Bridge River Project Water Use Plan

Seton River Habitat and Fish Monitoring

Implementation Year 9

Reference: BRGMON-9

Study Period: January 2021 – December 2021

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Implementation Year 9 (2021)

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Executive Summary

The Seton River Habitat and Fish Monitoring Project (BRGMON-9) monitors coincident habitat and fish population data (e.g., abundance, growth) in the Seton River to evaluate of the effects of flow releases from Seton Dam. Monitoring in 2021 (Year 9 of 10) consisted of:

1. Biological sampling and age analyses to evaluate juvenile and resident fish traits and condition.
2. Abundance estimates of Rainbow Trout (*Oncorhynchus mykiss*) fry.
3. Telemetry to inform species-specific distribution and habitat use.
4. Enumeration of adult Pacific Salmon and steelhead (anadromous *O. mykiss)*.
5. Measurement of physical habitat variables (i.e., water temperature, riverbed elevation, substrate size) to assess spatial and temporal conditions.
6. Habitat suitability surveys to evaluate species and life-stage specific habitat suitability during different flows and over time.

Management questions (MQ) were defined in 2012 and are assessed under two operational regimes: Water Use Plan (WUP; 2013-2014) and Modified Operations (MOD; 2015-2021). Target flow releases from the Seton Dam, specified under the WUP, were designed to mimic the river's natural flow regime, and ranged between 12 to 36 m^3/s (± 25%) with a minimum flow requirement of 5 m^3/s and target maximum of 60 m^3/s. Operations of the Bridge-Seton hydroelectric complex were modified in 2016 due to safety concerns at La Joie Dam (the uppermost dam in the system) and have resulted in intermittent exceedances of WUP target flows at the Seton Dam.

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?

Relative abundance was determined using a combination of mark-recapture electrofishing and snorkel surveys; estimates were limited to age-0 Rainbow Trout due to low capture rates of other species and age classes. Distribution of juvenile salmonids was assessed using recapture locations and Passive Integrated Transponder (PIT) arrays in both spawning channels and the Seton Dam fishway. Scales were collected from a subset of captured fish for age analysis to separate species-specific age classes to assist in abundance, growth, and distribution measures. A total of 16 fish species have been detected including seven salmonids; Rainbow Trout (age 0 to 3), Coho Salmon (*Oncorhynchus kisutch*, age 0 to 2), and Chinook Salmon (*Oncorhynchus tshawytscha*, age 0 to 2) were most common.

Captures and detections of tagged fish indicated that Rainbow Trout fry were distributed predominantly in the mainstem, while parr and adult fish were distributed between the spawning channels and mainstem. Rainbow Trout moved between the spawning channels and the mainstem Seton River, suggestive of a single population. Positive monthly growth was evident in age-0 to age-2 fish. Initial analyses indicated that rearing location and year (related to flow and habitat conditions) were significantly affecting the body condition of fry in that fish found in the USC and Reach 3 had a lower body condition compared to fish in Reach 1. Rainbow Trout fry abundance was greatest in 2014 (*N* = 13,481 fish; 95% credible interval of 9,734 - 18,981 fish) and lowest in 2021 (*N* = 1,839 fish; 95% credible interval of 1,113–3,025 fish). Given that 2014 had high variability in observed densities during electrofishing and was the only estimate under WUP target flows, there is a high degree of uncertainty in the estimate and comparative analyses between abundance and flow conditions were limited. Between 2014 and 2021, data indicated that suitable rearing habitat for Rainbow Trout fry has decreased, particularly in Reach 2.

Juvenile Coho Salmon were distributed between the mainstem and spawning channels regardless of age. Though recapture rate was low, all recaptures were in the same reach as the original capture, suggesting that Coho Salmon juveniles may show high site fidelity. Movement between spawning channels and the mainstem was detected during the typical period of seaward migration, as well as prior to the overwintering period. Data suggest that most juveniles leave the system in the spring at age 1 with a smaller number remaining for a second year. Growth metrics from month-to-month were positive with initial analyses showing that rearing location and year (related to flow and habitat conditions) were significantly affecting the body condition of fry.

Juvenile Chinook Salmon were found primarily in the mainstem Seton River close to the Fraser River. Juvenile Chinook Salmon spend up to 2 years in the Seton River prior to making their seaward migration, which occurs in the spring. While positive monthly growth was observed, growth metrics did not appear to be related to conditions in the Seton River system. DNA results from the 2016–2018 samples (results pending for 2019–2021; samples were also collected from juvenile Coho Salmon in 2020 and 2021) revealed that, on average, 52% of juvenile Chinook Salmon captured in the Seton River originated from other Fraser River populations (though not Bridge River). The Seton River evidently provides rearing habitat for many Chinook Salmon populations; given their imperiled status, the potential effects of Seton Dam operations on this species remains an important consideration.

Assessing adult anadromous salmonids in the Seton River has been challenging. Enumeration data for steelhead, Coho and Chinook Salmon were limited due to low densities and poor visibility in the mainstem. Pink Salmon (*Oncorynchus gorbuscha*) run on odd calendar years and were the most numerous species (counts ≈ 1,800 to 9,000 adults). All species appear to spawn in the Seton River (primarily in Reach 1) and spawning channels. Steelhead spawning has not been visually confirmed in the mainstem. Abundance estimates for steelhead that passed the Seton Dam (presumably to spawn in upstream habitats) were 25 fish in 2019, 17 fish in 2020, and 30 in 2021 based on resistivity counter data. Steelhead migration behaviour may be affected (delayed entry and passage of the fishway) during high flow releases from the Seton Dam, particularly > 60 m^3/s; though with only three years of monitoring, data are limited. Continued use of the Seton Dam resistivity counter (regardless if the year is considered MOD), in conjunction with joint collection of telemetry data with BRGMON-3 is recommended. Adult salmon were observed in much greater numbers in 2019 and 2020 (n = 235 Coho Salmon; n = 66 Chinook Salmon) following the Big Bar slide compared to previous years (n ≈ 30 to 110 Coho Salmon; n ≈ 0 to 5 Chinook Salmon); the effect on steelhead is unknown. Data from other programs indicated that the slide created a migration barrier and resulted in increased straying into non-natal streams. Continuing with the DNA analysis of stock origin for juvenile Chinook Salmon in addition to Coho Salmon is recommended for Year 10.

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

River-wide habitat suitability surveys at target flow releases demonstrated that the amount of suitable juvenile rearing habitat for Rainbow Trout (fry and parr), Coho Salmon and Chinook Salmon decreased from 12 to 60 m^3/s (though significant gains were made for all species from 25 to 40 m^3/s but are lost again from 40 to 60 m^3/s). Flows above the WUP target maximum were initially buffered by increased availability of side-channel habitats that became wetted; however, overall gains were lost once flows exceeded 100 m^3/s (based on a partial dataset). Data support the rejection of H_1, though we recommend completing the habitat suitability datasets above 100 m^3/s to fill in data gaps.

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?

Overall redd dewatering risk has been considered low; risk may be elevated in years where salmon abundance is high (i.e., during Pink Salmon runs) and moderate flows (lower than the maximum target but higher than the WUP target hydrograph) are maintained during the spawning period but then dropped later in the spawning period or incubation period. Stranded eggs were observed in 2019 under these conditions. Continued monitoring is required for reliable hypothesis testing; data do not clearly support or refute H_2.

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?

Substrate volume and size, particularly for spawning, have been maintained in the Reach 1 monitoring area. Results were variable, but current data suggest that although flow releases have resulted in mobilization of gravel (reject first part of H_3), the river has been able to recover from periods of scour (i.e., periods of net scour and increased substrate size associated with high flows were followed by net deposition and decreased substrate size). Considering that the Seton Dam is a barrier to recruitment of sediment in the Seton River, a self-sustaining supply of suitable spawning gravel is likely limited. Limited evidence (four years of random sub-samples throughout the mainstem) has indicated that the mean substrate size in all reaches has varied over time with observed increase in Reach 1 and both increase and decrease in Reaches 2 and 3 (all of which may be an artifact of the random transect selection from year to year rather than river-wide trends). Continued monitoring is required for reliable hypothesis testing; data do not clearly support or refute H_3. We recommend conducting riverbed elevation surveys and pebble counts annually (regardless if the year is considered MOD).

Keywords

Seton River, Rainbow Trout, Coho Salmon, Chinook Salmon, Pink Salmon, steelhead, rearing, spawning, habitat, flow, discharge, hydro, hydroelectric, dam, operations

BRGMON-9 status of objectives, management questions, and hypothesis after Implementation Year 9 (2021).

Management Questions	Management Hypotheses	Implementation Year 9 (2021) 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?		Continued to collect data on biological characteristics of target fish species in the Seton River. Rainbow Trout and steelhead, Coho and Chinook Salmon use the study area for rearing and spawning. While Pink Salmon use the area for spawning, they out-migrate soon after emergence. Other salmonids found in the study area (i.e., Bull Trout, Mountain Whitefish, Sockeye Salmon) were either found in very low densities or only use the area during migration and therefore were not assessed. Biological metrics included: length, weight, condition, age, DNA (Chinook and Coho Salmon juveniles only), relative density and abundance, migration and spawning timing, spawning locations, and habitat use. Analyses will continue to be refined. Rainbow Trout fry mostly used mainstem habitats, while parr and adults used the spawning channels and mainstem. Preliminary analyses indicated that rearing location and year (related to flow and habitat conditions) significantly affected the condition of fry in that individuals found in both the USC and Reach 3 had a lower body condition than in individuals found in Reach 1. Fry abundance was greatest in 2014 and lowest in 2021. The 2014 estimate had high variability in observed densities and was the only year of WUP flows. All estimates from MOD years were lower.

Management Questions	Management Hypotheses	Implementation Year 9 (2021) Status
		Juvenile Coho Salmon were distributed between the mainstem and spawning channels. Limited recapture data suggested juvenile Coho Salmon exhibit high site fidelity. Juveniles typically migrated at age 1 with some remaining for a second year. Initial analyses suggested that rearing location and year (related to flow and habitat conditions) significantly affected the body condition of Coho Salmon fry. Juvenile Chinook Salmon primarily used mainstem habitats near the Fraser River confluence. Juveniles spent up to 2 years rearing in the Seton River prior to migrating. DNA analyses found that approximately half of captured juvenile Chinook Salmon originated from upstream Fraser River populations (though not Bridge River). Understandably, given the variety of populations, initial analyses indicated that growth was not statistically related to conditions in the Seton River. All target species appeared to spawn in the Seton River (primarily in Reach 1) and spawning channels. Pink Salmon were the most abundant of the Pacific salmon (n ≈ 1,800 to 9,000 adults). The Big Bar slide created a migration barrier on the Fraser River and increased straying rates of salmon in 2019 and 2020. Adult Coho and Chinook Salmon abundance were generally low in other years (n ≈ 30 to 110 Coho Salmon; n ≈ 0 to 5 Chinook Salmon). Observed counts in 2021 were similar to those observed prior to the Big Bar slide. Based on resistivity counter data, abundance of steelhead that passed the Seton

Management Questions	**Management Hypotheses**	**Implementation Year 9 (2021) Status**
		Dam (presumably to spawn in upstream habitats) were 30, 17, 25 fish in 2021, 2020 and 2019, respectively; the effect of the Big Bar slide on steelhead is unknown.
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.	River-wide habitat suitability surveys demonstrated that rearing habitat for Rainbow Trout (fry and parr), Coho and Chinook Salmon was inversely related to flow releases from the Seton Dam; the amount of suitable habitat generally decreased as flow releases increased from 12 to 60 m^3/s, however increases between 25 and 40 m^3/s have been observed but are lost again between 40 and 60 m^3/s. At flow releases above the WUP target maximum, habitat suitability was initially buffered by newly flooded side-channels; however, overall gains in habitat suitability were lost once flows were ≥ 100 m^3/s. Current data support the rejection of H_1, though only partial datasets were available above 100 m^3/s flows. Completing habitat suitability surveys at these flows is recommended.

Management Questions	Management Hypotheses	Implementation Year 9 (2021) Status
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.	Redd dewatering risk has been considered low. Risk may be elevated in years when salmon abundance is high and moderate flows are maintained during the spawning period but then dropped later in the spawning or incubation period. Stranded eggs were observed in 2019, but not in 2021 under similar conditions. Continued monitoring is required for reliable hypothesis testing; data do not clearly support or refute H_2.
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.	Data suggested that substrate volume and size, particularly for spawning, have been maintained in the Reach 1 monitoring area over the study period. Though results were variable, flow releases from the Seton Dam resulted in mobilization of gravel (reject first part of H_3), but the river was able to recover from periods of scour thus far. Because Seton Dam is a barrier to the recruitment of sediment in the Seton River, a self-sustaining supply of suitable spawning gravel is likely limited. Limited evidence indicated that mean substrate size was variable and observed trends thus far may be an artifact of random transect sampling. Nevertheless, observed mean substrate size in Reach 1 has increased, and has both increased and decreased in Reach 2 and Reach 3 over time. Continued monitoring is required for reliable hypothesis testing; data do not clearly support or refute H_3.

Acknowledgements

The monitoring work completed for BRGMON-9 took place in the traditional territory of the St'át'imc Nation and includes: Sekw'el'was, Xwisten, T'it'q'et, Ts'kw'atkaxw, Xaxli'p, Tsal'alh First Nations. We thank each of them for allowing us to study the fish populations within Seton River.

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1. Introduction

The Seton River Habitat and Fish Monitoring Project, BRGMON-9, was designed to monitor responses of fish habitat and fish populations in the Seton River to flow releases from Seton Dam. The project was initiated in 2012 as a ten-year monitor with data collection beginning in 2013. Instream Fisheries Research (IFR) has been involved in the project since 2014 in partnership with St'át'imc Eco-resources (2014–2016) and Splitrock Environmental (2017–present).

1.1 Background

The Bridge-Seton hydroelectric power development regulates the water within the Seton and Bridge River watersheds, within the territory of the St'át'imc people (Figure 1.1). In the Seton Watershed, the Seton Dam impounds Seton Lake and controls flows into the Seton River, which runs east into the Fraser River. At the dam, water from Seton Lake can also be diverted into the Seton Power Canal for power generation at the Seton Generating Station, which discharges water into the Fraser River approximately 1 km downstream of the Seton and Fraser confluence. The Seton power project was completed in 1956 and was the final dam built as part of the Bridge-Seton hydroelectric power development. The Water Use Plan (BC Hydro 2011) contains detailed descriptions of all infrastructure and operating conditions associated with the development.

In 1999, a consultative process began and ultimately, in 2011, lead to the adoption of a Water Use Plan (WUP) to guide operations for the Bridge-Seton hydroelectric power development (BC Hydro 2011). During the development of the WUP, one of the concerns was to devise an acceptable instream flow regime for the Seton River which balanced concerns for competing water uses, while recognizing the interdependence of all Bridge-Seton system projects (BC Hydro 2012). A critical concern identified was the need for a flow regime that considered the high ecological and cultural values 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 (BC Hydro 2012).

The Seton Dam and generating station are a 'hydraulic bottleneck' in the Bridge-Seton system whereby operational changes at the Seton Dam significantly affect management of upstream reservoirs (i.e., Carpenter and Downton Reservoirs). Consequently, there are times when flow releases into the Seton River that are greater than WUP targets 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 into the Lower Bridge River. As well, natural variability in flow patterns on a seasonal and inter-annual basis can result in highly variable hydrographs in Seton River. Maintaining the WUP target flows at Seton Dam is a trade-off between minimizing the effects of a modified flow regime to fish and fish habitat in Seton River and protecting the productive capacity of other upstream waterways (i.e., Lower Bridge River) that are considered higher priority under the WUP.

Beginning in 2016, Bridge-Seton operations were modified because of safety concerns at La Joie Dam (i.e., the Modified Operations). Specifically, the maximum water elevation in Downton Reservoir was decreased from 749 m above sea level (ASL) to 734 m ASL, decreasing the reservoir's storage capacity. As a result of the change in Downton Reservoir storage and WUP prioritization of flows in the Bridge-Seton system, maximum flow releases from the Seton Dam have exceeded WUP targets every year since 2015 (Table 1.1). Reduced storage at Downton Reservoir is expected to continue until 2030, creating a period of Modified Operations in the Bridge-Seton system that will increase the likelihood of exceeding WUP targets for the Seton Dam.

During periods of high flow, the mainstem of the Seton River connects to off-channel habitat, including side channels. The effects of these high flows on some juvenile fish are hypothesized to be beneficial because 1) side-channels may provide favorable habitat conditions for juvenile and sub-adult fish, and 2) a possible "dynamic equilibrium" of suitable hydraulic conditions exists on the Seton River [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 given that data previously collected for BRGMON-9 has shown that the "dynamic equilibrium" hypothesis can be rejected for the WUP target hydrograph (Buchanan et al. 2018).

Additionally, for some salmonid species in the Seton River, changes in flow between the spawning period and fry emergence could lead to redd dewatering and desiccation of eggs. The potential for dewatering in the Seton River is largely unknown, but likely depends on where fish deposit eggs, and the interactions between channel geometry and observed flows during the incubation period. In addition to potential dewatering, Modified Operations may also have an impact on the quantity of suitable gravel for spawning because 1) it is assumed there is little (if any) gravel recruitment to the Seton River channel below the Seton Dam, and 2) high flows may mobilize suitable spawning gravel. The combination of potential redd dewatering and gravel mobilization may reduce the quantity and quality of suitable spawning habitat in the river over time.

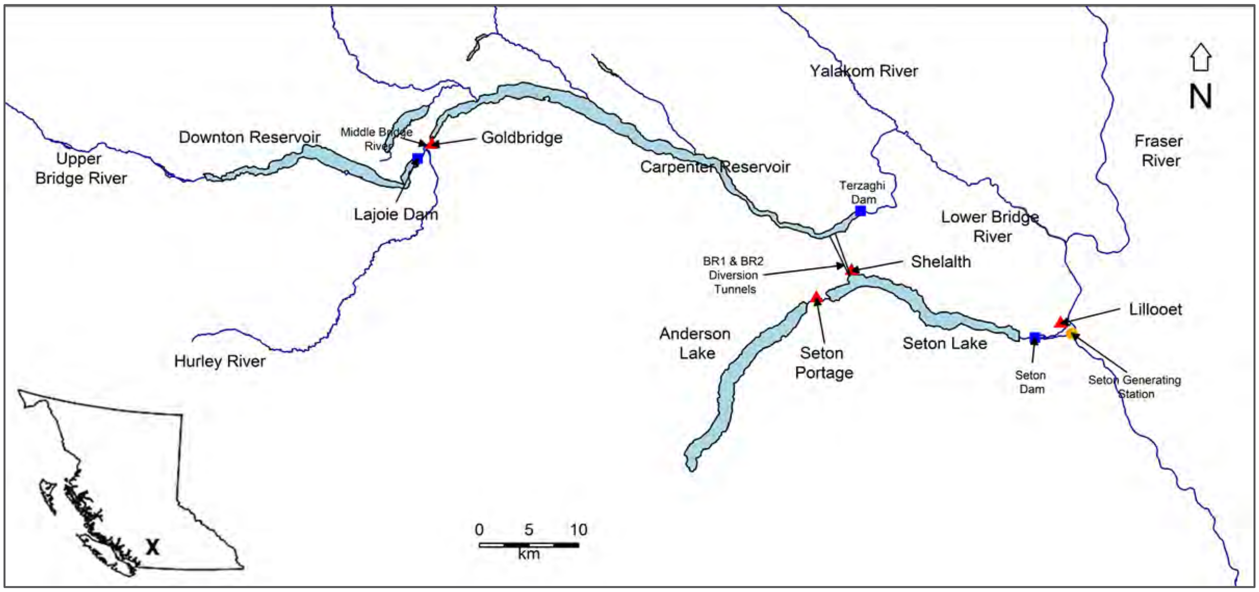


Figure 1.1 Map of the Bridge-Seton hydroelectric power system operated by BC Hydro.

Table 1.1 Annual flow statistics for the Seton River by condition, designated either as Water Use Plan (WUP) or Modified Operations (MOD), from 2013 to 2021. Data are daily average discharge from the Water Survey Canada station 08ME0003, located in the upper Seton River below the Seton Dam.

Year	Condition	Discharge (m^3/s)			
		Mean	SD	Minimum	Maximum
Target	WUP	19	8	11 (5)	36 (60)
2014	WUP	24	15	10	69
2015	WUP*	23	19	11	100
2016	MOD	36	33	13	114
2017	MOD	36	36	11	144
2018	MOD	24	19	10	93
2019	MOD	35	23	11	87
2020	MOD	30	20	11	92
2021	MOD	37	34	13	118

* Although 2015 was prior to Modified Operations, flows exceeded the target maximum of 60 m^3/s set forward in the WUP.

1.2 Management Questions

The purpose of BRGMON-9 is to document how flow releases from Seton Dam influence fish habitat in the Seton River, and specifically to addresses the following 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?

The data collected under BRGMON-9 can be used to develop and refine performance measures for fish resources in the Seton River. MQ1 to MQ4 are within the scope of this report; however, MQ5 is out of scope and therefore is not addressed herein.

1.3 Management Hypotheses

From the management questions above, three hypotheses and three sub-hypotheses were developed:

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 (BC Hydro 2012).

H_1 and its associated sub-hypotheses are related to MQ1 and designed to answer MQ2 through the collection of juvenile standing-crop biomass and habitat data. H_2 directly addresses MQ3 by assessing spawning and spawning habitat in the Seton River. H_3 addresses MQ4 by evaluating changes in substrate size and movement in key spawning areas of Seton River.

1.4 Objectives and Scope

The objective of BRGMON-9, as outlined in the Terms of Reference (BC Hydro 2012, 2018), is to monitor the response of fish habitat and fish populations to Seton Dam operations. The scope of work for BRGMON-9 in Implementation Year 9 (2021) was to:

1. Document hydraulic conditions 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 locations in the Seton River to assess potential risk of redd dewatering;
4. Monitor changes in the quantity, quality, and location of suitable spawning gravel in the Seton River;
5. Complete an annual report that summarizes 2021 monitoring results and incorporates all BRGMON-9 results to date and includes a literature review on the effects of increased flows during winter months.

The scope of work also included monitoring for any periods when flow releases from the Seton Dam exceeded the WUP target maximum of 60 m^3/s.

2. Methods

Fish species names and standard codes used in report tables and figures are listed in Table 2.1.

Table 2.1 Summary of fish species within the Seton River study area, as well as the standard species codes used in reporting.

Common Name	Scientific Name	Species Code
Bull Trout	*Salvelinus confluentus*	BT
Bridgelip Sucker	*Catastomus columbianus*	BSU
Sculpin sp. (general)	*Cottus* sp.	CC
Chinook Salmon	*Oncorhynchus tshawytscha*	CH
Coastrange Sculpin (formerly Aleutian Sculpin)	*Cottus aleuticus*	CAL
Coho Salmon	*Oncorhynchus kisutch*	CO
Dace sp. (general)	*Rhinichthys* sp; *Phoxinus* sp.	DC
Longnose Dace	*Rhynichthys cataractae*	
Lamprey sp. (general)	*Lampetra* sp.	L
Mountain Whitefish	*Prosopium williamsoni*	MW
Northern Pikeminnow	*Ptychocheilus oregonensis*	NSC
Peamouth Chub	*Mylocheilus caurinus*	PCC
Pink Salmon	*Oncorhynchus gorbuscha*	PK
Prickly Sculpin	*Cottus asper*	CAS
Rainbow Trout	*Oncorhynchus mykiss*	RB
Redside Shiner	*Richardsonius balteatus*	RSC
Slimy Sculpin	*Cottus cognatus*	CCG
Sockeye Salmon	*Oncorhynchus nerka*	SK
Steelhead (anadromous Rainbow Trout)	*Oncorhynchus mykiss*	ST
Sucker spp. (general)	*Catostomus* spp.	SU

2.1 Study Area

The Seton Dam is an 18-meter-high concrete dam that incorporates a fish ladder and a diversion canal. Water from Seton Lake is routed through the Seton Dam into the Seton River, or diverted via the Seton Power Canal to the Seton Generating Station (i.e., the powerhouse), which in turn discharges into the Fraser River (Figure 2.1). Below the dam, there are also two spawning channels for anadromous salmon; these sites are referred to as the lower spawning channel (LSC) and upper spawning channel (USC). Flows in both spawning channels are maintained from water diverted from the power canal. The Seton River has one tributary, Cayoosh Creek, which enters approximately 1.3 km downstream of the Seton Dam. Habitat encompassed by this monitoring program includes the mainstem Seton River and the spawning channels. Side-channels that become wetted above the maximum WUP flow target, referred to as off-channel habitats (OCH), are included under Modified Operations.

The Seton River mainstem was divided into three reaches, numbered in ascending order from the Seton Dam to the Fraser River confluence (Figure 2.1). 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 lower spawning channel intake to the Fraser River.

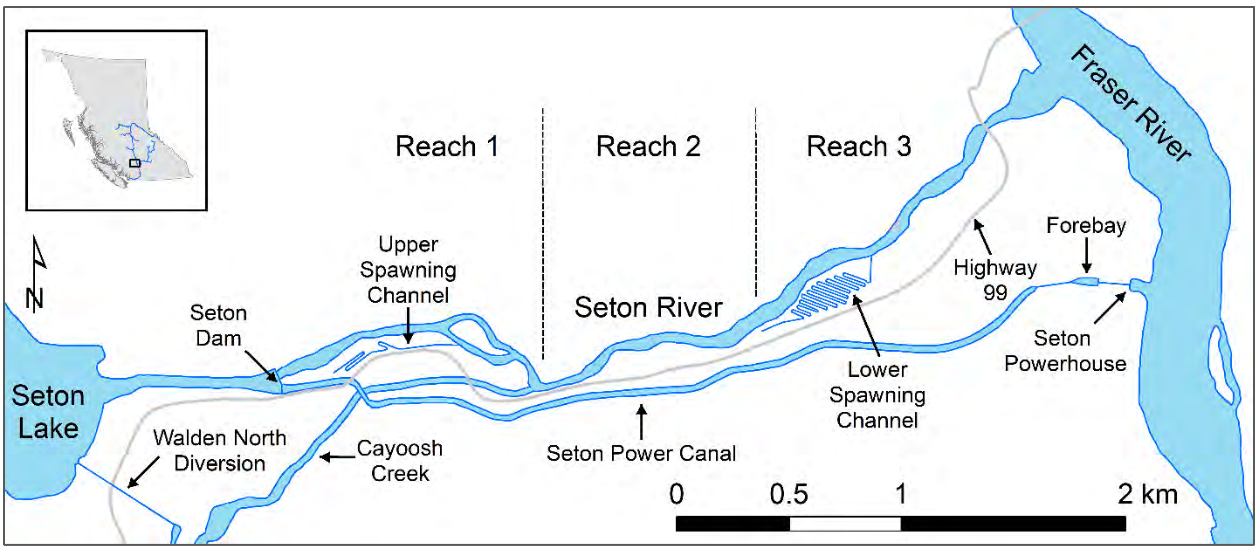


Figure 2.1 Detail of the Seton River study area bound by Seton Lake to the west and the Fraser River to the east. The study area was divided into three distinct reaches. Included on the map, but not included in the study, is Seton Power Canal and Cayoosh Creek.

2.2 Monitoring Sites

In 2013, Tisdale Environmental Consulting surveyed the entire length of Seton River and defined distinct hydrological habitat units (e.g., riffles, glides, pools; Ramos-Espinoza et al. 2014). Multiple sites within each individual habitat unit were identified resulting in a total of 125 sites throughout the mainstem; 76 sites originated on river right and 49 sites were on river left. River right and river left sites were matched where possible, creating 81 unique transects across the entire channel (Figure 2.2). These 81 full-channel transects have been used for all river-wide habitat suitability assessments (see Section 2.3.3). Monitoring locations for other components of BRGMON-9 are drawn out of the original group of 125 sites.

As a result of Modified Operations, there have been several changes to the locations of BRGMON-9 monitoring locations since 2016, including increased monitoring sites in the LSC and USC and the addition of sites within off-channel habitats (Ramos-Espinoza et al. 2016). Details on the number and specific locations of monitoring sites are discussed in the relevant sections below.

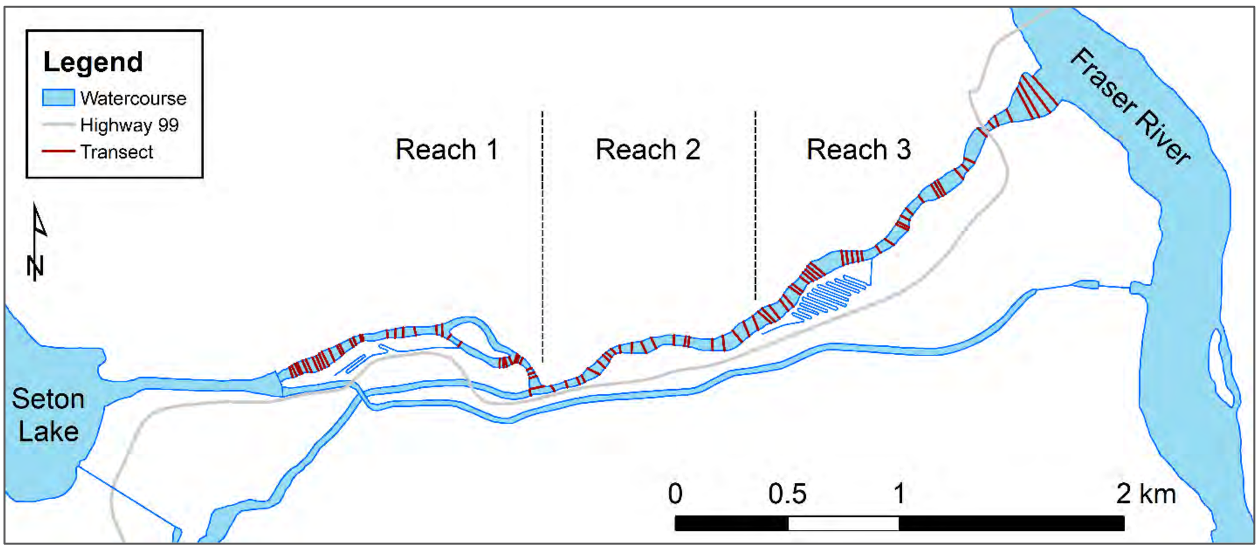


Figure 2.2 Locations of full-channel transects surveyed during river-wide habitat suitability assessments. Transects end point(s) represent the locations of the original group of 125 sites (n = 76 on river right; n = 49 on river left) that were used for other components of the BRGMON-9 Seton River Habitat and Fish Monitoring study.

2.3 Fish Habitat

2.3.1 Discharge

Instream flow data collected at five-minute intervals were obtained from the Water Survey of Canada (WSC) gauges at Seton River (08ME003) and Cayoosh Creek (08ME002; Figure 2.3). WSC data were used to calculate daily average flow for each gauge. The Seton River gauge was used for Reach 1 flow data, while data from both the Seton River and Cayoosh Creek gauges were combined and used for Reach 2 and 3 (Figure 2.3). The two spawning channels provide additional inflow, but their combined contribution is constant year-round (~2 m^3/s) and was not considered herein.

2.3.2 Water Temperature

Water temperature was recorded hourly for the duration of the study using Onset Tidbit water temperature data loggers (Bourne, Massachusetts, USA). Loggers were attached to solid features either on shore or within the river (e.g., pilings) using aircraft cable, and were weighed down using cinder blocks or lead weights. Loggers were downloaded regularly to reduce the risk of data loss (e.g., during high flow events that may wash out anchor lines). Water temperatures were monitored at five locations: the fishway of the Seton Dam, the mainstem Seton River immediately downstream of the dam in Reach 1 (“Upper Seton River”), the mainstem Seton River in Reach 2 (“Lower Seton River”), Cayoosh Creek approximately 300 m above the confluence with the Seton River, and in the USC and LSC (Figure 2.3). In December 2020, three additional loggers were installed at the following locations: Reach 1, Reach 3, and Cayoosh Creek (within 50 m of the existing logger) to act as backup in case an older logger failed.

An Analysis of Variance (ANOVA) using significance level of α = 0.05 was performed to test for differences in mean daily water temperatures by site and year. Assumptions were assessed using diagnostic plots. Only complete annual and site data sets were used. Significant ANOVA results were followed by Tukey’s pairwise hypothesis testing to determine statistical differences among groups; analyses were completed in R (R Core Development Team 2019) using the FSA package (Ogle 2016).

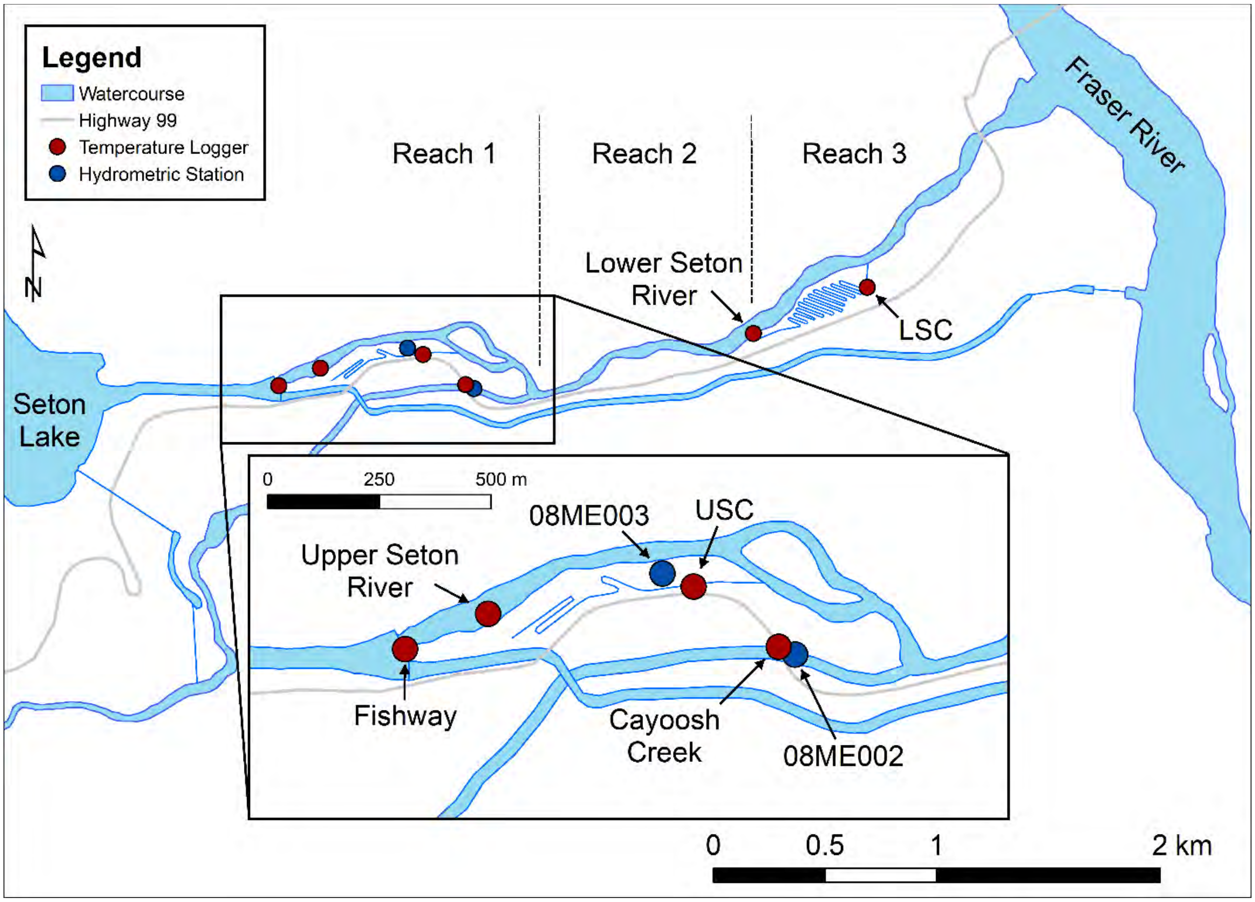


Figure 2.3 Locations of Water Survey of Canada hydrometric stations (used for instream flow data) and water temperature data loggers within the study area.

2.3.3 Habitat Suitability

Surveys were completed within the mainstem Seton River to assess the suitability of habitat for juvenile Rainbow Trout, Coho and Chinook Salmon, as well as assess how the quantity of habitat is affected by flow releases from the Seton Dam. Generally, habitat suitability surveys were completed under two scenarios: river-wide surveys at specific flow targets to estimate how habitat changes with flow, and at randomly selected sites based on the juvenile abundance surveys (see Section 2.4.3) that are completed annually during the baseflow period (~12-14 m^3/s). Field data collection followed the same methodology detailed in Ramos-Espinoza et al. (2015) based on the BC Instream Flow Methodology (Lewis et al. 2004).

Available habitat was estimated using a Weighted Useable Area (WUA) model based on species and life stage specific habitat suitability indices (HSIs) developed by the Government of British

Columbia (Ptolemy et al. 1994). HSI values correspond to specific physical habitat parameters, including water column depth, mean water column velocity, and substrate size class (Table 2.2, Figure 2.4). The model estimates the amount of suitable habitat available for different species and life stages at each surveyed flow. Each physical habitat parameter is weighted by an HSI value ranging from 0 (least suitable) to 1 (most suitable). The amount of suitable habitat (i.e., WUA) is quantified as the product of HSI values for each habitat parameter measured every meter along a transect, the width of the transect measurement (i.e., 1 m), and the length of the habitat unit. WUA is then summed across the transect, resulting in a total WUA for each habitat unit. Major assumptions of this methodology are that habitat is relatively uniform along the length of the river between each transect, and that each point along the transect represents an area of streambed bound by the halfway point to the neighbouring upstream and downstream transect (i.e., either the end of the hydrological habitat unit or the neighbouring transect).

From 2014 to 2019, river-wide habitat suitability surveys were completed at 12, 25, 60, and 86 m^3/s. To support ongoing assessment of the flow-habitat relationship, habitat suitability surveys were completed June 24–26, 2020, when the flow release from Seton Dam was 40 m^3/s. Because of the extremely high flows in Cayoosh Creek during this survey period, survey sites downstream of the Cayoosh Creek confluence (i.e., Reaches 2 and 3) were not surveyed. Reach 2 and 3 were surveyed on July 14-20, 2021 at 40 m^3/s to complete the survey. Results from 2014 to 2021 were combined to estimate at which flow(s) suitable habitat (i.e., WUA) is maximized and minimized for juvenile Rainbow Trout, Coho, and Chinook Salmon in the Seton River. Analyses also included applying a river-km to each survey site, measured to the nearest 0.1 km from the most downstream site near the confluence of the Fraser River, and qualitatively comparing spatial differences in available habitat longitudinally along the river at specific flows. Survey sites within the same 0.1 km were pooled together.

Habitat suitability surveys were also completed during approximate base flow conditions (i.e., ~12-15 m^3/s) in 2018, 2019, and 2020. In 2021, surveys were completed October 5-12, when flows in the Seton River ranged from 13.4–14.8 m^3/s, and flow contribution from Cayoosh Creek was minimal. These conditions are comparable to habitat suitability surveys conducted in 2014, 2018, 2019, and 2020 (Table 2.2, Table 2.3). The sites surveyed in 2018–2021 were the same randomly-selected sites used for juvenile abundance surveys during those years and provided a sub-sample of available habitat for the entire river. Consistency in flow conditions during these surveys allows for the detection of potential changes in habitat suitability over time during Modified Operations. Habitat for Rainbow Trout fry (the same species and life stage of the juvenile abundance analysis; see Section 2.4.3) was compared between 2014–2018, 2014–2019, 2014–2020 and 2014–2021 baseflow conditions longitudinally along the river. Because survey sites were randomly selected each year, only matching sites between paired years were considered to visualize trends in habitat suitability in Seton River over time during the period of Modified Operations. Although river-wide trends are examined, it should be noted that for evaluations of changes across years, results only represent a random subsample of the total habitat available.

Table 2.2 Species and life stage specific habitat suitability index values for substrate size classes of the Weighted Useable Area (WUA) model used to estimate habitat suitability for juvenile salmonids in the Seton River (Ptolemy et al. 1994, Lewis et al. 2004). Substrate was not included in Weighted Useable Area calculations for Chinook Salmon.

Substrate Size Class (mm)	Rainbow Trout Fry	Rainbow Trout Parr	Coho Salmon
Fines (< 2)	0.05	0	0.3
Small gravel (2–16)	1	0.5	1
Large gravel (16–64)	1	0.5	1
Small cobble (64–128)	1	0.6	0.8
Large cobble (128–256)	1	0.8	0.7
Boulder (256–4000)	1	1	0.4
Bedrock (> 4000)	0.2	0.4	0.2

Table 2.3 Flow conditions in 2014, 2018, 2019, and 2020 in Seton River (Reach 1) and in Cayoosh Creek during habitat suitability surveys used to compare changes in habitat suitability for juvenile Rainbow Trout, Coho Salmon, and Chinook Salmon.

Year	Survey Date	Mean Seton River Discharge (m^3/s)	Mean Cayoosh Discharge (m^3/s)
2014	Mar 18–Apr 9	12.3	1.5
2018	Sept 18–Oct 30	12.6	1.8
2019	Oct 10–21	14.1	2.2
2020	Dec 14–20	14.4	2.0
2021	Oct 5–12	13.7	3.0

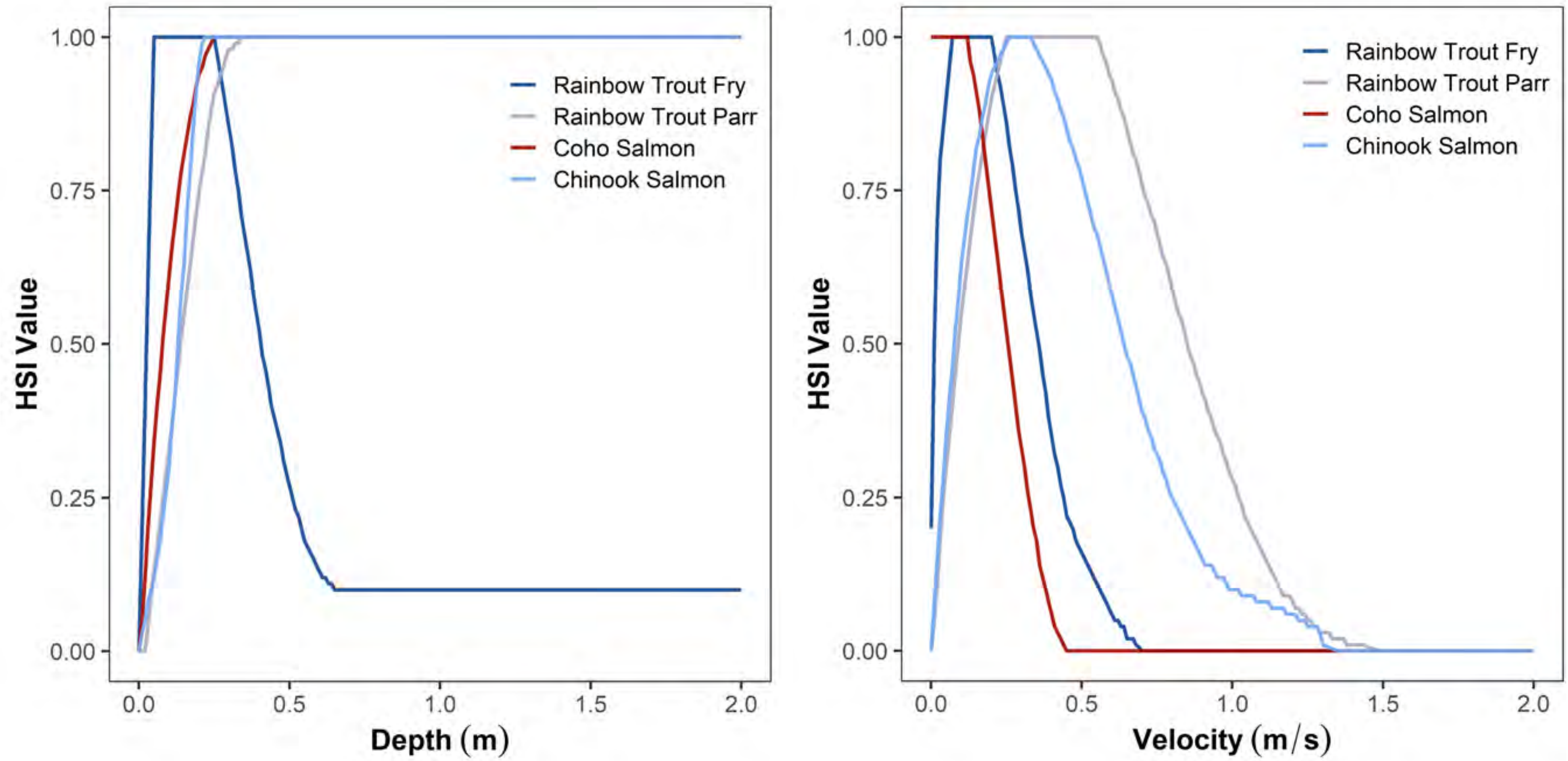


Figure 2.4 Species and life stage specific habitat suitability index (HSI) values for water column depth (left) and mean water column velocity (right) of the Weighted Useable Area (WUA) model used to estimate habitat suitability for juvenile salmonids in the Seton River (Ptolemy et al. 1994). HSI values continue indefinitely beyond 2 m and 2 m/s.

2.3.4 Riverbed Elevation

Detailed topographic (i.e., elevation) surveys have been used to monitor patterns of substrate scour/deposition (i.e., gravel mobilization) in Reach 1 approximately 150 m downstream of the Seton Dam, where the majority of salmon spawn outside of the spawning channels (Ramos-Espinoza et al. 2016; Figure 2.5). Surveys were scheduled every other year, occurring on odd years (e.g., 2013, 2015); additional surveys were added under Modified Operations if it was an 'off' year and flow releases from the Seton Dam exceeded 60 m^3/s. Bennett Land Surveying Ltd. (BLS) was contracted in 2013, 2015, 2016, 2017, 2019, 2020, and 2021 to conduct riverbed topographic surveys. The 8,300 m^2 survey area was identified as the major source of gravel for the Seton River. To characterize gravel mobilization over time, digital elevation models (DEMs) created from the survey data were differenced in QGIS Desktop 3.10.12 (QGIS Development Team 2019). In the resulting DEMs of differences (DoDs), positive values indicated where riverbed elevation increased (i.e., deposition) and negative values indicated where elevation decreased (i.e., scour). The vertical accuracy of the topographic surveys was approximately 0.03 m (BLS, pers. comm., December 2020), and so the maximum uncertainty in elevation change between two survey years was 0.06 m. Locations with differences between -0.06 m and 0.06 m were removed from further analysis, and then the areas and volumes of deposition and scour were calculated using ArcGIS Desktop 10.8.1. (ESRI 2020) for the periods between each survey.

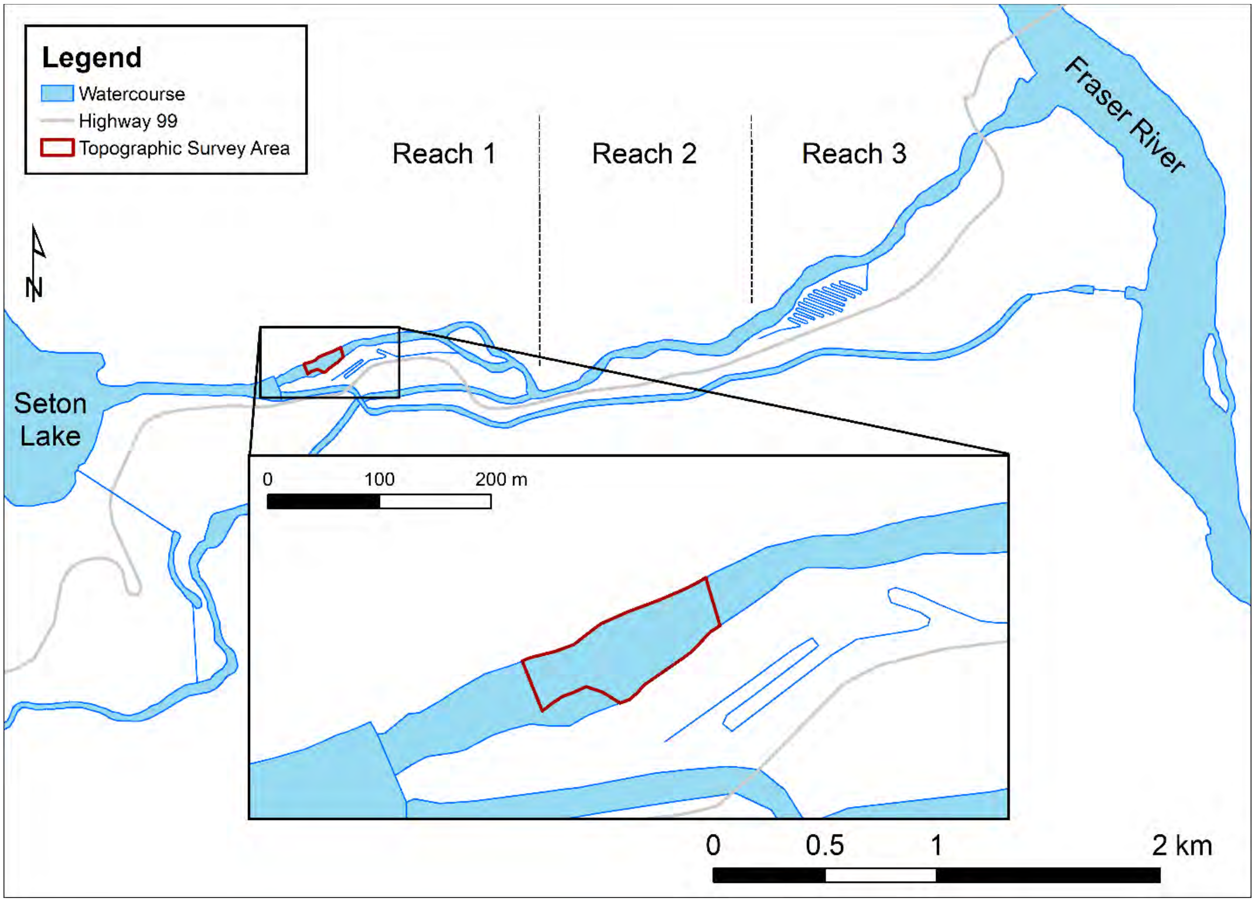


Figure 2.5 Location of topographic survey (i.e., riverbed elevation) within Reach 1 of the Seton River.

2.3.5 Substrate Size

Wolman pebble counts have been used to monitor change in substrate size over time. Pebble counts have been conducted annually since 2015 along transects within the riverbed elevation survey area, which was identified as a major source of gravel within the Seton River and was where most anadromous salmon spawn within the mainstem. At each transect, field crews would run a measuring tape across the width of the river, from bankfull pin to bankfull pin, or to a point of safe wading access. Along each transect, the size of 100 particles were measured along the intermediate axis to the nearest 1 mm. All particles < 2 mm were given a measurement of 1 mm. Particles were selected randomly along the transect. The geometric mean and geometric standard deviation of substrate sizes within each transect were calculated and compared among years.

Since 2018, Wolman pebble counts were also completed throughout the river at randomly selected survey sites that corresponded with juvenile abundance surveys each fall. Randomly selected particles were measured at each site using the same methods described above. Average substrate size was then compared by Reach to evaluate potential trends in substrate size throughout the river.

2.4 Juvenile & Resident Fish

2.4.1 Biological Characteristics

Biological surveys of juvenile and resident fish populations consisted of monthly open-site electrofishing (Smith-Root LR-24 backpack electrofisher) from April through October within the spawning channels and the Seton River. A crew of three experienced people performed single-pass electrofishing during daylight hours at established sites, each approximately 50 m in length (Figure 2.6). Crews moved in an upstream direction with one person operating the electrofisher and two people dip-netting fish. Sampling effort metrics (i.e., crew size, total electrofishing time in seconds, site length in m) for each site were recorded.

Sampling locations were based on permanently established locations throughout the study area (Figure 2.6). Originally, the project study plan aimed to sample six of the fourteen mainstem sites (MS1 to MS14) annually, in addition to random sampling (i.e., no established locations) at one site in each of the spawning channels. Since 2016, high flows from Modified Operations prevented sampling at some of the mainstem sites. Because the spawning channels were unaffected by flow releases from the Seton Dam, sampling effort within the spawning channels was increased and permanent sites were established. As well, sites within side-channels that were inundated at high flows were identified and surveyed (OCH1 to OCH9; Ramos-Espinoza et al. 2016). In 2020, two of the fourteen originally established mainstem sampling locations (MS12 and MS13) were removed due to new bridge construction along Highway 99. Overall, there were twelve permanent sampling sites in the mainstem of the Seton River, three permanent sampling sites in each of the spawning channels (six sites total), and four sampling sites in side-channel habitats (Figure 2.6).

All captured fish were kept in aerated bucket of river water. Fork length (to nearest mm) and weight (to nearest g) were measured for all fish. For all salmonids, scale samples were collected from the area above the lateral line and immediately below the dorsal fin and stored in labelled envelopes. Tissue samples (i.e., a fin clip) were collected from juvenile Chinook Salmon and used for a DNA analysis to determine stock identification since 2016. Tissue samples for DNA analysis have also been collected from juvenile Coho Salmon since 2020.

All Rainbow Trout, Coho Salmon, Chinook Salmon, Bull Trout, and Mountain Whitefish > 75 mm in length were targets for passive integrated transponder (PIT) tags that can then be used to assess distribution and movement (see Section 2.4.2). Each fish of the target species and size class were scanned for existing tags. Untagged fish were implanted with a 134.2 kHz 12 mm HDX PIT tag (Oregon, RFID, Portland, Oregon USA) using a 12-gauge needle. Fish < 150 mm were tagged in the ventral stomach cavity, while fish > 150 mm were tagged in the dorsal musculature. Tagging status (recapture of tagged fish, tag applied to untagged fish, or untagged fish with no tag applied) and the unique PIT tag identifier were recorded. Recaptured fish were re-measured to evaluate growth between capture events.

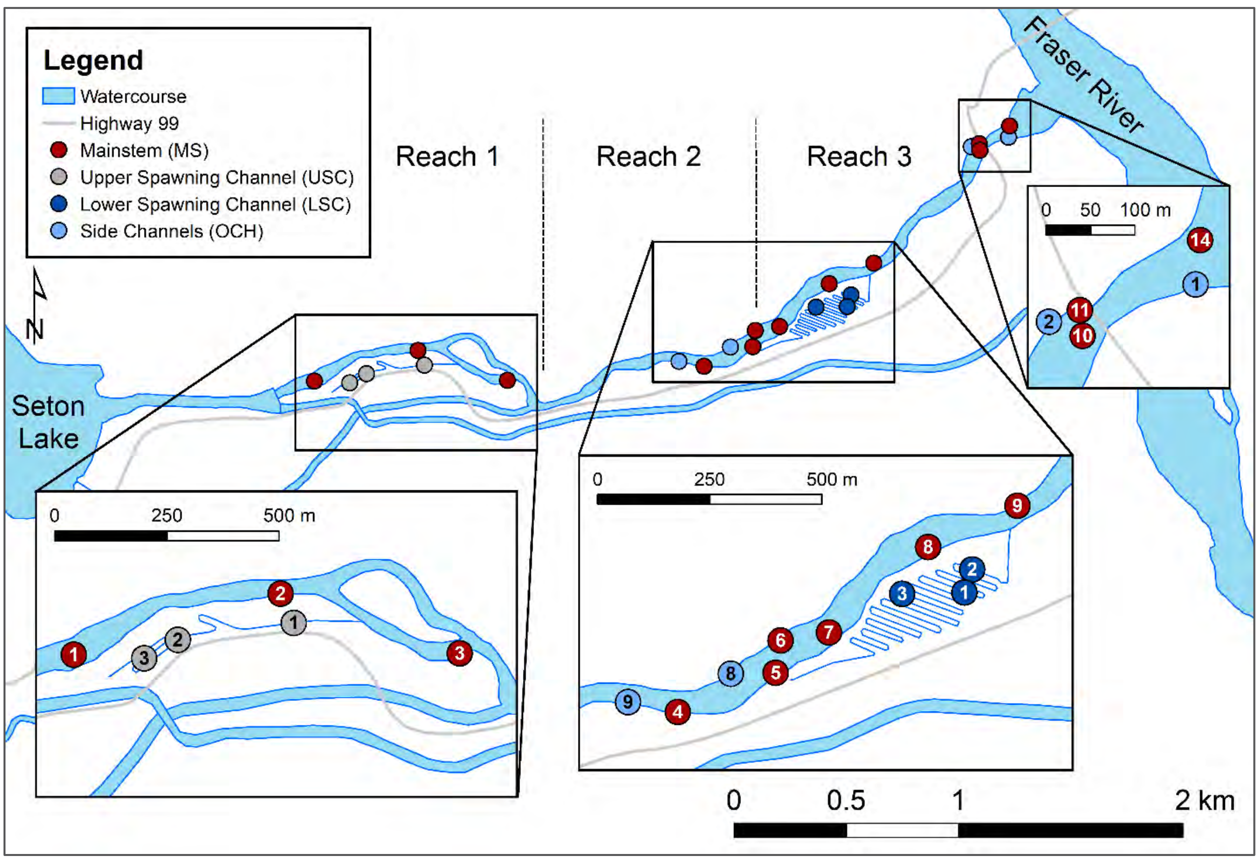


Figure 2.6 Sampling locations for biological surveys (electrofishing) of juvenile and resident fish in the Seton River from 2014 to 2021. Side-channel (OCH) sites were surveyed only if wetted (i.e., at flows greater than 60 m^3/s).

Age Analyses

Ageing analyses were used to better understand the basic biological characteristics and life histories of juvenile and resident fish. Scale samples were stratified by fish length (25-59 mm, 60-124 mm, 125-170 mm, >170 mm) based on approximate sizes of each age class as estimated through previous years' data. A maximum of thirty scales per species and length strata were selected for ageing each month. Scales were mounted directly onto glass slides, digitally photographed, and read under magnification. Age was estimated independently by two people without knowledge of identifying biological data to reduce bias (Zymonas and McMahon 2009).

Age-length keys (ALKs) were developed for Rainbow Trout, Coho Salmon, and Chinook Salmon. Fish from all years and capture locations were pooled for ALKs under the assumption that fish move freely between the spawning channels and mainstem Seton River. 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). Because of the rapid growth rates of juvenile fish, two seasonal ALKs were created for each species: one for March through June, and a second for July through October.

Body Condition

Fulton's Condition Factor (K_f), a measure of body condition that refers to the general plumpness or fatness of a fish relative to its length, was used to explore potential relationships between the condition of Rainbow Trout, Coho Salmon, and Chinook Salmon and flow releases from the Seton Dam. K_f was calculated according to Anderson and Neumann (1996):

$$\boldsymbol{K_f = \frac{W * 10^N}{L^3}} \qquad \text{Eq 1}$$

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 salmonids in the Seton River study area).

The effects of year and capture location were evaluated statistically for their effect on K_f in an Akaike Information Criterion (AIC) multi-model selection approach, with AIC values adjusted for small sample sizes (i.e., AICc; Burnham and Anderson, 2002). Year was used as a proxy for flow condition in analyses; while flow conditions have generally been high every year under Modified Operations, the magnitude, timing, and duration of peak flows have varied among study years.

We performed a Multivariate Analysis of Variance (MANOVA) using Type II sums of squares and significance level of $\alpha = 0.05$. MANOVA testing was performed for age-0 and age-1 Rainbow Trout, and for age-0 Chinook and Coho Salmon. Statistical testing was not performed for older age classes or other species, due to small sample sizes and missing data that resulted in highly imbalanced year-reach comparisons. Five candidate models were tested for each age class and species, and the model with the lowest AICc 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 AICc values were within two units of each other (i.e., Δ AICc < 2), models were considered to have equal support, and the most parsimonious model (i.e., model with the fewest parameters) was selected. Significant MANOVA results were followed by Tukey's pairwise hypothesis testing to determine statistical differences among groups; analyses were completed in R (R Core Development Team 2019) using the FSA package (Ogle 2016).

Stock Origin

Since 2016, DNA analysis has been used to assess the stock origin of juvenile Chinook Salmon found the Seton River. Although juvenile Chinook Salmon have been captured throughout the study, few adults have been observed (see Section 2.5.1). This has led to uncertainties regarding the use of the Seton River by adult Chinook Salmon for spawning. Unobserved adult Chinook Salmon may be spawning in the Seton River, or conversely, juvenile Chinook Salmon from other populations (e.g., Bridge River) may be rearing in or migrating to the Seton River. Tissue samples (fin clips) were analyzed using standardized genetic stock identification protocols by Fisheries and Oceans Canada at the Pacific Biological Station Molecular Genetics Lab in Nanaimo (Beacham et al. 1996).

2.4.2 Distribution

Location of initial captures and recaptures of marked juvenile and resident fish were recorded in all sampling programs. Capture data were pooled and then broken down by reach (i.e., Reach 1, Reach 2, Reach 3, LSC, USC), year, species, and age class to evaluate the distribution of juvenile and resident fish within the study area.

Passive Integrated Transponder (PIT) Telemetry

PIT telemetry was used to assess juvenile and resident fish distribution and movement within the study using hand-held PIT readers to evaluate recapture events and using fixed PIT antennas installed at key locations. Fish were tagged and recapture events were recorded during monthly biological surveys (see Section 2.4.1) and annual juvenile abundance surveys (see Section 2.4.3). Detection data were used to explore movement behaviour between the mainstem of the Seton

River, spawning channels, and Seton Lake (via the Seton Dam fishway). Fixed PIT antennas can be installed in several configurations that have implications for the types of data produced. A single antenna can determine the presence of tagged fish only, whereas two or more antennas installed as an array can determine direction of fish movement and antenna detection efficiency. Detection efficiency of the array was calculated as the number of fish detected on both antennas divided by the total number of fish detected on the first antenna. Low detection efficiencies indicate that fish were missed on one antenna but observed on the other and affects the ability to determine the direction of fish movements.

PIT antenna arrays varied slightly through the study period with three stations operating in 2021 (Figure 2.8). Double-antenna arrays were installed in the Seton Dam fishway in 2013 and in the USC in 2015; both have operated continuously since that time. In 2014, a single antenna was installed in the LSC and a second antenna was added in 2015. In 2018, detection efficiency on the LSC array was very low and so in 2019, the entire LSC PIT array was moved upstream to avoid electrical interference with the LSC resistivity counter operated seasonally by Splitrock Environmental. A double-antenna array in Cayoosh Creek was operated from October 2014 to April 2018, and then removed. The Seton Dam fishway and USC arrays were powered from main electrical sources, while the LSC array operated on a solar power and battery system.

2.4.3 Abundance

To estimate abundance of Seton River fish populations, we used a two-phase sampling protocol combining mark-recapture and index data like Korman et al. (2016) and Hagen et al. (2010). The mark-recapture portion consisted of a two-pass backpack electrofishing program used to estimate river-wide fish detection probability. Detection probabilities were applied to counts obtained from single-pass electrofishing at separate index sites to estimate abundances of target species and age classes for each reach of the Seton River.

The original goal was to incorporate index data from both fall open-site electrofishing surveys and spring snorkel surveys into a multi-gear model to estimate abundance of juvenile Coho Salmon, Chinook Salmon, 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, electrofishing detection probability is generally higher for juveniles relative to adults, whereas the opposite is true for snorkel surveys. The appropriateness of electrofishing and snorkel survey methods also varies with seasonal conditions; snorkel surveys perform better in deeper water but are not possible during high turbidity periods, while electrofishing surveys perform better in shallower water but are ineffective at higher flows. In the Seton River for both survey types, fish densities were too low to obtain abundance or index estimates for all species apart from Rainbow Trout. For Rainbow Trout, a hierarchical Bayesian model was used to estimate age-0 abundance using electrofishing mark-recapture and index data, while snorkel survey data were used to obtain simple indices of age-1 and age-2 fish.

Open-site electrofishing surveys for indexing and mark-recapture were completed in late September of each study year during baseflows of approximately 12-14 m^3/s. In some years (i.e., 2020), surveys were delayed by a month to late October, due to higher-than-normal flow releases from the Seton Dam. Index sites (n = 25) were randomly selected each year from the group of 125 sites (see Section 2.2) distributed throughout the mainstem Seton River (Figure 2.7). Sites with deep water habitats were excluded, as they could not be effectively surveyed with the electrofisher (Smith-Root LR-24 backpack electrofisher). Six mark-recapture sites (two in each reach) were also randomly selected from the group of 125 sites to represent shallow riffle and glide habitats; the actual number of mark-recapture sites included in the analysis varied from 4 to 6 depending on the study year, due to variability in river conditions and low catches in some years (Figure 2.7, Table 3.3).

A crew of three experienced people conducted electrofishing surveys during daylight hours at randomly selected sites. At each site, sampling effort was restricted to a single habitat unit (e.g., riffle, glide) over a distance of 50 m or less if the specific habitat unit was shorter than 50 m. Crews moved in an upstream direction with one person operating the electrofisher and two people dip-netting fish and attempting to capture all fish observed. At narrow sites, the entire width of the channel was sampled, while in wider sections, crews sampled as far into the river as was safe to wade. Sampling effort metrics (i.e., crew size, total electrofishing time in seconds, site length in m) for each site were recorded. Index sites were surveyed using a single pass. Mark-recapture sites were surveyed with two passes. During the first pass, all Rainbow Trout, Coho Salmon, and Chinook Salmon < 75 mm were marked with a fin clip and released at their original capture site; individuals > 75 mm were implanted with a 12 mm PIT tag. The second pass was performed after 24 hours, and the number of marked fish that were recaptured was recorded. All by-catch species and age classes were processed in the same manner described in Section 2.4.1.

Night-time snorkel surveys were completed annually since 2014 during base flows of approximately 12-14 m^3/s in March of each year with the exception of 2019 due to high flows (resulting from scheduled maintenance at the Seton Generating Station). Snorkel survey sites (n

= 20) were randomly selected from the same pool of 125 sites (see Section 2.2) and were focused on deeper pool, glide, or riffle habitats. Like electrofishing sites, survey effort was restricted to a single habitat unit (e.g., pool, glide) 50 m in length or less if the specific habitat unit was shorter than 50 m. Surveys involved a single swimmer navigating the site in an upstream direction searching for fish with the assistance of an under-water light, fish were identified by species and length was estimated to the nearest 10 mm; an onshore safety person recorded data. In some years, a second swimmer followed behind the lead swimmer and attempted to capture observed fish using small dip-nets. All captured fish were processed in the same manner described in Section 2.4.1. Data collected during the snorkel surveys add to the biological data set for juvenile and resident fish within the Seton River, particularly for age-1 and older fish; moreover, water temperatures in March are too cold for electrofishing methods and so snorkel surveys provided early spring data.

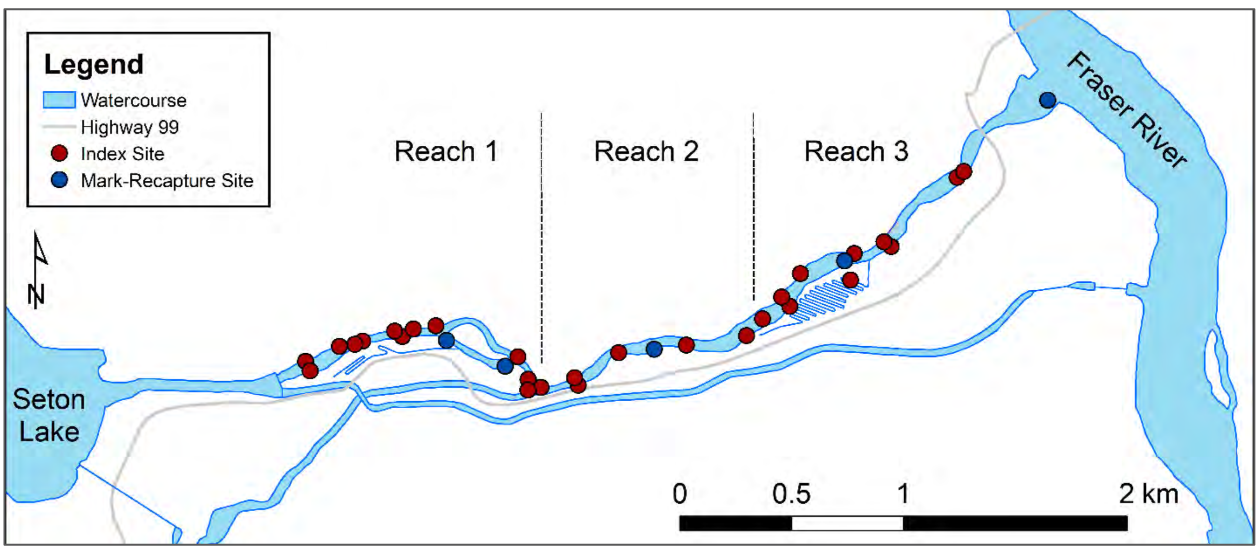


Figure 2.7 Location of electrofishing index and mark-recapture sites used to estimate juvenile fish abundance in 2020 within the Seton River in Reach 1, Reach 2, and Reach 3). Sites were randomly selected each year from an established group of 125 sites classified by habitat unit (pool, glide, riffle) and distributed throughout the mainstem of the Seton River.

Hierarchical Bayesian Analysis

A hierarchical Bayesian mark-recapture model was used to estimate year-specific abundance and density for age-0 Rainbow Trout in the Seton River. The Bayesian model has been used consistently for all project years, and a detailed description of parameters and model equations can be found in Buchanan et al. (2018) and Korman et al. (2016). The model was implemented through a hierarchical Bayesian framework in R software (R Core Development Team 2020) and JAGS using the package rjags (Plummer 2018).

The model consisted of two simultaneous levels: a detection model and a population model. The detection model used mark-recapture data from all sites and years to estimate a representative distribution of river-wide detection probabilities (Korman et al. 2016). This method assumes detection characteristics in the Seton River did not change over the entire study period. To maintain consistent detection efficiency, we used experienced field crews, standardized protocols to minimize the effect of sampling crew, and electrofishing took place during similar flow conditions each year (i.e., baseflows of ~12-14 m^3/s).

The population model used the detection probabilities estimated by the detection model to obtain site-specific abundance and density (i.e., fish/m) estimates for index sites and unsampled shoreline. The true abundance for each site was determined using the observed number of fish and a detection probability randomly drawn from the distribution created by the detection model. The abundance at each index site was then Poisson distributed with a mean equal the length of the site multiplied by the site-specific density estimated by the process model. All priors used during the hierarchical modeling were uninformative. Reported density and abundance estimates will change slightly from previous years, as the new mark-recapture data from the current year informs the capture probabilities in previous years.

2.5 Adult Anadromous Fish

2.5.1 Distribution and Abundance

Visual Counts

Visual surveys of the Seton River and spawning channels were performed weekly, as conditions allowed, during the adult migration and spawning periods for target species. Observed steelhead, Chinook Salmon, Coho Salmon and Pink Salmon were enumerated to provide an index of adult abundance. Any spawning pairs or visible redd sites were also noted and georeferenced. Survey methods consisted of two observers that walked along the riverbank in a downstream direction looking for adult fish and any spawning activity. Fish species, location (by reach or spawning channel), and viewing conditions (cloud cover %, precipitation, general water visibility) were recorded. Surveys targeting steelhead were scheduled for March or April to June of each year; however, high flows and low water visibility in the mainstem of the Seton River in all years except

2015 prevented surveys from being completed. Alternate methods have therefore been used to assess adult steelhead distribution and abundance (see Sections below: *Passive Integrated Transponder (PIT) Telemetry*, *Radio Telemetry*, and *Seton Dam Resistivity Counter*). Surveys targeting Chinook Salmon, Pink Salmon and Coho Salmon were completed each year from August to late November or mid-December (Table 2.4). In 2018, August surveys were not completed due to a miscommunication between contractors. Adult Sockeye Salmon were monitored under BRGMON-14, and so were not assessed during BRGMON-9 field programs (though field crews would record migrating adults observed during visual surveys).

Table 2.4 Annual timing of visual surveys for adult steelhead, Chinook, Coho and Pink Salmon during the migration and spawning periods.

Year	Steelhead	Chinook , Coho, and Pink Salmon
2015	Mar 4 – Jun 15	Aug 8 – Dec 15
2016	NA	Aug 16 – Dec 16
2017	NA	Aug 8 – Dec 12
2018	NA	Sep 25 – Nov 26
2019	NA	Aug 1 – Nov 26
2020	Mar 13 – Jun 15	Aug 4 – Dec 2
2021	Apr 6 - 19	Aug 24 – Nov 30

Passive Integrated Transponder (PIT) Telemetry

Since 2013, adult steelhead have been captured and tagged to assess their migration patterns and spawning locations under BRGMON-3. Fish were captured via angling throughout the Seton and Bridge watersheds, including at the confluence of the Seton and Fraser rivers. Upon capture, adult fish were implanted with a 134.2 kHz 32-mm HDX PIT tag in the dorsal musculature, in addition to a radio tag (see Section below: *Radio Telemetry*). Fork length and sex were recorded, and scale samples were taken for age analysis. To differentiate between adult Rainbow Trout and steelhead, a fork length cut-off was applied with fish ≥ 600 mm designated as steelhead. Unique identifiers associated with each PIT tag were searched against detection data from the PIT arrays at the Seton Dam fishway, LSC and USC.

Radio Telemetry

Radio telemetry has also been used under BRGMON-3 since 2013 to track adult steelhead during the migration and spawning period. Fish captured using the methods described above 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). After tagging, fish were held in a submersible holding tube for a minimum of 20 minutes prior to release to ensure full recovery, proper tag placement, and confirm the tag had not been regurgitated.

Tracking of radio-tagged steelhead was achieved through mobile surveys and fixed antennas. Weekly mobile tracking (by vehicle or foot) with a hand-held Lotek W31 radio receiver (Lotek Wireless Inc., Ontario, Canada) was conducted each year from mid-March (following the first fish tagged) to mid-May throughout the Seton River. Fish location and tag code were recorded, as well as any visual observations of tagged and untagged individuals of all fish species. Fixed station logging was generally conducted from early March to mid-June. Under BRGMON-3, one station consisting of an Orion receiver (Sigma Eight Inc., Ontario, Canada) linked to one Yagi 6-prong directional aerial antenna oriented downstream was installed in Reach 3 approximately 1.3 km upstream of the Fraser River confluence near the outlet of the LSC (‘Seton River DS’ station; Figure 2.8). In general, detection data from all fixed stations were used to build migration histories of tagged steelhead; reporting focused on fish that were detected more than once within the Seton River beyond the confluence with the Fraser River. While detections within the Bridge River were included, all sites within that watershed were combined and labeled as ‘Bridge River’ for simplicity (detailed data are available in the BRGMON-3 2020 report by White et al. [in review]). Detection data were also used to corroborate fish locations determined by mobile tracking, identify entry and exit timing, and generally inform our understanding of adult steelhead spawning migrations in the Seton River watershed. Overall, PIT and radio telemetry along with the resistivity counter acted as complementary detection methods to monitor adult steelhead.

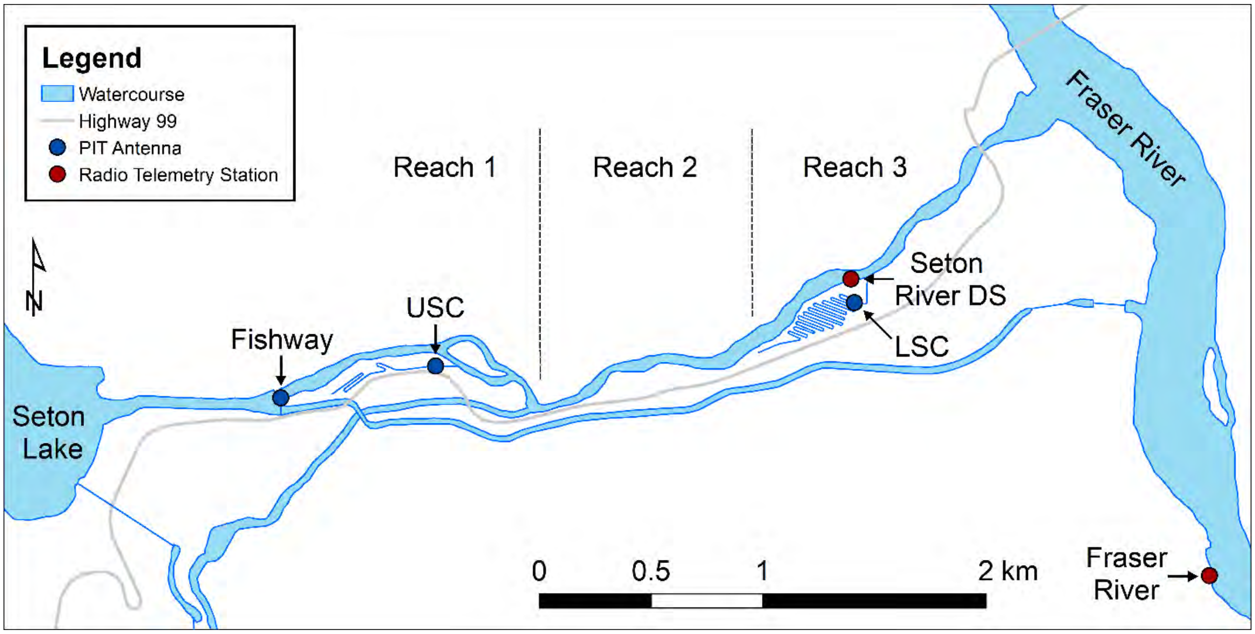


Figure 2.8 Locations of fixed telemetry stations within the BRGMON-9 Seton River Habitat and Fish Monitoring study area. All Passive Integrated Transponder (PIT) stations were double-antenna arrays.

Seton Dam Resistivity Counter

Since 2019, a resistivity tube counter was used to monitor fish passage through the Seton Dam fishway during the adult steelhead migration and spawning period from March to mid-June. The same counter was used in 2021 to enumerate the passage of Gates Creek Sockeye through Seton Dam from August through September (previously monitored under BRGMON-14). The resistivity counter consisted of eight tube sensors and two four-channel Logie 2100C resistivity electronic fish counters (Thurso, Caithness, Scotland), which counted upstream and downstream movement of fish through the Seton Dam fishway (Figure 2.9). Video equipment, that continuously monitored the tubes, consisted of digital video cameras attached to the upstream end of each counter tube (one camera per tube). Video data were collected and saved to a Digital Video Recorder (DVR) in five-minute increments. Each video camera was equipped with an underwater LED light to aid in species identification.

A detailed description of the operation of the Seton Dam fish counter can be found in the BRGMON-14 report by Casselman et al. (2013). Briefly, because a fish swimming through the resistivity sensor tube is more conductive than the water it is displacing, the counter measures a change in electrical resistance. An internal algorithm is then used to determine if a fish passed through, or if a fish entered the unit but failed to pass through. For each detection, the counter records the date, time, water conductivity, channel, direction of movement (upstream or downstream), and peak signal size (PSS) between 0 and 127. The PSS is a function of fish size, position in the sensor tube, electrode sensitivity, water conductivity, and bulk resistance (background resistance caused by flowing water). A minimum PSS threshold of 30 was used at the Seton Dam fish counter to eliminate resistance noise caused by surface air bubbles or debris passing through the sensor tubes (i.e., detection events with PSS < 30 were ignored by the counter).

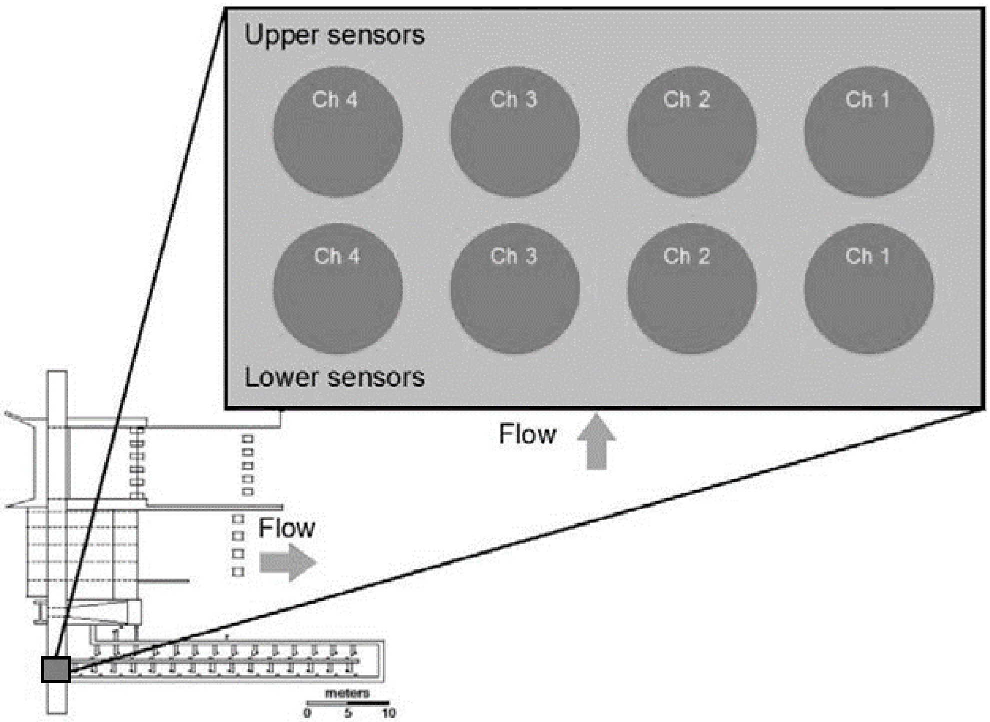


Figure 2.9 Schematic of the fish counter located at the exit of the Seton Dam fishway. The upper and lower sensors were monitored by two, four-channel resistivity counters.

Raw counter data were validated using the video record to determine the number of true positives, false positives, and false negatives (

Table 2.5), and to calculate tube-specific counter accuracy. We used a multi-step validation process that included targeted validation of counter up and down counts and random validation of additional video data (Figure 2.10).

During targeted validation, each graphical trace (up or down) was verified by watching the corresponding video data and an additional one minute of video before and after. The two-minute time bracket accounted for minor time-stamp discrepancies between the counter and the video records and allowed the analyst to verify movements that were recorded by the counter as multiple records (this occurred when a fish moved slowly through the counter or traveled in an erratic manner).

During random validation, a set of randomly selected video segments was reviewed to determine the number of false negatives (i.e., a fish was observed on the video but the counter recorded no trace). For each full day of video, either 20 randomly selected 10-minute segments or 10 randomly selected 20-minute segments of video were reviewed, and false negatives were recorded. The amount of video watched was based on estimated population size, number of fish expected to be validated, total number of hours available to be validated, and time constraints (Braun et al. 2016). The total number of false negatives was determined by expanding the validated count based on the proportion of video validated and total hours of video data collected (Table 2.6).

During the video validation process, each fish observed was identified to species. Tube counters are ideal for species identification, as fish must pass directly by the camera allowing distinguishing features (Table 2.7) to be examined. If species could not be determined or agreed upon by two independent analysts (e.g., during low visibility conditions), the species was classified as unknown.

The numbers of true positives (TP), false positives (FP), and false negatives (FN) were used to calculate counter accuracy (Equation 3), summarized by direction, species, and counter channel:

$$\boldsymbol{A = \frac{TP}{TP + FP + FN}} \qquad \text{Eq 3}$$

Accuracies were used to assess the performance of the counter, and to adjust the final estimate of abundance for steelhead that migrated through the Seton Dam fishway (*E*), as calculated using Equation 4:

$$E = \sum_{t=1}^{k} \frac{U_t - D_t}{A} \qquad \text{Eq 4}$$

where, U_t is the total number of upstream detections, D_t is the total number of downstream detections, U_t - D_t is the net upstream counts (accounting for species ratio is the counter accuracy), A is the counter accuracy, and k is the final day of the monitoring period. As no down counts were observed for steelhead, Equation 4 can be simplified to:

$$E = \sum_{t=1}^{k} \frac{U_t}{A} \qquad \text{Eq 5}$$

Table 2.5 Definition of error rates used to classify counter records during validation.

Error Category	Resistivity Counter	Video Review
True Positive	Graphical trace (up or down)	Fish observed and movement agrees with up or down classification
False Positive	Graphical trace (up or down)	No fish movement occurred
False Negative	No graphical trace	Fish movement occurred
Unclassified	Graphical trace (up or down)	Video data not available

Table 2.6 Summary of validation metrics for each year the Seton Dam fishway counter was operated for steelhead.

Year	Video Start and End Dates	Total Video (h)	Targeted Video Watched (h:m)	Random Video Watched (h:m)	% Total Video Watched
2019	Apr 01 to May 31	1,430	2:36	150:20	11
2020	Apr 09 to Jun 06	1,279	3:24	184:10	15
2021	Apr 15 to Jun 19*	1,380	11:14	168:50	13

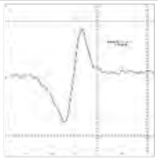
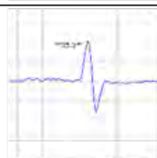
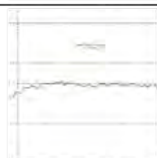
* Video outage in 2021 from May 4 – 10 and from June 2 – 4.

Table 2.7 Distinguishing features of commonly observed fish species moving through the Seton Dam fishway.

Species	Distinguishing Features
Steelhead	• Black spots on back, dorsal fin, and caudal fin • Cheeks and sides may be pink • Presence of radio tag antenna (fish tagged under BRGMON-3) • Length of fish is greater than the electrode spacing (30 cm) • PSS ≥ 30
Rainbow Trout	• Black spots on back, dorsal fin, and caudal fin • Cheeks and sides may be pink • Length of fish is less than the electrode spacing (30 cm) • PSS < 40
Bull Trout	• White leading edge on pectoral fins and mouth • Light coloured spots against dark coloured body on back
Mountain Whitefish	• Small mouth • Pointed nose • Adipose fin
Bridgelip Sucker	• Ventral sucker mouth • White ventral side • Large anal fin

Raw Counter Measurement

- Counter measures a change in resistance as fish passes over the counter site
- Each trace is examined by a reviewer and classified as an up, down, or event

Up		Fish is moving upstream, passing over all three electrodes
Down		Fish is moving downstream, passing over all three electrodes
Event		Partial or incomplete movement of fish over the counter electrodes (in any direction) caused by a fish nosing up, falling back, or sitting on electrodes. Events may also be caused by air entrainment or debris flow over the electrodes.

Targeted Video Validation

- All up and down records are reviewed on video
- Reviewer watches record plus 1 minute before and after record
- Record is classified as either:
    - **True positive**
    - **False positive**

Random Video Validation

- Subset of randomly-selected video segments is reviewed
- For each 24-hour period, 10 randomly selected 20-minute segments are reviewed
- Fish movement observed on video but without paired graphical trace are classified as **false negative**

Calculate Error Rates and Accuracy

- Determine the total number of true positives (TP), false positives (FP), and false negatives (FN), summarized by:
    - Movement direction (up and down)
    - Species
    - Counter channel
- Calculate accuracy (A) for each category

Figure 2.10 Flow diagram of the counter validation process used in estimating the annual abundance of steelhead migrating through the Seton Dam fishway.

2.6 Winter Modified Operations

In 2020, the Seton Generating Station experienced an outage from mid-August through to mid-October; while discharge through Seton Dam was increased during this period, it still resulted in a back-log of water in the Bridge-Seton system. To avoid spilling water into the Bridge River, variance flows to increase discharge to 21 m^3/s beginning on December 21, 2020 and continuing through to May 4, 2021 were approved for Seton Dam. Additional monitoring was added to the scope of BRGMON-9 to examine the potential effects of increased flows during winter months on water temperature and fish growth and distribution.

2.6.1 Water Temperature

Regulated rivers tend to be warmer in the winter and colder in the summer than their non-regulated counterparts (Heggenes 2021) and so increased discharge from Seton Dam in December through March could result in increased water temperatures in Seton River. Water temperature over the modified winter period was compared with the same period in previous years to determine if it fell within the natural variation. An Analysis of Variance (ANOVA) using significance level of $\alpha = 0.05$ was performed to test for differences in mean daily water temperatures by year. Assumptions were assessed using diagnostic plots. Only complete data sets were used. Significant ANOVA results were followed by Tukey's pairwise hypothesis testing to determine statistical differences among groups; analyses were completed in R (R Core Development Team 2019) using the FSA package (Ogle 2016).

2.6.2 Juvenile Fish

Higher discharge is known to push more fish into sheltered areas resulting in a per capita reduction of resources and therefore slower growth rates. To assess if increased discharge from Seton Dam was impacting the distribution and growth of juvenile salmonids, additional monitoring was conducted to determine where fish were residing and their body condition.

Distribution

Distribution and movement of juvenile salmonids was monitored through the PIT arrays in the spawning channels. Both arrays were serviced weekly during the winter to ensure data were downloaded on a regular basis and in the case of the LSC, battery levels maintained for continuous operation. Movements of tagged individuals during the winter period of modified operations were examined.

Growth and Body Condition

An additional electrofishing session was completed in early April 2021 to try and capture fish at the discharge they experienced through the winter and before they started to migrate. All juvenile salmonids were sampled for scales and their age was rolled into the standard BRGMON-9 age analysis (Section 3.2.1).

3. Results

3.1 Fish Habitat

3.1.1 Discharge

From 2016 to 2021, Modified Operations resulted in flow releases from the Seton Dam that exceeded the maximum WUP target of 60 m^3/s. Similar to most previous years, peak high flows occurred in the spring, with peak flows in 2021 occurring in May and reaching a maximum of 118 m^3/s at the Seton River hydrometric station in Reach 1 (Figure 3.1). Reaches 2 and 3 of the Seton River experienced peak flows of 162 m^3/s in May during the natural freshet period when flows from Cayoosh Creek also peaked (Figure 3.1). While flow releases from Seton Dam were maintained within the WUP target range of 15 – 60 m^3/s from January to mid-April and again from July through December, flows exceeded WUP targets except in March and September through November. From mid-April through June, flow releases were above the target maximum of 60 m^3/s. Similar to 2020, flow releases during the winter (starting in December 2021) were 20 m^3/s, almost two times greater than target baseflows (11 m^3/s). The mean annual discharge for Reach 1 was 37 m^3/s (SD = 34), while in Reaches 2 and 3 it was 44 m^3/s (SD = 42). Cayoosh Creek had a mean annual discharge of 7 m^3/s (SD = 13) in 2021. Over the study period, 2017 had the greatest maximum for all reaches of the Seton River and Cayoosh Creek, but mean annual discharge was greater in Reach 1 in 2021 than it was in 2017 (Table 1.1).

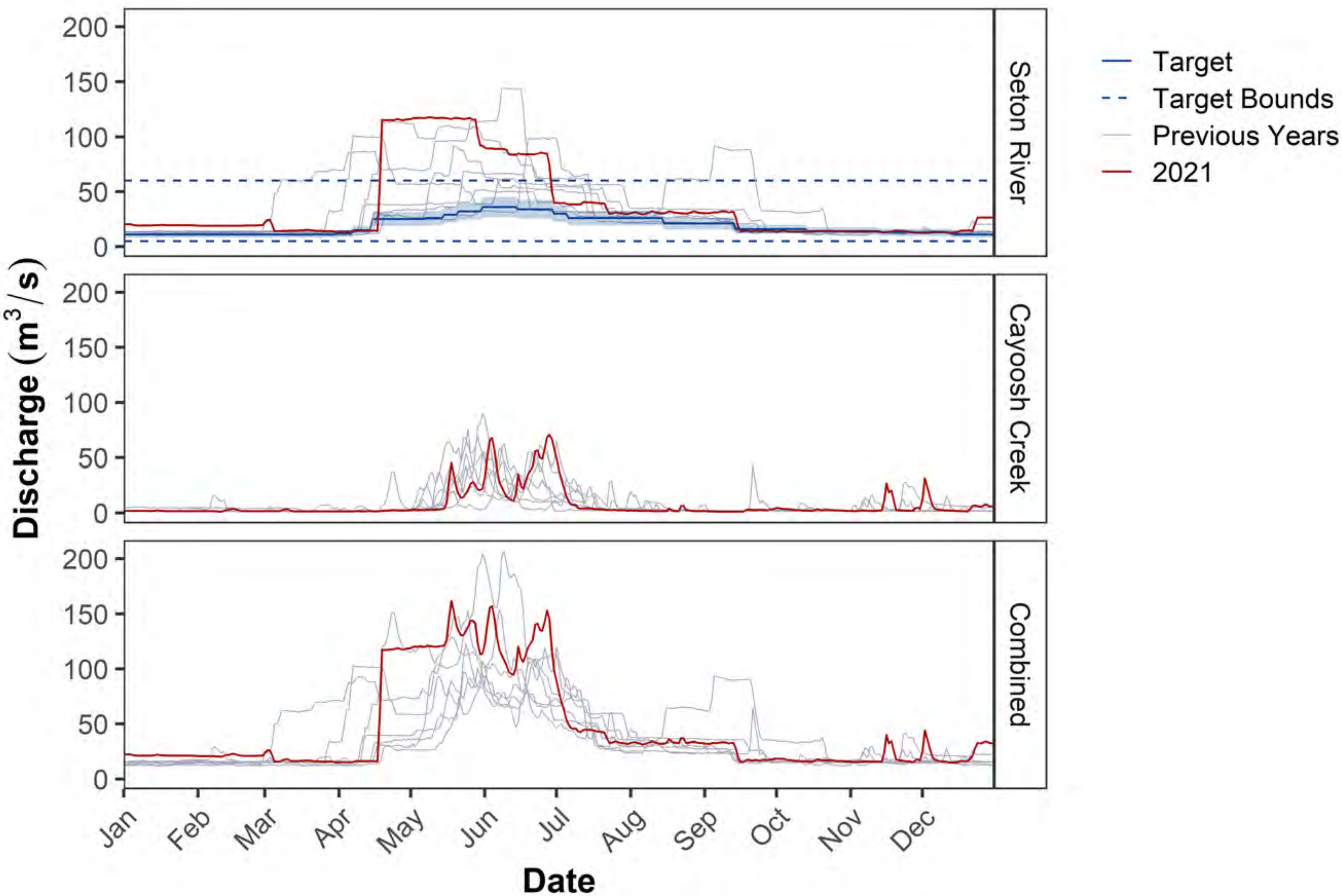


Figure 3.1 Hydrographs of flow releases from Seton Dam compared to Water Use Plan targets, as well as flow in Cayoosh Creek and the cumulative flow downstream of the confluence of the Seton River and Cayoosh Creek. Previous years represents 2014 to 2020.

3.1.2 Water Temperature

In the Seton River, the lowest water temperatures (typically 3-4°C) occurred from January to March; water temperatures gradually increased throughout the year until September when temperatures peaked at 20°C in the Seton Dam fishway and the Seton River (Figure 3.2). Water temperatures then decreased gradually through the fall until January when they again stabilized at their lowest points. Spawning channel temperatures followed the same pattern as the Seton River (data not shown). In 2021, the maximum water temperatures in the spawning channels reached 21°C. ANOVA results indicated mean daily water temperatures were significantly different among sites ($F_{5, 8522}$ = 21.5 P < 0.001). Tukey tests indicated the spawning channels were both significantly warmer than Reach 2 (P = 0.019). While the upper spawning channel, fishway and Reach 1 were statistically similar, the lower spawning channel was significantly warmer than all other sites. Cayoosh Creek was significantly cooler overall (Reach 2: P < 0.01, all other sites: P < 0.001; Figure 3.3). Year was also a significant determinant of water temperature ($F_{5, 8156}$ = 10.5, P < 0.001; Figure 3.3). Over the study period, 2021 had the warmest and 2019 had the coolest mean daily water temperatures (P = 0.036). Water temperatures in 2021 were the warmest recorded during the study period, but were statistically similar to conditions from 2014, 2015 and 2020.

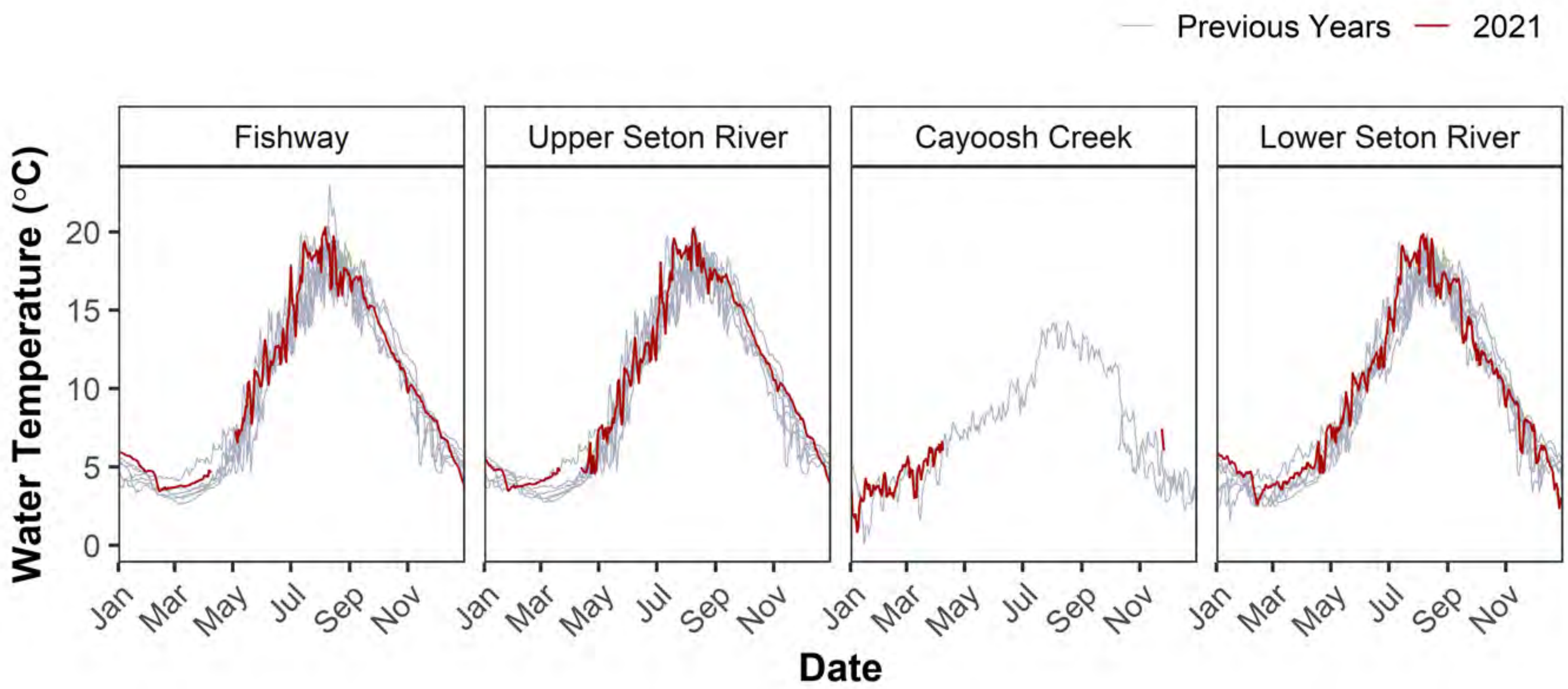


Figure 3.2 Mean daily temperature for the Seton Dam Fishway, Upper Seton River (Reach 1), Cayoosh Creek, and Lower Seton River (Reach 2) in 2021.

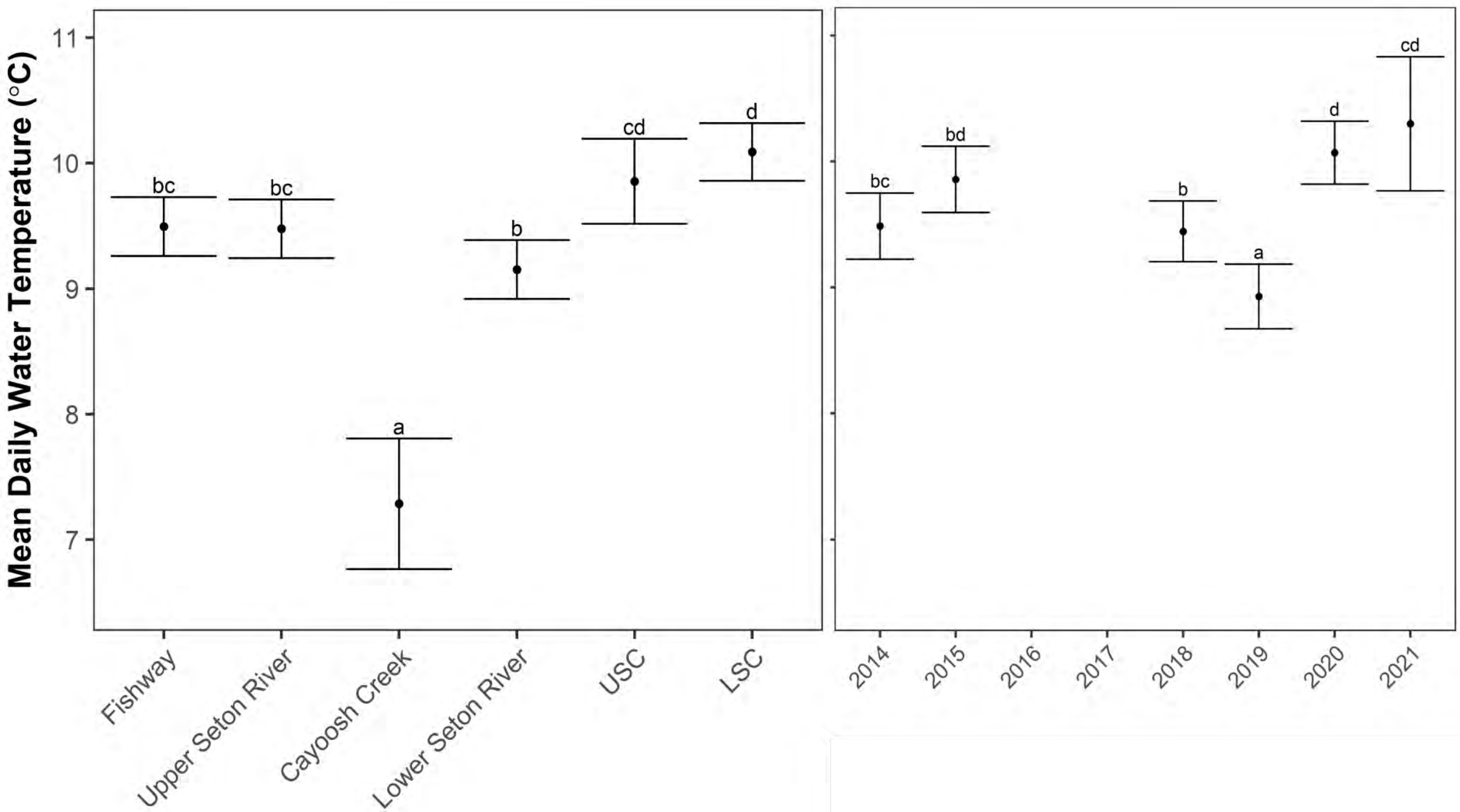


Figure 3.3 Mean daily water temperature (°C) with 95% confidence intervals by site (all years combined, left panel) and year (right panel) from 2014–2021, excluding 2016 and 2017 which had incomplete annual records. Data from Cayoosh Creek was also excluded from the year plot as complete data existed for 2020 only. Points within the same panel that do not share the same letter are statistically different from each other.

3.1.3 Habitat Suitability

River-wide Juvenile Rearing Habitat Suitability

Habitat suitability surveys conducted in 2021 provided additional insight to the flow-habitat relationship for juvenile salmonids in the Seton River. As observed in previous years, modelled habitat suitability sharply decreased at flows above 12 m^3/s for all four juvenile species and life stages, with slight increases from 60 to 86 m^3/s (Figure 3.4). However, results from 2021 surveys suggest that significant habitat is regained for all four juvenile species from 25 to 40 m^3/s, with the greatest increases being observed in Reach 2 and Reach 3.

Longitudinal profiles of habitat suitability along the Seton River showed that the largest increases in juvenile rearing habitat between 25 and 40 m^3/s were in Reach 3 (Figure 3.5, Figure 3.6). For example, there was an increase of more than 5,000 m^2 of habitat for Rainbow Trout parr and fry and juvenile Coho and Chinook Salmon at 40 m^3/s. Decreases in habitat at higher flows were relatively consistent throughout each reach for Coho and Chinook Salmon and Rainbow Trout fry. Rainbow Trout parr however, exhibited the least amount of change at 40 m^3/s.

Changes in Rainbow Trout Fry Habitat Suitability

To assess changes in habitat suitability at base flows over time, we compared results from sites sampled during the annual juvenile abundance surveys to matching sites from 2014. After pooling survey sites within the same 0.1 km, there were 22 matching habitat suitability sites between 2014 and 2018, 20 between 2014 and 2019, 20 between 2014 and 2020, and 22 between 2014 and 2021 (Figure 3.7). Differences in available habitat between 2014 and 2018 appeared to be relatively small throughout Seton River (<1%), while larger differences were observed from 2014 to 2019 (32%) and 2014 to 2020 (26%). Differences in available habitat between 2014 and 2021 were lower (17%) than what was observed in 2019 and 2020. It should be noted that these results only represent a random subsample of the total habitat available in the Seton River.

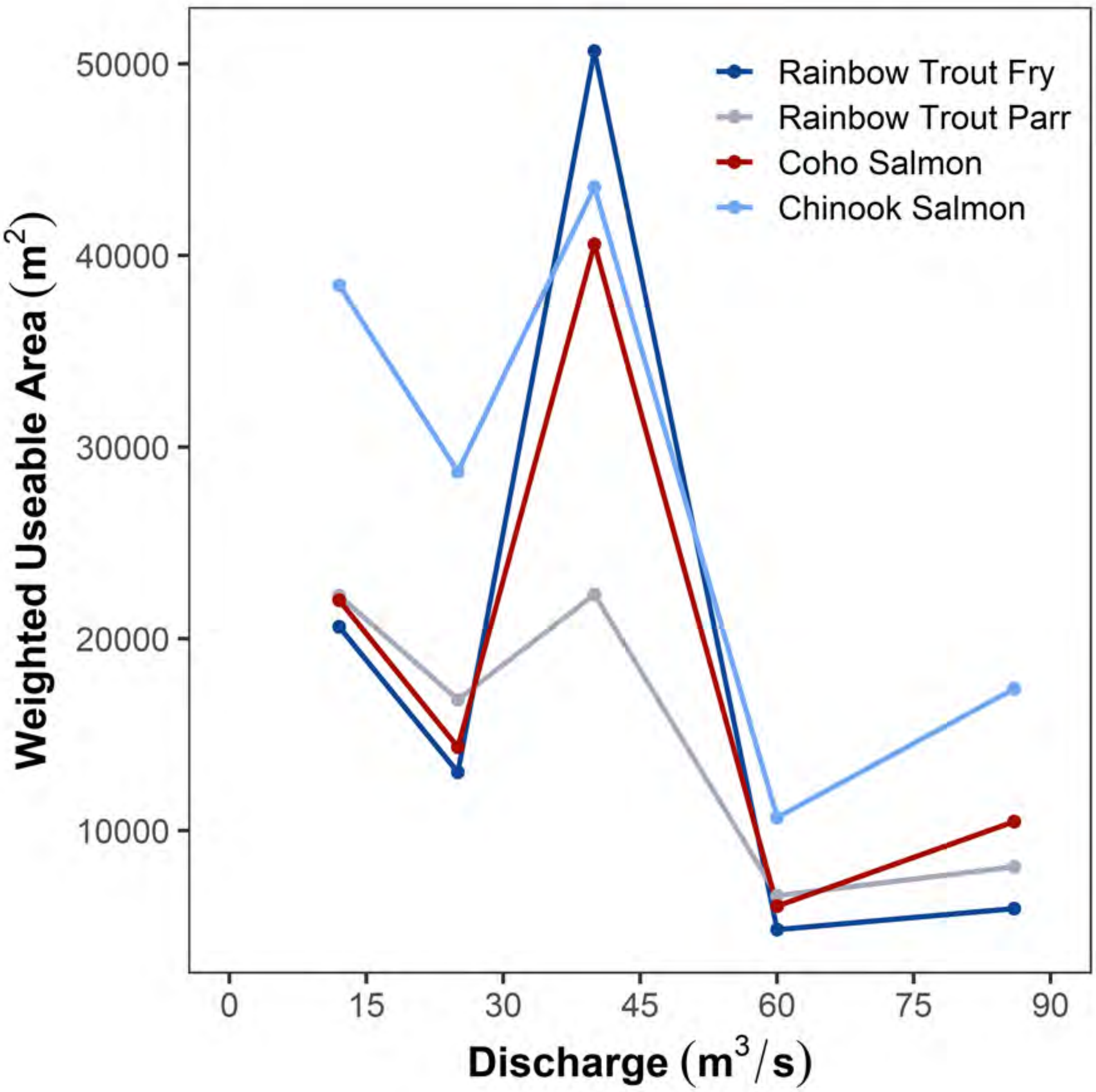


Figure 3.4 Total weighted useable area (WUA in m^2) in Seton River at various flow releases from Seton Dam for Rainbow Trout fry, Rainbow Trout parr, and juvenile Coho Salmon and Chinook Salmon.

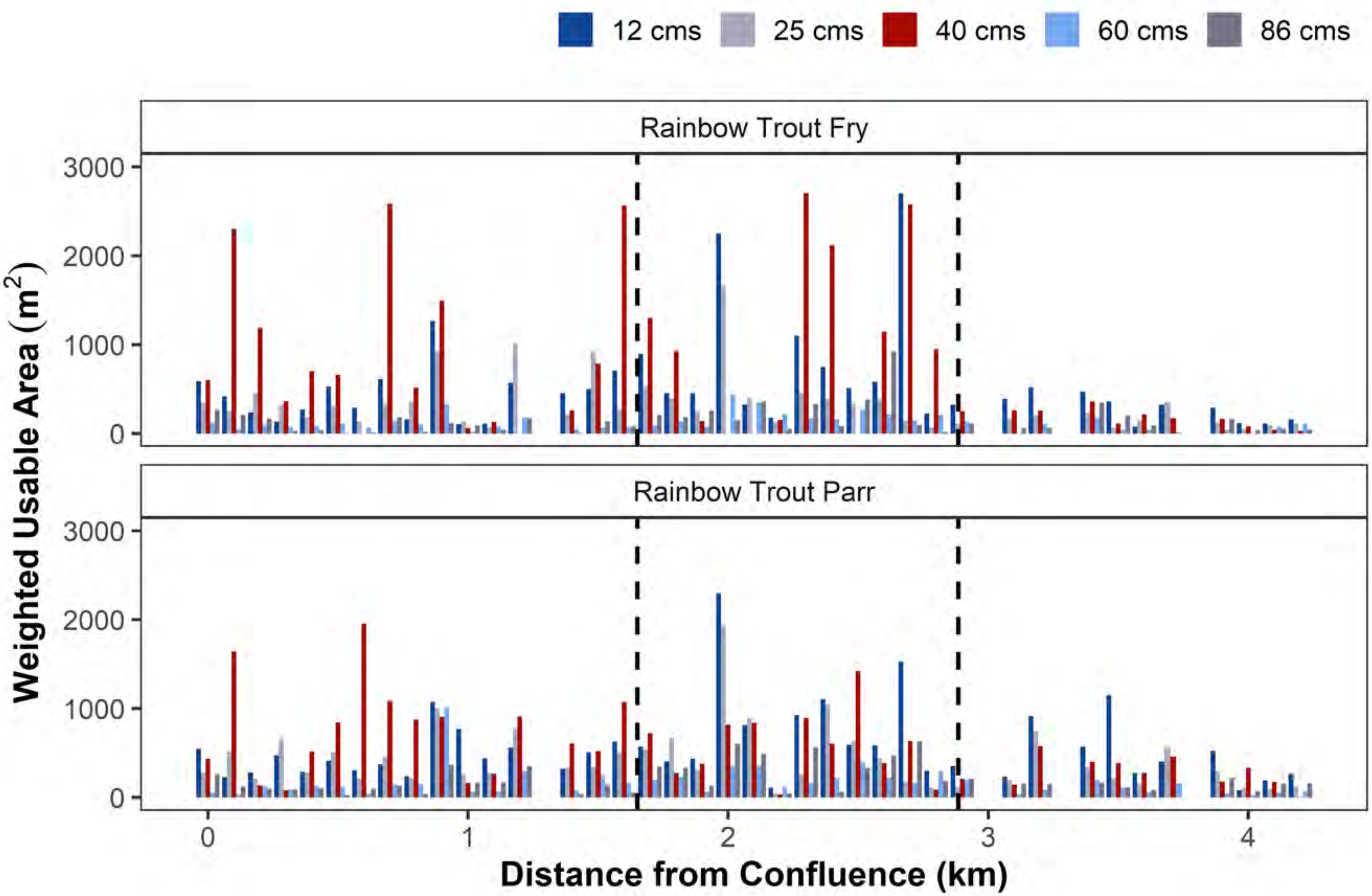


Figure 3.5 Total weighted useable area (WUA in m²) in Seton River at various flow releases from Seton Dam for Rainbow Trout fry and parr. Dashed lines indicate reach breaks.

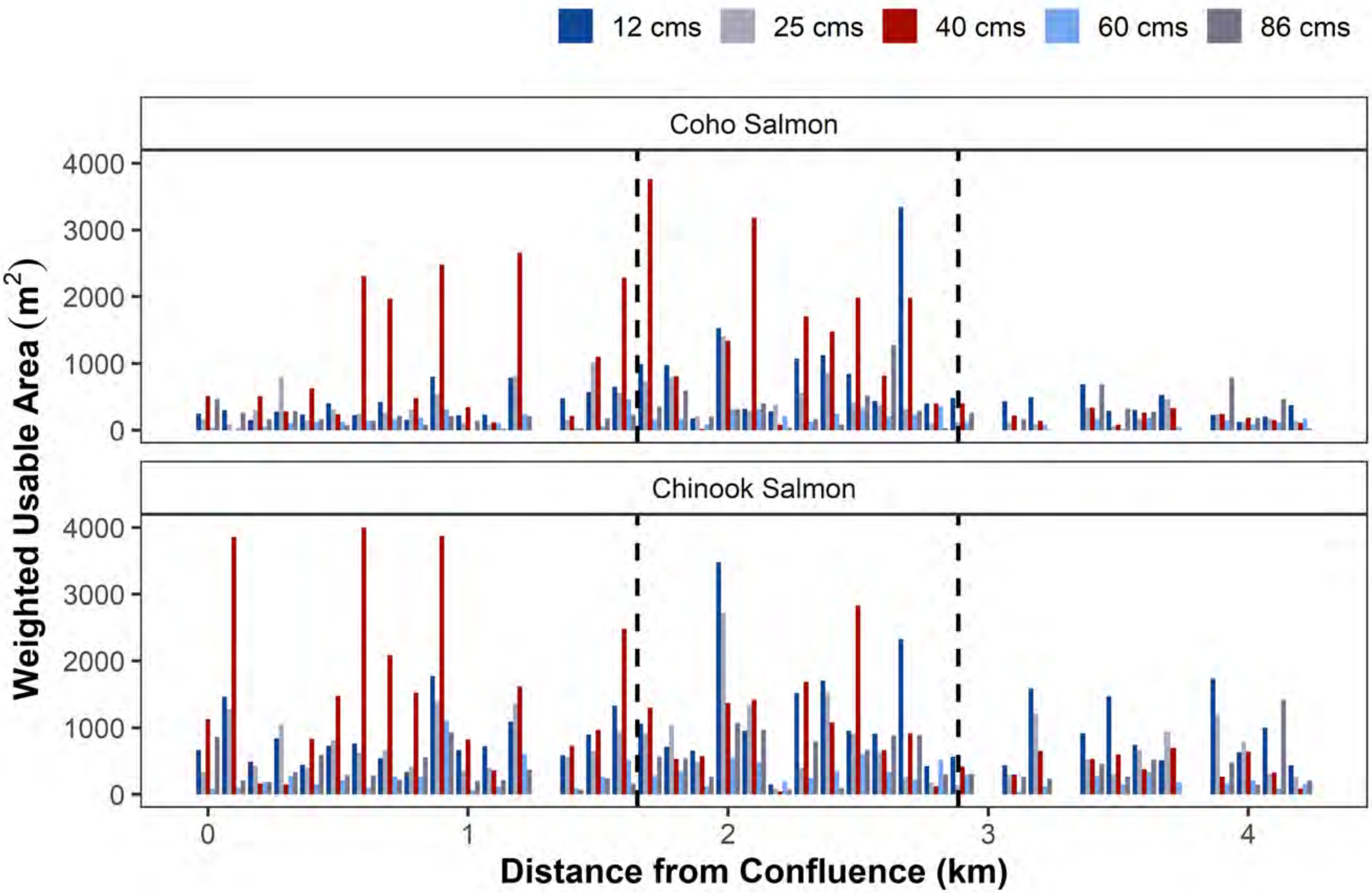


Figure 3.6 Total weighted useable area (WUA in m²) in Seton River at various flow releases from Seton Dam for juvenile Coho Salmon and Chinook Salmon. Dashed lines indicate reach breaks.

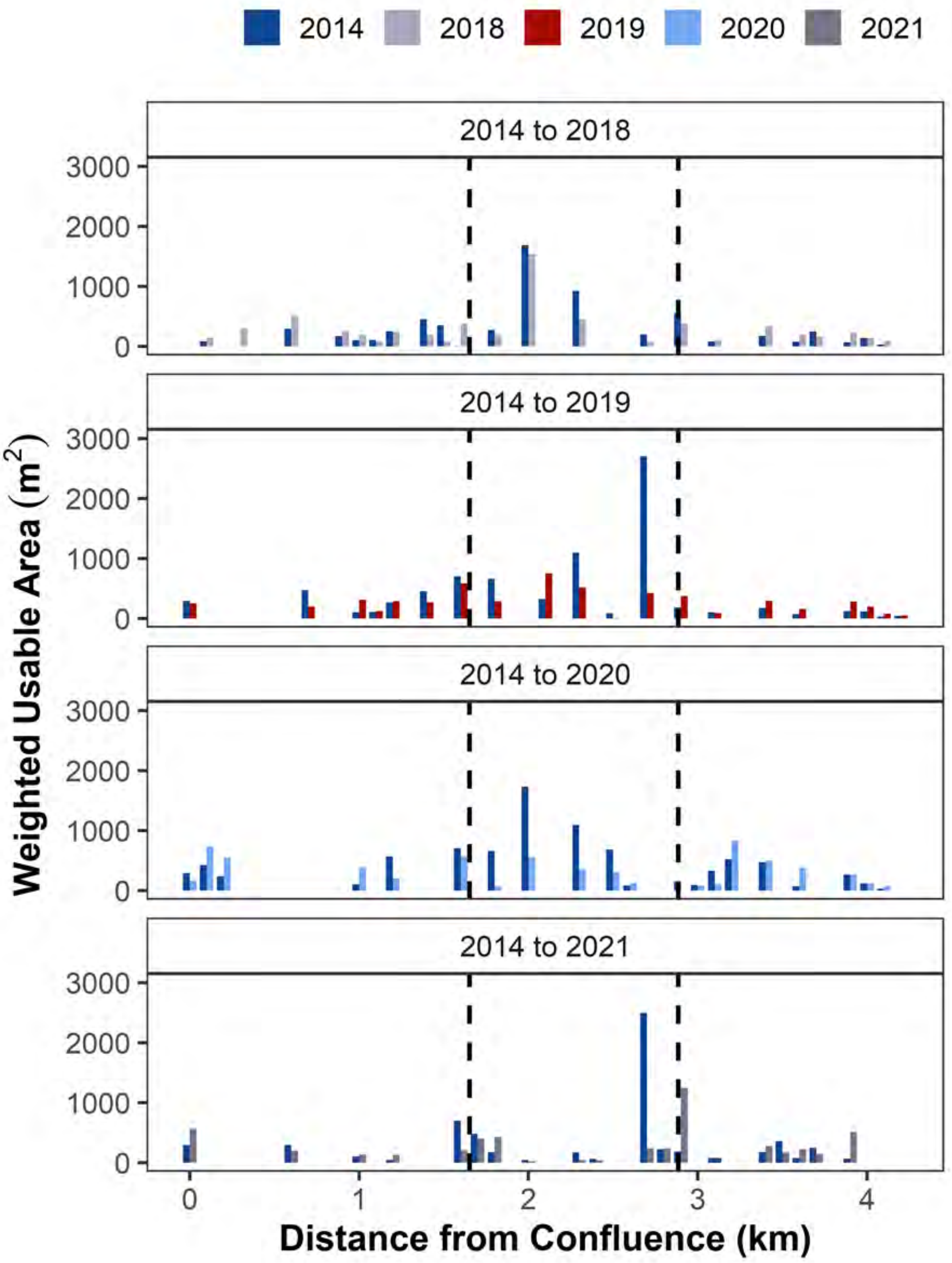


Figure 3.7 Total weighted useable area (WUA in m^2) for Rainbow Trout fry in the Seton River (at baseflow conditions of ~12–14 m^3/s) for matching sites between 2014 and 2018, 2014 and 2019, 2014 and 2020, and 2014 and 2021. Dashed lines indicate reach breaks.

3.1.4 Riverbed Elevation

Analysis of topographic data showed that that the riverbed within the spawning habitat area downstream of Seton Dam has experienced more deposition than scour since 2013; however, topographic change was spatially and temporally variable between survey periods (Table 3.1; Figure 3.8; Figure 3.9). The majority of the 8,300 m^2 area was between the -0.06 to 0.06 m elevation change level of uncertainty during all six periods and was not considered in this analysis. By area, there was slightly more deposition than scour during the 2013 to 2015 and 2015 to 2016 periods, more scour than deposition during the 2016 to 2017, 2019 to 2020, and 2020 to 2021 period, and much more deposition than scour during the 2017 to 2019 period. Net changes in volume for each survey period followed a similar pattern, except that the volume of scour and deposition were nearly equal during the 2015 to 2016 and 2019 to 2020 periods. Visually, areas of deposition and scour were spread relatively evenly throughout the channel.

Table 3.1 Areas and volumes of deposition and scour during each period between topographic surveys.

Period	Deposition Area (m^2)	Scour Area (m^2)	Net Area (m^2)	Deposition Volume (m^3)	Scour Volume (m^3)	Net Volume (m^3)
2013 to 2015	878	713	165	85	70	15
2015 to 2016	634	560	74	55	56	-1
2016 to 2017	465	1158	-693	39	123	-84
2017 to 2019	1795	302	1493	188	25	163
2019 to 2020	573	632	-58	50	53	-3
2020 to 2021	530	793	-263	45	70	-25
2013 to 2021	1526	579	947	140	63	77

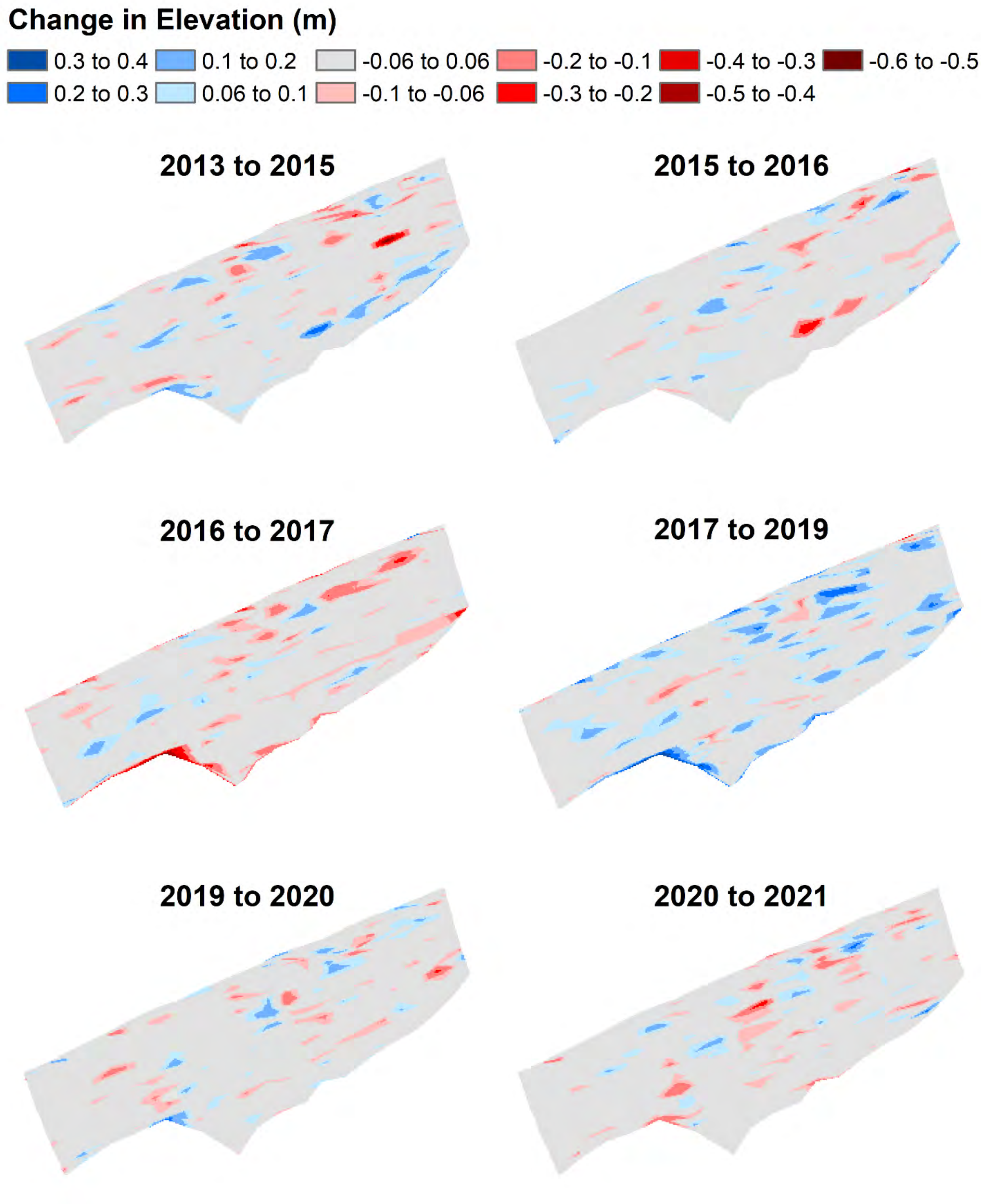


Figure 3.8 Digital elevation models of differences between topographic surveys within the main salmon spawning area of the Seton River, outside of the spawning channels. Blue represents deposition, while red represents scour.

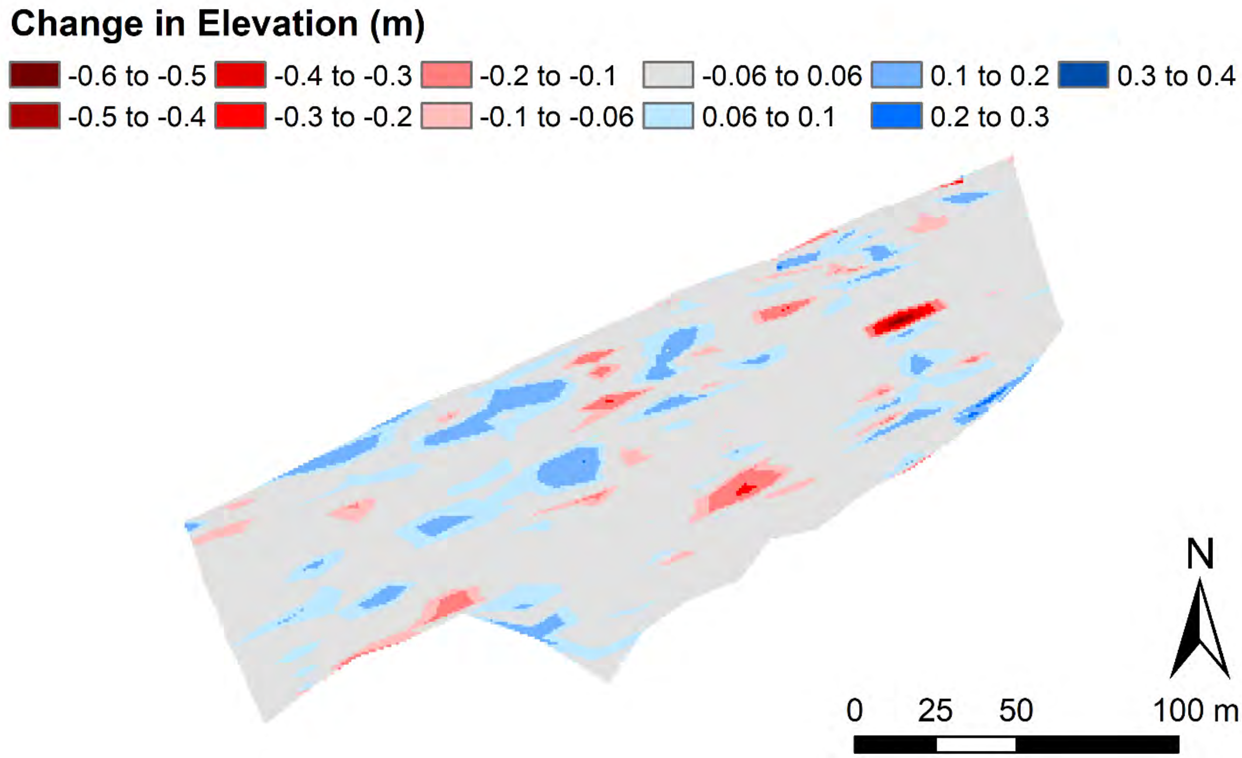


Figure 3.9 Digital elevation model of net difference between the 2013 and 2021 topographic surveys within the main salmon spawning area of the Seton River, outside of the spawning channels. Blue represents deposition, while red represents scour.

3.1.5 Substrate Size

Analysis of pebble count data from the spawning area showed a general increase in substrate size from 2015 to 2017, followed by a decrease from 2017 to 2020 (Table 3.2). The most upstream transect (G1B) increased in substrate size from 2015 to 2017, decreased from 2017 to 2019, then slightly increased in 2020. The second most upstream transect (G1D) slightly decreased in substrate size from 2015 to 2016, increased and peaked in 2017, and then decreased from 2017 to 2020. The third transect (G1F) increased in substrate size from 2015 to 2017, then decreased from 2017 to 2020. The most downstream transect (G1G) slightly decreased in substrate size from 2015 to 2016, increased in 2017, decreased from 2017 to 2019, slightly increased from 2019 to 2020, and then increased and peaked in 2021.

Pebble counts from 2018 through 2021 showed variable changes in substrate throughout the Seton River (Figure 3.10). Substrate sizes in Reach 1 appeared to be relatively consistent between 2018 to 2020 with a slight increase in size in 2021. Substrate sizes in Reaches 2 and 3 appeared to be increasing between 2018 and 2020 but returned to 2018/2019 levels in 2021.

Table 3.2 Geometric mean and geometric standard deviation of substrate particle size (mm) as measured using Wolman pebble counts at four transects in the Seton River from 2015 to 2021.

Transect	2015	2016	2017	2018	2019	2020	2021
G1B	11 ± 2.0	33 ± 1.8	41 ± 1.5	NA	18 ± 5.5	28 ± 3.7	NA
G1D	27 ± 2.6	23 ± 1.9	45 ± 2.5	NA	34 ± 4.8	24 ± 4.1	NA
G1F	25 ± 2.1	NA	55 ± 2.3	NA	NA	24 ± 4.2	NA
G1G	36 ± 1.7	28 ± 1.7	42 ± 2.0	NA	28 ± 4.1	29 ± 4.5	50 ± 3.1
All	23 ± 2.4	28 ± 1.8	45 ± 2.1	NA	26 ± 4.9	26 ± 4.1	NA

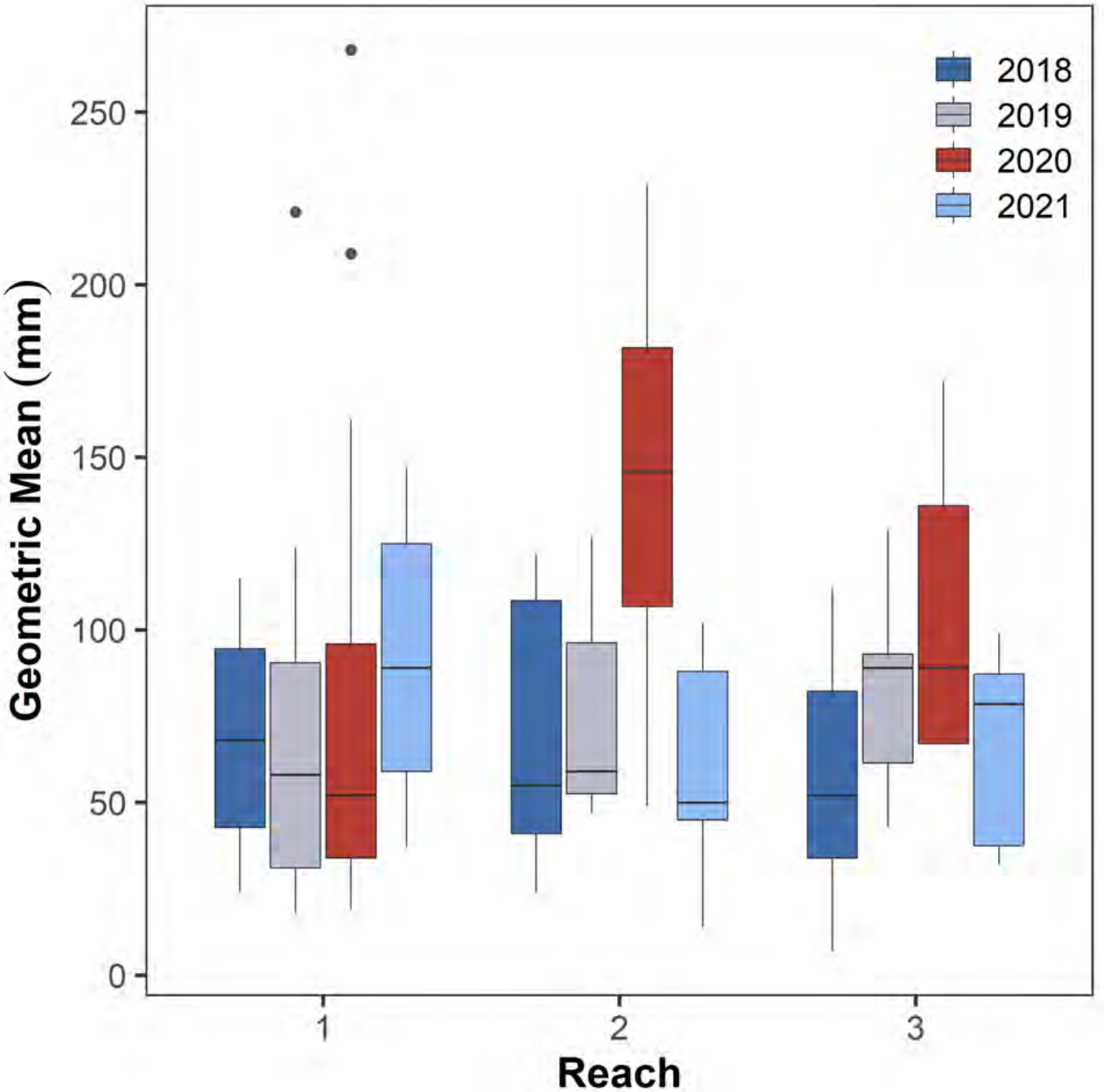


Figure 3.10 Boxplots of substrate sizes from pebble count surveys conducted in 2018, 2019, 2020, and 2021. Solid lines denote the annual median geometric mean substrate size, boxes represent the interquartile range (IQR). Vertical lines represent the range excluding outliers, which are shown individually as points.

3.2 Juvenile & Resident Fish

The approach and overall effort for sampling juvenile and resident fish has remained consistent over the study period, though program delivery was necessarily adjusted to account for high flows (largely as a results of Modified Operations) and in-season effort varied depending on the conditions at the time of sampling (Table 3.3). Total catch in 2021 was similar to previous years (Table 3.4). Total catch of Coho and Chinook Salmon in 2021 was greater than the average for 2014–2020 by 60% and 150%, respectively, while the total Rainbow Trout catch was lower by approximately 45% from average.

Table 3.3 Summary of sampling effort (including total number of sites [n]) during biological and abundance surveys targeting juvenile and resident fish within the Seton River and spawning channels from 2014 to 2021. For the biological surveys in 2014 to 2018, sampling sites in the lower and upper spawning channels were combined into one site each (i.e., LSC, USC) regardless of the different locations that were sampled. For the biological surveys in 2016 to 2021, side channels were sampled during high flow events, if wetted.

Year	Program	Method	Type	n	Mean Distance (m)	Mean Time
2014	Abundance surveys	Electrofishing	M-R	6	60	-
		Electrofishing	Index	25	54	-
		Snorkeling	Index	-	-	-
	Biological surveys*	Electrofishing	Index	16	79	-
2015	Abundance surveys	Electrofishing	M-R	4	48	1,141 s
		Electrofishing	Index	23	50	416 s
		Snorkeling	Index	10	50	-
	Biological surveys*	Electrofishing	Index	9	49	-
2016	Abundance surveys	Electrofishing	M-R	5	56	1,354 s
		Electrofishing	Index	23	50	744 s
		Snorkeling	Index	20	48	-
	Biological surveys*	Electrofishing	Index	11	54	1,080 s
2017	Abundance surveys	Electrofishing	M-R	6	52	841 s
		Electrofishing	Index	24	50	469 s
		Snorkeling	Index	20	48	-
	Biological surveys*	Electrofishing	Index	10	47	602 s
2018	Abundance surveys	Electrofishing	M-R	6	52	870 s
		Electrofishing	Index	21	43	502 s
		Snorkeling	Index	20	47	-
	Biological surveys*	Electrofishing	Index	13	42	529 s
2019	Abundance surveys	Electrofishing	M-R	4	49	**
		Electrofishing	Index	25	46	**
		Snorkeling	Index	-	-	-
	Biological surveys*	Electrofishing	Index	15	34	619 s

Year	Program	Method	Type	n	Mean Distance (m)	Mean Time
2020	Abundance surveys	Electrofishing	M-R	4	52	1,133 s
		Electrofishing	Index	25	48	391 s
		Snorkeling	Index	20	25	27 m
	Biological surveys*	Electrofishing	Index	19	48	379 s
		Minnow trapping†	Index	20	NA	19.5 h
2021	Abundance surveys	Electrofishing	M-R	5	50	832 s
		Electrofishing	Index	28	49	791 s
		Snorkeling	Index	20	46	13 m
	Biological surveys*	Electrofishing	Index	24	46	**

*Total possible sites (n) were: 16 in 2014-2015; 20 in 2016-2018; 24 in 2019; 22 in 2020; and 24 in 2021.

**Clock broke on electrofisher; sampling start and end times at each site were recorded but are not reported.

†Minnow trapping was conducted in April 2020 in response to the COVID-19 pandemic; remainder of biological surveys were conducted using electrofishing methods (with an adjusted crew size of 2 people, rather than the standard 3 people).

Table 3.4 Fish catch by species within the Seton River and spawning channels from 2014 to 2020. Species include Bull Trout (BT), sculpin spp. (CC), Chinook Salmon (CH), Coho Salmon (CO), dace (DC), lamprey (L), Mountain Whitefish (MW), Northern Pikeminnow (NSC), Peamouth Chub (PCC), Pink Salmon (PK), Rainbow Trout (RB), Redside Shiner (RSC), Sockeye Salmon (SK), and sucker (SU).

Year	BT	CC	CH	CO	DC	L	MW	NSC	PCC	PK	RB	RSC	SK	SU	Total
2014	4	182	22	674	400	0	14	0	0	36	1,377	59	6	30	2,804
2015	1	302	197	447	484	0	7	0	1	0	664	14	24	47	2,188
2016	1	119	211	143	111	2	6	16	6	0	684	19	4	12	1,334
2017	5	395	298	279	565	1	1	0	0	0	864	41	2	38	2,489
2018	4	431	120	457	804	0	0	0	0	5	968	72	0	166	3,027
2019	13	483	70	701	488	0	6	0	2	0	869	51	40	108	2,832
2020	1	334	117	191	200	0	1	0	0	1	386	14	1	72	1,318
2021	2	285	370	672	339	0	7	5	0	0	460	32	2	58	2,232

3.2.1 Biological Characteristics

Size-at-Age

Of the salmonids, only Rainbow Trout, Coho Salmon, and Chinook Salmon were captured in sufficient numbers to show the presence of discrete size classes for an Age-Length Key (ALK). Rainbow Trout up to age 6 have been captured in the study area, though individuals greater than age 3 were rare (n = 13 from 2014 to 2021). Fork length distributions for three age classes, age-0 to age-2, demonstrated positive monthly growth from March to October (Figure 3.11) and suggest that the ALKs adequately estimated age for juvenile Rainbow Trout. Growth in the older age classes was not as evident.

Three age classes of Coho Salmon were identified; age-0 fish were the most frequently captured followed by age-1 and then age-2. Fork length distributions demonstrated positive monthly growth from March to October at age 0 with the growth trajectory continuing at age 1 the following spring (Figure 3.12). Fork length distributions showed a size reduction in the age-1 Coho Salmon being captured between June and July followed by positive monthly growth from July at age 1 to until June at age 2 (Figure 3.12). Given the life history strategy of this species, it is assumed that larger age-1 Coho Salmon migrated out of the river as smolts, while the smaller age-1 cohort remained in the study area for a second year and this size-class was reflected in the capture data following the migration period. Data suggest that the ALKs adequately estimated age for juvenile Coho Salmon.

Overall, the age classes and growth patterns of Chinook Salmon were similar to Coho Salmon; however, the sample sizes, particularly for age-1 (n = 129) and age-2 fish (n = 34) were low. Low captures of age-1 (from July to October) and age-2 Chinook Salmon made it difficult for the ALK to partition fish with larger fork lengths; however, even with the small sample size, there was not a noticeable effect on monthly growth trajectories (Figure 3.13). Positive monthly growth was observed in age-0 fish which continued until presumptive smolting in June at age 1; smaller age-1 Chinook Salmon that remained in the study area continued to demonstrate positive monthly growth until the following spring at age-2.

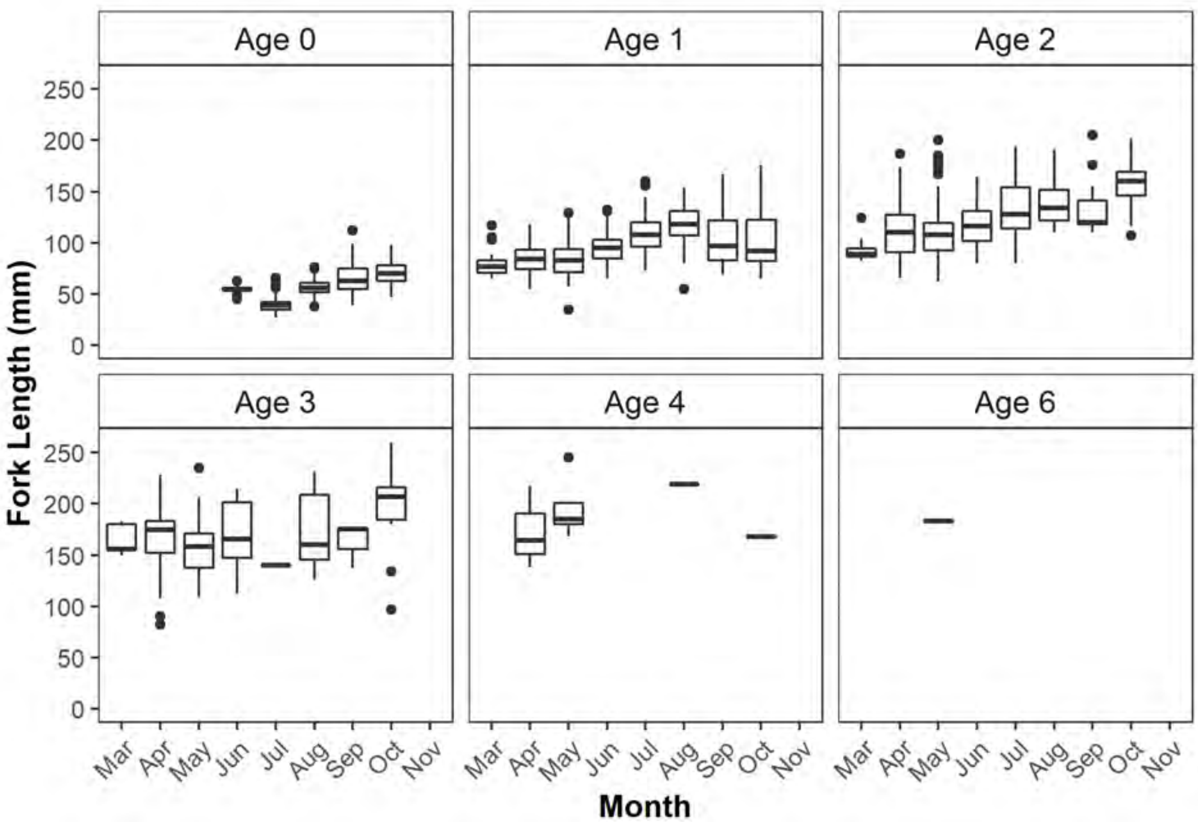


Figure 3.11 Boxplots of fork length (mm) by age class for Rainbow Trout captured in the Seton River and spawning channels from 2014 to 2021. Solid lines denote the median fork length, boxes represent the interquartile range. Vertical lines represent the range excluding outliers, which are shown individually as points.

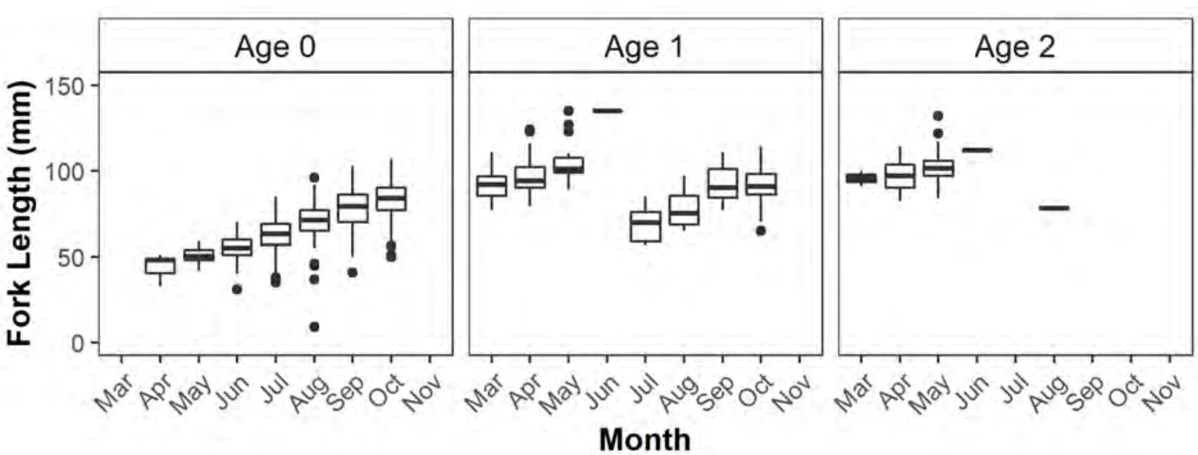


Figure 3.12 Boxplots of fork length (mm) by age class for juvenile Coho Salmon captured in the Seton River and spawning channels from 2014 to 2021. Solid lines denote the median fork length, boxes represent the interquartile range. Vertical lines represent the range excluding outliers, which are shown individually as points.

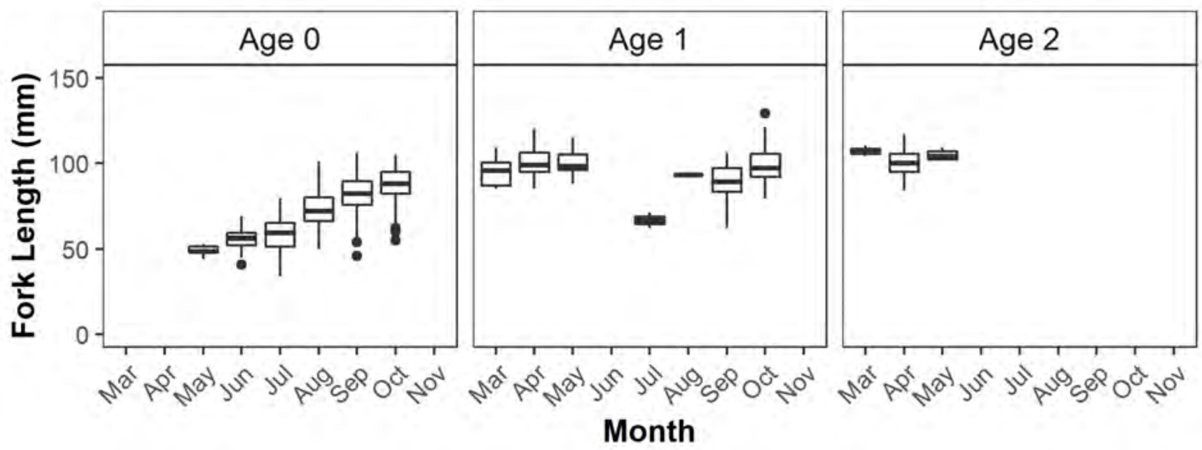


Figure 3.13 Boxplots of fork length (mm) by age class for juvenile Chinook Salmon captured in the Seton River and spawning channels from 2014 to 2021. Solid lines denote the median fork length, boxes represent the interquartile range (IQR). Vertical lines represent the range excluding outliers, which are shown individually as points.

Body Condition

We assessed the age-specific effect of year (as a proxy for flow conditions) and reach on mean body condition, specifically K_f, using a MANOVA and AICc model selection approach. We limited our analysis to age-0 and age-1 Rainbow Trout, and to age-0 Coho and Chinook Salmon.

Both age classes of Rainbow Trout had year and reach (without interactions) as the top model of body condition (Appendix 7-1, Appendix 7-2). For the age-0 model, year ($F_{7,\ 2824}$ = 3.2, P < 0.001) and reach ($F_{4,\ 2827}$ = 6.9, P < 0.001) were significant predictors of condition. Year ($F_{7,\ 823}$ = 2.9, P = 0.001) and reach ($F_{4,\ 826}$ = 3.41, P < 0.001) were also significant predictors of condition for the age-1 model. Tukey tests indicated that mean K_f values for age-0 Rainbow Trout were significantly greater in Reach 1 compared to the upper spawning channel (P = 0.02) and Reach 3 of the mainstem (P < 0.001; Figure 3.14). In previous years, the results comparing Reach 1 to Reach 2 indicated a statistically significant difference (Buchanan et al. 2020). Juvenile Rainbow Trout captured in 2019 for both age classes had generally greater body condition compared to other study years (Figure 3.14, Figure 3.15). Model results overall were not as strong for age-1 Rainbow Trout. Still, unlike for age-0 fish, mean condition of age-1 Rainbow Trout in the upper spawning channel were greater than those in mainstem sites (specifically, Reach 1 and 3).

For age-0 Coho Salmon, the most complex model with year ($F_{7,\ 2028}$ = 20.4, P < 0.001), reach ($F_{4,\ 20280}$ = 1.1, P = 0.34) and an interaction term ($F_{28,\ 2028}$ = 3.7, P < 0.001) had the lowest AICc value (Appendix 7-3); all other models tested had AICc difference values >38 indicating low support. Tukey tests suggested that juvenile Coho Salmon captured in the lower spawning channel had significantly greater K_f values compared to those in Reach 1 (P = 0.07; Figure 3.16). There were also significant differences in condition among years with juveniles in 2017 being a particularly strong cohort (Figure 3.16).

For age-0 Chinook Salmon, the intercept-only model was the top model, followed by the reach only model (ΔAICc = 3.3) and then the year only model (ΔAICc = 4.1, Appendix 7-4). However, none of the terms were a significant predictor of body condition in age-0 Chinook Salmon. Increased sample sizes may provide better results. Overall, the intercept-only model remained as the top model suggesting that neither year, reach, nor their interactions affected the mean body condition of juvenile Chinook Salmon in the Seton River (Figure 3.17). No further statistical testing was conducted post-hoc.

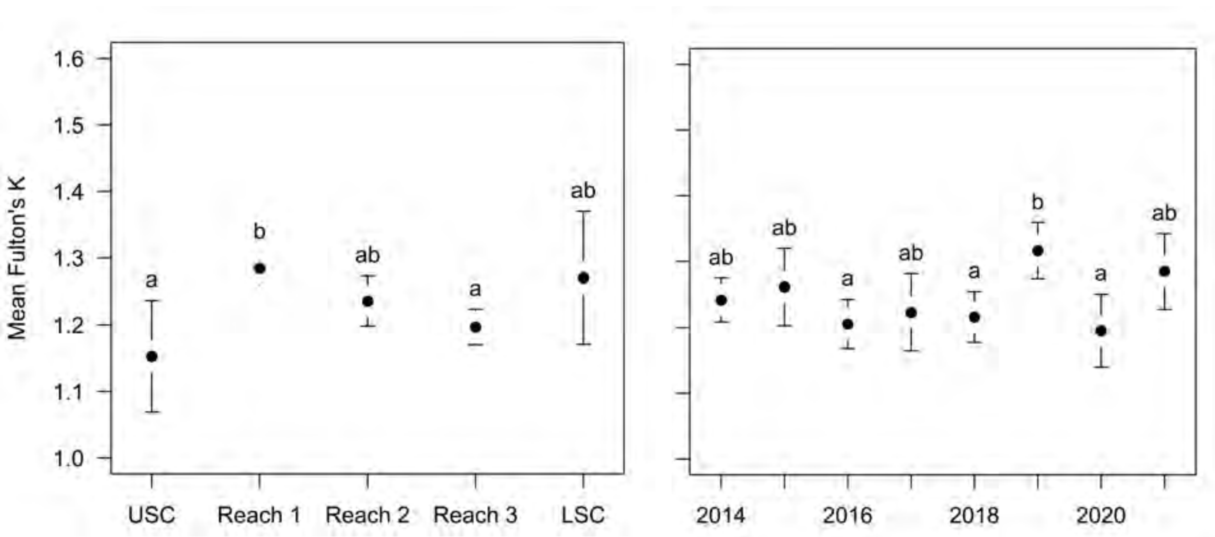


Figure 3.14 Mean condition factor with 95% confidence intervals of age-0 Rainbow Trout by reach (left panel) and year (right panel) from 2014–2021. Points within the same panel that do not share the same letter are statistically different from each other.

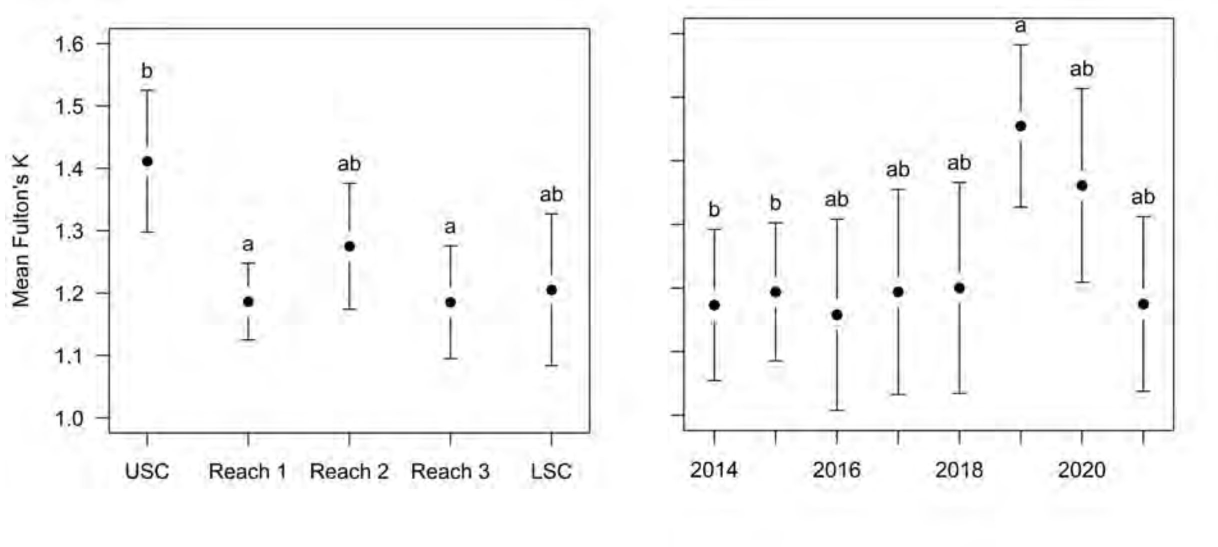


Figure 3.15 Mean condition factor with 95% confidence intervals of age-1 Rainbow Trout by reach (left panel) and year (right panel) from 2014–2021. Points within the same panel that do not share the same letter are statistically different from each other.

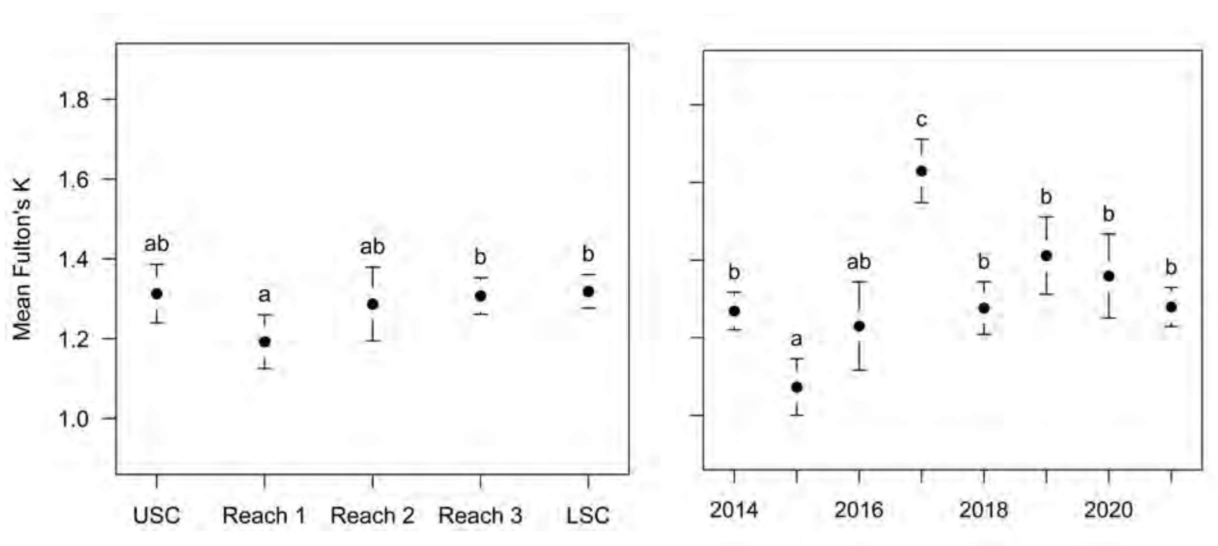


Figure 3.16 Mean condition factor with 95% confidence intervals of age-0 Coho Salmon by reach (left panel) and year (right panel) from 2014–2021. Points within the same panel that do not share the same letter are statistically different from each other.

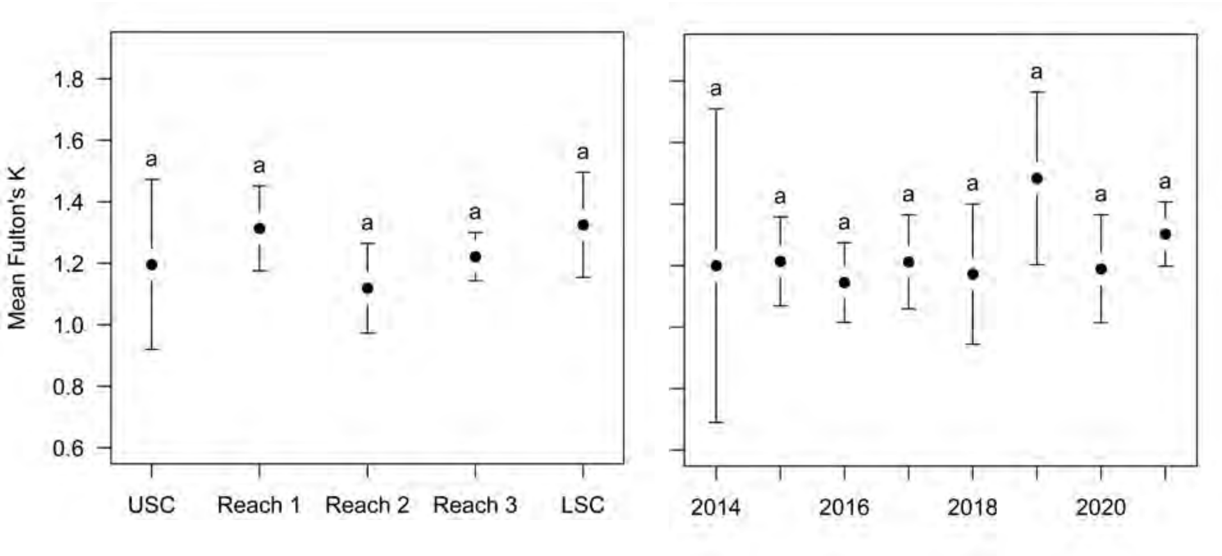


Figure 3.17 Mean condition factor with 95% confidence intervals of age-0 Chinook Salmon by reach (left panel) and year (right panel) from 2014–2021. Points within the same panel that do not share the same letter are statistically different from each other.

Stock Origin

Results of the DNA analysis for 2019, 2020, and 2021 samples taken from juvenile Chinook Salmon and Coho Salmon had not been received from the laboratory at the time of writing. The most recent results will be incorporated into future reports. As reported in Buchanan et al. (2020), DNA analysis for the 2016 to 2018 samples (n = 207) from juvenile Chinook Salmon indicated the most numerous stock groupings were Seton River/Portage Creek (n = 99; current molecular methods cannot distinguish Portage Creek from Seton River), Stuart (n = 30), Quesnel (n = 25), Nechako (n = 16), and Chilko (n = 14). An additional 23 samples were of Chinook Salmon from other watersheds (Buchanan et al. 2020). No Bridge River Chinook Salmon were captured in the Seton River or spawning channels. Stock proportions varied throughout the year. Seton River/Portage Creek fish were dominant in May and June; juvenile Chinook Salmon from other watersheds were dominant in August to October (Figure 3.18).

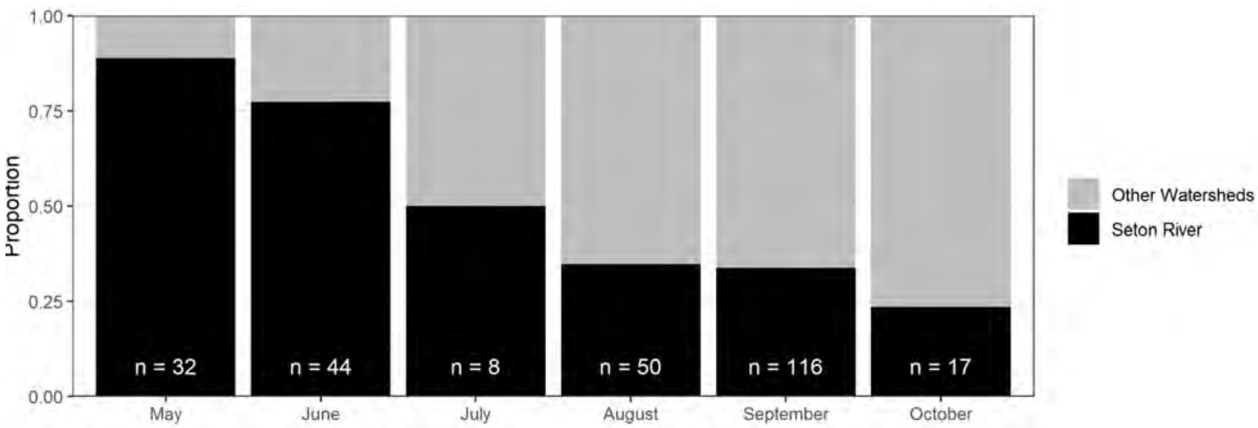


Figure 3.18 Proportion of juvenile Chinook Salmon identified as Seton River/Portage Creek origin relative to those of all other populations as determined through DNA analysis of tissue samples taken from captured fish monthly, 2016-2018 (2019-2021 results pending).

3.2.2 Distribution

Capture locations of juvenile fish varied by species and age class. In all years, age-0 Rainbow Trout were the most frequently captured and were primarily caught in the mainstem of the Seton River (Figure 3.19). Captures of age-1 and older Rainbow Trout were well distributed between the mainstem river and spawning channels. Juvenile Coho Salmon were distributed among the spawning channels and mainstem river, regardless of age class (Figure 3.20). Chinook Salmon, especially the age-0 fish, were most frequently captured within the mainstem river, and primarily in Reach 3; older juvenile Chinook Salmon were captured relatively equally between the mainstem river and spawning channels (Figure 3.21).

During 2021 field sampling, 228 PIT tags were implanted into juvenile and resident salmonids captured in the mainstem Seton River, with an additional 60 in the LSC and 54 in the USC. A total of 2,891 PIT tags have been deployed under BRGMON-9 since 2013. Additional PIT tags have been implanted through other BRGMON field programs. Tagged fish have been detected on a fixed PIT array or recaptured from 0 to 1,105 days (~3 years) following the initial capture event. In 2021, 21 unique tags (each representing an individual fish) were detected on the LSC PIT array (n = 14 RB, n = 6 CO, n = 1 CH), 44 tags were detected on the USC PIT array (n = 38 RB, n = 3 CO, n = 3 CH), and no tags were detected on the Seton Dam fishway array. Results for the Seton Dam fishway are reported below in Section 2.5.1.

In terms of detection efficiency, the USC array generally performed better than the LSC array (Table 3.5). Detection data for the LSC array, specifically during the spring smolt migration period of 2016 and 2017, would have been affected by a temporary fish fence installed to capture out-migrating smolts; the fence blocked movement of juvenile and adult fish into and out of the spawning channel (though the fence was modified in 2017 to allow upstream movement of adult fish; juvenile passage in 2017 was unknown). Alternate methods were used starting in 2018 to allow the free movement of adult and juvenile fish.

Fish movement data from the USC array showed two periods of increased detection activity; these were generally from April through June and then from September through November (Figure 3.22). The LSC array had a similar patter with lower detections; the exception was for 2019, which showed a single period of increased detection activity from June to October (Figure 3.22). There were no clear trends on directed movement in or out of the spawning channels, counts were nearly equal in all months. Determining directed movement periods was challenging, due to the resident behaviour of fish in this study. Direction could not be assigned for 59% of detections on the LSC array and 41% on the USC array. Incidences where direction could not be assigned resulted from: data from 2014 when the array was not designed to detect direction of movement; fish that were not detected on both antennas; and or fish that moved between the antennas numerous times, thus confusing the assignment of direction. A more detailed review of individual detection data in future reporting years may assist with interpretation of fish movements.

For all years and both spawning channels combined, Rainbow Trout accounted for most of the detections (n = 301) on the fixed PIT arrays, followed by Coho Salmon (n = 53) and then Chinook (n = 6). Movement data for all species was limited during January to March. Rainbow Trout movement activity occurred mainly from April to November with no distinct break; some movement also occurred in December. Bull Trout activity was like Rainbow Trout but with detections condensed from May to October and some activity in November and December. Rainbow Trout did not have clear patterns of directed movement in or out of the spawning channels. In total, 49% of Rainbow Trout detections could not be assigned direction. Coho Salmon movement had two distinct periods, one in the spring (March to June) and one later in the year from August to December. While 73% of Coho Salmon detections could not be assigned direction, movement out of the spawning channels was more frequent (17%).

Table 3.5 Detection efficiency of the fixed Passive Integrated Transponder (PIT) antenna arrays within the lower (LSC) and upper (USC) spawning channels of the Seton River from 2014 to 2021. Antenna 1 is always the downstream antenna, while Antenna 2 is always the upstream antenna.

Array	Year	Antenna	Detection Efficiency	Shared Detections	Detections on Array	Detections not on Array	Missed Detections
LSC	2014	1	NA	NA	12	NA	NA
		-	-	-	-	-	-
	2015	1	0.83	10	15	12	2
		2	0.67	10	12	15	5
	2016	1	0.14	1	3	7	6
		2	0.33	1	7	3	2
	2017	1	0.83	10	13	12	2
		2	0.77	10	12	13	3
	2018	1	0.20	1	6	5	4
		2	0.17	1	5	6	5
	2019	1	0.50	3	5	6	3
		2	0.60	3	6	5	2
	2019*	1	0.57	8	10	14	6
		2	0.80	8	14	10	2
	2020	1	0.70	7	24	10	3
		2	0.29	7	10	24	17
	2021	1	0.79	11	18	14	3
		2	0.61	11	14	18	7
USC	2015	1	0.39	12	13	31	19
		2	0.92	12	31	13	1
	2016	1	0.88	14	18	16	2
		2	0.78	14	16	18	4
	2017	1	1.00	15	18	15	0
		2	0.83	15	15	18	3
	2018	1	0.90	18	24	20	2
		2	0.75	18	20	24	6
	2019	1	0.76	41	46	54	13
		2	0.89	41	54	46	5
	2020	1	0.73	35	37	48	13
		2	0.95	35	48	37	2
	2021	1	0.79	30	35	38	8
		2	0.86	30	38	35	5

*PIT reader was replaced, and array was moved to new location within the LSC on May 30, 2019; these lines are detection efficiency after these changes.

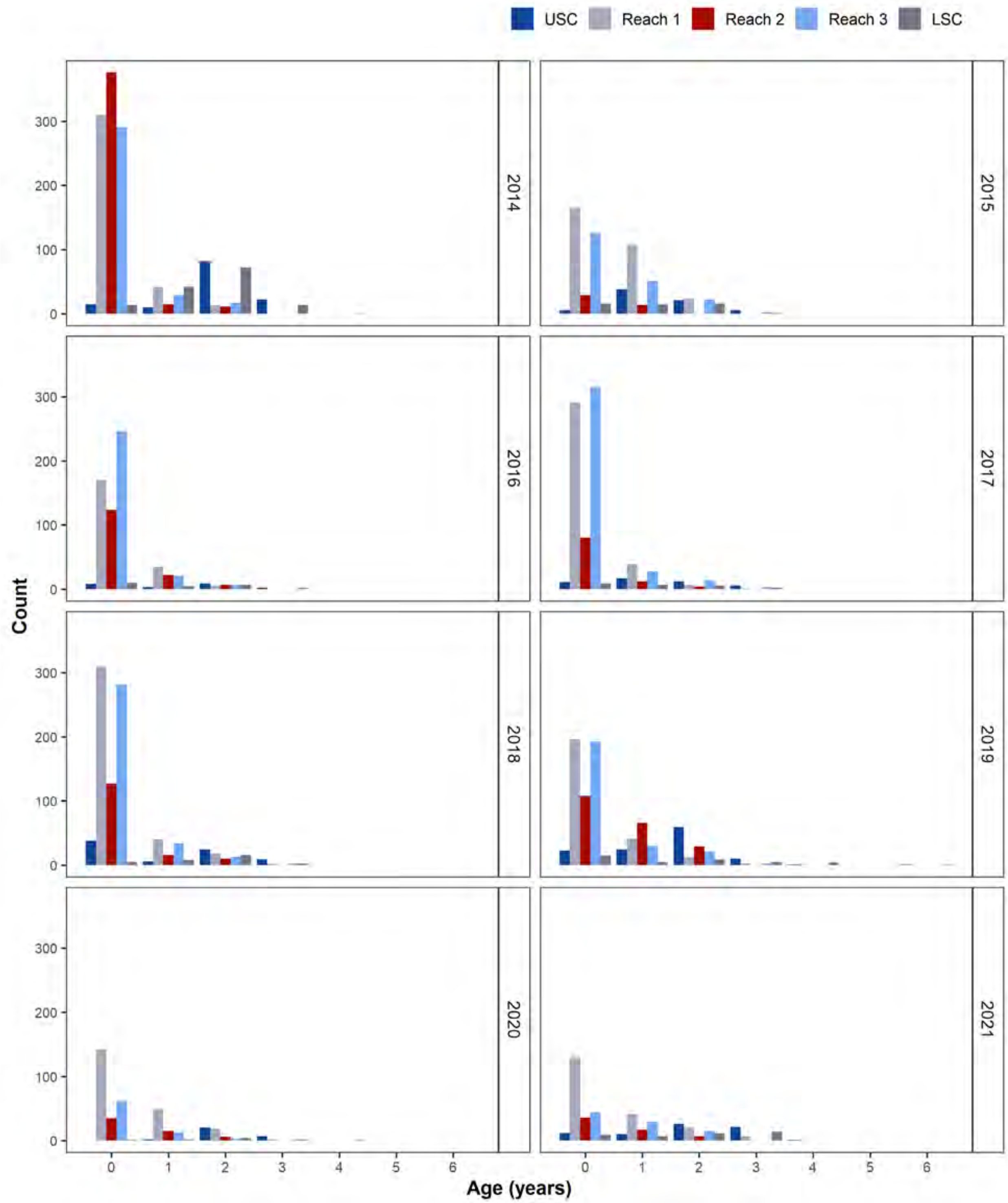


Figure 3.19 Number of Rainbow Trout collected from the Seton River mainstem (by reach) and spawning channels separated by age at capture.

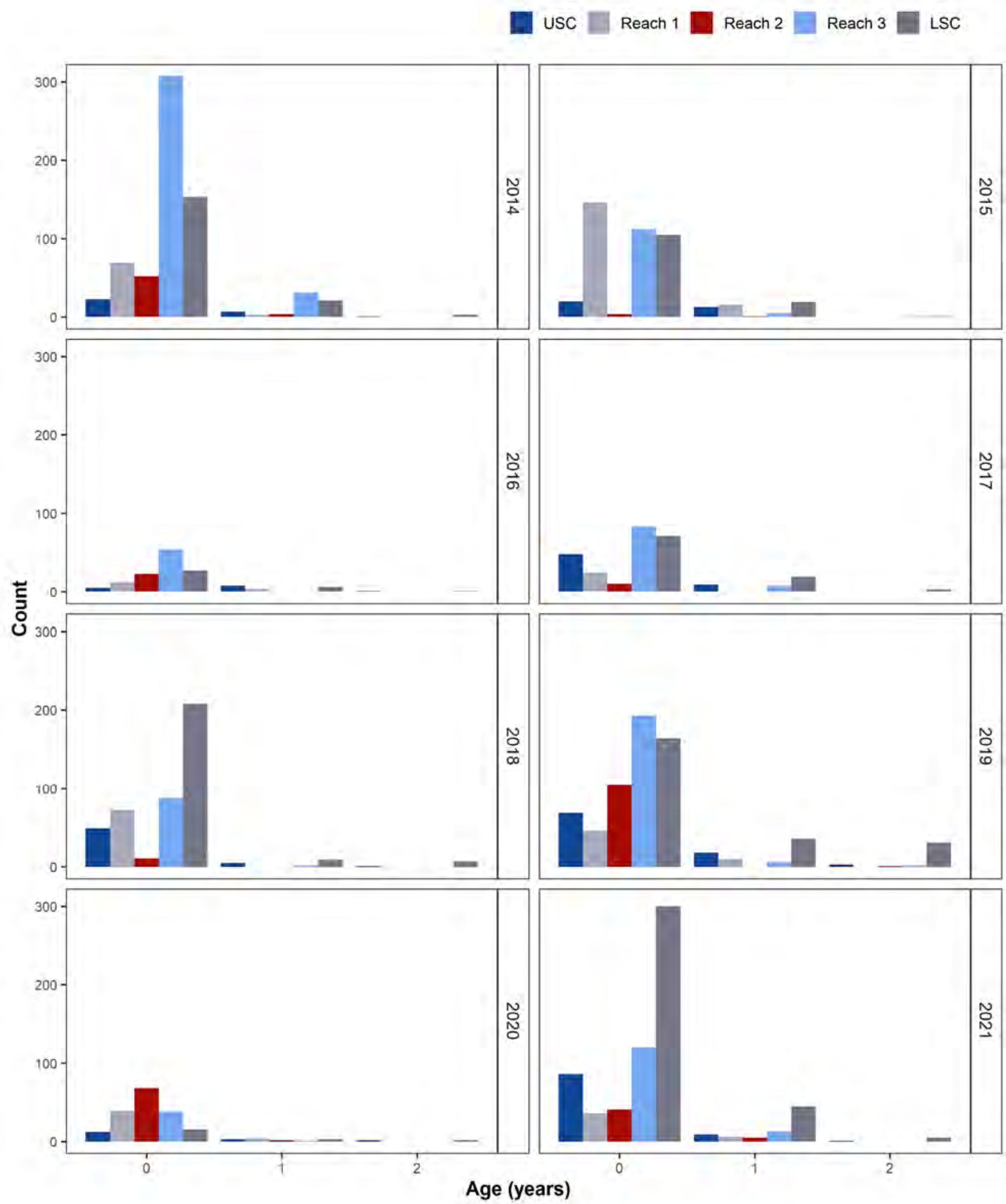


Figure 3.20 Number of Coho Salmon collected from the Seton River mainstem (by reach) and spawning channels separated by age at capture.

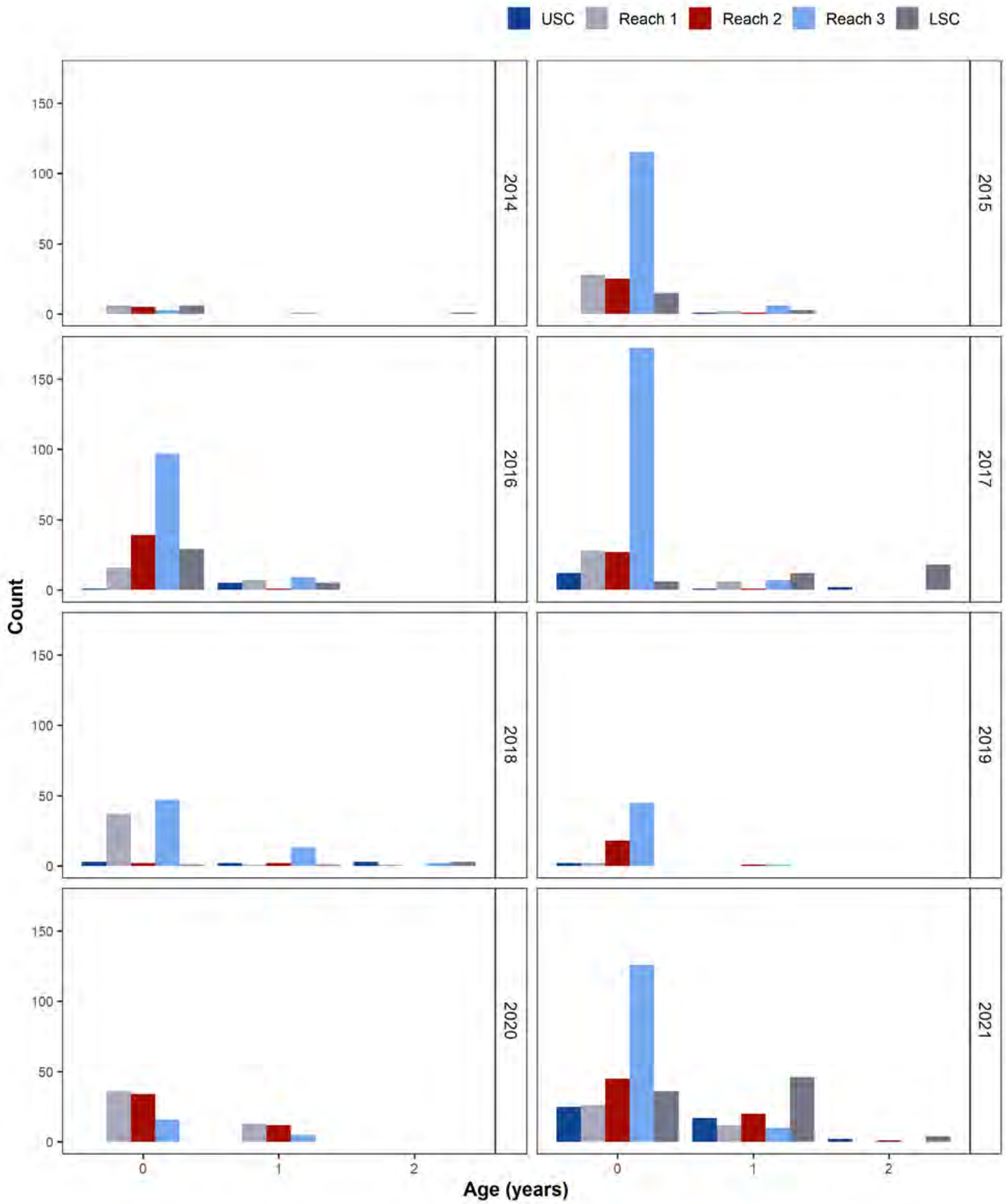


Figure 3.21 Number of Chinook Salmon collected from the Seton River mainstem (by reach) and spawning channels separated by age at capture.

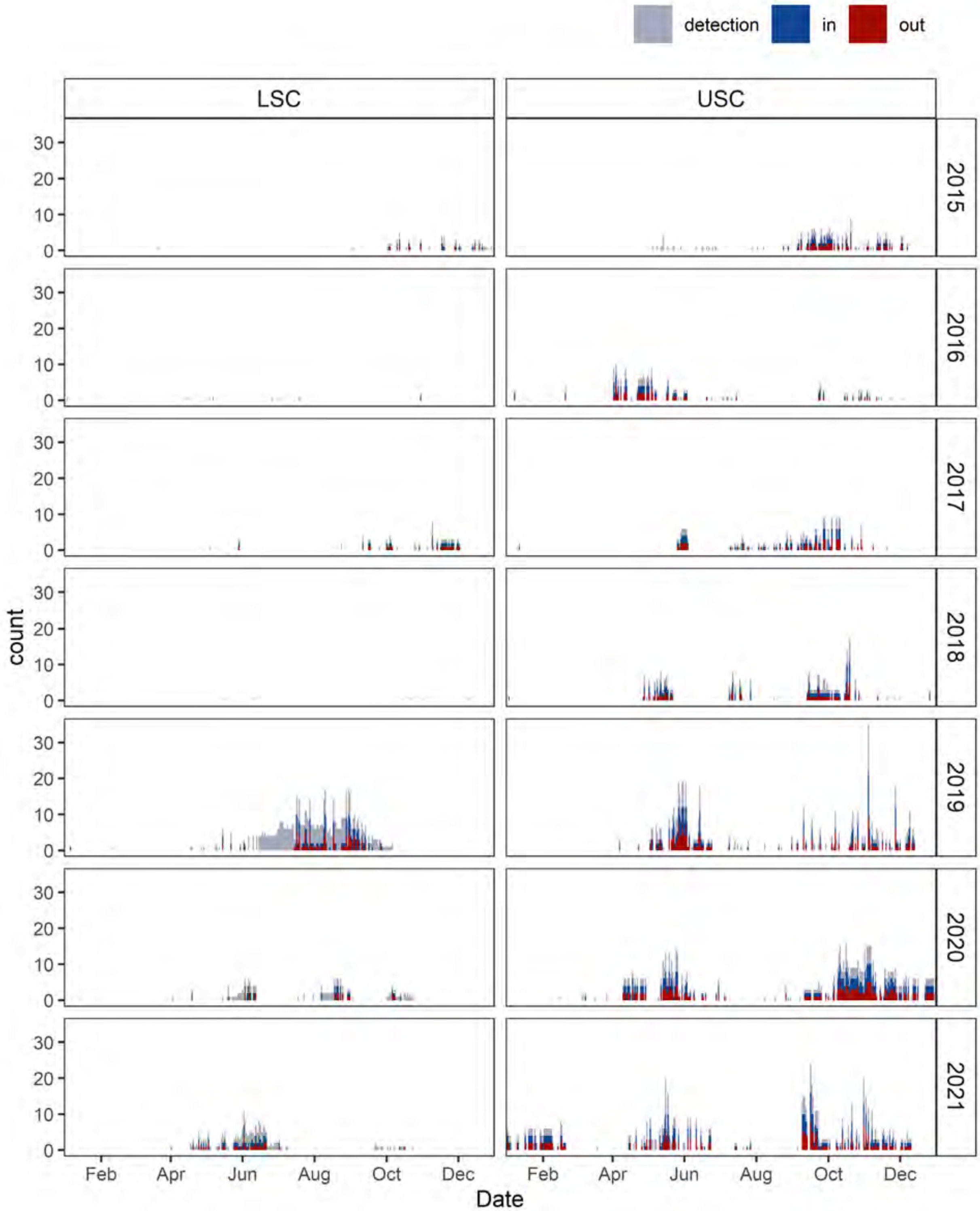


Figure 3.22 Counts of daily detections by direction of movement for passive integrated transponder (PIT) tagged juvenile and resident fish at the PIT antenna arrays in the lower (LSC) and upper (USC) spawning channels of the Seton River from 2015 to 2021. 2014 was removed as only one antenna was present in the LSC as apilot program and direction cannot be assigned to any records.

3.2.3 Abundance

During the spring of 2021, snorkel index sites (n = 20) were surveyed at ~15 m^3/s, which covered approximately 900 m (11 %) of the Seton River shoreline. The percentage of shoreline sampled in 2021 was the same as 2016–2018 but double what was completed in 2015 and 2020. This was a result of a method change in 2020, where 25 m site lengths were selected in combination with additional time spent at each site to catch fish and sample fish in an effort to increase PIT tags in age 2+ fish. Rainbow Trout were the most frequently observed (n = 45) and had an average estimated fork length of 124 mm (SD = 37; min = 80; max = 300); observed Rainbow Trout were within the size range expected for age 1 and older fish. Other species observed included Chinook Salmon (n = 7), Coho Salmon (n = 5), Pink Salmon (n = 38), Mountain Whitefish (n = 1), and sculpin (n = 2). During the surveys, crews attempted to capture target species using small aquarium nets. Overall, 31 Rainbow Trout, 4 Chinook Salmon and 3 Coho Salmon were captured from 13 of the 20 sites. All captured Rainbow Trout received a PIT tag; had a mean fork length of 108 mm (SD = 20; min = 75; max = 158) and were age 1 to 3 (median = 2). All captured Chinook Salmon were age 2 with a mean fork length of 110 mm (SD = 20; min = 75; max = 158). Two of the captured Coho Salmon were age 2 (FL = 90 and 112 mm) and one was age 1 (FL = 97 mm). Data from these fish were included in the analyses of biological characteristics and distribution. A mark-recapture experiment was not attempted, as sample sizes within each site were insufficient; thus, the abundance estimate included only Rainbow Trout sampled during fall electrofishing.

During the fall of 2021, electrofishing index sites (n = 16) covered approximately 16% of the total shoreline of the Seton River, which is consistent with the amount of shoreline sampled in previous years (Table 3.6). Only age-0 Rainbow Trout were captured in sufficient numbers to be used in the Bayesian hierarchical model. For mark-recapture sites (n = 5), the mean recapture rate was 23% (SD = 17), which was similar to previous study years (the lowest rate was 10% in 2015 and the highest rate was 35% in 2019; Table 3.6). The mean of the beta hyperparameter for detection probability estimated by the Bayesian hierarchical model for 2014 to 2021 was 0.25 (SD = 0.02), or a 25% detection probability (Figure 3.23).

The estimate of river-wide abundance for age-0 Rainbow Trout in 2021 was 1,839 fish with a 95% credible interval of 1,113 to 3,025 fish. Abundance in 2021 was the lowest since the study began and was close to the 2015 estimate. Although the 2014 abundance estimate was greater than in all other years, there is a high degree of uncertainty in this estimate due to variable densities observed during 2014 shoreline electrofishing (Figure 3.24, Figure 3.25). The hyper distribution of fish density for the Seton River in 2021 (mean density = 0.23 fish/m) is shown along with site-specific density estimates in Figure 3.26. The mean of the hyper distribution of fish density in 2021 was the lowest among the study years.

Table 3.6 Summary of recapture probabilities (recaptures/marks) calculated for mark-recapture sites, and percent of total shoreline sampled in the Seton River during juvenile abundance surveys from 2014 to 2021.

Year	Percent of Total Shoreline Sampled				Recapture Rate (%)	
	Reach 1	Reach 2	Reach 3	Entire River	Mean	SD
2014	18	23	10	17	29	8
2015	13	21	17	17	10	11
2016	14	25	10	16	27	15
2017	15	16	13	14	28	5
2018	8	6	18	11	21	9
2019	12	11	15	13	35	5
2020	17	15	12	15	22	16
2021	21	11	15	16	23	17

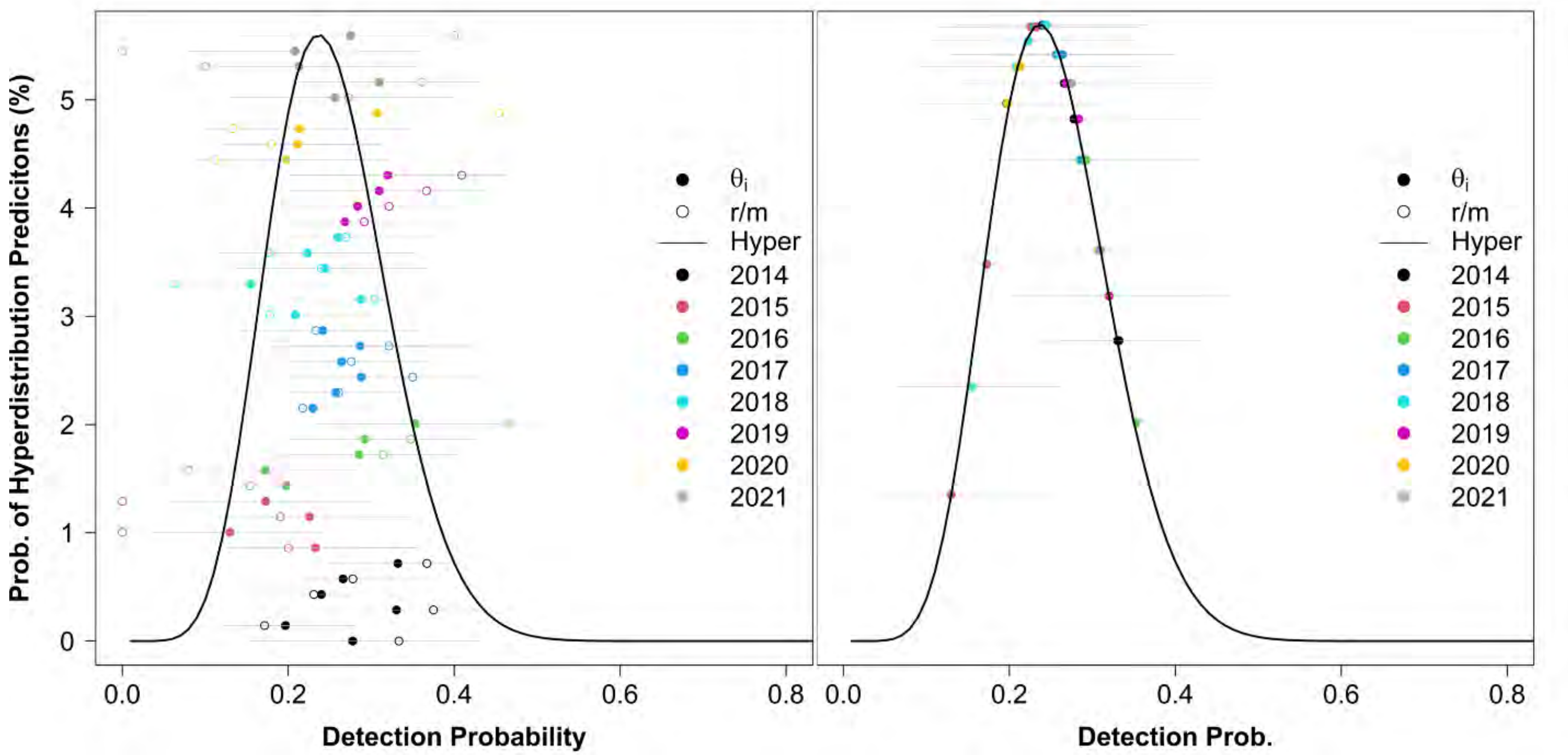


Figure 3.23 Parameter estimates from the hierarchical Bayesian model that estimates age-0 Rainbow Trout abundance. Shows the median hyperdistribution for detection probability (Hyper), as well as the median estimates of site-specific detection probability at mark-recapture sites with 95% credible intervals (θ_i) for each year from 2014 to 2021. The panel on the left shows each estimate ordered by year with an uninformative y-axis while the panel on the right shows each estimate fitted to the hyperdistribution.

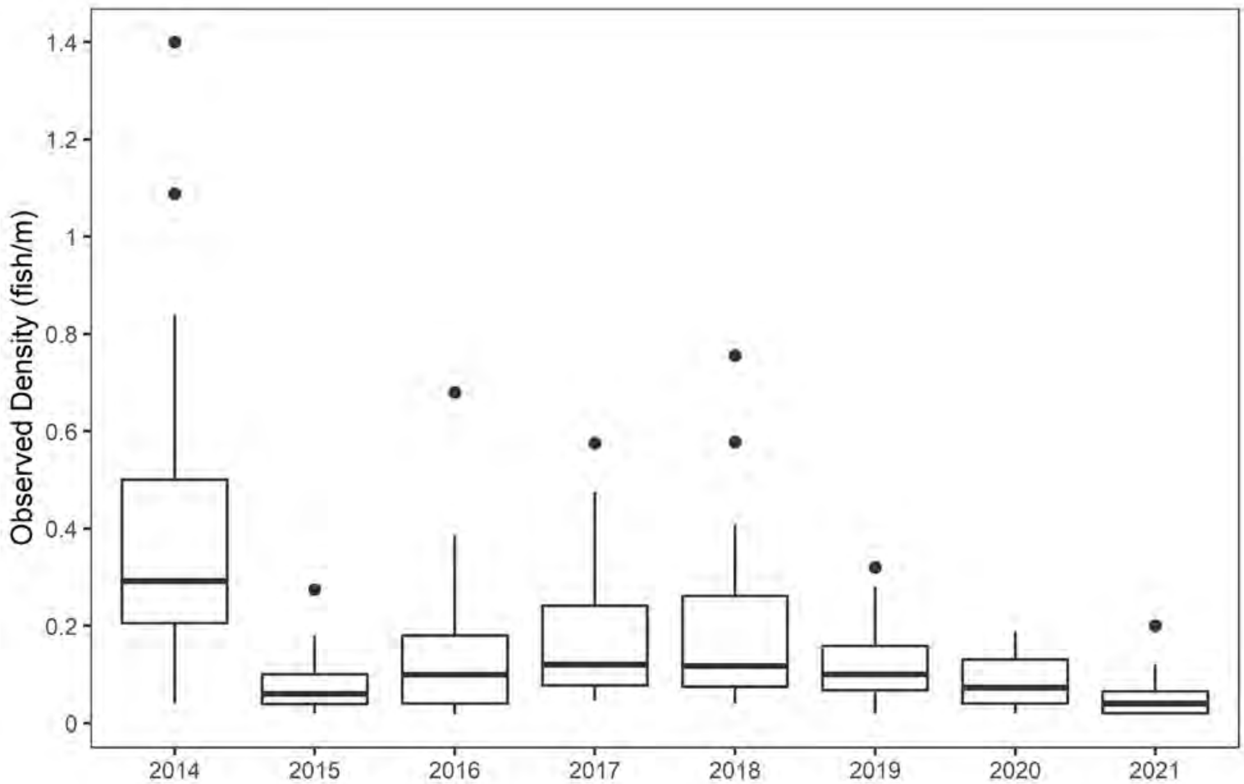


Figure 3.24 Density of age-0 Rainbow Trout (fish/m) directly calculated from shoreline electrofishing index sites (observed data) in the Seton River from 2014 to 2021. Solid lines denote the median observed density, boxes represent the interquartile range (IQR). Vertical lines represent the range excluding outliers, which are shown individually as points.

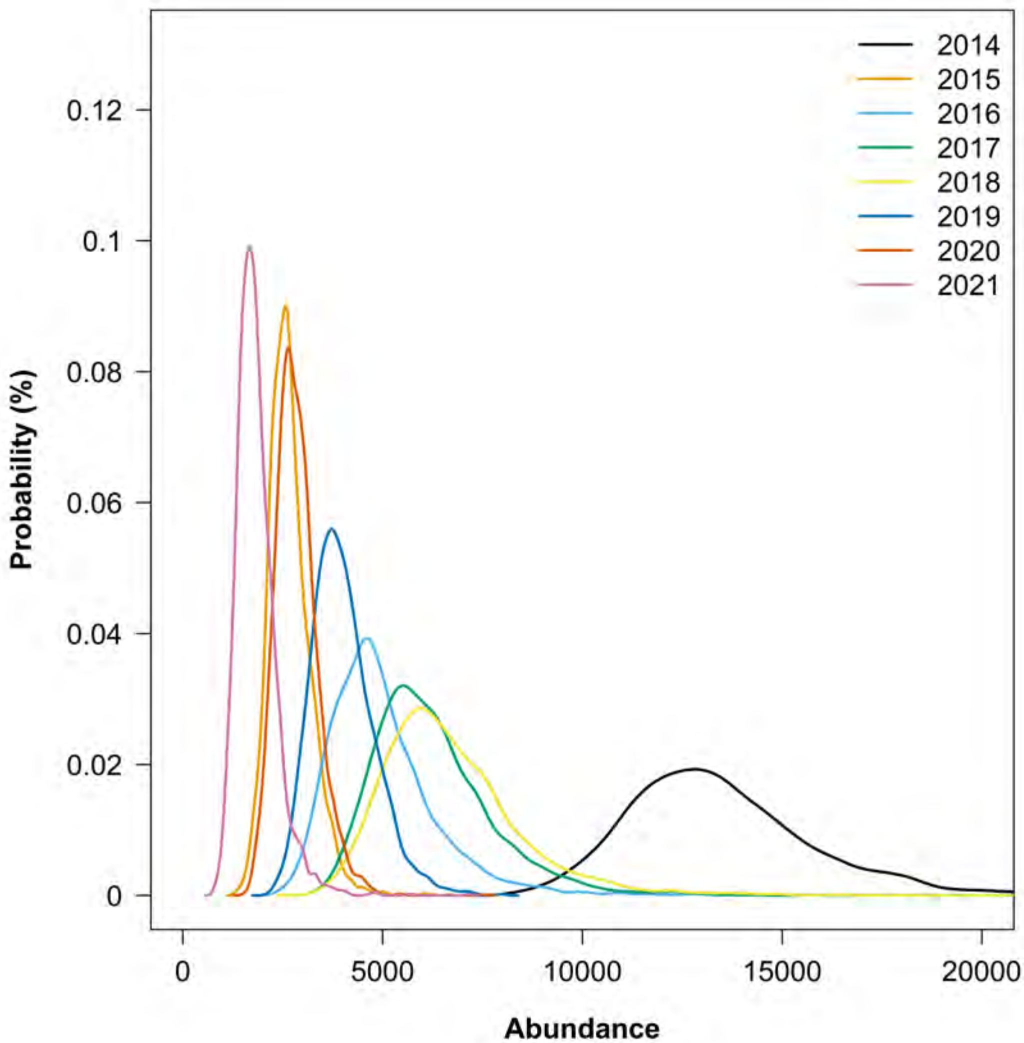


Figure 3.25 Posterior probability distributions for total river-wide abundance of age-0 Rainbow Trout in Seton River from 2014 to 2021.

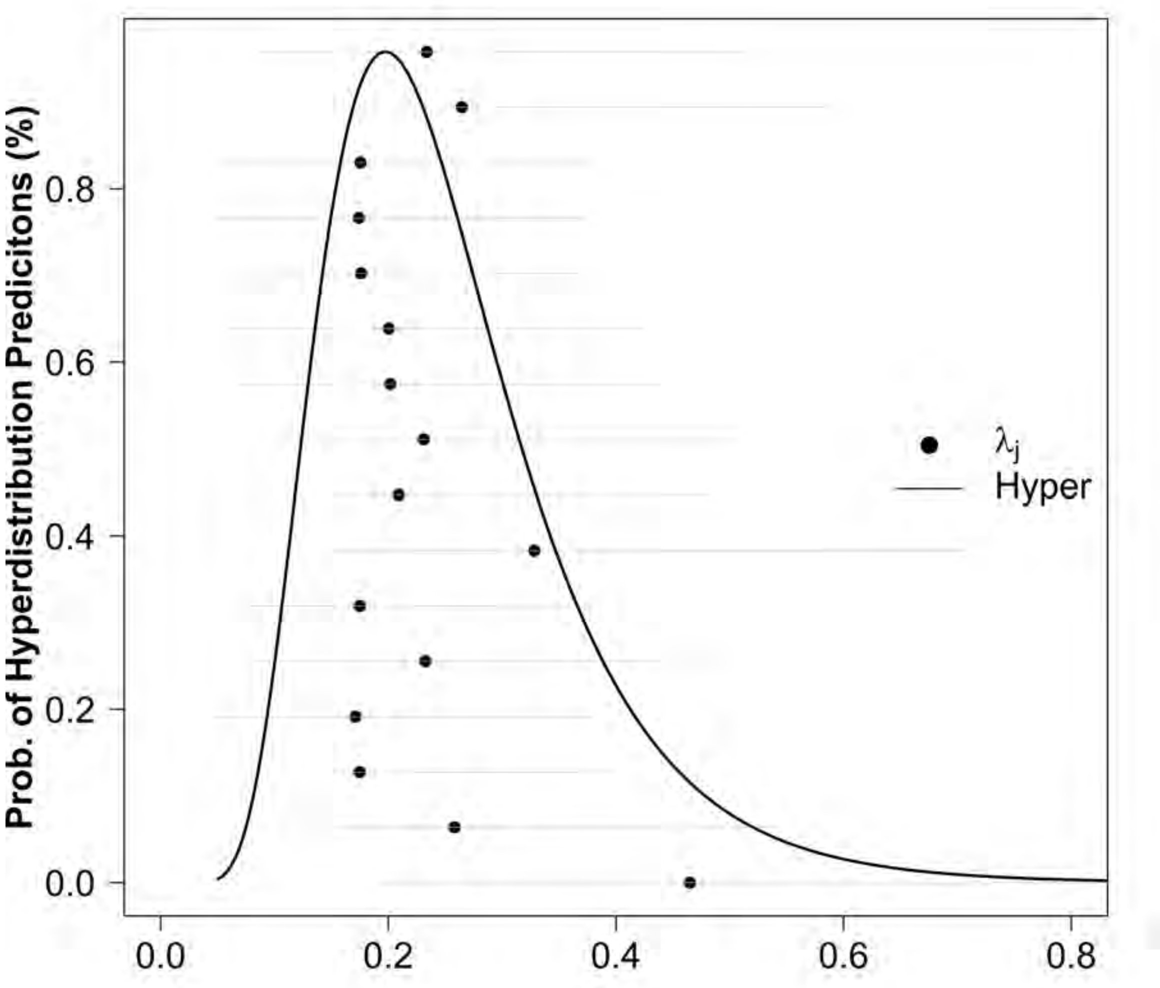


Figure 3.26 Estimates of fish density (fish/m) for age-0 Rainbow Trout in the Seton River in 2021. Filled points are the mean with 95% credible interval of individual index sites and the black line is the hyperdistribution based on the means of the hyperparameters estimated during the hierarchical Bayesian 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.

3.3 Adult Anadromous Fish

3.3.1 Distribution and Abundance

Redds were identified in the upper and lower spawning channels in October and November following the Pink, Sockeye and Coho Salmon migrations. No redds were identified in the mainstem of the Seton River though flow conditions always make visualizing redds in the mainstem more difficult. Seton River discharge was held at 30 m^3/s through August and was dropped to 12 m^3/s on September 16, 2021. Both Pink and Sockeye Salmon were present in Seton River during the ramp down but no stranded redds were observed.

Chinook Salmon, Coho Salmon, and Pink Salmon

Since the study began, observations of adult Chinook Salmon have generally been rare and sporadic (Figure 3.27). The exception was in 2019; after a slide on the Fraser River upstream of the confluence with the Bridge River (i.e., the Big Bar slide) impeded upstream migration, observations of adult Chinook Salmon increased considerably (n = 66) with counts peaking in early September. In 2021, observations of Chinook Salmon decreased (n = 4) and were similar to 2020 and the study years prior to the slide. All observed Chinook Salmon were found in the LSC.

Counts of adult Coho Salmon in 2021 (n = 61) were consistent with numbers observed in 2018 (n 65). As with Chinook Salmon, observations of Coho Salmon were greatest in 2019 (n = 235) coinciding with the Big Bar slide. In study years prior to the slide, counts of adult Coho Salmon ranged from 4 (2014) to 108 (2016). Adult Coho Salmon were primarily observed in the spawning channels. Of those observed within the mainstem Seton River, the greatest number occurred in Reach 1 (Figure 3.27). Peak counts consistently occurred in November (Figure 3.27).

Pink Salmon were observed in the study area on dominant run years, which fall on odd calendar years (i.e., 2015, 2017, 2019 and 2021; Figure 3.28). In 2021, a total of 3,228 Pink Salmon were observed. The largest return was in 2015 (n = 9,016). Peak counts were consistently in mid to late September with most adults observed in the spawning channels and Reach 1 of the Seton River (Figure 3.28).

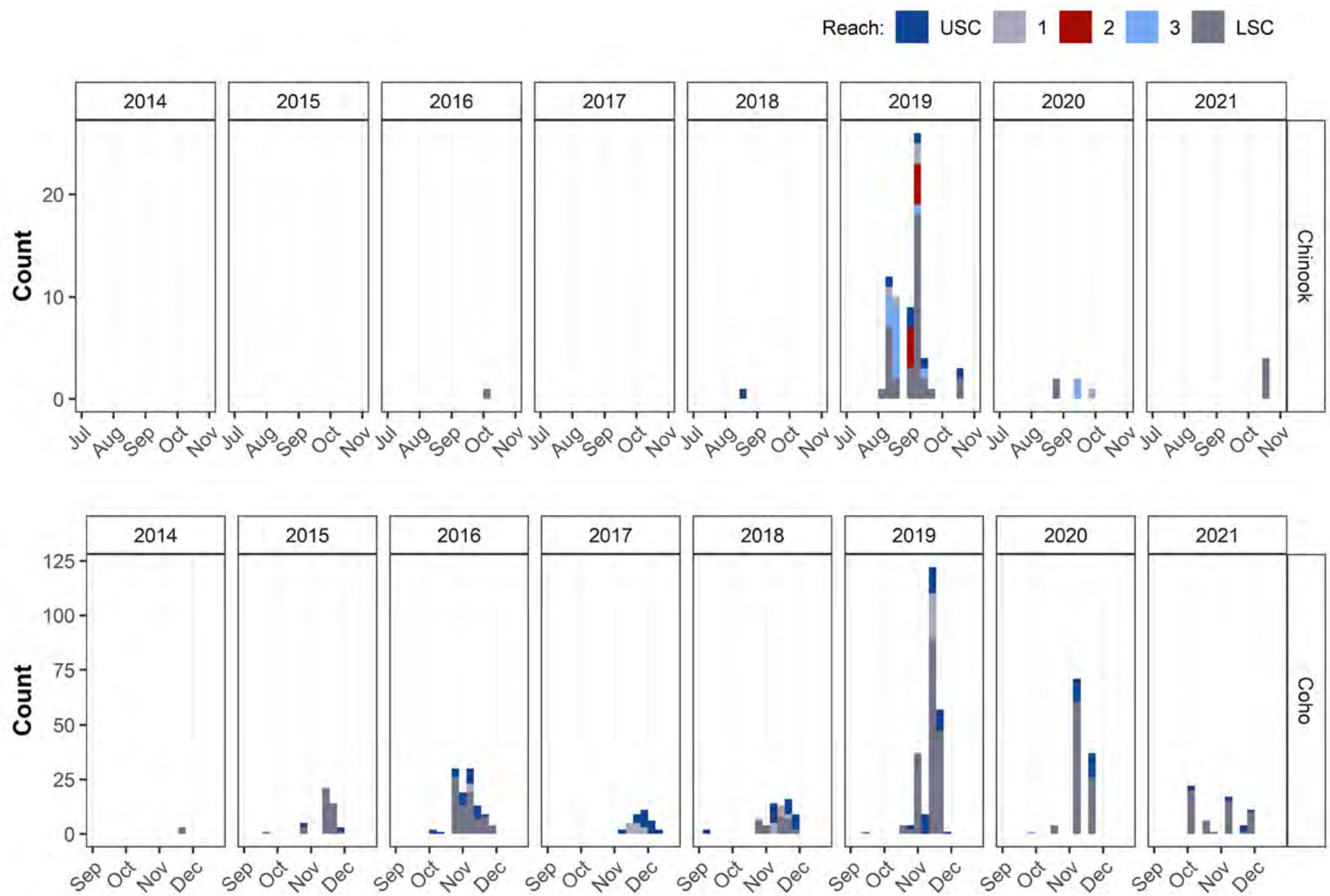


Figure 3.27 Counts of adult Chinook Salmon and adult Coho Salmon observed within each reach of Seton River, as well as in the upper (USC) and lower (LSC) spawning channels during weekly visual surveys from 2014 to 2021. Note differences in x- and y-axes.

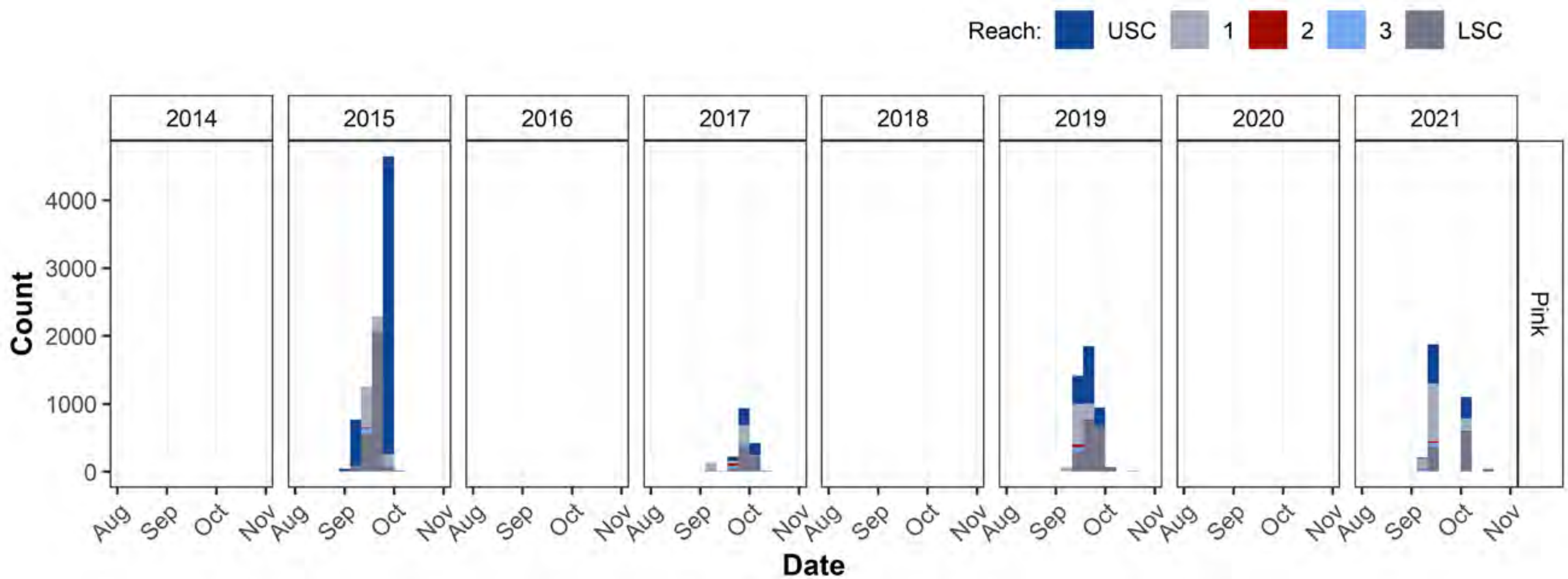


Figure 3.28 Counts of adult Pink Salmon observed within each reach of Seton River, as well as in the upper (USC) and lower (LSC) spawning channels during weekly visual surveys from 2014 to 2021.

Steelhead Trout

Visual counts of adult steelhead have had limited success due to high flows and turbidity during the migration and spawning period. In 2021, surveys were conducted from April 1-19 as discharge from Seton Dam was within WUP targets. Following April 19, increased flows inhibited visibility so surveys were ceased. Twenty-three (23) Steelhead were tagged under BRGMON3 in 2021 but none of them were detected on the Seton PIT arrays. Over the study period there have been six detections at the LSC (5 as unassigned direction and 1 as in) from three different fish (n = 2 in 2015; n = 1 in 2019). Steelhead have not been recorded on the USC PIT array during the duration of the study. While no Steelhead were detected on the Seton Fishway PIT antenna, one radio-tagged fish (code 202) was detected in Seton Lake during mobile tracking under BRGMON-3 (Figure 3.29). An additional fish (code 192) was observed at the LSC Outflow fixed receiver and during mobile tracking at the same site for 2 months following tagging.

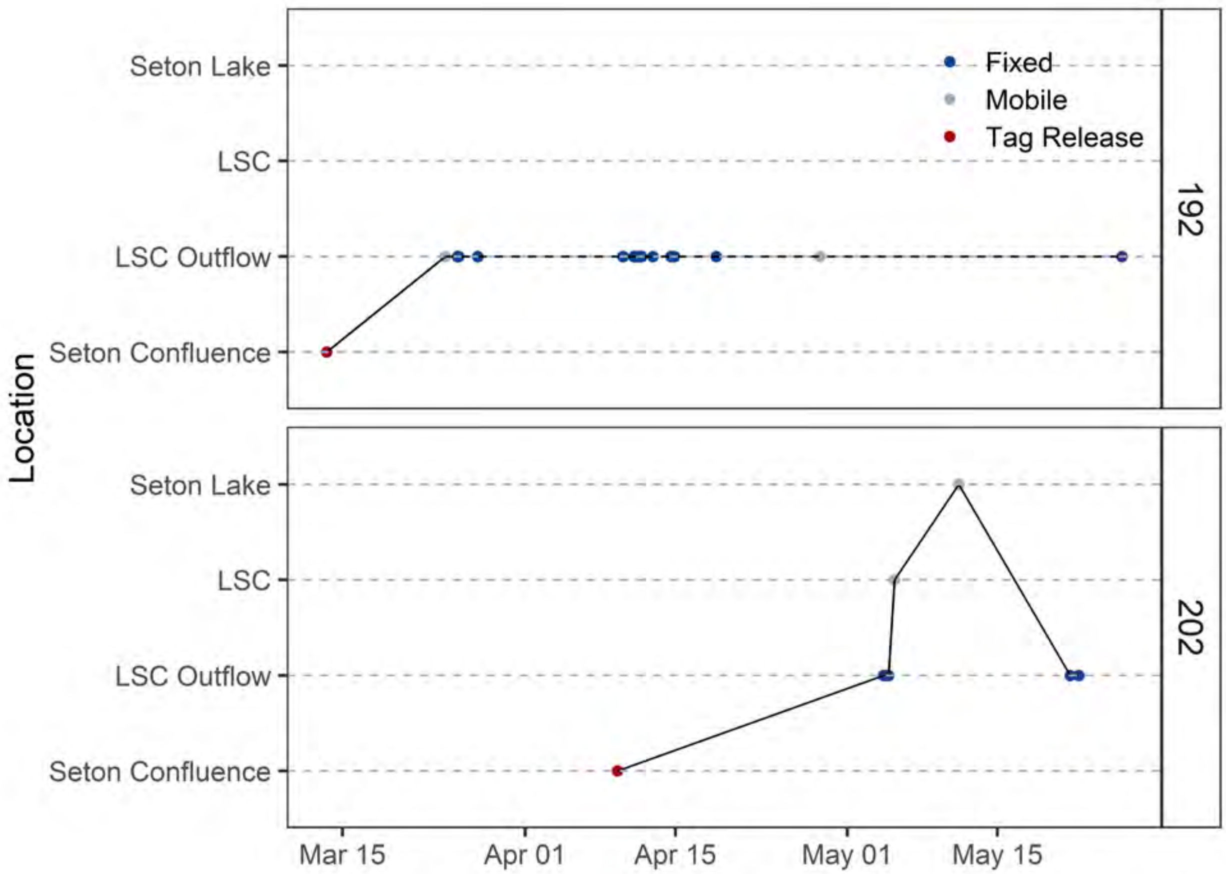


Figure 3.29 Detection histories of radio-tagged steelhead within the Seton River watershed and surrounding area during the 2021 migration and spawning period. The red point represents the release location during which fish were tracked with a mobile receiver. The remainder of detections, represented by the blue and grey points, are from fixed receiver stations and mobile tracking respectively. Only fish with >1 detection above the Seton River confluence are shown.

Seton Dam Resistivity Counter

In 2019 and 2020, the counter under-estimated the number of steelhead migrating upstream through the Seton Dam. However, in 2021 the counter overestimated the number of steelhead. Accuracies in 2021 were high (Table 3.7). No Steelhead were observed moving on channel 1 on either the lower or the upper counter. After correcting for accuracy, 30 steelhead were estimated to have migrated through the Seton Dam between April 15 and June 15, 2021 (counter n = 18), which was higher than the two previous years (Figure 3.30). The migration timing of steelhead in 2021 occurred from mid-April through mid-June. The delay in timing may have been the result of the annual counter cleaning being delayed until April 13, 2021, whereas in previous years, migrating steelhead have been observed using the counter as early as April 1. The onset of steelhead migration in 2021 also corresponded with an increase in discharge in Seton Dam. On April 18, discharge from Seton Dam was increased to 117 m^3/s where it remained until May 29 when it was reduced to 92 m^3/s. These discharges were higher than the maximum discharge during 2019 and 2020 over the same period (79 and 48 m^3/s, respectively). Additionally, the maximum daily abundance of steelhead was eight in 2021, which was similar to observations in 2019 (7). In 2020, only 2 fish were observed migrating through the Seton Dam in a single day (Figure 3.30). Overall, in years with higher discharges, steelhead migration through the Seton Dam has occurred in larger groups, and in years with lower discharges, migration occurred gradually with one or two fish passing through the dam per day.

Table 3.7 Number of classified events (n) and accuracy of each resistivity counter channel (tube) for detecting adult steelhead that are migrating upstream of the Seton Dam.

Year	Counter	Lower				Upper			
	Channel	1	2	3	4	1	2	3	4
2019	Accuracy	100%	75%	100%	18%	0%	100%	100%	NA
	n	3	3	4	3	1	4	2	0
2020	Accuracy	100%	100%	NA	NA	100%	75%	100%	NA
	n	6	3	0	0	1	4	1	0
2021	Accuracy	NA	100%	100%	33%	NA	100%	100%	100%
	n	0	2	3	6	0	3	3	1

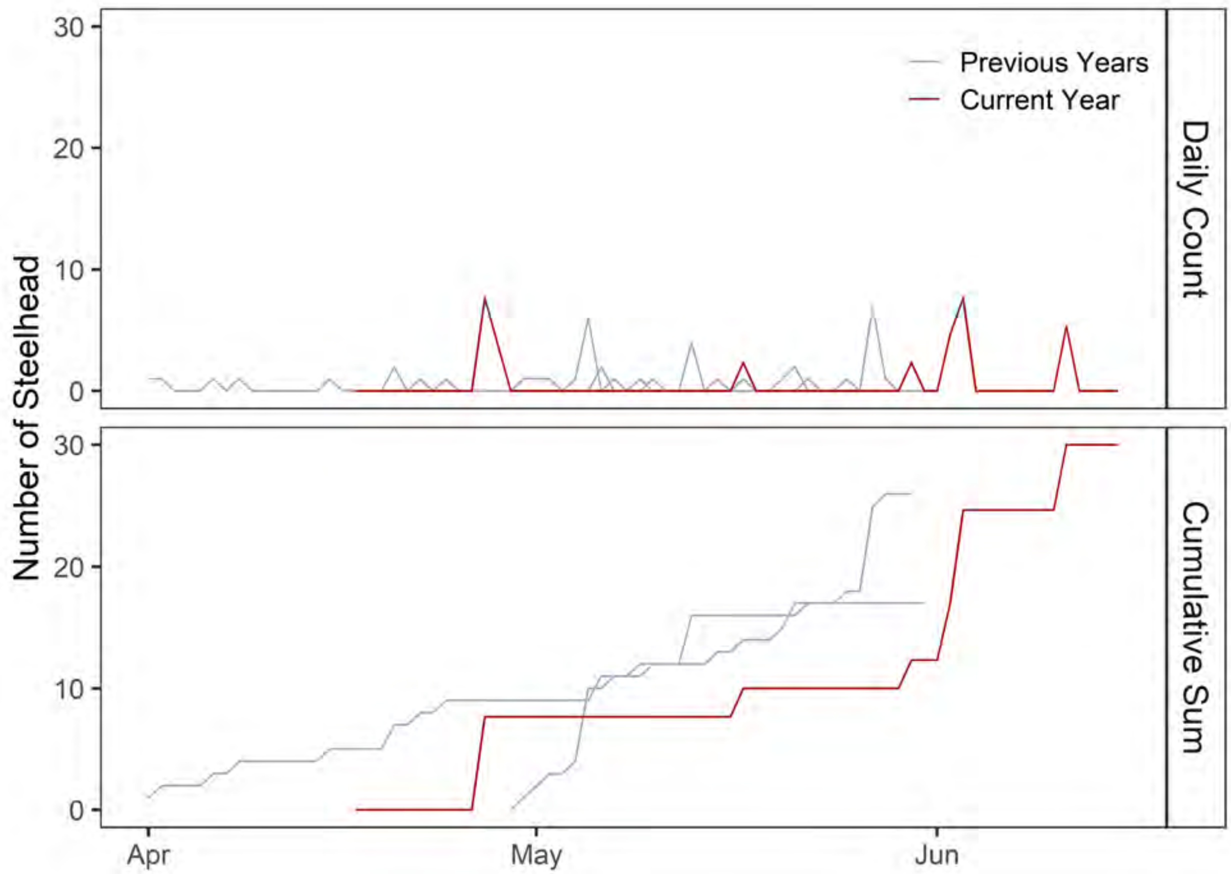


Figure 3.30 Daily and cumulative abundance estimates for adult steelhead that migrated upstream of the Seton Dam from April 1 to May 31. Previous years represents 2019 and 2020. 2019 was the first year the resistivity counter was operated during the steelhead migration period.

Gates Creek Sockeye Salmon

The counter algorithm was validated for two periods: before and after September 1. This represents the date that the number of fish passing Seton Dam exceeded 2,500. Because the counter is known to miscount when fish are crossing simultaneously, it was important to separate the two periods. From August 1 through to September 1, 22,848 records were validated and the algorithm accuracy for the upper and lower counters were 83% and 88%, respectively. From September 1 to 30, 64,055 records were validated and algorithm accuracy for the upper and lower counters were 77% and 72%, respectively (Figure 3.31).

Raw counts indicated that from August 1 to September 30, a total of 213,009 fish passed through the resistivity counter in Seton Dam (Figure 3.32). As Pink Salmon were also present in Seton

River in 2021, this number represents the combined total of Pink and Sockeye Salmon. The bimodal distribution of fish over this period is typical for Seton River, which supports two Sockeye Salmon runs: Gates Creek and Portage Creek. Typically, Gates Creek Sockeye Salmon are observed during the first peak in early September and Portage Creek Sockeye Salmon during the second peak in later September. Using September 15, 2021 as a cutoff point between the two runs, 131,986 and 81,023 fish are estimated to have passed during the Gates Creek and Portage Creek migrations, respectively. Peak migration occurred on September 11, 2021 with 13,913 fish moving past Seton Dam.

Video validation has not yet been completed for the Gates Creek Sockeye Salmon. Due to a file storage error, data had to be sent to a specialist for recovery. It has since been recovered and video validation was ongoing at the time of writing. Validation will allow for daily species ratios to be determined and therefore an accurate estimate of Sockeye Salmon to be developed. An addendum will be provided to this report with updated Gates Creek Sockeye Salmon counts when it is completed.

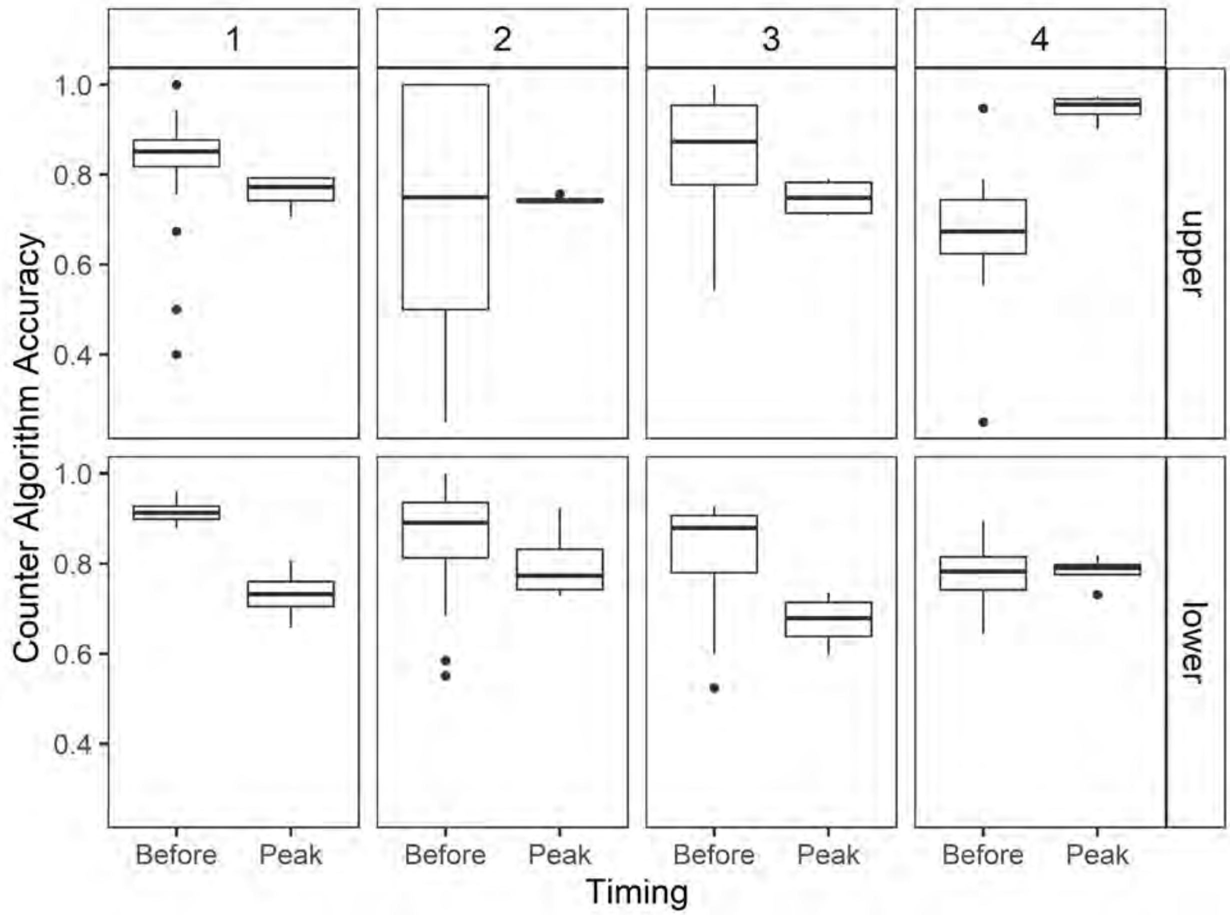


Figure 3.31. Resistivity counter algorithm accuracy before and during peak Sockeye Salmon migration for the upper and lower counters at the Seton Dam fishway. The two periods were separated, as the counter is susceptible to miscounting when many fish are crossing simultaneously. September 1, 2021, represented the threshold date when the number of fish passing Seton Dam exceeded 2,500 in a single day.

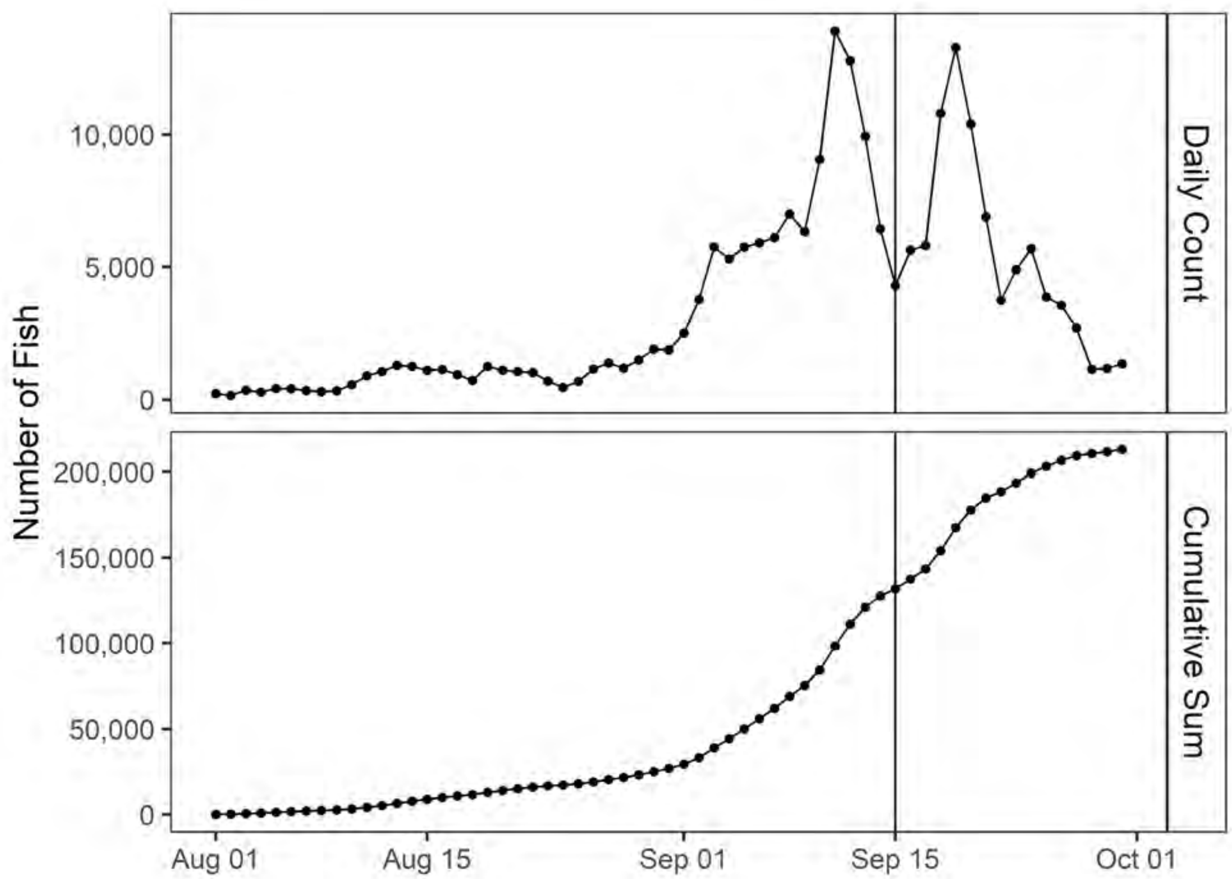


Figure 3.32. Daily and cumulative abundance estimates for fish (both Pink and Sockeye Salmon) passing the Seton Dam from August 1 to September 30, 2021. The vertical line represents the estimated cutoff timing between the Gates Creek and Portage Creek Sockeye Salmon migrations.

3.4 Winter Modified Operations

3.4.1 Water Temperature

Water temperature analyses focused on the period of December 1 to March 31. Data from the winters of 2015-16 and 2016-17 were removed due to incomplete datasets. Water temperatures in the upper Seton River (Reach 1) were generally warmer than the average across previous year's, except for a period from mid-January to early February when water temperatures dropped. In the lower Seton River (Reach 2 and 3), water temperature followed a similar pattern though the magnitude of the temperature difference was greater and the drop in temperatures was observed in early to mid-February. While only one previous year of data is available for Cayoosh Creek, water temperatures were similar ($F_{1,\,240}$ = 1.22, P = 0.27; Figure 3.33).

Water temperature in the upper Seton River (Reach 1) from December 1, 2020 to March 31, 2021 was statistically warmer from temperatures during the same period of 2013/2014 and 2017/2018 but was similar to all other previous winters ($F_{5,\ 720}$ = 3.7, P = 0.002; Figure 3.34). Temperatures in the lower Seton River (Reach 2) were significantly warmer than all other years on record ($F_{5,710}$ = 19.4, P < 0.001; Figure 3.34). However, the temperature from within Seton Fishway was also significantly warmer than all other years on record ($F_{5,\ 720}$ = 8.29, P = < 0.001;

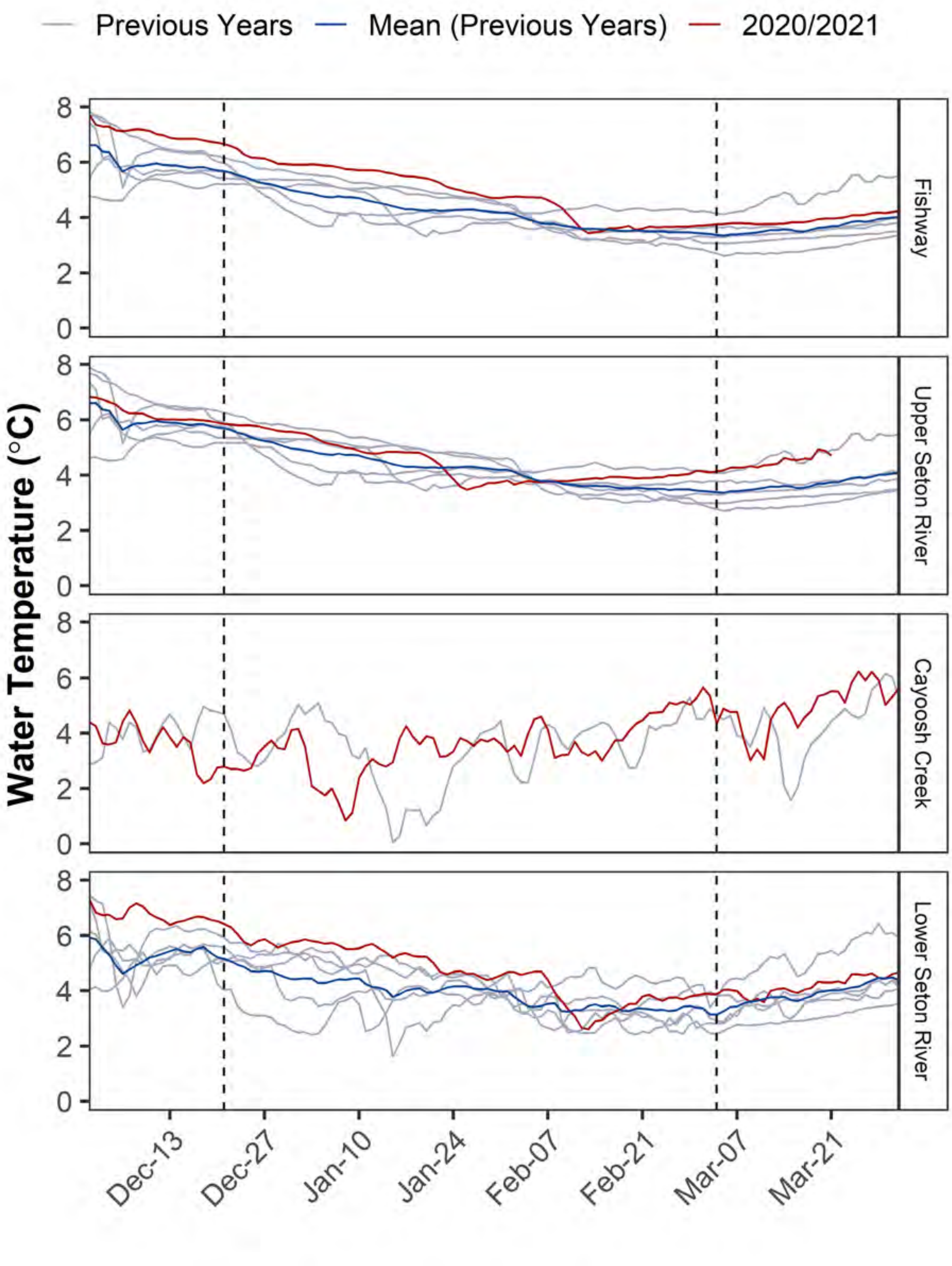


Figure 3.33 Mean daily water temperature for the upper Seton River (Reach 1), Cayoosh Creek, and lower Seton River (Reach 2) from December 1, 2020 to March 31. The dashed lines indicate the start and end of winter modified operations of the Seton Dam.

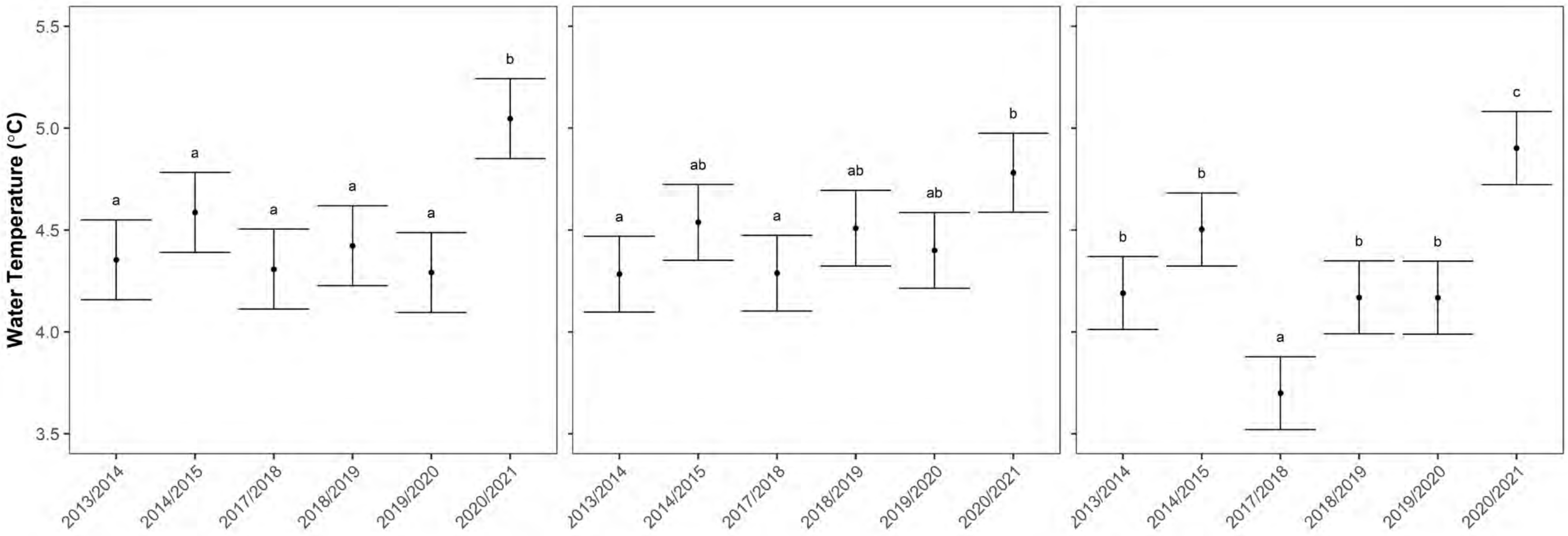


Figure 3.34 Mean water temperature with 95% confidence intervals from December 1 to March 31 for the Seton Dam Fishway (left panel), upper Seton River (Reach 1, centre panel) and the lower Seton River (Reach 2, right panel) from 2013 to 2021. The winters of 2015/2016 and 2016/2017 were removed due to incomplete datasets. Points within the same panel that do not share the same letter are statistically different from one another.

3.4.2 Juvenile Fish

Distribution

Fish detections for the period of December 1 to March 31 of each year were examined. Since 2014, the LSC and USC PIT arrays respectively have detected an average of 2 and 4 unique fish per year (SD = 1 and 5, respectively). Over the winter of 2020-21, no fish were detected on the LSC and eight (8) unique fish, all Rainbow Trout, were detected on the USC (Table 3.8). While fish were detected moving both in and out of the USC, 4 fish were detected but could not be assigned a direction, 3 fish moved out of the spawning channel, and the other fish moved repeatedly back and forth across the PIT array causing both in and out detections (Figure 3.35,). The Fishway PIT array has a low detection efficiency for 12 mm PIT tags and has only detected one fish in the given period in the winter of 2017-18. While it appears the change in flow regime over the winter may have increased activity within the USC, further years of monitoring at increased flows would be required to determine if 2020-21 falls within the normal range of variability.

Table 3.8 Mean ± standard deviation of the number of unique tagged fish detected on each PIT array during the period of December 1 to March 31 of each study year.

	Fishway	LSC	USC
Previous Years	0 ± 0*	2 ± 1	4 ± 5
2020/2021	0	0	8

* only one fish has been detected during the winter period on the Seton Dam Fishway array since 2014.

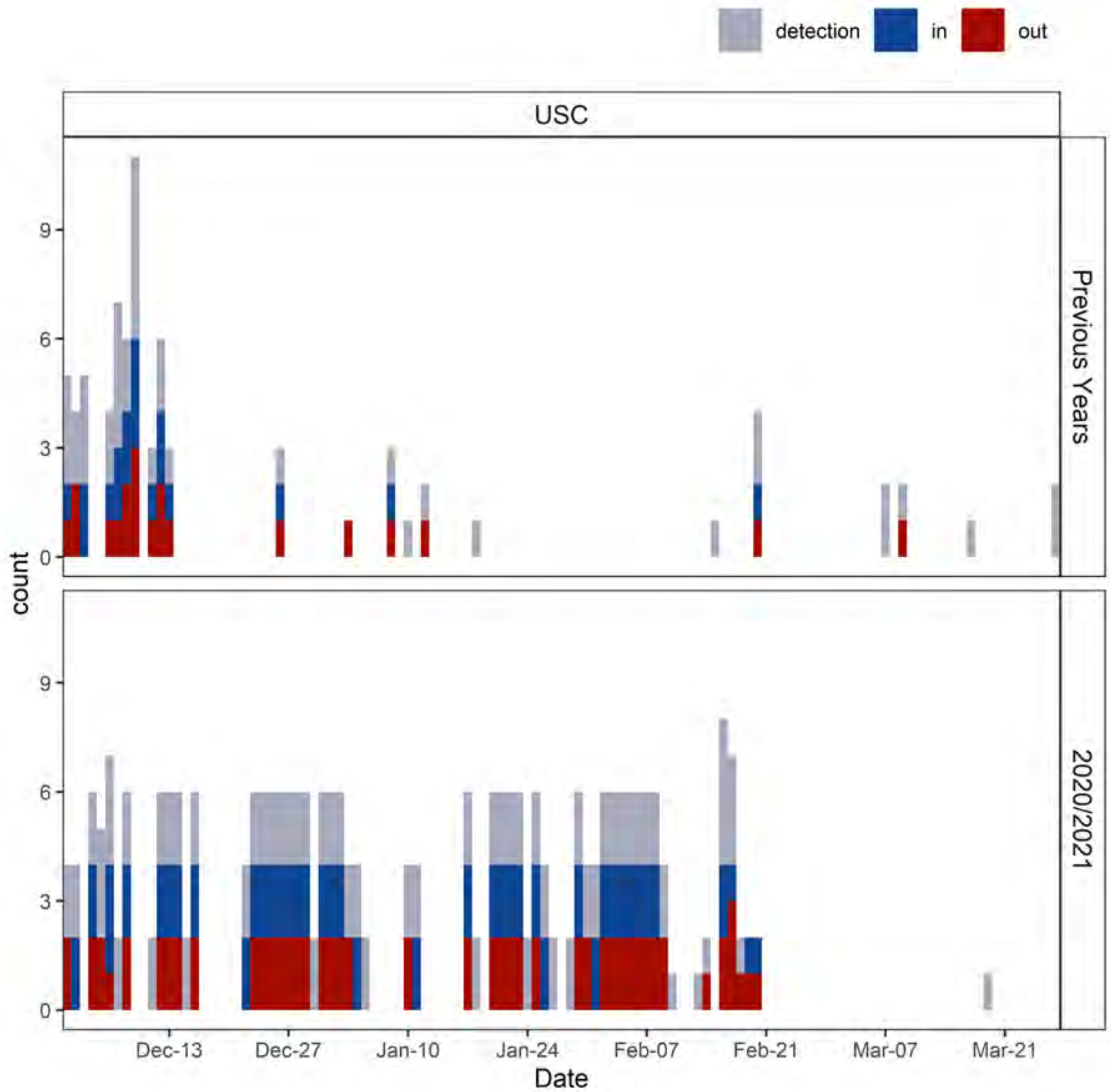


Figure 3.35 Counts of daily detections by direction of movement for passive integrated transponder (PIT) tagged juvenile and resident fish at the PIT antenna arrays in the upper spawning channel (USC) over the winter period of December 1 to March 31. All previous years(2015-2020) were combined to show trends in movement for the same period.

Growth and Body Condition

A winter electrofishing survey was conducted in early April 2021 and 55 scale samples were collected. Samples were aged and the analysis was included in the regular Size-at-Age and Body Condition models (see Section 3.2.1).

4. Discussion

Overall, the study objectives were to monitor responses of fish and fish habitat to Seton Dam operations, which could then be used to identify and refine potential performance measures. While the study was originally designed to monitor fish and fish habitat in the Seton River under WUP target flows, Modified Operations have resulted in intermittent flow releases beyond WUP targets since 2016 and as such, monitoring activities have been impacted and necessarily adjusted. Data collected in 2021 (Implementation Year 9 of 10) continued to build upon knowledge gained in previous years. Preliminary synthesis analyses have identified some trends but given natural and operational variability combined with method adjustments, additional monitoring is required to sufficiently address the management questions (MQs). Herein, findings to date are discussed in the context of each MQ within the scope of this document (i.e., MQ1–MQ4).

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?

MQ1 is of a descriptive nature with no hypothesis testing. The data collected under BRGMON-9 have improved our understanding of fish populations in the Seton River and will continue to do so each year.

A total of 16 fish species have been identified within the Seton River and spawning channels, including Coho Salmon, Chinook Salmon, Pink Salmon, Sockeye Salmon, Rainbow Trout and steelhead, Bull Trout, Mountain Whitefish, Longnose Dace (also dace sp.), Bridgelip Sucker (also sucker sp.), Coastrange Sculpin (formerly Aleutian Sculpin), Prickly Sculpin, Slimy Sculpin (also sculpin sp.), Peamouth Chub, Northern Pikeminnow, Redside Shiner, and lamprey sp.

Biological Characteristics of Rearing Populations in Seton River

Of the salmonids, juvenile Rainbow Trout, Coho Salmon and Chinook Salmon (in order of prevalence) were the most frequently captured, and thus these species were the focus of monitoring. Remaining salmonid species either do not typically use riverine habitats for rearing (e.g., Sockeye Salmon) or were found in low densities within the study area (e.g., Bull Trout, Mountain Whitefish).

Rainbow Trout

As juveniles, resident Rainbow Trout cannot be distinguished from steelhead (the anadromous life history form of the species). Movement data suggest that Rainbow Trout within the study area are a single population; fry primarily inhabit the mainstem Seton River and move between the mainstem and spawning channels as parr and adults (regardless of capture location). Movement activity occurred primarily during the spawning and growing season (spring through fall) with a reduction of activity evident in December and very limited movement during the winter. The timing of movements between the spawning channels and mainstem did not appear to be directly related to flow events, but instead were seasonal (e.g., overwintering, spawning).

Juvenile abundance surveys (i.e., standing crop) have been conducted annually since 2014 at baseflow conditions, making it the most consistent dataset collected under BRGMON-9. Although too few fish were observed during snorkel surveys to inform abundance estimates, results generally showed that large age-1 and age-2 Rainbow Trout occurred at lower densities. Abundance estimates of age-0 Rainbow Trout ranged from 1,839 (2021) to 12,269 (2014) individuals. All years from 2015 to 2021 have experienced high flow conditions and in these years fry densities and abundance have been substantially lower compared to 2014 – the only year of WUP flow conditions. However, comparative analyses between abundance and flow conditions were limited and there is a high degree of uncertainty in the 2014 estimate given high variability in observed densities during electrofishing. Statistical modelling to discern a potential relationship between abundance and flow conditions within the Seton River has not been performed; the current dataset was limited by a lack of data at WUP flow conditions. The lower Rainbow Trout abundances observed in high flow years could be related to the timing, magnitude, and or duration of high flows. Monthly biological sampling data suggest that Rainbow Trout fry emerge from their redds in June through July (and August), as is common with this species (Scott and Crossman 1973). If high flows occur during the emergence period, fry may be flushed out of the mainstem into the Fraser River or displaced from suitable habitat at a critical period. Habitat suitability analyses indicated that suitability in the mainstem for Rainbow Trout fry decreases considerably above 60 m^3/s and so under high flow conditions, fry may be rearing in less suitable habitat, potentially impacting growth and survival. In terms of duration, it would be expected that the potential effects of high flows would be exacerbated with increasing duration. Moreover, availability of suitable habitat for Rainbow Trout fry may decrease over time with consecutive years of high flows. Results from habitat suitability surveys at base flow conditions indicated this may be the case. The availability of suitable rearing habitat was lower in 2019, 2020, and 2021 compared to 2014 with the largest decreases at sites in Reach 2 (generally > 50%).

Positive monthly growth was evident in Rainbow Trout from age 0 to age 2, after which growth stabilized. Modeling suggested body condition in age-0 Rainbow Trout was affected by reach of capture and year. Results among mainstem reaches and spawning channels were variable, but there was indication that fry in Reach 1 may be in better condition overall. The effect of year was less pronounced; Rainbow Trout fry in 2019 were in significantly better condition than those in 2016, 2018, and 2020, but all other years were statistically similar.

Overall, while year and location may explain some of the variation in fry condition, the results were mixed and suggest other factors may be driving fry growth and body condition. Flows, water temperatures, habitat suitability (for a given flow), and fry densities also varied among reaches and years – all which could be expected to influence fish growth (Wetzel 2001). Rather than using year as a proxy variable in analyses, using variables that explicitly test the desired relationship(s) may be more meaningful. Alternate modelling approaches, not only for Rainbow Trout but also for Coho and Chinook Salmon, will be explored leading up to the Year 10 report. It should be noted, however, that another approach may still produce inconclusive results, as the potential for significant time lags between changes in river flow conditions and the response in fish populations should not be underestimated. Unlike for the Lower Bridge River through BRGMON-1 (Sneep et al. 2019), a comprehensive dataset of chemical and lower trophic level indicators (that are more directly linked with physical habitat variables, such as flow and water temperature, while also affecting fish growth) does not exist for the Seton River.

Coho Salmon

Juvenile Coho Salmon were captured up to age 2 within the study area and were distributed throughout the spawning channels and mainstem, regardless of age class. Coho Salmon spend 1 or 2 years in the Seton River study area prior to their seaward migration. Data suggest that most juveniles leave at age 1 with migration occurring in the spring to late June. Juveniles that remained in the system in July continued to rear for an additional year before beginning their seaward migration at age 2. No age-2 Coho Salmon were found after the spring out-migration period (i.e., June and later). Movement between the spawning channels and the mainstem may be related to high flow conditions and or seasonal behaviour with a clear period of increased activity in the spring, which is the out-migration period but also overlaps with freshet and typically increased flow releases from Seton Dam. Directed movement out of the spawning channels was more common and suggestive of migration activity. The second period of increased activity was in August (after the out-migration period) in the fall prior to overwintering. Very little movement was detected during the winter.

Positive monthly growth was evident in juvenile Coho Salmon to age 2 and modeling indicated that body condition of fry was influenced by reach of capture and year, as well as an interaction term. Interestingly, fry in the lower spawning channel were in significantly better condition than in Reach 1, but all other areas were statistically similar. In terms of year, the condition of Coho Salmon fry was the greatest in 2017, condition was similar among 2014 and 2018–2021, and was the lowest in 2015 – all of which were statistically significant differences. As discussed for Rainbow Trout, several variables (i.e., flow conditions, water temperatures, habitat suitability) varied among reaches and years. But unlike Rainbow Trout fry which primarily inhabit the mainstem, differences in Coho Salmon habitat preferences may make some fish – those that inhabit the spawning channels – less susceptible to variability in flow given that conditions in the spawning channels are maintained at consistent levels. Because juvenile Coho Salmon were not found in great enough densities for a mark-recapture program, observed densities and abundance were unknown throughout the study period and therefore only habitat variables can be used to inform growth and condition results. Exploration of alternate analyses, with the same caveats articulated above, is also recommended for age-0 Coho Salmon leading up to the Year 10 report.

Chinook Salmon

As juveniles, Chinook Salmon exhibit a variety of life history strategies that differ in residence times within freshwater and estuary habitats (and therefore timing of migrations), as well as age at seaward migration. While juvenile Chinook Salmon were found up to age 2, age-0 fish were the most common. Chinook Salmon fry were found primarily in mainstem habitats, especially in Reach 3. As with other juvenile salmonids, data collected during biological surveys indicated positive monthly growth, particularly between age 0 and age 1. Neither reach nor year were significant in analyses of body condition. DNA results from 2016–2018 revealed that juvenile Chinook Salmon in the study area were not only from Seton River/Portage Creek, but also from several upstream tributaries of the Fraser River (e.g., Stuart, Quesnel, Chilko; but not Bridge River). Results also demonstrated that the proportion of juveniles from other populations using the Seton River increased throughout the growing season. Fish originating from other rivers can explain why Seton River metrics (i.e., flow and reach) were not significant factors in body condition. Given that these fish were exposed to various rearing conditions prior to entering Seton River, the use of juvenile Chinook Salmon metrics as indicators may not be advisable. Still, results demonstrated that the Seton River provides rearing habitat for this species and local conditions would influence not only the local population but also those from throughout the Fraser River. Given the imperiled status of many Fraser River Chinook Salmon populations (Grant et al. 2019, DFO 2016), it is important to consider the potential effects of operations on this species. The increased numbers of Chinook Salmon juveniles observed in Seton River in 2021 may be the result of a hatchery release of Chinook in Portage Creek.

Biological Characteristics of Spawning Populations in River

Currently, information regarding adult salmon abundances is limited to relatively inconsistent count data, precluding robust analyses. Estimating spawner abundance in the Seton River has been difficult because few adults, apart from Pink Salmon, have been observed until 2019 following the Big Bar slide. Visual tagging of Pink Salmon was attempted in 2015 and 2017 to assess observer efficiency and create AUC estimates. However, insufficient numbers were captured to warrant releasing tags, and thus all count data should be considered an index of relative abundance. Pink Salmon were the most abundant (observed counts ≈ 1,800 to 9,000 adults) followed by Coho Salmon (observed count ≈ 30 to 200 adults). Chinook Salmon were rarely observed, except in 2019 (observed count = 66 adults) when adults were found throughout the mainstem and channels. Pink Salmon and Coho Salmon spawned mainly in the spawning channels followed by Reach 1 (largely in the area just downstream of the Seton Dam). But given differences in viewing conditions, counts were likely biased as it was easier to observe fish in the spawning channels compared to the mainstem. High flow conditions in the fall of 2020 further limited observability, and so numbers from this year should be interpreted with caution. Pink Salmon have been observed in the study area from late August to early October, while Coho Salmon were observed from October to early December. If observed, adult Chinook Salmon were present from August to October. Chinook Salmon may be using habitats downstream of Seton Dam for spawning or may be migrating through the Seton River to upstream spawning grounds such as Portage Creek; current stock classification does not distinguish between the two different spawning areas (John Candy, DFO, pers. comm.).

Increased adult counts for all salmon species in 2019 were likely the result of strays due to the Big Bar slide, which impeded upstream passage of fish in the Fraser River. Straying was supported by DNA analyses of adult Chinook Salmon in the Bridge River taken during brood stock collection; all sampled adults in 2019 were from populations upstream of the Big Bar slide (Evans and McHugh 2021). Though most adult salmon will home rather than stray, mature fish with advanced senescence may select the nearest available spawning location instead of migrating to their natal site (Keefer and Caudil 2014). If spawning was successful and the offspring survive, straying as a result of the Big Bar slide may result in increased numbers of salmon in the Seton River in future years when these cohorts would be expected to return to spawn.

To augment the population of Chinook Salmon in Portage Creek, a hatchery program released 10,000 juveniles into the river in 2021. These fish are expected to return in 2023 (Age 3) and 2023 (Age 4) and will move through Seton River and the Seton Dam fishway before ultimately reaching their destination

Enumeration of adult steelhead in the Seton River has been challenging. Visual surveys have been attempted annually since 2014, but poor viewing conditions from high flows and turbidity made it difficult to observe fish. Beginning in 2019, the Seton Dam Counter was operated during the steelhead migration and spawning period and proved an effective way to enumerate adults passing Seton Dam in the spring. When cross-validated with PIT and radio telemetry data collected under BRGMON-3, these data also support our improved understanding of adult steelhead distribution and movement behaviour. In all three years, steelhead abundance was low with up to 30 adults passing the Seton Dam in each year. Even so, stark differences in passage timing and flow releases from the Seton Dam in the last three years suggest operations may affect steelhead migrations. When flows were kept at or below ~40 m^3/s (2020), steelhead migration occurred over many weeks with small numbers of fish (1 or 2) moving through the fishway each day. On the other hand, when flows were 55 m^3/s or greater (2019 and 2021), adults may have delayed their migration, passing in larger groups (6 to 8 fish) and particularly after there were reductions in flows. In 2021, discharge from Seton Dam was held at 120 m^3/s and very few fish were observed passing the fishway until flows were reduced to 90 m^3/s further lending support that Modified Operations during the spring may delay Steelhead migration. Previous work at the Seton Dam through BRGMON-14 found that dam operations affected the ability of fish to find and ascend the fishway (Harrower et al. 2020). This could also be the case for steelhead. With only three years of data, caution must be taken before making any generalizations; additional years of monitoring are required to discern if such differences may be the result of natural variability in migration pattern or potential responses to operations.

Telemetry data from BRGMON-3 have consistently shown that steelhead move into the Seton River (White et al. 2021). Continued use of the counter on the Seton Dam (regardless of whether the monitoring year is considered WUP or MOD) in conjunction with the BRGMON-3 tagging program will increase understanding of Seton River steelhead, and potentially the effects of Seton Dam operations on this species. Moreover, increasing the telemetry infrastructure in the study area would leverage shared information between the two monitoring programs and more effectively address uncertainties regarding the basic biological characteristics of steelhead, a focus of both programs. To date, visual counts have been a poor method of enumeration and could be abandoned on the mainstem (i.e., only conducted on the spawning channels, which is completed by Splitrock under a different program) in favor of an increased focus on radio telemetry. Fixed radio telemetry stations have allowed us to confirm the presence of tagged fish and residence time. While the intent of mobile tracking is to confirm the presence of tagged fish in areas not covered by fixed radio stations or PIT antennas, steelhead detections in the Seton River during these surveys have been limited. The additional radio receivers deployed under BRGMON-13 in 2020 demonstrated the value of having additional fixed stations in determining the migration behaviour of steelhead. Deployment of one or two additional fixed radio stations focused on key areas of interest, such as the Seton Dam near the common spawning area, is recommended.

Though current data have provided insight on steelhead populations within the Seton River watershed, several uncertainties remain:

1. *Impacts of the Big Bar slide*

   Because there are no counter data for steelhead prior to 2019, it is unclear whether the abundance estimates collected since 2019 were within the range of “normal” or were anomalous due to higher straying rates following the Big Bar slide. As fish passage is restored through the slide area, future years of monitoring would provide a better indication of annual spawner abundance of steelhead using the Seton River watershed above the Seton Dam.
2. *Entrainment*

   Evidence from the steelhead telemetry study under BRGMON-3 indicate that entrainment is occurring in the Seton system (White et al. 2019). For example, in 2019, the same tagged steelhead migrated through the Seton Dam twice with no downstream movements recorded by the counter or PIT array, indicating that this fish entrained through Seton Dam or the Seton Generating Station. While entrainment can be monitored for tagged fish, it is difficult to quantify the rate of entrainment for non-tagged fish. If entrainment is occurring and fish are passing Seton Dam multiple times, calculated abundances may be over-estimates.
3. *Steelhead use of Seton River as Spawning Habitat*

   While the counter gives a good indication of fish moving through Seton Dam to spawn upstream, it does not provide an estimate of steelhead that may be using the Seton River for spawning. PIT data from previous study years indicated that some fish may use the spawning channels, but detections have been rare. Steelhead assessments conducted by the Government of British Columbia indicated that adults were migrating through the Seton River but ultimately spawning in Cayoosh Creek (Webb et al. no date).

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

The primary monitoring activity to address MQ2 was habitat suitability surveys withing the Seton River to obtain estimates of rearing habitat quantity. Across all sampling methods, juvenile Rainbow Trout, followed by Coho Salmon and Chinook Salmon, have been the dominant species captured, and were thus the focal species for evaluating effects of flow on rearing habitats.

Habitat suitability surveys were completed in the mainstem river and side-channel habitats at discharges ranging between 12 and 143 m^3/s. River-wide, results showed that habitat suitability for all species decreased as flows increased from 12 to 60 m^3/s. Habitat for Rainbow Trout fry and parr and juvenile Coho and Chinook Salmon increased at flows from 25 to 40 m^3/s before ultimately decreasing again at flows from 40 to 60 m^3/s. Flows above the WUP maximum target of 60 m^3/s wetted side-channel habitats, making them available to juvenile fish. Available side-channel rearing areas buffered habitat losses in the mainstem from 60 to 86 m^3/s; however, any gains in rearing habitat resulting from the flooding of side-channels were lost when flows exceeded 100 m^3/s (based on partial survey data; Buchanan et al. 2020). Results indicated that the amount of available habitat suitable for juvenile Rainbow Trout, Coho Salmon and Chinook Salmon was not independent of flow releases form Seton Dam discharge and therefore we can reject H_1. Additional surveys are needed at flows ≥100 m^3/s to complete the river-wide curve beyond 86 m^3/s.

To assess potential effects of Modified Operations on habitat suitability over time for Rainbow Trout only (as they were the subject of annual abundance estimates), habitat suitability surveys were completed in the fall during baseflow conditions each year since 2018 and then compared to 2014 results prior to modified operations. Results indicated that habitat has decreased over time during the period of Modified Operations, particularly in Reach 2 (Figure 3.7). It is possible that changes in habitat suitability have contributed to the lower density and abundance of Rainbow Trout fry observed in recent years. While conditions in Reach 1 are solely impacted by flow releases from the Seton Dam, conditions in Reach 2 and Reach 3 are also influenced by Cayoosh Creek which, while regulated, has a hydrograph that mimics what may be expected in a natural river with a large (but variable) peak flow during freshet. High flow releases from Seton Dam that coincide with high flows from Cayoosh Creek would disproportionally affect the lower reaches of the Seton River (Reach 2 and 3) and may partially explain habitat changes. Though these annual habitat surveys represent only a subsample of sites rather than river-wide changes, they nonetheless act as an indicator of potential long-term changes that may be occurring within the Seton River during the period of Modified Operations.

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 in the Seton River mainstem is limited for all salmonid species and can be attributed to the relatively restricted nature of the river that has been extensively dyked or armored throughout its four-kilometer stretch. These attributes create higher velocities and greater water depths in the river along with few areas for substrate to be deposited. Throughout the monitor, visual surveys have identified two areas in the mainstem where spawning occurs: immediately below Seton Dam in Reach 1 and near the outflow of the LSC in Reach 3. Risk of redd dewatering is related to the magnitude and timing of flow events in the Seton River; risk increases when flows are higher during the spawning period followed by lower flows during the incubation period. Typically, both identified spawning areas have remained wetted at the flows experienced in the Seton River since 2014.

During periods of Modified Operations, which have typically occurred during the steelhead migration and spawning period, side-channels become wetted; any redds in these areas would be at high-risk of dewatering prior to emergence (generally June to August for steelhead). Habitat surveys in 2017 indicated that the substrate in the OCH1 and OCH2 side-channels may be suitable for steelhead spawning (Buchanan et al. 2018). Though no adults have been observed using these locations for spawning, such conditions pose a risk of redd dewatering for steelhead.

In August to September of 2019, August to October of 2020, and August to September of 2021, flow releases from the Seton Dam were held higher than WUP targets (~35 m^3/s in 2019 and 2021; ~30, 60 and 90 m^3/s in 2020) coinciding with the spawning periods of Sockeye, Pink and Chinook Salmon. In 2019, once Seton Dam discharge returned to WUP targets, stranded eggs were observed on stream margins in several places. It is unknown whether these eggs were deposited along stream margins however, as eggs were distributed in larger substrate that was unlikely to have been true redds. Based on the timing of the flow event and that 2019 was a Pink Salmon year, it seems probable that these would have been Pink Salmon eggs (though they could have also been from Sockeye or Chinook Salmon). In 2020, while no redds in the Seton River were observed nor was any evidence of redd dewatering found when flows were reduced, observed spawner densities were lower (no Pink Salmon run). Moreover, flows in 2020 were significantly greater than in 2019 and so conditions may have been unsuitable for spawning in the mainstem downstream of Seton Dam where most of the known mainstem spawning occurs for all species. Conditions in 2021 mimicked those of 2019, in that the flow conditions were similar and Pink Salmon were present in the river, however, no stranded fish or redds were observed during streamwalk surveys (Splitrock Environmental, pers. comm., June 2022). As in previous years, designated redd surveys are recommended during similar flow events; surveys should be conducted prior to flow reductions (depending on the specific flow level and visibility) and especially following the rampdown to assess stranding.

Given the variability in flow releases during the steelhead and salmon spawning periods combined with the low density of redds (thereby making *in situ* redd monitoring and dewatering assessments challenging), it would be valuable to analyze spawning habitat suitability over a range of flows. This would help inform redd dewatering risk by giving insight to which conditions may preclude spawning, and therefore would be lower risk, and which conditions may pose greater risks of stranding. However, given that Seton River is too deep and fast to wade across and accurately obtain depth and velocity measurements at even base flow conditions, obtaining data to assess spawning habitat suitability at a range of flows may be unfeasible. Overall, redd stranding risk during WUP target flows is likely low in the Seton River, however, H_2 cannot be rejected until additional data are obtained to better assess the risk of redd stranding at higher discharges.

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?

High flows can mobilize gravel (Ellis et al. 2018). Therefore, periods of high flow during Modified Operations have the potential to impact gravel availability for spawning and egg incubation in Seton River. Paired topographic and pebble count surveys on the Seton River have been completed regularly to monitor changes in riverbed elevation and substrate composition downstream of Seton Dam. Analyses of these data indicated that changes in elevation have occurred since 2013, though results were variable. Changes in elevation corresponded with changes in substrate size, as indicated by pebble count surveys completed during the same period. For example, significant scouring was observed in the spawning area just downstream of Seton Dam from 2016 to 2017 following high flows during Modified Operations, which corresponded with an increase in substrate size in the same area. Since 2017, the area has experienced a net increase in elevation (indicative of deposition) and decrease in substrate size. These results suggest that from 2016 to 2017 high flows caused mobilization of smaller gravel in the area, leaving behind slightly larger substrate (reject the first part of H_3). This was followed by a period of deposition of smaller substrate from 2017 to 2019 that was maintained from 2019 to 2021. These results indicate that the river returned to pre-Modified Operations conditions after high flows from 2016 to 2017; however, the level of scour or deposition upstream between the Seton Dam is unknown. It is possible that an upstream gravel source provided recruitment to the survey area. And because the Seton Dam blocks the downstream supply of sediment to the Seton River, this self-sustaining recruitment of suitable spawning gravel after periods of scour is likely limited. Overall, substrate size in this important spawning area has remained within the size range suitable for steelhead, Coho Salmon and Chinook Salmon; however, the mean particle size was larger than what is typically used by Pink Salmon for spawning (Kondolf and Wolman 1993). While 2020 data initially suggested the substrate size was increasing in Reach 2 and 3, 2021 data indicated that the geometric mean has been maintained. Unlike the paired topographic and pebble count surveys, the reach-wide substrate data were based on random sub-samples of sites and so it possible the larger mean sizes in 2020 were an artifact of sampling. The next surveys are due to be completed in 2022 and will continue to inform these inferences regarding the influence of the flow releases from Seton Dam on suitable spawning gravel.

Winter Modified Operations

From August through to October of 2020, the Seton Generating Station experienced an outage resulting in a back-up of water in the Bridge-Seton system. To reduce the need to spill the excess water into the Bridge River, a variance was provided to BC Hydro that allowed them to increase the discharge into Seton River over the winter. Discharge was increased from 15 m^3/s to 20 m^3/s on December 21, 2020, where it remained until March 4, 2021. The change in discharge over the winter resulted in an increase of suitable habitat for juvenile Rainbow Trout, Coho Salmon, and Chinook Salmon (see Figure 3.4), and increased water temperatures in the mainstem Seton River. Increased movements of Rainbow Trout in and out of the Upper Spawning channel were also observed.

While the additional flow from Seton Dam during the winter of 2020/2021 did not significantly impact the water temperature of Seton River in Reach 1, temperatures in Reach 2 and 3 were significantly warmer when compared to the previous years' average during the same period. The increased temperatures in the lower reaches were likely the result of the increased discharge from Seton Dam (and therefore a greater proportion of water from Seton Reservoir) that reduced the typical cooling effects of Cayoosh Creek and is consistent with the literature (Murchie et al. 2008; Heggenes et al. 2021).

Modified stream temperatures directly affect salmonid egg incubation. Warmer winter water temperatures reduce incubation periods and can cause a temporal disparity between the emergence of fry and food availability (Heggenes et al. 2021). Many benthic macroinvertebrates are better at adapting to temperature changes than the juvenile salmon that consume them, and as a result, juvenile salmon may emerge before their primary food source (Heggenes, 2021). These temporal mismatches can lead to reduced growth and survival rates for juvenile salmonids. However, as most mainstem spawning occurs in Reach 1 and the temperature change in this reach was less dramatic than that observed in the lower reaches, it is unlikely that the change would significantly impact salmon, though further years of study will be needed to confirm.

While an increase in temperatures have not been found to be a dependent variable for growth in older fish, Davidson et al. (2010) found that higher growth rates coincide with higher winter temperatures which typically occur early and late in the season and may be an indicator for increasing seasonal food availability. The more food that is available, the less calorically intensive foraging for food is, and therefore, higher growth rates can be achieved (Parrish et al. 2004). However, when food is limited during winter, fish conserve energy by moving towards deep, low flow areas with shelter availability (Parrish et al, 2004; Teirchert et al. 2010; Heggenes et al. 2021). This means that the increased discharge in Seton River through the winter and the resulting increase in temperature may lead to increased growth rates in older resident fish.

There are few studies that empirically demonstrate the effects of increased flows on juvenile salmon growth rates over winter. However, stream velocity effects the growth rate of juvenile salmon by impacting their ability to drift feed. While higher flows may result in the delivery of more food, they also increase the energy juvenile salmon need to expend to forage (Davidson et al. 2010, Piccolo et al. 2008). Additionally, increasing flows are associated with elevated turbidity, which can make it difficult to identify food (Neuswanger et al. 2014). When flows become excessive, stream dwelling salmonids will abandon foraging efforts in preference of shelter (Davidson et al. 2010). Additional variables such as temperature, localized velocities and food availability and the interactions between them likely cause varying effects on growth rates (Railsback 2021).

Increased discharge in Seton River through the winter has the potential to modify the survival and growth rate of juvenile salmonids. While increased flows in regulated rivers, including the Seton River, have resulted in warmer winter stream temperatures that may negatively impact survival and growth rate of eggs and emergent young-of-the-year, we would expect such effects to be less applicable here, as most of the known spawning locations were outside of the areas with significant temperature increases. Instead, we would expect the effects associated with increased winter discharge to be most relevant for age-0 and age-1 salmonids going into their first or second winter. Increased flow may improve drift feeding conditions by increasing the delivery rate of food, but for conditions to be favorable, there must be enough food available such that the energetic cost of foraging is less than or equal to the calories gained from eating. If food is limited during higher flows, fish are likely to adopt a shelter and starvation survival strategy. Lower trophic levels (e.g., periphyton, benthic invertebrates, drift) are not monitored in the Seton River and directly testing the effects of modified operations on food availability is not possible with the given data. Scale samples of juvenile salmonids have been collected and archived over the monitor, and so using these samples to measure and compare growth rates can be achieved (though it is outside the scope of this report). Further monitoring and analysis will be required to determine if and how increased winter flows in the Seton River are impacting juvenile salmonid growth.

5. Recommendations

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

- Continued collection of tissue samples from Chinook Salmon and Coho Salmon for DNA analysis to determine stock origins will inform the life history strategies and habitat use of rearing populations (MQ1), in addition to teasing out the effects of the Big Bar slide (versus natural variability or operations) on fish populations within the Seton River watershed. Currently, this work is being completed collaboratively with Fisheries and Oceans Canada.

- Complete river-wide habitat suitability surveys above 100 m^3/s. Partial datasets currently exist. Having the full datasets would fill in gaps for the flow versus weighted useable area (a measure of habitat suitability for target species and life stages) relationship developed for the Seton River. These data directly support MQ2 and are important for evaluating the Modified Operations hydrograph, in addition to providing context for any changes that may be proposed during future planning processes. To complete river-wide surveys, flows from Seton Dam must be held at the target levels for approximately 2 weeks.

- Complete river-wide habitat suitability surveys at 30 and 50 m^3/s. Given the increase in habitat observed at 40 m^3/s, it is worthwhile to determine a range of discharges which maximize habitat for juvenile species. To complete river-wide surveys, flows from Seton Dam must be held at the target levels for approximately 2 weeks.

- Continued use of the Seton Dam counter (regardless of whether flows are within WUP targets or are considered Modified Operations) in conjunction with the BRGMON-3 telemetry program. Counter, PIT and radio telemetry data combined have increased the understanding of steelhead migration timing, distribution, and abundance within the Seton River watershed (directly informing MQ1), as well as the potential effects of Seton Dam operations on steelhead migration. More annual data is required to discern if data collected since 2019 were the result of natural variability in migrations or fish responses to operations. As well, having additional fixed radio telemetry stations (as demonstrated under BRGMON-13 in 2020) in combination with existing mobile tracking surveys would be of value, particularly during high flow periods. The use of additional radio telemetry stations with mobile tracking surveys could replace weekly visual surveys for steelhead on the mainstem as that method has been ineffective. To increase the likelihood of detections during mobile tracking surveys, additional time could be spent in key areas. These methods could inform potential spawning locations, which may then be monitored for potential dewatering (MQ3).

- Conduct riverbed elevation surveys in conjunction with pebble counts along transects in the survey area on an annual basis, regardless of flow releases, to quantify substrate scour and deposition, as well as substrate size. These data directly support MQ4 and will improve our understanding of the effects of operations on gravel availability.

- Further analysis of existing data should be completed in 2022 to explore whether management questions and hypothesis can be properly addressed for final report.

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7. Appendices

Appendix 7-1. AICc table for predicting body condition for Age 0 Rainbow Trout. Bolded text shows best fit model.

Model	AICc	Delta
K ~ 1	3220.528	28.41

K ~ year	3211.901	19.78
K ~ reach	3201.032	8.92
K ~ year + reach	**3192.116**	**0**
K ~ year * reach	3220.136	28.02

Appendix 7-2. AICc table for predicting body condition for Age 1 Rainbow Trout. Bolded text shows best fit model.

Model	AICc	Delta
K ~ 1	1479.116	10.31
K ~ year	1472.900	4.10
K ~ reach	1473.609	4.81
K ~ year + reach	**1468.803**	**0**
K ~ year * reach	1495.575	26.77

Appendix 7-3. AICc table for predicting body condition for Age 0 Coho Salmon. Bolded text shows best fit model.

Model	AICc	Delta
K ~ 1	3327.02	166.30
K ~ year	3203.03	42.31
K ~ reach	3326.40	165.69
K ~ year + reach	3206.74	46.02
K ~ year * reach	**3160.72**	**0**

Appendix 7-4. AICc table for predicting body condition for Age 0 Chinook Salmon. Bolded text shows best fit model.

Model	AICc	Delta
K ~ 1	**2174.85**	**0**
K ~ year	2178.98	4.13
K ~ reach	2178.12	3.27
K ~ year + reach	2181.02	6.17
K ~ year * reach	2216.82	41.97