪᓗᑎᒃ ᐱᔾᔪᑎᖃᕐᓗᑎᒃ ᐆᒪᔪᕕᓂᕐᓂᒃ ᐱᐅᓯᕆᐊᕈᑎᓄᑦ ᐊᒻᒪ ᐊᔾᔨᒌᒃᑕᙱᓐᓂᖓᓄᑦ ᓄᓇᖓᓂ ᓇᔪᒐᐃᑦ ᖃᓄᐃᓕᖓᓂᖏᑦ ᐃᒪᐃᓐᓇᐅᙱᖔᖅᓱᓂ ᑕᕆᐅᔾᔭᒃᓄᑦ ᓱᕈᕐᓇᖅᑐᑦ ᐱᔾᔪᑕᐅᓪᓗᐊᑕᙱᓱᑎᒃ. ᐊᒃᑐᐃᓃᑦ ᐊᒃᑐᐊᓂᖃᖅᑐᑦ ᐅᔾᔪᕐᓂᐊᕐᕕᖕᒥ ᐃᒪᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᐊᒻᒪ ᓄᓇᖓᑕ ᑕᐅᑦᑐᓇᓚᐅᖅᑕᖓᓄᑦ ᐅᑎᑦᑎᑦᑎᓇᓱᐊᕐᓃᑦ ᑭᓵᒥ 105-ᒥ ᑕᓯᖁᕐᒥ ᒪᓕᑐᐃᓐᓇᖅᑐᑦ ᐱᔾᔪᑕᐅᔪᓄᑦ ᓴᖅᑭᓯᒪᑐᐃᓐᓇᕆᐊᓕᖕᓄᑦ ᐊᓯᔾᔨᕐᓂᕐᒧᑦ SDLT1 BIC-ᓄᑦ 2024-ᒥ.

ᐃᒪᐅᑦ ᖃᓄᐃᓐᓂᖓ ᓄᓘᔾᔮᒃ ᑕᓯᑯᓗᐊᓂ ᑯᒃᑐᒥ 9-ᒥ (SDLT9) ᐊᒃᑐᖅᑕᐅᓯᒪᖕᒥᔪᖅ ᐅᔾᔪᕐᓂᐊᕐᓂᕐᒍᑦ ᐱᓕᕆᓂᕐᓂ, ᓴᖅᑮᓪᓗᓂ ᐱᕙᓪᓕᐊᓯᒪᓂᖕᓴᐅᔪᓂ ᐆᓇᖅᑐᖃᖅᑐᓄᑦ-ᐊᒃᑐᐊᔪᑦ ᐱᑕᖃᕐᕕᐅᕙᓪᓚᒃᑐᑦ (ᐊᒨᓂᐊ, ᐆᓇᖅᑐᑦ, ᐆᓇᖅᑐᓛᑦ, ᐊᒻᒪ ᑲᑎᕝᓱᒋᑦ ᐆᓇᖅᑐᖃᖅᑐᓂᒃ ᐆᒃᑐᕋᐅᑎ ᖃᐅᔨᓴᒐᒃᓴᓂ (TKN)) ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ 2023-ᒥ ᑯᕕᓯᕙᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ

ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑐᓂ (CREMP) (ᒥᓈ 2024ᐃ) ᐊᒻᒪᓗᒃᑲᓐᓂᖅ 2024-ᒥ. ᐊᔾᔨᐅᙱᑦᑐᖅ ᖃᐅᔨᓴᕐᓂᖅ, ᐱᖃᓱᐅᔾᔨᓱᒪᓪᓗᓂ ᐊᖏᓂᖅᓴᒥᒃ ᐊᖏᔪᒻᒪᕆᐊᓗᖕᒥ ᖃᐅᔨᓴᒐᒃᓴᓕᐅᓂᕐᒥᒃ ᐊᒻᒪ ᐱᔭᕇᖅᑕᐅᓯᒪᓪᓗᓂ ᐅᑭᐊᒃᖠᖑᓂ 2024-ᒥ ᓇᓗᓇᐃᖅᓯᓯᒪᔪᑦ ᐱᓕᕆᓃᑦ ᐱᓕᕆᐊᖑᓯᒪᔪᑦ ᑕᐃᓈ ᓅᐳ ᐊᕕᒃᑐᐃᑉᕕᖓᓂ (ᑕᐃᓅᒃᑯᑦ ᐱᓕᕆᑉᕕᖓ), ᑐᖅᑯᐃᓯᒪᔩᓐᓇᖅᑐᑦ ᐋᒨᓂᐊ ᐆᓐᓇᖅᑐᓕᖕᓂᒃ ᐅᔭᖃᖕᓂᐊᕐᕕᖕᒥ ᐊᒻᒪ ᐃᓂᖃᖅᓱᓂ ᖃᓂᑦᔭᖓᓂ SDLT9, ᐱᑉᕕᐅᓪᓗᐊᑕᔫᖅ ᑖᒃᑯᓇᙶᑦ ᐱᑕᖃᖅᑐᓂ. ᐊᒃᑐᐃᓂᕐᓗᖕᓂᖅᑕᖃᙱᑦᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᓂᓕᖕᓄᑦ ᐊᒃᑐᐃᓂᕐᓂᒃ ᐱᕈᖅᑐᕋᓖᓄᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᑭᓯᐊᓂᓕ ᐆᒪᔪᑦ ᓇᔪᖓᖏᑕ ᓯᓚᑎᖏᑕᓗ ᓇᔪᖓᖏᓐᓂ ᑐᑭᑖᖅᑎᑕᐅᑦᑎᐊᖅᓯᒪᓪᓗᑎᒃ ᐊᔾᔨᒌᙱᓂᑦ BIC ᐋᖅᑭᒃᓯᒪᓂᖏᓐᓂ ᑕᐃᑲᓂ SDLT9-ᒥ 20204-ᖑᑎᓪᓗᒍ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ (ᑭᐊᓯᓂᓕ BIC ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᓐᓄᑦ ᖃᐅᔨᒋᐊᕐᓂᖅ ᑖᒃᑯᓄᖓ ᖃᐅᔨᒋᐊᕐᕕᐅᔩᓐᓇᖅᑐᓄᑦ ᑰᒪᖏᖕᓄᑦ). ᐊᑐᖅᑕᐅᓪᓗᑎᒃ ᓄᓇᖓᓂ ᑕᐃᒪᐃᓕᖓᕙᒃᑐᑦ ᐊᕿᒍᒥᑦ-ᐊᕿᒍᒧᑦ ᐊᔾᔩᒃᑕᖃᑦᑕᙱᑦᑐᑦ ᓇᔪᒪᐃᑦ ᖃᓄᐃᓕᖓᓂᖏᓐᓄᑦ ᐃᓱᒪᒋᔭᐅᑐᐃᓐᓇᕆᐊᓕᒃ ᐊᓯᔾᔨᕐᓂᕐᓄᑦ BIC-ᓂ ᐊᒃᑐᐊᓃᑦ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ, ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᓃᑦ ᐊᒃᑐᐃᓃᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑕᐃᑲᓂ SDLT9-ᒥ 2024-ᒥ ᐃᒪᐃᓐᓇᓱᒋᓈᔪᑦ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᖃᕐᓂᖓᓂᒃ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᓂᓕᖕᓂ.

ᑕᐃᑲᓂ ᓄᓘᔭ ᑕᓯᑯᓗᐊᓂ ᑰᒪᖏᓂ 12 (SDLT12) 2024-ᒥ, ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᓂᖃᑐᐃᓐᓇᕆᐊᖏᑦ ᐊᒃᑐᐃᓃᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑭᒡᓕᐅᔪᓄᑦ ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᑕᕆᐅᖑᓂᖓ, ᑕᕆᐅᕈᔪᒃ, ᓴᐅᓂᒃᓴᖅ, ᑕᕆᐅᕈᔪᒃ, ᐆᒻᒪᖅᑯᑎᖃᕐᓂᖓ, ᑎᓯᓂᖓ, ᒪᔾᓃᓯᐊᒻ, ᓴᕕᕋᔭᒃᓴᖅ, ᐳᑖᓯᐊᒻ ᑕᕆᐅᖅ, ᓴᕕᕋᔭᐅᔭᖅ, ᑎᒃᔭᐅᖅᑐᒃᑖᑦ ᑕᖏᓖᑦ (TDS) ᐊᒻᒪ ᓄᖑᔪᐃᑦᑐᑦ ᐱᔭᐅᔪᑦ ᐅᐱᕐᖠᒃᑯᑦ ᐱᑕᖃᕐᓂᐅᔪᓂ ᐱᕗᓪᓕᐊᓯᒪᔪᓂ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᑕᑯᓪᓗᒋᑦ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᐊᒻᒪ ᖃᐅᔨᒋᐊᕐᕕᐅᔩᓐᓇᖅᑐᑦ ᐱᕗᓪᓕᐊᓂᖅᓴᐅᔪᓄᑦ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᓄᑦ / ᐱᐅᓯᐅᓕᖅᐸᒃᑐᓄᑦ ᑕᐃᒪᙵᓂᑦ ᐱᒋᐊᖅᑕᐅᓚᐅᕐᓂᖏᓐᓂᒃ ᖃᐅᔨᓴᒐᒃᓴᓕᐅᕐᓃᑦ ᑕᑉᐱᓂ ᐃᓂᒥ 2021-ᒥ. ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ SDLT12-ᒥ ᐊᒃᑐᐊᓂᖃᖅᑐᒃᓴᐅᔪᑦ ᐊᐳᒻᒧᑦ ᑲᑎᖅᓯᐃᑉᕕᖕᒥ ᐱᓕᕆᓂᐅᕙᒃᑐᓂᒃ ᐊᒻᒪ ᓯᐊᒻᒪᒃᑉᕕᐅᖃᑦᑕᕐᓂᖓᓄᑦ ᐳᔪᕋᕐᓂᒃ (ᑕᐃᑲᖓᖅᑐᒃᓴᐅᔪᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᓯᖃᓪᑎᕝᕕᖕᒥ) ᐃᒻᒥᖅᐸᒃᑐᓂ ᑰᒪᖏᓐᓂᑦ ᑕᐃᑲᓂ SDLT12-ᒥ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᕕᐅᔪᒥ.

ᑕᒪᕐᒥᖕᓂ ᓄᓘᔭᖕᓂ ᑕᓯᑯᓗᐊᓂ (ᐅᐊᖕᓇᔮᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ[NW] ᐊᒻᒪᓗ ᓂᒋᖅᐸᓯᖕᒥ ᑲᓇᖕᓇᔮᒥ [SE]) 2024-ᒥ, ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ–ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᒪᑯᓇᖓ ᐳᓪᓚᖅ, ᑕᕆᐅᕈᔪᒃ, ᓴᕕᕋᔭᒃᓴᖅ ᐊᒻᒪᓗ ᓄᖑᔪᐃᑦᑐᖅ, ᐊᒻᒪᓗᑕᐅᖅ ᑕᕆᐅᖃᕐᓂᖓ ᓄᓘᔭᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔮᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥᑐᖅᐊᓂᑐᐊᖅ. ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᐱᔾᔪᑕᐅᓪᓗᑎᒃ ᐃᒫᓂ ᐱᑕᖃᕐᓂᐅᔪᑦ ᐊᒃᑐᐊᓃᑦ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᐊᒻᒪᓗ/ᐅᕝᕙᓘᓐᓃᑦ ᖃᐅᔨᒋᐊᕐᕕᐅᔩᓐᓇᖅᑐᑦ 2024-ᒥ ᐊᒻᒪᓗᑦᑕᐅᖅ ᓇᓗᓇᐃᕈᑎᖃᕐᓗᑎᒃ ᐱᕗᓪᓕᐊᓂᖏᓐᓂᒃ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᑦ / ᐱᐅᓯᐅᓕᖅᐸᒃᑐᑦ ᑭᒡᓕᐅᔪᑦ ᐱᑕᖃᕐᓂᐅᔪᓂ, ᑕᐃᒪᙵᓂᑦ 2018 / 2019-ᒥᑦ ᐊᒻᒪ ᐱᑕᖃᐃᓐᓇᖅᓱᑎᒃ 2024-ᒥ. ᑕᒪᒃᑯᐊ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᑦ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᖏᑦ ᐱᓕᕆᓂᕐᓂ ᐊᑐᖅᑕᐅᕙᒃᑐᑦ ᑕᐃᓅᒃᑯᑦ ᐱᓕᕆᕕᐊᓂ (ᓄᓘᔮᑉ ᑕᓯᑯᓗᐊᓂ ᓂᒋᖅᐸᓯᖕᒥ ᑲᓇᖕᓇᔮᒥᑐᐊᖅ), ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐃᒪᕐᒥᒃ ᐊᐅᓚᑦᑎᓂᒃᑯᑦ ᑕᐃᑰᓇ ᑭᓛᒥᑕ 106 ᑕᓯᕋᕐᒥ, ᐊᒻᒪ ᐊᖏᓂᖅᓴᖅ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒥ ᐱᕐᑯᑎᕐᔪᐊᓂ ᐃᒻᒥᖅᑳᖅᑐᑦ ᑕᐃᑲᓂ ᓄᓘᔮᒃ ᑕᓯᑯᓗᐊᓂ.

ᓄᓘᔮᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔮᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ, ᓈᓴᐅᑎᒋᒐᔫᒃᑕᖏᑦ ᓴᕕᒃᓴᐃᑦ ᐱᑕᖃᕐᓂᐅᓂᖏᓐᓄᑦ ᓯᒃᔭᖓᓂ ᐊᒻᒪ ᐃᑎᓂᖓᓂ ᐃᓱᕐᓃᑦ ᐅᖓᑕᐅᔾᔨᓯᒪᔪᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᓄᑦ (AEMP)

ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ, ᓇᓴᐅᓯᕆᓂᖅᑎᒍᑦ ᐱᕙᓪᓕᐊᓂᖃᕐᔪᐊᖅᑐᑦ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᑦ ᑕᒪᐃᓐᓂᒃ ᐱᒋᐊᕈᑕᐅᔪᓂᒃ ᖃᐅᔨᓴᕈᑎᓂᒃ ᐊᒻᒪ ᐅᔭᕋᙵᓂᐊᕐᕕᖕᒥ ᐊᐅᓚᓂᖃᓕᖅᑎᓪᓗᒍ. ᐊᖏᔫᒃᑯᑦ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᖅ ᓴᕕᒃᓴᖃᕐᓂᐅᔪᓂᑦ ᑕᓯᕐᒥ ᐅᔾᔨᕆᔭᐅᓚᐅᕐᒥᔪᑦ ᐃᒪᐃᓐᓇᓱᑦᔭᐅᓪᓗᑎᒃ ᓴᖅᑭᓐᓂᖏᑦ ᐅᔭᕋᙵᓂᐊᕐᕕᖕᒍᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐃᓱᕐᓂᖓᓚ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ, ᐊᒃᑐᐊᓂᖃᑐᐃᓐᓇᕆᐊᓖᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓗᑎᒃ ᐃᓱᕐᓃᑦ ᑰᓚᓖᖅᓂ. ᐅᓪᓗᒥᒧᑦ, ᐊᒃᑐᐃᓂᕐᓗᖕᓂᖅᑕᖃᖏᑦᑐᖅ-ᐅᔭᕋᙵᓂᐊᕐᕕᖏᑦ ᐆᒪᔪᕈᔪᙱᓄᑦ ᐊᒃᑐᐃᓃᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᓇᓕᐊᖕᓂᑐᐃᓐᓇᖅ ᓄᓘᔾᔫᒃ ᑕᓯᑯᓗᐊᓂ.

ᖃᐅᔨᔭᐅᔪᑦ 2024-ᒥ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᑎᒍᑦ (CREMP) ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒥ ᓄᓘᔾᔫᒃ ᑕᓯᑯᓗᐊᓂ ᑲᒪᒋᔭᐅᔭᕆᐊᓖᑦ ᐊᑐᖅᐸᒋᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᓄᑦ (AEMP) ᐊᐅᓚᑦᑎᔾᔪᑎᓄᑦ ᑲᒪᒋᔭᐅᔾᔪᑎᖏᓐᓄᑦ ᐋᖅᑭᒃᓯᒪᓂᐅᔪᑦ. ᐊᑐᓕᕐᑯᔭᐅᔾᔪᑏᑦ ᐋᖅᑭᒃᑕᐅᓯᒪᓚᐅᖅᑐᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᑯᓪᓗᒋᑦ BIC-ᓂᒃ ᑕᐃᑲᓂ SDLT1-ᒥ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᑯᓪᓗᒋᑦ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᓕᙵᓄᑦ ᐆᒪᔪᓄᑦ ᓴᖅᑭᓯᒪᔪᑦ ᐅᔭᕋᙵᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑭᒡᓕᐅᔪᓄᑦ. ᖁᑉᐹᓂᐊᒍᑦ, 2025-ᒥ, ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑭᒡᓕᐅᔪᑦ ᓇᓗᓇᐃᖅᑕᐅᔪᖃᓚᐅᖅᑐᑦ ᐅᔭᕋᙵᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᒧᑦ ᐅᕝᕙᓘᑉᓃᑦ ᐅᔭᕋᙵᓂᐊᕐᓂᕐᒧᑦ–ᐊᒃᑐᐊᓂᖃᑐᐃᓐᓇᕆᐊᓖᑦ ᐊᒃᑐᐃᓃᑦ ᑕᐃᑲᓂ SDLT1, SDLT12, ᓄᓘᔾᔫᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖓᓈᔾᔨ ᐱᓇᖓᖅᐸᓯᖓᒥ ᐊᒻᒪᓗ/ᐅᕝᕙᓘᓐᓃᑦ ᓄᓘᔾᔫᒃ ᑕᓯᑯᓗᐊᓂ ᓂᒋᖅᐸᓯᖓᒥ ᑲᓇᖓᓈᔾᔨ, ᒫᓐᓇᓚᐅᑲᒃ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᓂᒃ ᕿᒥᕐᕈᓂᑦ ᐱᓕᕆᐊᖑᓛᖅᑐᑦ ᖃᐅᔨᓴᖅᑕᐅᒃᑲᓂᐅᔭᕋᓱᐊᕐᓗᑎᒃ ᒫᓐᓇᓚᐅᑲᒃ ᐃᓕᖅᑯᓯᐅᓕᖅᐸᒃᑐᑦ / ᐱᐅᓯᐅᓕᖅᐸᒃᑐᑦ, ᑲᑎᙵᒋᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᙱᙲᑕ ᑕᖏᖃᕈᓐᓃᖅᓯᒪᔪᑦ ᐱᑕᖃᕐᓂᐅᔪᑦ ᓴᕋᔭᙱᓐᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᓂᐊᖅᑐᑦ ᕿᒥᕐᕈᔭᐅᕐᑯᓪᓗᒋᑦ ᐆᒪᔪᕕᓃᑦ ᐊᑐᐃᓐᓇᐅᔪᑦ ᐊᒻᒪ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᕆᐊᓖᑦ ᐃᒫᓂ ᐆᒪᔪᕋᓛᓄᑦ, ᐊᒻᒪ ᐱᑕᖃᓕᕐᕕᐅᑐᐃᓐᓇᕆᐊᓖᑦ ᐊᒃᑐᐃᔪᑦ ᑕᓯᕐᓂᒃ/ᑯᒃᐸᒃᑐᓂᓗ ᑕᐃᑲᓂ ᓄᓘᔾᔫᖅ ᑕᓯᑯᓗᐊᓂ ᖃᐅᔨᓴᖅᑕᐅᓂᐊᖅᑐᑦ ᑐᑭᑖᖅᑎᑕᐅᑦᑎᐊᕐᓂᖅᓴᐅᕐᑯᓪᓗᒋᑦ ᐅᔭᕋᙵᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᓂᓛᑦ ᐊᒃᑐᐃᓃᑦ ᐊᒻᒪ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᖃᕐᓂᖓᓂᒃ ᐱᕙᓪᓕᐊᑎᑕᐅᖏᓐᓇᕐᓂᕐᓄᑦ. ᐱᕙᓪᓕᐊᑎᑕᐅᓂᖏᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᓄᑦ (AEMP) ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᑦ ᓄᙳᔪᐃᑦᑐᓄᑦ ᐃᓱᒪᒋᖃᓯᐅᔾᔭᐅᓂᐊᖅᑐᑦ ᐊᒻᒪ ᐃᑲᔪᖅᑕᐅᓗᑎᒃ ᕿᒥᕐᕈᓂᕐᒧᑦ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᓖᑦ ᐆᒪᔪᕈᔪᙱᓄᑦ ᐊᒃᑐᐃᓃᑦ ᐅᔾᔨᕆᔭᐅᔪᓂ ᐱᑕᖃᕐᓂᐅᔪᓂ.

ᐸᓚᐅᒪᑎᑦᑎᓇᓱᐊᕐᓃᑦ ᐊᑐᓕᖅᑎᑕᐅᓂᐊᖅᑐᑦ ᐱᐅᓯᕆᐊᖅᑕᐅᕐᑯᓪᓗᒋᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖏᑦ ᓄᓘᔾᔫᒃ ᑕᓯᑯᓗᐊᑕ ᑰᓚᖓᓂᖏᓐᓂ ᐊᒻᒪ ᓄᓘᔾᔫᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖓᓈᔾᔨ ᐱᓇᖓᖅᐸᓯᖓᒥ ᐊᒻᒪ ᓂᒋᖅᐸᓯᖓᒥ ᑲᓇᖓᓈᔾᔨ. ᑭᓛᒥᑕ 105-ᒥ ᑕᓯᖓᓂ, ᑐᕌᓚᐅᔪᑦ ᓄᓇᖓᓚ ᑕᐅᑦᑐᕆᓚᐅᖅᑕᖓᓄᑦ ᐅᑎᖅᑎᓇᓱᐊᕐᓃᑦ 2025-ᒥ ᐱᕙᓪᓕᐊᓂᖅᓴᐅᓕᕐᓂᐊᖅᑐᑦ ᐱᐅᓯᕆᐊᖅᑕᐅᕐᑯᓪᓗᒋᑦ ᐃᓱᕐᓂᖓᓚ ᐊᐅᓚᑕᐅᔾᔪᑎᒃᓴᖏᓐᓄᑦ ᐱᓕᕆᔾᔪᓯᕐᓄᑦ, ᐃᓚᓕᐅᑎᓗᒋᑦ ᐃᓚᐅᕈᑎᑎᒍᑦ ᑲᒪᒋᔭᐅᖃᑦᑕᕐᓂᑦ, ᓴᓗᒻᒪᖅᓴᐅᑎᓃᑦ, ᐊᒻᒪ ᐱᐅᓯᒋᐊᖅᑕᐅᓃᑦ ᐋᖅᑭᒃᓯᒪᓂᖏᓐᓄᑦ ᐋᖅᑭᓯᒪᔪᑦ ᐃᒪᐃᙲᖏᙵᕐᓗᑎᒃ ᖁᒃᑲᓐᓂᐊᒍᑦ ᐋᖅᑭᒃᓯᒪᓂᖏᑦ ᐋᖅᑭᒋᐊᖅᑕᐅᓗᑎᒃ. ᐋᖅᑭᒃᑕᐅᓂᖏᑦ ᓱᕈᔭᐃᒃᑯᑏᑦ ᑕᑉᐸᓃ ᑰᒃᑐᒥ ᐃᒪᕐᒧᑦ ᓇᐃᒪᓐᓯᒧᑦ ᖃᐅᔨᓴᕋᑰᑎᓄᑦ ᑐᓴᐅᒪᖃᑦᑕᐅᑎᔾᔪᑎᓄᑦ ᐱᓕᕆᐊᓄᑦ (SNP) ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᖃᐅᔨᓴᕐᕖᑦ MS-C-D (ᐃᓂᖃᖅᑎᑦ ᑰᓚᖓᓂᖏᓐᓂ SDTL1 ᐱᒋᐊᖅᓯᒪᔪᖅ ᓂᒋᖅᐸᓯᐊᓂ ᑲᓇᖓᓈᔾᔪᖓᓂ ᐊᒻᒪ ᑰᒃᑯᓂ SDLT1 ᑖᒃᑯᓇᓂ ᖃᐅᔨᓴᕐᕕᒃ D1-05 ᐊᒻᒪ D1-00) ᐸᕐᓇᒃᑕᐅᓯᒪᖕᒥᔪᑦ 2025-ᒧᑦ. ᐱᔾᔪᑖ ᑕᒪᑐᒧᖓ ᖁᑉᐹᓂᐊᒍᑦ ᐱᕐᑯᑎᕐᔪᐊᑦ ᐸᓚᐅᒪᑎᑦᑎᓇᓱᒃᐹᓂᕐᓂᕐᒧᑦ ᐅᔭᕋᙵᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐱᕙᓪᓕᐊᑎᑦᑎᔪᑦ ᑲᑎᓗᒃᑖᖅᐳᒋᑦ ᑲᖓᑦᑕᖅᑐᑦ ᑕᖏᓃᑦ (TSS) SDLT1-ᒧᑦ ᐊᒃᑐᐊᔪᑦ ᐳᔪᕋᕐᒧᑦ ᐊᒻᒪ ᐊᓯᖏᑦ ᐱᑕᖃᕐᕕᐅᔪᑦ ᑲᑎᓗᒃᑖᖅᐳᒋᑦ ᑲᖓᑦᑕᖅᑐᑦ ᑕᖏᓃᑦ (TSS) ᑰᓚᖓᓂᑦ ᐃᒻᒥᕐᕕᐅᕙᒃᑐᑦ. ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑐᓴᐅᒪᔾᔪᑏᑦ ᑲᑎᑕᐅᓯᒪᔪᑦ ᐱᓕᕆᐊᖃᖅᑎᓪᓗᒋᑦ 2025-ᒥ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ

ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᓄᑦ (CREMP) ᐊᑐᖅᑕᐅᓂᐊᖅᑐᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓐᓇᖅᑕᐅᕐᑯᓪᓗᒋᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑕᐃᑲᓂ SDLT1 ᐊᒻᒪ ᓄᓘᔮᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔮᖓᓂ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ ᐊᒻᒪ ᓂᒋᖅᐸᓯᖕᒥ ᑲᓇᖕᓇᔮᒥ ᒪᓕᒃᑕᐅᓗᑎᒃ ᑐᑭᓯᑎᑕᐅᕐᑯᓪᓗᒋᑦ ᐱᑕᖃᕆᐊᖃᑐᐃᓐᓇᕆᐊᖃᕐᓂᖕᒋᓐᓂᒃ ᖃᐅᔨᓴᖅᑕᐅᒃᑲᓐᓂᐅᔭᕆᐊᖃᕐᓃᑦ ᐊᒻᒪ ᐸᓚᐅᒪᑎᑦᑎᓇᓱᒃᑲᓐᓂᐅᔭᕐᓃᑦ.

ᐱᓕᕆᓂᐅᔪᓂᒃ ᑕᒻᒪᖅᓯᒪᔪᓯᐅᕐᓂᖅ ᐃᓱᒪᒋᓪᓗᒋᑦ ᐅᓯᑲᖅᑕᕐᓃᑦ, ᑐᖅᑯᐃᓃᑦ, ᐊᒻᒪ ᑲᒪᒋᖕᒋᔭᐅᓂᖕᒋᑦ ᑕᕆᐅᕈᔪᐃᑦ ᐱᓕᕆᓂᕐᓂ ᐱᓕᕆᐊᖑᔪᑦ ᑕᐃᓅᒃᑯᑦ ᐱᓕᕆᕝᕕᖓᓂ ᐊᑐᓕᖅᑎᑕᐅᕙᓪᓕᐊᔪᑦ, ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᖄᒃᑲᓂᐊᒍᑦ ᐃᒪᕐᒥᒃ ᖃᐅᔨᓴᒐᒃᓴᓕᐅᕐᓃᑦ ᓯᑯᖃᖏᑎᓪᓗᒍ 2025-ᒥ, ᑕᐃᒪᐃᑦᑕᕆᐊᖃᕈᑎᒃ, ᐃᑲᔪᕐᓂᐊᕐᓗᓂ ᓇᓗᓇᐃᔭᖅᑕᐅᓗᑎᒃ ᐱᑕᖃᕐᕕᐅᔪᖅ(ᑦ) ᐃᒫᓂ ᑕᕆᐅᕈᔪᖃᖅᑐᑦ ᐃᓂᖕᒋᓐᓂᒃ. ᐸᓚᐅᒪᑎᑦᑎᓇᓲᑏᑦ ᐱᓕᕆᔾᔪᔫᑦ ᐱᕙᓪᓕᐊᑎᑕᐅᓯᒪᔪᑦ ᐱᔾᔪᑎᒋᓪᓗᒋᑦ ᖃᐅᔮᔭᐅᔪᑦ. ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓐᓇᕐᓂᕐᖅ SDLT9-ᒥ ᑲᔪᓯᓂᐊᖅᑐᑦ 2025 ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᕿᒥᕐᕈᔭᐅᕐᑯᓪᓗᒋᑦ ᐊᒃᑑᑎᖃᑦᑎᐊᕐᓂᖏᑦ ᐸᓚᐅᒪᑎᑦᑎᓇᓲᑏᑦ ᑕᐃᓅ ᐱᓕᕆᕝᕕᐊᓂ ᐱᑕᖃᓗᐊᕿᓐᓃᖅᑎᕐᑯᓪᓗᖏᑦ ᐱᑕᖃᕐᓂᐅᔪᑦ ᐃᒫᓂ ᑕᕆᐅᖃᖅᑐᓄ ᐃᓂᐅᔪᓂ. ᑕᒪᓐᓇ ᐸᓚᐅᒪᑎᑦᑎᓇᓱᖕᓂᖅ ᐃᑲᔪᖅᑕᐅᑐᐃᓐᓇᕆᐊᖃᖅᑐᑦ ᐊᖏᒡᓕᒋᐊᕐᓗᒋᑦ ᐊᖏᔪᑎᒍᑦ ᖃᐅᔨᓴᒐᒃᓴᓕᐅᕐᓂᖅ ᐅᑭᐊᒃᓵᖓᓂ 2025-ᒥ, ᑕᐃᒪᐃᑦᑕᕆᐊᖃᕈᑎᒃ ᕿᒥᕐᕈᔭᐅᔭᐃᓐᓇᐅᕐᑯᓪᓗᒋᑦ ᐸᓚᐅᒪᑎᑦᑎᓇᓲᑏᑦ ᐊᒃᑑᑎᖃᕐᓂᖏᑦ.

ᑭᖑᓪᓕᖅᐹᒥᒃ, ᒫᓐᓇᓚᐅᑲᒃ ᐃᓕᖅᑯᓯᐅᑉᑐᓂᒃ ᕿᒥᕐᕈᓃᑦ ᓴᕝᕚᓂᒃ ᐱᑕᖃᕐᕕᐅᓃᑦ ᓯᒡᔭᖓᓂ ᐊᒻᒪ ᐃᑎᓂᖓᓂ ᐃᓱᕐᓂᕐᓂ ᓄᓘᔮᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔮᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ ᐱᓕᕆᐊᖑᒃᑲᓐᓂᕐᓂᐊᖅᑐᑦ ᐃᓚᓕᐅᔾᔨᓯᒪᓗᑎᒃ ᓄᑖᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖕᒋᓐᓇᕐᓂᕐᒧᑦ ᖃᐅᔨᓴᕈᑎᕕᓂᕐᓂᒃ ᕿᒥᕐᕈᔭᐅᕐᑯᓪᓗᒋᑦ ᐃᒪᐃᙱᒃᑲᓗᐊᕐᒪᖔᑦ ᐱᕙᓪᓕᐊᓂᐅᔪᑦ ᐃᓕᖅᑯᓯᐅᑉᑐᑦ ᓇᓗᓇᐃᔭᖅᑕᐅᖕᒋᓐᓇᕐᓗᑎᒃ ᐊᒻᒪ ᐃᑲᔫᑎᖃᕐᓗᑎᒃ ᓇᓗᓇᐃᖅᑕᐅᓂᖓ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓂᑦ ᓴᕝᕙᖃᑦ ᐱᑕᖃᐅᕐᕕᖕᒋᓐᓂ ᐊᔾᔨᐸᓗᒋᓗᒋᑦ ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᓄᑦ ᐊᒻᒪ ᐱᒋᐊᕈᑕᐅᔪᑦ ᖃᐅᔨᓴᕈᑏᑦ ᖃᓄᐃᓚᖓᓂᖏᓐᓄᑦ 2024-ᒥ. ᓱᓕᒃᑲᓐᓂᖅ, ᐊᖏᔪᑎᒍᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᕿᒥᕐᕈᔭᐅᓂᖏᑦ ᑖᒃᑯᓇᓂ ᓴᕝᕙᖃᑦ ᐱᑕᖃᕐᕕᐅᔪᑦ ᐃᓱᕐᓂᕐᒥ ᐃᓗᐊᓂ ᑕᓯᐅᑉ ᐱᔭᓃᖅᑕᐅᓂᐊᖅᑐᑦ ᐃᑲᔪᖅᑕᐅᕐᑯᓪᓗᒋᑦ ᓇᓗᓇᐃᖅᓯᓃᑦ ᐊᒃᑐᐃᓂᕐᓂ ᑕᓯᕐᓂ ᑰᓪᓗᐊᑕᖅᑐᓂ ᐊᒻᒪ ᐱᑕᖃᓕᖅᐹᓪᓕᖅᑦᑕᕐᓂᖓ ᐃᓱᕐᓃᑦ ᓯᕝᕕᒃᓴᐃᑦ ᓄᓘᔮᒃ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔮᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ.

ᓄᓘᔮᒃ ᑰᖓᓂ ᐊᒻᒪᓗ ᓄᓘᔮᑉ ᑕᓯᖓᓂ, ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᓂᓛᑦ ᐊᒃᑐᐃᓃᑦ 2024-ᒥ ᑭᒡᓕᖅᑲᓚᐅᖅᑐᑦ ᐊᒥᓲᖏᑦᑐᓄᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᑭᒡᓕᐅᔪᓄᑦ, ᐱᖃᓯᐅᑎᓪᓗᒋᑦ ᑕᕆᐅᖅ, ᑕᕆᐅᕈᔪᑦ, ᐊᒻᒪ ᓯᓕᓂᐊᒻ ᓄᓘᔮᒃ ᑰᖓᓂ ᑰᒐᓛᒃᑐᑦ-ᒥ (MRTF). ᑕᐃᑲᓂ ᓄᓘᔮᒃ ᑰᖓᓂ ᑰᒐᓛᒃᑐᑦ ᒥ–ᒥ (MRTF), ᐃᒫᓂ ᐱᑕᖃᕐᕕᐅᓂᖏᑦ ᑕᕆᐅᖅ, ᑕᕆᐅᕈᔪᓪᓗ ᐱᕙᓪᓕᐊᓯᒪᓚᐅᖅᑐᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᕿᒥᕐᕈᖅᑐᒋᑦ ᐱᒋᐊᕈᑕᐅᔪᓄᑦ ᖃᐅᔨᓴᕈᑎᓄᑦ ᐊᒻᒪ ᖃᐅᔨᒋᐊᕐᕕᐅᔪᓐᓇᖅᑐᓄᑦ ᐊᑕᐅᓯᖅ ᐊᕐᕌᒍᑉ ᐃᓚᐃᓐᓇᖕᓚᓂ 2024-ᒥ ᐊᒻᒪ ᐱᑕᖃᕐᓂᐅᔪᑦ ᑖᒃᑯᓇᓂ ᑭᒡᓕᐅᔪᓂ ᓴᖅᑭᔮᖅᑎᑦᑎᓯᒪᔪᑦ ᐱᕙᓪᓕᐊᓯᒪᓪᓗᑎᒃ ᐃᓕᖅᑯᓯᐅᑉᑐᑦ / ᐱᐅᓯᐅᑉᑐᑦ ᐱᒋᐊᓚᐅᖅᓯᒪᔪᑦ 2019-ᒥ ᐊᒻᒪ 2017-ᒥ (ᑕᕆᐅᕐᒧᑦ ᐊᒻᒪ ᑕᕆᐅᕈᔪᖕᒧᑦ, ᐊᓯᒪᒥᓪᓗᒋᑦ) ᑭᓯᐊᓂᓕ ᒪᓕᑐᐃᓐᓇᖅᓯᒪᙱᑦᑐᑦ ᐊᑯᓂᐅᔪᒃᑯᑦ, ᐃᒪᐃᓐᓇᐅᖅᑰᔫᓪᑐᓂ ᐱᕙᓪᓕᐊᓪᓚᓇᒡᓯᒪᙱᑦᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᒃ ᐊᐅᓚᓂᖅᑲᐃᓐᓇᖅᑎᓪᓗᒍ. ᓯᓕᓂᐊᒻᒥᒃ ᐱᑕᖃᕐᓂᐅᔪᑦ ᑕᐃᑲᓂ ᓄᓘᔮᒃ ᑰᖓᓂ ᑰᒐᓛᒃᑐᑦ ᒥ–ᒥ (MRTF) ᑐᖕᓕᓂᓚᔪᖅᑲᑦᑕᖅᓯᒪᔪᑦ ᖃᐅᔨᓴᖅᖄᑦ ᐅᓂᒃᑲᓚᐅᕐᓂᕐᒧᑦ ᑭᒡᓕᖏᓐᓄᑦ (LTL) ᐊᒻᒪᓗ ᕿᒥᕐᕈᔭᐅᓂᖏᑦ ᒫᓐᓇᓚᐅᑲᒃ ᐱᐅᓯᐅᑉᑐᑦ ᓇᓗᓇᖅᑐᑦ

ᐊᓯᔾᔨᖅᑕᕐᓂᖅᑎᓐᓄᑦ ᖃᐅᔨᓴᕐᕖᑦ ᐅᓂᒃᑳᓕᐅᕐᓂᕐᒧᑦ ᑭᒡᓕᖅᑎᓐᓄᑦ (LTL), ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᓇᓗᓇᐃᕈᑏᑦ ᐃᒪᓐᓇᓯᑎᓐᓈᙱᑦᑐᑦ ᐱᑕᖃᑐᐃᓐᓇᑎᐊᖃᖅᑐᑦ ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᓯᓚᓂᐊᒻᒧᑦ ᑕᐅᕙᓂ ᓄᓘᔮᒃ ᑯᖓᓂ ᑰᒪᓛᒃᑐᑦ ᒥ−ᒥ (MRTF). ᑕᐃᒪᐃᒃᑲᓗᐊᖅᑎᓪᓗᒋᑦ ᐊᒃᑐᐃᓂᐅᑐᐃᓐᓇᑎᐊᓛᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ, ᐊᒃᑐᐃᓂᖅᑕᖅᓯᒪᙱᑦᑐᖅ ᑯᒪᕈᕐᓄᑦ ᑕᐃᑲᓂ ᓄᓘᔮᒃ ᑯᖓᓂ ᑰᒪᓛᒃᑐᑦ ᒥ−ᒥ (MRTF) ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᓂ. ᐅᔭᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐅᒃᑐᐃᓃᑦ ᑐᑎᐅᕐᒥᑦ, ᑕᑎᐅᕈᔪᖕᒥᑦ ᐱᑕᖃᕐᕕᐅᔪᑦ ᑕᐃᑲᓂ ᓄᓘᔮᒃ ᑯᖓᓂ ᑰᒪᓛᒃᑐᑦ ᒥ−ᒥ (MRTF) ᐊᒃᑐᐊᓂᖃᑐᐃᓐᓇᑎᐊᓛᑦ ᓴᓗᒪᖅᓴᖅᓯᒪᔪᒧᑦ ᐃᒥᕐᒧᑦ ᑯᕕᔭᐅᖃᑦᑕᖅᑐᒥᑦ (ᓃᕐᓗ MS-08-ᒥᑦ ᑭᖑᓪᓕᖅᐹᒥᒃ ᑯᕕᓯᕝᕕᐅᓃᖅ [FDP] ᑕᐃᑲᓂ ᓄᓘᔮᒃ ᑯᖓᓂ ᑰᒪᓛᒃᑐᑦ ᒥ−ᒥ (MRTF)). ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ, ᐃᓱᕐᓃᑦ ᖃᓄᐃᑦᑑᓂᖏᓐᓄᑦ, ᐅᕝᕙᓘᓐᓃᑦ ᐆᒪᔪᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ ᐊᓯᖏᓐᓂ ᓄᓘᔮᒃ ᑰᖓᓂ ᐅᕝᕙᓘᓐᓃᑦ ᓄᓘᔮᒃ ᑕᓯᖓᓂ.

ᖃᐅᔨᔭᐅᔪᑦ 2024-ᖑᑎᓪᓗᒍ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖅᑎᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑎᓂᑦ (CREMP) ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖅᑎᓐᓇᕐᓂᕐᒧᑦ ᓄᓘᔮᒃ ᑯᖓᓂ ᑰᒪᓛᒃᑐᑦ ᒥ−ᒥ (MRTF) ᑲᒪᔾᔪᑎᖃᕆᐊᖃᖅᑐᑦ ᐊᑐᖅᓱᒋᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖅᑎᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᑦ (AEMP) ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᒪᔾᔪᑏᑦ ᐋᖅᑭᒃᓯᒪᓂᖅᑎᓐᓂ. ᐊᑐᓕᕐᑯᔾᔪᑏᑦ ᐋᖅᑭᒃᑕᐅᓚᐅᖅᑐᑦ ᐱᖃᓯᐅᔾᔨᓗᑎᒃ ᒫᓐᓇᓚᐅᑉ ᐃᓕᖅᑯᓯᐅᕙᒃᑐᓂᒃ ᕿᒥᕐᕈᓂᕐᒥᒃ ᐃᒫᓂ ᐱᑕᖃᕐᓂᐅᔪᓂ ᑕᑎᐅᒥᒃ ᐊᒻᒪ ᑕᑎᐅᕈᔪᖕᒥᒃ ᐱᓕᕆᐊᖑᓗᑎᒃ 2025-ᒥ ᖃᐅᔨᓴᖅᑕᐅᒃᑲᓐᓂᐅᔭᕐᑯᓗᒋᑦ ᒫᓐᓇᓚᐅᑉ ᐃᓕᖅᑯᓯᐅᕙᒃᑐᑦ / ᐱᐅᓯᐅᕙᒃᑐᑦ. ᓱᓕᒃᑲᓐᓂᖅ, 2025-ᒥ, ᐊᔾᔨᐅᙱᑦᑐᒥᒃ ᖃᐅᔨᓴᕐᓃᑦ ᐱᓕᕆᐊᖑᓂᐊᖅᑐᑦ ᕿᒥᕐᕈᓗᑎᒃ ᐃᒥᕐᒥᒃ ᓴᓗᒪᖅᓴᖅᓯᒪᔪᒥᒃ ᐊᒻᒪ ᑯᕕᓯᕝᕕᐅᕙᒃᑐᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᖃᐅᔨᓴᕈᑎᕕᓃᑦ ᑲᑎᑕᐅᓚᔪᖃᑦᑕᖅᑐᑦ ᐃᓚᒋᔭᐅᓪᓗᑎᒃ ᓴᕕᕋᔭᖕᓄᑦ ᐊᒻᒪᓗ ᐅᔭᖅᑲᓂᒃ ᑎᓯᔪᓂᒃ ᐅᔭᕋᖕᓂᐊᕐᓂᕐᒧᑦ ᓴᓗᒪᖅᓴᖅᓯᒪᔪᑦ ᐃᒦᑦ ᒪᓕᒐᖏᑦ (MDMER) ᐱᑕᖃᓇᐊᓛᑦ MS-08-ᒧᑦ ᑭᖑᓪᓕᖅᐹᒥᒃ ᑯᕕᓯᕝᕕᐅᓃᖅ (FDP) ᕿᒥᕐᕈᔭᐅᕐᑯᓪᓗᒋᑦ ᐊᒃᑐᐃᓃᑦ MS-08-ᒥ ᑭᖑᓪᓕᖅᐹᒥᒃ ᑯᕕᓯᕝᕕᐅᓃᖅ (FDP) ᐱᑕᖃᓕᕐᕕᐅᑐᐃᓐᓇᑎᐊᖃᖅᑯᓂ ᑕᑎᐅᕐᒥᒃ ᐊᒻᒪ ᑕᑎᐅᕈᔪᖕᒥᒃ ᓄᓘᔮᒃ ᑯᖓᓂ ᑰᒪᓛᒃᑐᑦ ᒥ−ᒥ (MRTF).

ᖃᐅᔨᔭᐅᔪᓯᒪᔪᑦ 2024-ᒥ ᑯᕕᓯᕝᕕᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖅᑎᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᕿᒥᕐᕈᔭᐅᓪᓗᑎᒃ ᓇᓚᐅᑦᑖᖅᑕᐅᔪᓄᑦ ᐱᕐᔪᐊᕐᓂᖅᑎᓐᓄᑦ ᐊᒃᑐᐃᓃᑦ ᐋᖅᑭᒃᑕᐅᓯᒪᔪᑦ ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᑦ (FEIS) ᐱᓕᕆᐊᒧᑦ (ᐹᕙᓖᓐ 2012). ᐊᑕᖏᖅᖢᑎᑦ ᖃᓄᐃᓐᓂᖅᓴᐅᓂᖏᑦ ᕿᒥᕐᕈᓪᓗᑎᑦ ᐃᒪᐅᑉ ᖃᓄᐃᑦᑑᓂᖓᓄᑦ ᐊᒻᒪ ᐃᓱᕐᓃᑦ ᖃᓄᐃᑦᑑᓂᖏᓐᓄᑦ ᖃᐅᔨᓴᕈᑎᕕᓂᕐᓄᑦ ᓇᔪᒐᐅᑉ ᑕᓯᖓᓂ, ᓄᓘᔮᑉ ᑕᓯᑯᓗᐊᓂ, ᐊᒻᒪ ᓄᓘᔮᖕᓂ ᑑᑐᓂ 2024-ᒥ ᑖᒃᑯᓄᖓ ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᑦ (FEIS) ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᑦ ᓇᓗᓇᐃᖅᓯᔪᐃᓐᓇᐃᑦ ᐊᑕᖏᖅᖢᑎᑦ ᑭᒡᓕᐅᔪᑦ ᐱᑕᖃᕐᓂᐅᔪᓂ ᐃᓗᐊᓃᓯᒪᔪᑦ ᐊᒃᑐᐊᔪᑦ ᐊᖏᓂᖅᑎᓐᓄᑦ ᓈᓴᐅᓯᖅᑐᖅᑕᐅᓯᒪᔪᓄᑦ ᐱᕙᓪᓕᐊᓂᕐᔪᐊᓄᑦ. ᑕᒪᓐᓇ ᑐᑭᖃᕐᒥᔪᖅ ᑖᒃᑯᐊ ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᓂᑦ (FEIS) ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᑦ (ᐱᑕᖃᙱᓐᓂᖅᑎᓐᓄᑦ) ᐊᒃᑐᐃᓃᑦ ᑕᑎᐅᕐᒥᐅᑕᓄᑦ ᐃᖃᓗᐃᑦ ᖃᓄᐃᙱᑦᑎᐊᕐᓂᖅᑎᓐᓄᑦ ᐊᒻᒪ ᖃᓄᐃᓕᖓᓂᖅᑎᓐᓄᑦ ᑎᑭᐅᒪᓚᐅᖅᑐᑦ. ᑕᐃᒪᐃᓐᓂᖓᓄᑦ ᑕᑎᐅᕐᒥᐅᑕᑦ ᐃᖃᓗᐃᑦ ᖃᓄᐃᙱᑦᑎᐊᕐᓂᖅᑎᑦ ᐊᒻᒪ ᖃᓄᐃᓕᖓᓂᖅᑎᑦ ᓇᔪᒐᕐᒥ ᑕᓯᕐᒥ, ᓄᓘᔮᑉ ᑕᓯᑯᓗᐊᓂ, ᐊᒻᒪ ᓄᓘᔮᒃ ᑰᖓᓂ ᑕᓯᖓᓂᓗ ᒪᓕᒃᑐᑦ ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᓄᑦ ᐋᖅᑭᒃᑕᐅᓯᒪᔪᑦ ᑕᐃᑲᓂ ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᓂᑦ (FEIS) (ᐹᕙᓖᓐ 2012). ᐱᓕᕆᐊᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐃᓱᕐᓃᑦ ᑲᑎᒻᐸᓪᓕᐊᔪᑦ ᐱᑕᖃᓗᐊᕌᕐᓂᖏᑦ ᑐᙵᓃᑦᑐᑦ 1 ᒥᓚᒦᑕ/ᐊᕐᕋᒍᒧᑦ ᓂᕆᐅᒋᔭᐅᓯᒪᔪᖅ ᑕᐃᑲᓂ ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᓂᑦ (FEIS) ᓴᖅᑭᓪᓗᑎᑦ ᐱᓗᐊᙱᑦᑐᓂᒃ ᐊᒃᑐᐃᓂᕐᓂᒃ ᑐᕐᑯᑕᐅᔾᔪᑎᖏᓐᓄᑦ ᑕᑎᐅᕐᒥᐅᑕᑦ ᐃᖃᓗᐃᑦ. ᓃᖃᐃᒻᒪ ᐃᓱᕐᓂᖅᑖᖅᐸᓪᓕᐊᓂᖏᓐᓄᑦ ᓲᒃᑲᓕᓂᐅᔪᑦ ᐊᑐᖅᑐᓂ 2023-ᒥ 2024-ᒧᑦ ᑕᑎᐅᕐᒥᐅᑕᑦ ᐃᖃᓗᐃᑦ ᐃᑦᓕᖕᒋᑦ ᑐᙵᓃᓯᒪᓚᐅᕐᒪᑕ 1 ᒥᓚᒦᑕ/ᐊᕐᕋᒍᒧᑦ ᑕᐅᕙᓂ ᓄᓘᔮᑉ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔾᔮᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ, ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᓂᑦ (FEIS) ᓇᓚᐅᑦᑖᖅᑕᐅᓯᒪᔪᓂ ᑕᒪᒃᑯᓄᖓ

(ᐱᑕᖅᙱᓐᓂᖏᓐᓄᑦ) ᑐᖁᑕᐅᔾᔨᑎᓪᓚᑦᑖᖏᑦ ᑕᕆᐅᕐᒥᐅᑕᑦ ᐃᖃᓗᐃᑦ ᑎᑭᐅᒪᓚᐅᖅᑐᑦ (ᒥᓅ 2025). ᑕᐃᒪᐃᓐᓂᖓᓄᑦ, ᐊᒃᑐᐊᓪᓚᑦᑖᖅᑐᑦ ᐃᖃᓗᐃᑦ ᑐᖁᖃᑦᑕᕐᓂᖏᓐᓄᑦ ᓄᓘᔾ ᑕᓯᑯᓗᐊᓂ ᐅᐊᙵᔫᒥ ᐱᓇᖕᓇᖅᐸᓯᖏᒥ ᒪᓕᑐᐃᓐᓇᓚᐅᖅᑐᑦ ᓇᓚᐅᖅᑖᖅᑕᐅᓯᒪᔪᓄᑦ ᑭᖑᓪᓕᖅᐹᒥᒃ ᐊᕙᑎᒧᑦ ᐊᒃᑐᐃᓂᐅᓂᐊᖅᑐᑦ ᐅᓂᒃᑳᓂᑦ (FEIS) (ᐹᕙᓖ 2012).

ᐊᑕᖏᖅᕿᑦ, ᐊᖏᓛᖑᔪᑦ ᐅᔾᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᔪᑦ ᐊᒃᑐᐃᓃᑦ ᐅᔾᔨᕆᔭᐅᓯᒪᔪᑦ ᑕᐃᕙᓂ ᓄᓘᔾ ᑕᓯᑯᓗᐊᓂ, ᐃᒪᐃᑦ/ᑕᔩᓪᓗ ᕿᒥᕐᕈᔭᐅᓯᓚᔪᐃᓐᓇᐸᓗᐃᑦ ᑕᐃᑲᓂ ᑯᕙᓯᕝᕙᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᓴᖅᑮᖅᑎᑦᑎᓯᒪᔪᑦ ᐃᓚᖓᓂ ᐅᔾᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᔪᓂᒃ ᐊᒃᑐᐃᓂᕐᓂᒃ, ᐊᒃᑐᐃᓃᑦ ᐅᖓᑖᓅᖅᓯᒪᓪᓗᑎᒃ BIC ᑰᒪᓛᓂ ᓄᓘᔾ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔫᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ (ᓱᕐᓗ SDLT1). ᐊᒃᑐᐊᓂᖏᑦ ᑖᒃᑯᓇᓂ ᐅᔾᕋᖕᓂᐊᕐᓂᕐᒥᑦ ᐱᓕᕆᓂᕐᓂ ᐃᓗᐊᓂ ᓄᓘᔾ ᑕᓯᑯᓗᐊᓂ ᐅᐊᖕᓇᔫᒥ ᐱᓇᖕᓇᖅᐸᓯᖕᒥ ᐊᒻᒪ ᐅᔾᔨᕆᔭᐅᔪᑦ ᐊᓯᔾᔨᐅᑏᑦ ᓇᓗᓇᐃᖅᑕᐅᓯᒪᔪᑦ, ᐊᒻᒪ ᒪᓕᒃᑯᑎᒃ ᐸᓚᐅᒪᑎᑦᑎᓇᓲᑎᓄᑦ ᐊᑐᓕᖁᔾᔨᔾᔪᑎᓄᓪᓗ ᐊᑐᐃᓐᓇᕈᖅᑕᐅᓯᒪᔪᑦ. ᐃᓚᖏᑦ ᐅᔾᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᒧᑦ ᐊᒃᑐᐃᓃᑦ ᓇᓗᓇᐃᖅᑕᐅᓚᐅᖅᑐᑦ ᓇᔪᒐᕐᒥ ᑕᓯᕐᒥ ᐊᒻᒪ ᓄᓘᔾᖓᓂ ᑰᖓᓂ/ᑕᓯᖏᓐᓂ, ᑕᒪᒃᑯᐊ ᐊᒃᑐᐃᓃᑦ ᓄᓇᖓᓂᑐᐊᖑᖅᑰᔾᔪᑦ ᐊᒻᒪᓗ, ᑕᐃᑲᓂ ᓇᔪᒐᐅᑉ ᑕᓯᖓᓂ, ᐊᒃᑐᖅᑕᐅᓯᒪᑐᐃᓐᓇᕆᐊᓖᑦ ᑕᐃᒪᐃᑐᐃᓐᓇᖅᐸᒃᑐᑦ ᐊᓯᔾᔨᖅᑕᕐᓂᖏᓐᓄᑦ. ᐊᑐᓕᖅᑎᑕᐅᕙᓪᓕᐊᖏᓐᓇᕐᓂᖏᑦ ᐊᕐᕌᒍᑕᒫᑦ ᑯᕙᓯᕝᕙᐅᓪᓗᐊᑕᖅᐸᒃᑐᓂ ᐊᒃᑐᐃᓂᕐᓂᒃ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐸᕐᓇᐅᑏᑦ (CREMP) ᕿᒥᕐᕈᔭᐅᖏᓐᓇᕐᓂᐊᖅᑐᑦ ᐱᑕᖃᑐᐃᓐᓇᕆᐊᖃᕐᓂᖏᓐᓄᑦ ᐅᔾᕋᖕᓂᐊᕐᕕᖕᒧᑦ-ᐊᒃᑐᐊᓂᓛᑦ ᐊᒃᑐᐃᓃᑦ ᐊᒻᒪ ᐊᐅᓚᑦᑎᔾᔪᑏᑦ ᐊᑐᖅᑕᐅᓗᑎᒃ ᐱᑕᖃᕆᐊᖃᕐᓂᖏᓐᓄᑦ ᒪᓕᒃᑯᒋᑦ ᐃᒪᕐᒧᑦ ᐊᒃᑐᐃᓂᕐᓄᑦ ᖃᐅᔨᓴᖅᐸᓪᓕᐊᖏᓐᓇᕐᓂᕐᒧᑦ ᐱᓕᕆᐊᑦ (AEMP) ᐊᐅᓚᑦᑎᓂᕐᒧᑦ ᑲᒪᔾᔨᑏᑦ ᐋᖅᑭᒃᓯᒪᓂᖏᑦ.

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ACRONYMS AND ABBREVIATIONS

AEMP – Aquatic Effects Monitoring Plan

ALS – ALS Canada Ltd.

ANCOVA – Analysis-of-Covariance

ANOVA – Analysis-of-Variance

Baffinland – Baffinland Iron Mines Corporation

BCENV – British Columbia Ministry of Environment and Climate Change Strategy

BCWQG – British Columbia Water Quality Guidelines

BIC – Benthic Invertebrate Community

CALA – Canadian Association for Laboratory Accreditation

CCME – Canadian Council of Minister of the Environment

CES – Critical Effect Size

CES_{BIC} – Critical Effect Size for the Benthic Invertebrate Community study

CES_{C} – Critical Effect Size (for Fish) Condition

cm – centimetres

CPUE – Catch-Per-Unit-Effort

CREMP – Core Receiving Environment Monitoring Program

CRM – Certified Reference Material

CSQG – Canadian Sediment Quality Guidelines

CWQG – Canadian Water Quality Guidelines

dbRDA – Distance-Based Redundancy Analysis

DELT – Deformities, Erosions, Lesions, and Tumours

DO – Dissolved Oxygen

DOC – Dissolved Organic Carbon

DQR – Data Quality Review

DSS – Digital Sampling System

EEM – Environmental Effects Monitoring

FDP – Final Discharge Point

FEIS – Final Environmental Impact Statement

FFG – Functional Feeding Group

GPS – Global Positioning System

HPG – Habit Preference Group

HSD – Honestly Significant Difference

ISQG – Interim Sediment Quality Guidelines

km – kilometre

K-M – Kaplan-Meier

K-S – Kolmogorov-Smirnov

K-W – Kruskal Wallis

L – Litre

LEL – Lower Effect Level

LFD – Length-Frequency Distributions

LRL – Laboratory Reporting Limit

m – Metre

MCT – Measure of Central Tendency

MDMER – Metal and Diamond Mining Effluent Regulations

mm – Millimetres

mg – Milligrams

Minnow – Minnow Environmental Inc.

MOD – Magnitude of Difference

MRTF – Mary River Tributary-F

Mt - Million Tonnes

NAD 83 – 1983 North American Datum

No - Number

NW – Northwest

NWB – Nunavut Water Board

pa – per annum

PAL – Protection of Aquatic Life

PEL – Probable Effect Level

Project – The Mary River Project

PSQG – Ontario Provincial Sediment Quality Guidelines

PWQO – Ontario Provincial Water Quality Objectives

QA/QC – Quality Assurance/Quality Control

SD – Standard Deviation

SD_{REF} – Reference Mean Standard Deviation

$SD_{BL\text{-}YR}$ – Baseline Year Standard Deviation

SE – Southeast

SEL – Severe Effect Levels

SPC – Specific Conductance

SQG – Sediment Quality Guidelines

TDS – Total Dissolved Solids

TKN – Total Kjeldahl Nitrogen

TOC – Total Organic Carbon

TSS – Total Suspended Solids

UTM – Universal Transverse Mercator

µg - Microgram

WQG – Water Quality Guidelines

YOY – Young-Of-The-Year

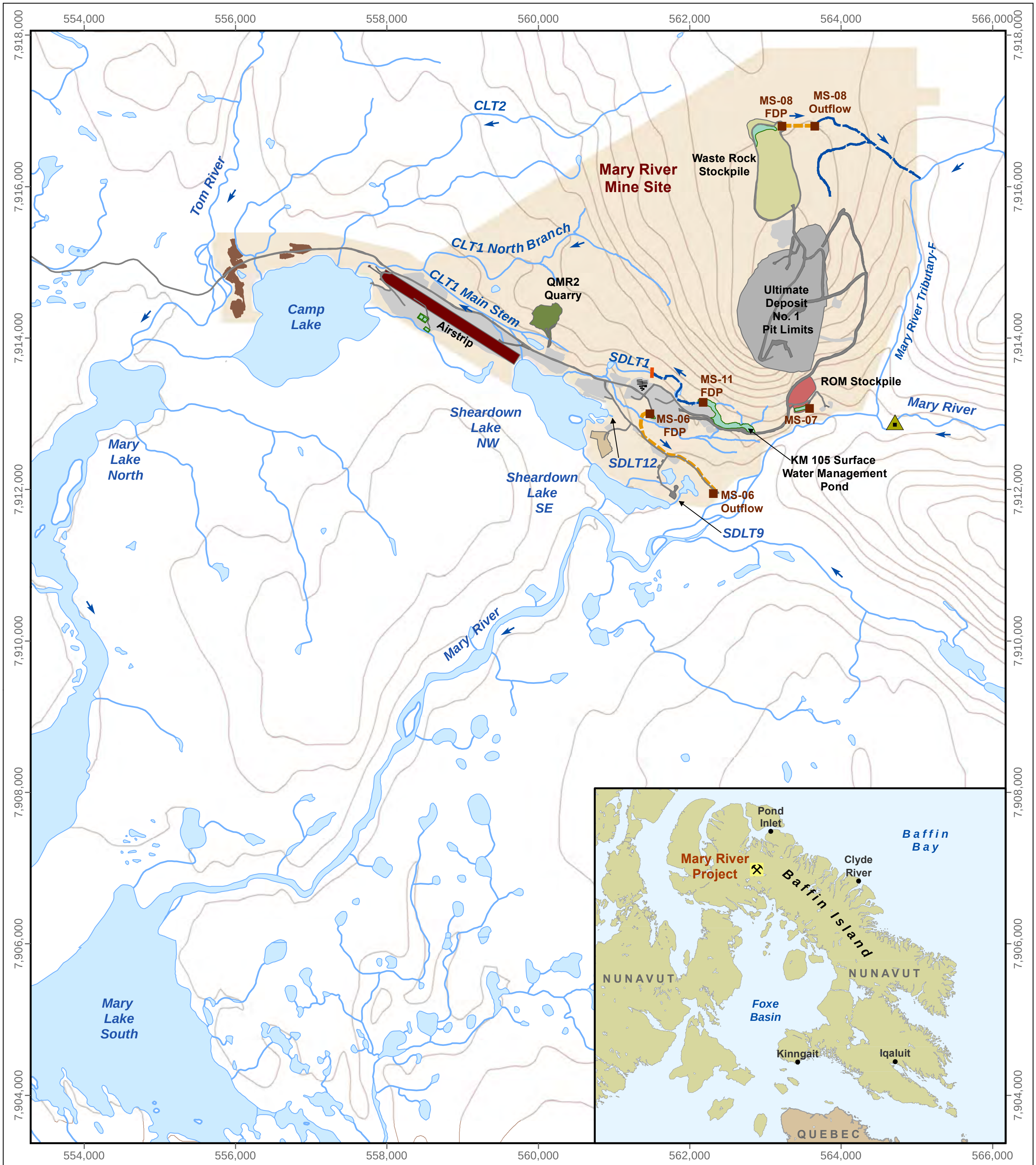
1 INTRODUCTION

1.1 Project Background

The Mary River Project (the Project), owned and operated by Baffinland Iron Mines Corporation (Baffinland), is a high-grade iron ore mining operation located in the Qikiqtani Region of northern Baffin Island, Nunavut (Figure 1.1). Commercial open pit mining, including pit bench development, ore haulage, stockpiling, and the crushing and screening of high-grade iron ore, began in 2015. During early years of mine operation (i.e., 2015 to 2017), the Project produced and transported up to 4.2 million tonnes per annum (Mtpa) of crushed and screen iron ore. Production increased between 2018 and 2024, during which time the Project was permitted to produce and transport up to 6 Mtpa to Milne Port. Over the entire operational period, ore has been transported from the Mine Site by off highway haul trucks along the Tote Road to Milne Port, which is located approximately 100 kilometres (km) north of the Mine Site. Upon arrival at Milne Port, the ore is stockpiled before being loaded onto bulk carrier ships for transport to international markets. No milling or additional processing takes place at the mine site, and as a result, no tailings are generated. The Project's mine waste management facilities include a waste rock stockpile and surface runoff collection/containment ponds which are located near the waste rock and ore stockpile areas. In addition to the periodic discharge of treated effluent from these facilities into the Mary River catchment, other potential mine-related inputs to adjacent aquatic systems include runoff and dust from ore (crusher) stockpiles in the Sheardown Lake catchment, discharge of treated sewage to the Mary River, runoff and explosives residue from quarry operations into the Camp Lake catchment, fugitive dust from mine activities, and general mine site runoff.

1.2 Monitoring Program Background

As required under the Mine's Type 'A' Water License (Number [No.] 2AM-MRY1325 Amendment No. 1) issued by the Nunavut Water Board (NWB), Baffinland developed an Aquatic Effects Monitoring Plan (AEMP) for the Project (Baffinland 2015). A key objective of the AEMP was to gather data and information to assess both short- and long-term effects of the Project on aquatic ecosystems. To achieve this, Baffinland established a Core Receiving Environment Monitoring Program (CREMP), which focuses on evaluating potential mine-related impacts on water quality, sediment quality, and aquatic biota, including phytoplankton, benthic invertebrates, and fish, in aquatic environments near the mine site (Baffinland 2015; KP 2014; NSC 2014). The primary receiving environments/systems monitored under the CREMP include the Camp Lake system (Tributaries 1 [CLT1] and 2 [CLT2], Camp Lake [JL0]), the Sheardown Lake system (Tributaries 1 [SDLT1], 9 [SDLT9], and 12 [SDLT12], Sheardown Lake Northwest [NW; DL0-01],



LEGEND

- Airstrip
- Mine Site Complex
- Pond
- ROM Stockpile
- Road
- Open Pit
- Waste Rock Facility
- Infrastructure
- Borrow Pit
- Quarry
- Landfill
- Project Development Area
- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Fish Barrier
- Overland Surface Water Flow
- HDPE Pipe
- Contour (20m)

Baffinland Iron Mines Corporation, Mary River Project

0 1.25 2.5 5 km

Map Projection: UTM Zone 17 W NAD 1983

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Date: March 2025

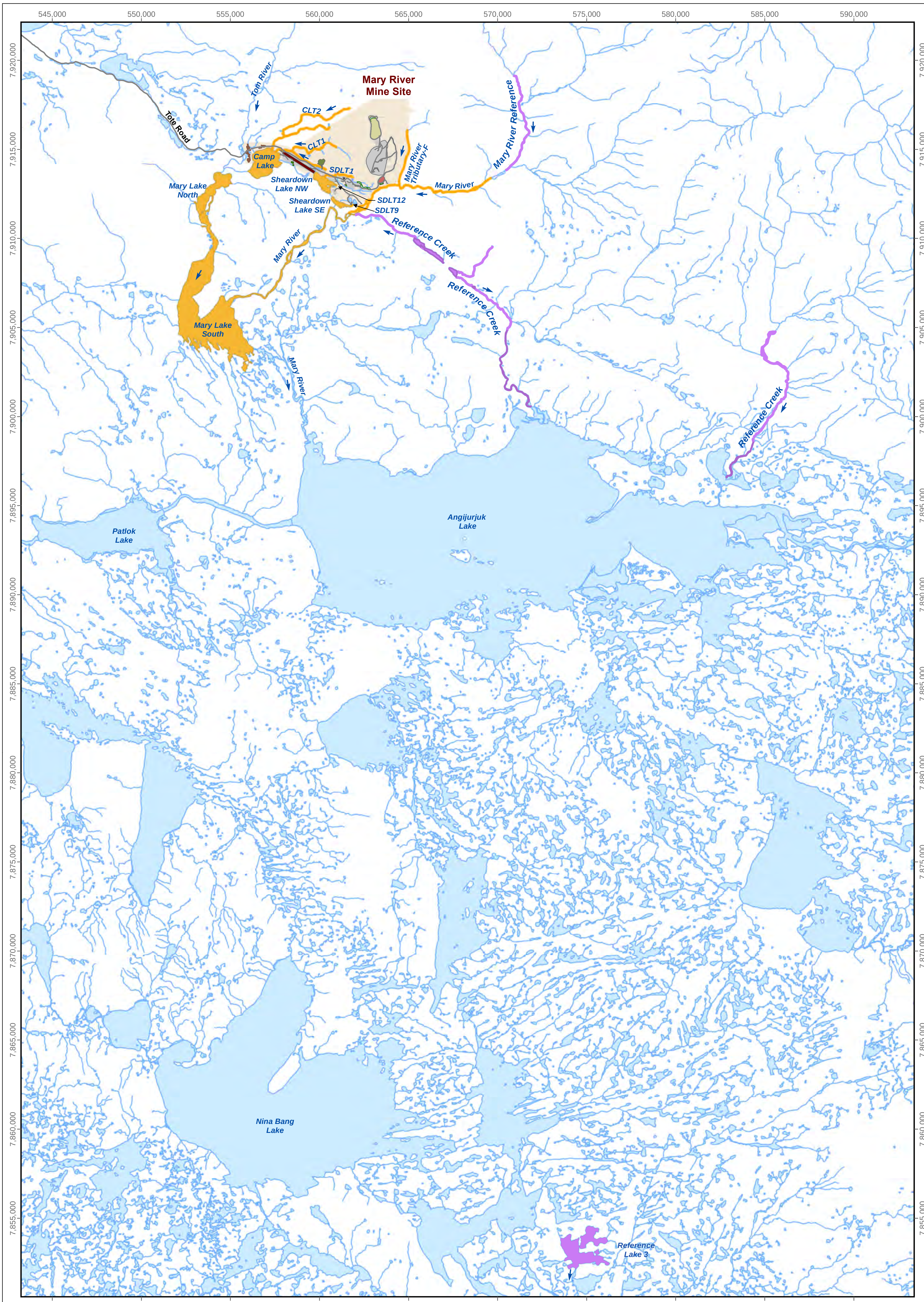
Project 247202.0075

minnow environmental inc.

Figure 1.1

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 1.1 Mary River Mine Site Location.mxd and Sheardown Lake Southeast [SE; DL0-02]), Mary River, and Mary Lake (north and south basins; BL0; Figure 1.1). Over the first nine years of mine operations, results from the CREMP have shown limited effects from Project activities to water and sediment quality in the receiving waterbodies. Potential mine-related effects, when identified, have largely been confined to tributaries flowing into Camp Lake and the Sheardown lakes, as well as Sheardown Lake NW and Sheardown Lake SE. Additionally, potential mine-related effects have been observed in Mary River Tributary-F (MRTF), a tributary to the Mary River that receives effluent from the MS-08 Final Discharge Point (FDP; Minnow 2016a, 2017, 2018, 2019, 2020, 2021b, 2022, 2023, 2024a). However, no adverse mine-related effects to phytoplankton, benthic invertebrates, or fish in the Camp Lake, Sheardown Lake, or Mary Lake systems were observed from 2015 to 2023, based on comparisons to reference waterbodies and pre-mine baseline data (Minnow 2016a, 2017, 2018, 2019, 2020, 2021b, 2022, 2023, 2024a).

This report outlines the methods and results of the 2024 CREMP, which includes an evaluation of potential Project-related effects on the chemical and biological conditions of waterbodies exposed to mine activities and covers the tenth full year of mine operation. Consistent with the previous nine years, the 2024 Mary River Project CREMP incorporated water quality, sediment quality, phytoplankton, and benthic invertebrate community (BIC) monitoring, as well as an evaluation of arctic charr (*Salvelinus alpinus*) populations. The 2024 CREMP was carried out in accordance with the study design under AEMP Revision 1 (Baffinland 2015), except for the inclusion of a reference lake for abiotic and biotic sampling, two additional water quality and phytoplankton monitoring stations, and three new BIC study areas (Figure 1.2, Table 1.1). In 2022, sediment quality and BIC data were collected in MRTF, near water quality Station F0-01 to support baseline aquatic inventory for potential future mining at the Project site and results were included in the 2022 CREMP report (Minnow 2023). However, baseline sampling in 2023 and 2024, was completed at other locations along MRTF and in Mary River, and no further sediment quality or BIC sampling was conducted at F0-01 after 2022; seasonal water quality monitoring continues at this station.



LEGEND

- Mine-exposed Stream/River System
- Reference Stream/River System
- Mine-exposed Lake
- Reference Lake
- Airstrip
- Mine Site Complex
- Pond
- ROM Stockpile
- Road
- Open Pit
- Waste Rock Facility
- Borrow Pit
- Quarry
- Infrastructure
- Project Development Area

Mary River Project 2024 CREMP Study Waterbodies

0 2.5 5 10
km

N E S W

Map Projection: UTM Zone 17N NAD 1983
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Figure 1.2

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 1.2 Mary River Project 2024 CREMP Study Waterbodies.mxd

Table 1.1: Additions to the Core Receiving Environment Monitoring Program Study Design from the Aquatic Effects Monitoring Plan Revision 1 (Baffinland 2015), Mary River Project CREMP 2024

Year New Scope was Added	Location ID	Description
2015	REF-03	An unnamed reference lake (REF-03) was added to the program to allow for a control/impact comparison for water quality, sediment quality, phytoplankton, benthic invertebrate community, and fish population endpoints.
2016	REF-CRK	A reference creek benthic invertebrate community study area (REF-CRK) was added to the program near the existing water quality reference area (MRY-REF2) in an unnamed tributary to Angijurjuk Lake to allow for a control/impact comparison for benthic invertebrate endpoints.
2021	SDLT12	A mine-exposed water quality and phytoplankton monitoring station (LDFG-OUT) was added at Sheardown Lake Tributary-12 (SDLT12) to support interpretation of biological data at the SDLT12 benthic invertebrate community monitoring area.
2021	SDLT9	A mine-exposed water quality monitoring and phytoplankton monitoring station (MS-C-G) was added at Sheardown Lake Tributary-9 (SDLT9) to support interpretation of biological data at the SDLT9 benthic invertebrate community monitoring area.
2021	CLT1-L2	A mine-exposed benthic invertebrate community sampling area (CLT1-L2) was added at the Camp Lake Tributary-1 Upper Main Stem to evaluate possible effects of elevated aqueous total aluminum and total iron concentrations (measured at water quality monitoring Station L2-03) on biota in this portion of the CLT1 system.
2024	DL0-01	A mine-exposed benthic invertebrate community sampling location (Station DL0-01-8) was added at the existing Sheardown Lake Northwest (DL0-01) sediment quality station to support the interpretation of sediment trap data collected for the Lake Sedimentation Monitoring Report.

2 METHODS

2.1 Overview

The CREMP includes water quality, sediment quality, phytoplankton (chlorophyll-a), BIC, and fish population monitoring (Baffinland 2015). As in previous years, the 2024 monitoring program involved water quality and phytoplankton sampling conducted in lakes and/or streams by Baffinland environment department personnel during four separate events: an ice-cover event in April (lakes only), an open-water season event corresponding to Arctic spring (freshet) in July (streams only), summer sampling in August, and fall sampling in September. Sediment quality, BIC, and fish population sampling were carried out by Minnow Environmental Inc. (Minnow) personnel, with assistance from Baffinland environment department personnel, between August 7th and 21st, 2024. This timing aligned with previous monitoring conducted during baseline (2005 to 2013), mine construction (2014), and mine operational (2015 to 2023) phases. The 2024 study included field sampling and standard field and laboratory quality assurance/quality control (QA/QC) measures for the water quality, sediment quality, phytoplankton (chlorophyll-a), BIC, and fish components, to support the assessment of overall data quality for each respective dataset (Appendix A).

The 2024 CREMP study areas included the same mine-exposed and reference waterbodies established in the original design document (Baffinland 2015), along with the reference lake added to the program in 2015, and additional water quality and BIC sampling locations as outlined in Section 1.2 (Figure 1.2, Table 1.1). To simplify the discussion of results, the mine-exposed study areas were grouped by lake catchment as follows:

- the Camp Lake system (Camp Lake Tributaries 1 [CLT1] and 2 [CLT2], and Camp Lake [JL0]);
- the Sheardown Lake system (Sheardown Lake Tributaries 1 [SDLT1], 9 [SDLT9], and 12 [SDLT12], Sheardown Lake NW [DL0-01], and Sheardown Lake SE [DL0-02]); and
- the Mary River (E0, C0, G0)/Mary Lake (BL0) system.

Reference Lake 3 (REF-03), which has served as a reference waterbody for lake environments since the 2015 CREMP, was again used as the reference lake for the 2024 CREMP. REF-03 is located well outside the area of potential mine influence, approximately 62 km south of the mine site (Figure 1.2). Streams used as reference areas in the current and previous CREMP studies included an unnamed tributary to Mary River and two unnamed tributaries to Angijurjuk Lake[1], all located southeast of the mine site. Consistent with earlier studies, an area of Mary River located far upstream from current mine activity (stations in the G0-09 series) served as a reference for the mine-exposed portion of Mary River in the 2024 study (Figures 1.2 and 2.1).

2.2 Water Quality

2.2.1 General Design

Surface water quality monitoring was conducted by Baffinland environment department staff at the sampling locations and frequencies stipulated in the CREMP design (Baffinland 2015), as well as at locations added in subsequent years (Table 1.1). The surface water sampling was conducted at as many as 59 stations during each sampling event (Table 2.1, Figures 2.1 and 2.2) and included collection of *in situ* water quality measurements and water chemistry data. Of the 59 stations, 57 are part of the core CREMP design, whereas two were added to the core design in fall 2021[2] (Section 2.5.2). The evaluation of potential mine-related effects on surface water near the mine site was based on comparisons of constituent concentrations to applicable reference data, baseline data, and guidelines, including site-specific AEMP benchmarks. The AEMP benchmarks were developed to help define potential effects from the Project to surface water quality, and to guide management response decisions when concentrations were above benchmarks, as part of a four-step Assessment Approach and Management Response Framework (Baffinland 2015).

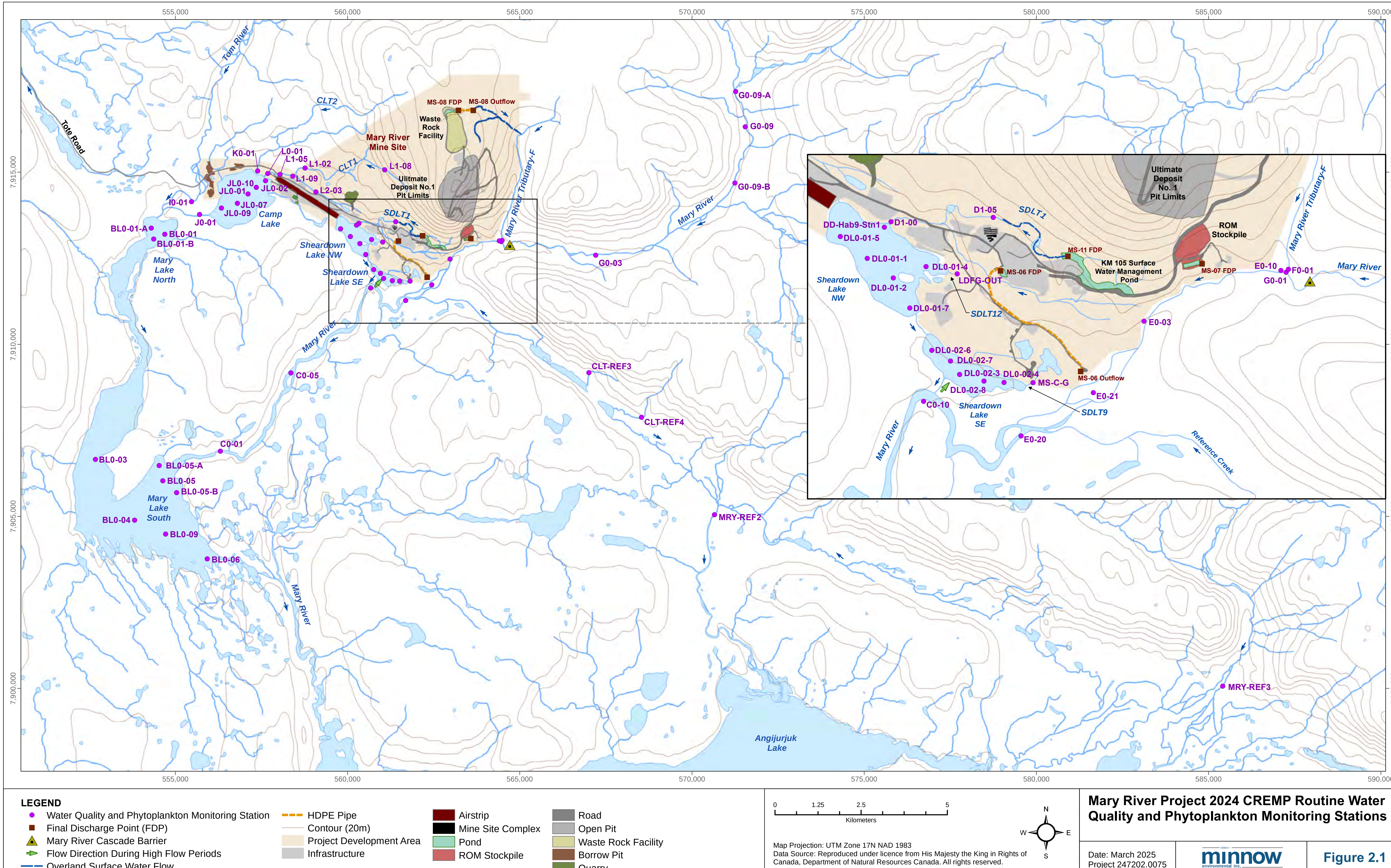
2.2.2 *In Situ* Water Quality

2.2.2.1 Sample Collection

In situ measurements of water temperature, dissolved oxygen (DO), pH, specific conductance (SPC; i.e., temperature standardized conductivity), and turbidity were taken mid-column at all lotic (stream) stations and as a vertical profile at one metre (m) intervals at each lentic (lake) water quality monitoring station during routine monitoring conducted by Baffinland personnel. These *in situ* measurements were also collected at the surface and bottom

[1] Referred to as Angajurjualuk Lake in earlier CREMP reporting (i.e., KP 2015; Minnow 2016a,b, 2017, 2018, 2021a,b, 2022). The name was changed beginning in the 2023 CREMP report (Minnow 2024a) based on an updated English translation of the Inuit place name.

[2] Water quality and phytoplankton monitoring stations were added in fall 2021 to Sheardown Tributary 12 (Station LDFG-OUT) and Sheardown Tributary 9 (Station MS-C-G). These stations were added based on recommendations made in the Mary River Mine 2020 CREMP (Minnow 2021b) to provide supporting information for BIC data interpretation (Section 2.5.2).



Mary River Project 2024 CREMP Routine Water Quality and Phytoplankton Monitoring Stations

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Figure 2.1

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 2.1 Mary River Project CREMP Routine WQ and Phytoplankton Monitoring Stations.mxd

Table 2.1: Mary River Project CREMP Water Quality and Phytoplankton (Chlorophyll-a) Monitoring Station Coordinates and Annual Sampling Schedule, 2024

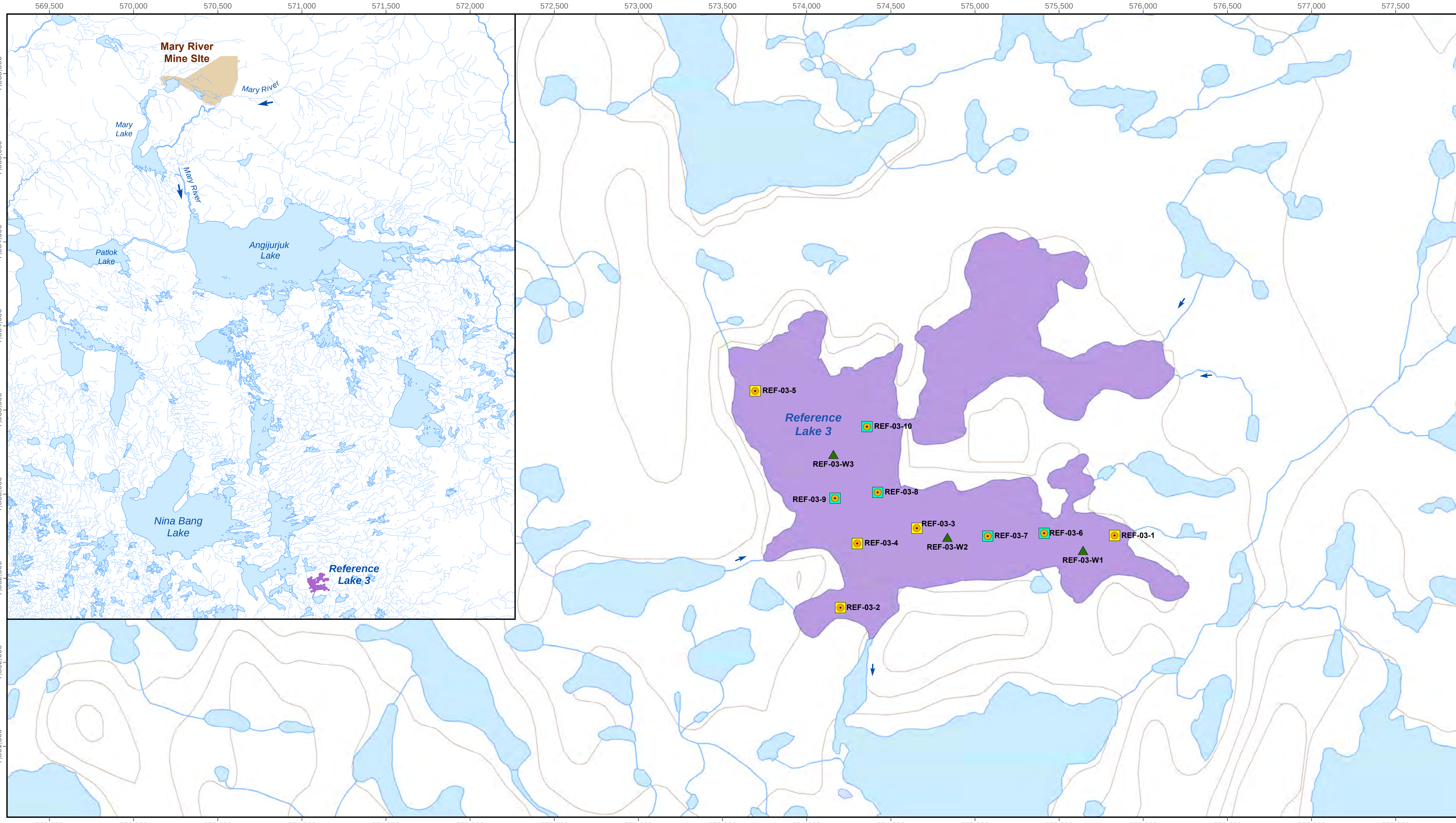
Study System	Water Body	Station ID	UTM Zone 17N, NAD83		Ref. Data Set [a]	Sampling Season			
			Easting	Northing		Winter (Apr. to May)	Spring (Jul.)	Summer (Aug.)	Fall (Sep.)
Reference Areas	Unnamed Reference Creek	CLT-REF3	567004	7909174	na	-	✓	✓	✓
	Unnamed Reference Creek	CLT-REF4	568533	7907874	na	-	✓	✓	✓
	Unnamed Reference Creek	MRY-REF3	585407	7900061	na	-	✓	✓	✓
	Unnamed Reference Creek	MRY-REF2	570650	7905045	na	-	✓	✓	✓
	Reference Lake 3	REF-03-W1	575642	7852666	na	-	-	✓	✓
		REF-03-W2	574836	7852744		-	-	✓	✓
		REF-03-W3	574158	7853237		-	-	✓	✓
	Mary River Reference	G0-09-A	571264	7917344	na	-	✓	✓	✓
		G0-09	571546	7916317		-	✓	✓	✓
		G0-09-B	571248	7914682		-	✓	✓	✓
Camp Lake System	Camp Lake Tributaries	J0-01	555701	7913773	a	-	✓	✓	✓
		K0-01	557390	7915030		-	✓	✓	✓
		L0-01	557681	7914959		-	✓	✓	✓
		L1-02	558765	7915121		-	✓	✓	✓
		L1-05	558040	7914935		-	✓	✓	✓
		L1-08	561076	7915068		-	✓	✓	✓
		L1-09	558407	7914885		-	✓	✓	✓
		L2-03	559081	7914425		-	✓	✓	✓
	Camp Lake	JL0-01	557108	7914369	b	✓	-	✓	✓
		JL0-02	557615	7914750		✓	-	✓	✓
		JL0-07	556800	7914094		✓	-	✓	✓
		JL0-09	556335	7913955		✓	-	✓	✓
		JL0-10	557346	7914562		✓	-	✓	✓
Sheardown Lake System	Sheardown Tributary 1	D1-00	560329	7913512	a	-	✓	✓	✓
		D1-05	561397	7913558		-	✓	✓	✓
	Sheardown Tributary 12	LDFG-OUT [b]	561021	7912967	a	-	✓	✓	✓ [c]
	Sheardown Tributary 9	MS-C-G [b]	561813	7911830	a	-	✓	✓	✓
	Sheardown Lake NW	DD-Hab9-Stn1	560259	7913455	b	✓	-	✓	✓
		DL0-01-1	560080	7913128		✓	-	✓	✓
		DL0-01-2	560353	7912924		✓	-	✓	✓
		DL0-01-4	560695	7913043		✓	-	✓	✓
		DL0-01-5	559798	7913356		✓	-	✓	✓
		DL0-01-7	560525	7912609		✓	-	✓	✓
	Sheardown Lake SE	DL0-02-3	561046	7911915	b	✓	-	✓	✓
		DL0-02-4	561511	7911832		✓	-	✓	✓
		DL0-02-6	560756	7912167		✓	-	✓	✓
		DL0-02-7	560952	7912054		✓	-	✓	✓
		DL0-02-8	561301	7911846		✓	-	✓	✓
Mary River and Mary Lake System	Mary River	G0-03	567204	7912587	c	-	✓	✓	✓
		G0-01	564459	7912984		-	✓	✓	✓
		F0-01	564483	7913015		-	✓	✓	✓
		E0-21	562444	7911724		-	✓	✓	✓
		E0-20	561688	7911272		-	✓	✓	✓
		E0-10	564405	7913004		-	✓	✓	✓
		E0-03	562974	7912472		-	✓	✓	✓
		C0-10	560669	7911633		-	✓	✓	✓
		C0-05	558352	7909170		-	✓	✓	✓
		C0-01	556305	7906894		-	✓	✓	✓
	Tom River	I0-01	555470	7914139	a	-	✓	✓	✓
	Mary Lake (North Basin)	BL0-01	554691	7913194	b	✓	-	✓	✓
		BL0-01-A	554300	7913378		✓	-	✓	✓
		BL0-01-B	554369	7913058		✓	-	✓	✓
	Mary Lake (South Basin)	BL0-03	552680	7906651	b	✓	-	✓	✓
		BL0-04	553817	7904886		✓	-	✓	✓
		BL0-05	554632	7906031		✓	-	✓	✓
		BL0-06	555924	7903760		✓	-	✓	✓
		BL0-05-A	554530	7906478		✓	-	✓	✓
		BL0-05-B	555034	7905692		✓	-	✓	✓
		BL0-09	554715	7904479		✓	-	✓	✓

Notes: "✓" = station is sampled during a given season. "-" = station is not sampled during a given season. na = not applicable.

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

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

[c] Station LDFG-OUT (Sheardown Lake Tributary 12) was dry during the fall sampling event in 2024; therefore, no data are available for this sampling period.



LEGEND

- Sediment and Benthic Monitoring Location
- Littoral Sampling Depth
- Profundal Sampling Depth
- Water Quality and Phytoplankton Monitoring Station
- Project Development Area

0 335 670 1,340
Meters

N W E S

Map Projection: UTM Zone 17N NAD 1983
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Mary River Project 2024 CREMP Reference Lake 3 (REF-03) Monitoring Stations

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Figure 2.2

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 2.2 CREMP Ref Lake 3 Water, Sediment, Phytoplankton, and BI Stations.mxd (i.e., approximately 30 centimetres [cm] above the water-sediment interface) at all lentic BIC stations during biological sampling conducted in August by Minnow staff. The *in situ* measurements were collected using one of three YSI Pro Digital Sampling System (DSS) meters equipped with a 4-port sensor (YSI Inc., Yellow Springs, OH). Meter readings of pH, SPC, DO, and turbidity were checked against standard solutions/using standard protocols and calibrated as necessary within 24 hours prior to the start of field sampling. If erroneous readings were identified in the field, they were investigated, and meters were re-calibrated as needed.

During the winter ice-cover sampling event, a 15-cm (6 inch) diameter electric powered ice auger was used to access the water column at lake water quality monitoring stations. Ice shavings were removed from the auger hole prior to the collection of *in situ* measures. To avoid confounding influences associated with snow/ice melt in the auger hole, the *in situ* measurements were collected just below the ice layer.

Additional supporting observations of water colour and clarity were recorded during water quality and biological sampling at all benthic stations; Secchi depth was measured at all lake stations during the summer and fall sampling periods, following methods outlined in Wetzel and Likens (2000).

2.2.2.2 Data Analysis

In situ water quality data collected at the mine-exposed stream and lake stations were compared to data from respective reference areas, applicable water quality guidelines (WQG[3]; for DO concentrations and pH only), and to baseline data for pH and conductivity. *In situ* water quality data were compared spatially within each stream (i.e., from upstream to downstream) and between littoral and profundal habitats of lake environments using both qualitative and statistical approaches. *In situ* water quality parameters were plotted to visually assess the data range and identify outliers. Values that appeared to be substantially outside the range of all other observations were flagged as potential outliers. These flagged values were cross-referenced with the original datasheets and either confirmed or corrected. In instances where verification was not possible and the data were deemed erroneous, the affected values were removed from the dataset. For the statistical analysis, both raw and log-transformed data were assessed for normality and homogeneity of variance before conducting pairwise comparisons or comparisons among multiple groups of similar habitats at mine-exposed and reference study areas using Analysis-of-Variance (ANOVA). The selection of untransformed or log-transformed data was determined based on which data best met the assumptions of ANOVA. In cases where

[3] Canadian Environmental Water Quality Guidelines (CCME 2024a) were used as the primary source for WQG, including those for DO concentrations and pH.

normality could not be achieved through data transformation, non-parametric Mann-Whitney U-tests and Kruskal Wallis (K-W) H-tests were used to conduct pairwise and multiple-group comparisons, respectively, on rank-transformed data. Similarly, in instances in which variances of normal data could not be homogenized by transformation, Student's t-tests assuming unequal variance were used for pairwise comparisons. In cases in which multiple-group comparisons were conducted, normally distributed data were subject to Tukey's Honestly Significant Difference (HSD) test and non-parametric *post hoc* tests were completed using Dunn's K-W Multiple Comparisons test (Dunn 1964). All statistical comparisons were conducted using R programming (R Core Team 2023) and an alpha (α; p-value) for defining differences of 0.05.

Vertical profiles of the *in situ* measurements taken from lake stations were plotted and visually assessed to evaluate thermal and chemical changes with depth and if they are associated with distinct layering (e.g., thermal stratification). The occurrence of a thermocline was conservatively assessed as a ≥ 0.5°C change in temperature per one m change in depth[4]. At each study lake, spatial and seasonal differences in the vertical profile plots were evaluated to provide a better understanding of natural conditions and/or mine-related influences on within-lake water quality. The vertical profile data collected at the mine-exposed study lakes were compared to those of the reference lake for each seasonal monitoring event using profile data averaged for each depth below the water surface at each lake.

2.2.3 Water Chemistry

2.2.3.1 Sample Collection and Laboratory Analysis

Surface water chemistry samples were collected from both lotic and lentic environments (Table 2.1). At lotic stations, water chemistry samples were taken from approximately the mid-water column using the grab sample method (by hand) and placed directly into pre-labeled sample bottles, which had been triple-rinsed with ambient water[5]. For samples requiring preservation, chemical preservatives were added before capping the bottles, or for bottles pre-dosed with preservatives, the sample was filled from a separate bottle. At lentic stations, two water chemistry samples were collected: one from approximately 1 m below the surface (or just below the ice layer during winter sampling) and another from approximately one m above the bottom using a non-metallic, vertically oriented 2.2-liter (L) TT Silicon Kemmerer bottle (Wildco Supply Co., Yulee, Florida). Sampling was primarily conducted by Baffinland environmental personnel

[4] Wetzel (2001) defines the thermocline as a ≥ 1˚C change in temperature per 1 m change in depth. As an outcome of discussions with regulatory agencies in 2017, regulatory agencies requested that a ≥ 0.5˚C change in temperature per 1 m change in depth be used to conservatively define a thermally stratified condition.

[5] Water sample bottles pre-dosed with preservatives could not be triple rinsed as it would result in loss of the preservative.

following Water Sampling Procedure BIM-5200-SOP-0017. If visual observations of a sample collected from one m above the bottom indicated that the lake bottom had been disturbed, the sample was rejected, and a new one was collected. During winter sampling, the water column was accessed using the same methods as for *in situ* measurements (see Section 2.2.2.1). Lake water collected with the Kemmerer bottle was transferred directly into sample bottles pre-dosed with the required chemical preservatives. For both lentic and lotic samples requiring filtration (e.g., for dissolved metals), filtration was performed in the field according to AEMP standard operating procedures (Baffinland 2015).

Following collection, water chemistry samples were placed into coolers in the field and maintained cold, but not frozen. Water chemistry sampling QA/QC included trip blanks, field blanks, equipment blanks, and field duplicates, which were collected at an approximate rate of 10% of the total number of samples collected for each sampling event (Appendix A).

Water chemistry samples were shipped on ice to ALS Canada Ltd. (ALS; Waterloo, Ontario) for analysis of pH, conductivity, hardness, total suspended solids (TSS), total dissolved solids (TDS), anions (alkalinity, bromide, chloride, sulphate), nutrients (ammonia, nitrate, nitrite, total Kjeldahl nitrogen [TKN], total phosphorus), dissolved and total organic carbon (DOC and TOC, respectively), mercury, total and dissolved metals, and phenols using standard laboratory methods[6]. The laboratories operated by ALS are accredited by the Canadian Association for Laboratory Accreditation Inc. (CALA).

2.2.3.2 Data Analysis

2.2.3.2.1 Standard Assessment

Water chemistry data were compared in the following ways: 1) among mine-exposed and reference areas for each lake catchment (Table 2.1); 2) spatially and seasonally at each mine-exposed waterbody; 3) to applicable WQG for the protection of aquatic life (Table 2.2) and/or site-specific benchmarks developed for the Mary River Project AEMP (i.e., AEMP benchmarks; Intrinsik 2014); and 4) to baseline data. To simplify the discussion of results, parameter concentration enrichment factors were calculated for data screening. These factors were determined by dividing the mean concentration of the parameter at the mine-exposed area by the mean concentration at the respective reference area/station. For temporal comparisons,

[6] The analytical methods used by ALS are developed using internationally recognized reference methods (where available), such as those published by the United States Environmental Protection Agency, American Public Health Association Standard Methods, ASTM International, International Organization for Standards, Environment Canada, British Columbia Ministry of Environment (BCENV), and Ontario Ministry of Environment.

Table 2.2: Guidelines and Benchmarks used for Water and Sediment Quality for the Mary River Project 2015 to 2024 CREMP Studies

Parameters		Units		Water Quality			Sediment Quality		Supporting Information and/or Calculations Used to Derive Hardness Dependent Water Quality Guideline
		WQ	SQ	Water Quality Guideline (WQG)[a]	AEMP Benchmark Stream	AEMP Benchmark Lake	Sediment Quality Guideline (SQG)[b]	AEMP Benchmark Lake	
Conventionals	pH (lab)	pH		6.5 - 9.0	-	-	-	-	-
	Total Organic Carbon (TOC)	%		-	-	-	10[γ]	-	-
Nutrients and Organics	Total Ammonia	mg/L	mg/kg	-	0.855	0.855	-	-	-
	Nitrate	mg/L	mg/kg	3	3	3	-	-	-
	Nitrite	mg/L	mg/kg	0.06	0.06	0.06	-	-	-
	Total Phosphorus	mg/L	mg/kg	0.020[α] or 0.030	-	-	-	-	Total phosphorus objective is 0.030 mg/L for lotic (rivers, streams) environments, and 0.020 mg/L for lentic (lake) environments.
	Phenols	mg/L	mg/kg	0.004[α]	-	-	-	-	-
Anions	Chloride (Cl)	mg/L	mg/kg	120	120	120	-	-	-
	Sulphate (SO_4)	mg/L	mg/kg	218[β]	218	218	-	-	Sulphate guideline is hardness (mg/L $CaCO_3$) dependent as follows: 128 mg/L at 0 to 30 hardness, 218 mg/L at 31 to 75 hardness, 309 mg/L at 76 to 180 hardness, and 429 mg/L at 181 to 250 hardness. Sample-specific or area-specific (mean) hardness was used for screening purposes in plotting. Values presented here and used in screening tables applicable to a conservative hardness value of 75 mg/L.
Total Metals	Aluminum (Al)	mg/L	mg/kg	0.100	0.179[hi], 0.966[j]	0.100[c], 0.179[f], 0.173[f], 0.130[g]	-	-	-
	Antimony (Sb)	mg/L	mg/kg	0.020[α]	-	-	-	-	-
	Arsenic (As)	mg/L	mg/kg	0.005	0.005	0.005	17	5.9[ceg], 6.2[d]	-
	Barium (Ba)	mg/L	mg/kg	1[β]	-	-	-	-	-
	Beryllium (Be)	mg/L	mg/kg	0.011[α]	-	-	-	-	For hardness less than 75 the guideline is 0.011, otherwise the guideline is 1.1. Sample-specific or area-specific (mean) hardness was used for screening purposes in plotting. Values presented here and used in screening tables applicable to a conservative hardness value of < 75 mg/L.
	Boron (B)	mg/L	mg/kg	1.5	-	-	-	-	-
	Cadmium (Cd)	mg/L	mg/kg	0.00012	0.00008[hi], 0.00006[j]	0.0001[c], 0.00009[de], 0.00006[g]	3.5	1.5	Cadmium guideline is hardness (mg/L $CaCO_3$) dependent. For hardness between 17 and 280 mg/L, the cadmium guideline is calculated using the equationCd (ug/L) = $10^{(0.83[\log(hardness)] -2.46)}$. Sample-specific or area-specific (mean) hardness was used for screening purposes in plotting. Values presented here and used in screening tables applicable to a conservative hardness value of 75 mg/L.
	Chromium (Cr)	mg/L	mg/kg	0.001	0.003	0.003	90	98[cg], 97[d], 79[e]	-
	Cobalt (Co)	mg/L	mg/kg	0.0009[α]	0.0040	0.0040	-	-	-
	Copper (Cu)	mg/L	mg/kg	0.0020	0.0022[hi], 0.0024[j]	0.0024[deg], 0.004[c]	110[γ]	50[cg], 58[d], 56[e]	Copper guideline is hardness (mg/L $CaCO^3$) dependent. At hardness <82 mg/L and >180 mg/L, the copper guideline is 2 and 4 ug/L, respectively. For hardness ranging from 82 to 180 mg/L, the copper guideline (ug/L) = 0.2 * $e^{(0.8545[\ln(hardness)] - 1.465)}$. Sample-specific or area-specific (mean) hardness was used for screening purposes in plotting. Values presented here and used in screening tables applicable to a conservative hardness value of 75 mg/L.
	Iron (Fe)	mg/L	mg/kg	0.30	0.326[hi], 0.874[j]	0.300	40,000[γ]	52,400[cg], 52,200[d], 34,400[e]	-

Note: "-" not applicable guideline or benchmark. Unless otherwise specified, guidelines and/or benchmarks apply to all lakes or streams.

[a] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024).

[b] Canadian Sediment Quality Guideline for the protection of aquatic life probable effects level (PEL; CCME 2024) except γ = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and δ = British Columbia Working SQG PEL (BCMOE 2024).

[c] AEMP benchmark is specific to Camp lake (JL0).

[d] AEMP benchmark is specific to Sheardown Lake NW (DL0-01).

[e] AEMP benchmark is specific to Sheardown Lake SE (DL0-02).

[f] Benchmark is 0.179 mg/L and 0.173 mg/L for shallow and deep stations, respectively at Sheardown Lake NW and Sheardown Lake SE (Intrinsik 2013).

[g] AEMP benchmark is specific to Mary Lake North (BL0-01) and South Basins (BL0-02).

[h] AEMP benchmark is specific to stations Camp Lake Tributaries (CLT and CLT2).

[i] AEMP benchmark is specific to Sheardown Lake Tributaries (SDLT).

[j] AEMP benchmark is specific to Mary River and Mary River Tributary-F (MRTF).

Table 2.2: Guidelines and Benchmarks used for Water and Sediment Quality for the Mary River Project 2015 to 2024 CREMP Studies

Parameters		Units		Water Quality			Sediment Quality		Supporting Information and/or Calculations Used to Derive Hardness Dependent Water Quality Guideline
		WQ	SQ	Water Quality Guideline (WQG)[a]	AEMP Benchmark Stream	AEMP Benchmark Lake	Sediment Quality Guideline (SQG)[b]	AEMP Benchmark Lake	
Total Metals	Lead (Pb)	mg/L	mg/kg	0.001	0.001	0.001	91	35	Lead guideline is hardness (mg/L $CaCO_3$) dependent. At hardness <60 mg/L and >180 mg/L, the lead guideline is 1 and 7 ug/L, respectively. For hardness ranging from 60 to 180 mg/L, the lead guideline (ug/L) = $e^{(1.273[\ln(hardness] - 4.705)}$. Sample-specific or area-specific (mean) hardness was used for screening purposes in plotting. Values presented here and used in screening tables applicable to a conservative hardness value of 75 mg/L.
	Magnesium (Mg)	mg/L	mg/kg	-	-	-	-	-	-
	Manganese (Mn)	mg/L	mg/kg	0.935[β]	-	-	1,100[γ,δ]	4,370[cg], 4,530[d], 657[e]	Manganese guideline is hardness (mg/L $CaCO_3$) dependent, and calculated using the equation Mn (ug/L) = 0.0044 * (hardness) + 0.605. Sample-specific or area-specific (mean) hardness was used for screening purposes. Value presented applicable to water with hardness of 75 mg/L.
	Mercury (Hg)	mg/L	mg/kg	0.000026	-	-	0.486	0.17	-
	Molybdenum (Mo)	mg/L	mg/kg	0.073	-	-	-	-	-
	Nickel (Ni)	mg/L	mg/kg	0.025	0.025	0.025	75[γ,δ]	72[cg], 77[d], 66[e]	Nickel guideline is hardness (mg/L $CaCO_3$) dependent. At hardness <60 mg/L and >180 mg/L, the nickel guideline is 25 and 150 ug/L, respectively. For hardness ranging from 60 to 180 mg/L, the nickel guideline (ug/L) = $e^{(0.76[\ln(hardness] + 1.06)}$. Sample-specific or area-specific (mean) hardness was used for screening purposes in plotting. Values presented here and used in screening tables applicable to a conservative hardness value of 75 mg/L.
	Potassium (K)	mg/L	mg/kg	-	-	-	-	-	-
	Phosphorus	mg/kg	mg/kg	-	-	-	2,000[γ]	1,580[cg], 1,958[d], 1,278[e]	-
	Selenium (Se)	mg/L	mg/kg	0.001	-	-	-	-	-
	Silicon (Si)	mg/L	mg/kg	-	-	-	-	-	-
	Silver (Ag)	mg/L	mg/kg	0.00025	0.0001	0.0001	-	-	-
	Sodium (Na)	mg/L	mg/kg	-	-	-	-	-	-
	Strontium (Sr)	mg/L	mg/kg	-	-	-	-	-	-
	Thallium (Tl)	mg/L	mg/kg	0.0008	0.0008	0.0008	-	-	-
	Tin (Sn)	mg/L	mg/kg	-	-	-	-	-	-
	Titanium (Ti)	mg/L	mg/kg	-	-	-	-	-	-
	Tungsten	mg/L	mg/kg	0.03[α]	-	-	-	-	-
	Uranium (U)	mg/L	mg/kg	0.015	-	-	-	-	-
	Vanadium (V)	mg/L	mg/kg	0.006[α]	0.006	0.006	-	-	-
	Zinc (Zn)	mg/L	mg/kg	0.02[α]	0.030	0.030	315	135	-

Note: "-" not applicable guideline or benchmark. Unless otherwise specified, guidelines and/or benchmarks apply to all lakes or streams.

[a] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024).

[b] Canadian Sediment Quality Guideline for the protection of aquatic life probable effects level (PEL; CCME 2024) except γ = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and δ = British Columbia Working SQG PEL (BCMOE 2024).

[c] AEMP benchmark is specific to Camp lake (JL0).

[d] AEMP benchmark is specific to Sheardown Lake NW (DL0-01).

[e] AEMP benchmark is specific to Sheardown Lake SE (DL0-02).

[f] Benchmark is 0.179 mg/L and 0.173 mg/L for shallow and deep stations, respectively at Sheardown Lake NW and Sheardown Lake SE (Intrinsik 2013).

[g] AEMP benchmark is specific to Mary Lake North (BL0-01) and South Basins (BL0-02).

[h] AEMP benchmark is specific to stations Camp Lake Tributaries (CLT and CLT2).

[i] AEMP benchmark is specific to Sheardown Lake Tributaries (SDLT).

[j] AEMP benchmark is specific to Mary River and Mary River Tributary-F (MRTF).

the enrichment factor was calculated by dividing the 2024 mean parameter concentration at a mine-exposed station/area by the baseline (2005 to 2013) mean concentration. The resulting enrichment factors were qualitatively categorized as slightly, moderately, or highly elevated compared to reference and/or baseline conditions, using the categorization outlined in Table 2.3.

Applicable WQG included the Canadian Water Quality Guidelines (CWQG; CCME 2024a) or, for parameters with no CWQG, the most conservative (i.e., lowest) criterion available from established Ontario Provincial Water Quality Objectives (PWQO; OMOEE 1994) or approved and/or working British Columbia Water Quality Guidelines (BCWQG; BCENV 2024; Table 2.2). For WQGs that are hardness dependent, the hardness of the individual sample was used to calculate the WQG for a specific parameter according to an established formula (Table 2.2). The AEMP water quality benchmarks were derived using an evaluation of background (i.e., baseline) water chemistry data together with existing generic WQGs that consider aquatic toxicity thresholds. These benchmarks were developed to inform management decisions under the AEMP Assessment Approach and Management Response Framework (Baffinland 2015). The concentration of a parameter being above the respective AEMP benchmark may trigger various actions, such as modifications to the sampling design, additional statistical assessments, and/or consideration of mitigation measures, to better understand and potentially reduce the effects (Section 2.5; Baffinland 2015). Water chemistry data for key parameters, those with concentrations that were higher at mine-exposed areas compared to reference areas and baseline conditions, were plotted to assess changes in concentrations over time and between baseline (2005 to 2013 data) and mine operational years (2015 to 2024). A mine-related effect was determined if the mean concentration of a parameter was categorized as at least slightly elevated relative to both reference and baseline concentrations, either across all seasons or in any individual season, and/or if qualitative evaluation of temporal plots suggested a mine-related increase over time.

2.2.3.2.2 Special Investigations

As recommended in the 2023 CREMP (Minnow 2024a; see Section 6 of the 2023 CREMP Report), special investigations were completed in instances where a mine-related influence was concluded for a surface water quality parameter (Table 2.4). One type of water quality special investigation involved temporal trend analyses for parameters that were elevated relative to AEMP benchmarks, baseline data, and/or reference area data across all seasons. Temporal trends were assessed using the non-parametric seasonal Kendall test described by Hirsch et al. (1982) using scripts written in R software (R Core Team 2023). The seasonal Kendall test assesses temporal trends separately for each season and combines the results for each season into an overall test for trend. The test is non-parametric and assesses whether there

Table 2.3: **Enrichment Factor Categories for Water and Sediment Chemistry Comparisons**

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

Table 2.4: **Summary of Special Investigations as a Result of Action Level Responses for Water Quality, Mary River Project CREMP, 2024**

Waterbody Type	Waterbody Name	Station(s)	Special Investigation Type	Parameter(s)	Year Identified	Results Section
Stream	Camp Lake Tributary 1 Upper Main Stem (CLT1)	L2-03	Temporal Trend Analysis	Total and dissolved iron, molybdenum, sodium, uranium. Total sulphate.	2023	3.1.1.3
Stream	Sheardown Lake Tributary 1 (SDLT1)	D1-05 D1-00	Temporal Trend Analysis	Total and dissolved aluminum, cadmium, iron, lithium, manganese, magnesium, potassium, strontium, uranium. Total chloride, nitrate, sulphate.	2023	4.1.1.2
Stream	Sheardown Lake Tributary 12 (SDLT12)	LDFG-OUT	Temporal Trend Analysis	Total sulphate.	2023	4.3.1.2
Stream	Mary River Tributary-F (MRTF)	F0-01	Temporal Trend Analysis	Total nitrate, sulphate.	2023	5.2.1.2
Lake	Sheardown Lake Northwest (DL0-01) and Southeast (DL0-02)	All	Analysis of Total vs. Dissolved Concentrations	Total and dissolved molybdenum, uranium.	2023	4.4.1.2 4.5.1.2

is a monotonic increasing or monotonic decreasing trend over time. The tests were conducted by calculating the test statistic (S_i),which is equal to the sum of the number of increases and decreases from a time period (t) to all time periods after t for each observation in season (i). The overall test statistic S was computed as the sum of S_i for all seasons. The significance of the observed S was determined by comparing it to a critical value of S (at the significance level α = 0.05) determined from the exact sampling distribution of S (calculated by determining all possible permutations and combinations of S based on the increases and decreases from the number of pairwise comparisons made; Hirsch et al. 1982). If more than 45 pairwise comparisons are made (equivalent to the number of pairwise comparisons for n = 10 in a single season), then the normal approximation was used to calculate a p-value and to assess significance (Hirsch et al. 1982). The standard normal deviate (Z) was calculated as:

$$Z = \begin{cases} \dfrac{S-1}{\sqrt{\sigma_S}} \, if \; S > 0 \\ 0 \quad if \; S = 0 \\ \dfrac{S+1}{\sqrt{\sigma_S}} \, if \; S < 0 \end{cases}$$

where $\sigma_S = \sum_{i=1}^{k} \frac{n_i(n_i-1)(2n_i+5)-\sum_{T_i} t_i(t_i-1)(2t_i+5)}{18}$ and n_i is the number of samples in season i, t_i is the number of tied values for each tied value T_i, and k is the number of seasons (Hirsch et al. 1982).

The trend slope over time was estimated by computing the median of all slopes between data pairs within the same season (Helsel and Hirsch 2002). The slope was reported as a percentage change in concentration per season and per year. The intercept of a line through the time series was estimated as the median intercept of all lines through each point with the estimated slope (Pohlert 2016). The trend analysis was conducted only when no fewer than five pairwise comparisons were possible (i.e., the minimum number required for all consecutive increases or decreases to be significant at α = 0.05). The seasonal averages used in the analysis were calculated using the Kaplan-Meier (K-M) method in R (R Core Team 2023) following the methods described in Helsel (2012).

The other type of water quality special investigation explored the relationship between total and dissolved concentrations of select parameters. In particular, increasing trends in aqueous concentrations of total and dissolved uranium and molybdenum were identified in Sheardown Lake NW and SE in 2023. Further analysis, comparing total to dissolved concentrations, was recommended to assess bioavailability and further evaluate the potential effects on aquatic biota. Exploratory analyses were conducted by creating scatterplots of total compared to dissolved concentrations measured since the baseline period (starting in 2006). Scatterplots were examined to qualitatively identify any temporal or seasonal changes in the relative concentrations of the two fractions that may suggest greater bioavailability or increased risk of potential effects to biota.

2.3 Sediment Quality

2.3.1 General Design

Sediment quality monitoring for the CREMP was designed to assess potential mine-related influence on sediment quality of lake environments using a gradient approach (Baffinland 2015). Sediment quality sampling in 2024 was conducted at five to ten stations per study lake to support physical and chemical characterization of sediments, as outlined under the AEMP. Additionally, physical sediment properties were characterized at four to six stations per study lake to support the interpretation of BIC data (Table 2.5; Figure 2.3). The lake sediment stations were classified as littoral or profundal based on a depth cutoff of 12 m, which was used to define lake zonation during baseline characterization studies (KP 2014, 2015). Similar to water quality, the evaluation of potential mine-related influence on sediments in the Project area lakes focused on the use of established AEMP benchmarks as well as comparisons to reference areas and baseline conditions. Baffinland conducts sediment sampling in the Camp Lake tributaries, Sheardown Lake tributaries, and Mary River on a three-year cycle. In line with this schedule, the 2024 CREMP did not include sampling at these locations, as it was completed in 2023 (Minnow 2024a).

2.3.2 Sample Collection and Laboratory Analysis

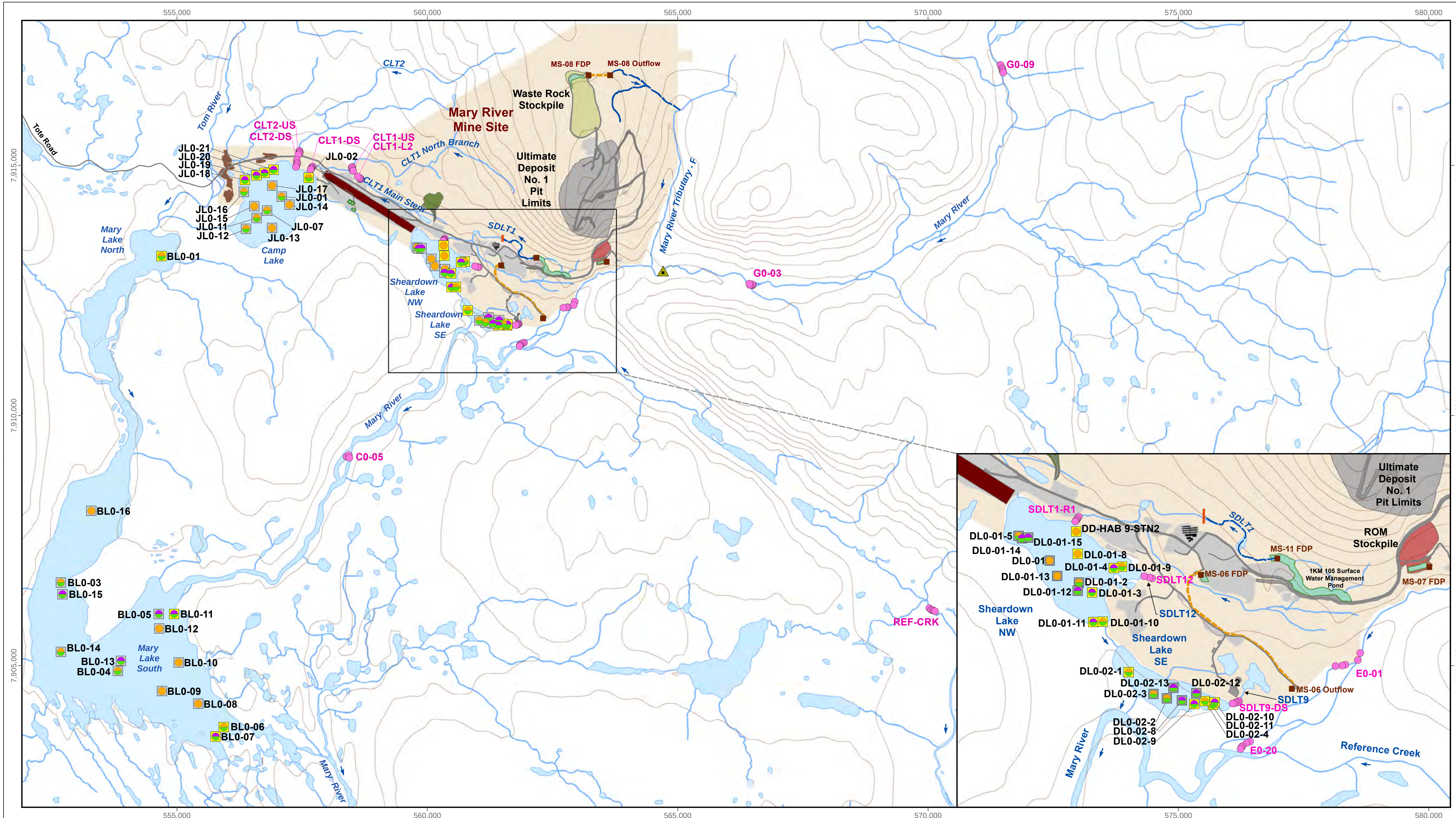
Sediment samples were collected using two methods at each of the mine-exposed study lakes: gravity core sampling for chemical/physical characterization and Petite Ponar sampling for physical characterization (Table 2.5, Figure 2.3). The sediment samples collected by Petite Ponar were taken from four to six BIC stations per study lake to assess the physical substrate, ensuring representation of sediment composition at BIC stations that had not been paired with sediment quality monitoring (i.e., sediment samples collected using the gravity core) under the AEMP (Baffinland 2015). No Petite Ponar sampling to support physical characterization of sediments was completed in Reference Lake 3 because all BIC sampling stations were paired with sediment quality monitoring stations (i.e., gravity core) in 2015 when sampling in this lake was initiated.

Coring was completed using a gravity corer (Hoskin Scientific Ltd., Model E-777-00) fitted with a clean 5.1 cm inside-diameter polycarbonate tube. From each retrieved, intact core that was representative of the sediment-water interface, the top two cm of sediment were manually extruded into a graded core collar, sectioned with a core knife, and placed into a pre-labeled plastic sample bag. If a core was not intact when retrieved, it was discarded, and a new one

Table 2.5: Lake Sediment Quality and Benthic Invertebrate Community Monitoring Station Identifiers and Coordinates Used for the Mary River Project CREMP, 2024

Waterbody	Station Code	UTM Zone 17N, NAD83		Sampling Habitat	Sample Type		
		Easting	Northing		Sediment Core[a]	Petite Ponar[b]	Benthic Invertebrate
Reference Lake 3	REF-03-1	575830	7852754	littoral	✓	-	✓
	REF-03-2	574201	7852325	littoral	✓	-	✓
	REF-03-3	574655	7852797	littoral	✓	-	✓
	REF-03-4	574301	7852706	littoral	✓	-	✓
	REF-03-5	573694	7853614	littoral	✓	-	✓
	REF-03-6	575411	7852767	profundal	✓	-	✓
	REF-03-7	575076	7852751	profundal	✓	-	✓
	REF-03-8	574421	7853011	profundal	✓	-	✓
	REF-03-9	574168	7852976	profundal	✓	-	✓
	REF-03-10	574358	7853401	profundal	✓	-	✓
Camp Lake	JL0-02	557630	7914751	littoral	✓	-	✓
	JL0-01	557090	7914376	profundal	✓	-	✓
	JL0-14	557244	7914216	profundal	✓	-	-
	JL0-17	556900	7914594	profundal	✓	-	-
	JL0-21	556926	7914913	littoral	-	✓	✓
	JL0-20	556748	7914850	littoral	-	✓	✓
	JL0-19	556585	7914800	littoral	-	✓	✓
	JL0-07	556800	7914099	profundal	✓	-	✓
	JL0-18	556355	7914708	littoral	-	✓	✓
	JL0-16	556336	7914470	profundal	✓	-	✓
	JL0-15	556542	7914184	profundal	✓	-	-
	JL0-11	556592	7913948	profundal	✓	-	✓
	JL0-13	556896	7913751	profundal	✓	-	-
	JL0-12	556378	7913732	profundal	✓	-	✓
Sheardown Lake Northwest (NW)	DL0-01-5	559792	7913360	profundal	✓	-	✓
	DL0-01-14	559824	7913338	profundal	-	✓	✓
	DL0-01-15	559882	7913345	profundal	-	✓	✓
	DD-HAB 9-STN2	560325	7913400	littoral	✓	-	-
	DL0-01-8	560338	7913194	littoral	✓	-	-
	DL0-01	560079	7913132	profundal	✓	-	-
	DL0-01-13	560149	7912990	profundal	✓	-	-
	DL0-01-2	560350	7912929	profundal	✓	-	✓
	DL0-01-12	560339	7912853	profundal	-	✓	✓
	DL0-01-9	560747	7913076	littoral	✓	-	✓
	DL0-01-4	560696	7913049	littoral	-	✓	✓
	DL0-01-3	560471	7912839	littoral	-	✓	✓
	DL0-01-11	560482	7912564	littoral	-	✓	✓
	DL0-01-10	560570	7912568	littoral	✓	-	✓
Sheardown Lake Southeast (SE)	DL0-02-1	560808	7912102	littoral	✓	-	✓
	DL0-02-11	561585	7911801	littoral	✓	-	✓
	DL0-02-10	561602	7911823	littoral	-	✓	✓
	DL0-02-4	561511	7911835	littoral	✓	-	✓
	DL0-02-12	561433	7911907	profundal	-	✓	✓
	DL0-02-9	561412	7911808	littoral	-	✓	✓
	DL0-02-8	561300	7911841	profundal	-	✓	✓
	DL0-02-13	561222	7911959	profundal	-	✓	✓
	DL0-02-2	561161	7911860	profundal	✓	-	✓
	DL0-02-3	561037	7911900	profundal	✓	-	✓
Mary Lake	BL0-01	554690	7913187	littoral	✓	-	✓
	BL0-16	553289	7908094	profundal	✓	-	-
	BL0-03	552679	7906662	profundal	✓	-	✓
	BL0-15	552714	7906428	profundal	-	✓	✓
	BL0-14	552679	7905276	profundal	✓	-	✓
	BL0-05	554635	7906034	profundal	-	✓	✓
	BL0-11	554942	7906034	littoral	-	✓	✓
	BL0-12	554644	7905742	profundal	✓	-	-
	BL0-13	553881	7905097	profundal	-	✓	✓
	BL0-04	553819	7904895	profundal	✓	-	✓
	BL0-10	555033	7905065	profundal	✓	-	-
	BL0-09	554700	7904493	profundal	✓	-	-
	BL0-08	555424	7904239	profundal	✓	-	-
	BL0-07	555770	7903584	littoral	-	✓	✓
	BL0-06	555932	7903773	littoral	✓	-	✓

Notes: "✓" = station is sampled for specified sample type; "-" = station is not sampled for specified sample type.
[a] Sediment core samples analyzed for particle size, total organic carbon (TOC), and total metals.
[b] Petite Ponar sediment grab samples analyzed for particle size only.



Mary River Project 2024 CREMP Sediment Quality and Benthic Invertebrate Community Sampling Stations

Date: March 2025
Project 247202.0075

minnow environmental inc.

Figure 2.3

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 2.3 Mary River Project CREMP Sediment Quality and BIC Sampling Stations.mxd was collected. Samples from three to four cores treated in this manner were composited to create a single sample at each station. Samples were placed into a labelled polyethylene sealable bag for analyses of particle size, moisture content, TOC content, and concentrations of metals/metalloids [hereafter collectively referred to as metals], including mercury. Supporting measurements of core penetration depth and depths of visually apparent redox boundaries/horizons, as well as notes regarding sediment texture and colour for each visible horizon, general sediment odour (e.g., hydrogen sulphide), and presence of algae or plants on or in the sediment, were recorded for each core sample.

Sediment samples for physical characterization were collected using a stainless-steel Petite Ponar (0.023 m^2 sampling area) from select sampling stations at each mine-exposed lake (Table 2.5, Figure 2.3). Each sample consisted of two grabs that were combined to create a composite sediment sample. If a grab was not complete to each edge of the sampler, or lacked an intact sediment-water surface layer, it was discarded, and a new grab was collected. If the grab was acceptable, the top two to three cm of sediment (i.e., the sediment fraction in which most benthic fauna generally reside [Kirchner 1975]) were removed and placed into a separate plastic tub. After two acceptable grabs were obtained, the sample was homogenized using a stainless-steel spoon. The homogenized sediment was then transferred to a labelled polyethylene sealable bag for analyses of moisture content, TOC content, and particle size.

A hand-held Garmin global positioning system (GPS; Garmin International Inc., Olathe, Kansas) was used to record GPS coordinates at each sediment sampling station in Universal Transverse Mercator (UTM) units and based on 1983 North American Datum (NAD 83). Following collection, all sediment samples were placed into a cooler and transported to the mine, where they were stored until shipment to a CALA-certified analytical laboratory, both under cool conditions.

Sediment samples (whole sample not field-sieved) were sent to ALS (Waterloo, Ontario) for analysis of moisture content, particle size, TOC content, and concentrations of metals using standard laboratory methods[7]. The QA/QC program included an assessment of laboratory sensitivity, accuracy, and precision (Province of British Columbia 2020). Specifically, data quality was evaluated based on the ability to achieve minimum laboratory reporting limits (LRL), as well as acceptable results for laboratory duplicate, spike recovery, blank, and certified reference material (CRM) samples (Appendix A).

[7] The analytical methods used by ALS are developed using internationally recognized reference methods (where available), such as those published by the United States Environmental Protection Agency, American Public Health Association Standard Methods, ASTM International, International Organization for Standards, Environment and Climate Change Canada, British Columbia Ministry of Environment, and Ontario Ministry of Environment.

2.3.3 Data Analysis

2.3.3.1 Standard Assessment

Similar to water quality, the evaluation of potential mine-related effects on sediments in mine-exposed lakes relied on established AEMP benchmarks. Sediment quality data from the mine-exposed lakes were compared to reference area data from similar habitats, applicable sediment quality guidelines/AEMP benchmarks, and, where available, baseline sediment quality data. Physical characteristics of sediment (e.g., moisture, particle size) and TOC at study area lakes, including the results collected via gravity core and Petite Ponar, were summarized by calculating the mean, standard deviation (SD), standard error, minima, and maxima for both littoral and profundal habitats. The physical sediment data from the mine-exposed lakes were compared to reference lake data using the same data transformations, test assumptions, statistical tests, and statistical software described previously for the statistical evaluation of *in situ* water quality (Section 2.2.2.2).

Sediment chemistry data from the mine-exposed lakes were initially assessed to identify potential gradients in metal concentrations with distance from known or suspected sources of mine influence. For each mine-exposed lake, data for each sediment chemistry parameter were averaged by habitat type (e.g., littoral and profundal habitat) and then compared to baseline and reference areas using enrichment factors, calculated and compared as described previously for evaluation of water chemistry data (Section 2.2.3.2; Table 2.3). The sediment chemistry data collected from lake environments were compared to applicable Canadian Sediment Quality Guidelines (CSQG; CCME 2024a) probable effect levels (PEL) or, for parameters without CSQG, to Ontario Provincial Sediment Quality Guidelines (PSQG; OMOE 1993) severe effect levels (SEL), collectively referred to as 'Sediment Quality Guidelines' (SQG) throughout this document. Additionally, the 2024 lake sediment chemistry data analyses included comparisons to Mary River Project AEMP sediment quality benchmarks, derived using baseline sediment chemistry data for each mine-exposed lake and existing generic CSQG interim or PSQG lowest effect level sediment quality guidelines (Intrinsik 2014, 2015).

As previously indicated, the AEMP benchmarks were developed to inform management decisions under the AEMP Assessment Approach and Management Response Framework (i.e., an increase in concentration to above the benchmark potentially triggers various actions to better understand and mitigate effects; Section 2.5; Baffinland 2015). Sediment chemistry data for key parameters, those with higher concentrations at mine-exposed areas compared to the reference area or baseline conditions, identified as site-specific parameters of concern in previous studies, or exceeding SQG and/or AEMP benchmarks, were plotted to qualitatively evaluate potential changes in concentrations in 2024 relative to baseline (2005 to 2013), construction (2014), and earlier mine operation years (2015 to 2023).

2.3.3.2 Special Investigations

As recommended in the 2023 CREMP (Minnow 2024a; see Section 6 of 2023 CREMP Report), special investigations were completed for sediment quality parameters in the study lakes where a potential for a mine-related influence was identified (Table 2.6). Temporal trend analyses were completed for parameters with concentrations that were elevated relative to AEMP benchmarks, baseline concentrations, and/or reference data. Only one sampling event was conducted each year and thus, temporal trends were first assessed for each station using a Mann-Kendall test for trend (Mann 1945). This approach uses the same methods described for the seasonal Kendall test used in special investigations of water chemistry temporal trends (Section 2.2.3.2.2) but only includes one "season" in the analysis. Trends were also assessed within each lake by adapting the non-parametric seasonal Kendall test (Hirsch et al.1982) to a regional Kendall test that assessed temporal trends separately for each station (or "region") within a lake and combined the results into an overall test for trend (Helsel and Frans 2006). The test was completed using customized R scripts (R Core Team 2023) and according to the same methods described for the seasonal Kendall test used in special investigations of water chemistry temporal trends (Section 2.2.3.2.2), but with trends combined for each station rather than over seasons. The trend slope over time was estimated by computing the median of all slopes between data pairs within the same station, rather than the same season (Helsel and Hirsch 2002).

A temporal assessment was also conducted for the CLT1 North Branch (Station CLT1-US) and SDLT1 (Station SDLT1-R1) study streams to determine whether changes in sediment iron concentrations have occurred during the mine operation period (i.e., since stream sediment sampling was initiated as a component of the CREMP in 2017). Because replicate samples (n = 3) were available at these study areas, an ANOVA was conducted to test significant differences among sampled years. Data were $\log_{10}$ transformed, if necessary, to meet the assumptions of normality and equal variance. If the ANOVA was significant (p-value < 0.05), pairwise *post hoc* contrasts were conducted with the number of comparisons corrected using a Tukey's HSD test. If the assumptions could not be met, equivalent non-parametric tests were used. A percent magnitude of difference (MOD) was calculated for significant pairwise comparisons as:

$$(MCT_{\text{later year}} - MCT_{\text{earlier year}})/MCT_{\text{earlier year}} \times 100\%$$

where the Measure of Central Tendency (MCT) was calculated according to the analysis type (mean for untransformed, geometric mean for $\log_{10}$ transformed, or medians for non-parametric).

Table 2.6: **Summary of Special Investigations as a Result of Action Level Response for Sediment Quality, Mary River Project CREMP, 2024**

Waterbody Type	Waterbody Name	Station(s)	Special Investigation Type	Parameter(s)	Year Identified	Results Section
Lake	Sheardown Lake Northwest (DL0-01)	All	Temporal Trend Analysis	Iron	2023	4.4.2
Stream	Camp Lake Tributary 1 North Branch (CLT1)	CLT1-US	Temporal Trend Analysis	Iron	2023	3.1.3
Stream	Sheardown Lake Tributary 1 (SDLT1)	SDLT1-R1	Temporal Trend Analysis	Iron	2023	4.1.3

2.4 Biological Assessment

2.4.1 Phytoplankton

2.4.1.1 Sample Collection and Laboratory Analysis

The CREMP measures aqueous chlorophyll-a concentrations to assess potential mine related influences on phytoplankton. Chlorophyll-a, the primary pigment of phytoplankton (including algae and other photosynthetic microbiota suspended in the water column), is commonly used as a surrogate to evaluate the abundance of photosynthetic microbiota in aquatic environments (Wetzel 2001), which helps to understand the productivity of a system. Water samples for analysis of aqueous chlorophyll-a concentrations were collected by Baffinland environmental department staff at the same stations and same time, and using the same methods and equipment, as those employed for the collection of water chemistry samples (Table 2.1; Figure 2.2; Section 2.2.3.1). The samples for chlorophyll-a analysis were collected into 1L glass amber bottles and maintained- in a cool and dark environment prior to submission to ALS (Mary River On-site Laboratory, Nunavut). Within 48 hours of sample collection, the on-site laboratory filtered the samples through a 0.45-micron (µm) cellulose acetate membrane filter using a vacuum pump. Following filtration, the membrane filter was wrapped in aluminum foil, inserted into a labelled envelope, and then frozen. At the completion of field collections for each seasonal sampling event, the filters were shipped frozen to ALS in Waterloo, Ontario for analysis of chlorophyll-a and phaeophytin-a analysis using standard laboratory methods[8,9]. The field QA/QC applied during collection of samples for chlorophyll-a analysis was similar to that described for water chemistry sampling (Section 2.2.3.1).

2.4.1.2 Data Analysis

The analysis of aqueous chlorophyll-a concentration data followed a similar approach to that used for the water quality evaluation. Chlorophyll-a concentrations were compared: 1) among mine-exposed and reference areas; 2) spatially and seasonally at each mine-exposed waterbody; 3) to AEMP benchmarks; and 4) to baseline data. Comparisons among the mine-exposed and reference areas/baseline conditions utilized both qualitative and statistical methods, with statistical tests, data transformations, test assumptions, statistical software, and α (p-value) for defining differences between study areas and/or relative to baseline consistent with those

[8] The analytical methods used by ALS are developed using internationally recognized reference methods (where available), such as those published by the United States Environmental Protection Agency, American Public Health Association Standard Methods, ASTM International, International Organization for Standards, Environment and Climate Change Canada, British Columbia Ministry of Environment, and Ontario Ministry of Environment.

[9] Samples for chlorophyll-a analysis are also analysed for phaeophytin-a, a breakdown product of chlorophyll-a that, if necessary, may be used to support determination and interpretation of chlorophyll-a analysis results.

outlined previously (Section 2.2.2.2). An AEMP benchmark chlorophyll-a concentration of 3.7 µg/L was established for the Mary River Project (Baffinland 2015), and the 2024 chlorophyll-a data were compared to this benchmark to assess potential mine-related nutrient enrichment in waterbodies near the mine site. Chlorophyll-a concentrations were plotted to qualitatively assess changes in concentrations over time and among baseline (2005 to 2013 data) and mine operational years (2015 to 2024). If the chlorophyll-a concentration was significantly greater than the concentration observed in both a representative reference area and the respective baseline condition and/or if qualitative evaluation of temporal plots suggested a mine-related trend, further investigation was warranted to determine whether effects are mine-related. This includes use of a weight-of-evidence approach, including consideration of results from other study components (e.g., water chemistry).

2.4.2 Benthic Invertebrate Community

2.4.2.1 General Design

The CREMP BIC survey design outlines a habitat-based approach for characterizing potential mine-related effects to benthic biota of lotic and lentic environments (Baffinland 2015). Lotic areas sampled for benthic invertebrates included CLT1 and CLT2 at historically established areas located upstream and downstream of the Tote Road; SDLT1, SDLT9, and SDLT12 near their respective outlets; and Mary River upstream (two areas; G0 series stations) and downstream (three areas; E0 and C0 series stations) from the mine site (Table 2.7; Figure 2.3)[10]. Beginning in 2016, to augment the original CREMP study design, BIC samples have also been collected at a reference stream, referred to as Unnamed Reference Creek herein, located within the same unnamed tributary to Angijurjuk Lake that is used for reference water quality sampling (Stations CLT-REF4 and MRY-REF2; Tables 1.1 and 2.6; Figure 2.3). Additionally, BIC data are collected every three years as a part of the Environmental Effects Monitoring (EEM) program at MRTF, both downstream (effluent-exposed) and upstream (reference) of the primary mine effluent discharge. The Phase 3 EEM was conducted concurrently with the CREMP in 2023, and BIC results were summarized in the 2023 CREMP report (Minnow 2024b. In alignment with the federal EEM program, the CREMP included sampling at five BIC stations in each lotic study area, with the exception of SDLT12, where only three stations are typically sampled due to limited suitable habitat. Similar to 2023, BIC sampling was not conducted at SDLT12 in 2024 because no streamflow was present during the CREMP BIC sampling window in August. As in studies conducted from 2015 to 2023, the level of replication used for lotic benthic sampling in 2024 was

[10] In 2016 and routinely beginning in 2021, BIC sampling area CLT1-L2 was included at CLT1. Aqueous total aluminum and total iron AEMP benchmarks were exceeded at this location in past studies (Minnow 2021b). As a result, BIC monitoring was added at this location to evaluate possible effects on biota in this portion of the CLT1 system.

Table 2.7: Stream and River Benthic Invertebrate Community Monitoring Station Identifiers and Coordinates Used for the Mary River Project CREMP, 2024

Lake System	Waterbody	Station Code	Station Type	UTM Zone 17W, NAD83		Benthic Invertebrate
				Easting	Northing	
Angijurjuk Lake	Unnamed Tributary	REF-CRK-B1	Reference	570025	7906149	✓
		REF-CRK-B2	Reference	570060	7906116	✓
		REF-CRK-B3	Reference	570093	7906111	✓
		REF-CRK-B4	Reference	570121	7906100	✓
		REF-CRK-B5	Reference	570137	7906087	✓
Camp Lake	Camp Lake Tributary 1	CLT1-US-B1	Lightly Mine-Exposed	558502	7914968	✓
		CLT1-US-B2	Lightly Mine-Exposed	558488	7914964	✓
		CLT1-US-B3	Lightly Mine-Exposed	558494	7914931	✓
		CLT1-US-B4	Lightly Mine-Exposed	558509	7914904	✓
		CLT1-US-B5	Lightly Mine-Exposed	558517	7914891	✓
		CLT1-L2-B1	Mine-Exposed	558670	7914728	✓
		CLT1-L2-B2	Mine-Exposed	558662	7914737	✓
		CLT1-L2-B3	Mine-Exposed	558657	7914742	✓
		CLT1-L2-B4	Mine-Exposed	558642	7914753	✓
		CLT1-L2-B5	Mine-Exposed	558613	7914782	✓
		CLT1-DS-B1	Mine-Exposed	557710	7914978	✓
		CLT1-DS-B2	Mine-Exposed	557693	7914958	✓
		CLT1-DS-B3	Mine-Exposed	557686	7914945	✓
		CLT1-DS-B4	Mine-Exposed	557678	7914933	✓
		CLT1-DS-B5	Mine-Exposed	557671	7914918	✓
	Camp Lake Tributary 2	CLT2-US-B1	Lightly Mine-Exposed	557441	7915292	✓
		CLT2-US-B2	Lightly Mine-Exposed	557451	7915276	✓
		CLT2-US-B3	Lightly Mine-Exposed	557449	7915252	✓
		CLT2-US-B4	Lightly Mine-Exposed	557441	7915238	✓
		CLT2-US-B5	Lightly Mine-Exposed	557423	7915216	✓
		CLT2-DS-B1	Mine-Exposed	557392	7915105	✓
		CLT2-DS-B2	Mine-Exposed	557398	7915054	✓
		CLT2-DS-B3	Mine-Exposed	557400	7915033	✓
		CLT2-DS-B4	Mine-Exposed	557383	7914995	✓
		CLT2-DS-B5	Mine-Exposed	557377	7914972	✓
Sheardown Lake Northwest (NW)	Sheardown Lake Tributary 1 (Reach 1)	SDLT1-R1-B1	Mine-Exposed	560349	7913537	✓
		SDLT1-R1-B2	Mine-Exposed	560337	7913521	✓
		SDLT1-R1-B3	Mine-Exposed	560328	7913508	✓
		SDLT1-R1-B4	Mine-Exposed	560320	7913498	✓
		SDLT1-R1-B5	Mine-Exposed	560317	7913494	✓
	Sheardown Lake Tributary 12	SDLT12-B1	Mine-Exposed	561026	7912969	- [a]
		SDLT12-B2	Mine-Exposed	561002	7912976	- [a]
		SDLT12-B3	Mine-Exposed	560953	7912989	- [a]
Sheardown Lake Southeast (SE)	Sheardown Lake Tributary 9	SDLT9-DS-B1	Mine-Exposed	561826	7911839	✓
		SDLT9-DS-B2	Mine-Exposed	561814	7911825	✓
		SDLT9-DS-B3	Mine-Exposed	561798	7911825	✓
		SDLT9-DS-B4	Mine-Exposed	561785	7911817	✓
		SDLT9-DS-B5	Mine-Exposed	561767	7911813	✓
Mary Lake	Mary River	G0-09-B1	Reference	571447	7917012	✓
		G0-09-B2	Reference	571479	7916947	✓
		G0-09-B3	Reference	571489	7916920	✓
		G0-09-B4	Reference	571499	7916884	✓
		G0-09-B5	Reference	571503	7916859	✓
		G0-03-B1	Mine-Exposed	566490	7912607	✓
		G0-03-B2	Mine-Exposed	566499	7912623	✓
		G0-03-B3	Mine-Exposed	566489	7912628	✓
		G0-03-B4	Mine-Exposed	566444	7912613	✓
		G0-03-B5	Mine-Exposed	566429	7912643	✓
		E0-01-B1	Mine-Exposed	562944	7912281	✓
		E0-01-B2	Mine-Exposed	562922	7912214	✓
		E0-01-B3	Mine-Exposed	562806	7912171	✓
		E0-01-B4	Mine-Exposed	562778	7912165	✓
		E0-01-B5	Mine-Exposed	562717	7912159	✓
		E0-20-B1	Mine-Exposed	561930	7911461	✓
		E0-20-B2	Mine-Exposed	561895	7911448	✓
		E0-20-B3	Mine-Exposed	561856	7911419	✓
		E0-20-B4	Mine-Exposed	561848	7911409	✓
		E0-20-B5	Mine-Exposed	561841	7911394	✓
		C0-05-B1	Mine-Exposed	558391	7909181	✓
		C0-05-B2	Mine-Exposed	558387	7909185	✓
		C0-05-B3	Mine-Exposed	558429	7909212	✓
		C0-05-B4	Mine-Exposed	558441	7909161	✓
		C0-05-B5	Mine-Exposed	558357	79099211	✓

Notes: "✓" = station is sampled for specified sample type; "-" = station is not sampled for specified sample type.
[a] Sample not collected in 2024 due to limited or no appropriate habitat (i.e., the stream was not flowing [SLDT12]).

greater than specified under the original CREMP design for consistency with EEM standards (Minnow 2016a). Where possible, the same station locations used in previous studies were sampled in 2024 to maintain continuity across historical baseline, construction, and operational period studies.

In lentic environments, BIC sampling was conducted at the 40 previously established stations described in the CREMP study design; these are distributed among the four mine-exposed study lakes (i.e., 10 stations in each of Camp, Sheardown Lake NW, Sheardown Lake SE, and Mary lakes), as well as at the same 10 stations established at REF-03 during the 2015 study (Table 2.5, Figures 2.2 and 2.3). Analysis of BIC data collected at REF-03 from 2015 to 2020 indicated that, similar to temperate lakes (Ward 1992), depth-related influences on BIC structure (such as density and richness) naturally occur in lakes within the study region (Minnow 2016a, 2018, 2019, 2020, 2021b, 2022). Due to the natural depth-related variations in BIC, the BIC stations at each mine-exposed and reference lake were categorized as either littoral zone (2 to 12 m depth) or profundal zone (greater than [>]12 m depth) stations, based on their respective depths (Table 2.5). To the extent possible, five littoral and five profundal stations were designated for each study lake based on the previously established suite of CREMP lentic benthic stations[11]. This approach ensured temporal continuity with the baseline studies and aligned with the original CREMP design (Table 2.5, Figures 2.2 and 2.3), while also facilitating data analysis in accordance with EEM standards. An ecologically meaningful difference in BIC metrics is defined as a MOD between the mine-exposed and reference area that exceeds a critical effect size (CES) for BIC metrics (CES_{BIC}) of ± 2 reference area standard deviations (SD_{REF}; Environment Canada 2012). This threshold is analogous to differences that would be expected beyond natural variability between two areas uninfluenced by anthropogenic inputs (i.e., between pristine reference areas; see Munkittrick et al. 2009; Environment Canada 2012). Therefore, differences beyond the CES_{BIC} are considered greater than those that would naturally occur (i.e., between two pristine reference areas). Sampling five stations from each zone at each study area provided adequate statistical power to detect ecologically meaningful differences in BIC metrics, as defined, using an equal α and beta (β) of 0.10 (Environment Canada 2012).

[11] At Sheardown Lake SE, depths > 12 m are spatially limited, so the five deepest CREMP stations were designated as profundal, despite one of them being less than (<) 12 m deep. At Mary Lake, six of the CREMP stations were located at depths > 12 m and were therefore designated as profundal, whereas the remaining four stations were designated as littoral.

2.4.2.2 Sample Collection and Laboratory Analysis

Two types of equipment and methods were used during the 2024 CREMP BIC survey to sample the lotic and lentic habitats, as follows:

- at **lotic stations** (i.e., areas with predominantly cobble and/or gravel substrate in flowing waters), BIC samples were collected using a Surber sampler (0.0929 m^2 sampling area) equipped with 500-µm mesh. At each erosional station, one composite sample was collected by combining three Surber sampler grabs (i.e., 0.279 m^2 area) to ensure adequate representation of the habitat. A concerted effort was made to ensure that water velocity and substrate characteristics were comparable between the respective lotic mine-exposed and reference area stations, minimizing natural influences on community variability. Once all three sub-samples were collected at each station, the material gathered in the Surber sampler net was transferred to a plastic sampling jar, which was labelled both externally and internally with the station identifier while working over a catch-tote.
- at **lentic stations** (i.e., areas with predominantly soft silt-sand, silt, and/or clay substrates with variable amounts of organics), BIC sampling was conducted using a Petite Ponar grab sampler (15.24 x 15.24 cm; 0.023 m^2 sampling area). A single composite sample, consisting of five grabs (i.e., 0.115 m^2 sampling area), was collected at each station, ensuring that each grab was acceptable (i.e., captured enough surface material to fill to the edges of the Petite Ponar). Any incomplete grabs were discarded. For each acceptable grab, the Petite Ponar was thoroughly rinsed, and the material was then field-sieved through 500-µm mesh. After sieving all five grabs, the retained material was carefully transferred into a plastic sampling jar, which was labelled externally and internally with the station identifier while working over a catch-tote.

Following collection, the BIC samples were preserved in 10% buffered formalin in ambient water. At lotic stations, supporting measurements and information, including sampling depth, water velocity, and a description of the presence of aquatic vegetation/algae, were collected at each replicate grab location. Additionally, *in situ* water quality measurements at the bottom of the water column, as well as GPS coordinates, were collected at each lotic BIC station. For each lake BIC station, supporting information recorded included substrate description, presence of aquatic vegetation/algae, sampling depth, *in situ* water quality at both the surface and bottom of the water column, and GPS coordinates.

Benthic samples were submitted to Zeas Inc. (Nobleton, Ontario) for processing, where standard sorting, identification, and counting methods were applied, as described in Environment

Canada 2014. Upon arrival at the laboratory, a biological stain was added to each sample to enhance sorting accuracy. The samples were first washed free of formalin in a 500-µm sieve, and the remaining sample material was then examined under a stereomicroscope at a magnification of at least 10 times. Benthic invertebrates were carefully removed from the sample debris and placed into vials containing 70% ethanol, organized by major taxonomic groups (typically at the order or family level). A senior taxonomist later identified and enumerated the organisms to the lowest practical level (usually genus or species) using up-to-date taxonomic keys. QA/QC procedures used during the laboratory processing included organism recovery and sub-sampling checks on up to 10% of the total samples collected for the 2024 CREMP (Appendix A).

2.4.2.3 Data Analysis

BIC data were evaluated separately for lotic, lentic-littoral, and lentic-profundal habitat datasets. BIC data were assessed using summary metrics, including mean invertebrate density (i.e., average number of organisms per m^2), mean taxonomic richness (number of taxa identified to lowest practical level), and Simpson's Evenness Index. Simpson's Evenness was calculated using the Krebs method (Smith and Wilson 1996). Additional analyses were performed using percent composition of dominant/indicator taxa, functional feeding groups (FFG), and habit preference groups (HPG). Percent composition of taxa and groups were calculated as the raw abundance of each respective group relative to the total number of organisms in the sample. Dominant/indicator taxonomic groups were defined as those groups representing, on average, > 5% of total raw organism abundance for a study area or any groups considered to be important indicators of environmental stress. The FFG and HPG were assigned based on Pennak (1989), Mandaville (2002), and/or Merritt et al. (2008) descriptions/designations for each taxon.

Statistical comparisons of BIC metrics and community composition endpoints were conducted using the same tests described for the *in situ* water quality comparisons (Section 2.2.2.2). Pairwise differences between the mine-exposed and reference areas were primarily tested using Student's t-tests on untransformed, normally distributed data. If the data were found to be non-normal, transformations including $\log_{10}$ and $\log_{10}(x+1)$ were applied, followed by re-evaluation for normality. The transformation that resulted in normal data with the highest p-value from a Shapiro-Wilks normality test was selected for analysis. In cases where normality could not be achieved through transformation, non-parametric Mann-Whitney U-tests were used for pairwise comparisons on rank transformation. Statistical comparisons were conducted using R programming (R Core Team 2023). A significant difference between BIC endpoints for any paired mine-exposed and reference areas was defined at a p-value of 0.10. For each endpoint that showed a significant difference, the MOD was calculated between study area means. Given that the benthic survey was designed to have sufficient power to detect a difference (effect size) of ± two SD, the MOD was expressed as the number of reference mean standard deviations (SD_{REF}), using equations provided by Environment Canada (2012). Metrics with MODs outside of the CES_{BIC} of ± 2 SD_{REF} indicated ecologically meaningful differences, and further evaluation was completed to determine whether the differences were mine-related. This evaluation considered the direction of the response and used a weight-of-evidence approach, incorporating results from other study components (e.g., water chemistry) and BIC endpoints.

The Bray-Curtis Index was used to evaluate community-level differences between study areas, with calculations and statistical assessments following the procedures recommended for federal EEM studies (see Borcard and Legendre 2013). Specifically, pairwise community-level differences between study areas were assessed using ln-transformed density data, and homogeneity of group variance was calculated according to the PERMDISP2 procedure (Anderson 2006). To further investigate differences in community structure, a Mantel Test and distance-based Redundancy Analysis (dbRDA) were applied using R statistical software (as per Borcard and Legendre 2013).

Temporal comparisons included statistical evaluations of primary BIC metrics (i.e., density, richness, Simpson's Evenness), dominant invertebrate groups, FFGs, and HPGs) between the baseline and operational period (2015 to 2024) data. These evaluations were conducted using univariate tests (e.g., ANOVA) and pairwise *post hoc* tests where appropriate. The temporal statistical comparisons followed the same tests, transformations, assumptions, and software as those used for the *in situ* water quality comparisons in the multiple group analysis (Section 2.2.2.2). As in the 2024 within-year statistical analyses, the MOD for endpoints that showed significant differences between years in the *post hoc* tests was calculated. This difference was then compared to the CES_{BIC}, defined as within ±2 SDs of the baseline year mean (abbreviated as ±2 $SD_{BL\text{-}year}$). A mine-related effect for temporal comparisons was assessed in a manner similar to spatial comparisons. An ecologically meaningful (significant) difference in BIC endpoint values between study years was concluded if the MOD fell outside of the CES_{BIC} of ± 2 SD of the respective baseline year mean, warranting further evaluation to determine whether the difference was mine-related. Again, this evaluation considered the direction of the response and used a weight-of-evidence approach, incorporating results from other study components (e.g., water chemistry) and BIC endpoints.

2.4.3 Fish Population

2.4.3.1 General Design

The CREMP fish population survey employs a non-lethal sampling design (Environment Canada 2012) to assess potential mine-related effects on the fish populations in mine-exposed lakes (Baffinland 2015). The survey targets arctic charr, as this species is the most abundant in the region's lakes and has sufficient baseline data for a before-after statistical evaluation. Arctic charr are also culturally significant, serving as a key subsistence food source for Inuit communities. The approach used in the CREMP survey closely aligns with the Environment Canada (2012) recommendations for non-lethal sampling. Specifically, the survey aims to collect approximately 100 arctic charr from nearshore lake habitats[12] and 100 arctic charr from littoral/profundal lake habitat[13]. Nearshore habitat sampling mainly captures small, juvenile fish, whereas littoral/profundal habitat sampling targets larger, sub-adult, and adult fish. By sampling both habitats, most age classes within the population are represented.

The fish survey focuses on four mine-exposed lakes: Camp Lake, Sheardown Lake NW, Sheardown Lake SE, and Mary Lake (Figure 2.4). These mine-exposed study lakes were sampled during 2002, 2005, 2006, 2007, 2008, and 2013[14] to establish baseline conditions. In 2015, Reference Lake 3 was added to the study and has been sampled for nearshore and littoral/profundal fish since (Figure 2.5). Overall, the sampling design allowed for statistical evaluation of potential health effects on arctic charr populations at mine-exposed lakes compared to a reference lake, as well as among baseline (2007 to 2013), construction (2014), and operational periods (2015 to 2024). Under the EEM, fish population surveys[15] are conducted every three years in the Mary River mine-exposed area and an unnamed tributary to Angijurjuk Lake, which serves as a reference area. The most recent Phase 3 EEM was completed in 2023, and its results were incorporated into the 2023 CREMP report (Minnow 2024a).

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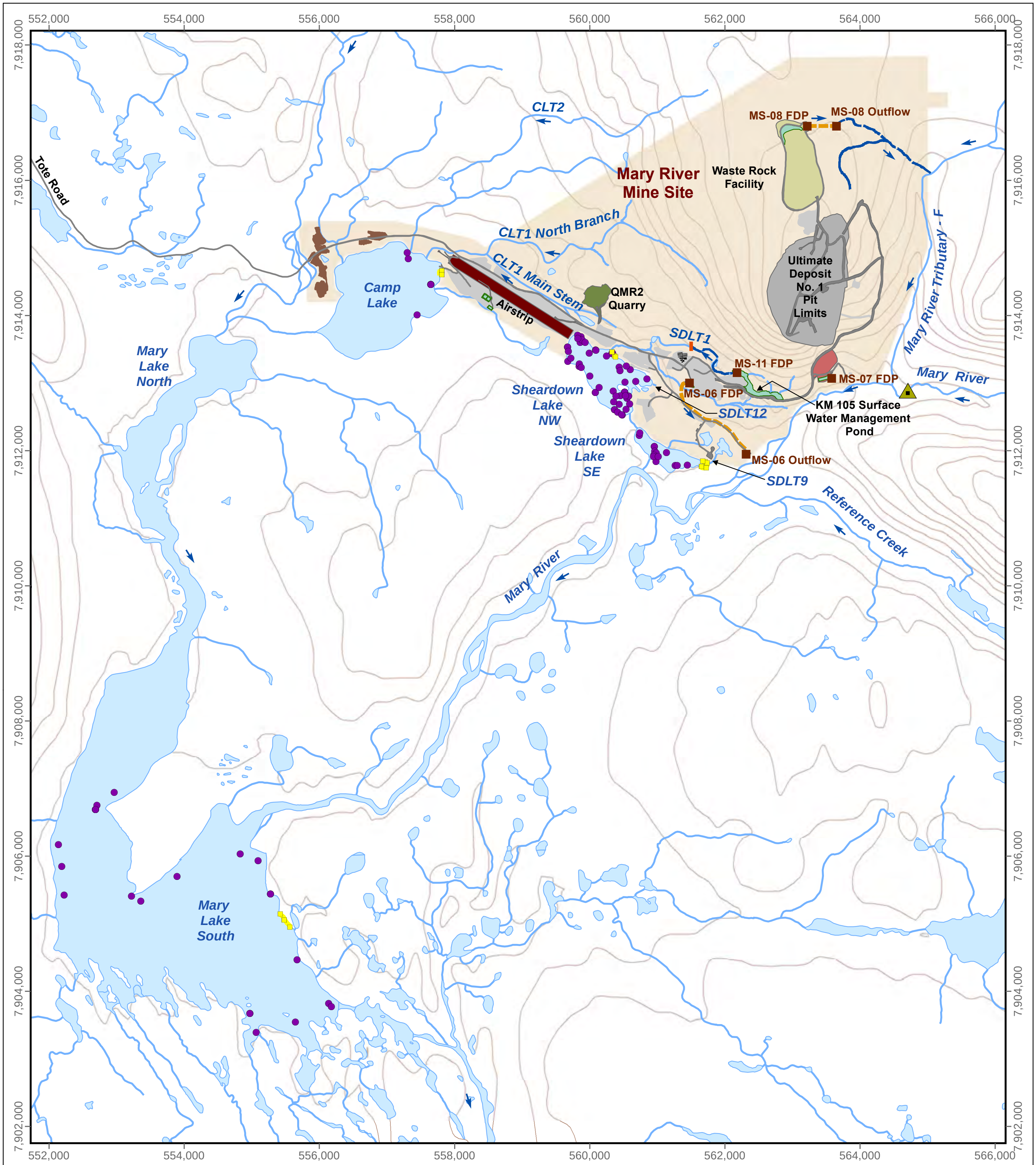
[12] Nearshore fish were collected from the lake shoreline using a backpack electrofisher. This method primarily captured arctic charr with fork lengths ranging from 2.5 cm to 18.8 cm in 2024, as well as small-bodied ninespine stickleback.

[13] Littoral/profundal fish were collected from the lake using gill nets with mesh sizes ranging from 38 to 76 mm (1.5" to 3"). This method primarily captured arctic charr with fork lengths ranging from 20.5 cm to 65.9 cm in 2024.

[14] Nearshore electrofishing baseline data were collected in 2013 in Camp Lake, in 2002/2005/2006/2008/2013 at Sheardown Lake NW, and in 2007 at Sheardown Lake SE. No nearshore electrofishing baseline data were collected at Mary Lake. Littoral/profundal gill netting baseline data were collected in 2006/2008/2013 at Camp Lake and Sheardown Lake NW, in 2007/2008 at Sheardown Lake SE, and in 2006/2007 at Mary Lake.

[15] The EEM fish survey included aspects of both lethal and non-lethal sampling designs to reflect the presence of fish in non-reproductive condition (e.g., juveniles) and the consequent inability to visually identify the sex of these individuals using external cues.



LEGEND

- Electrofishing
- Gill Netting
- Final Discharge Point (FDP)
- Mary River Cascade Barrier
- Electrofishing Path
- Fish Barrier
- Overland Surface Water Flow
- HDPE Pipe
- Airstrip
- Mine Site Complex
- Pond
- ROM Stockpile
- Road
- Open Pit
- Waste Rock Facility
- Borrow Pit
- Quarry

Mary River Project 2024 CREMP Fish Survey Sampling Locations

0 1.25 2.5 5 km

N E S W

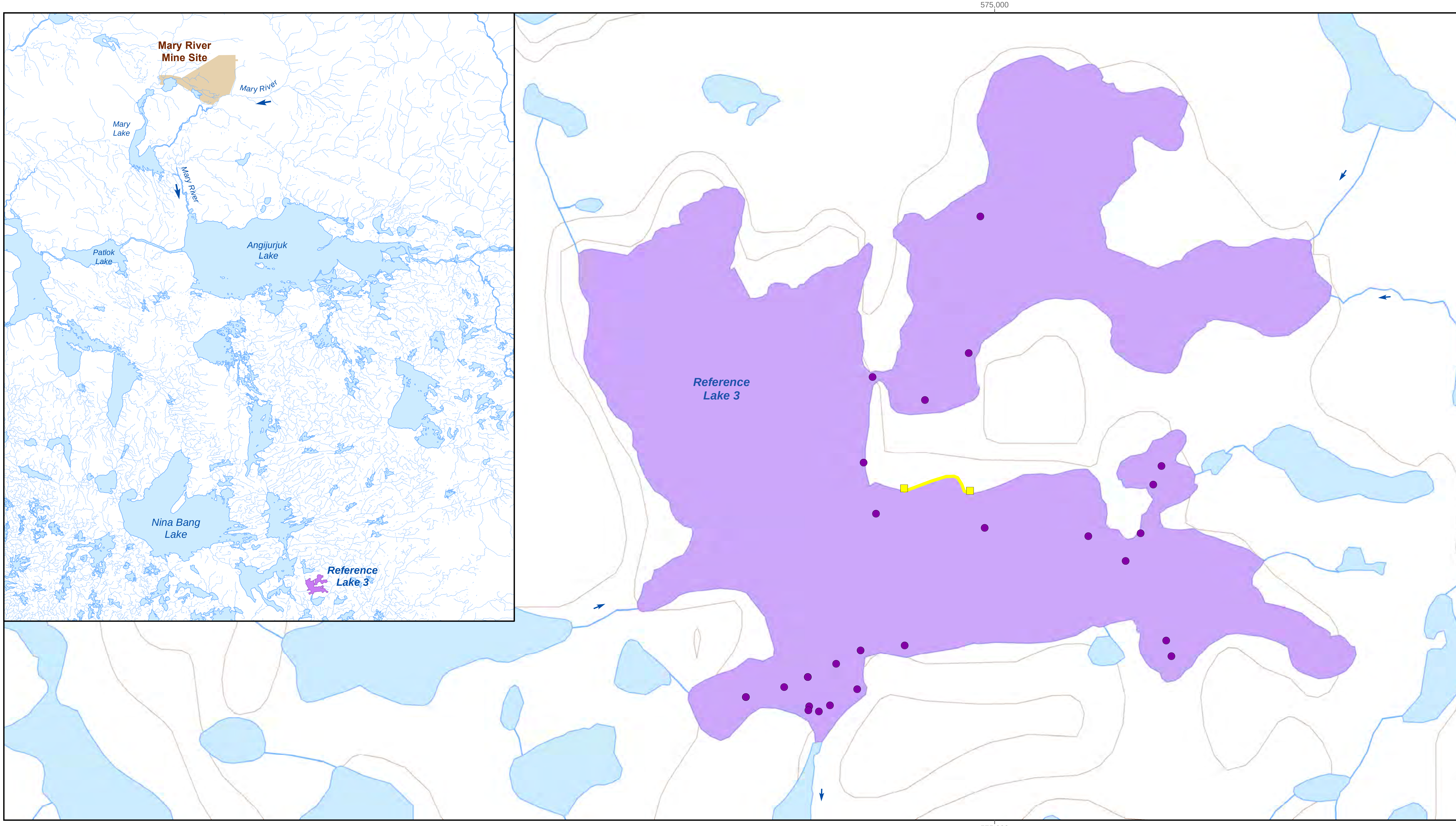
Map Projection: UTM Zone 17W NAD 1983
Data Source: Reproduced under licence from His Majesty the King in Rights of Canada, Department of Natural Resources Canada. All rights reserved.

Date: March 2025
Project 247202.0075

minnow environmental inc.
A Trinity Consultants Company

Figure 2.4

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 2.4 Mary River Project CREMP Mine Area Fish Survey Locations.mxd



LEGEND
- Gill Netting
- Electrofishing
- Electrofishing Path

0 200 400 800
Meters

Map Projection: UTM Zone 17N NAD 1983
Data Source: Reproduced under licence from His Majesty the King in Right of Canada, Department of Natural Resources Canada. All rights reserved.

Mary River Project CREMP Reference Lake 3 Fish Survey Locations

Date: March 2025
Project 247202.0075

minnow environmental inc.

Figure 2.5

Document Path: C:\Users\MLaPalme\Trinity Consultants, Inc\Baffinland Iron Mines - 247202.0075 - 2024 AEMP\D - GIS\CREMP\24-75 Figure 2.5 Mary River Project CREMP Reference Lake 3 Fish Survey Locations.mxd

2.4.3.2 Sample Collection

2.4.3.2.1 Nearshore Sampling

Nearshore areas of study lakes were sampled for arctic charr using a battery-powered backpack electrofishing unit (Model LR-24, Smith-Root Inc., Vancouver, Washington). An electrofishing team, consisting of the operator and a single netter, fished at up to two shoreline reaches per lake (Figures 2.4 and 2.5). The number of passes conducted at each reach was based on the initial catch success; additional passes were made if the target sample numbers were not achieved during the first pass. All fish captured during each pass were temporarily retained in buckets containing aerated water. At the conclusion of each pass, fishing effort (i.e., electrofishing seconds) was recorded to facilitate the calculation of time standardized catch rates. Captured fish were identified to species, enumerated, and any non-target species were released alive at the site of capture. All captured arctic charr were temporarily retained for processing using methods described in Section 2.4.3.3. Additional supporting data were collected during each electrofishing pass, including GPS coordinates at the boundaries of each electrofishing reach and a description of the habitat within the reach.

2.4.3.2.2 Littoral/Profundal Sampling

Littoral/profundal areas of the study lakes were sampled for arctic charr using experimental (gang index) gill nets. The gill nets were multi-panel, 2 m high, between 61 to 91 m (200’ to 300’) total length, and made up of bar mesh sizes ranging from 38 to 76 millimetres (mm; 1.5” to 3”). These nets were set on the lake bottom for short durations, ranging from 0.49 to 1.36 hours per set, with an average set duration of 0.89 hours during daylight hours. Upon retrieval, all captured fish were identified to species, enumerated, and processed separately according to the mesh size of the gill net panels. Processing details are described in Section 2.4.3.3. For each gill net set, the following information was recorded: net length, mesh size, duration of sampling, sampling depth range, GPS coordinates, and habitat descriptions.

2.4.3.3 Field and Laboratory Processing

Following completion of each electrofishing pass and retrieval of each individual gill net panel, all captured arctic charr were processed in the field. The external condition of each fish (live and incidental mortalities) was visually assessed for deformities, erosions, lesions, or tumours (DELT) and evidence of external and/or internal parasites. All observations were recorded on field sheets, and supporting photographs taken. Each fish was then measured for both fork and total length to the nearest mm using a standard measuring board. Fish captured by electrofishing were individually weighed to the nearest milligram (mg) using an Ohaus Model 123 Scout-Pro analytical balance (Ohaus Corp., Pine Brook, NJ) with a draft shield for accuracy. Fish captured by gill net were weighed using Pesola™ spring scales (Pesola AG, Baar, Switzerland), which are accurate to within ± 0.3% of the fish's mass. The scale was selected so that the weight of the fish was near the top of the scale's range to ensure a measurement resolution near 1%. Once measurements were taken, all live arctic charr, whether captured by electrofishing or gill netting, that were not selected for aging structure collection, were released near the location of capture. In line with the EEM guidelines (Environment Canada 2012) for non-lethal fish population surveys, approximately 10% of the arctic charr captured by electrofishing were sacrificed for collection of aging structures. The whole body of sacrificed fish and any incidental mortalities during electrofishing were retained for age determination. The fish were placed in a labelled WhirlPak™ bag with a unique fish identifier and frozen for storage before being shipped on ice to the aging laboratory.

The aging structures (otoliths) were extracted from whole body fish by North/South Consultants Inc. (Winnipeg, Manitoba) for age determination. The otoliths were processed by embedding the cleaned structures in epoxy resin, which was allowed to harden. Once hardened, the otoliths were sectioned near the center and mounted on glass slides using a mounting medium. The age from each otolith was determined by counting the annuli under a compound microscope using transmitted light. The age was recorded along with a condition index that assessed both qualitative characteristics (e.g., pattern clarity) and quantitative characteristics (e.g., repeatability of age assignment by the aging technician). A subsample of 10% of the otolith structures was aged by a different technician to ensure accuracy and consistency, as part of the QA/QC process (Appendix A).

2.4.3.4 Data Analysis

Fish community data from both the mine-exposed and reference areas were described based on total catch and catch-per-unit-effort (CPUE) for each sampling method. Electrofishing CPUE was calculated as the number of fish captured per electrofishing minute for each lake's nearshore area. Gill netting CPUE was determined by the number of fish captured per 100 metre·hours of net deployed in the littoral/profundal area of each study lake. Temporal comparisons of fish community assemblages were made qualitatively, using electrofishing CPUE and gill netting CPUE, to assess changes in fish catches at study lakes. These comparisons examined shifts in fish community composition/structure between baseline conditions and the years of mine operation from 2015 to 2024.

Health endpoints for arctic charr populations were assessed separately for nearshore and littoral/profundal datasets. Initial data analysis involved plotting length-frequency distributions from the nearshore dataset to differentiate young-of-the-year (YOY) individuals from older juvenile/adult age classes (i.e., non-YOY). Size and age classes were assessed and assigned qualitatively through visual inspection of length-frequency distributions. Where distinct gaps were observed in the length distribution between age-0 and age-1+ individuals, the upper size boundary of the age-0 group was designated as the YOY cutoff. This approach was applied individually to each lake dataset. A potential source of error in visually determining the YOY cutoff is an overlap in length between age-0 and age-1+ individuals, especially in populations with variable growth rates. This can lead to misclassification when size classes within length-frequency distributions are not clearly distinct. In these cases, supporting fish weight data were also used to further inform distinctions between age-0 and age-1+ individuals. Subjectivity in visual assessment may introduce inconsistencies across datasets or observers.

Lake nearshore fish survey endpoints were analysed separately for the YOY and non-YOY age classes with YOY analyses only completed if sample size permitted (i.e., n ≥ 10). Fish size endpoints, including fork length and fresh body weight, were summarized by calculating and reporting mean, median, minimum, maximum, SD, standard error, and sample size by age class within each study area. These measurement endpoints formed the basis for evaluating four key response categories, survival, growth, reproduction, and energy storage, as outlined by Environment Canada (2012) for EEM (Table 2.8). Relative length-frequency distributions were compared between the mine-exposed lakes and the reference lake using 2024 data (i.e., control-impact analysis), as well as between the combined baseline period and 2024 for individual lakes (i.e., before-after analysis), employing a non-parametric two-sample Kolmogorov Smirnov (K-S) test. To assess potential differences in reproductive success between paired study areas, the proportion of YOY arctic charr in the mine-exposed and reference areas was evaluated. Additionally, KS test results were compared both with and without the inclusion of YOY individuals in the datasets to determine if there was an approach-based impact on results and conclusions regarding reproductive success.

Mean fork length and body weight were compared between mine-exposed and reference study areas using data collected in 2024, as well as between the mine baseline period and 2024, separately for each study lake. Prior to statistical testing, the data were evaluated for normality and homogeneity of variance to determine the appropriateness of parametric statistical tests such as ANOVA. In cases where data did not meet ANOVA assumptions, even after log transformation, a non-parametric Mann-Whitney U-test was used to test for differences between study areas or study periods. Body weight at fork length (condition) was compared using Analysis-of-Covariance (ANCOVA). Prior to conducting the ANCOVA tests, scatter plots of all variable and covariate combinations were reviewed to identify outliers, high-leverage values, or other unusual data points. These plots were also examined to ensure adequate overlap between the 2024 data for mine-exposed and reference areas, as well as between the 2024 mine-exposed and baseline data, and to confirm a linear relationship between the variable and

Table 2.8: Fish Population Survey Endpoints Examined for the Mary River Project CREMP, 2024

Response Category	Endpoint	Statistical Procedure[c,d]	Critical Effect Size
Survival	Length-frequency distribution[a]	K-S Test	not applicable
Energy Use (size)	Size (fresh body weight)[b]	ANOVA	25%
	Size (fork length)[b]	ANOVA	25%
Energy Use (reproduction)	Relative abundance of YOY (% composition)[b]	K-S Test	not applicable
Energy Storage	Condition (body weight against length)[a]	ANCOVA	10%

Notes: YOY = young-of-the-year; ANOVA = analysis of variance; ANCOVA = Analysis of covariance, K-S Test = Kolmogorov-Smirnov test.

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

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

[c] ANOVA used except for non-normal data, where Mann Whitney U-tests were used.

[d] For the ANCOVA analyses, the first term in parentheses is the endpoint (dependent variable Y) that is analyzed to identify a potential effect. The second term in parentheses is the covariate, X (age, weight, or length).

the covariate. To verify the linearity of the relationship, linear regression analysis was performed for each area, with evaluation at an alpha (α) level of 0.05. If no significant linear relationship was found between the variable and covariate for the 2024 mine-exposed area and reference or baseline data, ANCOVA was not conducted.

Once it was determined that ANCOVA could be used for statistical analysis, the first step was to test whether the slopes of the regression lines between datasets were equal. This was done by including an interaction term (dependent × covariate) in the ANCOVA model and evaluating its significance. If the interaction term was significant, it indicated that the regression slopes were not equal between the datasets, and the resulting ANCOVA would provide spurious results. In cases where the interaction term was significant, the process for determining whether a full ANCOVA could proceed involved the following steps: 1) removal of influential points using Cook's distance and re-assessing the equality of slopes; and/or, 2) evaluating Coefficients of Determination (r^2), which considered slopes equal regardless of the interaction effect (Environment Canada 2012). For the Coefficients of Determination, the full ANCOVA was conducted to test for main effects. If the r^2 value of both the parallel regression model (with the interaction term) and full regression model were > 0.8 and within 0.02 units of each other, the parallel-regression ANCOVA model was considered valid (Environment Canada 2012). If neither method was acceptable, a statistically significant interaction effect (indicating unequal slopes) was noted, and the magnitude of effect was estimated at both the minimum and maximum overlap of covariates between the areas (Environment Canada 2012). If the interaction term was not significant (i.e., slopes were homogeneous between the two populations), the full ANCOVA model was run without the interaction term to test for differences in adjusted means between the two datasets. The adjusted mean was then used as an estimate of the population mean, accounting for the value of the covariate in the ANCOVA model.

For endpoints showing significant differences, the MOD between the 2024 mine-exposed and reference data or between 2024 mine-exposed and baseline data was calculated as outlined by Environment Canada (2012). This calculation used either mean values (for ANOVA), adjusted mean values (for ANCOVA with no significant interaction), or predicted values (for ANCOVA with a significant interaction). In cases where the endpoint values were $\log_{10}$transformed, the anti-log of the mean, adjusted mean, or predicted mean value was used in the equations. If no significant difference was found between datasets, the minimum detectable effect size was calculated as a percent difference from the reference mean or mine-exposed baseline mean for ANOVA, or from the adjusted reference mean or mine-exposed baseline mean for ANCOVA, at α and beta (β) level of 0.10. The square root of the mean square error (calculated during the ANOVA or ANCOVA procedures) was used as a measure of variability in the sample population, based on the formula provided by Environment Canada (2012). If outliers or high-leverage values were identified upon examination of scatter plots and residuals, those values were removed, and both ANOVA or ANCOVA tests were repeated using the reduced dataset. Similar to the CES applied in the BIC survey, a MOD of ± 10% was applied for condition (CES_C) to define ecologically relevant differences, in alignment with EEM guidelines (Table 2.8; Munkittrick et al. 2009; Environment Canada 2012).

Differences beyond the CES_C are considered greater than those that would naturally occur (i.e., between two pristine reference areas), and therefore warranted further evaluation to determine whether the difference was mine-related. This assessment considered the direction of the response and used a weight-of-evidence approach, incorporating results from other study components (e.g., water and sediment chemistry, phytoplankton, and BIC).

Finally, an *a priori* power analysis was conducted to determine the appropriate fish sample sizes for future surveys, as recommended by Environment Canada (2012). These analyses were based on the mean square error values generated during the ANOVA or ANCOVA procedures, with both α and β set equally at 0.10. The power analysis was based on two key assumptions: 1) representativeness of populations: it was assumed that fish caught in both the mine-exposed and reference areas in 2024, or at mine-exposed areas in 2024 and baseline periods, were representative of the larger population. This assumes similar distribution and variance for the examined parameters across these areas; 2) population stability: it was assumed that the characteristics of the fish populations would not change substantially between the 2024 survey and the next study. The results of the power analysis were reported as the minimum sample size (number of fish per area) required to detect a specified MOD (effect size) between the mine-exposed and reference areas (or between the mine exposed areas in 2024 and baseline) for each endpoint. The MOD was expressed as a percentage increase or decrease relative to the reference area/baseline mean for each endpoint, calculated using the observed pooled SD of the residuals from ANOVA or parallel slope ANCOVA model.

2.5 Effects Assessment

2.5.1 2024 CREMP Objective and Approach

2.5.1.1 2024 Effects Determination

An effects determination was conducted for all key waterbodies in the Camp Lake, Sheardown Lake, Mary River, and Mary Lake systems. This included a summary of instances where Mary River Project AEMP benchmarks for water and sediment quality were exceeded at waterbodies examined under the CREMP. A mine-related influence on water or sediment quality for a waterbody was determined if water or sediment quality parameters were consistently elevated relative to both reference and baseline conditions at mine-exposed areas across all sampling seasons, or in a single season in 2024 and/or qualitative evaluation suggested an increasing temporal trend.

Determining a mine-related effect on aquatic biota (e.g., phytoplankton, BIC, fish) involved a weight-of-evidence approach (Figure 2.6). This considered incidences where AEMP benchmarks were exceeded and/or mine-related influences were identified in water and sediment quality, alongside corroboration of adverse effects on aquatic biota from biological monitoring results, as described in Sections 2.4.1.2, 2.4.2.3, and 2.4.3.2. The effects determination aimed to identify potential biological effects at these waterbodies in 2024 and, where appropriate, provide recommendation(s) for future study to assist Baffinland in making informed decisions about management actions (Figure 2.6).

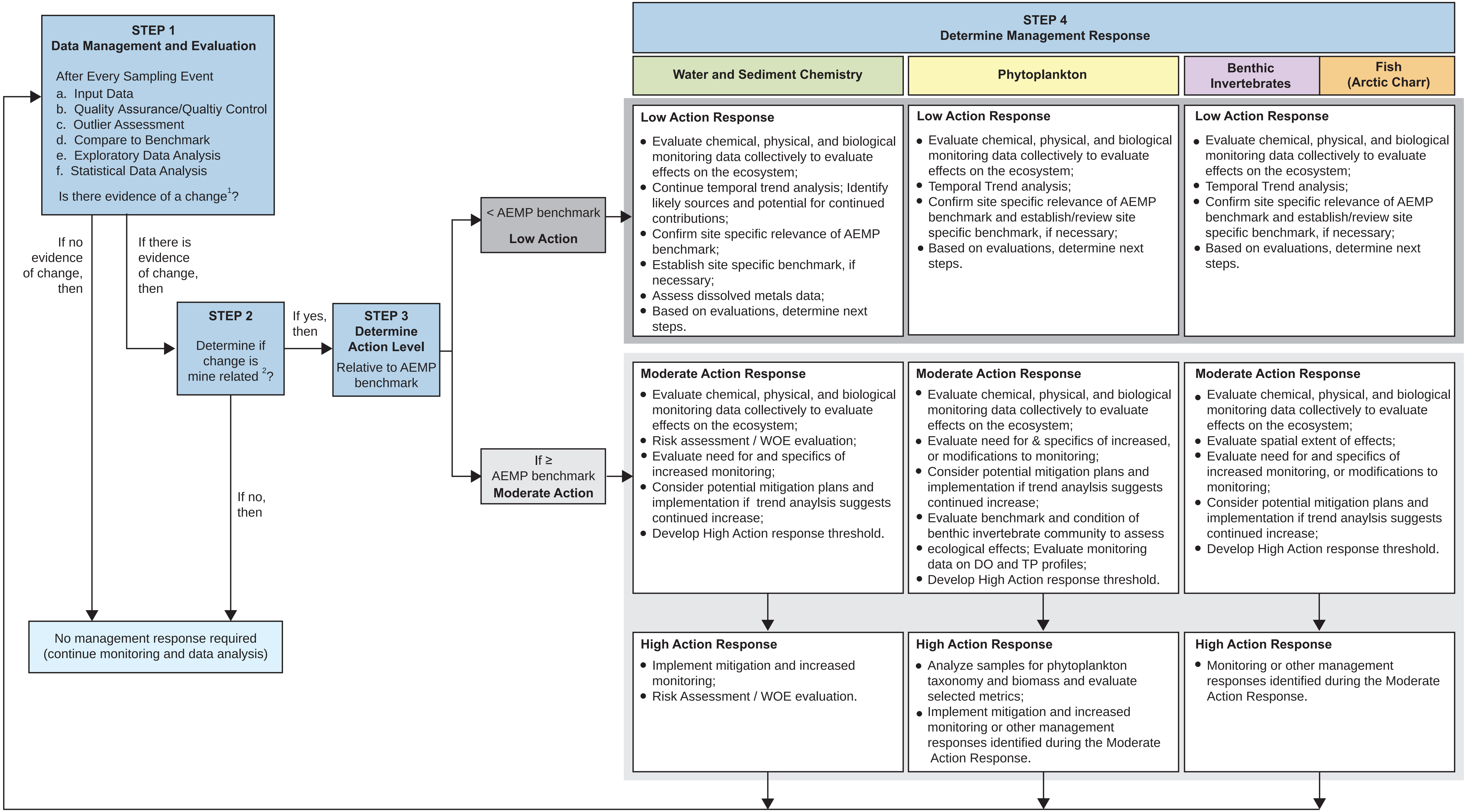
2.5.1.2 Comparisons to FEIS Predictions

The results of the 2024 CREMP were also compared to the effects predictions outlined in the Final Environmental Impact Statement (FEIS) for the Project (Baffinland 2012). Water and sediment quality data from the 2024 CREMP were compared to thresholds established in the FEIS for predicting the magnitude of potential effects on water quality from mine contact water and site runoff. Parameters were considered to conform to FEIS predictions if their concentrations fell within the predicted ranges based on the ‘magnitude of effect’ significance ratings for residual water and sediment quality impacts[16].

Water quality data were compared to FEIS predictions for the following non-point source emissions: SWSQ-2 (Site Water Management; all stations), SWSQ-4 (Explosives; CLT1 Upper and Lower Main Stems, and SDLT9 and Sheardown Lake SE), SWSQ-5 (Quarries and Borrow Areas; CLT1 Upper and Lower Main Stems), SWSQ-7 (Camps and Fuel Management; Camp Lake and Sheardown Lake systems), and SWSQ-9 (Airstrip and Airstrip Use; Camp Lake and Sheardown Lake Systems). In each case, water quality data were compared to a Level II magnitude of effect rating, as predicted in the FEIS for these specific sources.

Sediment quality data were compared to FEIS predictions for the effects of airborne emissions (i.e., fugitive dust; FEIS Issue SWSQ-17-3) on the Camp, Sheardown, and Mary lake systems. The sediment data were assessed against ‘magnitude of effect’ significance ratings of negligible (chromium, copper, lead, and zinc), Level I (nickel), Level II (arsenic and cadmium), and Level III (iron), as predicted for the airborne emissions in the FEIS.

[16] Significance ratings were: Negligible – concentrations of indicator(s) predicted to be less than threshold value(s); Level I - concentrations of indicator(s) predicted to be above but within an order of magnitude of threshold value(s) (1 to 10x the threshold); Level II - concentrations of indicator(s) predicted to exceed threshold value(s) by an order of magnitude or greater (10 to 100x the threshold); Level III - concentrations of indicator(s) predicted to exceed threshold value(s) by more than two orders of magnitude (greater than 100x the threshold).



Notes:

AEMP = Aquatic Effects Monitoring Plan; WOE = Weight of evidence; DO = dissolved oxygen; TP = total phosphorus.

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

Baffinland Mary River Project AEMP Data Assessment Approach and Response Framework

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Figure 2.6

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The FEIS fish health and condition predictions were based on water and sediment quality parameters meeting specific thresholds. Therefore, the 2024 CREMP water and sediment quality data were used to assess the conformance of arctic charr health and condition with the FEIS predictions (i.e., it was determined whether water and sediment quality were within the predicted 'magnitude of effect' significance ratings[17,18]). Water quality data were compared to predictions for the effects on freshwater biota (arctic charr health and condition) linked to the following project activities:

- Ore and waste rock dust generation and dispersion: sediment quality changes (Level II)
- Discharge of east waste rock, ore stockpile runoff, and pit water/run of mine stockpile to the Mary River (Level II); and
- Aqueous non-point sources (Level I).

Comparisons for effects on direct mortality of arctic charr were addressed in the Lake Sedimentation Monitoring Program 2023/2024 report (Minnow 2025), and were assessed based on sediment accumulation thicknesses not exceeding 1 mm/year in Sheardown Lake NW.

2.5.2 Implementation of 2023 Effects Assessment Recommendations

The effects Assessment Approach outlined above[19] was applied to data collected in 2023 for the Mary River Project 2023 CREMP, which led to development of recommendations for addressing instances where AEMP benchmarks for water quality and sediment quality were exceeded and/or a potential mine-related influence was determined for a water or sediment quality parameter (Minnow 2024a). A summary of effect determinations based on 2023 CREMP results (Minnow 2024a), along with recommendations and action-level responses implemented by

[17] Significance ratings for water quality were: Not Assessed (Level 0) – water quality change within the Canadian Council of Ministers of the Environment (CCME) water quality guidelines for the Protection of Aquatic Life (PAL) or other water quality thresholds; Level I – water quality change 1 to 10x the CCME PAL guidelines or other water quality thresholds; Level II – water quality change 10 to 100x the CCME PAL guidelines or other water quality thresholds; Level III – water quality change >100x the CCME PAL guidelines or other water quality thresholds (CCME 2011).

[18] Significance ratings for sediment quality were: Not Assessed (Level 0) – sediment quality change within sediment quality guidelines (Ontario Lower Effect Level [LEL] SQG and CCME Interim Sediment Quality Guidelines (ISQG) for the protection of aquatic life; Level I – sediment quality change greater than CCME ISQG or Ontario LEL but less than CCME PEL or Ontario Severe Effect Level (SEL); Level II – sediment quality change 1 to 10x times CCME ISQG or Ontario LEL but less than CCME PEL or Ontario SEL; Level III – sediment quality change >10x CCME ISQG or Ontario LEL (CCME 2003).

[19] A mine-related effect on water quality parameters was identified based on a magnitude of elevation of at least 3 to 5x (i.e., defined as at least slightly elevated) compared to reference and baseline conditions across all seasons in 2023. However, the approach was updated in this report to identify a mine-related effect if a water quality parameter was at least slightly elevated compared to reference and baseline conditions in any individual sampling season (as outlined in Section 2.5.1.1).

Baffinland in 2024, is presented in this report (Table 2.9). Table 2.9 tracks the management responses to previously identified mine-related influences on each applicable waterbody and aquatic system. Recommendations from the effects assessment in this report, based on the 2024 CREMP data, will be implemented and reported on as part of the Mary River Project 2025 CREMP.

Table 2.9: Summary of Effects Determination for 2023 and Associated Action Level Responses, Mary River Project CREMP, 2024

Waterbody	2023 Effects Determination Summary	Action Level Response
Camp Lake Tributary 1 (CLT1)	In 2023, total and dissolved uranium concentrations at the CLT1 upper main stem were elevated in all seasons compared to reference levels, with total uranium remaining elevated since 2019 compared to baseline concentrations. Total sulphate, and total and dissolved molybdenum, sodium, and uranium concentraitons were also elevated relative to reference and baseline conditions in all seasons, indicating a mine-related influence. A **Moderate Action Response** was triggered due to elevated iron concentrations exceeding the AEMP benchmark, while a **Low Action Response** was triggered for the elevated concentrations of sulphate, molybdenum, sodium, and uranium relative to water quality guidelines and reference/baseline conditions.	As a result, it was recommended to continue monitoring the benthic invertebrate community at Station CLT1-L2 (upper main stem) in 2024, as well as in future CREMP studies. Additionally, temporal trend analyses of aqueous concentrations of total sulphate, and total and dissolved molybdenum, sodium, and uranium was suggested to determine if the elevated concentrations are associated with mine-related influences were completed in 2024.
	Metal concentrations in sediment at the CLT1 north branch in 2023 were generally elevated compared to reference areas, but only the mean concentration of iron exceeded the SQG. The source of elevated sediment metal concentrations compared to reference in 2023 is unclear and temporally limited sediment quality data preclude comparison to baseline for a determination of mine-related influence. While **no action** is required under the Management Response Framework, further investigation was recommended to assess potential impacts.	To determine whether changes in iron concentrations in sediment of CLT1 north branch have occurred over the sampling period during mine operations (i.e., since 2017), evaluation of temporal plots and/or statistical temporal trend analyses of the available 2017, 2020, and 2023 sediment quality data for iron were completed in 2024.
Sheardown Lake Tributary 1 (SDLT1)	In 2023, aqueous concentrations of total aluminum (summer), cadmium (summer), copper (summer and fall), and iron (spring and summer) at SDLT1 were above AEMP benchmarks. Additionally, concentrations of several other parameters, including total chloride, nitrate, sulphate, lithium, and potassium, and total and dissolved magnesium, manganese, strontium, and uranium, were elevated relative to reference and baseline conditions across most seasonal sampling events. These elevated concentrations suggest a mine-related influence on water quality, potentially linked to the recent construction and discharge from the KM105 surface water management system. A temporal trend analysis indicated a significant increase in total and dissolved cadmium concentrations at SDLT1 stations since mine operations began in 2015, with cadmium exceeding AEMP benchmarks in 2022. Based on these findings, a **Low Action Response** is triggered for parameters below AEMP benchmarks, and a **Moderate Action Response** for those above benchmarks, as per the AEMP Management Response Framework.	In 2024, a trend analysis was completed for SDLT1 to assess the aqueous concentrations of total chloride, nitrate, and sulphate, and total and dissolved aluminum, iron, lithium, magnesium, manganese, potassium, strontium, and uranium, in order to determine if there are any statistically significant temporal trends. Upgrades and adjustments to facilities and systems associated with water management for the KM105 surface water management infrastructure in the upper SDLT1 system are ongoing, and therefore water quality information collected during the 2024 CREMP will be used to monitor water quality of SDLT1 and as a basis for informing the potential need for further investigations.
	Metal concentrations in sediment at SDLT1 in 2023 were generally elevated compared to reference areas, but only the mean concentration of iron exceeded the SQG. The source of elevated sediment metal concentrations compared to reference in 2023 is unclear and temporally limited sediment quality data preclude comparison to baseline for a determination of mine-related influence. While **no action** is required under the Management Response Framework, further investigation was recommended to assess potential impacts.	To determine whether changes in iron concentrations in sediment of SDLT1 have occurred over the sampling period during mine operations (i.e., since 2017), evaluation of temporal plots and/or statistical temporal trend analyses of the available 2017, 2020, and 2023 sediment quality data for iron were completed in 2024.
Sheardown Lake Tributary 9 (SDLT9)	In 2023, concentrations of certain water chemistry parameters, particularly nitrogen compounds such as total ammonia, nitrate, and TKN, exceeded AEMP benchmarks and WQGs in summer and fall sampling events triggering a **Moderate Action Response**. A mine-related influence at SDLT9, was also determined based on elevated ammonia, nitrate, and TKN concentrations relative to reference and baseline conditions, triggering a **Low Action Response**.	In 2024, a special investigation was completed, including the implementation of an expanded spatial water quality sampling program, to identify the sources of ammonia, nitrate, and TKN at SDLT9.
Sheardown Lake Tributary 12 (SDLT12)	In 2023, aqueous concentrations of sulphate were consistently elevated compared to both reference and baseline conditions in spring and summer sampling seasons. These results suggest a mine-related influence on water quality, though elevated sulphate concentrations did not exceed AEMP benchmarks, triggering a **Low Action Response**.	In 2024, a temporal trend analysis of aqueous sulphate concentrations was completed to assess whether sulphate levels are increasing over time as a result of mine operations.

Notes: WQG = Water Quality Guidelines. AEMP = Aquatic Effects Monitoring Program. CREMP = Core Receiving Environment Monitoring Program. FDP = Final Discharge Point. SQG = sediment quality guideline. TKN = Total Kjeldahl nitrogen.

Table 2.9: Summary of Effects Determination for 2023 and Associated Action Level Responses, Mary River Project CREMP, 2024

Waterbody	2023 Effects Determination Summary	Action Level Response
Sheardown Lake Northwest (DL0-01)	In 2023, a mine-related influence on the water quality of Sheardown Lake NW was suggested due to aqueous concentrations of sulphate that were elevated relative to reference and baseline conditions, as well as increasing trends in concentrations of total nitrate, chloride and sulphate, and both total and dissolved molybdenum and uranium, triggering a **Low Action Response**.	In 2024, a trend analysis for total and dissolved concentrations of molybdenum and uranium was completed. Additionally, an analysis comparing total to dissolved aqueous concentrations of molybdenum and uranium will be completed to assess bioavailability and potential biological effects. Potential sources of nitrate, chloride, sulphate, and total and dissolved molybdenum and uranium were investigated to better define potential mine-related influence on water quality of Sheardown Lake NW.
	In 2023, AEMP benchmarks for sediment quality at Sheardown Lake NW were exceeded for arsenic, copper, and iron. Temporal trend analysis found significant increasing trends since baseline for arsenic in sediment from littoral and profundal areas, reflecting a step-change in concentrations between mine construction and operation periods. Since sediment arsenic concentrations throughout the operation period have remained within a similar range for each habitat, including in 2023, a mine-related influence was not concluded. For iron in sediment from littoral areas, increasing trends since baseline and throughout the mine operation period were observed, with concentrations in 2022 and 2023 exceeding the historical range, and a spatial pattern in sediment iron concentration identified across the lake. These results suggest the emergence of a mine-related influence on sediment quality of the lake but further monitoring is required to support this conclusion. Therefore, while **no action** is required under the Management Response Framework, further investigation was recommended to assess potential impacts.	In 2024, a temporal trend analysis of iron sediment concentrations in Sheardown Lake NW was recommended to be repeated using new monitoring data.
Sheardown Lake Southeast (DL0-02)	In 2023, a mine-related influence on the water quality of Sheardown Lake NW was suggested due to increasing trends in aqueous concentrations of total nitrate and sulphate, and total and dissolved molybdenum and uranium over the mine operation period (2015 to 2023), triggering a **Low Action Response**.	In 2024, an analysis comparing total to dissolved aqueous concentrations of molybdenum and uranium will be conducted to assess bioavailability and potential biological effects. Potential sources of nitrate, sulphate, and total and dissolved molybdenum and uranium were investigated to better define potential influence on water quality of Sheardown Lake SE.
Mary River Tributary-F (MRTF)	Previous investigations into elevated aqueous concentrations of total aluminum and iron in the Mary River system suggested that natural surface runoff and fluvial transport, rather than a mine-related source, accounted for the elevated concentrations relative to AEMP benchmarks and/or WQGs at mine-exposed stations. Temporal trend analyses for total nitrate and sulphate concentrations, which were elevated compared to the Mary River upstream reference area in 2023 but remained below AEMP benchmarks and WQGs, revealed significant increasing trends over the mine operation period (2015 to 2023) and since the baseline period. These trends showed a step-change in total nitrate concentrations starting in 2019 and in total sulphate concentrations starting in 2017, rather than a gradual increase. The significantly increasing trends in total nitrate and sulphate suggest a mine-related influence on water quality, likely from effluent discharged to the MRTF at the MS-08 FDP. This triggers a **Low Action Response**.	In 2024, a temporal trend analysis of total nitrate and sulphate concentrations in MRTF was completed using new monitoring data. Effluent quality and MRTF water quality will be closely monitored and evaluated for any continually increasing trends in nitrate and/or sulphate concentrations that indicate the need for development of additional mitigation measures.

Notes: WQG = Water Quality Guidelines. AEMP = Aquatic Effects Monitoring Program. CREMP = Core Receiving Environment Monitoring Program. FDP = Final Discharge Point. SQG = sediment quality guideline. TKN = Total Kjeldahl nitrogen.

3 CAMP LAKE SYSTEM

3.1 Camp Lake Tributary 1 (CLT1)

3.1.1 Water Quality

3.1.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at CLT1 concurrent with water quality sampling in spring, summer, and fall (Figure 2.1), as well as concurrent with BIC sampling in August (Figure 2.3). During spring, summer, and fall monitoring events, DO at CLT1 was consistently near full saturation (all ≥ 94.2%) at the North Branch and Main Stem stations, and was comparable to or slightly higher than at the reference stream stations (Appendix Figure C.1, Appendix Tables C.1 to C.3). Although there was a significant difference in mean dissolved oxygen saturation between the CLT1 Lower Main Stem (114%) and North Branch (109%) and the reference stream (98.9%) in August, the percent differences were small (Figure 3.1, Appendix Tables C.11, C.12 and C.13). During all sampling events, DO concentrations at CLT1 North Branch, Upper Main Stem, Lower Main Stem, and reference stream stations were above the WQG lowest acceptable concentration for early life stages of cold-water biota (i.e., 9.5 mg/L; Figure 3.1, Appendix Figure C.1, Appendix Tables C.1 to C.3 and C.11).

Generally, pH increased slightly or remained similar with progression downstream through the CLT1 North Branch (Stations L1-08 to L1-02) and Main Stem (Stations L2-03 to L0-01) stations during spring, summer, and fall monitoring events (Appendix Figure C.1, Appendix Tables C.1 to C.3). In August 2024, pH was significantly higher at CLT1 Lower Main Stem (mean pH of 7.99), Upper Main Stem (mean pH of 7.98) and the North Branch (mean pH of 8.07) than at Unnamed Reference Creek (mean pH of 7.27), although actual differences were small (Figure 3.1, Appendix Tables C.12 and C.13). During all sampling events in 2024, pH at CLT1 was consistently within WQG limits (pH of > 6 and < 9; Figure 3.1, Appendix Figure C.1, Appendix Tables C.1 to C.3 and C.11).

Mean specific conductance at CLT1 in August was generally highest in the Upper Main Stem (mean = 285 µS/cm) and lowest in the North Branch (mean = 187 µS/cm), with intermediate values observed at the Lower Main Stem (mean = 213 µS/cm) reflecting mixing of these two branches and suggesting a potential mine-related source affecting water quality of the CLT1 Upper Main Stem (Figure 3.1, Appendix Tables C.11 and C.12). Specific conductance was consistently higher at CLT1 North Branch, Upper Main Stem, and Lower Main Stem than at the reference stream stations during the spring, summer, and fall sampling events (Appendix Figure C.1, Appendix Tables C.1 to C.3). Specific conductance was significantly higher at the CLT1

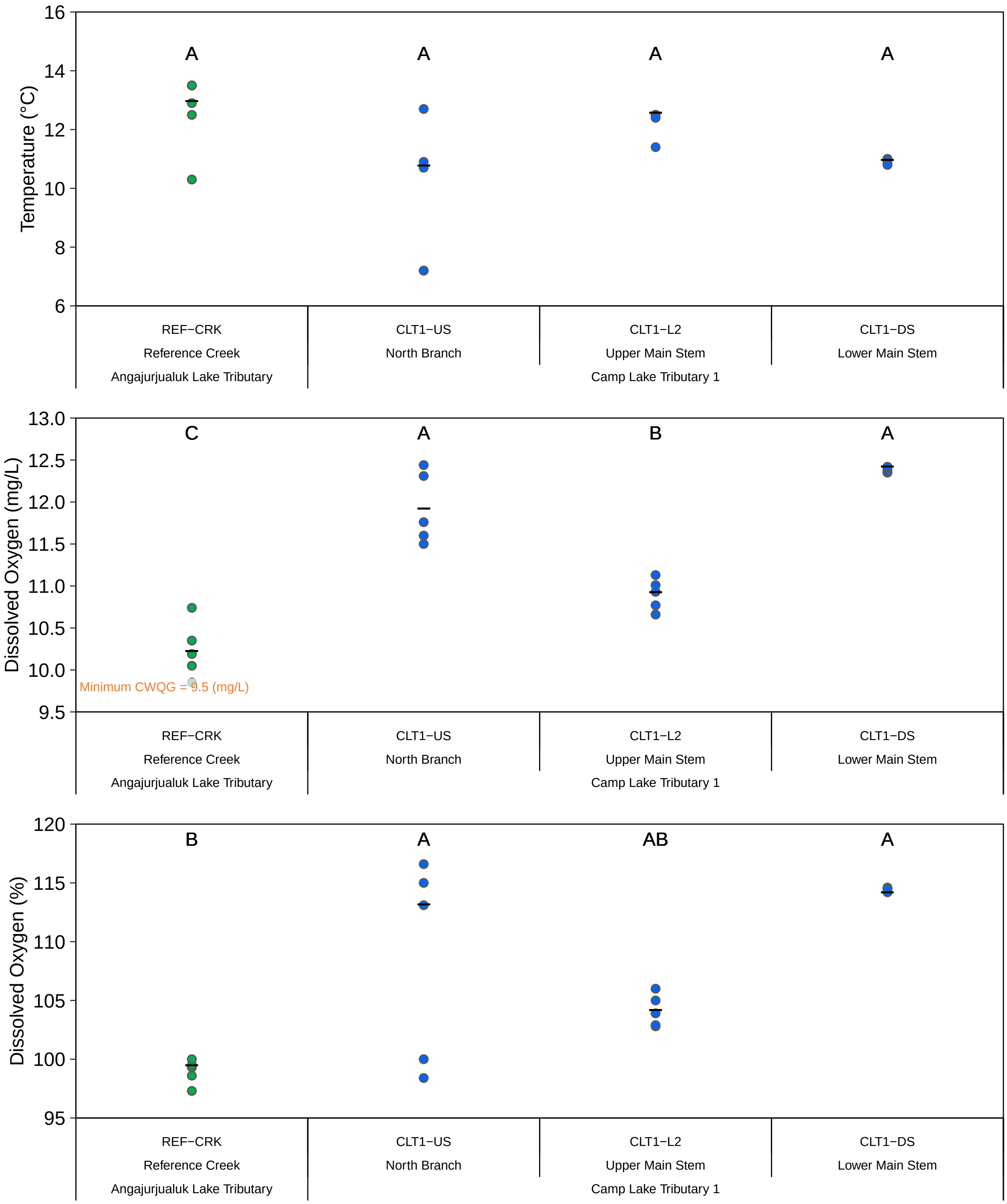


Figure 3.1: Comparison of *In Situ* Water Quality Measured at Camp Lake Tributary 1 (CLT1; Stations CLT1-US, CLT1-L2, CLT1-DS) and Reference Creek (REF−CRK) Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

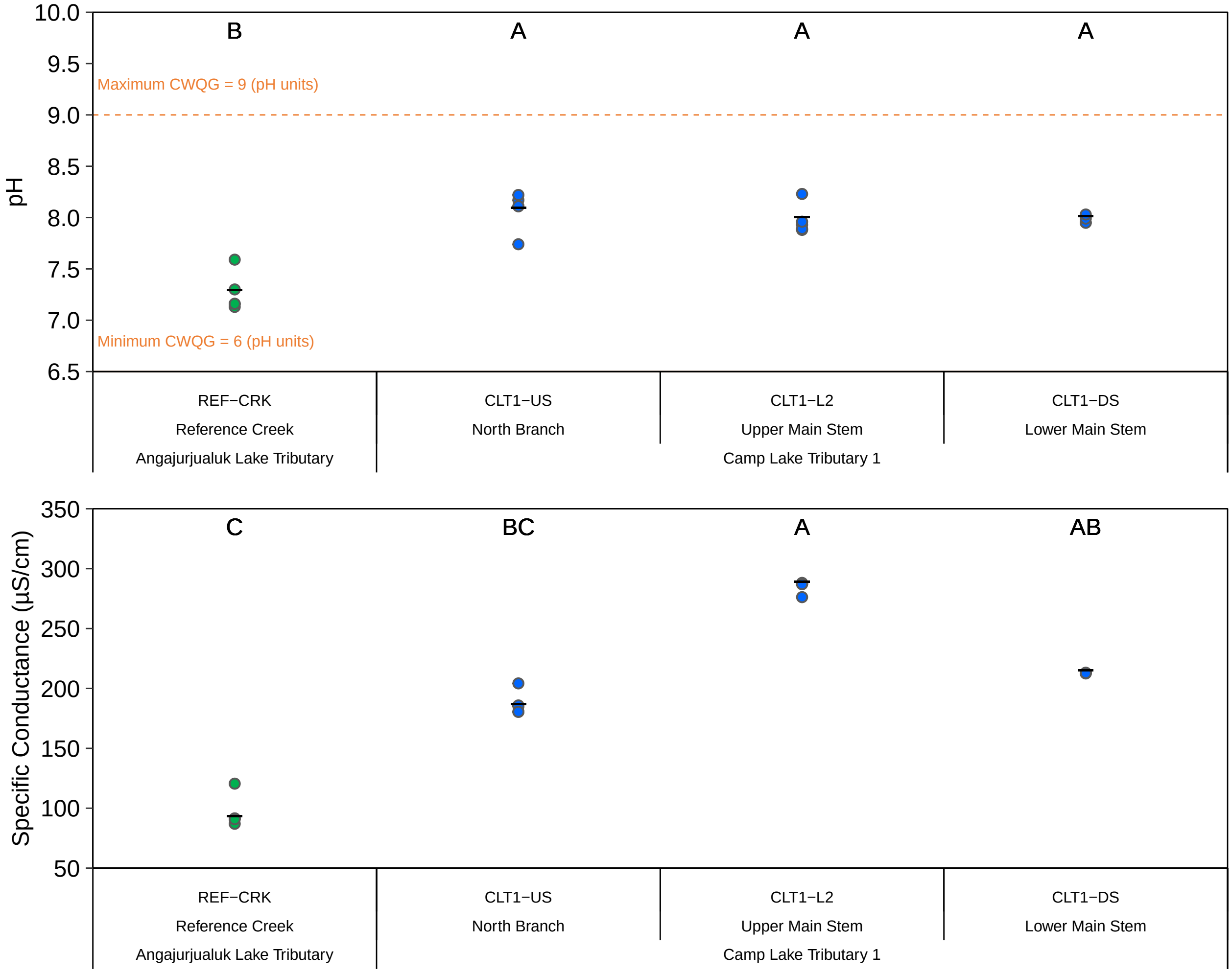


Figure 3.1: Comparison of *In Situ* Water Quality Measured at Camp Lake Tributary 1 (CLT1; Stations CLT1-US, CLT1-L2, CLT1-DS) and Reference Creek (REF−CRK) Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

Upper and Lower Main Stem than the reference area and higher at the Upper Main Stem than the North Branch during the August BIC sampling (Figure 3.1, Appendix Figure C.1, Appendix Tables C.11 to C.13). However, specific conductance did not differ between the CLT1 Lower Main Stem and the Upper Main Stem (Figure 3.1, Appendix Figure C.1, Appendix Tables C.11 to C.13) suggesting that the elevated specific conductance in the CLT1 system was largely associated with the Upper Main Stem, suggesting a potential mine-related influence on that portion of the CLT1 system.

3.1.1.2 CLT1 North Branch Water Chemistry

At the CLT1 North Branch stations (L1-08 and L1-02), mean and individual sample aqueous concentrations met AEMP benchmarks and WQGs for all parameters in 2024, with the exception of total copper. Mean and individual sample total copper concentrations slightly exceeded the AEMP benchmark (0.0022 mg/L) and WQG (0.002 mg/L) during the fall sampling event (mean = 0.00244 mg/L), while the total copper concentration in a single sample collected at Station L1-08 during the summer sampling event (0.00204 mg/L) slightly exceeded the WQG (Table 3.1, Appendix Table C.14). Mean total and dissolved copper concentrations at the CLT1 North Branch were not elevated relative to reference stream stations or baseline concentrations in any season in 2024, except for dissolved copper in the spring, which was slightly elevated (3 to 5 times higher) compared to the reference stream stations (Table 3.1, Appendix Tables C.14 to C.18). Temporal trend analyses of total and dissolved copper concentrations were conducted in the 2022 CREMP and found no significant trends in the CLT1 North Branch during the mine operation period from 2015 to 2022 (Minnow 2023). Ranges in total copper concentrations in both 2023 and 2024 were consistent with all previous years of mine production, and copper concentrations have generally remained within the range of baseline concentrations since the start of the operation period in 2015 (Appendix Figure C.2). Taken together, these findings suggest that mine operations have not contributed to elevated copper concentrations in water at the CLT1 North Branch; instead, concentrations above the AEMP benchmark and WQG are likely attributable to natural variation.

Total and dissolved water chemistry parameter concentrations that were slightly or moderately elevated at the CLT1 North Branch in 2024 relative to reference or baseline concentrations are identified in Appendix Tables C.15, C.17, and C.18. In 2024, there were no total or dissolved water chemistry parameter concentrations that were consistently elevated across all seasons (spring, summer, and fall) compared to the reference stream stations and/or to baseline concentrations (Appendix Figure C.2, Appendix Tables C.15, C.17, and C.18). Some total and dissolved parameter concentrations were slightly or moderately elevated in at least one season relative to reference or baseline concentrations (most frequently in spring), but none were

Table 3.1: Mean Water Chemistry at Camp Lake Tributary 1 (CLT1) Monitoring Stations During Spring, Summer, and Fall, Mary River Project CREMP, 2024

	Parameters	Units	Water Quality Guideline (WQG)[a,b]	AEMP Benchmark[c]	Reference Creeks (n=4)			North Branch (n=2)			Upper Main Stem L2-03 (n=1)			Lower Main Stem (n=3)		
					Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	28.1	82.5	107	92.2	154	188	219	275	317	114	193	240
	pH (lab)	pH	6.5 - 9.0	-	7.63	7.60	7.77	7.63	7.96	8.01	7.94	7.94	7.90	7.78	8.05	7.99
	Hardness (as $CaCO_3$)	mg/L	-	-	13.0	38.6	50.4	46.5	74.6	90.8	91.8	119	141	54	91.6	114
	Total Suspended Solids (TSS)	mg/L	-	-	2.65	1.27	<1	<1	<1	<1	3.40	1.00	<1	<1	<1	<1
	Total Dissolved Solids (TDS)	mg/L	-	-	25.2	48.2	48.5	54.0	77.5	97.5	122	148	162	61.3	105	123
	Turbidity	NTU	-	-	2.72	3.68	3.69	0.695	1.42	0.185	9.23	3.16	2.37	1.29	1.21	0.637
	Alkalinity (as $CaCO_3$)	mg/L	-	-	12.4	37.7	53.2	41	69.5	81.2	83.9	111	138	51.8	88.9	111
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00592	0.00520	0.00565	0.00580	<0.005	0.0144	0.0416	0.0111	0.0379	0.0128	0.00530	0.0193
	Nitrate	mg/L	3	3	<0.02	0.0240	<0.02	0.0215	0.0250	0.0275	0.466	0.141	0.329	0.0473	0.0293	0.0977
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.101	0.0768	0.0635	0.0715	0.0925	0.150	0.204	0.287	0.311	0.0993	0.119	0.196
	Dissolved Organic Carbon	mg/L	-	-	2.28	2.20	2.14	2.76	2.46	3.73	3.25	4.08	6.20	2.44	4.18	3.25
	Total Organic Carbon	mg/L	-	-	1.92	1.82	2.11	2.38	2.38	2.55	3.15	4.46	4.63	2.55	3.88	3.45
	Total Phosphorus	mg/L	0.030^{α}	-	0.00450	0.00318	0.00335	0.00210	0.00200	0.00210	0.00830	0.00500	0.00390	0.00263	<0.002	<0.002
	Phenols	mg/L	0.004^{α}	-	<0.001	<0.001	<0.001	0.00110	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	`	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	0.605	1.24	1.67	3.40	4.28	8.12	10.3	11.4	13.0	2.52	4.13	7.21
	Sulphate (SO_4)	mg/L	218^{β}	218	0.542	1.72	2.44	1.15	1.96	2.78	9.90	9.75	11.7	2.21	3.21	5.38
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0670	0.0832	0.160	0.0222	0.0132	0.0131	***0.249***	0.0493	0.0516	0.0346	0.0198	0.0180
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.000100	<0.0001	<0.0001	<0.0001	0.000150	0.000120	0.000130	<0.0001	<0.0001	<0.0001
	Barium (Ba)	mg/L	1^{β}	-	0.00201	0.00480	0.00714	0.00572	0.00951	0.0121	0.0126	0.0126	0.0148	0.00654	0.0108	0.0134
	Beryllium (Be)	mg/L	0.011^{α}	-	<0.00002	<0.00002	0.0000205	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.0160	0.0150	0.0160	<0.01	<0.01	<0.01
	Cadmium (Cd)	mg/L	0.00012	0.00008	0.00000532	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	0.00000730	0.00000550	<0.000005	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	2.56	7.68	10.3	8.74	14.2	18.6	18.0	23.4	28.6	10.3	17.6	23.3
	Chromium (Cr)	mg/L	0.0089	0.003	0.000522	0.000565	0.000672	<0.0005	<0.0005	<0.0005	0.000680	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Cobalt (Co)	mg/L	0.0009^{α}	0.0040	0.000102	0.000105	0.000125	<0.0001	<0.0001	<0.0001	0.000210	0.000110	0.000140	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.0022	0.000600	0.000852	0.00114	0.00160	0.00200	***0.00244***	0.00180	0.00127	0.00136	0.00136	0.00185	0.00211
	Iron (Fe)	mg/L	0.30	0.326	0.0810	0.0942	0.143	0.0280	0.0165	0.0140	***0.401***	0.273	***0.330***	0.0600	0.0623	0.0837
	Lead (Pb)	mg/L	0.001	0.001	0.000100	0.000114	0.000154	0.0000540	<0.00005	<0.00005	0.000342	0.0000720	0.0000740	0.0000507	<0.00005	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.00350	0.00290	0.00330	<0.001	0.00110	0.00183
	Magnesium (Mg)	mg/L	-	-	1.74	4.96	6.36	6.16	9.65	12.4	12.9	15.9	19.5	7.61	12.2	15.8
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00141	0.00126	0.00162	0.000650	0.000475	0.000495	0.0183	0.0228	0.0255	0.00276	0.00425	0.00487
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.0000752	0.000232	0.000420	0.000334	0.000732	0.00104	0.00262	0.00325	0.00310	0.000665	0.000997	0.00131
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	0.000500	0.000698	0.000510	0.000525	0.000600	0.00102	0.00110	0.00122	0.000570	0.000723	0.000977
	Potassium (K)	mg/L	-	-	0.313	0.559	0.789	1.14	1.70	2.24	4.26	3.50	3.37	1.55	2.00	2.38
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	0.0000500	0.0000760	0.0000840	0.000103	<0.00005	0.0000503	0.0000500
	Silicon (Si)	mg/L	-	-	0.475	0.932	1.20	0.610	0.860	1.03	0.970	0.940	1.14	0.587	0.973	1.22
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.383	1.12	1.60	0.558	0.841	1.24	8.10	8.84	10.2	2.15	2.62	4.34
	Strontium (Sr)	mg/L	-	-	0.00238	0.00758	0.0111	0.00514	0.00899	0.0133	0.0341	0.0329	0.0337	0.00947	0.0157	0.0201
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	0.0000105	0.0000100	0.0000110	0.0000115	0.0000100	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.00410	0.00471	0.00760	0.000850	0.000610	0.000450	0.00778	<0.003	0.00197	0.000983	0.000830	0.000630
	Uranium (U)	mg/L	0.015	-	0.000212	0.00115	0.00286	0.000497	0.00200	0.00373	0.0125	0.0190	0.0248	0.00205	0.00377	0.00713
	Vanadium (V)	mg/L	0.006^{α}	0.006	0.000508	0.000522	0.000625	<0.0005	<0.0005	0.000535	0.000510	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02^{α}	0.030	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

[a] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[b] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

consistently elevated in the same season compared to both reference and baseline (Appendix Tables C.15, C.17, and C.18), suggesting no mine-related influence on water quality at the CLT1 North Branch. Furthermore, except for copper concentrations that appear to be naturally above the AEMP benchmark and WQG, water quality has consistently met applicable AEMP benchmarks and WQG at the CLT1 North Branch since the commencement of the mine operational period in 2015.

3.1.1.3 CLT1 Main Stem Water Chemistry

3.1.1.3.1 CLT1 Upper Main Stem

At the CLT1 Upper Main Stem (Station L2-03), total aluminum, iron, and uranium were the only water chemistry parameters with concentrations that exceeded AEMP benchmarks and/or WQG in at least one seasonal sampling event in 2024. Total aluminum concentrations exceeded the AEMP benchmark (0.179 mg/L) and WQG (0.100 mg/L) in the spring (0.249 mg/L; Table 3.1, Appendix Table C.14). The total aluminum concentration was only consistently elevated compared to reference stream stations and baseline concentrations in the spring (slightly [3 to 5 times] and highly [≥ 10 times], respectively; Appendix Figure C.2, Appendix Table C.15). Dissolved aluminum concentrations were not seasonally elevated relative to the reference stream but were moderately elevated (5 to 10 times) in the spring and highly elevated in the fall compared to baseline (Appendix Tables C.17 and C.18). Visual assessment of total aluminum concentrations over time indicates that concentrations in the spring exceeded the AEMP benchmark in 2014 and consistently since 2018 (except for in 2022), though no clear increasing pattern is evident over this period (Appendix Figure C.2). Based on spring concentrations that were elevated relative to both reference and baseline conditions and exceedance of both the AEMP benchmark and WQGs, a potential mine-related influence was identified for total aluminum concentrations in the CLT1 Upper Main Stem in 2024. However, given that total but not dissolved aluminum concentrations in the spring were elevated compared to reference and baseline concentrations, the higher total concentrations are associated with suspended solids in the water column during freshet as indicated by turbidity that was also elevated compared to both reference and baseline conditions in spring (Appendix Table C.15). This suggests that contributions of total aluminum to the CLT1 Upper Main Stem may be related to the background minerology of the system mobilized by both natural (e.g., weathering and erosion) as well as mine-related (e.g., dust) processes.

Total iron concentrations in the CLT1 Upper Main Stem in 2024 exceeded both the AEMP benchmark (0.326 mg/L) and WQG (0.30 mg/L) in the spring (0.401 mg/L) and fall (0.330 mg/L; Table 3.1, Appendix Table C.14). Total iron concentrations were slightly elevated compared to reference and baseline concentrations only in the spring (Appendix Table C.15) while dissolved iron concentrations were moderately elevated in the summer and slightly elevated in the fall compared to the reference stream stations and no differences were observed compared to baseline (Appendix Figure C.2, Appendix Tables C.17 and C.18). Visual assessment of total iron concentrations over time indicates that concentrations in the spring, summer, and/or fall have exceeded the AEMP benchmark and/or WQG fairly consistently since the initiation of mine operations in 2015, though no clear patterns of increasing concentrations in any season over this period are evident (Appendix Figure C.2). Annual monitoring previously identified the potential for a relatively continuous mine-related source influencing aqueous iron concentration in the CLT1 Upper Main Stem. In response, annual BIC monitoring was initiated at CLT1-L2 in the CLT1 Upper Main Stem in 2020 (Minnow 2021b) with no adverse effects on the BIC identified to date (see Section 3.1.4). Temporal trend analyses of total and dissolved iron concentrations in the CLT1 Upper Main Stem were completed as part of the 2022 CREMP (Minnow 2023), and based on a Moderate Action Response recommended in the 2023 CREMP (Minnow 2024a), repeated in 2024 incorporating the most recent water quality results (Appendix Tables H.1 and H.2). In both cases, a significant increasing trend in total iron was observed over all sampling seasons combined since the baseline period (2005 to 2024; Minnow 2023; Appendix Figure C.2, Appendix Table H.1), but not during the mine operation period (2015 to 2024; Appendix Figure C.2, Appendix Table H.2). The 2024 analyses also investigated individual seasonal trends and found significant increasing trends since the baseline period in spring and fall (Appendix Table H.1). No significant temporal trends for dissolved iron were identified at the CLT1 Upper Main Stem since the baseline period or over the mine operational period (Minnow 2023; Appendix Table H.1 and H.2). Of the four reference streams, MRY-REF3 is the only one that showed significant increasing trends in total iron for all seasons combined and in spring, specifically in the 2024 analysis. The absence of similar trends in other reference streams suggests that a consistent increase in aqueous iron concentrations since the baseline period is not regionally occurring. Temporal trend analyses for dissolved iron could not be completed for all reference streams, as results were often below the LRL also suggesting that increasing dissolved iron concentrations are not regionally widespread. However, as with total aluminum concentrations, total but not dissolved iron concentrations in the spring were elevated compared to reference and baseline concentrations and temporal trend analyses for iron identified increasing trends for total but not dissolved concentrations. Therefore, higher total iron concentrations are associated with suspended solids in the water column during freshet and contributions of total iron to the CLT1 Upper Main Stem are likely related to the background minerology of the system mobilized by both natural (e.g., weathering and erosion) as well as mine-related (e.g., dust) processes.

Total iron concentrations that exceeded both the AEMP benchmark and WQG, were elevated relative to both reference and baseline conditions in spring 2024, and that have shown significant increasing trends since the baseline period (but not during the mine-operation period) suggest a potential mine-related influence in the CLT1 Upper Main Stem. However, the absence of increasing trends over the mine operation period or negative effects on the BIC suggest that this influence is not intensifying over time nor adversely influencing aquatic life.

Total uranium concentrations exceeded the WQG (0.015 mg/L) in the summer (0.019 mg/L) and fall (0.0248 mg/L; Table 3.1, Appendix Table C.14). Compared to the reference stream, both total and dissolved uranium concentrations were highly elevated in the spring and summer, and moderately elevated in the fall (Appendix Tables C.15, C.17, and C.18). Compared to baseline conditions, uranium concentrations were consistently highly elevated across all seasons (Appendix Tables C.15, C.17, and C.18). Since 2018, total uranium concentrations exceeded the WQG in the summer and fall, intermittently exceeded the WQG in the spring, and were higher than concentrations observed in the reference streams over the same period (Appendix Figure C.2). A potential mine-related effect on uranium at the CLT1 Upper Main Stem was identified in 2023, prompting the recommendation for a temporal trend analysis in the 2023 CREMP report (Low Action response; Minnow 2024a). The trend analysis found significant increases in both total and dissolved uranium concentrations across all seasons combined and in each individual season, since the baseline period (2005 to 2024) and in all seasons combined and in the summer during the mine operation period (2015 to 2024; Appendix Figure C.2, Appendix Tables H.1 and H.2). The trend analyses also found that three of four reference stream monitoring areas also had significant increasing trends in total uranium since 2014 in summer, suggesting potential naturally occurring regional influences in addition to those potentially related to mining activities (Appendix Table H.1). While uranium was included as a key indicator for water and sediment quality in the Mary River Project FEIS (Baffinland 2012), concentrations in Camp Lake and its tributaries where effects were predicted from waste rock stormwater discharge were not estimated (i.e., concentrations in waste rock stockpile seepage and open pit water were not modelled) and concentrations in ore dust runoff were not predicted (i.e., no data for the concentration of uranium in ore).

Increases in uranium concentrations in water can result from the disturbance of naturally occurring uranium-bearing minerals in rock formations during mining processes leading to uranium leaching into surrounding water sources (Dzimbanhete et al. 2025). Additionally and notably, the Arctic is warming at more than twice the global average rate due to climate feedback mechanisms (Koivurova et al. 2016), resulting in the destabilization and thawing of permafrost, which has functioned as a long-term geochemical sink for various elements over centuries, as well as deepening of the active layer, the uppermost soil layer that undergoes seasonal freeze-thaw cycles (Jin and Ma 2021, Hindshaw et al. 2018, Schnurr 2018). Prolonged seasonal thaw within the active layer may enhance geochemical mobility, increasing the potential for element transport (Jin and Ma 2021, Hindshaw et al. 2018, Schnurr 2018). Organic matter in the active layer has a high binding affinity for elements such as uranium, and when thaw occurs, decomposition of previously frozen organic matter can release bound elements into adjacent aquatic environments (MacDonald et al. 2000, Schnurr 2018, Skierszkan et al. 2021, Hindshaw et al. 2018). Anthropogenic activities, like mining, can further accelerate permafrost thaw by altering surface conditions and impacting the active layer through infrastructure development, vegetation removal, and ground disturbance, all of which reduce the insulating capacity of the surface and increase heat transfer into the ground (Langer et al. 2023).

Although an AEMP benchmark has not been established for uranium, consistent exceedance of the WQG, the notable elevation relative to reference and baseline concentrations across all seasons in 2024 and increasing trends in uranium concentrations both since the baseline period and over the mine operation period, suggest a mine-related influence on uranium concentrations at the CLT1 Upper Main Stem. A combination of climate-driven and mine-related factors may be contributing.

Total and dissolved water chemistry parameter concentrations that were slightly, moderately, or highly elevated relative to reference or baseline conditions at the CLT1 Upper Main Stem in 2024 are identified in Appendix Tables C.15, C.17, and C.18. In addition to total and dissolved uranium, total sulphate, and total and dissolved sodium and molybdenum concentrations were consistently elevated relative to both baseline and reference conditions in all sampling seasons in 2024 (except for dissolved sodium which was elevated compared to baseline only in spring and summer; Appendix Figure C.2, Appendix Tables C.15, C.17, and C.18). In 2023, elevated concentrations of sulphate, sodium, and molybdenum were identified as potentially mine-related, warranting temporal trend analysis as a special investigation (Low Action Response; Minnow 2024a). Sulphate, total sodium, and total and dissolved molybdenum concentrations increased significantly over time since the baseline period (2005 to 2024) when all seasons were combined as well as in some individual seasons (Appendix Figures C.2 and H.1, Appendix Tables H.1 and H.2). However, no significant increasing trends were observed in the concentrations of these parameters during the mine operation period (2015 to 2024; Appendix Figure C.2, Appendix Tables H.1 and H.2). Since baseline and during the mine operation period, reference streams have shown some increasing trends for these parameters (sulphate, sodium, molybdenum). Significant increasing trends for concentrations of these parameters were occasionally observed over all seasons combined and in some individual seasons at some of the reference streams, mostly since the baseline period (Appendix Tables H.1 and H.2), but results did not suggest that increasing trends in concentration for any individual parameter were regionally widespread.

Total sulphate and total and dissolved sodium and molybdenum concentrations at the CLT1 Upper Main Stem have generally been higher than those at other CLT1 water quality stations, the reference streams, and baseline concentrations since mine operations began suggesting a mine-related influence (Appendix Figures C.2 and H.1). However, the absence of significant increasing trends during the mine operation period suggests that mine-related influence is not intensifying with ongoing operations and concentrations that remain below applicable AEMP benchmarks and WQG suggest limited potential for adverse effects to aquatic biota. During both baseline and mine operational periods, reference streams showed inconsistent increasing trends for these parameters (sulphate, sodium, molybdenum), indicating that the elevations are not regionally widespread.

In addition to total aluminum and iron, total lead and strontium, and total and dissolved lithium and manganese concentrations at the CLT1 Upper Main Stem, were elevated relative to both baseline and reference conditions in spring 2024 (Appendix Tables C.15, C.17, and C.18). Compared to both reference and baseline conditions, concentrations of total nitrate were elevated in spring and fall, and concentrations of total and dissolved potassium were elevated in both spring and summer (Appendix Tables C.15, C.17, and C.18). Over time, total lead concentrations at the CLT1 Upper Main Stem have generally been higher than at most other CLT1 sampling stations (concentrations at these stations have typically been below the LRL) but have remained within the range of concentrations at the reference streams since 2017, with no consistent increasing or decreasing patterns observed (Appendix Figure C.2). Total strontium concentrations have been slightly elevated compared the reference streams since the start of mine operations but have remained within the range of baseline concentrations with no directional patterns noted (Appendix Figure C.2). Total nitrate, lithium, manganese, and potassium concentrations at the CLT1 Upper Main Stem have consistently been higher than at other CLT1 sampling stations and reference streams since mine operations began (Appendix Figure C.2). Total nitrate, manganese, and potassium concentrations have also been higher than baseline concentrations across all seasons, while total lithium concentrations have remained within the baseline range in summer and fall but have been higher in spring (Appendix Figure C.2). No consistent increasing or decreasing patterns in the concentrations of total manganese or lithium have been observed (Appendix Figure C.2). Total nitrate concentrations peaked between summer 2018 and fall 2019 (exceeding the AEMP benchmark) and have since consistently decreased, with concentrations in 2024 among the lowest observed over the mine operation period (Appendix Figure C.2). A potential source of nitrate is historical quarrying activities at the QMR2 pit. There has been no blasting activity at this quarry pit since 2019, and nitrate concentrations in the upper CLT1 Main Stem have decreased since that time, through remain above reference and baseline concentrations (Appendix Figure C.2). Total potassium concentrations steadily increased in all seasons from the baseline period until 2019 and have since remained relatively stable (Appendix Figure C.2). There is potential that progression of mining at Deposit 1 may have had a general influence on water quality given that until 2019, the top / west side of the mountain where Deposit 1 is located was being mined regularly. The majority of the mining since 2019 has been on the east side of the mountain where Deposit 1 is located, as mine progression continued in a more eastern direction.

As with total aluminum and iron concentrations, total but not dissolved lead and strontium concentrations were elevated compared to reference and baseline concentrations in the spring (Appendix Tables C.15, C.17, and C18). This suggests that higher total concentrations may be associated with suspended solids in the water column during freshet, and contributions of total concentrations to the CLT1 Upper Main Stem are likely related to the background minerology of the system, released through both natural (e.g., weathering and erosion) and potentially mine-related processes (e.g., dust). Parameters with elevated total and dissolved concentrations, in one or multiple seasons, compared to reference and baseline conditions (i.e., lithium, manganese, and potassium), more strongly suggest a mine-related influence. Overall, seasonally elevated concentrations of lead, strontium, lithium, manganese, nitrate, and potassium relative to reference and/or baseline concentrations were observed in 2024. For most of these parameters, a lack of any apparent directional temporal patterns suggests limited potential mine-related influences that have not intensified over the mine operations period at the CLT1 Upper Main Stem.

3.1.1.3.2 CLT1 Lower Main Stem

At the CLT1 Lower Main Stem (Stations L1-09, L1-05, and L0-01), water chemistry met all AEMP benchmarks and WQGs during the spring, summer, and fall sampling events in 2024, except for mean and individual sample copper concentrations in the fall (mean = 0.00211 mg/L), which slightly exceeded the WQG of 0.002 mg/L (Table 3.1, Appendix Table C.14). When compared to both the reference stream stations and baseline concentrations, total and dissolved copper concentrations in 2024 were not elevated in any season (Appendix Tables C.15, C.17, and C.18). Visual assessment of total copper over time indicates that though concentrations at the CLT1 Lower Main Stem have been higher than in the reference streams and frequently near or above the WQG, they have generally been within the baseline range and have shown no pattern of temporal increase (Appendix Figure C.2). Total and dissolved water chemistry parameter concentrations that were slightly, moderately, or highly elevated relative to reference or baseline concentrations are identified in Appendix Tables C.15, C.17, and C.18. While some total and dissolved parameter concentrations were slightly, moderately, or highly elevated in at least one season relative to reference or baseline concentrations, none were consistently elevated in the same season compared to both (Appendix Tables C.15, C.17, and C.18), suggesting no mine-related influence in the CLT1 Lower Main Stem throughout the open-water season. Overall, these results suggest that there were no mine-related influences on water quality at the CLT1 Lower Main Stem.

3.1.2 Phytoplankton

Chlorophyll-a concentrations at the CLT1 North Branch stations (Stations L1-08 and L1-02) were comparable to, or lower than, the concentrations observed at reference stream sites during the spring, summer, and fall sampling events in 2024, based on qualitative comparisons (Figure 3.2, Appendix Table E.1). This suggests similar or slightly lower phytoplankton abundance at the CLT1 North Branch stations, relative to reference stream stations (Figure 3.2, Appendix Table E.1). In contrast, chlorophyll-a concentrations within the CLT1 Main Stem were comparable to (summer) or higher than (spring and fall) reference stream concentrations (Figure 3.2, Appendix Tables E.1 and E.2). The highest concentrations were observed at the upstream-most CLT1 Main Stem station (L2-03) across all three seasonal sampling events (Figure 3.2, Appendix Table E.1). Results for Station L2-03 potentially reflected higher nutrient (e.g., nitrate) concentrations in surface waters compared to the reference streams and other CLT1 stations (Appendix Tables C.14 and C.15).

Chlorophyll-a concentrations at all CLT1 North Branch and Main Stem monitoring stations were well below the AEMP benchmark of 3.7 µg/L for all seasonal sampling events in 2024 (Figure 3.2). These results are indicative of low phytoplankton productivity, suggesting oligotrophic conditions. This classification aligns with Dodds et al. (1998) trophic status criteria for stream environments (i.e., chlorophyll-a concentrations were well below the oligotrophic-mesotrophic boundary of 8 µg/L; Appendix Table E.1). This trophic status classification was also consistent with an ultra-oligotrophic to oligotrophic categorization under the CCME Phosphorus Guidance Framework for the Management of Freshwater Systems (CCME 2024b). Specifically, aqueous total phosphorus concentrations were typically less than 10 µg/L at each of the CLT1 North Branch and Main Stem stations during all spring, summer, and fall sampling events (Table 3.1, Appendix Table C.14).

Mean chlorophyll-a concentrations at the CLT1 North Branch in 2024 were similar to, or lower than, the average concentrations observed during the baseline period, based on qualitative comparisons (2005 to 2013; Figure 3.3). At the CLT1 Main Stem, chlorophyll-a concentrations were higher in the spring and summer during the mine operational years (2015 to 2024) compared to the baseline period (Figure 3.3). However, in the fall, mean chlorophyll-a concentrations at the CLT1 Main Stem were comparable to baseline (Figure 3.3). No consistent patterns of increasing or decreasing mean chlorophyll-a concentrations were observed across the years of mine operation, either at the CLT1 North Branch or at the CLT1

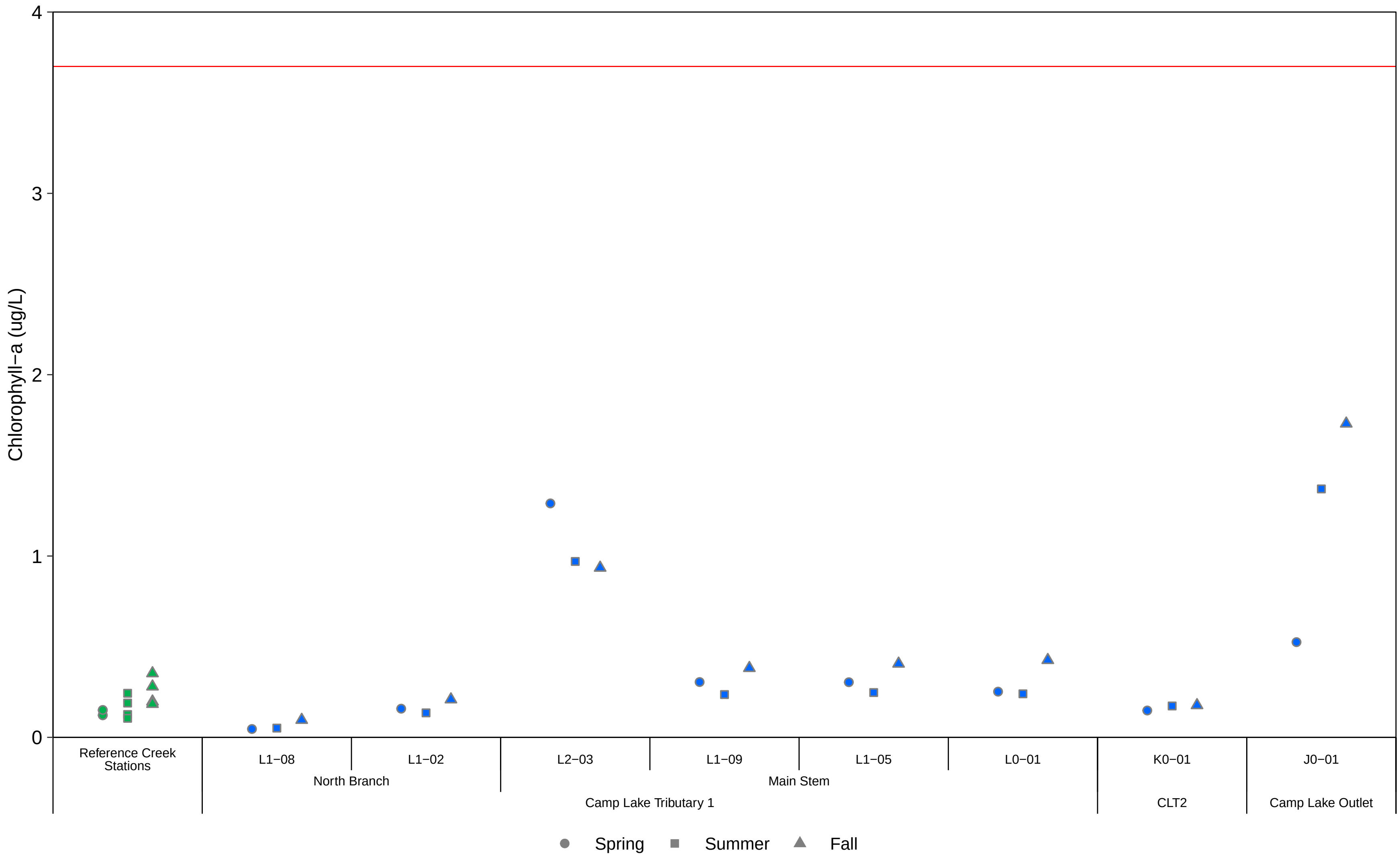


Figure 3.2: Chlorophyll−a Concentrations at Camp Lake Tributary 1 (CLT1) and Tributary 2 (CLT2) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2024

Notes: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Red line indicates AEMP Benchmark. Reference Creek Stations includes data from stations CLT−REF4, CLT−REF3, MRY−REF3, and MRY−REF2. Reference areas are shown in green and mine−exposed areas are shown in blue.

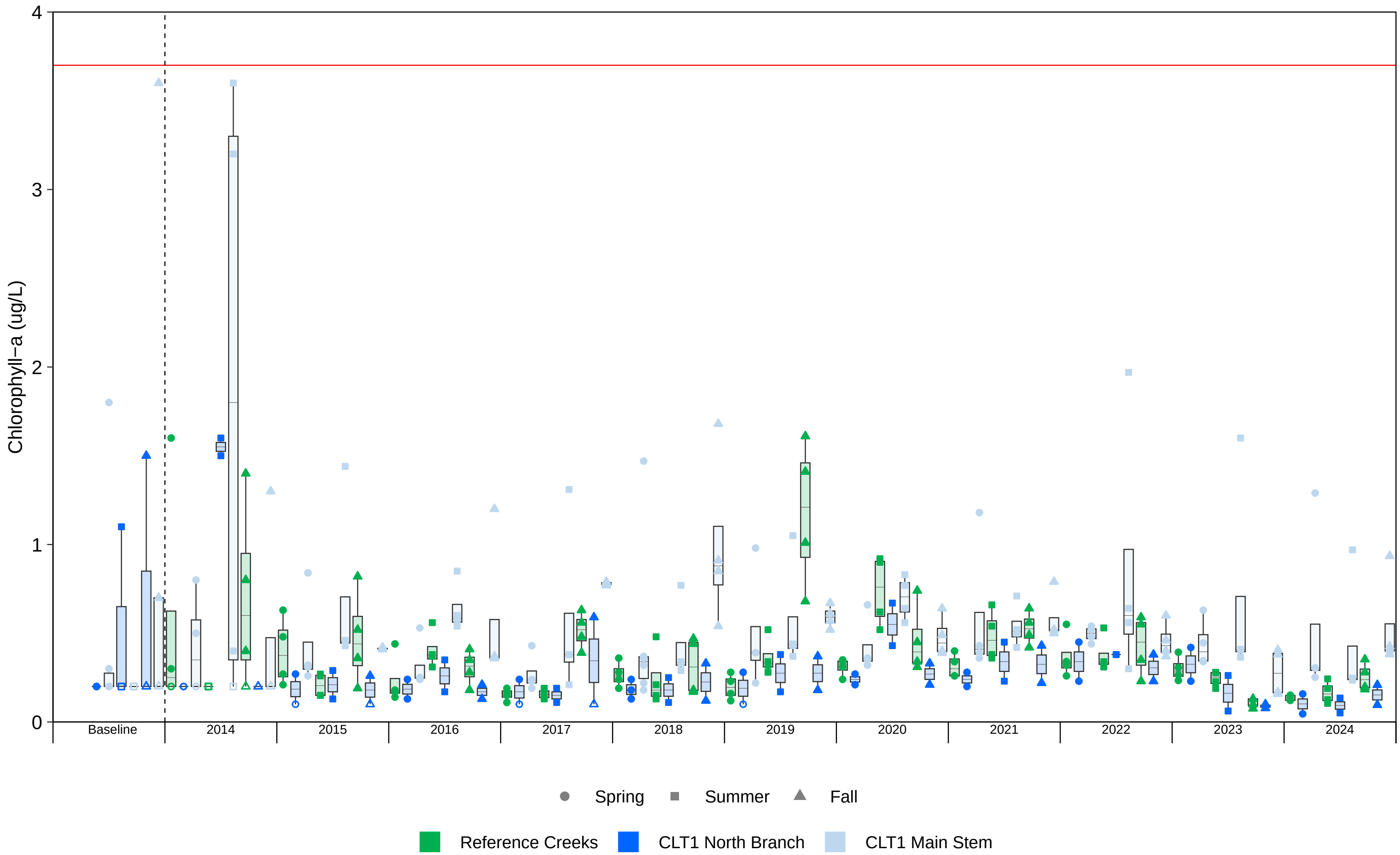


Figure 3.3: Temporal Comparison of Chlorophyll−a Concentrations at Camp Lake Tributary 1 (CLT−1) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Reference Creeks includes stations CLT−REF4, CLT−REF3, MRY−REF3, and MRY−REF2. CLT1 North Branch includes stations L1−08 and L1−02. CLT1 Main Stem includes stations L2−03, L1−09, L1−05, and L0−01. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

Main Stem. Additionally, chlorophyll-a concentrations remained consistently below the AEMP benchmark of 3.7 µg/L from 2015 to 2024 (Figure 3.3).

Overall, spatial and temporal analyses of mean chlorophyll-a concentrations suggest that mine operations may have contributed to higher phytoplankton abundance at the CLT1 Main Stem Station L2-03 during all 2024 sampling events, relative to reference streams. This relative increase in phytoplankton abundance in the CLT1 Main Stem has also been observed over the entire mine operational period, compared to baseline conditions. Elevated chlorophyll-a concentrations in the CLT1 Upper Main Stem were consistent with higher aqueous nutrient concentrations, particularly nitrate, relative to both reference streams and baseline levels (Figure 3.2, Appendix Figure C.2). While the potential source of these additional nutrients is historical quarrying activities at the QMR2 pit, there has been no blasting at this quarry pit since 2019, and nitrate concentrations in the CLT1 Upper Main Stem have decreased since that time, through remain above reference and baseline concentrations (Appendix Figure C.2). In contrast to the CLT1 Upper Main Stem, mean chlorophyll-a concentrations in the CLT1 North Branch in 2024 were similar to, or lower than, concentrations for both the mine operational period and baseline. However, these concentrations remained generally similar to the range observed in the reference streams in 2024.

Despite the slightly higher chlorophyll-a concentrations at the uppermost CLT1 Main Stem station compared to the reference streams and baseline conditions in 2024, both the CLT1 North Branch and Main Stem have remained oligotrophic and concentrations are below the AEMP benchmark. No consistent temporal patterns have been observed since the onset of commercial mine operations in 2015, based on qualitative comparisons. Overall, these results indicate no adverse mine-related effects on phytoplankton productivity at CLT1 in 2024.

3.1.3 Sediment Quality

Sediment is collected on a three-year cycle from the streams monitored in the CREMP, with samples taken in 2023. Therefore, no sediment quality sampling was completed at any of the CLT1 stream stations in 2024. In 2023, metal concentrations in sediment at CLT1 North Branch (Station CLT1-US) were generally elevated compared to reference areas, and the mean concentration of iron exceeded the SQG. The source of elevated sediment metal concentrations compared to reference in 2023 was unclear and stream sediment data are temporally limited (i.e., stream sediment sampling was initiated as a component of the CREMP in 2017), precluding comparison to baseline for a determination of mine-related influence. Though not required under the AEMP Management Response Framework, in the 2023 CREMP report (Minnow 2024a) it was recommended that temporal plots and/or statistical temporal trend analyses of the available 2017, 2020, and 2023 sediment quality data for iron be evaluated to determine whether changes in iron concentrations in sediment of the CLT1 North Branch have occurred over the sampling period during mine operations (i.e., since 2017). This was completed through a statistical temporal assessment in 2024 (see Section 2.3.3.2; Appendix Figure H.3 and Appendix Table H.7). Although iron concentrations in sediments collected in 2023 were more variable than in samples collected in 2017 and 2020, there were no statistically significant differences in mean concentrations among years (Appendix Figure H.3 and Appendix Table H.7). Therefore, the limited available data indicate that iron concentrations in sediment of the CLT1 North Branch have not increased over the mine operations period, since 2017.

In general, sediment sampling locations in CLT1, and other CREMP streams, are in highly erosional environments with very limited sediment accumulation. In this case, sediment is defined as silt and clay particles of < 63 µm as per the Wentworth particle size scale. Larger material (i.e., sand, gravel, etc.) is more geochemically inert and generally not a meaningful route of exposure to organisms. During routine CREMP stream sediment sampling events, sediment sampling is focused on locations containing the finest grain sizes available but as noted in 2023 (Minnow 2024a), fine sediments are rare, suggesting limited relevance of stream sediments as an exposure pathway to aquatic biota. Additionally, the collection of sediments in streams and rivers has the potential to produce temporally and spatially variable results for chemical analyses due to temporal fluctuations in hydrology, particularly in highly erosional environments. Therefore, the use of stream sediment chemistry results in evaluating for mine-related influences should be considered with caution.

3.1.4 Benthic Invertebrate Community

3.1.4.1 North Branch (CLT1-US)

In 2024, BIC endpoints for the CLT1 North Branch (Station CLT1-US) were generally comparable to those of the reference creek, except for the relative proportions of *Nematoda, Ostracoda, Simuliidae,* collector-gatherers, shredders, and clingers (Table 3.2, Appendix Figure F.1, Appendix Table F.6). However, the higher relative proportion of *Nematoda* in the BIC samples from CLT1-US (mean = 5.49%) relative to reference (mean = 0.704%) was the only difference that was considered ecologically meaningful based on based on a MOD beyond the CES_{BIC} (Table 3.2). The Bray-Curtis Index also showed significant differences between the CLT1 North Branch and the reference creek, further highlighting the structural or compositional differences between the BIC at CLT1 North Branch and the reference area (Appendix Table F.7).

The differences between BIC structure at CLT1 North Branch and the reference creek in 2024 are likely related to differences in habitat conditions. The higher relative proportions of *Nematoda* at CLT1 North Branch, relative to reference, are suggestive of differences in sediment or organic

Table 3.2: Statistical Comparison of Benthic Invertebrate Community Endpoints for Camp Lake Tributary 1 (CLT1) and Unnamed Reference Creek (REF-CRK) Study Areas, Mary River Project CREMP, August 2024

Endpoint	Overall Area Comparison[a]				Pair-wise, *post hoc* comparisons				
	Statistical Test	Transform-ation	Significant Difference between Areas?	P-value	Study Area	Mean	Standard Deviation	MOD[b]	Pairwise Comparison
Density (org/m^2)	ANOVA	log10	YES	0.003	Reference Creek (REF-CRK)	393	397	nc	B
					CLT1 North Branch (CLT1-US)	503	334	0.47	B
					CLT1 Upper Main Stem (CLT1-L2)	1,634	708	1.7	A
					CLT1 Lower Main Stem (CLT1-DS)	268	88.0	-0.019	B
Richness (No.Taxa)	ANOVA	log10	YES	0.008	Reference Creek (REF-CRK)	15.6	5.90	nc	A
					CLT1 North Branch (CLT1-US)	13.6	3.91	-0.30	A
					CLT1 Upper Main Stem (CLT1-L2)	14.6	2.07	-0.047	A
					CLT1 Lower Main Stem (CLT1-DS)	8.40	0.894	-1.5	B
Simpson's Evenness (Krebs)	ANOVA	none	YES	0.031	Reference Creek (REF-CRK)	0.949	0.0322	nc	A
					CLT1 North Branch (CLT1-US)	0.833	0.102	-3.6	AB
					CLT1 Upper Main Stem (CLT1-L2)	0.848	0.0261	-3.1	AB
					CLT1 Lower Main Stem (CLT1-DS)	0.751	0.149	-6.1	B
% Nematoda	ANOVA	none	YES	0.017	Reference Creek (REF-CRK)	0.704	1.00	nc	B
					CLT1 North Branch (CLT1-US)	5.49	2.98	4.8	A
					CLT1 Upper Main Stem (CLT1-L2)	2.16	1.40	1.4	AB
					CLT1 Lower Main Stem (CLT1-DS)	2.00	2.55	1.3	B
% Oligochaeta	K-W	rank	YES	0.002	Reference Creek (REF-CRK)	0.0690	0.154	nc	B
					CLT1 North Branch (CLT1-US)	0.372	0.534	nm	B
					CLT1 Upper Main Stem (CLT1-L2)	22.3	11.3	nm	A
					CLT1 Lower Main Stem (CLT1-DS)	0	0	nm	B
% Hydracarina	ANOVA	log10(x+1)	NO	0.525	Reference Creek (REF-CRK)	6.47	1.62	nc	A
					CLT1 North Branch (CLT1-US)	4.89	1.60	-0.99	A
					CLT1 Upper Main Stem (CLT1-L2)	5.00	3.78	-0.94	A
					CLT1 Lower Main Stem (CLT1-DS)	3.74	3.72	-1.7	A
% Ostracoda	K-W	rank	YES	0.008	Reference Creek (REF-CRK)	2.61	2.65	nc	A
					CLT1 North Branch (CLT1-US)	0	0	-0.72	B
					CLT1 Upper Main Stem (CLT1-L2)	0.912	1.10	-0.61	A
					CLT1 Lower Main Stem (CLT1-DS)	0	0	-0.72	B
% Chironomidae	ANOVA	log10	NO	0.113	Reference Creek (REF-CRK)	68.9	9.23	nc	A
					CLT1 North Branch (CLT1-US)	72.0	7.36	0.35	A
					CLT1 Upper Main Stem (CLT1-L2)	67.1	10.2	-0.21	A
					CLT1 Lower Main Stem (CLT1-DS)	81.3	8.51	1.2	A
% Metal Sensitive Chironomidae	ANOVA	log10(x+1)	NO	0.522	Reference Creek (REF-CRK)	8.62	10.7	nc	A
					CLT1 North Branch (CLT1-US)	17.2	9.76	0.80	A
					CLT1 Upper Main Stem (CLT1-L2)	14.0	8.67	0.52	A
					CLT1 Lower Main Stem (CLT1-DS)	11.5	8.58	0.29	A
% Simuliidae	K-W	rank	YES	0.005	Reference Creek (REF-CRK)	9.93	6.72	nc	A
					CLT1 North Branch (CLT1-US)	2.99	5.11	-1.7	B
					CLT1 Upper Main Stem (CLT1-L2)	0	0	-1.9	B
					CLT1 Lower Main Stem (CLT1-DS)	0.225	0.502	-1.9	B
% Tipulidae	K-W	rank	YES	0.075	Reference Creek (REF-CRK)	4.83	6.79	nc	B
					CLT1 North Branch (CLT1-US)	5.75	3.13	3.2	AB
					CLT1 Upper Main Stem (CLT1-L2)	1.49	0.786	0.46	B
					CLT1 Lower Main Stem (CLT1-DS)	8.47	2.94	4.9	A
% Collector Gatherers FFG	ANOVA	none	YES	0.002	Reference Creek (REF-CRK)	60.4	11.9	nc	A
					CLT1 North Branch (CLT1-US)	42.2	10.4	-1.5	BC
					CLT1 Upper Main Stem (CLT1-L2)	58.1	11.9	-0.19	AB
					CLT1 Lower Main Stem (CLT1-DS)	30.5	10.4	-2.5	C
% Filterers FFG	K-W	rank	YES	0.009	Reference Creek (REF-CRK)	1.94	1.79	nc	B
					CLT1 North Branch (CLT1-US)	1.51	2.54	-2.2	B
					CLT1 Upper Main Stem (CLT1-L2)	8.17	3.13	4.7	A
					CLT1 Lower Main Stem (CLT1-DS)	0.709	1.58	-2.9	B
% Shredders FFG	ANOVA	log10	YES	<0.001	Reference Creek (REF-CRK)	13.2	9.93	nc	C
					CLT1 North Branch (CLT1-US)	42.6	14.4	1.8	AB
					CLT1 Upper Main Stem (CLT1-L2)	27.2	9.36	1.2	B
					CLT1 Lower Main Stem (CLT1-DS)	59.8	14.3	2.2	A
% Clingers HPG	ANOVA	log10	YES	0.009	Reference Creek (REF-CRK)	24.7	11.1	nc	C
					CLT1 North Branch (CLT1-US)	44.6	12.9	1.2	AB
					CLT1 Upper Main Stem (CLT1-L2)	30.8	11.8	0.48	BC
					CLT1 Lower Main Stem (CLT1-DS)	56.0	13.6	1.7	A
% Sprawlers HPG	ANOVA	none	YES	0.084	Reference Creek (REF-CRK)	54.5	15.0	nc	A
					CLT1 North Branch (CLT1-US)	43.8	10.9	-0.72	AB
					CLT1 Upper Main Stem (CLT1-L2)	43.0	6.41	-0.77	AB
					CLT1 Lower Main Stem (CLT1-DS)	33.5	13.2	-1.4	B
% Burrowers HPG	ANOVA	log10	YES	0.030	Reference Creek (REF-CRK)	20.7	12.5	nc	AB
					CLT1 North Branch (CLT1-US)	11.6	3.07	-0.76	B
					CLT1 Upper Main Stem (CLT1-L2)	26.2	12.1	0.45	A
					CLT1 Lower Main Stem (CLT1-DS)	10.5	2.88	-0.94	B

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

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

Notes: MOD = Magnitude of Difference. nc = no comparison. nm = MOD could not be calculated due to SD = 0. FFG = Functional Feeding Group. HPG = Habitat Preference Group.

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

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

matter conditions. The sparse presence of bryophytes and algae at CLT1-US, compared to their near absence at the reference creek, may contribute to differences in organic matter conditions, potentially affecting BIC (Appendix Table F.1). Additionally, aqueous DOC concentrations were slightly higher at the CLT1 North Branch relative to the reference creeks in 2024 (though not indicative of any mine-related influence; Appendix Tables B.2 and C.14). Given that organic matter is the primary source of DOC, these higher levels at the CLT1 North Branch may also be contributing to differences in BIC. The lower relative proportions of *Ostracoda* and *Simuliidae* at the CLT1 North Branch versus reference may be attributed to substrates being significantly more embedded at the CLT1 North Branch than the reference creek in 2024 (Appendix Table F.3). Similarly, the higher relative proportions of shredders and clingers, alongside lower relative proportions of collector-gatherers, suggests a shift, relative to reference, toward taxa that process coarse organic material. Again, this observation is possibly linked to the differences in substrate embeddedness between areas and differences in bryophyte abundance (Appendix Tables F.1 and F.3). The lack of significant spatial differences in relative proportions of *Chironomidae* or metal-sensitive *Chironomidae*, which are key bioindicators, suggest that metal contamination is not a primary stressor driving the differences in the BIC at the CLT1 North Branch and reference creek. Continued monitoring is recommended to better characterize conditions contributing to the observed spatial differences in the BIC.

Most BIC endpoints for the CLT1 North Branch during mine operations years (2015 to 2024) differed significantly from at least one baseline year (2007 or 2011), except for taxonomic richness, and the relative proportions of metal sensitive *Chironomidae* and shredder FFG, which showed no significant change relative to 2007 and 2011 (Appendix Table F.8, Appendix Figure F.2). Ecologically meaningful differences over time were noted for Simpson's Evenness (2017 and 2022 relative to 2007 only) and the relative proportions of *Nematoda* (2017, 2021, 2024), *Oligiochaeta* (2020), *Hydracarina* (2016, 2017, and 2023 relative to 2007 only), *Tipulidae* (2018 relative to 2007 only), collector-gatherer FFG (2016, 2017, 2023, and 2024 relative to 2007 only), and filterer FFG (2020 relative to 2007 only; Appendix Table F.8, Appendix Figure F.2). These differences have been inconsistent over the mine operation period and were not observed for comparisons to both baseline years (2007 and 2011; Appendix Table F.8, Appendix Figure F.2). There were significantly higher relative proportions of *Nematoda* in the BIC samples from 2024 relative to both baseline years and the differences were considered ecologically meaningful (i.e., based on MODs beyond the CES_{BIC}; Appendix Table F.8, Appendix Figure F.2).

Chironomidae were the only taxon to show a consistent significant and ecologically meaningful decline across multiple mine operations years (i.e., 2015 to 2017, 2021, 2023, 2024) relative to the 2007 baseline (Appendix Table F.8, Appendix Figure F.2). However, it is possible these results are attributable, in part, to 2007 having relative proportions of *Chironomidae* that were on the high end of natural variability, given the significant differences between baseline years and the fact that the relative proportions of *Chironomidae* in the BIC during most mine operational years (i.e., from 2015 to 2017 and 2019 to 2024) were statistically similar to 2011 (Appendix Table F.8, Appendix Figure F.2). Further, the lack of significant temporal changes in the relative proportions of metal sensitive *Chironomidae* within the BIC at the CLT1 North Branch, coupled with relatively stable invertebrate densities and taxonomic richness over time, suggests the mine is not adversely affecting the BIC.

Overall, the BIC community at CLT1 North Branch in 2024 exhibited structural differences compared to reference and baseline conditions. The higher relative proportions of *Nematoda*, shredders, and clingers and lower relative proportions of *Ostracoda*, *Simuliidae*, and collector-gatherers at CLT1-US versus the reference creek may be related to differences in habitat (i.e., substrate embeddedness). A lack of significant differences in relative proportions of *Chironomidae* or metal-sensitive *Chironomidae*, which are key bioindicator taxa, between the CLT1 North Branch and reference creek suggests that metal contamination is not a primary stressor. Temporally, most BIC endpoints have differed significantly between baseline years and mine operation years; however there have been few consistent patterns over time. Although continued monitoring is necessary to track long-term patterns, results from up to 2024 do not suggest a mine-related influence on the BIC at CLT1 North Branch.

3.1.4.2 Upper Main Stem (CLT1-L2)

In 2024, a number of BIC endpoints at the CLT1 Upper Main Stem (Station CLT1-L2) were statistically significantly different from the reference creek (Table 3.2, Appendix Figure F.1, Appendix Table F.6). Specifically, density was significantly higher at the CLT1 Upper Main Stem relative to the reference creek, along with the relative proportions of *Oligochaeta*, filterer and shredder FFG, and clinger HPG (Table 3.2, Appendix Figure F.1, Appendix Table F.6). Conversely, the relative proportions of *Simuliidae* were significantly lower at the CLT1 Upper Main Stem compared to the reference creek (Table 3.2, Appendix Figure F.1, Appendix Table F.6). Of these differences, the higher relative proportion of filterers at CLT1-L2 versus the reference creek was the only one with an MOD indicative of an ecologically meaningful difference (i.e., the MOD was outside the CES_{BIC} threshold; Table 3.2). The Bray-Curtis Index was reflective of significant structural differences between the CLT1 Upper Main Stem and the reference creek BIC (Appendix Table F.7).

The differences in BIC structure between CLT1 Upper Main Stem and the reference creek in 2024 do not appear to be linked to differences in habitat between the two areas or potential mine-influence. The only significant physical habitat difference in 2024 was that substrate embeddedness was higher at CLT1 Upper Main Stem relative to the reference area (Appendix Table F.3). The higher relative proportions of *Oligochaeta*, filterers, shredders, and clingers at the CLT1 Upper Main Stem, relative to reference, suggest spatial differences in organic matter availability, and the lower proportions of *Simuliidae* at CLT-L2 may also point to potential differences in resource availability, as well as substrate composition[20]. However, no significant changes in physical habitat related to flow (Appendix Table F.3) or chlorophyll-a (a proxy for primary production) were observed (see Section 3.1.2), which does not support a conclusion of resource-driven (i.e., primary productivity) differences in community structure at CLT1 Upper Main Stem and the reference area. The greater presence of bryophytes and algae at CLT1 Upper Main Stem compared to the reference site (Appendix Table F.1), along with higher DOC concentrations (Appendix Tables B.2 and C.14), may however contribute to spatial differences in organic matter availability, potentially influencing BIC. A lack of significant differences in *Chironomidae* or metal-sensitive *Chironomidae* (key bioindicators) between CLT1 Upper Main Steam and the reference area suggests that metal contamination is not a primary stressor. The mixed responses across taxa and the absence of habitat or resource differences suggest that localized habitat variability, rather than mine-related influence is likely driving differences observed between areas in 2024.

BIC endpoints at the CLT1 Upper Main Stem were significantly different between at least one mine operational year (i.e., for at least one year between 2016 and 2024) and the baseline year (2011), except for richness and the relative proportions of *Nematoda*, *Oligochaeta*, *Chironomidae*, metal sensitive *Chironomidae*, and filterers (Appendix Table F.9, Appendix Figure F.2). BIC densities at CLT1-L2 appear to have decreased gradually over time, with differences (relative to baseline) being ecologically meaningful in 2023 and 2024 (Appendix Table F.9, Appendix Figure F.2). Other ecologically meaningful differences in BIC endpoints for 2024 relative to 2011 included higher relative proportions of *Tipulidae* and collector-gatherer FFG in the 2024 samples (Appendix Table F.9, Appendix Figure F.2). Relative proportions of *Tipulidae* in particular were significantly and meaningfully higher throughout 2021 to 2024 relative to baseline (Appendix Table F.9, Appendix Figure F.2). No other consistent, significant, and ecologically relevant patterns were identified in the temporal BIC data (Appendix Table F.9, Appendix Figure F.2).

Despite the decrease in BIC density over time at CLT1-L2, results for richness indicate that species diversity has remained stable, suggesting that the temporal decrease in BIC densities is not reflective of taxonomic groups being lost from the community over time. The increasing

[20] No statistical comparisons were made for substrate composition between CLT1 and the reference area (Minnow 2024a).

relative proportions of *Tipulidae* (crane fly larvae, which are shredders that depend on coarse organic matter) over time may reflect a rise in detrital inputs or changes in organic matter processing. Additionally, the lack of consistent temporal patterns across other endpoints, including for taxonomic groups considered as key bioindicators (e.g., metal-sensitive *Chironomidae*) are not suggestive of a progressive mine-related influence.

Overall, the 2024 BIC results for CLT1 Upper Main Stem highlight clear differences in community structure compared to reference and baseline conditions. However, results for key bioindicators (e.g., metal-sensitive *Chironomidae*) and species diversity have remained stable over time. It is possible that broader environmental factors/variability (e.g., sediment characteristics and/or organic matter availability) may be driving differences in the BIC at exposed and reference areas and among years. However, it is noteworthy that no major differences in primary production (chlorophyll-a) were detected (see Section 3.1.2), and there have been changes in water quality, as evidenced by emerging trends in concentrations of certain parameters (see Section 3.1.1). Although continued monitoring is necessary to track patterns in water quality, habitat variables, and the BIC to identify the primary factors influencing the BIC at CLT1 Upper Main Stem, results from up to 2024 do not support determination of a mine-related influence.

3.1.4.3 Lower Main Stem (CLT1-DS)

In 2024, similar to the CLT1 Upper Main Stem, a number of BIC endpoints at the CLT1 Lower Main Stem (Station CLT1-DS) were significantly different from the reference creek (Table 3.2, Appendix Figure F.1, Appendix Table F.6). Specifically, richness, Simpson's Evenness, and the relative proportions of *Ostracoda, Simuliidae,* collector-gatherer FFG, and sprawler HPG were significantly lower at the CLT1 Lower Main Stem relative to the reference creek (Table 3.2, Appendix Figure F.1, Appendix Table F.6). Conversely, the relative proportions of *Tipulidae,* shredders, and clingers were significantly higher at CLT1-DS compared to the reference creek (Table 3.2, Appendix Figure F.1, Appendix Table F.6). Of the endpoints that differed significantly between CLT1-DS and the reference area, Simpson's Evenness and the relative proportions of *Tipulidae*, and collector-gather and shredder FFG had ecologically meaningful differences, with MOD values outside the CES_{BIC} (Table 3.2). Further, the Bray-Curtis Index confirmed significant structural/compositional differences between the BIC at the CLT1 Lower Main Stem and the reference creek (Appendix Table F.7).

The higher relative proportions of *Tipulidae*, shredders, and clingers, along with lower relative proportions of *Ostracoda*, *Simuliidae*, collector-gatherers, and sprawlers suggests potential habitat modifications, such as differences in organic matter availability, hydrology, and/or substrate stability and function. However, substrate embeddedness was the only significant physical habitat difference observed between CLT1-DS (higher) and the reference (lower) creek (Appendix Table F.3). Analysis of chlorophyll-a concentrations (a proxy for primary production) show no significant changes at CLT1 compared to reference, nor signs of adverse effects that could be attributed to mining (see Section 3.1.2). However, the slightly higher presence of algae at CLT1 Lower Main Stem compared to the reference creek in 2024 (Appendix Table F.1), along with higher aqueous DOC concentrations relative to reference (Appendix Tables B.2 and C.14), may contribute to spatial differences in organic matter availability, potentially influencing BIC. The lack of significant differences in relative proportions of *Chironomidae* or metal-sensitive *Chironomidae*, key bioindicators of metal contamination, between the CLT1-DS and reference BIC suggests that mining-related stress is not likely a primary driver of the observed spatial patterns.

Temporally, most BIC endpoints at the CLT1 Lower Main Stem were significantly different during mine operation years (2015 to 2024) relative to baseline (2007, 2011), except for total invertebrate densities and the relative proportions of *Nematoda*, *Ostracoda*, *Tipulidae,* and filterer FFG (Appendix Table F.10, Appendix Figure F.2). In 2024 specifically, significant and ecologically meaningful differences between mine-operational and baseline years were identified for richness (relative to 2007 only) and relative proportions of *Hydracarina* (relative to 2011 only), both of which were lower in 2024 relative to baseline (Appendix Table F.10, Appendix Figure F.2). The relative proportions of *Hydracarina* (water mites) in the BIC at CLT1-DS have generally been lower since 2015, except in 2016 and 2022 when results were generally more comparable to baseline (Appendix Table F.10, Appendix Figure F.2). This long-term temporal pattern may be indicative of a shift in habitat conditions, possibly due to natural hydrological variability or reduced prey availability. However, because this pattern is only significant relative to 2011 (not 2007), and 2011 data are not statistically comparable to baseline data from 2007, it is considered likely that natural interannual variation is also a factor contributing to the observed results.

Historical temporal patterns (with ecological relevance [i.e., MOD was outside the CES_{BIC}]) for the relative proportions of *Oligochaeta* (2015 to 2017 higher relative to 2011), metal sensitive *Chironomidae* (2015 to 2017 lower relative to 2007), and collector-gatherer FFG (2015 to 2020 higher relative to 2011) have not continued into 2024 (Appendix Table F.10, Appendix Figure F.2). This suggests that whatever environmental changes initially influenced these taxa have not persisted, supporting the concept that conditions at CLT1-DS are variable rather than showing a progressive change over time in response to one or more stressors such as mine-influence. Other endpoints that were significantly different between mine operational years and baseline have shown inconsistent trends (Appendix Table F.10, Appendix Figure F.2), suggesting that while the community has changed over time, these shifts do not follow a clear trajectory indicative of a sustained mining-related impact.

Overall, the BIC at CLT1 Lower Main Stem in 2024 showed statistically significant differences from both reference and baseline conditions, but most were not ecologically meaningful. The lack of sustained spatial and temporal patterns across endpoints and the stability observed for endpoints associated with key bioindicators (e.g., metal-sensitive *Chironomidae*) suggest the observed differences relative to reference and over time are at least partially due to natural variability, rather than mining activities. However, it is noted that substrate embeddedness was the only significant physical habitat difference between CLT1 and the reference area in 2024. Sediment quality analyses from 2023 suggested that increased sedimentation is not occurring in the area and no significant or adverse changes in primary production (phytoplankton) were observed in 2024 (see Section 3.1.2; Appendix Table F.3; Minnow 2024a). However, higher presence of algae (Appendix Table F.1) and aqueous concentrations of DOC (Appendix Tables B.2 and C.14) at the CLT1 Lower Main Stem compared to reference areas may contribute to spatial differences in organic matter availability, potentially influencing BIC. Further, changes in water quality, as evidenced by emerging trends in aqueous concentrations of certain parameters, may also be contributing to the observed patterns in the BIC (see Section 3.1.1). Although continued monitoring is necessary to track patterns in water quality, habitat variables, and the BIC to identify the primary factors influencing the BIC at CLT1 Lower Main Stem, results from up to 2024 do not support determination of a mine-related influence.

3.1.5 Effects Assessment and Recommendations

3.1.5.1 CLT1 North Branch

At the CLT1 North Branch (Stations L1-08 and L1-02), the following AEMP benchmark was exceeded in 2024:

- The mean aqueous total copper concentration slightly exceeded the AEMP benchmark of 0.0022 mg/L, with a concentration of 0.00244 mg/L in the fall.

Although mean copper concentrations also exceeded the WQG (0.002 mg/L), they were not elevated compared to reference and baseline concentrations in any season. Furthermore, temporal trend analyses in the 2022 CREMP report indicated no significant trends in copper concentrations over the mine operation period (2015 to 2022; Minnow 2023); and copper concentrations in 2024 were consistent with mine operation and baseline ranges. Therefore, the concentration above the AEMP benchmark and WQG is likely associated with natural variation. Additionally, no other water quality parameters showed elevated concentrations compared to both the reference stream and baseline conditions in any season, suggesting no mine-related influence on water quality at the CLT1 North Branch.

Sediment is collected on a three-year cycle from the streams monitored in the CREMP, with samples taken in 2023. As a result, no stream sediment quality results from 2024 are presented in this report. A special investigation into sediment iron concentrations at the CLT1 North Branch, as recommended in the 2023 CREMP report (Minnow 2024a), found that despite sediment iron concentrations in 2023 that were generally elevated compared to reference areas and above the SQG, iron concentrations in sediment of the CLT1 North Branch have not significantly increased over the mine operations period, since 2017.

No adverse mine-related effects on chlorophyll-a (as a measure of primary productivity) or on the BIC were observed at CLT1 North Branch in 2024.

Under the Mary River Project AEMP Management Response Framework, the absence of a mine-related change in water chemistry parameters over time (or compared to background) and in biota, as concluded at CLT1 North Branch in 2024, requires no further management response (Figure 2.7).

Comparison to FEIS Predictions

A comparison of water quality at CLT1 north branch in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ2 (Site Water Management), SWSQ-7 (Camp Management), and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at CLT1 north branch was in conformance with predictions made in the Baffinland FEIS (Baffinland 2012).

3.1.5.2 CLT1 Main Stem

At the CLT1 Main Stem, the following AEMP benchmarks were exceeded in 2024:

- The aqueous total aluminum concentration exceeded the AEMP benchmark of 0.179 mg/L in spring (0.249 mg/L) at the Upper Main Stem (Station L2-03); and
- The aqueous total iron concentration exceeded the AEMP benchmark of 0.326 mg/L in spring (0.401 mg/L) and fall (0.330 mg/L) at the Upper Main Stem (Station L2-03).

Both aluminum and iron concentrations in individual seasonal samples in the Upper Main Stem also exceeded their respective WQGs (100 mg/L and 0.30 mg/L). When comparing water quality parameter concentrations to both reference and baseline across all seasons, or within a single season, the following parameters were elevated at the Upper Main Stem (Station L2-03), suggesting a potential mine-related effect:

- All seasons (spring, summer, and fall): total/dissolved molybdenum, total sodium, and sulphate;
- Spring: total aluminum, total iron, total lead, total/dissolved lithium, total/dissolved manganese, nitrate, total/dissolved potassium, total/dissolved sodium, total strontium, and total/dissolved uranium;
- Summer: total/dissolved potassium, total/dissolved sodium, and total/dissolved uranium; and
- Fall: nitrate.

Although the two parameters where AEMP benchmarks were exceeded (total aluminum and total iron) were identified in 2023 as elevated relative to reference and baseline concentrations, no mine-related influence was determined for aluminum at that time due to slight differences in the decision framework for determining a mine-related influence[21], but a mine-related influence was identified for iron. Temporal trend analyses for iron completed in 2024, based on the Moderate Action Response from the 2023 CREMP (Minnow 2024a), showed a significant increase in total iron across all seasons since the baseline period (2005 to 2024); however, no significant trends in total or dissolved iron were observed during the mine operation period (2015 to 2024). While total iron concentrations have repeatedly exceeded the AEMP benchmark and WQG, and shown increases since the baseline period, the absence of a trend during mine operations indicates a potential mine-related effect, but one that is not intensifying over time with ongoing mine operations. Similarly, for total aluminum concentrations, no clear increasing pattern is evident over the mine operation period, based on visual assessment of temporal data, though elevated spring concentrations relative to both reference and baseline, and the exceedance of both the AEMP benchmark and WQG suggest a potential mine-related influence. For both iron and aluminum, total but not dissolved concentrations in the spring were elevated compared to reference and baseline concentrations and temporal trend analyses for iron identified increasing trends for total but not dissolved concentrations. Therefore, higher total concentrations are associated with suspended solids in the water column during freshet and contributions to the CLT1 Upper Main Stem are likely related to the background minerology of the system mobilized by both natural (e.g., weathering and erosion) as well as potentially to mine-related (e.g., dust) processes.

[21] A mine-related influence designation previously required elevation in concentration relative to reference and baseline across all seasons; however, the approach has been updated in this report to also assess each season separately. If exceedances occur relative to both reference and baseline conditions in any season, a mine-related influence is now investigated for that season and/or across the open water period.

Of all the other water quality parameters with concentrations that were elevated compared to reference and baseline conditions in any given season, but were below the AEMP benchmark, total uranium was the only analyte to exceed its respective WQG of 0.015 mg/L in both the summer and fall. In 2023, uranium was identified as having a potential mine-related effect, leading to a recommended temporal trend analysis which showed significant increases in both total and dissolved uranium concentrations across all seasons since the baseline period (2005 to 2024) and during the mine operation period (2015 to 2024), particularly in the summer and when all seasons were combined. Exceedance of the WQG and increasing concentrations relative to reference and baseline levels indicate a mine-related influence. An AEMP benchmark for uranium has not yet been developed and it may be appropriate to establish one in order to better assess and manage potential mine-related influence moving forward.

Sulphate, sodium, and molybdenum were the only other parameters identified as having a potential mine-related effect in the 2023 CREMP report as well as in 2024. In 2024, these parameters had elevated concentrations relative to both reference and baseline in at least one monitoring season and exhibited statistically significant increasing temporal trends since the baseline period (2005 to 2024). While sulphate, sodium, and molybdenum concentrations that have shown increases since the baseline period indicate potential mine-related influences, the absence of statistically significant temporal trends during mine operations indicate that influences are not intensifying over time with ongoing mine operations and concentrations that remain below applicable AEMP benchmarks and WQG suggest limited potential for adverse effects to aquatic biota. For the remaining parameters at the CLT1 Upper Main Stem with concentrations that were elevated compared to reference and baseline in at least one season in 2024 (lead, lithium, manganese, nitrate, potassium, strontium), no potential mine-related influence was concluded based on a lack of any apparent directional temporal patterns observed though visual assessments.

There were no water quality parameters with concentrations above respective AEMP benchmarks or that were elevated compared to both reference and baseline conditions in any season of 2024 at the CLT1 Lower Main Stem (Stations L1-09, L1-05, and L0-01), indicating no mine-related influences on water chemistry at this area.

Sediment is collected on a three-year cycle from the streams monitored in the CREMP, with samples taken in 2023. As a result, no stream sediment quality results are presented in this report. No adverse mine-related effects on chlorophyll-a (as a measure of primary productivity) or on the BIC were determined at the any of the upper or lower stations of the CLT1 Main Stem in 2024.

Both total aluminum and total iron exceeded AEMP benchmarks in 2024, were elevated compared to reference and baseline concentrations in spring of 2024, and had increasing trends compared to baseline (though not during mine operations) suggesting a potential mine-related influence and requiring a Moderate Action Response under the AEMP Management Response Framework (Figure 2.6). Additionally, a Low Action Response is required for parameters that remained below the AEMP benchmark but showed elevated concentrations across one or more seasons in 2024 and had increasing trends over the baseline period (though not during mine operations) suggesting a potential mine-related influence (Figure 2.6). These parameters include sulphate, molybdenum, and sodium. Finally, a potential mine-related influence on uranium based on elevated concentrations relative to baseline and reference and increasing temporal trends since baseline and over the mine operations period, require a Low Action Response (Figure 2.6). Notably, iron, uranium, sulphate, molybdenum, and sodium were previously identified as having a potential mine-related influence in 2023 (Minnow 2024a).

As a Moderate Action Response within the AEMP Management Response Framework associated with aluminum and iron and a Low Action Response associated with sulphate, molybdenum, sodium, and uranium relative to reference and baseline conditions in at least one sampling season, the following actions are recommended:

- Continued monitoring of the BIC at CLT1-L2 is recommended in 2025 (and future CREMP studies) to monitor potential effects to biota and to support evaluation of elevated aluminum/iron concentrations above the AEMP benchmarks and uranium concentrations above the WQG at the CLT1 Upper Main Stem using a weight-of-evidence approach;
- In 2025, temporal trend analysis will be conducted for total and dissolved (where applicable) aqueous concentrations of sulphate, aluminum, iron, molybdenum, sodium, and uranium in the CLT1 Main Stem to continue to investigate temporal trends/patterns, evaluate for increasing trends that are indicative of intensifying mine-related influences, and confirm potential mine-related influences.
- In 2025, an analysis of total compared to dissolved aqueous concentrations of aluminum, iron, and uranium will be completed to investigate biological availability and further determine potential for effects on aquatic biota.
- Potential sources of sulphate, aluminum, iron, molybdenum, sodium, and uranium to CLT1 will be investigated to better define mine-related influence and the potential for continued contributions.
- Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this

benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.

According to the Mary River Project AEMP Management Response Framework, the absence of mine-related influences on phytoplankton (as a measure of primary productivity) and the BIC, means no further management response is required for these monitoring components at CLT1 in 2024 (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at CLT1 Upper and Lower Main Stem in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-4 (Explosives), SWSQ-5 (Quarries and Borrow Areas), SWSQ-7 (Camp Management) and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at CLT1 upper and lower main stem was in conformance with predictions made in the Baffinland FEIS (Baffinland 2012).

3.2 Camp Lake Tributary 2 (CLT2)

3.2.1 Water Quality

3.2.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at CLT2 concurrent with water quality sampling in spring, summer, and fall (Figure 2.1), as well as concurrent with BIC sampling in August (Figure 2.3). DO at CLT2 was consistently near saturation at the time of spring, summer, and fall monitoring events, and concentrations were comparable to those at the reference streams (Appendix Figure C.1, Appendix Tables C.1 to C.3). In August 2024, DO saturation was significantly higher at CLT2 upstream (mean of 114%) and CLT2 downstream (mean of 115%) than at the Unnamed Reference Creek (mean of 98.9%); and DO concentrations at CLT2 were well above the WQG lowest acceptable concentration for early life stages of cold-water biota (i.e., 9.5 mg/L; Figure 3.4, Appendix Tables C.11, C.12, and C.19).

Aqueous pH at CLT2 was generally slightly higher (i.e., more alkaline) than at the reference streams but consistently within WQG during the spring, summer, and fall sampling events in 2024 (Appendix Figure C.1, Appendix Tables C.1 to C.3). Similarly, during BIC sampling in August, pH was significantly higher at the upstream CLT2 study area compared to the Unnamed Reference Creek, but within WQG (Figure 3.4, Appendix Tables C.11, C.12, and C.19). No significant difference in pH was indicated between CLT2 study areas located downstream and upstream of

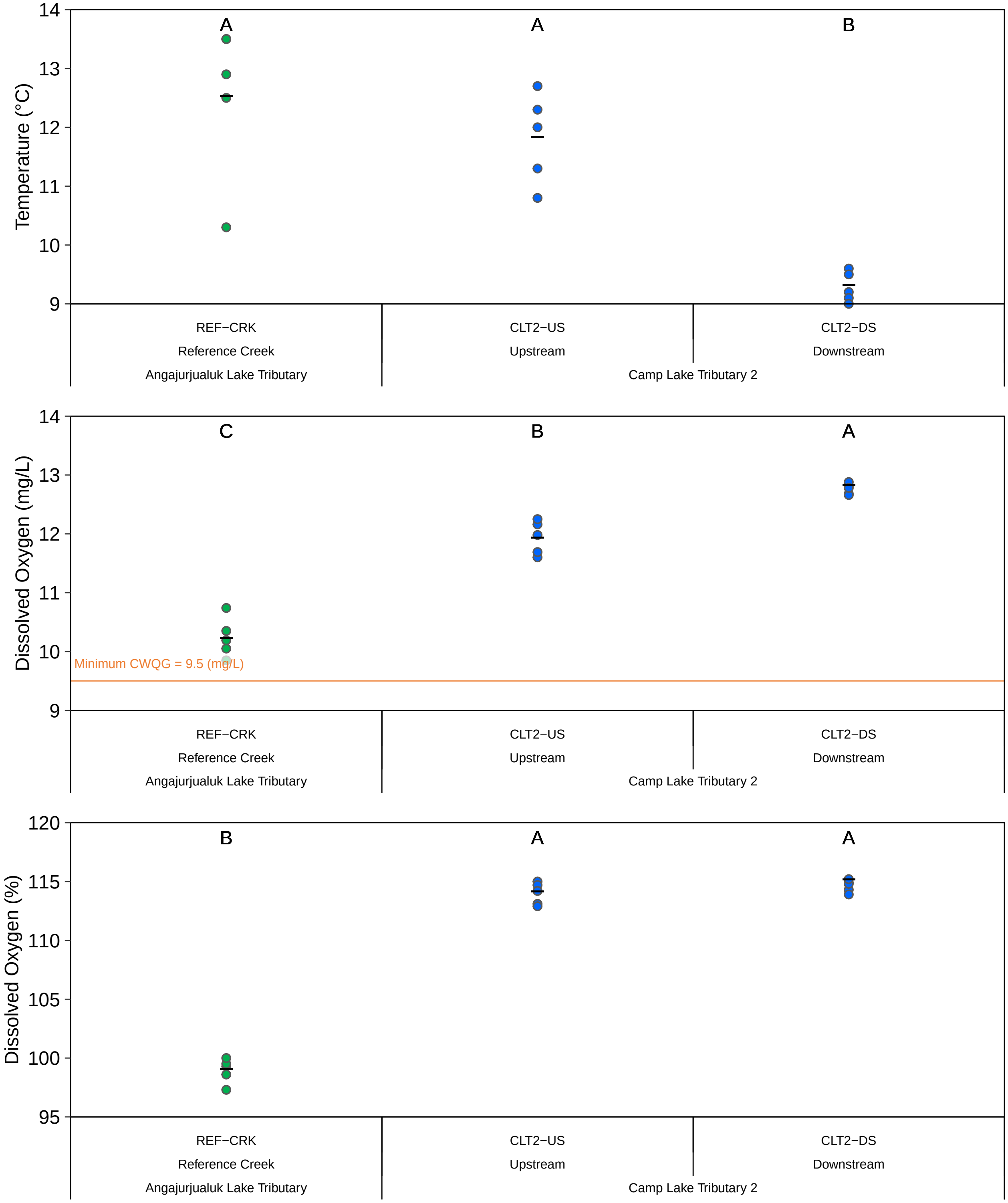


Figure 3.4: Comparison of *In Situ* Water Quality Measured at Camp Lake Tributary 2 (CLT2; Stations CLT2-US, CLT2-DS) and Reference Creek (REF−CRK) Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

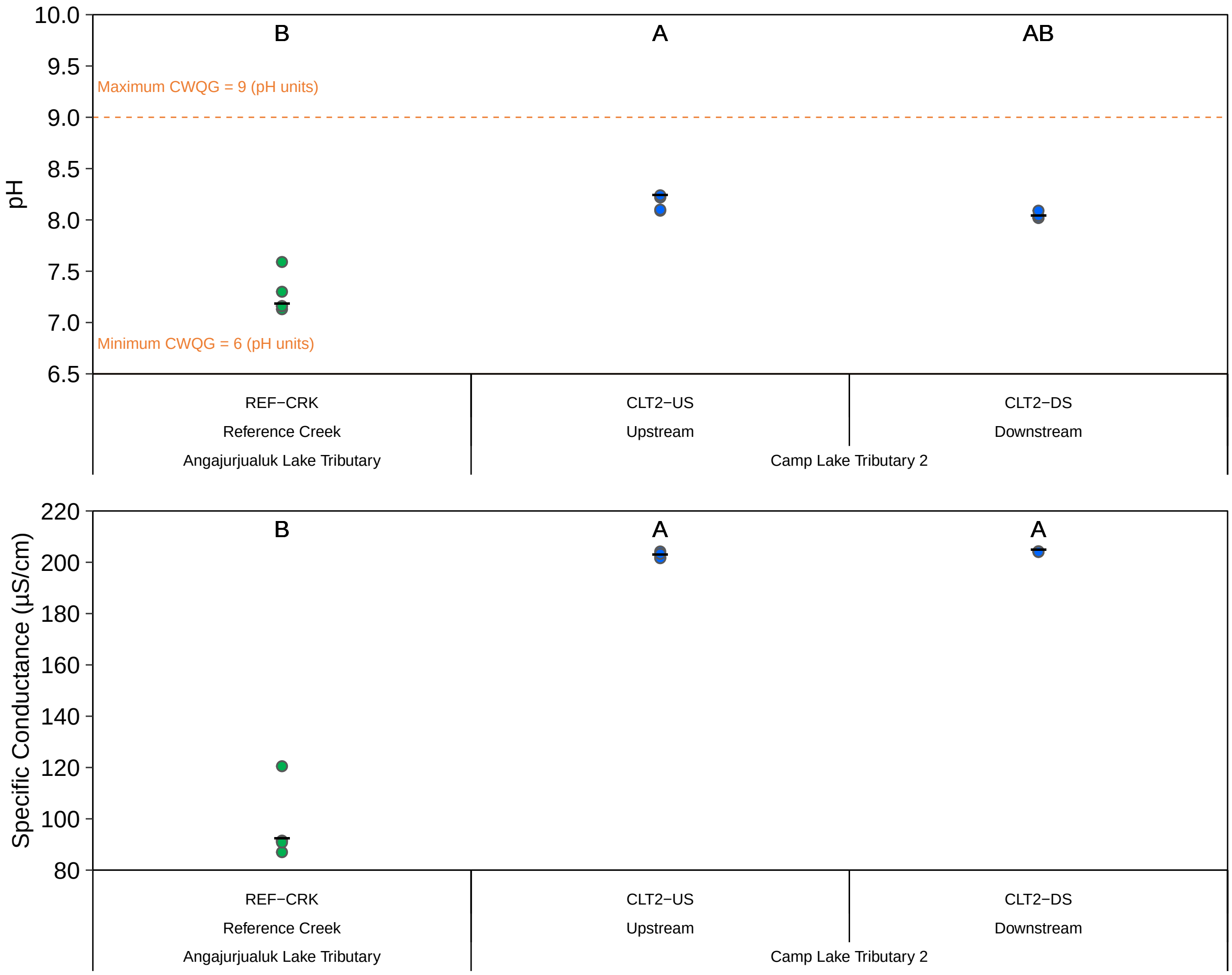


Figure 3.4: Comparison of *In Situ* Water Quality Measured at Camp Lake Tributary 2 (CLT2; Stations CLT2-US, CLT2-DS) and Reference Creek (REF−CRK) Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

the Tote Road suggesting that this road crossing did not influence the pH of CLT2 (Figure 3.4, Appendix Tables C.11, C.12, and C.19).

In situ specific conductance was consistently higher at CLT2 compared to the reference streams during spring, summer, and fall monitoring events (Appendix Figure C.1, Appendix Tables C.1 to C.3), and similarly was significantly higher at the CLT2 upstream and downstream areas compared to the Unnamed Reference Creek during BIC sampling in August 2024 (Figure 3.4, Appendix Tables C.11, C.12, and C.19). No significant difference in specific conductance was indicated between CLT2 study areas located downstream and upstream of the Tote Road, further suggesting no influence of the road crossing on water quality at the downstream CLT2 area (Figure 3.4, Appendix Tables C.11, C.12 and C.19).

3.2.1.2 Water Chemistry

Water chemistry at CLT2 (Station K0-01) met all AEMP benchmarks and WQGs during the spring, summer, and fall sampling events of 2024 (Table 3.3, Appendix Table C.14). Total and dissolved water chemistry parameter concentrations that were slightly, moderately, or highly elevated relative to reference or baseline concentrations are identified in Appendix Tables C.15, C.17, and C.18. No parameter concentrations (total or dissolved) were consistently elevated across all three seasons when compared to the reference stream stations or baseline conditions (Appendix Tables C.15 to C.18, Appendix Figure C.3). Similarly, no total or dissolved parameter concentrations were consistently elevated relative to reference and baseline conditions in any single season, except for TOC, which was moderately elevated (5 to 10 times higher) compared to reference and slightly elevated (3 to 5 times higher) compared to baseline in the spring (Appendix Tables C.15 to C.18). The TOC concentration at CLT2 in spring 2024 (15.6 mg/L) was the highest observed at CLT2 or the reference streams since the initiation of sampling in the baseline period (Appendix Figure C.3). Higher TOC in the spring can result from several factors, including increased runoff due to snow melt which carries terrestrially derived organic matter into aquatic systems, thawing of the permafrost active layer which releases additional organic material, and higher biological activity in warmer temperatures as microorganisms break down organic matter leading to temporary increases in TOC levels. Each of these influences has the potential to naturally vary annually depending on local temperature and melt conditions. However, other parameters indicative of an abnormal spring freshet event that could explain the high TOC concentration at CLT2, such as TSS, DOC, and pH, remained within the range observed in the reference streams (Appendix Tables B.2 and C.14). While TDS and conductivity were also higher at CLT2 in spring 2024 relative to reference areas, the overall similarity of freshet related parameters (i.e., TSS, DOC, pH, TDS, conductivity, TDS, conductivity) across all CLT sites, combined with the exceptionally high TOC concentration at CLT2 (up to five times greater

Table 3.3: Mean Water Chemistry at Camp Lake Tributary 2 (CLT2) Monitoring Stations During Spring, Summer, and Fall, Mary River Project CREMP, 2024

Parameters		Units	Water Quality Guideline (WQG)[a, b]	AEMP Benchmark[c]	Reference Creeks (n=4)			Camp Lake Tributary 2 (K0-01, n = 1)		
					Spring	Summer	Fall	Spring	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	28.1	82.5	107	97.8	190	236
	pH (lab)	pH	6.5 - 9.0	-	7.63	7.60	7.77	7.91	8.15	8.07
	Hardness (as $CaCO_3$)	mg/L	-	-	13.0	38.6	50.4	47.8	97.2	118
	Total Suspended Solids (TSS)	mg/L	-	-	2.65	1.27	<1	<1	<1	<1
	Total Dissolved Solids (TDS)	mg/L	-	-	25.2	48.2	48.5	55.0	112	74.0
	Turbidity	NTU	-	-	2.72	3.68	3.69	0.550	0.720	0.140
	Alkalinity (as $CaCO_3$)	mg/L	-	-	12.4	37.7	53.2	47.1	93.5	117
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00592	0.00520	0.00565	<0.005	<0.005	0.0107
	Nitrate	mg/L	3	3	<0.02	0.0240	<0.02	<0.02	<0.02	<0.02
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.101	0.0768	0.0635	0.0530	0.0880	0.164
	Dissolved Organic Carbon	mg/L	-	-	2.28	2.20	2.14	1.12	2.75	2.86
	Total Organic Carbon	mg/L	-	-	1.92	1.82	2.11	15.6	2.64	3.13
	Total Phosphorus	mg/L	0.030$^{\alpha}$	-	0.00450	0.00318	0.00335	0.00280	<0.002	<0.002
	Phenols	mg/L	0.004$^{\alpha}$	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	0.605	1.24	1.67	0.950	1.83	3.65
	Sulphate (SO_4)	mg/L	218$^{\beta}$	218	0.542	1.72	2.44	2.42	4.26	4.16
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0670	0.0832	0.160	0.0122	0.0103	0.00850
	Antimony (Sb)	mg/L	0.020$^{\alpha}$	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.000100	<0.0001	<0.0001	<0.0001
	Barium (Ba)	mg/L	1$^{\beta}$	-	0.00201	0.00480	0.00714	0.00514	0.0100	0.0126
	Beryllium (Be)	mg/L	0.011$^{\alpha}$	-	<0.00002	<0.00002	0.0000205	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Cadmium (Cd)	mg/L	0.00012	0.00008	0.00000532	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	2.56	7.68	10.3	8.95	19.2	24.0
	Chromium (Cr)	mg/L	0.0089	0.003	0.000522	0.000565	0.000672	<0.0005	<0.0005	<0.0005
	Cobalt (Co)	mg/L	0.0009$^{\alpha}$	0.0040	0.000102	0.000105	0.000125	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.0022	0.000600	0.000852	0.00114	0.000840	0.00128	0.00170
	Iron (Fe)	mg/L	0.30	0.326	0.0810	0.0942	0.143	0.0120	0.0140	0.0140
	Lead (Pb)	mg/L	0.001	0.001	0.000100	0.000114	0.000154	<0.00005	<0.00005	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.001	0.00100	0.00140
	Magnesium (Mg)	mg/L	-	-	1.74	4.96	6.36	6.66	13.2	15.8
	Manganese (Mn)	mg/L	0.935$^{\beta}$	-	0.00141	0.00126	0.00162	0.000400	0.000600	0.000660
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.0000752	0.000232	0.000420	0.000259	0.000443	0.000630
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	0.000500	0.000698	<0.0005	<0.0005	0.000660
	Potassium (K)	mg/L	-	-	0.313	0.559	0.789	0.955	1.47	1.80
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Silicon (Si)	mg/L	-	-	0.475	0.932	1.20	0.470	0.930	1.08
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.383	1.12	1.60	1.12	1.97	3.07
	Strontium (Sr)	mg/L	-	-	0.00238	0.00758	0.0111	0.00605	0.0119	0.0160
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	0.0000105	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.00410	0.00471	0.00760	0.000350	<0.0006	0.000360
	Uranium (U)	mg/L	0.015	-	0.000212	0.00115	0.00286	0.000375	0.00176	0.00270
	Vanadium (V)	mg/L	0.006$^{\alpha}$	0.006	0.000508	0.000522	0.000625	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02$^{\alpha}$	0.030	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003

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

BOLD Indicates parameter concentration above the AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

[a] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[b] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

than other CLT areas) suggests that the TOC value at CLT2 may be an anomaly or erroneous data point rather than the result of an abnormal freshet influence. Overall, no water chemistry parameter concentrations at CLT2 exceeded AEMP benchmarks or WQGs in 2024, and none were consistently elevated in any individual season when compared to reference and baseline conditions, except for TOC in spring, which was potentially an anomaly associated with natural processes occurring during spring freshet. These results indicated no mine-related influences on water quality within the CLT2 system in 2024.

3.2.2 Phytoplankton

Chlorophyll-a concentrations at CLT2 (Station K0-01) during the spring, summer, and fall sampling events of 2024 fell within the range observed at the reference streams and the average chlorophyll-a concentrations at CLT2 were consistently below the AEMP benchmark of 3.7 µg/L across all 2024 sampling events (Figure 3.2, Appendix Table E.1). Measured chlorophyll-a concentrations indicate low phytoplankton productivity and oligotrophic conditions based on chlorophyll-a concentrations (i.e., <8 µg/L; Dodds et al. (1998); Appendix Table E.1) and total phosphorus concentrations (i.e., <10 µg/L; CCME 2024b; Table 3.3, Appendix Table C.14; see Section 3.1.2 for additional trophic status classification details).

When qualitatively comparing 2024 results to baseline, construction, and mine-operational period concentrations, the average chlorophyll-a concentrations at CLT2 were comparable or lower across all seasons (Figure 3.5). Lower chlorophyll-a concentrations relative to baseline and earlier mine-operations years were also observed in reference streams, suggesting that the variation at CLT2 is likely due to natural annual fluctuations. Overall, chlorophyll-a concentrations at CLT2 exhibited no consistent directional (i.e., increasing or decreasing) temporal patterns, were consistent with those observed at reference streams, and remained well below the AEMP benchmark, indicating no mine-related effects on phytoplankton productivity at CLT2 in 2024.

3.2.3 Benthic Invertebrate Community

3.2.3.1 Upstream (CLT2-US)

In 2024, BIC endpoints for CLT2 Upstream (Station CLT2-US) were comparable to those of the reference creek, except for the relative proportion of *Ostracoda*, which was significantly lower at CLT2-US (Table 3.4, Appendix Table F.14, Appendix Figure F.3). The lower relative proportion of *Ostracoda* at CLT2-US versus the reference area in 2024 was not considered ecologically meaningful (Table 3.4). However, although only one endpoint differed significantly between areas, the Bray-Curtis Index suggested there were significant structural differences between the BIC at CLT2 Upstream and the reference creek (Appendix Table F.7).

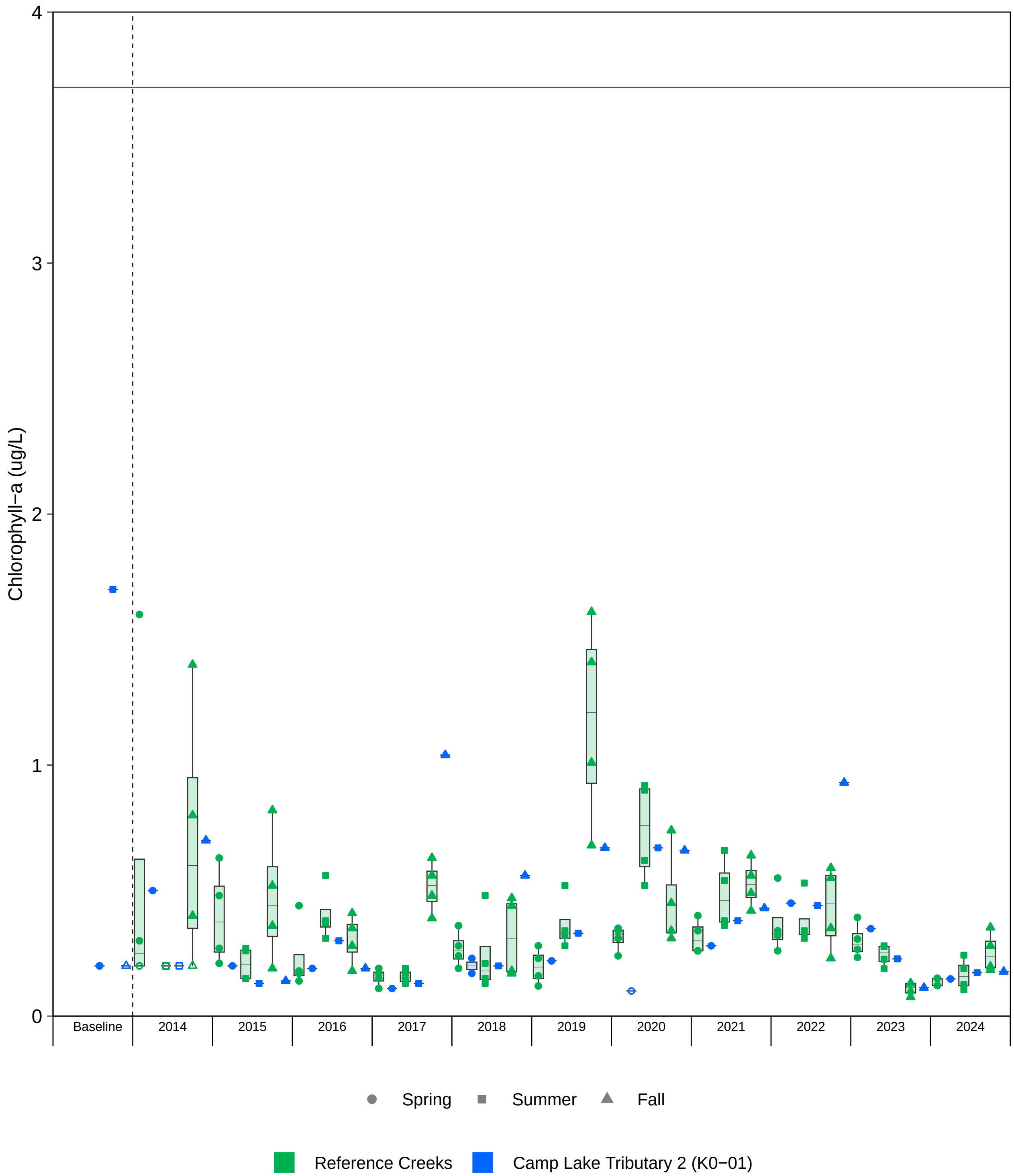


Figure 3.5: Temporal Comparison of Chlorophyll−a Concentrations at Camp Lake Tributary 2 (CLT−2) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

Table 3.4: Statistical Comparison of Benthic Invertebrate Community Endpoints for Camp Lake Tributary 2 (CLT2) and Unnamed Reference Creek (REF-CRK) Study Areas, Mary River Project CREMP, August 2024

Endpoint	Overall Area Comparison[a]				Pair-wise, *post hoc* comparisons				
	Statistical Test	Transform-ation	Significant Difference between Areas?	P-value	Study Area	Mean	Standard Deviation	MOD[b]	Pairwise Comparison
Density (org/m²)	ANOVA	log10	NO	0.916	Reference Creek (REF-CRK)	393	397	nc	A
					CLT2 Upstream (CLT2-US)	224	105	-0.21	A
					CLT2 Downstream (CLT2-DS)	409	557	-0.22	A
Richness (No. Taxa)	ANOVA	log10	NO	0.767	Reference Creek (REF-CRK)	15.6	5.90	nc	A
					CLT2 Upstream (CLT2-US)	13.6	4.22	-0.33	A
					CLT2 Downstream (CLT2-DS)	13.2	3.42	-0.36	A
Simpson's Evenness (Krebs)	ANOVA	none	NO	0.393	Reference Creek (REF-CRK)	0.949	0.0322	nc	A
					CLT2 Upstream (CLT2-US)	0.908	0.0493	-1.3	A
					CLT2 Downstream (CLT2-DS)	0.905	0.0735	-1.4	A
% Nematoda	K-W	rank	NO	0.393	Reference Creek (REF-CRK)	0.704	1.00	nc	A
					CLT2 Upstream (CLT2-US)	1.18	1.45	nm	A
					CLT2 Downstream (CLT2-DS)	11.6	19.9	nm	A
% Oligochaeta	ANOVA	log10(x+1)	NO	0.161	Reference Creek (REF-CRK)	0.0690	0.154	nc	A
					CLT2 Upstream (CLT2-US)	6.33	7.15	38	A
					CLT2 Downstream (CLT2-DS)	3.21	4.40	20	A
% Hydracarina	ANOVA	log10	YES	0.026	Reference Creek (REF-CRK)	6.47	1.62	nc	AB
					CLT2 Upstream (CLT2-US)	11.7	7.15	2.0	A
					CLT2 Downstream (CLT2-DS)	4.10	3.69	-3.1	B
% Ostracoda	K-W	rank	YES	0.032	Reference Creek (REF-CRK)	2.61	2.65	nc	A
					CLT2 Upstream (CLT2-US)	0	0	-0.72	B
					CLT2 Downstream (CLT2-DS)	0	0	-0.72	B
% Chironomidae	ANOVA	none	NO	0.481	Reference Creek (REF-CRK)	68.9	9.23	nc	A
					CLT2 Upstream (CLT2-US)	59.4	7.97	-1.0	A
					CLT2 Downstream (CLT2-DS)	64.0	16.9	-0.53	A
% Metal Sensitive Chironomidae	ANOVA	log10(x+1)	YES	0.075	Reference Creek (REF-CRK)	8.62	10.7	nc	B
					CLT2 Upstream (CLT2-US)	19.1	8.05	0.98	AB
					CLT2 Downstream (CLT2-DS)	27.8	16.9	1.7	A
% Simuliidae	ANOVA	log10	NO	0.109	Reference Creek (REF-CRK)	9.93	6.72	nc	A
					CLT2 Upstream (CLT2-US)	7.06	1.88	-0.26	A
					CLT2 Downstream (CLT2-DS)	4.39	3.71	-1.3	A
% Tipulidae	K-W	rank	NO	0.278	Reference Creek (REF-CRK)	4.83	6.79	nc	A
					CLT2 Upstream (CLT2-US)	2.53	2.40	0.47	A
					CLT2 Downstream (CLT2-DS)	7.40	5.02	2.9	A
% Collector Gatherers FFG	ANOVA	none	YES	0.052	Reference Creek (REF-CRK)	60.4	11.9	nc	AB
					CLT2 Upstream (CLT2-US)	55.1	12.0	-0.44	B
					CLT2 Downstream (CLT2-DS)	74.2	9.97	1.2	A
% Filterers FFG	ANOVA	none	YES	0.004	Reference Creek (REF-CRK)	1.94	1.79	nc	AB
					CLT2 Upstream (CLT2-US)	3.84	1.63	1.1	A
					CLT2 Downstream (CLT2-DS)	0.104	0.233	-1.0	B
% Shredders FFG	ANOVA	log10	NO	0.754	Reference Creek (REF-CRK)	13.2	9.93	nc	A
					CLT2 Upstream (CLT2-US)	15.0	11.3	0.10	A
					CLT2 Downstream (CLT2-DS)	16.8	6.99	0.46	A
% Clingers HPG	ANOVA	none	NO	0.351	Reference Creek (REF-CRK)	24.7	11.1	nc	A
					CLT2 Upstream (CLT2-US)	31.0	19.9	0.57	A
					CLT2 Downstream (CLT2-DS)	17.3	9.97	-0.67	A
% Sprawlers HPG	ANOVA	none	NO	0.852	Reference Creek (REF-CRK)	54.5	15.0	nc	A
					CLT2 Upstream (CLT2-US)	57.1	17.8	0.17	A
					CLT2 Downstream (CLT2-DS)	60.4	16.1	0.39	A
% Burrowers HPG	ANOVA	log10	NO	0.325	Reference Creek (REF-CRK)	20.7	12.5	nc	A
					CLT2 Upstream (CLT2-US)	11.9	8.94	-1.1	A
					CLT2 Downstream (CLT2-DS)	22.3	21.3	-0.086	A

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

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

Notes: MOD = Magnitude of Difference. nc = no comparison. nm = MOD could not be calculated due to SD = 0. FFG = Functional Feeding Groups. HPG = Habitat Preference Groups.

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

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

The lower relative proportion of *Ostracoda* at CLT2 Upstream, relative to reference, suggests there is minor, natural variation in community composition between areas; however, overall, the BIC data for 2024 are not suggestive of mine-related effects at CLT2 Upstream. The subtle differences in the BIC between CLT2-US and the reference creek may be attributed to differences in water depth and substrate embeddedness, which were significantly different between CLT2 Upstream and the reference area in 2024 (Appendix Table F.12). The conclusion that natural conditions rather than mine-related factors are the dominant influence on BIC is supported by the absence of adverse change in water quality and chlorophyll-a concentrations (a proxy for primary productivity) at CLT2-US compared to reference in 2024 (see Sections 3.2.1 and 3.2.2).

Most BIC endpoints at CLT2 Upstream varied significantly over time, with results from at least one mine operations year (2015 to 2024) being different from baseline (2007). Exceptions were total invertebrate densities and relative proportions of *Nematoda, Oligochaeta, Ostracoda*, and *Tipulidae* (Appendix Table F.15, Appendix Figure F.4). In 2024 specifically, the relative proportion of *Hydracarina* was significantly higher, whereas the relative proportion of *Chironomidae* was significantly lower, relative to baseline and both differences were ecologically meaningful (Appendix Table F.15, Appendix Figure F.4). *Chironomidae* relative proportions have gradually declined at CLT2 Upstream since 2015, and differences in 2021, 2023, and 2024, relative to baseline, were ecologically meaningful (Appendix Table F.15, Appendix Figure F.4). The long-term, gradual decrease in relative proportions of *Chironomidae* suggests a gradual shift in macroinvertebrate community structure. However, the absence of a similar temporal pattern for metal-sensitive *Chironomidae* at CLT2-US suggests that this pattern is not linked to mining-related metal contamination.

Longer-term ecologically significant temporal patterns that have been previously observed, such as higher Simpson's Evenness in 2015, 2017 to 2019, and 2023 and lower proportions of shredder FFG in 2017, and 2020 to 2022 relative to baseline, were not observed in 2024 (Appendix Table F.15, Appendix Figure F.4). This suggests short-term shifts in community structure rather than a sustained environmental stressor. For other endpoints that showed significant differences between mine operation years and baseline, patterns of increase or decrease and ecological significance have been inconsistent (Appendix Table F.15, Appendix Figure F.4), further suggesting that broader environmental variability or habitat shifts may be influencing the observed patterns.

Overall, the BIC at CLT2 Upstream in 2024 was similar to the reference creek (except for lower relative proportions of *Ostracoda*), which suggests there is no adverse mine-related influence on the BIC at CLT2 Upstream. This is further supported by the absence of adverse differences in water quality or chlorophyll-a concentrations at this location. The increase in relative proportions of *Hydracarina* and concomitant decrease in relative proportions of *Chironomidae* over time at CLT2 Upstream suggest shifts in habitat conditions and organic matter dynamics over time. However, the absence of sustained increasing or increasing patterns in other endpoints and the stability of key bioindicators (e.g., metal-sensitive *Chironomidae*) suggest the observed temporal changes are more likely due to natural variability than mine-related influence.

3.2.3.2 Downstream (CLT2-DS)

In 2024, similar to the upstream area, the BIC at CLT2 Downstream (Station CLT2-DS) was comparable to the reference creek except for lower relative proportions of *Ostracoda* and higher relative proportions of metal sensitive *Chironomidae* at CLT2-DS (Table 3.4, Appendix Table F.14, Appendix Figure F.3). These differences were not ecologically meaningful (Table 3.4, Appendix Table F.14, Appendix Figure F.3). Although only two endpoints differed between CLT2-DS and the reference creek, the Bray-Curtis Index suggested significant structural differences between the BIC at CLT2 Downstream and the reference creek (Appendix Table F.7).

The lower relative proportions of *Ostracoda* alongside higher relative proportions of metal-sensitive *Chironomidae* at CLT2 Downstream suggests that localized habitat variability, rather than mining-related influences, are driving differences in the BIC from reference conditions. However, it is unclear which environmental factors are driving the small number of observed differences between CLT2-DS and reference BIC, given no significant physical habitat differences were observed between CLT2 Downstream and the reference creek in 2024 (Appendix Table F.12).

Temporally, most BIC endpoints at CLT2 Downstream were statistically similar among mine operations years (2015 to 2024) and the baseline period (2007; Appendix Table F.16, Appendix Figure F.4). Specifically, relative proportions of *Chironomidae* (2015 and 2021 to 2024), *Tipulidae* (2017), and filterers (2024) were the only endpoints that differed during mine operations years relative to baseline (Appendix Table F.16, Appendix Figure F.4). Of these differences, only those for the relative proportion of *Chironomidae* were ecologically meaningful and sustained over consecutive years (i.e., 2021 to 2024; Appendix Table F.6). Although these results for *Chironomidae* may represent a recent shift in invertebrate community structure relative to baseline, it is possible that there was an earlier shift sometime between 2007 and 2015, given relative proportions of *Chironomidae* were statistically comparable throughout 2015 to 2024 (Appendix Table F.16, Appendix Figure F.4). Regardless, the absence of temporal changes in invertebrate densities, richness, and relative proportions of key bioindicators (e.g., metal-sensitive *Chironomidae*) suggests that the small number of differences observed in the CLT2 Downstream BIC over time are not linked to metal contamination or mining-related stressors.

Overall, the BIC at CLT2 Downstream in 2024 was largely similar to the reference creek and baseline. The absence of significant differences between CLT2-DS and reference areas, other than lower proportions of *Ostracoda* and higher proportions of metal-sensitive *Chironomidae* at CLT2-DS, suggests a lack of mine-related effects at CLT2 Downstream. Further, the consistency of nearly all BIC endpoints over time at CLT2 Downstream, from baseline through to 2024, including those for key bioindicators (e.g., metal-sensitive *Chironomidae*), supports the conclusion of no mine-related effects to the BIC at that location. Finally, water quality, chlorophyll-a concentrations (a proxy for primary productivity), and physical habitat assessments for 2024 do not indicate there have been changes in resource availability or habitat at CLT2 Downstream (see Sections 3.2.1 and 3.2.2; Appendix Table F.12).

3.2.4 Effects Assessment and Recommendations

In 2024, water chemistry parameter concentrations at CLT2 (Station K0-01) met all AEMP benchmarks, and no parameters had concentrations that were elevated compared to both reference and baseline in any season, indicating no mine-related influence on water quality at this location. Sediment is collected every three years from streams monitored under the CREMP, with samples taken in 2023; therefore, sediment quality results are not included in this report. No adverse mine-related effects on chlorophyll-a (a measure of primary productivity) or to the BIC were observed at CLT2 in 2024.

According to the Mary River Project AEMP Management Response Framework, the absence of any mine-related changes in water chemistry concentrations or to biota, as observed at CLT2 in 2024, requires no further management action (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at CLT2 in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-7 (Camp Management) and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at CLT2 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

3.3 Camp Lake (JL0)

3.3.1 Water Quality

3.3.1.1 *In Situ* Water Quality

In 2024, profiles were developed from *in situ* water quality measured concurrent with water quality sampling in winter, summer, and fall (Figure 2.1), and *in situ* water quality was measured at the top and bottom of the water column concurrent with BIC sampling in August (Figure 2.3). *In situ* water quality profiles at Camp Lake showed no substantial spatial differences in water temperature, DO, pH, or specific conductance. Specifically, there was no patterns detected with progression from the CLT1 inlet to the lake outlet (JL0-02, JL0-10, JL0-01, JL0-07 and JL0-09) during any of the winter, summer, or fall seasonal sampling events in 2024 (Appendix Figures C.4 to C.7, Appendix Tables C.20 to C.22). During August 2024 BIC sampling, the bottom temperature, pH, and specific conductance differed significantly between littoral and profundal stations within the lake but these small magnitude differences were expected based on known depth-related patterns of *in situ* water quality measures (Appendix Tables C.23 and C.24). Similar differences were observed for temperature at Reference Lake 3 (Appendix Table C.8).

The 2024 Camp Lake water column profiles (mean measures) indicated a slight increase in temperature from surface to bottom in the winter (i.e., up to approximately 2˚C), while in summer there was a slight decrease in temperature from top to bottom (i.e., up to approximately 2˚C; Figure 3.6). However, temperature was consistent from surface to bottom during the fall sampling event which captured the period of fall lake turnover (Figure 3.6). The temperature profile at Camp Lake in summer differed from the reference lake, as Camp Lake had a warmer surface layer that cooled with depth by approximately 2°C, while Reference Lake 3 was a consistent temperature throughout the depth profile (Figure 3.6). Conversely, the profile from the fall sampling event at Camp Lake was similar to that of Reference Lake 3 in 2024 where there was no stratification, but Camp Lake was approximately 0.5 ˚C warmer than the reference lake (Figure 3.6). During August 2024 biological monitoring, water temperature at both profundal and littoral stations differed significantly but slightly (~1°C difference in means) between Camp Lake and Reference Lake 3 (Figure 3.7, Appendix Tables C.23 and C.25).

Dissolved oxygen profiles at Camp Lake and Reference Lake 3 were relatively consistent from surface to bottom during the summer and fall, whereas in the winter (Camp Lake only), saturation and concentration of dissolved oxygen declined with depth beginning at approximately 10 m below surface (Figure 3.6). At the time of BIC sampling in August 2024, dissolved oxygen at the bottom of the water column was near full saturation (> 99 %) at littoral and profundal stations of Camp Lake and was similar to like-habitat in Reference Lake 3 (Figure 3.7, Appendix

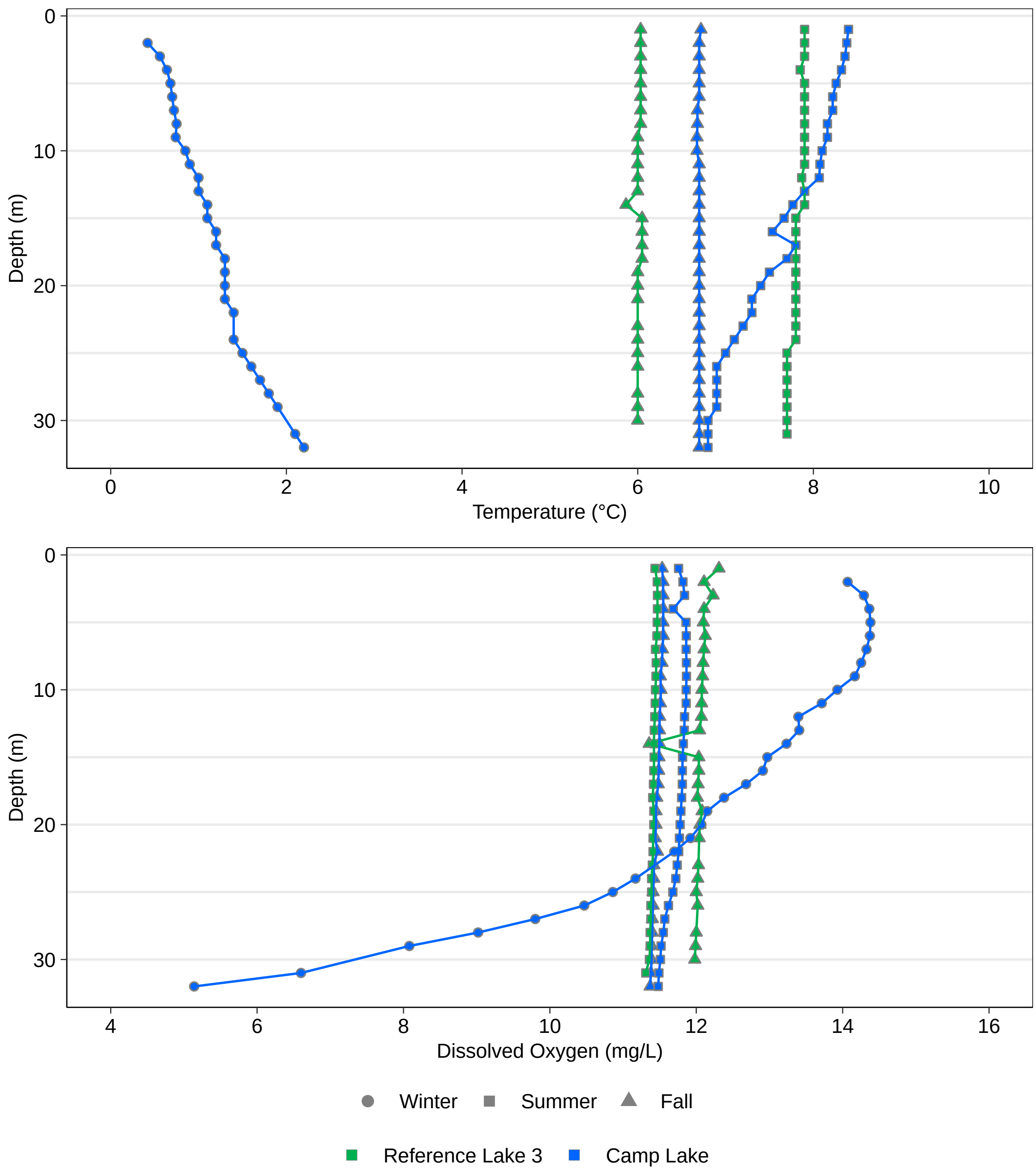
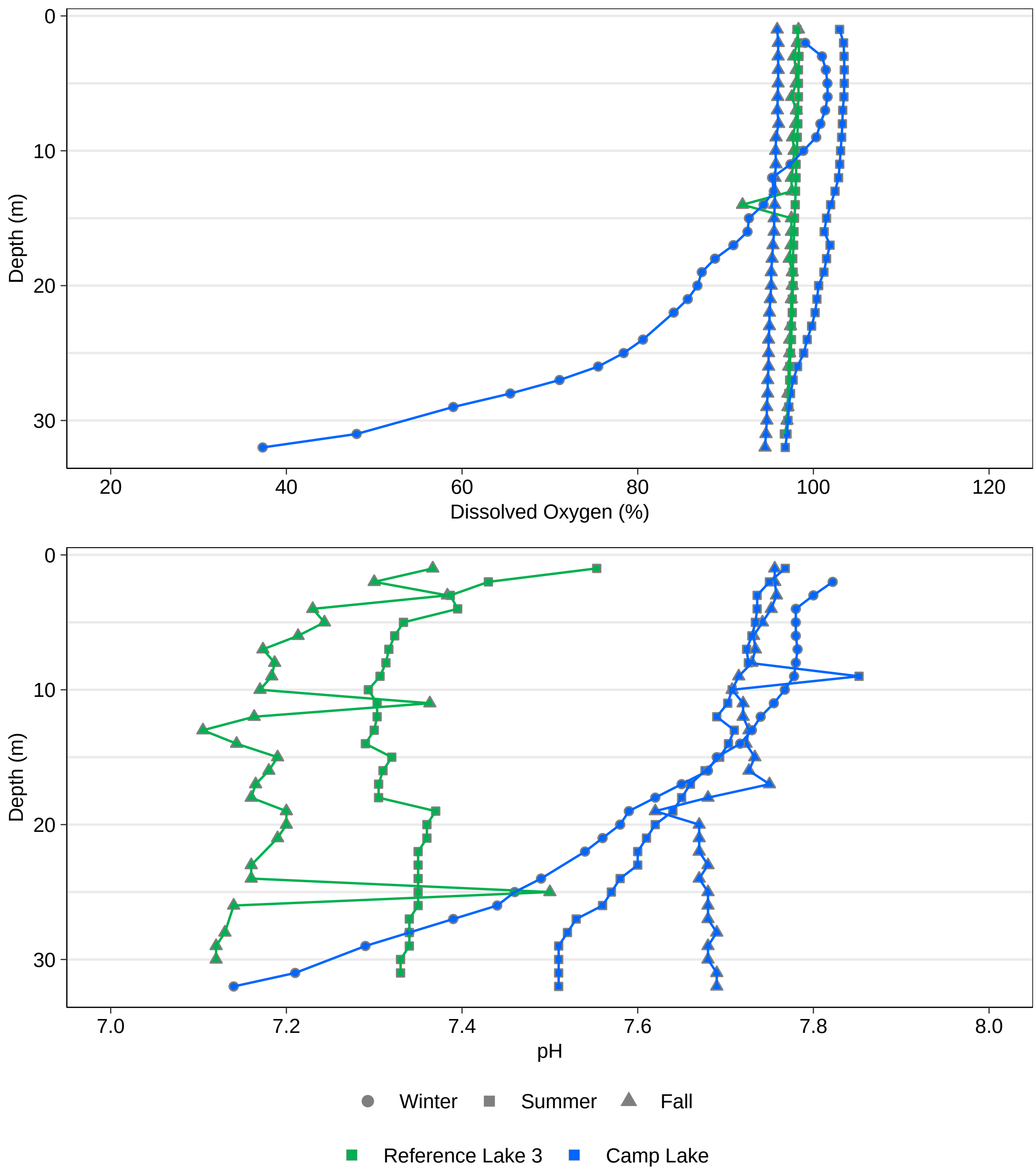


Figure 3.6: Average In Situ Water Quality with Depth from Surface at Camp Lake Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024



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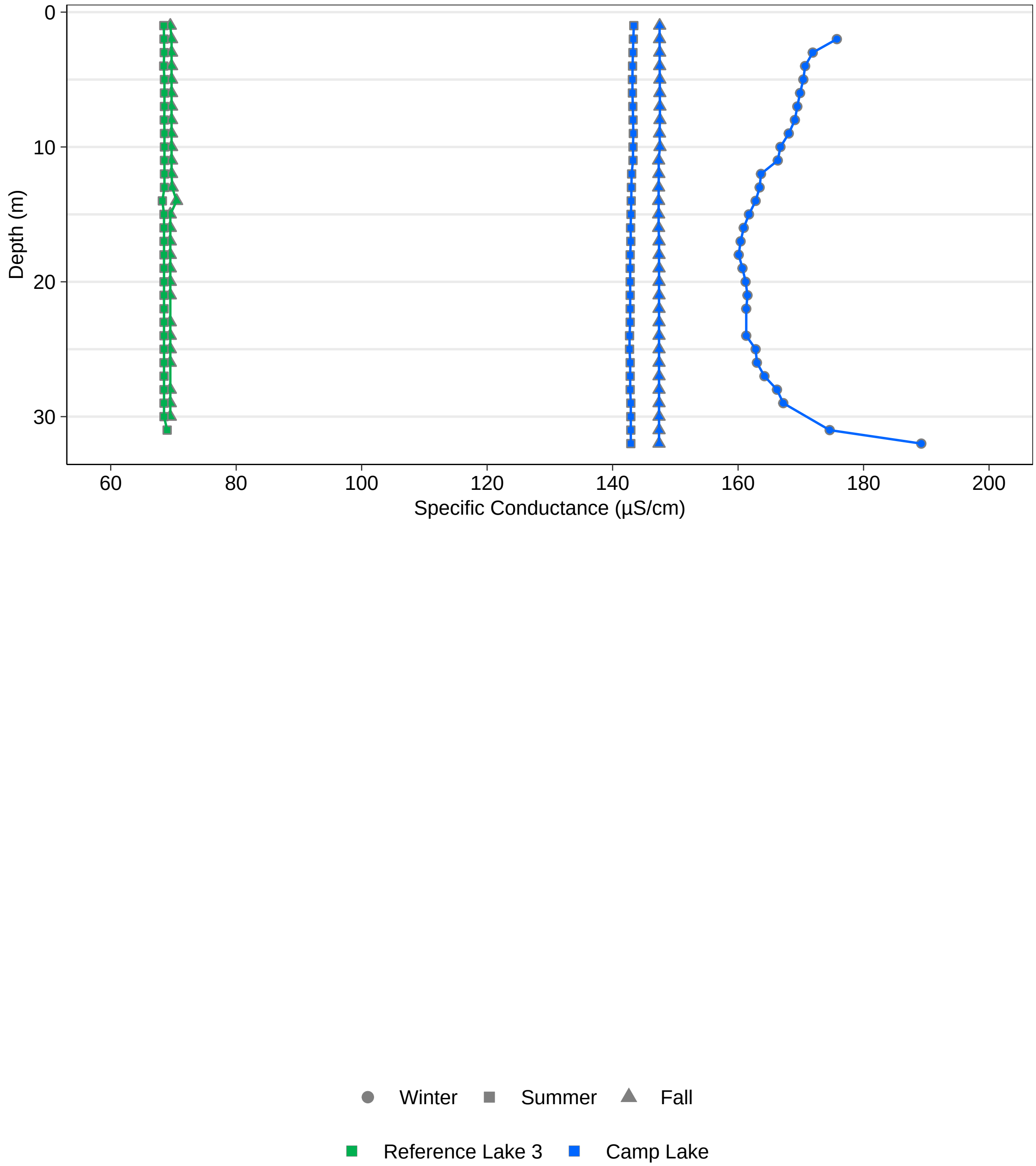

Figure 3.6: Average In Situ Water Quality with Depth from Surface at Camp Lake Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

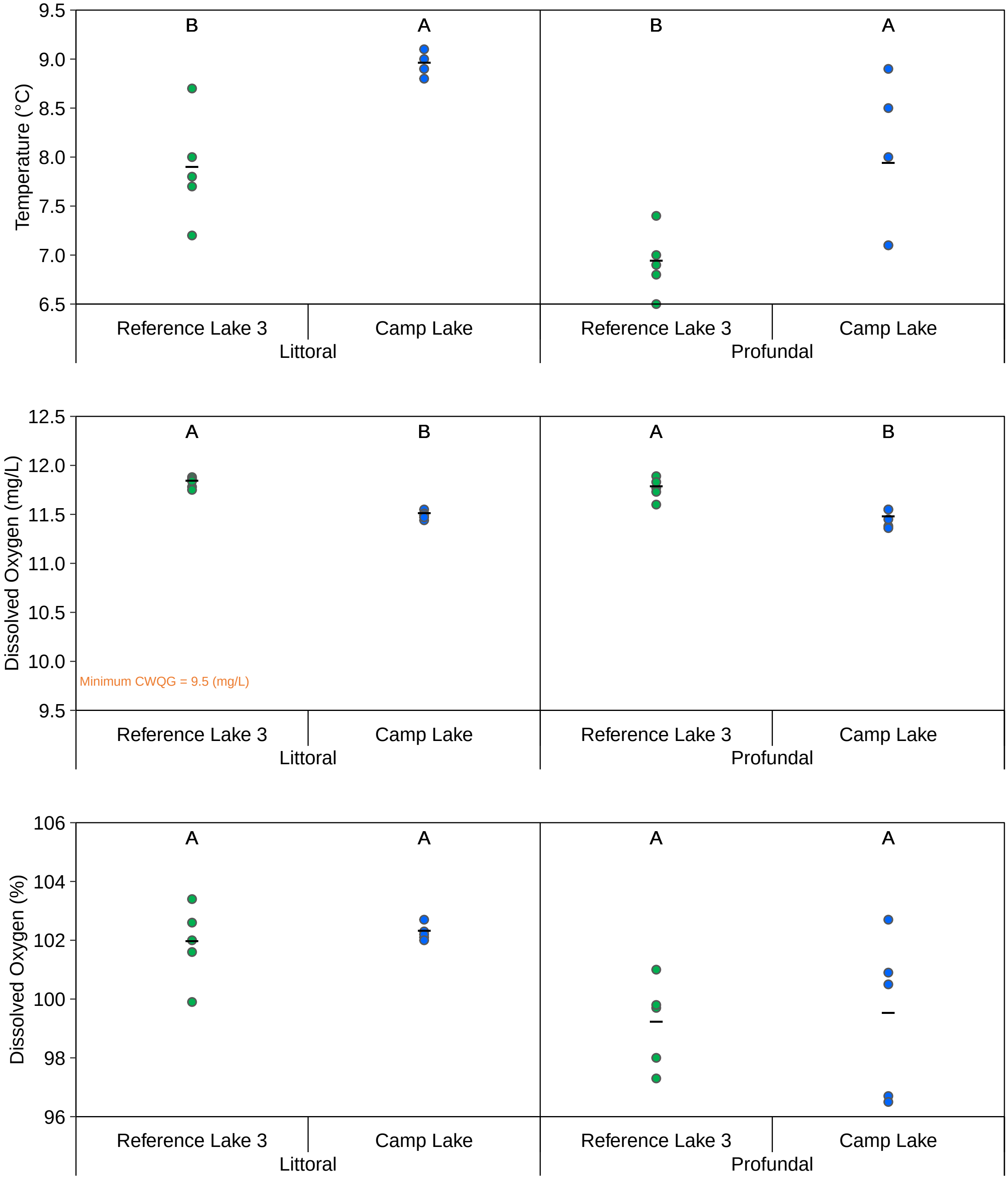

Figure 3.7: Comparison of *In Situ* Water Quality Measured at Camp Lake (JL0) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

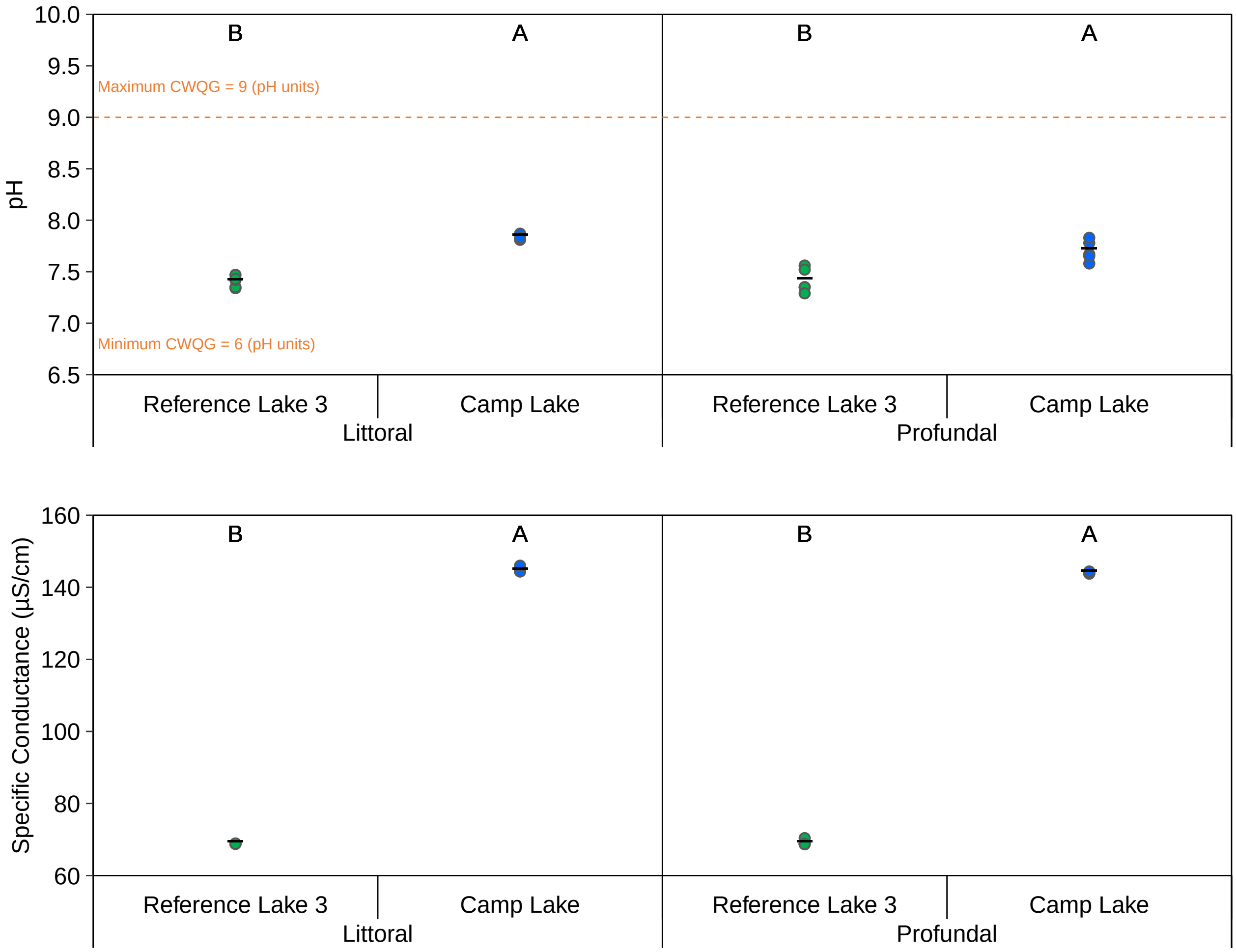

Figure 3.7: Comparison of *In Situ* Water Quality Measured at Camp Lake (JL0) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

Tables C.23 and C.25). Dissolved oxygen concentrations at Camp Lake were well above the WQG minimum for the protection of early life stages of cold-water biota (i.e., 9.5 mg/L) during all sampling events in 2024, except at water depths greater than 28 m (i.e., within 2 m of the lake bottom at the deepest station) in winter (Appendix Tables C.20 to C.22). This pattern is expected at depth, under-ice in arctic lakes and suggested that dissolved oxygen concentrations were not likely to be limiting to benthic invertebrates or fish at Camp Lake for most of the year, except for in the limited portion of the lake with a depth greater than 28 m during the winter.

In 2024, water column profiles showed a slight decrease in pH with depth at Camp Lake in the winter (pH ~7.8 to 7.1) and summer (pH ~ 7.8 to 7.5), and consistent pH (range between ~ 7.8 and 7.7) in the fall (Figure 3.6). This is generally consistent with Reference Lake 3, where water column profiles showed a slight decrease in pH with depth in both the summer (pH ~7.6 to 7.3) and the fall (pH ~7.3 to 7.1; Figure 3.6). Although pH values near the bottom at littoral and profundal stations of Camp Lake were significantly higher than at the reference lake during the August 2024 BIC sampling, the mean incremental difference in pH between lakes was small (i.e., approximately 0.4 pH units) and all pH values were consistently within WQG limits (Figure 3.7, Appendix Tables C.20 to C,22 and C.25).

Mean specific conductance profiles at Camp Lake in the winter showed a decrease with depth from approximately 1 to 17 m (178 to 160 µS/cm), before rising again from approximately 17 to 29 m (back to 167 µS/cm; Figure 3.6), as is a common pattern for arctic lakes during the winter (Vidal and Macintyre 2011). There were no depth-related trends in specific conductance during summer or fall (Figure 3.6). Overall, specific conductance was higher at Camp Lake in winter compared to other seasons (Figure 3.6), likely due to the absence of dilution originating from tributaries due to complete freezing of these watercourses. Specific conductance was consistently higher at Camp Lake compared to Reference Lake 3 during summer and fall sampling events (Figure 3.6), and significantly so during the August 2024 biological study (Figure 3.7, Appendix Table C.25). Specific conductance at CLT1 was elevated relative to reference streams in 2024 (see Section 3.1.1) but no spatial gradient was evident across Camp Lake water quality sampling stations (Appendix Figure C.7, Appendix Tables C.20 to C.23) suggesting limited influence of CLT1 inflows as a potential source of elevated specific conductance in Camp Lake. Secchi depth readings, which serve as a proxy for water clarity, were significantly lower at Camp Lake than at Reference Lake 3 during the August 2024 biological study (Appendix Figure C.8) indicating more suspended particulate material in waters of Camp Lake.

3.3.1.2 Water Chemistry

Water chemistry at Camp Lake met all AEMP benchmarks and WQGs during the winter, summer, and fall sampling events in 2024 (Table 3.5, Appendix Table C.26). Total and dissolved water chemistry parameter concentrations that were slightly, moderately, or highly elevated relative to reference or baseline concentrations are identified in Appendix Tables C.27 and C.29. No parameter concentrations (total or dissolved) were consistently elevated compared to both Reference Lake 3 and baseline conditions across all seasons (Appendix Tables C.27 and C.29, Appendix Figure C.9), and only total uranium concentrations were consistently elevated relative to Reference Lake 3 and baseline conditions in any single season (moderately elevated [5 to 10 times higher] compared to Reference Lake 3 and slightly elevated [3 to 5 times higher] compared to baseline concentrations during the summer; Appendix Table C.27, Appendix Figure C.9). Dissolved uranium concentrations were also moderately elevated relative to Reference Lake 3 in summer (Appendix Tables C.28 and C.29), suggesting that higher concentrations were also present in a bioavailable form.

Visual assessment of temporal data indicates that total uranium concentrations have been elevated relative to Reference Lake 3 since 2015 and relative to baseline since 2017, with a defined increase in concentration from 2017 to 2022 (Appendix Figure C.9). Although an AEMP benchmark has not been established for uranium, concentrations remained well below the WQG of 0.015 mg/L throughout the mine operation period (2015 to 2024; Table 3.1, Appendix Table C.26, Appendix Figure C.9). Notably, in 2024, total uranium concentrations were elevated relative to reference streams and baseline conditions across all seasons at the CLT1 Upper Main Stem (see Section 3.1.1.3.1), possibly contributing to the concentrations observed in Camp Lake. Interpretation of uranium concentration data from the CLT1 Upper Main Stem and reference streams suggests that inputs of uranium to the Camp Lake system (i.e., Camp Lake and CLT1 Upper Main Stem) may be influenced by potential naturally occurring regional increases in uranium concentrations in streams (as observed in some of the reference streams) but that increasing trends in uranium concentrations at the CLT1 Upper Main Stem, both since the baseline period and over the mine operation period, may also suggest a potential mine-related influence (see Section 3.1.1.3.1). In summary, comparisons of 2024 water chemistry at Camp Lake with Reference Lake 3 and baseline conditions revealed slightly to highly elevated concentrations of several parameters at Camp Lake. However, there were no parameters that were consistently elevated compared to both reference and baseline conditions in any individual season, except for total uranium. Nevertheless, since mean concentrations of all parameters have remained well below AEMP benchmarks and WQGs since the start of commercial mine operations in 2015, including in 2024, no adverse effects on biota are expected

Table 3.5: Mean Water Chemistry at Camp Lake (JL0) and Reference Lake 3 (REF-03) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

	Parameters	Units	Water Quality Guideline (WQG)[b,c]	AEMP Benchmark[d]	Reference Lake 3 (n = 3)		Camp Lake Stations (n = 5)		
					Summer	Fall	Winter	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	72.5	72.0	174	150	153
	pH (lab)	pH	6.5 - 9.0	-	7.51	7.50	7.63	7.92	7.92
	Hardness (as $CaCO_3$)	mg/L	-	-	34.8	35.3	68.6	71.3	74
	Total Suspended Solids (TSS)	mg/L	-	-	<1	3.30	<1	<1	<1
	Total Dissolved Solids (TDS)	mg/L	-	-	51.5	41.2	87.5	88.3	77.0
	Turbidity	NTU	-	-	0.323	0.267	0.121	0.406	0.291
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31.4	36.1	64.2	64.7	67.1
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00738	0.00837	0.00648	0.0126	0.00701
	Nitrate	mg/L	3	3	<0.02	<0.02	0.264	<0.02	<0.02
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.191	0.145	0.120	0.111	0.114
	Dissolved Organic Carbon	mg/L	-	-	3.62	3.44	2.08	2.25	2.10
	Total Organic Carbon	mg/L	-	-	3.01	3.51	1.88	2.19	2.60
	Total Phosphorus	mg/L	0.020^{α}	-	0.00467	0.00262	0.00347	0.00300	0.00326
	Phenols	mg/L	0.004^{α}	-	<0.001	0.00152	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	1.21	1.21	6.31	4.21	4.34
	Sulphate (SO_4)	mg/L	218^{β}	218	2.72	2.63	8.60	4.10	4.13
Total Metals	Aluminum (Al)	mg/L	0.100	0.1	0.0158	0.00605	0.00391	0.0121	0.00992
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.000117	<0.0001	0.000101	<0.0001	<0.0001
	Barium (Ba)	mg/L	1^{β}	-	0.00614	0.00598	0.00897	0.00783	0.00773
	Beryllium (Be)	mg/L	0.011^{α}	-	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.01	<0.01
	Cadmium (Cd)	mg/L	0.00012	0.0001	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	6.49	6.40	15.2	13.7	13.9
	Chromium (Cr)	mg/L	0.0089	0.003	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.004	0.000848	0.000823	0.00114	0.000991	0.00107
	Iron (Fe)	mg/L	0.30	0.300	0.0337	0.0112	<0.01	0.0154	0.0121
	Lead (Pb)	mg/L	0.001	0.001	0.0000528	<0.00005	<0.00005	<0.00005	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	0.00101	0.00105	0.00103
	Magnesium (Mg)	mg/L	-	-	4.26	4.48	10.3	9.11	9.39
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00136	0.000602	0.000581	0.00139	0.00133
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000139	0.000144	0.000470	0.000449	0.000456
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000664	0.000653	0.000653
	Potassium (K)	mg/L	-	-	0.888	0.831	1.55	1.46	1.44
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Silicon (Si)	mg/L	-	-	0.487	0.425	0.581	0.493	0.482
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.875	0.843	2.60	2.13	2.27
	Strontium (Sr)	mg/L	-	-	0.00783	0.00754	0.0127	0.0115	0.0119
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	0.000101	<0.0001	0.000125
	Titanium (Ti)	mg/L	-	-	0.000947	0.000308	<0.0003	0.000527	0.000343
	Uranium (U)	mg/L	0.015	-	0.000273	0.000260	0.00153	0.00140	0.00146
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02^{α}	0.030	<0.003	<0.003	<0.003	<0.003	<0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the applicable AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

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

[b] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[c] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

at Camp Lake and mine-related influences on water quality in the lake are limited to potentially increasing concentrations of uranium.

3.3.2 Sediment Quality

Surficial sediments (i.e., the top 2 cm) collected at the Camp Lake coring stations in 2024 were primarily composed of silt and sand; mean TOC content was <2% (Figure 3.8; Appendix Tables D.5 to D.8). There were no significant differences in the proportions of sand, silt, or clay in sediments from littoral or profundal areas at Camp Lake and Reference Lake 3 (Appendix Table D.7). Additionally, there was no detectable hydrogen sulphide scent from any of the Camp Lake or Reference Lake 3 cores. However, the TOC content in sediments at the profundal stations at Camp Lake was significantly lower than Reference Lake 3 (Figure 3.8; Appendix Table D.7). Regardless, sediment core observations suggested that, overall, sediment characteristics in Camp Lake and Reference Lake 3 were comparable at the time of sampling in 2024.

Sediment samples collected from littoral areas using a Petite Ponar, specifically collected to support the interpretation of BIC data, had higher proportions of sand relative to the surface sediment samples collected by coring; otherwise, particle size and TOC content were comparable between the two sample types (Appendix Table D.4). Additionally, surficial sediments collected using both sampling methods were generally described as reddish brown to brown or gray silt (Appendix Tables D.5 and D.6). Similar colours and physical features were observed in sediment cores from Reference Lake 3 (Appendix Table D.1), except that Reference Lake 3 contained more gray clay units (Appendix Table D.1).

Metal concentrations in sediment were generally similar between the inlet and outlet stations in Camp Lake. Exceptions were arsenic, barium, cobalt, iron, molybdenum, and phosphorus concentrations, which were slightly higher at the inlet station (JL0-02) than at the stations near the lake outlet (JL0-11 and JL0-12), based on qualitative comparisons (Appendix Table D.8). Overall, metal concentrations in littoral and profundal sediments at Camp Lake were comparable to those at Reference Lake 3 in 2024 (Table 3.6; Appendix Table D.9).

Mean metal concentrations in sediments from littoral and profundal stations in Camp Lake were below AEMP benchmarks and SQG, except for mean concentrations of iron (littoral sediments only) and manganese (profundal sediments only), which were above applicable SQG but below AEMP benchmarks (Table 3.6; Figure 3.9; Appendix Table D.8). Concentrations of arsenic, copper, iron, manganese, nickel, and phosphorus in sediments from profundal stations were above their respective Camp Lake AEMP benchmarks and/or SQG at low frequencies (Appendix Table D.8). Specifically, one profundal sample for each of arsenic, iron, manganese, nickel, and phosphorus, and two profundal samples for copper exceeded their

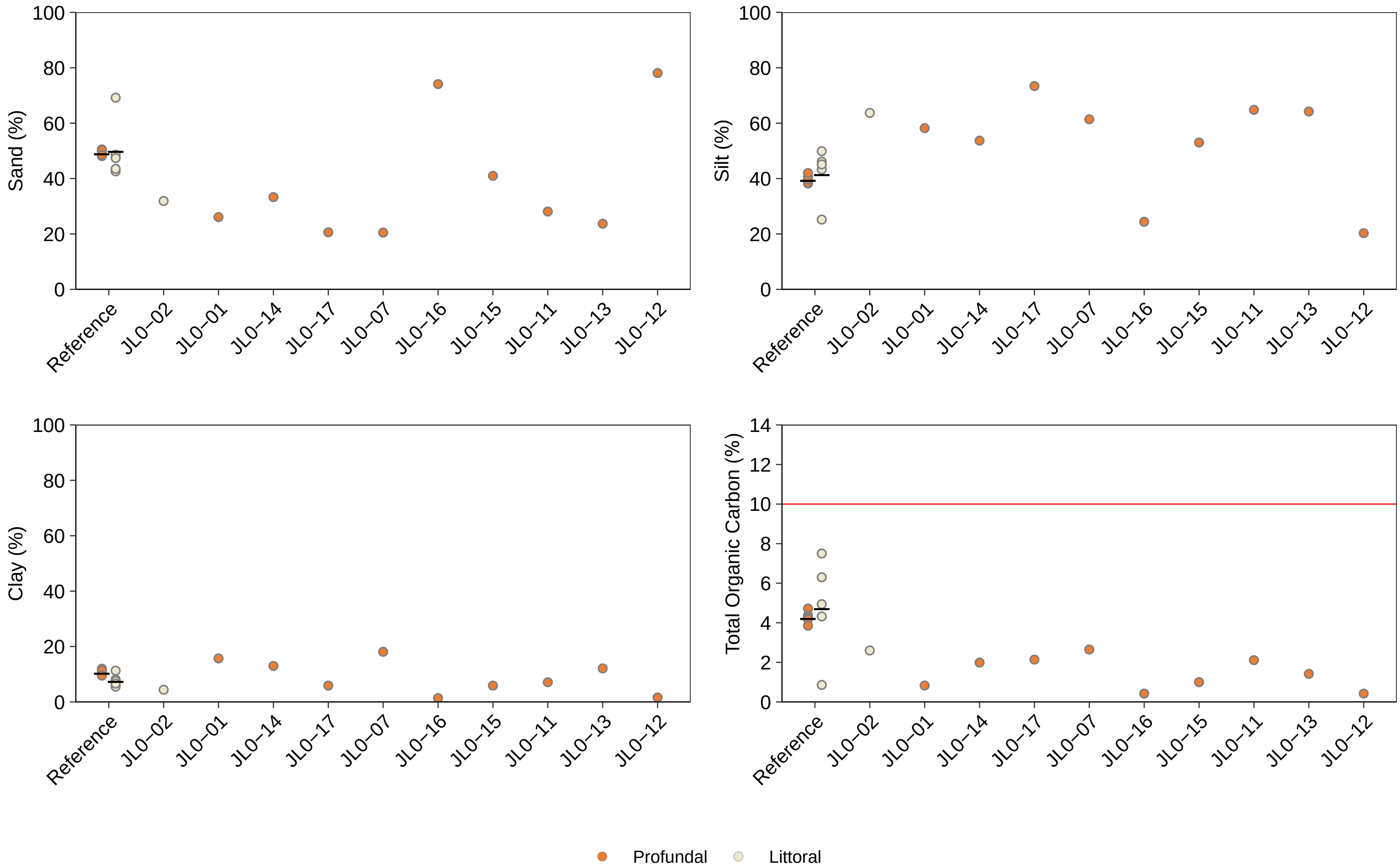

Figure 3.8: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Core Samples taken from Camp Lake (JL0) Sediment Monitoring Stations and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Black bars indicate mean of reference samples. Red line indicates AEMP Benchmark.

Table 3.6: Total Organic Carbon (TOC) Content and Metal Concentrations in Sediments at Monitoring Stations in Camp Lake (JL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

	Parameter	Units	SQG[a]	AEMP Benchmark[b]	Littoral Stations: Reference Lake (n = 5) Average ± SD	Littoral Stations: Camp Lake (n = 1) Average ± SD	Profundal Stations: Reference Lake (n = 5) Average ± SD	Profundal Stations: Camp Lake (n = 9) Average ± SD
	TOC	%	10[α]	-	4.78 ± 2.52	2.60 ± -	4.28 ± 0.315	1.44 ± 0.818
Metals	Aluminum (Al)	mg/kg	-	-	16,560 ± 3,306	17,000 ± -	23,060 ± 1,363	15,468 ± 4,690
	Antimony (Sb)	mg/kg	-	-	<0.1 ± -	<0.1 ± -	<0.1 ± -	<0.1 ± -
	Arsenic (As)	mg/kg	17	5.9	5.02 ± 1.55	4.83 ± -	5.07 ± 0.449	4.30 ± 2.15
	Barium (Ba)	mg/kg	-	-	115 ± 34.7	84.8 ± -	142 ± 20.5	80.1 ± 46.1
	Beryllium (Be)	mg/kg	-	-	0.646 ± 0.147	0.730 ± -	0.884 ± 0.0586	0.813 ± 0.241
	Bismuth (Bi)	mg/kg	-	-	<0.2 ± -	0.270 ± -	<0.2 ± -	0.267 ± 0.0588
	Boron (B)	mg/kg	-	-	13.3 ± 2.05	18.0 ± -	16.7 ± 0.879	22.5 ± 6.11
	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.0497	0.155 ± -	0.166 ± 0.0166	0.152 ± 0.0652
	Calcium (Ca)	mg/kg	-	-	4,716 ± 728	4,080 ± -	5,426 ± 237	4,550 ± 2,068
	Chromium (Cr)	mg/kg	90	98	55.1 ± 12.3	65.7 ± -	76.0 ± 4.65	65.5 ± 14.9
	Cobalt (Co)	mg/kg	-	-	11.5 ± 2.84	16.4 ± -	17.4 ± 1.70	14.8 ± 4.57
	Copper (Cu)	mg/kg	197	50	***67.5*** ± 21.3	39.8 ± -	***95.1*** ± 8.03	39.7 ± 15.0
	Iron (Fe)	mg/kg	40,000[α]	52,400	***58,760*** ± 25,999	49,200 ± -	49,820 ± 3,295	34,356 ± 10,365
	Lead (Pb)	mg/kg	91	35	13.7 ± 1.78	16.4 ± -	18.5 ± 1.01	17.8 ± 6.43
	Lithium (Li)	mg/kg	-	-	25.6 ± 5.12	25.8 ± -	36.2 ± 2.68	27.5 ± 7.37
	Magnesium (Mg)	mg/kg	-	-	11,308 ± 2,124	14,400 ± -	15,780 ± 841	12,966 ± 2,323
	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	862 ± 611	851 ± -	2,246 ± 2,318	1,707 ± 1,686
	Mercury (Hg)	mg/kg	0.486	0.17	0.0470 ± 0.0233	0.0344 ± -	0.0702 ± 0.0129	0.0353 ± 0.0214
	Molybdenum (Mo)	mg/kg	-	-	4.63 ± 1.94	1.80 ± -	2.83 ± 0.501	1.30 ± 0.918
	Nickel (Ni)	mg/kg	75[α,β]	72	39.2 ± 8.63	61.6 ± -	52.2 ± 3.77	59.2 ± 13.1
	Phosphorus (P)	mg/kg	2,000[α]	1,580	1,344 ± 713	964 ± -	999 ± 72	935 ± 316
	Potassium (K)	mg/kg	-	-	4,118 ± 630	4,150 ± -	5,600 ± 317	4,134 ± 1,303
	Selenium (Se)	mg/kg	-	-	0.740 ± 0.278	0.300 ± -	0.826 ± 0.133	0.336 ± 0.124
	Silver (Ag)	mg/kg	-	-	0.146 ± 0.0462	<0.1 ± -	0.238 ± 0.0192	0.123 ± 0.0306
	Sodium (Na)	mg/kg	-	-	311 ± 48.8	174 ± -	431 ± 20.9	200 ± 85.5
	Strontium (Sr)	mg/kg	-	-	11.1 ± 1.22	7.64 ± -	13.3 ± 0.458	11.3 ± 3.56
	Sulphur (S)	mg/kg	-	-	1,620 ± 403	<1,000 ± -	1,360 ± 114	1,033 ± -
	Thallium (Tl)	mg/kg	-	-	0.423 ± 0.145	0.396 ± -	0.748 ± 0.0562	0.403 ± 0.147
	Tin (Sn)	mg/kg	-	-	<2 ± -	<2 ± -	<2 ± -	<2 ± -
	Titanium (Ti)	mg/kg	-	-	958 ± 159	952 ± -	1,164 ± 37.1	814 ± 171
	Uranium (U)	mg/kg	-	-	15.3 ± 5.91	5.38 ± -	25.1 ± 2.33	5.15 ± 2.27
	Vanadium (V)	mg/kg	-	-	51.2 ± 9.67	53.4 ± -	67.7 ± 3.92	51.4 ± 14.2
	Zinc (Zn)	mg/kg	315	135	72.1 ± 14.9	55.9 ± -	95.2 ± 6.65	51.7 ± 16.5
	Zirconium (Zr)	mg/kg	-	-	4.26 ± 1.70	6.00 ± -	3.92 ± 0.455	5.37 ± 2.12

Indicates parameter concentration above SQG.

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation. "-" = data not available.

[a] Canadian SQG for the protection of aquatic life probable effect level (PEL; CCME 2024) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BCMOE 2024).

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

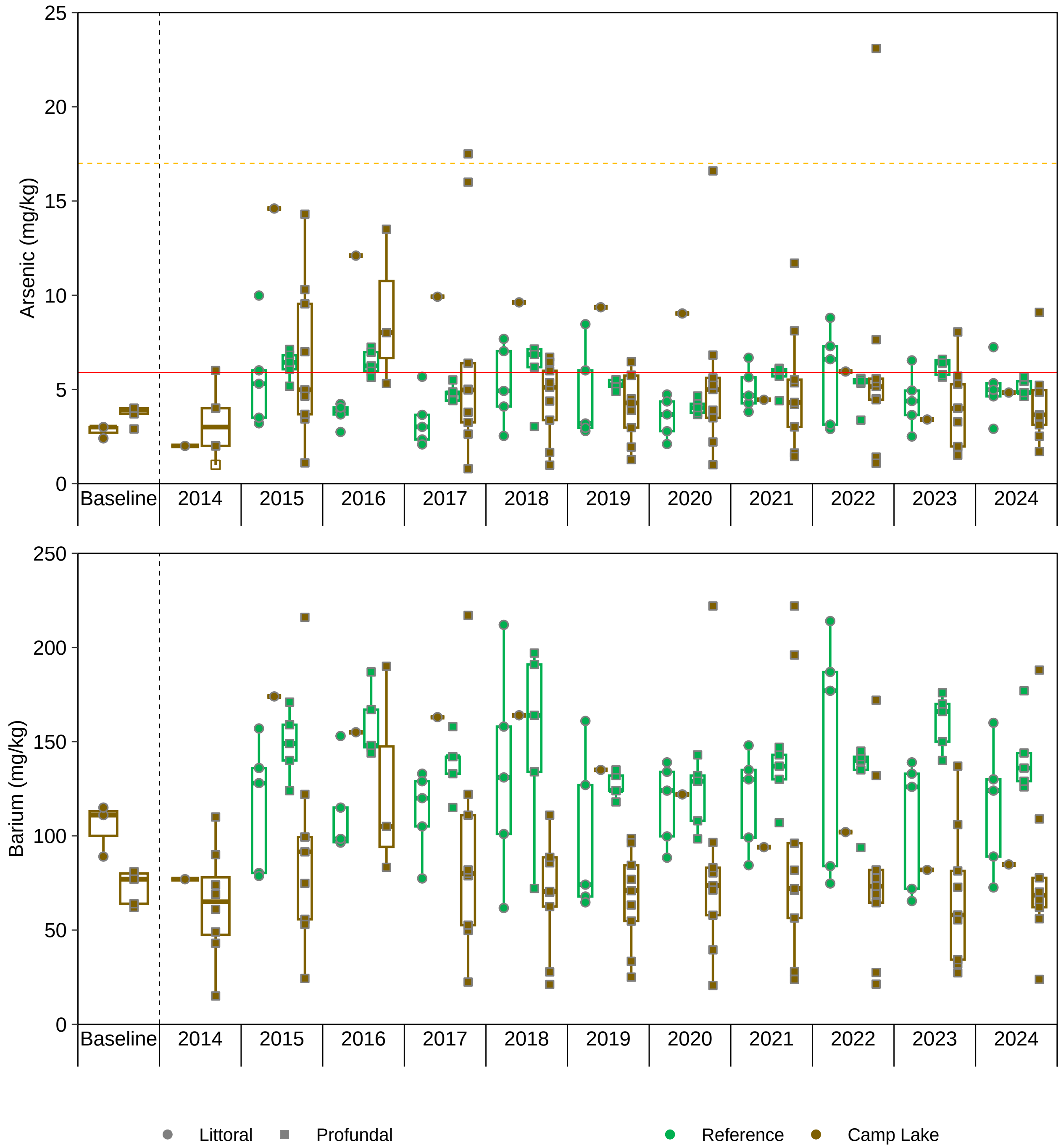

Figure 3.9: Temporal Comparison of Metal Concentrations in Sediments at Littoral and Profundal Stations of Camp Lake (JL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

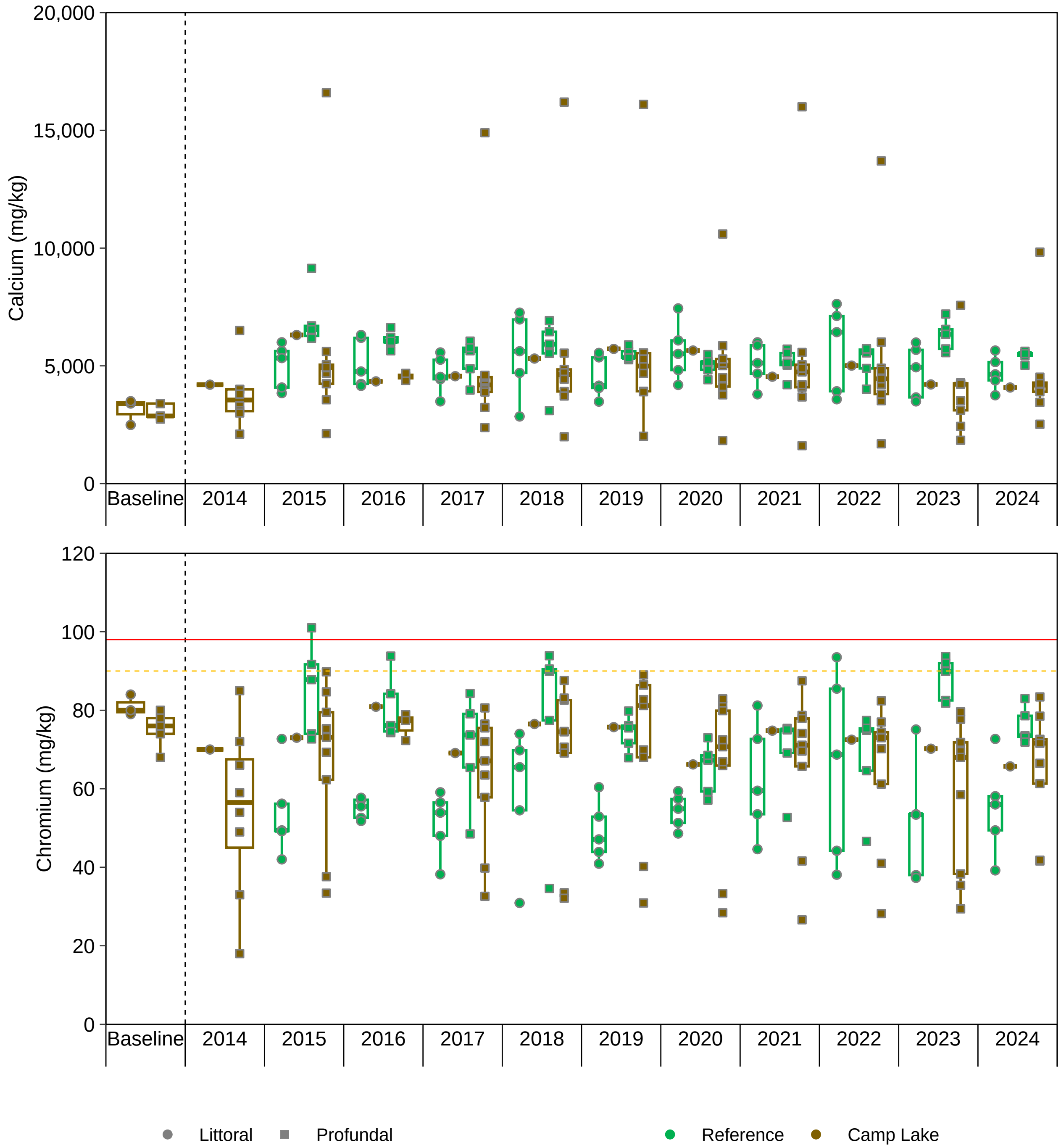

Figure 3.9: Temporal Comparison of Metal Concentrations in Sediments at Littoral and Profundal Stations of Camp Lake (JL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

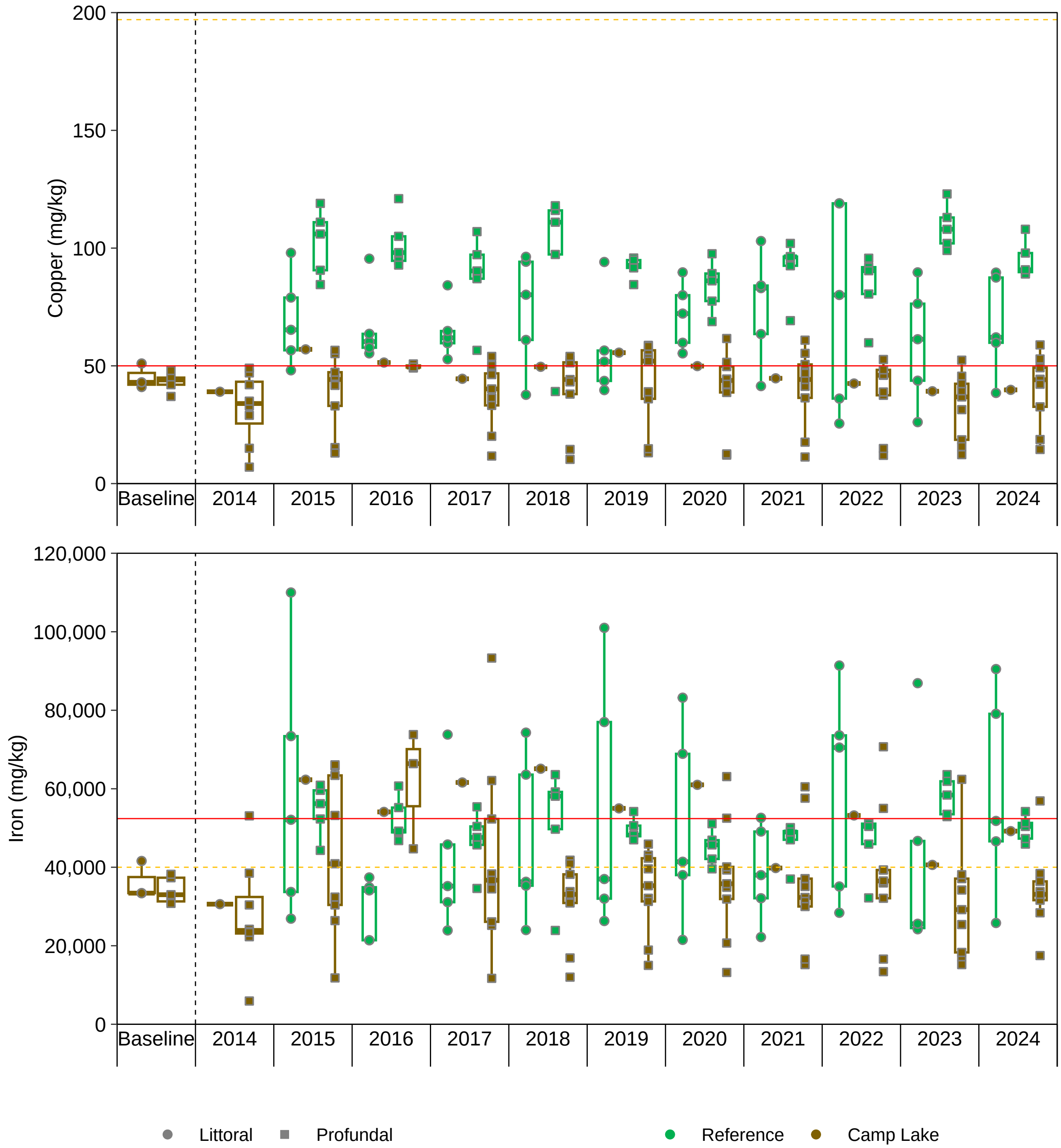

Figure 3.9: Temporal Comparison of Metal Concentrations in Sediments at Littoral and Profundal Stations of Camp Lake (JL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

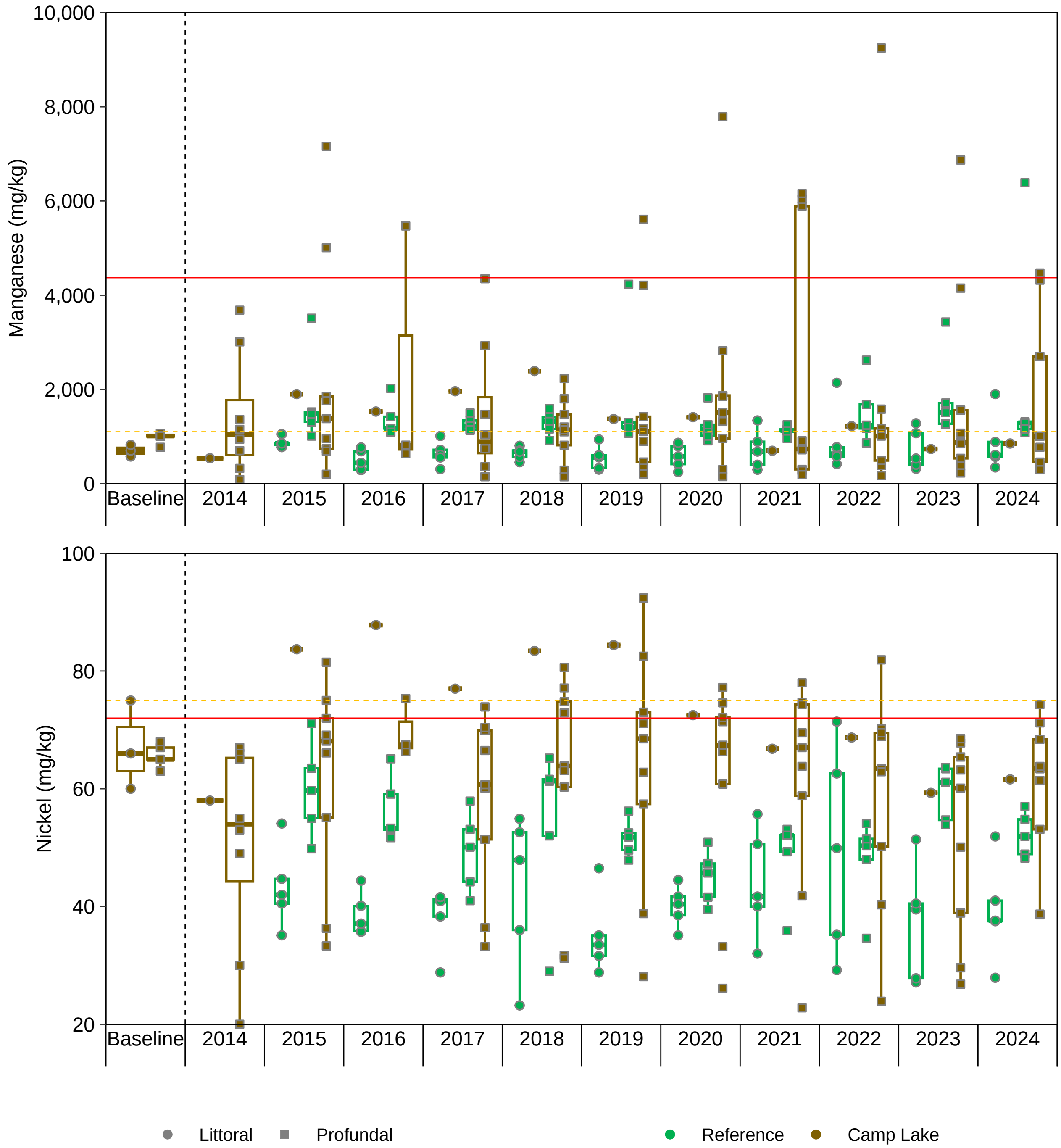

Figure 3.9: Temporal Comparison of Metal Concentrations in Sediments at Littoral and Profundal Stations of Camp Lake (JL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

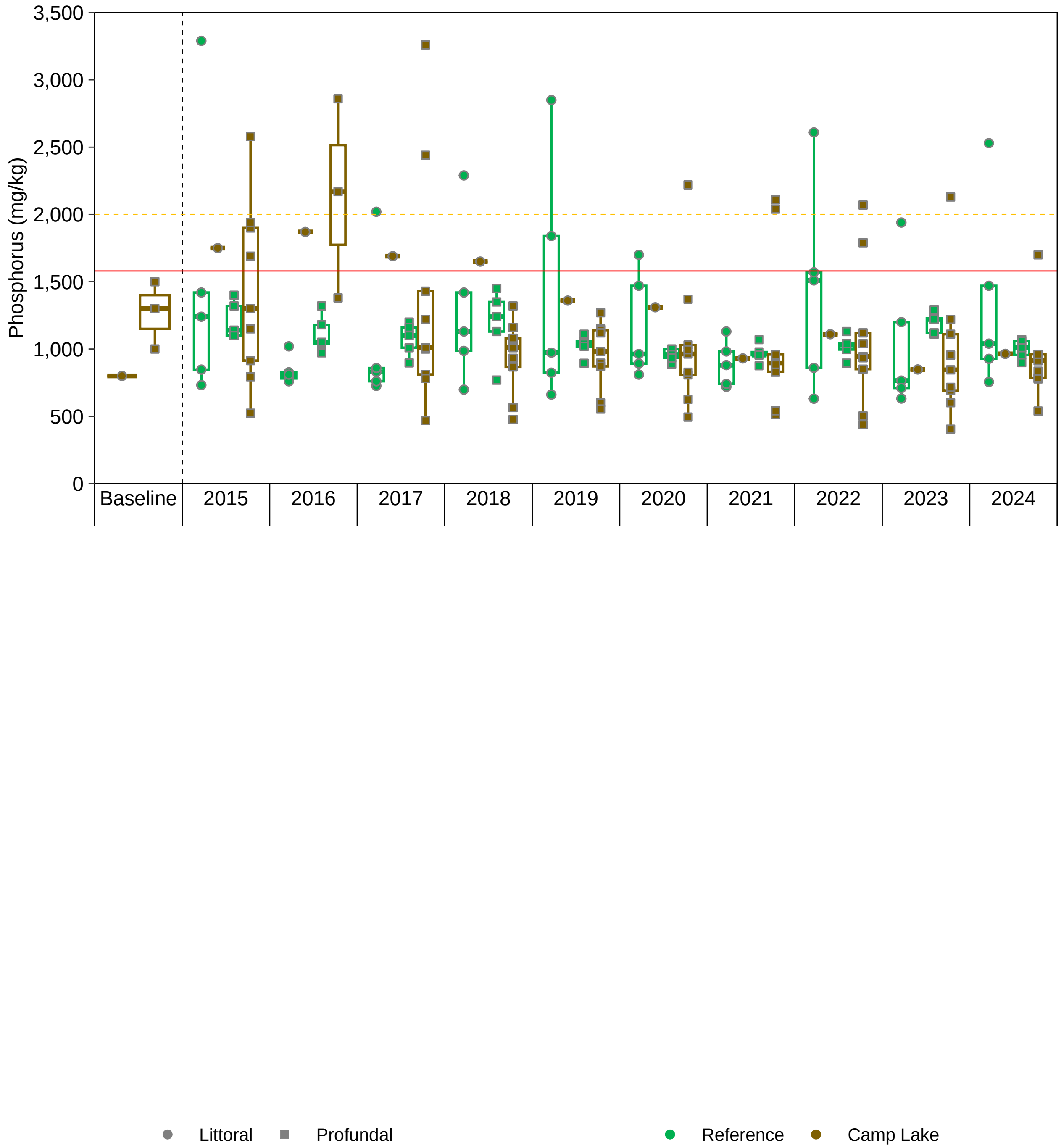

Figure 3.9: Temporal Comparison of Metal Concentrations in Sediments at Littoral and Profundal Stations of Camp Lake (JL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

respective AEMP benchmarks in 2024 (Appendix Table D.13). One profundal and one littoral sample as well as three profundal samples also exceeded the SQGs for iron and manganese, respectively (Appendix Table D.13). Iron and manganese concentrations in sediments from Camp Lake that were higher than AEMP benchmarks and/or SQGs are at least partially attributable to regional geological enrichment, which makes this area attractive for iron mining, and is supported by similar concentrations in sediments at the reference lake (Figure 3.9; Table 3.6; Appendix Tables D.2 and D.9). At Reference Lake 3, mean concentrations of iron and manganese were above AEMP benchmarks and SQG in 2024 (Figure 3.9; Appendix Table D.2). Chromium and phosphorus concentrations in sediments were rarely above AEMP benchmarks for Reference Lake 3 and/or SQG (i.e., in one to two samples; Figure 3.9; Appendix Table D.2).

Mean metal concentrations in sediment samples collected from littoral and profundal stations in Camp Lake in 2024 were comparable to those measured at the reference lake in 2024 and in Camp Lake during the baseline period (2005 to 2013), except for boron, which had concentrations that were highly elevated (i.e., concentration was 10 times higher than mean baseline concentration) compared to baseline at both littoral and profundal stations (18.0--and 11.3-times greater, respectively; Appendix Figure D.1, Appendix Table D.9)[22]. Additionally, metal concentrations in sediments from littoral and profundal stations in Camp Lake in 2024 were typically within the ranges previously observed during mine operations from 2015 to 2022 (Figure 3.9). There was no evidence to suggest concentrations of metals in sediments from Camp Lake increased over time during the mine operation period (2015 to 2024; Figure 3.9). Overall, sediment chemistry in Camp Lake is likely controlled by natural variation and geological sources and no substantial mine-related changes in sediment chemistry have been observed at Camp Lake since the commencement of mine operations in 2015.

3.3.3 Phytoplankton

In 2024, mean chlorophyll-a concentrations at Camp Lake did not show clear spatial gradients between the inlet (Station JL0-02, near the inflow from CLT1) and the lake outlet stations (Station JL0-09; Figures 3.10 and 2.2, Appendix Table E.3). Chlorophyll-a concentrations were significantly lower in winter compared to summer and fall, with no significant differences between the summer and fall values (Figure 3.10, Appendix Table E.4). Additionally, during both the

[22] Boron concentrations in sediments from 2015 to 2024 were considerably higher (i.e., 10- to 70-times) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences (i.e., probable under-recovery of boron in baseline and 2014). The analytical laboratory used for the baseline study differed from the current laboratory.

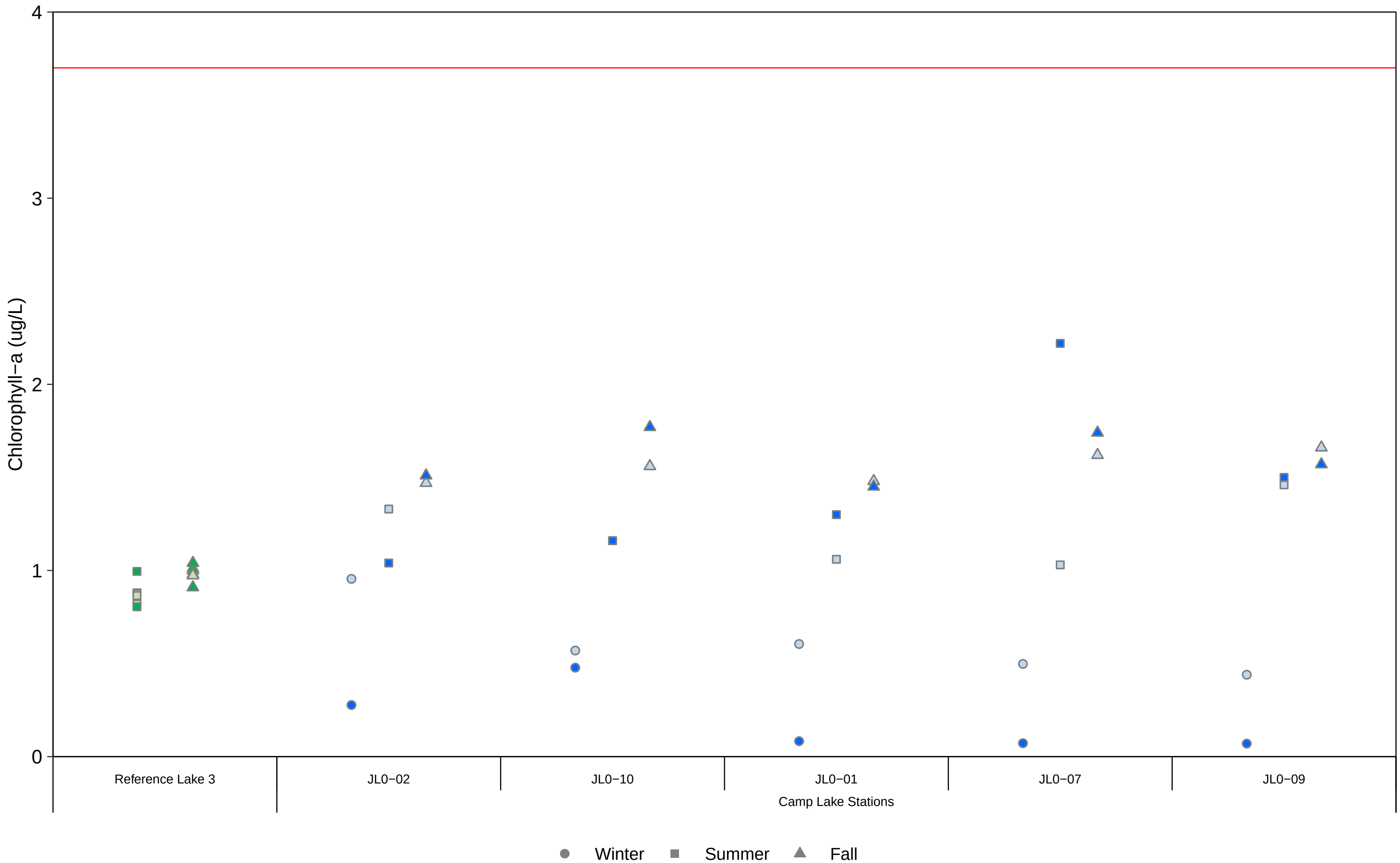

Figure 3.10: Chlorophyll−a Concentrations at Camp Lake (JL0) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2024

Notes: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Red line indicates AEMP Benchmark. Lighter shade of colour indicates surface sample, darker shade indicates bottom sample. Reference areas are shown in green and mine−exposed areas are shown in blue. Camp Lake Stations are presented in order of proximity to the lake inlet (left to right).

summer and fall sampling events, mean chlorophyll-a concentrations at Camp Lake were significantly higher than those at Reference Lake 3 (Appendix Tables E.5, E.6, and E.7). Despite this, chlorophyll-a concentrations at Camp Lake remained below the AEMP benchmark of 3.7 μg/L throughout all winter, summer, and fall sampling events in 2024 (Figure 3.10). The average chlorophyll-a concentrations at Camp Lake suggest relatively low phytoplankton abundance and an oligotrophic status, based on Wetzel's (2001) lake trophic classifications (i.e., chlorophyll-a < 4.5 μg/L; Appendix Table E.3). This classification was further supported by an ultraoligotrophic to oligotrophic designation under the CCME Phosphorus Guidance Framework for the Management of Freshwater Systems (CCME 2024b), as mean aqueous total phosphorus concentrations at Camp Lake were consistently below 10 μg/L during all seasonal sampling events (Table 3.5; Appendix Table C.26).

Although temporal comparisons of Camp Lake's chlorophyll-a concentrations differed significantly among years of mine construction and operation, both seasonally and annually, the data showed considerable temporal and seasonal variability, resulting in no consistent temporal patterns (Figure 3.11, Appendix Table E.8). Chlorophyll-a concentrations have also differed significantly in summer and fall among years at Reference Lake 3 (Appendix Table E.9). Chlorophyll-a concentrations in Camp Lake have consistently been slightly higher than those at Reference Lake 3 during at least one season each year since mining operations began (Figure 3.11; Minnow 2016a, 2017, 2018, 2019, 2020, 2021b, 2022, 2023, 2024a). However, the differences between the two lakes have been and remain minimal (i.e., less than 1 μg/L), with both lakes falling within the same trophic classification, suggesting no ecologically relevant mine-related influences on Camp Lake. The relatively small magnitude and consistency of the differences in chlorophyll-a concentrations between Camp Lake and Reference Lake 3 also suggest that they are due to natural factors, such as lake morphology and location (e.g., lake size and fetch, which affect lake mixing potential and the amount of sunlight received) rather than mine-related influences. In addition, the absence of consistent directional changes in mean chlorophyll-a concentrations across seasons and years at Camp Lake aligns with stable nutrient concentrations (i.e., nitrate) and water quality generally meeting WQGs over the ten years since mine operations began (Section 3.3.1). Baseline chlorophyll-a data (2005 to 2013) were not collected for Camp Lake, precluding comparisons to conditions prior to the construction of the mine.

Overall, chlorophyll-a concentrations in Camp Lake exhibited no consistent directional temporal patterns in any season, have generally remained consistent relative to concentrations observed at Reference Lake 3 since 2015, and remained well below the AEMP benchmark in 2024. These results indicate no adverse mine-related effects on phytoplankton productivity at Camp Lake in 2024.

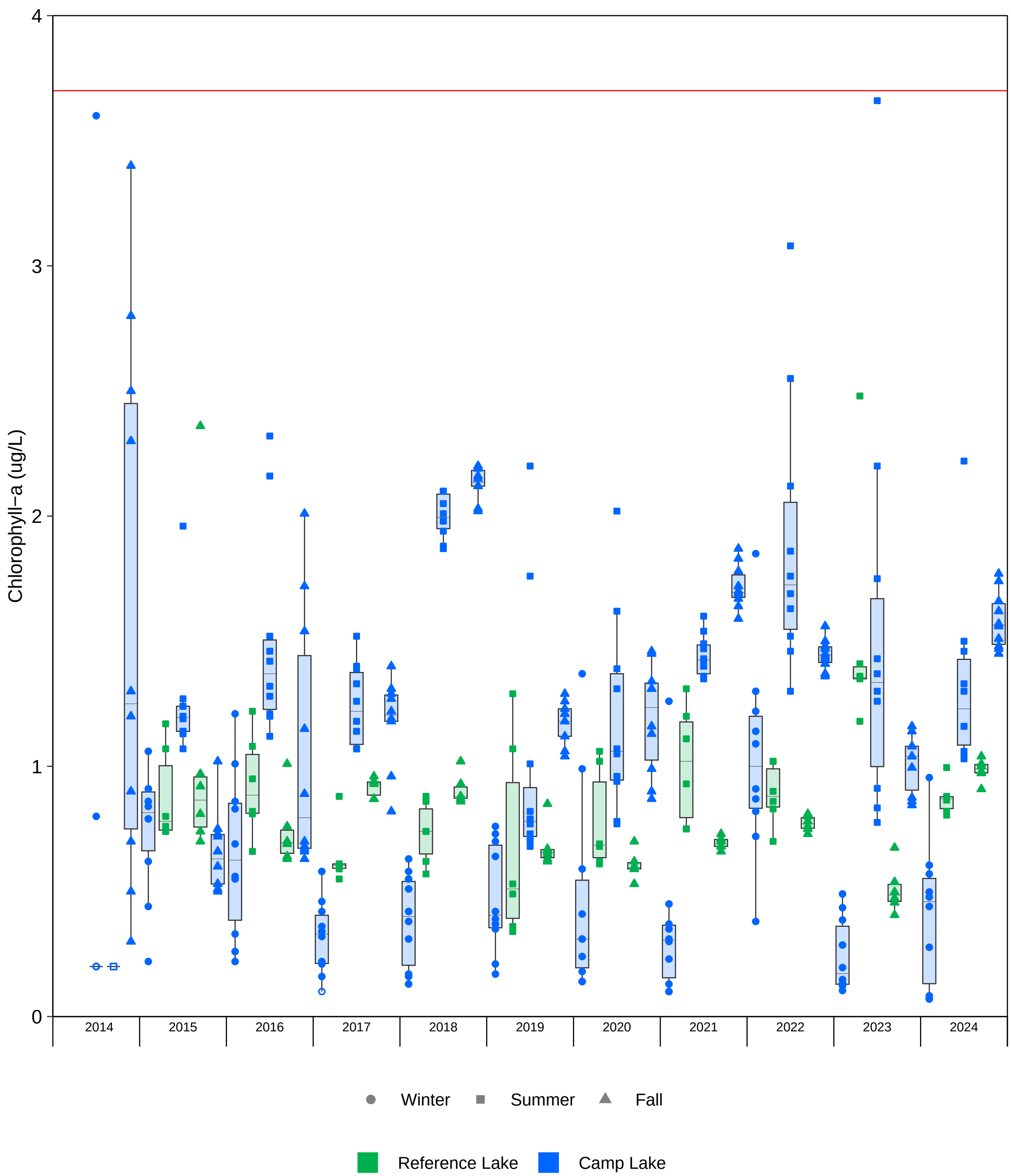

Figure 3.11: Temporal Comparison of Chlorophyll−a Concentrations Among Seasons between Camp Lake (JL0) and Reference Lake 3 (REF-03) for Construction (2014) and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

3.3.4 Benthic Invertebrate Community

In 2024, most BIC endpoints for littoral (shallow) habitats of Camp Lake were statistically similar to Reference Lake 3, except for Simpson's Evenness and relative proportions of Ostracoda and Chironomidae (Table 3.7). Simpson's Evenness was higher in Camp Lake relative to Reference Lake 3 (MCT = 0.911 and 0.759, respectively) and the difference between lakes was ecologically meaningful (i.e., the MOD was outside of the CES_{BIC}; Table 3.7). Camp Lake had significantly lower relative proportions of *Ostracoda* (MCT = 2.96%) and significantly higher relative proportions of *Chironomidae* (MCT = 91.9%) relative to Reference Lake 3 (MCT = 39.5% and 52.6%, respectively; Table 3.7). These differences were also considered ecologically meaningful (Table 3.7). Further, the structural differences in the BIC of littoral habitats were reflected in the comparisons of Bray-Curtis Indices, which differed significantly between Camp Lake and Reference Lake 3 (Appendix Table F.19).

In 2024, most BIC endpoints for profundal (deep) habitats of Camp Lake were statistically similar to Reference Lake 3, except for density and relative proportions of *Ostracoda, Chironomidae*, and burrowers (Table 3.8). Benthic invertebrate density in profundal habitats of Camp Lake (MCT = 834 individuals/m^2) was significantly higher relative to similar habitats at Reference Lake 3 (MCT = 202 individuals/m^2) and this difference was ecologically meaningful (i.e., the MOD was outside the CES_{BIC} of ± 2 SD_{REF}; Table 3.8). Profundal habitats in Camp Lake had significantly lower relative proportions of *Ostracoda* (MCT = 3.07%) relative to Reference Lake 3 (MCT = 8.37%), which were counterbalanced by higher relative proportions of *Chironomidae* (MCT = 95.3%) relative to Reference Lake 3 (MCT = 85.2%; Table 3.8). Overall, the lower relative proportions of Ostracoda and higher proportions of burrowing taxa in the profundal samples from Camp Lake were ecologically meaningful, based on MOD outside of the CES_{BIC} (Table 3.8). Similar to littoral habitats, these differences in BIC structure between lakes were reflected in the comparisons of Bray-Curtis Indices, which differed significantly between Camp Lake and Reference Lake 3 (Appendix Table F.19).

Chironomidae are commonly used as bioindicators of ecological conditions, with a high relative proportion of this group often signaling poor water and/or sediment quality due to their general tolerance to metals (Barbour et al. 1999 and Merritt et al. 2008). However, water and sediment quality in Camp Lake in 2024 generally met AEMP benchmarks and relevant WQGs (developed to be protective of aquatic life) suggesting that the higher proportions of *Chironomidae* in the lake, relative to reference, is not reflective of degraded environmental conditions. Further, since the start of mine operations in 2015, proportions of *Ostracoda* and *Chironomidae* in littoral and profundal habitats of Camp Lake have not changed significantly relative to baseline (Appendix Table F.20 and F.21) and qualitative assessment of temporal patterns indicates that

Table 3.7: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Littoral Habitats in Camp Lake (JL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake (Littoral Habitat)	MCT (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	none	NO	0.118	1.3	Reference Lake 3	1,049	901	403	215	982	2,514
						Camp Lake	2,242	1,226	548	543	1,963	3,496
Richness (Number of Taxa)	tequal	none	NO	0.311	0.84	Reference Lake 3	8.80	3.56	1.59	5.00	8.00	13.0
						Camp Lake	11.8	5.07	2.27	3.00	14.0	15.0
Simpson's Evenness (E)	tequal	log10	YES	0.003	2.2	Reference Lake 3	0.759	0.0621	0.0278	0.669	0.755	0.840
						Camp Lake	0.911	0.0466	0.0208	0.862	0.900	0.988
Shannon's Diversity	M-W	rank	NO	0.151	2.8	Reference Lake 3	2.01	0.243	0.109	1.72	2.01	2.28
						Camp Lake	2.68	0.637	0.285	1.57	2.87	3.14
Hydracarina (%)	tequal	log10(x+1)	NO	0.404	-0.45	Reference Lake 3	2.49	2.88	1.29	0	2.33	7.02
						Camp Lake	1.18	1.62	0.726	0	0.943	3.95
Ostracoda (%)	tequal	log10(x+1)	YES	<0.001	-2.5	Reference Lake 3	39.5	16.1	7.20	16.0	40.3	60.5
						Camp Lake	2.96	5.91	2.64	0	0	13.5
Chironomidae (%)	tequal	none	YES	<0.001	2.7	Reference Lake 3	52.6	14.4	6.46	30.7	56.6	68.0
						Camp Lake	91.9	6.27	2.81	82.4	92.1	100
Metal Sensitive Chironomidae (%)	tequal	none	NO	0.325	-0.55	Reference Lake 3	22.1	17.5	7.81	0	17.3	41.3
						Camp Lake	12.4	10.8	4.81	0	8.17	27.8
Collector Gatherers (%)	tequal	log10	NO	0.982	0.014	Reference Lake 3	74.9	18.1	8.08	56.4	77.0	100
						Camp Lake	74.8	16.3	7.28	55.1	71.0	100
Filterers (%)	tequal	log10(x+1)	NO	0.291	-0.59	Reference Lake 3	21.7	17.6	7.85	0	17.3	41.3
						Camp Lake	11.1	9.78	4.37	0	8.17	26.0
Shredders (%)	tequal	log10(x+1)	YES	0.082	1.6	Reference Lake 3	0.344	0.578	0.259	0	0	1.33
						Camp Lake	1.29	0.898	0.402	0	1.37	2.48
Clingers (%)	tequal	log10(x+1)	YES	0.075	-0.98	Reference Lake 3	24.0	17.9	8.01	0	19.0	43.6
						Camp Lake	6.53	6.09	2.72	0	5.89	13.4
Sprawlers (%)	tequal	none	NO	0.221	-0.76	Reference Lake 3	66.2	16.9	7.56	43.2	74.5	84.0
						Camp Lake	53.4	13.5	6.03	33.3	53.2	65.9
Burrowers (%)	tequal	log10	YES	0.005	1.9	Reference Lake 3	9.82	6.40	2.86	2.30	8.16	16.9
						Camp Lake	40.1	16.4	7.35	22.3	37.7	66.7

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

Table 3.8: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Profundal Habitats in Camp Lake (JL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake (Profundal Habitat)	MCT (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tunequal	none	YES	0.002	19	Reference Lake 3	202	33.7	15.1	146	207	233
						Camp Lake	834	202	90.1	611	896	1,033
Richness (Number of Taxa)	tequal	none	NO	0.104	3.2	Reference Lake 3	4.40	1.14	0.510	3.00	4.00	6.00
						Camp Lake	8.00	4.24	1.90	5.00	5.00	14.0
Simpson's Evenness (E)	tequal	log10	NO	0.491	-0.69	Reference Lake 3	0.582	0.169	0.0754	0.457	0.508	0.867
						Camp Lake	0.516	0.227	0.102	0.231	0.533	0.806
Shannon's Diversity	tequal	log10	NO	0.825	0.25	Reference Lake 3	1.28	0.318	0.142	0.834	1.27	1.71
						Camp Lake	1.51	0.838	0.375	0.646	1.27	2.67
Hydracarina (%)	tequal	log10(x+1)	NO	0.301	-0.50	Reference Lake 3	4.09	5.94	2.66	0	0	13.0
						Camp Lake	1.09	0.668	0.299	0	1.37	1.72
Ostracoda (%)	tequal	log10(x+1)	YES	0.045	-2.7	Reference Lake 3	8.37	2.08	0.929	5.88	8.33	11.5
						Camp Lake	3.07	4.61	2.06	0	0	10.3
Chironomidae (%)	tequal	none	YES	0.043	1.3	Reference Lake 3	85.2	7.71	3.45	76.9	85.2	94.1
						Camp Lake	95.3	5.30	2.37	86.2	98.1	98.6
Metal Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.920	0.056	Reference Lake 3	9.98	11.3	5.05	0	7.41	29.4
						Camp Lake	10.3	7.46	3.34	2.82	7.34	20.7
Collector Gatherers (%)	M-W	rank	NO	1.000	0.34	Reference Lake 3	85.2	16.2	7.24	57.6	88.2	100
						Camp Lake	88.8	5.02	2.24	81.0	90.4	94.4
Filterers (%)	M-W	rank	NO	0.824	-[a]	Reference Lake 3	6.70	12.8	5.72	0	0	29.4
						Camp Lake	5.69	7.84	3.51	0	3.67	19.0
Shredders (%)	M-W	rank	NO	1.000	-[a]	Reference Lake 3	0.833	1.86	0.833	0	0	4.17
						Camp Lake	0.385	0.860	0.385	0	0	1.92
Clingers (%)	M-W	rank	NO	0.398	-[a]	Reference Lake 3	9.96	18.4	8.23	0	0	42.4
						Camp Lake	6.59	5.32	2.38	1.37	6.91	13.6
Sprawlers (%)	tunequal	log10	NO	0.109	-3.4	Reference Lake 3	86.2	16.7	7.46	57.6	88.9	100
						Camp Lake	51.5	39.1	17.5	17.7	33.6	94.4
Burrowers (%)	tequal	log10(x+1)	YES	0.039	6.4	Reference Lake 3	3.88	4.71	2.11	0	3.70	11.5
						Camp Lake	41.9	34.4	15.4	4.23	56.7	74.3

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

[a] Contrast MODs could not be calculated because the MAD = 0.

the proportion of *Ostracoda* has been consistently lower and the proportion of *Chironomidae* has been consistently higher in both littoral and profundal habitats of Camp Lake than the reference lake (though the difference is more pronounced in littoral habitats; Appendix Figure F.6). Therefore, significant differences in the proportions of these organisms between like-habitats in Camp Lake and Reference Lake 3 in 2024 are not indicative of mine-related effects. Additionally, *Chironomidae* are a preferred prey item for juvenile arctic charr (Eloranta et al. 2010, Wight et al. 2023), suggesting that their high abundance may help support the higher densities of nearshore and littoral arctic charr observed in Camp Lake relative to reference (i.e., through bottom-up trophic cascades; see Section 3.3.5). This pattern may reflect a bottom-up trophic cascade, where increased prey availability supports juvenile growth and/or survival, which in turn may enhance food availability for larger, piscivorous adult charr that prey on smaller or mid-sized conspecifics (Eloranta et al. 2010, Wight et al. 2023). Overall, higher BIC density in profundal habitats of Camp Lake aligned with higher phytoplankton (chlorophyll-a) density during the growing season (i.e., summer and fall; see Section 3.3.3) and arctic charr densities (see Section 3.3.5), indicating that Camp Lake was more biologically productive than Reference Lake 3 in 2024.

Although benthic invertebrate density (2016, 2018, 2023, and 2024) and Simpson's Evenness (2015 only) in littoral habitats of Camp Lake were significantly lower relative to baseline during some mine operation years (2015 to 2024), no consistent ecologically meaningful differences were observed in BIC endpoints based on a MOD outside of the CES_{BIC}. This suggests that BIC endpoints in the littoral habitats of Camp Lake have, in general, remained comparable to the baseline year (i.e., 2013; Appendix Table F.20, Appendix Figure F.5). Additionally, the lower invertebrate densities observed in littoral habitats in 2023 and 2024 relative to baseline are comparable to densities observed in the same habitat type within Camp Lake in 2015 to 2018 (Appendix Table F.20).

Most benthic invertebrate endpoints for profundal habitats in Camp Lake were statistically comparable throughout baseline (2007, 2013) and mine operation years (2015 to 2024; Appendix Table F.21, Appendix Figure F.6). However, there were consistent significant differences at MODs outside of the CES_{BIC} when compared to baseline data from 2007 and 2013 in relative proportions of metal sensitive *Chironomidae* and filterers (consistently lower than baseline) and collector-gatherers (consistently higher than baseline)(Appendix Table F.21). Despite these differences, the relative proportions of these groups have remained relatively stable from 2018 to 2024 (Appendix Table F.21), suggesting that any shifts in community structure likely occurred starting in 2014 or 2015 (i.e., changes in community structure are not recent). The lower relative proportions of metal-sensitive *Chironomidae*, a recognized bioindicator of aquatic ecosystem health, in the BIC starting in 2015 is suggestive of a mine-related influence (Appendix Table F.21).

However, water and sediment quality data from 2024 are not suggestive of potential for adverse mine-related effects to biota in Camp Lake. The decline in metal-sensitive *Chironomidae* circa 2015 may have reduced competition and predation, allowing proportions of collector-gathers within the BIC to increase (Appendix Table F.21). Although the temporal increase in the relative proportion of collector-gatherers could indicate organic enrichment (Merritt et al. 2008), aqueous DOC and TOC concentrations at Camp Lake have remained consistent with baseline conditions (Appendix Table C.27), and TOC content of sediments from profundal habitats of Camp Lake were lower in 2024 compared to reference (Appendix Table D.7). The lower proportions of filterers after circa 2015 is not likely attributed to increased TSS (which can clog feeding structures) or reduced availability of organic matter (i.e., food; Merritt et al. 2008), given that aqueous TSS, DOC, and/or TOC have not shown any change compared to baseline in 2024 (Appendix Table C.27). Although shifts in BIC structure have occurred in profundal areas of Camp Lake since baseline (between 2013 and 2015), they appear to be consistent over time during the mine operation period with no newly identified concerns. Furthermore, supporting water and sediment quality data do not pinpoint specific mine-related influences that would be expected to affect BIC endpoints.

Overall, the 2024 BIC results for Camp Lake did not indicate any consistent differences from reference or baseline conditions. Therefore, consistent with limited changes in water and sediment quality since mine operations began in 2015, the data suggest there were no adverse mine-related effects on the BIC in Camp Lake as of 2024.

3.3.5 Fish Population

3.3.5.1 Fish Community

Arctic charr (*Salvelinus alpinus*) was the only fish species captured at Camp Lake in 2024 (Table 3.9). Ninespine stickleback (*Pungitius pungitius*) have also been captured in nearshore electrofishing surveys in Camp Lake in most CREMP monitoring years, typically at very low densities (CPUE ranged from 0.07 to 0.81 fish per electrofishing minute; Minnow 2017, 2018, 2019, 2020, 2021b, 2023, and 2024a); however, none were captured in 2024, which also occurred in 2015 and 2021 (Minnow 2016a and 2022). Ninespine stickleback have not been consistently captured in Camp Lake in CREMP monitoring programs, suggesting annual natural variability in habitat use and sampling conditions rather than changes in fish community richness over time.

The CPUE for arctic charr in electrofishing and in gill netting surveys was higher at Camp Lake compared to Reference Lake 3 in 2024, suggesting greater fish density at Camp Lake (Table 3.9, Appendix Tables G.1 and G.3). Fish density (based on CPUE) has typically been higher in Camp Lake than Reference Lake 3 since sampling was initiated in Reference Lake 3 in 2015

Table 3.9: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Camp Lake (JL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	105	15	120	2
		CPUE	1.12	0.16	1.28	
	Gill netting	No. Caught	84	-	84	
		CPUE	3.30	-	3.30	
Camp Lake	Electrofishing[b]	No. Caught	105	0	105	1
		CPUE	4.38	0.00	4.30	
	Gill netting	No. Caught	112	-	112	
		CPUE	31.0	-	31.00	

Notes: "-" indicates not applicable as ninespine stickleback are not captured by gill netting.

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

[b] Electrofishing effort for the second pass completed at Camp Lake was not recorded. CPUE is based on the first electrofishing pass only while the total number of fish caught represents cumulative catch from both electrofishing passes. See Appendix Table G.1 for detailed Camp Lake electrofishing records.

(coinciding with the start of the mine operational period; Figure 3.12)[23]. In general, higher fish density in Camp Lake relative to the reference lake since 2015 may be associated with greater primary and secondary productivity, as evidenced by higher chlorophyll-a concentrations (indicating greater phytoplankton density; Section 3.3.3) and higher benthic invertebrate density (Section 3.3.4) compared to Reference Lake 3, resulting in more abundant food sources for arctic charr.

In 2024, the electrofishing CPUE for arctic charr at Camp Lake was within the range observed during baseline studies (2007 to 2013) and during the previous nine years (2015 to 2023) of mine operation (Figure 3.12). The gill netting CPUE at Camp Lake in 2024 was higher than that observed during baseline and mine operational years 2015 to 2021 and 2023, but lower than in 2022 (Figure 3.12). Overall, gill netting CPUE at Camp Lake has been higher over the past five years (2020 to 2024) compared to earlier years of mine operation and construction, as well as to baseline years (Figure 3.12). No consistent temporal patterns in chlorophyll-a concentrations (i.e., phytoplankton density) or ecologically significant differences in BIC endpoints have been observed in Camp Lake that are consistent with an increase in gill net CPUE due to changes in productivity at lower trophic levels over this period (Sections 3.3.3 and 3.3.4). Therefore, the increased gill netting CPUE may be linked to other factors such as sampling influences, weather/climate conditions, or natural variability in fish spatial ecology or movement behaviour.

Small changes in sample timing (e.g., sampling conducted slightly earlier or later in August) and weather/climate conditions, can influence fish movement and access to sampling areas within the lakes. Water temperatures influence both fish movement and metabolic demands, with warmer temperatures typically driving increased fish movement as they seek food to meet their energy needs (Reist et al. 2006). Because gill netting is a 'passive' fish collection method that relies on fish movement into stationary nets, increased fish movement is expected to result in higher catch rates. Weather conditions, particularly wind speed and direction, can influence the locations in the lake that are safely accessible for sampling during the limited August sampling period which may also influence catch rates. Additionally, arctic charr are known to move between offshore and nearshore habitats, as well as across depth gradients, in response to factors such as temperature, prey availability, and life history stage (e.g., feeding or spawning; Klemetsen et al. 2003). Environmental factors that can vary naturally with annual weather and climate conditions, including altered thermal stratification, oxygen availability, or prey distribution, could contribute to greater or poorer aggregation of charr in sampled areas (Helland et al. 2011).

[23] Baseline fish community data (2005 to 2013) were not collected at Reference Lake 3, precluding comparisons of mine-exposed and reference conditions prior to the construction of the mine.

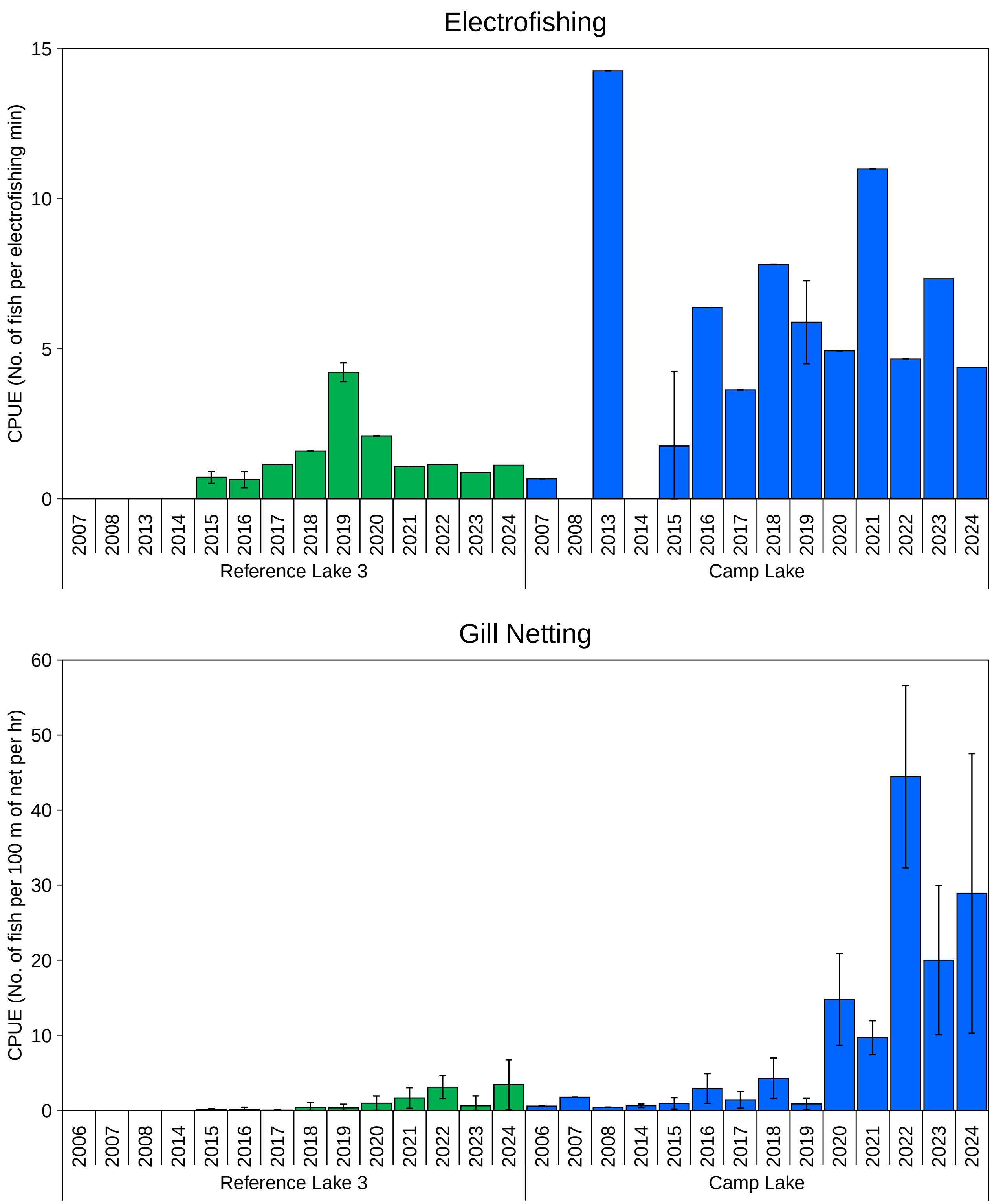

Figure 3.12: Catch−per−unit−effort (CPUE; mean ± standard deviation) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Camp Lake (JL0) and Reference Lake 3 (REF3), Mary River Project CREMP, 2006 to 2024

Notes: Data presented for fish sampling conducted in summer during baseline (2006, 2007, 2008, 2013), construction (2014), and operational (2015 to 2024) mine phases. Reference areas are shown in green and mine−exposed areas are shown in blue.

Gill netting catch rates in Reference Lake 3 have also varied over the sampling period, and like in Camp Lake, CPUE was higher from 2020 to 2024 compared to earlier years. Similar variability in CPUE in Camp Lake and the reference lake is additional evidence that factors other than changes in fish density such as slight differences in sampling locations and timing or natural variability in environmental conditions, may account for the observed variations in gill netting CPUE over the sampling period. Greater temporal similarity in electrofishing CPUE, which is an 'active' fish collection method with more consistent sampling locations between 2024 and previous mine operation years suggests no substantial changes in fish densities at either lake.

Because 2024 electrofishing and gill netting CPUE results fall within the range previously observed during the baseline and mine operations period and were greater than in Reference Lake 3, mine-related changes in fish densities at Camp Lake are not indicated

3.3.5.2 Fish Health Assessment

Nearshore Arctic Charr

In August 2024, a total of 100 and 102 arctic charr were sampled for assessment of fish health from the nearshore habitats of Camp Lake and Reference Lake 3, respectively (Appendix Tables G.4 and G.5)[24,25]. Arctic charr YOY were differentiated from older age classes (non-YOY) based on fork length: 4.5 cm for Camp Lake and 4.0 cm for Reference Lake 3, as determined from length-frequency distributions (LFD), with supporting length and weight measurements and age determinations (Figure 3.13, Appendix Tables G.4 and G.5). Because fewer than ten YOY arctic charr were captured in Camp Lake, statistical comparisons of health endpoints were limited to the non-YOY age classes (Table 3.10) except for LFD comparisons which were also conducted for the distribution of all fish lengths, regardless of age class (Figure 3.13, Table 3.10, Appendix Figure G.1, Appendix Table G.6). Both LFDs for nearshore arctic charr differed significantly between Camp Lake and Reference Lake 3, (Table 3.10, Appendix Table G.6), reflecting a higher proportion of larger individuals in Camp Lake compared to Reference Lake 3 in 2024 (Figure 3.13, Appendix Figure G.1). The LFD for nearshore arctic charr has consistently been different between Camp Lake and Reference Lake 3 over the period of mine operations years (2015 to

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[24] Sample sizes at Camp Lake in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.8.

[25] The total number of fish captured in Camp Lake and Reference Lake 3 by electrofishing (Table 3.9, Appendix Table G.1) was greater than the number of fish sampled for the fish health assessment. The study design requires 100 fish from each lake to be sampled (measured and weighed; Baffinland 2015). Once field crews were certain that the minimum target sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

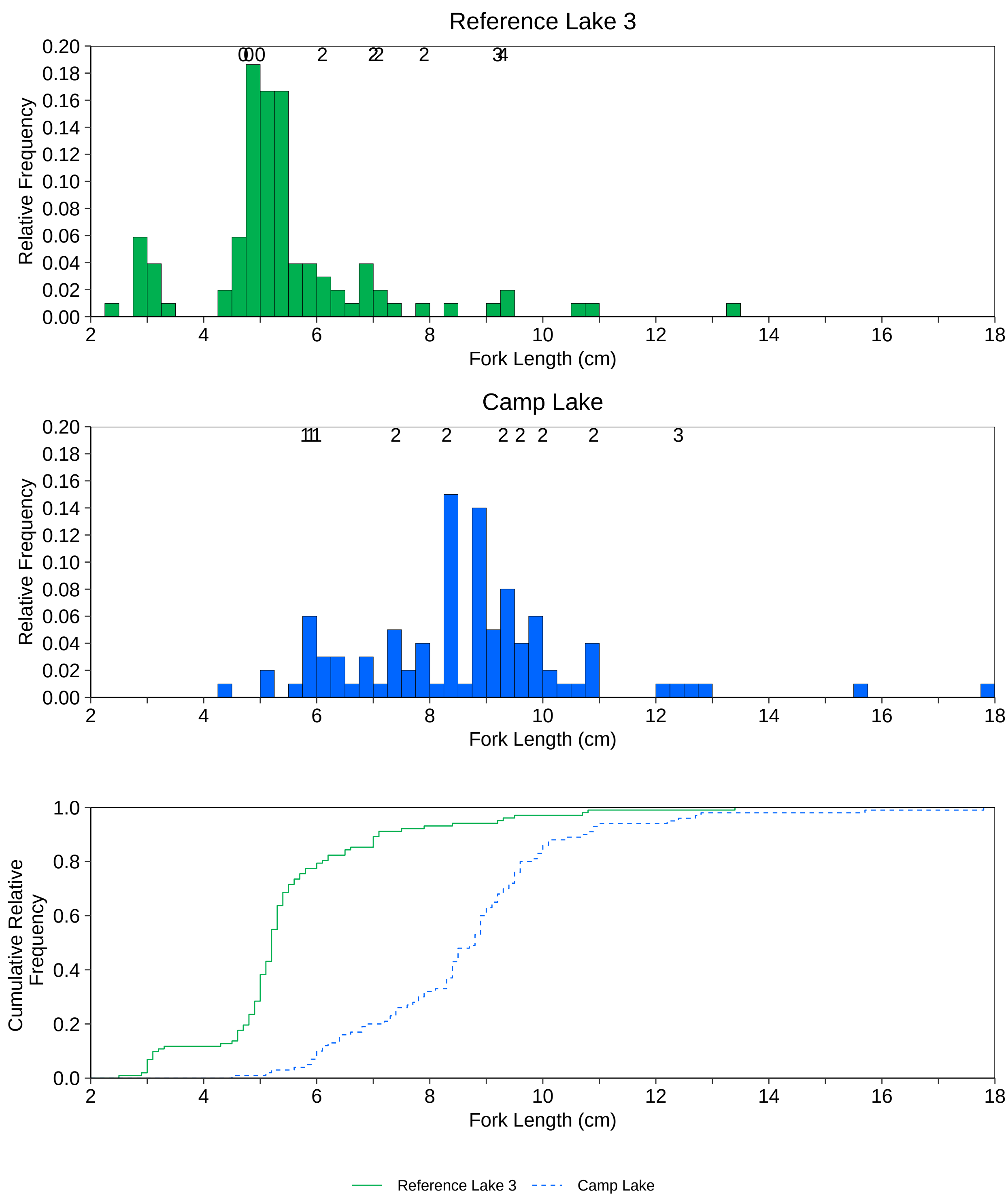


Figure 3.13: Relative Length−Frequency and Cumulative Length−Frequency Distributions for All Arctic Charr Captured by Backpack Electrofishing at Camp Lake (JL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Fish ages are shown above the bars, where available. Camp Lake n = 100; Reference Lake 3 n = 102.

Table 3.10: Summary of Statistical Results for Arctic Charr Population Comparisons between Camp Lake (JL0) and Reference Lake 3 (REF-03), and between Camp Lake Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP, 2015 to 2024

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]																			
			versus Reference Lake 3										versus Camp Lake baseline period data [b]									
			2015	2016	2017	2018	2019	2020	2021	2022	2023	2024	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Nearshore Electrofishing	Survival [c]	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	**Yes (+41%)**	No	**Yes (+17%)**	**Yes (+40%)**	**Yes (+10%)**	**Yes (+8%)**	**Yes (+134%)**	**Yes (+54%)**	**Yes (+5.7%)**	**Yes (+66%)**	**Yes (-15%)**	**Yes (-32%)**	**Yes (-35%)**	**Yes (-28%)**	No	**Yes (-22%)**	**Yes (+9.6%)**	**Yes (-22%)**	**Yes (-37%)**	**Yes (-25%)**
		Size (mean weight)	**Yes (+176%)**	No	**Yes (+51%)**	**Yes (+135%)**	**Yes (+29%)**	**Yes (+44%)**	**Yes (+1,103%)**	**Yes (+147%)**	**Yes (-4.2%)**	**Yes (+371%)**	**Yes (-42%)**	**Yes (-71%)**	**Yes (-74%)**	**Yes (-56%)**	No	**Yes (-52%)**	**Yes (+38%)**	**Yes (-56%)**	**Yes (-76%)**	**Yes (-64%)**
		Growth (weight-at-age)	**Yes (+154%)**	No	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
		Growth (fork length-at-age)	**Yes (+36%)**	**Yes (+18%)**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	No	**Yes (-6%)**	No	**Yes (-14%)**	**Yes (-7%)**	**Yes (+7%)**	**Yes (-7.4%)**	No	**Yes (-14%)**	**Yes (+5.7%)**	**Yes (-6%)**	**Yes (-10%)**	**Yes (-10%)**	**Yes (-9%)**	**Yes (-11%)**	No	**Yes (-3.5%)**	**Yes (-7.8%)**	**Yes (-6.3%)**	**Yes (-12%)**
Littoral/Profundal Gill Netting [d]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	-	-	-	-	-	-	-	-	-	-	**Yes (+48%)**	**Yes (+58%)**	**Yes (+ 46%)**	-	-	-	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	**Yes (+10%)**	**Yes (+28%)**	**Yes (+24%)**	**Yes (+33%)**	**Yes (+27%)**	No	**Yes (+33%)**	**Yes (+6%)**	No	**Yes (+12%)**	**Yes (+15%)**	**Yes (+17%)**	**Yes (+19%)**	**Yes (+21%)**	**Yes (+19%)**	**Yes (+17%)**	**Yes (+16%)**
		Size (mean weight)	-	-	-	**Yes (+46%)**	**Yes (+130%)**	**Yes (+129%)**	**Yes (+180%)**	**Yes (+158%)**	No	**Yes (+142%)**	No	No	**Yes (+37%)**	**Yes (+46%)**	**Yes (+44%)**	**Yes (+47%)**	**Yes (+52%)**	**Yes (+51%)**	**Yes (+34%)**	**Yes (+31%)**
		Growth (fork length-at-age)	-	-	-	-	-	-	-	-	-	-	No	**Yes (nc)**	No	-	-	-	-	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	-	-	-	-	No	**Yes (nc)**	No	-	-	-	-	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+12%)**	**Yes (+6%)**	**Yes (+18%)**	**Yes (+34%)**	**Yes (+30%)**	**Yes (-7.9%)**	**Yes (+20%)**	No	**Yes (-3%)**	No	No	No	No	**Yes (-4.4%)**	No	**Yes (-6.6%)**	**Yes (-5.8%)**

BOLD indicates a statistically significant difference.

Notes: "-" indicates data not available for comparison. YOY = Young-of-the-Year.

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

[b] Baseline period data included 2013 nearshore electrofishing data and 2006, 2008, and 2013 littoral/profundal gill netting data. nc = non-calculable magnitude.

[c] The length-frequency distribution to Reference Lake 3 includes all fish, whereas for baseline conditions, it only includes non-YOY fish.

[d] 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.

2023; Table 3.10), generally reflecting higher relative frequencies of larger fish in Camp Lake (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024).

Non-YOY arctic charr from Camp Lake were significantly longer (66%) and heavier (371%) than those from Reference Lake 3 in 2024 (Table 3.10, Appendix Figure G.3, Appendix Table G.6). Additionally, non-YOY arctic charr from Camp Lake had significantly greater condition (i.e., weight-at-length; 5.7%) compared to those collected from Reference Lake 3, the magnitude of which was within the CES_C of ± 10% indicating that it was not ecologically meaningful (Table 3.10, Appendix Table G.6; Appendix Figure G.3). Fork length and body weight have been consistently greater for nearshore arctic charr from Camp Lake compared to those from Reference Lake 3 from 2015 to 2024, with the exception of in 2016 when no significant difference was observed for either endpoint, and in 2023 when fish from Camp Lake were significantly lighter than fish from Reference Lake 3 (Table 3.10). Condition of non-YOY arctic charr from Camp Lake has varied from 2015 to 2024 relative to the reference lake, with higher condition in fish from Camp Lake in 2024, though most often fish from Camp Lake had lower condition than fish from Reference Lake 3 (Table 3.10). Generally lower condition of fish in Camp Lake compared to Reference Lake 3 despite greater weight and length reflects a common pattern of fish growth where weight does not increase proportionally with length. Relative increases in fish weight and length depend on energy allocation which can be influenced by multiple factors such as habitat use, food availability, and environmental conditions. MOD values for length, weight, and condition of fish from Camp Lake relative to Reference Lake 3 in 2024 fell within the ranges observed throughout the mine operational period (2015 to 2023; Table 3.10).

A significant difference in the LFD of non-YOY nearshore arctic charr from Camp Lake was observed between 2024 and the Camp Lake baseline period which is consistent with annual comparisons to baseline in previous mine operational years (Table 3.10, Appendix Figure G.2, Appendix Table G.7). Non-YOY arctic charr in 2024 were significantly shorter (-25%), lighter (-64%), and exhibited lower condition (-12%) compared to individuals captured during the baseline period (Table 3.10, Appendix Figure G.4, Appendix Table G.7). The observed difference in condition between fish captured in 2024 and baseline was outside the CES_C of ±10%, suggesting that this difference was ecologically meaningful (Table 3.10, Appendix Figure G.4, Appendix Table G.7). However, the observed differences in length and weight reflect a higher frequency of smaller individuals captured at Camp Lake in 2024, in contrast to the more uniform distribution of fish across the range of lengths recorded during the baseline period (Appendix Figure G.2). Results from 2024 are consistent with most previous years of mine operation, when non-YOY arctic charr from Camp Lake were significantly shorter, lighter, and exhibited lower condition compared to baseline, although the MODs for condition have typically been within the CES_C and therefore differences have not been considered ecologically meaningful in consecutive sampling years (Table 3.10, Appendix Table G.7, Appendix Figure G.4). General differences in the size and condition of nearshore arctic charr between Camp Lake and Reference Lake 3, as well as within Camp Lake between the mine operational and baseline periods, may reflect differences in lake productivity and fish density (Section 3.3.5.1). Camp Lake's higher productivity relative to the reference lake, as evidenced by higher chlorophyll-a concentrations (Section 3.3.3) and higher benthic invertebrate density (Section 3.3.4) could result in increased size and growth, while differences in density could lead to competition effects that variably influence these endpoints.

Overall, there have been no consistent changes in non-YOY condition in Camp Lake relative to Reference Lake 3 since 2015. Although the condition of non-YOY arctic charr in Camp Lake in 2024 was lower than in the baseline period and the absolute MOD was outside of the CES_C, this has not been the case in recent consecutive study years (i.e., 2022 and 2023) and generally the MOD for non-YOY condition at Camp Lake between mine operational years and the baseline period was within the CES_C which indicated that the difference is not ecologically meaningful. Together, these results suggest that there have been no adverse mine-related effects on the health of non-YOY arctic charr at Camp Lake since the onset of mine operations in 2015. This determination will be verified through ongoing annual assessment of fish health.

Littoral/Profundal Arctic Charr

In August 2024, a total of 110 and 84 arctic charr were sampled for fish health assessment from littoral and profundal habitat of Camp Lake and Reference Lake 3, respectively (Appendix Tables G.9 and G.11)[26,27]. The LFD of littoral/profundal arctic charr differed significantly between Camp Lake and Reference Lake 3; the majority of fish captured in Camp Lake had lengths between 30 and 45 cm long while fish captured in Reference Lake 3 were mostly less than 35 cm in length (Table 3.10, Figure 3.14, Appendix Table G.12). The LFD for littoral/profundal arctic charr has consistently been significantly different between Camp Lake and the reference lake since 2018 (Table 3.10), though there have been no consistent patterns in the relative frequencies of fish

[26] Sample sizes at Camp Lake in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.8.

[27] The total number of fish captured in Camp Lake and Reference Lake 3 by gill netting (Table 3.9, Appendix Tables G.2 and G.10) was greater than the number of fish sampled for the fish health assessment. The study design requires 100 fish from each lake to be sampled (measured and weighed; Baffinland 2015). Once field crews were certain that the minimum target sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

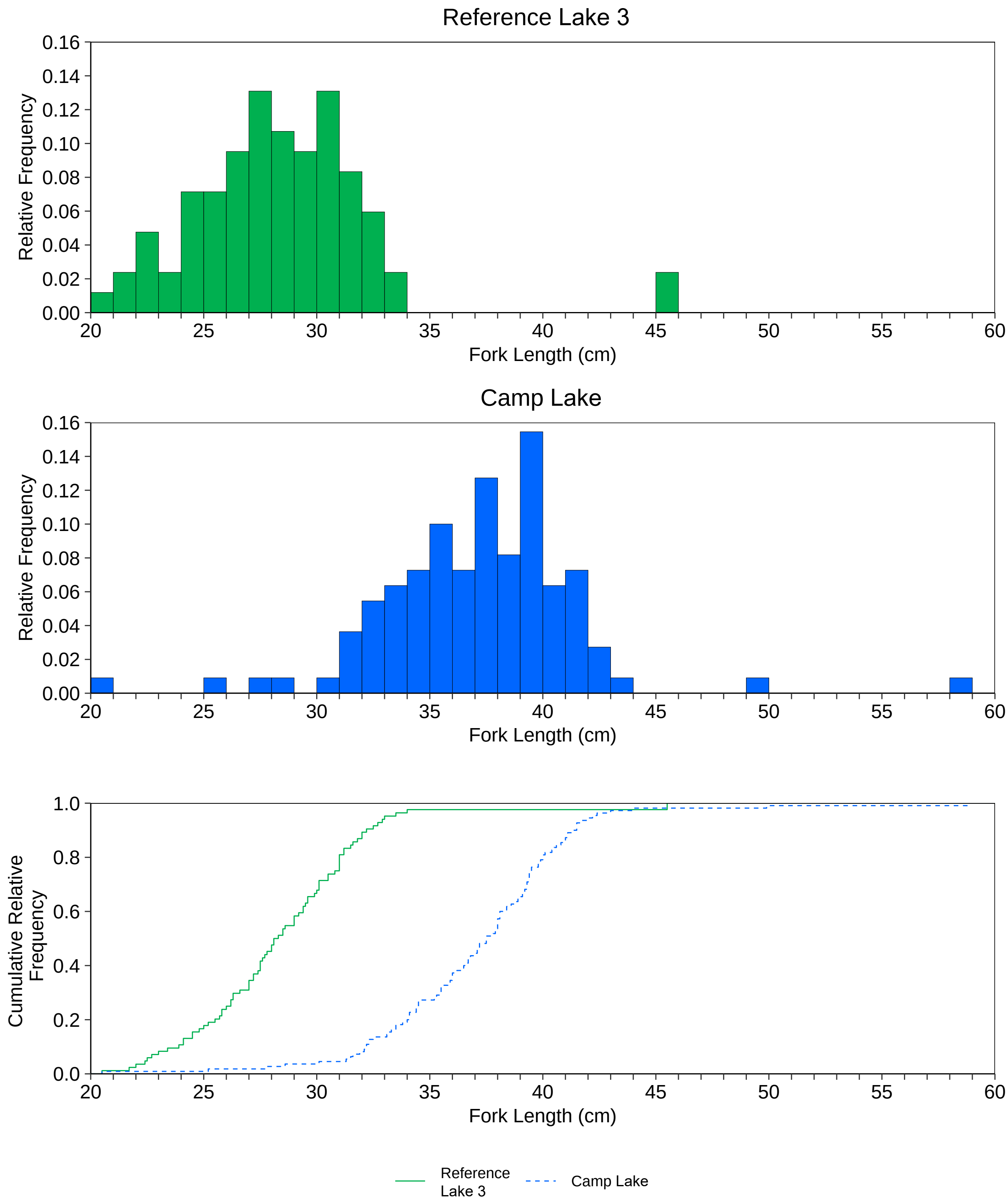


Figure 3.14: Relative Length−Frequency and Cumulative Length−Frequency Distributions for Arctic Charr Captured by Gill Netting at Camp Lake (JL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Camp n = 110; Reference Lake 3 n = 84.

lengths between the lakes (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024).

Arctic charr from Camp Lake were significantly longer (33%) and heavier (142%) compared to those from Reference Lake 3 in 2024 (Table 3.10, Appendix Figure G.6, Appendix Table G.12). Additionally, the condition of the arctic charr from Camp Lake was significantly higher (by 20%) than that of individuals from Reference Lake 3. The magnitude of this difference exceeded the CES_c of ± 10%, indicating that the difference was ecologically meaningful (Appendix Table G.12, Appendix Figure G.6, Appendix Table G.12). Fork length and body weight have been consistently greater for littoral/profundal arctic charr from Camp Lake compared to those from Reference Lake 3 from 2018 to 2024, with the exception of in 2023, when no significant difference was observed (Table 3.10). Condition of littoral/profundal arctic charr from Camp Lake has also been consistently higher than condition of fish from Reference Lake 3 since 2018 and while the MOD was outside of the CES_c in 2024, it fell within the historical range observed during mine operation (2018 to 2023; Table 3.10).

A significant difference in LFD of littoral/profundal arctic charr from Camp Lake was observed between 2024 and the combined Camp Lake baseline dataset which was consistent with results since 2018 (Table 3.10, Appendix Figure G.5). Camp Lake littoral/profundal arctic charr were significantly longer (16%) and heavier (31%) in 2024 compared to during the baseline period but exhibited a significantly lower condition factor (-5.8%; Table 3.10, Appendix Figure G.7, Appendix Table G.12). Fork length and body weight were significantly greater for littoral/profundal arctic charr captured at Camp Lake from 2017 to 2024 compared to the baseline period (Table 3.10, Appendix Figure G.7). Differences in baseline body condition have been inconsistent since 2015 with lower condition, within the CES_C was observed in Camp Lake relative to baseline in 2016, 2021, 2023, and 2024 and no significant differences in all other years (Table 3.10, Appendix Figure G.7). Although condition was significantly lower in 2024 compared to baseline, the absolute MOD was below the CES_C of ±10%, suggesting that this difference was not ecologically meaningful (Table 3.10, Appendix Table G.12). The greater size and differences in condition of littoral/profundal arctic charr from Camp Lake compared to fish from Reference Lake 3, as well as within Camp Lake between the mine operational and baseline periods, may have been influenced by the lake's higher productivity relative to the reference lake, as evidenced by higher chlorophyll-a concentrations (Section 3.3.3) and higher benthic invertebrate density (Section 3.3.4). However, multiple factors including littoral and profundal fish density and capture efficiency, as well as variation in nearshore fish density, size, and condition may have also contributed.

Overall, littoral/profundal arctic charr at Camp Lake have consistently been larger and exhibited greater condition over the mine operational period compared to those from Reference Lake 3. Furthermore, littoral/profundal arctic charr at Camp Lake were consistently larger but showed no ecologically significant difference in condition compared to baseline. Therefore, no mine-related adverse effects on the health of adult arctic charr at Camp Lake are indicated since the onset of mine operations in 2015.

3.3.6 Effects Assessment and Recommendations

In 2024, water chemistry parameter concentrations at Camp Lake met all AEMP benchmarks across all seasonal sampling events (spring, summer, fall). When comparing water quality parameter concentrations to both reference and baseline across all seasons or within a single season, only total uranium was elevated in summer in 2024, suggesting a potential mine-related influence.

Although uranium concentrations at Camp Lake have consistently remained well below the WQG of 0.0150 mg/L throughout the mine operational period (2015 to 2024), visual assessment of temporal data indicated a defined increase in total uranium concentrations in Camp Lake in all seasons between 2017 and 2022 though concentrations appear to have stabilized from 2022 to 2024.

In 2024, the following sediment quality AEMP benchmarks were exceeded at Camp Lake:

- Manganese concentrations in two individual profundal sediment samples exceeded the AEMP benchmark of 4,370 mg/kg at Stations JL0-17 and JL0-21 in August (4,470 mg/kg and 4,400 kg/kg, respectively); and
- Iron concentrations in one individual profundal sediment sample exceeded the AEMP benchmark of 52,400 mg/kg at Station JL0-17 in August (56,900 mg/kg).

Manganese and iron concentrations in sediment were not elevated compared to reference and baseline concentrations, indicating the exceedance of AEMP benchmarks is likely due to natural variation rather than a mine-related influence. Additionally, the mean concentrations of these parameters throughout profundal habitats in Camp Lake were below the AEMP benchmark in 2024. No other sediment quality parameters had elevated concentrations compared to Reference Lake 3 and baseline in 2024, indicating no mine-related influence on sediment quality at Camp Lake.

No adverse mine-related effects on phytoplankton, BIC, or fish (arctic charr) health were observed at Camp Lake in 2024, based on comparisons to Reference Lake 3 and baseline data from Camp Lake.

In accordance with the AEMP Management Response Framework, a Low Action Response is required based on determination of mine-related influence on total aqueous uranium concentrations in Camp Lake in 2024 (Figure 2.6). The following actions are recommended:

- In 2025, temporal trend analysis of aqueous total and dissolved uranium concentrations will be conducted for Camp Lake to further investigate temporal patterns.
- In 2025, an analysis of total compared to dissolved aqueous concentrations of uranium will be completed to investigate biological availability and further determine potential for effects on aquatic biota.
- Potential sources of uranium to Camp Lake will be investigated to better define mine-related influence and the potential for continued contributions.
- Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.

According to the Mary River Project AEMP Management Response Framework, the absence of any mine-related influences on sediment chemistry concentrations or biota, means no further management response is required for these monitoring components at Camp Lake in 2024 (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at Camp Lake in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-7 (Camp Management) and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at Camp Lake conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

Comparison of sediment quality at Camp Lake in 2024 to FEIS predictions related to Airborne Emission sources (i.e., fugitive dust; FEIS Issue SWSQ-17-3) indicated that all mean parameter concentrations were within the applicable significance rating magnitudes expected for lake sediments during mine operations. Therefore, sediment quality at Camp Lake conformed with predictions made in Baffinland FEIS (Baffinland 2012).

Water and sediment quality at Camp Lake in 2024 where parameter concentrations were within applicable FEIS significance rating magnitude predictions also meant that FEIS predictions for (absence of) effects on arctic charr health and condition were also met. Therefore, arctic charr health and condition at Camp Lake in 2024 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

4 SHEARDOWN LAKE SYSTEM

4.1 Sheardown Lake Tributary 1 (SDLT1)

4.1.1 Water Quality

4.1.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at SDLT1 (Stations D1-05 and D1-00) concurrent with water quality sampling in spring, summer, and fall (Figure 2.1), as well as concurrent with BIC sampling in August (Figure 2.3). At SDLT1, DO was consistently above saturation (> 91.5%; > 10.6 mg/L) during spring, summer, and fall sampling events in 2024 (Appendix Tables C.1 to C.3). During the August BIC sampling, DO concentrations at SDLT1 were significantly higher than those at the reference stream and were well above the WQG (i.e., lowest acceptable concentration for early life stages of cold-water biota of 9.5 mg/L; Figure 4.1, Appendix Tables C.30 to C.32). *In situ* pH was significantly higher at SDLT1 compared to Unnamed Reference Creek (Figure 4.1, Appendix Tables C.30 to C.32); however, both values are within WQG (Figure 4.1). *In situ* specific conductance was consistently higher at SDLT1 compared to the reference streams during spring, summer, and fall monitoring events (Appendix Figure C.10, Appendix Tables C.1 to C.3) and significantly higher (i.e., 400%) at SDLT1 compared to Unnamed Reference Creek during the August 2024 BIC sampling, suggesting a potential mine-related influence on specific conductance at SDLT1 (Figure 4.1, Appendix Tables C.30 to C.32).

4.1.1.2 Water Chemistry

At SDLT1 (Stations D1-05 and D1-00), mean concentrations of total nitrate, cadmium, cobalt, and copper were the only water chemistry parameters to exceed the AEMP benchmark and/or WQG during at least one seasonal sampling event in 2024 (Table 4.1, Appendix Table C.33). Total nitrate concentrations were above the AEMP benchmark (3 mg/L) and WQG (3 mg/L) in both the summer (mean = 5.43 mg/L) and fall (mean = 6.32; Table 4.1, Appendix Table C.33). Total nitrate concentrations were highly elevated (≥ 10 times) relative to reference and baseline conditions in all seasons in 2024 at both SDLT1 stations (D1-05 and D1-00; Appendix Table C.34). Nitrate was identified as potentially mine-influenced in 2023, triggering a Moderate Action Response under the AEMP Management Response Framework (i.e., temporal trend analyses Figure 2.6; Minnow 2024a). Temporal trend analyses revealed significant increasing trends in nitrate concentrations at both SDLT1 sampling stations in all seasons combined and in each individual season since the baseline period (2005 to 2024) and over the mine operation period (2015 to 2024; Appendix Figure C.11, Appendix Tables H.3 and H.4).

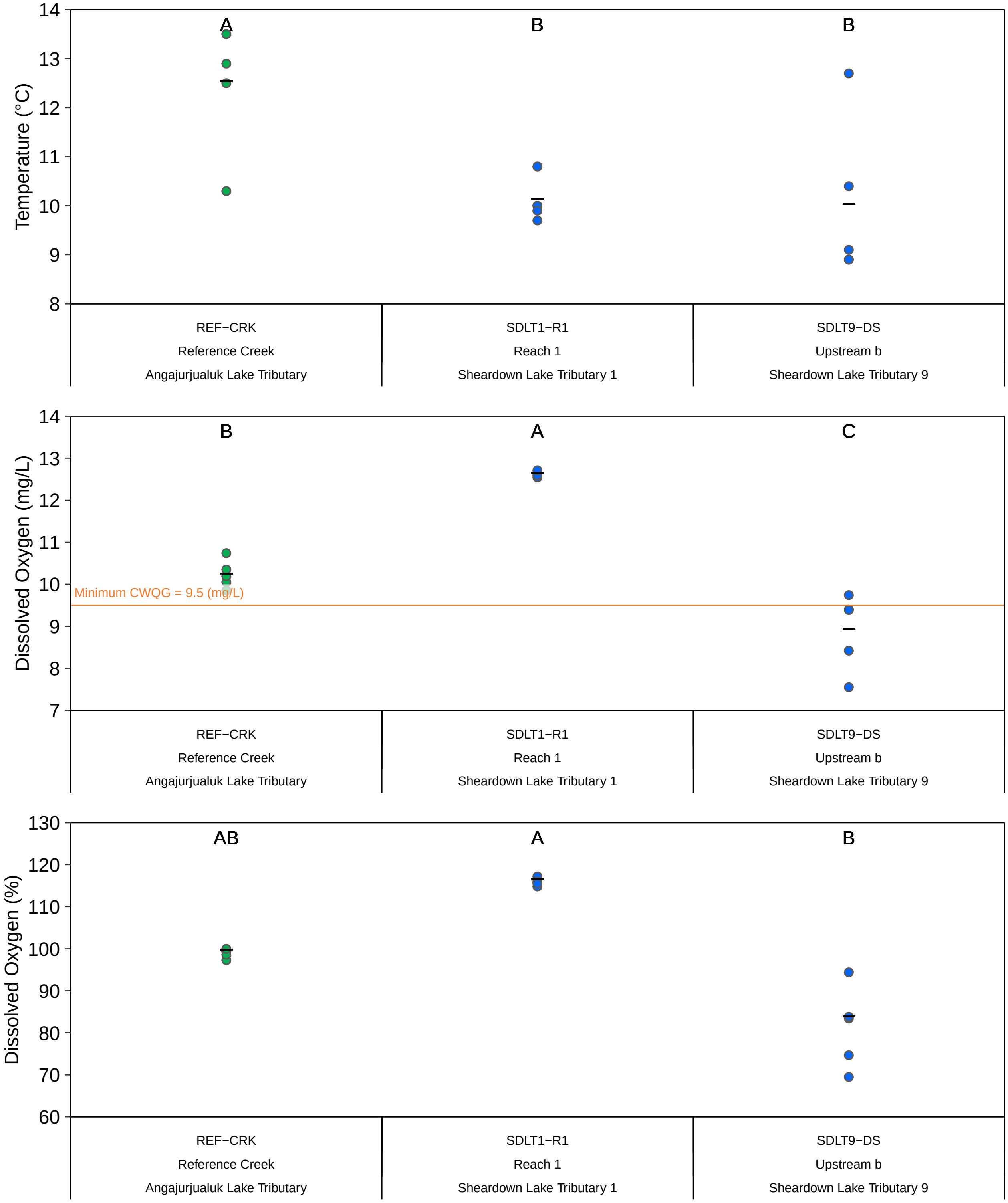


Figure 4.1: Comparison of *In Situ* Water Quality Measured at Sheardown Lake Tributaries (SDLT1 and SDLT9) and Reference Creek (REF−CRK) Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

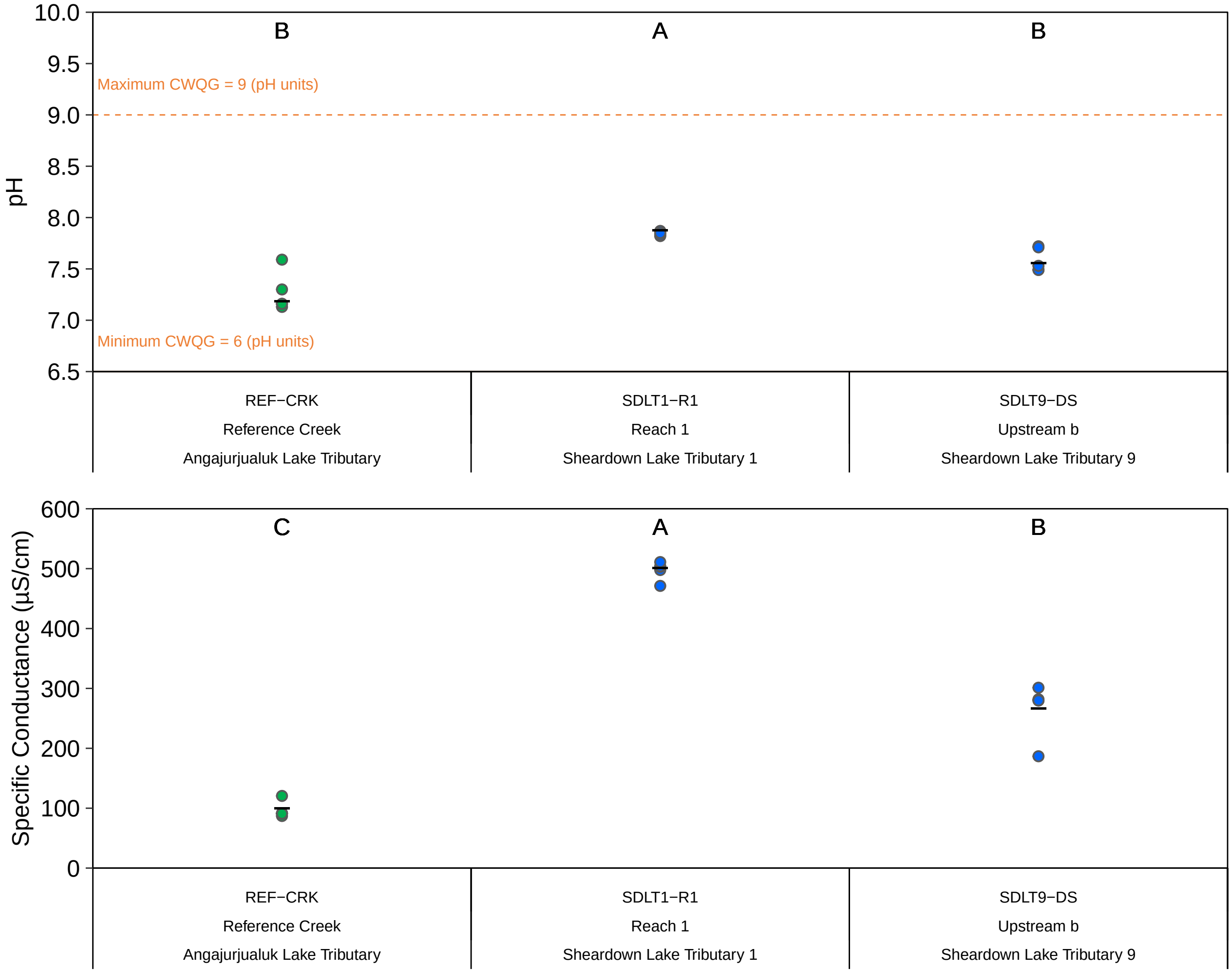


Figure 4.1: Comparison of *In Situ* Water Quality Measured at Sheardown Lake Tributaries (SDLT1 and SDLT9) and Reference Creek (REF−CRK) Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

Table 4.1: Mean Water Chemistry at Sheardown Lake Tributaries (SDLT1, SDLT12, and SDLT9) Monitoring Stations in Spring, Summer, and Fall, Mary River Project CREMP, 2024

	Parameters	Units	Water Quality Guideline (WQG)[a,b]	AEMP Bench-mark[c]	Reference Creek (n = 4)			Sheardown Lake Tributary 1 (n = 2)			Sheardown Lake Tributary 12[d] (LDFG-OUT)			Sheardown Lake Tributary 9 (MS-C-G; n = 1)		
					Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	28.1	82.5	107	322	678	550	261	-	-	167	242	243
	pH (lab)	pH	6.5 - 9.0	-	7.63	7.60	7.77	7.78	7.66	7.66	8.10	-	-	7.73	7.44	7.37
	Hardness (as $CaCO_3$)	mg/L	-	-	13.0	38.6	50.4	138	314	254	130	-	-	87.5	116	114
	Total Suspended Solids	mg/L	-	-	2.65	1.27	<1	1.40	<1	<1	1.00	-	-	<1	<1	<1
	Total Dissolved Solids	mg/L	-	-	25.2	48.2	48.5	194	440	340	133	-	-	93.0	143	128
	Turbidity	NTU	-	-	2.72	3.68	3.69	1.87	1.58	1.28	3.04	-	-	0.380	1.33	0.160
	Alkalinity (as $CaCO_3$)	mg/L	-	-	12.4	37.7	53.2	52.5	73	92.0	118	-	-	88.6	86.4	94.6
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00592	0.00520	0.00565	0.0257	0.0486	0.0884	<0.005	-	-	0.00800	<0.005	***1.45***
	Nitrate	mg/L	3	3	<0.02	0.0240	<0.02	2.16	***5.43***	***6.32***	0.0700	-	-	1.30	***8.12***	***7.08***
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	0.0220	0.0125	<0.01	-	-	<0.01	<0.01	0.0400
	Total Kjeldahl Nitrogen	mg/L	-	-	0.101	0.0768	0.0635	0.333	0.340	0.455	0.103	-	-	0.410	0.406	1.78
	Dissolved Organic Carbon	mg/L	-	-	2.28	2.20	2.14	2.06	2.63	2.92	2.99	-	-	2.15	2.46	2.70
	Total Organic Carbon	mg/L	-	-	1.92	1.82	2.11	1.96	2.23	2.56	3.06	-	-	2.35	2.71	3.00
	Total Phosphorus	mg/L	0.030[α]	-	0.00450	0.00318	0.00335	0.00285	<0.002	<0.002	0.00280	-	-	0.00280	<0.002	<0.002
	Phenols	mg/L	0.004[α]	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	-	-	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	0.130	<0.1	0.190	<0.1	-	-	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	0.605	1.24	1.67	15.6	32.6	19.6	3.23	-	-	0.860	1.16	1.16
	Sulphate (SO_4)	mg/L	218[β]	218	0.542	1.72	2.44	70.8	194	138	14.2	-	-	1.43	2.35	3.18
Total Metals	Aluminum (Al)	mg/L	0.100	0.179	0.0670	0.0832	0.160	0.0442	0.0345	0.0433	0.0311	-	-	0.00530	0.00340	0.00490
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	-	-	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	0.000100	<0.0001	<0.0001	<0.0001	0.000100	-	-	<0.0001	<0.0001	<0.0001
	Barium (Ba)	mg/L	1[β]	-	0.00201	0.00480	0.00714	0.0168	0.0302	0.0218	0.0147	-	-	0.00919	0.0119	0.0160
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00002	<0.00002	0.0000205	<0.00002	<0.00002	<0.00002	<0.00002	-	-	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	-	-	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	0.0145	0.0180	0.0175	0.0120	-	-	<0.01	<0.01	<0.01
	Cadmium (Cd)	mg/L	0.00012	0.00006	0.00000532	<0.000005	<0.000005	0.0000280	***0.000334***	***0.000206***	<0.000005	-	-	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	2.56	7.68	10.3	29.7	52.9	44.0	25.7	-	-	16.3	23.1	24.0
	Chromium (Cr)	mg/L	0.001	0.003	0.000522	0.000565	0.000672	0.000540	<0.0005	<0.0005	<0.0005	-	-	<0.0005	<0.0005	<0.0005
	Cobalt (Co)	mg/L	0.0009[α]	0.004	0.000102	0.000105	0.000125	0.000185	0.00124	0.00110	0.000160	-	-	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.0022	0.000600	0.000852	0.00114	0.00151	0.00159	0.00218	0.00169	-	-	0.000600	0.00113	0.00119
	Iron (Fe)	mg/L	0.30	0.326	0.0810	0.0942	0.143	0.0950	0.0720	0.0795	0.0440	-	-	0.0110	<0.01	0.0150
	Lead (Pb)	mg/L	0.001	0.001	0.000100	0.000114	0.000154	0.0000985	0.0000535	0.0000595	<0.00005	-	-	<0.00005	<0.00005	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	0.00415	0.00975	0.00725	0.00310	-	-	<0.001	0.00120	0.00130
	Magnesium (Mg)	mg/L	-	-	1.74	4.96	6.36	18.6	46.0	39.6	18.4	-	-	11.6	14.7	15.2
	Manganese (Mn)	mg/L	0.935[β]	-	0.00141	0.00126	0.00162	0.00542	0.641	0.279	0.00347	-	-	0.000780	0.000140	0.000880
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	-	-	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.0000752	0.000232	0.000420	0.00526	0.00708	0.00544	0.00201	-	-	0.000457	0.000581	0.000718
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	0.000500	0.000698	0.000935	0.00178	0.00204	0.00142	-	-	0.000870	0.00129	0.00130
	Potassium (K)	mg/L	-	-	0.313	0.559	0.789	5.63	6.80	5.01	3.02	-	-	1.24	1.39	1.64
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	0.000143	0.000708	0.000788	<0.00005	-	-	<0.00005	<0.00005	<0.00005
	Silicon (Si)	mg/L	-	-	0.475	0.932	1.20	1.17	1.63	1.83	0.810	-	-	1.06	0.980	0.880
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	-	-	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.383	1.12	1.60	3.20	6.82	6.08	3.58	-	-	1.16	1.16	1.50
	Strontium (Sr)	mg/L	-	-	0.00238	0.00758	0.0111	0.103	0.164	0.101	0.0176	-	-	0.00883	0.0133	0.0146
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	0.0000105	0.0000170	0.0000240	0.0000185	<0.00001	-	-	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	-	-	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.00410	0.00471	0.00760	<0.002	<0.002	<0.0015	<0.0008	-	-	<0.0003	<0.0003	<0.0003
	Uranium (U)	mg/L	0.015	-	0.000212	0.00115	0.00286	0.00298	0.00970	0.0106	0.00161	-	-	0.000430	0.000606	0.000723
	Vanadium (V)	mg/L	0.006[α]	0.006	0.000508	0.000522	0.000625	<0.0005	<0.0005	<0.0005	<0.0005	-	-	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02[α]	0.030	<0.003	<0.003	<0.003	<0.003	0.00325	<0.003	<0.003	-	-	<0.003	<0.003	<0.003

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

BOLD Indicates parameter concentration above the AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

[a] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[b] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

[d] Station LDFG-OUT (Sheardown Lake Tributary 12) was dry during summer and fall sampling events in 2024, therefore no data are available for these seasons

Visual assessment of temporal data indicated that while concentrations of total nitrate have generally been higher than baseline and reference concentrations over the mine operations period (i.e. since 2015), consistent seasonal patterns of increasing concentrations from year to year have only been observed since 2022, with concentrations above the AEMP benchmark and WQG in summer of 2023 and 2024 and fall of 2024 (Appendix Figure C.11). No significant temporal trends in nitrate concentrations were observed at reference streams (Appendix Figure C.11, Appendix Tables H.3 and H.4), indicating that elevated concentrations and increasing trends at SDLT1 are likely linked to mining activities rather than natural regional changes.

Total cadmium concentrations were above the AEMP benchmark (0.00008 mg/L) and WQG (0.00012 mg/L) in both the summer (mean = 0.000334 mg/L) and fall (mean = 0.000206 mg/L; Table 4.1; Appendix Table C.33). Compared to reference stream stations and baseline concentrations, total and dissolved cadmium concentrations were moderately (5 to 10 times higher) or highly (≥ 10 times higher) elevated in the summer and fall at both SDLT1 stations (Appendix Tables C.34, C.35, and C.36). Cadmium was identified as potentially mine-influenced in 2022 and 2023, triggering a Moderate Action Response in each year under the AEMP Management Response Framework (i.e., temporal trend analyses Figure 2.6; Minnow 2023 and 2024a). Temporal trend analyses completed in 2023 found a significant increasing trend in total cadmium concentrations at both SDLT1 sampling stations, as well as a significant increasing trend in dissolved cadmium concentrations at the downstream station (D1-00) over the years of mine operation (2015 to 2023), driven by increasing concentrations in the summer and fall of 2022 and 2023 relative to baseline and earlier mine operations years (Minnow 2024a). In 2024, the temporal trend analyses for total and dissolved cadmium concentrations at SDLT1 were repeated, incorporating the most recent data. These analyses found significant increasing trends for total and dissolved cadmium concentrations in all seasons combined and in individual sampling seasons (primarily summer and fall) at both SDLT1 sampling stations since the baseline period (2005 to 2024) and over the mine operational period (2015 to 2024; Appendix Figure C.11, Appendix Tables H.3 and H.4). Cadmium concentrations have increased in the summer and fall (open water season) over the period from 2022 to 2024, with an increasing frequency of concentrations that exceeded the AEMP benchmark, and the highest concentrations recorded in the summer of 2024 (Appendix Figure C.11). Reference streams did not indicate similar trends in cadmium concentrations based on the results of 2024 temporal trend analyses[28]

[28] Total and dissolved cadmium concentration data for reference streams only date as far back as 2014 (i.e., the construction period) and > 75% of measured concentrations were below LRL. Therefore, temporal trend analyses could not be formally completed for the reference streams. However, results regularly below the LRL indicate that increasing trends, like those identified at SDLT1, are not occurring at reference streams.

(Appendix Figure C.11, Appendix Tables H.3 and H.4), suggesting that natural regional increases in cadmium concentrations are not occurring and trends at SDLT1 likely associated with mining activities.

Average total cobalt concentrations exceeded the WQG (0.0009 mg/L) in both the summer (mean = 0.00124 mg/L) and fall (mean = 0.00110 mg/L; Table 4.1, Appendix Table C.33). When compared to reference stream stations and baseline concentrations, total and dissolved cobalt concentrations were moderately to highly elevated during summer and fall at both SDLT1 stations (Appendix Tables C.34 and C.36). Visual assessment of temporal data revealed a pattern of increasing cobalt concentrations in the summer and fall over the period from 2022 to 2024, when concentrations were higher than both reference and baseline (Appendix Figure C.11). While cobalt concentrations have remained below the AEMP benchmark (0.004 mg/L) throughout the operational period (2015 to 2024), elevated concentrations relative to both reference and baseline, along with an increasing pattern since 2022 during the summer and fall, suggest a seasonal mine-related influence.

Average total copper concentrations exceeded the WQG (0.002 mg/L) in the fall (mean = 0.00218 mg/L; Table 4.1, Appendix Table C.33). However, neither total nor dissolved copper concentrations were elevated relative to reference stream stations or baseline in any season in 2024 (Appendix Table C.34 and C.36). Fall 2024 sample concentrations were generally comparable to or lower than those collected over the mine operation period (2015 to 2024) and were similar to or below baseline concentrations, though higher than reference concentrations (Appendix Figure C.11). A special investigation into elevated copper concentrations above the AEMP benchmark at SDLT1 in 2021, which involved spatially expanded sampling, found no distinct source of copper to SDLT1, suggesting that copper concentrations that are typically above reference concentrations are likely due to a naturally occurring, rather than mine-related, source (Appendix Figure C.11; Minnow 2022).

In addition to total nitrate, cadmium, cobalt, and copper, total and dissolved water chemistry parameter concentrations that were slightly (3 to 5 times), moderately, or highly elevated relative to reference or baseline conditions are identified in Appendix Tables C.34 and C.36. Some parameter concentrations were elevated compared to both reference and baseline concentrations across all seasons in 2024. At both SDLT1 stations (D1-05 and D1-00), concentrations of sulphate, total lithium, and total and dissolved strontium were elevated across all seasons compared to reference and baseline conditions (Appendix Tables C.34 and C.36, Appendix Figure C.11). At Station D1-05, located upstream of the tote road on SDTL1, total magnesium and potassium concentrations were also elevated in all seasons relative to reference and baseline conditions (Appendix Figure C.11, Appendix Tables C.34 and C.36) and at Station

D1-00, downstream of the tote road on SDLT1, dissolved lithium and manganese concentrations were elevated across all seasons (Appendix Table C.36). These parameters with elevated total and/or dissolved concentrations throughout the open-water season, were also identified in the 2023 CREMP as potentially mine-influenced, triggering a Low Action Response under the AEMP Management Response Framework (Minnow 2024a). Temporal trend analyses revealed significant increasing trends at both SDLT1 sampling stations in all seasons combined and in each individual season for sulphate, total and dissolved lithium, magnesium, potassium, and strontium concentrations since the baseline period (2005 to 2024) and over the mine operation period (2015 to 2024; Appendix Figure C.11, Appendix Tables H.3 and H.4). This pattern differed slightly for concentrations of total and dissolved manganese for which there were significant increasing trends at both SDLT1 sampling stations when results from all seasons were combined and in most individual sampling seasons, but not at the upstream station (D1-05) in summer or fall for total manganese or spring, summer, or fall for dissolved manganese (Appendix Figure C.11, Appendix Tables H.3 and H.4). Increasing trends in concentrations for these water chemistry parameters were only occasionally observed (in all seasons combined or in individual sampling seasons) in reference streams (Appendix Figure C.11, Appendix Tables H.3 and H.4), indicating that elevated concentrations and increasing trends at SDLT1 are likely linked to mining activities rather than natural regional changes. Visual assessment of temporal data indicates that while concentrations of total sulphate, lithium, magnesium, potassium, and strontium have generally been higher than baseline and reference concentrations over the mine operations period (i.e. since 2015), consistent seasonal patterns of increasing concentrations from year to year have only been observed since 2022 (Appendix Figure C.11). Meanwhile, manganese concentrations, first peaked above baseline and reference concentrations in the summer of 2023, followed by dramatic increases in summer and fall of 2024 (Appendix Figure C.11). Although concentrations of these parameters have remained below AEMP benchmarks and WQGs, suggesting limited potential for adverse effects on aquatic biota, consistently elevated concentrations relative to reference and baseline conditions, coupled with increasing temporal trends indicate a mine-related influence.

Some total and dissolved water chemistry parameter concentrations were elevated compared to both reference and baseline concentrations in only one or two seasons in 2024 (Appendix Tables C.34 and C.36). Parameters with concentrations that were elevated in at least one season relative to reference and baseline concentrations at SDLT1 Station D1-05 and/or D1-00 included conductivity, TDS, TKN, chloride, total lithium and potassium, and total and dissolved barium, calcium, magnesium, manganese, molybdenum, selenium, sodium, and uranium (Appendix Tables C.34 and C.36). In general, water chemistry parameter concentrations were similar between upper SDLT1 (Station D1-05) and lower SDLT1 (Station D1-00) in 2024, indicating little dilution between these two monitoring points and minimal influence from the tributary that flows from the southeast into upper SDLT (Table 4.1, Figure 2.1, Appendix Tables C.33 to C.36).

Some parameter concentrations that were elevated in one or more seasons in 2024 compared to reference and baseline levels at Station D1-05 and/or D1-00, including chloride and uranium, also had elevated concentrations in 2023 that were considered to be mine-related, triggering a Low Action Response under the AEMP Management Response Framework (Minnow 2024a). A temporal trend analysis in 2024 found significant increasing trends in chloride and total and dissolved uranium concentrations in all seasons combined and in almost all individual seasons since the baseline period (2005 to 2024) and over the mine operational period (2015 to 2024) at both SDLT1 sampling stations[29] (Appendix Figure C.11, Appendix Tables H.3 and H.4). Consistent significant increasing trends in chloride and total and dissolved uranium were not observed in all seasons combined or in any individual season in the reference areas, indicating a mine-related influence rather than natural regional changes. Visual assessment of temporal data indicates that concentrations of chloride and total uranium during the mine operations period were generally slightly higher than reference concentrations, but within the baseline range until 2022 for chloride and 2020 for total uranium (Appendix Figure C.11). Chloride and total uranium concentrations were higher than during the previous years of mine operations starting in 2022 and 2020, respectively, and both peaked in the fall of 2022 before gradually declining in 2023 and 2024 (Appendix Figure C.11). Despite this gradual decline, and although chloride and total uranium concentrations were below respective AEMP benchmarks and/or WQGs in 2024, consistently elevated concentrations relative to reference and baseline conditions, coupled with increasing temporal trends indicate a mine-related influence on these parameters at SDLT1.

The remaining parameters with concentrations that were elevated compared to reference and baseline concentrations in at least one season at one or both SDLT1 sampling stations in 2024 (conductivity, TDS, TKN, and total and dissolved barium, calcium, molybdenum, selenium, and sodium), have not previously indicated potential mine-influences based on previous annual monitoring. Visual assessment of temporal data indicated that while conductivity, TDS and concentrations of total barium, and calcium have generally been higher than baseline and reference over the mine operations period (i.e. since 2015), concentrations increased relative to earlier mine operations years between 2022 and 2024, with the highest concentrations occurring in the summer and fall (Appendix Figure C.11). TKN concentrations showed a slight increase from 2023 to 2024 relative to previous mine operational years (Appendix Figure C.11).

[29] The only exception was for chloride in the fall at Station D1-05, where there was no significant trend since the baseline period or during the mine operations period.

Total molybdenum, and sodium concentrations have exhibited general increasing trends over the mine operations period, though concentrations of molybdenum decreased slightly in 2024 compared to the peak observed in 2023 (Appendix Figure C.11). Finally, total selenium concentrations appear to have increased at SDLT1 between 2023 and 2024, particularly in the summer and fall, though limited data above LRL prior to 2023 prevents evaluation of a longer-term temporal pattern (Appendix Figure C.11). While concentrations of total barium, molybdenum, and selenium in 2024 were below applicable WQGs), elevated concentrations relative to reference and baseline conditions in at least one season at one or both SDLT1 sampling stations, combined with the observed temporal patterns of increasing concentrations, particularly since 2022, suggest a potential mine-related influence on these water quality parameters at SDLT1.

Two parameters, aluminum and iron, had elevated concentrations at SDLT1 in 2023 that were considered to be potentially mine-related, triggering a Moderate Action Response in the 2023 CREMP (Minnow 2024a); however, they did not have concentrations that were elevated relative to reference and baseline conditions prior to 2023 or in 2024. As a Moderate Action Response to 2023 results, a temporal trend analysis was completed and found significant increasing trends in total aluminum and iron in all seasons combined and in the spring at Station D1-05, as well as in total iron at Station D1-00 for combined seasons, both since the baseline period (2005 to 2024) and over the mine operational period (2015 to 2024; Appendix Figure C.11, Appendix Tables H.3 and H.4). No significant temporal trends were found for dissolved aluminum or iron concentrations (Appendix Table H.3 and H.4). Similar trends were observed at the MRY-REF3 reference area but not at other reference sites. The inconsistency in trends among reference sites suggests a mine-related influence at SDLT1 rather than a broader natural regional trend. For both total aluminum and iron, concentrations substantially above the ranges of reference and baseline concentrations and exceeding the respective AEMP benchmark and WQG were observed in 2023 but not in any other mine operational year (including 2024 when concentrations were similar to reference and baseline and below AEMP benchmarks and WQG; Appendix Figure C.11, Appendix Tables C.33 to C.36). The elevated total aluminum and iron concentrations at SDLT1 in 2023 may be partially attributed to higher turbidity. In spring 2023, total aluminum, iron, and turbidity were elevated compared to reference, while aluminum and iron remained high in spring and summer relative to baseline, and turbidity was consistently high across all seasons (Minnow 2024a). The greater turbidity observed in all 2023 seasons compared to baseline may have partially reflected natural conditions related to the high flow observed in the mine site area in 2023 (Minnow 2024a) but was also likely influenced by mine-related factors that have not persisted in 2024 when results indicated no mine-related influence on iron and aluminum.

A large portion of the mine site infrastructure is located in the SDLT1 catchment resulting in the potential for mine-related influences from non-point source and airborne emissions (Baffinland 2012) and management plans are in place to manage and mitigate influences in the SDLT1 catchment associated with site water management, laydown areas, camps, waste management, and dust deposition. For parameters with concentrations that were elevated at SDLT1 in 2024 relative to reference and baseline concentrations and for which a mine-related influence was concluded, visual assessment of temporal data suggested that patterns of consistently increasing concentrations have only generally only been observed since 2022. The main potential mine-related influence that originated upstream of both SDLT1 monitoring stations in the period from 2022 to 2024 is likely site water management through the KM 105 Surface Water Management Pond (the KM 105 Pond; constructed in 2021 and 2022; Figure 2.2; Minnow 2024b). The KM 105 Pond was designed and constructed to enhance water management at the mine site, including by addressing sedimentation issues resulting from Deposit 1 through settling of TSS. However, the KM 105 has not performed as expected and has experienced persistent seepage and water quality challenges since its commissioning in 2022 (Baffinland 2025). Despite multiple remediation efforts targeted at structural reinforcements (e.g., cut-off trench, bentonite plugs, and a grout curtain) seepage has continued from multiple locations within the KM 105 Pond's dam structure through 2024 (Baffinland 2025). Efforts to reduce seepage and improve downstream water quality have included chemical treatments, filtration systems, and the installation of silt curtains, sediment curtains, and check dams. Engineering assessments on the KM 105 Pond dam determined that previous remediation strategies, including the grout curtain, were ineffective due to the unforeseen permeability of the substrate (Baffinland 2025).

Water quality in the receiving environment downstream of seepage and treated effluent released from the KM 105 Pond have been monitored and reported relative to criteria as per the Mine's Type 'A' Water License (i.e., the water license) and the Metal and Diamond Mining Effluent Regulations (MDMER, [Baffinland 2025, Minnow 2024b]). These water quality results have consistently met water license and MDMER limits, despite downstream water quality at SDLT1 reflecting mine-related influences based on the CREMP data assessment approach (see Section 2.2.3.2) and AEMP Management Response Framework (see Section 2.5.1). In particular, parameter concentrations that were elevated relative to reference and baseline conditions and for which there is evidence of increasing trends indicate a mine-related influence. This is the case despite parameter concentrations that remained below applicable AEMP benchmarks and WQG at SDLT1 in 2024 receiving environment water quality meeting water license and MDMER limits. To address the ongoing challenges of managing water from the KM 105 Pond, and in turn mitigate mine-related influences in the receiving environment at SDLT1, focus in 2025 will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications to the dam.

Mine-related influences other than site water management upstream of Station D1-05 (i.e., at the KM 105 Pond) have the potential to affect water chemistry in SDLT1 at downstream Station D1-00. These include influences from a tributary that originates from the southeast and flows into SDLT1 between Stations D1-05 and D1-00, the Sailivik Camp located in the lower portion of the SDLT1 watershed near the confluence with Sheardown Lake NW, and general site dust deposition (Figure 2.1). However, the similarity in water quality between Stations D1-00 and D1-05, particularly for parameters identified as having mine-related influence in 2024, suggests that mine-related influences driving water quality conditions in SDLT1 are from sources upstream of D1-05 (Figure 2.1).

Overall 19 parameters had total and/or dissolved concentrations that were elevated relative to reference and baseline concentrations in at least one season in 2024 at one or both of the SDLT1 sampling stations. These parameters included conductivity, TDS, nitrate, TKN, chloride, sulphate, barium, cadmium, calcium, cobalt, lithium, magnesium, manganese, molybdenum, potassium, selenium, strontium, sodium, and uranium. For each of these parameters, either statistical or visual evidence of increasing temporal trends/patterns supported comparisons to reference and baseline concentrations in indicating mine-related influences. Among them, cadmium was the only analyte to exceed the AEMP benchmark in 2024, while the others remained below respective benchmarks or lacked an established site-specific threshold. Seepage from the KM 105 Pond dam and associated remediation efforts are likely the primary contributor to the observed mine-related influences on these parameters.

4.1.2 Phytoplankton

In 2024, mean chlorophyll-a concentrations at upper SDLT1 (Station D1-05) were similar to those near the stream mouth (Station D1-00) during the spring, summer, and fall sampling events (Figure 4.2, Appendix Table E.1). Chlorophyll-a concentrations at SDLT1 were generally within or lower than the range of variability observed at reference streams for the same seasons in 2024 and were consistently below the AEMP benchmark of 3.7 µg/L during all sampling events (Figure 4.2, Appendix Table E.1). Measured chlorophyll-a concentrations indicate low phytoplankton productivity and oligotrophic conditions based on chlorophyll-a concentrations (i.e., <8 µg/L; Dodds et al. (1998); Appendix Table E.1) and total phosphorus concentrations (i.e., <10 µg/L; CCME 2024b; Table 4.1, Appendix Table C.33; see Section 3.1.2 for additional trophic status classification details).

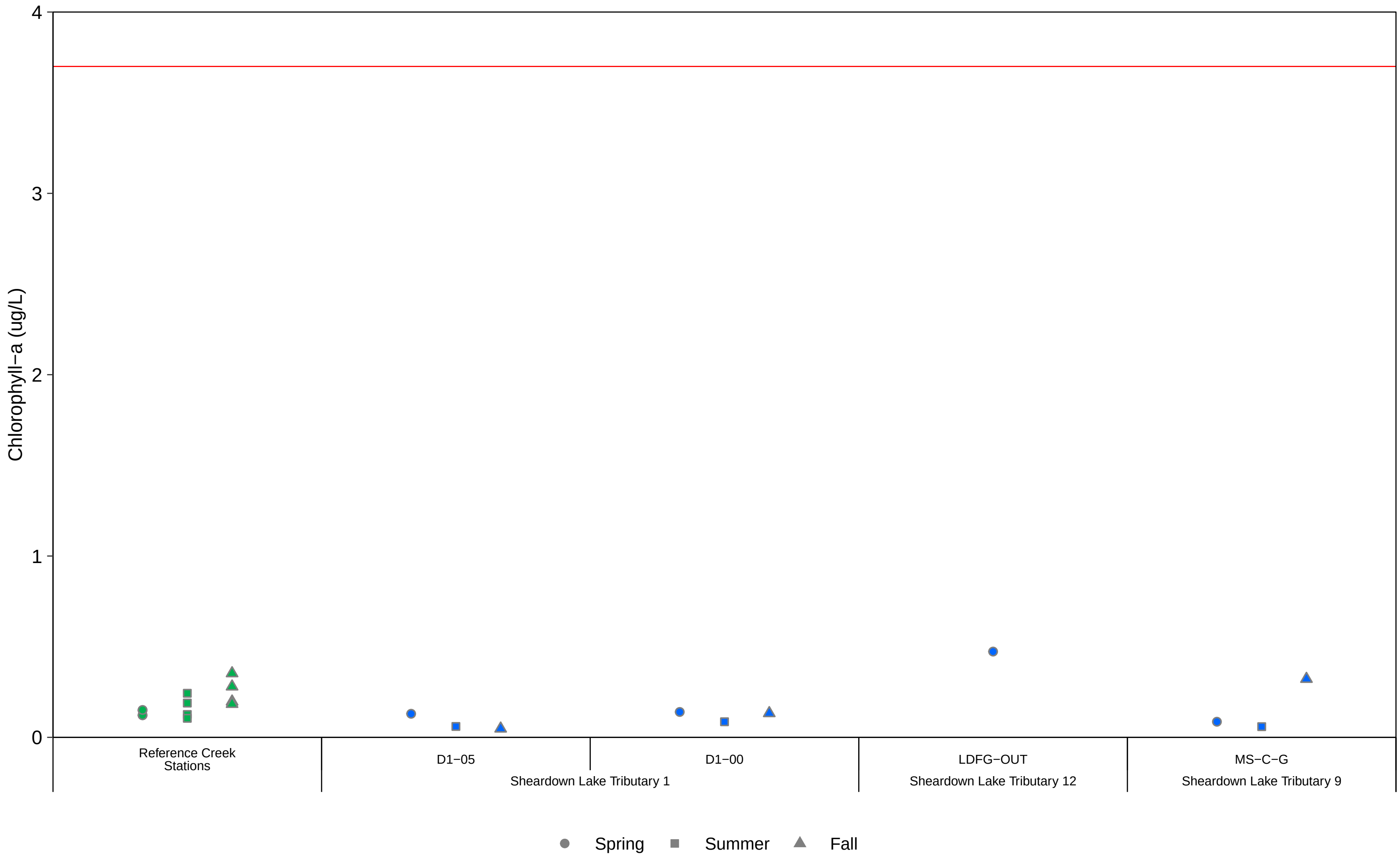


Figure 4.2: Chlorophyll−a Concentrations at Sheardown Lake Tributary (SDLT1, SDLT12, SDLT9) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2024

Notes: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Red line indicates AEMP Benchmark. Reference Creek Stations includes data from stations CLT−REF4, CLT−REF3, MRY−REF3, and MRY−REF2. Reference areas are shown in green and mine−exposed areas are shown in blue.

When visually comparing chlorophyll-a concentrations seasonally and among years, concentrations at SDLT1 stations in 2024 were lower than those observed during the baseline period (2005 to 2013; Figure 4.3). However, the baseline period featured relatively high LRLs compared to 2024, which may partially explain the observed difference (Figure 4.3). Despite the potential influence of shifting LRLs, lower chlorophyll-a concentrations were observed at SDLT1 in 2024 compared to the rest of the operational period (2015 to 2023; Figure 4.3). Specifically, chlorophyll-a concentrations were lower in the spring and summer of 2024 compared to 2023 but remained similar in the fall (Figure 4.3). Chlorophyll-a concentrations in reference stream samples in 2024 were also lower compared to the rest of the operational period and to spring and summer in 2023, suggesting that the difference in both SDLT1 and reference streams was likely due to natural inter-annual variation. On-going monitoring will continue to evaluate for evidence of temporal patterns in the data. Overall, the available data indicate no consistent directional changes (i.e., increasing or decreasing) in chlorophyll-a concentrations at SDLT1 across any seasonal sampling events during the baseline (2005 to 2013), construction (2014), and operational (2015 to 2023) periods; the stream has remained oligotrophic; and chlorophyll-a concentrations in 2024 remained well below the AEMP benchmark (Figure 4.2). These results indicate no adverse mine-related effects on phytoplankton productivity at SDLT1 in 2024.

4.1.3 Sediment Quality

Sediment is collected on a three-year cycle from the streams monitored in the CREMP, with samples taken in 2023. Therefore, no sediment quality sampling was completed at the SDLT1 stream station in 2024. In 2023, metal concentrations in sediment at SDLT1 (Station SDLT1-R1) were generally elevated compared to reference areas, and the mean concentration of iron exceeded the SQG. The source of elevated sediment metal concentrations compared to reference in 2023 was unclear and stream sediment data are temporally limited (i.e., stream sediment sampling was initiated as a component of the CREMP in 2017), precluding comparison to baseline for a determination of mine-related influence. Though not required under the AEMP Management Response Framework, in the 2023 CREMP report (Minnow 2024a) it was recommended that temporal plots and/or statistical temporal trend analyses of the available 2017, 2020, and 2023 sediment quality data for iron be evaluated to determine whether changes in iron concentrations in sediment of SDLT1 have occurred over the sampling period during mine operations (i.e., since 2017). This was completed through a statistical temporal assessment in 2024 (see Section 2.3.3.2; Appendix Figure H.3 and Appendix Table H.7). Results found no statistically significant differences in mean concentrations among years (Appendix Figure H.3 and Appendix Table H.7) and therefore, the limited available data suggest that iron concentrations in sediment of SDLT1 have not increased over the mine operations period, since 2017.

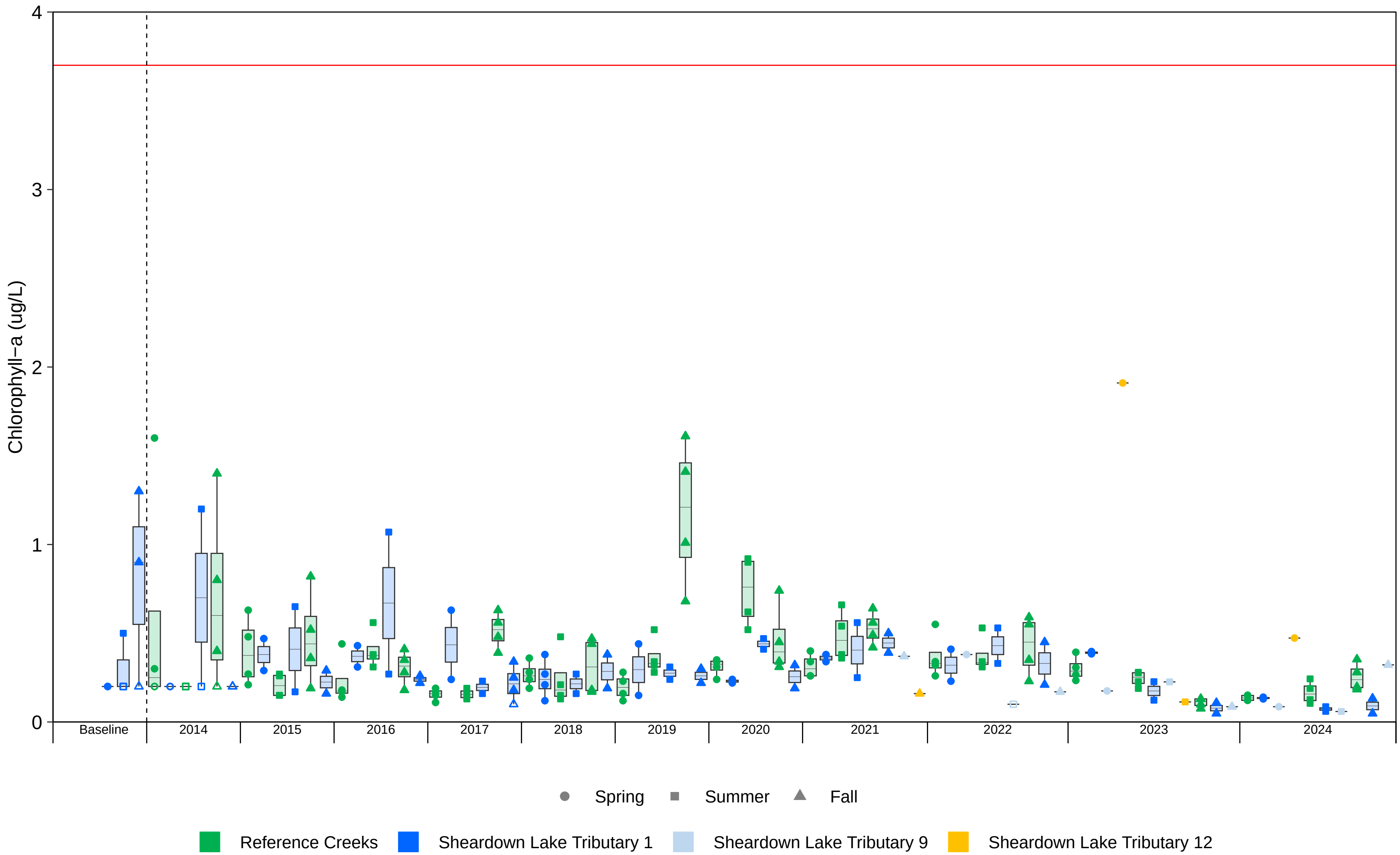


Figure 4.3: Temporal Comparison of Chlorophyll−a Concentrations at the Sheardown Lake Tributaries (SDLT1, SDLT12, and SDLT9) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Sheardown Lake Tributary 12 (SDLT12/LDFG−OUT) and Sheardown Lake Tributary 9 (SDLT9/MS−C−G) stations were added to the CREMP in fall 2021. Reference Creeks includes stations CLT−REF4, CLT−REF3, MRY−REF3, and MRY−REF2. Sheardown Lake Tributary 1 includes stations D1−05 and D1−00. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from whiskers.

As previously described for CLT1 (see Section 3.1.3), sediment sampling locations in CREMP streams are in highly erosional environments with very limited fine sediment accumulation. Although sediment sampling is focused on locations containing the finest grain sizes available, incorporation of larger material (i.e., sand, gravel, etc.) that is more geochemically inert suggests limited relevance of stream sediments as an exposure pathway to aquatic biota. Additionally, the collection of sediments in streams and rivers has the potential to produce temporally and spatially variable results for chemical analyses due to temporal fluctuations in hydrology, particularly in highly erosional environments. Therefore, the use of stream sediment chemistry results in evaluating for mine-related influences should be considered with caution.

4.1.4 Benthic Invertebrate Community

In 2024, most BIC endpoints at SDLT1, including Simpson's Evenness and relative proportions of *Hydracarina*, *Oligochaeta*, *Chironomidae, Simuliidae,* collector-gatherers, filterers, shredders, clingers, and sprawlers, were significantly different from the reference creek (Table 4.2, Appendix Table F.26, Appendix Figure F.7). Simpson's Evenness, along with the relative proportions of *Hydracarina*, *Simuliidae*, collector-gatherers, filterers, and sprawlers were significantly lower at SDLT1 relative to reference (Table 4.2, Appendix Table F.26, Appendix Figure F.7). Among these, Simpson's Evenness and the relative proportions of *Hydracarina*, collector-gatherer and shredder FFGs, and sprawler HPGs exhibited ecologically meaningful differences, as indicated by MODs outside the CES_{BIC} threshold of ±2 SD_{REF} (Table 4.2). In contrast, the relative proportions of *Chironomidae* were significantly higher at SDLT1 compared to the reference creek in 2024, and relative proportions of shredder FFG and clinger HPG were significantly and ecologically meaningfully higher at SDLT1 compared to the reference creek (Table 4.2, Appendix Table F.26, Appendix Figure F.7). Finally, the Bray-Curtis Index confirmed significant structural differences between SDLT1 and the reference creek, consistent with the differences noted above for relative proportions of indicator taxa, FFGs, and HPGs (Appendix Table F.27).

Lower Simpson's Evenness and relative proportions of *Hydracarina*, *Simuliidae*, collector-gatherers, filterers, and sprawlers in the BIC from SDLT1 versus the reference area suggest SDLT1 had a less diverse and less functionally balanced community. The higher proportions of *Chironomidae*, shredders, and clingers at SDLT1 relative to the reference area indicate a community dominated by pollution-tolerant and detritus-processing taxa. Observed differences in water quality at SDLT1 in 2024 relative to reference and baseline, including emerging increasing trends in certain parameter concentrations since the baseline period and over the mine operations period (see Section 4.1.1.2), suggest mine-related influences on water quality may be contributing to the increased abundance of pollution-tolerant taxa. However, sediment chemistry

Table 4.2: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Sheardown Lake Tributary 1 (SDLT1), Sheardown Lake Tributary 9 (SDLT9), and Unnamed Reference Creek (REF-CRK) Study Areas, Mary River Project CREMP, August 2024

Endpoint	Overall Three-Area Comparison[a]				Pair-wise, *post hoc* comparisons				
	Statistical Test	Transform-ation	Significant Difference between Areas?	P-value	Study Area	Mean	Standard Deviation	MOD[b]	Pairwise Comparison
Density (org/m^2)	ANOVA	log10	NO	0.167	Reference Creek	393	397	nc	A
					SDLT1	640	214	0.82	A
					SDLT9	2,148	2,082	1.4	A
Richness (No. Taxa)	ANOVA	log10	NO	0.248	Reference Creek	15.6	5.90	nc	A
					SDLT1	10.8	2.49	-0.90	A
					SDLT9	11.6	3.78	-0.76	A
Simpson's Evenness (Krebs)	ANOVA	none	YES	<0.001	Reference Creek	0.949	0.0322	nc	A
					SDLT1	0.525	0.0950	-13	C
					SDLT9	0.763	0.155	-5.8	B
% Nematoda	K-W	rank	YES	0.064	Reference Creek	0.704	1.00	nc	B
					SDLT1	1.73	1.26	nm	AB
					SDLT9	4.44	4.63	nm	A
% Hydracarina	ANOVA	none	YES	<0.001	Reference Creek	6.47	1.62	nc	A
					SDLT1	1.01	0.844	-3.4	B
					SDLT9	1.28	1.72	-3.2	B
% Ostracoda	K-W	rank	YES	0.044	Reference Creek	2.61	2.65	nc	AB
					SDLT1	0.179	0.401	-0.72	B
					SDLT9	7.13	10.1	0.081	A
% Oligochaeta	K-W	rank	YES	0.004	Reference Creek	0.0690	0.154	nc	B
					SDLT1	3.07	1.49	nm	A
					SDLT9	0.144	0.322	nm	B
% Chironomidae	K-W	rank	YES	0.054	Reference Creek	68.9	9.23	nc	B
					SDLT1	89.7	3.96	1.5	A
					SDLT9	77.0	23.6	1.3	AB
% Metal Sensitive Chironomidae	ANOVA	log10(x+1)	NO	0.733	Reference Creek	8.62	10.7	nc	A
					SDLT1	11.9	5.24	0.34	A
					SDLT9	12.4	9.52	0.37	A
% Simuliidae	K-W	rank	YES	0.005	Reference Creek	9.93	6.72	nc	A
					SDLT1	0	0	-1.9	B
					SDLT9	2.40	4.12	-1.8	A
% Tipulidae	K-W	rank	NO	0.429	Reference Creek	4.83	6.79	nc	A
					SDLT1	2.74	1.90	0.76	A
					SDLT9	1.50	2.36	-0.16	A
% Collector Gatherers FFG	ANOVA	none	YES	<0.001	Reference Creek	60.4	11.9	nc	A
					SDLT1	24.8	5.68	-3.0	B
					SDLT9	69.7	14.7	0.78	A
% Filterers FFG	ANOVA	log10(x+1)	YES	0.059	Reference Creek	1.94	1.79	nc	A
					SDLT1	0.214	0.297	-0.96	B
					SDLT9	0.393	0.657	-0.86	AB
% Shredders FFG	ANOVA	none	YES	<0.001	Reference Creek	13.2	9.93	nc	B
					SDLT1	73.6	5.51	6.1	A
					SDLT9	26.5	15.4	1.3	B
% Clingers HPG	ANOVA	none	YES	<0.001	Reference Creek	24.7	11.1	nc	B
					SDLT1	72.1	6.29	4.3	A
					SDLT9	27.9	13.9	0.28	B
% Sprawlers HPG	ANOVA	none	YES	<0.001	Reference Creek	54.5	15.0	nc	A
					SDLT1	20.2	5.48	-2.3	B
					SDLT9	62.2	7.68	0.51	A
% Burrowers HPG	ANOVA	log10	YES	0.049	Reference Creek	20.7	12.5	nc	A
					SDLT1	7.55	3.82	-1.6	AB
					SDLT9	6.16	4.59	-2.4	B

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

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

Notes: MOD = Magnitude of Difference. nc = no comparison. nm = MOD could not be calculated due to SD = 0. FFG = Functional Feeding Group. HPG = Habitat Preference Group

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

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

analyses from the 2023 CREMP report found no evidence of mine-related influence at SDLT1, with metal concentrations remaining below applicable SQGs (Minnow 2024a) suggesting no sediment quality influence on the BIC.

Factors such as organic matter inputs could also be influencing community composition. Higher presence of bryophytes (Appendix Table F.22) and slightly higher aqueous DOC concentrations (Appendix Tables B.2 and C.33), both compared to reference areas, may contribute to spatial differences in organic matter availability, potentially influencing the BIC at SDLT1. Additionally, water depth was significantly greater and substrate was significantly more embedded at SDLT1 relative to the reference creek in 2024, indicating that physical habitat differences may be contributing to the observed differences (Appendix Tables F.22 and F.24). Substrate embeddedness, where fine sediments fill interstitial spaces between larger particles, can reduce habitat availability for taxa that rely on clean, well-aerated substrate, such as sprawlers, filterers, and collector-gatherers, which typically require open spaces for attachment and feeding. In contrast, taxa such as *Chironomidae* and shredders, which are more tolerant of fine sediment accumulation and organic matter deposition, may benefit from these conditions. Similarly, increased water depth can indicate altered flow conditions (i.e., higher flows within streams) and reduce habitat complexity, potentially limiting the availability of preferred microhabitats for *Hydracarina*, *Ostracoda*, and *Simuliidae*, while favoring organisms that thrive in more stable, depositional environments. Although total invertebrate densities and richness were comparable between SDLT1 and the reference creek, the dominance of shredders and *Chironomidae* and lower proportions of filterers and sprawlers at SDLT1 suggest potential effects from mine-related influences on water quality as well as potential changes in organic matter availability and/or substrate composition.

Temporally, nearly all BIC endpoints reported for SDLT1 during mine operation years (2015 to 2024) differed significantly from baseline (2008, 2013), except for taxonomic richness and the relative proportion of filterer FFG (Appendix Table F.28, Appendix Figure F.8). In 2024, Simpson's Evenness was significantly lower, whereas the relative proportions of *Oligochaeta*, *Chironomidae*, metal sensitive *Chironomidae*, *Tipulidae* (relative to 2013 only), collector-gatherers, and shredders were significantly higher, compared to baseline (2008 and 2013; Appendix Table F.28, Appendix Figure F.8). Further, the temporal increases in relative proportions of *Oligochaetes* (since 2020), *Chironomidae* (since 2018), metal sensitive *Chironomidae* (since 2020), *Tipulidae* (since 2020), collector gatherer FFG (since 2019), and shredder FFG (since 2020) in the BIC from SDLT1 have been ecologically meaningful since 2018, 2019, or 2020 (Appendix Table F.28), suggesting a sustained shift in BIC structure, rather than natural variability. The other endpoints for which significant differences were observed, ecological significance has been inconsistent, showing no trends (Appendix Table F.28, Appendix Figure F.8).

The decline in Simpson's Evenness, combined with the consistent increases in relative proportions of *Oligochaeta*, *Chironomidae*, metal-sensitive *Chironomidae*, *Tipulidae*, and functional groups associated with organic matter processing, suggests a gradual, yet ecologically significant, restructuring of the macroinvertebrate community at SDLT1 over time. The increases in relative proportions of *Chironomidae*, *Oligochaetes*, and shredders point to organic matter enrichment or habitat modifications, whereas the increasing proportions of metal-sensitive *Chironomidae* suggests that metal contamination is not the primary factor driving changes in the BIC.

The main potential mine-related influence on water quality identified in SDLT1 in the period from 2022 to 2024 is likely site water management through the KM 105 Pond (see Section 4.1.1). Water chemistry parameters indicating mine-related influence at SDLT1 in 2024 included conductivity, TDS, nitrate, TKN, chloride, sulphate and multiple metals (see Section 4.1.1). However, all water chemistry parameter concentrations, except for total cadmium, remained below applicable AEMP benchmarks and WQG at SDLT1 in 2024, suggesting low potential for adverse effects to aquatic organisms. These results are consistent with differences from the reference area and temporal shifts in the SDLT1 BIC that are not indicative of metal contamination as the primary stressor. Through the process of management of water from the KM 105 Pond and attempted remediation efforts at the KM 105 Pond dam, there was the potential for increased influences of contact water on SDLT1, though concentrations of TSS remained below receiving environment water quality limits (i.e., water license and MDMER) downstream at CREMP water quality and BIC monitoring stations (Baffinland 2025). Although aqueous TSS concentrations were below limits expected to be protective of aquatic life, potential TSS inputs to the SDLT1 system could have influenced substrate embeddedness through the introduction of fine particles that filled interstitial spaces downstream. This is a potential mechanism for observed greater substrate embeddedness at SDLT1 compared to the reference area in 2024 that is reflected in differences in the composition of the BIC. Finally, elevated (relative to reference and baseline) and temporally increasing concentrations of nutrients such as nitrate and TKN could contribute to increased bryophyte growth at SDLT1, which is a potential factor influencing the availability of organic matter and potentially the BIC community.

Overall, the BIC at SDLT1 in 2024 exhibited statistically significant and ecologically meaningful differences compared to the reference creek and baseline conditions, suggesting a long-term shift in community composition, however, not benthic invertebrate density. Mine-related influences on water quality that were identified at SDLT1 in 2024 are generally consistent with mechanisms that may have resulted in the observed shift in the BIC and therefore support the determination of a mine-related influence on the SDLT1 BIC.

4.1.5 Effects Assessment and Recommendations

At SDLT1 (Stations D1-05 and D1-00), the following AEMP benchmarks were exceeded in 2024:

- Mean aqueous total cadmium concentrations were greater than the AEMP benchmark of 0.00008 mg/L in the summer (mean = 0.000334 mg/L) and fall (mean = 0.000206 mg/L).

When comparing water quality parameters to reference and baseline concentrations across all seasons, or within a single season, the following parameters were elevated, suggesting a potential mine-related effect:

- All seasons (spring, summer, and fall): total/dissolved lithium, total magnesium, dissolved manganese, nitrate, total potassium, total/dissolved strontium, and sulphate;
- Spring: total/dissolved barium, total/dissolved calcium, chloride, conductivity, total/dissolved molybdenum, total/dissolved potassium, total/dissolved sodium, TDS, and total/dissolved uranium;
- Summer: total/dissolved barium, total cadmium, total/dissolved calcium, total cobalt, conductivity, chloride, dissolved lithium, total/dissolved magnesium, total/dissolved manganese, total/dissolved molybdenum, total/dissolved potassium, and total/dissolved selenium, total/dissolved sodium, TDS, TKN, and total/dissolved uranium; and
- Fall: total cadmium, total cobalt, dissolved magnesium, total/dissolved manganese, total/dissolved selenium, and TKN.

Not only did total cadmium concentrations exceed the AEMP benchmark in the summer and fall of 2024, but they also exceeded the WQG of 0.00012 mg/L, were elevated compared to reference and baseline in the same seasons, and showed significant increasing trends across all seasons since the baseline period and over the mine operations period. Of the other water chemistry parameters with concentrations that were elevated relative to reference and baseline in at least one season in 2024, total cobalt is the only one what exceeded the WQG (0.0009 mg/L) in the summer (mean = 0.00124 mg/L) and fall (mean = 0.00110 mg/L), but remained below the AEMP benchmark of 0.004 mg/L. Visual assessment of temporal data indicated increasing total cobalt concentrations at SDLT1 in 2023 and 2024, particularly in the summer and fall. These results indicate a mine-related influence on cadmium and cobalt during the open water season in 2024.

The remaining parameters with total and/or dissolved concentrations that were elevated relative to reference and baseline in at least one season in 2024 (i.e., barium, cadmium, calcium, chloride, cobalt, conductivity, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulfate, TDS, TKN, uranium) did not exceed AEMP benchmarks or WQGs. However each demonstrated either statistically significant increasing trends or visually increasing patterns in total concentrations particularly from 2022 to 2024 suggesting a mine-related influence. Potential mine-related influences on aluminum, chloride, iron, lithium, magnesium, manganese, nitrate, potassium, strontium, sulphate, and uranium were identified in 2023 (Minnow 2024a), and aluminum and iron were the only parameters from this list that were found not to have a similar potential mine-related effect in 2024. Overall, mine-related influences on water quality parameters in SDLT1 are likely linked to site water management through the KM 105 Surface Water Management Pond (the KM 105 Pond; constructed in 2021 and 2022).

Sediment samples, collected every three years from streams monitored under the CREMP, were taken in 2023, therefore, sediment quality was not sampled in 2024 and results are not included in this report. A special investigation into sediment iron concentrations at SDLT1, as recommended in the 2023 CREMP report (Minnow 2024a), found that despite sediment iron concentrations in 2023 that were generally elevated compared to reference areas and above the SQG, iron concentrations in sediment of SDLT1 have not significantly increased over the mine operations period, since 2017.

No adverse mine-related effects on chlorophyll-a (a measure of primary productivity) were observed at SDLT1 in 2024. However, mine-related influences on the BIC were detected at SDLT1 in 2024, with results suggesting they were likely driven by organic matter enrichment and changes in physical habitat conditions rather than metal contamination as a primary stressor. Influences associated with site water management and remediation efforts at the KM 105 Pond are consistent with the factors that may have resulted in shifts to the SDLT1 BIC in 2024. Continued monitoring of BIC is necessary to assess potential long-term impacts.

Based on the AEMP Management Response Framework, a Moderate Action Response is required for cadmium due to exceedance of the AEMP benchmark, elevated concentrations relative to reference and baseline in the summer and fall, and consistent increasing trends since baseline and over operational years (Figure 2.6). Other parameters that were elevated in one or more seasons compared to reference and baseline concentrations, which also showed increasing trends/patterns over time– including barium, calcium, chloride, cobalt, conductivity, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, TDS, TKN, and uranium – require a Low Action Response, as per the AEMP Management Response Framework (Figure 2.6). The following actions are recommended:

- In 2025, a temporal trend analysis of aqueous conductivity and total and dissolved (where applicable) concentrations of barium, calcium, cadmium, chloride, cobalt, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, TDS, TKN, and uranium will be conducted to further investigate temporal trends/patterns. In 2025, an analysis of total compared to dissolved aqueous concentrations of metals determined to have mine-related effects at SDLT1 in 2024 will be completed to investigate biological availability and further determine potential for effects on aquatic biota.

- Continued monitoring of the BIC at SDLT1 is recommended in 2025 (and in future CREMP studies) to track potential effects on biota and support the evaluation parameter concentrations exceeding the AEMP benchmark (cadmium), as well as those that were elevated compared to reference and baseline conditions, using a weight-of-evidence approach; and
- Potential sources of elevated/increasing water quality parameters in SDLT1 will be further investigated to better define mine-related influence and the potential for continued contributions.
- Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.
- The focus in 2025 for the KM 105 Pond remediation efforts will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. Water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT1 as a basis for informing the potential need for further investigations and mitigation.
- Installation of a filter berm upstream of the water license Surveillance Network Program monitoring location Station MS-C-D (which is located on a tributary to SDTL1 that originates from the southeast and flows into SDLT1 between Stations D1-05 and D1-00) is planned in 2025 to further mitigate for mine-related contributions of TSS to SDLT1 associated with dust and other sources of TSS within the upstream catchment area.

The absence of confirmed mine-related influences on phytoplankton (as a measure of primary productivity) means no further management response is required for these monitoring components at SDLT1 in 2024 (Figure 2.6).

Determination of mine-related influence on the BIC at SDLT1 in 2024 requires a Low Action Response under the AEMP Management Response Framework (Figure 2.6). Because the main predicted sources of mine-related influence on water quality and BIC at SDLT1 are site water management and remediation efforts at the KM 105 Pond, recommended actions associated with water quality will also serve to appropriately investigated and mitigate effects on the BIC.

Comparison to FEIS Predictions

A comparison of water quality at SDLT1 in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-7 (Camp Management), and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude of effect (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at SDLT1 in 2024 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

4.2 Sheardown Lake Tributary 9 (SDLT9)

4.2.1 Water Quality

4.2.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at SDLT9 (Station MS-C-G) concurrent with water quality sampling in spring, summer, and fall (Figure 2.1), as well as concurrent with BIC sampling in August (Figure 2.3). At SDLT9, DO was lower (78.6 to 86.1% saturation and 9.84 to 10.8 mg/L) than at the reference streams during spring, summer, and fall sampling events in 2024; however, concentrations were all above the WQG minimum for the protection of early life stages of cold-water biota (i.e., 9.5 mg/L; Appendix Figure C.10, Appendix Tables C.1 to C.3; CCME 2024a). Similarly, during the August BIC sampling, DO was lower at SDLT9 than at the reference stream though the difference was only significant for mean concentration (not mean saturation; Figure 4.1, Appendix Tables C.30 to C.32). The mean concentration of DO at SDLT9 during the August BIC sampling was slightly below the WQG minimum for the protection of early life stages of cold-water biota; however, it was above the WQG minimum for the protection of all other life stages (i.e., 6.5 mg/L; Figure 4.1, Appendix Table C.31, CCME 2024a). *In situ* pH at SDLT9 did not differ significantly from that at the reference stream during the August 2024 BIC sampling (Figure 4.1, Appendix Tables C.30 to C.32). *In situ* specific conductance was consistently higher at SDLT9 (105 µs/cm, 228 µs/cm, 240 µs/cm) compared to the reference streams (12.9 to 37.9 µs/cm, 52.2 to 93.5 µs/cm, 71.8 to 115 µs/cm) during spring, summer, and fall monitoring events, respectively and significantly higher during the August 2024 BIC sampling event (mean = 266 [SDLT9] and 96.2 [REF-CRK]; Figure 4.1, Appendix Figure C.10, Appendix Tables C.1 to C.3 and C.30 to C.32).

4.2.1.2 Water Chemistry

In 2024, water chemistry at the SDLT9 water quality station (MS-C-G) met respective AEMP benchmarks and WQGs for all parameters across all seasons, except for total ammonia and nitrate (Table 4.1, Appendix Table C.33). The total ammonia concentration exceeded the AEMP benchmark of 0.855 mg/L in the fall (1.45 mg/L) and nitrate concentrations exceeded both the AEMP benchmark (3 mg/L) and WQG (3 mg/L) during the summer (8.12 mg/L) and fall (7.08 mg/L; Table 4.1, Appendix Table C.33). In 2022 and 2023, aqueous concentrations of nitrogen compounds (i.e., total ammonia, nitrate, nitrite, and/or TKN) at SDLT9 were higher than AEMP benchmarks, reference concentrations, and/or baseline concentrations, which indicated a mine-related influence (Minnow 2023 and 2024a). In 2024, concentrations of total nitrate and TKN were consistently elevated across all sampling seasons compared to both baseline and reference conditions (Appendix Table C.34). In the fall of 2024, total ammonia and nitrite concentrations ranged from slightly to highly elevated compared to reference and baseline concentrations (Appendix Table C.34, Appendix Table C.11).

Visual assessment of temporal data indicated that total ammonia, nitrate, nitrite, and TKN concentrations at SDLT9 were higher than reference and baseline concentrations consistently in the fall and frequently in the summer from 2021 to 2024 (Appendix Figure C.11). Total ammonia concentrations in the fall and total nitrate concentrations in the summer and fall have also consistently exceeded respective AEMP benchmarks and WQG (Appendix Figure C.11). Temporal data indicate an ongoing mine-related influence on aqueous nitrogen compounds in SDLT9 since 2021, however none of the parameter concentrations have demonstrated consistently increasing temporal patterns over the period from 2021 to 2024, suggesting that the influence has not been intensifying over time.

As a Moderate Action Response under the AEMP Management Response Framework based on conclusion of a potential mine-related influence on total ammonia, nitrate, and nitrate in the 2022 CREMP, temporal trend analyses were recommended (Minnow 2023). Temporal trend analyses for total ammonia, nitrate, and nitrite were conducted in 2023 but data availability affected the ability to conduct robust trend analyses for SDLT9[30] and a high proportion of measured concentrations below LRL prevented formal trend analyses for all parameters/stations in reference streams (Minnow 2024a). Therefore, results of the temporal trend analyses were generally inconclusive.

[30] Water chemistry sampling at SDLT9 during the mine operation period was only initiated in 2021. Analyses of total ammonia, nitrate, and nitrite concentrations at MS-C-G since the baseline period found no significant trend for any of the parameters but sample sizes were too low for exact p-values to be determined. Similar trend analyses could not be completed over the mine operation period only due to insufficient data (i.e., data available only for 2021 to 2023)

As a Moderate Action Response under the AEMP Management Response Framework based on conclusion of a mine-related influence on total ammonia, nitrate, and TKN in the 2023 CREMP, a special investigation was recommended (Minnow 2024a). This investigation was implemented in 2024 and included an expanded spatial water quality sampling program to identify the source(s) of ammonia, nitrate, and TKN to SDLT9 (Minnow 2024a). Detailed methods, results, and recommendations from this special investigation are presented in Appendix I. Ammonium nitrate is used at the mine site and is stored in containers in three locations, including the Dyno Nobel Emulsion Plant (Dyno facility), which is located upgradient to SLDT9 Station MS-C-G (Appendix Figure I.1). Aqueous nitrogen compounds (i.e., total ammonia, nitrate, nitrite, and TKN) and other water chemistry parameters (i.e., total phosphorus, TOC, and phenols) that are often correlated with nitrogen compounds under natural conditions, or through different types of anthropogenic influences on aquatic systems (e.g., wastewater) were sampled at eight stations upgradient and downgradient of the Dyno facility in September 2024 (Appendix Figure I.1). Nitrogen compounds were elevated at three stations (i.e., SDLT9-1, MS-C-H-US1, and MS-C-H-US2; Appendix Figure I.1, Appendix Table I.1) downstream from the Dyno facility, suggesting that activities occurring at the Dyno facility are the source of the elevated nitrogen compounds. The WQG and AEMP benchmark for nitrate were exceeded at each of these stations but the ammonia AEMP benchmark was not. Recommendations for further investigations and mitigative actions are provided in Section 4.2.4.

Total and dissolved water chemistry parameters that were slightly, moderately, or highly elevated relative to reference or baseline conditions at SDLT9 in 2024 are identified in Appendix Tables C.34 and C.35. Besides total ammonia, nitrate, and nitrite, the only parameter with concentrations that were elevated compared to reference and baseline in any season in 2024 was total sodium in spring (slightly elevated; Appendix Table C.34). While sodium concentrations were higher than reference and baseline concentrations in the spring, visual assessment of temporal data did not indicate any consistent increasing or decreasing patterns across seasons since the initiation of sampling in 2021, and concentrations in the summer and fall were within the range observed in reference creeks, suggesting limited potential for a mine-related influence.

4.2.2 Phytoplankton

Chlorophyll-a concentrations at SDLT9 (Station MS-C-G) in 2024 were generally within, or slightly lower than, the range of variability observed among reference streams during spring, summer, and fall sampling events and concentrations were consistently well below the AEMP benchmark of 3.7 µg/L throughout 2024 (Figure 4.2, Appendix Table E.1). Measured chlorophyll-a concentrations indicate low phytoplankton productivity and oligotrophic conditions based on chlorophyll-a concentrations (i.e., <8 µg/L; Dodds et al. (1998); Appendix Table E.1) and total phosphorus concentrations (i.e., <10 µg/L; CCME 2024b; Table 4.1, Appendix Table C.33; see Section 3.1.2 for additional trophic status classification details).

Because SDLT9 was added to the CREMP as a monitoring station in fall 2021, there are no baseline data or early mine-operational data available for this station, which precludes direct comparisons of chlorophyll-a concentrations to conditions prior to mine construction or during the initial seven years of operation. However, compared to previous years when data were collected at this site, chlorophyll-a concentrations across all seasons visually demonstrate a general decreasing pattern, with the exception of slightly higher concentrations in fall of 2024 relative to previous years (Figure 4.3). The overall pattern of decreasing concentrations aligns with a similar temporal pattern in the reference streams during the same period (Figure 4.3), indicating that the changes are likely due to natural inter-annual variation. Ongoing monitoring will continue to evaluate for a consistent temporal pattern in chlorophyll-a concentrations at SDLT9.

Overall, temporal changes in chlorophyll-a concentrations at SDLT9 are consistent with those observed in the reference streams, the stream has remained oligotrophic over the sampling period, and chlorophyll-a concentrations remained below the AEMP benchmark in 2024. These results indicate that there were no adverse mine-related effects on phytoplankton productivity at SDLT9 in 2024.

4.2.3 Benthic Invertebrate Community

In 2024, most BIC endpoints at SDLT9 were statistically comparable to those of the reference creek, except for higher relative proportions of *Nematoda* and lower Simpson's Evenness and relative proportions of *Hydracarina* and burrowers at SDLT9 (Table 4.2, Appendix Table F.26, Appendix Figure F.7). Of these, Simpson's Evenness and the relative proportions of *Hydracarina* and burrowers had MODs outside of the CES_{BIC}, indicating ecologically meaningful differences (Table 4.2). These significant structural differences between SDLT9 and the reference creek were reflected in the Bray-Curtis Index (Appendix Table F.27).

The differences between SDLT9 and reference creek BIC in 2024, specifically the significantly lower relative proportions of *Hydracarina* and burrowers at SDLT9, are likely due to differences in substrate conditions (e.g., stability, composition) between areas, rather than a mine-related influence. There were significant differences in water depth and velocity at SDLT9 compared to the reference creek (Appendix Table F.24), and these factors are known to influence physical sediment characteristics and the BIC. Although sediment composition was not statistically compared between SDLT9 and the reference creek it has been observed that the substrate at SDLT9 primarily consisted of gravel and cobble, with sand and gravel as interstitial material (Minnow 2024a). Conversely, the substrate at the reference area was characterized as predominantly sand and gravel (Minnow 2024a).

Water quality at SDLT9 in 2024 was influenced by activities occurring at the adjacent Dyno facility, as indicated by elevated aqueous concentrations of nitrogen compounds downstream of the facility compared to upstream (see Appendix I and Section 4.2.1). Elevated nitrogen levels could be influencing BIC composition at SDLT9, particularly given that mean and individual water sample concentrations of ammonia and nitrate exceeded the AEMP benchmark and/or WQG in summer and/or fall of 2024 (see Section 4.2.1). High nitrogen concentrations can negatively impact *Hydracarina* and burrowing BIC populations through oxygen depletion and affecting preferred habitat conditions. Excess nitrogen can fuel algal blooms that lead to hypoxic conditions, making sediments unsuitable for burrowing organisms. Notably in 2024, algae presence at SDLT9 was much higher than at the reference area (Appendix Table F.22), supporting the potential influence of high nitrogen levels driving algal bloom formation.

Most BIC endpoints assessed for SLDT9 differed from baseline (2007 and 2013) in at least one operational year (2015 to 2024; Appendix Table F.30, Appendix Figure F.9). In 2024 specifically, compared to both baseline years, richness was significantly lower, whereas the relative proportions of *Chironomidae*, metal sensitive *Chironomidae*, collector-gatherer FFG, and shredder FFG were significantly higher and all of these differences were ecologically meaningful (Appendix Table F.30, Appendix Figure F.9). Overall, the data for the mine operational period, when compared to baseline, suggests an overall shift in benthic invertebrate densities and community structure. This is especially evident in the period since approximately 2020, when ecologically meaningful increases in invertebrate densities and relative proportions of *Hydracarina, Chironomidae*, metal sensitive *Chironomidae, Tipulidae,* collector gatherers, and shredders became more apparent. Since 2020, in-stream vegetation has been more abundant and water depth generally lower at SDLT9 compared to the reference creek (Minnow 2021b, 2022, 2023, 2024a). These conditions enhance habitat complexity and food availability, leading to increased populations of *Hydracarina*, *Chironomidae* (including metal-sensitive species), *Tipulidae*, collector-gatherers, and shredders. Vegetation provides refuge, stabilizes sediments, and traps organic matter, supporting detritus-feeding invertebrates, while reduced water flow minimizes displacement and promotes organic matter accumulation. Consistently higher DOC concentrations at SDLT9 relative to reference areas since 2020 (Minnow 2021b, 2022, 2023, 2024a) further indicate increased organic matter input, as DOC primarily originates from decomposing vegetation. Additionally, vegetation improves oxygen availability through daytime photosynthesis (it also contributes to diel fluctuations in DO, with elevated levels during the day and reduced concentrations at night due to continued respiration) and helps mitigate metal stress, creating favorable conditions for metal-sensitive *Chironomidae*. These general patterns since 2020 were accompanied by a concomitant decrease in Simpson's Evenness relative to 2013 (Appendix Table F.30, Appendix Figure F.9).

Although there were clear, ecologically meaningful differences in BIC structure at SDLT9 from about 2020 onward with a concurrent increase in density, relative to baseline conditions (2007 and 2013), the BIC at SDLT9 was comparable to the reference creek in 2024. The temporal changes in the BIC at SDLT9 may be attributed to localized inter-annual variability in habitat conditions (e.g., hydrological changes, changes to organic matter inputs, or flow variability). However, sediment quality analyses from the 2023 CREMP report indicate that fine sediment accumulation is not a major factor affecting physical habitat at SDLT9 (Minnow 2024a). The observed temporal patterns may also be attributed to mine-related factors, such as elevated aqueous concentrations of nitrogen compounds downstream from the activities occurring at the adjacent Dyno facility compared to upstream (see Section 4.2.1). However, it is noteworthy that, despite these potential influences on the BIC at SDLT9 over time, BIC endpoints were comparable to reference in 2024. Therefore, mine-related factors potentially influenced the BIC at SDLT9 in 2024.

4.2.4 Effects Assessment and Recommendations

At SDLT9, the following AEMP benchmarks were exceeded in 2024:

- The aqueous total ammonia concentration exceeded the AEMP benchmark of 0.855 mg/L in fall (1.45 mg/L); and
- The aqueous nitrate concentration exceeded the AEMP benchmark/WQG of 3 mg/L in summer (8.12 mg/L) and fall (7.08 mg/L).

When comparing water quality parameter concentrations to both reference and baseline across all seasons, or within a single season, the following parameters were elevated, suggesting a potential mine-related influence:

- All seasons (spring, summer, fall): nitrate, and TKN;
- Spring: total sodium; and
- Fall: nitrite and total ammonia.

Nitrate concentrations also exceeded the WQG of 3 mg/L in both the summer and fall. The exceedance of the AEMP benchmark for total ammonia and nitrate, along with elevated concentrations relative to baseline and reference in at least one season in 2024 for nitrate, TKN, nitrite, and total ammonia indicate a mine-related influence on these parameters. Visual assessment of temporal data indicated that total ammonia, nitrate, nitrite, and TKN concentrations at SDLT9 were higher than reference and baseline concentrations consistently in the fall and frequently in the summer from 2021 to 2024, however none of the parameter concentrations have demonstrated consistently increasing temporal patterns over the period from

2021 to 2024, suggesting that the influence has not been intensifying over time. Mine-related influences on ammonia and nitrate at SDLT9 were previously identified in annual CREMP monitoring in 2022 and 2023 (Minnow 2023 and 2024a). In response, a special investigation was completed to determine the source(s) of nitrogen compounds to SDLT9 through an expanded spatial water quality sampling program in the fall of 2024 (see Appendix I). The investigation identified activities occurring at the Dyno facility, located adjacent to SDLT9, as the likely source, leading to recommendations for mitigation and further monitoring.

Despite elevated concentrations compared to baseline and reference in spring 2024, no mine-related influence is indicated for total sodium, as no consistent directional patterns in concentration have been observed and concentrations in the summer and fall were within the range observed in reference areas.

Sediment samples, collected every three years from streams monitored under the CREMP, were taken in 2023, therefore, sediment quality results are not included in this report. No adverse mine-related effects on chlorophyll-a (a measure of primary productivity) were observed at SDLT9 in 2024. There were ecologically meaningful differences in BIC structure at SDLT9 in 2024 compared to baseline, however the BIC was comparable to the reference creek. Though localized natural inter-annual variability in habitat conditions may account for changes in the BIC relative to baseline, mine-related influences on water quality at SDLT9 in 2024 also suggest the potential for a mine-related effect.

In accordance with the AEMP Management Response Framework, a Moderate Action Response is required based on determination of mine-related influence on total aqueous nitrate, TKN, nitrate, and ammonia concentrations and nitrate and ammonia concentrations above AEMP Benchmarks at SDLT9 in 2024 (Figure 2.6). The following actions are recommended:

- An activity audit concerning the transportation, storage, and handling of ammonium nitrate at the Dyno facility is being implemented, along with potential additional water sampling during the open water season in 2025, as needed, to help identify point source(s) of aqueous nitrogen compounds. Mitigation measures will be developed based on the findings.
- Water quality monitoring at SDLT9 will continue in the 2025 CREMP to assess the effectiveness of mitigation efforts at the Dyno facility in reducing the concentrations of aqueous nitrogen compounds. This monitoring may be supplemented by expanded spatial sampling in the fall of 2025 if necessary to fully evaluate mitigation effectiveness.

The absence of confirmed mine-related influences on phytoplankton (as a measure of primary productivity) and the BIC, means no further management response is required for these monitoring components at SDLT9 in 2024 (Figure 2.6). However, if observed changes in the BIC community structure at SDLT9 (identified as potentially mine-related) are associated with mine-related influences on aqueous concentrations of nitrogen compounds, it is anticipated that recommended actions associated with water quality will also serve to appropriately mitigate effects on the BIC.

Comparison to FEIS Predictions

A comparison of water quality at SDLT9 in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-4 (Explosives), SWSQ-7 (Camp Management), and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at SDLT9 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

4.3 Sheardown Lake Tributary 12 (SDLT12)

4.3.1 Water Quality

4.3.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at SDLT12 (Station LDFG-OUT) concurrent with water quality sampling in spring only (Figure 2.1) [31]. During the spring sampling event, DO levels at SDLT12 were similar to those of the reference streams and near saturation (> 93%; > 11.8 mg/L; Appendix Figure C.10, Appendix Table C.1). *In situ* pH was slightly higher than observed at the reference stream during the spring sampling event (Appendix Figure C.10, Appendix Table C.1). Despite minor differences in pH of water at SDLT12 compared to the reference stream, pH was consistently within WQG limits (Appendix Figure C.10). *In situ* specific conductance was higher at SDLT12 (76.6 µS/cm) compared to the reference streams (range 12.9 to 37.9 µS/cm) during spring monitoring events (Appendix Figure C.10; Appendix Table C.1).

4.3.1.2 Water Chemistry

Water chemistry at SDLT12 was assessed only during the spring sampling event in 2024 due to the natural absence of surface flow at this tributary in the summer and fall. During this time, all parameter concentrations were below AEMP benchmarks and WQGs (Table 4.1; Appendix Table C.33). Water chemistry parameters with concentrations that were slightly (3 to 5 times), moderately (5 to 10 times), or highly (≥ 10 times) elevated relative to reference or

[31] Station LDFG-OUT (SDLT12) was not flowing during the summer and fall water quality sampling events nor during the August BIC sampling event in 2024.

baseline conditions are identified in Appendix Tables C.34 and C.36. When compared to reference stream stations and baseline concentrations in the spring, multiple parameters including conductivity, hardness, TDS, alkalinity, total chloride and sulphate, total and dissolved barium, calcium, magnesium, molybdenum, potassium, sodium, strontium, and uranium, and dissolved manganese were slightly, moderately, or highly elevated (Appendix Tables C.33 to C.36, Appendix Figure C.11).

A potential mine-related influence was identified for sulphate concentrations at SDLT12 in the 2023 CREMP, triggering a Low Action Response (Minnow 2024a). Temporal trend analysis was conducted in response and found no significant increasing trends since the baseline period (2005 to 2024) or over the mine operational period for which data are available (2017 to 2024) in the spring and concentrations in 2024 were slightly lower than those observed in 2023 (Appendix Figure C.11, Appendix Tables H.5 and H.6). Visual assessment of temporal data indicates that spring concentrations of TDS, conductivity, hardness, and total alkalinity, chloride, barium, calcium, molybdenum, potassium, sodium, and uranium increased from 2021 to 2023 at SDLT12, with 2024 concentrations that were comparable to or lower than those recorded in 2023 (Appendix Figure C.11). In contrast, total magnesium and strontium have increased consistently in the spring from 2021 to 2024 (Appendix Figure C.11).

Mine site infrastructure is located in the SDLT12 catchment resulting in the potential for mine-related influences from non-point source and airborne emissions (Baffinland 2012) and management plans are in place to manage and mitigate influences in the SDLT12 catchment associated with site water management, waste management (i.e., the landfill), and dust deposition. Specifically, potential sources of mine-related influences in the SDLT12 catchment include the mine haul road and snow stockpiling activities which could contribute to runoff effects as well as general site dust deposition (Figure 2.1). However, implementation of concentrated dust suppression efforts at the crusher and along the haul road (e.g., increased road-watering with recycled water) in 2024 may have contributed to trend stabilization and reductions in concentration observed in 2024 compared to 2023 for water quality parameters with mine-related influence.

In general, water chemistry data for SDLT12 are relatively limited given that annual CREMP sampling at SDLT12 was only initiated in 2021 and the tributary is frequently dry during summer and fall sampling events. While evidence of increasing temporal patterns since 2021 and elevation relative to baseline and reference conditions for multiple water chemistry parameters suggest a mine-related influence on water quality at SDLT12, these results should be considered in the context of limited temporal and seasonal data availability and verified with ongoing annual monitoring. Further, concentrations of all parameters remained below AEMP benchmarks and WQGs in 2024 and therefore no adverse effects to biota are expected.

4.3.2 Phytoplankton

A single chlorophyll-a sample was collected at Station LDFG-OUT[32] in SDLT12 in the spring of 2024. The chlorophyll-a concentration at SDLT12 in spring 2024 was higher than observed at reference streams and all other Sheardown Lake Tributaries in 2024, but within the seasonal spring ranges at reference streams and other Sheardown Lake Tributaries in earlier mine operations years (Figure 4.2, Appendix Table E.1). Chlorophyll-a concentrations at SDLT12 remained well below the AEMP benchmark of 3.7 μg/L (Figure 4.2, Appendix Table E.1). Measured chlorophyll-a concentrations indicate low phytoplankton productivity and oligotrophic conditions based on chlorophyll-a concentrations (i.e., <8 μg/L; Dodds et al. (1998); Appendix Table E.1) and total phosphorus concentrations (i.e., <10 μg/L; CCME 2024b; Table 4.1, Appendix Table C.33; see Section 3.1.2 for additional trophic status classification details).

As with SDLT9, SDLT12 was added to the CREMP as a monitoring station in fall 2021; therefore, no baseline data or early mine-operational data are available for this station. This precludes direct comparisons of chlorophyll-a concentrations to conditions before mine construction or during the first seven years of mine operation. Additionally, data collected at SDLT12 since 2021 are limited due to ephemeral conditions preventing sampling to support evaluation of temporal patterns (Figure 4.3).

Overall, although the chlorophyll-a concentration at SDLT12 was slightly higher than those observed in reference streams and other Sheardown Lake Tributaries in spring 2024, based on qualitative comparisons, the concentration was within the previously observed reference range, the stream was classified as oligotrophic, and the single measured chlorophyll-a concentration was well below the AEMP benchmark. These results suggest no adverse mine-related effects on phytoplankton productivity at SDLT12 in 2024.

4.3.3 Effects Assessment and Recommendations

In 2024, water chemistry at SDLT12 (Station LDFG-OUT) met all AEMP benchmarks and WQGs during the spring sampling event, which was the only season for which data was collected due to the natural absence of surface flow at this tributary in the summer and fall. The following

[32] Station LDFG-OUT (SDLT12) was dry during the summer and fall sampling events in 2024; therefore, no data are available for these seasons.

parameters had concentrations that were elevated compared to reference and baseline, suggesting a potential mine-related influence:

- Spring: alkalinity, total/dissolved barium, total/dissolved calcium, chloride, conductivity, hardness, total/dissolved magnesium, total/dissolved molybdenum, total/dissolved potassium, total/dissolved sodium, total/dissolved strontium, sulphate, TDS, and total/dissolved uranium.

A potential mine-related influence on sulphate was previously identified in 2023 (Minnow 2024a), but temporal trend analyses conducted in 2024 found no significant trends since the baseline period (2005 to 2024) or over the mine operational period (2015 to 2024) suggesting no mine-related influence. Visual assessment of temporal data (total concentrations only) indicated increasing patterns and therefore mine-related influence for each of the other parameters with concentrations that were elevated compared to baseline and reference in spring 2024 (i.e., alkalinity, barium, calcium, chloride, conductivity, hardness, magnesium, molybdenum, potassium, sodium, strontium, TDS, and uranium). However, no parameter concentrations exceeded AEMP benchmarks of WQGs, suggesting that water quality remains within established criteria for protecting aquatic life.

Sediment samples were collected in 2023 as part of the CREMP and will be reported again in 2026; therefore, sediment quality results are not included in this report. No adverse mine-related effects on chlorophyll-a (a measure of primary productivity) were observed at SDLT12 in 2024.

According to the AEMP Management Response Framework, the presence of potential mine-related changes in water chemistry concentrations as observed at SDLT12 in 2024, requires a Low Action Response (Figure 2.6). The following actions are recommended:

- In 2025, temporal trend analysis of aqueous alkalinity, conductivity, hardness, and TDS, and total and dissolved (where applicable) concentrations of barium, calcium, chloride, magnesium, molybdenum, potassium, sodium, strontium, and uranium will be conducted for SDLT12 to further investigate temporal patterns. In 2025, an analysis of total compared to dissolved aqueous concentrations of barium, calcium, magnesium, molybdenum, potassium, sodium, strontium, and uranium will be completed to investigate biological availability and further determine potential for effects on aquatic biota.
- Potential sources of the water chemistry parameters at SDLT12 for which mine-related influence was indicated in 2024 will be investigated to better define mine-related influence and the potential for continued contributions.

The absence of confirmed mine-related influences on phytoplankton (as a measure of primary productivity) means no further management response is required for this monitoring component at SDLT12 in 2024 (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at SDLT12 in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-7 (Camp Management), and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at SDLT12 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

4.4 Sheardown Lake Northwest (DL0-01)

4.4.1 Water Quality

4.4.1.1 *In Situ* Water Quality

In 2024, profiles were developed from *in situ* water quality measured concurrent with water quality sampling in winter, summer, and fall (Figure 2.1), and *in situ* water quality was measured at the top and bottom of the water column concurrent with benthic invertebrate community sampling in August (Figure 2.3). Water quality profiles of *in situ* water temperature, dissolved oxygen, pH, and specific conductance measured at Sheardown Lake NW in 2024 showed no substantial station-to-station differences during any of the winter, summer, or fall sampling events (Appendix Figures C.12 to C.15, Appendix Tables C.37 to C.39). During August 2024 BIC sampling, the bottom temperature and pH differed significantly between littoral and profundal stations within the lake but these small magnitude differences were expected based on known depth-related patterns of *in situ* water quality measures (Appendix Tables C.40 and C.41).

In winter 2024, the mean water temperature of Sheardown Lake NW increased by approximately 1°C from the surface to the bottom, while in summer, the mean temperature decreased by < 1°C from the surface to the bottom (Figure 4.4, Appendix Figure C.12). These temperature changes were linear and occurred without the development of a pronounced thermocline, which was similar to the temperature profile at Reference Lake 3 during summer in 2024 (Figure 4.4). Mean temperature during fall sampling at both Sheardown Lake NW and Reference Lake 3 was uniform across all depths (Figure 4.4, Appendix Figure C.12). The mean water temperatures at the bottom of the water column at Sheardown Lake NW littoral and profundal stations was slightly (~1 to 1.5°C) warmer than Reference Lake 3 for like-habitats during the August 2024 BIC sampling (Figure 4.5, Appendix Table C.42).

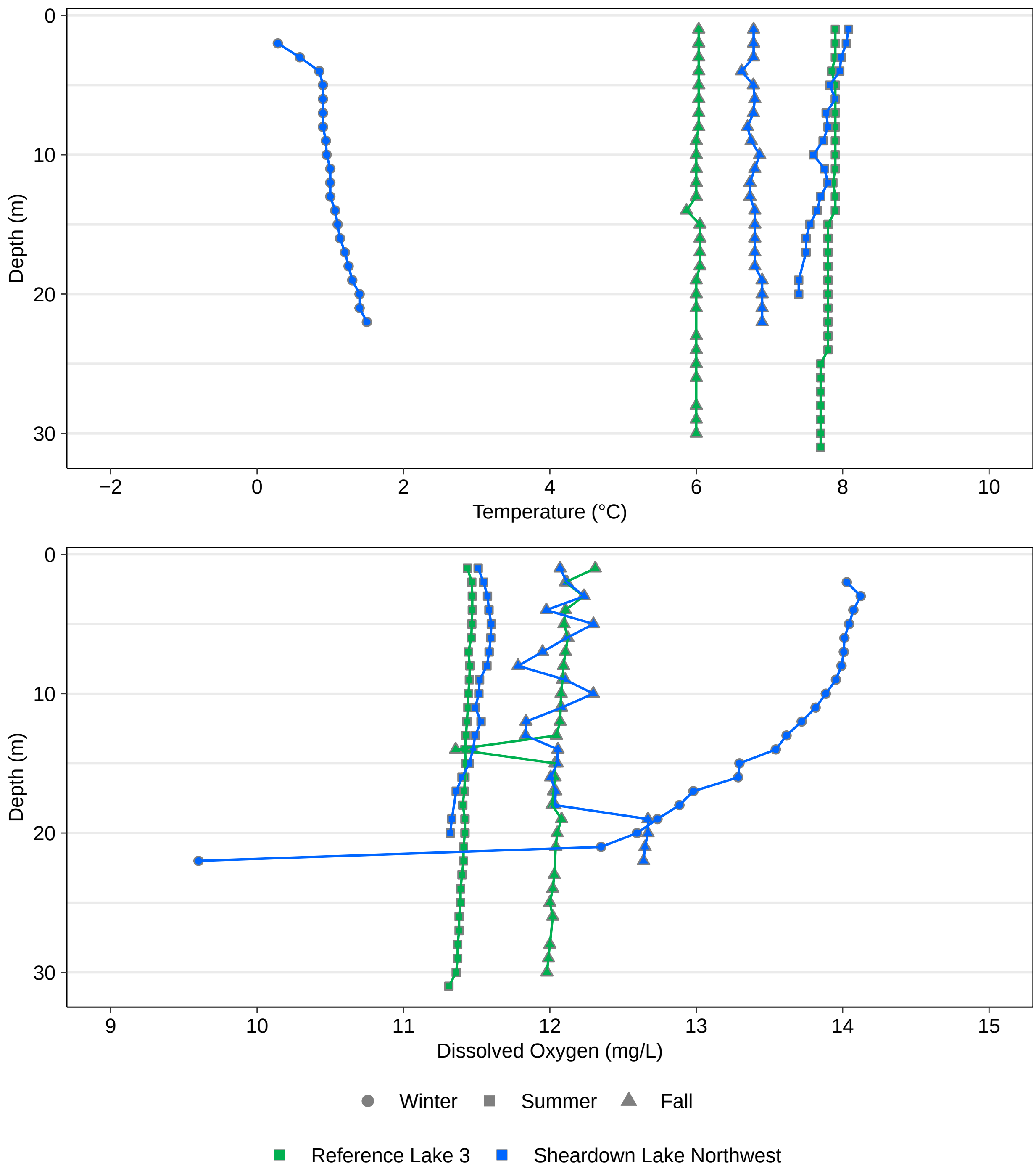
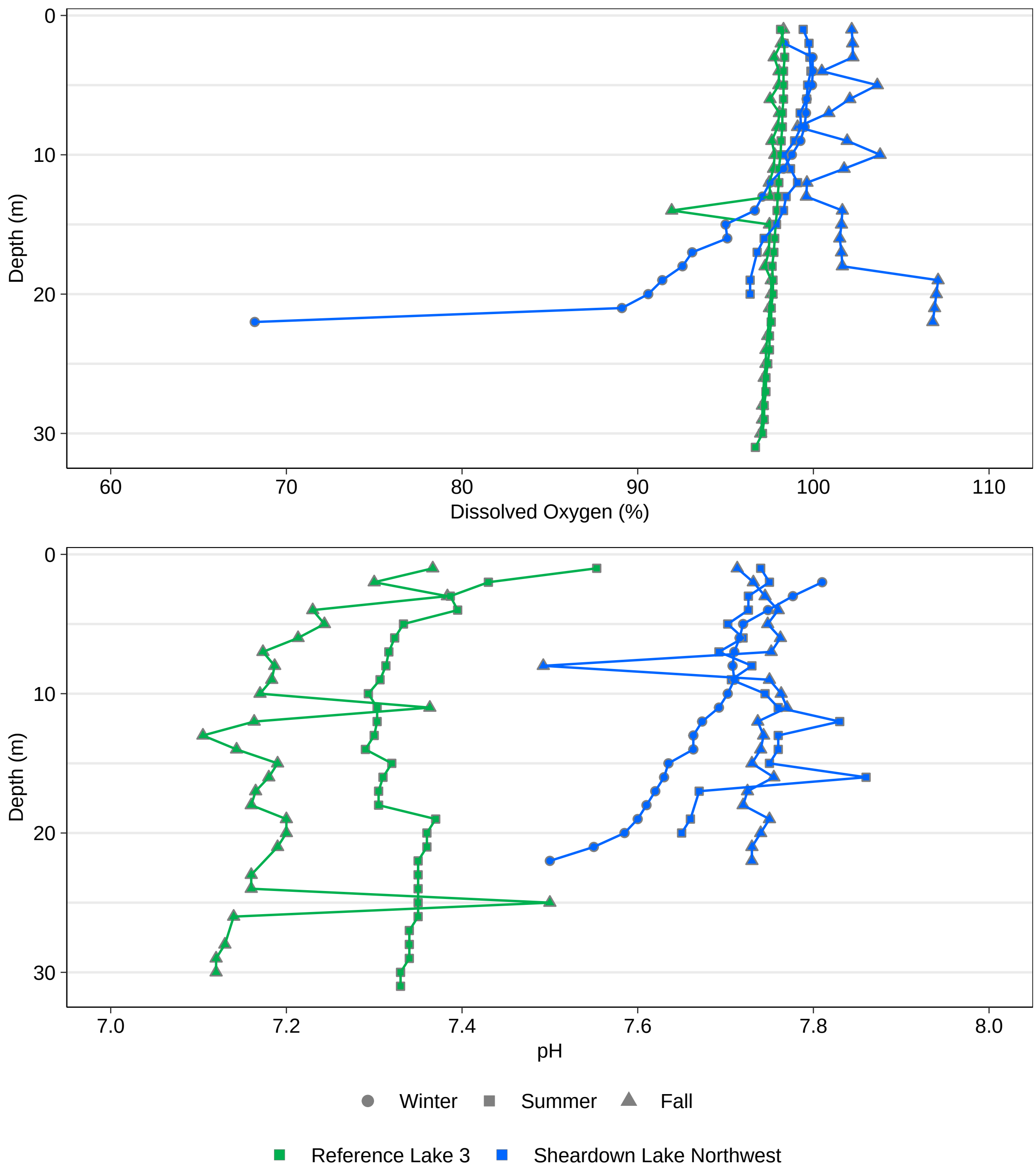


Figure 4.4: Average In Situ Water Quality with Depth from Surface at Sheardown Lake Northwest Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024



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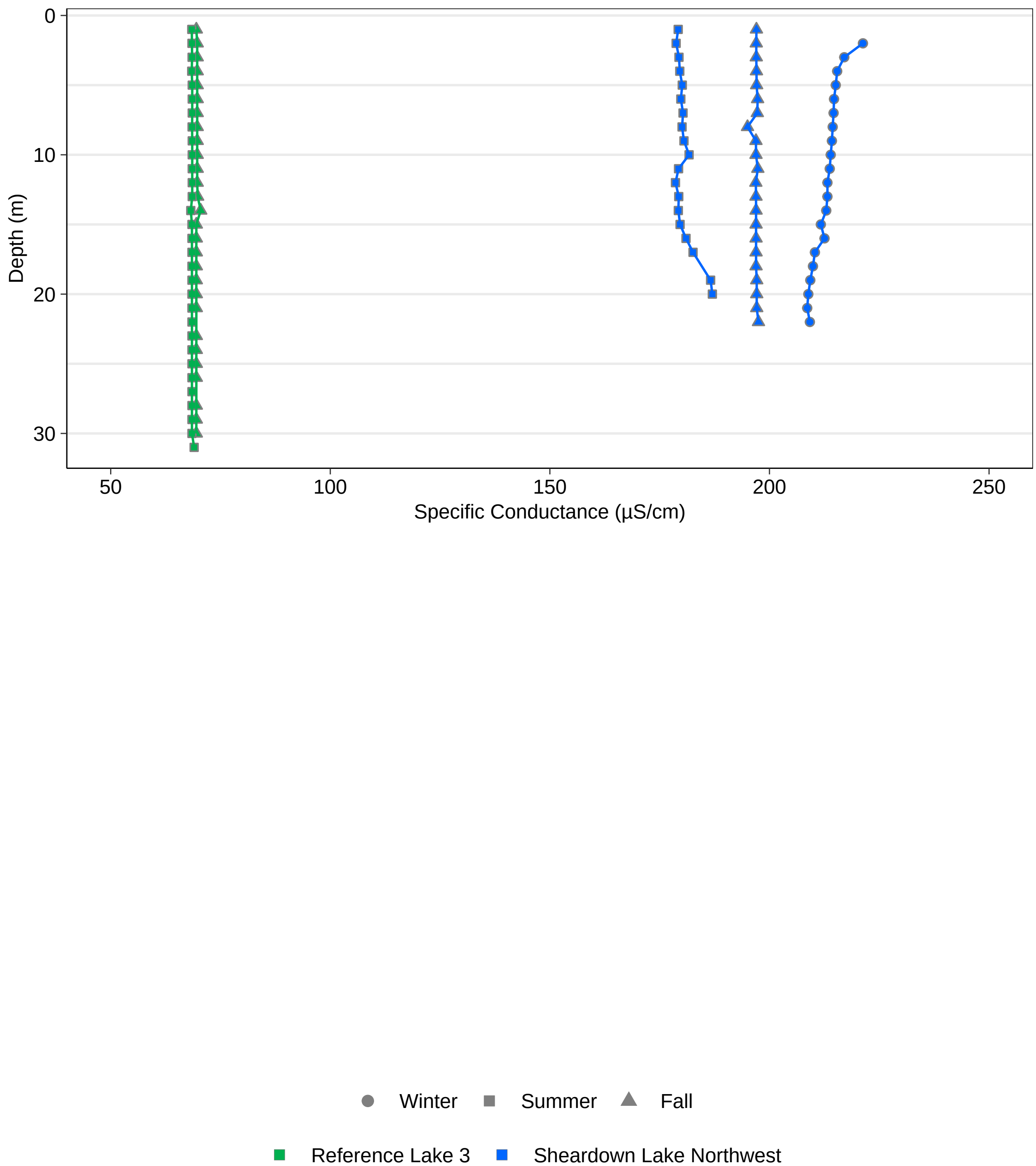

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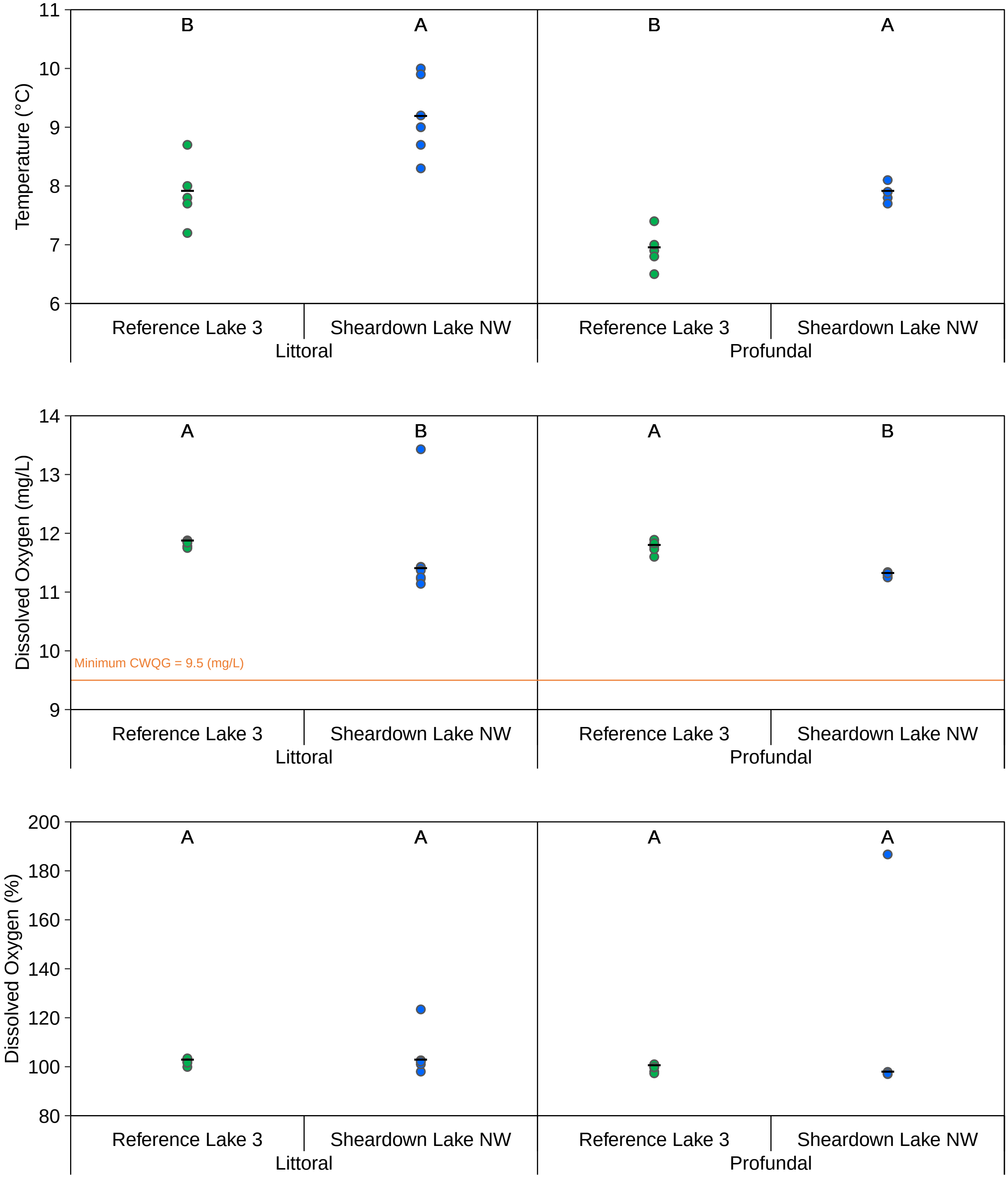


Figure 4.5: Comparison of *In Situ* Water Quality Measured at Sheardown Lake Northwest (NW; DL0−01) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

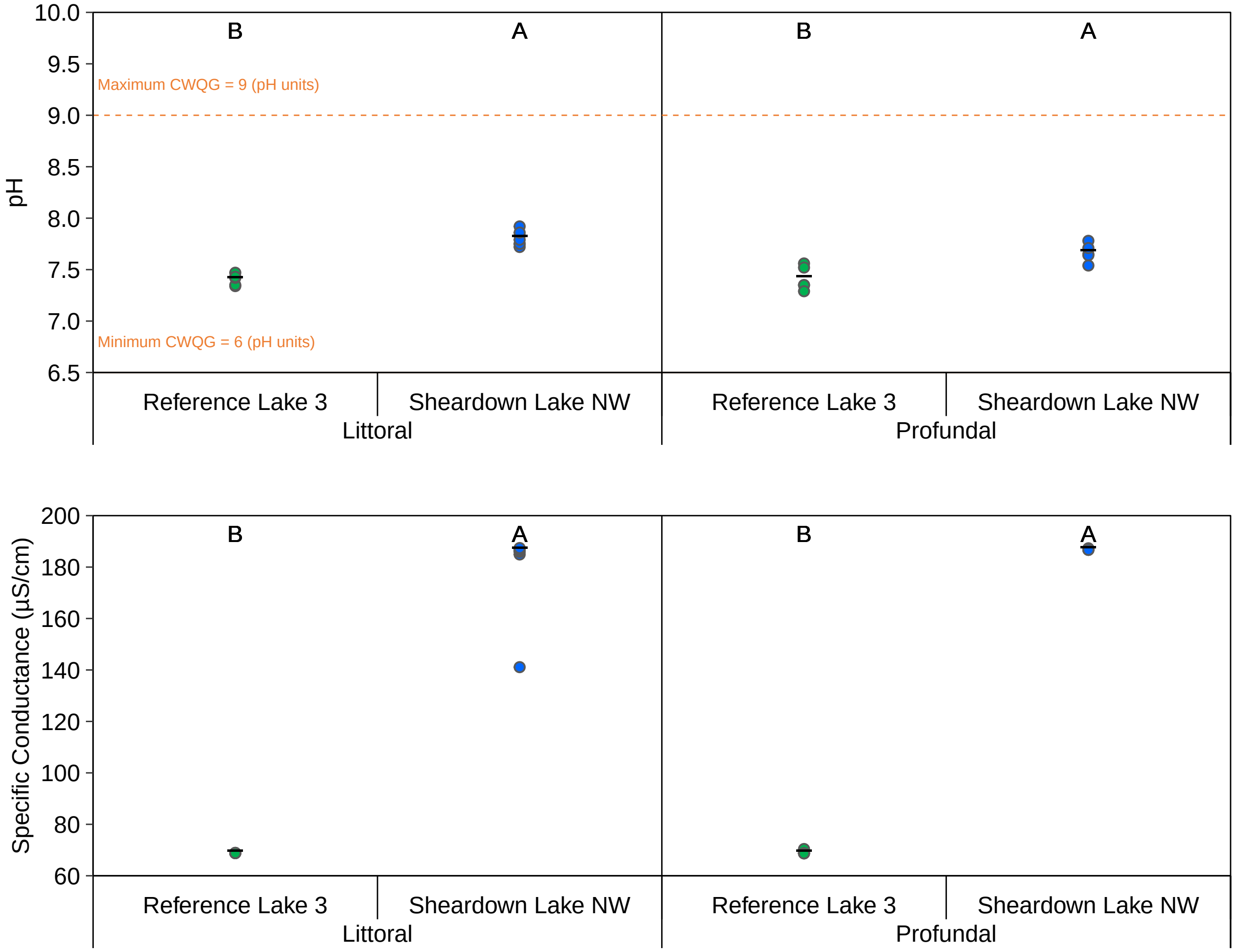


Figure 4.5: Comparison of *In Situ* Water Quality Measured at Sheardown Lake Northwest (NW; DL0−01) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

Dissolved oxygen profiles measured at Sheardown Lake NW in winter and summer 2024 showed decreasing saturation with depth while Reference Lake 3 showed a small decrease in oxygen saturation with increasing depth in summer (Figure 4.4). In the fall of 2024, Sheardown Lake NW and Reference Lake 3 showed relatively consistent mean oxygen saturation with depth (Figure 4.4). Reference Lake 3 showed a small decrease in oxygen saturation with increasing depth in summer (Figure 4.4). Compared to like-habitat stations at Reference Lake 3 during the August 2024 BIC sampling (Appendix Table C.39), dissolved oxygen concentrations were significantly lower at Sheardown Lake NW at both profundal and littoral stations than reference; however, the mean difference was small (0.2 to 0.5 mg/L) whereas no significant differences in dissolved oxygen saturation were indicated between lakes at either depth (Figure 4.5, Appendix Tables C.40 and C.42). Mean dissolved oxygen concentrations were well above the WQG of 9.5 mg/L (lowest acceptable concentration for early life stages of cold-water biota) near the bottom at littoral and profundal stations of Sheardown Lake NW and Reference Lake 3 during BIC sampling in August 2024, suggesting no ecologically meaningful differences in dissolved oxygen between lakes (Figure 4.5, Appendix Table C.40).

In 2024, water column profiles (mean measures) at Sheardown Lake NW showed a slight decrease in pH with depth in the winter and summer (~ 0.3 pH units), while values were consistent with depth during the fall (Figure 4.4). Comparatively, at Reference Lake 3, water column profiles showed a decrease in pH with depth (~ 0.5 pH units) in the summer and fall (Figure 4.4). The pH near the bottom at both littoral and profundal stations of Sheardown Lake NW was significantly higher (i.e., more alkaline) than at like-habitat for the reference lake during the August 2024 BIC sampling (Figure 4.5; Appendix Table C.42). However, pH values were consistently within WQG at Sheardown Lake NW and Reference Lake 3 (Figure 4.5, Appendix Table C.40 and C.42).

Mean specific conductance profiles at Sheardown Lake NW showed no substantial changes with depth in summer or fall of 2024, which was similar to Reference Lake 3 in both seasons (Figure 4.4). Similarly, beginning at depths of approximately 2 m below the ice, specific conductance profiles at Sheardown Lake NW in winter 2024 showed no marked change with depth (Figure 4.4). Specific conductance was higher at Sheardown Lake NW in winter compared to other seasons (Figure 4.4), likely due to an absence of dilution originating from tributaries due to their complete freezing. During spring, summer, and fall sampling events, specific conductance was higher at Sheardown Lake NW compared to the reference lake (Figure 4.4). Similarly, during the August 2024 BIC sampling, specific conductance near the bottom of the water column at littoral and profundal stations was significantly higher at Sheardown Lake NW than at like-habitats at the reference lake (Figure 4.5, Appendix Tables C.40 and C.42). Specific conductance at SDLT1 was elevated relative to reference streams in 2024 (see Section 4.1.1.1) but no spatial gradient was evident across Sheardown Lake NW water quality sampling stations (Appendix Figure C.15, Appendix Tables C.37 to C.39) suggesting limited influence of SDLT1 inflow as a potential source of elevated specific conductance in Sheardown Lake NW. Secchi depth readings, which serve as a proxy for water clarity, were significantly lower at Sheardown Lake NW than at Reference Lake 3 during the August 2024 BIC (Appendix Figure C.8; Appendix Table C.42) indicating more suspended particulate material in waters of Sheardown Lake NW.

4.4.1.2 Water Chemistry

Mean water chemistry parameter concentrations at Sheardown Lake NW met all AEMP benchmarks and WQGs during spring, summer, and fall sampling events in 2024 (Table 4.3; Appendix Table C.43). Only one individual sample (Station DL0-01-5 surface water sample in the winter) had a copper concentration (0.00215 mg/L) that was marginally above the WQG (0.002 mg/L; Appendix Table C.43). Parameters with total and dissolved concentrations that were slightly (3 to 5 times higher), moderately (5 to 10 times higher), or highly (≥ 10 times higher) elevated relative to reference or baseline concentrations are identified in Appendix Tables C.44 and C.46. Concentrations of nitrate, sulphate, and total and dissolved uranium were consistently elevated across all seasons compared to Reference Lake 3 and baseline concentrations in 2024 (Appendix Tables C.44 and C.46, Appendix Figure C.16). In the summer, chloride, and total and dissolved manganese concentrations were also slightly, moderately, or highly elevated relative to reference and baseline concentrations (Appendix Tables C.44 and C.46, Appendix Figure C.16). In the summer and fall, total and dissolved strontium concentrations were slightly elevated compared to both Reference Lake 3 and baseline concentrations (Appendix Tables C.44 and C.46, Appendix Figure C.16).

In 2022, aqueous concentrations of nitrate, chloride, sulphate, and total and/or dissolved molybdenum and uranium were elevated at Sheardown Lake NW compared to reference and/or baseline conditions in at least one seasonal sampling event (Minnow 2023). Under the AEMP Management Response Framework, a Low Action Response was recommended in the form of a temporal trend analysis for these parameters (Minnow 2023) and was completed in 2023 (Minnow 2024a). Significant increasing trends were identified for nitrate, chloride, sulphate, and total and dissolved molybdenum and uranium at all Sheardown Lake NW water quality stations since the baseline period (2007 to 2023) and over the mine operation period (2015 to 2023; Minnow 2024a). Visual assessment of temporal data indicated that these increasing trends generally started in 2018 or 2019 and persisted in 2024 (Appendix Figure C.16), including for total molybdenum, despite concentrations that were only elevated compared to

Table 4.3: Mean Water Chemistry at Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

Parameters		Units	Water Quality Guideline (WQG)[b,c]	AEMP Benchmark[d]	Reference Lake 3 (n = 3)		Sheardown Lake NW Stations (n = 5)		
					Summer	Fall	Winter	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	72.5	72.0	220	190	201
	pH (lab)	pH	6.5 - 9.0	-	7.51	7.50	7.59	7.88	7.91
	Hardness (as $CaCO_3$)	mg/L	-	-	34.8	35.3	100	86.8	90.7
	Total Suspended Solids (TSS)	mg/L	-	-	<1	3.30	<1	1.02	<1
	Total Dissolved Solids (TDS)	mg/L	-	-	51.5	41.2	115	95.2	99.8
	Turbidity	NTU	-	-	0.323	0.267	0.103	1.17	0.364
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31.4	36.1	73.5	56.3	58.4
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00738	0.00837	0.00693	0.00669	0.0147
	Nitrate	mg/L	3	3	<0.02	<0.02	0.432	0.466	0.566
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.191	0.145	0.131	0.143	0.159
	Dissolved Organic Carbon	mg/L	-	-	3.62	3.44	1.85	2.06	1.98
	Total Organic Carbon	mg/L	-	-	3.01	3.51	2.05	1.96	2.10
	Total Phosphorus	mg/L	0.020^{α}	-	0.00467	0.00262	0.00272	0.00272	0.00302
	Phenols	mg/L	0.004^{α}	-	<0.001	0.00152	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	1.21	1.21	8.30	7.23	7.37
	Sulphate (SO_4)	mg/L	218^{β}	218	2.72	2.63	22.9	22.4	25.0
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[e]	0.0158	0.00605	0.00785	0.0289	0.0116
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.000117	<0.0001	0.000105	<0.0001	<0.0001
	Barium (Ba)	mg/L	1^{β}	-	0.00614	0.00598	0.0115	0.00970	0.00958
	Beryllium (Be)	mg/L	0.011^{α}	-	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	0.0218	0.0173	0.0175
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000005	<0.000005	0.00000648	0.00000811	0.00000929
	Calcium (Ca)	mg/L	-	-	6.49	6.40	19.4	15.7	16.6
	Chromium (Cr)	mg/L	0.001	0.003	<0.0005	<0.0005	<0.0005	<0.0005	0.000508
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	<0.0001	<0.0001	<0.0001	0.000101	<0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.000848	0.000823	0.00121	0.000988	0.000853
	Iron (Fe)	mg/L	0.30	0.300	0.0337	0.0112	0.0190	0.0355	0.0122
	Lead (Pb)	mg/L	0.001	0.001	0.0000528	<0.00005	<0.00005	0.0000510	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	0.00265	0.00174	0.00208
	Magnesium (Mg)	mg/L	-	-	4.26	4.48	13.0	11.0	11.9
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00136	0.000602	0.000339	0.00619	0.00465
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000139	0.000144	0.00208	0.00164	0.00173
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.00114	0.000637	0.000695
	Potassium (K)	mg/L	-	-	0.888	0.831	2.26	2.02	2.01
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	0.0000562	0.0000671	0.0000812
	Silicon (Si)	mg/L	-	-	0.487	0.425	0.790	0.641	0.554
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.875	0.843	2.66	2.23	2.42
	Strontium (Sr)	mg/L	-	-	0.00783	0.00754	0.0262	0.0238	0.0270
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.000947	0.000308	0.000340	0.00149	<0.0003
	Uranium (U)	mg/L	0.015	-	0.000273	0.000260	0.00301	0.00224	0.00252
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02^{α}	0.030	<0.003	<0.003	<0.003	<0.003	<0.003

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

BOLD Indicates parameter concentration above the AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

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

[b] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[c] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

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

Reference Lake 3 (not relative to baseline) in 2024. In all cases except for sulphate, the rate of increase also appears have been greater since 2022 (Appendix Figure C.16).

Total strontium concentration has consistently increased during the mine operations period in Sheardown Lake NW, starting in 2019, though no consistent increasing or decreasing patterns in manganese concentrations were observed, with concentrations remaining well below the WQG (Appendix Tables C.43, Appendix Figure C.16).

Based on the identification of significant increasing trends in total and dissolved molybdenum and uranium in Sheardown Lake NW in the 2023 temporal trend analyses, and the resulting determination of a mine-related influence, further response was recommended under the AEMP Management Response Framework (Minnow 2024a). This response included analysis of total compared to dissolved aqueous concentrations of molybdenum and uranium to investigate biological availability and further determine potential for effects on aquatic biota. Exploratory analyses were conducted by creating scatterplots of total compared to dissolved concentrations measured since the baseline period (starting in 2006) to identify any temporal or seasonal changes in the relative concentrations of the two fractions that may suggest greater bioavailability or increased risk of potential effects to biota (see Section 2.2.3.2.2). For both molybdenum and uranium, a strong relationship between total and dissolved concentrations was observed from 2006 to 2024 (Appendix Figure H.4). The dissolved phase of these parameters is the form most readily available for uptake by aquatic organisms. The special investigation indicated that despite increasing total and dissolved concentrations of molybdenum and uranium in Sheardown Lake NW due to mine-related influences, the total:dissolved concentration ratios have not changed over time, meaning mine-related processes contributing to these metal concentrations are not altering the proportion of the bioavailable fraction. Notably, based on a total:dissolved concentration ratio near 1:1 for both molybdenum and uranium since the baseline period, nearly all of their total concentrations were in dissolved form rather than bound to suspended particles. A 1:1 total:dissolved ratio for molybdenum and uranium is commonly found in oxygenated waters with low turbidity and low organic matter, but this is not universally the case (Zhang et al. 2023, Adams et al. 2019). Geochemical factors such as alkaline pH, low turbidity, and oxidizing conditions, can enhance the solubility and mobility of metals such as molybdenum and uranium (Zhang et al. 2023, Adams et al. 2019). At Sheardown Lake NW, pH levels do not indicate alkaline conditions (Appendix Table C.43). Therefore, the observed near 1:1 total:dissolved concentration ratio for molybdenum and uranium is likely driven by the consistently low turbidity across all monitoring stations and the presence of oxidizing conditions in the area (i.e., low DOC suggests oxidizing conditions where organic carbon is efficiently broken down; Zhang et al. 2023, Adams et al. 2019; Appendix Table C.43).

Dissolved metals are typically more biologically relevant in aquatic environments than particulate-bound forms, as they can be directly absorbed by aquatic organisms through gill membranes, ingestion, or passive diffusion, whereas particulate-bound metals are generally less bioavailable unless environmental conditions promote their release (Adams et al. 2019). A high proportion of dissolved metals within the total concentration suggests potential for biological uptake and toxicity but, factors such as pH, hardness, alkalinity, dissolved organic matter, and phosphorus also modify the bioavailability and toxicity of molybdenum and uranium (Wood et al. 2011). Though they have increased since the baseline period and over the mine operations period, concentrations of both total molybdenum and uranium at Sheardown Lake NW have remained more than an order of magnitude below their respective WQGs since baseline monitoring began (Appendix Figure C.16), suggesting limited risk of adverse effects on aquatic biota.

While concentrations of nitrate, chloride, sulphate, and total molybdenum, and uranium remained below applicable AEMP benchmarks and WQGs in Sheardown Lake NW in 2024, observed temporal patterns and/or elevated concentrations relative to reference and baseline concentrations indicate an ongoing mine-related influence on these parameters, as well as for total strontium for which there is no AEMP benchmark or WQG. Most of the mine site infrastructure is located within the Sheardown Lake System catchment, resulting in the potential for mine-related influences from non-point source and airborne emissions (Baffinland 2012) and management plans are in place to manage and mitigate influences in the Sheardown Lake NW catchment associated with site water management, laydown areas, camps, waste management (including the landfill), and dust deposition. Increasing concentrations of nitrate, chloride, sulphate, and total molybdenum, and uranium in Sheardown Lake NW may be associated with inflows from SDLT1 and/or SDLT12. Concentrations of all parameters determined to be influenced by the mine in Sheardown Lake NW also indicate mine-related influence in SDLT1, the catchment of which contains mine site infrastructure, including the Deposit 1 pit and subsequent drainage from Deposit 1 as well as more recently, the KM 105 Pond (see Section 4.1.1). At SDLT12, concentrations of total chloride, molybdenum, and uranium in 2024 suggested potential mine-related influences during the spring[33] (see Section 4.3.1). Furthermore, the landfill, located within the Sheardown Lake NW catchment could contribute to runoff or groundwater influences Sheardown Lake NW as could general site dust deposition. While additional investigation is needed to identify other potential sources of these

[33] SDLT12 has typically been dry in the summer and/or fall since the initiation of CREMP monitoring in that tributary in 2021.

elevated parameters[34], water chemistry in Sheardown Lake NW met AEMP benchmarks and WQG in 2024, indicating limited risk of adverse effects on aquatic biota related to water quality.

4.4.2 Sediment Quality

Surficial sediments (i.e., the top 2 cm) collected from coring stations in Sheardown Lake NW in 2024 were primarily composed of silt or silt-sand (Figure 4.6) and were described as red-brown to yellow-red silt (Figure 4.6; Appendix Table D.11). A redox layer was observed in one core collected at station DL0-01-10 (Appendix Table D.11). Surficial sediments that were collected from Sheardown Lake NW using a Petite Ponar and to specifically support interpretation of the BIC data were described as brown-gray silts and clays (Appendix Table D.10).

The littoral and profundal sediment sampling (coring) stations at Sheardown Lake NW had higher proportions of silt and lower proportions of sand relative to Reference Lake 3, whereas proportions of clay in the littoral and profundal sediment samples were similar between lakes (Appendix Table D.12). Overall, the littoral and profundal stations of Sheardown Lake NW had significantly finer substrate compared to littoral and profundal stations of Reference Lake 3 (Appendix Table D.12). Additionally, profundal sediments in Sheardown Lake NW had significantly lower TOC content compared to Reference Lake 3 (Figure 4.6; Appendix Table D.12). The sediment samples collected using a Petite Ponar and to support interpretation of BIC data were composed of similar particle size fractions and TOC content as the core samples and indicate that samples collected at differing depths and using different methodologies were consistent for these variables (Appendix Table D.4).

Some spatial differences in sediment chemistry were identified in Sheardown Lake NW in 2024, based on concentrations of metals in sediment located nearest to the inflows from SDLT1 and SDLT12 (i.e., Stations DD-HAB-9-STN2 and DL0-01-9, respectively) compared to the station located near the Sheardown Lake NW outflow (i.e., Station DL0-01-10; Appendix Table D.13). For example, in Sheardown Lake NW, arsenic and iron concentrations were highest in the sediment samples from DD-HAB-9-STN2 and DL0-01-9 in 2024 and were lower at the outflow station (DL0-01-10; Appendix Table D.13). Sediment copper and nickel concentrations were highest near the SDLT12 inflow (i.e., at DL0-01-9), where the highest proportions of TOC were also observed (Appendix Table D.13). The spatial distribution of concentrations of these metals and TOC in Sheardown Lake NW suggests possible localized influences of tributaries and association of metals with TOC.

[34] Investigation of potential sources of elevated parameters in Sheardown Lake NW will include consideration of dustfall (EDI 2024) and groundwater (WSP 2024) influences.

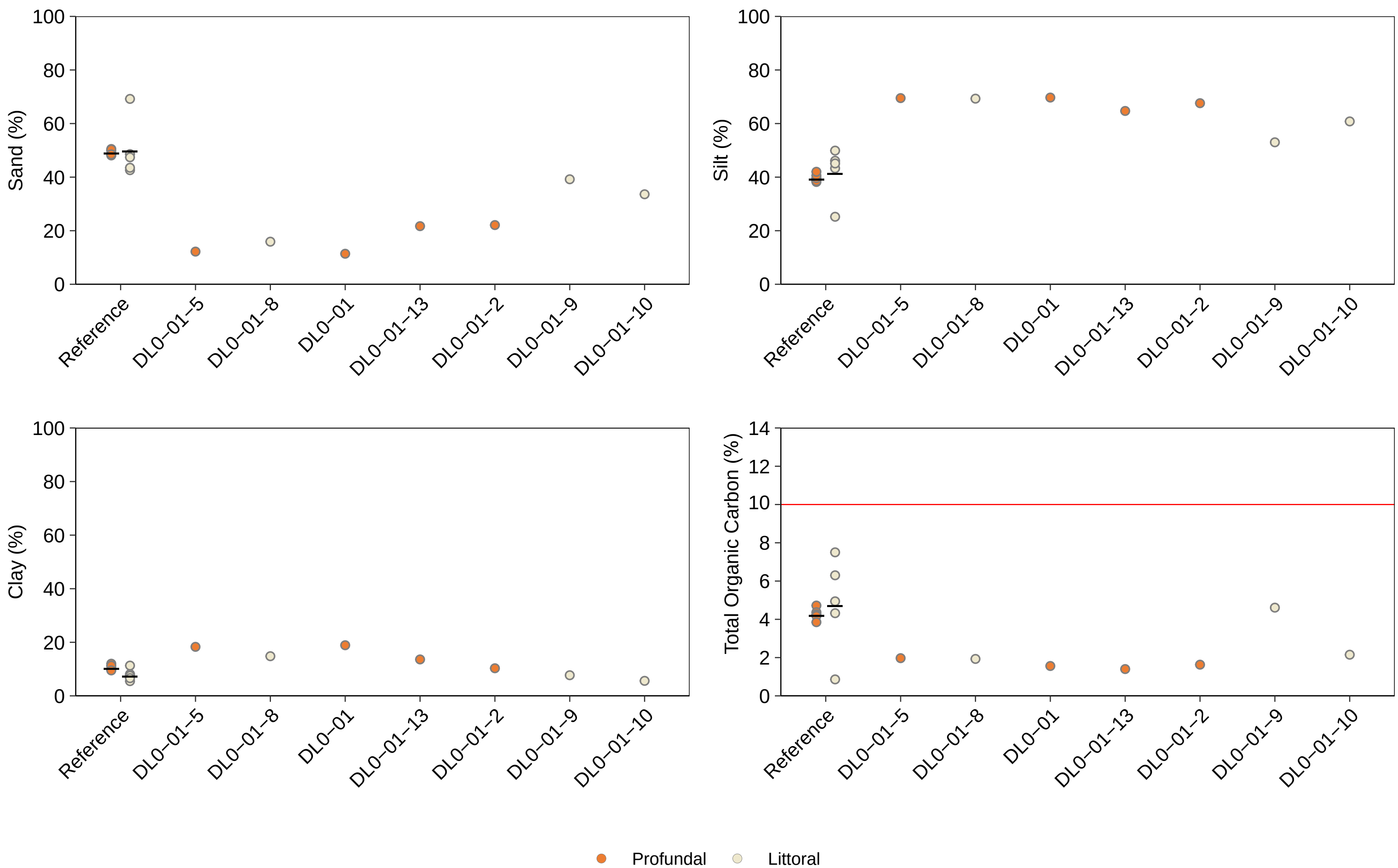


Figure 4.6: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sediment Core Samples Taken from Sheardown Lake Northwest (NW; DL0−01) Sediment Monitoring Stations and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

In littoral and profundal sediments collected from Sheardown Lake NW in 2024, mean concentrations of iron were above AEMP benchmarks and SQG and mean concentrations of manganese were above SQG (Table 4.4; Appendix Table D.13). Similar to Camp Lake (Section 3.3.2), higher iron concentrations (relative to AEMP benchmarks and SQG) in sediments from Sheardown Lake NW are influenced by the regional geology and geogenic enrichment may contribute to naturally high iron concentrations, which makes this area attractive for iron mining. This conclusion is supported by similarly high mean iron concentrations in Reference Lake 3 sediments in 2024 and in Sheardown Lake NW during the baseline period (Table 4.4; Appendix Tables D.2 and D.14). Similar to iron, manganese concentrations in sediments from the reference lake in 2024 and Sheardown Lake NW during the baseline period suggest the elevated (relative to SQG) manganese concentrations in sediments from Sheardown Lake NW are at least partially geogenic (Table 4.4; Appendix Tables D.2 and D.14).

Apart from iron and manganese, no other metals in sediments from Sheardown Lake NW had mean concentrations above AEMP benchmarks and/or SQG in 2024 (Table 4.4; Appendix Table D.13). However, specific individual sediment samples with high (relative to AEMP benchmarks and SQG) iron and/or manganese concentrations (e.g., DD-HAB-9-STN2 and DL0-01-9) also had arsenic, copper, and nickel concentrations above AEMP benchmarks and/or SQG, likely due to sorption characteristics of the sediment (Appendix Table D.13). Specifically, iron and manganese (oxy)hydroxides are known to sorb metal cations (e.g., nickel) and anions (e.g., arsenic; Bendell-Young et al. 1992). Individual sediment samples for some metals exceeded their respective AEMP benchmarks in 2024 including two littoral samples for arsenic, two littoral and one profundal samples for nickel, and one littoral sample for copper (Appendix Table D.13). One profundal sample and two littoral and two profundal samples also exceeded the SQGs for chromium and nickel, respectively (Appendix Table D.13). Overall, in 2024, mean concentrations of metals in sediments of Sheardown Lake NW, including for iron and manganese, were similar to or below (e.g., copper) those measured at Reference Lake 3 (Appendix Table D.14).

Mean metal concentrations in sediments collected from littoral and profundal stations in Sheardown Lake NW in 2024 were comparable to concentrations measured during the

Table 4.4: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Sheardown Lake NW (DL0-01) and Reference Lake 3 (REF-03) Sediment Monitoring Stations, Mary River Project CREMP, August 2024

	Parameter	Units	Canadian or Provincial SQG Criteria[a]	AEMP Benchmark[b]	Littoral		Profundal	
					Reference Lake (n = 5)	Sheardown Lake NW (n = 3)	Reference Lake (n = 5)	Sheardown Lake NW (n = 4)
					Average ± SD	Average ± SD	Average ± SD	Average ± SD
	TOC	%	10[α]	-	4.78 ± 2.52	2.90 ± 1.49	4.28 ± 0.315	1.64 ± 0.240
Metals	Aluminum (Al)	mg/kg	-	-	16,560 ± 3,306	19,133 ± 3,821	23,060 ± 1,363	23,625 ± 1,310
	Antimony (Sb)	mg/kg	-	-	<0.1 ± -	0.103 ± 0.00667	<0.1 ± -	0.102 ± -
	Arsenic (As)	mg/kg	17	6.2	5.02 ± 1.55	5.83 ± 2.13	5.07 ± 0.449	4.55 ± 0.589
	Barium (Ba)	mg/kg	-	-	115 ± 34.7	103 ± 26.4	142 ± 20.5	113 ± 10.9
	Beryllium (Be)	mg/kg	-	-	0.646 ± 0.147	0.897 ± 0.174	0.884 ± 0.0586	1.14 ± 0.0750
	Bismuth (Bi)	mg/kg	-	-	<0.2 ± -	0.290 ± 0.0346	<0.2 ± -	0.328 ± 0.0665
	Boron (B)	mg/kg	-	-	13.3 ± 2.05	26.6 ± 6.61	16.7 ± 0.879	33.7 ± 2.17
	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.0497	0.315 ± 0.146	0.166 ± 0.0166	0.277 ± 0.0302
	Calcium (Ca)	mg/kg	-	-	4,716 ± 728	4,460 ± 783	5,426 ± 237	4,572 ± 174
	Chromium (Cr)	mg/kg	90	97	55.1 ± 12.3	74.0 ± 10.6	76.0 ± 4.65	88.3 ± 3.76
	Cobalt (Co)	mg/kg	-	-	11.5 ± 2.84	16.2 ± 2.45	17.4 ± 1.70	18.7 ± 0.556
	Copper (Cu)	mg/kg	197	58	***67.5*** ± 21.3	48.9 ± 18.1	***95.1*** ± 8.03	51.6 ± 5.28
	Iron (Fe)	mg/kg	40,000[α]	52,200	***58,760*** ± 25,999	***81,967*** ± 23,061	49,820 ± 3,295	***62,000*** ± 8,078
	Lead (Pb)	mg/kg	91.3	35	13.7 ± 1.78	19.0 ± 4.05	18.5 ± 1.01	23.8 ± 1.93
	Lithium (Li)	mg/kg	-	-	25.6 ± 5.12	28.6 ± 6.91	36.2 ± 2.68	37.8 ± 2.68
	Magnesium (Mg)	mg/kg	-	-	11,308 ± 2,124	13,933 ± 1,501	15,780 ± 841	16,075 ± 1,513
	Manganese (Mn)	mg/kg	1,100[α,β]	4,530	862 ± 611	1,284 ± 1,177	2,246 ± 2,318	2,320 ± 2,019
	Mercury (Hg)	mg/kg	0.486	0.17	0.0470 ± 0.0233	0.0405 ± 0.0163	0.0702 ± 0.0129	0.0451 ± 0.0125
	Molybdenum (Mo)	mg/kg	-	-	4.63 ± 1.94	7.01 ± 2.17	2.83 ± 0.501	4.16 ± 2.12
	Nickel (Ni)	mg/kg	75[α,β]	77	39.2 ± 8.63	73.5 ± 14.1	52.2 ± 3.77	75 ± 2.29
	Phosphorus (P)	mg/kg	2,000[α]	1,958	1,344 ± 713	857 ± 62.4	999 ± 72	903 ± 44.6
	Potassium (K)	mg/kg	-	-	4,118 ± 630	5,113 ± 1,053	5,600 ± 317	6,158 ± 425
	Selenium (Se)	mg/kg	-	-	0.740 ± 0.278	0.540 ± 0.226	0.826 ± 0.133	0.488 ± 0.0842
	Silver (Ag)	mg/kg	-	-	0.146 ± 0.0462	0.143 ± 0.0404	0.238 ± 0.0192	0.188 ± 0.0411
	Sodium (Na)	mg/kg	-	-	311 ± 48.8	272 ± 57.8	431 ± 20.9	357 ± 29.1
	Strontium (Sr)	mg/kg	-	-	11.1 ± 1.22	11.4 ± 1.67	13.3 ± 0.458	13.6 ± 0.681
	Sulphur (S)	mg/kg	-	-	1,620 ± 403	1,533 ± 533	1,360 ± 114	1,050 ± -
	Thallium (Tl)	mg/kg	-	-	0.423 ± 0.145	0.486 ± 0.151	0.748 ± 0.0562	0.598 ± 0.0277
	Tin (Sn)	mg/kg	-	-	<2 ± -	<2 ± -	<2 ± -	<2 ± -
	Titanium (Ti)	mg/kg	-	-	958 ± 159	1,183 ± 185	1,164 ± 37.1	1,462 ± 46.5
	Uranium (U)	mg/kg	-	-	15.3 ± 5.91	9.52 ± 4.84	25.1 ± 2.33	8.88 ± 1.91
	Vanadium (V)	mg/kg	-	-	51.2 ± 9.67	53.9 ± 11.6	67.7 ± 3.92	65.8 ± 1.95
	Zinc (Zn)	mg/kg	315	135	72.1 ± 14.9	71.0 ± 10.3	95.2 ± 6.65	81.3 ± 7.87
	Zirconium (Zr)	mg/kg	-	-	4.26 ± 1.70	11.7 ± 4.67	3.92 ± 0.455	10.0 ± 4.53

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

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation. "-" indicates data not available.

[a] Canadian SQG for the protection of aquatic life probable effect level (PEL; CCME 2024) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BCMOE 2024).

[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 Sheardown Lake NW

baseline period, except for boron, which had concentrations that were moderately to highly elevated relative to baseline[35] (Appendix Figure D.1, Appendix Table D.14). Metal concentrations in sediments from littoral and profundal stations in Sheardown Lake NW in 2024 were also typically within the ranges observed during mine operations from 2015 to 2023 (Figure 4.7). These results further indicate that metals with mean concentrations above AEMP benchmarks and/or SQG (i.e., iron and manganese) are likely naturally elevated.

Under the AEMP Management Response Framework, sediment quality in Sheardown Lake NW initially triggered a Moderate Action Response in 2022 based on mean concentrations of iron exceeding the AEMP benchmark (Minnow 2023). The recommended Action Response, which was completed in the 2023 CREMP (Minnow 2024a), was a special investigation into iron concentrations in Sheardown Lake NW using a temporal trend analysis. Based on the results of this investigation in the 2023 CREMP (Minnow 2024a), it was recommended that the temporal trend analysis be repeated in the 2024 CREMP, incorporating 2024 results (Table 2.9).

Trend analyses conducted for Sheardown Lake NW in 2024 identified significant increasing trends in mean iron concentrations in sediment from littoral and profundal habitats since baseline (2007 to 2024; Appendix Table H.8) and over the mine operation period (2015 to 2024; Appendix Table H.9). Similar temporal trends were not observed for iron concentrations at Reference Lake 3, where data are only available for the period of mine operation (2015 to 2024; Appendix Tables H.8 and H.9). Iron concentrations in littoral sediments at Sheardown Lake NW have often been above both the AEMP benchmark and SQG since 2015 (Figure 4.7). Though a slight step-change/increasing trend in sediment iron concentration for littoral and profundal habitats between the baseline/construction periods (up to 2014) and the operation period (starting in 2015) may have contributed to the significant increasing trend since baseline, the highest concentrations of iron in littoral and profundal sediments were observed in 2022, 2023, and 2024, suggesting that the increasing temporal trend over the mine operations period primarily reflects higher concentrations in recent years (Figure 4.7, Appendix Figure H.5). Collectively, increasing iron concentrations in sediments, the observed spatial variability (i.e., highest concentrations near tributary inflows) within Sheardown Lake NW, and the highest iron concentrations in sediments being observed in recent years (i.e., 2023 and 2024), suggest an emerging mine-related influence on sediment quality within the lake.

[35] Boron concentrations in sediments collected from 2015 to 2024 were considerably higher (i.e., 10- to 70-times) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences (i.e., probable under-recovery of boron in baseline and 2014). The analytical laboratory used for the baseline study differed from the current laboratory.

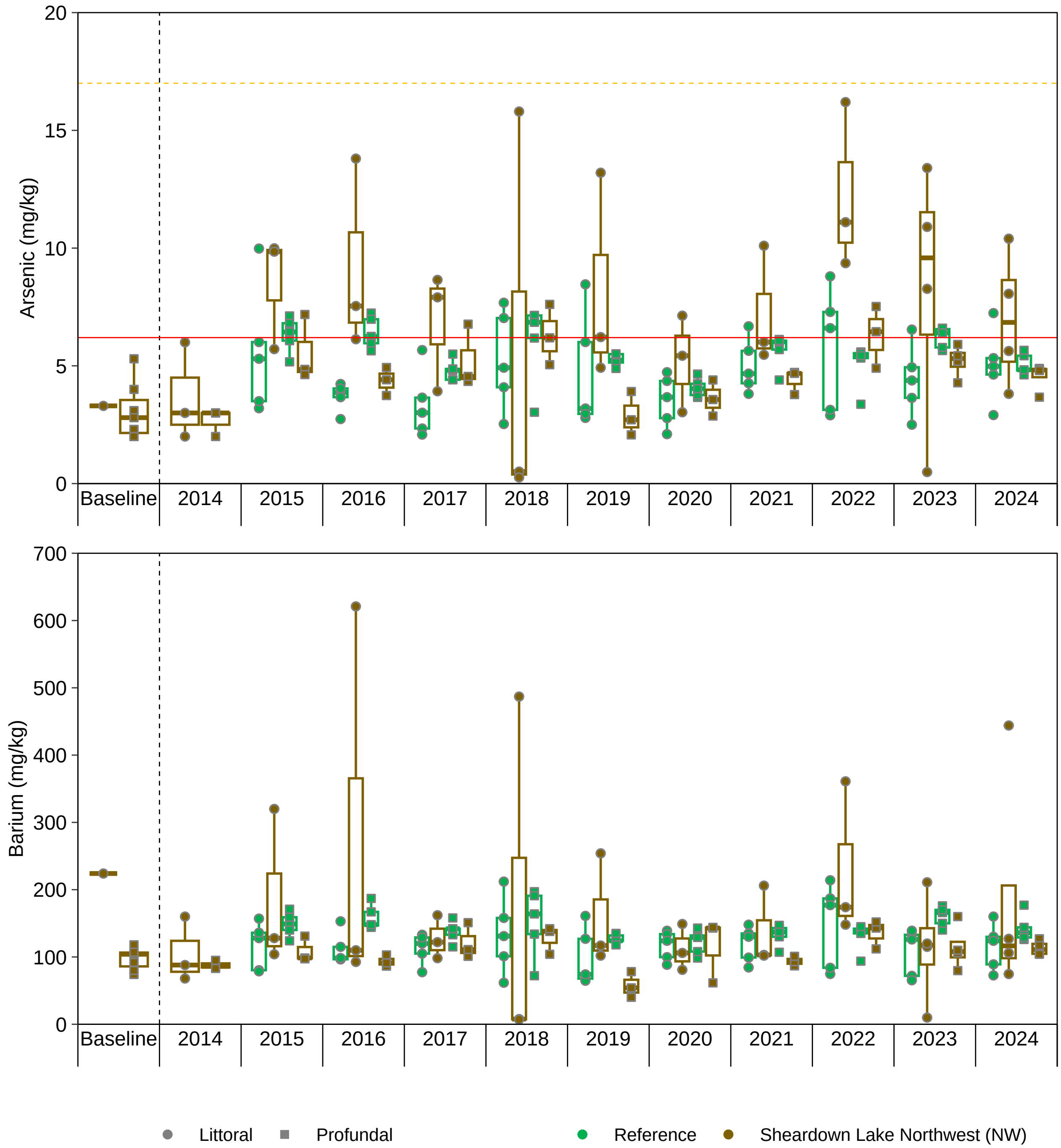


Figure 4.7: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

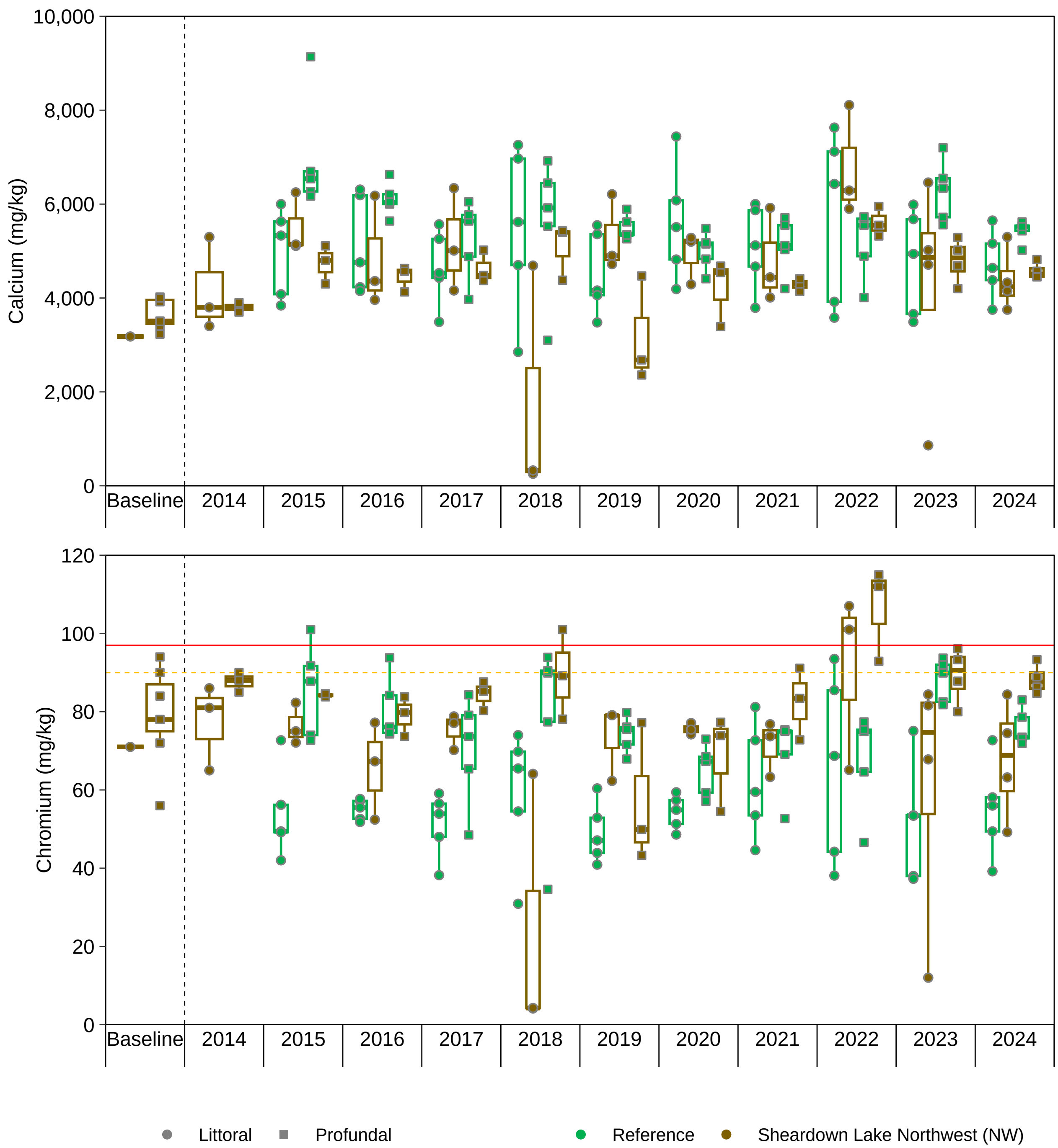


Figure 4.7: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

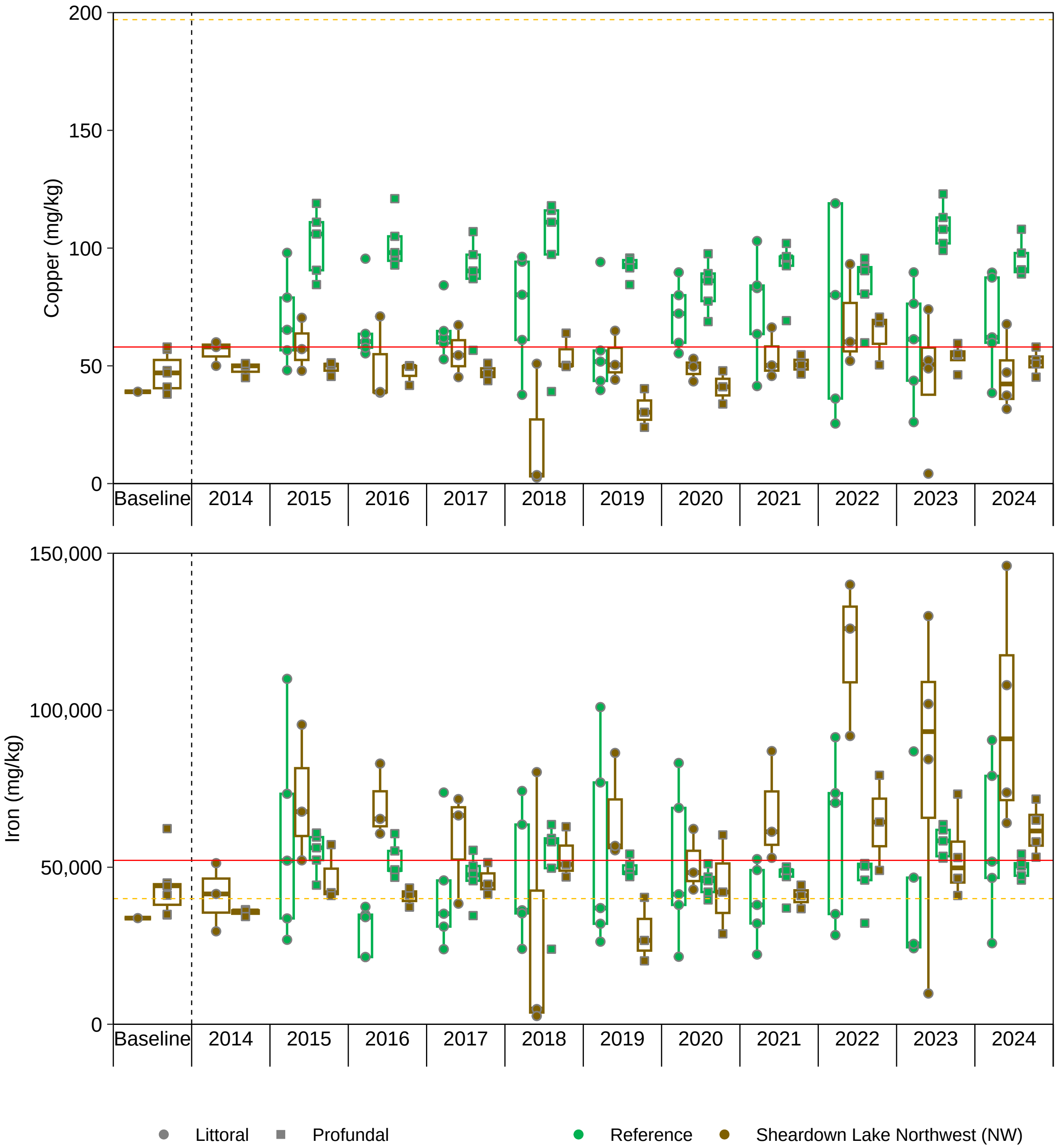


Figure 4.7: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

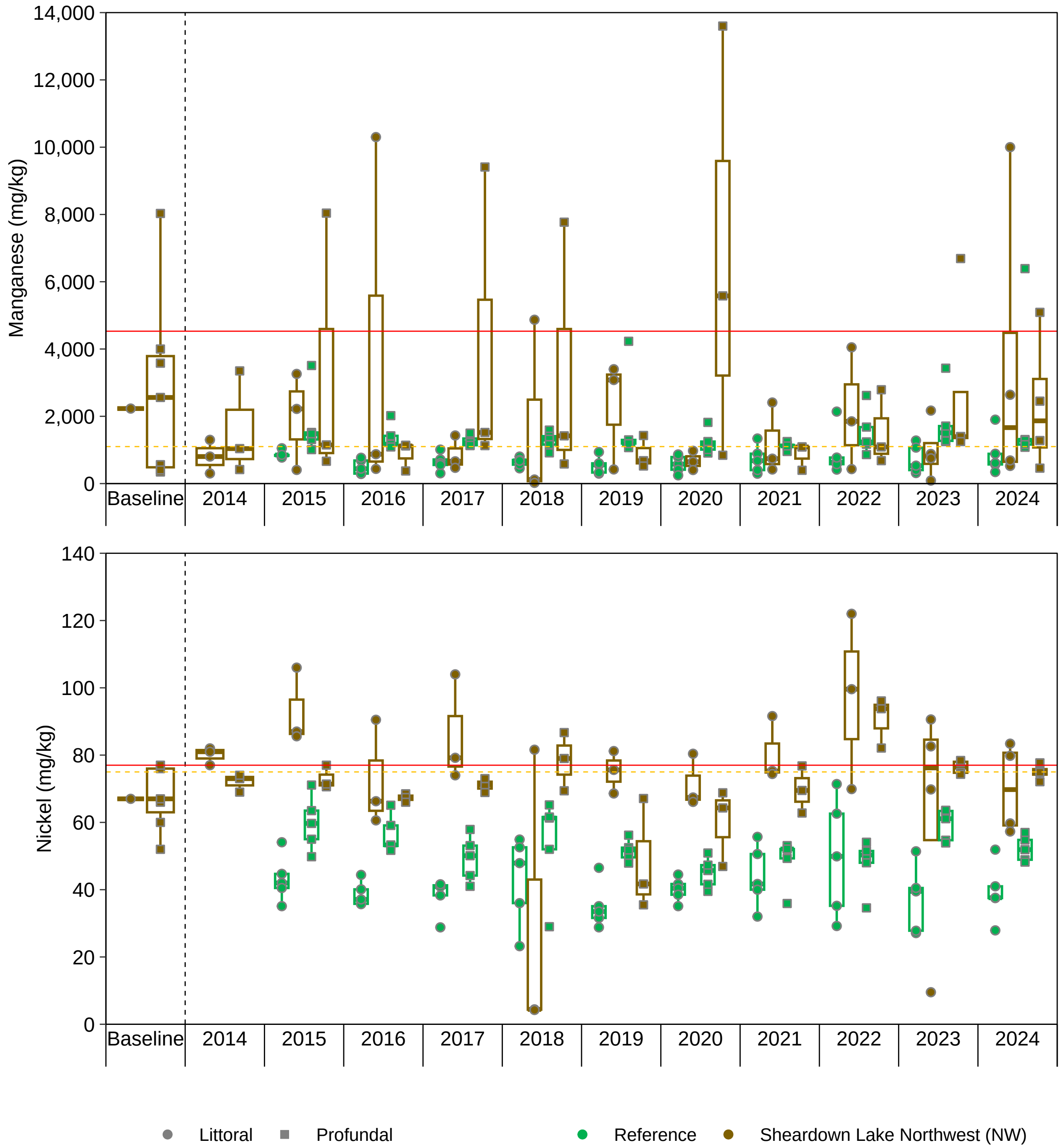


Figure 4.7: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

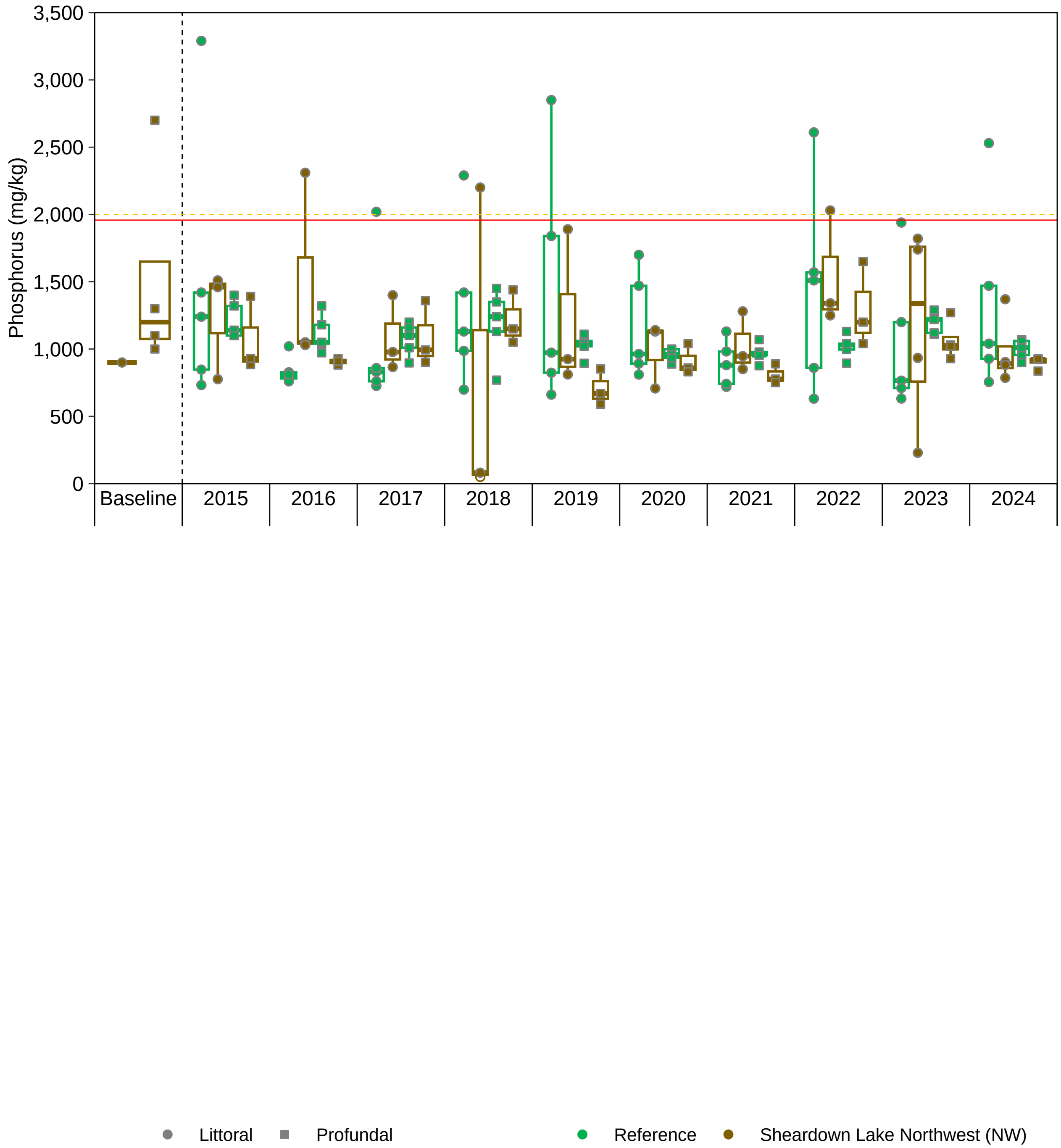


Figure 4.7: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

Results of trend analyses completed for individual sediment sampling stations in Sheardown Lake NW are indicative of significant increasing monotonic trends in iron concentrations over time at Stations DD-HAB-9-STN2, DL0-01-9, and DL0-01-05 (west of SDLT1; Appendix Tables H.10 and H.11). Overall, the temporal trend in sediment iron concentration at Station DD-HAB-9-STN2 and DL0-01-9 mirrored the general trend in Sheardown Lake NW where there was a slight increasing trend between the baseline/construction periods (up to 2014) and the operation period (starting in 2015; Appendix Figure H.5). Concentrations appeared to be increasing following a linear trend with the highest concentrations observed in 2022, 2023, and 2024 indicating that the increasing temporal trend over the mine operations period primarily reflects higher concentrations in recent years (Appendix Figure H.5). Iron in lake sediments at DD-HAB-9-STN2, which is near the SDLT1 tributary inflow, may reflect tributary influences, including TOC inputs and/or metal sources to the tributary. Evidence of tributary influences is further supported by iron concentrations in sediments at DL0-01-09 (the station nearest to the SDLT12 inflow in Sheardown Lake NW); these were the second highest iron concentrations of any sediment sampling station in 2024 and followed a similar temporal pattern as concentrations at DD-HAB-9-STN2 (Appendix Figure H.5).

In the 2023 CREMP, temporal trend analyses were completed for other sediment metal (i.e., arsenic, chromium, copper, nickel, and phosphorus) concentrations and no increasing or decreasing monotonic trends (since baseline or over the period of mine operations) over time were identified at individual sampling stations, or for littoral or profundal habitat areas of Sheardown Lake NW (Minnow 2024a). Finally, despite mean concentrations above the SQG in 2024, manganese concentrations in Sheardown Lake NW sediments did not indicate an increase in concentration with time in littoral or profundal areas based on visual assessment of temporal patterns (Figure 4.7). These results suggest there have been no mine-related influences on concentrations of these parameters in sediments of Sheardown Lake NW.

In summary, spatial and temporal trends in sediment iron concentrations suggest an emerging mine-related influence on sediment quality in Sheardown Lake NW; however, there is no apparent mine-related influence on concentrations of other metals in the sediments of Sheardown Lake NW nor on metal-sensitive BIC taxa (e.g., *Chironomidae*; see section 4.4.4).

4.4.3 Phytoplankton

Mean chlorophyll-a concentrations at Sheardown Lake NW in 2024 showed no consistent spatial gradients with distance from the lake inlet across winter, summer, and fall sampling events (Figure 4.8 and 2.2). While significant differences were observed between winter and both summer and fall, no significant difference was found between summer and fall chlorophyll-a concentrations (Figure 4.8, Appendix Table E.4). Fall exhibited the highest

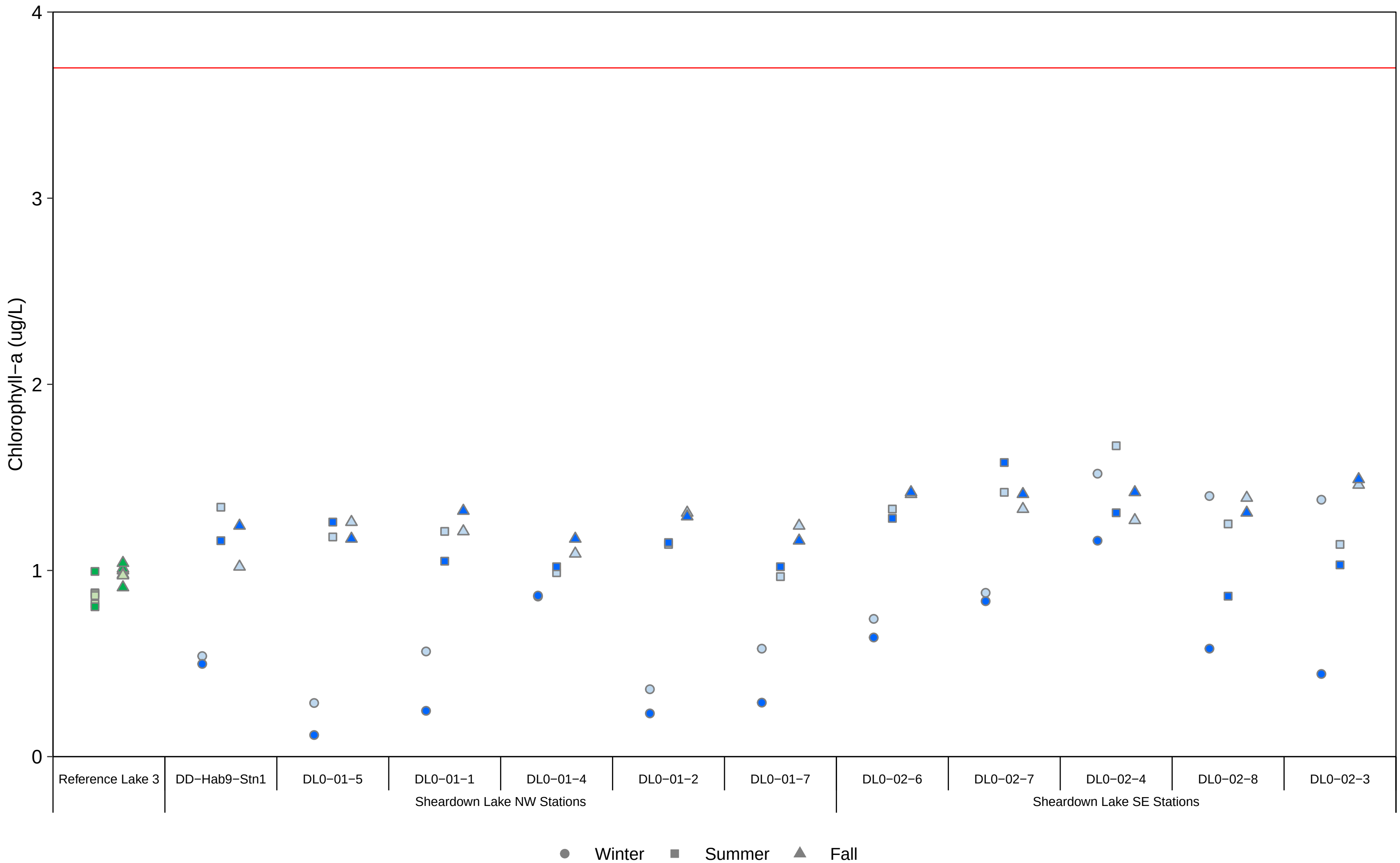


Figure 4.8: Chlorophyll−a Concentrations at Sheardown Lake Northwest (NW; DL0-01) and Sheardown Lake Southeast (SE; DL0-02) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2024

Notes: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Red line indicates AEMP Benchmark. Lighter shade of colour indicates surface sample, darker shade indicates bottom sample. Reference areas are shown in green and mine−exposed areas are shown in blue. Sheardown Lake NW Stations are presented in order of proximity to the lake inlet (left to right). In Sheardown Lake SE, Station DL0-02-6 is proximal to the lake inlet from Sheardown Lake NW, Station DL0-02-4 is proximal to the lake inlet from Sheardown Lake Tributary 9, and Station DL0-02-3 is proximal to the lake outlet to the Mary River.

chlorophyll-a concentrations, while winter had the lowest (Figure 4.8, Appendix Table E.10). Chlorophyll-a concentrations at Sheardown Lake NW were significantly higher than those at the reference lake during both the summer and fall sampling events (Appendix Tables E.6 and E.7). Despite these higher concentrations in the summer and fall relative to the reference lake, chlorophyll-a concentrations in all seasonal events at Sheardown Lake NW remained well below the AEMP benchmark of 3.7 µg/L (Figure 4.8) and chlorophyll-a concentrations <4.5 ug/L (Appendix Table E.10) and total phosphorus concentrations <10 µg/L (Table 4.3, Appendix Table C.43) indicated an oligotrophic status for Sheardown Lake NW (Wetzel 2001, CCME 2024b; see Section 3.3.3 for additional trophic status classification details).

Although chlorophyll-a concentrations at Sheardown Lake NW differed significantly among years of mine construction and operation, both seasonally and annually, the data showed considerable temporal and seasonal variability (Figure 4.9, Appendix Table E.11). The 2024 concentrations fell within the seasonal ranges observed from 2014 to 2023, and there were no consistent directional changes across winter, summer, or fall (Figure 4.9, Appendix Table E.11). Chlorophyll-a concentrations in Sheardown Lake NW have consistently been slightly higher than those at Reference Lake 3 during at least one season each year since mining operations began (Figure 4.9; Minnow 2016a, 2017, 2018, 2019, 2020, 2021b, 2022, 2023, 2024a). However, the differences between the two lakes have been and remain minimal (i.e., less than 1.5 µg/L), with both lakes falling within the same trophic classification, suggesting no ecologically relevant mine-related influences on Sheardown Lake NW. The relatively small magnitude and consistency of differences in chlorophyll-a concentrations between Sheardown Lake NW and Reference Lake 3 also suggest that they are due natural factors, such as lake morphology and location (e.g., lake size and fetch which affect lake mixing potential and the amount of sunlight received) rather than mine-related influences. In addition, due to the absence of chlorophyll-a data for the baseline period (2005 to 2013), comparisons to pre-mining conditions could not be made. Overall, chlorophyll-a concentrations in Sheardown Lake NW exhibited no consistent directional temporal trends in any season, have generally remained consistent relative to concentrations observed at Reference Lake 3 since 2015, and remained well below the AEMP benchmark in 2024. These results indicate no adverse mine-related effects on phytoplankton productivity at Sheardown Lake NW in 2024.

4.4.4 Benthic Invertebrate Community

In 2024, most BIC endpoints for littoral (shallow) habitats of Sheardown Lake NW were statistically similar to those of Reference Lake 3, except for higher total density and lower relative proportion of clinger taxa (Table 4.5). Benthic invertebrate density in littoral habitats was significantly higher (MCT = 4,684 individuals/m^2) in Sheardown Lake NW compared to similar habitats at

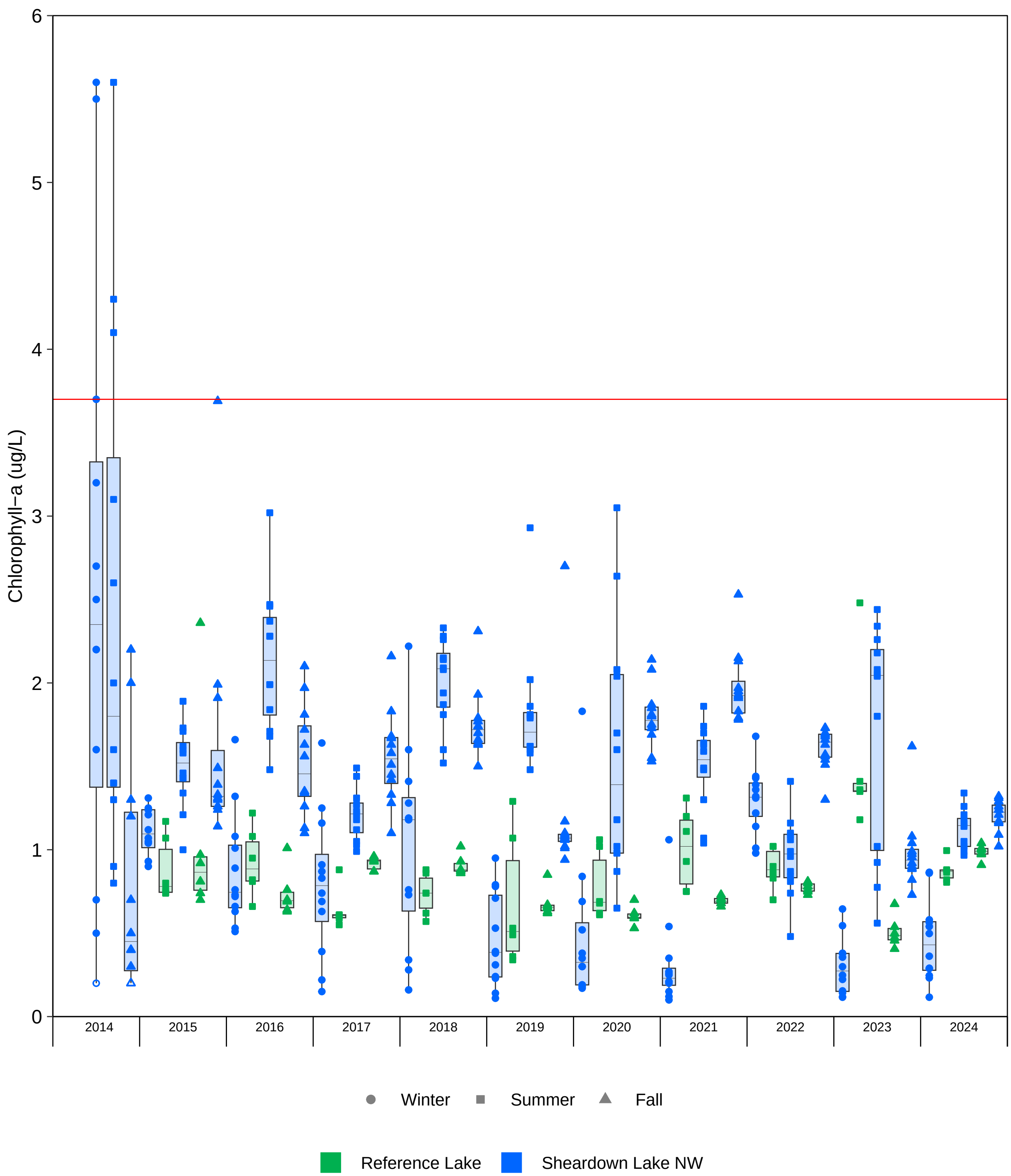


Figure 4.9: Temporal Comparison of Chlorophyll−a Concentrations Among Seasons between Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03) for Construction (2014) and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

Table 4.5: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Littoral Habitats in Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake Littoral Habitat	MCT (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	log10	YES	0.008	1.8	Reference Lake 3	1,049	901	403	215	982	2,514
						Sheardown Lake Northwest (NW)	4,684	2,227	996	1,860	4,719	7,784
Richness (Number of Taxa)	tequal	log10	NO	0.243	0.67	Reference Lake 3	8.80	3.56	1.59	5.00	8.00	13.0
						Sheardown Lake Northwest (NW)	11.2	3.11	1.39	8.00	11.0	16.0
Simpson's Evenness (E)	tequal	log10	NO	0.177	-1.5	Reference Lake 3	0.759	0.0621	0.0278	0.669	0.755	0.840
						Sheardown Lake Northwest (NW)	0.677	0.113	0.0504	0.541	0.652	0.851
Shannon's Diversity	tequal	none	NO	0.710	0.31	Reference Lake 3	2.01	0.243	0.109	1.72	2.01	2.28
						Sheardown Lake Northwest (NW)	2.08	0.362	0.162	1.62	2.10	2.57
Hydracarina (%)	tequal	log10(x+1)	NO	0.491	-0.35	Reference Lake 3	2.49	2.88	1.29	0	2.33	7.02
						Sheardown Lake Northwest (NW)	1.46	1.20	0.538	0	0.990	2.92
Ostracoda (%)	tequal	none	NO	0.231	0.91	Reference Lake 3	39.5	16.1	7.20	16.0	40.3	60.5
						Sheardown Lake Northwest (NW)	54.1	19.4	8.66	21.2	61.3	71.6
Chironomidae (%)	tequal	log10	NO	0.373	-0.69	Reference Lake 3	52.6	14.4	6.46	30.7	56.6	68.0
						Sheardown Lake Northwest (NW)	43.9	19.6	8.79	25.9	37.0	77.0
Metal Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.295	-0.67	Reference Lake 3	22.1	17.5	7.81	0	17.3	41.3
						Sheardown Lake Northwest (NW)	10.6	15.3	6.86	2.92	4.38	38.0
Collector Gatherers (%)	tequal	none	NO	0.594	0.31	Reference Lake 3	74.9	18.1	8.08	56.4	77.0	100
						Sheardown Lake Northwest (NW)	80.5	13.7	6.12	56.6	85.2	90.1
Filterers (%)	tequal	log10(x+1)	NO	0.298	-0.67	Reference Lake 3	21.7	17.6	7.85	0	17.3	41.3
						Sheardown Lake Northwest (NW)	10.3	15.6	6.95	1.98	3.65	38.0
Shredders (%)	tequal	log10(x+1)	NO	0.248	1.1	Reference Lake 3	0.344	0.578	0.259	0	0	1.33
						Sheardown Lake Northwest (NW)	1.01	1.05	0.468	0	0.885	2.19
Clingers (%)	tequal	none	YES	0.045	-1.1	Reference Lake 3	24.0	17.9	8.01	0	19.0	43.6
						Sheardown Lake Northwest (NW)	4.77	2.56	1.15	1.98	3.85	8.64
Sprawlers (%)	tequal	none	NO	0.113	0.84	Reference Lake 3	66.2	16.9	7.56	43.2	74.5	84.0
						Sheardown Lake Northwest (NW)	80.5	6.03	2.70	73.3	83.9	86.4
Burrowers (%)	tequal	none	NO	0.319	0.77	Reference Lake 3	9.82	6.40	2.86	2.30	8.16	16.9
						Sheardown Lake Northwest (NW)	14.8	8.17	3.66	4.94	12.4	24.8

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

Reference Lake 3 (MCT = 1,049 individuals/m^2); however, the difference between the two lakes was not ecologically meaningful based on a MOD outside of the CES_{BIC} of ± 2 SD_{REF} (Table 4.5). The lower relative proportion of clinger taxa in the samples from littoral habitats of Sheardown Lake NW relative to reference was also not ecologically meaningful but was likely reflected in the results for Bray-Curtis Index, which indicated compositional differences in the littoral BIC of the two lakes (Appendix Table F.19). The majority of the benthic invertebrates collected from littoral habitats in Sheardown Lake NW as part of the CREMP in 2024 were chironomids (i.e., midges).

The BIC endpoints for profundal habitats of Sheardown Lake NW were generally statistically similar to those for the same habitats in Reference Lake 3 in 2024. Exceptions were significantly higher invertebrate density and taxa richness and lower proportion of *Ostracoda* in Sheardown Lake NW versus the reference lake, and overall differences in Bray-Curtis Index signaling compositional differences in the BIC of profundal habitats (Table 4.6; Appendix Table F.19). The difference between benthic invertebrate density in profundal habitats of Sheardown Lake NW (MCT = 873 individuals/m^2) and Reference Lake 3 (MCT = 202 individuals/m^2) was ecologically meaningful, based on a MOD outside of the CES_{BIC} of ± 2 SD_{REF} (Table 4.6). The lower proportion of *Ostracoda* in the profundal BIC samples from Sheardown Lake NW relative to Reference Lake 3 was also ecologically meaningful, whereas significant differences in taxa richness were not (i.e., the MOD was smaller than the CES_{BIC}; Table 4.6). Since the start of mine operations in 2015, proportions of *Ostracoda* in profundal habitats of Sheardown Lake NW have not changed significantly relative to both baseline years (Appendix Table F.33) and qualitative assessment of temporal patterns indicates that the proportion of *Ostracoda* in profundal habitats of Sheardown Lake NW has varied relative to the reference lake and there have been no consistent directional (i.e., increasing or decreasing) changes (Appendix Figure F.10).

For both habitat types in Sheardown Lake NW, no significant and ecologically meaningful differences were found in the relative proportions of FFGs and HPGs between the two lakes (Tables 4.5 and 4.6). This suggests that both lakes provide similar food resources and habitat availability for benthic invertebrates.

For the littoral habitat of Sheardown Lake NW, significant differences in BIC endpoints relative to baseline that had MOD outside of the CES_{BIC} of ± 2 SD_{REF} (i.e., differences considered ecologically meaningful) included richness, Simpson's Evenness, and relative proportions of *Ostracoda*, *Chironomidae*, and collector-gatherers (Appendix Table F.32, Appendix Figure F.9). There was no consistent directionality (i.e., consistent positive or negative MODs for comparisons of mine operational years relative to baseline) for the endpoints with ecologically meaningful differences (i.e., those with a MOD outside of the CES_{BIC} of ± 2 SD_{REF};

Table 4.6: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Profundal Habitats in Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake Littoral Habitat	MCT (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	M-W	rank	YES	0.008	14	Reference Lake 3	202	33.7	15.1	146	207	233
						Sheardown Lake Northwest (NW)	873	711	318	448	551	2,136
Richness (Number of Taxa)	tequal	log10	YES	0.035	1.5	Reference Lake 3	4.40	1.14	0.510	3.00	4.00	6.00
						Sheardown Lake Northwest (NW)	6.40	1.34	0.600	5.00	7.00	8.00
Simpson's Evenness (E)	tequal	log10	NO	0.693	0.28	Reference Lake 3	0.582	0.169	0.0754	0.457	0.508	0.867
						Sheardown Lake Northwest (NW)	0.629	0.177	0.0794	0.385	0.662	0.874
Shannon's Diversity	tequal	none	NO	0.238	0.91	Reference Lake 3	1.28	0.318	0.142	0.834	1.27	1.71
						Sheardown Lake Northwest (NW)	1.56	0.391	0.175	0.936	1.60	1.96
Hydracarina (%)	M-W	rank	NO	0.290	-[a]	Reference Lake 3	4.09	5.94	2.66	0	0	13.0
						Sheardown Lake Northwest (NW)	8.86	9.11	4.07	1.59	4.69	24.2
Ostracoda (%)	tequal	log10(x+1)	YES	0.035	-2.5	Reference Lake 3	8.37	2.08	0.929	5.88	8.33	11.5
						Sheardown Lake Northwest (NW)	3.35	3.92	1.75	0	1.59	7.66
Chironomidae (%)	tequal	none	NO	0.707	0.33	Reference Lake 3	85.2	7.71	3.45	76.9	85.2	94.1
						Sheardown Lake Northwest (NW)	87.8	12.5	5.58	68.2	95.3	96.8
Metal Sensitive Chironomidae (%)	M-W	rank	NO	0.421	-1.1	Reference Lake 3	9.98	11.3	5.05	0	7.41	29.4
						Sheardown Lake Northwest (NW)	4.16	4.28	1.91	0.411	3.12	11.5
Collector Gatherers (%)	tequal	none	NO	0.287	-0.58	Reference Lake 3	85.2	16.2	7.24	57.6	88.2	100
						Sheardown Lake Northwest (NW)	75.8	8.51	3.81	68.8	71.2	87.1
Filterers (%)	M-W	rank	NO	0.180	-[a]	Reference Lake 3	6.70	12.8	5.72	0	0	29.4
						Sheardown Lake Northwest (NW)	0	0	0	0	0	0
Shredders (%)	M-W	rank	NO	0.424	-[a]	Reference Lake 3	0.833	1.86	0.833	0	0	4.17
						Sheardown Lake Northwest (NW)	0	0	0	0	0	0
Clingers (%)	M-W	rank	NO	0.398	-[a]	Reference Lake 3	9.96	18.4	8.23	0	0	42.4
						Sheardown Lake Northwest (NW)	8.86	9.11	4.07	1.59	4.69	24.2
Sprawlers (%)	tequal	none	NO	0.722	-0.19	Reference Lake 3	86.2	16.7	7.46	57.6	88.9	100
						Sheardown Lake Northwest (NW)	83.0	9.77	4.37	71.2	84.4	95.2
Burrowers (%)	tequal	log10(x+1)	NO	0.431	0.85	Reference Lake 3	3.88	4.71	2.11	0	3.70	11.5
						Sheardown Lake Northwest (NW)	8.17	10.3	4.60	0.411	3.23	25.0

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

[a] Contrast MODs could not be calculated because the MAD = 0.

Appendix Table F.32, Appendix Figure F.9). However, relative proportions of *Ostracoda* were notably higher in 2023 and 2024 relative to baseline years 2007 and 2008, whereas relative proportions of *Chironomidae* were lower, signalling a relatively recent shift in community composition relative to baseline (Appendix Table F.32). Qualitative assessment of temporal patterns also suggests that, in littoral habitats, the proportion of *Ostracoda* has been increasing and the proportion of *Chironomidae* has been decreasing since 2022 in Sheardown Lake NW (Appendix Figure F.9). Results of the 2024 Lake Sedimentation Monitoring Program suggest this shift may be attributed to higher sedimentation rates and accumulation thicknesses at BIC stations co-located with sediment trap monitoring stations in littoral habitats of Sheardown Lake NW in 2023 and 2024 relative to baseline (2015 for sedimentation), based on sedimentation data for the open-water season (Minnow 2025). Specifically, relative proportions of *Chironomidae* were strongly and negatively correlated with sedimentation rate and accumulation thickness, and relative proportions of *Ostracoda* were shown to be strongly and positively correlated with sediment accumulation thickness at the littoral BIC stations co-located with the sediment trap monitoring stations (Minnow 2025). Although this localized, potential mine-related influence of sedimentation on BIC was detected through the Lake Sedimentation Monitoring Program (Minnow 2025), metal-sensitive *Chironomidae*, key bioindicators of contaminant stress, did not exhibit any significant change in abundance over time in CREMP monitoring. This suggests that while sedimentation may be influencing BIC structure in certain areas of the lake, there is currently no indication of widespread ecological degradation or metal-related toxicity in the littoral BIC community of Sheardown Lake NW.

In the profundal habitat of Sheardown Lake NW, significant, ecologically meaningful (i.e., based on MOD outside of the CES_{BIC} of ± 2 SD_{REF}) differences among mine operational years (2015 to 2024) and baseline (2007 and/or 2013) were identified for total invertebrate density and relative proportions of *Ostracoda*, *Chironomidae*, metal-sensitive *Chironomidae*, and collector-gatherers (Appendix Table F.33, Appendix Figure F.10). For years during which ecologically meaningful differences in invertebrate density were identified relative to one or both baseline years, densities were lower during mine operational years (particularly in 2021 and 2023; Appendix Table F.33, Appendix Figure F.10). This may be attributable to higher sedimentation rates at profundal areas during the open water season of most mine operational years (i.e., 2015 to 2018 and 2022 to 2023), given that benthic invertebrate densities appear to be significantly and strongly negatively correlated with sedimentation rate (Minnow 2025). Compared to 2007, relative proportions of *Ostracoda* were significantly and meaningfully higher in 2017 and 2020 through 2023 but not in 2024 and results for mine operational years were generally comparable with each other and baseline data from 2013 (Appendix Table F.33). A similar pattern was observed for relative proportions of metal sensitive

Chironomidae; results for mine operational years were generally comparable with each other and baseline data from 2013, despite relative proportions of metal sensitive *Chironomidae* being higher in most mine operational years relative to 2007 (Appendix Table F.33).

Overall, when differences in BIC endpoints over time (i.e., relative to baseline) were found to be outside the CES_{BIC}, they were not typically significantly different than all baseline years (Appendix Tables F.32 and F.33), suggesting that these variations may, at least in part, reflect natural variability. As noted above, the 2023 and 2024 data for Sheardown Lake NW may be indicative of a potential sedimentation-related shift in the littoral BIC (i.e., an increase in the relative proportion of *Ostracoda* and a concomitant decrease in the relative proportion of *Chironomidae*). However, this potential link between a BIC community shift in Sheardown Lake NW and sedimentation was based on a localized analysis and may not be occurring across BIC habitat in the lake.

Throughout the mine operation period (2015 to 2024), benthic invertebrate densities in both the littoral and profundal habitats, as well as richness in the profundal habitat, at Sheardown Lake NW have consistently been higher than at Reference Lake 3 (Appendix Figures F.10 and F.11). This indicates that Sheardown Lake NW is more biologically productive, as also seen in differences in primary productivity (i.e., phytoplankton density as measured by chlorophyll-a; see Section 4.4.3) between the two lakes. However, the relatively few ecologically meaningful differences between the littoral and profundal BIC of Sheardown Lake NW and Reference Lake 3 in 2024, along with few consistent (i.e., increasing or decreasing over time) differences between the mine operational period and baseline, suggest no mine-related effects on the majority of BIC endpoints at Sheardown Lake NW. Shifts in the BIC correlated with sedimentation rate and accumulation thickness will continue to be investigated through the Lake Sedimentation Monitoring Program in 2025 to assess for potential mine-related influences.

4.4.5 Fish Population

4.4.5.1 Fish Community

Arctic charr (*Salvelinus alpinus*) was the only fish species captured at Sheardown Lake NW in 2024 (Table 4.7). Ninespine stickleback (*Pungitius pungitius*) have also been captured at very low densities in nearshore electrofishing surveys in Sheardown Lake NW in some CREMP monitoring years (CPUE ranged from 0.04 to 0.23 fish per electrofishing minute; Minnow 2020, 2021b, 2022, and 2024a); however, none were captured in 2024, which was also the case from 2015 to 2018 or in 2022 (Minnow 2016a, 2017, 2018, 2019 and 2023). Ninespine stickleback have not been consistently captured in Sheardown Lake NW in CREMP monitoring programs,

Table 4.7: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Sheardown Lake Northwest (NW; DL0-01), Sheardown Lake Southeast (SE; DL0-02), and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	105	15	120	2
		CPUE	1.12	0.16	1.28	
	Gill netting	No. Caught	84	-	84	
		CPUE	3.30	-	3.30	
Sheardown Lake Northwest	Electrofishing	No. Caught	109	0	109	1
		CPUE	3.63	0.00	3.63	
	Gill netting	No. Caught	109	-	109	
		CPUE	2.15	-	2.15	
Sheardown Lake Southeast	Electrofishing	No. Caught	105	26	131	2
		CPUE	2.53	0.63	3.15	
	Gill netting	No. Caught	102	-	102	
		CPUE	8.17	-	8.17	

Note: "-" indicates not applicable as ninespine stickleback are not captured by gill netting.

[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.

suggesting annual natural variability in habitat use and sampling conditions rather than changes in fish community richness over time.

In 2024, the CPUE for arctic charr captured by backpack electrofishing was higher at Sheardown Lake NW compared to Reference Lake 3 and fell within the range observed during baseline studies (2007 to 2013), construction (2014), and mine operation years (2015 to 2023; Figure 4.10, Table 4.7, Appendix Table G.1). Fish density (based on CPUE) has typically been higher in nearshore areas (i.e., electrofishing surveys) in Sheardown Lake NW than Reference Lake 3 since sampling was initiated in Reference Lake 3 in 2015 (coinciding with the start of the mine operational period; Figure 4.10)[36]. In general, higher fish density in the nearshore area of Sheardown Lake NW relative to the reference lake since 2015 may be associated with greater primary and secondary productivity, as evidenced by higher chlorophyll-a concentrations in the summer and fall (indicating greater phytoplankton density; Section 4.4.3) and higher benthic invertebrate density in littoral areas (Section 4.4.4) compared to Reference Lake 3, resulting in more abundant food sources for arctic charr.

Gill netting CPUE, representing the density of larger, littoral/profundal fish was lower in Sheardown Lake NW in 2024 compared to Reference Lake 3 (Figure 4.10, Table 4.7, Appendix Table G.3), despite higher chlorophyll-a concentrations (Section 4.4.3) and higher benthic invertebrate density in profundal areas (Section 4.4.4). The CPUE from gill netting surveys in Sheardown Lake NW in 2024 was lower than in the previous three years (2021 to 2023) but remained within the range observed during baseline studies, construction, and mine operation years (Figure 4.10). No consistent temporal patterns in chlorophyll-a concentrations (i.e., phytoplankton density) or ecologically significant differences in BIC endpoints have been observed in Sheardown Lake NW that are consistent with the temporal pattern in gill net CPUE (Sections 4.4.3 and 4.4.4). These results suggest that factors other than lake productivity differences may have resulted in variability in gill net CPUE in Sheardown NW. As described in Section 3.3.5.1, sampling related influences (e.g., seasonal timing and access to sampling locations) or naturally influenced environmental factors (e.g., water temperature) that affect fish movement behaviour, spatial ecology, and metabolic demands have the potential to influence fish catch rates, particularly in 'passive' gill net surveys.

Although gill net CPUE in Sheardown Lake NW in 2024 was lower relative to Reference Lake 3 and to recent monitoring years (i.e., 2021 to 2023), 2024 results fell within the range of previously observed CPUE for both electrofishing and gill netting surveys. Therefore, negative mine-related

[36] Baseline fish community data (2005 to 2013) were not collected at Reference Lake 3, precluding comparisons of mine-exposed to reference conditions prior to the construction of the mine.

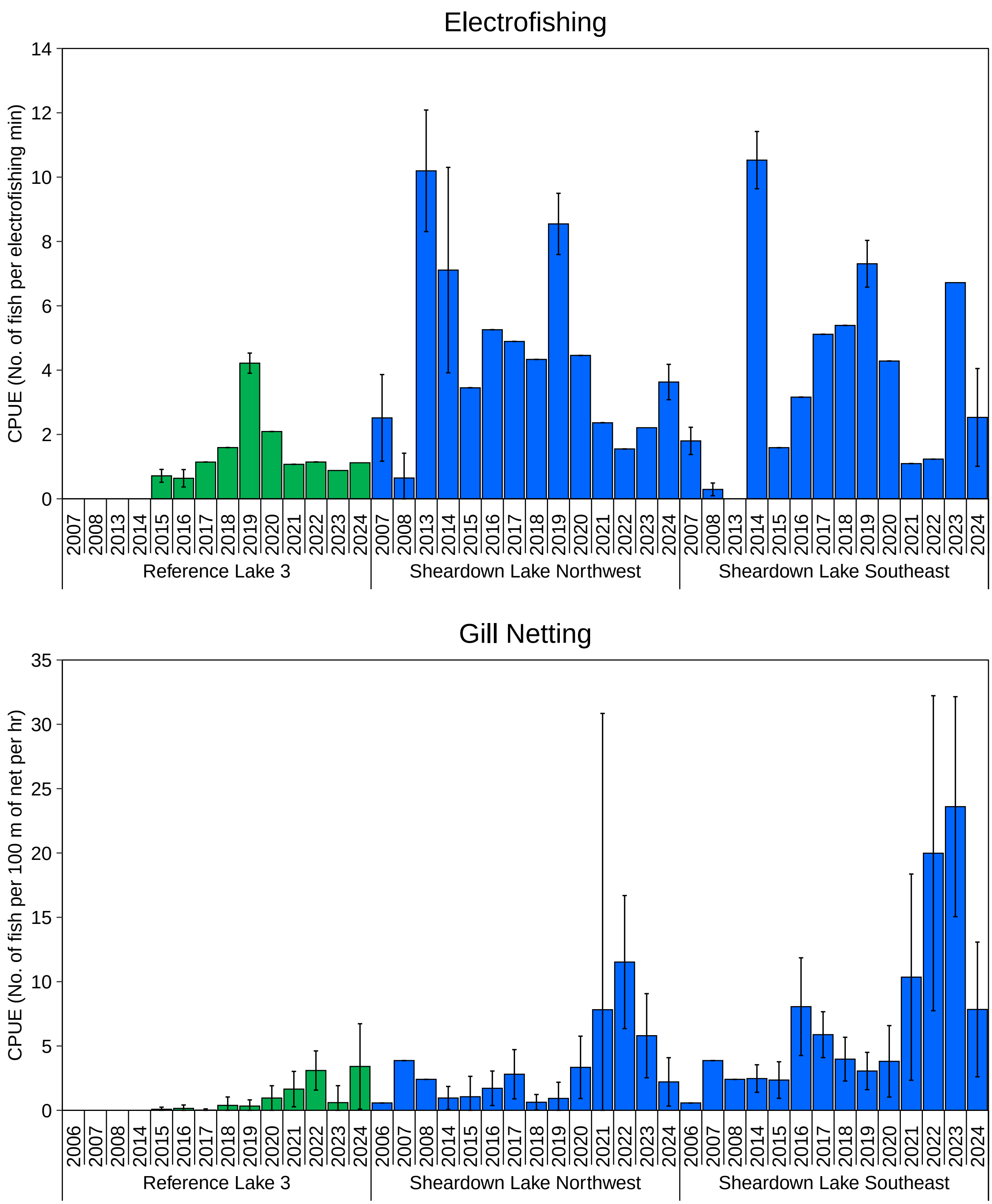


Figure 4.10: Catch−per−unit−effort (CPUE; mean ± standard deviation) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Sheardown Lake NW (DL0−01) and Sheardown Lake SE (DL0−02), Mary River Project CREMP, 2006 to 2024

Notes: Data presented for fish sampling conducted in fall during baseline (2006, 2007, 2008, 2013), construction (2014), and operational (2015 to 2023) mine phases. Lake basins (i.e., NW or SE) were not differentiated for baseline gill netting catches. Reference areas are shown in green and mine−exposed areas are shown in blue.

changes in fish densities at Sheardown Lake NW are not indicated. Ongoing evaluation of CPUE in addition to fish health and lower trophic level endpoints will be used to verify this determination.

4.4.5.2 Fish Health Assessment

Nearshore Arctic Charr

In August 2024, a total of 100 and 102 arctic charr were sampled for assessment of fish health from the nearshore habitats of Sheardown Lake NW and Reference Lake 3, respectively (Appendix Tables G.4 and G.13)[37,38]. Arctic charr YOY were distinguished from older (non-YOY) age classes using a fork length cut-off of 5.0 cm for Sheardown Lake NW and 4.0 cm for Reference Lake 3, based on the evaluation of LFD coupled with supporting length and weight measurements and age determinations (Figure 4.11, Appendix Figure G.8, Appendix Tables G.4 and G.13). Due to fewer than ten YOY arctic charr being captured in Sheardown Lake NW, statistical comparisons of health endpoints could only be made for the non-YOY population, except for comparisons of LFD which were conducted for the full distribution of fish lengths (Figure 4.11, Table 4.8, Appendix Table G.14) and the distribution of non-YOY lengths (Table 4.8, Appendix Figure G.8). Both LFDs for nearshore arctic charr differed significantly between Sheardown Lake NW and Reference Lake 3 (Figure 4.11, Table 4.8; Appendix Table G.14). The Reference Lake 3 distribution had dominant size classes between 4.75 and 5.5 cm (classified as age 1+) and few fish greater than 6.0 cm in length, while the Sheardown Lake NW distribution had relatively equal proportions of fish across all size classes from 5.25 to 14.5 cm, with no individuals classified as YOY (Figure 4.11, Appendix Figure G.8, Appendix Table G.13). Despite a lack of YOY captures in Sheardown Lake NW in 2024, significant differences in LFDs for all fish and non-YOY fish between Sheardown Lake NW and Reference Lake 3 suggest that the relative abundance of YOY is not significantly different between the lakes (Environment Canada 2012). The LFD for nearshore arctic charr from Sheardown Lake NW has consistently been significantly different from that of Reference Lake 3 since 2015 (Table 4.8) though there have been no consistent patterns in the relative frequencies of fish lengths between the lakes (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024).

[37] Sample sizes at Sheardown Lake NW in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.15.

[38] The total number of fish captured in Sheardown Lake NW and Reference Lake 3 by electrofishing (Table 4.7, Appendix Table G.1) was greater than the number of fish sampled for the fish health assessment. The study design targets 100 fish from each lake for sampling (measurement of length and weight; Baffinland 2015). Once field crews were certain that the minimum target sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

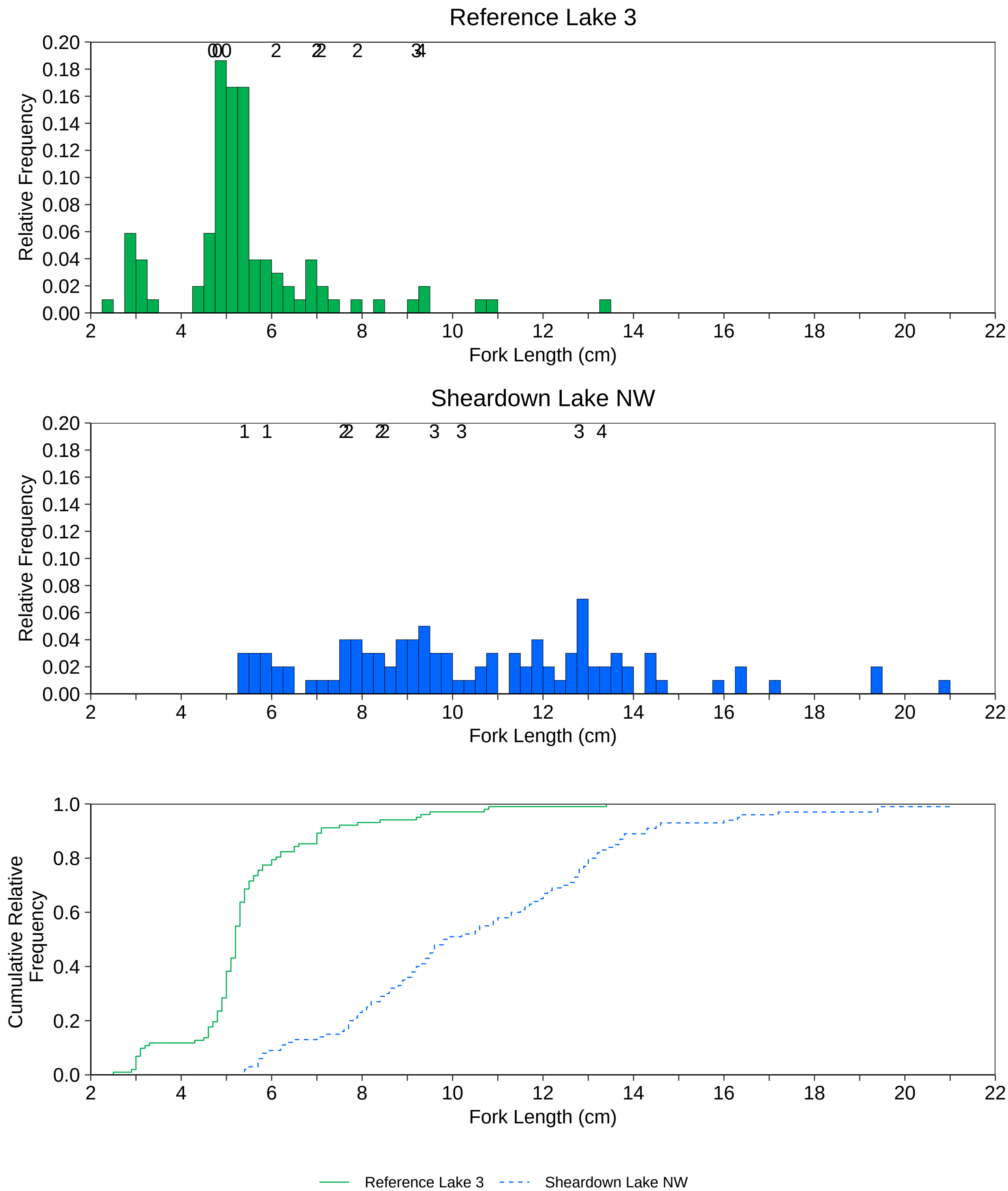


Figure 4.11: Relative Length−Frequency and Cumulative Length−Frequency Distributions for All Arctic Charr Captured by Backpack Electrofishing at Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Fish ages are shown above the bars, where available. Sheardown Lake NW n = 100; Reference Lake 3 n = 102.

Table 4.8: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03), and between Sheardown Lake Northwest Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP, 2015 to 2024

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]																			
			versus Reference Lake 3										versus Sheardown Lake NW baseline period data [b]									
			2015	2016	2017	2018	2019	2020	2021	2022	2023	2024	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Nearshore Electrofishing	Survival [c]	Length-Frequency Distribution	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**
		Age	No	No	No	-	-	-	-	-	-	-	No	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	**Yes (+29%)**	**Yes (+17%)**	**Yes (+20%)**	**Yes (+24%)**	**Yes (-10%)**	**Yes (+22%)**	**Yes (+48%)**	**Yes (+34%)**	No	**Yes (+86%)**	No	No	No	**Yes (-12%)**	No	**Yes (+13%)**	No	No	**Yes (-20%)**	**Yes (+19%)**
		Size (mean weight)	**Yes (+121%)**	**Yes (+60%)**	No	**Yes (+83%)**	**Yes (-24%)**	**Yes (+99%)**	**Yes (+234%)**	**Yes (+115%)**	No	**Yes (+660%)**	No	**Yes (-29%)**	No	**Yes (-50%)**	No	No	No	**Yes (-28%)**	**Yes (-49%)**	No
		Growth (weight-at-age)	**Yes (+156%)**	**Yes (+66%)**	-	-	-	-	-	-	-	-	No	-	-	-	-	-	-	-	-	-
		Growth (fork length-at-age)	**Yes (+38%)**	**Yes (+24%)**	-	-	-	-	-	-	-	-	No	-	-	-	-	-	-	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	**Yes (+7%)**	**Yes (-5%)**	**Yes (+4%)**	**Yes (+10%)**	**Yes (+4.6%)**	**Yes (+5.2%)**	No	**Yes (+8.4%)**	**Yes (-13%)**	**Yes (-12%)**	**Yes (-9%)**	**Yes (-10%)**	**Yes (-13%)**	**Yes (-9%)**	**Yes (-7.5%)**	**Yes (-12%)**	No	**Yes (-16%)**
	Energy Use (YOY)	Size (mean fork length)	-	-	-	-	-	No	**Yes (+20%)**	**Yes (+8.3%)**	-	-	-	-	-	-	-	-	-	-	-	-
		Size (mean weight)	-	-	-	-	-	**Yes (+44%)**	**Yes (+81%)**	**Yes (+24%)**	-	-	-	-	-	-	-	-	-	-	-	-
	Energy Storage (YOY)	Condition (body weight-at-fork length)	-	-	-	-	-	**Yes (+7.2%/+9.0)**	No	No	-	-	-	-	-	-	-	-	-	-	-	-
Littoral/Profundal Gill Netting [d]	Survival	Length Frequency Distribution	-	-	-	No	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	No	**Yes**	No	No	**Yes**
		Age	-	-	-	-	-	-	-	-	-	-	**Yes (-35%)**	**Yes (-28%)**	**Yes (-26%)**	-	-	-	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (+22%)**	**Yes (+18%)**	**Yes (+13%)**	**Yes (+18%)**	No	**Yes (+18%)**	**Yes (-21%)**	**Yes (-14%)**	**Yes (-6%)**	No	No	No	**Yes (-7.7%)**	No	**Yes (-3.8%)**	**Yes (-7%)**
		Size (mean weight)	-	-	-	No	**Yes (+92%)**	**Yes (+94%)**	**Yes (+68%)**	**Yes (+115%)**	No	**Yes (+80%)**	**Yes (-47%)**	**Yes (-31%)**	**Yes (-9%)**	No	No	No	**Yes (-20%)**	No	**Yes (-12%)**	**Yes (-14%)**
		Growth (fork length-at-age)	-	-	-	-	-	-	-	-	-	-	No	No	No	-	-	-	-	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	-	-	-	-	No	No	**Yes (+24%)**	-	-	-	-	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+4%)**	No	**Yes (+11%)**	**Yes (+20%)**	**Yes (+20%)**	**Yes (-14%)**	**Yes (+18%)**	**Yes (+8%)**	**Yes (+11%)**	**Yes (+6%)**	No	No	No	**Yes (+6.0%)**	**Yes (+5.5%)**	No	**Yes (+5.3%)**

BOLD Indicates a statistically significant difference.

Notes: "-" indicates data not available for comparison. YOY = Young-of-the-Year.

[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] The length-frequency distribution for Reference Lake 3 includes all fish, whereas for baseline conditions, it only includes non-YOY fish.

[d] 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.

In 2024, non-YOY arctic charr from Sheardown Lake NW were significantly longer (86%), heavier (660%), and exhibited greater condition (8.4%) relative to those from Reference Lake 3 (Table 4.8, Appendix Figure G.10, Appendix Table G.14). The observed difference in condition did not have an MOD outside of the CES_C of ±10%, indicating it was not ecologically meaningful (Table 4.8, Appendix Table G.14, Appendix Figure G.10). Nearshore arctic charr from Sheardown NW were generally longer and heavier during mine operations years from 2015 to 2024 relative to Reference Lake 3, with the exceptions of 2017, when no significant difference was observed for weight, 2019, when both length and weight were significantly lower, and 2023, when no significant difference was noted for either metric (Table 4.8). MOD values for length and weight of fish from Sheardown Lake NW relative to Reference Lake 3 in 2024 were the highest observed during the mine operational period (2015 to 2023; Table 4.8). Condition of non-YOY arctic charr from Sheardown Lake NW relative to the reference lake has varied from 2015 to 2024, but fish from Sheardown Lake NW have generally had greater condition at MOD within the CES_C and the MOD for condition in 2024 fell within the range observed during the mine operational period (Table 4.8). Greater size and improved condition of arctic charr from Sheardown Lake NW compared to fish from Reference Lake 3 is consistent with the lake's higher productivity relative to the reference lake, as evidenced by higher chlorophyll-a concentrations (Section 4.4.3) and higher benthic invertebrate density (Section 4.4.4). Further, results indicate that potential shifts in the BIC correlated with sedimentation rate and accumulation thickness identified through the Lake Sedimentation Monitoring Program in 2024 (Minnow 2025) have not resulted in changes in food source availability that have adversely affected fish health in Sheardown Lake NW relative to reference conditions.

A significant difference in LFD of non-YOY nearshore arctic charr from Sheardown Lake NW was observed between 2024 and the combined Sheardown Lake NW baseline period which is consistent with annual comparisons to baseline in previous mine operational years (Table 4.8, Appendix Figure G.9). Arctic charr captured at Sheardown Lake NW in 2024 were significantly longer (19%) but not significantly different in weight and therefore exhibited lower condition (-16%) compared to individuals captured during the baseline period (Table 4.8, Appendix Figure G.11, Appendix Table G.7). The observed difference in condition between the 2024 samples and baseline exceeded the CES_C of ± 10%, indicating an ecologically meaningful difference (Table 4.8, Appendix Figure G.11, Appendix Table G.7). Fork length and body weight have shown no consistent pattern relative to baseline over time (Table 4.8) and while differences from baseline for these metrics were frequently not significant, when significant differences were observed, MODs were always outside of the CES_C (Table 4.8). In contrast, except in 2023, body condition has consistently been lower than baseline at MODs near or outside of the CES_C

(Table 4.8). However, MODs do not indicate a consistent ecologically meaningful difference in condition from baseline in consecutive sampling years.

Overall, there have been no consistent changes in non-YOY condition in Sheardown Lake NW relative to Reference Lake 3 since 2015. The absolute MOD for the condition of non-YOY arctic charr in Sheardown Lake NW in 2024 compared to the baseline period was outside of the CES_C, but similar potentially ecologically meaningful differences have not been consistently observed in recent study years (i.e., no significant difference in 2023). Therefore, no adverse mine-related effects on the health of non-YOY arctic charr at Sheardown Lake NW are indicated. This determination will be verified through ongoing annual assessment of fish health.

Littoral/Profundal Arctic Charr

In August 2024, a total of 104 and 84 arctic charr were sampled for fish health assessment from littoral and profundal habitats of Sheardown Lake NW and Reference Lake 3, respectively (Appendix Table G.9 and G.17)[39,40]. The LFD of littoral/profundal arctic charr differed significantly between Sheardown Lake NW and Reference Lake 3, with the lengths of fish captured in Reference Lake 3 being mostly less than 35 cm while the majority of fish captured in Sheardown Lake NW were between 30 and 40 cm long (Table 4.8, Figure 4.12, Appendix TableG.18). The LFD for littoral/profundal arctic charr has generally been significantly different between Sheardown Lake NW and Reference Lake 3 since 2019 (Table 4.8, Figure 4.12) generally reflecting higher relative frequencies of larger fish in Sheardown Lake NW (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024).

Littoral/profundal arctic charr from Sheardown Lake NW were significantly longer (18%) and heavier (80%) than those from Reference Lake 3 in 2024 (Table 4.8, Appendix Figure G.13, Appendix Table G.18). Additionally, the condition of arctic charr from Sheardown Lake NW was significantly higher (by 18%) than that of individuals from Reference Lake 3. The magnitude of this difference exceeded the CES_C of ± 10%, indicating that the difference was ecologically meaningful (Table 4.8, Appendix Figure G.13, Appendix Table G.18). Fork length

[39] Sample sizes at Sheardown Lake NW in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.15.

[40] The total number of fish captured in Sheardown Lake NW by gill netting (Table 4.7, Appendix Tables G.2 and G.16) was greater than the number of fish sampled for the fish health assessment. The study design targets 100 fish from each lake for sampling (measurement of length and weight; Baffinland 2015). Once field crews were certain that the minimum sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

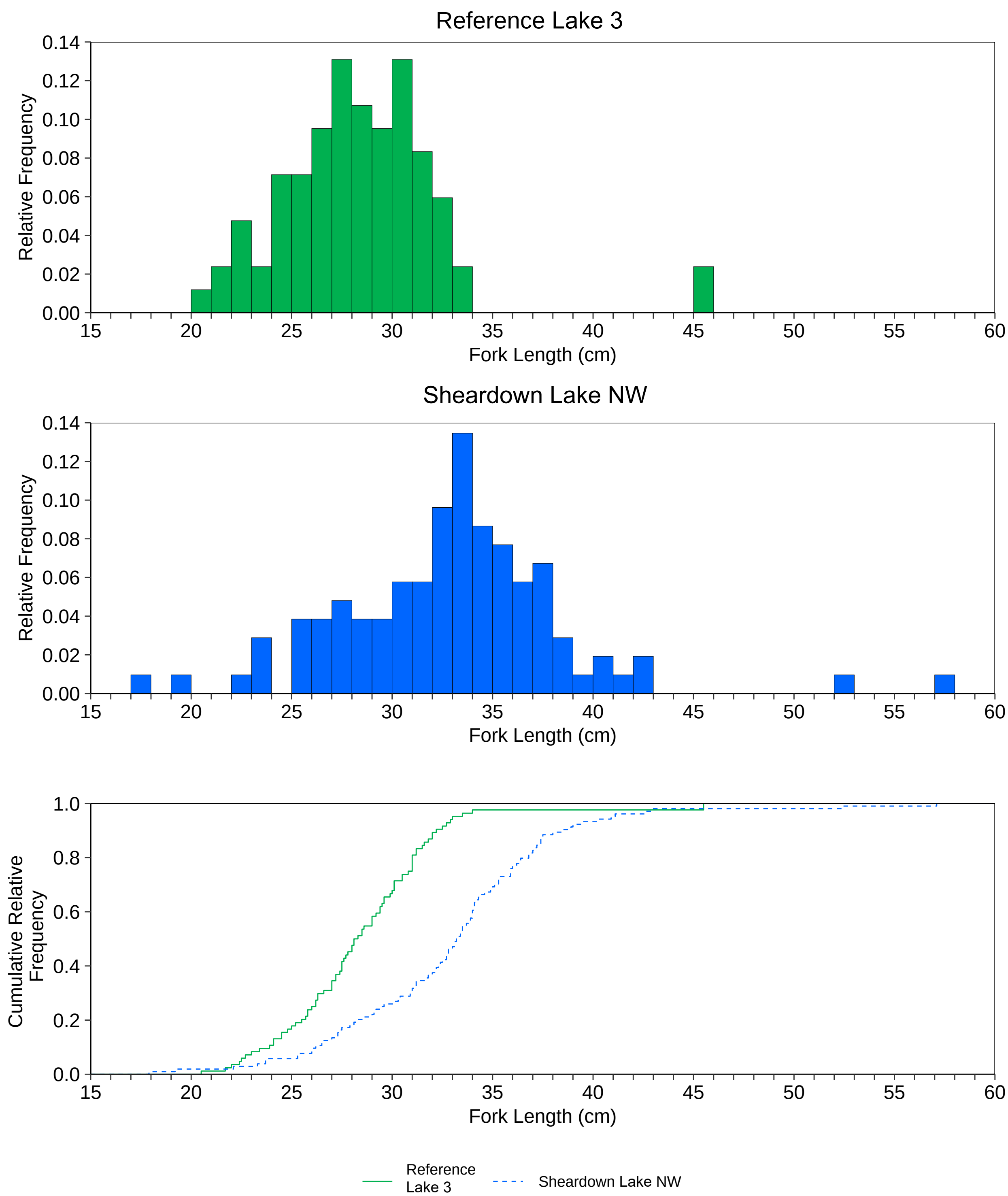


Figure 4.12: Relative Length−Frequency and Cumulative Length−Frequency Distributions for Arctic Charr Captured by Gill Netting at Sheardown Lake Northwest (NW; DL0-01) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Sheardown Lake NW n = 104; Reference Lake 3 n = 84.

and body weight for littoral/profundal arctic charr from Sheardown Lake NW have been consistently greater than those from Reference Lake 3 between 2018 and 2024, with the exception of in 2018 and 2023, when no significant differences were observed (Table 4.8). Condition of littoral/profundal arctic charr in Sheardown Lake NW relative to the reference lake has varied, but most often, condition has been better at Sheardown Lake NW at MOD outside of the CES_C (Table 4.8). MOD values for length, weight, and condition of arctic charr from Sheardown Lake NW relative to Reference Lake 3 in 2024 fell within the ranges observed since 2018 (Table 4.8). Greater size and condition of littoral/profundal arctic charr from Sheardown Lake NW compared to fish from Reference Lake 3 may have been influenced by the lake's higher productivity relative to the reference lake, as evidenced by higher chlorophyll-a concentrations (Section 4.4.3) and higher benthic invertebrate density (Section 4.4.4). However, multiple factors including littoral and profundal fish density and capture efficiency, as well as variation in nearshore fish density, size, and condition may have also been factors.

A significant difference in LFD of littoral/profundal arctic charr from Sheardown Lake NW was observed in 2024 relative to the baseline period, as was the case in some earlier mine operational years (2015, 2016, 2017, 2019, and 2021; Table 4.8, Appendix Figure G.12). Littoral/profundal arctic charr from Sheardown Lake NW were significantly shorter (-7%) and lighter (-14%) in 2024 compared to the baseline period, although they exhibited greater condition (5.3%; Table 4.8, Appendix Table G.18, Appendix Figure G.14). These differences in size reflect a higher proportion of mid-length fish captured in Sheardown Lake NW in 2024, compared to the more uniform distribution over a greater size range observed during the baseline period (Appendix Figure G.12). Although the lengths, weights, and condition of littoral/profundal arctic charr have not differed consistently relative to the baseline period at Sheardown Lake NW, when significant differences were detected, lengths and weights were lower and condition was higher than baseline though the MOD for condition was only outside of the CES_C once (in 2016; Table 4.8)

Arctic charr from littoral and profundal habitats at Sheardown Lake NW have consistently been larger and had greater condition during the mine operational years compared to those from Reference Lake 3. In contrast, while littoral/profundal arctic charr at Sheardown Lake NW over the mine operations period have sometimes been smaller and lighter than the baseline period, generally their condition was higher in these years though not by a great enough magnitude to be outside of the CES_C and thus the difference was not considered ecologically meaningful. These results suggest that there have been no adverse mine-related effects on the health of littoral/profundal arctic charr at Sheardown Lake NW since the onset of mine operations in 2015.

4.4.6 Effects Assessment and Recommendations

In 2024, water chemistry at Sheardown Lake NW met all AEMP benchmarks across all seasonal sampling events (spring, summer, fall). Only one individual sample (Station DL0-01-5 surface water sample in the winter) had a copper concentration that was marginally above the WQG (0.002 mg/L). When comparing water quality parameter concentrations to reference and baseline across all seasons or within a single season, the following parameters were elevated, indicating a potential mine-related effect:

- All seasons (spring, summer, fall): nitrate, sulphate, total/dissolved uranium;
- Summer: chloride, total/dissolved manganese, total/dissolved strontium; and
- Fall: total/dissolved strontium.

None of these parameters exceeded WQGs, and there is no AEMP benchmark for uranium or strontium. Since 2022, concentrations of nitrate, chloride, sulphate, and total and/or dissolved uranium, as well as total and/or dissolved molybdenum (which was only elevated relative to Reference Lake 3 in 2024) have had elevated concentrations relative to reference and baseline (Minnow 2023 and 2024a). In temporal trend analyses completed in the 2023 CREMP total and/or dissolved concentrations of each showed statistically significant increasing trends since the baseline period and over the mine operations period (Minnow 2024a) and visual assessment of temporal data indicated that these increasing trends generally started in 2018 or 2019 and persisted in 2024. Total strontium concentration has also consistently increased during the mine operations period in Sheardown Lake NW, starting in 2019, though no consistent increasing or decreasing patterns in manganese concentrations were observed. Overall, these results indicated a mine-related influence on nitrate, chloride, sulphate, total/dissolved molybdenum and uranium, and total strontium at Sheardown Lake NW in 2024. A special investigation into analysis of total compared to dissolved aqueous concentrations of molybdenum and uranium in 2024 found that the dissolved fraction constituted almost the entire total fraction for both parameters but that there has been no change in the total:dissolved ratio over time, indicating that mine activities have likely not influenced the bioavailability of either parameter. In general, mine-related influences on water chemistry at Sheardown Lake NW in 2024 were attributed to mine infrastructure located in the upstream watershed, primarily influences of the water management at the KM 105 Pond and related influences at SDLT1, a tributary to Sheardown Lake NW.

In 2024, the following sediment quality AEMP benchmarks were exceeded at Sheardown Lake NW:

- Mean iron concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 52,200 mg/kg at all sediment monitoring stations in August (mean = 70,557 mg/kg); and
- Manganese concentrations exceeded the AEMP benchmark of 4,530 mg/kg in one littoral (Station DD-HAB 9-STN; 10,000 mg/kg) and one profundal (Station DL0-01-13; 5,090 mg/kg) sediment sample in August.

Despite these exceedances, manganese concentrations were not elevated compared to reference and baseline, suggesting no mine-related impact on littoral and profundal sediment quality in 2024. It is likely that the elevated concentrations are due to natural processes. For iron in sediment, increasing trends since baseline and over the mine operation were significant in littoral and profundal areas. Iron sediment concentrations from littoral areas were also above the historical range in 2023 and 2024 and a spatial pattern in sediment iron concentration was identified in the lake. Biological monitoring results do not reflect any adverse mine-related effects. Temporal trends as part of the special investigation of iron concentrations in sediment in 2024 suggest the emergence of a mine-related influence on sediment quality in the lake.

No adverse mine-related effects on phytoplankton, BIC, or fish (arctic charr) health were observed at Sheardown Lake NW in 2024, based on comparisons to Reference Lake 3 and to Sheardown Lake NW baseline data.

Under the AEMP Management Response Framework, a Low Action Response is required based on determination of mine-related influences on chloride, nitrate, sulphate, molybdenum, uranium, and strontium due to elevated total and/or dissolved concentrations that were elevated compared to reference and baseline in at least one season in 2024 and/or evidence of increasing trends/patterns over the mine operations period (Figure 2.6). The following actions are recommended:

- In 2025, temporal trend analysis of aqueous total and dissolved (where applicable) concentrations of chloride, nitrate, sulphate, molybdenum, uranium, and strontium will be conducted for Sheardown Lake NW to further investigate temporal trends/patterns.
- Potential sources of chloride, nitrate, sulphate, molybdenum, uranium, and strontium to Sheardown Lake NW will be investigated to better define mine-related influence and the potential for continued contributions.
- Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.

- The focus in 2025 for the KM 105 Pond remediation efforts will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. Given likely influences of water management at the KM 105 Pond on water quality at Sheardown Lake NW (through inputs from SDLT1), water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT1 and Sheardown Lake NW as a basis for informing the potential need for further investigations and mitigation.

Following the AEMP Management Response Framework, a Low Action Response is required based on sediment quality results for iron in Sheardown Lake NW that indicate a mine-related influence. The following actions are recommended:

- In 2025, temporal trend analysis of iron concentrations in littoral and profundal sediment should be repeated with the inclusion of new monitoring data to evaluate whether an increasing trend continues to be identified and to contribute to determination of mine-related influences despite iron sediment concentrations that were similar to reference and baseline conditions in 2024; and
- Further spatial comparisons between iron concentrations in sediment within the lake for the determination of the influence of key lake tributaries on the influx of iron into Sheardown Lake NW.

The absence of confirmed mine-related influences on phytoplankton (as a measure of primary productivity), the BIC, or fish means no further management response is required for these monitoring components at Sheardown Lake NW in 2024 (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at Sheardown Lake NW in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-7 (Camp Management), and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at Sheardown Lake NW conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

Comparison of sediment quality at Sheardown Lake NW in 2024 to FEIS predictions related to Airborne Emission sources (i.e., fugitive dust; FEIS Issue SWSQ-17-3) indicated that all mean parameter concentrations were within the applicable significance rating magnitudes expected for lake sediments during mine operations. Therefore, sediment quality at Sheardown Lake NW conformed with predictions made in Baffinland FEIS (Baffinland 2012).

Water and sediment quality at Sheardown Lake NW in 2024 where parameter concentrations were within applicable FEIS significance rating magnitude predictions also meant that FEIS predictions for (absence of) effects on arctic charr health and condition were also met. Therefore, arctic charr health and condition at Sheardown Lake NW in 2024 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

4.5 Sheardown Lake Southeast (DL0-02)

4.5.1 Water Quality

4.5.1.1 *In Situ* Water Quality

In 2024, profiles were developed from *in situ* water quality measured concurrent with water quality sampling in winter, summer, and fall (Figure 2.1), and *in situ* water quality was measured at the top and bottom of the water column concurrent with benthic invertebrate community sampling in August (Figure 2.3). Vertical profiles of *in situ* water temperature, dissolved oxygen, pH, and specific conductance measured at Sheardown Lake SE showed few substantial within-season differences among stations during any of the winter, summer, or fall sampling events in 2024 (Appendix Figures C.17 to C.20). Specific conductance at DL0-02-6 in the winter and summer and at DL0-02-7 near the surface in summer was higher than at other stations and, at DL0-02-7 in the summer, decreased between one and three meters of depth (Appendix Figure C.20). During August 2024 BIC sampling, there were no differences in *in situ* water quality parameters between littoral and profundal stations within the lake (Appendix Tables C.50 and C.51).

The 2024 Sheardown Lake SE water column temperature profiles indicated a colder layer under the ice, which warmed with depth during the winter sampling event, while there was a decrease in temperature with increasing depth during the summer sampling event (Figure 4.13). Water temperatures were uniform with depth at Sheardown Lake SE in the fall, which is similar to Reference Lake 3 during the summer and fall in 2024 (Figure 4.13). The mean water temperature at the bottom of the water column at Sheardown Lake SE littoral and profundal stations was significantly higher than at Reference Lake 3 during the August 2024 BIC sampling (by ~1.5 and 2.5°C, respectively; Figure 4.14, Appendix Tables C.50 and C.52). Sheardown Lake SE is a smaller and shallower waterbody than Reference Lake 3 (see Figure 1.2, Appendix Table B.1) and therefore heat distribution patterns (i.e., thermal profiles) are expected to differ naturally between these lakes.

Mean dissolved oxygen profiles at Sheardown Lake SE in 2024 generally showed a gradient of decreasing saturation with depth in winter and summer (Figure 4.13). During the fall sampling event, mean dissolved oxygen saturation profiles showed well oxygenated water

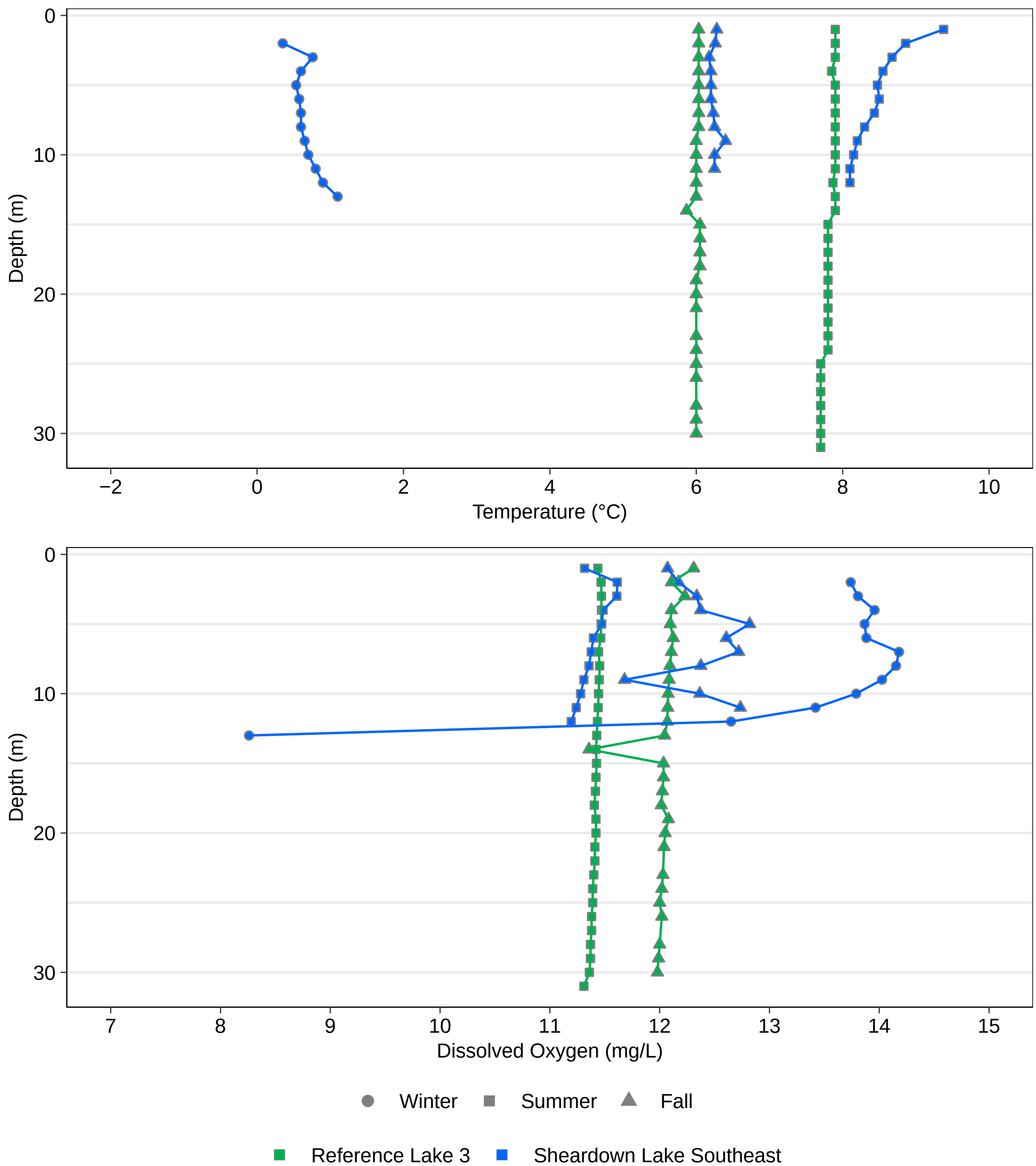


Figure 4.13: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake Southeast Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

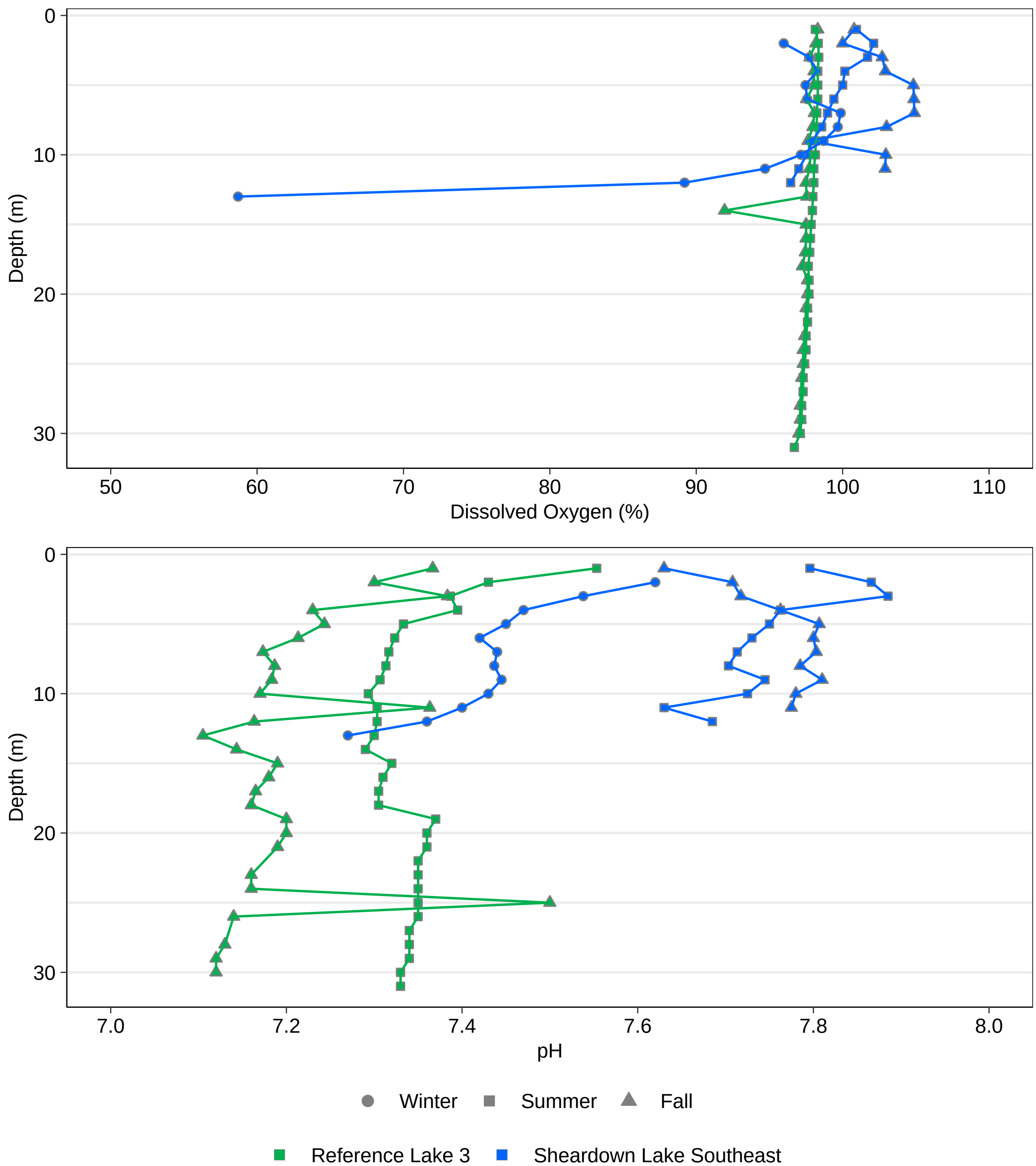


Figure 4.13: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake Southeast Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

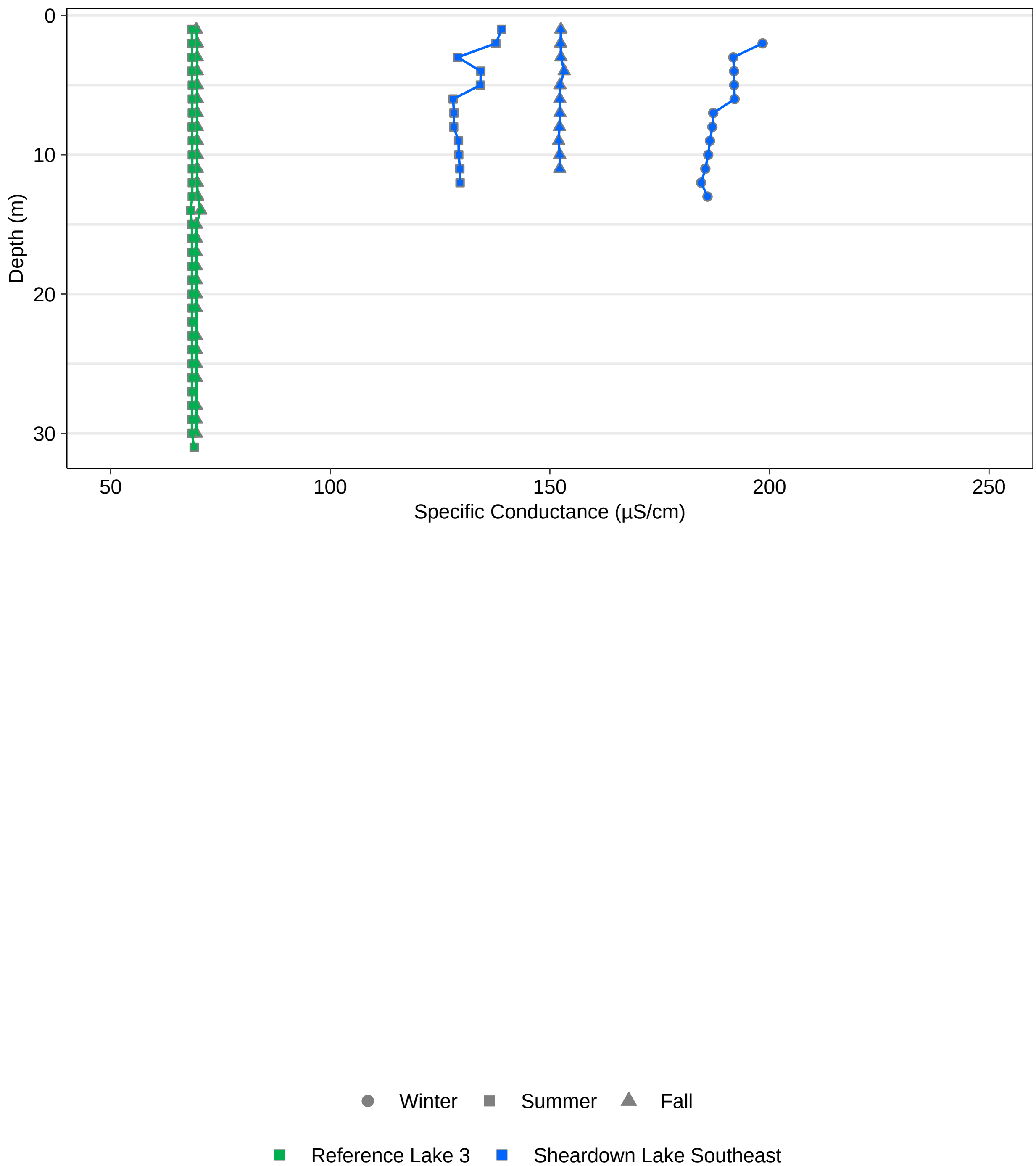


Figure 4.13: Average *In Situ* Water Quality with Depth from Surface at Sheardown Lake Southeast Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

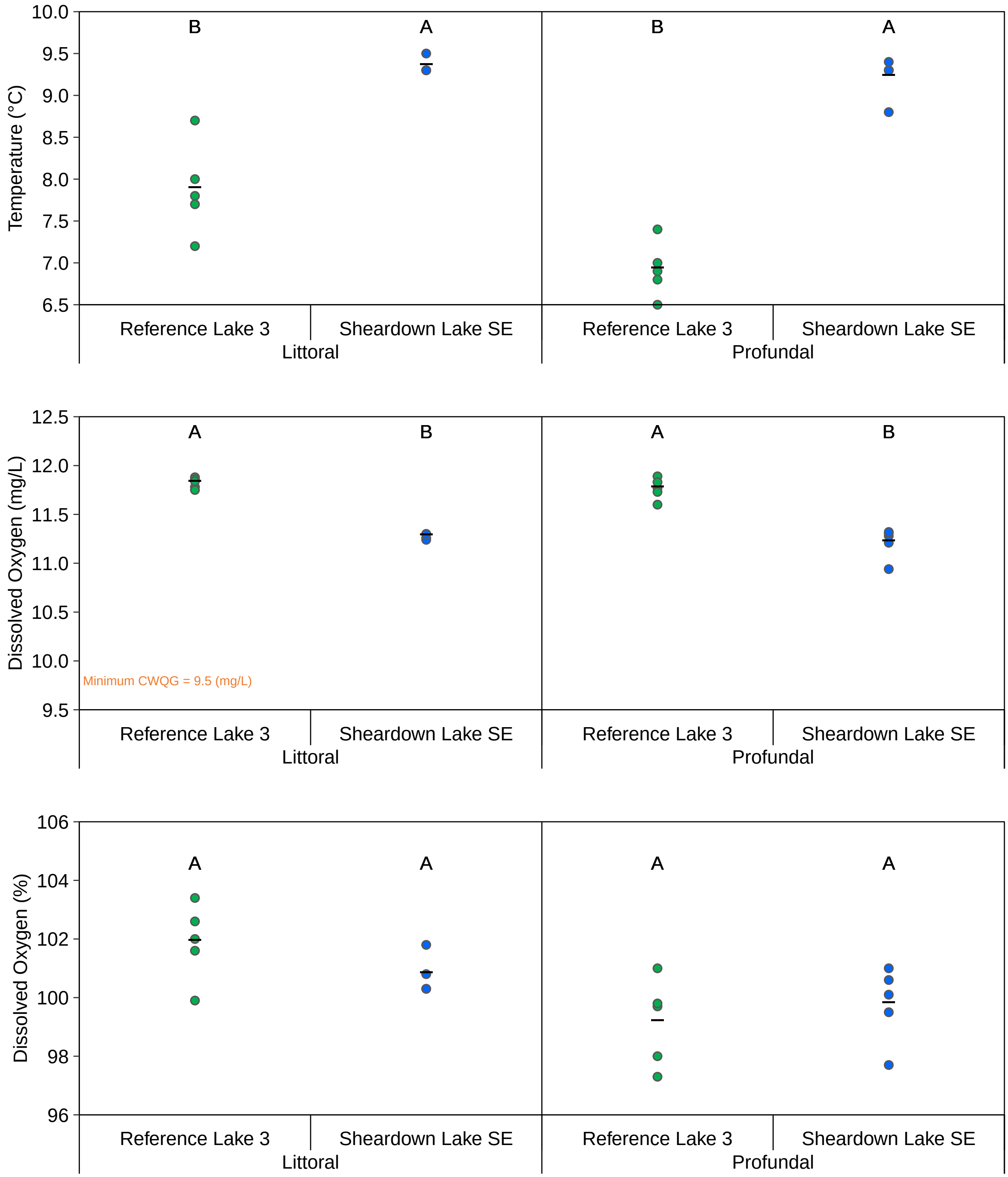


Figure 4.14: Comparison of *In Situ* Water Quality Measured at Sheardown Lake Southeast (SE; DL0−02) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

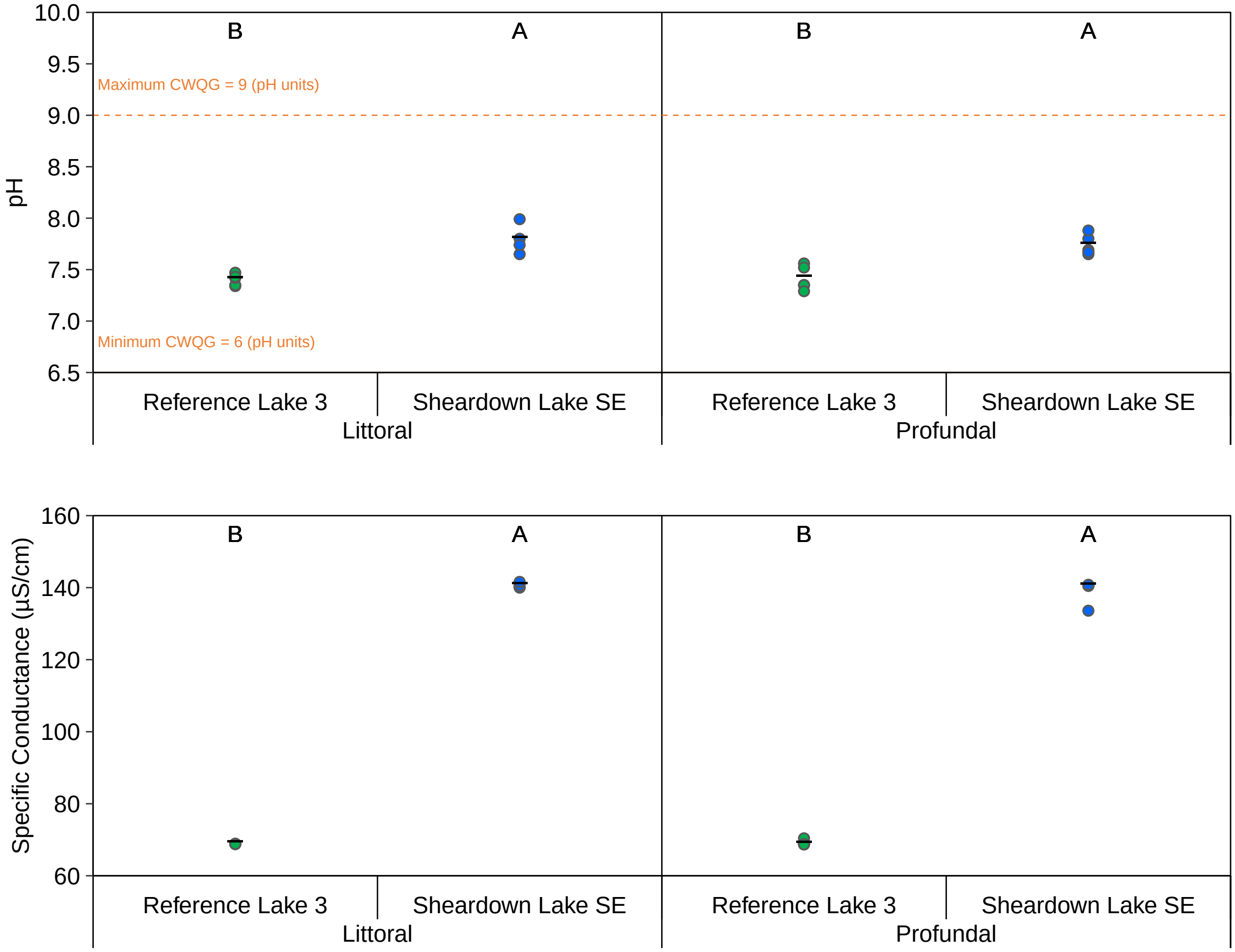


Figure 4.14: Comparison of *In Situ* Water Quality Measured at Sheardown Lake Southeast (SE; DL0−02) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

extending from the surface to the bottom of Sheardown Lake SE, similar to the reference lake (Figure 4.13). Dissolved oxygen concentrations near the bottom of the water column at littoral and profundal stations in Sheardown Lake SE were significantly lower than those at Reference Lake 3 during the August 2024 BIC sampling (Figure 4.14, Appendix Tables C.50 and C.52). However, there were no significant differences in dissolved oxygen saturation between in like-habitats between the two lakes, the absolute magnitude of the differences in concentration were small (i.e., ~0.6 mg/L), and mean dissolved oxygen concentrations were above the WQG for the protection of early life stages of cold-water biota (i.e., 9.5 mg/L) near the bottom at littoral and profundal stations of both Sheardown Lake SE and Reference Lake 3 at the time of BIC sampling (Figure 4.14, Appendix Tables C.50 and C.52).

In 2024, water column mean profiles at Sheardown Lake SE showed a slight decrease in pH with depth in the winter and summer (~ 0.1 to 0.3 pH units), and a slight increase with depth during the fall (~0.2 pH units; Figure 4.13). Comparatively, at Reference Lake 3, water column mean profiles showed a slight decrease in pH with depth (~ 0.5 pH units) in the summer and fall (Figure 4.4). The pH near the bottom at both littoral and profundal stations of Sheardown Lake NW was significantly higher (i.e., more alkaline) than at like-habitat for the reference lake during the August 2024 BIC sampling (Figure 4.14, Appendix Tables C.50 and 5.2). However, pH values were consistently within WQG at Sheardown Lake NW and Reference Lake 3 (Figure 4.5, Appendix Table C.40 and C.42).

Specific conductance profiles at Sheardown Lake SE showed no substantial changes with depth in summer 2024, which was similar to Reference Lake 3 in both summer and fall (Figure 4.13). In winter and summer, mean specific conductance profiles at Sheardown Lake SE decreased slightly with depth, particularly at depths less than 5 m (Figure 4.13). Specific conductance was higher at Sheardown Lake SE in winter compared to other seasons (Figure 4.13), likely due to an absence of dilution originating from tributaries due to their complete freezing. During spring, summer, and fall sampling events, specific conductance was higher at Sheardown Lake SE compared to the reference lake (Figure 4.13). Similarly, during the August 2024 BIC sampling, specific conductance values near the bottom of the water column at littoral and profundal stations were significantly higher at Sheardown Lake SE than at like-habitats at the reference lake (Figure 4.14, Appendix Tables C.50 and C.52). Specific conductance at SDLT9 was elevated relative to reference streams in 2024 (see Section 4.2.1.1) but was similar to other Sheardown Lake SE sampling stations at DL0-02-4, the station nearest to the SDLT9 inflow suggesting limited influence of SDLT9 inflow as a potential source of elevated specific conductance in Sheardown Lake SE (Appendix Figure C.20, Appendix Tables C.47 to 49). The highest specific conductance at sampling station DL0-02-6, which is nearest to the Sheardown Lake NW inflow, suggests a potential influence of the upstream lake on specific conductance in Sheardown Lake SE. Secchi depth readings, which serve as a proxy for water clarity, were significantly lower at Sheardown Lake SE than at Reference Lake 3 during the August 2024 BIC (Appendix Figure C.8; Appendix Table C.52) indicating more suspended particulate material in waters of Sheardown Lake SE.

4.5.1.2 Water Chemistry

Mean and individual sample water chemistry parameter concentrations at Sheardown Lake SE met all AEMP benchmarks and WQGs during spring, summer, and fall sampling events in 2024 (Table 4.9; Appendix Table C.53). Parameters with total and dissolved concentrations that were slightly (3 to 5 times higher), moderately (5 to 10 times higher), or highly (≥ 10 times higher) elevated relative to reference or baseline concentrations are identified in Appendix Tables C.44 and C.46. Concentrations of sulphate were consistently slightly to highly elevated across all seasons when compared to both Reference Lake 3 and baseline concentrations (Appendix Tables C.44 and C.46, Appendix Figure C.21). In the fall, nitrate concentrations were also slightly to highly elevated relative to reference and baseline concentrations (Appendix Tables C.44 and C.46, Appendix Figure C.21).

In 2022, aqueous concentrations of nitrate and sulphate and total and dissolved molybdenum, and uranium were elevated at Sheardown Lake SE compared to reference and/or baseline (Minnow 2023). Under the AEMP Management Response Framework, a Low Action Response was recommended in the form of a temporal trend analysis for these parameters (Minnow 2023) and was completed in 2023 (Minnow 2024a). Significant increasing trends were identified for nitrate, sulphate, and total and dissolved molybdenum and uranium at Sheardown Lake SE stations over the mine operation period (2015 to 2023; all stations), since the construction period (2014 to 2023; Stations DL0-02-7 and DL-02-8), and since baseline (2007 to 2023; Stations DL0-02-3 and DL0-02-4; Minnow 2024a). Visual assessment of temporal data indicated that these increasing trends have persisted in 2024 (Appendix Figure C.21), including for total molybdenum and uranium despite total and dissolved concentrations of these parameters that were only elevated compared to Reference Lake 3 (not relative to baseline) in 2024.

Similar to in Sheardown Lake NW, significant increasing trends in total and dissolved molybdenum and uranium concentrations were observed at Sheardown Lake SE in the 2023 temporal trend analysis and a mine-related influence was identified (Minnow 2024a). This resulted in a recommendation for further response through the investigation of the relationship between total and dissolved aqueous concentrations of these parameters (Minnow 2024a; see Section 2.2.3.2.2).

As at Sheardown Lake NW (see Section 4.4.1.2), for both molybdenum and uranium, a strong relationship between total and dissolved concentrations was observed from 2006 to 2024 at

Table 4.9: Mean Water Chemistry at Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03) Monitoring Stations[a] During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

Parameters		Units	Water Quality Guideline (WQG)[b,c]	AEMP Benchmark[d]	Reference Lake 3 (n = 3)		Sheardown Lake SE Stations (n = 5)		
					Summer	Fall	Winter	Summer	Fall
Conventionals[b]	Conductivity (lab)	µmho/cm	-	-	72.5	72.0	195	146	163
	pH (lab)	pH	6.5 - 9.0	-	7.51	7.50	7.31	7.86	7.88
	Hardness (as $CaCO_3$)	mg/L	-	-	34.8	35.3	90.5	65.2	75.2
	Total Suspended Solids (TSS)	mg/L	-	-	<1	3.30	<1	1.43	<1
	Total Dissolved Solids (TDS)	mg/L	-	-	51.5	41.2	112	75.1	83.9
	Turbidity	NTU	-	-	0.323	0.267	0.144	2.66	0.669
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31.4	36.1	74.1	48.2	55.1
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00738	0.00837	0.00664	0.00667	0.0106
	Nitrate	mg/L	3	3	<0.02	<0.02	0.328	0.240	0.331
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.191	0.145	0.136	0.119	0.122
	Dissolved Organic Carbon	mg/L	-	-	3.62	3.44	2.06	2.01	2.07
	Total Organic Carbon	mg/L	-	-	3.01	3.51	1.91	1.96	2.30
	Total Phosphorus	mg/L	0.020^{α}	-	0.00467	0.00262	0.00304	0.00362	0.00376
	Phenols	mg/L	0.004^{α}	-	<0.001	0.00152	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	1.21	1.21	6.08	4.64	5.18
	Sulphate (SO_4)	mg/L	218^{β}	218	2.72	2.63	14.9	12.8	15.2
Total Metals	Aluminum (Al)	mg/L	0.100	0.179, 0.173[e]	0.0158	0.00605	0.00312	0.0445	0.0235
	Antimony (Sb)	mg/L	0.020^{α}	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.000117	<0.0001	<0.0001	0.000100	<0.0001
	Barium (Ba)	mg/L	1^{β}	-	0.00614	0.00598	0.0108	0.00783	0.00843
	Beryllium (Be)	mg/L	0.011^{α}	-	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	0.0201	0.0125	0.0142
	Cadmium (Cd)	mg/L	0.00012	0.00009	<0.000005	<0.000005	<0.000005	0.00000512	0.00000548
	Calcium (Ca)	mg/L	-	-	6.49	6.40	17.1	12.2	14.0
	Chromium (Cr)	mg/L	0.001	0.003	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Cobalt (Co)	mg/L	0.0009^{α}	0.004	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.000848	0.000823	0.000836	0.000814	0.000861
	Iron (Fe)	mg/L	0.300	0.300	0.0337	0.0112	0.0155	0.0479	0.0264
	Lead (Pb)	mg/L	0.001	0.001	0.0000528	<0.00005	<0.00005	0.0000512	<0.00005
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	0.00192	0.00112	0.00148
	Magnesium (Mg)	mg/L	-	-	4.26	4.48	11.5	8.32	9.68
	Manganese (Mn)	mg/L	0.935^{β}	-	0.00136	0.000602	0.00234	0.00244	0.00260
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.000139	0.000144	0.00119	0.000998	0.00115
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000638	0.000531	0.000595
	Potassium (K)	mg/L	-	-	0.888	0.831	1.84	1.45	1.55
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	<0.00005	0.0000548
	Silicon (Si)	mg/L	-	-	0.487	0.425	0.658	0.599	0.509
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.875	0.843	2.42	1.71	2.00
	Strontium (Sr)	mg/L	-	-	0.00783	0.00754	0.0194	0.0158	0.0193
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.000947	0.000308	<0.0003	0.00232	0.00108
	Uranium (U)	mg/L	0.015	-	0.000273	0.000260	0.00180	0.00145	0.00177
	Vanadium (V)	mg/L	0.006^{α}	0.006	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02^{α}	0.030	<0.003	<0.003	<0.003	<0.003	<0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

Note: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

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

[b] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[c] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

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

Sheardown Lake SE (Appendix Figure H.4). The special investigation indicated that despite increasing total and dissolved concentrations of molybdenum and uranium in Sheardown Lake SE due to mine-related influences, the total:dissolved concentration ratios have not changed over time, meaning mine-related processes contributing to these metal concentrations are not altering the proportion of the bioavailable fraction. Notably, based on a total:dissolved concentration ratio near 1:1 for both molybdenum and uranium since the baseline period, nearly all of their total concentrations were in dissolved form and a high proportion of dissolved metals suggests potential for biological uptake and toxicity (see Section 4.4.1.2 for additional details). However, though they have increased since the baseline period and over the mine operations period, concentrations of both total molybdenum and uranium at Sheardown Lake SE have remained more than an order of magnitude below their respective WQGs since baseline monitoring began (Appendix Figure C.21), suggesting limited risk of adverse effects on aquatic biota.

Overall, nitrate and sulphate concentrations were consistently elevated in Sheardown Lake SE across all sampling seasons in 2024 compared to reference and baseline levels, and increasing temporal patterns were observed, suggesting a mine-related influence on water quality. In addition, increasing concentrations of total and dissolved molybdenum and uranium have also been observed over the baseline, construction, and mine operation periods. However, concentrations of all parameters remained below applicable AEMP benchmarks and WQGs in Sheardown Lake SE in 2024. Most of the mine site infrastructure is located within the Sheardown Lake System catchment, resulting in the potential for mine-related influences from non-point source and airborne emissions (Baffinland 2012) and management plans are in place to manage and mitigate influences in the Sheardown Lake SE catchment associated with site water management, laydown areas, explosives (including the Dyno facility), waste management (including the landfill), and dust deposition. Increasing concentrations of nitrate, sulphate, total molybdenum, and uranium are indicative of a mine-related influence at Sheardown Lake SE, with a potential source being inflow from Sheardown Lake NW, which also had increasing trends in the same parameters over the same periods (see Section 4.1.1.2). Nitrate concentrations may further be influenced by inflow from SDLT9, where concentrations of aqueous nitrogen compounds were elevated compared to reference and baseline concentrations in 2024 and have shown increasing trends over time, likely linked to activities occurring at the adjacent Dyno facility (see Section 4.1.1.2 and Appendix I). While additional investigation is needed to identify other potential sources of these elevated parameters[41], water chemistry in Sheardown Lake SE met

[41] Investigation of potential sources of elevated parameters in Sheardown Lake NW will include consideration of dustfall (EDI 2024) and groundwater (WSP 2024) influences.

AEMP benchmarks and WQG in 2024, indicating limited risk of adverse effects on aquatic biota related to water quality.

4.5.2 Sediment Quality

Surficial sediments (i.e., top 2 cm) collected from sediment coring stations at Sheardown Lake SE in 2024 were primarily composed of silt; two of the littoral samples were also reddish in colour (Figure 4.15; Appendix Table D.16). Sediment samples collected using a Petite Ponar and collected specifically to support the interpretation of BIC data had similar particle size distribution and TOC content as the core samples, except there was less sand in the Petite Ponar, relative to core, samples (Appendix Table D.4). A brown to red-brown layer overlying gray substrates was observed on the surface of sediment samples collected using a Petite Ponar (Appendix Table D.15).

Overall, observations of colour and texture for sediments collected in sediment cores and Petite Ponar grabs from Sheardown Lake SE were comparable to Reference Lake 3 (Appendix Table D.1). However, sediment particle sizes at littoral and profundal stations of Sheardown Lake SE differed significantly from Reference Lake 3; sediments from Sheardown Lake SE generally had finer sediment in both habitat types (i.e., lower proportion of sand and higher proportion of silt and clay) (Appendix Table D.17). The relatively high proportion of fines (i.e., silt and clay) in substrate of Sheardown Lake SE is potentially due to the receipt of Mary River backflow during high flow periods, which is expected to result in the deposition of naturally suspended, fine-grained material. The TOC content of littoral and profundal sediments in Sheardown Lake SE was significantly lower than in sediment at Reference Lake 3 (Appendix Table D.17).

Metal concentrations in sediment at Sheardown Lake SE showed no clear spatial patterns with progression towards the lake outlet in 2024, suggesting no point source inputs of metals to the lake (Appendix Table D.18).[42] Mean concentrations of iron and manganese in littoral and profundal sediments at Sheardown Lake SE were higher than applicable AEMP benchmarks and SQG; however, concentrations were similar to mean concentrations of iron and manganese in sediment at Reference Lake 3, indicating that iron and manganese are naturally elevated in the study area lakes (Table 4.10 and Appendix Tables D.2 and D.19).[43]

[42] The sediment core station closest to the Sheardown Lake SE inlet is DL0-02-1 and the station located closest to the outlet is DL0-02-3.

[43] Iron concentrations in sediments from the reference lake were above AEMP benchmarks and SQG. Similarly, manganese concentrations in sediments from the reference area were above AEMP benchmarks (littoral and profundal) and SQG (littoral only) in 2024 (Table 4.10; Appendix Tables D.2 and D.19).

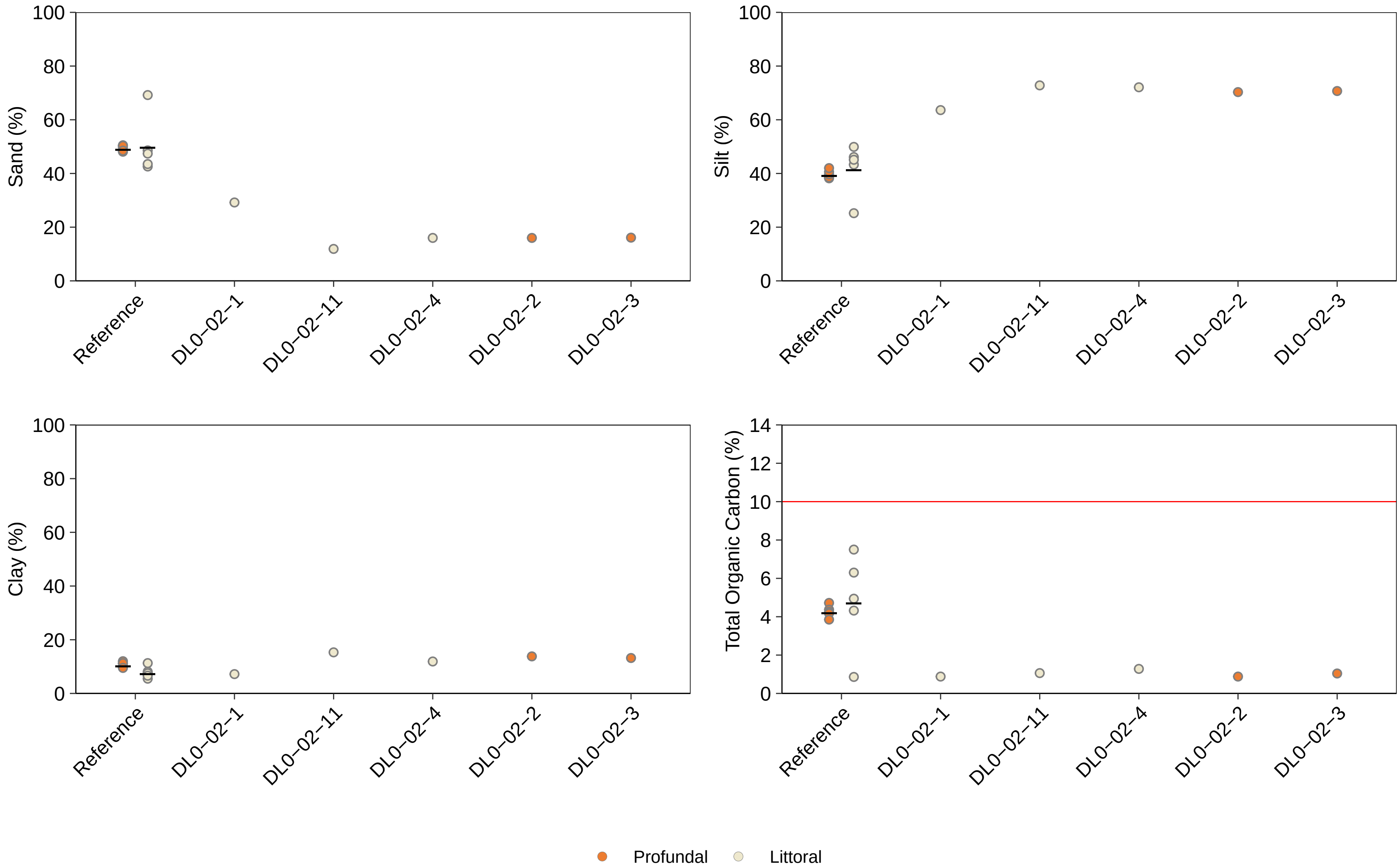


Figure 4.15: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons in Sediment Cores among Sheardown Lake Southeast (SE; DL0-02) Sediment Monitoring Stations and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

Table 4.10: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Sheardown Lake SE (DL0-02) and Reference Lake 3 (REF-03) Sediment Monitoring Stations, Mary River Project CREMP, August 2024

	Parameter	Units	SQG[a]	AEMP Benchmark[b]	Littoral: Reference Lake (n = 5) Average ± SD	Littoral: Sheardown Lake SE (n = 3) Average ± SD	Profundal: Reference Lake (n = 5) Average ± SD	Profundal: Sheardown Lake SE (n = 2) Average ± SD
	TOC	%	10[α]	-	4.78 ± 2.52	1.07 ± 0.200	4.28 ± 0.315	0.960 ± 0.113
Metals	Aluminum (Al)	mg/kg	-	-	16,560 ± 3,306	20,367 ± 2,359	23,060 ± 1,363	19,200 ± 1,273
Metals	Antimony (Sb)	mg/kg	-	-	<0.1 ± -	<0.1 ± -	<0.1 ± -	<0.1 ± -
Metals	Arsenic (As)	mg/kg	17	5.9	5.02 ± 1.55	4.81 ± 1.28	5.07 ± 0.449	3.00 ± 0.00707
Metals	Barium (Ba)	mg/kg	-	-	115 ± 34.7	121 ± 38.4	142 ± 20.5	90.5 ± 4.31
Metals	Beryllium (Be)	mg/kg	-	-	0.646 ± 0.147	0.913 ± 0.162	0.884 ± 0.0586	0.920 ± 0
Metals	Bismuth (Bi)	mg/kg	-	-	<0.2 ± -	0.247 ± 0.0321	<0.2 ± -	0.215 ± 0.00707
Metals	Boron (B)	mg/kg	-	-	13.3 ± 2.05	27.2 ± 6.71	16.7 ± 0.879	30.6 ± 1.06
Metals	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.0497	0.126 ± 0.0250	0.166 ± 0.0166	0.120 ± 0.0163
Metals	Calcium (Ca)	mg/kg	-	-	4,716 ± 728	5,167 ± 911	5,426 ± 237	5,860 ± 56.6
Metals	Chromium (Cr)	mg/kg	90	79	55.1 ± 12.3	***89*** ± 16.3	76.0 ± 4.65	***87.3*** ± 3.54
Metals	Cobalt (Co)	mg/kg	-	-	11.5 ± 2.84	15.2 ± 2.07	17.4 ± 1.70	14.2 ± 0.778
Metals	Copper (Cu)	mg/kg	197	56	***67.5*** ± 21.3	30.8 ± 4.10	***95.1*** ± 8.03	29.6 ± 1.91
Metals	Iron (Fe)	mg/kg	40,000[α]	34,400	***58,760*** ± 25,999	***57,100*** ± 11,303	***49,820*** ± 3,295	***40,400*** ± 1,697
Metals	Lead (Pb)	mg/kg	91.3	35	13.7 ± 1.78	18.4 ± 3.44	18.5 ± 1.01	18.1 ± 0.566
Metals	Lithium (Li)	mg/kg	-	-	25.6 ± 5.12	33.4 ± 5.48	36.2 ± 2.68	36.1 ± 0.707
Metals	Magnesium (Mg)	mg/kg	-	-	11,308 ± 2,124	15,833 ± 1,531	15,780 ± 841	15,650 ± 778
Metals	Manganese (Mn)	mg/kg	1,100[α,β]	657	***862*** ± 611	***2,243*** ± 1,470	***2,246*** ± 2,318	***796*** ± 58.7
Metals	Mercury (Hg)	mg/kg	0.486	0.17	0.0470 ± 0.0233	0.0228 ± 0.00502	0.0702 ± 0.0129	0.0244 ± 0.00233
Metals	Molybdenum (Mo)	mg/kg	-	-	4.63 ± 1.94	2.78 ± 1.05	2.83 ± 0.501	1.50 ± 0.0495
Metals	Nickel (Ni)	mg/kg	75[α,β]	66	39.2 ± 8.63	65.4 ± 10.3	52.2 ± 3.77	62.5 ± 2.40
Metals	Phosphorus (P)	mg/kg	2,000[α]	1,278	***1,344*** ± 713	1,113 ± 133	999 ± 72	918 ± 7.78
Metals	Potassium (K)	mg/kg	-	-	4,118 ± 630	5,143 ± 798	5,600 ± 317	4,970 ± 396
Metals	Selenium (Se)	mg/kg	-	-	0.740 ± 0.278	0.227 ± 0.0379	0.826 ± 0.133	0.210 ± -
Metals	Silver (Ag)	mg/kg	-	-	0.146 ± 0.0462	0.113 ± 0	0.238 ± 0.0192	0.120 ± 0
Metals	Sodium (Na)	mg/kg	-	-	311 ± 48.8	297 ± 60	431 ± 20.9	320 ± 19.1
Metals	Strontium (Sr)	mg/kg	-	-	11.1 ± 1.22	13.5 ± 2.06	13.3 ± 0.458	13.6 ± 0.0707
Metals	Sulphur (S)	mg/kg	-	-	1,620 ± 403	<1,000 ± -	1,360 ± 114	<1,000 ± -
Metals	Thallium (Tl)	mg/kg	-	-	0.423 ± 0.145	0.416 ± 0.0909	0.748 ± 0.0562	0.398 ± 0.00707
Metals	Tin (Sn)	mg/kg	-	-	<2 ± -	<2 ± -	<2 ± -	<2 ± -
Metals	Titanium (Ti)	mg/kg	-	-	958 ± 159	1,433 ± 136	1,164 ± 37.1	1,520 ± 84.9
Metals	Uranium (U)	mg/kg	-	-	15.3 ± 5.91	6.00 ± 1.25	25.1 ± 2.33	5.86 ± 0.438
Metals	Vanadium (V)	mg/kg	-	-	51.2 ± 9.67	54.9 ± 7.06	67.7 ± 3.92	54.8 ± 2.97
Metals	Zinc (Zn)	mg/kg	315	135	72.1 ± 14.9	64.5 ± 6.85	95.2 ± 6.65	60.9 ± 3.32
Metals	Zirconium (Zr)	mg/kg	-	-	4.26 ± 1.70	17.0 ± 4.54	3.92 ± 0.455	24.4 ± 0.212

Indicates parameter concentration above Sediment Quality Guideline (SQG).

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation. "-" indicates data not available.

[a] Canadian SQG for the protection of aquatic life probable effect level (PEL; CCME 2024) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BCMOE 2024).

[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 Sheardown Lake SE

Similar to iron and manganese, mean concentrations of chromium in sediments from Sheardown Lake SE were also above AEMP benchmarks in 2024 (Table 4.10; Appendix Table D.18). Three sediment sampling stations in Sheardown Lake SE (DL0-02-4, DL0-02-12, and DL0-02-13) also had arsenic concentrations that were above AEMP benchmarks, despite the mean concentration being below the AEMP benchmarks (Table 4.10; Appendix Table D.18). Sediment samples with high (relative to AEMP benchmarks and/or SQGs) iron and/or manganese concentrations also had concentrations of arsenic and/or chromium above AEMP benchmarks and/or SQGs, likely due to sorption characteristics of the sediment (Appendix Table D.18). Specifically, iron and manganese (oxy)hydroxides are known to sorb metal cations (e.g., chromium) and anions (e.g., arsenic; Bendell-Young et al. 1992).

Concentrations of other metals in sediments from Sheardown Lake SE were similar to those observed at Reference Lake 3 in 2024. The only exception was zirconium, which had concentrations that were four and six times higher at littoral and profundal areas, respectively, in Sheardown Lake SE relative to Reference Lake 3 (Appendix Table D.19). Analysis of zirconium as a sediment chemistry parameter was initiated as part of the CREMP in 2020, therefore there are no baseline or early mine operation period data available for comparison to results from 2024 and insufficient data for statistical analysis of temporal trends. As in 2024, zirconium concentrations in sediments from Sheardown Lake SE have been 2 to 4 times higher at littoral areas and 4 to 6 times higher at profundal areas than at the reference lake since 2020 (Minnow 2021b, 2022, 2023, and 2024a, Appendix Table D.19). One source of zirconium, the mineral zircon (zirconium orthosilicate), may be found in sedimentary deposits that experience natural weathering processes resulting in alluvial deposits in rivers and streams (Perks and Mudd 2019). Zirconium in sediments in Sheardown Lake SE may therefore originate from a naturally occurring source in the Mary River catchment and from Mary River backflow that enters the lake during high flow periods. The Mary River as a potential source of zirconium in Sheardown Lake SE sediments is consistent with concentrations of zirconium in Mary Lake sediments that have also been elevated by 3 to 6 times compared to Reference Lake 3 since 2020 when analysis of zirconium in sediments began (see Section 5.3.2, Minnow 2021b, 2022, 2023, and 2024a). Zirconium was not identified as a parameter that was anticipated to affect water or sediment quality due to an association with the mine ore bodies, site contact water or runoff, or ore dust in the FEIS (Baffinland 2012) and therefore a mine-related influence on sediment concentrations is not anticipated. Concentrations will continue to be monitored annually for ongoing evaluation of potential mine-related influence.

Mean metal concentrations in sediments at both littoral and profundal stations at Sheardown Lake SE in 2024 were similar to the baseline period (2005 to 2013), except for the concentrations of boron in littoral and profundal sediments and manganese in littoral sediments

(Appendix Table D.19). Boron concentrations in littoral and profundal sediments were approximately 10- and 20-times higher, respectively, in 2024 compared to baseline (Appendix Figure D.1)[44]. Manganese concentrations in littoral sediments were approximately five times higher relative to baseline in 2014 but showed no consistent increasing temporal pattern and were within the range of concentrations observed over the mine operations period (2015 to 2023) and similar to concentrations in Reference Lake 3 (Figure 4.16, Appendix Table D.18).

Concentrations of metals in littoral and profundal sediments collected from Sheardown Lake SE in 2024 were generally within ranges observed in previous years of mine operation (2015 to 2023; Figure 4.16). Additionally, there was no evidence of consistent increasing patterns of metal concentrations, including for boron, in Sheardown Lake SE sediments over the 2015 to 2024 period of mine operation, based on visual examination of the data (Figure 4.16, Appendix Figure D.1). While there was no consistent temporal pattern for mean iron or manganese concentrations in littoral sediments, the mean concentrations measured in 2024 were some of the highest observed since the start of mine operations and, despite similarity to reference conditions, mean iron and manganese concentrations in the sediments of Sheardown Lake SE have consistently been above their respective AEMP benchmarks (Figure 4.16). Therefore, further investigation is recommended to investigate potential mine-related influence (see Section 4.5.6). Additionally, zirconium concentrations were elevated relative to the reference lake in littoral and profundal sediments of Sheardown Lake SE in 2024. Overall, results indicate no substantial mine-related changes in sediment chemistry have been observed at Sheardown Lake SE following the commencement of mine operations in 2015, and that sediment chemistry is likely controlled by natural/geogenic processes, though further investigation into iron and manganese, and ongoing monitoring of zirconium concentrations is recommended.

4.5.3 Phytoplankton

Mean chlorophyll-a concentrations at Sheardown Lake SE in 2024 showed no spatial gradients with distance from the lake inlet during summer, fall, and winter sampling events (Figures 4.8 and 2.2). In 2024, chlorophyll-a concentrations at Sheardown Lake SE were significantly lower in the winter compared to both summer and fall, with no significant difference observed between

[44] Boron concentrations in sediments from 2015 to 2024 were considerably higher (i.e., 10- to 70-times) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences (i.e., probable under-recovery of boron in baseline and 2014). The analytical laboratory used for the baseline study differed from the current laboratory.

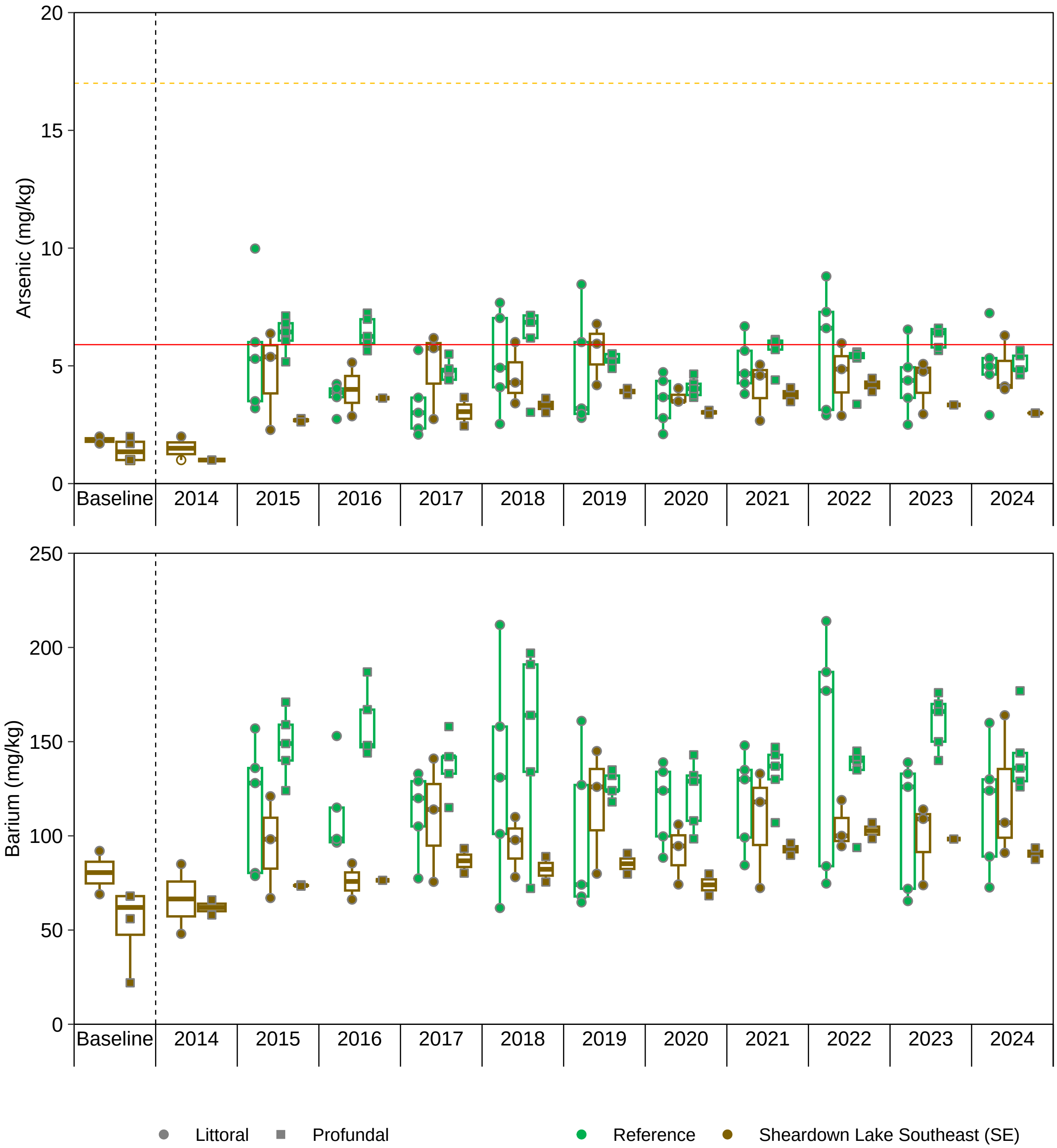
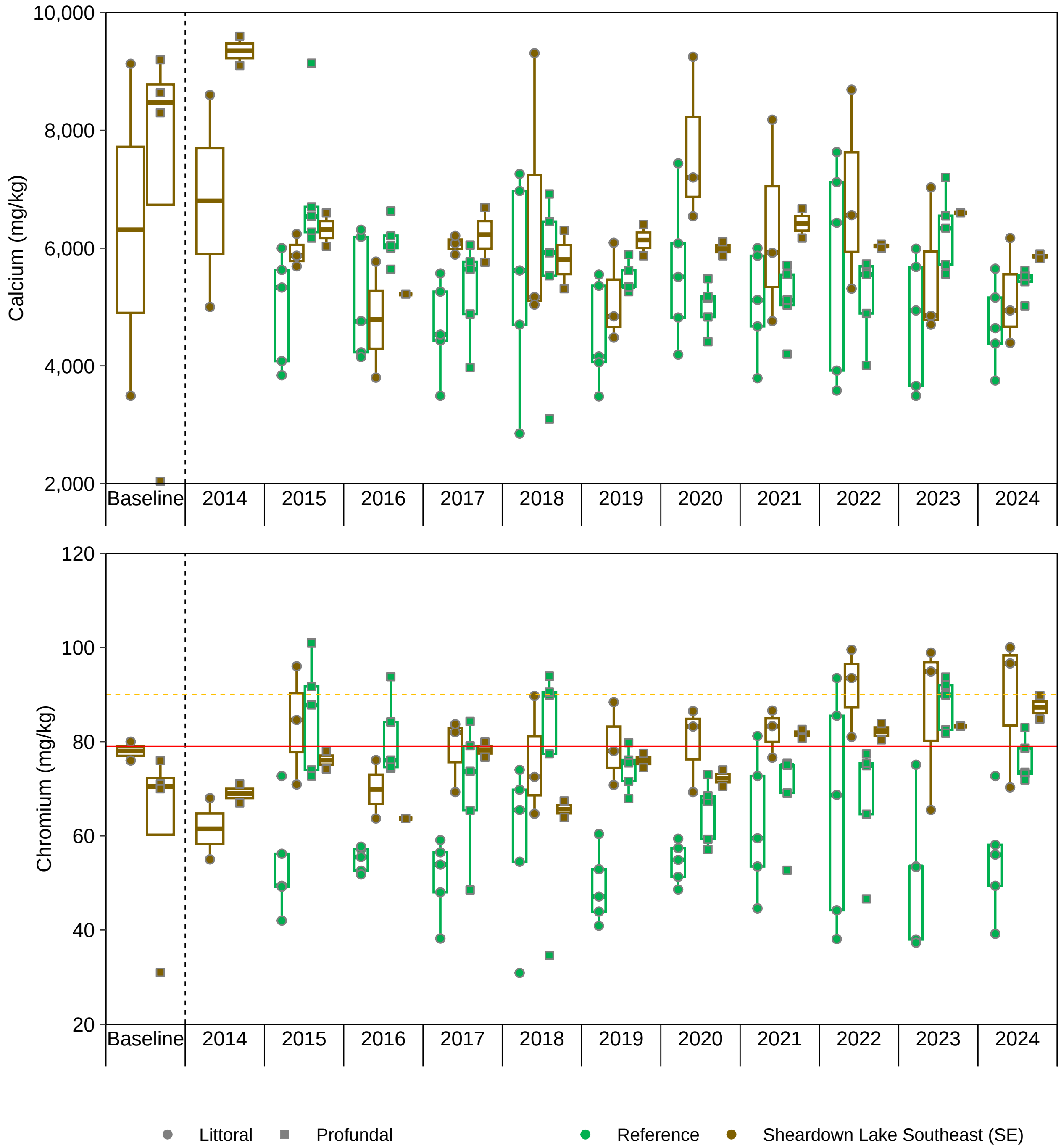


Figure 4.16: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).



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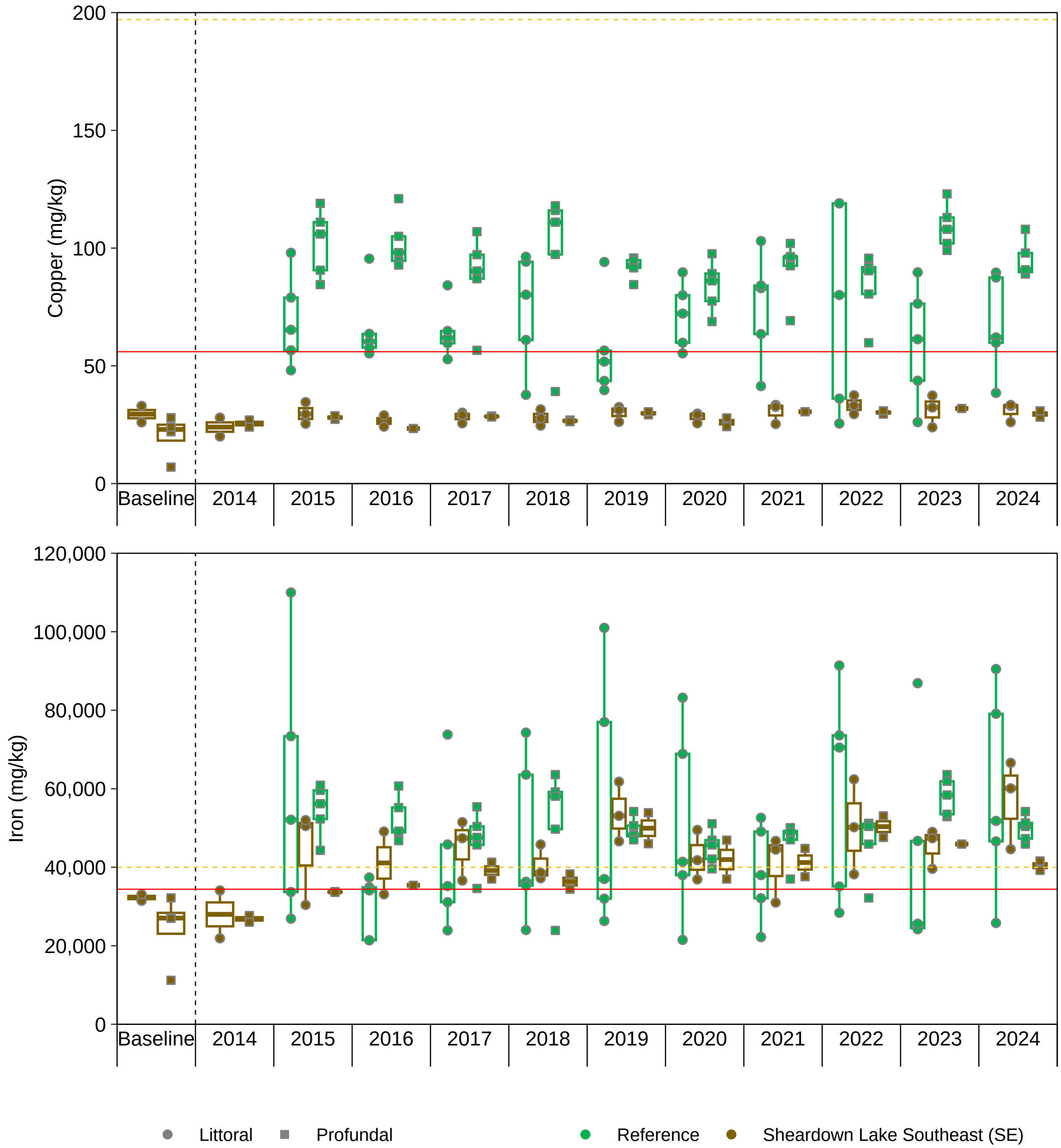

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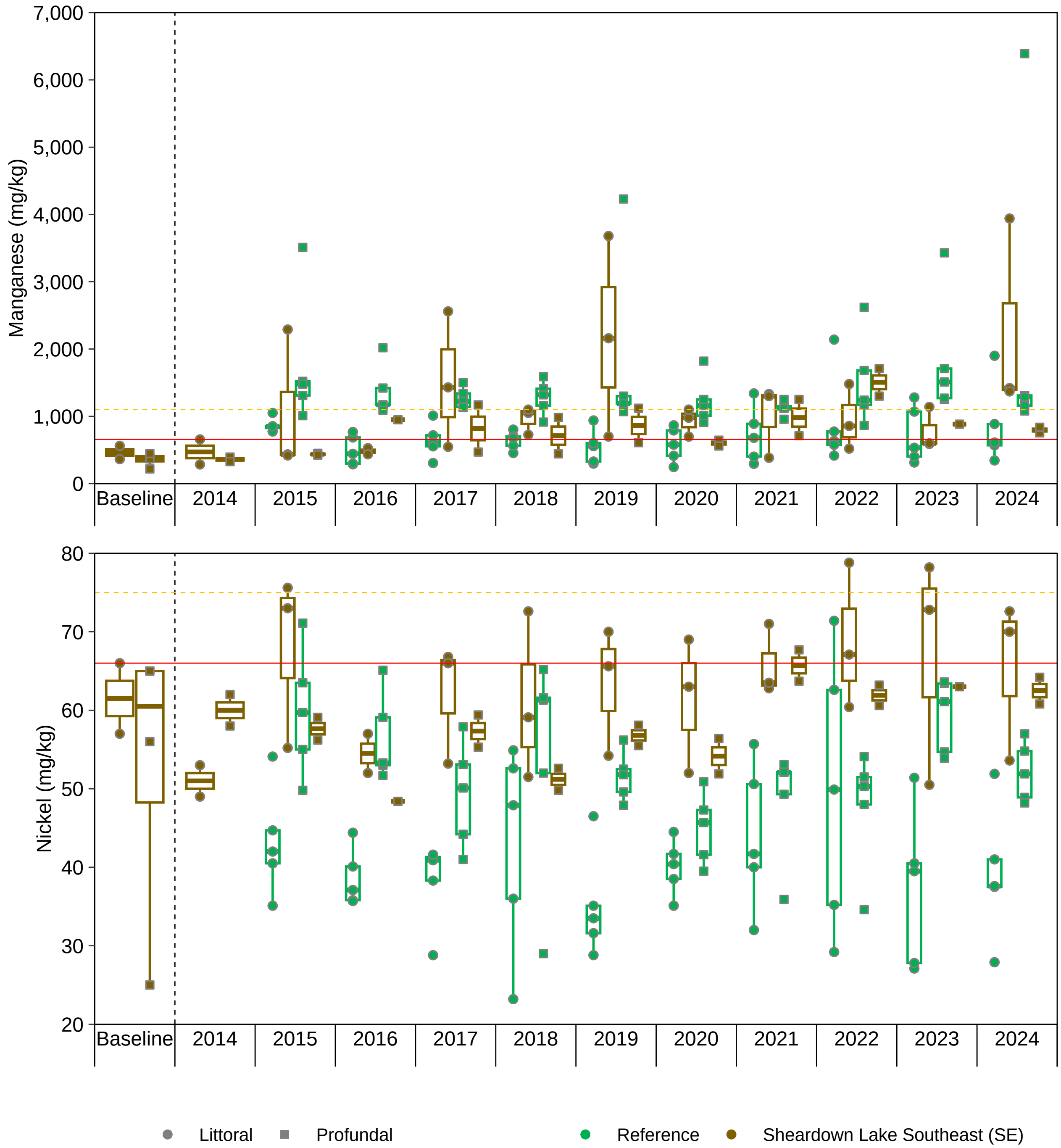

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Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

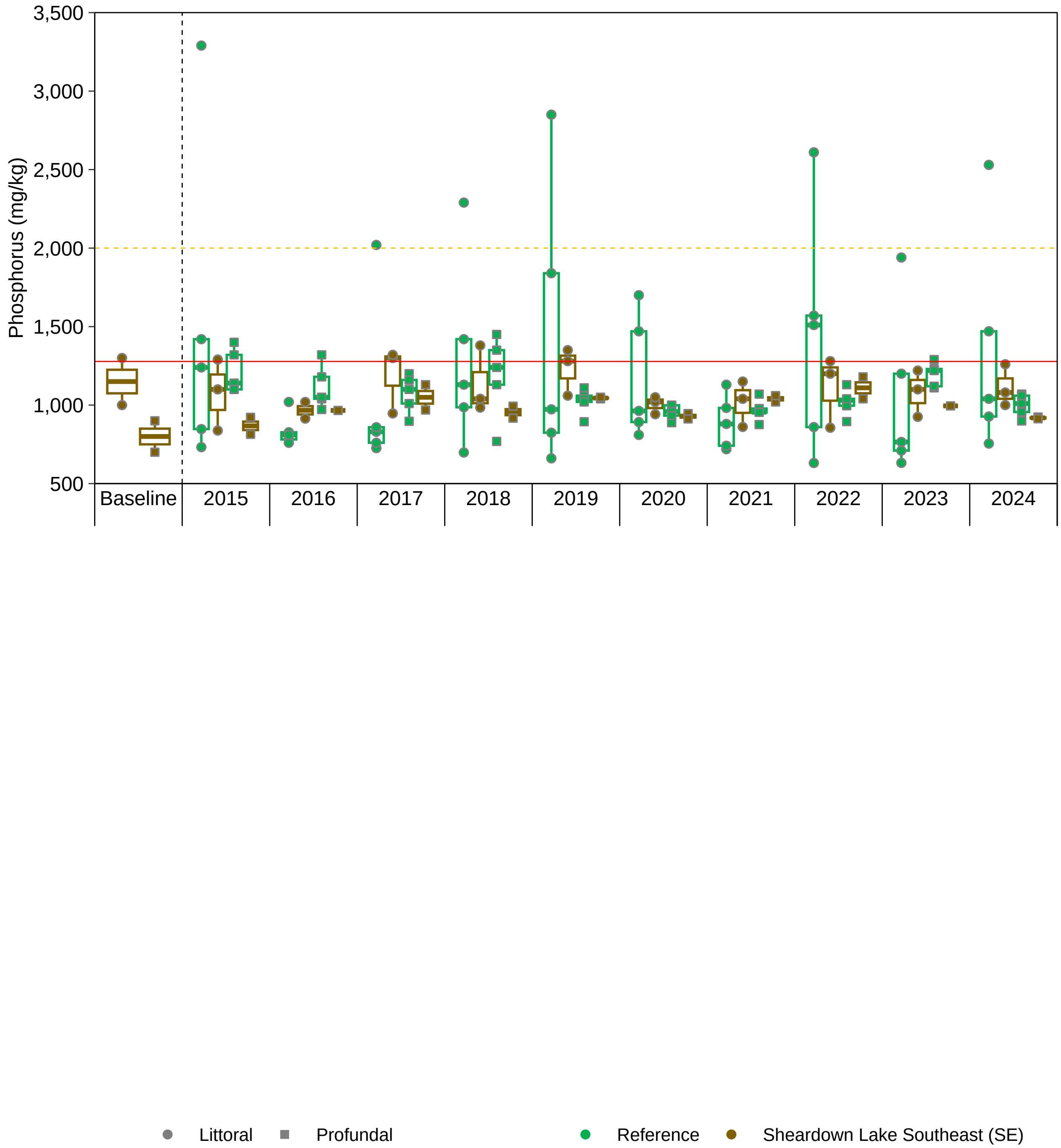

Figure 4.16: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

summer and fall, consistent with the pattern observed at Sheardown Lake NW (Figure 4.8, Appendix Tables E.4 and E.12). As with Sheardown Lake NW, chlorophyll-a concentrations at Sheardown Lake SE were significantly higher than those at the reference lake during both summer and fall sampling events (Figure 4.8, Appendix Tables E.6 and E.7). Despite these elevated concentrations in the summer and fall relative to the reference lake, chlorophyll-a concentrations at all Sheardown Lake SE stations in 2024 remained well below the AEMP benchmark of 3.7 µg/L (Figure 4.8). Chlorophyll-a concentrations <4.5 ug/L (Appendix Table E.12) and total phosphorus concentrations <10 µg/L (Table 4.9, Appendix Table C.53) also indicated an oligotrophic status for Sheardown Lake SE (Wetzel 2001, CCME 2024b; see Section 3.3.3 for additional trophic status classification details).

Although chlorophyll-a concentrations at Sheardown Lake SE varied significantly among years of mine construction and operation, both seasonally and annually, the data demonstrated considerable temporal and seasonal variability (Figure 4.17, Appendix Table E.13). The 2024 concentrations were within seasonal ranges observed from 2014 to 2023, and there were no consistent directional changes across winter, summer, or fall (Figure 4.17, Appendix Table E.13). As in Sheardown Lake NW, chlorophyll-a concentrations in Sheardown Lake SE have consistently been slightly higher than those at Reference Lake 3 during at least one season each year since mining operations began (Figure 4.17; Minnow 2016a, 2017, 2018, 2019, 2020, 2021b, 2022, 2023, 2024a). However, the differences between Sheardown Lake SE and Reference Lake 3 have been and remain minimal (i.e., less than 1.5 µg/L), with both lakes falling within the same trophic classification, suggesting no ecologically relevant mine-related influences on Sheardown Lake SE. The relatively small magnitude and consistency of differences in chlorophyll-a concentrations between the two lakes also suggest that they are due natural factors, such as lake morphology and location (e.g., lake size and fetch which affect lake mixing potential and the amount of sunlight received) rather than mine-related influences.Due to the absence of chlorophyll-a data for the baseline period (2005 to 2013), comparisons to pre-mining conditions could not be made for Sheardown Lake SE.

Overall, chlorophyll-a concentrations in Sheardown Lake SE exhibited no consistent directional temporal patterns in any season, have generally remained consistent relative to concentrations observed at Reference Lake 3 since 2015, and remained well below the AEMP benchmark in 2024. These results indicate no adverse mine-related effects on phytoplankton productivity at Sheardown Lake SE in 2024.

4.5.4 Benthic Invertebrate Community

In 2024, most BIC endpoints for littoral habitats in Sheardown Lake SE were statistically comparable to those for littoral habitats in Reference Lake 3, except invertebrate density,

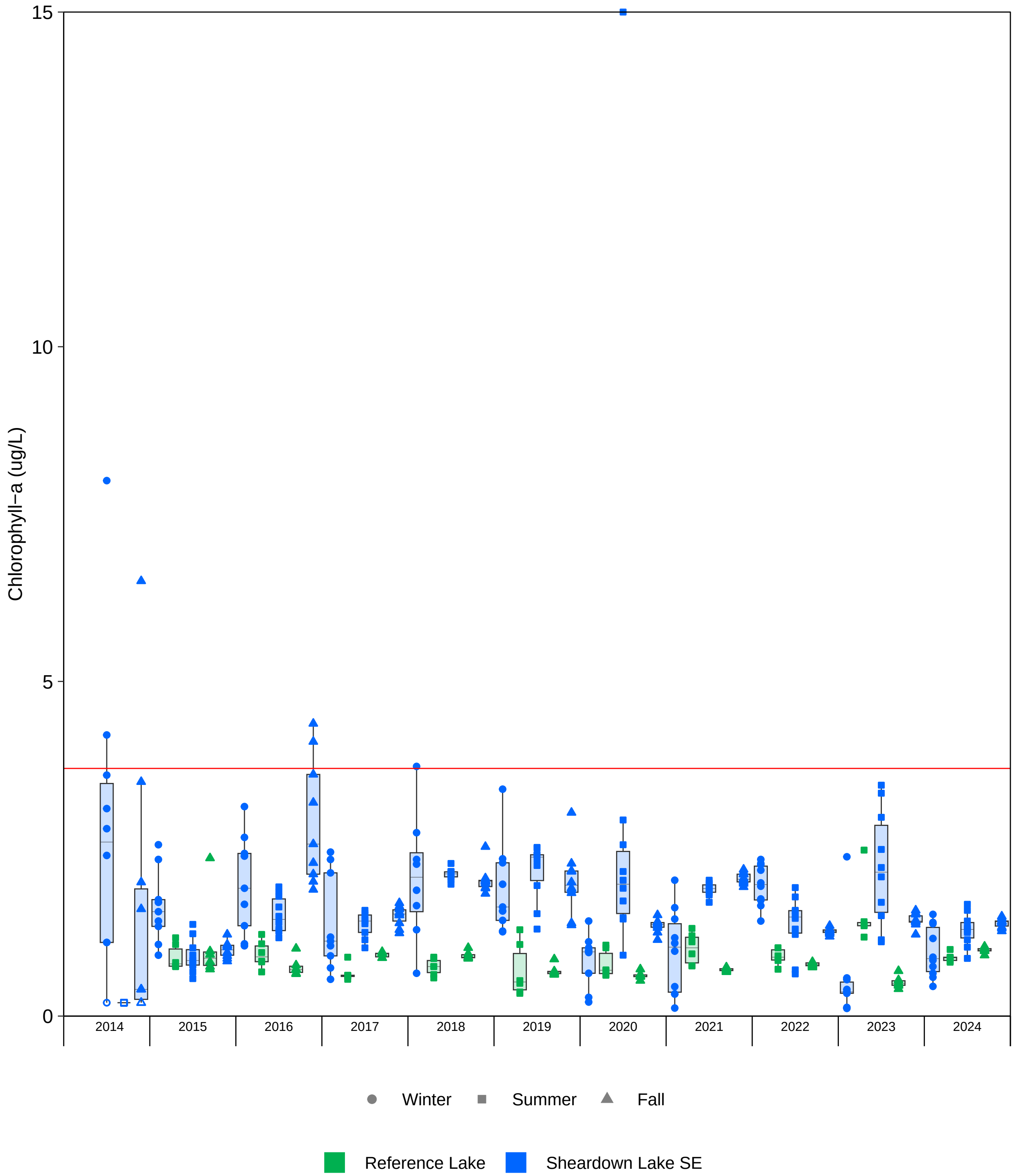

Figure 4.17: Temporal Comparison of Chlorophyll−a Concentrations Among Seasons between Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03) for Construction (2014) and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

richness, Bray-Curtis Index, and relative proportions of *Ostracoda*, *Chironomidae*, collector-gatherers, and burrowers (Table 4.11; Appendix Table F.19). Benthic invertebrate density (MCT = 4,728 individuals/m^2 [DL0-02] and 1,049 individuals/m^2 [REF-03]) and richness (MCT = 13.0 [DL0-02] and 8.8 [REF-03]) were significantly higher in littoral habitats of Sheardown Lake SE compared to similar habitats at Reference Lake 3 (Table 4.11). However, these differences were not ecologically meaningful in 2024, based on a CES_{BIC} of ± 2 SD_{REF} (Table 4.11). In fact, the only difference that was considered ecologically meaningful in 2024, based on littoral BIC data from Sheardown Lake SE and Reference Lake 3 and a CES_{BIC} of ± 2 SD_{REF}, was the relative proportion of *Ostracoda*, which was lower in Sheardown Lake SE relative to reference (Table 4.11). Qualitative assessment of temporal patterns indicates that the proportion of *Ostracoda* has been consistently lower in littoral habitats of Sheardown Lake SE than the reference lake since the start of mine operations in 2015 (Appendix Figure F.11).

Based on comparisons for profundal habitats in 2024, significant differences in BIC endpoints between Sheardown Lake SE and Reference Lake 3 were identified for invertebrate density, richness, Shannon's Diversity Index, and relative proportions of collector-gatherers, sprawlers, and burrowers (Table 4.12). For the other BIC endpoints, no significant differences relative to reference were identified for profundal habitats of Sheardown Lake SE in 2024 (Table 4.12). Benthic invertebrate density (MCT = 2,010 individuals/m^2 [DL0-02] and 202 individuals/m^2 [REF-03]) and richness (MCT = 9.6 [DL0-02] and 4.4 [REF-03]) were significantly higher in Sheardown Lake SE versus Reference Lake 3 profundal habitats (Table 4.12). Further, the differences for density and richness exceeded the CES_{BIC} of ± 2 SD_{REF}, making them ecologically meaningful (Table 4.12). Differences between the proportions of sprawlers and burrowers in profundal habitats were also ecologically meaningful; Sheardown Lake SE had a lower proportion of sprawlers and a higher proportion of burrowing taxa relative to reference (Table 4.12). These compositional differences were evident in the Bray-Curtis Index, with significant differences observed between Sheardown Lake SE and Reference Lake 3 (Appendix Table F.19).

The differences between relative proportion of FFGs and HPGs in both littoral and profundal habitats of Sheardown Lake SE and Reference Lake 3 may reflect lower TOC in sediment at Sheardown Lake SE, relative to reference (see Section 4.3.2; Figure 4.15, Table 4.10). Because TOC is an organic food source for invertebrates, low sediment TOC can limit food resources for collector-gatherers, leading to a lower proportion of this FFG (Merritt et al. 2008). Additionally, the higher proportion of burrowers in Sheardown Lake SE relative to reference may be related to the higher proportion of silt in Sheardown Lake SE sediment compared to Reference Lake 3 (Figure 4.15, Appendix Table D.17). The higher silt content is likely a natural characteristic of Sheardown Lake SE, influenced by high turbidity backflow from the Mary River

Table 4.11: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Littoral Habitats in Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake Littoral Habitat	MCT (n = 5)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	log10	YES	0.005	1.9	Reference Lake 3	1,049	901	403	215	982	2,514
						Sheardown Lake Southeast (SE)	4,728	1,639	733	2,428	4,633	6,398
Richness (Number of Taxa)	tequal	log10	YES	0.069	1.1	Reference Lake 3	8.80	3.56	1.59	5.00	8.00	13.0
						Sheardown Lake Southeast (SE)	13.0	2.55	1.14	9.00	14.0	15.0
Simpson's Evenness (E)	tequal	none	NO	0.723	0.40	Reference Lake 3	0.759	0.0621	0.0278	0.669	0.755	0.840
						Sheardown Lake Southeast (SE)	0.783	0.137	0.0613	0.570	0.793	0.935
Shannon's Diversity	tequal	none	NO	0.158	1.6	Reference Lake 3	2.01	0.243	0.109	1.72	2.01	2.28
						Sheardown Lake Southeast (SE)	2.39	0.492	0.220	1.87	2.20	3.01
Hydracarina (%)	tequal	log10(x+1)	NO	0.416	0.45	Reference Lake 3	2.49	2.88	1.29	0	2.33	7.02
						Sheardown Lake Southeast (SE)	3.77	1.82	0.813	1.49	3.10	6.03
Ostracoda (%)	tequal	log10	YES	0.047	-3.2	Reference Lake 3	39.5	16.1	7.20	16.0	40.3	60.5
						Sheardown Lake Southeast (SE)	13.7	13.9	6.22	1.06	12.3	35.4
Chironomidae (%)	tequal	log10	YES	0.020	1.5	Reference Lake 3	52.6	14.4	6.46	30.7	56.6	68.0
						Sheardown Lake Southeast (SE)	82.0	13.5	6.04	60.5	82.1	94.3
Metal Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.781	-0.17	Reference Lake 3	22.1	17.5	7.81	0	17.3	41.3
						Sheardown Lake Southeast (SE)	18.9	15.2	6.81	4.61	16.2	39.2
Collector Gatherers (%)	tequal	log10	YES	0.071	-2.0	Reference Lake 3	74.9	18.1	8.08	56.4	77.0	100
						Sheardown Lake Southeast (SE)	49.1	19.0	8.48	22.6	50.6	73.0
Filterers (%)	tequal	log10(x+1)	NO	0.793	-0.16	Reference Lake 3	21.7	17.6	7.85	0	17.3	41.3
						Sheardown Lake Southeast (SE)	18.7	15.4	6.91	3.90	16.2	39.2
Shredders (%)	tequal	log10(x+1)	NO	0.432	0.50	Reference Lake 3	0.344	0.578	0.259	0	0	1.33
						Sheardown Lake Southeast (SE)	0.630	0.519	0.232	0	0.557	1.42
Clingers (%)	tequal	log10(x+1)	NO	0.714	-0.21	Reference Lake 3	24.0	17.9	8.01	0	19.0	43.6
						Sheardown Lake Southeast (SE)	19.7	14.1	6.32	9.22	12.3	42.3
Sprawlers (%)	tequal	none	NO	0.237	-0.89	Reference Lake 3	66.2	16.9	7.56	43.2	74.5	84.0
						Sheardown Lake Southeast (SE)	51.2	20.1	9.01	23.4	53.5	76.3
Burrowers (%)	tequal	log10	YES	0.063	1.3	Reference Lake 3	9.82	6.40	2.86	2.30	8.16	16.9
						Sheardown Lake Southeast (SE)	29.1	23.6	10.6	11.4	16.7	67.4

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref}$. MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

Table 4.12: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Profundal Habitats in Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake Littoral Habitat	MCT (n = 6)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tunequal	none	YES	0.001	54	Reference Lake 3	202	33.7	15.1	146	207	233
						Sheardown Lake Southeast (SE)	2,010	524	234	1,361	2,110	2,609
Richness (Number of Taxa)	tequal	log10	YES	<0.001	3.0	Reference Lake 3	4.40	1.14	0.510	3.00	4.00	6.00
						Sheardown Lake Southeast (SE)	9.60	1.67	0.748	7.00	10.0	11.0
Simpson's Evenness (E)	tequal	log10	NO	0.126	0.80	Reference Lake 3	0.582	0.169	0.0754	0.457	0.508	0.867
						Sheardown Lake Southeast (SE)	0.699	0.0597	0.0267	0.637	0.697	0.787
Shannon's Diversity	tequal	none	YES	0.006	1.8	Reference Lake 3	1.28	0.318	0.142	0.834	1.27	1.71
						Sheardown Lake Southeast (SE)	1.86	0.143	0.0639	1.67	1.88	2.00
Hydracarina (%)	tequal	log10(x+1)	NO	0.979	-0.013	Reference Lake 3	4.09	5.94	2.66	0	0	13.0
						Sheardown Lake Southeast (SE)	3.90	1.90	0.850	0.660	4.35	5.71
Ostracoda (%)	tequal	log10	NO	0.110	-5.2	Reference Lake 3	8.37	2.08	0.929	5.88	8.33	11.5
						Sheardown Lake Southeast (SE)	5.34	6.78	3.03	0.330	1.27	16.3
Chironomidae (%)	tequal	none	NO	0.288	0.71	Reference Lake 3	85.2	7.71	3.45	76.9	85.2	94.1
						Sheardown Lake Southeast (SE)	90.7	7.44	3.33	79.3	92.7	99.0
Metal Sensitive Chironomidae (%)	M-W	rank	NO	0.841	-0.38	Reference Lake 3	9.98	11.3	5.05	0	7.41	29.4
						Sheardown Lake Southeast (SE)	6.38	3.13	1.40	2.53	5.95	11.2
Collector Gatherers (%)	tequal	none	YES	0.026	-1.8	Reference Lake 3	85.2	16.2	7.24	57.6	88.2	100
						Sheardown Lake Southeast (SE)	56.1	17.5	7.81	34.1	60.0	77.5
Filterers (%)	M-W	rank	NO	0.398	-[a]	Reference Lake 3	6.70	12.8	5.72	0	0	29.4
						Sheardown Lake Southeast (SE)	4.86	4.02	1.80	0.633	3.99	11.2
Shredders (%)	M-W	rank	NO	1.000	-[a]	Reference Lake 3	0.833	1.86	0.833	0	0	4.17
						Sheardown Lake Southeast (SE)	0.108	0.242	0.108	0	0	0.541
Clingers (%)	M-W	rank	NO	0.398	-[a]	Reference Lake 3	9.96	18.4	8.23	0	0	42.4
						Sheardown Lake Southeast (SE)	8.75	4.67	2.09	4.65	7.03	15.6
Sprawlers (%)	tequal	none	YES	0.003	-2.7	Reference Lake 3	86.2	16.7	7.46	57.6	88.9	100
						Sheardown Lake Southeast (SE)	41.6	17.7	7.91	18.8	41.1	66.7
Burrowers (%)	tequal	none	YES	0.002	9.7	Reference Lake 3	3.88	4.71	2.11	0	3.70	11.5
						Sheardown Lake Southeast (SE)	49.7	22.1	9.90	17.8	51.9	76.5

(grey box) P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = (MCT_{Exp} - MCT_{Ref})/$SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

[a] Contrast MODs could not be calculated because the MAD = 0.

during periods of high flow. Overall, supporting water and sediment quality data do not pinpoint specific mine-related chemical (i.e., versus physical habitat) influences affecting these groups.

Similar to Sheardown Lake NW, higher BIC density and richness, combined with elevated primary productivity during the growing season (summer and fall), indicate that Sheardown Lake SE is more biologically productive than Reference Lake 3 (see Section 4.5.3). Further, as noted above, differences in sediment characteristics between Sheardown Lake SE and the reference lake likely contribute to the observed differences in BIC endpoints. The shallower profundal depths at Sheardown Lake SE compared to Reference Lake 3 may also play a role in the observed differences in BIC. Natural depth-related influences on BIC structure are well documented, with lower density and richness typically found at greater depths in lake environments (Ward 1992; Armitage et al. 1995). The maximum depth of Sheardown Lake SE is approximately 15 m, with profundal samples collected near this depth (Appendix Table D.15), whereas the profundal stations at Reference Lake 3 have a mean depth of 20 m.

Benthic invertebrate density, richness, Simpson's Evenness, and relative proportions of *Ostracoda*, metal sensitive *Chironomidae*, collector-gatherers, and filterers in littoral habitats of Sheardown Lake SE showed significant differences during years of mine operation (2015 to 2024) when compared to baseline (2013; Appendix Table F.35, Appendix Figure F.12). However, few of these differences were ecologically meaningful based on a MOD outside of the CES_{BIC} of $\pm 2\ SD_{REF}$ (i.e., lower total invertebrate densities and higher relative proportions of collector-gatherers in the littoral BIC of Sheardown Lake SE in 2024, relative to 2013; Appendix Table F.35). Further, reported values for most remaining endpoints were generally similar among mine operational years (i.e., from 2015 to 2024), despite being different relative to 2013 baseline, which signals a potential step change or shift in BIC endpoints between 2013 and 2015.

Ecologically significant differences in profundal BIC endpoints for mine operational (2015 to 2024) versus baseline (2007 and 2013) years of data for Sheardown Lake SE were identified for invertebrate density, Simpson's Evenness, and relative proportions of *Ostracoda* and metal sensitive *Chironomidae* (Appendix Table F.36, Appendix Figure F.13). Invertebrate densities in the profundal habitat were almost consistently meaningfully lower during all mine operational years (i.e., except 2019) relative to 2013, and lower in 2015, 2018, 2023, and 2024 relative to 2007 (Appendix Table F.36, Appendix Figure F.13). The relative proportion of *Ostracoda* was the only other BIC endpoint for profundal habitats that exhibited a temporal pattern that was both ecologically meaningful and with a consistent directionality (i.e., higher or lower) relative to baseline. Specifically, relative proportions of *Ostracoda* were higher throughout 2019 to 2024, relative to 2013 (Appendix Table F.36, Appendix Figure F.13). This particular pattern for

Ostracoda may reflect an increase in fine sediment in Sheardown Lake SE, creating more favorable conditions for *Ostracoda*, which are well-documented to thrive in such environments (Moquin et al 2014).

Throughout the mine operation period, Sheardown Lake SE has consistently had higher benthic invertebrate density and richness in both the littoral and profundal habitats compared to Reference Lake 3 (Appendix Figures F.11 and F.12). This pattern is similar to that observed in Sheardown Lake NW, which is expected given their connectivity, and these results suggest that Sheardown Lake SE is more biologically productive than Reference Lake 3. This finding is further supported by higher primary productivity (i.e., phytoplankton levels as measured by chlorophyll-a; see Section 4.5.3). Additionally, lower sediment TOC and higher fine sediment content (i.e., silt and clay) may contribute to observed patterns in the BIC (see Section 4.5.2). Overall, the relatively few differences in 2024 between the BIC of Sheardown Lake SE and Reference Lake 3, along with minimal variation among mine operation years and between mine operation and baseline data, as well as the absence of adverse changes in water and sediment quality, suggest no mine-related influence on the BIC at Sheardown Lake SE.

4.5.5 Fish Population

4.5.5.1 Fish Community

In 2024, the fish community of Sheardown Lake SE consisted of arctic charr (captured in electrofishing and gill netting surveys) and ninespine stickleback (*Pungitius pungitius*; captured by electrofishing only; Table 4.7). The same species were captured in previous CREMP monitoring years at Sheardown Lake SE (except for 2015 when no ninespine stickleback were captured; Minnow 2016a and 2024a). The CPUE for both species in electrofishing surveys and for arctic charr in gill netting surveys was higher at Sheardown Lake SE than at Reference Lake 3 in 2024, indicating greater densities of both species in Sheardown Lake SE (Figure 4.10, Table 4.7, Appendix Tables G.1 and G.3). Similar to at Camp Lake and littoral areas of Sheardown Lake NW, these results were supported by higher chlorophyll-a concentrations in the summer and fall (indicating greater phytoplankton density; Section 4.5.3) and higher benthic invertebrate density (Section 4.5.4) compared to Reference Lake 3, suggesting more abundant food sources for arctic charr.

In 2024, electrofishing CPUE at Sheardown Lake SE was within the range observed over the previous nine years of mine operation (2015 to 2023) and was higher than in baseline studies (2007 and 2008; Figure 4.10). Likewise, gill netting CPUE for arctic charr in 2024 was within the range of earlier mine operation years and greater than during baseline, although it was lower than in the previous three years (2021 to 2023; Figure 4.10). No consistent temporal patterns in chlorophyll-a concentrations (i.e., phytoplankton density) or ecologically significant differences in BIC endpoints have been observed in Sheardown Lake SE that are consistent with the temporal pattern in gill net CPUE (Sections 4.4.3 and 4.4.4). These results suggest that factors, such as changes in spatial ecology, other than lake productivity differences may have resulted in the temporal variability in gill net CPUE in Sheardown SE. As described in Section 3.3.5.1, sampling related influences (e.g., seasonal timing and access to sampling locations) or naturally influenced environmental factors (e.g., water temperature) that affect fish movement behaviour, spatial ecology, and metabolic demands, have the potential to influence fish catch rates, particularly in 'passive' gill net surveys.

In 2024, CPUE in electrofishing and in gill netting surveys fell within the range of previously observed CPUE during the mine operations period, were greater than during the baseline period, and were greater than in Reference Lake 3. Therefore, mine-related changes in fish densities at Sheardown Lake SE are not indicated.

4.5.5.2 Fish Health Assessment

Nearshore Arctic Charr

In August 2024, a total of 101 and 102 arctic charr were sampled for assessment of fish health from nearshore habitats of Sheardown Lake SE and Reference Lake 3, respectively (Appendix Tables G.4 and G.19)[45,46]. To distinguish arctic charr YOY from non-YOY, fork length cut-offs of 5.0 cm for Sheardown Lake SE and 4.0 cm for Reference Lake 3 were applied, based on an analysis of LFD and supporting length and weight measurements and age determinations (Figure 4.18, Appendix Tables G.4 and G.19). Since at least ten YOY arctic charr were captured from each lake (n = 12 and 58 for Reference Lake 3 and Sheardown Lake SE, respectively; Appendix Table G.20), statistical comparisons of health endpoints were conducted separately for YOY and non-YOY groups.

Length frequency distributions for all fish and for non-YOY age classes were significantly different between Sheardown Lake SE and Reference Lake 3 in 2024 (Table 4.13, Figure 4.18, Appendix Figure G.15, Appendix Table G.20). In the LFD for all fish, two distinct dominant size

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[45] Sample sizes at Sheardown Lake SE met minimum requirements to detect a ±10% difference in condition based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.21.

[46] The total number of fish captured in Sheardown Lake SE and Reference Lake 3 by electrofishing (Table 4.7, Appendix Table G.1) was greater than the number of fish sampled for the fish health assessment. The study design targets 100 fish from each lake for sampling (measurement of length and weight; Baffinland 2015). Once field crews were certain that the minimum target sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

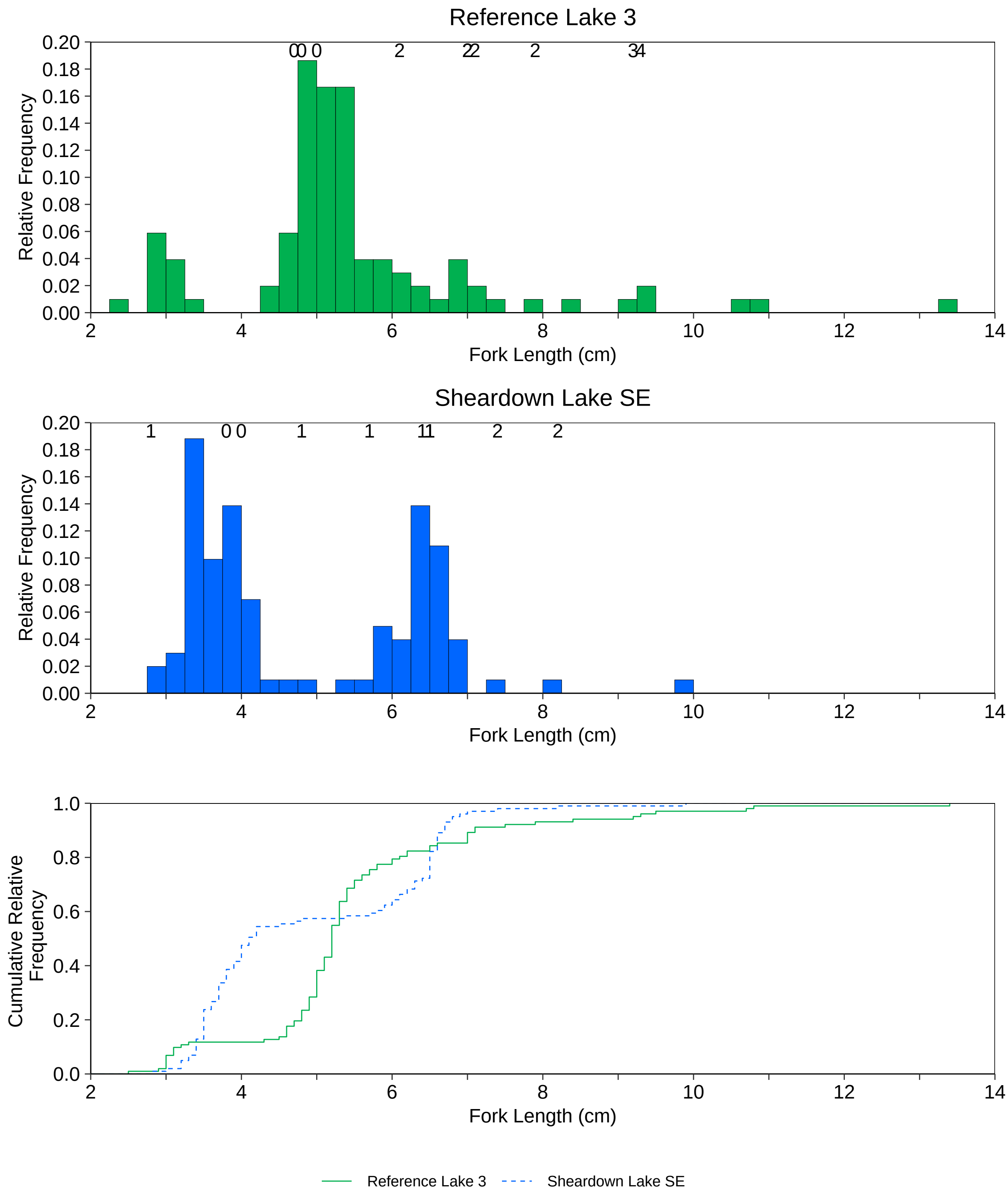

Figure 4.18: Relative Length−Frequency and Cumulative Length−Frequency Distributions for All Arctic Charr Captured by Backpack Electrofishing at Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Fish ages are shown above the bars, where available. Sheardown Lake SE n = 101; Reference Lake 3 n = 102.

Table 4.13: Summary of Statistical Results for Arctic Charr Population Comparisons between Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03), and between Sheardown Lake Southeast Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP, 2015 to 2024

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]																			
			Sheardown Lake SE vs Reference Lake 3										Sheardown Lake SE Mine Operational Year vs Baseline Period [b]									
			2015	2016	2017	2018	2019	2020	2021	2022	2023	2024	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Nearshore Electrofishing	Survival [c]	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**
		Age	No	No	No	-	-	-	-	-	-	-	**Yes (+273%)**	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	No	No	**Yes (+12%)**	**Yes (+21%)**	**Yes (-28%)**	**Yes (+7%)**	**Yes (+26%)**	**Yes (-20%)**	No	**Yes (+23%)**	**Yes (+7%)**	**Yes (-15%)**	**Yes (+19%)**	**Yes (-47%)**	No	**Yes (+30%)**	**Yes (+11%)**	No	**Yes (+4.8%)**	No
		Size (mean weight)	No	No	**Yes (+55%)**	**Yes (+59%)**	**Yes (-59%)**	**Yes (+53%)**	**Yes (+124%)**	**Yes (-62%)**	**Yes (-32%)**	**Yes (+127%)**	No	**Yes (-43%)**	**Yes (+54%)**	No	No	**Yes (+117%)**	**Yes (+32%)**	No	No	No
		Growth (weight-at-age)	**Yes (+85%)**	**Yes (+120%)**	-	-	-	-	-	-	-	-	No	-	-	-	-	-	-	-	-	-
		Growth (fork length-at-age)	**Yes (+21%)**	**Yes (+34%)**	-	-	-	-	-	-	-	-	No	-	-	-	-	-	-	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+4%)**	No	**Yes (+9%)**	**Yes (-13%)**	**Yes (+4%)**	**Yes (+14%)**	**Yes (+10%)**	**Yes (-8.9%)**	**Yes (-16%)**	**Yes (+21%)**	**Yes (-14%)**	**Yes (-16%)**	No	**Yes (-15%)**	**Yes (-13%)**	No	No	**Yes (-17%)**	**Yes (+14%/-27%)**	**Yes (+26%/-20%)**
	Energy Use (YOY)	Size (mean fork length)	-	-	-	-	-	-	-	-	**Yes (+20%)**	**Yes (+23%)**	-	-	-	-	-	-	-	-	-	-
		Size (mean weight)	-	-	-	-	-	-	-	-	No	**Yes (+89%)**	-	-	-	-	-	-	-	-	-	-
	Energy Storage (YOY)	Condition (body weight-at-fork length)	-	-	-	-	-	-	-	-	**Yes (-34%/-21%)**	**Yes (+42%)**	-	-	-	-	-	-	-	-	-	-
Littoral/Profundal Gill Netting [d]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	No	No	**Yes**	No	**Yes**
		Age	-	-	-	-	-	-	-	-	-	-	**Yes (-13%)**	No	No	-	-	-	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	No	**Yes (+23%)**	**Yes (+21%)**	**Yes (+27%)**	**Yes (+20%)**	No	**Yes (+29%)**	**Yes (-9%)**	**Yes (-7%)**	**Yes (-5%)**	**Yes (-4%)**	**Yes (-2%)**	No	No	**Yes (-2.3%)**	No	No
		Size (mean weight)	-	-	-	No	**Yes (+102%)**	**Yes (+107%)**	**Yes (+158%)**	**Yes (+122%)**	No	**Yes (+139%)**	**Yes (-26%)**	**Yes (-20%)**	**Yes (-16%)**	**Yes (-16%)**	**Yes (-11%)**	**Yes (-7.0%)**	No	**Yes (-9.0%)**	**Yes (-9.0%)**	**Yes (-9.0%)**
		Growth (fork length-at-age)	-	-	-	-	-	-	-	-	-	-	No	No	No	-	-	-	-	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	-	-	-	-	**Yes (+18%)**	**Yes (+24%)**	No	-	-	-	-	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+7%)**	No	**Yes (+14%)**	**Yes (+24%)**	**Yes (+33%)**	**Yes (-12%)**	**Yes (+19%)**	No	No	**Yes (-6%)**	**Yes (-7%)**	**Yes (-6%)**	**Yes (-5.0%)**	**Yes (-3.6%)**	No	**Yes (-11%)**	**Yes (-5.7%)**

BOLD indicates a statistically significant difference.

Notes: "-" indicates data not available for comparison. YOY = Young-of-the-Year.

[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] The length-frequency distribution for Reference Lake 3 includes all fish, whereas for baseline conditions, it only includes non-YOY fish.

[d] 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.

classes of fish were evident in Sheardown Lake SE (i.e., fish in the 3 to 4 cm length range, classified as YOY and fish in the 6 to 7 cm range, likely aged 1+; Figure 4.18,). In Reference Lake 3, most fish were less than 6 cm in length and either classified as YOY (<4 cm) or were likely age 1+ (4 to 6 cm; Appendix Figure G.15). The LFD for nearshore arctic charr has consistently been different between Sheardown Lake SE and Reference Lake 3 over the period of mine operations (Table 4.13) with two distinct dominant size classes of fish frequently evident in Sheardown Lake SE compared to a more normal distribution of lengths in Reference Lake 3 (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024).

Arctic charr from Sheardown Lake SE, both YOY and non-YOY, were significantly longer (YOY = 23%, non-YOY= 23%), heavier (YOY = 89%, non-YOY = 127%), and had better condition (YOY = 42%, non-YOY = 21%) compared to those from Reference Lake 3 in 2024 (Table 4.13, Appendix Table G.20, Appendix Figures G.17 and G.18). The differences in condition for both YOY and non-YOY exceeded the CES_C of ± 10%, suggesting an ecologically meaningful difference between populations (Table 4.13, Appendix Table G.20, Appendix Figures G.17 and G.18). In all previous monitoring years except 2023, insufficient YOY arctic charr were captured at Sheardown Lake SE[47] and/or Reference Lake 3 to allow for comparisons of fish health endpoints in the YOY age class. However, in 2023 and 2024, differences in body weight and condition in Sheardown Lake SE were inconsistent relative to Reference Lake 3, though the relative direction and magnitude of difference in length was similar (Table 4.13). For non-YOY arctic charr, no consistent directional differences in size (length and weight) or condition were observed between Sheardown Lake SE and Reference Lake 3 from 2015 to 2024 (Table 4.13). Although fish from Sheardown Lake SE were frequently longer and heavier than those from Reference Lake 3, relative condition varied and the MOD for condition in 2024 was the highest recorded since 2015 (Table 4.13).

A significant difference in LFD of nearshore non-YOY arctic charr from Sheardown Lake SE was observed between 2024 and the combined Sheardown Lake SE baseline dataset, consistent with annual comparisons to baseline in previous mine operational years (Table 4.13, Appendix Figure G.16). In 2024, body size (length and weight) of non-YOY arctic charr from Sheardown Lake SE was not significantly different in compared to the baseline period (Table 4.13, Appendix Figure G.19, Appendix Table G.7). However, condition was significantly different from baseline in 2024, with a MOD between +26% at the lower end of the overlapping length

[47] In 2022 the sampling location in Sheardown Lake SE shifted from the channel connecting the Sheardown lakes to the southeast end of the lake. This change represents a subtle shift in the habitat type sampled. The original site featured deeper water with larger cobble and boulder substrates, while the new site sampled from 2022 to 2024 was characterized primarily by cobble and gravel substrates in shallower areas. These habitat differences may explain the greater proportion of YOY arctic charr captured in 2023 and 2024.

distribution between the two lakes and -20% at the upper end, in both cases exceeding the CES_C of ± 10% and indicating an ecologically meaningful difference (Table 4.13, Appendix Figure G.19, Appendix Table G.7)[48]. Similarity in body size was reflected in a dominant size class of fish between 6 and 7 cm in both 2024 and the baseline period (Appendix Figure G.16). Fork length and body weight of non-YOY nearshore arctic charr in Sheardown Lake SE have been inconsistent relative to baseline between 2015 and 2024 (Table 4.13). While body size has varied over the mine operational period, condition of fish from Sheardown Lake SE has frequently been lower than during the baseline period, though the MODs in 2023 and 2024 did not indicate a consistent directional difference from baseline over the length range of fish captured (i.e., the condition of smaller fish in 2024 was greater than during the baseline period while the condition of larger fish was lower; Table 4.13, Appendix Figure G.19, Appendix Table G.7). A similar difference in fish condition in 2023 and 2024 relative to the baseline period could reflect higher fish density (Section 4.5.5.1). Higher density can lead to competition effects that potentially influence condition and growth rates.

There have been no consistent differences in nearshore non-YOY arctic charr condition in Sheardown Lake SE relative to Reference Lake 3 since 2015 and limited results (i.e., 2023 and 2024 only) identify a similar pattern in the YOY age class. The absolute MODs for the condition of non-YOY arctic charr in Sheardown Lake SE in 2024 compared to the reference lake and baseline period exceeded the CES_C of ±10%. While the direction of differences relative to Reference Lake 3 have not been consistent in recent study years suggesting no adverse mine-related influence, in both 2023 and 2024, MODs for condition relative to the baseline period indicated a similar pattern, outside of the CES_C, which was potentially ecologically meaningful. Ongoing monitoring in 2025 will be used to confirm these results and to further evaluate whether the difference is mine-related.

Littoral/Profundal Arctic Charr

In August 2024, a total of 100 and 84 arctic charr were sampled for fish health assessment from littoral and profundal habitats of Sheardown Lake SE and Reference Lake 3, respectively

[48] Given a significant interaction term in the ANCOVA model and unequal regression slopes based on coefficients of determination, the MOD was estimated at both the minimum and maximum overlap of covariates between the lakes (Section 2.4.3.4).

(Appendix Tables G.9 and G.23)[49,50]. The LFD for littoral/profundal arctic charr differed significantly between the two lakes, with the lengths of fish captured in Reference Lake 3 being mostly less than 35 cm while the majority of fish captured in Sheardown Lake SE were between 29 and 45 cm long (Table 4.13, Figure 4.19, Appendix Table and G.24). The LFD for littoral/profundal arctic charr has consistently been different between Sheardown Lake SE and the reference lake since 2018 (Table 4.13, Figure 4.19) generally reflecting higher relative frequencies of larger fish in Sheardown Lake SE (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024).

Arctic charr from Sheardown Lake SE were significantly longer (29%), heavier (139%), and exhibited significantly better body condition (19%) than those from Reference Lake 3 (Table 4.13, Appendix Table G.24, Appendix Figure G.21). The absolute MOD for condition exceeded the CES_C of ± 10%, indicating that the observed difference was ecologically meaningful (Table 4.13, Appendix Table G.24). Fork length and body weight for littoral/profundal arctic charr from Sheardown Lake SE have been consistently greater than those from Reference Lake 3 between 2019 and 2024, except in 2023, when no significant difference was observed (Table 4.13). Condition of littoral/profundal arctic charr from Sheardown Lake SE has also generally been higher than condition of fish from Reference Lake 3 since 2018, except in 2023 when it was lower (Table 4.13). While the MOD for condition between the two lakes was outside of the CES_C in 2024, it fell within the range observed during the mine operational period from 2018 to 2023 (Table 4.13). The greater size and condition of littoral/profundal arctic charr from Sheardown Lake SE between 2019 and 2024 compared to fish from Reference Lake 3, may have been influenced by the lake's higher productivity relative to the reference lake, as evidenced by higher chlorophyll-a concentrations (Section 4.5.3) and higher benthic invertebrate density (Section 4.5.4). However, multiple factors including littoral and profundal fish density and capture efficiency, as well as variation in nearshore fish density, size, and condition may have also been factors.

The LFD of littoral/profundal arctic charr from Sheardown Lake SE in 2024 was significantly different from baseline, but this has not consistently been the case over years of mine operations

[49] Sample sizes at Sheardown Lake SE in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.21.

[50] The total number of fish captured in Sheardown Lake SE by gill netting (Table 4.7, Appendix Tables G.2 and G.22) was greater than the number of fish sampled for the fish health assessment. The study design targets 100 fish from each lake for sampling (measurement of length and weight; Baffinland 2015). Once field crews were certain that the minimum sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

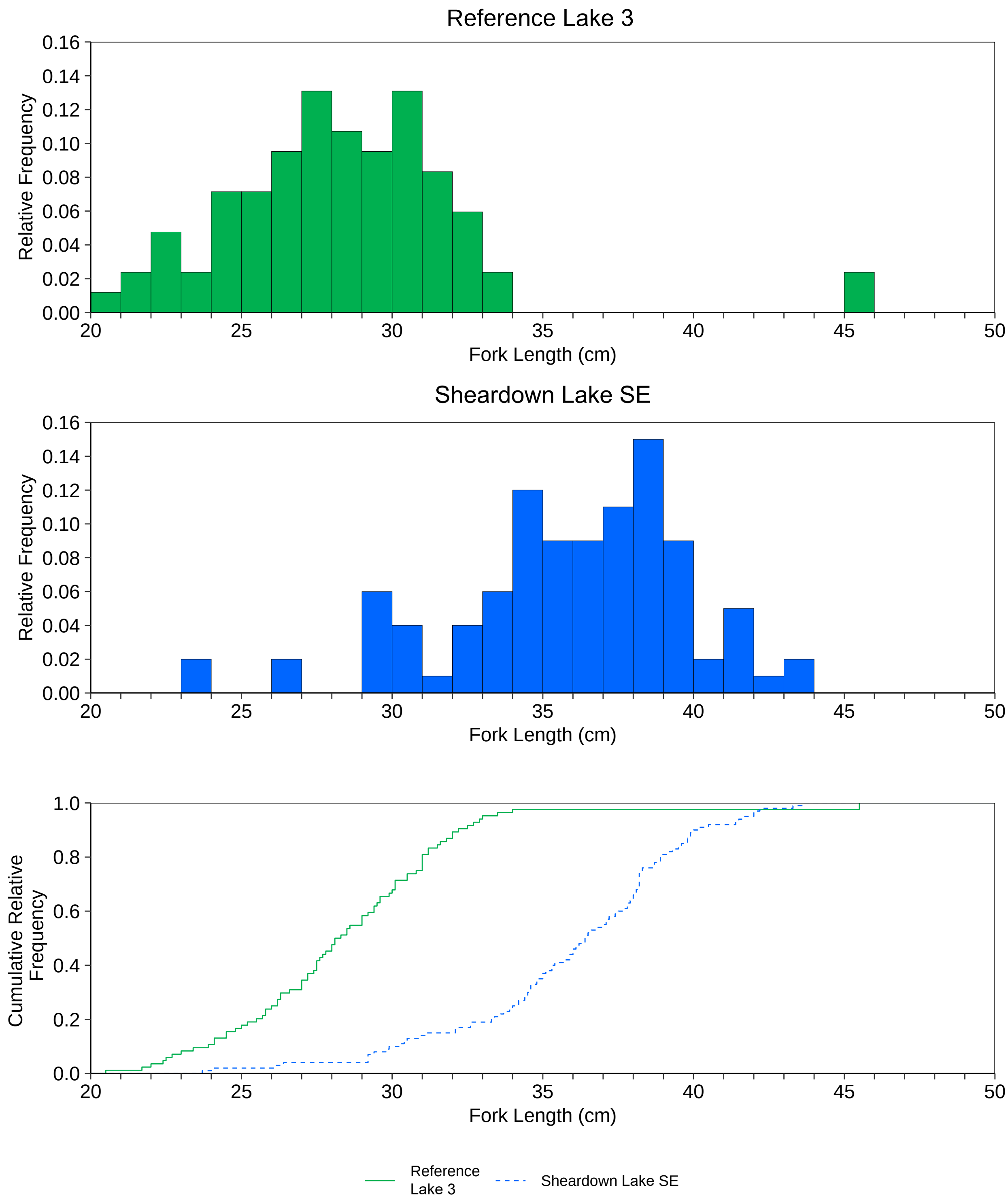

Figure 4.19: Relative Length−Frequency and Cumulative Length−Frequency Distributions for Arctic Charr Captured by Gill Netting at Sheardown Lake Southeast (SE; DL0-02) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Sheardown Lake SE n = 100; Reference Lake 3 n = 84.

(2015 to 2023; Table 4.13, Appendix Figure G.20, Appendix Table G.24). Although the distribution of fish lengths was similar to baseline in 2024, there was a slightly greater proportion of fish longer than 36 cm during the baseline period (Appendix Figure G.20). When comparing adult arctic charr in 2024 to baseline data, individuals from Sheardown Lake SE were significantly lighter (-9%) and had significantly lower condition (-5.1%), although the MOD for condition was within the CES_C of ± 10%, indicting that this difference was not ecologically meaningful (Table 4.13, Appendix Table G.24, Appendix Figure G.22). There was no significant difference in length in 2024 compared to baseline (Table 4.13, Appendix Table G.24, Appendix Figure G.22). Fork length, body weight, and condition have generally remained consistent relative to baseline over the mine operation period (2015 to 2024), with arctic charr from Sheardown Lake SE typically being similar to or slightly smaller, lighter, and in poorer condition compared to those from the baseline period (Table 4.13, Appendix Table G.24). MODs for fish weight and condition between 2024 and the baseline period were also within the range observed over the mine operational period (Table 4.13).

Littoral/profundal arctic charr at Sheardown Lake SE have consistently been significantly larger and in better condition over the mine operational period compared to Reference Lake 3. In contrast, when compared to baseline, adult arctic charr from Sheardown Lake SE were generally smaller and had lower condition though the magnitudes of these differences were not ecologically meaningful, including in 2024 when the MOD for condition was within the CES_C. Overall, these results suggest no mine-related effects on the health of non-YOY arctic charr at Sheardown Lake SE over the mine operational period.

4.5.6 Effects Assessment and Recommendations

In 2024, water chemistry at Sheardown Lake SE met AEMP benchmarks and WQGs across all seasonal sampling events (spring, summer, fall). When comparing water quality parameter concentrations to reference and baseline across all seasons or within a single season, the following parameters were elevated, indicating a potential mine-related effect:

- All seasons (spring, summer, fall): nitrate and sulphate.

Since 2022, concentrations of nitrate and sulphate, and total and/or dissolved molybdenum and uranium have had elevated concentrations relative to reference and/or baseline (Minnow 2023 and 2024a). In temporal trend analyses completed in the 2023 CREMP total and/or dissolved concentrations of each showed statistically significant increasing trends since the baseline period and over the mine operations period (Minnow 2024a) and visual assessment of temporal data indicated that these increasing trends persisted in 2024. Overall, these results indicated a mine-related influence on nitrate, sulphate, and total/dissolved molybdenum and uranium at Sheardown Lake SE in 2024. A special investigation into analysis of total compared to dissolved aqueous concentrations of molybdenum and uranium in 2024 found that the dissolved fraction constituted almost the entire total fraction for both parameters but that there has been no change in the total:dissolved ratio over time, indicating that mine activities have likely not influenced the bioavailability of either parameter. In general, mine-related influences on water chemistry at Sheardown Lake SE in 2024 were attributed to mine infrastructure located in the upstream watershed, including influences on aqueous nitrogen compounds from activities occurring at the Dyno facility and related influences at SDLT9, a tributary to Sheardown Lake SE as well as mine-related influences identified in upstream Sheardown Lake NW.

In 2024, the following sediment quality AEMP benchmarks were exceeded at Sheardown Lake SE:

- Arsenic concentrations exceeded the AEMP benchmark of 5.9 mg/kg in one littoral (Station DL0-02-4; 6.29 mg/kg) and two profundal (Stations DL0-02-12 and DL0-02-13; 6.42 and 6.68 mg/kg, respectively) sediment samples in August;
- Mean chromium concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 79 mg/kg at all sediment monitoring stations except for DL0-02-1 in August (mean = 85.7 mg/kg);
- Mean iron concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 34,400 mg/kg at all sediment monitoring stations in August (mean = 54, 920 mg/kg); and
- Mean manganese concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 657 mg/kg at all sediment monitoring stations in August (mean = 2,574 mg/kg);
- Nickel concentrations exceeded the AEMP benchmark of 66 mg/kg in two littoral (Stations DL0-02-11 and DL0-02-4; 72.6 and 70 mg/kg, respectively) and two profundal (Stations DL0-02-12 and DL0-02-13; 76.3 and 66.9 mg/kg, respectively) sediment samples in August; and
- Phosphorus concentrations exceeded the AEMP benchmark of 1,278 mg/kg in two profundal sediment samples at Station DL0-02-12 (1,300 mg/kg) and DL0-02-13 (1,310 mg/kg) in August.

Despite these exceedances, none of these parameters had concentrations that were elevated compared to both reference and baseline, suggesting no or minimal mine-related impact on littoral and profundal sediment quality in 2024. It is likely that the elevated concentrations are due to natural processes causing high variability.

No adverse mine-related effects on phytoplankton, BIC, or fish (arctic charr) health were observed at Sheardown Lake SE in 2024, based on comparisons to Reference Lake 3 and to Sheardown Lake SE baseline data. Under the AEMP Management Response Framework, a Low Action Response is required based on determination of mine-related influences on nitrate, sulphate, molybdenum and uranium due to total and/or dissolved concentrations that were elevated compared to reference and baseline in at least one season in 2024 and/or evidence of increasing trends/patterns over the mine operations period (Figure 2.6). The following actions are recommended:

- In 2025, temporal trend analysis of aqueous total and dissolved (where applicable) concentrations of nitrate, sulphate, molybdenum, and uranium will be conducted for Sheardown Lake NW to further investigate temporal trends/patterns.
- Spatial comparisons of the concentrations of nitrate within the lake will be completed in 2025 for evaluation of the overall influence of inputs from activities occurring at the Dyno facility (via SDLT9) into Sheardown Lake SE. Water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT9 and Sheardown Lake SE as a basis for informing the potential need for further investigations and mitigation.
- Potential sources of nitrate, sulphate, molybdenum, and uranium to Sheardown Lake SE will be investigated to better define mine-related influence and the potential for continued contributions.
- Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.
- The focus in 2025 for the KM 105 Pond remediation efforts will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. Given potential influences of water management at the KM 105 Pond on water quality at Sheardown Lake SE (through inputs from Sheardown Lake NW), water quality information collected during the 2025 CREMP will be used to monitor water quality of Sheardown Lake NW and Sheardown Lake SE as a basis for informing the potential need for further investigations and mitigation.

According to the Mary River Project AEMP Management Response Framework, the absence of any mine-related influences on sediment chemistry concentrations or biota, means no further management response is required for these monitoring components at Sheardown Lake SE in 2024 (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at Sheardown Lake SE in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management), SWSQ-4 (Explosives), SWSQ-7 (Camps and Fuel Management), and SWSQ-9 (Airstrips and Airstrip Use) indicated all parameter concentrations were within the Level II significance rating for magnitude (or Level I for SWSQ-7) expected for the watercourse during mine operations. Therefore, water quality at Sheardown Lake SE conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

Comparison of sediment quality at Sheardown Lake SE in 2024 to FEIS predictions related to Airborne Emission sources (i.e., fugitive dust; FEIS Issue SWSQ-17-3) indicated all mean parameter concentrations were within the applicable significance rating magnitudes expected for lake sediments during mine operations. Therefore, sediment quality at Sheardown Lake SE conformed with predictions made in Baffinland FEIS (Baffinland 2012).

Water and sediment quality at Sheardown Lake SE in 2024 where all parameter concentrations were within applicable FEIS significance rating magnitude predictions also meant that FEIS predictions for (absence of) effects on arctic charr health and condition were also met. Therefore, arctic charr health and condition at Sheardown Lake SE in 2024 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

5 MARY RIVER AND MARY LAKE SYSTEM

5.1 Mary River

5.1.1 Water Quality

5.1.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at the Mary River concurrent with water quality sampling in spring, summer, and fall (Figure 2.1), as well as concurrent with BIC sampling in August (Figure 2.3). Dissolved oxygen in water at Mary River stations was consistently near or above saturation (> 95%) during all spring, summer, and fall monitoring events, and showed comparable saturation among the G0-09 series Mary River reference stations and Mary River areas upstream of (G0 series), adjacent to (E0 series), and downstream of (C0 series) the mine as well as the Tom River (I0-01) for each respective seasonal sampling event in 2024 (Appendix Figure C.22, Appendix Tables C.1 to C.3). Dissolved oxygen concentrations were significantly lower at Mary River mine-exposed BIC sampling stations located upstream of the mine (G0-03) than at the upstream reference (G0-09) stations in August 2024 (Figure 5.1, Appendix Tables C.55 to C.57). There were no significant differences in DO saturation among any of the Mary River reference or mine-exposed BIC sampling stations (Figure 5.1, Appendix Tables C.55 to C.57). In addition, mean dissolved oxygen concentrations were well above the WQG of 9.5 mg/L (lowest acceptable concentration for early life stages of cold-water biota) at all Mary River stations during all sampling in 2024 (Figure 5.1, Appendix Table C.56), indicating that differences in dissolved oxygen concentrations among the Mary River study areas were unlikely to be ecologically meaningful.

In situ pH at all Mary River mine-exposed stations was generally comparable to pH at the G0-09 series reference stations during the spring, summer, and fall sampling events in 2024 (Appendix Figure C.22). Only the downstream station (C0-05) had significantly different (i.e., lower) pH than the reference station (G0-09) during the August 2024 BIC sampling though there were also significant differences among the mine-adjacent (E0-01 and E0-20 and downstream (C0-05) stations (Figure 5.1, Appendix Tables C.55 to C.57). However, pH at all Mary River areas in all seasons was consistently within WQG limits (Figure 5.1, Appendix Figure C.22, Appendix Tables C.1 to C.3 and C.55 to C.57).

Specific conductance was consistently lowest in spring and highest in fall at all Mary River stations (Appendix Figure C.22, Appendix Tables C.1 to C.3), reflecting natural seasonal differences related to proportion of flow from surface runoff (e.g., spring snowmelt) and baseflow/groundwater sources. Within the Mary River, specific conductance was

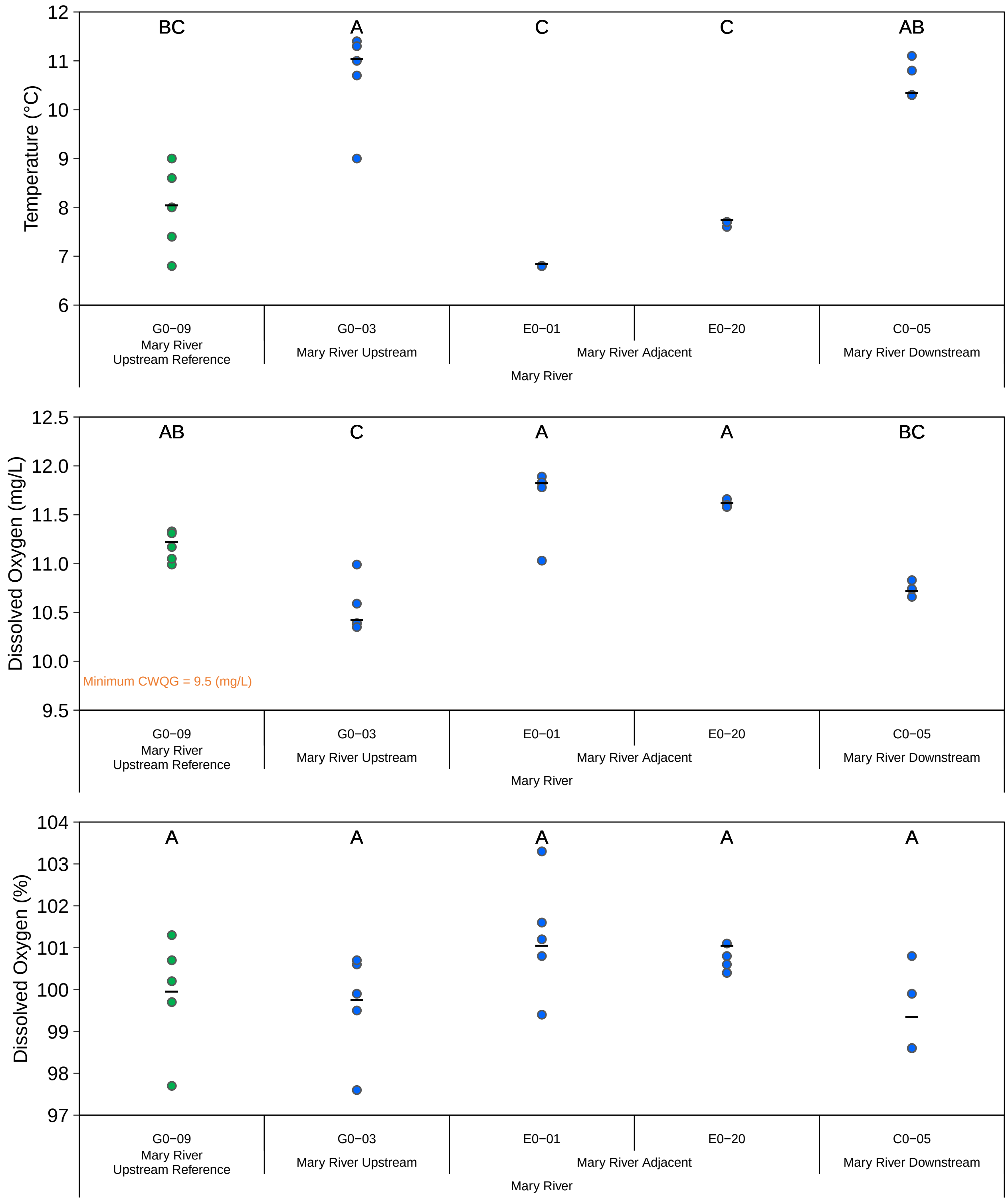

Figure 5.1: Comparison of In Situ Water Quality Measured at Mary River Reference and Mine−Exposed Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

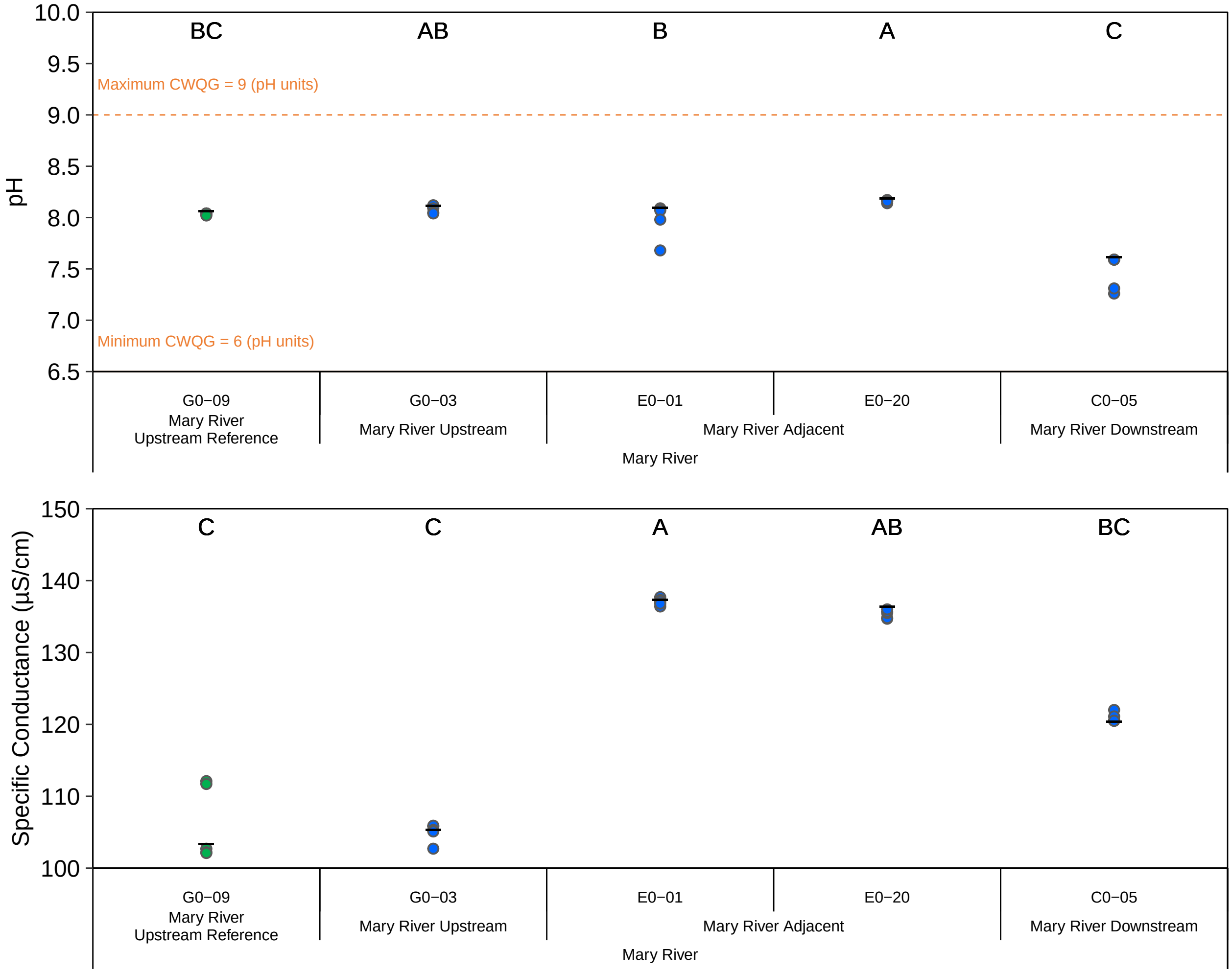

Figure 5.1: Comparison of In Situ Water Quality Measured at Mary River Reference and Mine−Exposed Benthic Invertebrate Community Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

significantly higher at the BIC sampling stations adjacent to the mine (E0-01 and E0-20) compared to both the upstream (G0-03) and reference (G0-09) stations in August 2024 (Figure 5.1, Appendix Tables C.55 to C.57), indicating a mine-related influence on the water quality of the Mary River.

5.1.1.2 Water Chemistry

Water chemistry parameters at Mary River mine-exposed areas (G0 Upstream, E0 Adjacent, and C0 Downstream series stations) met all AEMP benchmarks and WQGs during spring, summer, and fall sampling events in 2024 (Table 5.1), except for individual sample concentrations of total aluminum, which slightly exceeded the WQG of 0.100 mg/L at the G0-01 upstream station in the spring (0.102 mg/L) and summer (0.104 mg/L) and total chromium which exceeded the WQG of 0.001 mg/L at the E0-20 mine-adjacent station in the summer (0.00221 mg/L; Appendix Table C.58). At the Mary River reference area (G0-09 series) the mean total aluminum concentration in the summer (0.103 mg/L) and individual station concentrations in the spring (0.141 mg/L at G0-09-B) and summer (0.118 mg/L) were also higher than the WQG (Table 5.1, Appendix Table C.58). Total and dissolved aluminum concentrations at mine-exposed Mary River stations were not elevated in any season in 2024 compared to either reference (G0-09 series) or baseline concentrations, except for the dissolved aluminum at one downstream area station (C0-10), which was moderately elevated compared to reference in the fall (Appendix Tables C.59 and C.61). Although total aluminum concentrations at Mary River mine-exposed stations were relatively high compared to other mine operation years in 2019 and 2020, no consistent increasing or decreasing temporal patterns since baseline were evident based on visual assessment, concentrations in 2024 were among the lowest observed in the mine operations period, and concentrations at the reference area were elevated relative to WQGs (indicative of regional change) suggest no mine-related influence on this parameter (Appendix Figure C.23). Relatively high concentrations of total aluminum (and other parameters including iron and copper) in Mary River mine-exposed and reference areas in 2019 and 2020 (Appendix Figure C.23) were attributed to suspended inorganic material introduced through natural surface runoff and fluvial transport, based on significant positive correlations with turbidity for total but not dissolved concentrations of these parameters (Minnow 2022).

The total concentration of chromium at mine-adjacent Mary River station E0-20 was slightly elevated relative to baseline and reference concentrations in the summer in 2024, while the total concentration in spring and fall and the dissolved concentration in all seasons was similar to both baseline and reference (Appendix Tables C.59 and C.51). Chromium concentrations at the E0 series stations and throughout the Mary River have generally been below the LRL, WQG, and the AEMP benchmark and there have been no temporal increasing patterns in chromium throughout

Table 5.1: Mean Water Chemistry at Mary River (G0, E0, F0, and C0 Series) Monitoring Stations During Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

	Parameters	Units	Water Quality Guideline (WQG)[a, b]	AEMP Benchmark[c]	G0-09 Reference (n = 3)			G0 Upstream (n = 2)			Mary River Tributary F			E0 Adjacent (n = 4)			C0 Downstream (n = 3)		
					Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall	Spring	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	40.7	105	164	30.0	110	146	103	208	309	33.2	134	160	30.9	116	161
	pH (lab)	pH	6.5 - 9.0	-	7.31	7.96	7.91	7.64	8.00	7.95	7.80	8.04	8.04	7.44	7.90	7.92	7.52	7.84	7.82
	Hardness (as $CaCO_3$)	mg/L	-	-	19.5	48.2	74.2	13.7	53.7	68.8	51.0	101	156	15.9	63.8	75.8	14.3	53.9	75.9
	Total Suspended Solids (TSS)	mg/L	-	-	2.33	<1	<1	1.00	<1	<1	<1	<1	<1	2.35	<1	<1	1.17	<1	<1
	Total Dissolved Solids (TDS)	mg/L	-	-	30.3	57.0	79.3	30.0	59.5	66.5	54.0	102	175	21.5	70.0	81.2	22.0	66.0	87.0
	Turbidity	NTU	-	-	2.48	4.42	1.87	4.10	3.28	2.30	0.780	0.900	0.210	3.06	2.66	2.14	2.83	2.92	1.75
	Alkalinity (as $CaCO_3$)	mg/L	-	-	19.9	49.1	81.6	13.2	51.5	68.2	42.8	81.5	107	14.6	58.1	74.1	12.7	53.3	72.7
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	<0.005	<0.005	0.00947	<0.005	0.00540	0.00515	<0.005	<0.005	<0.005	<0.005	<0.005	0.00588	0.00887	<0.005	0.00517
	Nitrate	mg/L	3	3	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	0.201	0.305	1.39	0.0292	0.0860	0.0758	0.0243	0.0233	0.124
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	<0.05	0.0733	0.113	0.0515	0.0750	0.106	0.0560	0.0650	0.313	<0.05	0.0742	0.140	0.0933	0.0680	0.147
	Dissolved Organic Carbon	mg/L	-	-	1.52	2.58	1.71	1.44	1.72	1.56	0.960	1.21	1.42	1.47	2.85	2.14	1.40	1.59	3.73
	Total Organic Carbon	mg/L	-	-	1.63	1.35	1.73	1.49	1.62	1.70	1.69	1.19	1.86	2.21	1.57	1.78	1.70	1.72	2.19
	Total Phosphorus	mg/L	0.020[α]	-	0.00470	0.00333	0.00203	0.00440	0.00310	<0.002	<0.002	<0.002	<0.002	0.00478	0.00217	0.00200	0.00453	0.00210	<0.002
	Phenols	mg/L	0.004[α]	-	0.00103	<0.001	<0.001	0.00105	<0.001	<0.001	0.00110	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	<0.5	1.75	3.53	<0.5	1.92	3.40	0.540	1.38	3.85	0.540	1.70	3.39	0.690	1.73	3.26
	Sulphate (SO_4)	mg/L	218[β]	218	<0.3	1.31	2.56	<0.3	1.25	2.24	7.43	20.1	45.6	0.925	6.39	4.76	0.863	2.45	6.10
Total Metals	Aluminum (Al)	mg/L	0.100	0.966	0.0766	0.103	0.0874	0.0974	0.0896	0.0950	0.0267	0.0115	0.00800	0.0772	0.0654	0.0927	0.0839	0.0710	0.0737
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Barium (Ba)	mg/L	1[β]	-	0.00288	0.00662	0.00947	0.00262	0.00696	0.00966	0.00404	0.00746	0.0136	0.00240	0.00697	0.0103	0.00237	0.00669	0.0102
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Cadmium (Cd)	mg/L	0.00012	0.00006	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	3.91	9.65	15.0	2.80	10.6	14.8	8.81	16.9	27.7	3.06	12.1	15.5	2.73	10.8	15.6
	Chromium (Cr)	mg/L	0.001	0.003	<0.0005	0.000507	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	0.000928	<0.0005	<0.0005	<0.0005	<0.0005
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.000110	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.000503	0.000873	0.000967	0.000530	0.000865	0.00103	<0.0005	0.000510	0.000800	<0.0005	0.000748	0.00100	<0.0005	0.000767	0.00110
	Iron (Fe)	mg/L	0.30	0.874	0.0693	0.0997	0.0610	0.0870	0.0775	0.0630	0.0250	0.0110	<0.01	0.0762	0.0612	0.0650	0.0777	0.0670	0.0607
	Lead (Pb)	mg/L	0.001	0.001	0.0000700	0.0000830	0.0000597	0.0000875	0.0000680	0.0000620	<0.00005	<0.00005	<0.00005	0.0000752	0.000192	0.0000630	0.0000740	0.0000553	0.0000523
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.00150	<0.001	0.00100	<0.001	<0.001	<0.001	<0.001
	Magnesium (Mg)	mg/L	-	-	2.32	6.02	9.12	1.67	6.84	9.06	6.71	15.5	23.7	2.11	8.60	9.94	1.96	7.21	10.2
	Manganese (Mn)	mg/L	0.935[β]	-	0.00145	0.00125	0.000717	0.00167	0.00112	0.000765	0.000680	0.000400	0.000300	0.00201	0.00101	0.00178	0.00204	0.00126	0.00200
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005
	Molybdenum (Mo)	mg/L	0.073	-	0.0000530	0.000276	0.000402	0.0000535	0.000220	0.000376	0.000176	0.000246	0.000385	0.0000602	0.000257	0.000486	0.0000727	0.000243	0.000487
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	0.000775	0.000555	<0.0005	<0.0005	0.000743
	Potassium (K)	mg/L	-	-	0.398	0.839	1.09	0.360	0.838	1.03	0.582	0.990	1.38	0.358	0.830	1.09	0.372	0.792	1.07
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	0.0000930	0.000154	0.000493	<0.00005	0.0000732	0.0000505	<0.00005	<0.00005	0.0000570
	Silicon (Si)	mg/L	-	-	0.560	0.963	1.02	0.540	1.06	1.08	0.510	0.800	0.750	0.448	0.940	1.08	0.443	0.967	1.03
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.461	1.89	3.08	0.377	1.84	2.73	0.410	1.05	2.43	0.337	1.50	2.64	0.404	1.70	2.60
	Strontium (Sr)	mg/L	-	-	0.00335	0.0110	0.0166	0.00304	0.0122	0.0158	0.00653	0.0149	0.0228	0.00260	0.0113	0.0159	0.00258	0.00998	0.0153
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.000108	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.00412	0.00573	0.00347	0.00509	0.00451	0.00364	0.00116	<0.0006	<0.0005	0.00433	0.00338	0.00361	0.00438	0.00358	0.00286
	Uranium (U)	mg/L	0.015	-	0.000143	0.00187	0.00415	0.000122	0.00147	0.00320	0.000396	0.00172	0.00316	0.000123	0.00148	0.00317	0.000141	0.00133	0.00270
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02[α]	0.030	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

[a] Canadian Water Quality Guideline for the protection of aquatic life (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[b] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

the mine operational period, indicating very low concentrations and no mine-related influence (Appendix Figure C.23).

Total and dissolved water chemistry parameter concentrations that were slightly (3 to 5 times higher), moderately (5 to 10 times higher), or highly (≥ 10 times higher) elevated at mine-exposed Mary River stations relative to reference or baseline concentrations are detailed in Appendix Tables C.59 and C.61. In addition to the total chromium concentration at Station E0-20 in the summer, the sulphate concentration at Station E0-10 and total lead concentration at Station E0-20 were also elevated relative to both reference and baseline in the summer (Appendix Table C.59, Appendix Figure C.23). In the fall, the DOC concentration at downstream Station C0-10 was elevated compared to both reference and baseline (Appendix Table C.59, Appendix Figure C.23). A visual assessment of temporal data indicated no consistent directional patterns for sulphate at the E0 series stations during the summer, with concentrations remaining below the AEMP benchmark and WQG since mine operations began in 2015 (Appendix Figure C.23). Lead concentrations at the E0 series stations have generally been within the baseline range, similar to reference concentrations, and below the AEMP benchmark and WQG (Appendix Figure C.23), except in 2019 and 2020, when concentrations were higher across all Mary River sites (attributed to particulate bound concentrations and relatively high turbidity; Minnow 2022). At the C0 series stations, there have been no consistent increasing or decreasing patterns in DOC concentrations over the mine operational period; though summer and/or fall concentrations at Mary River reference, E0 mine-adjacent, and C0 downstream stations were more variable and higher in 2024 than during the baseline period and recent mine operational years (i.e., 2022 and 2023). DOC concentrations may be influenced by factors that vary naturally from year to year depending on local temperature and melt conditions, including organic matter decomposition rates, seasonal permafrost thaw, and the amount of runoff transporting DOC from surrounding soils and vegetation. The similar pattern in DOC concentrations observed at Mary River reference stations and the E0 and C0 series stations suggest natural variability is the dominant factor influencing DOC in the Mary River. Ongoing water quality monitoring will continue to assess for further evidence of potential mine-related influences.

Overall, water chemistry at mine-exposed areas of the Mary River in 2024 were generally consistent with reference and baseline, met AEMP benchmarks and WQGs, and demonstrated no consistent increasing temporal patterns over the mine operations period. These results suggest no adverse mine-related influences on water quality in the Mary River.

5.1.2 Phytoplankton

Chlorophyll-a concentrations at various locations along the Mary River – upstream (G0 series of stations), adjacent to (E0 series of stations), and downstream (C0 series of stations) from the mine – were generally consistent with concentrations observed at the G0-09 Mary River reference station and other reference stream stations during the 2024 spring, summer, and fall sampling events, based on qualitative comparisons (Figure 5.2, Appendix Table E.14). Chlorophyll-a concentrations remained well below the AEMP benchmark of 3.7 µg/L at all Mary River sampling stations and seasonal sampling events in 2024 (Figure 5.2, Appendix Table E.14). Measured chlorophyll-a concentrations indicate low phytoplankton productivity and oligotrophic conditions based on chlorophyll-a concentrations (i.e., <8 ug/L; Dodds et al. (1998); Appendix Table E.1) and total phosphorus concentrations (i.e., <10 ug/L; CCME 2024b; Table 5.1, Appendix Table C.58; see Section 3.1.2 for additional trophic status classification details).

Chlorophyll-a concentrations at the mine-exposed stations in 2024 were consistently lower across all seasons when compared to baseline values and were similar to concentrations at the reference stations (Figure 5.3). Concentrations during the baseline and construction periods were more variable compared to the operational period, and the baseline period featured relatively high LRLs compared to 2024, which may partially explain the observed difference between baseline and 2024 concentrations (Figure 5.3). Despite the potential influence of shifting LRLs, chlorophyll-a concentrations at both mine-exposed and reference stations in 2024 were generally similar to (fall), or slightly lower than (spring and summer), the range of concentrations observed during previous operational years (Figure 5.3). However, concentrations in reference stream samples in spring and summer 2024 were also lower compared to the rest of the operational period and to spring and summer in 2023, suggesting that the difference in both the Mary River and reference streams was likely due to natural inter-annual variation. Ongoing monitoring will continue to evaluate for evidence of consistent temporal patterns.

Overall, the available data indicate no consistent directional (i.e., increasing or decreasing) changes in chlorophyll-a concentrations at the Mary River mine-exposed areas across any seasonal sampling events during the baseline (2005 to 2013), construction (2014), and operational (2015 to 2023) periods. Additionally, the stream has remained oligotrophic and chlorophyll-a concentrations in 2024 remained well below the AEMP benchmark (Figure 5.3). These results indicate no adverse mine-related effects on phytoplankton productivity at the Mary River in 2024.

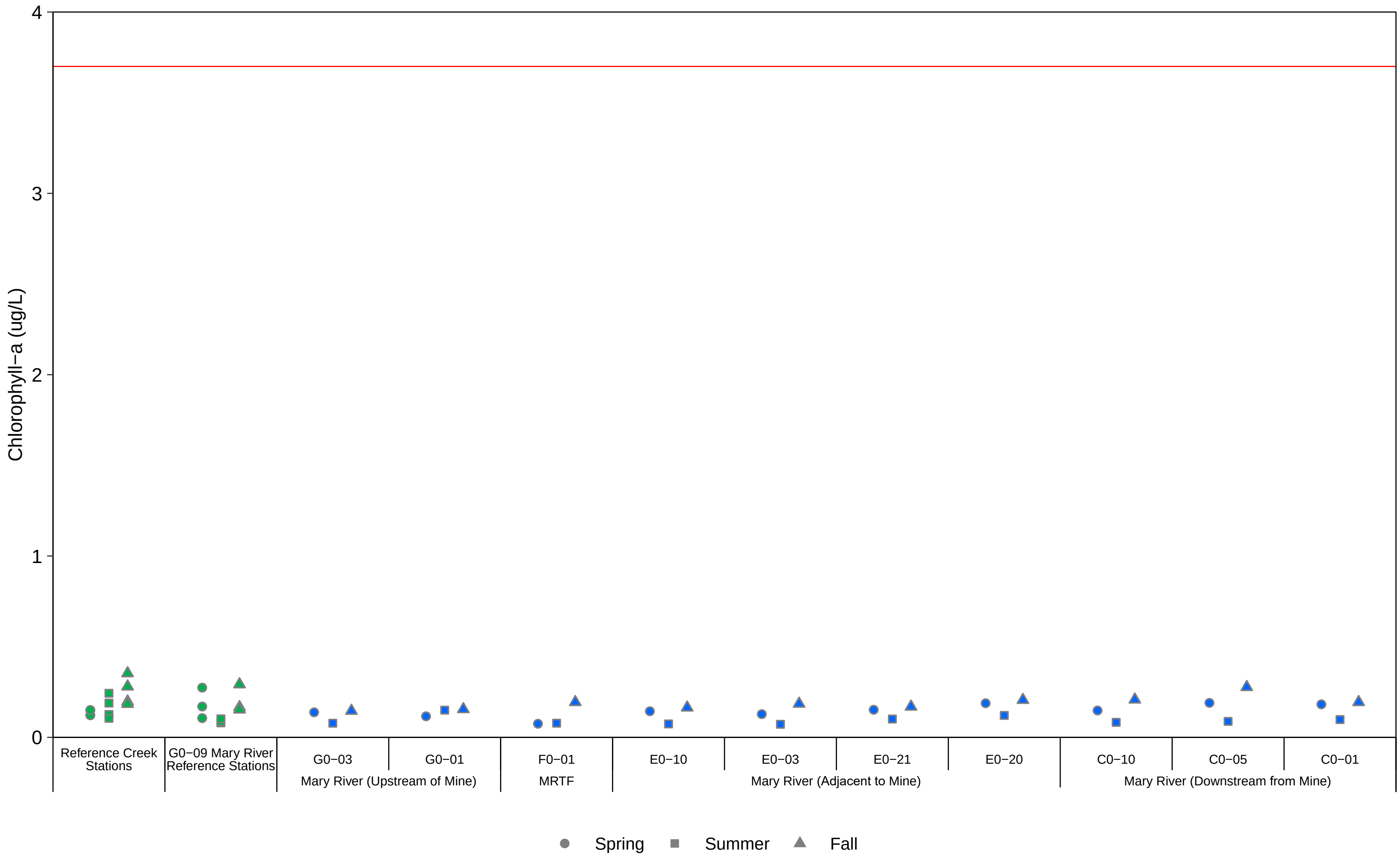

Figure 5.2: Chlorophyll−a Concentrations at Mary River (G0, F0, E0, and C0 Series) Phytoplankton Monitoring Stations Located Upstream, Adjacent to, and Downstream from the Mine, Mary River Project CREMP, 2024

Notes: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Red line indicates AEMP Benchmark. Reference Creek Stations includes data from stations CLT−REF4, CLT−REF3, MRY−REF3, and MRY−REF2. G0−09 Mary River Reference Stations includes data from stations G0−09, G0−09−A, and G0−09−B. MRTF = Mary River Tributary−F. Reference areas are shown in green and mine−exposed areas are shown in blue.

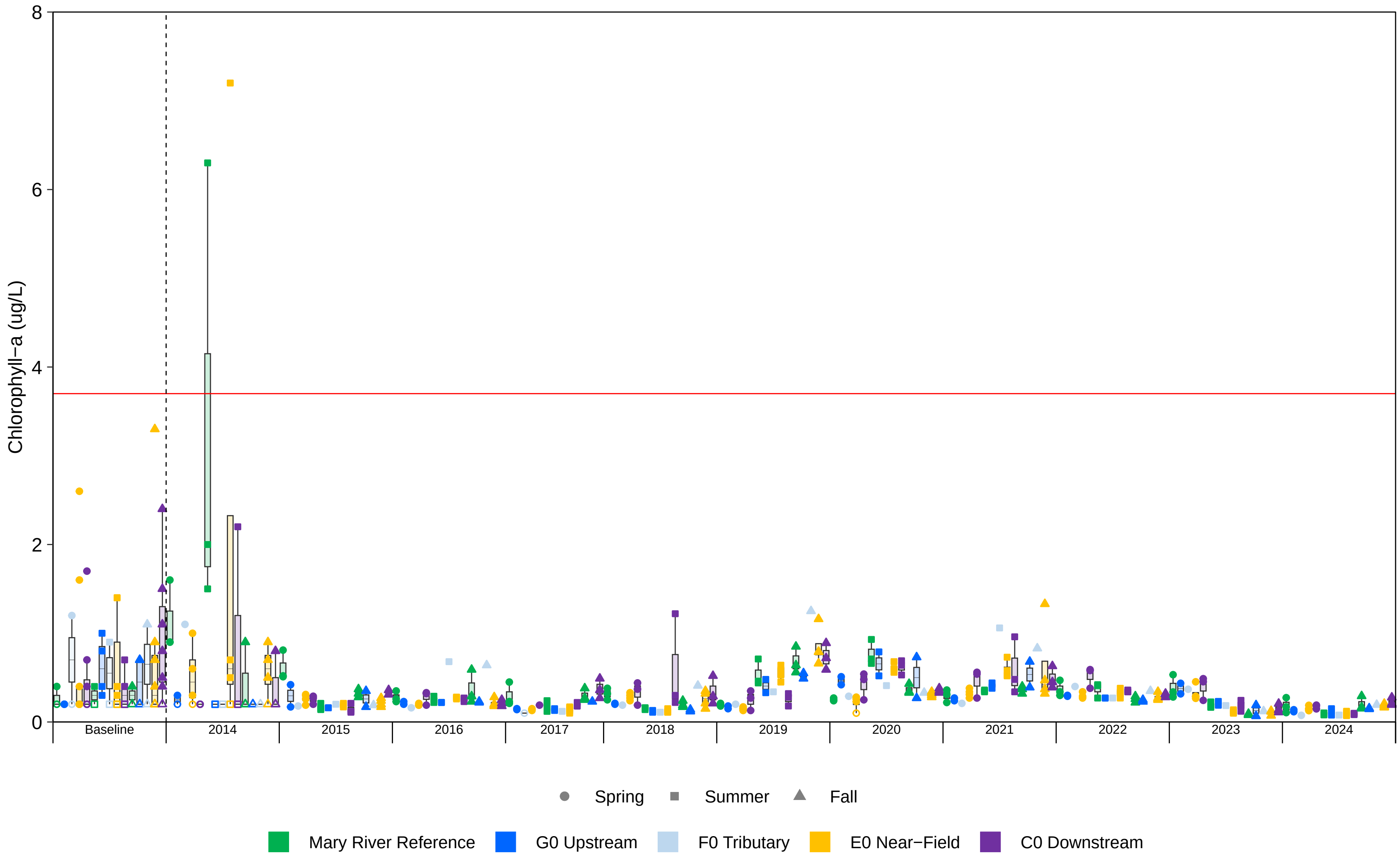

Figure 5.3: Temporal Comparison of Chlorophyll−a Concentrations at Mary River Stations (G0, F0, E0, and C0 Series) for Mine Baseline (2005 to 2013), Construction (2014), and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Tributary station is station F0−01. E0 Near−Field is stations E0−10, E0−03, E0−21, and E0−20. C0 Downstream is stations C0−10, C0−05, and C0−01. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

5.1.3 Benthic Invertebrate Community

In 2024, the BIC at the Mary River mine-exposed areas (G0, E0, and C0 series stations) was largely comparable to the BIC at the Mary River reference area (Station G0-09), except for significant differences in density, richness, and the relative proportions of *Simuliidae* and burrowers (Table 5.2, Appendix Figure F.14, Appendix Table F.45). Specifically, density was significantly higher at Station C0-05 compared to the reference area and the MOD was outside of the CES_{BIC} of ± 2 SD, suggesting an ecologically meaningful difference relative to reference (Table 5.2, Appendix Figure F.14, Appendix Table F.45). Taxonomic richness and relative proportions of burrowers were significantly higher at Stations E0-20 and C0-05 relative to the reference area; however, the differences were only considered ecologically meaningful for richness (Table 5.2, Appendix Figure F.14, Appendix Table F.45). The relative proportion of *Simuliidae* was significantly lower at Station E0-01 compared to the reference area but the MOD remained within the CES_{BIC} of ± 2 SD, indicating the difference was not ecologically meaningful (Table 5.2, Appendix Figure F.14, Appendix Table F.45). Despite there being few significant differences among exposed and reference areas for the other BIC endpoints, significant differences in overall BIC composition, as indicated by the Bray-Curtis Index, were observed at Stations G0-03 and C0-05 relative to the reference Station G0-09 (Appendix Table F.46).

The results of the spatial comparisons indicate that, although the overall BIC composition in mine-exposed areas of the Mary River remains largely comparable to reference conditions, there are subtle, localized differences in density and taxonomic richness at specific stations. The significantly higher density and richness observed at Stations C0-05 and E0-20 may be due to increased organic matter availability or habitat conditions that support greater invertebrate productivity and diversity. Notably, bryophytes were more common at Station C0-05 compared to the reference station, whereas the E0 series stations had lower bryophyte and algae presence relative to reference (Appendix Table F.37). These differences, along with higher aqueous DOC concentrations at the E0 and C0 series stations compared to the reference Station G0-09 (Appendix Table C.58), support the concept that variations in organic matter availability between mine-exposed and reference areas of Mary River may be influencing BIC composition. Significantly greater water depth at Station C0-05 compared to the reference area (Appendix Table F.39) supports that habitat differences may be influencing BIC composition. In addition to organic matter availability and habitat conditions, potential mine-related influences on water quality could be contributing to the observed differences in the BIC between mine-exposed and reference areas. The E0 and C0 series stations are located downstream of the MRTF confluence, which receives effluent from the MS-08 FDP (Figure 2.4) and where significant increasing trends of aqueous nitrate and sulphate concentrations have been identified

Table 5.2: Statistical Comparisons of Benthic Invertebrate Community Endpoints among Mary River Reference (G0-09), Upstream (G0-03), and Mine-Exposed (E0-01, E0-20, C0-05) Study Areas, Mary River Project CREMP, August 2024

Endpoint	Overall 5-Area Comparison				Pair-wise, *post hoc* comparisons				
	Statistical Test[a]	Data Transformation	Significant Difference Among Areas?	P-value	Area	Mean	Standard Deviation	MOD[b]	Pairwise Comparison
Density (org/m^2)	ANOVA	log10	YES	0.003	G0-09 Ref	50.2	29.6	nc	BC
					G0-03	101	70.8	1.1	BC
					E0-01	57.4	41.4	-0.38	C
					E0-20	524	717	3.0	AB
					C0-05	612	479	4.4	A
Richness (No. Taxa)	ANOVA	none	YES	0.002	G0-09 Ref	4.20	0.837	nc	C
					G0-03	6.40	2.70	2.6	BC
					E0-01	5.20	2.77	1.2	BC
					E0-20	9.60	4.51	6.5	AB
					C0-05	13.0	3.74	10	A
Simpson's Evenness (Krebs)	ANOVA	none	YES	0.027	G0-09 Ref	0.864	0.0705	nc	AB
					G0-03	0.930	0.0363	0.94	A
					E0-01	0.875	0.0492	0.16	AB
					E0-20	0.875	0.0524	0.16	AB
					C0-05	0.733	0.160	-1.9	B
% Nemata	K-W	rank	NO	0.714	G0-09 Ref	1.67	3.73	nc	A
					G0-03	0.800	1.79	nm	A
					E0-01	0.667	1.49	nm	A
					E0-20	1.30	2.58	nm	A
					C0-05	2.20	3.28	nm	A
% Hydracarina	K-W	rank	NO	0.714	G0-09 Ref	1.67	3.73	nc	A
					G0-03	2.00	4.47	nm	A
					E0-01	3.42	3.26	nm	A
					E0-20	0.162	0.264	nm	A
					C0-05	0.351	0.589	nm	A
% Chironomidae	K-W	rank	NO	0.288	G0-09 Ref	66.0	24.6	nc	A
					G0-03	61.2	18.2	0.017	A
					E0-01	65.3	38.0	0.22	A
					E0-20	79.3	10.9	0.36	A
					C0-05	83.9	7.38	0.51	A
% Metal Sensitive Chironomidae	K-W	rank	NO	0.513	G0-09 Ref	44.7	29.0	nc	A
					G0-03	31.4	6.67	-1.4	A
					E0-01	45.1	26.0	-0.34	A
					E0-20	39.6	16.0	-0.96	A
					C0-05	49.1	22.0	-0.19	A
% Simuliidae	ANOVA	none	YES	0.020	G0-09 Ref	29.1	18.8	nc	A
					G0-03	30.0	13.8	0.048	A
					E0-01	8.25	8.18	-1.1	B
					E0-20	14.5	8.18	-0.77	AB
					C0-05	9.85	7.60	-1.0	AB
% Tipulidae	K-W	rank	NO	0.336	G0-09 Ref	0	0	nc	A
					G0-03	5.24	9.65	nm	A
					E0-01	20.0	44.7	nm	A
					E0-20	0.0821	0.184	nm	A
					C0-05	1.35	1.63	nm	A
% Collector Gatherers FFG	ANOVA	none	NO	0.575	G0-09 Ref	69.3	20.5	nc	A
					G0-03	51.3	16.2	-0.87	A
					E0-01	65.0	38.0	-0.21	A
					E0-20	70.0	13.2	0.034	A
					C0-05	73.4	14.2	0.20	A
% Filterers FFG	K-W	rank	NO	0.403	G0-09 Ref	0	0	nc	A
					G0-03	14.4	22.1	nm	A
					E0-01	1.88	4.19	nm	A
					E0-20	3.31	5.90	nm	A
					C0-05	1.28	1.82	nm	A
% Shredders FFG	K-W	rank	NO	0.108	G0-09 Ref	0	0	nc	A
					G0-03	15.7	13.0	nm	A
					E0-01	22.6	43.4	nm	A
					E0-20	8.70	14.1	nm	A
					C0-05	5.03	4.13	nm	A
% Clingers HPG	ANOVA	none	YES	0.062	G0-09 Ref	30.7	20.5	nc	AB
					G0-03	42.4	17.9	0.57	A
					E0-01	15.0	13.9	-0.77	B
					E0-20	23.4	16.8	-0.36	AB
					C0-05	14.2	9.52	-0.81	B
% Sprawlers HPG	ANOVA	none	NO	0.599	G0-09 Ref	67.6	22.7	nc	A
					G0-03	49.7	12.7	-0.79	A
					E0-01	64.4	37.9	-0.14	A
					E0-20	68.7	15.3	0.047	A
					C0-05	70.3	11.5	0.12	A
% Burrowers HPG	K-W	rank	YES	0.048	G0-09 Ref	1.67	3.73	nc	C
					G0-03	7.84	9.42	nm	ABC
					E0-01	20.7	44.4	nm	BC
					E0-20	7.92	2.06	nm	AB
					C0-05	15.5	7.64	nm	A

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

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

Notes: MOD = Magnitude of Difference. nc = no comparison. nm = MOD could not be calculated due to SD = 0. FFG = Functional Feeding Group. HPG = Habitat Preference Group.

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

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

since the baseline period and over the mine operational period, though concentrations have remained below AEMP benchmarks (see Section 5.2.1.2). These potential mine-related influences on water quality in MRTF have the potential to contribute to the observed differences in BIC downstream in the Mary River; however, water quality results indicate that most effects observed in MRTF are diluted and not observed in the Mary River at the E0 and C0 series stations (see Sections 5.1.1.2 and 5.2.2.2). The lack of consistent spatial patterns throughout the Mary River suggests no mine-related influence on BIC.

Most BIC endpoints assessed for the Mary River mine-exposed areas had at least one operational year (2015 to 2024) that significantly differed from baseline (2007 and/or 2011; Appendix Tables F.47 to F.51, Appendix Figure F.15). At Station G0-03 in 2024, Simpson's Evenness and relative proportions of metal sensitive *Chironomidae* were significantly higher, whereas the relative proportions of *Chironomidae* and collector gatherers was significantly lower compared to the 2007 baseline (Appendix Table F.48, Appendix Figure F.15). Although Simpson's Evenness was almost consistently higher during mine operations than baseline, the differences were not ecologically meaningful (Appendix Table F.48, Appendix Figure F.15). Relative proportions of *Chironomidae* at Station G0-03 were typically lower after 2015 compared to baseline in 2007, with ecologically meaningful differences observed in 2015, 2020, 2022, and 2024 (Appendix Table F.48, Appendix Figure F.15). However, it is worth noting that relative proportions of *Chironomidae* at the reference Station G0-09 were also lower after 2015 relative to 2007 (Appendix Table F.47, Appendix Figure F.15), which suggests there may have been regional, rather than mine-related factors, at play. A similar pattern was observed for collector gatherers at Station G0-03 (exposed) and the reference station (Appendix Table F.48, Appendix Figure F.15). Relative proportions of metal-sensitive *Chironomidae* at Station G0-03 showed an ecologically significant increase in 2024 compared to the 2007 baseline for the first time since 2018 (Appendix Table F.48, Appendix Figure F.15). At Station E0-01 in 2024, both density and taxonomic richness were significantly lower compared to the 2007 baseline but differences were not considered ecologically meaningful (Appendix Table F.49, Appendix Figure F.15). The relative proportion of filterers was also significantly lower at Station E0-01 in 2024 relative to baseline, with ecologically meaningful differences (Appendix Table F.49, Appendix Figure F.15). In contrast, Simpson's Evenness at E0-01 has been consistently higher than baseline since 2015, and the MODs suggest ecologically meaningful differences (Appendix Table F.49, Appendix Figure F.15). Similarly, Simpson's Evenness at Station E0-20 has significantly increased since 2015 compared to the 2011 baseline, with ecologically meaningful differences observed since 2017 (Appendix Table F.50, Appendix Figure F.15). Notably, Simpson's Evenness at the reference Station G0-09 was also significantly and meaningfully higher in most mine operational years (i.e., 2015, 2016, 2018, 2020 to 2022, and 2024) relative to 2007 (Appendix Table F.47, Appendix Figure F.15). Also, at Station E0-20, relative proportions of collector-gatherers have significantly increased since 2015, with ecologically meaningful differences in every operational year except 2020. In 2024 specifically, the relative proportions of metal-sensitive *Chironomidae* were higher, whereas relative proportions of *Tipulidae* decreased relative to baseline, though only metal-sensitive *Chironomidae* showed an ecologically meaningful MOD (Appendix Table F.50, Appendix Figure F.15). At Station C0-05, the relative proportions of *Chironomidae* were significantly lower from 2020 to 2024 compared to 2007, whereas relative proportions of metal-sensitive *Chironomidae* increased significantly in 2024 compared to 2011, with the differences being ecologically meaningful (Appendix Table F.51, Appendix Figure F.15). Relative proportions of collector-gatherers have shown significant and ecologically meaningful increases since 2018 compared to 2007, whereas filterers have frequently exhibited significant decreases (2015, 2016, 2019, 2021, 2022, and 2024) compared to 2011 (Appendix Table F.51, Appendix Figure F.15).

These findings indicate that while the overall BIC composition in Mary River mine-exposed areas has changed since baseline years, the ecological significance of these shifts varies by station and taxonomic group. Further, shifts in BIC endpoints over time have also been observed at the upstream reference area, suggesting there are broader regional factors (e.g., interannual variability in air and water temperatures and/or flows) influencing the BIC at the Mary River stations. For example, the higher Simpson's Evenness across several stations since 2015, including at the G0-09 reference station, suggests a shift in community structure throughout the system, again, potentially reflecting habitat changes or even altered resource availability. At Station E0-01, the concurrent decrease in density, taxonomic richness, and filterers may be due to potential habitat changes (e.g., absence of in-stream vegetation). Meanwhile, the increase in collector-gatherers at Station E0-20 and C0-05 may reflect greater organic matter deposition, which may be influenced by both natural and mine-related factors.

Overall, the BIC community in Mary River mine-exposed areas in 2024 remained largely comparable to reference conditions; however, localized differences suggest habitat and environmental variations across stations. The significantly higher density and richness at Stations C0-05 and E0-20, alongside differences in bryophyte presence and aqueous DOC concentrations, indicate potential differences in organic matter availability, which may be influencing BIC structure. Although some stations, such as G0-03 and C0-05, exhibited significant differences in overall BIC composition relative to reference, the lack of widespread, ecologically meaningful differences is not suggestive of mine-related influence on BIC in the Mary River. Temporal analysis indicates that many BIC endpoints across Mary River BIC stations have ecologically meaningfully differed between baseline and operational years, with shifts in community structure observed at both exposed and reference stations. Although continued monitoring is essential to track long-term patterns and potential effects, the current findings do not suggest mine-related effects on BIC across the Mary River system.

5.1.4 Effects Assessment and Recommendations

In 2024, water chemistry at the Mary River (G0, E0, and C0 series stations) met all AEMP benchmarks and WQGs across all seasonal sampling events (spring, summer, fall) except for individual sample concentrations of total aluminum, which slightly exceeded the WQG of 0.100 mg/L at the G0-01 upstream station in the spring and summer and total chromium which exceeded the WQG of 0.001 mg/L at the E0-20 mine-adjacent station in the summer. In comparisons of water quality parameter concentrations to reference and baseline across all seasons or within a single season, the following parameters were elevated, indicating a potential mine-related effect:

- Summer: total chromium (Station E0-20), total lead (Station E0-20), sulphate (Staton E0-10); and
- Fall: DOC (Station C0-10).

Aluminum and chromium concentrations were either not elevated compared to reference and baseline in individual seasons in 2024 and/or showed no evidence of increasing patterns over time at Mary River sampling stations indicating no-mine related influence. Similarly, visual assessment of temporal data indicated no consistent increasing patterns for sulphate or lead at the E0 series stations, or for DOC at the C0 series stations since mine operations began in 2015 also indicated no mine-related influence.

Sediment is collected every three years from streams monitored under the CREMP, with most recent samples taken in 2023; therefore, sediment quality results are not included in this report. No adverse mine-related effects on chlorophyll-a (i.e., primary productivity) or to the BIC were observed in 2024.

According to the Mary River Project AEMP Management Response Framework, the absence of any mine-related changes in water chemistry concentrations or to biota, as observed at the Mary River in 2024, requires no further management action (Figure 2.6).

Comparison to FEIS Predictions

TA comparison of water quality at Mary River (G0, E0, and C0) in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management) indicated all parameter concentrations were within the Level II significance rating for magnitude expected for the watercourse during mine operations. Therefore, Mary River water quality conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

Water quality at Mary River in 2024 where parameter concentrations were within applicable FEIS significance rating magnitude predictions also meant that FEIS predictions for (absence of) effects on arctic charr health and condition were also met. Therefore, arctic charr health and condition at Mary River in 2024 conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

5.2 Mary River Tributary-F (MRTF)

5.2.1 Water Quality

5.2.1.1 *In Situ* Water Quality

In 2024, *in situ* water quality was assessed at Mary River Tributary-F (Station F0-01) concurrent with water quality sampling in spring, summer, and fall (Figure 2.1). Dissolved oxygen in water at MRTF was consistently near or above saturation (> 97%) during all spring, summer, and fall monitoring events (Appendix Figure C.22; Appendix Tables C.1 to C.3). In addition, dissolved oxygen concentrations were consistently well above the WQG of 9.5 mg/L (lowest acceptable concentration for early life stages of cold-water biota) at MRTF during all sampling in 2024 (Appendix Tables C.1 to C.3). *In situ* pH at MRTF was generally comparable to pH at the G0-09 series reference stations during the spring, summer, and fall sampling events in 2024 and was consistently within WQG limits (Appendix Figure C.22; Appendix Tables C.1 to C.3). Specific conductance was lowest in spring and highest in summer at MRTF (Appendix Figure C.22; Appendix Tables C.1 to C.3), reflecting natural seasonal differences related to proportion of flow from surface runoff (e.g., spring snowmelt) and baseflow/groundwater sources. Specific conductance was consistently higher at MRTF (F001) than at Mary River monitoring stations in all seasons (Appendix Figure C.22; Appendix Tables C.1 to C.3).

Mine effluent is discharged to MRTF from the MS-08 FDP (Figures 1.1 and 2.2) and therefore this tributary is a known source of mine-related influence to the Mary River (Minnow 2024b; Section 5.2.1.2) and has been monitored through EEM under the MDMER. The most recent EEM study at MRTF was completed in 2023 (Minnow 2024b)[51]. *In situ* water quality at the EEM study areas in 2023 indicated that dissolved oxygen concentrations did not differ significantly between the MRTF effluent-exposed and reference study areas and were well above the WQG lowest

[51] Under the MDMER, Baffinland is required to conduct EEM studies as a condition governing the authority to discharge effluent from the site. Three studies (phases), completed in 2017, 2020, and 2023 have been conducted to meet EEM biological sampling requirements at the Mary River Project.

acceptable concentration for early life stages of cold-water biota at both study areas (Minnow 2024b). Although pH was significantly lower at the effluent-exposed area than at the reference area of MRTF, the mean incremental difference in pH between areas was small (i.e., 0.27 units) and pH values were well within the WQG acceptable range for the protection of aquatic life (Minnow 2024b). Specific conductance was significantly higher at the effluent-exposed area than at the reference area at the time of the August 2023 EEM biological field study, indicating an effluent-related influence on water quality of the tributary (Minnow 2024b). At the time of benthic invertebrate community sampling, the corresponding proportion of MS-08 effluent at the MRTF effluent-exposed area below the effluent channel confluence was estimated as 13.6% (mean of estimated concentrations at the five benthic invertebrate community sampling stations; range of 2.5% to 21.4%)[52], based on extrapolation using specific conductance measures of reference water, effluent, and water downstream of effluent discharge (Minnow 2024b).

5.2.1.2 Water Chemistry

Water chemistry parameter concentrations at MRTF (Station F0-01) met all AEMP benchmarks and WQG during spring, summer, and fall sampling events in 2024 (Table 5.1, Appendix Table C.58). Total and dissolved parameter concentrations that were slightly (3 to 5 times higher), moderately (5 to 10 times higher), or highly (≥ 10 times higher) elevated relative to the Mary River reference area (G0-09 series) or baseline levels are outlined in Appendix Tables C.59 and C.61. Sulphate concentrations in 2024 were consistently slightly to highly elevated compared to reference and baseline concentrations across all seasons (Appendix Table C.59, Appendix Figure C.23). During the summer and fall, nitrate concentrations were moderately to highly elevated relative to reference and baseline conditions (Appendix Table C.59, Appendix Figure C.23). Finally, the total selenium concentration was moderately and highly elevated compared to reference and baseline concentrations, respectively in the fall (Appendix Table C.59, Appendix Figure C.23).

As part of the effects assessment of the 2022 CREMP, action level responses associated with MRTF water quality were not recommended (Minnow 2023). However, through the 2022 CREMP report review process, intervenors requested a trend analysis of MRTF water quality, focusing on nitrate and sulphate concentrations. In response, temporal trend analyses were conducted in 2023 and found significant increasing trends in nitrate and sulphate concentrations at MRTF since the baseline period (2005 to 2023) and over the years of mine operation (2015 to 2023; Minnow 2024a). Data for reference streams only dated as far back as 2014 but no similar trends

[52] The range in effluent concentrations estimated over the approximately 280 m of Mary River Tributary-F between MRTF-EXP1 and MRTF-EXP5 potentially represents an area of effluent mixing.

were observed at any of the reference streams (Minnow 2024a). Based on these findings, aqueous nitrate and sulphate were identified as requiring a Low Action Response within the AEMP Management Response Framework in 2023 (Minnow 2024a).

In 2024, temporal trend analyses were repeated, incorporating the most recent nitrate and sulphate concentration data. These analyses confirmed significant increasing trends in nitrate and sulphate concentrations across all sampling seasons combined, and in each individual sampling season since the baseline period (2005 to 2024), as well as in all sampling seasons combined and in spring for nitrate concentrations only over the operational period (2015 to 2024; Appendix Tables H.12 and H.13). Again, no similar trends were observed in reference streams, which indicates a mine-related influence rather than naturally increasing concentrations at a regional scale. Visual assessment of temporal data suggests that concentrations of nitrate and sulphate began increasing at MRTF in 2019 and 2017, respectively, although for both parameters, concentrations did not increase consistently over time (Appendix Figure C.23). Concentrations of both nitrate and sulphate in 2023 were lower than in previous years of mine operations and more similar to reference and baseline (Appendix Figure C.23). In 2024, fall concentrations increased again but remained below AEMP benchmarks and WQGs (Appendix Figure C.23). Temporal patterns of nitrate and sulphate concentrations at MRTF therefore do not suggest that mine-related influences have intensified over the mine operations period.

Selenium concentrations at MRTF have frequently been below the LRL and have consistently been below the WQG throughout mine operations, suggesting very low concentrations and no potential for effects on aquatic biota (Appendix Figure C.23). Evaluation for a temporal pattern is confounded by changing LRLs, where the LRL for most samples in 2020, 2021, and 2022 was high (0.001 mg/L) relative to measured concentrations in spring and summer of 2023 (0.000125 and 0.000192 mg/L, respectively) and 2024 (0.000154 and 0.000493 mg/L, respectively). Given the few results above LRLs, evidence does not suggest a potential mine-related influence on selenium concentrations at MRTF. Ongoing monitoring will continue to evaluate for patterns of increasing concentrations over time, and relative to the WQG.

Mine-related influences on nitrate and sulphate concentrations may be associated with effluent discharge (i.e., from the MS-08 FDP into MRTF) as indicated by the highest concentrations of these parameters throughout the Mary River System occurring at MRTF Station F0-01 in 2024 (Appendix Table G.58). However, the effluent quality at the MS-08 FDP is routinely monitored for compliance with MDMER regulations, with parameter concentrations consistently meeting effluent quality limits and mean annual receiving environment concentrations of aqueous nitrate and sulphate below WQGs over the Phase 3 EEM from 2021 to 2023 (i.e., at the Mary River Effluent-Exposed station downstream of the MS-08 FDP [MS-08-DS]; Minnow 2024b).

Further, concentrations of sulphate and nitrate downstream of F0-01 (e.g., at the Mary River E0 and C0 series stations) were not generally elevated compared to reference and baseline suggesting that effluent discharge from MS-08 FDP is likely being rapidly diluted in the Mary River before reaching Mary Lake. This dilution effect reduces the concentrations of mine-influenced parameters, such that no mine-related influences were detected downstream along the Mary River system (see Section 5.1.1.2).

Overall, results from 2024 indicated a mine-related influence on nitrate and sulphate concentrations at MRTF. However, based on qualitative assessment of temporal patterns, this influence is not intensifying over time . Finally, all parameters remained below AEMP benchmarks and WQGs, indicating no potential for adverse effects to aquatic biota.

5.2.2 Phytoplankton

Chlorophyll-a concentrations at MRTF (Station F0-01), located downstream from the mine, were generally within the range of concentrations observed at the mine-exposed G0, E0, and C0 Station series along May River, as well as reference stream stations, during the 2024 spring, summer, and fall sampling events (Figure 5.2, Appendix Table E.14). These concentrations were below the AEMP benchmark of 3.7 µg/L during all seasonal sampling events at MRTF in 2024 (Figure 5.2, Appendix Table E.14). Measured chlorophyll-a concentrations indicate low phytoplankton productivity and oligotrophic conditions based on chlorophyll-a concentrations (i.e., <8 µg/L; Dodds et al. (1998); Appendix Table E.1) and total phosphorus concentrations (i.e., <10 µg/L; CCME 2024b; Table 5.1, Appendix Table C.58; see Section 3.1.2 for additional trophic status classification details).

In 2024, chlorophyll-a concentrations at MRTF stations were consistently lower across all seasons compared to baseline values but were similar to concentrations at reference stations (Figure 5.3). During the baseline and construction periods, concentrations showed greater variability than the more consistent levels observed during the mine operational period (Figure 5.3). The baseline period featured relatively high LRLs compared to 2024, which may partially explain the observed difference between these periods (Figure 5.3). Despite the potential influence of shifting LRLs, chlorophyll-a concentrations at both mine-exposed and reference stations in 2024 were generally similar to or slightly lower than those observed during previous operational years (i.e., fall concentrations aligned with past concentrations, whereas spring and summer concentrations were slightly lower; Figure 5.3). However, spring and summer chlorophyll-a concentrations in reference stream samples in 2024 were lower than in previous operational years and compared to spring and summer 2023, suggesting that the difference in both MRTF and reference streams are likely due to natural inter-annual variation. Ongoing monitoring will continue to evaluate for evidence of consistent temporal patterns.

Overall, the available data indicate no consistent directional (i.e., increasing or decreasing) changes in chlorophyll-a concentrations at MRTF mine-exposed areas across any seasonal sampling events during the baseline (2005 to 2013), construction (2014), and operational (2015 to 2023) periods. The stream has remained oligotrophic, with chlorophyll-a concentrations in 2024 well below the AEMP benchmark (Figure 5.3). These results indicate no adverse mine-related effects on phytoplankton productivity at MRTF in 2024.

5.2.3 Effects Assessment and Recommendations

In 2024, water chemistry at MRTF (Station F0-01) met all AEMP benchmarks and WQGs across all seasonal sampling events (spring, summer, fall). In comparisons of water quality parameter concentrations to reference and baseline across all seasons or within a single season, the following parameters were elevated, indicating a potential mine-related effect:

- All seasons (spring, summer, fall): sulphate;
- Summer: nitrate; and
- Fall: nitrate and total selenium.

Since 2022, concentrations of nitrate and sulphate have been identified as having potential mine-related influence (Minnow 2023 and 2024a). In temporal trend analyses completed in the 2023 CREMP and again in 2024, each showed statistically significant increasing trends since the baseline period and over the mine operations period (Minnow 2024a) and visual assessment of temporal data indicated that these increasing trends generally started in 2019 and 2017 for nitrate and sulphate, respectively. However, for both parameters concentrations have not increased consistently over time suggesting that while a mine-related influence is present, it is not intensifying over time. Selenium concentrations at MRTF have frequently been below the LRL and have consistently been below the WQG throughout mine operations, suggesting very low concentrations and no potential for effects on aquatic biota. Although visual assessment of the selenium concentrations in summer and fall of 2023 and 2024 suggests that concentrations at MRTF increased from 2023 to 2024, evaluation for a temporal pattern is confounded by changing LRLs and therefore evidence does not suggest a potential mine-related effect on selenium at MRTF. Mine-related influences on nitrate and sulphate concentrations may be associated with effluent discharge (i.e., from the MS-08 FDP into MRTF)

Sediment is collected every three years from streams monitored under the CREMP, with most recent samples taken in 2023; therefore, sediment quality results are not included in this report. No adverse mine-related effects on chlorophyll-a (i.e., primary productivity) were observed in 2024.

Under the AEMP Management Response Framework, a Low Action Response is required based on determination of mine-related influences on nitrate and sulphate due to concentrations that were elevated compared to reference and baseline in at least one season in 2024 and/or evidence of increasing trends/patterns over the mine operations period (Figure 2.6). The following actions are recommended:

- In 2025, temporal trend analysis of aqueous concentrations of nitrate and sulphate will be conducted for MRTF to further investigate temporal trends/patterns.
- In 2025, a special investigation will be conducted evaluating effluent and receiving water quality data that are routinely collected as part of MDMER requirements for the MS-08 FDP to evaluate influence of the MS-08 FDP as a potential source of nitrate and sulphate to MRTF.

According to the Mary River Project AEMP Management Response Framework, the absence of any mine-related effects on phytoplankton (as a measure of primary productivity), as observed at the MRTF in 2024, requires no further management action (Figure 2.6).

Comparison to FEIS Predictions

TA comparison of water quality at Mary River Tributary-F (F0-01) in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ-2 (Site Water Management) indicated all parameter concentrations were within the Level II significance rating for magnitude expected for the watercourse during mine operations. Therefore, Mary River water quality conformed with predictions made in the Baffinland FEIS (Baffinland 2012).

5.3 Mary Lake (BL0)

5.3.1 Water Quality

5.3.1.1 *In Situ* Water Quality

In 2024, profiles were developed from *in situ* water quality measured concurrent with water quality sampling in winter, summer, and fall (Figure 2.1), and *in situ* water quality was measured at the top and bottom of the water column concurrent with benthic invertebrate community sampling in August (Figure 2.3). Vertical profiles of *in situ* water temperature, dissolved oxygen, pH, and specific conductance measured at Mary Lake showed few substantial within-season differences between stations for each of the north and south basins during any of the winter, summer, or fall sampling events in 2023 (Appendix Figures C.25 to C.28, Appendix Tables C.62 to C.64). Water temperature was generally similar in winter, cooler in fall, and warmer in summer of 2024 in the north basin than in the south basin (Appendix Figure C.25). Compared to the south basin, dissolved oxygen saturation was similar in all seasons, and pH and specific conductance were higher in all seasons (Appendix Figures C.26 to C.28). Higher pH and specific conductance at the north basin likely reflects the predominant influence of the Tom River at the north basin versus the predominant influence of the Mary River at the south basin (Appendix Figures C.27 and C.28).

At the north and south basins of Mary Lake in 2024, mean water temperature increased with depth in the winter, decreased with depth in the summer, and was consistent throughout the water column in the fall (Figures 5.4 and 5.5). The reference lake had cooler and warmer surface temperatures than the Mary Lake north and south basins in summer and fall, respectively (Figures 5.4 and 5.5). During the August 2024 BIC sampling, the mean water temperature at the bottom of the water column at Mary Lake profundal stations was significantly warmer than at Reference Lake 3 but there was no difference between the lakes at littoral stations (Figure 5.6, Appendix Table C.65 and C.67).

Dissolved oxygen profiles showed declining concentrations and saturation with depth through the entire water column at Mary Lake north in the winter but, similar to at Reference Lake 3, concentrations and saturation were relatively consistent throughout the water column during the summer and fall (Figure 5.4). At the Mary Lake south basin, dissolved oxygen concentration and saturation increased with depth until approximately 5 m in all seasons before decreasing again toward the lake bottom in winter and summer and remaining consistent in fall (Figure 5.5). Compared to like-habitat stations at Reference Lake 3 during the August 2024 BIC sampling, dissolved oxygen concentrations were significantly lower at Mary Lake littoral and profundal stations and dissolved oxygen saturation was significantly lower at Mary Lake littoral stations; however, the differences were small (0.3 mg/L and 4%, respectively; Figure 5.6, Appendix Tables C.65 and C.67). Mean dissolved oxygen concentrations were well above the WQG of 9.5 mg/L (lowest acceptable concentration for early life stages of cold-water biota) near the bottom at littoral and profundal stations of Mary Lake and Reference Lake 3 during BIC sampling in August 2024, suggesting no ecologically meaningful differences in dissolved oxygen between lakes (Figure 4.5; Appendix Table C.41).

Water column profiles showed slightly decreasing pH with depth at Mary Lake north and south basins during summer, fall, and winter sampling events in 2024 (Figure 5.4). Mary Lake mean pH at Mary Lake north and south basins was generally higher (~ 0.3 to 0.7 pH units) than Reference Lake 3 during all seasons in 2024 (Figure 5.4 and 5.5). The pH near the bottom of the water column at profundal stations of Mary Lake was significantly lower than like-habitat at the reference lake areas during the August 2024 BIC sampling but all values were consistently within WQG (Figure 5.6, Appendix Tables C.65 and C.67).

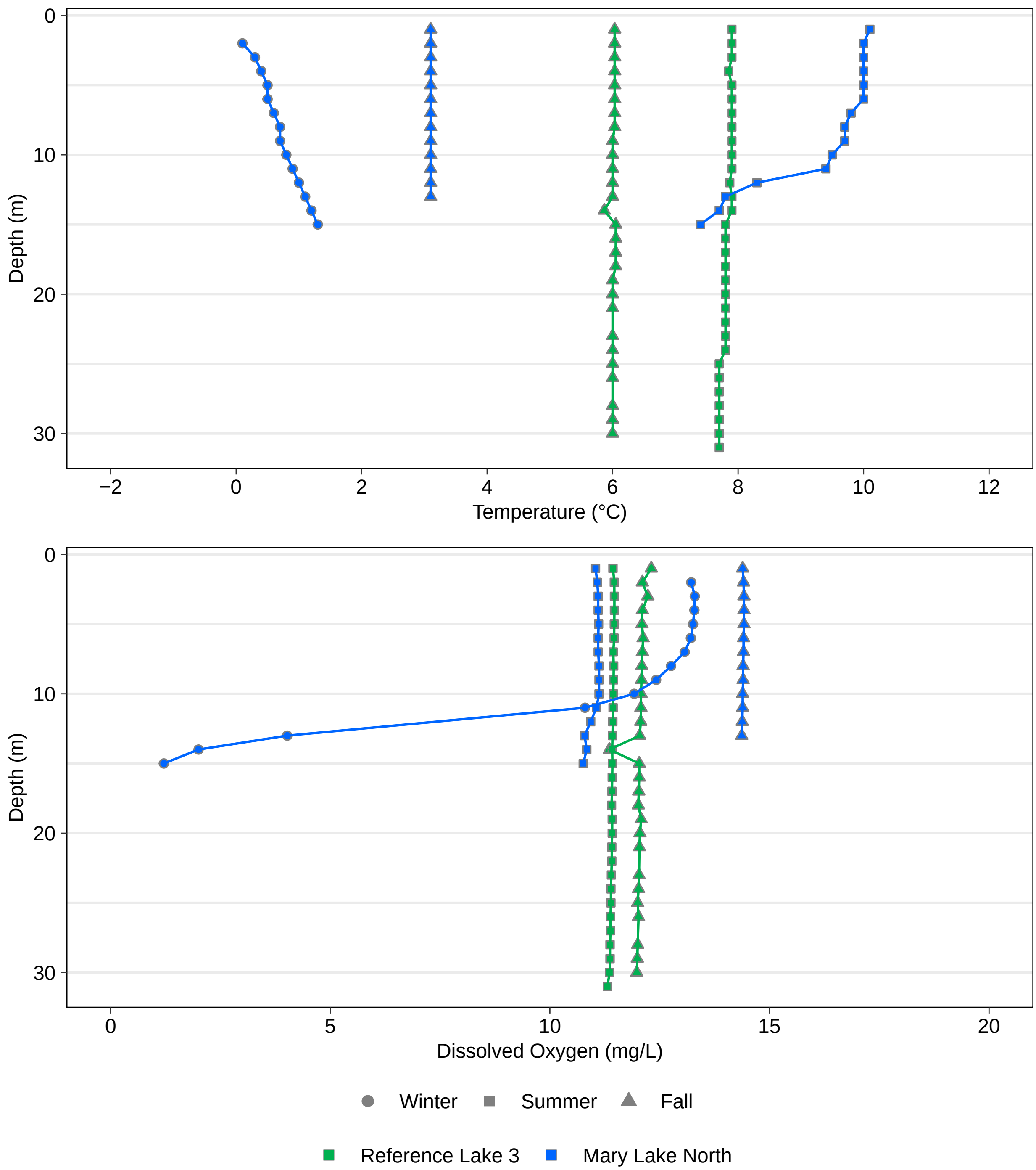

Figure 5.4: Average In Situ Water Quality with Depth from Surface at Mary Lake North Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

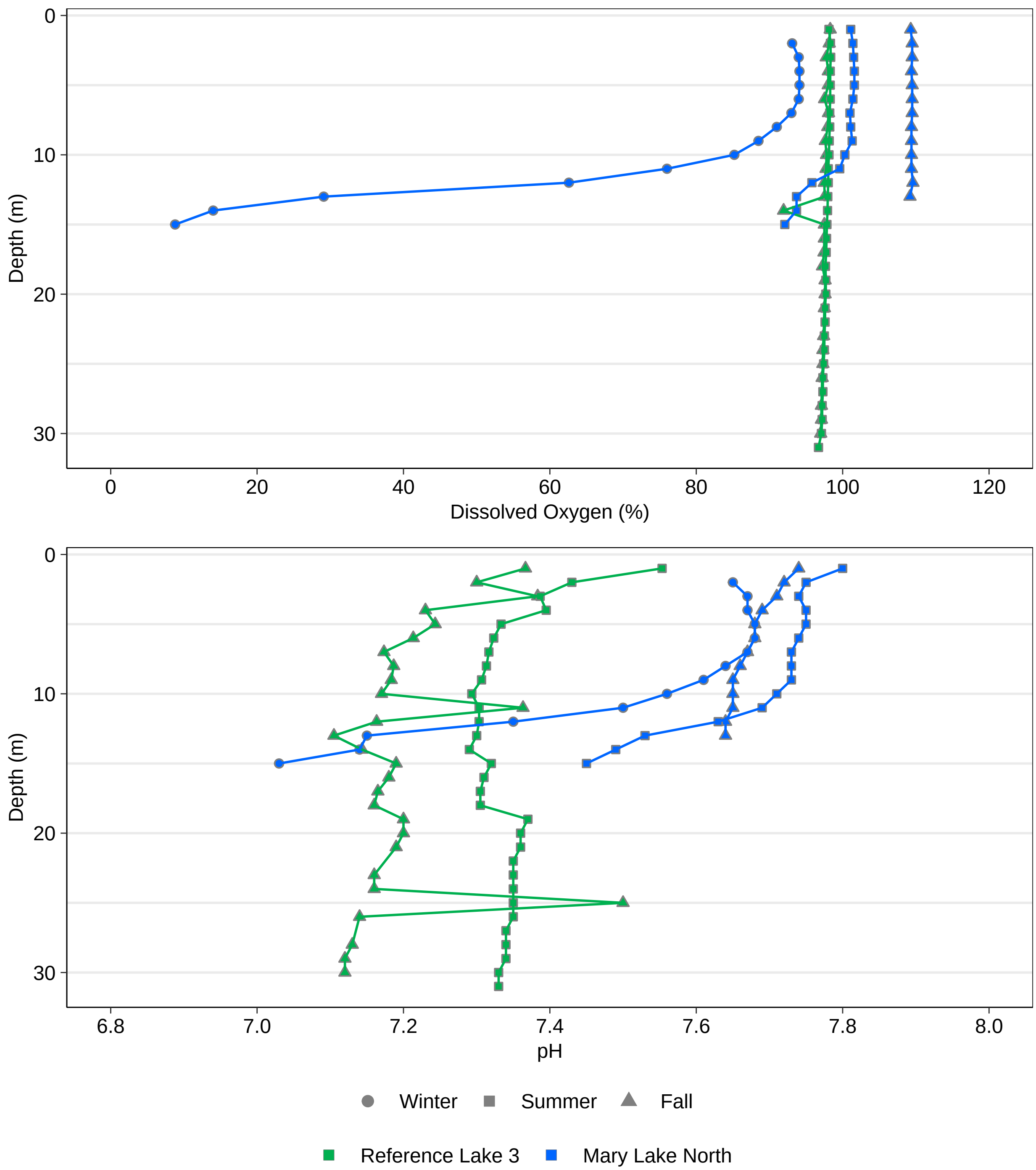

Figure 5.4: Average In Situ Water Quality with Depth from Surface at Mary Lake North Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

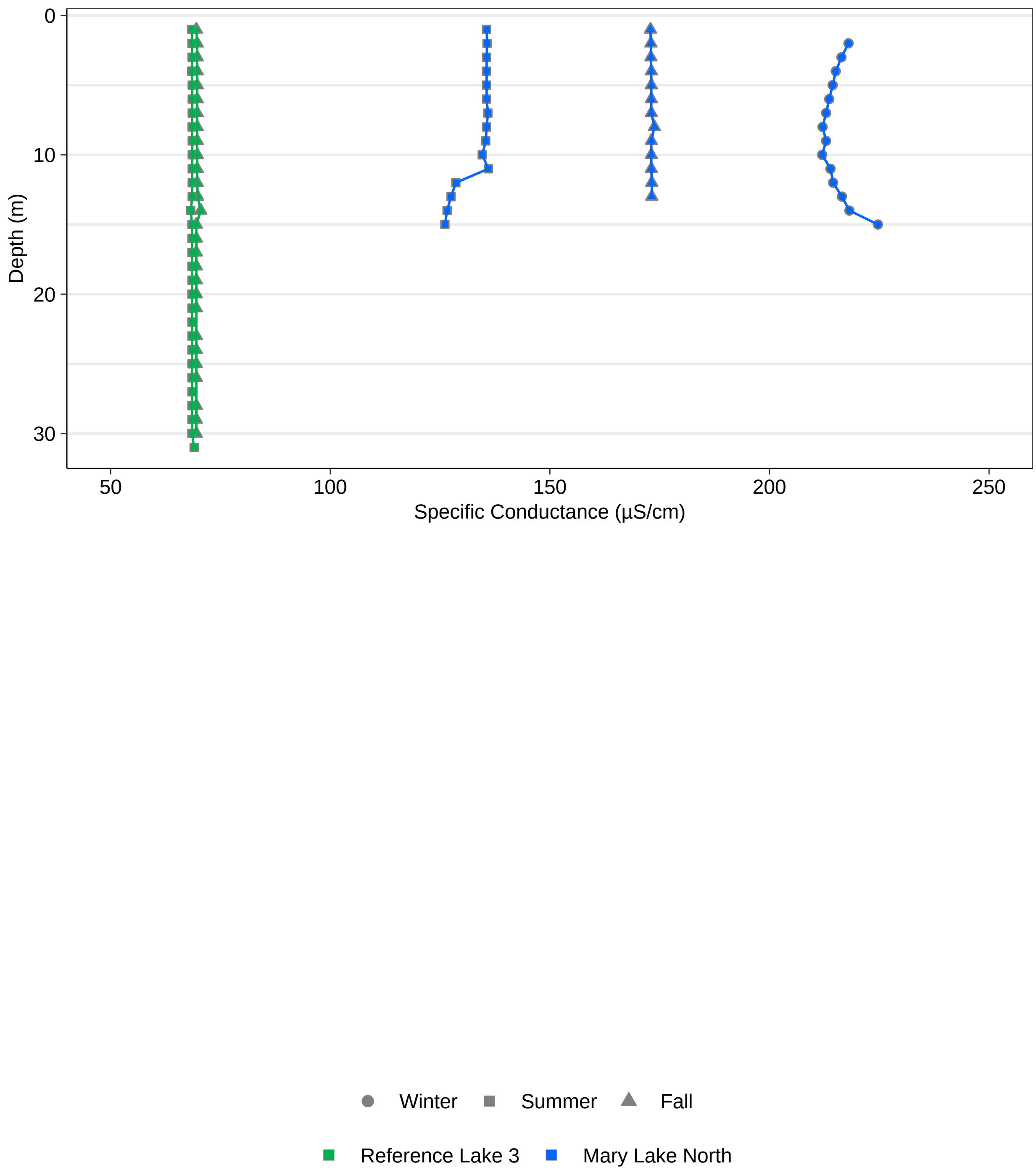

Figure 5.4: Average In Situ Water Quality with Depth from Surface at Mary Lake North Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

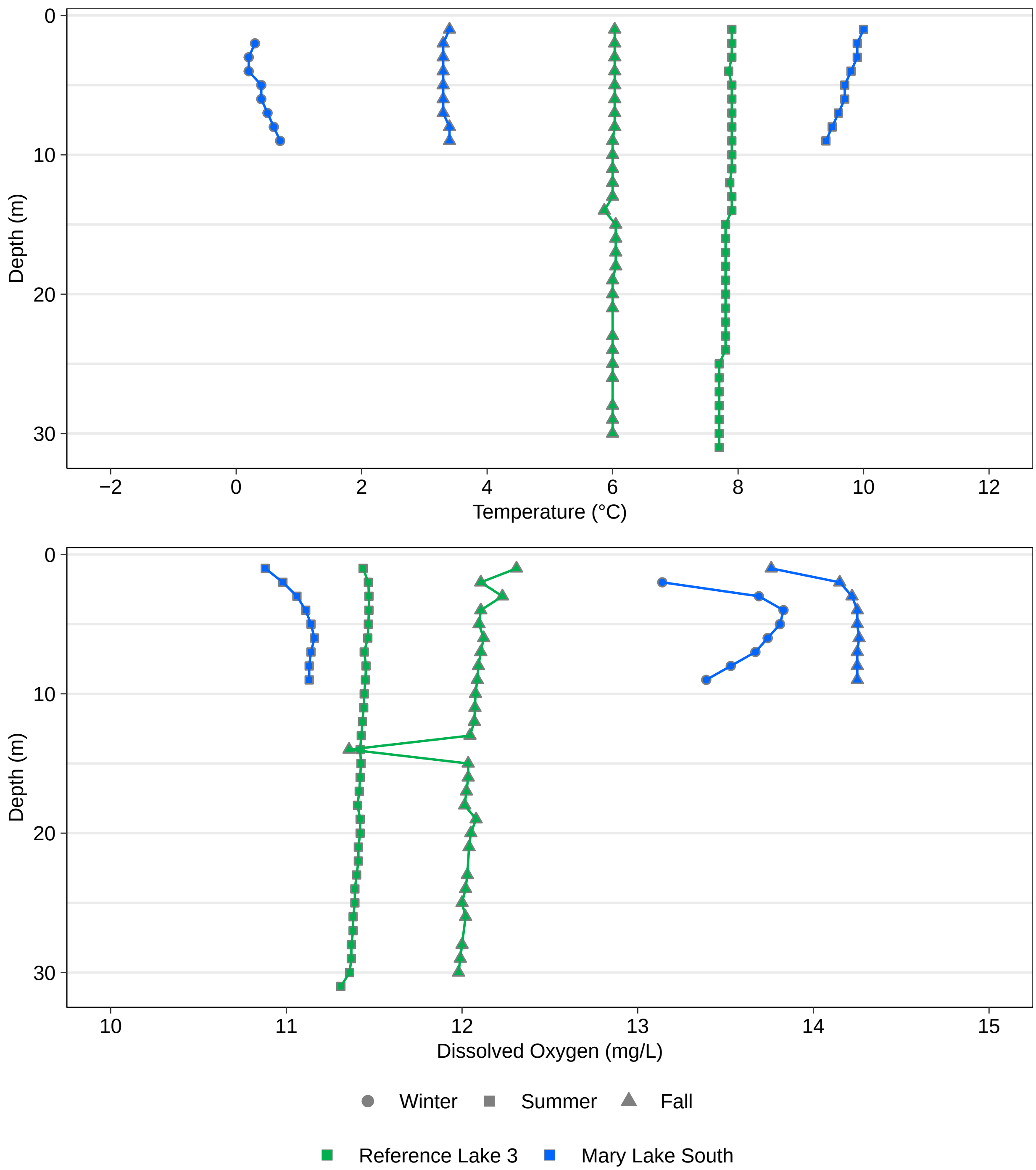

Figure 5.5: Average In Situ Water Quality with Depth from Surface at Mary Lake South Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

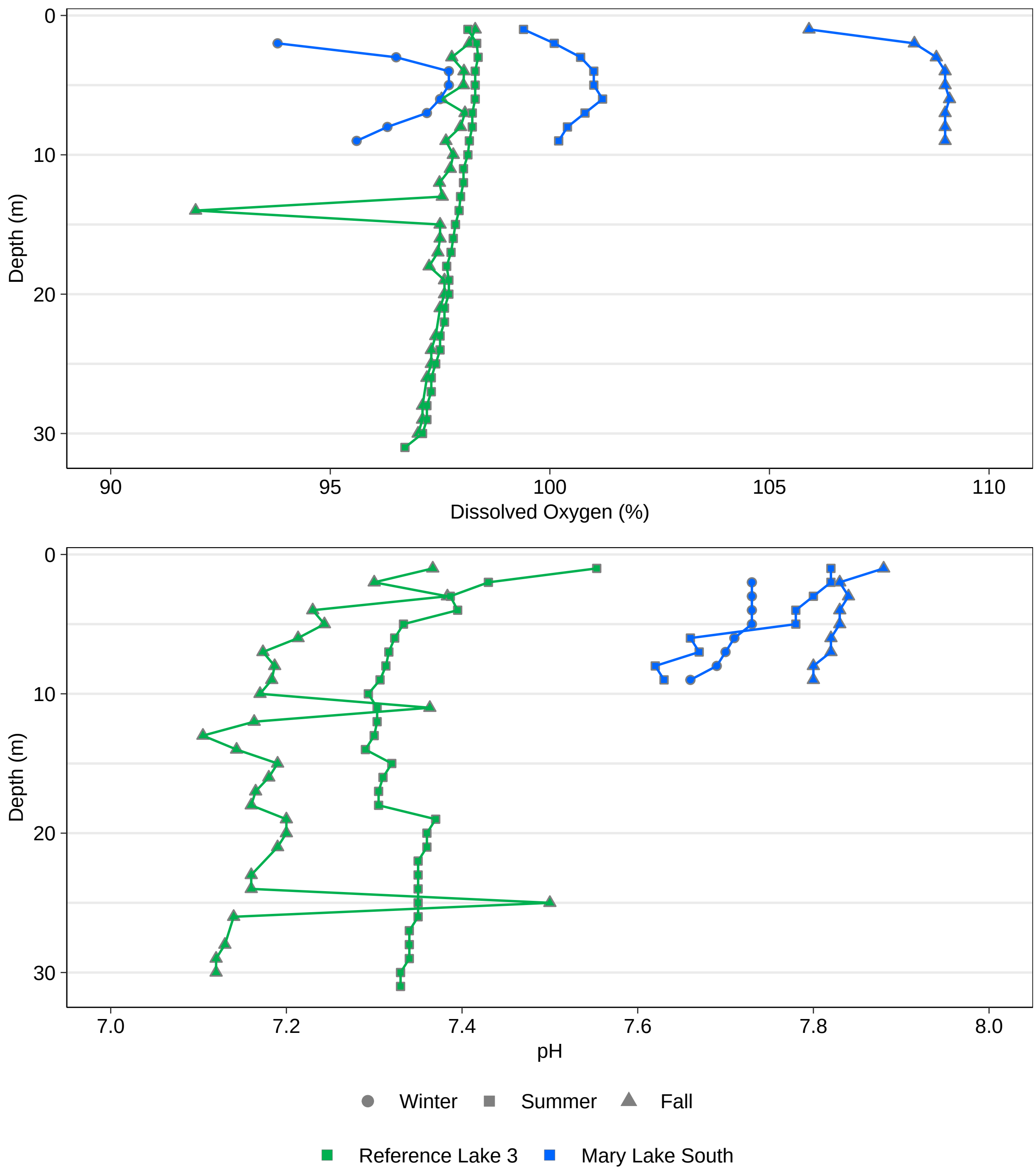

Figure 5.5: Average In Situ Water Quality with Depth from Surface at Mary Lake South Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

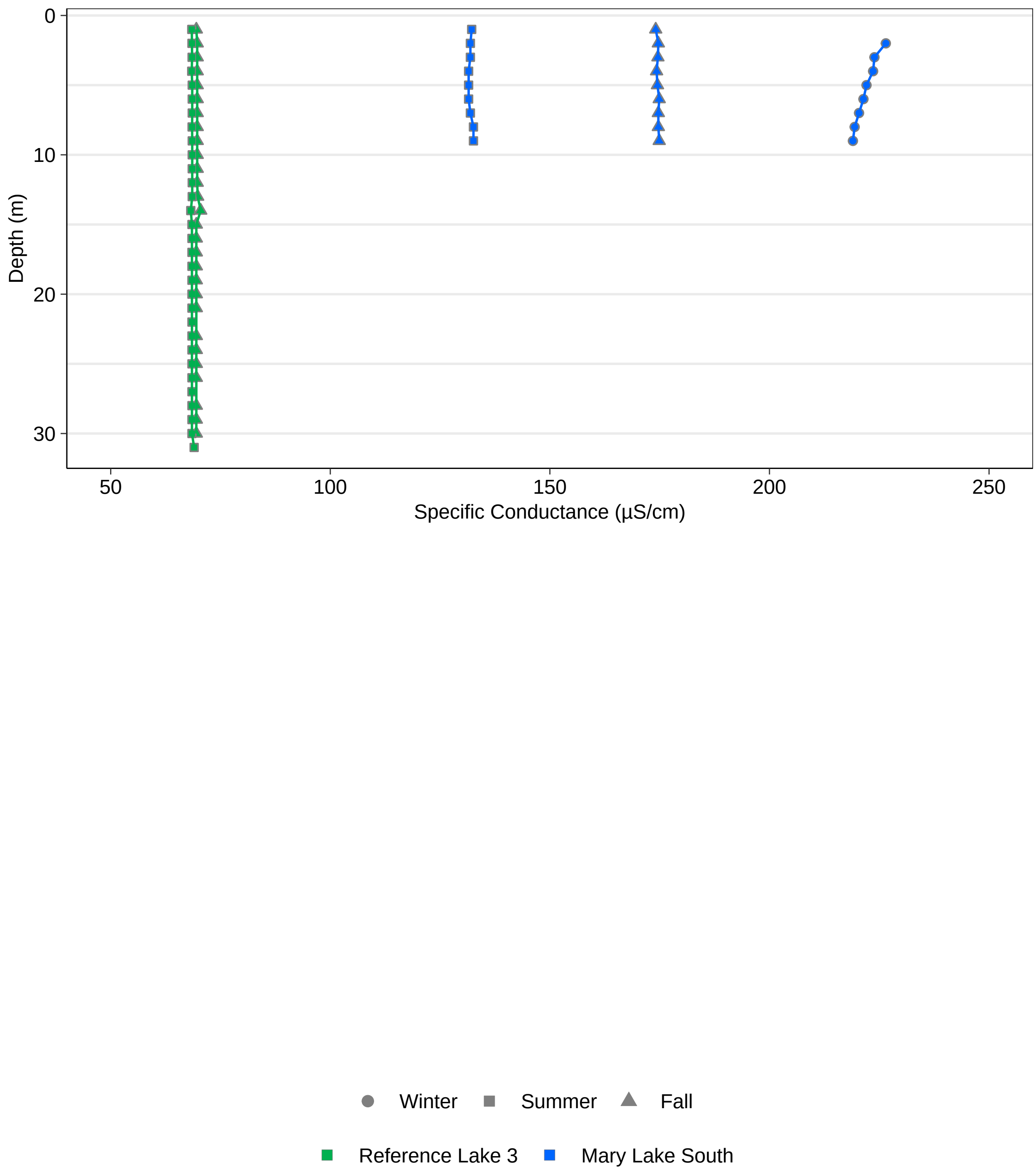

Figure 5.5: Average In Situ Water Quality with Depth from Surface at Mary Lake South Compared to Reference Lake 3 (REF3) during Spring, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

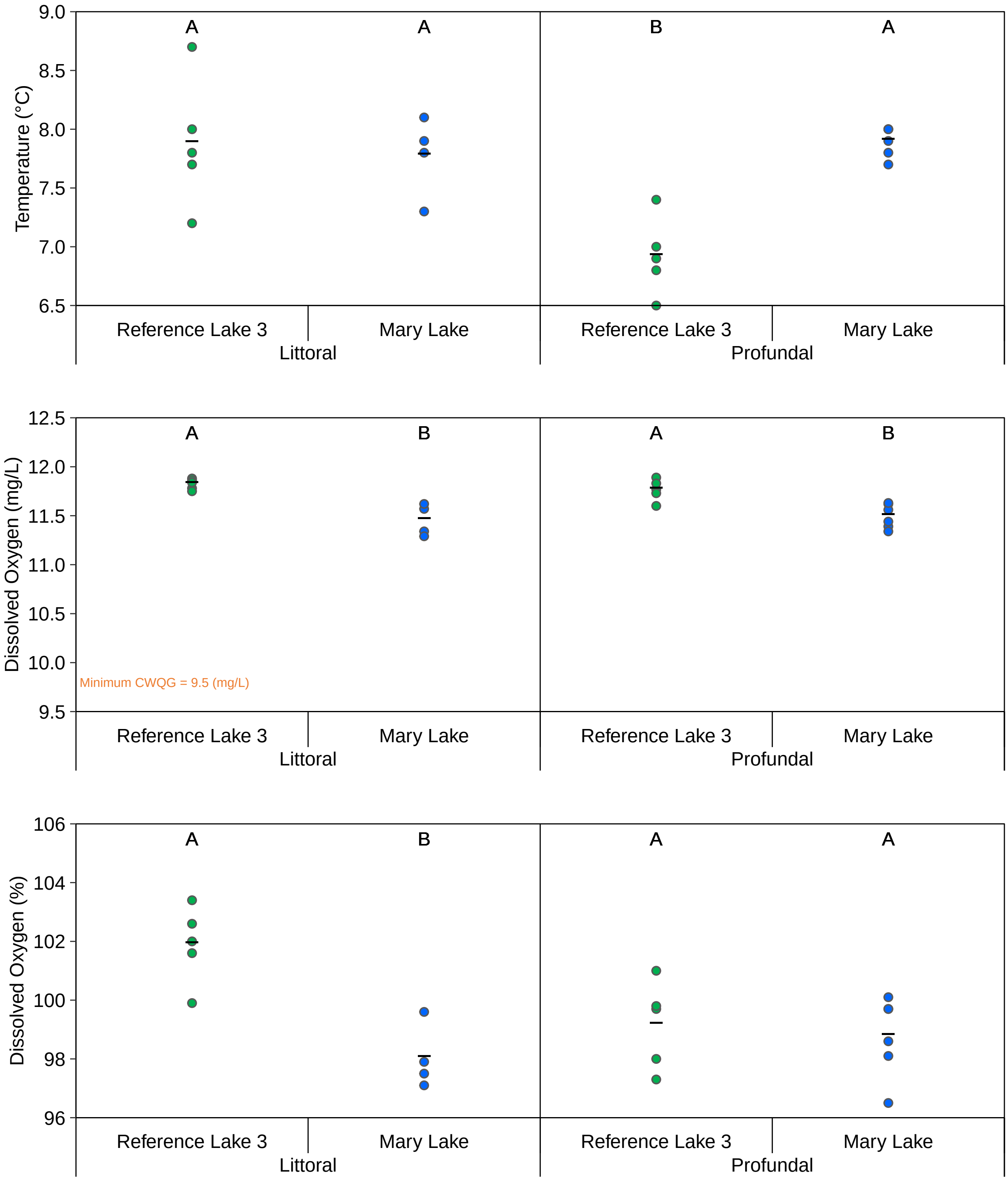

Figure 5.6: Comparison of *In Situ* Water Quality Measured at Mary Lake (North Basin [BL0-01]; and South Basin [BL0]) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

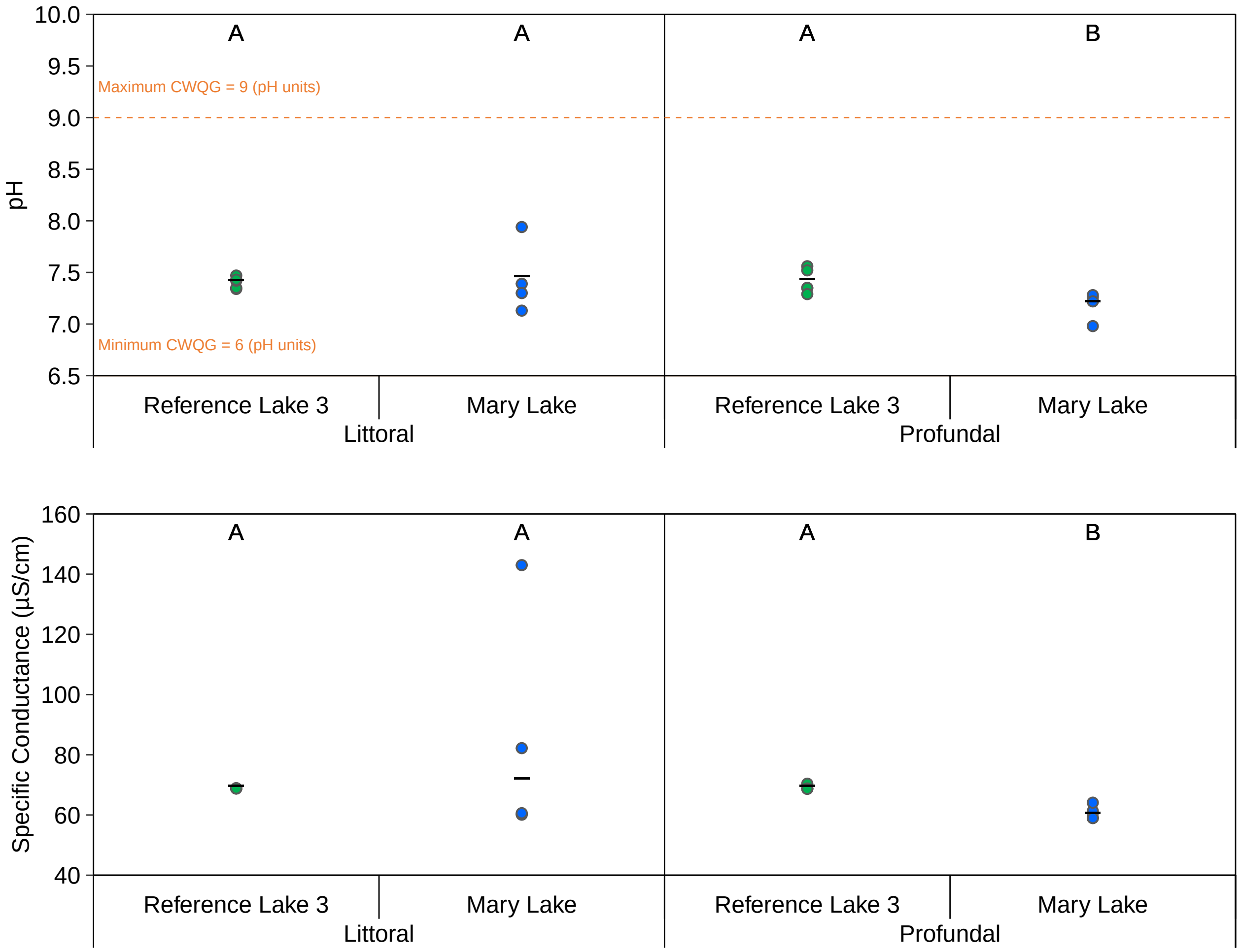

Figure 5.6: Comparison of *In Situ* Water Quality Measured at Mary Lake (North Basin [BL0-01]; and South Basin [BL0]) and Reference Lake (REF-03) Littoral and Profundal Benthic Invertebrate Community (BIC) Stations, Mary River Project CREMP, August 2024

Notes: Green represents reference stations and blue represents mine−exposed stations. Areas that share a letter do not differ significantly (p−value = 0.05). Bars indicate measures of central tendency of the statistical tests. Orange lines indicate Canadian Water Quality Guidelines (CWQG). Minimum dissolved oxygen WQG is for the protection of early life stages of cold−water biota, all other life stages are 6.5 mg/L.

Specific conductance was higher in winter, summer, and fall at the north basin compared to the south basin of Mary Lake (Figures 5.4 and 5.5, Appendix Figure C.28, Appendix Tables C.62 to C.64), likely reflecting natural differences in dominant inflow sources to Mary Lake (i.e., Tom River inflow to the north basin, and the Mary River inflow to the south basin). Mean specific conductance profiles were relatively uniform from the surface to bottom of the water column at the north and south basins of Mary Lake and Reference Lake 3 in summer and fall 2024 (Figures 5.4 and 5.5, Appendix Figure C.28, Appendix Tables C.62 to C.64). In winter in the south basin, mean specific conductance decreased with depth, while in the north basin it decreased until approximately 10 m then increased (Figures 5.4 and 5.5, Appendix Figure C.28, Appendix Tables C.62 to C.64). Specific conductance near the bottom of the water column at profundal stations of Mary Lake was significantly lower than at like-habitat stations at Reference Lake 3 during the August 2024 BIC sampling, while there was no difference at littoral stations (Figure 5.6, Appendix Tables C.65 and C.67). Water clarity, as determined by Secchi depth, was significantly lower at Mary Lake compared to Reference Lake 3 in August 2024 (Appendix Figure C.8, Appendix Table C.67), which may indicate an influence of suspended particles from inflow of the Mary River and the Tom River.

5.3.1.2 Water Chemistry

Mean water chemistry at the Mary Lake North and South Basins met all AEMP benchmarks and WQGs during spring, summer, and fall sampling events in 2024 (Table 5.3). In individual samples, only the concentration of chromium at the bottom at Station BL0-01-B in Mary Lake North (0.00114 mg/L) marginally exceeded the WQG of 0.001 mg/L (Appendix Table C.68), while all other parameter concentrations from individual samples were below respective AEMP benchmarks and WQGs (Appendix Tables C.68, C.70, C.72, C.73). Total and dissolved parameter concentrations that were slightly (3 to 5 times higher), moderately (5 to 10 times higher), or highly (≥ 10 times higher) elevated relative to reference or baseline concentrations are outlined in Appendix Tables C.69 and C.71. Comparisons of Mary Lake water chemistry to Reference Lake 3 in 2024 and to Mary Lake baseline indicated slightly to moderately elevated concentrations of some parameters, but none were consistently elevated across all sampling seasons, or within a single season, compared to both reference and baseline conditions. Additionally, visual assessment of temporal data did not indicate any consistent increasing or decreasing concentration patterns and mean concentrations have remained well below AEMP benchmarks and WQGs since commercial mine operations began in 2015, including in 2024 (Appendix Figure C.23). As a result, no adverse effects to biota are expected at Mary Lake and the mine-related influence on water quality is considered negligible.

Table 5.3: Mean Water Chemistry at Mary Lake North (BL0-01) and South (BL0) Basins Monitoring Stations[a], During Winter, Summer, and Fall Sampling Events, Mary River Project CREMP, 2024

Parameters		Units	Water Quality Guideline (WQG)[b,c]	AEMP Benchmark[d]	Reference Lake 3 (n = 3)		Mary Lake North Basin Stations (n = 3)			Mary Lake South Basin Stations (n = 7)		
					Summer	Fall	Winter	Summer	Fall	Winter	Summer	Fall
Conventionals	Conductivity (lab)	µmho/cm	-	-	72.5	72.0	222	139	178	85.3	62.0	81.9
	pH (lab)	pH	6.5 - 9.0	-	7.51	7.50	7.57	7.89	8.00	7.34	7.49	7.56
	Hardness (as $CaCO_3$)	mg/L	-	-	34.8	35.3	111	67.8	90.3	41.5	29.0	37.7
	Total Suspended Solids (TSS)	mg/L	-	-	<1	3.30	<1	<1	<1	<1	<1	1.00
	Total Dissolved Solids (TDS)	mg/L	-	-	51.5	41.2	114	65.5	88.3	47.6	39.3	39.9
	Turbidity	NTU	-	-	0.323	0.267	0.185	1.00	0.677	0.107	1.37	0.754
	Alkalinity (as $CaCO_3$)	mg/L	-	-	31.4	36.1	107	65.5	98.4	42.8	28.6	44.5
Nutrients and Organics	Total Ammonia	mg/L	-	0.855	0.00738	0.00837	<0.005	0.0108	0.00843	0.00518	0.00827	0.00529
	Nitrate	mg/L	3	3	<0.02	<0.02	0.0793	<0.02	<0.02	0.0644	0.0207	0.0211
	Nitrite	mg/L	0.06	0.06	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Total Kjeldahl Nitrogen (TKN)	mg/L	-	-	0.191	0.145	0.131	0.124	0.100	0.0630	0.0828	0.105
	Dissolved Organic Carbon	mg/L	-	-	3.62	3.44	2.62	2.54	2.61	1.59	2.02	2.00
	Total Organic Carbon	mg/L	-	-	3.01	3.51	2.50	2.16	2.58	1.54	1.74	2.26
	Total Phosphorus	mg/L	0.020[α]	-	0.00467	0.00262	0.00225	0.00387	0.00303	0.00241	0.00347	0.00387
	Phenols	mg/L	0.004[α]	-	<0.001	0.00152	<0.001	<0.001	0.00165	<0.001	<0.001	0.00144
Anions	Bromide (Br)	mg/L	-	-	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chloride (Cl)	mg/L	120	120	1.21	1.21	7.19	2.37	3.57	2.30	1.34	1.66
	Sulphate (SO_4)	mg/L	218[β]	218	2.72	2.63	3.26	1.30	1.78	1.84	1.09	1.40
Total Metals	Aluminum (Al)	mg/L	0.100	0.130	0.0158	0.00605	0.00422	0.0240	0.0182	0.00421	0.0458	0.0309
	Antimony (Sb)	mg/L	0.020[α]	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Arsenic (As)	mg/L	0.005	0.005	0.000117	<0.0001	0.000107	0.000102	<0.0001	<0.0001	0.000102	<0.0001
	Barium (Ba)	mg/L	1[β]	-	0.00614	0.00598	0.0106	0.00701	0.00784	0.00474	0.00352	0.00456
	Beryllium (Be)	mg/L	0.011[α]	-	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002	<0.00002
	Bismuth (Bi)	mg/L	-	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Boron (B)	mg/L	1.5	-	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Cadmium (Cd)	mg/L	0.00012	0.00006	<0.000005	<0.000005	<0.000005	0.00000500	<0.000005	<0.000005	<0.000005	<0.000005
	Calcium (Ca)	mg/L	-	-	6.49	6.40	21.2	13.2	16.6	7.93	5.40	7.63
	Chromium (Cr)	mg/L	0.001	0.003	<0.0005	<0.0005	<0.0005	<0.0005	0.000607	0.000515	0.000515	<0.0005
	Cobalt (Co)	mg/L	0.0009[α]	0.004	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Copper (Cu)	mg/L	0.002	0.0024	0.000848	0.000823	0.00112	0.000960	0.000928	0.000569	0.000601	0.000709
	Iron (Fe)	mg/L	0.30	0.300	0.0337	0.0112	0.0237	0.0375	0.0297	<0.01	0.0484	0.0346
	Lead (Pb)	mg/L	0.001	0.001	0.0000528	<0.00005	<0.00005	0.0000533	<0.00005	<0.00005	0.0000541	0.0000502
	Lithium (Li)	mg/L	-	-	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Magnesium (Mg)	mg/L	-	-	4.26	4.48	13.6	8.55	11.6	4.81	3.53	4.88
	Manganese (Mn)	mg/L	0.935[β]	-	0.00136	0.000602	0.0124	0.00373	0.00171	0.000371	0.00167	0.00145
	Mercury (Hg)	mg/L	0.000026	-	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	<0.000005	0.00000576
	Molybdenum (Mo)	mg/L	0.073	-	0.000139	0.000144	0.000312	0.000213	0.000247	0.000169	0.000123	0.000166
	Nickel (Ni)	mg/L	0.025	0.025	<0.0005	<0.0005	0.000605	0.000502	0.000525	<0.0005	0.000513	0.000505
	Potassium (K)	mg/L	-	-	0.888	0.831	1.13	0.912	0.922	0.586	0.491	0.612
	Selenium (Se)	mg/L	0.001	-	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
	Silicon (Si)	mg/L	-	-	0.487	0.425	1.34	0.837	0.933	0.454	0.484	0.469
	Silver (Ag)	mg/L	0.00025	0.0001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Sodium (Na)	mg/L	-	-	0.875	0.843	3.67	1.89	2.66	1.27	0.850	1.20
	Strontium (Sr)	mg/L	-	-	0.00783	0.00754	0.0157	0.00972	0.0118	0.00716	0.00465	0.00669
	Thallium (Tl)	mg/L	0.0008	0.0008	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
	Tin (Sn)	mg/L	-	-	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	Titanium (Ti)	mg/L	-	-	0.000947	0.000308	<0.0003	0.00116	0.000857	<0.0003	0.00227	0.00134
	Uranium (U)	mg/L	0.015	-	0.000273	0.000260	0.00260	0.00123	0.00220	0.000739	0.000375	0.000780
	Vanadium (V)	mg/L	0.006[α]	0.006	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
	Zinc (Zn)	mg/L	0.02[α]	0.030	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003	<0.003

Indicates parameter concentration above applicable Water Quality Guideline.

BOLD Indicates parameter concentration above the AEMP benchmark.

Notes: AEMP: Aquatic Effects Monitoring Plan. "-" indicates no applicable WQG or AEMP benchmark.

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

[b] Canadian Water Quality Guideline (CCME 2024) except those indicated by α (Ontario Provincial Water Quality Objective [PWQO]; OMOE 1994) and β (British Columbia Water Quality Guideline [BCWQG]; BCMOE 2024). See Table 2.2 for information regarding WQG criteria.

[c] A conservative hardness value of 75 mg/L was used for guideline calculations dependent on hardness (i.e., sulphate, beryllium, cadmium, copper, lead, manganese, and nickel).

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

5.3.2 Sediment Quality

Most surficial sediments (i.e., top 2 cm) collected at the Mary Lake coring stations in 2024 were primarily composed of reddish brown to brown silt and sand (Figure 5.7; Appendix Table D.21). These samples also contained very little TOC (i.e., <1.5%; Figure 5.7; Appendix Table D.22). Sediment samples collected using a Petite Ponar, specifically to support BIC data interpretation, contained a higher proportion of sand and less silt relative to the core samples (Appendix Table D.4). The sediment samples collected using a Petite Ponar were also characterized as having lower TOC content relative to the core samples and no detectable hydrogen sulphide odour (Appendix Tables D.4 and D.20).

In 2024, the particle size distributions for littoral and profundal sediments at Mary Lake differed significantly from the distributions at Reference Lake 3, and field crews noted the presence of a gray clay layer in the samples from Reference Lake 3 that was absent from the Mary Lake samples (Appendix Tables D.1, D.21, and D.22). Littoral sediments from Mary Lake contained significantly more silt and less sand than the reference lake (Appendix Table D.22). Profundal sediments from Mary Lake contained significantly less sand and more clay relative to Reference Lake 3 in 2024 (Appendix Table D.22). Additionally, the average proportion of TOC in profundal sediments from Mary Lake was significantly lower than at the reference lake (Appendix Table D.22).

Metal concentrations in sediments from the Mary Lake south basin showed no clear spatial gradients or patterns with progression from the Mary River inflow to the lake outlet (Appendix Table D.23)[53]. Mean iron (littoral and profundal) and manganese (profundal) concentrations in sediments from Mary Lake were above SQG in 2024, but not AEMP benchmarks (Table 5.4; Appendix Table D.42). Similar to other sediment monitoring areas for the CREMP (e.g., Camp Lake), the results for iron and manganese reflect high natural (i.e., geogenic) concentrations of iron and manganese in the region (Table 5.4; see also Appendix Tables D.2 and D.24). Arsenic and chromium concentrations were above AEMP benchmarks at one profundal station each (BL0-16 and BL0-08, respectively) in 2024 and chromium concentrations at four stations (BL0-10, BL0-09, BL0-08, and BL0-06) were above the SQG (the mean was still below the SQG; Table 5.4; Appendix Table D.23). As noted in Sheardown Lake SE, sediment samples with high (relative to SQG) iron and/or manganese concentrations also had concentrations of arsenic and/or chromium above AEMP benchmarks and/or SQGs (Appendix Table D.23). This is likely due to sorption characteristics of the sediment as

[53] Spatially, the order of sediment quality stations going from closest to Mary River to the lake outlet was BL0-12, BL0-10, BL0-09, BL0-08, and BL0-06 (Figure 2.3). All stations, except BL0-06, were profundal.

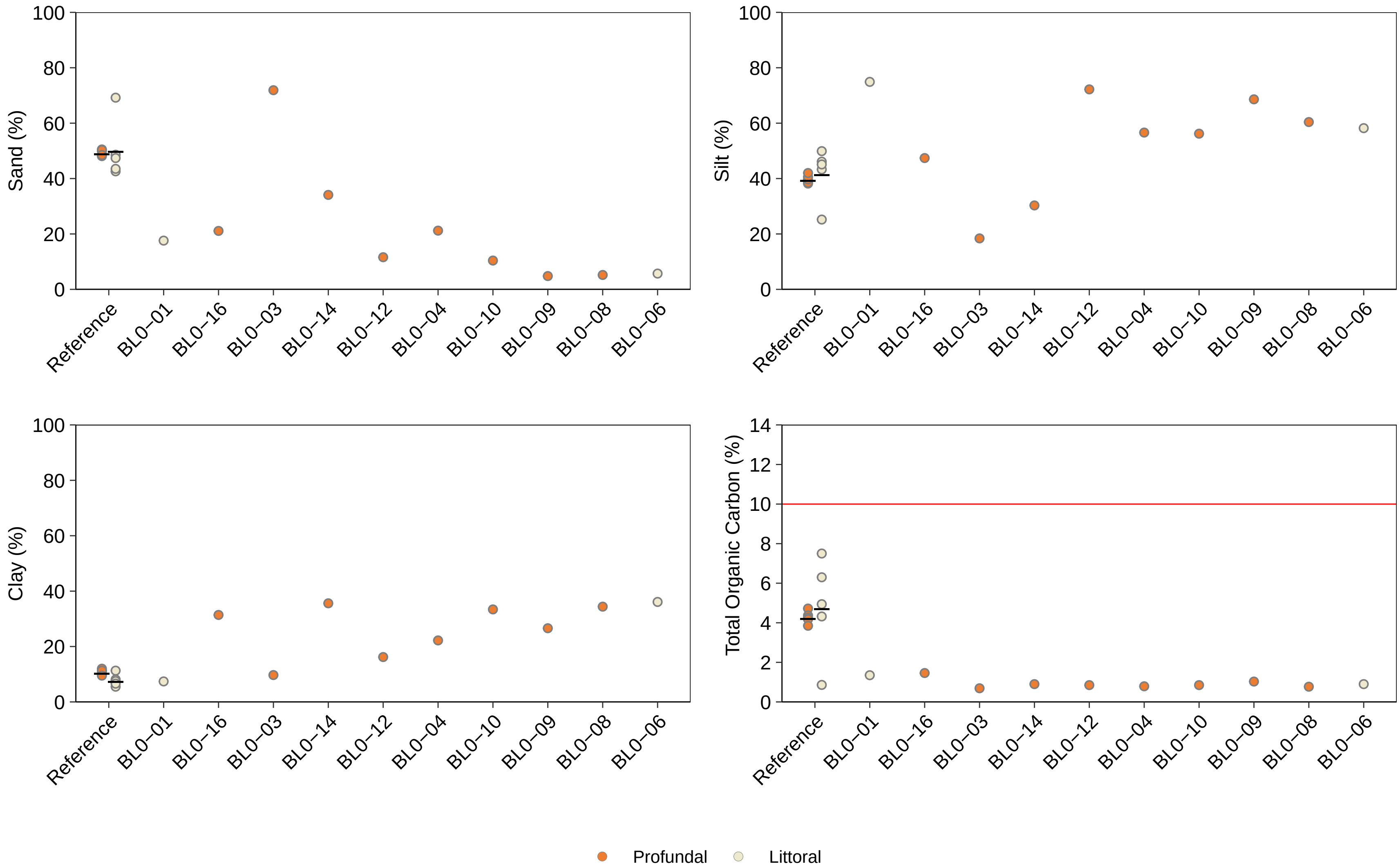

Figure 5.7: Sediment Particle Size and Total Organic Carbon (TOC) Content Comparisons among Sediment Cores taken from Mary Lake (BL0) Sediment Monitoring Stations and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Black bars indicate average of reference samples. Red line indicates AEMP Benchmark.

Table 5.4: Sediment Total Organic Carbon (TOC) and Metal Concentrations at Mary Lake North (BL0-01) and South (BL0) Basins and Reference Lake 3 (REF-03) Sediment Monitoring Stations, Mary River Project CREMP, August 2024

Parameter		Units	SQG[a]	AEMP Benchmark[b]	Littoral		Profundal	
					Reference Lake (n = 5)	Mary Lake (n = 2)	Reference Lake (n = 5)	Mary Lake (n = 8)
					Average ± SD	Average ± SD	Average ± SD	Average ± SD
TOC		%	10[α]	-	4.78 ± 2.52	1.12 ± 0.318	4.28 ± 0.315	0.918 ± 0.241
Metals	Aluminum (Al)	mg/kg	-	-	16,560 ± 3,306	20,950 ± 7,283	23,060 ± 1,363	22,229 ± 5,573
	Antimony (Sb)	mg/kg	-	-	<0.1 ± -	<0.1 ± -	<0.1 ± -	<0.1 ± -
	Arsenic (As)	mg/kg	17	5.9	5.02 ± 1.55	4.33 ± 0.877	5.07 ± 0.449	3.86 ± 1.98
	Barium (Ba)	mg/kg	-	-	115 ± 34.7	91.3 ± 12.2	142 ± 20.5	91.6 ± 24.6
	Beryllium (Be)	mg/kg	-	-	0.646 ± 0.147	1.10 ± 0.354	0.884 ± 0.0586	1.19 ± 0.296
	Bismuth (Bi)	mg/kg	-	-	<0.2 ± -	0.220 ± -	<0.2 ± -	0.255 ± 0.0153
	Boron (B)	mg/kg	-	-	13.3 ± 2.05	36.2 ± 12.8	16.7 ± 0.879	37.8 ± 9.83
	Cadmium (Cd)	mg/kg	3.5	1.5	0.146 ± 0.0497	0.121 ± 0.0297	0.166 ± 0.0166	0.144 ± 0.0390
	Calcium (Ca)	mg/kg	-	-	4,716 ± 728	6,855 ± 2,779	5,426 ± 237	4,421 ± 1,135
	Chromium (Cr)	mg/kg	90	98	55.1 ± 12.3	81.0 ± 17.1	76.0 ± 4.65	84.8 ± 21.2
	Cobalt (Co)	mg/kg	-	-	11.5 ± 2.84	15.8 ± 2.12	17.4 ± 1.70	16.0 ± 3.80
	Copper (Cu)	mg/kg	197	50	***67.5*** ± 21.3	32.0 ± 3.96	***95.1*** ± 8.03	32.2 ± 7.86
	Iron (Fe)	mg/kg	40,000[α]	52,400	***58,760*** ± 25,999	41,500 ± 4,667	49,820 ± 3,295	41,175 ± 10,555
	Lead (Pb)	mg/kg	91.3	35	13.7 ± 1.78	20.8 ± 6.51	18.5 ± 1.01	22.1 ± 5.43
	Lithium (Li)	mg/kg	-	-	25.6 ± 5.12	42.5 ± 13.6	36.2 ± 2.68	42.9 ± 11.5
	Magnesium (Mg)	mg/kg	-	-	11,308 ± 2,124	16,350 ± 1,626	15,780 ± 841	15,432 ± 3,807
	Manganese (Mn)	mg/kg	1,100[α,β]	4,370	862 ± 611	1,090 ± 70.7	2,246 ± 2,318	1,558 ± 915
	Mercury (Hg)	mg/kg	0.486	0.170	0.0470 ± 0.0233	0.0359 ± 0.0134	0.0702 ± 0.0129	0.0548 ± 0.0189
	Molybdenum (Mo)	mg/kg	-	-	4.63 ± 1.94	0.775 ± 0.134	2.83 ± 0.501	1.11 ± 0.387
	Nickel (Ni)	mg/kg	75[α,β]	72	39.2 ± 8.63	57.8 ± 2.05	52.2 ± 3.77	59.1 ± 14.3
	Phosphorus (P)	mg/kg	2,000[α]	1,580	1,344 ± 713	886 ± 189	999 ± 72	857 ± 314
	Potassium (K)	mg/kg	-	-	4,118 ± 630	5,700 ± 2,461	5,600 ± 317	6,016 ± 1,534
	Selenium (Se)	mg/kg	-	-	0.740 ± 0.278	0.220 ± 0.0283	0.826 ± 0.133	0.269 ± 0.0483
	Silver (Ag)	mg/kg	-	-	0.146 ± 0.0462	0.130 ± -	0.238 ± 0.0192	0.162 ± 0.0166
	Sodium (Na)	mg/kg	-	-	311 ± 48.8	360 ± 121	431 ± 20.9	405 ± 127
	Strontium (Sr)	mg/kg	-	-	11.1 ± 1.22	15.0 ± 2.33	13.3 ± 0.458	14.9 ± 3.75
	Sulphur (S)	mg/kg	-	-	1,620 ± 403	<1,000 ± -	1,360 ± 114	<1,000 ± -
	Thallium (Tl)	mg/kg	-	-	0.423 ± 0.145	0.507 ± 0.211	0.748 ± 0.0562	0.539 ± 0.134
	Tin (Sn)	mg/kg	-	-	<2 ± -	2.00 ± -	<2 ± -	<2 ± -
	Titanium (Ti)	mg/kg	-	-	958 ± 159	1,515 ± 544	1,164 ± 37.1	1,521 ± 388
	Uranium (U)	mg/kg	-	-	15.3 ± 5.91	7.00 ± 3.50	25.1 ± 2.33	9.02 ± 2.22
	Vanadium (V)	mg/kg	-	-	51.2 ± 9.67	62.5 ± 15.9	67.7 ± 3.92	62.8 ± 15.4
	Zinc (Zn)	mg/kg	315	135	72.1 ± 14.9	72 ± 23.7	95.2 ± 6.65	74.4 ± 18.2
	Zirconium (Zr)	mg/kg	-	-	4.26 ± 1.70	19.3 ± 10.3	3.92 ± 0.455	21.0 ± 6.56

[shaded] Indicates parameter concentration above SQG.

BOLD Indicates parameter concentration above the AEMP Benchmark.

Notes: TOC = total organic carbon. SQG = sediment quality guideline. n = number of samples. SD = standard deviation. "-" = data not available.

[a] Canadian SQG for the protection of aquatic life probable effect level (PEL; CCME 2024) except α = Ontario Provincial Sediment Quality Guideline (PSQG) severe effect level (SEL; OMOE 1993) and β = British Columbia Working SQG PEL (BCMOE 2024).

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

iron and manganese (oxy)hydroxides are known to sorb metal cations (e.g., chromium) and anions (e.g., arsenic; Bendell-Young et al. 1992).

Mean concentrations of metals in sediment at Mary Lake were comparable to those at Reference Lake 3 in 2024, except for zirconium, which had concentrations that were approximately four and five times higher at littoral and profundal habitats, respectively, within Mary Lake (Appendix Table D.24). Similar to Sheardown Lake SE, concentrations of zirconium in Mary Lake sediments been elevated by 3 to 6 times compared to Reference Lake 3 since 2020, when zirconium was first included in sediment chemistry analyses (Minnow 2021b, 2022, 2023a, and 2024a). As described in Section 4.5.2, zirconium was not a parameter that was identified in the FEIS as having the potential to have mine-related effects on water or sediment quality (Baffinland 2012) and a potential source of zirconium to Mary Lake may be from naturally occurring zircon mineral weathered from sedimentary and alluvial deposits in the Mary River catchment. Because analysis of zirconium as a sediment chemistry parameter was initiated as part of the CREMP only in 2020, there are no baseline or early mine operation period data available for comparison to results from 2024 and insufficient data for statistical analysis of temporal trends. Concentrations will continue to be monitored annually for ongoing evaluation of potential mine-related influence.

Concentrations of metals, in sediment, except boron, at littoral and profundal stations of Mary Lake in 2024 have not changed substantially from those observed during the baseline period. Boron concentrations were approximately 20 to 40-times higher in 2024 relative to baseline (2005 to 2013; Appendix Figure D.1, Appendix Table D.24).[54] On average, metal concentrations in sediment from Mary Lake in 2023 were within the range of those observed during mine operations from 2015 to 2023, and there was no evidence of concentrations increasing over time for any metal (Figure 5.8).

Overall, there were no apparent mine-related changes in metal concentrations in sediments[55] at Mary Lake since the initiation of commercial mine operations in 2015 though ongoing monitoring of zirconium concentrations is recommended.

[54] Boron concentrations in sediments from 2015 to 2024 were considerably higher (i.e., 10- to 70-times) than those reported during both the baseline and 2014 studies at all mine-exposed lakes. The lack of any distinct gradient in the magnitude of the elevation in boron concentrations among stations within each lake and among study lakes suggested that the stark contrast in boron concentrations between recent data and data collected prior to 2015 was likely due to laboratory-based analytical differences (i.e., probable under-recovery of boron in baseline and 2014). The analytical laboratory used for the baseline study differed from the current laboratory.

[55] Except for boron, which was considered to be due to analytical differences.

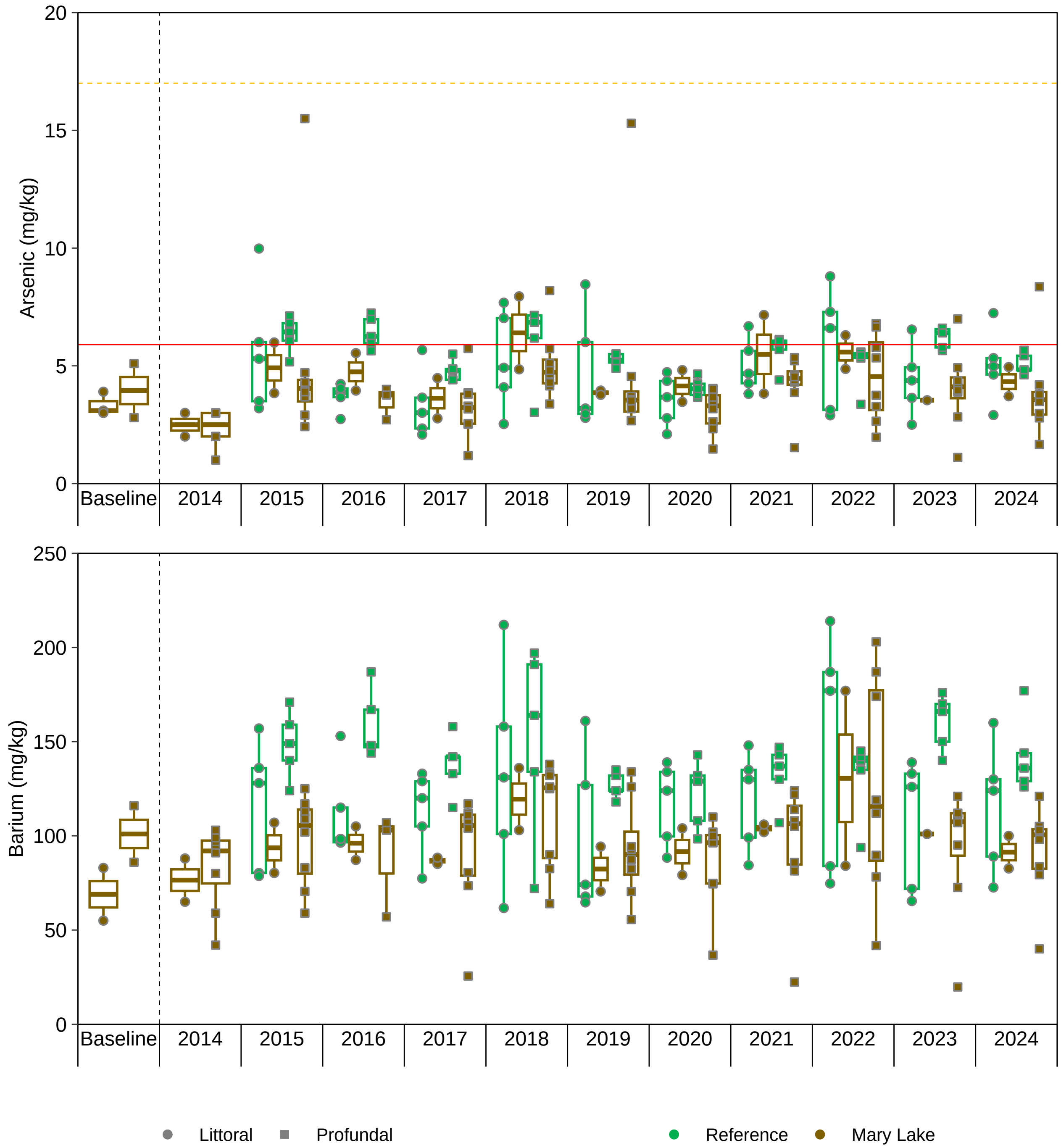

Figure 5.8: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Mary Lake (BL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

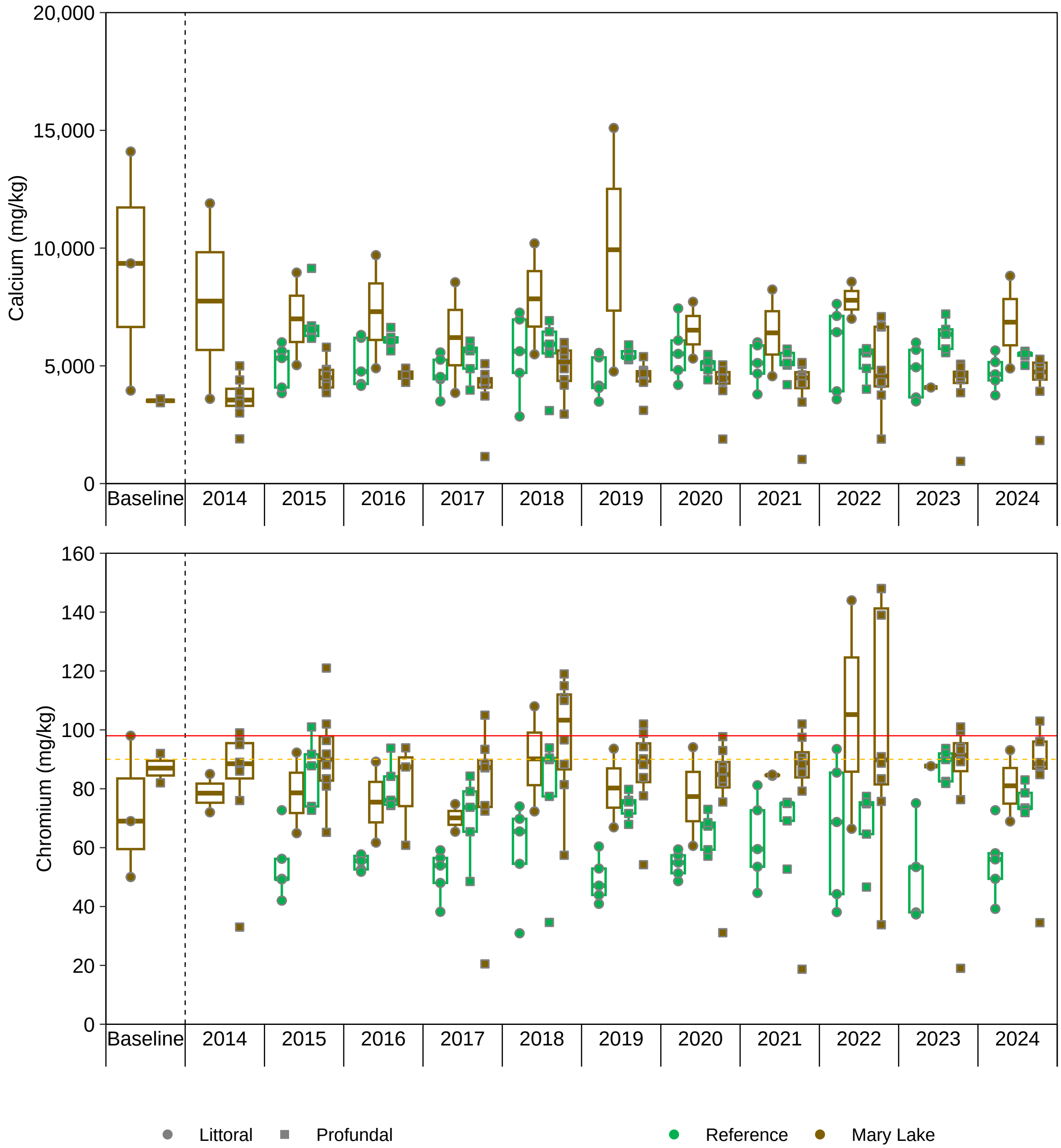

Figure 5.8: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Mary Lake (BL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

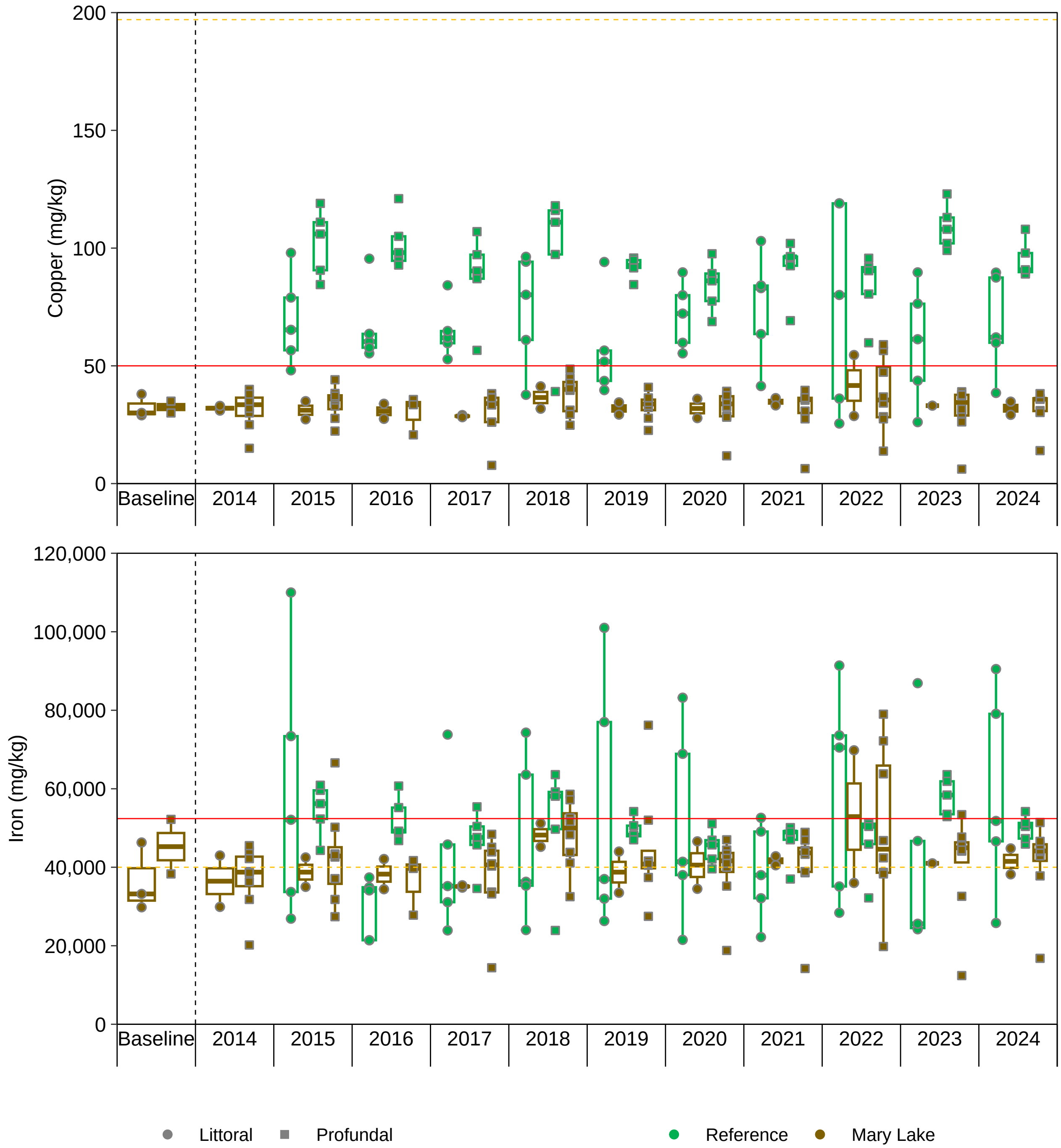

Figure 5.8: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Mary Lake (BL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

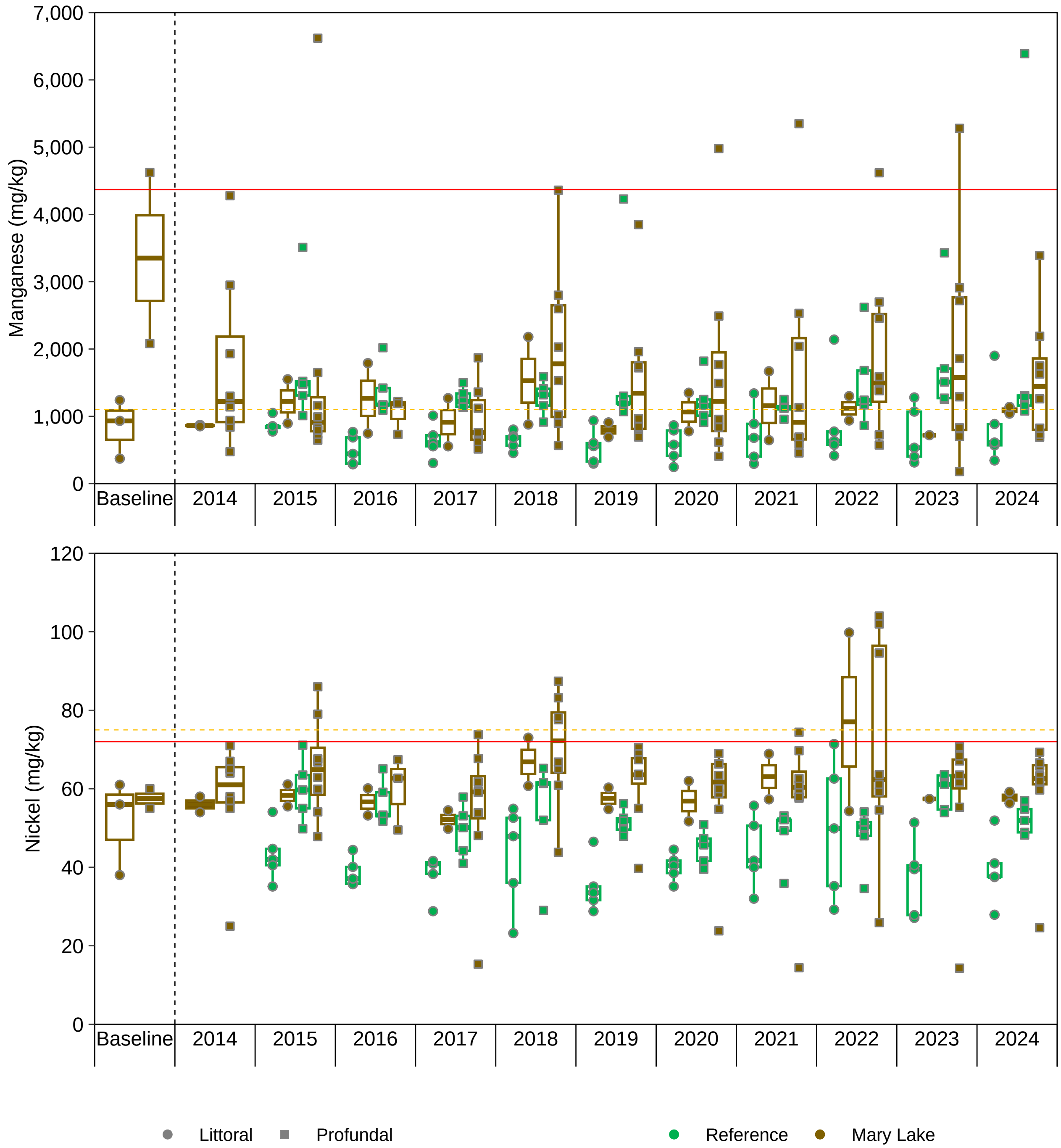

Figure 5.8: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Mary Lake (BL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

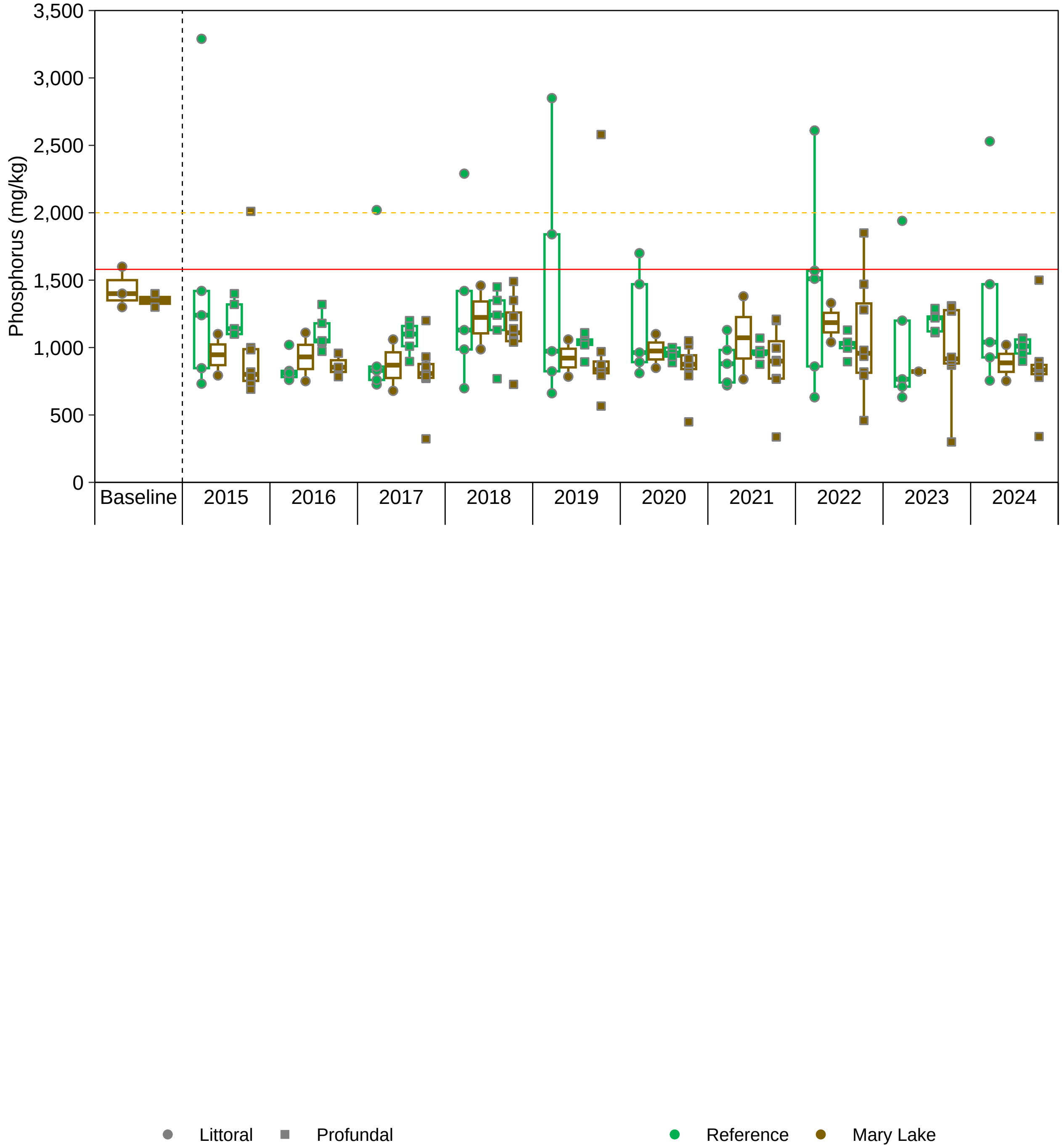

Figure 5.8: Temporal Comparison of Metal Concentrations in Sediment at Littoral and Profundal Stations of Mary Lake (BL0) and Reference Lake 3 (REF-03) for Mine Baseline (2005 to 2013), Construction (2014), and Operations (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark; orange dashed line indicates Canadian Sediment Quality Guideline Probable Effect Level or Ontario Provincial Sediment Quality Guideline Severe Effect Level. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum (excluding potential outliers).

5.3.3 Phytoplankton

Chlorophyll-a concentrations at Mary Lake showed no spatial gradients with distance from either the Tom River inlet or the Mary River inlet during any of the winter, summer, or fall sampling events in 2024, although fall concentrations throughout the Mary Lake south basin tended to be higher than in the north basin, based on visual comparisons (Figure 5.9). In 2024, chlorophyll-a concentrations were lowest in winter on both Mary Lake basins, whereas summer concentrations were highest in the north basin and fall concentrations were highest in the south basin (Figure 5.9, Appendix Tables E.4 and E.15). When compared to Reference Lake 3, chlorophyll-a concentrations in the Mary Lake north and south basins were significantly lower in the summer of 2024 (Appendix Table E.6). However, in the fall of 2024, concentrations in the Mary Lake south basin were significantly higher than those in Reference Lake 3, and chlorophyll-a concentrations in the Mary Lake north basin were significantly lower (Appendix Table E.7). Although fall chlorophyll-a concentrations in the Mary Lake south basin have not typically been higher than both Reference Lake 3 and Mary Lake north basin concentrations over the mine operational period, the concentrations in 2024 were within the ranges that have been observed at Mary Lake and Reference Lake 3 since 2015 (Figure 5.9).

Despite seasonal variations, chlorophyll-a concentrations in both the north and south basins of Mary Lake remained well below the AEMP benchmark of 3.7 µg/L during all seasonal sampling events in 2024 (Figure 5.9). Chlorophyll-a concentrations <4.5 ug/L (Appendix Table E.15) and total phosphorus concentrations <10 µg/L (Table 5.3, Appendix Table C.68 andC.72) indicated an oligotrophic status for both basins of Mary Lake (Wetzel 2001, CCME 2024b; see Section 3.3.3 for additional trophic status classification details).

Although significant differences in chlorophyll-a concentrations were observed at Mary Lake north and south basins across years of mine construction and operation in all seasons, concentrations in 2024 generally fell within the seasonal ranges observed from 2014 to 2023 (Figure 5.10, Appendix Tables E.16 and E.17). There were no consistent directional changes (i.e., increasing or decreasing) in chlorophyll-a concentrations for any of the winter, summer, or fall seasons over time (Figure 5.10, Appendix Tables E.16 and E.17). Moreover, there have been no consistent directional patterns in annual average chlorophyll-a concentrations at either basin since mine construction was completed (Figure 5.10, Appendix Tables E.16 and E.17). No chlorophyll-a data were available for Mary Lake north or south basins from the baseline period (2005 to 2013), which precludes comparisons to conditions prior to the mine’s construction.

Overall, chlorophyll-a concentrations in Mary Lake exhibited no consistent directional temporal patterns in any season, have generally remained consistent relative to those observed at Reference Lake 3 since 2015, and remained well below the AEMP benchmark in 2024.

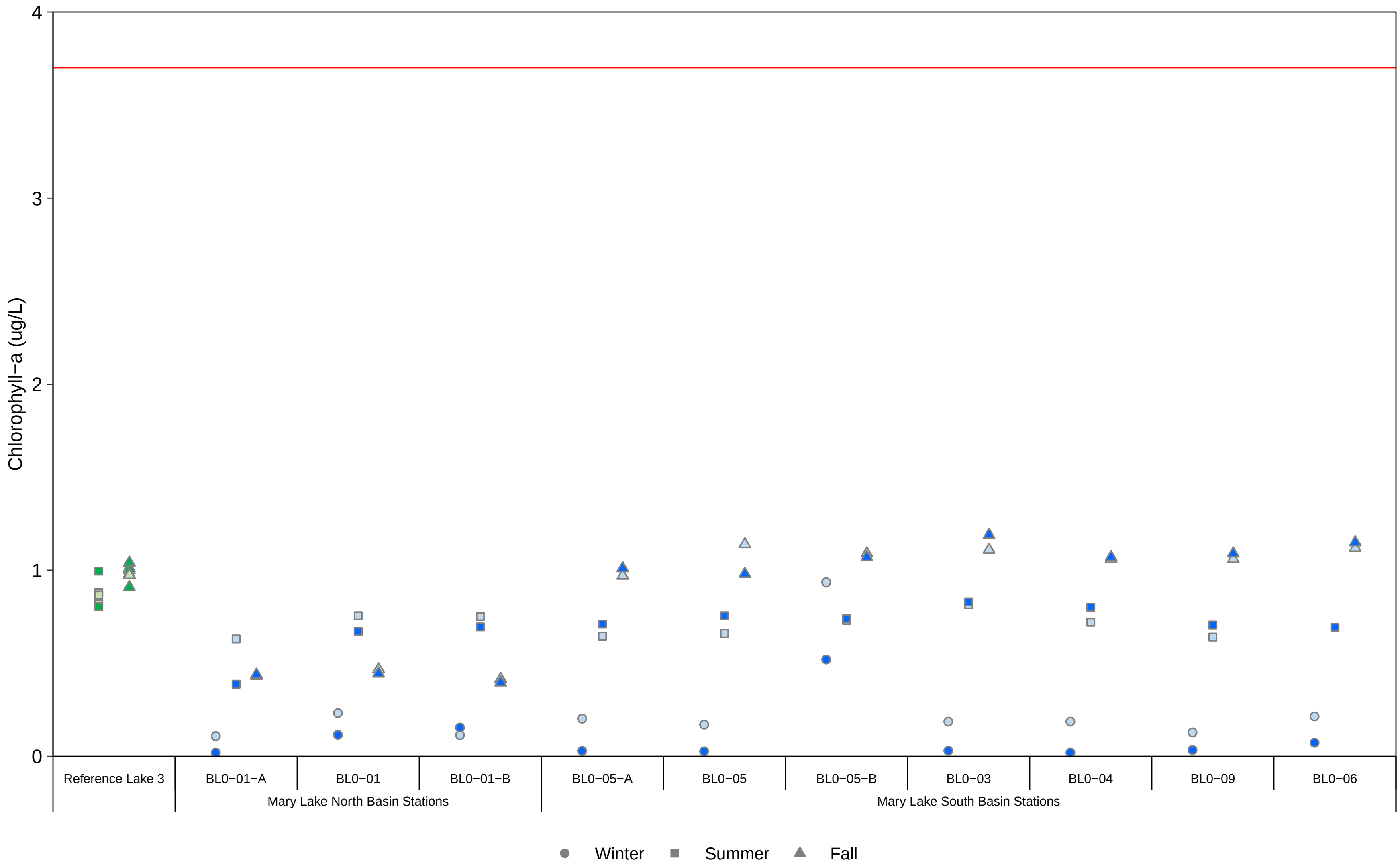

Figure 5.9: Chlorophyll−a Concentrations at Mary Lake (BL0) Phytoplankton Monitoring Stations, Mary River Project CREMP, 2024

Notes: Concentrations reported below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL. Red line indicates AEMP Benchmark. Lighter shade of colour indicates surface sample, darker shade indicates bottom sample. Reference areas are shown in green and mine−exposed areas are shown in blue. Mary Lake North Basin Stations are presented in order of proximity to the lake inlet from Camp Lake (left to right). In Mary Lake South Basin, Station BL0-05-A is proximal to the lake inlet from the Mary River, Station BL0-03 is proximal to the inlet from the North Basin, and Station BL0-06 is proximal to the lake outlet to the Mary River.

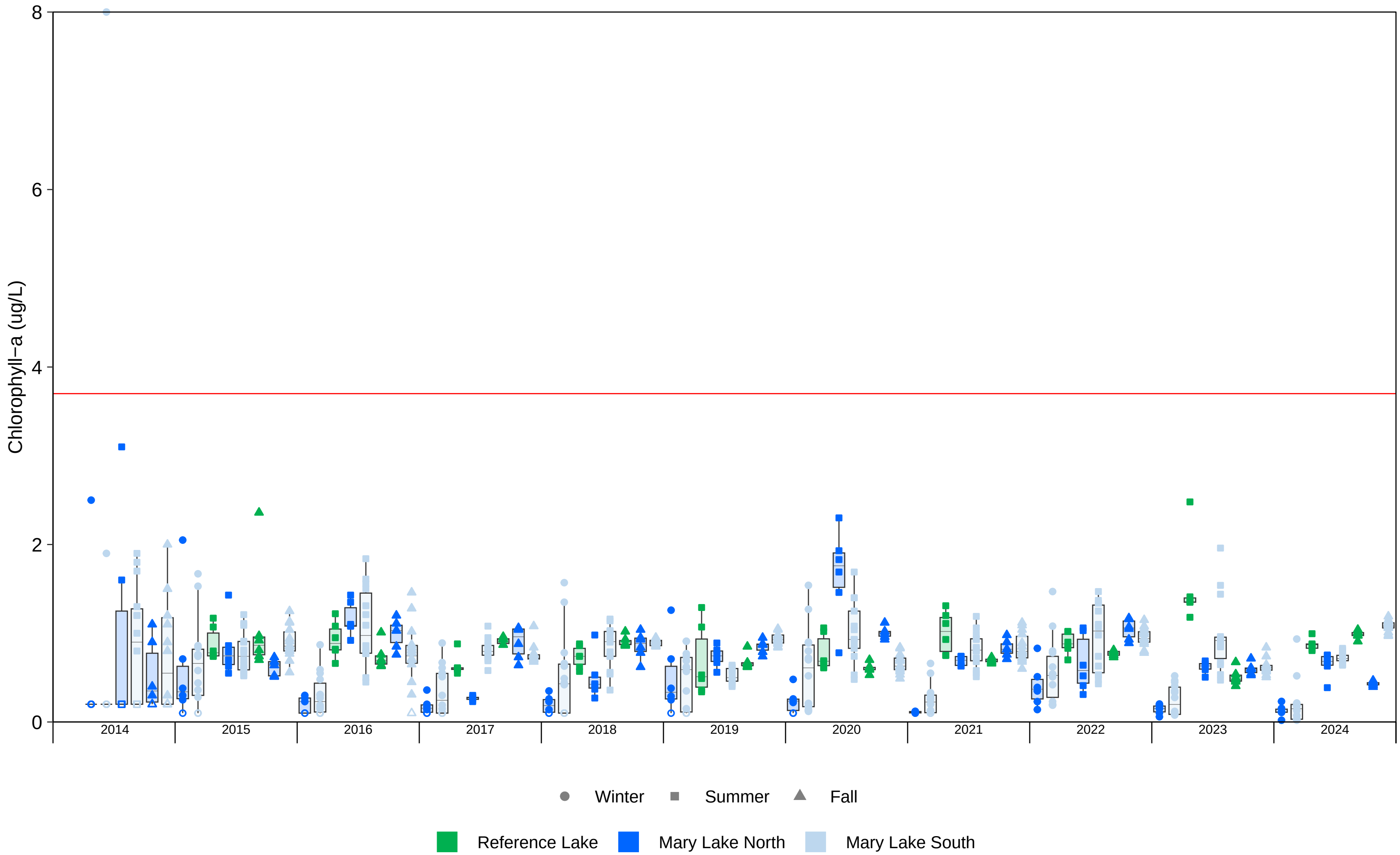

Figure 5.10: Temporal Comparison of Chlorophyll−a Concentrations Among Seasons between Mary Lake (BL0) and Reference Lake 3 (REF-03) for Construction (2014) and Operational (2015 to 2024) Periods, Mary River Project CREMP, 2024

Notes: Concentrations below the laboratory reporting limit (LRL) are plotted as open symbols at the LRL and the open symbol represents one or more values reported below the LRL. Red line indicates AEMP Benchmark. Black bars indicate average of samples. Boxplot lines show the 25th percentile, median, and 75th percentile with the boxplots whiskers showing the minimum and maximum. Potential outliers, defined as values outside three times the interquartile range, are excluded from the whiskers.

These results indicate no adverse mine-related effects on phytoplankton productivity at Mary Lake in 2024.

5.3.4 Benthic Invertebrate Community

In 2024, there were few significant differences in BIC endpoints between the littoral habitats of Mary Lake and those of Reference Lake 3, other than relative proportions of *Ostracoda*, *Chironomidae*, filterers, clingers, and burrowers (Table 5.5). Specifically, relative proportions of *Ostracoda* were significantly lower (MCT = 1.03%) compared to Reference Lake 3 (MCT = 39.5%), with the absolute difference falling outside the CES_{BIC} of ± 2 SD_{REF} (i.e., the difference was ecologically meaningful; Table 5.5). Conversely, the relative proportion of *Chironomidae* was significantly higher in littoral habitats of Mary Lake (MCT = 96.7%) relative to Reference Lake 3 (MCT = 52.6%), with this difference also being ecologically meaningful (Table 5.5). These results suggest that conditions in Mary Lake are more favourable for Chironomids. Since the start of mine operations in 2015, proportions of *Ostracoda* and *Chironomidae* in littoral habitats of Mary Lake have not changed in an ecologically meaningful way relative to baseline (Appendix Table F.53) and qualitative assessment of temporal patterns indicates that the proportion of *Ostracoda* has been consistently lower and the proportion of *Chironomidae* has been consistently higher in littoral habitats of Camp Lake than the reference lake (Appendix Figure F.14). Therefore, significant differences in the proportions of these organisms between like-habitats in Mary Lake and Reference Lake 3 in 2024 are not indicative of mine-related effects. Finally, the Bray-Curtis Index was reflective of the noted structural differences in the littoral BIC between the Mary Lake and Reference Lake 3 (Appendix Table F.19).

In 2024, significant differences in BIC endpoints between profundal habitats in Mary Lake and Reference Lake 3 were limited to lower proportions of *Ostracoda* and higher proportions of burrower taxa in Mary Lake relative to reference (Table 5.6). These differences between Mary Lake and Reference Lake 3 were considered ecologically meaningful based on MODs outside the CES_{BIC} of ± 2 SD_{REF} (Table 5.6). Since the start of mine operations in 2015, proportions of *Ostracoda* in profundal habitats of Mary Lake have not changed significantly relative baseline (Appendix Table F.54) and qualitative assessment of temporal patterns indicates that the proportion of *Ostracoda* in profundal habitats of Mary Lake has varied relative to the reference lake and there have been no consistent directional (i.e., increasing or decreasing) changes (Appendix Figure F.15).Similar to littoral habitats, the Bray-Curtis Index was also reflective of these structural differences in the BIC between the two lakes (Appendix Table F.19).

Aqueous concentrations of TSS, DOC, and TOC can influence FFG and HPG composition based on diet and habitat preferences (Merritt et al. 2008); however, no differences were observed in

Table 5.5: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Littoral Habitats in Mary Lake (BL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake Littoral Habitat	MCT (n = 4)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	log10	NO	0.990	0.015	Reference Lake 3	1,049	901	403	215	982	2,514
						Mary Lake	2,454	3,474	1,737	43.1	1,089	7,595
Richness (Number of Taxa)	tequal	log10	NO	0.694	-0.30	Reference Lake 3	8.80	3.56	1.59	5.00	8.00	13.0
						Mary Lake	8.00	4.24	2.12	4.00	7.00	14.0
Simpson's Evenness (E)	tequal	none	NO	0.389	0.86	Reference Lake 3	0.759	0.0621	0.0278	0.669	0.755	0.840
						Mary Lake	0.812	0.111	0.0553	0.695	0.796	0.960
Shannon's Diversity	tequal	log10	NO	0.981	0.019	Reference Lake 3	2.01	0.243	0.109	1.72	2.01	2.28
						Mary Lake	2.02	0.347	0.173	1.77	1.89	2.53
Hydracarina (%)	tequal	log10(x+1)	NO	0.202	-0.72	Reference Lake 3	2.49	2.88	1.29	0	2.33	7.02
						Mary Lake	0.404	0.549	0.275	0	0.227	1.16
Ostracoda (%)	tequal	none	YES	0.002	-2.4	Reference Lake 3	39.5	16.1	7.20	16.0	40.3	60.5
						Mary Lake	1.03	1.21	0.607	0	0.907	2.33
Chironomidae (%)	tequal	none	YES	<0.001	3.1	Reference Lake 3	52.6	14.4	6.46	30.7	56.6	68.0
						Mary Lake	96.7	4.40	2.20	90.7	98.1	100
Metal Sensitive Chironomidae (%)	tequal	log10(x+1)	NO	0.249	-0.71	Reference Lake 3	22.1	17.5	7.81	0	17.3	41.3
						Mary Lake	9.46	9.26	4.63	0	8.91	20.0
Collector Gatherers (%)	tequal	none	NO	0.806	-0.15	Reference Lake 3	74.9	18.1	8.08	56.4	77.0	100
						Mary Lake	72.3	10.9	5.46	60.0	73.2	82.7
Filterers (%)	tequal	none	YES	0.096	-1.0	Reference Lake 3	21.7	17.6	7.85	0	17.3	41.3
						Mary Lake	3.92	5.69	2.84	0	1.81	12.1
Shredders (%)	M-W	rank	NO	0.687	-[a]	Reference Lake 3	0.344	0.578	0.259	0	0	1.33
						Mary Lake	1.21	2.04	1.02	0	0.301	4.25
Clingers (%)	tequal	none	YES	0.097	-1.0	Reference Lake 3	24.0	17.9	8.01	0	19.0	43.6
						Mary Lake	5.38	7.73	3.87	0	2.39	16.8
Sprawlers (%)	tequal	log10	NO	0.695	-0.37	Reference Lake 3	66.2	16.9	7.56	43.2	74.5	84.0
						Mary Lake	62.9	27.6	13.8	33.1	63.3	91.9
Burrowers (%)	tequal	none	YES	0.090	3.4	Reference Lake 3	9.82	6.40	2.86	2.30	8.16	16.9
						Mary Lake	31.7	24.3	12.1	6.98	28.3	63.2

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

[a] Contrast MODs could not be calculated because the MAD = 0.

Table 5.6: Statistical Comparisons of Benthic Invertebrate Community Endpoints for Profundal Habitats in Mary Lake (BL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Endpoint	Statistical Test Results					Summary Statistics						
	Statistical Test	Data Transform-ation	Significant Difference Between Areas?	P-value	MOD	Study Lake Littoral Habitat	MCT (n = 6)	Standard Deviation	Standard Error	Minimum	Median	Maximum
Density (Individuals/m^2)	tequal	log10	NO	0.366	3.2	Reference Lake 3	202	33.7	15.1	146	207	233
						Mary Lake	713	955	390	60.3	495	2,618
Richness (Number of Taxa)	tequal	log10	NO	0.198	1.5	Reference Lake 3	4.40	1.14	0.510	3.00	4.00	6.00
						Mary Lake	7.17	3.97	1.62	3.00	6.00	13.0
Simpson's Evenness (E)	tequal	none	NO	0.437	0.56	Reference Lake 3	0.582	0.169	0.0754	0.457	0.508	0.867
						Mary Lake	0.676	0.206	0.0841	0.298	0.688	0.869
Shannon's Diversity	tequal	none	NO	0.224	1.4	Reference Lake 3	1.28	0.318	0.142	0.834	1.27	1.71
						Mary Lake	1.72	0.692	0.283	0.865	1.67	2.78
Hydracarina (%)	tequal	none	NO	0.921	-0.048	Reference Lake 3	4.09	5.94	2.66	0	0	13.0
						Mary Lake	3.80	3.32	1.35	0	2.82	8.33
Ostracoda (%)	M-W	rank	YES	0.082	-4.3	Reference Lake 3	8.37	2.08	0.929	5.88	8.33	11.5
						Mary Lake	5.33	8.43	3.44	0	2.44	22.0
Chironomidae (%)	tequal	none	NO	0.751	0.18	Reference Lake 3	85.2	7.71	3.45	76.9	85.2	94.1
						Mary Lake	86.6	6.55	2.67	76.3	88.2	93.2
Metal Sensitive Chironomidae (%)	M-W	rank	NO	0.855	0.14	Reference Lake 3	9.98	11.3	5.05	0	7.41	29.4
						Mary Lake	11.7	15.1	6.17	0	7.95	41.7
Collector Gatherers (%)	M-W	rank	NO	0.429	-0.42	Reference Lake 3	85.2	16.2	7.24	57.6	88.2	100
						Mary Lake	87.2	4.94	2.02	83.3	85.5	96.6
Filterers (%)	M-W	rank	NO	0.751	-[a]	Reference Lake 3	6.70	12.8	5.72	0	0	29.4
						Mary Lake	1.49	2.36	0.962	0	0	5.26
Shredders (%)	M-W	rank	NO	0.892	-[a]	Reference Lake 3	0.833	1.86	0.833	0	0	4.17
						Mary Lake	0.459	1.12	0.459	0	0	2.75
Clingers (%)	M-W	rank	NO	0.454	-[a]	Reference Lake 3	9.96	18.4	8.23	0	0	42.4
						Mary Lake	10.4	14.4	5.89	0	5.01	38.2
Sprawlers (%)	tequal	none	NO	0.303	-0.85	Reference Lake 3	86.2	16.7	7.46	57.6	88.9	100
						Mary Lake	72.0	24.4	9.97	35.5	79.3	94.9
Burrowers (%)	tequal	log10(x+1)	YES	0.066	2.7	Reference Lake 3	3.88	4.71	2.11	0	3.70	11.5
						Mary Lake	17.5	14.4	5.88	3.39	14.4	41.7

P-value < 0.1.

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

Notes: MOD = Magnitude of Difference = $(MCT_{Exp} - MCT_{Ref})/SD_{Ref.}$ MCT = Measure of Central Tendency. SD = Standard Deviation. MAD = Median Absolute Deviation. MCT and SD reported as median and MAD for rank-transformed data, as transformed means and SD for log transformed data, and as untransformed means and SD for untransformed data.

[a] Contrast MODs could not be calculated because the MAD = 0.

these parameters compared to reference or baseline conditions in 2024 (Appendix Table C.69). Additionally, the absence of a mine-related influence on water and sediment chemistry suggests that the structural differences in the BIC of Mary Lake and Reference Lake 3 are not attributed to the mine.

No significant, ecologically meaningful differences in BIC endpoints were observed in littoral or profundal habitats of Mary Lake over time based on comparisons among mine operational years (i.e., 2015 to 2024) and the baseline year (2007; Appendix Tables F.53 and F.54, Appendix Figures F.15 and F.16). Although relative proportions of *Ostracoda* in littoral habitats were higher in 2018 and 2020 through 2023 relative to 2007, as noted above, the differences were not ecologically meaningful, and results for most mine operational years (i.e., 2015 to 2019 and 2021 to 2023) were comparable (Appendix Table F.53, Appendix Figure F.15). Similarly, invertebrate density in profundal habitats was comparable among most mine operational years (i.e., 2015 to 2023), and differences relative to baseline (2007) were not ecologically meaningful, based on the CES_{BIC} (Appendix Table F.54, Appendix Figure F.16).

Overall, few differences in BIC endpoints were observed between Mary Lake and Reference Lake 3 in 2024, as well as among mine operational and baseline years in Mary Lake, indicating no adverse mine-related impacts on benthic invertebrates. This is consistent with the lack of substantial changes in water and sediment quality at Mary Lake compared to reference and baseline conditions.

5.3.5 Fish Population

5.3.5.1 Fish Community

Arctic charr (*Salvelinus alpinus*) was the only fish species captured at Mary Lake in 2024 (Table 5.7). Ninespine stickleback (*Pungitius pungitius*) have been captured in nearshore electrofishing surveys in Mary Lake in all previous CREMP monitoring years at very low densities (CPUE ranged from 0.01 to 0.75 fish per electrofishing minute; Minnow 2017, 2018, 2019, 2020, 2021b, 2023, and 2024a). Given previously low densities, and because at all other mine-exposed study lakes, as well as at Reference Lake 3, there have been monitoring years where no ninespine stickleback were captured despite their confirmed presence in the lake (see Sections 3.3.5.1, 4.4.5.1, and 4.5.5.1), 2024 nearshore fish captures do not suggest a change in fish community richness in Mary Lake.

The CPUE for arctic charr in electrofishing and in gill netting surveys was higher at Mary Lake compared to Reference Lake 3 in 2024, suggesting greater fish density at Mary Lake (Table 5.7, Appendix Tables G.1 and G.3). In 2024, electrofishing CPUE at Mary Lake was within the range observed over the previous nine years of mine operation (2015 to 2023) and baseline studies

Table 5.7: Fish Catch and Community Summary from Backpack Electrofishing and Gill Netting Conducted at Mary Lake (BL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Lake	Method [a]		Arctic Charr	Ninespine Stickleback	Total by Method	Total No. of Species
Reference Lake 3	Electrofishing	No. Caught	105	15	120	2
		CPUE	1.12	0.16	1.28	
	Gill netting	No. Caught	84	-	84	
		CPUE	3.30	-	3.30	
Mary Lake	Electrofishing	No. Caught	111	0	111	2
		CPUE	1.69	0.00	1.69	
	Gill netting	No. Caught	126	-	126	
		CPUE	7.91	-	7.91	

Note: "-" indicates not applicable as ninespine stickleback are not captured by gill netting.

[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.

(2007 and 2008; Figure 5.11). Gill netting CPUE for arctic charr in 2024 was also within the range of earlier mine operation years but greater than during baseline (Figure 5.11). While gill netting CPUE at Mary Lake in 2023 was more than two-fold higher than any previous sampling year, it decreased in 2024 to a level that was more consistent with earlier mine operational years. Fish density based on CPUE in electrofishing and in gill netting surveys at Mary Lake has frequently been greater than reference conditions (since sampling was initiated in Reference Lake 3 in 2015; Figure 5.11)[56] and greater than baseline. Although occasional differences in BIC community structure have been observed between Mary Lake and Reference Lake 3 (Section 5.3.4), no consistent temporal patterns in chlorophyll-a concentrations (i.e., phytoplankton density) or ecologically significant differences in BIC endpoints have been observed in Mary Lake that explain observed patterns in fish density (Sections 5.3.3 and 5.3.4). Factors, such as changes in spatial ecology, other than lake productivity, may have influenced CPUE in Mary Lake over time. As described in Section 3.3.5.1, sampling related influences (e.g., seasonal timing and access to sampling locations) or naturally influenced environmental factors (e.g., water temperature) that affect fish movement behaviour, spatial ecology, and metabolic demands, have the potential to influence fish catch rates, particularly in 'passive' gill net surveys. Mary Lake is also substantially larger than the other study lakes, including the designated reference lake (Appendix Table B.1), which may influence the home range size and movement patterns of resident fish. In larger lakes, fish typically have access to a broader array of habitats and greater spatial extent, often resulting in more extensive movements and larger home ranges (Woolnough et al. 2009). As a result, catch rates may be more variable compared to smaller, more spatially constrained systems (Power et al. 2000).

In 2024, CPUE in both electrofishing and gill results fell within the range of previously observed CPUE during the mine operations period and were greater than during the baseline period and Reference Lake 3 in 2024. Therefore, mine-related changes in fish densities at Mary Lake are not indicated.

5.3.5.2 Fish Health Assessment

Nearshore Arctic Charr

In August 2024, a total of 100 and 102 arctic charr were sampled for assessment of fish health from nearshore habitats at Mary Lake and Reference Lake 3, respectively (Appendix Tables G.4

[56] Baseline fish community data (2005 to 2013) were not collected at Reference Lake 3, precluding comparisons of mine-exposed and reference conditions prior to the construction of the mine.

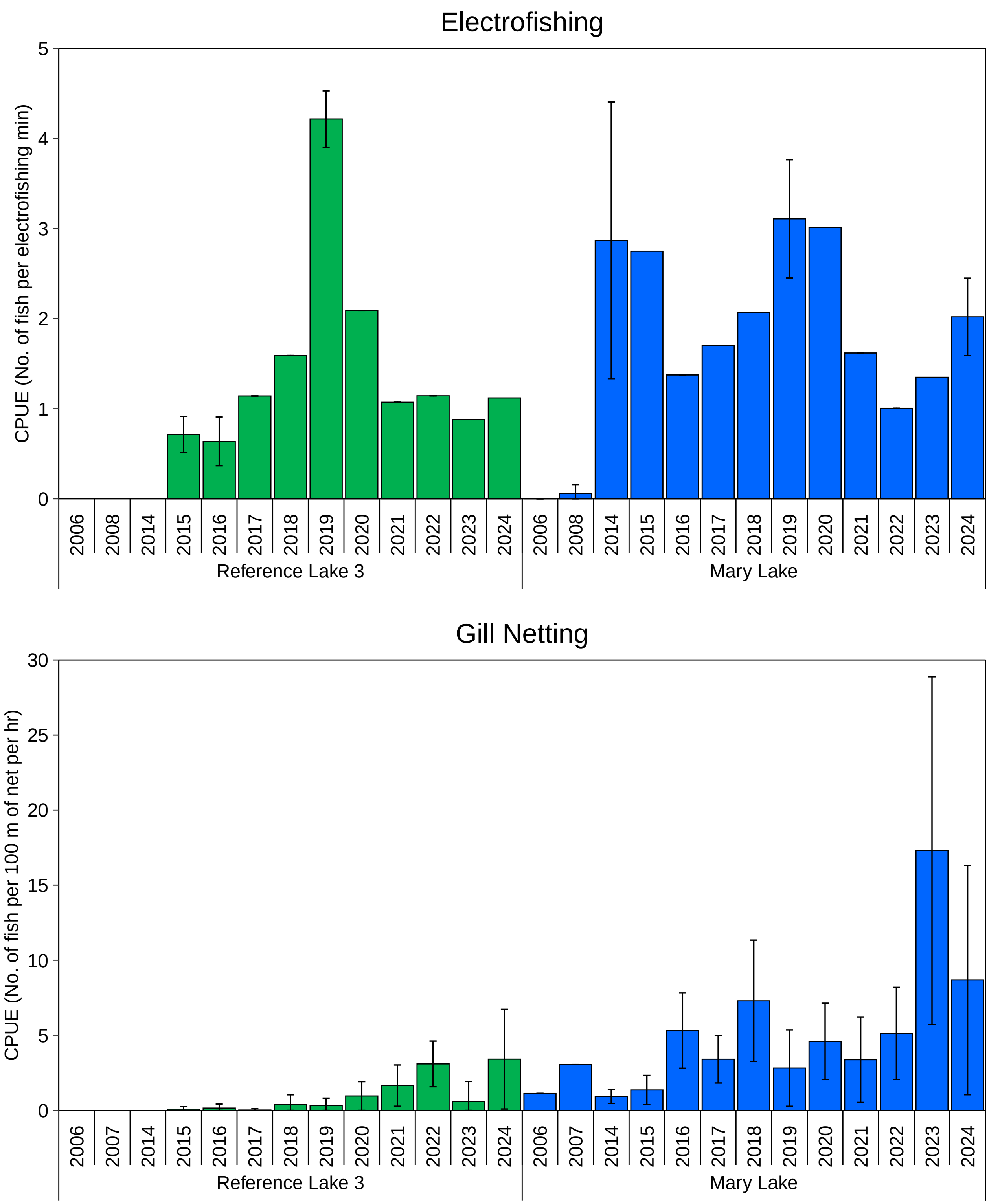

Figure 5.11: Catch−per−unit−effort (CPUE; mean ± standard deviation) of Arctic Charr Captured by Backpack Electrofishing and Gill Netting at Mary Lake (BL0), Mary River Project CREMP, 2006 to 2024

Notes: Data presented for fish sampling conducted in fall during baseline (2006, 2007), construction (2014), and operational (2015 to 2023) mine phases. Reference areas are shown in green and mine−exposed areas are shown in blue.

and G.25)[57,58]. Arctic charr YOY were distinguished from non-YOY using fork length cut-offs of 5.0 cm for Mary Lake and 4.0 cm for Reference Lake 3, based on analysis of LFDs coupled with supporting length and weight measurements and age determinations (Figure 5.12, Appendix Tables G.4 and G.25). Since at least ten YOY arctic charr were captured from each lake (n = 12 and 10 for Reference Lake 3 and Mary Lake, respectively; Appendix Table G.26), statistical comparisons of health endpoints were conducted separately for YOY and non-YOY groups.

LFD for all fish and non-YOY age classes were significantly different between Mary Lake and Reference Lake 3 (Table 5.8, Figure 5.12, Appendix Figure G.23, Appendix Table G.26). Reference Lake 3 had dominant size classes between 4.75 and 5.5 cm (classified as age 1+) and few fish greater than 6.0 cm in length, while Mary Lake had more equal proportions of fish across all size classes from 5.25 to 10 cm and a small group of individuals less than 5 cm classified as YOY (Figure 5.12, Appendix Figure G.23). The LFD for nearshore arctic charr has consistently been different between Mary Lake and Reference Lake 3 over the period of mine operations since 2016 (Table 5.8) though there have been no consistent patterns in the relative frequencies of fish lengths between the lakes (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024a).

Arctic charr from Mary Lake, both YOY and non-YOY, were significantly longer (YOY= 38%, non-YOY = 42%) and heavier (YOY = 173%, non-YOY = 205%) compared to those from Reference Lake 3 (Table 5.8, Appendix Table G.26, Appendix Figures G.24 and G.25). No significant difference in the condition was observed for YOY, but non-YOY from Mary Lake had significantly greater condition than Reference Lake 3 (11%), with a MOD exceeding the CES_C of ± 10%, indicating an ecologically meaningful difference (Table 5.8, Appendix Table G.26). Sufficient YOY arctic charr to allow for statistical comparisons of fish health have been captured at both Mary Lake and Reference Lake 3 only in 2017, 2018, 2022, and 2012 (Table 5.8). In these years, YOY from Mary Lake were generally longer and heavier, with inconsistent differences in condition relative to Reference Lake 3 (Table 5.8). For non-YOY fish from Mary Lake were frequently longer and heavier and had better condition than those from

[57] Sample sizes at Mary Lake in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.27.

[58] The total number of fish captured in Mary Lake and Reference Lake 3 by electrofishing (Table 4.7, Appendix Table G.1) was greater than the number of fish sampled for the fish health assessment. The study design requires 100 fish from each lake to be sampled (measured and weighed; Baffinland 2015). Once field crews were certain that the minimum target sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

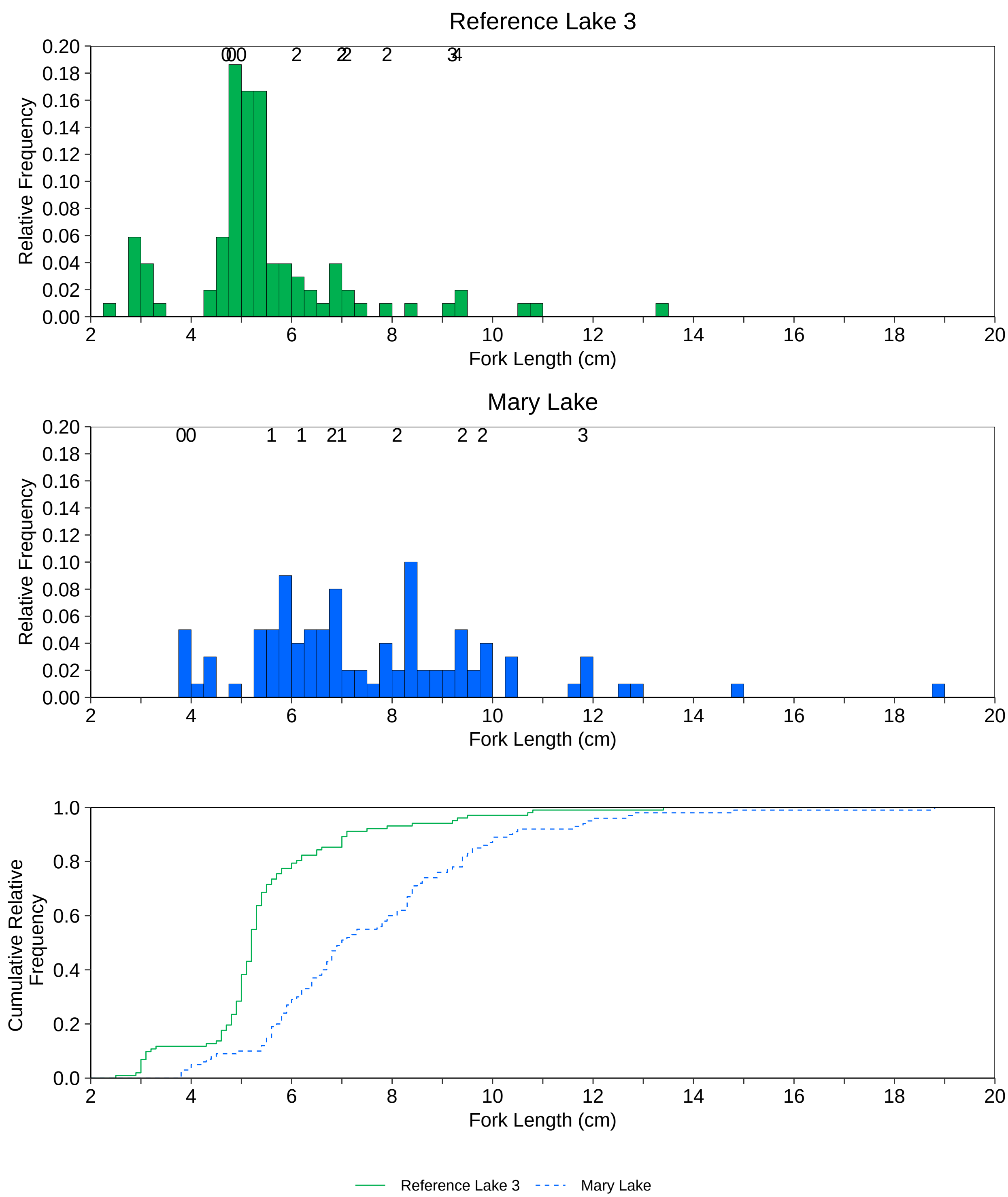

Figure 5.12: Relative Length−Frequency and Cumulative Length−Frequency Distributions for All Arctic Charr Captured by Backpack Electrofishing at Mary Lake (BL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Fish ages are shown above the bars, where available. Mary Lake n = 100; Reference Lake 3 n = 102.

Table 5.8: Summary of Statistical Results for Arctic Charr Population Comparisons between Mary Lake (BL0) and Reference Lake 3 (REF-03), and between Mary Lake Mine Operational and Baseline Period Data, for Fish Captured by Electrofishing and Gill Netting Methods, Mary River Project CREMP, 2015 to 2024

Data Set by Sampling Method	Response Category	Endpoint	Statistically Significant Differences Observed? [a]																			
			Mary Lake vs Reference Lake 3										Mary Lake Mine Operational Year vs Baseline Period [b]									
			2015	2016	2017	2018	2019	2020	2021	2022	2023	2024	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Electrofishing Samples	Survival [c]	Length-Frequency Distribution	No	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	-	-	-	-	-	-	-	-	-	-
		Age	**Yes (-43%)**	No	No	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Energy Use (YOY)	Size (mean fork length)	-	-	**Yes (+14%)**	No	-	-	-	**Yes (+17%)**	-	**Yes (+38%)**	-	-	-	-	-	-	-	-	-	-
		Size (mean weight)	-	-	**Yes (+40%)**	No	-	-	-	**Yes (+70%)**	-	**Yes (+173%)**	-	-	-	-	-	-	-	-	-	-
	Energy Storage (YOY)	Condition (body weight-at-fork length)	-	-	**Yes (-7.7%)**	**Yes (-11%)**	-	-	-	**Yes (+50%)**	-	No	-	-	-	-	-	-	-	-	-	-
	Energy Use (non-YOY)	Size (mean fork length)	No	No	**Yes (+17%)**	**Yes (+10%)**	**Yes (-27%)**	No	**Yes (+39%)**	**Yes (+28%)**	No	**Yes (+42%)**	-	-	-	-	-	-	-	-	-	-
		Size (mean weight)	No	No	**Yes (+51%)**	No	**Yes (-61%)**	No	**Yes (+185%)**	**Yes (+89%)**	No	**Yes (+205%)**	-	-	-	-	-	-	-	-	-	-
		Growth (weight-at-age)	**Yes (+99%)**	No	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
		Growth (fork length-at-age)	**Yes (+23%)**	No	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Energy Storage (non-YOY)	Condition (body weight-at-fork length)	**Yes (+3%)**	No	No	**Yes (-8%)**	**Yes (+4%)**	**Yes (+2.6%)**	**Yes (+5.1%)**	**Yes (+4.0%)**	**Yes (-17%)**	**Yes (+11%)**	-	-	-	-	-	-	-	-	-	-
Gill Netting Samples [d]	Survival	Length Frequency Distribution	-	-	-	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	**Yes**	No	**Yes**	No	No
		Age	-	-	-	-	-	-	-	-	-	-	No	**Yes (-14%)**	No	-	-	-	-	-	-	-
	Energy Use	Size (mean fork length)	-	-	-	**Yes (+12%)**	**Yes (+24%)**	**Yes (+23%)**	**Yes (+32%)**	**Yes (+24%)**	No	**Yes (+36%)**	**Yes (+6%)**	No	**Yes (-5%)**	No	**Yes (-4%)**	No	No	No	No	No
		Size (mean weight)	-	-	-	**Yes (+51%)**	**Yes (+96%)**	**Yes (+118%)**	**Yes (+186%)**	**Yes (+164%)**	No	**Yes (+155%)**	**Yes (+19%)**	No	**Yes (-9%)**	No	**Yes (-14%)**	No	**Yes (+8.5%)**	No	No	No
		Growth (fork length-at-age)	-	-	-	-	-	-	-	-	-	-	No	**Yes (nc)**	No	-	-	-	-	-	-	-
		Growth (weight-at-age)	-	-	-	-	-	-	-	-	-	-	No	**Yes (nc)**	No	-	-	-	-	-	-	-
	Energy Storage	Condition (body weight-at-fork length)	-	-	-	**Yes (+3%)**	**Yes (+3%)**	**Yes (+14%)**	**Yes (+28%)**	**Yes (+26%)**	**Yes (-6.9%)**	**Yes (+13%)**	No	**Yes (+3%)**	**Yes (+5%)**	**Yes (-3%)**	**Yes (-5%)**	No	**Yes (+6.0%)**	**Yes (+6.9%)**	No	No

BOLD indicates a statistically significant difference.

Notes: "-" indicates data not available for comparison. YOY = Young-of-the-Year. nc = non-calculable magnitude.

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

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

[c] The length-frequency distribution for Reference Lake 3 includes all fish, whereas for baseline conditions, it only includes non-YOY fish.

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

Reference Lake 3 but no consistent directional differences in size or condition were observed compared to Reference Lake 3 from 2015 to 2024 and MODs for condition between the two lakes were below the CES_C in all years except 2023 (Table 5.8). There were no clear patterns in Mary Lake's productivity relative to the reference lake (e.g., chlorophyll-a concentrations [Section 5.3.3) and benthic invertebrate density [Section 5.3.4]) that suggested lake productivity was a major contributing factor to greater size of arctic charr from Mary Lake compared to fish from Reference Lake 3. No baseline data for nearshore arctic charr were collected at Mary Lake, precluding a before-after comparison.

Overall, results of comparisons to Reference Lake 3 in 2024 and over the period of mine operations do not indicate adverse mine-related effects on the health of nearshore arctic charr in Mary Lake.

Littoral/Profundal Arctic Charr

In August, 2024 a total of 101 and 84 arctic charr were sampled for fish health assessment from littoral and profundal habitat of Mary Lake and Reference Lake 3, respectively (Appendix Tables G.9 and G.29)[59,60]. The LFD for littoral/profundal arctic charr differed significantly between the two lakes, with the lengths of fish captured in Reference Lake 3 being mostly less than 35 cm while the majority of fish captured in Mary Lake were between 31 and 45 cm long (Table 5.8, Figure 5.13, Appendix Table G.30). The LFD for littoral/profundal arctic charr has consistently been different between Mary Lake and the reference lake since 2018 (Table 5.8) generally reflecting higher relative frequencies of larger fish in Mary Lake (Minnow 2016a 2017, 2018, 2019, 2020, 2021b, 2022, 2023, and 2024a).

Arctic charr from Mary Lake were significantly longer (36%), heavier (155%), and exhibited significantly better body condition (13%) than those from Reference Lake 3 (Table 5.8, Appendix Table G.30, Appendix Figure G.27). The absolute MOD in condition was outside the CES_C of ± 10%, indicating that the observed difference was ecologically meaningful (Table 5.8, Appendix Table G.30). Fork length, body weight, and condition for littoral/profundal arctic charr from Mary Lake have been consistently greater than those from Reference Lake 3 from 2018 to 2024, except in 2023, when no significant differences were observed for length and weight and

[59] Sample sizes at Mary Lake in 2024 met minimum requirements to detect a ±10% difference in condition relative to Reference Lake 3 and baseline data based on *a priori* power analysis using 2023 data (Minnow 2024a). *A priori* power analysis was also conducted in 2024 to determine the appropriate fish sample sizes required to detect various effect sizes in future surveys with results presented in Appendix Table G.27.

[60] The total number of fish captured in Mary Lake by gill netting (Table 5.7, Appendix Tables G.2 and G.28) was greater than the number of fish sampled for the fish health assessment. The study design targets 100 fish from each lake for sampling (measurement of length and weight; Baffinland 2015). Once field crews were certain that the minimum target sample size was reached, additional fish were enumerated only in order to limit stress resulting from fish handling.

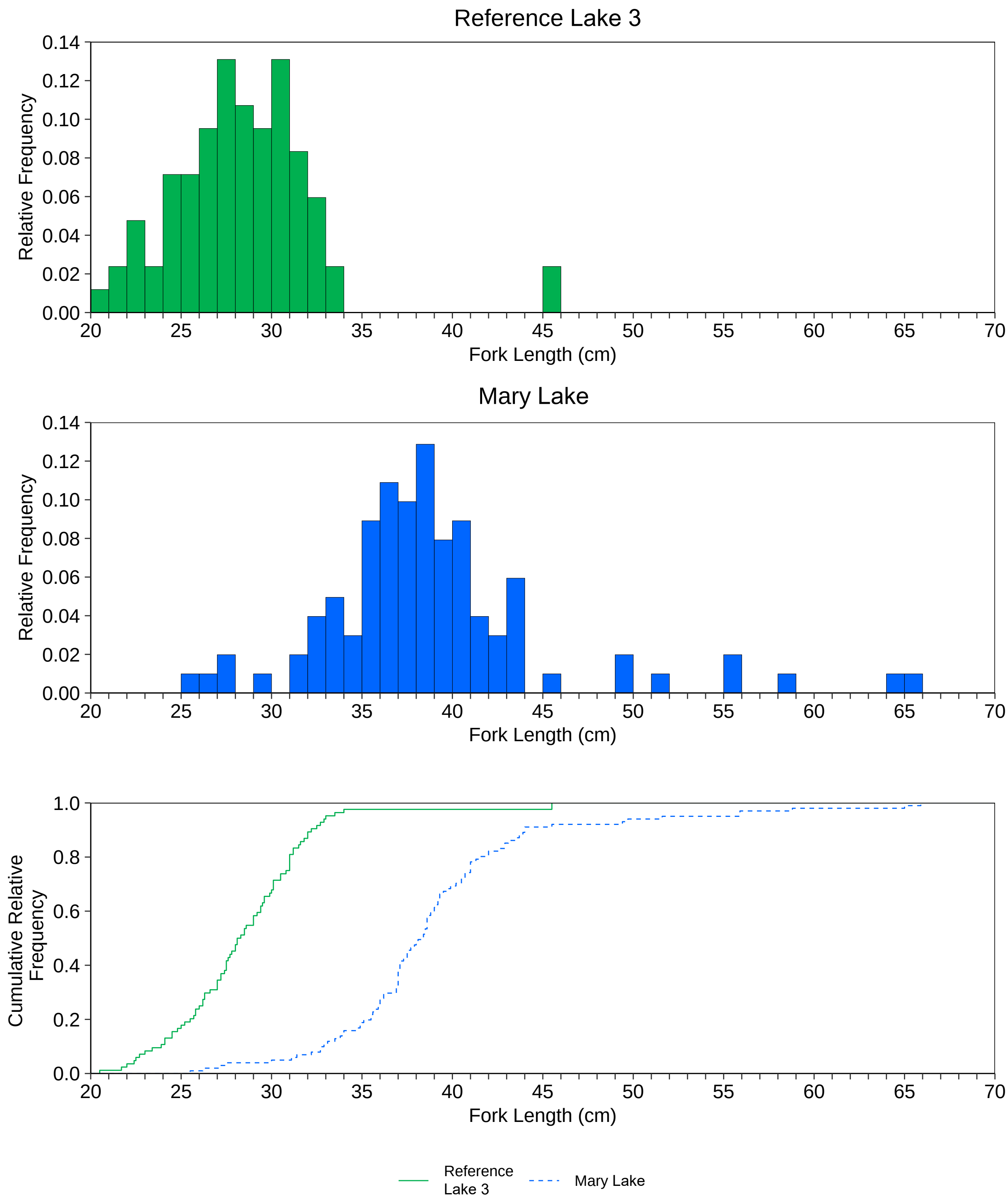

Figure 5.13: Relative Length−Frequency and Cumulative Length−Frequency Distributions for Arctic Charr Captured by Gill Netting at Mary Lake (BL0) and Reference Lake 3 (REF-03), Mary River Project CREMP, August 2024

Notes: Mary Lake n = 101; Reference Lake 3 n = 84.

when condition was significantly lower in fish from Mary Lake (Table 5.8). Although the condition of fish from Mary Lake was significantly higher in 2024 relative to the reference lake, with a MOD outside the CES_C , the MOD remained within the range observed during the mine operations period (2018 to 2023; Table 5.8). There were no clear patterns in Mary Lake's productivity relative to the reference lake (e.g., chlorophyll-a concentrations [Section 5.3.3) and benthic invertebrate density [Section 5.3.4]) that suggested lake productivity was a major contributing factor to greater size of littoral/profundal arctic charr from Mary Lake compared to fish from Reference Lake 3. However, multiple factors including littoral and profundal fish density and capture efficiency, as well as variation in nearshore fish density, size, and condition may have also been factors.

When comparing adult arctic charr from Mary Lake in 2024 to the baseline period, no significant differences were observed in LFD, body size (length and weight), or condition (Table 5.8, Appendix Table G.30, Appendix Figures G.26 and G.28). Since 2015, arctic charr length, weight, and condition in Mary Lake have varied directionally relative to baseline, where they have frequently either not been significantly different or, for condition, where differences have been within the CES_C (Table 5.8).

Arctic charr from the littoral and profundal zones of Mary Lake were generally larger and had greater condition compared to Reference Lake 3 from 2018 to 2024. Furthermore, littoral/profundal arctic charr at Mary Lake showed no consistent differences in length or weight and no ecologically relevant differences in condition compared to baseline. Therefore, no mine-related adverse effects on the health of adult arctic charr at Mary Lake are indicated since the onset of mine operations in 2015.

5.3.6 Effects Assessment and Recommendations

In 2024, water chemistry in Mary Lake met all AEMP benchmarks and WQGs, and no parameters were elevated compared to both reference and baseline levels in any season, indicating no mine-related influence on water quality.

In 2024, the following sediment quality AEMP benchmarks were exceeded at Mary Lake:

- Arsenic concentrations in one profundal sediment sample exceeded the AEMP benchmark of 5.9 mg/kg at Station BL0-16 in August (8.36 mg/kg); and
- Chromium concentrations in one individual profundal sediment sample exceeded the AEMP benchmark of 98 mg/kg at Station BL0-06 in August (103 mg/kg). This chromium sample also exceeded the SQG of 90 mg/kg.

Since arsenic and chromium concentrations in sediment were not elevated compared to reference and baseline concentrations, the exceedance of AEMP benchmarks is likely due to natural variation. The mean concentrations of these parameters throughout profundal habitats in Mary Lake were below the AEMP benchmark in 2024. No other sediment quality parameters had elevated concentrations compared to Reference Lake 3 and baseline in 2024, indicating no mine-related influence on sediment quality at Mary Lake.

No adverse mine-related effects on chlorophyll-a (primary productivity), BIC, or fish health (arctic charr) were observed at Mary Lake in 2024.

According to the Mary River Project AEMP Management Response Framework, the absence of any mine-related changes in water or sediment chemistry concentrations or to biota, as observed at Mary Lake in 2024, requires no further management action (Figure 2.6).

Comparison to FEIS Predictions

A comparison of water quality at Mary Lake in the 2024 spring, summer, and fall seasons to FEIS predictions for Aqueous Non-point Source Emissions effects related to applicable SWSQ2 (Site Water Management) indicated all parameter concentrations were within the Level II significance rating for magnitude expected for the watercourse during mine operations. Therefore, Mary River water quality was in conformance with predictions made in the Baffinland FEIS (Baffinland 2012).

Comparison of sediment quality in Mary Lake in 2024 to FEIS predictions related to Airborne Emission sources (i.e., fugitive dust; FEIS Issue SWSQ-17-3) indicated all mean parameter concentrations were within the applicable significance rating magnitudes expected for lake sediments during mine operations. Therefore, sediment quality at Mary Lake was in conformance with predictions made in Baffinland FEIS (Baffinland 2012).

Water and sediment quality at Mary Lake in 2024 where parameter concentrations were within applicable FEIS significance rating magnitude predictions also meant that FEIS predictions for (absence of) effects on arctic charr health and condition were also met. Therefore, arctic charr health and condition at Mary Lake in 2024 was in conformance with predictions made in the Baffinland FEIS (Baffinland 2012).

6 CONCLUSIONS

6.1 Overview

The objective of the 2024 Mary River Project CREMP was to assess potential mine-related impacts on the chemical and biological conditions of aquatic environments near the mine after ten years of operations. The CREMP employs an effects-based approach that includes standard EEM techniques that were applied to evaluate water quality, sediment quality, phytoplankton, BIC, and fish populations in mine-exposed areas of the Camp Lake, Sheardown Lake, and Mary River/Mary Lake systems. Potential mine-related effects were assessed by comparing 2024 data to applicable reference conditions, baseline data, and site-specific AEMP benchmarks, which guide management response decisions within a four-step Management Response Framework (Baffinland 2015). Effects determinations for key waterbodies in each system were based on weight-of-evidence, considering AEMP benchmark exceedances, mine-related influences on water or sediment quality, and associated mine-related adverse effects on aquatic biota. Where necessary, recommendations for further study were provided to support decisions regarding appropriate management actions. A summary of results, including determinations of mine-related effects identified in the 2024 CREMP is provided for each of the CREMP study systems including the Camp, Sheardown, and Mary River/Lake systems in Table 6.1.

6.2 Camp Lake System

In 2024, the Mary River Project CREMP identified potential mine-related effects on abiotic and biotic factors within the Camp Lake system. Mine-related influences on water quality were observed in the CLT1 Main Stem, particularly in the Upper Main Stem (Station L2-03), where concentrations of aluminum, iron, uranium, sulphate, sodium, and molybdenum indicated a potential mine-related influence based on concentrations that were elevated compared to baseline and reference in 2024 and increasing trends/patterns since the baseline period (see Section 3.1.1.3; Table 6.1). However, there were no similar increasing trends/patterns for these parameters over the mine operations period suggesting that potential mine-related influence has not been intensifying with ongoing mine operations (see Section 3.1.1.3; Table 6.1). Of these parameters, only aqueous concentrations of total aluminum and iron exceeded their respective AEMP benchmarks (in summer and summer and fall, respectively). Increasing concentrations, since the baseline period and over the period of mine operations, were observed for total and dissolved uranium and indicated a mine-related influence (see Section 3.1.1.1; Table 6.1). No corresponding adverse effects on phytoplankton or the BIC in the CLT1 Main Stem were noted (see Sections 3.1.2 and 3.1.4; Table 6.1).

Table 6.1: Summary of AEMP Benchmark Exceedances, Effects Determinations, and Management Response Framework Recommendations, Mary River Project CREMP, 2024

System/ Waterbody			AEMP Benchmark Exceedance	Effects Determination Summary	Action Level Response Under the AEMP Management Response Framework	Recommendations	Comparison to FEIS Predictions[a]
Camp Lake System	Camp Lake Tributary 1	CLT1 North Branch	The mean aqueous total copper concentration slightly exceeded the AEMP benchmark of 0.0022 mg/L, with a concentration of 0.00244 mg/L in the fall.	While the mean aqueous total copper concentration slightly exceeded the AEMP benchmark and WQG (0.002 mg/L) in fall 2024, total and dissolved copper concentrations in 2024 were consistent with reference and baseline conditions and total copper concentrations have not significantly increased over the mine operations period. This suggests the exceedance is due to natural variation rather than mine influence. No other water quality parameter concentrations indicated mine-related influence and no adverse mine-related effects on phytoplankton or the BIC were observed.	No action required.	N/A	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
Camp Lake System	Camp Lake Tributary 1	CLT1 Main Stem	The aqueous total aluminum concentration exceeded the AEMP benchmark of 0.179 mg/L in spring (0.249 mg/L) at the Upper Main Stem (Station L2-03). The aqueous total iron concentration exceeded the AEMP benchmark of 0.326 mg/L in spring (0.401 mg/L) and fall (0.330 mg/L) at the Upper Main Stem (Station L2-03).	At the CLT1 Upper Main Stem, aqueous concentrations of total aluminum, iron, and sulphate, along with total and dissolved uranium (total concentration exceeded the WQG of 0.015 mg/L in summer and fall), sodium, and molybdenum, had elevated concentrations compared to reference and baseline in 2024. Significant increasing temporal trends since the baseline period (2005 to 2024) were observed for total iron and sulphate and total and dissolved sodium and molybdenum but there were no similar significant increasing trends for these parameters over the mine operations period (2015 to 2024). Similarly, visual assessment of temporal patterns for aluminum indicated higher concentrations in mine operations vs baseline periods but no clear increasing pattern over the mine operations period. Results indicate a potential mine-related influence on aluminum, iron, sulphate, sodium, and molybdenum that has not been intensifying with ongoing mine operations. Significant increasing temporal trends since the baseline period (2005 to 2024) and over the mine operations period (2015 to 2024) were observed for total and dissolved uranium concentrations indicating a mine-related influence. No other water quality parameter concentrations at the CLT1 Lower Main Stem indicated mine-related influence and no adverse mine-related effects on phytoplankton or the BIC were observed.	Moderate Action Response is required for aqueous aluminum and iron based on AEMP benchmark exceedances and determination of potential mine-related influence. Low Action Response is required for aqueous uranium, sulphate, molybdenum, and sodium based on determination of mine-related or potential mine-related influence.	1. Continued monitoring of the BIC at CLT1-L2 is recommended in 2025 (and future CREMP studies) to monitor potential effects to biota and to support evaluation of elevated aluminum/iron concentrations above the AEMP benchmarks and uranium concentrations above the WQG at the CLT1 Upper Main Stem using a weight-of-evidence approach; 2. In 2025, temporal trend analysis will be conducted for total and dissolved (where applicable) aqueous concentrations of sulphate, aluminum, iron, molybdenum, sodium, and uranium in the CLT1 Main Stem to continue to investigate temporal trends/patterns, evaluate for increasing trends that are indicative of intensifying mine-related influences, and confirm potential mine-related influences. 3. In 2025, an analysis of total compared to dissolved aqueous concentrations of aluminum, iron, and uranium will be completed to investigate biological availability and further determine potential for effects on aquatic biota. 4. Potential sources of sulphate, aluminum, iron, molybdenum, sodium, and uranium to CLT1 will be investigated to better define mine-related influence and the potential for continued contributions. 5. Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
Camp Lake System	Camp Lake Tributary 2	CLT2	Water chemistry parameter concentrations were below AEMP benchmarks at CLT2.	No water quality parameter concentrations at CLT2 indicated mine-related influence and no adverse mine-related effects on phytoplankton or the BIC were observed.	No action required.	N/A	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
Camp Lake System	Camp Lake	JL0	Water chemistry parameter concentrations were below AEMP benchmarks at Camp Lake. Manganese concentrations in two individual profundal sediment samples exceeded the AEMP benchmark of 4,370 mg/kg at Stations JL0-17 and JL0-21 in August (4,470 mg/kg and 4,400 kg/kg, respectively). Iron concentrations in one individual profundal sediment sample exceeded the AEMP benchmark of 52,400 mg/kg at Station JL0-17 in August (56,900 mg/kg).	Aqueous uranium concentrations were below the WQG (0.015 mg/L), but were elevated compared to reference and baseline in summer 2024. A visual assessment of temporal patterns indicated a defined increase in total uranium concentrations in all seasons between 2017 and 2022 though concentrations appear to have stabilized from 2022 to 2024. Results indicate a mine-related influence on uranium. Sediment quality exceeded AEMP benchmarks for manganese and iron in individual samples, but concentrations were not elevated compared reference and baseline suggesting results above benchmarks were due to natural variation rather than mine-related influence. No other sediment quality parameters indicated mine-related influence. No adverse mine-related effects on phytoplankton, the BIC, or arctic charr health were observed.	Low Action Response is required for aqueous uranium based on determination of mine-related influence.	1. In 2025, temporal trend analysis of aqueous total and dissolved uranium concentrations will be conducted for Camp Lake to further investigate temporal patterns. 2. In 2025, an analysis of total compared to dissolved aqueous concentrations of uranium will be completed to investigate biological availability and further determine potential for effects on aquatic biota. 3. Potential sources of uranium to Camp Lake will be investigated to better define mine-related influence and the potential for continued contributions. 4. Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site.	Water and sediment quality parameter concentrations were within the significance ratings for magnitude expected for the waterbody during mine operations and therefore conformed with FEIS predictions.

Notes: N/A = not applicable. AEMP = Aquatic Effects Monitoring Plan. WQG = water quality guideline. MDMER = Metal and Diamond Mining Effluent Regulations. LRL = laboratory reporting limits. FDP = final discharge point. TDS = total dissolved solids. TKN = total Kjeldahl nitrogen. CLT = Camp Lake Tributary. SDLT = Sheardown Lake Tributary. MRTF = Mary River Tributary-F. BIC = benthic invertebrate community. FEIS = Final Environmental Impact Statement.

[a] Comparisons to FEIS predictions were completed according to methods described in Section 2.5.1.2

Table 6.1: Summary of AEMP Benchmark Exceedances, Effects Determinations, and Management Response Framework Recommendations, Mary River Project CREMP, 2024

System/ Waterbody			AEMP Benchmark Exceedance	Effects Determination Summary	Action Level Response Under the AEMP Management Response Framework	Recommendations	Comparison to FEIS Predictions[a]
Sheardown Lake System	Sheardown Lake Tributary 1	SDLT1	Mean aqueous total cadmium concentrations were greater than the AEMP benchmark of 0.00008 mg/L in the summer (mean = 0.000334 mg/L) and fall (mean = 0.000206 mg/L).	Aqueous total cadmium concentrations exceeded the AEMP benchmark and WQG (0.00012 mg/L) in summer and fall, were elevated compared to reference and baseline in the same seasons in 2024, and showed significant increasing trends across all seasons since the baseline period (2005 to 2024) and over the mine operations period (2015 to 2024) indicating a mine-related influence. Several other water quality parameters, including barium, calcium, chloride, cobalt, conductivity, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, TDS, TKN, and uranium, had concentrations that were elevated compared to reference and baseline in at least one season in 2024. These parameters also showed either statistically significant increasing trends or visually increasing patterns in total concentrations particularly from 2022 to 2024 indicating mine-related influence. Mine-related influences on water quality parameters in SDLT1 are likely linked to site water management through the KM 105 Surface Water Management Pond (the KM 105 Pond; constructed in 2021 and 2022). No adverse mine-related effects on phytoplankton were observed. Mine-related influences on the BIC were detected, with results suggesting they were likely driven by organic matter enrichment and differences in physical habitat conditions rather than metal contamination as a primary stressor. Influences associated with site water management and remediation efforts at the KM 105 Pond are consistent with the factors that may have resulted in shifts to the SDLT1 BIC in 2024.	Moderate Action Response is required for aqueous cadmium based on AEMP benchmark exceedances and determination of mine-related influence. Low Action Response is required for aqueous barium, calcium, chloride, cobalt, conductivity, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, TDS, TKN, and uranium as well as the BIC based on determination of mine-related influence.	1. Continued monitoring of the BIC at SDLT1 is recommended in 2025 (and in future CREMP studies) to track potential effects on biota and support the evaluation of elevated parameter concentrations exceeding the AEMP benchmark (cadmium), as well as those that were elevated compared to reference and baseline conditions, using a weight-of-evidence approach. 2. In 2025, a temporal trend analysis of aqueous conductivity and total and dissolved (where applicable) concentrations of barium, calcium, cadmium, chloride, cobalt, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, TDS, TKN, and uranium will be conducted to further investigate temporal trends/patterns. 3. In 2025, an analysis of total compared to dissolved aqueous concentrations of metals determined to have mine-related effects at SDLT1 in 2024 will be completed to investigate biological availability and further determine potential for effects on aquatic biota. 4. Potential sources of elevated/increasing water quality parameters in SDLT1 will be further investigated to better define mine-related influence and the potential for continued contributions. 5. Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site. 6. The focus in 2025 for the KM 105 Pond remediation efforts will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. Water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT1 as a basis for informing the potential need for further investigations and mitigation. 7. Installation of a filter berm upstream of the water license Surveillance Network Program monitoring location Station MS-C-D (which is located on a tributary to SDTL1 that originates from the southeast and flows into SDLT1 between Stations D1-05 and D1-00) is planned in 2025 to further mitigate for mine-related contributions of TSS to SDLT1 associated with dust and other sources of TSS within the upstream catchment area.	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
	Sheardown Lake Tributary 9	SDLT9	The aqueous total ammonia concentration exceeded the AEMP benchmark of 0.855 mg/L in fall (1.45 mg/L). The aqueous nitrate concentration exceeded the AEMP benchmark/WQG of 3 mg/L in summer (8.12 mg/L) and fall (7.08 mg/L).	The aqueous total ammonia concentration exceeded the AEMP benchmark in summer and the aqueous nitrate concentration exceeded the AEMP benchmark and WQG in summer and fall of 2024. Concentrations of nitrate, TKN, nitrite, and total ammonia were elevated compared to reference and baseline in at least one season in 2024 and visual assessment of temporal data indicated that there parameters had concentrations that were higher than reference and baseline consistently in the fall and frequently in the summer from 2021 to 2024 . Mine-related influences on ammonia and nitrate at SDLT9 were previously identified in annual CREMP monitoring in 2022 and 2023 , prompting an expanded spatial water quality sampling program to determine the source(s) of nitrogen compounds to SDLT9 in the fall of 2024. The study identified the Dyno Nobel Emulsion Plant (Dyno facility), located adjacent to SDLT9, as the likely source. No adverse mine-related effects on phytoplankton were observed. There were ecologically meaningful differences in BIC structure at SDLT9 in 2024 compared to baseline, however the BIC was comparable to the reference creek. Though localized natural inter-annual variability in habitat conditions may account for changes in the BIC relative to baseline, mine-related influences on water quality at SDLT9 in 2024 also suggest the potential for a mine-related effect.	Moderate Action Response is required for aqueous total ammonia and nitrate based on AEMP benchmark exceedances and determination of mine-related influence. Low Action Response is required for aqueous nitrite and TKN based on determination of mine-related influence. No action is required for BIC based on the absence of confirmed mine-related influences. However, if observed changes in the BIC community structure at SDLT9 (identified as potentially mine-related) are associated with mine-related influences on aqueous concentrations of nitrogen compounds, it is anticipated that recommended actions associated with water quality parameters will also serve to appropriately mitigate effects on the BIC.	1. An activity audit concerning the transportation, storage, and handling of ammonium nitrate at the Dyno facility is being implemented, along with potential additional water and/or seepage sampling during the open water season in 2025, as needed, to help identify point source(s) of aqueous nitrogen compounds. Mitigation measures will be developed based on the findings. 2. Water quality monitoring at SDLT9 will continue in the 2025 CREMP to assess the effectiveness of mitigation efforts at the Dyno facility in reducing the concentrations of aqueous nitrogen compounds. This monitoring may be supplemented by expanded spatial sampling in the fall of 2025, if necessary to fully evaluate mitigation effectiveness.	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.

Notes: N/A = not applicable. AEMP = Aquatic Effects Monitoring Plan. WQG = water quality guideline. MDMER = Metal and Diamond Mining Effluent Regulations. LRL = laboratory reporting limits. FDP = final discharge point. TDS = total dissolved solids. TKN = total Kjeldahl nitrogen. CLT = Camp Lake Tributary. SDLT = Sheardown Lake Tributary. MRTF = Mary River Tributary-F. BIC = benthic invertebrate community. FEIS = Final Environmental Impact Statement.

[a] Comparisons to FEIS predictions were completed according to methods described in Section 2.5.1.2

Table 6.1: Summary of AEMP Benchmark Exceedances, Effects Determinations, and Management Response Framework Recommendations, Mary River Project CREMP, 2024

System/ Waterbody			AEMP Benchmark Exceedance	Effects Determination Summary	Action Level Response Under the AEMP Management Response Framework	Recommendations	Comparison to FEIS Predictions[a]
Sheardown Lake System	Sheardown Lake Tributary 12	SDLT12	Water chemistry parameter concentrations were below AEMP benchmarks at SDLT12.	Aqueous alkalinity, conductivity, and hardness, as well as concentrations of chloride, and TDS and total and dissolved barium, calcium, magnesium, molybdenum, potassium, sodium, strontium, and uranium were elevated compared to reference and baseline in spring 2024 as well as patterns of increasing spring concentrations over time since the initiation of sampling in 2021. Results indicate a potential mine-related influence on these water quality parameters. No adverse mine-related effects on phytoplankton were observed.	Low Action Response is required for aqueous alkalinity, barium, calcium, chloride, conductivity, hardness, magnesium, molybdenum, potassium, sodium, strontium, TDS, and uranium based on determination of potential mine-related influence.	1. In 2025, temporal trend analysis of aqueous alkalinity, conductivity, and hardness, and total and dissolved (where applicable) concentrations of barium, calcium, chloride, magnesium, molybdenum, potassium, sodium, strontium, TDS, and uranium will be conducted for SDLT12 to further investigate temporal patterns. 2. In 2025, an analysis of total compared to dissolved aqueous concentrations of barium, calcium, magnesium, molybdenum, potassium, sodium, strontium, and uranium will be completed to investigate biological availability and further determine potential for effects on aquatic biota. 3. Potential sources of the water chemistry parameters at SDLT12 for which mine-related influence was indicated in 2024 will be investigated to better define mine-related influence and the potential for continued contributions.	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
	Sheardown Lake Northwest	DL0-01	Water chemistry parameter concentrations were below AEMP benchmarks at Sheardown Lake NW. Mean iron concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 52,200 mg/kg at all sediment monitoring stations in August (mean = 70,557 mg/kg). Manganese concentrations exceeded the AEMP benchmark of 4,530 mg/kg in one littoral (Station DD-HAB 9-STN; 10,000 mg/kg) and one profundal (Station DL0-01-13; 5,090 mg/kg) sediment sample in August.	Aqueous concentrations of nitrate, sulphate, chloride and total and dissolved manganese, strontium, and uranium were elevated compared to reference and baseline in at least one season in 2024. Total and/or dissolved concentrations of each of these parameters, except manganese, showed statistically significant increasing trends since the baseline period and over the mine operations period and/or visual patterns of increasing trends, generally starting in 2018 or 2019 and persisting in 2024. Manganese concentrations showed no evidence of increasing patterns over time indicating no mine-related influence. Total and/or dissolved concentrations of molybdenum (although only elevated compared to reference in 2024) also showed statistically significant increasing trends since the baseline period and over the mine operations period. Results indicate a mine-related influence on nitrate, sulphate, chloride, molybdenum, strontium, and uranium. Mean iron concentrations in littoral and profundal sediments exceeded the AEMP benchmark, with statistically significant increasing trends over both baseline and mine operation periods. Spatial patterns in iron concentration within the lake were also observed. These findings suggest the emergence of a mine-related influence on sediment quality. Manganese concentrations in sediments from individual sampling stations exceeded the AEMP benchmark. Despite these exceedances, manganese concentrations were not elevated compared to reference and baseline suggesting that elevated concentrations are likely due to natural processes rather than mine-related influence. No adverse mine-related effects on phytoplankton, the benthic invertebrate community, or arctic char health were observed.	Low Action Response is required for aqueous chloride, nitrate, sulphate, molybdenum, uranium, and strontium based on determination of mine-related influence. Low Action Response is also required for sediment iron based on determination of mine-related influence.	1. In 2025, temporal trend analysis of aqueous total and dissolved (where applicable) concentrations of chloride, nitrate, sulphate, molybdenum, uranium, and strontium will be conducted for Sheardown Lake NW to further investigate temporal trends/patterns. 2. The focus in 2025 for the KM 105 Pond remediation efforts will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. Given likely influences of water management at the KM 105 Pond on water quality at Sheardown Lake NW (through inputs from SDLT1), water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT1 and Sheardown Lake NW as a basis for informing the potential need for further investigations and mitigation. 3. Potential sources of chloride, nitrate, sulphate, molybdenum, uranium, and strontium to Sheardown Lake NW will be investigated to better define mine-related influence and the potential for continued contributions. 4. Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site. 5. In 2025, temporal trend analysis of iron concentrations in littoral and profundal sediment will be repeated with the inclusion of new monitoring data to evaluate whether an increasing trend continues to be identified and to contribute to determination of mine related influences despite iron sediment concentrations that were similar to reference and baseline conditions in 2024; and 6. Further spatial comparisons between iron concentrations in sediment within the lake for the determination of the influence of key lake tributaries on the influx of sediment iron into Sheardown Lake NW.	Water and sediment quality parameter concentrations were within the significance ratings for magnitude expected for the waterbody during mine operations and therefore conformed with FEIS predictions.

Notes: N/A = not applicable. AEMP = Aquatic Effects Monitoring Plan. WQG = water quality guideline. MDMER = Metal and Diamond Mining Effluent Regulations. LRL = laboratory reporting limits. FDP = final discharge point. TDS = total dissolved solids. TKN = total Kjeldahl nitrogen. CLT = Camp Lake Tributary. SDLT = Sheardown Lake Tributary. MRTF = Mary River Tributary-F. BIC = benthic invertebrate community. FEIS = Final Environmental Impact Statement.

[a] Comparisons to FEIS predictions were completed according to methods described in Section 2.5.1.2

Table 6.1: Summary of AEMP Benchmark Exceedances, Effects Determinations, and Management Response Framework Recommendations, Mary River Project CREMP, 2024

System/ Waterbody			AEMP Benchmark Exceedance	Effects Determination Summary	Action Level Response Under the AEMP Management Response Framework	Recommendations	Comparison to FEIS Predictions[a]
Sheardown Lake System	Sheardown Lake Southeast	DL0-02	Water chemistry parameter concentrations were below AEMP benchmarks at Sheardown Lake SE. Arsenic concentrations exceeded the AEMP benchmark of 5.9 mg/kg in one littoral (Station DL0-02-4; 6.29 mg/kg) and two profundal (Stations DL0-02-12 and DL0-02-13; 6.42 and 6.68 mg/kg, respectively) sediment samples in August. Mean chromium concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 79 mg/kg at all sediment monitoring stations except for DL0-02-1 in August (mean = 85.7 mg/kg). Mean iron concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 34,400 mg/kg at all sediment monitoring stations in August (mean = 54, 920 mg/kg). Mean manganese concentrations in littoral and profundal sediment samples exceeded the AEMP benchmark of 657 mg/kg at all sediment monitoring stations in August (mean = 2,574 mg/kg). Nickel concentrations exceeded the AEMP benchmark of 66 mg/kg in two littoral (Stations DL0-02-11 and DL0-02-4; 72.6 and 70 mg/kg, respectively) and two profundal (Stations DL0-02-12 and DL0-02-13; 76.3 and 66.9 mg/kg, respectively) sediment samples in August. Phosphorus concentrations exceeded the AEMP benchmark of 1,278 mg/kg in two profundal sediment samples at Station DL0-02-12 (1,300 mg/kg) and DL0-02-13 (1,310 mg/kg) in August.	Aqueous concentrations of nitrate and sulphate were elevated compared to reference and baseline in all seasons (spring, summer, fall) in 2024. Total concentrations of nitrate and sulphate, as well as total and/or dissolved molybdenum and uranium showed statistically significant increasing trends since the baseline period and over the mine operations period. Results indicate a mine-related influence on these water quality parameters. Arsenic, chromium, iron, manganese, nickel, and phosphorus exceeded AEMP benchmarks in littoral and profundal sediments. However, none of these parameters had concentrations that were elevated compared to both reference and baseline, suggesting that elevated concentrations are likely due to natural processes rather than mine-related influence. No adverse mine-related effects on phytoplankton, the benthic invertebrate community, or arctic char health were observed.	Low Action Response is required for aqueous nitrate, sulphate, molybdenum, and uranium based on determination of mine-related influence.	1. In 2025, temporal trend analysis of aqueous total and dissolved (where applicable) concentrations of nitrate, sulphate, molybdenum, and uranium will be conducted for Sheardown Lake NW to further investigate temporal trends/patterns. 2. Spatial comparisons of the concentrations of nitrate within the lake will be completed in 2025 for evaluation of the overall influence of inputs from the Dyno facility (via SDLT9) into Sheardown Lake SE. Water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT9 and Sheardown Lake SE as a basis for informing the potential need for further investigations and mitigation. 3. Potential sources of nitrate, sulphate, molybdenum, and uranium to Sheardown Lake SE will be investigated to better define mine-related influence and the potential for continued contributions. 4. Development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations. The development of this benchmark may include review of baseline and reference concentrations as well as review of potential toxicological effects relevant to the aquatic biota present near the mine site. 5. The focus in 2025 for the KM 105 Pond remediation efforts will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. Given potential influences of water management at the KM 105 Pond on water quality at Sheardown Lake SE (through inputs from Sheardown Lake NW), water quality information collected during the 2025 CREMP will be used to monitor water quality of Sheardown Lake NW and Sheardown Lake SE as a basis for informing the potential need for further investigations and mitigation.	Water and sediment quality parameter concentrations were within the significance ratings for magnitude expected for the waterbody during mine operations and therefore conformed with FEIS predictions.

Notes: N/A = not applicable. AEMP = Aquatic Effects Monitoring Plan. WQG = water quality guideline. MDMER = Metal and Diamond Mining Effluent Regulations. LRL = laboratory reporting limits. FDP = final discharge point. TDS = total dissolved solids. TKN = total Kjeldahl nitrogen. CLT = Camp Lake Tributary. SDLT = Sheardown Lake Tributary. MRTF = Mary River Tributary-F. BIC = benthic invertebrate community. FEIS = Final Environmental Impact Statement.

[a] Comparisons to FEIS predictions were completed according to methods described in Section 2.5.1.2

Table 6.1: Summary of AEMP Benchmark Exceedances, Effects Determinations, and Management Response Framework Recommendations, Mary River Project CREMP, 2024

System/ Waterbody			AEMP Benchmark Exceedance	Effects Determination Summary	Action Level Response Under the AEMP Management Response Framework	Recommendations	Comparison to FEIS Predictions[a]
Mary River and Mary Lake System	Mary River	G0, E0, C0 Series	Water chemistry parameter concentrations were below AEMP benchmarks at the Mary River.	The aqueous concentration of total aluminum slightly exceeded the WQG of 0.100 mg/L at the G0-01 upstream station in the spring and summer and the aqueous concentration of total chromium exceeded the WQG of 0.001 mg/L at the E0-20 mine-adjacent station in the summer. Aluminum concentrations were not elevated compared to reference and baseline in individual seasons in 2024 and/or showed no evidence of increasing patterns over time at Mary River sampling stations indicating no-mine related influence. Aqueous total chromium, total lead, sulphate, and dissolved organic carbon concentrations were elevated compared to reference and baseline in either summer or fall in 2024. Visual assessment of temporal data indicated no consistent increasing patterns since mine operations began in 2015, indicating no mine-related influence. No adverse mine-related effects on phytoplankton or the BIC were observed.	No action required.	N/A	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
Mary River and Mary Lake System	Mary River Tributary-F	MRTF	Water chemistry parameter concentrations were below AEMP benchmarks at MRTF.	Aqueous concentrations of sulphate, nitrate, and selenium were elevated compared to reference and baseline in at least one season in 2024. Nitrate and sulphate concentrations showed statistically significant increasing trends since the baseline period and over the mine operations period indicating a mine-related influence. Visual assessment of temporal data showed increases started in 2019 and 2017 for nitrate and sulphate, respectively, but have not be consistent over time, suggesting they are not intensifying with ongoing mine operations. Selenium concentrations at MRTF have frequently been below the LRL and evaluation for a temporal pattern is confounded by changing LRLs, therefore evidence does not suggest a potential mine-related effect on selenium at MRTF. Mine-related influences on nitrate and sulphate concentrations at MRTF may be associated with effluent discharge (i.e., from the MS-08 FDP into MRTF) No adverse mine-related effects on phytoplankton were observed.	Low Action Response is required for aqueous nitrate and sulphate based on mine-related influence.	1. In 2025, temporal trend analysis of aqueous concentrations of nitrate and sulphate will be conducted for MRTF to further investigate temporal trends/patterns. 2. In 2025, a special investigation will be conducted evaluating effluent and receiving water quality data that are routinely collected as part of MDMER requirements for the MS-08 FDP to evaluate the influence of the MS-08 FDP as a potential source of nitrate and sulphate to MRTF.	Water quality parameter concentrations were within the significance rating for magnitude expected for the watercourse during mine operations and therefore conformed with FEIS predictions.
Mary River and Mary Lake System	Mary Lake	BL0	Water chemistry parameter concentrations were below AEMP benchmarks at Mary Lake. Arsenic concentrations in one profundal sediment sample exceeded the AEMP benchmark of 5.9 mg/kg at Station BL0-16 in August (8.36 mg/kg). Chromium concentrations in one individual profundal sediment sample exceeded the AEMP benchmark of 98 mg/kg at Station BL0-06 in August (103 mg/kg).	No water quality parameter concentrations at Mary Lake indicated mine-related influence. Arsenic and chromium exceeded AEMP benchmarks in individual profundal sediment samples. Chromium also exceeded the SQG of 90 mg/kg. However, since mean concentrations remained below AEMP benchmarks and no increasing trends or elevated concentrations were observed compared to reference and baseline, these exceedances are likely due to natural variation rather than mine-related influences. No adverse effects on chlorophyll-a (primary productivity), the benthic invertebrate community, or arctic char health were observed.	No action required.	N/A	Water and sediment quality parameter concentrations were within the significance ratings for magnitude expected for the waterbody during mine operations and therefore conformed with FEIS predictions.

Notes: N/A = not applicable. AEMP = Aquatic Effects Monitoring Plan. WQG = water quality guideline. MDMER = Metal and Diamond Mining Effluent Regulations. LRL = laboratory reporting limits. FDP = final discharge point. TDS = total dissolved solids. TKN = total Kjeldahl nitrogen. CLT = Camp Lake Tributary. SDLT = Sheardown Lake Tributary. MRTF = Mary River Tributary-F. BIC = benthic invertebrate community. FEIS = Final Environmental Impact Statement.

[a] Comparisons to FEIS predictions were completed according to methods described in Section 2.5.1.2

In Camp Lake, uranium was the only water quality parameter for which elevated concentrations relative to baseline and reference and increasing temporal patterns indicated a mine-related influence, concentrations remained below the WQG (see Section 3.3.1.2; Table 6.1). No other mining-related effects were identified within the Camp Lake system.

Results of 2024 CREMP monitoring in the Camp Lake system require response actions under the AEMP Management Response Framework (Table 6.1). Recommendations were made for continued monitoring of the BIC in the CLT1 mainstem to monitor for potential effects to biota resulting from mine-related influences on water quality parameters. Additionally, in 2025, for water quality parameters for which there was a determination of mine-related or potential mine-related influence, temporal trend analyses will be conducted to further investigate temporal trends/patterns, total compared to dissolved concentrations of metals will be investigated to assess biological availability and potential effects on aquatic biota, and potential sources to affected waterbodies/watercourses in the Camp Lake system will be investigated to better define mine-related influence and the potential for continued contributions. Finally, development of an AEMP benchmark for uranium will be considered to support evaluation of the potential biological effects of observed concentrations.

6.3 Sheardown Lake System

In 2024, within the Sheardown Lake system, SDLT1 exhibited the most pronounced mine-related influences on water quality across the entire CREMP monitoring area. Mine-related influence was determined for several water quality parameters including barium, cadmium, calcium, chloride, cobalt, conductivity, lithium, magnesium, manganese, molybdenum, nitrate, potassium, selenium, sodium, strontium, sulphate, TDS, TKN, and uranium based on concentrations that were elevated compared to baseline and reference and increasing trends/patterns, particularly from 2022 to 2024 (see Section 4.1.1.2; Table 6.1). Of these parameters, only aqueous concentrations of total cadmium exceeded the AEMP benchmark (in summer and fall). Mine-related influence at SDLT1 is likely linked to the extensive mine site infrastructure within the SDLT1 catchment area, particularly site water management through the KM 105 Surface Water Management Pond (KM 105 Pond). Since its commissioning in 2022, the pond has not performed as expected, leading to persistent seepage and water quality challenges and multiple remediation efforts (see Section 4.1.1.2). Mine-related influences on the BIC at SDLT 1 were also detected, with results suggesting they were likely driven by organic matter enrichment and differences in physical habitat conditions rather than metal contamination as a primary stressor (see Section 4.1.4; Table 6.1). Influences associated with site water management and remediation efforts at the KM 105 Pond are consistent with the factors that may have resulted in shifts to the SDLT1 BIC in 2024 (see Section 4.1.4).

Water quality at SDLT9 has also been influenced by mining activities, resulting in elevated nitrogen-related compounds (ammonia, nitrate, nitrite, and TKN) as identified in the 2023 CREMP (Minnow 2024a) and again in 2024 (see Section 4.2.1.2; Table 6.1). A special investigation, involving expanded spatial sampling and completed in the fall of 2024 identified activities occurring at the Dyno facility, which stores ammonium nitrate on-site and is located adjacent to SDLT9, as the primary source of these compounds (see Appendix I). No adverse mine-related influences on phytoplankton were determined (see Section 4.2.2) but there were ecologically meaningful differences in BIC structure at SDLT9 in 2024 compared to baseline (however the BIC was comparable to the reference creek). Though localized natural inter-annual variability in habitat conditions may account for changes in the BIC relative to baseline, mine-related influences on water quality at SDLT9 in 2024 also suggest the potential for a mine-related effect (see Section 4.2.3; Table 6.1).

At SDLT12 in 2024, potential mine-related influence was determined on water quality parameters including alkalinity, barium, calcium, chloride, conductivity, hardness, magnesium, molybdenum, potassium, sodium, strontium, TDS, and uranium based on spring concentrations that were elevated compared to baseline and reference and increasing trends/patterns since the initiation of sampling at this location in 2021 (see Section 4.3.1.2; Table 6.1). Mine-related influence on water quality at SDLT12 is likely linked to snow stockpiling activities and inputs from dust deposition (mostly originating from the mine site crusher facility) in the catchment area upstream of the SDLT12 monitoring location.

At both Sheardown Lakes (NW and SE) in 2024, mine-related influences on water quality were determined for nitrate, sulphate, molybdenum, and uranium, as well as for chloride in Sheardown Lake NW only (see Sections 4.4.1.2 and 4.5.1.2; Table 6.1). Determinations were based on elevated aqueous concentrations relative to baseline and/or reference in 2024 as well as evidence of increasing trends/patterns in parameter concentrations, generally since 2018/2019 and persisting in 2024 (see Sections 4.4.1.2 and 4.5.1.2; Table 6.1). These trends suggest potential influences from activities occurring at the Dyno Facility (in Sheardown Lake SE only), site water management through the KM 105 Pond, and the broader mine site infrastructure within the catchments of the Sheardown Lakes.

At Sheardown Lake NW, mean iron concentrations in littoral and profundal sediments exceeded the AEMP benchmark, with statistically significant increasing trends over both baseline and mine operation periods (see Section 4.4.2; Table 6.1). Spatial patterns in iron concentration within the lake were also observed suggesting the emergence of a mine-related influence on sediment quality, that may be linked to contributions of sediment from tributaries (see Section 4.4.2). To date, no adverse-mine related biological effects have been identified in either of the Sheardown Lakes (see Sections 4.4.3 to 4.4.5 and 4.5.3 to 4.5.5; Table 6.1).

Results of 2024 CREMP monitoring in the Sheardown Lake system require response actions under the AEMP Management Response Framework (Table 6.1). Recommendations were made for continued monitoring of the BIC in at SDLT1 to monitor for potential effects to biota resulting from mine-related influences on water quality parameters. Additionally, in 2025, for water quality parameters for which there was a determination of mine-related or potential mine-related influence at SDLT1, SDLT12, Sheardown Lake NW and/or Sheardown Lake SE, temporal trend analyses will be conducted to further investigate temporal trends/patterns, total compared to dissolved concentrations of metals will be investigated to assess biological availability and potential effects on aquatic biota, and potential sources to affected waterbodies/watercourses in the Sheardown Lake system will be investigated to better define mine-related influence and the potential for continued contributions. Development of an AEMP benchmark for uranium will also be considered to support evaluation of the potential biological effects of observed concentrations.

Mitigation efforts will be implemented to improve water quality in the Sheardown Lake Tributaries and Sheardown Lakes NW and SE. At the KM 105 Pond, the focus for remediation efforts in 2025 will shift toward enhanced sediment control measures, incorporating chemical treatment, filtration, and improved settling structures rather than additional structural modifications. The installation of a filter berm upstream of the water license Surveillance Network Program (SNP) monitoring location Station MS-C-D (which is located on a tributary to SDTL1 that originates from the southeast and flows into SDLT1 between Stations D1-05 and D1-00; Figure 2.1) is also planned in 2025. The purpose of this additional infrastructure is to further mitigate for mine-related contributions of TSS to SDLT1 associated with dust and other sources of TSS within the upstream catchment area. Water quality information collected during the 2025 CREMP will be used to monitor water quality of SDLT1 and Sheardown Lakes NW and SE as a basis for informing the potential need for further investigations and mitigation.

An activity audit concerning the transportation, storage, and handling of ammonium nitrate at the Dyno facility is being implemented , along with potential additional water sampling during the open water season in 2025, as needed, to help identify point source(s) of aqueous nitrogen compounds. Mitigation measures will be developed based on the findings. Water quality monitoring at SDLT9 will continue in the 2025 CREMP to assess the effectiveness of mitigation efforts at the Dyno facility in reducing the concentrations of aqueous nitrogen compounds. This monitoring may be supplemented by expanded spatial sampling in the fall of 2025, if necessary to fully evaluate mitigation effectiveness.

Finally, temporal trend analysis of iron concentrations in littoral and profundal sediment in Sheardown Lake NW will be repeated with the inclusion of new monitoring data to evaluate whether an increasing trend continues to be identified and to contribute to determination of mine related influences despite iron sediment concentrations that were similar to reference and baseline conditions in 2024. Further, spatial comparisons between iron concentrations in sediment within the lake will be completed to support determination of the influence of key lake tributaries on the influx of sediment iron into Sheardown Lake NW.

6.4 Mary Lake System

Within the Mary River and Mary Lake System, mine-related influences in 2024 were limited to a small number of water quality parameters, including nitrate, sulphate, and selenium in MRTF; see Section 5.2.1.2; Table 6.1). At MRTF, aqueous concentrations of nitrate and sulphate were elevated compared to baseline and reference in at least one season in 2024 and concentrations of these parameters have shown increasing trends/patterns that started in 2019 and 2017 (for nitrate and sulphate, respectively) but have not be consistent over time, suggesting they are not intensifying with ongoing mine operations (see Section 5.2.1.2). Selenium concentrations at MRTF have frequently been below the LRL and evaluation for a temporal pattern is confounded by changing LRLs, therefore evidence does not suggest a potential mine-related effect on selenium at MRTF (see Section 5.2.1.2). Despite potential influences on water quality, no effects on phytoplankton at MRTF were determined (see Sections 5.2.2 and 5.2.3; Table 6.1). Mine-related influences on nitrate and sulphate concentrations at MRTF may be associated with effluent discharge (i.e., from the MS-08 FDP into MRTF). No mine-related influences on water quality, sediment quality, or biota were identified elsewhere in the Mary River or in Mary Lake (see Sections 5.3.1 to 5.3.5; Table 6.1).

Results of 2024 CREMP monitoring in MRTF require response actions under the AEMP Management Response Framework (Table 6.1). Recommendations were made including temporal trend analysis of aqueous concentrations of nitrate and sulphate to be conducted in 2025 to further investigate temporal trends/patterns. Further, in 2025, a special investigation will be conducted evaluating effluent and receiving water quality data that are routinely collected as part of MDMER requirements for the MS-08 FDP to evaluate influence of the MS-08 FDP as a potential source of nitrate and sulphate to MRTF.

6.5 Comparison to FEIS Predictions

Comparisons of water quality data to the FEIS SWSQ-2 (Site Wave Management; all stations), SWSQ-4 (Explosives; CLT1 upper and lower main stems, and SDLT9 and SDSE), SWSQ-5 (Quarries and Borrow Areas; CLT1 upper and lower main stems), SWSQ-7 (Camps and

Fuel Management; Camp Lake and Sheardown Lake systems), and SWSQ-9 (Airstrip and Airstrip Use; Camp Lake and Sheardown Lake) issues indicated all parameter concentrations were within the Level II significance rating for magnitude of effect for the applicable watercourse during mine operations. Therefore, water quality at all CREMP waterbodies and watercourses were in conformance with predictions made in the FEIS (Baffinland 2012).

The comparisons of sediment quality at Camp, Sheardown, and Mary Lakes in 2024 to FEIS predictions related to Airborne Emission sources (i.e., fugitive dust; FEIS Issue SWSQ-17-3) indicated all parameter concentrations were within the applicable significance ratings for magnitude of effect expected for lake sediments during mine operations. Therefore sediment quality at Camp, Sheardown, and Mary lakes were in conformance with predictions made in the FEIS (Baffinland 2012).

The overall comparisons of water quality and sediment quality data within the Camp Lake, Sheardown Lake, and Mary Lake systems in 2024 to FEIS predictions indicated all parameter concentrations were within applicable significance ratings for magnitude meant that FEIS predictions for (absence of) effects on arctic charr health and condition were also met. Therefore arctic charr health and condition at Camp Lake, Sheardown Lake, and Mary River and Lake were in conformance with predictions made in the FEIS (Baffinland 2012).

Project-related sedimentation accumulation thickness of less than 1 mm/year was predicted in the FEIS to result in negligible effects on direct mortality of arctic charr. Because the sediment accumulation rate over the 2023 to 2024 arctic charr egg incubation period was well below 1 mm/y at Sheardown Lake NW, FEIS predictions for (absence of) direct mortality of arctic charr were met (Minnow 2025). Therefore, direct fish mortality effects at Sheardown Lake NW were in conformance with predictions made in the FEIS (Baffinland 2012).

6.6 Conclusion

Overall, the most significant mine-related influences have been observed within the Sheardown Lake System, where most watercourses/waterbodies assessed in the CREMP have shown some degree of mine-related influence, with effects extending to the BIC in tributaries of Sheardown Lake NW (i.e., SDLT1) and Sheardown Lake SE (i.e., SDLT9). Links between mining activities within the Sheardown Lake System and the observed changes have been identified, and corresponding mitigation measures and recommendations have been provided. While some mine-related influences were noted in the Camp Lake and Mary River/Lake Systems, these effects appear to be more localized and, in the Camp Lake system, may be influenced by natural variation. Ongoing implementation of the annual CREMP will continue to assess potential mine-related influences and management actions will be applied as required according to the AEMP Management Response Framework.

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