Bridge River Project Water Use Plan

Seton Lake Resident Fish Habitat and Population Monitoring

Implementation Year 9

Reference: BRGMON-8

BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 9 (2021) Results

Study Period: April 1, 2021 to March 31, 2022

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BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 9 (2021) Results

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Reference: BRGMON-8

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Executive Summary

Data collection for Year 9 of this proposed 10-year study was completed in 2021. Results for Years 1 to 8 are provided in the previous reports produced for this program (Sneep 2022; Sneep 2021; Sneep 2019a; Sneep 2019b; Sneep 2018a, Sneep 2018b; Sneep 2015). Where relevant, comparisons across monitoring years have been included in this report. A synthesis of all results will be conducted following the final year of data collection which is scheduled for 2022. The primary objective of this monitoring program is to "collect better information on the relative abundance, life history and habitat use of resident fish populations in Seton Lake" (BC Hydro 2012).

Field studies for the Seton Lake Resident Fish Habitat and Population Monitoring Program (BRGMON-8) were conducted in both Seton and Anderson lakes. Starting as a pilot effort in Year 3 (2015), data collection in Anderson Lake has been included in each year since to provide context and comparison for the Seton Lake results. The two lakes are comparably sized, located within the same watershed, and have similar natural inflows; however, Seton Lake is impacted by the diversion from Carpenter Reservoir whereas Anderson Lake is not. As in Years 4 to 8 (2016 to 2020), sampling effort was fully extended to Anderson Lake in Year 9 (2021), including the full lake length for the annual fish population index sampling.

The general approach to this monitoring program is to collect a multi-year data set on the populations of selected resident fish species as well as key habitat conditions in these lakes in order to resolve data gaps and better inform the trade-off decisions made during the Water Use Planning process. The target species selected for this program were bull trout, rainbow trout and gwenis based on their ecological and social value in this context, and their potential for response to diversion effects. To-date, gwenis and bull trout have been sampled effectively for contributing to trend monitoring and evaluation of linkages with potential effects variables. However, rainbow trout catches have been consistently low, such that they will not be an appropriate species for addressing management questions but will continue to be included in qualitative/descriptive assessments in the reporting.

Four primary data collection methods were employed in Year 9 (2021) to document the biological characteristics of the resident fish population, generate an annual abundance index, and characterize relevant physical conditions that may impact fish habitats. These methods included:

- Thermal profile monitoring;
- Sedimentation rate monitoring;
- Resident fish population index survey in the lakes (by gill netting); and,
- Analysis of collected ageing structures (fin rays, otoliths, scales from target species for determining age distributions.

Since Year 3 (2015), sampling for the resident fish population index survey has been conducted by gill netting, which incorporated both nearshore and offshore habitats. In order to allow

concurrent sampling coverage of both Seton and Anderson lakes with the available budget, fish indexing effort was concentrated into one longer session in early fall, rather than dividing effort across two shorter sessions (spring and fall) as was the case in Years 1 and 2 (2013 and 2014).

Physical characteristics in the two lakes were described by characterizing the annual and seasonal characteristics of Carpenter diversion operations, temperature profiles, and sedimentation deposition. Analysis of this information documented differences in diversion inflow volumes among years and seasons, and differences in temperatures and sedimentation that can be attributed to the diversion inputs. Relative to conditions in Anderson Lake, the diversion resulted in colder water temperatures throughout the water column (by up to 7°C) and higher inputs of fine sediments (by 54 to 85 mg/day), according to depth and seasonal period at the inflow end of Seton Lake in 2021. There was also a gradient of effects across the length of Seton Lake (i.e., according to depth and season, temperatures were up to 6°C warmer, and there was 44 to 146 mg/day less sediment deposition at the outflow end, relative to the inflow end).

Approximately 319 net-hours of gill netting effort were employed in Seton (160 net-hours) and Anderson (159 net-hours) lakes over 8 days in late September and early October 2021. In total, 529 fish were captured from 66 sampling locations (33 on Seton Lake and 33 on Anderson Lake). This was a decrease of 176 fish relative to 2020; however, most of this was due to a decreased catch of non-target species. The gwenis catch was very similar to 2020 (only lower by 12 fish), but these were two of the three lowest catch-years for this species since the annual gill net sampling began in 2015 (the other lowest year was in 2016). As before, the sampling sites were distributed spatially throughout three longitudinal zones (i.e., inflow, mid, and outflow) in each lake. Sampling depths ranged from 0 to 60 m below the surface, and included surface, mid-column, and bottom sets. Captured fish included nine different resident species; target species made up 65% of the total (gwenis, bull trout, and rainbow trout comprised 57%, 5%, and 3% of the catch, respectively).

Catch-per-unit-effort (CPUE) values were generated for target species in Year 9 (2021). As reported in past years, highest CPUE for gwenis was recorded in Seton Lake, and lowest values were in Anderson Lake in 2021. Gwenis were again more numerous in nearshore sets in Seton Lake (nearshore= 134.5 fish·net-hour^{-10}; offshore= 8.5 fish·net-hour^{-10}), whereas the gwenis in Anderson Lake were more abundant in offshore habitats (nearshore= 0.0 fish·net-hour^{-10}; offshore= 2.3 fish·net-hour^{-10}). Highest CPUE for bull trout was in the nearshore zone of Anderson Lake (26.3 fish·net-hour^{-10}) and catch rates for rainbow trout were relatively low in each sampled habitat (0 to 3.1 fish·net-hour^{-10}). Generation of these catch statistics for each monitoring year will be used to establish whether the population trends for target species are increasing, staying the same, or decreasing across the period of monitoring years.

During the fall fish sampling session (late September to early October) the majority of mature gwenis in spawning-ready condition were sampled in the bottom-set nets at depths ≥ 20 m, and ≥ 45 m horizontal distance from the lake edge in Seton Lake. As such, these spatial distribution characteristics may represent potential spawning habitat characteristics in this lake. Anderson Lake gwenis were not in spawning-ready condition due to the later spawn timing for this population (estimated to be in November or December; Morris et al. 2003), and they were distributed close to shore (for juvenile age classes) or in the water column in pelagic habitats, reflecting the typical rearing and feeding distribution for this species.

Based on analysis of size, gwenis tended to be larger in Anderson Lake, particularly after Age-2, and reached a maximum age of 4 years. The Seton Lake gwenis were smaller and had a maximum age of 3 years (at which they were sexually mature), reflecting growth and age-at-maturity differences between these populations. Bull trout captured in Seton Lake in 2021 (*n*= 5) ranged from Age-5 to Age-11 (length range = 345 to 733 mm), and in Anderson Lake (n= 21) from Age-4 to Age-11 (length range = 280 to 612 mm), which were similar to the size and age ranges described for these populations in previous years. Based on median size-at-age values, bull trout appear to grow faster in Seton Lake than in Anderson Lake, which is likely related to reduced predator densities and better foraging conditions (i.e., larger prey) in Seton Lake.

As in previous years, assessment of bull trout stomach contents in Year 9 (2021) further contributed to the body of evidence that the various lifestages of *O. nerka* (i.e., sockeye or gwenis; eggs, juveniles and adults) comprise the dominant food source for this species in both lakes in the fall. Larger bull trout in Seton Lake are able to capitalize on the mature gwenis, which are smaller bodied in Seton Lake, whereas juvenile gwenis were the dominant food items in Anderson Lake.

Summary of BRGMON-8 Management Questions and Interim (Year 9 – 2021) Status

Primary Objectives	Management Questions	Year 9 (2021) Status Based on Results To-Date
To collect better information on the relative abundance, life history and habitat use of resident fish populations in Seton Lake.	1. What are the basic biological characteristics of resident fish populations in Seton Lake and its tributaries?	**Species Composition:** Sampling has documented nine resident fish species, which were present in both lakes. Gwenis, bull trout and rainbow trout have been identified as target species for monitoring. Sample sizes for rainbow trout have been consistently low, so the summary that follows focusses on gwenis and bull trout, for which there is more representative data. **<u>Gwenis</u>** **Relative Abundance:** Gwenis are the most abundant resident species in the Seton Lake catch but are much less abundant in the Anderson Lake catch (likely because they are more dispersed in the pelagic zone at the time of sampling). **Size:** Adult gwenis are substantially larger in Anderson Lake, particularly after Age-2. **Age/Maturity:** Gwenis in Seton Lake ranged in age from 1 to 3 years (and were sexually mature at Age-3); Anderson Lake gwenis had a maximum age of 4 years, similar to the typical spawning age for sockeye. **Distribution/Habitat Use:** At the time of the survey (late Sep to early Oct), gwenis in Seton Lake were more abundant in nearshore sets (between ~45 and 90 m from shore) and >20 m depth, which likely coincides with spawning location characteristics for this population based on evidence of spawning-readiness. By longitudinal zone, abundance in Seton Lake was highest at the outflow end again in Year 9 (2021), and lowest in the inflow section. Gwenis in Anderson Lake have been either <15 m from shore near the surface (juveniles), or in the offshore sets (>75 m from shore) within the epilimnion and metalimnion thermal layers (i.e., 0 to 30 m depth) for adults. These locations likely correspond with their distribution in the lake for rearing and feeding. Highest catch rates were in the outflow end of the lake, although differences among zones have been much smaller compared to Seton Lake in most years. **Diet:** Zooplankton **<u>Bull Trout</u>** **Relative Abundance:** Bull trout were again the sixth most abundant species in the Seton Lake catch (behind gwenis, northern pikeminnow, peamouth chub, redside shiner, and bridgelip sucker), and third in Anderson Lake (behind gwenis and northern pikeminnow). **Size:** Analysis of median size-at-age and growth rates across years seems to confirm that bull trout grow faster in Seton Lake. Based on the available sample size, bull trout growth appears to slow after Age-5 in Anderson Lake or Age-6 in Seton Lake. Additional size and age data will be incorporated in the analysis once the final remaining year of data are collected. **Age/Maturity:** Captured bull trout have ranged in age between 2 and 12 years old and based on the minimum size of tagged fish that moved into Gates Creek during the spawning period, bull trout in this system may become mature by ~Age-3.

<table>
<tr><th>Primary Objectives</th><th>Management Questions</th><th>Year 9 (2021) Status Based on Results To-Date</th></tr>
<tr><td rowspan="3">To collect better information on the relative abundance, life history and habitat use of resident fish populations in Seton Lake.</td><td>1. What are the basic biological characteristics of resident fish populations in Seton Lake and its tributaries?</td><td>Distribution/Habitat Use: Bull trout distribution in Seton Lake corresponded directly with gwenis distribution in this lake. In Anderson, they were distributed in nearshore habitats between 10 and 90 m from shore and across the full range of sampled depths. Highest catch rates for this species have typically been near the Gates Creek mouth at the Inflow end of the lake, or near the outflow into Portage Creek in the Outflow end.
Diet: Gwenis adults, juveniles & O. nerka eggs

See Sections 3.2 and 4.</td></tr>
<tr><td>2. Will the selected alternative (N2-2P) result in positive, negative or neutral impact on abundance or index of abundance and diversity of target fish populations in Seton Lake?</td><td>Annual CPUE (# of fish per 10 net-hours):
<table>
<tr><th>Year</th><th>Seton Lake Gwenis</th><th>Anderson Lake Gwenis</th><th>Seton Lake Bull Trout</th><th>Anderson Lake Bull Trout</th></tr>
<tr><td>2015</td><td>44.9 (±16.1)</td><td>1.8 (±0.8)</td><td>0.7 (±0.4)</td><td>1.9 (±1.3)</td></tr>
<tr><td>2016</td><td>47.1 (±13.6)</td><td>0.6 (±0.3)</td><td>1.4 (±0.9)</td><td>31.5 (±9.7)</td></tr>
<tr><td>2017</td><td>63.1 (±15.4)</td><td>2.7 (±1.3)</td><td>4.9 (±2.3)</td><td>32.7 (±10.3)</td></tr>
<tr><td>2018</td><td>200.2 (±40.5)</td><td>2.6 (±1.3)</td><td>2.2 (±1.3)</td><td>16.4(±6.6)</td></tr>
<tr><td>2019</td><td>140.4 (±37.3)</td><td>1.3 (±0.8)</td><td>1.9 (±1.1)</td><td>7.0 (±3.1)</td></tr>
<tr><td>2020</td><td>56.5 (±15.6)</td><td>1.2 (±0.6)</td><td>1.4 (±1.1)</td><td>13.1 (±4.5)</td></tr>
<tr><td>2021</td><td>100.5 (±23.9)</td><td>1.5 (±0.9)</td><td>0.7 (±0.7)</td><td>19.2 (±5.3)</td></tr>
</table>
The Seton Lake gwenis abundance index was 100.5 (±23.9) fish per 10 net-hours in 2021, which was in between the lowest values (44.9–63.1 fish per 10 net-hours) in 2015–2017 and 2020, and the highest values (200.2 and 140.4 fish per 10 net-hours) in 2018 and 2019, respectively. Seton Lake bull trout have remained relatively stable other than a bit of an increase in 2017. The 2021 value was tied for lowest with 2015 (0.7 fish per 10 net-hours in both cases). The trend in the abundance index for gwenis in Seton Lake across the monitoring years has been slightly positive (see Figure 3.12); however, this was heavily influenced by the high values in 2018 and 2019. These inter-year differences likely have more to do with dominant/sub-dominant years for this species than an operations-related effect. The trend lines for gwenis in Anderson Lake and bull trout in both lakes were relatively flat (consistent with no effect or neutral impact). A before-after treatment comparison was not possible for this monitor due to the prior implementation timing of operating alternative N2-2P. See Sections 3.2 and 4.</td></tr>
<tr><td>3. Is there a relationship between the quality, quantity, and timing of water diverted from Carpenter Reservoir on the productivity of Seton Lake resident fish populations?</td><td>Two of the anticipated effects of the Carpenter diversion on Seton Lake were on the thermal regime and the introduction of fine particulate sediments. Based on data available from Years 4 to 9 (2016 to 2021), the diversion operations have an effect on both temperature and sediment deposition in Seton Lake, particularly at the inflow end, with a gradient of effect across the length of the lake. Specifically, temperature effects can have an influence on incubation conditions for spawned eggs, productive capacity of food organisms and growth rate for fish during the rearing period, as well as foraging habitat based on the depth and</td></tr>
</table>

Primary Objectives	Management Questions	Year 9 (2021) Status Based on Results To-Date
To collect better information on the relative abundance, life history and habitat use of resident fish populations in Seton Lake.		vertical extent of the thermocline. Fine sediments can impact productivity by increasing turbidity and introducing particulates that deposit on spawning substrates in the lake (i.e., for gwenis). Fish may compensate for these effects by utilizing areas of the lake that are least directly affected by the diversion inputs (e.g., the outflow section) and capitalizing on areas that receive groundwater upwelling effects (as has been observed for kokanee populations in other lakes; Hebert 2019). The presence of artesian upwelling has been documented for Seton Lake (Wade and Taylor 1979). MQ 3 will be addressed with the continuation of temperature profile and sedimentation rate monitoring (coincident with seasonal Carpenter diversion characteristics). Establishment of potential linkages with the fish abundance index information will continue to be explored, but potential correlations will not be included until all the annual data points are available (i.e., including Year 10). Relevant results & analysis from BRGMON-6 may also be incorporated with the results from this program by the end of the study period to inform the response to this question. **See Sections 3.1, 3.2 and 4**.
	4. Can refinements be made to the selected alternative to improve habitat conditions or enhance resident fish populations in Seton Lake?	We cannot answer this MQ at this stage. The program is intended to provide relevant information, coupled with applicable results provided by other programs (i.e., BRGMON-6), for answering this MQ. Relevant inputs from BRGMON-8 include seasonal water temperature profile and sedimentation rate effects of the diversion, as well as general fish population trends* for target species across the monitoring period. Providing more conclusive inputs (based on observed effects and relationships among monitored variables) for making management decisions about diversion operations, will require the full 10-year data set (i.e., the full duration of data collection for this program). *Note: It is anticipated that this program would be able to detect large-scale changes in relative abundance of target species, but not likely small-scale changes. Finer resolution in the results would require different methods (e.g., “Seabird” profiles, hydroacoustics), effort and budget. Also, based on the time lag between potential causal effects on gwenis spawning (diversion operations, temperature, sedimentation) and the measurement of fish population response (i.e., annual CPUE of gwenis at Age-3) there will be a limited number of years for linking effect and response at the end of the current monitoring period in 2022. See more information provided about this issue in **Section 4**.

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1. Introduction

1.1. Background

Seton Lake receives inflows from a combination of natural and regulated sources; however, since development of the hydroelectric infrastructure, inputs from regulated sources account for ~76% of total inflows, whereas natural inflows contribute ~24% by volume. Natural inflow sources include small tributaries that drain directly into the lake from the north and south sides of the valley, as well as Portage Creek at the west end, which conveys all of the attenuated inflows from the upper portion of the watershed. Regulated inflow sources include the Carpenter Reservoir diversion flows which are harnessed by BC Hydro's Bridge 1 (BR1) and Bridge 2 (BR2) Generating Stations for power production, and discharge into Seton Lake at Shalalth; and the Cayoosh diversion outflow at the public beach on the lake's east end. Outflows are regulated by BC Hydro's Seton Dam and Generating Station, which discharge into the Seton River and Fraser River, respectively.

The entire Bridge-Seton hydroelectric complex is integrated and the operations of each reservoir and facility are managed based on storage, conveyance, and generation decisions that account for water management priorities, electricity demands, plant maintenance requirements, fisheries impacts, as well as other values. Seton Lake and its associated BC Hydro facilities are situated at the downstream end of the Bridge-Seton system. Surface elevations in Seton Lake are managed within a narrow range (i.e., ≤0.6 m) relative to other reservoirs in the system. Daily and seasonal elevations and lake turn-over are driven by a wide range of factors: BR1 and BR2 operation; Seton Dam discharge; Seton Generating Station operation; Cayoosh Creek diversion inflows; and tributary inflows.

The Bridge-Seton Water Use Planning Consultative Committee (BRG CC) developed aquatic ecosystem objectives for Seton Lake that were established in terms of abundance and diversity of fish populations present in the lake. The Seton-Anderson watershed provides habitat for a wide range of anadromous and resident species, which are valued from a commercial, recreational, and cultural perspective. Use of the Seton-Anderson watershed by anadromous species, and trends in their relative abundance, have been assessed as a part of some of the other Bridge/Seton monitoring programs (i.e., BRGMONs #6, #13 and #14). However, there was also a lot of uncertainty about the basic biological characteristics of the *resident* fish species inhabiting Seton Lake, particularly gwenis, rainbow trout and bull trout.

The BRG CC agreed that resident species play a significant role in the functioning and overall productivity of the ecosystem and are of special importance because they have long been valued by First Nations as a source of food and for the significant cultural values that they embody (i.e., gwenis). While there were no systematic studies on these populations prior to hydroelectric development, observations and oral testimony from local St'at'imc people have suggested that there has been a significant decline in the abundance of resident species associated with the operation of the Bridge River Generating Stations. However, there was a fundamental lack of any data confirming the current species composition, relative abundance, habitat requirements, and life history of resident fish, as well as the impacts of the Carpenter Reservoir diversion, to directly support decision-making during the WUP.

During the BRG WUP process it was decided that changes to the operation of Seton Lake elevations (operating range ≤0.6 m) would not be considered because of physical constraints associated with discharge facilities and the power canal at Seton Dam. Thus, consideration of potential changes to BC Hydro operations were focussed on the seasonal timing of diversion flows from Carpenter Reservoir into Seton Lake. Trade-off decisions to define the preferred operating alternative were made using generalized ecosystem level indicators rather than explicit performance measures. The general ecosystem indicators were:

1) expected changes in productivity in Seton Lake associated with the Bridge River diversion are believed to be linked to the food base for resident species of Seton Lake, and
2) the estimated transfer of suspended sediment which was hypothesized to impact the success of lake/shore spawning species (e.g., gwenis).

The application of the general performance measures allowed trade-off decisions to be made however they required an extensive amount of qualitative judgment about which factors limited fish population abundance and diversity. As these judgments could not be supported with technical data or observation, significant uncertainty and risk associated with how well the assessments actually reflected resident fish population response to different operating strategies at the Bridge Generating Stations remained. To resolve these data gaps, reduce uncertainties, and reduce risk of further impacts to resident fish populations the BRG CC recommended monitoring to obtain more comprehensive information on Seton Lake habitats and the biological characteristics of the fish populations that use them.

The Bridge River Power Development Water Use Plan was accepted by the provincial Comptroller of Water Rights in March 2011. Terms of Reference for the Seton Lake Resident Fish Habitat and Population Monitoring program were developed and approved by late 2012, and field data collection activities were initiated in 2013. Under the WUP, monitoring for this program is scheduled to continue annually until 2022. Data collection for Year 9 of this proposed 10-year study was completed in 2021.

It should be noted that due to lessons learned during the first two years of sampling (2013 and 2014), key deficiencies in data collection methodologies and issues with the testability of some of the hypotheses included in the original study Terms of Reference (ToR) were identified. As per the ToR Addendum (March 2015): the management questions remained the same, but the hypotheses changed from those in the original ToR and new methods for fish sampling were proposed (i.e., gill netting instead of boat electrofishing).

1.2. Objectives, Management Questions and Study Hypotheses

The primary objectives of this monitoring program are: 1) to collect scientifically rigorous information on the species composition, relative abundance, life history and habitat use of resident fish populations in Seton Lake; and 2) to provide information required to link the effects of the Carpenter Reservoir diversion on fish populations to a) document impacts of the operating alternative on resident fish populations, and, b) support future decisions regarding the operation of BC Hydro facilities.

A set of management questions related to fisheries management goals and associated hypotheses regarding potential environment responses to the selected WUP operations were also defined to provide direction for the study.

The primary management questions to be addressed by this monitoring program are:

1. **What are the basic biological characteristics of resident fish populations in Seton Lake and its tributaries?**

This management question will be evaluated using fish population abundance or index of abundance, fish distribution and biological characteristics data. Target species include rainbow trout, bull trout and Kokanee (Gwenis).

2. **Will the selected alternative (N2-2P) result in positive, negative or neutral impact on abundance and diversity of fish populations in Seton Lake?**

This management question will be evaluated using weight-of-evidence as exhibited by trends in fish abundance indices and trends in their biological characteristics in conjunction with the range of Carpenter diversion characteristics. The underlying operational cause-effect relationship associated with any response may not be evident from this analysis alone. However, results from BRGMON-6 (Seton Lake Aquatic Productivity Monitoring) will be used to evaluate WUP operations impacts on lake productivity that could in turn be linked to impacts on productivity of the resident fish population.

3. **Is there a relationship between the quality, quantity, and timing of water diverted from Carpenter Reservoir on the productivity of Seton Lake target resident fish populations?**

This management question will be evaluated using basic habitat quality and diversion timing data collected in the lake in conjunction with trends in fish abundance and productivity data collected through BRGMON-6 study.

4. **Can refinements be made to the selected alternative to improve habitat conditions or enhance resident fish populations in Seton Lake?**

This management question will be evaluated based on insights gained from results under management questions 1-3.

The primary hypotheses (and sub-hypotheses) associated with these management questions from the Terms of Reference Addendum are:

H_1: The index of target species abundance in Seton Lake is stable over the monitoring period.

H_2: The measured habitat variables (temperature, turbidity) do not explain observed patterns of fish distribution in Seton Lake.

- **H_{2a}:** Patterns of fish distribution are not correlated with temperature profile.
- **H_{2b}:** Fish are distributed evenly within the lake (upstream vs. downstream).
- **H_{2c}:** Patterns of fish distribution are not correlated with turbidity.

H_3: The measured habitat variables (described in H_{2a} and H_{2c} above) do not substantially change between operation and shutdown events of the BR1 and BR2 plants over the monitoring period.

H_4: Potential food source variables explain observed patterns of target fish distribution in Seton Lake.

- **H_{4a}:** Patterns of Gwenis distribution are correlated with zooplankton abundance.
- **H_{4b}:** Patterns of bull trout distribution are correlated with *Oncorhynchus nerka* distribution.

H_5: The annual abundance index of target species is independent of discharge from the BR1 and BR2 plants.

- **H_{5a}:** The annual abundance index (by species) is independent of total BR1 and BR2 discharge.
- **H_{5b}:** The annual abundance index (by species) is independent of the within-year variability in BR1 and BR2 discharge.

These hypotheses reflect the generalized effects of BC Hydro operations that were understood to influence habitat suitability and resident fish population abundance in Seton Lake. The goal is to test these hypotheses by analyzing general fish population trends, habitat use, and general habitat characteristics in the lake, and making comparisons with data collected in Anderson Lake. Inferences about the impacts of the diversion from Carpenter Reservoir will be based on a weight-of-evidence approach that ultimately incorporates findings from the BRGMON-6 study with the time-series data collected under this program once all of the data are available.

1.3. Study Area

Field studies for the Seton Lake Resident Fish Habitat and Population Monitoring Program (BRGMON-8) were conducted in Seton and Anderson lakes in Year 9 (2021; Figure 1.1). For the purposes of monitoring the relative influence of the Carpenter Diversion inflows, as well as the main natural inflows and outflows (Gates Creek, Portage Creek and Seton River), the lakes were divided into three, approximately equal sections along their longitudinal axes. These are referred to as the: inflow, mid and outflow sections. It was assumed that the diversion influence would generally be correlated with proximity to the Bridge 1 and Bridge 2 Generating Station outflows, and that there could be different temperature, sediment deposition, and fish distribution patterns according to distance from these inputs. Each lake was divided in the same way to facilitate comparison of the results.

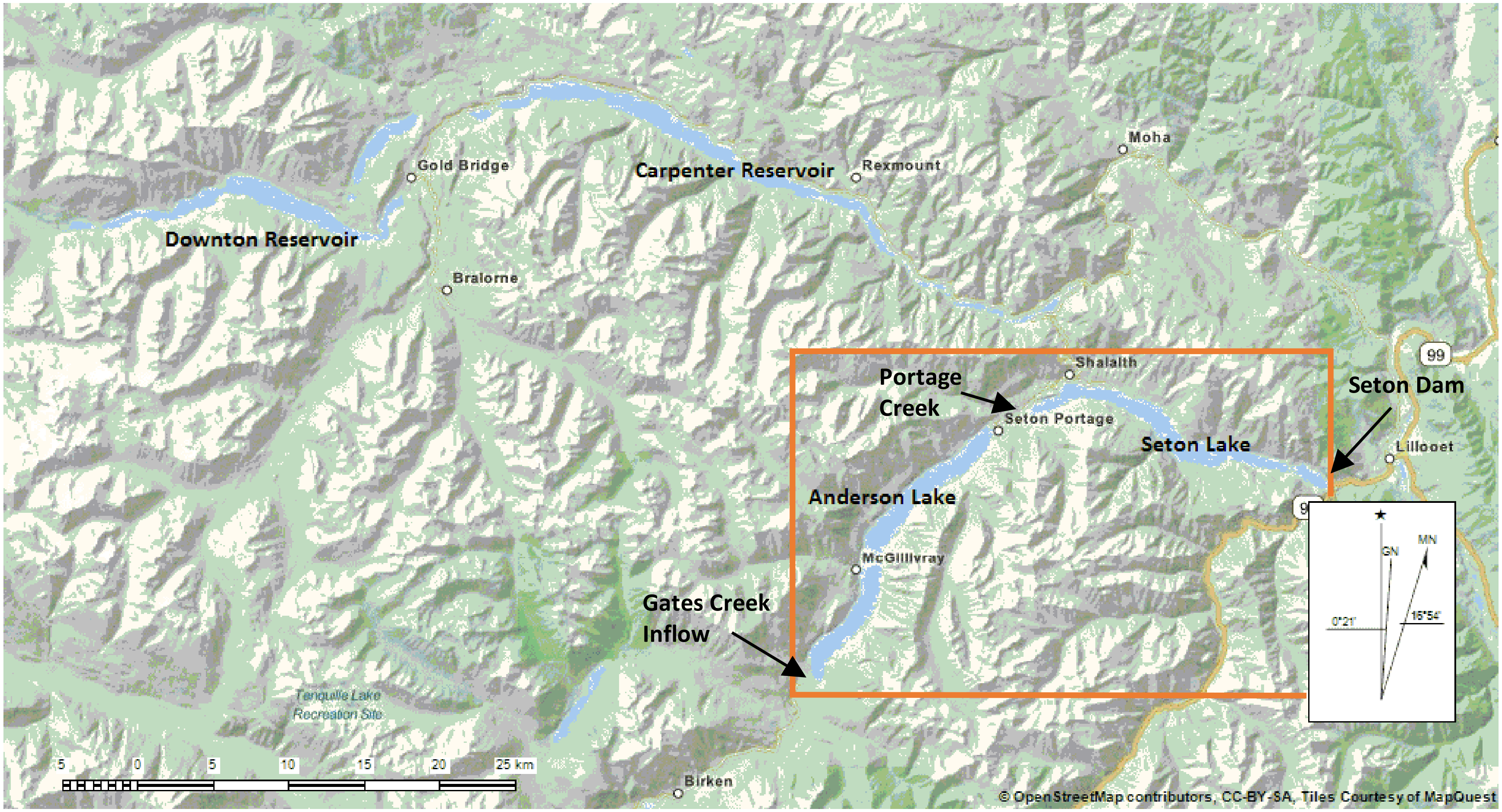


Figure 1.1 Overview of the Bridge and Seton watersheds. The extent of the BRGMON-8 study area, which includes all of Seton and Anderson lakes between the Gates Creek inflow and Seton Dam, is outlined by the orange rectangle.

1.4. Diversion Operations Context

In the context of the Bridge-Seton hydroelectric system, total average inflows to Downton and Carpenter reservoirs are approximately 40 $m^3 \cdot s^{-1}$ and 51 $m^3 \cdot s^{-1}$, respectively, for a combined total average diversion typically about 91 $m^3 \cdot s^{-1}$ into Seton Lake (BC Hydro 1993). Water is diverted through tunnels and penstocks from Carpenter Reservoir to two powerhouses on Seton Lake called Bridge River 1 (BR1) and Bridge River 2 (BR2). The maximum licensed discharge from these generating stations is 160 $m^3 \cdot s^{-1}$ (BR1 = 65.0 $m^3 \cdot s^{-1}$; BR2 = 95.0 $m^3 \cdot s^{-1}$) (BC Hydro 2011).

In 2016, BC Hydro identified issues with some of its infrastructure associated with water storage and flow conveyance within the Bridge-Seton hydroelectric complex. As a result, the storage capacity of Downton Reservoir has been reduced and conveyance of water through the system, including diversion of flows from Carpenter Reservoir to Seton Lake (via the diversion tunnels and generating units at Bridge 1 and 2), was affected. This resulted in a change from the typical N2-2P (i.e., post-Water Use Plan) operations to modified operations. Modified operations were first implemented in 2016 and are expected to continue for a number of years to mitigate the storage and conveyance issues and allow for the associated infrastructure to be fixed or replaced.

The changes that pertained specifically to Seton Lake since the modified operations have been implemented include increased diversion flow volume from BR1 and BR2 in spring (when Carpenter Reservoir reaches minimum elevations and begins to fill), and reduced volume during fall (when Carpenter Reservoir reaches maximum elevations), relative to previous (N2-2P) monitoring years (Table 1.1). However, *total* diversion volumes and mean discharge rates from the BR1 and BR2 generating stations (i.e., across all seasons combined) did not change between the two operational regimes.

Table 1.1 Summary of Diversion Flow Volumes and Average Discharge Rates from the Bridge Generating Stations (BR1 and BR2) for the BRGMON-8 Monitoring Years to-date based on hourly data provided by BC Hydro. Years affected by modified operations have been highlighted in yellow.

Study Year	Diversion Volume (Mm^3)					Average Discharge Rate ($m^3 \cdot s^{-1}$)
	Spring [a]	Summer	Fall	Winter	All Seasons Total	
1 (2013)	454	556	706	742	**2,457**	**78**
2 (2014)	249	644	659	863	**2,415**	**76**
3 (2015)	498	739	645	779	**2,661**	**85**
N2-2P Means (±SD)	**400 (±133)**	**646 (±92)**	**670 (±32)**	**795 (±62)**	**2,511 (±132)**	**80 (±5)**
4 (2016)	805	646	494	821	**2,765**	**87**
5 (2017)	525	589	612	788	**2,515**	**79**
6 (2018)	529	635	450	722	**2,335**	**74**
7 (2019)	497	877	590	597	**2,561**	**81**
8 (2020)	689	574	586	591	**2,441**	**78**
9 (2021)	734	583	619	738	**2,674**	**85**
Mod. Ops. Means (±SD)	**630 (±129)**	**651 (±115)**	**559 (±70)**	**710 (±96)**	**2,549 (±156)**	**81 (±5)**

[a] Seasonal periods were defined as follows: Spring = 21 Mar to 20 Jun; Summer = 21 Jun to 20 Sep; Fall = 21 Sep to 20 Dec; Winter = 21 Dec to 20 Mar for each year.

1.5. Sampling Design and Implementation To-Date

Monitoring programs in large lake contexts such as this one face significant challenges in that, despite extensive, rigorous sampling effort, they commonly fail to achieve the statistical certainty required to obtain precise population estimates and determine cause and effect. Challenges typically include low capture and re-capture rates, migration and 'open populations,' and a complex inter-relationship of variables affecting recruitment, growth and survival of fish populations. Despite these challenges, these programs can collect important inventory, life history and general trend information that is valuable to better understand the populations of interest and potential effects of operations based on weight-of-evidence inferences.

A great deal of learning about sampling conditions and fish distribution, densities, and catchability occurred during the first two years of monitoring, which helped inform the approach and strategy for this monitoring program going forward. There has also been key learning about deficiencies in data collection methodologies and issues with the testability of some of the hypotheses included in the original ToR. These issues necessitated revision to the original approach; these revisions were described in a ToR Addendum completed by BC Hydro and submitted to the provincial Comptroller of Water Rights in March 2015 (BC Hydro 2015).

A summary of the methods employed from 2013-2021 (study years 1 to 9) for accomplishing the goals and objectives of the BRGMON-8 program are provided in Table 1.2, for reference. For more information about the methods employed during past years, and the rationale behind them, please refer to the appropriate annual monitoring reports for those years (Sneep 2015; Sneep 2018a; Sneep 2018b; Sneep 2019a; Sneep 2019b; Sneep 2021; Sneep 2022).

Table 1.2 Methods implementation by Study Year to-date. For more details on the specific methods employed, refer to the annual monitoring report for each year.

Monitoring Method	Study Year								
	1	2	3	4	5	6	7	8	9 [a]
BC Hydro Operations Data	■	■	■	■	■	■	■	■	■
Temperature Monitoring (Continuous)									
• Tributaries			■	■	■	■	■	■	■
• In-lake Profile Arrays			■	■	■	■	■	■	■
Sedimentation Rate Monitoring				■	■	■	■	■	■
Shoreline Habitat Mapping			■		■				
Fish Population Index Surveys									
• Nearshore Boat Electrofishing	■	■							
• Gill Netting (Littoral & Pelagic)			■	■	■	■	■	■	■
Suppl. Tagging of Target Species (Angling)	■	■							
Tributary Spawner Surveys									
• Rainbow Trout (RB)	■	■							
• Bull Trout (BT)			■						
Radio Tagging & Telemetry (RB & BT)				■					
Stomach Contents Assessment (Bull Trout)			■	■	■	■	■	■	■
Scale, Fin Ray & Otolith Ageing			■	■	■	■	■	■	■

[a] The specific dates that each of the Year 9 (2021) field activities were completed are provided in Section 1.6, Table 1.3.

In Year 9 (2021), field activities for this program were again focussed on providing data to meet the primary objectives and management questions by building the BRGMON-8 data set and contributing an annual data point towards trends analysis to be completed at the conclusion of the 10-year monitoring program. Given the challenges and limitations outlined above, efforts have been focussed on establishing an annual index of abundance rather than attempting to quantify population sizes within the study area.

The study design in Year 9 (2021) was consistent with monitoring since the ToR Revision changes in 2015, and included the same three primary monitoring components:

- Thermal profile monitoring;
- Sedimentation rate monitoring;
- Resident fish population index survey (by gill netting).

Tributary spawner surveys were discontinued in Year 4 (2016) due to challenging conditions (e.g., turbidity, high flows) in the surveyed streams, and the limited information that they provided for answering the management questions (Sneep 2018a). It was not possible to incorporate mark-resight methods to quantify observer efficiency and residence time within the available budget for this component, and documented use of surveyed areas by the target species selected for this program (i.e., gwenis, bull trout and rainbow trout) was not considered a representative means of tracking population trends in this context. Among the target species, gwenis spawn in the lakes rather than tributary streams, bull trout are highly migratory and spawn in streams both within and outside the study area, and rainbow trout abundance (spawner and otherwise) appears to be quite low, particularly in Seton Lake.

The radio tagging and tracking component that was trialed in Year 4 (2016) has not been repeated since then because it relied on in-kind contributions and existing tagging efforts and telemetry infrastructure from the Seton Entrainment Study and BRGMON-14 programs that have not been available to this program since then. And lastly, the shoreline habitat mapping component that was completed in Seton Lake in 2015 (Year 3) and Anderson Lake in 2017 (Year 5), has not been repeated because the full extents of each lake were mapped in those previous years, and there were not any changes expected to the measured parameters based on conditions and operations since then. Instead, efforts and available budget resources for Year 9 (2021) were focussed on the consistent and thorough application of the primary methods (outlined above), along with the requisite sample collection, laboratory analyses, data analyses and reporting.

The fish sampling gear employed for this program (RIC gill nets; see Section 2.2) tends to sample a broad range of species and size classes of fish reasonably well; however, the smallest juveniles (e.g., Age-0+ and Age-1 bull trout, gwenis, or rainbow trout) are not sampled as effectively due to their small body size and habitat use. These juveniles were sampled more effectively by the trawling method incorporated for the fish sampling component of BRGMON-6 ("Seton Lake Aquatic Productivity Monitoring") on Seton and Anderson lakes and were more effectively inventoried as a part of that work (Limnotek 2016).

1.6. Year 9 (2021) Sampling Schedule

As per the original ToR, the activities associated with this monitoring program were recommended by the BRG WUP Consultative Committee for a total of 10 years. The study year covered by this report (2021) represents monitoring Year 9. The general schedule of field sampling activities is presented in Table 1.3.

Table 1.3 Schedule of Field Sampling Sessions and Activities in Year 9.

Field Activities	Dates (Year 9 - 2021)
Temperature array retrieval (R) and deployment (D)	5 May, 28 May[a] (R & D); 15, 16 Jul (R & D); 20, 21 Oct (R & D)
Sedimentation sampler retrieval (R) and deployment (D)	5 May, 28 May[a] (R & D); 15, 16 Jul (R & D); 20, 21 Oct (R & D)
Resident Fish Population Index Survey	28 Sep to 1 Oct (Seton); 5 to 8 Oct (Anderson)

[a] The outflow end of Anderson Lake could not be accessed in April or throughout most of the month of May 2021 due to low lake levels affecting the useability of the boat launch. As such, the field crew had to wait until late May when lake levels had increased.

2. Methods

The general approach to this monitoring program is to collect a long-term data set on selected resident fish species and physical habitat conditions in Seton Lake in order to detect trends, resolve data gaps, and better inform the trade-off decisions made during the WUP process. Following the successful extension of sampling in Years 3 – 8 (2015 to 2020), collection of comparable data from Anderson Lake was continued for all Year 9 (2021) activities. The intent was to provide additional context from a similar lake in the same watershed with shared ecology and analogous development impacts (i.e., railway, transmission lines, recreational cabins, and some residential), but no direct diversion impacts. Given the benefit of having comparable information from Anderson Lake to understand results and potential trends in context, attempts will be made to continue collecting data from both lakes within the constraints of the existing budget for the remainder of monitoring in Year 10.

Collection of coincident information on diversion operations from Carpenter Reservoir, in-lake habitat (i.e., sedimentation and temperature) conditions, and the resident fish population (including life history information, age structure and an index of abundance) is intended to allow identification of potential broad scale changes among years and across the 10-year monitoring period. Trends in these changes over time can be used to test hypotheses (presented in Section 1.2) about the relationships between diversion operations, the measured physical habitat variables, and fish population responses using a weight-of-evidence approach.

The target species selected for this program are bull trout, rainbow trout and gwenis based on their ecological and social value in this context, and their potential for response to diversion effects. Bull trout are a species of regional concern, rainbow trout are popular with recreational anglers, and gwenis are a historically significant winter food source for St'at'imc communities.

2.1. Physical Conditions

BC Hydro Operations

Records of BR1 and BR2 discharge rates (i.e., Carpenter diversion inflows to Seton Lake in $m^3 \cdot s^{-1}$) and Seton Lake surface elevations (measured in the forebay of Seton Dam in metres above sea level) were provided by BC Hydro Power Records as hourly values for each study year to-date. These data facilitated comparison of diversion inflows (rates and volumes) and management of lake levels among years. Diversion inflow volumes were also summarized by season to assess differences in flow delivery by time-of-year. The seasons were defined according to the following date ranges: Spring = March 21 to June 20; Summer = June 21 to September 20; Fall = September 21 to December 20; and Winter = December 21 to March 20 each year.

Thermal Profile Monitoring

Continuing since initial deployment in Year 3 (2015), temperature logger arrays were deployed to monitor the thermal profiles of the water column at the outflow end of Anderson Lake and both the inflow and outflow ends of Seton Lake throughout the year. Individual temperature loggers were deployed in M'sut Creek and Portage Creek, to monitor water temperatures from these natural inflow sources, and in the Seton Dam forebay. To monitor the temperature of the diversion inflows directly, a logger was placed in the BR1 tailrace starting in Year 6 (initially deployed on 19 July 2018). The locations of the temperature arrays and other logger locations in the study area are provided in Figure 2.1 in Section 2.2, below. Universal Transverse Mercator (UTM) coordinates for each temperature profile array and individual temperature logger locations are provided in Table 2.1. Since thermal profile monitoring was initiated in Year 3 (2015), temperature data were not available for years 1 and 2 (2013 and 2014).

Table 2.1 Universal Transverse Mercator (UTM) coordinates for temperature monitoring locations in Seton and Anderson lakes.

Location	UTM Coordinates (Zone 10U)	
	Easting	**Northing**
In-lake Temperature Arrays		
• Anderson Lake Outflow	548140	5614932
• Seton Lake Inflow	555060	5618911
• Seton Lake Outflow	569860	5613582
Individual Temperature Loggers		
• Portage Creek	550340	5617682
• BR1 Tailrace	553735	5620016
• M'sut Creek	560562	5616154
• Seton Dam Approach Channel	572103	5613519

The temperature loggers were TidbiT v2 loggers (model UTBI-001) manufactured by Onset Computer Corporation. For each array, loggers were attached at prescribed intervals to a line suspended vertically between a concrete anchor at the bottom and a float just below the surface. This arrangement was intended to span the thermal layers when the water column is stratified. Originally (i.e., in Years 3 to 6 and most of Year 7; 2015-2019), the depth intervals for the loggers were: 1, 10, 20, 25, 30, 40, 50, 60, and 70 m. However, in order to provide better resolution of temperatures within and around the thermocline, additional loggers were added at the following depths in Year 7: 15 m (installed on 17 July 2019); and 5, 12.5, 17.5, 22.5, and 35 m (installed on 24 October 2019). The deepest loggers at 60 m and 70 m were removed from each array on 24 October 2019 since hypolimnion temperatures were adequately characterized by the 40 m and 50 m loggers. These changes increased the total number of loggers on each array from 9 to 13, spanning a depth range of 1 to 50 m. This revised arrangement on each array was maintained in Years 8 and 9 (2020, 2021) and will be so for the remaining monitoring year (Year 10; 2022).

Starting in Year 6 (2018), a depth logger (SENSUS PRO model supplied by ReefNet Inc. (Mississauga, ON)) was deployed at the bottom of each line across the entire measurement period each year. The data from these loggers was used to calibrate the actual depths (± 0.3 m) of each temperature logger (at the prescribed intervals on each array as described above) since the placement (and therefore total depth) of the array was not exactly the same between deployments. A sinking line was run along the bottom from the array anchor to a fixed point on shore (i.e., tree trunk) to facilitate retrieval of the arrays, since the float was not visible at the lake surface.

Thermal layers that naturally form within a lake during the period of stratification (spring to fall in the northern hemisphere), are called the epilimnion, metalimnion, and hypolimnion. These terms are defined as follows:

Epilimnion: the mixed layer nearest the surface of the lake. It is the warmest layer during the period of stratification, and typically has a higher pH, dissolved oxygen concentration, and receives more sunlight than the lower layers.

Metalimnion: (also known as the thermocline) the distinct layer in which temperature changes more rapidly with depth than in the layers above or below. Seasonal weather changes and wind events can affect the depth and thickness of this layer.

Hypolimnion: the calm, dense layer that extends below the thermocline to the bottom of the lake. Temperatures in this layer are the lowest and most consistent across the year. Being the deepest layer, it is isolated from wind-mixing and receives little to no irradiance (light).

The thermal profile monitoring was intended to document the depths and temperature characteristics of each of these layers in Seton and Anderson lakes during each monitoring year going forward. Documenting the specific depths and extents of these layers is relevant to the resident fish sampling because pelagic fish species (such as gwenis) migrate among these thermal layers on a diel cycle for the purposes of feeding and could be useful for evaluating the effect of Carpenter Reservoir inflows (timing, magnitude and duration) on temperature profiles across the length of Seton Lake by the end of the monitor.

Scripts in R were written to produce colour-filled isopleths of the temperature profile data recorded by the logger arrays across the year and at a range of depths at the outflow end of Anderson Lake, and the inflow and outflow ends of Seton Lake. The R scripts and plots, written for presentation of temperature profile data for BRGMON-6 (Seton Lake aquatic productivity monitoring) by Annika Putt (Biologist with Instream Fisheries Research), were used for generating similar plots for BRGMON-8 to facilitate visual comparison of similarities and differences in thermal profile characteristics among locations and across the year.

Loggers deployed in Portage Creek, M'sut Creek, the BR1 tailrace and the Seton Dam forebay were fixed to a weight (i.e., a brick or concrete anchor, as appropriate to flow conditions and depth) that was connected to an anchor point on shore using a length of cable or sinking line. Measurement depth for the creek and forebay loggers was ~0.5–1.0 m below the surface, the BR1 tailrace logger depth was ~10 m. All of the temperature loggers were retrieved, downloaded, and redeployed approximately every 2 to 3 months during the field season (see Table 1.3 in Section 1.6 for 2021 dates). Data were downloaded onto a waterproof shuttle in the field and then transferred to a computer upon return to the office.

Sedimentation Monitoring

In addition to potential changes in temperature, the diversion supplying the BR1 and BR2 generating stations has introduced turbid water from the glacier-headed Bridge River valley. Drawn near the bottom of Carpenter Reservoir, these inflows routinely contain fine sediment particulates that are delivered to Seton Lake resulting in changes to colour, turbidity and sediment deposition. While differences in seasonal turbidity characteristics in Seton Lake have been assessed under the BRGMON-6 program, we undertook to investigate the seasonal and spatial differences in sedimentation rate (i.e., the amount of fine particles that settle out of suspension by season and distance from the BR1 and BR2 outflows).

In order to monitor the extent of this sedimentation and more closely document it by season and diversion flow volume, a set of sedimentation samplers have been continuously deployed in each of the three longitudinal sections (inflow, mid, and outflow) of Seton Lake, and the outflow section of Anderson Lake, since they were established in Year 4 (see Table 2.2; and Figure 2.1 in Section 2.2 for deployment locations). The samplers were suspended in the water column at ~40 m below the surface, which was just below the depths associated with highest gwenis spawner abundance (i.e., 20-35 m, based on the annual fish population index survey results from Years 3–9). The intention was to gather data that corresponds with potential spawning depths for this species. Samples were collected three times during the year (i.e., spring, summer, and fall; see Table 1.3 in Section 1.6 for 2021 dates).

Table 2.2 UTM coordinates for sedimentation monitoring locations in Seton and Anderson lakes.

Location	UTM Coordinates (Zone 10U)	
	Easting	Northing
Anderson Lake Outflow	548257	5615092
Seton Lake Inflow	555399	5618909
Seton Lake Mid	560750	5615572
Seton Lake Outflow	569664	5613606

The samplers were loaned to the project by Chris Perrin (Limnotek Research & Development Inc.) and consist of two open PVC tubes (dimensions: 40 cm long x 11 cm inside diameter) mounted side-by-side with metal brackets (Photo 2.1). Removable sampling cups (Photo 2.1 inset; 12 cm long x 11 cm inside diameter) were mounted to the bottom of each tube with a rubber gasket and two adjustable hose clamps. The top of each tube was fitted with a course plastic grate to keep large organic materials (e.g., leaves, etc.) out of the sample. The samplers were suspended vertically in the water column by two lines: one extended up to a submerged float, and the other extended down to a concrete anchor on the lake bottom.

Photo 2.1 Sedimentation sampler deployed year-round in Seton and Anderson lakes since Year 4 (2016). The removable sampling cups from which the sediment samples were collected is also shown (inset).

As in previous years, the samplers were continuously deployed for the entire year in 2021, other than the dates they were retrieved for sample collection (i.e., Seton Lake = 5 May, 16 July and 20 October 2021; Anderson Lake = 28 May, 15 July, and 21 October 2021). On each retrieval date, the samplers were pulled up from the sampling depth and lifted into the boat; care was taken to maintain the vertical orientation of the tubes so the collected sample was not disturbed or lost. Once the samplers were secured in the boat, a siphon pump, fitted with suction and discharge hoses, was used to draw the water in the tubes down to below the level of the rubber gasket at the top of the sampling cup. Then the sampling cup was removed and the water level was drawn further down with the pump to the top of the sediment layer, taking care not to disturb the collected sample. The remaining water and sediment sample was poured into a sample jar labelled with the sample date, location, and replicate number (i.e., tube 1 or 2). The bottom of the cup was scraped with a plastic spoon and all remaining sample was rinsed into the sample jar using a wash bottle. Following sample collection, the sample jars were sealed with water-tight lids.

Once the sample was collected, the tubes and sampling cups were scrubbed with pipe brushes to clean off algae and any other residual material to ensure they were clean to start the next sample period. The samplers were then reassembled for re-deployment in approximately the same position and depth as previous. Once the boat was manoeuvred into position (based on GPS coordinates and the length of the line tethered to shore), the anchor was lowered over the side, the tubes were allowed to slowly fill with water, and then the float was submerged as the sampler slowly descended to its sampling depth.

Same as for the temperature logger arrays, a depth logger (SENSUS PRO model supplied by ReefNet Inc. (Mississauga, ON)) was deployed on each sampler to provide actual depths (± 0.3 m) across the sampling period since the placement of each sampler was not exactly the same between deployments. The depth loggers were added to the sedimentation samplers for the first time during the May 2018 retrieval and redeployment.

All sediment samples were submitted to Caro Analytical Services (Kelowna, BC, Canada) for analysis of dry weight in grams. For the analyses of the dry weight data presented in this report, the average of the replicates (i.e., from each tube of the sampler) at each location was calculated, along with standard deviation.

2.2. Resident Fish Population Index Survey

The resident fish population index surveys are intended to provide information on the inter-annual variation in the relative abundance, distribution and size-at-age for target species (i.e., bull trout, rainbow trout and gwenis) in the study area. In addition to the focus on Seton Lake, 2021 sampling included all of Anderson Lake, as was done in Years 4 to 8 (2016-2020). The index survey data were collected in both the nearshore and offshore zones (i.e., ≤ 100 m and > 100 m horizontal distance from shore, respectively) of each lake by a standardized gill netting method, which covered a range of depths from 1 to 60 m from the lake surface in 2021.

Sampling effort was combined into one extended survey in the fall (i.e., late September to early October). This timing was selected because fish would have completed another season of growth and the lakes remain thermally stratified during this period; Gwenis orient around the thermocline depth for feeding purposes or near the substrate at depth for spawning in the fall. While in the lakes, bull trout may orient to the depths of prey species (e.g., gwenis, among others), and rainbows likely feed nearer the surface and at creek mouths, based on the results of this program to-date.

In BC, standardized gear specifications have been developed by the Resources Information Standards Committee (RISC) for the use of gill nets in lakes for indexing-level surveys (B.C. Ministry of Environment, Lands and Parks 1997). The standard gill nets are 91.2 m long and 2.4 m deep and consist of six panels (each 15.2 m long) of different mesh sizes that are strung together in a "gang". The mesh size is measured from knot to knot of a single, diagonally stretched mesh. Each mesh size is generally selective for certain size fish (Table 2.3), therefore, the individual panels used in the net have been chosen so the net is capable of catching a wide range of species and size classes across panels.

Table 2.3 The standard order of the panels based on mesh size, the corresponding filament size used in the construction of the net and the mean fork length of the fish typically caught by each of the mesh sizes.

Panel Order	Mesh Size (mm)	Filament Size (mm)	Mean Fork Length (mm)
1	25	0.20	141 mm
2	76	0.25	289 mm
3	51	0.20	225 mm
4	89	0.30	336 mm
5	38	0.20	203 mm
6	64	0.25	298 mm

Gill nets were fished at 33 sites in Seton Lake (24 nearshore and 9 offshore sets), and 33 sites in Anderson Lake (24 nearshore and 9 offshore sets). The distribution of sites spanned the three longitudinal zones (i.e., inflow, mid, and outflow sections) in both lakes (Figure 2.1). Set duration was different for nearshore versus offshore sets in Years 4 – 9 (2016 to 2021). This was due to the significantly more abundant catches and incidence of mortality observed for the nearshore sets compared to offshore sets in Year 3 (2015), most of which were fished overnight in that first year of gill netting. To mitigate the rate of mortality, starting in Year 4 the nearshore zone was fished using short-duration sets (target = 15 to 30 minutes) throughout the day so that the fish could be removed from the nets and processed much more quickly. Due to substantially lower catch rates, the offshore sets were still fished overnight (i.e., set late in the day and retrieved the following morning).

Nearshore nets were set perpendicular from shore. A length of rope connected one end of a sinking RIC gill net to a secure anchor point on shore (i.e., tree trunk) and ensured that the shallow end of the net was deployed in an adequate depth of water (>2 m) for proper net deployment. The net was deployed off the bow of the boat as it was operated at slow speed away from the shoreline in reverse. A concrete anchor was attached to the lead line at the offshore end to hold the net in place and align it with the slope of the lake bottom. A line with a large orange buoy was attached between the anchor and the surface to facilitate net retrieval. Panel order was generally alternated between nearshore sets (i.e., panel 1 vs. panel 6 nearest to shore).

Offshore nets were set parallel to the longitudinal axis of the lake where water column depths ranged from ~ 70 to 130 m in Seton Lake and ~ 60 to 125 m in Anderson Lake. At each location, three six-panel gangs of RIC nets were deployed in a row, each set at a different sampling depth between the surface and the thermocline (i.e., 0, 20, or 25 m from the surface to the top of each net). The height of the gill nets extended an additional 2.4 m below these depths. Once the crew was in position to begin deployment, a large concrete anchor was lowered off the front of the boat to the bottom of the lake and was connected by an adequate length of rope to a large orange buoy at the surface. The nets were deployed from the buoy using pre-measured dropper lines (attached to small foam floats) to control the sampling depth across the length of each net. Buoys were also deployed between each net gang, and another concrete anchor with buoy was deployed at the end of the third net. Flashing lights were deployed with each buoy for overnight sets to make them visible to boaters during the hours of darkness.

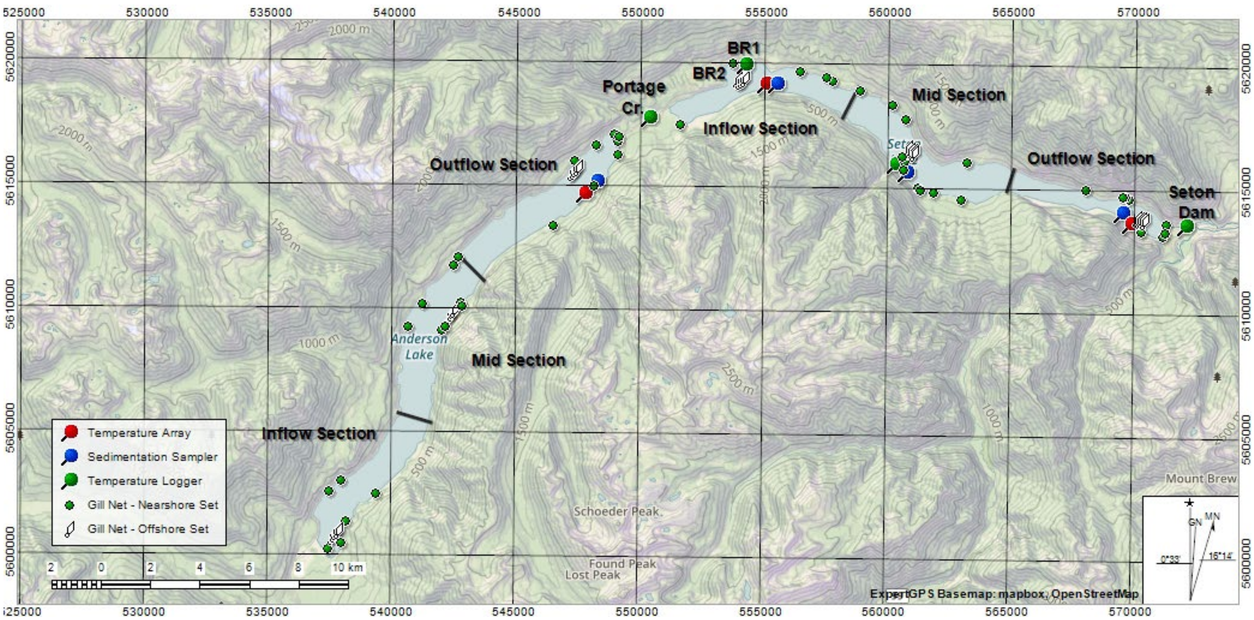


Figure 2.1 BRGMON-8 study area showing longitudinal sections and the locations of temperature arrays (red pins), individual temperature loggers (green pins), sedimentation samplers (blue pins), and fish sampling locations (green dots and white markers) in Seton and Anderson lakes during Year 9 (2021). The locations of the Bridge 1 and 2 (BR1, BR2) Generating Stations and Seton Dam are also shown. Note: the UTM coordinates for each gill net sampling location are provided in Appendix A.

Offshore nets were generally retrieved in the reverse order that they were deployed (end buoy to start buoy; 25 m net, 20 m net and then surface net – unless a change of wind direction dictated otherwise). Nearshore nets were retrieved from the offshore buoy end towards shore. Fish were removed from the nets as they were retrieved and placed into separate holding containers for each gill net panel. Each container of fish was labelled with the net identifier and panel number which were subsequently recorded on the catch data sheets for each captured fish. Bucket aerators were used to maintain oxygen levels for live fish until release. Following processing, fish mortalities were cut open to assess sex and returned to the lake near the point of capture.

All captured fish were identified to species, measured for length and weight, and evaluated for sex and sexual maturity (as possible); appropriate aging structures were collected from a subset of fish for target species (see Section 2.3 for more information). Bull trout and rainbow trout that were in good condition, and could be released alive, were marked with PIT tags to facilitate identification of any recaptures during subsequent surveys. Gwenis were not marked as the majority were mature spawners that would die following their spawning period. Stomach content samples and otoliths were opportunistically collected from bull trout that had succumbed to the sampling. Additional data recorded at each sampling location included habitat type (nearshore or offshore), net orientation (parallel or perpendicular to shore), set and retrieval dates/times for the nets, set depths (at both ends of each net), first and last panel mesh sizes, UTM coordinates, water temperature and secchi depth.

By accruing fish size and age data across monitoring years, we were able to fit growth curves to the median values for bull trout from both Seton and Anderson Lakes using the von Bertalanffy growth equation:

$$L_t = L_\infty(1 - e^{-K(t-t_0)})$$

Where:

L_t = Length at Age $_t$

L_∞ , the "asymptotic length" = 977 (Seton); 721 (Anderson)

K, the "curvature parameter" = 0.078 (Seton); 0.116 (Anderson)

t_0, the "initial condition parameter" = -2.53 (Seton); -2.07 (Anderson)

2.3. Laboratory Analysis

To assist in developing an understanding of the life history and age class structure of the target resident fish populations in Seton and Anderson lakes, fish sampling included collection of age structures (i.e., scales, fin rays and otoliths) from captured fish. Approximately five to ten scales were collected from selected gwenis and rainbow trout from the preferred area above the lateral line and immediately behind the dorsal fin. Pectoral fin rays were collected from all captured bull trout and otoliths were additionally collected from any bull trout mortalities to provide a secondary ageing structure. The ageing structures were placed in coin envelopes marked with appropriate data for cross-reference with the data sheets.

Ageing analysis was conducted on the scale samples by Angela Ratzburg and Dani Ramos-Espinoza (Instream Fisheries Research). After a period of air-drying, scales were pressed under heat to transfer precise images onto soft plastic strips by a St'at'imc Eco-Resources technician (Bailee Phillips). The images were magnified using a microfiche reader following the methods of Mackay et al. (1990). Processing and age-reading for fin ray and otolith samples was completed by Mike Stamford (Stamford Environmental). After a period of air-drying, the fin ray samples were trimmed, set in epoxy, and cut into transverse cross-sections. The sections and otoliths were polished using 400 to 1200 grit wet-dry sandpaper and then affixed to a microscope slide for reading. Images of age structures, from which age reading was completed, are available upon request.

2.4. Data Management

All field data collected for this project were recorded into field notebooks or on standardized datasheets specifically developed for this program. A standardized data entry template was developed in MS Excel, and all data entry was conducted by SER technicians. Data quality assurance (QA) checks were completed by the Project Manager/Biologist (Jeff Sneep).

All entered data were compiled into an active Microsoft Excel (for Microsoft 365) database that already included the data from years 1 to 8 of this monitoring program. As this program proceeds, this database will: facilitate data sharing between monitoring programs; continue to be updated each year as new data are collected and entered; and be stored in multiple locations (i.e., office computer, external hard drive, and online storage such as Microsoft “OneDrive”). All data and document files have been backed up to ensure data security and integrity.

3. Results

3.1. Physical Conditions

BC Hydro Operations

Some of the operational differences observed among years, and described in the text that follows, were due to identified system constraints (see general description in Section 1.4). For instance, a number of the BR1 and BR2 units had been “de-rated” during the earlier BRGMON-8 study years (i.e., 2013 to 2015) which constrained the diversion volume that could be run through the generating units in those years (Matt Casselman, BC Hydro, pers. comm.). From 2016 to 2021, works to upgrade the generating units at the Bridge generating station coupled with reduced storage in Downton Reservoir contributed to higher diversion volumes, particularly during spring in those years. Once the capital improvements at BR1 and BR2 are complete, the units will be operational at their rated capacities and the constraints on diversion conveyance will be alleviated.

The pattern of BR1 and BR2 discharges in 2021 were higher than the Modified Operations averages in all seasons except summer and there were only two extended (i.e., ≥ 24 hr) outage periods (Figure 3.1). For total differences among seasons, refer to Table 1.1 in Section 1.4. Peak diversion discharges in Year 9 (2021) were between 120 $m^3 \cdot s^{-1}$ and 130 $m^3 \cdot s^{-1}$ on a few dates in April and December, which were lower than peak rates in 2020 (i.e., 148 $m^3 \cdot s^{-1}$) but on par with the highest rates in all other monitoring years. Discharges varied between maximum and minimum values on only a few dates as the units were cycled between on and off. However, overall, the amount of within-day cycling throughout the year from 2015 to 2021 was much lower than what was implemented in the earlier study years (i.e., 2013 and 2014; Sneep 2015).

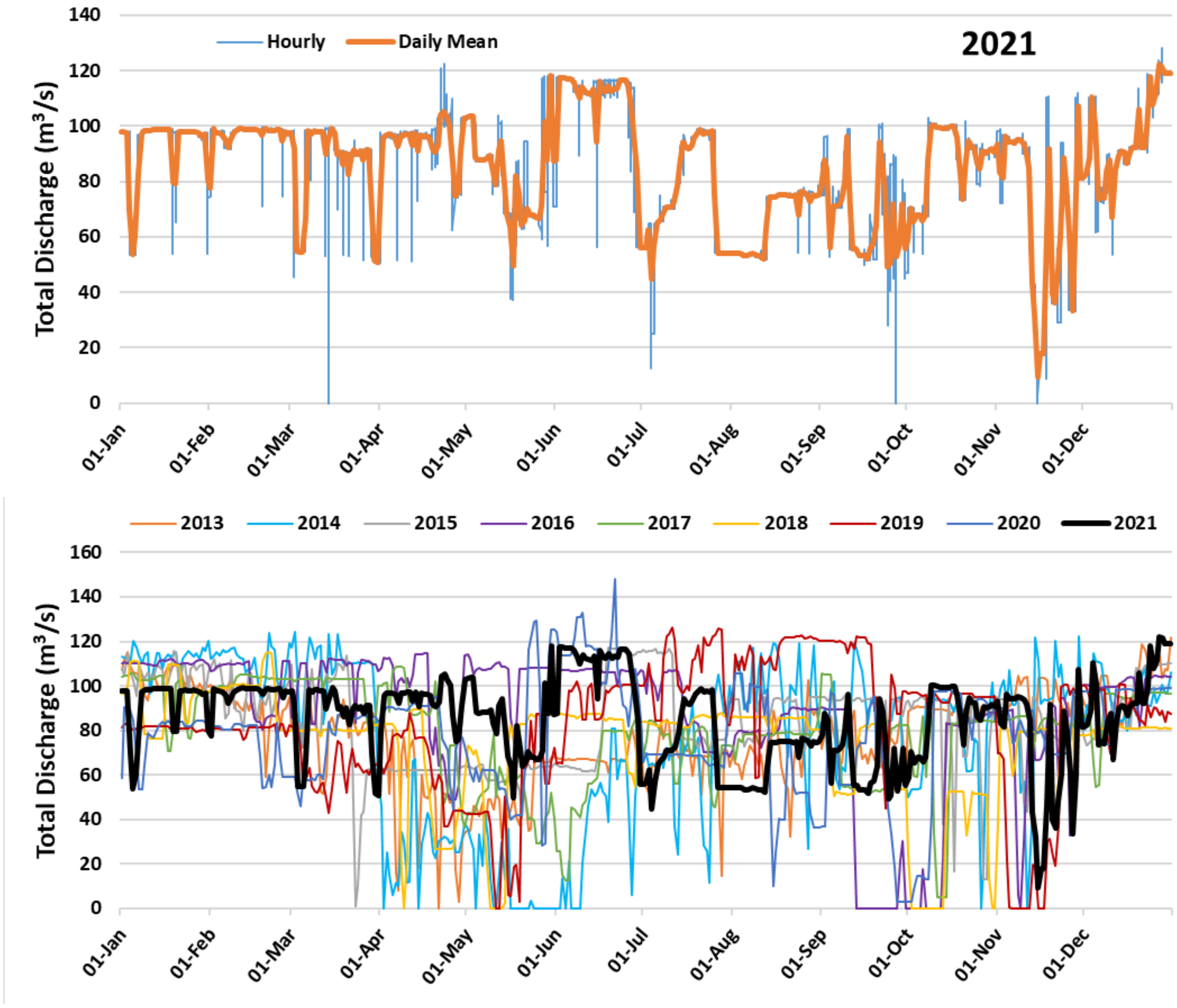


Figure 3.1 Total Discharges from the Bridge 1 and Bridge 2 Generating Stations into Seton Lake from January to December. Hourly and daily means for Year 9 (2021) are shown in the upper plot, and daily means for each monitoring year (2013 - 2021) are in the lower plot.

A summary of "outages" (or shutdowns of the generating units) among study years is provided in Table 3.1 for comparison. Since outages can be brief (i.e., <1-hour duration), they are not always reflected as a zero value in the hourly discharge record. For this reason, we conservatively tallied the number of hours per season and per year when mean hourly discharge was <20 $m^3 \cdot s^{-1}$ (combined for all generating units) to reflect the relative number of hourly periods that included a shutdown (or near shutdown) of all units. The total number of station outage-hours at BR1 and BR2 were markedly reduced starting in Year 3 (2015 to 2021 range = 197 to 765 hours) relative to Years 1 and 2 (range = 1084 to 1,814 hours) (Table 3.1). The number in 2021 (*n*= 88) was the lowest documented to-date. On a seasonal basis, the highest number occurred during fall (*n*= 83). As in Years 3, 5, 6, 7 and 8, shutdowns were very minimal during the summer of Year 9, and were virtually non-existent in winter, similar to Years 2, 4, 5, 6, 7 and 8. Outages of ≥24-hour duration in 2021 (*n*= 2) were at the low end of the range across years (e.g., 1–29), and all occurred in fall.

Table 3.1 Summary of the number of hourly "outages" (periods when mean hourly discharge was <20 $m^3 \cdot s^{-1}$ combined total for all generating units) and ≥24-hour outages (shown in brackets) by season and study year.

Study Year	# of Hourly Outages (and ≥24-hour Outages) by Season				
	Spring	Summer	Fall	Winter	All Seasons
1 (2013)	430 (1)	532 (0)	26 (0)	96 (0)	**1,084 (1)**
2 (2014)	1,252 (27)	316 (0)	241 (2)	5 (0)	**1,814 (29)**
3 (2015)	66 (1)	2 (0)	104 (2)	64 (0)	**236 (3)**
4 (2016)	0 (0)	182 (7)	582 (22)	1 (0)	**765 (29)**
5 (2017)	75 (2)	0 (0)	121 (4)	1 (0)	**197 (6)**
6 (2018)	212 (6)	0 (0)	404 (14)	1 (0)	**617 (20)**
7 (2019)	181 (4)	21 (0)	337 (11)	1 (0)	**540 (15)**
8 (2020)	30 (0)	25 (0)	311 (12)	1 (0)	**367 (12)**
9 (2021)	0 (0)	4 (0)	83 (2)	1 (0)	**88 (2)**

The effects of varying inflow volumes, as well as year-specific management decisions and operating system constraints on Carpenter diversion operations were also reflected by the comparison of cumulative diversion discharge by season and year (Figure 3.2 below, and Table 1.1 in Section 1.4), which highlighted that the greatest proportion of water was released in late spring (i.e., June), with closer to average proportions in summer, fall and winter in Year 9 (2021). Based on a comparison among years, the total discharge volume in 2021 (2,674 million m^3) was between 4% and 13% higher than 2013, 2014 and 2017–2020 (i.e., 2,457, 2,415, 2,515, 2,335, 2,561 and 2,441 million m^3, respectively); nearly identical to 2015 (2,661 million m^3); and 3% lower than 2016 (2,765 million m^3).

Under the terms included in the Water Use Plan, the licensed operating range for Seton Lake is 0.60 m between 235.76 and 236.36 meters above sea level (masl; measured in the vicinity of BR1) to manage water storage for generation, fish habitat, and to reduce foreshore erosion rates (BC Hydro 2011). Assessment of surface elevations in Seton Lake among years has not revealed any obvious seasonal patterns (Figure 3.3). The total range of elevations is low relative to other reservoirs in the system (i.e., Carpenter and Downton); the most observed has been 0.60 m between minimum and maximum levels in 2018, due to a period of low lake levels in November and December that year (which were at the bottom end of the licensed range). The maximum *daily* rate of change observed has been between 13 and 26 cm for each study year. The lowest elevations in Year 9 were recorded in mid February (Week 7; 235.80 masl) and the highest was 236.36 masl on 16 May 2021 (Week 20), which was a span of 0.56 m across the year. Slight differences in elevation between reported values and the terms in the WUP may be due to differences between the current measurement location (forebay of Seton Dam) versus the compliance location (in the vicinity of BR1).

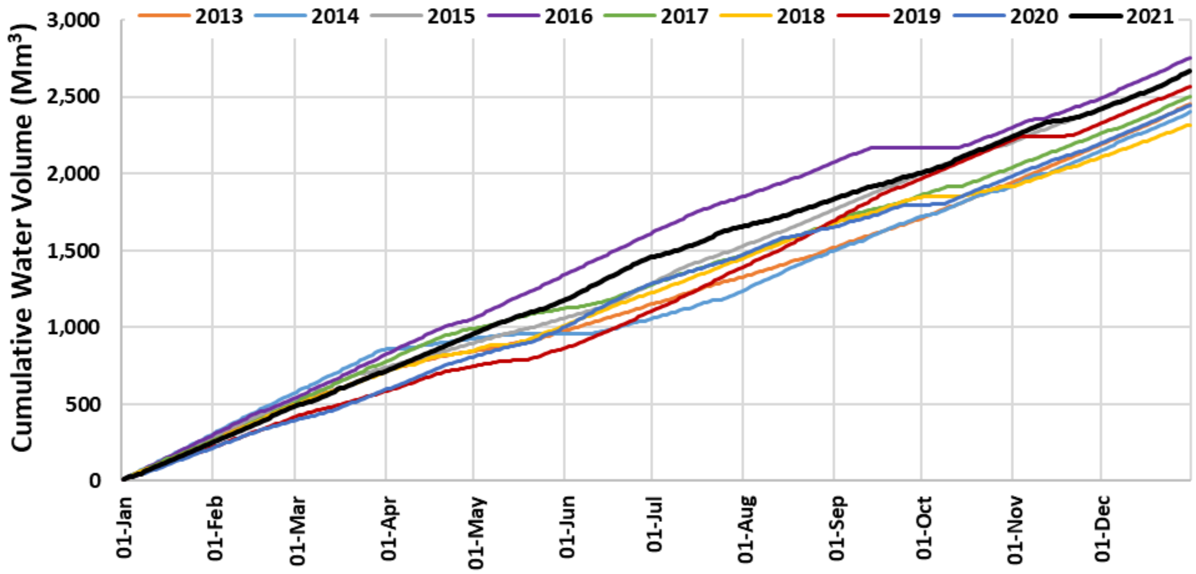


Figure 3.2 The daily cumulative inflow volume from the Bridge 1 and 2 Generating Stations, 2013 to 2021 (Monitoring Years 1 to 9).

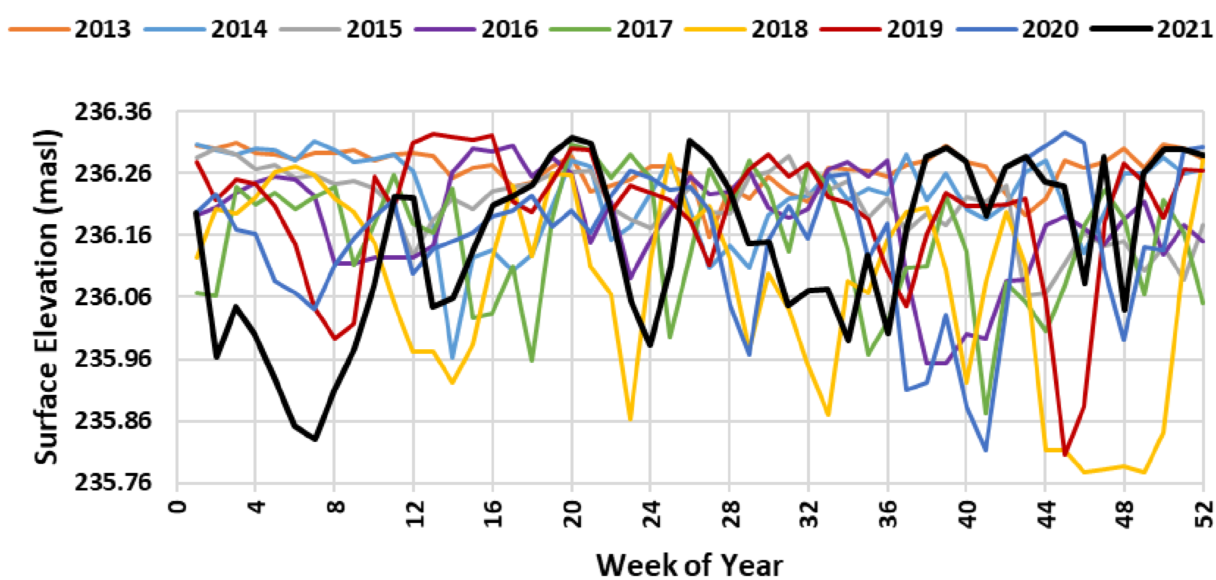


Figure 3.3 Mean weekly surface elevations of Seton Lake recorded in the forebay of Seton Dam across the year, 2013 to 2021. Note: the y-axis range spans the licensed operating range referenced in the Water Use Plan (BC Hydro 2011).

In terms of the range of elevations among years, Seton Lake levels were only below 236.1 masl (the approximate mid-point of the observed range) less than 10% of the time for study years 1 to

3 (2013 to 2015; i.e., the "normal" operations years). In Years 4 to 9 (2016–2021; i.e., years affected by "modified operations" in the system), the levels were below 236.1 masl approx. 17%, 33%, 49%, 14%, 30% and 37% of the time, respectively, due to an increased number or duration of "drawdown" events in those years. However, the range of lake surface elevations has been within the licensed range in all cases. Based on observed patterns of fish distribution (see Section 3.2), the range of variation in observed surface elevations was not expected to impact the target species for this monitoring program.

Thermal Profile Monitoring

Year 9 (2021) hourly water temperatures for the January to December period at a range of depths in the outflow end of Anderson Lake, and the inflow and outflow ends of Seton Lake, are displayed in Figure 3.4 (top to bottom panels, respectively). Patterns in the temperature profile data were very similar to those described for Years 4 to 8 (2016 to 2020). Since temperature monitoring for the BRGMON-8 program was initiated at the end of July 2015 (i.e., mid-way through Year 3), it was not possible to make comparisons with the first three monitoring years.

Consistent with the normal lacustrine process of thermal stratification, significant temperature differences developed among the various depths across the seasons at each array location in both lakes. At the start of the year (in mid winter) temperatures within each lake were consistent at all depths, reflecting isothermic conditions of between 3° and 6°C, which persisted until mid- to late April.

Typically beginning in late April or early May, the temperature profiles began to stratify. As expected, in both lakes the surface (or epilimnion) layer, had the highest degree of warming since it interfaces most directly with air temperatures and solar heating, relative to the deeper layers. However, maximum temperatures among the lakes, and across the length of Seton Lake, varied. Peak summer surface temperature was 23.5°C at the outflow end of Anderson Lake (on 30 July), 16.8°C at the inflow end of Seton Lake (on 12 August), and 22.5°C at the outflow end of Seton Lake (on 30 July). These were on par with the highest peak temperatures in previous monitoring years. Relative to the inflow end of Seton Lake, the greatest daily differences in surface temperatures among stations was up to 7.8°C at the outflow end of Anderson Lake and 7.3°C at the outflow end of Seton Lake in 2021. At each monitoring location, turn over in the fall (collapse of stratification) progressed across the month of November, as mixing among layers occurred and the lakes returned to a fully isothermic condition by early December.

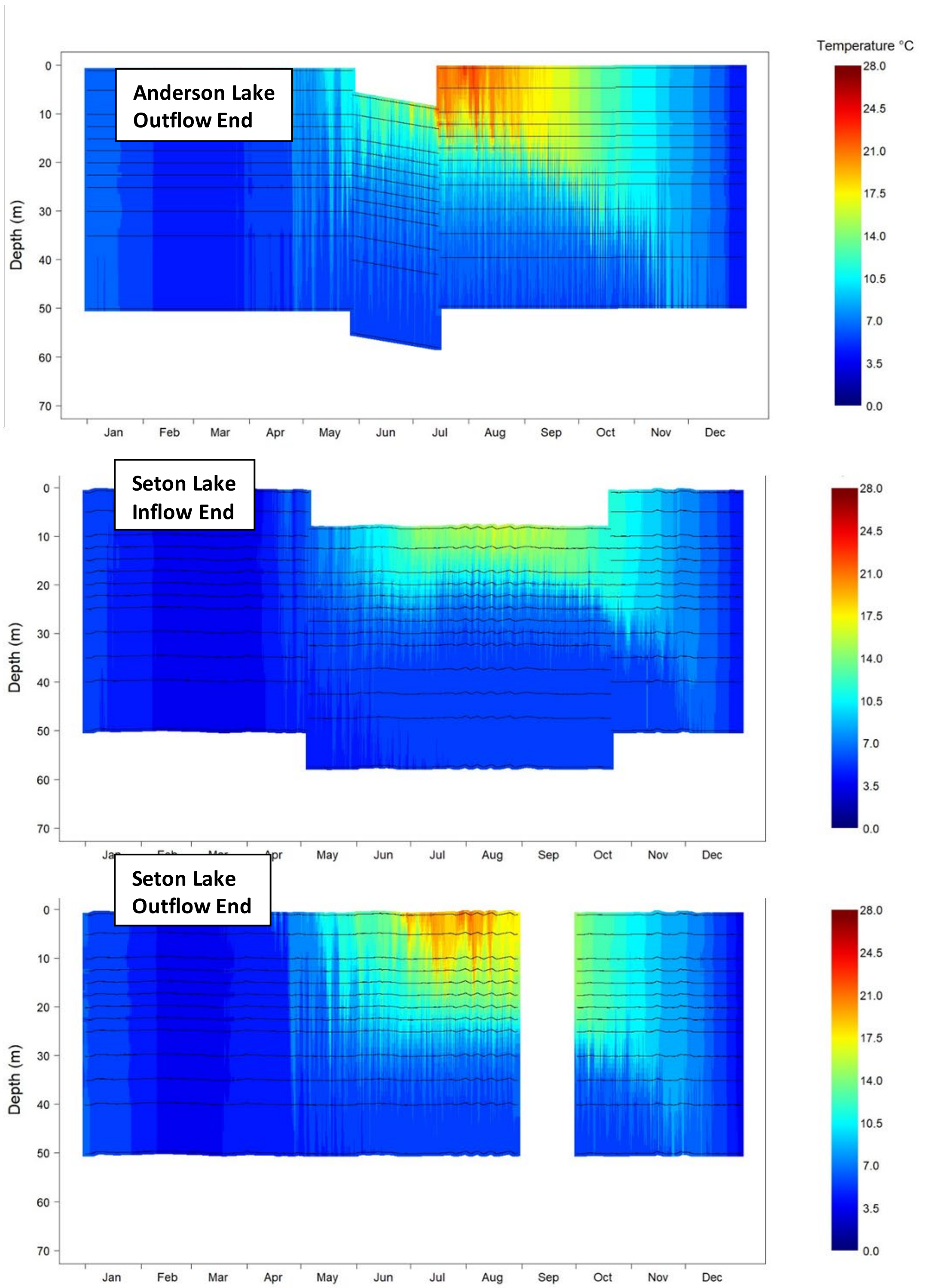


Figure 3.4 **Water temperature profiles recorded at the outflow end of Anderson Lake (upper plot) and at the inflow (middle plot) and outflow (lower plot) ends of Seton Lake from January to December 2021. The horizontal lines indicate the measurement depths. Temperatures between those depths were linearly interpolated.**

In both Seton and Anderson lakes the mid (or metalimnion) layer had the greatest temperature differential by increment of depth of any layer, particularly during the summer months when stratification was most established. However, the amount of warming and overall depth of this layer varied among array locations and between the lakes. The maximum differentials for the metalimnion were: 11.3°C (from 19.9° to 8.6°C between 15 m and 26 m depth) at the outflow end of Anderson Lake, 4.4°C (from 10.9° to 6.5°C between 15 m and 21 m depth) at the inflow end of Seton Lake; and 7.3°C (from 16.8° to 9.4°C between 15 m and 23 m depth) at the outflow end of Seton Lake. Temperatures in the deepest (hypolimnion) layer were the most stable, changing by ≤ 5°C across the entire year.

The temperatures of each thermal layer at the three monitoring locations were also summarized for the period when the lakes were stratified using a set of box-and-whisker plots (Figure 3.5). Based on these plots, the temperatures at the two ends of Seton Lake differed most substantially in the epilimnion and mesolimnion layers during summer, reflecting that the inflow end of the lake is typically cooler than the outflow end due to the influence of the cool diversion inflows on these layers. Median summer temperatures were 12.7°C and 17.2°C in the epilimnion, and 7.0°C and 12.9°C in the mesolimnion for the inflow and outflow ends of Seton Lake in Year 9 (2021), respectively. The temperatures of each layer became more consistent among locations in the fall: Median temperatures were 10.1°C and 9.9°C in the epilimnion, and 8.0°C and 9.8°C in the mesolimnion for the inflow and outflow ends, respectively. Temperatures in the hypolimnion were quite consistent among locations and seasons (i.e., median range = 4.8°C to 7.2°C). The temperature ranges for each thermal layer at the outflow end of Seton Lake were generally similar to the temperatures at the outflow end of Anderson Lake during each seasonal period.

As noted in past reports, the depths of the epilimnion, metalimnion, and hypolimnion layers have varied to some extent by location and among years, within the limits of precision based on the logger depth intervals (Table 3.2). In Year 9 (2021), the epilimnion depth (generally represented by the yellow and red colours in Figure 3.4) extended from the surface to ~15 m depth at all array locations. The metalimnion layer (generally represented by the extent of the turquoise colour in Figure 3.4) was thickest in Anderson Lake, spanning ~11 m (from ~15 m to ~26 m below the surface), whereas it was narrower in Seton Lake (spanning ~6 m and ~8 m below the depths of the epilimnion layer at the inflow and outflow ends, respectively). The top of the hypolimnion layer was slightly shallower at the inflow end of Seton Lake (>21 m depth) than at the outflow end of either lake (>23 m or >26 m depth). Many of these differences were consistent across the period of thermal stratification (particularly between June and October), which likely reflect effects of the colder diversion inflows and possibly differences in the aspect of the longitudinal axis of each lake (southwest to northeast for Anderson Lake, and east to west for Seton Lake) to some extent.

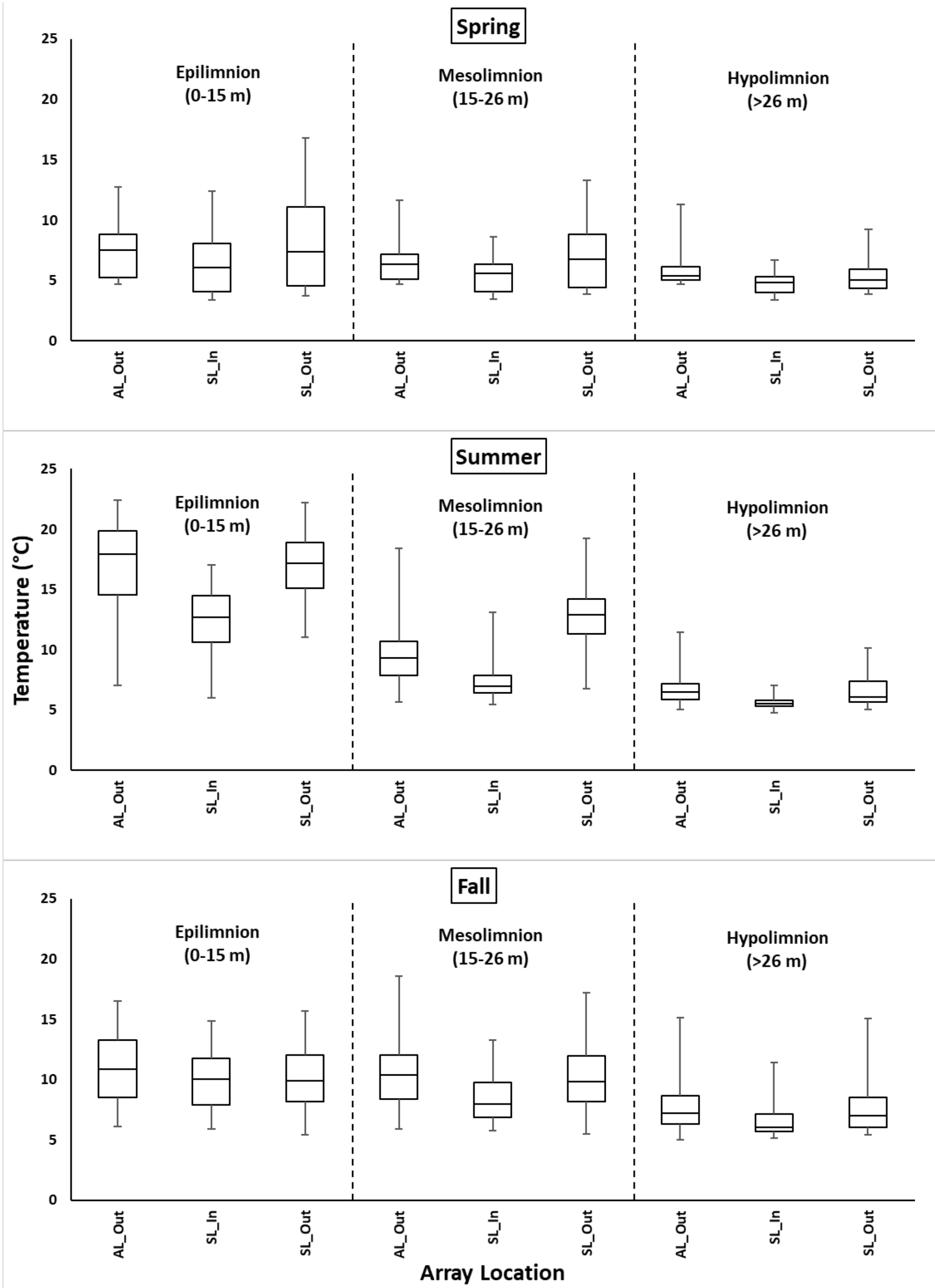


Figure 3.5 **“Box-and-whisker” plots representing water temperatures in the epilimnion (surface), mesolimnion (thermocline), and hypolimnion (deep) layers at the outflow end of Anderson Lake (AL_Out) and the inflow and outflow ends of Seton Lake (SL_In and SL_Out). The boxes are bounded on the top by the 75th percentile, and on the bottom by the 25th percentile. The median divides each box. The whiskers represent the minimum and maximum values.**

There was also evidence in the temperature profile data of seiche activity (i.e., temperatures within the epilimnion and metalimnion bouncing up and down on a near-daily basis). This was apparent as vertical lines in the profiles that showed warmer water episodically extending deeper and resulted in "fuzzy-looking" rather than distinct transitions between the thermal layers in Figure 3.4. These lines extended deepest at the outflow end of Anderson Lake and were more evident at the outflow end of Seton Lake than the inflow end. This was likely because the typical afternoon "Katabatic" winds generally blow down the valley (i.e., from the inflow end to the outflow end). Because of this typical wind direction, fetch (i.e., the horizontal distance over which wave-generating winds blow) is greatest at the outflow end of each lake.

Table 3.2 Summary of depths (in meters) for the epilimnion, metalimnion and hypolimnion at each monitoring location in Anderson and Seton lakes during the period of thermal stratification (Spring, Summer and Fall) in Years 3 to 9 (2015-2021). Note: data were not available for Anderson Lake in 2019.

Season	Epilimnion			Metalimnion			Hypolimnion		
	SL Inflow	SL Outflow	AL Outflow	SL Inflow	SL Outflow	AL Outflow	SL Inflow	SL Outflow	AL Outflow
2015	0-15	0-20	0-20	15-28	20-28	20-30	>28	>28	>30
2016	0-20	0-23	0-20	20-25	23-29	20-30	>25	>29	>30
2017	0-13	0-21	0-17	13-19	21-29	17-30	>19	>29	>30
2018	0-20	0-20	0-18	20-28	20-28	18-28	>28	>28	>28
2019	0-20	0-20		20-28	20-28		>28	>28	
2020	0-17	0-18	0-20	17-24	18-25	20-29	>24	>25	>29
2021	**0-15**	**0-15**	**0-15**	**15-21**	**15-23**	**15-26**	**>21**	**>23**	**>26**

Water temperatures were also monitored for various Seton Lake inflows, including Portage Creek, the BR1 tailrace, M'sut Creek, as well as the Seton Dam forebay (Figure 3.6). Surface temperatures at the inflow and outflow ends of Seton Lake (i.e., Inflow 1 m and Outflow 1 m) are also included on the figure for reference. Water temperatures in Portage Creek warmed from ~5°C at the end of March to a peak of ~20°C between mid July and early August. Portage Creek draws from the surface at the outflow end of Anderson Lake and receives cooler inflows from a couple of smaller tributaries (i.e., Whitecap Creek and Spider Creek in Seton Portage) before flowing into the western end of Seton Lake. Temperature inputs from the diversion (i.e., BR1 tailrace) were typically between 1° and 8°C cooler than the Portage Creek inflows, according to season. The differences in the temperatures of the diversion inflows (i.e., relative to Portage Creek), which contribute the largest volume to Seton Lake by far, account for the routinely cooler temperatures at the inflow end of Seton Lake compared to the outflow ends of both Anderson and Seton lakes, which are shown on Figure 3.4 and described in the paragraphs above. M'sut Creek temperatures were the coolest of any source and discharge into the middle section of Seton Lake, but the flow volumes from this creek are very small relative to the diversion inputs.

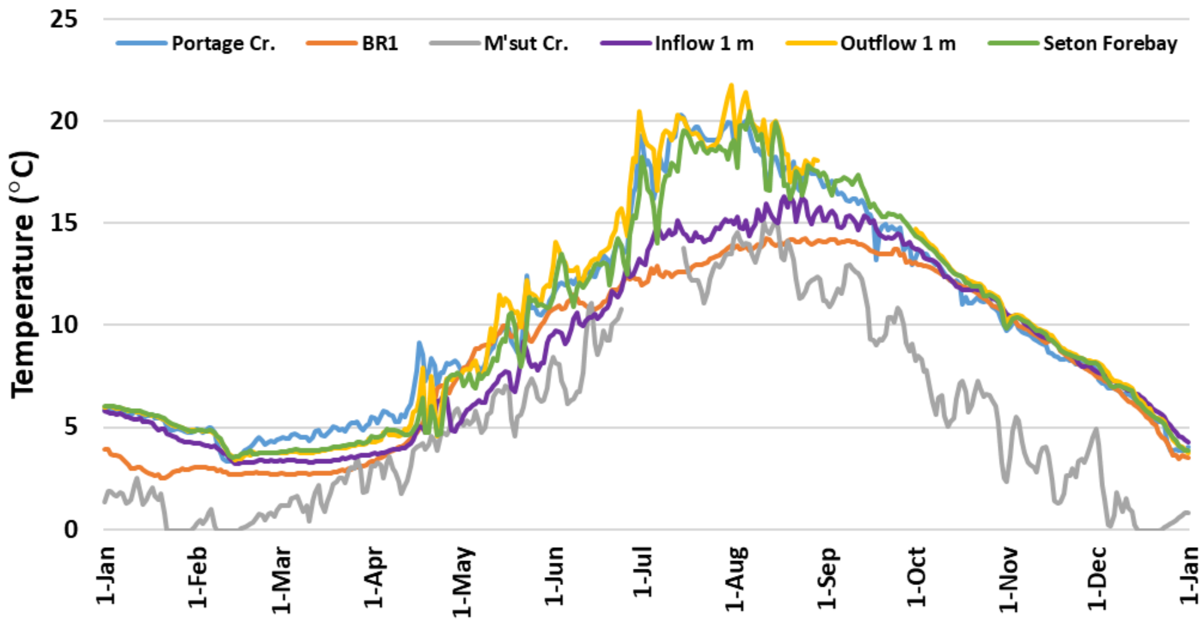


Figure 3.6 Temperatures of various inflow sources to Seton Lake (Portage Cr., BR1 and M'sut Cr.), as well as in-lake surface temperatures (Inflow 1.0 m and Outflow 1.0 m) and the Seton Dam forebay (Seton Forebay) in Year 9 (2021).

During the summer months, epilimnion and metalimnion temperatures warmed across the length of Seton Lake, such that temperatures at the Seton Dam forebay (drawn from the surface at the outflow end of Seton Lake) were generally a similar magnitude and followed the same trajectory across this period as Portage Creek (drawn from the surface at the outflow end of Anderson Lake).

Sedimentation Monitoring

As reported for previous study years (i.e., 2016 to 2020), the rate of sedimentation (mg dry weight accumulated per day) in 2021 was consistently highest at the inflow end of Seton Lake (mean = 62 to 167 mg/day, according to season), which is closest to the diversion inputs, and generally lowest in the Anderson Lake samples (mean = 8 to 85 mg/day, according to season), which are outside the influence of the Carpenter diversion (Figure 3.7). The spring sample in Anderson Lake was much larger than usual (i.e., relative to previous years) which may have been caused by a period of extremely high sediment-laden inflows related to a record high temperature event (i.e., the "heat dome") that formed over the region in late June and early July 2021. Higher inflows from this record weather event may have affected the spring samples at the other monitoring locations to some extent as well.

The rate of sedimentation in Seton Lake tended to diminish from the inflow to the mid section, and further from the mid to the outflow section. Mean rates in the mid section ranged from 31 to 105 mg/day, and in the outflow section from 18 to 47 mg/day, according to season. This sedimentation gradient was generally maintained across each sampled season; however, the outflow sample was larger than the inflow sample in Seton Lake during fall 2021. This may have been caused by a large landslide event that occurred in summer 2021 on the north shore of Seton Lake a short distance from the outflow sampler.

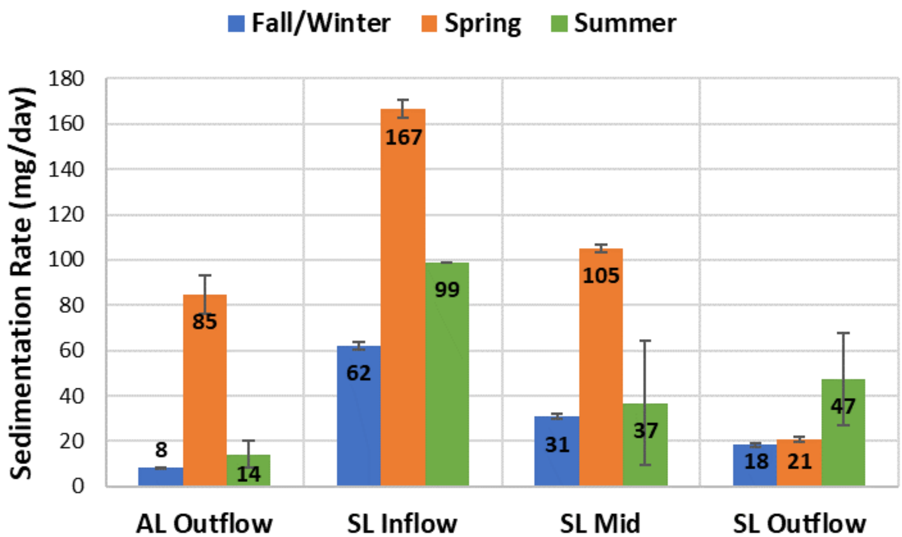


Figure 3.7 Mean sedimentation rate by sampler location (lake/longitudinal zone) and season during Year 9 (2021). AL = Anderson Lake; SL = Seton Lake. The error bars represent ± 1 SD among the replicates for each sample. Fall/Winter samples were collected from October 2020 to May 2021; Spring samples were collected from May to July 2021; and Summer samples were collected from July to October 2021.

Measured sediment accumulations have varied for each season and location among years, particularly for the samplers in Seton Lake (Table 3.3). These differences are due to a combination of factors among years, including: natural inputs (reflected by the relative differences in Anderson Lake values), diversion volumes, and Carpenter Reservoir elevations. These factors are explored further in the tables and figures below.

In all Seton Lake sections, the rates of sedimentation were greatest in spring, corresponding with high water-diversion volumes (i.e., 734 Mm^3 in 2021; see Table 1.1 in Section 1.4) and lowest drawdown elevations in Carpenter Reservoir (i.e., the source of the diversion flows; 2021 minimum = ~621 masl) during that sampling period. Relative to spring, summer and fall/winter sedimentation rates were lower in each lake section when Carpenter diversion volumes were lower (summer = 583 Mm^3 and fall/winter = 586/591 Mm^3) and Carpenter elevations were higher.

Table 3.3 Summary of sediment accumulations at each monitoring location (AL = Anderson Lake; SL = Seton Lake) by study year and season sampled.

Study Year	Location	Sediment Accumulation - mg dry weight per day (± SD)			
		Fall/Winter	Spring	Summer	Mean
2016	AL Outflow	- [a]	6 (0)	-	-
	SL Inflow	-	133 (1)	66 (1)	-
	SL Mid	-	76 (1)	-	-
	SL Outflow	-	56 (2)	28 (0)	-
2017	AL Outflow	10 (0)	36 (1)	7 (0)	**14 (0)**
	SL Inflow	67 (0)	150 (8)	-	-
	SL Mid	38 (4)	103 (2)	77 (10)	**58 (5)**
	SL Outflow	25 (0)	66 (1)	19 (2)	**31 (1)**
2018	AL Outflow	13 (2)	5 (1)	7 (2)	**11 (1)**
	SL Inflow	66 (0)	114 (0)	83 (1)	**83 (1)**
	SL Mid	43 (4)	64 (4)	49 (3)	**49 (4)**
	SL Outflow	20 (0)	46 (3)	14 (2)	**28 (2)**
2019	AL Outflow	14 (2)	4 (1)	5 (2)	**11 (1)**
	SL Inflow	53 (2)	120 (2)	106 (1)	**77 (2)**
	SL Mid	34 (2)	68 (1)	66 (2)	**48 (2)**
	SL Outflow	16 (0)	34 (2)	43 (2)	**26 (1)**
2020	AL Outflow	23 (0)	9 (0)	8 (2)	**14 (1)**
	SL Inflow	51 (1)	90 (13)	50 (0)	**64 (5)**
	SL Mid	-	62 (0)	42 (1)	**52 (1)**
	SL Outflow	27 (3)	44 (7)	19 (0)	**30 (6)**
2021	AL Outflow	8 (0)	85 (8)	14 (6)	**36 (5)**
	SL Inflow	62 (2)	167 (4)	99 (0)	**109 (2)**
	SL Mid	31 (1)	105 (2)	37 (27)	**58 (10)**
	SL Outflow	18 (1)	21 (1)	47 (20)	**29 (7)**

[a] "-" indicates sample not available for this period.

To supplement the analysis by season, a comparison of sediment accumulation (measured as dry weight of sediment in each sample) in Seton Lake with the total volume of water diverted from Carpenter Reservoir during each sampling interval was generated (Figure 3.8). Based on the data available to-date, the accumulation of sediments is positively correlated with diversion volume, and the slope of the regressions varied according to longitudinal zone (Table 3.4). The highest slope for this relationship was for the sampler nearest the diversion inflow and was lower in the middle and outflow sections of Seton Lake. In fact, relative to the mid and outflow sections, the slope was 1.9 and 2.8 times greater at the inflow end, respectively. In other words, the accumulation increases at a greater rate with diversion volume nearest the inflow than it does at further longitudinal distances down the lake.

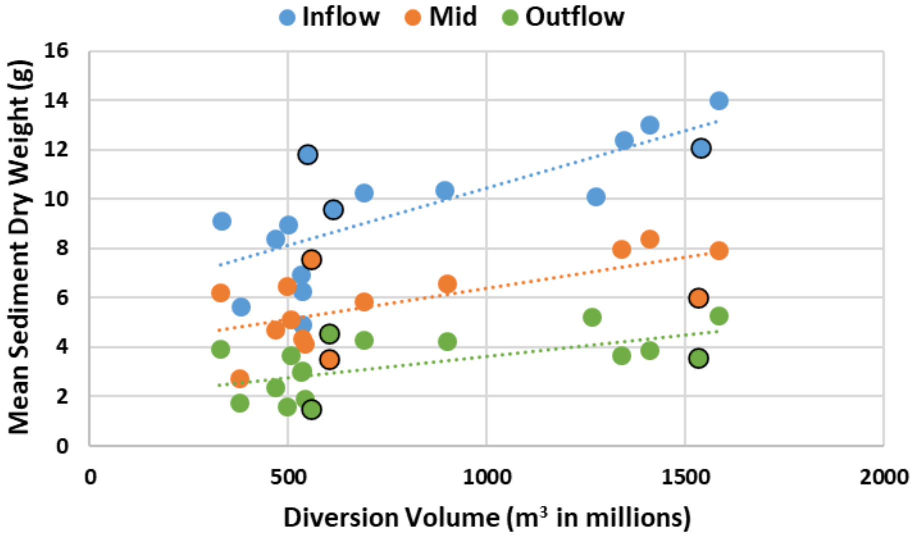


Figure 3.8 **Sediment accumulation in each longitudinal zone (i.e., inflow, mid, and outflow) of Seton Lake according to diversion volume from Carpenter Reservoir based on Years 4 to 9 (2016–2021) results. Points with black border represent 2021 data; Plain points are data from previous years. Linear regressions are shown (see Table 3.4 for regression parameters).**

Table 3.4 **Sample size (*n*), slope, y-intercept and R^2 values for the relationship between discharge from the BR1 and BR2 Generating Stations and sediment accumulation for each longitudinal zone of Seton Lake, based on data from all available study years, 2016 to 2021.**

Longitudinal Zone of Seton Lake	n	slope	y-intercept	R^2
Inflow	16	0.0047	5.802	0.59
Mid	15	0.0025	3.850	0.42
Outflow	17	0.0017	1.909	0.41

As acknowledged above, Carpenter Reservoir elevations are another aspect of BC Hydro operations that may influence sediment transfer to Seton Lake from the diversion. When Carpenter Reservoir elevations are low, sediments recruited from the upper portion of the Bridge River valley and mobilized within the drawdown area of Carpenter Reservoir may be more readily entrained by the diversion. Conversely, at higher Carpenter Reservoir elevations, a larger proportion of these sediments are likely deposited within the reservoir before reaching the diversion intakes.

Plotting sediment accumulation measured at each location in Seton Lake with the mean elevation of Carpenter Reservoir during each sampling interval, suggested that the correlations were negative (most strongly in the inflow section) (Figure 3.9); however, there was more scatter among the points (reflected by the lower R^2 values; Table 3.5) relative to the sedimentation versus diversion volume relationships. In other words, the measured sedimentation rates in Seton Lake may have been influenced by Carpenter Reservoir elevation, but this relationship is weaker, likely because diversion volume and the seasonal aspect of sediment transport are more significant drivers.

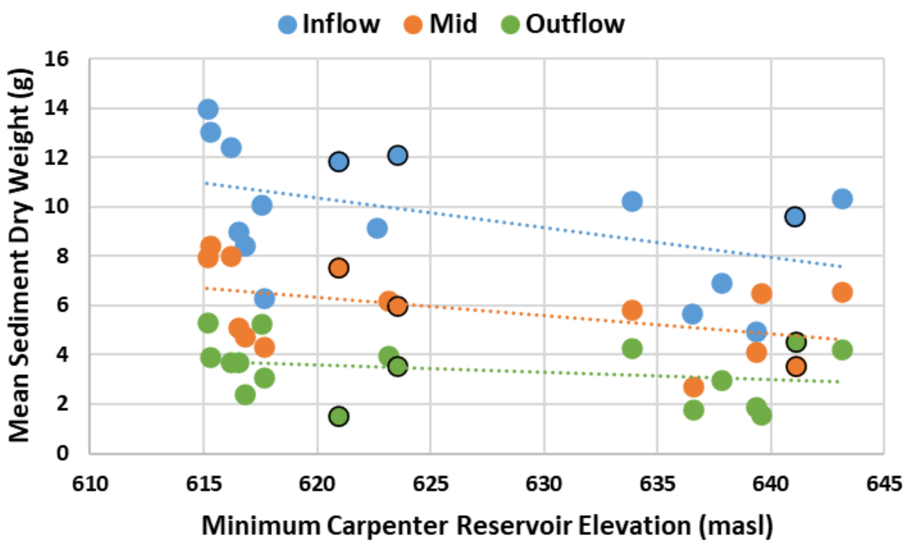


Figure 3.9 **Sediment accumulation in each longitudinal zone (i.e., inflow, mid, and outflow) of Seton Lake according to minimum Carpenter Reservoir elevations during each seasonal collection period based on Years 4 to 9 (2016–2021) results. Points with black border represent 2021 data; Plain points are data from previous years. Linear regressions are shown (see Table 3.5 for regression parameters).**

Table 3.5 **Sample size (n), slope, y-intercept and R^2 values for the relationship between minimum Carpenter Reservoir elevation and sediment accumulation during the collection period for each longitudinal zone of Seton Lake, based on data from all available study years, 2016 to 2021.**

Longitudinal Zone of Seton Lake	n	slope	y-intercept	R^2
Inflow	16	-0.1205	85.03	0.23
Mid	15	-0.0728	51.48	0.21
Outflow	17	-0.0295	21.86	0.07

It should be noted, however, that there is interaction between reservoir elevation and season in this context (i.e., Carpenter Reservoir elevations are consistently lowest in spring when sediment transport rates tend to be highest, and highest in late summer and fall when sediment transport tends to be lower). There has also been some interaction between diversion volume and season, but as noted in Section 3.1 – *BC Hydro Operations* (above) and summarized in Table 1.1 in Section 1.4, that interaction has been less consistent among the study years to-date.

Particle size analysis of the accumulated sediments and total organic content were assessed for the samples collected in Years 5 and 6 (2017 and 2018) which revealed that the samples in Seton Lake were consistently characterized by smaller particle sizes (i.e., >97% silt and clay) and lower organic carbon content (i.e., 1-2% total organic carbon) relative to the Anderson Lake samples. This was understood to reflect the difference in the primary source of sediment inputs among the two lakes (i.e., tributary for Anderson Lake and diversion inputs for Seton Lake) and confirmed that the samples were almost entirely composed of inorganic sediments. These analyses were not conducted on the samples in Years 7 to 9 (2019–2021) since the analyses were costly, were limited by minimum sample volume requirements, and because these results were not expected to change over time. For more information about these results refer to the annual report for Year 6 (2018) (Sneep 2019a).

Additional sedimentation rate data will be collected in the Year 10 (2022). Ideally, samples collected across the range of operational conditions (i.e., diversion release volumes and Carpenter Reservoir elevations) among seasons will continue to augment these relationships and strengthen the conclusions drawn from them by the end of the monitor.

3.2. Resident Fish Population Index Survey

A total of 529 fish were captured by gill netting during the annual resident fish index survey in Year 9 (2021; Seton Lake *n*= 393; Anderson Lake *n*= 136), including 9 resident species (Table 3.6). Target species made up 65% of the catch, and the other 35% were non-target species including: northern pikeminnow, peamouth chub, redside shiner, bridgelip sucker, mountain whitefish, and sculpins (in decreasing order of abundance). Thirty-three sites were sampled in Seton Lake, including 24 nearshore and 9 offshore sets; and 33 sites were sampled in Anderson Lake, including 24 nearshore and 9 offshore sets. The total sampling effort was 319.4 net-hours (Seton Lake = 159.9 net-hours; Anderson Lake = 159.5 net-hours), or just over 50 net-hours for each longitudinal section in each lake.

For target species, the 2021 catches of bull trout in Seton Lake (*n*= 5) and Anderson Lake (*n*= 21) were within the range of catches from previous years (range of *n*= 3–10 for Seton Lake and *n*= 12–38 for Anderson Lake). Catches of gwenis were similar to 2019 and 2020, which were all substantially lower than peak catches in 2018, primarily due to reduced catches in Seton Lake (2018 *n*= 801; 2019 *n*= 292; 2020 *n*= 196; 2021 *n*= 240). Catches of gwenis in Anderson Lake (2021 *n*= 35) were at the low end of the range of catches for this lake in recent years (range of *n*= 30–

73 from 2016 to 2020). Numbers of juvenile *O. nerka* in the 2021 catch (*n*= 26 overall) were also within the range observed in previous years. Increased catches of *O. nerka* in 2016 and 2017 likely reflected a stronger juvenile cohort (i.e., Age-1 and Age-2 fish) in those years that contributed to the high adult gwenis numbers in 2018. Other than in 2020, a higher proportion of the *O. nerka* catch has been consistently documented in Seton Lake for each year to-date. As reported previously, rainbow trout catches have generally been low each year, ranging from 0 to 3 fish in Seton Lake and 3 to 17 fish in Anderson Lake (2021 Seton Lake *n*= 3; Anderson Lake *n*= 12).

Table 3.6 Catch totals for all resident fish species from gill net sampling in Seton and Anderson lakes from Year 3 (2015) to Year 9 (2021). Gill net sampling was not conducted in Years 1 and 2 (2013 and 2014).

Study Year	Lake	Species [a]									
		BT	GW	ON	RB	MW	PMC	NSC	BSU	RSC	CC
3 (2015)[b]	Seton	10	563	20	2	2	77	49	27	46	14
	Anderson	12	11	2	3	3	36	23	3	3	2
	2015 Totals	**22**	**574**	**22**	**5**	**5**	**113**	**72**	**30**	**49**	**16**
4 (2016)	Seton	7	182	45		3	32	52	3	7	
	Anderson	37	30	11	8	3	5	42			
	2016 Totals	**44**	**212**	**56**	**8**	**6**	**37**	**94**	**3**	**7**	**0**
5 (2017)	Seton	7	349	36	1	3	38	35	17	14	
	Anderson	38	70	30	17	2	11	30	1	12	
	2017 Totals	**45**	**419**	**66**	**18**	**5**	**49**	**65**	**18**	**26**	**0**
6 (2018)	Seton	3	801	14	2		76	83	15	31	1
	Anderson	22	73	8	6	1	2	22	2	5	
	2018 Totals	**25**	**874**	**22**	**8**	**1**	**78**	**105**	**17**	**36**	**1**
7 (2019)	Seton	5	292	16	1	3	55	61	13	14	1
	Anderson	20	66	5	7		6	57	1	1	2
	2019 Totals	**25**	**358**	**21**	**8**	**3**	**61**	**118**	**14**	**15**	**3**
8 (2020)	Seton	5	196	13	3	1	35	92	14	105	
	Anderson	24	67	46	9		34	52		8	1
	2020 Totals	**29**	**263**	**59**	**12**	**1**	**69**	**144**	**14**	**113**	**1**
9 (2021)	Seton	5	240	22	3		40	62	9	11	1
	Anderson	21	35	4	12	2	10	46	3	3	
	2021 Totals	**26**	**275**	**26**	**15**	**2**	**50**	**108**	**12**	**14**	**1**

[a] Species codes: BT = bull trout; GW = gwenis; ON = *Oncorhynchus nerka* juveniles; RB = rainbow trout; MW = mountain whitefish; PMC = peamouth chub; NSC = northern pikeminnow; BSU = bridgelip sucker; RSC = redside shiner; CC = sculpin sp.

[b] Note: all nets were set overnight in 2015; and in Anderson Lake that year, only the outflow section was sampled.

Fourteen bull trout (Seton Lake *n*= 1; Anderson Lake *n*= 13) and six rainbow trout (all in Anderson Lake) were marked with PIT tags in 2021. Only fish that were alive and in robust condition were tagged. The total numbers of fish tagged during each monitoring year are summarized in Table 3.7. No fish that had been tagged during preceding study years (i.e., 2013 to 2020) were recaptured in Year 6 (2018), Year 7 (2019), Year 8 (2020) or Year 9 (2021). Original capture and recapture information for all recaptured fish to-date is summarized in Table 3.8.

Table 3.7 Total numbers of fish from target species marked with PIT tags during each monitoring year in Seton and Anderson lakes.

Study Year	Lake	# of PIT tags applied		
		Bull Trout	Rainbow Trout	Total
1 (2013)[a]	Seton	5	27	32
	Anderson	-[b]	-[b]	0
	2013 Totals	**5**	**27**	**32**
2 (2014)[a]	Seton	9	10	19
	Anderson	-[b]	-[b]	0
	2014 Totals	**9**	**10**	**19**
3 (2015)[b]	Seton	5	1	6
	Anderson	3	3	6
	2015 Totals	**8**	**4**	**12**
4 (2016)	Seton	2		2
	Anderson	28	6	34
	2016 Totals	**30**	**6**	**36**
5 (2017)	Seton	3		3
	Anderson	24	2	26
	2017 Totals	**27**	**2**	**29**
6 (2018)	Seton	2	1	3
	Anderson	13	3	16
	2018 Totals	**15**	**4**	**19**
7 (2019)	Seton	3		3
	Anderson	11	3	14
	2019 Totals	**14**	**3**	**17**
8 (2020)	Seton	3	1	4
	Anderson	6	2	8
	2020 Totals	**9**	**3**	**12**
9 (2021)	Seton	1		1
	Anderson	13	6	19
	2021 Totals	**14**	**6**	**20**
Grand Totals		**131**	**65**	**196**

[a] Fish were captured by boat electrofishing and angling in Years 1 and 2; Gill net sampling has been employed in every other monitoring year.

[B] Note: Anderson Lake was not sampled in Years 1 and 2; and in Year 3, only the outflow section of Anderson Lake was sampled.

Table 3.8 Summary of all inter-year bull trout recaptures to-date (as of Year 9 – 2021). Note: No marked fish were recaptured in 2018, 2019, 2020 or 2021, and no tagged fish from other species have been recaptured to-date.

Tag Code [a]	Original Capture Data			Recapture Data			Dist. (km)	Growth (mm/yr.)
	Date	Zone	FL (mm)	Date	Zone	FL (mm)		
888688	5-Oct-16	AL-Mid	445	4-Oct-17	AL-Mid	474	0.0	29
888710	6-Oct-16	AL-Mid	350	5-Oct-17	AL-Mid	396	0.0	46
888933	30-Sep-16	SL-Inflow	590	24-Sep-17	SL-Mid	683	8.8	93

[a] The prefix to each of these tag codes is: 900 226000

As reported for previous years, the highest catch-per-unit-effort (CPUE or catch rate) values for bull trout and gwenis were in nearshore nets in 2021 (Table 3.9). In Seton Lake, a substantially higher proportion of gwenis were sampled in nearshore sets likely because the survey timing corresponded with the start of spawning for that population. Based on catches, there were spawning locations within the range of the nearshore nets along the lake bottom (sampling extended down to a maximum of 56 m in Seton Lake and 60 m in Anderson Lake). It is possible that spawning or habitat use by this species also occurs beyond those depths in these lakes, but that was outside the reach of our sampling methods. In Anderson Lake, mature gwenis were primarily distributed in offshore (pelagic) habitats within or around the metalimnion thermal layer. Juvenile gwenis (*O. nerka* juveniles) and rainbow trout were only captured in offshore sets in Seton Lake but were each captured in nearshore and offshore sets in Anderson Lake. In general, mean CPUE values for *O. nerka* juveniles and rainbow trout have been low in both lakes (ranging from 0–6 fish/10 net-hours) relative to the catch rates for bull trout and mature gwenis every year.

An exception to this was the CPUE for *O. nerka* juveniles in the nearshore sets at the inflow end of Anderson Lake (CPUE = 208.3 fish/10 net-hours) in 2020 (Year 8). However, this value was biased high due to a single net catch of 25 *O. nerka* juveniles during a set duration of only 9 minutes (resulting in an individual net CPUE of 1,666.7 fish/10 net-hours for this set). This catch result highlighted the spotty or uneven nature of juvenile *O. nerka* distribution in these lakes.

Table 3.9 Summary of fish catch-per-unit-effort results for target species during the annual resident fish population indexing survey in Year 9, 28 September to 8 October 2021.

Location	Zone	Catch per 10 Net-Hours for Target Species							
		Bull Trout		Gwenis		*O. nerka* juv.[a]		Rainbow Trout	
		Nearshore	Offshore	Nearshore	Offshore	Nearshore	Offshore	Nearshore	Offshore
Seton Lake	Inflow	-	-	62.8	13.2	-	2.8	-	-
	Mid	-	-	100.4	2.8	-	0.6	-	0.6
	Outflow	2.7	0.8	240.3	9.3	-	1.0	-	-
Seton Lake Average		**0.9**	**0.3**	**134.5**	**8.5**	**0.0**	**1.5**	**0.0**	**0.2**
Anderson Lake	Inflow	21.0	0.4	-	2.7	3.4	-	3.4	0.6
	Mid	21.9	0.4	-	0.4	-	0.4	5.8	0.9
	Outflow	36.1	0.2	-	3.9	-	0.2	-	0.4
Anderson Average		**26.3**	**0.3**	**0.0**	**2.3**	**1.1**	**0.2**	**3.1**	**0.6**

[a] Values in these columns represent immature *Oncorhynchus nerka* that could not be differentiated between juvenile sockeye and gwenis in the field. Based on scale ageing results, juvenile *O. nerka* (Age-1 and Age-2) were all ≤188 mm in Seton Lake and <240 mm in Anderson Lake.

Bull trout distribution reflected the locations of their dominant food items in each lake (i.e., mature gwenis in Seton Lake; juvenile O. nerka and salmon eggs in Anderson Lake – see more on catches by depth and distance from shore, and bull trout stomach contents in the sub-sections that follow). As has generally been the case every year, rainbow trout catches were entirely at the surface (i.e., ≤2.5 m) in Seton Lake. Rainbow trout were also primarily near the surface in Anderson Lake in 2021, although a few were distributed down to 30 m in this lake. This wider range of depth distribution in Anderson Lake may be related to the differences in water clarity between the two lakes; however, it has been difficult to characterize patterns for this species due to persistent low catches each year.

In terms of the longitudinal sections, highest catch rates for gwenis were in nearshore habitats at the outflow end and in the middle section of Seton Lake, followed by the inflow section, in decreasing order (Figure 3.10, upper plot). Relative to the nearshore, gwenis were generally much less abundant in offshore catches in Seton Lake and trends among sections were much less apparent, especially considering the completely overlapping error bars. Catches of gwenis in Anderson Lake were much lower in each section than in Seton Lake. The differences among sections were also much smaller overall and there was substantial (or complete) overlap in error bars among sections. Juvenile *O. nerka* were also captured, but in smaller numbers than mature gwenis (likely due to the capture size limitations of the gill nets). They were marginally more abundant in the inflow section of Seton Lake (offshore habitat) and the inflow section of Anderson Lake (nearshore habitat) than in the other sections of each lake.

It was not possible to differentiate these juvenile *O. nerka* as gwenis vs. sockeye progeny in the field. However, the results of analyses included under BRGMON-6 provide a useful description of size classes based on scale ageing, and stock origin based on DNA analysis (Limnotek 2016). They determined a probability for each captured fish whether it belonged to one of three stocks:

Portage Creek sockeye, Gates Creek sockeye, or gwenis (total P=100%). Most of their juvenile fish (195/204) collected during a summer survey were identified to a specific stock with a >80% probability and no identified fish had <56% probability of belonging to that stock. According to their assessment, all of the *O. nerka* greater than 75 mm were identified as gwenis (Limnotek 2016). Since all of the *O. nerka* captured for the BRGMON-8 program to-date were ≥95 mm, all of these fish can likely be considered gwenis based on size (and have been included as such in the remainder of analyses for this report).

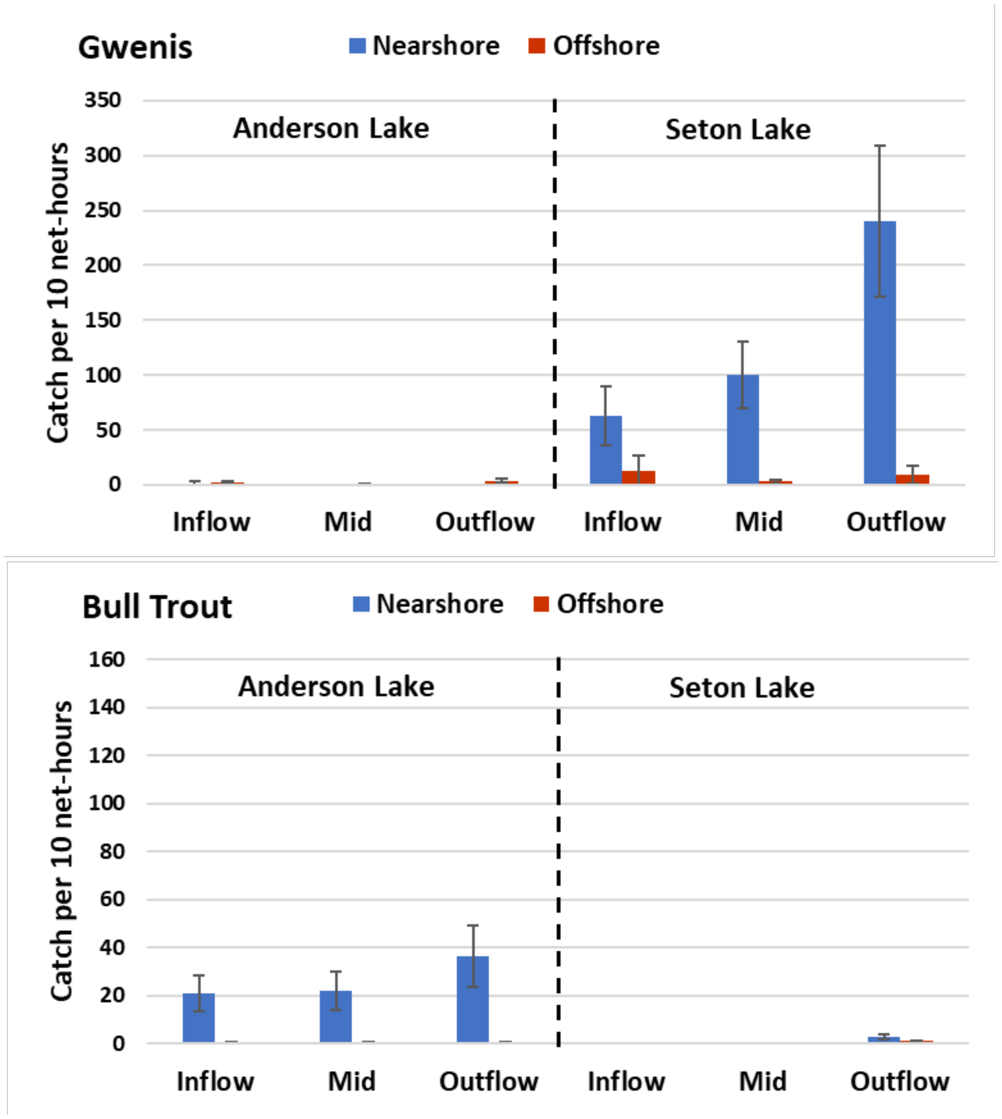


Figure 3.10 Catch-per-unit-effort summary for gwenis (top) and bull trout (bottom) comparing nearshore vs. offshore habitats in each longitudinal zone for Seton and Anderson lakes based on Year 9 (2021) sampling.

Between the two lakes, bull trout showed the opposite pattern of catch rates to gwenis (Figure 3.10, lower plot). CPUEs were lower in Seton Lake than in Anderson Lake, as has been the case every year. Nearly all of the catch for this species was in the nearshore sets in both lakes. In Seton

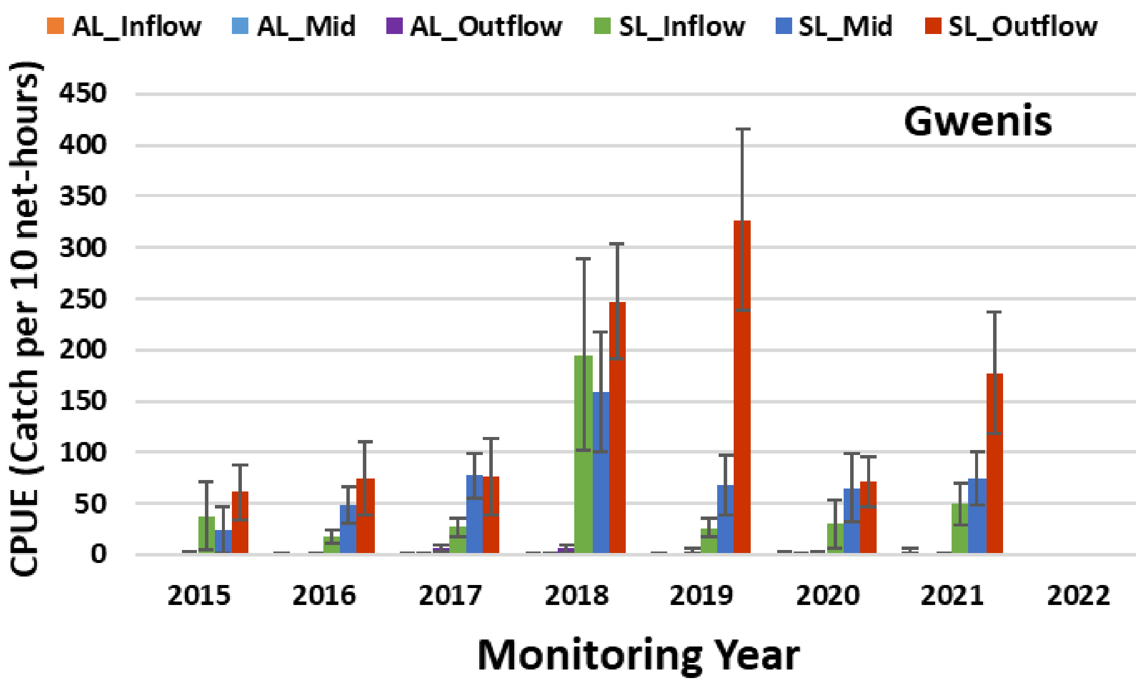
Lake, catches occurred solely in the outflow section. Bull trout were not captured in the inflow or mid sections of Seton Lake in Year 9 (2021). In Anderson Lake, catch rates for bull trout were highest in the outflow end; whereas the mid and inflow sections were slightly lower and equivalent with each other.

As in previous years, catches of rainbow trout in Seton Lake were low with only 3 of this species captured (435, 441 and 447 mm fork lengths; 2021 CPUE = 0.1 fish/10 net-hours). A total of 12 rainbow trout were captured in Anderson Lake (ranging from 204 to 460 mm; 2021 CPUE = 3.1 fish/10 net-hours in the nearshore and 0.6 fish/10 net-hours in the offshore). By longitudinal zone, rainbow trout were captured in all three sections of Anderson Lake but only in the mid section of Seton Lake in 2021. Among years rainbow trout have been captured in low numbers in each of the longitudinal sections of both lakes.

Due to the consistently low catches, particularly in Seton Lake, rainbow trout will likely not be assessed for trends in CPUE by monitoring year, lake section, or diversion inflows as performance metrics for this program. We will continue to collect data on the available rainbow trout and tag them, as possible, so we can identify any recaptures, as well as growth and movement between capture events. Furthermore, PIT tagging data from this program are being shared annually with the BRGMON-9 program to facilitate potential tracking of movements by this species (and bull trout) past Seton Dam and the generating station (i.e., entrainment).

A summary of CPUE values by longitudinal section of each lake, for the available monitoring years to-date, is provided in Figure 3.11 and Table 3.10. Fish sampling in Years 1 & 2 (2013 & 2014) was conducted by boat electrofishing around the perimeter of Seton Lake only. The catch rates for target species in those years were very low and not comparable with the gill netting method initiated in Year 3 (2015) and going forward, so those years were not included in the figure. It should also be noted that gill net sampling in 2015 employed overnight sampling for each set (nearshore and offshore), so the sampling events spanned the diel period of vertical migration for gwenis among the thermal layers, but also resulted in much higher rates of mortality (i.e., near 100%) that year.

Starting in Year 4 (2016), sampling in nearshore habitats (where highest catch rates tend to occur) was changed to short-duration sets in the interest of reducing the mortality rate (given that this sampling was planned for multiple consecutive years) and expanding the number of locations that could be sampled. This approach has been effectively implemented from 2016 to 2021. Going forward (i.e., Year 10), the plan is to maintain the sampling methods and effort to ensure the direct comparability of results across years, within the limits of our control.



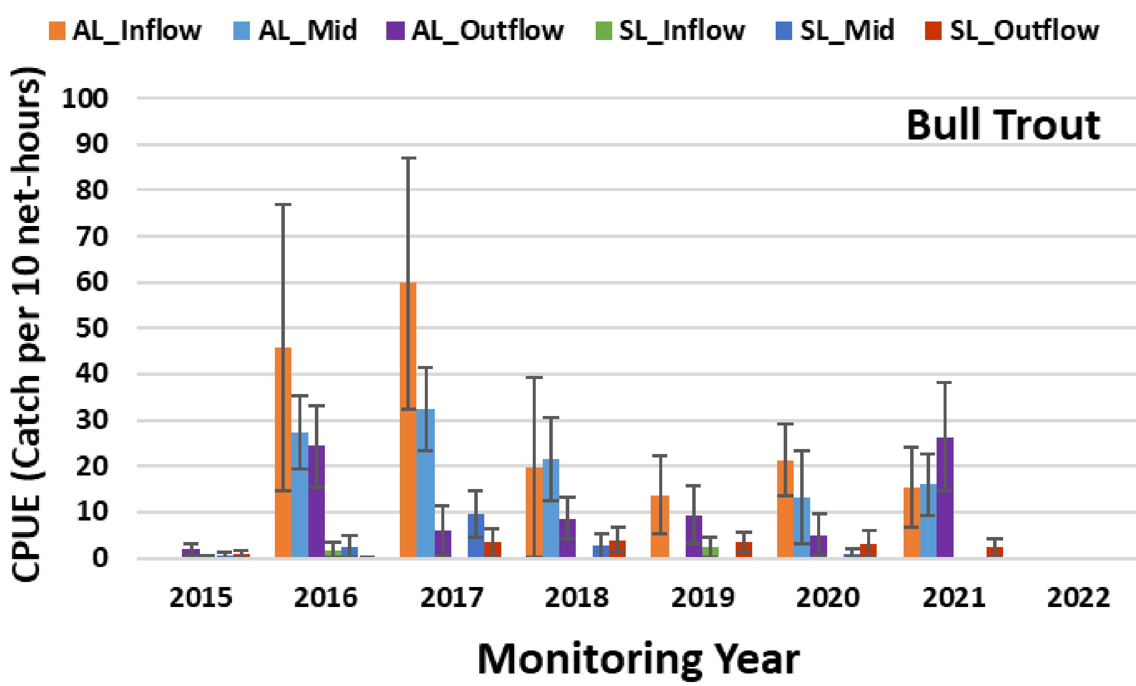


Figure 3.11 **Catch-per-unit-effort summary for gwenis (top) and bull trout (bottom) by longitudinal zone in Seton and Anderson lakes for each monitoring year from 2015 (Year 3) to 2022 (Year 10). Values for years 1 and 2 (2013 & 2014) were not applicable to this summary because sampling was done by boat electrofishing in those years. Anderson Lake inflow and mid zones were not sampled in 2015. Note the different y-axis scales between the two plots.**

Table 3.10 Summary of catch-per-unit-effort (±SE) for gwenis and bull trout by longitudinal zone in Seton and Anderson lakes for each monitoring year to-date. Values for years 1 and 2 (2013 & 2014) were not included – see text for rationale.

Species	Study Year	Anderson Lake						Seton Lake					
		Inflow		Mid		Outflow		Inflow		Mid		Outflow	
Gwenis	2015	-		-		**1.8**	(0.8)	**37.7**	(33.2)	**24.1**	(22.6)	**60.9**	(27.3)
	2016	**0.8**	(0.4)	**0.2**	(0.2)	**0.9**	(0.6)	**18.1**	(6.7)	**48.4**	(17.9)	**74.8**	(35.5)
	2017	**0.9**	(0.7)	**0.8**	(0.5)	**6.5**	(3.7)	**26.7**	(9.2)	**77.4**	(21.6)	**75.9**	(37.7)
	2018	**0.8**	(0.5)	**1.2**	(0.7)	**5.5**	(3.4)	**195.1**	(93.7)	**158.5**	(58.3)	**246.9**	(56.1)
	2019	**0.6**	(0.4)	**0.1**	(0.1)	**3.1**	(2.4)	**26.6**	(9.2)	**67.8**	(29.7)	**327.0**	(88.3)
	2020	**1.4**	(1.1)	**0.7**	(0.5)	**1.6**	(1.4)	**30.2**	(23.8)	**65.3**	(33.7)	**71.3**	(24.1)
	2021	**3.2**	(2.4)	**0.2**	(0.1)	**1.1**	(0.8)	**50.0**	(20.3)	**74.0**	(25.7)	**177.6**	(59.0)
Bull Trout	2015	-		-		**1.9**	(1.3)	**0.3**	(0.3)	**0.6**	(0.6)	**1.0**	(0.7)
	2016	**45.7**	(31.1)	**27.2**	(7.9)	**24.3**	(9.0)	**1.7**	(1.6)	**2.4**	(2.4)	**0.1**	(0.1)
	2017	**59.7**	(27.2)	**32.4**	(9.0)	**6.0**	(5.5)	**0.0**	-	**9.6**	(5.1)	**3.3**	(3.2)
	2018	**19.7**	(19.6)	**21.5**	(9.1)	**8.6**	(4.4)	**0.0**	-	**2.6**	(2.6)	**3.9**	(2.7)
	2019	**13.7**	(8.6)	**0.0**	-	**9.4**	(6.2)	**2.4**	(2.3)	**0.0**	-	**3.5**	(2.3)
	2020	**21.2**	(7.8)	**13.2**	(10.1)	**4.9**	(4.5)	**0.0**	-	**1.0**	(1.0)	**3.0**	(2.9)
	2021	**15.4**	(8.7)	**16.0**	(6.7)	**26.3**	(11.8)	**0.0**	-	**0.0**	-	**2.2**	(1.9)

The total catch rate for gwenis in Seton Lake in 2021 of 100.5 (± 23.9) fish/10 net-hours was in between the catch rates in 2015, 2016, 2017 and 2020 (i.e., 44.9 (± 16.1), 47.1 (± 13.6), 63.1 (± 15.4) and 56.5 (± 15.6), respectively) and the increased catch rates documented for this species in 2018 and 2019 (i.e., 200.2 (± 40.5) and 140.4 (± 37.3), respectively). This has resulted in a slight positive trend (slope = 9.39) for Seton Lake gwenis across the monitoring period to-date (Figure 3.12; upper plot). An emerging trend among years appears to be that highest CPUE values are typically in the outflow section of Seton Lake, with lowest values at the inflow end. This was true again in 2021 (as described above), and the differences between these sections were significant.

Relative to Seton Lake, CPUE values for gwenis in Anderson Lake have been consistently much lower (see 2° y-axis on Figure 3.12 upper plot), and this was again the case in 2021. The CPUE values in Anderson Lake were slightly higher in 2017 and 2018 (i.e., 2.7 (±1.3) and 2.6 (±1.3) fish/10 net-hours, respectively) relative to the other monitoring years (i.e., 1.8 (±0.8), 0.6 (±0.3), 1.3 (±0.8), 1.2 (±0.6) and 1.5 (±0.9) fish/10 net-hours for 2015, 2016, 2019, 2020 and 2021, respectively), although the overlap in error margins suggests that these differences were not significant. The trend in CPUE for Anderson Lake gwenis was fairly flat (slope = -0.03) across the period of monitoring based on the results to-date.

Total 2021 bull trout CPUE in Seton Lake (0.7 (±0.7) fish/10 net-hours) was within the range from previous study years (i.e., 0.7 to 2.2 (±0.4 to 1.3) fish/10 net-hours), except 2017 which was higher (4.9 (±2.3) fish/10 net-hours) (Figure 3.12; lower plot). Among years, catch rates have usually been highest in the outflow or mid sections and lowest in the inflow section, the same as the pattern observed for gwenis in this lake. Conversely, in Anderson Lake, highest CPUE values for bull trout have varied among each section across years. Total 2021 CPUE for bull trout in Anderson Lake (19.2 (±5.3) fish/10 net-hours) was lower than in 2016 and 2017 (31.5 (±9.7) and 32.7 (±10.3) fish/10 net-hours, respectively), but on par or higher than the other monitoring years to-date. The trends in bull trout CPUE have had a slight negative slope (i.e., -0.10 and -0.35 for Seton and Anderson lakes, respectively; Figure 3.12; lower plot).

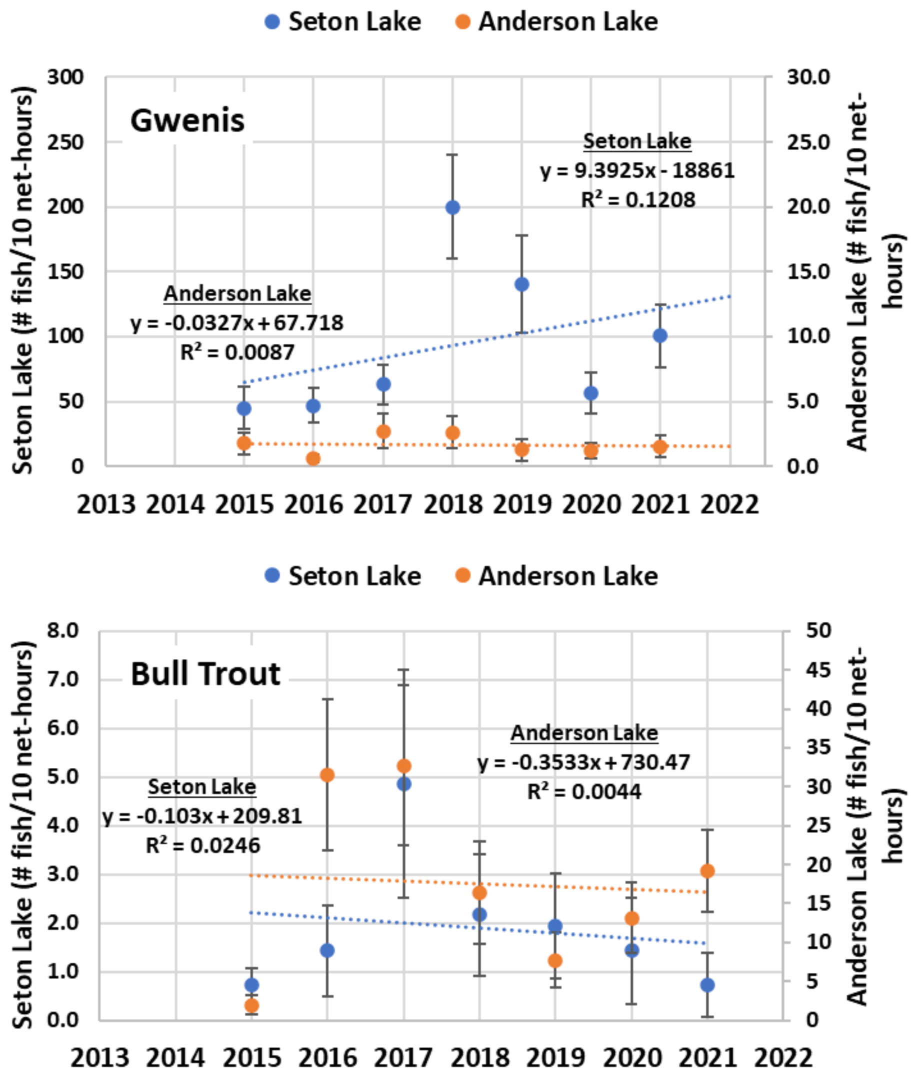


Figure 3.12 Total annual catch-per-unit-effort for gwenis (top) and bull trout (bottom) in Seton and Anderson lakes by study year (2015-2021). Note the different y-axes for each lake in both plots.

The 2021 distribution of gwenis and bull trout catches according to individual sampling locations in Seton and Anderson lakes are depicted in Figure 3.13 and Figure 3.14. From these figures it is additionally apparent that the greatest proportions of dots depicting higher catch rates for gwenis by site (i.e., *blue* or *green* dots) in Seton Lake occurred within the outflow and mid sections, and there were very few coloured pins (which depict bull trout catches) associated with this set of sampling locations (Figure 3.13). In the Anderson Lake figure (Figure 3.14), there were more white dots for the nearshore sets, reflecting the difference in catch and distribution of gwenis in this lake at the time of the survey, and there were more coloured pins (depicting bull trout catches) than in Seton Lake. The magnitude of the error margins around the CPUE values (included in Table 3.10 and shown on Figure 3.11 and Figure 3.12) was a reflection of the fact that catches tended to be highly variable among sites.

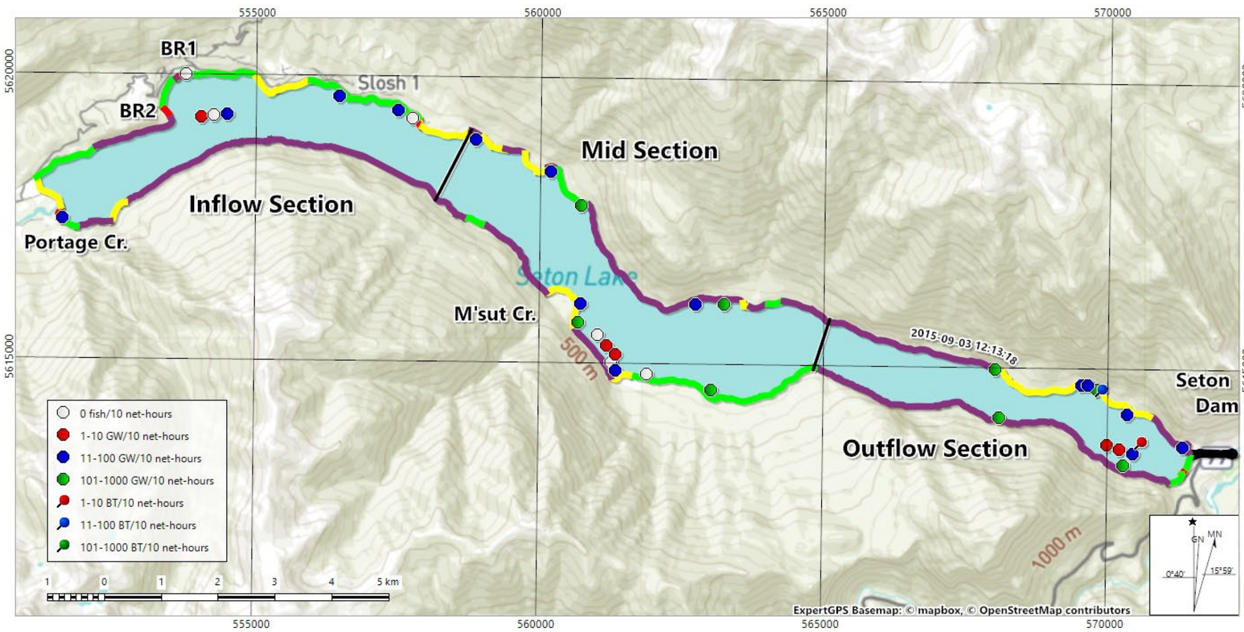


Figure 3.13 CPUE for gwenis (circles) and bull trout (pins) according to capture location in Seton Lake during Year 9 (2021). The CPUE ranges for each colour are Indicated in the legend. The coloured shorelines reference the shoreline habitat type: Red = Tributary or Diversion inflows; Yellow = Fans; Green = Shallow Slopes; Purple = Steep Slopes.

Shoreline habitat types (as documented by the habitat mapping in 2015 and 2017) around the entire perimeter of each lake are depicted by coloured lines in Figure 3.13 and Figure 3.14. However, potential trends in catches (or CPUE) according to shoreline habitat type were not plotted or evaluated because catches of target fish tend to be much deeper and further from shore in these lakes than the visual range of the habitat mapping surveys was able to describe (for an explanation of the habitat mapping methods, refer to the Year 5 (2017) report). Therefore, it was not certain that the habitats at the depths and distance from shore where fish are captured (see next sub-section, below) have any similarity or direct correlation with the habitats observed and characterized at the lake edge by the habitat surveys.

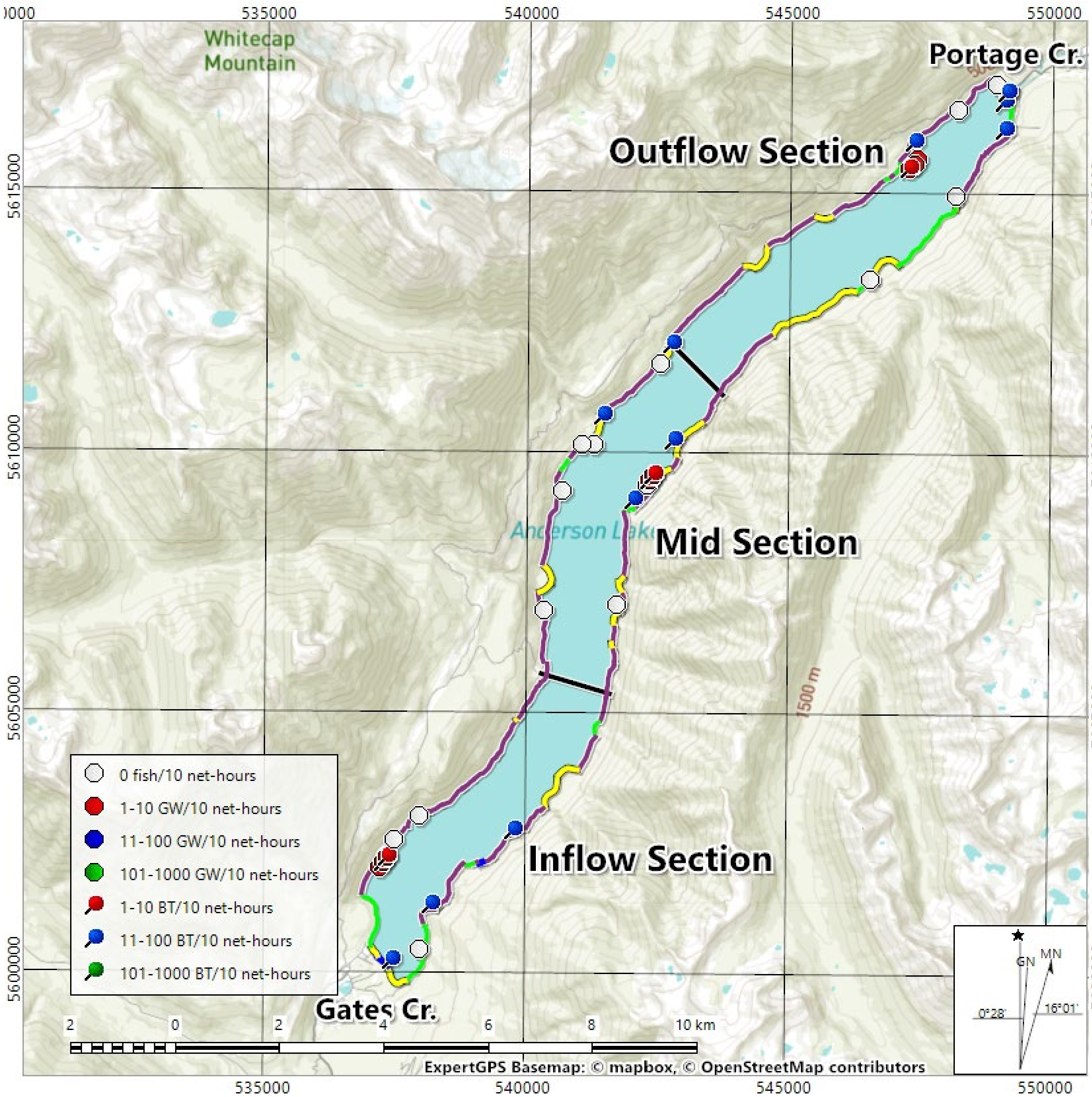


Figure 3.14 CPUE for gwenis (circles) and bull trout (pins) according to capture location in Anderson Lake during Year 9 (2021). See caption under Figure 3.13 for additional information.

We also considered plotting the relationships between diversion volume (from BR1 and BR2 operations) and sedimentation rate (in Spawning Year $_t$) on gwenis recruitment (in Spawning Year$_{t+4}$; since the majority of the sample are Age-3 – i.e., 4 years from spawning year to when captured by this program in Seton Lake). However, the operations (diversion volume and sediment deposition) in 2013 to 2017 would have theoretically affected recruitment of the cohorts that were Age-3 in 2017 to 2021, respectively. Because of that timespan between the effect variable and the response variable, there would only be five corresponding data points (for diversion volume vs. Age-3 CPUE), and only two corresponding data points for sedimentation (since that monitoring component started in 2016), within the timeframe of the study period to-date. As such we decided that this analysis should be reserved for the Year 10 (2022) report when all of the data from this program will be available.

Since gwenis and bull trout catches have been variable among years, we plotted the annual index of abundance for gwenis versus bull trout in each lake to test for a potential effect of bull trout population size on the numbers of gwenis, since they are a preferred prey item (as juvenile fish in Anderson Lake, and across their entire size range in Seton Lake – see sub-section on bull trout stomach contents, below) (Figure 3.15). The index of gwenis abundance (CPUE) has varied greatly in Seton Lake with much less corresponding variation in bull trout abundance. Whereas the opposite was true in Anderson Lake. Based on the available data there was the unlikely result of a slight positive correlation for each lake (i.e., gwenis abundance increases with bull trout abundance); however, the fit was poor due to the variability among the data points. As such, there is presently no evidence for a direct effect of the observed range of bull trout abundances on gwenis population size within the span of years monitored.

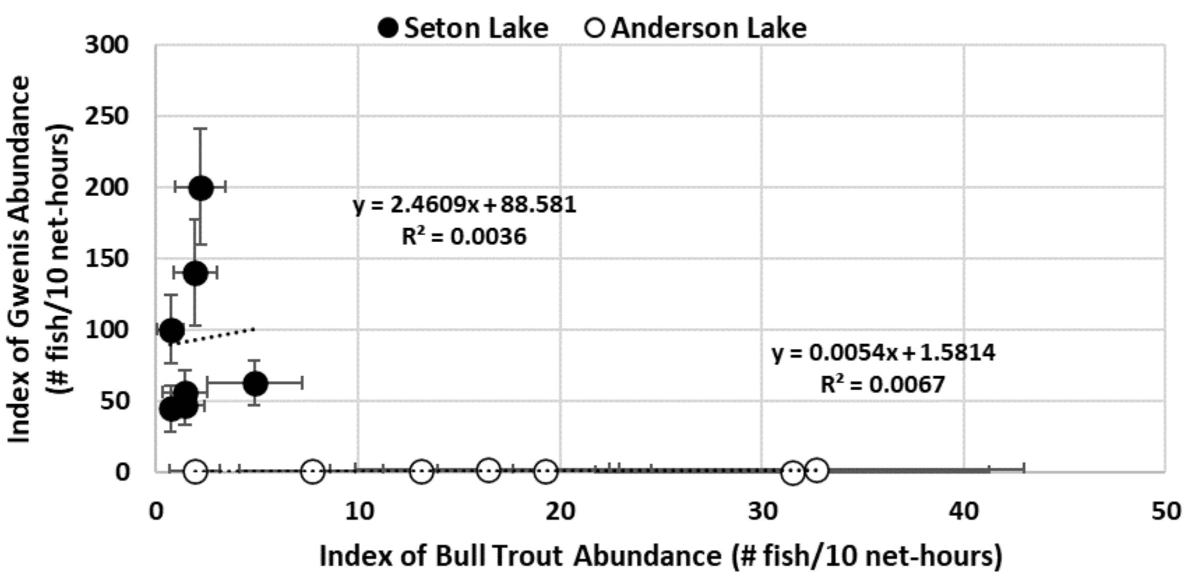


Figure 3.15 Regressions of the index of abundance (annual CPUE) for gwenis versus bull trout in Seton Lake (solid circles) and Anderson Lake (open circles) for all available study years to-date (2015–2021).

Conversely, this relationship could also be viewed in the opposite way: that gwenis abundance directly influences bull trout abundance. However, when plotted this way (gwenis catch rates as the independent variable and bull trout catch rates as the dependent variable) the fit of the regressions is equivalently poor indicating that other factors likely have a stronger influence on their abundance patterns than the prey-availability or predation-risk relationships between them.

Catches by depth and distance from shore

As described in previous monitoring reports, gwenis and bull trout were captured across a broad range of depths in Year 9 (2021; Figure 3.16). For gwenis in Seton Lake, their distribution spanned the range of sampling depths, but they were disproportionately associated with the metalimnion thermal layer. Approximately 8% of the gwenis were captured in the epilimnion layer (0 to 20 m depth), 63% were in the narrow metalimnion (20 to 30 m), and 29% were in the sampled portion of the hypolimnion (30 to 56 m). As in past years, over 90% tended to be below 20 m from the surface. The majority of these gwenis were sampled near the lake bottom at these depths (just above the lead line of the net) in the nearshore sets and were assessed to be mature and in spawn-ready (i.e., gravid or ripe) condition at the time of the survey. Bull trout in Seton Lake also tended to be fairly deep and, though catch numbers were comparatively small, their depth distribution appeared to correlate with the depths where gwenis abundance was the greatest (i.e., the metalimnion).

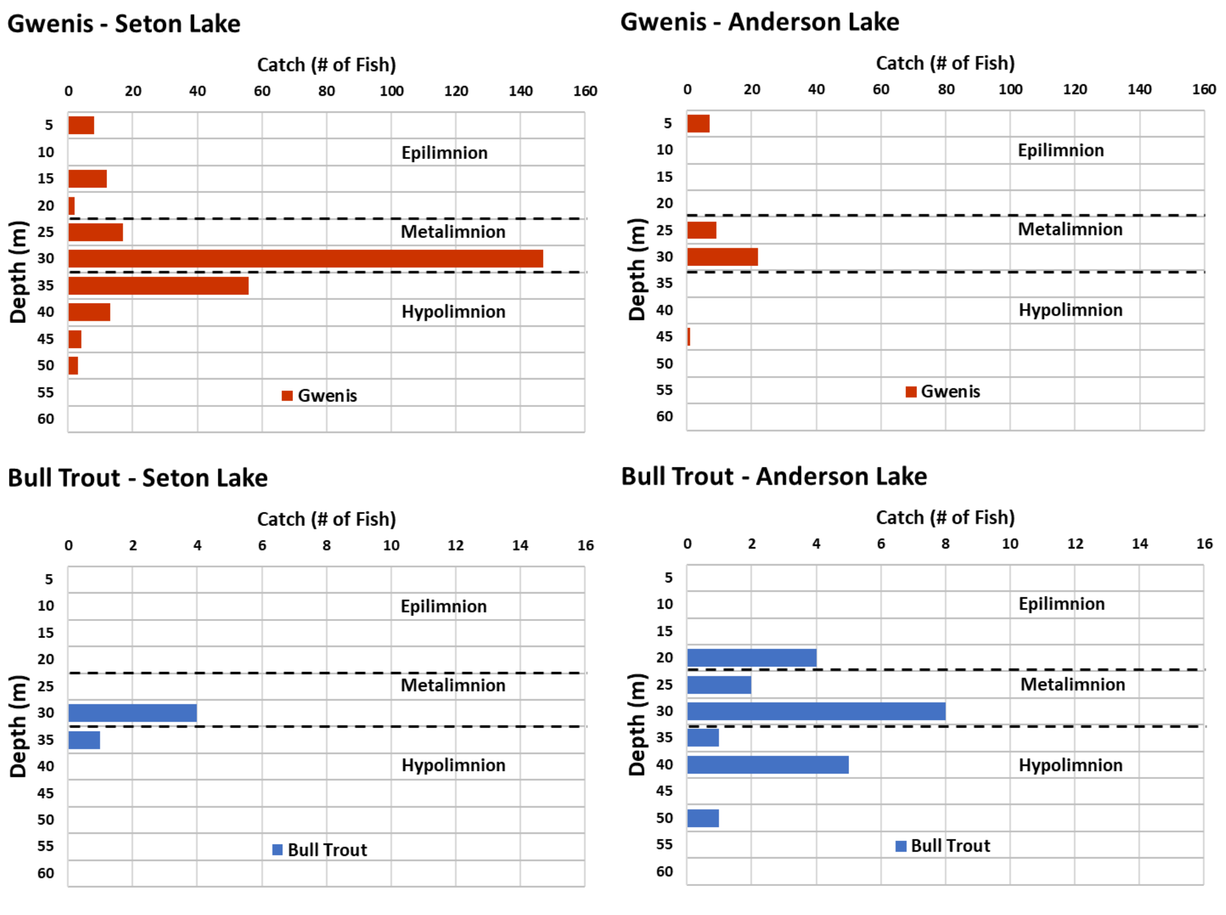


Figure 3.16 Numbers of gwenis (upper plots) and bull trout (lower plots) by capture depth in Seton Lake (left plots) and Anderson Lake (right plots) during the annual population indexing survey in Year 9 (2021). Note the different x-axis scales between upper and lower plots.

In terms of horizontal distribution from shore, the vast majority (94%) of gwenis in Seton Lake tended to be >45 m from the lake edge (Figure 3.17). This pattern was also evidenced in the distance from shore distribution for bull trout, which reflected that the bull trout tended to be found in panels that also captured gwenis (despite dramatically different body size). Stomach content assessment also identified exclusively gwenis in bull trout stomachs, further documenting that Seton Lake bull trout are targeting gwenis (more on this in bull trout stomach contents sub-section, below).

In Anderson Lake, adult gwenis (which were not in spawn-ready condition at the time of the survey) were predominantly captured within the metalimnion thermal layer in offshore net sets or the panel furthest from shore for nearshore nets (i.e., ≥100 m and ~90 m horizontal distance from shore, respectively). As in Seton Lake, the *O. nerka* caught in <10 m depth were likely immature gwenis based on size (all were ≥122 mm).

Anderson Lake bull trout were distributed across a wider range of depths than in Seton Lake but were similarly most abundant within the metalimnion layer. They also tended to be widely distributed across the range of sampled distance from shore, though the highest catch rates were at distances between 30 m and 90 m from the lake edge (i.e., in the nearshore sets). The catch rate diminished beyond that (i.e., in offshore sets) and, notably, the distance from shore pattern reflected that the spatial distribution of bull trout and adult gwenis only minimally overlapped in this lake.

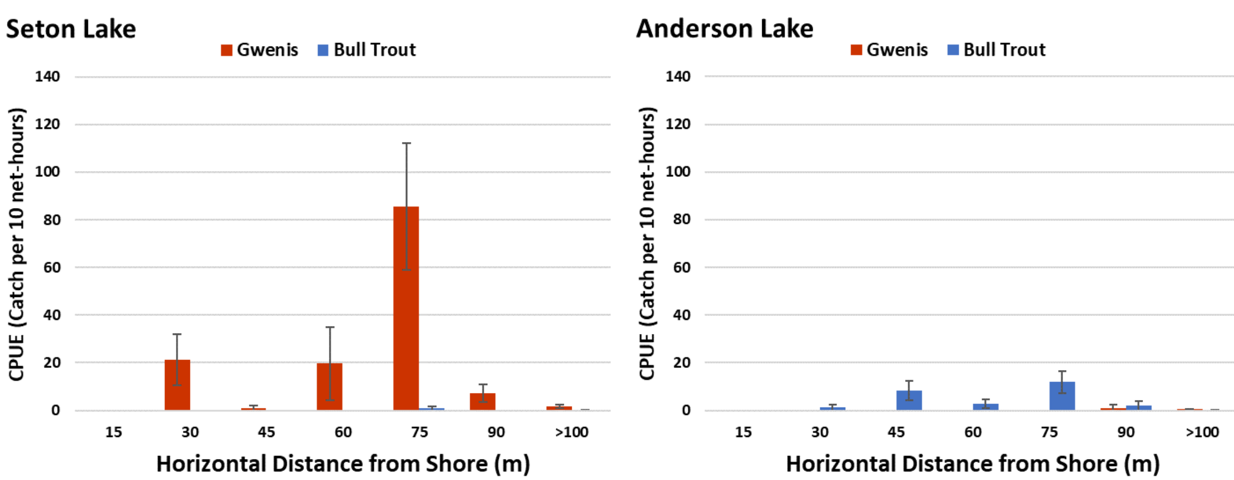


Figure 3.17 CPUE for gwenis (red columns) and bull trout (blue columns) by horizontal distance from shore in Seton Lake (left plot) and Anderson Lake (right plot) during the annual population indexing survey in Year 9 (2021). Distances less than 100 m are from nearshore sets; offshore sets are represented by the >100 m column.

Size-at-Age

The following results are based on the fish size measurements (fork lengths) and ages (based on reading of age structures) for all study years, updated to include the Year 9 (2021) data.

Juvenile gwenis (Age-1 and Age-2) based on the scale ageing results were all ≤197 mm in Seton Lake and ≤241 mm in Anderson Lake. Minimum sizes in the sample (across years) have been 95 mm and 106 mm for each lake, respectively, which represented the minimum size limitation of the sampling gear rather than the smallest size of this species in the two lakes. Median fork length for Age-1 gwenis was 125 mm in Seton Lake (*n*= 61) and 128 mm in Anderson Lake (*n*= 14) (Figure 3.18). Median Age-2 fork lengths were 165 mm (Seton; *n*= 418) and 140 mm (Anderson; *n*= 118). In Seton Lake, all of the mature gwenis were Age-3, the same as what was reported for Years 3 to 8 (2015 to 2020) results, as well as the BRGMON-6 ageing results (Limnotek 2016) and ranged in size from 158 to 240 mm (median = 195 mm; *n*= 1634).

In Anderson Lake, the mature gwenis were up to 4 years old and larger than the Seton Lake fish, ranging in size from 142 to 369 mm (median = 258 mm (*n*= 89) and 310 mm (*n*= 227) for Ages 3 and 4, respectively). All of the gwenis captured in Anderson Lake were also very chrome-coloured and not in spawn-ready condition at the time of the survey (28 September to 8 October), further confirming that spawn-timing for this population is later (as previously reported in Morris et al. 2003, Limnotek 2016, and Sneep 2022) than the timing for the Seton Lake population.

Gwenis

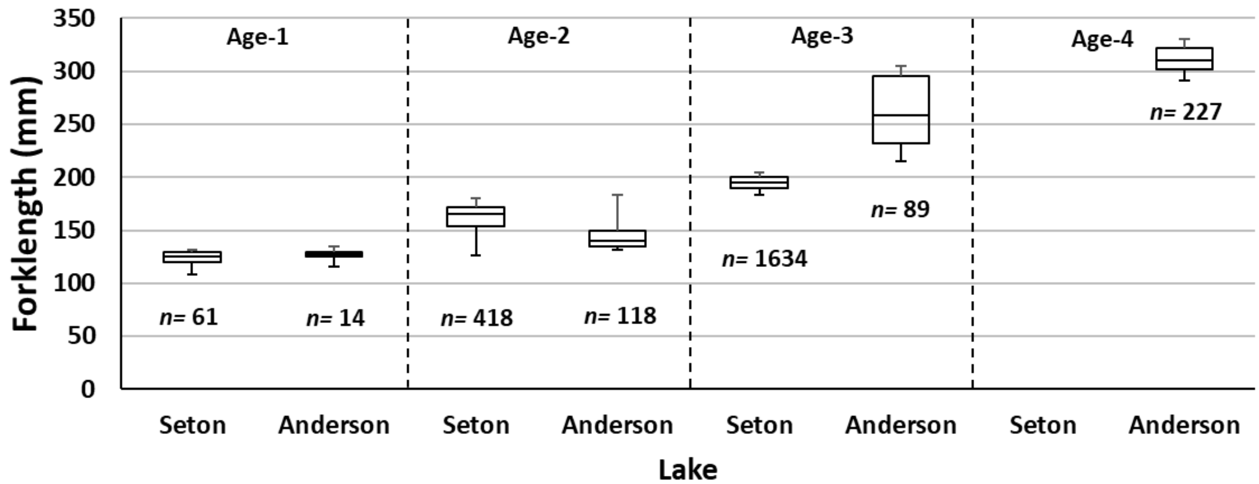


Figure 3.18 Size-at-age box plot for gwenis captured during the annual gill net survey in Seton and Anderson lakes from late September to early October, 2015 to 2021 (all years combined). The boxes are bounded by the 75th and 25th percentiles. The median divides each box and the whiskers represent the 10th and 90th percentiles. Note that sample sizes for each age group differ among the two lakes.

Given that the gwenis spawn earlier in Seton Lake and temperatures at depth where they spawn are similar during the incubation period among the two lakes, it is likely that the new year class of gwenis emerge earlier in Seton Lake than in Anderson Lake. As a result, the Seton fish are larger than the Anderson fish in their first summer (Age-0+), owing to their earlier start in the growing season, as reported by BRGMON-6 (Limnotek 2016). The Seton Lake fish may also benefit from shallower daytime habitat use afforded by the turbidity, and lower predation risk (due to lower predator abundance), which may also contribute to the higher growth rates observed for the early age classes of *O. nerka*. However, the Anderson Lake gwenis appear to catch up in size after Age-2 and are consistently larger at each age thereafter, likely due to lower conspecific densities coupled with much higher zooplankton abundance in Anderson Lake, which is their primary food source (Limnotek 2016). So, interestingly, the Seton Lake gwenis mature earlier (at Age-3) despite being smaller, and the Anderson Lake gwenis mature at Age-4 despite faster growth after Age-2 (Figure 3.18).

For the bull trout sampled in Seton Lake in 2021 (*n*= 5), two were Age-5 (both 345 mm in length), one was Age-6 (425 mm) and two were Age-11 (720 and 733 mm) based on fin ray and otolith ageing. The bull trout captured in Anderson Lake in 2021 (*n*= 21) were Age-4 (*n*= 4; 280–412 mm); Age-5 (*n*= 4; 380–406 mm); Age-6 (*n*= 1; 445 mm); Age-7 (*n*= 3; 414–465 mm); Age-8 (*n*= 2; 474–483 mm); Age-9 (*n*= 1; 512 mm); Age-10 (*n*= 2; 535–612 mm); and Age-11 (*n*= 1; 550 mm). Across all the monitoring years to-date, the total size distribution of captured bull trout has been from

126 to 754 mm in Seton Lake and 213 to 760 mm in Anderson Lake, and the majority have been between Age-3 and Age-9 in both lakes (Figure 3.19).

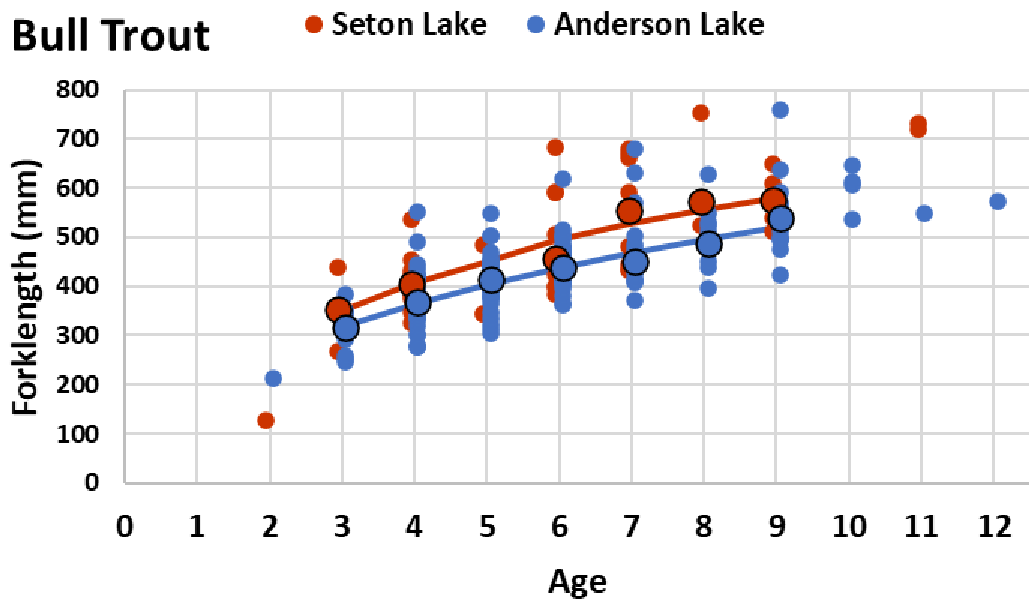


Figure 3.19 Size-at-age plot based on fin ray and otolith ageing data from bull trout captured during the annual gill net surveys in Seton and Anderson lakes from late September to early October, 2015 to 2021 (all years combined). The lines represent the von Bertalanffy growth curves based on median size values for each lake. Note: growth curves were not be extended below Age-3 or beyond Age-9 due to insufficient sample sizes for the youngest and oldest age classes.

The median size values of bull trout have been consistently smaller in Anderson Lake than in Seton Lake (for each age that had a median length value for both lakes), and the predicted growth curve based on the available data is also lower. The inclusion of the Year 9 (2021) data did not change this result overall; however, the annual sample size was again much smaller in Seton Lake. As described in previous reports, bull trout may be larger in Seton Lake due to lower density of this species, resulting in less competition for food resources relative to Anderson Lake where bull trout abundance is higher. There is a trade-off with foraging opportunity, however, since predation efficiency in Seton Lake may be limited by the turbidity resulting from the diversion inflows (Limnotek 2016). In addition, we speculate that the inclusion of mature gwenis in the diet of the Seton Lake bull trout (due to appropriate prey body size) may lend itself to better growth since the gwenis are relatively abundant in that lake.

The larger body size of most mature gwenis in Anderson Lake may preclude them as a prey item for bull trout. Some evidence of bull trout movements (Seton Entrainment Study; BRGMON-8 Year 4 results) also suggests that Seton Lake bull trout may be more migratory – opportunistically moving into and out of Seton Lake to and from the Seton River corridor, Fraser River, or adjacent watersheds (e.g., Bridge River, Yalakom River) to capitalize on the best feeding locations wherever they occur at different times of year (Burnett and Parkinson 2018; Sneep 2018a).

A total of 15 rainbow trout was captured in Year 9 (2021) and, similar to previous years, a larger proportion of them (2021 *n*= 12 or 80%) was in Anderson Lake. The three rainbow trout in Seton Lake were 435, 441 and 447 mm, and the Anderson Lake fish ranged in size from 204 to 460 mm. Scales were collected from thirteen of these fish and, based on scale reading, they were Age-2 (225 mm), Age-3 (204 and 215 mm), Age-4 (318 mm), Age-5 (336 to 460 mm) and Age-6 (430 and 435 mm). Across all monitoring years, the age range for rainbow trout has been Age-1 to Age-6 (129 to 447 mm; *n*= 12) in Seton Lake, and Age-1 to Age-8 (112 to 460 mm; *n*= 62) in Anderson Lake.

Bull trout stomach contents

Stomach contents were assessed in the field for nine bull trout in Year 9 (2021); Four were from Seton Lake and five were from Anderson Lake (Table 3.11). As documented in previous reports, most of the identifiable prey items in bull trout from both lakes were various life stages of the species *O. nerka* (i.e., sockeye or gwenis; eggs, juveniles and adults). Based on the identifiable contents in 2021, the bull trout in Seton Lake had been feeding on juvenile and mature gwenis, which were between approximately 100 mm and 180 mm in length. Four of the Anderson Lake bull trout had been feeding on juvenile *O. nerka* (size range: approx. 100 to 130 mm). In three cases the stomachs were either completely empty or the contents had been digested to the point that the remaining prey items could not be identified to species or estimated for original size.

Table 3.11 Summary of stomach contents assessed for bull trout from Seton and Anderson lakes during the annual gill netting survey, 28 September to 8 October 2021.

Lake	Bull Trout Size		Stomach Contents			
	Fork length (mm)	Weight (g)	Species	#	Approx. Prey Size (mm)	Comments
Seton	345	459	O. nerka	1	100 - 120	Juvenile
	345	470	Stomach empty – contents digested			
	425	754	Stomach empty – contents digested			
	720	4280	Gwenis	5	160 - 180	Spawners
Anderson	280	188	O. nerka	1	100 - 120	Juvenile
	380	473	O. nerka	1	100 - 120	Juvenile
	445	909	O. nerka	2	100 - 130	Juveniles
	512	852[a]	O. nerka	3	100 - 130	Juveniles
	612	2639	Stomach empty – contents digested			

[a] This fish was noted as being very skinny for its length.

4. Discussion

Another year of data were collected and analysed in 2021 (Year 9) that will contribute to answering the management questions by the end of the monitoring program in 2023 after 10 years of study. Field and laboratory work was completed in Year 9 (2021) to contribute information towards addressing the first 3 of 4 management questions for this program. Data and interpretations to address question 4 will be possible once the full time-series of data are available, and a possible synthesis of relevant information from the BRGMON-6 program can be integrated into the analyses.

MQ 1: What are the basic biological characteristics of resident fish populations in Seton Lake and its tributaries?

The following response to this management question is based on the available information and learning from this program to-date and has been updated to include all relevant and new information collected in Year 9 (2021). Overall, the 2021 results further corroborated the conclusions described for previous study years and built upon our growing understanding of the resident fish populations in Seton and Anderson lakes.

Gwenis

As in Years 3 to 8 (2015 to 2020), gwenis were the most abundant species in the annual resident fish sampling in Year 9 (2021) and continued to be a well-suited target species for monitoring. The 2021 gwenis catch rate in Seton Lake of 100.5 (± 23.9) fish/10 net-hours was in between the lower catch rates in 2015, 2016, 2017 and 2020 (i.e., 44.9 (± 16.1), 47.1 (± 13.6), 63.1 (± 15.4) and 56.5 (± 15.6), respectively) and the increased catch rates documented for this species in 2018 and 2019 (i.e., 200.2 (± 40.5) and 140.4 (± 37.3), respectively). The higher CPUEs in 2018 and 2019 may be related to the natural cycle of dominant versus sub-dominant years for the respective parent populations of sockeye (i.e., Gates and Portage); however, the timing of dominant years for sockeye may not directly correspond with the dominant years for gwenis due to their different life histories in this system (i.e., maximum age of 4 for sockeye versus maximum age of 3 for gwenis in Seton Lake). It is possible that in-lake habitat conditions (e.g., diversion inflows, temperature, sedimentation) could also be factors contributing to observed differences among years; however, we haven't yet matched effects variables (in $Year_t$) with fish population response (in $Year_{t+4}$) at this point. Further explanation is provided in the responses to Management Questions 2 and 3, below. Our plan is to plot these relationships for the Year 10 (2022) report once all the years of monitoring data are available.

In each study year, gwenis adults have been substantially more numerous in the Seton Lake catch, particularly in nearshore sets (between ~45 and 90 m from shore), than in Anderson Lake. However, it is certainly possible that the gwenis in Seton Lake are more vulnerable to the sampling due their distribution on the bottom within the range of the nearshore sets related to spawning aggregations, whereas the gwenis in Anderson Lake are less densely distributed throughout the pelagic zone at the time of sampling. For the purposes of this program, we are less interested in attempting to quantify the absolute population size differences between these populations and more focussed on assessing the relative population trends over the years of monitoring (for addressing management question #2).

In nearshore habitats, there has been a gradient of CPUE values for gwenis among the three longitudinal sections in Seton Lake: highest catch rates have consistently been in the outflow section, followed generally by the mid section and then the inflow section. This result was true again in 2021 and catches in the outflow were substantially greater than the mid and inflow sections. The catch rates for gwenis in Anderson Lake tended to be much lower than in Seton Lake and more similar among the longitudinal sections (the differences between sections were not significant when error margins are considered).

Since the timing of sampling coincides with the start of spawning for this species in Seton Lake, this apparent gradient of abundance along the longitudinal axis of the lake appears to reflect differential spawning use among the inflow, mid and outflow sections. This could be related to differences in the temperature profiles and/or the varying degrees of fine sediment deposition according to proximity to the diversion inflows, as described by the physical conditions monitoring (Section 3.1). Other factors contributing to the observed distribution could also include differential food base availability or foraging opportunities among the sections (which may also link to the associated physical conditions). The 2021 results provided an additional set of data points contributing to the described patterns. Data from the final monitoring year in 2022 will be useful for contributing to the weight of evidence.

As in each previous year, the Seton Lake gwenis were noticeably smaller than the Anderson Lake gwenis, particularly after Age-2 (median sizes at Age-3 = 195 mm vs. 258 mm, respectively, as of Year 9 (2021)). As reported by the BRGMON-6 program, zooplankton (which is the primary food item for gwenis) were less than half as abundant in Seton Lake than in Anderson Lake (Limnotek 2016). This smaller food base coupled with the larger population size in Seton Lake (of both gwenis and Gates/Portage sockeye juveniles) may be important factors contributing to the reduced growth relative to Anderson Lake gwenis. There could also be other factors contributing to the different growth rates among the age classes, such as: changes in habitat use (e.g., nearshore vs. offshore); changes in foraging behaviour (e.g., extent of diel vertical migration), differences in interaction with the stratification of temperature or turbidity, etc. as the fish age and grow within each of the study lakes. However, based on the information available from this program, weighting the relevance of these explanations would be entirely speculative.

As reported previously, the maximum assessed ages (based on scale ageing) were different between the lakes. Again in Year 9 (2021), the oldest fish in Seton Lake were Age-3 (and sexually mature) versus Age-4 in Anderson Lake, which is the same as what was reported for BRGMON-6 (Limnotek 2016). Also, the *O. nerka* juveniles (Ages 1 and 2; <180 mm) in the catch from both lakes were likely all gwenis offspring based on size at the time of sampling.

As in previous study years, the majority (i.e., 86%) of the gwenis sampled in Seton Lake during the late September to early October 2021 survey were mature and in some stage of spawning readiness (assessed as green (34%), gravid (18%), or ripe (49%) by gently squeezing the belly to express gametes). This suggests that spawning in Seton Lake by this species occurs around the time of the annual fish population index sampling, or shortly thereafter. Also, >90% of mature gwenis in spawning-ready condition were sampled in the bottom-set nets at depths ≥ 20 m, and between 45 to 90 m horizontal distance from the lake edge again in 2021. Since these results have been consistent during each monitoring survey to-date, these locations of capture likely reflect the spatial distribution of these fish during their spawning period in this lake.

We plotted the distribution of catch rates according to the habitat types around the perimeter of each lake (based on the shoreline habitat mapping results from Years 3 and 4) in Figure 3.13 and Figure 3.14 in Section 3.2 to visually represent the distribution of catches in 2021 with the adjacent shoreline habitat types. In general, the highest catch rates were on shallow slopes and adjacent to fans; however, these were also the nearshore types that were most readily sampled by the gill netting method. Other available habitat types comprised a much smaller proportion of the sample because there were either much fewer of them (i.e., creek mouths), or they were generally not as conducive to effectively setting gill nets (i.e., steep shorelines). Also, as explained in the Results section, we cannot be certain that the habitat characteristics selected by gwenis at the depths and distance from shore where they are captured have a direct correlation with the habitat types observed and characterized at the lake edge by the habitat surveys.

Gwenis in Seton and Anderson lakes display the same phenotypic expression for spawning (black colouration as opposed to the typical red and green for sockeye or other kokanee populations) and select spawning sites deep within their respective lakes. In Anderson Lake, none of the gwenis were considered spawning-ready (i.e., chrome colouration, tight bellies, no gametes expressed), reflecting the later spawn-timing for this population. Morris et al. (2003) estimated gwenis spawning in Anderson Lake to occur during December and January, based on the observation of carcasses on shorelines in January; however, this estimate has not been directly corroborated by other field studies to-date. Also, gwenis in Anderson Lake have been almost exclusively either <15 m from shore (typically juveniles) or in offshore habitats (i.e., >100 m from shore), and either near the surface (0 to 10 m depth) or between 20 and 30 m depth (in the metalimnion layer). These locations likely reflected the spatial distribution associated with rearing and feeding for the different age classes at the time of the survey in this lake. Other than for spawning or early rearing as juveniles, gwenis are generally a pelagic species that migrates above and below the thermocline (i.e., vertical movements among the thermal layers) between night and day periods for the purposes of feeding (Limnotek 2016).

Different spawn timing has also been documented for the two populations of sockeye (i.e., Gates Creek and Portage Creek runs) in the Seton/Anderson watershed. Differences in apparent spawn timing for the resident gwenis populations in each lake could be related to the differential spawn timing of their respective parent populations of sockeye. However, Gates Creek sockeye that rear in both Seton and Anderson lakes spawn approx. 1.5 months earlier than Portage Creek sockeye that apparently rear only in Seton Lake (Limnotek 2016).

On the other hand, it's also possible that the spawn timing difference reflects localized adaptations to differing habitat conditions among the two lakes. Studies in other systems have shown that *O. nerka* spawning habitat use or ecotypes (e.g., stream versus in-lake spawning), as well as spawn timing, are highly adaptive according to conditions, and sympatric populations with different ecotypes and run timings can be found within the same systems (Whitlock et al. 2018). The results of a study on *O. nerka* in Lake Washington suggested that rapid differentiation in these traits among population groups was the result of local adaptation to particular spawning and incubation environments (Hendry et al. 1998).

The combination of differential spawn timing, maximum age (and possibly age-at-maturity), and adult body size differences presents both the possibility that a) the populations of gwenis in Seton and Anderson lakes may be distinct from one-another, or that b) they could be the same but have adapted to local conditions in each lake and the populations are no longer connected. Earlier work by Moreira (2014) suggested that the two populations may be genetically distinct. The differences detected by that study were modest, but statistically significant. Further genetic analyses would be required to confirm stock identification, but this could not be accommodated within the existing budget and is not within the current scope of the BRGMON-8 program.

Bull Trout

Relative to the gwenis, catch-per-unit-effort values for bull trout were lower, particularly in Seton Lake. Total 2021 bull trout CPUE in Seton Lake (0.7 (±0.7) fish/10 net-hours) was within the range from previous study years (i.e., 0.7 to 2.2 (±0.4 to 1.3) fish/10 net-hours), except 2017 which was higher (4.9 (±2.3) fish/10 net-hours). Total 2021 CPUE for bull trout in Anderson Lake (19.2 (±5.3) fish/10 net-hours) was lower than in 2016 and 2017 (31.5 (±9.7) and 32.7 (±10.3) fish/10 net-hours, respectively), but on par or higher than the other monitoring years for this lake to-date.

Bull trout are known to be an effective piscivore that opportunistically prey on gwenis (among other species). This was confirmed by assessment of stomach contents from bull trout that had succumbed to the sampling procedure at the time of the fall survey (2021 *n*= 9 among both lakes). Across all monitoring years to-date, almost all of the identifiable prey items in the bull trout stomachs at the time of the surveys have been various life stages of the species *O. nerka* (i.e., sockeye or gwenis; eggs, juveniles and adults). In some cases, several (up to 5 or 6) adult gwenis have been noted in the stomachs of some of the larger bull trout in Seton Lake. This capacity to forage on adult gwenis as a primary food item may contribute to the larger mean size-at-age and growth rate for Seton Lake bull trout relative to the Anderson Lake fish. As such, the relative abundance of bull trout over the course of the monitoring program was initially considered a factor that may contribute to potential changes in the gwenis abundance index across years (in addition to potential operations effects). We have attempted to evaluate this by plotting regressions of the index of abundance (annual CPUE) for gwenis versus bull trout in each lake. However, the relationships were weak and non-informative based on the years of data available. While it seems unlikely at this point that the bull trout populations strongly influence the size of the gwenis populations in these lakes based on the magnitude of differences in their abundance, we will continue to assess this as a possible contributing factor to population trends with each year of data that is accrued.

As documented in each previous study year, bull trout distribution in Seton Lake tended to fully overlap with the locations where gwenis were most prevalent (i.e., both vertically in the water column and spatially in the lake). Bull trout were more numerous in Anderson Lake, but unlike Seton Lake, their spatial distribution did not coincide with the habitats where mature gwenis tended to be most abundant: The bull trout were captured across the full range of sampled depths in Anderson Lake, but almost exclusively in nearshore sets (i.e., ≤90 m from shore). Also, across all monitoring years combined, bull trout have been most abundant in the inflow and mid sections of Anderson Lake, and lowest at the outflow end, whereas gwenis were most abundant at the outflow end and less so in the mid and inflow sections. As noted here and in previous reports, the body size of mature gwenis in Anderson Lake may be too large to be a prey item for the size range of bull trout in the lake, so they focus on the juvenile life stage and/or other available prey items in nearshore habitats or at creek mouths.

Bull trout in Seton and Anderson lakes are adfluvial, migrating from the lakes into streams to spawn. In past years, spawning by this species was noted in Portage Creek (particularly at the top end near the outflow of Anderson Lake), as well as Spider Creek and Whitecap Creek (which are both tributaries of Portage Creek); however, the relative use of other streams within or outside of the Seton/Anderson watershed were unknown. Based on the radio telemetry monitoring that was available during Year 4 (2016), bull trout movements in Seton Lake tended to have a downstream orientation during the monitored period (27 July to 12 October 2016), with the majority of detections at the fixed stations in the Seton Dam approach channel and below (Burnett and Parkinson 2018). Upstream movements were detected for only 2 of the 30 tagged fish from Seton Lake during the typical spawning period (i.e., mid September to end of October): 1 was detected in Portage Creek, and 1 was detected in Gates Creek.

On the other hand, a higher proportion of detected bull trout tagged in Anderson Lake (during the available monitoring period from July to mid October 2016) tended to migrate in an upstream direction, reflecting possible spawning movements and/or foraging opportunities associated with the Gates Creek sockeye returns (to-date, the gill net site at the mouth of Gates Creek has consistently had the highest catch rate for bull trout of any individual site in the study area). Of the sample size of 10 fish, 4 made upstream movements into Gates Creek and 2 made downstream movements (1 into Portage Creek, and 1 into the inflow end of Seton Lake) during the spawning period. Four of the tagged bull trout were not detected after initial capture in Year 4 (2016) and may have remained in Anderson Lake during the monitored period. As there was no fixed telemetry station in Portage Creek, it was possible that Anderson Lake bull trout could have migrated to Seton Lake (or vice versa) between the weekly mobile tracking surveys in Portage

Creek, undetected. None of the tagged Anderson Lake bull trout were detected by the receivers at the outflow end of Seton Lake during the continuous period of operation (June 2015 to Oct 2017).

Recaptures of PIT tagged bull trout can also shed light on their movements within or among sample sessions. PIT tagged fish were not recaptured in 2018, 2019, 2020 or 2021; however, a total of three tagged bull trout have been recaptured by the program in prior years (two in Anderson Lake and one in Seton Lake in 2017). Each of these fish were recaptured in the same lake where they were originally captured, further suggesting some degree of lake-based fidelity. This is a small sample size to-date but capture and recapture locations for PIT tagged fish will continue to be monitored going forward, to build on this dataset of bull trout movement information within the Seton-Anderson watershed.

Rainbow Trout

Rainbow trout have only been sampled in very low numbers during all eight years of this monitoring program to-date, and the majority have been captured in Anderson Lake. Only three rainbow trout were captured in Seton Lake in 2021, all in the mid section and near the surface (i.e., between 0 and 2.5 m depth). Another 12 individuals were captured in Anderson Lake during Year 9 (2021) between 0 and 30 m depth across all three longitudinal sections. Across years, the rainbow trout have spanned a deeper range in the epilimnion and metalimnion layers of Anderson Lake but have almost exclusively been at the surface in Seton Lake. Among the longitudinal sections they have been more prevalent at the outflow end of Seton Lake and in the inflow and mid sections of Anderson Lake overall. The sizes of these fish have ranged from 112 mm to 460 mm (2021 range = 204 to 460 mm), and the aged individuals (n= 64) have ranged from Age-1 to Age-8 (2021 range = Age-2 to Age-6) based on scale reading.

Like the bull trout, this population is likely adfluvial, migrating to nearby streams in the spring to spawn. The combination of small population size and their life history characteristics makes this species less suitable for trend monitoring and linking observed population characteristics to operations, relative to gwenis and bull trout. However, data for rainbow trout will continue to be collected to support an understanding of the basic biology of this species, and conclusions or recommendations at the end of the monitor, if possible.

Other Resident Fish Species

Northern pikeminnow (*Ptychocheilus oregonensis*), peamouth (*Mylocheilus caurinus*), redside shiner (*Richardsonius balteatus*), mountain whitefish (*Prosopium williamsoni*), and bridgelip sucker (*Catostomus columbianus*) are other resident species that have been consistently documented in the catch each year (see Table 3.6). The summary of life history information that follows for these species was based on the information available from the BRGMON-8 sampling results, supplemented by information referenced from “The Freshwater Fishes of British Columbia” (McPhail 2007), which incorporates information from other watersheds in BC.

Across years, northern pikeminnow have been the most abundant of this group (*n*= 706) and contributed to the sample from each lake, although they tended to be more prevalent in Seton Lake (*n*= 434 or 61%) versus Anderson Lake (*n*= 272 or 39%). Catches for this species in 2021 (*n*= 108) was among the highest of the study years to-date (*n*= 72, 94, 65, 105, 118 and 144 for 2015, 2016, 2017, 2018, 2019 and 2020, respectively). They have ranged in size from 102 to 533 mm (2021 range = 113 to 470 mm in Seton Lake and 195 to 435 mm in Anderson Lake). Like bull trout, pikeminnows are a piscivore that may feed on gwenis (among other prey species) in these lakes. For this reason, tracking their abundance across the duration of this monitor could be important (for the same reason as bull trout, as stated above). Pikeminnow are spring spawners (i.e., May and June), and spawning can occur in both flowing water (inlet streams) and in lakes (McPhail 2007).

Peamouth were the next most abundant (*n*= 457, all years combined; 2021 *n*= 50), but were more prevalent in the Seton Lake catch (*n*= 353 or 77%) than the Anderson Lake catch (*n*= 104 or 23%). They have ranged in size from 97 to 293 mm (2021 range = 155 to 258 mm in Seton Lake and 175 to 240 mm in Anderson Lake). Peamouth also spawn in the spring (threshold temperature is approx. 9°C). Some populations spawn in lakes over gravel beaches, but most lacustrine populations spawn in inlet or outlet streams. Peamouth are insectivores and are primarily water column foragers, but they can also take insect prey from the bottom and surface (McPhail 2007).

Redside shiner (*n*= 260) and bridgelip sucker (*n*= 108) have been more abundant in the Seton Lake catches (*n*= 228 and 98, respectively) than in Anderson Lake (*n*= 32 and 10, respectively), and have ranged in size from 79 to 122 mm, and 97 to 472 mm, respectively. Both species are spring spawners (mid April to mid June). Redside shiners can spawn in lakes or streams, whereas bridgelip suckers spawn in streams only. Shiners primarily eat aquatic and terrestrial insects from the bottom, mid-column, or surface of the littoral zone. Food sources for suckers include periphyton, filamentous algae, and detritus.

Mountain whitefish (*n*= 23, all years combined) have been captured in equivalently low numbers in each lake. In 2021 only two individuals (219 and 379 mm fork length) were captured in Anderson Lake. Across years the catches have ranged in size from 120 to 403 mm. Mountain whitefish can exhibit different life history patterns: lacustrine, riverine, or adfluvial. They are fall spawners (October and November), and most lake populations migrate into streams to spawn. The main food items for mountain whitefish include plankton, snails, surface insects, and occasionally, young fish (McPhail 2007).

Each of these species have diverse life histories, are generally less directly sensitive to water quality and in-lake habitat changes than the target salmonid species, contribute limited abundance to the sample in most cases, and tend to have less social value relative to gwenis, bull trout and rainbow trout. Due to the combination of these factors, none of these "other" species have been selected as target species for monitoring the effects of diversion operations by this monitoring program. Going forward, their catch information will continue to be fully documented; however, analyses and discussion of these other resident species will largely be limited to opportunistic observations from the collected data.

MQ 2: Will the selected alternative (N2-2P) result in positive, negative or neutral impact on abundance and diversity of fish populations in Seton Lake?

Relative to the Year 1 and 2 results, the gill netting method employed in Years 3 to 9 (2015 to 2021) has proved much more effective for capturing target species, particularly gwenis and bull trout, and has been much better suited for establishing an annual index of abundance for target species that can be compared across the duration of the program. So far, the annual abundance index for gwenis has been relatively stable and low in Anderson Lake across the years of monitoring available to-date (2015 to 2021; CPUE range = 0.6 to 2.7 fish/10 net-hours). The index for gwenis has been consistently higher in Seton Lake and was relatively stable from 2015 to 2017 (CPUE range = 44.9 to 63.1 fish/10 net-hours), then increased by between 2- to 4-fold in 2018 and 2019 (CPUE = 200.2 and 140.4 fish/10 net-hours, respectively), returned to a similar level as those first three years of gill netting results in 2020 (CPUE = 56.5 fish/10 net-hours), and then increased in 2021 (CPUE = 100.5 fish/10 net-hours). These results suggest that the selected alternative (or the range of operations implemented since the start of this monitoring program) have coincided with a net positive trend in the abundance index for gwenis in Seton Lake; however, this is currently heavily influenced by the high index values in 2018, 2019 and 2021 so it is not feasible to rule out that the impact could be neutral by the end of the monitor if the CPUE value for the remaining year (2022) is lower. With the data points currently available, it seems most apparent that operations have not caused a net negative impact on this species within the period of monitoring to-date.

The potential reasons for the increase in abundance index results for Seton Lake in 2018 and 2019 are varied and may include: dominant/sub-dominant cycle of years for parent populations of sockeye, changes to in-lake habitat quality related to diversion operations and/or natural factors, differences in abundance of predator species (i.e., bull trout and northern pikeminnow) among years, etc. Each of these factors could have a direct or indirect influence on recruitment, growth rate and survival for gwenis. However, understanding the relative significance of the observed differences in the abundance index among years requires a long enough time series to put the observed highs and lows into context, determine long-term patterns or trends, and allow for correlation of the timing of changes with potential causal factors (including replication). The gill netting method for the annual population index of target species has been employed for seven years to-date (or six years based on short-duration sets covering the full length of each lake). However, final determination of whether the implemented operations have influenced the abundance and diversity of fish populations in Seton Lake will be reserved for the final report after Year 10 (2022).

Gwenis continued to be the best-suited resident species for trend monitoring in Seton Lake for the following reasons: a) their ecological and social value, b) the fact that they carry out their entire life cycle within the lake, and c) their potential for response to diversion effects. In addition, the Seton and Anderson populations may not mix such that the indices of abundance and size, etc. may specifically link to the conditions within the respective lake where they reside.

Due to their importance as a top predator species, and direct interaction with gwenis as a prey species in both lakes, bull trout are considered the next most important of the target resident species to directly monitor as a part of this program. For instance, it may be possible that changes to the abundance of bull trout could have an impact on the abundance of prey species (e.g., gwenis), so the abundance index for this species may be useful for informing whether this is a factor in gwenis population trends across the duration of the monitor. Or, conversely, bull trout abundance among years could be influenced by population trends for gwenis. Overall, catch rates for bull trout in 2021 were on par with every other available year except 2017 (in Seton Lake) and 2016 and 2017 (in Anderson Lake) which were significantly higher. We made a preliminary attempt to plot a regression between bull trout CPUE and gwenis CPUE among years up to 2021 (see Figure 3.15), but the slopes were fairly flat and the correlations were weak based on the available data.

In terms of their suitability as a species for monitoring operations effects, however, it's important to acknowledge that the bull trout in these lakes are adfluvial, and evidence suggests that they are migratory: opportunistically moving out of, and back into, Seton Lake according to where feeding opportunities are throughout the system at different times of year, or either lake for spawning purposes. Bull trout may move between Seton and Anderson lakes as well, but the number of recaptures has been too limited for detecting these movements to-date. Three bull trout that had been captured and tagged in Year 4 (2016) were recaptured in Year 5 (2017), but each of these fish were in the same lake as their original capture event, and 2 of them were recaptured in the exact same locations. Nonetheless, because bull trout have a propensity to access habitats and resources outside of the study area as needs dictate, changes or differences in abundance or life history characteristics for this species may be less directly linked to impacts associated with the Carpenter diversion operations.

In terms of the species diversity aspect of MQ 2, gill netting is an effective method for sampling a broad range of species and size classes, and providing information on their relative abundance, distribution and habitat use (see response to MQ 1). As of Year 9 (2021) results, there have been no dramatic shifts in species composition or the relative abundance of species detected in either lake based on our sampling. Results for all non-target species sampled will continue to be collected but will be considered more as incidental and supplementary information relative to the results for gwenis and bull trout.

Three important comments about implications of the scope, approach and methods being implemented for this program are as follows:

1) We are not monitoring a "before-after" treatment scenario with a distinct change in operations divided into representative sample sizes for each treatment. One of the main

objectives defined for this program was to monitor *existing* operations (with inherent variability among years) and assess for any changes in fish population across the monitoring period (i.e., does the general trend appear to be increasing, decreasing or staying the same under N2-2P operations). However, this does not specifically set up any known amount of replication for the potential range of operations (diversion magnitude and timing) that may be required for confirming differences or changes.

2) The resolution for detecting change may also be an issue. As acknowledged in Section 1.5, sampling programs such as this one typically face challenges achieving statistical certainty given low (or undefined) capture efficiencies and the complex interaction of variables affecting recruitment, growth and survival of fish populations. Within the limits of the existing scope and budget, this program may be able to detect large-scale changes (which may not necessarily result from the extent of operations effects within the study timeframe), but not smaller ones that may be more likely. This is possibly true regardless of which specific age class(es) we focus on for gwenis, and operations effects on bull trout are likely more indirect (than for gwenis) due to life history differences among these species.
3) There may be an issue with the number of years of data available for linking resident fish population trends with the measured habitat metrics (i.e., temperature, sedimentation) related to diversion operations by the end of the 10-year monitoring period in 2022. The time lag between potential causal factors (e.g., diversion operations effects on spawning) and measured population response (e.g., gwenis abundance index at Age-3) may limit the number of data points for a regression to assess potential correlations with adequate replication. See more on this potential issue described in the response to MQ 3, below.

MQ 3: Is there a relationship between the quality, quantity, and timing of water diverted from Carpenter Reservoir on the productivity of Seton Lake target resident fish populations?

As explained in the results, sampling methods changed across the early years (Years 1-3; 2013-2015) of the monitoring program from boat electrofishing to gill netting, and then from overnight gill netting to short duration gill netting based on learning generated about species composition, abundance, distribution and sampling mortality from those efforts. For every year since (i.e., Years 4-9; 2016-2021), the sampling approach and methodology have remained consistent, providing results that are directly comparable among the two lakes (i.e., Seton and Anderson) and among those years. For the final remaining monitoring year ahead, the plan is to maintain the current sampling methods and effort to ensure the direct comparability of results across years for addressing the management questions, within the limits of our control.

As of this report, there are currently 7 years of comparable data for fish abundance and distribution, but fewer than that for linking abundance with in-lake habitat quality metrics (temperature & sedimentation) (Table 4.1). By the end of the prescribed 10-year monitoring period, there will be 8 years of comparable fish population information, and 3-4 years of sedimentation rate and temperature profile data (respectively) that can be linked to potential population response for gwenis given the 4-year timeframe (i.e., spawning year to Age-3) between cause (e.g., sedimentation in Year$_t$) and effect (e.g., gwenis abundance in Year$_{t+4}$) based on their life history and the age at sampling. However, based on the data currently available, the relationship between diversion flow volume and sedimentation rate is becoming clearer (see Figure 3.8). As such, it may be possible to reasonably back-cast expected sedimentation rates given known inflow volumes for years prior to sediment sampler deployment (i.e., 2012–2015). This approach may allow us to fill gaps from early in the program to support synthesis analyses across all eight years (listed in Table 4.1) at the end of the monitor. However, since the full set of monitoring data will be required for these analyses to be meaningful for addressing this management question, we decided not to include this in the Year 9 (2021) report.

Table 4.1 Summary of the time lag (i.e., 4 years) between measurement of effect variables (e.g., diversion volume, temperature, sedimentation rate) and the measurement of potential gwenis population response (e.g., abundance index at Age-3) across the 10-year monitoring period for BRGMON-8. Comparable years of response variable data available to-date are bolded.

Effect Variable Measured during Gwenis Spawning Year			Response Variable Measured in Fish Capture Year (at Age-3)
Diversion Operations	Temperature	Sedimentation	
2011	- [a]	- [a]	**2015 (Year 3)[b]**
2012	-	-	**2016 (Year 4)**
2013	-	-	**2017 (Year 5)**
2014	-	-	**2018 (Year 6)**
2015	2015	-	**2019 (Year 7)**
2016	2016	2016	**2020 (Year 8)**
2017	2017	2017	**2021 (Year 9)**
2018	2018	2018	2022 (Year 10)

[a] Note that temperature profile monitoring started in Year 3 (2015) and sedimentation monitoring started in Year 4 (2016).

[b] Note that fish population index sampling was done by boat electrofishing in Years 1 and 2 which did not provide comparable results to the gill netting method that began in Year 3 (2015).

There have been a range of diversion operations across monitoring years to-date (see Figure 3.1 lower panel and Figure 3.2 in Section 3.1). Based on a comparison among years, the total discharge volumes from BR1 and BR2 into Seton Lake were lowest in 2013, 2014, 2018 and 2020 (2,457, 2,415, 2,335 and 2,441 million m^3, respectively), in the middle of the range in 2017 and 2019 (2,515 and 2,561 million m^3, respectively), and highest in 2015, 2016 and 2021 (2,661, 2,765 and 2,674 million m^3, respectively). Similarly, there has been variation in which season had the highest diversion volumes among years as well (see Table 1.1 in Section 1.4). These differences in inflow magnitudes among seasons and years may provide an effect signal that is detectable in the selected response variables being monitored by this program. However, since the full number of years of replicate data is not yet available, as described above, we cannot confirm if that is the case at this point. The following discussion provides some of the information pertaining to this question that we have learned to-date.

Some concern was raised during the WUP process that fluctuations in the lake surface elevation may have the potential to impact Gwenis spawning locations based on the assumption that selected spawning habitats may occur at elevations within the lake surface elevation range (i.e., that they spawn in shallow habitats near the shoreline, as occurs for some lake spawning populations of kokanee). Gwenis spawn timing has been observed to occur in fall (Morris et al. 2003, Limnotek 2016, and this program) in Seton Lake. To-date there is no evidence for shore-spawning use in Seton Lake; but, rather, that gwenis spawn at depth and would not be directly impacted by the degree of surface elevation changes implemented under N2-2P operations.

Based on thermal profile monitoring, temperatures have tended to be warmest at the outflow end of Anderson Lake, which of course is not affected by the generally colder, turbid inflows from the diversion. The inflow end of Seton Lake was the coolest (particularly in the epilimnion and mesolimnion layers compared to the other stations) and most turbid of the monitored locations since it is the most directly affected by the diversion operations. Temperatures at the outflow end of Seton Lake were most similar to the outflow of Anderson Lake, but slightly cooler across most of the profile.

There has also been evidence of some differences in the depths of the various thermal layers among the three temperature profile monitoring locations (within the limitations of the depth intervals of the temperature loggers). The depths of the epilimnion and mesolimnion layers have tended to be broadest at the Anderson Lake outflow station and narrowest at the Seton Lake inflow station. The metalimnion layer at the outflow of Seton Lake has generally been slightly narrower than in Anderson and deeper than the inflow end of Seton. These differences are also likely due to the effects of the diversion, which were particularly evident at the inflow end of Seton Lake.

The main source of natural inflows to the top end of Seton Lake are from Portage Creek which draws directly from the epilimnion layer at the outflow end of Anderson Lake (also receiving inputs from Whitecap and Spider creeks). As such, the temperatures from this source could be expected to be similar to the Anderson Lake regime. However, Portage Creek and other natural inputs only contribute approximately ≤10% of the inflows to Seton Lake relative to the Carpenter diversion flows from BR1 and BR2, which contribute ≥90% of total volume. Therefore, it is clear that the colder temperatures documented at the inflow end of Seton Lake are a direct result of the diversion influence (see the similarity between the orange and purple lines in Figure 3.6; Section 3.1). The diversion effects on temperature in Seton Lake are most acute at the inflow end nearest the BR1 and BR2 Generating Station inputs and there is a gradient of effect across the length of the lake, particularly during the period of thermal stratification (i.e., spring and summer months). The influence is still apparent at the outflow end of Seton Lake but is mitigated by normal lacustrine thermal processes relative to the inflow end. Since the volume of the diversion inputs dominate the natural inflows to such a large extent throughout the year, it is unlikely that anything short of major changes to diversion operations and the volume of inputs would change the current temperature profiles and gradient across the lake. In other words, minor tweaks to the quantity and timing of diversion operations are unlikely to have a substantial effect on the existing temperature patterns.

Sediment inputs from the Carpenter diversion that settle on the bottom of the lake have the potential to impact gwenis production by covering or infiltrating spawning substrates over time. The particle sizes deposited could potentially affect oxygen exchange for the buried eggs, which could, theoretically, result in reduced recruitment success and declining population abundance over time. However, this effect has not been observed as the index of gwenis population abundance has been stable, or even increased (based on the 2018, 2019 and 2021 data points) across the available monitoring years to-date. It is also possible that gwenis spawning activities (assuming some form of redd excavation similar to stream-spawning salmon) or upwelling groundwater from aquifers could be “cleaning” the substrates in selected areas and maintaining them, thereby enabling the spawners to return to the same locations each year. While it’s not possible for our sampling to document all of the potential spawning locations in the lake, if there are consistent trends in the relative distribution of gwenis just prior to spawning (e.g., higher proportions in the mid and outflow ends of the lake than the inflow end, as has been observed to-date) that may contribute to a weight of evidence suggesting that areas subject to higher sediment deposition correlate with reduced spawning use. Anything beyond that would likely be beyond the current scope and budget of this program.

Based on six years of data collection to-date, sedimentation rate was lowest in Anderson Lake and highest at the inflow end of Seton Lake. Also, there was a gradient of effect across the length of Seton Lake. These patterns were consistent across seasons, but sediment inputs to Seton Lake were highest in spring (i.e., mass per day and mass per diversion volume). Sedimentation rates in summer were generally higher than fall/winter, except at the outflow end of both lakes. Sedimentation rates in 2021 were generally on par or higher than in previous years due to generally higher diversion inflow volumes (particularly in spring) that year. There is a positive correlation between diversion volume (which is also seasonal) and sedimentation rate (see Figure 3.8 in Section 3.1), and the slope of this relationship is steepest at the inflow end of Seton Lake (i.e., 0.0047) with a decreasing gradient across the mid (i.e., 0.0025) and outflow (i.e., 0.0017) sections.

We also plotted regressions of Carpenter Reservoir elevation (which is also seasonal) with sedimentation rate to see if lower reservoir elevations contribute to higher volumes of sediment being entrained. This relationship is also pertinent because Carpenter Reservoir has been drawn down more deeply in some years (i.e., ~615-618 masl in 2017–2020) compared to 2015 and 2016 (i.e., ~631 m and 633 m, respectively) in order to manage spill magnitude at Terzaghi Dam. Minimum Carpenter Reservoir elevations in 2021 were ~621 m. There is a negative slope to this correlation; however, there was more scatter among the points (reflected by the lower R^2 values; see Table 3.5) relative to the sedimentation versus diversion volume relationships (Table 3.4). In other words, the measured sedimentation rates in Seton Lake were likely influenced by Carpenter Reservoir elevation, but this relationship was weaker, likely because diversion volume and the seasonal aspect of sediment transport are more significant drivers.

It is likely that the introduction of fine sediment particulates into Seton Lake (and resulting turbidity) associated with the diversion from Carpenter Reservoir contributes to the different biological characteristics and spatial distribution patterns of gwenis and bull trout observed between the lakes, as described for the Year 3 to 9 (2015 to 2021) datasets. While the potential for effects of sedimentation on gwenis production seems fairly direct (e.g., deposition on spawning substrates on the lake bottom), the effects on bull trout, which are adfluvial and migratory, may be less direct relative to the availability and spatiality of feeding opportunities in each context. Ideally, the cumulative dataset on physical habitat parameters (including diversion volume and sediment inputs) and population abundance indices collected across the full duration of this monitoring program will shed light on these linkages for these two focus species.

Specifically, once the full set of this data are in hand, it may be possible to investigate any correlation between gwenis abundance index in $Year_t$ with sedimentation rate in $Year_{t-4}$ (i.e., spawning year based on the evidence that mature fish in Seton Lake are 3 years old). Also, evaluating gwenis distribution patterns (i.e., possible spawning distribution in Seton Lake during the survey period) with temperature characteristics and sediment deposition will also be informative for determining effects of the diversion operations on this target species. The additional data points for these components provided by the upcoming data collection in 2022 should provide a sufficient number for conducting these analyses. As such, we plan to assess and include these correlations in the Year 10 report.

For the remaining year of the program, the continued collection of temperature and sedimentation data in Seton and Anderson lakes will support analysis of correlations between temperature characteristics or sedimentation rate with the inflow volume of the Carpenter diversion on both seasonal and annual bases. This analysis will be useful for characterizing the seasonality of diversion effects and support recommendation of potential refinements to the N2-2P operating alternative as a part of addressing management question 4 (below) by the end of the monitoring period. Relevant results & analysis from BRGMON-6 will also be incorporated with the results from this program by the end of the study period (i.e., 2022) to inform the response to these questions.

MQ 4: Can refinements be made to the selected alternative to improve habitat conditions or enhance resident fish populations in Seton Lake?

This management question will be evaluated based on insights gained from results under management questions 1-3. It is not expected that this question will be able to be answered until the completion of monitoring in 2022.

5. Recommendations

The following recommendations are provided based on the learning generated by this monitoring program to-date. The recommendations following the Year 9 (2021) data collection activities focus on maintaining the sampling approach established over the last several years with only minor tweaking to improve precision where possible. Implementation of these recommendations for the final monitoring year in 2022 are intended to preserve the integrity of the results for answering the management questions within the allocated budget framework.

- Repeat fish population index sampling using gill nets in pelagic and littoral habitats of both Seton and Anderson Lakes. Ensure that sample timing, effort, and methods remain as consistent and comparable as possible for all remaining monitoring years.
- Focus analysis efforts on gwenis and bull trout as target species for assessing potential linkages with operations effects. Continue documenting sampling results for other resident species to support species composition and biological characteristics information under MQ 1.
- Continue year-round thermal profile monitoring and sedimentation rate monitoring as initiated in 2015 and 2016, respectively. Ensure sample timing, effort, and methods remain as consistent and comparable as possible to facilitate comparisons among years and operational ranges.
- Starting at the end of Year 7 (2019), loggers deployed on the thermal profile arrays in each lake at 60 m and 70 m depth were moved up and 12 additional loggers were purchased to reduce the spacing between temperature measurements within the epilimnion and mesolimnion so temperatures and depth ranges for these layers can be defined with more precision. With these changes, the revised logger depth intervals on each array were: 1, 5, 10, 12.5, 15, 17.5, 20, 22.5, 25, 30, 35, 40 and 50 m below the surface. As such, we have discontinued logging measurements at 60 m and 70 m since temperatures at these depths tended to remain the most stable across the year and consistent among locations. Going forward, we intend to maintain this set up of loggers on the arrays for the remaining monitoring year in 2022.
- Continue to evaluate the success of BRGMON-8 data collection methods for their capacity to provide relevant information for answering the management questions. Potential issues include unknown replication of different operations among years (i.e., lack of designated “treatments”), coupled with potentially limited precision or resolution to detect small changes in the abundance of target species, which may limit the strength of conclusions. Also, the time lag between measurement of effects variables and resulting fish population response (as described in the Discussion for MQ 3), may limit the number of years available for evaluating linkages by the end of the monitoring period for this program in 2022.

6. References

BC Hydro. March 2015. Bridge-Seton Water Use Plan – Monitoring Program Terms of Reference. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring – Addendum 1. Prepared for the BC Comptroller of Water Rights.

BC Hydro. January 2012. Bridge-Seton Water Use Plan – Monitoring Program Terms of Reference. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring. Prepared for the BC Comptroller of Water Rights.

BC Hydro. 2011. Bridge River power development water use plan: revised for acceptance for the Comptroller of Water Rights. pp. 1–79.

BC Hydro. 1993. Project Report: Bridge River Operation Study. Hydroelectric Engineering Division, Report No. H2721, 19 p.

B.C. Ministry of Environment, Lands and Parks. 1997. Fish Collection Methods and Standards – Version 4.0. Prepared by the Fish Inventory Unit for the Aquatic Ecosystems Task Force, Resource Inventory Committee.

Burnett, N.J., and E. Parkinson. 2018. Bull Trout and Rainbow Trout Tagging and Assessment of Fish Entrainment in Seton Lake Reservoir – Final Report. Report prepared for BC Hydro by InStream Fisheries Research Inc. and Eric Parkinson. 51 p. + 3 Apps.

Hebert, A.S. 2019. Reproductive ecology and habitats of deep spawning Kokanee and Sockeye Salmon Oncorhynchus nerka. Master's thesis, The University of British Columbia, Vancouver, British Columbia.

Hendry, A.P., J.E. Hensleigh, and R.R. Reisenbichler. 1998. Incubation temperature, developmental biology, and the divergence of Sockeye Salmon (Oncorhynchus nerka) within Lake Washington. Canadian Journal of Fisheries and Aquatic Sciences 55:1387–1394.

Limnotek. 2016. Seton Lake aquatic productivity monitoring (BRGMON6): Progress in 2015-16. Report prepared for BC Hydro, May 2016. 172p.

Limnotek. 2015. Seton Lake aquatic productivity monitoring (BRGMON6): Progress in 2014-15. Report prepared for BC Hydro, March 2015. 81p.

Mackay, W.C., G.R. Ash and H.J. Norris. 1990. Fish ageing methods for Alberta. R.L. & L. Environmental Services Ltd. in association with Alberta Fish and Wildlife Division and University of Alberta, Edmonton. 133p.

McPhail, J.D. 2007. The Freshwater Fishes of British Columbia. The University of Alberta Press, Edmonton, Alberta, Canada. ISBN 978-0-88864-467-1.

Moreira, A.L. 2014. The evolutionary origin of “black” kokanee (Oncorhynchus nerka) and their genetic and phenotypic diversity in the Anderson and Seton lakes system. M.Sc. Thesis University of British Columbia, Vancouver. 123p.

Morris, A.R., E. Braumandl, H. Andrusak, and A. Caverly. 2003. 2002/2003 Seton and Anderson Lakes Kokanee Assessment – Feasibility Study and Study Design. Prepared for British Columbia Conservation Foundation and Ministry of Water, Land and Air Protection.

Perrin, C., N. Swain, M. Hocking, P. Dinn, T. Hatfield and A. Marriner. 2016. JHTMON4: Upper and Lower Campbell Reservoirs Littoral Productivity Assessment Year 1 Annual Monitoring Report. Draft V2. Consultant’s report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership, Limnotek Research and Development Inc. and Ecofish Research Ltd., July 15, 2016.

Sneep, J. 2022. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 8 (2020) Results. Annual Report prepared for St’at’imc Eco-Resources, November 2022.

Sneep, J. 2021. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 7 (2019) Results. Annual Report prepared for St’at’imc Eco-Resources, January 2021.

Sneep, J. 2019a. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 6 (2018) Results. Annual Report prepared for St’at’imc Eco-Resources, November 2019.

Sneep, J. 2019b. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 5 (2017) Results. Annual Report prepared for St’at’imc Eco-Resources, January 2019.

Sneep, J. 2018a. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 4 (2016) Results. Annual Report prepared for St’at’imc Eco-Resources, June 2018.

Sneep, J. 2018b. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, Year 3 (2015) Results. Annual Report prepared for St’at’imc Eco-Resources, February 2018.

Sneep, J. 2015. BRGMON-8 Seton Lake Resident Fish Habitat and Population Monitoring, 2013 and 2014 Results. Annual Data Report prepared for St’at’imc Eco-Resources, March 2015.

Wade, N.H. and H. Taylor. 1979. Control of artesian pressure by bleeder wells. Can. Geotech J., 16, 488-496 (1979).

Whitlock, S.L., M.R. Campbell, M.C. Quist, and A.M. Dux. 2018. Using Genetic and Phenotypic Comparisons to Evaluate Apparent Segregation among Kokanee Spawning Groups. Transactions of the American Fisheries Society 147:43–60, 2018.

Appendix A – Fish Sampling (Gill Net) Site Locations

Table A1 Habitat type, set depth and UTM coordinates for each gill net location in Seton Lake (left) and Anderson Lake (right) during Year 9 (2021).

Seton Lake

Site Name	Habitat Type	Set Depth	Easting	Northing
SL21-01	littoral	bottom	569919.72	5614592.62
SL21-02	littoral	bottom	569714.00	5614737.56
SL21-03	littoral	bottom	569549.02	5614773.29
SL21-04	littoral	bottom	568123.41	5614971.96
SL21-93	littoral	bottom	568187.39	5614052.94
SL21-05	littoral	bottom	570252.68	5613197.78
SL21-07	pelagic	0m	569921.59	5613585.60
SL21-08	pelagic	20m	570020.23	5613512.85
SL21-09	pelagic	25m	570124.29	5613464.31
SL21-10	littoral	bottom	560653.78	5615976.56
SL21-11	littoral	bottom	560585.75	5615689.85
SL21-12	littoral	bottom	561189.04	5614942.08
SL21-13	littoral	bottom	561376.26	5614786.36
SL21-14	littoral	bottom	561863.76	5614705.91
SL21-15	littoral	bottom	562945.49	5614480.60
SL21-31	littoral	bottom	563271.42	5616153.93
SL21-95	littoral	bottom	562780.34	5616148.23
SL21-16	pelagic	0m	560972.95	5615556.56
SL21-17	pelagic	20m	561027.72	5615482.02
SL21-18	pelagic	25m	561097.44	5615386.08
SL21-21	littoral	bottom	558943.32	5618974.35
SL21-22	littoral	bottom	557857.95	5619227.45
SL21-23	littoral	bottom	557581.87	5619449.94
SL21-24	littoral	bottom	556532.14	5619700.59
SL21-28	littoral	bottom	553666.89	5619983.75
SL21-29	littoral	bottom	551547.21	5617449.78
SL21-25	pelagic	0m	553925.47	5619249.10
SL21-26	pelagic	20m	554038.75	5619237.37
SL21-27	pelagic	25m	554134.67	5619280.29
SL21-19	littoral	bottom	560818.09	5617785.19
SL21-20	littoral	bottom	560253.42	5618436.91
SL21-33	littoral	bottom	571411.79	5613716.82
SL21-94	littoral	bottom	570417.57	5614280.55

Anderson Lake

Site Name	Habitat Type	Set Depth	Easting	Northing
AL21-01	littoral	bottom	549208.58	5616702.29
AL21-02	littoral	bottom	548991.27	5617153.75
AL21-03	littoral	bottom	548211.82	5616672.88
AL21-04	littoral	bottom	547326.62	5616037.81
AL21-05	littoral	bottom	548249.84	5614870.47
AL21-06	littoral	bottom	546643.92	5613345.52
AL21-30	littoral	bottom	549158.68	5616119.77
AL21-31	littoral	bottom	549242.09	5616923.35
AL21-07	pelagic	0m	547495.72	5615687.16
AL21-08	pelagic	20m	547439.46	5615628.26
AL21-09	pelagic	25m	547380.73	5615529.87
AL21-10	littoral	bottom	542678.38	5612030.73
AL21-11	littoral	bottom	542584.79	5611813.23
AL21-17	littoral	bottom	542898.35	5610175.99
AL21-18	littoral	bottom	542107.81	5608958.69
AL21-12	littoral	bottom	541235.74	5610242.15
AL21-13	littoral	bottom	540595.35	5609250.56
AL21-35	littoral	bottom	541095.79	5610235.16
AL21-36	littoral	bottom	541372.34	5610708.83
AL21-14	pelagic	0m	542229.40	5609299.17
AL21-15	pelagic	20m	542297.78	5609361.66
AL21-16	pelagic	25m	542362.01	5609456.48
AL21-26	littoral	bottom	541831.43	5607012.99
AL21-27	littoral	bottom	540278.01	5606847.93
AL21-37	littoral	bottom	539735.28	5602644.27
AL21-20	littoral	bottom	538309.32	5601226.43
AL21-21	littoral	bottom	538102.08	5600384.25
AL21-22	littoral	bottom	537375.76	5600150.97
AL21-23	pelagic	0m	537172.87	5601931.06
AL21-24	pelagic	20m	537223.61	5602006.70
AL21-25	pelagic	25m	537298.70	5602092.97
AL21-33	littoral	bottom	537955.26	5603055.00
AL21-34	littoral	bottom	537453.66	5602578.69