Campbell River Project Water Use Plan

Upper Campbell, Lower Campbell and John Hart Reservoirs and Diversion Lakes Littoral versus Pelagic Fish Production Assessment

Implementation Year 2

Reference: JHTMON-5

Year 2 Annual Monitoring Report

Study Period: March 1, 2015 to April 30, 2016

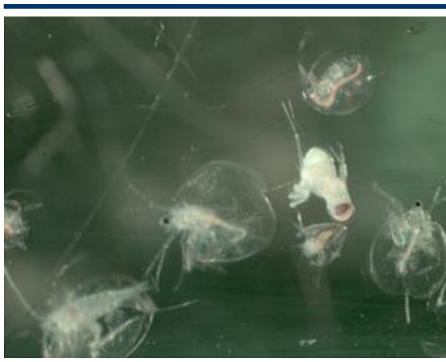
Laich-Kwil-Tach Environmental Assessment Ltd. Partnership
and Ecofish Research Ltd.

October 17, 2016

JHTMON-5: Littoral versus Pelagic Fish Production Assessment

Year 2 Annual Monitoring Report



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EXECUTIVE SUMMARY

Water Use Plans (WUPs) were developed for all of BC Hydro's hydroelectric facilities through a consultative process and monitoring is being undertaken to address outstanding management questions in the years following implementation of a WUP. As the Campbell River Water Use Plan process reached completion, a number of uncertainties remained with respect to the effects of BC Hydro operations on aquatic resources. The *Upper Campbell, Lower Campbell, John Hart Reservoirs and Diversion Lakes Littoral versus Pelagic Fish Production Assessment* (JTHMON-5) is part of wider monitoring of the Campbell River WUP. JTHMON-5 is designed to assess the extent to which fish production is driven by littoral vs. pelagic production and how this relates to BC Hydro operations.

The Campbell River WUP project area is complex and includes facilities and operations in the Campbell, Quinsam and Salmon watersheds. In addition to the mainstem rivers, there are three large reservoirs, nine diversion lakes influenced by water diverted from the Quinsam and Salmon rivers, and many tributaries and small lakes that are not directly affected by operations. During development of the Campbell River WUP, the Fish Technical Committee (FTC) hypothesized that fish production in Upper and Lower Campbell reservoirs was negatively impacted by large fluctuations in water level through its effect on littoral production. The FTC also hypothesized that short water residence time of the diversion lakes as a result of the BC Hydro diversion operations could negatively impact pelagic productivity.

The JTHMON-5 monitoring program aims to address the following two management questions:

1. To what extent do stabilized reservoir levels, as affected by BC hydro operations, benefit fish populations?
2. What is the relationship between residence time (as affected by diversion rate) and lake productivity?

JHTMON-5 is scheduled for 10 years and has two components: stable isotope analysis of food webs in reservoirs and diversion lakes, and production estimates of pelagic bacteria in reservoirs and diversion lakes. This report presents data from Year 2 of the stable isotope analysis of food webs component. Under the current TOR, sampling using stable isotope methods is scheduled for years 1, 2 and 3 of JHTMON-5, with a potential for a fourth year of sampling to be determined at the end of Year 3. Estimates of pelagic bacteria as an indicator of pelagic productivity will be addressed in years 7, 8, and 9 and thus will be discussed in future years of the program.

Substantial information regarding the structure and functioning of lake food webs can be gained by using stable isotopes to reconstruct the diets of lake biota. JHTMON-5 uses stable isotope analysis of nitrogen and carbon in fish tissues and their potential diet items to assess relative energy flows to fish from littoral vs. pelagic areas. Sampling in Year 2 was completed for Lower Campbell Reservoir, Beavertail Lake, Snakehead Lake and Upper Quinsam Lake. Lower Campbell Reservoir was chosen because it experiences some fluctuations in water levels but less than those observed at Upper Campbell Reservoir, which was sampled in Year 1. Beavertail Lake, Snakehead Lake and Upper

Quinsam Lake were chosen because they are either part of the Quinsam River diversion or are nearby control lakes. Snakehead Lake is a receiving lake that is part of the same diversion as Middle Quinsam (donor lake) and Gooseneck Lake (receiving lake) that were sampled in Year 1. Upper Quinsam is above the water diversion and is therefore a control lake. Beavertail is a nearby control lake.

The primary species of interest in JHTMON-5 are Cutthroat Trout (*Oncorhynchus clarkii*) and Rainbow Trout (*O. mykiss*). Sampling was designed to improve understanding of the diets and energy sources of these two fish species, which are the resident fish species of primary management concern in reservoirs and lakes of the Campbell River system. Additional sampling of Dolly Varden (*Salvelinus malma*) was also completed. Gill netting, trap netting and minnow trapping was completed in June through October of 2015 to obtain representative tissue samples from Cutthroat Trout, Rainbow Trout, Dolly Varden and their prey fish including Threespine Stickleback (*Gasterosteus aculeatus*), Sculpin spp. (*Cottus* spp.), and juvenile trout (*Oncorhynchus* spp.) from Lower Campbell Reservoir and Beavertail, Snakehead and Upper Quinsam lakes. Primary diet items for Cutthroat Trout and Rainbow Trout also include zooplankton (pelagic source), and benthic/littoral, stream and terrestrial invertebrates (littoral source). Invertebrate sampling occurred in June, July and August to obtain representative samples from the four lakes. Invertebrates were sorted and counted in the laboratory to order and family by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society).

Invertebrate and fish samples were processed for nitrogen and carbon stable isotopes at the Stable Isotope in Nature Laboratory located within the Canadian Rivers Institute at the University of New Brunswick in Fredericton, New Brunswick. A total of 246 samples of invertebrates and fish were sent for analysis. The relative contributions of pelagic vs. littoral sources to Cutthroat Trout, Rainbow Trout and Dolly Varden diets were assessed through dual isotope ($\delta^{13}C$ and $\delta^{15}N$), four to six-source Bayesian isotopic mixing models implemented in the program SIAR (Stable Isotope Analysis in R). SIAR takes isotope data from consumers (fish) and sources (diet items) along with estimates of diet-tissue isotopic fractionation, and fits Bayesian models that estimate source contributions to diet.

In the Year 1 report, we recommended that water residence time be determined for all lakes so that estimates of littoral versus pelagic production in the target fish species could be modelled as a function of water residence. Water residence time was calculated for all lakes sampled in Year 1 and Year 2 of JHTMON-5 during Year 2. Water residence time was calculated for the whole year and for the growing season stratified period using a water balance method. To estimate residence time during the stratified period, an effective lake volume was used, which is defined as the average volume of the surface mixed layer (epilimnion) times the proportion of year that each study lake becomes stratified. Based on historical lake temperature records, thermal stratification was assumed to establish within each lake around May 15 and to break down around September 30.

Nitrogen and carbon stable isotope signatures of all fish and invertebrates were similar among all seven lakes and reservoirs sampled in Year 1 and Year 2. Large Cutthroat and Rainbow trout had the highest $\delta^{15}N$ levels consistent with their top position within lake food webs, followed by smaller prey fish with intermediate trophic level positions. Zooplankton had the lowest $\delta^{13}C$ levels consistent with their pelagic habitat, while littoral, stream, and terrestrial invertebrates had higher $\delta^{13}C$ isotopic signatures, consistent with their terrestrial and littoral sources of carbon in diet. Among the small prey fish, Threespine Stickleback had the lowest $\delta^{13}C$ levels indicative of a pelagic dominated diet.

Nitrogen and carbon stable isotope signatures in bulk zooplankton varied by month of collection. Across all lakes, $\delta^{15}N$ signatures in zooplankton were significantly higher in August compared to June or July while $\delta^{13}C$ signatures were significantly higher in both July and August compared to June.

In Lower Campbell Reservoir, only 8% of Cutthroat Trout diet is estimated to be derived from pelagic sources and 92% is estimated to be derived from littoral sources, while Rainbow Trout have a pelagic contribution to diet of 23% and a littoral contribution to diet of 77%. In contrast, in Year 1 of data collection from Upper Campbell Reservoir, 26% of the diet of Cutthroat Trout was estimated to be derived from pelagic sources, while 74% was estimated to be derived from littoral sources. Rainbow Trout were estimated to have a pelagic contribution to diet of 44% and a littoral contribution to diet of 56%.

Analysis confirmed the initial hypothesis that top fish consumers have a reduced littoral contribution to diet in Upper Campbell Reservoir compared to Lower Campbell Reservoir. Upper Campbell Reservoir has greater fluctuations in water levels than Lower Campbell Reservoir, which may reduce littoral production for fish. However, there are other factors that may explain the patterns we observed. For example, the seasonal water residence time at Lower Campbell Reservoir was found to be one of the shortest among all study lakes, and possibly shorter than Snakehead Lake, which is less than 1/100th its area. A short water residence time is due to the relatively large inflows and outflows of water, which create conditions during the stratification period that may limit pelagic production. Despite the large pelagic areas of each reservoir, the top fish consumers in both reservoirs appear to be supported by littoral production to a greater extent than pelagic production.

Shorter water residence time is hypothesized to decrease the pelagic contribution to fish diets in diversion lakes and ultimately decrease fish production. There was some evidence to support this hypothesis, although a more complete synthesis analysis will be conducted after Year 3 of sampling. Estimates for pelagic sources of production to Cutthroat Trout in the lakes and reservoirs ranged from 14% at Middle Quinsam Lake to 24% at Snakehead Lake and 8% in Lower Campbell Reservoir to 26% in Upper Campbell Reservoir. These estimates for pelagic contributions to diet were compared to estimates of annual and seasonal water residence time for each lake. An asymptotic relationship was found with reduced but variable pelagic contribution to diet observed when seasonal water residence is less than ~25 days.

Gooseneck and Snakehead lakes receive water from the Quinsam River diversion whereas Middle Quinsam Lake has water diverted upstream of the lake and thus is a donor lake. The original prediction was that water residence time will be shorter in Gooseneck and Snakehead lakes (receiving lakes) than Middle Quinsam Lake (donor lake), which could result in decreased zooplankton production. Seasonal water residence time was indeed found to be lower in both Gooseneck (6.1 days) and Snakehead (4.1 days) lakes compared to Middle Quinsam Lake (26.7 days). However, the annual residence time at Gooseneck Lake (75.6 days) was estimated to be over three times longer than the annual residence time at Middle Quinsam Lake (21.1 days). The estimates of % pelagic production at these lakes do not always follow the predictions of water residence time; a greater pelagic source of production (ultimately from plankton) in both Gooseneck (21%) and Snakehead (24%) lakes was observed in Cutthroat Trout diets compared to Middle Quinsam Lake (14%). Absent from this current analysis, however, is an estimate for the total littoral habitat in each lake relative to total lake area. For example, Middle Quinsam Lake has a very similar surface area to Gooseneck Lake but is much shallower and has a greater percentage of its lake area dominated by shoal habitat.

We observed a strong negative relationship between lake volume and the $\delta^{13}C$ signature of zooplankton. This suggests that carbon from non-phytoplankton sources increasingly contributes to zooplankton production as lake volume declines, which may further explain some of the variability in fish diets across lakes that have short lake water residence time. Therefore, carbon originating from terrestrial sources (e.g., leaf litter) and/or lake macrophytes seems to be relatively more important in smaller lakes. This suggests that declines in pelagic production due to reduction in water residence times may be buffered in small lakes by large contributions of alternative carbon sources to zooplankton production.

In Year 3, a more complete synthesis analysis is planned. A significant goal for Year 3 is to add more lakes into the analysis and to finalize a model between water residence time and pelagic versus littoral contribution to diet. However, % pelagic production will be modeled as a function of water residence time and % littoral habitat in the same model. This will enable predictions of how different water diversion scenarios affect pelagic contributions to fish diets in the diversion lakes while controlling for the amount of littoral habitat available.

Conclusions and recommendations for Year 3 of the program include the following:

1. The following lakes will be sampled in Year 3: Gray Lake, Brewster Lake and Whymper Lake. In addition, we propose to sample John Hart Reservoir, although it is necessary to confirm whether field crews can access the reservoir given the current works that are underway to replace the generating station.
2. Stable isotope analysis of nitrogen and carbon, combined with the use of Bayesian mixing models, was used successfully in Year 2 to understand the diets of species or functional groups in lake food webs, and ultimately to provide estimates of total littoral and pelagic contributions to diets of adult Cutthroat Trout, Rainbow Trout and Dolly Varden. These

methods will be continued in Year 3 to address the management questions posed in the TOR.

3. The amount of littoral habitat in each lake will influence the proportion of fish diets derived from littoral versus pelagic sources. In Year 3, we recommend that the relative area of littoral habitat in each study lake is estimated. This can then be used as an independent variable in models to predict the relative pelagic contributions to fish diets. This desktop exercise will require analysis of bathymetry data that have been collected, or will be collected in Year 3 (Grey and Whymper lakes). We anticipate that this task can be completed within the scope of the outstanding water residency time analysis.

4. The lake levels of the three reservoirs are monitored continuously by BC Hydro. We recommend that metrics relating to the frequency and range of water level fluctuations be identified and compared across the three reservoirs. We propose to integrate this into the scope of the final data analysis tasks.

5. We recommend undertaking invertebrate sampling as planned, which will include three separate trips to each lake in June, July and August. Minnow traps should be deployed during each of these trips with the primary aim of catching Sculpin spp. and reducing effort necessary in the main fish sampling trip in late August or early September. This trip will include gill netting, and we also recommend that trap netting is undertaken with the aim of sampling Threespine Stickleback, Sculpin spp., and juvenile trout. We do not recommend that a separate trip is undertaken in June to sample Threespine Stickleback; this was undertaken in Year 2 and was unsuccessful.

6. There is high overlap in the $\delta^{13}C$ and $\delta^{15}N$ isotope signatures of the three littoral invertebrate prey sources (benthic/littoral, stream and terrestrial invertebrate groups). In Year 3, we recommend that the Bayesian isotope mixing model be simplified to fewer sources by aggregating the three littoral invertebrate prey sources into one prey group.

7. As undertaken in Year 2, we recommend that all zooplankton samples collected in Year 3 are enumerated so an estimate of zooplankton biomass can be made for each lake. This will involve collecting body length measurements for a sub-sample of individuals to estimate mean body mass. This is important because zooplankton biomass provides a direct measure of food availability to fish and we aim to examine relationships between this variable and lake water residence time. We plan to integrate this work into the existing scope of the zooplankton sample analysis. In addition, we recommend that zooplankton sample analysis is undertaken after each sampling trip, rather than at the end of the field season. This will break up the work, which will aid scheduling and allow for preliminary analysis of results before the sampling is completed.

8. We recommend that lake water temperature profiles are collected during each zooplankton sampling trip. This will provide data regarding how the thermocline depth changes throughout the growing season, which will support the water residency time analysis.

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1. INTRODUCTION

1.1. Background to Water Use Planning

Water use planning exemplifies sustainable work in practice at BC Hydro. The goal is to provide a balance between the competing uses of water that include fish and wildlife, recreation and power generation. Water Use Plans (WUPs) were developed for all of BC Hydro's hydroelectric facilities through a consultative process involving local stakeholders, government agencies and First Nations. The framework for water use planning requires that a WUP be reviewed on a periodic basis and there is expected to be monitoring to address outstanding management questions in the years following the implementation of a WUP.

As the Campbell River Water Use Plan (BC Hydro 2012) process reached completion, a number of uncertainties remained with respect to the effects of BC Hydro operations on aquatic resources. A key question throughout the WUP process was "what limits fish abundance?" For example, are fish abundance and biomass in lakes limited by pelagic or littoral sources of production? Answering this question is an important step to better understanding how human activities in a watershed affect fisheries, and to effectively manage water uses to protect and enhance aquatic resources. To address this uncertainty, monitoring programs were designed to assess whether fish benefits are being realized under the WUP operating regime and to evaluate whether limits to fish production could be improved by modifying operations in the future.

In lakes and reservoirs, fish production is assumed to be proportional to overall aquatic productivity, but there is considerable uncertainty over the extent to which fish production is driven by littoral vs. pelagic production and whether this is influenced by operations. BC Hydro affects lake littoral production through drawdowns, and pelagic production through alterations of water residence time (e.g., by manipulation of inflows and outflows). The *Upper Campbell, Lower Campbell, John Hart Reservoirs and Diversion Lakes Littoral versus Pelagic Fish Production Assessment* (JTHMON-5) is part of wider monitoring of the Campbell River WUP. JTHMON-5 is designed to assess the extent to which fish production is driven by littoral vs. pelagic production and how this relates to BC Hydro operations.

1.2. BC Hydro Infrastructure, Operations and the Monitoring Context

The Campbell River WUP project area is complex and includes facilities and operations in the Campbell, Quinsam and Salmon watersheds. In addition to the mainstem rivers, there are three large reservoirs, nine diversion lakes influenced by water diverted from the Quinsam and Salmon rivers, and many tributaries and small lakes that are not directly affected by operations (Map 1). Details of BC Hydro's Campbell River infrastructure and operations are provided in the Campbell River System WUP (BC Hydro 2012).

1.2.1. Reservoirs

Strathcona, Ladore and John Hart dams regulate reservoir water levels for Buttle/Upper Campbell, Lower Campbell, and John Hart reservoirs respectively. Buttle/Upper Campbell Reservoir varies the most in water levels, whereas John Hart Reservoir water levels vary the least. During development of the Campbell River WUP, the Fish Technical Committee (FTC) hypothesized that fish production in Upper and Lower Campbell reservoirs was negatively impacted by large fluctuations in water level through its effect on littoral production. Stable reservoir levels were assumed to have a positive influence on fish production. Evaluation of reservoir operations relied heavily on the Effective Littoral Zone (ELZ) Performance Measure (PM) with the assumption that increasing littoral development would lead to increases in fish productivity. This assumes a strong link between littoral and fish production. JHTMON-4 is designed to investigate the effect of operations on littoral primary production, and JHTMON-5 is designed to test the assumption that improvements in littoral production lead to corresponding increases in fish production. This information will then be used to directly evaluate the impact of the Campbell River WUP on reservoir fish production, help refine reservoir-related PMs and assess their relative importance for future WUP review processes. The understanding gained through the present monitoring program may also help guide the development of alternative management strategies for reservoir operations.

1.2.2. Diversion Lakes

The Quinsam and Salmon diversions divert water through several smaller lakes and into Lower Campbell Reservoir (Map 1). Among the diversion-affected lakes, there are lakes that receive water diverted from adjacent watersheds and thus have lower water residence time (e.g., Gooseneck, Fry and Gray lakes) and lakes that have water diverted away from them and thus have increased water residence time (e.g., Middle Quinsam, Lower Quinsam). During the WUP process, the FTC hypothesized that short water residence time as a result of the BC Hydro diversion operations could negatively impact pelagic productivity. Simple chemostat modelling exercises showed that high inflows flush pelagic organisms from the system. The loss in pelagic productivity from high inflows was thought to have a potential impact on fish production in these lakes. However, the hypothesis could not be tested during the WUP due to time and resource constraints. The FTC therefore assumed for decision-making purposes that there was limited impact, but strongly recommended that the test of this hypothesis be part of a monitoring program. Information collected in JHTMON-5 will be used to evaluate the effect of Campbell River WUP operations on diversion lake productivity, and help refine PMs for future WUP reviews.

1.3. Management Questions and Hypotheses

The JTHMON-5 monitoring program aims to address the following two management questions:

1) To what extent do stabilized reservoir levels, as affected by BC hydro operations, benefit fish populations?

2) What is the relationship between residence time (as affected by diversion rate) and lake productivity?

In addressing the questions, the monitoring program is designed to test the following three null hypotheses:

$H_0$1: The extent of littoral development in lakes, as governed by the magnitude and frequency of water level fluctuations, is not correlated with the ratio of littoral vs. pelagic energy flows to reservoir fish populations.

$H_0$2: The extent of pelagic production in lakes, as governed by the average water residence time, is not correlated with the ratio of littoral vs. pelagic energy flows to diversion lake fish populations.

$H_0$3: Standing crop of pelagic bacteria is not correlated with water residence time.

1.4. Scope of the JHTMON 5 Study

1.4.1. Overview

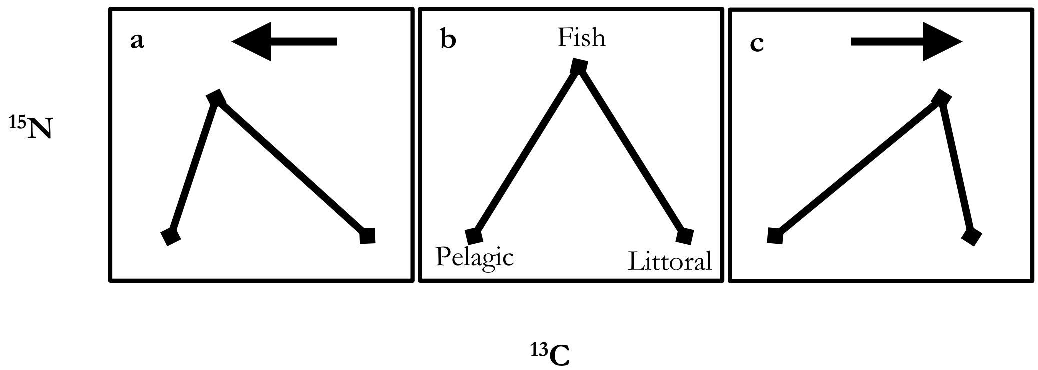
JHTMON-5 is scheduled for 10 years and has two components: stable isotope analysis of food webs in reservoirs and diversion lakes (used to address $H_0$1 and $H_0$2 above), and production estimates of pelagic bacteria in reservoirs and diversion lakes (used to address $H_0$3 above). Data from these two study components will be analyzed separately and together to assess linkages between littoral and pelagic production and the effect of BC Hydro operations on fish production in reservoirs and diversion lakes. This report presents data from Year 2 of the stable isotope analysis of food webs component used to test $H_0$1 and $H_0$2 above. Under the current TOR, sampling using stable isotope methods is scheduled for years 1, 2 and 3 of JHTMON-5, with a potential for a fourth year of sampling to be determined at the end of Year 3 (BC Hydro 2013). Estimates of pelagic bacteria as an indicator of pelagic productivity will be addressed in years 7, 8, and 9 and thus will be discussed in future years of the program.

1.4.2. Summary of the Main Method to Test Management Questions

Substantial information regarding the structure and functioning of lake food webs can be gained by using stable isotopes to reconstruct the diets of lake biota (Vander Zanden and Vadeboncoeur 2002, McIntyre *et al*. 2006). JHTMON-5 uses stable isotope analysis (SIA) of nitrogen and carbon of fish tissues and their potential diet items to assess relative energy flows to fish from littoral vs. pelagic areas. Nitrogen isotope ratios ($\delta^{15}N$) are commonly used to assess the trophic position of species in a food web (DeNiro and Epstein 1981, Peterson and Fry 1987), whereas carbon isotope ratios ($\delta^{13}C$) are commonly used to indicate the sources of primary production (DeNiro and Epstein 1978, Peterson and Fry 1987). The main premise is that the isotopic ratios in the tissues of consumers represent the isotopic ratios of their diet. In other words, you are what you eat. In lakes, fish that are high in the lake food web tend to have the highest $\delta^{15}N$ signatures. Further, carbon isotopes can be used to determine the relative contributions of littoral vs. pelagic sources of production because $\delta^{13}C$ signatures tend to be higher in littoral and benthic areas than pelagic areas.

Figure 1 represents a conceptual framework where energy flow through the aquatic food web (i.e., trophic level) is described by ^{15}N and energy source is described by ^{13}C. Figure 1b represents a natural system where fish receive quantities of energy from benthos and plankton at some natural system-specific ratio. When littoral production is negatively affected (relative to pelagic production), the peak of the triangle is shifted to the left, as fish obtain relatively more energy from plankton than benthos (Figure 1a). When pelagic production decreases (relative to littoral production) the peak is shifted to the right (Figure 1c) as energy production becomes increasingly dominated by benthos. The magnitude of the peak shifts will define the effect of the treatment impact.

Figure 1. Conceptual framework for the interpretation of stable isotope analysis (SIA) data where b) is the pre-treatment state, a) is dominance of a pelagic-derived energy in fish diet, and c) is dominance of littoral-derived energy in fish diet.



Using both $\delta^{15}N$ and $\delta^{13}C$ together allows for the development of stable isotope mixing models which can estimate the contributions of different prey sources to a consumers diet (Semmens *et al.* 2009, Parnell *et al.* 2010). The primary species of interest in JHTMON-5 are Cutthroat Trout (*Oncorhynchus clarkii*) and Rainbow Trout (*O. mykiss*). Sampling is geared toward understanding the diets and energy sources of these two fish species, which are the resident fish species of primary management concern in reservoirs and lakes of the Campbell River system. Resident Dolly Varden (*Salvelinus malma*) is also present in some lakes of the Campbell River system and is a secondary management interest. Primary diet items for Cutthroat Trout, Rainbow Trout and Dolly Varden include zooplankton (pelagic source), benthic/littoral invertebrates (littoral source), stream invertebrates that wash into littoral areas (allochthonous littoral source), terrestrial invertebrates that fall into littoral areas (allochthonous littoral source), and other fish including Threespine Stickleback (*Gasterosteus aculeatus*), Sculpin spp. (*Cottus* spp.), and juvenile trout (*Oncorhynchus* spp.). Thus the JHTMON-5 study was geared towards obtaining representative samples of Cutthroat Trout, Rainbow Trout and Dolly Varden and potential diet items from each reservoir and lake sampled. Stable isotope data can be obtained from tissue samples of individuals (e.g., fin clips, muscle samples), from whole organisms (e.g., whole insects), or from composite samples from multiple individuals (e.g., zooplankton samples).

1.4.3. Sampling in Year 2

Year 2 of JHTMON-5 was planned and implemented as a full sampling year based on the results and recommendations of the pilot Year 1. Sampling was completed for Lower Campbell Reservoir, Beavertail Lake, Snakehead Lake and Upper Quinsam Lake (Map 2, Map 3, Map 4, Map 5).

To what extent do stabilized reservoir water levels, as affected by BC Hydro operations, benefit fish populations? It is hypothesized that less variation in reservoir water levels benefits littoral production and increases reservoir fish production. Upper Campbell Reservoir, which experiences the greatest fluctuations in water levels, was sampled in Year 1. Lower Campbell Reservoir was chosen for sampling in Year 2 because it experiences intermediate fluctuations in water levels compared to Upper Campbell Reservoir and John Hart Reservoir. Sampling in John Hart Reservoir, which experiences the lowest fluctuations in water levels, is planned for Year 3. After Year 3, contrasts of BC Hydro operations across reservoirs will be possible using this design.

What is the relationship between residence time (as affected by diversion rate) and lake productivity? It is hypothesized that shorter water residence times decreases zooplankton production and thus the pelagic source to fish production. In Year 1, Gooseneck Lake and Middle Quinsam Lake were chosen because they are part of the same diversion system (Quinsam River); Middle Quinsam Lake experiences greater residence time (donor lake) and Gooseneck Lake experiences reduced residence time (recipient lake). Based on this contrast in BC Hydro operations it is predicted that Gooseneck Lake will have a lower pelagic contribution to fish production and a greater reliance on littoral sources of production than Middle Quinsam Lake. However, an important recommendation from the Year 1 report was that an estimate of water residence time be developed for each lake based on lake volume and hydrology. For example, it is possible that Middle Quinsam Lake has a lower water residence time than Gooseneck Lake based on its unique morphology and hydrology. It was also recommended in the Year 1 report that 10 or more lakes be sampled across the program years that vary in lake water residence time. This would enable simple regression models relating water residence time in each lake to % littoral or % pelagic contributions to fish diets (Figure 2). Operations from water diversions could then be integrated into predictive models of how water inputs or extractions can change lake water residence time, lake productivity and food webs.

In Year 2, Beavertail Lake, Snakehead Lake and Upper Quinsam Lake were chosen because they are part of the same diversion system (Quinsam River) or, in the case of Beavertail Lake, are a nearby control lake. Snakehead Lake experiences reduced water residence time (recipient lake), while Upper Quinsam Lake is a control lake above the Quinsam River water diversion. Taken together, the five lakes sampled across Year 1 and Year 2 include two recipient lakes (Gooseneck and Snakehead lakes), one donor lake (Middle Quinsam Lake) and two control lakes (Beavertail and Upper Quinsam lakes).

The selection of Lower Campbell Reservoir, Beavertail Lake, Snakehead Lake and Upper Quinsam Lake support an examination of $H_0 1$ and $H_0 2$, particularly when the data from Year 1 and Year 2 are combined. At each lake, representative pelagic and littoral sampling sites were chosen to collect invertebrate prey sources (zooplankton, littoral invertebrates, stream invertebrates, and terrestrial invertebrates) and fish. The representative littoral sites were located near stream inflows at each lake.

In Year 3, sampling is planned to occur in John Hart Reservoir, and in several diversion and control lakes with varying water residence times that are a part of the Salmon River diversion. The following lakes and reservoirs have been highlighted for potential study within the JHTMON-5 program:

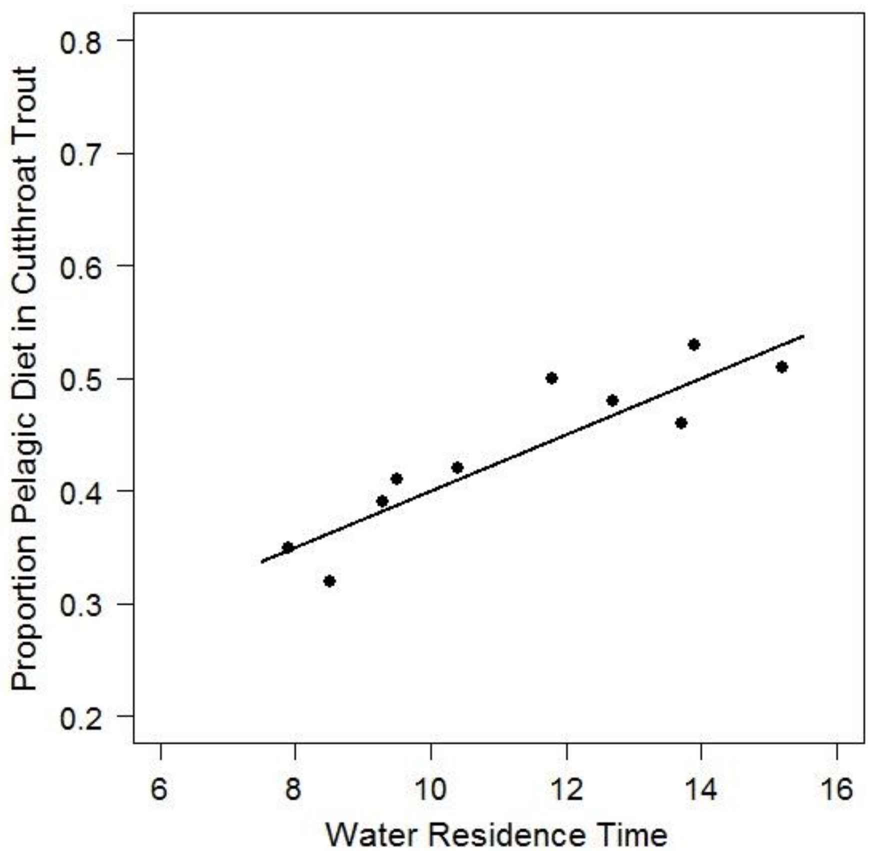
Reservoirs: Buttle/Upper Campbell, Lower Campbell, John Hart

Diversion Lakes: Brewster, Fry, Gooseneck, Gray, Lower Quinsam, McIvor, Middle Quinsam, Snakehead, Whymper

Crest, Upper Drum, and Lower Drum are decommissioned diversion lakes.

"Control" Lakes: Amor, Beavertail, Boot, Gentian, Gosling, Long, Merrill, Mohun, Paterson, Roberts, Upper Quinsam, Wokas

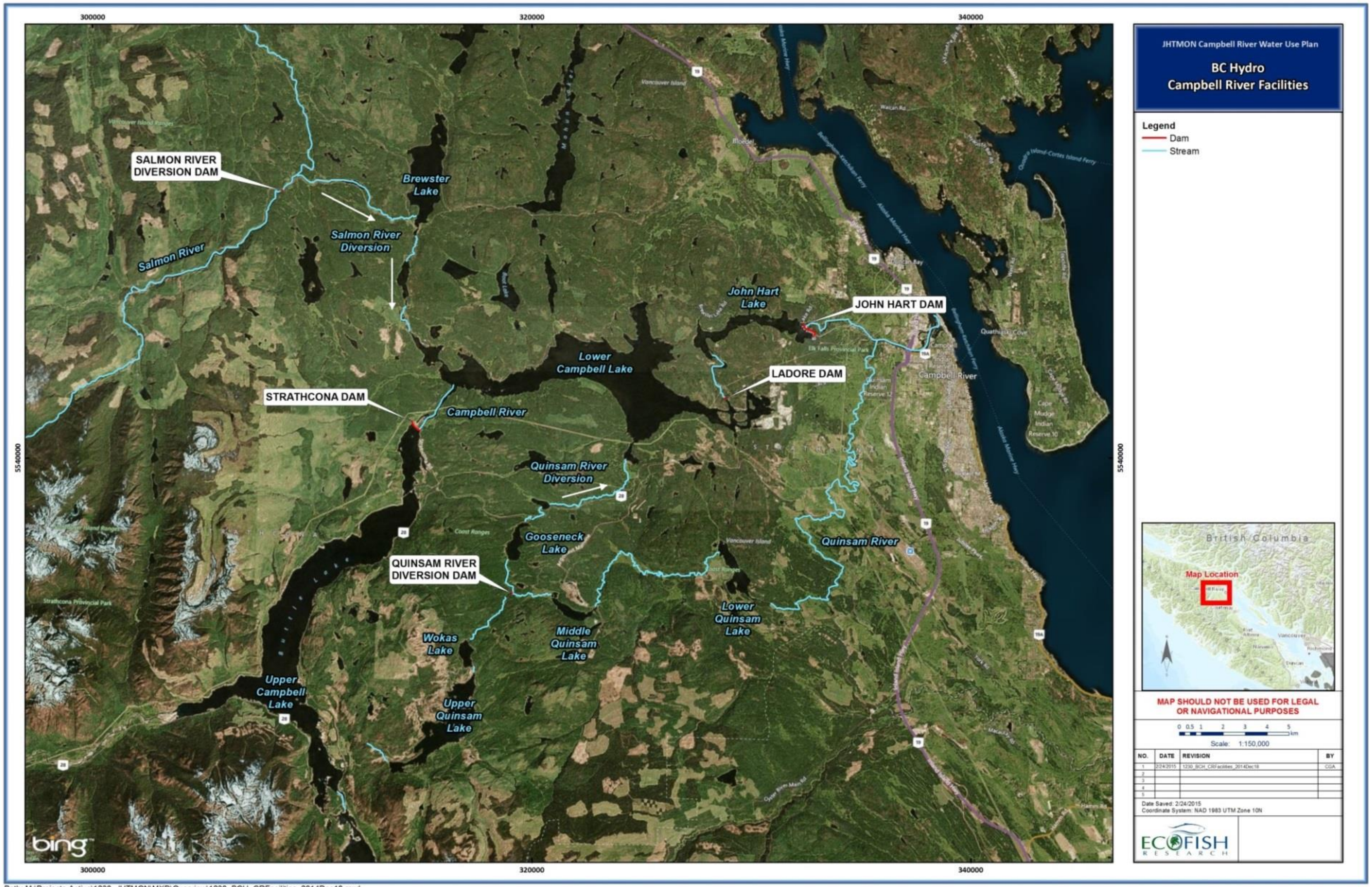
Figure 2. Hypothetical relationship between lake water residence time and pelagic productivity as indicated by the proportion of pelagic diet in Cutthroat Trout. Each point represents a different diversion or control lake with data accumulated over several years of the JHTMON-5 program.



1.4.4. Water Residence Time

To address management question 2, estimates of water residence time were determined for all Year 1 and Year 2 study lakes and reservoirs including Upper Campbell and Lower Campbell reservoirs, and Beavertail, Gooseneck, Middle Quinsam, Snakehead, and Upper Quinsam lakes. This same approach is planned for the Year 3 study lakes. The theoretical residence time of a lake is calculated by relating the annual amount of water passing through the lake to the volume of the whole basin. A lake's residence time is often calculated on an annual basis so that seasonal variation does not unduly influence the results. The effective residence time may, however, be influenced by a number of factors such as the timing of stratification, the depth of the thermocline, and the variability of inflows (George and Hurley 2002, Rueda *et al.* 2006, Vidal *et al*. 2012). These factors may be particularly important during the summer months, when fish production is greatest. Since this monitor is focused on assessing the effects of residence time on pelagic production, a seasonal residence time was computed in addition to the annual residence time. To understand inter-annual variability, residence times were computed for each lake for the years 2012 to 2015.

Map 1. Overview of BC Hydro Campbell River facilities.



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2. METHODS

2.1. Invertebrate sampling

Primary invertebrate diet items for Cutthroat Trout, Rainbow Trout and Dolly Varden include zooplankton (pelagic source), benthic/littoral invertebrates (littoral source), stream invertebrates that wash into littoral areas (allochthonous littoral source), and terrestrial invertebrates that fall into littoral areas (allochthonous littoral source). Zooplankton are the primary pelagic source of production to fish in lakes. The benthic/littoral invertebrates, stream invertebrates and terrestrial invertebrates are three different littoral sources of production for fish.

2.1.1. Zooplankton

2.1.1.1. Field

Zooplankton are typically the main source of secondary pelagic production in lakes for upper trophic level consumers such as trout. In stable isotope studies of lakes, it is important to use an appropriate isotopic baseline for zooplankton that is representative of the isotopic signatures experienced by the consumers. Based on studies from other lakes and reservoirs on Vancouver Island, the nitrogen and carbon stable isotope signatures in zooplankton are known to vary seasonally (Matthews and Mazumder 2003, 2005). Therefore, zooplankton was sampled at three time periods (late June, late July and early September) to obtain a representative sample of zooplankton in each lake. Zooplankton was sampled at three sites on each lake, thus extending the spatial coverage from Year 1 when only one site was sampled in the diversion lakes.

Zooplankton was sampled at three central sites located in the deepest areas of each lake (Table 1). These sites were georeferenced with a GPS and revisited on each of the three sampling dates. Zooplankton sampling sites were typically close to the gill net fish sampling sites in the pelagic area of each lake (see Map 2, Map 3, Map 4, Map 5).

Zooplankton was sampled using a tow net with a 30 cm diameter aperture and a mesh size of 80 μm (Figure 3). Sampling involved one or two upwards vertical tows at a rate of approximately 0.5 m/s, from a depth of ≤ 20 m to the surface (Table 1). The net was rinsed with deionized water prior to each tow and care was taken to ensure the net did not touch the bed. Triplicate samples were collected at each site on each sampling date. All samples were preserved in 95% ethanol (Figure 4).

Figure 3. Zooplankton net.

Figure 4. Zooplankton samples prior to adding ethanol. Note high density of captured specimens.

2.1.1.2. Laboratory: Taxonomic Identification

The first of the three zooplankton samples was used for identification, a component of the second was transferred to a smaller vial and sent for stable isotope analysis, and the third was kept as a backup.

Zooplankton was primarily identified to family by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society). Zooplankton was primarily classified into the following taxa, based on Witty (2004):

- Order Cladocera: Families – Daphniidae, Bosminidae, Sididae, Leptodoridae, Polyphemidae
- Order Calanoida and Cyclopoida
- Nauplii (unidentified)

Counts of less common taxa were also recorded.

Each zooplankton sample for taxonomic analysis was split into subsamples using a Folsom Plankton Splitter (Aquatic Research Instruments, Idaho). Each sample was split between zero and five times until subsamples comprised ~100 to 400 individuals of the most dominant taxon. The final subsample was concentrated into a zooplankton counting chamber for counting and identification with a binocular microscope. Counts were expressed as individuals per sample by multiplying counts by the appropriate dilution factor. These counts were then expressed volumetrically (#/L) by dividing the counts by the volume of water (L) sampled in the field. The sampling volume was calculated as the product of the sampling depth (m), the number of vertical tows per sample, and the area of the net aperture (circular aperture with diameter = 0.3 m).

2.1.1.3. Laboratory: Biomass Determination

The trophic position occupied by zooplankton between phytoplankton and fish in the pelagic zone means that zooplankton biomass represents an important 'food web channel' through which carbon fixed by plants within the lake (i.e., autochthonous carbon) is transferred to fish populations that occupy higher trophic levels (Sterner 2009). Zooplankton biomass was therefore estimated for each sample enumerated in Year 2 to provide a metric of the pelagic productivity in each lake that is available to fish. Taxon-specific length measurements made in Year 2 were also used to estimate biomass in samples collected in Year 1. These data will be used to address Management Question 2, which relates to the relationship between residence time and lake productivity.

Biomass (dry weight) of crustacean zooplankton (comprising all taxa sampled) was determined using published relationships between body length and body mass for individual taxa, using methods based on the US EPA (2003) protocol. This approach was chosen instead of methods that involve weighing or measuring displacement of bulk samples, as these techniques are susceptible to error due to other material (e.g., seston) present in samples. Briefly, the approach involved calculating a mean length that was representative of each dominant taxon. These lengths were then used to estimate the biomass of zooplankton in each sample using established biomass–length relationships. Finally, biomass was calculated on a volumetric basis (µg/L) for each lake by dividing the biomass in each sample by the total volume of water that was sampled.

Sub-samples of dominant taxa were measured using an ocular ruler (100× magnification) with a binocular microscope. Preliminary analysis was undertaken to inform the approach of sub-sampling taxa for length measurements, with the aim of ensuring that sufficient length measurements were made to adequately reflect sources of variability between samples, while minimizing the number of samples that needed to be processed. Specifically, variance in animal body length between lakes and sampling months was initially examined for subsamples of Daphniidae individuals; these were typically the dominant taxon in each sample that made the greatest estimated contribution to biomass. These results were used to decide whether it was necessary to measure body length of each taxon separately for each lake and sampling month, or whether estimates for an individual lake/month could be applied across all lakes/months. Results of this preliminary analysis showed that length measurements were relatively consistent between lakes and sampling months (Figure 5). Two-way ANOVA showed that there was no statistically significant difference (significance level = 0.05) in mean lengths between lakes ($p = 0.48$, $F = 0.84$, $df = 3$), although there was a small but statistically significantly difference between sampling months ($p = 0.002$, $F = 6.42$, $df = 2$). This reflected a statistically significantly higher mean length between the June and September samples (Tukey's HSD test, adj. $p = 0.004$, Δ 298 μm), with visual inspection of the data indicating a slight increase in lengths as the growing season progressed (compare June, July and September data for Upper Quinsam Lake in Figure 5).

Based on the consistency in length measurements between lakes, we consequently chose to measure mean body lengths of remaining dominant taxa using samples collected from a single lake, and apply these to respective taxa in all lakes. In addition, we chose to use only samples collected in July to measure mean body lengths of remaining taxa, as measurements for this month were deemed to be most representative of the growing season in general, based on the indication that there was a slight increase in zooplankton size as the season progressed (i.e., July was approximately in the middle of the sampling program).

Taxon-specific mean body length (L) measurements were converted to dry biomass (W; μg) using relationships listed in US EPA (2003) and Watkins *et al.* (2011). An exception was naupilii, for which a constant biomass of 40 μg was assigned, independent of length (Hawkins and Evans 1979 cited in US EPA 2003).

W–L relationships followed the general power equation:

$$W = \alpha L^{\beta}$$

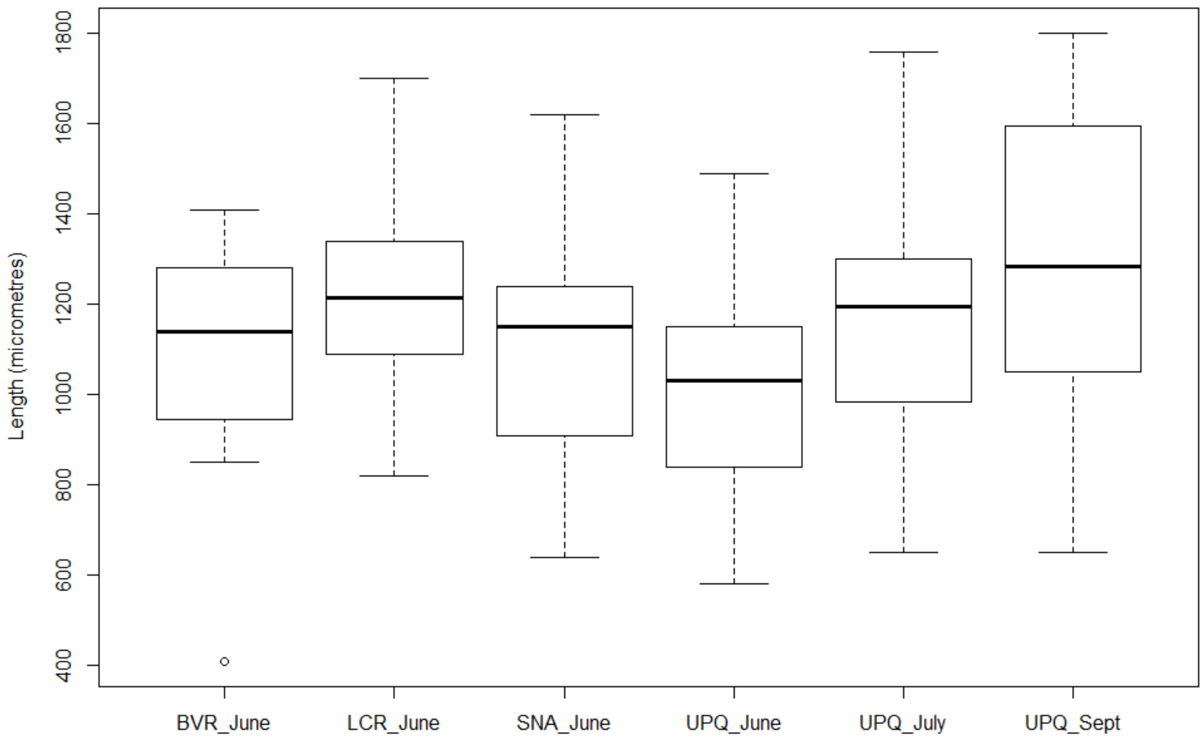
where W is biomass (μg), L is mean body length (mm) and α and β are constants specific to each taxon. Relationships were converted to linear form by logarithmic transformation:

$$\ln W = \ln \alpha + \beta \cdot \overline{\ln L}$$

where $\overline{\ln L}$ is calculated as the mean of the transformed length measurements in mm. Mean individual biomass for each taxon was then calculated following back transformation. Corrections were not made to reflect logarithmic transformation bias. This potential source of error is not considered in either US EPA (2003) or Watkins *et al.* (2011), and the information necessary to estimate this (e.g., the residual mean square of the original regression) is not typically reported with published biomass–length relationships. McCauley (1984) estimates that failure to consider this source of bias may result in error of 2–11%, which was considered tolerable given that the objective was to primarily compare biomass estimates between study systems, rather than with systems elsewhere.

Estimates of the biomass of dominant taxa in each sample were calculated as the product of total sample count data and taxon–specific mean biomass values ($\bar{W}$). These estimates were then standardized on a volumetric basis (µg/L) by dividing the total biomass (µg) in each sample by the volume (L) of water that was sampled.

Figure 5. Body lengths of Daphniidae (*n*=15–20) measured in samples collected from Beavertail Lake (BVR), Lower Campbell Lake reservoir (LCR), Snakehead Lake (SNA) and Upper Quinsam Lake (UPQ). Data are presented for samples collected from each lake in June, plus July and September samples for UPQ. Bold horizontal lines denote medians, boxes denote interquartile ranges and whiskers denote ranges.



2.1.2. Littoral[1] Invertebrates

2.1.2.1. Field

Samples of littoral invertebrates were collected from the littoral zone of each waterbody once during the July sampling trip. Littoral invertebrate sampling sites are shown on Map 2, 3, 4 and 5 (see 'BIV' sites). Different sampling methods were used (Table 1) depending on the specific habitat characteristics at each lake, e.g., substrate type, productivity, macrophyte coverage. Sampling was conducted with the objectives of maximizing both the numbers of individuals, and the range of taxa collected within the time available. Sampling was non–quantitative and all samples were preserved in the field in 95% ethanol.

[1] This sampling component corresponds to 'benthic invertebrate' sampling in Year 1, although we use the term 'littoral invertebrates' in this report to reflect that several sampling methods were used to collect invertebrates from different lentic habitats (including the water column in the littoral zone), as opposed to the sole use of a Ponar grab in Year 1.

Samples were collected from Lower Campbell Reservoir using a Ponar grab ('Petite' model) with an aperture of 152 mm × 152 mm. The Ponar grab was deployed six times in the littoral zone by wading to a depth of 0.5 m, approximately 5 m from shore (Figure 6). All visible invertebrates were removed with forceps and placed in a clean sample jar for preservation. Two samples were collected.

Samples were collected from Beavertail and Upper Quinsam lakes using 'travelling kick and sweep' sampling (CCME 2011). This is a standard sampling method suitable for sampling moderately productive lakes with littoral zones that contain abundant emergent and submergent macrophytes (Figure 7, Figure 8). Sampling involved wading along transects in the littoral zone (area sampled ~ 20 m^2) and gently kicking the substrate to suspend invertebrates in benthic sediments or attached to macrophytes. A hand–held aquarium net (mesh size < 500 µm) was then repeatedly drawn through the water column in a sweeping motion to collect a sample of suspended material. This was then sorted on a tray and all visible invertebrates were removed with forceps. Three samples were collected from each lake.

Samples were collected from Snakehead Lake using hand searches; trial sampling using a Ponar grab indicated that invertebrate abundance was very low in benthic sediments, while numerous cobbles and dense stands of emergent vegetation (*Juncus* spp.) inhibited efficient travelling kick and sweep sampling. Invertebrates were predominantly picked from submergent and emergent vegetation, cobbles and large woody debris. Additional macroinvertebrates (notably freshwater mussels; Figure 9) were picked from the lake bed. Three samples were collected.

Figure 6. Ponar grab sampling at Lower Campbell Reservoir (site LCR–BIV01)

Figure 7. Littoral habitat sampled using travelling kick and sweep sampling at Beavertail Lake (site BVR–BIV01; left) and example sample (right).

Figure 8. Littoral habitat sampled using travelling kick and sweep sampling at Upper Quinsam Lake (site UPQ–BIV01).

Figure 9. Littoral habitat sampled on Snakehead Lake (left), with freshwater mussels (family: Unionidae) collected from lake bed (right).

2.1.2.2. Laboratory

Littoral invertebrates were sorted and counted in the laboratory to order, and, where possible, family by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society). Identification was made with reference to Iowa State University (2015).

2.1.3. Stream Invertebrates

2.1.3.1. Field

Stream invertebrates were sampled once in one stream inflow or outflow to each lake during the July sampling trip. Stream invertebrate sampling sites are shown on Map 2, 3, 4 and 5 (see 'SIV' sites). Preference was given to selecting stream sites in inflowing streams, although sites were located on the lake outflow at Snakehead and Beavertail lakes. This was primarily due to drought conditions, which meant that surface flows in stream inflows were negligible. In addition, the main inflows to these two lakes enter the lake via wetland habitats, whereas the stream outflows had defined channels and were better representative of lotic habitats that were the target of the stream sampling.

As with lake invertebrate sampling (Section 2.1.2.1), stream sampling was non–quantitative and was undertake with the objectives of maximising both the numbers of individuals, and the range of taxa collected within the time available. Triplicate samples were collected for each water body and all samples were preserved in the field in 95% ethanol.

Sampling methods varied depending on the substrate type and flow conditions (Table 1). Kick sampling was used to collect stream invertebrates at the stream site at Lower Campbell Lake reservoir. This site was on Beavertail Creek, with samples collected from riffle sections with predominantly gravel substrate. A single drift net (mesh size = 250 µm) was secured to the stream bed using rebar and the upstream substrate was agitated for three minutes using a wading boot. This was undertaken at a total of three sub–sites (< 5 m apart) to collect a single composite sample of material that was thoroughly inspected, with all individuals picked. This was repeated two further times to collect triplicate samples.

The kick sampling method was also used to sample stream invertebrates at Beavertail Lake. Low flow conditions (Figure 10), however, limited the effectiveness of this sampling technique at this site, and samples were supplemented by employing hand searches to pick individuals from the underside of large gravel and cobbles.

Very low current velocity and shallow depth prohibited kick sampling to sample stream invertebrates at Snakehead Lake (Figure 11). Instead, cobbles were overturned in a small riffle section with very low water depth (< 0.03 m), and individuals were picked using tweezers. Five medium (~ 0.15 m diameter) cobbles were picked per sample.

Very low current velocity prohibited standard stream kick sampling to sample stream invertebrates at Upper Quinsam Lake (Figure 13). Instead, travelling kick and sweep sampling (see Section 2.1.2.1) was used to sample pools (~0.2 m to 1.0 m deep) in a stream inflow, with one pool (~4 m^2) sampled per invertebrate sample.

2.1.3.2. Laboratory

Stream invertebrates were sorted and counted in the laboratory to order, and, where possible, family by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society). Identification was made with reference to Iowa State University (2015).

Figure 10. Stream invertebrate sampling site at an inflow to Lower Campbell Reservoir (site BVR–SIV01; left) and example sample (right).

Figure 11. Stream invertebrate sampling site at the outflow of Beavertail Lake (site BVR–SIV01).

Figure 12. Stream invertebrate sampling site at the outflow of Snakehead Lake (site BVR–SIV01).

Figure 13. Stream invertebrate sampling site at an inflow to Upper Quinsam Lake (site UPQ–SIV01).

2.1.4. Terrestrial Invertebrates

2.1.4.1. Field

A sample of terrestrial invertebrates was collected at each lake during each of the three sampling trips using a malaise trap placed on the lake shorelines (see 'TIV' sites on Map 2, 3, 4 and 5). Three terrestrial invertebrate samples were therefore collected at each lake; this represents an increase in sampling scope relative to Year 1 when only a single terrestrial invertebrate sample was collected at each lake. The malaise trap consisted of a square–shaped tent (1.2 m long × 1.2 m wide × 2.1 m high) with openings at the side (Figure 8). Insects fly into the tent and climb upwards into a collecting jar. The trap was deployed for 2.0 to 5.5 hours at a single site on the shoreline of each lake. No chemical attractants or killing agents were used and samples were preserved using 95% ethanol.

Figure 14. Malaise net deployed at Upper Quinsam Lake in September 2015.

2.1.4.2. Laboratory

Terrestrial invertebrates were sorted and counted in the laboratory to order, and, where possible, family by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society). Identification was made with reference to Iowa State University (2015).

Table 1. Summary of Year 2 invertebrate sampling sites.

Sampling type	Waterbody	Site	Method	Sampling dates	UTM (NAD 83)			Site depth (m)	Depth sampled (m)
					Zone	E (m)	N (m)		
Zooplankton	Lower Campbell Reservoir	LCR-LKZP01	Vertical plankton tow	Jun-24, Jul-20	10U	326112	5542580	50.0	20.0
		LCR-LKZP02			10U	324730	5543888	30.0	20.0
		LCR-LKZP03			10U	322197	5544324	44.0	20.0
	Beavertail Lake	BVR-LKZP01	Vertical plankton tow	Jun-23, Jul-23	10U	319990	5539765	18.0	14.0–16.0
		BVR-LKZP02			10U	320271	5539872	17.0	15.0–16.0
		BVR-LKZP03			10U	320595	5539736	16.0	15.0
	Snakehead Lake	SNA-LKZP01	Vertical plankton tow	Jun-25, Jul-21	10U	320191	5538070	6.0	4.0–4.5
		SNA-LKZP02			10U	320311	5538001	6.5	4.5–5.5
		SNA-LKZP03			10U	320503	5537936	9.5	7.0–8.0
	Upper Quinsam Lake	UPQ-LKZP01	Vertical plankton tow	Jun-22, Jul-22	10U	317002	5529342		12.0–20.0
		UPQ-LKZP02			10U	316822	5528930		12.0–20.0
		UPQ-LKZP03			10U	316584	5528193		14.0–20.0
Littoral invertebrates	Lower Campbell Reservoir	LCR-BIV01	Ponar grab	Jul-20	10U	324299	5541198	0.5	0.5
	Beavertail Lake	BVR-BIV01	Travelling kick and sweep	Jul-23	10U	320992	5539774	0.1–1.0	0.1–1.0
	Snakehead Lake	SNA-BIV01	Hand searches	Jul-21	10U	320823	5537879	0.1–1.0	0.1–1.0
	Upper Quinsam Lake	UPQ-BIV01	Travelling kick and sweep	Jul-22	10U	313472	5526110	0.1–1.0	0.1–1.0
Stream invertebrates	Lower Campbell Reservoir	LCR-SIV01	Kick sampling	Jul-20	10U	324238	5541125	0.1–0.2	0.1–0.2
	Beavertail Lake	BVR-SIV01	Kick sampling, hand search	Jul-23	10U	320833	5540045	0.1–0.2	0.1–0.2
	Snakehead Lake	SNA-SIV01	Hand search	Jul-21	10U	320850	5537901	0.1–0.2	0.1–0.2
	Upper Quinsam Lake	UPQ-SIV01	Travelling kick and sweep	Jul-22	10U	313404	5526155	0.1–0.5	0.1–0.5
Terrestrial invertebrates	Lower Campbell Reservoir	LCR-TIV01	Malaise trap	Jun-24	10U	324620	5540694	-	-
		LCR-TIV02	Malaise trap	Jul-20	10U	324271	5541202	-	-
	Beavertail Lake	BVR-TIV01	Malaise trap	Jun-23, Jul-23	10U	320375	5539398	-	-
	Snakehead Lake	SNA-TIV01	Malaise trap	Jun-25, Jul-21	10U	320824	5537868	-	-
	Upper Quinsam Lake	UPQ-TIV01	Malaise trap	Jun-22, Jul-22	10U	320375	5539398	-	-

2.2. Fish Sampling

Fish sampling was undertaken to obtain representative stable isotope samples of the target fish species of Cutthroat Trout, Rainbow Trout and Dolly Varden and potential fish prey items including Threespine Stickleback, Sculpin spp., and juvenile trout. Several fishing methods were used in order to maximize catch of these food web components including gill netting, minnow trapping and trap netting. In addition to obtaining tissue and stomach samples for diet analyses, the fish sampling methods enabled analyses of fish catch per-unit-effort (CPUE), and fish size and age distributions by species.

2.2.1. Gill Netting

Gill netting was undertaken from August 17 to October 4, 2015 in Lower Campbell Reservoir, Beavertail Lake, Snakehead Lake and Upper Quinsam Lake (Figure 15). Gill netting was primarily used to sample Cutthroat Trout and Rainbow Trout. Two littoral and two pelagic sites were sampled from Beavertail Lake, three littoral and three pelagic sites were sampled from Lower Campbell Reservoir, and one littoral and one pelagic site was sampled from each of Snakehead and Upper Quinsam lakes (Map 2, Map 3, Map 4, Map 5, Table 2).

Sinking gill nets were used to target different depths within the water column. At the littoral sites, nets were set on the bed, perpendicular to shore. At pelagic sites, nets were set perpendicular to depth contours, with sinking nets suspended in the water column at a depth of 5 m (Snakehead Lake only) or 10 m below the surface, close to the assumed thermocline depth. Unlike Year 1, nets were not set on the bed at pelagic sites as Year 1 results indicated that catch success was higher when nets were suspended. RISC standard gill nets were used; the nets consist of six panels, each 15.2 m long and of different mesh sizes, strung together in a "gang" to form a net 91.2 m long and 2.4 m deep. The mesh sizes were as follows: 25 mm, 76 mm, 51 mm, 89 mm, 38 mm, and 64 mm. This sequence of mesh sizes captures a range of size classes of fish.

When setting a net, the boat operator ensured the proper location and depth of the site using a GPS and depth sounder and positioned the net according to depth contours and wind conditions. The net was held in place with a net anchor at each end of the net. Nets were set overnight with soak times of 19–26 hours. Floating lights were attached to each net to mark their location overnight for boater safety. Individual fish processing is described in Section 2.2.4.

Fish CPUE from gill netting was computed for Cutthroat Trout, Rainbow Trout and Dolly Varden and compared across all lakes in both years of sampling including Upper Campbell and Lower Campbell reservoirs and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes.

Table 2. Gill netting sampling site summary for Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

Waterbody	Site	Sampling Date	UTM			Location	Clarity	Water Temp. (°C)
			Zone	Easting	Northing			
Lower Campbell Reservoir	LCR-LKGN01	23-Aug-15	10U	322373	5545290	Littoral	Clear	20.2
	LCR-LKGN02	23-Aug-15	10U	326112	5542580	Pelagic	Clear	20.6
	LCR-LKGN03	04-Oct-15	10U	324420	5541275	Littoral	Clear	15.5
	LCR-LKGN04	04-Oct-15	10U	326112	5542580	Pelagic	Clear	15.8
	LCR-LKGN05	04-Oct-15	10U	324798	5544003	Pelagic	Clear	15.8
	LCR-LKGN06	04-Oct-15	10U	322364	5545336	Littoral	Clear	15.6
Beavertail Lake	BVR-LKGN01	17-Aug-15	10U	320988	5539764	Littoral	Clear	21.0
	BVR-LKGN02	17-Aug-15	10U	320375	5539398	Littoral	Clear	21.0
	BVR-LKGN03	17-Aug-15	10U	319990	5539765	Pelagic	Clear	21.0
	BVR-LKGN04	17-Aug-15	10U	320595	5539736	Pelagic	Clear	22.0
Snakehead Lake	SNA-LKGN01	21-Aug-15	10U	320729	5537745	Littoral	Clear	21.0
	SNA-LKGN02	21-Aug-15	10U	320503	5537936	Pelagic	Clear	21.5
Upper Quinsam Lake	UPQ-LKGN01	19-Aug-15	10U	317098	5528861	Littoral	Clear	21.3
	UPQ-LKGN02	20-Aug-15	10U	316585	5528193	Pelagic	Clear	21.7

Figure 15. Buoys marking location of suspended gill net at UPQ-LKGN02.

2.2.2. Trap Netting

Trap netting was undertaken from August 31 to October 7, 2015 in Lower Campbell and Upper Campbell reservoirs, and Beavertail, Gooseneck, Snakehead, Middle Quinsam and Upper Quinsam lakes (Table 3, Figure 16). Trap netting was primarily used to sample Threespine Stickleback as the target number of this species was not sampled from each lake during Year 1, or during minnow trapping undertaken at Year 2 study lakes in June 2015. The exception was Upper Campbell Reservoir, where trap nets were also used for fish population abundance sampling as part of JHTMON-3. Two sites were sampled on Lower Campbell Reservoir, six sites were sampled on Upper Campbell Reservoir and a single site was sampled at the remaining lakes (Table 3, Map 2, Map 3, Map 4, Map 5). Traps were set overnight in littoral areas with a target soak time of 24 hours. Sites were selected for suitability for trap netting based on site depths and absence of underwater hazards.

When setting a net, the boat operator ensured the proper location and depth of the site using a GPS and depth sounder and positioned the net according to depth contours and wind conditions. The net was held in place with a net anchor. Nets were set overnight with soak times of 19–29 hours. Individual fish processing is described in Section 2.2.4.

Fish CPUE from trap netting was computed for Threespine Stickleback, Cutthroat Trout and Sculpin spp. and compared across all lakes including Upper Campbell and Lower Campbell reservoirs and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes.

Table 3. Trap netting sampling site summary for Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

Waterbody	Site	Sampling Date	UTM			Water Temp.
			Zone	Easting	Northing	
Lower Campbell Reservoir	LCR-LKTN01	04-Oct-15	10U	324300	5541180	15.5
	LCR-LKTN02	04-Oct-15	10U	322197	5545373	16.5
Beavertail Lake	BVR-LKTN01	07-Oct-15	10U	320413	5539412	15.0
Snakehead Lake	SNA-LKTN01	07-Oct-15	10U	320230	5538106	15.0
Upper Quinsam Lake	UPQ-LKTN01	03-Oct-15	10U	317109	5529006	14.8
Upper Campbell Reservoir	UCR-LKTN01	31-Aug-15	10U	305365	5528924	UNK
	UCR-LKTN02	01-Sep-15	10U	309922	5527439	UNK
	UCR-LKTN03	02-Sep-15	10U	314793	5539470	UNK
	UCR-LKTN04	02-Sep-15	10U	312231	5536469	UNK
	UCR-LKTN05	03-Sep-15	10U	310532	5535870	UNK
	UCR-LKTN06	03-Sep-15	10U	310046	5525736	UNK
Gooseneck Lake	GOO-LKTN01	06-Oct-15	10U	318850	5535730	15.0
Middle Quinsam Lake	QUN-LKTN01	06-Oct-15	10U	321328	5533391	15.0

Figure 16. Trap net set at UPQ-LKTN01.

2.2.3. Minnow Trapping

Minnow trapping was undertaken during June 22 to 25 2015 to specifically target Threespine Stickleback in Year 2 study lakes, and Gooseneck Lake and Middle Quinsam Lake, which are both Year 1 study lakes where this species was not caught last year. Sampling was undertaken in June to target individuals present in near-shore areas where lacustrine populations construct nests and spawn during spring and early summer (McPhail 2007). Minnow trapping was also undertaken at the four Year 2 study lakes around the period of the other fish sampling, during August 17 to 24 2015. Target species were Sculpin spp., juvenile trout and Threespine Stickleback (Table 4).

Multiple sites were established on each lake, with 4–10 Gee type minnow traps deployed at each site (Table 4). Traps were either deployed on the bed and secured to the shoreline or suspended at a range of depths (0.5–10 m beneath a buoy).

Each trap was baited with a small amount fish roe placed in a film container perforated with holes, which allowed the scent to escape but prevented the attractant from being consumed. Traps were marked with a float, and UTM co–ordinates, depth, time, and mesh size of trap were recorded. Traps were fished overnight, with soak times ranging from 20-26 hours. Captured fish were separated by site and trap number and then brought back to shore for processing. Individual fish processing is described in Section 2.2.4.

Fish CPUE from minnow trapping was computed for Sculpin spp., juvenile trout and Threespine Stickleback and compared across all lakes in both years of sampling including Upper Campbell and Lower Campbell reservoirs and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes.

Table 4. Minnow trapping sampling site summary for Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

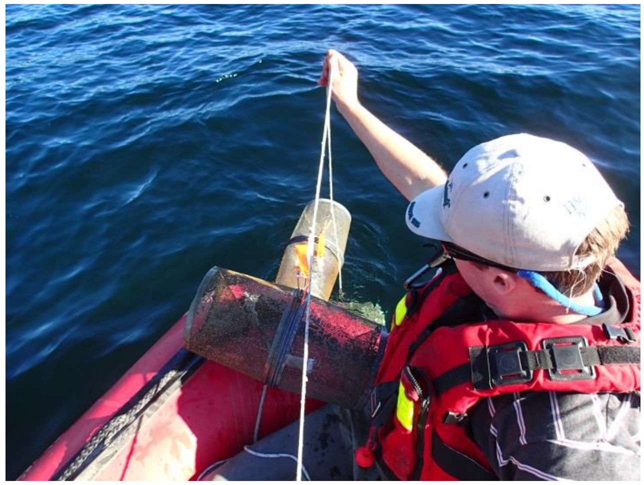
Waterbody	Site	Sampling Date	UTM Zone	UTM Easting	UTM Northing	Water Temp. (°C)	Number of Traps	Position[1]
Lower Campbell Reservoir	LCR-LKMT01	25-Jun-15	10U	324667	5541047	22.0	5	S
	LCR-LKMT02	25-Jun-15	10U	324656	5540794	22.0	5	S[2]
	LCR-LKMT03	23-Aug-15	10U	326042	5542621	22.0	5	S
	LCR-LKMT04	23-Aug-15	10U	324271	5541202	20.7	5	S
	LCR-LKMT05	23-Aug-15	10U	322351	5545333	20.3	5	S
	LCR-LKMT06	23-Aug-15	10U	322250	5544312	20.3	5	S
Beavertail Lake	BVR-LKMT01	22-Jun-15	10U	320432	5539370	22.0	5	B
	BVR-LKMT02	22-Jun-15	10U	320988	5539764	22.0	6	B
	BVR-LKMT03	17-Aug-15	10U	319990	5539754	22.0	5	S
	BVR-LKMT04	17-Aug-15	10U	320619	5539735	22.0	5	S
	BVR-LKMT05	17-Aug-15	10U	320930	5539771	22.0	5	S
	BVR-LKMT06	17-Aug-15	10U	320355	5539468	22.5	5	S
Snakehead Lake	SNA-LKMT01	24-Jun-15	10U	320729	5537745	UNK	10	B
	SNA-LKMT02	21-Aug-15	10U	320570	5537918	21.5	5	S
	SNA-LKMT03	21-Aug-15	10U	320192	5538070	21.5	5	S
	SNA-LKMT04	21-Aug-15	10U	320748	5537866	21.5	5	B
	SNA-LKMT05	21-Aug-15	10U	320310	5538070	21.5	5	B
Upper Quinsam Lake	UPQ-LKMT01	19-Aug-15	10U	317098	5528861	22.1	5	B
	UPQ-LKMT01	22-Jun-15	10U	317098	5528861	24.0	10	B
	UPQ-LKMT02	19-Aug-15	10U	313454	5526107	22.6	5	B
	UPQ-LKMT03	21-Aug-15	10U	320192	5538070	21.5	5	S
	UPQ-LKMT04	21-Aug-15	10U	320748	5537866	21.5	5	S
Middle Quinsam Lake	QUN-LKMT03	25-Jun-15	10U	321264	5533433	24.0	5	B
	QUN-LKMT04	25-Jun-15	10U	322643	5533050	24.0	3	B
Gooseneck Lake	GOO-LKMT03	25-Jun-15	10U	318953	5535887	23.0	5	S
	GOO-LKMT04	25-Jun-15	10U	318810	5535854	23.0	4	S[3]

[1] S, suspended in the watercolumn beneath a buoy; B, secured on the bed

[2] Suspended at depths 2-6 m from tree stumps, not from a buoy.

[3] Suspended at a depth of 5 m from posts on a jetty, not from a buoy.

Figure 17. Deploying minnow traps suspended beneath a buoy at LCK-LKMT01.



2.2.4. Individual Fish Analysis

All fish captured by gill netting, trap netting, or minnow trapping were processed as soon as possible after capture. Sampling details, including target numbers of each species, are presented in Table 5.

The majority of gill netted fish (>90%) did not survive and had already died by the time of net retrieval. Fish were picked out of the net as they were encountered and placed in a tote filled with water. Fork length was measured to nearest 1 mm and mass was measured to the nearest 0.1 g or 1 g for fish over 200 g. Photographs of all processed fish were taken. Some fish that were still alive were quickly measured for fork length and then released. Fish captured using minnow traps and trap nets were all still alive upon capture. These fish were anaesthetized using ENO®, processed as above, allowed to recover in a tote filled with water and then released. Any mortality was noted.

Fin clip samples were collected for stable isotope analysis, with the aim of meeting the sampling targets for each species in Table 5. Small fin clips were taken from the caudal fin of individuals and stored in small vials with 95% ethanol.

A total of 331 scale samples were collected: 20 Cutthroat Trout, 4 Rainbow Trout and 7 Dolly Varden from Beavertail Lake, 32 Cutthroat Trout and 70 Rainbow Trout from Lower Campbell Reservoir, 22 Cutthroat Trout from Snakehead Lake, and 20 Cutthroat Trout from Upper Quinsam Lake. Scale samples were taken from individuals across a range of sizes to ensure that a range of fish ages were captured and so that length-at-age relationships could be built for each species.

Scale samples were examined under a dissecting microscope to determine age at the Ecofish Campbell River laboratory. Representative scales were photographed and apparent annuli were noted using landmarks on a digital image (Figure 18). All scale samples collected at Beavertail, Snakehead, and Upper Quinsam lakes were aged. A subsample of scales was aged from Lower Campbell Reservoir (Cutthroat Trout: n = 18; Rainbow Trout; n = 25). Fish age was determined by a QAQC methodology where by individual scales were initially aged by a junior staff (A-Tlegay and Ecofish) then ages were confirmed by a senior biologist (Ecofish).

Further analysis consisted of defining age class structure and describing other characteristics of the fish populations such as the length-weight relationship, Fulton's condition factor (K), and length at age. Fulton's condition factor (K) was calculated for all captured fish as:

$$K = weight * length^{-3} * 100{,}000$$

DNA samples of Rainbow and Cutthroat Trout were also collected from a subsample of individuals and are currently stored at the Ecofish Laboratory in Campbell River.

Table 5. Sampling requirements for Year 2 fish sampling.

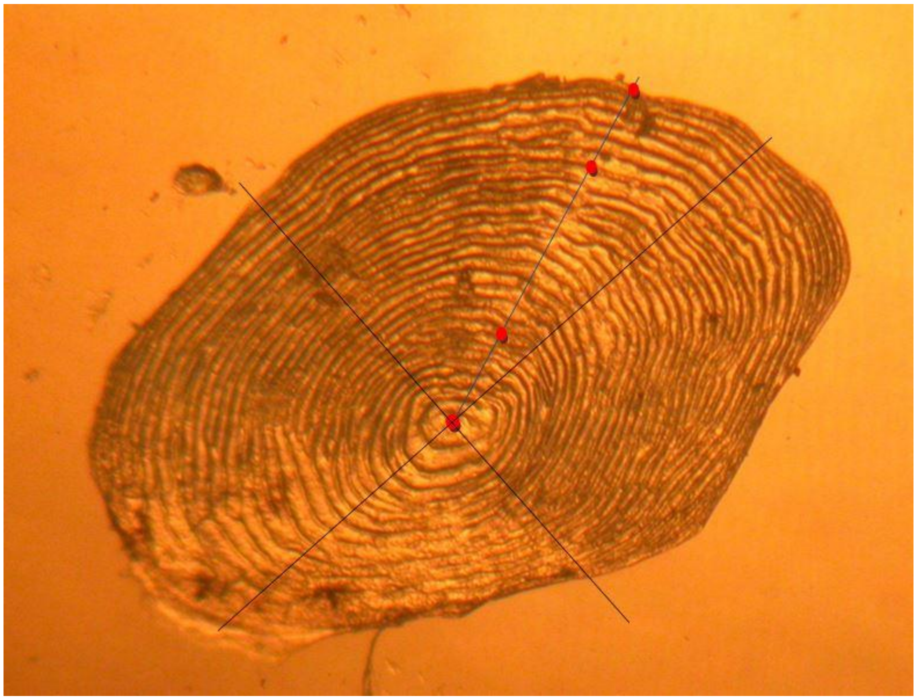
a) Non-lethal sampling requirements

Fish Species	Fork Length (mm)	Target #	Maximum #	Sampling Requirements (non-lethal)
Cutthroat Trout	>150 mm	20	40	• Upper caudal fin clip for SIA, pelvic fin clip for DNA, scales for age analysis, fork length, body mass, photograph, sex/maturity (if possible) • If possible, evenly distribute samples among the full range of body sizes sampled at each lake
Rainbow Trout	>150 mm	20	40	
Juvenile Trout spp.	around 150 mm or less	10	20	• Upper caudal fin clip for SIA, fork length, body mass, photograph • If possible, evenly distribute samples among the full range of body sizes sampled at each lake
Sculpin spp.	all sizes	10	20	
Stickleback	all sizes	10	20	
Dolly Varden	all sizes	10	20	
All species	all fork lengths	all remaining fish		• Once target is reached count and take the fork length of all remaining fish in each lake

b) Lethal sampling requirements

Fish Species	Fork Length	Target #	Maximum #	Sampling Requirements (lethal)
Cutthroat Trout	>150 mm	10	10	• Retain stomachs in formalin from 10 of the large Cutthroat and Rainbow Trout sampled above
Rainbow Trout	>150 mm	10	10	

Figure 18. An example scale aged as a 2+ fish with age annuli indicated with landmarks. The initial landmark and landmark on the outer edge are not counted in the fish age.



2.2.5. Stomach Contents

For comparison to isotope results, fish stomachs were extracted from 17 Cutthroat Trout and 4 Rainbow Trout from Beavertail Lake, 7 Cutthroat Trout and 17 Rainbow Trout from Lower Campbell Reservoir, 22 Cutthroat Trout from Snakehead Lake, and 27 Cutthroat Trout from Upper Quinsam Lake. Stomach contents were examined in the field and were separated into the following broad categories by mass: % zooplankton, % littoral invertebrates (sum of any littoral and terrestrial invertebrates), and % fish.

2.3. Stable Isotope Data

2.3.1. Stable Isotope Processing

Invertebrate and fish samples were processed for nitrogen and carbon stable isotopes at the Stable Isotope in Nature Laboratory (SINLAB) (http://www.unb.ca/research/institutes/cri/sinlab/) located within the Canadian Rivers Institute at the University of New Brunswick in Fredericton, New Brunswick. Dr. Brian Hayden, the Science Manager of SINLAB, was the primary contact.

A total of 246 samples of invertebrates and fish were sent for analysis (Table 6). Invertebrates were sent as whole individuals, while most fish were sent as fin clip samples. Threespine Stickleback was an important target fish species, although individuals were only caught in trap nets from Lower Campbell Reservoir.

All samples were rinsed with distilled water, dried for 48 hours at 60°C and ground into a fine homogeneous powder using a pestle and mortar. Samples were then weighed into tin capsules and loaded into either a PN150 or Costech Zeroblank autosampler. Samples were converted to gases by combustion by a Carlo Erba NC2500 or Costech 4010 Elemental Analyzer (EA) and then analyzed for $\delta^{15}N$ and $\delta^{13}C$ using a Delta Plus or a Delta XP continuous flow isotope-ratio mass spectrometer (CF-IRMS) (ThermoFinnigan; Bremen, Germany) (see SINLAB website).

Isotopic signatures are expressed in delta notation (δ) as ratios relative to known isotopic standards of atmospheric N_2 and Vienna Pee Dee Belemnite (V-PDB) carbon. This is expressed in parts per thousand (‰) according to:

$$\delta^{15}N \text{ or } \delta^{13}C\ (‰) = (R_{sample}/R_{standard} - 1) * 1000$$

where R is the ratio of the heavy isotope (^{15}N or ^{13}C)/ light isotope (^{14}N or ^{12}C).

Thirteen samples were run in duplicate to test repeatability of the stable isotope results. The absolute mean difference in $\delta^{15}N$ between repeats was 0.19±0.15‰. The absolute mean difference in $\delta^{13}C$ between repeats was 0.22±0.16‰.

Table 6. Invertebrate and fish samples sent for stable isotope processing at SINLAB.

Taxa	Lower Campbell Reservoir	Beavertail Lake	Snakehead Lake	Upper Quinsam Lake	Total
Zooplankton	9	9	9	9	36
Littoral Invertebrates	7	3	4	1	15
Stream Invertebrates	1	6	2	4	13
Terrestrial Invertebrates	3	3	3	3	12
Sculpin spp.	11	5	0	5	21
Juvenile Trout	6	1	5	0	12
Threespine Stickleback	10	0	0	0	10
Dolly Varden	0	6	0	0	6
Cutthroat Trout	29	20	21	20	90
Rainbow Trout	27	4	0	0	31
Total	103	57	44	42	246

2.3.2. Assessing Fish Diet Using Mixing Models

The relative contributions of pelagic and littoral sources to Cutthroat Trout, Rainbow Trout, and Dolly Varden diets were assessed through dual isotope ($\delta^{13}C$ and $\delta^{15}N$), four to six-source Bayesian isotopic mixing models implemented in the program SIAR (Stable Isotope Analysis in R; Parnell and Jackson 2013). SIAR takes isotope data from consumers (fish) and sources (diet items) along with estimates of diet-tissue isotopic fractionation, and fits Bayesian models based on Gaussian likelihoods with a Dirichlet prior mixture on the mean, which provide posterior distribution estimates of source contributions to diet (Parnell *et al.* 2010). The diet-tissue fractionation values used in the models were 1.50 ± 1.16 for $\delta^{13}C$ and 2.79 ± 1.46 for $\delta^{15}N$. These are average diet-tissue fractionation rates across several fish species and tissue types (Sweeting *et al.* 2007a, b).

Two models were run for each of the seven lakes. The first model estimated diet contributions to large Cutthroat Trout, Rainbow Trout, and Dolly Varden (Age >2+, FL ≥ 159 mm). Six potential diet sources (mean $\delta^{13}C$ and $\delta^{15}N$ ± SD) for the three large fish species were included in this model: 1) zooplankton, 2) littoral invertebrates, 3) stream invertebrates, 4) terrestrial invertebrates, 5) littoral prey fish (juvenile trout (Age ≤ 2, FL ≤ 152) and sculpin (FL ≤ 170 mm)), and 6) Threespine Stickleback (FL ≤ 64 mm). The second model run for each lake estimated the diet contributions to the smaller prey fish (littoral prey fish, and Threespine Stickleback). Four potential diet sources (mean $\delta^{13}C$ and $\delta^{15}N$ ± SD) were used to estimate the smaller prey fish diets: 1) zooplankton, 2) littoral invertebrates, 3) stream invertebrates, and 4) terrestrial invertebrates.

The two models were run to assess the total relative contributions of pelagic vs. littoral sources of production to large Cutthroat Trout, Rainbow Trout, and Dolly Varden via direct and indirect pathways. The total littoral vs. pelagic contribution can be derived by summing the contributions of the invertebrate prey to large trout and Dolly Varden diet in model one (direct pathway) with the relative contributions of invertebrate prey to the diets of small fish (model 2) that occur in the diets of large trout and Dolly Varden (indirect pathway). The direct pathway (model 1) is the contribution of zooplankton (pelagic) and summed contribution from littoral, stream, and terrestrial invertebrates (littoral) to large trout and Dolly Varden diets. The indirect contribution (derived from model 1 and model 2) is the proportional contribution of pelagic and littoral sources to the small prey fish diets that is carried forward to the diets of large trout and Dolly Varden.

2.3.3. Correlations with Fish Size and Age

As fish become larger they tend to eat larger prey. It is also possible that Cutthroat Trout and Rainbow Trout shift among pelagic and littoral sources of production as they grow and age. Basic linear regression models were built for both Cutthroat Trout and Rainbow Trout to test the relationships between $\delta^{15}N$ and $\delta^{13}C$ stable isotope signatures and fish length or fish age. All analyses were conducted using the statistical program R (R Core Team 2014).

2.4. Water Residency Analysis

2.4.1. General

Annual and seasonal water residence time was estimated for Upper Campbell and Lower Campbell reservoirs, and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes. Where surface outflow of the lake is directly measured, the water residence time (τ_w) was calculated by dividing the annual average outflow (Q_o) rate by the volume of the lake basin (V). The seasonal average elevation of the lake was used for the seasonal water residence time computations, and the average annual elevation of the lake was used for the annual water residence time computations. To estimate residence time during the growing season, an effective lake volume, defined as the average volume of the mixed layer (epilimnion) times the proportion of year that the lake is stratified, was used. Based on historical lake temperature records, stratification is assumed to set in within each lake around May 15 and to break down around September 30 (Hatfield 2000).

Lake volumes were obtained from stage-volume relationships developed by BCH (Bruce 2001) and bathymetric maps. The volume of the Upper Campbell Reservoir included the storage of both Upper Campbell Lake and Buttle Lake (BCH 2012). The volume of the epilimnion was determined from the average thermocline depth of the lakes (Section 2.4.2). The thermocline depth was then related to volume using the stage-volume relationships developed by BCH (Bruce 2001).

Where surface outflow is unknown, the outflow rate (Q_o) was estimated from the following:

$$Q_o = Q_i + ((P - E) \times A) \quad (1)$$

where Q_i is inflow rate (m^3/day), P is precipitation (m/day), E is evaporation (m/day), and A is the area of the lake (m^2). Where Q_o is known (e.g., Upper Campbell Reservoir and Lower Campbell Reservoir) the estimated Q_o was compared to the measured Q_o to determine relative accuracy of the computed outflow rates using Equation (1). Equation (1) is a water balance method that neglects potential groundwater recharge or net change in storage.

Where the inflow rate (Q_i) is unknown, it was estimated by computing a runoff coefficient for the watershed. The runoff coefficient was estimated from local precipitation and climate data, with refinements for geology, local hydrology, slope, and land use (Ward and Elliot 1995). Where Q_i is known (e.g., Lower Campbell Reservoir), the estimated runoff coefficient was compared to the measured Q_i to determine relative accuracy of the computed runoff. BC Hydro also operates flow gauges on either side of the Quinsam diversion that provide a check on the order of magnitude of inflow data computed for Gooseneck, Middle Quinsam, and Snakehead lakes. A summary of the hydrometric stations used in the inflow and outflow calculations is provided in Table 7; Map 6 shows the locations of the stations relative to the study lakes.

Table 7. Hydrometric stations used in the water residence time computations.

Lake/Reservoir	Hydrometric Station Name	Station No.	Parameters Recorded		
			Level (m)	Inflow (Q_i) (m^3/s)	Outflow (Q_o) (m^3/s)
Upper Campbell	Buttle Lake Above Campbell Lake	08HD033	x[1]		
	Elk River Above Campbell Lake	08HD018	x	x	
	Upper Campbell Lake at Strathcona Dam	08HD031	x		x[2]
Lower Campbell	Upper Campbell Lake at Strathcona Dam	08HD031	x	x[1]	
	Salmon River Diversion Near Campbell River	08HD020	x	x	
	BC Hydro Ladore Dam	n/a			x[2]
Gooseneck	Quinsam Diversion Near Campbell River	08HD026	x	x	
Middle Quinsam	Quinsam River at Argonaut Bridge	08HD021	x	x	

[1]Data used only to determine connectivity between Upper Campbell Reservoir and Buttle Lake.

[2]Data provided by BC Hydro.

2.4.2. Thermocline Depth

The presence of a thermocline can influence the water residence time in lakes. Lakes often exhibit strong stratification of temperature due to density differences of water, which occurs most often during summer months. Seasonal stratification limits the extent of vertical mass transport. Under stratified conditions, inflows will form intrusions of limited vertical extent, outflows will be typically drawn from a narrow range of depths, and there can be parts of the lake that become isolated from the inflow and outflow processes. These isolated pieces of water can remain in the lake for much longer periods of time than predicted from the ratio of lake volume and annual flow rate. The presence and depth of the thermocline was used as a means to characterize stratified conditions within the reservoirs and lakes of the Campbell River system.

To determine the presence and depth of a thermocline, lake temperature profiles were collected at six of the study lakes (Lower Campbell Reservoir, Beavertail Lake, Gooseneck Lake, Middle Quinsam Lake, Snakehead Lake and Upper Quinsam Lake) in September 2015. TidbiT v.2 temperature data loggers (Onset) were attached at 1-2 m intervals to a rope with an anchor and lowered at the deepest point of the lake, which was ascertained from bathymetric maps of the lakes. The lake temperature profiles collected represent only a snapshot of the thermal conditions for a particular location on a particular date, without any spatial or temporal variation. As the profiles were collected in late summer, the estimated growing season thermocline depths are likely biased high. To increase the sample size, additional temperature measurements were used to define the depth of the thermocline in the selected lakes. These temperature measurements were extracted from a number of unpublished reports catalogued in the library of MELP (Vancouver Island Region, Nanaimo). A summary of these data are provided in Hatfield (2000), Appendix A.

2.4.3. Precipitation and Evaporation

Local precipitation data were obtained to determine the daily atmospheric inputs to the study lakes, and to derive water inflow from their drainage basins. There are a number of climate stations within the WUP study area; however, only two stations had continuous precipitation data within the vicinity of the study lakes. Daily precipitation data were obtained from the BCH climate stations at Strathcona Dam (SCA) and Salmon River above the Diversion (QIN) (both accessed from PCIC 2013) (Table 8). In 2015, precipitation data were only available from Station SCA. For the other years, data were obtained from the climate station located within the lake watersheds, and/or the station at a similar elevation to the lake (Table 9). The SCA and QIN climate stations only record minimum and maximum daily air temperature. The closest climate station with a continuous record of daily air temperature for the study period is Environment Canada's Campbell River A Station. The average daily air temperature recorded at this station was used to compute lake evaporation (Table 10). The locations of the climate stations are shown in Map 1.

Table 8. Climate stations used in the water residence time computations.

Climate Station Name	Station No.	Elevation (m)	Parameters Recorded		Time Interval
			Air Temperature (deg. C)	Precipitation (mm)	
Strathcona Dam (SCA)	2501	227	x	x	Daily 2012 - 2015
Quinsam River at Argonaut Bridge (QIN)	2498	280	x	x	Daily 2012 - 2014
Campbell River Airport Climate Station (A)	6678	108	x	x	Daily 2012 - 2015

Table 9. Summary of the climate station used to obtain precipitation data for each study waterbody

Waterbody	Precipitation	
	SCA	QIN
Upper Campbell Reservoir	x	
Lower Campbell Reservoir	x	
Beavertail Lake		x
Gooseneck Lake		x
Middle Quinsam Lake		x
Snakehead Lake		x
Upper Quinsam Lake		x

Lake evaporation estimates are required to determine the evaporative losses from the study lakes. There are no direct measurements of evaporation from the WUP study lakes, and continuous measurements of net radiation, wind speed, and humidity are not available to compute evaporation. Two empirical methods were used to estimate monthly total potential evapotranspiration (PET) from the study lakes based on air temperature. Evaporation from open water is equivalent to potential evapotranspiration, primarily because in both conditions the supply of water is non-limiting, meaning there is an adequate supply of water for evaporative processes.

The Thornthwaite formula (1948) is commonly used to estimate evaporation from lakes when data are sparse and is given by,

$$ET_i = 16\left(\frac{10T_i}{I}\right)^{\propto} \qquad (2)$$

where ET_i is potential evapotranspiration for month i (mm/month); T_i is mean monthly air temperature (°C), obtained from Environment Canada's Campbell River Station A (PCIC 2013); I is the local heat index given by,

$$I = \sum_{i=1}^{12}\left(\frac{T_i}{5}\right)^{1.514} \qquad (3)$$

and the coefficient $\alpha = \left(0.675 \times I^3 - 77.1 \times I^2 + 17{,}920 \times I + 492{,}390\right) \times 10^{-6}$ (Xu and Singh 2001, Equations 4a and 4b). The formula is for a month with 30 days and was adjusted for latitude and number of days in a month by multiplying the calculated ET_i by a correction factor (Dunne and Leopold 1978, Table 5-2). Thornthwaite evaporation values have been found to compare well with values obtained using the Bowen-ratio energy budget method for a small mountain lake in northeastern USA (Rosenberry *et al.* 2007), and energy budget values for a wetland in North Dakota, USA (Rosenberry *et al.* 2004). However, Thornthwaite values have been have been found to overestimate potential evaporation in the summer months when compared to results from the Penman PET formula (Shaw 1994), a physically-based formula for estimating potential evapotranspiration.

Due to the uncertainty in Thornthwaite evaporation estimates during summer months, the Hamon method (1961) was also used to derive monthly estimates of potential evapotranspiration. The Hamon method has been shown to provide reasonable estimates of evaporation when compared to evaporation computed from energy budget method, and within 20% of annual reservoir evaporation from pan data (pan estimates of evaporation) (Dalton *et al.* 2004, Rosenberry *et al.* 2007). The Hamon method is based on the mean air temperature and is expressed as

$$ET = 0.55\left(\frac{D}{12}\right)^2 \cdot \frac{SVD}{100} \cdot 25.4 \quad (4)$$

where ET is potential evapotranspiration (mm/day), D is the hours of daylight for a given day (in units of 12 h) and Pt is a saturated water vapour density term calculated by

$$Pt = \frac{4.95e^{(0.062T_a)}}{100} \quad (5)$$

where *Ta* is daily mean air temperature (°C), SVD saturated vapor density at mean air temperature (g m^{-3}).

Total annual and seasonal (May 15 to October 1) lake evaporation estimates varied considerably between the two methods (Table 10). The difference in the average total annual precipitation calculated was 150 ± 6 mm; the seasonal difference in total evaporation estimates was even greater (232 ± 47 mm). A literature search was conducted to determine the most reliable estimates of lake evaporation for the study lakes. Lake evaporation estimates are available for two areas on Vancouver Island: Ladysmith and Salt Spring Island (Sprague 2007). Though these are southern locations, some of these lakes are at a similar elevation to the lakes within the Campbell River area.

Lake evaporation was estimated for lakes within the Ladysmith area on Vancouver Island (Tetra Tech EBA 2014). The mean annual evaporation for these lakes was determined to be 642 mm. Similar estimates of annual evaporation were determined for lakes on Salt Spring Island (713 mm and 585 mm) (reported in Sprague 2007). Evaporation was found to lower lake levels on Salt Spring Island by about 0.3 m during the growing season (Sprague 2007). Another study of the same lakes, estimated total seasonal lake evaporation to be approximately 411 mm (Barnett et al. 1993). These estimates agree well with the annual and seasonal evaporation estimates computed from the Thornthwaite formula (1948) (Table 10), and were therefore used to compute lake evaporation.

Table 10. Total annual and seasonal lake evaporation computed from the Hamon (1961) method and Thornthwaite (1948) formula for the years 2012-2015.

	Thornthwaite Formula **Evaporation[1] (mm)**	Hamon Method **Evaporation[1] (mm)**
Annual		
2012	616	467
2013	629	472
2014	624	480
2015	654	498
Seasonal[2]		
2012	432	229
2013	444	230
2014	541	240
2015	454	243

[1] Computed using average daily air temperature recorded at Environment Canada's Climate Station A
[2] May 15 - October 1

The average annual and seasonal precipitation and potential evapotranspiration data are provided in Table 11. Precipitation varied according to location and year. On average, the difference in annual precipitation between the two stations was 119 mm, and the average seasonal (May 15-October 1) difference in precipitation was 9 mm. The annual precipitation was lowest at both stations in 2013 with values 393 mm (SCA) and 338 mm (QIN) less than the highest precipitation recorded in other years. During the same year, the precipitation from May 15 to October 1 was the highest relative to the other years. For historical context, the average total annual precipitation recorded at SCA from 1981 to 2015 was 1359 mm; the average total seasonal precipitation for the same period was 236 mm. Precipitation data were available at the QIN station from 1994-2014. During this time, the average total annual precipitation was 1183 mm, and the average total seasonal precipitation was 242 mm. The range in annual and seasonal precipitation values over the study period (2012-2015) provide valuable information on how water residence times vary between years with lower than average, average, and higher than average precipitation.

Total annual lake evaporation ranged from 616 to 654 mm (Table 11). In 2012, there was a significantly greater estimated annual loss of water at those lakes in similar proximity and/or elevation to the Quinsam (QIN) BC Hydro climate station than those near the Strathcona Dam climate station (SCA). Seasonal (May 15 to October 1) lake evaporation estimates ranged from 432 to 541 mm, resulting in a net loss of lake water in all years (Table 11).

Table 11. Total annual and seasonal precipitation and estimated lake evaporation data used in the water residence time computations.

	SCA[1] Precipitation (mm)	QIN[2] Precipitation (mm)	Evaporation[3] (mm)	SCA P- E (mm)	QIN P- E (mm)
Annual					
2012	1424	1185	616	808	569
2013	1031	902	629	402	273
2014	1229	1240	624	605	616
2015	1253	no data available	654	599	SCA data used
Seasonal [4]					
2012	204	183	432	-228	-249
2013	400	414	444	-43	-30
2014	175	208	541	-367	-333
2015	220	no data available	454	-233	SCA data used

[1] BC Hydro climate station at Strathcona Dam (SCA).
[2] BC Hydro climate station at Salmon River above the Diversion (QIN).
[3] The Thornthwaite (1948) formula was used to determine evaporation.
[4] May 15 - October 1

2.4.4. Inflow Rate

To estimate the inflow rate (Q_i), an algorithm was created using a modified Soil Conservation Service (SCS) runoff curve number (CN) method (SCS 1972). The runoff curve number (CN) is a coefficient that reduces the total precipitation to runoff potential, after losses to evaporation, absorption, transpiration, and surface storage have been taken into account. In its modified form, the model is based on using the following equation:

$$Q_i = \frac{(P-I_a)}{P-I_a+S} \qquad (6)$$

where Q_i is runoff (mm), P is precipitation (mm), I_a is initial abstractions (water retained in surface depressions, intercepted by vegetation, evaporation, and infiltration expressed in mm), and S is the potential maximum capacity of retention after runoff begins (mm).

Parameter I_a is equivalent to

$$I_a = 0.2 \cdot S \cdot \beta, \qquad (7)$$

where coefficient β accounts for the water retention capacity of vegetation as function of different land uses of a watershed (Crăciun *et al.* 2009). The ratio of I_a/S=0.2 is commonly used in the scientific literature, as this was the original relationship published. However, this ratio was developed through studies of many small agricultural watersheds. Hawkins *et al.* (2002) examined the ratio of I_a/S using rainfall and runoff data from numerous watersheds in the U.S. They found that over 90 percent of the ratios were less than 0.2. In this study, the ratio is adjusted by the coefficient β, as it provides a physical basis for adjusting the I_a/S relationship that is directly related to each individual watershed. Coefficient β was determined by computing an area-weighted runoff coefficient for each watershed that reflects the percent cover of different vegetation and surface types within the watershed. The type and area (km^2) of the different surface covers were determined using the British Columbia Forest and Vegetation Cover Resources Inventory (MFLNRO 2016a) and GIS spatial analysis and mapping functions.

Parameter S, the potential maximum water retention, is expressed as

$$S = \frac{25.400}{CN} - 254 \qquad (8)$$

where the CN index is a determined according to land use, soil hydrological group (A, B, C, D) and antecedent moisture conditions. The higher the CN value, the higher the runoff potential will be. The majority of the surface soils within the study area are Humo-Ferric Podzols (Valentine *et al.* 1978). These soils are classified as Soil Group B (Soil Conservation Service, 1979), and are moderately well drained (i.e., the upper meter of soil is not saturated for long durations).

The watershed's curve number represents the spatial variability of runoff and was derived for hydrologic soil group B and the various land uses and hydrologic conditions of the watershed, from tabulated values published in Chapter 9 of the National Engineering Handbook of Hydrology (USDA-SCS 1985). The type and area (km^2) of the different surface land uses were determined using the British Columbia Baseline Thematic Mapping Present Land Use Inventory (MFLNRO 2016b) and GIS spatial analysis and mapping functions. An area-weighted curve number was computed based on the different land uses within the watershed, and then corrected for the average slope of the watershed.

For each day, the area-weighted-average curve number was adjusted according to the five-day antecedent rainfall amount to account for the temporal variability of runoff in the watershed. The adjustments were different for the growing season (April 1-September 30) and the dormant season (October 1 to March 31) (Table 12).

Table 12. Antecedent moisture conditions used to adjust area-weighted curve numbers.

Antecedent Condition	Growing Season[1]	Dormant Season[2]
	5-Day Antecendant Rainfall	5-Day Antecendant Rainfall
Dry AMC I	< 35 mm	< 12 mm
Average AMC II	35-53 mm	12-28 mm
Wet AMC III	> 53 mm	> 28 mm

[1] April 1 to September 30

[2] October 1 to March 31

The watershed characteristics used to determine inflow rate are presented in Table 13. A breakdown of inflow, evaporative loss, and outflow computed for each of the study lakes is provided in Table 14 and Table 15 for the annual and seasonal periods, respectively. Upper Campbell and Lower Campbell reservoirs had the highest annual and seasonal inflows and outflows, and Snakehead and Beavertail lakes had the lowest.

Table 13. Watershed characteristics, including drainage area, average slope of the watershed, runoff coefficients, runoff curve numbers (*CN*), and soil retention capacity (*S*).

Waterbody	Watershed Area[1] (km^2)	Average Slope (%)	Runoff Coefficient	*CN* (I)[2]	*CN* (II)[3]	*CN* (III)[4]	*S* (I)[2] (mm)	*S* (II)[3] (mm)	*S* (III)[4] (mm)
Upper Campbell Reservoir	1192.8	56.3	0.46	43	69	97	338	113	8
Lower Campbell Reservoir	1422.3	49.2	0.46	48	66	80	275	132	65
Beavertail Lake	5.7	10.3	0.47	39	58	76	396	182	81
Gooseneck Lake	99.1	15.1	0.48	40	60	78	382	172	74
Middle Quinsam Lake	111.7	25.6	0.46	40	60	78	379	170	72
Snakehead Lake	19.8	13.7	0.48	40	60	77	382	172	74
Upper Quinsam Lake	84.6	30.9	0.46	40	60	78	380	171	73

[1]Watershed area for Gooseneck and Middle Quinsam lakes include the local watershed area plus the area upstream of the diversion.

[2]Area-averaged, slope corrected curve number (*CN*) and soil retention (*S*) used for dry soil conditions.

[3]Area-averaged, slope corrected curve number (*CN*) and soil retention (*S*) used for average soil conditions.

[4]Area-averaged, slope corrected curve number (*CN*) and soil retention (*S*) used for wet soil conditions.

Table 14. Total annual inflow (Q_i), precipitation (P) minus evaporation (E) multiplied by lake area, and lake outflow (Q_o) computed for 2012-2015.

Waterbody	2015			2014			2013			2012		
	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)
Upper Campbell Reservoir	1.48E+09	4.E+07	1.52E+09	2.26E+09	4.15E+07	2.30E+09	1.26E+09	2.76E+07	1.29E+09	2.06E+09	5.55E+07	2.12E+09
Lower Campbell Reservoir	1.59E+09	2.E+07	1.60E+09	9.27E+08	1.30E+07	9.40E+08	8.26E+08	8.64E+06	8.35E+08	9.22E+08	1.74E+07	9.39E+08
Beavertail Lake	3.52E+06	6.E+05	4.13E+06	3.96E+06	6.34E+05	4.59E+06	4.08E+06	2.81E+05	4.36E+06	3.89E+06	5.85E+05	4.47E+06
Gooseneck Lake	3.07E+07	5.E+05	3.12E+07	3.81E+07	4.81E+05	3.86E+07	4.61E+07	2.13E+05	4.63E+07	3.25E+07	4.44E+05	3.29E+07
Middle Quinsam Lake	6.22E+07	4.E+05	6.27E+07	7.11E+07	4.47E+05	7.16E+07	6.72E+07	1.98E+05	6.74E+07	7.18E+07	4.12E+05	7.22E+07
Snakehead Lake	3.40E+07	1.E+05	3.41E+07	4.02E+07	1.25E+05	4.03E+07	4.61E+07	5.52E+04	4.62E+07	3.55E+07	1.15E+05	3.56E+07
Upper Quinsam Lake	5.97E+07	3.E+06	6.24E+07	7.06E+07	2.70E+06	7.33E+07	7.28E+07	1.20E+06	7.40E+07	6.71E+07	2.49E+06	6.96E+07

Table 15. Total inflow (Q_i), precipitation (P) minus evaporation (E) multiplied by lake area, and lake outflow (Q_o) computed for from May 15 to October 1, 2012-2015.

Waterbody	2015			2014			2013			2012		
	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)	Q_i (m^3)	(P-E)*Lake Area (m^3)	Q_o (m^3)
Upper Campbell Reservoir	4.15E+08	-8.81E+06	4.07E+08	4.26E+08	-1.39E+07	4.12E+08	4.60E+08	5.08E+06	4.66E+08	4.47E+08	-8.66E+06	4.38E+08
Lower Campbell Reservoir	1.45E+09	-3.40E+06	1.45E+09	4.11E+08	-4.36E+06	4.06E+08	3.77E+08	1.59E+06	3.79E+08	4.28E+08	-2.71E+06	4.26E+08
Beavertail Lake	1.59E+06	-1.32E+05	1.46E+06	2.16E+06	-1.75E+05	1.98E+06	1.93E+06	9.06E+04	2.02E+06	2.23E+06	-1.51E+05	2.08E+06
Gooseneck Lake	2.14E+07	-1.00E+05	2.13E+07	2.09E+07	-1.33E+05	2.07E+07	1.83E+07	6.86E+04	1.84E+07	1.98E+07	-1.15E+05	1.97E+07
Middle Quinsam Lake	1.91E+07	-9.29E+04	1.90E+07	2.91E+07	-1.23E+05	2.90E+07	2.84E+07	6.38E+04	2.84E+07	4.03E+07	-1.06E+05	4.02E+07
Snakehead Lake	2.29E+07	-2.59E+04	2.28E+07	2.12E+07	-3.43E+04	2.11E+07	2.11E+07	1.78E+04	2.11E+07	2.07E+07	-2.97E+04	2.07E+07
Upper Quinsam Lake	2.60E+07	-5.62E+05	2.54E+07	3.28E+07	-7.45E+05	3.21E+07	3.00E+07	3.86E+05	3.04E+07	3.87E+07	-6.44E+05	3.80E+07

3. RESULTS

3.1. Invertebrate Sampling

3.1.1. Zooplankton

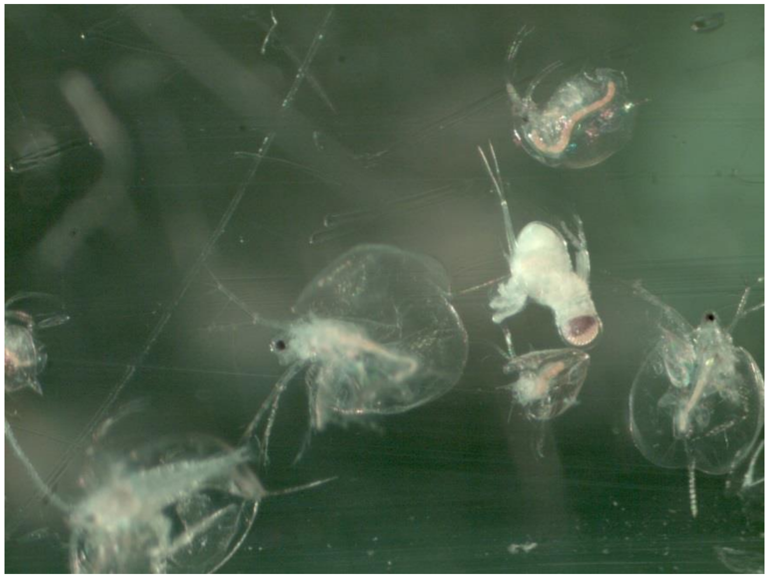
3.1.1.1. Abundance

Dominant zooplankton taxa were: Daphniidae, Cyclopoida, Calanoida, Bosminidae, Sididae, Polyphemidae and Leptodoridae. Together, these taxa comprised >99% of the individuals sampled in 2015, with 85% of individuals belonging to one of three taxa: Daphniidae, Calanoida and Cyclopoida (Table 16). Daphniidae was the most abundant taxon in Lower Campbell Reservoir and Snakehead Lake, where it respectively comprised 51% and 50% of the individuals sampled (Figure 19). Cyclopoida was the most abundant taxon in Beavertail Lake and Upper Quinsam Lake, where it respectively comprised 42% and 32% of the individuals sampled. Between 9% and 17% of individuals in each lake belonged to the Calanoida[2] order. Individuals belonging to the following taxa were also occasionally recorded, typically in only a single sample collected throughout the season: Arachnida, Radiolaria, Oligochaeta, Tricladida and Gammaridae.

The taxonomic compositions of samples were broadly similar to those of the three lakes sampled in Year 1, when Daphniidae was the most abundant family in each lake (Hocking *et al.* 2015). It is notable though that copepods comprised a greater relative proportion of the samples in Year 2. For example, only 0.2–1.4% of individuals from each lake were assigned to the family Cyclopoida in Year 1, whereas 19–42% of individuals from each lake were assigned to this family in Year 2.

[2] This order was not identified to a lower taxonomic level but, based on the results of Year 1 sampling, this order likely comprised *Diaptomus* spp. (Hocking *et al.*, 2015).

Figure 19. Example of zooplankters collected during Year 2 sampling. The photograph shows multiple Daphniidae individuals and a single Polyphemidae individual (*Polyphemus pediculus*; note fused compound eyes)



3.1.1.2. Biomass

Table 17 presents a summary of length measurements, published length–biomass relationships and mean biomass values by taxa that were used to estimate total zooplankton biomass by lake in Year 1 and Year 2 samples. Zooplankton biomass was not estimated during Year 1; instead, taxon–specific biomass values estimated in Year 2 were used to estimate biomass (µ/L) for samples collected during both years. Consequently, zooplankton biomass estimates for both Year 1 (Figure 20) and Year 2 (Figure 21) are presented here.

In Year 1, zooplankton biomass in each sample was 9–106 µg/L (mean = 34 µg/L; standard deviation = 27 µg/L; Figure 20). In Year 1, biomass was highest in Middle Quinsam Lake in June and July (40–106 µg/L) and highest in Gooseneck Lake in August (48–49 µg/L). Zooplankton biomass was consistently lowest in Upper Campbell Lake reservoir in Year 1 (9–21 µg/L).

In Year 2, zooplankton biomass was 3–47 µg/L (mean = 19 µg/L; standard deviation = 13 µg/L; Figure 21). There were no consistent differences in biomass between lakes, although biomass was generally greatest in Beavertail Lake in June and July. There was indication of a relative decline in biomass in the samples collected in early September in Year 2, with biomass in all lakes generally lower during this month relative to earlier in the season (Figure 21). This contrasts with evidence that there was a slight increase in mean length of individuals as the growing season progressed (Figure 5), and it was qualitatively observed during taxonomic analysis that lipid content (and thus presumably biomass) increased in individuals that were sampled in September. Thus, it seems that abundance and overall biomass generally declined during September, although the mean biomass of individuals may have been slightly higher.

It is notable that there was sometimes substantial variation in biomass between replicate samples collected at the same site (e.g., compare replicates in June 2014 for GOO–LKZP01) or samples collected at different sites during one day on the same lake (e.g., compare data for LCR sites sampled in June 2015).

Daphniidae was typically the family that made the dominant contribution to biomass in samples collected during both Year 1 (Figure 20) and Year 2 (Figure 21), reflecting both the relative high abundance of this taxon (Section 3.1.1.1), and the relatively high estimated biomass of individuals (Table 17). Daphniidae individuals comprised >50% of the biomass of most samples. Bosminidae individuals usually made the second–greatest contribution to biomass in samples for Year 1, when they comprised an average of 22% of the biomass in samples. Note that the relatively high biomass of Bosminidae individuals (6.20 µg; Table 17) meant that, relative to abundance, this taxon made a disproportionately higher contribution to biomass in each sample compared with copepods (Calanoida and Cyclopoida), which were estimated to have much lower biomass (2.60 µg and 0.96 µg respectively; Table 17) based in length measurements. In Year 2, the taxon that made the second–greatest contribution to biomass was typically either Bosminidae, Calanoida or Cyclopoida; the mean contribution to overall biomass of each of these three taxa was 11% across all Year 2 samples.

Table 16. Zooplankton abundance, 2015.

Waterbody	Site	Month	Zooplankton abundance (individuals/L)													
			Arachnida	Bosminidae	Calanoida	Cyclopoida	Daphniidae	Gammaridae	Leptodoridae	Nauplii	Oligochaeta	Onychopoda	Polyphemoidea	Radiolarian	Sididae	Tricladida
Lower Campbell Reservoir	LCR-LKZP01	June	23	1517	204	1200	5908	0	181	45	0	45	0	0	136	0
		July	23	226	385	1324	3610	0	170	34	0	147	0	0	147	0
		Sept	0	147	447	945	390	0	28	11	0	28	0	0	85	0
	LCR-LKZP02	June	0	110	11	81	338	0	20	0	0	7	0	0	4	0
		July	11	79	192	1086	2071	0	147	23	0	45	0	0	45	0
		Sept	0	221	821	1630	985	0	17	6	0	23	0	0	108	0
	LCR-LKZP03	June	34	656	238	883	2139	11	79	0	0	0	0	0	34	0
		July	57	181	453	1143	3407	0	204	45	0	170	0	0	124	0
		Sept	11	430	577	1324	724	0	34	6	0	11	0	0	153	0
Abundance (all months; %)			**<1%**	**9%**	**9%**	**25%**	**51%**	**<1%**	**2%**	**0%**	**0%**	**1%**	**0%**	**0%**	**2%**	**0%**
Beavertail Lake	BVR-LKZP01	June	0	244	279	3517	6338	0	279	0	0	70	0	0	139	0
		July	0	296	1097	6251	3291	0	70	0	0	0	0	0	435	35
		Sept	0	200	897	3160	688	0	0	0	0	26	0	0	322	0
	BVR-LKZP02	June	0	302	1117	3712	5221	0	151	0	0	0	272	0	302	0
		July	0	211	1086	6278	3712	0	60	30	0	30	0	0	483	0
		Sept	0	83	445	2015	792	0	30	0	0	0	0	23	264	0
	BVR-LKZP03	June	0	0	7666	0	362	0	0	181	0	2052	0	0	483	0
		July	0	91	1720	8451	4678	0	91	30	0	91	0	0	785	0
		Sept	0	91	226	1690	672	0	38	0	0	8	0	0	400	0
Abundance (all months; %)			**0%**	**2%**	**17%**	**42%**	**31%**	**0%**	**1%**	**0%**	**0%**	**3%**	**<1%**	**<1%**	**4%**	**0%**
Snakehead Lake	SNA-LKZP01	June	0	85	821	1188	4499	0	198	0	0	198	0	0	340	0
		July	0	0	1273	990	6196	0	368	0	0	85	0	0	198	0
		Sept	0	212	863	2065	297	0		0	0		0	0	2617	0
	SNA-LKZP02	June	0	0	792	616	3182	0	126	0	0	25	0	0	126	0
		July	0	0	855	453	2301	0	365	0	0	13	0	0	340	0
		Sept	0	13	453	1408	1459	0	101	0	13	13	0	0	1169	0
	SNA-LKZP03	June	0	32	125	566	1265	0	36	4	4	12	0	0	16	0
		July	0	14	137	216	622	0	30	4	0	18	0	0	67	0
		Sept	0	8	356	614	954	0	40	0	0	8	0	0	420	0
Abundance (all months; %)			**0%**	**1%**	**14%**	**19%**	**50%**	**0%**	**3%**	**<1%**	**0%**	**1%**	**0%**	**0%**	**13%**	**0%**
Upper Quinsam Lake	UPQ-LKZP01	June	0	634	498	1256	1369	0	91	11	0	11	0	0	0	0
		July	23	487	736	1596	1890	0	34	0	0	102	0	0	407	0
		Sept	0	91	215	1850	181	0	0	0	0	17	0	0	147	0
	UPQ-LKZP02	June	0	1007	328	1154	770	0	57	0	0	0	0	0	11	0
		July	0	170	1053	1483	1743	0	45	0	0	23	0	0	713	0
		Sept	0	136	385	2162	249	0	57	6	0	6	0	0	158	0
	UPQ-LKZP03	June	0	487	538	838	407	0	6	0	0	17	0	0	317	0
		July	0	407	453	1211	1437	0	23	0	0	45	0	0	260	0
		Sept	0	108	232	877	390	0	119	0	0	11	0	0	130	0
Abundance (all months; %)			**<1%**	**8%**	**11%**	**30%**	**20%**	**0%**	**1%**	**<1%**	**0%**	**1%**	**0%**	**0%**	**5%**	**0%**
Total abundance (all samples; %)			**0%**	**5%**	**14%**	**33%**	**38%**	**0%**	**2%**	**0%**	**0%**	**2%**	**0%**	**0%**	**6%**	**0%**

Table 17. Summary of data and relationships used to estimate the mean biomass of dominant zooplankton taxa.

Taxon	Sample: Lake	Sample: Month	# of individuals measured	Mean length (µm)	Std. deviation (µm)	Biomass (*W*) ~ length (*L*) relationship	Estimated mean biomass (µg/individual)	Reference
Daphniidae	Beavertail	June	16	1103	259	$\ln W = 1.51 + 2.56 \cdot \ln L$	5.30	Dumont *et al*. (1975)
	Lower Campbell	June	18	1231	208		7.44	
	Snakehead	June	15	1113	266		5.53	
	Upper Quinsam	June	20	1002	240		4.21	
	Upper Quinsam	July	20	1143	266		5.93	
	Upper Quinsam	Sept	20	1300	321		8.16	
Bosminidae	Beavertail	July	15	932	979	$\ln W = 2.711 + 2.529 \cdot \ln L$	6.20	Bottrell *et al*. (1976)
Sididae (*Diaphanosoma* spp.)	Beavertail	July	15	930	531	$\ln W = 1.289 + 3.039 \cdot \ln L$	1.90	Rosen (1981)
Polyphemidae (*Polyphemus pediculus*)	-	-	-	700 [1]	-	$\ln W = 2.779 + 2.152 \cdot \ln L$	7.47	Rosen (1981)
Leptodoridae (*Leptodora kindtii*)	Beavertail	July	4	1763	655	$\ln W = -0.822 + 2.67 \cdot \ln L$	1.70	Rosen (1981)
Cyclopoida	Beavertail	July	19			$\ln W = 1.953 + 2.399 \cdot \ln L$	0.96	Bottrell *et al*. (1976)
Calanoida	Beavertail	July	19			$\ln W = 1.953 + 2.399 \cdot \ln L$	2.60	Bottrell *et al*. (1976)
Naupilii	-	-	-	-	-	Constant biomass assigned.	0.40	Hawkins and Evans (1979)

1. Body lengths were not measured for this species, which was typically present in low abundance (usually <20 individuals/sample). *L* based on the middle of the range of body lengths (0.7 mm) measured by Rosen (1981).

Figure 20. Zooplankton biomass for Year 1 samples. Lower case letters in site descriptions denote replicates.

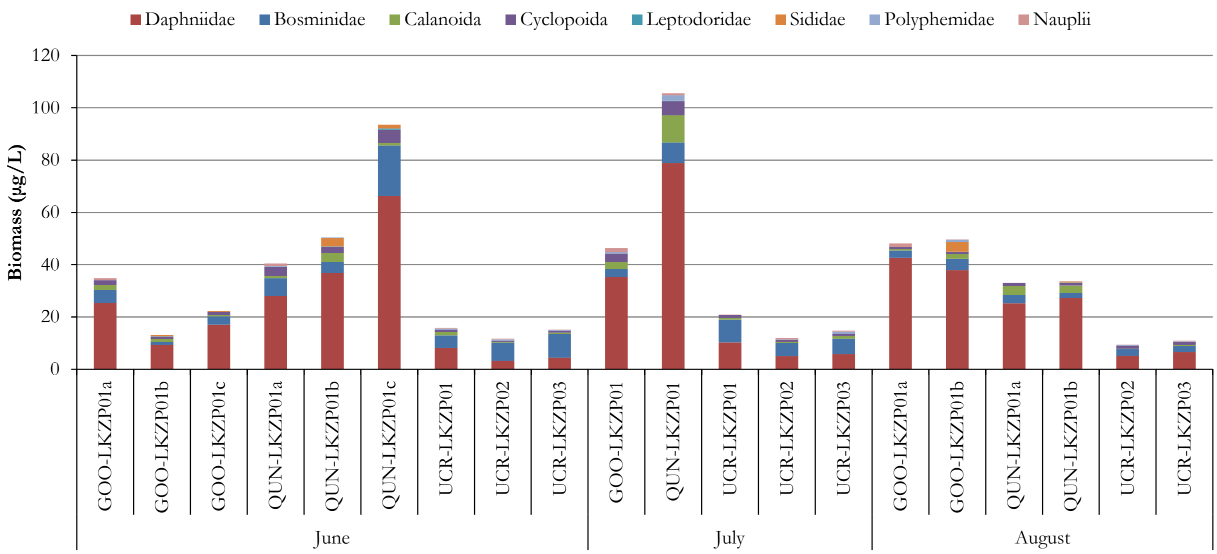


Figure 21. Zooplankton biomass for Year 2 samples.

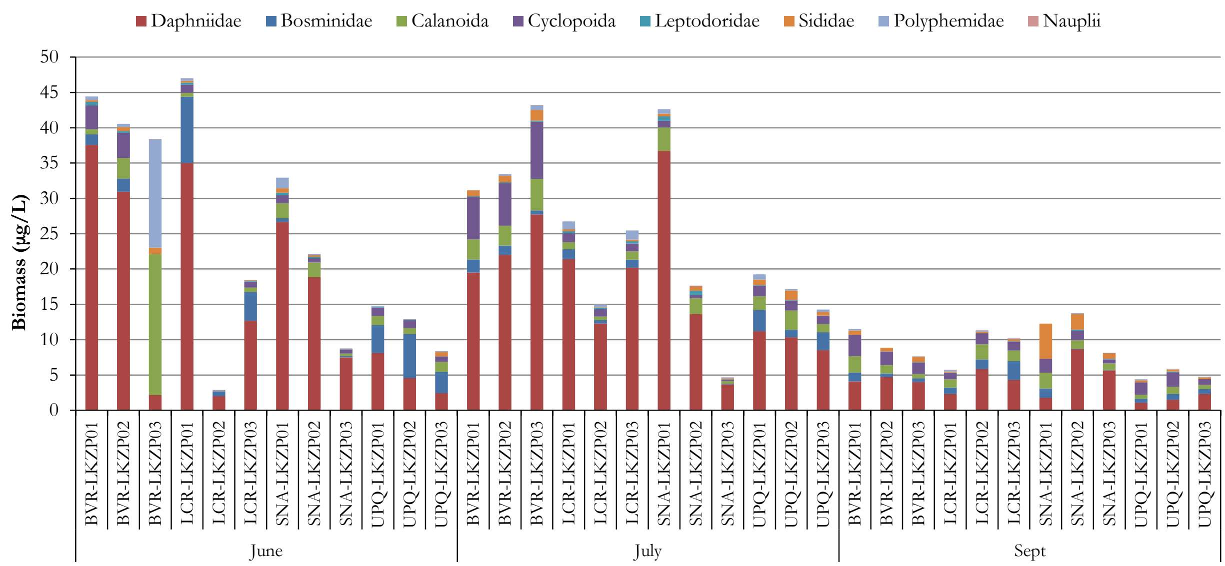
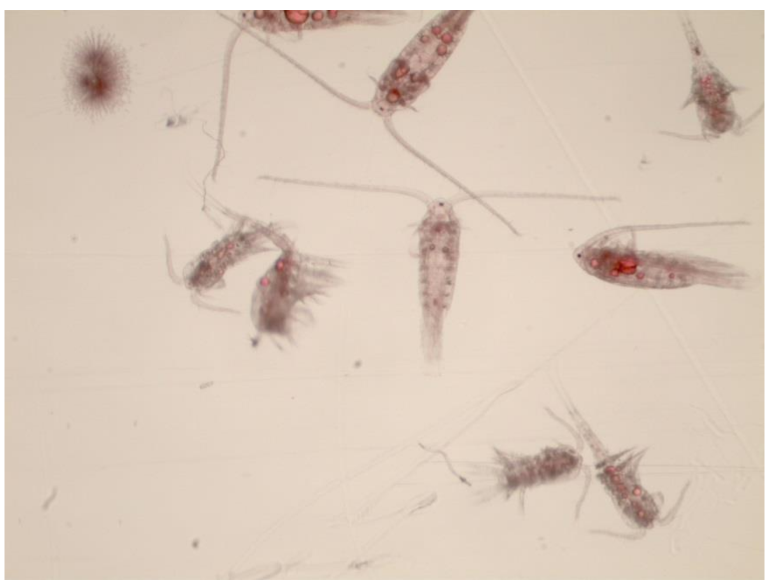


Figure 22. Copepods in a zooplankton sample collected in September 2015. Note stored lipids (red vesicles), which were qualitatively observed to be more abundant in individuals sampled in September relative to earlier in the growing season.



3.1.2. Littoral Invertebrates

Littoral invertebrates sampled at all lakes in July 2015 are summarized in Table 18. When all samples are combined, the most abundant taxa were: Hemiptera (33%), Diptera (14%), Amphipoda (13%), Coleoptera (10%) and Odonata (7%). Sample composition varied substantially between lakes.

3.1.3. Stream Invertebrates

Stream invertebrates sampled at all lakes in July 2015 are summarized in Table 19. When all samples are combined, the most abundant taxa were: Coleoptera (39%), Diptera (15%), Ephemeroptera (13%), Plecoptera (9%), Trichoptera (8%) and Hemiptera (7%). Sample composition varied substantially between lakes.

3.1.4. Terrestrial Invertebrates

Terrestrial invertebrates sampled at all lakes during each of the three sampling months are summarized in Table 20. At each lake, there was considerable variability in sample composition between months. However, when all months are combined, Diptera was the most abundant order at all lakes (79–86% of individuals), except for Snakehead Lake where Hymenoptera was most abundant overall (51% of individuals), notably in June and July. Individuals from another seven orders were collected during Year 2, although the abundance of these orders was generally low. Four orders were represented by only a single individual collected during all sampling events.

Table 18. Littoral invertebrate abundance, 2015.

Waterbody	Site	Rep.	Abundance (individuals/sample)																	
			Amphipoda	Araneae	Bivalvia	Coleoptera	Diptera	Ephemeroptera	Gastropoda	Hemiptera	Hydracarina	Hymenoptera	Odonata	Oligochaeta	Planorbidae	Plecoptera	Pulmonata	Trichoptera	Tricladida	Trombidiformes
Lower Campbell Reservoir	LCR-BIV01	1	15	0	0	0	9	0	0	3	0	0	0	1	0	0	0	0	0	1
		2	9	0	0	0	2	0	0	2	0	0	1	2	0	0	0	0	0	0
	Abundance (all replicates; %)		53%	0%	0%	0%	24%	0%	0%	11%	0%	0%	2%	7%	0%	0%	0%	0%	0%	2%
Beavertail Lake	BVR-BIV01	1	3	0	0	8	6	1	5	9	0	0	0	0	0	1	0	4	0	1
		2	0	1	1	7	0	0	3	7	2	0	4	0	0	0	0	0	0	0
		3	0	0	2	5	1	2	5	18	0	0	2	0	0	1	0	0	0	0
	Abundance (all replicates; %)		3%	1%	3%	20%	7%	3%	13%	34%	2%	0%	6%	0%	0%	2%	0%	4%	0%	1%
Snakehead Lake	SNA-BIV01	1	2	0	0	2	1	0	0	0	0	0	6	0	0	0	0	0	0	0
		2	0	0	0	0	1	2	0	0	0	1	3	0	3	0	0	0	0	0
		3	6	0	1	0	10	4	0	1	0	0	1	0	0	0	0	0	0	0
	Abundance (all replicates; %)		18%	0%	2%	5%	27%	14%	0%	2%	0%	2%	23%	0%	7%	0%	0%	0%	0%	0%
Upper Quinsam Lake	UPQ-BIV01	1	0	0	0	2	0	0	0	18	0	0	1	0	0	0	9	0	1	0
		2	0	0	0	1	4	0	0	9	0	0	1	0	0	0	0	0	0	0
		3	0	0	0	0	3	0	0	20	0	0	0	0	0	0	3	0	0	0
	Abundance (all replicates; %)		0%	0%	0%	4%	10%	0%	0%	65%	0%	0%	3%	0%	0%	0%	17%	0%	1%	0%
Total abundance (all samples; %)			13%	0%	2%	10%	14%	3%	5%	33%	1%	0%	7%	1%	1%	1%	5%	2%	0%	1%

Table 19. Stream invertebrate abundance, 2015.

Waterbody	Date	Site	Replicate	Abundance (individuals/sample)														
				Araneae	Bivalvia[1]	Coleoptera	Diptera	Ephemeroptera	Gastropoda	Haplotaxida	Hemiptera	Hydracarina	Megaloptera	Odonata	Oligochaeta[1]	Physidae	Plecoptera	Trichoptera
Lower Campbell Reservoir	Jul 20, 2015	LCR-SIV01	1	0	0	0	5	0	0	0	0	0	0	5	0	0	0	2
			2	0	0	1	1	12	0	0	1	0	0	0	0	0	4	0
			3	0	0	1	0	0	0	0	0	0	0	1	0	0	4	2
	Abundance (all replicates; %)			0%	0%	5%	15%	31%	0%	0%	3%	0%	0%	15%	0%	0%	21%	10%
Beavertail Lake	Jul 23, 2015	BVR-SIV01	1	0	1	0	2	5	4	0	3	0	0	0	1	0	0	1
			2	0	1	1	12	7	0	3	0	1	0	2	0	0	13	14
			3	0	0	0	9	16	0	0	1	0	1	3	0	0	8	8
	Abundance (all replicates; %)			0%	2%	1%	20%	24%	3%	3%	3%	1%	1%	4%	1%	0%	18%	20%
Snakehead Lake	Jul 21, 2015	SNA-SIV01	1	0	0	45	1	0	0	0	0	0	0	0	0	0	0	0
			2	0	0	36	4	0	1	0	0	0	0	0	0	0	1	0
			3	0	0	38	0	0	0	0	0	0	0	0	0	0	0	0
	Abundance (all replicates; %)			0%	0%	94%	4%	0%	1%	0%	0%	0%	0%	0%	0%	0%	1%	0%
Upper Quinsam Lake	Jul 22, 2015	UPQ-SIV01	1	1	0	0	0	2	0	0	12	0	0	0	0	2	0	0
			2	0	0	6	14	0	0	0	2	0	0	1	0	0	0	0
			3	1	0	0	2	0	0	0	3	0	0	0	0	0	0	0
	Abundance (all replicates; %)			4%	0%	13%	35%	4%	0%	0%	37%	0%	0%	2%	0%	4%	0%	0%
Total abundance (all samples; %)				**1%**	**1%**	**39%**	**15%**	**13%**	**2%**	**1%**	**7%**	**<1%**	**<1%**	**4%**	**<1%**	**1%**	**9%**	**8%**

1. Only enumerated to class.

Table 20. Terrestrial invertebrate abundance, 2015.

Waterbody	Site	Month	Abundance (individuals/sample)								
			Arachnida	Coleoptera	Diptera	Hemiptera	Hymenoptera	Neuroptera	Odonata	Orthoptera	Trichoptera
Lower Campbell Reservoir	LCR-TIV01	June	0	3	13	1	0	0	0	0	0
		July	0	0	8	0	1	0	1	0	0
		Sept	0	0	23	0	1	0	0	0	0
	Abundance (all months; %)		0%	6%	86%	2%	4%	0%	2%	0%	0%
Beavertail Lake	BVR-TIV01	June	0	0	20	0	1	0	0	0	0
		July	0	0	8	2	2	1	0	0	0
		Sept	0	0	6	0	3	0	0	0	0
	Abundance (all months; %)		0%	0%	79%	5%	14%	2%	0%	0%	0%
Snakehead Lake	SNA-TIV01	June	0	0	11	0	14	0	0	0	0
		July	0	0	6	0	9	0	0	0	0
		Sept	5	0	0	0	1	0	0	0	1
	Abundance (all months; %)		11%	0%	36%	0%	51%	0%	0%	0%	2%
Upper Quinsam Lake	UPQ-TIV01	June	1	0	5	0	0	0	0	0	0
		July	0	0	5	0	1	0	0	1	0
		Sept	0	0	15	0	1	0	0	0	0
	Abundance (all months; %)		3%	0%	86%	0%	7%	0%	0%	3%	0%
Total abundance (all samples; %)			**4%**	**2%**	**71%**	**2%**	**20%**	**1%**	**1%**	**1%**	**1%**

3.2. Fish Catch

3.2.1. Gill Netting

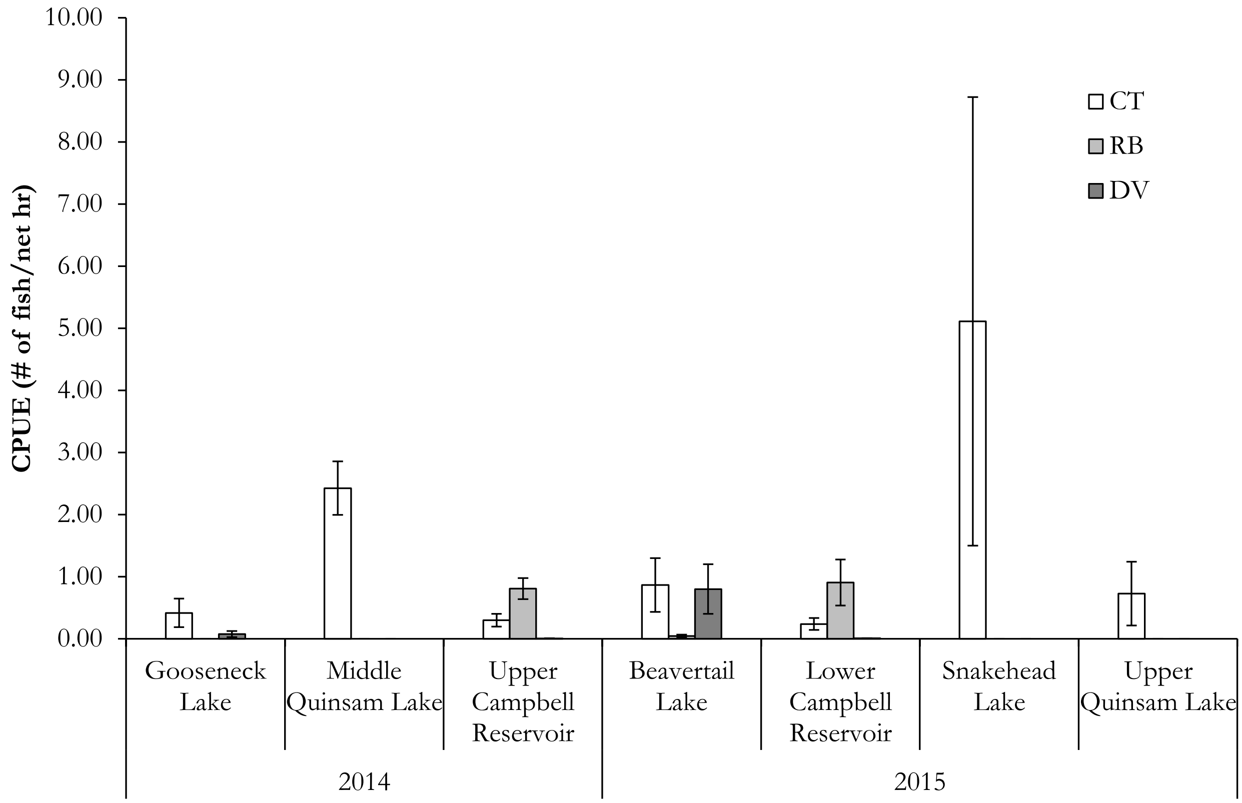
Gill netting catch (# of fish) and catch-per-unit-effort (CPUE) by species and lake from Year 2 is shown in Table 21. Average CPUE by species was compared across all lakes, including the lakes sampled in Year 1 (Figure 23). Average CPUE for Cutthroat Trout was highest at Snakehead Lake, followed by Middle Quinsam Lake, and lowest in both Upper and Lower Campbell Reservoirs (Figure 23). CPUE for Rainbow Trout was highest in Upper and Lower Campbell Reservoirs (Figure 23). Rainbow Trout were present in Beavertail Lake and may be absent in the remaining lakes.

Table 21. Gill netting capture results from the Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

Waterbody	Site	Sampling Date	No. of Sets	Gill Netting Effort (hrs)	Gill Net Catch (# of fish)[1]					Gill Net CPUE (# of fish/net hr)[1]				
					CT	RB	DV	CC	TSS	CT	RB	DV	CC	TSB
Lower Campbell Reservoir	LCR-LKGN01	23-Aug-15	1	23.9	11	2	0	1	0	0.46	0.08	0.00	0.04	0.00
	LCR-LKGN02	23-Aug-15	1	22.8	0	27	0	0	0	0.00	1.19	0.00	0.00	0.00
	LCR-LKGN03	04-Oct-15	1	23.7	13	52	0	7	0	0.55	2.20	0.00	0.30	0.00
	LCR-LKGN04	04-Oct-15	1	25.2	0	28	1	0	0	0.00	1.11	0.04	0.00	0.00
	LCR-LKGN05	04-Oct-15	1	24.6	0	8	0	0	0	0.00	0.32	0.00	0.00	0.00
	LCR-LKGN06	04-Oct-15	1	24.5	10	13	0	1	0	0.41	0.53	0.00	0.04	0.00
		Total	**6**	**144.7**	**34**	**130**	**1**	**9**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**24.1**	**6**	**22**	**0**	**2**	**0**	**0.24**	**0.91**	**0.01**	**0.06**	**0.00**
		SD	**n/a**	**0.9**	**6**	**18**	**0**	**3**	**0**	**0.26**	**0.77**	**0.02**	**0.12**	**0.00**
Beavertail Lake	BVR-LKGN01	17-Aug-15	1	22.5	35	1	16	0	0	1.55	0.04	0.71	0.00	0.00
	BVR-LKGN02	17-Aug-15	1	23.5	15	3	0	11	0	0.64	0.13	0.00	0.47	0.00
	BVR-LKGN03	17-Aug-15	1	23.6	13	0	27	0	0	0.55	0.00	1.15	0.00	0.00
	BVR-LKGN04	17-Aug-15	1	23.9	17	0	32	0	0	0.71	0.00	1.34	0.00	0.00
		Total	**4**	**93.4**	**80**	**4**	**75**	**11**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**23.4**	**20**	**1**	**19**	**3**	**0**	**0.86**	**0.04**	**0.80**	**0.12**	**0.00**
		SD	**n/a**	**0.6**	**10**	**1**	**14**	**6**	**0**	**0.46**	**0.06**	**0.59**	**0.23**	**0.00**
Snakehead Lake	SNA-LKGN01	21-Aug-15	1	20.6	142	0	0	0	0	6.88	0.00	0.00	0.00	0.00
	SNA-LKGN02	21-Aug-15	1	19.8	66	0	0	0	0	3.34	0.00	0.00	0.00	0.00
		Total	**2**	**40.4**	**208**	**0**	**0**	**0**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**20.2**	**104**	**0**	**0**	**0**	**0**	**5.11**	**0.00**	**0.00**	**0.00**	**0.00**
		SD	**n/a**	**0.6**	**54**	**0**	**0**	**0**	**0**	**2.51**	**0.00**	**0.00**	**0.00**	**0.00**
Upper Quinsam Lake	UPQ-LKGN01	19-Aug-15	1	24.5	14	0	0	0	0	0.57	0.00	0.00	0.00	0.00
	UPQ-LKGN02	20-Aug-15	1	23.9	21	0	0	0	0	0.88	0.00	0.00	0.00	0.00
		Total	**2**	**48.4**	**35**	**0**	**0**	**0**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**24.2**	**18**	**0**	**0**	**0**	**0**	**0.73**	**0.00**	**0.00**	**0.00**	**0.00**
		SD	**n/a**	**0.4**	**5**	**0**	**0**	**0**	**0**	**0.22**	**0.00**	**0.00**	**0.00**	**0.00**

[1] CT- Cutthroat Trout, RB - Rainbow Trout, DV - Dolly Varden, CC - Sculpin general, TSB - Threespine Stickleback.

Figure 23. Catch-per-unit-effort (CPUE) during gill netting of Cutthroat Trout (CT), Rainbow Trout (RT) and Dolly Varden (DV) from all lakes sampled in Year 1 and Year 2 of the JHTMON-5 Program.



3.2.2. Trap Netting

Threespine sticklebacks were the primary fish species targeted by trap net fishing in Beavertail, Gooseneck, Lower Campbell, Upper Campbell, Snakehead, Upper Quinsam, and Middle Quinsam lakes. In the Lower Campbell Reservoir a total of 21 Threespine Stickleback were captured and in Upper Campbell Reservoir a total of 32 Stickleback were captured. No Threespine Stickleback were captured during trap netting efforts in Beavertail, Gooseneck, Snakehead, Middle Quinsam and Upper Quinsam lakes, suggesting that Threespine Stickleback may be absent from these lakes (Table 22). CPUE for Threespine Stickleback was 0.44 fish/trap hour (±0.43 SD) in Lower Campbell Reservoir and 0.19 fish/trap hour (±0.27 SD) in Upper Campbell Reservoir (Table 22, Figure 24).

Cutthroat Trout and sculpin spp. were caught using trap netting from most lakes in 2015 (Table 22, Figure 24). In contrast, Rainbow Trout and Dolly Varden were only captured in Upper Campbell Reservoir using this method.

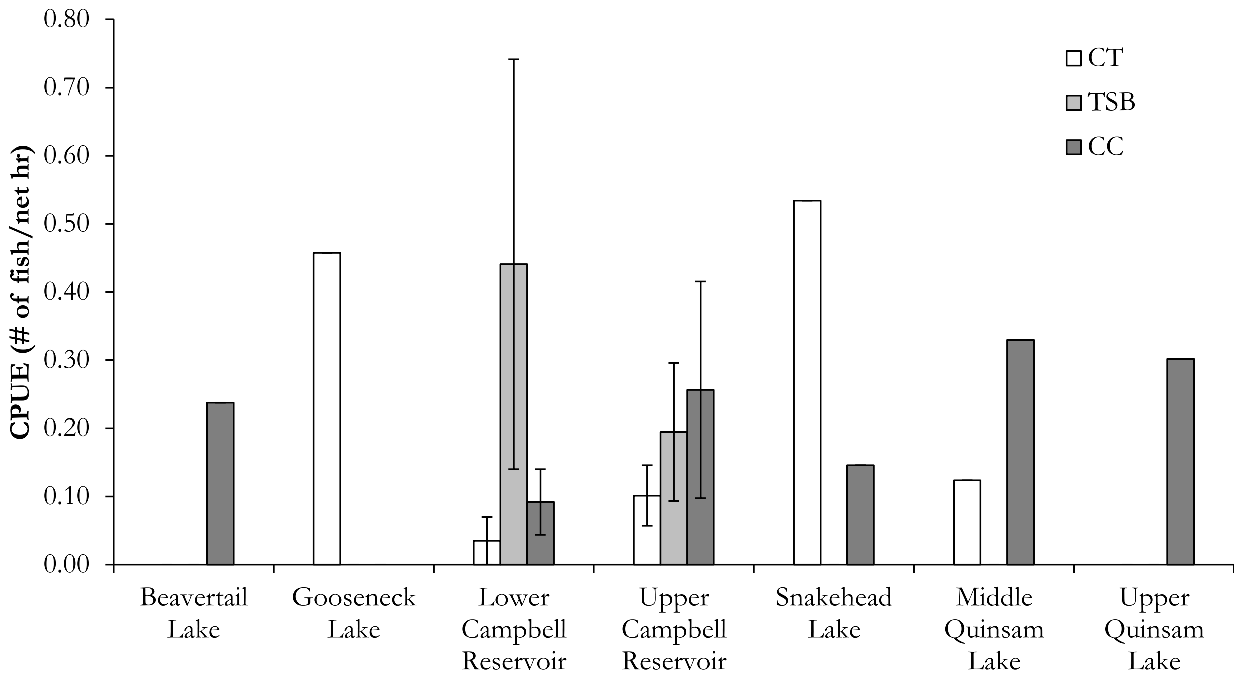
Table 22. Trap netting capture results from Upper Campbell and Lower Campbell reservoirs, and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes, 2015.

Waterbody	Site	Sampling Date	No. of Sets	Gill Netting Effort (hrs)	Gill Net Catch (# of fish)[1]					Gill Net CPUE (# of fish/net hr)[1]				
					CT	RB	DV	CC	TSB	CT	RB	DV	CC	TSB
Beavertail Lake	BVR-LKTN01	07-Oct-15	1	21.0	0	0	0	5	0	0.00	0.00	0.00	0.24	0.00
		Total	**1**	**21.0**	**0**	**0**	**0**	**5**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**21.0**	**0.0**	**0.0**	**0.0**	**5.0**	**0.0**	**0.00**	**0.00**	**0.00**	**0.24**	**0.00**
		SD[2]	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
Gooseneck Lake	GOO-LKTN01	06-Oct-15	1	24.1	11	0	0	0	0	0.46	0.00	0.00	0.00	0.00
		Total	**1**	**24.1**	**11**	**0**	**0**	**0**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**24.1**	**11.0**	**0.0**	**0.0**	**0.0**	**0.0**	**0.46**	**0.00**	**0.00**	**0.00**	**0.00**
		SD	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
Lower Campbell Lake Reservoir	LCR-LKTN01	04-Oct-15	1	28.6	2	0	0	4	4	0.07	0.00	0.00	0.14	0.14
	LCR-LKTN02	04-Oct-15	1	22.9	0	0	0	1	17	0.00	0.00	0.00	0.04	0.74
		Total	**2**	**51.5**	**2**	**0**	**0**	**5**	**21**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**25.7**	**1.0**	**0.0**	**0.0**	**2.5**	**10.5**	**0.04**	**0.00**	**0.00**	**0.09**	**0.44**
		SD	**n/a**	**4.0**	**1.4**	**0.0**	**0.0**	**2.1**	**9.2**	**0.05**	**0.00**	**0.00**	**0.07**	**0.43**
Upper Campbell Reservoir	UCR-LKTN01	31-Aug-15	1	25.9	0	2	6	31	20	0.00	0.08	0.23	1.20	0.77
	UCR-LKTN01	01-Sep-15	1	18.5	6	20	2	3	1	0.32	1.08	0.11	0.16	0.05
	UCR-LKTN02	01-Sep-15	1	19.9	2	0	0	0	0	0.10	0.00	0.00	0.00	0.00
	UCR-LKTN03	02-Sep-15	1	19.4	0	0	0	3	4	0.00	0.00	0.00	0.15	0.21
	UCR-LKTN04	02-Sep-15	1	19.6	1	0	0	0	0	0.05	0.00	0.00	0.00	0.00
	UCR-LKTN05	03-Sep-15	1	21.6	4	0	0	3	3	0.19	0.00	0.00	0.14	0.14
	UCR-LKTN06	03-Sep-15	1	20.9	1	0	0	3	4	0.05	0.00	0.00	0.14	0.19
		Total	**7**	**145.8**	**14**	**22**	**8**	**43**	**32**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**20.8**	**2.0**	**3.1**	**1.1**	**6.1**	**4.6**	**0.10**	**0.17**	**0.05**	**0.26**	**0.19**
		SD[2]	**n/a**	**2.5**	**2.2**	**7.5**	**2.3**	**11.1**	**7.0**	**0.12**	**0.41**	**0.09**	**0.42**	**0.27**
Snakehead Lake	SNA-LKTN01	07-Oct-15	**1**	20.6	11	0	0	3	0	0.53	0.00	0.00	0.15	0.00
		Total	**1**	**20.6**	**11**	**0**	**0**	**3**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**20.6**	**11.0**	**0.0**	**0.0**	**3.0**	**0.0**	**0.53**	**0.00**	**0.00**	**0.15**	**0.00**
		SD[2]	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
Middle Quinsam Lake	QUN-LKTN01	06-Oct-15	**1**	24.3	3	0	0	8	0	0.12	0.00	0.00	0.33	0.00
		Total	**1**	**24.3**	**3**	**0**	**0**	**8**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**24.3**	**3.0**	**0.0**	**0.0**	**8.0**	**0.0**	**0.12**	**0.00**	**0.00**	**0.33**	**0.00**
		SD[2]	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
Upper Quinsam Lake	UPQ-LKTN01	03-Oct-15	**1**	26.5	0	0	0	8	0	0.00	0.00	0.00	0.30	0.00
		Total	**1**	**26.5**	**0**	**0**	**0**	**8**	**0**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**
		Average	**1**	**26.5**	**0.0**	**0.0**	**0.0**	**8.0**	**0.0**	**0.00**	**0.00**	**0.00**	**0.30**	**0.00**
		SD[2]	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**

[1] CT- Cutthroat Trout, RB - Rainbow Trout, DV - Dolly Varden, CC - Sculpin general, TSB - Threespine Stickleback.

[2] No standard deviation is calculated if only one sample occurs.

Figure 24. **Catch-per-unit-effort (CPUE) of Cutthroat Trout (CT), Threespine Stickleback (TSB) and Sculpin spp. (CC) during trap netting from Upper Campbell and Lower Campbell reservoirs, and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes, 2015.**



3.2.3. Minnow Trapping

No Threespine Stickleback individuals were caught during the June trip. The remainder of this section focuses on minnow trapping undertaken during the main Year 2 fish sampling program undertaken in late summer. Data collected during this trip provide results that can be compared with lakes sampled during this period in other years of JHTMON-5.

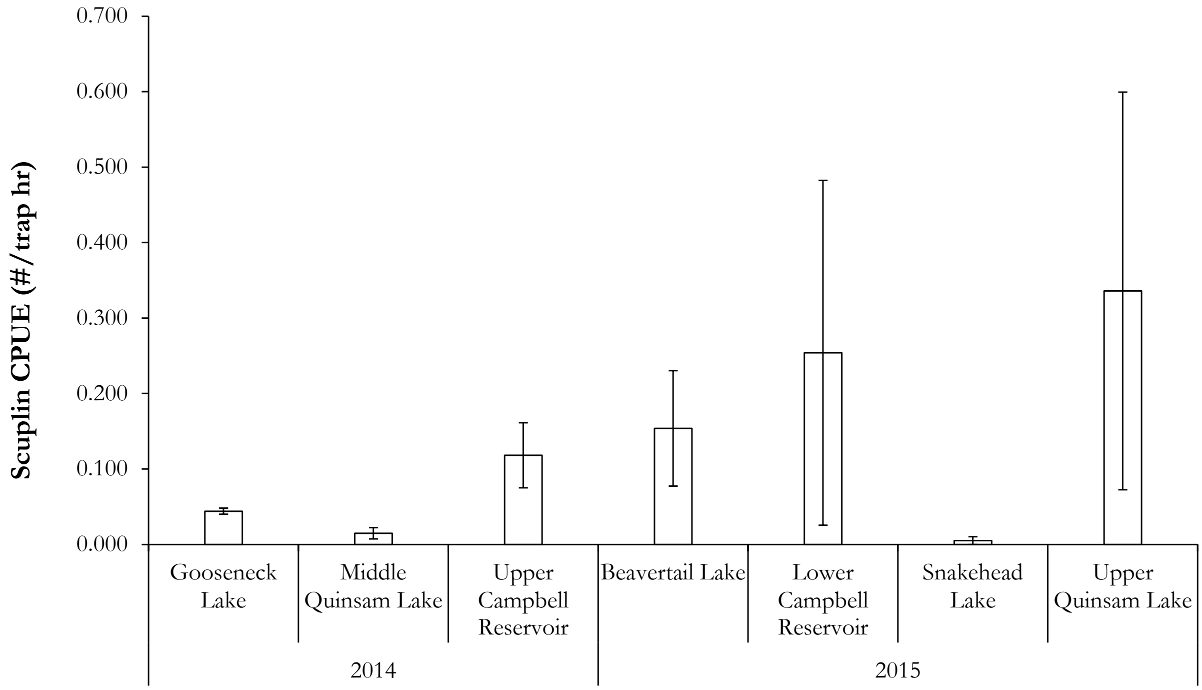
Sculpin spp. and Threespine Stickleback (*n*=1) were the only two species captured using minnow traps in Year 2 sampling (Table 23). Sculpin spp. were caught across all lakes, including all lakes sampled in Year 1 using this method (Figure 25). Sculpin spp. CPUE was highest in the largest lakes and reservoirs (Upper Quinsam, Lower Campbell, Beavertail and Upper Campbell) and was the lowest in the smallest lake (Snakehead Lake). Not all sculpin that were captured were identified to species. Some individual sculpin were identified as Prickly Sculpin (*Cottus asper*) or Coastrange Sculpin (*Cottus aleuticus*); therefore all captured sculpin have been categorized as sculpin spp. (CC). A single Threespine Stickleback was captured in the Lower Campbell Lake using this method (Table 23). No other Threespine Stickleback were caught.

Table 23. Minnow trapping capture results from Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

Waterbody	Site	Sampling Date	No. of Minnow	Minnow Trapping	Minnow Trapping Catch		Minnow Trapping CPUE	
					CC	TSB	CC	TSB
Lower Campbell Reservoir	LCR-LKMT04	23-Aug-15	5	118.1	57	1	0.48	0.01
	LCR-LKMT05	23-Aug-15	5	118.0	3	0	0.03	0.00
		Total	**10**	**236.1**	**60**	**1**	**0.51**	**0.01**
		Average	**5**	**118.1**	**30**	**1**	**0.25**	**0.004**
		SD	**n/a**	**0.1**	**38.2**	**0.7**	**0.32**	**0.006**
Beavertail Lake	BVR-LKMT05	17-Aug-15	5	117.2	27	0	0.23	0.00
	BVR-LKMT06	17-Aug-15	5	116.2	9	0	0.08	0.00
		Total	**10**	**233.4**	**36**	**0**	**0.31**	**0.00**
		Average	**5**	**116.7**	**18**	**0**	**0.15**	**0.00**
		SD	**n/a**	**0.7**	**12.7**	**0.0**	**0.11**	**0.00**
Snakehead Lake	SNA-LKMT04	21-Aug-15	5	101.5	0	0	0.00	0.00
	SNA-LKMT05	21-Aug-15	5	98.8	1	0	0.01	0.00
		Total	**10**	**200.3**	**1**	**0**	**0.01**	**0.00**
		Average	**5**	**100.1**	**1**	**0**	**0.01**	**0.00**
		SD	**n/a**	**1.9**	**0.7**	**0.0**	**0.01**	**0.00**
Upper Quinsam Lake	UPQ-LKMT01	19-Aug-15	5	125.1	75	0	0.60	0.00
	UPQ-LKMT02	19-Aug-15	5	124.1	9	0	0.07	0.00
		Total	**10**	**249.2**	**84**	**0**	**0.67**	**0.00**
		Average	**5**	**124.6**	**42**	**0**	**0.34**	**0.00**
		SD	**n/a**	**0.7**	**46.7**	**0.0**	**0.37**	**0.00**

[1] CC - Sculpin general and TSB - Threespine Stickleback.

Figure 25. Catch-per-unit-effort (CPUE) of Sculpin spp. during minnow trapping from all lakes sampled in Year 1 and Year 2 of the JHTMON-5 Program.



3.2.4. Individual Fish Analysis

Cutthroat Trout and Rainbow Trout captured in gill nets varied substantially in size. Length frequency histograms of Cutthroat Trout and Rainbow Trout captured in Beavertail, Lower Campbell, Snakehead, and Upper Quinsam lakes are presented in Figure 26, Figure 27, Figure 28, and Figure 29. In Beavertail Lake, the fork length of Cutthroat Trout ranged from 149 mm to 308 mm, and Rainbow Trout ranged from 206 mm to 235 mm. In Lower Campbell Reservoir, the fork length of Cutthroat Trout ranged from 195 mm to 418 mm, and Rainbow Trout ranged from 111 mm to 336 mm. The fork length of Cutthroat Trout captured in Snakehead and Upper Quinsam lakes ranged from 115 mm to 314 mm and 203 mm to 364 mm, respectively.

Cutthroat Trout and Rainbow Trout ranged from two to six and two to five years, respectively, in age across all lakes sampled in Year 2. Fish age was positively related to fish length in all lakes sampled (Figure 30, Figure 31, Figure 32, and Figure 33). In Lower Campbell Reservoir, our subsample of Cutthroat Trout ranged in age from three to six years, and Rainbow Trout ranged in age from two to five years (Figure 30). In Beavertail Lake, Cutthroat Trout ranged in age from 3 to 5 years, and all assessed Rainbow Trout were 4 years of age (Figure 31). Cutthroat Trout sampled from Snakehead Lake ranged in age from three to six years (Figure 32), while Cutthroat Trout from Upper Quinsam Lake ranged in age from two to six years (Figure 33).

Length at age distributions for Cutthroat Trout and Rainbow Trout were used to define discrete fork length ranges for each age class by lake (Table 24). These discrete fork length ranges allow all captured and measured fish to be assigned an age class based on fork length. For example, in Beavertail Lake, 2+ Cutthroat Trout vary from 149 to 153 mm in length, while 5+ Cutthroat Trout vary from 255 to 308 mm in length. One trend observed with these data is that Cutthroat Trout in Lower Campbell Reservoir attain greater lengths at the same age compared to Cutthroat Trout in the other lakes.

Figure 26. Length-frequency histogram of Cutthroat Trout (CT) and Rainbow Trout (RB) captured in Lower Campbell Reservoir between June and October, 2015.

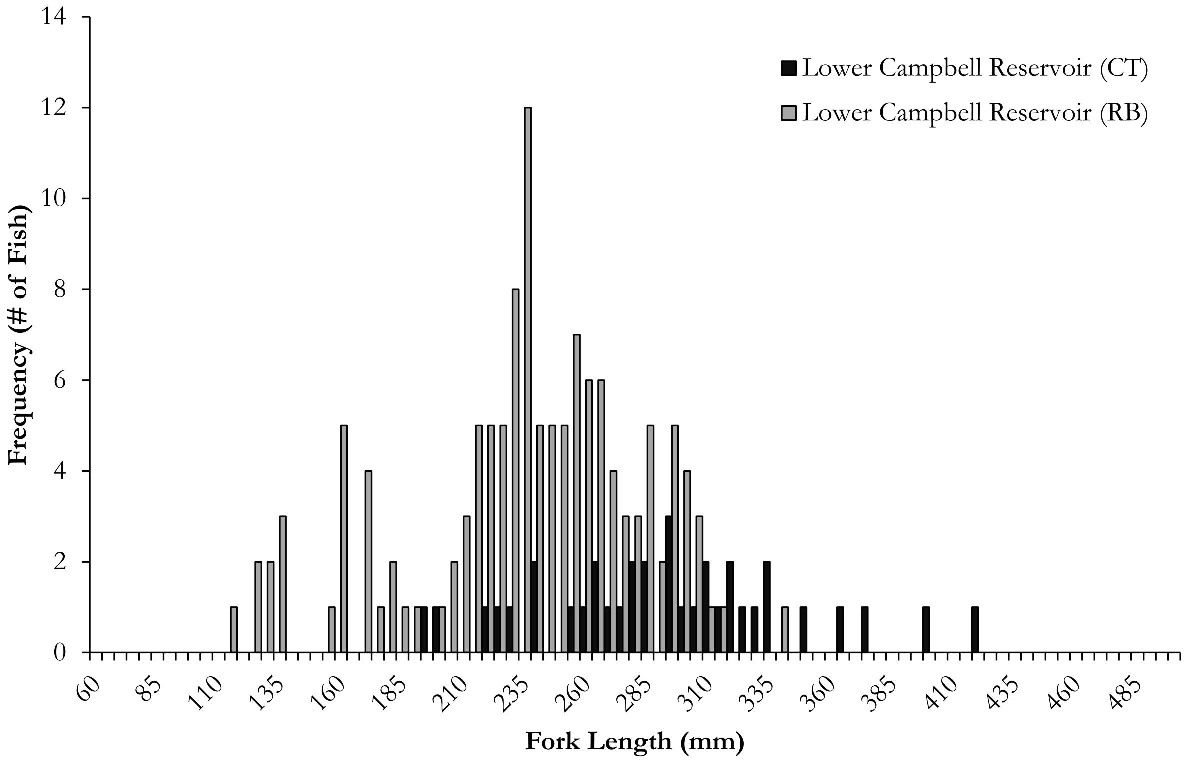


Figure 27. Length-frequency histogram of Cutthroat Trout (CT) and Rainbow Trout (RB) captured in Beavertail Lake between June and October, 2015.

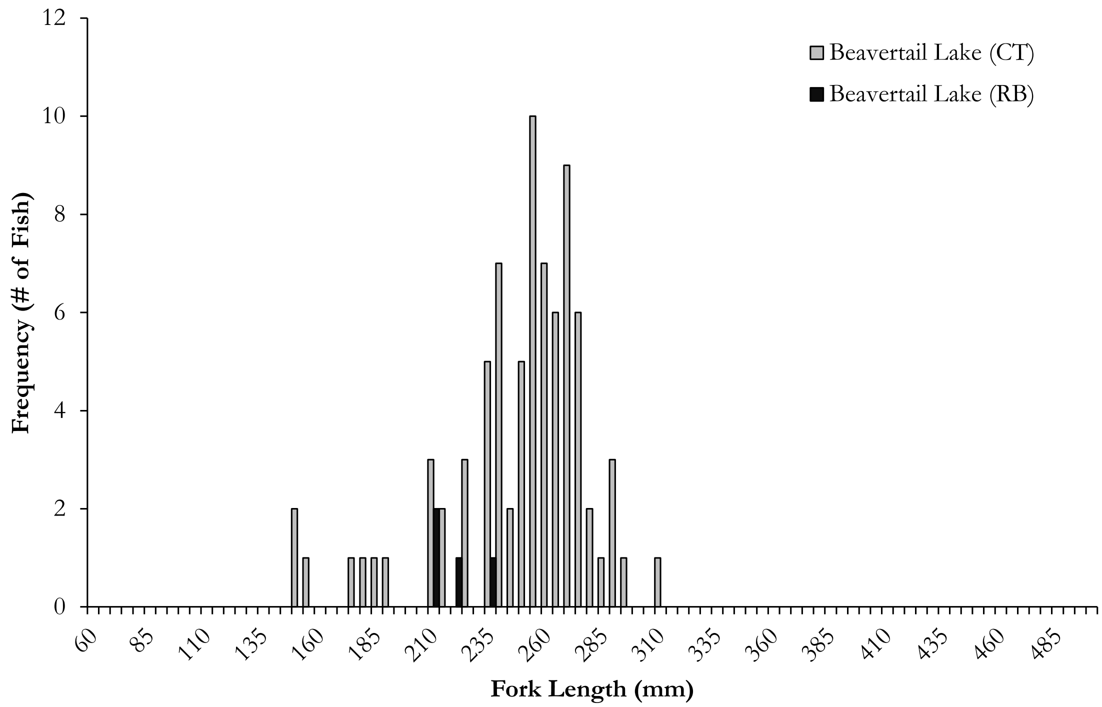


Figure 28. Length-frequency histogram of Cutthroat Trout captured in Snakehead Lake between June and October, 2015.

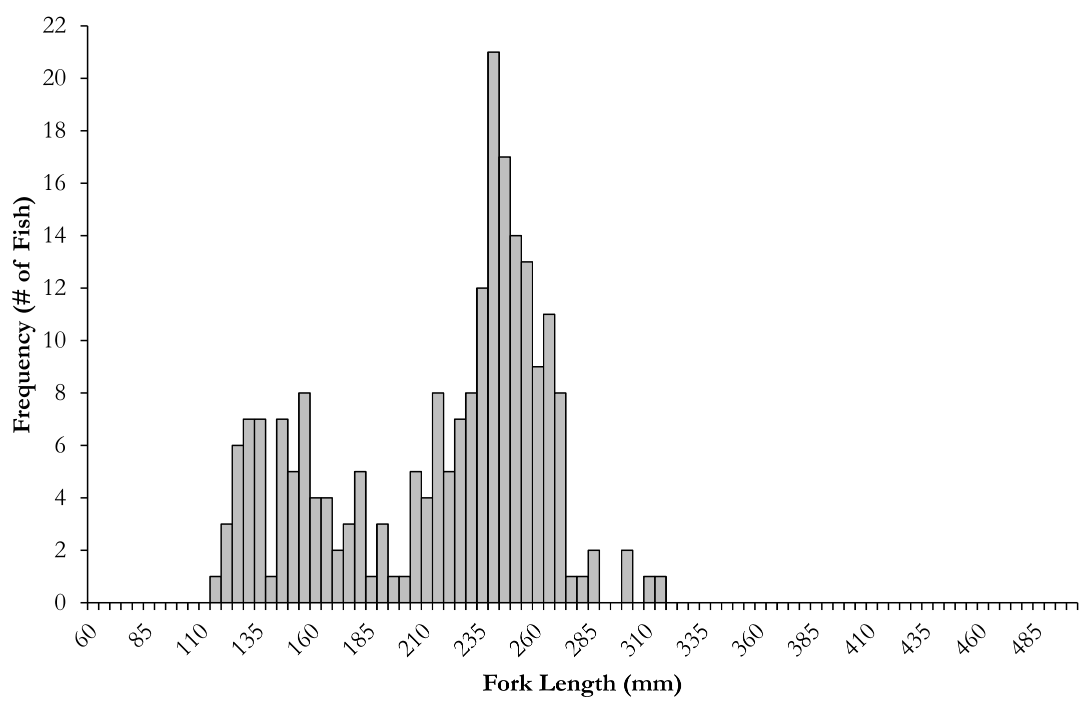


Figure 29. Length-frequency histogram of Cutthroat Trout captured in Upper Quinsam Lake between June and October, 2015.

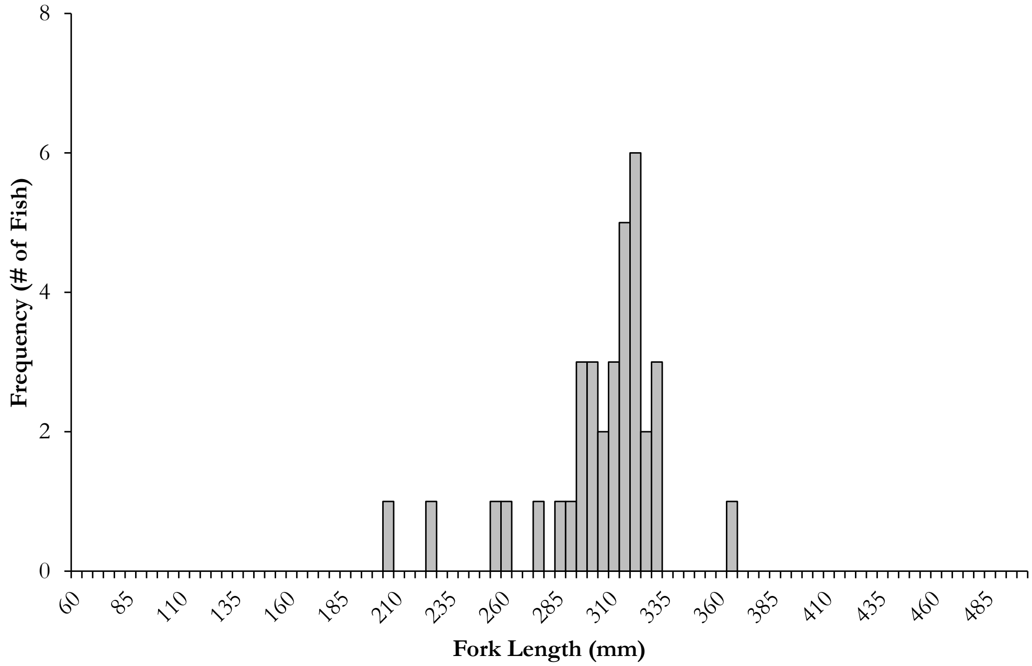


Figure 30. Cutthroat Trout (CT) and Rainbow Trout (RT) length at age in Lower Campbell Reservoir, 2015.

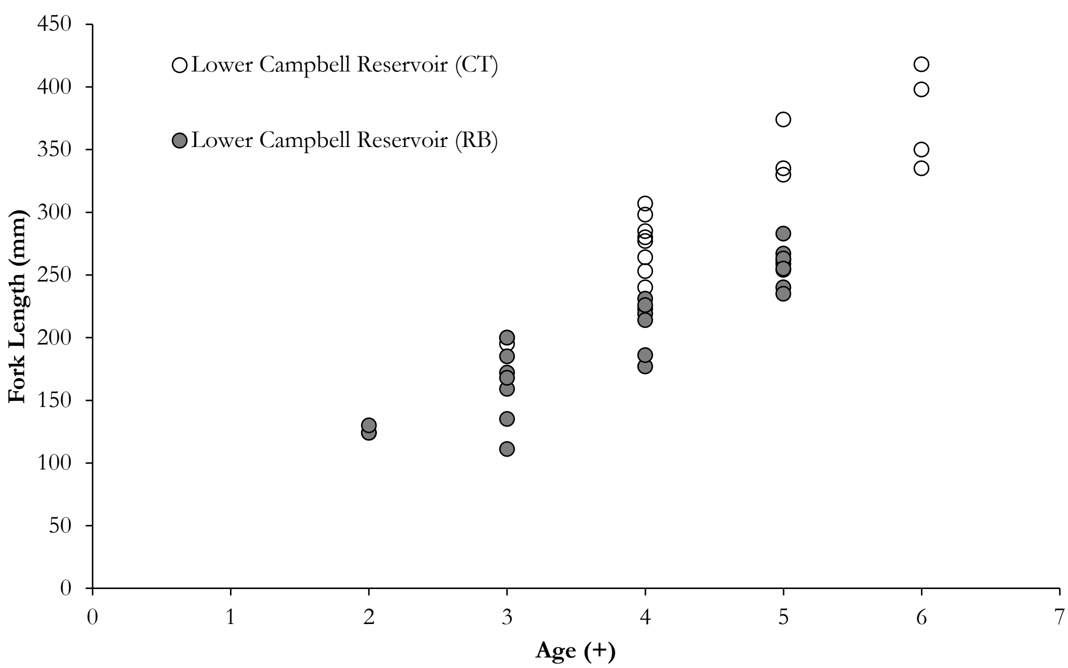


Figure 31. Cutthroat Trout (CT) and Rainbow Trout (RT) length at age in Beavertail Lake, 2015.

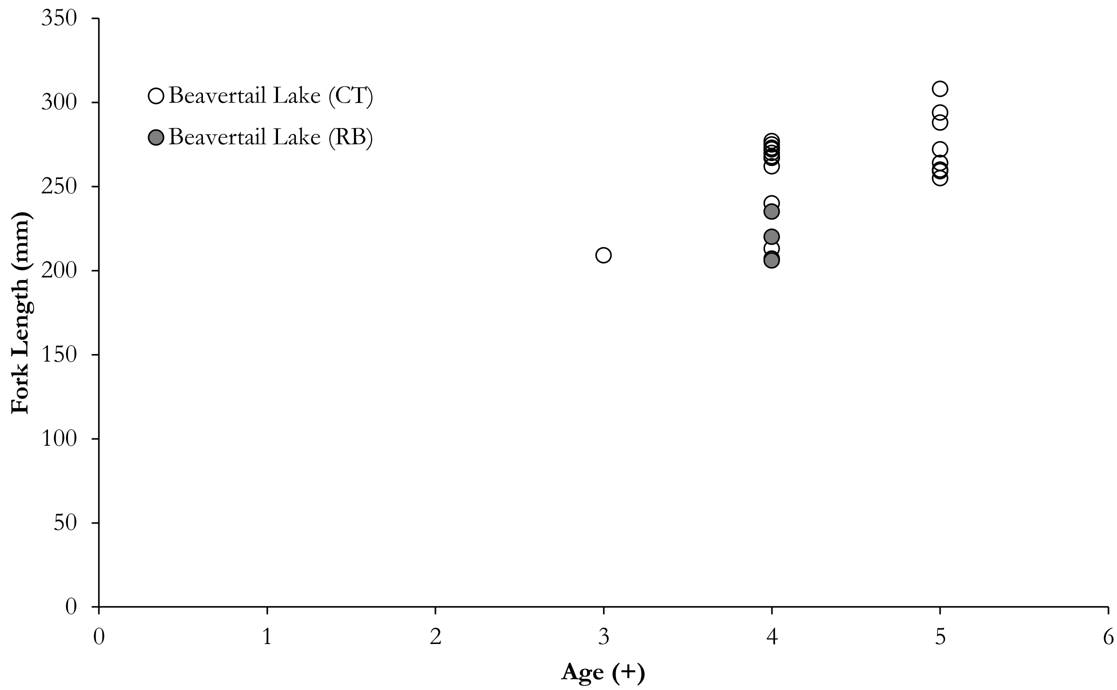


Figure 32. Cutthroat Trout (CT) length at age in Snakehead Lake, 2015.

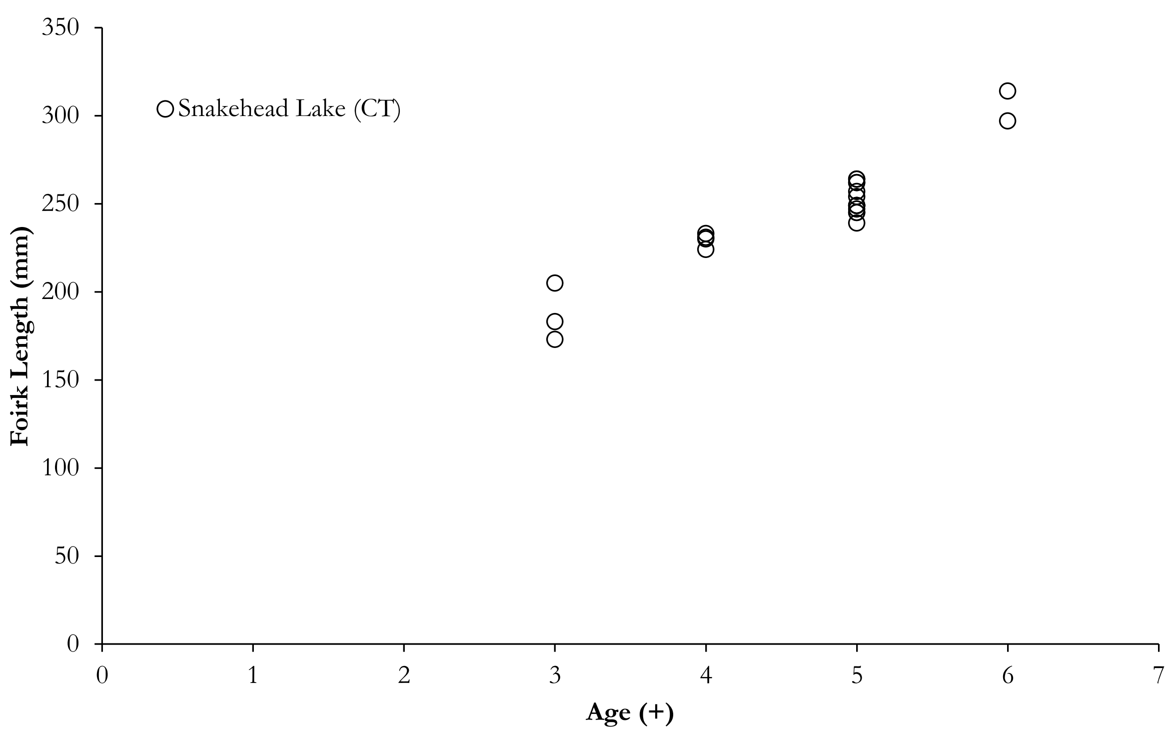


Figure 33. Cutthroat Trout (CT) length at age in Upper Quinsam Lake, 2015.

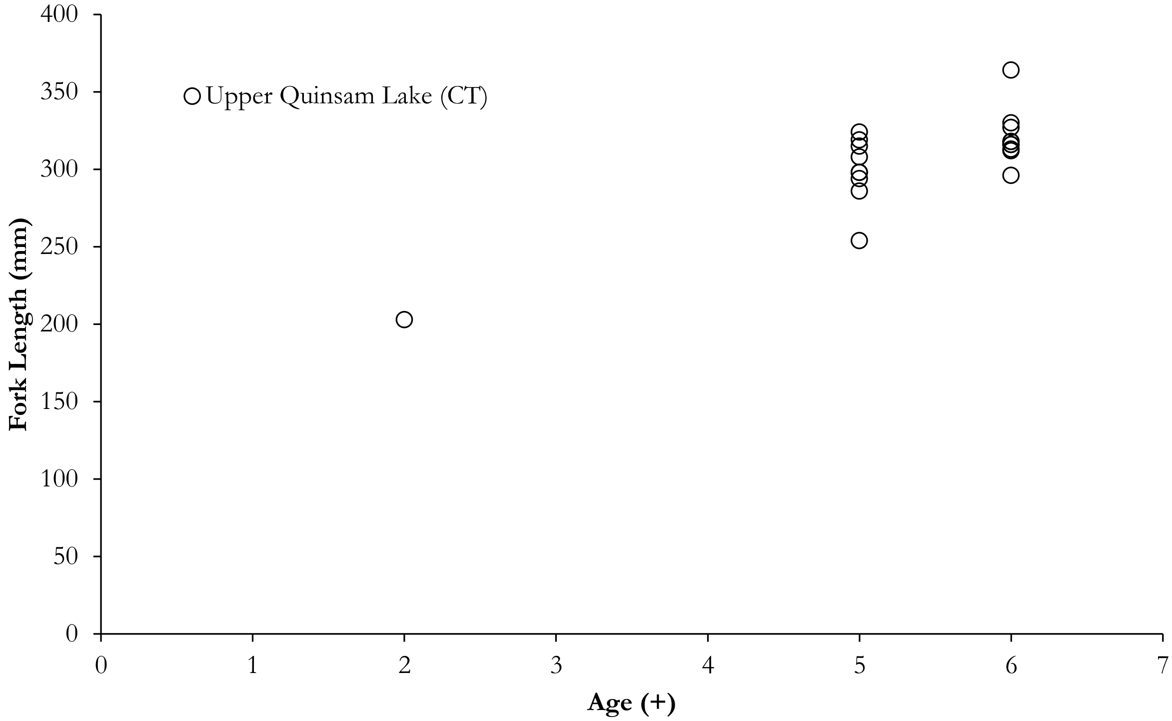


Table 24. Fork length ranges used to define age classes of Cutthroat Trout (CT) and Rainbow Trout (RT) in Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

Age Class	Fork Length Range (mm)					
	Lower Campbell Reservoir (CT)	Lower Campbell Reservoir (RB)	Beavertail Lake (CT)	Beavertail Lake (RB)	Snakehead Lake (CT)	Upper Quinsam Lake (CT)
Fry (0+)	-	-	-	-	-	-
Parr (1+)	-	-	-	-	115 - 136	-
Parr (2+)	-	124 - 135	149 - 153	-	137 - 172	-
Adult (3+)	195 - 252	153 - 185	173 - 212	-	173 - 223	-
Adult (4+)	253 - 309	186 - 239	213 - 254	206 - 235	224 - 238	-
Adult (5+)	310 - 363	240 - 283	255 - 308	-	239 - 296	254 - 295
Adult (6+)	364 - 418	284+	-	-	297 - 314	296 -364
Adult (7+)	-	-	-	-	-	-

3.2.5. Stomach Contents

Stomach content results for Cutthroat Trout and Rainbow Trout sampled from Beavertail, Lower Campbell, Snakehead and Upper Quinsam lakes are shown in Table 26. Cutthroat Trout diet was dominated by littoral invertebrates and by fish, with diet composition varying by lake. The proportion of zooplankton observed in Cutthroat Trout diet was low and did not exceed 14% across all lakes. In contrast, Rainbow Trout diets from Lower Campbell Reservoir and Beavertail Lake were dominated by zooplankton and littoral invertebrates. No fish were observed in Rainbow Trout diets in either lake.

Table 25. Stomach content results by volume for Cutthroat Trout and Rainbow Trout sampled from Upper Campbell Reservoir and Gooseneck and Middle Quinsam lakes, 2014.

Diet item	Upper Campbell Reservoir		Gooseneck Lake	Middle Quinsam Lake
	Cutthroat Trout	Rainbow Trout	Cutthroat Trout	Cutthroat Trout
Littoral invertebrates	1.2%	41.7%	35.6%	87.5%
Zooplankton	0.0%	58.3%	53.3%	4.2%
Fish	98.8%	0.0%	11.1%	8.3%

Table 26. Stomach content results by volume for Cutthroat Trout (CT) and Rainbow Trout (RB) sampled from Lower Campbell Reservoir and Beavertail, Snakehead, and Upper Quinsam lakes, 2015.

Diet Item	Lower Campbell Reservoir		Beavertail Lake		Snakehead Lake	Upper Quinsam Lake
	CT	RB	CT	RB	CT	CT
Littoral Invertebrates	42.9%	41.2%	53.5%	77.5%	90.9%	68.5%
Zooplankton	0.0%	58.8%	14.1%	22.5%	4.5%	7.4%
Fish	57.1%	0.0%	32.4%	0.0%	4.5%	24.1%

3.3. Stable Isotope Data

3.3.1. Summary of Stable Isotope Signatures by Taxa

Nitrogen and carbon stable isotope signatures of all fish and invertebrates were similar among the three lakes sampled in 2014 and four lakes sampled in 2015 (Figure 34 and Figure 35 respectively). Cutthroat Trout had high δ^{15}N levels consistent with their top position within lake food webs. Rainbow Trout had lower δ^{15}N and δ^{13}C values than Cutthroat Trout, indicating increased pelagic contribution to diet. Dolly Varden had high δ^{15}N levels consistent with a piscivorous diet, but had similar δ^{13}C to Rainbow Trout. Smaller prey fish generally had lower δ^{15}N and δ^{13}C than large trout and Dolly Varden consistent with their intermediate trophic level positions. Littoral invertebrates and zooplankton had the lowest δ^{15}N signatures consistent with their lower relative food web positions. Zooplankton in particular had the lowest $\delta^{13}C$ levels consistent with their pelagic habitat, while littoral, stream, and terrestrial invertebrates had higher $\delta^{13}C$ isotopic signatures, consistent with the allochthonous source of carbon in their diet. Among the small prey fish, Threespine Stickleback had the lowest $\delta^{13}C$ levels indicative of a pelagic dominated diet, whereas juvenile trout and Sculpin spp (combined under "prey fish") had $\delta^{13}C$ values that overlapped with the littoral invertebrates.

Figure 34. Carbon – nitrogen stable isotope bi-plots (mean ± SD) of fish and invertebrates from Upper Campbell Reservoir Gooseneck Lake, and Middle Quinsam Lake in 2014.

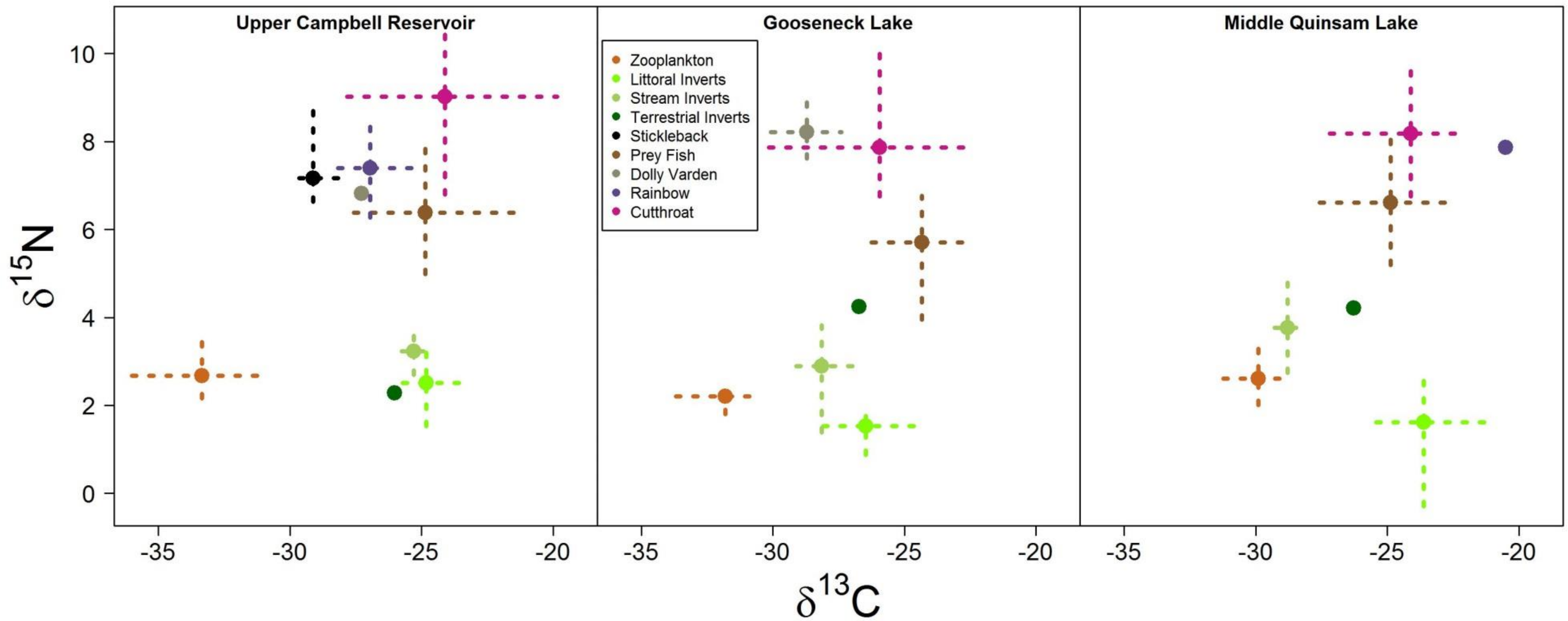


Figure 35. Carbon – nitrogen stable isotope bi-plots (mean ± SD) of fish and invertebrates from Lower Campbell Reservoir, Upper Quinsam Lake, Snakehead Lake, and Beavertail Lake in 2015

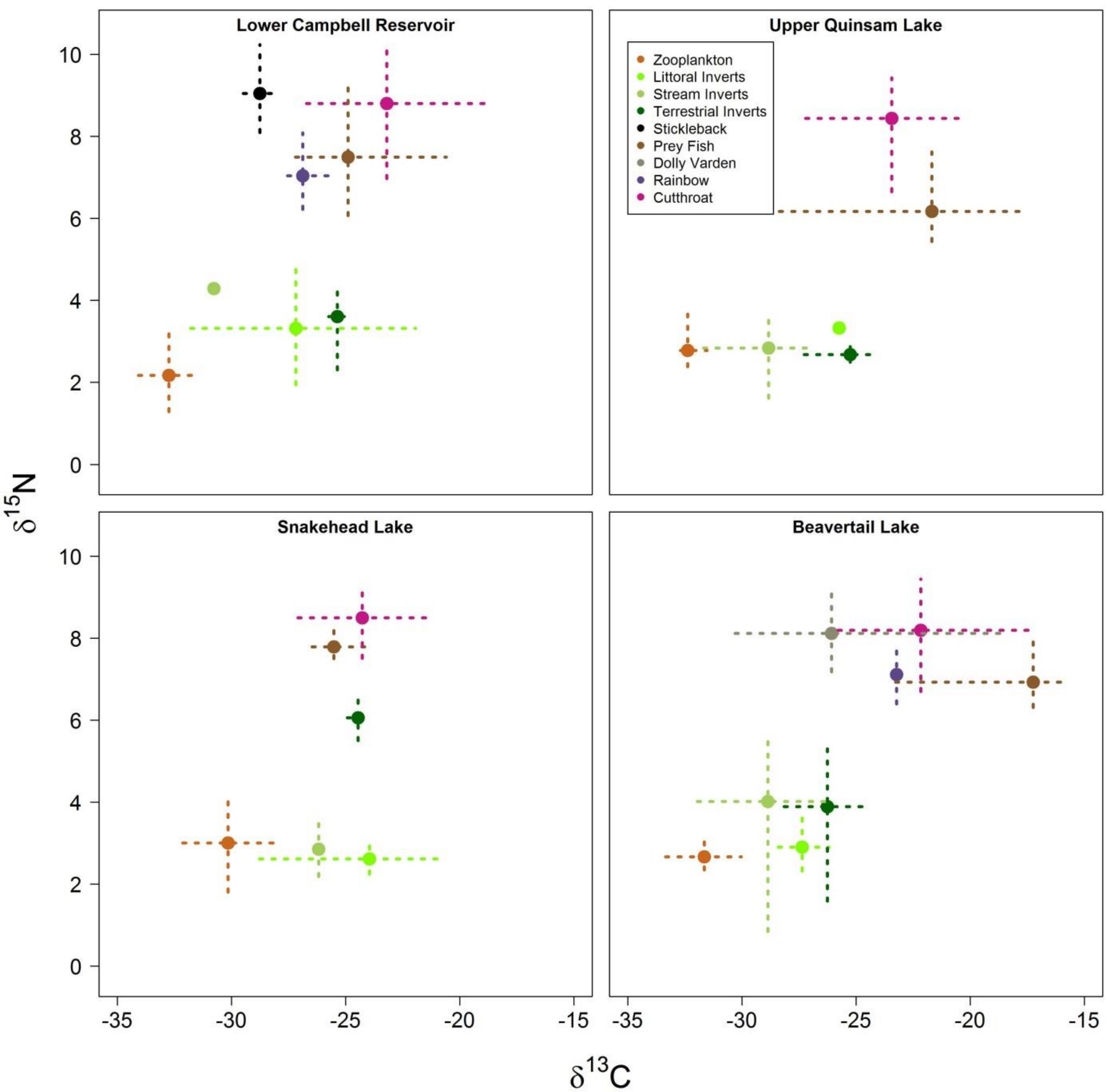


Figure 36. Average $\delta^{15}N$ stable isotope signatures in all taxa sampled across all four lakes in 2015. ZOO = Zooplankton, LI = Littoral Invertebrates, SI = Stream Invertebrates, TI = Terrestrial Invertebrates, TSB = Threespine Stickleback, SC = Sculpin spp., JT = Juvenile Trout, DV = Dolly Varden, RB = Rainbow Trout > 150 mm, CT = Cutthroat Trout > 150 mm.

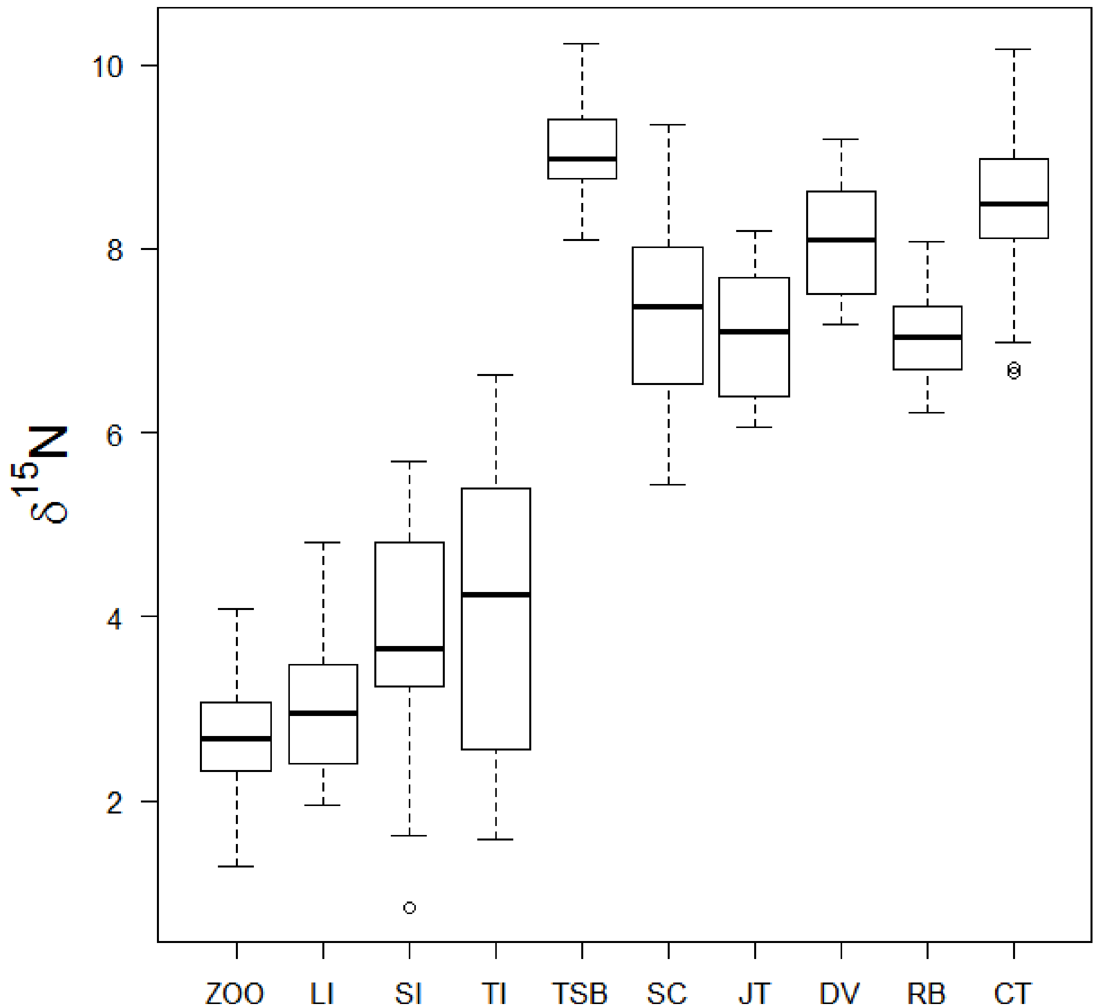
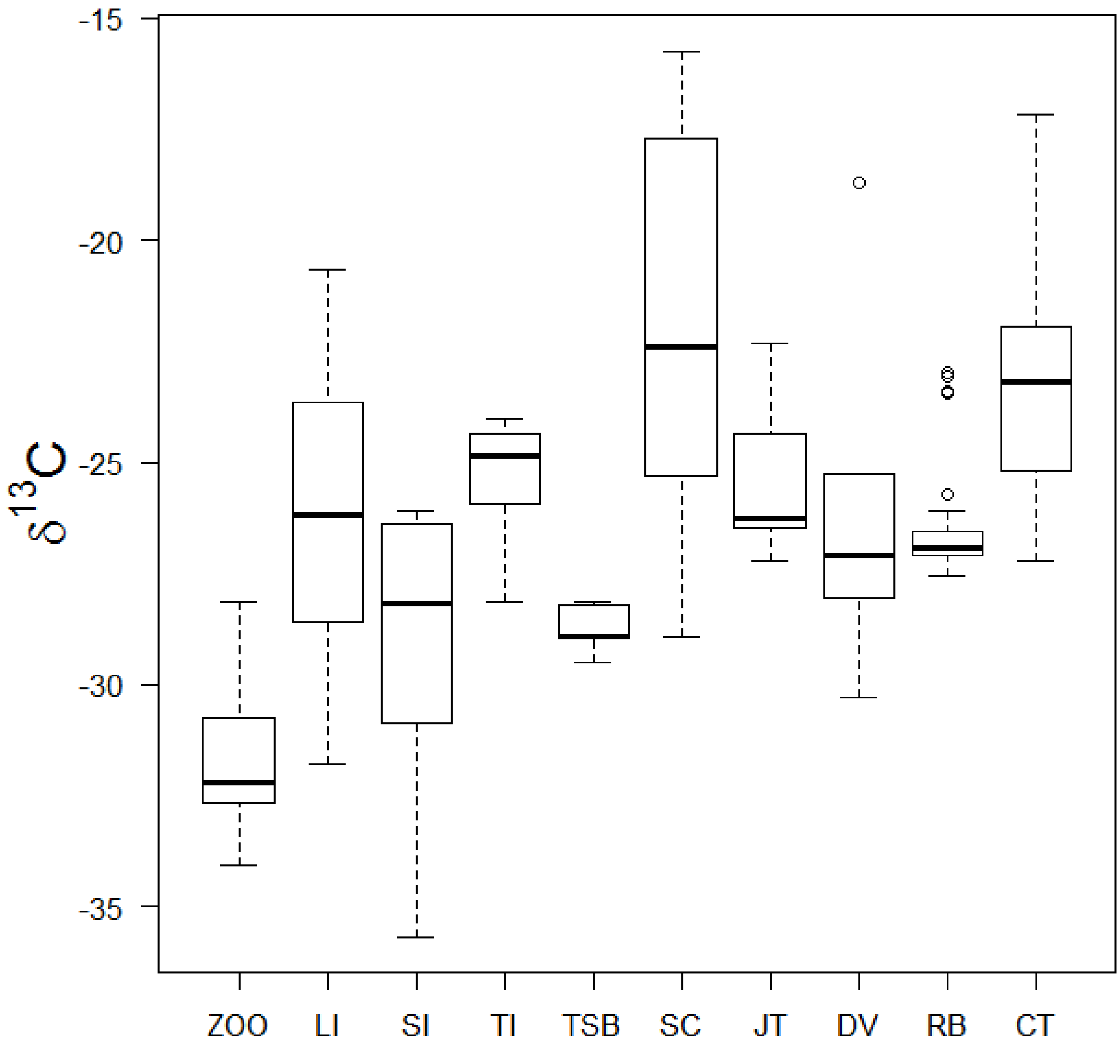


Figure 37. Average $\delta^{13}C$ stable isotope signatures in all taxa sampled across all four lakes in 2015. ZOO = Zooplankton, LI = Littoral Invertebrates, SI = Stream Invertebrates, TI = Terrestrial Invertebrates, TSB = Threespine Stickleback, SC = Sculpin spp., JT = Juvenile Trout, DV = Dolly Varden, RB = Rainbow Trout > 150 mm, CT = Cutthroat Trout > 150 mm.

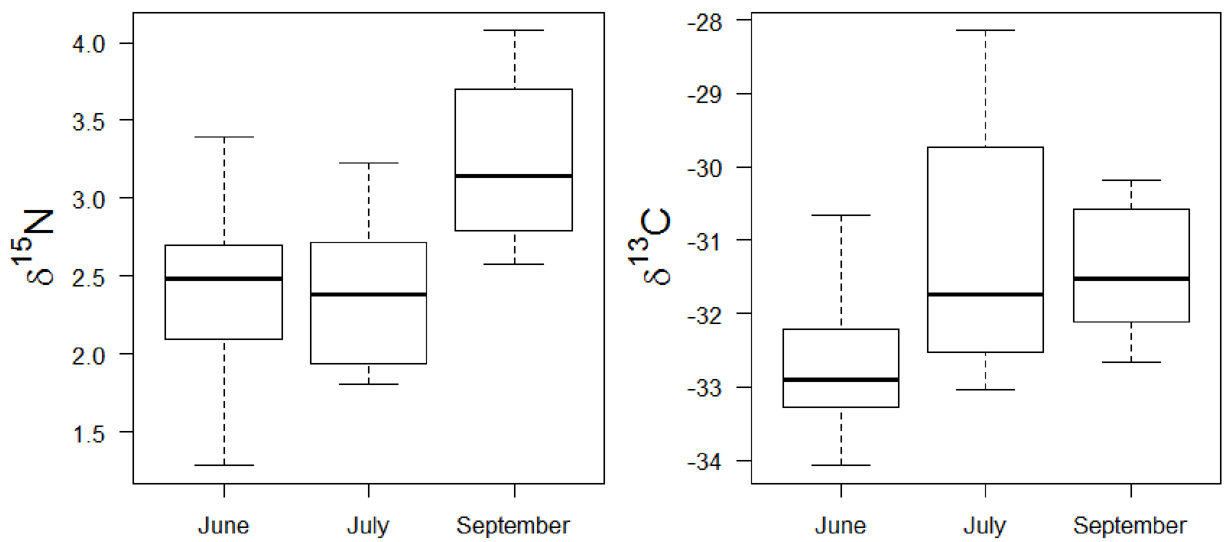


3.3.2. Seasonal Variation in Zooplankton

Nitrogen and carbon stable isotope signatures in bulk zooplankton varied by month of collection. Across all lakes, $\delta^{15}N$ signatures in zooplankton were significantly higher in September compared to June or July (Figure 38, ANOVA: $F_{2,30} = 14.7$, $p < 0.001$). $\delta^{13}C$ signatures in zooplankton were significantly higher in both July and September compared to June (Figure 38, ANOVA: $F_{2,30} = 13.4$, $p < 0.001$). These results are nearly identical to those from the 2014 report. $\delta^{15}N$ and $\delta^{13}C$ signatures in zooplankton were also significantly different by lake. The most enriched

$\delta^{15}N$ and $\delta^{13}C$ were observed in Snakehead Lake and the most depleted $\delta^{15}N$ and $\delta^{13}C$ were observed in Lower Campbell Reservoir (ANOVA: $F_{3,30} > 5.7$, $p < 0.01$).

Figure 38. Monthly variation in the $\delta^{15}N$ and $\delta^{13}C$ stable isotope signatures in zooplankton across all four lakes sampled in 2015.



3.3.3. Assessing Fish Diet Using Mixing Models

Mean estimates of diet contributions of pelagic and littoral sources to fish diets were fairly similar across the seven lakes and are discussed for each lake in the sections below.

3.3.3.1. Lower Campbell and Upper Campbell reservoirs

Cutthroat Trout (age >2+) diets were dominated by littoral prey fish (Sculpin spp., and juvenile trout) in Lower Campbell and Upper Campbell reservoirs (44% and 53% respectively) followed by terrestrial invertebrates in Lower Campbell Reservoir (24%) and Threespine Stickleback in Upper Campbell Reservoir (27%) (Figure 39, Table 27). This is consistent with the biology of Cutthroat Trout as a top piscivorous predator. Zooplankton and the three littoral invertebrate groups each made up < 12% of the diet (exception: terrestrial invertebrates in Lower Campbell Reservoir). In contrast, Rainbow Trout diets had a much higher prevalence of zooplankton (20% and 25% in Lower Campbell and Upper Campbell reservoirs respectively) and had a much lower contribution of littoral prey fish (≤16% in both reservoirs).

These patterns in Cutthroat Trout and Rainbow Trout diet are similar to that observed in the stomach contents. Prey fish made up a high percentage of Cutthroat Trout stomach contents (57-95% in Lower Campbell Reservoir and Upper Campbell Reservoir respectively), and Rainbow Trout had a high percentage of zooplankton (58%) in both reservoirs.

Invertebrate contributions to the diets of Sculpin spp. and juvenile trout were similar, which justified combining these two species into a littoral prey fish group in models, but differed from that of

Threespine Stickleback (Figure 40, Table 27). Littoral prey fish diets were dominated by littoral, stream and terrestrial invertebrates (16-56% for each of the three invertebrate groups at each lake). When the diet contributions of these three invertebrate groups are summed, a total of 87-96% of the littoral prey fish diet is found to consist of these prey items. In contrast, Threespine Stickleback diets were dominated by zooplankton (60%) in Upper Campbell Reservoir and by zooplankton (27%) and stream invertebrates (50%) in Lower Campbell Reservoir. Based on these results for prey fish, only 4% to 13% of juvenile trout and Sculpin diets are pelagic, with the remainder of their diets made up of littoral invertebrate sources. In contrast, pelagic zooplankton makes up 27% to 60% of Threespine Stickleback diet depending on the reservoir.

Figure 39. Estimated proportions of invertebrate and fish diet sources to Cutthroat Trout and Rainbow Trout in Lower Campbell and Upper Campbell reservoirs. Estimates are means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources.

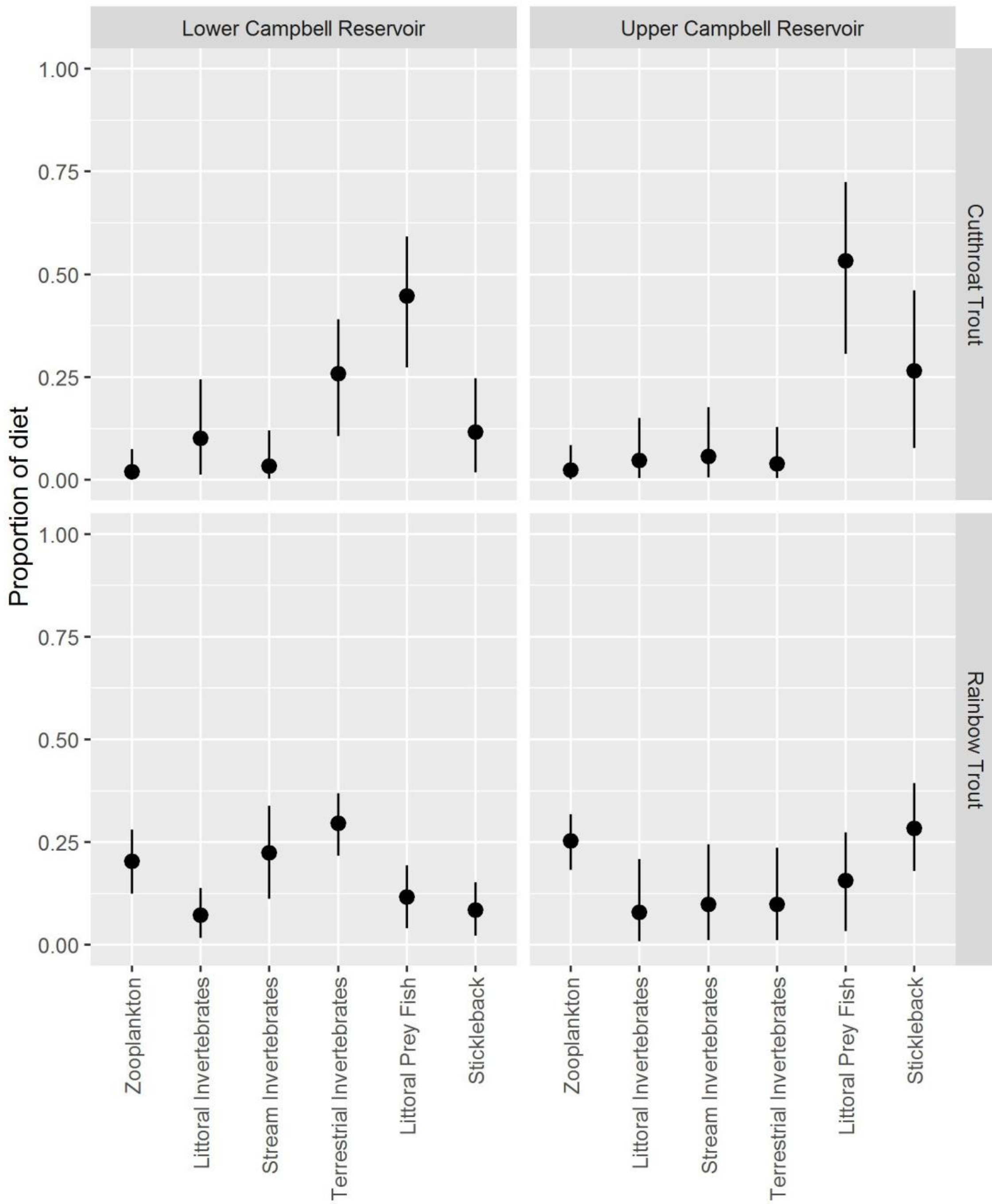


Figure 40. Estimated proportions of invertebrate diet sources to littoral prey fish (juvenile trout and Sculpin spp.) and Threespine Stickleback in Lower Campbell and Upper Campbell reservoirs. Estimates are calculated as means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from prey fish and their potential diet sources.

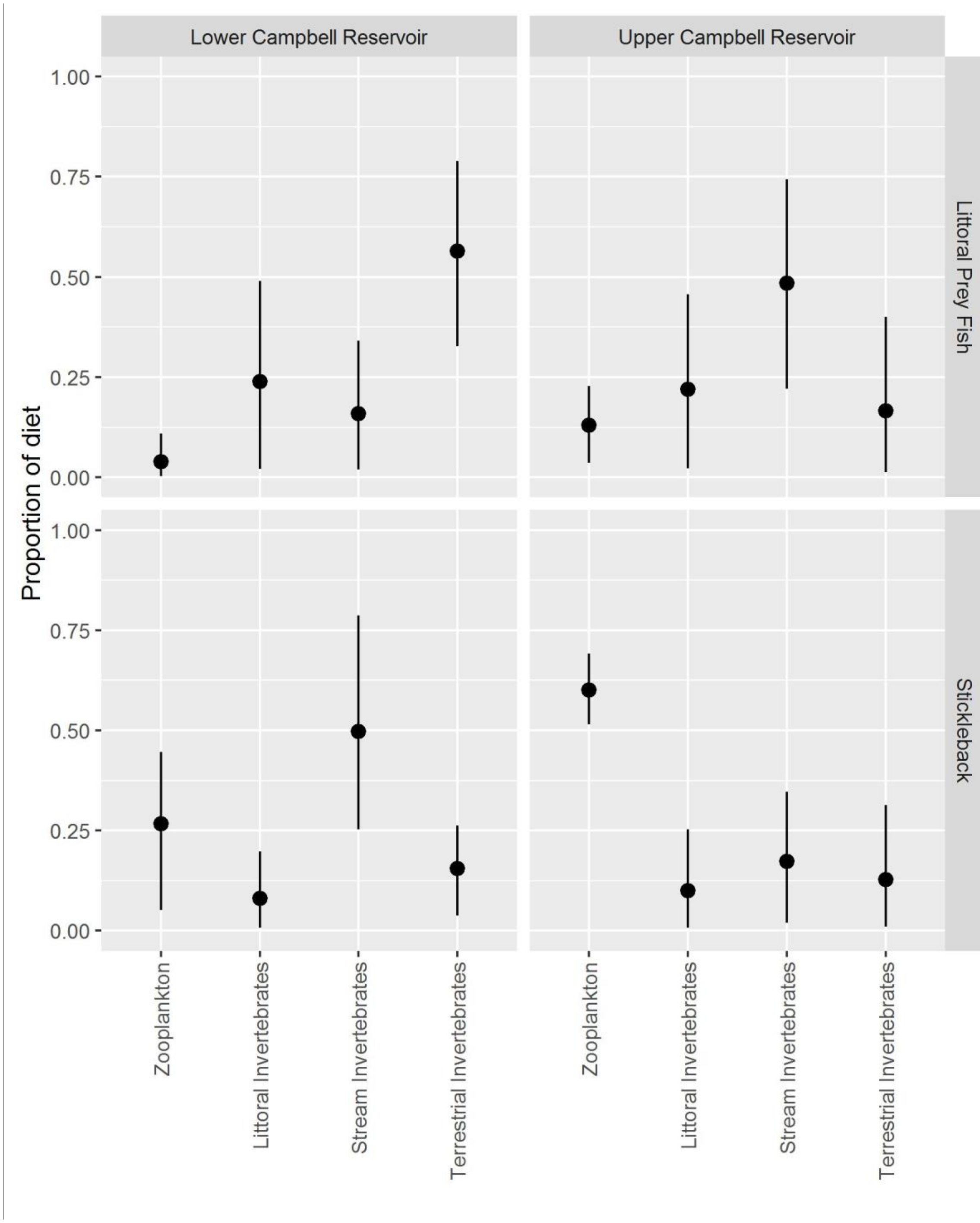


Table 27. SIA-based estimates of diet contribution from A) pelagic and littoral invertebrate and B) prey fish in Upper Campbell and Lower Campbell reservoirs, and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes. Estimates are calculated as means with 5% and 95% quartile range of posterior distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources. Sampling was completed in 2014 and 2015.

A)

Year	Waterbody	Consumer	Estimated Invertebrate Contributions to Diet											
			Pelagic			Littoral								
			Zooplankton			Littoral Invertebrates			Stream Invertebrates			Terrestrial Invertebrates		
			Mean	Q 5%	Q 95%	Mean	Q 5%	Q 95%	Mean	Q 5%	Q 95%	Mean	Q 5%	Q 95%
2014	Upper Campbell Reservoir	Cutthroat Trout	0.03	0.00	0.08	0.06	0.00	0.15	0.07	0.01	0.18	0.05	0.00	0.13
		Rainbow Trout	0.25	0.18	0.32	0.09	0.01	0.21	0.11	0.01	0.24	0.11	0.01	0.24
		Littoral Prey Fish	0.13	0.04	0.23	0.22	0.02	0.46	0.48	0.22	0.74	0.17	0.01	0.40
		Stickleback	0.60	0.52	0.69	0.10	0.01	0.25	0.17	0.02	0.35	0.13	0.01	0.31
	Gooseneck Lake	Cutthroat Trout	0.19	0.04	0.32	0.03	0.00	0.10	0.10	0.01	0.29	0.24	0.03	0.50
		Dolly Varden	0.36	0.07	0.63	0.10	0.01	0.27	0.18	0.02	0.40	0.19	0.02	0.40
		Littoral Prey Fish	0.05	0.00	0.15	0.40	0.21	0.58	0.16	0.01	0.38	0.39	0.22	0.57
	Middle Quinsam Lake	Cutthroat Trout	0.06	0.01	0.16	0.08	0.01	0.17	0.09	0.01	0.22	0.22	0.04	0.43
		Littoral Prey Fish	0.13	0.01	0.31	0.22	0.06	0.39	0.24	0.03	0.44	0.40	0.16	0.67
2015	Lower Campbell Reservoir	Cutthroat Trout	0.03	0.00	0.07	0.11	0.01	0.24	0.04	0.00	0.12	0.25	0.11	0.39
		Rainbow Trout	0.20	0.13	0.28	0.07	0.02	0.14	0.22	0.11	0.34	0.30	0.22	0.37
		Littoral Prey Fish	0.04	0.00	0.11	0.24	0.02	0.49	0.16	0.02	0.34	0.56	0.33	0.79
		Stickleback	0.27	0.05	0.45	0.08	0.01	0.20	0.50	0.25	0.79	0.16	0.04	0.26
	Upper Quinsam Lake	Cutthroat Trout	0.12	0.02	0.24	0.13	0.01	0.30	0.10	0.01	0.24	0.07	0.00	0.18
		Littoral Prey Fish	0.19	0.02	0.41	0.30	0.06	0.55	0.23	0.03	0.45	0.28	0.05	0.52
	Beavertail Lake	Cutthroat Trout	0.13	0.02	0.28	0.11	0.01	0.26	0.16	0.03	0.32	0.13	0.01	0.28
		Rainbow Trout	0.16	0.02	0.32	0.21	0.04	0.38	0.16	0.02	0.32	0.19	0.03	0.36
		Dolly Varden	0.17	0.02	0.35	0.15	0.01	0.33	0.22	0.03	0.41	0.19	0.02	0.37
		Littoral Prey Fish	0.19	0.02	0.40	0.25	0.04	0.47	0.27	0.05	0.48	0.28	0.06	0.49
	Snakehead Lake	Cutthroat Trout	0.12	0.03	0.22	0.11	0.02	0.23	0.12	0.02	0.25	0.27	0.11	0.42
		Littoral Prey Fish	0.33	0.15	0.47	0.11	0.01	0.29	0.17	0.02	0.38	0.39	0.19	0.56

Table 27. Continued.

B) Year	Waterbody	Consumer	Estimated Vertebrate Contributions to Diet					
			Threespine Stickleback			Littoral Prey Fish		
			Mean	Q 5%	Q 95%	Mean	Q 5%	Q 95%
2014	Upper Campbell Reservoir	Cutthroat Trout	0.27	0.08	0.46	0.53	0.31	0.72
		Rainbow Trout	0.28	0.18	0.39	0.16	0.03	0.27
	Gooseneck Lake	Cutthroat Trout		NA		0.44	0.25	0.61
		Dolly Varden				0.17	0.01	0.43
	Middle Quinsam Lake	Cutthroat Trout		NA		0.54	0.42	0.65
2015	Lower Campbell Reservoir	Cutthroat Trout	0.12	0.02	0.25	0.44	0.27	0.59
		Rainbow Trout	0.09	0.02	0.15	0.12	0.04	0.19
	Upper Quinsam Lake	Cutthroat Trout		NA		0.59	0.48	0.68
	Beavertail Lake	Cutthroat Trout				0.47	0.38	0.55
		Rainbow Trout		NA		0.29	0.19	0.38
		Dolly Varden				0.27	0.10	0.44
	Snakehead Lake	Cutthroat Trout		NA		0.38	0.27	0.49

3.3.3.2. Diversion Lakes

Prey fish and littoral invertebrates made up the majority of Cutthroat Trout (age >2+) diets across the five diversion lakes (Figure 41, Table 27). Prey fish contributed an estimated 38% to 59% to Cutthroat Trout diet in the diversion lakes. Invertebrates from the three littoral groups (littoral, stream and terrestrial invertebrates) varied in their individual contribution to Cutthroat Trout diets (3% to 27%). When the diet contributions of the three littoral invertebrate groups were summed, the contribution to Cutthroat Trout diet varied from 30% at Upper Quinsam Lake to 50% at Snakehead Lake. In contrast, zooplankton contributed little to Cutthroat Trout diets across all five diversion lakes (6% to 19%).

The diet estimates for Cutthroat Trout based on stable isotope analysis are similar to the stomach content analysis results, except that the stable isotope results estimate a greater contribution of prey fish to Cutthroat Trout diet. Stomach content analyses generally showed low contribution of zooplankton to Cutthroat Trout diet, which matches that observed in the stable isotope modeling. One exception is Gooseneck Lake, which had an unusually high estimate for zooplankton of 53% of diet based on stomach content analysis of 10 Cutthroat Trout individuals. The contribution of zooplankton to Cutthroat Trout diet was also highest at Gooseneck Lake using stable isotope methods compared to the other lakes, although it was much lower at 19%. Littoral invertebrates were found to contribute 36% to 91% to diet using stomach content methods versus 30% to 50% using the stable isotope methods. The difference in relative contribution between methods was made up for in increased contribution of prey fish to Cutthroat Trout diet estimated by the stable isotope method.

The diets of Rainbow Trout and Dolly Varden were relatively evenly split among the range of prey sources in Beavertail and Gooseneck Lake (Figure 42, Table 27). The diet contribution of littoral invertebrates was 7% to 30% considering each of the three invertebrate groups separately, or 47% to 59% when summing the average contribution across the three groups. Prey fish made up 27% and 29% of Rainbow Trout and Dolly Varden diets respectively in Beavertail Lake, and 17% of Dolly Varden diets in Gooseneck Lake. Zooplankton contributed an estimated 16-17% to Rainbow Trout and Dolly Varden diets in Beavertail Lake, and 36% to Dolly Varden diet in Gooseneck Lake.

As in the Lower Campbell and Upper Campbell Reservoirs, the diets of prey fish sampled in the five diversion lakes were dominated by littoral invertebrates, with each group making up 11-40% of diets, or 67-95% of diets when combined. Again, zooplankton contributed significantly less to prey fish diets (5-19%). The exception to this was in Snakehead Lake, where zooplankton contributed to 33% of prey fish diets.

Figure 41. Estimated proportions of invertebrate and vertebrate diet sources to Cutthroat Trout in Beavertail, Snakehead, and Upper Quinsam lakes in 2015 (top row) and Gooseneck and Middle Quinsam lakes in 2014 (bottom row). Estimates are means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources.

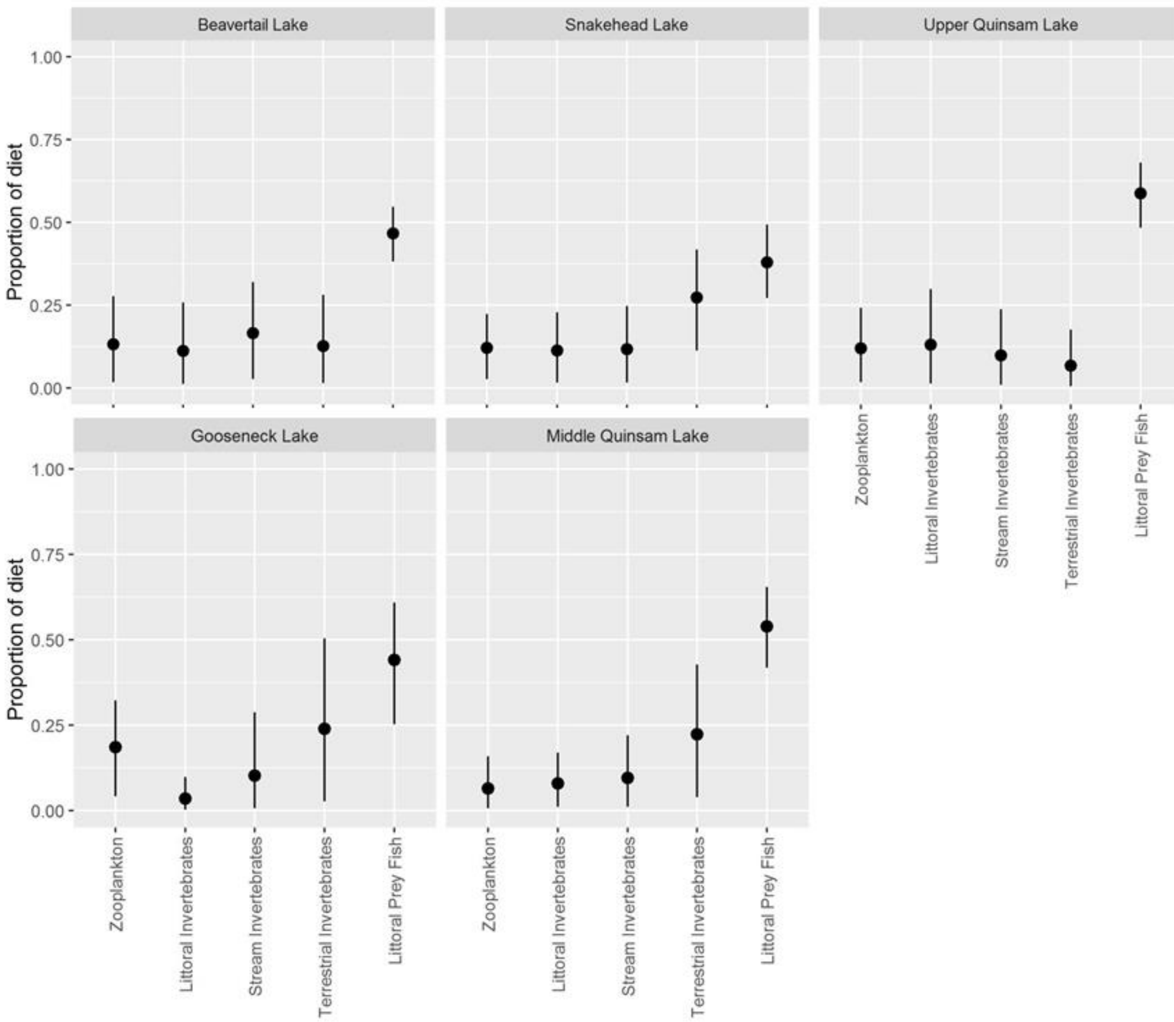


Figure 42. Estimated proportions of invertebrate and vertebrate diet sources to Rainbow Trout and Dolly Varden in Beavertail Lake and Dolly Varden in Gooseneck Lake. Estimates are means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources.

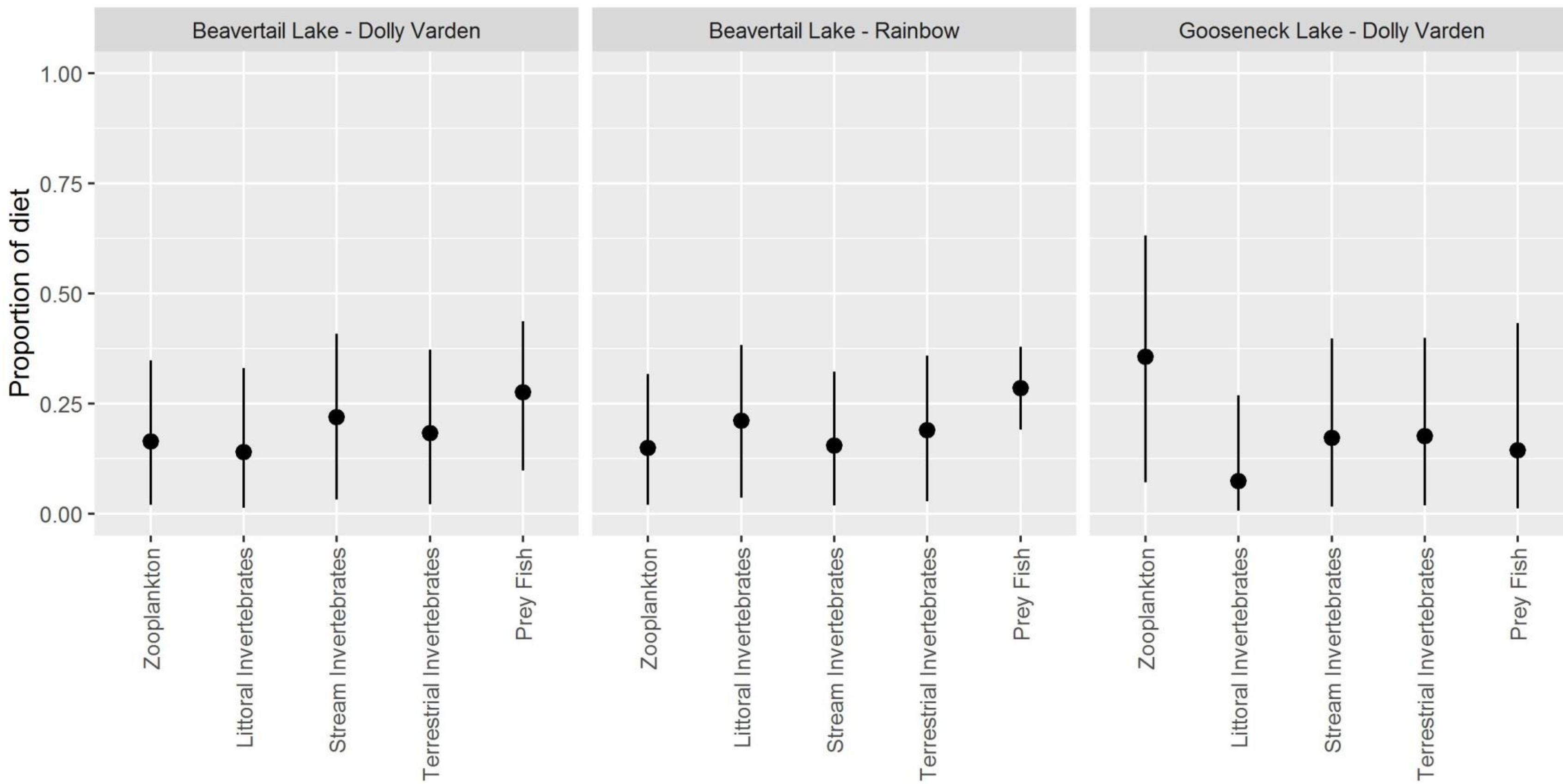
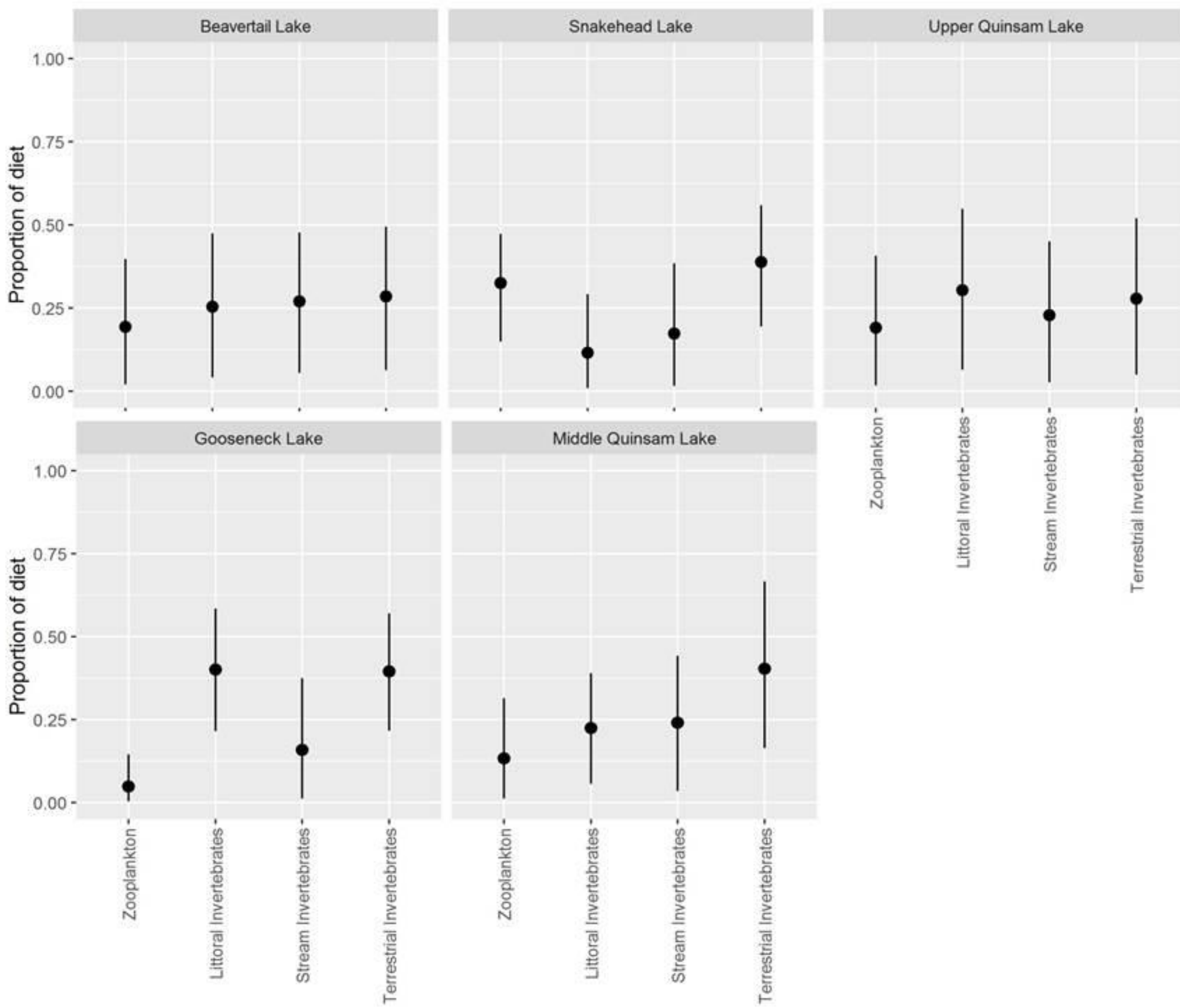


Figure 43. Estimated proportions of invertebrate diet sources to littoral prey fish (Sculpin spp. and juvenile trout) in Beavertail, Snakehead, and Upper Quinsam lakes in 2015 (top row) and Gooseneck and Middle Quinsam lakes in 2014 (bottom row). Estimates are calculated as means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from prey fish and their potential diet sources.



3.3.4. Diet Variation with Fish Size

Cutthroat Trout and Rainbow Trout δ^{15}N signatures are both highly positively correlated to fork length in Lower Campbell Reservoir (Figure 44, Cutthroat $F_{1,25} = 6.8$, Rainbow $F_{1,27} = 21.1$, both $p < 0.02$). This indicates that larger Cutthroat Trout and Rainbow Trout are more piscivorous and eat higher in the food web than smaller individuals. Both Cutthroat Trout and Rainbow Trout δ^{13}C signatures are not related to fork length (Cutthroat $F_{1,25} = 1.9$, Rainbow $F_{1,27} = 0.1$, both $p > 0.18$). This suggests that the dominant source of carbon (Cutthroat Trout = littoral; Rainbow Trout = pelagic) does not vary by fish age and body size.

Cutthroat Trout δ^{15}N signatures were positively related to fork length in Beavertail, Snakehead and Upper Quinsam lakes (all $p < 0.01$) (Figure 45). Cutthroat Trout δ^{13}C signatures were positively related to fork length in Upper Quinsam Lake ($F_{1,18} = 19.9$, $p < 0.001$), but were not related to fork length in either Beavertail or Snakehead lakes ($p > 0.08$). These results suggest that that larger Cutthroat Trout are more piscivorous and eat higher in the food web than smaller individuals.

Figure 44. δ^{15}N and δ^{13}C stable isotope signatures by fork length (mm) in Cutthroat Trout (open circles) and Rainbow Trout (closed triangles) from Lower Campbell Reservoir.

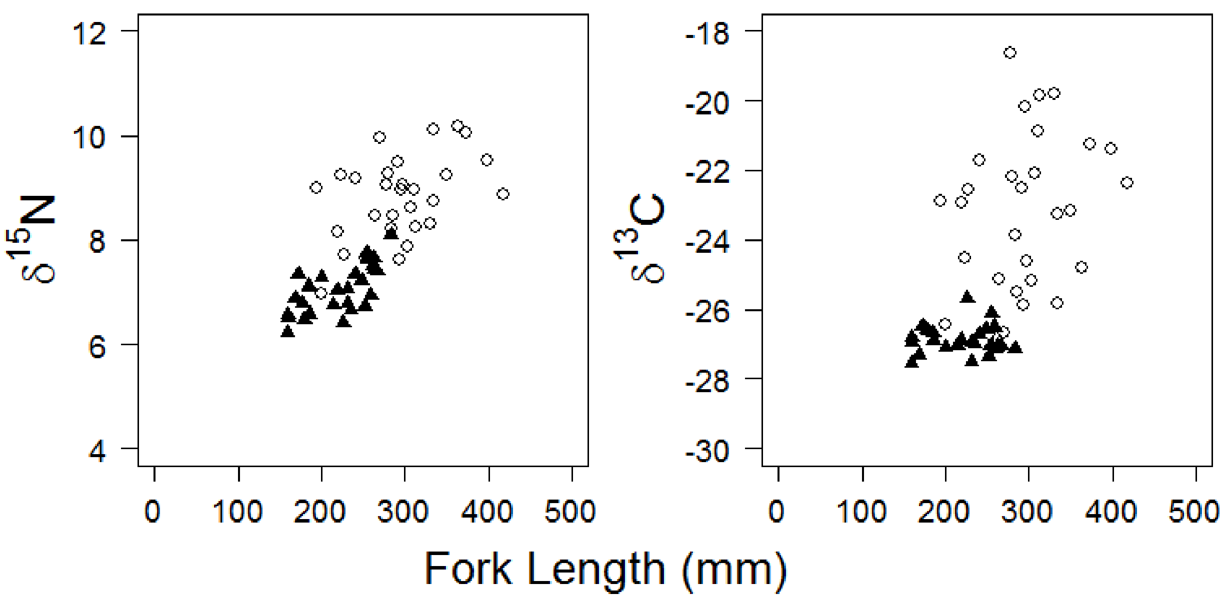
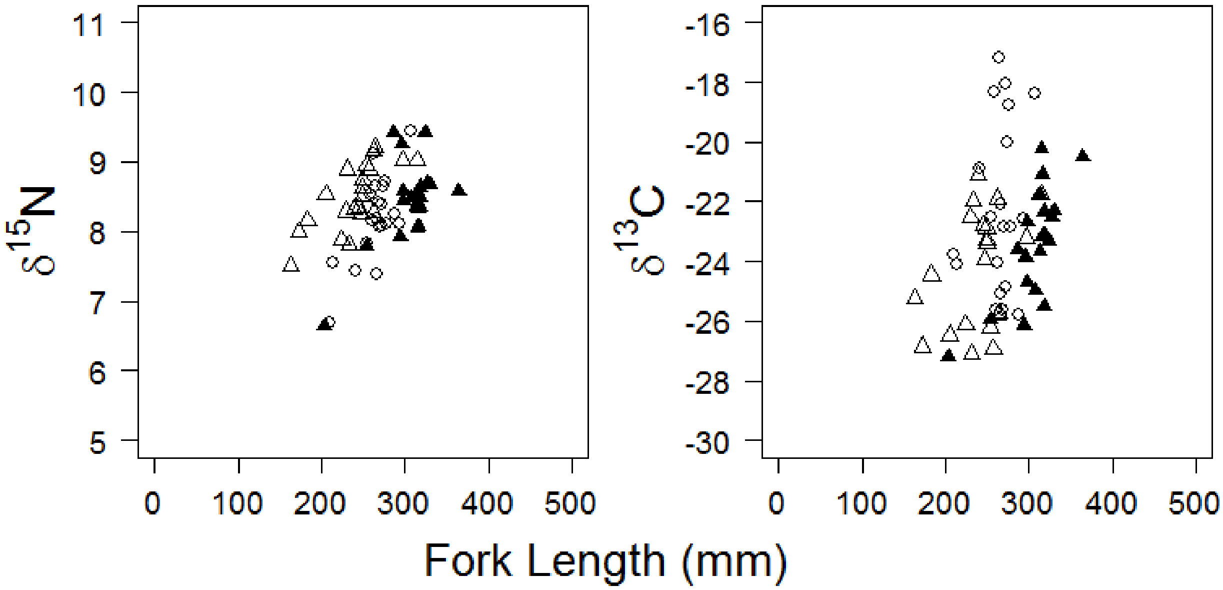


Figure 45. $\delta^{15}N$ and $\delta^{13}C$ stable isotope signatures by fork length (mm) in Cutthroat Trout from Beavertail Lake (open circles), Snakehead Lake (open triangles) and Upper Quinsam Lake (closed triangles).



3.4. Water Residence Time

The physical characteristics of the study lakes and reservoirs are presented in Table 28. Upper Quinsam Lake is situated at the highest elevation (358 m), and Lower Campbell Reservoir is at the lowest elevation (178 m). The reservoirs have the greatest surface area, and the diversion lakes have smallest surface areas. Lower Campbell Reservoir and Upper Quinsam Lake are the deepest of the study lakes; Snakehead is the shallowest lake. Thermocline depth is greatestin the reservoirs and shallowest in Middle Quinsam Lake. For all study years, Upper Campbell Reservoir and Lower Campbell Reservoir had the highest annual and seasonal inflows and outflows, and Gooseneck and Beavertail lakes had the lowest (Table 29, Table 30).

Table 28. Physical characteristics of the study lakes.

Waterbody	Description	Elevation (m)	Surface Area (km^2)	Mean Water Depth[1] (m)	Max Water Depth[1] (m)	Depth to Thermocline[2] (m)
Upper Campbell	Reservoir	221	68.7	12.2	39.6	20.0
Lower Campbell	Reservoir	178	26.5	18.0	69.7	17.0
Beavertail Lake	Control	270	1.03	10.8	26.0	11.9
Gooseneck Lake	Diversion (Receiving Lake)	290	0.78	9.80	38.0	10.0
Middle Quinsam Lake	Diversion (Donor Lake)	270	0.72	4.00	14.6	7.3
Snakehead Lake	Diversion (Receiving Lake)	283	0.20	3.50	9.00	8.0
Upper Quinsam Lake	Control	335	4.38	13.1	48.0	10.3

[1] Average of the depths reported in Hatfield (2000) and on bathymetric maps.

[2] Average depth of thermocline from Hatfield (2000) and the data collected by Ecofish in September 2015.

Estimates of water residence time by lake using the water balance method and the gauged flow data method are shown in Table 29 and Table 30. Estimates of water residence time were derived for the whole year (Table 29) and for stratified period (Table 30), which spans May 15th to September 30th (also referred to as seasonal water residence time).

Upper Campbell Reservoir, Lower Campbell Reservoir, and Upper Quinsam Lake had annual average water residence times exceeding 100 days of 526, 114 and 277 days respectively computed using the water balance method. These long annual water residence times are largely due to the large surface area and water volume of these lakes. Although Upper Quinsam Lake has a lower volume than Lower Campbell Reservoir, it has substantially less input and output of water, which increases its water residence time (Table 14 and Table 29).

Beavertail Lake had the longest average annual and seasonal water residence time (913 and 395 days respectively), computed using the water balance method. The water residence times for Beavertail Lake were relatively long considering the surface area, volume, and watershed area. The long residence time at Beavertail Lake is likely due to its landscape position. Beavertail Lake occurs in the upper reaches of a relatively isolated watershed, disconnected from the larger drainage system that comprises the other diversion lakes (Map 6), and thus receives less water input than the other lakes. Water residence time in Beavertail Lake is long because it has a large volume relative to water inputs (Table 29, Table 30).

Snakehead Lake had the shortest annual water residence time (7 days), while Lower Campbell Reservoir had the shortest seasonal water residence time (1.4 days). The short residence time of Snakehead Lake is expected given its small volume and watershed size. The short seasonal residence time for Lower Campbell Reservoir is somewhat surprising given its large surface area. However, Lower Campbell Reservoir also has relatively large inflows and outflows relative to its volume in the summer, which reduces its seasonal water residence time. A comparison of seasonal water residence time computed from the gauged data estimates Lower Campbell Reservoir to have a seasonal residence time of 9.1 days in 2015, which is 6.4 days greater than the residence time computed from the water balance method for the same year (Table 30). While both methods support the conclusion that seasonal water residence in Lower Campbell Reservoir is short, the water balance method may moderately underestimate residence time for this reservoir.

Annual water residence time was longer than the water residence time computed for the stratified period for all lakes except Middle Quinsam Lake (Table 29, Table 30). This may be the result of a large decrease in lake outflow (1.2 m^3/s) computed at Middle Quinsam during the stratified season, which is the largest drop in outflow compared to all of the lakes. The difference between annual and seasonal water residence times varied, on average, between 3.2 to 518 days, with Lower Campbell Reservoir and Upper Quinsam Lake showing the greatest reduction in water residence time during the stratified period (Table 30).

3.4.1. Data Limitations and Error Analysis

The estimates for water residence time by lake are only as accurate as the data used to derive them, and should be used for comparative purposes and not as absolute values. This section provides an assessment of the accuracy of the computed water residence times using the water balance method and some of the data limitations.

There is some uncertainty around our estimates for evaporation at each lake. The heat transferred into the lake by inflows and outflows of water may be a significant factor in the energy budget of the lakes and thus the evaporation rates, which was not taken into account. Seasonal variations in the evaporation rate can be significantly affected by the heat storage capacity of the water body which is, to a large extent, determined by its depth. In addition, advective energy, driven largely by wind, typically increases evaporation rates. Due to lack of data and resources, these factors were not taken into account in the lake evaporation estimates. However, even large errors in estimates of evaporation (e.g., 100 mm) play a minor role in water residency time. This is because water inflows and outflows were one to two orders of magnitude greater than evaporation from the lakes (Table 14 and Table 15). Due to the lack of data, groundwater seepage was also not taken into account in the water balance computation, but like evaporation, is expected to be a minor factor in estimates of water residency time.

The annual water residence time calculations were particularly dependent on lake volume and watershed area. Stage-volume data from 2001 were used to determine lake volume (Bruce 2001). However, it is unclear when the bathymetric surveys were conducted in Bruce (2001). If the data were collected under low water conditions, then the resultant volumes would be underestimated. Watershed area was used to determine the rate of inflow; therefore, accurate delineation of watershed area is important to the water residence computations. When delineating the watersheds, it was assumed that the entire watershed area upstream of the lake outlet contributes water inputs to the lake, rather than the local watershed area.

Likewise, the lake volumes used to determine seasonal water residence time have some important assumptions that require further investigation. Thermocline depth was used as a boundary condition for flow within the lake during the stratified season. However, the thermocline data were relatively limited in extent and did not incorporate the spatial and temporal variation of thermal conditions at each lake. In addition, few measurements of thermocline depth were used to ascertain the stratification period (May 15th to September 30th), which was assumed to be the same for all lakes, in all years. Although mean water residence time in a lake is unaffected by stratification, we estimated the residence time of the epilimnion during the growing season because it is more representative of euphotic waters that plankton predominantly inhabit and where autotrophic production occurs. Error in these estimates of growing season water residence time will therefore reflect the assumptions that summer stratified conditions extended for the same period for each lake and a the thermocline depth remained constant in each lake.

As a quality assurance check, water residence was computed using gauged inflow and outflow data for Gooseneck and Middle Quinsam lakes and Upper Campbell and Lower Campbell reservoirs, and compared to the water residence times using the water balance method derived from Equation (1) (Table 31, Table 32). The computed water residence times (derived from Equation 1) compare well with residence times derived from gauged data, particularly for the smaller lakes. The most significant exception is found with the estimates for annual water residence time of the reservoirs, particularly Upper Campbell Reservoir (Table 32). The water residence time computed from the water balance method (Equation 1) was 205 days greater than that derived from the gauged data. The difference could be due to a number of factors including errors in the computation of watershed area, inaccurate volume of the reservoir, missing or incorrect data related to land surface type and land use that would result in an inaccurate runoff coefficient, and/or underestimation of soil retention capacity (S). The errors of the gauged data are not known, but are likely dependant on the stage-discharge relationship developed for each gauge. Overall, the similarity in estimates between the two methods provides us with confidence that water residence time estimates can be reasonably compared across lakes to understand how this variable may affect lake food webs and fish production.

Table 29. Lake volume, annual lake outflow (Q_o), and annual water residence time (WRT) computed for 2012-2015 from the water balance method (WB) and gauged flow data method (Gauged).

Waterbody	Volume (m^3)	2015 Q_o (m^3/year)	2015 WRT (days)	2014 Q_o (m^3/year)	2014 WRT (days)	2013 Q_o (m^3/year)	2013 WRT (days)	2012 Q_o (m^3/s)	2012 WRT (days)	Average WB WRT (days)	Average WB WRT (years)	Average Gauged WRT (days)
Upper Campbell Reservoir	2.46E+09	1.52E+09	590.4	2.30E+09	390.4	1.29E+09	697.7	2.12E+09	423.5	525.5	1.44	384.9
Lower Campbell Reservoir	3.16E+08	1.60E+09	72.0	9.40E+08	122.6	8.35E+08	138.2	9.39E+08	122.8	113.9	0.31	46.9
Beavertail Lake	1.10E+07	4.13E+06	968.2	4.59E+06	871.1	4.36E+06	917.9	4.47E+06	894.8	913.0	2.50	-
Gooseneck Lake	7.53E+06	3.12E+07	88.2	3.86E+07	71.3	4.63E+07	59.3	3.29E+07	83.5	75.6	0.21	76.4
Middle Quinsam Lake	2.82E+06	6.27E+07	16.4	7.16E+07	14.4	6.74E+07	15.3	7.22E+07	38.1	21.0	0.06	22.6
Snakehead Lake	7.62E+05	3.41E+07	8.1	4.03E+07	6.9	4.62E+07	6.0	3.56E+07	7.8	7.2	0.02	-
Upper Quinsam Lake	5.28E+07	6.24E+07	309.0	7.33E+07	262.8	7.40E+07	260.4	6.96E+07	277.0	277.3	0.76	-

Table 30. Estimated epilimnion volume, seasonal lake outflow (Q_o), and seasonal water residence time (WRT) computed for 2012-2015 from the water balance method (WB) and gauged flow data method (Gauged).

Waterbody	Volume (m^3)	2015.0 Q_o (m^3/s)	2015.0 WRT (days)	2014.0 Q_o (m^3/s)	2014.0 WRT (days)	2013.0 Q_o (m^3/s)	2013.0 WRT (days)	2012.0 Q_{out} (m^3/s)	2012.0 WRT (days)	Average WB WRT (days)	Average WB WRT (years)	Average Gauged WRT (days)
Upper Campbell Reservoir	3.21E+08	12.7	293.8	12.7	292.4	14.8	252.0	13.7	272.0	277.5	0.8	293.8
Lower Campbell Reservoir	1.05E+07	45.8	2.7	12.8	1.0	12.0	1.0	13.4	0.9	1.4	0.0	9.1
Beavertail Lake	1.89E+06	0.0	511.0	0.1	378.9	0.1	340.7	0.1	348.1	394.7	1.1	-
Gooseneck Lake	3.33E+05	0.7	5.7	0.7	5.9	0.6	6.6	0.6	6.2	6.1	0.0	8.5
Middle Quinsam Lake	1.99E+06	0.6	38.4	0.9	25.1	0.9	25.5	1.3	18.1	26.7	0.1	35.8
Snakehead Lake	2.36E+05	0.7	3.8	0.7	4.1	0.7	4.1	0.7	4.2	4.0	0.0	-
Upper Quinsam Lake	1.84E+06	0.8	26.9	1.0	21.4	1.0	22.1	1.2	17.9	22.1	0.1	-

Table 31. Annual and seasonal water residence time (WRT) computed from gauged inflow data (Gauged Q_i), and derived inflow data (Computed Q_i) for Gooseneck and Middle Quinsam lakes for years where gauged data were available.

Waterbody		2015		2013		2012		Average	Average	Average
		Gauged Q_i (m^3/s)	WB Q_i (m^3/s)	Gauged Q_i (m^3/s)	WB Q_i (m^3/s)	Gauged Q_i (m^3/s)	WB Q_i (m^3/s)	Gauged WRT (days)	WB WRT (days)	Difference WRT (days)
Gooseneck Lake	Annual	1.1	1.0	1.2	1.5	1.1	1.0	76.4	77.0	0.6
	Seasonal	1.0	0.7	1.0	0.6	1.0	0.6	8.5	5.7	-2.8
Middle Quinsam Lake	Annual	2.4	2.0	1.6	2.1	2.6	2.3	22.6	15.3	-7.3
	Seasonal	0.6	0.6	1.5	0.9	2.2	1.3	35.8	38.4	2.6

Table 32. Annual and seasonal water residence time (WRT) computed from gauged outflow data (Gauged Q_o) and derived outflow data (Computed Q_o) for the Upper Campbell and Lower Campbell reservoirs in 2015.

Waterbody		Gauged Q_o (m^3/s)	Computed Q_o (m^3/s)	Gauged WRT (days)	Computed WRT (days)	Difference WRT (days)
Upper Campbell Reservoir	Annual	73.9	48.2	384.9	590.4	205.4
	Seasonal	14.3	12.7	260.0	293.8	33.8
Lower Campbell Reservoir	Annual	77.9	50.8	46.9	72.0	25.0
	Seasonal	13.4	45.8	9.1	2.7	-6.4

3.5. Analysis of Management Questions

3.5.1. To what extent do stabilized reservoir levels, as affected by BC hydro operations, benefit fish populations?

The total littoral vs. pelagic contribution to Cutthroat Trout and Rainbow Trout diet for Upper Campbell and Lower Campbell reservoirs can be estimated by summing the contributions of the invertebrate prey to Cutthroat Trout and Rainbow Trout diet (direct pathway) with the relative contributions of invertebrate prey to the prey fish in the Cutthroat Trout and Rainbow Trout diet (indirect pathway) (Table 33). Using this method, a total of 26% of the diet of Cutthroat Trout (age >2+) in Upper Campbell Reservoir is estimated derived from pelagic sources while 74% is estimated derived from littoral sources. In Lower Campbell Reservoir, only 8% of Cutthroat Trout diet is estimated to be derived from pelagic sources while 92% is estimated to be derived from littoral sources. In contrast, Rainbow Trout in Upper Campbell Reservoir have a pelagic contribution to diet of 44% and a littoral contribution to diet of 56%. In Lower Campbell Reservoir, Rainbow Trout have a pelagic contribution to diet of 23% and a littoral contribution to diet of 77%.

In summary, Lower Campbell Reservoir supports greater littoral contribution to diet than Upper Campbell Reservoir for both Cutthroat Trout and Rainbow Trout. The SIA results are consistent with the expected differences in littoral production due to drawdown magnitude in the two reservoirs. Rainbow Trout have a greater pelagic contribution to diet than Cutthroat Trout in both reservoirs. Despite the large lake areas, the top fish consumers in both reservoirs appear to be supported by littoral production to a greater extent than pelagic production.

3.5.2. What is the relationship between residence time (as affected by diversion rate) and lake productivity?

The total littoral vs. pelagic contribution to Cutthroat Trout diet for all diversion and control lakes sampled in 2014 and 2015 can be estimated by summing the contributions of the invertebrate prey to Cutthroat Trout diet (direct pathway) with the relative contributions of invertebrate prey to the small fish in Cutthroat Trout diet (indirect pathway) (Table 33). Using this method, a total of 14% to 24% of the diet of Cutthroat Trout (age >2+) is estimated derived from pelagic sources across all diversion and control lakes sampled in Year 1 and Year 2; estimates of littoral contribution to diet ranged from 76% to 86% across all lakes.

Estimates of the total littoral or pelagic contribution to Cutthroat Trout diet across lakes was compared to annual and seasonal water residence time for each lake (Figure 46). No clear linear relationship was observed between annual and seasonal water residence time and total pelagic contribution to Cutthroat Trout diet. However, there appeared to be an asymptotic relationship between both annual and seasonal water residence time, and pelagic contribution to Cutthroat Trout diet. In lakes with short water residence times there is substantial variability in estimates of pelagic contribution to diet. With longer water residence time, pelagic contribution to diet ranged from ~22% to 26%, although sample size is low.

Gooseneck and Snakehead lakes receive water from the Quinsam River diversion whereas Middle Quinsam Lake has water diverted upstream of the lake and thus is a donor lake. We estimated how different water diversion scenarios could affect the water residence time of Gooseneck, Snakehead and Middle Quinsam lakes in three different years (Table 34, Table 35). Water diversion scenarios included a significant diversion scenario (90% of flow), average diversion (35% of flow), minor diversion (10% of flow) and no diversion (0% of flow). Under no diversion, Gooseneck and Snakehead lakes have annual water residence time of 33.9 and 18.9 days respectively. Under average diversion conditions of 35%, annual water residence time decreases by roughly 65% in each lake to 11 days in Gooseneck Lake and 7.2 days in Snakehead Lake. In contrast, the annual water residence time of Middle Quinsam Lake increases from 17.4 days under no diversion to 23.7 days with average diversion conditions. Similar trends were observed for scenarios for seasonal water residence time. Scenarios of water residence time with water diversion can be incorporated into predictive models of pelagic vs. littoral contributions to fish diet. This will be completed in Year 3.

One assumption that is being made is that pelagic zooplankton derives all energy from autochthonous production of phytoplankton. However, recent research has shown that in small lakes 20% to 40% of zooplankton carbon can be derived from terrestrial sources (e.g., Cole et al. 2011). Such carbon is first processed by bacteria, which are then consumed by Protozoa and small metazoans that are incorporated into the diet of zooplankton – the so called 'microbial loop' (Moss 2010). To test this assumption, we modeled the relationship between the $\delta^{13}C$ signatures of zooplankton by lake volume across all lakes and reservoirs sampled in Year 1 and Year 2. We observed a strong negative relationship between lake volume and the $\delta^{13}C$ signature of zooplankton, which suggests that terrestrial carbon (and/or carbon from lake macrophytes) increasingly contributes to zooplankton production as lake volume declines (Figure 47). This result may help explain some of the variability with short lake water residence time shown in Figure 46. It also suggests that declines in pelagic production with decreased water residence may be buffered in small lakes by large contributions of terrestrial carbon to zooplankton.

Table 33. Total mean contributions of pelagic vs. littoral sources to Cutthroat Trout, Rainbow Trout, and Dolly Varden diets in Upper Campbell and Lower Campbell reservoirs, and Beavertail, Gooseneck, Middle Quinsam, Snakehead and Upper Quinsam lakes. Pelagic and littoral contributions are derived from direct (via invertebrates) and indirect (via prey fish) sources.

Year	Waterbody	Consumer	Pelagic Contributions			Littoral Contributions		
			Direct	Indirect	Total	Direct	Indirect	Total
2014	Upper Campbell Reservoir	Cutthroat Trout	0.03	0.23	**0.26**	0.18	0.56	**0.74**
		Rainbow Trout	0.25	0.19	**0.44**	0.31	0.25	**0.56**
	Gooseneck Lake	Cutthroat Trout	0.19	0.02	**0.21**	0.37	0.42	**0.79**
		Dolly Varden	0.36	0.01	**0.37**	0.47	0.16	**0.63**
	Middle Quinsam Lake	Cutthroat Trout	0.06	0.07	**0.14**	0.40	0.47	**0.86**
2015	Lower Campbell Reservoir	Cutthroat Trout	0.03	0.05	**0.08**	0.41	0.51	**0.92**
		Rainbow Trout	0.20	0.03	**0.23**	0.59	0.18	**0.77**
	Upper Quinsam Lake	Cutthroat Trout	0.12	0.11	**0.23**	0.29	0.48	**0.77**
	Beavertail Lake	Cutthroat Trout	0.13	0.09	**0.22**	0.40	0.38	**0.78**
		Rainbow Trout	0.16	0.05	**0.21**	0.56	0.23	**0.79**
		Dolly Varden	0.17	0.05	**0.22**	0.56	0.22	**0.78**
	Snakehead Lake	Cutthroat Trout	0.12	0.12	**0.24**	0.50	0.26	**0.76**

Figure 46. Percent pelagic contribution to Cutthroat Trout diet by A) annual and B) seasonal water residence time (days) across all study lakes and reservoirs sampled in 2014 and 2015. Data are fit with the model: %Pelagic.CT = a×WRT^b.

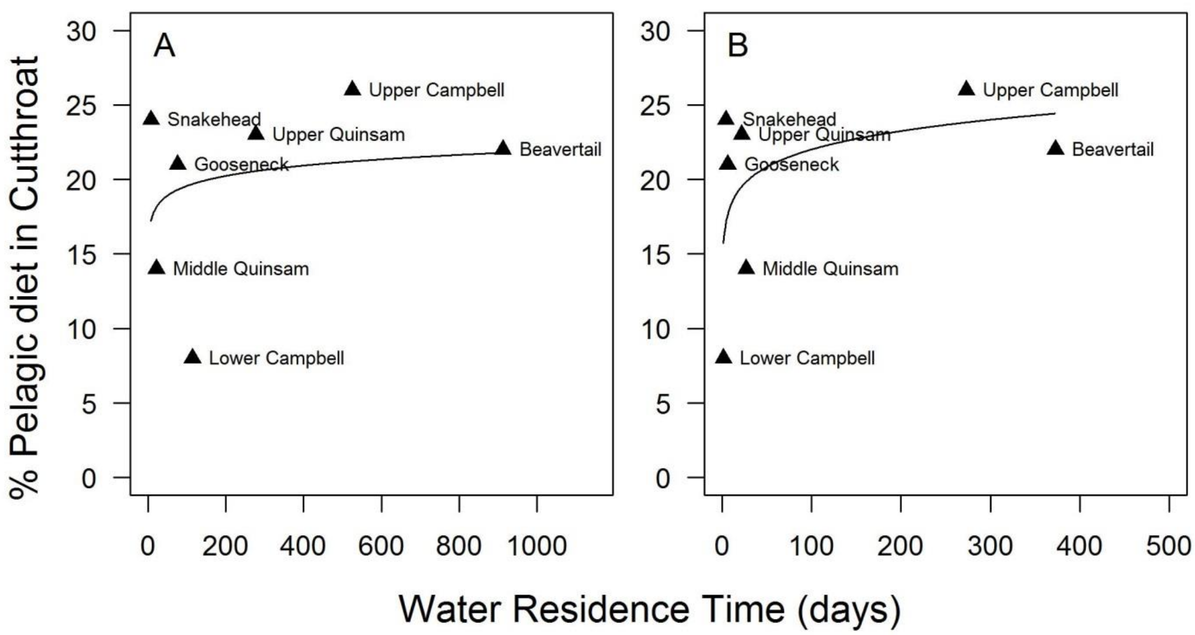


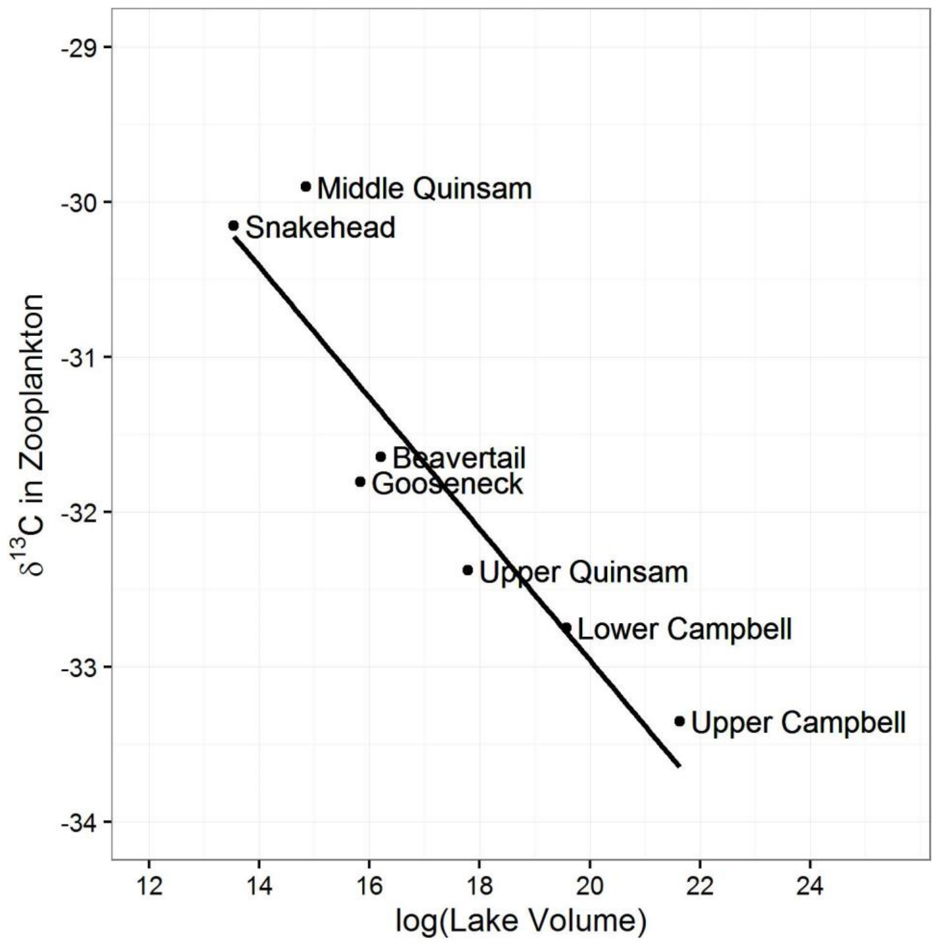
Table 34. Estimates of annual water residence time (days) under different water diversion scenarios for Gooseneck Lake (GL), Middle Quinsam Lake (MQ) and Snakehead Lake (SN).

	2012			2013			2015			Average		
	GL	MQ	SN	GL	MQ	SN	GL	MQ	SN	GL	MQ	SN
Scenario 1:												
Significant Diversion (90/10)	36.2	93.2	3.6	4.9	32.4	3.5	5.9	39.0	3.9	36.8	54.9	3.7
Scenario 2:												
Average Diversion (35/65)	74.5	40.3	7.0	10.0	14.0	6.9	12.1	16.9	7.8	75.7	23.7	7.2
Scenario 3												
Minor Diversion (10/90)	143.9	32.0	12.6	19.5	11.1	12.4	23.4	13.4	13.9	146.3	18.8	13.0
Scenario 4:												
No Diversion (0/100)	228.8	29.5	18.4	31.2	10.2	18.1	37.2	12.4	20.3	233.1	17.4	18.9

Table 35. Estimates of seasonal water residence time (days) under different water diversion scenarios for Gooseneck Lake (GL), Middle Quinsam Lake (MQ) and Snakehead Lake (SN).

	2012			2013			2015			Average		
	GL	MQ	SN	GL	MQ	SN	GL	MQ	SN	GL	MQ	SN
Scenario 1:												
Significant Diversion (90/10)	2.8	43.6	2.0	3.6	55.4	2.3	4.2	65.0	2.8	3.5	54.7	2.4
Scenario 2:												
Average Diversion (35/65)	4.9	20.7	3.4	6.2	26.5	3.9	7.3	30.9	4.7	6.1	26.0	4.0
Scenario 3												
Minor Diversion (10/90)	11.4	14.7	7.1	14.4	18.9	8.1	17.0	22.0	9.9	14.3	18.5	8.4
Scenario 4:												
No Diversion (0/100)	18.5	13.6	10.4	23.2	17.5	11.9	27.7	20.3	14.6	23.1	17.1	12.3

Figure 47. Average $\delta^{13}C$ signatures of zooplankton by lake volume (natural log scale) across all lakes and reservoirs sampled in Year 1 and Year 2 of JHTMON-5. Less negative $\delta^{13}C$ signatures are associated with increased contribution of terrestrial-derived carbon.



4. CONCLUSIONS

4.1. Reservoir Levels and Benefits to Fish Populations

The initial hypothesis that top fish consumers have a reduced littoral contribution to diet in Upper Campbell Reservoir compared to Lower Campbell Reservoir was confirmed by this analysis. Estimates of littoral-derived production were 74% at Upper Campbell Reservoir and 92% at Lower Campbell Reservoir. Upper Campbell Reservoir has greater fluctuations in water levels than Lower Campbell Reservoir, which may reduce littoral production for fish. However, there are other factors that may explain the patterns we observed. First, Lower Campbell Reservoir may have a higher proportion of littoral habitat than Upper Campbell Reservoir, which would produce fish diets that are more littoral. In Year 3, estimates of the proportion of littoral to pelagic habitat (e.g., shoal area in each lake) will be compiled for all reservoirs and diversion lakes based on bathymetric data and incorporated into the analysis. Second, the seasonal water residence time at Lower Campbell Reservoir was found to be one of the shortest among all study lakes, and possibly shorter than Snakehead Lake, which is less than 1/100th its area. This is due to the large inflows and outflows of water, which create conditions during stratification that may limit pelagic production due to direct flushing effects. Therefore, despite that we observed reduced littoral contribution to diet in Upper Campbell Reservoir compared to Lower Campbell Reservoir, it is not yet certain if water level fluctuations cause reduced littoral dependence in Upper Campbell Reservoir.

There is strong evidence that the pelagic contribution to Rainbow Trout diet is higher than the pelagic contribution to Cutthroat Trout diet in both Upper Campbell and Lower Campbell reservoirs. Rainbow Trout appear to be eating a higher proportion of zooplankton and a lower proportion of juvenile trout and Sculpin compared to the highly piscivorous Cutthroat Trout. The fish that Cutthroat Trout are eating (e.g., Sculpin, juvenile trout) are themselves highly dependent on littoral resources. Cutthroat Trout also eat a significant proportion of littoral invertebrates from various sources (e.g., benthic, terrestrial and stream invertebrates).

In Year 3, it will be important to obtain estimates of fish diets for John Hart Reservoir to allow contrasts in the sources of production across Upper Campbell, Lower Campbell and John Hart reservoirs that differ in operations. The large dependence on littoral resources across all sampled lakes and reservoirs hints that Cutthroat Trout may have a relatively fixed dependence on littoral production with limited ability to switch to a pelagic-dominated diet. Further analyses will be conducted in Year 3 to integrate data on the proportion of littoral habitat and estimates of water residence time and how they may affect estimates of littoral versus pelagic production to Cutthroat and Rainbow Trout.

4.2. Water Residence Time and Lake Productivity

Shorter water residence time is hypothesized to decrease the pelagic contribution to fish diets in diversion lakes and ultimately decrease fish production. There was some evidence to support this hypothesis, although a more complete synthesis analysis will be conducted after Year 3 of sampling. Stable isotope data indicate that the general food web structure is fairly similar among lake systems despite large differences in waterbody size and operational influences. The total littoral contribution to Cutthroat Trout diet is higher than the contribution from pelagic sources across all lakes. Estimates for pelagic sources of production to Cutthroat Trout in the lakes and reservoirs ranged from 14% at Middle Quinsam Lake to 24% at Snakehead Lake and 8% in Lower Campbell Reservoir to 26% in Upper Campbell Reservoir. These estimates for pelagic contributions to diet were compared to estimates of annual and seasonal water residence time for each lake (Figure 46). An asymptotic relationship was found with reduced, but variable, pelagic contribution to diet observed when seasonal water residence is less than ~25 days. Interestingly, this time period is consistent with average generation times of crustacean zooplankton at mean water temperatures of ~15°C (Gillooly 2000).

Gooseneck and Snakehead lakes receive water from the Quinsam River diversion, whereas Middle Quinsam Lake has water diverted upstream of the lake and thus is a donor lake. The original prediction was that water residence time will be shorter in Gooseneck and Snakehead lakes (receiving lakes) than Middle Quinsam Lake (donor lake), which could result in decreased zooplankton production. Seasonal water residence time was indeed found to be lower in both Gooseneck (6.1 days) and Snakehead (4.1 days) lakes compared to Middle Quinsam Lake (26.7 days). However, the annual residence time at Gooseneck Lake (75.6 days) was estimated to be over three times longer than the annual residence time at Middle Quinsam Lake (21.1 days). The estimates of % pelagic production at these lakes do not always follow the predictions of water residence time; a greater pelagic source of production (ultimately from plankton) in both Gooseneck (21%) and Snakehead (24%) lakes was observed in Cutthroat Trout diets compared to Middle Quinsam Lake (14%). Absent from this current analysis, however, is an estimate for the total littoral habitat in each lake relative to total lake area. For example, Middle Quinsam Lake has a very similar surface area to Gooseneck Lake but is much shallower and has a greater percentage of its lake area dominated by shoal habitat, with well-established macrophyte communities. Therefore, it would be expected that % pelagic production would be lower in Middle Quinsam Lake than Gooseneck Lake, independent of water residence time.

We observed a strong negative relationship between lake volume and the $\delta^{13}C$ signature of zooplankton (Figure 47). This suggests that terrestrial (and/or macrophyte) carbon increasingly contributes to zooplankton production as lake volume declines, which may explain some of the variability with short lake water residence time shown in Figure 46. It also suggests that declines in pelagic production with decreased water residence times may be buffered in small lakes by large contributions of terrestrial carbon to zooplankton production.

In Year 3, a more complete synthesis analysis is planned. A significant goal for Year 3 is to add more lakes into the analysis and to finalize a model between water residence time and pelagic vs. littoral contribution to diet as shown in Figure 46. However, % pelagic production will be modeled as a function of water residence time and % littoral habitat in the same model. This will enable predictions of how different water diversion scenarios shown in Table 34 and Table 35 affect pelagic contributions to fish diets in the diversion lakes while controlling for the amount of littoral habitat available. Further models are also possible using different response variables. For example, the biomass of zooplankton shown in section 3.1.1 could be regressed against water residence time and estimates of terrestrial carbon contribution to zooplankton to test if zooplankton production declines with reduced water residence time. Another synthesis model that could be developed is the CPUE data for different fish species shown in section 3.2. Does Cutthroat Trout population density differ across lakes with different water residence times? A more complete synthesis analysis will be conducted in Year 3 once all of the lakes and reservoirs have been sampled.

5. RECOMMENDATIONS

1. The following lakes will be sampled in Year 3: Gray Lake, Brewster Lake and Whymper Lake. In addition, we propose to sample John Hart Reservoir, although it is necessary to confirm whether field crews can access the reservoir given the current works that are underway to replace the generating station.
2. Stable isotope analysis of nitrogen and carbon, combined with the use of Bayesian mixing models, was used successfully in Year 2 to understand the diets of species or functional groups in lake food webs, and ultimately to provide estimates of total littoral and pelagic contributions to diets of adult Cutthroat Trout, Rainbow Trout and Dolly Varden. These methods will be continued in Year 3 to address the management questions posed in the TOR.
3. The amount of littoral habitat in each lake will influence the proportion of fish diets derived from littoral versus pelagic sources. In Year 3, we recommend that the relative area of littoral habitat in each study lake is estimated. This can then be used as an independent variable in models to predict the relative pelagic contributions to fish diets. This desktop exercise will require analysis of bathymetry data that have been collected, or will be collected in Year 3 (Grey and Whymper lakes). We anticipate that this task can be completed within the scope of the outstanding water residency time analysis.
4. The lake levels of the three reservoirs are monitored continuously by BC Hydro. We recommend that metrics relating to the frequency and range of water level fluctuations be identified and compared across the three reservoirs. We propose to integrate this into the scope of the final data analysis tasks.
5. We recommend undertaking invertebrate sampling as planned, which will include three separate trips to each lake in June, July and August. Minnow traps should be deployed during each of these trips with the primary aim of catching Sculpin spp. and reducing effort necessary in the main fish sampling trip in late August or early September. This trip will include gill netting, and we also recommend that trap netting is undertaken with the aim of sampling Threespine Stickleback, Sculpin spp., and juvenile trout. We do not recommend that a separate trip is undertaken in June to sample Threespine Stickleback; this was undertaken in Year 2 and was unsuccessful.
6. There is high overlap in the $\delta^{13}C$ and $\delta^{15}N$ isotope signatures of the three littoral invertebrate prey sources (benthic/littoral, stream and terrestrial invertebrate groups). In Year 3, we recommend that the Bayesian isotope mixing model be simplified to fewer sources by aggregating the three littoral invertebrate prey sources into one prey group.
7. As undertaken in Year 2, we recommend that all zooplankton samples collected in Year 3 are enumerated so an estimate of zooplankton biomass can be made for each lake. This will involve collecting body length measurements for a sub-sample of individuals to estimate

mean body mass. This is important because zooplankton biomass provides a direct measure of food availability to fish and we aim to examine relationships between this variable and lake water residence time. We plan to integrate this work into the existing scope of the zooplankton sample analysis. In addition, we recommend that zooplankton sample analysis is undertaken after each sampling trip, rather than at the end of the field season. This will break up the work, which will aid scheduling and allow for preliminary analysis of results before the sampling is completed.

8. We recommend that lake water temperature profiles are collected during each zooplankton sampling trip. This will provide data regarding how the thermocline depth changes throughout the growing season, which will support the water residency time analysis.

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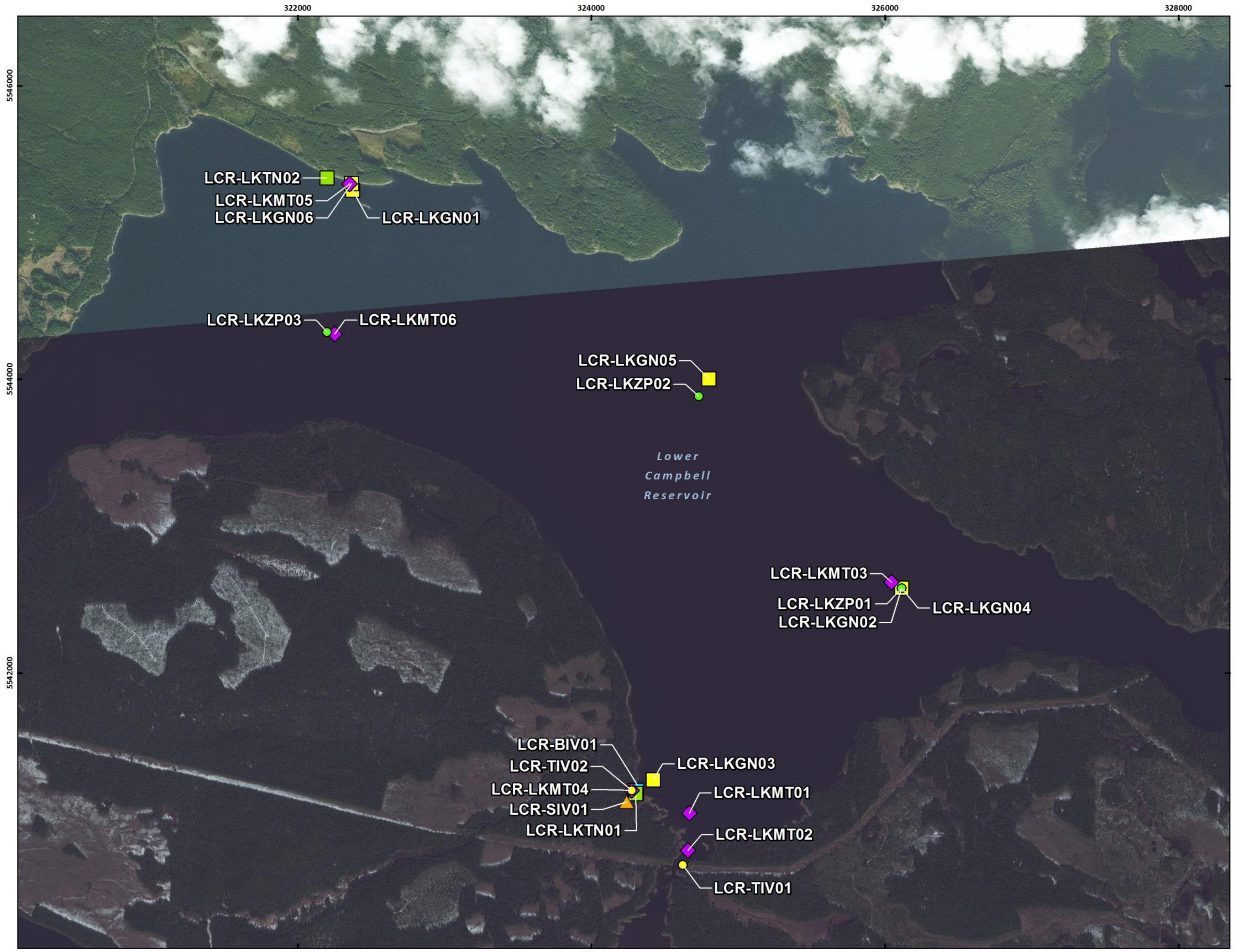
PROJECT MAPS

JHTMON Campbell River Water Use Plan

Lower Campbell Reservoir Lake Sampling Locations



Legend

Sample Sites

- Zooplankton Sampling
- Minnow Trapping - Fish
- Gill Netting - Fish
- Trap Netting - Fish
- Terrestrial Invertebrate Sampling
- Stream Invertebrate Sampling
- Benthic Invertebrate Sampling

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 100 200 400 600 800 1,000 m

Scale: 1:25,000

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Coordinate System: NAD 1983 UTM Zone 10N

ECOFISH RESEARCH

Map 2.

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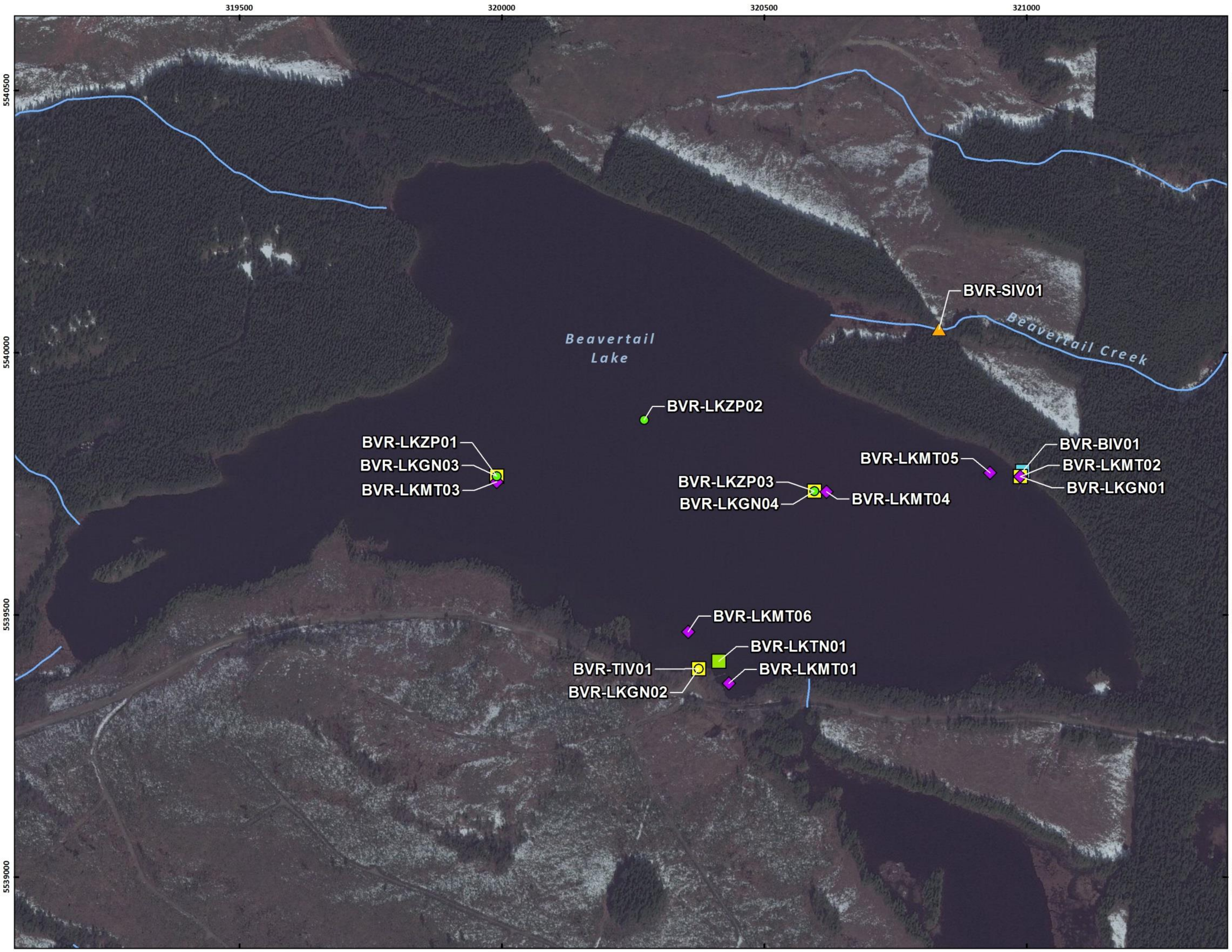

JHTMON Campbell River Water Use Plan

Beavertail Lake
Lake Sampling Locations

Legend
Sample Sites

- Zooplankton Sampling
- Minnow Trapping - Fish
- Gill Netting - Fish
- Trap Netting - Fish
- Terrestrial Invertebrate Sampling
- Stream Invertebrate Sampling
- Benthic Invertebrate Sampling

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

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Scale: 1:7,000

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ECOFISH RESEARCH

Map 3.

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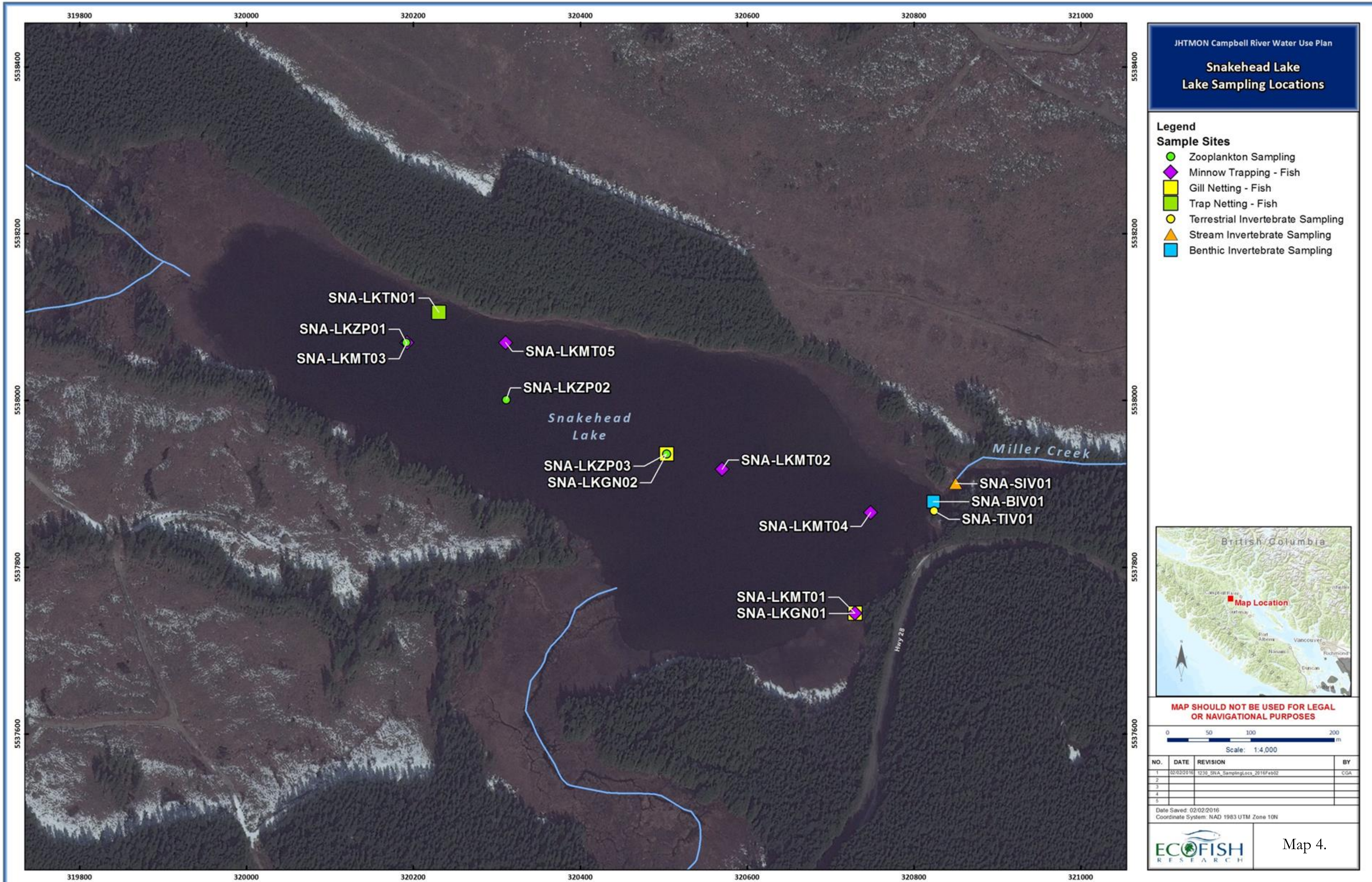

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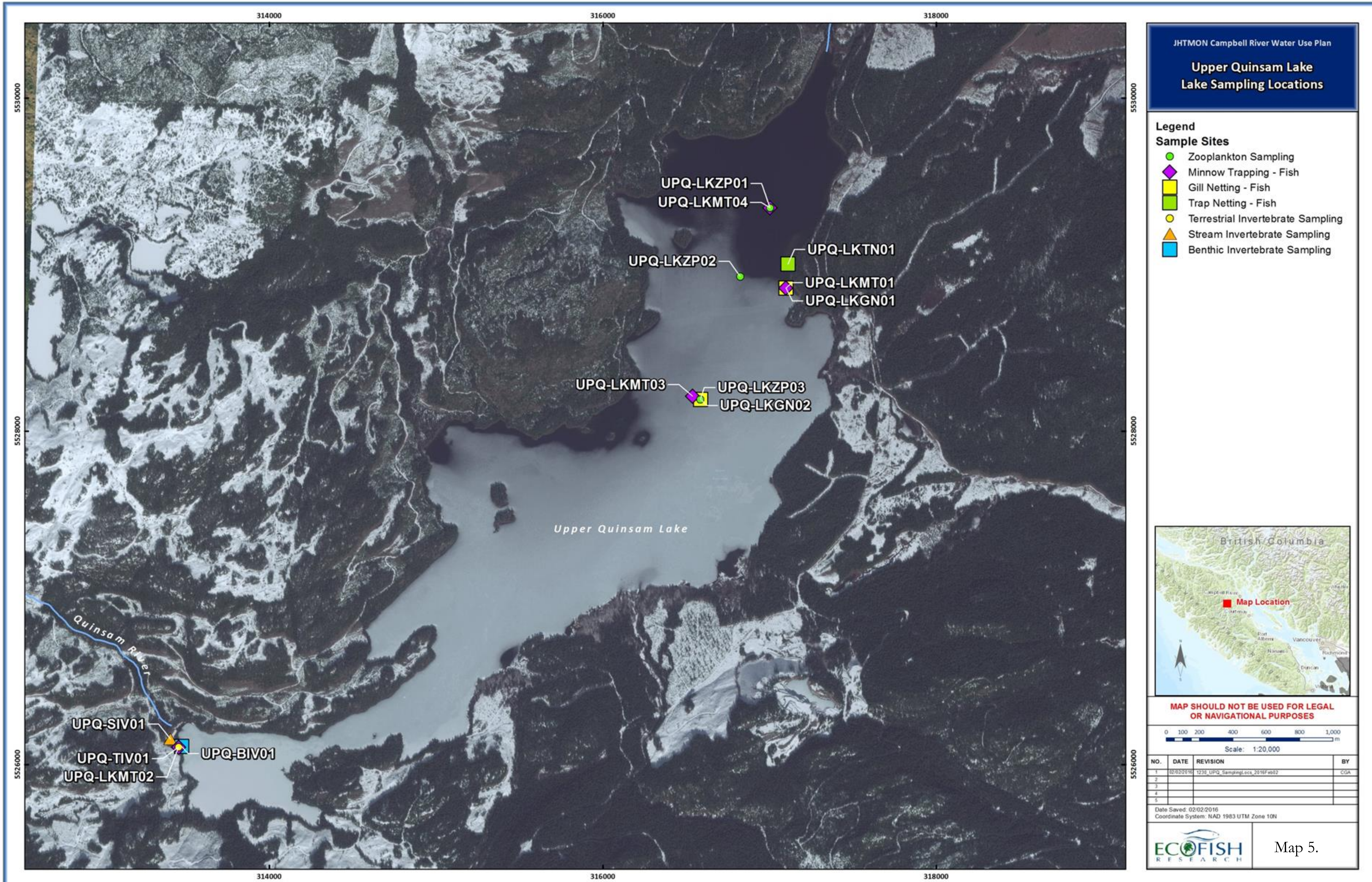

Map 5.

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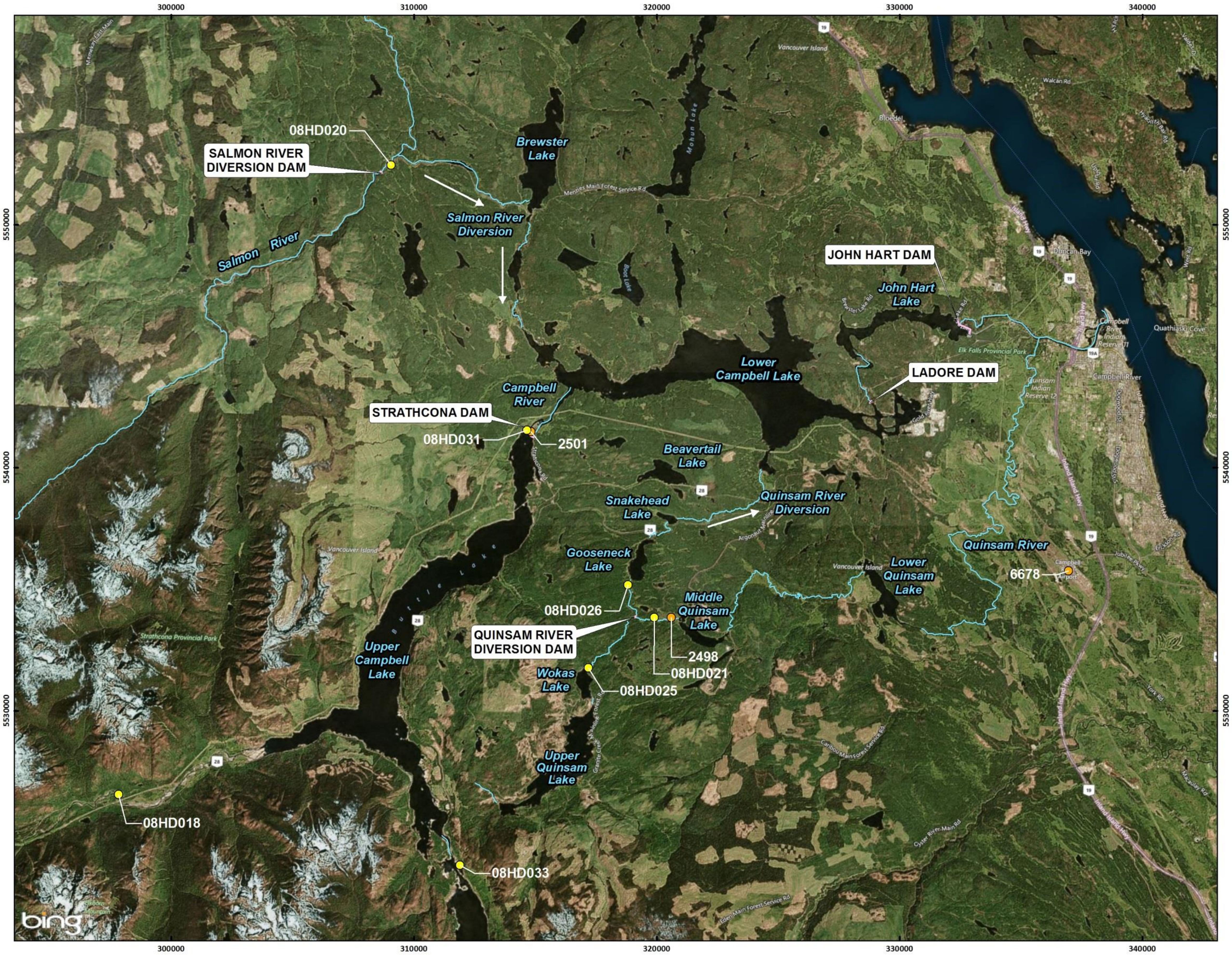
JHTMON Campbell River Water Use Plan

Hydrometric and Climate Stations



Legend

- Hydrometric Gauge
- Climate Station
- Dam
- Stream

British Columbia

Map Location

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 0.5 1 2 3 4 5 km

Scale: 1:150,000

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Date Saved: 01/04/2016
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ECOFISH RESEARCH

Map 6.

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