Coquitlam-Buntzen Project Water Use Plan

Lower Coquitlam River Fish Productivity Index

Implementation Year 15

Reference: COQMON-7

Coquitlam River Fish Monitoring Program

Study Period: 2000-2020

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Coquitlam River Fish Monitoring Program: 2000-2020 Results

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Executive Summary

As part of the Coquitlam River Water Use Plan (LB1 WUP), a long-term adaptive management study is being conducted in the Coquitlam River to compare anadromous fish production under two experimental flow regimes. Fish population monitoring under the first flow regime (Treatment 1, dam release flows from 0.8-1.4 m^3/s) occurred from 2000 until the completion of the Coquitlam Dam seismic upgrade in October 2008, with the exception of 2001(8 years). Fish production under Treatment 2 (release flows from 1.1-6.1 m^3/s) was initially proposed to be monitored until 2017 (9 years). This has since been extended for an additional 3 years (ending 2020) to increase the level of confidence in the inferences drawn from the study in the lead-up to the Water Use Plan Order Review. The overall objective of the Lower Coquitlam River Fish Productivity Index Monitoring Program (COQMON-07) is to provide the information necessary to better understand the trade-offs between fisheries, domestic water and power generation. To do this, COQMON-07 was designed to evaluate the fisheries benefits of each flow regime by monitoring adult escapement, and smolt and fry outmigration of four anadromous species including Steelhead Trout (*Oncorhynchus mykiss*), Coho Salmon (*Oncorhynchus kisutch*), Chum Salmon (*Oncorhynchus keta*) and Pink Salmon (*Oncorhynchus gorbuscha*). Higher returns during 2007-2016 allowed Chinook Salmon (*Oncorhynchus tshawytscha*) escapements to be estimated as well. Since 2006, night snorkeling surveys have also been included in the monitoring program to provide estimates of late summer standing stocks of juvenile Coho and Steelhead. This report summarizes monitoring results for the 8 years during Treatment 1 (2000-2008) and the 12 years of Treatment 2 (2009-2020) for the four major study components of the COQMON-07: adult salmon escapement surveys, Steelhead redd counts, juvenile standing stock surveys, and smolt and fry outmigration trapping. For Coho and Steelhead, the primary metrics for evaluating the fisheries benefits of each flow regime are smolt production. For Chum and Pink, it is the number of outmigrating fry produced per adult spawner. The evaluation also includes survival between various freshwater life-stages, fish size and habitat use. Adult abundance is given limited importance because of the large role that ocean survival plays in the number of salmon and Steelhead Trout returning to the Coquitlam River.

The primary emphasis of this report is on fall standing stock, smolt and fry outmigration, Steelhead escapement in 2020, and the 2019 escapement for Chum, Pink, Coho and Chinook salmon. Summaries of all data years for each species and life stage are presented and discussed as well. Estimates of adult escapement, late summer juvenile standing stocks and egg-to-smolt survival estimates should be considered preliminary and will change as additional observer efficiency data are accumulated in future years. Note that within this report, the precision of estimates represents the 95% confidence intervals or 95% credible intervals, the equivalent form for estimates relying on Bayesian statistics.

Coho

An estimated 11,670 Coho adults returned to the Coquitlam River study area in 2019 and continues the trend of higher returns 2010 onward. As with past year, we are unable to estimate precision due to insufficient observer efficiency data. The 2020 late summer standing stock fry estimate was 43,386 ± 30% based on night snorkel surveys, which was well within the range of previous years (18,405-91,367 fry), and above average for Treatment 2 (55,232 fry). During the spring of 2020, 8,490 ± 10% Coho smolts originating from mainstem habitats outmigrated past the lowermost trapping site. This is above the mean for Treatment 2 (7,822 smolts) but still less than the maximum for Treatment 1 (11,036 smolts) and Treatment 2 (10,953 smolts). An additional 1,680 smolts outmigrated from the four constructed off-channel habitats above RST2. This continues the trend of low production from these off-channel habitats from a high of 56 smolts/m^2 in 1997 to 5.5 smolts/m^2 in 2019. It is unlikely this decrease is the result of the flow treatment considering that supply flows to these habitats were largely unrelated to mainstem flow levels.

Using the combined Coho smolt yield from mainstem and off-channel habitat as the primary measure of freshwater carrying capacity, there was no significant difference in mean smolt freshwater production between Treatment 1 and Treatment 2 (mean: 12,949 and 12,114 smolts, respectively; 2-tailed t-test $p = 0.53$). When using only smolt yield from the mainstem of the Coquitlam River, which we consider a better indicator of treatment effects, there was also no significant difference between Treatment 1 and 2 (mean: 6,173 and 7,822, respectively; 2-tailed t-test p=0.12), even though mean yield increased 27%. Maximum smolt yield was similar during each treatment period but minimum yield was lower and occurred more frequently during Treatment 1 than during Treatment 2. This suggests that freshwater carrying capacity was not necessarily increased by Treatment 2, rather, that the minimum smolt yield was increased.

We are uncertain how much the change in Coho smolt yield was due to Treatment 2 flows, adult escapement or other regional factors that occurred over a similar time. Note that MON-7 uses a Before-After (BA) study design, which relies on the assumption the change in smolt yield is primarily the result of the flow treatments. For Coho, the moderate effect of adult escapement on smolt abundance and the rise in smolt yield in other watersheds raises the possibility that the increase was primarily the result of higher escapement rather than from the flow treatment. There is also some indication that ramping related stranding reduced fall fry abundance. The index of ramping related stranding is primarily a result of the number of Coho fry stranded through the June rampdowns during Treatment 2.

Steelhead

In 2020, 131 unique redds were counted, which we used to estimate the total escapement of 218 Steelhead. This was the lowest escapement since redd surveys began in 2005 (range: 230-870 Steelhead). Redd counts suggests that Steelhead escapements during 2005-2020 (24 to 80 adults/km, or 37,000-149,000 eggs/km,) were above that necessary to seed available juvenile habitat based on stock and recruitment data for the Keogh River (Ward et al. 1993), a well-studied coastal stream, and based on our preliminary stock-recruitment analysis from the Coquitlam River. The late summer standing stocks estimates of Steelhead fry, age 1+ and 2+ parr for 2020 was 47,408 ±34%, 6,863 ±21% and 2,057±52%, respectively. Smolt yield for the mainstem upstream of the lowermost trap (RST2) was 3,789 ± 21% in 2020, which is within the study precision target.

Mean Steelhead smolt yield for the Coquitlam River mainstem increased 29% between Treatment 1 and Treatment 2 (mean: 3,701 and 4,759 smolts, respectively) and was significantly different (2-tailed t-test, $p<0.05$). This was almost entirely a product of increased smolt yield from reach 4, which increased 144% between Treatment 1 and 2 (mean: 925 and 2,253 smolts, respectively; 2-tailed t-test, $p< 0.01$). As well, abundance in reach 4 has been at least two-fold higher than during Treatment 1 in all but one year since 2009. This continues the trend of generally increasing abundance in reach 4 from less than 400 smolts in 1996 (prior to the start of Treatment 1) to over 2,800 smolts in recent years. For reaches 2 and 3, there was no significant change in smolt yield between Treatment 1 and 2 (-6% and -8% change, respectively; 2-tailed t-test, $p< 0.05$).

Similar to Coho, we are uncertain whether a change in Steelhead smolt yield was due to Treatment 2 flows or other regional factors that occurred over a similar time. The adult-to-smolt in the Ricker stock-recruitment relationship suggests that the juvenile carrying capacity rather than adult escapement has the largest impact on smolt abundance. However, there was no clear indication that month and season specific flow metrics, or flow treatment had a substantive effect on either adult-to-fall fry survival or fall fry-to-fall age-1 parr survival.

Chum

An estimated 10,970 Chum adults returned to the Coquitlam River study area in 2019. As with past year, we are unable to estimate precision due to insufficient observer efficiency data. In 2020, 0.84 million ± 5% Chum fry outmigrated past the lowermost trap. This is below the range for Treatment 2 (range: 1.9-12.7 million fry) and at the low end of the range for Treatment 1 (0.8-3.4 million fry). Egg-to-fry survival ranged from 3.7%-26.8%, and averaged 10.0% during Treatment 1. Egg-to-fry survival during Treatment 2 averaged 24.7% and ranged from 12%-40%. Some or all our survival estimates could be biased high as they exceed the published values for Pacific Northwest streams (Bradford 1995). Survival estimates may be better interpreted as an index, only comparable within the Coquitlam River Monitoring Program. Mean survival was significantly higher during Treatment 2 then during Treatment 1 (2-tailed t-test $p=0.01$). Preliminary stock-recruitment analysis also suggests that Treatment 2 likely increased fry production compared with Treatment 1. These findings could change as we further refine the Chum escapement model as well as with further comparisons with other rivers. Chum salmon returns to Coquitlam River were greatly improved in 2002-2018 compared to escapements in years prior to the implementation of the Treatment 1 flow regime in 1997.

Pink

Pink escapement for 2019 was 1,211, the lowest for the study period. The 2020 estimate of fry yield of 197,838 ± 36% was at the low end for Treatment 2 (range: 114,000-6.7 million fry). Fry yields during Treatment 2 have been up to 20-fold higher than Treatment 1 and mirrors the significant changes in fry yield in the Cheakamus River since 2006. Egg-to-fry survival ranged from 5.1%-9.7% up to 2009, which was comparable to reported values for other streams and potentially biased high survival rates of 24%-48% for the 2011-2015 broods. The good fit of the linear adult-to-fry stock-recruitment relationship including all Treatment 1 and 2 years (R^2 = 0.87) or only those years with comparable adult escapement (R^2 = 0.90) suggest that adult escapement was the single largest driver of fry abundance and that fry abundance was minimally affected by the availability of spawning habitat. With only two run-years under Treatment 2 conditions at comparable escapement levels to Treatment 1, between-treatment comparisons are weak. For years with comparable escapements, there was also no significant difference in egg-to-fry survival. If future escapement continues at the lower levels similar to Treatment 1, additional monitoring will greatly improve the reliability our ability to assess whether productivity differs between Treatments 1 and 2.

Chinook

The Chinook escapement estimate for 2019 is 591 spawners. Escapement 2007-2016 ranged from 360-8,000 adults, and was likely less than 300 adults prior to this period. The highest Chinook escapement occurred in 2010 (8,018 adults). Too few Chinook smolts (114) or fry (415) were captured in downstream trapping to estimate the number outmigrating out of the Coquitlam River.

COQMON-07 Status of Objectives, Management Questions and Hypothesis after Year 21 of Treatment 1 and 2

Primary Objective	Management Question	Management Hypothesis	Year 21 Status
To determine the fisheries benefits associated with the two test flows : Treatment 1 – 2FVC Treatment2 – STP6	Has juvenile rearing capacity of the Coquitlam river changed as a result of flow treatments for Steelhead and Coho?	H_0 –Steelhead smolt production does not differ between Treatment 1 and 2	H_0 –status unclear. Reject if based only on change in Coquitlam River smolt production. Unclear of the effect of flow treatment on increased production. Section 6.2
		H_{01} – Coho smolt production does not differ between Treatment 1 and 2	H_{01} – status unclear. Possibly reject if based only on change in Coquitlam River smolt production. Unclear of the effect of flow treatment on increased production. Section 6.1
	Has Chum and Pink juvenile productivity changed as a result of flow treatments in the Coquitlam River?	H_{03} –Each adult Chum produced the same fry yield during Treatment 1 and 2. Stock-recruitment relationships unchanged	H_{03} – possibly rejected. High moderate support of a treatment effect. Section 6.3
		H_{04} –Each adult Pink produced the same fry yield during Treatment 1 and 2. Stock-recruitment relationships unchanged.	H_{04a} – not rejected H_{04b} – not rejected No indication of a treatment effect but sample size is very limited. No indication of habitat limitation under high Treatment 2 escapement levels. Section 6.4

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1. Introduction

The development of the Coquitlam-Buntzen facilities Water Use Plan (LB1 WUP) was initiated in September 1999 and completed in March 2003. As part of the LB1 WUP, the LB1 WUP Consultative Committee (CC) made recommendations on dam releases in the Coquitlam River based on trade-offs between power, drinking water and fisheries values (BC Hydro 2003). The LB1 WUP was also designed as a long-term adaptive management experiment to compare different flow regimes for the Coquitlam River below the Coquitlam Lake Reservoir. The effect of different flows and other types of enhancements on the productivity of anadromous salmonid populations are often difficult to detect because of the high degree of natural variation in both freshwater and ocean survival (Keeley and Walters 1994; Bradford 1995). Relying on a study by Higgins *et al*. (2002) that looked at the statistical power to detect changes in fish production in the Coquitlam River under different flow regimes, the CC selected two flow regimes for comparison: the regime of two fish valves fully open (2FV, Treatment 1), and the Share the Pain#6 flow regime (STP#6, Treatment 2) prescribed by the CC (Table 1.1) that attempts to improve spawning and rearing habitat conditions in the Coquitlam River relative to Treatment 1.

Table 1.1 Scheduled monthly flow releases from Coquitlam Dam under Treatments 1 and 2 of the Coquitlam River Water Use Plan (BC Hydro 2003a).

	Reservoir diversion schedule (m^3/s)					
	Domestic water		Coquitlam Dam releases			
			Treatment 1	Treatment 2		
Period	Target	Min		Target	Min	Target species and life stage
Jan 1-15	11.9	10.7	1.0	5.9	3.6	Chinook spawning
Jan 15-31	11.9	10.7	1.0	2.9	2.9	Chinook incubation
Feb	11.9	10.7	1.0	2.9	1.8	Chinook incubation
Mar	11.9	10.7	0.8	4.3	1.1	Steelhead spawning
Apr	12.0	10.8	0.8	3.5	1.1	Steelhead spawning
May	12.0	11.0	1.1	2.9	1.1	Steelhead spawning
Jun	12.0	10.9	1.4	1.1	1.1	Steelhead parr
Jul	18.0	15.8	1.4	1.2	1.1	Steelhead parr
Aug	23.0	20.2	1.1	2.7	1.1	Steelhead parr
Sep	23.0	20.9	0.8	2.2	1.1	Steelhead parr
Oct	12.0	10.8	0.8	6.1	3.6	Chinook spawning
Nov	12.0	10.8	1.1	4.0	1.5	Chinook spawning
Dec	11.9	10.7	1.1	5.0	2.5	Chinook spawning

1.1 Background

The lower Coquitlam River flows 17 km from the base of the Coquitlam Dam to its confluence with the Fraser River. The stream was first dammed in 1903. The present dam dates back to 1914. As part of LB1 WUP (BC Hydro 2003a), flows in the lower Coquitlam River are regulated through the Coquitlam Dam's low-level outlets, which release flows from the Coquitlam Reservoir. The Coquitlam Reservoir supplies drinking water for the Greater Vancouver Regional District (GVRD) and water for power via a diversion tunnel to Buntzen Lake.

Typical of lotic habitats downstream of dams, spawning and rearing habitat in the lower Coquitlam River (hereafter referred to as the Coquitlam River) has been impacted over the last hundred years by reduced gravel recruitment from upstream sources and increased sedimentation due to reduced peak flows (NHC, 2001). Several gravel pit operations adjacent to Coquitlam River have also affected habitats by contributing large amounts of fine sediment directly to the system. Other impacts are typical of urban streams, and include extensive channelization and dyke construction, road and bridge crossings, alteration of natural drainage patterns and tributary inflows, and the increased discharge of pollutants. Peak, post-dam flows in Coquitlam River can exceed 200 cubic metres per second (m^3/s, Water Survey of Canada, Station 08MH141). Prior to June 1997, flow releases from the dam ranged from 0.06 to 0.5 m^3/s (not including occasional spill events). From 1997 to October 2008, minimum flow releases were increased to a range of 0.8 to 1.4 m^3/s, depending on the time of year. This represents the Treatment 1 regime of two fish valves fully open (2FV), and is the baseline for this adaptive management study.

The Treatment 2 flow regime, Share the Pain#6 (STP#6) was initiated on October 22, 2008, with seasonal target flow releases from Coquitlam Dam ranging from 1.1 to 6.1 m^3/s (Table 1.1). After the knife gate was put into operation, BC Hydro personnel conducted measurements of actual flows during the spring and summer of 2009, and these were compared to modeled flows to verify the theoretical discharge rating curve through the various flow ranges. These measurements indicated that actual flows were consistently higher than predicted (2 m^3/s higher on average than seasonal target releases, Table 1.1). During August and early September, 2009, BC Hydro's Engineering Group worked on updating the flow rating curve for the new gate facility. Once sufficient data was collected, the discharge rating curve was adjusted and brought into service on September 15, 2009. As a consequence, with respect to the flow experiment, 2009 was not strictly representative of Treatment 2. However, given the planned 9 year duration of Treatment 2, this was not considered to have a significant impact on the comparison of the two treatment periods.

The Coquitlam River historically supported all six Pacific salmon, as well as Cutthroat Trout (*Oncorhynchus clarki*), which are still present in low numbers, and Dolly Varden (*Salvelinus malma*) char, which appear to have been extirpated. Dam construction also resulted in the extirpation of an anadromous stock of summer Sockeye Salmon (*Oncorhynchus nerka*), but this species stills exists in the Coquitlam Reservoir in its resident form (Kokanee). Other species inhabiting the Coquitlam River below the dam include Longnose Dace (*Rhinichthys cataractae*), Prickly Sculpin (*Cottus asper*), Redside Shiner (*Richardsonius balteatus*) Pacific Lamprey (*Entosphenus tridentatus*), and Three-Spine Stickleback (*Gasterosteous aculeatus*).

1.2 Study design

Prior to the implementation of the monitoring program, the CC evaluated several potential flow regimes using flow-habitat models for target species and life histories, with habitat treated as a surrogate for fish productivity (BC Hydro 2003b). Habitat modelling suggested that increased base flows in late summer under Treatment 2 could increase the quantity and quality of juvenile rearing habitat for species with longer freshwater residency periods including Chinook Salmon (*Oncorhynchus tshawytscha*) and Steelhead Trout (*Oncorhynchus mykiss*), and that increased fall and spring base flows could improve spawning success for all anadromous salmonids.

The LB1 WUP was developed as an adaptive management study with the objective of ensuring sufficient information is in place to determine the fisheries benefits of the two test flows and to enable a better understanding of the trade-offs between fisheries, domestic water and power generation. From this a primary management question was developed:

What are the fisheries benefits of flow Treatment 1 (2FV) and Treatment 2 (STP6)?

To answer this, a Before – After (BA) experimental design was developed that included juvenile outmigration as well as adult escapement monitoring with six years of monitoring Treatment 1 flow conditions followed by nine years of monitoring under Treatment 2 conditions. While Steelhead Trout and Chinook Salmon were identified as the key species of interest, lower Chinook abundance necessitated the use of Coho Salmon (*Oncorhynchus kisutch*), Chum Salmon (*Oncorhynchus keta*) and Pink Salmon (*Oncorhynchus gorbuscha*) as surrogates for monitoring.

An expected effect size (increased freshwater productivity) from the Treatment 2 flow regime was not defined in the TOR, however the Fisheries Technical Committee (FTC) predictions ranged from a 0% - 100% increase in productivity from Treatment 1 to Treatment 2. Higgins *et al*. (2002) summarized that this uncertainty in the fish benefits from alternative flow regimes, including STP#6, were mainly due to: 1) the fundamental uncertainty about the functional relationships between flow-habitat-fish for the Coquitlam River, 2) the poor contrast in available hydrometric data needed to calibrate hydraulic models which necessitated extrapolation of habitat predictions to flow levels above that observed but under consideration by the WUP Consultative Committee, and 3) the unknown influence of spawning substrate quality issues from gravel mining operations and the practical extent to which that impact could be mitigated by implementing deliberate flushing flows from the Coquitlam Dam.

The Terms of Reference (BC Hydro 2006) anticipated this experimental design to have a probability of 0.51-0.59 (power) to detect a 50% increase in abundance (effect size) and characterizes this level of statistical power as ‘moderate’. However, moderate power typically reflects a probability of 0.8, whereas a probability of 0.5 is more typically considered ‘low’. Power levels of 0.51-0.59 reflect the odds of detecting a given change only slightly better than flipping a coin. This review will use 0.8 as the benchmark for acceptable power, which is consistent with the approach used in the power analysis of the Coquitlam WUP Monitoring Program (Higgins *et al*. 2002).

COQMON-07 focuses on four species: Steelhead Trout, and Coho, Chum, and Pink Salmon. Other fish species in the Coquitlam River are either of too low abundance to effectively monitor, although this appears to be changing for Chinook (see Section 1.2.1), or are not considered to be as high in economic, recreational, or cultural importance. Adult escapement and smolt/fry outmigration are monitored for all four target species. In addition, beginning in 2006, fall juvenile standing stock was assessed for Coho and Steelhead. Coho and Steelhead smolt production is the primary performance measure for the flow experiment. Coho and Steelhead have lengthy freshwater residencies relative to other target species, and smolt production for these species was judged to be the best indicator of the effects of flow management and dam operation on freshwater production. There is much research (e.g., Bradford and Taylor 1996; Ward and Slaney 1993) suggesting that Coho and Steelhead smolt production is limited primarily by habitat carrying capacity at all but very low levels of adult escapement. If adult returns are insufficient to seed available juvenile habitat, then recruitment effects may confound the relationship between smolt production and habitat. Monitoring escapement in addition to smolt production for Coho and Steelhead allows freshwater production to be evaluated under a scenario of recruitment-limited smolt production by substituting smolts per spawner or egg-to-smolt survival for absolute smolt production, but only if enough years of data are available to reliably define the stock-recruitment relationships. At the least, monitoring escapement provides a means of assessing whether escapement was adequate to seed available habitat based on comparisons with other systems for which reliable stock-recruitment data are available. Monitoring the fall standing stock of juvenile Coho and Steelhead, together with smolt production, is potentially useful in addressing questions about freshwater production bottlenecks in the Coquitlam River. Such as, is overwintering habitat more important than summer rearing habitat in limiting juvenile carrying capacity?

For Chum and Pink, which migrate to saltwater shortly after emergence, habitat conditions in the Coquitlam River determine the quantity and quality of available spawning substrate and incubation conditions for eggs. For these species, fry production and egg-to-fry survival are the most important indicators of freshwater production. Figure 1.1 provides a periodicity chart for different life stages of numerous anadromous salmonids in the Coquitlam River.

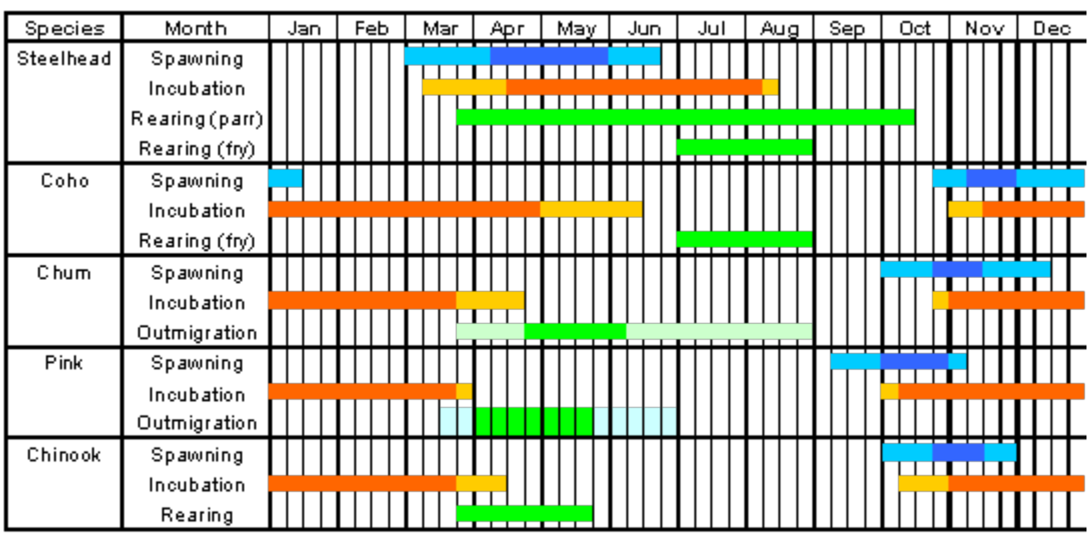

Figure 1.1 Life stage periodicity chart for some anadromous salmonids in the Coquitlam River.

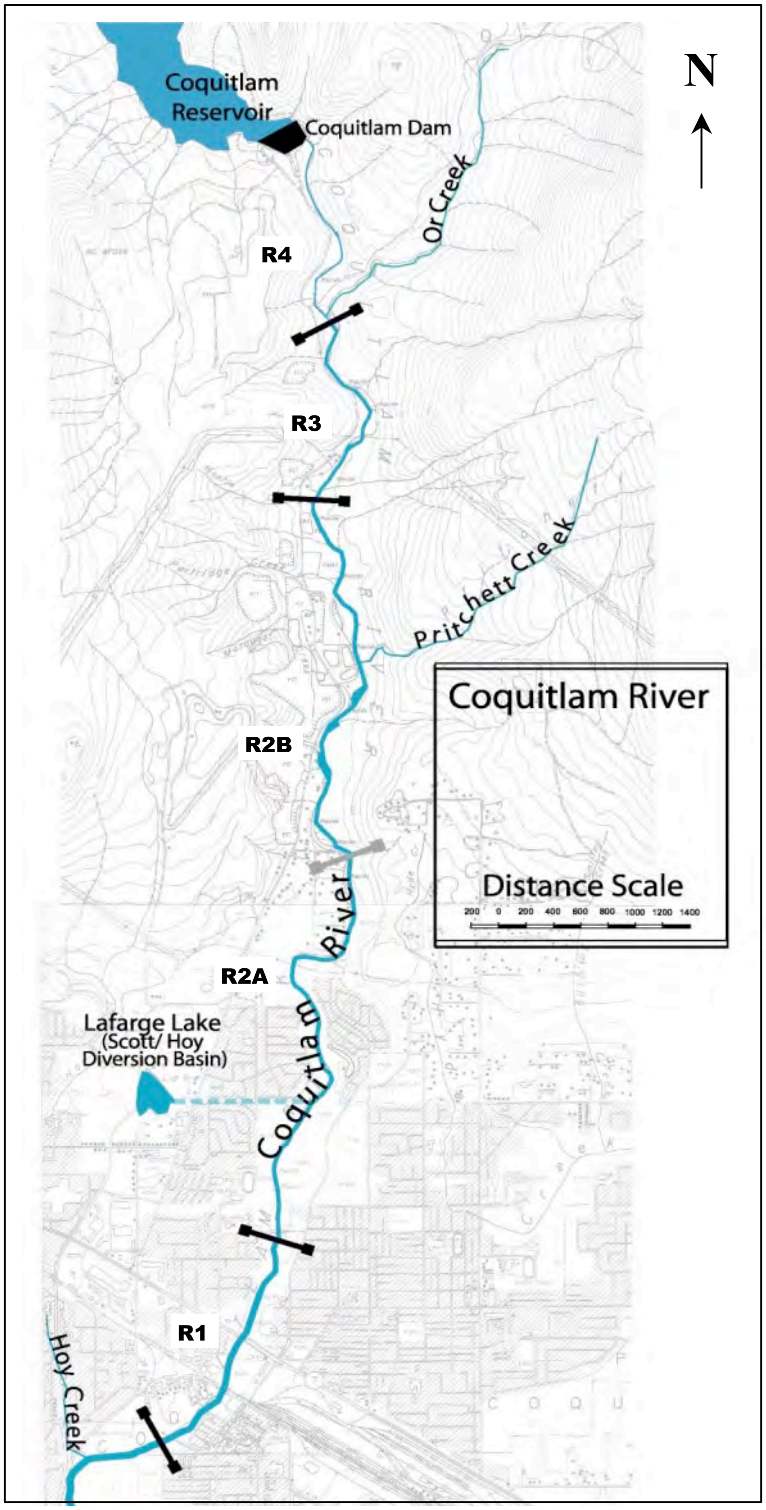

Figure 1.2 Map of lower Coquitlam River study area with stream reaches defined by the Coquitlam-Buntzen Water Use Plan Consultative Committee.

1. Introduction

COQMON-07 focuses on the effects of dam releases on the fish productivity of mainstem habitats in reaches 2a, 2b, 3 and 4, of the Coquitlam River (Figure 1.2). These reaches contain the majority of productive spawning and rearing habitat in the Coquitlam River (Riley *et al*. 1997; Macnair 2005). The actual boundaries of the study area vary somewhat among components of the monitoring program due to sampling constraints or species distribution (see Sections 1.2.1-1.2.4). Within reaches 2-4, spawning and rearing for Steelhead, Chum and Pink is largely confined to the mainstem (Macnair 2005; Decker *et al*. 2006). Or Creek is a high gradient, nutrient-poor stream, with limited accessible length, and is the only significant tributary (Figure 1.2). There are several other tributaries, but they are small, with accessible lengths limited to a few hundred metres. In addition to natural habitat, six large off-channel habitats, totalling about 27,000 m^2 have been constructed in reaches 2-4 (Decker and Foy 2000). The contribution of tributaries and off-channel sites to the production of Steelhead, Chum and Pink is low, but off-channel sites are used extensively by Coho for spawning and rearing. Constructed off-channel habitat contributes 33%-77% of Coho smolt production in reaches 2-4 (Decker *et al*. 2009).

The principal objective of this report is to summarize the fish productivity in the Coquitlam River during Treatment 1 and the first 12 years of Treatment 2 by providing population estimates at each monitored life stage for the four target species. This report also provides a thorough description of the study design and sampling methodologies for each component of COQMON-07, an evaluation of potential limitations or problems with the existing study design, and recommended changes to be applied for future years. The remainder of the report is organized in six parts (Sections 2-7). The first four parts (Sections 2-6) address methods and results for the four monitoring components of the COQMON-07: adult salmon escapement surveys, Steelhead redd surveys, juvenile standing stock surveys, and smolt trapping respectively. A discussion of the technical aspects and issues with each monitoring component is included at the end of each of these sections. The rationale for each of the four components and a summary of work completed to date are provided in Sections 1.2.1-1.2.4 below. In the final section of the report (Section 7), fish production across life stages is synthesized for each species for the study period to date. Where possible, we compare productivity data for the Coquitlam River to that in other regulated and non-regulated control streams within the region in order to assess the relative productivity of the Coquitlam River in its current state, and to examine whether recent trends in the Coquitlam River have followed those observed in other streams.

1.2.1 Adult salmon escapement

Formal surveys of adult salmon escapement were included as a component of COQMON-07 beginning with Chum and Coho salmon in 2002, and Pink salmon during odd years in 2003. Chinook were also enumerated during surveys in all years, but in monitoring years prior to 2007 Chinook escapements were negligible, and were not estimated as part of the monitoring program. During 2007-2014, Chinook escapement increased substantially, largely as a result of hatchery enhancements (M. Coulter-Boisvert, DFO, pers. comm.), and we were able to generate escapement estimates for these years. It should be noted that because adult salmon monitoring was started after smolt monitoring, estimates of egg-to-smolt survival for Treatment 1 were limited to six, three and five years' of data for Chum, Pink and Coho, respectively (smolt abundance was not estimated for Chinook).

During 2002-2016, 2018 and 2019, weekly total counts of live adults by shore-based observers and area-under-the-curve (AUC) methodology was used to estimate adult salmon abundance. The AUC approach requires accurate information about observer efficiency and average spawner survey life (Perrin and Irvine 1990). In 2006 we began conducting mark-recapture studies to generate observer efficiency and survey life estimates for Chum and Pink salmon in the Coquitlam River. Salmon escapement estimates appearing in this report differ from previous years' estimates due to the incorporation of new observer efficiency and survey life data based on mark-recapture experiments and the integration of subjective estimates of observer efficiency made by survey crews for individual years, surveys and stream sections. Escapement estimates will continue to evolve in future years as more mark-recapture data is collected and the escapement model is refined. This report describes in detail the results of the 2019 adult salmon escapement results and summarizes population estimates for 2002-2016, 2018 and 2019. Due to a transition in the monitoring program, no salmon escapement monitoring was completed during the fall of 2017.

1.2.2 Adult Steelhead escapement

The assessment of adult winter Steelhead escapement, in the form of redd surveys, was included as a component of COQMON-07 starting in 2005. Because Steelhead escapement monitoring was not included as part of the flow experiment until 2005, estimates of egg-to-smolt survival were available only for 2007 onward, which limits egg-to-smolt survival estimates to just one year for Treatment 1, which includes the yield of age-2 and age-3 smolts in 2007 and 2008, respectively, from the 2005 escapement year. Prior to 2005, snorkeling crews conducted periodic counts of adult Steelhead in some years (2001-2004) but no attempt was made to relate these counts to actual escapement. With the exception of 1999, when redd counts were conducted in reaches 3 and 4 (see Decker and Lewis 1999), pre-2005 surveys did not include counts of Steelhead redds. Because of the protracted migration and spawning period for winter Steelhead in the Coquitlam River (4-5 months), the high variation in stream residence time among individual fish (Korman *et al*. 2002), and the highly variable survey conditions within the spawning period, reliable information about residence time and observer efficiency would be needed in order to estimate escapement using counts of adult Steelhead and the area-under-the-curve methodology (Korman *et al*. 2002). This was considered unfeasible within the scope and budget of the monitoring program given the considerable cost of collecting such information, and the difficulty tagging sufficient numbers of individuals each year from this relatively small population.

Alternatively, in streams that are well suited to the method, redd count surveys can provide a more reliable index of inter-annual trends in escapement than the AUC-type adult surveys. Redd counts can be excellent predictors (R^2 values > 0.9) of Steelhead escapement as estimated by direct trap counts (Freeman and Foley 1985), resistivity counters (Korman and Schick 2015), mark-recapture studies (Jacobs *et al*. 2002) or the AUC methodology (Gallagher and Gallagher 2005). A pilot study conducted in reaches 3 and 4 in 1999 (Decker and Lewis 1999) indicated that conditions during the spring Steelhead spawning period in the Coquitlam River were, for the most part, well suited to redd count surveys. Estimating uncertainty (95% confidence bounds) for Steelhead escapement estimates derived from redd counts would require the concurrent use of a second more accurate method (e.g., resistivity counter or mark-recapture program). This is beyond the scope of the current study. Thus, estimates of Steelhead escapement and egg deposition for the Coquitlam River are reported without estimates of precision. Our reliance on literature values for sex ratio and fecundity values could reduce the accuracy of escapement estimates, however, they likely remain reliable as an index of relative change across years.

1.2.3 Juvenile Coho and Steelhead standing stock

In 2006 the CC requested that a juvenile standing stock survey component be added to the Coquitlam River Monitoring Program to provide an index of annual abundance for age-0+ Coho and age-0+ to age-2+ Steelhead. These data, together with adult escapement and smolt abundance estimates, are useful for examining freshwater production bottlenecks at specific juvenile life stages that may relate to specific habitat or flow issues. In September 2006, we conducted a feasibility study to determine the best method for sampling juvenile populations. We compared closed-site three-pass removal electrofishing to open-site night snorkeling counts at 20 m long, one-shoreline sites. We also compared results from shoreline sites and sites that spanned the entire stream channel, using snorkeling counts only. The results suggested that sampling juvenile abundance at full channel sites using night snorkeling counts would be the most effective method for monitoring annual juvenile standing stocks in the Coquitlam River (Decker *et al*. 2007). Juvenile standing stocks were assessed during 2006 onward using this methodology; mark-recapture experiments were conducted during 2007-2013 to estimate snorkeling detection probability (the percentage of fish present that snorkelers detect), so that snorkeling counts could be expanded to population estimates. This report describes in detail the results of the 2020 juvenile standing stock survey, and summarizes preliminary population estimates for 2006-2020.

1.2.4 Smolt outmigrant trapping

Smolt trapping has occurred in the Coquitlam River in various years since 1993 (see Decker and Lewis 2000 for a summary of earlier work). However, earlier studies were intended to compare smolt production at several constructed off-channel habitat sites to that in reach 4 of the Coquitlam River mainstem, as opposed to assessing production in the mainstem as a whole. During 2000-20120, numbers of Coho and Steelhead smolt outmigrants were assessed for a 7.5 km long section of Coquitlam River mainstem that included reaches 3 and 4 and most of reach 2a. Smolt numbers were also assessed for individual mainstem reaches and for the four off-channel sites. Chum and Pink smolt numbers were monitored for the same section of the mainstem beginning in 2003. Smolt numbers in the mainstem were assessed using mark-recapture methodology and rotary screw or incline plane traps. Full-span downstream weirs were used at the off-channel sites. This report describes in detail the results of the 2020 smolt trapping program and summarizes population estimates for all species and reaches for 2000-2020.

2 Adult Salmon Escapement

2.1 Methods

Salmon escapements are often estimated by obtaining repeat counts of the number of fish present over the spawning migration. The number of live spawners present that are detected by the survey crew (observer efficiency) and the proportion of the total run that is present must both be estimated on each survey to determine the total escapement. The total number of fish present on a survey is simply the difference between the cumulative arrivals and departures on that date. The departure schedule will be determined based on the arrival schedule and the length of time spawners remain in the survey area (survey life). The proportion of the run that is present on any survey date can therefore be estimated from data on at least two of the three run timing components: arrival timing, survey life, and departure timing.

Analytical approaches for estimating escapement from repeat count data have advanced considerably from the original AUC methodology (e.g. English *et al*. 1992). Hilborn *et al*. (1999) used a maximum likelihood approach to estimate escapement and arrival timing parameters by assuming that survey life was constant, and that on average, all fish present in the survey area were counted. Korman *et al*. (2002) estimated escapement from repeat mark-recapture experiments in conjunction with more flexible arrival timing and survey life models. Escapement estimates will be uncertain if there are no post peak counts (Hilborn *et al*. 1999, Adkison and Su 2001), or if peak and post peak surveys occur during periods of low catchability (Korman *et al*. 2002). In these situations, the possibility of a large number of fish entering at the peak or late in the run cannot be discounted in the estimation process because there is little information about arrival timing in the repeat count data.

2.1.1 Stratified index survey design

Returning spawners to the Coquitlam River were enumerated by stream walk surveys conducted on an annual basis during 2002-2016, 2018-2019 for Chum and Coho, and during odd years for Pink. Chinook were also counted during this time period, but prior to 2007 peak live counts were only 21 to 87 fish (J. Macnair, Living Resources Environmental Consultants, data on file), suggesting annual escapements of < 100 to 300 fish. During 2008-2013 counts of Chinook were substantially higher, largely as a result of hatchery enhancement (S. Ducharme, DFO, pers. comm.; Appendix 2.1), and we have included estimates of Chinook escapement 2008 onward in this report. In this report, we have included escapement results for all four species for 2002-2019. No escapement monitoring was completed during the fall of 2017.

For adult salmon, the study area extends downstream from Coquitlam Dam to the downstream boundary of Reach 1 at the Maple Creek confluence, encompassing reaches 1-4 in their entirety (Figure 1.2). Reach 0 (Fraser River confluence to Maple Creek) was excluded as it contains little spawning habitat and because fish entering the Hoy/Scott Creek system often hold in this reach and could be confused with fish destined for upper reaches in the Coquitlam River.

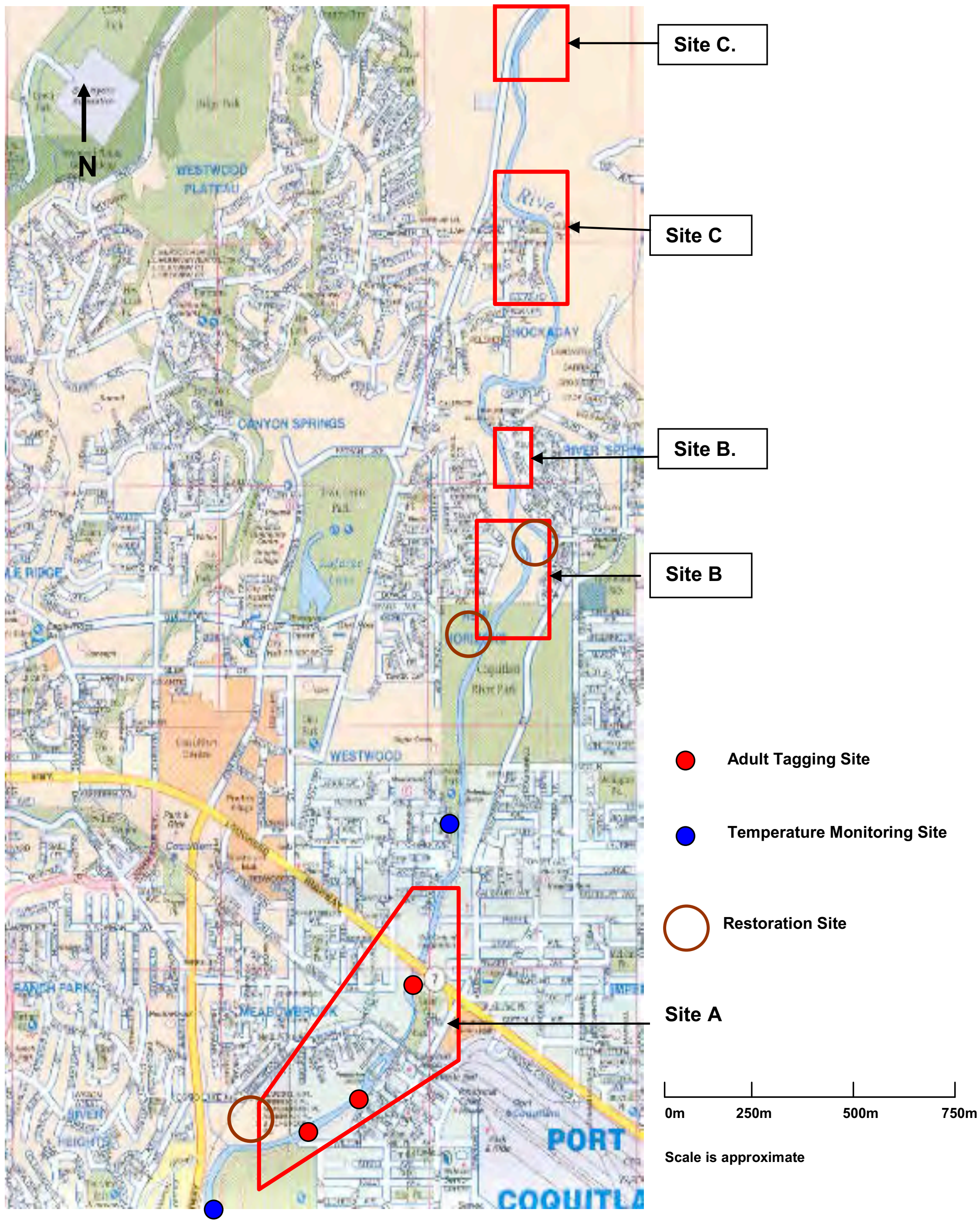

Figure 2.1 Map showing adult spawning index sites A-C in the lower portion of Coquitlam River study area (reaches 1, 2a)

2. Adult Salmon Escapement

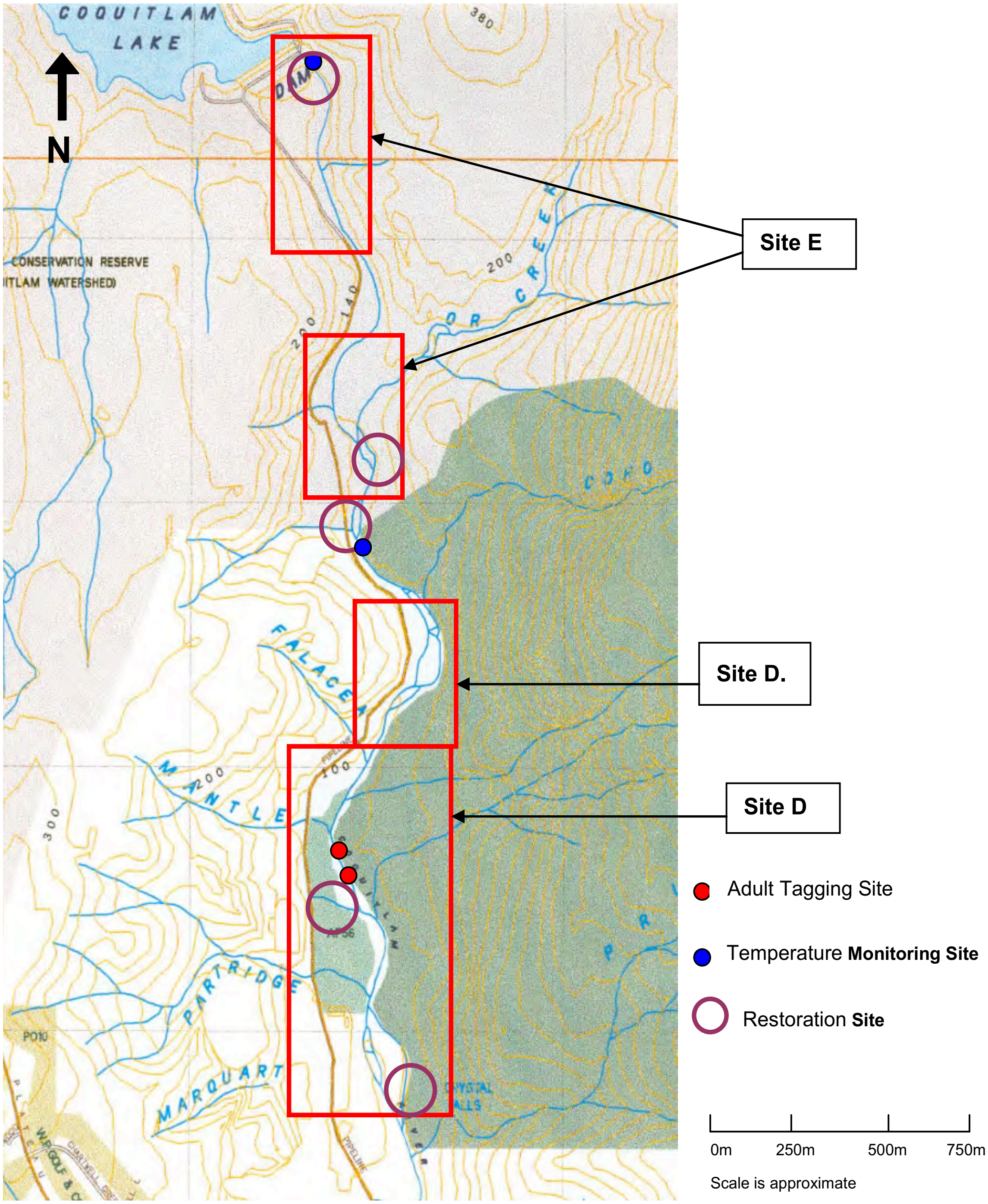

Figure 2.2 Map showing adult spawning index sites D and E, in the upper portion of Coquitlam River study area (reaches 2b, 3 and 4).

2. Adult Salmon Escapement

Considerable overlap exists for the spawning periods of Pink (early to mid-September – late October), Chinook (mid-September – mid-November), Chum (mid-October – early-December), and Coho (mid-October – mid-January). To address this, we conducted concurrent counts for whichever species were present during a particular survey. Surveys were scheduled to occur weekly throughout the entire spawning period, with the first survey date adjusted to capture the arrival of Pink and/or Chinook, and the last survey date dependent on the end of the spawning period for Coho. However, surveys were often cancelled or postponed due to poor water clarity conditions.

Due to the length of the study area (approximately 12.8 km), and the concentration of spawning activity within specific sections, sampling efficiency was improved by stratifying the survey to focus on five key areas hereafter referred to as index sites A-E (Figures 2.1, 2.2). Irvine *et al*. (1992) demonstrated that using a stratified index design to select areas to conduct visual surveys for adult Coho provided accurate estimates of escapement at a lower cost than more intensive methods such as mark-recapture or operation of counting fences. Coquitlam River index sites were originally developed from spawning distribution maps developed as part of the LB1 WUP. The boundaries of these sites were refined during the first several years of the study under Treatment 1, and were further refined over the first several years of the study under Treatment 2. The five index sites have a collective length of approximately 9 km, or 63% of the total length of the survey area, but account for a higher percentage of the total fish present during any one survey because they encompass the majority of available spawning habitat. All potential holding and spawning habitats were surveyed within each index site, including mainstem areas, natural side-channels and braids, and constructed off-channel habitat.

To account for spawners present in the study area, but not in one of the five index sites, on several occasions each year, the survey was extended to include the entire 12.8 km length of the study area. We attempted to complete three full surveys of the study area during the spawning period for each species, with some dates providing full surveys for more than one species. To address possible temporal variation in the proportion of spawners in non-index sites, surveys were scheduled in an attempt to capture early, peak, and late portions of the spawning period for each species. There were occasions each year when it was not possible to survey all five index sites due to poor water visibility. We used data from complete surveys of the study area to 'fill-in' counts for non-surveyed index sites and non-index sites on occasions when not all of the study area was surveyed (see Section 2.1.2).

Spawner surveys were performed by a crew of two people, equipped with chest waders and polarized glasses, who traveled in an upstream direction, with one person on either side of the river. The survey team has been very consistent over the project life (1st Crew member: Jason McNair, 2002-2016; 2nd crew member: Gord Lewis 2002-2006; Kris Kehler 2007, 2015-2016; Thibault Doix 2008-2015). This relative consistency likely reduced between-observer variance. The survey crew minimized the likelihood of making duplicate counts by regularly discussing which portions of the river channel each person was responsible for. Surveyors carried walking staffs that they used to probe under cutbanks and LWD accumulations in order to detect fish that were not in plain view. Total numbers of live and dead adults were recorded during each survey, but only data for live fish were used to estimate escapement. In most cases, stratified counts of the five index sites were completed in one day, while surveys of the entire study area were completed over two days.

With the onset of Treatment 2 in October 2008, dam releases during the spawning period increased, particularly during the latter part when the majority of Coho spawning occurred. In 2009, the survey crew concluded that, shore-based observations were less effective for Coho under the new flow regime because of increased water depths and turbulence in many areas where these fish were found. During the latter part of the survey period in 2009 (December-January), the survey crew opted to modify the survey design by incorporating one crewperson equipped with a dry suit and snorkeling gear, in addition to 1-2 shore-based observers. Comparisons of counts made by snorkelers and shore-based observers suggested that snorkelers detected 4- to 6-fold higher numbers of Coho than shore-based observers under Treatment 2. The effect of this shift in protocol with regard to estimating Coho escapements is discussed in Section 2.2.2. Field crew did not report an obvious difference in the detectability of other salmon species between Treatments 1 and 2, and there was some support for this based on similar mark-recapture derived estimates of observer efficiency for Chum salmon under the two treatments (see Section 2.2.2).

2.1.2 Partial surveys, observer efficiency, and survey life

Frequent high flow events and associated high turbidity during the fall and winter spawning period contribute substantially to the uncertainty of salmon escapement estimates in the Coquitlam River (Decker *et al*. 2008). During 2002-2019 it was common for surveys to be postponed for as long as three weeks, or for some portions of the study area to be excluded from a survey, due to poor water visibility. In some cases, this resulted in poorly defined run timing curves for one or more species. The COQMON-07 Terms of Reference and previous analyses of spawner survey data for Coquitlam River (Macnair 2003, 2004, 2005, and 2006) do not explicitly consider negative bias in escapement estimates caused by partial surveys. In computing escapement estimates presented in this report, we corrected for negative bias arising from partial surveys by deterministically 'infilling' (i.e., approximating) counts for missed index or non-index sites prior to running the escapement model. Uncertainty from this process was not incorporated into abundance estimates. We used year-specific ratios of spawner counts in missed sites to spawner counts for the entire study area to infill missing counts for specific sites during specific surveys. First, for each year, we computed the ratio of spawners counted in each index site (and for the non-index sites as a whole) to the total spawner count for all complete surveys. These values were then averaged across complete surveys to obtain an average ratio for each site for each year. These ratios were then used to infill missing counts for each site. For example, if, for Coho salmon, the average ratio of counts at the non-index sites to counts for the entire study area in 2009 was 0.15, and the non-index sites were not surveyed on December 13, the total count for the study area for the December 13 survey would be expanded such that:

$$\text{Expanded total count} = (\text{total count}_{\text{sites A-E}}) / (1 - 0.15). \quad (2.1)$$

Information about observer efficiency and survey life is essential for the accurate estimation of salmon escapement (Irvine *et al*. 1992; Korman *et al*. 2002). During 2006-2016, we conducted 22 mark-recapture experiments to obtain estimates of observer efficiency and survey life for the four salmon species in the Coquitlam River (Table 2.2; Appendix 2.2). Note that no additional mark-recapture experiments were conducted in 2016 due to the frequency of high-water events. Mark-recapture experiments did not occur for Coho and Chinook until 2010 because these species are less abundant in the Coquitlam River, and it was decided at the beginning of the mark-recapture program that resources were insufficient to provide for the amount of fieldwork that would be required to capture and tag sufficient numbers of these fish. Due to the greater need for Chum and Pink escapement estimates for addressing the management question, we have discontinued the mark-recapture program for Chinook and Coho, instead shifting these resources to Chum and Pink. We attempted to minimize the length of time from when a fish arrived in the study area to when it was tagged (i.e., minimize negative bias in estimated survey life) by tagging fish near the downstream boundary of the study area, under the assumption that these would be predominately new arrivals. We also concentrated on fish holding in pools rather than those actively spawning, and avoided tagging fish exhibiting the physical characteristics of advanced sexual maturation. However, in some cases it was necessary to capture and tag salmon at locations further upstream in order to deploy an adequate number of tags and this could negatively bias survey life estimates (see Section 2.2.2.2). Beach seining was used as the primary method of capturing fish, but monofilament tangle nets were sometimes used as well when turbidity was very low. Standard Petersen disc tags were used to tag fish, with different colours used to distinguish temporal mark groups.

To provide estimates of observer efficiency (i.e., proportion of marked fish seen during a survey); we attempted to conduct a complete survey of the study area within two days of a tagging event so that the number of tags lost to mortality and emigration would be minimized. To estimate survey life, for each tagging group, we attempted to complete as many additional surveys as possible, given the constraints of river conditions and work schedules. Ideally, surveys would be repeated every 3-4 days following a tagging event, but this was not always possible. Perrin and Irvine (1990) describe several methods for estimating survey life from tagging data, two of which are applicable to this study. Both methods underestimate survey life when tagged fish are present in the study area for any length of time prior to tagging. With the first method, numbers of tagged fish from an individual tagging event that are observed on subsequent surveys are plotted against time to produce a tag depletion curve, and survey life is estimated as the area-under-the-tag-depletion curve divided by the total number of tags applied. In the second method, individually numbered tags are recovered from carcasses, and survey life is computed as the average number of days between fish tagging and carcass recovery. We estimated survey life using the area-under-the-tag-depletion curve. Fish tagging efforts during 2006-2008 suggested that the second method was not feasible in the Coquitlam River because once they die, tagged fish are quickly flushed out of the study area, and only a negligible number of tagged carcasses are recovered (a carcass fence would likely be necessary to apply this method).

In addition to causing missed surveys, variable flows and turbidity in the Coquitlam River during the salmon spawning period likely resulted in substantial variability in observer efficiency between surveys within a year, and in some cases, between years as well (see Section 2.2.2). Substantial variation in water visibility (and hence observer efficiency) among index sites during individual surveys was also common. This source of error is important because variation in observer efficiency among years that is unaccounted for will bias comparisons of adult abundance and egg-to-smolt survival among years and therefore, between flow treatments. To address this, during 2002-2018, the survey crew developed a relative index of survey conditions by subjectively ‘guesstimating’ observer efficiency (0%-100%) for each index site during all surveys. It is not feasible to estimate observer efficiency using mark-recapture during each survey. Instead, the guesstimate is used to estimate observer efficiency for a survey based on the predictive relationship between surveyor guesstimates and mark-recapture derived estimates of observer efficiency (see Section 2.1.3.2).

Beginning in 2007, the survey crew also began collecting quantitative water visibility data. To index water visibility for each survey, a 1.5 m wading staff that was clearly marked at 5cm intervals, was placed in the water column, and the depth at which the tip of the staff was no longer visible was recorded. Measurements were taken at permanently marked locations in each index site. However, based on mark-recapture experiments completed to date, estimates of water visibility have proven to be a less useful predictor of variation in observer efficiency compared to surveyor guesstimates (Decker *et al*. 2012).

2.1.3 Escapement model structure and parameter estimation

The escapement model consists of two main elements: i) a simple process model predicts the number of fish present on each day of the run and the departure schedule based on the total escapement and parametric relationships simulating arrival timing and survey life, and ii) an observation model simulates the number of fish counted on each survey based on the predicted numbers present and detection probabilities.

2.1.3.1 Process model

To estimate total escapement from repeat count data, the proportion of the total run present on each survey day must be determined. This can be calculated by estimating run timing parameters that describe the cumulative proportion that has arrived and departed for each model day, which forms the process model. In the description that follows, note that lower case Arabic letters denote either model array indices (subscripts) or data. Upper case Arabic letters denote state variables (variables predicted by the model), and Greek letters denote variables that are estimated (parameters).

The proportion of the total escapement entering the survey area on day ‘t’ (PA_t) of the run is predicted by a beta distribution, where α and β are parameters of the beta distribution and p_t represents the proportional day of the run. The total number of model days for Chum, Pink, Coho, and Chinook were 119 (September 3-December 30), 82 (September 1-November 21), 130 (September 20-January 27), and 99 (September 3-December 10), respectively.

$$PA_t = p_t^{\alpha-1}(1-p_t)^{\beta-1} \qquad (2.2)$$

The beta distribution is reparametrized so that β is calculated based on estimates of the day when the peak arrival rate occurs (μ) and the variance (standard deviation) in the proportion of the run arriving over time (σ), using the transformations:

$$\alpha = \mu * \frac{1}{\sigma^2}$$
$$\beta = (1-\mu) * \frac{1}{\sigma^2} \tag{2.3}$$

For Pacific salmon, survey life, which is the number of days a fish spends in the survey area, is normally longer for fish that arrive earlier in the spawning period (Perrin and Irvine 1990; Su *et al*. 2001). Survey life was modeled such that it varied with the day of entry into the spawning area using a decaying exponential relationship,

$$SL_t = \lambda_c e^{-\lambda_s t} \tag{2.4}$$

where, SL_t is the survey life for a fish entering on day t, λ_c is the maximum survey life, and λ_s is the slope of the relationship. The day that a fish arriving on day t has exceeded its survey life is simply $D_t = t + SL_t$, and the proportion of the run that has departed on day t is,

$$PD_t = \sum_t PA_t \mid t = D_t \tag{2.5}$$

The total number of fish present in the survey area on each day (N_t) is the product of the total escapement (E) and the proportion present on any survey day, estimated as the difference between the cumulative arrivals and departures on that day.

$$N_t = E\left[\int_1^t PA - \int_1^t PD\right] \tag{2.6}$$

2.1.3.2 Observation model

Escapement (E) and arrival timing parameters (μ, σ), and those defining the observation process are jointly estimated by assuming that the count data arise from an over-dispersed Poisson distribution which accounts for the extra variation associated with the non-random distribution of fish on any survey (i.e., clumping),

$$n_t \sim Poisson(N_t \theta_t e^{\varepsilon_t}) \tag{2.7}$$

where, n_t is the total number of fish counted on day t, θ_t is an estimate of the survey-specific detection probability, and ε_t is a survey-specific deviate used to model over-dispersion in the data (McCarthy 2007; Royle and Dorazio 2008). ε_t is drawn from a normal distribution with a mean of 0 and a precision $\tau.o$ (i.e., ε_t~dnorm(0, $\tau.o$), where $\sigma.o=\tau.o^{-0.5}$). The term "~" denotes that the value to the left of the term is a random variable sampled from the probability distribution defined on the right. This equation is often referred to as the likelihood component of the model because it describes the likelihood of the data, given the parameter values. Note that n_t will be greater than the total fish counted across sites surveyed on day t if the entire survey area was not surveyed. In this case, an adjustment is required to account for areas that were not surveyed (see data description above).

Survey-specific detection probability is predicted based on the relationship between detection probability and estimated detection probability developed from mark-recapture data,

$$\gamma_i = \frac{e^{\beta_0+\beta_1 * v_i}}{1+e^{\beta_0+\beta_1 * v_i}} \tag{2.8}$$

where γ_i is the predicted detection probability for mark-recapture experiment i, and β_0 and β_1 are the constant and slope of a linear relationship predicting γ_i as a function of the estimated detection probability from visual methods for that experiment (v_i), respectively. We assume that the number of marks detected on these experiments is a binomially-distributed random variable,

$$r_i \sim dbin(\gamma_i, m_i) \tag{2.9}$$

where r_i and m_i are the number of marks detected and the total marks released for each experiment. Given estimates of β_0 and β_1, it is then possible to predict survey-specific detection probabilities (θ_t) from equation 2.8 given a visual estimate of detection probability on each survey (v_t).

The escapement model is implemented in a Bayesian framework and therefore requires that prior probability distributions are specified for all estimated parameters. We used uninformative priors in all cases,

$$\begin{aligned}
&E \sim \text{dnorm}(2000,1.0\text{E-6})\ \text{I}(0,) &(2.10)\\
&\mu \sim \text{dunif}(0,1)\\
&\sigma \sim \text{dunif}(0,10)\\
&\tau.o \sim \text{dgamma}(5,5)\\
&\beta_0 \sim \text{dnorm}(0,1.0\text{E-6})\\
&\beta_1 \sim \text{dnorm}(0,1.0\text{E-6})
\end{aligned}$$

where dnorm, dunif, and dgamma refer to normal, uniform, and gamma distributions respectively. The first and second terms in dnorm represent the mean and precision, respectively. The I(0,) term associated with the prior for escapement indicates that the normal distribution is truncated at 0 as negative escapement values are not possible. The first and second values for the uniform distributions represent the minimum and maximum values, respectively. The first and second values in the gamma distribution represent the shape and scale parameters, respectively. Values of 5 were used in each case so that model fit, as assessed by Bayesian *p*-values (see below), was adequate.

2.1.3.3 Parameter Estimation and assessing model fit

Posterior probability distributions of model parameters were estimated using a Monte Carlo Markov Chain (MCMC) algorithm as implemented in WinBUGS (Spiegelhalter *et al*. 1999). We called WinBUGS from the R2WinBUGS (Sturtz *et al*. 2005) library from R (R Development Core Team 2009). We used the mean of the posterior to represent the expected value for the parameter, and the ratio of the standard deviation of the posterior to the mean as a measure of relative parameter uncertainty. The 95% credible intervals were determined from the lower 2.5 and upper 97.5 percentiles of the posterior distribution. Posterior distributions were based on a total of 2,000 MCMC samples. These samples were obtained by drawing every 2nd sample from a total of 5,000 simulations after excluding the first 1,000 samples to remove the effects of initial values. This strategy was sufficient to achieve convergence in all cases. Model convergence was evaluated by visually inspecting the MCMC chains for evidence of non-stationarity and poor mixing.

We used posterior predicted *p*-values, often called Bayesian *p*-values, to statistically evaluate the fit of the models (Gelman *et al*. 2004). The concept behind this statistic is that data simulated from the model will resemble the real data if and only if the model fits the data well (Brooks *et al*. 2000; Gelman *et al*. 2004). Bayesian *p*-values are similar to the statistic generated from classical goodness-of-fit tests, but are based on multiple measures of discrepancy determined from the posterior distribution of predictions, rather than the single best-fit prediction determined by maximum likelihood estimation in the latter case. Bayesian *p*-values are computed by replicating a data set based on the model predictions for each MCMC trial. Measures of discrepancy between the replicated data and model predictions (D`), and observed data and model predictions (D), are then compared. The fraction of MCMC trials were D`>D is the Bayesian *p*-value. Low *p*-values indicate the model under-fits the data, that is, there is too much scatter around the curve describing the number of fish observed over the run, either because the run-timing model is not flexible enough (under-parameterized) and/or does not explain enough of the variability in the data given the assumed error model. High *p*-values indicate that the model over-fits the data, that is, the model explains more variation than expected, either because the run-timing model is too flexible or because the assumed error structure is too complex. Bayesian *p*-values of approximately 0.5 indicate an ideal fit. We used the Freeman-Tukey statistic as the measure of discrepancy as recommended by Brooks *et al*. (2000) for the analysis of mark-recapture models. This measure assigns less weight to outcomes with small expected counts (similar to Pearson's χ^2), and provides more robust assessments of model fit when outcomes are close to zero as is sometimes the case with count data.

2.2 Results and Discussion

2.2.1 Survey conditions and run timing

Unadjusted survey counts from all surveys during 2002-2016 and 2018-2019 are shown for Chum, Coho, Pink and Chinook in Appendices 2.3-2.6. The typical period of peak spawning was the last week of October for Chum, the second week of December for Coho, and the last half of October for Chinook. The reliability of estimates depends on surveys encompassing the entire migration but particularly peak conditions, as was the case for all four species in 2019 (Figure 2.3). In other years, run curve peaks for Chum, Coho and Chinook were poorly defined as a result of missed or partial surveys during high water events (see interim data reports of individual years for more details; Decker and Macnair 2009; Macnair 2004, 2005, 2006). In 2003 and 2005, the run timing curve was poorly defined for Pink Salmon because substantial numbers of Pinks were already present in the spawning area at the time of the first survey, and survey data was sparse in the latter half of the spawning period on account of high flows (Appendix 2.3). For Chum and Chinook (with the exception of 2007 for Chinook), the beginning, peak and end of the spawning period was generally well defined each year, other than in 2012 when surveys missed the peak (Appendices 2.4, 2.6). The beginning of the spawning period was well defined for Coho, but in some years of the study (2002, 2004, 2005, 2011 and 2018); significant numbers of Coho were still present during the final survey in late January (Appendix 2.5). For modeling purposes, the maximum length of the spawning period for Chum, Pink, Coho, and Chinook was 119 (September 3-December 30), 82 (September 1-November 21), 130 (September 20-January 27), and 99 (September 3-December 10), days respectively.

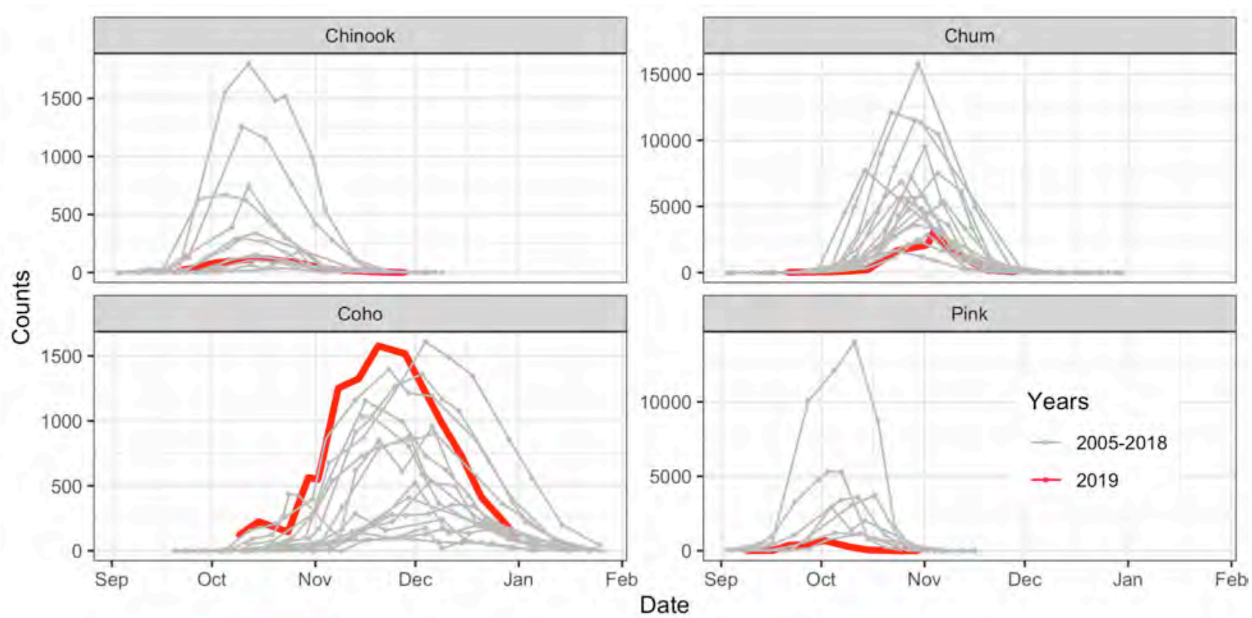

Figure 2.3 Spawning run timing base on survey counts for Chinook, Chum, Coho and Pink Salmon in the Coquitlam River during 2018 (red line) and 2002-2016 (grey).

Survey conditions during the 2019 varied considerably. Water column visibility ranged from 0.7m to more than 3m across the surveys with generally lower visibility as the survey season progressed (Table 2.1). This corresponded to a general trend of decreasing observer efficiency 'guestimates' as the season progressed (Table 2.1). Even with a high frequency of storm events in the 2019 survey season, all but three surveys occurred at or near base flows and with a discharge of less than 10 m^3/s (Figure 2.4). This increased the likelihood that survey conditions are sufficient in all index sites, which improves the reliability of survey counts.

2. Adult Salmon Escapement

Table 2.1 Water column visibility (m) at permanent measurement points at index sites A-E and surveyor 'guesstimates' of observer efficiency for Chum salmon (see Section 2.1.2) during surveys of the Coquitlam River for the 2019 brood year.

	Estimated water column visibility (m)						
Escapement Year	Date	site A	site B	site C	site D	site E	non-index
2019	09-Sep	>3	>3	>3	>3	>3	
2019	17-Sep	1.2	1.2	1.2	1.2	1.4	
2019	21-Sep	1.2	1.2	1.2	1.2	1.3	
2019	27-Sep	1.1	1.1	1.1	1.2	>3	1.2
2019	02-Oct	1.0	1.1	1.1	1.2	1.2	
2019	09-Oct	1.1	1.2	1.1	1.1	1.3	
2019	15-Oct	1.1	1.2	1.2	1.2	1.2	1.2
2019	24-Oct	0.9	0.9	1.0	1.1	1.2	
2019	30-Oct	1.0	1.0	0.9	1.1	1.1	1.0
2019	02-Nov	0.9	0.9	1.0	1.1	1.1	1.0
2019	04-Nov	0.9	0.9	1.0	1.1	1.1	1.0
2019	08-Nov	0.9	0.9	1.0	1.1	1.1	1.0
2019	14-Nov	0.9	0.9	1.0	1.2	1.2	
2019	20-Nov	0.8	0.8	0.9	1.1	1.1	
2019	28-Nov	0.8	0.8	0.8	1.1	1.1	
2019	09-Dec		0.7	0.8	1.1	1.1	
2019	15-Dec		0.7	0.8	1.1	1.0	
2019	21-Dec		0.6	0.7	1.1	1.0	
2019	27-Dec		0.7	0.7	0.9	1.0	
2019	31-Dec		0.7	0.7	0.9	1.1	

Surveyor "guesstimates" of observer efficiency (0.0-1.0): (chum salmon example)							
Escapement Year	Date	site A	site B	site C	site D	site E	non-index
2019	09-Sep	0.70	0.7	0.70	0.75	0.80	
2019	17-Sep	0.60	0.65	0.65	0.65	0.75	
2019	21-Sep	0.60	0.65	0.65	0.65	0.75	
2019	27-Sep	0.60	0.65	0.60	0.65	0.75	0.70
2019	02-Oct	0.60	0.60	0.65	0.65	0.70	
2019	09-Oct	0.60	0.60	0.65	0.65	0.70	
2019	15-Oct	0.60	0.60	0.65	0.65	0.70	0.65
2019	24-Oct	0.50	0.50	0.60	0.60	0.70	
2019	30-Oct	0.60	0.60	0.65	0.65	0.70	0.65
2019	02-Nov	0.60	0.60	0.65	0.65	0.70	0.65
2019	04-Nov	0.60	0.60	0.65	0.65	0.70	0.65
2019	08-Nov	0.60	0.60	0.60	0.65	0.70	0.65
2019	14-Nov	0.60	0.60	0.60	0.60	0.60	
2019	20-Nov	0.50	0.60	0.60	0.60	0.60	
2019	28-Nov	0.55	0.60	0.60	0.60	0.60	
2019	09-Dec	0.40	0.50	0.50	0.60	0.65	
2019	15-Dec		0.50	0.50	0.60	0.65	
2019	21-Dec		0.40	0.40	0.50	0.60	
2019	27-Dec		0.50	0.50	0.60	0.60	
2019	31-Dec		0.50	0.50	0.60	0.60	

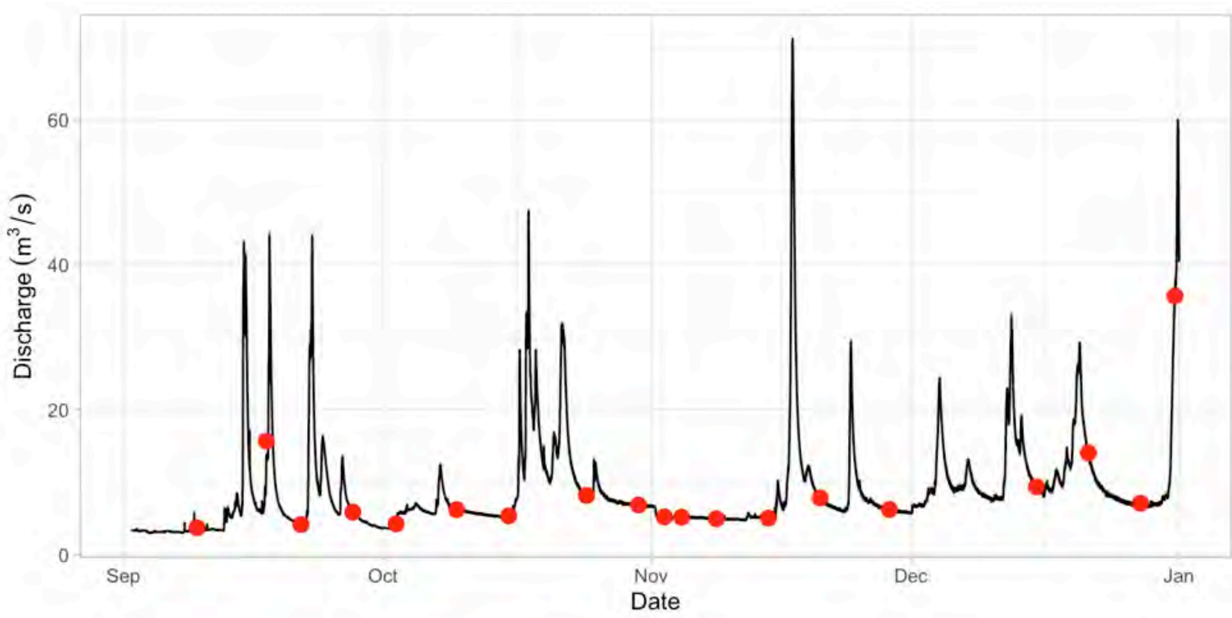

Figure 2.4 Daily flows in Coquitlam River at Port Coquitlam during the fall and winter spawning period in 2019-2020 (Water Survey of Canada, stn. 08MH141). Survey days are represented by red dots.

2.2.2 Observer efficiency and survey life

2.2.2.1 Observer efficiency

During 2006-2019, 24 estimates of observer efficiency were obtained for all species combined. One observer efficiency estimate was obtained during 2019. In some of the 24 cases, the field crew was unable to capture and mark adequate numbers of fish to provide reliable estimates of observer efficiency, while in other cases, salmon were tagged, but no estimates of observer efficiency were obtained because poor visibility conditions prevented a complete survey from being conducting within two days (Appendix 2.2). The opportunity exists in future monitoring years to collect additional mark-recapture data under Treatment 2. This is not possible for Treatment 1; estimates of observer efficiency under Treatment 1 (across all years) are limited to four years for Chum, one for Pink and none for Coho and Chinook (Appendix 2.2).

Table 2.2 Averages and absolute ranges for observer efficiency estimates (proportion of live salmon present that are visually detected) derived from mark-recapture experiments, and subjective 'guesstimates' of observer efficiency made by the survey crew for the same surveys during which the mark-recapture experiments occurred (see Section 2.1.2).

	Chum	Pink	Coho	Chinook	All species
Mark-recapture-derived estimates of observer efficiency					
Number of estimates	11	7	3	2	23
mean	0.55	0.66	0.70	0.60	0.60
minimum	0.33	0.49	0.67	0.53	0.33
maximum	0.69	0.85	0.73	0.67	0.85
Surveyor guesstimates of observer efficiency					
mean	0.75	0.82	0.71	0.77	0.76
minimum	0.55	0.55	0.60	0.75	0.55
maximum	0.90	1.00	0.85	0.78	1.00
Survey life (days)					
Number of estimates	6	5	3	2	16
mean of estimates	8.4	10.7	16.4	7.7	
range of estimates	6.5 - 9.9	6.8 - 15.5	11.6 - 15.2	7.7 - 8.5	
maximum survey life for individual fish	16	20	28	25	

Observer efficiency for Chum averaged 48% across 10 mark-recapture experiments during 2006-2019 (range: 33%-69%; Table 2.2); with similar means for Treatment 1 and Treatment 2 (50% and 48%, respectively; Appendix 2.2). For Pink, seven mark-recapture experiments yielded an average observer efficiency estimate of 66% (range: 49%-85%; Table 2.2, Appendix 2.2). For Coho, three mark-recapture experiments under Treatment 2 provided average observer efficiency estimates of 70%. The value for Coho is relatively high compared to observer efficiency estimates reported for Coho in other streams (Irvine *et al*. 1992). The addition of an underwater observer to the survey crew, beginning in 2009 (see Section 2.1.1), was presumably a contributing factor. In the absence of an underwater observer, observer efficiency during Treatment 1 for Coho in the Coquitlam River was likely lower than the Treatment 2 average of 70%; and was likely lower than the mean value of 47% for Chum, which spawn earlier in the season, and are less associated with cover and deep pools. For Chinook, two mark-recapture experiments under Treatment 2 provided average observer efficiency estimates of 60%. Due to the small sample size, we combined Coho and Chinook observer efficiency estimates for generating Coho population estimates. Also, given the absence or near absence of data, the observer efficiency during Treatment 1 can only be approximated for Coho, Chinook and Pink (see Section 2.2.2.3).

Mark-recapture experiments completed to date have been limited to the early or middle (peak) portions of the spawning period for each species, with no tagging events occurring after November 1 for any species except Coho. For Chum and Coho, which spawn later in the fall when poor survey conditions occur more frequently, observer efficiency estimates likely represent the upper range, rather than average values for the Coquitlam River. This is because the same poor river conditions that lead to low observer efficiency also make it difficult to capture fish for the mark-recapture experiments. Without this information, we were unable to confirm observer efficiency through the full range of survey conditions. We consider this source of uncertainty one of the primary reasons the HBM is unable to estimate the precision of escapement estimates. It is important that future efforts be made to conduct mark-recapture experiments as late in the season as possible, and during periods of higher flows and lower visibility, so that the actual range in observer efficiencies can be better represented in the escapement model.

The issue of poor spatial distributions of marked populations of Chum, Pink and Chinook has improved since 2007 when marking occurred in only one location which provided little information about observer efficiency in the remainder of the survey area. During 2006-2019, 13 of 23 mark-recapture experiments included marking at two different sections of the Coquitlam River. This provided more spatially representative estimates of observer efficiency, but rendered the data less reliable for assessing survey life because fish captured in the upper river were less likely to be new arrivals to the study area (see below). See Decker *et al.* 2010 for the rational for distributing marking sites throughout the entire survey area.

In some cases, marked populations of Chum, Pink and Chinook were skewed to males, and were likely unrepresentative of the sex ratios of the population as a whole (Appendix 2.2). Bias in the sex ratio of marked populations can result in a bias in observer efficiency and survey life, if these parameters differ for male and female spawners (Perrin and Irvine 1990).

2.2.2.2 Survey life

Mark-recapture data for 2006-2019 provided limited information about survey life (number of days fish are in the survey area) for each species. Obtaining estimates of survey life requires conducting multiple consecutive surveys (minimum of three) of the entire study area every few days following a tagging event, and this was frequently impossible due to unsuitable survey conditions. A total of 16 estimates of survey life were obtained, seven for Chum, three for Coho, six for Pink and two for Chinook (Table 2.2). Based on the area-under-the-tag-depletion curve method (Perrin and Irvine 1990), mean survey life values ranged from 7.5 days for Chum, to 16.4 days for Coho and decrease as the date they enter the study area increases (Figure 2.5). Based on estimates from the Coquitlam River, the maximum survey life (maximum number of days between when a fish was tagged and subsequently detected) ranged from 16 days for Chum to 28 days for Coho (Table 2.2; Appendix 2.2). Survey life estimates for salmon in the Coquitlam River were less than mean values reported for the same species in other streams, but were still within the reported range (see next section), suggesting that survey life is relatively short in the Coquitlam River. However, survey life estimates for the Coquitlam River are biased low to some degree because salmon were present in the study area for an unknown period of time prior to being captured (as opposed to being captured while migrating past a weir). This bias was further exacerbated by the fact that during many of the mark-recapture experiments, fish were captured and tagged in the spawning areas at the upstream index sites in order to better distribute tags for the purpose of estimating observer efficiency (see above). Additionally, in order to compute estimates of survey life it was necessary to assume that observer efficiency remained constant across a series of surveys following a tagging event. Yet, in several of the mark-recapture experiments, the number of tagged fish detected actually increased from one survey to the next, indicating that observer efficiency had increased over time, rather than remaining constant. While it is apparent this assumption isn't satisfied when observer efficiency increases substantially, survey life bias would also occur with a drop in observer efficiency, but would go undetected due to the inability to distinguish between the effects of observer efficiency and survey life under these conditions.

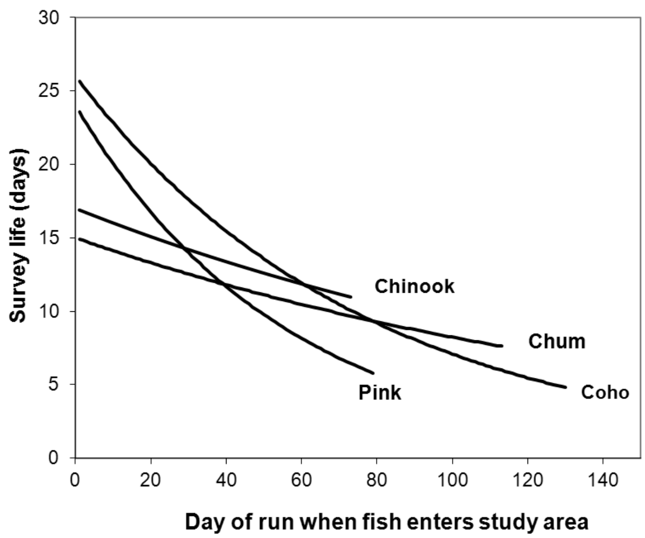

Figure 2.5 Modeled relationship between mean survey life and day of arrival in the study area for Chum, Pink, Coho, and Chinook salmon in the Coquitlam River based on empirical data from other streams.

Similar to observer efficiency, survey life experiments were skewed towards the early- to mid-spawning period. For Pacific salmon, survey life tends to be greater for early-run fish compared with those spawning later in the season (Perrin and Irvine, 1990). Without more survey life experiments later in the spawning season, we weren't able to confirm the magnitude of difference between early and late run fish.

With a biased estimate of survey life, escapement estimates can still provide a reliable index of adult abundance as long as the actual survey life was relatively consistent across years. An inaccurate estimate of survey life will bias escapement estimates to a similar degree, yet the relative change between years could still be accurate.

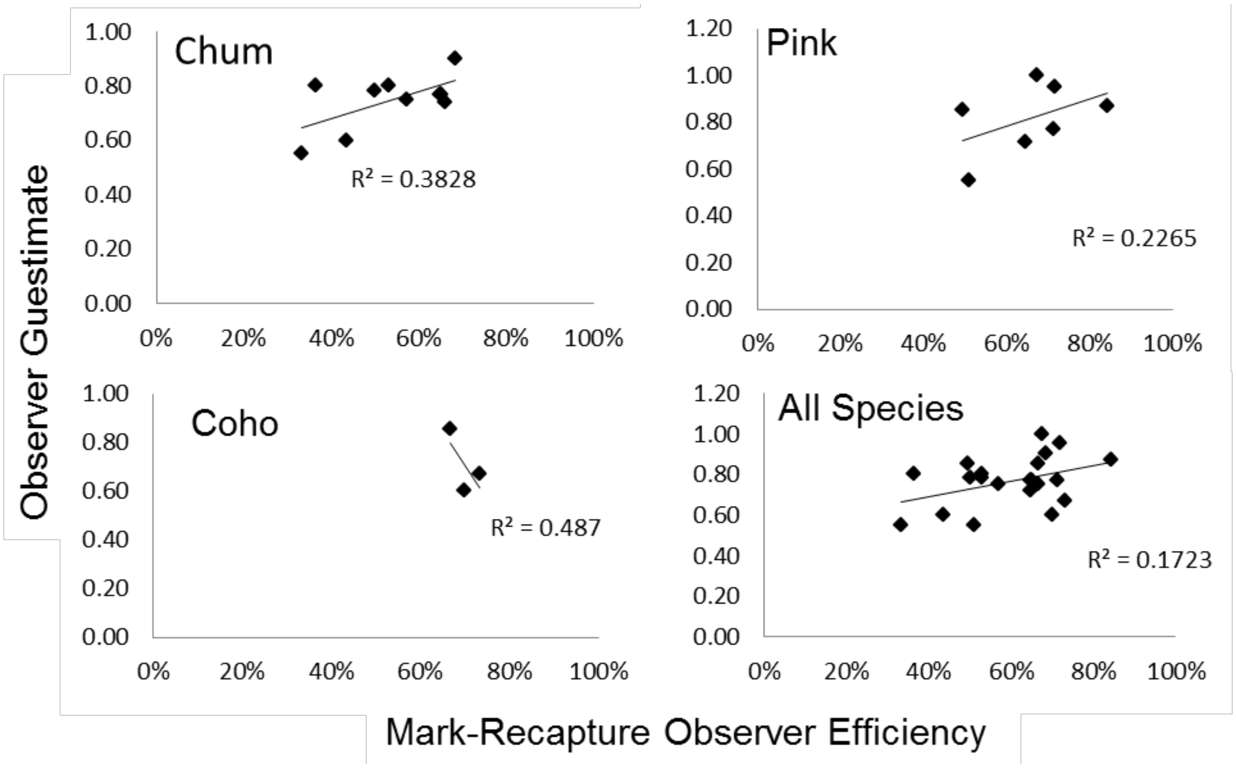

Figure 2.6 Relationship between the surveyor's 'guesstimate' of observer efficiency and observer efficiency estimated from mark recapture experiments for Chum, Pink and all species combined conducted opportunistically since 2006 in the Coquitlam River.

2.2.2.3 Modeling observer efficiency and survey life

For Chum, subjective guesstimates of observer efficiency made by the survey crew for surveys where mark-recapture estimates of observer efficiency were available ranged from 55%-100%, and averaged 76% (Table 2.2). When compared to mark-recapture estimates of observer efficiency, the surveyor guesstimates were biased high, but were moderately useful predictors of observer efficiency for Chum (linear regression, n=10; R^2=0.38; Figure 2.6). For Pinks, surveyor guesstimates ranged from 55% to 100% for five surveys for which mark-recapture data were available (Table 2.2). Surveyor guesstimates explained less than one quarter of the variation in mark-recapture derived estimates of observer efficiency among surveys for Pinks (n=7; R^2=0.23; Figure 2.6). However, this relationship is highly uncertain, being based on only seven observations. For Chum and Pink, we used the regression relationships in the escapement model to estimate observer efficiency for individual surveys based on the guesstimates of observer efficiency, and to model error in estimated observer efficiency (see equations 2.8 and 2.9). For Coho, surveyor guesstimates over the three surveys ranged from 67% to 73%, which is a poor reflection of common survey conditions (range for all years, 0.45-1.0). Unfortunately, the guesstimates were negatively related to the mark-recapture data (linear regression, n=3; R^2=0.49; Figure 2.6), which increases uncertainty in the population estimation model. For Chinook, there were only two mark-recapture estimates of observer efficiency available (Table 2.2), which provides no useful information about the relationship between surveyor guesstimates and actual observer efficiency, or even what the average observer efficiency in the Coquitlam River might be. In light of the poor relationship for Coho and limited information for Chinook, in order to model observer efficiency, we regressed surveyor guesstimates against mark-recapture derived estimates of observer efficiency using pooled data for all four species (*n*=22; R^2=0.17; Figure 2.6). Mean observer efficiency (based on mark-recapture) across all species was 60% compared to observed means of 76% and 60% for Coho and Chinook, respectively (Table 2.2). Furthermore, existing information is too sparse and variable to evaluate whether an assumption for pooling (similar relationship between mark-recapture observer efficiency and guestimates for pooled species) is satisfied.

Given the limited and uncertain survey life data for all four salmon species in the Coquitlam River, we relied on reported values from other studies to inform the parameterization of survey life in the escapement model. Perrin and Irvine (1990) summarized studies of survey life for Pacific salmon. They reported that for Chum, average survey life was 12 days (range= 4-21 days, n=54), and varied little between early and late portions of the spawning period (14 and 11 days, respectively). Average survey life for Pink was 17 days overall, and 24 and 15 days respectively, for the early and late portions of the spawning period (range =5-41 days, n=36). Average survey life for Coho was 11 days (range =3-15 days, n=22), with no information available about differences between early and late portions of the spawning period. Average survey life for Chinook was 12 days (range = 3-20 days, n=38). We adjusted the λ_c and λ_s parameter values for each species to provide early, mean, and late period survey life values for Chum (12, 10, and 9 days, respectively); Pink (18, 13, and 9 days); Coho (18, 12, and 8 days); and Chinook (15, 12, and 11 days) (Figure 2.3). To model error in survey life, the coefficient of variation for survey life (λ_v) was set at 0.65 for each species, based on an intensive study of survey life of Pink salmon by Su *et al*. 2001.

2.2.3 Escapement model

We evaluated the performance of the escapement model using data for each species and for different years within each species that provided contrasts in the amount of information available about run timing and the shape of the run timing curve. We found that, in general, it was not possible to obtain realistic estimates of uncertainty in escapement (i.e., 95% credible intervals), while at the same time obtaining plausible mean estimates of escapement and run timing (i.e., predicted run timing curves that provide a good fit to the observed counts; see Appendices 2.7a-d). If the priors that control the extent of overdispersion were set to allow for sufficient overdispersion in the data, as indicated by Bayesian *P* values of ~0.5 (see Section 2.1.3.3), then the predicted escapement tended to be unrealistically low. Conversely, if the priors were adjusted to reduce the extent of overdispersion, the model provided a good fit to the count data, but the Bayesian *p*-values were too low (indicating that the error distribution was wrong and the 95% credible interval for the escapement estimate was unrealistically narrow). The underlying problem with the model is that there is no independent data to estimate the extent of overdispersion. A similar model to ours worked very well to estimate the uncertainty in estimates of adult bull trout abundance in the Cheakamus River (Ladell *et al*. 2010), but in that study radiotelemetry data provided much better information about observer efficiency and run timing than was available for salmon in the Coquitlam River. As a result, the model was able to estimate the extent of overdispersion in escapement estimates in the absence of the confounding effect of uncertainty in the other parameters. Given the model-fitting problems described for the Coquitlam data and the very limited amount of observer efficiency and survey life information collected to date, we concluded that the best approach at this point would be to use a version of the model that assumed no overdispersion in the data, and to compute point estimates of escapement only, without attempting to estimate uncertainty in these estimates.

The "no overdispersion" version of the escapement model provided good fits of predicted numbers of fish present (i.e., predicted run timing survey) to observed weekly counts of Chum, Pink, Coho and Chinook, allowing for plausible estimates of escapement and run timing. An example of model fit for 2012 data for Coho is provided in Appendix 2.8. However, because true error in the count data exceeded that assumed by a regular Poisson distribution (as opposed to an overdispersed Poisson distribution; see Section 2.1.3.2), 95% credible intervals for escapement estimates were unrealistically low, and were therefore not reported.

2.2.4 Escapement estimates

Estimates of escapement for all species in all years are summarized in Table 2.3. Among years, estimated escapements ranged from 7,000-78,000 for Chum; 900-14,000 for Coho; 1,200-34,000 for Pink; and 123-8,000 for Chinook. It is important to note that escapement is an insensitive measure for comparing fisheries benefits of Treatment 1 and 2 flows owing to the large role of ocean survival (particularly its' variability) on the number of adult returns. Trends reported here are products of freshwater and marine conditions. For all species, escapement has been much higher during Treatment 2 than during Treatment 1 (Table 2.4). Mean escapement has increased four-fold for Pink and Chinook, three-fold for Coho and two-fold for Chum compared with Treatment 1. While the mean escapement for Pink and Chum have been higher during Treatment 2, the escapement estimates for 2019 were the lowest and second lowest, respectively, since surveys began in 2002 (Table 2.4). Escapement estimates for Coho and Chinook during Treatment 1 years should be treated as approximations and are likely not comparable with Treatment 2 (See section 2.2.2.1). Estimates reflected here for Coho and Chinook during Treatment 2 years may also be biased low if the limited species specific mark-recapture information collected to date more accurately reflects observer efficiency than the pooled mark-recapture data for all species, which we used to increase the sample size, and thus precision of escapement estimates (see Section 2.2.2.3).

Escapement estimates for 2002-2019 may differ to some degree in future reports from those reported in Table 2.3, as observer efficiency and survey life is re-estimated with additional trials. Escapement estimates are highly sensitive to estimates of observer efficiency and survey life (e.g., a decrease in estimated observer efficiency from 0.8 to 0.4 results is a doubling of the escapement estimate). Ideally, enough mark-recapture experiments should be conducted during future years to provide reliable estimates of survey life and observer efficiency specific to each of the four salmon species in the Coquitlam River, at least for Treatment 2. We were unable to say how many mark-recapture experiments will be necessary to achieve this since survey life data is too sparse to assess the uncertainty of escapement estimates.

Table 2.3 Annual escapement estimates for Chum, Pink, Coho and Chinook salmon for the years 2002-2016 and 2018-2019. Also included are the mean abundance estimates during Treatment 1 (2002-2008) and Treatment 2 (2009-2019).

Year	Treatment	Chum	Pink	Coho	Chinook
2002	1	15,378	-	2,648	-
2003	1	18,301	5,418	1,562	-
2004	1	27,992	-	2,562	-
2005	1	24,559	4,279	1,334	-
2006	1	51,860	-	939	-
2007	1	11,066	2,944	2,401	360
2008	1	18,224		878	952
2009	2	19,600	10,698	3,175	1,529
2010	2	6,931	-	12,338	8,018
2011	2	27,410	10,427	8,414	4,918
2012	2	57,300	-	11,320	1,632
2013	2	42,220	34,280	13,290	2,413
2014	2	8,491	-	4,957	572
2015	2	23,410	9,327	4,979	123
2016	2	78,120	-	6,867	511
2018	2	26,490	-	13,910	456
2019	2	10,970	1,211	11,670	591
Treatment 1 Mean		23,911	4,214	1,761	656
Treatment 2 Mean		30,094	13,189	9,092	2,076

2.2.6 Adult habitat distribution and access to off-channel sites

Chum salmon in particular show a preference for mainstem spawning habitat in the Coquitlam River (Table 2.4). This preference has been noted in many studies for Chum salmon in medium-sized rivers (Salo, 1991). For Chum, there was a reduction of approximately 10% in the proportion of Chum spawning in mainstem habitats following the initiation of the Treatment 2 flow regime in 2008 (Treatment : 82%-90%, Treatment 2: 69%-81%; Table 2.4). It is unclear whether this was an artifact of reduced observer efficiency in the mainstem when flows had increased after October 22; whether it was due to the increased availability of off-channel habitats; or a combination of both. Higher mainstem flows under Treatment 2 gave salmon easier access to off-channel habitats, and increased the amount of available spawning habitat in some constructed off-channel sites, mainstem areas and natural side-channels. In addition, adult Chum show a preference for spawning in the lower reaches of the Coquitlam River, (an average of 63% of adult Chum spawning in index sites A-C during 2002-2018; Table 2.5). Chum salmon have a brief freshwater residency and often spawn exclusively in the lower reaches of river systems (Salo 1991). Spawning gravels are also more abundant in the lower reaches of the Coquitlam River.

During Treatment 1 and 2, Pinks made greater use of mainstem sites for spawning than off-channel habitats (Treatment 1: 55%-71%, Treatment: 259-76%; Table 2.4). Pink Salmon also have a brief freshwater residency period, but unlike Chum, Pink spawners make greater use of spawning areas in upper reaches of the Coquitlam River. Depending on the year, the proportion of Pink spawning in the two uppermost sites (D and E) ranged from 44%-72% (Table 2.5).

Table 2.4 Adult spawning distribution by habitat type during Treatment 1 and 2, and the 2008 transition year. Treatment 2 flows were initiated October 22, 2008. Proportions shown are calculated based on counts of actively spawning fish only, during surveys when all five index sites were completed. M/S = mainstem, NOC = natural off-channel, OCR = off-channel restoration site, and OC = off-channel sites combined.

		Treatment 1						Transition	Treatment 2										
Species	Habitat	2003	2004	2005	2006	2007	mean	2008	2009	2010	2011	2012	2013	2014	2015	2016	2018	2019	mean
Chum	M/S	0.87	0.82	0.87	0.90	0.84	0.86	0.76	0.77	0.73	0.77	0.69	0.78	0.79	0.81	0.75	0.79	0.81	0.76
	NOC	0.06	0.06	0.07	0.06	0.10	0.07	0.12	0.13	0.16	0.18	0.20	0.13	0.11	0.11	0.15	0.14	0.12	0.15
	OCR	0.08	0.12	0.06	0.04	0.06	0.07	0.12	0.11	0.11	0.05	0.10	0.09	0.10	0.08	0.10	0.07	0.08	0.09
	OC	0.13	0.18	0.13	0.10	0.16	0.14	0.24	0.23	0.27	0.23	0.31	0.22	0.21	0.19	0.25	0.21	0.19	0.24
Pink	M/S	0.55		0.65		0.71	0.64		0.76		0.59		0.77		0.74			0.68	0.72
	NOC	0.19		0.22		0.20	0.20		0.12		0.22		0.12		0.15			0.12	0.15
	OCR	0.26		0.13		0.09	0.16		0.12		0.19		0.11		0.11			0.20	0.13
	OC	0.45		0.35		0.29	0.36		0.24		0.41		0.23		0.26			0.32	0.28
Coho	M/S	0.32	0.27	0.46	0.80	0.70	0.51	0.74	0.84	0.87	0.91	0.90	0.84	0.92	0.94	0.90	0.86	0.92	0.89
	NOC	0.15	0.38	0.33	0.07	0.11	0.21	0.04	0.03	0.01	0.01	0.01	0.02	0.00	0.03	0.01	0.01	0.02	0.02
	OCR	0.53	0.35	0.21	0.13	0.19	0.28	0.22	0.13	0.12	0.08	0.09	0.14	0.08	0.03	0.09	0.14	0.08	0.10
	OC	0.68	0.73	0.51	0.20	0.29	0.48	0.26	0.16	0.13	0.09	0.10	0.16	0.08	0.06	0.10	0.15	0.10	0.11
Chinook	M/S	0.82	0.71	0.84	0.99	0.87	0.85	0.77	0.94	0.94	0.98	0.88	0.98	0.99	0.96	0.93	0.97	0.95	0.95
	NOC	0.06	0.02	0.07	0.01	0.04	0.04	0.02	0.02	0.02	0.01	0.05	0.01	0.00	0.01	0.01	0.01	0.01	0.02
	OCR	0.12	0.26	0.10	0.00	0.09	0.11	0.21	0.04	0.04	0.01	0.07	0.01	0.01	0.03	0.06	0.03	0.05	0.03
	OCR	0.18	0.29	0.16	0.01	0.13	0.15	0.23	0.06	0.06	0.02	0.12	0.02	0.01	0.04	0.07	0.03	0.05	0.05

The trend for Coho of low natural or enhanced off-channel habitat use during Treatment 2 continued in 2019 with usage of 10% (Table 2.4). The combined natural and enhanced off-channel habitat use dropped from 20%-73% during 2002-2007 to 6%-16% during 2009-2018 (Table 2.4). This shift commenced prior to Treatment 2 and coincided with the modifications to the Coquitlam Dam and dewatering of the Grant's Tomb off-channel site in 2005, which accounted for the majority of off-channel use. The change in relative use between mainstem and off-channel habitat may reflect the higher carrying capacity of the mainstem as well as changes in off-channel habitat capacity. In terms of the number of Coho, the maximum off-channel use has remained relevantly consistent across treatment periods but varied considerably across years (Figure 2.7). With the several fold higher escapement levels from 2010 onward, use increased disproportionately in the mainstem habitat. Coho salmon showed a preference for the upper reaches of the Coquitlam River (sites D and E accounted for 59%-99% of Coho spawning during 2002-2018; Table 2.5).

Evidence of movement barriers for spawning adults was not apparent at any time during Treatment 1. Fish arriving during the late summer low flow period (which in 2006 lasted until the middle of October), were observed at all index sites. However, observations by the survey crew suggest that low flows did impede access to natural and enhanced off-channel sites in some instances. For example, during a low flow period in 2005, Pink did not enter off-channel sites until October 2, and in 2006, Chum avoided off-channel sites until October 13. Delayed migration into off-channel areas was not apparent during 2008 when flows were increased under Treatment 2. Under Treatment 2, all index sites continue to be accessible to spawning adults throughout the entire escapement period.

Table 2.5 Estimated average proportion of Chum, Pink, Coho and Chinook salmon spawning populations present at each index site (A-E) and at non-index (NI) sites during 2002-2016 and 2018-2019.

Species	Site	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2018	2019
Chum	A	0.25	0.52	0.39	0.48	0.33	0.45	0.35	0.32	0.27	0.66	0.29	0.42	0.55	0.59	0.36	0.44	0.53
	B	0.04	0.07	0.07	0.05	0.07	0.09	0.07	0.05	0.03	0.05	0.06	0.06	0.06	0.06	0.11	0.04	0.07
	C	0.27	0.19	0.18	0.26	0.11	0.14	0.13	0.19	0.09	0.09	0.16	0.11	0.06	0.09	0.15	0.05	0.18
	D	0.30	0.15	0.23	0.14	0.25	0.22	0.26	0.26	0.31	0.11	0.32	0.24	0.17	0.15	0.21	0.25	0.04
	E	0.07	0.01	0.02	0.03	0.08	0.03	0.11	0.12	0.18	0.04	0.08	0.10	0.11	0.04	0.10	0.13	0.16
	NI	0.07	0.06	0.11	0.04	0.15	0.07	0.08	0.06	0.12	0.06	0.09	0.07	0.05	0.07	0.08	0.08	0.03
Pink	A	-	0.16	-	0.17	-	0.21	-	0.13	-	0.17	-	0.21	-	0.30	-	-	0.09
	B	-	0.10	-	0.05	-	0.03	-	0.06	-	0.02	-	0.05	-	0.03	-	-	0.00
	C	-	0.20	-	0.11	-	0.08	-	0.12	-	0.06	-	0.13	-	0.12	-	-	0.06
	D	-	0.21	-	0.20	-	0.24	-	0.25	-	0.19	-	0.22	-	0.17	-	-	0.05
	E	-	0.24	-	0.42	-	0.33	-	0.36	-	0.53	-	0.32	-	0.29	-	-	0.69
	NI	-	0.10	-	0.05	-	0.11	-	0.08	-	0.04	-	0.07	-	0.08	-	-	0.10
Coho	A	0.00	<0.001	0.02	0.09	<0.01	0.01	0.05	<0.01	<0.01	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.01
	B	0.01	0.06	0.03	0.02	0.04	0.07	0.03	0.02	0.04	0.02	0.03	0.02	0.03	0.03	0.02	0.00	0.02
	C	0.00	0.04	0.06	0.28	0.24	0.04	0.08	0.10	0.11	0.11	0.10	0.08	0.09	0.06	0.06	0.03	0.08
	D	0.19	0.20	0.20	0.36	0.43	0.32	0.18	0.14	0.21	0.30	0.28	0.17	0.23	0.24	0.22	0.04	0.20
	E	0.80	0.70	0.63	0.23	0.24	0.56	0.65	0.74	0.57	0.52	0.54	0.66	0.55	0.55	0.68	0.85	0.65
	NI	<0.01	<0.01	0.06	0.02	0.05	<0.001	0.01	<0.01	0.07	0.05	0.05	0.06	0.09	0.12	0.02	0.07	0.04
Chinook	A						0.02	0.02	0.06	0.04	0.03	0.02	0.03	0.01	0.01	0.02	0.00	0.01
	B						0.00	0.01	0.03	0.01	0.02	0.00	0.02	0.00	0.00	0.00	0.01	0.02
	C						0.10	0.05	0.08	0.07	0.07	0.00	0.07	0.02	0.02	0.01	0.05	0.08
	D						0.06	0.11	0.09	0.18	0.22	0.08	0.22	0.10	0.10	0.04	0.12	0.21
	E						0.64	0.76	0.70	0.60	0.61	0.84	0.61	0.86	0.86	0.90	0.80	0.68
	NI						0.18	0.05	0.04	0.10	0.06	0.06	0.06	0.00	0.00	0.03	0.02	0.00

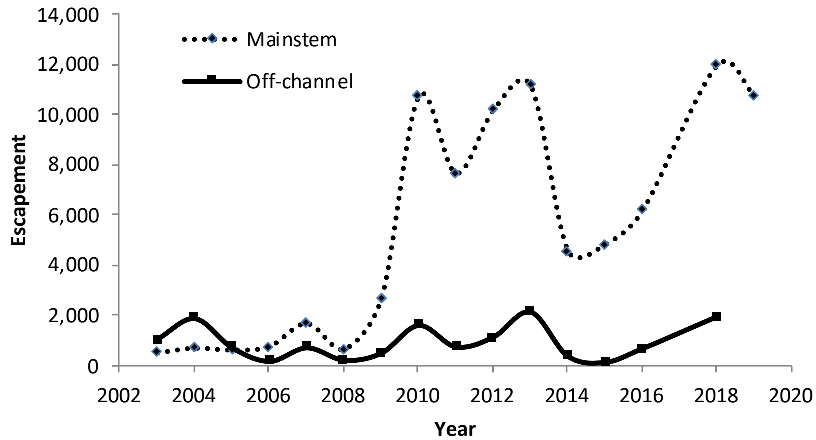

Figure 2.7 Estimated numbers of Coho spawning in mainstem and side-channel habitat in the Coquitlam River 2003-2019. Note that habitat type was not recorded during the 2002 surveys.

2.2.7 Temperature

Optimal spawning temperatures range from 4.4-9.4°C for Coho, to 4.4-10°C for Chum, to 7.2-12.8°C for Pink (McCullough 1999). Optimal temperatures during the incubation stage range from between 4.4 and 14°C for all species (McCullough 1999). Stream temperatures in the Coquitlam River have typically fallen within the optimal range for Chum and Coho during their mid-October – late November and November – January spawning periods, respectively. The same was true for Pink spawning in late September-late October, with the exception of higher than optimal temperatures in reach 4 during most of September 2009 (18-20°C).

2.3 Implication for hypothesis testing

Adult escapement monitoring is providing sufficient information to evaluate the fisheries benefits of Treatment 1 and 2 for Coho but not for Pink or Chum. For Coho, the evaluation of flow treatments depends primarily on smolt production estimates, given that the stock-recruitment relationship to date suggest smolt production is limited by rearing habitat. In this situation escapement estimates only serve the purpose of confirming that escapement was sufficient to reach the juvenile carrying capacity (see Figure 6.1). Beyond this minimum value (~800 fish), smolt production appears insensitive to escapement. Furthermore, we do not recommend using Coho escapement for any between-treatment comparisons since survey methods differed between Treatment 1 and 2, and all Coho mark-recapture experiments occurred during Treatment 2. Given the switch to replacing one of the foot-surveyors with a snorkeler during Treatment 2, it is likely observer efficiency was higher than during Treatment 1. Using the same observer efficiency for Treatment 1 and 2 could lead to underestimates for Treatment 1.

For Chum and Pink, our inability to calculate the precision for escapement estimates reduces the reliability of inferences drawn from this data. Unlike for Coho, the Chum and Pink stock-recruitment relationships to date point to escapement-limited fry production (Figure 6.6), which depends on both juvenile and adult estimates to detect treatment effects. This type of regression based analysis assumes that the adult abundance (x axis) is without error, or at least of much less error than juvenile abundance (Zar 1999). The Coquitlam River data is far from satisfying this assumption. While stock-recruitment analyses rarely satisfy this assumption, knowing the precision of our estimates would allow us to exclude very imprecise estimates, which may increase our ability to detect differences between treatments. Using mean counts as an index of escapement can corroborate HBM results when surveys include peak counts but are still subject to error with sparse data. As mentioned in Section 2.2.3, we think that our inability to calculate credible, precise estimates stems from the lack of Coquitlam-specific survey life information and/or insufficient mark-recapture data.

2. Adult Salmon Escapement

3 Adult Steelhead Escapement

3.1 Methods

During 2005-2020, we conducted periodic redd surveys to assess the cumulative number of redds constructed during the spawning period. To convert redd counts to indices of adult winter Steelhead abundance and potential egg deposition, we used empirical data from studies of winter Steelhead in other coastal streams to approximate the number of redds constructed by each female, the average sex ratio, and mean fecundity per female (see section 3.1.4). The variation in redd counts among observers was not investigated, but was minimized by having the same crew conduct all surveys. Steelhead redds become increasingly difficult to detect over time as their characteristic features become obscured by algal growth and substrate movement during high flows. In a small number of years it was necessary to use redd survey life data (i.e., the period of time following initial construction during which a redd can be positively identified) to adjust redd counts upwards to account for redds that we failed to detect due to survey intervals exceeding redd survey life (see Section 3.1.3).

3.1.1 Description of study area and survey methods

For Steelhead redds, the study area extended approximately 10.8 km from the Coquitlam Dam downstream to the Patricia Footbridge, and included reaches 2a, 2b, 3, and 4 (Figures 3.1, 3.2). Reach 1 was omitted as minimal Steelhead spawning occurs there. During 2005-2006, it was found that the majority of Steelhead spawning occurred during a seven-week period (mid-March – early May). The analysis of previous years' data indicated that all redds remained visible with a seven day survey interval and that only a very small proportion went undetected with a 14 day survey interval (see Section 3.1.3). As such, our target survey frequency was every two weeks for the entire spawning period. We attempted to conduct surveys just prior to high flow events in order to minimize the number of new redds becoming obscured by substrate movement before they could be detected. Owing to the length of the study area, each survey was completed over a two-day period.

Redd surveys were conducted by two trained technicians who were familiar with Steelhead spawning locations in the Coquitlam River and had considerable experience identifying Steelhead redds. During each survey, one crewmember wore a dry suit and snorkeling gear and searched for redds in deep water, while the other wore chest waders and searched for redds in shallower water along the banks. The shoreline observer marked the location of each redd detected by either crew member with numbered flags and a global positioning system (GPS) to prevent double counting on future surveys, and to provide estimates of redd survey life (see section 3.1.3). Additional data collected for each redd included width and length, specific location within the stream channel, and average substrate size. The crew also recorded the number of live adults observed on each survey, along with their location, and, if possible, their approximate fork length, sex, and the absence of an adipose fin (indicating hatchery origin). The observation data of live adults collected during redd counts was not used to estimate escapement.

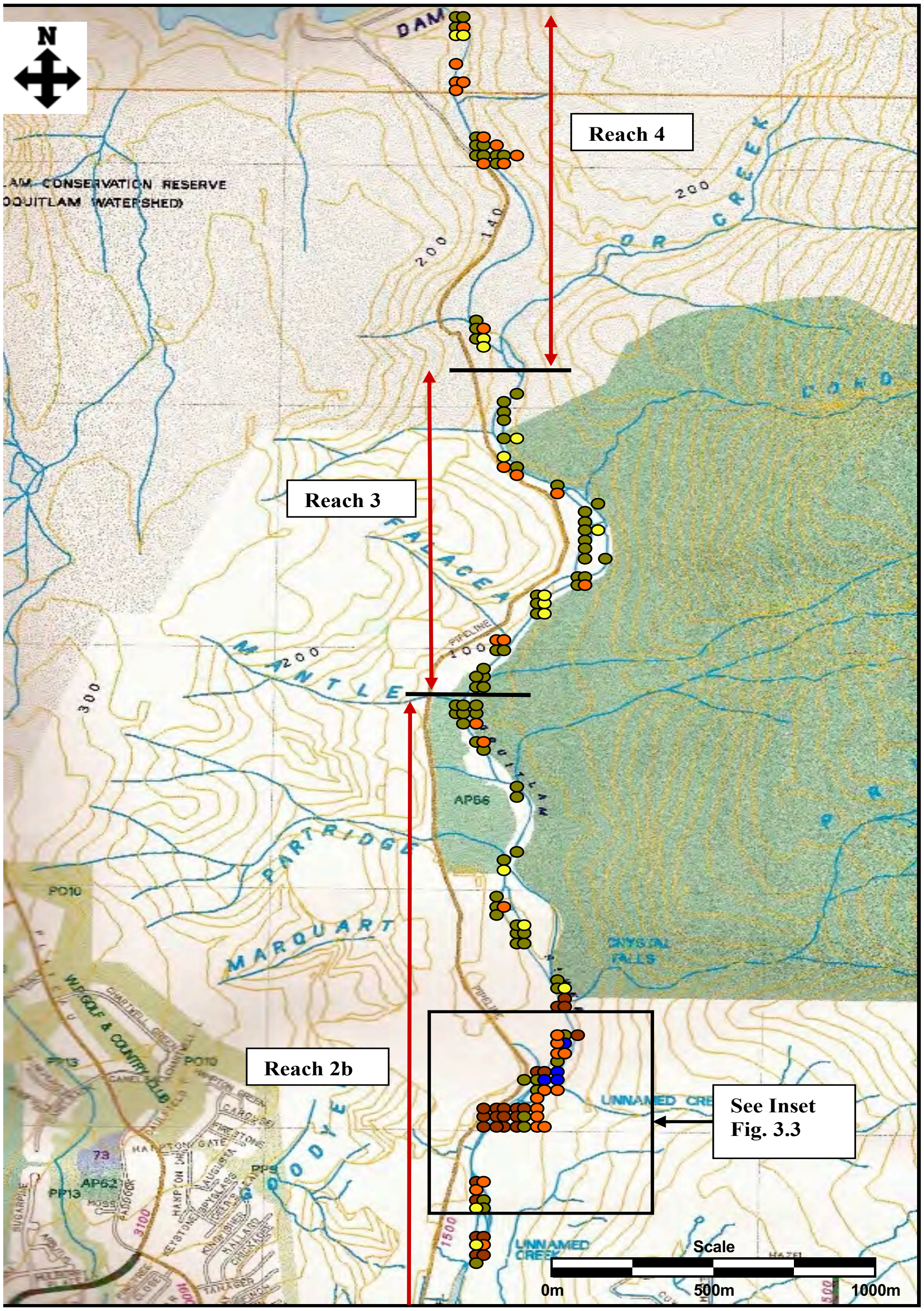

Figure 3.1 Steelhead redd locations in reaches 2b-4 in Coquitlam River in 2006, which was the highest escapement year during 2005-2020. Coquitlam Dam is the upstream boundary of the survey area. See Figure 3.2 for redd symbol legend.

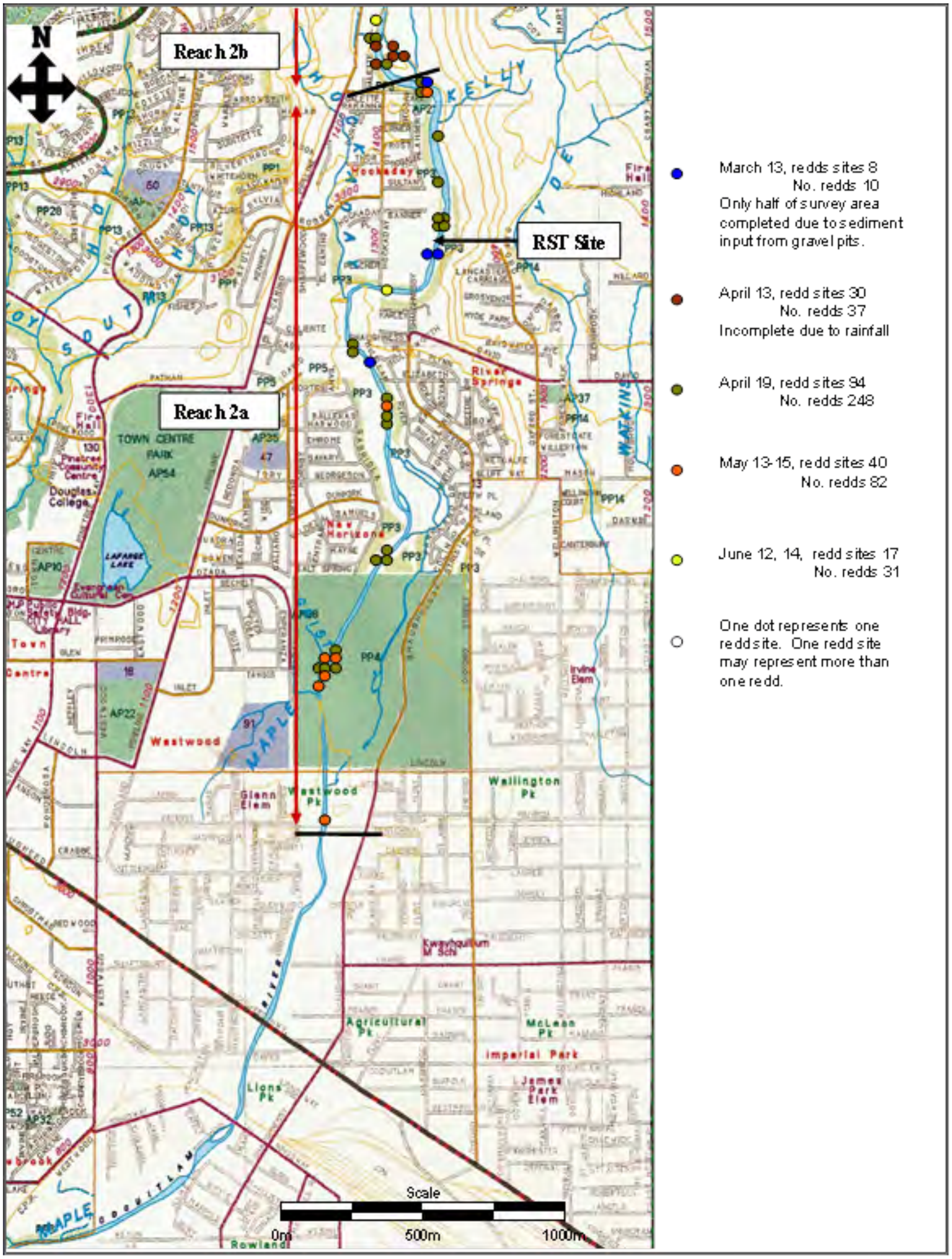

Figure 3.2 Steelhead redd locations in reaches 2a-2b in Coquitlam River in 2006. The downstream boundary of reach 2a is also the survey area boundary.

3. Adult Steelhead Escapement

3.1.2 Redd Identification

Redds were identified as approximately dish-shaped excavations in the bed material, often of brighter appearance than surrounding substrates, accompanied by a deposit beginning in the excavated pit and spilling out of it in a downstream direction. Disturbances in the bed material caused by fish were discriminated from natural scour by:

i) the presence of tail stroke marks;
ii) an over-steepened (as opposed to smooth) pit wall often accompanied by perched substrate that could be easily dislodged down into the pit, and often demarcated by sand deposited in the velocity break caused by the front wall;
iii) excavation marks alongside the front portion of the deposit demarcating the pit associated with earlier egg laying events, and
iv) a highly characteristic overall shape that included a 'backstop' of gravel deposited onto the unexcavated substrates, a deposit made up of gravels continuous with this backstop and continuing upstream into the pit, and a pit typically broader than the deposit and of a circular shape resulting from the sweeping of gravels from all sides to cover the eggs (in a portion of redds gravels are swept into the pit from only one side, often a shallow gravel bar on the shore side).

A second important determination was whether fish had actually spawned at a location where an excavation had been started. 'Test digs' were considered to be pits, often small, accompanied by substrate mounded up on the unexcavated bed material downstream but with no accompanying gravel mound downstream of the pit, which would denote at least one egg deposition event. In the case of a test dig determination, the mound of gravels would typically be short and narrow around the downstream side of a relatively small pit. Potential test digs were tagged and re-examined on subsequent surveys to determine if they had been further developed into actual redds.

Redds constructed by resident cutthroat or rainbow trout, or lamprey were distinguished from Steelhead redds by their considerably smaller size, lack of a large deposit downstream of the pit, and a conical, or bowl shape, rather than a rectangular shape.

In areas of limited gravel or high redd abundance, or where spawning site selection is highly specific, superimposition of redds can occur (Baxter and McPhail 1996). Owing to the relatively high survey frequency (see below), undercounting of Steelhead redds as a result of redd superimposition is not likely to be a substantial source of negative bias in estimates of Steelhead spawner abundance in the Coquitlam River. Redds are usually detected and their locations recorded before new redds are superimposed. In cases where we do encounter superimposed redds, we count redds based on a subjective evaluation, with the most recent complete redd(s) counted and the disturbed remains of prior redds being estimated in relation to it. A greatly extended deposit length (subjectively evaluated to be at least twice the length of a 'typical' deposit length) constitutes grounds to consider whether a second female had made use of the pit created by a first to construct a separate redd.

3.1.3 Redd survey life

In most cases, Steelhead redds can be readily detected upon initial construction, but over time, they become undetectable as they are obscured by scour or deposition, regrowth of periphyton, or superimposition of new redds. Thus, survey frequency is an important consideration in designing redd surveys, particularly for streams like the Coquitlam River, where moderately high flow events can occur during the Steelhead spawning period. If the length of time between surveys exceeds average redd survey life, then undercounting will occur. Freymond and Foley (1985) reported winter Steelhead redds remaining easily identifiable for a period of 14 to 30 days in coastal Washington State streams. Based on five years' of data from several coastal Oregon State streams, Jacobs *et al.* (2002) concluded that, on average, 95% of winter Steelhead redds remain visible one week after completion, while 86% remain visible after two weeks.

Across all years, our target of bi-weekly surveys, had we met them, would likely have resulted in negligible undercounting of redds due to survey intervals exceeding average redd survey life. In 2020, with intervals of 15 days or less, we estimated the number of redds simply as the sum of new redds (x_i) counted during n surveys (Equation 3.1). Only if the case where the interval exceeds 15 days did we use the redd life model to estimate the number of missed redds. See Decker *et al.* 2010 for a description of methods used to estimate redd survey life and how this is used to estimate the number of redds not visible when survey intervals exceed 2 weeks. Numbered flags were used to identify new redds (or groups of redds) during each survey. The visibility of previously flagged redds was evaluated during each survey to further refine the redd survey life model.

$$\boldsymbol{N} = \left(\sum_{i=1}^{n} \boldsymbol{x_i}\right) \qquad (3.1)$$

3.1.4 Female escapement and egg deposition

The objective of the Steelhead redd survey component is to enable smolt production to be related to spawning effort. Redd numbers are a direct measure of spawning effort and egg deposition. So, for the purposes of this study, we considered that estimating the total number of redds is as useful as estimating the total adult escapement. However, the number of recruits per spawner is commonly expressed as the number of smolts per female. Following this convention, we converted our estimates of total redd abundance to total female abundance by relying on empirical estimates of the average number of redds per female for winter Steelhead in Pacific coastal streams. Gallagher and Gallagher (2005) reported redds/female values for winter Steelhead in several streams, but their estimates were based on mark-recapture and AUC estimates that were themselves highly imprecise. Freeman and Foley (1985) reported the average number of redds per adult in Snow Creek, Washington State, but not the average number per female. The most robust estimate we were able to obtain was from a study of winter Steelhead in Oregon State coastal streams by Jacobs *et al*. (2002) that compared total redd counts to accurate estimates of female escapement for four streams over three years using total counts at full-span upstream fences, or at upstream fences coupled with intensive mark-recapture methodology. The number of redds per female derived by Jacobs *et al.* (2002) ranged from 0.75

to 1.63 and averaged 1.2, with relatively little variability among years for individual streams. We used this value (1.2 redds/female) to convert total redd numbers to female escapement.

The total number of adult female Steelhead in the surveyed portion of Coquitlam River (N) was approximated as:

$$\boldsymbol{N = \left(\sum_{i=1}^{n} x_i\right) \div 1.2} \tag{3.2}$$

Where x_i is the cumulative number of new redds summed across n surveys and 1.2 is a constant representing the number of redds per individual female spawner. In the absence of fecundity data for Coquitlam River Steelhead, we substituted average fecundity for winter Steelhead (3,700 eggs/female) in the Keogh River on northern Vancouver Island (Ward and Slaney 1993). We assumed 50% of adult Steelhead in the Coquitlam River were female, which is commonly reported for Pacific coastal winter Steelhead (Jacobs *et al*. 2002). To reflect the uncertainty in the Steelhead escapement estimates arising from uncertainty about the average number of redds per female and sex ratio, the possible minimum and maximum range in escapement in any given year was approximated by arbitrarily varying redds/female by 1.0-2.0, and the proportion of females in the population using sex ratios from five other winter Steelhead streams (0.42-0.63; Jacobs *et al*. 2000, 2002).

3.2 Results and Discussion

During 2005-2020, the dates of the first and last redd survey ranged from February 15 to March 24, and from June 5 to June 13, respectively (Table 3.1). During all years except 2007, the periods when discharge exceeded the 20 m^3/s limit for ideal surveys were relatively infrequent and typically lasted less than two days, based on flows at Port Coquitlam (WSC 08MH002, Appendix 3.1). In contrast, during 2007, mean daily flows remained above 20 m^3/s for 17 days during March. Surveys during Treatment 1 were completed at lower flows (2-10 m^3/s, Table 3.1) than during Treatment 2 (2-26 m^3/s). This reflects the higher minimum flows during Treatment 2 as a result of increased releases from the LLO during the March-May spawning period. In 2020, surveys were completed at flows that were well below the 20 m^3/s upper limit for ideal surveys (Figure 3.2).

Poor stream visibility conditions, which occurred at low as well as high flows, limited the frequency of surveys in all years (see Section 3.2.1). The goal of conducting bi-weekly surveys during the major portion of the spawning period was met for all 8 surveys in 2020; across the entire survey period, the length of time between surveys ranged from 9 - 15 days, and averaged 12 days (Appendix 3.2,). In previous years, the length of time between surveys has ranged from 6 to 37 days (Appendix 3.2).

Table 3.1 The start and end date for Steelhead surveys, and the mean, minimum and maximum discharge (m^3/s) in Coquitlam River at Port Coquitlam during surveys for the years 2005-2020 (Water Survey of Canada station 08MH002).

	Date		Discharge (m^3/s)		
Year	Start	End	Mean	Minimum	Maximum
2005	24-Mar	05-Jun	4.9	2.7	9.6
2006	15-Feb	12-Jun	3.5	3.1	4.7
2007	02-Mar	13-Jun	4.6	3.1	5.9
2008	21-Mar	13-Jun	4.2	2.0	7.9
2009	11-Mar	08-Jun	7.6	6.1	8.3
2010	09-Mar	14-Jun	6.1	4.4	7.4
2011	22-Mar	06-Jun	8.3	5.2	10.6
2012	08-Mar	07-Jun	7.0	5.5	8.4
2013	10-Mar	08-Jun	6.8	4.0	10.3
2014	15-Mar	08-Jun	6.3	2.2	13.5
2015	16-Mar	04-Jun	4.9	2.1	9.1
2016	17-Mar	08-Jun	6.7	3.7	10.8
2017	20-Mar	07-Jun	9.3	7.3	12.4
2018	11-Mar	07-Jun	13.2	4.0	26.5
2019	14-Mar	09-Jun	5.6	1.8	7.9
2020	15-Mar	09-Jun	8.5	5.9	12.2

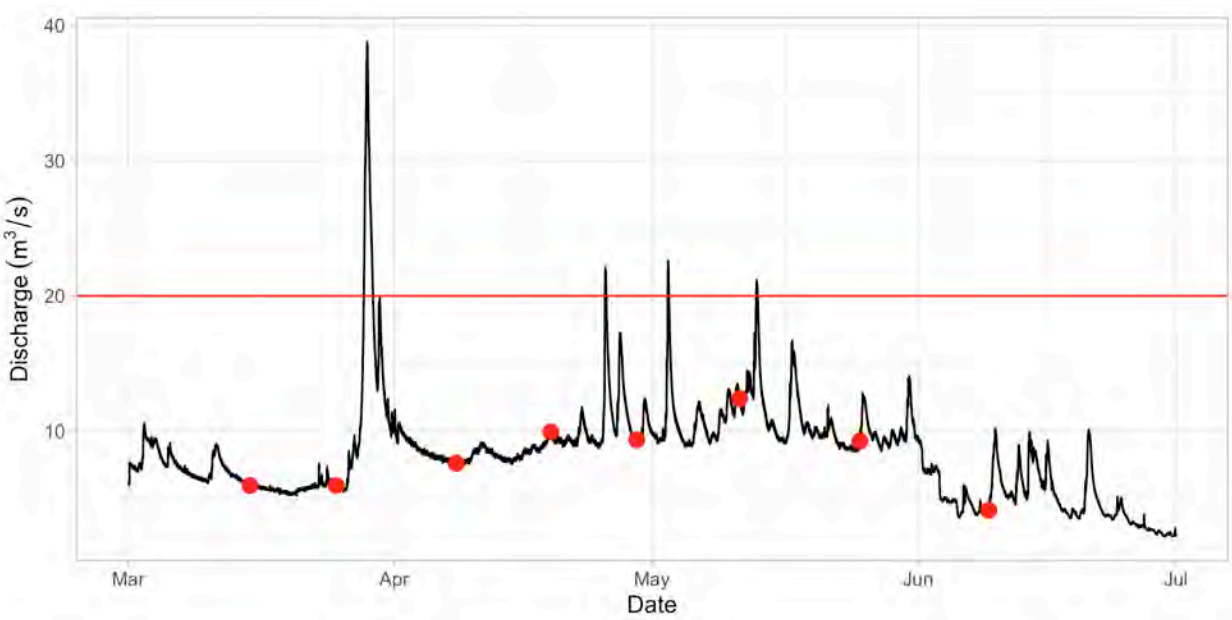

Figure 3.3 Discharge (m^3/s) in Coquitlam River at Port Coquitlam during Steelhead spawning period in 2020 (Water Survey of Canada station 08MH002). The red line represents the upper flow limit for ideal survey conditions. The red dots represent survey dates.

3. Adult Steelhead Escapement

3.2.1 Escapement and egg deposition

In 2020, a total of 131 novel redds were counted (Table 3.2). This represents a total escapement of 218 Steelhead (Table 3.2), which was a lower than average value for the 2005-2019 period (mean = 401; respectively). The highest and lowest escapements occurred in 2006 (868 adults; Table 3.2) and 2009 (225 adults), respectively. Average Steelhead redd density in the study area of the Coquitlam River was 12.1 redds/km in 2020, and ranged from 13-48 redds/km during 2005-2019 (Appendix 3.2). Among reaches and years, redd density ranged from 6-71 redds/km (Table 3.3). Spawning distribution during 2020 was lower in the lower reaches of the river (2a and 2b, 35%) and higher in the upper reaches (3 and 4, 65%, Table 3.3), which is atypical to the trend of equal or more spawning in the lower reaches for past years. Figures 3.1 and 3.2 illustrate the fine-scale distribution of redds in the study area for 2006.

Table 3.2 Annual estimates of the total number of redds, redd/km, and the estimated number and density of female spawners, eggs, total escapement based on 1.2 females/redd, 3700 eggs/female and 2 spawners/redd. Peak counts represent the highest count of adults per year.

Year	Total number of redds	Redds /km	Total female spawners	Total egg deposition	Eggs /km	Total adult escapement	Peak count
2005	224	20.7	187	691,000	64,000	373	22
2006	521	48.2	434	1,606,000	149,000	868	95
2007	156	14.4	130	481,000	45,000	260	45
2008	178	16.5	148	549,000	51,000	297	44
2009	135	12.5	113	416,000	39,000	225	37
2010	200	18.5	167	617,000	57,000	333	60
2011	247	22.9	206	762,000	71,000	412	103
2012	337	31.2	281	1,039,000	96,000	562	148
2013	297	27.5	248	916,000	85,000	495	113
2014	190	17.6	158	586,000	54,000	317	88
2015	301	27.9	251	928,000	86,000	502	117
2016	246	22.8	205	759,000	70,000	410	90
2017	239	22.1	199	737,000	68,000	398	81
2018	161	14.9	134	496,000	46,000	268	61
2019	175	16.2	146	540,000	50,000	292	69
2020	131	12.1	109	404,000	37,000	218	49

Table 3.3 Proportion of redds in reaches 2-4 of the Coquitlam River since redd surveys began in 2005.

	Reach			
Year	2a	2b	3	4
2005	13%	34%	28%	25%
2006	14%	41%	22%	23%
2007	16%	41%	35%	8%
2008	24%	47%	23%	6%
2009	22%	40%	26%	12%
2010	16%	36%	33%	16%
2011	17%	23%	34%	26%
2012	18%	30%	30%	21%
2013	8%	31%	31%	30%
2014	16%	32%	28%	25%
2015	12%	34%	23%	31%
2016	12%	35%	30%	23%
2017	11%	41%	29%	19%
2018	7%	32%	32%	28%
2019	19%	29%	30%	22%
2020	7%	28%	31%	34%

The principal sources of uncertainty in deriving Steelhead escapement estimates for the Coquitlam River from redd counts are the sex ratio and the average number of redds constructed by each female; error in escapement estimates will be directly proportional to error in either of these parameters. We used average values of 1:1 for sex ratio, and 1.2 redds per female (Jacobs *et al*. 2002) to develop escapement estimates. Jacobs *et al*. (2002) reported a 2-fold variation in the number of redds per female among streams, but noted relatively little variation among years within individual streams. Jacobs *et al*. (2002) also observed fairly consistent sex ratios of 1:1. For the purposes of indexing Steelhead escapement in the Coquitlam River during Treatment 1 and 2, this is encouraging, since the accuracy of the estimates is of secondary importance as long as the sex ratio and redds/female ratios remained constant between treatments.

There was no indication that the higher flows during 2020 led to redds becoming obscured by scour. This would have been apparent if individually marked redds were not detected in subsequent surveys. In 2020, all redds were confirmed during at least one survey after they were initially observed.

In 2020, there was a substantial number of redd counts at the beginning of the survey period and relatively few on the last survey. Ideally, counts on the first and last survey would be relatively low to indicate surveys spanned the entire survey period. The moderate count on the first survey suggests the assumption that surveys spanned the early spawning period was not clearly met and that spawning started earlier than past years. The relatively high count of live adults on the first survey (33 adults) compared to the peak count (49) also suggests an earlier start to spawning (Table 3.4). However, it is unlikely that the start of spawning was early enough that redds were obscured to the point they were undetected, given that all redds counted on the first survey were seen again on a later survey. This suggests spawning still started within two to three weeks of the survey start date. If spawning started prior to this, we would expect some redds to 'fade' and go undetected on the second survey. Across all years, no or minimal spawning (<5% of annual total) was observed by surveys prior to March 14. Excluding 2018 and 2020, 9% - 18% of new redds were constructed by the end of March (Appendix 3.3). In 2018, there was increased spawning during March (37% of redds formed prior to March 31). The March spawning activity for 2020 falls between 2018 and the other years (24% of redds formed prior to March 31). The results for 2005 to 2020 suggest that Steelhead typically begin spawning in the Coquitlam River in early March but, with the exception of 2018 and 2020 that most spawning occurs (80-90%) during a six-week period spanning early April to mid-May (Figure 3.4).

Table 3.4 Survey dates with raw counts of Steelhead redds, estimated new redds, and live adult counts for all surveys during 2020. Estimated new redds includes the sum of the raw count and the estimated number of redds that were constructed and then obscured by substrate movement prior to a scheduled survey, based on a redd survey life model. Appendix 3.3 lists this for all years.

Year	Survey date	Days since previous survey	Raw count of new redds	Estimated # new redds	# Live adults observed
2020	15-Mar	0	21	21	33
2020	25-Mar	10	11	11	40
2020	8-Apr	14	21	21	49
2020	19-Apr	11	41	41	44
2020	29-Apr	10	19	19	19
2020	11-May	12	9	9	15
2020	25-May	14	7	7	13
2020	9-Jun	15	2	2	0

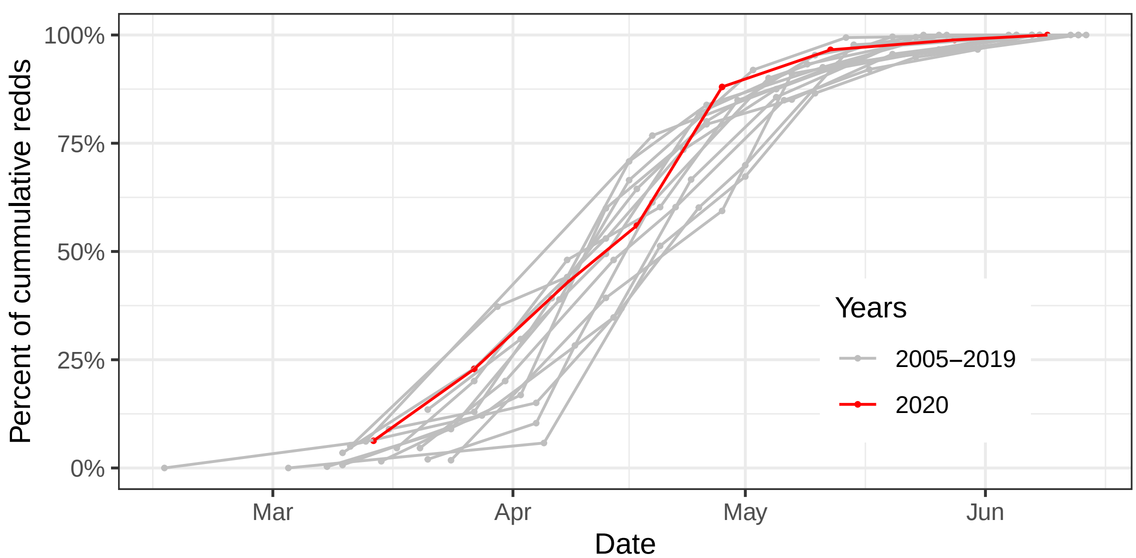

Figure 3.4 Cumulative proportion of the total Steelhead redd count observed over time during 2020 (red) and 2005 - 2019 (grey).

3. Adult Steelhead Escapement

Spawning Steelhead preferred mainstem habitat as compared to natural side channel and constructed off-channel habitat by a large margin during all survey years (83%-99%). In 2020, 89% of the total number of redds were observed in mainstem sites. Average redd size was about 2 m^2 during all years. Misidentification of resident trout or lamprey redds as Steelhead redds did not appear to be an issue, as the former were much smaller than Steelhead redds, and, in the case of trout, spawning was largely complete prior to the beginning of Steelhead spawning.

Increased base flows under Treatment 2 in 2009-2020 reduced the ability of the survey crew to spot adult Steelhead compared to previous years under Treatment 1. Several sections of the river had increased turbulence that prevented ideal conditions for the observation of adults, and higher current velocities made it difficult for the snorkeler to slow down enough for careful observation. Nevertheless, the peak number of live adults observed on a single survey during Treatment 2 (37-148) was generally higher than during Treatment 1 (22-95, Table 3.2). During 2001-2004, when snorkel counts of adult Steelhead occurred as part of a larger survey of Steelhead escapement in BC Lower Mainland streams (BCCF, Lower Mainland Branch, data on file), the maximum number of adult Steelhead observed on any one survey ranged from 20-64. However, peak counts should be considered a less reliable index of year-to-year differences in total escapement compared to redd counts. Unadjusted live peak counts of winter Steelhead are often poorly correlated with actual escapement due to the lengthy spawning period, and the immigration and emigration of fish into the counting area over the course of the survey period (Korman *et al*. 2002). This is also the case on the Coquitlam River. Peak counts explain only 40% of the variability of escapement based on redd counts largely as a result of far fewer live counts than expected in 2006 (Figure 3.5).

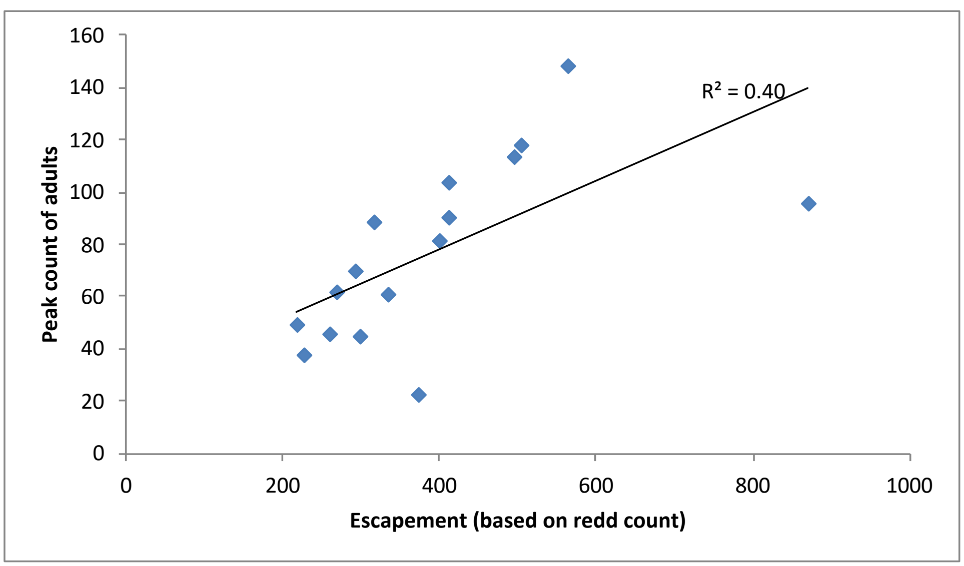

Figure 3.5 Relationship between Adult Steelhead escapement estimates based on redd counts and peak counts of live adults during surveys 2005-2020 in the Coquitlam River.

No redds were found during the once-a-year reconnaissance survey of Reach 1 in 2020. Reach 1 is not included in the annual Steelhead redd survey, and thus redd counts for this section were not included in estimates of adult escapement for the purposes of continuity with past years.

3.2.1 Redd survey life

In 2020, the period between surveys was typically short enough (≤ 16 days) to assume that a negligible number of redds became obscured from one survey to the next, based on the evaluation of redd survey life during 2005-2020. The survey intervals were sufficiently short that redd life model estimated less than one additional redd for all surveys. Thus, the estimated number of new redds was considered equal to the raw count of new redds (Table 3.2). Between 2005-2020, 2006 was the only year where the number of redds estimated using the redd survey life model was substantially higher (21%) than the unadjusted counts due to redds becoming undetectable over the 37-day gap between surveys that spanned the peak spawning period (Table 3.1). See Decker *et al.* 2010 for further discussion of trends in survey life.

3.3 Implications for hypothesis testing

The Coquitlam River is well suited to conducting Steelhead redd surveys and provides useful information for assessing the benefits to Steelhead of Treatment 1 and 2. Estimates of adult Steelhead abundance and egg deposition based on total redd counts may be systematically biased high or low due to uncertainty about the number of redds each females constructs, but can still be expected to provide a sensitive and reliable index of recruitment during 2005-2020.

4 Juvenile Salmonid Standing Stock

In 2006, the CC requested that a juvenile standing stock survey component be added to the monitoring program to provide estimates of total abundance in late summer for Coho and Steelhead fry (age-0+), and Steelhead parr (age-1+ and 2+) in the Coquitlam River mainstem. The purpose of this monitoring was that this data together with adult escapement and smolt abundance estimates, could be used to investigate freshwater production bottlenecks at specific juvenile life stages that may be related to specific habitat or flow issues. We conducted a feasibility study in 2006 to determine the best method for collecting annual juvenile standing stock data. The study compared three-pass removal electrofishing at 20 m long enclosed sites along one shoreline with night snorkeling counts at sites that extended across the entire stream channel (Decker *et al*. 2007). During 2007-2018 we proceeded with annual juvenile surveys based on night snorkeling counts, as this method proved to be the most effective for the purpose of estimating juvenile standing stocks (Decker *et al*. 2007). A multi-year mark-recapture study was also initiated in 2007 to provide estimates of snorkeling detection probability (percentage of fish present at a site that snorkelers detect), which is necessary to expand raw snorkeling counts to population estimates. In this report, we present a new Hierarchical Bayesian Model (HBM) that was developed to provide estimates of juvenile standing stocks in the Coquitlam River during 2006-2020 (see *Section 4.1.5*); this model replaces a bootstrap model used in previous years (Decker *et al*. 2012).

During 2007-2020 we also conducted a separate 3-pass electrofishing survey (with input and assistance from Ron Ptolemy, MOE stock assessment). As per the COQMON-07 Terms of Reference, the electrofishing data was collected to provide a comparison of fish densities in specific habitats in the Coquitlam River with fish densities from similar habitats in other streams that were sampled using the same methods (BC MOE juvenile electrofishing database; see Ptolemy 2007). The collected electrofishing data was not used to estimate juvenile standing stocks in the Coquitlam River.

4.1 Methods

4.1.1 Study area

The study area extends 10.3 km from the Coquitlam Dam downstream to the Patricia Footbridge just upstream of the Lougheed highway (i.e., reaches 2a, 2b, 3, and 4; Figure 4.1), and includes all mainstem, braid and side-channel habitat. Natural and man-made off-channel habitats in Coquitlam River were not included, and juvenile fish populations in these habitats were therefore not included in juvenile standing stock values reported in this section or in Section 6.

4.1.2 Sampling design

We employed a two-stage sampling design (Cochrane 1977) to generate juvenile standing stock estimates by species and age-class for the Coquitlam River study area. The first stage consisted of a single-pass snorkeling count at each of the 12 index sites for 2007-2014 that were sampled each year with another 12 index sites added in 2014 increasing the sites surveyed to 24 for 2014-2020. The second stage consisted of conducting mark-recapture experiments at a subsample of these sites to quantify snorkeling detection probability. Fish abundance at each site was estimated by expanding the observed number of fish by the estimate of detection probability (global mean across all mark-recapture sites in all years for each species/size class). The abundance of fish in the remaining length of the Coquitlam River study area that was not sampled (i.e., total stream length $-\Sigma$ stream length $_{\text{index sites 1-12}}$) was estimated based on estimates of the mean and variance in fish density for the sampled sites. Total standing stock estimate for the study area was the sum of estimates for sampled and unsampled stream length.

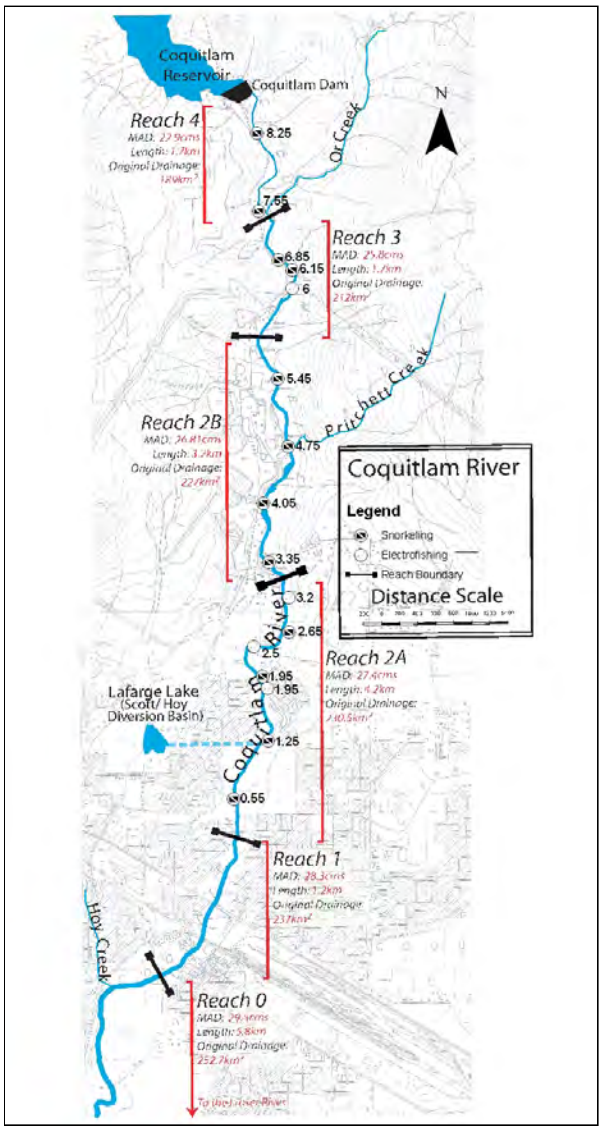

Figure 4.1 Map of Coquitlam River showing juvenile standing stock study area, reach breaks and original 12 sampling sites.

3. Adult Steelhead Escapement

For this type of sampling design, error in the estimation of fish standing stocks is the result of both process error (spatial variation in fish abundance among sampling sites) and measurement error (error in the estimation of fish abundance within an individual site). Measurement error includes variation in detection probability caused by differences in fish behaviour and habitat characteristics among sites, and differences among snorkelers in their ability to spot and identify fish. The HBM was used to estimate posterior distributions of the fish standing stocks, from which expected values (mean and median), and 95% credible intervals (Bayesian equivalent of confidence intervals) could be computed.

4.1.3 Night snorkeling

Snorkeling sites were chosen using a simple (non-stratified) systematic sampling design (SSS). Sampling was not stratified by reach or habitat type due to the limited number of sites sampled. During 2007-2013, the 10 sites originally selected in 2006 were re-sampled, and an additional two sites were added in Reach 4 to maintain a uniform sampling interval of ~ 0.85 km (Figure 4.1). The additional 12 sites added in 2014 (for a total of 24 sites) were distributed equidistance between the existing sites. Initial site selection was accomplished using a hand-held GPS unit to determine the straight-line distance from Patricia Footbridge to Coquitlam Dam, and dividing this distance by the total number of sites to obtain a uniform sampling interval. The downstream boundary of each site was then located according to the appropriate pre-determined distance from Patricia Footbridge. Each site was 25 m in length and spanned the entire stream channel. If the stream was split into two or more wetted channels at the selected site location, the entire wetted width of all channels was surveyed as part of the 25 m site to ensure that the site accurately represented available habitat for a particular channel cross-section. Snorkeling surveys were scheduled for early September when precipitation is normally low and target discharge from Coquitlam Dam was 0.8 m^3/s under Treatment 1 (2006-2008) and 2.2 m^3/s under Treatment 2 (2009-2020). Snorkeling counts were performed once at each site by a two-person crew. Counts were performed at night because numerous studies have shown that daytime concealment behaviour is common in juvenile salmonids (e.g., Bradford and Higgins 2000 and references therein). We limited snorkeling surveys to a four-hour period beginning 0.5 hours after dusk. We based this on Bradford and Higgins' (2000) finding that, throughout the year, the highest counts of juvenile salmonids during a 24-hour period were consistently recorded during a 3-4 hour period after dusk. To illuminate the sampling sites at night, snorkelers used handheld dive lights that cast diffuse rather than direct beams to minimize the disturbance to fish. Snorkelers surveyed the stream's entire wetted width, with each snorkeler entering the site at its downstream end and systematically sweeping in an upstream direction the area between his bank and the agreed upon mid-point of the site. Regular communication between snorkelers was essential to avoid duplicating counts, particularly in the instances where fish were present in mid-channel areas.

To address the potential concern that age-0+ salmonids, which occupy shallow, near-shore habitats, would be difficult to survey effectively by snorkeling (Griffith 1981; Campbell and Neuner 1985; Hillman *et al.* 1992), snorkelers delineated areas that were too shallow to view from an underwater position. Following the completion of an underwater search of the remainder of the site, conducted a separate visual survey of these areas on foot with masks removed. During the 2006 pilot study it was evident that small fish along the stream margin remained relatively stationary at night and could be identified to species and size class, and if necessary, could be captured with a small net to confirm observations. At sites where these shallow areas were not well delineated from the rest of the site, and the risk of double counting fish was apparent, the two snorkelers worked parallel to one another, with one person searching shallow near-shore areas, and the other searching adjacent off-shore areas. Each person communicated movements of detected fish to the other. This procedure was then repeated for the other half of the site. Other studies have shown that streamside visual counts can be excellent predictors of juvenile salmonid abundance when calibrated using more accurate methods (Bozek and Rahel 1991; Decker and Hagen 2009). Snorkelers identified to species and visually estimated the fork lengths of all fish observed and recorded their observations in waterproof notebooks. To aid in the estimation of fish lengths, snorkelers drew ruled scales on the cover of their notebooks. Snorkelers were typically able to hold the notebooks within 30 cm of a fish to measure its length without disturbing it. Although there was no attempt to assess the accuracy of fish length estimates made by snorkelers, in two similar studies (Korman *et al*. 2011; Decker and Hagen 2009) in which some of the same snorkelers from this study participated, it was found that snorkelers could estimate juvenile fish lengths relatively precisely with little negative or positive bias (R^2 values for regressions of estimated versus measured fork length ranged from 0.94 to 0.97).

4.1.4 Mark-recapture experiments to estimate snorkeling detection probability

To derive population estimates from snorkeling counts, an estimate of snorkeling detection probability (proportion of total fish at a site that snorkelers detect) is also required. The juvenile standing stock study design calls for 2-4 mark-recapture experiments to be completed during each year until enough data are obtained to provide a reliable model of detection probability. We conducted a total of 23 mark-recapture experiments during 2007-2013 towards this end. By distributing the mark-recapture experiments over several years and equally among the 12 annual sampling sites, bias resulting from differences in detection probability among years or habitat types was minimized. Now with well-defined detection probability information for Coho, age-0 fry and age-1 Steelhead parr, further mark-recapture experiments were suspended in 2014. Further mark-recapture experiments would not increase the precision of the standing stock estimates as much as doubling the number of index sites would, by possibly shifting effort from mark-recapture to index sampling. This prevents further refinements to the age-2 Steelhead parr detection probability estimate but this age-class has minimal use for estimating survival or other productivity metrics because a portion of this year-class undergoes smolting prior to fall surveys.

To estimate detection probability discretely for each target species/age-class at a sampling site, one night prior to conducting the normal snorkeling survey, a single snorkeler captured and marked fish throughout the site using one or two large aquarium nets affixed to handles of ~80 cm in length. The snorkeler searched for and captured fish throughout the site with the goal of marking 10-20 individuals for Coho fry, and for each length class of Steelhead (see below). Minimizing disturbance to marked and unmarked fish was a primary goal of the marking methodology. Captured fish were handed to a second crewmember on shore, who immediately measured the fish (fork length to nearest 5 mm), marked it, and returned it to its original location after the snorkeler had moved on. Anticipating that detection probability would differ for smaller and larger juvenile Steelhead over the size range occurring in the Coquitlam River

(Hagen *et al*. 2011; Korman *et al*. 2011), we used colour-coded tags to obtain five discrete mark groups for Steelhead (40-50 mm, 50-69 mm, 70-99 mm, 100-140 mm, and >140 mm). The smaller two length classes represent age-0+ fry, while the larger three represent age-1+ and 2+ parr. Marking consisted of inserting a custom-made tag into the fish's back at the insertion of the dorsal fin. Tags consisted of size 16-20 barbed fish hooks (size 16 for fish > 140 mm fork length, size 18 for fish 70-140 mm, and size 20 for fish < 70 mm), with a length of coloured plastic chenille (8-15 mm depending on fish size) attached at the hook eye with heat shrink tubing (Hagen *et al*. 2011). Tags were sized so that snorkelers could readily detect a mark on a fish, without the mark increasing the likelihood of the fish being seen relative to an unmarked one. Captured fish were not anaesthetized because of the uncertainty associated with behavioural effects from the anaesthetic. During the re-sighting event snorkelers recorded marked and unmarked fish separately.

Snorkeling detection probability was estimated for individual sites, species and length classes by dividing the number of marked fish seen by the number marked (R/M). This type of mark-recapture study assumes a closed population, whereas our sites were not enclosed with nets. However, over sufficiently short time periods, if study animals restrict their movements to a defined area, physically open sites can be treated as closed without introducing significant bias (Pollock 1982; Bohlin *et al*. 1989; Mitro and Zale 2002). We chose to conduct the underwater surveys 24 hours after marking because we considered this to be the shortest time period that would still allow fish to recover from marking and complete a diurnal cycle of movement and redistribution within the site, but would minimize movement from the site. We investigated the assumption of site closure by surveying an additional distance of approximately half the site length adjoining both the upstream and downstream site boundaries, so that the total distance surveyed for marks was approximately two times the length of the original site where fish were marked. Marked fish that had moved beyond the original site boundaries were recorded separately. The number of marked fish that emigrated from the original site was estimated as the number of marks observed in the adjoining sections divided by R/M.

4.1.5 Estimation of fish standing stocks and mean densities

There are predominately three age-classes of juvenile Steelhead (age-0+, 1+ and 2+) in the Coquitlam River during late summer; older fish are relatively uncommon and likely to be resident rainbow trout. We computed separate population statistics for each of the three age-classes, and also pooled age-1+ and age-2+ Steelhead data to compute aggregate population statistics for Steelhead parr. Steelhead ages were estimated based on an analysis of length frequency histograms generated from both the electrofishing and snorkeling data, as well as from length-age data derived from Steelhead smolts from the Coquitlam River (see *Section 5.2.2*). A small proportion of juvenile Coho salmon spend two winters in the Coquitlam River prior to migrating seaward, but we did not stratify our standing stock estimates for Coho by age.

To estimate juvenile standing stocks for the entire study area, and to quantify uncertainty in these estimates, we relied on a modified version of an HBM originally developed by Korman *et al*. (2010) to estimate juvenile Steelhead abundance in the Cheakamus River. Their model is in turn a derivation of a model originally proposed by Wyatt (2002, 2003). The sampling (night snorkeling) and calibration methods (mark-recapture) employed in the Korman *et al*. (2010) study were similar to those used in this study. The hierarchical structure of the HBM approach is well suited to two-stage sampling designs where it is necessary to combine error sources arising at different levels or hierarchies of the sampling design (Wyatt 2002).

The mark-recapture experiments indicated that the snorkeling detection probability for Steelhead was size-dependant (see *Sections 4.1.4 and 4.2.1.1*). In order to account for this, the HBM incorporates stratification by generating independent standing stock estimates for six Steelhead age-class/size-class strata (0+ < 50 mm; 0+ 50-70 mm; 1+ 70-99 mm; 1+ 100-149; 2+ 100-149 mm; and 2+ > 149 mm). To generate a standing stock estimate for a particular Steelhead age-class, the HBM sums estimates across the appropriate size-class strata.

Descriptions of all parameters, variables, constants, subscripts and equations used in the HBM are provided in Appendices 4.1 and 4.2. For the observation (detection) component of the HBM, the number of marked fish observed at snorkeling mark-recapture site *i* during the recapture event was assumed to be binomially distributed and to depend on the detection probability and number of marks released during the initial marking event (Appendix 4.2, Equation 4.1). The between-site variation in detection probability at mark-recapture sites was assumed to follow a beta hyper-distribution (Equation 4.2). The number of fish observed at index site *j* (regular sampling site as opposed to a mark-recapture site) was assumed to be binomially distributed and to depend on abundance at the site and a randomly selected detection probability taken from the hyper-distribution of detection probabilities (Equations 4.3 and 4.4). The process component of the HBM assumes that variation in juvenile abundance across sample sites follows a Poisson/log-normal mixture. That is, abundance within a site is Poisson-distributed with a mean equal to the product of fish density and length of stream that was sampled (Equation 4.5), and the log of fish density across index sites is normally distributed (Equation 4.6).

The total standing stock for the study area (Equation 4.9) was computed as the sum of the standing stock estimates from the 12 sampled index sites (Equation 4.7) and the standing stock estimate for the unsampled stream length within the stratum (Equation 4.8). The latter value was computed as the product of the back-transformed mean density from the lognormal density hyper distribution (μ_λ) with lognormal bias correction ($0.5\tau_\lambda$), and the length of the unsampled portion of the stratum.

Posterior distributions of parameters and standing stock estimates from the HBM were estimated using WinBUGS (Spiegelhalter *et al*. 1999) called from the R2WinBUGS library (Sturtz *et al*. 2005) from the "R" statistical package (R Development Core Team 2009). Uninformative prior distributions for hyper-parameters were used if possible for size-specific strata. As well, an uninformative uniform distribution, and an uninformative half-Cauchy distribution were used as priors for the mean and standard deviation of the hyper-distribution for age-, and size-specific detection probability, respectively (Appendix 4.2, Equations 10 and 11, respectively). An uninformative normal prior was used for the mean of the hyper-distribution for log fish density, and an uninformative half-Cauchy distribution was used as a prior for the standard deviation of log fish density (Equation 4.12). The half-Cauchy prior, also referred to as a 'folded t distribution', is useful in cases where it is difficult to estimate the variance of hyper-distributions in hierarchical Bayesian models due to limited information in the data (Gelman 2006).

4. Juvenile Salmonid Standing Stock

In a few cases, estimates of the variance in the hyper-distributions of detection probability or log fish density were unstable based on these uninformative priors. This occurred because there were either too few fish of a specific size class marked during the mark-recapture experiments to reliably estimate the standard deviation in detection probability ($\tau_{\theta,g}$, Equation 4.11), or the number of fish of a specific size class present in the index sites was too low and variable to reliably estimate the standard deviation in fish density among the index sites (τ_{λ}, Equation 4.13). In these cases, which are described in Appendix 4.3, rather than estimate $\tau_{\theta,g}$ and τ_{λ}, we used fixed values that were equal to the estimated parameter values for an adjacent size class. The means of the hyper-distributions ($\mu_{\theta,g}$ and $\mu_{\lambda,s}$; Equations 4.10 and 4.11) were still estimated separately for each fish size class.

Posterior distributions were estimated by taking every second sample from a total of 10,000 simulations after excluding the first 1000 'burn in' samples. This sample size and sampling strategy was sufficient to achieve adequate model convergence in all cases. Model parameters were estimated in two stages. In the first stage, the posterior distributions of site-specific detection probabilities and hyper-parameters were estimated (Equations 4.1 and 4.2). In the second stage, posterior distributions for the parameters in the population model were estimated. The $\theta_{j,g}$ values required for the population model were simulated from beta hyper-distributions whose parameters were determined from the median values of the posterior distributions estimated in the first stage. This two-phased estimation approach reflects our two-stage sampling design, and ensures that the hyper-distribution for detection probability is not influenced by data from the regular snorkeling index sites. Ideally, we could have sampled from the full range of detection probability hyper-distributions of detection probability in the second estimation phase. This latter approach, which integrates over the full uncertainty in detection probability hyper-parameters, increases computational time by two to three orders of magnitude. During the initial model development of a similar HBM for the Cheakamus River, Korman *et al*. (2010) compared uncertainty in juvenile Steelhead standing stock estimates based on the median versus fully integrated two-phased estimation approaches and found the increase in uncertainty under the latter approach was relatively modest (a few %). Based on their results, we adopted the more computationally efficient median approach. Korman *et al*. (2010) also used computer simulations to evaluate the extent of bias in standing stock estimates and hyper-parameters generated from the Cheakamus River HBM and found that bias to be negligible in all cases.

To describe the precision of the standing stock estimates in this report, we have used percent relative error, which we computed as the average half credible interval (upper 95% credible limit minus the lower credible limit divided by two and then divided by the mean and expressed as a percentage Krebs 1999). It is important to note that standing stock estimates and confidence intervals reported here will differ in future years' reports if estimates of size-specific snorkeling detection probability are further refined by additional mark-recapture experiments, or in the case where site-specific habitat or environmental variables (e.g., temperature, mean depth, etc.) are incorporated into the observation component of the HBM models, if they are found to be significant predictors of snorkeling detection probability.

Fish/km was calculated by dividing the standing stock estimate by the total length of the Coquitlam River (10.3 km). Fish/100 m^2 was calculated as the average density of the sampling sites. This is because the total area of the studied zone of the Coquitlam River is not assessed each year.

4.1.6 Day electrofishing survey

In 2020, we resurveyed four shoreline electrofishing sites previously sampled during 2007-2018. These sites were non-randomly chosen based on MOE protocols to represent fast-water habitats (riffle/cascades with relatively large mean substrate size) that were presumed to be ideal habitats for both Steelhead fry and parr (Ptolemy 2007). Sites were fully enclosed by upstream and downstream stop nets placed perpendicular to the shore, and a third offshore net that was placed parallel to the bank, and attached to the other two nets. Nets were held in place using a system of metal bipods, anchors and ropes, and cobbles and boulders placed along the bottom apron of each net. The offshore net was placed as far from shore as water depth and velocity permitted, usually 5-8 m.

Three-pass depletion electrofishing was conducted during daylight hours. Electrofishing was initiated at the downstream net, and consisted of a thorough search in an upstream direction, followed by a systematic sweep back towards the downstream net. Electrofishing sites were ‘rested’ for a minimum of one hour between passes to minimize decline in capture efficiency over subsequent passes (Bohlin and Sundstrom 1977). All salmonids captured were anaesthetized, identified to species, measured for fork length (nearest mm), allowed to recover and released back into the site following the completion of sampling.

Population estimates were generated for age 0+, 1+ and 2+ Steelhead (see *Section 4.1.5*, par. 1) and Coho at each site using a maximum likelihood (ML) algorithm (Otis *et al*. 1978).

4.1.7 Physical characteristics of snorkeling and electrofishing sites

We conducted simple habitat surveys to describe the physical characteristics of the sampling sites. At each site, depth was measured at five stations along each of three transects spanning the width of the site. During 2009-2020 we also estimated current velocity at each station using a propeller-type current meter. Stations were uniformly-spaced along transects, and transects were uniformly-spaced along the length of the site. We also recorded maximum depth, substrate composition (boulder, cobble, gravel, and fines as percentages of the site area), D90 and D50 (diameters of substrate particles for which 90% and 50%, respectively, of the site area consist of smaller particles), site length, site width, cover (categories included: overhead vegetation, turbulence, deep water and boulder as percentages of the site area, undercut bank as a percentage of the combined length of the stream banks, and the total area of the site covered by wood debris). Other information collected for each site included location (UTMs), and water quality parameters (water temperature, pH, and total alkalinity).

4.2 Results

4.2.1 Night snorkeling

In 2020, night snorkeling surveys were completed over six days between Aug 23-30 at flows of 3.1-3.4 m^3/s (WSC station 08MH002). Previous surveys were conducted at flows of 0.8-2 m^3/s during Treatment 1, and 2-7 m^3/s during Treatment 2. Water temperatures ranged from 19.6-20.9°C during 2020, similar to previous years. In 2020, horizontal underwater visibility exceeded 4 meters at all sites. In past years, visibility has been adequate to good at all sites (2008, 3-4 metres; all other years, >4 metres). This is more than adequate for conducting snorkeling counts (Hagen *et al*. 2011) and within the range of conditions that detection probability experiments were conducted. Physical characteristics of the snorkel sites are listed in Appendix 4.4

4.2.1.1 Mark-recapture experiments to estimate snorkeling detection probability

No additional mark-recapture experiments were carried out since 2013. This section will remain unchanged in future years until the need arises for additional experiments.

From 2007-2013, we marked totals of 454 Coho fry, 450 Steelhead fry, and 428 Steelhead parr at 23 mark-recapture sites (Table 4.1). Based on detection of marked fish by snorkelers during the survey 24 hours after marking, for Coho, the maximum likelihood estimate of mean snorkeling detection probability was 39% (Table 4.1, Figure 4.2), whereas for Steelhead, detection probability ranged from 26% for the < 50 mm length class of Steelhead, to 66% for the 70-99 mm class. For Steelhead, the results suggest an asymptotic relationship between detection probability and body size (Figure 4.2). Estimated detection probability for larger (> 140 mm) age-2+ parr (45%) remains highly uncertain given the limited number of tagged fish for this size class (24 fish across all sites and years). Steelhead larger than 140 mm at the end of summer are relatively uncommon in the Coquitlam River, representing only about 7% of the total standing stock of age-1+ and older parr.

The numbers of marked fish resighted by snorkelers in upstream and downstream sections adjacent to mark-recapture sites suggests that the assumption of population closure was largely met when mark-recapture sites were expanded to account for small-scale fish movement. Across the 23 mark-recapture sites, 27 marked Coho, 27 marked Steelhead fry, and 32 marked Steelhead parr were detected in adjacent upstream and downstream sections as opposed to the original marking site (Table 4.2). When adjusted for detection probability, these values suggest that 50 of 454 marked Coho (11.1%), 73 of 450 marked Steelhead fry (16.0%), and 52 of 428 marked Steelhead parr (age-1+ and 2+ combined: 14.8%), had moved from the original marking site to one of the adjacent sections during the 24-hour interval between the marking and re-sighting events. However, snorkelers noted that the majority of marked fish detected in the adjacent upstream and downstream sections had moved only a few metres beyond the original marking site.

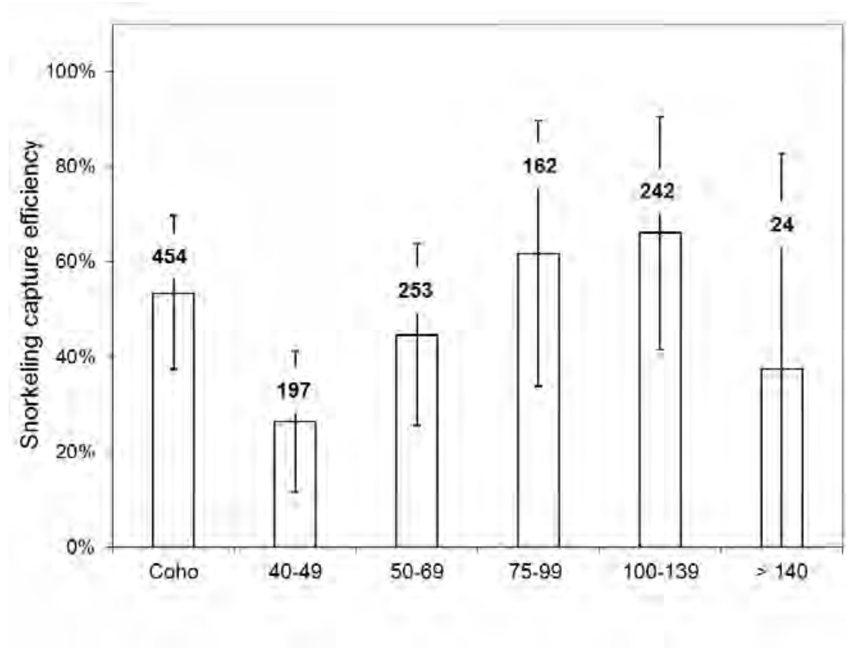

Figure 4.2 Maximum likelihood estimates of mean snorkeling detection probability for juvenile Coho and Steelhead by forklength class (Steelhead only) at 16 sites in the Coquitlam River during 2007-2013. Errors bars represent ±1 standard deviation of the mean. Values above bars are total numbers of marked fish for each category.

Table 4.1 Summary of mark-recapture results and snorkeling detection probability estimates for 16 sites in Coquitlam River collected 2007-2013.

Species	Fork length class (mm)	Total marks (M)	Total resighted marks (R)	Mean snorkeling efficiency	SD	No. of marks resighted in sections adjacent to original marking site	Estimated no. marks actually present in sections adjacent to original marking site
Coho	all	454	258	0.57	0.18	27	48
Steelhead	40-49	197	53	0.27	0.16	8	30
Steelhead	50-69	253	123	0.49	0.21	19	39
Steelhead	70-99	162	104	0.64	0.30	9	14
Steelhead	100-140	242	166	0.69	0.27	21	31
Steelhead	>140	24	9	0.38	0.45	2	5

4.2.1.2 Juvenile fish distribution and abundance

Abundance and density estimates for all years, species and age classes are listed in Appendix 4.6. The 2020 Coho fry abundance and 95% confidence intervals for the Coquitlam River mainstem was 43,389 ± 30%, which was slightly below average compared with previous years (mean: 50,830; range: 18,450-91,367 fish, Figure 4.3). Precision of estimates has increased substantially with the doubling of the number of sampling sites from ±38-44% for 2006-2013 to ±30-31% for 2014-2020 (Figure 4.3). Averaged individually for Treatment 1 and 2, Coho density generally increased with distance upstream during Treatment 2 (R^2 = 0.76) whereas during Treatment 1 density was lower between km 8-11 but was variable above this point (Figure 4.4).

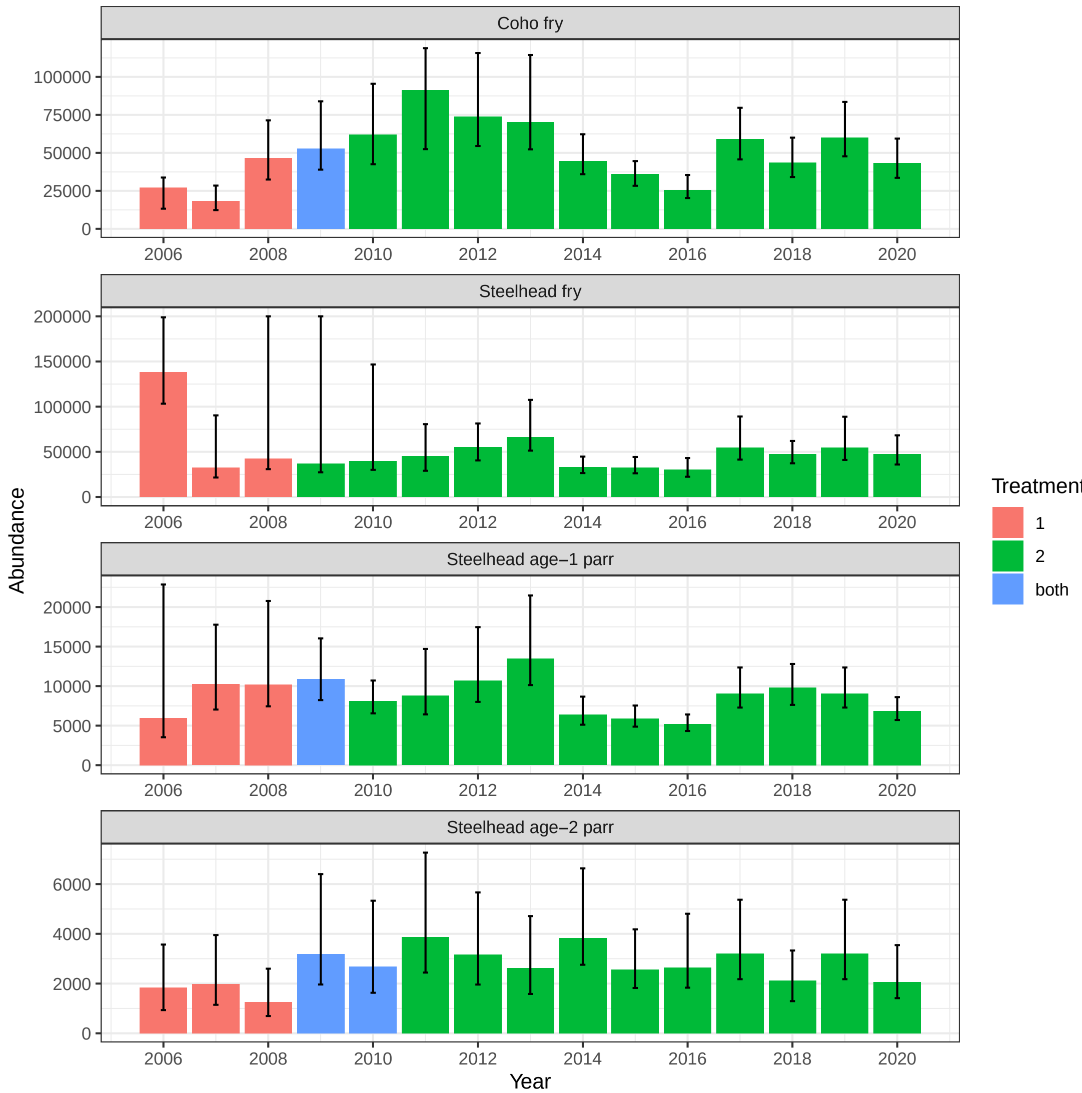

Figure 4.3 Estimates of juvenile standing stock, and 95% confidence intervals by species and age class in Coquitlam River during 2006-2020. Estimates were derived from night snorkeling counts with the exception of 2011 Steelhead (0+), which were based on electrofishing. Bar colour indicates cohorts that were reared entirely under Treatment 1 flows (red), Treatment 2 (green) or both (blue).

The 2020 Steelhead fry abundance of 47,408 ±34% was average compared to previous years 2006-2020 (mean: 48,878; range: 21,949-138,132 fish, Figure 4.3). We consider the 2020 fry estimate credible (minimally biased) since the assumptions underlying the mark-recapture methodology were largely satisfied. Similar to 2017, 2018 and 2019, there was a higher proportion of fish less than 40mm fork length (13%) compared with the average from past years excluding the 2011 outlier year (5%, Appendix 4.5), which was the minimum size included in mark-recapture experiments (>40mm). This may have biased the fry estimate low since observer efficiency decreases with the size of fry (Korman *et al*. 2011) and yet we used the observer efficiency based on fry with a fork length of 40-49mm for smaller fry as well (< 40 mm). However, the underestimation would likely be relatively small considering that the proportion of observations for fry less than 40mm from electroshocking – a more reliable sampling method for Steelhead fry – was far more similar to an average year than to 2011, when the large shift towards smaller sized fry led to a very unreliable snorkeling based estimate (see Schick *et al*. 2012, Appendix4.4). Relative precision was moderate in 2020. The precision of fry estimates increased substantially since doubling the number of index site in 2014 (±28-44%) compared to prior years (±37%-1790%, Figure 4.3). During Treatment 1, Steelhead fry densities were substantially higher in the middle portion of the study area (km 11-13) compared to upper and lower reaches (Figure 4.4), whereas during Treatment 2, density was far less variable and with no clear trend.

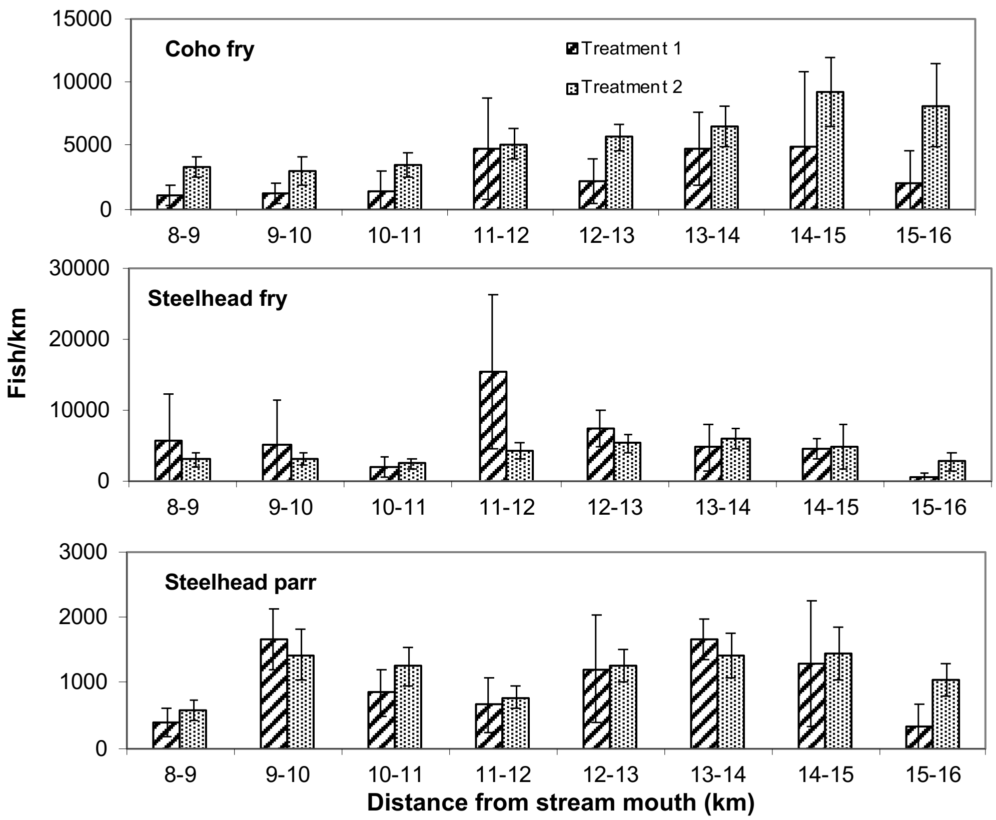

Figure 4.4 Linear distribution of juvenile salmonids in the Coquitlam River during Treatment 1(2006-2008) and Treatment 2 (2009-2020) Bars represent mean abundance estimates and 95% confidence intervals for years under flow Treatments 1 and 2. Estimates are based on calibrated snorkeling counts at 10-12 sampling sites 2006-2013 and 24 sites 2014-2020.

The 2020 standing stock estimate of age-1+ Steelhead parr of 6,863 ±21% was below the average abundance since commencing snorkel surveys (mean: 8,824; range: 5,889-13,456 fish, 2006-2018, Figure 4.3). The density of age-1+ parr in 2020 was 858 fish/km or 4.5 fish/100m^2.

The 2020 standing stock estimate for age-2 Steelhead parr was 2,057±52%. Mean density of age-2+ Steelhead parr in 2020 was 200 fish/km or 1.1 fish/100m^2. On average, this trend corresponds well with the transition between flow treatments for age-2+ parr that have spent one year under Treatment 2 conditions even though 2011 would be the first estimate for age-2+ parr that reared entirely under Treatment 2 conditions (Figure 4.3). It is difficult to draw conclusions about the impact of flow treatment on productivity from age-2+ parr abundance since it reflects both survival to age and changes in life history. Age 2+ parr in the Coquitlam River can smolt during the spring prior to the fall standing stock surveys or in the following spring. Thus, an increased abundance of age-2+ parr could reflect greater freshwater survival and/or an increase in the proportion of smolting the following spring. There was no strong longitudinal pattern in Steelhead parr density when averaged by treatment period and was also relatively similar between the two treatment periods (Figure 4.4).

4.2.2 Assumptions of estimates based on snorkeling counts

A key assumption of our mark-recapture calibration method was that marked and unmarked fish had equal probabilities of being seen by snorkelers the night following marking. Testing for this type of bias was beyond the scope of this study, but we made considerable effort to minimize the effects of handling and marking on fish behaviour: fish were captured in a relatively low impact manner (hand nets), were not anaesthetized prior to marking, were released into the same location that they had been captured from (or first seen in), and were allowed a 24-hour recovery period prior to the re-sighting event. Snorkelers noted that, after 24 hours, marked fish occupied comparable locations to unmarked ones and behaved in a similar way.

A second assumption of our mark-recapture methodology was that the populations were closed between marking and re-sighting events. While our sites were not enclosed, we treated the fish populations within as being closed over the 24-hour period between marking and the snorkeler survey. Some marked fish did move from the original marking site to adjacent upstream and downstream sections during the 24-hour period, with “movers” representing 11% (Coho fry) to 22% (Steelhead 2+ parr) of the total number marked. We included these movers as part of the re-sighted population to account for small-scale movement, but this would not have accounted for larger-scale movements (i.e., marked fish moving beyond the adjacent sections of each mark-recapture site to areas not surveyed by the snorkelers). While movement beyond the adjacent sections would lead to negative bias in our estimates of snorkeling detection probability, we assumed that larger-scale movements of marked fish were relatively uncommon considering that almost all of the marked fish that were detected by snorkelers beyond the original marking site had remained within a short distance (< 5 m) of the original site boundaries.

Detection probabilities derived from mark recapture estimates always refer to the catchable population. All Coho fry are treated as one population while juvenile Steelhead are partitioned into several sub-populations, based on fish length, to minimize the variability in detection probabilities within each sub-population or size-class. During 2008-2010 and 2012-2016 the size class during mark-recapture experiments matched that during index sampling. However, this was not the case in 2011 when the smaller-than-usual Steelhead fry were likely less visible than the years upon which the mark-recapture results were based. As mentioned, this was also the case in 2017-2020, but to a lesser degree than 2011, leading us to treat these estimates as reliable.

4.2.3 Stream-wide fish abundance estimates based on snorkeling counts

The snorkeling surveys indicate that Coho and Steelhead fry and parr are broadly distributed within the study area of the Coquitlam River mainstem, although Coho production was concentrated in the upper portion of the study area during most years. The majority of adult Coho spawn in the upper river. Steelhead fry densities are low in reach 4 relative to downstream reaches. Whereas the channel is relatively confined and deep in Reach 4, in the remaining reaches downstream, it is much broader, with more frequent braids and side-channel and shallow margin areas, which are preferred fry habitats (Hume and Parkinson 1987).

Riley *et al*. (1997) surveyed juvenile abundance in the Coquitlam River in 1997, prior to the installation of the 'fish flow' valves and the implementation of Treatment 1. Although their sampling methodology differed as they used three-pass electrofishing, lower flows enabled them to enclose sample sites across the entire wetted width of the channel, similar to our channel-wide snorkeling sites. Comparing the results of the two studies would suggest that mean densities of Coho fry in the Coquitlam River mainstem during 2006-2020 (13-42 fish/100 m^2) ranged from just over two and a half to nine times that in 1997 (5 fish/100 m^2, Riley *et al*. 1997). Compared to the Steelhead fry density in 1997 (12 fish/100 m^2), Steelhead fry densities in 2006-2020 were up to more than 2-fold higher (8-29 fish/100 m^2). Steelhead parr densities were 6-19 times higher during 2006-2020 (3.7-8.1 fish/100 m^2, respectively) compared to 1997 (0.5 fish/100 m^2). However, electrofishing removal estimates obtained in 1997 were biased-low, particularly for Steelhead parr, as a result of low conductivity and ineffective electrofishing in deeper mid-channel habitats (Riley *et al*. 1997), thus exaggerating the apparent increases in standing stock from 1997 to 2006-07. Nevertheless, the differences in Coho fry and Steelhead parr densities between 2006-2020 and 1997 are likely too large to be explained by negative bias alone in electrofishing depletion estimates (Bohlin and Sundstrom 1977; Peterson *et al*. 2004). While other factors may have also played a role, increased flow releases from the dam during Treatment 1 and 2 relative to earlier years (0.06 to 0.5 m^3/s) likely contributed to increased juvenile Coho and Steelhead production in the Coquitlam River.

Based on the calibrated snorkeling data, Steelhead fry density in the Coquitlam River in 2006- 2020 (98-29 fish/100 m^2) was typically low to average compared to published values for other streams. For example, Hume and Parkinson (1987) considered 30 Steelhead fry/100 m^2 to be about average for BC coastal streams, while Ward and Slaney (1993) reported that Steelhead fry densities in Keogh River averaged 34 fish/100 m^2 one month after emergence.

Snorkeling-derived estimates of Steelhead parr densities in the Coquitlam River (3.3-8.3 fish/100 m^2) were comparable to parr density estimates derived from daytime snorkeling counts in Oregon streams (Satterthwaite 2002), and from night snorkeling counts in tributaries of the Thompson River, BC (Decker *et al*. 2009). However, some of the streams sampled by Satterthwaite (2002) had Steelhead parr densities that were considerably higher (up to 20 fish/100 m^2).

4.2.4 Fish densities in 'optimal' habitats based on electrofishing

Appendix 4.7 contains the linear and areal density estimates based on electrofishing. Density estimates were not used to estimate river-wide abundance since sampling sites were not representative of the unsampled areas in the study area. Density estimates based on electrofishing and snorkelling across years for all species and age-classes were poorly correlated ($R < 0.4$, Table 4.2). Differences in density estimates derived from the two methods are expected, given that snorkeling was conducted at randomly chosen sites that spanned the entire channel width, whereas electrofishing sites were deliberately chosen to represent 'optimal' Steelhead habitat and encompassed only a portion of the channel width.

Table 4.2 Comparison of backpack electroshocking and night snorkeling fish density estimates (fish/km) for juvenile Coho and Steelhead in the Coquitlam River including: sample size (N), correlation coefficient (R) ,and the minimum and maximum ratio of estimates based on electroshocking to snorkeling 2006-2020.

				EF:SN	
Species	Age Class	N	R	Min	Max
Coho	0+	15	0.21	0.9	41.4
Steelhead	0+	15	0.24	0.4	2.9
Steelhead	1+	15	0.34	0.6	4.86

Electrofishing surveys in Coquitlam River during 2007-2020 followed a standardized methodology developed by Ron Ptolemy (BC MOE) to facilitate among-stream comparison of relative Steelhead abundance in 'optimal' habitat. Ptolemy (2007) proposed an empirical maximum carrying capacity biomass of 272 g/100m^2 for individual age-classes of Steelhead (combined age-classes would exceed this value) in suitable habitats in the Coquitlam River. This value represents the 95th percentile of the distribution of observed fish densities versus mean weights (Ptolemy 2007; Allan plot on p. 4). This distribution included electrofishing data from 2007 and from previous MOE electrofishing surveys in the Coquitlam River (pre-1998). The Coquitlam River is located in the Coast and Mountains Ecoprovince, and comparisons within this landscape unit are appropriate. A maximum biomass of 272 g/100m^2 places the Coquitlam River at about the 65% percentile for this landscape unit, which includes data for 86 streams (R. Ptolemy, MOE Fisheries Branch, pers. comm.). The model suggests that the potential carrying capacity for the Coquitlam River exceeds the average for the Ecoprovince. Using electrofishing and alkalinity data from streams in all provincial landscape units, Ptolemy developed a model to predict maximum salmonid biomass based on total alkalinity, as an index of nutrient status (R. Ptolemy, pers. comm.). The observed maximum biomass of 272 g/100m^2 exceeded the model prediction for the Coquitlam River of 200 g/100m^2 (based on very low alkalinity; e.g., 8-13 mg/l in 2006), which suggests above-average carrying capacity for the Coquitlam River relative to streams of comparable nutrient richness as described by this model.

Assuming a mean weight of 14 g for age-1+ Steelhead (R. Ptolemy, pers. comm.), maximum biomass values observed at electrofishing sites in the Coquitlam River were 139-236 g/100m^2 during Treatment 1 (2006-2008) and 38-94 g/100m^2 during Treatment 2 (2009-2017). Based on a mean weight of 2.5 g for age-0 fry, maximum biomass values observed at electrofishing sites in the Coquitlam River were 123-342 g/100m^2 and 64-127 g/100m^2 during Treatment 1 and 2, respectively. Thus, observed maximum values during 2006-2020 were mostly below or well below the 'historical' observed maximum of 272 g/100m^2 . However, given the limited number of sampling sites each year it is possible that electrofishing surveys in 2006-2020 failed to include ‘optimal’ sites where maximum Steelhead biomass would be expected.

4.3 Implications for hypothesis testing

Standing stock monitoring was designed to provide stock-recruitment information at a shorter timescale than possible using smolt outmigration but also at a lower level of precision. As well, it was meant to inform on the distribution of abundance throughout the lower Coquitlam River. To this end, it is satisfying its objective. However, it was not intended as the primary metric for evaluating the fisheries benefits of Treatment 1 and 2.

Standing stock monitoring data based on snorkel surveys provides accurate abundance estimates for mainstem Coho, Steelhead fry and age-1+ Steelhead along with a consistent index of age-2+ Steelhead abundance. While the precision of 2006-2020 standing stock estimates are lower than the average for smolt and fry outmigration, it does provide useful information for distinguishing at what life-stage abundance may become limited by adult escapement versus rearing habitat availability.

5 Smolt and Fry Outmigration

5.1 Methods

5.1.1 Coho and Steelhead smolt enumeration

In 2020, downstream migrating Coho and Steelhead smolts were captured at three locations in the Coquitlam River mainstem (RST2, RST3, RST4) using rotary screw traps (RST), and at the outlets of four constructed off-channel sites using full span weirs (Figure 5.1). Mark-recapture data collected at RSTs were used to estimate smolt numbers for three mainstem reaches and for the entire Coquitlam River upstream of Port Coquitlam (Figure 5.1).

Our approach assumes all fish passing full span weirs are captured (capture efficiency = 100%) but that only a portion passing each RST site are caught (capture efficiency ≠ 100%). We use variations of the Peterson two-sample mark-recapture method (Seber 1982) to estimate the proportion of fish passing each RST that are caught and not caught. This information is then used to adjust the catch from the RST site by the capture efficiency of the trap. For example, if an RST caught 50 of the 100 marked fish released upstream, the trap would have a 50% capture efficiency or that the trap captures half of the fish that pass by. And if a total of 1,000 fish were captured at that same trap, then we would estimate that another 1,000 fish passed without capture for a total estimate of 2,000 fish passing the trap. Our marking approach divides the outmigration period into a number of time periods (strata) that allows for individual estimates of capture efficiency for each stratum.

The Peterson mark-recapture method relies on several assumptions in order to produce unbiased estimates. Arnason *et al*.(1996) lists the three for time stratified mark-recapture as:

Closed population – There is no recruitment, emigration or mortality between sampling events.

No tag loss – Tag loss can take several forms in the context of this study. Tagging can induce mortality. Tags can be lost or go undetected. This includes that marks remain identifiable when marked fish pass the recapture site.

Equal catchability – That the probability of capturing a fish at the recapture sites is independent of whether it was marked or not.

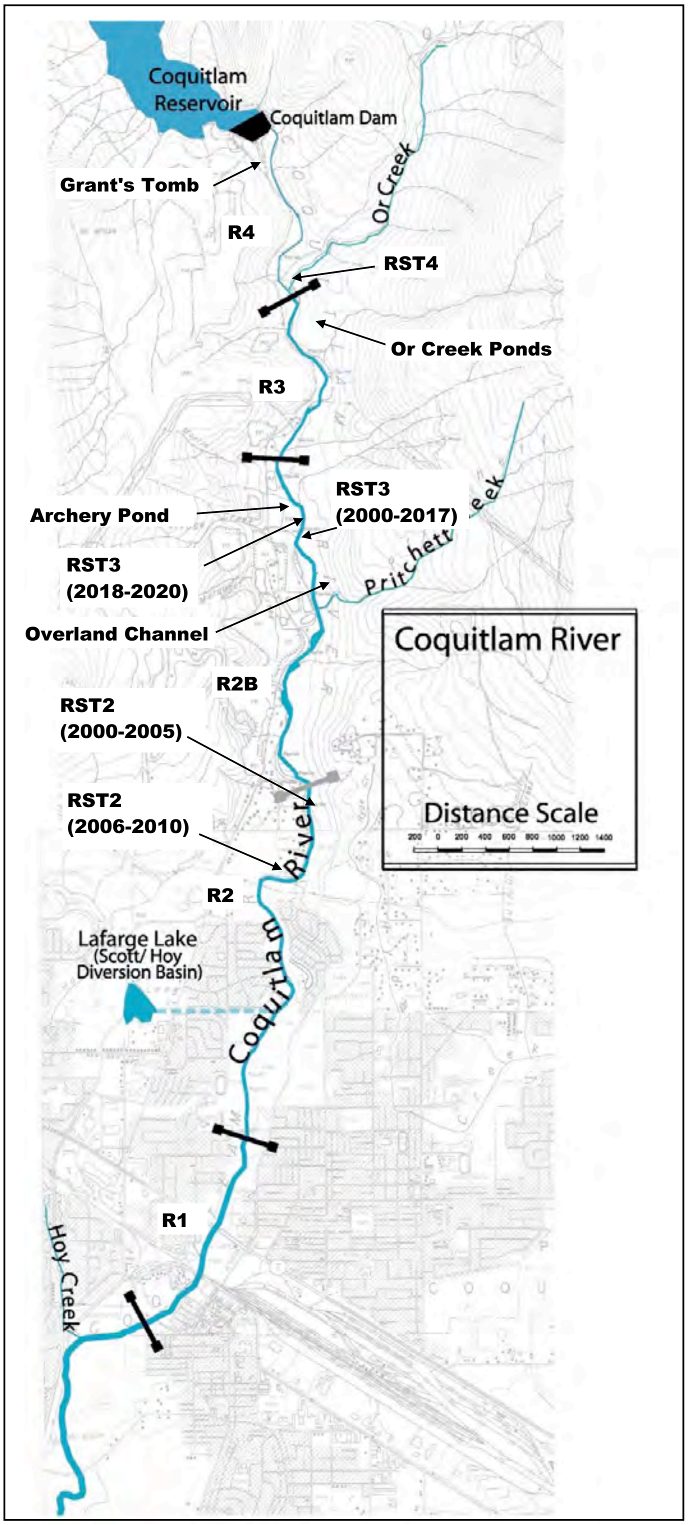

Figure 5.1 Map of the Coquitlam River showing constructed off-channel habitat sites, mainstem reach breaks and the locations of mainstem rotary screw traps (RSTs).

5. Smolt and Fry Outmigration

5.1.1.1 Location and description of downstream traps

Ideally, RST trapping would be conducted at the downstream end of Reach 1 at Port Coquitlam (the upper limit of tidal influence), so as to estimate smolt yield for the entire study area of the Coquitlam River. However, because of problems with site security, and given the limited number of sites that possess adequate water depth and velocity, RSTs were not installed at the downstream reach boundaries (Figure 5.1). Until 2005, our lowermost trapping site (RST2) was located just downstream of the upper boundary of reach 2a, 5.1 km upstream of the Reach 1 downstream boundary (Figure 5.1). The 2.6 km long section between RST2 and RST3 immediately upstream includes most of reach 2b and the upper portion of reach 2a, and is referred to in this report as reach 2. For 2006-2018, the RST2 site was moved 600 m downstream (a high water event infilled the former trapping site), increasing the length of the 'reach 2' section to 3.2 km. In 2018, the RST3 site was moved 300m upstream of its previous location to improve access. We refer to the 2.7 km long section between RST3 and RST4 as reach 3 (Figure 5.1), but it should be noted that this section also includes the upper 900 m portion of reach 2b. The uppermost trap site (RST4) was installed 1.6 km below the Coquitlam Dam, trapping a section that includes all but 100 m of reach 4 (Figure 5.1)[1].

In annual reports prior to 2009, smolt yield for the entire study area included the 4.5km of reach 1 and 2 downstream of the lowest trapping site. To allow for this, we approximated smolt numbers for reach 1 and the portion of reach 2a downstream of RST2 (4.5 km of habitat) based on extrapolation of smolt densities in reach 2 immediately upstream of RST2 site (i.e., reach 2b and a portion of reach 2a). However, this represents a potentially serious source of bias depending on the degree to which actual smolt densities in the 4.5 km section downstream of RST2 differ from those immediately upstream. For example, extrapolating relatively high Steelhead smolt density in reach 2 in 2008 (3.1 smolts/100m^2) to the 4.5 km section downstream, resulted in an estimate of 9,245 Steelhead smolts for the Coquitlam River mainstem based on 5,480 smolts passing RST2 (see Decker *et al*. 2009). This suggests that the unsampled lower 4.5 km section produced 41% of mainstem Steelhead smolts, despite relatively low densities of Steelhead redds (Figure 3.2) and parr (Figure 4.3). With the exception of Chum, spawning occurs primarily upstream of RST2 for the four species included in the monitoring program (Coho: 92%; Chum: 50%; Pink: 74%; Steelhead: 88%; mean values across years). To eliminate potential bias associated with extrapolation of smolt numbers downstream of RST2, estimates of smolt yield for the Coquitlam River in all years reported here are for the 7.5 km long section upstream of RST2 only, rather than for the entire 12.0 km long study area extending from the dam to the downstream boundary of reach 1 (see Figure 5.1). With respect to stock-recruitment relationships, and egg-to-smolt survival estimates, this assumes that all juvenile recruits from spawning upstream of RST2 will remain upstream of RST2 until they emigrate as smolts. However, downstream movement of pre-smolt juveniles occurs in the spring as evidenced by significant catches of age-1 Steelhead parr at the RSTs (and likely occurs during other portions of the years as well).This results in some degree of negative bias in our estimates of egg-to-smolt survival.

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[1] Prior to 2002, a full-span downstream weir was used in place of an RST in reach 4 (see Decker and Lewis 2000).

There are four large constructed off-channel sites (Or Creek, Grant's Tomb, Overland Channel, and Archery Pond) located between Coquitlam Dam and RST2, totaling about 27,000 m^2 of habitat (Figure 5.1). The enumeration of smolts from the off-channel sites was necessary for two reasons: 1) to distinguish between smolt production in constructed off-channel habitat that is largely unaffected by flow releases from the dam, and production in natural mainstem habitat that is directly affected by flow releases; and 2) to provide additional marked smolts to improve the precision of smolt abundance estimates for downstream mainstem reaches.

We relied on total counts at full-span downstream weirs (Conlin and Tutty 1979) to estimate smolt yield from three of the sites (Or Creek, Grant's Tomb, Overland Channel). Overland Channel consists of two ponds that are connected, with each pond also having its own outlet channel. We installed an inclusion fence at the outlet of the upper pond at the Overland Channel sites, forcing all smolts to migrate through a single weir installed in the outlet of the lower pond. Detailed descriptions of these sites and the design of the downstream weirs can be found in previous reports (e.g., Decker 1998).

A full span weir was used to enumerate smolts leaving Archery Pond. This approach was used prior to 2008 and 2014 onward. During 2009-2013, the method switched to use mark-recapture by minnow trapping to estimate pre-smolt abundance. This was to shift resources to the sites that historically produced greater numbers of smolts. However, during 2013, too few Coho pre-smolts were captured in Archery Pond for population estimates (6 fish with 200 "Gee" brand wire mesh minnow traps baited with 2 g of preserved roe and set for 24 hours).

5.1.1.2 Downstream trap operation

In 2020, one 2.4 m diameter RST was operated at the reach 4 trapping site (RST4), one 1.8 m RST was operated in reach 3 (RST3) and two 1.8 m RSTs (RST2; Figure 5.1) were operated in close proximity to one another in reach 2. Using two smolt traps at the RST2 location was intended to increase the capture efficiency, which is key to produce precise mainstem population estimates. A 2.4 m diameter RST was used previously at the reach 4 site, with the exception of 2018. The decision to return to the 2.4 m RST at this site was based on the low capture efficiency in 2018. Screening used on all of these RSTs was 12 mm in diameter on rotating drum and 9mm for retention box. An additional 1.3m diameter RST (RST2.2) with 2.5mm mesh size was operated at the RST2 location to capture outmigrating Chum and Pink fry.

The objective is for all off-channel weirs and the mainstem RSTs to operate continuously from early-April until mid-June, with the exception of RST2.2, used for Chum and Pink enumeration, which starts operation in early-March. Table 5.1 lists the dates traps were installed, decommissioned and the start dates of marking strata. During some years, one or more RSTs were installed as early as February to monitor for early outmigration but operation was sporadic and was not designed to produce reliable population estimates for the period prior to full operation. Thus, data from the early monitoring was not used in the outmigration estimates. All juvenile fish captured at the weirs and RSTs were identified to species, counted and measured for fork length (nearest mm). Unmarked Coho and Steelhead smolts and Steelhead parr were injected with Visual Indicator Elastomer (VIE) to apply a unique mark identifying capture period and location (see Section 5.1.1.3). Prior to 2018, only fin clips were used to mark for release period and location. To minimize behavioural effects from handling, every effort was made to reduce the stress on fish during the sampling and marking process, and, once recovered, fish were immediately released.

We assumed that all downstream migrating Coho larger than 60 mm fork length were smolts. Steelhead smolts in the Coquitlam River range from two to four years in age. We assumed that all Steelhead 120-230 mm in length were seaward migrating smolts, while fish <120 mm were yearlings and smaller age-2 fish that would remain in the river for at least one more year (see section 5.3.2 for a discussion of this assumption). Frequency histograms of Steelhead fork length from previous years suggest that most two-year old Steelhead smolts are greater than 120 mm in length during the spring migration. We recorded daily catches of Steelhead parr (< 120 mm) caught at each downstream trapping site, but, because there was no way of knowing what proportion of the total parr population these downstream migrants represented, we did not attempt to estimate parr populations by mark-recapture. Conversely, it was reasonable to assume that all smolts were downstream migrants. However, trapping personnel have noted marks from previous years on captured Steelhead smolts, indicating that at least a small portion of Steelhead >120 mm that are counted as smolts are actually parr that remain in the river for an additional year. This resulted in some degree of positive bias in estimates of annual smolt yield. During 2005-2020, we collected scale samples from randomly selected Steelhead captured at the RSTs in order to estimate the proportions of age-2+ and age-3+ fish in the smolt population. This was necessary for estimating egg-to-smolt survival since the progeny from one spawning event will outmigrate after either two or three winters in freshwater.

5.1.1.3 Differential marking by period and initial capture location

As in previous years, we estimated smolt abundance in mainstem reaches of the Coquitlam River using a stratified mark-recapture method (Arnason *et al*. 1996, Schwarz *et al*. 2009). Significant temporal variation in capture efficiency (% of marked smolts recovered) is common when mark-recapture methods are used to estimate the abundance of a migrating population (Seber 1982), and stratifying marking by period allows for unbiased estimates when temporal variation in capture efficiency is expected.

To provide distinct mark groups over time, all unmarked Coho and Steelhead smolts captured at the off-channel weirs and the upstream RSTs (RST3, RST4) were differentially batch-marked according to date (week) and location of initial capture (Table 5.1). In addition, unmarked Steelhead captured at RST2 were uniquely marked so that they could be released upstream (≈ 1 km) rather than downstream in order to increase the size of the marked population available for capture at RST2. Similarly, for the uppermost RST site (RST4; Figure 5.1), marked populations of Coho and Steelhead originating from the Grant's Tomb off-channel site were augmented by marking and releasing captured mainstem smolts at a site about 1 km upstream of RST4.

Prior to 2018, the unique mark types consisted of a small clip at one of several fin locations. In 2018, VIE was introduced based on its long mark-retention, ease of distinguishing mark types and greater number of unique mark types possible than with fin clips. VIE used six colours (green, pink, orange, yellow, red and blue) injected into the caudal fin or a clear tissue deposit posterior to the eye. The duration of each marking period was set at 7 days for the 2020 season. This is a departure from years prior to 2018 in which the duration was based on achieving a minimum recapture target of 40 Coho and 10 Steelhead smolts from each mark group and trapping location. There were two reasons for this change. First, we planned to use a different model to estimate time-stratified capture data that benefits from set length marking strata as well as a higher number of marking strata. Second, VIE allows for a larger number of marking strata than fin clips. To evaluate that detection rates were similar for VIE and fin clips, we compared recapture rates for each method over two weeks. To do this, half the fish at each trap received the VIE mark and the other half received a combination of caudal clips. Capture efficiency for each species, and trap were then tested whether they were significantly different.

Coho and Steelhead smolts as well as Steelhead fry were also given a unique mark based on where they were initially captured (see Appendix 5.1 and paragraph below). This mark served two purposes. First, it allowed for separate population estimates for mainstem and off-channel habitats. Since all off-channel fish were marked, all unmarked fish captured at the RSTs were assumed to be of mainstem origin and then could be used to estimate the number of mainstem fish passing each RST. The second purpose was to evaluate whether fish marked at off-channel weirs had similar capture efficiencies at each RST as mainstem fish. If they were similar, both could be used to estimate the number of unmarked smolts (mainstem origin) passing each RST. If they were different, then only mainstem marked fish would be used. This is more of an issue for Coho than Steelhead since very few Steelhead smolts originated from the off-channel sites while a large proportion of marked Coho smolts do. This is a concern for Coho because previous work in the Coquitlam River has shown significant differences in capture efficiencies for smolts originating from these two habitat types (Decker and Lewis 2000; Decker *et al*. 2003), suggesting that estimates based on combined marked populations could be biased. By separately analyzing marking and recovery data for these different mark groups, we were able to evaluate the capture efficiency of smolts marked as originating from mainstem versus off-channel habitat passing the same RST.

Since the precision of mark-recapture estimates generally improves with the number of smolts marked, it is advantageous to generate estimates based on combining data for different mark groups. To decide which spatial mark groups could be included in the final mark-recapture dataset for a particular RST, we used the following rationale and statistical tests:

1. We assumed that capture efficiency for unmarked smolts from the mainstem would be better approximated by observed capture efficiency for marked mainstem smolts than by observed capture efficiency for marked off-channel smolts, although we were not able to test this (see section 5.1.1.5),

2. Using Fisher's exact test, we tested whether overall capture efficiency (pooled data for temporal mark groups) differed ($P < 0.05$) for marked smolts from the off-channel and

mainstem areas. For example, capture efficiencies (CE) for off-channel and mainstem smolts at RST2 were computed as:

$$\frac{\sum_{i}^{6} R_{off-channel,i}}{\sum_{i}^{6} M_{off-channel,i}} \quad \text{and} \quad \frac{\sum_{i}^{6} R_{RST2,RST3,RST4,i}}{\sum_{i}^{6} M_{RST2,RST3,RST4,i}} \tag{5.1}$$

where

$R_{off\text{-}channel,i}$ = number of marked off-channel smolts from marking period i that were recaptured at RST2

$M_{off\text{-}channel,i}$ = number of off-channel smolts marked during marking period i

$R_{RST1,RST2,RST3}$ = number of marked mainstem smolts (all mainstem trapping locations summed) from marking period i that were recaptured at RST2

$M_{RST2,\ RST3,RST4,I}$ = number of mainstem smolts that were marked during marking period i

3. If we failed to detect a difference in CE, all mark groups were considered in the dataset used to compute the final mark-recapture estimate. On the other hand, if a difference was detected, the final dataset was limited to data for the mainstem mark groups only.
4. Off-channel mark groups were also rejected if when compared to the estimate using only the mainstem mark group, the estimate using both mainstem and off-channel mark groups was either 1) less precise or 2) differed by more than the standard error of either estimate.

5.1.1.4 Population estimates

For the three off-channel sites where full-span weirs were operated, in the absence of evidence to the contrary, we assumed a capture efficiency of 100% for each weir, and used the total number of smolts captured to estimate smolt production.

We estimated the number of smolts passing each RST on the mainstem of the Coquitlam River using one or more analytical approaches that all rely on a two-sample mark-recapture method. In this study, smolts captured and marked at the weirs or upstream RSTs constituted the marking sample and smolts recovered at an RST represent the recovery sample. With stratified mark-recapture methodology, both the marking and recovery samples are stratified.

Darroch Maximum Likelihood Estimator

For 2000-2017, we used the maximum likelihood (ML) model developed by Darroch (1961) and modified by Plante (1990) for stratified mark-recapture data. We refer to this model as the Darroch ML model. Darroch ML population estimates and confidence intervals were computed using a software package that is available to the public (SPAS, http//www.cs.umanitoba.ca/~popan/). A description of the ML estimator and the use of the SPAS software is provided by Arnason *et al*. (1996). In general, we delineated six marking and recovery periods although in some cases, it was necessary to pool strata to avoid small sample and numeric problems that may have prevented the maximum likelihood iterations from converging. When pooling strata, we followed the recommendations of Arnason *et al*. (1996). When there were no recaptures in a recovery stratum, it was dropped rather than pooled (Schwarz and Taylor 1998).

Bayesian Time Stratified Petersen Analysis System

In 2018, we started using a Bayesian spline model using the package BTSPAS (Bayesian Time Stratified Petersen Analysis System) developed by Bonner and Schwarz (2014), which was run in the R statistical programming environment (R Core Team 2016). The model was configured for non-diagonal data since marked smolts are often recaptured in the strata they were marked in as well as the next strata. The method has two main components, 1) modeling the general shape of abundance during outmigration using a spline and 2) using a hierarchical Bayesian model that can share capture probabilities (capture efficiency) across strata. BTSPAS is similar to the Darroch ML estimator but can deal with several commonly encountered data problems. First, it helps reduce problems from low numbers of recaptures in some temporal strata by "sharing" capture efficiency information among adjacent strata. Second, the model automatically adjusts the amount it "shares" mark-recapture data between strata based on the amount they vary across periods. It will share capture efficiency data when it is similar (analogous to pooling) but when it isn't, it estimates realistic precision when variation is high. Third, it can interpolate capture rates for strata with missing or minimal mark-recapture data or for strata with missing unmarked fish data.

The model assumes continuous operation of the recapture trap within a stratum. In cases where the recapture trap operates in only a portion of the stratum, the heterogeneity in capture efficiency within the strata is not properly incorporated into the model, leading to overly optimistic estimates of precision as well as bias. Instead, these strata must be flagged as problematic so the data is treated as missing and abundance imputed based on the spline.

Each of the three MCMC chains was run for 20,000 iterations. The first 10,000 were discarded and the remainder thinned at a ratio of 1:50. This provided 6,000 iterations for each estimate.

Model fit was assessed using Bayesian p-values to quantify the similarity between expected and observed recaptures, unmarked captures and both combined. Values close to 0.5 reflect good fit. Values near 0 or 1 indicate potential problems. These values are based on the discrepancy between observed and expected data using either Freeman-Tukey (Freeman-Tukey 1950) or deviance statistics (Schwarz *et al*. 2009). Second, we checked for model convergence using the Brooks-Rubin-Gelman statistic (BRG, Brooks and Gelman (1998). Values close to 1, or less than 1.2 were considered adequate.

The present version of BTSPAS does not account for incomplete sampling in capture efficiency estimates. This can lead to biased estimates depending on whether a) incomplete sampling occurred at both the marking and recapture site, b) the marked fraction was constant within each strata and c) marking or recapture was impacted by the incomplete sampling (Schwarz pers.com.). It is possible to "adjust" the number of marked, recaptured or unmarked fish under some circumstances to generate unbiased results. However, precision would be biased high in such situations since the uncertainty from any adjustments are not incorporated into the model. Details of when and how this was done are included in section 5.2. Uncertainty is incorporated into the estimate for strata with incomplete sampling only by excluding all recaptured and unmarked fish within the strata, leaving abundance to be imputed based on the spline.

Pooled Peterson Estimator

For the years or RST sites with inadequate numbers of marked or recaptured fish to use the Darroch ML model, we used the pooled Peterson Estimator (PPE) for a point estimates of abundance but without confidence intervals. Estimates using the PPE are often biased when capture efficiency varies through the survey period as has often been the case (Arnason *et al*. 1996) and generally overestimates the precision of the estimate (Schwarz *et al*. 2009).

Reach Level Abundance

To estimate smolt abundance originating in each mainstem reach (N_{reach}), we computed an estimate of mainstem smolts passing a RST at the downstream end of that reach, and then subtracted from this the estimate for the next RST upstream:

$$N_{reach\ 2} = N_{RST2} - N_{RST3} \tag{5.5}$$

$$N_{reach\ 3} = N_{RST3} - N_{RST4} \tag{5.6}$$

$$N_{reach\ 4} = N_{RST4} \tag{5.7}$$

where $N_{reach\ i}$ was the estimated abundance from reach *i* and $N_{RSTi,}$ represents the estimated number of mainstem fish passing an RST site. Note: due to the catch data input into the Darroch ML estimator, N_{RSTi} is estimated directly. However, BTSPAS generates estimates of both the total number of smolts ($N_{RST)}$ and unmarked smolts (U_{RSTi}). With BTSPAS, we used the estimated the number of unmarked fish passing each RST (U_{RSTi}) and then added the number of fish marked (M) upstream minus any marked mortalities (m) to estimate the total number of mainstem fish passing the RST ($N_{RSTi,}$). Both approaches result in the same estimate of N but including or not including marked fish in N is simplified with the latter approach. Note that when smolts are marked, transported upstream and released, they are not included in N for the trap where they were marked. Marked fish (M) are added to U for each trapping location as follows:

$$N_{RST2} = U_{RST2} + M_{RST3} + M_{RST4} - (m_{RST3} + m_{RST4}) \quad (5.8)$$

$$N_{RST3} = U_{RST3} + M_{RST4} - m_{RST4} \quad (5.9)$$

$$N_{RST4} = U_{RST4} \quad (5.10)$$

The key to estimating the abundance of smolts only originating from the Coquitlam River mainstem was that all off-channel smolts were marked, thus allowing them to be distinguished from mainstem smolts. Therefore, we assume that all unmarked smolts captured at mainstem sites are mainstem origin fish.

To compute 95% confidence intervals for $N_{reach\ 2}$ and $N_{reach\ 3}$, we summed variances for all relevant upstream RST or minnow-trapping mark-recapture estimates. For example, the 95% confidence interval for smolt estimates for reach 2 would be:

$$\pm 95\%\ CI\left(N_{reach\ 2}\right) = \pm 1.96\sqrt{Var\left(N_{RST2}\right) + Var\left(N_{RST3}\right)} \quad (5.11)$$

Since reach 4 is the uppermost reach, the variance of population estimates is not affected by the uncertainty of mark-recapture estimates for trapping sites upstream:

$$\pm 95\%\ CI\left(N_{reach\ 4}\right) = \pm 1.96\sqrt{Var\left(N_{RST4}\right)} \quad (5.12)$$

Coho and Steelhead smolt production for the Coquitlam River mainstem upstream of RST2 is simply N_{RST2} with a 95% confidence interval of:

$$\pm 95\%\ CI\left(N_{mainstem}\right) = \pm 1.96\sqrt{Var\left(N_{RST2}\right)} \quad (5.13)$$

The estimate for total smolt numbers for the Coquitlam River study area upstream of RST2 including the four off-channel sites was computed as:

$$N_{total} = N_{RST2} + N_{Off\text{-}channel} \quad (5.14)$$

with a 95% confidence interval of:

$$\pm 95\%\ CI\left(N_{total}\right) = \pm 1.96\sqrt{Var\left(N_{RST2}\right)} \quad (5.15)$$

5.1.1.5 Mark-recapture assumptions

We evaluated the assumption of population closure by plotting a frequency histogram of daily smolt catches for each weir or RST and then comparing the numbers of smolts captured at the beginning and end of the trapping period to captures during the peak of the migration. Very low catches at the tails of the trapping period relative to catches during the peak were taken as an indication that most smolts emigrated during the trapping period. We assumed 100% mark retention and 0% marking-induced mortality based on two earlier studies using similar marking procedures (Decker 1998; Decker and Lewis 1999). With respect to the assumption of equal capture efficiency for marked and unmarked smolts, we assumed marking did not change capture efficiency at the RSTs, but we did not test this directly. To do so would require that there be more than one potential recapture event for individual fish with similar effort for each trapping period (Seber 1982). In our study, individual fish may be recaptured at more than one RST site, but trapping effort is not equal among sites because the efficiency of each RST depends on its location. The steps taken to address potential differences in capture efficiency between marked and unmarked smolts are described in section 5.1.1.3. With respect to the assumptions of constant capture efficiency and proportions of marked to unmarked smolts over time, the use of a stratified mark-recapture design minimizes or avoids violations of these two assumptions by stratifying both the marking and recovery periods. We limited the time period during which capture efficiency and the proportion of marked to unmarked smolts were assumed to be constant within each weekly stratum (Appendix 5.1).

5.1.2 Chum and Pink fry enumeration

5.1.2.1 Downstream trapping

To estimate Chum and Pink fry out-migrant numbers, we relied on similar methodology to that employed by Cope (2002) on the nearby Alouette River. Prior to 2008 we used two incline plane traps (IPTs) to capture Chum and Pink fry. Beginning in 2008, a 1.3 m diameter RST was used in place of the IPTs (two RSTs were used in 2009). The substitution of an RST for the IPTs reduced cleaning and maintenance demands as well as fish mortality. The RST targeting Chum was operated in reach 2 at the same location as the two RSTs used to trap Coho and Steelhead smolts (RST2 site; Figure 5.1). The main difference between fry and smolt traps is that fry traps use a smaller screening that smolt traps (2.5 mm versus 12 mm, respectively).

5.1.2.2 Differential marking over time

To generate temporally stratified mark-recapture estimates, the trapping season was divided into weekly strata. Each week, catches of fry from 1-2 days were marked and released immediately downstream of RST3, approximately 3.2 km upstream of the RST2 trapping site. Releases were limited to the first half of the week to allow sufficient time for marked fish to pass the RST2 prior to the end of each week. Thus, all recaptures are assumed to occur within the stratum they are released in. This provided temporally stratified data without the need for different marks. This differed from the approach taken for Coho and Steelhead smolts in that marking was not continuous across all days and recaptures occurred over several subsequent strata.

We mass-marked fry by placing them in a solution of Bismark brown Y, a vital stain (Deacon 1963), and water (1:100,000 concentration) for one hour. Adequate oxygen levels within the solution were maintained using bottled oxygen and a flow meter. Fry were held in a live box and released at dusk to reduce predation. Mortalities prior to release were noted and subtracted from the count for each mark group. Mark loss was not assessed, but Deacon (1963) suggests that fry marked with Bismark brown are readily identifiable for at least 5 days following staining, which agreed with our own observations. Daily captures of Chum were individually sorted from other species (Coho, Chinook and Steelhead), counted and inspected for marks.

5.1.2.3 Population estimates

The population estimate and 95% confidence interval for Chum passing the RST2 site was computed using the same methodology as that for Coho and Steelhead (i.e., N_{RST2}; see section 5.1.1.4). This included the Darroch ML estimator for 2000-2018 and BTSPAS for 2018-2020 only (see section 5.1.1.4). The estimation process for Darroch ML and BTSPAS is simplified for Chum and Pink fry since all recaptures occur within the marking strata owing to the typically 1-2 days for marked fry to pass RST2. This is referred to as diagonal captures.

5.2 Results

5.2.1 Off-channel sites

In 2020, off-channel weirs were operational from March 11 to June 15 (Table 5.1), similar to most previous years. Trap operation has occasionally started in mid-February to monitor for early outmigration and as late as early April, as a consequence of logistical constraints related to the transition to a new contractor (2018). Traps operated continuously during this period with the exceptions of a one-day decommissioning for the Archery trap (April 20), and two-day decommissioning for the Or Creek and Overland traps (April 19-20). The impact of these outages was likely minimal considering that daily captures were relatively low for Coho and Steelhead smolts before and after the outages (Figure 5.3-5.4). There were no trap breaches and no indications smolts were able to bypass weirs without capture.

Table 5.1 Start and end dates of trap operation, trapping season length, trapping days and percent season trapped for the 2020.

Downstream RST trapping site	Start date	End date	Season length	Trapping days	% trapping
Reach 2 (RST2.2, chum, pink)	2-18	5-16	89	80	90%
Reach 2 (RST2.4, coho, steelhead)	3-15	6-20	98	89	91%
Reach 2 (RST2.5, coho, steelhead)	3-15	6-20	98	89	91%
Reach 3 (RST3, coho, steelhead)	3-15	6-20	98	89	91%
Reach 4 (RST4, coho, steelhead)	3-15	6-20	98	90	92%
Archery Pond	3-15	6-20	98	91	93%
Overland Ponds	3-15	6-20	98	91	93%
Or Creek Ponds	3-15	6-20	98	83	85%
Grants Tomb Pond	3-15	6-20	98	85	87%

Daily catches of Coho smolts at the off-channel weirs at the beginning of the trapping period were very low compared to catches during the peak of the migration at all off-channel trapping sites (Figure 5.3). Daily captures were relatively low by the end of trapping at all sites except Archery Ponds, where they peaked during the second to last week of trapping. This supports the assumption that trapping included the entire smolt migration from Grant's Tomb, Or Creek and Overland off-channel areas but less so for Archery Ponds due to the possibility substantial outmigration continued after trapping ended mid-June. However, even if this did occur, post-trapping outmigration from Archery Ponds is unlikely to have a meaningful impact on off-channel and total river abundance estimates. For these reasons, we consider that weir counts likely represent Coho smolt outmigration from off-channel sites during 2020. For Steelhead, daily captures were too low and sporadic to assess whether off-channel weir operation spanned the entire outmigration period (Figure 5.4). Observed mortality for Coho was 0-2% at the off-channel weirs. There were no Steelhead mortalities at the four off-channel weirs.

In 2020, an aggregate total of 1,680 Coho and 47 Steelhead smolts were captured at the downstream weirs as they outmigrated from the Overland, Or Creek, Archery Ponds and Grant's Tomb off-channel sites (Table 5.1). This is the second lowest outmigration for each off-channel habitat and for all combined since 2000, with 2019 as the lowest (Figure 5.5). It is unlikely that the lower counts were the product of smolts evading capture or a shorter than usual trapping period considering there was no indication of trap breaches and that traps operated over a similar period to past years.

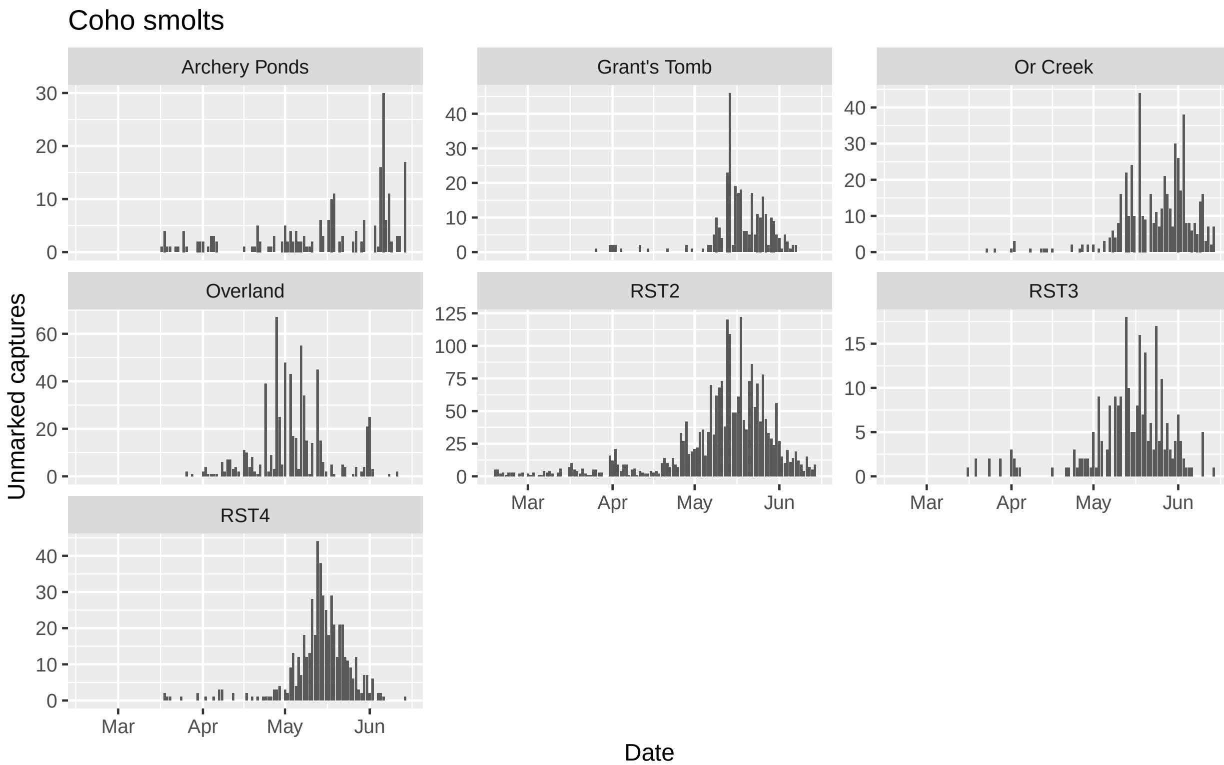


Figure 5.3 Daily catches of Coho smolts at downstream weirs in four off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2020. See Table 5.1 for start and end dates for individual trapping sites.

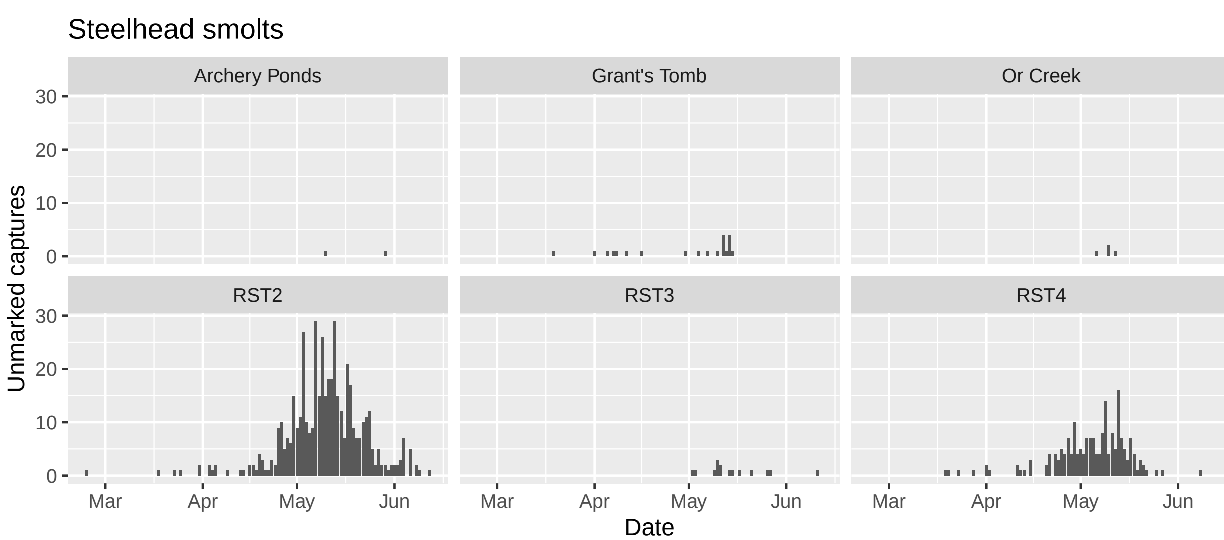
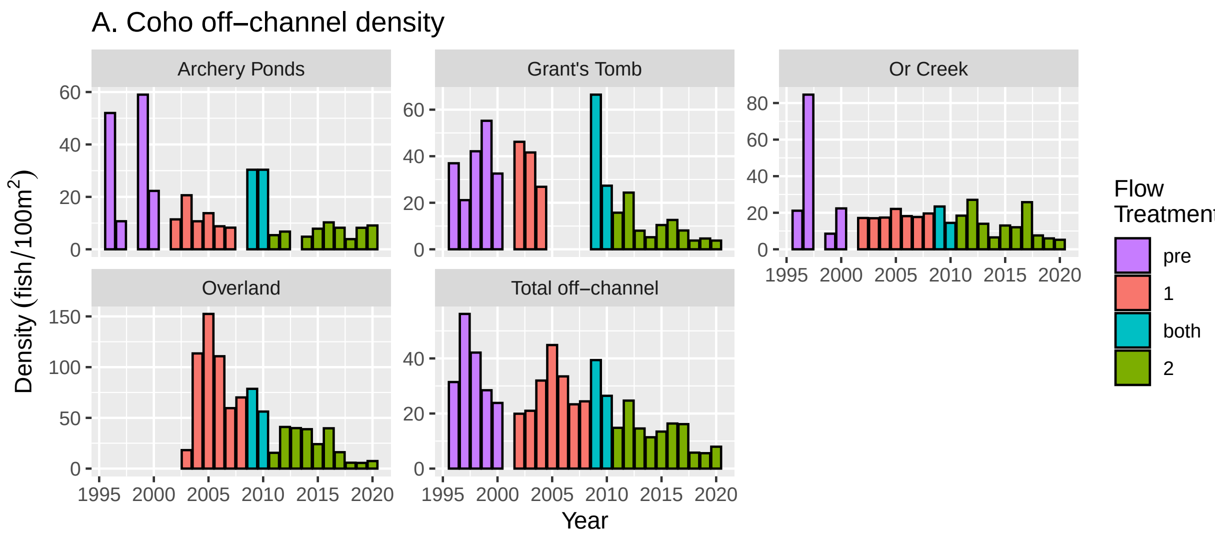


Figure 5.4 Daily catches of Steelhead smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2020. See Table 5.1 for start and end dates for individual trapping sites.

Table 5.2 Summary of estimated smolt numbers and densities by species in 2020 for four off-channel sites, for reaches 2-4 of the Coquitlam River mainstem, and for the total Coquitlam River mainstem including and excluding the off-channel sites.

Site	Length (km)	Area (m^2)	N smolts	CI (+/-)	CI (%)	Density (no./100m^2)	Density (no./km)
Coho							
Off-channel sites							
Grant's Tomb	-	3,300	300	-	-	9.1	-
Or Creek	-	13,336	500	-	-	3.7	-
Archery Pond	-	4,500	234	-	-	5.2	-
Overland Channel	-	8,700	646	-	-	7.4	-
Total Off-channel	-	21,136	1,680	-	-	7.9	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	3,099	1,839	59%	3.7	968
Reach 3, Coquitlam River	2.7	46,920	5,391	1,638	30%	11.5	1,997
Reach 4, Coquitlam River	1.6	19,200	1,073	208	19.4%	5.6	671
Total Mainstem	7.5	149,898	8,490	860	10%	5.7	1,132
Coquitlam R.incl. off-channel s	7.5	171,034	10,170	860	8%	5.9	1,356
Steelhead							
Off-channel sites							
Grant's Tomb	-	3,300	30	-	-	0.9	-
Or Creek site	-	13,336	6	-	-	0.0	-
Archery Pond	-	4,500	10	-	-	0.2	-
Overland Channel	-	8,700	1	-	-	0.0	-
Total Off-channel	-	21,136	47	-	-	0.2	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	3,796	813	21%	4.5	1,186
Reach 3, Coquitlam River	2.7	46,920	-	-	-	-	-
Reach 4, Coquitlam River	1.6	19,200	1,061	378	36%	5.5	663
Total Mainstem	7.5	149,898	3,789	813	21%	2.5	505
Coquitlam R.incl. off-channel s	7.5	171,034	3,836	813	21%	2.2	511
Chum							
Coquitlam R.incl. off-channel s	7.5	171,034	841,880	45,999	5%	492	112,251
Pink							
Coquitlam R.incl. off-channel s	7.5	171,034	197,838	70,570	36%	116	26,378

The mean weighted density of Coho smolts in the off-channel sites was 7.9 smolts/100 m^2 while Steelhead smolt density was 0.2 smolts/100 m^2 (Table 5.2). In terms of areal density, trends across years and flow treatment periods for Coho were flat for Or Creek Ponds and declining for Archery Ponds, Grant's Tomb, and Overland (Figure 5.5a). The density of Steelhead has been variable at the off-channel sites, with no clear trends across years (Figure 5.5b). Across all years, 17%-68% for Coho and 0.3%-9% for Steelhead of production for the entire Coquitlam River upstream of RST2 originated from the constructed off-channel sites.



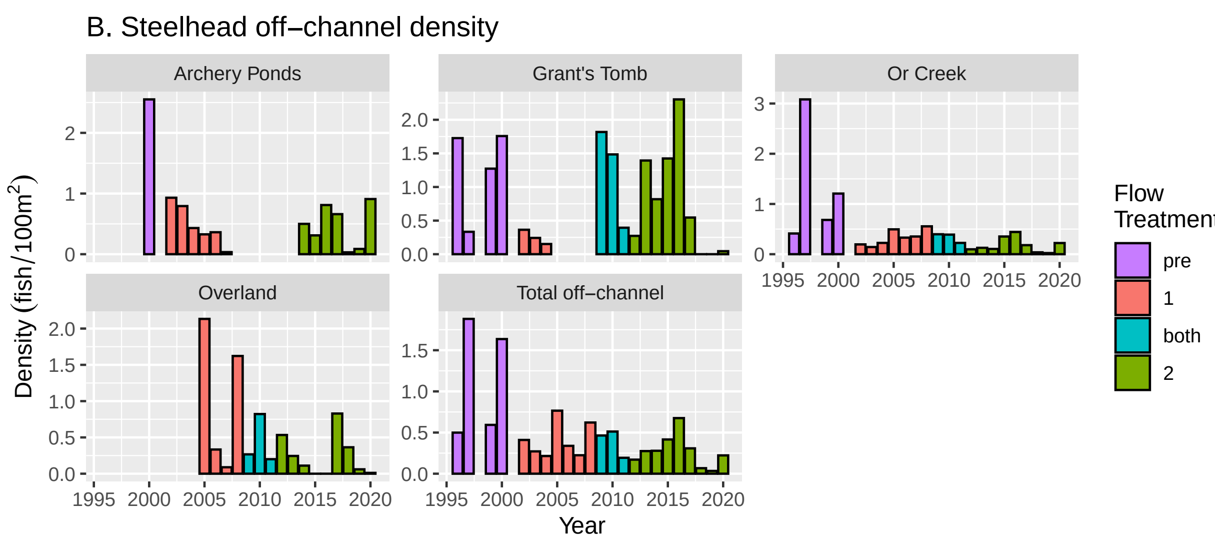


Figure 5.5 a-b Areal density of Coho and Steelhead smolt (smolts/100m2) in four constructed off-channel habitats along the Coquitlam River and for all four combined (Total) previous to Treatment 1 (1996-2000), Treatment 1 (2002-2008), when smolt cohorts reared under both treatments (2009) and Treatment 2 (2010-2020). Years with zero fish represent those when the off-channel habitats were not in operation or were not monitored.

5.2.2 Coquitlam River mainstem

5.2.2.1 Discharge and trap operations

In 2020, fry trapping at RST2 began February 18 and smolt trapping began March 15 at all sites. Appendix 5.1 lists the dates traps were installed and removed, and the start day of each marking period. Discharge in the Coquitlam River during the spring trapping period was variable and was similar to other Treatment 2 years, with the exception of 2018 when discharge from the LLO was increased from 3.5 m^3/s (April) and 2.9 m^3/s (May) to 8 m^3/s as part of the Kwikwetlem Sockeye Restoration Program (KSRP). Daily mean flows exceeded the 12 m^3/s limit for Chum fry trapping on seven occasions and exceeded the 20 m^3/s limit for Steelhead and Coho smolt trapping on four occasions (Figure 5.6). Due to flows exceeding the trap capacity, fry trapping operations at RST2 were suspended for one day on one occasion, for two days on two occasions and for four days on one occasion for a total of nine lost trapping days during the 89-day monitoring period. Smolt trapping at all three RST sites was near continuous. Trap operations were suspended for two days at RST2 and RST3 and one day at RST 4 during March 29-30. The description of how the missed trapping days were accounted for in the population estimates is included in Sections 5.2.2.3-5.2.2.5.

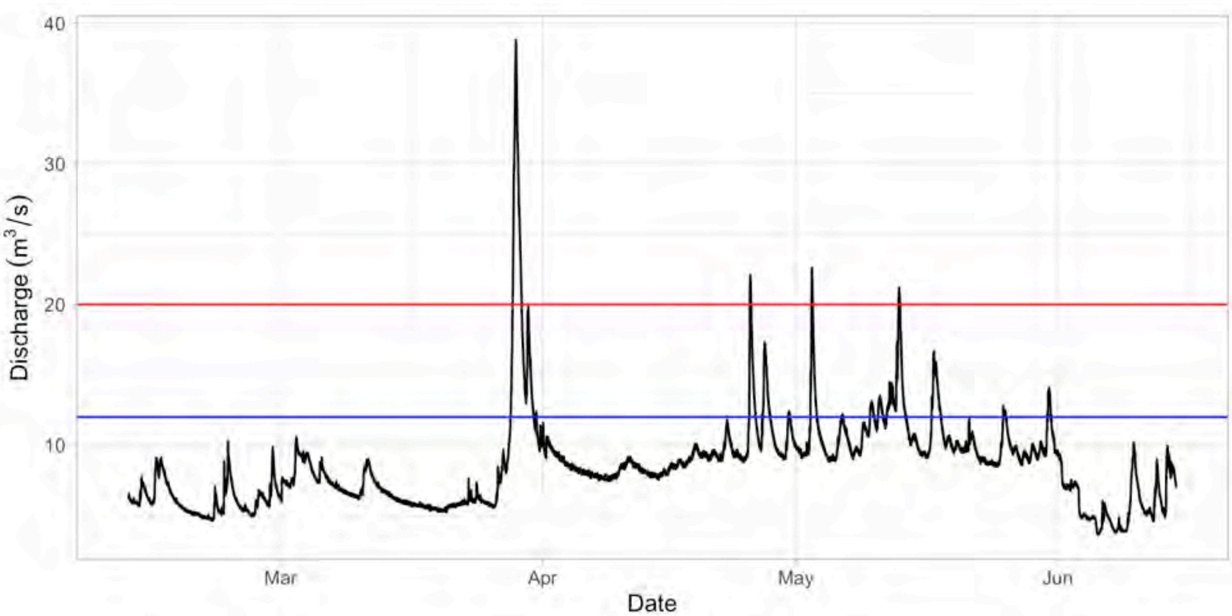


Figure 5.6 Mean daily flows in Coquitlam River at Port Coquitlam during the smolt and fry trapping period in 2020. (Water Survey of Canada, stn. 08MH002). The maximum discharge for operating the smolt traps is shown by the red line (20 m^3/s) and for the fry trap by the blue line (12 m^3/s).

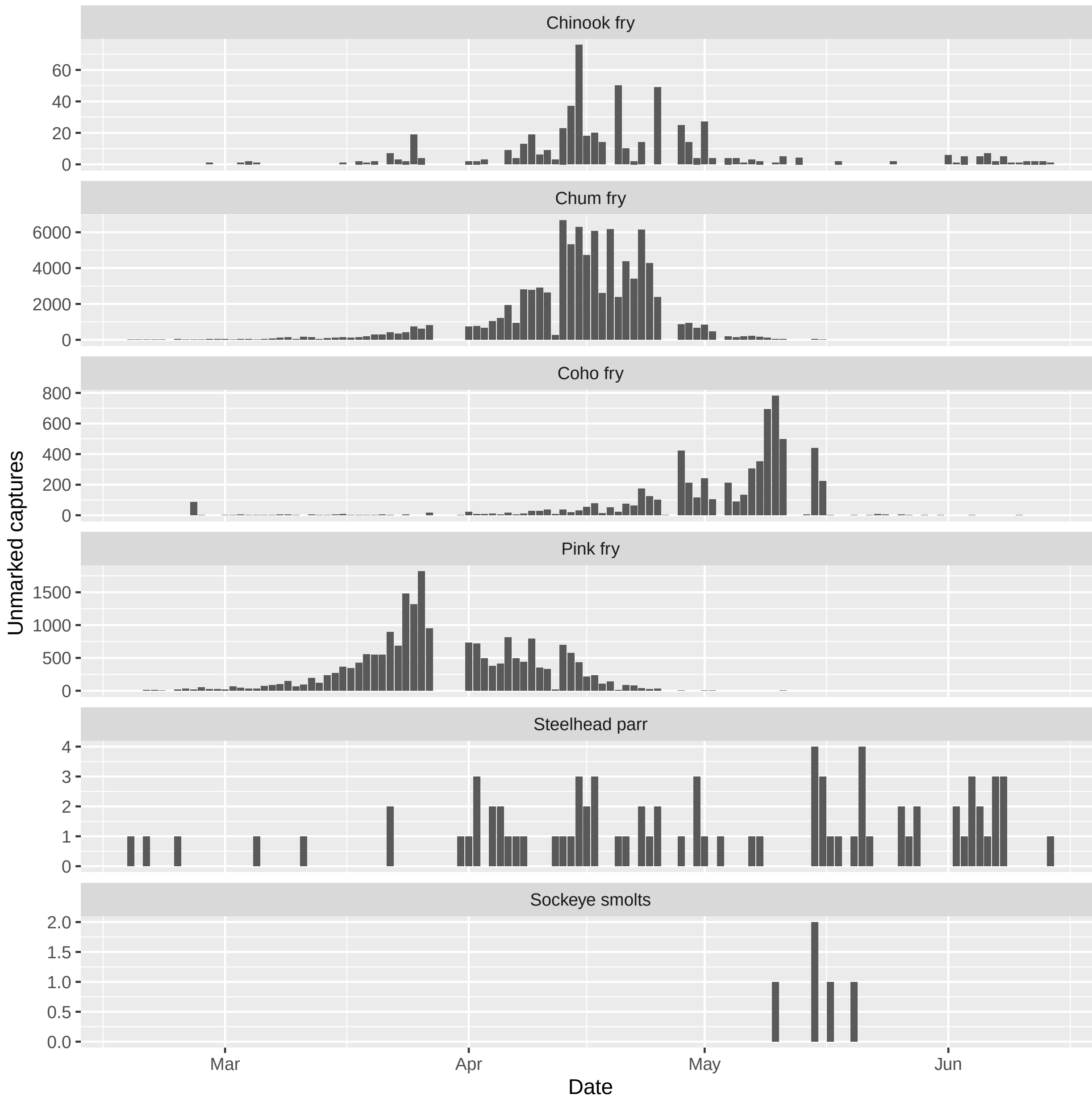


Figure 5.7 Daily catches of Chinook fry, Chum fry, Coho fry, Pink fry, Steelhead parr and Sockeye smolts at the RST2 trapping site in Reach 2 in the Coquitlam River in 2020. See Table 5.1 for start and end dates of downstream trapping.

Overall, observed mortality at the RSTs was 1.0% for Coho and 1.2% for Steelhead smolts, 4.9% for Pink fry and 1.7% for Chum fry. This is similar to 2018 rates of 1.2% for Coho and 1.0% for Steelhead smolts, and lower for Chum at 10% for Chum fry, though the connection to lower the flows in 2020 than 2018 is less clear. The higher dam releases during April and May 2018 increased the number of days compared to other Treatment 2 years traps were operating near their upper flow limit based on crew safety as well as fish health. At higher discharges, fry and smolts are at increased risk of trauma as they are captured in the RST and while being held in the trap box.

Daily catches of all species at the beginning and end of the trapping period were very low compared to catches during the peak of the migration (Figures 5.3-5.4, 5.7). This suggests that population closure was largely met for all species.

Stratified mark-recapture data (catch tables) used to generate estimates of the number of Coho and Steelhead passing each RST site are shown in Appendix 5.3.

5.2.2.2 Coho

Table 5.1 lists estimates of mainstem smolt production for Reach 2-4 individually, as well as for all three reaches combined, with and without off-channel sites.

Table 5.3 Differences in capture efficiency (proportion of marked smolts that were recaptured) for Coho and Steelhead from off-channel sites and the Coquitlam River mainstem at three rotary screw traps (RSTs) sites in the Coquitlam River mainstem in 2020. Stratified marking periods were pooled prior to testing (see Equation 5.1). Equal capture efficiency for mark groups was tested using Fisher's exact test. $P < 0.05$ indicates a significant difference in capture efficiency.

		Capture efficiency		
Species	Recapture site	Mainstem mark group	Off-channel mark group	Fisher's exact test (P)
Coho	RST 2	0.31	0.29	0.29
Coho	RST 3	0.04	0.05	0.20
Coho	RST 4	0.49	0.38	0.00
Steelhead	RST 2	0.14	0.04	0.12
Steelhead	RST 3	0.01	0.00	0.59
Steelhead	RST 4	0.18	0.19	0.89

At RST4, capture efficiency was significantly different between the off-channel and mainstem mark groups (49% and 38%, respectively, Fisher's exact test, $P<0.01$, Table 5.3). For this reason, we used only the mainstem mark group for the estimate of mainstem smolts passing RST4. Using BTSPAS, we estimated that 1,073 Coho smolts (95% CI: ± 19%) passed the reach 4 trapping site (Table 5.4), which also represents the smolt production for this reach. Note precision from BTSPAS is represented by 95% credible intervals, which in this report are considered analogous to 95% confidence intervals from the Darroch ML estimator used prior to 2018. Precision of the 2020 estimate is below the mean for Treatment 2 prior to 2018 (± 8%) even though there were relatively high capture efficiency and numbers of recaptured smolts. We expect lower precision using BTSPAS since it includes the additional uncertainty when estimating abundance during strata with low recaptures, which wasn't included in the Darroch ML estimator as well as when incomplete sample requires spline-fitting for the estimates.

Table 5.4 Summary of estimated numbers of Coho, Steelhead and Chum smolts passing the three RST trapping locations (not reach estimates) in the Coquitlam River mainstem in 2020. Mark group indicates the location where fish were initially captured and marked. Also shown are numbers of marked (M), recaptured (R) smolts, unmarked captures (U) and estimated capture efficiencies (R/M).

Species	Site	Mark group(s)	M	R	U	Capture efficiency	N mainstem smolts	CI (+/-)	CI (%)
Coho	RST 2	all	2,310	680	2,008	29%	8,490	860	10%
	RST 3	all	1,463	72	228	5%	5,391	1,625	30%
	RST 4	mainstem	503	248	494	49%	1,073	208	19%
Steelhead	RST 2	all	669	92	424	14%	3,789	813	21%
	RST 3	-	-	-		-	-	-	-
	RST 4	all	200	36	187	18%	1,061	378	36%
Chum	RST 2	RST 2	7,836	17,201	1,102	220%	841,880	45,999	5%
Pink	RST 2	RST 2	20,430	2,157	83,918	11%	197,838	70,570	35.7%

At RST3, capture efficiency using marked off-channel and mainstem Coho smolts were not significantly different (4% and 5%, respectively, Fisher's exact test, P=0.20; Table 5.3). Considering this, we used both mainstem and off-channel mark groups to estimate that 5,391 ± 30% mainstem smolts passed RST3 (Table 5.4). Once smolts from reach 4 were subtracted, we estimated the production from reach 3 to be 4,318 ± 38% smolts. The moderate precision of the reach 3 estimate is a product of lower recaptures at RST 3 combined with the reduction due to subtracting one estimate from another.

At RST2, capture efficiency was not significantly different for the mainstem mark and off-channel mark groups (31% and 29%, respectively, P =0.29; Table 5.3). Using both mark groups, we estimated smolt production from reach 2 at 3,099 ± 59% (Table 5.2), once the contribution from reaches 3 and 4 were removed. We estimated smolt production for the entire mainstem upstream of RST2 at 8,490 ± 10% smolts and including off-channel areas was at 10,170 ± 8%. Precision of the mainstem estimate is similar to past Treatment 2 years (mean = ± 10%). This is due to adequate recaptures during strata spanning peak migration.

Average estimated Coho smolt density in the Coquitlam River mainstem was 5.7 smolts/100 m^2 and 5.8 smolts/100 m^2 including the off-channel sites. Areal Coho density was three-fold higher in reaches 3 (11.5 smolts/100m^2; Table 5.2), than in reach 2 (3.7 smolts /100 m^2) and two-fold higher than reach 4 (5.6 smolts/100m^2). The precision of estimates for individual reaches ranged from ± 36% for the estimate for reach 4, to ± 59% for the smolt estimate for reach 4 (Table 5.2). This is similar to 2018 and 2019 but lower than many previous years. This is still in the range to provide useful information for evaluating flow effects. The lower precision for 2018-2020 are a product of the way BTSPAS includes the uncertainty when abundance for a stratum with missing mark-recapture data is based on fitting the spline and during strata with few recaptures. These uncertainties existed prior to 2018 but Darroch ML estimator did not have the capacity to estimate it.

Abundance estimates across all years and reaches for Coho are reported in Appendix 5.4. The analysis of abundance estimate trends across all study years and low treatment periods is included in Section 6.

5.2.2.3 Steelhead

At RST4, we used both mark groups to estimate that 1,061 ± 36% Steelhead smolts passed RST4 (Table 5.4). We used both mark groups since capture efficiency was not significantly different (18% and 19%, respectively, P =0.89; Table 5.3). The lower capture efficiency of recent years continued in 2020 (18%, Table 5.4), and is reflected in the moderate precision of the estimates. Since recaptures at RST4 depend on the number of smolts marked at this site, released upstream and then recaptured, low capture efficiency reduced both the number marked and the proportion of those recaptured. The capture efficiency was similar to 2019 (13%) and higher than for 2018 (2%), but far lower than other prior years (30-60%). The increased capture efficiency from 2018 was due to a return to using the larger trap (diameter = 2.4m) and lower dam discharge.

The lower catch at RST4 also reduces the precision of the RST2 and RST3 estimates by reducing the number of marked fish available for recapture at the downstream traps and the relationship between the number of recaptures and precision. The effect would be largest for RST3, which depends entirely on RST4 for all marked fish, and less so for RST3 since only 25-50% of marked fish are from RST4.

At RST3, only two of the 214 Steelhead smolts marked at RST4 were recaptured. Due to this, we are unable to estimate the number of Steelhead smolts passing the trap. Capture efficiency at RST3 decreased from 15-20% to 1% in 2020. The continued low capture efficiency since 2018 is likely a product of the move to a new location that lacks the well-defined thalweg that concentrates fish within a portion of the channel. The RST3 site was moved approximately 200 m upstream due to changes in site access. The channel width and depth at the new site was considered insufficient for traps larger than 1.5 m.

At RST2, capture efficiency was not significantly different for mainstem and off-channel mark groups (14% and 4%, respectively, P =0.12; Table 5.3) but as with previous years, this was largely a result of insufficient power to detect a difference. We used both mark groups for the estimate that 3,789 ± 21% mainstem Steelhead smolts passed RST2. This also represents the estimate of mainstem production for the Coquitlam River upstream of RST2. When we included Steelhead from off-channel habitats, the estimate for all areas upstream of RST2 was 3,836 ± 21% (Table 5.2). Precision is at the high end of the range during Treatment 2 prior to 2018 (range: ± 9% to 38%) and a large improvement from 2018 (± 76%). This is due to sufficient numbers of recaptures during the period when the majority of outmigration occurred.

With only two lost trapping days that were outside of the peak migration period, we did not flag any strata as 'bad', which triggers spline fitting in place of missing data, as this would have minimal impact on the mainstem estimate.

Average Steelhead density in the Coquitlam River mainstem was 2.5 smolts/100 m^2 and 2.2 smolts/100 m^2 in the Coquitlam River including the off-channel sites (Table 5.1). Smolt density in reach 4 was considerably higher than the average of the entire mainstem as has been the case during Treatment 2 (5.5 smolts/100 m^2, Table 5.2)

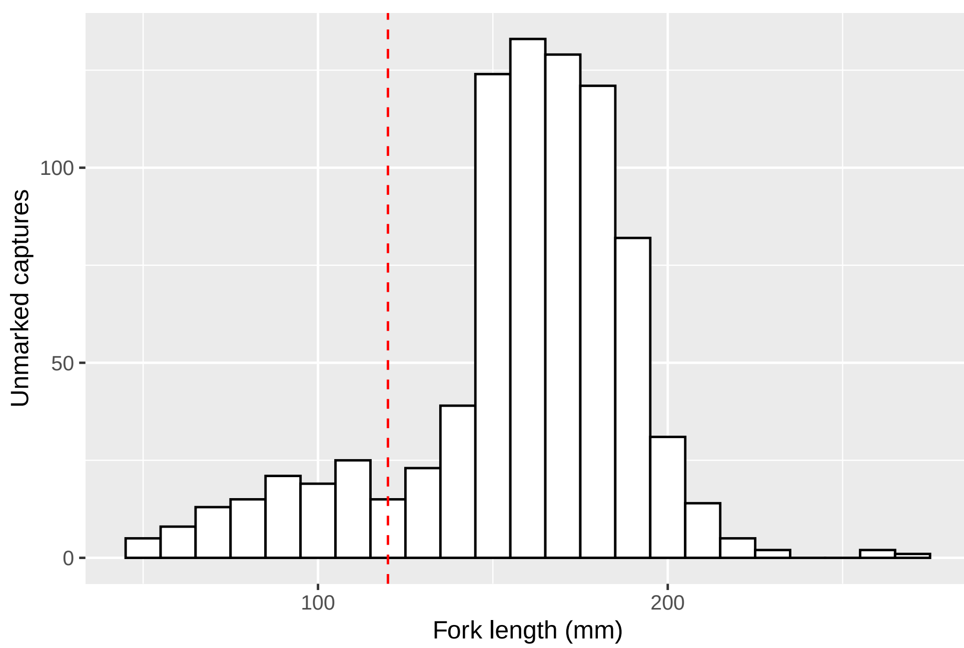


Figure 5.8 Length-frequency histogram for unmarked Steelhead captured in the Coquitlam River in 2020 (data pooled for all trap sites).

We assumed all Steelhead with a fork length of 120-230 mm to be smolts. As in previous years, 120 mm corresponded to the minima between two defined modes representing age-1+ and age-2+ and older juveniles, respectively (Figure 5.8). This was corroborated by scale samples collected for Steelhead in this size range during 2005-2018 (Appendix 5.3). Scale analysis of 789 individuals indicated a broad overlap (130-188 mm) in the absolute ranges in fork length for age-2 and age-3 smolts, but most smolts greater than 160 mm in length were age-3 (Appendix 5.5). Age-4 smolts were also present in the scale sample, ranging in length from 171 mm to 222 mm. Age-4 smolts comprised 5%-10% of smolts 165-194mm fork length and 30%-60% of smolts 195-220mm. To estimate Steelhead adult-to-smolt survival for the 2005-2017 escapement years (the broods of later escapement have yet to smolt), we used age-2/age-3 length cut-offs of 160-170 mm (depending on the year) to estimate the proportions of age-2 smolts in the smolt populations. The proportion of age-2 smolts ranged from 50%-67% among years. Fish larger than 230 mm had the general appearance of resident Rainbow Trout (i.e., cryptic colouring, heavily spotting) as opposed to smolts (bright silver), and some were sexually mature.

Table 5.5 Captures of Chinook and Coho fry, Steelhead parr and Sockeye/Kokanee during 2020 at RST 2-4.

Site	Chinook smolts/fry	Coho fry	Steelhead parr	Sockeye smolts
RST 2	571	6040	100	5
RST 3	4	9	10	0
RST 4	6	6	5	7

We assumed that age-1+ Steelhead (fork length < 120mm) will outmigrate after one or two additional winters in the Coquitlam River. In 2020, we captured 100, 10 and 5 Steelhead parr at RST2-4, respectively (Table 5.5). While Steelhead parr (fork length < 120mm) are caught at the RSTs, it has been unclear whether this reflects downstream movement, representing unaccounted for production, or if caught parr were intercepted during local movement confined within a reach. The proportion of unmarked captures of Steelhead < 120mm at RST2 since 2011 ranged from 21% to 52% of total captures (Table 5.6). To evaluate this, we marked Steelhead parr at all three mainstem trapping sites. Downstream movement would be indicated by recaptures of parr at traps downstream of their marking location where as recaptures at the marking trap would indicate primarily local movement. Of the 25 parr marked at RST4, one was recaptured at RST2 and another at RST4. These low recaptures at each site is too low to distinguish whether parr are primarily moving downstream or remaining in the vicinity of the trapping site.

Table 5.6 Percent of all unmarked juvenile Steelhead captures that were considered Age-1 parr based on fork length (< 120mm) at RST 2-4 in the Coquitlam River. 120mm fork length has been the minimum length to be considers smolts since 2012.

	Year								
Location	2011	2012	2013	2014	2015	2016	2017	2018	2019
RST2	38%	33%	52%	40%	43%	21%	37%	30%	27%
RST3	20%	20%	29%	42%	33%	29%	23%	23%	63%
RST4	15%	13%	16%	9%	13%	11%	5%	23%	11%

Tests to measure whether capture efficiency differed between VIE from fin clips were not repeated in 2019 or 2020. Tests completed in 2018 found no indication that a switch to marking smolts using VIE from fin clips impacted the capture efficiency. For Coho, capture efficiency estimates were nearly identical for both marking methods (clipped = 0.33, VIE = 0.32; Appendix 5.4). The difference in capture efficiency was also very small for Steelhead (clipped = 0.21, VIE = 0.18). The Chi square test of independence also support the finding that mark type did not impact capture efficiency (Chi square test, Coho $p = 0.63$, Steelhead $p = 0.55$; Appendix 5.6). Comparing capture efficiency between the two marking methods reflects the sum total of all ways that marking methods could impact the number of reported recaptures including: survival between marking and recapture; behavioural changes impacting catchability; mark loss and mark detection.

Abundance across all years and reaches for Steelhead are reported in Appendix 5.4. Analysis of abundance trends across years or by flow treatment period are included in Section 6.

5.2.2.4 Chum and Pink

Chum and Pink fry were trapped from February 18 to May 16 at the RST2 location in reach 2 (Appendix 5.1). For Chum capture efficiency varied from 5.8%-14.2% (Appendix 5.3), and averaged 11% (Table 5.4). Out of the four trap outages, only one occurred when there were sufficient daily captures to mark, and this was prior to peak outmigration. We accounted for this by flagging unmarked captures as missing for strata 7. In this case, BTSPAS used spline-fitting to estimate the number of unmarked captures. During 2020, an estimated 842,880 ± 5% Chum fry (Table 5.2) migrated past the RST2 trapping site. The estimated density in the mainstem of the Coquitlam River for Chum was 112,251 fry/km or 562 fry/100 m^2 (Table 5.2).

Pink capture efficiency varied from 8.1%-20.6% (Appendix 5.3), and averaged 14% (Table 5.4). The estimated density in the mainstem of the Coquitlam River for Pink was 26,378 fry/km or 116 fry/100 m^2 (Table 5.2). The 4-day trap outage had a relatively large impact on unmarked captures since it occurred just after peak outmigration. Similar to Chum, we accounted for this by using spline-fitting to estimate unmarked captures during stratum 7. While this improved the accuracy of the estimate, it reduced precision due having to estimate migration with incomplete information for a substantial proportion of the total cohort outmigrating. During 2020, an estimated 197,838 ± 36% Pink fry (Table 5.2) migrated past the RST2 trapping site. This includes the uncertainty from using spline-fitting to estimate the number of fry passing the trap site during the 4-day outage.

We consider this a relatively accurate estimate even with the four day outage. BTSPAS spline-fitting reduced bias and incorporated the uncertainty from trap outages. Trap operation was also similar to most prior years and included the majority of the outmigration period (Figure 5.4).

5.2.2.5 Sockeye/Kokanee

In 2020, catches of Sockeye smolts at RST 2-4 were 5, 0 and 7; respectively (Table 5.5). No mortalities occurred at RST 2-4. Sockeye captured each year at all traps during Treatments 1 and 2 have ranged from 10's of fish to several hundred (2005-2007). 2020 captures are at the low end of this range. Given the limited number of fish captured, no attempt was made to mark fish or generate population estimates.

5.2.2.6 Chinook

In 2020, 55 Chinook juveniles were captured by the two smolt traps at the RST2 location and another 497 were captured in the fry trap at RST2 (Table 5.5). Chinook captured in the smolt traps had a fork length range of 40-85 mm, suggesting most were age-0 fish. As in past years, there was no attempt to distinguish between the age-classes and they were not included in the mark-recapture program, which would have been necessary to estimate the number of outmigrants.

5.2.2.7 Coho Fry

In addition to Coho smolt production, an additional 6,093 Coho fry were captured at the RST2 fry trap in 2020 (Table 5.5). Fry catch has ranged from 2,200-30,000 since 2008. As with previous years, there was no attempt in 2020 to include them in the mark-recapture program or estimate the number of outmigrants.

5.3 Discussion

Tables 6.1a and 6.1b in the next section provide estimates of annual escapement, juvenile standing stocks, and smolt production for the Coquitlam River upstream of RST2, along with survival rates from one life stage to the next.

5.3.1 Assumptions of the study design

We assumed all two year and older Steelhead (120-230 mm in length) were smolts. However, there was probably a small proportion of smaller Steelhead in this size range, which were likely parr that were dispersing to downstream habitats, and would ultimately be smolting at age-3, or even age-4 (Withers 1966). As well, some of the larger fish in this size range were likely mature residents. In past years we excluded a small number of fish that the trapping crew identified as being resident Rainbow Trout based on cryptic colouring and heavy spotting as opposed to the typical silvery colouration of a smolt. A number of these fish were confirmed to be sexually mature males or females as they released milt or eggs when light abdominal pressure was applied. However, the vast majority of Steelhead that were captured and recorded as smolts were silvery in appearance (e.g., >97% in 2002 and 2005 when physical characteristics were categorized for all Steelhead captured). Moreover, the average fork length of Steelhead smolts during 1996-2020 varied from 154 mm to 171 mm, which is in good agreement with the expected mean length at ocean entry for Steelhead stocks in the North Pacific (160 mm; CV = 10%-15%; Burgner *et al*. 1992).

We have assumed that captures of Steelhead parr represent within-river movement rather than outmigration yet this has not been confirmed during this monitoring program. If Steelhead do exit the study reaches as parr, this would represent unaccounted for juvenile production. The low number of parr marked at RST3 and 4 did not provide sufficient information about the validity of this assumption.

For Chum and Pink estimates, the primary assumption was the capture efficiency estimated from the 1-3 days per weekly strata reflects the capture efficiency during the remaining days, when no marked fish passed the trap. To minimize the extent of this assumption, the number of release days was increased from one to 2-3 days per weekly strata period from the single release prior to 2019.

5.3.2 Reliability of estimates and implications for the flow experiment

Results to date suggest that, for the most part, the downstream trapping program in its current form is adequate for the purposes of generating sufficiently precise and reliable estimates of smolt and fry abundance for all species to meet COQMON-07 objectives.

Higgins *et al*. (2002) demonstrated that the statistical power to detect differences in fish production in the Coquitlam River under different flow regimes was strongly influenced by the precision of annual estimates of smolt abundance. Specifically, they showed that for a simulated 12 year-long experiment, power (β) decreases significantly over a range of increasing observation error ($\sigma_{sm,o}$ in their paper) from a high of β: 0.6-0.7 with no observation error to a low of β: 0.3. While this falls short of the goal of 'moderate' power (β > 0.8), the study suggested that there was relatively little drop in power at smolt observation error levels up to 0.1-0.2, which expressed as a 95% confidence interval are ±20% to ±40% of the estimate.

The precision of the 2020 Coho smolt abundance estimate in the Coquitlam River mainstem was high (95% CI: ±10%) and similar to estimates for 2000-2017 (95% CI: ±6%-14%). It is well above the theoretical optimal value of ±20% predicted by Higgins et al (2002). Precision of the 2020 mainstem Steelhead smolt estimate was moderate (95% CI: ±21%) compared with estimates since 2000 (95% CI: ±11%-37%) and near the predicted theoretical optimum. For both Steelhead and Coho smolts, this was the result of maximizing the number of recaptures at RST2 by marking smolts at RST2-4 for Steelhead and RST3-4 for Coho combined with relatively high capture efficiency at RST2. It was also due to the near continuous trap operation, which minimized the reliance on spline-fitting to estimates abundance during periods of incomplete trapping.

The precision of population estimates for Chum fry at the RST2 in 2020 was high (95% CI: ±5%) compared with previous years using rotary screw traps (95% CI: 6%-18% during 2008-2018). This includes the uncertainty from using spline-fitting to estimate abundance during the period traps were not operating. The reduction in precision from spline-fitting was minimal in 2020 since trap outages were not during the period when most Chum were outmigrating. Unlike prior years, when the uncertainty from missed trapping was not included, estimates 2018 onward likely include a better representation of their true uncertainty.

The precision of Pink fry estimates in 2020 (95% CI: ±36%) was far lower than past rotary screw trap estimates (95% CI: ±12-19% during 2008-2017) but similar to 2018 (95% CI: ±40%). In 2020, this was largely to consequence of having to rely on spline-fitting to estimates abundance during the trap outage that coincided with the high outmigration period. While this improved accuracy, spline-fitting does not have the same precision as capture data.

6 Fish Productivity during Treatment 1 and 2

The COQMON-07 uses a Before-After (BA) experimental design where fish metrics (i.e. abundance, survival, productivity) measured under flow Treatment 1 conditions are compared to those under Treatment 2. The expectation of the Consultative Committee (CC) was that increased base flows (all months except June and July) would benefit spawning and/or juvenile rearing of target species (Table 1.1, Figure1.1). The CC chose an empirical approach to evaluating the benefits of either flow regime. Thus, the primary objective of monitoring is to address the management question:

What are the fisheries benefits of Flow Treatment 1 (2FV) and Treatment 2 (STP6)?

The analysis approach to evaluate the fisheries benefits for Coho and Steelhead relies upon overall smolt production and to a lesser degree on standing stock estimates and survival from one age-class to another. For Pink and Chum salmon, we use egg-to-smolt survival (productivity) as the main metric for the evaluation. We break the evaluation process into two questions.

1. Did fish production or productivity change between Treatment 1 and 2, and by how much? We used the 2-tailed T-test ($\alpha<0.05$) to test for a statistically significant change in Steelhead and Coho smolt yield. To estimate the amount that Steelhead and Coho smolt yield changed, we used the mean and standard deviation of abundance during each treatment period to generate a posterior distribution of the mean difference between treatments. From this, we estimated the probability that at least a 10%, 20% or 50% change in abundance occurred. Posterior distributions for mean abundance during each treatment as well as the mean difference between treatments were estimated using JAGS (Plummer 2012) called using the BEST (Kruschke 2013) from the "R" statistical package (R Development Core Team 2009). To evaluate whether Chum and Pink Salmon productivity changed between treatments, we used the Analysis of Covariance (ANCOVA) to test whether the flow treatment was a statistically significant predictor of productivity ($\alpha<0.05$).
2. How much of a change was the consequence of the flow treatments versus other factors? For this, we compared the amount of change in abundance that occurred in the Coquitlam River compared to the abundance changes that occurred over a similar time period at other comparable rivers that did not undergo flow manipulations. These "control" rivers reflect abundance changes due to factors unrelated to the change from Treatment 1 to Treatment 2.
3. What were the main factors impacting freshwater survival? For this we compared the effectiveness of Ricker stock-recruitment models with and without a covariate to explain year-to-year changes in juvenile abundance. Covariates include flow treatment, season specific flow metrics and for Coho, stranding estimates. With this, we can examine the influence of stock size (e.g. adult spawners), density dependent effects, and in combination with the covariate effects (e.g. flow treatment). This allows for comparisons of the models using the deviance information criterion (DIC). DIC is similar to the Akaike information criterion used to compare support

for different models (Burnham and Anderson 2002) but is more compatible with the hierarchical modeling method used in this analysis. In this context, models differ in the variable used to explain the fish data and whether one or two variables is included. Models with more variables may fit the data better but variable estimates will be less precise. The most parsimonious model from a group of models is the one which exhibits the best trade-off between fit and precision, and will have the lowest DIC value. Models with ΔDIC >2 are considered to have strong support, 2-7 moderate support and >7 low support. We also evaluated the overall effectiveness of the model by the amount of variation explained by a model (R^2). This is important since the model with most support (lowest DIC value) may not be a useful model if it explains only a small amount of the variation. For example, a model that explains 80% of the variation is more useful than one that explains 10%. To evaluate the covariate effects on survival, we fit the following covariate Ricker stock-recruitment model to the data,

$$R_t = S_t \cdot e^{\alpha+\beta\cdot S_t+\gamma\cdot X_t} \qquad \text{Eq 6.1}$$

where R_t is the abundance of one life stage (e.g., outmigrating Chum fry) in year t, S is the estimated abundance of the previous life stage (e.g., Chum spawners), α is the log of the maximum survival rate when there are no density or flow effects (productivity), β is a density-dependent effect, and γ is the effect of flow covariate X_t. The product of γ and X_t therefore represents the vertical shift in the stock-recruitment curve in log space in year t due to the value of the flow covariate in that year. As X_t is a standardized annual covariate value ($X_t = \frac{x_t - \mu}{\sigma}$), this formulation results in a base recruitment curve at the mean level of the covariate value, since the standardized value would be 0 in this case (thus $\exp(\gamma\cdot X_t) = 1$). A variety of flow covariates were computed using mean hourly discharge records at the WSC Port Coquitlam gauge.

The WUP did not specify the amount of change in fish production or productivity to conclude that a meaningful amount had occurred. The CC predicted increases in abundance ranging from 10%-100% (Higgins *et al.* 2002). The expectation of the study was that it would have an 80% probability of detecting a 50% change in abundance, but this was not to say that smaller changes would not be of interest to the CC and in decisions about future flows.

6.1 Treatment 1 and 2 flows

Treatment 1 and 2 discharges are based on releases from the low level outlet gates (LLOG) at Coquitlam Dam. The LLOG releases for Treatment 2 were categorized as either targeted or minimum discharge. Releases are based on targeted levels other than during periods of limited reservoir capacity or inflows based on a set of rules set out in the Water Use Plan. Generally, targeted releases were higher throughout the year compared with Treatment 1 with the exception of June and July, which were lowered slightly (Table 1.1). Minimum release levels were still higher than Treatment 1 minimums for October to February but similar to Treatment 1 levels for March to September.

Based on flows measured at the Water Survey of Canada station at Port Coquitlam, the increased release from the LLOG during Treatment 2 resulted in higher minimum flows for all months but June-August (Figure 6.1). The absence of an increase in minimum flows for August reflects the frequency that LLOG releases were based on minimum levels rather than on targeted levels. Mean discharge increases were most pronounced during February-April and September-October. This timing corresponds to periods with increased targeted releases as well as relatively low inflows from tributaries below the dam. The difference between minimum and mean discharge was most pronounced during late fall and winter when the contribution from tributaries and spill events represents a large portion of total flow. For the months with higher minimum flows, flows were also more stable during Treatment 2 (Figure 6.1).

Another notable difference between Treatment 1 and 2 was the increase in rampdown events, which occurred almost entirely as a consequence of the monthly change in releases from the LLOGs. While the number of unscheduled rampdowns following higher flow events was similar for each treatment, there were from three to six scheduled rampdowns each year during Treatment 2 while none of comparable size occurred previously (MacNair 2018).

5. Smolt and Fry Outmigration

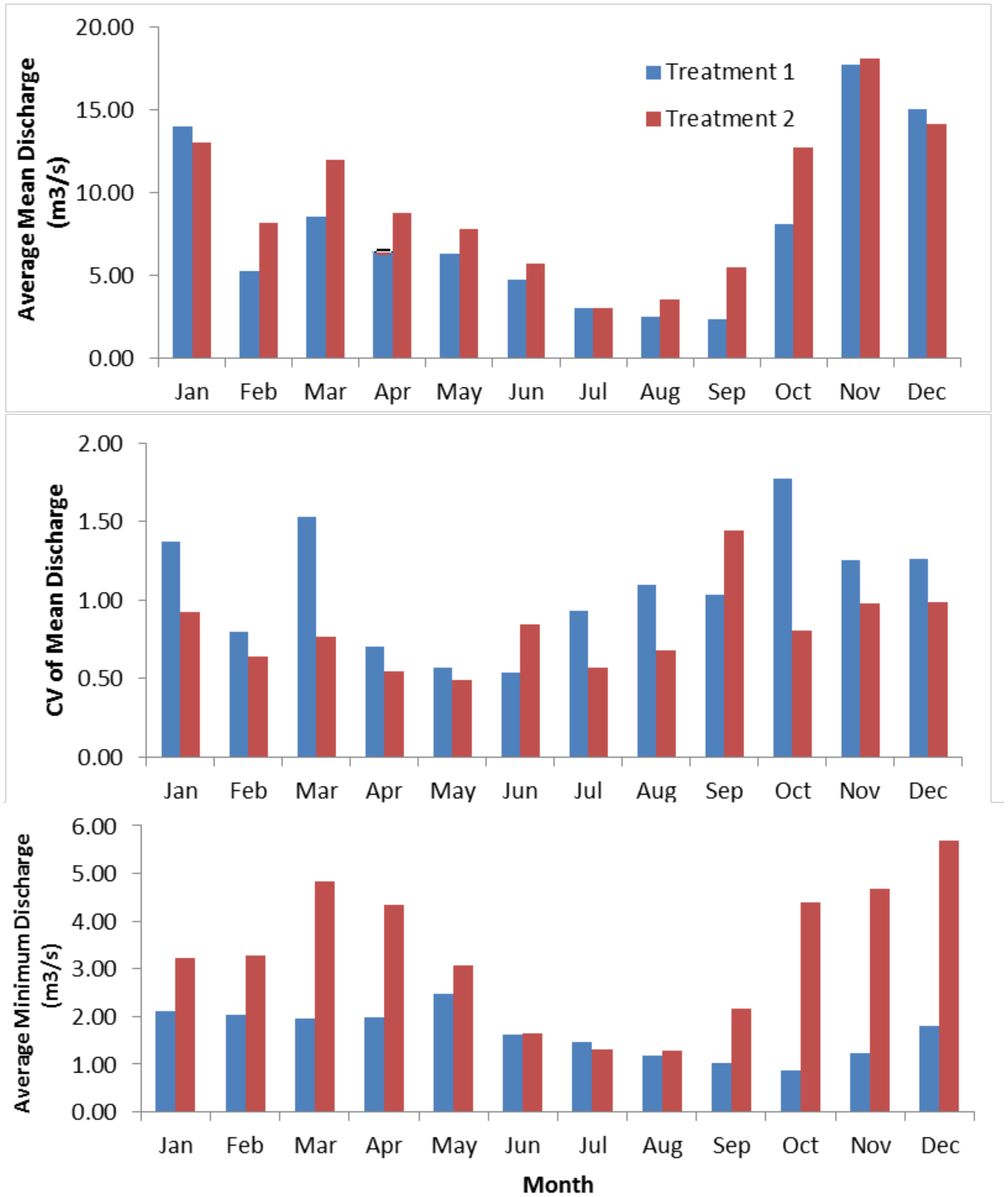


Figure 6.1 Monthly discharge statistics for the Coquitlam River during Flow Treatment 1 (2000-2008) and Treatment 2 (2009-2020) measured at Port Coquitlam (Water Survey of Canada, stn. 08MH141) including monthly mean and minimum discharge as well as the coefficient of variation in monthly mean discharge.

5. Smolt and Fry Outmigration

6.2 Coho

6.2.1 Off-channel smolt abundance

Smolt production from the four constructed off-channel habitats[2] was not considered one of the primary metrics for evaluating the impact of dam regulated flows on freshwater productivity. This is because 1) flows in off-channel areas are stabilized by groundwater inputs and regulated mainstem intakes and 2) continued maintenance of off-channel areas has a large impact on productive capacity. This is not to say that productivity in these areas is unaffected by mainstem flows, but that effects could be less than in the mainstem and, more importantly, mainstem flow effects could be confounded with other factors impacting the productivity of off-channel areas. Production from the four off-channel habitats is reported here because it represents a considerable portion of smolt production for the Coquitlam River and changes in productivity could have population level consequences.

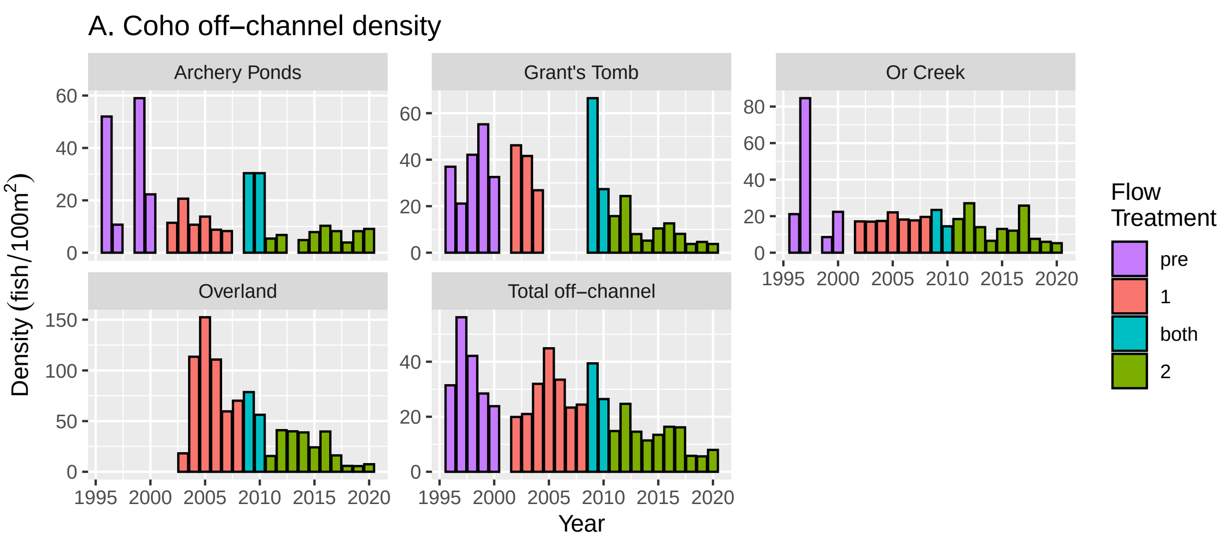


Figure 6.2 Annual Coho smolts density for four constructed off-channel habitats connected to the Coquitlam River. The colours of the bars reflect the flow treatment period of the cohort outmigrating during that year: pre-Treatment (purple), Treatment 1(red), both treatment conditions (teal), Treatment 2 (green).

The four monitored off-channel habitats represent about 10% of available habitat in the Coquitlam River study area and supported from 17% to 77% of the overwintering Coho smolt population during 2000-2020. Since monitoring began in 1995, Total off-channel smolt density has generally declined in all four areas but the decline was most pronounced in Grant's Tomb and Overland Channel (Figure 6.2). Mean off-channel smolt density decreased from 28 smolts/100m^2 during Treatment 1 to 14 smolts/100m^2 during Treatment 2 (2-tailed t-test, p = 0.01). The annual mean density of Coho smolts in the mainstem portion of the study area ranged from 1.9 to 9.2 smolts/100m^2, which was several times lower than that the average for all the off-

[2] There are seven major off-channel habitat sites in Coquitlam River, four in the smolt study area, including Grant's Tomb, which was dewatered during 2005-2008 to facilitate repairs to Coquitlam Dam, and three downstream of the study area.

channel sites combined (5.8 to 44.9 smolts/100m^2). While constructed off-channel habitat may represent relatively productive Coho habitat in the Coquitlam River, the smolt densities in Coquitlam River off-channel sites were below the average densities reported for constructed side-channels and ponds in other Pacific Northwest streams (67 and 69 smolts/100m^2, respectively; Koning and Keeley 1997).

6.2.2 Mainstem smolt abundance

During 2000-2020, the Coho smolt yield for the 7.5 km long section of the lower Coquitlam River mainstem upstream of the RST2 trapping site ranged from 2,900 to 13,800, with considerable year-to-year variation across the entire study period (Figure 6.3, Appendix 6.1a). To compare changes between Treatment 1 and 2, we only compare abundance for cohorts that reared entirely under Treatment 1 or 2. In the 2016 and 2017 reports, this meant we excluded estimates for the years 2000 and 2009. We interpreted that the 2000 outmigration cohort reared under Treatment 1 and pre-Treatment 1 and that the 2009 cohort reared under both Treatment 1 and 2. However, we have recently realized that 2000 shouldn't have been excluded for Coho since Treatment 1 was implemented in 1997, not 2000 as we previously understood. There is also no justification for considering 2000 as an outlier year as it is comparable to both 2003 and 2008. As a consequence, for this report we included Treatment 1 smolt estimates from 2000-2008 and Treatment 2 estimates from 2010-2020. Changes in smolt abundance from mainstem habitats only were a more sensitive measure of the effect of flow treatments than changes that included estimates from constructed off-channel habitats, which are buffered from mainstem flows either by groundwater effects or have independently controlled water intakes. For this reason, we only used the mainstem Coho abundance estimates for evaluating flow treatment effects.

The mean abundance of mainstem smolts was 6,173 during Treatment 1 and 8,193 during Treatment 2 (Figure 6.4) and did not amount to a statistically significant change in abundance (2-tailed t-test, p = 0.12; Table 6.1). Generally, smolt estimates varied more during Treatment 1 (range: 2,870-11,036) than during Treatment 2 (range: 6,573-10,935; Figure 6.3). This consistency in the year-to-year mainstem smolt yield during Treatment 2 is a potentially important outcome of the flow experiment. Maximum smolt yield was similar during each treatment period but minimum yield was lower and occurred more frequently during Treatment 1 (Figure 6.3). This suggests that freshwater carrying capacity was not increased for all years during Treatment 2, rather, that it increased primarily for lower abundance years. We would have expected an increase in carrying capacity to result in an upward shift from the Treatment 1 smolt yield or just an increase in maximum yield. An increase in the minimum smolt yield could indicate a more complex relationship with habitat carrying capacity mediated by an additional variable, such as the type of carrying capacity, which may only be influenced by the flow treatment under some conditions. An alternative explanation is that carrying capacity was unchanged and the higher minimum smolt yield was a product of higher adult escapement.

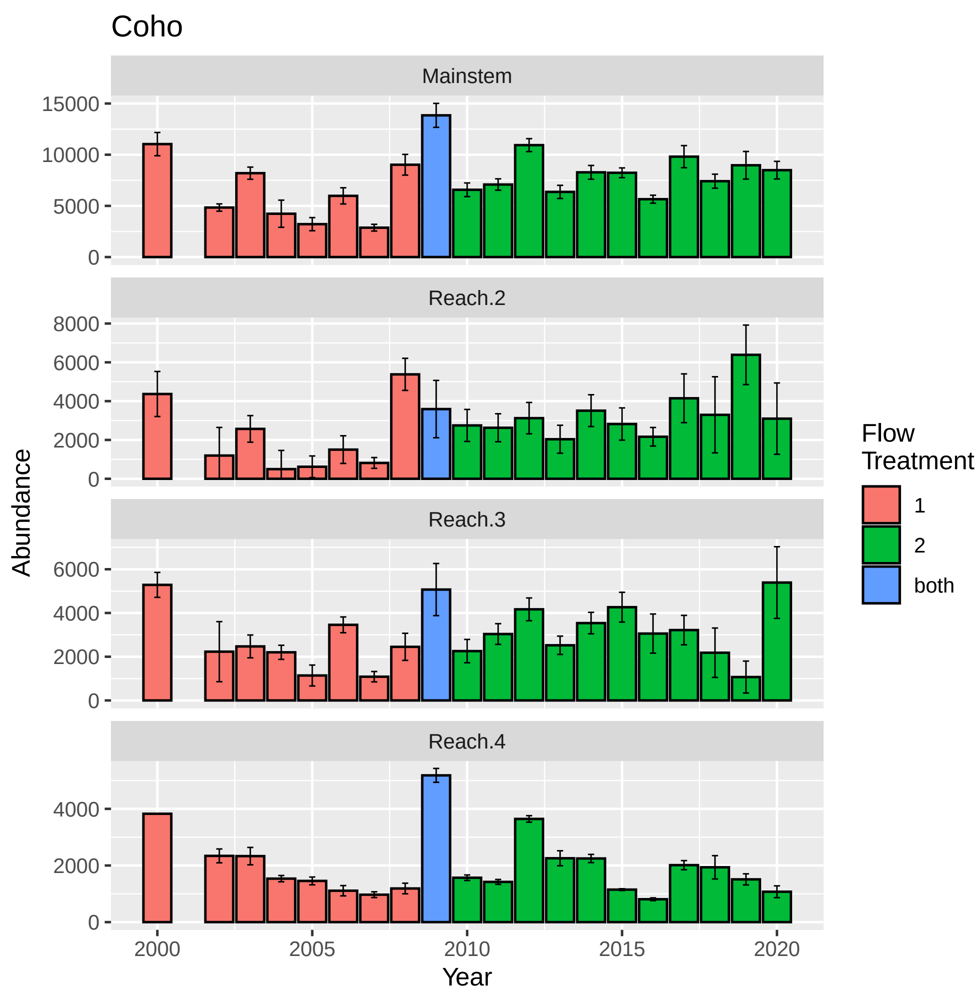


Figure 6.3 Annual Coho smolts yield and 95% confidence intervals for the 7.5km study section of the Coquitlam River mainstem as well for individual reaches 2-4. The colours of the bars reflect the flow treatment period that cohorts were reared under: Treatment 1(red), both treatment conditions (blue), Treatment 2 (green).

Table 6.1 Comparison of mean smolt yield, standard deviation (SD) and number of annual estimates (N) during Treatment 1 and Treatment 2 in the Coquitlam River including the p-values for the two-tailed t tests and percent change in mean abundance between Treatment 1 and 2. Only annual estimates for cohorts that reared exclusively under either Treatment 1 or Treatment 2 conditions were included. For Coho, this includes 2000-2008 for Treatment 1 and 2010-2020 for Treatment 2.

	Treatment 1			Treatment 2			t test		
Smolt yield	Mean	SD	N	Mean	SD	N	p value	% change	Null Hypothesis of no change (p<0.05)
Coho (Total)	12,949	2,719	8	12,114	2,640	11	0.53	-6%	do not reject
Coho (Mainstem)	6,173	2,954	8	7,822	1,601	11	0.12	27%	do not reject
Coho (Reach 2)	2,118	1,728	8	2,900	726	10	0.15	37%	do not reject
Coho (Reach 3)	2,541	823	8	3,138	757	10	0.51	24%	do not reject
Coho (Reach 4)	1,844	563	8	1,898	815	10	0.97	3%	do not reject

5. Smolt and Fry Outmigration

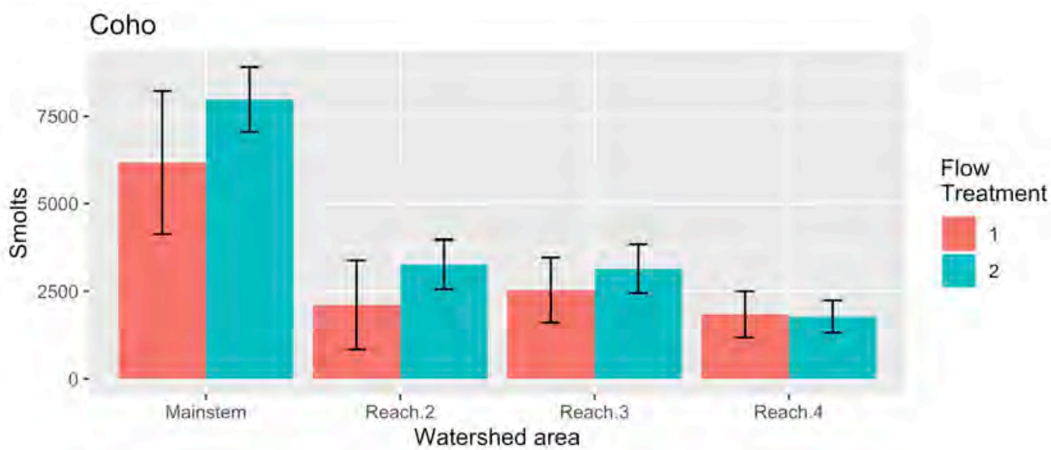


Figure 6.4 Mean Coho smolt yield and 95% confidence intervals for Treatment 1 and Treatment 2 for the 7.5km study section of the Coquitlam River mainstem and in reaches 2-4. Only annual estimates for cohorts that reared exclusively under either Treatment 1 or Treatment 2 conditions were included. For Coho, this includes 2002-2008 for Treatment 1 and 2010-2020 for Treatment 2.

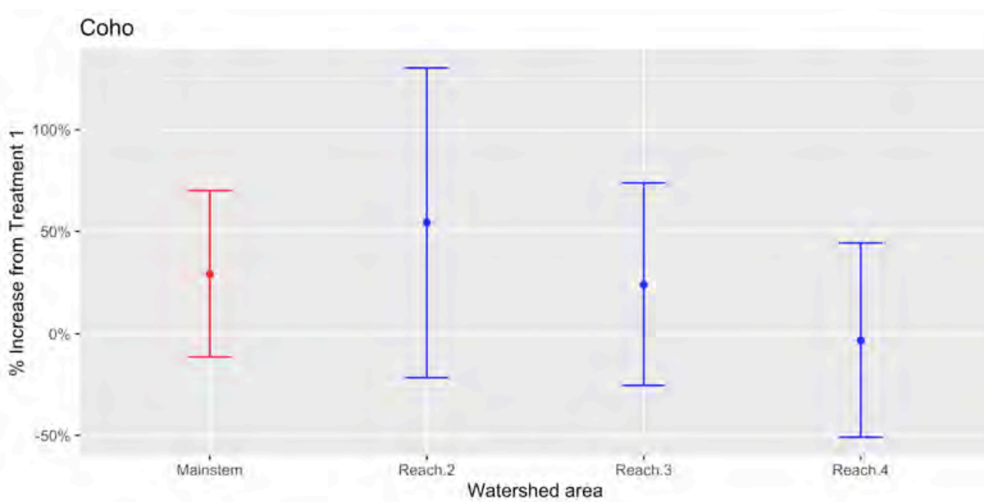


Figure 6.5 Average effect size and 95% confidence intervals of the change in smolt yield from Treatment 1 to 2 for Coho and Steelhead for the 7.5km study section of the Coquitlam River mainstem (red) and for individual reaches 2-4 (blue). For Coho, this includes smolt estimates from 2000-2008 for Treatment 1 and 2010-2020 for Treatment 2.

The amount that abundance changed between treatments (effect size) is a more useful measure for evaluating changes in smolt yield than testing merely whether the means are statistically different. The mean increase in mainstem abundance estimates between Treatment 1

5. Smolt and Fry Outmigration

and Treatment 2 was 27%, but with broad confidence intervals that span the null hypothesis from no change to a 50% increase (Figure 6.5). This is an ambiguous result since it includes both a decrease and increase in smolt yield. The mean increase was highest for reach 2 (53%), followed by reach 3 (24%) and reach 4 (3%, Figure 6.5). The high uncertainty for reach 2 is largely a result of the high variability between annual estimates, particularly during Treatment 1 (Figure 6.3).

We also compared the relative change between Treatment 1 and 2 in the Coquitlam to the change in abundance in other monitored watersheds on the south coast of BC, Vancouver Island and northern Washington State (Appendix 6.2 and 6.3) to attempt to distinguish between the treatment effects and other factors, such as adult escapement and regional changes in freshwater productivity. This is a common approach when conducting experiments in natural settings where a host of non-treatment factors cannot be experimentally controlled. With a BACI analysis, the change in smolt production between Treatment 1 and 2 in the Coquitlam is compared with the change that occurred in the control streams between the same time periods. The change that occurs at the control streams would potentially represent what would have happened in the Coquitlam without any experimental manipulation. For example, if the production increased in the Coquitlam by 50% and there was a 10% increase in the control streams then the effect size of the flow treatment would be 40% and would suggest the assumption of the BA comparison was largely met. For the Coquitlam, the expectation was that Treatment 2 flows would have a neutral to positive affect on smolt production, and thus, the between treatment increase would be similar (neutral) or larger (positive) than the change that happened over a similar time period in the control streams. Control streams were considered based on proximity to the Coquitlam, consistent flow regulation during 2000-2018, and the availability of at least three abundance estimates per treatment period. We then compared abundance trends in the Coquitlam to each of the control streams for the years 2000-2009. We used this time period to test the correlation since the primary expectation is that Coquitlam and control streams are comparable prior to the application of the treatment (Treatment 2 flows) even though trends may continue afterwards. Only those with a correlation coefficient ≥ 0.5 were included from the analysis. Three streams met the criteria: Alouette R. ($R = 0.75$), Keogh R. ($R = 0.62$) and Sakinaw Cr. ($R = 0.60$, Appendix 6.4). We then completed the BACI analysis using the mean relative change of the three control streams, the Alouette River individually, and the Keogh and Sakinaw combined. We included the Alouette River analysis since we consider this the most similar monitored watershed in terms of watershed size, location, geomorphology, fish assemblage and abundance trends. We included the analysis using the Keogh and Sakinaw due to concerns about the amount of bias between the pre- and post 2008 estimates for the Alouette.

Results from the BACI analysis provide no support that the increase smolt production between Treatment 1 and 2 was the result of the flow treatment or that BA comparison was relatively unaffected by other factors. The mean change in productivity between Treatment 1 and 2 in the Coquitlam River ranged from near zero when compared to the control streams excluding the Alouette, to a 13% decline when compared to all control streams (Figure 6.6). This is contrary to the expectation if the BA assumption was satisfied. Mean smolt yield in the control streams increased a similar or greater amount as the Coquitlam, depending on whether the Alouette River is included as a control stream or not, (Figure 6.7) instead of a greater increase for the Coquitlam than the control streams if the BA assumption was satisfied. The reliability of these results depends on how well the control streams reflect the non-treatment factors influencing smolt yield in the Coquitlam River, which we estimate at no higher than moderate. For the Keogh and Sakinaw rivers, which were only moderately correlated with the Coquitlam pre-Treatment 2, it isn't certain that these streams are adequate controls for the Coquitlam. This also applies to the Alouette but with the additional uncertainty about the amount the pre-2008 estimates were biased low. While the Alouette River is possibly the most suitable control stream, based on proximity, watershed similarity and fish assemblage, Cope (2015) concluded that smolt yield estimates for the Alouette River were biased low prior to 2008, when the trapping site was repositioned further upstream to avoid tidal-driven backwatering. The BACI analysis is not intended to replace or supersede smolt yield as the primary metric for evaluating fisheries benefits. It does provide useful context to interpret the likely rise in smolt yield, and raises the possibility that some of the increased yield was due to non-treatment effects.

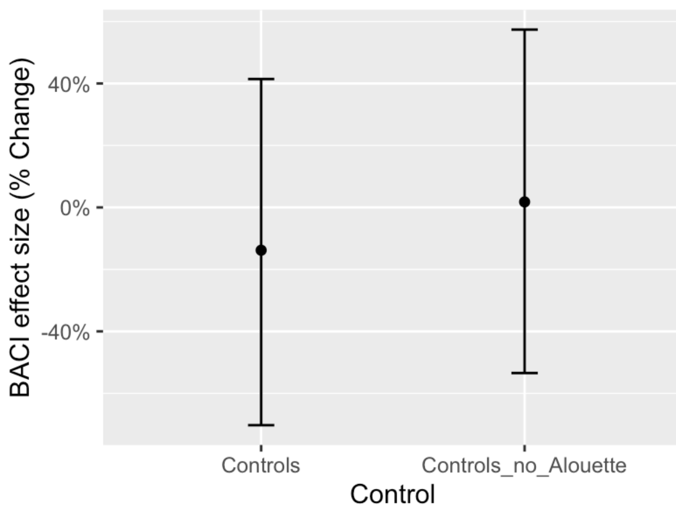


Figure 6.6 Average effect size and 95% confidence intervals for Coho smolt yield between Treatment 1 and 2 based on the BACI analysis using the Alouette, Keogh and Sakinaw as controls (Controls) and using only the Keogh and Sakinaw as controls (Controls_no_Alouette). The BACI effect represents the change in Coquitlam mainstem smolt production between Treatment 1 and 2 that is attributed to flow treatment. Effect size greater than 0 represents an increased productivity as a result of Treatment 2 whereas negative values indicates a decrease. This includes cohorts that reared entirely during Treatment 1 (2000-2008) or Treatment 2 (2010-2018). Note that the Alouette and Sakinaw data did not included all years.

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5. Smolt and Fry Outmigration

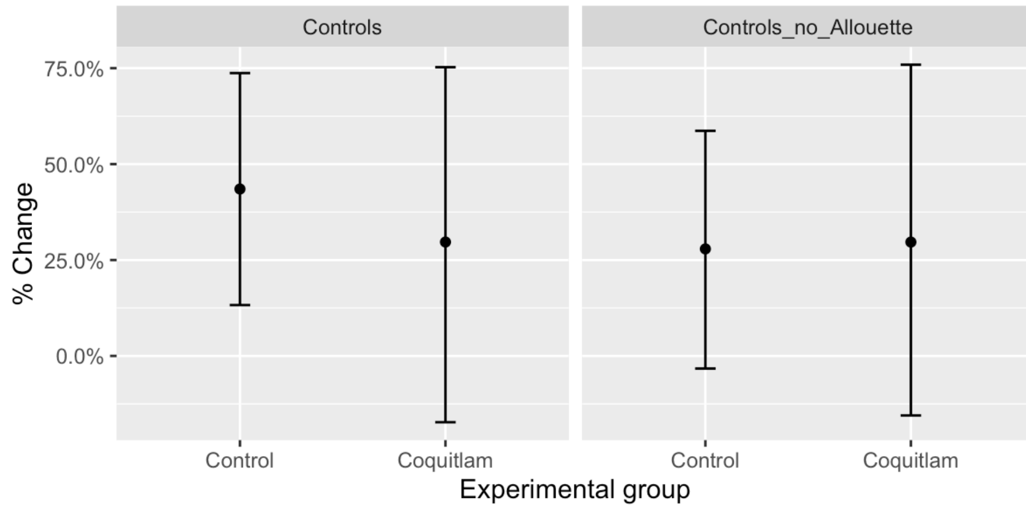

Figure 6.7 Mean percent change in Coho smolt yield and 95% confidence intervals from Treatment 1 (2000-2008) to Treatment 2 (2010-2018) in mainstem habitats in the Coquitlam River and control streams (Alouette and Keogh rivers, and Sakinaw Creek) with including all streams (Controls) or excluding the Alouette River (Controls_no_Alouette). This includes smolts estimates for cohorts that reared entirely under only Treatment 1 or 2, including 2000-2008 for Treatment 1 and 2010-2018 for Treatment 2. Note that Alouette and Sakinaw data sets did not included all years.

In reach 4, where annual downstream smolt trapping has occurred over a longer time period (1997-present), there is a systematic trend of generally decreasing Coho smolt yields from 1997-2008 and then again from 2009-2020, but with high variability for adjacent years (Figure 6.8). This highlights the need to consider that the flow treatments were only one of several factors influencing smolt yield during this time period.

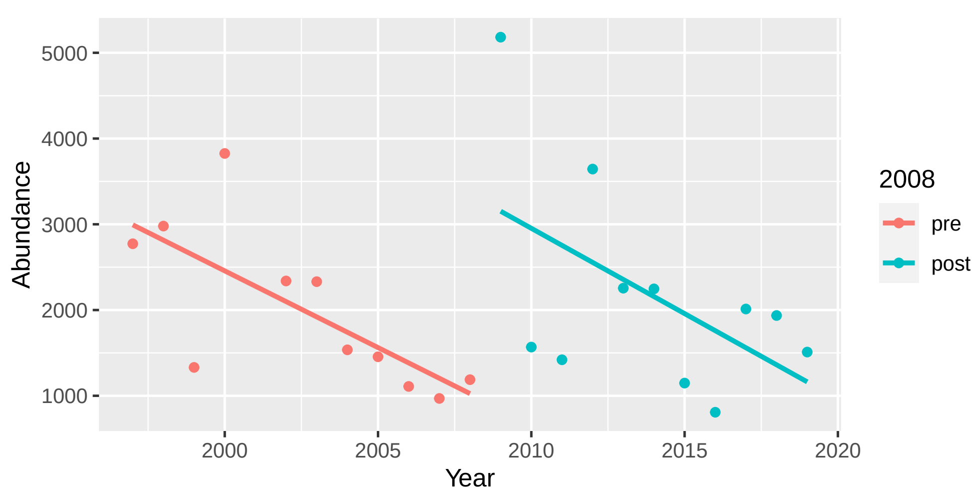

Figure 6.8 Annual numbers of Coho smolts in reach 4 of Coquitlam River during 1997-2020.

5. Smolt and Fry Outmigration

Coho egg-to-smolt survival remained consistently low (0.03-0.55%) for the 2002-2020 brood years (Table 6.1b), with the highest values associated with the lowest escapements. These values decreased from Treatment 1 and 2 (mean: 0.3% and 0.1%, respectively; 2-tailed t-test p=0.01). We are cautious about any between-treatment evaluations using adult escapement estimates, which are likely not comparable because of the change in survey methods employed during Treatment 2.

Table 6.2 Comparison of alternative Ricker stock-recruitment model predicting Coho smolt abundance as a function of adult abundance. Mean covariate effect denote the mean value of γ (covariate effect). R^2 is the proportion of observed variance in log fry abundance predicted by the model, and ΔDIC is the difference in the deviance information criteria for each model relative to the model with the lowest value (the best model). This includes the 2002-2018 brood years. Shaded rows indicate models with high model support (ΔDIC< 2).

Model #	Model	Mean covariate effect	R^2	Δ DIC
1	Base Ricker	NA	0.66	0.00
2	SPAWN_MEAN	0.23	0.61	23.46
3	INC_MEAN	-0.032	0.65	38.27
4	Emerg_MEAN	-0.155	0.59	31.10
5	SUMMER_MEAN	-0.159	0.55	38.75
6	WINTER_MEAN	-0.188	0.72	6.91
7	WINTER_70	-0.162	0.72	20.73
8	AUG_5.4cms	0.054	0.67	26.86
9	Stranded	-0.449	0.51	13.68
10	Treatment	-0.005	0.66	36.63

The covariate analysis suggests that adult abundance alone has the largest effect on Coho smolt abundance. Appendix 6.5 lists the covariates, including the calculation methods. The Base Ricker model, which does not include a covariate, was the only model with high support (ΔDIC<2, Table 6.2) and explained 66% of the variation in smolt abundance. While several other models explained slightly more variation, only Model 6 – Winter Mean had even moderate model support (ΔDIC 4-7), with low support for all others. It is important to note that DIC values are lowest for the model with the best fit and fewest parameters. Since the Base Ricker model has one less parameter than those with a covariate, it can have the highest model support even though it does not have the highest R^2 value. Model 10, representing the flow treatment effect had low support (ΔDIC = 36) and a relatively small covariate effect. With the Base Ricker model, smolt abundance varied considerably around the best-fit line particularly at low escapement levels (Figure 6.9 upper-left panel).

5. Smolt and Fry Outmigration

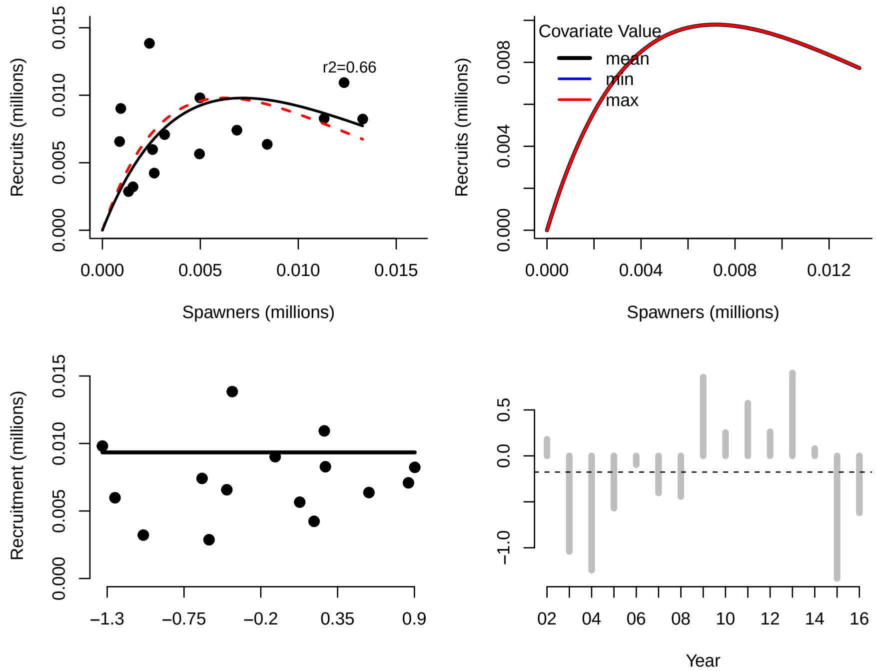

Figure 6.9 Fit of Ricker adult-to-smolt stock-recruitment covariate model for the Coquitlam River during Treatment 1 and 2 years (2002-2018 brood years). Since the Base Ricker model had the highest support, no covariates were included in the plot. Upper left shows the best fit line for the relationship between spawners and smolts, and model fit (R^2).

Since adult escapement estimates for Treatment 1are likely biased, we repeated this analysis only including brood years during Treatment 2. The Base Ricker model still had the highest support but four other models also had high support (ΔDIC <2, Table 6.3). With the Base Ricker model explaining 66-75% of smolt abundance depending on if Treatment 1 years are included, and that no model explained substantially more, suggests adult escapement may be the primary driver of freshwater productivity. While this adds support of the importance of adult escapement, it does not eliminate the possibility that the flow treatment also affected smolt production due to the uncertainty about the amount that escapement was biased.

Table 6.3 Comparison of alternative Ricker stock-recruitment model predicting Coho smolt abundance as a function of adult abundance. This includes the 2008-2018 brood years. Shaded rows indicate models with high model support (ΔDIC< 2).

Model #	Model	Mean covariate effect	R^2	Δ DIC
1	Base Ricker	NA	0.75	0.00
2	SPAWN_MEAN	0.1	0.50	2.82
3	INC_MEAN	-0.378	0.61	0.68
4	Emerg_MEAN	-0.286	0.45	1.75
5	SUMMER_MEAN	-0.17	0.44	2.45
6	WINTER_MEAN	-0.008	0.77	3.78
7	WINTER_70	0.004	0.75	3.02
8	AUG_5.4cms	0.366	0.60	0.34
9	Stranded	-0.39	0.61	0.41

5. Smolt and Fry Outmigration

6.2.3 Mainstem fall fry abundance

The abundance of Coho fry from the snorkel survey based standing stock assessment has varied from a low of 18,000 in 2007 to 91,000 in 2011 (Figure 4.3). Mean fry abundance increased 83% from Treatment 1 to Treatment 2 (2-tailed t-test: *p*=0.05, Figure 6.10). However, any change in fry abundance between the treatments could be explained by both the change in flow treatment as well as the change in adult escapements. Adult Coho escapement during the three years of Treatment 1 was far lower than during the years of Treatment 2 (Treatment 1: 940-2400 adults; Treatment 2: 3,200-13,000). Mean estimated egg-to-fall fry survival during Treatment 1 and 2 was 1.3% and 0.9% respectively, though they were not significantly different (2-tailed t-test, $p = 0.60$). All three years of Treatment 1 survival estimates correspond to very low escapements.

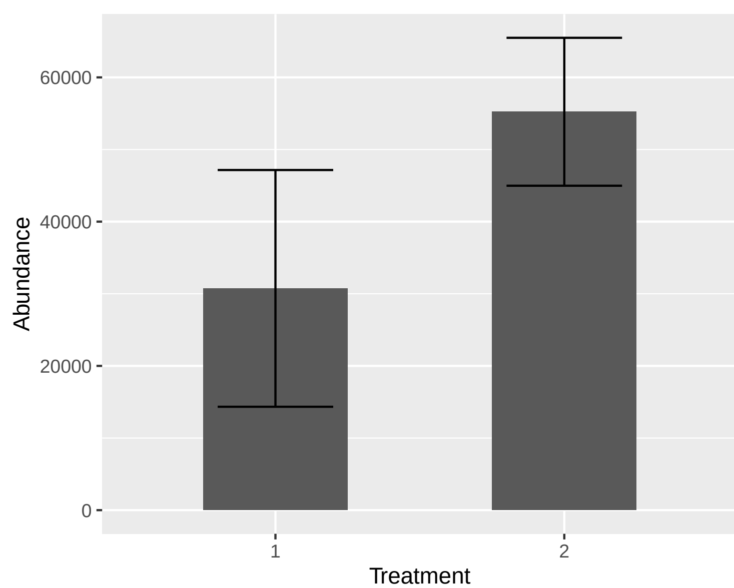

Figure 6.10 Mean Coho fall fry abundance and 95% confidence intervals during Treatment 1(2006-2008) and Treatment 2 (2009-2020) for the 7.5km study section of the Coquitlam River mainstem.

We used a Ricker stock-recruitment covariate to predict fall fry abundance as a product of adult escapement and a covariate reflecting either flow treatment, ramping related fry stranding and season specific flow. Appendix 6.6 lists the covariates, including the calculation methods. Of these, the model with fry stranding as the covariate best predicted fall fry abundance. Model 8 - Stranded was the only model with high support (lowest ΔDIC), while all other models had low support (ΔDIC > 7, Table 6.4). Model 8, along with five other models explained similarly high amounts of fall fry abundance (R2: 0.71-0.79). Model 8 predicted that fall fry capacity had not been reached with the escapement levels thus far, and that that the variation in fry abundance at high escapement levels was explained by the stranding effect (Figure 6.11, upper-left pane). This is evident from the large vertical offset (dashed lines) relative to the predicted fry abundance due to adults alone. While the high support for Model 8 and high covariate effect suggests stranding may have had a substantial effect on fall fry abundance, the wide spread of the minimum and maximum covariate values represent the high uncertainty of the model (Figure 6.11, upper-right pane). Model 7 – Treatment, representing the flow treatment effect had both low support and explained considerably less of the fall fry abundance (ΔDIC > 7, $R^2 = 0.54$). What is surprising about the relatively high effect of the stranding model is that the covariate uses the number of fish salvaged, which is not a direct measure of mortalities since many are returned to the river alive. If the model does reflect a true effect from standing, this raises the possibility that stranding may largely not be mitigated by salvage.

Table 6.4 Comparison of alternative Ricker covariate stock-recruitment model predicting fall fry Coho abundance as a function of adult abundance. This includes the 2006-2019 brood years. Shaded rows indicate models with high model support (ΔDIC< 2).

Model #	Model	Mean covariate effect	R^2	Δ DIC
1	Base Ricker	NA	0.71	12.84
2	Spawning Mean	-0.087	0.79	18.30
3	Incubation Mean	-0.121	0.68	15.89
4	Emergence Mean	-0.114	0.57	15.88
5	Summer Mean	0.074	0.78	17.70
6	Proportion Aug> 5.4cms	0.054	0.76	18.32
7	Treatment	-0.243	0.54	12.62
8	Stranded	0.538	0.75	0.00

5. Smolt and Fry Outmigration

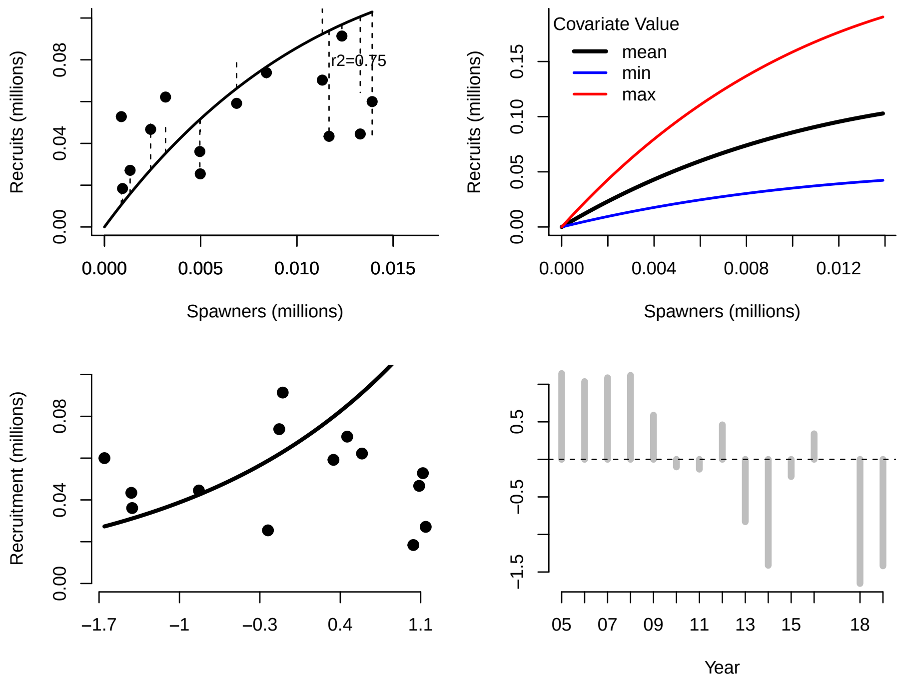

Figure 6.11 Fit of Coho adult-to-fall fry Ricker covariate model using number of fish stranded as the covariate for the Coquitlam River during Treatment 1 and 2 years (2006-2019 brood years). The upper-left panel shows the best fit line for the relationship between spawners and smolts, and model fit (R^2). This model shifts the stock-recruitment curve each year based on the number of stranded Coho fry salvaged following ramping events March-September. The vertical dashed lines show the covariate offset due to the covariate.

6.3 Steelhead

6.3.1 Off-channel smolt abundance

Steelhead smolt yield from off-channel habitats represents only a small proportion of the total yield (0.3%-9% of total annual yield). Thus, off-channel smolt production is not used to evaluate the fisheries benefits of Treatment 1 and 2 flow regimes.

6.3.2 Mainstem smolt abundance

During 2000-2020 the estimated number of Steelhead smolts outmigrating from the 7.5 km study section of the Coquitlam River mainstem upstream of the RST2 trapping site ranged from 2,200 to 5,500 (Figure 6.12). Mean smolt yields for cohorts that reared exclusively under Treatment 1 or Treatment 2 were statistically different (3,716 smolts and 4,759 smolts; respectively; 2-tailed t-test $p = 0.04$, Table 6.2). This included smolt estimates from the years 2002-2008 for Treatment 1 and years 2012-2020 for Treatment 2. This was driven primarily by the increase in smolt abundance in reach 4. Mean abundance increased more than two-fold in reach 4 from Treatment 1 to Treatment 2 (mean: 925 and 2,253 smolts, respectively; t-test, *p*<0.01; Table 6.2, Figure 6.12), whereas yields from reaches 2 and 3 were similar during both treatment periods. This result is consistent with the hypothesis that the relatively confined channel in reach 4 combined with the higher Treatment 2 base flows would produce a higher energy flow environment, which would be more favorable to juvenile Steelhead.

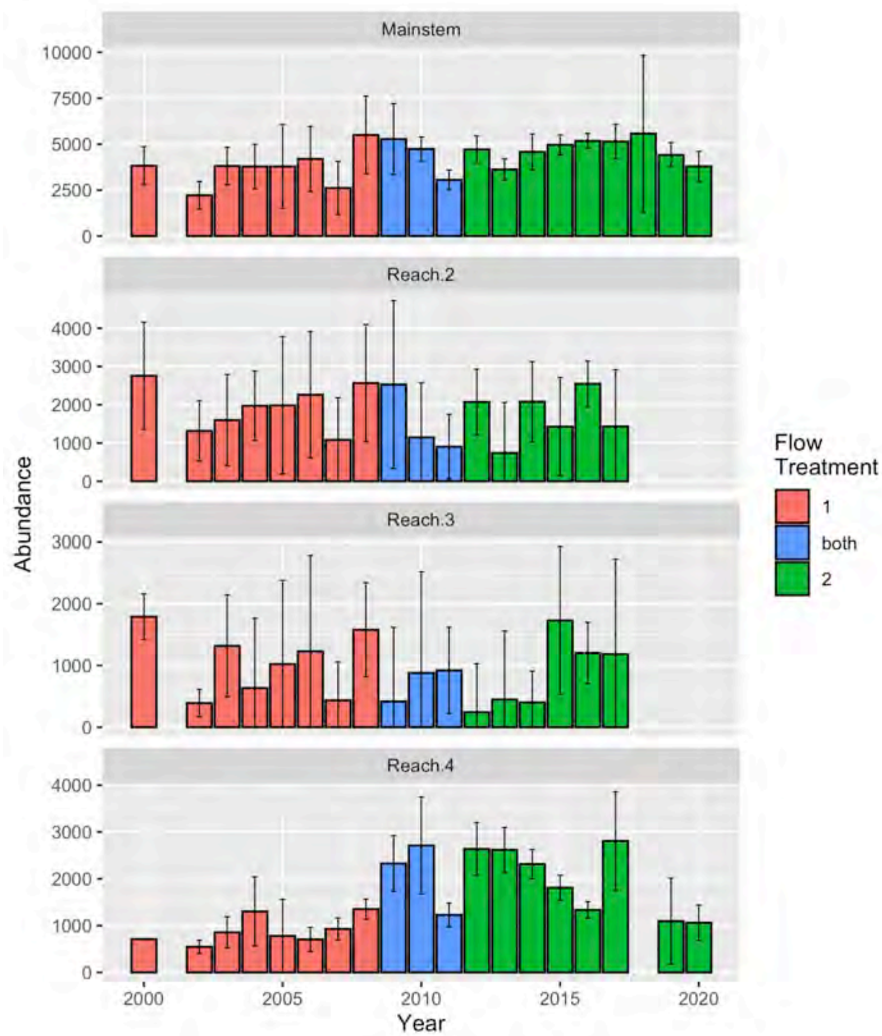

Figure 6.12 Annual Steelhead smolts yield and 95% confidence intervals for the 7.5km study section of the Coquitlam River mainstem as well for individual reaches 2-4. The colours of the bars reflect the flow treatment period that cohorts were reared under: Treatment 1(red), both treatment conditions (blue), Treatment 2 (green).

5. Smolt and Fry Outmigration

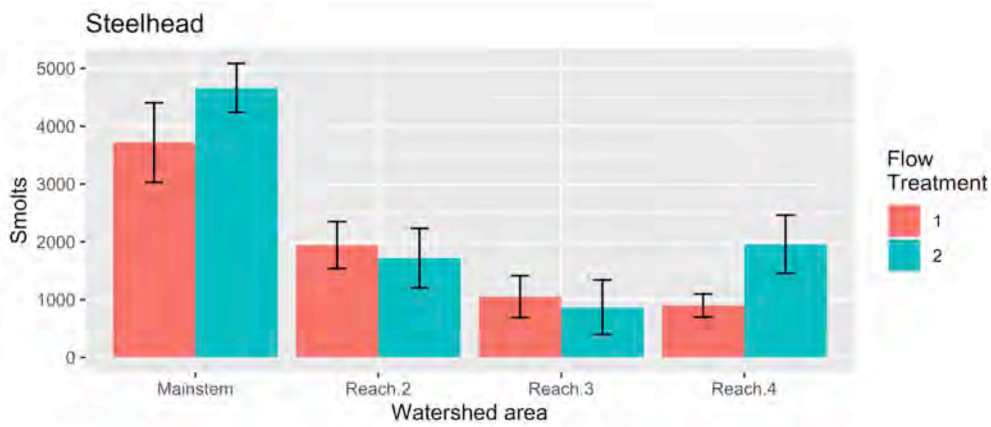

Figure 6.13 Mean Steelhead smolt yield and 95% confidence intervals for Treatment 1 and Treatment 2 for the 7.5km study section of the Coquitlam River mainstem and in reaches 2-4. Only annual estimates for cohorts that reared exclusively under either Treatment 1 or Treatment 2 conditions were included. This includes 2002-2008 for Treatment 1 and 2012-2020 for Treatment 2.

Table 6.5 Comparison of mean smolt yield, standard deviation (SD) and number of annual estimates (N) during Treatment 1 and Treatment 2 in the Coquitlam River including the p-values for the two-tailed t tests and percent change in mean abundance between from Treatment 1 to 2. Only annual estimates for cohorts that reared exclusively under either Treatment 1 or Treatment 2 conditions were included. For Steelhead, this includes 2000-2008 for Treatment 1 and 2012-2020 for Treatment 2.

	Treatment 1			Treatment 2			t test		
Smolt yield	Mean	SD	N	Mean	SD	N	p value	% change	Null Hypothesis of no change (p<0.05)
Steelhead (Total)	3,799	993	7	4,825	272	9	0.05	27%	do not reject
Steelhead (Mainstem)	3,701	978	7	4,759	177	9	0.04	29%	reject
Steelhead (Reach 2)	1,827	524	7	1,716	643	6	0.97	-6%	do not reject
Steelhead (Reach 3)	944	464	7	868	590	6	0.80	-8%	do not reject
Steelhead (Reach 4)	925	301	7	2,253	571	7	0.00	144%	reject

The mean increase in abundance from Treatment 1 to 2 was 29% (Figure 6.14). For reaches 2 and 3, there was no clear indication that abundance changed from Treatment 1 to 2. The mean change was above 100% (or a doubling in abundance) in reach 4 but was also highly uncertain. The high uncertainty for individual reaches isn't surprising given the relatively high variability within each treatment period compared to estimates for the entire mainstem (Figure 6.11). As with Coho, to interpret this change as a response to the Treatment 2, we must assume other factors, such as adult escapement, had a relatively small influence on smolt yield during the flow experiment.

5. Smolt and Fry Outmigration

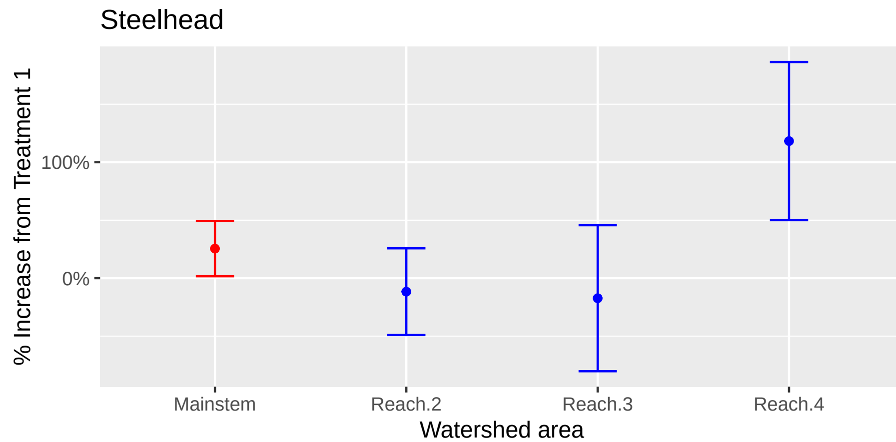

Figure 6.14 Average effect size and 95% confidence intervals of the change in smolt yield from Treatment 1 to 2 for Coho and Steelhead for the 7.5km study section of the Coquitlam River mainstem (red) and for individual reaches 2-4 (blue). This includes 2002-2008 for Treatment 1 and 2012-2020 for Treatment 2.

Using the same method as for Coho, we compared the relative increase in smolt yield in the Coquitlam River to comparable watersheds to distinguish between flow treatment effects and regional factors that could influence smolt yield. The BACI analysis is intended to addressed whether the smolt abundance increased more in the Coquitlam River than in the comparison streams over a similar time period. Appendix 6.7 lists the annual smolt yield for the two Coastal BC streams considered for use as controls. We were limited to only the Alouette River for this comparison since it was the only stream with adequately similar smolt yield trends for cohorts that reared prior to Treatment 2 (correlation coefficient: Keogh $R < 0.01$, Alouette $R = 0.52$, Appendix 6.8). For the BACI comparison, we only included commonly monitored years during Treatment 1 (2002-2008) and Treatment 2 (2012-2014). The mean percent increase in smolt yield between treatment periods was higher for the Alouette (74%) than for the Coquitlam (30%), with considerable overlap in their confidence intervals (Figure 6.15).

The BACI analysis depends on the credibility of the Alouette River smolt data as an index of region wide productivity. As mentioned for Coho, the estimates for the Alouette prior to 2008 may be biased low due to backwatering at the trapping site and shorter monitoring period (Cope 2015). If the bias was large, what we are considering as a non-treatment effects on productivity would instead be an artifact of the changes in monitoring. Support for using the Alouette River relies on is its close proximity, consistent flow regulation, and similar geomorphology, fish assemblage and abundance trends to the Coquitlam. On this basis, it is a useful control. Its weakness is due to the uncertainty in the amount pre-2008 smolt estimates are biased. The possibilities to resolve this rely on accessing additional data from other coastal BC watersheds and further investigating the degree that the Alouette smolt estimates are biased. Given this uncertainty and that no other comparison streams are included, it is difficult to determine whether the BACI analysis is sufficiently reliable to consider in the assessment of the fisheries benefits.

5. Smolt and Fry Outmigration

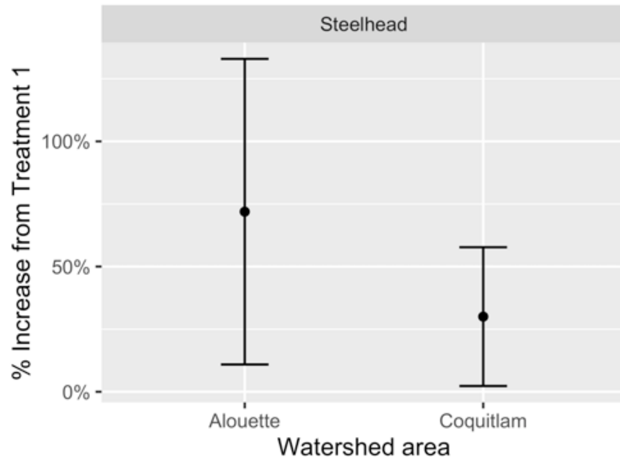

Figure 6.15 Average percent change in Steelhead smolt yield and 95% confidence intervals from Treatment 1 (2000-2008) to Treatment 2 (2010-2018) in mainstem habitats in the Coquitlam River and Alouette River. This includes 2002-2008 during Treatment 1 and 2012-2018 during Treatment 2. Note that for the Alouette River, estimates were only available during Treatment 2 up to 2014.

On average, fork lengths of Steelhead smolts were 6-7mm less during Treatment 1 than Treatment 2 in reach 4 ,and reaches 2 and 3 combined (2-tailed t-test $p < 0.01$ for both). Age-1+ spring migrant parr in reach 4 were on average 10mm larger than those in reaches 2 and 3 (t-test $p = 0.01$).

Steelhead spawner densities appeared to be well above levels thought to be required for full seeding of juvenile habitat for during 2005-2020. During 2005-2020, Steelhead spawner densities in the Coquitlam River ranged from 24 to 80 fish/km (mean: 38 fish/km). This translates to 39,000-149,000 eggs/km deposited in the Coquitlam River during 2005-2017 (Table 3.2). In the Keogh River 13,300 eggs/km was estimated as the minimum required to achieve optimal smolt yield (derived from Ward and Slaney 1993). Moreover, the horizontal spread of data points in the Ricker stock-recruitment relationship for the 2005-2017 brood years suggests the variation in adult abundance had a relatively small effect on smolt abundance (Figure 6.16).

These results suggest that the juvenile carrying capacity of the Coquitlam River, as opposed to adult escapement, had a major influence on Steelhead smolt yield for these adult brood years. With the 2005 brood as the only one to rear entirely under Treatment 1 conditions and all others reared under both treatments or Treatment 2, adult-to-smolt survival has little use for directly assessing the effect of the flow treatment. Instead, we used a stock-recruitment covariate model for the adult-to-fall fry and fall fry-to-fall age-1 parr life stages to address this question.

5. Smolt and Fry Outmigration

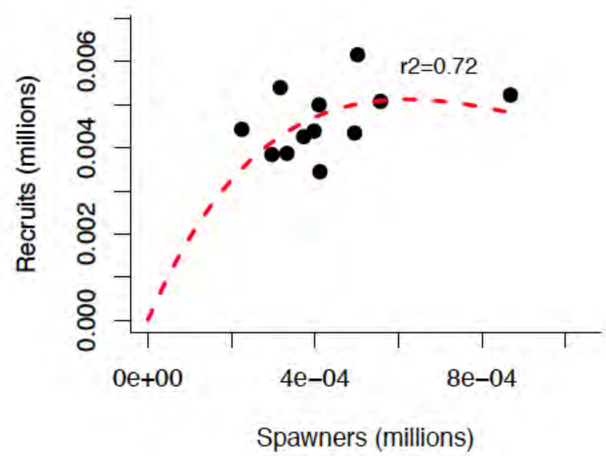

Figure 6.16 Fit of Ricker stock-recruitment model to adult-to-smolt abundance for the Coquitlam River for the 2005-2017 brood years.

6.3.3 Mainstem fry and parr abundance

Based on snorkeling surveys alone, average fall abundance of fry was 70,963 for Treatment 1 and 44,797 for Treatment 2, though this difference was not significant (t-test $p = 0.12$, Figure 6.17). The higher mean and lower precision for Treatment 1 is largely due to 2006, when high escapement resulted in over two-folder higher fry abundance than any other years during the study (Figure 6.18). Without 2006, the mean abundance during Treatment 1 of 37,300 is far more similar to Treatment 2 (44,797). However, relying on only two Treatment 2 sampling years is too vulnerable to bias to represent fry abundance across all of Treatment 1. The 2006 escapement was over 50% higher than the next highest year during the entire study (Table 3.2). Age 1+ parr abundance was similar between Treatment 1 and 2 (age 1+ parr 8,812 and 8,484; respectively; t-test $p = 0.84$). Age 2 + parr were only half as abundant during Treatment 1 than during Treatment 2 (age 2+ parr 1,691 and 2,928; respectively; t-test $p < 0.01$). However, the abundance of age 2+ parr is the product of surviving to that age as well as not smolting during the prior spring. Coquitlam Steelhead smolt after their 2nd or 3rd winter, which is size and growth dependent (Kendall *et al* 2015). The increase in age 2+ parr could be a result of increased survival during their 2nd winter, increased proportion of age 3 smolts and movement of downstream parr into the study area.

5. Smolt and Fry Outmigration

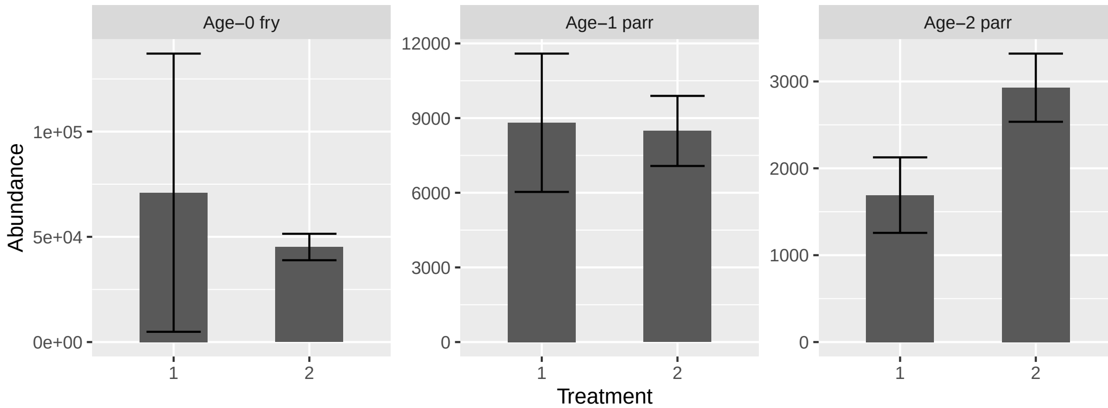

Figure 6.17 Average abundance and 95% confidence limits for Steelhead fry, age-1 parr and age-2 parr during Treatment 1 and 2 in reaches 2-4 of the Coquitlam River. The start of Treatment 2 for Steelhead fry, age-1 parr and age-2 parr is 2009, 2010 and 2011; respectively.

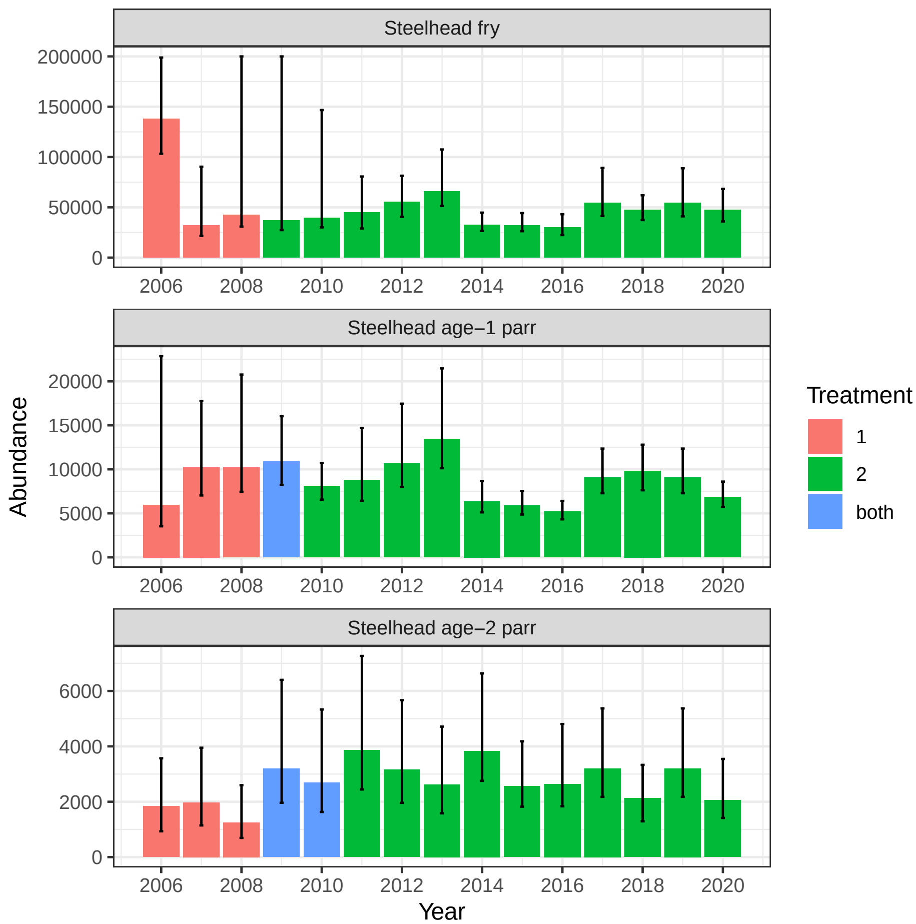

Figure 6.18 Estimates of juvenile standing stock, and 95% confidence intervals by species and age class in Coquitlam River during 2006-2020. Estimates were derived from night snorkeling counts with the exception of 2011 Steelhead (0+), which were based on electrofishing. Bar colour indicates cohorts that were reared entirely under Treatment 1 flows (red), Treatment 2 (green) or both (blue).

5. Smolt and Fry Outmigration

Steelhead egg-to-fry survival averaged 7.7% during Treatment 1 and 5.8% during Treatment 2 and were not significantly different (t-test $p = 0.13$, Appendix 6.1b). This is similar to the Keogh River (mean: 6.5%, Ward and Slaney 1993) but far lower than the Cheakamus (mean: 22.8%, Korman and Schick 2018). Fall fry-to-fall age-1+ parr survival averaged were also similar to Treatment 1 and 2 (20% and 30%, respectively; t-test $p = 0.70$; Appendix 6.1b). Age-1+ parr-to-smolt survival were also similar during Treatment 1 and 2 (55% and 57%, respectively; t-test $p = 0.87$; Appendix 6.1b). This is comparable to parr-to-smolt survival for Steelhead in the Keogh River (48.8%; Tautz *et al*. 1992).

The Ricker stock-recruitment models using adult escapement with and without a covariate were all poor predictors of fall fry abundance ($R^2 < 0.35$, Table 6.6). While Model 4, using June mean discharge as the covariate, had the highest support (ΔDIC <2), it has limited usefulness for explaining fry abundance given the poor fit of the model ($R^2 < 0.35$). To test whether the poor fit was a product of the 2006 brood year, which had both much higher escapement and fry abundance (Figure 6.19, upper left pane), we excluded this year in a repeat analysis. However, there was no improvement in fit with 2006 excluded ($R^2 < 0.36$).

Table 6.6 Comparison of alternative Ricker covariate stock-recruitment model predicting Steelhead fall fry abundance as a function of adult abundance. This includes the 2006-2020 brood years. Shaded rows indicate models with high support (ΔDIC< 2).

Model #	Model	Mean covariate effect	R^2	Δ DIC
1	Base Ricker	NA	0.10	3.53
2	Spawning Mean	-0.037	0.01	5.81
3	Incubation Mean	0.136	0.15	3.78
4	June Mean	0.209	0.35	0.00
5	July Mean	0.01	0.00	6.05
6	August Mean	-0.003	0.00	6.23
7	Difference Spawn Incub	-0.104	0.09	4.83
8	Proportion Aug> 5.4cms	0.122	0.12	4.72
9	Treatment	-0.052	0.02	5.87

5. Smolt and Fry Outmigration

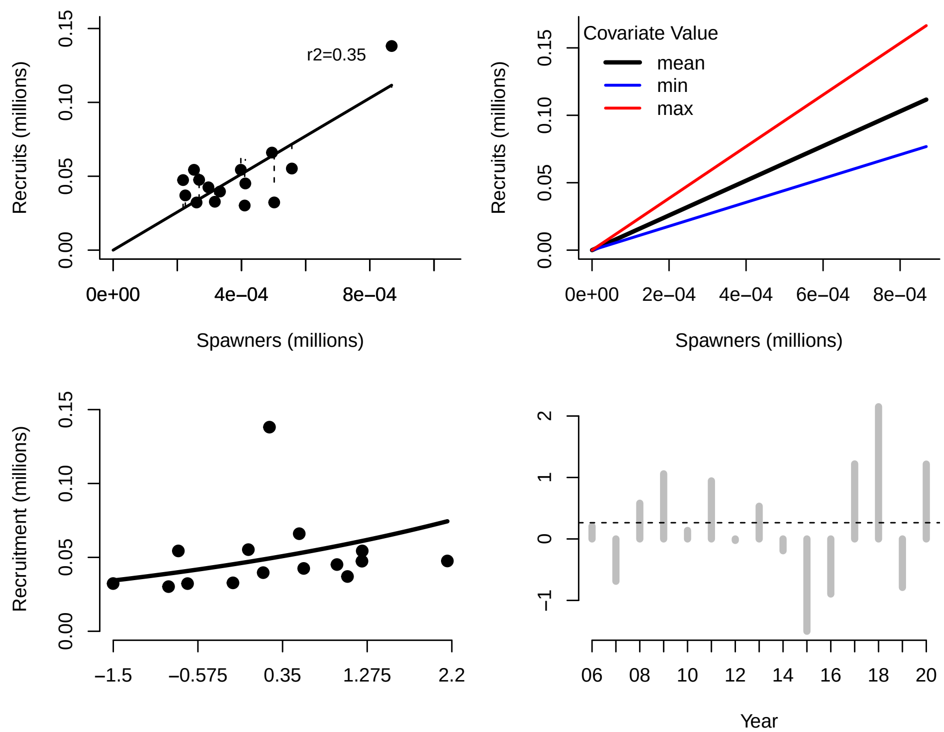

Figure 6.19 Fit of Ricker adult-to-fall fry covariate model to adult-to-fall fry using June mean discharge as the covariate for the Coquitlam River during Treatment 1 and 2 years (2006-2020 brood years). The upper-left panel shows the best fit line for the relationship between spawners and smolts, and model fit (R^2). This model shifts the stock-recruitment curve each year based on mean June discharge at the Port Coquitlam gauge. The vertical dashed lines show the offset due to the covariate.

The Ricker covariate stock-recruitment models using fall fry abundance were all moderate predictors of fall age-1 parr abundance (R^2 0.61-67, Table 6.6), providing little useful information about the primary productivity driver . Model 5 – Spring Mean and Model 9 – Treatment had the most model support (ΔDIC <2). The flow treatment model predicted higher parr abundance for Treatment 1 broods than Treatment 2 (negative covariate effect, Table 6.7). However, with only two Treatment 1 years, the effect could be due to coincidence. For models other than these two, covariates had only small effects, thus, provided no information to understand age-1 parr abundance. The near identical abundance trends for fall fry and fall age-1 parr suggests a common factor is affecting both age classes (Figure 6.18) but that this factor was not one of those evaluated thus far. This may be addressed by expanding the range of covariates. While this can lead to spurious results, it is useful for generating new hypotheses.

5. Smolt and Fry Outmigration

Table 6.7 Comparison of alternative Ricker covariate stock-recruitment model predicting Steelhead fall age-1 parr abundance as a function of fall fry. This includes the 2006-2020 brood years. Shaded rows indicate models with high support (ΔDIC< 2).

Model #	Model	Mean covariate effect	R^2	Δ DIC
1	Base Ricker	NA	0.61	10.79
2	Proportion Sept> 5.4cms	0.016	0.61	3.17
3	Fall Mean	0.035	0.61	2.88
4	Winter Mean	-0.041	0.61	3.18
5	Spring Mean	0.097	0.66	0.91
6	Summer Mean	0.03	0.61	3.21
7	Proportion Aug> 5.4cms	-0.056	0.62	2.19
8	Winter days > 70 cms	-0.07	0.63	2.21
9	Treatment	-0.126	0.67	0.00

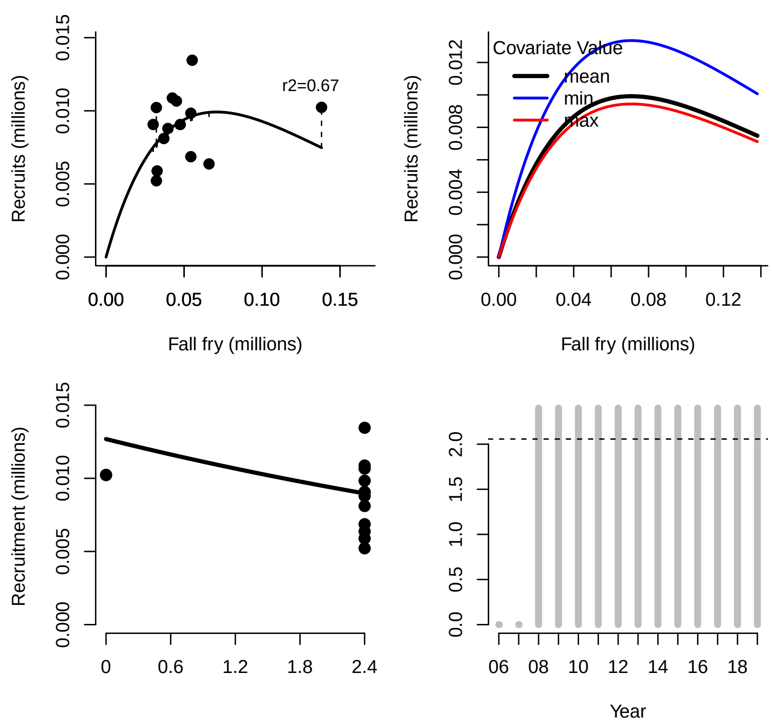

Figure 6.20 Fit of Ricker fall fry-to-fall age-1 parr covariate model using flow treatment as the covariate for the Coquitlam River during Treatment 1 and 2 years (2006-2020 brood years). The upper-left panel shows the best fit

5. Smolt and Fry Outmigration

line for the relationship between spawners and smolts, and model fit (R^2). This model shifts the stock-recruitment curve each year based on whether rearing occurred under Treatment 1 flows (2006-2007 brood years) or Treatment 2 (2008-2020 brood years). The vertical dashed lines show the offset due to the covariate.

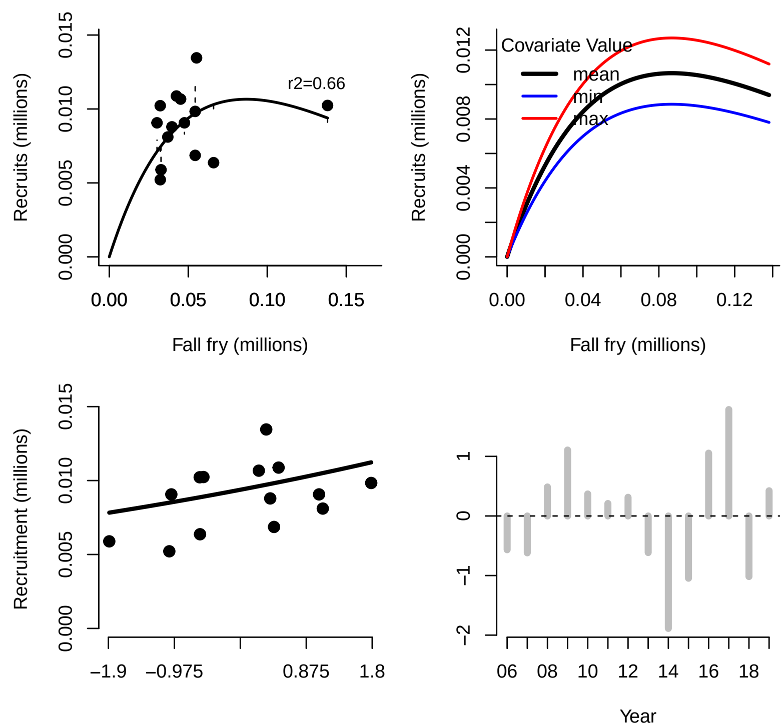

Figure 6.21 Fit of Ricker fall fry-to-fall age-1 parr covariate model using mean flow during spring (April-May) as the covariate for the Coquitlam River during Treatment 1 and 2 years (2006-2019 brood years). The upper-left panel shows the best fit line for the relationship between spawners and smolts, and model fit (R^2). This model shifts the stock-recruitment curve each year based the mean April-May discharge. The vertical dashed lines show the offset due to the covariate.

6.4 Chum

6.4.1 River-wide abundance and survival

Similar to that for Coho and Pink, escapement and egg-to-fry survival estimates for Chum should be considered preliminary and will likely change as adult salmon observer efficiency and survey life data are collected in future years. Adult returns of Chum salmon to Coquitlam River (including Reach 1) have ranged from 12,000-78,000 (Appendix 6.1a), while fry production upstream of RST2 has ranged from 0.8 to 12.7 million. During Treatment 1 (2002-2007 brood years) Chum egg-to-smolt survival ranged from 3.7% to 14.1% and 18.1% to 40.0% during Treatment 2 (Appendix 6.1b). Mean survival increased from 10.2% for Treatment 1 to 22.8% for Treatment 2 (t-test $p < 0.01$, Figure 6.22).

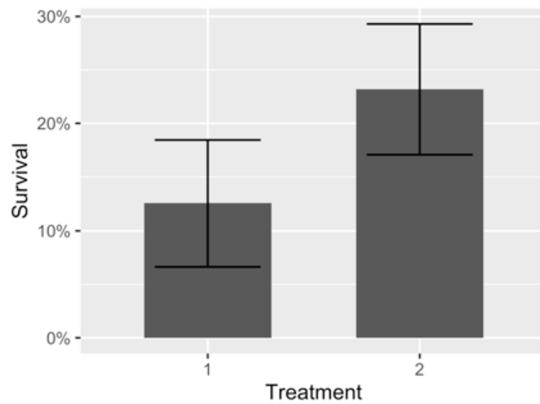

Figure 6.22 Mean egg-to-fry survival and 95% confidence intervals for Chum Salmon broods reared under Treatment 1 (2002-2007) and Treatment 2 (2008-2019) in the Coquitlam River study area.

Bradford (1995) reported an average egg-to-smolt survival rate of 6.7% (±1 standard deviation = 3.3%-13.5%) for Chum populations in nine Pacific Northwest streams. The Chum egg-to-fry survival estimates for the 2005, 2008, 2010, 2013 and 2014 brood years in the Coquitlam River exceeds published values for this species, and these, and possibly all, may be biased high. The most plausible source of this bias would be an underestimate of Chum escapement (see Section 2.2) as opposed to an overestimate of Chum fry. The study design for estimating fry yield is robust: sampling was nearly daily over the entire outmigration period; it accounts for changes in capture efficiency over the outmigration period by stratifying into 8-10 distinct periods; and used a large number of marked fish (800-2000) to estimate capture efficiency for each strata. Escapement estimates depend on estimating both observer efficiency and survey life. Even with considerable effort, these two parameters remain highly uncertain. For instance, there is only a weak relationship between an observer's guess of their efficiency and the mark-recapture based estimate of their efficiency ($R^2 = 0.38$). This level of uncertainty remains too large for the escapement model to generate both credible escapement estimates and precision estimates (see section 2.2.3). Given the weaknesses of adult salmon escapement estimates we view them as an index of abundance rather than accurate measures of escapement. As well, considering that our survival estimates could be biased and unknown (but likely high) uncertainty of adult escapement, we encourage a conservative approach to interpreting any statistical tests using this metrics.

There was strong support of a linear escapement-to-fry relationship during Treatment 2 ($R^2 = 0.72$, Figure 6.23) but only moderate support of one during Treatment 1 ($R^2 = 0.36$). The lower fit during Treatment 1 was largely the result of 2005, when fry production was considerably higher than expected. If this was considered an outlier and excluded, there would be a high level of support for a linear relationship during Treatment ($R^2 = 0.78$). A linear relationship confirms that with changes in egg-to-fry survival or fry- per-adult are more appropriate methods for evaluating the fisheries benefits of flow treatments than juvenile production alone.

5. Smolt and Fry Outmigration

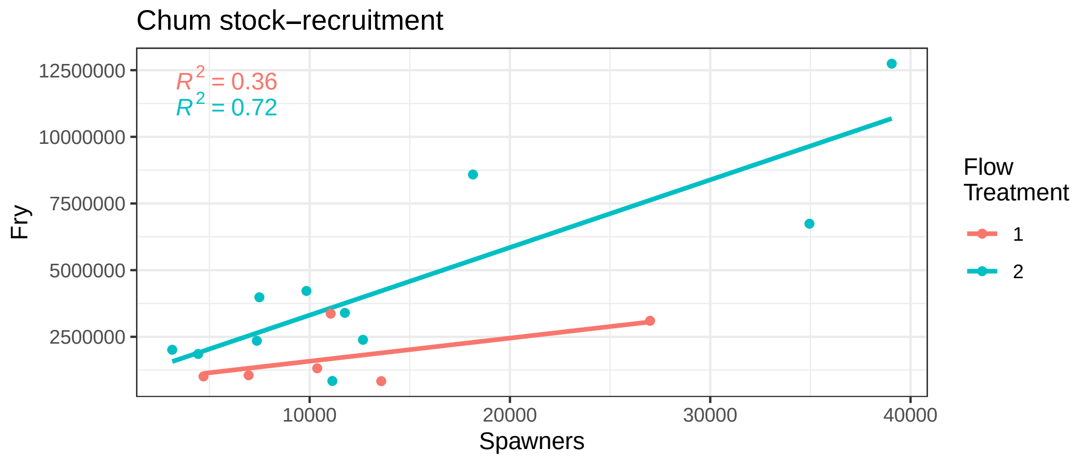

Figure 6.23 Escapement-to-fry stock-recruitment relationships of Chum Salmon during flow Treatment 1 (2002-2008) and Treatment 2 (2009-2019) for the Coquitlam River study area. The best-fit lines intercept the x and y axis at 0 as is typical of stock-recruitment relationships. R^2 values reflect the fit of stock-recruitment.

Based on an analysis of covariance (ANCOVA), to evaluate the Chum fry-per-adult relationship for Treatment 1 and 2 using the package STATS in R (R Development Core Team 2009). The test whether flow treatment had a significant effect on recruitment, we evaluated whether the y-intercepts differed for the ln (fry/adult) ~ adult relationship between Treatment 1 and 2 (Figure 6.24). Using ln (fry/adult) instead of fry in the analysis reduces the violation of the assumption that linear stock-recruitment intercept at 0 adults and 0 juveniles. In effect, it becomes a test of whether adult-to-fry survival changes by flow treatment. Appendix 6.11 provides outputs of this analysis for Chum 2003-2016 and 2018-2019 brood years. The results are consistent with the comparison of egg-to-fry survival above. The significant values for Treatment and intercept (both $p<0.001$) support hypothesis that fry-per-adult recruitment differed between Treatment 1 and 2. However, as previously discussed, unaccounted for bias and uncertainty in the adult escapement data also impacts this test. Thus, results should be interpreted with this in mind.

5. Smolt and Fry Outmigration

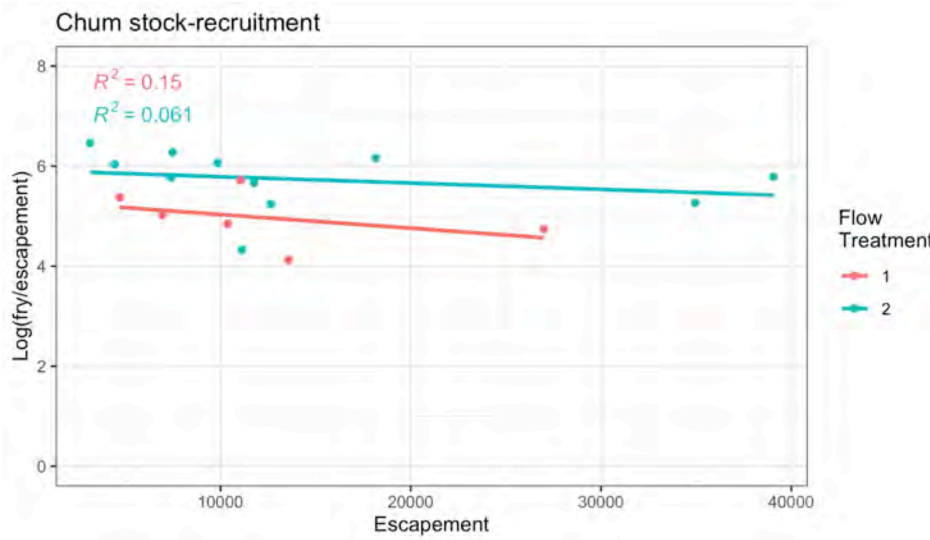

Figure 6.24 Adult-to-ln(fry/adult) stock-recruitment relationships of Chum during flow Treatment 1 (2002-2008) and Treatment 2 (2009-2019) from 7.5 km of the Coquitlam River used for the ANCOVA tests as to whether flow treatment had a significant effect of productivity.

As for Coho and Steelhead, an effective way to distinguish between the effects of the flow treatment and other factors influencing freshwater conditions is by comparison with other watersheds. Chum escapement and juvenile abundance has been monitored in the Alouette and Cheakamus River. During 2002-2014, both Chum escapement and fry yield in the Coquitlam River are only minimally to moderately correlated with that in the Alouette River (escapement: R = 0.55, fry yield: R=0.03; Figure 6.25a; Cope 2015), which reduces the viability of using the Alouette River as a control of region-wide factors influencing Chum productivity. With the end of monitoring on the Alouette in 2014, these values will remain unchanged. Chum escapement and fry yield were strongly and moderately correlated with the Cheakamus River (R = 0.79 and R = 0.57; Figure 6.25b) suggesting it has a moderate use as a comparison when evaluating whether changes in productivity in the Coquitlam are the result of flow treatments or region wide factors. However, a drawback in using the Cheakamus is that the switch in 2016 between IFA and WUP flow regimes occurred at a similar time as the change between Treatment 1 and 2 on the Coquitlam. Even with a potential increase in productivity for the Cheakamus under WUP flows, there was a relative increase in fry abundance 2008 onwards in the Coquitlam. While this is possibly an effect of increased escapement, if could also be due to the transition to Treatment 2.

5. Smolt and Fry Outmigration

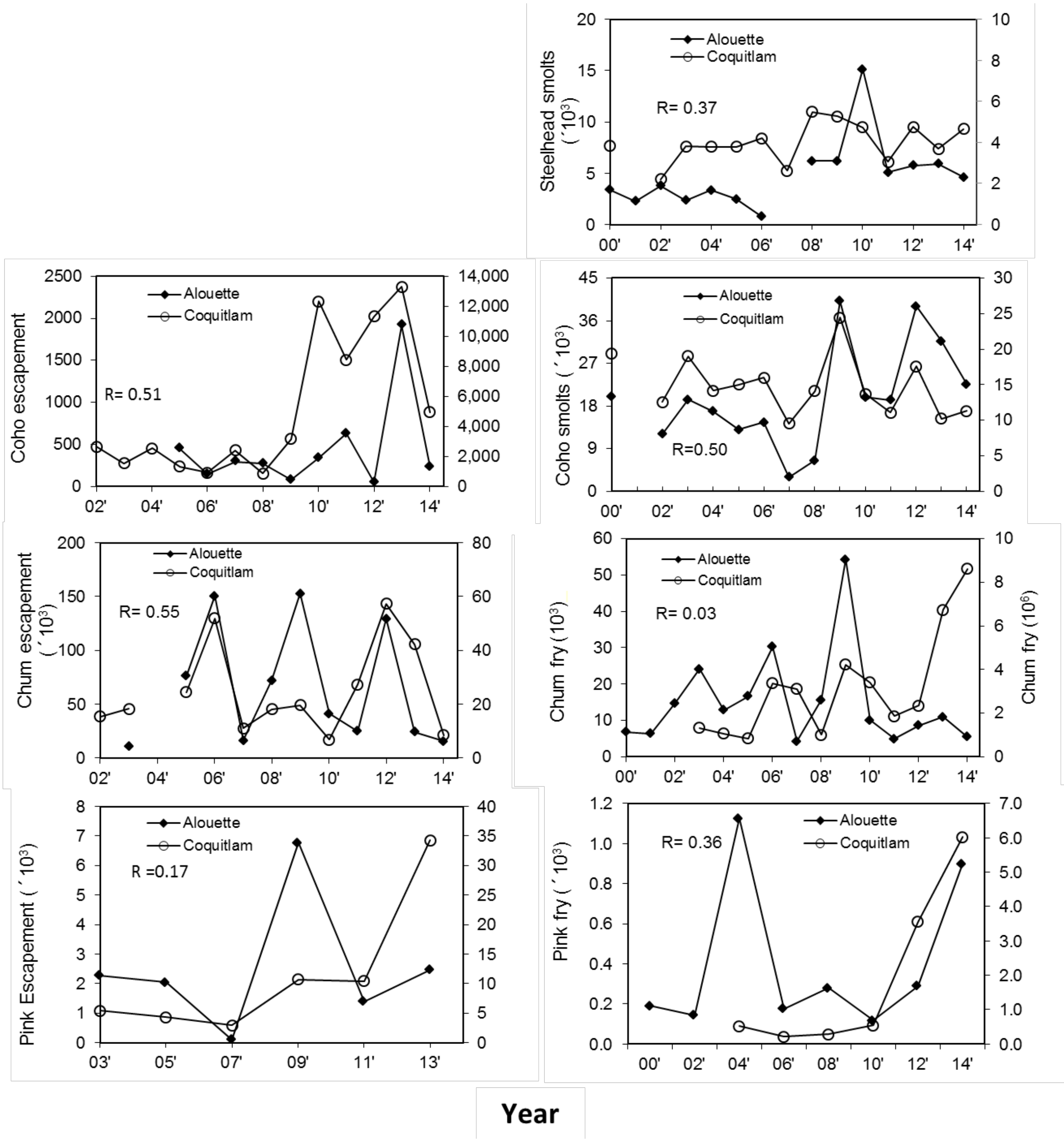

Figure 6.25a Annual escapement and smolt yield in the Coquitlam River study area versus that in the Alouette River during 2002-2014. Values for the Coquitlam are given on the right-hand axis, and values for the Alouette are given on the left-hand axis.

5. Smolt and Fry Outmigration

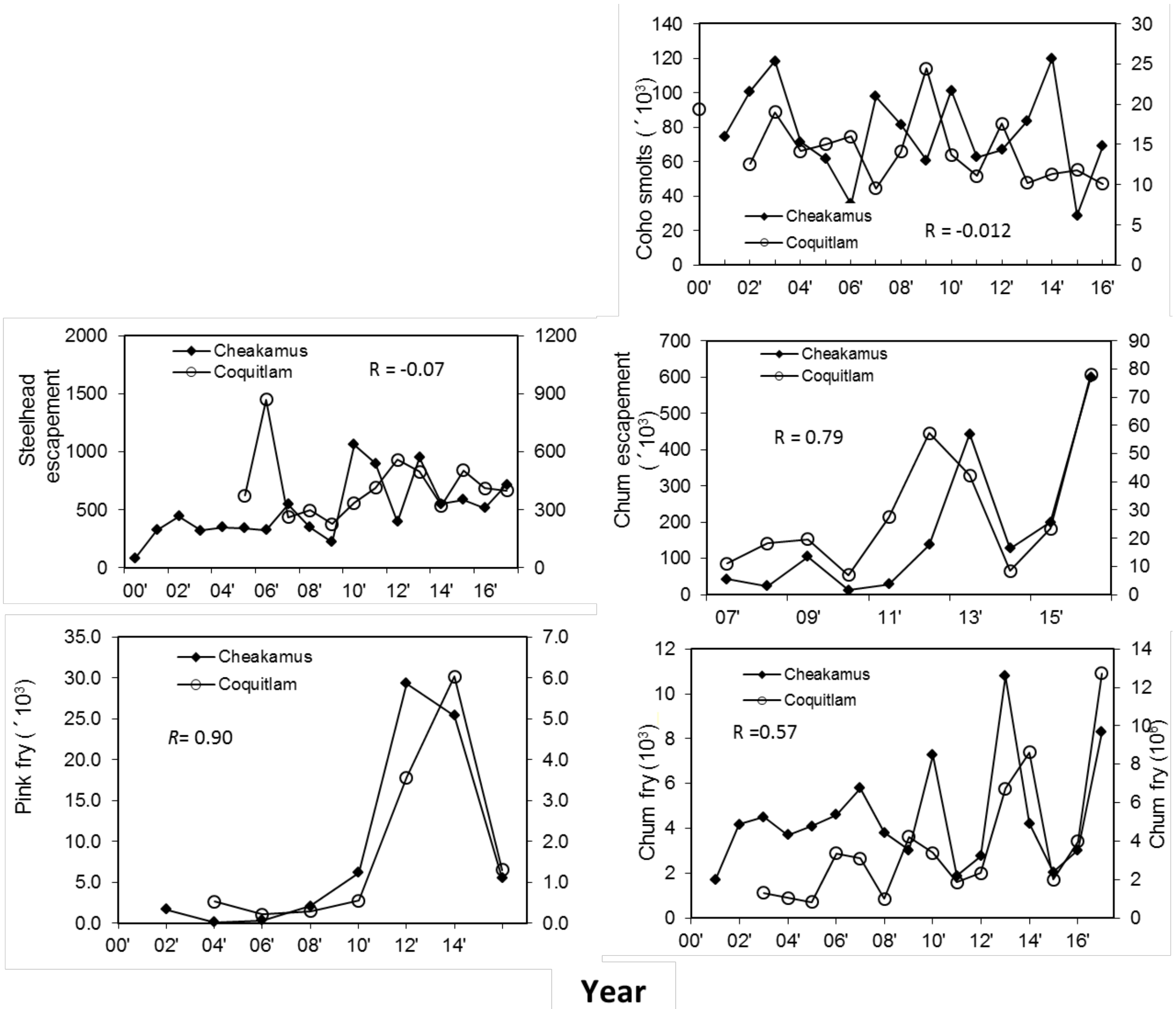

Figure 6.25b Annual escapement, fry and smolt yield in the Coquitlam River versus that in the Cheakamus River during 2002-2017. Values for the Coquitlam are given on the right-hand axis, and values for the Cheakamus are given on the left-hand axis.

Egg-to-fry survival was strongly influence by combined effect of flow treatment and the number of days winter flows exceeded 70m^3/s (Dec-Feb). Results from the comparison of models with one of several life-stage specific flow metrics (spawning, incubation and winter) both singly and in combination with flow treatment are listed in Table 6.8. Model 10 (Treatment + days above 70m^3/s Dec-Feb) had strong support ($\Delta AIC_c<2$) and the majority of the variance in survival (R^2 = 0.80). Mode 13 (Treatment + mean discharge Dec-Feb) was the second ranked model, had moderate support (ΔAIC_c=6.1) and had a R^2 = 0.69. This suggests survival increases with frequency and magnitude of flows during the incubation period. Results provided no support that flow during spawning had a meaningful impact on survival.

5. Smolt and Fry Outmigration

Table 6.8 Model performance of Chum egg-to-fry survival for 2002-2019 brood years by each of the explanatory variables evaluated using, linear (r^2), Akaike information criterion (AIC), and the distance from the lowest AIC value (ΔAIC). AIC include the penalty for the number of parameters at each sample size. Shading marks the model(s) with the most support based on ΔAICc.

Chum

Model #	Response variable	Explanatory Variable	R^2	AICc	Δ AIC
1	Survival (egg-to-fry)	Treatment	0.47	-26.4	9.6
2	Survival (egg-to-fry)	Days above 70 cms (Oct-Feb)	0.01	-17.6	18.4
3	Survival (egg-to-fry)	Days above 70 cms (Oct-Nov)	0.23	-21.1	14.9
4	Survival (egg-to-fry)	Days above 70 cms (Dec-Feb)	0.20	-20.6	15.4
5	Survival (egg-to-fry)	Mean discharge (Oct-Feb)	0.00	-17.5	18.5
6	Survival (egg-to-fry)	Mean discharge (Oct-Nov)	0.17	-20.1	15.9
7	Survival (egg-to-fry)	Mean discharges (Dec-Feb)	0.38	-24.2	11.8
8	Survival (egg-to-fry)	Treatment + Days above 70 cms (Oct-Feb)	0.49	-22.9	13.0
9	Survival (egg-to-fry)	Treatment + Days above 70 cms (Oct-Nov)	0.51	-23.3	12.7
10	Survival (egg-to-fry)	Treatment + Days above 70 cms (Dec-Feb)	0.80	-36.0	0.0
11	Survival (egg-to-fry)	Treatment + Mean discharge (Oct-Feb)	0.47	-22.4	13.6
12	Survival (egg-to-fry)	Treatment + Mean discharge (Oct-Nov)	0.53	-24.0	12.0
13	Survival (egg-to-fry)	Treatment + Mean discharges (Dec-Feb)	0.69	-29.9	6.1

5. Smolt and Fry Outmigration

6.5 Pink

Estimated adult Pink salmon returns to Coquitlam River ranged from 2,900-34,280 adults, with significantly increasing abundance starting in 2009 (Appendix 6.1a). Fry production upstream of RST2 ranged from 148,000-6,030,000 (Figure 6.26, Appendix 6.1a), with substantial increases since 2008. The egg-to-fry survival for 2003-2009 Pink broods (4.9%-10.1%, Appendix 6.1b) was comparable to the range reported for Pink populations in 18 other streams (mean: 7.4%; ±1 standard deviation: 3.2%-17.0%; Bradford 1995). However, the 2011, 2013 and 2015 brood egg-to-fry survival far exceeded this range (range: 27%-48%), which signals they could be non-credible or at least, biased high. An unrealistically high value would occur if escapement was biased low or if fry production was biased high. There were no indications of high bias in the escapement or fry estimates for these years, making it difficult to isolate the cause of the high survival rate. However, we have generally lower confidence in escapement estimates considering they depend heavily on assumptions about observer efficiency, survey life and fecundity (see section 2.2 on how this relates to bias and precision). We will gain a better understanding of the accuracy and precision of Pink escapement estimates if the escapement model is provided with sufficient observer efficiency and survey life information.

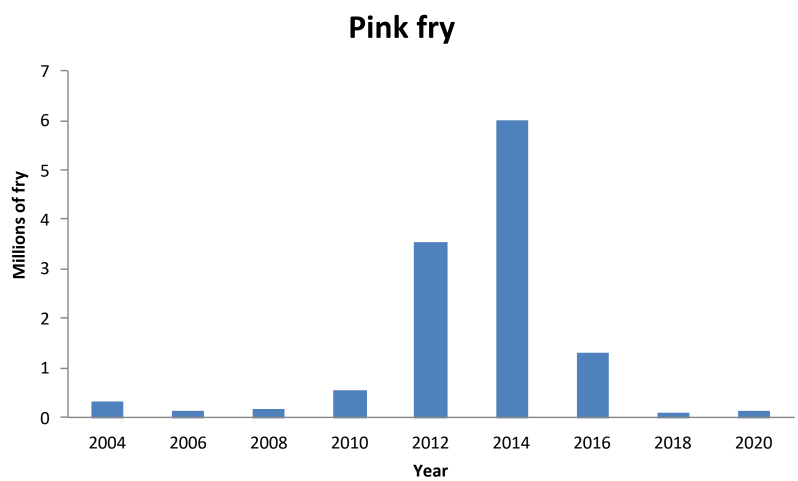

Figure 6.26 Estimated abundance of Pink fry outmigrating for the 7.5km study section of the Coquitlam River. Pink return to south coast rivers on odd years and thus, outmigrate during the spring of even years.

There was weak support for a linear escapement-to-fry stock-recruitment relationship during Treatment 1 ($R^2 = 0.36$) and strong support for one during Treatment 2 ($R^2 = 0.81$, Figure 6.27). However, the fit improved when both treatment periods were included ($R^2 = 0.87$), providing some support that the stock-recruitment relationship was similar during Treatment 1 and 2. The good fit of the single linear stock-recruitment relationship over a wide range of escapements suggests that fry abundance was minimally effected by the availability of spawning habitat. Given that all high escapement years were during Treatment 2, this provides no information as to whether high escapement under Treatment 1 flows would have resulted in a similar increase in fry abundance. It is still an important finding that freshwater carrying capacity was exceeded under Treatment 2 with adult escapement as high as 25,000 fish, but there is no information to indicate the same would not have occurred under Treatment 1 conditions.

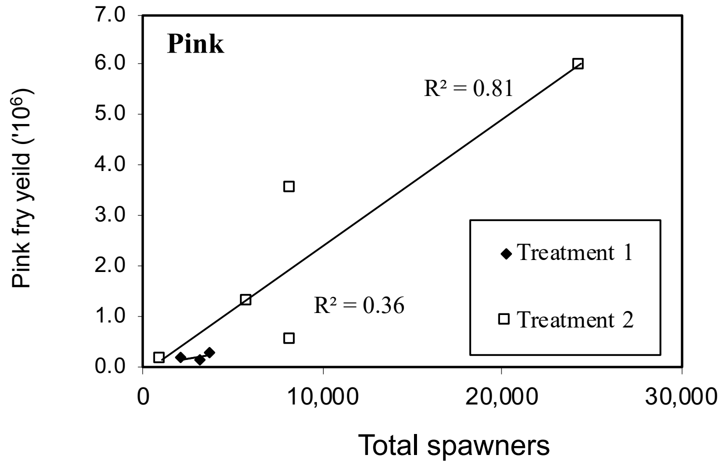

Figure 6.27 Escapement-to-fry stock-recruitment relationships of Pink Salmon during flow Treatment 1 (2002-2008) and Treatment 2 (2009-2019) from 7.5 km of the Coquitlam River. The best-fit lines intercept the x and y axis at 0 as is typical of stock-recruitment relationships. R^2 values reflect the fit of stock-recruitment relationship.

The large effect of adult escapement and lack of a clear indication of flow treatment effect was also found when comparing only years with low escapement. With the low escapement in 2009 and 2019 providing two years with low escapement under Treatment 2, it is now possible to compare egg-to-fry survival under similar escapement levels between Treatment 1 and 2. With these years combined, the fit of a linear stock-recruitment relationship also very high (R^2 = 0.90). Average egg-to-fry survival was 8.1% for Treatment 1 and 11.4% for Treatment 2 with broadly overlapping confidence intervals, indicting they are not significantly different.

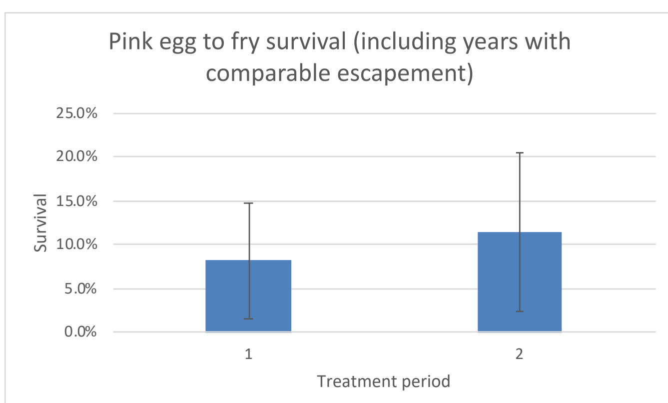

Figure 6.28 Mean egg-to-fry survival for Pink Salmon in the Coquitlam River during Treatment 1 (2003-2007 brood years) and Treatment 2 (2009 and 2019 brood years). The 2011-2017 brood years with escapement above 10,000 were excluded.

6. Fish Productivity during Treatment 1 and 2

Pink escapement was poorly correlated with that in the Alouette River (R=0.17, Figure 6.25a). As well, there was little correlation between Pink fry yield in the Coquitlam River and that in the Alouette River ($R^2 = 0.36$) but a strong correlation exists with the Cheakamus River fry production (R = 0.90, Figure 6.25b). Lacking escapement estimates for the Cheakamus, we cannot say this is because of similarities in escapement over this time period, but if it is, this would suggest that escapement is the largest driver of juvenile abundance.

While our ability to distinguish treatment effects from region-wide abundance trends remains low, results to date strongly suggest that adult escapement is the dominant driver of juvenile production in the Coquitlam River across both treatment periods. Given the lack of a reliable control stream, an alternative approach for distinguishing flow effect from the effects of escapement is to either return to Treatment 1 flow conditions or a switch to a third flow treatment. This would apply to the other three target species as well.

Pink salmon were successfully reintroduced to the Coquitlam River in 1995 following their extirpation in the 1960's. Increased minimum flows in Coquitlam River beginning in 1997 likely improved migration and spawning conditions for Pinks. There is some indication that larger dam releases under Treatment 2 have further improved access to spawning habitats for Pink salmon (Macnair 2010b) and may account for the lack of density dependent interactions under higher escapements during this period.

7 Conclusions

Study results suggest that the production of juvenile Coho, Chum and Pink Salmon as well as Steelhead have been higher in the Coquitlam River study area during Treatment 2 compared with Treatment 1, but only statistically higher for Steelhead and Chum. For smolts reared in the Coquitlam River mainstem, the mean increased by 27% for Coho and 29% for Steelhead. For Chum fry, this reflects a near doubling of the egg-fry productivity. For Pink fry, it is unclear whether egg-fry productivity increased during Treatment 2, however, there is no indication that spawning habitat limited Pink productivity at high escapement levels during Treatment 2.

We are less certain about whether the increases were a product of the Treatment 2 flows or other factors, such as higher adult escapement or regional environmental changes. If the assumption is correct that these differences were primarily the result of the flow treatment, these results support a conclusion that Treatment 2 flows caused increased freshwater productivity for some species compared to Treatment 1. However, support for this assumption varies by species and is generally uncertain at this point. For Coho, the moderate effect of adult escapement on smolt abundance and the rise in smolt yield in other watersheds raises the possibility that the increase was more the result higher escapement rather than from the flow treatment. There is some indication that the level of ramping related stranding primarily due to the June rampdowns during Treatment 2 reduced fall fry abundance. Adult escapement also appears to explain the majority of the variation in Steelhead smolt abundance. However, there was no clear indication that month and season specific flow metrics, or flow treatment had a substantive effect on either adult-to-fall fry survival or fall fry-to-fall age-1 parr survival. For Chum, results to date suggest that the combination of adult abundance, flow treatment and duration of high flows during incubation are the main factors affecting juvenile abundance. For Pink, it is adult abundance alone that explains the 80-90% of the juvenile abundance during Treatment 1 and 2. The number of abundance estimates is at the minimum to reliably compare the influence of multiple factors.

Current answers to the primary management question:

What are the fisheries benefits associated with each of the proposed test flows evaluated over the review period?

Coho – Mean smolt yield likely increased from Treatment 1 to Treatment 2 (mean increase 27%). Maximum smolt capacity was relatively unchanged between treatments but there were a higher number of years with abundance near the maximum carrying capacity during Treatment 2. It remains unclear at this time whether the increased productivity was the result of the Treatment 2, but there is support that adult escapement has a large effect.

Steelhead – Mean smolt yield very likely increased from Treatment 1 to Treatment 2 (mean increase 29%). Maximum smolt capacity was relatively unchanged between treatments but there were a higher number of years with abundance near the maximum carrying capacity during Treatment 2. It remains unclear at this time whether the increased productivity was the result of the Treatment 2. There is support that adult escapement had a large effect on smolt productivity.

There was no clear indication of the effect of flow treatment or other seasonal flow metrics on adult-to-fall fry survival or fall fry-to-fall age-1 parr survival.

Chum – Egg-fry recruitment increased significantly from Treatment 1 to Treatment 2, even when accounting for the number of adult spawners. While we lack information to confirm that this change was the result of the flow treatment, there is strong support that productivity was largely the result of adult escapement, flow treatment and high discharge during incubation.

Pink – Egg-fry recruitment possibly increased from Treatment 1 to Treatment 2 but this may be a product of the up to 10-fold higher spawner abundance during Treatment 2. While there was no clear effect of flow treatment on productivity, there was no indication that juvenile carrying capacity was reached during Treatment 2. Juvenile production was almost entirely the result of adult escapement and showed no indication of habitat limitations during Treatment 2.

7. Conclusions

8 Recommendations

8.1 Adult Salmon escapement

1. If adult escapement for Chum and Pink continue, conduct at least four mark-recapture experiments per year for Chum and for Pink during odd years, prioritizing for Chum (see section 6.5). Data derived from these experiments is critical for generating reliable estimates of observer efficiency and survey life. The lack of this information limits our ability to confirm the accuracy of the escapement estimates or to report on the precision of the escapement estimates. With the project nearing its end and unpredictable river conditions from year-to-year, a comprehensive approach to obtaining this information during years with favorable river conditions has the best chance of obtaining sufficient data.

2. If adult escapement for Chum and Pink continue, continue reconnaissance surveys at the beginning of the arrival of Pinks during odd years in order to confirm the absence of spawners from the study area prior to the first survey. This is needed to minimize uncertainty in the arrival and departure timing models. Pink assessments for access are normally conducted during the low-flow period in late August– early September period as a separate requirement of the Coquitlam-Buntzen WUP, but could also serve as reconnaissance surveys to determine the start date of the Pink run in odd years.

8.2 Adult Steelhead escapement

3. If flow increases for Sockeye outmigration are continued, increase the frequency of redd surveys to weekly from bi-weekly to reduce the possibility of redds being obscured due to scour.

8.3 Juvenile salmonid standing stock

4. As much as possible, continue sampling at least 24 sites to maintain adequate precision.

5. Mark-recapture experiments no longer need to be conducted for any age-class but for age-2+ Steelhead since the Coquitlam River-specific model of snorkeling detection probability is sufficiently refined for all but this species age-class. If we find that precision would improve from further refining the detection probability, benefits from this would be applied to all previous sampling.

8.4 Smolt and fry outmigration

6. Continue to maximize the number of Steelhead recaptures at RST2 by maintaining high capture efficiency at RST2 and smolt marking at all RSTs (2-4). The length of the trapping period and the trap configurations and locations for Coho and Steelhead were adequate to include the vast majority of the smolt and fry outmigrations in recent years. A similar approach should be applied for future years. The repositioning of the RST3 trap to the pre-2018 location on private land should be pursued if possible, to re-establish the higher catch efficiency for that trap location. This would improve the precision of mainstem estimates and, for Steelhead, allow real-specific estimates for reaches 2 and 3.

9 References

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10 Appendices

10.1 Appendices for Chapter 2

Appendix 2.1 The number of hatchery Chinook smolts released in the Coquitlam River 2004-2018 (Scott Ducharme, DFO).

Release year	Number
2004	38,000
2005	142,244
2006	195,000
2007	171,050
2008	300,000
2009	200,456
2010	245,000
2011	122,943
2012	22,800
2013	50,000
2014	50,000
2015	93,116
2016	49,713
2017	50,000
2018	69,070

Appendix 2.2 Results of the 2006-2018 mark-recapture study to estimate observer efficiency and survey life for Chum, Pink, Coho and Chinook salmon in the Coquitlam River. Only shaded values provide estimates of mean observer efficiency, as they represent cases where the proportion of tagged fish detected was based on a complete survey of the study area within two days of tagging.

Species	Treat-ment	Year	Index site	Tag group	Tagging date	Recovery date	Duration (days)	Marks (*M*)	Recoveries (*R*)	*R/M*	Surveyor guess	% females	Recoveries by section A	B	C	D	E
chum	1	2006	below A	1	Oct 17	Oct 21-22	4.5	11	1	9%	0.90	18%	1	0	0	0	0
chum	1	2006	below A	1	Oct 17	Oct 31-Nov 1	14.5	11	0	0%	0.70	18%	0	0	0	0	0
chum	1	2006	C	2	Oct 19-20	Oct 21-22	1-3	89	61	69%	0.90	33%	0	0	0	49	12
chum	1	2006	C	2	Oct 19-20	Oct 31-Nov 1	11-13	89	1	1%	0.85	33%	0	0	0	1	0
chum	1	2006	C	3	Oct 24, 28-30	Oct 31-Nov 1	2.5-7.5	59	27	46%	0.85	44%	0	1	2	71	14
chum	1	2006	C	3	Oct 24, 28-30	Nov 30	31.5-36.5	59	0	0%	0.70	44%	0	0	0	0	0
chum	1	2007	A	1	Oct 11	Oct 13	2	33	11	33%	0.55	45%	7	4	0	0	0
chum	1	2007	A	1	Oct 11	Oct 26-27	15-16	33	3	9%	0.70	45%	1	0	0	1	1
chum	1	2007	A	2	Oct-25	Oct 26-27	1-2	62	27	44%	0.60	55%	22	4	0	0	1
chum	1	2007	A	2	Oct-25	Oct 31-Nov 1	6-7	62	19	31%	0.60	55%	16	1	0	2	0
chum	1	2008	A	1	Oct 15	Oct 16-17	1-2	81	43	53%	0.80	37%	27	3	8	5	0
chum	2	2008	A	1	Oct 15	Oct 23-24	7-8	81	18	22%	0.80	37%	14	0	2	2	0
chum	2	2008	A	1	Oct 15	Oct 28-29	13-14	81	2	2%	0.65	37%	0	0	1	1	0
chum	2	2008	A	1	Oct 15	Nov 4-5	20-21	81	0	0%	0.65	37%	0	0	0	0	0
chum	2	2008	A/D	2	Oct 21	Oct 23-24	1-2	93	34	37%	0.80	35%	15	1	1	14	3
chum	2	2008	A/D	2	Oct 21	Oct 28-29	7-8	93	37	40%	0.70	35%	10	2	3	15	7
chum	2	2008	A/D	2	Oct 21	Nov 4-5	14-15	93	3	3%	0.50	35%	0	2	0	0	1
chum	2	2008	A/D	2	Oct 21	Nov 15	24-25	93	0	0%	0.50	35%	0	0	0	0	0
chum	2	2010	A/D	1	Oct 19	Oct 20	1	64	32	50%	0.78	56%	14	0	3	15	0
chum	2	2010	A/D	1	Oct 19	Oct 23	4	64	22	34%	0.80	56%	8	0	0	11	3
chum	2	2010	A/D	1	Oct 19	Oct 31	12	64	4	6%	0.80	56%	0	0	0	4	0
chum	2	2010	A/D	1	Oct 19	Nov 4	16	64	0	0%	0.80	56%	0	0	0	0	0
chum	2	2011	A/D	1	Oct 24	Oct 25	1	70	40	57%	0.75	49%	28	1	1	10	0
chum	2	2011	A/D	1	Oct 24	Nov 1	8	70	12	17%	0.75	49%	6	1	2	2	0
chum	2	2011	A/D	1	Oct 24	Nov 7	14	70	0	0%	0.75	49%	0	0	0	0	0
chum	2	2013	A/D	1	Oct 17	Oct 18	1	88	57	65%	0.77	50%	22	0	0	29	6
chum	2	2013	A/D	1	Oct 18	Oct 23	6	88	31	35%	0.77	50%	14	1	2	9	5
chum	2	2013	A/D	1	Oct 19	Oct 29	12	88	6	7%	0.77	50%	1	0	0	4	1
chum	2	2013	A/D	2	Oct 28	Oct 29	1	92	60	65%	0.77	48%	33	2	0	23	2
chum	2	2013	A/D	2	Oct 29	Nov 6	9	92	22	24%	0.77	48%	9	2	1	7	3
chum	1	2015	A/D	1	19-Oct	20-Oct	1	133	88	66%	74.2%	0.579	45	7	7	25	4
chum	1	2015	A/D	1	19-Oct	25-Oct	6	133	73	55%	74.0%	0.579	32	6	7	21	7
chum	1	2015	A/D	1	19-Oct	03-Nov	15	133	13	10%	65.0%	0.579	6	1	0	4	2
pink	1	2007	A	1	Oct 9-11	Oct 13	2-4	45	23	51%	0.55	22%	19	4	0	0	0
pink	1	2007	A	1	Oct 9-11	Oct 26-27	17-19	45	0	0%	0.65	22%	0	0	0	0	0
pink	2	2009	A/D	1	Sept 22	Sept 23	1	32	23	72%	0.95	59%	4	2	7	9	1
pink	2	2009	A/D	1	Sept 22	Oct 7	15	32	6	19%	0.85	59%	2	0	1	2	1
pink	2	2009	A/D	1	Sept 22	Oct 12	20	32	3	9%	0.85	59%	1	0	0	1	1
pink	2	2009	A/D	1	Sept 22	Oct 28	36	32	0	0%	0.85	59%	0	0	0	0	0

Appendix 2.2. continued

Species	Treat-ment	Year	Index site	Tag group	Tagging date	Recovery date	Duration (days)	Marks (*M*)	Recoveries (*R*)	*R/M*	surveyor guess	% females	Recoveries by section A	B	C	D	E
pink	2	2009	A/B	2	Oct 6	Oct 7	1	79	39	49%	0.85	65%	11	9	1	14	4
pink	2	2009	A/B	2	Oct 6	Oct 12	6	79	41	52%	0.85	65%	17	0	11	8	5
pink	2	2009	A/B	2	Oct 6	Oct 28	22	79	0	0%	0.80	65%	0	0	0	0	0
pink	2	2013	D	1	Sept 26	Sept 27	1	142	120	85%	0.87	33%	1	0	1	76	42
pink	2	2013	D	1	Sept 26	Oct 5	8	142	59	42%	0.70	33%	1	2	1	23	32
pink	2	2013	D	1	Sept 26	Oct 10	13	142	31	22%	0.77	33%	1	1	1	7	21
pink	2	2013	D	1	Sept 26	Oct 18	21	142	5	4%	0.77	33%	0	0	0	3	2
pink	2	2013	A/D	2	Oct 17	Oct 18	1	35	25	71%	0.77	40%	7	1	1	14	2
pink	2	2013	A/D	2	Oct 18	Oct 23	6	35	7	20%	0.75	40%	3	0	0	2	2
pink	2	2015	D	1	22-Sep	23-Sep	1	77	52	68%	1.00	51.9%			34	18	4.1
pink	2	2015	D	1	22-Sep	30-Sep	8	77	27	35%	1.00	51.9%	1		15	10	3.2
pink	2	2015	D	1	22-Sep	03-Oct	12	77	17	22%	1.00	51.9%			8	9	6.4
pink	2	2015	D	1	22-Sep	07-Oct	16	77	4	5%	0.86	51.9%			1	3	6.1
pink	2	2015	A	2	29-Sep	30-Sep	1	102	66	65%		34.3%	2				3.2
pink	2	2015	A	2	29-Sep	03-Oct	5	102	63	62%		34.3%	4				6.4
pink	2	2015	A	2	29-Sep	07-Oct	9	102	23	23%		34.3%	3	2	3		6.1
pink	2	2015	A	2	29-Sep	15-Oct	17	102	2	2%		34.3%					6.8
chinook	2	2010	A/D	1	Oct 19	Oct 20	1	17	9	53%	0.75	41%	1	1	0	4	3
chinook	2	2010	A/D	1	Oct 19	Oct 23	4	17	6	35%	0.80	41%	1	0	1	3	1
chinook	2	2010	A/D	1	Oct 19	Oct 31	12	17	3	18%	0.72	41%	0	0	0	1	2
chinook	2	2010	A/D	1	Oct 19	Nov 4	16	17	0	0%	0.70	41%	0	0	0	0	0
chinook	2	2010	A/D	1	Oct 19	Nov 13	25	17	1	6%	0.65	41%	0	0	0	1	0
chinook	2	2010	A/D	1	Oct 19	Nov 23	35	17	0	0%	0.65	41%	0	0	0	0	0
chinook	2	2011	A/D	1	Oct 24	Oct 25	1	12	8	67%		25%	3	0	5	0	0
chinook	2	2011	A/D	1	Oct 24	Nov 1	8	12	3	25%		25%	1	0	0	2	0
chinook	2	2011	A/D	1	Oct 24	Nov 7	14	12	0	0%		25%	0	0	0	0	0
coho	2	2010	A/D	1	Dec 4	Dec 5	1	20	14	70%	0.60	60%	0	0	0	0	14
coho	2	2010	A/D	1	Dec 4	Dec 19	15	20	9	45%	0.60	60%	0	0	0	0	9
coho	2	2010	A/D	1	Dec 4	Dec 31	27	20	6	30%	0.60	60%	0	0	0	0	6
coho	2	2010	A/D	1	Dec 4	Jan 11	38	20	0	0%		60%	0	0	0	0	0
coho	2	2011	E	1	Nov 20	Nov 21	1	15	10	67%		47%	0	0	0	5	5
coho	2	2011	E	1	Nov 20	Dec 1	11	15	5	33%		47%	0	0	0	1	4
coho	2	2011	E	1	Nov 20	Dec 7	16	15	5	33%		47%	0	0	0	1	4
coho	2	2011	E	1	Nov 20	Dec 19	28	15	2	13%		47%	0	0	0	0	2
coho	2	2012	D/E	1	Nov 24	Nov 25	1	30	22	73%		44%	0	0	0	8	14
coho	2	2012	D/E	1	Nov 24	Nov 29	5	30	19	63%		44%	0	0	0	6	13
coho	2	2012	D/E	1	Nov 24	Dec 8	14	30	12	40%		44%	0	0	0	3	9
coho	2	2012	D/E	1	Nov 24	Dec 14	20	30	5	17%		44%	0	0	0	0	5

Appendix 2.3 Unadjusted live counts of Pink salmon during 2003-2019.

			No. sites	ınt of the number of adults present					
Year	Date	Run day	surveyed	site A	site B	site C	site D	site E	non-index
2003	16-Sep	13	5	1	0	0	19	0	-
2003	22-Sep	19	5	18	0	39	15	9	-
2003	27-Sep	24	5	255	227	266	12	10	-
2003	04-Oct	31	6	378	511	907	642	159	340
2003	10-Oct	37	5	243	12	135	504	379	-
2003	14-Oct	41	6	270	18	105	350	1200	77
2003	02-Nov	60	6	0	0	0	0	0	0
2003	07-Nov	65	5	0	0	0	0	0	-
2003	13-Nov	71	3	0	0	0	-	-	-
2005	23-Sep	20	5	93	109	47	54	16	-
2005	05-Oct	32	5	201	37	149	294	403	-
2005	12-Oct	39	4	162	3	-	215	656	-
2005	24-Oct	51	6	34	0	13	59	356	29
2005	01-Nov	59	1	-	-	-	-	0	-
2005	09-Nov	67	2	-	0	-	0	-	-
2005	16-Nov	74	6	0	0	0	0	0	0
2007	04-Sep	1	5	0	0	0	0	2	-
2007	14-Sep	11	5	0	0	0	1	2	-
2007	20-Sep	17	5	0	1	6	4	2	-
2007	27-Sep	24	6	11	2	5	95	31	8
2007	03-Oct	30	5	128	31	53	222	233	-
2007	13-Oct	40	6	227	21	90	209	475	132
2007	17-Oct	44	2	-	-	-	152	329	-
2007	27-Oct	54	6	3	0	1	4	0	3
2007	31-Oct	58	6	0	0	1	0	2	0
2007	06-Nov	64	5	1	0	0	0	0	-
2007	29-Nov	87	5	0	0	0	0	0	-
2009	03-Sep	3	5	0	0	0	28	37	-
2009	12-Sep	12	5	46	24	50	223	56	-
2009	17-Sep	17	5	9	13	32	182	181	64
2009	23-Sep	23	5	68	181	86	180	435	114
2009	07-Oct	37	6	440	188	465	971	1071	283
2009	12-Oct	42	6	700	136	452	746	1299	264
2009	20-Oct	50	3	-	-	-	-	784	-
2009	28-Oct	58	6	1	7	0	2	88	0
2009	04-Nov	65	5	0	0	0	0	0	-

Appendix 2.3 continued (Pink)

Year	Date	Run day	No. sites surveyed	ount of the number of adults present site A	site B	site C	site D	site E	non-index
2011	04-Sep	4	5	6	0	0	7	31	-
2011	10-Sep	10	5	4	0	1	3	41	-
2011	17-Sep	17	5	27	3	11	12	43	-
2011	24-Sep	24	5	42	22	92	141	101	-
2011	10-Oct	40	6	550	75	89	505	1753	98
2011	17-Oct	47	6	590	39	352	794	1809	122
2011	25-Oct	55	6	97	2	19	98	311	43
2011	01-Nov	62	6	3	1	0	0	41	1
2011	06-Nov	67	6	0	0	0	0	4	0
2011	15-Nov	76	5	0	0	0	0	0	-
2013	16-Sep	16	5	50	223	20	27	26	-
2013	27-Sep	27	6	961	1074	2426	2800	1762	1082
2013	05-Oct	35	6	2024	773	1269	2894	4520	656
2013	11-Oct	41	6	4075	232	1429	2396	5099	782
2013	18-Oct	48	6	2679	55	907	1797	2681	593
2013	23-Oct	53	6	243	9	31	274	980	82
2013	29-Oct	59	6	7	0	0	0	27	1
2015	08-Sep	8	5	0	2	2	0	5	
2015	16-Sep	16	5	26	80	219	246	362	
2015	23-Sep	23	6	540	333	535	704	939	227
2015	30-Sep	30	6	1206	60	598	949	1300	637
2015	03-Oct	33	6	2036	66	566	853	1404	386
2015	07-Oct	37	6	1939	128	685	647	1539	331
2015	15-Oct	45	6	145	13	39	126	425	45
2015	20-Oct	50	6	45	2	15	43	110	0
2015	25-Oct	55	5	8	0	1	9	19	-
2019	09-Sep	9	5	0	0	2	2	1	
2019	17-Sep	17	5	0	2	5	15	12	
2019	21-Sep	21	5	21	4	48	167	128	
2019	27-Sep	27	5	30	25	84	49	258	
2019	02-Oct	32	6	62	2	49	34	479	54
2019	09-Oct	39	6	24	0	12	17	193	43
2019	15-Oct	45	5	4	2	0	7	45	
2019	24-Oct	54	5	1	0	0	3	3	
2019	30-Oct	60	6	0	0	0	0	1	0

Appendix 2.4 Unadjusted live counts of Chum salmon during 2002-2016 and 2018-2019.

			No. sites	Unadjusted count of the number of adults present					
Year	Date	Run day	surveyed	site A	site B	site C	site D	site E	non-index
2002	5-Oct	26	5	23	1	15	28	6	-
2002	11-Oct	32	5	83	17	48	120	7	-
2002	18-Oct	39	5	530	83	229	432	26	-
2002	22-Oct	43	5	1209	223	453	577	202	-
2002	31-Oct	52	6	1808	228	731	1416	361	330
2002	5-Nov	57	2	-	-	-	1294	117	-
2002	13-Nov	65	5	198	123	360	979	198	-
2002	24-Nov	76	5	29	0	98	97	64	-
2002	28-Nov	80	5	8	1	116	13	6	-
2002	5-Dec	87	5	3	0	2	4	0	-
2003	16-Sep	7	5	0	0	0	0	0	-
2003	22-Sep	13	5	0	0	0	0	0	-
2003	27-Sep	18	5	0	0	0	0	0	-
2003	4-Oct	25	5	120	13	6	0	0	-
2003	10-Oct	31	5	94	9	64	43	0	-
2003	14-Oct	35	6	231	7	213	594	52	82
2003	2-Nov	54	6	2172	422	502	1076	33	314
2003	7-Nov	59	5	3268	483	642	540	20	-
2003	13-Nov	65	3	1844	327	484	-	-	-
2003	22-Nov	74	5	177	149	165	115	0	-
2003	9-Dec	91	5	23	0	17	3	0	-
2003	16-Dec	98	5	0	0	0	0	0	-
2003	23-Dec	103	4	0	0	-	0	0	-
2003	30-Dec	110	5	0	0	0	0	0	-
2004	29-Sep	20	5	10	4	4	1	0	-
2004	5-Oct	26	5	60	14	6	11	0	-
2004	13-Oct	34	6	346	35	75	316	20	127
2004	20-Oct	41	5	928	175	279	766	38	-
2004	27-Oct	48	6	1727	392	863	1104	87	556
2004	5-Nov	57	5	3313	295	-	1577	239	649
2004	12-Nov	64	5	1857	520	1226	1502	242	-
2004	21-Nov	73	5	296	62	287	245	17	-
2004	30-Nov	82	5	23	1	16	38	0	-
2004	11-Dec	93	4	-	0	0	0	0	-
2004	23-Dec	103	5	0	0	0	0	0	-
2004	29-Dec	109	5	0	0	0	0	0	-
2005	23-Sep	14	5	2	0	0	0	0	-
2005	5-Oct	26	5	258	52	79	120	12	-
2005	12-Oct	33	4	719	50	-	383	175	-
2005	24-Oct	45	6	2230	393	1080	1059	283	547
2005	1-Nov	53	1	-	-	-	-	290	-
2005	9-Nov	61	2	-	95	-	472	-	-
2005	16-Nov	68	6	505	104	95	280	26	64
2005	24-Nov	76	5	183	24	104	16	0	-
2005	4-Dec	86	5	18	0	24	0	0	-
2005	9-Dec	91	5	0	0	0	0	0	-
2005	17-Dec	99	5	0	0	0	0	0	-
2005	23-Dec	103	4	0	0	-	0	0	-
2005	28-Dec	108	4	0	0	-	0	0	-

Appendix 2.4 continued (Chum)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present					
				site A	site B	site C	site D	site E	non-index
2006	27-Sep	18	5	40	0	2	2	0	-
2006	4-Oct	25	5	187	34	49	97	3	-
2006	11-Oct	32	6	1544	391	548	1241	258	1034
2006	22-Oct	43	5	3844	900	1152	3137	1123	-
2006	31-Oct	52	6	3657	737	1408	3180	1048	1318
2006	30-Nov	82	5	48	6	63	47	0	-
2006	8-Dec	90	2	-	-	-	0	0	-
2006	15-Dec	97	2	-	-	-	0	0	-
2006	24-Dec	106	2	-	-	-	0	0	-
2006	30-Dec	112	3	-	-	0	0	0	-
2007	14-Sep	6	5	0	0	0	0	0	-
2007	20-Sep	12	11	0	0	0	0	0	-
2007	27-Sep	19	19	0	0	0	0	0	-
2007	3-Oct	25	24	18	3	2	2	0	-
2007	13-Oct	35	34	97	28	31	170	5	48
2007	17-Oct	39	38	-	-	-	313	39	-
2007	27-Oct	49	48	742	144	363	595	121	155
2007	31-Oct	53	52	939	220	406	457	141	124
2007	6-Nov	59	58	603	143	281	373	114	-
2007	29-Nov	82	81	44	2	10	6	0	-
2007	5-Dec	88	87	-	-	-	-	0	-
2007	21-Dec	104	103	0	0	0	0	0	-
2008	29-Sep	21	5	9	11	4	11	20	-
2008	6-Oct	28	5	40	9	18	102	28	-
2008	10-Oct	32	6	208	20	110	85	49	86
2008	17-Oct	39	6	841	80	245	438	83	127
2008	23-Oct	45	6	1096	95	336	730	246	231
2008	29-Oct	51	6	1316	156	393	1019	455	247
2008	5-Nov	58	5	959	353	300	828	608	-
2008	15-Nov	68	5	123	106	159	392	148	-
2008	24-Nov	77	5	17	1	26	17	1	-
2008	4-Dec	87	5	0	3	3	0	0	-
2008	9-Dec	92	5	0	0	0	0	0	-
2009	12-Sep	4	5	0	0	0	0	0	-
2009	17-Sep	9	5	0	0	0	0	0	0
2009	23-Sep	15	6	2	4	0	0	0	0
2009	7-Oct	29	6	57	5	31	82	24	42
2009	12-Oct	34	6	505	75	108	127	37	95
2009	20-Oct	42	3	-	-	-	-	749	-
2009	28-Oct	50	6	2585	247	1131	1870	1031	321
2009	4-Nov	57	5	1042	279	1014	1161	454	-
2009	12-Nov	65	5	180	57	244	276	34	-
2009	24-Nov	77	3	-	-	0	8	17	-
2009	5-Dec	88	5	0	0	0	0	0	-
2010	3-Sep	1	5	3	0	0	0	0	-
2010	10-Sep	8	6	4	0	0	0	0	0
2010	21-Sep	19	5	0	2	0	4	0	-
2010	5-Oct	33	5	50	17	54	37	57	-
2010	12-Oct	40	6	311	35	118	283	191	89
2010	20-Oct	48	6	331	40	102	474	305	165
2010	23-Oct	51	6	553	33	119	388	288	278
2010	31-Oct	59	5	-	37	119	415	257	57
2010	4-Nov	63	6	176	42	108	382	139	51
2010	13-Nov	72	4	61	-	53	86	1	-
2010	23-Nov	82	5	0	0	0	2	0	-
2010	29-Nov	88	4	-	0	0	0	0	-

Appendix 2.4 continued (Chum)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present					
				site A	site B	site C	site D	site E	non-index
2011	04-Sep	2	5	0	0	0	0	0	-
2011	10-Sep	8	5	1	0	0	0	0	-
2011	17-Sep	15	5	0	0	1	0	0	-
2011	24-Sep	22	5	1	0	0	0	1	-
2011	10-Oct	38	6	238	51	63	36	7	9
2011	17-Oct	45	6	790	66	144	229	32	14
2011	25-Oct	53	6	3056	224	557	849	434	495
2011	01-Nov	60	6	6757	575	702	794	183	493
2011	06-Nov	65	6	3785	240	475	446	162	257
2011	15-Nov	74	5	692	132	185	230	42	-
2011	21-Nov	80	5	238	21	114	67	6	-
2011	01-Dec	90	6	23	5	33	8	0	0
2011	07-Dec	96	5	-	0	5	0	0	0
2011	19-Dec	108	5	-	0	0	0	0	0
2012	10-Sep	8	5	1	0	0	0	0	-
2012	17-Sep	15	5	3	0	0	1	0	-
2012	24-Sep	22	5	1	0	0	0	5	-
2012	30-Sep	28	5	81	2	4	20	31	-
2012	8-Oct	36	6	1349	93	747	1475	361	477
2012	14-Oct	42	3	-		928	1808	403	-
2012	15-Nov	74	6	224	214	108	273	65	27
2012	18-Nov	77	5	154	72	89	203	35	-
2012	25-Nov	84	6	25	7	25	11	0	13
2012	3-Dec	92	4	3	0	-	6	0	-
2012	9-Dec	98	6	0	0	1	0	0	0
2013	16-Sep	14	5	2	0	1	5	4	-
2013	27-Sep	25	6	14	10	20	10	0	0
2013	05-Oct	33	6	73	41	40	105	66	10
2013	11-Oct	39	6	570	57	89	207	159	68
2013	18-Oct	46	6	1928	127	490	1003	447	537
2013	23-Oct	51	6	3073	527	1020	1849	804	693
2013	29-Oct	57	6	4273	767	1288	3353	1136	681
2013	05-Nov	64	6	5212	534	1014	2110	980	605
2013	15-Nov	74	5	1682	88	353	885	380	-
2013	21-Nov	80	5	114	45	115	155	9	-
2013	27-Nov	86	5	27	5	33	8	0	-
2013	04-Dec	93	6	0	0	0	0	1	0
2013	11-Dec	100	5	0	2	0	0	1	-
2014	17-Sep	15	5	0	0	0	0	3	-
2014	26-Sep	24	6	4	0	0	1	2	0
2014	07-Oct	35	5	51	3	15	29	16	-
2014	14-Oct	42	2	-	-	-	27	62	-
2014	18-Oct	46	6	292	0	72	100	112	72
2014	23-Oct	51	3	-	-	76	174	148	-
2014	30-Oct	58	3	-	-	119	477	208	-
2014	02-Nov	61	6	1656	201	127	513	272	85
2014	13-Nov	72	5	160	37	43	80	28	-
2014	18-Nov	77	6	52	12	23	16	21	36
2014	29-Nov	88	3	-	-	11	1	0	-

Appendix 2.4 continued (Chum)

			No. sites	Unadjusted count of the number of adults present					
Year	Date	Run day	surveyed	site A	site B	site C	site D	site E	non-index
2015	16-Sep	14	5	6	0	1	0	0	
2015	23-Sep	21	6	4	1	2	0	0	3
2015	30-Sep	28	6	30	3	6	6	7	16
2015	03-Oct	31	6	164	11	20	21	14	37
2015	07-Oct	35	6	453	16	28	13	16	93
2015	15-Oct	43	6	2604	186	247	482	113	174
2015	20-Oct	48	6	2945	383	672	1018	289	435
2015	25-Oct	53	5	3382	627	902	1531	447	-
2015	03-Nov	62	5	1451	755	323	585	309	-
2015	09-Nov	68	5	705	309	156	233	72	-
2015	20-Nov	79	5	40	4	8	11	3	-
2016	05-Sep	3	5	0	0	2	0	0	-
2016	15-Sep	13	5	0	0	0	0	4	-
2016	21-Sep	19	5	0	0	0	1	3	-
2016	29-Sep	27	6	43	8	5	3	8	9
2016	05-Oct	33	5	85	22	41	135	23	-
2016	12-Oct	40	6	1691	57	178	535	361	187
2016	19-Oct	47	1	-	-	-	-	684	-
2016	30-Oct	58	6	5331	2014	2457	3384	1045	1541
2016	16-Nov	75	6	1643	522	1020	1196	259	316
2016	27-Nov	86	5	170	21	68	31	7	-
2016	04-Dec	93	5	10	0	4	6	1	-
2018	14-Sep	12	5	0	0	0	0	1	
2018	23-Sep	21	5	0	0	0	1	4	
2018	29-Sep	27	5	23	0	2	17	6	
2018	06-Oct	34	6	34	0	17	37	22	18
2018	13-Oct	41	5	394	22	35	167	87	
2018	19-Oct	47	6	609	44	97	481	211	154
2018	22-Oct	50	6	1170	76	142	680	318	216
2018	24-Oct	52	6	2117	175	209	999	441	352
2018	27-Oct	55	6	2251	270	285	1371	818	374
2018	08-Nov	67	5	1522	298	195	1096	490	
2018	13-Nov	72	3		195		80	263	
2018	20-Nov	79	5	236	66	99	96	98	
2018	27-Nov	86	5	38	11	31	13	14	
2018	06-Dec	95	5	4	0	1	5	4	
2018	12-Dec	101	4		0	1	0	0	
2019	21-Sep	19	5	0	5	9	1	6	
2019	27-Sep	25	6	0	8	11	3	15	
2019	02-Oct	30	5	6	16	8	3	12	0
2019	09-Oct	37	5	50	7	9	7	29	3
2019	15-Oct	43	6	75	17	48	10	60	
2019	24-Oct	52	5	838	108	313	92	326	
2019	30-Oct	58	6	897	121	372	110	340	103
2019	02-Nov	61	6	1008	114	411	115	327	119
2019	04-Nov	63	6	1778	120	533	154	411	130
2019	08-Nov	67	6	975	113	317	48	305	89
2019	14-Nov	73	5	442	136	201	26	134	0
2019	20-Nov	79	5	87	24	65	2	14	
2019	28-Nov	87	4	32	2	13	0	5	

Appendix 2.5 Unadjusted live counts of Coho salmon during 2002-2016 and 2018-2019.

			No. sites	Unadjusted count of the number of adults present					
Year	Date	Run day	surveyed	site A	site B	site C	site D	site E	non-index
2002	5-Oct	16	5	0	0	0	0	0	-
2002	11-Oct	22	5	0	0	0	0	0	-
2002	18-Oct	29	5	0	0	0	1	0	-
2002	22-Oct	33	5	0	0	0	1	0	-
2002	31-Oct	42	6	0	0	0	0	0	0
2002	5-Nov	47	2	-	-	-	0	0	-
2002	13-Nov	55	5	0	0	0	8	97	-
2002	24-Nov	66	5	0	0	0	80	192	-
2002	28-Nov	70	5	0	0	0	36	231	-
2002	5-Dec	77	5	0	0	0	88	189	-
2002	12-Dec	84	2	-	-	-	50	296	-
2002	18-Dec	90	3	-	0	-	70	268	-
2002	26-Dec	98	3	-	11	-	22	169	-
2002	12-Jan	115	3	-	7	-	1	35	-
2003	27-Sep	8	5	0	0	0	0	0	-
2003	4-Oct	15	5	0	0	0	0	0	-
2003	10-Oct	21	5	0	0	0	0	0	-
2003	14-Oct	25	6	0	0	0	0	0	0
2003	2-Nov	44	6	1	1	6	58	0	0
2003	9-Nov	51	5	0	18	3	62	81	-
2003	13-Nov	55	3	0	8	48	-	-	-
2003	22-Nov	64	5	0	1	3	55	97	-
2003	9-Dec	81	5	0	50	2	50	135	-
2003	16-Dec	88	5	0	19	0	10	55	-
2003	23-Dec	95	4	0	0	-	1	44	-
2003	30-Dec	102	5	0	0	0	2	31	-
2003	5-Jan	108	5	0	0	0	0	1	-
2004	29-Sep	10	5	0	0	0	0	0	-
2004	5-Oct	16	5	2	0	0	2	0	-
2004	14-Oct	25	6	1	3	0	8	8	0
2004	21-Oct	32	5	1	0	0	15	0	-
2004	28-Oct	39	6	0	1	0	20	3	0
2004	5-Nov	47	4	1	2	-	25	13	9
2004	12-Nov	54	5	21	4	19	27	62	-
2004	21-Nov	63	5	13	0	65	50	110	-
2004	1-Dec	73	5	0	7	30	95	379	-
2004	11-Dec	83	4	-	16	0	38	76	-
2004	23-Dec	95	5	0	11	0	11	195	-
2004	29-Dec	101	5	0	5	0	6	94	-
2005	23-Sep	4	5	0	0	0	0	0	-
2005	5-Oct	16	5	0	0	0	0	0	-
2005	12-Oct	23	4	1	0	-	2	0	-
2005	24-Oct	35	6	0	0	0	0	4	0
2005	1-Nov	43	1	-	-	-	-	0	-
2005	9-Nov	51	2	-	0	-	0	-	-
2005	16-Nov	58	6	9	0	5	54	14	0
2005	24-Nov	66	5	19	9	50	10	7	-
2005	4-Dec	76	5	12	2	54	42	13	-

Appendix 2.5 continued (Coho)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present					
				site A	site B	site C	site D	site E	non-index
2005	9-Dec	81	6	32	0	55	70	7	6
2005	17-Dec	89	5	10	2	56	49	12	-
2005	23-Dec	95	4	0	4	-	33	65	-
2005	28-Dec	100	4	0	0	-	34	55	-
2005	5-Jan	108	4	0	0	-	19	2	-
2006	27-Sep	8	5	0	0	0	0	0	-
2006	4-Oct	15	5	0	0	2	0	0	-
2006	11-Oct	22	6	0	0	1	12	0	0
2006	22-Oct	33	5	1	2	20	18	4	-
2006	31-Oct	42	6	0	3	19	29	7	0
2006	17-Nov	59	1	-	-	-	-	27	-
2006	30-Nov	72	6	0	4	0	12	59	16
2006	8-Dec	80	2	-	-	-	9	37	-
2006	15-Dec	87	2	-	-	-	32	12	-
2006	24-Dec	96	2	-	-	-	23	18	-
2006	30-Dec	102	3	-	-	1	8	6	-
2006	16-Jan	119	2	-	-	-	0	1	-
2007	3-Oct	14	5	0	0	0	0	0	-
2007	13-Oct	24	6	2	0	0	2	0	0
2007	31-Oct	42	6	0	0	4	0	2	0
2007	6-Nov	48	5	0	5	0	6	6	-
2007	29-Nov	71	5	7	30	16	130	217	-
2007	21-Dec	93	5	0	14	8	76	99	-
2007	29-Dec	101	5	0	2	2	19	60	-
2007	4-Jan	107	2	-	-	-	9	39	-
2007	16-Jan	119	3	-	-	0	3	6	-
2007	26-Jan	129	3	-	-	0	0	0	-
2008	10-Oct	21	6	0	0	0	0	0	0
2008	17-Oct	28	6	2	0	2	0	0	0
2008	23-Oct	34	6	3	0	0	0	6	0
2008	29-Oct	40	6	0	0	0	3	14	0
2008	5-Nov	47	5	0	0	0	20	24	-
2008	15-Nov	57	5	6	11	14	8	95	-
2008	24-Nov	66	5	4	9	10	5	68	-
2008	4-Dec	76	6	0	4	8	60	103	2
2008	9-Dec	81	2	-	1	-	-	11	-
2008	15-Dec	87	4	-	0	4	25	41	-
2008	21-Dec	93	6	0	0	7	12	44	0
2008	29-Dec	101	3	-	-	3	7	17	-
2008	6-Jan	109	3	-	-	0	2	9	-
2008	14-Jan	117	3	-	-	0	0	5	-
2008	22-Jan	125	3	-	-	0	0	0	-
2009	28-Oct	39	6	0	0	0	0	0	0
2009	4-Nov	46	5	0	0	18	14	26	-
2009	12-Nov	54	5	0	0	11	8	122	-
2009	24-Nov	66	3	0	12	0	12	195	-
2009	5-Dec	77	5	0	7	26	52	431	-
2009	13-Dec	85	5	0	7	26	39	415	-
2009	20-Dec	92	2	-	-	-	15	161	-
2009	29-Dec	101	3	-	-	3	33	119	-
2009	7-Jan	110	3	-	-	0	13	36	-
2009	14-Jan	117	3	-	-	0	3	10	-
2009	26-Jan	129	2	-	-	-	0	0	-

Appendix 2.5 continued (Coho)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present site A	site B	site C	site D	site E	non-index
2010	21-Sep	2	5	0	0	0	0	0	
2010	5-Oct	16	5	0	0	0	0	8	0
2010	12-Oct	23	6	0	20	30	59	29	19
2010	20-Oct	31	6	0	12	19	60	106	10
2010	23-Oct	34	6	1	7	26	55	153	19
2010	31-Oct	42	5		3	0	121	237	34
2010	4-Nov	46	6	2	12	86	139	565	61
2010	13-Nov	55	4	3		137	162	761	-
2010	23-Nov	65	5	0	21	129	329	813	-
2010	29-Nov	71	4	0	7	64	203	863	77
2010	6-Dec	78	-	-	0	0	235	866	-
2010	19-Dec	91	-	-	25	21	181	567	-
2010	30-Dec	102	-	-	16	7	109	239	-
2010	11-Jan	114	-	-	3	0	31	64	-
2010	18-Jan	121	-	-	1.4	0	17	21	-
2011	24-Sep	5	5	0	0	0	0	0	-
2011	10-Oct	21	6	0	0	1	0	0	0
2011	17-Oct	28	6	0	0	0	24	5	2
2011	25-Oct	36	6	1	0	1	37	58	1
2011	1-Nov	43	6	0	12	25	132	128	4
2011	6-Nov	48	6	3	12	71	167	242	45
2011	15-Nov	57	6	20	27	108	218	318	39
2011	21-Nov	63	6	0	0	135	235	399	40
2011	1-Dec	73	6	0	3	40	184	596	38
2011	7-Dec	79	5	-	0	70	167	623	39
2011	19-Dec	91	5	-	0	20	103	426	26
2011	27-Dec	99	5	-	2	0	56	281	20
2011	2-Jan	105	5	-	14	3	38	194	17
2011	12-Jan	115	5	-	2	0	1	69	8
2011	21-Jan	124	5	-	0	0	0	10	2
2012	15-Nov	57	6	0	29	115	328	526	36
2012	18-Nov	60	5	0	11	75	357	538	-
2012	25-Nov	67	6	3	24	146	384	633	79
2012	3-Dec	75	4	4	67	-	267	820	-
2012	9-Dec	81	6	0	37	74	260	725	69
2012	14-Dec	86	4	-	35	50	137	800	-
2012	23-Dec	95	4	-	11	32	94	520	-
2012	30-Dec	102	5	-	1	13	62	290	12
2012	3-Jan	106	4	-	0	0	21	188	-
2012	11-Jan	114	4	-	-	0	7	87	3
2012	17-Jan	120	3	-	-	0	3	24	-
2012	24-Jan	127	3	-	-	0	1	9	-

Appendix 2.5 continued (Coho)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present site A	site B	site C	site D	site E	non-index
2013	11-Oct	22	6	0	0	0	0	2	0
2013	18-Oct	29	6	0	0	4	18	11	0
2013	29-Oct	40	6	0	0	13	18	69	0
2013	05-Nov	47	6	3	27	144	126	378	-
2013	15-Nov	57	5	19	47	204	148	449	-
2013	21-Nov	63	5	14	48	243	161	619	-
2013	27-Nov	69	5	13	54	265	262	708	-
2013	04-Dec	76	6	4	43	117	268	1044	131
2013	11-Dec	83	5	0	30	82	318	1060	-
2013	18-Dec	90	6	0	23	128	224	919	54
2013	29-Dec	101	4	-	90	54	129	582	-
2013	5-Jan	108	4	-	70	42	77	365	-
2013	14-Jan	117	4	-	29	10	36	123	-
2013	25-Jan	128	4	-	3	2	13	28	-
2014	07-Oct	18	5	0	0	0	0	9	-
2014	14-Oct	25	3	-	-	0	0	16	-
2014	18-Oct	29	3	-	-	0	2	21	3
2014	23-Oct	34	4	4		9	11	43	-
2014	30-Oct	41	4	-	4	11	13	56	-
2014	02-Nov	44	6	2	0	9	23	106	31
2014	13-Nov	55	5	12	36	134	170	482	-
2014	18-Nov	60	6	6	34	105	252	538	94
2014	29-Nov	71	3	9	23	67	141	495	-
2014	03-Dec	75	5	2	20	47	115	352	-
2014	08-Dec	80	6	0	18	21	89	221	16
2014	20-Dec	92	4	0	12	7	32	123	-
2014	29-Dec	101	4	0	7	4	8	67	-
2014	04-Jan	107	4	0	6	2	8	44	-
2014	09-Jan	112	4	0	3	0	8	25	-
2014	18-Jan	121	4	0	0	0	4	16	-
2014	23-Jan	126	3	0	-	0	1	6	-
2015	25-Oct	36	5	0	0	0	0	10	-
2015	03-Nov	45	5	0	25	0	20	31	-
2015	09-Nov	51	5	0	68	50	142	76	-
2015	20-Nov	62	5	0	51	36	311	388	-
2015	29-Nov	71	6	0	29	53	214	483	102
2015	04-Dec	76	3	-		35	150	352	-
2015	21-Dec	93	4	-	2	15	66	141	-
2015	29-Dec	101	4	-	0	13	25	55	-
2015	07-Jan	110	4	-	0	14	2	28	-
2015	13-Jan	116	4	-	0	5	2	21	-
2015	20-Jan	123	4	-	0	2	3	9	-
2016	12-Oct	23	6	1	7	0	0	27	16
2016	19-Oct	30	1	-	-	-	-	52	-
2016	31-Oct	42	6	1	0	11	66	186	19
2016	16-Nov	58	6	0	0	19	227	877	33
2016	27-Nov	69	4	0	23	80	229	665	-
2016	04-Dec	76	4	0	18	70	184	457	-
2016	10-Dec	82	4	0	12	60	144	408	-
2016	17-Dec	89	4	-	8	38	100	279	-
2016	22-Dec	94	4	-	6	20	59	159	-
2016	30-Dec	102	4	-	2	3	32	98	-
2016	06-Jan	109	4	-	1	3	15	68	-
2016	14-Jan	117	4	-	0	2	11	46	-
2016	20-Jan	123	4	-	0	0	6	23	-
2016	24-Jan	127	4	-	0	0	1	10	-

Appendix 2.5 continued (Coho)

			No. sites	Unadjusted count of the number of adults present					
Year	Date	Run day	surveyed	site A	site B	site C	site D	site E	non-index
2018	06-Oct	17	6	0	0	0	0	6	0
2018	13-Oct	24	5	0	1	0	0	90	0
2018	19-Oct	30	6	1	0	0	2	116	4
2018	22-Oct	33	6	1	1	7	1	233	9
2018	24-Oct	35	6	0	0	13	9	375	38
2018	27-Oct	38	6	0	0	15	41	321	46
2018	08-Nov	50	5	0	0	0	3	93	
2018	13-Nov	55	3		1		148	418	
2018	20-Nov	62	5	2	40	92	137	577	
2018	27-Nov	69	5	0	150	86	141	236	
2018	06-Dec	78	5	0	89	25	289	552	
2018	12-Dec	84	4		33	17	170	361	
2019	09-Oct	20	5	0	0	21	50	48	
2019	15-Oct	26	6	0	0	31	66	126	1
2019	24-Oct	35	5	0	0	51	90	5	
2019	30-Oct	41	6	4	0	15	97	382	65
2019	02-Nov	44	6	4	8	15	84	410	31
2019	04-Nov	46	6	7	22	69	176	490	17
2019	08-Nov	50	6	3	34	142	250	791	32
2019	14-Nov	56	5	71	46	136	263	809	
2019	20-Nov	62	5	14	50	95	345	1073	
2019	28-Nov	70	5	83	97	60	376	902	
2019	09-Dec	81	5	20	70	37	201	660	
2019	15-Dec	87	4		78	26	129	497	
2019	21-Dec	93	4		24	9	67	312	
2019	27-Dec	99	4		12	6	24	199	
2019	31-Dec	103	4		6	0	14	96	

Appendix 2.6 Unadjusted live counts of Chinook salmon during 2007-2016, 2018-2019.

			No. sites	unt of the number of adults present					
Year	Date	Run day	surveyed	site A	site B	site C	site D	site E	non-index
2007	27-Sep	10	5	0	0	0	0	0	-
2007	3-Oct	16	6	0	0	0	0	0	2
2007	13-Oct	26	5	1	0	0	22	44	-
2007	17-Oct	30	2	-	-	-	0 '	27 '	-
2007	27-Oct	40	6	2	0	34	7	134	5
2007	31-Oct	44	6	3	0	6	0	49	0
2007	6-Nov	50	5	0	0	0	1	22	-
2007	29-Nov	73	5	0	0	0	0	0	-
2008	23-Sep	6	5	5	5	0	0	0	-
2008	29-Sep	12	5	0	5	4	7	90	-
2008	6-Oct	19	5	6	0	1	22	166	-
2008	10-Oct	23	6	11	3	3	13	242	23
2008	17-Oct	30	6	3	1	24	36	190	10
2008	23-Oct	36	6	3	0	24	36	107	3
2008	29-Oct	42	6	0	0	0	9	68	0
2008	5-Nov	49	5	0	0	0	2	9	-
2008	15-Nov	59	5	1	0	0	1	2	-
2008	24-Nov	68	5	0	0	0	0	0	-
2009	3-Sep	1	5	0	0	0	0	0	-
2009	12-Sep	10	5	5	7	6	5	2	-
2009	17-Sep	15	5	2	0	2	10	12	-
2009	23-Sep	21	6	3	6	3	8	107	16
2009	7-Oct	35	6	7	6	9	81	250	35
2009	12-Oct	40	6	89	29	40	84	495	6
2009	20-Oct	48	3	-	-	-	-	263	-
2009	28-Oct	56	6	3	0	41	19	126	0
2009	4-Nov	63	5	0	0	0	0	27	-
2009	12-Nov	71	5	0	0	0	0	8	-
2009	24-Nov	83	3	-	-	0	0	3	0
2009	5-Dec	94	5	0	0	0	0	0	-
2010	3-Sep	1	5	0	0	0	0	0	-
2010	10-Sep	8	6	2	0	0	0	2	1
2010	21-Sep	19	5	0	0	0	5	5	-
2010	5-Oct	33	5	56	49	159	86	1025	-
2010	12-Oct	40	6	52	18	150	250	1036	292
2010	20-Oct	48	6	52	22	97	281	915	114
2010	23-Oct	51	6	69	4	86	343	911	105
2010	31-Oct	59	5		0	43	213	625	69
2010	4-Nov	63	6	25	0	30	101	331	34
2010	13-Nov	72	4	11		8	30	58	-
2010	23-Nov	82	5	0	0	0	1	10	-
2010	29-Nov	88	4		0	0	0	0	-
2011	10-Sep	8	5	1	0	0	0	0	-
2011	17-Sep	15	5	3	1	0	0	0	-
2011	24-Sep	22	5	10	9	31	38	32	-
2011	10-Oct	38	6	17	20	75	268	800	74
2011	17-Oct	45	6	12	5	95	246	730	66
2011	25-Oct	53	6	4	9	38	181	505	33
2011	1-Nov	60	6	53	21	44	103	167	36
2011	6-Nov	65	6	23	7	10	62	159	19
2011	15-Nov	74	5	7	3	11	16	22	-
2011	21-Nov	80	5	5	0	4	1	6	-
2011	1-Dec	90	6	0	0	0	0	2	0
2011	7-Dec	96	5	-	0	5	0	0	0

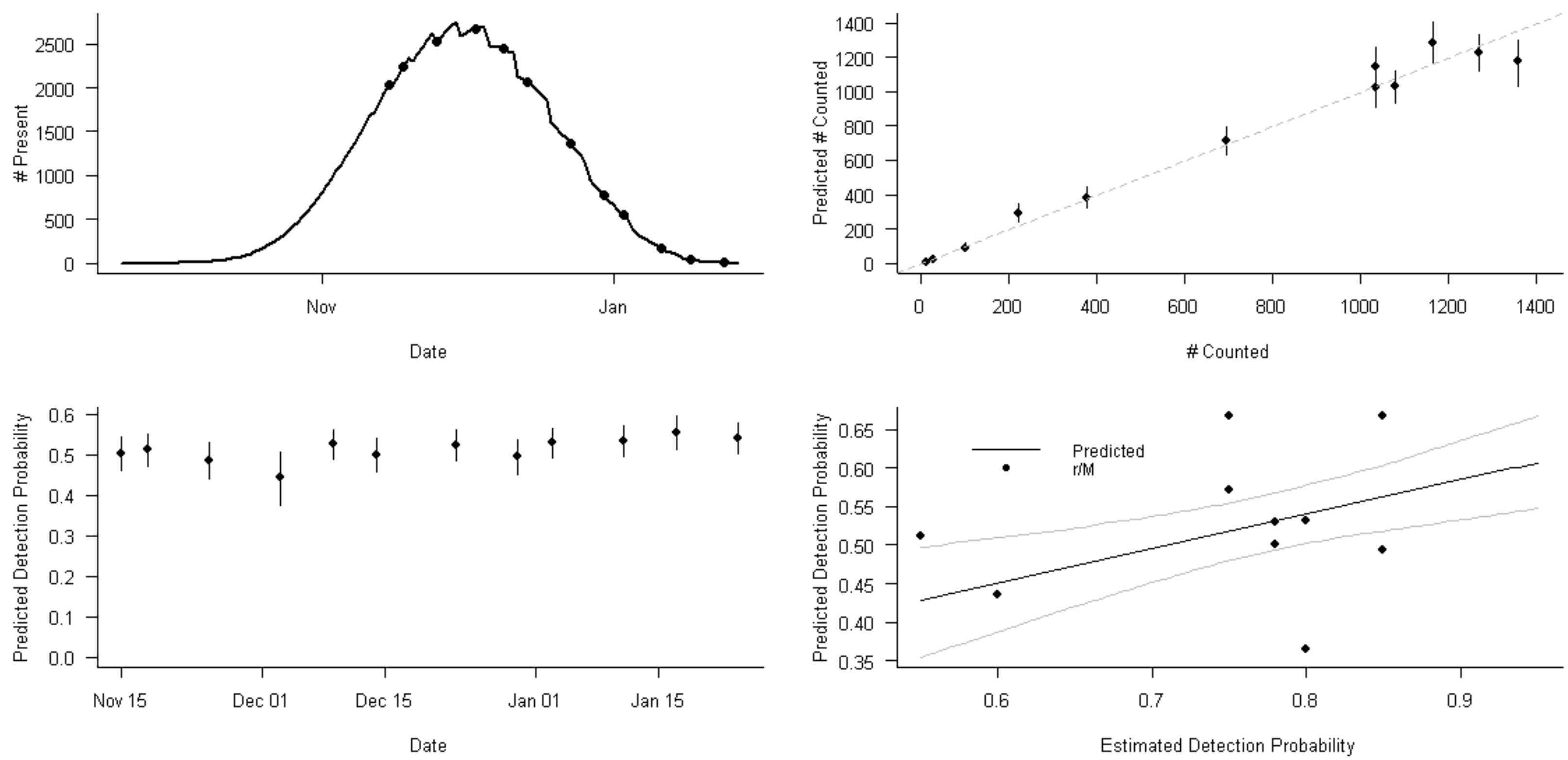
Appendix 2.6 continued (Chinook)

Year	Date	Run day	No. sites surveyed	unt of the number of adults present					
				site A	site B	site C	site D	site E	non-index
2012	10-Sep	8	5	1	0	0	0	0	-
2012	17-Sep	15	5	3	0	1	0	0	-
2012	24-Sep	22	5	3	0	0	0	34	-
2012	30-Sep	28	5	0	0	0	1	137	-
2012	08-Oct	36	6	6	0	0	22	246	18
2012	14-Oct	42	3	-	-	0	23	239	-
2012	15-Nov	99	6	0	0	0	0	6	0
2012	18-Nov	77	5	0	0	0	2	4	-
2012	25-Nov	84	6	0	0	0	1	1	0
2012	03-Dec	92	4	0	0	-	0	1	-
2012	09-Dec	98	6	0	0	0	0	0	0
2013	09-Sep	7	5	1	0	0	0	0	-
2013	16-Sep	14	5	3	0	1	0	0	-
2013	27-Sep	25	6	3	21	4	35	563	8
2013	05-Oct	33	6	9	3	22	53	533	49
2013	11-Oct	39	6	3	5	19	28	549	22
2013	18-Oct	46	6	2	1	32	42	345	24
2013	23-Oct	51	6	12	1	14	24	230	17
2013	29-Oct	57	6	7	3	11	14	146	0
2013	05-Nov	64	6	0	0	2	2	17	0
2013	15-Nov	74	5	0	0	0	3	12	-
2014	09-Sep	7	5	1	0	1	0	0	-
2014	17-Sep	15	5	0	0	2	0	5	-
2014	26-Sep	24	6	2	0	0	2	2	0
2014	07-Oct	35	5	3	0	4	2	103	-
2014	14-Oct	42	2	-	-	-	17	130	-
2014	18-Oct	46	6	0	0	4	12	113	0
2014	23-Oct	51	3	-	-	2	7	96	-
2014	30-Oct	58	3	-	-	11	5	36	-
2014	02-Nov	61	6	0	0	0	3	29	0
2014	13-Nov	72	5	0	0	0	0	8	-
2014	18-Nov	77	6	0	0	0	0	2	0
2015	16-Sep	14	5	0	0	2	3	1	
2015	23-Sep	21	6	0	0	2	2	1	0
2015	30-Sep	28	6	0	0	2	7	5	0
2015	03-Oct	31	6	1	0	3	8	8	0
2015	07-Oct	35	6	0	0	1	4	11	0
2015	15-Oct	43	6	0	0	0	0	20	0
2015	20-Oct	48	6	0	2	4	8	23	0
2015	25-Oct	53	5	0	0	3	1	31	-
2015	03-Nov	62	5	0	0	0	0	4	-
2015	09-Nov	68	5	0	0	0	0	1	-
2016	15-Sep	13	5	0	0	2	3	1	-
2016	21-Sep	19	5	0	0	2	2	1	-
2016	29-Sep	27	6	0	0	2	7	5	0
2016	05-Oct	33	5	0	0	0	0	53	-
2016	12-Oct	40	6	7	0	0	2	89	7
2016	19-Oct	47	1	-	-	-	-	78	-
2016	30-Oct	58	6	0	0	0	0	53	2
2016	16-Nov	75	6	0	0	0	3	27	0
2016	27-Nov	86	5	0	0	0	0	4	-
2016	04-Dec	93	5	0	0	0	0	1	-

Appendix 2.6 continued (Chinook)

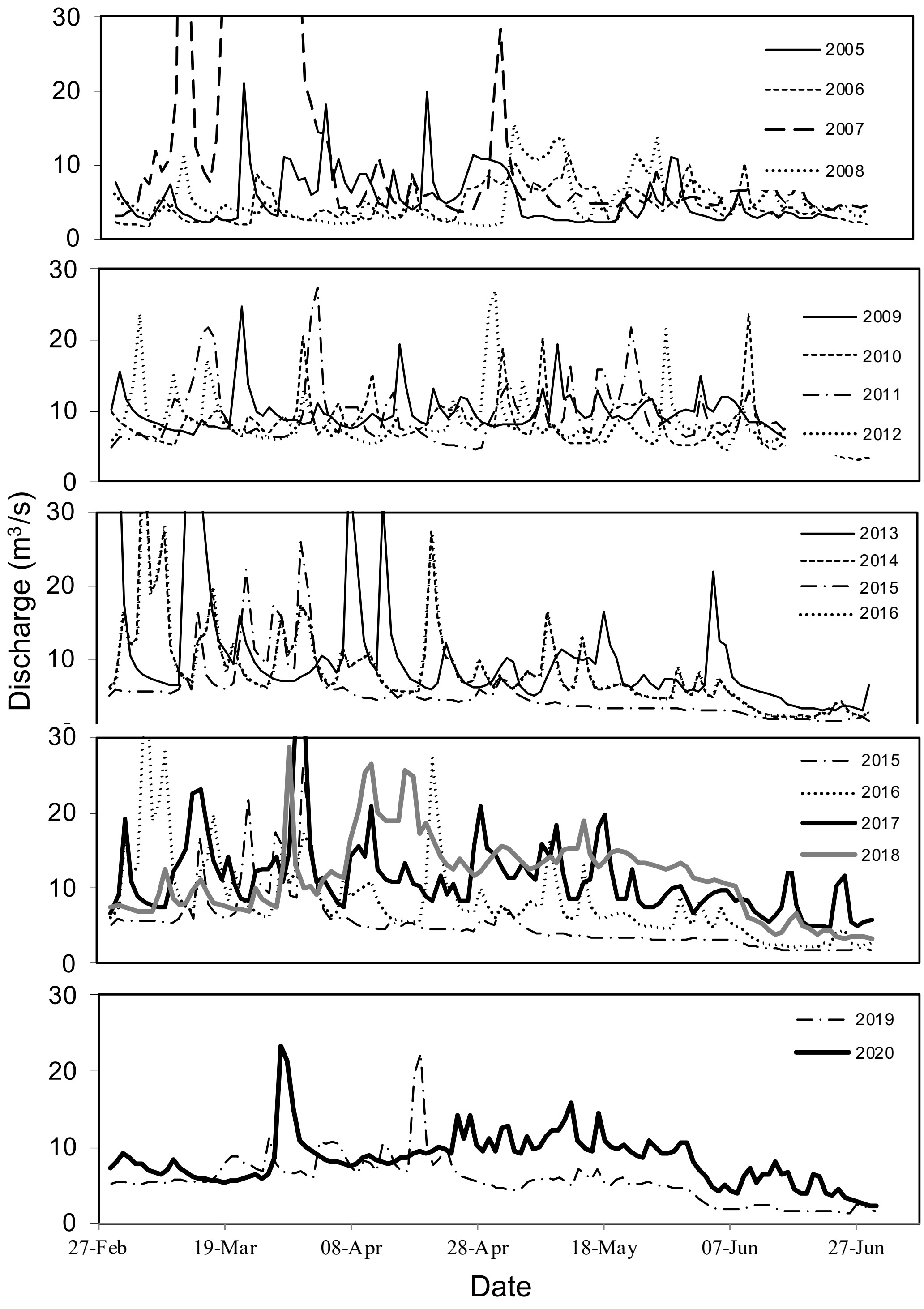
Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present					
				site A	site B	site C	site D	site E	non-index
2018	14-Sep	34	5	0	1	0	0	0	
2018	23-Sep	41	5	0	0	0	1	3	
2018	29-Sep	47	5	0	0	1	0	4	
2018	06-Oct	50	6	0	2	0	6	105	2
2018	13-Oct	52	5	1	4	11	16	103	
2018	19-Oct	55	6	1	0	3	19	96	7
2018	22-Oct	67	6	0	0	7	13	73	4
2018	24-Oct	72	6	0	0	8	9	58	2
2018	27-Oct	19	6	0	0	3	10	42	0
2018	08-Nov	25	5	0	0	0	4	14	
2018	13-Nov	30	3		0		0	8	
2019	21-Sep		5	0	0	3	13	0	
2019	27-Sep		6	0	0	7	10	27	0
2019	02-Oct		5	0	0	8	15	62	
2019	09-Oct		5	1	0	4	15	83	
2019	15-Oct		6	1	7	13	13	90	0
2019	24-Oct		5	4	2	10	14	66	
2019	30-Oct		6	2	0	4	22	39	0
2019	02-Nov		6	0	0	3	17	26	0
2019	04-Nov		6	0	0	1	7	23	0
2019	08-Nov		6	0	0	0	6	17	0
2019	14-Nov		5	0	0	0	4	9	
2019	20-Nov		5	0	0	0	1	4	
2019	28-Nov		5	0	0	0	0	1	

Appendix 2.7 An example of diagnostic graphs used to evaluate model fit to the observed data (Coho 2008). Top-left graph shows fit of predicted run timing curve (line) to unadjusted counts of spawners over time. Top-right shows relationship of predicted to observed counts with 95% credible intervals for predicted counts. Lower-left graph shows variation in predicted observer efficiency across surveys. Lower-right graph shows the regression relationship between surveyor guesstimates of observer efficiency (horizontal axis) and mark-recapture derived estimates of observer efficiency (vertical axis), with 95% credible intervals shown for the estimated regression slope.



10.2 Appendices for Chapter 3

Appendix 3.1 Discharge (m^3/s) in Coquitlam River at Port Coquitlam during Steelhead spawning period in 2005 – 2020 (Water Survey of Canada station 08MH002).



Appendix 3.2 Summary statistics for Steelhead escapement to Coquitlam River during 2005-2020 based on redd counts. Minimum and maximum range in escapement reflects uncertainty about the number of redds constructed by each female, and about sex ratio (see Section 3.1.4).

Year	Reach	Total number of redds	Redds /km	Total female spawners	Total egg deposition	Eggs /km	Total adult escapement	Range in escapement	Adults /km
2005	2a	30	7.1	25	92,000	22,000	50		12
	2b	76	23.8	63	234,000	73,000	127		40
	3	63	36.9	52	193,000	114,000	104		61
	4	55	32.6	46	171,000	101,000	92		54
	Total	224	20.7	187	691,000	64,000	373	(172-640)	35
2006	2a	72	17.0	60	220,000	52,000	119		28
	2b	215	67.0	179	661,000	207,000	358		112
	3	114	66.9	95	350,000	206,000	189		111
	4	121	71.4	101	374,000	220,000	202		119
	Total	521	48.2	434	1,606,000	149,000	868	(401-1,489)	80
2007	2a	25	6.0	21	77,000	18,000	42		10
	2b	64	20.0	53	197,000	62,000	106		33
	3	54	32.0	45	168,000	99,000	91		53
	4	13	7.4	10	39,000	23,000	21		12
	Total	156	14.4	130	481,000	45,000	260	(120-446)	24
2008	2a	42	10.0	35	130,000	31,000	70		17
	2b	84	26.3	70	259,000	81,000	140		44
	3	41	24.1	34	126,000	74,000	68		40
	4	11	6.5	9	34,000	20,000	18		11
	Total	178	16.5	148	549,000	51,000	297	(137-509)	
2009	2a	30	7.1	25	93,000	22,000	50		12
	2b	54	16.9	45	167,000	52,000	90		28
	3	35	20.6	29	108,000	64,000	58		34
	4	16	9.4	13	49,000	29,000	27		16
	Total	135	12.5	113	416,000	39,000	225	(104-386)	21
2010	2a	32	7.6	27	99,000	24,000	53		13
	2b	71	22.2	59	219,000	68,000	118		37
	3	66	38.8	55	204,000	120,000	110		65
	4	31	18.2	26	96,000	56,000	52		30
	Total	200	18.5	167	617,000	57,000	333	(154-571)	31
2011	2a	42	10.0	35	130,000	31,000	70		17
	2b	58	18.1	48	179,000	56,000	97		30
	3	84	49.4	70	259,000	152,000	140		82
	4	63	37.1	53	194,000	114,000	105		62
	Total	247	22.9	206	762,000	71,000	412	(190-760)	38
2012	2a	60	14.3	50	185,000	44,000	100		24
	2b	102	31.9	85	315,000	98,000	170		53
	3	102	60.0	85	315,000	185,000	170		100
	4	70	41.2	58	216,000	127,000	117		69
	Total	337	31.2	281	1,039,000	96,000	562	(257-954)	52

Appendix 3.2 cont'd

Year	Reach	Total number of redds	Redds /km	Total female spawners	Total egg deposition	Eggs /km	Total adult escapement	Range in escapement	Adults /km
2013	2a	24	5.6	20	73,000	17,000	39		9
	2b	91	28.6	76	282,000	88,000	152		48
	3	91	53.8	76	282,000	166,000	152		90
	4	90	53.2	75	279,000	164,000	151		89
	Total	297	27.5	248	916,000	85,000	495	(222-826)	46
2014	2a	30	7.1	25	93,000	22,000	50		12
	2b	60	18.8	50	185,000	58,000	100		31
	3	53	31.2	44	163,000	96,000	88		52
	4	47	27.6	39	145,000	85,000	78		46
	Total	190	17.6	158	586,000	54,000	317	(146-543)	29
2015	2a	37	8.8	31	114,000	27,000	62		15
	2b	102	31.9	85	315,000	98,000	170		53
	3	68	40.0	57	210,000	124,000	113		67
	4	94	55.3	78	290,000	171,000	157		92
	Total	301	27.9	251	928,000	86,000	502	(232-860)	46
2016	2a	32	7.6	27	99,000	24,000	53		13
	2b	86	26.9	72	265,000	83,000	143		45
	3	72	42.4	60	222,000	131,000	120		71
	4	56	32.9	47	173,000	102,000	93		55
	Total	246	22.8	205	759,000	70,000	410	(189-703)	38
2017	2a	27	6.4	23	83,000	20,000	45		11
	2b	97	30.3	81	299,000	93,000	162		51
	3	70	41.2	58	216,000	127,000	117		69
	4	45	26.5	38	139,000	82,000	75		44
	Total	239	22.1	199	737,000	68,000	398	(184-683)	37
2018	2a	12	2.9	10	37,000	9,000	20		5
	2b	52	16.3	43	160,000	50,000	87		27
	3	52	30.6	43	160,000	94,000	87		51
	4	45	26.5	38	139,000	82,000	75		44
	Total	161	14.9	134	496,000	46,000	268	(124-460)	25
2019	2a	33	7.9	28	102,000	24,000	55		13
	2b	51	15.9	43	157,000	49,000	85		27
	3	52	30.6	43	160,000	94,000	87		51
	4	39	22.9	33	120,000	71,000	65		38
	Total	175	16.2	146	540,000	50,000	292	(135-500)	27
2020	2a	15	3.6	13	46,000	11,000	25		6
	2b	47	14.7	39	145,000	45,000	78		24
	3	42	24.7	35	130,000	76,000	70		41
	4	27	15.9	23	83,000	49,000	45		26
	Total	131	12.1	109	404,000	37,000	218	(101-374)	20

Appendix 3.3 Survey dates with raw counts of Steelhead redds, estimated new redds, and live adult counts for all surveys during 2005-2020. Estimated new redds includes the sum of the raw count and the estimated number of redds that were constructed and then obscured by substrate movement prior to a scheduled survey, based on a redd survey life model.

Year	Survey date	Days since previous survey	Raw count of new redds	Estimated # new redds	# Live adults observed
2005	24-Mar	-	4	4	0
2005	13-Apr	20	81	84	2 [2]
2005	28-Apr	15	45	45	11 [2]
2005	07-May	9	71	71	22 [2]
2005	05-Jun	28	17	20	4
2005	**Total**		**218**	**224**	**peak = 22**
2006	15-Feb	-	0	0	29
2006	13-Mar	27	32 [1]	32	11
2006	19-Apr	37	285 [3]	368	95
2006	13-May	24	82	86	37
2006	12-Jun	29	31	35	3
2006	**Total**		**430**	**521**	**peak = 95**
2007	02-Mar	-	0	0	20
2007	04-Apr	32	5	9	45
2007	19-Apr	15	68	71	43
2007	30-Apr	11	25	25	33
2007	09-May	9	30	30	24
2007	22-May	13	13	13	13
2007	13-Jun	22	8	8	0
2007	**Total**		**149**	**156**	**peak = 45**
2008	21-Mar	-	24	24	17
2008	02-Apr	12	29	29	37
2008	13-Apr	11	35	35	24
2008	25-Apr	12	58	58	45
2008	09-May	14	20	20	28
2008	27-May	18	12	12	17
2008	13-Jun	17	0	0	3
2008	**Total**		**178**	**178**	**peak = 45**
2009	11-Mar	-	9 [1]	9	11
2009	04-Apr	24	13	13	25
2009	15-Apr	11	29	29	23
2009	25-Apr	10	31	31	37
2009	01-May	6	13	13	20
2009	15-May	14	37	37	24
2009	08-Jun	24	3	3	4
2009	**Total**		**135**	**135**	**peak = 37**

Appendix 3.3 continued

Year	Survey date	Days since previous survey	Raw count of new redds	Estimated # new redds	# Live adults observed
2010	09-Mar	-	7	7	33
2010	27-Mar	18	39	39	30
2010	13-Apr	17	60	60	51
2010	23-Apr	10	41	41	60
2010	05-May	12	28	28	44
2010	23-May	18	24	24	12
2010	14-Jun	22	1	1	1
2010	**Total**		**200**	**200**	**peak = 60**
2011	22-Mar	-	5	5	43
2011	05-Apr	14	21	21	61
2011	10-Apr	5	45	45	97
2011	20-Apr	10	83	83	103
2011	05-May	15	68	68	67
2011	21-May	16	24	24	36
2011	06-Jun	16	1	1	11
2011	**Total**		**247**	**247**	**peak = 103**
2012	8-Mar	-	1	1	38
2012	24-Mar	16	29	29	68
2012	7-Apr	14	100	100	95
2012	16-Apr	9	92	92	148
2012	2-May	16	85	87	76
2012	14-May	12	25	25	44
2012	7-Jun	24	2	3	11
2012	**Total**		**334**	**337**	**peak = 148**
2013	10-Mar	-	2	2	31
2013	28-Mar	18	32	34	59 [4]
2013	14-Apr	17	64	67	70 [4]
2013	24-Apr	10	94	95	113
2013	5-May	11	56	56	88
2013	20-May	15	34	35	47
2013	8-Jun	19	7	8	9
2013	**Total**		**289**	**297**	**peak = 113**
2014	15-Mar	-	3	3	29
2014	2-Apr	18	28	30	57
2014	13-Apr	11	83	83	88
2014	26-Apr	13	37	37	71
2014	7-May	11	11	11	63
2014	20-May	13	20	20	22
2014	8-Jun	19	8	9	6
	Total		**190**	**193**	**peak = 88**

Appendix 3.3 continued

Year	Survey date	Days since previous survey	Raw count of new redds	Estimated # new redds	# Live adults observed
2015	16-Mar	0	27	27	41
2015	27-Mar	11	12	12	57
2015	6-Apr	10	79	79	91
2015	17-Apr	11	76	76	117
2015	26-Apr	9	47	47	80
2015	10-May	14	46	46	61
2015	24-May	14	14	14	34
2015	4-Jun	10	0	0	8
	Total		**301**	**301**	**peak = 117**
2016	17-Mar	0	8	8	31
2016	27-Mar	10	31	31	59
2016	8-Apr	12	74	74	78
2016	20-Apr	12	51	51	90
2016	30-Apr	10	33	33	67
2016	12-May	12	34	34	48
2016	26-May	14	14	14	26
2016	8-Jun	13	1	1	7
	Total		**246**	**246**	**peak = 90**
2017	20-Mar	0	11	11	23
2017	31-Mar	11	37	37	49
2017	14-Apr	14	63	63	81
2017	22-Apr	8	33	33	77
2017	6-May	14	59	59	59
2017	17-May	11	17	17	41
2017	31-May	14	11	11	19
2017	7-Jun	7	8	8	6
	Total		**239**	**239**	**peak = 81**
2018	11-Mar	0	8	8	29
2018	30-Mar	19	52	52	51
2018	8-Apr	9	11	11	29
2018	16-Apr	8	43	43	61
2018	26-Apr	10	21	21	22
2018	11-May	15	14	14	0 [4]
2018	26-May	15	12	12	12
2018	7-Jun	12	0	0	0
	Total		**161**	**161**	**peak = 61**
2019	14-Mar	0	11	11	20
2019	27-Mar	13	29	29	34
2019	8-Apr	12	34	34	57
2019	17-Apr	9	24	24	69
2019	28-Apr	11	56	56	33
2019	12-May	14	15	15	21
2019	28-May	16	4	4	10
2019	9-Jun	12	2	2	0
	Total		**175**	**175**	**peak = 69**

Appendix 3.3 continued

Year	Survey date	Days since previous survey	Raw count of new redds	Estimated # new redds	# Live adults observed
2020	15-Mar	0	21	21	33
2020	25-Mar	10	11	11	40
2020	8-Apr	14	21	21	49
2020	19-Apr	11	41	41	44
2020	29-Apr	10	19	19	19
2020	11-May	12	9	9	15
2020	25-May	14	7	7	13
2020	9-Jun	15	2	2	0
	Total		**131**	**131**	**peak = 49**

[1]Redd survey incomplete due to poor conditions
[2] Live adult totals incomplete
[3]Redd totals from aborted April 13 survey added to April 19 survey
[4]Adult count incomplete due to poor survey conditions

Appendix 3.4 An example of how raw survey counts were expanded to account for redds that were completed and subsequently became undetectable between surveys (see section 3.2.1).

April 19, 2007 redd survey

Total # new redds observed	68
Number days from previous survey (CSI)	15
Number of redds constructed per day since previous sui assuming uniform distribution of spawning over time	4.53
Run day for the spawning period (***R***) (March 1 = day one)	50
Redd survey life equation	% redds lost = 0.029***CSI*** - 0.002***R*** - 0.1572

Day	Loss rate	Adjusted # redds
1	0.0000	4.53
2	0.0000	4.53
3	0.0000	4.53
4	0.0000	4.53
5	0.0000	4.53
6	0.0000	4.53
7	0.0000	4.53
8	0.0000	4.53
9	0.0070	4.57
10	0.0358	4.70
11	0.0646	4.85
12	0.0934	5.00
13	0.1222	5.16
14	0.1510	5.34
15	0.1798	5.53

10.3 Appendices for Chapter 4

Appendix 4.1 Definition of variables of the hierarchical Bayesian model used to estimate juvenile Coho and Steelhead abundance in the Coquitlam River system. Index sites refer to the 12 sites in the Coquitlam River where fish abundance is sampled each year by night snorkeling. Fish size strata (subscript g) apply only to Steelhead (see Section 4.1.5).

Variable	Description
Data	
$r_{i,g}$	Marks detected at snorkeling mark-recapture site i, fish size strata g
$m_{i,g}$	Marks released at mark-recapture site i, strata g
$c_{j,g}$	Fish detected at index site j for strata g
l_j	Stream length for index site j
Site-Specific Parameters	
$\theta_{i,g}$	Estimated detection probability at mark-recapture site i for fish size strata g
$\theta_{j,g}$	Simulated detection probability for index site j for strata g
λ_j	Estimated density (fish/m) at index site j
Hyper-Parameters	
$\mu_{\theta,g}$	Mean of beta hyper-distribution for detection probability for strata g
$\tau_{\theta,g}$	Precision of beta hyper-distribution for detection probability for strata g
μ_λ	Mean of normal hyper-distribution for log fish density
τ_λ	Precision of normal hyper-distribution for log fish density
Derived Variables	
$\alpha_{i,g}$	Parameter for beta hyper distribution of detection probability for strata g
$\beta_{i,g}$	Parameter for beta hyper distribution of detection probability for strata g
$N_{j,g}$	Abundance at index site j for strata g
Ns	Total abundance across all index sites
Nus_s	Total abundance in unsampled stream length
Nt	Total abundance in the Coquitlam River study area
Indices and Constants	
i	Index for snorkeling mark-recapture site
j	Index for snorkeling index site
g	Index for fish size strata
l_j	Shoreline length for index site j
L	Total shoreline length for the Coquitlam River study area

Appendix 4.2 Equations of the hierarchical Bayesian model used to estimate juvenile Steelhead abundance in the Coquitlam River. See Appendix 4.1 for definitions of model parameters, constants, and subscripts. Lower case Arabic letters denote data or indices (if subscripts). Capital Arabic letters denoted derived variables, which are computed as a function of estimated parameters. Greek letters denote estimated parameters. Parameters with Greek letter subscripts are hyper-parameters.

Detection Model

(4.1) $$r_{i,g} \sim dbin(\theta_{i,g}, m_{i,g})$$

(4.2) $$d_{i,p,g} \sim dbeta(\theta_{i,g}, n_{i,p-1,g})$$

Population Model

(4.3) $$\theta_{j,g} \sim dbeta(\alpha_g, \beta_g)$$

(4.4) $$c_{j,g} \sim dbin(\theta_{j,g}, N_{j,g})$$

(4.4) $$N_{j,g} \sim dpois(\lambda_j l_j)$$

(4.6) $$\log(\lambda_j) \sim dnorm(\mu_\lambda, \tau_\lambda)$$

(4.7) $$Ns = \sum_g \sum_{j \in r} n_{j,g}$$

(4.8) $$Nus = \exp[\mu_\lambda + 0.5\tau_\lambda^{-1}](h_r - \sum_{j \in s} l_j)$$

(4.9) $$Nt = Ns + Nus$$

Appendix 4.2 (continued)

Priors and Transformation

$$\mu_{\theta,g} \sim dunif(0,1)$$
$$\sigma_{\theta,g} \sim dhcauchy(0,0.5) \qquad (4.10)$$

$$\tau_{\theta,g} = \sigma_{\theta,g}^{-2}$$
$$\alpha_g = \mu_{\theta,g}\tau_{\theta,g}$$
$$\beta_g = (1-\mu_{\theta,g})\tau_{\theta,g} \qquad (4.11)$$

$$\mu_\lambda \sim dnorm(0,0.1.0E-6)$$
$$\sigma_\lambda \sim dhcauchy(0,0.5) \qquad (4.12)$$

$$\tau_\lambda = \sigma_\lambda^{-2} \qquad (4.13)$$

Appendix 4.3 Summary of data deficiencies and alternate approaches taken with respect to estimation of parameters and hyper-distributions in the Hierarchical Bayesian Model (HBM) used to estimate juvenile Steelhead and Coho standing stocks in the Coquitlam River during 2006-2019.

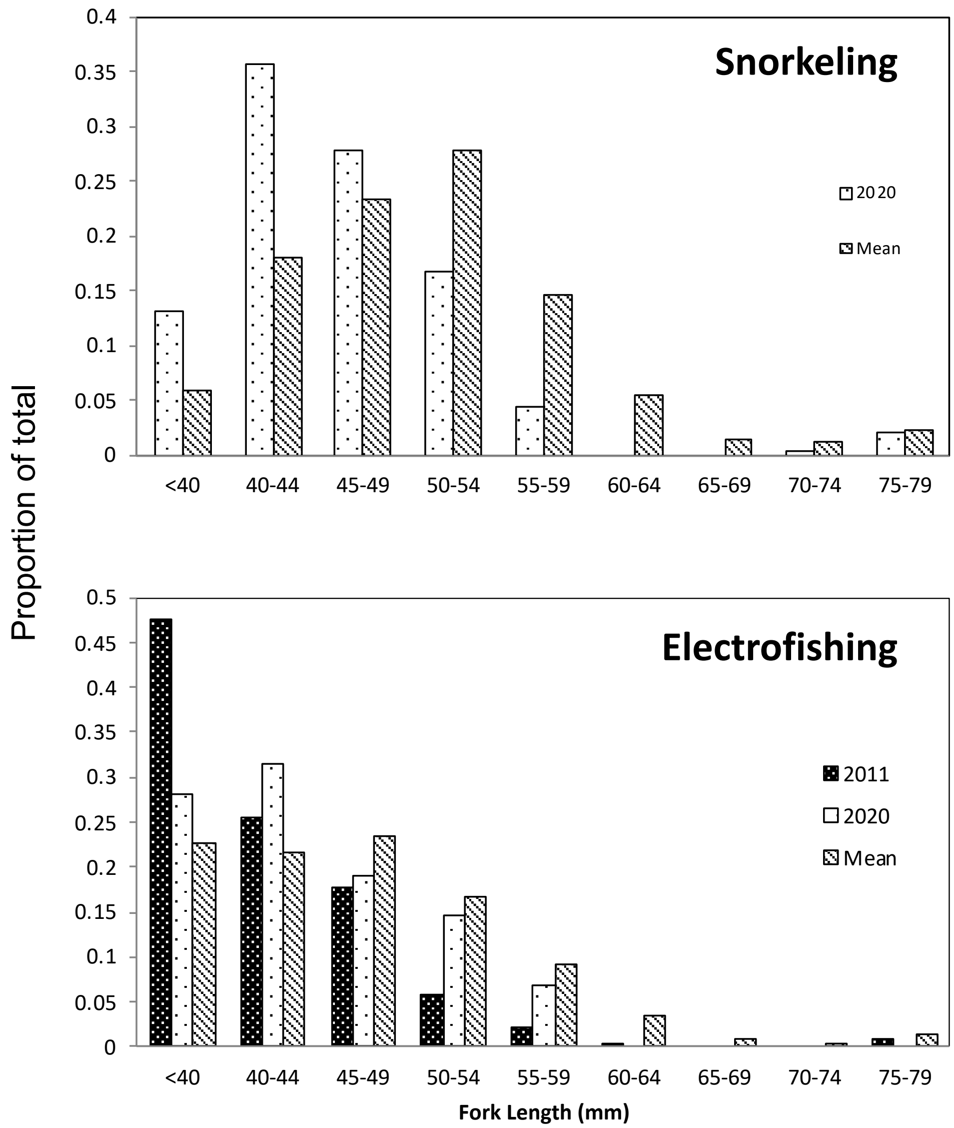
1. *Large-sized Steelhead parr (>140 mm) and small-sized Steelhead fry (<50 mm).* For these size categories of Steelhead, there were relatively few fish marked and resighted at the mark-recapture sites, and numbers observed by snorkelers in the index sites were low and quite variable. This led to an insufficient amount of data for the HBM to reliably estimate standard deviation in snorkeling detection probability and fish density among index sites. These deficiencies were addressed by substituting parameter estimates for medium-sized Steelhead parr (100-140 mm) in the case of large-sized Steelhead, and parameter estimates for large-sized Steelhead fry (50-69 mm) in the case of small-sized Steelhead fry.

2. *Coho fry in 2006 and large -sized Steelhead fry (50-69 mm) in 2007.* In these cases, numbers observed by snorkelers in the index sites were low and quite variable, leading to an insufficient amount of data for the HBM to reliably estimate standard deviation fish density among index sites. These deficiencies were addressed by substituting the mean of standard deviation estimates for other years for these species/size classes.

Appendix 4.4 Summary of habitat data for night snorkeling and day electrofishing sites in Coquitlam River in 2020.

Sampling method	Site no.	Upstream distance (km)	Site area (m^2)	Mean length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m)	Dmax (m)	Boulder (%)	Cobble (%)	Gravel (%)	Fines (%)
snorkeling	0.55	8.25	534	25	21	0.41	0.29	1.95	50	30	10	10
snorkeling	0.90	8.60	366	25	15	0.59	0.36	2.10	25	40	25	10
snorkeling	1.25	8.95	443	25	18	0.35	0.46	1.90	60	20	15	5
snorkeling	1.60	9.30	808	25	32	0.34	0.40	1.80	35	40	20	5
snorkeling	1.95	9.65	421	25	17	0.37	0.50	3.30	65	25	5	5
snorkeling	2.65	10.35	391	25	16	0.62	0.21	2.20	50	30	10	10
snorkeling	3.00	10.70	697	25	28	0.39	0.37	3.10	65	30	5	0
snorkeling	3.35	11.05	474	25	19	0.37	0.42	1.40	10	43	38	10
snorkeling	3.70	11.40	587	25	23	0.53	0.34	2.25	50	30	10	10
snorkeling	4.05	11.75	827	25	33	0.30	0.40	1.30	20	50	20	10
snorkeling	4.40	12.10	711	25	28	0.50	0.30	3.70	60	25	10	5
snorkeling	4.75	12.45	501	25	20	0.43	0.45	1.70	30	40	20	10
snorkeling	5.00	12.70	492	25	20	0.45	0.27	1.65	50	30	15	5
snorkeling	5.20	12.90	578	25	23	0.33	0.40		40	35	15	10
snorkeling	5.45	13.15	439	25	18	0.52	0.35	1.80	55	28	10	8
snorkeling	5.60	13.30	432	25	17	0.44	0.36	1.80	35	30	30	5
snorkeling	5.70	13.40	600	25	24	0.33	0.26	2.60	35	40	20	5
snorkeling	5.80	13.50	416	25	17	0.45	0.36	1.50	55	30	10	5
snorkeling	6.15	13.85	380	25	15	0.43	0.45	2.30	40	35	20	5
snorkeling	6.85	14.55	436	25	17	0.51	0.34	1.60	33	38	20	10
snorkeling	7.20	14.90	265	25	11	0.47	0.54	2.35	60	30	10	0
snorkeling	7.55	15.25	427	25	17	0.55	0.35	2.65	53	33	10	5
snorkeling	7.90	15.60	334	25	13	0.45	0.42	2.10	25	50	20	5
snorkeling	8.25	15.95	396	25	16	0.30	0.44	1.50	38	35	18	10
electrofishing	1.95	9.7	132	18.2	7	0.44	0.35		60	25	10	5.0
electrofishing	2.50	10.2	137	20	7	0.39	0.38		35	40	20	5.0
electrofishing	3.20	10.9	122	19	6	0.24	0.19		40	40	15	5.0
electrofishing	6.00	13.7	118	18	7	0.39	0.39		35	35	20	10.0

Appendix 4.5 Length-frequency histogram (proportion of total catch less <80mm forklength) for Steelhead fry captured by electrofishing and counted during snorkeling in the Coquitlam River averaged for 2008-2010,2012-2019 and 2020 (data pooled for all sites). 2011 is also shown for electroshocking as an example of a year with considerable shift towards small sized fry.



Appendix 4.6 Estimates of juvenile fish density, standing stock, and 95% confidence intervals by species and age class in Coquitlam River during 2006-2020. Estimates were derived from night snorkeling counts with the exception of 2011 Steelhead (0+), which used the electrofishing estimate.

Species/age class	Year	Density (fish/100m^2)	Density (fish/km)	Standing stock	Lower 95% CI	Upper 95% CI	± 95% CI	sd
coho (0+)	2006	14.6	2,632	27,111	13,249	33,791	38%	5240
coho (0+)	2007	13.1	1,787	18,405	12,340	28,481	44%	4118
coho (0+)	2008	22.8	4,536	46,719	32,460	71,416	42%	9938
coho (0+)	2009	20.7	5,126	52,794	38,958	83,936	43%	11474
coho (0+)	2010	24.1	6,037	62,178	42,549	95,461	43%	13498
coho (0+)	2011	42.8	8,871	91,367	52,468	118,903	36%	16948
coho (0+)	2012	42.9	7,170	73,846	54,509	115,705	41%	15611
coho (0+)	2013	39.2	6,823	70,279	52,339	114,403	44%	15833
coho (0+)	2014	22.8	4,321	44,507	35,979	62,261	31%	7074
coho (0+)	2015	19.7	3,505	36,101	28,320	44,591	31%	5738
coho (0+)	2016	14.8	2,468	25,424	20,270	35,420	30%	3865
coho (0+)	2017	37.8	5,744	59,166	45,730	79,661	29%	8656
coho (0+)	2018	24.7	4,247	43,740	34,079	60,020	30%	6618
coho (0+)	2019	38.6	5,824	59,992	47,747	83,531	30%	9129
coho (0+)	2020	28.0	4,212	43,386	33,560	59,400	30%	6592
								0
steelhead (0+)	2006	28.9	13,411	138,132	108,971	257,522	54%	37896
steelhead (0+)	2007	9.4	3,131	32,251	22,193	139,860	182%	30017
steelhead (0+)	2008	9.4	4,127	42,506	32,185	660,106	739%	160184
steelhead (0+)	2009	8.1	3,597	37,047	29,002	1,355,054	1790%	338279
steelhead (0+)	2010	10.6	3,850	39,657	29,627	151,626	154%	31122
steelhead (0+)[1]	2011	9.6	2,131	21,949	-	-	-	
steelhead (0+)	2012	14.5	5,362	55,232	40,520	81,398	37%	10428
steelhead (0+)	2013	19.9	6,409	66,017	51,319	107,519	43%	14337
steelhead (0+)	2014	7.9	3,179	32,746	26,499	44,724	28%	4649
steelhead (0+)	2015	10.4	3,134	32,277	26,270	44,291	28%	4597
steelhead (0+)	2016	8.9	2,932	30,203	22,396	43,135	34%	5291
steelhead (0+)	2017	14.8	5,277	54,358	41,408	89,120	44%	12171
steelhead (0+)	2018	12.0	4,618	47,565	37,341	62,031	26%	6332
steelhead (0+)	2019	14.7	5,277	54,358	41,030	88,839	44%	12196
steelhead (0+)	2020	12.8	4,603	47,408	35,965	68,281	34%	8244
								0
steelhead (1+)	2006	2.9	580	5,976	3,532	22,859	162%	4930
steelhead (1+)	2007	6.6	994	10,237	7,036	17,771	52%	2739
steelhead (1+)	2008	4.6	992	10,222	7,446	20,770	65%	3399
steelhead (1+)	2009	3.8	1,056	10,876	8,229	16,041	36%	1993
steelhead (1+)	2010	3.7	787	8,106	6,556	10,710	26%	1060
steelhead (1+)	2011	4.2	853	8,791	6,425	14,701	47%	2111
steelhead (1+)	2012	6.0	1,036	10,668	8,002	17,462	44%	2413
steelhead (1+)	2013	6.9	1,306	13,456	10,129	21,470	42%	2893
steelhead (1+)	2014	3.2	618	6,369	5,115	8,669	28%	907
steelhead (1+)	2015	2.9	572	5,889	4,869	7,546	23%	683
steelhead (1+)	2016	2.9	506	5,216	4,321	6,416	20%	534
steelhead (1+)	2017	4.8	880	9,064	7,287	12,360	28%	1294
steelhead (1+)	2018	5.0	955	9,836	7,623	12,800	27%	1364
steelhead (1+)	2019	4.9	880	9,064	7,287	12,360	28%	1294
steelhead (1+)	2020	3.9	666	6,863	5,710	8,607	21%	739

Appendix 4.6 cont'd

Species/age class	Year	Density (fish/100m^2)	Density (fish/km)	Standing stock	Lower 95% CI	Upper 95% CI	± 95% CI
steelhead (2+)	2006	1.1	179	1,841	933	3,569	72%
steelhead (2+)	2007	1.3	192	1,978	1,145	3,950	71%
steelhead (2+)	2008	1.0	122	1,255	694	2,598	76%
steelhead (2+)	2009	1.9	310	3,196	1,963	6,402	69%
steelhead (2+)	2010	1.7	261	2,690	1,630	5,331	69%
steelhead (2+)	2011	1.9	375	3,862	2,443	7,266	62%
steelhead (2+)	2012	1.8	307	3,160	1,961	5,666	59%
steelhead (2+)	2013	1.6	255	2,625	1,582	4,713	60%
steelhead (2+)	2014	2.1	372	3,831	2,756	6,634	51%
steelhead (2+)	2015	1.3	249	2,561	1,822	4,181	46%
steelhead (2+)	2016	#N/A	257	2,642	1,835	4,807	56%
steelhead (2+)	2017	#N/A	311	3,207	2,177	5,371	50%
steelhead (2+)	2018	1.7	207	2,127	1,293	3,330	52%
steelhead (2+)	2019	2.1	311	3,207	2,177	5,371	50%
steelhead (2+)	2020	1.1	200	2,057	1,414	3,546	52%
steelhead (parr)	2006	3.7	759	7,817	-	-	-
steelhead (parr)	2007	7.6	1,186	12,215	-	-	-
steelhead (parr)	2008	5.4	1,114	11,477	-	-	-
steelhead (parr)	2009	5.3	1,366	14,072	-	-	-
steelhead (parr)	2010	4.1	1,048	10,796	-	-	-
steelhead (parr)	2011	#N/A	1,228	12,653	-	-	-
steelhead (parr)	2012	#N/A	1,343	13,828	-	-	-
steelhead (parr)	2013	#N/A	1,561	16,081	-	-	-
steelhead (parr)	2014	4.4	990	10,200	-	-	-
steelhead (parr)	2015	3.9	820	8,450	-	-	-
steelhead (parr)	2016	3.7	763	7,858	-	-	-
steelhead (parr)	2017	6.3	1,191	12,271	-	-	-
steelhead (parr)	2018	6.3	1,161	11,963	-	-	-
steelhead (parr)	2019	8.0	1,191	12,271	-	-	-
steelhead (parr)	2020	5.5	866	8,920	-	-	-

Appendix 4.7 Mean 3-pass depletion electrofishing density estimates at four one-shoreline sites in the Coquitlam River for 2006-2020. The electrofishing survey was conducted at the same four sites during 2007-2020, whereas in 2006 electrofishing was conducted at 10 shoreline sites located within the annual snorkeling index sites (Decker et al. 2007).

			Density	
Year	Species	Age	fish/100m^2	fish/km
2006	Coho	0	10	591
2007	Coho	0	3	211
2008	Coho	0	1	90
2009	Coho	0	8	606
2010	Coho	0	3	200
2011	Coho	0	13	1072
2012	Coho	0	7	1073
2013	Coho	0	20	2759
2014	Coho	0	28	4011
2015	Coho	0	16	2263
2016	Coho	0	18	2862
2017	Coho	0	24	3685
2018	Coho	0	16	2388
2019	Coho	0	15	1989
2020	Coho	0	20	1308
2006	Steelhead	0	50	3055
2007	Steelhead	0	27	2154
2008	Steelhead	0	31	2224
2009	Steelhead	0	20	1530
2010	Steelhead	0	25	1648
2011	Steelhead	0	51	4179
2012	Steelhead	0	23	1704
2013	Steelhead	0	36	2418
2014	Steelhead	0	34	2364
2015	Steelhead	0	22	1507
2016	Steelhead	0	20	1543
2017	Steelhead	0	20	1524
2018	Steelhead	0	20	3205
2019	Steelhead	0	27	1814
2020	Steelhead	0	25	1695
2006	Steelhead	1	3	206
2007	Steelhead	1	11	891
2008	Steelhead	1	7	493
2009	Steelhead	1	7	505
2010	Steelhead	1	3	200
2011	Steelhead	1	5	425
2012	Steelhead	1	3	211
2013	Steelhead	1	5	344
2014	Steelhead	1	5	347
2015	Steelhead	1	7	460
2016	Steelhead	1	3	209
2017	Steelhead	1	8	568
2018	Steelhead	1	8	738
2019	Steelhead	1	7	452
2020	Steelhead	1	8	515

Appendix 4.8 Summary of 3-pass depletion electrofishing results at four one-shoreline sites in the Coquitlam River in 2020.

Year	Site	Pass 1	Pass 2	Pass 3	Population estimate	Lower 95% CI	Upper 95% CI	Mean density fish/100m^2	Mean density fish/km
Coho fry									
2020	1.95	6	2	3	12	6	18	9	1,319
2020	2.50	7	6	4	24	3	45	17	2,462
2020	3.20	10	10	5	35	11	59	29	3,684
2020	6.00	14	8	3	27	21	33	23	3,000
Steelhead fry									
2020	1.95	25	6	5	37	33	41	28	2,033
2020	2.50	7	4	1	12	10	14	9	615
2020	3.20	12	8	1	31	0	125	25	1,632
2020	6.00	4	5	4	45	45	45	38	2,500
Steelhead parr (1+)									
2020	1.95	10	5	0	15	14	16	11.3	824
2020	2.50	1	1	0	2	0	7	1.5	103
2020	3.20	4	5	1	11	6	16	9.0	579
2020	6.00	6	3	1	10	8	12	8.5	556

10.4 Appendices for Chapter 5

Appendix 5.1 Trapping start and end dates, season length, number of trap operating days (Trapping days) and percent of trapping season with trap operating (% trapping) for rotary screw traps at the trapping sites at the downstream end of Reach 2-4 as well as fish weirs at the outlets of the four constructed off-channel habitats for 2020.

Downstream RST trapping site	Start date	End date	Season length	Trapping days	% trapping
Reach 2 (RST2.2, chum, pink)	2-18	5-16	88	80	91%
Reach 2 (RST2.4, coho, steelhead)	3-15	6-20	97	89	92%
Reach 2 (RST2.5, coho, steelhead)	3-15	6-20	97	89	92%
Reach 3 (RST3, coho, steelhead)	3-15	6-20	97	89	92%
Reach 4 (RST4, coho, steelhead)	3-15	6-20	97	90	93%
Archery Pond	3-15	6-20	97	91	94%
Overland Ponds	3-15	6-20	97	91	94%
Or Creek Ponds	3-15	6-20	97	83	86%
Grants Tomb Pond	3-15	6-20	97	85	88%

Appendix 5.2 Summary of estimated numbers of Coho, Steelhead and Chum smolts passing the three RST trapping locations (not reach estimates) in the Coquitlam River mainstem in 2020. Mark group indicates the location where fish were initially captured and marked. Also shown are numbers of marked (M), recaptured (R) smolts, unmarked captures (U) and estimated capture efficiencies (R/M).

Species	Site	Mark group(s)	M	R	U	Capture efficiency	N mainstem smolts	CI (+/-)	CI (%)
Coho	RST 2	all	2,310	680	2,008	29%	8,490	860	10%
	RST 3	all	1,463	72	228	5%	5,391	1,625	30%
	RST 4	mainstem	503	248	494	49%	1,073	208	19%
Steelhead	RST 2	all	669	92	424	14%	3,789	813	21%
	RST 3	all	214	2	12	-	-	-	-
	RST 4	all	200	36	187	18%	1,061	378	36%
Chum	RST 2	RST 2	7,836	17,201	1,102	220%	841,880	45,999	5%
Pink	RST 2	RST 2	20,430	2,157	83,918	11%	197,838	70,570	35.7%

Appendix 5.3 Mark-recapture data used for the BTSPAS estimates for Coho, Steelhead, Chum and Pink at three rotary screw trap sites (RST2, RST3, RST4) in the Coquitlam River mainstem in 2020. Tables include numbers of fish marked in strata *i* , numbers recaptured within the release strata (i+0) and in subsequent strata (i+n), and proportion of each strata in which sampling occurred.

Coho

Recovery site: RST 2
All Mark Group

Release Strata (i)	Marks	Recoveries in strata post release i+0	i+1	i+2	i+3	i+4	i+5	i+6	Unmarked
1	12	1	1	0	1	0	0	0	34
2	17	0	1	0	0	0	0	0	20
3	36	1	2	0	0	2	0	0	62
4	36	1	2	0	0	0	2	1	35
5	45	2	2	3	2	2	1	0	20
6	81	10	6	1	2	1	0	0	71
7	202	45	9	7	2	3	0	0	181
8	340	81	31	3	2	1	1	0	284
9	526	75	63	3	1	0	0	0	506
10	390	101	23	1	0	0	0	0	474
11	271	55	24	0	0	0	0	0	321
12	266	59	21	0	0	0	0	0	153
13	88	18	1	0	0	0	0	0	71
14	0	0	0	0	0	0	0	0	9

Recovery site: RST 3
All Mark Group

Release Strata (i)	Marks	Recoveries in strata post release i+0	i+1	i+2	i+3	i+4	i+5	i+6	Unmarked
1	9	0	0	0	0	0	0	0	3
2	10	1	0	0	0	0	0	0	4
3	24	1	0	0	0	0	0	0	7
4	15	0	0	0	0	0	0	0	0
5	10	0	0	0	0	0	0	0	1
6	18	1	0	0	0	0	0	0	6
7	37	0	0	0	0	1	0	0	15
8	129	5	3	0	0	0	0	0	33
9	393	8	7	2	0	1	0	0	55
10	319	9	2	0	0	0	0	0	58
11	215	9	1	0	0	0	0	0	46
12	204	13	2	0	0	0	0	0	20
13	80	6	0	0	0	0	0	0	5
14	0	0	0	0	0	0	0	0	1

Recovery site: RST 4
Mainstem Mark Group

Release Strata (i)	Marks	Recoveries in strata post release i+0	i+1	i+2	i+3	i+4	i+5	i+6	Unmarked
1	4	1	0	0	0	0	0	0	4
2	1	0	0	0	0	0	0	0	1
3	10	1	0	0	0	0	0	0	3
4	9	0	1	0	0	0	0	0	7
5	5	3	0	0	0	0	0	0	4
6	6	4	0	0	1	0	0	0	5
7	16	6	4	0	0	0	0	0	16
8	94	30	16	0	0	1	0	0	75
9	293	101	29	2	0	0	0	0	195
10	202	73	15	1	0	0	0	0	134
11	116	36	11	0	0	0	0	0	50
12	39	21	1	0	0	0	0	0	20
13	2	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0	1

Appendix 5.3 continued

Steelhead

Recovery site: RST 2
All Mark Group

Release Strata (i)	Marks	Recoveries in strata post release i+0	i+1	i+2	i+3	i+4	i+5	i+6	Unmarked
1	4	0	0	0	0	0	0	0	1
2	3	0	0	0	0	0	0	0	2
3	8	0	1	0	0	1	0	0	5
4	9	0	0	0	0	1	0	0	3
5	12	3	1	0	0	0	0	0	7
6	42	0	3	2	0	0	0	0	23
7	104	6	8	0	0	0	0	0	63
8	172	7	12	0	0	0	0	0	124
9	176	13	11	0	0	0	0	0	114
10	90	9	2	0	0	0	0	0	82
11	31	5	1	0	0	0	0	0	29
12	18	5	0	0	0	0	0	0	21
13	4	0	0	0	0	0	0	0	4
14	0	0	0	0	0	0	0	0	0

Recovery site: RST 3
All Mark Group

Release Strata (i)	Marks	Recoveries in strata post release i+0	i+1	i+2	i+3	i+4	i+5	i+6	Unmarked
1	3	0	0	0	0	0	0	0	0
2	2	0	0	0	0	0	0	0	0
3	4	0	0	0	0	0	0	0	0
4	6	0	0	0	0	0	0	0	0
5	6	0	0	0	0	0	0	0	0
6	18	0	0	0	0	0	0	0	0
7	40	0	1	0	0	0	0	0	1
8	53	0	0	0	0	0	0	0	2
9	62	1	0	0	0	0	0	0	7
10	17	0	0	0	0	0	0	0	2
11	2	0	0	0	0	0	0	0	2
12	1	0	0	0	0	0	0	0	0
13	5	0	0	0	0	0	0	0	1
14	1	0	0	0	0	0	0	0	0

Recovery site: RST 4
All Mark Group

Release Strata (i)	Marks	Recoveries in strata post release i+0	i+1	i+2	i+3	i+4	i+5	i+6	Unmarked
1	3	0	0	0	0	0	0	0	2
2	2	0	0	0	0	0	0	0	2
3	4	1	0	0	0	0	0	0	3
4	6	0	1	0	0	1	0	0	2
5	5	0	0	0	0	0	0	0	5
6	18	1	3	0	1	0	0	0	18
7	35	3	3	0	0	0	0	0	38
8	51	2	3	0	0	0	0	0	51
9	59	10	4	0	0	0	0	0	48
10	16	2	1	0	0	0	0	0	18
11	1	0	0	0	0	0	0	0	2
12	0	0	0	0	0	0	0	0	0
13	1	1	0	0	0	0	0	0	1
14	0	0	0	0	0	0	0	0	0

Appendix 5.3 continued

Chum

Recovery site: RST 2.2

All mark groups

Release strata	Marks	Recaptures	Unmarked	Capture efficiency
1	0	0	14	-
2	0	0	178	-
3	0	0	312	-
4	0	0	732	-
5	0	0	1516	-
6	1386	197	2712	14.2%
7	725	67	4327	9.2%
8	4594	565	13666	12.3%
9	6000	706	37141	11.8%
10	6000	522	22394	8.7%
11	1725	100	3678	5.8%
12	0	0	1062	-
13	0	0	92	-
Untagged Fish		1,977	0	

Pink

Recovery site: RST 2.2

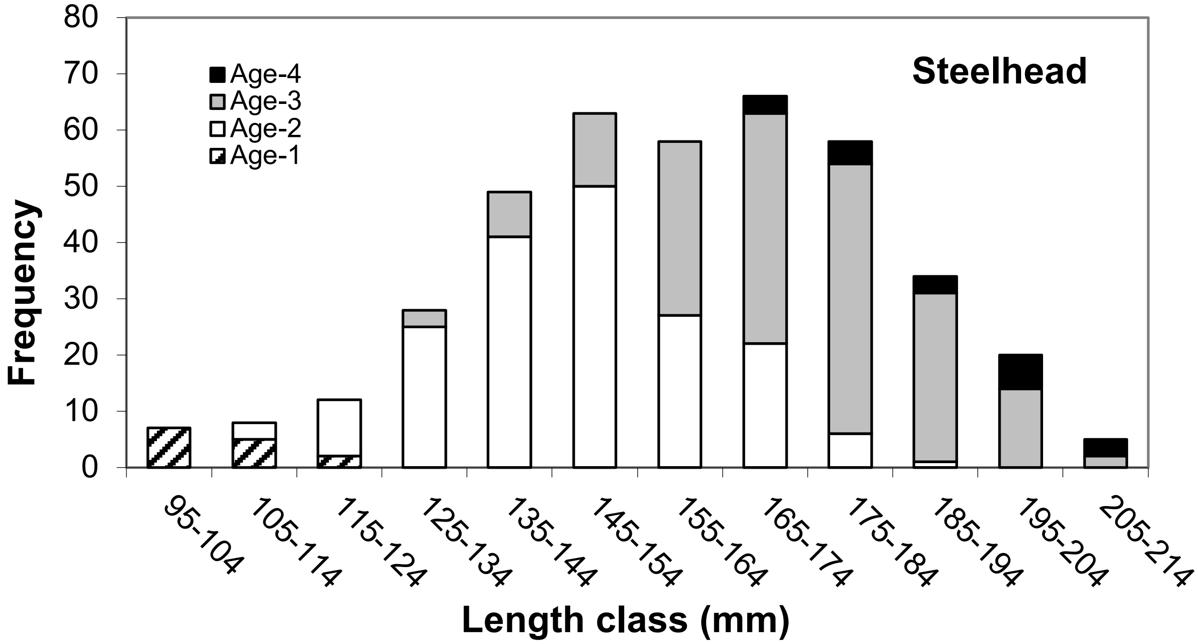
All mark groups

Release strata	Marks	Recaptures	Unmarked	Capture efficiency
1	0	26	0	-
2	0	185	0	-
3	0	429	0	-
4	0	1121	0	-
5	1064	3601	86	338.4%
6	2989	5632	615	188.4%
7	676	2652	77	392.3%
8	1546	3077	163	199.0%
9	1561	2239	161	143.4%
10	0	262	0	-
11	0	9	0	
12	0	1	0	-
13	0	0	0	-

Appendix 5.4 Estimated of the number of Coho and Steelhead smolts outmigrating from reaches 2-4 of the Coquitlam River 1996-2020. Individual estimates for four constructed off-channel habitats and mainstem reaches 2-4, both individually and combined, which extends 7.5 km downstream from the Coquitlam Dam.

	Year																							
Site	1996	1997	1998	1999	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Coho																								
Off-channel sites																								
Grant's Tomb	1,220	697	1,390	1,822	1,074	1,524	1,373	886	0	0	0	0	2193	902	519	804	264	171	344	416	267	124	153	300
Or Creek Ponds	2,814	11,281	-	1,138	2,982	2,283	2,266	2,315	2,945	2,420	2,357	2,614	3,121	1,926	2,454	3,608	1,862	868	1,735	1,608	3,437	1,010	790	500
Archery Pond	3,016	621	-	3,422	1,292	662	1,196	620	799	509	479	-	1,761	1,761	313	392	-	279	456	596	274	251	189	234
Overland Channel	-	-	-	-	-	-	819	5,108	6,860	4,983	2,681	3,156	3,538	2,529	700	1,846	1,796	1,747	1,085	1,788	370	175	370	646
Total	7,050	12,599	1,390	6,382	5,348	4,469	5,654	8,929	10,604	7,912	5,517	5,770	10,613	7,118	3,986	6,650	3,922	3,065	3,620	4,408	4,348	1,560	1,502	1,680
Mainstem																								
Reach 2	-	-	-	-	4,368	1,196	2,570	497	619	1,502	815	5,380	3,591	2,748	2,629	3,124	2,037	3,510	2,821	1,788	4,148	3,767	5,663	3,099
Reach 3	-	-	-	2,405	5,285	2,231	2,471	2,201	1,141	3,458	1,086	2,452	5,071	2,257	3,037	4,167	2,523	3,538	4,265	3,058	3,218	2,183	911	5,391
Reach 4	290 [1]	2,773	2,979	1,331	3,826	2,339	2,331	1,536	1,455	1,109	969	1,188	5,182	1,568	1,420	3,644	2,255	2,247	1,148	848	2,012	1,936	1,510	1,073
Total	-	-	-	-	11,036	4,838	8,195	4,234	3,215	5,979	2,870	9,020	13,844	6,573	7,086	10,935	6,366	8,278	8,234	5,654	9,810	7,886	8,084	8,490
Coquitlam River (incld. off-channel)	-		-	-	16,384	9,307	13,849	13,163	13,819	13,891	8,387	14,790	24,457	13,691	11,072	17,585	10,288	11,343	11,854	10,062	14,158	9,446	9,586	10,170
Steelhead																								
Off-channel sites																								
Grant's Tomb	57	11	-	42	58	12	8	5	0	0	0	0	60	49	13	9	46	27	47	76	19	11	4	30
Or Creek Ponds	55	411	-	91	161	26	19	30	66	44	47	74	53	52	30	13	17	14	47	59	22	2	2	6
Archery Pond	-	-	-	-	148	54	46	25	19	21	2	-	-	-	-	-	-	29	18	47	24	5	3	10
Overland Channel	-	-	-	-	-	-	-	-	96	15	4	73	12	37	9	24	11	5	0	0	18	0	0	1
Total	112	422	-	133	367	92	73	60	181	80	53	147	125	138	52	46	74	75	112	182	83	18	9	47
Mainstem																								
Reach 2	-	-	-	-	2,756	1,317	1,598	1,974	1,984	2,262	1,085	2,567	2,529	1,146	903	2,071	739	2,080	1,428	2,547	1,433	5,572	-	3,796
Reach 3	-	-	-	1,781	1,790	391	1,318	636	1,022	1,230	435	1,578	417	879	921	243	449	402	1,730	1,204	1,182	-	-	-
Reach 4	258	207	421	526	711	547	857	1,303	779	705	929	1,352	2,327	2,711	1,228	2,636	2,618	2,314	1,808	1,335	2,807	-	1,097	1,061
Total	-	-	-	-	3,824	2,216	3,812	3,782	3,785	4,197	2,615	5,497	5,273	4,736	3,052	4,712	3,622	4,579	4,966	5,086	5,142	5,572	4,397	3,789
Coquitlam River (incld. off-channel)	-	-	-	-	4,191	2,308	3,885	3,842	3,966	4,277	2,668	5,644	5,398	4,874	3,104	4,758	3,696	4,654	5,078	5,268	5,225	5,590	4,406	3,836

Appendix 5.5 Age-fork length relationships for Steelhead parr and smolts in the Coquitlam River during 2005-2020 derived from scale-aging analysis.



Appendix 5.6 Mark-recapture data used to test whether the capture efficiency differed between smolts marked using VIE or fin clips for Coho and Steelhead captures at RST 2 during strata 8 and 9 in 2018. Proportions recaptured are considered unaffected by marking method with p values greater than 0.05.

Species	Marking method	Marked	Recaptured	Capture efficiency	Chi square test p value
Coho	Adipose clip	278	93	0.33	0.63
	VIE	276	87	0.32	
Steelhead	Adipose clip	159	33	0.21	0.55
	VIE	160	29	0.18	

10.5 Appendices for Chapter 6

Appendix 6.1a Summary of all population estimates for all life stages and species in Coquitlam River, 2000-2019. Values shown for the different life stages for a given year do not correspond in most cases (i.e., columns do not line up), as values are shown for the year in which they occurred rather than the brood year. Abundances for the different life stage are also not strictly comparable because study areas differ somewhat for the different components of the monitoring program.

Life stage	Species	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Adult	chum	-	15,378	18,301	27,992	24,559	51,860	11,066	18,224	19,600	6,931	27,410	57,300	42,220	8,491	23,410	78,120		26,490	10,970
escapement	pink	-	-	5,418	-	4,406	-	2,876	-	10,698	-	10,427	-	34,280		9,327				
	coho	-	2,648	1,562	2,562	1,334	939	2,401	878	3,175	12,338	8,428	11,320	13,290	4,957	4,979	6,867		13,910	11,670
	Chinook	-	-	<300	<100	<100	<100	438	952	1,529	8,018	4,918	363	2,413	572	123	511		456	591
	steelhead (female)	-	-	-	-	187	434	130	148	113	167	206	278	248	158	251	205	199	134	126
	steelhead (total)	-	-	-	-	373	868	260	297	225	333	412	557	495	317	502	410	398	268	252
Fall standing	coho	-	-	-	-	-	27,111	18,405	46,719	52,794	62,178	91,367	73,846	70,279	44,507	36,101	25,424	59,166	43,740	59,992
stock	0+ steelhead fry	-	-	-	-	-	138,132	32,251	42,506	37,047	39,657	21,949	55,232	66,017	32,746	32,277	30,203	54,358	47,565	54,358
	1+ steelhead parr	-	-	-	-	-	5,976	10,237	10,222	10,876	8,106	8,791	10,668	13,456	6,369	5,889	5,216	9,064	9,836	9,064
	2+ steelhead parr	-	-	-	-	-	1,841	1,978	1,255	3,196	2,690	3,862	3,160	2,625	3,831	2,561	2,642	3,207	2,127	3,207
Smolt yield	chum (total - million:	-	-	1.3	1.1	0.8	3.4	3.1	1.0	4.2	3.4	1.9	2.3	6.7	8.6	2.0	4.0	12.7	4.0	2.4
	pink (total - millions	-	-		0.32	-	0.15	-	0.18	-	0.55	-	3.56	-	6.03	-	1.31	-	0.11	-
	coho (total)	16,384	9,307	13,849	13,163	13,819	13,891	8,387	14,790	24,457	13,691	11,072	17,585	10,288	11,343	11,854	10,062	14,158	8,973	10,473
	coho (mainstem)	11,036	4,838	8,195	4,234	3,215	5,979	2,870	9,020	13,844	6,573	7,086	10,935	6,351	8,080	8,234	5,654	9,810	7,413	8,967
	steelhead (total)	4,191	2,308	3,885	3,842	3,966	4,277	2,668	5,644	5,398	4,874	3,104	4,758	3,622	4,654	5,078	5,268	5,225	5,590	4,419
	steelhead (mainster	3,824	2,216	3,812	3,782	3,785	4,197	2,615	5,497	5,273	4,736	3,052	4,712	3,622	4,579	4,966	5,086	5,142	5,572	4,410
	steelhead (2+)	-	-	-	-	-	-	1,412	2,795	2,968	2,588	1,848	2,177	1,927	3,134	3,134	4,056	3,882	3,180	2,444
	steelhead (3+)	-	-	-	-	-	-	-	2,849	2,430	2,286	1,256	2,581	1,695	1,520	1,944	1,212	1,343	2,410	1,975
	steelhead smolts by brood year					4,261	5,225	5,254	3,843	4,429	3,871	3,447	5,078	4,347	5,399	6,292	5,155			

Appendix 6.1b Summary of survival estimates across all life stages and species for 2000-2019 brood escapements for Coho, Chum and Pink Salmon and 2005-2018 brood escapement for Steelhead in the Coquitlam River. Egg-to-smolt survival estimates are based on adult escapement upstream of the lowermost smolt trapping site (RST2). Unlike Table 6.1a, year corresponds to the adult return year (brood year), as opposed to the year when the juvenile life stage was present. For survival rates among the juvenile life stages of Steelhead (e.g., fry to age 1+ parr), year corresponds to the younger life stage. Biased-high survival rate estimates (i.e., >100%) are shown in red (see Section 6.2).

Species	Survival by life stage	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Coho	Egg-to-fall fry [1]	-	-	-	-	1.35%	1.31%	1.30%	4.01%	1.31%	0.49%	0.58%	0.41%	0.22%	0.49%	0.34%	0.57%	-	0.29%	
Coho	Fall fry-to-smolt	-	-	-	-	-	2.08%	27.97%	32.57%	17.74%	17.87%	49.82%	11.50%	12.24%	25.15%	17.52%	32.48%	13.64%	18.85%	
Coho	Egg-to-smolt		0.17%	0.28%	0.18%	0.35%	0.30%	0.21%	0.93%	0.14%	0.03%	0.07%	0.03%	0.03%	0.08%	0.07%	0.07%	-	0.03%	
Steelhead	Egg-to-fall fry [1]	-	-	-	-	-	8.6%	6.7%	7.8%	8.9%	6.4%	2.9%	5.4%	7.2%	5.6%	3.5%	4.0%	7.4%	9.6%	11.7%
Steelhead	Egg-to-parr [1]	-	-	-	-	0.9%	0.6%	2.1%	2.0%	1.9%	1.4%	1.4%	1.3%	0.7%	1.0%	0.6%	1.2%	1.3%	1.8%	
Steelhead	Egg-to-smolt [1,2]	-	-	-	-	0.7%	0.4%	1.1%	0.8%	1.3%	0.7%	0.5%	0.5%	0.5%	1.0%	0.7%	0.8%			
Steelhead	Fry to age-1+ parr	-	-	-	-	-	7.4%	31.7%	25.6%	21.9%	22.2%	48.6%	24.4%	9.6%	18.0%	16.2%	30.0%	18.1%	19.1%	
Steelhead	Fry to age-2+ parr	-	-	-	-	-	0.9%	9.9%	6.3%	10.4%	8.0%	12.0%	6.9%	3.9%	8.1%	9.9%	7.0%	5.9%		
Steelhead	Age 1+ parr to smol	-	-	-	-	-	68.6%	40.4%	44.1%	35%	55%	44%	32%	38%	68%	92%	121%	57%		
Steelhead	Age 2+ parr to smol	-	-	-	-	-	68.2%	144.0%	193.6%	71.5%	46.7%	66.8%	53.6%	57.9%	50.7%	47.3%	50.8%	75.1%	92.9%	
Chum	Egg-to-fry [1]	-	7.9%	9.5%	3.8%	19.0%	7.2%	13.5%	26.8%	18.1%	26.1%	19.9%	12.1%	29.7%	40.0%	33.5%	20.4%		11.8%	
Pink	Egg-to-fry [1]	-	-	9.6%	-	5.1%	-	9.7%	-	7.4%	-	48.0%	-	27.5%	-	24.9%	-			

[1] Assuming a 1:1 sex ratio for all species and average fecundity values of 3,200, 1,800, 3000, and 3,700 eggs/female for Chum, Pink, Coho, and Steelhead (Groot and Margolis 1991; Ward and Slaney 1993).

[2] Derived from yield of age-2 and age-3 smolts in subsequent years (see Section 5.2.2.2).

Appendix 6.2 References for annual smolt estimates considered for use as controls steams in the BACI analysis for treatment effects for Coho and Steelhead.

Alouette River – Coho and Steelhead

Cope, S. 2015. Alouette River Salmonid Smolt Migration Enumeration: 2014 Data Report. Unpublished report prepared for the Alouette River Management Committee and BC Hydro by Westslope Fisheries, Cranbrook BC, 90 p.

Black Creek, Carnation Creek, Cherry Creek, Keogh River and Sakinaw Creek – Coho

Wade, J. and Irvine, J.R., 2018. Synthesis of smolt and spawner abundance information for Coho Salmon from South Coast British Columbia streams. Can. Manuscr. Rep. Fish. Aquat. Sci. 3161: vi + 39 p.

Keogh River – Steelhead and Coho (post 2015)

Reports prepared by Instream Fisheries Research

Green River – Coho

Topping, P.C. and J.H. Anderson. 2018. Green River Juvenile Salmonid Production Evaluation: 2017 Annual Report. Report prepared by Washington Department of Fish and Wildlife, Fish Program, Science Division

Appendix 6.3 Mean annual Coho smolt abundance estimates 1998-2018 for the Coquitlam River mainstem and watersheds considered for use as controls for regional changes in freshwater productivity.

		Coquitlam Mainstem	Alouette R	Black Cr	Carnation Cr	Sakinaw Cr	Cherry Cr	Keogh	Green R
Year	Flow Treatment	South Coast BC	South Coast BC	Vancouver Island	Vancouver Island	South Coast BC	Vancouver Island	Vancouver Island	Northern Washington
1998	pre		16,200		4,865				
1999	pre		10,200		2,842				
2000	1	11,036	20,003	153,966	4,828		16,438	74,400	
2001	1			42,628	2,205		9,078	59,931	
2002	1	4,838	12,100	89,257	4,740		13,853	60,044	194,393
2003	1	8,195	19,358	81,973	4,539	30,592	5,345	93,578	207,442
2004	1	4,234	16,880	41,250	4,433	13,934	3,476	22,428	
2005	1	3,215	13,020	46,028	2,974	16,264	6,499	58,900	
2006	1	5,979	14,591	124,864	2,248	21,465	8,750	50,403	31,460
2007	1	2,870	3,040	35,370	1,100	15,986	4,708	56,187	22,671
2008	1	9,020	6,508	34,700	682	9,197	2,017	72,064	
2009	both	13,844	40,156	68,517	3,205	32,648	10,306	77,822	81,079
2010	2	6,573	19,885	27,750	2,617		15,943	61,495	43,763
2011	2	7,086	19,240	27,975	2,757	19,830	11,662	89,788	62,280
2012	2	10,935	39,050	32,274	2,861	33,864	11,676	108,063	48,148
2013	2	6,351	31,670	56,856	819	26,860	15,147	111,042	50,642
2014	2	8,080	22,620	55,964	1,386	21,149		66,765	106,365
2015	2	8,234		19,419	1,203	4,094		111,718	42,564
2016	2	5,654		25,424		12,473		91,582	62,074
2017	2	9,810		34,473		2,966		81,502	79,491
2018	2	7,413		40,322				65,084	

Appendix 6.4 The Pearson correlation coefficient and the sample size (N) for the comparison between Coho smolt abundance in the Coquitlam River mainstem and the streams considered as controls for regional changes in freshwater productivity in the Before-After-Control-Impact (BACI) analysis. The comparison included the years 2000-2009, which represents cohorts reared entirely or mostly under Treatment 1. We used correlation coefficient value of 0.5 as the minimum value to include a stream in the BACI analysis.

Stream	Correlation coefficient (R)	N	Included in BACI analysis
Alouette R	0.75	9	TRUE
Keogh	0.62	10	TRUE
Sakinaw Cr	0.60	7	TRUE
Black Cr	0.38	10	FALSE
Cherry Cr	0.36	10	FALSE
Carnation Cr	0.14	10	FALSE
Green River	-0.04	7	FALSE

Appendix 6.5 Covariates used for Coho adult-to-fall fry stock-recruitment analysis. All flow metrics based on average 15 minute or hourly discharge for Coquitlam River at Port Coquitlam (Water Survey of Canada, stn. 08MH002). Covariates were transformed to standard deviations with a mean of zero to maintain equal weighting in the model.

Model #	Model	Mean covariate effect
1	Base Ricker	No covariate
2	Spawning Mean	Mean discharge Nov-Jan
3	Incubation Mean	Mean discharge Feb-Mar
4	Emergence Mean	Mean discharge April-May
5	Summer Mean	Mean discharge Jun-Sept
6	Winter Mean	Mean discharge Nov-March
7	Winter days > 70 cms	Days Nov-March with discharge > 70cms
8	Proportion Aug> 5.4cms	Proportion of days in August above 5.4cms (20%MAD)
9	Stranded	Number of Coho fry salvaged alive and dead following rampdowns March-Sept
10	Treatment	Flow Treatment 1 or 2

Appendix 6.6 Covariates used for Coho adult-to-fall fry stock-recruitment analysis. All flow metrics based on average 15 minute or hourly discharge for Coquitlam River at Port Coquitlam (Water Survey of Canada, stn. 08MH002). Covariates were transformed to standard deviations with a mean of zero to maintain equal weighting in the model.

Model #	Model	Covariate
1	Base Ricker	No covariate
2	Spawning Mean	Mean discharge Nov-Jan
3	Incubation Mean	Mean discharge Feb-Mar
4	Emergence Mean	Mean discharge April-May
5	Summer Mean	Mean discharge Jun-Sept
6	Proportion Aug> 5.4cms	Proportion of days in August above 5.4cms (20%MAD)
7	Treatment	Flow Treatment 1 or 2
8	Stranded	Number of Coho fry salvaged alive and dead following rampdowns March-Sept

Appendix 6.7 Annual Steelhead smolt abundance during the Treatment 1 and 2 flow period in the Coquitlam River mainstem and the Alouette River. The Alouette River was used as control for regional changes in freshwater productivity.

Year	Treatment Period	Coquitlam Mainstem South Coast BC	Alouette R South Coast BC	Keogh R Vancouver Island
2000	pre	3,824	3,392	2,344
2001	pre		2,286	2,010
2002	pre	2,216	3,768	1,892
2003	1	3,812	2,364	4,865
2004	1	3,782	3,355	511
2005	1	3,785	2,493	4,676
2006	1	4,197	784	2,051
2007	1	2,615		965
2008	1	5,497	6,204	1,455
2009	both	5,273	6,191	1,561
2010	both	4,736	15,130	1,889
2011	both	3,052	5,077	1,638
2012	2	4,712	5,778	3,193
2013	2	3,622	5,917	2,620
2014	2	4,579	4,610	2,382
2015	2	4,966		2,981
2016	2	5,086		2,009
2017	2	5,142		2,396
2018	2	5,572		1,471

Appendix 6.8 The Pearson correlation coefficient and the sample size (N) for the comparison between Steelhead smolt abundance in the Coquitlam River mainstem and the streams considered as controls for regional changes in freshwater productivity in the Before-After-Control-Impact (BACI) analysis. The comparison included the years 2000-2009, which represents cohorts reared entirely or mostly under pre-Treatment 2 conditions. We used correlation coefficient value of 0.5 as the minimum value to include a stream in the BACI analysis.

Stream	Correlation coefficient (R)	N	Included in BACI analysis
Alouette R	0.521850241	9	TRUE
Keogh R	-0.025447947	10	FALSE

Appendix 6.9 Covariates used for Steelhead adult-to-fall fry stock-recruitment analysis. All flow metrics based on average 15 minute or hourly discharge for Coquitlam River at Port Coquitlam (Water Survey of Canada, stn. 08MH002). Covariates were transformed to standard deviations with a mean of zero to maintain equal weighting in the model.

Model #	Model	Covariate
1	Base Ricker	No covariate
2	Spawning Mean	Mean discharge March-May
3	Incubation Mean	Mean discharge June-July
4	June Mean	Mean discharge June
5	July Mean	Mean discharge July
6	August Mean	Mean discharge Aug
7	Difference Spawn Incub	Difference in mean discharge between spawning and incubation
8	Proportion Aug> 5.4cms	Proportion of days in August above 5.4cms (20%MAD)
9	Treatment	Flow Treatment 1 or 2

Appendix 6.10 Covariates used for Steelhead fall fry-to-fall parr stock-recruitment analysis. All flow metrics based on average 15 minute or hourly discharge for Coquitlam River at Port Coquitlam (Water Survey of Canada, stn. 08MH002). Covariates were transformed to standard deviations with a mean of zero to maintain equal weighting in the model.

Model #	Model	Covariate
1	Base Ricker	No covariate
2	Proportion Sept> 5.4cms	Proportion of days in September above 5.4cms (20%MAD)
3	Fall Mean	Mean discharge Sept-Oct
4	Winter Mean	Mean discharge Nov-March
5	Spring Mean	Mean discharge April-May
6	Summer Mean	Mean discharge June-Aug
7	Proportion Aug> 5.4cms	Proportion of days in August above 5.4cms (20%MAD)
8	Winter days > 70 cms	Days Nov-March with discharge > 70cms
9	Treatment	Flow Treatment 1 or 2

Appendix 6.11 Preliminary ANCOVA results for Chum 2003-2016 brood years to examine the significance of flow treatment on fry yield during Treatment 1 (2000-2008) and Treatment 2 (2009-2019) in the Coquitlam River including the significance of F values. The null hypothesis in all cases is that the predictive variable is not a significant predictor of fry yield. Escapement x Treatment represents the interaction effect that would produce different slopes of the stock-recruitment relationships for Treatment 1 and 2.

Coefficient	F value	Significance level probablilty (>F)	Null hypothesis prob < 0.05
Intercept	5.2	<0.001	reject
Escapement	27.4	0.06	do not reject
Treatment	9.1	<0.001	reject