Bridge River Project Water Use Plan

Seton River Habitat and Fish Monitoring

Implementation Year 6

Reference: BRGMON-9

Study Period: January 2018 to December 2018

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Executive Summary

The overall objective of the BRGMON-9 program is to monitor responses of fish habitat and fish populations in the Seton River to Seton Dam flow operations. Currently in Year 6 of 10, this monitoring program was developed to address a series of management questions that aim to: 1) better understand the basic biological characteristics of the rearing and spawning fish populations in Seton River, 2) determine how the Seton River hydrograph influences the hydraulic condition of juvenile fish rearing habitats and fish populations, 3) evaluate potential risks of salmon and steelhead redds dewatering due to changes in the Seton River hydrograph, 4) assess how the Seton River hydrograph influences availability of gravel suitable for spawning, and 5) estimate the effects of discharge from the Seton River Generating Station on fish habitat in the Fraser River. Fish stranding in the upper Fraser River following Seton Generating Station (SGS) shutdowns was monitored in previous years, however this aspect of the monitor was discontinued due to low stranding risk. Lower Fraser River stranding was added to BRGMON-9 as part of a TOR Addendum (BC Hydro 2018) and is reported separately. A preliminary synthesis of key data from the monitoring period to date (2014-2018) is provided.

Seton Dam represents a hydraulic bottleneck in the Bridge-Seton Hydroelectric complex in that decisions made upstream have an impact on Seton River Flows. Seton Dam discharge, as per WUP targets, generally mimics natural seasonal flows and falls between 12 m^3/s and 36 m^3/s. In 2014, operations at Seton Dam followed WUP target flows for Seton River with only minor periods requiring releases above target levels (max discharge 68 m^3/s). Beginning in 2015 changes to the Seton River hydrograph substantially increased in-river flows bringing Seton River discharge to 100 m^3/s. In response to dam safety risks, BC Hydro modified operations at La Joie Dam in 2016, decreasing the storage capacity of Downton Reservoir. Periods of modified operations have continued each subsequent spring and reached a peak discharge of 145 m^3/s in 2017. Higher discharges as a result of modified operations impacted monitoring activities such as monthly bio-sampling surveys.

Monthly juvenile surveys conducted annually since 2014 from April to October observed 14 species of fish including seven salmonids, of which age could be determined for Rainbow Trout (Age 0-3), Coho, and Chinook Salmon (both Age 0-2). Preliminary PIT telemetry data from 2014-2018 indicates that Rainbow Trout move between the spawning channels and the mainstem Seton River, suggesting a lack of distinct populations within these habitats. Juvenile salmonids also appear to use spawning channels

during specific times of the year (i.e., for overwintering). Notably, juvenile Chinook abundance has increased since 2015, yet few adults have been observed. DNA analysis revealed that 52% of juvenile Chinook captured in the Seton River from 2016-2018 are of different stock origin, suggesting they rear in the Seton River.

A two-level sampling strategy using electrofishing (September) and snorkel surveys (March) was used to estimate juvenile Rainbow Trout abundance in Seton River annually since 2014. To date, a relationship between standing crop and the Seton Dam hydrograph has not been identified. Variable flow conditions throughout the four years of modified operations make it difficult to determine if the reduced abundance of Rainbow Trout (11,157 in 2014 to 2,487 in 2015) are truly the result of operational changes. Additionally, 2014 represents the only year where flows were within the WUP targets limiting comparative analyses. Further data of modified operations and WUP target flows would help identify if a relationship exists between flow condition and Rainbow Trout abundance. Coho and Chinook are also collected throughout these surveys but are not captured in high enough quantities to accurately estimate their abundance.

Assessing the basic biological characteristics of adult salmonid populations in the Seton River has been challenging. Enumeration data for Chinook and Coho Salmon and Rainbow and Steelhead Trout, the focus of this monitor, are limited due to low densities and poor visibility during visual surveys. However, observational and telemetry data do confirm that all three species spawn in the Seton River and associated spawning channels. Steelhead spawning has not been visually confirmed for mainstem habitat due to poor visibility.

Monitoring activities were not affected in 2015 but in 2016 a monitoring strategy was developed for the period of modified operations when the WUP target flows are exceeded. Focus shifted to surveying side-channel habitats created at discharges >60 m^3/s and bio-sampling increased in the spawning channels rather than the Seton River mainstem, where many survey sites were made inaccessible. As only one year (2014) of pre-modified operations data were available, bio-sampling data collected through the monthly juvenile surveys was used in statistical models to assess whether changes in growth and condition across years, could be used as an in-season measure of flow effects. Results to-date have been inconsistent, and at this time no conclusions can be drawn regarding the effects of flow to individual fish condition and growth due to the single year of WUP target hydrograph data available for comparable analyses. Continued bio-sampling will add to the long-term data set useful in assessing how modified operations influence fish populations when Seton Dam returns to the WUP target hydrograph. Until that time, growth will not be used as a measure of in-season discharge effects.

Habitat suitability surveys at standing stock sites were completed in 2018 to assess changes in Weighted Useable Area (WUA) and substrate as a result of modified operations. These surveys are directly compared to habitat suitability surveys done in 2014 (WUP) at the same discharge (12 m^3/s). A large proportion (10/25) of transects had larger substrate size in 2018 over 2014 indicating mobilization of smaller substrate at high discharge may be occurring. Mobilization of smaller substrate downstream is corroborated by riverbed topographic surveys completed in other years of this monitor (Buchanan et al., 2017). Changes to WUA for juveniles have been variable, with more suitable habitat being observed from Seton Dam to the Cayoosh confluence (Reach 1) and less from the Cayoosh confluence to the LSC (Reach 2) relative to 2014. The amount of habitat suitable for spawning was also assessed for Coho and Chinook salmon at standing stock sites and known spawning locations; overall, both species experienced a reduction relative to 2014 (-22% for Coho and -18% for Chinook). However, when only known spawning locations were evaluated, there was a 2% (Coho) and 23% (Chinook) increase in suitable spawning habitat indicating that Coho and Chinook may be actively selecting these areas. Salmon and Steelhead redds in the mainstem are unlikely at risk of dewatering due to changes in the Seton River hydrograph because known spawning areas remain wetted throughout the year.

Modified operations of Seton Dam will continue to affect how various components of the monitor are carried out. We recommend several additions be made to the monitoring program including: added habitat suitability surveys between 60 m^3/s and 100 m^3/s for side-channel habitats wetted during modified operations (> 60 m^3/s), modifications to PIT antennas to increase detection efficiency to better detect if fish movements are linked to changes in flow, and marking Coho juveniles to determine if they show high site fidelity therefore allowing the spawning channels to be used as control for flow for this species. The monitoring approach will continue to be adaptive, with the goal of answering established management questions.

Status of BRGMON-9 objectives, management questions and hypotheses after Year 6 (2018)

Management Questions and Hypotheses	Status
1: What are the basic biological characteristics of the rearing and spawning populations in Seton River in terms of relative abundance, distribution, and life history?	- Monthly bio-sampling surveys have been conducted since 2014. Monitoring has identified 14 species of fish, including seven salmonids. Coho and Chinook Salmon juveniles are present, but samples are dominated by Rainbow Trout. Individuals captured through this program are sampled for length, weight, and age (through scale samples). DNA samples are also collected from Chinook juveniles to better understand their life history. All Rainbow Trout >75 mm are implanted with a 12 mm PIT tag for long term monitoring. - Observing adults in Seton River has been difficult. Modified operations at Seton Dam during Steelhead migration limits the effectiveness of visual surveys. Adult Coho Salmon are predominantly observed using the constructed spawning channels. Only 3 adult Chinook have ever been observed in Seton River. DNA analyses show that many Chinook juveniles captured in Seton River originate from other Fraser River stocks (e.g., up to 72% in 2016), suggesting that the Seton River provides important rearing habitat. - Juvenile standing crop surveys have been completed since 2014. To date a relationship between standing crop and discharge for Rainbow Trout has not been identified. Coho and Chinook are not captured in high enough densities to calculate standing crop.
2: How does the proposed Seton hydrograph influence the hydraulic condition of juvenile fish rearing habitats downstream of Seton Dam? H_1: The amount of hydraulic habitat that can be inhabited by juvenile fish is independent of discharge from Seton Dam *H_{1A}: Juvenile standing crop biomass per unit area is inversely related to flow velocity.* *H_{1B}: Juvenile standing crop biomass per unit area is independent of flow depth.* *H_{1C}: Juvenile standing crop biomass per unit area is independent of both flow velocity and depth.*	- Data collected to date suggests that the amount of hydraulic habitat available to juvenile fish varies with Seton Dam discharge (Reject H_1). Habitat suitability has been assessed at Seton Dam Discharge of 12 – 145 m^3/s. As flows increase, the amount of habitat available to juvenile salmonids decreases. Above 60 m^3/s, side-channels begin to become wetted which does buffer some of the juvenile habitat loss occurring in the mainstem. Habitat suitability surveys are expected to continue in future years at discharges between 60 and 100 m^3/s to determine at what discharge the amount of hydraulic habitat is maximized for juvenile fish in the side-channels of Seton River. - Sub-hypotheses have not been explicitly tested. A robust data set exists for rainbow trout abundance, but no other species given data limitations. Rainbow trout abundance could be qualitatively compared to discharge conditions in a given year, but no analysis currently differentiates between flow velocity and flow depth. Juvenile Rainbow Trout abundance was highest in 2014 under the WUP hydrograph, with 2015 showing a large reduction in abundance with discharge reaching 99.7 m3/s June 25. Small recoveries have been made every subsequent year indicating there may be a link between juvenile abundance and flows, or at the very least timing of high flow releases from Seton Dam. Analyses are limited by only one year of baseline data (2014). - Condition of fish is examined through monthly bio-sampling surveys. While no trends in the data are visible to date due to variability of flow within the 'Modified Operations' from year to year, the data collected builds upon a growing long-term biological data set. This dataset will be invaluable when Seton River flows return to the WUP target hydrograph and make it possible to examine results from WUP target discharges against discharges during 'Modified

	Operations'. As such, current bio-sampling surveys should be considered baseline data collection.
3: What is the potential risk for salmon and Steelhead redds dewatering due to changes in flow between spawning and incubation periods imposed by the Seton hydrograph? H_2: The selected Seton River hydrograph does not result in dewatering of salmon or Steelhead redds	- To date, no redd dewatering events have been observed as the primary spawning area for Pink, Coho, Sockeye and Steelhead remains wetted throughout the year. However, with the modified operations discharges at Seton Dam, several side-channel habitats have been wetted during Steelhead migration (April – June) and subsequently dry when flows return to the WUP target hydrograph in July. As such, H_2 cannot be rejected and further years of surveys should be completed to determine the risk for redd stranding in these side-channel habitats.
4: How will the Seton hydrograph influence the short-term availability of gravel suitable for use by anadromous and resident species for spawning and egg incubation? H_3: The selected Seton River hydrograph does not result in mobilization of gravel or net loss of gravel from the system.	- Riverbed elevation surveys (2013, 2015, 2016, 2017) of a key spawning area immediately downstream of Seton Dam have shown minor changes in elevation and substrate composition. Some sections of the area have eroded while other sections have shown deposition; there has also been some movement of smaller substrate (gravel and small cobble) downstream. The 2017 survey shows increases in elevation or deposition of gravel and possible recruitment of substrate. - The data supports rejecting the first part of H_3, that the Seton River hydrograph does not result in mobilization of gravel, but the deposition results show that it is still undetermined if there is a net loss of gravel. Riverbed elevation surveys are due to be repeated in 2019 if flows greater than the WUP targets occur that year. - Substrate surveys have been added throughout Seton River to determine if gravel is moving downstream as a result of Seton Dam's modified operations.
5: Does discharge from Seton Generating Station impact fish habitat in the Fraser River above and beyond natural variation in Fraser River discharge?	- Fish stranding in the Fraser River as a result of shutdowns from Seton Generating Station over three shutdowns (2015-2017) was found to be 5 individuals. Actual area dewatered during each survey varied based on Fraser River discharge at the time of the shutdown. Due to the low stranding risk in the Fraser River near the Seton Generating Station, a TOR Addendum (BC Hydro 2018) was prepared to address remaining uncertainties in the effects of winter Seton Generating Station shutdowns on adult redd stranding risk in the lower Fraser River. These results of this assessment will be reported separately.

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List of Abbreviations

AIC – Akike Information Criterion

ALK – Age-Length Key

CPUE – Catch Per Unit Effort

BRG-CC – Bridge-Seton Consultative Committee

EF – Electrofishing

HSI – Habitat Suitability Index

K_f – Fulton's Condition Factor

LSC – Lower Spawning Channel

M-R – mark-recapture

MANOVA – multivariate analysis of variance

MASL – meters above sea level

MOE – Ministry of Environment

MS – mainstem bio-sampling site

OCH – off-channel or side-channel habitat

PIT – passive integrated transponder

SGS – Seton Generating Station

TOR – Terms of Reference

USC – Upper Spawning Channel

WSC – Water Survey of Canada

WUA – weighted useable area

WUP – Water Use Plan

1.0 INTRODUCTION

1.1 Background

The Seton River is a four-km river bound by the Fraser River to the east and Seton Dam to the west (Figure 1-1). Seton Dam was completed in 1956 and was the final dam built as part of the Bridge River hydroelectric development structures. Since construction, Seton Dam has regulated Seton River flows to control the amount of water received by Seton Generating Station (SGS) and manage water levels in Seton Lake.

Adopted in 2011, the Bridge River Water Use Plan (WUP) was developed as part of a consultative process that began in 1999 and aimed to develop an acceptable instream flow regime for the Seton River which balanced environmental, social and economic concerns for competing water uses and recognized the interdependence of projects within the Bridge River system (BC Hydro, 2012). A critical environmental concern identified through this process was the need for a flow regime that considered the high ecological value, in terms of fish and wildlife, that the Seton River provides to local communities. The Bridge-Seton Consultative Committee (BRG CC) therefore set environmental objectives for Seton River that are measured in terms of abundance and diversity of fish populations within the river (BC Hydro, 2012). As a result of the WUP, BRGMON-9 was initiated in 2012 as a ten-year monitoring program, with data collection beginning in 2013.

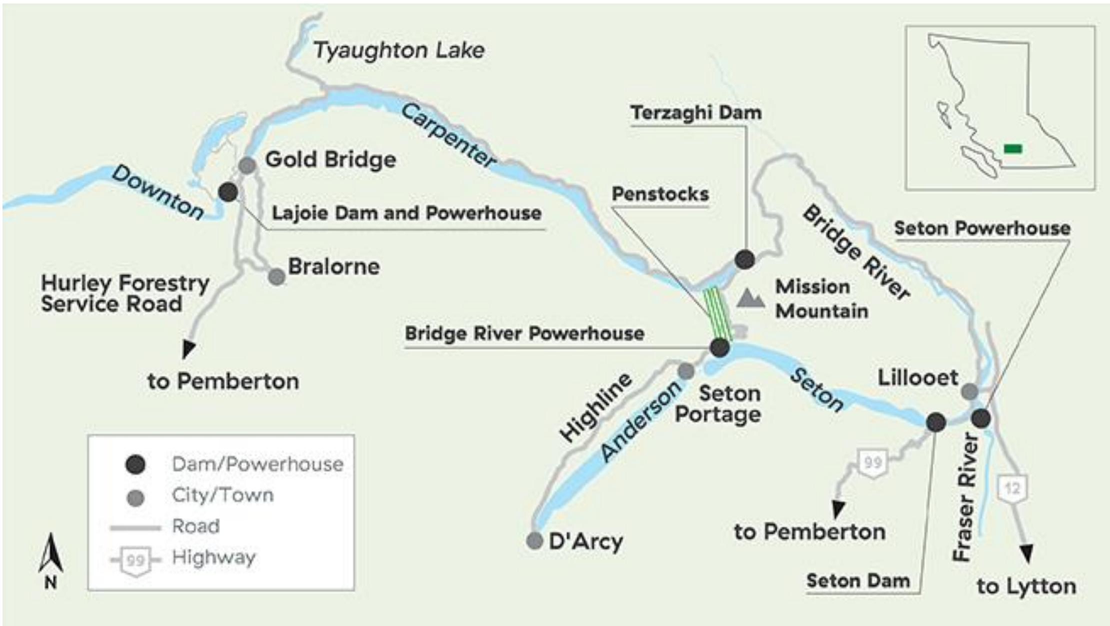


Figure 1-1. A map of the Bridge-Seton hydroelectric structures operated by BC Hydro (BC Hydro, 2016). This report is focused on the effects of flows from the Seton Dam, the downstream-most structure in the system.

1.2 Changes to the Seton Dam Hydrograph During Monitoring

The Seton Dam and generating station are a 'hydraulic bottleneck' in the Bridge-Seton system whereby management changes at the upstream Carpenter and Downton reservoirs and Bridge Powerhouse can have considerable impact on Seton River flows. This hydraulic characteristic has two practical consequences. First, there are periodic discharges above the WUP target hydrograph in the Seton River that are necessitated by water management concerns upstream. For example, in high inflow years, water in the Bridge-Seton system is managed to prevent excessive flow releases from Terzaghi Dam, limiting environmental impacts to the lower Bridge River. Because the quantity of water that can be 'generated' out of the system is limited by the Seton power canal, water releases from Seton Dam that are greater than the target hydrograph for the Seton River may be required. Second, natural variability in flow patterns to the system on seasonal and inter-annual basis can result in highly variable annual hydrographs in Seton River. Maintaining the WUP target hydrograph at Seton Dam is a trade-off between minimizing impacts of instream flow regimes to fish and fish habitat in Seton River and the higher WUP priority of protecting the productive capacity of other upstream waterways (i.e., Lower Bridge River).

Seton River discharges beyond the WUP target hydrograph occurred during the monitoring period beginning in 2015. In response to dam safety risks, BC Hydro modified operations at La Joie Dam in 2016, decreasing the storage capacity of Downton Reservoir. Specifically, maximum water elevation in Downton Reservoir was decreased from 749 meters above sea level (MASL) to 734 MASL, significantly decreasing the storage capacity. Downton Reservoir was drawn down to the new operating level in 2016 and the lower reservoir elevation was maintained through to 2018. The result of the change in Downton Reservoir storage and WUP prioritization of flows in the Bridge-Seton system was Seton Dam flow releases in 2016-2018 that exceeded the WUP target hydrograph (Table 1-1). Reduced storage at Downton Reservoir is expected to continue indefinitely, creating a period of 'Modified Operations' in the Bridge-Seton system that will increase the likelihood that the WUP target hydrograph for Seton Dam will be exceeded.

Table 1-1. Flow statistics by condition (WUP or modified operations) for Seton Dam 2013-2018

Year	Flow Condition	Flow Statistics		
		Mean Annual Discharge (m^3/s)	Minimum Discharge (m^3/s)	Maximum Discharge (m^3/s)
2013	WUP - target	19	11	36
2014	WUP	24	10	68
2015	WUP*	23	11	100
2016	MOD	36	13	114
2017	MOD	36	11	145
2018	MOD	24	10	93

Though flows above the WUP target Seton River hydrograph were not initially expected in 2018, discharges did exceed Seton River WUP targets, reaching 93 m^3/s on May 18 and remaining high through most of the spring and summer as they did in 2016 and 2017, having implications for monitoring activities and introducing potentially confounding effects not considered in the Terms of Reference (TOR). However, these changes also presented an opportunity to perform monitoring at flows above the WUP target hydrograph to compare the effects of the WUP target hydrograph and higher discharge hydrographs during periods of modified operations on Seton River fish and fish habitat.

During periods of high discharges, water overflows the Seton River mainstem creating side-channels. Effects of high discharge to juvenile fish are hypothesized to be buffered because 1) these side-channels may provide favorable habitat for juvenile and sub-adult fish and 2) a possible "dynamic equilibrium" of suitable hydraulic conditions exists [i.e., for different flow levels there is a fixed volume of hydraulic habitat that conforms to tolerances or preferences of small fish, (BC Hydro, 2012)]. However, it is unknown whether this 'dynamic equilibrium'' hypothesis is valid during modified operations. For example, seasonal changes in flow regimes between the spawning period and the emergence of fry could lead to redd dewatering. The potential for dewatering, dependent on where fish deposit eggs and the interaction between channel geometry and observed flows, is largely unknown. High discharges under modified operations may also impact the quantity of suitable gravel for spawning because 1) it is assumed there is little (if any) gravel recruitment to the river channel below the dam and 2) high discharges may mobilize spawning gravel. The combination of redd dewatering and gravel mobilization may erode the quantity and effectiveness of spawning habitats in the river.

Changes to the study sites as a result of modified operations meant that some BRGMON-9 monitoring activities had to be modified following 2016. In 2016, efforts were focused on identifying side-channel habitats and a new monitoring strategy appropriate for periods of high discharges during modified operations was developed. In 2017 side-channel habitats were surveyed at various instream flows to quantify habitat characteristics and verify fish presence and use. In 2018, side-channel habitats were included in the monthly bio-sampling surveys whenever wetted. However, as higher discharges due to modified operations occurred in the spring and summer, BRGMON-9 monitoring occurring in the fall remained unchanged and results would be able to capture the effects of higher discharges.

1.3 Scope and Objectives

The main objective of the BRGMON-9 program, as outlined in the TOR, is to monitor the response of fish habitat and fish populations to Seton Dam operations. A second objective is to identify key physical and biological indicators for monitoring the effects of the implemented Seton River hydrograph.

The scope of BRGMON-9 in Year 6 (2018) was to:

1) Document the hydraulic condition in the Seton River;
2) Collect information on juvenile fish habitat use in the Seton river as it relates to the instream flow regime;

3) Monitor anadromous salmon spawning location to assess the potential impacts for redd dewatering;
4) Monitor changes in the quantity, quality, and location of suitable spawning gravel;
5) Complete an annual report that summarize the 2018 monitoring results and incorporate all BRGMON-9 results to date.

Though discharges in Seton River in 2018 were not initially forecasted to exceed the WUP target maximum of 60 m^3/s, the scope included modified operations monitoring.

1.4 Management Questions

The purpose of this monitoring program is to document how the implemented Seton Dam hydrograph (either WUP target or modified operations) influences habitat availability, to inform and refine future performance measures for fish resources in Seton River, and to provide information on the most suitable shape of the hydrograph for fish productivity.

This monitor addresses five management questions (MQ):

1. What are the basic biological characteristics of the rearing and spawning populations in Seton River in terms of relative abundance, distribution, and life history?
2. How does the proposed Seton hydrograph influence the hydraulic condition of juvenile fish rearing habitats in downstream of Seton Dam?
3. What is the potential risk for salmon and steelhead redds dewatering due to changes in flow between spawning and incubation periods imposed by the Seton hydrograph?
4. How will the Seton hydrograph influence the short term and long-term availability of gravel suitable for use by anadromous and resident species for spawning and egg incubation?
5. Does discharge from Seton Generating Station impact fish habitat in Fraser River above and beyond natural variation in Fraser River discharge?

Note that MQ5, while still under BRGMON-9, was monitored by another organization in 2018 and as such, no data is presented in this report.

1.5 Management Hypotheses

From the management questions above, three hypotheses and three sub-hypotheses were developed. H_1 and its associated sub-hypotheses are designed to answer MQ1 and MQ2 through the collection of standing-crop biomass and habitat data. H_2 directly addresses MQ3 by looking for spawning adult salmon and assessing Seton River for suitable spawning habitat. H_3 addresses MQ4 by assessing gravel movement in key spawning areas of Seton River. No hypotheses were created for MQ5.

Data from this program will be collected to explicitly test the following null hypotheses (and sub-hypotheses):

H_1: The amount of hydraulic habitat that can be inhabited by juvenile fish is independent of discharge from Seton Dam.

H_{1A}: Juvenile standing crop biomass per unit area is inversely related to flow velocity.

H_{1B}: Juvenile standing crop biomass per unit area is independent of flow depth.

H_{1C}: Juvenile standing crop biomass per unit area is independent of both flow velocity and depth.

H_2: The selected Seton River hydrograph does not result in dewatering of salmon or Steelhead redds.

H_3: The selected Seton River hydrograph does not result in mobilization of gravel or net loss of gravel from the system.

1.6 Monitoring Approach

The monitoring schedule is designed to collect coincident habitat, abundance and growth information on Seton River fish populations. These data can be used to better understand the effects of the Seton Dam hydrograph on critical habitat characteristics, and to relate how habitat conditions influence habitat use and relative productivity. Annual surveys are conducted to index population abundance and distribution in relation to habitat conditions, quantify redd dewatering, and determine changes in spawning gravel location and quantity. Standardized data management, analysis, and base mapping continues to be improved to better determine the linkage between fish use and abundance observations and habitat inventories.

2.0 METHODS

2.1 Study Sites

The Seton dam is an 18-meter high concrete dam that incorporates a fish ladder and a diversion canal. From the dam, a portion of the Seton River's flow is diverted via the Seton Canal to the Seton Powerhouse, which in turn drains into the Fraser River (Figure 2-1). Cayoosh Creek enters the Seton River approximately 1.3 km downstream of Seton Dam. High flows from Cayoosh Creek can further increase the flows in the Seton River downstream of the confluence. There are also two constructed restoration channels designed as habitat for spawning salmon: The Lower Spawning Channel (LSC) and the Upper Spawning Channel (USC).

Habitat encompassed by this monitoring program includes the Seton River, the spawning channels, and certain side-channel habitats created during modified operations (referred to as off-channel habitat or OCH).

Using data collected during site selection surveys (Ramos-Espinoza et al. 2014) and visually from Google Maps satellite imagery, the Seton River was divided into three distinct reaches, numbered in ascending order from Seton Dam to the Fraser River confluence (Figure 2-1). As defined in Johnston and Slaney (1996), a reach is a homogeneous section of river. Reach 1 extends from the dam to the confluence of Cayoosh Creek. Reach 2 extends from the Cayoosh Creek confluence to the intake of the Lower Spawning Channel. Reach 3 extends from the lower spawning channel intake to the Fraser River.

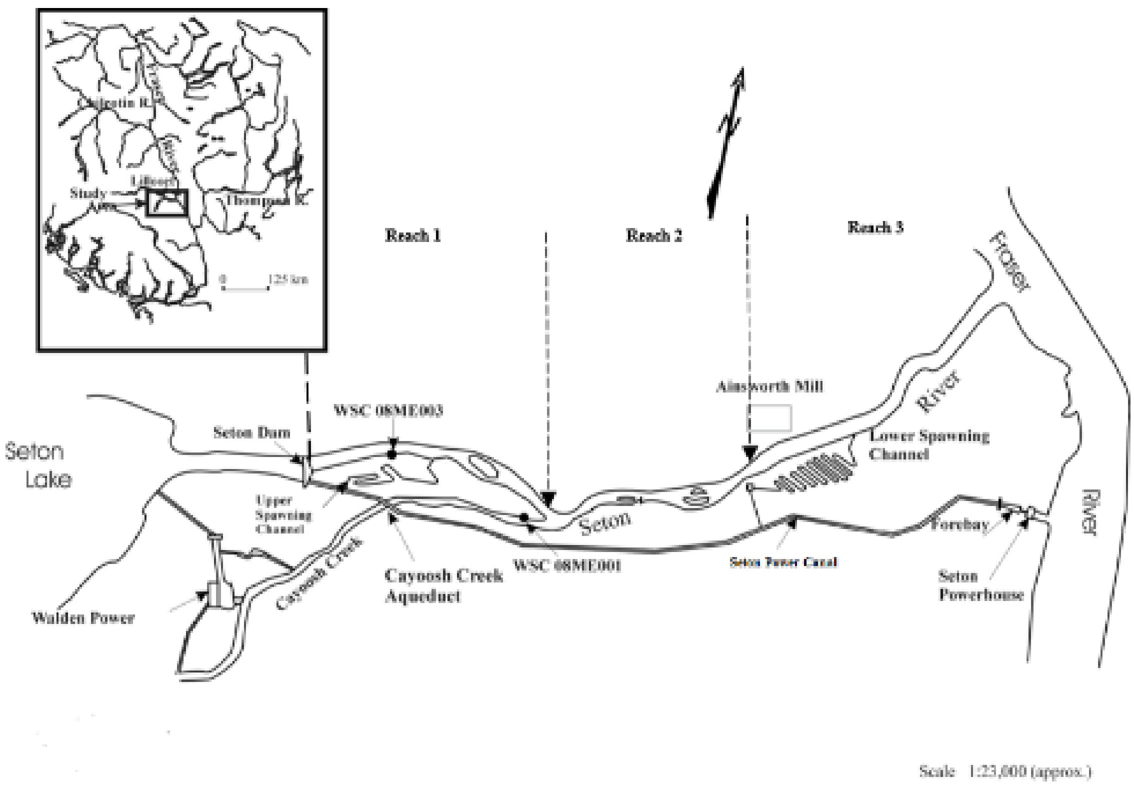


Figure 2-1. Detail of the Seton River study area bound by Seton Lake to the west and the Fraser River to the east. The study area was divided into three distinct reaches. Included on the map, but not included in the study, is Seton Power Canal and Cayoosh Creek.

2.1.1 Discharge

Discharge data was obtained from the Water Survey of Canada (WSC) gauges at Seton River near Lillooet (08ME003) and at Cayoosh Creek (08ME002). Due to the influence of Cayoosh Creek on the Seton River below the confluence, the discharge data for Reach 1 was taken from the Seton River gauge, located upstream of the confluence (Figure 2-1). For Reach 2 and 3 the discharge data from both gauges were combined to create a cumulative discharge. The two spawning channels also provide additional inflow throughout the year, but their combined contribution is constant all year round (~2 m^3/s) and thus was not considered.

2.1.2 Temperature

Water temperature is recorded hourly for the duration of the study using Onset Tidbit Water Temperature Data Loggers (Bourne, Massachusetts, USA). Loggers are attached to solid features either on shore or within the river (e.g. pilings) using aircraft cable and are weighted down using cinder blocks or a lead weight. Loggers are downloaded regularly during monthly bio-sampling surveys to reduce the risk of data loss in the event of high flows blowing out anchor lines.

Water temperatures are monitored in five locations: in the fishway of Seton Dam, the Seton River immediately downstream of the dam, downstream near the inflow to the LSC, and within the USC and LSC (Figure 2-2). Splitrock Environmental monitors temperature within the USC and LSC.

Figure 2-2. Location of temperature loggers in Seton River and the Spawning Channels

2.2 Habitat Suitability Assessments

2.2.1 Juvenile Rearing Habitat Suitability

Habitat Suitability Index (HSI) surveys were completed for mainstem Seton River in 2014 to assess the suitability for juvenile Rainbow Trout, Coho, and Chinook. To enable comparisons between higher discharges under modified operations and the WUP target hydrograph of 2014, transect surveys were repeated in September and October 2018 during base flow conditions (12 m^3/s) at the randomly selected stock assessment sites (Figure 2-3). Consistency in flow conditions during surveys allows changes to habitat suitability for these species as a result of higher Seton Dam discharges under modified operations to be detected. The same methodology used in 2014 and 2015 (as detailed in Ramos-Espinoza et al. 2015) was applied in 2018 for field surveys.

Weighted Useable Area (WUA) is calculated using a model developed by the Ministry of Environment [(MOE), Ptolmey et al. 1994], which is based on HSI scores. The MOE provided species and life stage specific HSI scores corresponding to depth, velocity and substrate preferences. The model estimates the amount of suitable habitat available for different species and life stages at any given discharge. Each parameter is weighted by an HSI score ranging from 0 (unsuitable) to 1 (optimal). The amount of suitable habitat is quantified as the product of HSI scores for each habitat value (i.e., water depth, velocity, and substrate) and the wetted width of the transect.

This methodology assumes that the habitat is relatively uniform along the length of each habitat unit, and that each point along the transect represents an area of streambed bound by the halfway point to the neighbouring vertical and the upstream and downstream boundaries (i.e., either the end of the hydrological habitat unit or the neighbouring transect, Mosley 1985).

WUA within each transect was summed to create a total WUA for each habitat unit. The total WUA for each habitat unit in 2018 was directly compared to the WUA for that same habitat unit assessed in 2014 to determine whether a change had occurred for each species and life stage. River-wide trends are examined but it should be noted that the results only represent a random subsample of the total habitat available.

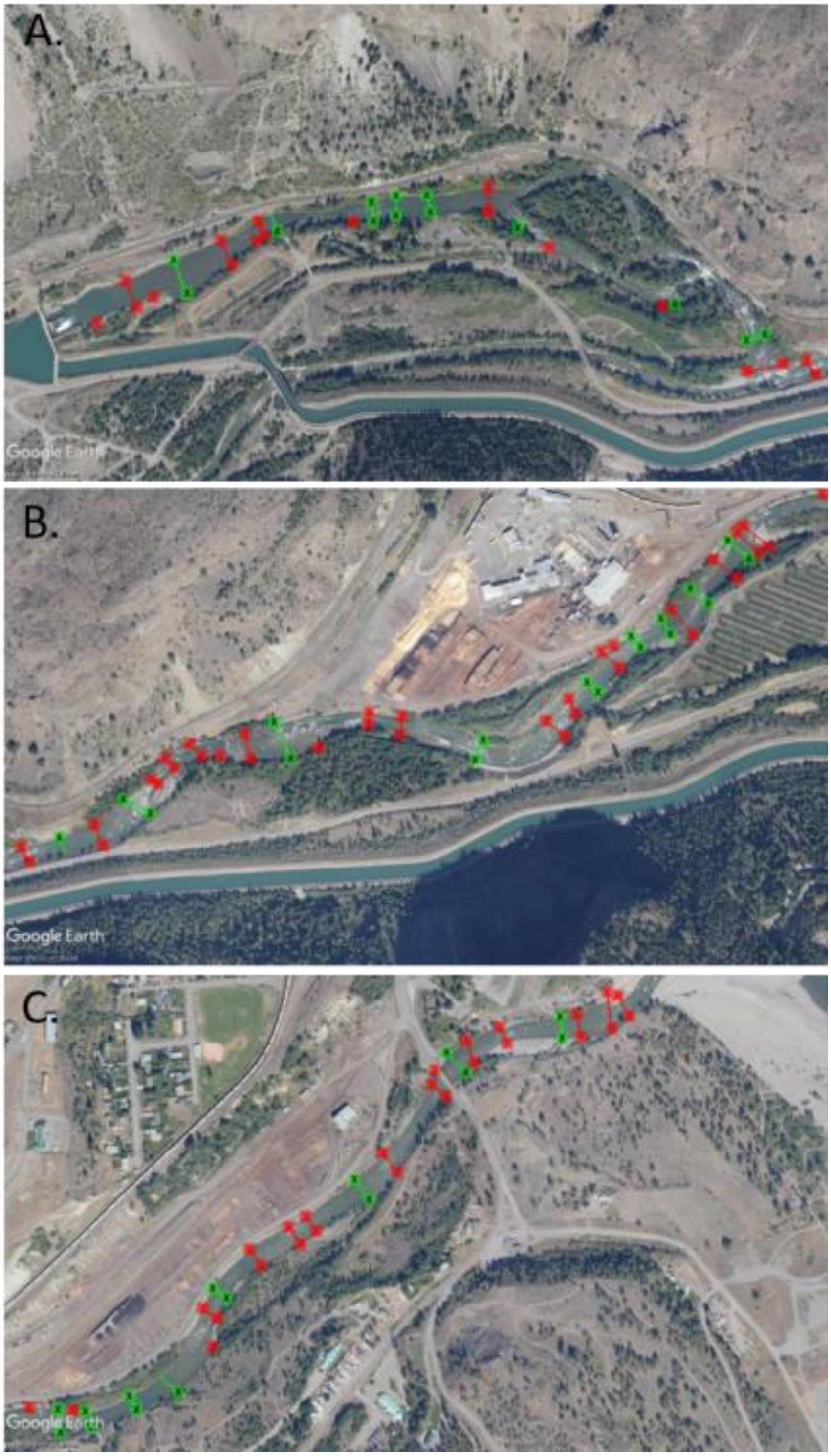


Figure 2-3. Location of Weighted Useable Area transects in Seton River. Red lines indicate the locations of transects done in the mainstem of the river 2014-2015 but not re-surveyed in 2018. Green lines indicate the locations of transects surveyed in 2014 and 2018.

2.2.2 Spawning Habitat Suitability

Using the same methodology described above (see Section 2.2.1), the suitability of Seton River for spawning Coho and Chinook Salmon was compared between 2014 and 2018 as both surveys are present in the river during surveys. WUA was calculated using spawning suitability curves at randomly selected standing stock sites and in known spawning locations (Figure 2-3) and in known spawning locations in Reach 1 and 3 where salmon have been observed spawning in the past (Figure 2-4). Estimates of useable spawning area should be considered minimum estimates as the thalwag of Seton River is frequently too deep or too fast for surveyors to safely assess habitat suitability but could be used by spawning salmon.

Figure 2-4. Location of Weighted Useable Area transects in Seton River at the two key spawning locations in Reach 1 (A) and Reach 3 (B) surveyed in 2014 and 2018.

2.2.3 Substrate Changes

A detailed substrate survey was completed in March of 2014 at base low flow (12 m^3/s), to identify the dominant substrate type at each habitat transect (Figure 2-3 and Figure 2-4) . Substrate type was classified using the Wentworth scale (Wentworth, 1922), which splits substrate into 7 categories (fines, sand, small gravel, large gravel, small cobble, large cobble, boulder and bedrock). Within a transect, each substrate type was visually estimated to the nearest 5% within a 1 m^2 section of riverbed along the entire length of the transect line or to a point of safe access. This survey was repeated at the selected sites in 2018 (Figure 2-3) to determine if changes in dominate and sub-dominate substrate classes within each habitat unit have occurred as a result of higher discharges under modified operations (Figure 2-3). Changes in substrate were classified as larger (2018 > 2014), smaller (2018 < 2014), and similar. The similar classification includes transects where no substrate change was observed as well as transects that the dominant and sub-dominant substrate size were reversed in 2018 relative to 2014.

2.3 Bio-sampling of Juveniles

From April through October we conducted monthly open-site electrofishing (Smith-Root LR-24 backpack electrofisher) in the spawning channels and the Seton River between the Seton Dam and the confluence of the Seton and Fraser Rivers. Sampling crews of three experienced technicians performed single-pass electrofishing at established sites (~50 m in length; Figure 2-5). Technicians moved upstream, with one operating the electrofisher and two dip-netting fish. Fork length and weight were measured for all captured fish. To determine age, scales were collected from the area above the lateral line and immediately below the dorsal fin and stored in labelled coin envelopes. During each sampling period, 30 fish of each species and age-class within each reach were sampled if numbers allowed.

All captured Rainbow Trout, Bull Trout, and Mountain Whitefish >75 mm in length were scanned for passive integrated transponder (PIT) tags, and untagged fish were implanted with a 12 mm PIT tag (Oregon, RFID, Portland, Oregon USA). Tags were inserted into the body cavity using a 12-gauge needle. Fish <150 mm were tagged in the ventral stomach cavity and fish >150 mm were tagged in the dorsal musculature. Recaptured fish were re-measured to evaluate growth between capture events.

We initially planned annual sampling in 6 of the 13 identified sites in the mainstem (MS1 to MS13, in addition to random sampling in the spawning channels, but high discharges due to modified operations in 2016-2018 prevented sampling at many pre-established sites. We sampled the pre-established sites when flows permitted, but also added sampling sites in side-channel habitat during periods of modified operations when discharges were higher and side-channels were activated (OCH1 to OCH9; Ramos-Espinoza et al. 2016). Table 2-1 provides a summary of the number of sites sampled in each year from 2014 to 2018.

In July 2018, a pilot program was initiated to angle for older/larger resident species which were under-represented in mainstem estimates. In conjunction with the monthly electrofishing surveys, two technicians angled for Rainbow Trout for four hours, moving every hour if no fish were caught. Catch Per Unit Effort (CPUE, fish captured per hour of sampling) was calculated using the equation:

$$CPUE = \frac{Catch}{Time(hours)} \qquad \text{Eq 2}$$

All Rainbow Trout, Bull Trout and Whitefish captured through angling were measured, weighed, and applied a 12 mm PIT tag. A scale sample was also taken to determine age. Age data from fish sampled through angling is included with the electrofishing samples.

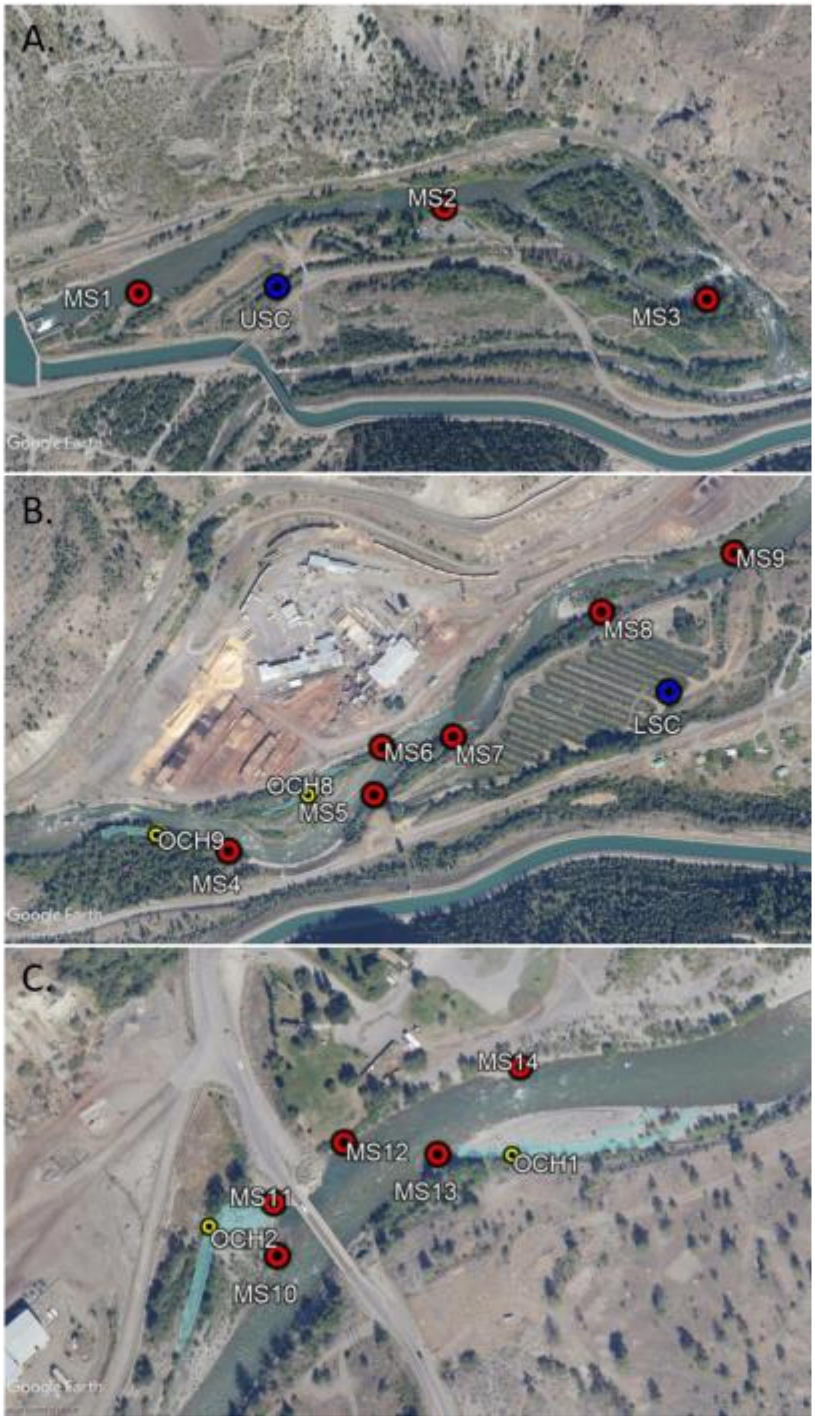
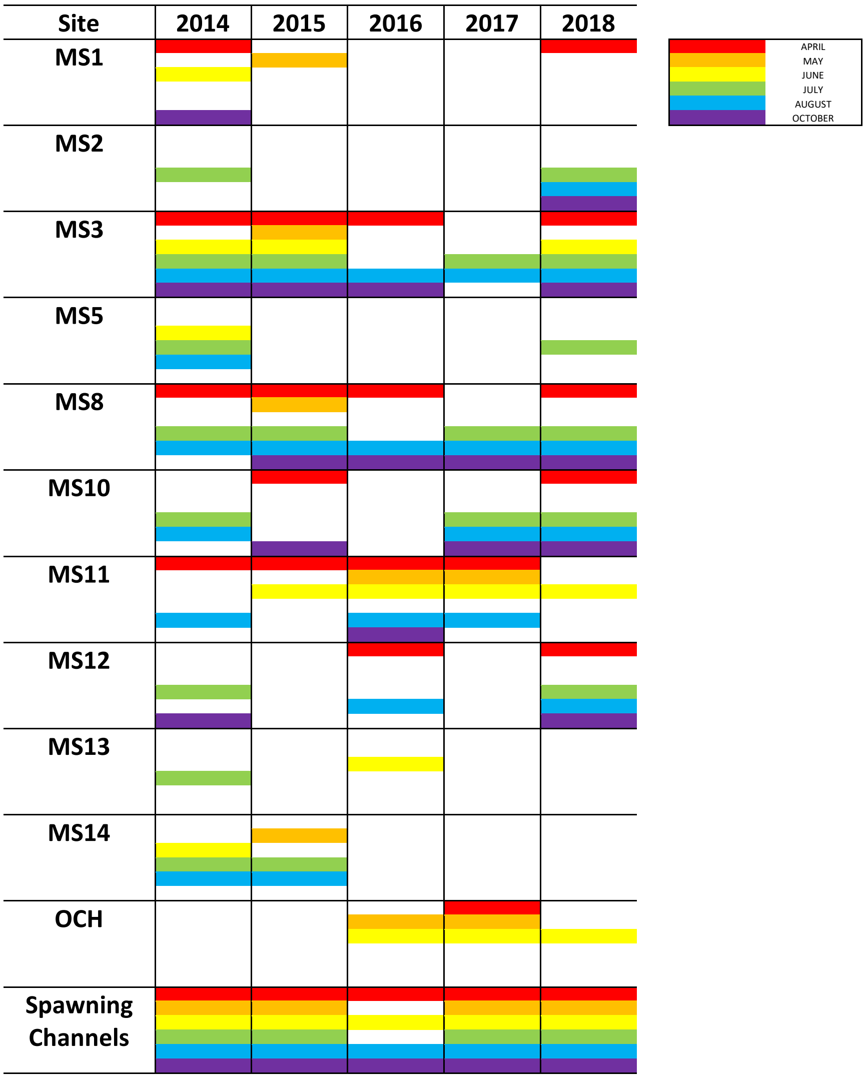


Figure 2-5. Location of juvenile growth sampling sites in the mainstem Seton River (MS; red circles), Upper (USC) and Lower (LSC) spawning channels (blue circles; sites sampled randomly within the channels), and side-channels (OCH; yellow circles) in 2014-2018 in A) Reach 1, B) Reach 2, C) Reach 3. For reference, Seton Dam can be seen on the far-left side of Panel A, and the Seton – Fraser confluence can be seen on the far-right of Panel C. OCH sites were only surveyed in 2017 and 2018.

Table 2-1. Growth sampling sites in the mainstem Seton River (MS) and the spawning channels (USC and LSC combined) and the years and months in which they were sampled from 2014 – 2018. Side-channel habitats (OCH) are only accessed when mainstem flows exceed 60 m^3/s.



2.3.1 Ageing Analysis

Ageing analyses add to our understanding of the basic biological characteristics of fish in the Seton River. Scale samples were stratified by fish length (25-59 mm, 60-124 mm, 125-170 mm, > 170 mm). A maximum of thirty scales per category per month were selected for ageing. Scales were mounted directly onto glass slides, digitally photographed, and each scale was read under magnification by two independent technicians to determine age (Zymonas and McMahon 2009).

Age-length keys (ALKs) were developed for Rainbow Trout, Coho Salmon, and Chinook Salmon. An ALK is a population-specific probability matrix that determines the probability that a fish from a length class is a given age class, and vice versa (Guy and Brown 2007; Ogle 2016). Probabilities are then used to determine proportions of fish from each length class assigned to each age class, from which age can be estimated for unaged fish in a population (Isermann and Knight 2005). Due to the rapid growth rates of juvenile fish, we created two seasonal ALKs for each species: one for March through June and a second for July through October.

2.3.2 Growth and Body Condition

Two distinct growth and body condition metrics were used to explore potential relationships between fish condition and discharge (i.e., the Seton River hydrograph) for Rainbow Trout, Coho Salmon, and Chinook Salmon:

1. Fulton's Condition Factor (K_f): A measure of body condition, referring to the general plumpness or fatness of fish relative to length.
2. Length vs Weight: The predicted weight (or predicted incremental change in weight) given length at a given capture time and location.

The effects of year and capture location were evaluated for their effect on the above evaluation metrics. Because flow conditions varied so distinctly among study years, year can be used as a proxy for flow treatment in analyses.

Body Condition

K_f was calculated according to Anderson and Neumann (1996):

$$K_f = \frac{W * 10^N}{L^3} \qquad \text{Eq 3}$$

where *W* is weight in grams, *L* is fork length in millimeters, and *N* is an integer that scales the condition factor close to a value of one (generally *N*=5 for Seton River salmonids). We performed multivariate ANOVA (MANOVA) tests (α= 0.05) to determine the effects of year and reach on average K_f values for Rainbow Trout, Coho Salmon, and Chinook Salmon. For Rainbow Trout, age-specific MANOVA testing was performed for age 0 and age 1, while only age 0 were tested for Chinook Salmon and Coho Salmon. MANOVAs were not performed for higher age classes of Chinook and Coho because small sample sizes and missing data resulted in highly imbalanced year-reach comparisons. Five candidate models were tested, and the model with the lowest AIC value was selected as the best-fit model:

1. K_f = 1 (intercept-only model)
2. K_f = year
3. K_f = reach
4. K_f = year + reach
5. K_f = year*reach

When AIC values were within two units of each other (ΔAIC < 2), models were considered to have equal support and the most parsimonious model (the model with the fewest parameters) was selected as the best-fit model. Significant MANOVAs were followed by Tukey's pairwise hypothesis testing to determine statistical differences among groups (completed using the R package FSA at α = 0.5; Ogle 2016).

Length vs Weight

Length and weight are generally highly correlated for fish within a habitat and the relationship can be used to monitor gross changes in fish growth given variable environmental conditions. For example, increases in slope would suggest improved body condition (i.e., more weight per unit of length). Multiple log-linear regression modelling was used to describe the fork length (L) vs weight (W) relationships for Rainbow Trout, Coho Salmon, and Chinook Salmon pooled for all age classes captured in the Seton River and its spawning channels according to (Ogle 2016):

$$\log(W_i) = \log(\alpha) + \beta \log(L_i) + \epsilon_i \qquad \text{Eq 4}$$

where α and β are intercept and slope parameters and ε is multiplicative model error. A multiple linear regression was performed to evaluate the effects of the categorical covariates of year and capture location (i.e., *reach*) on the length vs weight relationship. Ten candidate linear models were evaluated using AIC model selection, and the model with the lowest AIC was selected as the best-fit candidate model:

1. log(W) = log(L)
2. log(W) = log(L) + reach
3. log(W) = log(L) + year

4. log(W) = log(L) + reach + year
5. log(W) = log(L)*reach
6. log(W) = log(L)*reach + year
7. log(W) = log(L)*year
8. log(W) = log(L) *year + reach
9. log(W) = log(L) + year*reach
10. log(W) = log(L)*reach*year

Due to the categorical nature of year and reach variables (each with five categories), the number of estimated parameters in candidate models with interaction terms was quite large. The most complex model with a three-way interaction between length, year, and reach had 50 parameters, which suggests that a large sample size is required to fit this model (i.e., 500-750 samples). Models with large numbers of parameters may overfit data. We evaluated the degree of model over-fitting using AIC model selection (where models are penalized with each additional parameter) and by comparing the model R-squared values to the predicted R-squared values (a measure of how well the model predicts individual observations). All models were assessed for linearity and homogeneity of variances.

In all cases the most complicated model best described the data, whereby length, year, reach, and their interactions were all statistically significant predictors of fish weight. Differences among capture locations were expected due to the inherent differences in habitat characteristics between different reaches and the spawning channels that occur regardless of BC Hydro management actions (e.g., flow rate, water temperature, depth, etc.). In contrast, differences in the length-weight relationship between years may indicate an effect of flow management decisions on fish growth. We performed reach-specific AIC modelling considering three candidate linear models to separate the effect of year on the length vs weight relationship in each reach:

1. log(W) = log(L)
2. log(W) = log(L) + year
3. log(W) = log(L)*year

These reach-specific models are simpler to interpret and help to determine how annual flow characteristics may be affecting growth of juvenile fish in the Seton River and its spawning channels.

2.3.3 DNA Sampling

Although age 0 Chinook have been captured throughout the monitor, few adults have been observed (see Section 2.6). This has led to uncertainties regarding the presence and use of the Seton River by adult Chinook for spawning. Unobserved Chinook may be spawning in the Seton River or, conversely, juvenile Chinook from other populations may be rearing and/or migrating in the Seton River, specifically those from Bridge River. Caudal fin-clip samples have been collected to obtain DNA from a subset of Chinook during bio-sampling and standing crop surveys (Sections 2.3 and 2.4.1, respectively) since 2016. Samples were analyzed using standardized genetic stock identification protocols at the Pacific Biological Station Molecular Genetics Lab in Nanaimo (Beacham et al. 1996).

2.4 Juvenile Abundance Estimation

We performed backpack electroshocking and nighttime snorkel surveys in the Seton River annually from 2014 to 2018 to estimate juvenile population abundance. A two-phase sampling protocol combines mark-recapture and index data to determine site- and river-wide population abundances [as in Korman et al. (2016) and Hagen et al. (2010)]. In the Seton River, the mark-recapture portion consisted of a two-pass backpack electrofishing program used to estimate river-wide fish detection probability. This detection probability was then applied to counts from separate index sites to obtain abundances for three reaches of the Seton River. The model was implemented through a hierarchical Bayesian framework in R Project Software (R Core Team 2017) and JAGS using the R package jagsUI (Kellner 2017).

Our goal was to incorporate index data from both fall open-site electroshocking and spring snorkel surveys in a multi-gear model to estimate juvenile abundance of Coho, Chinook, Rainbow Trout/steelhead, Bull Trout, and Mountain Whitefish in the Seton River. A multi-gear sampling design can account for variation in detection probability across different life stages and habitat types (Korman et al. 2016). For example, electroshocking detection probability is generally higher for juveniles relative to adults, whereas the opposite is true during snorkel surveys. The appropriateness of snorkeling and electroshocking also varies with seasonal discharges; snorkeling is not possible during high turbidity periods, while electrofishing is ineffective at high discharges. During both electrofishing and snorkeling, densities were too low to obtain abundance or index estimates for all species apart from Rainbow Trout. For Rainbow Trout, the hierarchical Bayesian model was used to estimate age 0 abundance using electroshocking mark-recapture and index data, while snorkel survey data were used to obtain annual indices of age 1 and age 2 abundance.

2.4.1 Electrofishing Surveys

Electrofishing surveys for indexing and mark-recapture were completed in September of each study year. Twenty-five electrofishing index sites were randomly selected each year from 125 sites previously assessed, distributed throughout Seton River from Seton Dam to the Seton-Fraser confluence (Figure 2-6). Index sites include shallow riffle, shallow glide, and deep habitats (Korman 2010), but deep habitats were excluded from the juvenile electrofishing abundance surveys as they cannot be efficiently surveyed with an electrofisher. An additional six mark-recapture sites were selected each year to represent shallow riffle and glide habitat in each of the three reaches to calculate capture efficiencies to be applied to the index sites. The actual number of mark-recapture sites included in the analysis varied each year due to annual conditions in the river and low to zero catches in some years (Table 2-2).

All open-site electrofishing surveys were performed during daylight hours by a three-person crew using a Smith-Root LR-24 backpack electrofisher. Electrofishing sites were 50 m long (shorter where habitat units were not 50 m in length) and were sampled systematically in an upstream direction, attempting to capture all fish observed. In side-channels and narrow sites, the entire width of the river was sampled, while in wider sections the crews sampled as far into the river as was safe to wade. Index sites were surveyed using a single pass, while mark-recapture sites were surveyed with two passes. During the first pass, fish were marked with a fin clip and released in their original capture site. A second pass was performed after 24 hours, and the number of marked fish re-caught recorded. All by-catch salmonid species were also weighed, measured, and sampled for ageing structure.

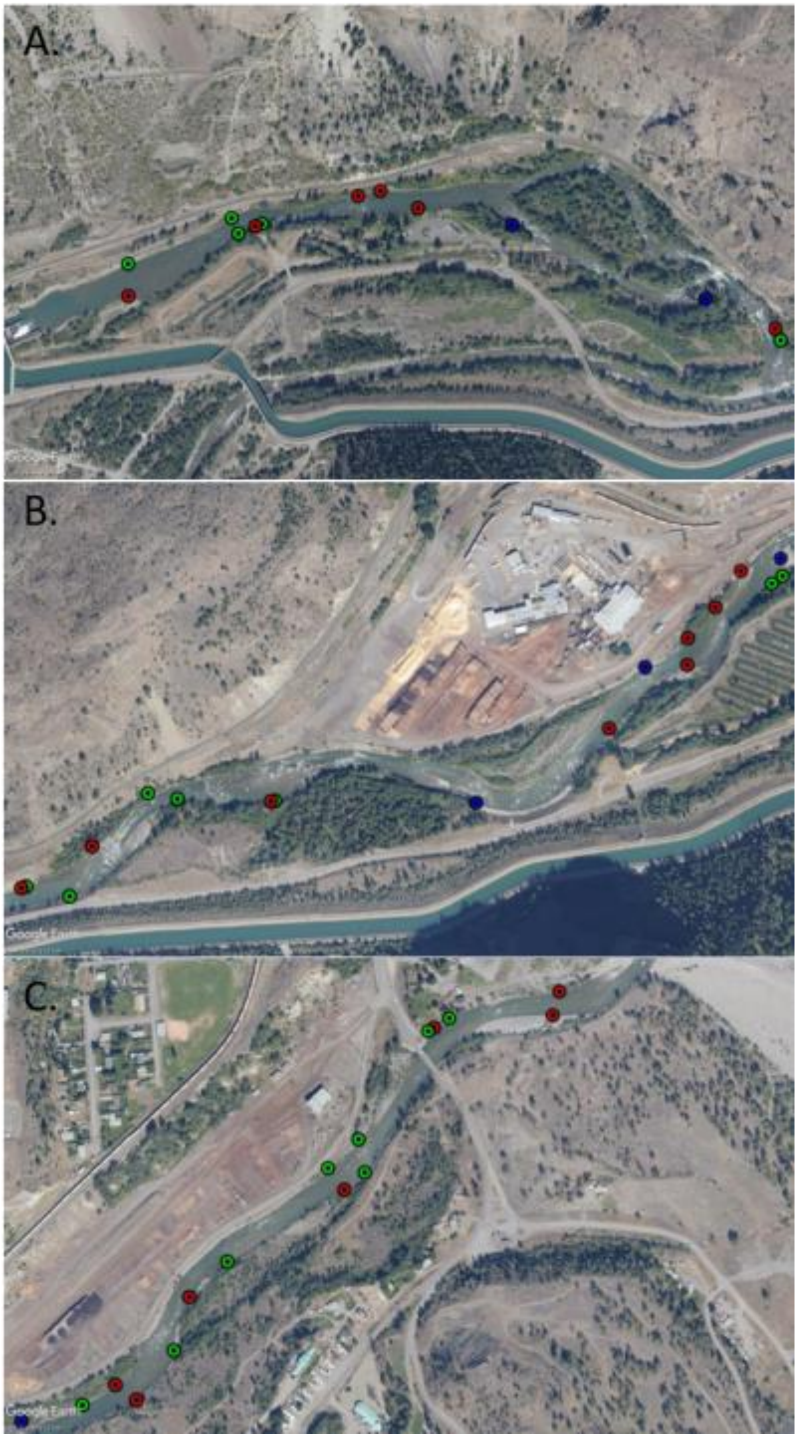


Figure 2-6. Location of juvenile standing crop sites in 2018 within Seton River in Reach 1 (A), Reach 2 (B), and Reach 3 (C). Sites were chosen randomly and cover both river right and river left. Red circles represent index-electrofishing sites, blue circles represent mark-recapture electrofishing sites and green circles represent snorkel survey sites.

Table 2-2. Summary of sites sampled from 2014-2018 in Seton River for indexing and mark-recapture (M-R) (EF = Electrofishing).

Year	Site Type	N	Mean	
			Site Length (m)	Time Shocked (s)
2014	EF M-R (Pass 1)	6	59	-
	EF M-R (Pass 2)	6	60	-
	EF Index	25	54	-
	Snorkeling	-	-	NA
2015	EF M-R (Pass 1)	4	48	1448
	EF M-R (Pass 2)	4	47	834
	EF Index	23	50	416
	Snorkeling	10	50	NA
2016	EF M-R (Pass 1)	5	56	1559
	EF M-R (Pass 2)	5	56	1148
	EF Index	23	50	744
	Snorkeling	20	48	NA
2017	EF M-R (Pass 1)	6	52	916
	EF M-R (Pass 2)	6	52	766
	EF Index	24	50	469
	Snorkeling	20	48	NA
2018	EF M-R (Pass 1)	6	52	1075
	EF M-R (Pass 2)	6	52	666
	EF Index	21	43	502
	Snorkeling	20	47	NA

2.4.2 Snorkel Surveys

Snorkel surveys were completed annually in the Seton River during March 2014-2018. Twenty indexing sites were randomly selected from deep habitats (deep riffles, deep glides, and pools) that could not be effectively sampled via shoreline electrofishing. Snorkel indexing sites were distributed throughout Seton River from Seton Dam to the Seton-Fraser confluence (Figure 2-6).

Snorkel surveys were performed during night hours using methods from Decker et al. (2009). A single snorkeler traversed the site in an upstream direction using an underwater light to observe and enumerate fish. Like the electrofishing survey sites, snorkel sites were up to 50 m long, dependent on characteristics of the habitat unit, and were surveyed as far into the river that was safe for swimmers

(0.5 to 5 m). Fork length was estimated to the nearest 5 mm by the snorkeler using a handheld underwater ruler. Count and length data was recorded by an on-shore safety observer.

2.4.3 Hierarchical Bayesian Analysis

The hierarchical Bayesian mark-recapture model consisted of two simultaneous levels: a detection model and a population model. The detection model used mark-recapture data from all sites sampled during the program to estimate a hyperdistribution for river-wide detection probability (Korman et al. 2016). Detection probability at site *i* (ϑ_i) was the ratio of marked fish recaptured in the second pass at site *i* (r_i) relative to the total number marked during the first pass (m_i; see variable definitions in (Table 2-3). The variation in detection probability among sites was assumed to be beta-distributed, while the variation among recapture rates was modelled using a binomial distribution. The overall hyperdistribution estimated by the detection model characterizes the mean (μ_ϑ) and variance of detection probability (τ_ϑ) from individual mark-recapture sites. During the Bayesian modelling, the hyperparameters act as prior distributions for the site-specific detection probabilities, such that sites with little mark-recapture data are heavily influenced by the hyperdistribution and sites with high numbers of marks and recaptures will be more site-specific and use data collected for that specific site.

Mark-recapture data from 2014 to 2018 were pooled to estimate hyperparameters for detection probability for the Seton River, thus assuming detection characteristics in the Seton River did not change over the five-year study period. To maintain consistent detection efficiency, we used experienced field crews and standardized protocols to minimize the effect of sampling crew, and electrofishing took place during similar discharge levels each year (~12 m^3/s).

The hyperparameters of detection probability estimated by the detection model were used by the process (population) model to obtain site-specific abundance estimates for the *j* index sites (N_j) as well as abundances for the unsampled shoreline (N_{us}; Table 2-3 and Table 2-4). The number of fish observed at an index site (c_j; used to determine density at site *j* [λ_j]) was assumed to binomially distributed based on the true site abundance (N_j) and a detection probability (ϑ_j) randomly drawn from the hyperdistribution of detection probability developed by the detection model. The abundance at each index site (N_j) was Poisson distributed with a mean equal the length of the site (l_j) multiplied by the site-specific density (fish/meter) estimated by the process model (λ_j). Abundance for the unsampled portion

(N_{us}) was the product of the length of unsampled shoreline and an expected fish density based on the mean and precision of the log-normal hyperdistribution of fish density ($\log(\lambda_j)$). The total river-wide abundance (N_t) was then calculated as the sum of abundances from sampled and unsampled shorelines.

All priors used during the hierarchical modeling were uninformative. The detection and population models were run simultaneously in JAGS using three separate chains of 10,000 iterations. The first half of the samples were unrepresentative of the equilibrium distribution and discarded through a 'burn in' process. The remaining samples comprised the final posterior distributions. A convergence threshold of 1.1 was used to ensure the model adequately fit the data.

Table 2-3. Definitions of variables used in hierarchical Bayesian model to estimate juvenile abundance.

Variable		Definition
Data	m_i	Marks released at mark-recapture site *i*
	r_i	Recaptured marked fish at mark-recapture site *i*
	c_j	Fish caught at index site *j*
	l_j	Length of index site *j*
Site-specific parameters	ϑ_i	Estimated detection probability for mark-recapture site *i*
	ϑ_j	Simulated detection probability at index site *j*
	λ_j	Estimated density (fish/m) at index site *j*
Hyperparameters	μ_θ	Mean of beta hyperdistribution for detection probability
	τ_θ	Precision of beta hyperdistribution for detection probability
	μ_λ	Mean of normal hyperdistribution for log density
	τ_λ	Precision of normal hyperdistribution for log density
Derived variables	α_i	Transformed hyperparameter for detection probability
	β_i	Transformed hyperparameter for detection probability
	N_j	Abundance at index site *j*
	N_s	Total abundance in sampled shorelines
	N_{us}	Total abundance in unsampled shorelines
	N_t	Total abundance in river

Table 2-4. Equations for hierarchical Bayesian model. The letters *i* and *j* represent the mark-recapture and index sites, respectively.

Model	Equation
Detection	$r_i \sim dbin(\vartheta_i, m_i)$
	$\vartheta_i \sim dbeta(\alpha, \beta)$
Population	$\vartheta_j \sim dbeta(\alpha, \beta)$
	$c_j \sim dbin(\vartheta_j, N_j)$
	$N_j \sim dpois(\lambda_j, l_j)$
	$\log(\lambda_j) \sim dnorm(\mu_\lambda, \tau_\lambda)$
Priors and Transformations	$\mu_\vartheta \sim dunif(0, 1)$
	$\sigma_\vartheta \sim dunif(0.05, 1)$
	$\tau_\vartheta = \sigma_\vartheta^{-2}$
	$\alpha \sim \mu_\vartheta\, \tau_\vartheta$
	$\beta = (1 - \mu_\vartheta)\, \tau_\vartheta$
	$\mu_\lambda \sim norm(0, 0.01)$
	$\sigma_\lambda \sim dhcauchy(0.1, 0.5)$
	$\tau_\lambda \sim \sigma_\lambda^{-2}$

2.5 Telemetry

2.5.1 Adult Radio Telemetry

Tagging and Bio-sampling

Adult Steelhead Trout have been tagged since 2013 to determine spawning locations. Skilled anglers attempt to capture fish throughout the Seton-Bridge complex, including the Seton-Fraser River confluence (Ramos-Espinoza et al. 2016). Fish captured were gastrically implanted with a TX-PSC-I-1200-M radio tag (44 x 16 x 16 mm; Sigma Eight Inc., Ontario, Canada) using methods described in Burnett et al. (2016). A 32 mm HDX PIT tag was also implanted into the dorsal musculature of each fish. Fork length and sex were recorded during tagging and scale samples were taken from all adults for ageing (see Section 2.3.1). After tagging, fish were held in a submersible holding tube for a minimum of 30 minutes prior to release to ensure full recovery, proper tag placement, and confirm the tag had not been regurgitated.

Tagging effort was distributed throughout the migration period. An effort was made to ensure even distribution of tags between sexes, in consideration of sex-specific migration behaviour and run timing

(Korman et al. 2010; Troffe et al. 2010). The tagging schedule was adaptive because suitable capture locations are limited on the Seton River. Tag releases were dependent on capture success, angling conditions, and fish behaviour.

From 2013-2015, effort was made to radio-tag migrating Coho and Chinook Salmon as well but angling efforts were unsuccessful, with only one fish tagged for each species over the 3 years. Angling for Coho and Chinook was discontinued in the Seton River in 2016.

Tag Tracking

Weekly mobile tracking with a hand-held Lotek W31 radio receiver (Lotek Wireless Inc., Ontario, Canada) was conducted for Steelhead in each year from mid-March (following the first fish tagged) to mid-May throughout the Seton River. Mobile tracking was completed by vehicle or foot and coincided with weekly visual surveys (see Section 2.6) when possible, but in isolation of the technicians conducting the visual surveys to avoid observer bias. Fish location and tag code were recorded, as well as visual sightings of tagged and untagged individuals of all species.

Fixed station logging was conducted from March 19 to June 22, 2018 at one site located on the Seton River, 1.3 km upstream of the Seton - Fraser River confluence (Figure 2-7). The fixed station used a Lotek W31 receiver linked to one Yagi 6-prong directional aerial oriented downstream. Fixed station data were used to corroborate fish locations determined by mobile tracking, identify entry and exit timing into the Seton River, and collect basic data on Steelhead adult migration and spawning in the Seton River.

Figure 2-7. Location of fixed telemetry stations on Seton River. PIT antennas are present near the mouth of the Upper Spawning Channel (USC) and Lower Spawning Channel (LSC) and in the Seton Fishway and Cayoosh Creek. A fixed radio antenna is located upstream of the confluence of the Lower Spawning Channel and Seton River.

2.5.2 Passive Integrated Transponder (PIT) Telemetry

As part of the monthly bio-sampling protocol conducted from 2014-2018 (see Section 2.3), all Rainbow Trout >75 mm were implanted with a PIT tag in the ventral cavity. PIT tag data were used to explore movement behaviour relative to changes in discharge from Seton Dam and if the spawning channels sustain populations distinct from the mainstem Seton River, or if a single population is seeded by the spawning channels.

PIT arrays

PIT antennas were installed in both spawning channels. Array characteristics varied slightly through the study period. The LSC only had one antenna in 2014, allowing for detection of tagged fish but not directionality of movement or detection efficiency. In 2015, a second antenna was added to the system to create an array (Figure 2-7). A two-antenna PIT array was installed in the USC in May 2015 (Figure 2-7), and thus detections span from 2015-2018.

Detection efficiency is calculated as the number of fish detected on both antennas divided by the total number of fish detected on the first. Low detection efficiencies indicate that fish were missed on one antenna but observed on the other (e.g., if detection efficiency is 13% for the upstream antenna, only

13% of the fish detected on the downstream antenna were also detected on the upstream antenna). Low detection efficiency has implications for determining direction of fish movements.

2.6 Adult Visual Counts

Visual stream counts were performed weekly as conditions allowed throughout the Seton River and spawning channels during the adult salmon migration period. Spawning Steelhead, Chinook and Coho Salmon were enumerated to provide an index of adult abundance, and any visible redds were noted and georeferenced. Survey methods followed those outlined in BRGMON-3 (Burnett et al. 2016), whereby two observers walk along the riverbank in a downstream direction looking for fish and any spawning activity. We recorded fish species, location, and viewing conditions, including cloud cover and lateral water visibility. Steelhead surveys were scheduled to be completed from March to June of each year but the surveys were not completed in the mainstem Seton River in 2016 through 2018 due to modified operations and low water visibility. Chinook surveys commence in August of each year and continue through to October, while Coho surveys begin in October and are completed by mid-December (Table 2-5). In 2018, the August portion of the Chinook run was not surveyed due to a miscommunication between contractors. Visual count surveys resumed in September.

Table 2-5. Timing of Adult Visual Surveys for Steelhead, Chinook and Coho for each year.

	2015	2016	2017	2018
Steelhead	Mar 4 – Jun 15	NA	NA	NA
Chinook	Aug 8 – Oct 6	Aug 16 – Oct 7	Aug 8 – Oct 4	Sep 25 – Oct 15
Coho	Oct 6 – Dec 15	Oct 7 – Dec 16	Oct 4 – Dec 12	Sep 25 – Nov 26

3.0 RESULTS

3.1 Physical Parameters

3.1.1 Discharge

As in 2016 and 2017, modified operations in the Bridge-Seton system in 2018 resulted in Seton Dam discharges which significantly exceeded the WUP target hydrograph. Starting on April 4, 2018 flows increased steadily from the WUP target flows of 15 m^3/s, reaching a maximum of 93 m^3/s on May 19 (Figure 3-1). Following this peak, flows were reduced to 53 m^3/s over a three-week period. Flows returned to WUP targets on July 27th and were maintained for the remainder of the year.

Discharge experienced in Reach 2 and 3 of the Seton River requires addition of Cayoosh Creek and Seton Dam discharge. Flows from Cayoosh in 2018 ranged from 1.65 – 75.8 m^3/s (Figure 3-1).

3.1.2 Water Temperature

Annual low water temperatures occur in March (4°C) and increase gradually throughout the year until September when temperatures peak around 18°C. Water temperatures in 2018 followed this pattern reaching approximately the same/higher/lower temperatures as 2014-2017. Water temperatures decrease gradually through the fall until settling around 5°C in December or early January. Spawning Channel temperatures follow the same profile as the mainstem Seton River (Figure 3-2). Cayoosh Creek frequently influences the temperature of the Lower Seton, cooling it by ~1°C.

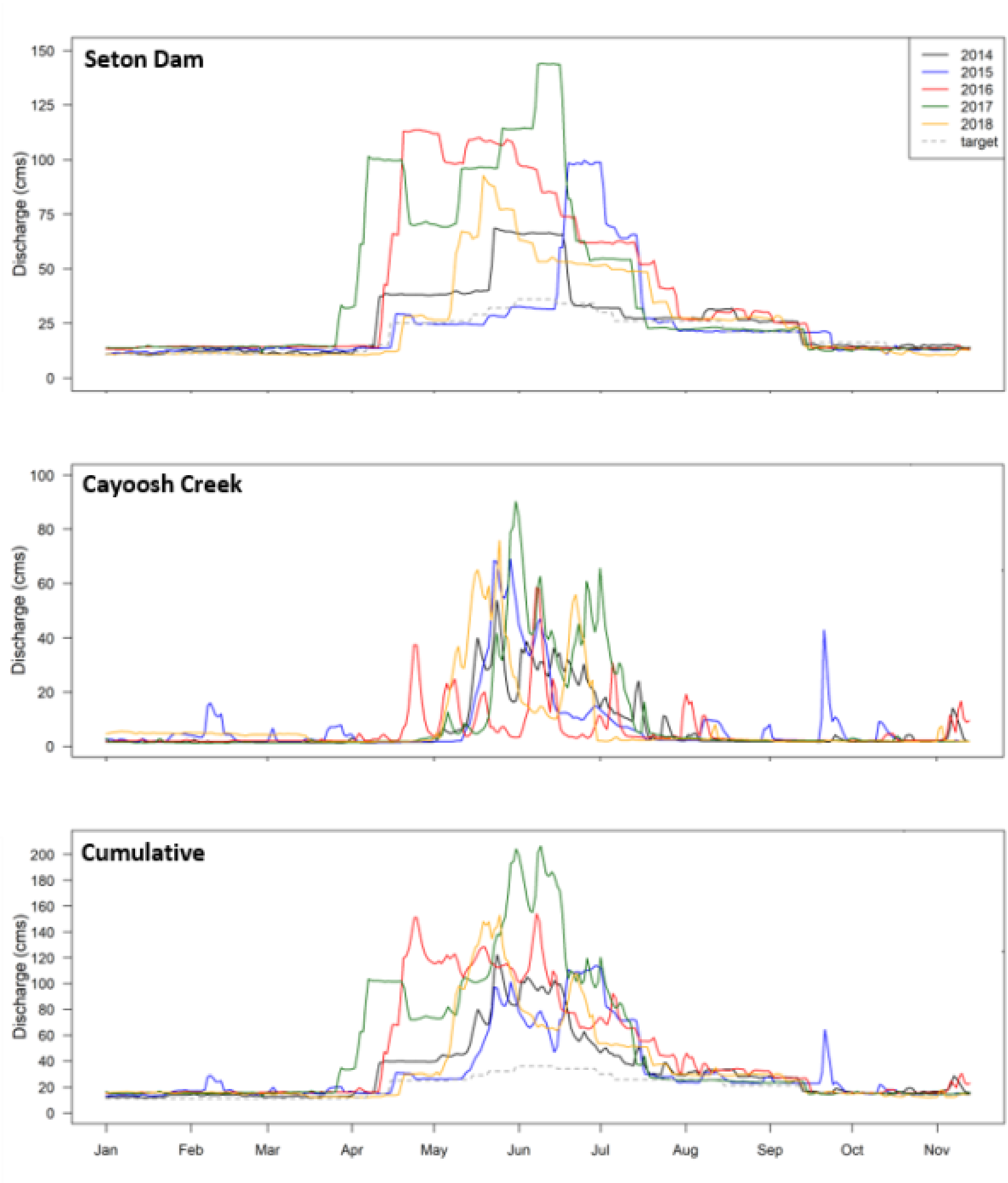


Figure 3-1. Seton Dam and Cayoosh Creek discharge for BRGMON-9 study years and the cumulative flow (Seton River and Cayoosh Creek) in Reach 2 and 3 of Seton River for BRGMON-9 study years.

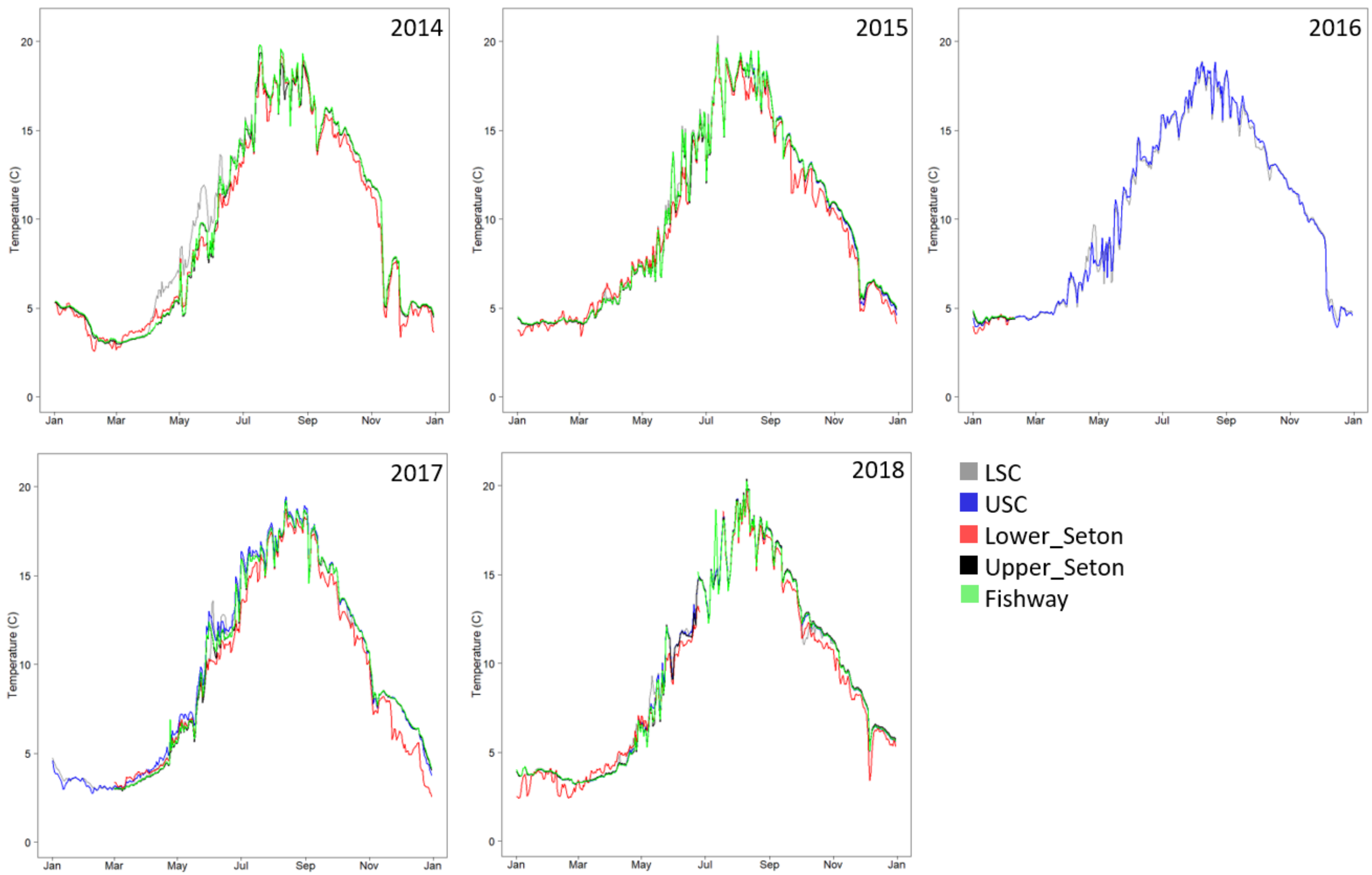


Figure 3-2. Temperature profile for each site for each year. Data is missing for most of 2016 for the mainstem Seton River sites due to modified operations at Seton Dam.

3.2 Habitat Suitability Assessments

Habitat Suitability Assessments were completed at each of the stock assessment sites from September 18 – October 30, 2018 when flows from Seton Dam ranged from 10.3-14.6 m^3/s with a mean flow of 12.6 m^3/s and flow contribution from Cayoosh Creek was minimal (1.8 m^3/s). These conditions are comparable to the 2014 Survey 1 conditions which occurred at a mean flow of 12.3 m^3/s from Seton Dam and 1.5 m^3/s from Cayoosh. While 2014 surveys were completed for the entire Seton River, only those transects repeated in 2018 are compared below.

3.2.1 Juvenile Rearing Habitat Suitability

In Reach 1 sites (n = 9), the WUA for Rainbow Trout, Coho and Chinook was greater in 2018 relative to 2014, while the opposite was observed in Reach 2 (n = 4). In Reach 3 (n = 12), results were variable; WUA values were higher in 2018 for Rainbow Trout fry and Coho but higher in 2014 for Rainbow Trout parr and Chinook. Overall across all reaches, a net loss in useable habitat was observed for Rainbow Trout fry and parr (-14% and -41%, respectively) and Chinook (-42%) while habitat for Coho increased by 7% (Figure 3-3 and detailed site breakdown in Appendix 7-1).

3.2.2 Spawning Habitat Suitability

The WUA, at the common sites, for spawning Coho and Chinook was lower in 2018 compared to 2014 in Reach 1 and Reach 2, but the opposite was true for Reach 3. Overall across all reaches, a net loss in useable spawning habitat was observed for both Coho and Chinook (-22% and -18% respectively; Figure 3-3 and detailed site breakdown in Appendix 7-2).

WUA was also measured at three transects within each of the two known spawning areas in 2014 and 2018. In Reach 1, the known spawning area saw a reduction in habitat suitable for spawning in 2018 relative to 2014 for Coho, but an increase for Chinook. WUA has increased for both species in the Reach 3 known spawning area. Overall, a net increase was observed in these known spawning locations for Coho and Chinook from 2014 to 2018 (2% and 23% respectively, Figure 3-3).

Figure 3-3. Percent change in WUA (2018 in relation to 2014) for each reach of Seton River. WUA is displayed for juvenile rearing areas, spawning areas (Coho and Chinook only), and areas where spawning is known to occur (Coho and Chinook only). There are no known spawning areas in Reach 2. The dotted line represents no net change in WUA.

3.2.3 Substrate Changes

Relative to 2014, dominant and sub-dominant substrate classes at each transect in 2018 is trending larger in Reach 1 and 3 but has stayed the same in Reach 2 (Figure 3-4, Appendix 7-3).

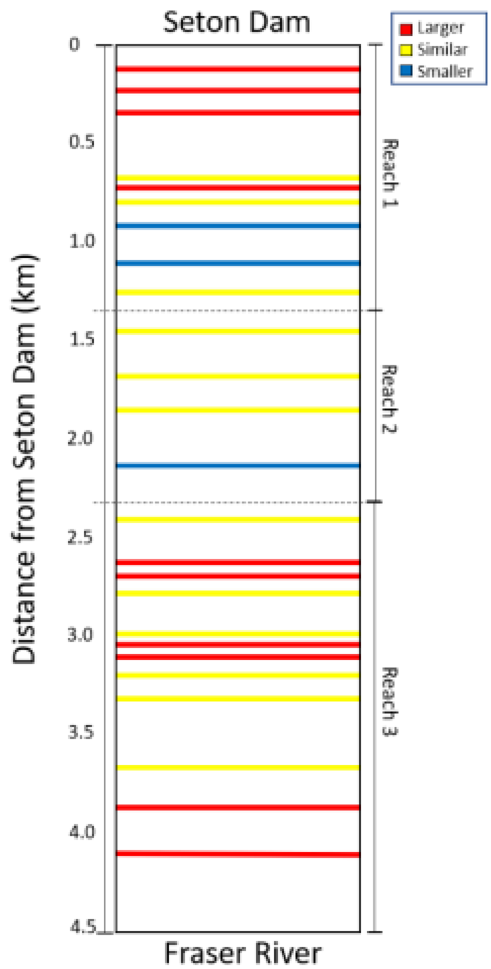


Figure 3-4. Change in substrate (Larger, smaller, or similar) for each transect surveyed in 2014 and 2018 in relation to Seton Dam.

3.3 Bio-sampling of Juveniles

Juvenile salmonids were sampled monthly from April to October in 2014-2018. Due to modified operations, some mainstem sites could not be accessed and additional side-channel habitats were sampled when wetted (See Table 2-1 in Section 2.3 for a detailed summary). A wide range of species were captured including six species of salmonids and numerous non-salmonids (Table 3-1).

Four Rainbow Trout were caught in 2018 during angling surveys resulting in a CPUE of 0.25 fish/hr.

Table 3-1: Total number of fish species caught during juvenile bio-sampling surveys in all years of BRGMON-9 monitoring. Effort increased in the spawning channels in 2017 and 2018 which may account for the increased abundance of some species.

Species	Count				
	2014	2015	2016	2017	2018
Bridgelip Sucker	30	47	12	38	162
Bull Trout	4	1	1	5	4
Sculpin *Spp.*	182	302	119	395	431
Chinook Salmon	22	197	211	298	121
Coho Salmon	674	447	143	279	456
Longnose Dace	400	484	111	565	801
Lamprey	0	0	2	1	0
Mountain Whitefish	14	7	6	1	0
Northern Pikeminnow	0	0	16	0	0
Peamouth Chub	0	1	6	0	0
Pink Salmon	36	0	0	0	5
Rainbow Trout	1377	664	684	864	966
Red-sided Shiner	59	14	19	41	72
Sockeye Salmon	6	24	4	2	0

3.3.1 Ageing Analysis

Of salmonids, only Rainbow Trout, Coho, and Chinook Salmon were captured in sufficient numbers (i.e., >500 compared to <50 Pink or Sockeye) to show the presence of discrete size classes and ALKs were produced to estimate ages of fish not aged by scale reading. Fish from all years and capture locations were pooled for ALKs under the assumption that fish move freely between the spawning channels and mainstem (see Section 3.5.2).

Rainbow Trout

Four distinct age classes were identified for Rainbow Trout. The most frequently captured age class was age 0 followed by age 1, while catch rates for age 2 and age 3 Rainbow Trout were lower. In all years, captures of age 1 through 3 Rainbow Trout were well distributed between the mainstem river habitats and the two spawning channels, while age 0 Rainbow Trout were primarily captured in the mainstem river (see Appendix 7-4). Fork length distributions for all Rainbow Trout age classes demonstrate clear monthly growth from March to October (Figure 3-5) and suggest that the ALKs adequately estimated age for juvenile Rainbow Trout.

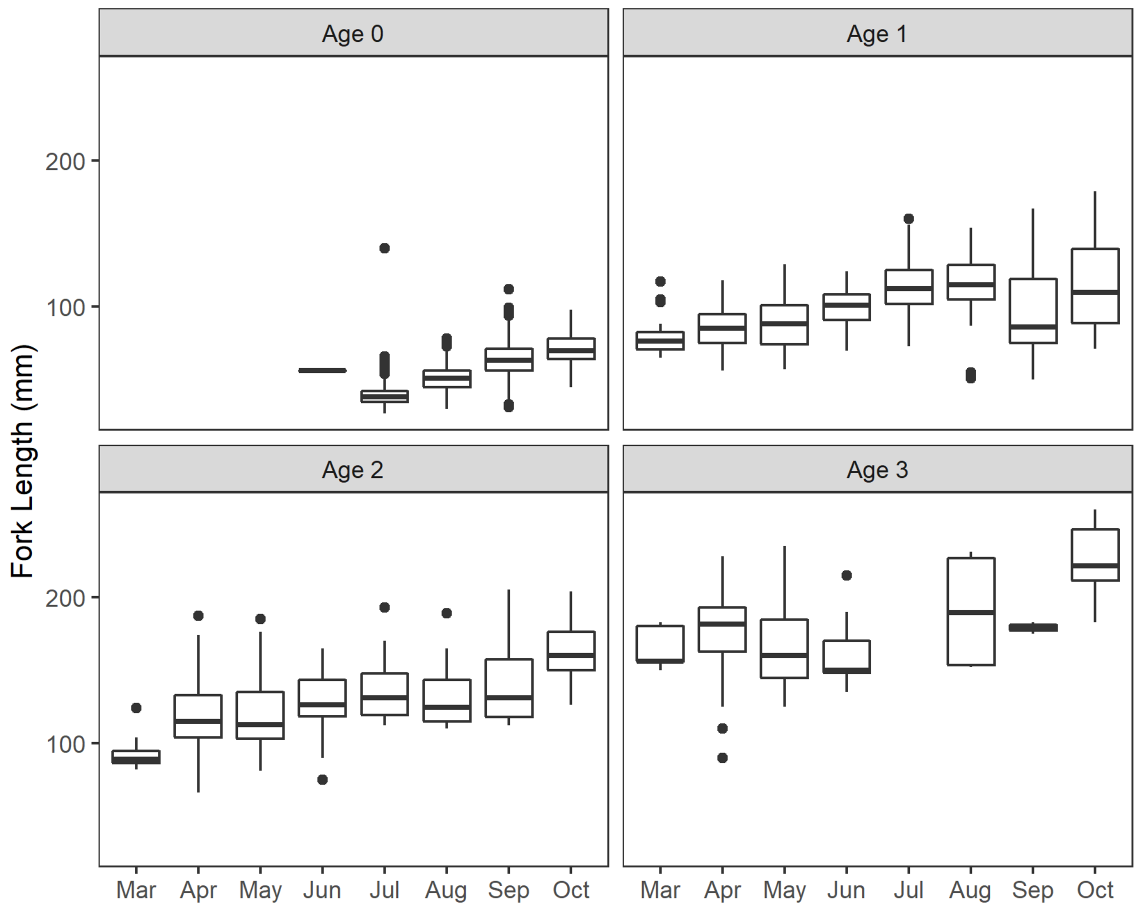


Figure 3-5. Boxplots of fork length for Rainbow Trout aged 0 to 3 captured in the Seton River and the spawning channels from 2014 to 2018.

Coho Salmon

Three distinct age classes were identified for Coho. Age 0 were the most frequently captured followed by age 1; only 17 age 2 Coho have been observed throughout the program. In 2014 through 2017, captures of Coho were relatively well distributed between the mainstem river habitats and the two spawning channels (see Appendix 7-5); however, in 2018, the number of Coho captured in the LSC was more than double that of any other reach sampled. Low captures of age 2 Coho made it difficult for the

ALK to partition fish with larger fork lengths; however, as all age 2 fish were selected for ageing, the small sample size did not have a noticeable effect on monthly growth trajectories (Figure 3-6).

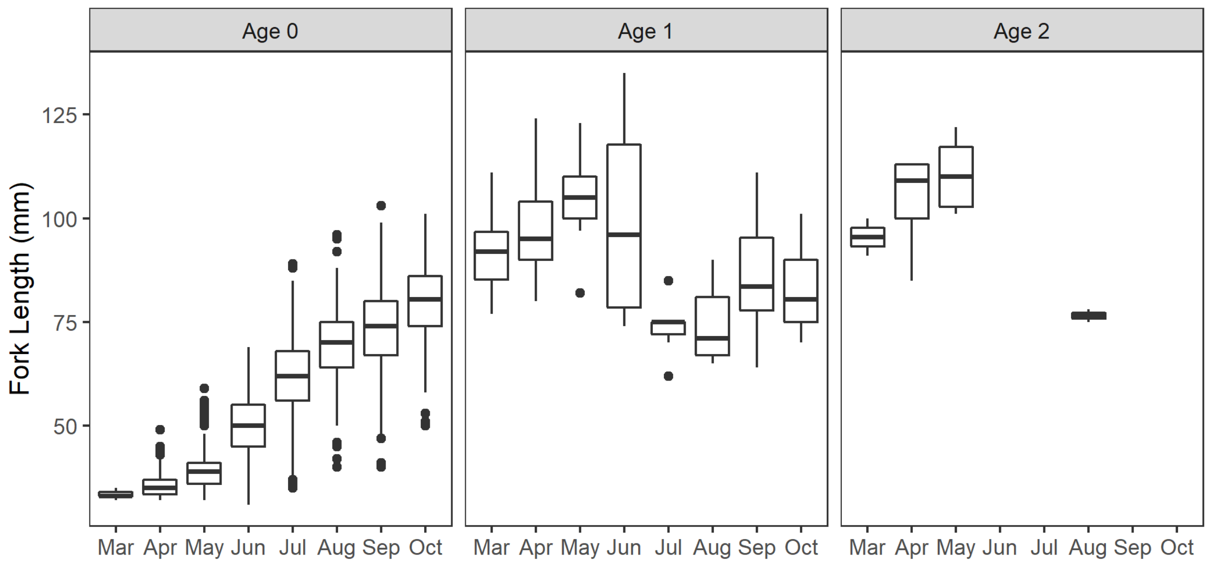


Figure 3-6. Boxplots of fork length for Coho Salmon aged 0 to 2 captured in the Seton River and the spawning channels from 2014 to 2018.

Chinook Salmon

Three distinct age classes were identified for Chinook Salmon. The most frequently captured age class was age 0 followed by age 1; only 30 age 2 Chinook were seen throughout the program. Captures of Chinook were variable between the mainstem river and spawning channels, with no consistent pattern in catch rates by location (see Appendix 7-6). Low captures of age 2 Chinook made it difficult for the ALK to partition fish with larger fork lengths; however, as all age 2 fish were selected for ageing, the small sample size did not have a noticeable effect on monthly growth trajectories (Figure 3-7).

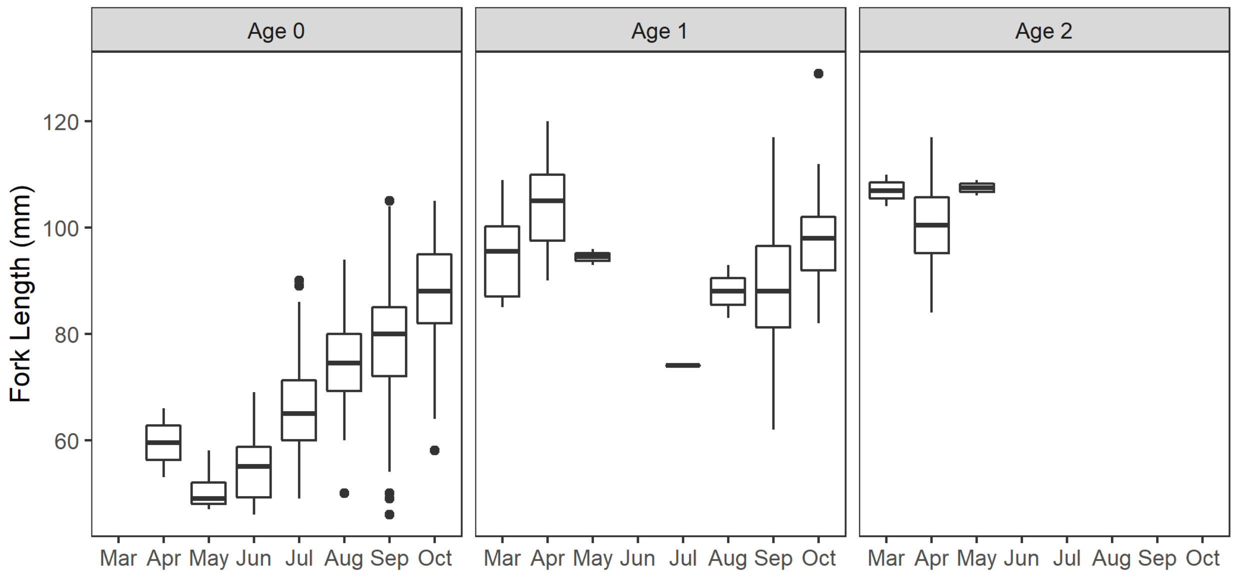


Figure 3-7. Boxplots of fork length for Chinook Salmon aged 0 to 2 captured in the Seton River and the spawning channels from 2014 to 2018.

3.3.2 Growth and Body Condition

Rainbow Trout

Body Condition (K_f)

We determined the age-specific effect of year and reach on mean K_f using MANOVA analyses and AIC model selection. For age 0 Rainbow Trout, the most complex model (year*reach), and the model with reach alone had equal AIC support (ΔAIC < 2), and the reach-only model was selected as the top model (Appendix 7-7). Tukey's hypothesis testing suggested that for age 0 Rainbow Trout, mean K_f values were statistically similar among LSC and USC, while mean K_f in Reach 1 was higher than in Reach 2 and 3 (ANOVA p-value 1.87e-07; Figure 3-8). For age 0 Rainbow Trout, capture location affected body condition while year did not. For age 1 Rainbow Trout, the intercept-only model and the model with year alone had equal AIC support, and the intercept-only model was selected as the best-fit model; neither year nor location significantly affected Rainbow age 1 body condition. No difference between years for body condition of age 0 Rainbow Trout.

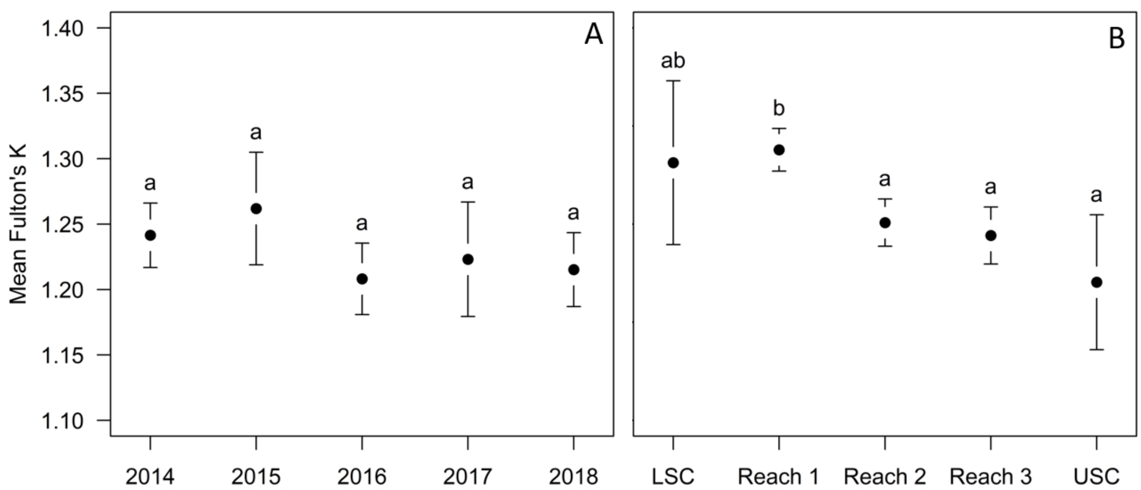


Figure 3-8. Mean condition factor of age 0 Rainbow Trout (A) each year (2014-2018) and (B) in the mainstem Seton River (Reach 1, 2 and 3) and the upper and lower spawning channels (USC and LSC, respectively) for all years. Years/Locations that do not share the same letter are statistically different from each other.

Length Vs. Weight Analyses

Effects of year and location on length and weight relationships were assessed using a multiple log-linear regression model with length and weight data pooled for all age classes. The top model according to AIC model selection was the most complex model: log(length)*year*reach (Appendix 7-8). The ΔAIC for the second model was 30, suggesting strong support for the best-fit model. Results from the best-fit model suggest a good model fit with R-squared of 0.971 ($F_{(49, 2597)}$ = 2015, p-value <2.2e-16). AIC and predicted R-squared (0.969) suggested that overfitting was not significant, and residual diagnostics suggested the assumptions of linear regression were not violated.

Given inherent difficulties in interpreting such complex interactive models, best-fit models for reach-specific length vs weight modelling were also determined using AIC. Year was found to be a significant predictor of weight in all reach-specific models except for in the LSC (Table 3-2). In Reach 1 and Reach 2, year had a significant impact on the slope and intercept of the length vs weight model (i.e., both the rate of change between length and weight and the average weight were different between years), while in

Reach 3 and in the USC, year had a significant impact on just the intercept (i.e., the average weight was different between years but year had no effect on the rate of change in length given weight).

Coho Salmon

Body Condition (K_f)

We determined the age-specific effect of year and reach on mean K_f using MANOVA analyses and AIC model averaging. For age 0 Coho, the most complex model (year*reach) had the highest AIC support and was selected as the best-fit model (Appendix 7-7). This suggests that year, reach, and their interactions affect mean body condition of Coho in the Seton River. To simplify the effects of year and reach on body condition we performed post-hoc Tukey's hypothesis testing on one-way ANOVAs of reach and year. Tukey's hypothesis testing suggested that mean K_f values were statistically similar in 2014, 2016, and 2018, while 2015 was low relative to all other years and 2017 was high relative to all other years (ANOVA p-value <2.2e-16; Figure 3-9). With all years of data pooled, condition was statistically similar in all reaches, but condition in Reach 1 was statistically lower than in the LSC (ANOVA p-value 0.002; **Error! Reference source not found.**).

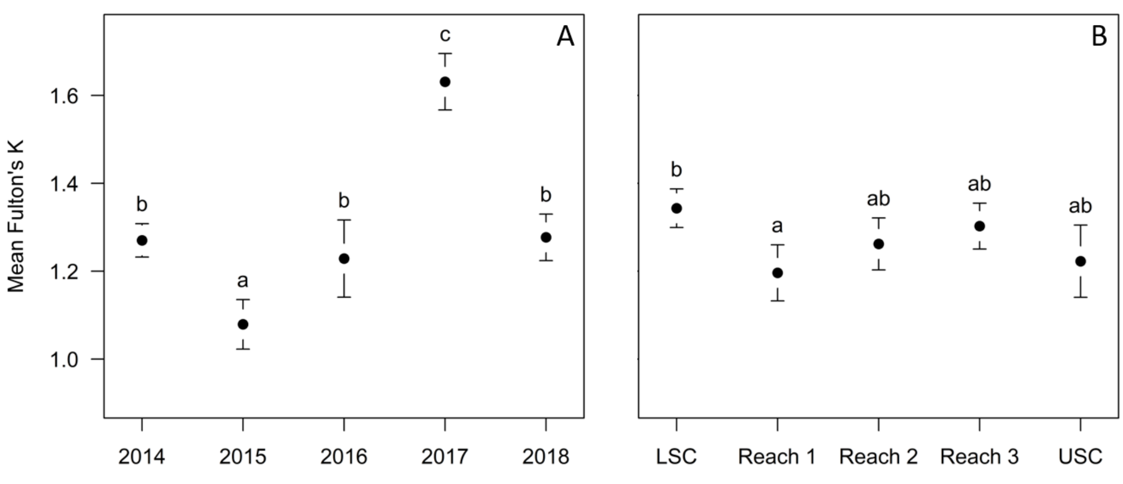


Figure 3-9. Mean condition factor of age 0 Coho Salmon (A) each year (2014-2018) and (B) in the mainstem Seton River (Reach 1, 2 and 3) and the upper and lower spawning channels (USC and LSC, respectively) for all years. Years/Locations that do not share the same letter are statistically different from each other.

Length Vs. Weight Analyses

Effects of year and location on length and weight relationships were assessed using a multiple log-linear regression model with length and weight data pooled for all age classes. The top model according to AIC model selection was the most complex model: log(length)*year*reach (Appendix 7-8). Results from the best-fit model suggest a good model with R-squared of 0.908 ($F_{(49,\ 1477)}$ = 298.6, p-value <2.2e-16). The ΔAIC for the second model was 74, suggesting strong support for the best-fit model. AIC and predicted R-squared (0.908) suggested that overfitting was not significant, and residual diagnostics suggested the assumptions of linear regression were not violated.

Best-fit models for reach-specific length vs weight modelling were determined using AIC. Year was found to be a significant predictor of weight in all reach-specific models except for in the USC (Table 3-3). In all other habitats (Reach 1-3 and the LSC), year had a significant impact on the slope and intercept of the length vs weight model (i.e., both the rate of change between length and weight and the average weight were different between years).

Chinook Salmon

Body Condition (K_f)

We determined the age-specific effect of year and reach on mean K_f using MANOVA analyses and AIC model selection. The best fit model according to AIC was the intercept-only model, suggesting mean K_f did not differ statistically between capture locations or years for age 0 Chinook Salmon (Appendix 7-7).

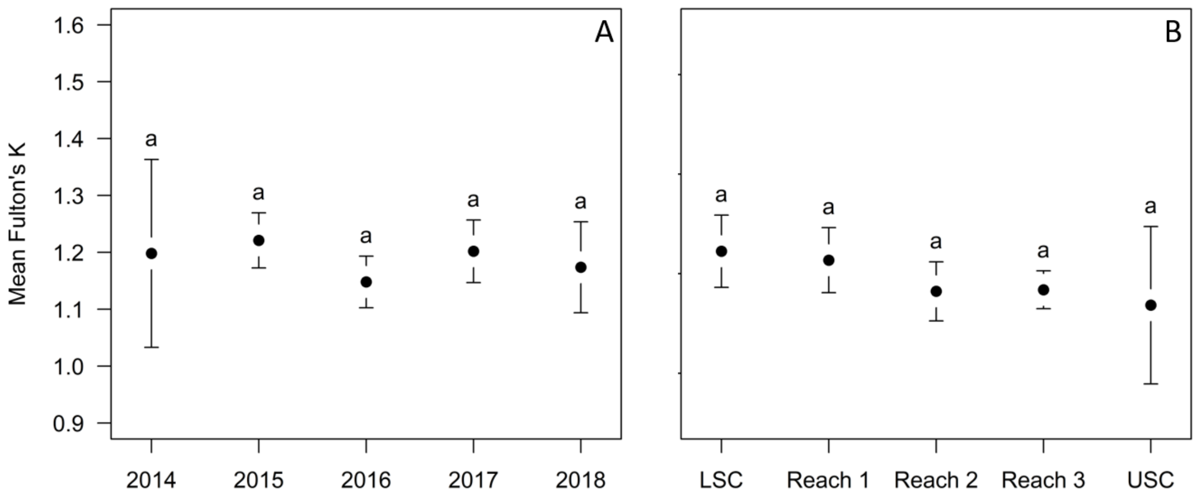


Figure 3-10. Mean condition factor of age 0 Chinook Salmon (A) each year (2014-2018) and (B) in the mainstem Seton River (Reach 1, 2 and 3) and the upper and lower spawning channels (USC and LSC, respectively) for all years. Years/Locations that do not share the same letter are statistically different from each other.

Length Vs. Weight Analyses

Effects of year and location on length and weight relationships were assessed using a multiple log-linear regression model with length and weight data pooled for all age classes. The most complex model: log(length)*year*reach, and the simplest model: log(length) alone, had equal AIC support (ΔAIC < 2), and the simplest model was therefore selected as the best-fit model (Appendix 7-8). Results from the best-fit model suggest a good model fit with R-squared of 0.913 ($F_{(1,\ 515)}$ = 5408, p-value <2.2e-16). AIC values suggested that more complex models with interactive terms tended to overfit the Chinook dataset. Overfitting may be due to a smaller sample size of Chinook over the 5-year sampling period (n total = 517), particularly in the USC (Reach 1: n = 100, Reach 2: 103, Reach 3: 264, USC: 26, LSC: 101).

Table 3-2. Direction and significance level of coefficients in reach-specific length vs weight modelling of Rainbow Trout in the Seton River from 2014 to 2018. Red values represent a negative effect on intercept or slope, and blue represents a positive effect. Significance codes for coefficient p-values are: * = <0.001, ** = <0.01, * = <0.05, and = <0.1. The sample size for each reach-year combination is shown in parentheses.**

			Year Effect on Intercept Relative to 2014				Effect on Slope Relative to 2014			
Reach	Model	n 2014	Int 2015	Int 2016	Int 2017	Int 2018	Slope 2015	Slope 2016	Slope 2017	Slope 2018
Reach 1	Log(L)*year	(240)	↓*** (248)	↑** (188)	↓ (147)	↑ (259)	↑**	↓*	↑	↓
Reach 2	Log(L)*year	(341)	↓ (57)	↑. (172)	↓*** (58)	↓ (159)	↑	↓.	↑***	↑
Reach 3	Log(L)+year	(144)	↑* (19)	↓* (213)	↓* (54)	↓ (103)				
USC	Log(L)+year	(129)	↑** (71)	↑ (19)	↑. (43)	↑** (78)				
LSC	Log(L)	(143)	(48)	(24)	(24)	(31)				

Table 3-3. Direction and significance level of coefficients in reach-specific length vs weight modelling of Coho Salmon in the Seton River from 2014 to 2018. Red values represent a negative effect on intercept or slope, and blue represents a positive effect. Significance codes for coefficient p-values are: * = <0.001, ** = <0.01, * = <0.05, and = <0.1. The sample size for each reach-year combination is shown in parentheses.**

			Year Effect on Intercept Relative to 2014				Effect on Slope Relative to 2014			
Reach	Model	n 2014	Int 2015	Int 2016	Int 2017	Int 2018	Slope 2015	Slope 2016	Slope 2017	Slope 2018
Reach 1	Log(L)*year	(43)	↓* (129)	↓ (16)	↓ (24)	↓ (16)	↑*	↑	↑	↑
Reach 2	Log(L)*year	(191)	↓*** (30)	↑ (24)	↓ (16)	↓*** (8)	↑***	↓	↑	↑***
Reach 3	Log(L)*year	(178)	↓ (30)	↓ (24)	↑*** (40)	↑ (61)	↑	↑	↓***	↓
USC	Log(L)	(30)	(29)	(14)	(57)	(39)				
LSC	Log(L)*year	(156)	↓*** (70)	↑ (34)	↑*** (94)	↓* (172)	↑***	↓	↓***	↑*

3.3.3 DNA Sampling

In total, 207 of the 240 samples sent for analysis were identified to stock origin. The top five Chinook stock origins present within Seton River are Seton River/Portage Creek (n = 99), Stuart (n = 30), Quesnel (n = 25), Nechako (n = 16) and Chilko (n = 14). An additional 23 Chinook were detected from other watersheds (Appendix 7-9). 58% of the juvenile Chinook originating from other watersheds (all years combined) were found in Reach 3 closest to the Fraser River confluence of the Seton River but were also observed throughout Seton River in lower numbers. The proportion of Seton River/Portage Creek fish in each reach relative to those originating from other populations is displayed in Table 3-4. Bridge River Chinook were not captured in Seton River.

No designated Seton River DNA profile exists for Chinook Salmon so Portage Creek and Seton River Chinook cannot be distinguished. They are presented as Seton River/Portage Creek below.

Table 3-4. Proportion of Seton River/Portage Creek Chinook relative to other populations captured in each sampling location of Seton River by year. Sample sizes presented in parenthesis. LSC and USC refer to Lower and Upper Spawning Channels, respectively.

Reach	2016	2017	2018	All years combined
1	0.57 (14)	0.67 (9)	0.95 (21)	0.77 (44)
2	0.36 (28)	0.79 (14)	0.50 (2)	0.50 (44)
3	0.15 (40)	0.70 (30)	0.31 (29)	0.36 (99)
LSC	0.08 (12)	0.50 (2)	1.0 (1)	0.20 (15)
USC	1.0 (1)	0.25 (4)	1.0 (1)	0.50 (6)

Juvenile Chinook were sampled throughout the year and proportions of each stock varied. Portage origin Chinook were sampled in consistent numbers (n = 4-26), but there appeared to be a higher number of other stock origin Chinook present in late summer and early fall months (August - October; Table 3-5). Juvenile Chinook from Portage Creek are similar in size to those from other watersheds throughout the sampling period, except in May and June when Portage Creek are smaller than those from other watersheds (Table 3-5).

Table 3-5. Mean fork length (mm ± standard deviation) and sample sizes of juvenile Chinook identified to be of Seton River/Portage Creek origin, compared to juvenile Chinook from other watersheds. Results are presented by month, and combine results from 2016, 2017 and 2018.

	May	June	July	August	September	October
Seton River	45.2 ± 4.0 (n = 24)	47.4 ± 8.6 (n = 24)	68.3 ± 7.8 (n = 4)	74.6± 9.1 (n = 16)	84.0 ± 6.3 (n = 26)	87.0 ± 5.5 (n = 4)
Other Watersheds	85.3 ± 38.4 (n = 3)	60.7 ± 4.9 (n = 7)	62.5 ± 9.2 (n = 4)	76.8 ± 8.3 (n =30)	84.0 ± 11.7 (n = 51)	100.2 ± 11.0 (n = 13)

3.4 Juvenile Abundance Estimation

3.4.1 Electrofishing Surveys

During September 2018, electroshocking covered approximately 15% of the total shoreline of Seton River, consistent with previous years (Table 3-6).

Although all species encountered were enumerated, weighed, and measured, only age 0 Rainbow Trout were captured in sufficient densities to be used in the Bayesian hierarchical modeling. Average recapture percentages calculated using mark-recapture data from 2014 through 2018 (i.e., recaptures/marks * 100) ranged from 10% in 2015 to 35% in 2016 (Table 3-7). The mean of the beta hyperparameter for detection probability estimated by the Bayesian hierarchical model for 2014 through 2018 was 0.28 (i.e., 28% detection probability; Figure 3-11) with an SD of 0.02.

The total river-wide abundance of age 0 Rainbow Trout in the Seton River was 4,290 fish with a 95% credible interval of 2,717 – 7,095 fish (Table 3-8). Abundance in 2018 was greater than estimated for 2015 (2,487 fish) and 2016 (3,871 fish) but less than 2017 (4,898 fish) and less than half the abundance estimated in 2014 (11,157 fish; Table 3-8). Although the 2014 abundance estimate was substantially higher than in other years, there is a high degree of uncertainty in this estimate due to variable densities observed during 2014 shoreline electroshocking (Figure 3-12, Figure 3-13). The hyperdistribution of fish density for the Seton River in 2018 (mean density 0.36) is shown along with site-specific density estimates in Figure 3-14. The mean of the hyperdistribution of fish density in 2018 was the third highest amongst all sample years behind 2014 (1.05 fish/m) and 2017 (0.40 fish/m).

Table 3-6. Percentage of shoreline sampled during electrofishing at shoreline index sites in the Seton River from 2014 to 2018.

Reach	Percent Sampled				
	2014	2015	2016	2017	2018
1	17	17	11	13	12
2	28	27	28	18	8
3	12	19	11	15	24
Total	**18**	**21**	**16**	**15**	**17**

Table 3-7. Recapture probabilities (recaptures/marks) calculated for mark-recapture sites in the Seton River during shoreline electroshocking from 2014 to 2018.

Year	Avg Recapture %	SD Recapture %	N
2014	29	8	6
2015	10	11	4
2016	35	13	5
2017	29	4	6
2018	30	12	6

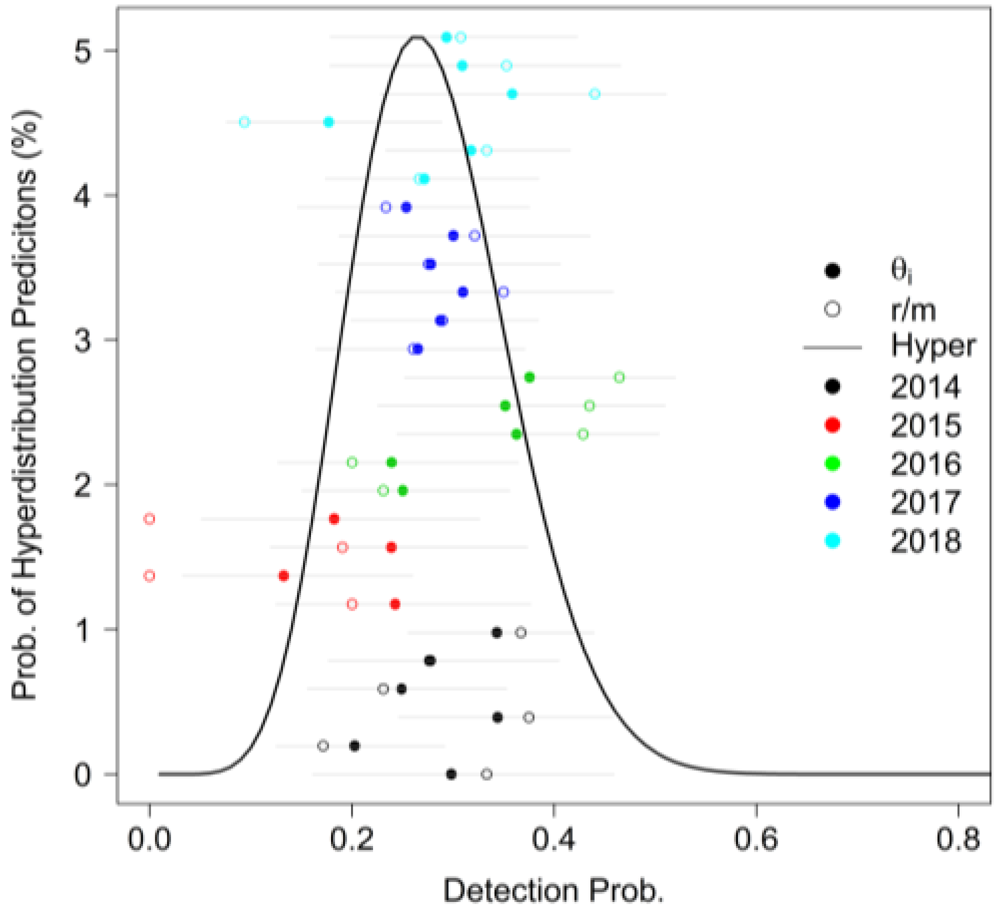


Figure 3-11. Parameter estimates from the hierarchical Bayesian model that estimate age 0 juvenile Rainbow Trout abundance. Shows the median hyperdistribution for detection probability, the median estimates of site-specific detection probability at mark-recapture sites and 95% credible interval (θ_i), and expected values (r/m).

Table 3-8. Mean posterior estimate of abundance and 95% credible interval (in parentheses) for Seton River Reach 1, 2 and 3 from 2014 to 2018.

Reach	2014	2015	2016	2017	2018
1	3,644	828	1,226	1,521	1,345
	(2,495, 5,371)	(5,551, 1,237)	(637, 2,528)	(861, 2,841)	(821, 2,288)
2	3,251	718	1,210	1,550	1,397
	(2,270, 4,725)	(479, 1,078)	(637, 2,272)	(913, 2784)	(885, 2,333)
3	4,262	941	1,435	1,827	1,545
	(2,912, 6,271)	(629, 1,407)	(756, 2,910)	(1,061, 3,325)	(995, 2,517)
Total	11,157	2,487	3,871	4,898	4,290
	(7,894, 16,064)	(1,709, 3,612)	(2,178, 7,452)	(2,948, 8,670)	(2,717, 7,095)

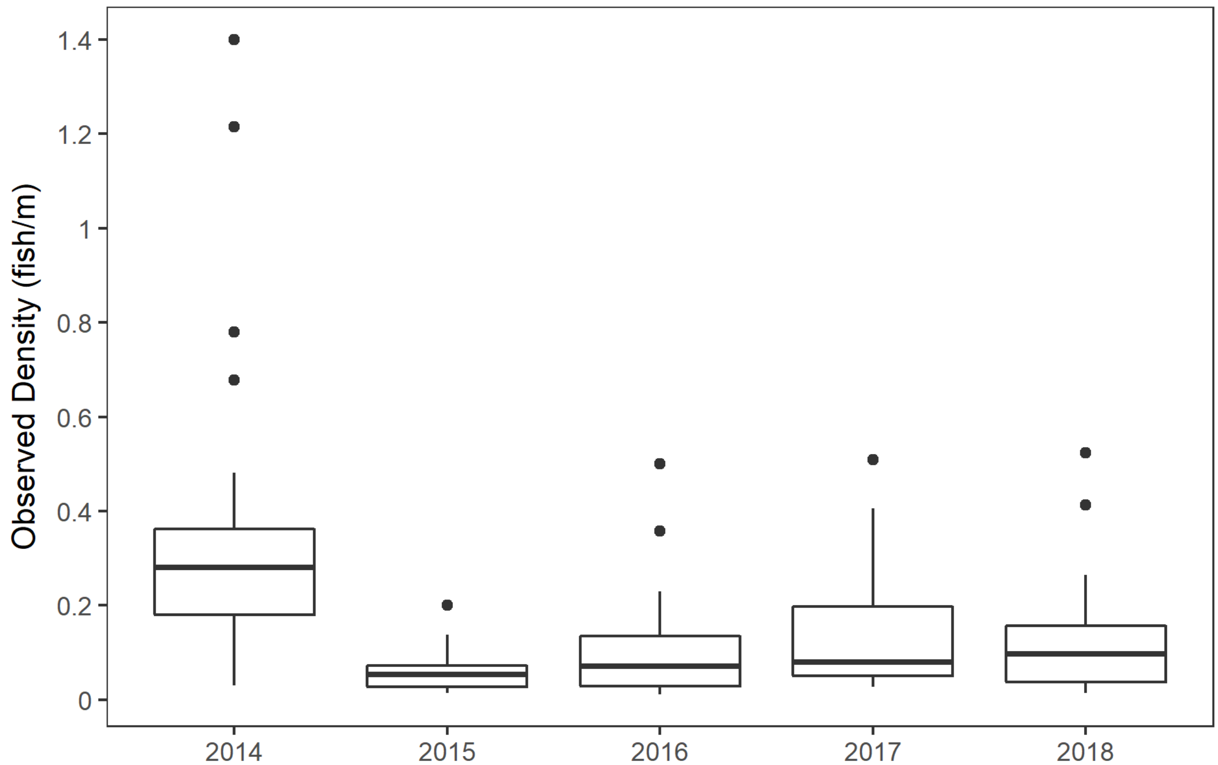


Figure 3-12. Density of age 0 Rainbow Trout (fish/m) directly calculated from shoreline electrofishing index sites (observed data) in the Seton River from 2014 to 2018.

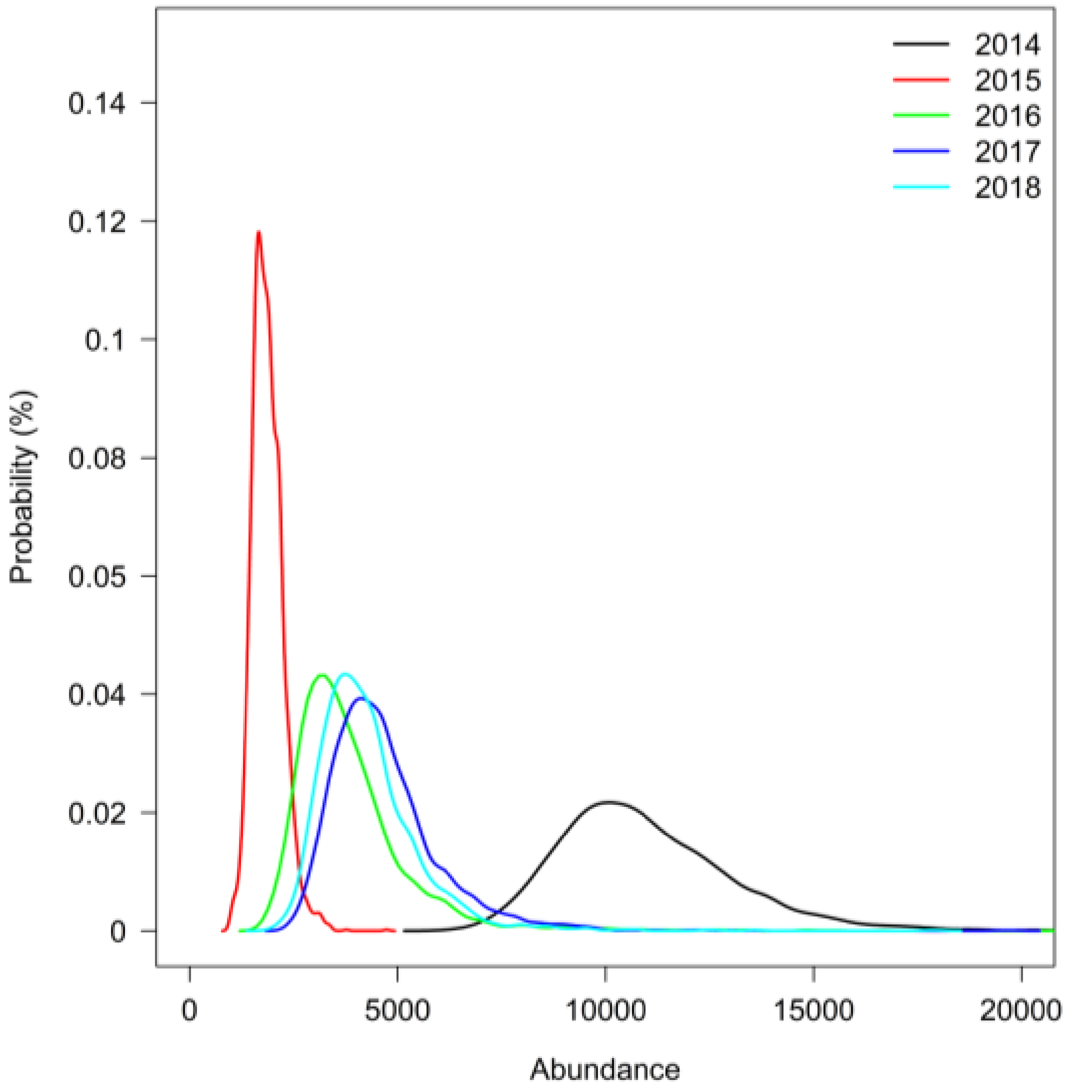


Figure 3-13. Posterior probability distributions for total abundance of age 0 Rainbow Trout in Seton River from 2014 to 2018.

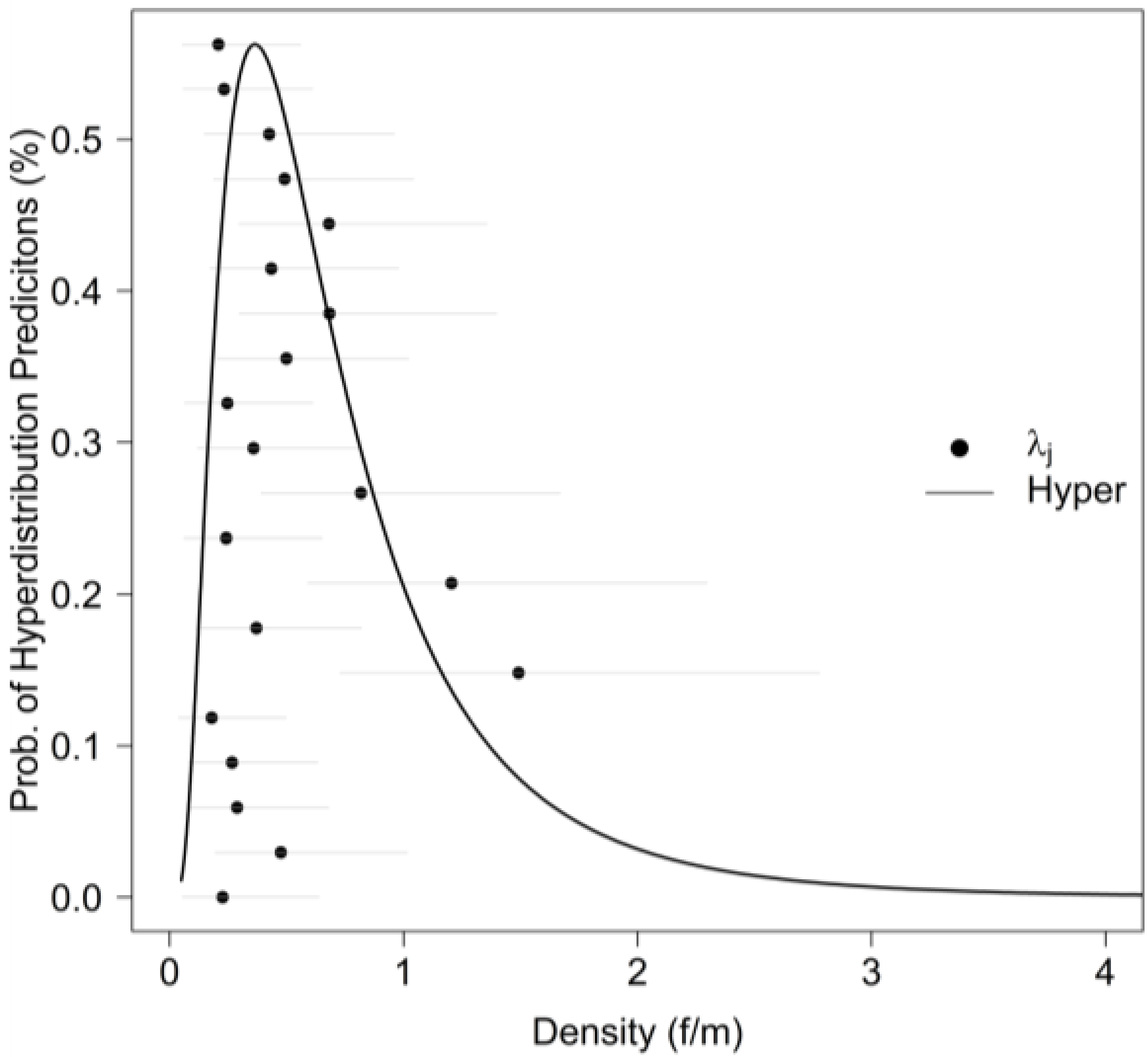


Figure 3-14. Estimates of fish density (fish/m) for age 0 Rainbow Trout in the Seton River in 2018. Filled points are the mean and 95% CI of individual index sites and the black line is the hyperdistribution based on the means of the hyperparameters estimated during the hierarchical Bayesian modeling. The vertical order of the site-specific estimates shows their position in the river from downstream to upstream and is unrelated to the numerical y-axis.

3.4.2 Snorkel Surveys

During March 2018, 20 snorkel index sites were surveyed, accounting for a total of 947 m (12 %) of shoreline (Table 3-9). The percentage of shoreline sampled has remained constant since 2016 (+/- 1%) but was double of what was sampled in 2015. Discharge at the WSC gauge (08ME003) was ~14.0 m^3/s during the March survey.

From 2015 to 2018 eleven species have been observed in total (Appendix 7-10), with Rainbow Trout being the most abundant (n = 394), followed by Coho (n = 129). Chinook were not observed in 2015 and 2016 but were in 2017 (n = 48) and 2018 (n = 22). A mark-recapture experiment was not attempted as sample sizes within each site were insufficient.

Table 3-9. Percentage of shoreline sampled during snorkel surveys at shoreline index sites in Seton River from 2015 to 2018.

Year	Shoreline Sampled (m)	Mean Site Length (m)	Percentage Sampled
2015	500	50	6%
2016	1015	48.3	13%
2017	950.5	47.5	12%
2018	947	47.4	12%

3.5 Telemetry

3.5.1 Adult Radio Telemetry

Radio tags were detected by fixed telemetry stations and through mobile tracking on the Seton and Lower Bridge Rivers. None of the 20 Steelhead Trout tagged at the Seton – Fraser confluence (via BRGMON-3) in 2018 were detected on the radio receiver located at the LSC confluence (1.42 km upstream of Seton-Fraser confluence; Figure 2-7) but this may have been because of a receiver malfunction that was resolved on May 27. Two Steelhead Trout (Codes 5 and 9) were detected through mobile radio tracking and PIT antennas at Seton Dam (Figure 2-7), passing Seton Dam on April 20 and 24, respectively. Code 9 was also detected through mobile tracking on May 23 in Seton Lake.

Of the other 18 Steelhead Trout tagged at the Seton – Fraser confluence in 2018, 16 moved into the Bridge River and were monitored under BRGMON-3, the other two may have continued upstream in the Fraser River to spawn elsewhere.

3.5.2 Passive Integrated Transponder (PIT) Telemetry

From April 4, 2013 to October 18, 2018, a total of 1109 Rainbow Trout were PIT tagged in the USC (n=215) and LSC (n=190) and mainstem Seton River (n=704).

With the LSC array only having one antenna in 2014, detection efficiency could not be calculated in that year. For other years, detection efficiency for the downstream antenna ranged from 0% in 2018 to 85% in 2017 and from 0% in 2016 to 82% in 2015 for the upstream antenna (see Appendix 7-11). From 2014-2018, 59 Rainbow Trout were detected on the LSC PIT array. As determination of fish direction is directly related to the efficiency of each individual PIT antenna, direction can only be confidently confirmed for 31 fish (Figure 3-15). The remaining 28 were either detected in 2014 when directionality could not be assigned, were not detected on both antennas, or moved between antennas, confusing the assignment of direction. Unfortunately, several challenges limited the effectiveness of PIT arrays in 2018. While fish were detected by each antenna individually, the detection efficiency was 0 and as such, direction cannot be assigned to any of the LSC detections.

In the spring of 2016, movement into the LSC for juvenile and adult fish was blocked by a temporary fish fence designed to capture out-migrating smolts. In the spring of 2017, the fence was re-installed but altered to allow adult fish passage through a tube. The ability for juvenile fish to pass through the tube was questionable, as flows were high and a potential barrier. Sampling of out-migrating juveniles switched to a modified Incline Plane Trap (IPT) in 2018 which allowed for free passage of adults and juveniles in and out of the spawning channel.

The USC array had two antennas for the entire monitoring period. Detection efficiency for the downstream antenna ranged from 40% in 2015 to 100% in 2017 and from 73% in 2018 to 92% in 2017 for the upstream antenna (Appendix 7-12). From 2015-2018, 80 Rainbow Trout were detected on the USC PIT array, of which direction can be confidently identified for 65 individuals (Figure 3-16). The remaining 15 detections could not be assigned a direction.

Analysis of movement data indicate that Rainbow Trout move in and out of spawning channels from mid-March to December. Movements do not appear to be associated with flow changes in the mainstem Seton River and in 2015 through 2018, there appeared to be a directed movement into spawning channels in the fall (Figure 3-15, Figure 3-16). This may indicate that juveniles overwinter in the spawning channels and suggests that fish from the Seton River mainstem and spawning channels are from one population. Corroborating this suggestion are ten fish that were detected on both the USC and LSC PIT arrays occurring from 2015-2018 and for both rearing and spawning purposes (detailed life history of each fish in Appendix 7-13). Generally, fish moving into the spawning channels were age 2 or older.

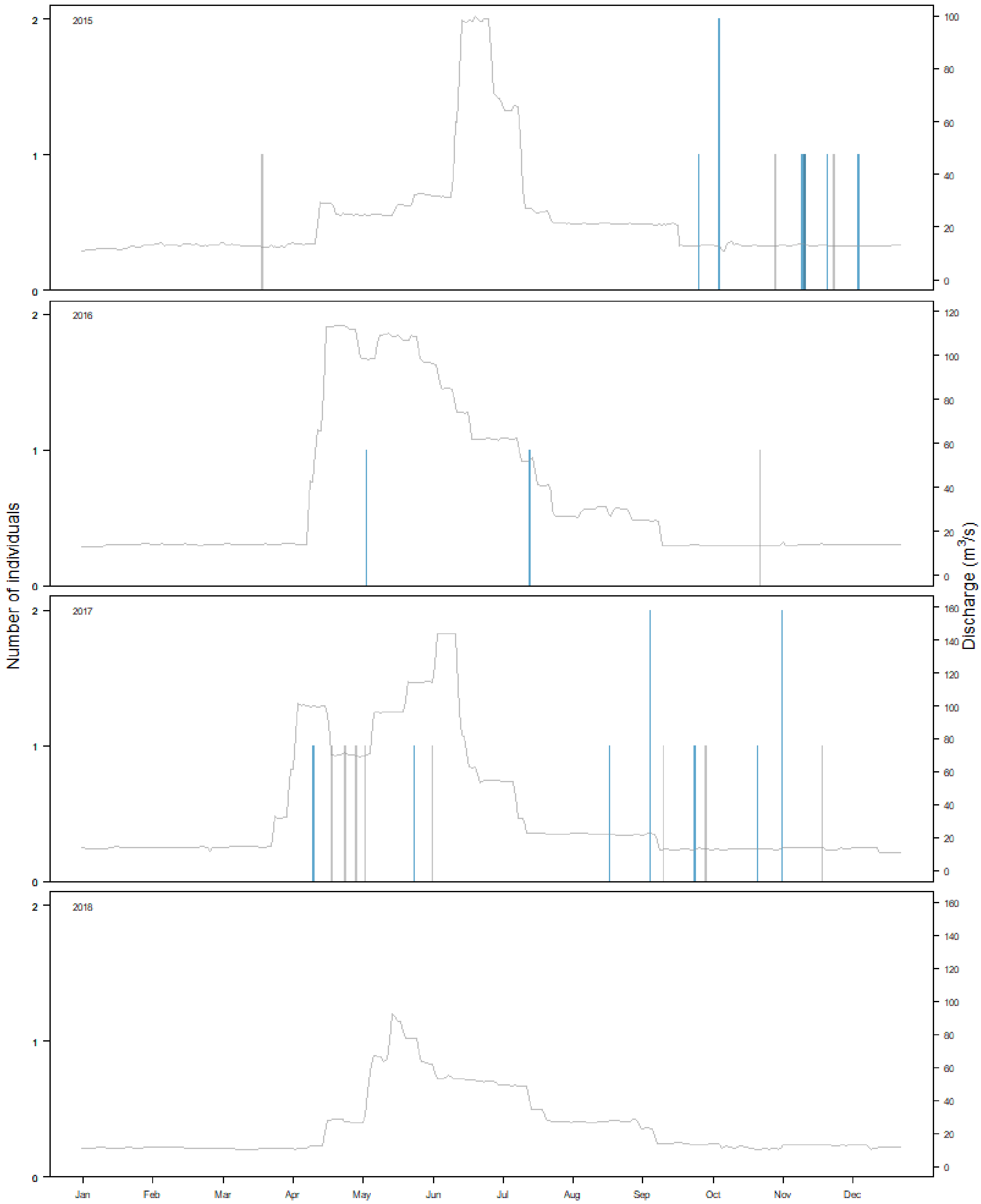


Figure 3-15. Rainbow Trout detections (for which direction of movement can be assigned) on the Lower Spawning Channel PIT array in 2015, 2016, 2017 and 2018 in relation to discharge. Movements into the spawning channel are shown in blue and movements out of the spawning channel are shown in grey. Movements could not be assigned in 2018 due to low detection efficiencies.

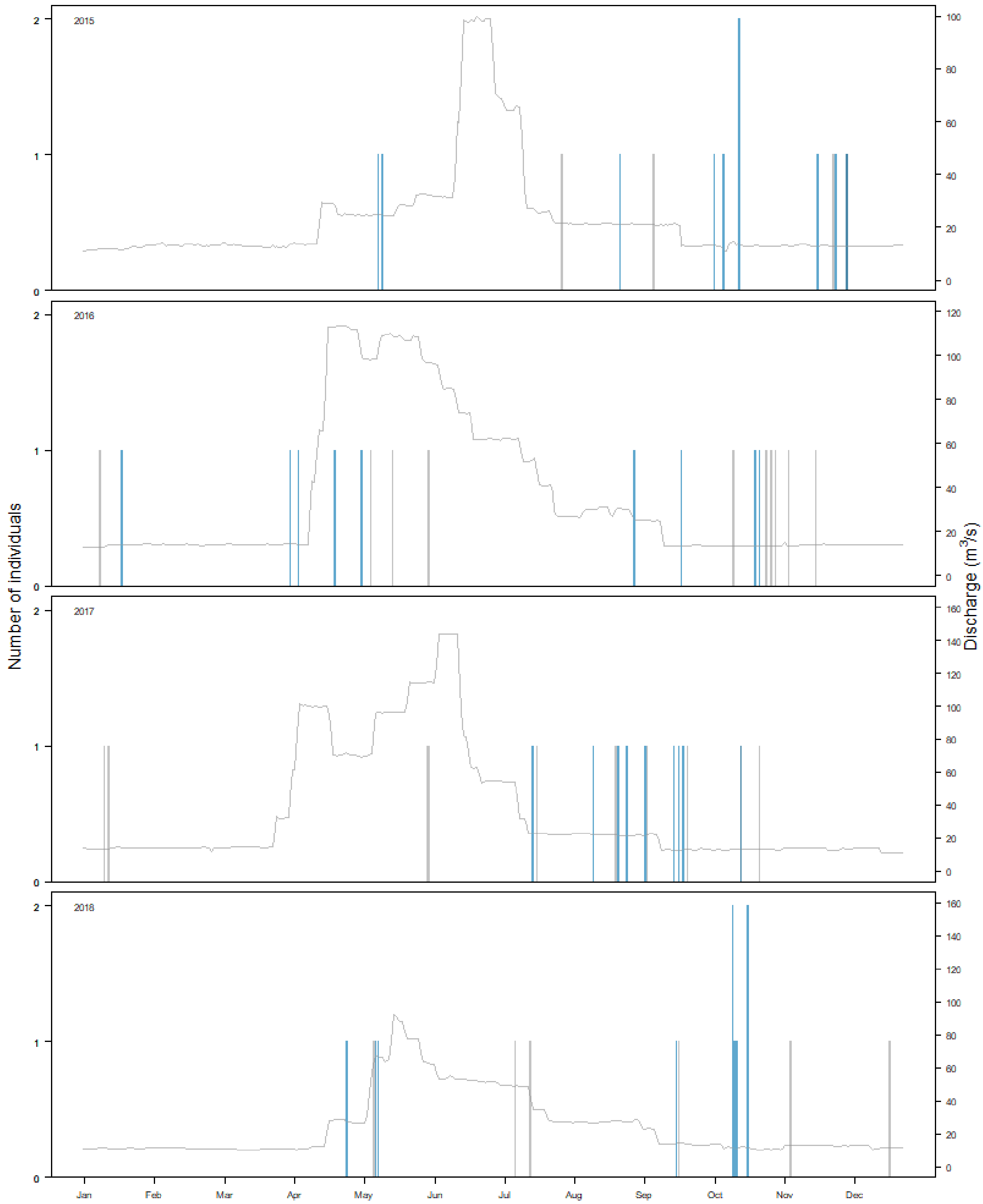


Figure 3-16. Daily Rainbow Trout detections (for which direction of movement can be assigned) on the Upper Spawning Channel PIT array in 2015, 2016, 2017 and 2018 in relation to discharge. Movements into the spawning channel are shown in blue and movements out of the spawning channel are shown in grey.

3.6 Adult Visual Counts

Observations of adult spawning salmonids have generally been low and variable among years and locations (Table 3-10). Only two Steelhead were observed in 2015, the only year Steelhead visual counts were conducted. Chinook Salmon were only observed in 2016 in Reach 1 and the LSC. In 2015, 2016, and 2018 Coho salmon were primarily observed in the spawning channels, but only in Reach 1 in 2017. Observed Coho numbers were higher in 2018 compared to 2015 and 2017 but lower than 2016. Pink salmon runs were present in the Seton River in 2015 and 2017, when adults were predominantly observed in the spawning channels.

Table 3-10. Streamwalk counts in 2015, 2016, 2017, and 2018 for Steelhead, Chinook Salmon, Coho Salmon and Pink Salmon.

	2015					2016					2017					2018				
Reach	**1**	**2**	**3**	**USC**	**LSC**	**1**	**2**	**3**	**USC**	**LSC**	**1**	**2**	**3**	**USC**	**LSC**	**1**	**2**	**3**	**USC**	**LSC**
Steelhead	1	0	0	0	1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Chinook	0	0	0	0	0	2	0	0	0	1	0	0	0	0	0	-	-	-	-	-
Coho	0	0	0	4	18	4	0	0	25	64	13	0	0	0	0	13	0	0	22	27
Pink	976	16	106	2577	2887	-	-	-	-	-	474	48	57	495	727	-	-	-	-	-

4.0 DISCUSSION

Objectives of this program are to monitor responses of fish habitat and fish populations to Seton Dam operations, and to identify potential indicators of the effects of the implemented Seton River hydrograph. The program was originally designed to monitor fish and fish habitat under WUP target flows. However, due to the modified operations at La Joie Dam, the Seton River hydrograph increased outside of these target flows, impacting monitoring activities at designated study sites. To ensure that management questions could still be addressed during periods when discharge from Seton Dam exceeded WUP targets, new '*Modified Operations*' monitoring methods were adopted in 2016 and have continued through to 2018. For example, bio-sampling during periods of modified operations focused on side-channel habitats and spawning channels rather than the mainstem, where many sites were inaccessible for sampling above 60 m^3/s. Conversely, sampling efforts to evaluate juvenile standing stock biomass were not affected by the increased Seton River hydrograph because the surveys are completed when flows return to WUP targets.

As the sixth year of a 10-year program, data collected in 2018 continues to build upon knowledge gained in previous years. Although preliminary synthesis analyses have identified some trends, continued monitoring is required to fully address each MQ. Herein, findings to date are discussed in the context of MQs. Many methods are not specific to a given MQ, and collected data therefore often informs multiple MQs. The first two MQs were addressed by conducting bio-sampling, visual count surveys, habitat surveys, and tagging. Under MQ1, data collection aims to understand basic biological characteristics of the rearing and spawning populations in Seton River. The intention of MQ2 is to determine how the Seton River hydrograph influences the hydraulic condition of juvenile fish rearing habitats. Streamwalks and redd surveys were conducted to address MQ3, which aims to evaluate potential risks of salmon and steelhead redds dewatering due to changes in the Seton River hydrograph. Riverbed topographic surveys were conducted to address MQ4, which asked how the Seton River hydrograph influences availably of gravel suitable for spawning. To determine potential effects of shutdowns at the Seton Generating Station on fish habitat in the Fraser River (MQ5), stranding surveys were conducted in the Fraser River.

MQ1: What are the basic biological characteristics of the rearing and spawning populations in Seton River in terms of relative abundance, distribution, and life history?

As no hypothesis fall under MQ1, there is no direct testing or conclusions to be drawn. Most data collected under BRGMON-9 contributes to the understanding of fish populations in the Seton River and will continue to do so each year.

Biological Characteristics of Rearing Populations in Seton River

Monthly juvenile bio-sampling surveys have identified 14 fish species in the Seton River, including seven salmonids (Coho, Chinook, Pink, Sockeye, Rainbow Trout, Bull Trout, and Whitefish). Rainbow Trout are the most prevalent during sampling, followed by Coho and Chinook Salmon, thus these three species are the focus of monitoring. Four age classes of Rainbow Trout (0-3) and three age classes of both Coho and Chinook (0-2) have been identified. Chinook juveniles have been tested for DNA since 2016 and results show that individuals are present from both Portage Creek (Seton River) origin and upstream tributaries of the Fraser River (e.g., Chilcotin, Quesnel) indicating that Seton River is important rearing habitat for these fish. No Bridge River origin Chinook have been observed using Seton River.

All juvenile Rainbow Trout > 75 mm captured through monthly bio-sampling surveys are given a 12 mm PIT tag, allowing movements and distribution to be tracked either through recapture events or on PIT arrays in the LSC, USC and Seton Dam. PIT detections (2015-2018) on the LSC and USC arrays indicate that the Rainbow Trout move between the spawning channels and the mainstem regardless of initial capture location with 10 Rainbow Trout detected moving between the two spawning channels. Movements between the mainstem and spawning channels suggest that distinct populations do not exist within each spawning channel. The timing of movements in and out of the spawning channels do not correlate with discharge events from Seton Dam but may be seasonal (e.g, over-wintering or spawning). Modifications to the PIT arrays to improve detection efficiency will improve the quality of PIT data and allow seasonal migrations of Rainbow Trout to be better observed.

Biological Characteristics of Spawning Populations in River

Currently, information regarding adult abundances is limited to inconsistent count data, precluding abilities to conduct any analyses. Estimating adult salmonid abundance has been difficult because very few fish have been observed, and during modified operation years, high discharge from Seton Dam reduces visibility in the spring.

Visual surveys for Steelhead Trout were only conducted in 2015 during high discharges and two individuals were observed. Visual surveys for Steelhead Trout were subsequently suspended indefinitely due to poor visibility. However, both PIT and radio telemetry allow for Steelhead Trout tagged under BRGMON-3 to be monitored as they migrate through the Seton River system. From 2015-2018, 11 Steelhead have been detected, most of which used the Seton Dam fishway and therefore it is assumed spawning occurs upstream of the Seton River.

The most frequently observed species are Coho Salmon and in odd number years (i.e., 2015, 2017), Pink Salmon. Both species are predominately observed in spawning channels and therefore their numbers relative to other species may be biased as it is easier to observe fish in these areas than in the mainstem Seton River. Coho Salmon numbers are still low and variable, ranging from 13 individuals in 2017 to 93 in 2016. When present, Pink salmon migrate in much higher numbers than other salmon species (6,562 in 2015 and 1,801 in 2017). In the mainstem, Coho and Pink Salmon are observed predominantly downstream of Seton Dam in reach 1. Visual tagging of Pink was attempted in 2015 and 2017 to assess observer efficiency and create AUC estimates. However, insufficient numbers were captured to release tags into the river thus all estimates should be considered and index of the relative abundance.

Historically, Seton River was not assessed to have a distinct population of Chinook salmon and any observed have been assumed to be migrating to Portage Creek or strays from other Fraser River tributaries. While adult Chinook have not specifically been tested for DNA, results from juvenile DNA samples suggest that at least some Portage Creek/Seton River fish are present in the system. Three Chinook salmon were observed during visual surveys in 2016 and none have been observed since.

MQ2: How does the proposed Seton hydrograph influence the hydraulic condition of juvenile fish rearing habitats in downstream of Seton Dam?

The primary monitoring activity to address MQ2 was habitat suitability assessments of juvenile rearing habitats, providing estimates of both habitat quality and quantity. However, monitoring activities also evaluated effects of flow to fish populations. Analyses of various metrics of juvenile fish growth were assessed as indicators of the in-season effects of high discharge, and trends in estimates of standing crop biomass over various years may also elucidate effects of flow. Across all sampling methods, fish captures have been dominated by Rainbow Trout, followed by Coho and Chinook Salmon, which are thus the focus when evaluating effects of flow.

Effects of Flow to Juvenile Fish Rearing Habitats

Habitat suitability surveys for juvenile salmonids have been completed for the mainstem river and side-channel habitats at a range of discharges (12 – 143 m^3/s, Buchanan et al., 2017). Mainstem surveys completed 2014-2016 show habitat availability decreases as dam discharge increases. High discharges as a result of modified operations wetted side-channel habitats making them available to juvenile fish. Though these additional habitats do buffer habitat changes in the mainstem, they do not adequately compensate for the loss of habitat. Results indicate that the amount of available habitat suitable for Rainbow Trout, Coho and Chinook juveniles varies with Seton Dam discharge and therefore we can reject H_1. Additional surveys are needed for Sent Dam releases below 40 m^3/s and between 60 and 100 m^3/s to determine where habitat is maximized for juvenile fish in the mainstem and side-channels, respectively.

To assess impacts of three years of high discharge conditions to juvenile rearing habitat as a result of modified operations, habitat suitability surveys were completed in the fall at 12 m^3/s WUP target flows in 2018. Surveys were completed at a subset of locations initially surveyed in 2014 at 12 m^3/s prior to modified operations. These surveys show that there has been changes in habitat suitability, and when compared to 2014 conditions, 2018 had a net loss in overall WUA for Rainbow Trout Fry and Parr, and Chinook juveniles but WUA for Coho juveniles increased slightly. These changes are inconsistent across reaches, with Reach 1 exhibiting a net increase for all species, and Reach 2, a net decrease. These inconsistencies are likely a factor of the varying discharge conditions in each reach; conditions in Reach 1 are solely impacted by flow releases from Seton Dam and Reach 2 and 3 are also influenced by Cayoosh River which, while regulated, can be seasonally unpredictable with maximum discharge from 2014 to 2018 ranging from 53.9 m^3/s (2014) to 90.2 m^3/s (2017). While surveys only represent a subsample of the entire river, they represent long-term changes that may be occurring within each reach as a result of the modified operations of Seton Dam.

Effects of flow to juvenile fish populations

Standing crop surveys have been conducted annually since 2014 at base flow conditions (i.e., September) and therefore have been unimpacted by modified operations. Sufficient data is only available to provide estimates for age 0 Rainbow Trout, which has ranged from 2,485 (2015) to 11,157 (2014) individuals. To date, a relationship between standing crop and discharge for Rainbow Trout has not been identified as the data set is limited by only one year of sampling during the WUP target hydrograph. This prevents comparative analyses. Until more data are collected during years with the WUP target hydrograph, it is impossible to know whether 2014 was anomalous or indicative of Rainbow Trout abundance under the target flow regime.

As it stands, 2014 had the lowest maximum discharge and the highest overall abundance of Rainbow Trout. Conversely, max discharges from 2015 to 2018 exceeded WUP targets and abundance numbers for Rainbow Trout were considerably lower, indicating there may be a link between discharge and Rainbow Trout abundance. The reduction in Rainbow Trout numbers in subsequent years may be due to the timing of high flows (i.e., during emergence). Monthly bio-sampling data suggest that Rainbow Trout fry emerge from their redds in late-June or early July. If discharge from Seton Dam is high during this time, fry may be flushed into the Fraser or displaced from suitable habitat. Another explanation for low Rainbow Trout abundances is that the Seton River discharge from 2015 to 2018 may have crossed a threshold value (i.e., 68.6 m^3/s – 2014 maximum discharge), above which habitat is too limited to support a greater population of juvenile Rainbow Trout. Habitat suitability surveys indicate that the amount of habitat available to Rainbow Trout fry decreases considerably above 60 m^3/s meaning that emergent fry are rearing in less suitable habitat, potentially impacting survival.

To further investigate if flows have an effect on juvenile populations, sampling effort was increased in the spawning channels in 2018 with the idea that if two distinct populations (mainstem vs spawning channel) of Rainbow Trout existed, the fish sampled in spawning channels would be unimpacted by modified operations and thus serve as a control. However, location as a model factor yielded inconsistent results and PIT data also indicates that Rainbow Trout move between the spawning channels and the mainstem. Therefore, location is likely not representative of the flow conditions experienced by this species. However, location may provide a good proxy for flow condition for Coho juveniles, which have been shown to exhibit higher site fidelity in summer and winter months

(Weybright and Giannico 2017). A mark-recapture study is recommended to determine if Coho juveniles to determine if these individuals are moving between the mainstem and the spawning channels.

Condition of fish is examined through monthly bio-sampling surveys. With lower densities of fish observed since 2015, it could be expected that body condition may have increased as fewer fish lead to less competition for resources. Seeing an increase in body condition would indicate that there is a density dependent factor driving the size of Rainbow Trout in Seton River. However, no trends in the data are visible to date. This may be due to variability of flow within the modified operations from year to year. It is unknown whether results would be the same if the hydrograph remained the same from year to year.

Additional monitoring during years with the WUP target hydrograph will be needed to properly assess the effects of the WUP and modified operation hydrographs on juvenile fish populations and effectively answer MQ2. In the interim, each year of modified operations data collection should be considered baseline data that contributes to a long-term biological data set. These data will be able to provide valuable comparisons and aid in management decisions regarding the best hydrograph for juvenile salmonids and inform the effects of potential future discharges above the WUP target hydrograph.

MQ3: What is the potential risk for salmon and Steelhead redds, dewatering due to changes in flow between spawning and incubation periods imposed by the Seton hydrograph?

Spawning habitat for all species is limited in the Seton mainstem and can be attributed to the relatively restricted nature of the river that has been extensively dyked or armored throughout. This creates higher velocities in the river and few areas for substrate to be deposited. Visual surveys of spawning Steelhead and Salmon have identified two areas in the mainstem Seton River where spawning occurs; immediately below Seton Dam and at the outflow of the LSC. To date, no redd dewatering has been observed as a result of changes in flow imposed by the Seton hydrograph as both identified spawning areas remain wetted at all flows. H_2 can therefore be accepted at this time.

During periods of modified operations, side-channels become wetted during the Steelhead migration and spawning period. If redds were present in the side-channels, they would be at risk of becoming dewatered if the Seton Dam hydrograph returned to WUP targets prior to emergence. However, habitat surveys in 2017 indicate that the substrate in the side-channels is unsuitable for spawning Steelhead and therefore the potential risk of redd dewatering in side-channel habitats is deemed low.

MQ4: How will the Seton hydrograph influence the short term and long-term availability of gravel suitable for use by anadromous and resident species for spawning and egg incubation?

Periods of high discharge as a result of modified operations have the potential to impact substrate availability in Seton River as higher velocity flows are known to mobilize gravel. Riverbed topographic surveys are generally completed every other year to monitor changes in streambed elevation and substrate downstream of Seton Dam. A detailed topographic survey was not completed in 2018 but results from 2013, 2015, 2016 and 2017 indicate that changes in elevation have occurred though results are variable. This may be due to the variable high discharges being released from the dam. Studies have shown that there is a threshold that needs to be reached before substrate is mobilized. For example, in the Bridge River, studies completed by KWL in 2017 (Ellis et al. 2018) found that discharges between 20 and 50 m^3/s are required to keep spawning gravel in the system, but that this may vary depending on channel characteristics and substrate composition. Prior to modified operations (2013-2015) scouring was observed but deposition was observed in both 2016 and 2017 following high discharge events as a result of modified operations. Results suggest that movement of smaller substrate from the top section downstream may be occurring (reject H_3). The next detailed topographic survey is due to be completed in 2019 and will continue to inform these inferences regarding the influence of the Seton hydrograph on spawning gravel over the long-term.

Substrate suitability surveys were added in 2018 at standing stock assessment sites to assess relationships between the Seton hydrograph and the short- and long-term availability of gravel suitable for anadromous and resident fish species throughout Seton River. Changes in substrate type from 2014 to 2018 are difficult to quantify due to the qualitative nature of the data collected in 2014. Coarse analyses do suggest a trend towards larger substrate size in 2018 relative to 2014. However, no trend exists with respect to distance from Seton dam as would be expected if changes were attributed solely to dam operations. In 2018, methods were improved upon and pebble counts were conducted at each transect, a more suitable method for comparing substrate size between years. Both pebble counts and substrate classification will be continued for the rest of the monitor.

MQ5: Does discharge from Seton Generating Station impact fish habitat in the Fraser River above and beyond natural variation in Fraser River Discharge.

Stranding risk was assessed at two sites in the Fraser River approximately 2 km and 11 km downstream from the Seton Generating Station from 2015-2017. A total of three shutdowns were monitored. The area dewatered on each shutdown was largely dependent on Fraser River discharge at the time of shutdown and although ramping rates exceeded the 5 cm/h recommended by DFO, only 5 individuals were observed stranded. As such, stranding risk was assessed to be low for these sites and monitoring discontinued. An addendum to the BRGMON-9 program was put forth in 2018 to address stranding concerns further downstream. This monitoring is conducted by a different organization and reported separately (BC Hydro, 2018).

5.0 RECOMMENDATIONS

The following recommendations are suggested to inform the management questions and address data gaps:

1. To test if Coho juveniles show high site fidelity, effort should be made to PIT tag fish and determine if the same fish are recaptured in the same sites throughout the season. Smaller fish could be marked with VIE or a small fin clip until they are large enough for a 12mm PIT tag. If Coho show strong site fidelity the spawning channels could be used as a control for flow scenario for this species. Differential growth and body condition of Coho in spawning channels vs. mainstem would inform high discharge effects/operational changes.
2. Weighted Useable Area has been calculated for Seton River at 12, 25, 60 m^3/s. An additional partial estimate is available at 100 m^3/s. To determine the best flow for juvenile salmonids, an additional habitat survey should be done between 25 and 60 m^3/s (for mainstem) and between 60 and 100 m^3/s (for mainstem and side-channels). For river-wide estimates to occur, flows from Seton Dam must be held at target flows for approximately 2 weeks.
3. Low read ranges on the LSC PIT antenna are the result of interference from the seasonally run counter. The PIT array should be moved further upstream to avoid interference which may require alterations to the power supply (i.e., switch from mains power to a solar powered battery bank).

4. Seton-specific habitat suitability curves have recently been found by BCH. Full comparison of the two sets of curves and how they were developed should be done to determine which is most appropriate to used moving forward for this study.
5. Monitoring should continue through the period of '*Modified Operations'* to determine impacts of the Seton River hydrograph on fish and fish habitats downstream of Seton Dam. Continuous, long-term data sets will be needed to make comparisons between discharges during modified operations and the WUP target hydrograph when upstream flow management and reservoir storage issues are resolved. return to normal in the future.

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7.0 APPENDIX

Appendix 7-1. Summary of juvenile salmonid weighted useable area (WUA; m^2) estimated in 2014 and 2018 at 12 m^3/s. WUA estimates are provided for each species/life stage in each reach in addition to the percent net change in WUA.

		WUA (m^2)											
		RB Fry			RB Parr			Coho			Chinook		
Reach	**Site**	**2014**	**2018**	**%Δ**	**2014**	**2018**	**%Δ**	**2014**	**2018**	**%Δ**	**2014**	**2018**	**%Δ**
1	G2B	3	79	25.3	0	67	NA	206	346	0.7	248	206	0.0
	G2C	73	118	0.6	276	209	-0.2	100	172	0.7	494	400	-0.2
	P3BR5A	89	144	0.6	155	140	-0.1	302	245	-0.2	271	234	-0.1
	R2CG2A	248	162	-0.3	177	251	0.4	174	357	1.1	227	413	0.8
	R3B	175	335	0.9	43	372	7.7	157	212	0.4	69	578	7.4
	SC2B	81	102	0.3	29	15	-0.5	82	109	0.3	57	52	-0.1
	total	**668**	**940**	**41**	**680**	**1054**	**55**	**1021**	**1442**	**41**	**1367**	**1883**	**38**
2	G5B	926	449	-0.5	804	335	-0.6	796	509	-0.4	1259	537	-0.6
	G6C	1687	1237	-0.3	1855	150	-0.9	1080	1858	0.7	2793	308	-0.9
	LG3B	200	81	-0.6	345	60	-0.8	401	288	-0.3	584	116	-0.8
	P4BG7A	277	185	-0.3	326	170	-0.5	614	258	-0.6	600	336	-0.4
	R6B	460	231	-0.5	387	162	-0.6	321	230	-0.3	612	224	-0.6
	total	**3550**	**2183**	**-39**	**3718**	**878**	**-76**	**3212**	**3143**	**-2**	**3848**	**1520**	**-74**
3	G10BR11A	105	191	0.8	769	522	-0.3	222	213	0.0	666	701	0.1
	G11ER12A	292	508	0.7	305	221	-0.3	223	419	0.9	760	403	-0.5
	G8B	7	373	52.3	0	307	NA	79	337	3.3	208	645	2.1
	G8DP6A	347	80	-0.8	287	139	-0.5	462	109	-0.8	509	194	-0.6
	G9BR10A	454	197	-0.6	324	66	-0.8	477	419	-0.1	587	208	-0.6
	LG7A2	5	17	2.4	0	108	NA	105	48	-0.5	279	340	0.2
	LG7B	110	75	-0.3	439	254	-0.4	228	319	0.4	721	603	-0.2
	R10DG10A	250	225	-0.1	317	81	-0.7	252	152	-0.4	412	150	-0.6
	R11EG11A	170	257	0.5	228	172	-0.2	140	256	0.8	435	276	-0.4
	R12F	85	146	0.7	10	360	35.0	150	168	0.1	1120	517	-0.5
	total	**1825**	**2070**	**13**	**2680**	**2229**	**-17**	**2339**	**2439**	**4**	**5696**	**4037**	**-29**
RIVER TOTAL		**6044**	**5192**	**-14**	**7078**	**4161**	**-41**	**6579**	**7024**	**7**	**12911**	**7440**	**-42**

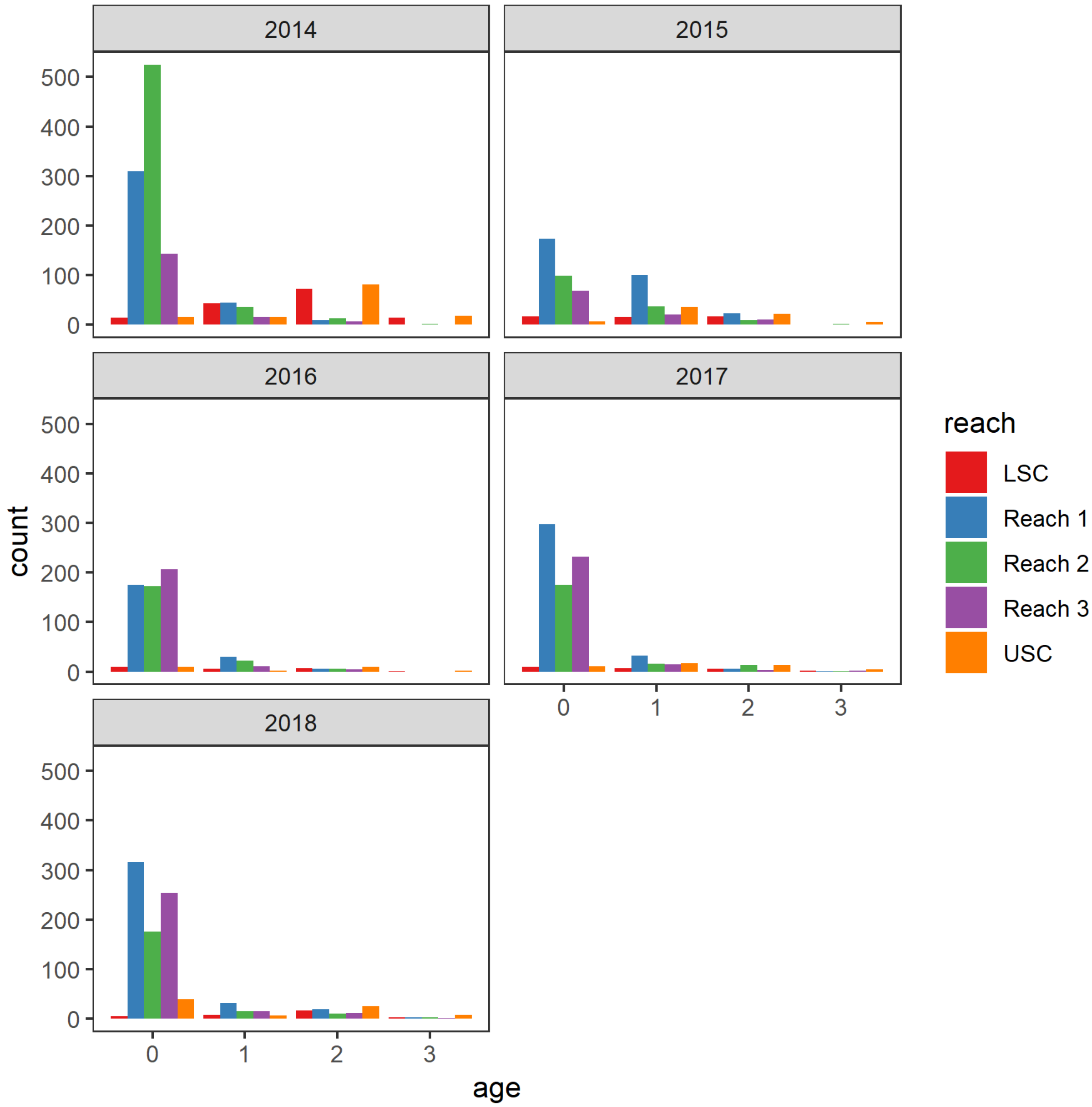
Appendix 7-2. Summary of weighted useable area (WUA; m^2) for Coho and Chinook spawning estimated in 2014 and 2018 at 12 m^3/s (fall spawning timing). Sites in red are in known spawning locations.

		WUA (m^2)					
		Coho			Chinook		
Reach	**Site**	**2014**	**2018**	**%Δ**	**2014**	**2018**	**%Δ**
1	G1B	1073	468	-0.6	737	346	-0.5
	G1D	768	293	-0.6	329	173	-0.5
	G1F	1004	648	-0.4	314	432	0.4
	G2B	0	57	NA	0	11	NA
	G2C	246	121	-0.5	355	209	-0.4
	P3BR5A	0	21	NA	31	37	0.2
	R2CG2A	0	199	NA	92	164	0.8
	R3B	33	393	10.9	23	190	7.3
	SC2B	0	2	NA	3	0	-1.0
	total	**3124**	**2201**	**-30**	**1885**	**1562**	**-17**
2	G5B	635	128	-0.8	912	191	-0.8
	G6C	0	1	NA	145	0	-1.0
	LG3B	0	0	NA	103	0	-1.0
	R6B	0	27	NA	52	28	-0.5
	P4BG7A	0	110	NA	47	92	1.0
	total	**635**	**156**	**-75**	**1258**	**219**	**-83**
3	G10BR11A	0	668	NA	533	1086	1.0
	G11ER12A	530	105	-0.8	267	52	-0.8
	G8B	0	198	NA	0	112	NA
	G8DP6A	443	95	-0.8	147	81	-0.4
	G9BR10A	259	317	0.2	198	225	0.1
	LG7A2	0	407	NA	0	280	NA
	LG7B	849	240	-0.7	626	268	-0.6
	R10DG10A	0	2	NA	97	0	-1.0
	R11EG11A	180	0	-1.0	291	78	-0.7
	R12F	253	388	0.5	184	448	1.4
	total	**2513**	**2531**	**1**	**2345**	**2721**	**16**
RIVER TOTAL		**6273**	**4887**	**-22**	**5488**	**4502**	**-18**
Known Spawning Location Total		**3694**	**3755**	**2**	**2539**	**3124**	**23**

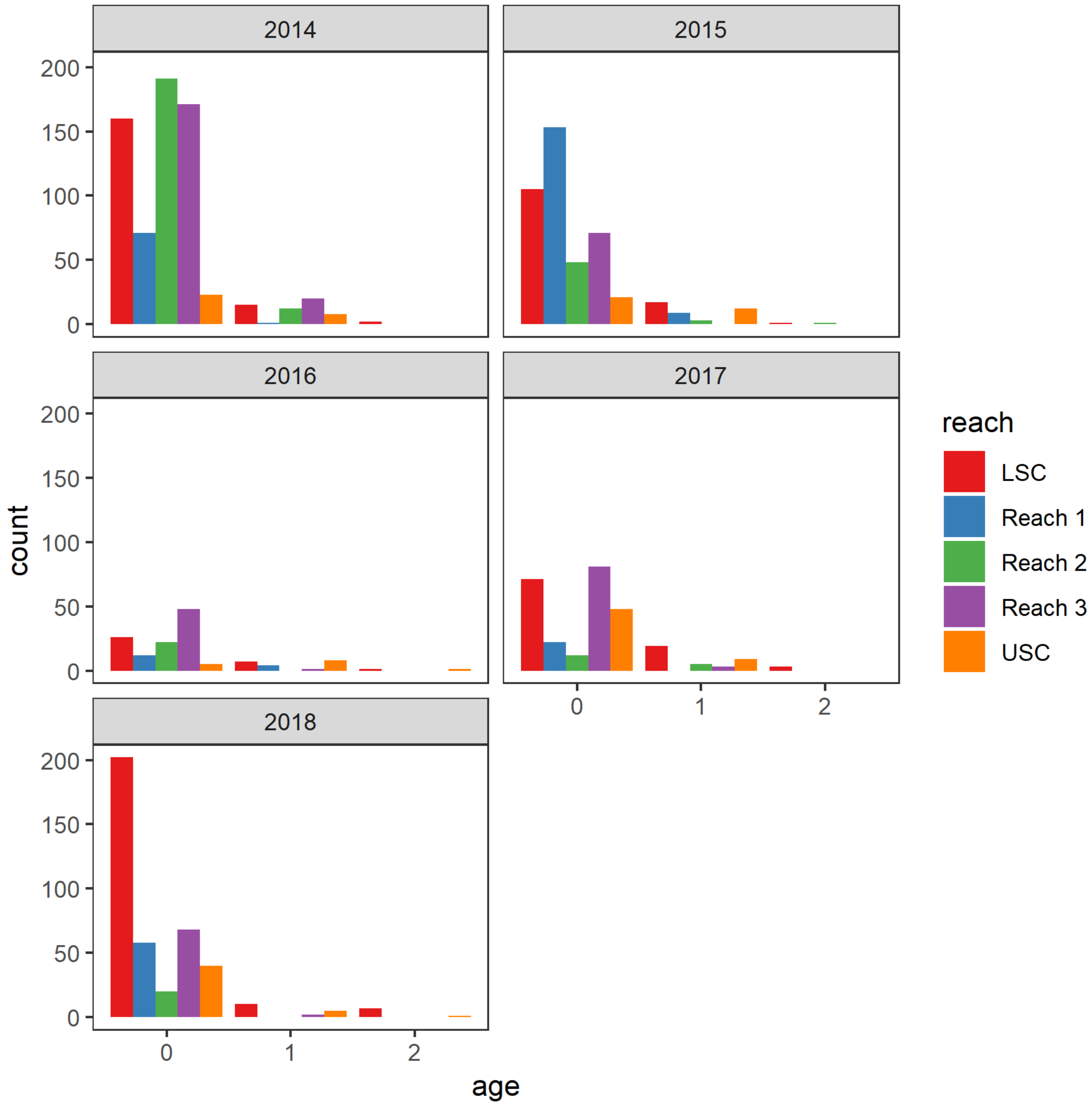
Appendix 7-3. Dominant and sub-dominant substrate size in 2014 and 2018 for the randomly selected standing crop sites selected for survey in 2018. Substrate classes, from largest to smallest, are as follows: B – Boulder, LC – Large Cobble, SC – small cobble, LG – Large Gravel, SG – Small Gravel, F – fine material.

	2014		2018	
Site	**Dominant**	**Sub-dominant**	**Dominant**	**Sub-Dominant**
G1B	LG	SG	LC	SC
G1D	LG	SG	LC	SC
G1F	LG	SG	SC	LG/LC
R2C/G2A	B	LC	B	LC
G2B	F	LG	SC	LC
G2C	LC	LG/SC	LC	SC
R3B	LC	B	SC	LC
SC2B	B	SC	LC	B
P3B/R5A	B	LC	B	LC
LG3B	B	LC	B	LC
R6B	B	LC	B	LC
G5B	LC	B	B	LC
G6C	B	LC	LC	SC
P4B/G7A	B	LC	LC	B
G8B	F	LC	LC	B
G8D/P6A	SC	B	B	LC
G9B/R10A	LC	B	LC	B
R10D/G10A	B	LC	B	LC
LG7A2	F	B	LG	SC
LG7B	SC	LC	LC	SC
G10B/R11A	SC	LC	LC	SC
R11E/G11A	LC	B	B	LC
G11E/R12A	LC	B	LC	B
LP6/R12A	F	LC	SC	LG
R12F	SG	SC	LC	SC

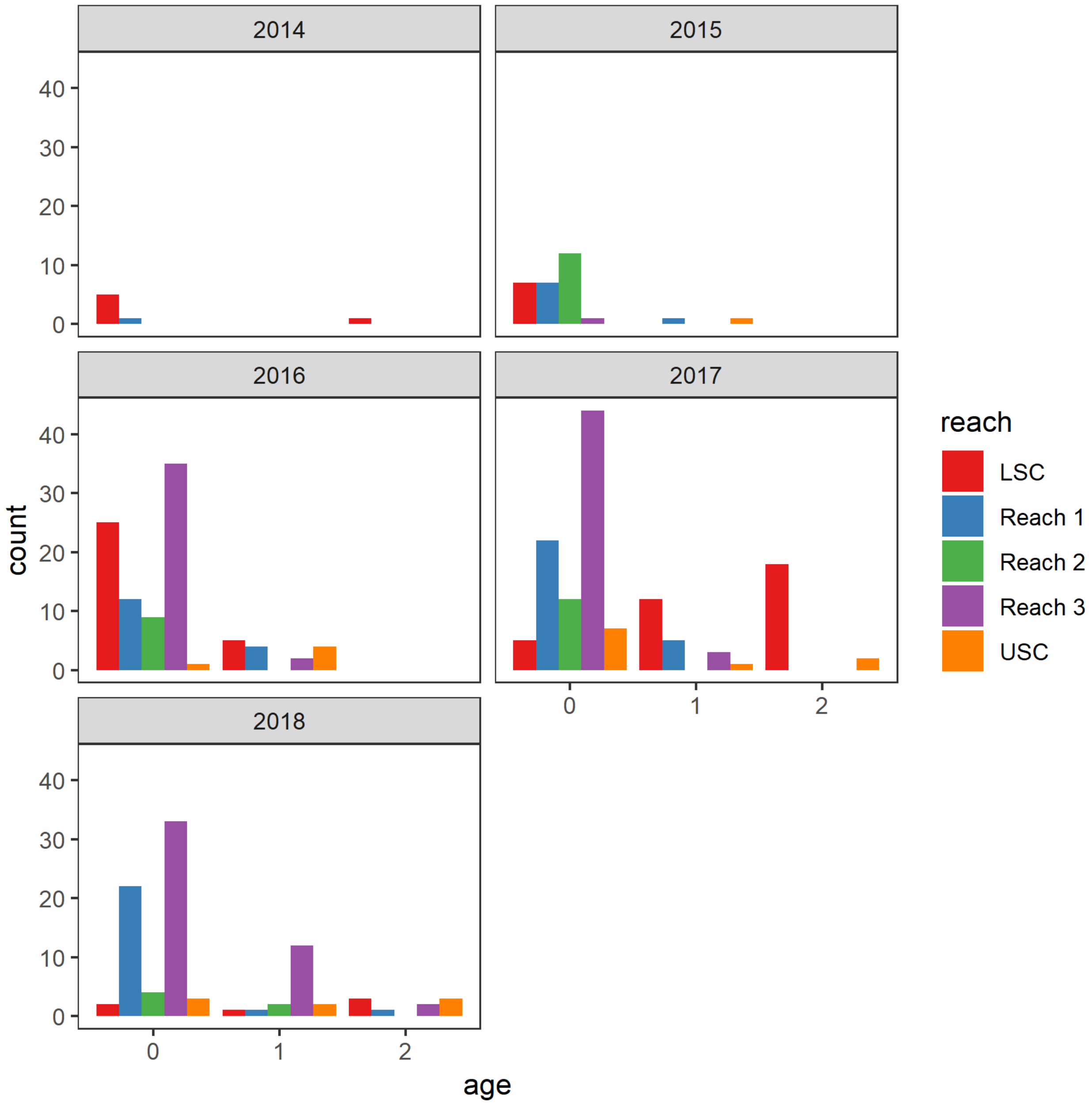
Appendix 7-4. Captures of Rainbow Trout in the Seton River mainstem and spawning channels from 2014 to 2018 separated by age and location of capture.



Appendix 7-5. Captures of Coho Salmon in the Seton River mainstem and spawning channels from 2014 to 2018 separated by age and location of capture.



Appendix 7-6. Captures of Chinook Salmon in the Seton River mainstem and spawning channels from 2014 to 2018 separated by age and location of capture.



Appendix 7-7. AIC model selection results for body condition modelling in the Seton River for Rainbow Trout, Coho Salmon, and Chinook Salmon. Bold values represent the best fit model (lowest AIC and fewest model parameters).

	RB Age 0		RB Age 1		CO Age 0		CHK Age 0	
Model	**AIC**	**ΔAIC**	**AIC**	**ΔAIC**	**AIC**	**ΔAIC**	**AIC**	**ΔAIC**
K ~ 1	1128.78	28.66	**-277.70**	**0.26**	1811.39	224.38	**128.48**	**0.00**
K ~ year	1129.99	29.87	-277.96	0.00	1662.49	75.48	131.03	2.55
K ~ reach	**1100.27**	**0.15**	-274.05	3.91	1801.95	214.94	130.73	2.25
K ~ year + reach	1102.46	2.34	-274.34	3.62	1650.03	63.02	131.87	3.39
K ~ year*reach	1100.12	0.00	-260.23	17.73	**1587.01**	**0.00**	145.06	16.58

Appendix 7-8. AIC model selection for length vs weight modelling of Rainbow Trout, Coho Salmon, and Chinook Salmon in the Seton River from 2014-2018 (all age classes combined). Bold values represent the best-fit model (lowest AIC and fewest model parameters).

	Rainbow Trout		Coho Salmon		Chinook Salmon	
Model	**AIC**	**ΔAIC**	**AIC**	**ΔAIC**	**AIC**	**ΔAIC**
Log(W) ~ Log(L)	-740.07	78.17	883.07	391.36	**-216.37**	**1.80**
Log(W) ~ Log(L) + reach	-760.23	58.01	878.77	387.06	-211.95	6.23
Log(W) ~ Log(L) + year	-745.73	72.51	749.00	257.29	-216.63	1.54
Log(W) ~ Log(L) + reach + year	-762.17	56.07	747.53	255.81	-212.57	5.60
Log(W) ~ Log(L)*year	-764.91	53.33	567.23	75.51	-213.30	4.88
Log(W) ~ Log(L)*year + reach	-788.40	29.84	565.66	73.94	-209.11	9.07
Log(W) ~ Log(L)*reach	-772.39	45.85	854.77	363.06	-210.56	7.61
Log(W) ~ Log(L)*reach + year	-774.91	43.33	727.20	235.49	-212.66	5.51
Log(W) ~ Log(L) + year*reach	-775.12	43.12	716.49	224.77	-210.57	7.61
Log(W) ~ Log(L)*year*reach	**-818.24**	**0.00**	**491.72**	**0.00**	-218.18	0.00

Appendix 7-9. DNA results for Chinook Salmon juveniles caught in Seton River in 2016, 2017 and 2018.

	Number of Chinook		
Stock Origin	**2016**	**2017**	**2018**
Chilko River	10	2	2
Cottonwood River	2	0	0
Deadman Creek	0	1	0
Fontoniko Creek	4	0	0
Indianpoint Creek	0	0	1
Cariboo River	0	1	0
Chilcotin River	2	0	0
Nechako River	13	3	0
Portage (Seton)	26	41	32
Quesnel River	14	0	11
Salmon River	2	0	2
Slim Creek	6	1	0
Stuart River	13	11	6
Willow River	1	0	0

Appendix 7-10. Mean and standard deviation (SD) of fork lengths (FL; mm) and sample sizes (n) of fish species observed during snorkel surveys from 2015 to 2018.

	2015			2016			2017			2018		
Species	n	FL		n	FL		n	FL		n	FL	
		Mean	SD		Mean	SD		Mean	SD		Mean	SD
Bridgelip Sucker	-	-	-	1	130	-	-	-	-	-	-	-
Bull Trout	1	180	-	1	175	-	-	-	-	-	-	-
Chinook	-	-	-	-	-	-	48	84.2	11.8	22	100.5	5.8
Coho	27	-	-	42	90.1	14.8	26	85	15.8	34	89.1	16.8
Pink										7	30	0
Sculpin	-	-	-	4	127.5	84.2	7	72.9	12.5	2	75	35.4
Lamprey	-	-	-	1	110					-	-	-
Mountain Whitefish	1	-	-	16	243.1	75.3	13	169.2	73.3	1	110	-
Rainbow Trout	102	88.7	27.1	129	121.5	42.7	90	102.7	39.7	73	118.1	42.6
Redsided shiner	8	85	0	-	-	-	-	-	-	1	150	-
Steelhead	-	-	-	1	600		-	-	-	-	-	-

Appendix 7-11. Detection efficiency calculated using PITR package for each of the Lower Spawning Channel PIT antennas (downstream = antenna 1, upstream = antenna 2), summarized by year.

Year	Antenna	Detection efficiency	Shared detections	Detections on array	Detections not on array	Missed detections
2015	1	0.82	9	11	11	2
2015	2	0.82	9	11	11	2
2016	1	0.14	1	3	7	6
2016	2	0.33	1	7	3	2
2017	1	0.85	11	18	13	2
2017	2	0.61	11	13	18	7
2018	1	0	0	4	3	3
2018	2	0	0	3	4	4

Appendix 7-12. Detection efficiency calculated using PITR package for each of the Upper Spawning Channel PIT antennas (downstream = antenna 1, upstream = antenna 2), summarized by year.

Year	Antenna	Detection efficiency	Shared detections	Detections on array	Detections not on array	Missed detections
2015	1	0.40	12	13	30	18
2015	2	0.92	12	30	13	1
2016	1	0.88	15	19	17	2
2016	2	0.79	15	17	19	4
2017	1	1.00	14	17	14	0
2017	2	0.82	14	14	17	3
2018	1	0.89	16	22	18	2
2018	2	0.73	16	18	22	6

Appendix 7-13. Detections for the 10 Rainbow Trout that moved between the two Spawning Channels in which the direction of movement can be confidently assigned.

Date of Detection	PIT code	Tag Date	Age at Tag	FL	Age at Detection	Antenna	Direction	Purpose
03-Oct-15	586038	Jun 2014	2	118	3	LSC	in	rearing
09-Oct-15	586038	Jun 2014	2	118	3	USC	in	rearing
01-Dec-15	657744	Jul 2015	1	160	2	USC	out	rearing
02-Dec-15	657744	Jul 2015	1	160	2	LSC	out	rearing
22-Apr-16	586036	Apr 2014	2	103	4	USC	in	spawning
07-May-16	586036	Apr 2014	2	103	4	LSC	in	spawning
18-May-16	586036	Apr 2014	2	103	4	USC	out	spawning
17-Oct-16	657061	Sep 2015	1	78	2	USC	out	rearing
30-Oct-16	657061	Sep 2015	1	78	2	LSC	out	rearing
27-Aug-17	656873	Mar 2017	NA	88	NA	USC	In	rearing
08-Sep-17	656806	Mar 2017	2	101	2	USC	in	rearing
11-Sep-17	656806	Mar 2017	2	101	2	LSC	in	rearing
27-Sep-17	657877	Sep 2016	0	76	1	USC	in	rearing
25-Oct-17	656873	Mar 2017	NA	88	NA	LSC	In	rearing
21-Oct-17	734906	Oct 2016	0	87	1	USC	out	rearing
09-Nov-17	734906	Oct 2016	0	87	1	LSC	in	rearing
09-Nov-17	657877	Sep 2016	0	76	1	LSC	in	rearing
27-Nov-17	734906	Oct 2016	0	87	1	LSC	out	rearing
17-Oct-18	656876	Mar 2017	NA	72	NA	USC	In	rearing
26-Dec-18	656876	Mar 2017	NA	72	NA	USC	In	rearing