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 3

Reference: JHTMON-5

Year 3 Annual Monitoring Report

Study Period: April 1, 2016 to May 31, 2017

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

October 23, 2017

JHTMON-5: Littoral versus Pelagic Fish Production Assessment

Year 3 Annual Monitoring Report




Prepared for:

BC Hydro Water License Requirements
6911 Southpoint Drive, 11th Floor
Burnaby, BC, V3N 4X8

October 23, 2017

Prepared by:

Laich-Kwil-Tach Environmental Assessment Ltd. Partnership

and

Ecofish Research Ltd.

Published by Ecofish Research Ltd., Suite F, 450 8th St., Courtenay, B.C., V9N 1N5

Citation:

Hocking, M., J. Abell, N. Swain, N. Wright, and T. Hatfield. 2017. JHTMON5 – Littoral versus Pelagic Fish Production Assessment. Year 3 Annual Monitoring Report. Consultant's report prepared for BC Hydro by Laich-Kwil Tach Environmental Assessment Ltd. Partnership and Ecofish Research Ltd, October 23, 2017.

Certification: *stamped version on file*

Senior Reviewer:

Todd Hatfield, Ph.D., R.P.Bio. No. 927

Senior Environmental Scientist

Technical Leads:

Morgan Hocking, Ph.D., R.P.Bio. No. 2752

Senior Fisheries Biologist

Jonathan Abell, Ph.D.

Environmental Scientist, Limnologist

Noel Swain, M.Sc.

Fisheries Biologist

Nicole Wright, Ph.D., A.Sc.T, PWS

Hydrologist and Wetland Scientist

Todd Hatfield, Ph.D., R.P.Bio. No. 927

Senior Environmental Scientist

Disclaimer:

EXECUTIVE SUMMARY

Water Use Plans (WUPs) were developed for all of BC Hydro's hydroelectric facilities through a consultative process. As the Campbell River WUP process reached completion, a number of uncertainties remained with respect to the effects of BC Hydro operations on aquatic resources. Monitoring is now being undertaken to address outstanding management questions that relate to assumptions made during the WUP development process. The *Upper Campbell, Lower Campbell, John Hart Reservoirs and Diversion Lakes Littoral versus Pelagic Fish Production Assessment* (JTHMON-5) is part of this wider monitoring of the Campbell River WUP. JTHMON-5 is designed to assess the extent to which fish production is driven by littoral versus pelagic production, and how this relates to BC Hydro operations. An overall summary of the study objective, management questions, hypotheses and results is presented in Table 1.

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 that cause reduced littoral production. The FTC also hypothesized that reduced water residence time of the diversion lakes as a result of the BC Hydro diversion operations could negatively impact pelagic productivity by flushing plankton.

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. In addressing the management 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 versus 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 versus pelagic energy flows to diversion lake fish populations.

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

This report presents data from the final year (Year 3) of the stable isotope analysis of food webs component of the program. The second component (testing $H_0 3$) is intended to be addressed 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 versus pelagic sources. The primary species of interest 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*) and Kokanee (*Oncorhynchus nerka*) was also completed. Gill netting, trap netting and minnow trapping was completed in June through October of 2014, 2015 and 2016 to obtain representative tissue samples from Cutthroat Trout, Rainbow Trout, Dolly Varden, Kokanee and their prey fish including Threespine Stickleback (*Gasterosteus aculeatus*), Sculpin spp. (*Cottus* spp.), and juvenile trout (*Oncorhynchus* spp.) from Upper Campbell Lower Campbell and John Hart reservoirs and eight diversion or control lakes in the Campbell River WUP system. 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 all lakes and reservoirs. Invertebrates were sorted and counted in the laboratory to order and family by Ecofish staff or by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society).

To test hypothesis $H_0 1$, littoral energy contributions to fish diets were compared across the three reservoirs that differ in BC Hydro operations. Upper Campbell Reservoir, which experiences the greatest fluctuations in water levels, was sampled in Year 1 and in Year 3. Lower Campbell Reservoir experiences intermediate fluctuations in water levels and was sampled in Year 2. John Hart Reservoir, which experiences the lowest fluctuations in water levels, was sampled in Year 3.

To test hypothesis $H_0 2$, pelagic energy contributions to fish diets were compared across eight lakes that differ in water diversion and water residence time. Taken together, the eight lakes sampled across Years 1 to 3 include five recipient lakes (Gooseneck, Snakehead, Brewster, Gray and Whymper lakes), one donor lake (Middle Quinsam Lake) and two control lakes (Beavertail and Upper Quinsam lakes) that are part of either the Quinsam River or Salmon River diversions. For each waterbody, annual and seasonal water residence times were calculated, with seasonal residence time accounting for the occurrence of thermal stratification (and thus a reduced mixed layer) during the growing season.

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 773 samples of invertebrates and fish were analyzed. Invertebrates were analyzed as whole individuals, while most fish samples were fin clips. The relative contributions of pelagic versus littoral sources to Cutthroat Trout and Rainbow Trout 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.

The food webs consisting of fish and invertebrates were similar among all eleven lakes and reservoirs sampled. Large Cutthroat Trout 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.

There was clear diet and habitat separation between Cutthroat Trout and Rainbow Trout in the lakes and reservoirs of the Campbell system. Cutthroat Trout were caught in higher abundance in littoral habitats than in pelagic habitats, were more piscivorous, and had little contribution of zooplankton to their diets, as evidenced by stomach contents and stable isotope analysis. In contrast, Rainbow Trout were caught in higher abundance in pelagic habitats, were not piscivorous, and had a higher dependence on zooplankton production in their diet. Somewhat surprisingly, both species have a high dependence on terrestrial invertebrates in their diets and, despite the large lake areas, are supported by littoral/terrestrial production often to a greater extent than pelagic production. Even Rainbow Trout, which are known to be highly planktivorous, had a roughly 50:50 contribution of terrestrial invertebrates and zooplankton in their diets from the Campbell reservoirs. In addition, the prey fish that Cutthroat Trout eat (Threespine Stickleback, Sculpin spp., and juvenile trout) also have a high dependence on terrestrial invertebrates in their diets. Together, these observations predict that Cutthroat Trout have the potential to be more affected by drawdown than Rainbow Trout due to their increased dependence on littoral resources. It also raises the question of whether drawdown influences the terrestrial linkage to littoral production via the input of terrestrial vegetation and invertebrates.

Upper Campbell Reservoir had the lowest littoral contributions to Cutthroat Trout diets, followed by Lower Campbell Reservoir, and then John Hart Reservoir. This follows the predicted effects of drawdown, which provides some support for the rejection of the null hypothesis H_01, and implies an effect from water management. In Upper Campbell Reservoir, 75% and 86% of Cutthroat Trout (age >2+) diets were estimated to be derived from littoral energy sources in 2014 and 2016, respectively. In Lower Campbell Reservoir in 2015, 87% of Cutthroat Trout diets were estimated to be derived from littoral energy sources, while in John Hart Reservoir in 2016, 89% of Cutthroat Trout diets were estimated to be derived from littoral energy sources.

In contrast, Rainbow Trout had higher littoral contributions to diet in Upper Campbell Reservoir and Lower Campbell Reservoir compared to John Hart Reservoir, which is opposite to predicted effects from drawdown. In Upper Campbell Reservoir, 64% and 60% of Rainbow Trout (age >2+) diets were estimated to be derived from littoral energy sources in 2014 and 2016, respectively. In Lower Campbell Reservoir in 2015, 61% of Rainbow Trout diets were estimated to be derived from littoral energy sources, while in John Hart Reservoir in 2016, 45% of Rainbow Trout diets were estimated to be derived from littoral sources. These observations suggest that drawdown could result in tradeoffs in production between Cutthroat Trout and Rainbow Trout, although the low sample size (three waterbodies) limits the strength of inference that can be drawn.

This analysis provides some support to the hypothesis that top fish consumers (Cutthroat Trout) have a reduced littoral contribution to diet with greater fluctuations in water levels from drawdown. However, there are other factors that may also contribute to the patterns we observed. First, the proportion of littoral habitat in Upper Campbell Reservoir is less than Lower Campbell or John Hart reservoirs, with only 12.8% of the reservoir comprising shoal habitat <6 m. Further, the seasonal water residence time at Lower Campbell and John Hart reservoirs was found to be two of the shortest among all study lakes, and possibly shorter than Snakehead Lake, which is less than 1/100th the area of Lower Campbell Reservoir. 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 due to high transport losses (flushing) of plankton.

Water residence time was estimated for all JHTMON-5 waterbodies and incorporated into across-lake models to predict energy flows to Cutthroat Trout and Rainbow Trout. The water residence time was calculated by dividing the effective volume of the basin by the annual average outflow rate. 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) multiplied by the proportion of year that the lake is 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. Annual and seasonal water residence time was also calculated for several water diversion scenarios including average diversion during pre-WUP (October 1997 – October 2004), and post-WUP (October 2012 – October 2016) periods.

A model selection approach was used to determine the most important lake variables to predict pelagic energy flows to Cutthroat Trout and Rainbow Trout across all sampled lakes and reservoirs. Lake variables included annual and seasonal water residence time, lake volume and % shoal habitat, which is a measure of the amount of shallow littoral habitat relative to total lake volume.

The pelagic contribution to Cutthroat Trout diets increased with annual water residence time and with the proportion of littoral habitat relative to pelagic habitat (% shoal habitat) in each waterbody. The top model for Cutthroat Trout included both annual water residence time and % shoal habitat, which was more than two times more likely than the second ranked model. Seasonal water residence time and lake volume were not strong predictors of the pelagic energy flows to diet in Cutthroat Trout. This suggests that the contribution of zooplankton to Cutthroat Trout diets increases with longer annual water residence time, which is a rejection of the null hypothesis $H_0 2$, and therefore implies an effect from water management. However, the underlying causes of these results remain uncertain because the sample size (11 waterbodies) is relatively small.

For Rainbow Trout, the top model predicting pelagic contribution to diet was the null model. The pelagic energy sources to Rainbow Trout diets were not influenced by annual or seasonal water residence time, lake volume or % shoal habitat. This indicates that the null hypothesis $H_0 2$ should be retained for Rainbow Trout, and that Rainbow Trout are less influenced by water diversion than Cutthroat Trout. Given the high importance of zooplankton as a food source for Rainbow Trout, it was expected that this species would be more sensitive than Cutthroat Trout to water management actions that reduce water residence time sufficiently to cause high transport losses (flushing) of zooplankton. A caveat is that Rainbow Trout were only detected in seven of the 11 waterbodies that were sampled, yielding a small sample size and less power to detect effects of water residence time.

Lake productivity was also analyzed across all lakes and reservoirs sampled in JHTMON-5 using zooplankton biomass and Cutthroat Trout CPUE and Rainbow Trout CPUE as response indices. Zooplankton biomass was positively predicted by % shoal habitat in each waterbody and not annual or seasonal water residence time or lake volume. Cutthroat Trout CPUE was positively predicted by annual water residence time and % shoal habitat, which are the same predictor variables included in the top model for pelagic energy flows to Cutthroat Trout. This suggests that water management through diversion may affect Cutthroat Trout abundance. For Rainbow Trout CPUE, the top model included only lake volume. Rainbow Trout catches decreased with decreasing lake size. Overall, it seems that Cutthroat Trout out-compete Rainbow Trout in the smaller and shallower lakes where there is less potential for the two species to occupy separate niches. In these waterbodies, Cutthroat Trout increase their use of pelagic energy sources by increasing their diet of zooplankton.

A strong positive relationship was also observed between % shoal habitat and the $\delta^{13}C$ signature of zooplankton. This suggests that terrestrial (and/or macrophyte) carbon increasingly contributes to zooplankton production as the amount of shallow littoral habitat increases relative to total lake volume. This is also supported by the observation that zooplankton biomass increases with % shoal habitat in each waterbody, and 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. These interactions between water residence time, trophic state and terrestrial contributions to pelagic productivity remain an uncertainty.

Scenarios of annual water residence time with water diversion were incorporated into the top statistical model predicting energy flows to Cutthroat Trout to illustrate how water diversion may influence Cutthroat Trout diets. In the Quinsam diversion, pre-WUP (1998 to 2004) rates diversion were, on average, 36.6% of annual available flow. This decreased to 24.7% post-WUP (2013 to current). This operational change is predicted to have increased pelagic energy flows to Cutthroat Trout in Gooseneck and Snakehead lakes by around 1.5%. In Middle Quinsam Lake, pelagic energy flows to Cutthroat Trout are predicted to have decreased by 0.7%.

In the Salmon River diversion, pre-WUP average rates diversion were 34.0% of annual available flow. Post-WUP this percentage of diversion decreased to 13.1% of annual available flow. This operational change is predicted to have increased pelagic energy flows to Cutthroat Trout in Brewster and Gray lakes by up to 3%. Pelagic energy flows to Cutthroat Trout in Whymper Lake has remained similar over this period. These results suggest that changes to operations have potential to cause subtle changes to habitat use by Cutthroat Trout.

Table 1. Status of JHTMON-5 objective, management questions and hypotheses after completing the stable isotope analysis of food webs component (Year 1 to Year 3).

Study Objectives	Management Questions	Management Hypotheses	Year 3 (Final) Status
Assess the extent to which trout production is driven by littoral versus pelagic production and evaluate how this relates to BC Hydro operations	1. To what extent do stabilized reservoir levels, as affected by BC Hydro operations, benefit fish populations?	$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 versus pelagic energy flows to reservoir fish populations.	In the three reservoirs, the contribution of littoral energy sources to Cutthroat Trout diets declined with increasing drawdown. This implies an effect from water management and supports rejection of the null hypothesis $H_0$1 for Cutthroat Trout. For Rainbow Trout, the opposite trend was observed with greater contribution of littoral energy sources in Upper and Lower Campbell reservoirs compared to John Hart Reservoir. This implies that the effects of water management through drawdown will be reduced for Rainbow Trout compared to Cutthroat Trout. These conclusions are qualitative in nature due to the fact that only three reservoirs could be compared and that other reservoir factors may influence energy contributions to fish populations. When both species are present, Cutthroat Trout and Rainbow Trout occupy distinct ecological niches in the lakes and reservoirs of the Campbell River system. Cutthroat Trout are more dependent on littoral habitats while Rainbow Trout are more dependent on pelagic habitats. Cutthroat Trout strongly out-compete Rainbow Trout in the shallower lakes with limited pelagic zones (e.g., Snakehead Lake). A caveat is that terrestrial invertebrates are an important food source for both species, meaning that impacts to riparian vegetation from drawdown may adversely affect both species. Across all waterbodies, the contribution of littoral energy sources to Cutthroat Trout diets declines with increasing shallow (<6 m) littoral volume relative to total lake volume. This result is counterintuitive but likely reflects a combination of niche expansion by Cutthroat Trout in smaller lakes without Rainbow Trout and increased productivity of smaller and shallower lakes that is driven by terrestrial carbon sources and results in higher zooplankton biomass (a pelagic food source). The relative volume of shallow littoral habitat was not related to the contribution of littoral energy sources to Rainbow Trout diet. Zooplankton (a pelagic food source) makes a high contribution to Rainbow Trout diets, although this contribution is reduced when other pelagic species (e.g., Kokanee) are present.

Table 1. Continued.

Study Objectives	Management Questions	Management Hypotheses	Year 3 (Final) Status
Assess the extent to which trout production is driven by littoral versus pelagic production and evaluate how this relates to BC Hydro operations	2. What is the relationship between residence time (as affected by diversion rate) and lake productivity?	$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 versus pelagic energy flows to diversion lake fish populations.	Across all waterbodies sampled, the pelagic energy flows to Cutthroat Trout increased with annual water residence time and with % shoal habitat in each waterbody. This suggests that Cutthroat Trout feed on zooplankton to a greater extent in shallow waterbodies with longer annual water residence times, which supports rejection of the null hypothesis $H_0$2 and implies an effect of water management through diversion. The contributions of pelagic energy sources to Rainbow Trout diets were not influenced by any of the lake variables tested, including annual or seasonal water residence time, lake volume or % shoal habitat. This indicates that the null hypothesis $H_0$2 should be retained for Rainbow Trout. An important caveat however is the reduced sample size in number of lakes where Rainbow Trout were sampled, which reduces the power to detect effects of water residence time. Lake productivity was also analyzed across all lakes and reservoirs sampled in JHTMON-5 using zooplankton biomass and Cutthroat Trout catch per-unit-effort (CPUE) and Rainbow Trout CPUE as response variables. Cutthroat Trout CPUE was positively predicted by annual water residence time and % shoal habitat, which suggests that water management through diversion may affect Cutthroat Trout abundance. For Rainbow Trout, only lake volume was an important predictor of CPUE, indicating that Rainbow Trout abundance decreases with decreasing lake size. Zooplankton biomass increased with % shoal habitat in each waterbody and not annual or seasonal water residence time, which may be driven by large terrestrial carbon inputs to zooplankton in smaller lakes. Scenarios of annual water residence time with water diversion were generated and simulated with the top statistical model predicting energy flows to Cutthroat Trout. Decreases in diversion post-WUP versus pre-WUP are predicted to have increased pelagic energy flows to Cutthroat Trout by a few percent. However, these pelagic energy flows may be influenced by terrestrial contributions to pelagic bacteria and ultimately incorporated into zooplankton production. The interaction between water residence time, trophic state and terrestrial contributions to pelagic productivity remains an uncertainty.
		$H_0$3: Standing crop of pelagic bacteria is not correlated with water residence time.	This hypothesis is scheduled to be addressed in the second component of JHTMON-5.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF MAPS

LIST OF APPENDICES

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 may 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 versus 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 versus 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 (Map 1). Buttle/Upper Campbell Reservoir varies the most in water levels, whereas John Hart Reservoir water levels vary the least. Specifically, the vertical ranges of historical operations are 11.0 m in Upper Campbell Reservoir, 4.3 m in Lower Campbell reservoir, and 0.6 m for John Hart Reservoir (BC Hydro 2012). 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-5 is designed to test the assumption that improvements in littoral production lead to corresponding increases in fish production. JHTMON-4, which is complementary to JHTMON-5, is designed to investigate the effect of operations on littoral primary 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 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 versus 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 versus 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

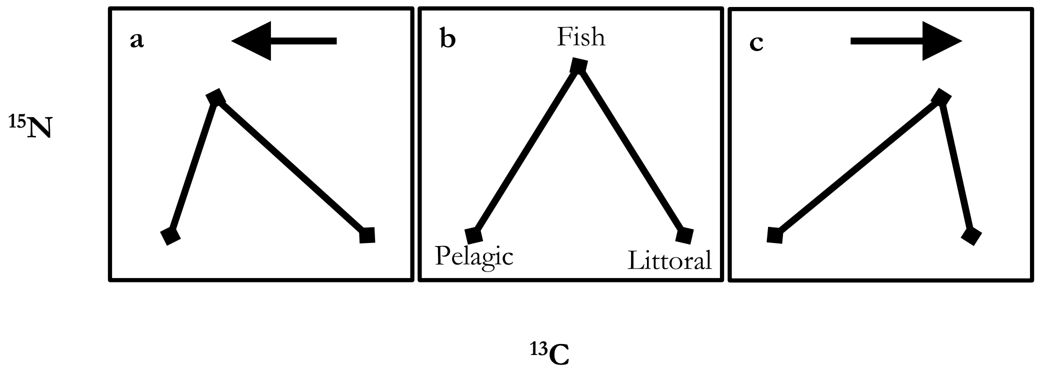
JHTMON-5 is scheduled for 10 years and has two components: stable isotope analysis of food webs in reservoirs and diversion lakes (used to address $H_0$1 and $H_0$2 above), and production estimates of pelagic bacteria in reservoirs and diversion lakes (used to address $H_0$3 above). Data from these two 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 the final year (Year 3) of the stable isotope analysis; data from all three years of sampling are used to test $H_0$1 and $H_0$2 above. Under the current TOR, sampling using stable isotope methods is scheduled for years 1, 2 and 3 of JHTMON-5. Estimates of pelagic bacteria as an indicator of pelagic productivity ($H_0$3 above) will be addressed 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 versus 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 reflect 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 versus 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 hypothetical 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 treatment effect.

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 was 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 were 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 3

Year 3 of JHTMON-5 was planned and implemented as a full sampling year based on the results and recommendations of Year 1 and Year 2. Sampling was completed for John Hart Reservoir, Brewster Lake, Gray Lake and Whymper Lake. In addition, based on existing sampling programs at Upper Campbell Reservoir within JHTMON-3 for fish and JHTMON-4 for invertebrates, it was possible to add Upper Campbell Reservoir as a fifth waterbody sampled in JHTMON-5 for Year 3. Upper Campbell Reservoir was also sampled in Year 1 and is the only waterbody to be sampled twice over the three years of sampling.

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 and in Year 3. Lower Campbell Reservoir experiences intermediate fluctuations in water levels and was sampled in Year 2. John Hart Reservoir, which experiences the lowest fluctuations in water levels, was sampled in Year 3. The goal of sampling was to contrast the littoral contribution to fish diets across these reservoirs that differ in BC Hydro operations.

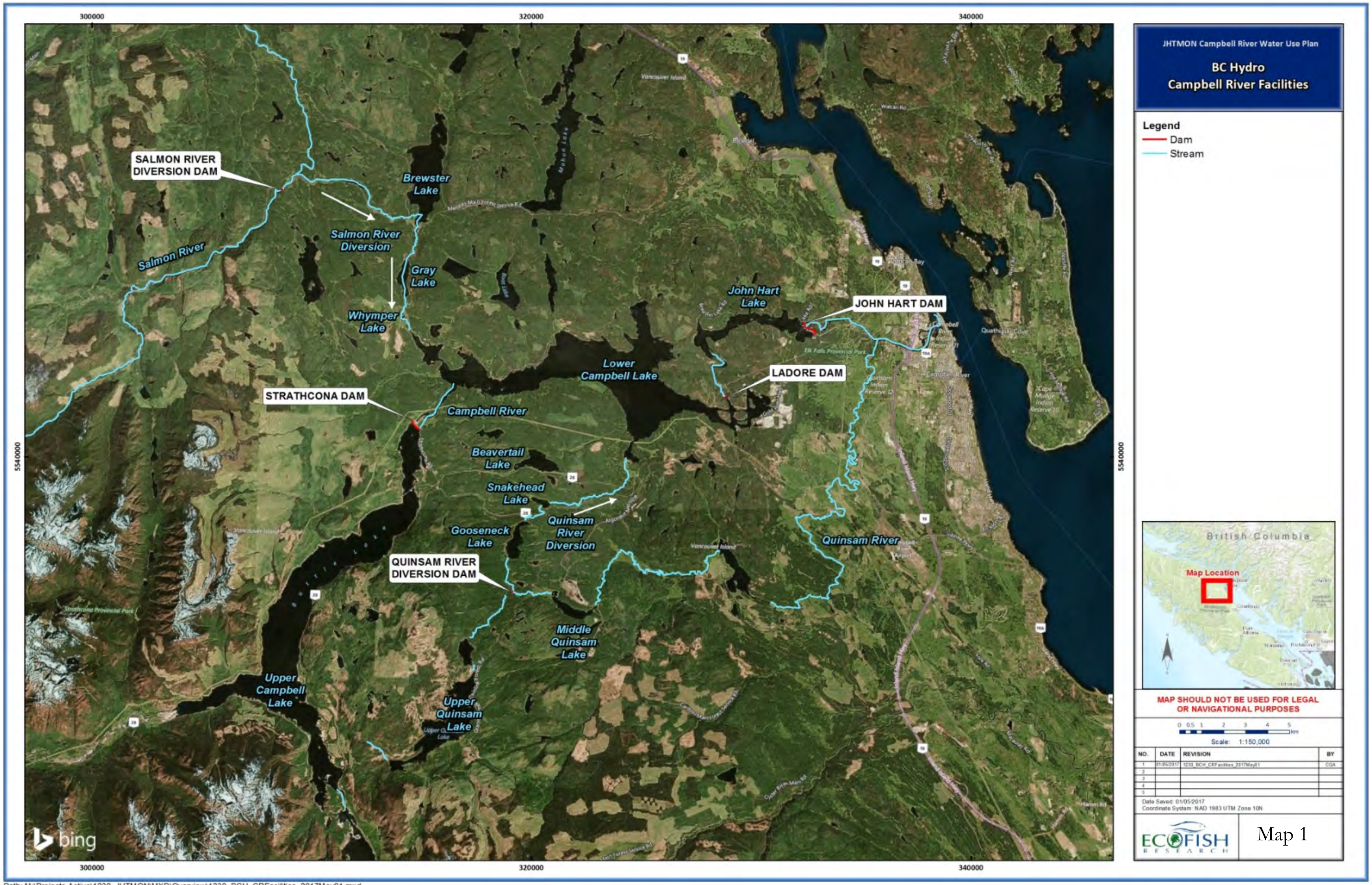
What is the relationship between residence time (as affected by diversion rate) and lake productivity? It is hypothesized that shorter water residence times decreases zooplankton production and thus the pelagic source to fish production. In Year 1, Gooseneck Lake and Middle Quinsam Lake were chosen because 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). In Year 2, Beavertail Lake, Snakehead Lake and Upper Quinsam Lake were sampled because they are also part of the Quinsam diversion system 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. In Year 3, Brewster Lake, Gray Lake and Whymper Lake were sampled, which are all part of the Salmon River diversion system. All three of these lakes are recipient lakes. Taken together, the eight lakes sampled across Years 1 to 3 include five recipient lakes (Gooseneck, Snakehead, Brewster, Gray and Whymper lakes), one donor lake (Middle Quinsam Lake) and two control lakes (Beavertail and Upper Quinsam lakes).

The selection of John Hart Reservoir, Upper Campbell Reservoir, Brewster Lake, Gray Lake and Whymper Lake in Year 3 support an examination of $H_0$1 and $H_0$2, particularly when the data from all years 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.

1.4.4. Water Residence Time

To address management question 2, estimates of water residence time were determined for all study lakes and reservoirs including Upper Campbell, Lower Campbell and John Hart reservoirs, and Beavertail, Brewster, Gooseneck, Gray, Middle Quinsam, Snakehead, Upper Quinsam and Whymper lakes. The annual 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 can 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. Therefore, both annual and seasonal water residence times were computed for all lakes and reservoirs sampled.

Map 1. Overview of BC Hydro Campbell River facilities.



Path: M:\Projects-Active\1230_JHTMON\MXD\Overview\1230_BCH_CRFacilities_2017May01.mxd

2. METHODS

2.1. Physical Characteristics of the Waterbodies

The physical characteristics of the JHTMON-5 waterbodies are presented in Table 2. Upper Quinsam Lake is situated at the highest elevation (358 m), and John Hart Reservoir is at the lowest elevation (139 m). The reservoirs have the greatest surface area. Lower Campbell Reservoir is the deepest of the waterbodies and Whymper Lake is the shallowest. John Hart Reservoir and Whymper Lake were only weakly stratified.

Table 2. Physical characteristics of the waterbodies.

Waterbody	Description	Elevation (m)	Surface Area (km^2)	Mean Water Depth[1] (m)	Max Water Depth[1] (m)	Thermocline Depth[2] (m)
Upper Campbell	Reservoir	221	68.7	12.2	39.6	20.0
Upper Quinsam Lake	Control	335	5.07	13.1	48.0	10.3
Middle Quinsam Lake	Diversion (Donor Lake)	270	0.72	4.00	14.6	7.25
Gooseneck Lake	Diversion (Receiving Lake)	290	0.78	9.80	38.0	10.0
Snakehead Lake	Diversion (Receiving Lake)	283	0.20	3.50	9.00	8.00
Beavertail Lake	Control	270	1.03	10.8	26.0	11.9
Lower Campbell	Reservoir	178	26.5	18.0	69.7	17.0
Brewster Lake	Diversion (Receiving Lake)	213	7.70	14.2	20.0	9.00
Gray Lake	Diversion (Receiving Lake)	170	0.63	10.0	18.7	5.6
Whymper Lake	Diversion (Receiving Lake)	183	0.08	3.00	5.50	-
John Hart Reservoir	Reservoir	130	3.43	12.2	22.9	-

[1] Average depth from Hatfield (2000), bathymetric maps, and data collected by Ecofish in June and July 2016.

[2] Average depth of thermocline from data collected by Ecofish in September 2015, and June and July, 2016. John Hart Reservoir and Whymper Lake have poorly defined thermoclines.

2.2. Invertebrate Sampling

2.2.1. Overview

Invertebrates are an important component of the diets of Cutthroat Trout, Rainbow Trout and Dolly Varden (McPhail 2007). Accordingly, the following invertebrate groups were sampled that represent potential diet items of the fish species of interest: zooplankton (pelagic source), littoral invertebrates (littoral source), stream invertebrates that drift into littoral areas (allochthonous littoral source), and terrestrial invertebrates that fall into littoral areas (allochthonous littoral source). The isotopic signatures of invertebrate samples were then determined to support fish diet analysis (Section 2.3.1).

2.2.2. Field Methods

2.2.2.1. Zooplankton

Zooplankton is typically the main source of secondary pelagic production in lakes, providing an important food source for tertiary consumers such as trout (e.g., Beauchamp 1990). As such, zooplankton represents an important 'food web channel' through which carbon fixed by phytoplankton (i.e., autochthonous carbon) is transferred to fish populations that occupy higher trophic levels (Sterner 2009). Studies of other lakes and reservoirs on Vancouver Island have shown that the nitrogen and carbon stable isotope signatures of zooplankton can vary substantially throughout the growing season (Matthews and Mazumder 2003), although this variability can be integrated by measuring average signatures for the growing season (Matthews and Mazumder 2005). Therefore, zooplankton was sampled three times during a single growing season (late June, late July or early August, and mid-August or early September) to obtain representative samples of zooplankton in each waterbody (Table 3). Upper Campbell Reservoir was sampled in 2014 and 2016 to provide temporal replication. The remaining ten waterbodies were sampled during a single growing season.

Zooplankton was sampled at three sites on each waterbody, with the exceptions of Middle Quinsam Lake (0.72 km^2) and Gooseneck Lake (0.78 km^2), which were two of the smallest waterbodies sampled and were only sampled at one site. Sites located in the deepest areas of each waterbody; sites were georeferenced 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 waterbody (see Map 2 to Map 12).

Zooplankton was sampled using a tow net with a 30 cm diameter aperture and a mesh size of 80 μm (Figure 2). 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 3). 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: the first was enumerated to describe taxonomic composition (Section 2.2.3.1), a component of the second was transferred to a smaller vial and sent for stable isotope analysis (Section 2.4), and the third was retained as a backup. All samples were preserved in 95% ethanol (Figure 3).

Table 3. Summary of zooplankton sampling sites.

Year	Waterbody	Site	Sampling Dates	UTM (NAD 83)			Site depth (m)	Depth sampled (m)
				Zone	E (m)	N (m)		
2014	Upper Campbell Reservoir	UCR-LKZP01	Jun-24, Jul-27, Aug-20	10U	308460	5530799	25.0	20.0
		UCR-LKZP02		10U	311771	5535849	45.0	20.0
		UCR-LKZP03		10U	309887	5534278	65.0	20.0
	Gooseneck Lake	GOO-LKZP01	Jun-25, Jul-29, Aug-19	10U	319337	5536221	17.0	10.0
	Middle Quinsam Lake	QUN-LKZP01	Jun-25, Jul-28, Aug-19	10U	322169	5332796	13.5	10.0
2015	Lower Campbell Reservoir	LCR-LKZP01	Jun-24, Jul-20, Sep-9	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
	Upper Quinsam Lake	UPQ-LKZP01	Jun-22, Jul-22, Sep-10	10U	317002	5529342	>20.0	12.0–20.0
		UPQ-LKZP02		10U	316822	5528930	>20.0	12.0–20.0
		UPQ-LKZP03		10U	316584	5528193	>20.0	14.0–20.0
	Snakehead Lake	SNA-LKZP01	Jun-25, Jul-21, Sep-11	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
	Beavertail Lake	BVR-LKZP01	Jun-23, Jul-23, Sep-9	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
2016	Upper Campbell Reservoir	UCR-LKZP01	Jun-28, Aug-2, Sep-6	10U	308460	5530799	25.0	20.0
		UCR-LKZP02		10U	311771	5535849	45.0	20.0
		UCR-LKZP03		10U	309887	5534278	65.0	20.0
	John Hart Reservoir	JHT-LKZP01	Jun-21, Jul-21, Aug-21	10U	327691	5545436	19.0	15.0
		JHT-LKZP02		10U	328718	5545643	20.0	15.0
		JHT-LKZP03		10U	330211	5546358	23.0	15.0
	Brewster Lake	BRE-LKZP01	Jun-22, Jul-20, Aug-19	10U	314914	5553854	53.0	15.0
		BRE-LKZP02		10U	315243	5552932	46.0	15.0
		BRE-LKZP03		10U	315214	5551895	25.9	15.0
	Whymper Lake	WHM-LKZP01	Jun-20, Jul-19, Aug-15	10U	314077	5546428	5.5	4.0
		WHM-LKZP02		10U	314061	5546374	5.5	4.0
		WHM-LKZP03		10U	314077	5546308	6.0	4.0
	Gray Lake	GRY-LKZP01	Jun-23, Jul-18, Aug-17	10U	314198	5548648	17.2	14.0
		GRY-LKZP02		10U	314085	5548342	24.3	14.0
		GRY-LKZP03		10U	314084	5548167	18.1	14.0

Figure 2. Zooplankton sampling at John Hart Reservoir, July 21, 2016.

Figure 3. Zooplankton samples collected at Upper Campbell Reservoir in June 24, 2014, prior to adding ethanol.

2.2.2.2. Littoral Invertebrates

Samples of macroinvertebrates were collected from the littoral zone of each waterbody on one to three occasions during a single growing season (Table 4). Littoral invertebrate sampling sites are shown on Map 2 to Map 12 (see 'BIV' sites). Littoral invertebrate sampling occurred in the same year as fish sampling on each waterbody, with the exception of Upper Campbell Reservoir in 2016. Littoral invertebrates were sampled from Upper Campbell Reservoir in 2014 to coincide with fish sampling but not again in 2016. The aim was to capture a range of taxa that were representative of the potential macroinvertebrate prey available to salmonids in the littoral zone of each waterbody. Different sampling methods were used (described below; Table 4) depending on the habitat characteristics at each waterbody, 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: effort was made to standardize the sampling effort at each waterbody; however, it was necessary to adapt this effort depending on the abundance of macroinvertebrates that were present. One to four samples were collected during each sampling event, although these samples were not true replicates as sampling effort and methods sometimes varied between samples from the same waterbody. Accordingly, multiple samples were pooled when preparing samples for stable isotope analysis (Section 2.4). Samples were preserved in the field in 95% ethanol.

Samples were collected from nine waterbodies (Table 4) using a Ponar grab ('Petite' model; Wildco, FL, USA) with an aperture of 152 mm × 152 mm. The Ponar grab was deployed in the littoral zone to a depth of 0.4–1.0 m, approximately 2–5 m from shore (Figure 4). Sediments were then placed in a clean tray and all visible invertebrates were removed with forceps and placed in a clean sample jar for preservation. The Ponar grab was deployed 1–5 times per sample, depending on macroinvertebrate abundance. Relative to effort, this method typically yielded the lowest abundance of macroinvertebrates, although it was the most suitable method at Upper and Lower Campbell reservoirs where there was limited vegetation in the littoral zone.

Samples were collected from six waterbodies (Table 4) 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 5). 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.

Samples were collected from Snakehead Lake in 2015 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 6) were picked from the lake bed.

Table 4. Littoral invertebrate sampling site details.

Year	Waterbody	Site	Sampling Dates	Sampling Method	UTM (NAD 83)		
					Zone	E (m)	N (m)
2014	Upper Campbell Reservoir	UCR-BIV01	Jul-27	Ponar grab	10U	308517	5533680
		UCR-BIV02	Aug-20	Ponar grab	10U	310962	5525914
	Gooseneck Lake	GOO-BIV01	Jul-29, Aug-19	Ponar grab	10U	318792	5535968
	Middle Quinsam Lake	QUN-BIV01	Jul-27, Aug-20	Ponar grab	10U	322652	5533051
2015	Lower Campbell	LCR-BIV01	Jul-20	Ponar grab	10U	324299	5541198
	Upper Quinsam Lake	UPQ-BIV01	Jul-22	Travelling kick and sweep	10U	313472	5526110
	Snakehead Lake	SNA-BIV01	Jul-21	Hand searches	10U	320823	5537879
	Beavertail Lake	BVR-BIV01	Jul-23	Travelling kick and sweep	10U	320992	5539774
2016	John Hart Reservoir	JHT-BIV01	Jun-21, Jul 21	Travelling kick and sweep, Ponar grab	10U	330802	5546303
	Brewster Lake	BRE-BIV01	Jun 22, Jul 20	Travelling kick and sweep, Ponar grab	10U	314645	5550864
	Whymper Lake	WHM-BIV01	Jun 20, Jul 19	Travelling kick and sweep, Ponar grab	10U	314028	5546327
	Gray Lake	GRY-BIV01	Jun 23, Jul 18	Travelling kick and sweep, Ponar grab	10U	314280	5547398

Figure 4. Left: Ponar grab sampling at Gray Lake (GRY–BIV01) on June 23, 2016; right: sample collected using the Ponar grab at Whymper Lake (WHM–BIV01) on July 19, 2016.

Figure 5. Left: kick and sweep sampling at John Hart Reservoir (JHT–BIV01) on July 21, 2016; right: resulting sample.

Figure 6. Littoral habitat sampled on Snakehead Lake (left) on July 21, 2015, with freshwater mussels (family: Unionidae) collected from lake bed (right).

2.2.2.3. Stream Invertebrates

Stream macroinvertebrates were sampled in one stream inflow or outflow to each waterbody on one to three occasions during a single growing season (Table 5). Stream invertebrate sampling sites are shown on Map 2 to Map 12 (see 'SIV' sites). Stream invertebrate sampling occurred in the same year as fish sampling on each waterbody, with the exception of Upper Campbell Reservoir in 2016. Stream invertebrates were sampled from Upper Campbell Reservoir in 2014 to coincide with fish sampling but not again in 2016. 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. Sampling was undertaken < 100 m upstream or downstream of the waterbody with the aim of sampling macroinvertebrates that provide a potential food source to lacustrine salmonids. As with littoral invertebrate sampling (Section 2.2.2.2), stream sampling was non–quantitative and was undertaken with the objectives of maximising both the numbers of individuals, and the range of taxa collected within the time available. Three samples were collected during each sampling event and all samples were preserved in the field in 95% ethanol.

Sampling methods varied depending on the substrate type and flow conditions (Table 5). Kick sampling was used at nine waterbodies (Table 5) and was the predominant method employed. Kick sampling was undertaken in riffle habitats by securing a single drift net (mesh size = 250 μm) to the stream bed using rebar and then agitating the upstream substrate for three minutes using a wading boot (Figure 7). 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.

Drought conditions in 2015 meant that there was insufficient current velocity to effectively use kick sampling at two lakes. At Upper Quinsam Lake, travelling kick and sweep sampling (see Section 2.2.2.2) 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. At Snakehead Lake, cobbles were overturned in a small riffle section with very low water depth (< 0.03 m), and individuals were picked using tweezers. Five medium cobbles (~ 0.15 m diameter) were picked per sample.

Table 5. Stream invertebrate sampling site details.

Year	Waterbody	Site	Sampling Date	Sampling Method	UTM (NAD 83)		
					Zone	E (m)	N (m)
2014	Upper Campbell Reservoir	UCR-SIV01	Jul-27	Kick sampling	10U	308517	5533680
	Gooseneck Lake	GOO-SIV01	Jul-29	Kick sampling	10U	318792	5535968
	Middle Quinsam Lake	QUN-SIV01	Jul-28	Kick sampling	10U	320848	5533816
2015	Lower Campbell Reservoir	LCR-SIV01	Jul-20	Kick sampling	10U	324238	5541125
	Upper Quinsam Lake	UPQ-SIV01	Jul-22	Travelling kick and sweep	10U	313404	5526155
	Snakehead Lake	SNA-SIV01	Jul-21	Hand search	10U	320850	5537901
	Beavertail Lake	BVR-SIV01	Jul-23	Kick sampling, hand search	10U	320833	5540045
2016	John Hart Reservoir	JHT-SIV01	Jul-21	Kick sampling	10U	327603	5546395
	Brewster Lake	BRE-SIV01	Jul-20	Kick sampling	10U	314592	5551006
	Whymper Lake	WHM-SIV01	Jul-19	Kick sampling	10U	314095	5546627
	Gray Lake	GRA-BIV01	Jul-18	Kick sampling	10U	314594	5549467

Figure 7. Kick sampling at the main inflow (Salmon River Diversion Route) to Whymper Lake (WHMSIV-01) on July 19, 2017.

2.2.2.4. Terrestrial Invertebrates

Terrestrial invertebrates were sampled in the riparian zone of each waterbody on one to three occasions during a single growing season (Table 6). Sampling sites are shown on Map 2 to Map 12 (see 'TIV' sites). Terrestrial invertebrate sampling occurred in the same year as fish sampling on each waterbody, with the exception of Upper Campbell Reservoir in 2016. Terrestrial invertebrates were sampled from Upper Campbell Reservoir in 2014 to coincide with fish sampling but not again in 2016. Samples were collected using a malaise trap, which 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 trap and climb upwards into a collecting jar. The trap was deployed for 2.0 to 5.5 hours at each site to collect a representative sample of taxa that could potentially land on the waterbody and provide food for salmonids. The duration of trap deployment was adjusted to reflect trapping efficiency – the abundance of insects was lower later in the growing season, and also declined on cool, wet or windy days. No chemical attractants or killing agents were used and samples were preserved using 95% ethanol.

Table 6. Terrestrial invertebrate sampling site details.

Year	Waterbody	Site	Sampling Date	UTM (NAD 83)		
				Zone	E (m)	N (m)
2014	Upper Campbell Reservoir	UCR-TIV01	27-Jul-14	10U	308517	5533680
	Gooseneck Lake	GOO-TIV01	29-Jul-14	10U	318792	5535968
	Middle Quinsam Lake	QUN-TIV01	28-Jul-14	10U	322652	5533051
2015	Lower Campbell Reservoir	LCR-TIV01	Jun-24, Sep-10	10U	324620	5540694
		LCR-TIV02	Jul-20	10U	324271	5541202
	Upper Quinsam Lake	UPQ-TIV01	Jun-22, Jul-22, Sep-10	10U	320375	5539398
	Snakehead Lake	SNA-TIV01	Jun-25, Jul-21, Sep-11	10U	320824	5537868
	Beavertail Lake	BVR-TIV01	Jun-23, Jul-23, Sep-9	10U	320375	5539398
2016	John Hart Reservoir	JHT-TIV01	Jun-21, Jul-21	10U	314244	5547369
	Brewster Lake	BRE-TIV01	Jun-22, Jul-20	10U	314644	5550876
	Whymper Lake	WHM-TIV01	Jun-20, Jul-19	10U	314025	5546314
	Gray Lake	GRA-TIV01	Jun-23, Jul-18	10U	314244	5547369

Figure 8. Malaise net deployed at John Hart Reservoir (JHT-TIV01) on July 21, 2016.

2.2.3. Laboratory Methods

2.2.3.1. Taxonomic Identification

Zooplankton samples were identified primarily to family. In Year 1, zooplankton samples were enumerated by Lech Dolecki of Ecofish Research Ltd. (Ecofish) and Casey Inrig of A-Tlegay Fisheries Society. In Year 2 and Year 3, zooplankton was primarily identified by Elan Downey (BC Centre for Aquatic Health Sciences), who was supported by Casey Inrig and Terri Henderson (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; and
- Nauplii (unidentified).

Counts of less common taxa were also recorded.

Zooplankton samples were split into subsamples using a Folsom Plankton Splitter (Aquatic Research Instruments, Idaho). Samples were 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 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).

Littoral, stream and terrestrial invertebrates were sorted and counted in the laboratory to order, and, where possible, family. Identification was made with reference to Iowa State University (2015). In Year 1 (2014) and 2 (2015), identification was undertaken by Elan Downey (BC Centre for Aquatic Health Sciences) and Casey Inrig (A-Tlegay Fisheries Society). In Year 3 (2016), identification was undertaken by Jared Ellenor (Ecofish) and Terri Henderson (A-Tlegay Fisheries Society).

2.2.3.2. Zooplankton Biomass Determination

The biomass of crustacean zooplankton was estimated for each sample to provide a metric of the pelagic productivity in each lake that is available to fish. Biomass (dry weight) of zooplankton taxa 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 the presence of other material (e.g., seston) 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 (40–100× 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, which were typically the dominant taxon in each sample that made the greatest 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 9). 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 9).

Based on the consistency in length measurements between lakes, we chose to measure mean body lengths of remaining dominant taxa using samples collected from a single lake (Beavertail Lake; Table 7), 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 seasonal sampling program). In Year 1 and 2, the mean body length of Daphniidae individuals was based on measurements collected at Upper Quinsam Lake in July (Figure 9, Table 7). In Year 3, the mean body length of Daphniidae individuals in each sample was estimated based on measurements collected from each waterbody during each sampling month (Table 7).

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:

$$W = \alpha L^{\beta} \qquad (1)$$

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:

$$\ln W = \ln \alpha + \beta \cdot \overline{\ln L} \qquad (2)$$

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 among study lakes, rather than with lakes 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 9. 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, whiskers denote ranges and open circles are outliers.

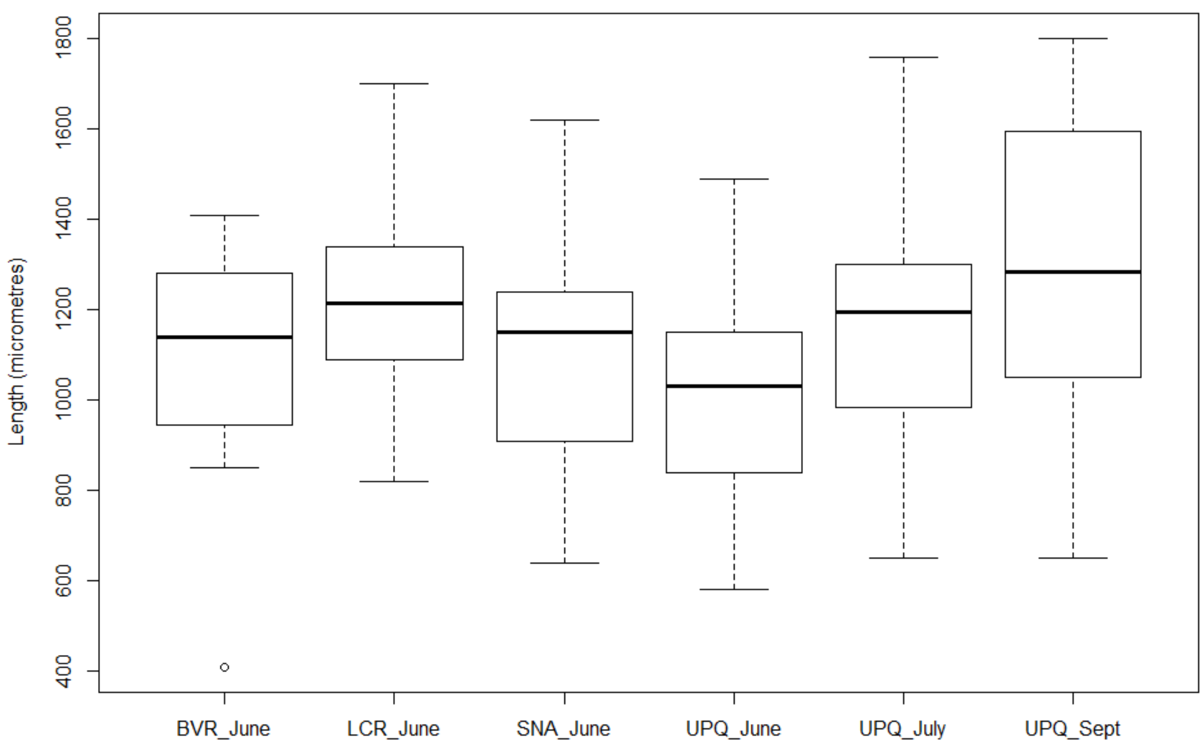


Table 7. Summary of data and relationships used to estimate the mean biomass of dominant zooplankton taxa.

Taxon	Sample: Lake	Sample: Month	# of Individuals Measured	Mean Length (µm)	Std. Dev. (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	
	Brewster	June	20	1138	268		6.94	
	Brewster	July	19	1245	321		8.93	
	Brewster	Aug	21	1374	352		11.46	
	John Hart	June	20	1009	244		5.13	
	John Hart	July	19	1022	140		4.96	
	John Hart	Aug	20	938	239		4.29	
	Gray	June	20	801	204		2.88	
	Gray	July	20	1070	246		5.91	
	Gray	Aug	21	1157	158		6.81	
	Whymper	June	20	720	198		2.24	
	Whymper	July	20	973	197		4.54	
	Whymper	Aug	21	860	158		3.27	
	Upper Campbell	June (2016)	20	1140	290		7.10	
	Upper Campbell	Aug (2016)	20	900	900		3.96	
	Upper Campbell	Sept (2016)	20	1074	254		6.00	
Other taxa								
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)
Onchopoda (*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	442	80	$\ln W = 1.953 + 2.399 \cdot \ln L$	0.96	Bottrell *et al*. (1976)
Calanoida	Beavertail	July	19	666	84	$\ln W = 1.953 + 2.399 \cdot \ln L$	2.60	Bottrell *et al*. (1976)
Naupilii	-	-	-	-	-	Constant biomass assignec	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).

2.3. Fish Sampling

2.3.1. Overview

Fish sampling was undertaken to obtain representative stable isotope samples of Cutthroat Trout, Rainbow Trout and Dolly Varden and potential fish prey items including Threespine Stickleback, Sculpin spp., and juvenile trout. The following fishing methods were used to maximize catches: gill netting, minnow trapping and trap netting. Fish sampling enabled analyses of tissue, stomach samples for diet analyses, catch per-unit-effort (CPUE), and fish size and age distributions by species.

2.3.2. Gill Netting

Gill netting was undertaken at all waterbodies towards the end of the growing season, during mid-August to early October (Table 8). Each waterbody was sampled during one growing season with the exception of Upper Campbell Reservoir, which was sampled in Year 1 and Year 3. Gill netting (Figure 10) was primarily used to sample Cutthroat Trout and Rainbow Trout. The pelagic and littoral zones were sampled on all waterbodies. The two exceptions were Whymper Lake where only one net was set and Upper Campbell Reservoir in 2016 where only littoral nets were set (Table 8). For Whymper Lake this was done because it is the smallest lake and is 100% littoral habitat. For Upper Campbell Reservoir in 2016, this was done because fish sampling was being conducted under the JHTMON-3 program. Site locations are shown in Map 2 to Map 12.

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, while at pelagic sites, nets were set perpendicular to depth contours. In Year 1, nets were set on the bed at pelagic sites, as well as suspended in the water column at a depth of 10 m to approximately target the thermocline. Year 1 results showed that catch rates were higher for the suspended nets (Hocking *et al.* 2015); accordingly, all pelagic nets were suspended in Year 2 and 3 at a depth of 5 to 10 m. RISC standard gill nets were used, which consisted 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. The mesh sizes were as follows: 25 mm, 76 mm, 51 mm, 89 mm, 38 mm, and 64 mm. This sequence of mesh sizes targeted 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 17–23 hours. Floating lights were attached to each net to mark their location overnight for boater safety. Individual fish processing is described in Section 2.3.5.

Fish CPUE from gill netting was computed for Cutthroat Trout, Rainbow Trout and Dolly Varden and compared across all waterbodies.

Table 8. Summary of gill net sampling.

Waterbody	Site	Sampling Date	UTM (Zone 10U) E (m)	UTM (Zone 10U) N (m)	Location[1]	Water Temp. (°C)[2]
Upper Campbell Reservoir	UCR-LKGN01	26-Aug-2014	314096	5539930	L	UNK
	UCR-LKGN02	26-Aug-2014	314629	5537246	L	UNK
	UCR-LKGN03	26-Aug-2014	313301	5536669	P	UNK
	UCR-LKGN04	26-Aug-2014	308638	5533904	L	UNK
	UCR-LKGN05	26-Aug-2014	309356	5530967	L	UNK
	UCR-LKGN06	26-Aug-2014	309419	5527967	L	UNK
	UCR-LKGN07	26-Aug-2014	310848	5526008	L	UNK
	UCR-LKGN08	26-Aug-2014	305645	5529532	L	UNK
Gooseneck Lake	GOO-LKGN01	20-Aug-2014	319321	5536247	P	24.0
	GOO-LKGN02	20-Aug-2014	319173	5536140	P	24.0
	GOO-LKGN03	20-Aug-2014	319255	5536146	P	24.0
	GOO-LKGN04	20-Aug-2014	318869	5536021	L	24.0
	GOO-LKGN05	20-Aug-2014	318916	5535760	L	24.0
	GOO-LKGN06	20-Aug-2014	318895	5535937	L	24.0
Middle Quinsam Lake	QUN-LKGN01	22-Aug-2014	321997	5532814	P	24.0
	QUN-LKGN02	22-Aug-2014	322080	5532804	P	24.0
	QUN-LKGN03	22-Aug-2014	321200	5533276	L	24.0
	QUN-LKGN04	22-Aug-2014	321323	5533388	L	24.0
Lower Campbell Reservoir	LCR-LKGN01	23-Aug-2015	322373	5545290	L	20.2
	LCR-LKGN02	23-Aug-2015	326112	5542580	P	20.6
	LCR-LKGN03	4-Oct-2015	324420	5541275	L	15.5
	LCR-LKGN04	4-Oct-2015	326112	5542580	P	15.8
	LCR-LKGN05	4-Oct-2015	324798	5544003	P	15.8
	LCR-LKGN06	4-Oct-2015	322364	5545336	L	15.6
Upper Quinsam Lake	UPQ-LKGN01	19-Aug-2015	317098	5528861	L	21.3
	UPQ-LKGN02	20-Aug-2015	316585	5528193	P	21.7
Snakehead Lake	SNA-LKGN01	21-Aug-2015	320729	5537745	L	21.0
	SNA-LKGN02	21-Aug-2015	320503	5537936	P	21.5
Beavertail Lake	BVR-LKGN01	17-Aug-2015	320988	5539764	L	21.0
	BVR-LKGN02	17-Aug-2015	320375	5539398	L	21.0
	BVR-LKGN03	17-Aug-2015	319990	5539765	P	21.0
	BVR-LKGN04	17-Aug-2015	320595	5539736	P	22.0
John Hart Reservoir	JHT-LKGN01	21-Aug-2016	328436	5545510	P	19.6
	JHT-LKGN02	21-Aug-2016	330819	5546423	L	19.6
Brewster Lake	BRE-LKGN01	19-Aug-2016	314670	5552673	P	23.4
	BRE-LKGN02	19-Aug-2016	314715	5550918	L	24.4
Gray Lake	GRY-LKGN01	21-Aug-2016	314090	5548284	P	20.9
	GRY-LKGN02	17-Aug-2016	313996	5547924	L	21.6
Whymper Lake	WHM-LKGN01	15-Aug-2016	314078	5546282	L	23.5
Upper Campbell Reservoir	UCR-LKGN01	29-Aug-2016	314096	5539930	L	19.7
	UCR-LKGN02	29-Aug-2016	314629	5537246	L	19.8
	UCR-LKGN04	30-Aug-2016	308638	5533904	L	20.1
	UCR-LKGN06	31-Aug-2016	309419	5527967	L	20.7
	UCR-LKGN07	31-Aug-2016	310848	5526008	L	20.7
	UCR-LKGN08	30-Aug-2016	305645	5529532	L	19.9

[1] P - pelagic zone, and L - littoral zone

[2] UNK, unknown

Figure 10. Buoys marking location of suspended gill net at UCR-LKGN08 on August 30, 2016.

2.3.3. Trap Netting

Trap netting was undertaken on all waterbodies towards the end of the growing season, during mid-August to early October in Year 2 and Year 3 (Table 9). Each waterbody was sampled during one growing season with the exception of Upper Campbell Reservoir, which was sampled in Year 2 and Year 3. Trap netting was initiated in Year 2 as an alternative method to primarily sample Threespine Stickleback. Trap netting was employed because the target number of Threespine Stickleback was not collected from each waterbody during Year 1, or during minnow trapping conducted during the early growing season (June) in Year 2 that was designed to capture Threespine Stickleback present in the littoral zone following spawning.

Eight sites were sampled on Upper Campbell Reservoir (note that sampling of this waterbody was undertaken also supported the JHTMON-3 study). Two sites were sampled at Brewster Lake, John Hart Reservoir, and Lower Campbell Reservoir while one site was sampled at the remaining lakes (Table 9, Map 2 to Map 12). Traps were set overnight in littoral areas with a target soak time of 17-29 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.

Fish CPUE from trap netting was computed for Threespine Stickleback, Cutthroat Trout and Sculpin spp. and compared among all waterbodies.

Table 9. Trap netting sampling site summary.

Waterbody	Site	Sampling Date	UTM Zone (10U) E (m)	N (m)	Water Temp. (^{0}C)[1]
Upper Campbell Reservoir	UCR-LKTN01	31-Aug-2015	305365	5528924	UNK
	UCR-LKTN02	1-Sep-2015	309922	5527439	UNK
	UCR-LKTN03	2-Sep-2015	314793	5539470	UNK
	UCR-LKTN04	2-Sep-2015	312231	5536469	UNK
	UCR-LKTN05	3-Sep-2015	310532	5535870	UNK
	UCR-LKTN06	3-Sep-2015	310046	5525736	UNK
	UCR-LKTN09	1-Sep-2016	310788	5526437	20.2
	UCR-LKTN10	1-Sep-2016	305221	5529108	19.5
Gooseneck Lake	GOO-LKTN01	6-Oct-2015	318850	5535730	15
Middle Quinsam Lake	QUN-LKTN01	6-Oct-2015	321328	5533391	15
Lower Campbell Reservoir	LCR-LKTN01	4-Oct-2015	324300	5541180	15.5
	LCR-LKTN02	4-Oct-2015	322197	5545373	16.5
Upper Quinsam Lake	UPQ-LKTN01	3-Oct-2015	317109	5529006	14.8
Snakehead Lake	SNA-LKTN01	7-Oct-2015	320230	5538106	15
Beavertail Lake	BVR-LKTN01	7-Oct-2015	320413	5539412	15
John Hart Reservoir	JHT-LKTN01	21-Aug-2016	328607	5545852	20
	JHT-LKTN02	21-Aug-2016	329324	5545976	20.1
Brewster Lake	BRE-LKTN01	19-Aug-2016	314749	5550553	23
	BRE-LKTN02	19-Aug-2016	316014	5554783	23
Gray Lake	GRY-LKTN01	17-Aug-2016	314119	5547364	21.5
Whymper Lake	WHM-LKTN01	15-Aug-2016	314104	5546445	22

[1] UNK, unknown

Figure 11. Trap net set at BRE-LKTN01 on August 19, 2016.

2.3.4. Minnow Trapping

Minnow trapping was undertaken on all waterbodies towards the end of the growing season, during mid-August to early September to target Sculpin spp., juvenile trout and Threespine Stickleback (Table 10, Map 2 to Map 12). Some waterbodies were also sampled in June in Year 2, specifically to target Threespine Stickleback present in littoral areas where lacustrine populations construct nests and spawn during spring and early summer (McPhail 2007).

Multiple sites were established on each lake, with 3–10 Gee type minnow traps deployed at each site (Table 10). Traps were deployed on the lake bed and secured to floats. 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 15–26 hours (mean = 22 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 2.3.5.

Fish CPUE from minnow trapping was computed for Sculpin spp., juvenile trout and Threespine Stickleback and compared among all waterbodies, and between years of sampling.

Table 10. Minnow trapping sampling site summary.

Waterbody	Site	Sampling Date	UTM Zone (10U) E (m)	UTM Zone (10U) N (m)	Water Temp. (^{0}C)[1]	Number of Traps
Upper Campbell Reservoir	UCR-LKMT01	28-Aug-2014	314096	5539930	UNK	3
	UCR-LKMT02	28-Aug-2014	314629	5537246	UNK	3
	UCR-LKMT04	26-Aug-2014	308638	5533904	UNK	3
	UCR-LKMT05	26-Aug-2014	309356	5530967	UNK	3
	UCR-LKMT06	27-Aug-2014	305645	5529532	UNK	3
	UCR-LKMT07	27-Aug-2014	310848	5526008	UNK	3
	UCR-LKMT08	26-Aug-2014	309419	5527967	UNK	3
	UCR-LKMT08	30-Aug-2014	309419	5527967	UNK	3
	UCR-LKMT09	1-Sep-2016	310754	5526441	20.2	5
	UCR-LKMT10	1-Sep-2016	305250	5529086	19.5	5
Gooseneck Lake	GOO-LKMT01	20-Aug-2014	318903	5535982	21	6
	GOO-LKMT02	20-Aug-2014	318856	5535974	21	6
	GOO-LKMT03	25-Jun-2015	318953	5535887	23	5
	GOO-LKMT04	25-Jun-2015	318810	5535854	23	4
Middle Quinsam Lake	QUN-LKMT01	22-Aug-2014	321331	5533391	24	6
	QUN-LKMT02	22-Aug-2014	321325	5533314	24	6
	QUN-LKMT03	25-Jun-2015	321264	5533433	24	5
	QUN-LKMT04	25-Jun-2015	322643	5533050	24	3
Lower Campbell Reservoir	LCR-LKMT01	25-Jun-2015	324667	5541047	22	5
	LCR-LKMT02	25-Jun-2015	324656	5540794	22	5
	LCR-LKMT03	23-Aug-2015	326042	5542621	22	5
	LCR-LKMT04	23-Aug-2015	324271	5541202	20.7	5
	LCR-LKMT05	23-Aug-2015	322351	5545333	20.3	5
	LCR-LKMT06	23-Aug-2015	322250	5544312	20.3	5
Upper Quinsam Lake	UPQ-LKMT01	19-Aug-2015	317098	5528861	22.1	5
	UPQ-LKMT01	22-Jun-2015	317098	5528861	24	10
	UPQ-LKMT02	19-Aug-2015	313454	5526107	22.6	5
	UPQ-LKMT03	21-Aug-2015	320192	5538070	21.5	5
	UPQ-LKMT04	21-Aug-2015	320748	5537866	21.5	5
Snakehead Lake	SNA-LKMT01	24-Jun-2015	320729	5537745	UNK	10
	SNA-LKMT02	21-Aug-2015	320570	5537918	21.5	5
	SNA-LKMT03	21-Aug-2015	320192	5538070	21.5	5
	SNA-LKMT04	21-Aug-2015	320748	5537866	21.5	5
	SNA-LKMT05	21-Aug-2015	320310	5538070	21.5	5
Beavertail Lake	BVR-LKMT01	22-Jun-2015	320432	5539370	22	5
	BVR-LKMT02	22-Jun-2015	320988	5539764	22	6
	BVR-LKMT03	17-Aug-2015	319990	5539754	22	5
	BVR-LKMT04	17-Aug-2015	320619	5539735	22	5
	BVR-LKMT05	17-Aug-2015	320930	5539771	22	5
	BVR-LKMT06	17-Aug-2015	320355	5539468	22.5	5
John Hart Reservoir	JHT-LKMT01	21-Aug-2016	329242	5546505	20.2	5
	JHT-LKMT02	21-Aug-2016	329370	5546545	20.3	5
Brewster Lake	BRE-LKMT01	19-Aug-2016	315860	5554707	21.9	5
	BRE-LKMT02	19-Aug-2016	316078	5554741	22.7	5
Gray Lake	GRY-LKMT01	17-Aug-2016	314122	5547430	21.6	5
	GRY-LKMT02	17-Aug-2016	314358	5548451	20.9	5
Whymper Lake	WHM-LKMT01	15-Aug-2016	314071	5546445	22.9	5
	WHM-LKMT02	15-Aug-2016	314053	5546315	22.9	5

[1] UNK, unknown

Laich-Kwil-Tach ENVIRONMENTAL ASSESSMENT LP

ECOFISH RESEARCH

Figure 12. Buoys marking locations of minnow traps at GRY-LKMT01 on August 17, 2016.

2.3.5. 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 11.

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 antacid (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 11. Small fin clips were taken from the caudal fin of individuals and stored in small vials with 95% ethanol.

Analysis involved describing characteristics of the fish populations, including 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:

$$K = \text{weight (g)} \times \text{length}^{-3} \text{ (mm)} \times 100{,}000 \qquad (3)$$

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. Similarly, age data for a subsample of salmonids captured in Year 1 and 2 are retained on file, based on scale analysis conducted at the Ecofish laboratory in Campbell River.

Table 11. Sampling requirements for JHTMON-5 fish sampling.

a) Non-lethal sampling requirements

Fish Species	Fork Length (mm)	Target #	Maximum #	Sampling Requirements (non-lethal)
Cutthroat Trout Rainbow Trout	>150 mm >150 mm	20 20	40 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
Juvenile Trout spp. Sculpin spp. Stickleback Dolly Varden	around 150 mm or less all sizes all sizes all sizes	10 10 10 10	20 20 20 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
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 Rainbow Trout	>150 mm >150 mm	10 10	10 10	• Retain stomachs in formalin from 10 of the large Cutthroat and Rainbow Trout sampled above

2.3.6. Stomach Contents

Fish stomachs were extracted from Cutthroat Trout and Rainbow Trout that were sampled from all waterbodies using gill netting, based on the targets shown in Table 11. Stomach contents analysis was conducted to provide information about fish diet to complement stable isotope results. 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.

2.4. Stable Isotope Data

2.4.1. 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 773 samples of invertebrates and fish from surveys between 2014 and 2016 were sent for analysis (Table 12). Invertebrates were sent as whole individuals, while most fish were sent as fin clip samples.

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:

$$\boldsymbol{\delta^{15}N \text{ or } \delta^{13}C}\ (‰) = (\mathbf{R}_{sample}/\mathbf{R}_{stamdard} - 1) * 1000 \qquad (4)$$

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

Taxa	2014			2015				2016					Total
	Gooseneck Lake	Middle Quinsam Lake	Upper Campbell Reservoir	Beavertail Lake	Lower Campbell Reservoir	Snakehead Lake	Upper Quinsam Lake	Brewster Lake	Gray Lake	John Hart Reservoir	Upper Campbell Reservoir	Whymper Lake	
Zooplankton	3	3	8	9	9	9	9	9	9	9	9	9	**95**
Littoral Invertebrates	4	4	3	2	8	2	1	4	4	4	3	4	**43**
Benthic Invertebrates					4						4		**8**
Stream Invertebrates	1	1	2	3	1	1	2	1	1	1		1	**15**
Terrestrial Invertebrates	1	1	1	3	3	3	3	2	2	2		2	**23**
Sculpin spp.	6	4	6	5	12		5	9	10	10	9	10	**86**
Threespine Stickleback			10		10			10	10	9	9	10	**68**
Juvenile Trout	5	4	12	1	5	5		6	5	6	4	14	**67**
Kokanee Salmon								6	15				**21**
Dolly Varden	6		1	6				4					**17**
Rainbow Trout		1	18	4	27			11		12	20	3	**96**
Cutthroat Trout	20	20	20	20	29	21	20	21	15	16	21	11	**234**
Total	**46**	**38**	**81**	**53**	**108**	**41**	**40**	**83**	**71**	**69**	**79**	**64**	**773**

2.4.2. Assessing Fish Diet Using Mixing Models

The relative contributions of pelagic and littoral sources to Cutthroat Trout, Rainbow Trout, Dolly Varden and Kokanee 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 eleven lakes. The first model estimated diet contributions to large Cutthroat Trout and Dolly Varden (Age >2+, FL ≥ 154 mm). Four potential diet sources (mean $\delta^{13}C$ and $\delta^{15}N$ ± SD) for the two large fish species were included in this model: 1) zooplankton, 2) littoral invertebrates (including benthic invertebrates), 3) terrestrial invertebrates, and 4) prey fish, which included juvenile trout (Age ≤ 2, FL ≤ 152), Sculpin spp. (FL ≤ 170 mm), and Threespine Stickleback (FL ≤ 80 mm). Stream invertebrates were excluded because their stable isotope signatures overlapped strongly with that of littoral invertebrates. Prey fish were aggregated because of their overlap in isotope signatures within and across lakes.

The second model run for each lake estimated the diet contributions to adult Rainbow Trout (Age >2+, FL ≥ 154 mm) and the prey fish. Three potential diet sources (mean $\delta^{13}C$ and $\delta^{15}N$ ± SD) were used to estimate the Rainbow Trout and prey fish diets: 1) zooplankton, 2) littoral invertebrates, and 3) terrestrial invertebrates. Models were also run for adult Kokanee Salmon (Age >2+, FL ≥ 173 mm) in the two systems where they were captured (Brewster Lake and Gray Lake). Adult Rainbow Trout and Kokanee diets are dominated by zooplankton and small invertebrates (e.g., Chapman *et al.* 1967, McPhail 2007 and references therein). All Rainbow Trout individuals across all lakes were less than 400 mm in length, which is a threshold identified for diet shifts of Rainbow Trout towards other fish. Excluding prey fish as a potential diet item in Rainbow Trout models was further supported by a lack of prey fish found in Rainbow Trout stomach contents (section 3.2.5).

The two models were run to assess the total relative contributions of pelagic versus littoral sources of production to target large Cutthroat Trout, Rainbow Trout and Dolly Varden via direct and indirect pathways. For Cutthroat Trout and Dolly Varden, both pathways needed to be considered since a proportion of their diet is derived from prey fish. Therefore, for Cutthroat Trout and Dolly Varden, the total littoral versus pelagic contribution equals the sum of the contributions of the invertebrate prey to target fish diets in model one (direct pathway) with the contributions of the invertebrate prey to target fish that occur via consumption of prey fish (model 2) (indirect pathway). The direct pathway (model 1) is the contribution of zooplankton (pelagic) and summed contribution from littoral and terrestrial invertebrates (littoral) to Cutthroat 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 prey fish diets that is carried forward to the diets of Cutthroat Trout and Dolly Varden.

2.4.3. 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).

2.5. Water Residence Time

2.5.1. General

Water residence time was estimated for all JHTMON-5 waterbodies. The water residence time (WRT) was calculated by dividing the effective volume of the basin (V) by the annual average outflow (Q_o) rate. To estimate residence time during the growing season, an effective volume, defined as the average volume of the mixed layer (epilimnion), was used in the calculations. This reflects that WRT during the growing season can be strongly influenced by the presence of thermal stratification, which reduces the effective volume of the waterbody where mixing occurs (Rueda *et al.* 2006). Based on historical water temperature records, stratification is assumed to set in within each waterbody around May 15 and to break down around September 30 (Hatfield 2000).

Waterbody volumes were obtained from stage-volume relationships developed by BC Hydro (Bruce 2001, Perrin 2012) and bathymetric maps. With the exception of John Hart Reservoir and Whymper Lake, the volume of the epilimnion was determined from the average thermocline depth of the waterbodies (Section 2.5.2). The thermocline depth was then related to volume using the stage-volume relationships developed by BC Hydro (Bruce 2001). The full waterbody volume was used in the seasonal WRT calculations for John Hart Reservoir and Whymper Lake, as they were only weakly stratified (Section 2.5.2).

The outflow rate (Q_o) was estimated from the following:

$$\boldsymbol{Q_o = Q_i + ((P - E) \times A)} \quad (5)$$

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). Equation (5) is a water balance method that neglects groundwater fluxes or net change in storage. Where $Q_o$is gauged (e.g., Upper Campbell, Lower Campbell, and John Hart reservoirs), the gauged outflow rate was compared to the computed Q_o from Equation (5) to determine relative accuracy of the water balance method.

Inflow rate (Q_i) was estimated by computing a runoff coefficient for the watershed (Section 2.5.4). 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 feasible, these estimates were validated with data collected at hydrometric gauges. Water Survey of Canada operates flow gauges upstream and downstream of the Quinsam diversion that provide data to check the inflow data computed for Gooseneck, Middle Quinsam, and Snakehead lakes. Water Survey of Canada also operates a flow gauge on the Salmon River diversion; this inflow was added to Brewster Lake inflows computed from Equation (5) to reflect that the diversion canal conveys water to the lake from outside of the watershed. A summary of the hydrometric stations used in the inflow and outflow calculations is provided in Table 13; Map 13 shows the locations of the stations relative to the study lakes.

Table 13. Hydrometric stations used in the WRT computations.

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

[1]Data used only to determine connectivity between Upper Campbell Reservoir and Buttle Lake.
[2]Data provided by BC Hydro.

2.5.2. Thermocline Depth

The presence of a thermocline can influence the water residence time in waterbodies. Waterbodies often exhibit vertical temperature stratification due to variability in water density with temperature. During summer, relatively warm water in the surface mixed layer (epilimnion) can overlie cooler bottom water (hypolimnion), with the thermocline delineating the zone of maximum vertical change in temperature. Seasonal stratification occurs in the majority of the JHTMON-5 waterbodies, although stratification is weak or absent in Whymper Lake and John Hart Reservoir, which are relatively shallow and have short water residence times. Seasonal stratification limits the extent of vertical mass transport: under stratified conditions, inflows do not mix with the full water column, outflows are typically drawn from a confined range of depths, and the hypolimnion becomes isolated from the inflow and outflow processes. Hypolimnetic water can remain in the waterbody for longer than predicted from the ratio of waterbody volume and annual outflow rate. The presence and depth of the thermocline was used to characterize stratified conditions within the JHTMON-5 waterbodies to calculate effective volume used to estimate WRT. During stratified conditions, the effective volume of the epilimnion was used to calculate the seasonal residence time, i.e., it was assumed that inflows mixed only with the surface mixed layer.

To determine the presence and depth of a thermocline, temperature profiles were collected at the majority of JHTMON-5 waterbodies in September 2015, and June and July 2016 (Table 14). TidbiT v.2 temperature data loggers (Onset) were attached at 1–4 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. Surface water temperature measurements were collected with an alcohol thermometer on some waterbodies. The 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. To increase the sample size, additional temperature measurements were used to define the depth of the thermocline in the selected lakes, where available. 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. The temperature data provided in the reports are similar to the temperature data collected in September 2015 and June and July 2016 in this study.

John Hart Reservoir and Whymper Lake were only weakly stratified. Therefore, the full lake volume was used in the seasonal WRT calculations, i.e., not just the surface mixed layer.

Table 14. Temperature profile sampling dates for JHTMON-5 waterbodies[1].

Waterbody	September 9-11 2015	June 21-23 2016	July 18-21 2016
Gooseneck Lake	x		
Middle Quinsam Lake	x		
Lower Campbell Reservoir	x		
Upper Quinsam Lake	x		
Snakehead Lake	x		
Beavertail Lake	x		
John Hart Reservoir		x	x
Brewster Lake		x	x
Gray Lake		x	x
Whymper Lake			x

[1]No data were collected from Upper Campbell Reservoir in 2015 or 2016.

2.5.3. Precipitation and Evaporation

Local precipitation data were obtained to determine direct daily inputs to the waterbodies, and to derive water inflow from drainage basins. There are a number of climate stations within the WUP study area; however, only three stations had continuous data within the vicinity and at similar elevations to the study lakes. Daily precipitation and air temperature data were obtained from the BC

Hydro climate stations at Strathcona Dam (SCA), Quinsam River at Argonaut Bridge (QIN), and Salmon River above the Diversion (SAM) (accessed from PCIC 2013). For each watershed, data from the closest available station were used in the calculations. Summaries of the climate stations used to source data for the WRT calculations are provided in Table 15 and Table 16; Map 13 shows the locations of the stations relative to the JHTMON-5 waterbodies.

Table 15. Climate stations used to compute WRT.

Climate Station Name	Station No.	Elevation (m)	Parameters Recorded		Time Interval
			Air Temperature (ºC)	Precipitation (mm)	
Strathcona Dam (SCA)	2501	227	x	x	Daily Oct 2011 - Nov 2016
Quinsam river at Argonaut Bridge (QIN)	2498	280	x	x	Daily Oct 2011 - Nov 2016
Salmon River above the Diversion (SAM)	2500	215	x	x	Daily Oct 2011 - Nov 2016

Table 16. Climate stations (Table 15) used to compute WRT for each JHTMON-5 waterbody.

Waterbody	Climate Station		
	SCA	QIN	SAM
Upper Campbell Reservoir	x		
Gooseneck Lake		x	
Middle Quinsam Lake		x	
Lower Campbell Reservoir	x		
Upper Quinsam Lake		x	
Snakehead Lake		x	
Beavertail Lake		x	
John Hart Reservoir			x
Brewster Lake			x
Gray Lake			x
Whymper Lake			x

Evaporation estimates were required to determine the evaporative losses from the waterbodies. There are no direct measurements of evaporation from the WUP waterbodies, and continuous measurements of net radiation, wind speed, and humidity are not available to compute evaporation. Accordingly, an empirical method was used to estimate monthly total potential evapotranspiration from the waterbodies based on air temperature. Evaporation from open water approximates potential evapotranspiration because the supply of water is non-limiting in both conditions, meaning there is an adequate supply of water for evaporative processes.

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

$$ET_i = 16\left(\frac{10T_i}{I}\right)^{\propto} \qquad (6)$$

where ET_i is potential evapotranspiration for month i (mm/month); T_i is mean monthly air temperature (°C); I is the local heat index given by,

$$I = \sum_{i=1}^{12}\left(\frac{T_i}{5}\right)^{1.514} \qquad (7)$$

and the coefficient $\alpha = \left(0.675 \times I^3 - 77.1 \times I^2 + 17{,}920 \times I + 492{,}390\right) \times 10^{-6}$ (Equations 4a and 4b ; Xu and Singh 2001). 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 (Table 5-2 in Dunne and Leopold 1978). The Thornthwaite formula (1948) was used to derive annual (October 1 to October 1) and seasonal (May 15 to October 1) evaporation estimates for JHTMON-05 study lakes.

The average annual and seasonal precipitation and potential evapotranspiration data are provided in Table 17. The annual precipitation was lowest in 2013–2014 with values 409 mm to 695 mm less (depending on the climate station) than the highest precipitation recorded in other years. During 2014, precipitation from May 15 to October 1 was also lowest (206–231 mm depending on climate station location) relative to the other years. For historical context, the average total annual precipitation recorded at SCA from October 1, 1981 to October 1, 2016 was 1,369 mm; the average total seasonal precipitation (May 15 to October 1) for the same period was 239.5 mm. Based on the ranges shown in Table 17, the annual and seasonal precipitation values over the study period (2012–2016) provide valuable information on how water residence times vary between years with lower than average, average, and higher than average precipitation.

Total estimated annual evaporation from the waterbodies ranged from 587 mm to 647 mm (Table 17). Seasonal (May 15 to October 1) evaporation estimates ranged from 424 mm to 457 mm (Table 17). A literature search was conducted to confirm whether these estimates were consistent with published estimates for comparable lakes. Evaporation estimates for lakes in two areas on Vancouver Island (Ladysmith and Salt Spring Island) are presented in Sprague (2007). Though these locations are further south, these lakes are at a similar elevation to the waterbodies within the Campbell River area. Mean annual evaporation reported for lakes near Ladysmith was 642 mm (Tetra Tech EBA 2014 cited in Sprague 2007), while similar estimates were determined for lakes on Salt Spring Island (713 mm and 585 mm) (reported in Sprague 2007). Another study of the same lakes estimated total seasonal lake evaporation to be approximately 411 mm (Barnett *et al.* 1993). These estimates are consistent with the annual and seasonal evaporation estimates for the JHTMON-5 waterbodies that were computed from the Thornthwaite formula (1948) (Table 17).

Table 17. Total annual and seasonal precipitation and estimated lake evaporation data used in the WRT computations.

Year	SCA[1] Precipitation (mm)	QIN[2] Precipitation (mm)	SAM[3] Precipitation (mm)	Evaporation (mm)	SCA P- E (mm)	QIN P- E (mm)	SAM P- E (mm)
Annual [4]							
2011-2012	1343	1057	1627	587	756	470	1040
2012-2013	1319	1170	1711	610	709	560	1101
2013-2014	871	847	1224	605	266	242	619
2014-2015	1343	1249	1731	647	696	602	1084
2015-2016	1377	1256	1919	634	743	622	1285
Seasonal [5]							
2012	204	183	223	424	-220	-241	-201
2013	400	414	448	438	-38	-24	10
2014	175	208	217	457	-282	-249	-240
2015	295	262	324	447	-152	-186	-123
2016	270	330	381	431	-161	-101	-50

[1] BC Hydro climate station at Strathcona Dam (SCA).

[2] BC Hydro climate station Quinsam River at Argonaut Bridge (QIN).

[3] BC Hydro climate station at Salmon River above the Diversion (SAM).

[4] October 1 to October 1

[5] May 15 to October 1

2.5.4. Inflow Rate

To estimate the inflow rate (Q_i), an algorithm was created using a modified Natural Resources Conservation Services Curve Number (NRCS-CN), formerly known as the Soil Conservation Service Curve Number (SCS-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:

$$Q_i = \frac{(P - I_a)}{P - I_a + S} \qquad (8)$$

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 before runoff begins, expressed in mm), and S is the potential maximum capacity of retention after runoff begins (mm). Equation (8) is valid for $P > I_a$, otherwise, $Q_i = 0$.

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

$$S = \frac{25400}{CN} - 254 \qquad (9)$$

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 (SCS 1972), and are moderately well-drained (i.e., the upper 1 m 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 (Huang *et al.* 2006).

Parameter I_a is equivalent to

$$I_a = \lambda \cdot S \qquad (10)$$

where λ is the initial abstraction ratio. The ratio of I_a/S=0.2 is commonly used in the scientific literature, as this was the original relationship published. This ratio was found to overestimate runoff in the JHTMON-5 watersheds, when compared to available gauged values. Hawkins *et al.* (2002) examined the ratio of I_a/S using rainfall and runoff data from numerous watersheds in the U.S and found that over 90% of the ratios were less than 0.2, and were on average 0.05. Other studies have found similar results (Elhakeem and Papanicolaou 2009, White *et al.* 2009, Lamont *et al.* 2008, Baltas *et al.* 2007).

To determine the best estimate of I_a, the curve numbers were calibrated using locally gauged rainfall and runoff. An iterative least squares function was used to find the best fit for both λ and S to Equation (10), i.e., by minimizing the sum of squared differences between rank-ordered rainfall depth and gauged runoff, where available (Table 13) (Baltas *et al.* 2007). For ungauged watersheds, λ was assumed to be 0.05. Equation (10) was developed for λ = 0.2, and thus had to be adjusted. For example, a transfer to equivalent CNs having λ=0.05 was attained from the empirically derived equation: $S_{0.05} = 1.33(S_{0.20})^{1.15}$ (Jiang 2001).

The watershed characteristics used to determine inflow rate are presented in Table 18. For each day, the area-weighted-average curve number was adjusted according to the five-day antecedent runoff condition (ARC) to account for the temporal variability of potential water retentions (dry, average, or wet) in the watersheds (Hawkins *et. al.* 1985). A breakdown of inflow, evaporative loss, and outflow computed for each of the study lakes is provided in Table 19 and Table 20 for the annual and seasonal periods, respectively. John Hart Reservoir had the highest annual and seasonal inflows, and Beavertail Lake had the lowest.

Table 18. Watershed characteristics, including drainage area, average slope of the watershed, runoff curve numbers (*CN*), soil retention capacity (*S*), and the average initial abstractions (I_a) by soil retention capacity ratio (I_a/S).

Waterbody	Watershed Area[1] (km²)	Average Slope (%)	*CN*			*S* (mm)			I_a / S
			(I)[2]	(II)[3]	(III)[4]	(I)[2]	(II)[3]	(III)[4]	
Upper Campbell Reservoir	1,192.8	56.3	55	73	86	210	96	40	0.18
Upper Quinsam Lake	84.6	30.9	35	56	77	477	198	75	0.10
Middle Quinsam Lake	94.6	25.6	35	56	77	456	189	71	0.10
Gooseneck Lake	31.8	15.1	25	46	69	744	301	112	0.05
Snakehead Lake	36.9	13.7	25	46	69	746	302	112	0.05
Beavertail Lake	5.7	10.3	24	44	68	794	320	119	0.05
Lower Campbell Reservoir	1,422.3	49.2	54	72	86	213	98	41	0.18
Brewster Lake	45.6	16.3	26	47	70	710	288	107	0.05
Gray Lake	56.8	16.0	26	47	70	718	291	108	0.05
Whymper Lake	68.5	15.7	26	47	70	717	291	108	0.05
John Hart Reservoir	1,449.7	48.5	51	70	85	244	110	44	0.17

[1]Watershed area includes the local watershed area plus the contributing area upstream. The watershed area for Gooseneck, Middle Quinsam, and Snakehead Lakes represents the average upstream area contributing to inflows, taking into account the % of diversion flows in 2012-2016.

[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 19. Total annual inflow (Q_i), precipitation (P) minus evaporation (E) multiplied by lake area, and lake outflow (Q_o) computed for each hydrologic year (October 1 to September 30) during 2011–2016.

Waterbody	2016			2015			2014			2013			2012		
	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)
Upper Campbell Reservoir	76.6	1.62	78.2	62.3	1.52	63.8	69.1	0.58	69.7	71.2	1.55	72.7	75.2	1.65	76.8
Upper Quinsam Lake	2.72	0.10	2.82	2.59	0.10	2.69	2.81	0.04	2.85	2.95	0.09	3.04	2.87	0.08	2.95
Middle Quinsam Lake	2.51	0.01	2.52	2.68	0.01	2.69	2.18	0.01	2.19	2.35	0.01	2.36	2.59	0.01	2.60
Gooseneck Lake	0.53	0.02	0.55	0.35	0.01	0.36	0.61	0.01	0.61	0.57	0.01	0.58	0.46	0.01	0.47
Snakehead Lake	0.63	0.00	0.64	0.44	0.00	0.45	0.70	0.00	0.70	0.66	0.00	0.66	0.56	0.00	0.56
Beavertail Lake	0.08	0.02	0.10	0.08	0.02	0.10	0.07	0.01	0.08	0.08	0.02	0.09	0.08	0.02	0.09
Lower Campbell Reservoir	94.6	0.51	95.2	76.9	0.47	77.4	85.6	0.18	85.7	88.6	0.48	89.1	93.8	0.51	94.3
Brewster Lake	3.36	0.31	3.68	3.52	0.27	3.79	1.70	0.15	1.85	0.52	0.17	0.70	0.60	0.25	0.85
Gray Lake	3.76	0.03	3.79	3.83	0.02	3.85	1.95	0.01	1.97	0.77	0.01	0.79	0.88	0.02	0.90
Whymper Lake	3.96	0.00	3.97	3.97	0.00	3.97	2.02	0.00	2.02	0.85	0.00	0.85	0.97	0.00	0.97
John Hart Reservoir	103.0	0.14	103.2	86.8	0.12	86.9	90.5	0.07	90.6	86.7	0.12	86.8	96.4	0.11	96.5

Table 20. Total inflow (Q_i), precipitation (P) minus evaporation (E) multiplied by lake area, and lake outflow (Q_o) computed for each summer stratified period (assumed May 15 to October 1) during 2012–2016.

Waterbody	2016			2015			2014			2013			2012		
	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)	Q_i (m^3/s)	(P-E)*Lake Area (m^3/s)	Q_o (m^3/s)
Upper Campbell Reservoir	52.6	-0.92	51.7	42.1	-0.87	41.3	56.4	-1.62	54.8	49.0	-0.21	48.8	58.6	-1.26	57.4
Upper Quinsam Lake	1.29	-0.04	1.24	1.12	-0.08	1.04	1.02	-0.11	0.92	0.97	-0.01	0.96	1.28	-0.10	1.18
Middle Quinsam Lake	1.07	-0.01	1.06	0.95	-0.01	0.94	0.73	-0.02	0.71	0.70	0.00	0.70	1.04	-0.01	1.03
Gooseneck Lake	0.23	-0.01	0.23	0.14	-0.01	0.13	0.25	-0.02	0.24	0.23	0.00	0.23	0.16	-0.02	0.15
Snakehead Lake	0.25	0.00	0.24	0.16	0.00	0.16	0.23	0.00	0.23	0.21	0.00	0.21	0.17	0.00	0.17
Beavertail Lake	0.09	-0.01	0.08	0.08	-0.02	0.06	0.08	-0.02	0.06	0.08	0.00	0.07	0.10	-0.02	0.07
Lower Campbell Reservoir	56.6	-0.29	56.3	45.9	-0.27	45.6	61.0	-0.50	60.5	52.8	-0.07	52.7	63.4	-0.39	63.0
Brewster Lake	1.28	-0.03	1.25	4.84	-0.08	4.76	1.29	-0.15	1.14	0.56	0.01	0.56	0.62	-0.13	0.49
Gray Lake	1.36	0.00	1.36	4.90	-0.01	4.89	1.29	-0.01	1.28	0.59	0.00	0.60	0.63	-0.01	0.62
Whymper Lake	1.45	0.00	1.45	4.97	0.00	4.97	1.33	0.00	1.33	0.66	0.00	0.66	0.69	0.00	0.69
John Hart Reservoir	67.5	-0.01	67.5	56.0	-0.04	56.0	59.8	-0.07	59.8	61.0	0.00	61.0	71.6	-0.06	71.6

2.5.5. Analysis of Water Diversion Scenarios

We estimated how different water diversion scenarios could affect the annual and seasonal WRT of six diversion lakes over the last 20 years (October 1997- October 2016), Pre-WUP (October 1997 – October 2004), and Post-WUP (October 2012 – October 2016). Annual water diversion scenarios included a significant diversion scenario (90% of flow), above average diversion (70% of flow), average diversion (24% of the flow for the Salmon Diversion and 30% of the flow for the Quinsam diversion), average diversion Pre-WUP (34% of the flow for the Salmon Diversion and 37% of the flow for the Quinsam diversion), average diversion Post-WUP (13% of the flow for the Salmon Diversion and 25% of the flow for the Quinsam diversion), and no diversion (0% of flow) (Table 21, Table 22). Seasonal water diversion scenarios were similar, and included a significant diversion scenario (90% of flow), above average diversion (70% of flow), average diversion (35% of the flow for the Salmon Diversion and 30% of the flow for the Quinsam diversion), average diversion Pre-WUP (57% of the flow for the Salmon Diversion and 43% of the flow for the Quinsam diversion), average diversion Post-WUP (15% of the flow for the Salmon Diversion and 10% of the flow for the Quinsam diversion), and no diversion (0% of flow) (Table 21, Table 22).

In the Quinsam River watershed, total available lake inflow used for scenario analysis was configured by adjusting flow measured at WSC 08HD026 (Gooseneck and Snakehead) and WSC 08HD021 (Middle Quinsam) (Table 21). For the three lakes downstream of the Salmon River Diversion (Brewster, Gray, and Whymper lakes), water diversion scenarios were configured by adjusting flows recorded at the WSC 08HD015 gauge, after subtracting inflows recorded at the WSC08HD020 gauge (Table 22).

Table 21. Total annual and seasonal inflow used to compute water diversion scenarios for the diversion lakes in the Quinsam River watershed in 2012-2016.

Year	WSC 08HD021 + WSC 08HD026 Total Available Inflow[1]		WSC 08HD021 Regulated Flow		WSC 08HD026 Regulated Flow		Middle Quinsam Lake Regulated Flow		Gooseneck and Snakehead Lake Regulated Flow	
	Mean Annual Q (m^3/s)	Mean Seasonal Q (m^3/s)	Mean Annual Q (m^3/s)	Mean Seasonal Q (m^3/s)	Mean Annual Q (m^3/s)	Mean Seasonal Q (m^3/s)	Annual % Q Diverted	Seasonal % Q Diverted	Annual % Q Diverted	Seasonal % Q Diverted
2012	3.55	2.26	3.06	2.23	0.48	0.02	86.4	98.9	13.6	1.08
2013	2.39	1.67	1.78	1.42	0.61	0.26	74.6	84.7	25.4	15.3
2014	1.93	0.98	1.08	0.81	0.85	0.17	56.1	82.9	43.9	17.1
2015	3.20	0.65	2.93	0.65	0.28	0.00	91.4	99.5	8.64	0.46
2016	3.36	1.19	2.66	1.10	0.70	0.09	79.1	92.4	20.9	7.57
Past 20-Year Mean Q	3.11	1.83	2.20	1.36	0.92	0.47	68.69	71.82	31.31	28.18
Pre-WUP Mean Q	3.11	1.81	2.02	1.17	1.09	0.64	63.38	57.38	36.62	42.62
Post-WUP Mean Q	2.72	1.12	2.11	0.99	0.61	0.13	75.28	89.89	24.72	10.11

[1]WSC 08HD021 is the Quinsam River at Argonaut Bridge gauge; WSC 08HD026 is the Quinsam Diversion near Campbell River gauge. Combined flow data from WSC 08HD021 and 08HD026 represents flow released from the Quinsam and Wokas Lakes dams.

Table 22. Total annual and seasonal inflow used to compute water diversion scenarios for the diversion lakes in the Salmon River watershed in 2012-2016.

Year	WSC 08HD015[1] Total Available Inflow		WSC 08HD020[2] Regulated Flow		Brewster, Gray and Whymper Lakes	
	Mean Annual Q (m^3/s)	Mean Seasonal Q (m^3/s)	Mean Annual Q (m^3/s)	Mean Seasonal Q (m^3/s)	Annual % Q Diverted	Seasonal % Q Diverted
2012	17.05	14.46	0.00	0.00	0.0	0.0
2013	12.22	7.80	0.00	0.00	0.0	0.0
2014	8.34	3.52	1.21	0.79	14.6	40.9
2015	14.70	3.06	2.68	4.33	18.3	0.0
2016	13.67	3.96	2.66	0.64	19.5	19.3
Past 20-Year Mean Q	14.08	9.13	3.09	2.15	23.5	34.5
Pre-WUP Mean Q	13.48	8.75	4.28	3.19	34.0	57.1
Post-WUP Mean Q	12.23	4.59	1.64	1.44	13.07	15.05

[1]WSC 08HD015 is the Salmon River above Diversion gauge

[2]WSC 08HD020 is the Salmon River Diversion gauge.

2.6. Littoral Area Calculations

Two horizontal zones can be delineated in lakes and reservoirs: the littoral zone and the pelagic zone. The littoral zone is the nearshore area that is sufficiently shallow to allow growth of attached autotrophs (rooted macrophytes and periphyton) on the bed of the lake, while the pelagic zone encompasses deeper areas, where phytoplankton comprises the only group of autotrophs (Lewis 2009). The boundary between these two zones is commonly approximated as the depth at which 1% of photosynthetically active radiation (PAR) present at the water surface reaches the bed – this depth at which net photosynthesis occurs is termed the euphotic depth (Lewis 2009, Moss 2010). Depending on water clarity, the euphotic depth varies among waterbodies, and seasonally within waterbodies, with typical values of 0.4–40 m (Moss 2010).

The area (ha) of the littoral zone and pelagic zone was estimated for each waterbody. For Upper and Lower Campbell reservoirs, the euphotic depth (20.0 m) was calculated based on PAR measurements collected in summer and fall by Perrin *et al.* (2016). For the remaining waterbodies, euphotic depth was estimated as twice the Secchi depth (c.f. Moss 2010), which was estimated based either on data collected during previous studies, or measurements collected in this study (Table 24).

The areas of the littoral and pelagic zones in each waterbody were then calculated based on estimates of euphotic depth and hypsographic tables derived for each waterbody based on existing bathymetry data provided by BC Hydro, or bathymetry data collected during JHTMON-5. For the reservoirs that are drawn down during the growing season (Upper Campbell and Lower Campbell), the littoral zone was defined as the area where the euphotic depth extends to the bed below the mean water level for the growing season (May–October). The percentage of each zone (by area and volume) was calculated for each waterbody. In addition, the percentage of each waterbody (by area and volume) that is < 6.0 m deep was also calculated. The 6.0 m criterion is the threshold to define the littoral zone that is presented in provincial standards for fish habitat inventory (RIC 2001). This criterion is shallower than the euphotic depth estimates presented in Table 24 and is therefore likely to be representative of the most productive area of the littoral zone. The estimated relative area and volume of each zone provide metrics to standardize variability in fish diets between waterbodies to isolate any variability that is due to variability in water residence time.

Table 23. Estimated euphotic depths used to calculate the area of the littoral zone in each waterbody.

Waterbody	Euphotic depth (m)	Reference/Rationale
Beavertail Lake	24.0	Twice Secchi depth reported by Silvestri and Fosker (2003)
Brewster Lake	22.0	Twice Secchi depth reported by Dayton and Knight (2001)
Gooseneck Lake	22.0	Set equal to Brewster Lake (downstream)
Gray Lake	19.0	Twice Secchi depth reported by Dayton and Knight (2001)
John Hart Reservoir	21.0	Twice Secchi depth reported by Perrin *et al*. (2012)
Lower Campbell Reservoir	20.0	Euphotic depth measured with a PAR sensor by Perrin *et al*. (2016). Based on a measurements in summer and fall.
Middle Quinsam Lake	17.0	Mean of multiple measurements by MacIsaac and Stockner (1985)
Snakehead Lake	15.0	Twice Secchi depth reported by Dayton and Knight (2001)
Upper Campbell Reservoir	20.0	Euphotic depth measured with a PAR sensor by Limnotek during Year 2 of MON-4.
Upper Quinsam Lake	22.0	Twice Secchi depth reported by Craig and Kehler (2009)
Whymper Lake	10.0	Twice Secchi depth measured on July 19, 2016

2.7. Analysis of Management Questions

A synthesis analysis was completed to integrate the invertebrate, fish, stable isotope, water residence time and littoral area datasets and address the management questions and hypotheses H_01 and H_02 of JHTMON-5.

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

To address this management question and hypothesis H_01, the littoral contribution to Cutthroat Trout and Rainbow Trout diets as estimated using stable isotope analysis was compared between Upper Campbell Reservoir, which experiences the greatest fluctuations in water levels, Lower Campbell Reservoir, which experiences intermediate fluctuations in water levels, and John Hart Reservoir, which experiences the lowest fluctuations in water levels. In addition, the CPUE data for Cutthroat Trout and Rainbow Trout was compared between the three reservoirs to test for differences in fish productivity.

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

To address this management question and hypothesis H_02, the pelagic contribution to Cutthroat Trout and Rainbow Trout diets as estimated using stable isotope analysis was analyzed across all 11 lakes and reservoirs sampled in JHTMON-5. General linear models were built to predict pelagic contribution to Cutthroat Trout and Rainbow Trout diets as a function of water residence time (either annual or seasonal), the proportion of littoral habitat available in each lake and lake volume. Analyses were completed using the package MuMIn in R, where all possible model combinations were competed against one another to best predict pelagic contribution to trout diets. Residuals were visually inspected to confirm normal distributions. Annual and seasonal water residence times and lake volume were log transformed for the analysis. Additional response variables tested using this method included the biomass of zooplankton and the CPUE of Cutthroat Trout and Rainbow Trout, all indices of lake productivity. The top model was extracted from each of the model combinations and presented.

As a final step, the top model equations were applied to the diversion scenarios presented in Section 2.4.5 to evaluate how changes in water residence time in the Quinsam River and Salmon River diversions are predicted to affect pelagic contribution to trout diets and lake productivity.

3. RESULTS

3.1. Invertebrate Sampling

3.1.1. Zooplankton

3.1.1.1. Abundance

Dominant zooplankton taxa were: Daphniidae, Cyclopoida, Calanoida, Bosminidae, Sididae, Onychopoda, Polyphemidae and Leptodoridae (Tables 1–3 in Appendix A). Cladocera (e.g., Daphniidae and Bosminidae) were typically the dominant zooplankton group, comprising a mean of 60% (range: 17–94%) of the individuals in each sample. Daphniidae spp. (Figure 13) were typically the most abundant cladocerans e.g., this taxon comprised a mean of ≥50% of all zooplankton individuals in several waterbodies (Lower Campbell Reservoir, Snakehead Lake, and Whymper Lake; Appendix A). Copepoda (Cyclopoida, Calanoida and Harpacticoid) were also common, comprising a mean of 40% (range: 6–83%) of the individuals in each sample. Of these, Cyclopoida and Calanoida (Figure 14) were the most common orders. Individuals belonging to the following taxa were also occasionally recorded in some lakes, typically in only a single sample collected throughout the season: Arachnida, Chironomidae, Radiolaria, Oligochaeta, Tricladida and Gammaridae.

Figure 13. Gravid Daphniidae individual sampled in Year 3.

Figure 14. Examples of zooplankters collected during Year 3 sampling. The photograph shows four Bosminidae (Cladocera) individuals and two Calanoid copepods.

3.1.1.2. Biomass

Zooplankton biomass was typically lower in the three reservoirs than in the eight lakes, as shown in Figure 15, which presents variation in mean zooplankton biomass (averaged across sites) among sampling dates for each waterbody. The lowest mean biomass was measured in Upper Campbell Reservoir, with comparable values measured between the two sampling years (Year 1 mean = 13.2 µg/L; Year 2 mean = 11.2 µg/L). The highest mean biomass was measured in Middle Quinsam Lake (66.8 µg/L), with a maximum monthly mean of 105.6 µg/L.

Daphniidae individuals generally made the dominant contribution to biomass in each waterbody, with the second–greatest contribution to biomass typically made by either Bosminidae, Calanoida or Cyclopoida (Figure 16, Figure 17, Figure 18). Considering all samples, the mean contributions that these taxa made to total sample biomass were: 55% (Daphniidae), 11% (Bosminidae), 8% (Calanoida), and 4% (Cyclopoida). There was no clear seasonal pattern in zooplankton biomass, although zooplankton biomass was generally lower for the samples collected in early-mid September during Year 2, relative to the samples collected earlier in the growing season (Figure 17). For individual waterbodies, zooplankton composition at each site was generally similar on individual dates, although there were some marked differences in biomass among sites, e.g., compare June samples collected at Lower Campbell Reservoir (Figure 17). This variability supported our approach of sampling multiple sites on each waterbody (except for Middle Quinsam and Gooseneck lakes, where multiple replicates from a single central lake site were analyzed).

For context, the mean zooplankton biomass measured for waterbodies in this study (11.2–66.8 µg/L; Figure 15) was at the low end of the range for a dataset of 49 lakes (10.7–786 µg/L) compiled by Hanson *et al.* (1984), which spanned a wide range of trophic states and had a geometric mean of 88.0 µg/L. This likely reflects the relatively low productivity of the

JHTMON-5 study lakes, as is typical of lakes on Vancouver Island. Zooplankton biomass measured in this study was higher than mean annual measurements for Sproat Lake on Vancouver Island (6.1–10.1 µg/L), and generally comparable with mean annual measurements for Quesnel Lake, interior BC (18.6–24.6 µg/L), as reported by Stockner and Shortreed (1989).

Figure 15. Mean total zooplankton biomass in JHTMON-5 study lakes. The plot shows mean biomass measured at three sites for three dates during a single growing season at each waterbody. Whiskers denote the range, boxes denote the inter-quartile range and bold horizontal lines denote medians.

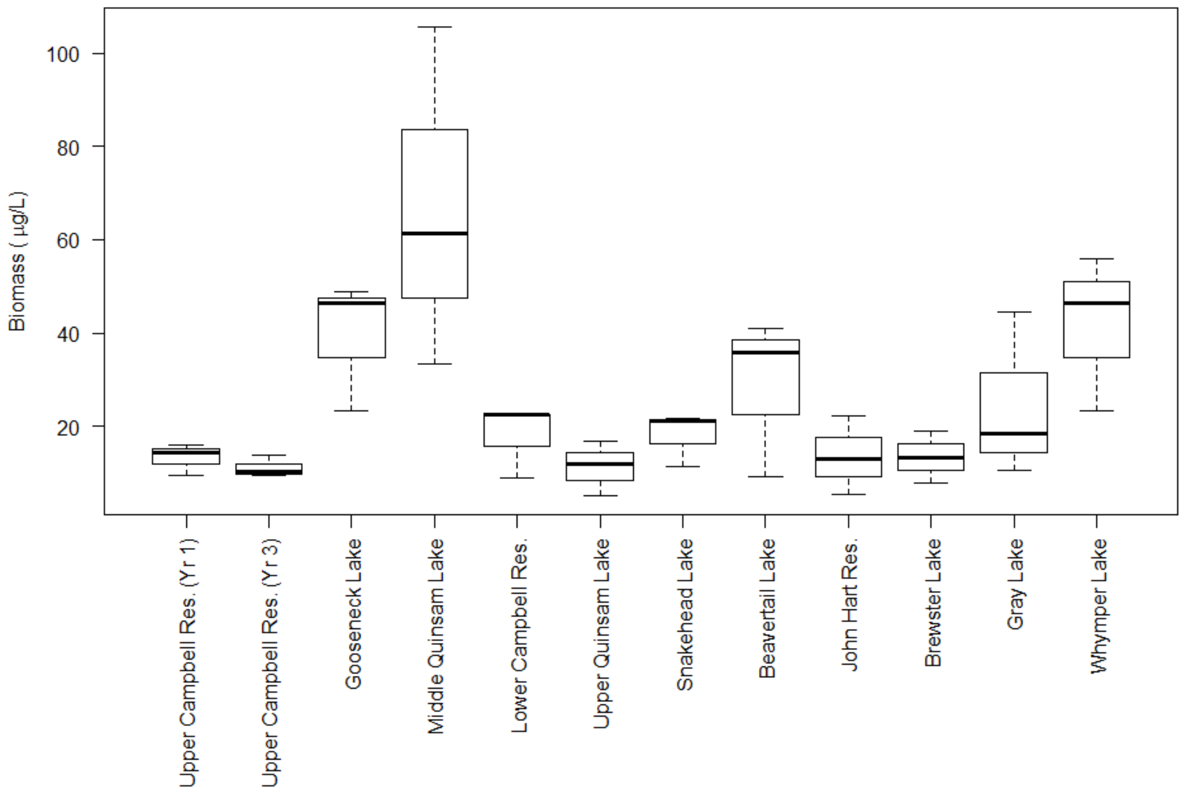


Figure 16. Zooplankton biomass for Year 1 samples. Lower case letters in site descriptions denote replicates.

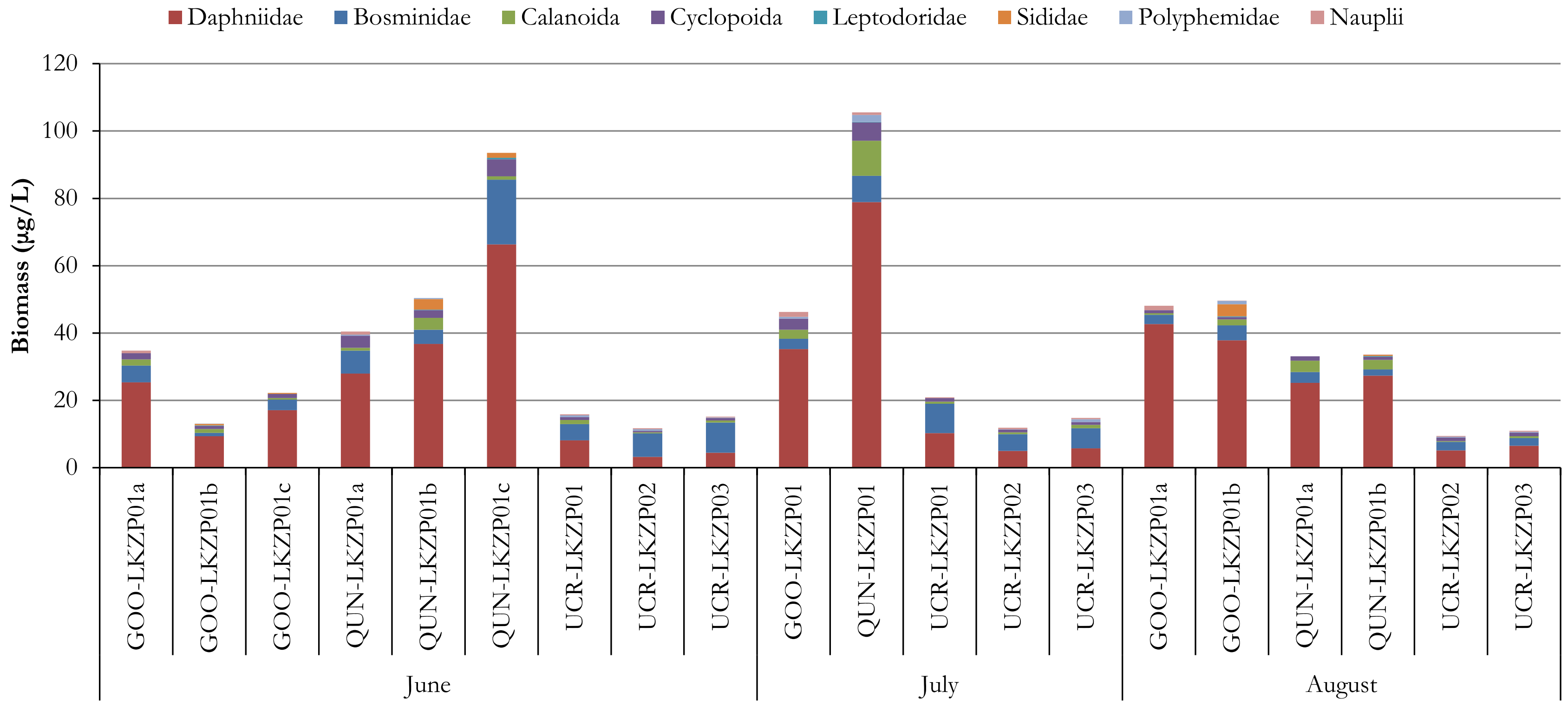


Figure 17. Zooplankton biomass for Year 2 samples.

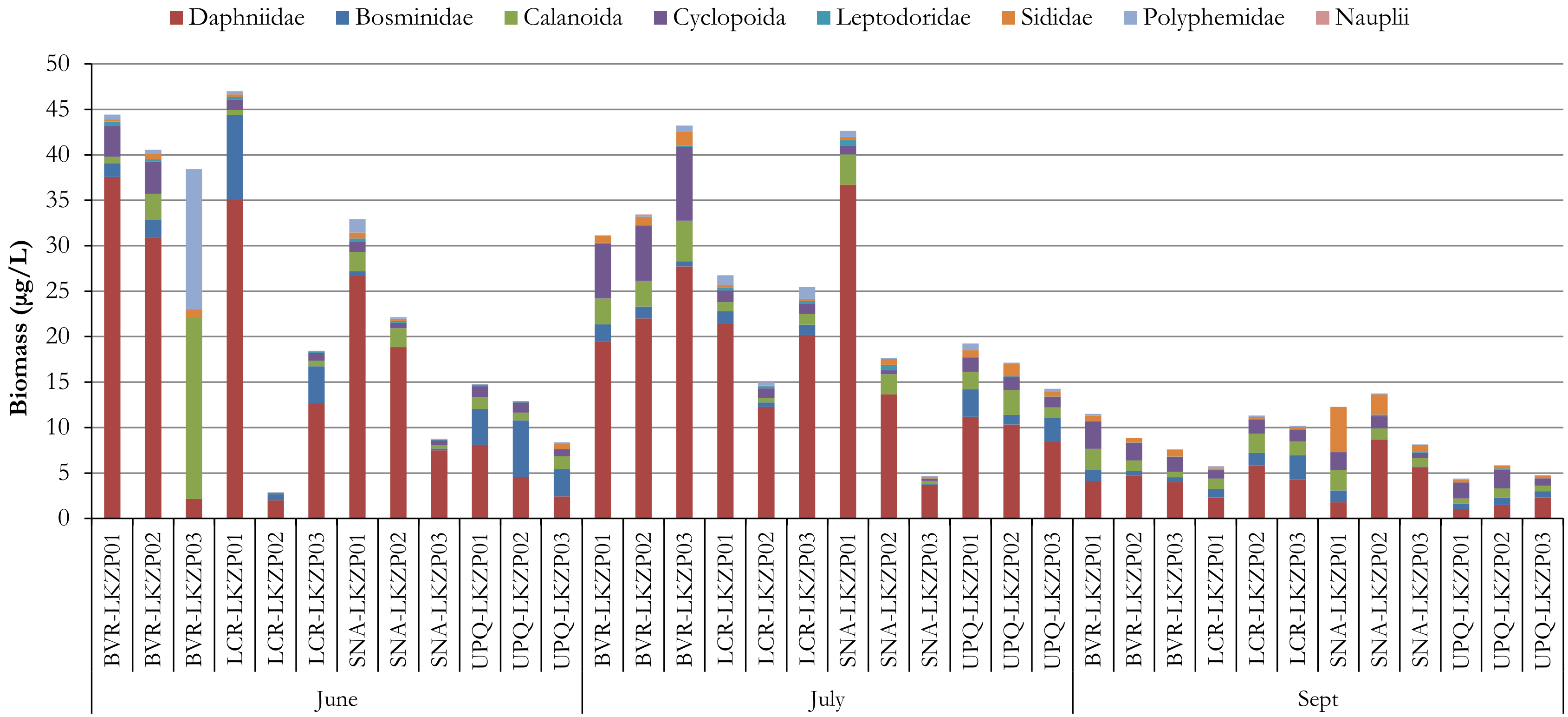
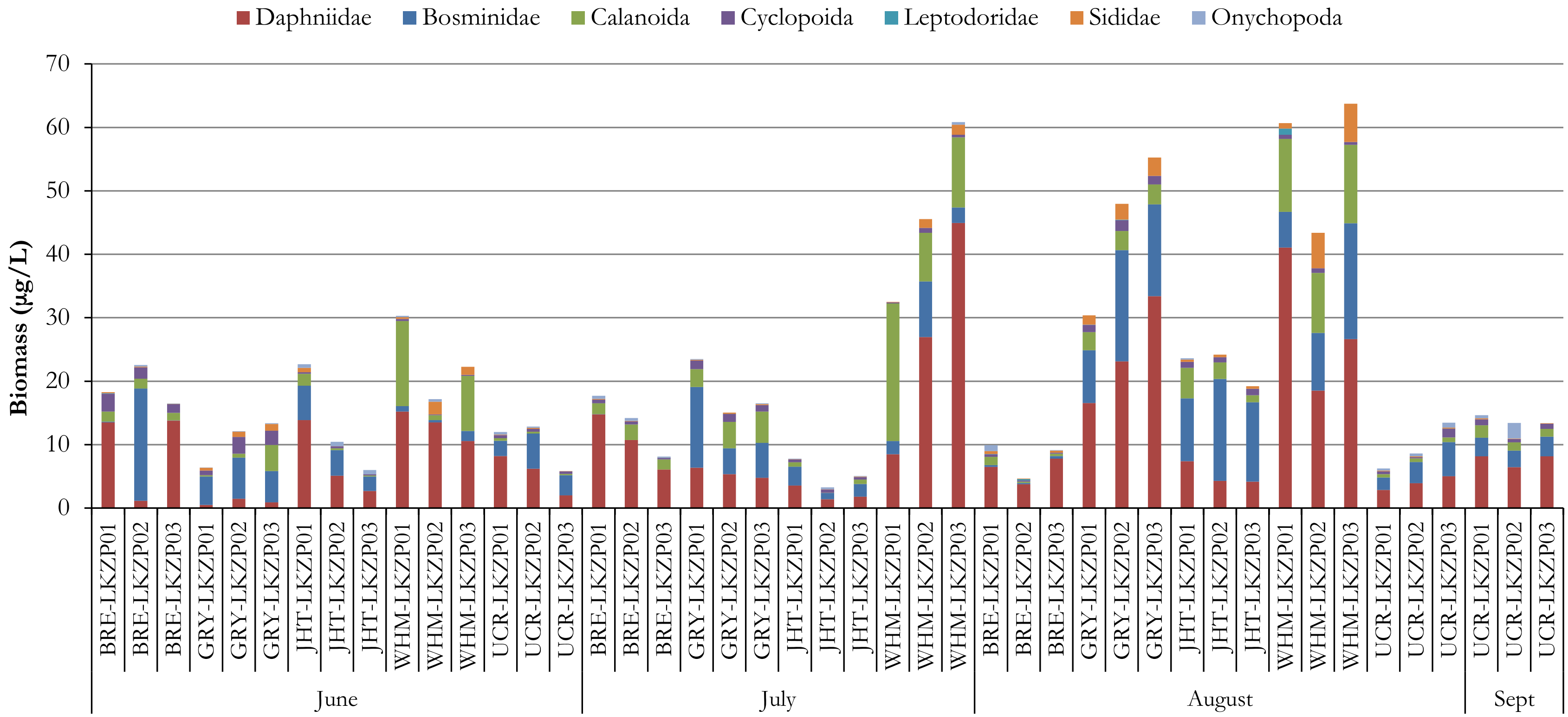


Figure 18. Zooplankton biomass for Year 3 samples.



3.1.2. Littoral Invertebrates

The abundance of littoral invertebrates sampled at all lakes is summarized in Table 24. When all samples are combined, the most abundant orders were: Hemiptera (true bugs; 21%), Amphipoda (amphipods; 18%), Odonata (dragonflies and damselflies; 11%) and Diptera (true flies; 10%). Composition varied substantially between samples.

3.1.3. Stream Invertebrates

The abundance of stream invertebrates sampled at all lakes during July is summarized in Table 25. When all samples are combined, the most abundant orders were: Coleoptera (beetles; 26%), Diptera (16%), Ephemeroptera (mayflies; 13%), Plecoptera (stoneflies; 12%), Trichoptera (caddisflies; 12%) and Hemiptera (8%). Composition varied substantially among samples.

3.1.4. Terrestrial Invertebrates

Terrestrial invertebrates sampled at all lakes are summarized in Table 26. When all samples are combined, the most abundant order was Diptera (76%), which encompassed multiple families including: Syrphridae (hover fly), Muscidae (house fly), Lonchopteridae (spear wing fly), Chironomidae (midge), Calliphoridae (blow fly), Empididae (dance fly), Ephydridae (shore fly), Simuliidae (black fly), Pipunculidae (big-headed fly) and Tabanidae (horse fly). Hymenoptera (sawflies, wasps, bees and ant) comprised 16% of all individuals. The remaining nine orders that were enumerated each comprised ≤1% of the individuals that were sampled.

Table 24. Littoral invertebrate abundance.

Waterbody	Site	Month	Rep.	Abundance (individuals/sample)											
				Amphipoda	Bivalvia	Coleoptera	Diptera	Ephemeroptera	Gastropoda	Hemiptera	Odonata	Physidae	Pulomonata	Trichoptera	Other
Upper Campbell Reservoir	UCR-LKIV01	July	1	0	0	0	1	3	0	0	0	0	0	0	0
		August	1	1	1	0	1	1	0	0	3	0	2	2	0
Gooseneck Lake	GOO-LKIV01	July		32	0	2	3	0	0	2	2	0	0	2	0
		August		0	0	0	0	0	0	0	2	0	0	0	0
Middle Quinsam Lake	QUN-LKIV01	July	1	4	0	0	1	0	0	0	2	0	0	0	0
		August	1	14	0	0	1	3	0	0	1	0	0	0	1
Lower Campbell Reservoir	LCR-BIV01	July	1	15	0	0	9	0	0	3	0	0	0	0	2
			2	9	0	0	2	0	0	2	1	0	0	0	2
Beavertail Lake	BVR-BIV01	July	1	3	0	8	6	1	5	9	0	0	0	4	2
			2	0	1	7	0	0	3	7	4	0	0	0	3
			3	0	2	5	1	2	5	18	2	0	0	0	1
Snakehead Lake	SNA-BIV01	July	1	2	0	2	1	0	0	0	6	0	0	0	0
			2	0	0	0	1	2	0	0	3	0	0	0	4
			3	6	1	0	10	4	0	1	1	0	0	0	1
Upper Quinsam Lake	UPQ-BIV01	July	1	0	0	2	0	0	0	18	1	0	9	0	1
			2	0	0	1	4	0	0	9	1	0	0	0	1
			3	0	0	0	3	0	0	20	0	0	3	0	0
John Hart Reservoir	JHT-BIV01	June	1	4	0	2	14	2	0	9	9	4	0	0	0
		July	1	7	1	2	2	8	0	5	8	3	0	0	4
Brewster Lake	BRE-BIV01	June	1	3	1	5	3	4	0	7	8	3	0	2	0
		July	1	0	0	2	1	14	0	9	7	12	0	2	2
Whymper Lake	WHM-BIV01	June	1	6	2	1	0	1	0	5	2	9	0	3	0
		July	1	11	3	9	3	0	0	2	3	9	0	8	1
Gray Lake	GRY-BIV01	June	1	7	0	3	3	4	0	7	5	3	0	5	5
		July	1	0	1	4	0	0	0	10	4	12	0	7	2
Total abundance (%)				**18**	**2**	**8**	**10**	**7**	**2**	**21**	**11**	**8**	**2**	**5**	**5**

Table 25. Stream invertebrate abundance.

Waterbody	Site	Rep.	Abundance (individuals/sample)												
			Amphipoda	Araneae	Bivalvia	Coleoptera	Diptera	Ephem-eroptera	Gastropoda	Hemiptera	Odonata	Oligo-chaeta	Plec-optera	Trich-optera	Other
Upper Campbell Reservoir	UCR-SIV01	1	0	0	0	1	3	17	0	0	0	1	10	3	0
		2	0	0	0	0	2	10	0	0	0	1	8	4	1
Gooseneck Lake	GOO-SIV01	1	1	0	0	20	0	3	7	12	4	0	1	0	2
		2	2	0	0	9	0	0	2	11	5	0	0	0	0
Middle Quinsam Lake	QUN-SIV01	1	1	0	0	0	0	4	0	0	11	4	7	1	0
		2	0	0	0	0	0	2	0	0	3	2	2	0	0
Lower Campbell Reservoir	LCR-SIV01	1	0	0	0	0	5	0	0	0	5	0	0	2	0
		2	0	0	0	1	1	12	0	1	0	0	4	0	0
		3	0	0	0	1	0	0	0	0	1	0	4	2	0
Beavertail Lake	BVR-SIV01	1	0	0	1	0	2	5	4	3	0	1	0	1	0
		2	0	0	1	1	12	7	0	0	2	0	13	14	4
		3	0	0	0	0	9	16	0	1	3	0	8	8	1
Snakehead Lake	SNA-SIV01	1	0	0	0	45	1	0	0	0	0	0	0	0	0
		2	0	0	0	36	4	0	1	0	0	0	1	0	0
		3	0	0	0	38	0	0	0	0	0	0	0	0	0
Upper Quinsam Lake	UPQ-SIV01	1	0	1	0	0	0	2	0	12	0	0	0	0	2
		2	0	0	0	6	14	0	0	2	1	0	0	0	0
		3	0	1	0	0	2	0	0	3	0	0	0	0	0
John Hart Reservoir	JHT-SIV01	1	0	2	0	1	9	0	0	2	1	0	10	5	0
Whymper Lake	WHM-SIV01	1	0	0	0	1	9	0	0	0	0	0	0	20	0
Brewster Lake	BRE-SIV01	1	0	0	0	3	9	3	0	2	1	0	5	5	1
Gray Lake	GRY-SIV01	1	0	0	0	0	20	0	0	0	1	0	0	9	0
Total abundance (%)			**1**	**1**	**<1%**	**26**	**16**	**13**	**2**	**8**	**6**	**1**	**12**	**12**	**2**

Table 26. Terrestrial invertebrate abundance.

Waterbody	Site	Month	Abundance (individuals/sample)										
			Arachnid	Coleoptera	Diptera	Ephemer-optera	Hemiptera	Homoptera	Hymen-optera	Neuroptera	Odonata	Orthoptera	Trich-optera
Upper Campbell Reservoir	UCR-TIV01	July	0	0	28	3	0	0	1	0	0	0	0
Middle Quinsam Lake	QUN-LKIV01	July	0	0	34	0	0	6	10	0	0	0	0
Gooseneck Lake	GOO-LKIV01	July	0	0	53	0	2	0	2	0	0	0	0
Lower Campbell Reservoir	LCR-TIV01	June	0	3	13	0	1	0	0	0	0	0	0
		July	0	0	8	0	0	0	1	0	1	0	0
		Sept	0	0	23	0	0	0	1	0	0	0	0
Beavertail Lake	BVR-TIV01	June	0	0	20	0	0	0	1	0	0	0	0
		July	0	0	8	0	2	0	2	1	0	0	0
		Sept	0	0	6	0	0	0	3	0	0	0	0
Snakehead Lake	SNA-TIV01	June	0	0	11	0	0	0	14	0	0	0	0
		July	0	0	6	0	0	0	9	0	0	0	0
		Sept	5	0	0	0	0	0	1	0	0	0	1
Upper Quinsam Lake	UPQ-TIV01	June	1	0	5	0	0	0	0	0	0	0	0
		July	0	0	5	0	0	0	1	0	0	1	0
		Sept	0	0	15	0	0	0	1	0	0	0	0
John Hart Reservoir	JHT-TIV01	June	0	0	3	0	0	0	5	0	0	0	0
		July	0	1	11	0	0	0	2	0	1	0	0
Whymper Lake	WHM-TIV01	June	0	0	52	0	0	0	4	0	2	0	0
		July	0	0	40	0	0	0	10	0	0	0	0
Brewster Lake	BRE-TIV01	June	0	0	34	0	0	0	5	0	0	0	0
		July	0	0	19	0	0	0	8	0	0	0	0
Gray Lake	GRY-TIV01	June	0	2	5	0	0	0	0	0	1	0	7
		July	0	0	8	0	0	0	5	0	0	0	0
Total abundance (%)			**1**	**1**	**76**	**1**	**1**	**1**	**16**	**<1%**	**1**	**<1%**	**1**

3.2. Fish Sampling

3.2.1. Gill Netting

Gill netting catch (# of fish) and catch-per-unit-effort (CPUE) by species and lake from Year 3 is shown in Table 27. Across all years, the average CPUE for Cutthroat Trout was highest at Snakehead Lake and Middle Quinsam Lake, and lowest in both Upper Campbell and Lower Campbell reservoirs (Figure 19). In contrast, the CPUE for Rainbow Trout was highest in Upper Campbell and Lower Campbell Reservoirs (Figure 19). Rainbow Trout were also present in the largest lakes sampled (e.g., Brewster and Beavertail lakes) but were absent or rare in many of the smaller lakes (e.g., Middle Quinsam, Gooseneck and Snakehead lakes). Dolly Varden was also generally more abundant in the largest lakes, including Upper Campbell and Lower Campbell reservoirs, and Beavertail, Brewster and Gooseneck lakes. Kokanee were only found in two lakes of the Salmon River diversion system, Brewster Lake and Gray Lake.

In lakes where both Cutthroat Trout and Rainbow Trout were caught, Cutthroat Trout have higher CPUE in gill nets set in littoral habitats compared to pelagic habitats (Figure 20a). In contrast for Rainbow Trout, either roughly equal or higher CPUE was observed in pelagic sets than littoral sets (Figure 20b). These catch results highlight a spatial separation between these two species.

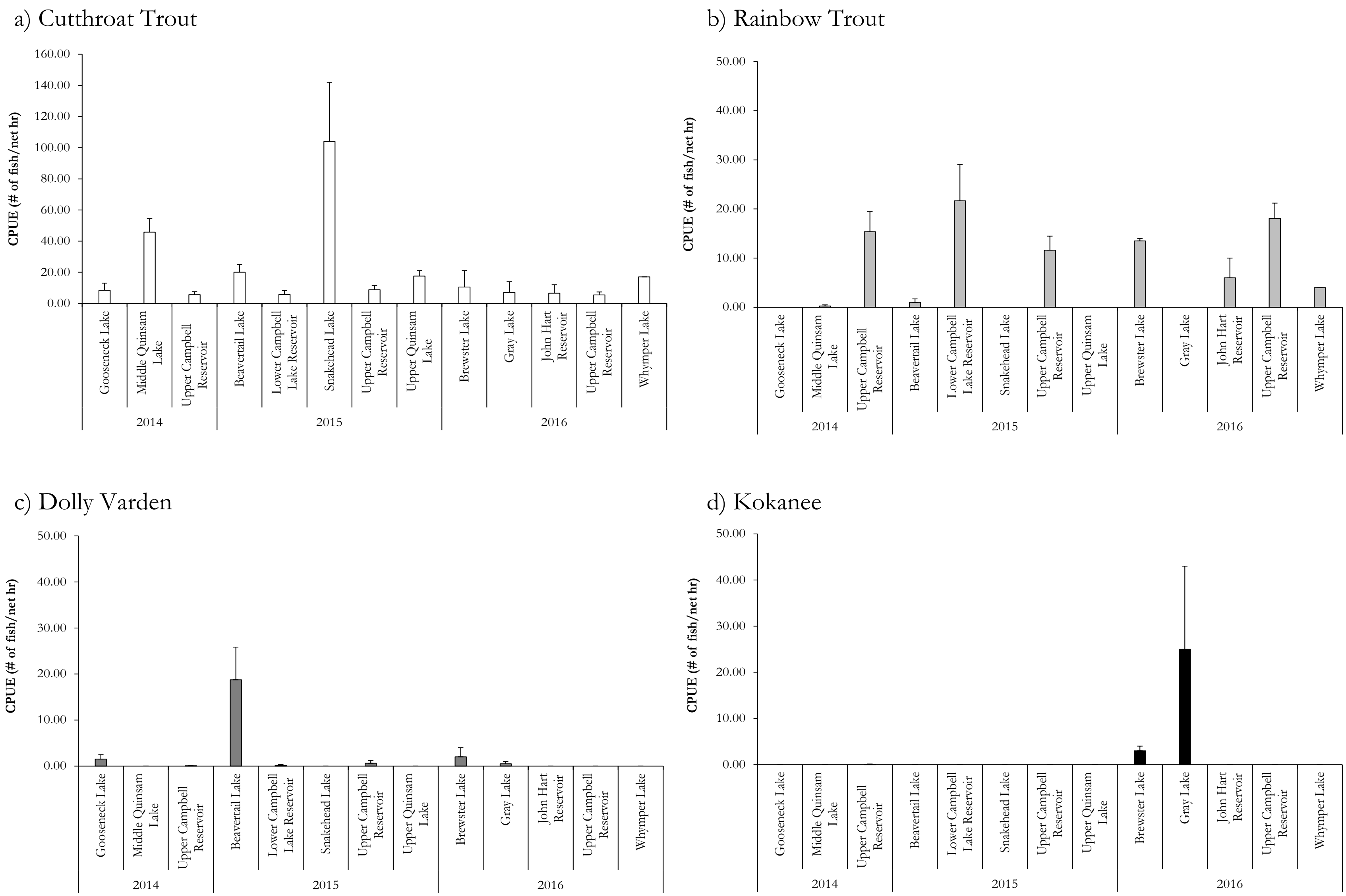
Table 27. Gill netting capture results from the Gray, Whymper, and Brewster Lakes and John Hart and Upper Campbell reservoirs, 2016.

Waterbody	Site	Sampling Date	Set Number	Gill Netting Effort (hrs)	Gill Net Catch (# of fish)[1]							Gill Net CPUE (# of fish/net hr)[1]						
					CT	RB	DV	CC	KO	CT/RB	TSB	CT	RB	DV	CC	KO	CT/RB	TSB
Gray Lake	GRY-LKGN01	21-Aug-16	1	18.7	0	0	0	0	43	0	0	0.00	0.00	0.00	0.00	2.30	0.00	0.00
	GRY-LKGN02	17-Aug-16	1	21.0	14	0	1	1	7	0	0	0.67	0.00	0.05	0.05	0.33	0.00	0.00
			Total	**39.7**	**14**	**0**	**1**	**1**	**50**	**0**	**0**	**0.35**	**0.00**	**0.03**	**0.03**	**1.26**	**0.00**	**0.00**
			Average	**19.8**	**7**	**0**	**0.5**	**0.5**	**25**	**0**	**0**	**0.33**	**0.00**	**0.02**	**0.02**	**1.31**	**0.00**	**0.00**
			SD[2]	**1.6**	**9.9**	**0**	**0.7**	**0.7**	**25.5**	**0**	**0**	**0.47**	**0.00**	**0.03**	**0.03**	**1.39**	**0.00**	**0.00**
Whymper Lake	WHM-LKGN01	15-Aug-16	1	22.8	17	4	0	0	0	0	0	0.75	0.18	0.00	0.00	0.00	0.00	0.00
			Total	**22.8**	**17**	**4**	**0**	**0**	**0**	**0**	**0**	**0.75**	**0.18**	**0**	**0**	**0**	**0**	**0**
			Average	**22.8**	**17**	**4**	**0**	**0**	**0**	**0**	**0**	**0.75**	**0.18**	**0**	**0**	**0**	**0**	**0**
			SD[2]	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**
Brewster Lake	BRE-LKGN01	19-Aug-16	1	17.8	0	14	4	0	4	1	0	0.00	0.79	0.23	0.00	0.23	0.06	0.00
	BRE-LKGN02	19-Aug-16	1	18.2	21	13	0	1	2	0	0	1.15	0.71	0.00	0.05	0.11	0.00	0.00
			Total	**36.0**	**21**	**27**	**4**	**1**	**6**	**1**	**0**	**0.58**	**0.75**	**0.11**	**0.03**	**0.17**	**0.03**	**0.00**
			Average	**18.0**	**10.5**	**13.5**	**2**	**0.5**	**3**	**0.5**	**0**	**0.58**	**0.75**	**0.11**	**0.03**	**0.17**	**0.03**	**0.00**
			SD[2]	**0.3**	**14.8**	**0.7**	**2.8**	**0.7**	**1.4**	**0.7**	**0.0**	**0.81**	**0.05**	**0.16**	**0.04**	**0.08**	**0.04**	**0.00**
John Hart Reservoir	JHT-LKGN01	21-Aug-16	1	18.9	1	10	0	0	0	0	0	0.05	0.53	0.00	0.00	0.00	0.00	0.00
	JHT-LKGN02	21-Aug-16	1	18.0	12	2	0	1	0	0	0	0.67	0.11	0.00	0.06	0.00	0.00	0.00
			Total	**36.9**	**13**	**12**	**0**	**1**	**0**	**0**	**0**	**0.35**	**0.32**	**0.00**	**0.03**	**0.00**	**0.00**	**0.00**
			Average	**18.5**	**6.5**	**6**	**0**	**0.5**	**0**	**0**	**0**	**0.36**	**0.32**	**0.00**	**0.03**	**0.00**	**0.00**	**0.00**
			SD[2]	**0.6**	**7.8**	**5.7**	**0.0**	**0.7**	**0.0**	**0.0**	**0.0**	**0.43**	**0.30**	**0.00**	**0.04**	**0.00**	**0.00**	**0.00**
Upper Campbell Reservoir	UCR-LKGN01	29-Aug-16	1	23.2	7	16	0	5	0	0	0	**0.30**	**0.69**	**0.00**	**0.22**	**0.00**	**0.00**	**0.00**
			2	22.9	0	28	0	0	0	0	0	**0.00**	**1.22**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
	UCR-LKGN02	29-Aug-16	1	22.7	4	23	0	0	0	0	0	**0.18**	**1.01**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
			2	22.8	0	0	0	0	0	0	0	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
	UCR-LKGN04	30-Aug-16	1	18.7	6	5	0	0	0	0	0	**0.32**	**0.27**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
			2	18.0	0	26	0	0	0	0	0	**0.00**	**1.44**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
	UCR-LKGN06	31-Aug-16	1	19.6	3	26	0	0	0	0	0	**0.15**	**1.32**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
			2	19.0	0	26	0	1	0	0	0	**0.00**	**1.37**	**0.00**	**0.05**	**0.00**	**0.00**	**0.00**
	UCR-LKGN07	31-Aug-16	1	21.9	16	34	0	0	0	0	0	**0.73**	**1.55**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
			2	21.4	0	9	0	0	0	0	0	**0.00**	**0.42**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
	UCR-LKGN08	30-Aug-16	1	18.6	20	17	0	0	0	2	0	**1.08**	**0.92**	**0.00**	**0.00**	**0.00**	**0.11**	**0.00**
			2	17.8	0	0	0	0	0	0	0	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**
			Total	**246.6**	**56**	**210**	**0**	**6**	**0**	**2**	**0**	**2.76**	**10.22**	**0.00**	**0.27**	**0.00**	**0.11**	**0.00**
			Average	**20.6**	**4.7**	**17.5**	**0.0**	**0.5**	**0.0**	**0.2**	**0.0**	**0.23**	**0.85**	**0.00**	**0.02**	**0.00**	**0.01**	**0.00**
			SD[2]	**2.1**	**6.8**	**11.6**	**0.0**	**1.4**	**0.0**	**0.6**	**0.0**	**0.34**	**0.56**	**0.00**	**0.06**	**0.00**	**0.03**	**0.00**

[1] CT- Cutthroat Trout, RB - Rainbow Trout, DV - Dolly Varden, CC - Sculpin general, KO - Kokanee, CT/RB - Cutthroat Trout/Rainbow Trout hybrid, TSB - Threespine Stickleback

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

Figure 19. Catch-per-unit-effort (CPUE) during gill netting from all lakes sampled in Years 1, 2, and 3 of the JHTMON-5 Program for a) Cutthroat Trout, b) Rainbow Trout, c) Dolly Varden, and d) Kokanee.

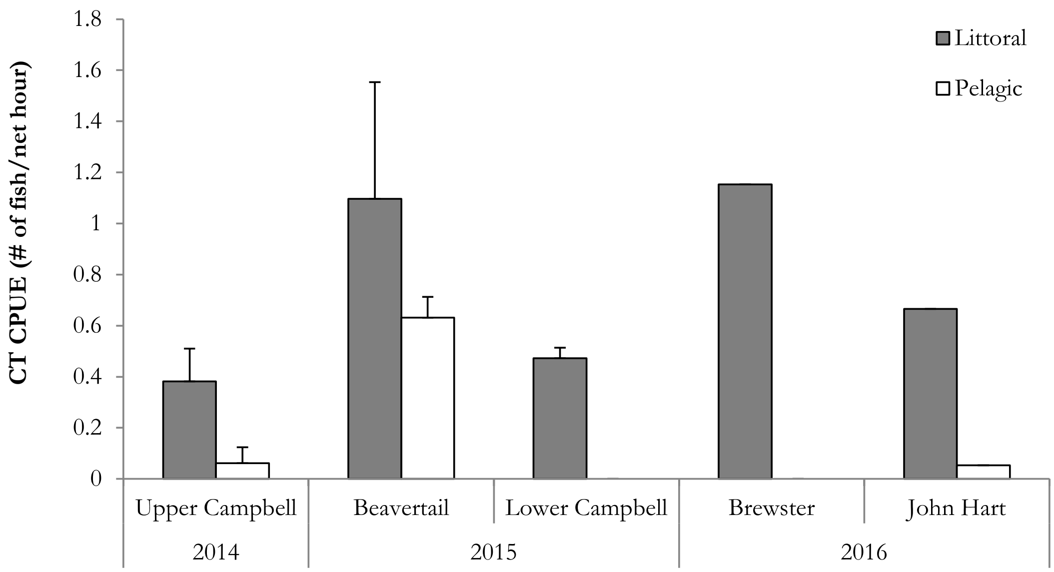


Laich-Kwil-Tach

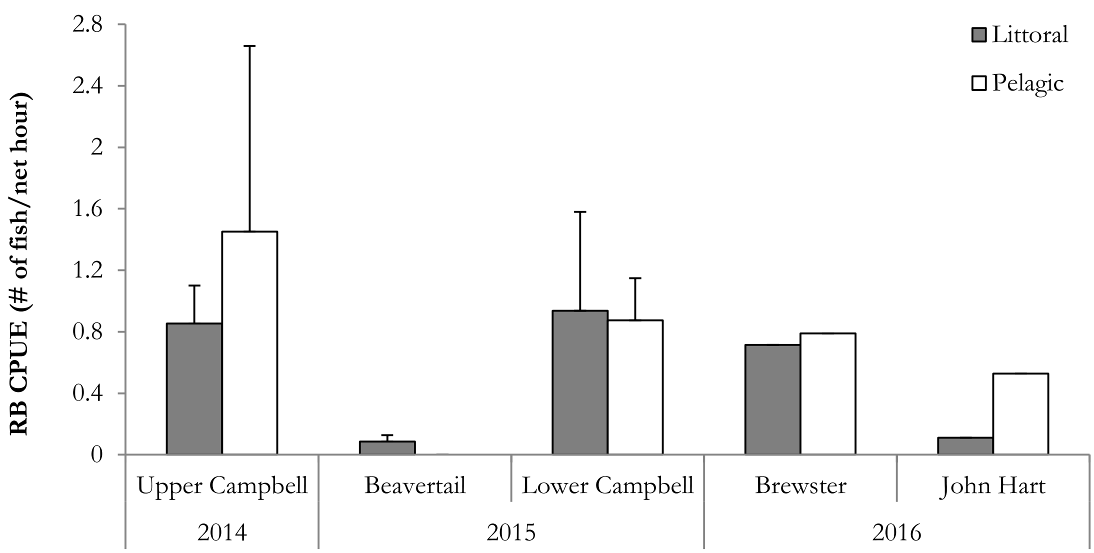
ECOFISH RESEARCH

Figure 20. Comparison of Catch-per-unit-effort (CPUE) from Upper Campbell Reservoir, Beavertail Lake, Lower Campbell Reservoir, Brewster Lake, and John Hart Reservoir across littoral and pelagic sets in 2014 to 2016 for a) Cutthroat Trout and b) Rainbow Trout.

a) Cutthroat Trout



b) Rainbow Trout



3.2.2. Trap Netting

The primary goal of the trap netting was to capture representative samples of Threespine Stickleback. Minnow trapping methods failed to catch Threespine Stickleback in 2014 and therefore trap netting was initiated in Year 2 at all lakes. Trap netting catch (# of fish) and catch-per-unit-effort (CPUE) by species and lake from Year 3 is shown in Table 28.

Across all years, the average CPUE for Threespine Stickleback was highest in Upper Campbell Reservoir and John Hart Reservoir (Figure 21a). Threespine Stickleback were present in all three reservoirs and in all lakes of the Salmon River diversion. No Threespine Stickleback were caught in the Quinsam River diversion system or in Beavertail Lake.

Trap netting was also successful in catching Sculpin spp. (Figure 21b). Sculpin were identified as Prickly Sculpin (*Cottus asper*) or Coastrange Sculpin (*Cottus aleuticus*), although not all individuals were identified to species. Therefore, all captured sculpin were categorized as Sculpin spp. (CC). Across all years, the average CPUE for Sculpin spp was highest in Upper Campbell Reservoir, John Hart Reservoir and the lakes of the Salmon River diversion. Lower Sculpin spp. catch occurred in the lakes of the Quinsam diversion and in Lower Campbell Reservoir.

Table 28. Trap netting capture results from Gray, Whymper, and Brewster lakes and Upper Campbell and John Hart reservoirs, 2016.

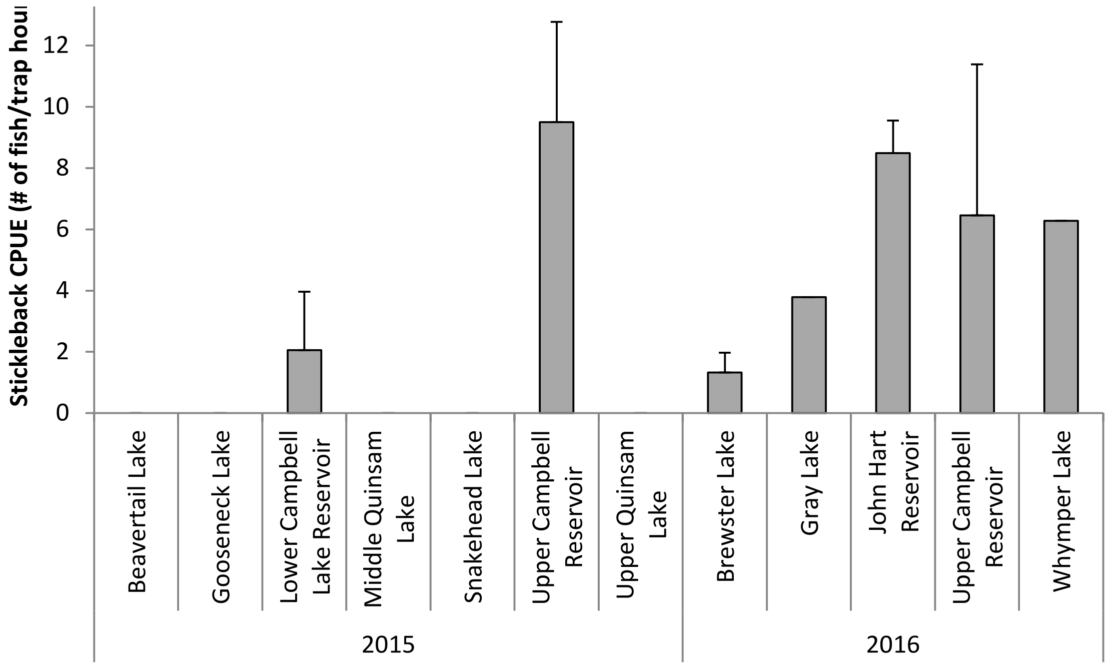
Waterbody	Site	Sampling Date	No. of Sets	Trap Netting Effort (hrs)	Trap Net Catch (# of fish)[1]						Trap Net CPUE (# of fish/net hr)[1]					
					CT	RB	DV	CC	TSB	CT/RB	CT	RB	DV	CC	TSB	CT/RB
Gray Lake	GRY-LKTN01	17-Aug-16	1	22.97	1	3	0	10	87	0	0.04	0.13	0.00	0.44	3.79	0.00
			Total	**22.97**	**1**	**3**	**0**	**10**	**87**	**0**	**0.04**	**0.13**	**0.00**	**0.44**	**3.79**	**0.00**
			Average	**22.97**	**1.0**	**3.0**	**0.0**	**10.0**	**87.0**	**0.0**	**0.04**	**0.13**	**0.00**	**0.44**	**3.79**	**0.00**
			SD	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**
Whymper Lake	WHM-LKTN01	15-Aug-16	1	22.1	3	4	0	13	139	0	0.14	0.18	0.00	0.59	6.28	0.00
			Total	**22.13**	**3**	**4**	**0**	**13**	**139**	**0**	**0.14**	**0.18**	**0.00**	**0.59**	**6.28**	**0.00**
			Average	**22.13**	**3.0**	**4.0**	**0.0**	**13.0**	**139.0**	**0.0**	**0.14**	**0.18**	**0.00**	**0.59**	**6.28**	**0.00**
			SD[2]	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**
Brewster Lake	BRE-LKTN01	19-Aug-16	1	21.8	1	2	0	15	43	0	0.05	0.09	0.00	0.69	1.98	0.00
	BRE-LKTN02	19-Aug-16	1	20.5	5	0	0	5	14	1	0.24	0.00	0.00	0.24	0.68	0.05
			Total	**42.2**	**6**	**2**	**0**	**20**	**57**	**1**	**0.14**	**0.05**	**0.00**	**0.47**	**1.35**	**0.02**
			Average	**21.1**	**3.0**	**1.0**	**0.0**	**10.0**	**28.5**	**0.5**	**0.15**	**0.05**	**0.00**	**0.47**	**1.33**	**0.02**
			SD	**0.9**	**2.8**	**1.4**	**0.0**	**7.1**	**20.5**	**0.7**	**0.14**	**0.07**	**0.00**	**0.32**	**0.91**	**0.03**
Upper Campbell Reservoir	UCR-LKTN09	01-Sep-16	1	17.0	4	2	0	6	193	0	0.24	0.12	0.00	0.35	11.39	0.00
	UCR-LKTN10	01-Sep-16	1	18.3	4	2	0	7	28	0	0.22	0.11	0.00	0.38	1.53	0.00
			Total	**16.95**	**8**	**4**	**0**	**13**	**221**	**0**	**0.45**	**0.23**	**0.00**	**0.74**	**12.91**	**0.00**
			Average	**16.95**	**4.0**	**2.0**	**0.0**	**6.5**	**110.5**	**0.0**	**0.24**	**0.12**	**0.00**	**0.35**	**11.39**	**0.00**
			SD	**0.98**	**0.0**	**0.0**	**0.0**	**0.7**	**116.7**	**0.0**	**0.01**	**0.01**	**0.00**	**0.02**	**6.97**	**0.00**
John Hart Reservoir	JHT-LKTN01	21-Aug-16	1	26.0	5	0	0	24	193	0	0	0.00	0.00	0.92	7.41	0.00
	JHT-LKTN02	21-Aug-16	1	25.3	4	0	0	13	242	0	0.16	0.00	0.00	0.51	9.55	0.00
			Total	**51.4**	**9**	**0**	**0**	**37**	**435**	**0**	**0.18**	**0.00**	**0.00**	**0.72**	**8.47**	**0.00**
			Average	**25.7**	**4.5**	**0.0**	**0.0**	**18.5**	**218**	**0.0**	**0.08**	**0.00**	**0.00**	**0.72**	**8.48**	**0.00**
			SD	**0.5**	**0.7**	**0.0**	**0.0**	**7.8**	**34.6**	**0.0**	**0.11**	**0.00**	**0.00**	**0.29**	**1.51**	**0.00**

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

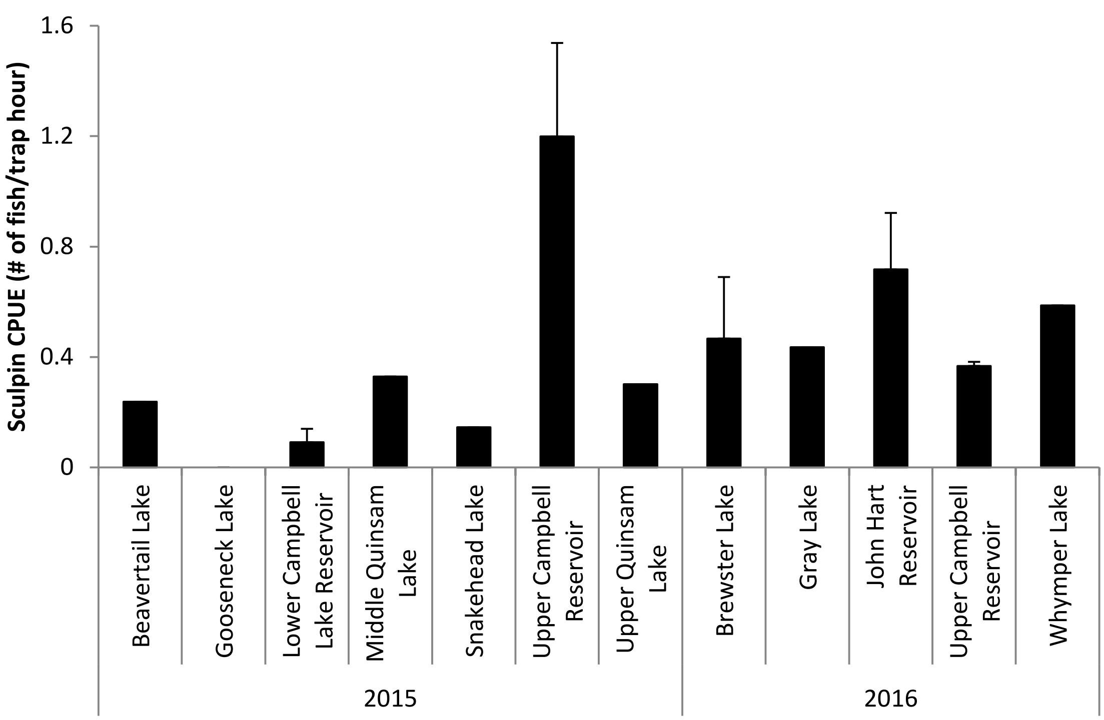
[2] No standard deviation is calculated if only one sample occurs.

Figure 21. Catch-per-unit-effort (CPUE) during trap netting from all lakes sampled in Year 2 and 3 of the JHTMON-5 Program for a) Threespine Stickleback and b) Sculpin spp.

a) Threespine Stickleback



b) Sculpin spp.



3.2.3. Minnow Trapping

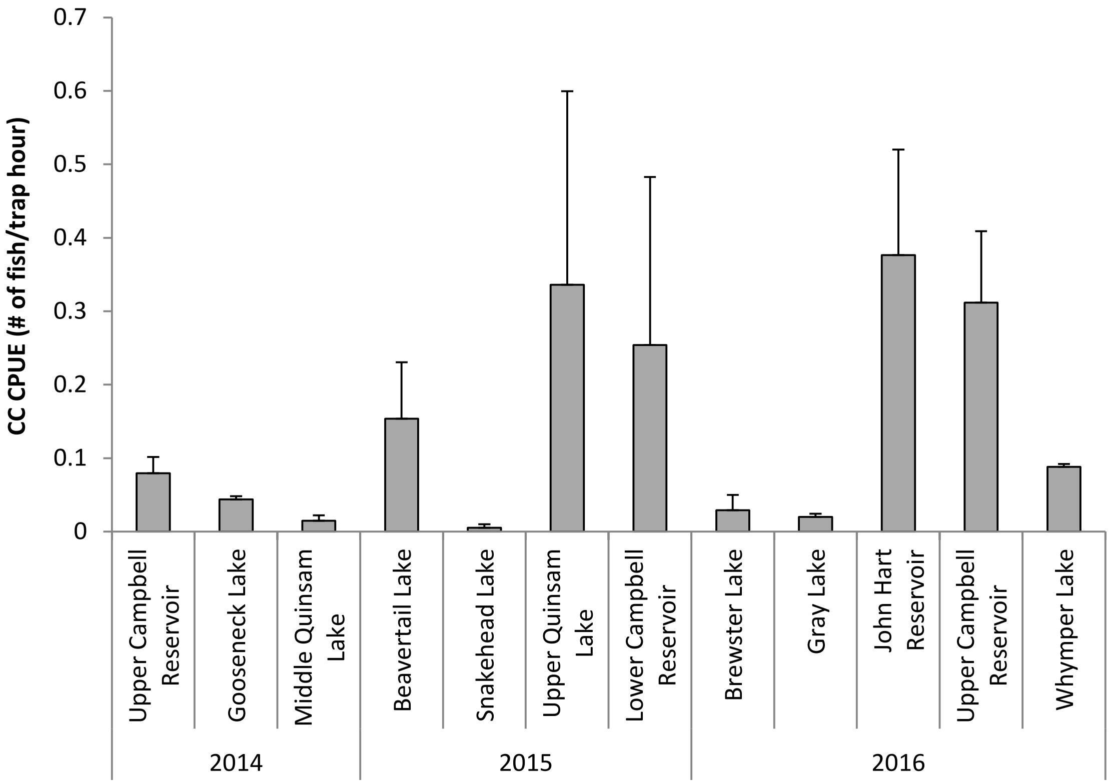
Sculpin spp. and Threespine Stickleback were the only two species captured using minnow traps in Year 3 sampling (Table 29). Sculpin spp. were caught in all lakes and reservoirs using minnow traps across all three years (Figure 22). Sculpin spp. CPUE was highest in some of the largest lakes and reservoirs (e.g., Upper Quinsam and Beavertail lakes, Upper and Lower Campbell reservoirs) and was the lowest in several of the smallest lakes (e.g., Middle Quinsam and Snakehead lakes).

Table 29. Minnow trapping capture results from Grey, Whymper, Brewster lakes and Upper Campbell and John Hart reservoirs, 2016.

Waterbody	Site	Sampling Date	No. of Minnow Traps	Minnow Trapping Effort (hrs)	Minnow Trapping Catch (Number of fish)[1]		Minnow Trapping CPUE (Number of fish/trap hr)[1]	
					CC	TSB	CC	TSB
Gray Lake	GRY-LKMT02	17-Aug-16	5	123.2	3	0	0.02	0.00
	GRY-LKMT01	17-Aug-16	5	129.3	2	10	0.02	0.08
		Total	**10**	**252.6**	**5**	**10**	**0.02**	**0.04**
		Average	**5**	**126.3**	**3**	**5**	**0.02**	**0.04**
		SD	**n/a**	**4.3**	**0.7**	**7.1**	**0.01**	**0.05**
Whymper Lake	WHM-LKMT02	15-Aug-16	5	119.4	11	7	0.09	0.06
	WHM-LKMT01	15-Aug-16	5	119.4	10	43	0.08	0.36
		Total	**10**	**238.8**	**21**	**50**	**0.09**	**0.21**
		Average	**5**	**119.4**	**11**	**25**	**0.09**	**0.21**
		SD	**n/a**	**0.1**	**0.7**	**25.5**	**0.01**	**0.21**
Brewster Lake	BRE-LKMT02	19-Aug-16	5	120.3	6	3	0.05	0.02
	BRE-LKMT01	19-Aug-16	5	119.5	1	0	0.01	0.00
		Total	**10**	**239.9**	**7**	**3**	**0.03**	**0.01**
		Average	**5**	**120**	**4**	**2**	**0.03**	**0.01**
		SD	**n/a**	**85.0**	**3.7**	**15.9**	**0.03**	**0.13**
Upper Campbell Reservoir	UCR-LKMT09	01-Sep-16	5	92.9	38	0	0.41	0.00
	UCR-LKMT10	01-Sep-16	5	93.3	20	0	0.21	0.00
		Total	**10**	**186.3**	**58**	**0**	**0.31**	**0.00**
		Average	**5**	**93.1**	**29**	**0**	**0.31**	**0.00**
		SD	**n/a**	**0.3**	**12.7**	**0.0**	**0.14**	**0.00**
John Hart Reservoir	JHT-LKMT01	21-Aug-16	5	99.0	23	8	0.23	0.08
	JHT-LKMT02	21-Aug-16	5	90.4	47	30	0.52	0.33
		Total	**10**	**189**	**70**	**38**	**0.37**	**0.20**
		Average	**5**	**95**	**35**	**19**	**0.38**	**0.21**
		SD	**n/a**	**6.1**	**17.0**	**15.6**	**0.20**	**0.18**

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

Figure 22. Catch-per-unit-effort (CPUE) of Sculpin spp. during minnow trapping from all lakes sampled in Year 1, 2, and 3 of the JHTMON-5 Program.

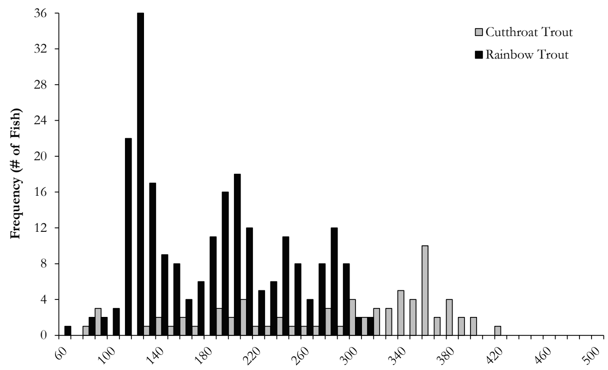


3.2.4. Individual Fish Analysis

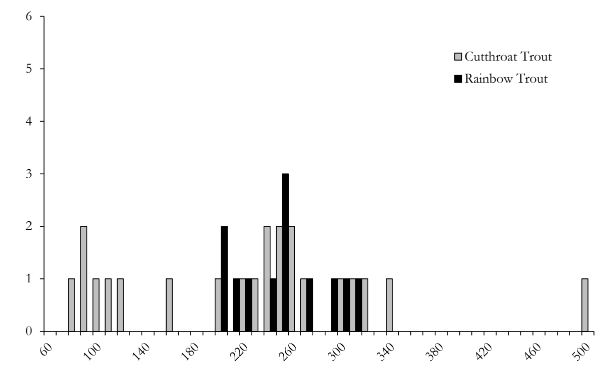
Cutthroat Trout and Rainbow Trout captured in gill nets varied substantially in size. Length frequency histograms of Cutthroat Trout and Rainbow Trout captured in 2016 are presented in Figure 23. Cutthroat Trout tend to be larger than Rainbow Trout in all lakes and attain a larger maximum body size. Note that these length-frequency histograms do not adequately represent relative abundance across, in particular, the smallest size classes.

Figure 23. Length-frequency histogram of Cutthroat Trout and Rainbow Trout captured in a) Upper Campbell Reservoir, b) John Hart Reservoir, c) Brewster Lake, d) Gray Lake and e) Whymper Lake.

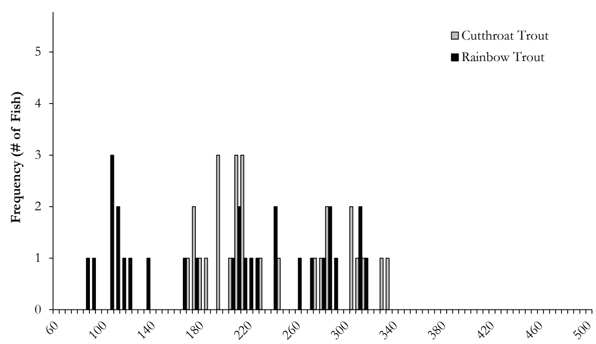
a) Upper Campbell Reservoir



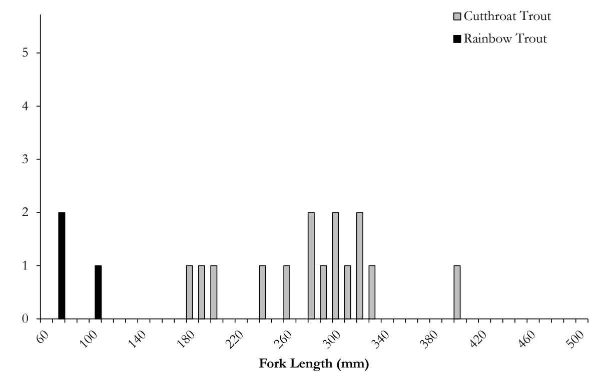
b) John Hart Reservoir



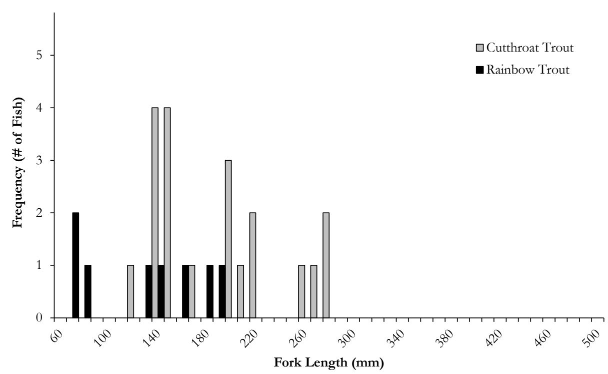
c) Brewster Lake



e) Gray Lake



d) Whymper Lake



3.2.5. Stomach Contents

Stomach content data for Cutthroat Trout and Rainbow Trout across all lakes sampled in Years 1 to 3 are shown in Table 30. Cutthroat Trout diet varied by lake but overall was dominated by littoral/terrestrial invertebrates (average = 46.5%) and by fish (average = 45.6%). The contribution of zooplankton to Cutthroat Trout diet was small (average = 7.9%). In contrast, Rainbow Trout diets were composed primarily of littoral/terrestrial invertebrates (average = 65.7%) and zooplankton (average = 34.3%), while no fish were observed in Rainbow Trout diets in any system.

Table 30. Stomach content results by volume for Cutthroat Trout (CT) and Rainbow Trout (RB).

Year	Waterbody	Fish Species	Littoral/Terrestrial Invertebrates	Zooplankton	Fish
2014	Upper Campbell Reservoir	CT	1.3%	0.0%	98.8%
		RB	41.7%	58.3%	0.0%
	Gooseneck Lake	CT	35.6%	53.3%	11.1%
	Middle Quinsam Lake	CT	86.4%	4.5%	9.1%
		RB	100.0%	0.0%	0.0%
2015	Upper Campbell Reservoir	CT	22.2%	0.0%	77.8%
		RB	75.0%	25.0%	0.0%
	Lower Campbell Reservoir	CT	42.9%	0.0%	57.1%
		RB	41.2%	58.8%	0.0%
	Snakehead Lake	CT	90.9%	4.5%	4.5%
	Beavertail Lake	CT	53.5%	14.1%	32.4%
		RB	77.5%	22.5%	0.0%
	Gooseneck Lake	CT	-	-	-
	Middle Quinsam Lake	CT	-	-	-
	Upper Quinsam Lake	CT	68.5%	7.4%	24.1%
2016	Upper Campbell Reservoir	CT	50.0%	0.0%	50.0%
		RB	-	-	-
	John Hart Reservoir	CT	46.4%	0.0%	53.6%
		RB	22.2%	77.8%	0.0%
	Brewster Lake	CT	52.6%	0.0%	47.4%
		RB	68.2%	31.8%	0.0%
	Gray Lake	CT	4.2%	0.0%	95.8%
		RB	-	-	-
	Whymper Lake	CT	50.0%	18.2%	31.8%
		RB	100.0%	0.0%	0.0%
	Average	*CT*	46.5%	7.9%	45.6%
		RB	65.7%	34.3%	0.0%

"-" denotes where no fish were sampled

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 across all lakes and reservoirs sampled from 2014 to 2016 (Figure 24, Figure 25). Cutthroat Trout had high $\delta^{15}N$ levels consistent with their top position within lake food webs. Rainbow Trout had lower $\delta^{15}N$ and $\delta^{13}C$ values than Cutthroat Trout, indicating an increased pelagic contribution to their diet. Dolly Varden had high $\delta^{15}N$ levels consistent with a piscivorous diet, but had similar $\delta^{13}C$ to Rainbow Trout. Adult Kokanee had similar $\delta^{15}N$ and $\delta^{13}C$ values to Rainbow Trout, indicative of their largely pelagic diet. Smaller prey fish (juvenile trout, Sculpin spp., and Threespine Stickleback) generally had lower $\delta^{15}N$ and $\delta^{13}C$ than large trout and Dolly Varden consistent with their intermediate trophic level positions. Littoral invertebrates and zooplankton had the lowest $\delta^{15}N$ 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, benthic, stream, and terrestrial invertebrates had higher $\delta^{13}C$ isotopic signatures, consistent with the largely allochthonous sources of carbon in their diets. For comparison, the isotope signatures of attached algae (ALG), small particulate organic matter (SPO), leaf litter (LL) and littoral periphyton (PER) are also shown in Figure 25, which were sampled as a part of JHTMON-4 in Upper Campbell and Lower Campbell reservoirs.

Figure 24. Carbon – nitrogen stable isotope bi-plots (mean ± SD) of fish and invertebrates from the three reservoirs (Upper Campbell, Lower Campbell, and John Hart), and eight diversion lakes (Gooseneck, Middle and Upper Quinsam, Snakehead, Beavertail, Brewster, Gray and Whymper) in 2014, 2015, and 2016.

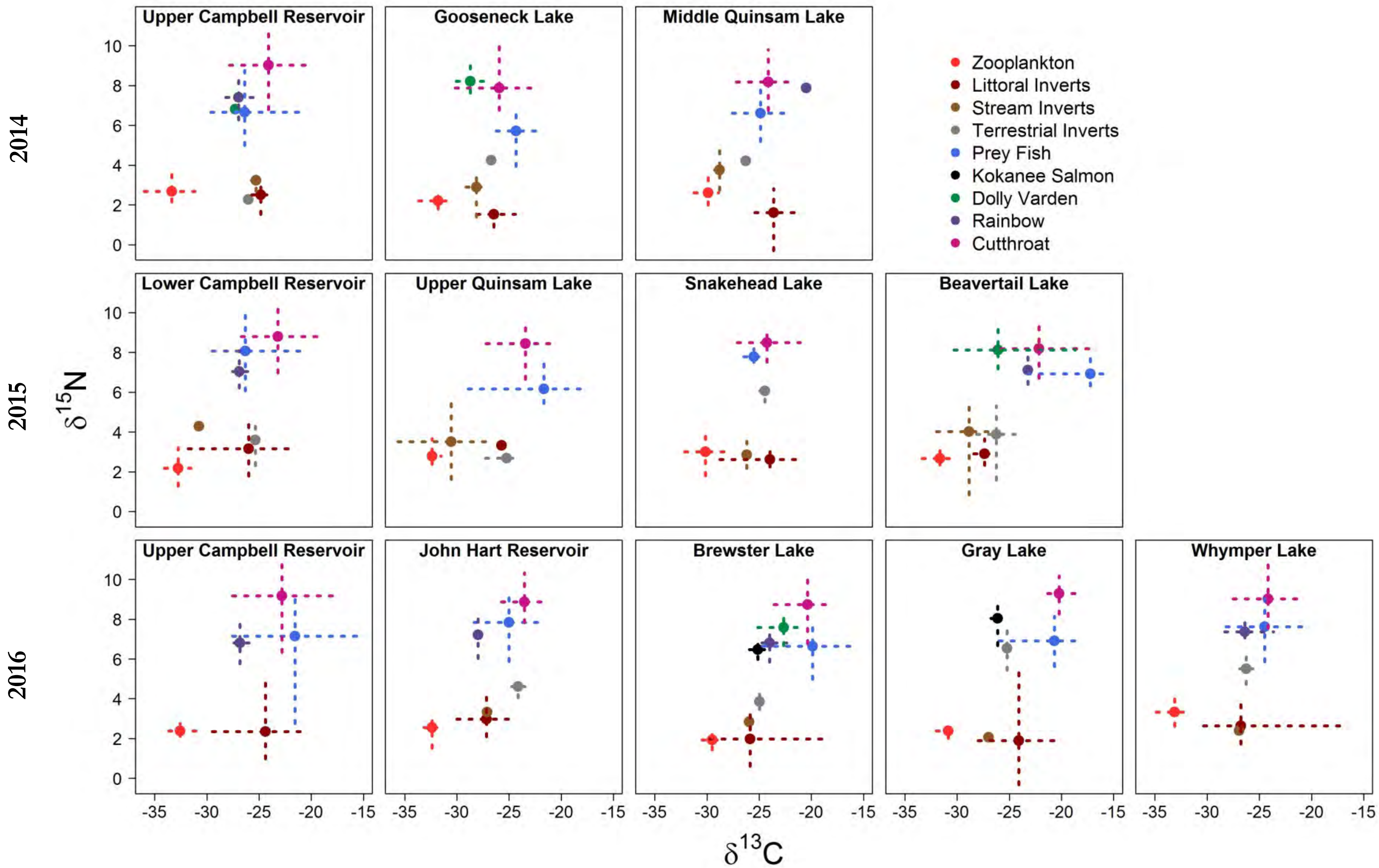
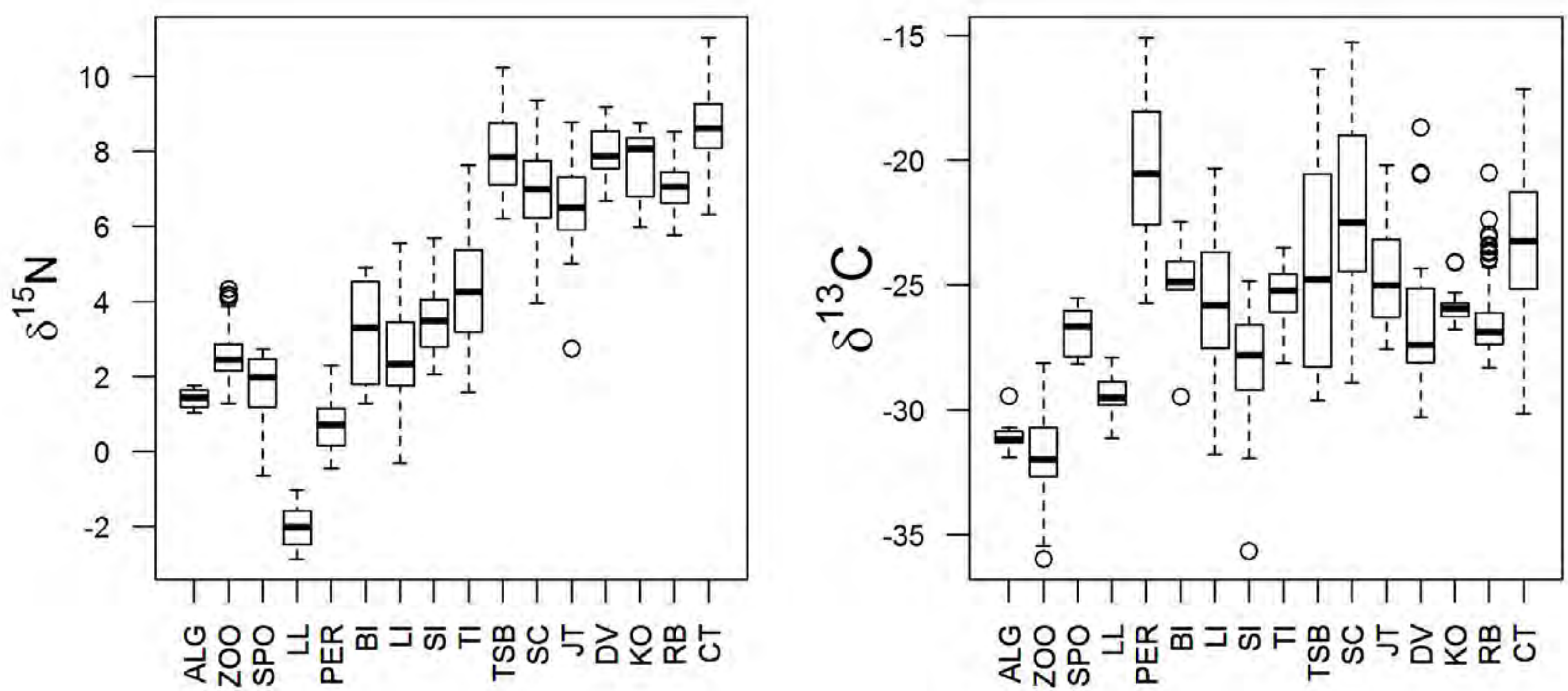


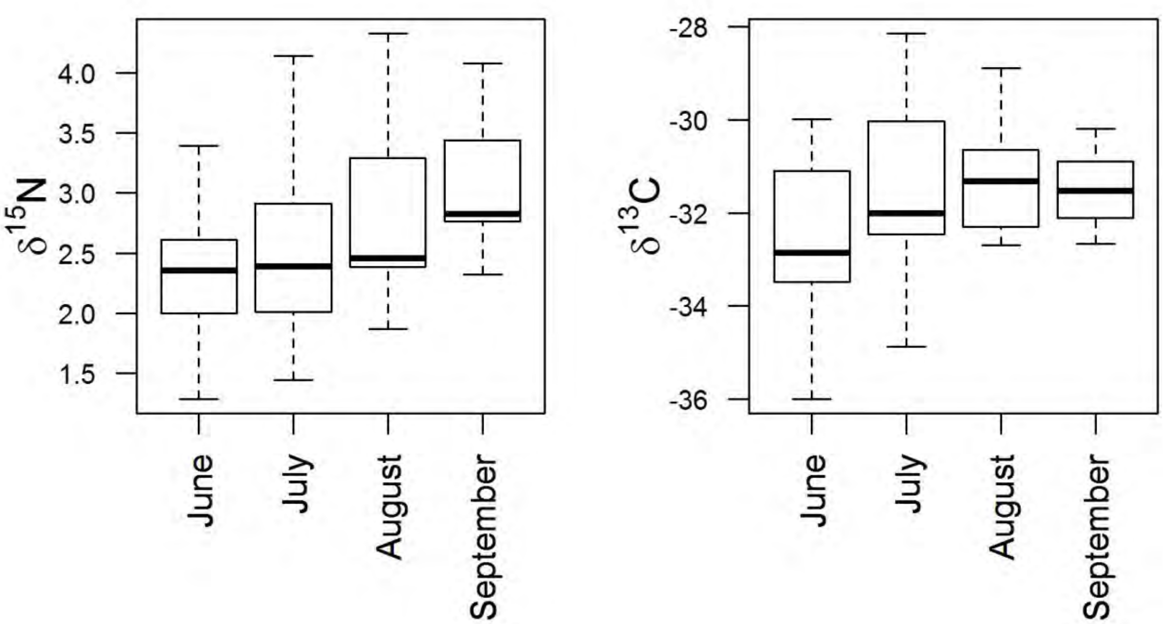
Figure 25. Average $\delta^{15}N$ and $\delta^{13}C$ stable isotope signatures in all taxa sampled across all 11 lakes between 2014 and 2016. ZOO = Zooplankton, LI = Littoral Invertebrates, SI = Stream Invertebrates, TI = Terrestrial Invertebrates, TSB = Threespine Stickleback, SC = Sculpin spp., JT = Juvenile Trout, KO=Kokanee Salmon, DV = Dolly Varden, RB = Rainbow Trout > 150 mm, CT = Cutthroat Trout > 150 mm. Additional taxa sampled within JHTMON-4 are also shown for comparison: BI = Benthic Invertebrates, ALG = Attached Algae, SPO = Small Particulate Organic Matter, LL = Leaf Litter, PER = Littoral Periphyton.



3.3.2. Isotopic Variation in Zooplankton

Nitrogen and carbon stable isotope signatures in bulk zooplankton varied by month of collection. Across all lakes and years, $\delta^{15}N$ signatures in zooplankton were significantly higher in September compared to June or July, and intermediate in August (Figure 26, ANOVA: $F_{3,81} = 8.3$, $p < 0.0001$). $\delta^{13}C$ signatures in zooplankton were significantly higher in both August and September compared to June and July (Figure 26, ANOVA: $F_{3,81} = 10.0$, $p < 0.0001$). $\delta^{15}N$ and $\delta^{13}C$ signatures in zooplankton were also significantly different by lake (ANOVA: $F_{3,30} > 6.5$, $p < 0.0001$). The most enriched $\delta^{15}N$ signatures were observed in Whymper Lake and Snakehead Lake, and the most depleted $\delta^{15}N$ signatures were observed in Brewster Lake and Lower Campbell Reservoir. For $\delta^{13}C$, the most enriched signatures were observed in Brewster, Middle Quinsam, and Snakehead lakes, while the most depleted signatures were observed in Upper Campbell Reservoir, Lower Campbell Reservoir, and Whymper Lake.

Figure 26. Monthly variation in the $\delta^{15}N$ and $\delta^{13}C$ stable isotope signatures in zooplankton across all 11 lakes sampled from 2014, 2015 and 2016.



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 11 lakes and are discussed for each lake in the sections below.

3.3.3.1. Upper Campbell, Lower Campbell and John Hart Reservoirs

Cutthroat Trout (age >2+) diets were dominated by prey fish in Upper Campbell, Lower Campbell and John Hart reservoirs (62-71%, 50%, and 53% respectively), followed by terrestrial invertebrates

(19%, 30%, and 28%, respectively) (Figure 27, Table 31). This is consistent with the trophic position of Cutthroat Trout as top piscivorous predators, but also shows a surprisingly high contribution of terrestrial invertebrates to their diets. Pelagic contributions to Cutthroat Trout diets were low in all three reservoirs, with zooplankton diet contributions ranging from only 3% to 11% on average. Littoral invertebrate contributions to the diets of these fish were also low ranging from 7% to 18%.

In contrast, Rainbow Trout diets had a high contribution from zooplankton at 36-40%, 39%, and 55% in Upper Campbell, Lower Campbell and John Hart reservoirs, respectively. Similar to Cutthroat Trout, Rainbow Trout had a relatively high contribution of terrestrial invertebrates in their diets (37% to 54% across all three reservoirs).

These patterns in Cutthroat Trout and Rainbow Trout diet are similar to those observed in their stomach contents. Prey fish made up a high percentage of Cutthroat Trout stomach contents (54-78% in the three reservoirs), while Rainbow Trout had a high percentage of zooplankton (42%-78%) in the reservoirs. One difference between stable isotope estimates of diet and stomach contents was that no zooplankton were observed in Cutthroat Trout stomachs.

Invertebrate contributions to the diets of Sculpin spp., juvenile trout, and Threespine Stickleback were relatively similar, which justified combining these three prey fish groups into a single prey fish group in models (Figure 28, Table 31). Prey fish diets were dominated by terrestrial invertebrates (48-77%). The contribution of zooplankton and littoral invertebrates varied by reservoir and year, with zooplankton making up 30% of prey fish diets in Upper Campbell Reservoir in 2014, but only 4% in 2016, and littoral invertebrates making up 22% of their diets in this reservoir in both years. Similarly, zooplankton made up 22% and 13% of prey fish diets in Lower Campbell Reservoir and John Hart Reservoir, respectively. These results are influenced by the relative proportion of prey fish made up of Threespine Stickleback, which fed predominantly on zooplankton in the Campbell reservoirs in 2014 and 2015. Based on these results for prey fish, only 4% to 30% of prey fish diets are pelagic, with the remainder of their diets made up of littoral or terrestrial invertebrate sources.

Figure 27. Estimated proportions of invertebrate and fish diet sources to Cutthroat Trout (top row) and Rainbow Trout (bottom row) in Upper Campbell, Lower Campbell, and John Hart 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.

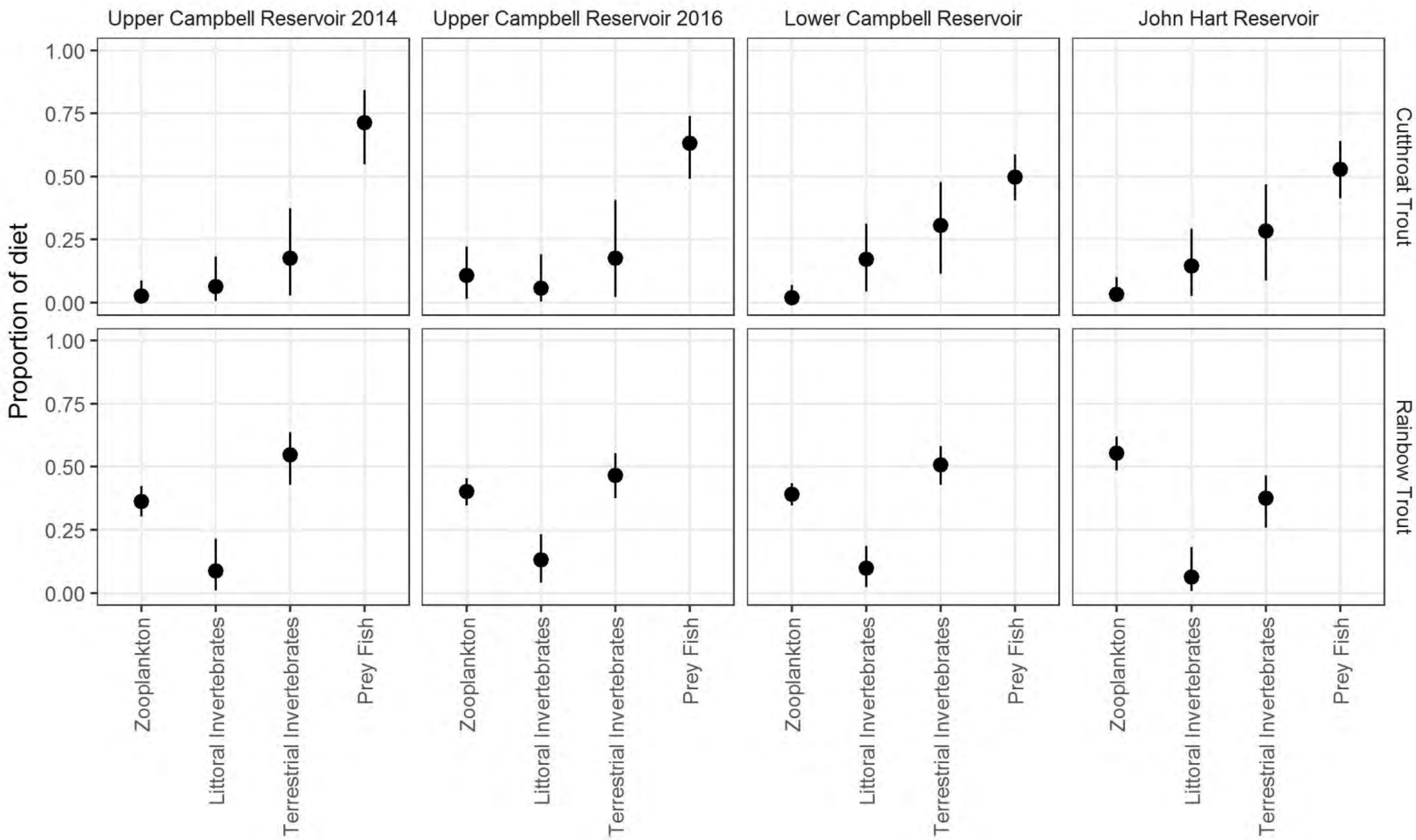


Figure 28. Estimated proportions of invertebrate diet sources to prey fish (juvenile trout, Sculpin spp., and Threespine Stickleback) in Upper Campbell, Lower Campbell and John Hart reservoirs. Estimates are calculated as means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from prey fish and their potential diet sources.

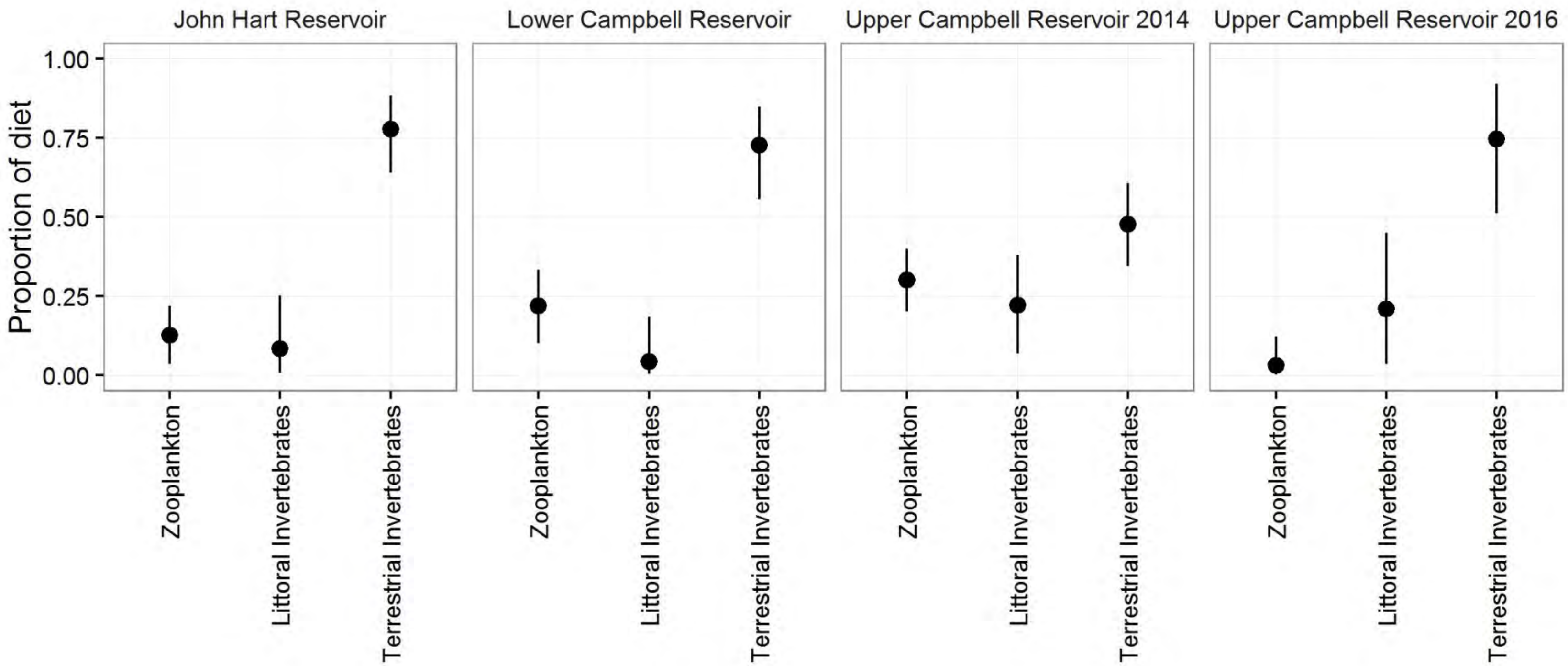


Table 31. SIA-based estimates of diet contribution from A) zooplankton and littoral and terrestrial invertebrates and B) prey fish to large trout and Dolly Varden in the three reservoirs and eight diversion lakes across the three years of sampling (2014 to 2016). Diet estimates are calculated as means 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.

Year	Waterbody	Consumer[1]	Estimated Invertebrate Contributions to Diet								
			Pelagic			Littoral					
			Zooplankton			Littoral Invertebrates[2]			Terrestrial Invertebrates		
			Mean	Q 5%	Q 95%	Mean	Q 5%	Q 95%	Mean	Q 5%	Q 95%
2014	Upper Campbell Reservoir	CT	0.03	0.00	0.09	0.07	0.01	0.18	0.19	0.03	0.38
		RB	0.36	0.30	0.42	0.10	0.01	0.21	0.54	0.43	0.64
		PF	0.30	0.20	0.40	0.22	0.07	0.38	0.48	0.35	0.61
	Gooseneck Lake	CT	0.30	0.19	0.39	0.05	0.00	0.14	0.16	0.01	0.39
		DV	0.44	0.08	0.73	0.12	0.01	0.33	0.19	0.02	0.44
		PF	0.13	0.03	0.24	0.43	0.24	0.60	0.44	0.26	0.64
	Middle Quinsam Lake	CT	0.10	0.02	0.20	0.13	0.02	0.27	0.24	0.06	0.44
		PF	0.23	0.07	0.40	0.32	0.09	0.52	0.45	0.26	0.66
2015	Lower Campbell Reservoir	CT	0.03	0.00	0.07	0.18	0.04	0.31	0.30	0.11	0.48
		RB	0.39	0.35	0.44	0.10	0.02	0.19	0.51	0.43	0.58
		PF	0.22	0.10	0.33	0.06	0.00	0.18	0.72	0.56	0.85
	Beavertail Lake	CT	0.22	0.08	0.36	0.09	0.01	0.22	0.22	0.04	0.41
		RB	0.11	0.01	0.33	0.36	0.08	0.61	0.53	0.26	0.82
		DV	0.25	0.04	0.45	0.16	0.01	0.37	0.26	0.04	0.49
		PF	0.23	0.02	0.48	0.32	0.06	0.58	0.45	0.21	0.71
	Snakehead Lake	CT	0.12	0.02	0.23	0.17	0.04	0.31	0.21	0.06	0.38
		PF	0.31	0.11	0.49	0.22	0.03	0.46	0.46	0.23	0.69
	Upper Quinsam Lake	CT	0.13	0.03	0.24	0.07	0.01	0.17	0.31	0.11	0.50
		PF	0.26	0.03	0.52	0.37	0.10	0.64	0.37	0.12	0.63
2016	Upper Campbell Reservoir	CT	0.11	0.02	0.22	0.07	0.01	0.19	0.19	0.02	0.41
		RB	0.40	0.35	0.45	0.13	0.04	0.23	0.47	0.37	0.55
		PF	0.04	0.00	0.12	0.22	0.04	0.45	0.74	0.51	0.92
	John Hart Reservoir	CT	0.04	0.00	0.10	0.15	0.03	0.29	0.28	0.09	0.47
		RB	0.55	0.49	0.62	0.07	0.01	0.18	0.37	0.26	0.47
		PF	0.13	0.04	0.22	0.10	0.01	0.25	0.77	0.64	0.88
	Brewster Lake	CT	0.06	0.00	0.14	0.08	0.01	0.19	0.22	0.06	0.37
		RB	0.13	0.02	0.24	0.30	0.11	0.47	0.58	0.41	0.75
		DV	0.19	0.02	0.37	0.21	0.03	0.43	0.25	0.04	0.47
		KO	0.31	0.13	0.46	0.27	0.06	0.47	0.43	0.25	0.61
		PF	0.09	0.01	0.22	0.38	0.20	0.54	0.53	0.39	0.68
	Gray Lake	CT	0.04	0.00	0.11	0.07	0.01	0.17	0.16	0.03	0.31
		KO	0.38	0.30	0.46	0.10	0.01	0.23	0.52	0.38	0.64
		PF	0.05	0.00	0.15	0.36	0.19	0.52	0.59	0.44	0.74
	Whymper Lake	CT	0.10	0.01	0.21	0.12	0.01	0.29	0.22	0.02	0.45
		RB	0.32	0.09	0.54	0.30	0.05	0.57	0.38	0.10	0.64
		PF	0.09	0.02	0.17	0.16	0.03	0.30	0.75	0.63	0.88

[1] CT = Cutthroat Trout, RB = Rainbow Trout, DV = Dolly Varden, KO = Kokanee Salmon, PF = Prey Fish (juvenile trout, sculpin, and threespine Stickleback.

[2] Includes benthic and littoral invertebrates.

Table 31. Continued.

B) Year	Waterbody	Consumer[1]	Estimated Vertabrate Contributions to Diet		
			Mean	Q 5%	Q 95%
2014	Upper Campbell Reservoir	CT	0.71	0.55	0.84
	Gooseneck Lake	CT	0.50	0.30	0.64
		DV	0.24	0.02	0.54
	Middle Quinsam Lake	CT	0.52	0.39	0.66
2015	Lower Campbell Reservoir	CT	0.50	0.40	0.59
	Beavertail Lake	CT	0.47	0.36	0.56
		DV	0.33	0.12	0.51
	Snakehead Lake	CT	0.50	0.40	0.59
	Upper Quinsam Lake	CT	0.49	0.36	0.62
2016	Upper Campbell Reservoir	CT	0.63	0.49	0.74
	John Hart Reservoir	CT	0.53	0.41	0.64
	Brewster Lake	CT	0.64	0.55	0.73
		DV	0.35	0.18	0.52
	Gray Lake	CT	0.72	0.62	0.82
	Whymper Lake	CT	0.57	0.39	0.73

[1] CT = Cutthroat Trout, DV = Dolly Varden.

3.3.3.2. Diversion Lakes

Prey fish and terrestrial invertebrates made up the majority of Cutthroat Trout (age >2+) diets across the eight diversion lakes based on stable isotope analyses (Figure 29, Table 31). Prey fish contributed an estimated 47% to 72% to Cutthroat Trout diet in the diversion lakes, while terrestrial invertebrates contributed an estimated 16% to 31%. Similar to the reservoirs, pelagic contributions to Cutthroat Trout diets were low, with zooplankton contributions ranging from only 4% to 22%. The exception to this was in Gooseneck Lake, where zooplankton made up an estimated 30% of Cutthroat diets. Similar to the reservoirs, the contribution of littoral invertebrates to the Cutthroat diets was low, ranging from 5% to 17%.

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 diets. Stomach content analyses generally showed low contributions of zooplankton to Cutthroat Trout diets, which matches results from the stable isotope analysis. One exception is Gooseneck Lake, which had an unusually high estimate for zooplankton contributions to diet (53%) based on stomach content analysis of 10 individual Cutthroat Trout. The contribution of zooplankton to Cutthroat Trout diets in Gooseneck Lake was also higher than in other lakes based on the stable isotope analysis, although it was estimated at 30%. The proportion of Cutthroat Trout stomach contents made up of combined terrestrial and littoral invertebrates was highly variable, ranging from 4% to 91% versus contributions of 21% to 39% estimated using stable isotope methods.

The diets of Rainbow Trout differed among Beavertail Lake, Brewster Lake, and Whymper Lake based on stable isotope analyses (Figure 30, Table 31). The contribution of the three prey sources to their diets in Whymper Lake were similar (ranging from 30% to 38% for littoral and terrestrial invertebrates, respectively). In contrast, Zooplankton contributed the least to Rainbow Trout diets in Beavertail Lake and Brewster Lake (11% and 13%, respectively), and terrestrial invertebrates contributed the most (53% and 58%, respectively).

The diet estimates for Rainbow Trout based on stable isotope analysis were similar to those derived from the analysis of stomach contents in the diversion lakes. Stomach content analyses estimated zooplankton contributions ranging from 0% to 32%, and combined littoral and terrestrial invertebrate contributions ranging from 68% to 100%. No prey fish were observed within Rainbow Trout stomach contents samples, validating the exclusion of prey fish from potential diet items included in isotope models for this species.

The diets of Dolly Varden were more variable than those of Cutthroat and Rainbow Trout based on stable isotope analyses (Figure 31, Table 31). Prey fish and terrestrial invertebrates contributed the most to the diets of Dolly Varden, with estimates ranging from 24-35% and 19-26%, respectively. Littoral invertebrate contributions to Dolly Varden diets were lower, ranging from an estimated 12% to 21%, while zooplankton contributions were the most variable, ranging from only 19% in Brewster Lake, to 45% in Gooseneck Lake.

Based on stable isotope analyses, terrestrial invertebrates contributed an estimated 43% to 52% to the diets of Kokanee Salmon, in Brewster Lake and Gray Lake, respectively, whereas zooplankton contributed an estimated 31% and 38%, respectively (Figure 31, Table 31). Littoral invertebrates contributed the least to Kokanee diets with mean estimated diet contributions ranging from only 10% to 27%.

As in the three reservoirs, the diets of prey fish sampled in the eight diversion lakes were dominated by littoral and terrestrial invertebrates, which made up, on average, an estimated 16-43% and 37-75% of their diets, respectively based on stable isotope analyses. Again, zooplankton contributed significantly less to prey fish diets (5% to 31%). The highest contribution of zooplankton to prey fish diets was observed in Snakehead Lake, at an estimated 31%.

Figure 29. Estimated proportions of invertebrate and vertebrate diet sources to Cutthroat Trout in Gooseneck and Middle Quinsam lakes in 2014 (top row), Beavertail, Snakehead, and Upper Quinsam lakes in 2015 (middle row) and Brewster, Gray, and Whymper lakes in 2016 (bottom row). Diet 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.

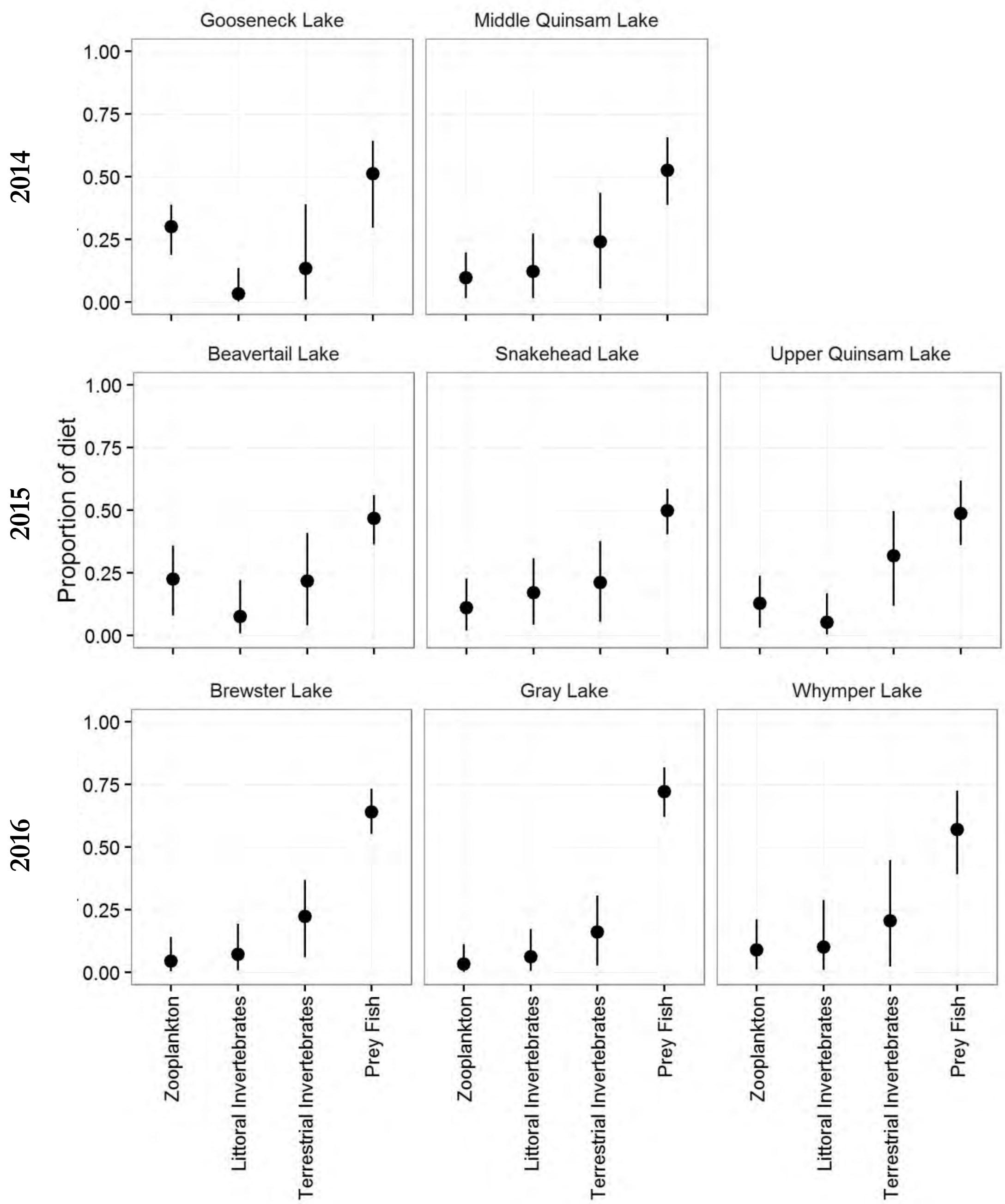


Figure 30. **Estimated proportions of invertebrate diet sources to Rainbow Trout in Beavertail, Brewster, and Whymper lakes. Diet estimates are means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources.**

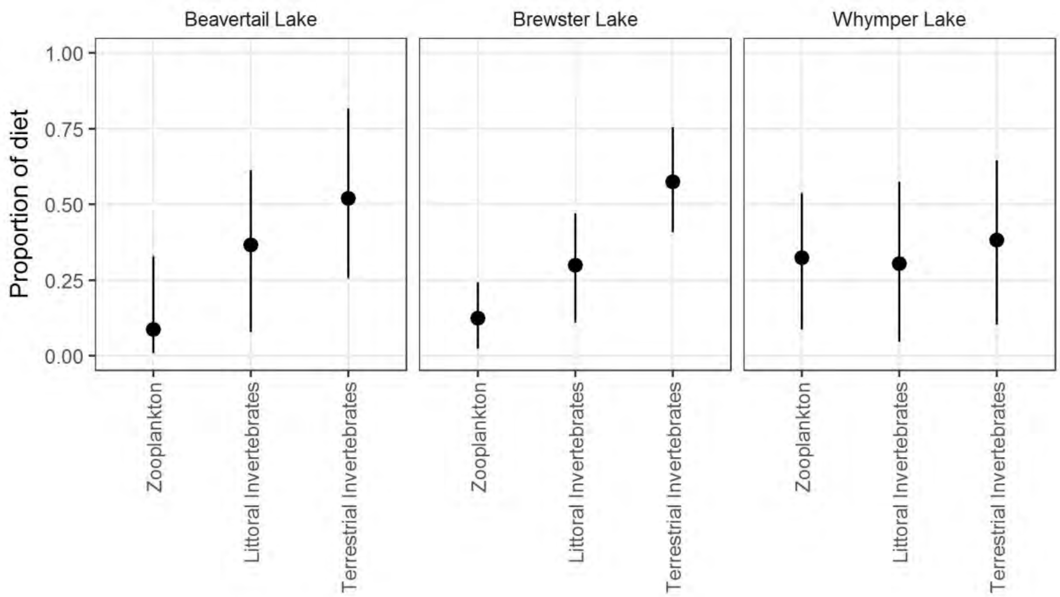


Figure 31. **Estimated proportions of invertebrate and vertebrate diet sources to Dolly Varden in Beavertail, Brewster, and Gooseneck lakes. Diet estimates are means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources.**

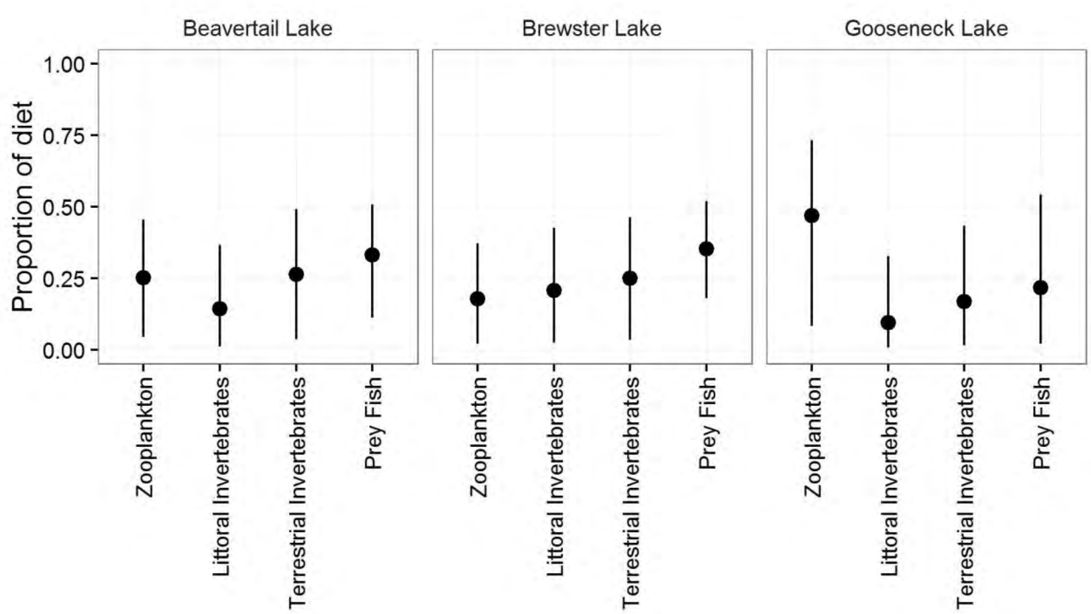


Figure 32. Estimated proportions of invertebrate diet sources to prey fish (Sculpin spp., juvenile trout, and Threespine Stickleback) in Gooseneck and Middle Quinsam lakes in 2014 (top row), Beavertail, Snakehead, and Upper Quinsam lakes in 2015 (middle row), and Brewster, Gray, and Whymper lakes in 2016 (bottom row). Diet 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.

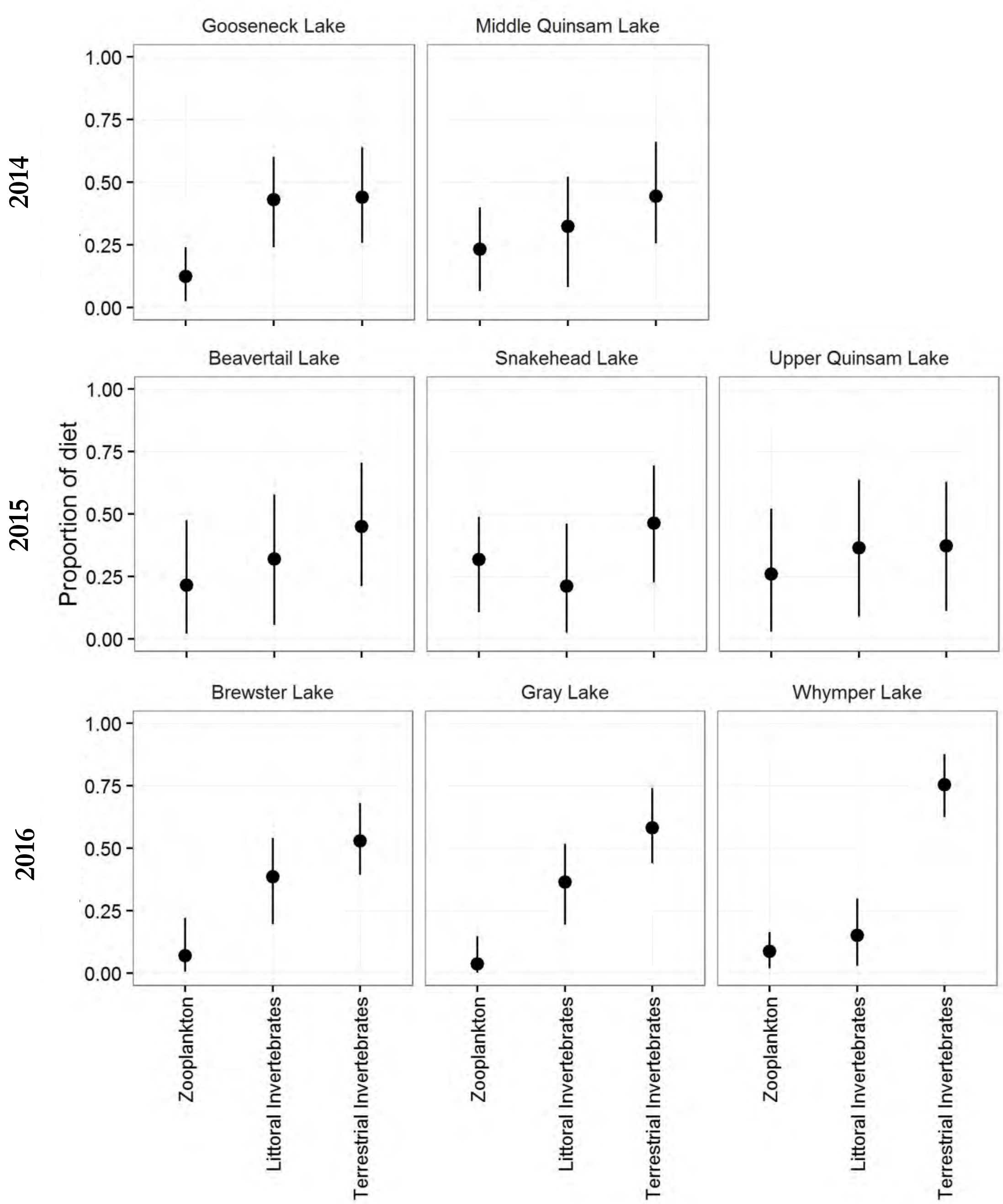
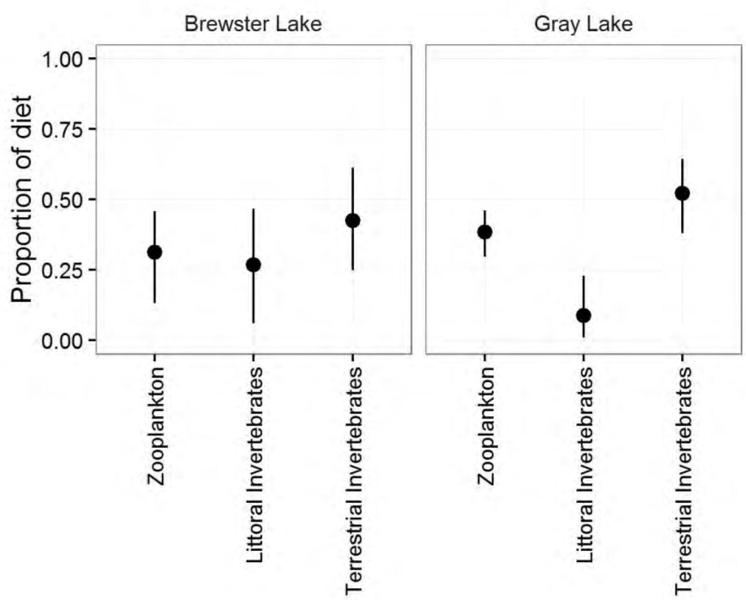


Figure 33. Estimated proportions of invertebrate diet sources to adult Kokanee Salmon in Brewster and Gray lakes in 2016. Diet estimates are means with 5% and 95% quartile ranges of posterior probability distributions from carbon – nitrogen Bayesian mixing models based on isotopic signatures from these consumers and their potential diet sources.

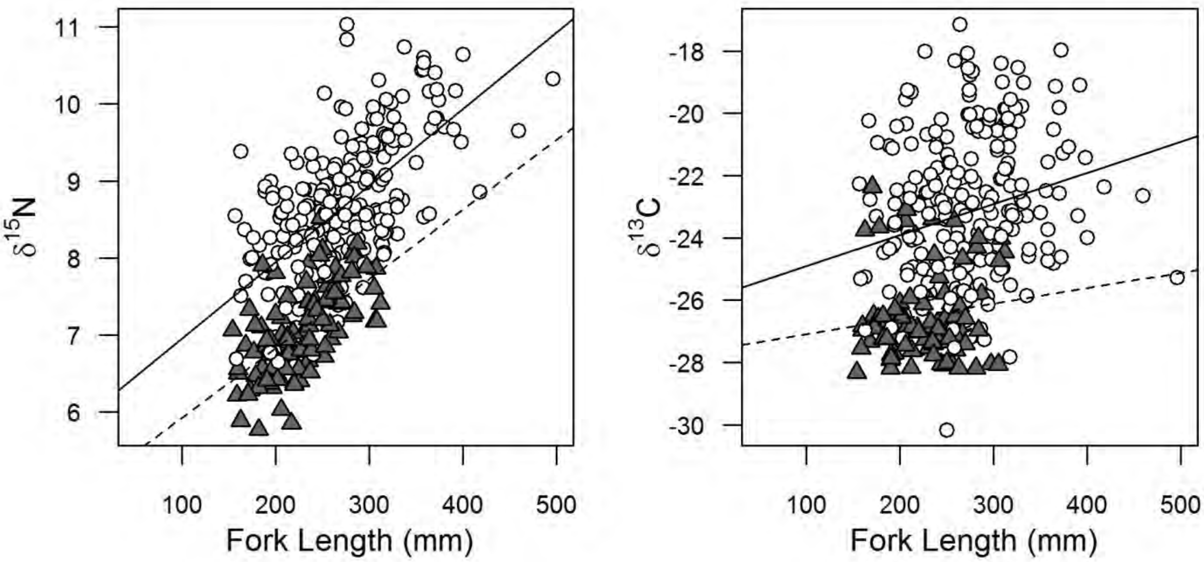


3.3.4. Diet Variation with Fish Size

Cutthroat Trout and Rainbow Trout $\delta^{15}N$ signatures and fork length were positively correlated for fish sampled from all 11 lakes (Figure 34, Cutthroat $F_{1,232} = 140$, Rainbow $F_{1,94} = 58$, both $p < 0.001$). This indicates that larger trout are more piscivorous and feed at higher trophic positions in the food web than smaller trout.

A positive relationship between Cutthroat Trout $\delta^{13}C$ signatures and fork length ($F_{1,232} = 13$, $R^2 = 0.05$, $p < 0.001$) was also observed, indicating that the relative contribution of littoral and terrestrial sources to their diet may increase slightly with size. In contrast, the relationship between Rainbow Trout $\delta^{13}C$ signatures and fork length was not significant ($F_{1,94} = 1.6$, $p > 0.2$), indicating that the dominant sources of carbon (e.g., pelagic and terrestrial) do not vary by fish age and body size for this species.

Figure 34. $\delta^{15}N$ and $\delta^{13}C$ stable isotope signatures by fork length (mm) in Cutthroat Trout (open circles; solid regression line) and Rainbow Trout (closed triangles; dotted regression line) from all 11 lakes.



3.4. Water Residence Time

3.4.1. Water Residence Time Estimates

Estimates of WRT were derived for the whole hydrological year (Table 32) and for the stratified period (Table 33), which was assumed to span May 15th to September 30th (also referred to as seasonal WRT). Annual WRT was, on average, longer than the WRT computed for the stratified period for all waterbodies except Middle Quinsam and Whymper lakes and John Hart Reservoir (Figure 35). Unlike the other waterbodies, Whymper Lake and John Hart Reservoir were only weakly stratified; therefore, the full lake volume was used in the seasonal WRT calculations. This, combined with lower seasonal water inputs, resulted in longer seasonal water residence times. Similarly, Middle Quinsam Lake had the smallest reduction in volume (with the exception of Whymper Lake and John Hart Reservoir) and largest decrease in outflow during the stratified season compared to all of the lakes (Figure 35). The difference between annual and seasonal water residence times across all the waterbodies varied, on average, between <1 to 1,053 days, with Upper Campbell Reservoir and Beavertail Lake showing the greatest reduction in WRT during the stratified period (Table 33).

Upper Campbell Reservoir, and Upper Quinsam, Gooseneck, Beavertail, and Brewster lakes had annual average water residence times exceeding 100 days; annual WRT at Upper Campbell Reservoir and Beavertail and Brewster lakes exceeded 365 days. These long annual water residence times are largely due to the large volume of these waterbodies. Although Upper Quinsam Lake has a lower volume than Lower Campbell Reservoir, it has substantially lower outflow, which increases its water residence time.

Beavertail Lake had the longest average annual WRT (3.8 years), followed by Brewster Lake (2.8 years). Brewster Lake has the longest seasonal WRT (875 days), followed by Beavertail Lake (321 days). The water residence times for Beavertail Lake were relatively long considering the surface area, volume, and watershed area. The long residence times at Beavertail Lake reflects low outflow, which is due to its landscape position. Beavertail Lake is a control lake in the upper reaches of a relatively isolated watershed, disconnected from the larger watersheds of the other diversion lakes (Map 6). Water residence time in Beavertail Lake is therefore long because it has a large volume relative to water inputs (Table 32 and Table 33). Likewise, the relatively long water residence times for Brewster Lake are mainly due to the large water volume of this lake (Figure 35 and Table 32), as well as its landscape position; although Brewster Lake receives some inputs from the Salmon River Diversion canal, surface inflows are low relative to other diversion lakes (Map 10).

Whymper Lake had the shortest annual and seasonal WRT (<1 day), reflecting its small volume. Gray Lake (directly upstream) had similar seasonal WRT (<1 day). Of the three reservoirs, John Hart Reservoir had the shortest annual WRT (5.3 days) owing to a smaller volume and larger outflows relative to the other reservoirs. The short seasonal residence time for Lower Campbell Reservoir (~ 1 day) is somewhat surprising given its large surface area; however, Lower Campbell Reservoir has relatively large inflows and outflows in the summer relative to its volume, which reduces its seasonal water residence time.

Figure 35. Average Annual and Seasonal WRT of the JHTMON-5 waterbodies from October 2011 to October 2016, calculated using the water balance method.

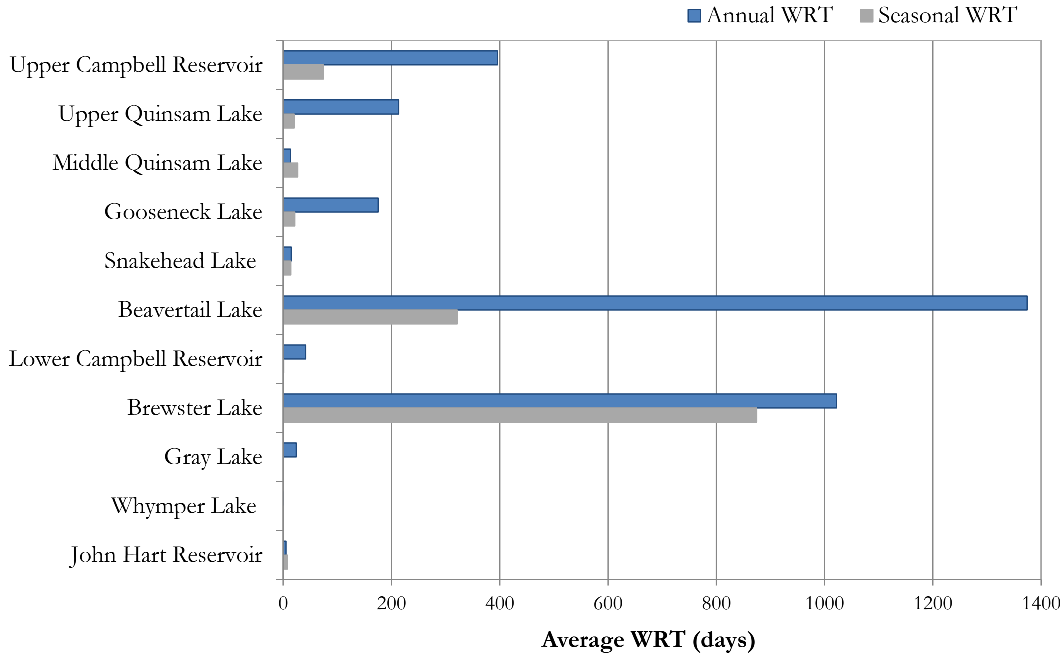


Table 32. Lake volume, annual lake outflow (Q_o), and annual water residence time (WRT) computed for 2012-2016 from the water balance method.

Waterbody	Volume (m^3)	2015-2016		2014-2015		2013-2014		2012-2013		2011-2012		Average
		Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	WRT (days)
Upper Campbell Reservoir	2.46E+09	78.2	363.8	63.8	446.2	69.7	408.3	72.7	391.3	76.8	370.4	396.0
Upper Quinsam Lake	5.28E+07	2.82	216.5	2.69	227.2	2.85	214.7	3.04	200.7	2.95	207.3	213.3
Middle Quinsam Lake	2.82E+06	2.52	12.9	2.69	12.1	2.19	14.9	2.36	13.8	2.60	12.5	13.3
Gooseneck Lake	7.53E+06	0.55	159.2	0.36	240.6	0.61	141.9	0.58	150.5	0.47	184.7	175.4
Snakehead Lake	7.62E+05	0.64	13.8	0.45	19.7	0.70	12.5	0.66	13.3	0.56	15.7	15.0
Beavertail Lake	1.10E+07	0.10	1,276.8	0.10	1,327.2	0.08	1,553.4	0.09	1,346.7	0.09	1,365.2	1,373.9
Lower Campbell Reservoir	3.16E+08	95.2	38.4	77.4	47.2	85.7	42.7	89.1	41.0	94.3	38.8	41.6
Brewster Lake	1.20E+08	3.68	376.8	3.79	365.8	1.85	747.1	0.70	1,992.3	0.85	1,628.8	1,022.1
Gray Lake	3.06E+06	3.79	9.4	3.85	9.2	1.97	18.0	0.79	45.0	0.90	39.3	24.2
Whymper Lake	4.45E+04	3.97	0.1	3.97	0.1	2.02	0.3	0.85	0.6	0.97	0.5	0.33
John Hart Reservoir	4.26E+07	103.2	4.8	86.9	5.7	90.6	5.4	86.8	5.7	96.5	5.1	5.3

Table 33. Estimated epilimnion volume, proportion of epilmnion volume to total volume, seasonal lake outflow (Q_o), and seasonal water residence time (WRT) computed for 2012-2016 from the water balance method (WB) and gauged flow data method (Gauged).

Waterbody	Epilimnion Volume (m^3)	Proportion of Epilimnion Volume to Total Volume (%)	2016		2015		2014		2013		2012		Average
			Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	Q_o (m^3/s)	WRT (days)	Q_{out} (m^3/s)	WRT (days)	WRT (days)
Upper Campbell Reservoir	3.21E+08	13	51.69	72.0	41.26	90.2	54.83	67.8	48.78	76.3	57.36	64.8	74.2
Upper Quinsam Lake	1.84E+06	3	1.24	17.1	1.04	20.5	0.92	23.2	0.96	22.1	1.18	18.1	20.2
Middle Quinsam Lake	1.99E+06	70	1.06	21.6	0.94	24.5	0.71	32.2	0.70	32.7	1.03	22.3	26.7
Gooseneck Lake	3.33E+05	4	0.23	17.1	0.13	30.6	0.24	16.2	0.23	17.0	0.15	26.0	21.4
Snakehead Lake	2.36E+05	31	0.24	11.2	0.16	17.4	0.23	12.1	0.21	13.1	0.17	16.0	14.0
Beavertail Lake	1.89E+06	17	0.08	274.6	0.06	368.8	0.06	373.6	0.07	297.0	0.07	293.1	321.4
Lower Campbell Reservoir	1.05E+07	3	56.32	2.2	45.64	2.7	60.48	0.2	52.69	0.2	63.03	0.2	1.1
Brewster Lake	6.63E+07	55	1.25	612.3	4.76	161.1	1.14	675.3	0.56	1,364.8	0.49	1,559.5	874.6
Gray Lake	5.76E+04	2	1.36	0.5	4.89	0.1	1.28	0.5	0.60	1.1	0.62	1.1	0.67
Whymper Lake	4.45E+04	100	1.45	0.4	4.97	0.1	1.33	0.4	0.66	0.8	0.69	0.7	0.47
John Hart Reservoir	4.26E+07	100	67.50	7.3	55.98	8.8	59.77	8.2	61.04	8.1	71.58	6.9	7.9

3.4.2. Error Analysis and Data Limitations

The accuracy of the estimates for WRT by waterbody is dependent on the data used to derive them. 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.

As a quality assurance check, WRT was computed using gauged inflow data for Gooseneck and Middle Quinsam lakes, and gauged outflow data for Upper Quinsam Lake, and Upper Campbell, Lower Campbell, and John Hart reservoirs, and compared to the WRTs estimated using the water balance method derived from Equation (5). Although there is error associated with the rating curves for each gauge, we assume that this is negligible and therefore differences between the two WRT estimates for these waterbodies quantify the error in the watershed water balance method.

On average, the computed water residence times (derived from Equation (5)) compare well with residence times derived from gauged data (Table 34). The differences in the estimates are considered to be well within the uncertainty of both methods used to compute water residence. The mean difference in computed minus measured annual WRT for all waterbodies was 1.44 days (or 1.04 %), and 1.29 days (or 5.40%) for seasonal WRT. Computed WRT neither consistently overestimated nor underestimated WRT over the 5-year period for any of the waterbodies, when compared to the WRT derived from the gauged data (Figure 36). The generally low error and absence of bias suggests that the water balance method well-approximated the main hydrological inputs and outputs to each waterbody.

The difference in computed minus measured annual WRT ranged from −54.4 days (Upper Quinsam Lake in 2013) to 117.6 days (Upper Campbell Reservoir in 2015), where a negative number of days denotes computed WRT was less than measured WRT and a positive number denotes it was greater. The difference in computed minus measured seasonal WRT ranged from −418.7 days to 456.8 days (Figure 36). The greatest differences in estimates of seasonal WRT were computed for Gooseneck Lake in 2012 (−418.7 days) and 2015 (456.8 days) (see two outlier values of seasonal WRT in Figure 36). These values were associated with estimated discharge (based on the water balance method) that was two orders of magnitude higher than the discharge gauged upstream at the Quinsam River Diversion. The gauged flow in 2012 was less than estimated evaporation, hence the negative WRT. The difference between computed and measured WRT for waterbodies was greatest in 2015, and the least in 2016.

Differences could be due to a number of factors. These are described further below and include errors in: precipitation associated with spatial variability that is not captured at climate stations, missing or incorrect data related to land surface type and land use that would result in an inaccurate runoff curve numbers, the initial abstraction ratio, watershed delineation, and waterbody volume estimates. 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. Note, that there were few data available to verify the water residence times of Brewster, Gray, and Whymper lakes, and therefore the relative accuracy of these estimates is unknown, although we expect it to be consistent with the error quantified for the other waterbodies (Figure 36).

There was a relatively high degree of spatial variability in precipitation recorded at the three climate stations, and it is likely that the precipitation was underestimated for watersheds in higher elevations. There is also some uncertainty around our estimates for evaporation at each lake. 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 19 and Table 20).

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, e.g. based on the watershed geology. In addition, the change in storage in the reservoirs was not taken into account in the water balance computations.

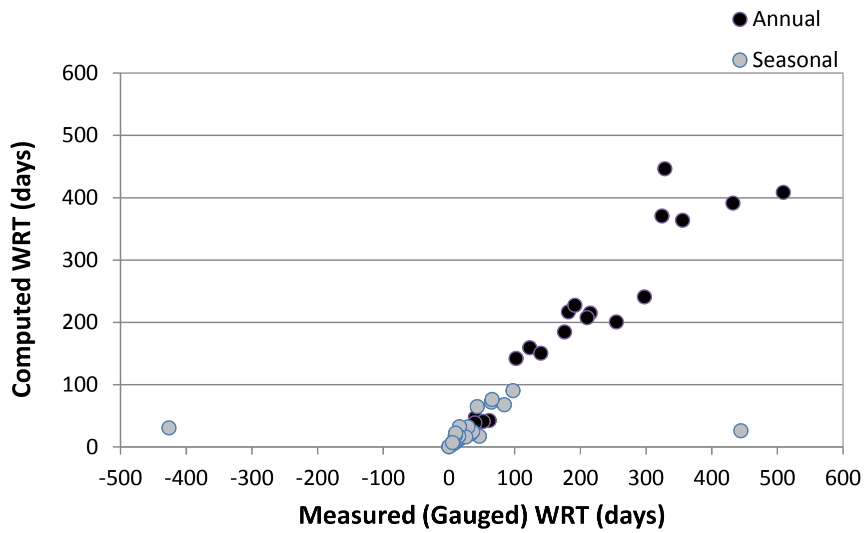
The annual water residence time calculations were particularly dependent on lake volume and watershed area. Stage-volume data from 2001 were used to determine lake volume (Bruce 2001). However, it is unclear when the bathymetric surveys were conducted in Bruce (2001). If the data were collected under 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, some important assumptions were associated with the lake volumes used to determine seasonal water residence time. 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 15 to September 30), which was assumed to be the same for all lakes, in all years. Although mean annual 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 the thermocline depth remained constant in each lake.

Table 34. Annual and seasonal water residence time computed from derived flow data (from WB method) and gauged flow data for Upper Quinsam, Middle Quinsam and Gooseneck lakes, and Upper Campbell, Lower Campbell, and John Hart reservoirs.

Waterbody		2016		2015		2014		2013		2012		Average		
		Computed WRT (days)	Gauged WRT (days)	Computed WRT (days)	Gauged WRT (days)	Computed WRT (days)	Gauged WRT (days)	Computed WRT (days)	Gauged WRT (days)	Computed WRT (days)	Gauged WRT (days)	Computed WRT (days)	Gauged WRT (days)	Difference WRT (days)
Upper Campbell Reservoir	Annual	363.8	355.7	446.2	328.6	408.3	509.1	391.3	432.6	370.4	324.4	396.0	390.1	5.92
	Seasonal	72.0	64.6	90.2	97.6	67.8	84.3	76.3	65.7	64.8	43.2	74.2	71.1	3.13
Upper Quinsam Lake	Annual	216.5	182.0	227.2	191.8	214.7	215.2	200.7	255.1	207.3	210.4	213.3	210.9	2.40
	Seasonal	17.1	18.0	20.5	32.9	23.2	21.8	22.1	12.7	18.1	9.4	20.2	19.0	1.26
Middle Quinsam Lake	Annual	12.9	12.2	12.1	11.2	14.9	16.3	13.8	18.1	12.5	13.4	13.3	14.2	-0.98
	Seasonal	21.6	21.1	24.5	36.2	32.2	28.9	32.7	16.2	22.3	10.4	26.7	22.6	4.11
Gooseneck Lake	Annual	159.2	123.2	240.6	297.7	141.9	102.5	150.5	139.9	184.7	176.3	175.4	167.9	7.49
	Seasonal	17.1	46.3	30.6	-426.2	16.2	25.5	17.0	15.2	26.0	444.7	21.4	21.1	0.31
Lower Campbell Reservoir	Annual	38.4	43.0	47.2	40.2	42.7	61.6	41.0	52.2	38.8	39.6	41.6	47.3	-5.69
	Seasonal	2.2	2.3	2.7	3.4	0.2	0.2	0.2	0.2	0.2	0.1	1.1	1.3	-0.17
John Hart Reservoir	Annual	4.8	5.2	5.7	5.0	5.4	7.5	5.7	6.4	5.1	5.0	5.3	5.8	-0.47
	Seasonal	7.3	8.3	8.8	12.1	8.2	10.4	8.1	7.5	6.9	5.4	7.9	8.7	-0.88

Figure 36. Annual and seasonal WRT computed from the water balance method and measured (gauged) for all JHTMON-5 waterbodies from October 2011 to October 2016.



3.4.3. Analysis of Water Diversion Scenarios

We estimated how different water diversion scenarios could affect the annual and seasonal WRT of six diversion lakes over the last 20 years (October 1997- October 2016), as well as during Pre-WUP conditions (October 1997 – October 2004), and Post-WUP conditions (October 2012 – October 2016) (Table 35, Table 36). Water diversion scenarios also included a significant diversion scenario (90% of flow), above average diversion (70% of flow), and no diversion (0% of flow).

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. Under no diversion, Gooseneck and Snakehead lakes have annual water residence time of 1025.8 and 103.8 days respectively (Table 35). Under average diversion conditions of 30%, annual water residence time decreases by roughly 65% in each lake to 92.9 days in Gooseneck Lake and 9.5 days in Snakehead Lake. In contrast, the annual water residence time of Middle Quinsam Lake increases from 9.7 days under no diversion to 13.0 days with average diversion conditions. Similar trends were observed for scenarios for seasonal water residence time (Table 36).

Brewster, Gray and Whymper lakes receive water from the Salmon River diversion. Under no diversion, Brewster, Gray and Whymper lakes have annual water residence time of 1129.5, 26.6 and 0.36 days respectively (Table 35). Under average diversion conditions of 24%, annual water residence time decreases by roughly 75% in each lake. Similar trends were observed for scenarios for seasonal water residence time except that seasonal water residence time in Gray Lake is much shorter than annual water residence time (Table 36).

Scenarios of water residence time with water diversion are incorporated into predictive models of pelagic versus littoral contributions to fish diet in Section 3.6.2.

Table 35. Estimates of annual water residence time (days) under different water diversion scenarios for the diversion lakes in the Salmon River and Quinsam River watersheds.

Scenario	Salmon Diversion			Quinsam Diversion		
	Brewster Lake	Gray Lake	Whymper Lake	Gooseneck Lake	Middle Quinsam Lake	Snakehead Lake
Scenario 1: Significant Diversion (90/10)	108.0	2.7	0.04	33.7	46.6	3.4
Scenario 2: Above Average Diversion (60/40)	152.6	3.8	0.06	49.4	19.9	5.0
Scenario 3: Average Diversion[1]	305.2	7.6	0.11	92.9	13.0	9.5
Scenario 4: Average Pre-WUP Diversion[2]	238.4	6.0	0.09	77.0	14.2	7.9
Scenario 5: Average Post-WUP Diversion[3]	445.8	11.0	0.16	109.0	12.3	11.1
Scenario 6: No Diversion (0/100)	1129.5	26.6	0.36	1025.8	9.7	103.8

[1] Scenario 3: 20-Year Average Salmon Diversion is 24/76 and 30/70 for the Quinsam Diversion.

[2] Scenario 4: Average Pre-WUP Salmon Diversion is 34/66 and 37/63 for the Quinsam Diversion.

[3] Scenario 5: Average Post-WUP Salmon Diversion is 13/87 and 25/75 for the Quinsam Diversion.

Table 36. **Estimates of seasonal water residence time (days) under different water diversion scenarios for the diversion lakes in the Salmon River and Quinsam River watersheds.**

Scenario	Salmon Diversion			Quinsam Diversion		
	Brewster Lake	Gray Lake	Whymper Lake	Gooseneck Lake	Middle Quinsam Lake	Snakehead Lake
Scenario 1: Significant Diversion (90/10)	161.2	0.14	0.10	1.4	12.1	1.0
Scenario 2: Above Average Diversion (60/40)	228.1	0.19	0.15	2.0	8.2	1.4
Scenario 3: Average Diversion[1]	349.5	0.29	0.21	3.7	6.2	2.6
Scenario 4: Average Pre-WUP Diversion[2]	330.9	0.20	0.15	2.7	7.0	1.9
Scenario 5: Average Post-WUP Diversion[3]	761.6	0.48	0.35	8.8	5.4	6.0
Scenario 6: No Diversion (0/100)	1577.3	1.10	0.76	122.6	5.0	85.4

[1] Scenario 3: 20-Year Average Salmon Diversion is 35/65 and 30/70 for the Quinsam Diversion.

[2] Scenario 4: Pre-WUP Average Salmon Diversion is 57/43 and 43/57 for the Quinsam Diversion.

[3] Scenario 5: Post-WUP Average Salmon Diversion is 15/85 and 10/90 for the Quinsam Diversion.

3.5. Littoral Area Calculations

The estimated area and volume of each waterbody that comprise littoral and shoal (<6 m) habitats are presented in Table 37. Shallow lakes comprise greater proportions of littoral and shoal habitat. The entire areas of Middle Quinsam, Snakehead and Whymper lakes were estimated to comprise littoral habitat, indicating that there is sufficient light present for net primary productivity to occur throughout the full range of depths present in these lakes.

Table 37. Percentage of each water body (area and volume) within the littoral zone and with depth < 6.0 m. The littoral zone was defined as the region with depth less than the estimated euphotic depth (Table 23).

Waterbody	Littoral		< 6 m	
	Area (%)	Volume (%)	Area (%)	Volume (%)
Beavertail Lake	99.4	99.9	32.8	45.5
Brewster Lake	46.9	61.7	12.0	23.1
Gooseneck Lake	93.3	95.4	38.4	50.6
Gray Lake	98.0	99.5	64.9	65.2
John Hart Reservoir	93.3	99.2	21.3	45.1
Lower Campbell Reservoir	72.0	79.0	17.1	33.4
Middle Quinsam Lake	100.0	100.0	67.4	76.3
Snakehead Lake	100.0	100.0	80.2	91.0
Upper Campbell Reservoir	65.1	88.1	12.8	28.1
Upper Quinsam Lake	84.4	92.4	27.7	38.3
Whymper Lake	100.0	100.0	99.6	99.8

3.6. Analysis of Management Questions

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

The total littoral versus pelagic contribution to Cutthroat Trout diets in Upper Campbell, Lower Campbell and John Hart reservoirs can be estimated by summing the contributions of the invertebrate prey types to Cutthroat Trout diets (direct pathway) with the relative contributions of invertebrate prey to the diets of prey fish in the Cutthroat Trout diets (indirect pathway) (Table 38). In contrast, the total littoral versus pelagic contribution to Rainbow Trout diets in these reservoirs is simply the contribution of littoral and pelagic invertebrate prey types.

Upper Campbell Reservoir had the lowest littoral contributions to Cutthroat Trout diets, followed by Lower Campbell Reservoir, and then John Hart Reservoir (Figure 37a). In Upper Campbell Reservoir, which was the only waterbody sampled twice during the JHTMON-5 program, 25% and 14% of Cutthroat Trout (age >2+) diets were estimated to be derived from pelagic energy sources in

2014 and 2016, respectively, while 75% and 86% were estimated to be derived from littoral energy sources. In Lower Campbell Reservoir in 2015, 13% of Cutthroat Trout diets were estimated to be derived from pelagic energy sources, while 87% were estimated to be derived from littoral energy sources. In John Hart Reservoir in 2016, 11% of Cutthroat Trout diets were estimated to be derived from pelagic energy sources while 89% were estimated to be derived from littoral energy sources. The contribution of littoral areas to Cutthroat Trout diets therefore increases as drawdown range decreases, i.e., as water levels become more stable (see Section 1.2.1 for description of water level operations). This provides some support for rejection of the null hypothesis H_01, and therefore implies an effect from water management. However, the estimated littoral energy contributions to Cutthroat Trout diet varied by 9% between 2014 and 2016 in Upper Campbell Reservoir and only by 3% between all three reservoirs in 2016. The low sample size (number of waterbodies and years of comparison) therefore limits the strength of inference that can be drawn associated with potential effects of drawdown. A further confounding factor is that there is a negative correlation between drawdown range and the relative littoral area in each waterbody, i.e., the littoral zone comprises 53.2%, 72.0% and 93.3% of the total areas of each of Upper Campbell, Lower Campbell and John Hart reservoirs, respectively (Table 37).

In contrast, Rainbow Trout have higher littoral contributions to diet in Upper Campbell Reservoir and Lower Campbell Reservoir compared to John Hart Reservoir (Figure 37b), which is opposite to predicted effects from drawdown. In Upper Campbell Reservoir, 36% and 40% of Rainbow Trout (age >2+) diets were estimated to be derived from pelagic energy sources in 2014 and 2016, respectively, while 64% and 60% were estimated derived from littoral sources. In Lower Campbell Reservoir in 2015, 39% of Rainbow Trout diets were estimated to be derived from pelagic sources while 61% were estimated to be derived from littoral sources. In John Hart Reservoir in 2016, 55% of Rainbow Trout diets were estimated to be derived from pelagic sources, while 45% were estimated to be derived from littoral sources.

Cutthroat Trout relative abundance (CPUE) based on gill net catch was similar between the three reservoirs (Figure 37c). In contrast, Rainbow Trout had the highest rate of catch in Lower Campbell Reservoir, followed by Upper Campbell Reservoir, and then John Hart Reservoir (Figure 37d). Littoral catch of Rainbow Trout was particularly low in John Hart Reservoir, which matches the low littoral contribution to Rainbow Trout diet in John Hart Reservoir (Figure 37b), and emphasizes the spatial separation between Cutthroat Trout and Rainbow Trout in this reservoir (Figure 20).

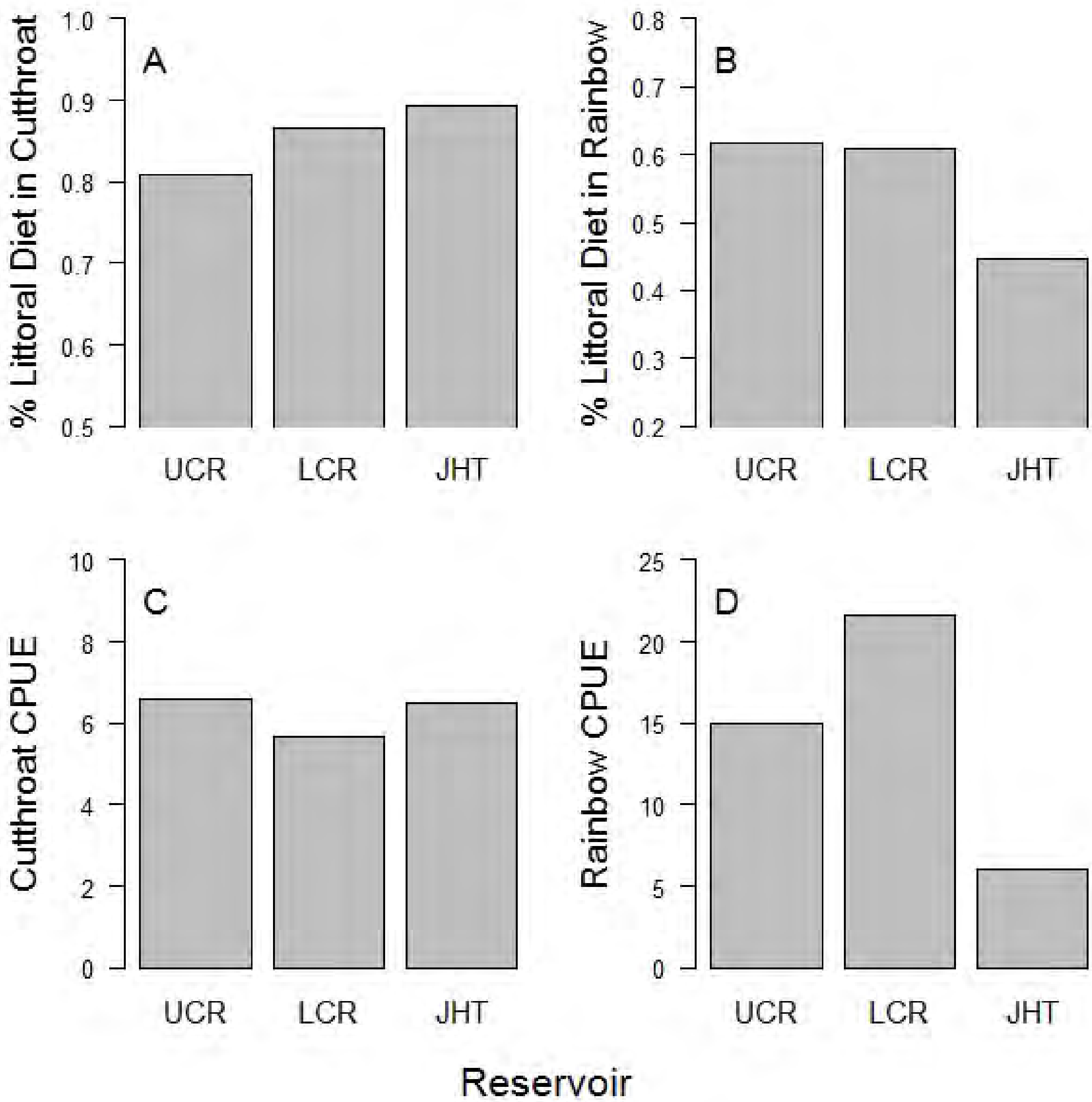
In summary, there appears to be clear diet and habitat separation between Cutthroat Trout and Rainbow Trout in the lakes and reservoirs of the Campbell system. Cutthroat Trout are more piscivorous and have little contribution of zooplankton to their diets as evidenced by stomach contents and stable isotope analysis. Cutthroat Trout are also caught in higher abundance in littoral habitats than in pelagic habitats in these reservoirs. In contrast, Rainbow Trout are caught in higher abundance in pelagic habitats and have a significant dependence on zooplankton production in their diet. Somewhat surprisingly, both species have a high dependence on terrestrial invertebrates in their diets and, despite the large lake areas, are supported by littoral/terrestrial production often to a greater extent than pelagic production. In addition, the prey fish that Cutthroat Trout eat (Threespine Stickleback, Sculpin spp., and juvenile trout) also have a high dependence on terrestrial invertebrates in their diets. Together, these observations predict that Cutthroat Trout have the potential to be more affected by drawdown than Rainbow Trout due to their increased dependence on littoral resources. This is supported by the data, which show a modest reduction in the percentage of littoral contributions in Cutthroat Trout diets in Upper Campbell Reservoir (greatest drawdown range) compared to those in the Lower Campbell and John Hart reservoirs. Rainbow Trout show the opposite trend with the lowest littoral contributions to their diets in John Hart Reservoir, where we expect the littoral habitat to be the most productive relative to the pelagic habitat. In John Hart Reservoir, there was the strongest spatial separation between the two fish species (Figure 20) and there was also the greatest divergence in their diets, as evidenced by both stable isotope and stomach contents analyses (Table 30, Table 38). This means that drawdown could result in tradeoffs in production between Cutthroat Trout and Rainbow Trout. It also raises the question of whether drawdown influences the terrestrial linkage to littoral production via the input of terrestrial vegetation and invertebrates.

Table 38. Total mean contributions of pelagic versus littoral sources to Cutthroat Trout, Rainbow Trout, and Dolly Varden diets in all reservoirs and diversion lakes sampled from 2014 to 2016. Pelagic and littoral contributions are derived from direct (via invertebrates) and indirect (via prey fish) sources.

Year	Waterbody	Consumer[1]	Pelagic Contributions			Littoral Contributions		
			Direct	Indirect	Total	Direct	Indirect	Total
2014	Upper Campbell Reservoir	CT	0.03	0.21	0.25	0.26	0.49	0.75
		RB	0.36	0.00	0.36	0.64	0.00	0.64
	Gooseneck Lake	CT	0.30	0.06	0.36	0.21	0.43	0.64
		DV	0.44	0.03	0.47	0.32	0.21	0.53
	Middle Quinsam Lake	CT	0.10	0.12	0.22	0.37	0.40	0.78
2015	Lower Campbell Reservoir	CT	0.03	0.11	0.13	0.48	0.39	0.87
		RB	0.39	0.00	0.39	0.61	0.00	0.61
	Beavertail Lake	CT	0.22	0.11	0.33	0.31	0.36	0.67
		RB	0.11	0.00	0.11	0.89	0.00	0.89
		DV	0.25	0.07	0.32	0.42	0.25	0.68
	Snakehead Lake	CT	0.12	0.16	0.27	0.39	0.34	0.73
	Upper Quinsam Lake	CT	0.13	0.13	0.26	0.38	0.36	0.74
2016	Upper Campbell Reservoir	CT	0.11	0.03	0.14	0.26	0.60	0.86
		RB	0.40	0.00	0.40	0.60	0.00	0.60
	John Hart Reservoir	CT	0.04	0.07	0.11	0.43	0.46	0.89
		RB	0.55	0.00	0.55	0.45	0.00	0.45
	Brewster Lake	CT	0.06	0.06	0.11	0.30	0.58	0.89
		RB	0.13	0.00	0.13	0.87	0.00	0.87
		DV	0.19	0.03	0.22	0.46	0.32	0.78
	Gray Lake	CT	0.04	0.04	0.08	0.24	0.68	0.92
	Whymper Lake	CT	0.10	0.05	0.15	0.34	0.51	0.85
		RB	0.32	0.00	0.32	0.68	0.00	0.68

[1]CT = Cutthroat Trout, DV = Dolly Varden, RB = Rainbow Trout

Figure 37. Cutthroat Trout (A, C) and Rainbow Trout (B, D) estimates of % littoral energy sources in diet (A, B) and CPUE (C, D) in Upper Campbell (UCR), Lower Campbell (LCR) and John Hart (JHT) reservoirs. Note that Upper Campbell Reservoir data represent the average of two years (2014 and 2016).



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

The total littoral versus pelagic contributions to Cutthroat Trout diets among all diversion and control lakes sampled in 2014 to 2016 can be estimated by summing the contributions of the invertebrate prey to Cutthroat Trout diets (direct pathway) with the relative contributions of invertebrate prey to the prey fish in Cutthroat Trout diets (indirect pathway) (Table 38). Using this method, a total of 8% to 36% of the diets of Cutthroat Trout (age >2+) are estimated to be derived from pelagic sources across all diversion and control lakes. Estimates of littoral contributions to Cutthroat Trout diets ranged from 64% to 92% across all lakes.

Fewer Rainbow Trout were caught in the diversion/control lakes compared to the three reservoirs, and they were completely absent in gill net or trap net catches in Upper Quinsam, Gooseneck, and Snakehead lakes. The largest two lakes sampled, Beavertail Lake and Brewster Lake, had the highest Rainbow Trout catches. Estimates of pelagic contributions to Rainbow Trout diets ranged from 11% to 32% across the three lakes (Beavertail, Brewster, and Whymper lakes) based on Rainbow Trout stable isotope data (Table 38).

The total pelagic contributions to Cutthroat Trout and Rainbow Trout diets were compared to estimates of annual and seasonal water residence time across all 11 lakes and reservoirs sampled in the JHTMON-5 program (Figure 38a,b, Figure 39a,b). No clear linear relationship was observed between annual and seasonal water residence time and total pelagic contributions to Cutthroat Trout or Rainbow Trout diets. However, these relationships may be non-linear (e.g., asymptotic) and may also be influenced by other lake specific variables such as the availability of littoral habitat relative to pelagic habitat in each waterbody and by lake size (Table 37, Figure 38c,d, Figure 39c,d). Therefore, we used a model selection approach to determine the most important variables to predict Cutthroat Trout and Rainbow Trout diets across all sampled lakes and reservoirs. Estimates for water residence time and lake volume were log transformed for the analysis to reduce the influence from outliers and because the relationships may be asymptotic.

The pelagic contribution to Cutthroat Trout diets increased with annual water residence time ($p = 0.008$) and with the proportion of littoral habitat relative to pelagic habitat (% shoal habitat) ($p = 0.021$) in each waterbody (Table 39). The top model for this species included both annual water residence time and % shoal habitat. This model is more than two times more likely than the second ranked model to explain Cutthroat Trout diets. Seasonal water residence time and lake volume were not strong predictors of the pelagic contributions to diet in Cutthroat Trout. This suggests that the contribution of zooplankton to Cutthroat Trout diets increases with longer annual water residence time, which is a rejection of the null hypothesis $H_0 2$, and therefore implies an effect from water management. It also suggests that the relative contribution of zooplankton to Cutthroat Trout diets increases with increased coverage of shallow littoral habitat relative to total lake volume. This relationship between % shoal habitat and pelagic contributions to diet is somewhat counterintuitive, although it can be explained by the observation that zooplankton biomass increases with % shoal habitat in each waterbody (Figure 40). Thus, zooplankton biomass was greater in shallower lakes, likely reflecting higher trophic status of these lakes.

For Rainbow Trout, the top model predicting pelagic contribution to diet was the null model (Table 39). The pelagic energy sources to Rainbow Trout diets were therefore not influenced by annual or seasonal water residence time, lake volume or % shoal habitat. The non-significant effects of water residence time to pelagic energy contributions to Rainbow Trout diet indicates that we cannot reject the null hypothesis $H_0 2$ for Rainbow Trout, implying less of an effect of water management for this species.

Lake productivity was also analyzed across all lakes and reservoirs sampled in JHTMON-5 using zooplankton biomass and Cutthroat Trout CPUE and Rainbow Trout CPUE as response indices (Figure 40, Figure 41, and Figure 42). The top model predicting zooplankton biomass only includes the % shoal habitat in each waterbody ($p = 0.04$) and not annual or seasonal water residence time or lake volume (Table 39). Cutthroat Trout CPUE was positively predicted by annual residence time ($p = 0.06$) and by % shoal habitat ($p = 0.004$), and not by seasonal water residence time or lake volume. The top model for Cutthroat Trout CPUE included the same predictor variables as the top model predicting the pelagic energy sources to Cutthroat Trout diets. For Rainbow Trout CPUE, the top model included only lake volume ($p = 0.003$), which was the only significant ($p < 0.05$) variable in the top model predicting pelagic energy sources to Rainbow Trout diets. This indicates that Rainbow Trout catches increased with increasing lake/reservoir volume and were not affected by annual or seasonal water residence time or % shoal habitat (Table 39).

Classical models of lake food webs assume that all energy sequestered by pelagic zooplankton is derived from autochthonous production of phytoplankton. However, more recent research has shown that in small lakes a significant portion 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). Because terrestrial vegetation $\delta^{13}C$ signatures are typically more enriched than lake algae and phytoplankton $\delta^{13}C$, the carbon isotope signature of zooplankton can be used as an indicator of terrestrial carbon input to lake production. To test this relationship, the $\delta^{13}C$ signatures of zooplankton were modeled as a function of annual and seasonal water residence time, % shoal habitat, and lake volume across the lakes and reservoirs sampled in JHTMON-5. Two outlier lakes were excluded from the analysis (Brewster Lake and Whymper Lake). A positive relationship was observed between % shoal habitat 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 the amount of littoral habitat available increases relative to total lake area (Figure 43). The top model explaining zooplankton $\delta^{13}C$ only included % shoal habitat as a predictor ($p < 0.0001$) (Table 39). This result may help explain the increase in zooplankton biomass with increased % shoal habitat. 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. The reason why Brewster Lake and Whymper Lake did not follow the trend shown by the other lakes is unknown.

Scenarios of annual water residence time with water diversion were incorporated into predictive models derived above to illustrate how water diversion may influence pelagic energy contributions to Cutthroat Trout diet (Table 40). This was completed using the water diversion scenarios for annual water residence time shown in Table 35, the percentage of shoal habitat volume (< 6 m habitat volume) in Table 37 and the model parameter estimates for Cutthroat Trout pelagic diet shown in

Table 39. Pre-WUP (1998 to 2004) average rates diversion were 36.6% of annual available flow in the Quinsam diversion. Post-WUP (2013 to current) this percentage of diversion decreased to 24.7% of annual available flow[1]. This reduction in diversion is predicted to have increased pelagic energy flows to Cutthroat Trout in Gooseneck and Snakehead lakes by around 1.5%. In Middle Quinsam Lake, pelagic energy flows to Cutthroat Trout are predicted to have decreased by 0.7%.

In the Salmon River diversion, pre-WUP average rates diversion were 34.0% of annual available flow in the Salmon River. Post-WUP this percentage of diversion decreased to 13.1% of annual available flow. This operational change is predicted to have increased pelagic energy flows to Cutthroat Trout in Brewster and Gray lakes by up to 3%. Pelagic energy flows to Cutthroat Trout in Whymper Lake has remained similar.

[1] We recognize that this reduction at least partly reflects that the Salmon River Diversion was not operating for part of this period due to physical upgrades.

Figure 38. Percent pelagic contribution to Cutthroat Trout diet by A) annual and B) seasonal water residence time (days), C) percent shoal habitat and D) lake volume (m^3) across all study lakes and reservoirs sampled from 2014 to 2016.

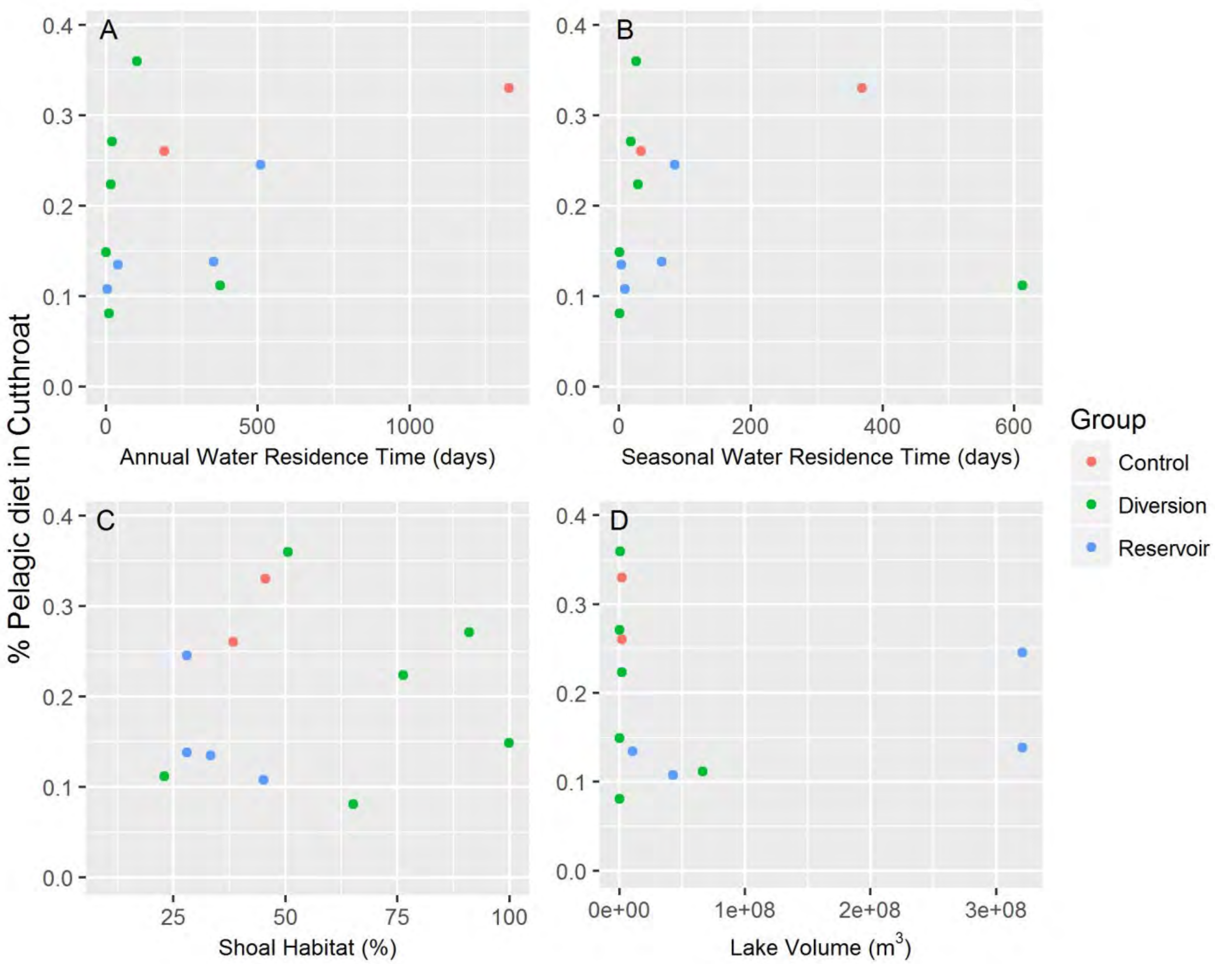


Figure 39. Percent pelagic contribution to Rainbow Trout diet by A) annual and B) seasonal water residence time (days), C) percent shoal habitat and D) lake volume (m^3) across all study lakes and reservoirs sampled from 2014 to 2016.

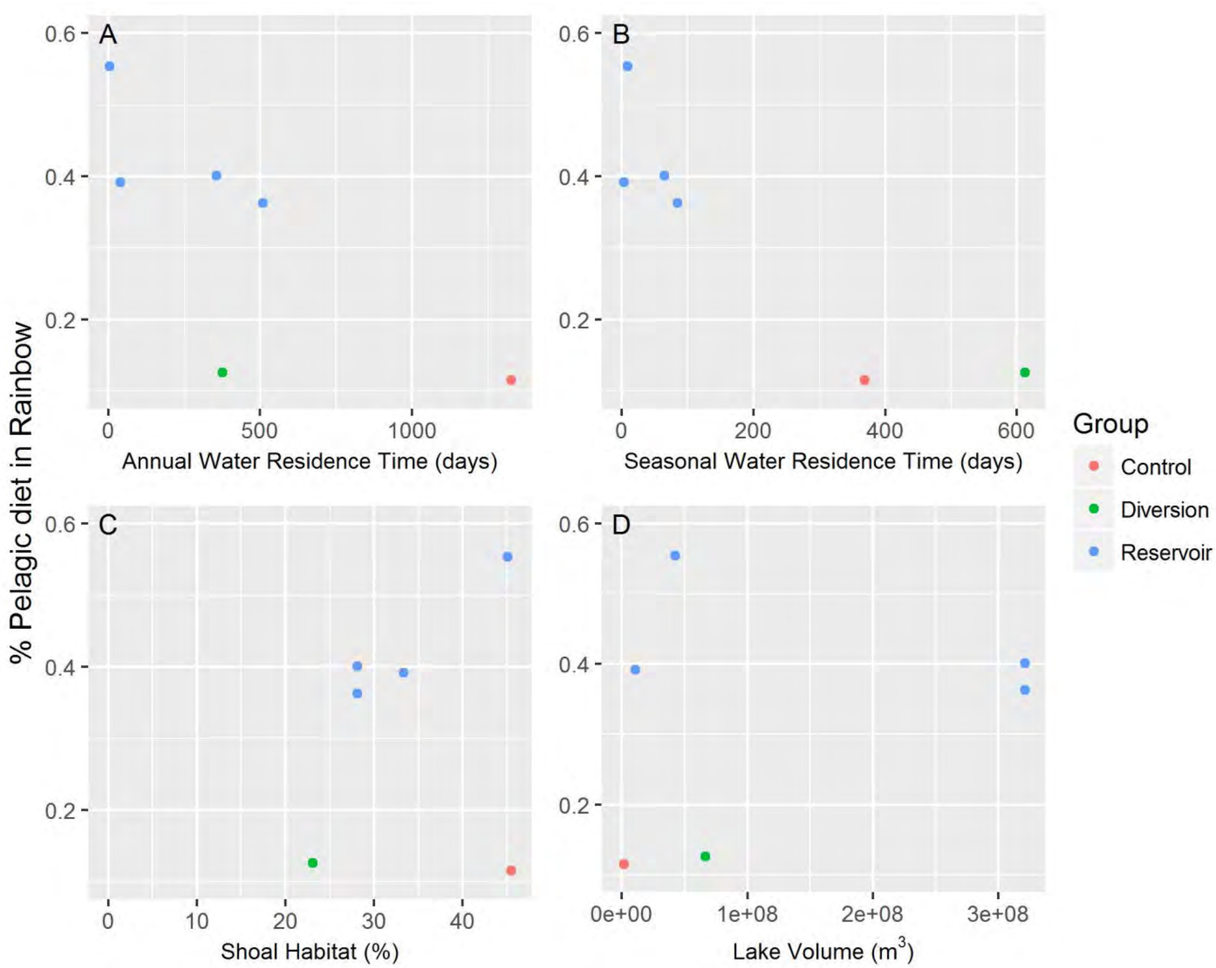


Table 39. Top models predicting Cutthroat and Rainbow Trout energy sources in diet and CPUE, and zooplankton biomass and $\delta^{13}C$ across all study lakes and reservoirs sampled from 2014 to 2016. WRT = water residence time.

Response variable	Variables in Top Model	Estimate	t-value	p-value
% Pelagic Diet in Cutthroat Trout	Intercept	-0.18	-1.52	0.16
	%Shoal Habitat	0.0034	2.80	0.02
	Log Annual WRT	0.049	3.38	0.008
% Pelagic Diet in Rainbow Trout	Intercept	0.32	4.6	0.0057
Zooplankton Biomass	Intercept	4.40	0.46	0.66
	%Shoal Habitat	0.40	2.37	0.04
CPUE of Cutthroat Trout	Intercept	-57.8	-2.39	0.04
	%Shoal Habitat	1.03	3.71	0.004
	Log Annual WRT	7.92	2.11	0.06
CPUE of Rainbow Trout	Intercept	-21.3	-2.75	0.02
	Log Lake Volume	1.84	3.73	0.003
$\delta^{13}C$ of Zooplankton	Intercept	-34.2	-105.46	<0.0001
	%Shoal Habitat	0.05	8.30	<0.0001

Figure 40. Zooplankton biomass (µg/L) by A) annual and B) seasonal water residence time (days), C) percent shoal habitat and D) lake volume (m^3) across all study lakes and reservoirs sampled from 2014 to 2016.

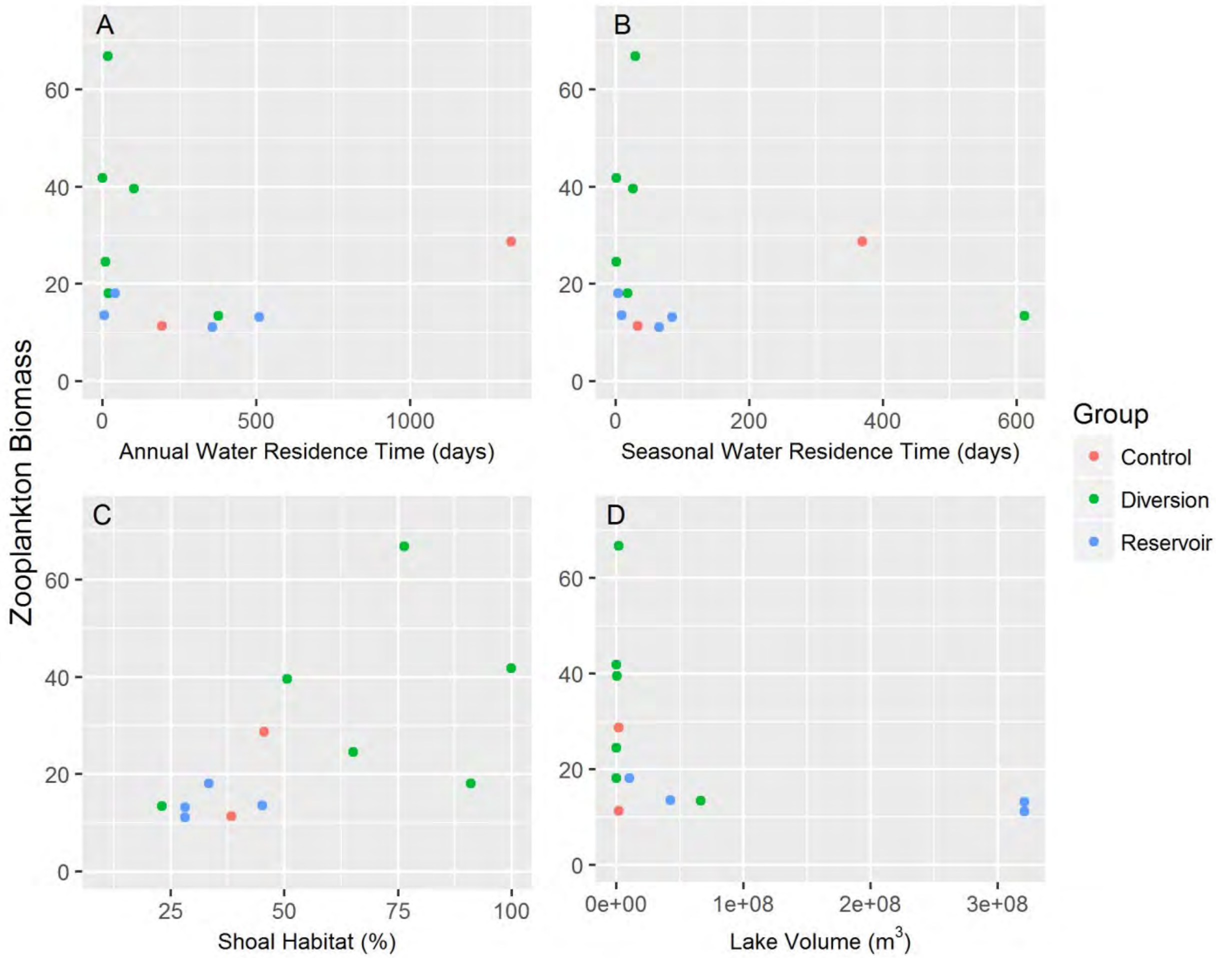


Figure 41. Cutthroat Trout CPUE (# fish per net hour) by A) annual and B) seasonal water residence time (days), C) percent shoal habitat and D) lake volume (m^3) across all study lakes and reservoirs sampled from 2014 to 2016.

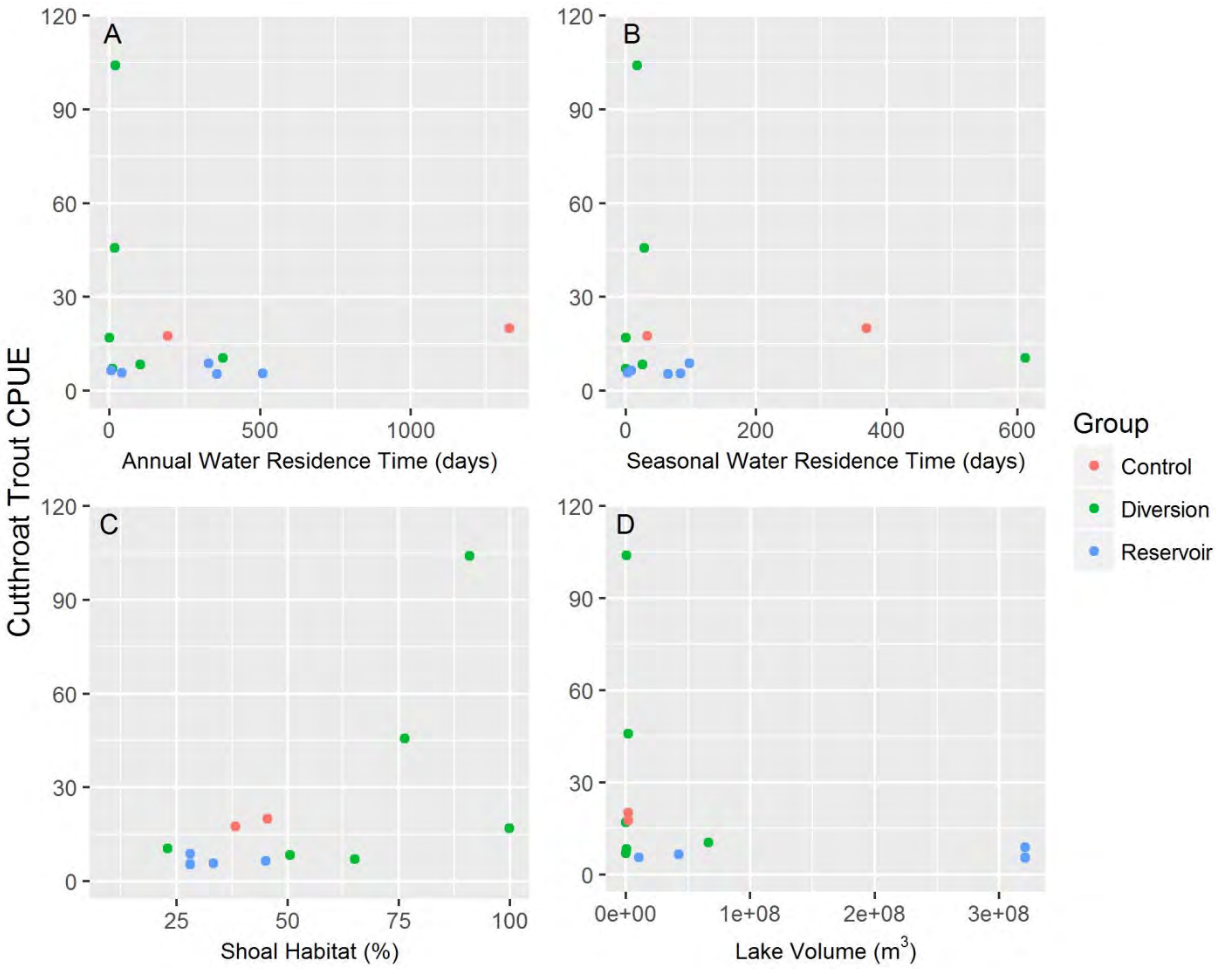


Figure 42. Rainbow Trout CPUE (# fish per net hour) by A) annual and B) seasonal water residence time (days), C) percent shoal habitat and D) lake volume (m^3) across all study lakes and reservoirs sampled from 2014 to 2016.

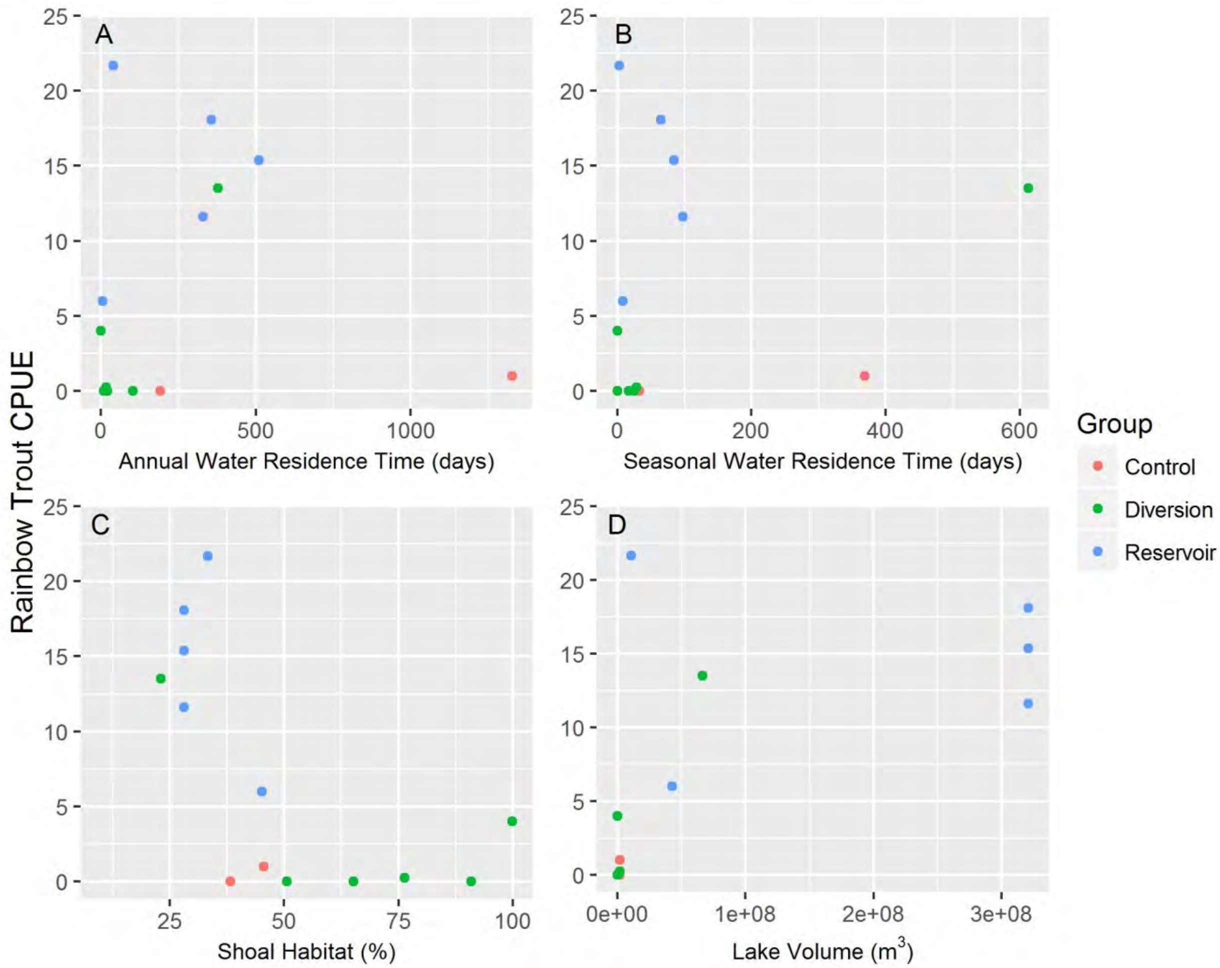


Figure 43. Zooplankton δ[13]C stable isotope signatures by percent shoal habitat across all study lakes and reservoirs sampled from 2014 to 2016 (exception: Brewster and Whymper lakes were excluded as outliers).

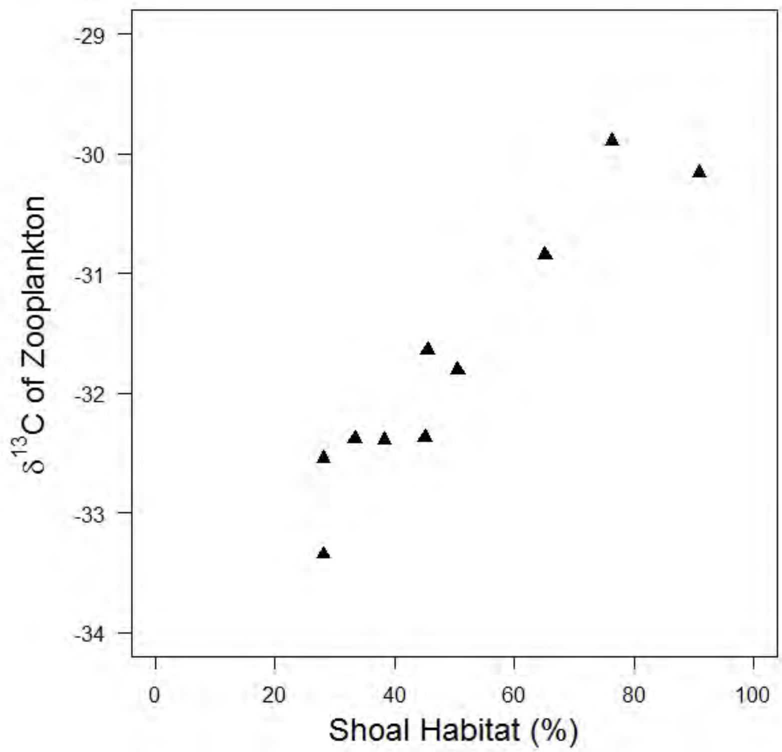


Table 40. Estimated changes in the pelagic energy sources to Cutthroat Trout diet under different water diversion scenarios in the Salmon River and Quinsam River diversion lakes.

Scenario	Salmon Diversion			Quinsam Diversion		
	Brewster Lake	Gray Lake	Whymper Lake	Gooseneck Lake	Middle Quinsam Lake	Snakehead Lake
Scenario 1: Significant Diversion (90/10)	12.8%	10.6%	16.1%	16.6%	26.9%	20.2%
Scenario 2: Above Average Diversion (60/40)	14.5%	11.9%	16.2%	18.4%	22.8%	21.7%
Scenario 3: Average Diversion	17.9%	14.7%	16.4%	21.5%	20.9%	24.5%
Scenario 4: Average Pre-WUP Diversion	16.7%	13.7%	16.4%	20.6%	21.3%	23.7%
Scenario 5: Average Post-WUP Diversion	19.8%	16.3%	16.7%	22.2%	20.6%	25.2%
Scenario 6: No Diversion (0/100)	24.3%	20.4%	17.4%	33.2%	19.6%	35.7%

4. CONCLUSIONS

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

This first component of JHTMON-5 addressed the following hypothesis $H_0 1$:

> $H_0 1$: 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 versus pelagic energy flows to reservoir fish populations.

This hypothesis was tested by comparing the ratio of littoral versus pelagic energy flows to fish across Upper Campbell, Lower Campbell, and John Hart reservoirs that differ in water level fluctuations due to drawdown. With only three waterbodies to study, a statistical test of this hypothesis was not possible. The test was largely qualitative in nature, through the comparison of food webs among reservoirs.

Upper Campbell Reservoir had the lowest littoral contributions to Cutthroat Trout diets, followed by Lower Campbell Reservoir, and then John Hart Reservoir. This suggests that drawdown influences the littoral contribution to Cutthroat Trout diet, which supports a rejection of the null hypothesis $H_0 1$, and therefore implies an effect from water management. This result implies that drawdown reduces littoral productivity (cf. Furey *et al*. 2004), and a shift in the diet of Cutthroat Trout.

In contrast, Rainbow Trout have higher littoral contributions to their diet in Upper Campbell Reservoir and Lower Campbell Reservoir compared to John Hart Reservoir, which is opposite to predicted effects from drawdown.

There is strong evidence for niche separation between Cutthroat Trout and Rainbow Trout in the lakes and reservoirs of the Campbell system. Cutthroat Trout are more dependent on littoral resources and are caught in higher abundance in littoral habitats than in pelagic habitats in these reservoirs. In contrast, Rainbow Trout are caught in higher abundance in pelagic habitats, have a substantial proportion of zooplankton in their diets, and appear to be excluded by Cutthroat Trout in smaller lakes dominated by littoral habitat. This means that drawdown likely has greater effects on Cutthroat Trout than Rainbow Trout due to Cutthroat dependence on littoral resources; it may even result in tradeoffs in production between Cutthroat Trout and Rainbow Trout.

Both Cutthroat Trout and Rainbow Trout have a relatively high dependence on terrestrial invertebrates in their diets and, despite the large surface area of the reservoirs, are supported by littoral/terrestrial production to a greater extent than pelagic production. In addition, the prey fish that Cutthroat Trout eat (Threespine Stickleback, Sculpin spp., and juvenile trout) also have a high dependence on terrestrial invertebrates in their diets. This raises the question of whether drawdown influences the terrestrial linkage to littoral production via the input of terrestrial vegetation and invertebrates – a question that would require additional data to address.

4.2. <u>What is the relationship between residence time (as affected by diversion rate) and lake productivity?</u>

This first component of JHTMON-5 addressed the following hypothesis $H_0 2$:

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

This hypothesis was tested by comparing the ratio of littoral versus pelagic energy flows to fish across 11 lakes and reservoirs in the Campbell River WUP system that differ in average water residence times and effects of water diversion. Because 11 waterbodies were sampled, including Upper Campbell Reservoir that was sampled twice, it was possible to build a statistical model to test the relationship between annual and seasonal water residence time and pelagic energy flows to lake fish populations. In addition to water residence time, % shoal habitat, and lake volume were also included in statistical models to account for other important characteristics of lakes that may influence energy flows to fish.

Pelagic energy flows to Cutthroat Trout increased with annual water residence time and with % shoal habitat in each waterbody. This suggests that Cutthroat Trout feed on zooplankton to a greater extent in waterbodies with longer annual water residence times, which is a rejection of the null hypothesis $H_0 2$, and therefore implies an effect from water management. It also suggests that the contribution of zooplankton to Cutthroat Trout diets increases with increased area of shallow littoral habitat relative to total lake volume. This relationship between % shoal habitat and pelagic contribution to diets is somewhat counterintuitive, although it can be explained by the observation that zooplankton biomass increases with % shoal habitat in each waterbody, potentially because shallower lakes have greater contribution of terrestrially-derived nutrients. However, the underlying causes of these results remain uncertain because the sample size (11 waterbodies) is relatively still relatively small.

For Rainbow Trout, the top model predicting pelagic contribution to diet was the null model. The pelagic energy sources to Rainbow Trout diets were not influenced by annual or seasonal water residence time, lake volume or % shoal habitat. This indicates that the null hypothesis $H_0 2$ should be retained for Rainbow Trout and that Rainbow Trout are less influenced by diversion water management than Cutthroat Trout. Given the high importance of zooplankton as a food source for Rainbow Trout (Table 38) it was expected that this species would be more sensitive than Cutthroat Trout to water management actions that reduce water residence time sufficiently to cause high transport losses (flushing) of zooplankton. A qualification is that Rainbow Trout were only detected in seven of the 11 waterbodies that were sampled, yielding a smaller sample size, and reduced power to detect effects of water residence time on pelagic energy flows to Rainbow Trout.

Lake productivity was also analyzed across all lakes and reservoirs sampled in JHTMON-5 using zooplankton biomass and Cutthroat Trout CPUE and Rainbow Trout CPUE as response variables. The top model predicting zooplankton biomass includes only % shoal habitat in each waterbody and not annual or seasonal water residence time. Cutthroat Trout CPUE was positively predicted by annual water residence time and % shoal habitat, which are the same predictor variables included in the top model for pelagic energy flows to Cutthroat Trout. This suggests that water management through diversion may affect Cutthroat Trout abundance. For Rainbow Trout CPUE, the top model included only lake volume ($p = 0.003$). Rainbow Trout catches decreased with decreasing lake size and these fish were absent from many of the small diversion lakes that were sampled. Thus, it seems that Cutthroat Trout out-compete Rainbow Trout in the smaller and shallower lakes where there is less potential for the two species to occupy separate niches. In these waterbodies, Cutthroat Trout increase their use of pelagic energy sources by increasing their feeding on zooplankton.

A strong positive relationship was also observed between % shoal habitat and the $\delta^{13}C$ signature of zooplankton. This suggests that terrestrial (and/or macrophyte) carbon increasingly contributes to zooplankton production as the amount of shallow littoral habitat increases relative to total lake volume. It also suggests that declines in pelagic production with decreased water residence times may be buffered in small lakes by large relative contributions of terrestrial carbon to zooplankton. These interactions between water residence time, trophic state and terrestrial contributions to pelagic productivity remains an uncertainty.

Gooseneck Lake and Snakehead Lake receive water from the Quinsam River diversion, whereas Middle Quinsam Lake has water diverted upstream of the lake and thus is a donor lake. Annual and seasonal water residence time in Gooseneck Lake and Snakehead Lake decreases with water diversion. Estimated decreases in annual water residence time are from roughly 1,026 to 93 days in Gooseneck Lake and 104 to 10 days in Snakehead Lake. In contrast, annual water residence in Middle Quinsam Lake is expected to increase from roughly 10 days to 13 days with average diversion.

Brewster, Gray, and Whymper lakes receive water from the Salmon River diversion, which decreases annual and seasonal water residence time. Estimated decreases in annual water residence time are from roughly 1,130 to 305 days in Brewster Lake, and 27 to 7.6 days in Gray Lake. The annual water residence time in Whymper Lake is very short and is predicted to change from 0.36 days to 0.11 days with average diversion.

Scenarios of annual water residence time with water diversion were simulated with the top statistical model predicting pelagic energy flows to Cutthroat Trout. In the Quinsam diversion, pre-WUP (1998 to 2004) rates diversion were, on average, 36.6% of annual available flow. This decreased to 24.7% post-WUP (2013 to current). This operational change is predicted to have increased pelagic energy flows to Cutthroat Trout in Gooseneck and Snakehead lakes by around 1.5%. In Middle Quinsam Lake, pelagic energy flows to Cutthroat Trout are predicted to have decreased by 0.7%.

In the Salmon River diversion, pre-WUP average rates diversion were 34.0% of annual available flow. Post-WUP this percentage of diversion decreased to 13.1% of annual available flow. This operational change is predicted to have increased pelagic energy flows to Cutthroat Trout in Brewster and Gray lakes by up to 3%. Pelagic energy flows to Cutthroat Trout in Whymper Lake has remained approximately the same. These results suggest that changes to operations have potential to cause subtle changes to habitat use by Cutthroat Trout.

REFERENCES

Baltas, E.A., N.A., Dervos, and M.A. 2007. Determination of the SCS initial abstraction ratio in an experimental watershed in Greece. Hydrology and Earth System Sciences, 11, 1825-1829.

Barnett, L., B. Bruno, W. Van Bruggen. 1993. Saltspring Island Water Allocation Plan. Province of British Columbia Ministry of Environment, Lands and Parks, Vancouver Island Region. Victoria, B.C., November 1993.

BC Hydro. 2012. Campbell River System Water Use Plan Revised for Acceptance by the Comptroller of Water Rights. November 21, 2012 v6. 46 p.

Beauchamp, D.A. 1990. Seasonal and diel food habits of rainbow trout stocked as juveniles in Lake Washington. Transactions of the American Fisheries Society 119: 475–482.

Bottrell, H.H., A. Duncan, Z.M. Gliwicz, E. Grygierek, A. Herzig, A. Hillbricht-Ilkowska, H. Kurasawa, P. Larsson and T. Weglenska. 1976. A review of some problems in zooplankton production studies. Norwegian Journal of Zoology 24: 419-456.

Bruce, J. 2001. Stage-Area/Volume Relationships for Campbell River Project Reservoirs. Water Use Plan Technical Note WUP JHT-TN-F01, BC Hydro, November 20, 2001.

CCME (Canadian Council of Ministers of the Environment) 2011 Protocols Manual for Water Quality Sampling in Canada. Available online: http://www.ccme.ca/files/Resources/water/water_quality/protocols_document_e_final_101.pdf. Accessed February 3, 2017. 180 p.

Chapman, D.W., H.J. Campbell and J.D. Fortune. 1967. Summer distribution and food of Kokanee salmon in Elk Lake, Oregon. Trans. Am. Fish. Soc. 96:308-312.

Cole, J.J, S.R. Carpenter, J. Kitchell, M.L. Pace, C.T. Solomon, and B. Weidel. 2011. Strong evidence for terrestrial support of zooplankton in small lakes based on stable isotopes of carbon, nitrogen and hydrogen. PNAS 108: 1975-1980.

Craig. J.D.C. and M. Kehler. 2009. Negative Storage Feasibility on Upper Quinsam and Wokas Lakes - Environmental Issues Associated with Maintaining Quinsam River Minimum Fisheries Flows. Report prepared for Ministry of Environment. 55 p.

Dayton & Knight Ltd. 2001. District of Campbell River Watershed Management Plan.

DeNiro, M.J., and S. Epstein. 1978. Influence of diet on the distribution of nitrogen isotopes in animals. Geochimica et Cosmochimica Acta 42: 495-506.

DeNiro, M.J., and S. Epstein. 1981. Influence of diet on the distribution of nitrogen isotopes in animals. Geochimica et Cosmochimica Acta 45: 341-351.

Dumont, H.T, I. van de Velde, and S. Dumont. 1975. The dry weight estimate of biomass in a selection of Cladocera, Copepoda and Rotifera from the plankton, periphyton, and benthos of continental waters. Oecologia 19: 225-246.

Dunne, T. and L.B. Leopold. 1978. Water in Environmental Planning. W.H. Freeman and Company, New York.

Elhakeem, M. and A. N. Papanicolaou. 2009. Estimation of the Runoff Curve Number via Direct Rainfall Simulator Measurements in the State of Iowa, USA. Water Resources Management 23 (12):2455–2473.

Furey, P.C., R.N. Nordin, A. Mazumder. 2004. Water level drawdown affects physical and biogeochemical properties of littoral sediments of a reservoir and a natural lake. Lake and Reservoir Management 20: 280–295.

George D.G. and M.A. Hurley. 2002. Using a continuous function for residence time to quantify the impact of climate change on the dynamics of thermally stratified lakes. Journal of Limnology 62 (Suppl. 1): 21-26.

Gillooly, J. F. 2000. Effect of body size and temperature on generation time in zooplankton. Journal of Plankton Research 22: 241–251.

Hanson, J.M. and R.H. Peters. 1984. Empirical prediction of crustacean zooplankton biomass and profundal macrobenthos biomass in lakes. Canadian Journal of Fisheries and Aquatic Sciences 41: 439–445.

Hatfield, T. 2000. A review of the physical, chemical, and ecological processes in reservoirs and diversion-affected lakes in the Campbell River area, Vancouver Island, British Columbia. A consultant's report prepared for BC Hydro, October 12, 2000.

Hawkins, B.E., and M. S. Evans. 1979 Seasonal cycles of zooplankton biomass in Southeastern Lake Michigan. Journal of Great Lakes Research. 5: 256-263.

Hawkins, R.H., A.T. Hjelmfelt Jr., and A.W. Zevenbergen. 1985. Runoff probability, storm depth and curve numbers. Journal of Irrigation and Drainage Engineering (ASCE) Vol. 111 (4): 330–340

Hawkins, R.H., R. Jiang, D.E. Woodward, A.T. Hjelmfelt, and J.A. Van Mullem. 2002. Runoff Curve Number Method: Examination of the Initial Abstraction Ratio. Proceedings of the Second Federal Interagency Hydrologic Modeling Conference, Las Vegas, Nevada. U.S. Geological Survey. doi:10.1111/j.1752-1688.2006.tb04481

Hocking, M.D., A. Yeomans-Routledge, N. Swain, M. Marquardson, J. Abell, and T. Hatfield. 2015. JHTMON5 – Year 1 Annual Monitoring Report. Draft V1. Consultant's report prepared for BC Hydro by Ecofish Research Ltd, March 12, 2015.

Huang, M., G. Jacques, Z. Wang, and G. Monique. 2006. A modification to the soil conservation service curve number method for steep slopes in the Loess Plateau of China. Hydrological Processes 20 (3), 579.

Iowa State University. 2015. Bug Guide online resource. Available online at: http://bugguide.net/node/view/15740. Accessed on December 4, 2015.

Jiang, R. 2001. Investigation of Runoff Curve Number Initial Abstraction Ratio. MS Thesis Watershed Management, University of Arizona. 120pp.

Lamont, S.J., R.N. Eli, and J.J. Fletcher. 2008. Continuous Hydrologic Models and Curve Numbers: A Path Forward. Journal of Hydrologic Engineering 13(7), 621-636.

Lewis, W.M. 2009. Ecological Zonation in Lakes. Encyclopedia of Inland Waters:416–422.

MacIsaac, E.A. and J.G. Stockner. 1985. Current Trophic State and Potential Impacts of Coal Mine Development on Productivity of Middle Quinsam and Long Lake. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1381. 69 p.

Matthews, B., and A. Mazumder. 2003. Compositional and inter-lake variability of zooplankton affect baseline stable isotope signatures. Limnology and Oceanography 48: 1977-1987.

Matthews, B., and A. Mazumder. 2005. Consequences of large temporal variability of zooplankton $\delta^{15}N$ for modeling fish trophic position and variation. Limnology and Oceanography 50: 1404-1414.

McCauley, E. 1984. Chapter 7: The estimation of the abundance and biomass of zooplankton in samples. In: Downing, J.A. and Rigler, F.H. (eds.) A manual on methods for the assessment of secondary production in fresh waters. 2nd edition. IBP Handbook 17. Blackwell Scientific Publications.

McIntyre, J.K., D.A. Beauchamp, M.M. Mazur, and N.C. Overman. 2006. Ontogenetic trophic interactions and benthopelagic coupling in Lake Washington: evidence from stable isotopes and diet analysis. Transactions of the American Fisheries Society 135: 1312-1328.

McPhail, J.D. 2007. The freshwater fishes of British Columbia. The University of Alberta Press, Edmonton, Alberta.

MFLNRO (BC Ministry of Forest, Lands, and Natural Resources). 2016b. Baseline Thematic Mapping Present Land Use Layers. Open Sourced Licensed Data. Available online: http://catalogue.data.gov.bc.ca/dataset/baseline-thematic-mapping-present-land-use-version-1-spatial-layer Accessed November 7, 2016.

McIntyre, J.K., D.A. Beauchamp, M.M. Mazur, and N.C. Overman. 2006. Ontogenetic trophic interactions and benthopelagic coupling in Lake Washington: evidence from stable isotopes and diet analysis. Transactions of the American Fisheries Society 135: 1312-1328.

McPhail, J.D. 2007. The Freshwater Fishes of British Columbia. The University of Alberta Press, Canada.

Moss, B. 2010. Ecology of Freshwaters: A View for the 21st Century, 4th edition. Wiley-Blackwell. Oxford, UK.

Parnell, A.C., and A.L. Jackson. 2011. R package SIAR: Stable Isotope Analysis in R.

Parnell, A.C., and A.L. Jackson. 2013. siar: Stable Isotope Analysis in R. R package version 4.2. Available from: http://CRAN.R-project.org/package=siar.

Parnell, A.C., R. Inger, S. Bearhop, and A.L. Jackson. 2010. Source partitioning using stable isotopes: coping with too much variation. PloS One 5: e9672.

PCIC (Pacific Climate Impacts Consortium) 2013. BC Station Data. Available online at: http://tools.pacificclimate.org/dataportal/pcds/map/Information. Accessed on October 27, 2016.

Perrin, C.J., T.B. Stables, and S. Bennett. 2012. Limnology of John Hart Reservoir in 2010-2011. Prepared by Limnotek Research and Development Inc. and Hemmera for BC Hydro. 58p.

Perrin, C., N. Swain, M. Hocking, P. Dinn, T. Hatfield and A. Marriner. 2016. JHTMON4: Upper and Lower Campbell Reservoirs Littoral Productivity Assessment Year 1 Annual Monitoring Report. Draft V1. Consultant's report prepared for BC Hydro by Laich-Kwil-Tach Environmental Assessment Ltd. Partnership, Limnotek Research and Development Inc. and Ecofish Research Ltd., May 9, 2016.

Peterson, B.J., and B. Fry. 1987. Stable isotopes in ecosystem studies. Annual Review of Ecology and Systematics 18: 293-320.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

RIC (Resources Inventory Committee). 2001. Reconnaissance (1:20 000) Fish and Fish Habitat Inventory: Standards and Procedures. Prepared by BC Fisheries Information Services Branch for the Resources Inventory Committee. April 2001. Version 2.0. Available online at: https://www.for.gov.bc.ca/hts/risc/pubs/aquatic/recon/. Accessed on May 29, 2017.

Rosen, R.A. 1981. Length-dry weight relationships of some freshwater zooplankton. Journal of Freshwater Ecology 1: 225-229.

Rosenberry, D.O., T.C. Winter, D.C., Buso, and G.E. Likens. 2007. Comparison of 15 evaporation methods applied to a small mountain lake in the northeastern USA. Journal of Hydrology 340, 149– 166.

Rueda, F., E. Moreno-Ostos, and J. Armengol. 2006. The residence time of river water in reservoirs. Ecological Modelling 191(2), 260–274.

SCS (Soil Conservation Service). 1972. National Engineering Handbook, Section 4: Hydrology. U.S. Department of Agriculture, Washington D.C., 1979340, 149–166.

Semmens, B.X., J.W. Moore, and E.J. Ward. 2009. Improving Bayesian isotope mixing models: a response to Jackson *et al.* (2009). Ecology Letters 12: E6-E8.

Silvestri, S. and Fosker, G. 2003. Beavertail Lake Rainbow Trout Strain Evaluation and Stocking Assessment, 2003. Ministry of Water, Land and Air Protection. 52 p.

Sprague, J.B. 2007. Nine lakes on Salt Spring Island, B.C.: size, watershed, inflow, precipitation, runoff and evaporation. Sprague Associates Ltd., Salt Spring Island, B.C. Report, 18 p.

Sterner, R.W. 2009. Role of Zooplankton in Aquatic Ecosystems, in: Likens, G. E. (ed) Encyclopedia of Inland Waters. Academic Press, Oxford, pp. 513-523.

Stockner, J.G. and K.S. Shortreed. 1989. Algal picoplankton production and contribution to food-webs in oligotrophic British Columbia lakes. Hydrobiologia 173: 151–166.

Sweeting, C., J. Barry, C. Barnes, N. Polunin, and S. Jennings. 2007a. Effects of body size and environment on diet-tissue $\delta^{15}N$ fractionation in fishes. Journal of Experimental Marine Biology and Ecology 340: 1–10.

Sweeting, C., J. Barry, N. Polunin, and S. Jennings. 2007b. Effects of body size and environment on diet-tissue $\delta^{13}C$ fractionation in fishes. Journal of Experimental Marine Biology and Ecology 352: 165–176.

Tetra Tech EBA. 2014. Holland Lake and Stocking Lake Hydrology Update. Consultant's report prepared for the Town of Ladysmith, September 2014.File No. V13203036-01.

Thornthwaite, C. W. 1948. An approach toward a rational classification of climate. Geographical Review 38 (1): 55–94. doi:10.2307/210739.

US EPA (US Environmental Protection Agency). 2003. Standard operating procedure for zooplankton analysis. LG403. Revision 03, February 2003. 20 p.

USDA-SCS. 1985. Chapter 9 Hydrologic Soil-Cover Complexes. National Engineering Handbook, Section 4 - Hydrology. Washington, D.C. USDA-SCS. Available online at: http://www.wcc.nrcs.usda.gov/ftpref/wntsc/H&H/NEHhydrology/ch9.pdf. Accessed on November 7, 2016

Valentine, K.W.G., P.N. Sprout, T.E. Baker, L.M. Lawkulich (Eds.). 1978. The Soil Landscapes of British Columbia. BC Ministry of Environment, Resource Analysis Branch. 197p. Available online: http://www.env.gov.bc.ca/soils/landscape/index.html Accessed on November 7, 2016.

Vander Zanden, M.J., and Y. Vadeboncoeur. 2002. Fishes as integrators of benthic and pelagic food webs in lakes. Ecology 83: 2152-2161.

Vidal, J., T. Serra, J. Colomer, R. Marce´, F. J. Rueda, and X. Casamitjana. 2012. Localized algal blooms induced by river inflows. Aquat. Sci. 74: 315–327.

Ward, A.D., and W.J. Elliot. 1995. Environmental hydrology. Lewis-CRC, New York.

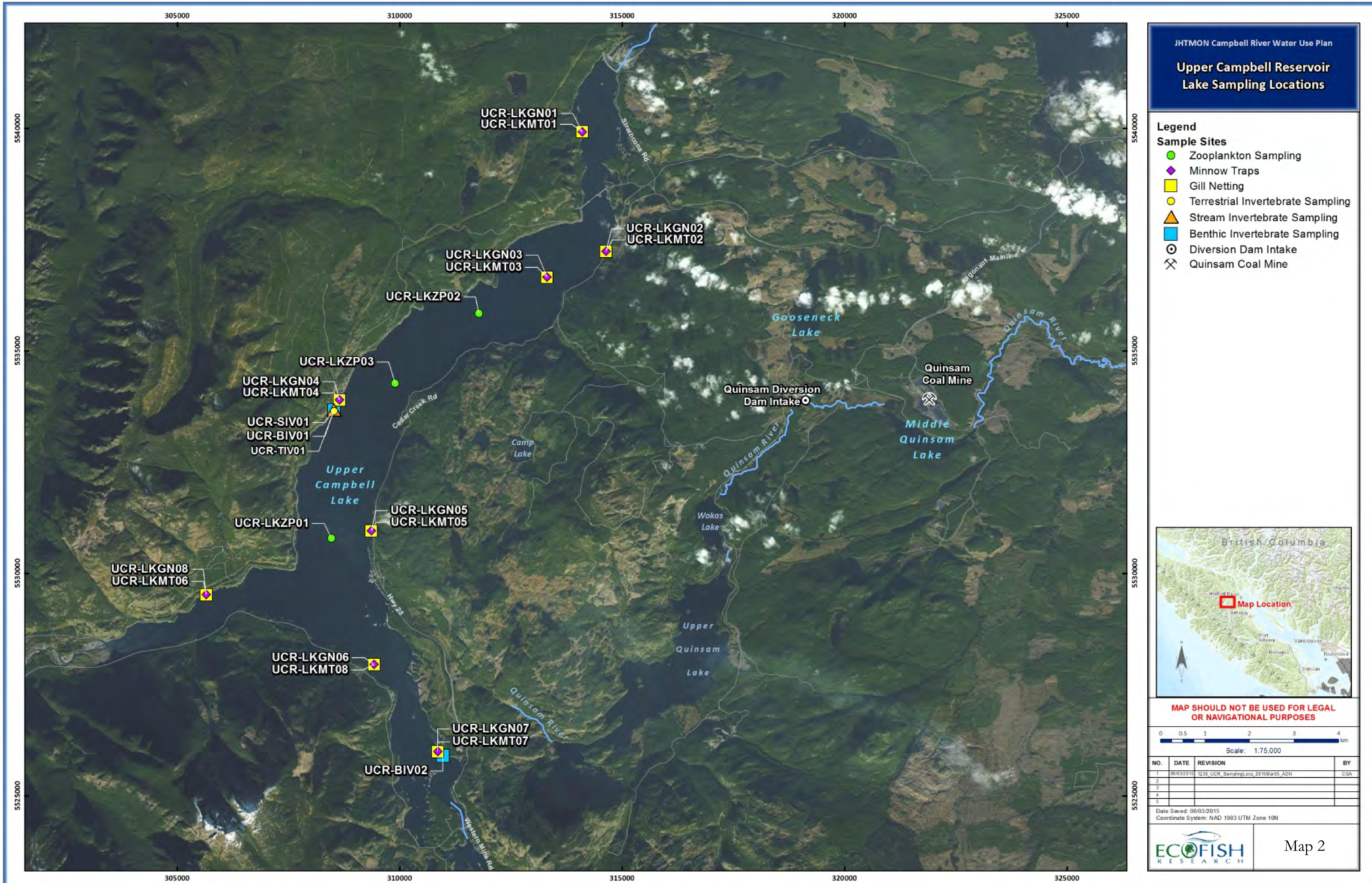
Watkins, J., L. Rudstam, and K. Holeck. 2011. Length–weight regressions for zooplankton biomass calculations – A review and a suggestion for standard equations. Cornell Biological Field Station Technical Report. Cornell University, USA. 17 p.

White, E.D., G.W. Feyereisen, T.L. Veith, and D.D. Bosch. 2009. Improved Daily Water Yield Estimates in the Little River Watershed: SWAT Adjustments. Transaction ASABE 52(1), 69-79.

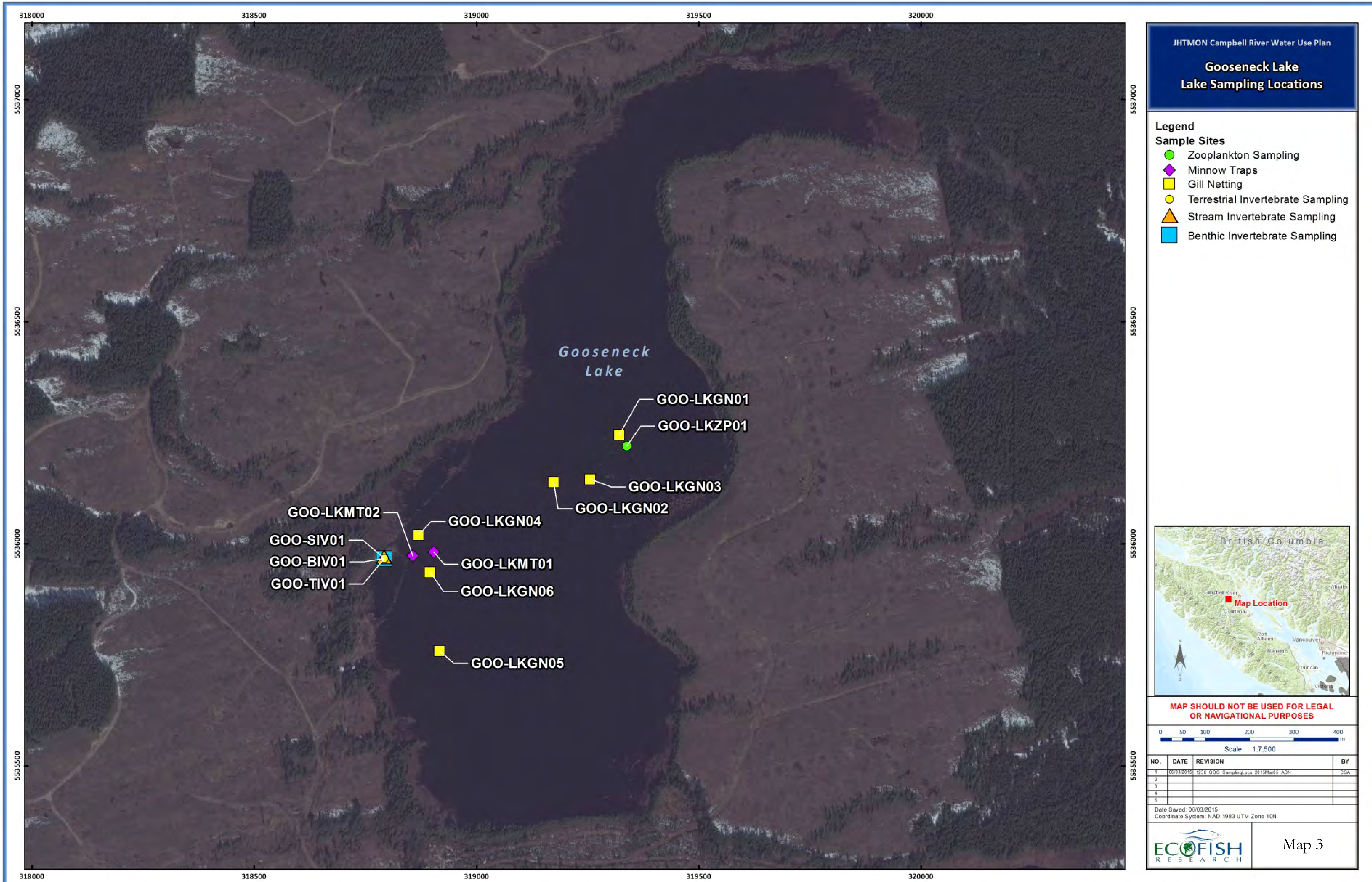
Witty, L. M. 2004. Practical guide to identifying freshwater crustacean zooplankton. Cooperative Freshwater Ecology Unit, Ontario, Canada. 60 p. Available online at: http://142.51.79.168/NR/rdonlyres/F65B174D-EAFA-47DF-8278-862E499FABB8/0/ZooplanktonGuidetoTaxonomy.pdf. Accessed on December 04, 2015.

Xu, C.Y. and V.P Singh. 2001. Evaluation and generalization of temperature-based methods for calculating evaporation. Hydrological Processes 15: 305–319.

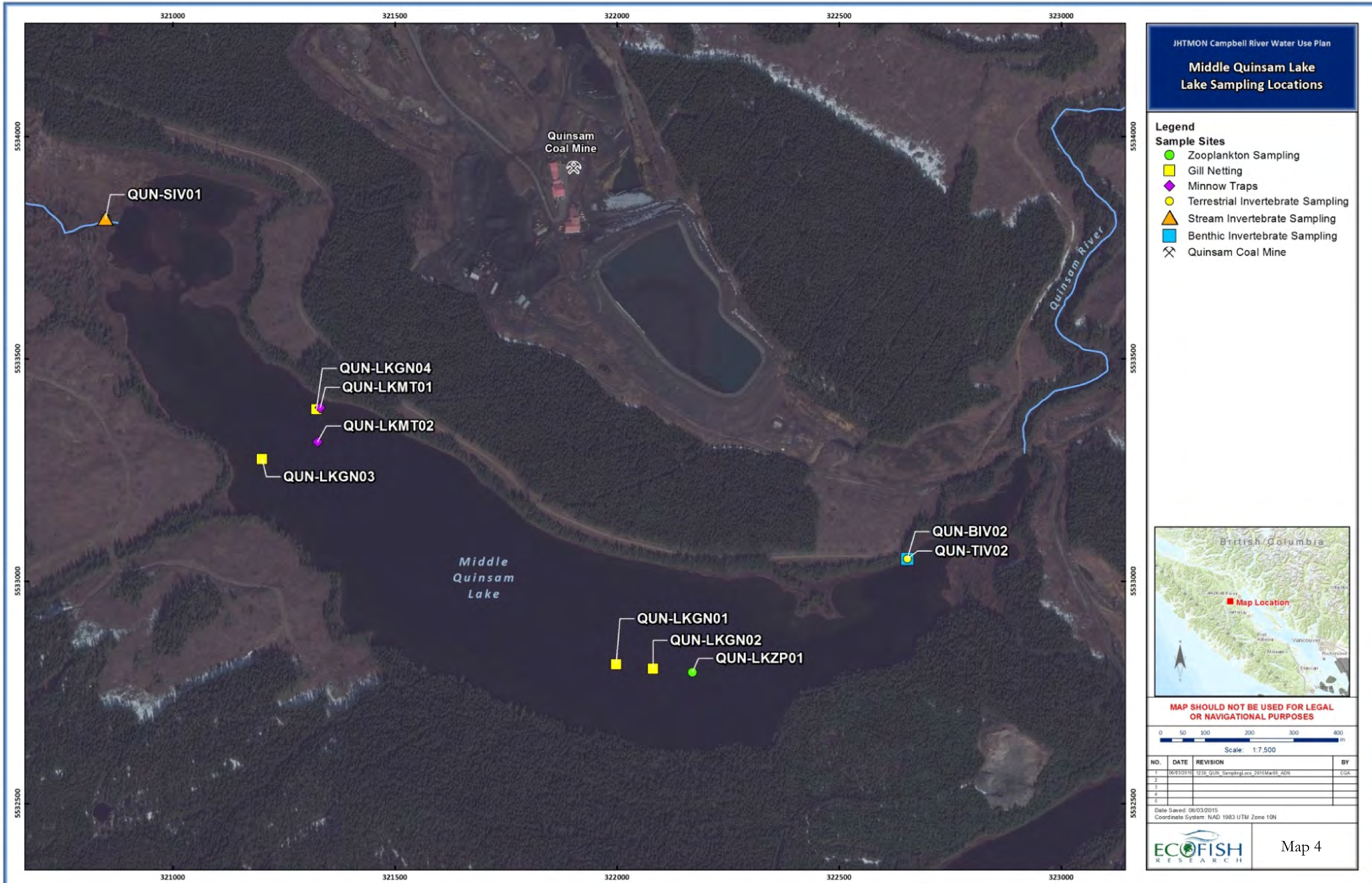
PROJECT MAPS



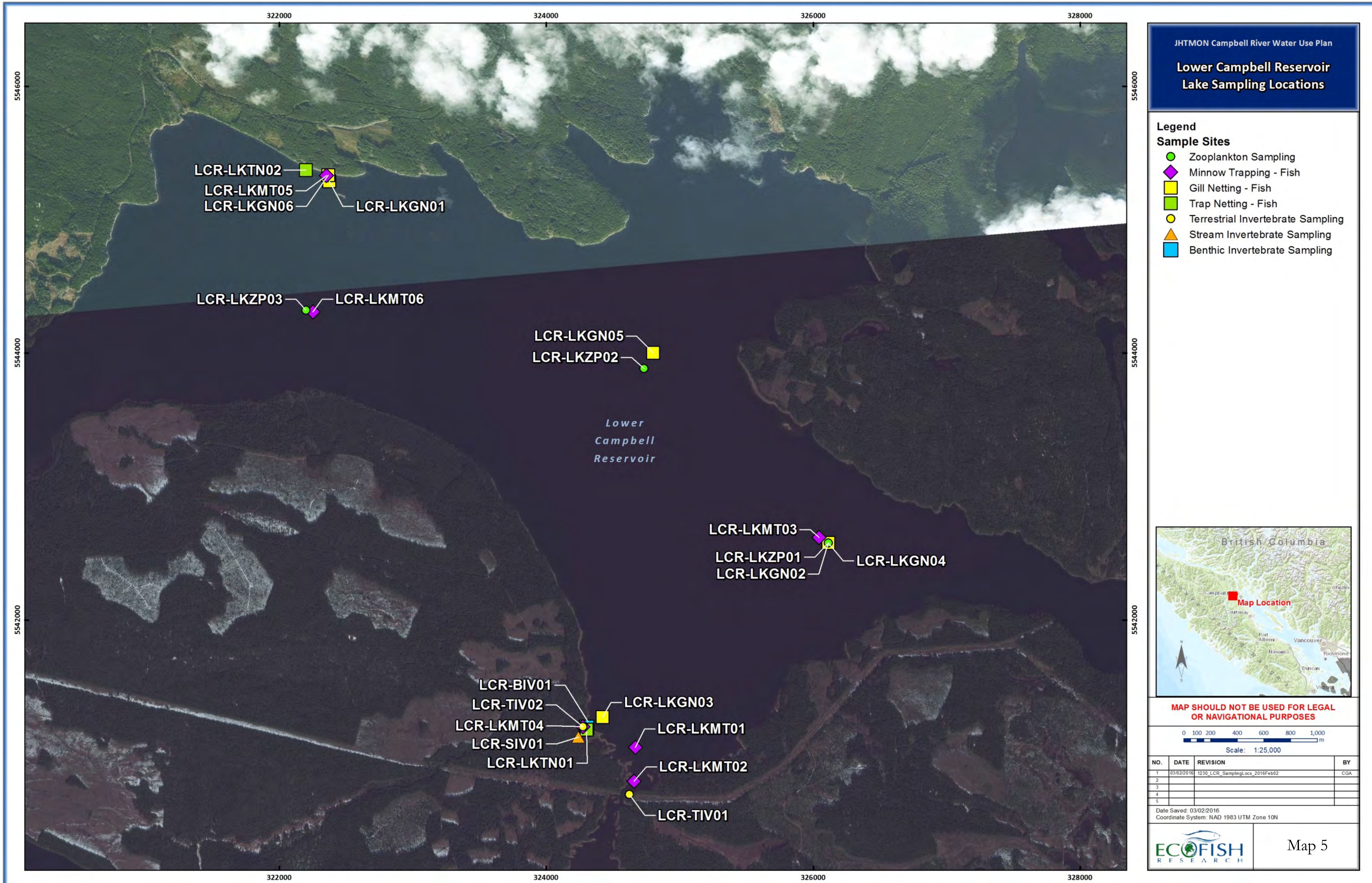
Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_UCR_SamplingLocs_2015Mar05_ADN.mxd



Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_GOO_SamplingLocs_2015Mar05_ADN.mxd



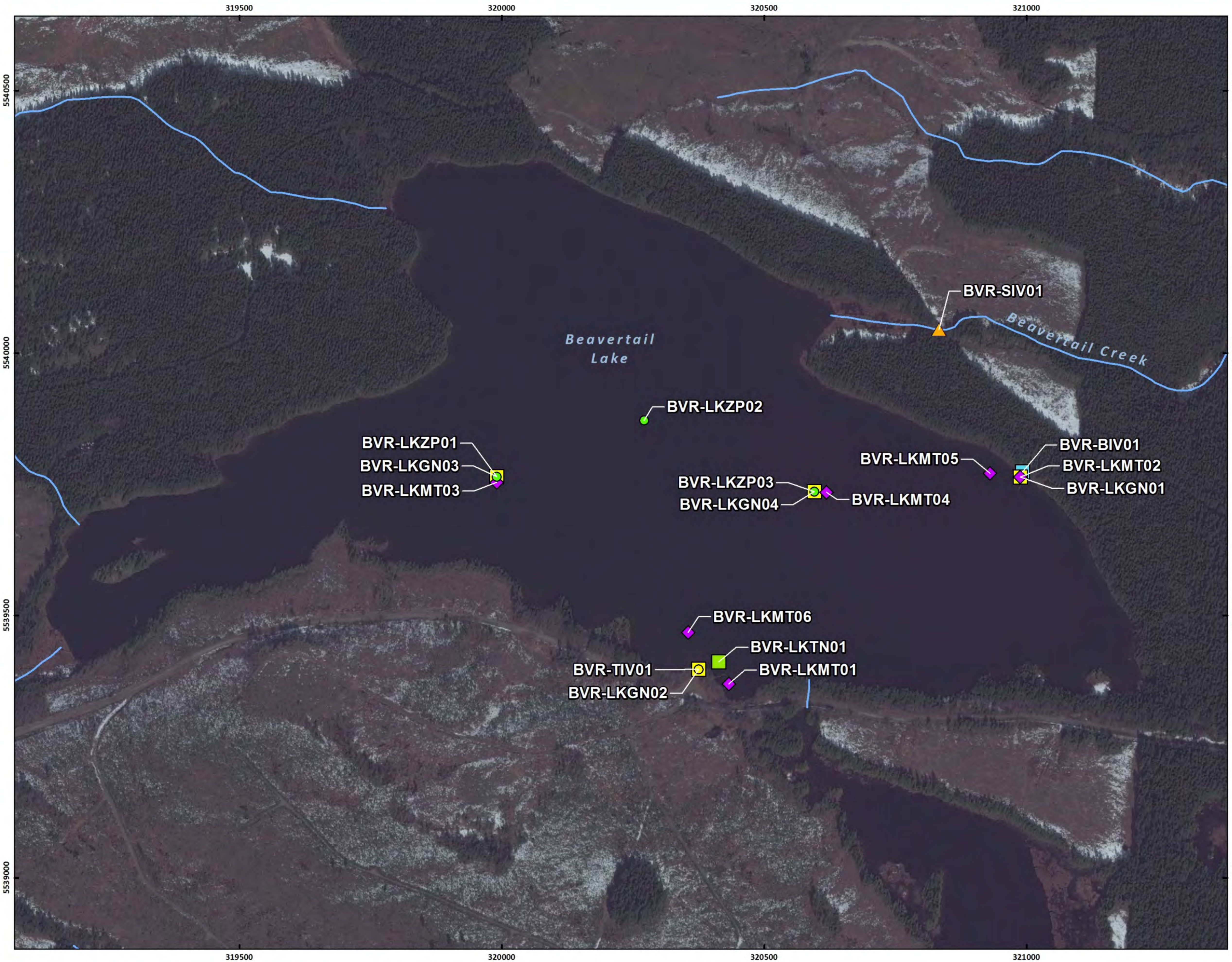
Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_QUN_SamplingLocs_2015Mar05_ADN.mxd



Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_LCR_SamplingLocs_2016Feb02.mxd

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

0 50 100 200 300 m

Scale: 1:7,000

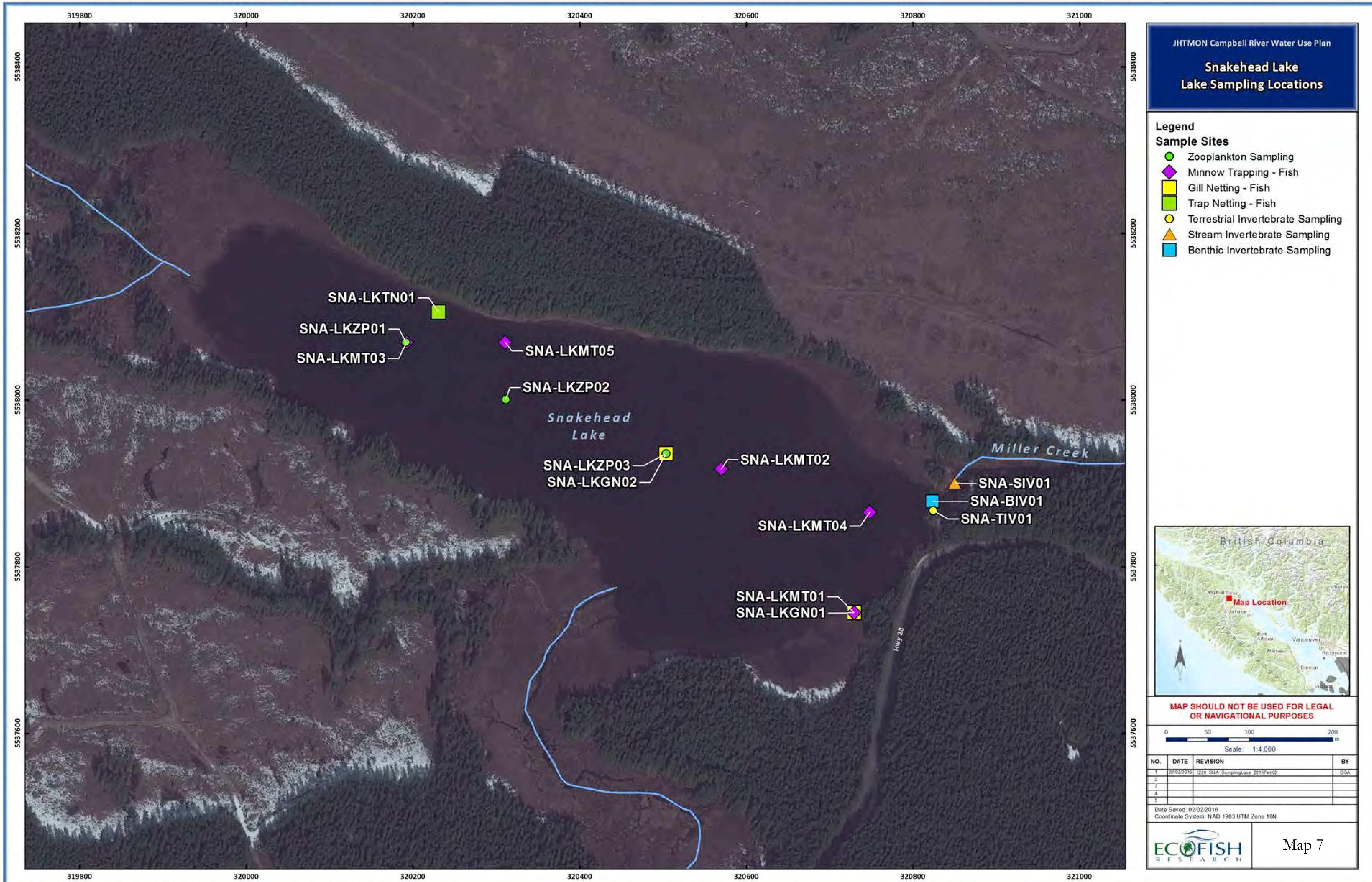
NO.	DATE	REVISION	BY
1	02/02/2016	1230_BVR_SamplingLocs_2016Feb02	CGA
2			
3			
4			
5			

Date Saved: 02/02/2016
Coordinate System: NAD 1983 UTM Zone 10N

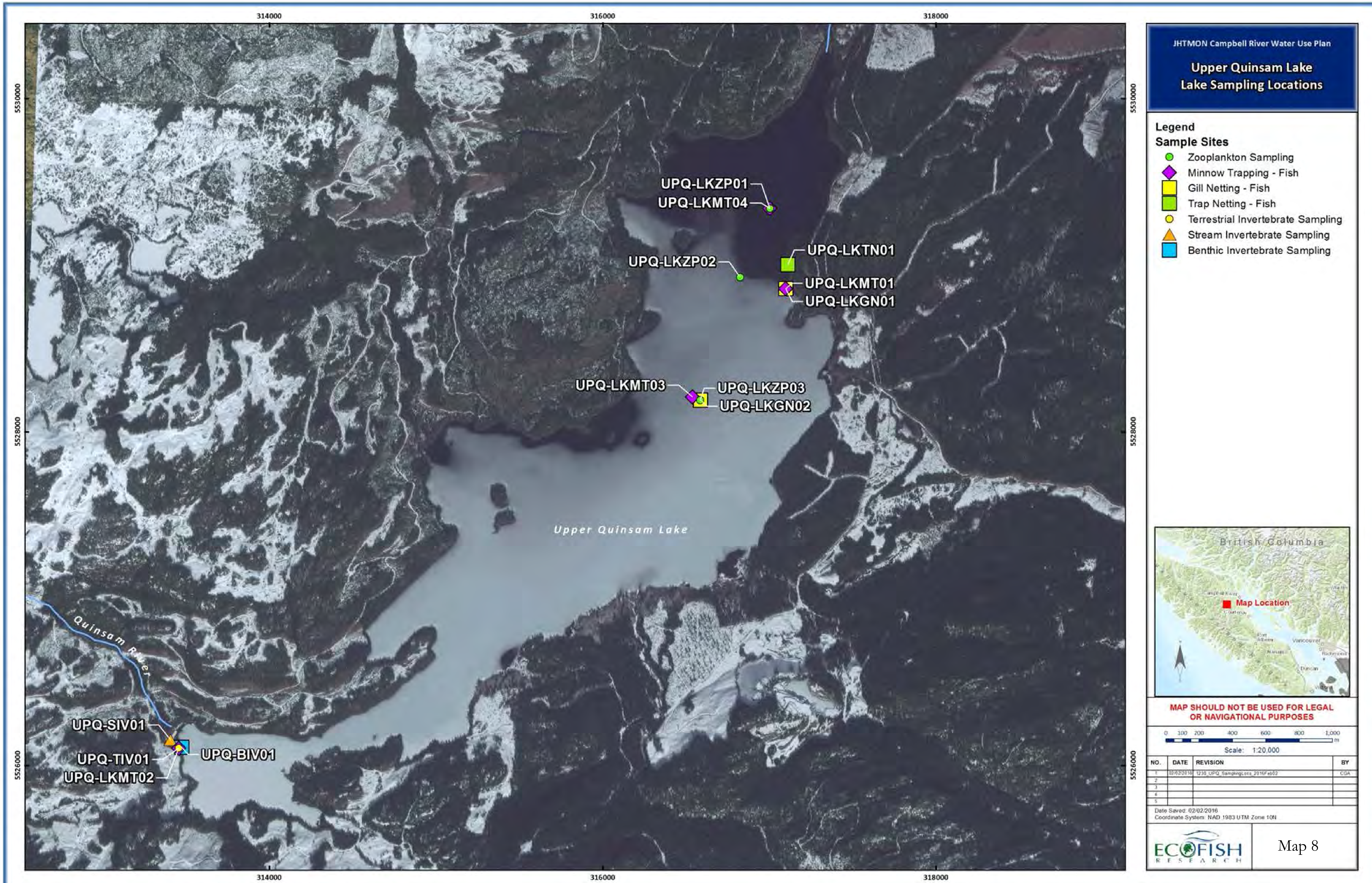
ECOFISH RESEARCH

Map 6

Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_BVR_SamplingLocs_2016Feb02.mxd



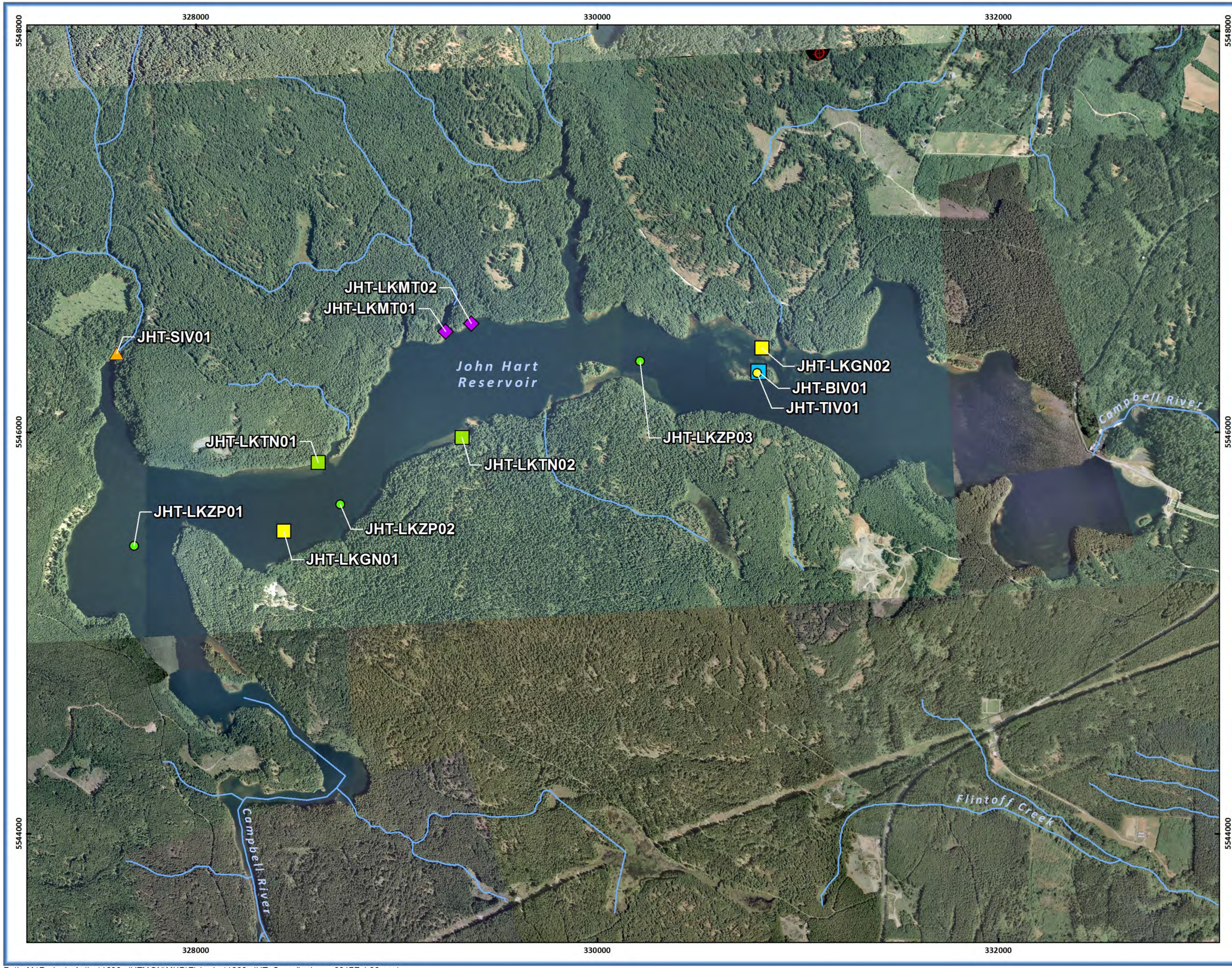
Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_SNA_SamplingLocs_2016Feb02.mxd



Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_UPQ_SamplingLocs_2016Feb02.mxd

JHTMON Campbell River Water Use Plan

John Hart Reservoir Lake Sampling Locations



Legend

Sample Sites

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

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 100 200 400 600 800 1,000 m

Scale: 1:18,000

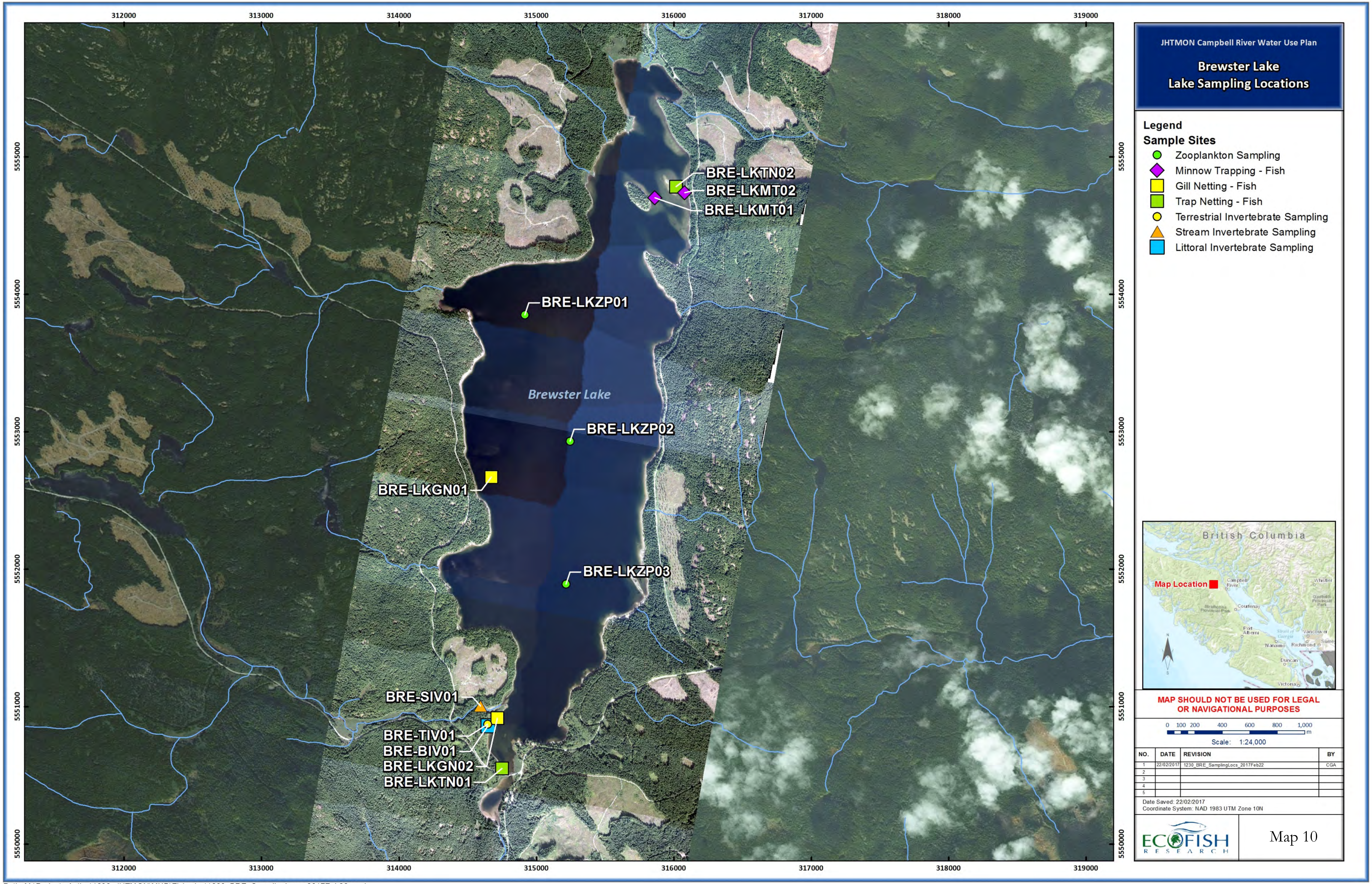
NO.	DATE	REVISION	BY
1	22/02/2017	1230_JHT_SamplingLocs_2017Feb22	CGA
2			
3			
4			
5			

Date Saved: 22/02/2017
Coordinate System: NAD 1983 UTM Zone 10N

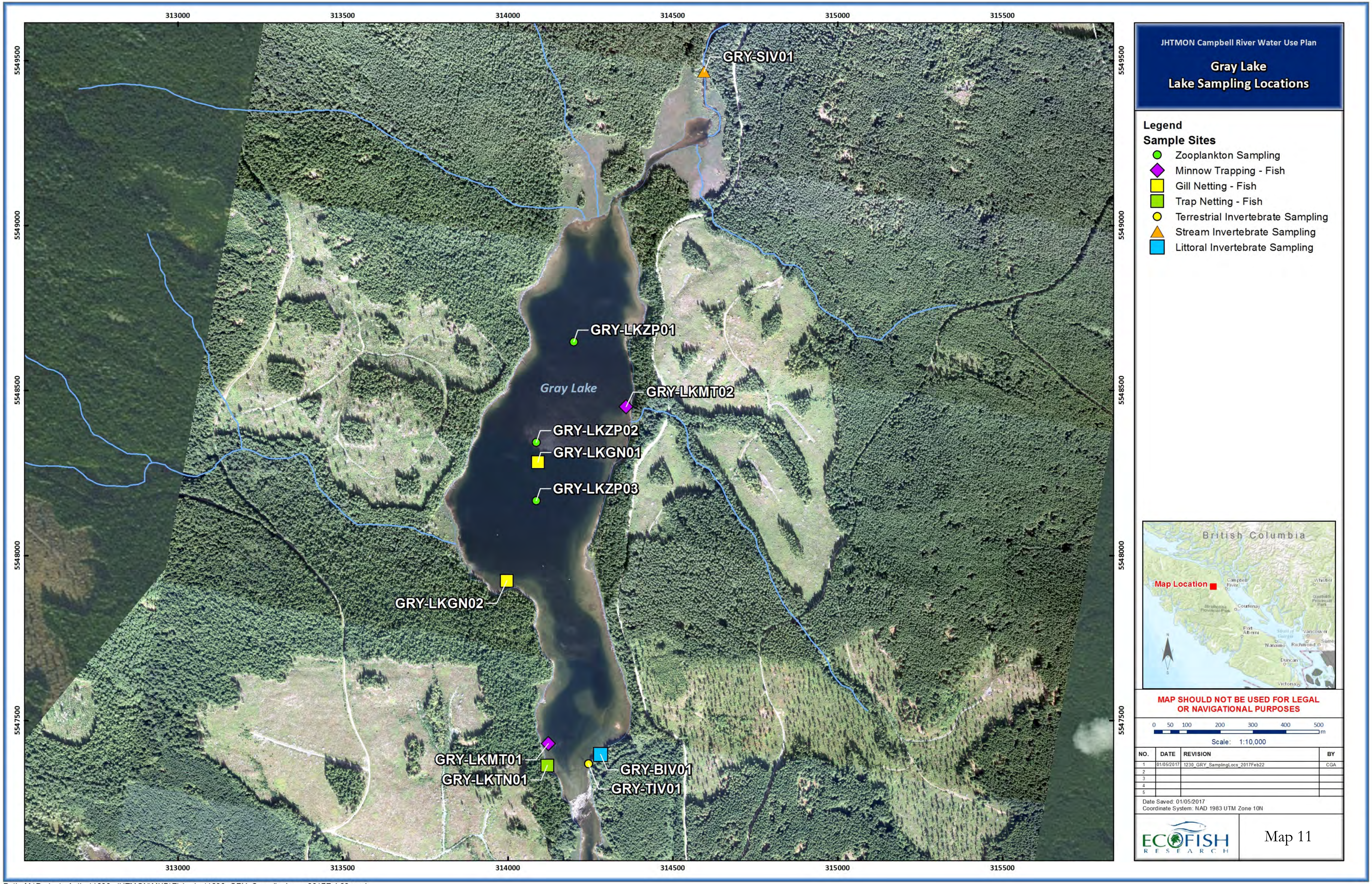
ECOFISH RESEARCH

Map 9

Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_JHT_SamplingLocs_2017Feb22.mxd



Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_BRE_SamplingLocs_2017Feb22.mxd



Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_GRY_SamplingLocs_2017Feb22.mxd

JHTMON Campbell River Water Use Plan

Whymper Lake Lake Sampling Locations

Legend

Sample Sites

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

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 50 100 m

Scale: 1:2,500

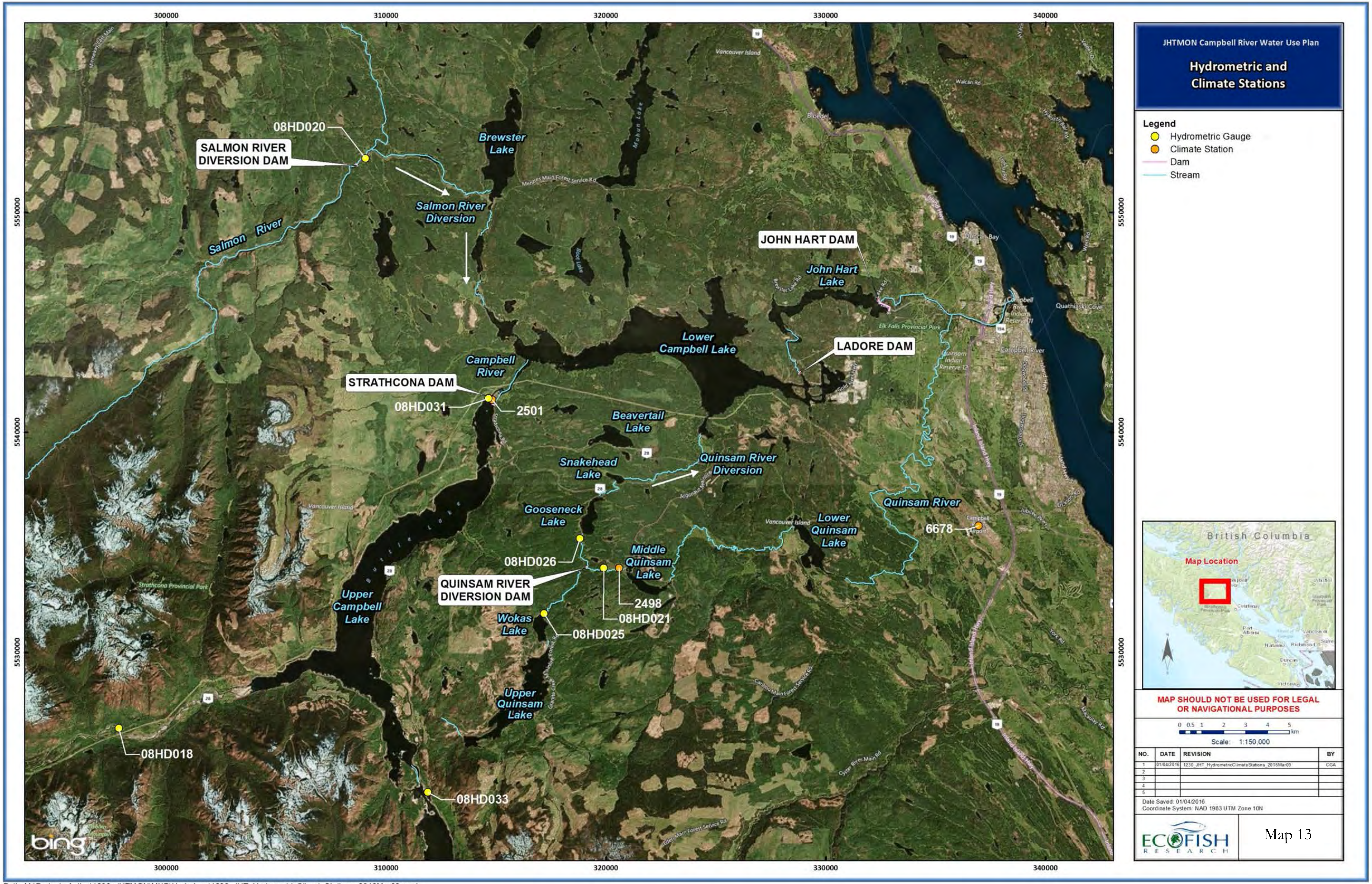
NO.	DATE	REVISION	BY
1	01/05/2017	1230_WHM_SamplingLocs_2017Feb22	CGA
2			
3			
4			
5			

Date Saved: 01/05/2017
Coordinate System: NAD 1983 UTM Zone 10N

ECOFISH RESEARCH

Map 12

Path: M:\Projects-Active\1230_JHTMON\MXD\Fisheries\1230_WHM_SamplingLocs_2017Feb22.mxd



Path: M:\Projects-Active\1230_JHTMON\MXD\Hydrology\1230_JHT_HydrometricClimateStations_2016Mar09.mxd

APPENDICES

Appendix A. Zooplankton abundance

LIST OF TABLES

Table 1. Zooplankton abundance, Year 1 (2014).

Waterbody	Sample[1]	Month	Zooplankton abundance (individuals/L)[2]							
			Bosminidae	Calanoida	Cyclopoida	Daphniidae	Leptodoridae	Nauplii	Onychopoda	Sididae
Upper Campbell Reservoir	UCR-LKZP01a	June	770	475	917	1381	0	781	68	0
		July	1403	249	1064	1743	0	656	0	0
		Aug	-	-	-	-	-	-	-	-
	UCR-LKZP02a	June	1132	102	509	543	0	804	57	0
		July	804	204	871	849	0	1109	11	0
		Aug	407	102	1064	871	0	555	34	0
	UCR-LKZP03a	June	1437	260	747	758	0	532	23	0
		July	962	396	860	973	0	939	113	0
		Aug	373	204	1154	1098	0	679	34	0
Abundance (all months, 'a' replicates only; %)			**23%**	**6%**	**23%**	**26%**	**0%**	**19%**	**1%**	**0%**
Gooseneck Lake	GOO-LKZP01a	June	799	719	1837	4287	0	1997	0	0
		July	488	1045	3413	5955	0	3656	70	0
		Aug	453	181	951	7198	0	3214	0	0
	GOO-LKZP01b	June	160	426	999	1584	93	0	13	240
		July	-	-	-	-	-	-	-	-
		Aug	724	679	679	6383	136	45	136	1901
	GOO-LKZP01c	June	506	160	1278	2889	40	0	0	146
		July	-	-	-	-	-	-	-	-
		Aug	-	-	-	-	-	-	-	-
Abundance (all months, 'a' replicates only; %)			**5%**	**5%**	**17%**	**48%**	**0%**	**24%**	**0%**	**0%**
Middle Quinsam Lake	QUN-LKZP01a	June	1099	323	3719	4721	0	2231	49	0
		July	1268	3984	5704	13310	91	1992	272	0
		Aug	521	1290	1358	4255	0	0	0	0
	QUN-LKZP01b	June	679	1358	2393	6209	65	0	32	1681
		July	-	-	-	-	-	-	-	-
		Aug	294	1086	1019	4618	136	45	0	181
	QUN-LKZP01c	June	3104	388	5303	11188	259	0	0	776
		July	-	-	-	-	-	-	-	-
		Aug	-	-	-	-	-	-	-	-
Abundance (all months, 'a' replicates only; %)			**6%**	**12%**	**23%**	**48%**	**<1%**	**9%**	**1%**	**0%**
Total abundance (all '1a' samples; %)			**7%**	**9%**	**21%**	**47%**	**<1%**	**16%**	**<1%**	**<1%**

1. Lower case letters denote replicates ('a', 'b' or 'c')
2. Some samples were not enumerated (denoted by '-')

Table 2. Zooplankton abundance, Year 2 (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; %)			**<1%**	**5%**	**14%**	**33%**	**38%**	**<1%**	**2%**	**<1%**	**<1%**	**2%**	**<1%**	**<1%**	**6%**	**<1%**

ECOFISH RESEARCH

Table 3. Zooplankton abundance, Year 3.

Waterbody	Site	Month	Zooplankton abundance (individuals/L)										
			Arachnida	Bosminidae	Calanoida	Chironomidae	Cyclopoida	Daphniidae	Harpacticoid	Leptodoridae	Onychopoda	Sididae	Other
Upper Campbell Reservoir	UCR-LKZP01	June	11	390	153	0	492	1154	0	0	57	45	0
		Aug	28	317	198	6	458	724	23	0	45	57	0
		Sept	23	475	747	0	973	1358	0	0	68	91	0
	UCR-LKZP02	June	6	900	108	0	470	871	0	0	28	79	6
		Aug	23	538	226	0	198	990	0	0	62	57	0
		Sept	11	419	498	11	589	1075	0	0	328	23	0
	UCR-LKZP03	June	6	515	68	0	453	283	0	0	0	11	0
		Aug	45	871	283	0	1432	1262	0	6	108	79	0
		Sept	23	498	475	0	838	1358	0	0	226	45	0
Abundance (all months; %)			**1%**	**20%**	**11%**	**0%**	**24%**	**37%**	**0%**	**0%**	**4%**	**2%**	**0%**
John Hart Reservoir	JHT-LKZP01	June	45	875	724	0	241	2701	0	0	75	377	0
		July	15	483	257	0	490	717	0	0	8	23	0
		Aug	121	1600	1841	0	996	1720	0	0	30	181	0
	JHT-LKZP02	June	15	649	121	0	302	996	0	0	91	15	0
		July	0	158	15	0	551	279	75	0	38	23	0
		Aug	30	2596	996	0	875	996	0	0	0	211	0
	JHT-LKZP03	June	0	370	53	0	211	528	0	0	91	0	0
		July	0	324	257	8	453	362	128	0	15	30	0
		Aug	30	2022	423	0	1056	966	0	0	0	211	0
Abundance (all months; %)			**1%**	**30%**	**16%**	**0%**	**17%**	**31%**	**1%**	**0%**	**1%**	**4%**	**0%**
Brewster Lake	BRE-LKZP01	June	0	15	619	0	2943	1947	679	45	0	75	0
		July	0	11	645	0	622	1652	181	0	68	57	11
		Aug	0	45	509	0	396	566	0	23	124	249	0
	BRE-LKZP02	June	0	2852	604	0	1856	166	332	0	30	75	0
		July	0	0	951	0	487	1200	79	0	57	57	0
		Aug	11	34	68	0	238	328	0	124	0	34	0
	BRE-LKZP03	June	0	8	460	0	1418	1984	189	8	8	15	15
		July	0	0	622	0	255	679	45	6	23	11	0
		Aug	0	68	147	0	136	679	0	0	11	170	0
Abundance (all months; %)			**0%**	**11%**	**17%**	**0%**	**30%**	**33%**	**5%**	**1%**	**1%**	**3%**	**0%**
Whymper Lake	WHM-LKZP01	June	0	141	5150	0	340	6791	0	0	28	141	0
		July	0	340	8318	0	170	1867	0	0	0	57	0
		Sept	0	905	4414	113	679	12563	0	566	0	453	0
	WHM-LKZP02	June	0	57	311	0	113	6027	0	0	57	1047	0
		July	0	1415	2943	0	792	5942	57	0	0	736	0
		Sept	0	1471	3622	0	792	5659	0	0	0	2943	0
	WHM-LKZP03	June	0	255	3339	28	170	4725	0	0	0	651	0
		July	0	396	4244	0	396	9903	0	0	57	849	0
		Sept	0	2943	4753	0	453	8149	113	0	0	3169	0
Abundance (all months; %)			**0%**	**7%**	**31%**	**0%**	**3%**	**51%**	**0%**	**0%**	**0%**	**8%**	**0%**
Gray Lake	GRY-LKZP01	June	0	728	73	0	760	170	32	0	4	238	0
		July	0	2056	1075	0	1415	1075	151	0	19	57	0
		Sept	0	1339	1094	0	1226	2433	0	19	0	773	0
	GRY-LKZP02	June	0	1043	243	0	2740	517	0	0	16	420	8
		July	0	660	1584	0	1320	905	94	19	0	94	0
		Sept	0	2829	1169	0	1773	3395	0	38	0	1320	0
	GRY-LKZP03	June	0	792	1593	0	2320	315	0	8	16	550	0
		July	0	887	1886	0	1094	811	75	0	19	75	0
		Sept	0	2339	1207	0	1396	4904	38	0	0	1509	0
Abundance (all months; %)			**0%**	**22%**	**17%**	**0%**	**25%**	**26%**	**1%**	**0%**	**0%**	**9%**	**0%**
Total abundance (all samples; %)			**<1%**	**14%**	**23%**	**<1%**	**14%**	**40%**	**1%**	**<1%**	**1%**	**7%**	**<1%**