Coquitlam-Buntzen Project Water Use Plan

Lower Coquitlam River Fish Productivity Index

Implementation Year 12

Reference: COQMON-7

Study Period: 2000-2017 Results

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Coquitlam River Fish Monitoring Program: 2000-2017 Results

Final Report

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BC Hydro

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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 cms) 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 cms) will be monitored for up to 9 years. The Lower Coquitlam River Fish Productivity Index Monitoring Program (COQMON-07) focuses on four anadromous species: Steelhead Trout (*Oncorhynchus mykiss*) and Coho Salmon (*Oncorhynchus kisutch*), Chum Salmon (*Oncorhynchus keta*) and Pink Salmon (*Oncorhynchus gorbuscha*), and includes adult escapement and smolt outmigration monitoring for each species. Higher returns during 2007-2014 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 of 8 years during Treatment 1 (2000-2008) and the 9 years of Treatment 2 (2009-2017) for the four major components of the COQMON-07: adult salmon escapement surveys, Steelhead redd counts, juvenile standing stock surveys, and smolt trapping. The primary emphasis of this report is on 2016 fall salmon escapement estimates and outmigration, fall standing stock and Steelhead escapement in 2017. 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.

Coho escapement to the Coquitlam River in 2002-2016 (880-13,290 adults) likely exceeded that necessary to seed available juvenile habitat based on preliminary stock-recruitment analysis. The 2015 escapement estimate of 6,867 was based on 14 surveys under relatively favourable survey conditions that spanned the entire spawning period. High water events were minimal and short in duration. As with the Pink, Chum and Chinook, there was insufficient survey life and observer efficiency information to calculate the uncertainty of Coho escapement estimates. The 2017 late summer standing stock fry estimate was 59,166 (95%CI ±29%) based on night snorkel surveys, which was above average compared with previous years (18,405-91,367 fish). During 2017, 9,810 (95%CI ± 1,080) Coho smolts originating from mainstem habitats outmigrated past the lowermost trapping site and an additional 4,340 smolts outmigrated from the four off-channel habitats above RST2. Using smolt yield as the primary measure of freshwater carrying capacity, there was no difference in mean smolt abundance between Treatment 1 and Treatment 2 when including off-channel smolts (mean: 12,949 and 12,507 smolts, respectively; 2-tailed t-test p=0.97) but is significantly different for mainstem origin smolts (mean: 5,479and 7,848, respectively; 2-tailed t-test p=0.05), when excluding the 2009 transition year. This reflects a 43% increase in mean abundance from Treatment 1 to Treatment 2. However, the 95% confidence intervals of the percent change in smolt yield from Treatment 1 to Treatment 2 are broad, ranging from no-increase to a nearly two-fold increase in abundance (95% CL: -0.5% to 87%). While this may represent a meaningful change in abundance, we are uncertain whether the

change was due to Treatment 2 flows or a regional increase in freshwater productivity that occurred over a similar time period. If smolt yield from the Alouette River– an adjacent watershed with similar hydrology – is a reliable control for changes in productivity in the Coquitlam River that would have occurred if there was no change in flow treatment, Treatment 2 may have had no effect or even a negative effect on productivity. This is because there was a greater increase in smolt abundance in the Alouette River (117%) over a similar time period than in the Coquitlam (mean increase: 43%). However, the reliability of the Alouette Rivers as a control is uncertain because of its low precision of the estimated change in abundance between treatments (95% CI: 25%-250%), including only the first five of nine Treatment 2 years, and the possibility that pre-2008 estimates were biased low.

Redd counts suggested that Steelhead escapements during 2005-2015 (230-870 adults, 24 to 80 adults/km, 39,000-149,000 eggs/km,) were well above that necessary to seed available juvenile habitat based on stock and recruitment data for the Keogh River, a well-studied coastal stream, and based on preliminary stock-recruitment analysis from the Coquitlam River. The 2017 estimate of 398 adults was minimally influenced by modeling redd loss since the period between surveys was sufficiently short that virtually all redds constructed after one survey remained visible during the subsequent survey. The late summer standing stocks and 95%CI of Steelhead fry, age 1+ and 2+ parr for 2015 was 54,358 ±44%, 9,064 ±28% and 3,207 ±50%, respectively. Smolt yield for the mainstem upstream of the lowermost trap (RST 2) was 3,696 ±16% in 2017.

Mean smolt yield for the Coquitlam River mainstem increased 24% from Treatment 1 to Treatment 2 (mean: 3,716 and 4,684 smolts, respectively). Although this result isn't statistically significant, we expected that it will with additional monitoring if current trends continue (2-tailed t-test, p=0.07). The 95% confidence intervals of the percent change in smolt abundance from Treatment 1 to 2 ranged from -5% to 53%. This increase was almost entirely the result of increased smolt yield in reach 4. Mean abundance in reach 4 increased significantly from Treatment 1 to Treatment 2 (926 and 2,253 smolts, respectively; 2-tailed t-test, $p< 0.01$). As well, reach 4 abundance 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.

Similar to Coho, increased Steelhead production may be the product of regional increases in freshwater productivity rather than Treatment 2 flows. If smolt yield from the Alouette River is a reliable control for changes in productivity in the Steelhead in the Coquitlam River that would have occurred if there was no change in flow treatment, Treatment 2 may have had no effect or even a negative effect on productivity. There has been a greater increase in mean smolt abundance in the Alouette River (78%) than for the Coquitlam River (24%) between Treatment 1 and 2. We are uncertain about the reliability of the Alouette River as a control for the same reasons as for Coho: low precision of the estimated change in abundance between treatments (95% CI: 10%-132%), small Treatment 2 sample size (3 years) and the possibility that pre-2008 estimates were biased low. Distinguishing between flow treatment and non-treatment effects will require further assessment of the reliability of the Alouette River as a control and or including considering other nearby rivers as controls.

Chum escapement for 2016 was the highest since monitoring began in 2002 (78,120 adults). This was based on 11 evenly spaced surveys with moderate to good survey conditions. Considering this, the 2016 escapement estimate is likely a reliable index for evaluating freshwater production even though we still lack adequate information about survey life and observer efficiency. In 2017, 12.7 million (95% CI ± 17%) Chum fry outmigrated past the lowermost trap. This is likely a product of the exceptionally high escapement in 2016. 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. 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 2002-2017 compared to escapements in years prior to the implementation of the Treatment 1 flow regime in 1997.

No Pink escapement estimates were generated for 2016 or outmigration estimates for 2017 as spawning occurs during odd years for the Coquitlam River. Escapement during the most recent run year, 2015 was comparable to the 2009 and 2011 runs and less than 1/3 of 2013 returns. Fry yield has ranged from 150,000-6.7 million with yield during Treatment 2 10- to 20-fold over Treatment 1 fry yield 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. With only two run-years under Treatment 2 conditions, between-treatment comparisons are weak and inconclusive. Future evaluations of the fisheries benefits of test flows may be complicated by non-comparable escapements during Treatments 1 and 2 if the current abundance trends continue and will likely rely on comparisons with other rivers.

The Chinook escapement in 2017 was 511 adults, which has been common since 2014. Escapement ranged from 360-8,000 adults during 2007-2014, and was likely less than 300 adults prior to this period. The highest Chinook escapement occurred in 2010 (8,018 adults).

COQMON-07 Status of Objectives, Management Questions and Hypothesis after Year 17

Primary Objective	Management Question	Management Hypothesis	Year 17 (2017) 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 Treatments 1 and 2	H_0 -not rejected As precision improves, a change to reject expected. Uncertainty distinguishing between flow treatment effects versus regional effects. Section 6.2
		H_{01} – Coho smolt production does not differ between Treatments 1 and 2	H_{01} –reject for mainstem Coho smolts but no support for rejecting off-channel and mainstem smolts combined. Uncertainty distinguishing between flow treatment effects versus regional effects. 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 Treatments 1 and 2. :	H_{03} – possibly rejected Awaiting larger Treatment 2 sample size and incorporation of year-specific uncertainty. Uncertainty distinguishing between flow treatment effects versus regional effects. Section 6.3
		H_{04} –Each adult Pink produced the same fry yield during Treatments 1 and 2.	H_{04} – not rejected Insufficient data for analysis. Awaiting larger Treatment 2 sample size and incorporation of year-specific uncertainty. Section 6.4

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1.0 Introduction

The Coquitlam-Buntzen facilities Water Use Plan (LB1 WUP) was initiated in September 1999 and concluded 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 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 current regime of two fish valves fully open (Treatment 1), and a new schedule of monthly flow releases prescribed by -CC (Treatment 2; Table 1.1) that attempts to improve spawning and rearing habitat conditions in the Coquitlam River relative to Treatment 1.

1.1 Background

The lower Coquitlam River flows 17 km from the base of Coquitlam Dam to its confluence with the Fraser River. The stream was first dammed in 1903. The present dam dates from 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 that release flows from Coquitlam Reservoir. The Coquitlam Reservoir also 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 also contribute 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 discharge of pollutants. Peak, post-dam flows in Coquitlam River can exceed 200 cms (Water Survey of Canada, Station 08MH141). Prior to June 1997, flow releases from the dam ranged from 0.06 to 0.5 cms (not including occasional spill events). From 1997 to October 2008, minimum flow releases were increased to a range of 0.8 to 1.4 cms, depending on the time of year. This represents the Treatment 1 regime of two fish valves fully open, and is the baseline for this adaptive management study.

The Treatment 2 flow regime (i.e., Coquitlam River CQD LLOG3 knife gate) was initiated on October 22, 2008, with seasonal target flow releases from Coquitlam Dam ranging from 1.1 to 6.1 cms (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 cms 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. With respect to the flow experiment then, 2009 was not strictly representative of Treatment 2. However, given the planned 9-year duration of Treatment 2, this is not likely 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 at low numbers, and Dolly Varden (*Salvelinus malma*) char, which appear to have been extirpated. Dam construction resulted in the extirpation of an anadromous stock of summer Sockeye Salmon (*Oncorhynchus nerka*), but this species stills exists in Coquitlam Reservoir in its resident form (Kokanee). Other species inhabiting 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, 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 long freshwater residency periods (Coho and Steelhead), and that increased fall and spring base flows could improve spawning success for all anadromous salmonids.

LB1 WUP was developed as an adaptive management study with the objective to 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 1flow conditions followed by nine years under Treatment 2 conditions. While Steelhead Trout and Chinook Salmon were identified as the key species of interest, low Chinook abundance necessitated the use of Coho, Chum and Pink salmon 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 predictions ranged from a 0% - 100% increase in productivity from Treatment 1 to Treatment 2.

The Terms of Reference (BC Hydro 2006) anticipated this experimental design to have a 0.51-0.59 probability (power) to detect a 50% increase in abundance (effect size) and characterizes this level of power as ‘moderate’. However, this level of power would more typically be considered ‘low’. Power levels of 0.51-0.59 reflect the odds of detecting a given changes of slightly better flipping a coin. The power analysis of the Coquitlam WUP Monitoring program (Higgins et al. 2002) defined reliable inferences as having a power approaching or exceeding 0.8. This review will use 0.8 as the benchmark for acceptable power.

COQMON-07 focuses on four species: Steelhead Trout (*Oncorhynchus mykiss*), Coho Salmon (*Oncorhynchus kisutch*), Chum Salmon (*Oncorhynchus keta*), and Pink Salmon (*Oncorhynchus gorbuscha*). Other fish species are either of too low abundance to effectively monitor (this appears to be changing for Chinook Salmon (*Oncorhynchus tshawytscha*), see Section 1.2.1), or are considered to be of lower 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. However, 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 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 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 (e.g., is overwintering habitat more important than summer rearing habitat in limiting juvenile carrying capacity?).

For Chum and Pink, which emigrate 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 anadromous salmonids in the Coquitlam River.

COQMON-07 focuses on the effects of dam releases on fish productivity in mainstem habitat in reaches 2a, 2b, 3 and 4, of the Coquitlam River (Figure 1.2). This section contains 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, a high gradient, nutrient-poor stream, with limited accessible length, is the only significant tributary (Figure 1.2). There are several other tributaries, but they are very 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 of habitat have been constructed in reaches 2-4 (Decker and Foy 2000). The contribution of tributaries and off-channel sites to 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 lower reaches of several of the small natural tributaries are also used by Coho for spawning.

The principal objective of this report is to summarize fish productivity in the Coquitlam River during Treatment 1 and the first eight 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 existing study designs, and recommended changes to be applied in 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 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), 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 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 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 enhancement (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 will be limited to six, three and five years' data for Chum, Pink and Coho, respectively (smolt abundance is not estimated for Chinook).

During 2002-2017, 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 includes escapement results for returns up to the 2015 spawning period. Results from spawning during the fall of 2016 will be included in the 2000-2017 summary report.

1.2.2 Adult Steelhead escapement

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 will be available for 2007 onward only, which limits egg-to-smolt survival estimates to just one year for Treatment 1 (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), high variation among individual fish in stream residence time (Korman *et al.* 2002), and 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 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 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 counter (Korman and Schick 2015), mark-recapture (Jacobs *et al.* 2002) or 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 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 (based on redd counts and assumed sex ratio and fecundity values) should properly be considered indices of abundance.

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 2017 juvenile standing stock survey, and summarizes preliminary population estimates for 2006-2017.

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 2000-2017, 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 2017 smolt trapping program and summarizes population estimates for all species and reaches for 2000-2017.

2.0 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. 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 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 (M. Coulter-Boisvert, DFO, pers. comm.), 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 2002-2016. 2016 escapements have not been reported previously.

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.

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 will likely be 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 are surveyed within each index site, including mainstem areas, natural side-channels and braids, and constructed off-channel habitat.

To account for spawners that are present in the study area, but not in one of the five index sites, on several occasions each year, the survey is 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 are occasions each year when it is 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 unsurveyed 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 consistency likely reduces 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, for Coho, shore-based observations were less effective 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 snorkelling gear, in addition to 1-2 shore-based observers. Comparisons of counts made by snorkelers and shore-based observers suggested that snorkelers detected four- to six-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 2002-2016 it was not uncommon 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. 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. 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 2.1). 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 (see Section 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 periods likely results in substantial variability in observer efficiency among surveys within years, and, in some cases, among years as well (see Section 2.2.2). Substantial variation in water visibility (and hence observer efficiency) among index sites during individual surveys is also common. This source of error is potentially important because variation in observer efficiency among years that is unaccounted for could bias comparisons of adult abundance and egg-to-smolt survival among years and between flow treatments. To address this, during 2002-2016, the survey crew developed a relative index of survey conditions by subjectively 'guesstimating' observer efficiency (0%-100%) for each index site during all surveys. While these guesstimates do not reflect actual observer efficiency, they are potentially useful predictors of mark-recapture-derived estimates of observer efficiency. Since the surveyors record their guesstimates of observer efficiency for every site during every survey, these data were used to model variation in observer efficiency among surveys in the escapement model based on a 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, 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 reliable 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.

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

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 – the number of days a fish spends in the survey area (i.e., are visible to an observer) – 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 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 overdispersed 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 overdispersion 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,

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

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,

$$r_i \sim dbin(\gamma_i, m_i) \qquad (2.9)$$

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,

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

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 are shown for Chum, Coho, Pink and Chinook in Appendices 2.2-2.5. 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. This resulted in the survey period encompassing nearly the entire migration of all target species (Appendices 2.2-2.4). The reliability of estimates depends on surveys encompassing the entire migration but particularly peak conditions as we the case for the three species present in 2016 (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 for 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 were sparse in the latter half of the spawning period on account of high flows (Appendix 2.6). 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 2.3, 2.5). The beginning of the spawning period was well defined for Coho, but in some years of the study (2002, 2004, 2005, and 2011); significant numbers of Coho were still present during the final survey (Appendix 2.4). For modeling purposes, the maximum length of the spawning period 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.

Water column visibility ranged from 0.7 - >3.0 m and averaged 1.5 m among surveyed sites in 2016 (Table 2.1). During 2016, survey conditions were moderate allowing for 19 surveys with 15 of these less than 10 days apart and four 10-17 days apart. If omitting the October 19 survey, which included only Site E, this would increase the maximum period between surveys to 18 days. As has been the case during previous years, visibility during 2016 was highest during September (>3m) and variable but typically lower during the remaining surveys (0.7-1.4m). Visibility in the lower river below the gravel mines was sufficient to not precluded surveys at index sites A and B during 2016. Once Chum spawning was complete (late November) surveys excluded Site A since Coho counts from this section are typically less than 1% of the total for any survey (see Table 2.4).

2.2.2 Observer efficiency and survey life

2.2.2.1 Observer efficiency

During 2006-2016, 22 estimates of observer efficiency were obtained for all species combined. None were conducted during 2015 due to the frequency of high flow events. In some of the 22 cases, the field crew were 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 2.1). 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 for Chum, one for Pink and none for Coho and Chinook (Appendix 2.1).

Observer efficiency for Chum averaged 48% across 10 mark-recapture experiments during 2006-2015 (range: 33%-69%; Table 2.2); with similar means for Treatment 1 and Treatment 2 (50% and 48%, respectively; Appendix 2.1). For Pink, seven mark-recapture experiments yielded an average observer efficiency estimate of 66% (range: 49%-85%; Table 2.2, Appendix 2.1). 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 probably lower than the Treatment 2 average of 71%; and was likely at least as low as 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, 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 than Pink and Chinook, when poor survey conditions occur more frequently, observer efficiency estimates obtained likely represent the upper range for the Coquitlam River, rather than average values. 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 are unable to confirm how closely the observer guestimate matches mark-recapture derived observer efficiency through the full range of survey conditions. The uncertainty from this is considered one of the reasons the HBM is unable to estimate precision of escapement estimates. It is important that every effort be made in future to conduct mark-recapture experiments later in the season, and during periods of higher flows and lower visibility, so that the actual range in observer efficiencies is captured by 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 2006-2016, 13 of 22 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 2.1). Bias in sex ratio of marked populations will result in 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 2006-2016 provided limited information about survey life for each species. No estimates of survey life were conducted during 2015 due to the frequency of high flow events. 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 not possible due to unsuitable survey conditions. A total of 18 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. 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 2.1). 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 problem was likely exacerbated by the fact that during many of the mark-recapture experiments, fish were captured and tagged in spawning areas in 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, which would lead to a negative bias in the estimate of survey life. By the same token, a decline in observer efficiency over time would lead to positive bias in estimates of survey life.

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 won’t be able to confirm the magnitude of difference between early and late run fish.

With a biased estimate of survey life, escapement estimates would still provide a reliable index of adult abundance as long as the bias is similar for all years. Survey life acts as a scaler in the HBM and since a similar survey life value is applied to all years for a given species, the escapement estimates by be biased they would still reflect the relative abundance between years.

2.2.2.3 Modeling observer efficiency and survey life

For Chum, subjective guesstimates of observer efficiency made by the survey crew for surveys for which mark-recapture estimates of observer efficiency were available ranged from 55% to 90%, and average 74% (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 2.5). For Pink, surveyor guesstimates ranged from 55% to 95% 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 Pink (n=7; R^2=0.23; Figure 2.5). However, this relationship is highly uncertain, being based on only seven observations. The regression relationships for Chum and Pink were used in the escapement model to estimate observer efficiency for individual surveys based on surveyor guesstimates of observer efficiency, and to model error in estimated observer efficiency (see equations 2.8 and 2.9). For Coho, over the three surveys surveyor guesstimates 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 2.5), 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 average observer efficiency in the Coquitlam River might be. In light of the poor relationship for Coho and limited information for Chinook, 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 2.5). 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 suffucient 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 2.7. 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 2.4. Among years, estimated escapements ranged from 7,000-78,000 for Chum; 900-13,000 for Coho; 3,000-34,000 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 how it varies) on the number of adult returns. Trends reported here are products of freshwater and/or marine conditions. For all species, escapement has been much higher during Treatment 2 than during Treatment 1 (Table 2.4). For Pink, Coho and Chinook, mean escapement has been increased four-fold for Pink and Chinook, three-fold for Coho and two-fold for Chum compared with Treatment 1. Escapement estimates for Coho and Chinook during Treatment 1 years should be treated as approximations and are likely non-comparable to Treatment 2 (See section 2.2.2.1). Estimates shown here for Coho and Chinook during Treatment 2 years may be biased low if the limited mark-recapture information collected for these species to date is in fact representative of observer efficiency (we used pooled mark-recapture data for all species to estimate observer efficiency for Coho and Chinook; see Section 2.2.2.3).

Escapement estimates generated for 2002-2016 in future reports will differ to some degree from those reported in Table 2.4, as more information about observer efficiency and survey life is collected. 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 of the study 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 are not able to say how many mark-recapture experiments are necessary to achieve this since the model, and survey life, data is too sparse to estimate the uncertainty of escapement estimates.

2.2.5 Alternative approach to monitoring changes in escapement

The HBM approach to estimating adult escapement still has several shortcomings, which make evaluating their accuracy and precisions difficult; see sections 2.2.2 and 2.2.3. While efforts continue to collect more information on survey life and observer efficiency, we also explored alternative approaches to estimate escapement both to corroborate the HBM results and as a possible replacement. Escapement estimates for Pink, Coho and Chinook during 2002-2016 are highly correlated with the mean counts of annual surveys (Pink, n=7 R=0.92; Coho, n=15, R=0.96 and Chinook n=9, R=0.94). Under some conditions, particularly consistent survey methods across years and evenly distributed surveys, mean counts can be an equally or more effective metric for detecting changes in escapement than mark-recapture and AUC (Holt and Cox 2008). Also, since it depends entirely on raw count data, it isn’t affected by the uncertain estimates of survey life and observer efficiency. However, mean count is susceptible to under-estimation when surveys do not include peak run (Holt and Cox 2008). This was the case in 2012 for Chum where high flows prevented surveys during the peak run. The mean count would have indicated a below average escapement in 2012 whereas the HBM, which accounts for this with run timing priors, estimated the second largest escapement to date.

We previously proposed to use the peak count as an index for escapement rather than continue with the HBM approach for Coho and Chinook as a cost saving measure and since there is a very low chance of collecting sufficient observer efficiency or survey life information during the study period (Schick *et al.* 2014). For Coho, we still support this approach since we are more interested knowing that the minimum escapement has been reached to fully seed the Coquitlam River with juveniles and there has been little gained from post-peak surveys. For Chinook, there is no additional cost for surveys using either the HBM or mean count method so there is no advantage of switching methods at this point. Under this approach, Coho surveys would end after the peak count (early December), and for both species, survey life and observer efficiency experiments would no longer continue. The reduced survey effort could then be redirected to Chum and Pink to increase the number of survey life and observer efficiency experiments to a level necessary to estimate the precision of escapement.

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 2.5). This preference has been noted in many studies for Chum salmon in medium-sized rivers (Salo 1991). 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 2002-2016; Table 2.3). 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.

Pink salmon also have a brief freshwater residency period, but unlike Chum, Pink spawners made 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 2.3). During Treatment 1 and 2, Pink salmon made greater use of mainstem sites for spawning than off-channel (Treatment 1: 55%-71%, Treatment 2:59-76%; Table 2.5). 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 2.5). It is not clear if this is an artifact of reduced observer efficiency in the mainstem when flows increased after October 22 or to the increased availability of off-channel habitats. Higher mainstem flows under Treatment 2 gave salmon easier access to off-channel habitats, and increased the amount of available habitat in some constructed off-channel sites and natural side-channels. The increased flows also provided new spawning habitat in previously unused side-channel and mainstem areas.

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 2002-2016; Table 2.3). The trend of low natural or enhanced off-channel habitat use during Treatment 2 continued in 2016 with usage of 6% (Table 2.5). The combined natural and enhanced off-channel habitat use dropped from 20%-73% during 2002-2007 to 6%-16% during 2009-2016 (Table 2.5). This shift commenced prior to Treatment 2 and coincided with the modifications to 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 varies considerably below this level and not in relation to the overall escapement level (Figure 2.5). With the several fold higher escapement levels 2010 onward, use increased disproportionately in the mainstem habitat.

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.

2.2.7 Temperature

Optimal 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). Reach specific stream temperature monitoring did not occur during the 2016 spawning and incubation period.

2.3 Implication for hypothesis testing

Adult escapement monitoring is providing sufficient information to evaluate the fisheries benefits of Treatments 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 fully seed the river (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 yet all Coho mark-recapture experiments occurred during Treatment 2.

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, Chum and Pink stock-recruitment relationships to date point to escapement-limited fry production (Figure 6.6), which depend 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 still subject to error with sparse data. As mentioned in Section 2.2.3, we think that our inability to calculate credible precision estimates stems from the lack of Coquitlam-specific survey life information and/or insufficient mark-recapture data.

3.0 Adult Steelhead Escapement

3.1 Methods

During 2005-2017, 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). 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 some cases 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 Coquitlam Dam downstream to 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 to early-May). Analysis of previous years' data suggested that conducting weekly surveys ensures that redd survey life exceeds the survey interval but that very minimal redd loss would occur if less than 14 days (see Section 3.1.3). 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 that 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 shallow 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 forklength, sex, and whether they lacked an adipose fin indicating hatchery origin. Data for live adults were not used to estimate 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 (i.e., 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 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 streams. Based on five years' data from several coastal Oregon 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 2017, with intervals of 14 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 14 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.

$$\boldsymbol{N} = \left(\sum_{i=1}^{n} \boldsymbol{x}_i\right) \tag{3.1}$$

3.1.4 Female escapement and egg deposition

The objective of the Steelhead redd survey component is to allow smolt production to be related to spawning effort. Redd numbers are a direct measure of spawning effort and egg deposition. So, for our purposes, estimating the total number of redds is arguably as useful as estimating 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, 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 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 from this study ranged from 0.75 to 1.63 and averaged 1.2, with relatively little variably 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:

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

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 in the Keogh River on northern Vancouver Island (3,700 eggs/female, Ward and Slaney 1993). We assumed 50% of adult Steelhead in the Coquitlam River were female, which is commonly reported for 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 2005-2017, 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, periods of high discharge were relatively infrequent during the spawning period, with mean daily discharge rarely exceeding 20 cms at Port Coquitlam (WSC 08MH002, Figure 3.3). In contrast, during 2007, mean daily flows remained above 10 cms for most of March, with a peak flow of 118 cms occurring on March 11. However, during April and May, 2007, when most spawning occurred, flows remained for the most part below 10 cms. During 2005-2008, surveys were conducted at flows of between 2-4 cms. During 2009-2017, increased discharge from Coquitlam Dam under Treatment 2 resulted in higher base flows compared to previous years. Mean daily flow exceeded 10 cms on 23-50% of days during the major spawning period during Treatment 2, versus 4-24% in previous years, Figure 3.3), while average discharge during the Treatment 2 spawning periods (8.75-10.7 cms) was about double that in previous years (4.3-5.6 cms), with the exception of 2007 (13.0 cms). On days when redd surveys were conducted during Treatment 2, average discharge (7.1 cms) was more than double that on most survey days during Treatment 1 (2-4 cms). Frequent 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 2017; across the entire survey period, the length of time between surveys ranged from 8 - 14 days, and averaged 10 days (Table 3.1). In previous years, the length of time between surveys has ranged from 6 to 37 days (Table 3.1).

In 2017, the first survey was conducted on March 17. Similar to past years, live adult Steelhead were observed (23 fish, Table 3.1), along with a small amount of spawning (11 redds were counted). Across all years, no or minimal spawning (<5% of annual total) occurred by the time of surveys conducted prior to March 12 and 9% - 18% of new redds were constructed by the end of March (Table 3.1). These results suggest that Steelhead typically begin spawning in the Coquitlam River in early March but that most spawning occurs (80-90%) during a six-week period spanning early April to mid-May (Table 3.1, Figure 3.4). In 2017, 80% of new redds were counted on surveys conducted from March 31 to May 6.

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 2017, 85% 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 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 2009-2017 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 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 has been generally higher than during Treatment 1 (37 – 148, Table 3.1). 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 (Table 3.1). However, peak counts should be considered a less reliable index of year-to-year differences in total escapement compared to redd counts. Unadjusted peak live 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 35% of variability of escapement based on redd counts (Figure 3.5), in large part because of a small number of years when adult peak counts were far less then would be expected from redd counts.

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

3.2.1 Redd survey life

In 2017, the period between surveys was typically sufficiently short (≤ 14 days) to assume that only a negligible number of redds became obscured from one survey to the next based on the evaluation of redd survey life during 2005 - 2017. Thus, the estimated number of new redds was equal to the raw count of new redds (Table 3.1). From 2005-2017, 2006 remains the only year where the number of redds estimated using the redd survey life model was substantially higher (21%) than unadjusted counts due to a 37-day gap between surveys during the peak spawning period (Table 3.1). See Decker *et al.* 2010 for further discussion of trends in survey life.

3.2.2 Female escapement and egg deposition

Estimated adult female escapement in 2017 was 199 females (Table 3.2), an average value for the 2005-2017 period. Highest and lowest female escapements occurred in 2006 (434 females; Table 3.2) and 2009 (113 females), respectively. Average Steelhead redd density in the study area of the Coquitlam River was 22 redds/km in 2017, and ranged from 13-48 redds/km during 2005-2017 (Table 3.2). Among reaches and years, redd density ranged from 6-71 redds/km (Table 3.2). Spawning distribution during 2017 was balanced between the lower river, reaches 2a and 2b, and the upper river, reaches 3 and 4 (52% and 48%, respectively, Table 3.2) similar to 2005-2007 and 2010-2012. 2008 and 2009 had the proportion of total redds found in the upper river reduced to 29% and 38%, respectively, whereas 2013 over 60% of total redds were found in that section of the Coquitlam River. Figures 3.1 and 3.2 illustrate the fine-scale distribution of redds in the study area in 2006.

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 parameter. We used average values of 1:1 for sex ratio, and 1.2 for the number of redds per female based on empirical data from several coastal streams (Jacobs *et al.* 2002; see Methods) to develop escapement estimates. Jacobs *et al.* (2002) reported two-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 Treatments 1 and 2, this is encouraging, since the accuracy of the estimates is of secondary importance, so long as sex ratio and redds/female ratios remain constant between treatments.

3.2.3 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 Treatments 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 2005-2017.

4.0 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 being that these data together with adult escapement and smolt abundance estimates, could be used to investigate freshwater production bottlenecks at specific juvenile life stages that may relate 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 and night snorkeling counts at sites that extended across the entire stream channel (Decker *et al.* 2007). During 2007-2017 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 that was developed to provide estimates of juvenile standing stocks in the Coquitlam River during 2006-2017 (see *Section 4.1.5*); this model replaces a bootstrap model used in previous years (Decker *et al.* 2012).

During 2007-2017 we also conducted a separate electrofishing survey (with input and assistance from Ron Ptolemy, MOE stock assessment). As per the COQMON-07 Terms of Reference, the electrofishing data were collected to provide a comparison of fish densities in specific habitats in the Coquitlam River with fish densities in similar habitats in other streams that were sampled using the same methods (BC MOE juvenile electrofishing database; see Ptolemy 2007). The electrofishing data were 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 Lougheed highway (i.e., reaches 2a, 2b, 3, and 4; Figure 4.1), and includes all mainstem, braid and sidechannel habitat. Natural and man-made off-channel habitats in the Coquitlam River were not included, and juvenile fish populations in these habitats are 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 2007-2014 that are sampled each year with another 12 index sites added in 2014 increasing the sites surveyed 2014-2016 to 24. 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 lengths.

For this type of sampling design, error in the estimation of fish standing stock 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 fish. The Hierarchical Bayesian Model 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 (unstratified) systematic sampling design (SSS). Sampling was not stratified by reach or habitat type on account of 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 $\approx$ 0.85 km (Figure 4.1; reach 4 was not sampled in 2006). The additional 12 sites added in 2014 (for a total of 24 sites) were placed 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 cms under Treatment 1 (2006-2008) and 2.2 cms under Treatment 2 (2009-2017). 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, and, 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 forklengths 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 we did not 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 forklengths 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 at 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 will be minimized. Now that we have well defined detection probability information for Coho, age-0 fry and age-1 Steelhead parr, we suspended further mark-recapture experiments in 2014. Further mark-recapture experiments would not increase the precision of the standing stock estimates as much as a doubling of the number of index sites would, possible by 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 smolt 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 as described above, a single snorkeler captured and marked fish throughout the site using one or two large aquarium nets affixed to handles of approximately 80 cm in length. The snorkeler searched for and captured fish throughout the site; with the goal of obtaining 10-20 marked individuals each 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 (forklength to nearest 5 mm), marked it, and returned it to its original location once 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 forklength, 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 uncertainty about 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. Over sufficiently short time periods, however, and 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 in 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 a Hierarchical Bayesian Model (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 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).

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 snorkelling 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 in the observation component of the HBM models, if 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 to total area of the studied zone of the Coquitlam River is not measured each year.

4.1.6 Day electrofishing survey

In 2017 we resurveyed four shoreline electrofishing sites previously sampled during 2007-2016. 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 as to species, measured for forklength (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 2009-2017 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 taken at the time of sampling at each site).

4.2 Results

4.2.1 Night snorkeling

In 2017, night snorkeling surveys were completed during August 30-September 1 and September 4, 6, 9 and 11 at a flow of 2.7-3.0 cms (WSC station 08MH002). Previous surveys were conducted at flows of 0.8-2 cms during Treatment 1, and 2-7 cms during Treatment 2. Water temperatures ranged from 19-20°C during 2017, similar to previous years. In 2017, 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.

4.2.1.1 Mark-recapture experiments to estimate snorkeling detection probability

No additional mark-recapture experiments were carried out in 2017. 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 4.2). 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 4.2, 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.

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.

4.2.1.2 Juvenile fish distribution and abundance

The 2017 Coho fry abundance in Coquitlam River mainstem was 59,166 (±29%), which was near average compared with previous years (mean: 50,658 fish, Table 4.3). Precision of estimates has increased substantially with the doubling of the number of sampling sites from ±38-44% 2006-2013 to ±30-31% 2014-2016 (Table 4.3). Averaged by treatment, Coho density generally increased with distance upstream during Treatment 2 (R^2 = 0.74) whereas during Treatment 1 density was lower between km 8-11 but was variable above this point (Figure 4.3).

The 2017 Steelhead fry abundance of 54,358 (±44%) was near average compared to previous years 2006-2016 (mean: 48,531, Table 4.3). We consider the 2017 fry estimate relatively credible (minimally biased) since the assumptions underlying the mark-recapture methodology were largely satisfied. There was an increased proportion of fish less than 40mm fork length, which was the minimum size included in mark-recapture experiments (<40mm). . In 2017 this was 10% whereas it has been close to 5% in past years, with the exception of 2011. This may have biased the fry estimate low since observer efficiency decreases with the size of fry (Korman *et al.* 2011) and yet the HBM use an observer efficiency values for fry 40-49mm. The underestimation would likely be relatively small considering that the proportion of observations for fry less than 40mm from electroshocking was far more similar to an average year than 2011, when the large shift towards smaller sized fry led to a very unreliable snorkeler based estimate (see Schick *et al.* 2012, Appendix4.4). Relative precision increased substantially since doubling the number of index site in 2014 (±28-44%) compared to prior years (±37%-1790%). 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 4.3), whereas in 2008-2017, density was far less variable and with no clear trend.

The 2017 standing stock estimate of age-1+ Steelhead parr of 9,064 ±28% was near the average abundance since commencing snorkel surveys (mean: 8,739 fish, 2006-2016, Table 4.3). The density of age-1+ parr in 2017 was 880 fish/km or 4.8 fish/100m^2. Mean density of age-2+ Steelhead parr in 2017 was 311 fish/km, which continues the trend of consistently higher densities since 2009 (199-372 fish/km during 2009-2016; Table 4.3) than that during 2006-2008 (112-177 fish/km). This trend corresponds well with the transition between flow treatments for age-2+ parr that have spent one year under Treatment 2 conditions. 2011 would be the first estimate for age-2+ parr that reared entirely under Treatment 2 conditions. However, 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 4.3).

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 case in 2017 but to a far less degree than 2011 leading us to treat this estimate 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 from ours (three-pass electrofishing), lower flows allowed them to extend 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 2006-2017 (13-42 fish/100 m^2) were 3-9 times that in 1997 (5 fish/100 m^2, Riley *et al.* 1997). Compared to Steelhead fry density in 1997 (12 fish/100 m^2), Steelhead fry densities in 2006-2017 were up to two-fold higher (8-28 fish/100 m^2). Steelhead parr densities were 6-19 times higher during 2006-2017 (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 2006-2017 and 1997 are likely too large to be explained by negative bias 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 Treatments 1 and 2 relative to earlier years (0.06 to 0.5 cms) likely contributed to increased juvenile fish production in the Coquitlam River.

Based on the calibrated snorkeling data, Steelhead fry density in Coquitlam River in 2006-2017 (9-29 fish/100 m^2) was typically low compared to published values for other streams. For example, Hume and Parkinson (1987) considered 30 Steelhead fry/100 m^2 to be about average in BC coastal streams. Ward and Slaney (1993) reported that Steelhead fry densities in Keogh River averaged 34 fish/100 m^2 one month after emergence. High Steelhead fry density in the Coquitlam River in 2006, 2011 and 2013 was associated with a relatively high brood escapement (see Section 3), which is consistent with the positive linear relationship between Steelhead escapement and fry abundance that has been observed in other streams (e.g., Keogh River, Ward and Slaney 1993).

Snorkeling-derived estimates of Steelhead parr density 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).

Coho densities in the Coquitlam River (15-42 fish/100 m^2) were much lower than the range of mean Coho densities observed at annual index sites in 15 other Lower Mainland streams (59-455 fish/100 m^2, respectively; DFO, data on file), although these streams were considerably smaller, and were sampled at sites chosen to represent 'good' Coho habitat. It is important to note that constructed off-channel habitat contributes about half of Coho smolt production in the Coquitlam River, and numbers of Coho fry from off-channel areas were not included in our estimates of mean densities and standing stocks in Table 4.3.

Overall, these comparisons suggest that the Coquitlam River mainstem may be a more productive stream for Steelhead than Coho, which is not surprising given its relatively high gradient and large substrate.

4.2.4 Fish densities in 'optimal' habitats based on electrofishing

Table 4.4 contains density and river-wide abundance estimates based on electrofishing. There was little agreement between density estimates based on electrofishing and snorkelling across years for all species and age classes ($R^2 < 0.2$, Table 4.6). 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.

Electrofishing surveys in the Coquitlam River during 2007-2017 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.). This suggests that carrying capacity in the Coquitlam River exceeds the average for its 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 in the Coquitlam River relatively to streams of comparable nutrient richness.

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), respectively. 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 Treatments 1 and 2, respectively. Thus, observed maximum values during 2006-2017 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 2006-2017 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 snorkelling 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 2006-2017 standing stock estimates are likely too low to detect between-treatment differences for all species age classes, it does provide useful information for distinguishing at what life-stage abundance may become limited by adult escapement versus rearing habitat availability.

5.0 Smolt and Fry Production

5.1 Methods

5.1.1 Coho and Steelhead smolt enumeration

In 2017, 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).

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. During 2006-2017, 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. 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 fourth RST (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 was estimated. 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

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

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), and this will result in some degree of negative bias in our estimates of egg-to-smolt survival.

There are four large constructed off-channel sites (Or Creek, Grant's Tomb, Overland Channel, and Archery Pond) located between Coquitlam Dam and RST2, totally about 27,000 m^2 of habitat (Figure 5.1). 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 2017, 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 producing precise mainstem population estimates. 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.

All off-channel weirs and the mainstem RSTs operated continuously from early-April until mid-June with the exception of RST2.2, used for Chum and Pink enumeration, which started operation early-March (Table 5.1). One or more RST's 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. All juvenile fish captured at the weirs and RSTs were identified to species, counted measured for forklength (nearest mm). Unmarked smolts were given a unique fin clip identifying capture period and location (see Section 5.1.1.3). 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 forklength were smolts. Steelhead smolts in 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 two year olds 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 forklength 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 will remain in the river for an additional year. This will result in some degree of positive bias in estimates of annual smolt yield. During 2005, 2007 – 2017, 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 is 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). 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 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 upstream) 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.

A unique mark type consisted of a small clip at one of several fin locations. The duration of the marking period was determined with the objective of achieving a minimum recapture target of 40 Coho smolts from each group at each RST (10 recaptures for Steelhead smolts). We monitored daily catch totals to meet this target and relied on observations of migration patterns in previous years to plan strata duration.

While almost all unmarked Steelhead smolts originated from the mainstem, a large proportion of marked Coho smolts originated from off-channel sites. This is of concern because previous work in the Coquitlam River has shown significant differences in capture efficiency 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. To address this, in addition to the mark given to identity capture period, smolts were given a second unique mark identifying their original capture location (see Table 5.1 and paragraph below). By separately analyzing marking and recovery data for these different mark groups, we were able to generate several independent estimates of the number of smolts passing the same RST. For example, independent estimates of Steelhead smolt abundance at RST2 could be generated using four different mark groups (off-channel, RST2, RST3 and RST4). Stratification of marking by location was achieved by assigning one unique fin-clip mark for all of the off-channel weirs, and additional unique marks for each of the three RST trapping locations (Table 5.1).

Since the precision of a mark-recapture estimate improves with the number of smolts marked, it is advantageous to generate estimates based on pooled 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:

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

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 when using both mainstem and off-channel mark groups were 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 CE of 100% for each weir, and used the total number of smolts captured to estimate smolt production.

For mainstem reaches of the Coquitlam River, the number of smolts passing each RST was estimated using a maximum likelihood (ML) model developed by Darroch (1961) and modified by Plante (1990) for stratified mark-recapture data. 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. All smolt 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 (Table 5.1), although in some cases, it was necessary to pool strata to avoid small sample and numeric problems that may prevent the maximum likelihood iterations from converging. When pooling strata, we followed the recommendations of Arnason *et al.* (1996). If numbers of marked and recaptured smolts in the majority of strata were too low to use the stratified estimator, data from all marking and recovery periods were pooled and the standard pooled Petersen estimator for unstratified data were used (see Arnason *et al.* 1996 and for a discussion of the problems associated with pooling sparse data).

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 that at RST 2 and 4, where smolts were marked and then recapture was the same trapping locations, it was only the initial capture event that was used to estimate the number of mainstem fish passing that trapping location. The numbers of recaptures were used only for estimating capture efficiency for each RST location.

Key to estimating the abundance of only smolts originating from the Coquitlam River mainstem was that all off-channel smolts were marked, thus allowing them to be distinguished from mainstem smolts by either having a mainstem mark or no mark.

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:

$$\pm 95\%\ CI\left(N_{reach\ 2}\right) = \pm 1.96\sqrt{Var\left(N_{RST\,2}\right) + Var\left(N_{RST\,3}\right)} \tag{5.8}$$

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:

$$\pm 95\%\ CI\left(N_{reach\ 4}\right) = \pm 1.96\sqrt{Var\left(N_{RST\,4}\right)} \tag{5.9}$$

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

$$\pm 95\%\ CI\left(N_{mainstem}\right) = \pm 1.96\sqrt{Var\left(N_{RST\,2}\right)} \tag{5.10}$$

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

5. Smolt and Fry Production

$$N_{total} = N_{RST2} + N_{Off\text{-}channel} \quad (5.11)$$

with a 95% confidence interval of:

$$\pm 95\%\ CI(N_{total}) = \pm 1.96\sqrt{Var(N_{RST2})} \quad (5.12)$$

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 CE 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 CE between marked and unmarked smolts are described in section 5.1.1.3. With respect to the assumptions of constant CE 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 CE and the proportion of marked to unmarked smolts were assumed to be constant to less than 10 days for most strata (Table 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 and fish mortality substantially. The RST targeting Chum were operated in reach 2 at the same location as the two RSTs used to trap Coho and Steelhead smolts (RST2 site; Figure 5.1), and differed from these larger traps mainly in that its drum was constructed of smaller screening (2.5 mm versus 12 mm).

5.1.2.2 Differential marking over time

To generate temporally stratified mark-recapture estimates, single day catches of fry were periodically marked and released at RST3, approximately 3.2 km upstream of the trapping site at RST2. This differed from the approach taken for Coho and Steelhead smolts in that marking was not continuous. We distributed marking events at least five days apart to allow for all marked fry from one group to pass RST2 before the next group was released, and also because the mark used remained detectable for only about five days. This provided temporally stratified data without the need for different marks.

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 was 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) and 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).

5.2 Results

5.2.1 Off-channel sites

In 2017, daily catches of Coho smolts at the off-channel weirs at the beginning and end of the trapping period were very low compared to catches during the peak of the migration (Figure 5.3). Therefore, we assumed that population closure was largely met for Coho, and that captures at the weirs accurately represented total smolt output. For Steelhead, daily captures were too low and sporadic to assess whether off-channel weirs operation spanned the entire outmigration period (Figure 5.4). Observed mortality was < 1% for all target species at the off-channel weirs with the exception of Coho at Overland Channel with 5.6% mortality. Only a single breach occurred that the off-channel sites in 2016. A tree fell on the Archery weir trap box April 25 that would have allowed fish to escape capture for a one day period. However, this would have no meaningful impact on off-channel estimates since captures during the five days before and after this period was very low for Steelhead and Coho smolts (< 1 smolt / day, Figures 5.3-4).

In 2017, an aggregate total of 4,334 Coho and 83 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 5.2). Mean weighted density of Coho smolts in the off-channel sites was 21 smolts/100 m^2 while Steelhead smolt density was 0.4 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). Density of Steelhead has been variable at the off-channel sites, with no clear trends across years (Figure 5.6). The proportion of production for the entire Coquitlam River upstream of RST2 originating from constructed off-channel sites was 27%-68% for Coho and 2%-9% for Steelhead.

5.2.2 Coquitlam River mainstem

During 2017, discharge in the Coquitlam River during the spring trapping period was variable, with daily mean flows exceeding 20 cms on four occasions during Chum and Pink fry migration and three occasions during Steelhead and Coho migration (Figure 5.2). Traps remained in near continuous operation during the entire Chum, Coho and Steelhead outmigration. Chum fry trapping was suspended March 28-29 during the largest storm event of the season with mean daily flows reaching 35 cms. Chum captures were still low at this early stage of outmigration that had a minimal impact on the overall population estimate. As well, all traps were decommissioned during May 5 to reduce mortality of the 5,000 Sockeye released immediately below the dam. A cool and wet spring combined with high winter snow pack led to higher monthly discharge during the March – June outmigration period compared to other Treatment 1 or 2 years (Table 5.4). Overall, observed mortality at the RSTs was 0.5% for Coho and 0.2% for Steelhead smolts, 5.8% for Chum fry, and 6.0% for Chinook fry and smolts. For Steelhead smolts, Coho and Chum, daily catches 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, and 5.7), suggesting that population closure was largely met. There was no monitoring for early season downstream movement of Coho in 2018.

Appendix 5.1 provides a summary of mark recapture statistics (all release and recovery strata pooled) for each species and mark group, and estimates of the number of smolts passing each RST (not to be confused with estimates of smolt yield from each reach). A summary of which marking and recovery strata were pooled (if any) in order to generate population estimates is provided in Appendix 5.2. 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.

5.2.2.1 Coho

At RST4, CE was not significantly different using off-channel smolts and mainstem smolts that were captured, marked and released upstream (50% and 54%, respectively, Fisher's exact test, P=0.25, Table 5.3, Figure 5.8). The combined mark group produced a similar estimate as the mainstem mark group (difference < 1%) and slightly higher precision (95% CI: ± 9% both, ± 12% mainstem), therefore we used both mark groups to generate a population estimate of 1,753 Coho smolts (95% CI: ± 161 smolts) for reach 4 (Table 5.2). Note that throughout this report, precision will always represent the 95% confidence intervals.

At RST3, CE were the same for marked off-channel and mainstem Coho smolts (25%, Fisher's exact test, P<0.68; Table 5.3, Figure 5.8). Therefore, we used the combined mark groups to generate a population estimate of 3,940 smolts (±673 smolts, Table 5.2) in reach 3, after smolt numbers from reach 4 were subtracted.

At RST2, CE were significantly different for the mainstem mark and off-channel mark groups (25% and 32%, respectively, P <0.01; Table 5.3, Figure 5.8). The combined mark group produced an estimate 15% lower than using the mainstem mark group. This difference (1,495 smolts) was considerably larger than the standard error (SE) of the mainstem estimate (551smolts), a condition for rejecting the use of combined mark groups (Arnason *et al*., 1996). Considering this, we used only the mainstem mark group for the reach 2 population estimate of 4,148 ± 1,257 Coho smolts (Table 5.2), which incorporated the downward adjustment for the presence of smolts from reaches 3 and 4, and the four off-channel sites.

Based on the mainstem mark group, the estimated number of Coho smolts outmigrating from the mainstem of the Coquitlam River upstream of RST2 in 2017 was 9,810 ± 1,080 (14,158 ± 1,080 smolts including those from the four off-channel sites, Table 5.2). Average Coho smolt density in the Coquitlam River was 6.5 smolts/100 m^2 (8.0 smolts/100 m^2 including the off-channel sites, Table 5.2). Areal Coho density was two-fold higher in reach 3 (20.5 smolts/100-m^2, Table 5.2), than in reach 2 (8.9 smolts /100 m^2) and over five-fold higher than in reach 4 (3.2 smolts /100 m^2). The precision of estimates for individual reaches ranged from ± 9% for the estimate for reach 4, to ± 30% for the smolt estimate for reach 2 (Table 5.2).

Abundance across all years and reaches for Coho are reported in Table 5.6. Analysis of abundance trends across years or by flow treatment period were not completed for this report. This will be included in the 2018 report, which provides a preliminary summary of the initial Treatment 1 and 2 monitoring periods.

5.2.2.2 Steelhead

At RST4, CE was significantly different between marked off-channel Steelhead smolts (from Grant's Tomb) and mainstem smolts that were captured at RST4, marked and released upstream (17 % and 40%, respectively, Fisher's exact test, P=0.02; Table 5.3, Figure 5.8). Therefore we used only the mainstem mark group to generate a population estimate of 2,807 Steelhead smolts (±1,051) for reach 4 (Table 5.2). The relatively low precision in 2017 was largely the product of lower CE of the mainstem mark group during the 2nd and 3rd recapture periods, which coincided with peak outmigration. This had the effect of the Darroch ML model having high uncertainty during peak outmigration.

At RST3, CE was not significantly different between off-channel and mainstem mark groups (14% and 16%, respectively, P = 0.73). Therefore, we used both mark groups to estimate that 3,802 mainstem smolts (±1,233) passed RST3 (Appendix 5.1). This resulted in an estimate of 995 smolts (±1,620 smolts, Table 5.2) originated in reach 3, after smolt numbers from reach 4 were subtracted. Though the precision was low for smolts originating in reach 3, this has been common in past years since precision from reach 3 and 4 estimates are combined for this calculation (see equation 5.8).

At RST2, CE was significantly different for the two groups (9% off-channel and 19% mainstem, P = 0.02; Table 5.3). Therefore we used only the mainstem mark group. The resultant estimate of smolts originating from reach 2 was 1,433 (± 1,480 smolts, Table 5.2).

Based on the mainstem mark group, the estimated number of Steelhead smolts outmigrating from the Coquitlam River mainstem upstream of RST2 was 5,142 ± 939 (5,225± 939 smolts when off-channel sites were included, Table 5.2). Average Steelhead density in the Coquitlam River mainstem was 3.4 smolts/100 m^2 (3.1 smolts/100 m^2 in the Coquitlam River including the off-channel sites, Table 5.2). Areal Steelhead smolt density was over five-fold higher in reach 4 (14.6 smolts/100m^2, Table 5.2) than in reaches 3 and 2 (2.5 and 1.7 smolts/100m^2, respectively). The precision of the abundance estimates ranged from ± 18% for the estimate smolt abundance the Coquitlam River including off-channel areas, to ± 130% for smolt abundance in reach 3 (Table 5.2).

We assumed all Steelhead 120-230 mm in forklength 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 5.6). This was corroborated by scale samples collected for Steelhead in this size range in 2005-2017 (Appendix 5.4). Scale analysis of 567 individuals indicated a broad overlap (132-188 mm) in the absolute ranges in forklength for age-2 and age-3 smolts, but most smolts greater than 160 mm in length were age-3 (Appendix 5.4). 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 forklength and 30%-60% of smolts 195-220mm. To estimate Steelhead adult-to-smolt survival for the 2005-2013 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 2007-2016 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.

We assumed that age-1+ Steelhead (forklength < 120mm) will outmigrate after one or two additional winters in the Coquitlam River, yet this assumption has not been substantiated. The proportion of captures of Steelhead < 120mm was largest and most variable at RST2 based on captures from the last five years (15% - 42% of total captures, Table 5.5). We currently lack any evidence to resolve whether these captures represent localized fish movement – where a similar amount of upstream movement occurs – or a net downstream movement. If it is the latter, these captures would represent freshwater production not included in the estimates to date.

Abundance across all years and reaches for Coho are reported in Table 5.6. Analysis of abundance trends across years or by flow treatment period were not completed for this report. This will be included in the 2018 report, which provides a preliminary summary of the initial Treatment 1 and 2 monitoring periods.

5.2.2.3 Chum and Pink

Only Chum salmon fry were present in the Coquitlam River during spring 2017. Chum were trapped continuously from March 6 to June 12 at the RST2 location in reach 2 with the exception of one high water event March 28 and May 5 during the Sockeye smolt release. Chum were batch-marked on 10 separate occasions with the first starting April 2 (Appendix 5.3). The capture efficiency varied from 0%-11.0% (Appendix 5.3), and averaged 6% (all strata pooled Appendix 5.1). The 0% capture efficiency, the 5th release stratum, did not reflect capture efficiency during this period. The lack of recaptures was due to a near total mortality event the day following the release of the 5th mark group (April 24) due to high flows, a closed refuge box gate and heavy debris load that rendered marked fry indistinguishable from unmarked. The total count April 24 would have included some marked fry but this would have been a relatively small number given the average capture efficiency (6%) during 2017. To compensate for the missing release group, the capture efficiency for the 6th strata was used for both strata 5 and 6 since it was the adjacent stratum with the most similar discharge.

During 2017, an estimated 12.7 million Chum fry (± 2.2 million, Table 5.2) migrated past the RST2 trapping site. This equates to a Chum density for the mainstem of the Coquitlam River of 1.7 million fry/km or 7,450 fry/100 m^2 (Table 5.2).

5.2.2.4 Sockeye/Kokanee

In 2017, catches of Sockeye/Kokanee smolts ranged at RST 2-4 were 12, 18 and 64; respectively (Table 5.7). Mortality rates were 11%-59% with no distinction between those that died before and after capture. In addition to these, 5,000 hatchery raised Sockeye were released May 4th near the low level outlet at the base of the Coquitlam dam and another 260 were released in the Coquitlam Reservoir (Alexis Hall, pers. Com.). Captures of these at RST 2-4 were 32, 21 and 21; respectively with mortality rates of 0%-5%. Sockeye/Kokanee captured each year at all traps during Treatments 1 and 2 have ranged from 10's of fish to several hundred (2005-2007). Given the limited number of fish captured, no attempt was made to mark fish or generate population estimates.

5.2.2.5 Chinook

191 Chinook juveniles were captured by the two smolts traps at the RST2 location and another 1,507 were captured in the fry trap at RST2 in 2017. As in past years, there was no attempt to distinguish between the age-classes and they were not included in the mark-recapture program necessary to estimate the number of outmigrants.

5.3 Discussion

Tables 6.1a and 6.1b in the next section provides 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, yet, a proportion (probably small) of smaller Steelhead in this size range were likely parr that were dispersing to downstream habitats, ultimately 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 (they released milt or eggs when light 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 forklength of Steelhead smolts during 1996-2016 varied from 154 mm to 171 mm, which is in good agreement with 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 exit the study reaches as parr they are not included in productivity estimates leading these to be biased low. Using the same marking approach as for smolts (distinct mark for capture locations) would provide information about whether parr moved out of the study area but not about the proportion of parr moving below the study area. This is because the results from a mark-recapture approach - such as 20% of parr marked in the study area were recaptured at RST2 – are a product of the capture efficiency of RST2 and the proportion of parr that moved past this point. Estimating the proportion of parr moving out of the study area would depend on assumed trap capture efficiency though there is the potential to estimate this using additional methods in future years.

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 al 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 on power at smolt observation errors 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 2017 Coho smolt abundance estimate in the Coquitlam River mainstem was moderate (95% confidence interval: ±11%) compared with estimates during 2000 - 2016 (95% confidence interval: ±6% – 14%) and was much better than the theoretical optimal value of ±20% (in their paper as $\sigma_{sm,o} \approx 0.1$). Precision of the 2017 mainstem Steelhead smolt estimate was moderate (95% confidence interval: ±18%) compared with estimates since 2000 (95% confidence interval: ±11% – 37%) and similar to the theoretical optimum. For both species, the satisfactory precision was the product of intensive marking and recapture efforts of mainstem and off-channel smolts. Significant numbers of smolts were marked at RST 3-4 for Coho, and RST 2-4 for Steelhead, and thus susceptible for recapture at RST2, the site responsible for the mainstem river estimate. As well, using two rotary screw traps for smolt trapping at the most downstream site (RST2) increases capture efficiency, and since precision generally increases with capture efficiency, resulted in relatively high precision for the mainstem smolt estimates. Marking off-channel smolts improves the precision of Coho estimates but are too low to improve the precision of Steelhead estimates.

The precision of fry population estimates for Chum salmon at the RST2 in 2017 was low compared with previous years using rotary screw traps (95% confidence interval: ±17% and ±6%-18% during 2008-2016) but high compared to years using incline plane traps (±19 – 25%). This was the product of the relatively moderate capture efficiency throughout (6.0%). Precision improves with capture efficiency, particularly during peak outmigration, as was the case in 2016 but not in 2017. The need to apply the capture efficiency from the strata 6 to strata 5 adds uncertainty to the 2017 estimate and likely biased the estimates low given that capture efficiency was likely lower during strata 5, which had average daily flows reach 21 m^3/sec whereas they never exceeded 12 m^3/sec strata 6.

5. Smolt and Fry Production

6.0 Fish Productivity during Treatment 1 and Treatment 2

6.1 Coho

During 2000-2017 Coho smolt yields for the 7.5 km long section of the 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 6.1 Table 6.1a). Annual Coho smolt yield for mainstem and constructed off-channel habitat combined, were, on average, double that for the mainstem alone, with less variation from year to year (8,400-24,500; Table 6.1a). To compare changes between Treatment 1 and 2, we only compare metrics for cohorts that reared entirely under only Treatment 1 or 2. For smolt abundance, this excludes estimates from the years 2000 and 2009. Mean abundance from mainstem habitats for Treatment 1and 2 were not statistically different for the mainstem and off-channel smolts combined (2-tailed t-test, $p = 0.96$, Table 6.2). Changes in smolt abundance for mainstem habitats is a more sensitive measure of the effect of flow treatments than when combined constructed off-channel habitats, which are buffered from mainstem flows either by groundwater effects or have independently controlled water intakes. Mean abundance of mainstem smolts alone increased significantly from 5,479 in Treatment 1 to 7,840 during Treatment 2 (2-tailed t-test, $p = 0.048$; Table 6.2) even though there was considerable overlap in the 95% confidence intervals of mean abundance during Treatment 1 and 2 (Figure 6.3). This suggests that juvenile Coho production has increased under the Treatment 2 flow regime. This change was the product of higher yields in Reaches 2 and 3 (Figure 6.3). In spite of a similar increase in the mean yield in both of these reaches, the change was only significant for Reach 3 (2-tailed t-test, $p = 0.02$).

A useful way to interpret the amount of change between treatments is by the effect size - the percent change in yield from Treatment 1 to Treatment 2. With this approach, it is the confidence limits are the key values to focus on rather than the mean change. Figure 6.5 provides a framework for interpreting effect size results based on the approach used by Bradford *et al.* (2005). For mainstem Coho, the increase in smolt yield between Treatment 1 and 2 could be anywhere from just above 0% to 95%, thus representing a significant increase. Though this doesn't fit clearly into one of the categories from Bradford *et al.* (2005), this is arguably a partial success based on the lower confidence limit extending very near to 0%, representing increases too small for considering that significant fisheries benefits were achieved, but also larger values that clearly do. Reach level results were ambiguous for Reaches 2 and 4, including both negative and positive change. The confidence intervals for Reach 3 were well above zero indicating the high probability of a significant increase, which is consistent with t-test results above.

Attributing the likely change in mainstem smolt abundance to Treatment 2 flow regime requires comparison with one or more systems unaffected by flow treatments but are similar enough to reflect abundance trends in the Coquitlam if not change in flow treatment had occurred. The nearby Alouette River shares many characteristics with the Coquitlam River: hydrology, fish assemblage, seasonal climate trends. Importantly, it has not experienced substantial flow manipulations during this time period. Annual trends in smolt abundance have been highly correlated during 2000-2008 (R=0.86) and 2009-2014 (R=0.74, Figure 6.8). Monitoring overlapped with all of Treatment 1 but only up to 2014 during Treatment 2. For cohorts that reared exclusively under either Treatment 1 (2002-2008) or Treatment 2 (2010-2014) conditions, and during years with monitoring in both rivers, Coho smolt yield increased approximately 117% (95% CI: 25%-250%) in the Alouette, which was considerably more than the Coquitlam (Figure 6.6). This reflects that regionally freshwater productivity increased during Treatment 2 years – as reflected by change in yield for the Alouette during Treatment 2. If the Alouette River is a reliable control stream, this suggests Treatment 2 had no or even a negative effect on productivity in the Coquitlam River and that the change in smolt yield was the product of regional increases in productivity. This would be a complete change to the interpretation of the effect of flow treatments on Coho production. However, we are not yet confident in the use of the Alouette data for this purpose. First, we are uncertain whether yields before and after 2008 are comparable. Cope (2014) speculated that smolt yield estimates for the Alouette River were biased low for some or all years prior to 2008, when the trapping site was repositioned further upstream to avoid tidal-driven backwatering of the trap. Resolving there was bias, and if needed correcting for it, would require re-evaluating the Alouette outmigration estimates with a model that incorporates back-watering effects. Completing this prior to the WUP order review would address a key uncertainty necessary to understand the fisheries benefits of the flow treatments. Another uncertainty is whether abundance trends for the Alouette during 2010-2014 reflect trends during the remainder of Treatment 2. This could be addressed either by additional Alouette River monitoring or including additional control rivers that overlap with Treatment 1 and 2.

For the purpose of comparing the average Coho productivity across all years in the Coquitlam River to that in other streams, an empirical smolt production model developed by Bradford *et al.* (2006) provides a relevant benchmark. For Pacific Northwest streams of similar latitude to the Coquitlam River (48-50° N), the model would predict an average yield of 1,664 smolts/km. By comparison, mean Coho smolt yield from the Coquitlam River, including off-channel habitat (which is appropriate given the dataset used by Bradford *et al.* (2006), was 1,789 smolts/km (range: 1,118-3,261 smolts/km, 2000-2017). This suggests that Coho smolt productivity in the Coquitlam River study area is comparable to the average for streams at this latitude.

In reach 4, where annual downstream trapping has occurred over a longer time period (1997-present), there was little evidence of a systematic trend across all survey years. While smolt yields generally decreased 1997-2007, 2008 onward has been a period of widely ranging abundance with no systematic trends (Figure 6.7). Late summer snorkeling surveys suggest that densities of Coho fry in reaches 2 and 3 were several-fold higher during 2006-2017 compared to density estimates obtained during an electrofishing survey in 1997 (Riley *et al.* 1997; see Section 4.3.2) prior the implementation of Treatment 1 when dam releases were considerably lower (see Section 1).

The constructed off-channel habitats included in the study[2], which represent about 10% of available habitat in the Coquitlam River study area, supported from 29% to 77% of the overwintering Coho smolt population during 2000-2017. Mean off-channel smolt density decreased from 32 smolts/100m^2 during Treatment 1 to 21 smolts/100m^2 during Treatment 2 (2-tailed t-test, p = 0.04). Using density for the comparison takes into account the four years that Grant's Tomb was decommissioned. The 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 in off-channel sites (19.9 to 44.9 smolts/100m^2). While constructed off-channel habitat may represent relatively productive Coho habitat in the Coquitlam River, smolt densities in Coquitlam River off-channel sites were below 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).

The Beverton-Holt escapement-to-smolt stock-recruitment relationship was indicative of a highly productive system that reaches carrying capacity at relatively low escapement levels and that beyond this threshold is relatively insensitive to additional spawners (Figure 6.10). The 2009 smolt yield stands as an outlier in this stock-recruitment relationship. Smolt yield during this year was over 50% higher than smolt yields produced from comparable spawner returns during Treatment 1and 2. However, Coho smolt yield in the nearby Alouette River was also two-fold higher in 2009 compared to other years (Cope 2011), suggesting that 2009 represents a recruitment anomaly caused by some factor acting at a larger regional scale. Estimated Coho spawner densities during 2002-2016 ranged from 70 to 1,038 fish/km (Table 2.4), or 34 to 519 females/km, assuming a 1:1 sex ratio. These values exceeded, by 1.8- to 25-fold, a theoretical minimum threshold of 19 females/km necessary to achieve maximum Coho smolt yield in an average coastal stream, as suggested by a meta-analysis of empirical data (Bradford and Myers 2000). The average proportion of adults counted in mainstem habitats increased substantially from Treatment 1 (51%) to Treatment 2 (87%, Figure 2.5). This increase coincided with higher escapements 2009 onward, which may be a product of the change in survey methods as well as an actual increase in escapement.

The abundance of Coho fry from the snorkeler based standing stock assessment has varied from a low of 18,000 in 2007 to 91,000 in 2011 (Table 6.1). Mean abundance during Treatment 1 and 2 was 30,745 and 57,296 fry, respectively. Though not statistically significant (2-tailed t-test: p=0.07), this represents a nearly two-fold increase in abundance. The high variability in abundance within each treatment period combined with only three monitoring years during Treatment 1 were responsible for the lack of a significant difference between Treatments 1 and 2. It is also important to note that adult escapement for these Treatment 1 cohorts were substantially lower than all but one of the cohorts reared during Treatment 2 (Table 2.4). While we are concerned about the comparability of Coho escapement estimates between treatment periods (see section 2.3), the very low escapement for the three Treatment 1 cohorts raise the possibility that the lower fry abundance during Treatment 1 was the consequence of low escapement rather than

[2] There are seven major off-channel habitat sites in the 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.

reduced productivity. The strong linear relationship between escapement and fall fry yield reported in prior reports did not continue when incorporating results from recent years. Linear and Beverton-Holt stock-recruitment models had poor to moderate fit ($R^2 < 0.01$ and $R^2 = 0.43$, respectively, Figure 6.10).

Mean size of Coho smolts in Reach 4 was slightly greater during 2000-2017 (mean: 96 mm) compared to the period preceding Treatment 1 (1996-1998; mean: 89 mm; t-test, p=0.02; Figure 6.8). No size data exist for reaches 2 and 3 prior to 1999. During 2000-2017, Coho smolts were consistently larger in reach 4 (mean: 94mm) than in reaches 2 and 3 (mean: 89mm; t-test, $p<0.01$), and larger in mainstem versus off-channel habitat (Figure 6.4). Mean lengths were not significantly different between Treatment 1 and 2 for reach 4 or reaches 2 and 3 combined (p=0.29 and p=0.65. respectively).

Annual Coho smolt yield in the Coquitlam River during Treatment 1, Treatment 2 and both treatments combined was positively correlated with that in the Alouette River (R = 0.86, 0.71 and 0.51, respectively; Figure 6.8, Cope 2011). Rearing conditions varied less within each treatment period than between them, indicating similar abundance trends within each treatment period but also of a shift in the trend between treatments. As discussed above, this could be the result of changes in productivity or of biased low estimates for the Alouette prior to 2009.

Coho egg-to-smolt survival remained consistently low (0.03-0.55%) for the 2002-2017 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 would be cautious about any between-treatment evaluations reliant on adult escapement estimates, which are likely not comparable between treatments due to the change in survey methods during Treatment 2. By comparison, the average egg-to-smolt survival rate for Coho populations in nine other Pacific coastal streams were considerably higher (1.5%, ±1 SD of 0.7%-3.0%; Bradford 1995). It should be noted, however, that high uncertainty in the estimates of Coho escapement to the Coquitlam River directly affects egg-to-smolt survival estimates and atypically low egg-to-smolt survival estimates for Coquitlam River Coho may be an artefact of biased-low estimates of observer efficiency or survey life for adults. So egg-to-smolt survival may be useful for evaluating within-river changes but not between rivers. As well, Coho escapements to the Coquitlam River include substantial numbers of first generation hatchery fish spawning in natural habitat. These fish presumably have reduced reproductive fitness compared to wild fish (Fleming and Gross 1993).

6.2 Steelhead

During 2000-2017 the estimated number of Steelhead smolts outmigrating from the 7.5 km long section of the Coquitlam River mainstem upstream of the RST2 trapping site ranged from 2,200 to 5,500. Smolt yield from off-channel habitats represents only a small proportion of total yield (1%-9% of total annual yield). Considering this, smolt yields for the mainstem only are reported here. Mean smolt yields for cohorts that reared exclusively under Treatment 1 or Treatment 2 were not statistically different (3,716 smolts and 4,684 smolts; respectively; 2-tailed t-test p = 0.07, Table 6.2). This reflects a 24% increase in mean abundance during Treatment 2.

Though not yet statistically significant, we expect it will with additional monitoring if recent abundance levels continue. This included smolt estimates from the years 2002-2008 for Treatment 1 and 2012-2017 for Treatment 2. Smolt yield for individual reaches has been more variable than the mainstem 2000-2017 and show very little similarity in trends across years (Figure 6.2). 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=0.02; Table 6.2, Figure 6.3) whereas the yields from reaches 2 and 3 were similar during both Treatments periods. Smolt yield from reach 4 has increased substantially since 1996 (R^2=0.66; 1996-2017; Figure 6.7). This was likely a product of the higher Treatment 2 base flows combined with the relatively narrow channel width in reach 4. This would be consistent with the prediction that a higher energy flow environment is more favorable to juvenile Steelhead. In terms of effect size, smolt yield in the mainstem has increased anywhere from near 0% to over 50% (Figure 6.5). Though the range of the increase was largely greater than 0%, it remains uncertain whether this represents a meaningful fisheries benefit. A clearly substantial increase in smolt yield occurred in reach 4 between treatment periods (90% and 200%). The mean effect size for reaches 3 and 4 were close to 0%.

As with Coho, we compared the relative increase in smolt yield in the Coquitlam and Alouette Rivers during commonly monitored years during Treatment 1 (2002-2008) and Treatment 2 (2012-2014) to evaluate the effect of Treatment 2 compared with any changes yield that would have occurred without a change in flow treatments. The mean percent increase in smolt yield in the Alouette was 71% (95% CI: 10%-132%) compared to the Coquitlam at 24% (95% CI: -5%-53%; Figure 6.6). If results for the Alouette are considered unbiased across Treatment 1 and 2, in spite of the change of trap location and duration in 2008, then this would suggest smolt yield in the Coquitlam was lower than what would have occurred if without the change to Treatment 2 flows. In other words, Treatment to either had no effect or a negative effect on productivity. This conclusion is highly speculative at this point. In addition to the potential bias, there was only moderate support that smolt yield across years for the two rivers were effected by common regional factors (R = 0.37; Figure 6.6). A high correlation would suggest that variation in annual Steelhead smolt production in the two streams was largely the product of similar regional conditions rather than conditions specific to each watershed.

During 2000-2017, Steelhead smolt densities for the mainstem study area upstream of RST2 as a whole averaged 2.7 smolts/100m^2 (range = 1.7-3.7 smolts/100m^2), which exceeded the provincial Steelhead biostandard of 2.0 smolts/100m^2 (Tautz *et al.* 1992). With the exception of 2000, areal smolt densities were highest in Reach 4, particularly in 2009, 2010 and 2017, but this was partly due to greater wetted width in downstream reaches. Differences in linear densities among reaches have become more pronounced since 2009, with markedly higher density in reach 4. In many cases, Steelhead population estimates for individual reaches were highly uncertain due to low numbers of marked and recovered fish, or, in the case of downstream reaches, compounding error (see Section 5.1.1.4).

The mean forklength of spring migrant Steelhead parr (age-1+) was nearly identical during Treatment 1 and 2 for captures in reach 4 (99-101 mm) and reaches 2 and 3 combined (89-91 mm) with relatively little variation across years (Figure 6.8).. 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; Figure 6.8). On average, forklength of Steelhead smolts reared entire under Treatment 1 flows were 5-6 mm smaller than those reared entirely during Treatment 2 for reach 4 and reaches 2 and 3 combined, though the difference was close to- but not statistically significant (2-tailed t-test, $p= 0.08$ and 0.06, respectively; Figure 6.11). Given that outmigrating smolts include at least two age classes, mean size can increase as a result of an increase in the size of an age-class and/or by an increase in the relative abundance of the older age classes. Almost all smolts are either age-2+ or 3+ smolts based ages derived from scale samples (Appendix 5.4). Though not statistically significant, the mean proportion of all captures that were age-2+ smolts increased from 57% during Treatment 1 to 64% during Treatment 2. The age at smoltification is determined by freshwater growth conditions (Randall *et al.*1987). The combination of increased smolt size and an increase of proportion of age-2+ smolts could suggest improved rearing conditions during Treatment 2.

Snorkeling surveys indicated that during 2006-2017, Steelhead fry density in the Coquitlam River in late summer averaged 12.7 fish/100m^2, while parr density averaged 5.5 fish /100m^2. In general, these values are low for coastal Steelhead streams (see Section 4.3.2). Compared to estimates of Steelhead fry and parr abundance in the Coquitlam River in 1997 prior to the start of Treatment 1 that were derived from electrofishing surveys (Riley *et al.* 1997), estimates for 2006-2017 derived from both electrofishing and snorkeling surveys suggest several-fold higher densities of fry and parr (see Section 4.3.2). Based on snorkeling surveys alone, average fall abundance of fall fry was 70,963 for Treatment 1 and 41,555 for Treatment 2 thought this difference was not significant (t-test $p = 0.15$). Age 1+ parr abundance was similar between Treatments 1 and 2 (age 1+ parr 8,812 and 8,715; respectively; t-test $p = 0.96$) yet age 2 + parr were only half as abundant during Treatment 1 than during Treatment 2 (age 2+ parr 1,691 and 3,072; respectively; t-test $p < 0.01$). However, the abundance of age 2+ parr is a factor of the survival-to-age and to the proportion that smolt prior to becoming an age 2+ parr (e.g. smolting after their 2nd versus 3rd winter). The higher age-2+ parr abundance did not translate into significantly higher smolt yield for smolts overwintering exclusively under Treatment 2 conditions (2010-2017) compared with Treatment 1 (4,654 smolts, 3,848 smolts; respectively; t-test $p = 0.081$, Table 6.2). At this point, we would not reject the null hypothesis that smolt production is unaffected by changes between Treatment 1 and 2 flows. We expect this will change with additional monitoring if current population trends continue.

During 2005-2017, Steelhead spawner densities in the Coquitlam River ranged from 24 to 80 fish/km (mean: 39 fish/km). Comparisons of Steelhead spawner densities in the Coquitlam River, relative to those in other streams was limited by a lack of reliable data (for other streams), and by the limited time series for the Coquitlam River. AUC-based estimates of Steelhead escapement to the Cheakamus River, a nearby stream that is also regulated, ranged from 6-100 fish/km during 2002-2017 (mean: 35 fish/km), but were not correlated with Coquitlam River escapements ($R = 0.07$; Figure 6.9).

Estimated Steelhead egg deposition in the Coquitlam River during 2005-2017 ranged from 39,000-149,000 eggs/km (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). The

Coquitlam River is likely to be a more productive Steelhead stream than the Keogh River, considering that mean smolt age is less (≈60% of smolts in the Coquitlam River are age-2 compared to an average of 33% in the Keogh River; Ward and Slaney 1993), and therefore a somewhat higher egg deposition per unit area is likely required. Nevertheless, one would not expect egg deposition to be an important limiting factor to smolt yield in the Coquitlam River during 2005-2017, taking into account that egg deposition per unit area exceeded the threshold value for the Keogh River by three- to 11-fold. The stock-recruitment data supports this assertion: a Steelhead escapement of only 260 adults in 2007 produced almost as many age-1+ parr (11,904; Figure 6.12) as did the estimated 896 adults that returned in 2006 (12,926), indicating strong density-dependent survival factors during the first year of life. Additional years of monitoring at very low escapements would be required to define the range in which adult recruitment strongly affects Steelhead smolt production in the Coquitlam River.

Steelhead egg-to-fry survivals ranged from 2.9% to 8.9% (mean: 6.3%, Table 6.1b) during 2006-2017. These values were comparable to the 1976-1985 average of 6.5% for Keogh River Steelhead (range = 1.8%-11.5%; Ward and Slaney 1993). Egg-to-age-1+ parr survival for Coquitlam River Steelhead ranged from 0.6%-2.1% (mean: 1.3%) which was somewhat higher than the average of two years' data for the Keogh River (0.65%, derived from Ward and Slaney 1993). Steelhead egg-to-smolt survival for Coquitlam River Steelhead ranged from 0.4%-1.1% for 2005-2014 brood-years (Table 6.1b; derived from age-2 and age-3 smolt yields in subsequent years). Ward and Slaney (1993) reported a similar range (0.4%-1.3%) for Steelhead egg-to-smolt survival in the Keogh River. Fry-to-age-1+ parr survival for the 2006-2016 fry Cohorts ranged from 10%-37% (Table 6.1b). Age-1+ parr-to-smolt survival averaged 52% (ranged 33%-92%) for the 2006-2015 age-1+ Cohorts (Table 6.1b; derived from age-2 and age-3 smolt yields in subsequent years). This is comparable to parr-to-smolt survival for Steelhead in the Keogh River (48.8%; Tautz *et al.* 1992), and for Atlantic salmon populations in several eastern Canadian streams (≈40%; Symons 1979). In two of five cases, survival estimates exceeded 100% for the age-2+ parr to age-3 smolt life stage (range: 59%-148%; Table 6.1b), indicating positive bias. The most likely source of this bias is either underestimation of age-2+ parr abundance in 2007 and 2008, or underestimation of the mean fork-length criteria used to delineate age-2 and age-3 smolts, which leads to overestimation of the proportion of age-3 smolts (see Section 5.2.2.2, last paragraph; Appendix 5.4). Over 500 scale samples have been collected to date. This is sufficient for defining multi-year averages of size-at-age and relative proportions of age-2 and age-3 but is not sufficient for year-specific size-at-age relationships.

6.3 Chum

Similar to that for Coho and Pink, escapement and egg-to-smolt 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 (Table 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% (mean: 10.2%; Table 6.1b); thus far during Treatment 2, egg-to-fry survivals have been significantly higher (t-test $p < 0.01$), ranging from 18.1% to 40.0% (mean: 25.2%, Table 6.1b). 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 other streams. The Chum egg-to-smolt survival estimates for the 2008, 2010, 2013 and 2014 brood years in the Coquitlam River exceeds published values for this species, and these, and possibly all, were possibly 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 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 week relationship between an observer's guess of their efficiency and the mark-recapture based estimate of their efficiency. 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.

There was strong support of a linear escapement-to-fry relationship during Treatment 2 (R^2 = 0.77, Figure 6.13) but only moderate support of one during Treatment 1 (R^2 = 0.36). The lower fit during Treatment 2 was largely the result of 2005, when the number of fry production was considerably higher than expected. If this was considered an outlier and excluded, there would be a higher level of support for a linear relationship during Treatment (R^2 = 0.79). A linear relationship suggests that within the escapement range thus far, fry abundance was largely a product of the number of spawners with no indication of an influence of habitat limitations.

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 6.2; Cope 2014), 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 are moderately correlated with the Cheakamus River (R = 0.63 and R^2 = 0.60), suggest 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.

Overall, Chum Salmon returns to Coquitlam River were markedly improved in 2002-2017 compared to previous years. Chum salmon escapement was not rigorously assessed until 2002, but qualitative surveys by DFO field staff over several decades suggest that total escapement was typically less than 1000 adults prior to the implementation of the Treatment 1 flow regime in 1997 (DFO, SEDS).

6.4 Pink

All stock-recruitment relationship and egg-to-fry survival estimates for Pink are also preliminary at this stage due to the same reason as for Coho and Chum. Estimated adult Pink salmon returns to Coquitlam River ranged from 2,900-34,280 adults, significantly increasing abundance starting in 2009 (Table 6.1a). Fry production upstream of RST2 ranged from 148,000-6,030,000 (Table 6.1a), with a substantially increases since 2008. The egg-to-fry survival for 2003-2009 Pink broods (4.9%-9.9%, Table 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%), even when incorporating the 95% confidence limits of the 2012, 2014 and 2016 fry estimates, which signals they could be non-credible or at least, biased high result. 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.

There was weak support of a linear escapement-to-fry stock-recruitment relationship during Treatment 1 (R^2 = 0.36) and strong support to of one during Treatment 2 (R^2 = 0.75, Figure 6.10). However, the fit improved when including both treatment periods (R^2 = 0.83) providing weak support that the stock-recruitment relationship was similar during Treatment 1 and 2.

Pink escapement was poorly correlated with that and in the Alouette River (R=0.16, Figure 6.8). As well, there was little correlation between Pink fry yield in the Coquitlam River and that in the Alouette River (R^2 = 0.10) but a strong correlation exists with the Cheakamus River fry production (R = 0.90, Figure 6.9).

Our ability to distinguish treatment effects from region-wide abundance trends remains low given that escapement and flow treatments are confounded. So far escapement has been significantly higher under Treatment 2 conditions than during Treatment 1. Using the Alouette River, with many physical and biotic similarities, as a control could separate the effects of Treatment versus escapement; however results to date do not suggest it is sufficiently similar for this purpose. The strong correlation in fry abundance with the Cheakamus River suggest the high fry outmigration during recent years are not isolated to Coquitlam however the lack of adult monitoring prevents controlling for escapement. An alternative approach that would also account for the trend of increasing escapement is either a return to Treatment 1 flow conditions or a switch to a third flow treatment.

Pink Salmon were successfully reintroduced to 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 interaction under higher escapement during this period.

6.5 Comparison of fisheries benefits in Treatments 1 and 2

COQMON-07 generates abundance data at two or more life stages for four salmonid species in the Coquitlam River. However, at the end of the study, not all of these data will play an equally important role in assessing possible differences in fish productivity between treatments. In some cases, the number of years of data will be insufficient to allow for statistical comparisons between treatments. This is particularly true for Steelhead stock-recruitment and egg-to-smolt survival estimates since a single data point depends on cohorts from a number of years. Cohorts that reared in part under condition prior to Treatment 1 or during both Treatment 1 and 2 were removed from the dataset used to compare treatment effects. Also, the number of years available during Treatment 1 was also reduced for some life stages since monitoring did not begin until several years into the treatment period (Table 6.1a). In other cases, because of density-dependent mortality and population bottlenecks within the Coquitlam River, or extraneous survival factors (e.g., marine survival), abundance at one life stage will be more directly affected by the flow regime in the Coquitlam River than another. It is also important to note that release flows from Coquitlam Dam in 2009 were 2.0 cms higher on average than seasonal targets for Treatment 2. Thus, year 1 of Treatment 2 represents somewhat of an outlier in the flow experiment, though more similar to Treatment 2 conditions, but given the planned 9-year duration of Treatment 2, this is not likely to have a significant impact on the comparison of the two treatments.

For Coho and Steelhead, annual smolt abundance will likely be the best performance measure for comparing Treatment 1 and Treatment 2 freshwater productivity. Smolt abundance estimates were obtained during eight years for each species under Treatment 1 (Table 6.1a). Smolt abundance is arguably the best metric for comparing flow treatments because it is a direct measure of carrying capacity. It has the added advantage of relying solely on the juvenile data, which has relatively high precision and no indication of bias. Table 6.2 provides sample results of such a comparison. Other metrics of stream productivity such as smolts-per-spawner or egg-to-smolt survival are preferable only if recruitment falls below that required to fully seed juvenile habitat. Moreover, the number of years available for comparison under Treatment 1 versus Treatment 2 is reduced for these latter metrics. During Treatment 1, adult escapement was estimated for four and seven years, respectively, for Steelhead and Coho (2005-2008 and 2002-2008, respectively). This provides only one stock-recruitment datapoint for Steelhead (age-2 and age-3 smolt yield in 2007 and 2008, respectively, for the 2005 brood year), and five datapoints for Coho. In the case of Coho, the reliability of the stock-recruitment relationship is questionable due to the large uncertainty in the estimates of escapement. Fortunately, in years when escapements were estimated, Coho and Steelhead spawner densities appeared to be well above levels thought to be required for full seeding of juvenile habitat across both Treatments 1 and 2. This supports the notion that smolt abundance is the best metric for comparing flow treatments. Moreover, inter-annual variation in Coho and Steelhead smolt abundance was relatively low within Treatment 1 and 2 (Table 6.1a), and there was evidence of density-dependent survival at older juvenile life stages (Figures 6.1 and 6.5). These results suggest that the juvenile carrying capacity of the Coquitlam River had a major influence on Coho and Steelhead smolt yield thus far during the flow experiment.

Conversely, for Chum and Pink, there was evidence that recruitment accounted for a substantial portion of the variation in smolt yield among years. Therefore, when comparing productivity between Treatment 1 and Treatment 2 for Chum and Pink, it will likely be necessary to account for variation in escapement by using fry- per-recruit, egg-to-fry survival or other stock-recruitment model parameters as the performance measure for comparing treatments. The former has the capability of comparing if productivity has changed between treatments, both in terms of the slopes of the fry-per-recruit relationships and the offset between treatments (examples a and b, respectively; Figure 6.7). We used an analysis of covariance (ANCOVA) to evaluate the Chum fry-to-escapement relationship for Treatments 1 and 2 using the package STATS in R (R Development Core Team 2009). It provides a basic framework for eventual hypothesis testing, examples of analysis outputs and indicates the capacity of available date for hypothesis testing. Table 6.3 provides sample outputs of this analysis for Chum 2003-2016. The significant values for Escapement and Treatment but non-significant values for the interaction between' Escapement x Treatment' suggests that, to date, the two factors are strong predictor of fry yield, but that there is no interaction between the two variables. This analysis will be further developed in future years to incorporate the uncertainty of individual fry and escapement estimates.

By the end of nine years of monitoring during Treatment 2, for Chum, there will be six datapoints for Treatment 1 and eight in Treatment 2 but for Pink, there would be only half this number, assuming nine years of Treatment 2 monitoring. While this may be adequate to evaluate treatment effects for Chum, it is likely insufficient for Pink. For Pink, we are also concerned that between-treatment comparison of any performance measure may not be valid unless future escapements include the range during Treatment 1, which to date, have been at least two-fold higher than any during Treatment 1 (Figure 6.6). The problem is that without similar or at least overlapping escapement between treatments then we cannot separate between flow effects and escapement effects on productivity with the current experimental design. If higher escapements continue, thus continuing the non-overlapping escapements between Treatments 1 and 2 and the weaknesses it causes in an ANCOVA analysis, we would instead focus on a BACI analysis involving one or a combination of other systems.

The moderate to strong correlations during Treatment 1 and 2 so far between the Coquitlam and Alouette rivers for Coho smolt abundance and Cheakamus River for Chum and Pink fry (Figure 6.2) suggests the possibility of using these rivers as controls for the flow experiment in the Coquitlam River. While the Coquitlam and Alouette rivers share many similarities: they are both regulated by dams and flow diversions and headed by large reservoirs, they are comparable in size, gradient, and morphology, and they support similar fish communities, similarities with the Cheakamus are weaker. Smolt abundance estimates for the four species of interest in the Coquitlam River are also available for the Alouette River during most years of Treatment 1. Escapement data are available for some species in some years in the Alouette River as well (Cope 2011), but with monitoring on the Alouette River in 2014, analysis will remain relatively unchanged from what is currently available.

The inclusion of the Alouette River or others as a control stream would allow for a before-after control-impact (BACI) experimental design (Stewart-Oaten *et al.* 1986). A BACI design can be a robust method for assessing ecological impacts or manipulations at larger scales and are the key method for distinguishing between treatment effects and regional effects (Stewart-Oaten *et al.* 1986; McDonald *et al.* 2000). In the case of this study, including a control stream reduces the likelihood of committing a type 1 error (i.e., falsely attributing an observed change in fish productivity during Treatment 2 to higher flows when the change was actually cause by a different factor such as escapement, local climate pattern, etc.). Faces with the potential of higher productivity under Treatment 2 conditions, attributing a change to the flow treatment will hinge on the level of understanding of how productivity would have changed without the change in flow treatment. This remains uncertain for all species at the present time.

6. Fish Productivity during Treatment 1 and 2

7.0 Recommendations

7.1 Adult Escapement

1. Conduct at least four mark-recapture experiments per year for Chum and Pink, prioritizing those for Chum above all others (see section 6.5). Relatively few mark-recapture experiments have occurred during recent years of the study, yet data derived from these experiments is of critical importance 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, an aggressive approach to obtaining this information during years with favorable river conditions has the best chance of obtaining sufficient data.

2. Discontinue using the HBM-based approach to estimate Chinook and Coho escapement. Instead, estimate escapement using mean count for Chinook and peak abundance for Coho, which has provided nearly the same information as the HBM approach over the life of the project and over a wide range of run sizes. This would allow the elimination of three to five late season Coho surveys and the elimination of all future mark-recapture and survey life experiments for both species. These resources could then be redirected to Chum and possibly Pink. Monitoring of adult Chinook abundance was not included in the original study design, and did not commence until the end of Treatment 1 in 2008. Given the lengthy freshwater residency of juvenile Coho in the Coquitlam River, adult Coho escapement is also not considered a key metric for evaluating the flow experiment, and mainly serves to provide evidence that juvenile habitat is fully seeded each year (see Section 1.2). Provided that more reliable escapement estimates are not needed for Coho and Chinook for other management purposes, future mark-recapture efforts should be focused on Chum and Pink.

3. Continue reconnaissance surveys at the beginning of the arrival of Pink 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 access assessments 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.

7.2 Adult Steelhead Escapement

4. Under a scenario of no additional resources, we recommend continuing bi-weekly redd surveys from mid-March to early June. This will provide a reliable index of adult abundance and likely an unbiased adult escapement estimate. Under a scenario of additional resources, either from reallocation or increased funding, surveys should be scheduled every 7 -10 days from April 1 through the first half of May in order to minimize the number of new redds that are constructed and lost between surveys. Although our estimates of redd survey life suggest that most redds remained detectable for up to 20 days, the shorter survey interval is beneficial for two reasons: 1) our estimates of redd survey life may be biased high as they are based on the untested assumption that all new redds detected on each survey were constructed at the midpoint in time between the current and previous surveys and will remain detectable until the midpoint in time between the current and subsequent surveys; and 2) if the survey interval is set at 2 weeks, the actual interval will often be longer due to interruptions caused by poor survey conditions.

5. Continue using only one survey crew. While there remains a risk of aborting surveys due to the onset of poor conditions, incomplete or missed redd surveys were not a large source of uncertainty in 2005-2015. A second crew, without the additional resources required to test for consistency with past survey methods, can introduce significant bias and uncertainty into redd counts.

7.3 Juvenile Salmonid Standing Stock

6. As much as possible, continue sampling the original 12 snorkeling sites and the 12 new sites added in 2014 to maintain adequate precision.

7. 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 sufficient refined for all but this age class. If we find that precision would improve from further refining the detection probability, benefits from this would be apply to all previous sampling.

7.4 Smolt and Fry Production

8. Top priority should continue to be given to maximizing the number of Steelhead recaptures at RST2 by maintaining high capture efficiency at RST2 and smolt marking at RST2-4. The length of the trapping period and the trap configurations and locations for

Coho and Steelhead were appropriate in recent years, and a similar approach should be applied the future.

9. If resources allow, mark Steelhead parr by capture location to better understand the extent of downstream movement and, in particular, the proportion that are moving downstream of the RST 2 trapping site. Marking would also provide estimates of the capture efficiency of at least RST 2 for this size fish. RST captures of Steelhead parr have not been considered and could represent additional production not accounted thus far.

8.0 References

Adkison, M.D. and Z. Su. 2001. A comparison of salmon escapement estimates using a hierarchical Bayesian approach versus separate maximum likelihood estimation of each year's return. Can. J. Fish. Aquat. Sci. 58:1663-1671.

Arnason, A.N., C.W. Kirby, C.J. Swartz, J.R, Irvine. 1996. Computer analysis of data from stratified mark-recovery experiments for estimation of salmon escapements and other populations. Can. Tech. Rep. Fish. Aquat. Sci. 2106.

BC Hydro. 2003a. Addendum to the Coquitlam Dam Water Use Plan Consultative Report. Prepared for the BC Hydro Coquitlam Dam Water Use Plan project, Burnaby, BC.

BC Hydro. 2003b. Lower Coquitlam River 2003 instream flow needs assessment interim report on transect data collection. Prepared for the BC Hydro Coquitlam Dam Water Use Plan project, Burnaby, BC.

Bohlin, T., and B. Sundstrom. 1977. Influence of unequal catchability on population estimates using the Lincoln Index and the depletion method applied to electro-fishing. Oikos 28:123-129.

Bozek, M. A., F.J. Rahel. 1991. Comparison of streamside visual counts to electrofishing estimates of Colorado River cutthroat trout fry and adults. N. Amer. J. Fish. Mange. 11:38-42.

Bradford, M. J. 1995. Comparative review of Pacific salmon survival rates. Can. J. Fish. Aquat. Sci. 52: 1327-1338.

Bradford, M.J., J. Korman, P. S. Higgins. 2005. Using confidence intervals to estimate the response of salmon populations (*Oncorhynchus* spp.) to experimental habitat alterations. Canadian Journal of Fisheries and Aquatic Sciences, 2005, 62:2716-2726

Bradford, M. J., G. C. Taylor, et al. 1996. Empirical review of coho salmon smolt abundance and the prediction of smolt production at the regional level. Trans. Amer. Fish. Soc. 126:49-64.

Bradford, M. J. and P. S. Higgins. 2000. Habitat-, season, and size-specific variation in diel activity patterns of juvenile Chinook salmon (Oncorhynchus tshawytscha) and Steelhead trout (Oncorhynchus mykiss). Canadian Journal of Fisheries and Aquatic Sciences 58:1-10.

Bradford, M. J., R. A. Myers, et al. 2000. Reference points for coho salmon (Oncorhynchus kisutch) harvest rates and escapement goals based on freshwater production. Can. J. Fish. Aquat. Sci. 57:677-686.

Brooks, S.P., E.A. Catchpole, and B.J.T. Morgan. 2000. Bayesian animal survival estimation. Statistical Science 15:357-376.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference, 2nd ed. Springer. 454 pp.

Campbell, R. F. and J. H. Neuner. 1985. Seasonal and diurnal shifts in habitat utilized by resident rainbow trout (Salmo gairderi) observed in western Washington Cascade Mountain streams. Pages 39-49 in F. W. Olson, R. G. White, and R. H. Hamre, editors. Symposium on Small Hydropower and Fisheries. American Fisheries Society, Western Division and Engineering Section, Bethesda, Maryland.

Conlin, K. and B.D. Tutty. 1979. Juvenile field trapping manual. Fish. and Mar. Serv. Man. Rep. No. 1530.

Cope, S. 2002. Alouette River salmonid smolt migration enumeration. Unpublished report prepared for the Alouette River Management Committee and BC Hydro by Westslope Fisheries, Cranbrook BC.

Cope, S. 2011. Alouette River salmonid smolt migration enumeration. Unpublished report prepared for the Alouette River Management Committee and BC Hydro by Westslope Fisheries, Cranbrook BC.

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.

Darroch, J.N. 1961. The two-sample capture-recapture census when tagging and sampling are stratified. Biometrika 48: 241-260.

Decker, S. 1998. Influence of off-channel habitat restoration and other enhancement on the abundance and distribution of salmonids in the Coquitlam River. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C. 35 p.

Decker, S., and G. Lewis. 1999. Response of salmonids to off-channel habitat restoration and increased minimum flows in the Coquitlam River. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., December 1999. 49 p.

Decker, S., and G. Lewis. 2000. Fish Response to Increased Minimum Flows and Habitat Restoration in the Coquitlam River: a 5 Year Review. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., December 2000. 40 p.

Decker. A.S., and M. Foy. 2000. The contribution of restored off-channel habitat to smolt production in the Coquitlam River. British Columbia Ministry of Environment, Watershed Restoration Technical Bulletin 4(3):22-24.

Decker, S., J. Schick, and G. Lewis. 2003. 2002 results for the Coquitlam River salmonid smolt enumeration program. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., April 2003. 32 p.

Decker, S., J. Schick, and G. Lewis. 2006. Coquitlam River salmonid smolt enumeration program: 2006 results. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., January 2006. 48 p.

Decker, A.S., and J. Hagen. 2007. The distribution and abundance of juvenile Chinook salmon in the lower Thompson River basin in relation to spawner abundance and habitat characteristics. Unpublished report prepared for the Pacific Salmon Commission, June 2007. 88 pages.

Decker, S., J. Macnair, G. Lewis, and J. Korman. 2007. Coquitlam River Fish Monitoring Program: 2000-2006 Results. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., June 2007. 116 p.

Decker, S., J. Macnair, G. Lewis, and J. Korman. 2008. Coquitlam River Fish Monitoring Program: 2000-2007 Results. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., June 2007. 125 p.

Decker, A.S., and J. Hagen. 2009. Stock assessment of juvenile Gerrard rainbow trout in the Lardeau/Duncan System: feasibility study 2005-2008. Prepared for Habitat Conservation Trust Fund, Victoria, British Columbia, September 2006. 71 p.

Decker, A.S., J. Hagen, and R. G. Bison. 2009. Annual distribution and abundance of Steelhead parr in the lower Thompson River basin during 2001-2008 in relation to spawner abundance and habitat characteristics. Consultant Report Prepared for BC Ministry of Environment, Kamloops BC. 78 p.

Decker, A.S., J. Macnair, G. Lewis. 2012. Coquitlam River Fish Monitoring Program: 2000-2010 Results. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C. March 2012. 147 p.

English, K.K., Bocking, R.C., and Irvine, J.R. 1992. A robust procedure for estimating salmon escapement based on the area-under-the-curve method. Can. J. Fish. Aquat. Sci. 59:1982-1989.

Fell, C., D.J.F McCubbing, L.J. Wilson, and C.C. Melville. 2013. Cheakamus River Chum Salmon Escapement Monitoring and Mainstem Spawning Groundwater Survey 2001-2013. Unpublished report prepared for B.C. Hydro Power Facilities, Burnaby, B.C., January 2013, 63p.

Fleming, I.A., and M.R. Gross. 1993. Breeding success of hatchery and wild coho salmon (Oncorhynchus kisutch) in competition. Ecological applications 3:230-245.

Freymond, B and S. Foley. 1985. Wild Steelhead: Spawning escapement estimates for Bolt Case Area Rivers –1985. Washington State Game Department, Fisheries Management Division. Report No. 86-12. 204 pp.

Gallagher, S.P., and C.M. Gallagher. 2005. Discrimination of Chinook salmon, coho salmon, and Steelhead redds and evaluation of the use of redd data for estimating escapement in several unregulated streams in northern California. N. Amer. J. Fish. Mange. 25:284-300.

Gelman, A., Carlin, J.B. Stern, H.S., and D.B. Rubin. 2004. Bayesian data analysis. Chapman and Hall/CRC. 668 pp.

Griffith, J. S. 1981. Estimation of the age-frequency distribution of stream-dwelling trout by underwater observation. Prog. Fish. Cult. 43:51-52.

Groot, C., and L. Margolis. 1991. Pacific salmon life histories. UBC Press, Vancouver BC.

Hagen, J., A. S. Decker, J. Korman, and R. Bison. 2010. Effectiveness of night snorkeling for estimating Steelhead parr abundance in a large river basin. North American Journal of Fisheries Management 30:1280-1302.

Higgins, P.S., J. Korman, and M.J. Bradford. 2002. Statistical power of monitoring inferences derived from experimental flow comparisons planned for Coquitlam-Buntzen Water Use Plan. B.C. Hydro internal report. Burnaby, B.C. 20 p.

Hillman, T. W., J. W. Mullan, and J. S. Griffith. 1992. Accuracy of underwater counts of juvenile Chinook salmon, coho salmon, and Steelhead. N. Amer. J. Fish. Manage. 12:598-603.

Hume, J.M.B., and E.A. Parkinson. 1987. Effect of density on the survival, growth, and dispersal of Steelhead trout fry (*Salmo gairdneri*). Can. J. Fish. Aquat. Sci. 44:271-281.

Irvine, J.R., R.C. Bocking, K.K. English, and M. Labelle. 1992. Estimating coho salmon (*Oncorhynchus kisutch*) spawning escapements by conducting visual surveys in areas surveyed using stratified random and stratified index sampling designs. Can. J. Fish. Aquat. Sci. 49:1972-1981.

Jacobs S., J. Firman, G. Susac, D. Stewart and J. Weybright. 2002. Status of Oregon coastal stocks of anadromous salmonids, 2000-2001 and 2001-2002; Monitoring Program Report Number OPSW-ODFW-2002-3, Oregon Department of Fish and Wildlife, Portland, Oregon.

Keeley, E. R. and C. J. Walters 1994. The British Columbia Watershed Restoration Program: summary of the experimental design, monitoring and restoration techniques workshop. Vancouver, B.C., B.C. Ministry of Environment, Lands and Parks.

Koning, C.W., and E.R. Keeley. 1997. Salmonid biostandards for estimating production benefits of fish habitat rehabilitation techniques. In P.A. Slaney and D. Zaldokas. [eds.] Fish habitat rehabilitation procedures. Watershed Restoration Technical Circular No. 9.

Korman, J., R.N.M. Ahrens, P.S. Higgins, C.J. Walters. 2002. Effects of observer efficiency, arrival timing, and survey life on estimates of escapement for Steelhead trout (*Oncorhynchus mykiss*) derived from repeat mark and recapture experiments. Can. J. Fish. Aquat. Sci. 59:1116-1131.

Korman, J., J. Schick, and A. Clarke. 2010. Cheakamus River Steelhead Juvenile and Adult Abundance Monitoring: Fall 2008 to Spring 2009. Report prepared for BC Hydro, Burnaby, BC, February 2010.

Korman, J., A. S. Decker, B. Mossop, and J. Hagen. 2011. A comparison of electrofishing- and snorkeling-based mark-recapture methods for estimating detection probability and abundance of juvenile Steelhead in a medium-sized river. North American Journal of Fisheries Management 30:1303-1314.

Koski, K.V. 1975. The survival and fitness of two stocks of chum salmon from egg deposition to emergence in a controlled stream environment at Big Beef Creek. Doctoral Dissertation, University of Washington, Washington.

Krebs, C.J. 1999. Ecological Methodology. Benjamin/Cummings, Menlo Park. 620 pp.

Ladell, J., J. Korman, D.J.F. McCubbing. 2010. Cheakamus River bull trout radiotelemetry and enumeration program 2007-2009. Consultant report prepared for Canadian National Railway Company, CN Environment, Surrey, BC, July 2010.

Leake, A.C. 2006. Coquitlam-Buntzen Water Use Plan Terms of Reference: COQMON#7 - Lower Coquitlam River Fish Productivity Index. BC Hydro, Water Use Plan Monitoring Report, November 14, 2006.

Macnair, J. 2003. Coquitlam River water use plan: 2003 chum fry enumeration. Unpublished report prepared for BC Generation Sustainability and Aboriginal Relations, Burnaby BC by Living Resources Environmental Services.

Macnair, J. 2004. 2003 Coquitlam River adult salmon escapement survey. Prepared for BC Generation Sustainability and Aboriginal Relations, Burnaby BC by Living Resources Environmental Services. 21 p.

Macnair, J. 2005. 2004 Coquitlam River adult salmon escapement survey. Prepared for BC Generation Sustainability and Aboriginal Relations, Burnaby BC by Living Resources Environmental Services. 29 p.

Macnair, J. 2006. Progress Report: 2005 Coquitlam River adult salmon escapement survey. Prepared for BC Generation Sustainability and Aboriginal Relations, Burnaby BC by Living Resources Environmental Services. 36 p.

Macnair, J. 2010a. Coquitlam River Fish Monitoring Program 2010 adult Steelhead results summary. Prepared for BC Generation Sustainability and Aboriginal Relations, Burnaby BC by Living Resources Environmental Services, July 2010. 22 p.

Macnair, J. 2010b. Coquitlam River Fish Monitoring Program: 2009 adult escapement interim data report. Prepared for BC Generation Sustainability and Aboriginal Relations, Burnaby BC by Living Resources Environmental Services. 36 p.

McCarthy, M.A. 2007. Bayesian methods in ecology. Cambridge University Press, New York. 296 p.

McCullough, D.A. 1999. A review and synthesis of effects and alterations to the water temperature regime on freshwater life stages of salmonids with special reference to Chinook salmon. Report prepared for the U.S. Environmental Protection Agency, Region 10, Seattle, Washington, July 1999. EPA Report No. 910-R-99-010.

McDonald, T.L., W.P. Erickson, and L.L. McDonald. 2000. Analysis of count data from before-after control-impact studies. Journal of Agricultural, Biological, and Environmental Statistics. 5:262-279.

Myers, J, 2003. Historical population structure of Willamette and lower Columbia River basin Pacific salmonids. National Marine Fisheries Service, Washington Dept. of Fish and Wildlife.188 p.

Otis, D. L., K. P. Burnham, G. C. White, and D. R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wild. Monogr. 62:1-135.

Perrin C.J., J.R. Irvine. 1990. A review of survey life estimates as they apply to the area-under-the-curve method for estimating the spawning escapement of Pacific salmon. Can. Tech. Rep. Of Fish and Aquat. Sci. No.1733.

Peterson, J.T., R.F. Thurow, and J.W. Guzevich. 2004. An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. Trans. Amer. Fish. Soc. 113:462-475.

Plante, N. 1990. Estimation de la taillee d’une population animale a l’aide d’un modele de capture-recapture avec stratification. M.Sc. thesis, Universite Laval, Quebec.

Ptolemy, R. A. 1993. Maximum salmonid densities in fluvial habitats in British Columbia. Pages 223-250 in L. Berg and P. W. Delaney, editors. Proceedings of the Coho Workshop, Nanaimo, BC, May 26-28, 1992. Canada Department of Fisheries and Oceans.

Ptolemy, R.A. 2007. Trial electrofishing and training using shore-based units; basic findings and observations for Coquitlam River sampling of riffle/rapid habitats during September 10-11, 2007. Management brief prepared for the Coquitlam-Buntzen Water Use Plan Consultative Committee, November 21, 2007. (included as Appendix 6.1 in this document).

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada 191.

Riley, S.C., J. Korman, S. Decker. 1997. Coquitlam River salmonid stock assessment. Unpublished draft report prepared for B.C. Hydro, Burnaby, B.C. 20 p.

Royle, J.A., and R.M. Dorazio. 2008. Hierarchical modeling and inference in ecology: the analysis of data from populations, metapopulations and communities. Academic Press, London, UK. 444 p.

Salo, E.O. 1991. Life history of chum salmon, (*Oncorhynchus keta*). *In* Groot, C., and L. Margolis (eds.), Pacific salmon life histories, p. 231-309. Univ. B. C. Press, Vancouver, B. C., Canada.

Satterthwaite, T.D. 2002. Population health goals and assessment methods for Steelhead in the Klamath mountains province. Monitoring Program Report Number OPSW-ODFW-2002-08, Oregon Department of Fish and Wildlife, Portland, OR.

Seber, G.A.F. 1982. The estimation of animal abundance. 2nd ed. Griffin, London.

Spiegelhalter, D.J., Thomas A., Best, N.G., and Lunn, D. 1999. WinBUGS User Manual: Version 1.4. Cambridge: MRC Biostatistics Unit, 60 pp.

Stewart-Oaten, A., W.W. Murdoch, and K.R. Parker. 1986. Environmental impact assessment: psedoreplication in time? Ecology 67:929-940.

Sturtz, S., Legges U., and Gelman A. 2005. R2WinBGS: a package for running WinBUGS from R. Journal of statistical software 3: 1-16.

Su, Z., Adkison, M.D., and Van Alen, B.W. 2001. A hierarchical Bayesian model for estimating historical salmon escapement and escapement timing. Can. J. Fish. Aquat. Sci. 58: 1648-1662.

Symons, P. E. K. 1979. Estimated escapement of Atlantic salmon (*Salmo salar*) for maximum smolt production in rivers of different productivity. J. Fish. Res. Bd. Can. 36:132-140.

Tautz, A.F., B.R. Ward, and R. A. Ptolemy. 1992. Steelhead trout productivity and stream carrying capacity. PSARC Working Paper S92-68.

Thurow, R. F., J. Peterson, J.W. Guzevich. 2006. Utility and validation of day and night snorkel counts for estimating bull trout abundance in first- to third-order streams. N. Amer. J. Fish. Mange. 26:217-232.

Ward, B. R. and P. A. Slaney. 1993. Egg-to-smolt survival and fry-to-smolt density dependence of Keogh River Steelhead trout. Can. Spec. Publ. Fish. Aquat. Sci. 118.

Wyatt, R. J. 2002. Estimating riverine fish population size from single- and multiple-pass removal sampling using a hierarchical model. Canadian Journal of Fisheries and Aquatic Sciences 59:695-706.

Wyatt, R.J. 2003. Mapping the abundance of riverine fish populations: integrating hierarchical Bayesian models with a geographic information system (GIS). Canadian Journal of Fisheries and Aquatic Sciences 60: 997-1007.

Zar, J.H. 1999. Biostatistical Analysis. 4th Edition. Pearson Prentice-Hall. 944 pp

9.0 Figures, Tables and Appendices

9.1 Figures and Tables for Chapter 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 2		
Period	Target	Min	Treatment 1	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

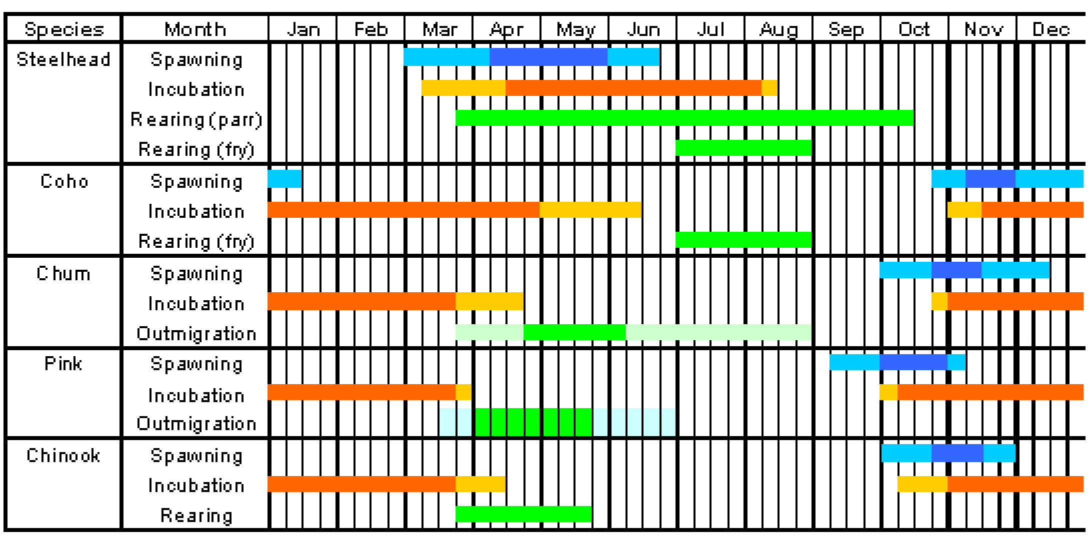
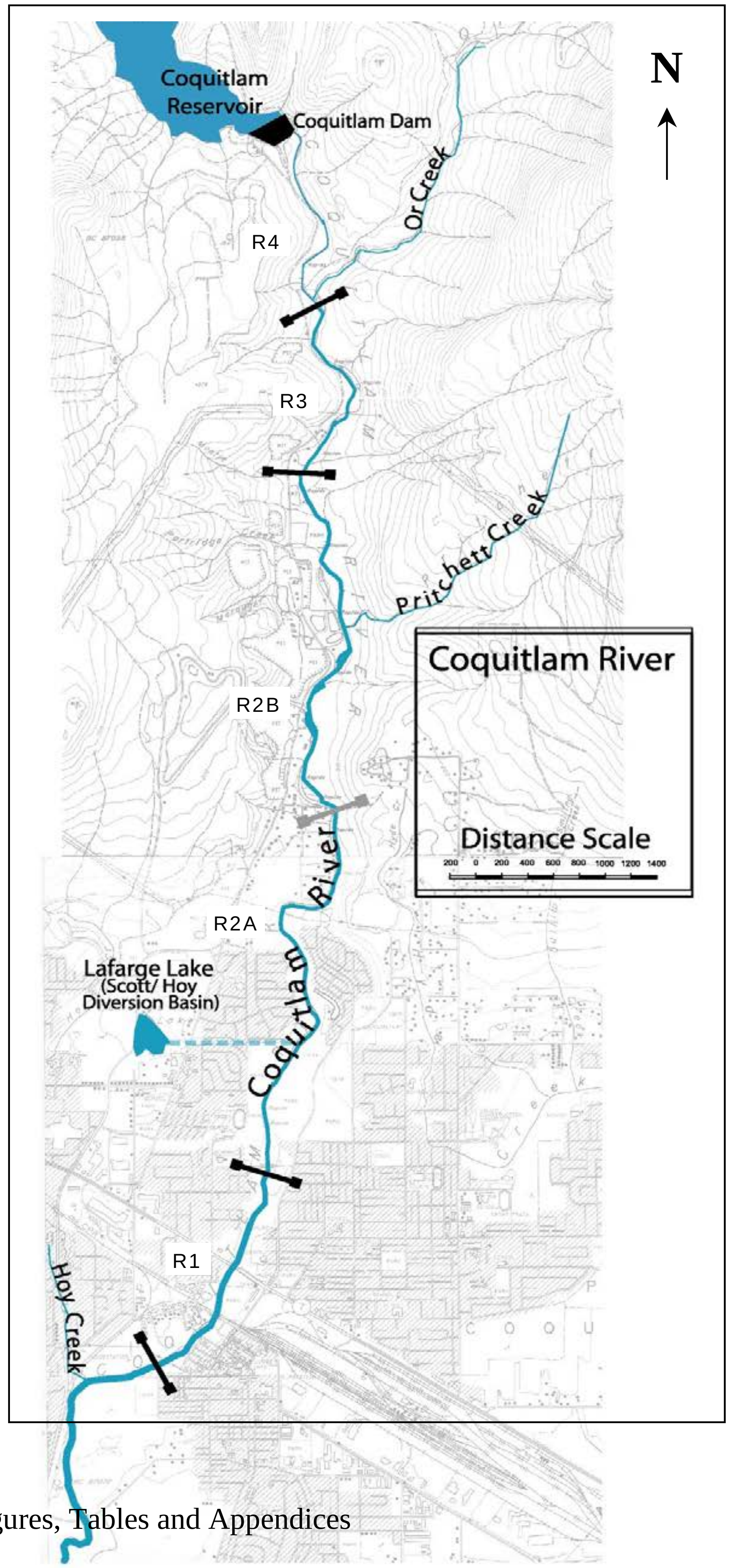


Figure 1.1 Life stage periodicity chart for anadromous salmonids in Coquitlam River.



9. Figures, Tables and Appendices

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

9. Figures, Tables and Appendices

9.2 Figures, Tables and Appendices for Chapter 2

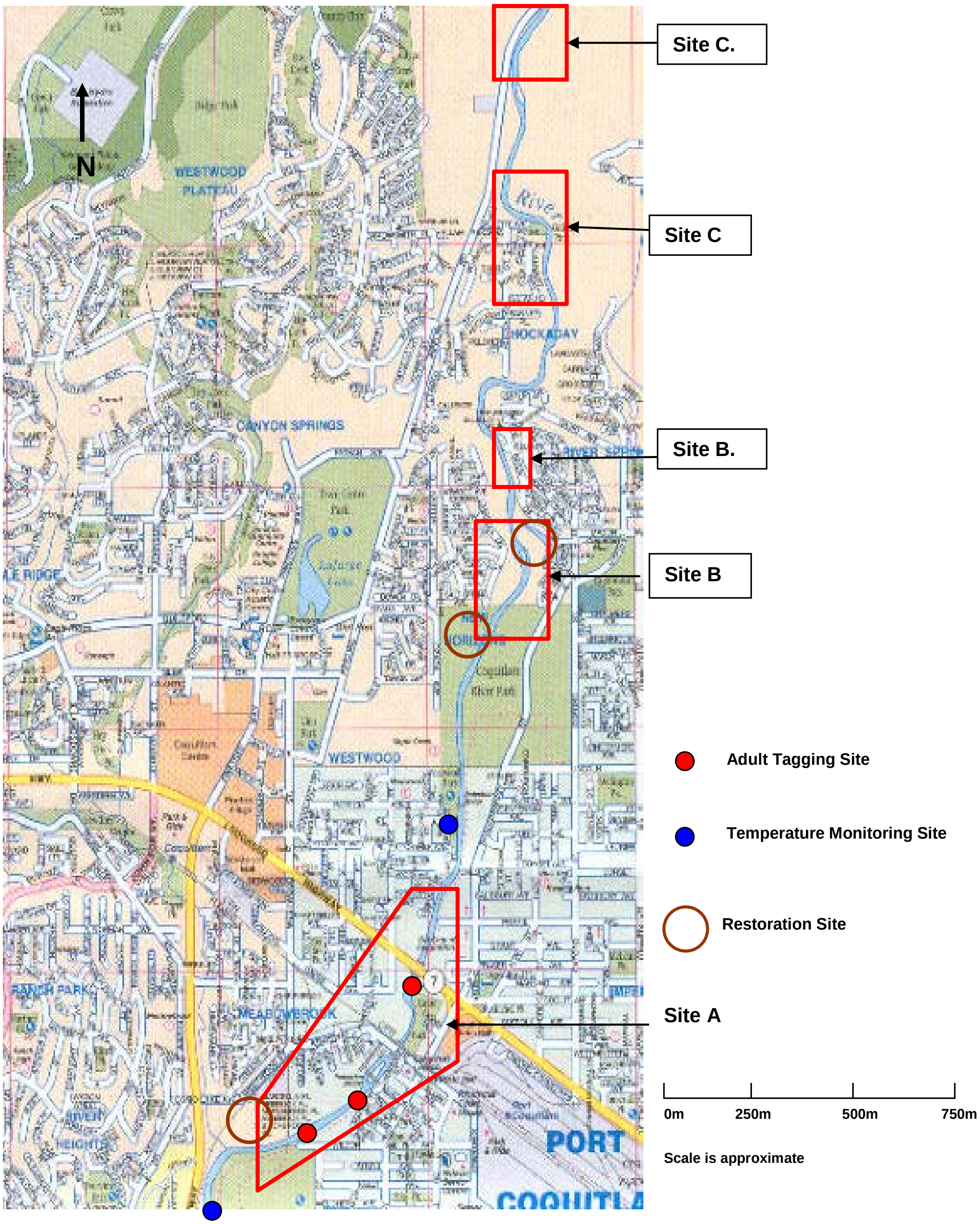


Figure 2.1 Map showing adult spawning index sites A-C in the lower portion of Coquitlam River study area (reaches 1, 2a)

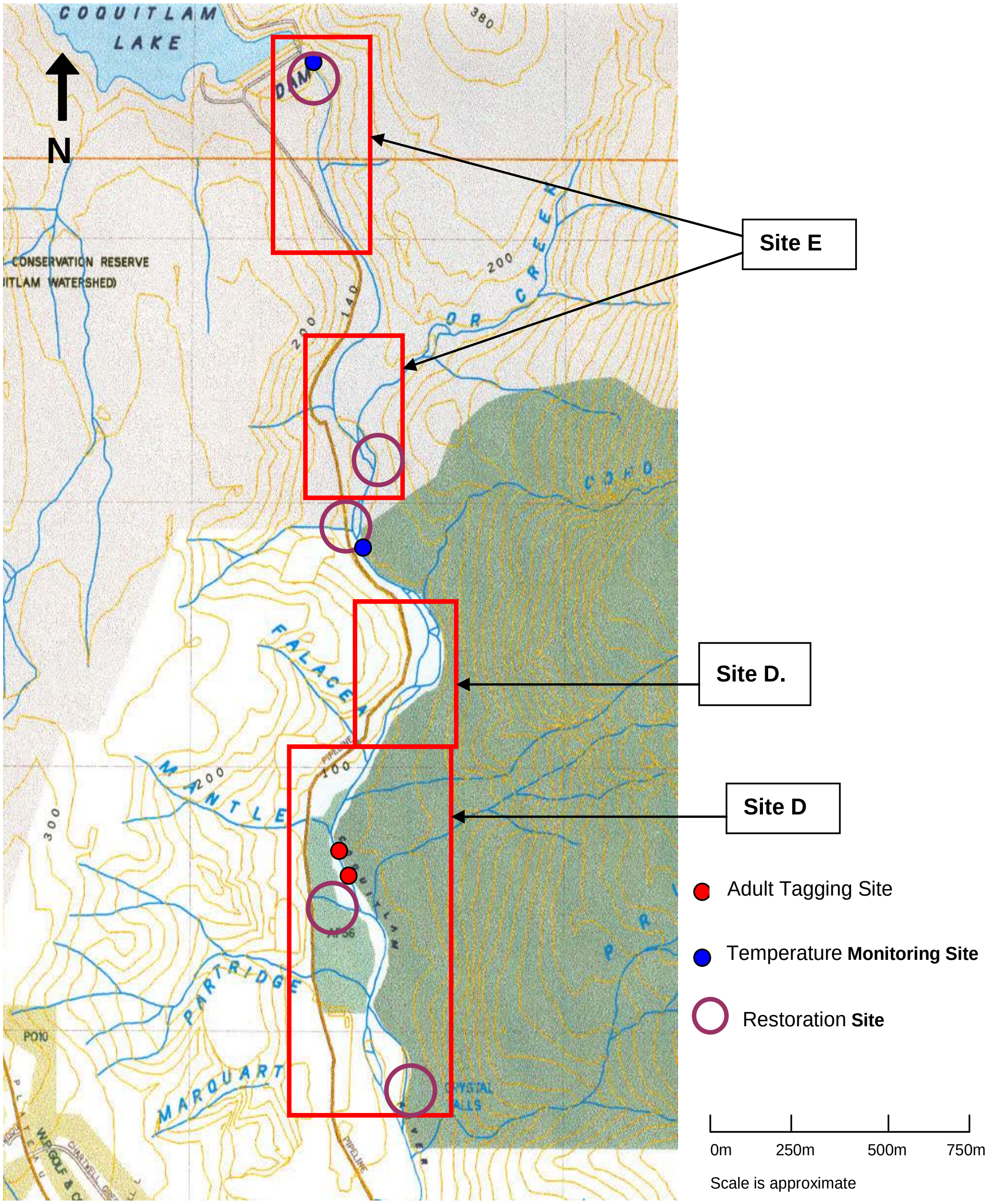


Figure 2.2 Map showing adult spawning index sites D and E, in the upper portion of Coquitlam River study area (reaches 2b, 3 and 4).

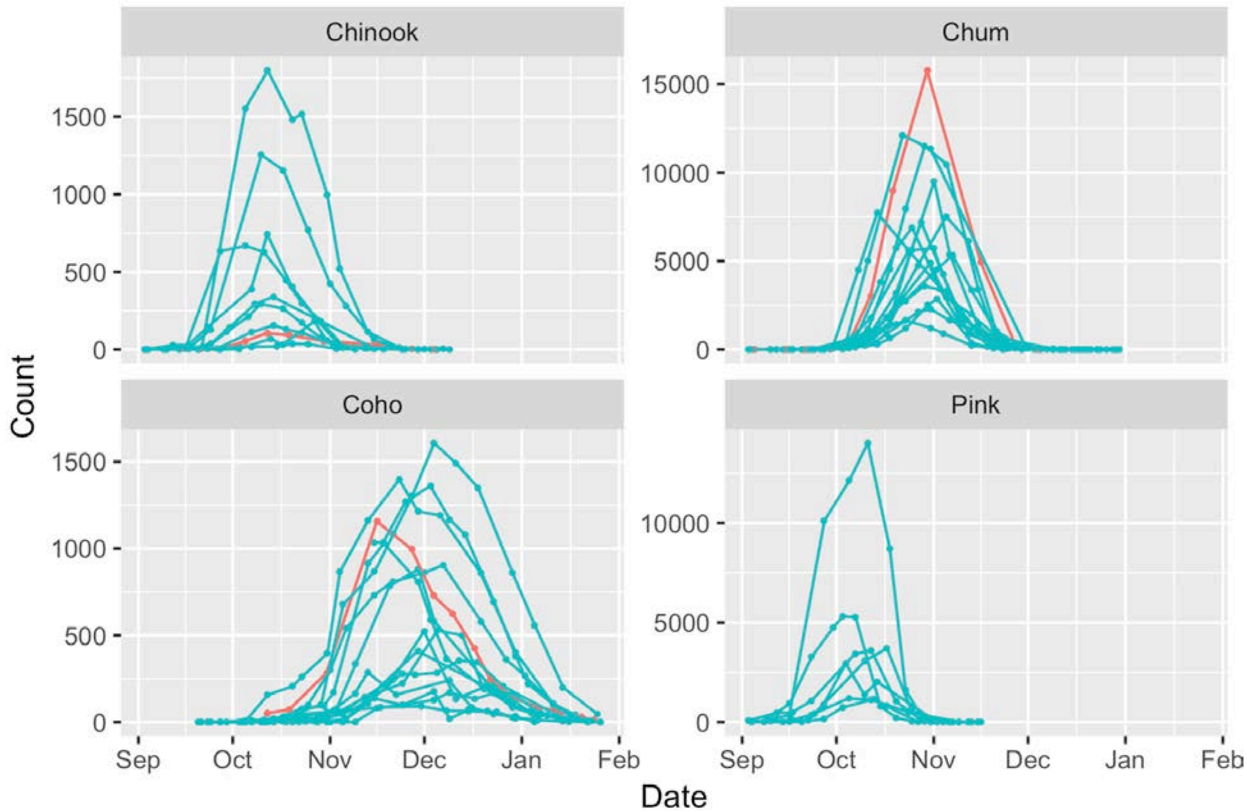


Figure 2.3 Spawning run timing base on survey counts for Chinook, Chum, Coho and Pink Salmon in the Coquitlam River during 2016 (red line) and 2002-2015 (teal).

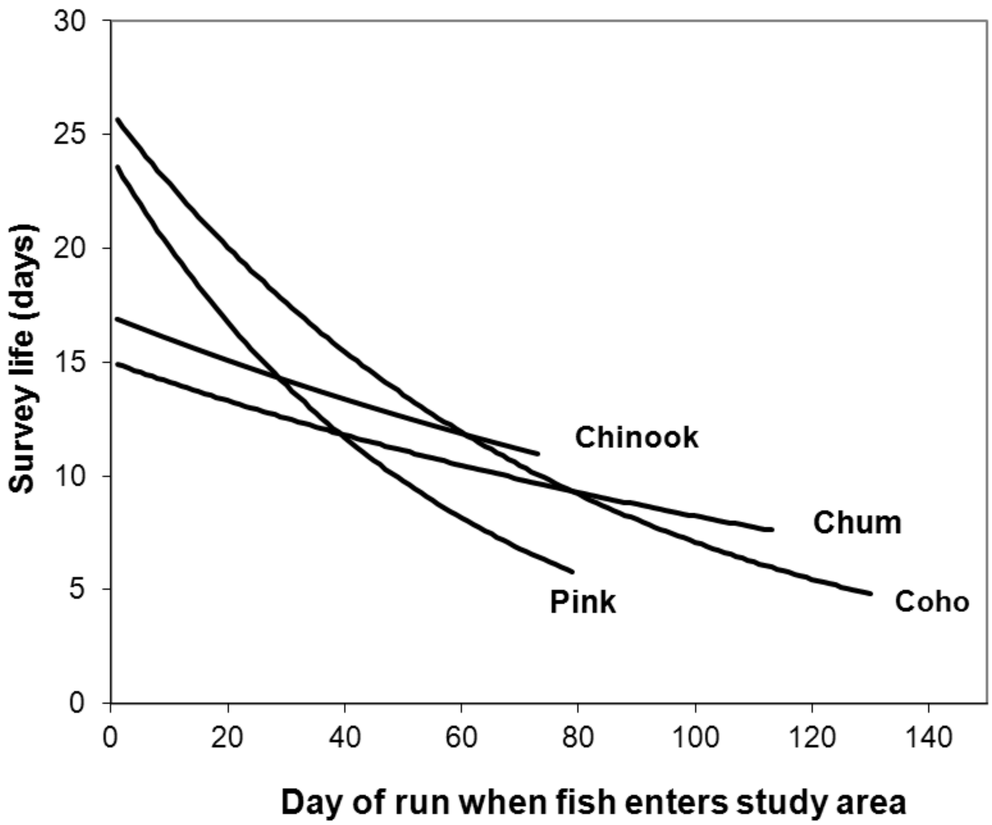


Figure 2.4 Modeled relationship between mean survey life and day of arrival in the study area for Chum, Pink, Coho, and Chinook salmon in the Coquitlam River based on empirical data from other streams.

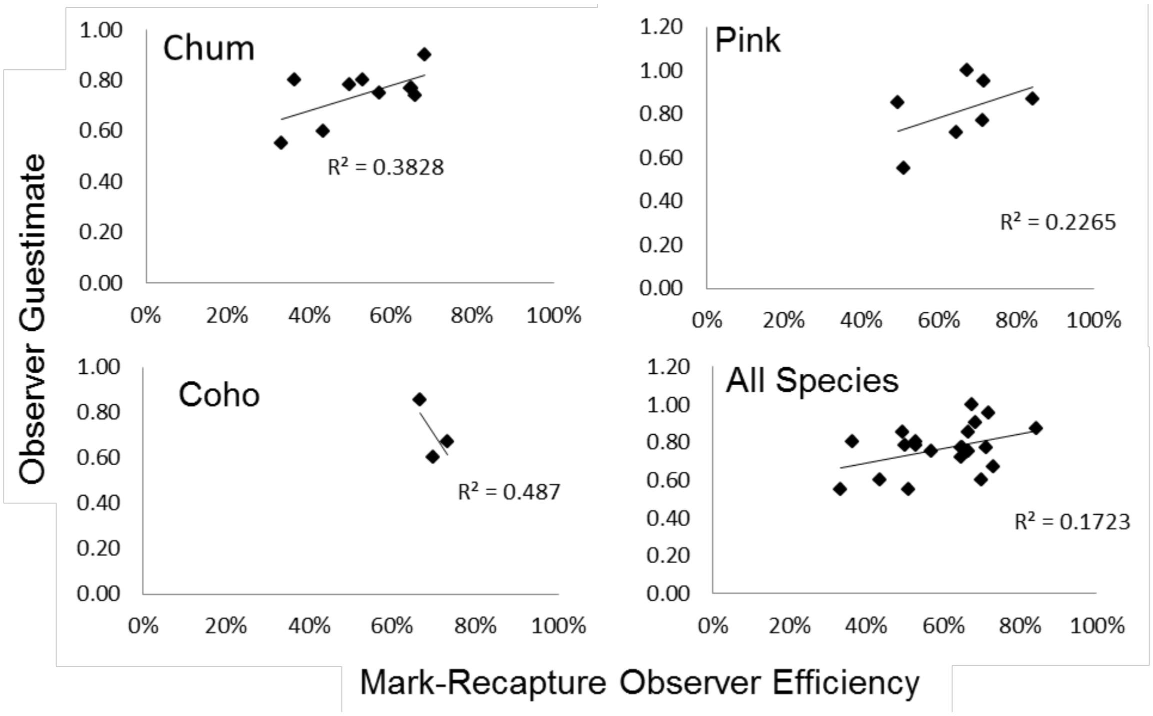


Figure 2.5 Relationship between the surveyor's 'guesstimate' of observer efficiency and observer efficiency estimated from mark recapture experiments for Chum, Pink and all species combined conducted opportunistically since 2006 in the Coquitlam River.

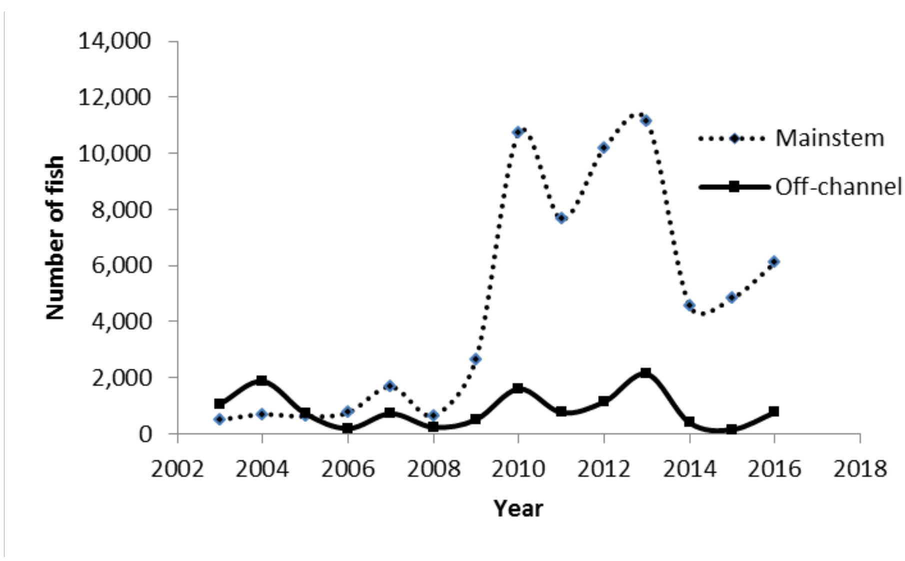


Figure 2.6 Estimated numbers of Coho spawning in mainstem and side-channel habitat in the Coquitlam River 2003-2016. Note that although escapement was estimated for 2000, surveys did not differentiate between habitat types.

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 2016 brood year.

Escapement Year	Date	Estimated water column visibility (m)					
		site A	site B	site C	site D	site E	non-index
2016	05-Sep	>3	>3	>3	>3	>3	
2016	15-Sep	>3	>3	>3	>3	>3	
2016	21-Sep	>3	>3	>3	>3	>3	>3
2016	29-Sep	>3	>3	>3	>3	>3	>3
2016	05-Oct	0.80	0.80	0.80	0.90	1.10	0.85
2016	12-Oct	0.90	0.90	0.90	1.10	1.20	0.70
2016	19-Oct					1.00	0.70
2016	30-Oct	0.75	0.80	0.80	0.9	1.10	0.75
2016	16-Nov	0.80	0.90	0.90	0.9	1.10	
2016	27-Nov	0.90	0.90	0.90	1.00	1.20	
2016	04-Dec	0.90	0.90	0.90	1.00	1.20	
2016	10-Dec	1.00	0.80	1.00	1.00	1.10	
2016	17-Dec	1.1	1.20	1.20	1.20	1.30	0.70
2016	22-Dec		1.00	1.00	1.00	1.30	
2016	30-Dec		1.10	1.10	1.10	1.40	
2016	06-Jan		1.20	1.10	1.10	1.40	
2016	14-Jan		1.00	1.00	1.10	1.40	
2016	20-Jan		1.00	1.00	1.20	1.40	
2016	24-Jan		1.00	1.00	1.20	1.30	

Surveyor "guesstimates" of observer efficiency (0.0-1.0): (chum salmon example)							
2016	05-Sep	1	1	1	1	1	
2016	15-Sep	1	1	1	1	1	
2016	21-Sep	1	1	1	1	1	1
2016	29-Sep	1	1	1	1	1	1
2016	05-Oct	0.65	0.65	0.65	0.7	0.75	1.5
2016	12-Oct	0.7	0.7	0.75	0.8	0.8	1.2
2016	19-Oct					0.7	1.2
2016	30-Oct	0.6	0.6	0.6	0.6	0.7	1.3
2016	16-Nov	0.7	0.7	0.7	0.7	0.8	
2016	27-Nov	0.6	0.6	0.7	0.7	0.8	
2016	04-Dec	0.6	0.6	0.6	0.7	0.8	
2016	10-Dec	0.6	0.7	0.7	0.7	0.7	
2016	17-Dec	0.8	0.8	0.8	0.8	0.9	1.2
2016	22-Dec		0.7	0.7	0.7	0.8	
2016	30-Dec		0.7	0.7	0.7	0.8	
2016	06-Jan		0.7	0.7	0.7	0.8	
2016	14-Jan		0.7	0.7	0.7	0.7	
2016	20-Jan		0.8	0.8	0.8	0.9	
2016	24-Jan		0.8	0.8	0.8	0.9	

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	10	7	3	2	22
mean	0.52	0.66	0.70	0.60	0.59
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	0.95
Survey life (days)					
Number of estimates	6	4	3	2	
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	

Table 2.3 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.

Species	Site	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
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
	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
	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
	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
	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
	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
Pink	A	-	0.16	-	0.17	-	0.21	-	0.13	-	0.17	-	0.21	-	0.30	-
	B	-	0.10	-	0.05	-	0.03	-	0.06	-	0.02	-	0.05	-	0.03	-
	C	-	0.20	-	0.11	-	0.08	-	0.12	-	0.06	-	0.13	-	0.12	-
	D	-	0.21	-	0.20	-	0.24	-	0.25	-	0.19	-	0.22	-	0.17	-
	E	-	0.24	-	0.42	-	0.33	-	0.36	-	0.53	-	0.32	-	0.29	-
	NI	-	0.10	-	0.05	-	0.11	-	0.08	-	0.04	-	0.07	-	0.08	-
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
	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
	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
	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
	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
	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
Chinook	A						0.02	0.02	0.06	0.04	0.03	0.02	0.03	0.01	0.01	0.02
	B						0.00	0.01	0.03	0.01	0.02	0.00	0.02	0.00	0.00	0.00
	C						0.10	0.05	0.08	0.07	0.07	0.00	0.07	0.02	0.02	0.01
	D						0.06	0.11	0.09	0.18	0.22	0.08	0.22	0.10	0.10	0.04
	E						0.64	0.76	0.70	0.60	0.61	0.84	0.61	0.86	0.86	0.90
	NI						0.18	0.05	0.04	0.10	0.06	0.06	0.06	0.00	0.00	0.03

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Table 2.4 Annual escapement estimates for Chum, Pink, Coho and Chinook salmon for the years 2002-2016.

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

Table 2.5 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	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	0.76
	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	0.15
	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	0.09
	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	0.24
Pink	M/S	0.55		0.65		0.71	0.64		0.76		0.59		0.77		0.74		0.72
	NOC	0.19		0.22		0.20	0.20		0.12		0.22		0.12		0.15		0.15
	OCR	0.26		0.13		0.09	0.16		0.12		0.19		0.11		0.11		0.13
	OC	0.45		0.35		0.29	0.36		0.24		0.41		0.23		0.26		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	0.89
	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	0.02
	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	0.09
	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	0.11
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	0.95
	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	0.02
	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
	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	0.05

Appendix 2.1. Results of the 2006-2016 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 2.1. 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 2.2 Unadjusted live counts of Pink salmon during 2003-2015.

			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
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 2.2 continued (Pink)

			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
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	-

Appendix 2.3 Unadjusted live counts of Chum salmon during 2002-2016.

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
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 2.3 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 2.3 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 2.3 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
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	-

Appendix 2.4 Unadjusted live counts of Coho salmon during 2002-2016.

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
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 2.4 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
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 2.4 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
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 2.4 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
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	-

9. Figures, Tables and Appendices

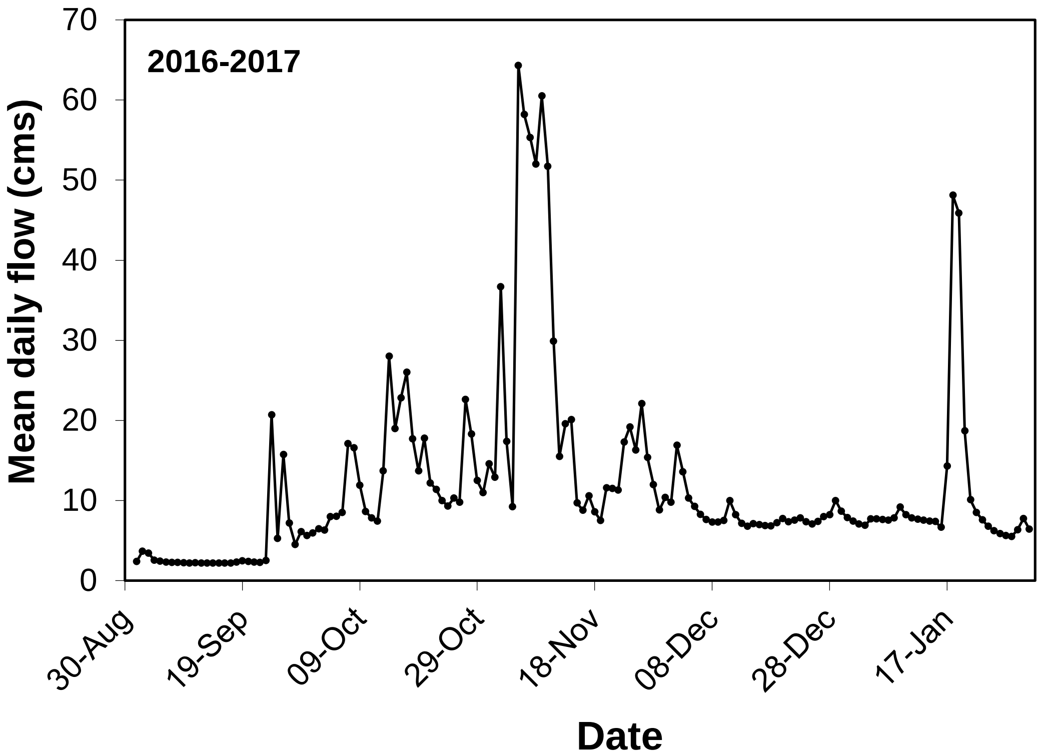
Appendix 2.5 Unadjusted live counts of Chinook salmon during 2007-2016.

			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 [1]	27 [1]	-
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

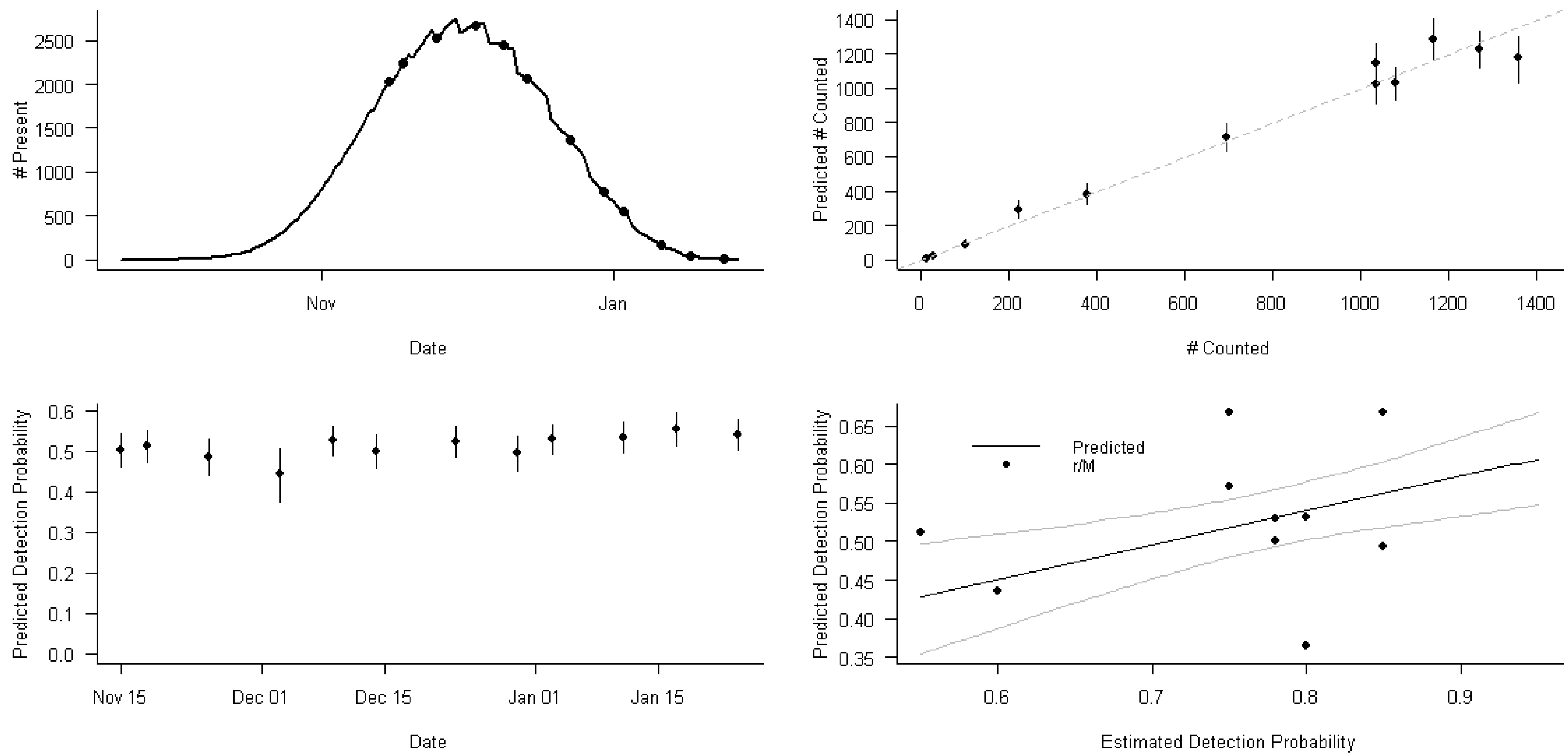
Appendix 2.5 continued (Chinook)

			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
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 Mean daily flows in Coquitlam River at Port Coquitlam during the fall and winter spawning period in 2016-2017 (Water Survey of Canada, stn. 08MH141).



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.



9.3 Figures, Tables and Appendices for Chapter 3

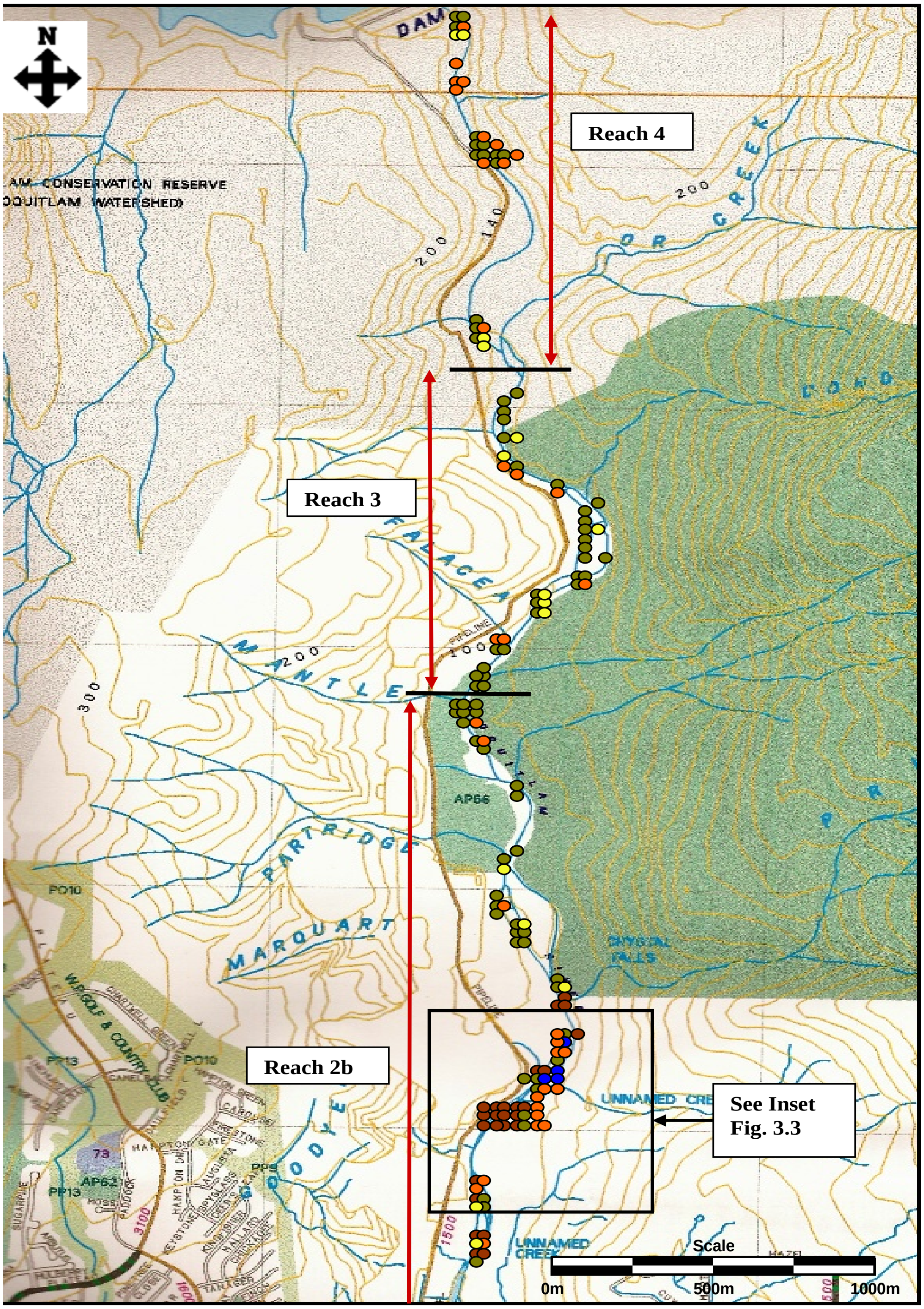


Figure 3.1 Steelhead redd locations in reaches 2b-4 in Coquitlam River in 2006, which was the highest escapement year during 2005-2016. Coquitlam Dam is the upstream boundary of the survey area. See Figure 3.2 for redd symbol legend.

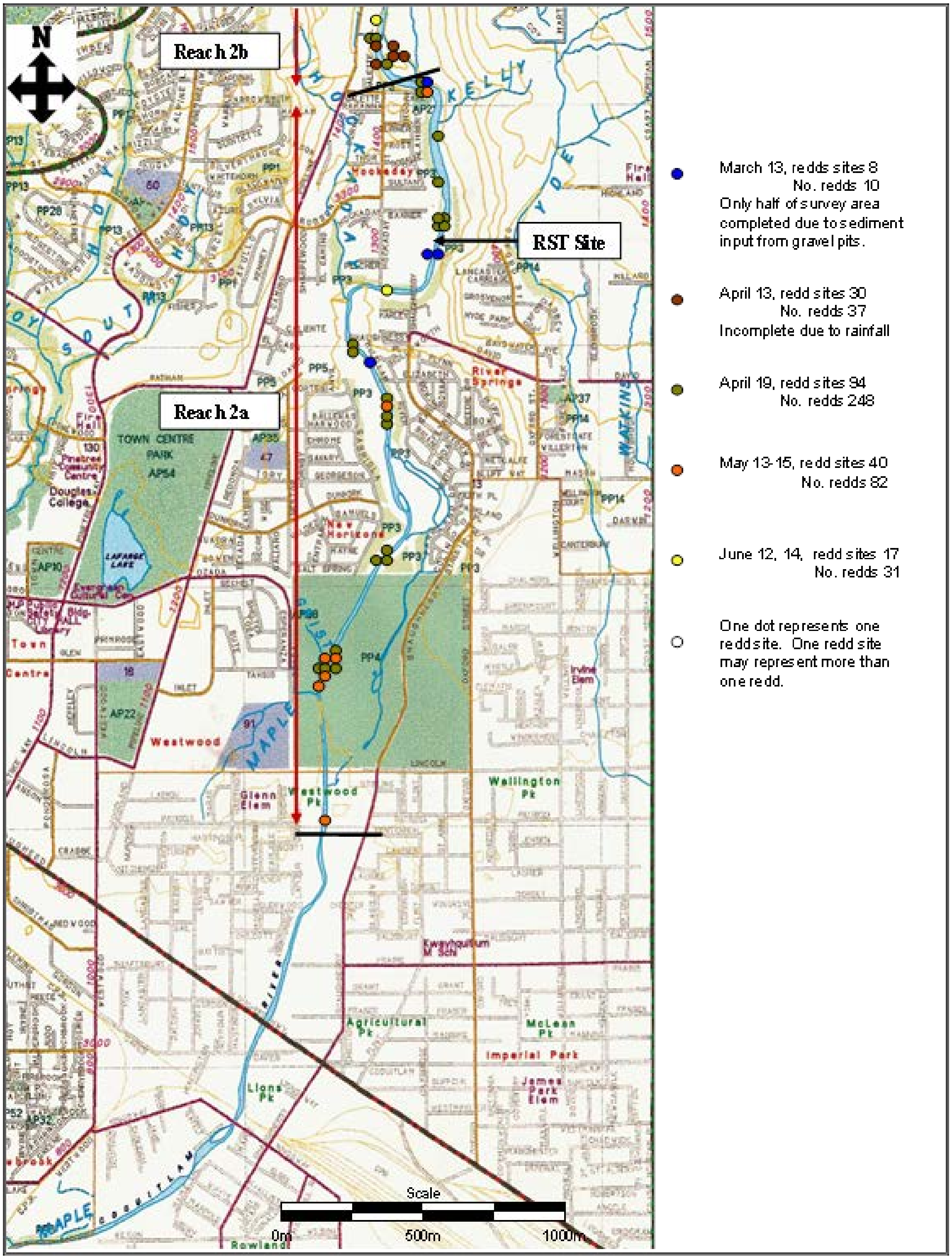


Figure 3.2 Steelhead redd locations in reaches 2a-2b in Coquitlam River in 2006. The downstream boundary of reach 2a is also the survey area boundary.

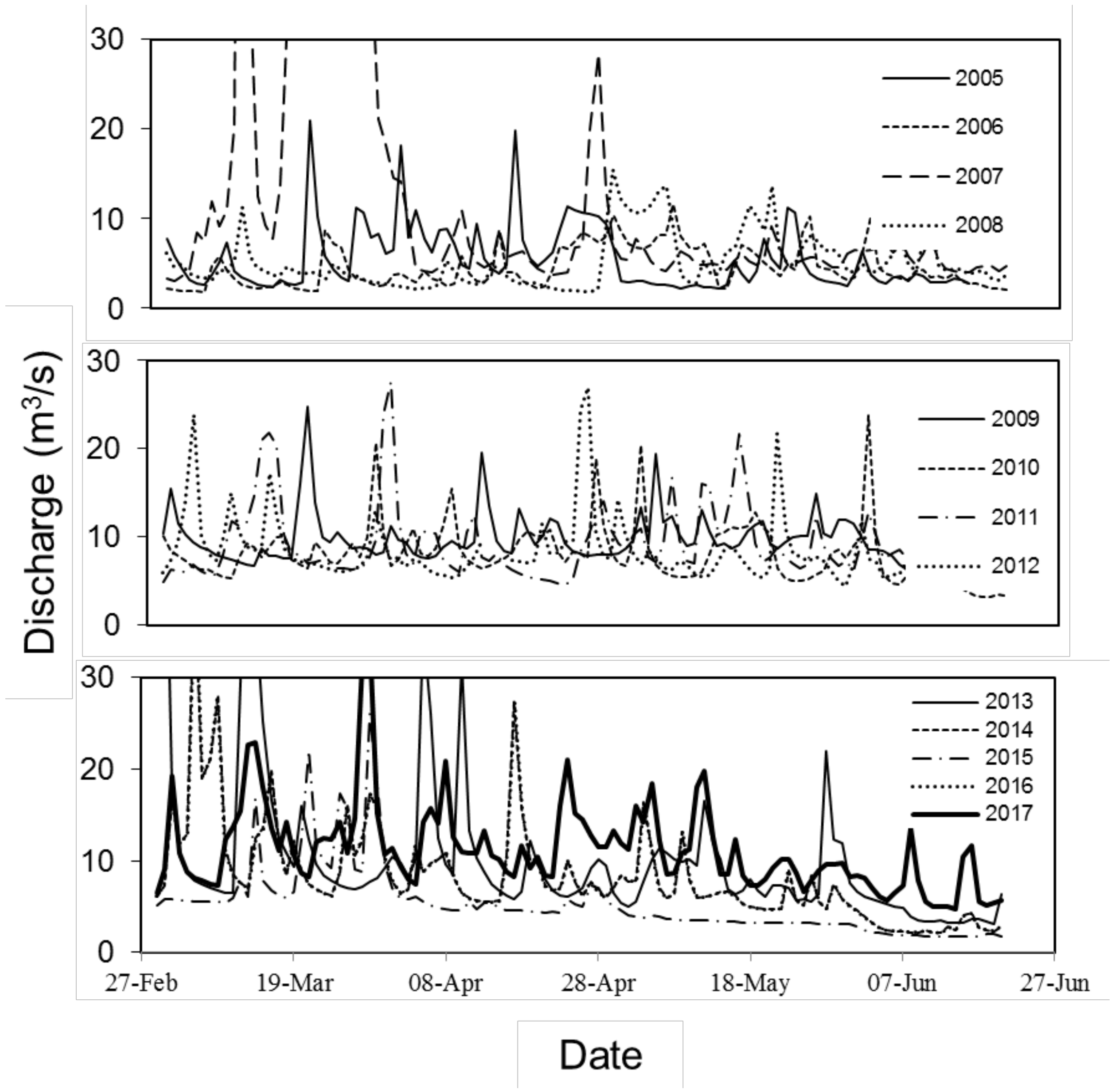


Figure 3.3 Discharge (cms) in Coquitlam River at Port Coquitlam during Steelhead spawning period in 2005 – 2017 (Water Survey of Canada station 08MH002).

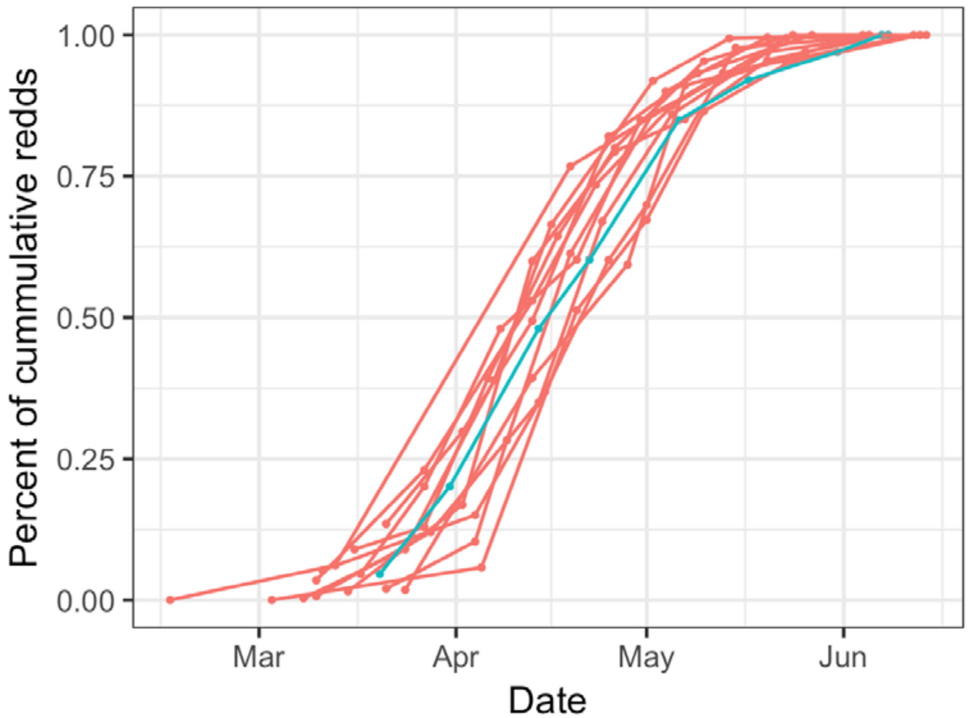


Figure 3.4 Cumulative proportion of the total Steelhead redd count observed over time during 2017 (teal) and 2005 - 2016 (red).

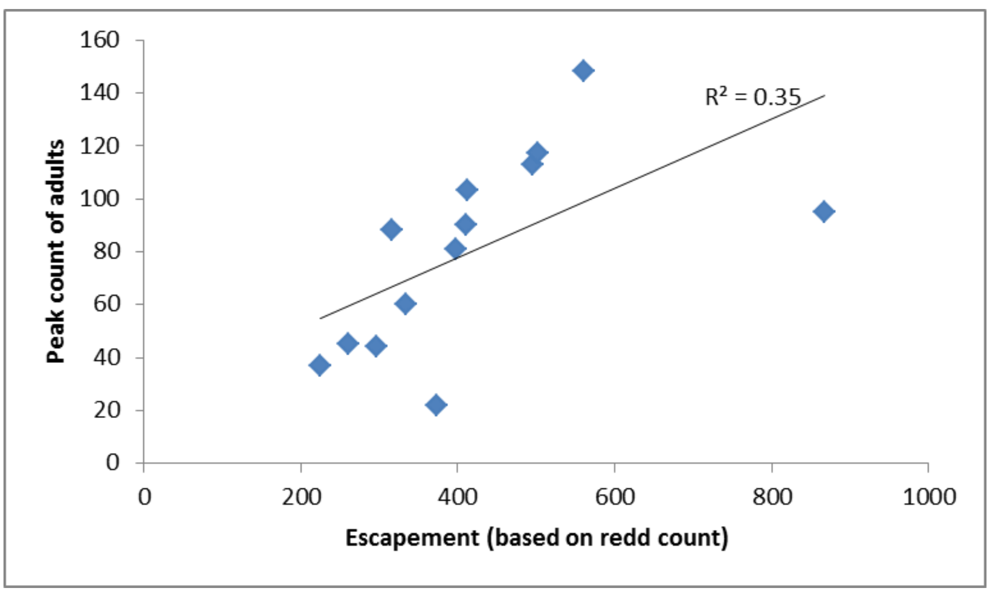


Figure 3.5 Relationship between Adult Steelhead escapement estimates based on redd counts and peak counts of live adults during surveys 2005-2017 in the Coquitlam River.

Table 3.1 Survey dates with raw counts of Steelhead redds, estimated new redds, and live adult counts for all surveys during 2005-2017. 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**

Table 3.1 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**

Table 3.1 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**

[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

Table 3.2 Summary statistics for Steelhead escapement to Coquitlam River during 2005-2017 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

Table 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

Appendix 3.1 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 sur 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

Total new redds adjusted for redd survey life **71.41**

9.4 Figures, Tables and Appendices for Chapter 4

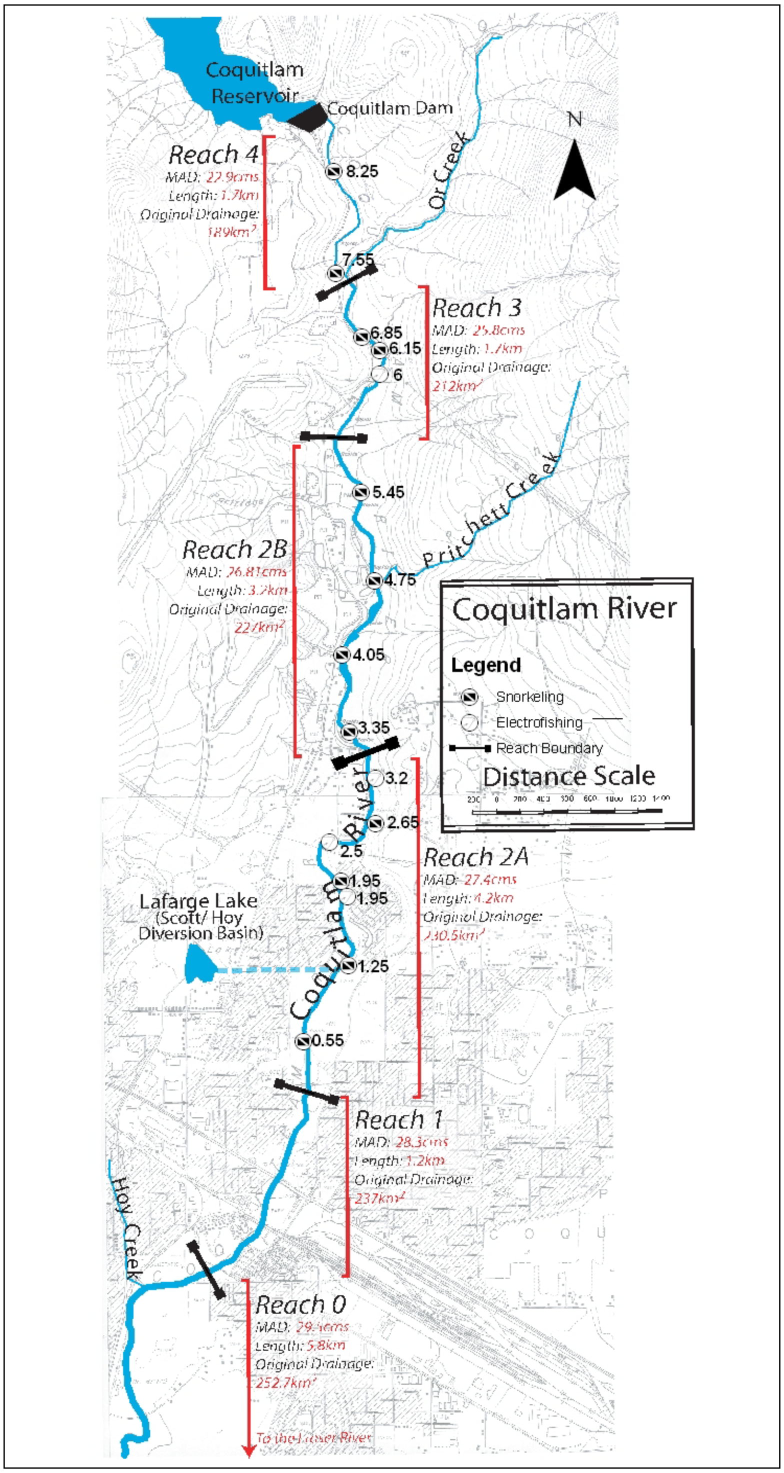


Figure 4.1 Map of Coquitlam River showing juvenile standing stock study area, reach breaks and original 12 sampling sites.

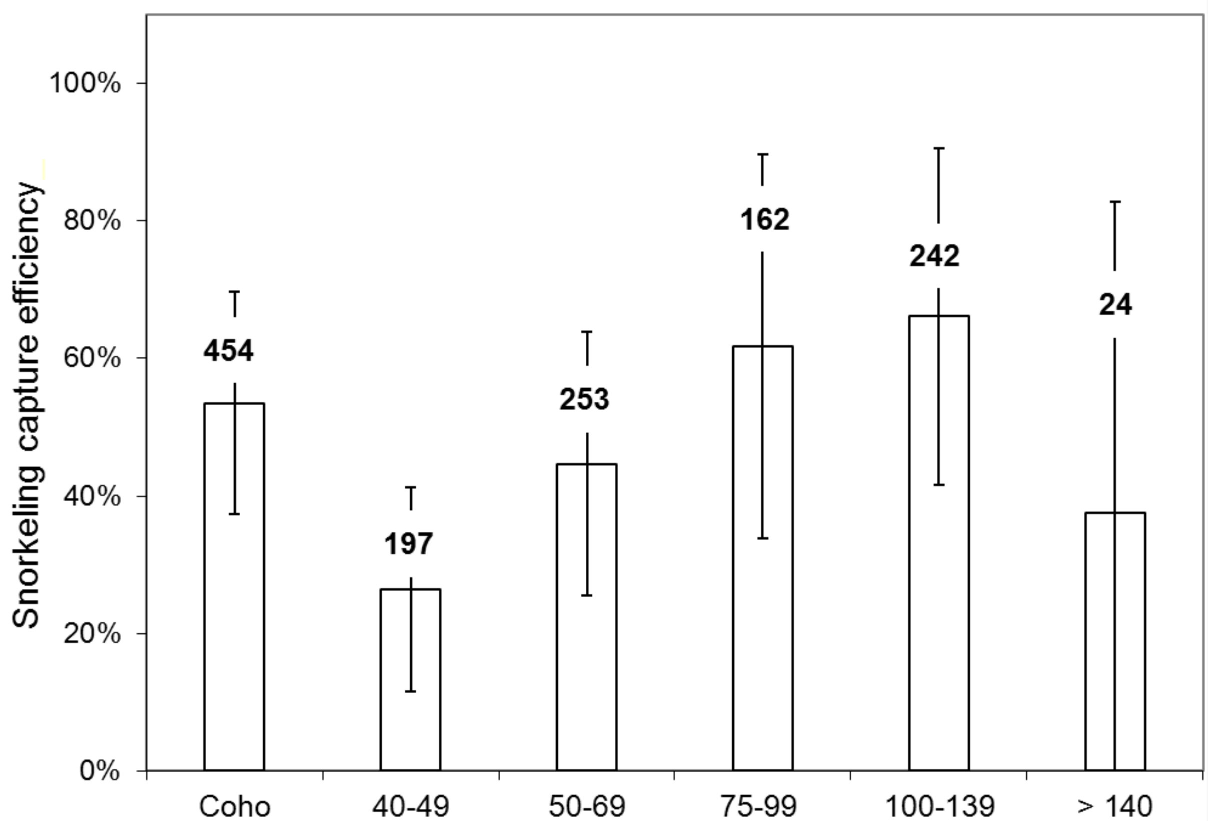


Figure 4.2 Maximum likelihood estimates of mean snorkeling detection probability for juvenile Coho and Steelhead by forklength class (Steelhead only) at 16 sites in the Coquitlam River during 2007-2013. Errors bars represent ±1 standard deviation of the mean. Values above bars are total numbers of marked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

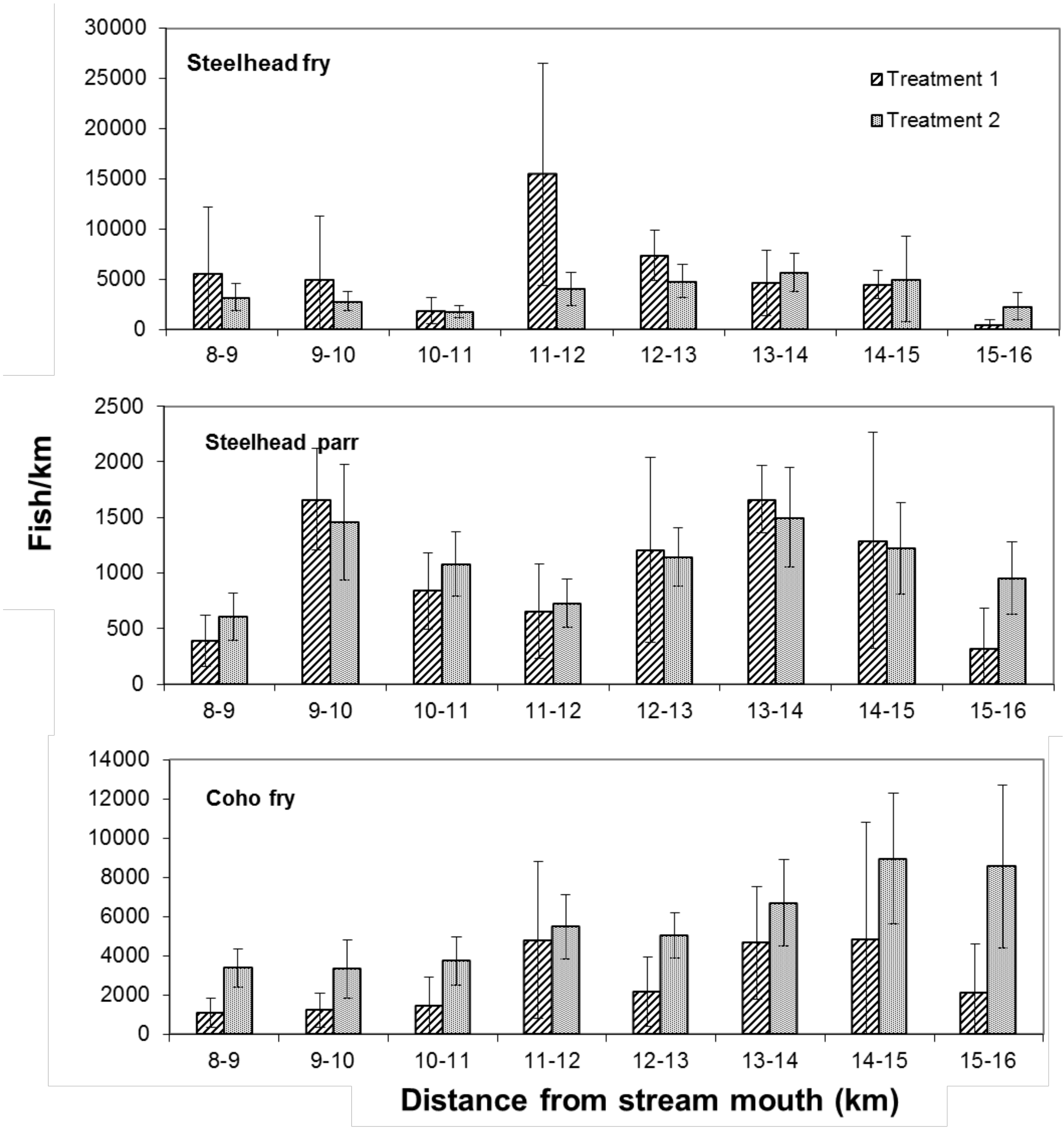


Figure 4.3 Linear distribution of juvenile salmonids in the Coquitlam River during Treatment 1(2006-2008) and Treatment 2 (2009-2017). Bars represent mean abundance estimates and 95% confidence intervals for years under flow Treatments 1 and 2. Estimates are based on calibrated snorkeling counts at 10-12 sampling sites 2006-2013 and 24 sites 2014-2016.

Table 4.1 Summary of habitat data for night snorkeling and day electrofishing sites in Coquitlam River in 2017.

Sampling method	Site no.	Upstream distance (km)	Site area (m^2)	Mean length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m)	D90 (m)	Boulder (%)	Cobble (%)	Gravel (%)	Fines (%)
snorkeling	0.55	8.25	540.0	25	22	0.46	0.25	-	60	25	10	5
snorkeling	0.90	8.60	384.2	25	15	0.61	0.25	-	45	30	15	10
snorkeling	1.25	8.95	465.8	25	19	0.40	0.48	-	40	35	20	5
snorkeling	1.60	9.30	797.5	25	32	0.29	0.23	-	30	40	25	5
snorkeling	1.95	9.65	573.3	25	23	0.37	0.25	-	65	20	10	5
snorkeling	2.65	10.35	419.2	25	17	0.73	0.20	-	45	30	15	10
snorkeling	3.00	10.70	461.7	25	18	0.29	0.16	-	65	25	5	5
snorkeling	3.35	11.05	408.3	25	16	0.36	0.45	-	20	40	30	10
snorkeling	3.70	11.40	601.7	25	24	0.49	0.20	-	50	25	15	10
snorkeling	4.05	11.75	918.3	25	37	0.22	0.32	-	20	40	30	10
snorkeling	4.40	12.10	757.5	25	30	0.47	0.40	-	55	20	15	10
snorkeling	4.75	12.45	479.2	25	19	0.46	0.30	-	50	35	10	5
snorkeling	5.00	12.70	438.3	25	18	0.48	0.26	-	35	40	20	5
snorkeling	5.20	12.90	607.5	25	24	0.36	0.41	-	50	30	15	5
snorkeling	5.45	13.15	226.3	25	9	0.33	0.30	-	50	30	15	5
snorkeling	5.60	13.30	467.5	25	19	0.46	0.33	-	30	40	20	10
snorkeling	5.70	13.40	624.2	25	25	0.27	0.35	-	35	40	20	5
snorkeling	5.80	13.50	479.2	25	19	0.33	0.42	-	30	35	25	10
snorkeling	6.15	13.85	370.8	25	15	0.35	0.32	-	30	40	20	10
snorkeling	6.85	14.55	226.3	25	9	0.36	0.24	-	45	35	15	5
snorkeling	7.20	14.90	266.7	25	11	0.55	0.57	-	60	20	10	10
snorkeling	7.55	15.25	221.7	25	9	0.31	0.20	-	60	25	10	5
snorkeling	7.90	15.60	390.8	25	16	0.53	0.36	-	50	30	15	5
snorkeling	8.25	15.95	194.2	25	8	0.29	0.24	-	50	35	10	5
electrofishing	1.95	10.2	122.9	19	6	0.49	0.39	-	55	25	15	5
electrofishing	2.50	10.7	125.4	19	7	0.34	0.45	-	35	45	15	5
electrofishing	3.20	11.5	122.7	20	6	0.24	0.16	-	40	30	20	10
electrofishing	6.00	14.5	130.7	20	7	0.37	0.34	-	35	40	20	5

Table 4.2 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

Table 4.3 Estimates of juvenile fish density, standing stock, and 95% confidence intervals by species and age class in Coquitlam River during 2006-2017. Estimates were derived from night snorkeling counts with the exception of 2011 Steelhead (0+).

Species/age class	Year	Density (fish/km)	Density (fish/100m^2)	Standing stock	Lower 95% CI	Upper 95% CI	± 95% CI
coho (0+)	2006	2,632	14.6	27,111	13,249	33,791	38%
coho (0+)	2007	1,787	13.1	18,405	12,340	28,481	44%
coho (0+)	2008	4,536	22.8	46,719	32,460	71,416	42%
coho (0+)	2009	5,126	20.7	52,794	38,958	83,936	43%
coho (0+)	2010	6,037	24.1	62,178	42,549	95,461	43%
coho (0+)	2011	8,871	42.8	91,367	52,468	118,903	36%
coho (0+)	2012	7,170	42.9	73,846	54,509	115,705	41%
coho (0+)	2013	6,823	39.2	70,279	52,339	114,403	44%
coho (0+)	2014	4,321	22.8	44,507	35,979	62,261	30%
coho (0+)	2015	3,505	19.7	36,101	28,320	44,591	23%
coho (0+)	2016	2,468	14.8	25,424	20,270	35,420	30%
coho (0+)	2017	5,744	37.8	59,166	45,730	79,661	29%
steelhead (0+)	2006	13,411	28.9	138,132	103,399	196,021	34%
steelhead (0+)	2007	3,131	9.4	32,251	21,784	89,848	106%
steelhead (0+)	2008	4,127	9.4	42,506	32,185	660,106	739%
steelhead (0+)	2009	3,597	8.1	37,047	29,002	1,355,054	1790%
steelhead (0+)	2010	3,850	10.6	39,657	29,627	151,626	154%
steelhead (0+)[1]	2011	2,131	9.6	21,949	-	-	-
steelhead (0+)	2012	5,362	14.5	55,232	40,520	81,398	37%
steelhead (0+)	2013	6,409	19.9	66,017	51,319	107,519	43%
steelhead (0+)	2014	3,179	7.9	32,746	26,499	44,724	28%
steelhead (0+)	2015	3,134	10.4	32,277	26,270	44,291	28%
steelhead (0+)	2016	2,932	8.9	30,203	22,396	43,135	34%
steelhead (0+)	2017	5,277	14.8	54,358	41,408	89,120	44%
steelhead (1+)	2006	580	2.9	5,976	3,532	22,859	162%
steelhead (1+)	2007	994	6.6	10,237	7,036	17,771	52%
steelhead (1+)	2008	992	4.6	10,222	7,446	20,770	65%
steelhead (1+)	2009	1,056	3.8	10,876	8,229	16,041	36%
steelhead (1+)	2010	787	3.7	8,106	6,556	10,710	26%
steelhead (1+)	2011	853	4.2	8,791	6,425	14,701	47%
steelhead (1+)	2012	1,036	6.0	10,668	8,002	17,462	44%
steelhead (1+)	2013	1,306	6.9	13,456	10,129	21,470	42%
steelhead (1+)	2014	618	3.2	6,369	5,115	8,669	28%
steelhead (1+)	2015	572	2.9	5,889	4,869	7,546	23%
steelhead (1+)	2016	506	2.9	5,216	4,321	6,416	20%
steelhead (1+)	2017	880	4.8	9,064	7,287	12,360	28%

[1]Biased low estimate due to overestimate of age-0+ detection probability (see section 4.3.1 for explanation)

Table 4.3 cont'd

Species/age class	Year	Density (fish/km)	Density (fish/100m^2)	Standing stock	Lower 95% CI	Upper 95% CI	± 95% CI
steelhead (2+)	2006	179	1.1	1,841	933	3,569	72%
steelhead (2+)	2007	192	1.3	1,978	1,145	3,950	71%
steelhead (2+)	2008	122	1.0	1,255	694	2,598	76%
steelhead (2+)	2009	310	1.9	3,196	1,963	6,402	69%
steelhead (2+)	2010	261	1.7	2,690	1,630	5,331	69%
steelhead (2+)	2011	375	1.9	3,862	2,443	7,266	62%
steelhead (2+)	2012	307	1.8	3,160	1,961	5,666	59%
steelhead (2+)	2013	255	1.6	2,625	1,582	4,713	60%
steelhead (2+)	2014	372	2.1	3,831	2,756	6,634	51%
steelhead (2+)	2015	249	1.3	2,561	1,822	4,181	46%
steelhead (2+)	2016	257	1.1	2,642	1,835	4,807	56%
steelhead (2+)	2017	311	1.9	3,207	2,177	5,371	50%
steelhead (parr)	2006	759	3.7	7,817	-	-	-
steelhead (parr)	2007	1,186	7.6	12,215	-	-	-
steelhead (parr)	2008	1,114	5.4	11,477	-	-	-
steelhead (parr)	2009	1,366	5.3	14,072	-	-	-
steelhead (parr)	2010	1,048	4.1	10,796	-	-	-
steelhead (parr)	2011	1,228	5.6	12,653	-	-	-
steelhead (parr)	2012	1,343	7.3	13,828	-	-	-
steelhead (parr)	2013	1,561	8.1	16,081	-	-	-
steelhead (parr)	2014	990	4.4	10,200	-	-	-
steelhead (parr)	2015	820	3.9	8,450	-	-	-
steelhead (parr)	2016	763	3.7	7,858	-	-	-
steelhead (parr)	2017	1,191	6.3	12,271	-	-	-

Table 4.4 Summary of day electrofishing results at four one-shoreline sites in the Coquitlam River in 2017. Mean fish density estimates are also shown for 2006-2017. The electrofishing survey was conducted at the same four sites during 2007-2017, whereas in 2006 electrofishing was conducted at 10 shoreline sites located within the annual snorkeling index sites (Decker et al. 2007).

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				
2016	1.95	5	2	1	8	6	10	7	851
2016	2.50	3	4	3	20	0	73	15	2,326
2016	3.20	15	7	5	30	22	38	23	3,774
2016	6.00	21	8	6	38	31	45	28	4,497
2006	all sites							10	591
2007	all sites							3	211
2008	all sites							1	90
2009	all sites							8	606
2010	all sites							3	200
2011	all sites							13	1072
2012	all sites							7	1073
2013	all sites							20	2759
2014	all sites							28	4011
2015	all sites							16	2263
2016	all sites							18	2862
					Steelhead fry				
2016	1.95	17	8	2	28	25	31	23	1,489
2016	2.50	11	6		17	16	18	13	988
2016	3.20	15	7	1	23	21	25	18	1,447
2016	6.00	13	10	6	38	18	58	28	2,249
2006	all sites							50	3,055
2007	all sites							27	2,154
2008	all sites							31	2,224
2009	all sites							20	1,530
2010	all sites							25	1,648
2011	all sites							51	4,179
2012	all sites							23	1,704
2013	all sites							36	2,418
2014	all sites							34	2,364
2015	all sites							22	1,507
2016	all sites							20	1,543
					Steelhead parr (1+)				
2016	1.95	3	3	1	7	4	10	5.7	372
2016	2.50	1	1	2	8	0	50	6.1	465
2016	3.20	0	0	0	0	-	-	0.0	0
2016	6.00	0	0	0	0	-	-	0.0	0
2006	all sites							3.4	206
2007	all sites							11.0	891
2008	all sites							6.8	493
2009	all sites							6.7	505
2010	all sites							2.7	200
2011	all sites							5.4	425
2012	all sites							2.8	211
2013	all sites							4.9	344
2014	all sites							4.9	347
2015	all sites							6.6	460
2016	all sites							2.9	209

Table 4.4 cont'd

Year	Site	Pass 1	Pass 2	Pass 3	Population estimate	Lower 95% CI	Upper 95% CI	Mean density (fish/100m^2)	fish/km
				Steelhead parr (2+)					
2006	all sites							0.3	21
2007	all sites							0.0	0
2008	all sites							0.4	30
2009	all sites							0.0	0
2010	all sites							0.0	0
2011	all sites							0.0	0
2012	all sites							0.0	0
2013	all sites							0.0	0
2014	all sites							0.0	0
2015	all sites							0.0	0
2016	all sites							0.0	0

Table 4.5 Average monthly discharge (cms) in Coquitlam River at Port Coquitlam during Steelhead spawning period in 2006 – 2017 (Water Survey of Canada station 08MH002).

	Discharge (m^3/sec)		
Year	June	July	Aug
2006	4.55	2.02	1.60
2007	5.54	6.41	2.48
2008	5.40	2.94	4.15
2009	5.91	4.58	6.18
2010	12.78	2.52	3.26
2011	6.58	5.43	4.14
2012	7.11	4.16	3.44
2013	4.60	1.99	3.14
2014	2.85	2.20	2.75
2015	1.85	1.53	2.72
2016	3.61	2.43	2.95
2017	6.22	2.64	3.20

Table 4.6 Comparison of backpack electroshocking and night snorkeling fish density estimates (fish/km) for juvenile Coho and Steelhead in the Coquitlam River including: R^2 and the mean, minimum and maximum ratio of estimates based on electroshocking to snorkeling 2006-2017.

Species	Age Class	N	R^2	EF:SN Min	EF:SN Max
Coho	0+	12	0.04	0.9	41.4
Steelhead	0+	12	0.11	0.4	2.0
Steelhead	1+	12	0.10	1.1	4.86

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
$N_{i,g}$	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

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

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

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

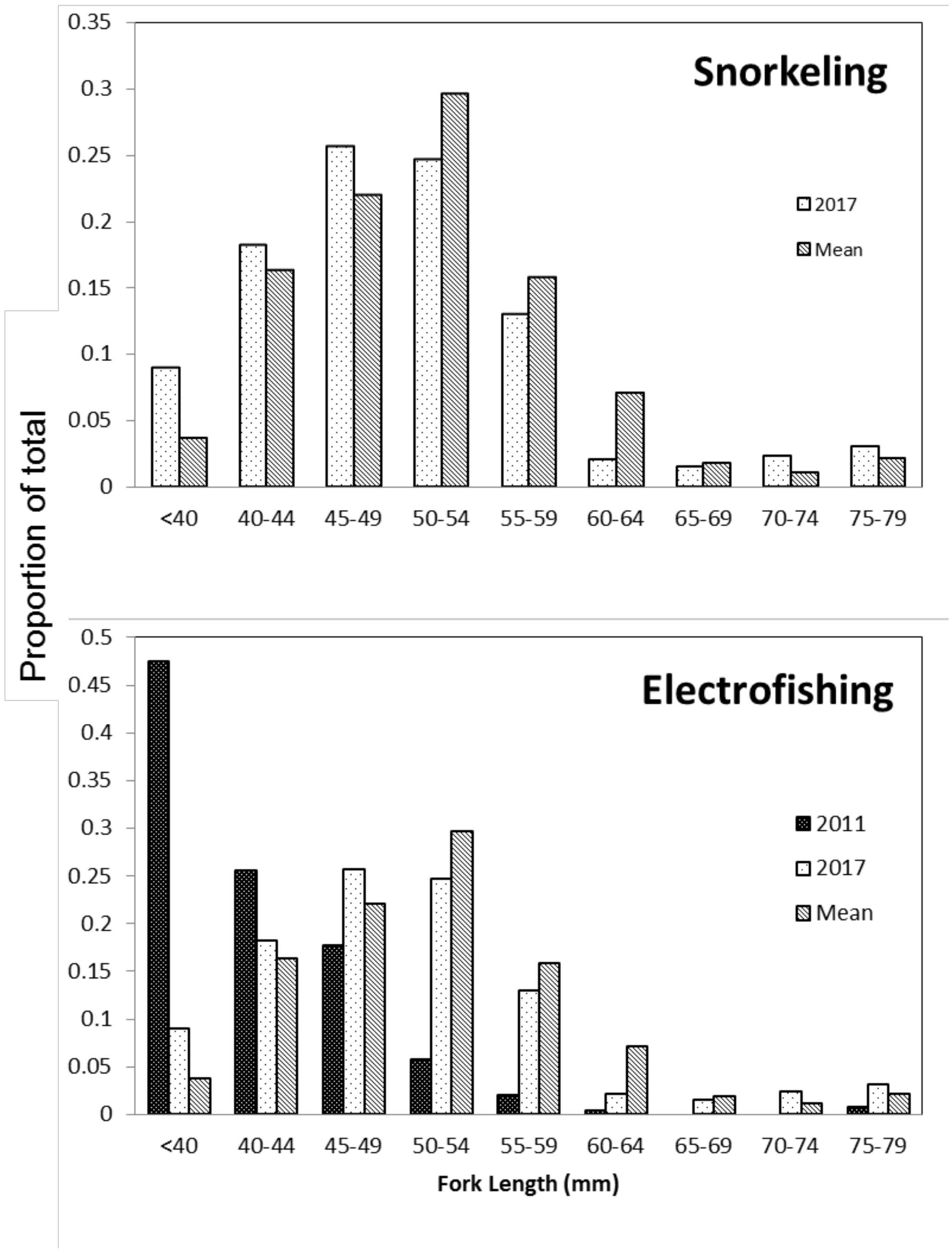
(4.13)
$$\tau_{\lambda} = \sigma_{\lambda}^{-2}$$

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 2006-2012.

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 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 2008-2016 and 2017 (data pooled for all sites). 2011 is also shown for electroshocking as an example of a year with considerable shift towards small sized fry.



9.5 Figures, Tables and Appendices for Chapter 5

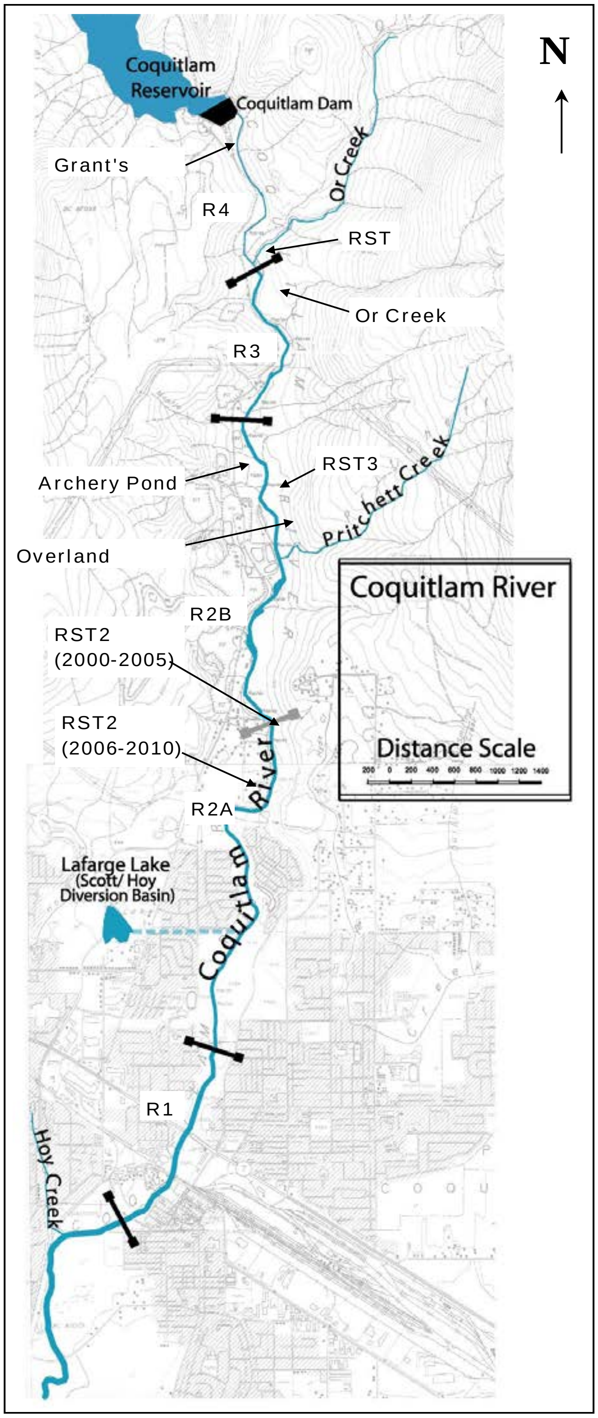


Figure 5.1 Map of the Coquitlam River showing constructed off-channel habitat sites, mainstem reach breaks and the locations of mainstem rotary screw traps (RSTs).

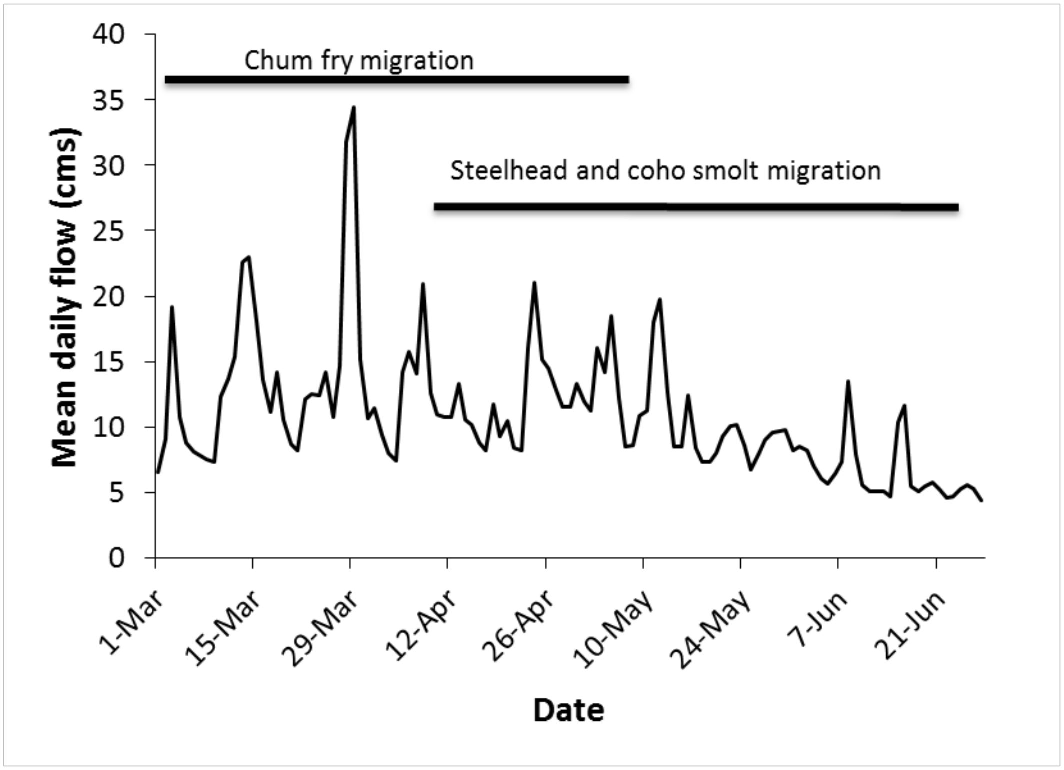


Figure 5.2 Mean daily flows in Coquitlam River at Port Coquitlam during the smolt trapping period in 2017. (Water Survey of Canada, stn. 08MH141). Approximate start times of Chum fry and Steelhead and Coho smolt migration based on captures at all trapping locations.

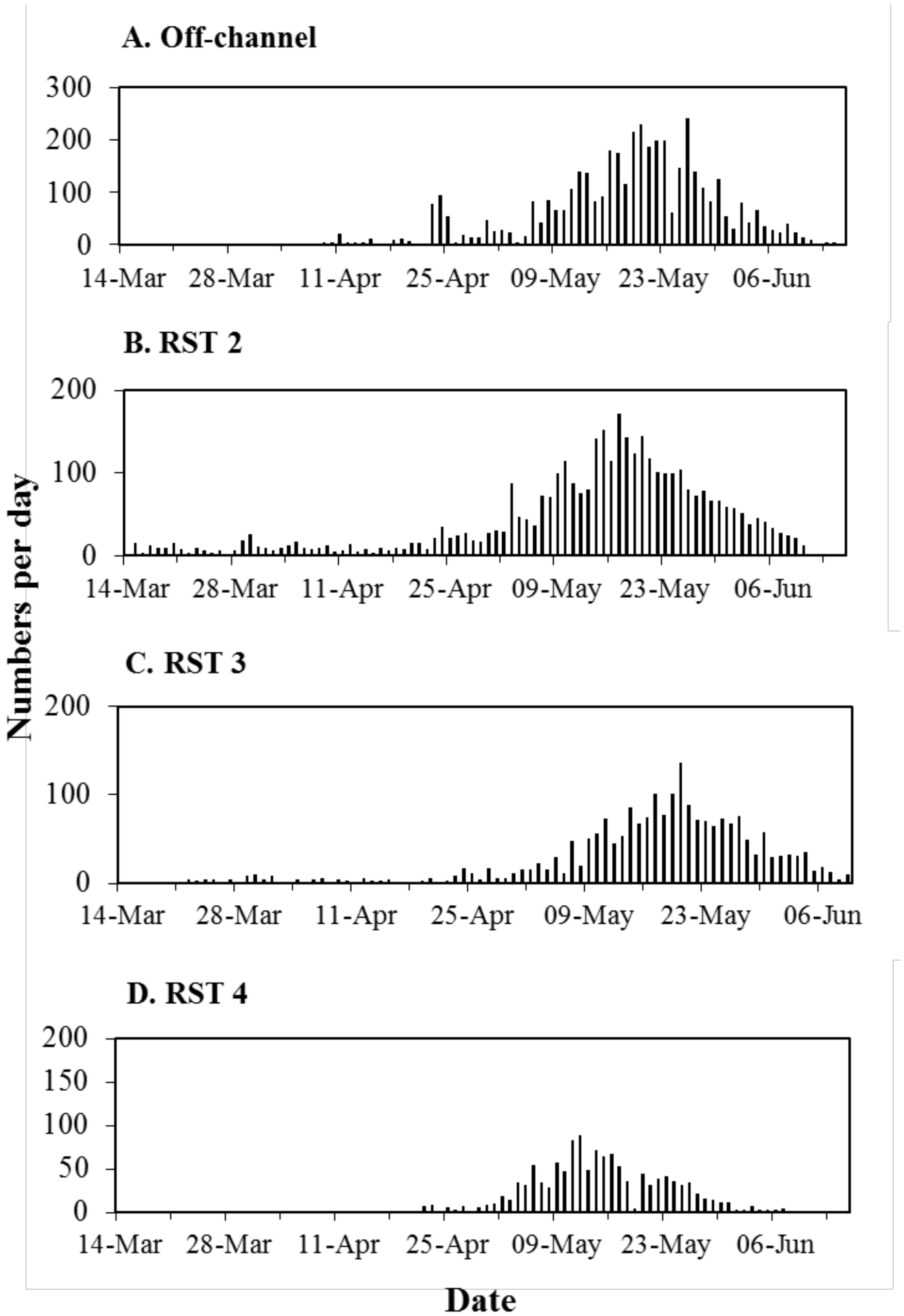


Figure 5.3 Daily catches of Coho smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2017. See Table 5.1 for start and end dates for individual trapping sites.

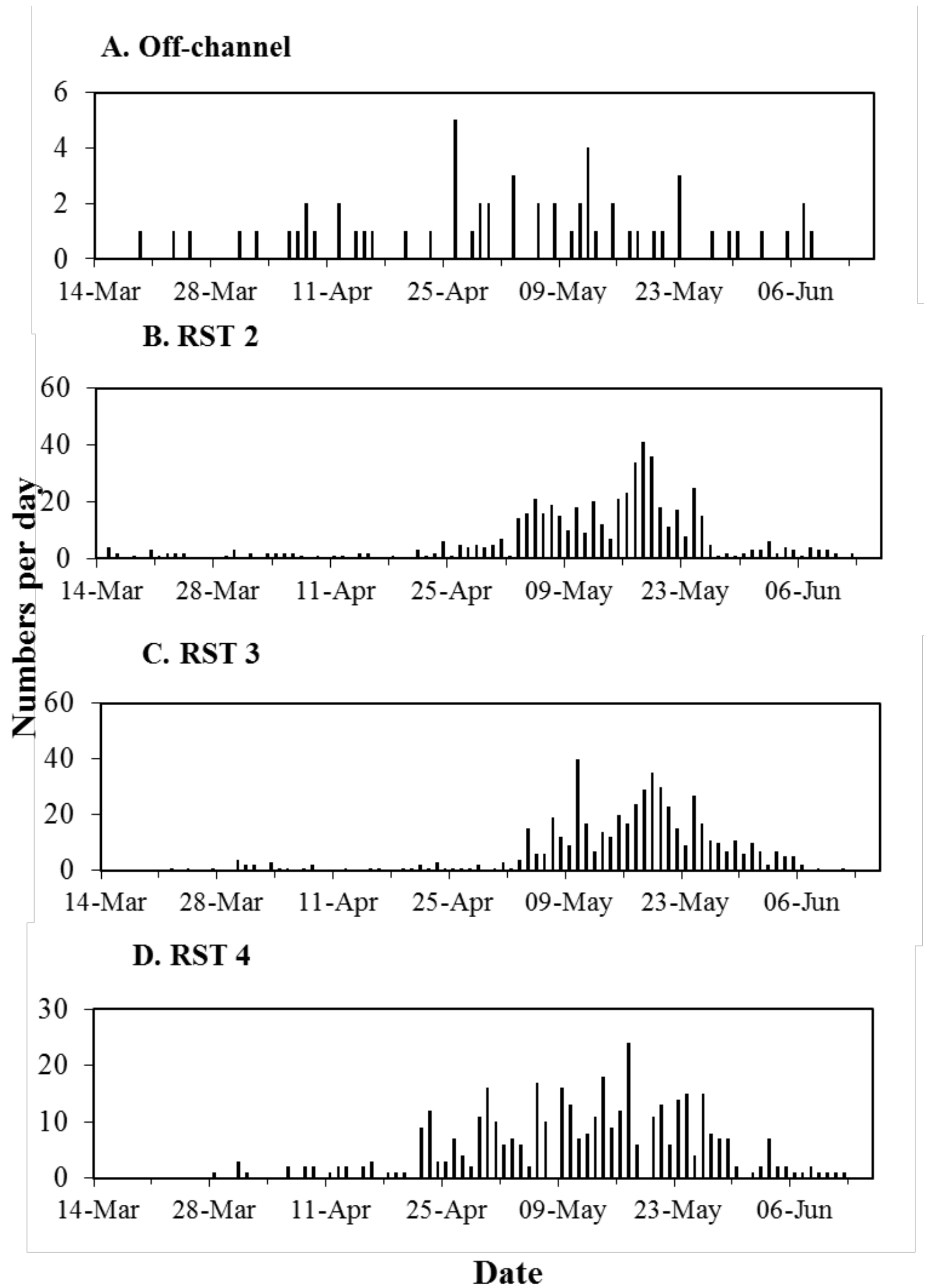


Figure 5.4 Daily catches of Steelhead smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2017. See Table 5.1 for start and end dates for individual trapping sites.

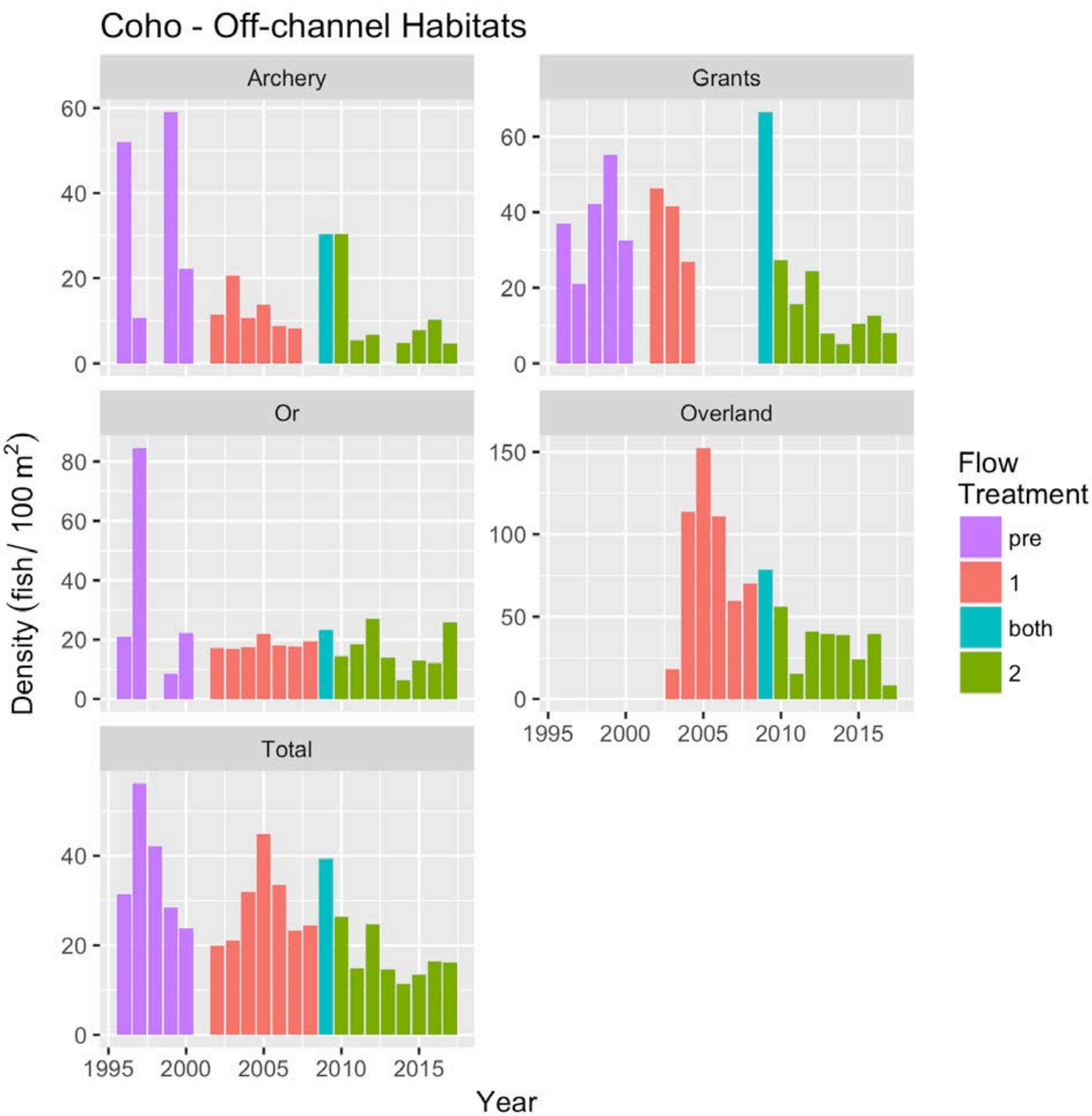


Figure 5.5a Areal of Coho smolts density (smolts/100m^2) in four constructed off-channel habitats along the Coquitlam River and for all four combined (Total) previous to Treatment 1 (1996-2000), Treatment 1 (2002-2008), when smolt cohorts reared under both treatments (2009) and Treatment 2 (2010-2017). Years with zero fish represent those when the off-channel habitats were not in operation or were not monitored.

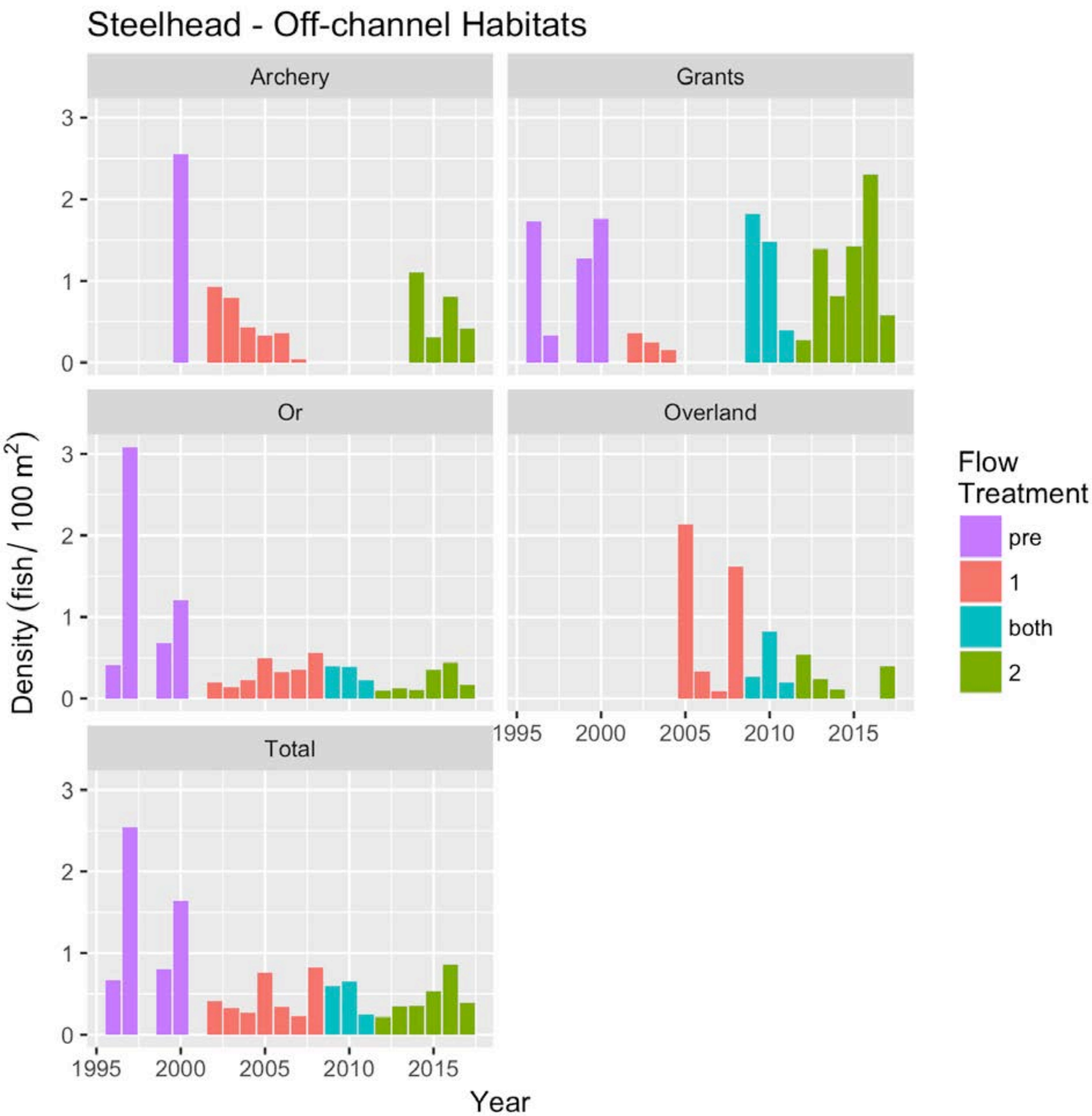


Figure 5.5b Areal density of Steelhead smolts (smolts/100m^2) in four constructed off-channel habitats along the Coquitlam River and for all four combined (Total) previous to Treatment 1 (1996-2000), Treatment 1 (2002-2008), when smolt cohorts reared under both treatments (2009) and Treatment 2 (2010-2017). Years with zero fish represent those when the off-channel habitats were not in operation or were not monitored.

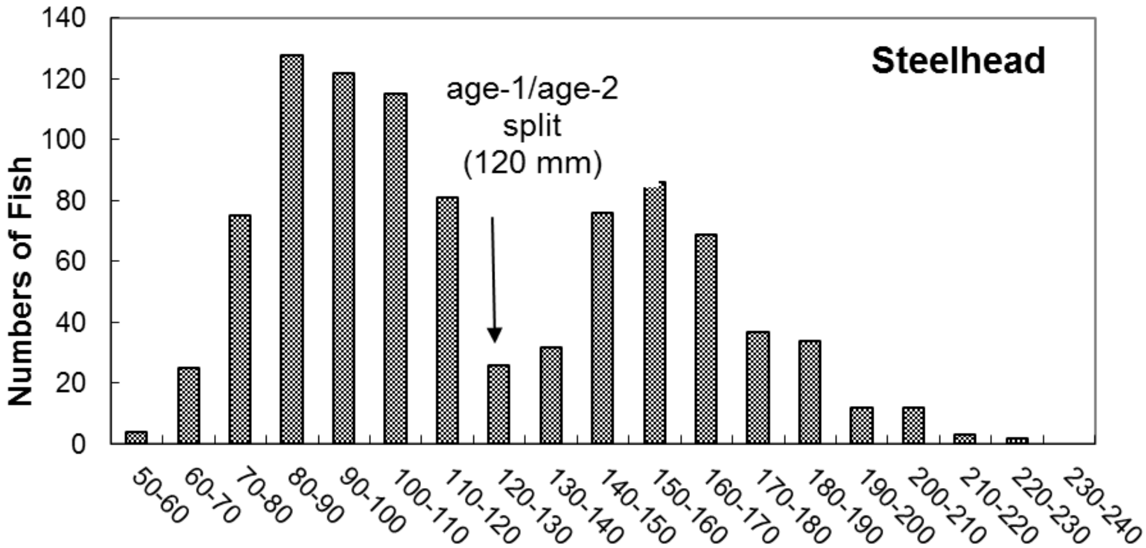


Figure 5.6 Length-frequency histogram for Steelhead captured in the Coquitlam River in 2017 (data pooled for all trap sites).

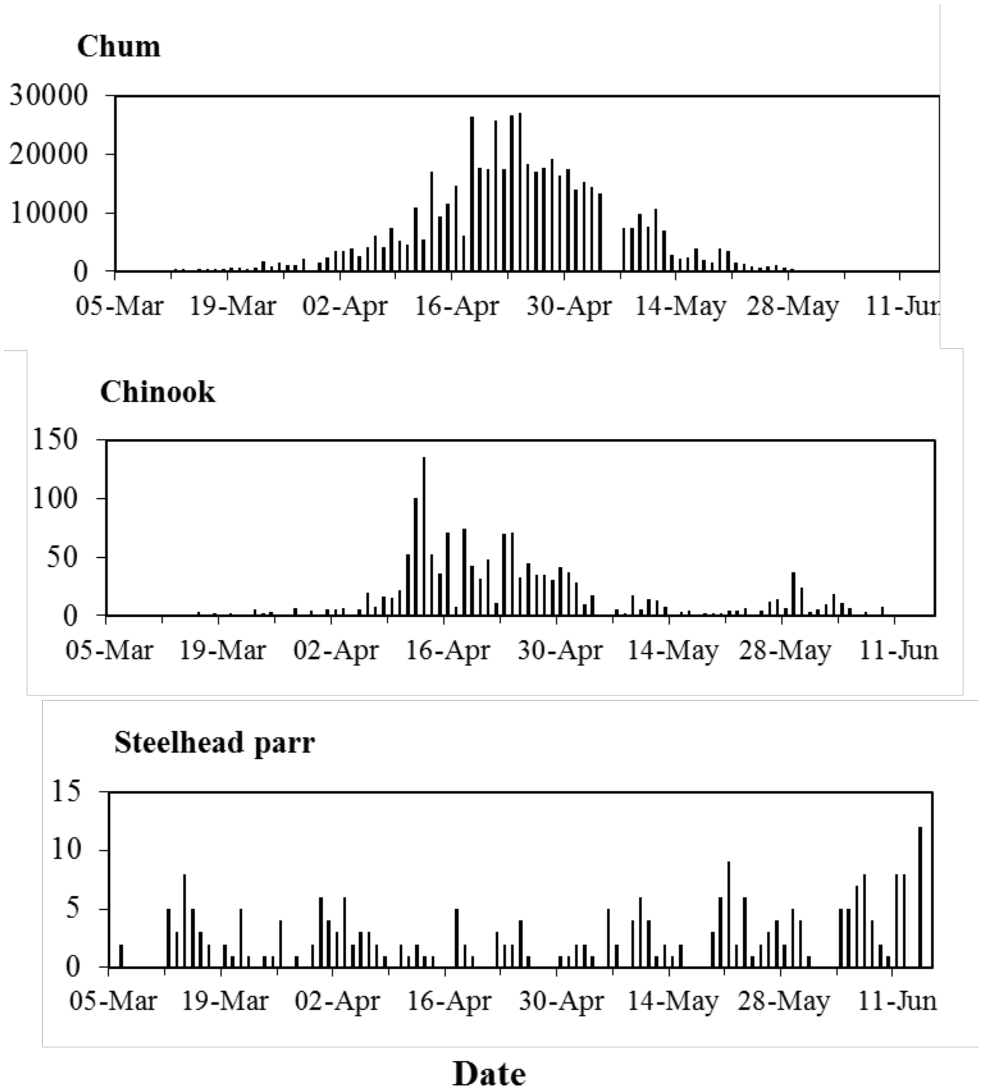


Figure 5.7 Daily catches of Chum fry, Chinook fry and smolts and Steelhead parr at the RST2 trapping site in reach 2 in the Coquitlam River in 2017. See Table 5.1 for start and end dates of downstream trapping.

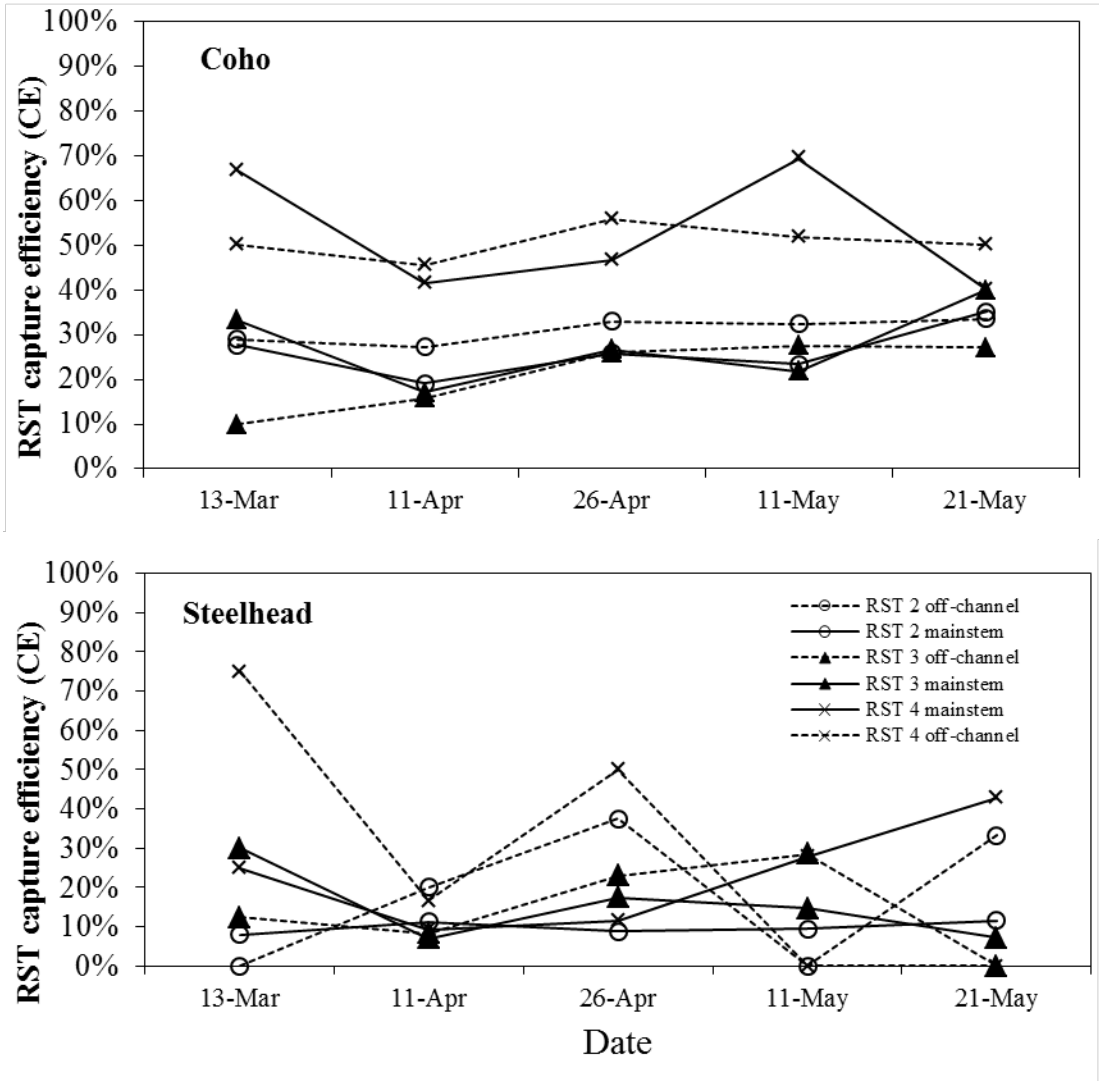


Figure 5.8 Estimated capture efficiencies (across six marking periods) at three rotary screw traps (RSTs) in the Coquitlam River for mark groups of Coho and Steelhead smolts from off-channel (dotted lines) and mainstem (solid lines) habitats in 2017. Dates on the horizontal axis indicate the start point for each marking period.

Table 5.1 Description of the stratification of fish marking by location and period for Coho and Steelhead smolts in the Coquitlam River in 2017. The start date for each temporal marking period at each RST trap site is also shown. Removal dates are also given.

Downstream RST trapping site	Mark type by location	Mark stratification by period 1	2	3	4	5	6	7	8	9	10	Traps removed
Reach 2 (RST2.2, chum)	mark E	4-2	4-9	4-14	4-19	4-23	4-28	5-3	5-10	5-14	5-21	6-12
Reach 2 (RST2.4, coho, steelhead)	mark E	3-30	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Reach 2 (RST2.5, coho, steelhead)	mark E	3-30	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Reach 3 (RST3, coho, steelhead)	mark D	3-30	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Reach 4 (RST4, coho, steelhead)	mark B	3-30	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Archery Pond	mark A	3-30	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Overland Ponds	mark A	3-30	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Or Creek Ponds	mark A	4-10	4-25	5-9	5-24	6-4	-	-	-	-		6-15
Grants Tomb Pond	mark A	4-1	4-25	5-9	5-24	6-4	-	-	-	-		6-15

Table 5.2 Summary of estimated smolt numbers and densities by species in 2017 for four off-channel sites, reaches 2-4 of the Coquitlam River mainstem and the total Coquitlam River mainstem including and excluding the off-channel sites. Note that only captures of Chinook juveniles are reported since there were too few to estimate population size.

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	267	-	-	8.1	-
Or Creek	-	13,336	3,431	-	-	25.7	-
Archery Pond	-	4,500	272	-	-	6.0	-
Overland Channel	-	8,700	370	-	-	4.3	-
Total Off-channel	-	21,136	4,340	-	-	20.5	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	4,148	1,257	30%	5.0	1,296
Reach 3, Coquitlam River	2.7	46,920	3,218	673	21%	6.9	1,192
Reach 4, Coquitlam River	1.6	19,200	2,012	161	8.0%	10.5	1,258
Total Mainstem	7.5	149,898	9,810	1,080	11%	6.5	1,308
Coquitlam R.incl. off-channel	7.5	171,034	14,150	1,080	8%	8.3	1,887
Steelhead							
Off-channel sites							
Grant's Tomb	-	3,300	19	-	-	0.6	-
Or Creek site	-	13,336	22	-	-	0.2	-
Archery Pond	-	4,500	24	-	-	0.5	-
Overland Channel	-	8,700	18	-	-	0.2	-
Total Off-channel	-	21,136	83	-	-	0.4	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	1,433	1,480	103%	1.7	448
Reach 3, Coquitlam River	2.7	46,920	1,182	1,538	130%	2.5	438
Reach 4, Coquitlam River	1.6	19,200	2,807	1,051	37%	14.6	1,754
Total Mainstem	7.5	149,898	5,142	939	18%	3.4	686
Coquitlam R.incl. off-channel	7.5	171,034	5,225	939	18%	3.1	697
Chinook							
Off-channel sites							
Grant's Tomb	-	3,300	0	-	-	-	-
Or Creek site	-	13,336	10	-	-	0.07	-
Archery Pond	-	5,800	7	-	-	0.12	-
Overland channel	-	4,500	0	-	-	-	-
Total Off-channel	-	23,636	17	-	-	0.072	-
Mainstem	191 fry and parr captured at rst 2.4 and 2.5 1507 fry captured at rst 2.2						
Chum							
Coquitlam R.incl. off-channel	7.5	171,034	12,742,642	2,173,848	17%	7,450	1,699,019

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 2017. 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	0.25	0.32	0.00
Coho	RST 3	0.25	0.25	0.68
Coho	RST 4	0.50	0.54	0.25
Steelhead	RST 2	0.09	0.19	0.02
Steelhead	RST 3	0.14	0.16	0.73
Steelhead	RST 4	0.17	0.40	0.02

Table 5.4 Mean monthly flows during Treatment 1 (2000-2008), Treatment 2(2009-2016) and 2017 in Coquitlam River at Port Coquitlam during the smolt and fry trapping period. (Water Survey of Canada, stn. 08MH141).

	Mean monthly flow (m^3s)		
Month	Treatment 1	Treatment 2	2017
March	8.5	11.8	13.7
April	6.4	8.5	12.0
May	6.5	7.6	10.7
June 1-15	5.8	5.1	7.1

Table 5.5 Percent of all juvenile Steelhead captures that were less than 120mm forklength at RST 2-4 in the Coquitlam River. 120mm forklength has been the minimum length to be considers smolts since 2012.

	Year					
Site	2012	2013	2014	2015	2016	2017
RST 2	27%	42%	33%	36%	15%	32%
RST 3	16%	26%	35%	28%	23%	21%
RST 4	11%	14%	7%	10%	8%	4%

Table 5.6 Estimated of the number of Coho and Steelhead smolts outmigrating from the lower Coquitlam River 1996-2017. 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 River Dam.

	Year																				
Site	1996	1997	1998	1999	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
	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
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
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
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
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
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
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
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
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
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
	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
Or Creek Ponds	55	411	-	91	161	26	19	30	66	44	47	74	53	52	30	13	17	14	47	59	22
Archery Pond	-	-	-	-	148	54	46	25	19	21	2	-	-	-	-	-	-	29	18	47	24
Overland Channel	-	-	-	-	-	-	-	-	96	15	4	73	12	37	9	24	11	5	0	0	18
Total	112	422	-	133	367	92	73	60	181	80	53	147	125	138	52	46	74	75	112	182	83
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
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
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
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

Table 5.7 Captures and mortality rate of wild and hatchery Sockeye/Kokanee during 2017 at RST 2-4.

	Origin			
	Hatchery		Wild	
Site	Catch	% mortalities	Catch	% mortalities
RST2	32	0%	12	25%
RST3	21	0%	18	11%
RST4	21	5%	64	59%

Appendix 5.1 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 2017. 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), estimated capture efficiencies (R/M), 95% confidence intervals, and relative percent errors.

Species	Site	Mark group(s)	M	R	U	Capture efficiency	N smolts	CI (+/-)	CI (%)
Coho	RST 2	mainstem	1,712	427	1,915	25%	9,810	1,080	11.0%
	RST 3	all	4,541	1,145	1,037	25%	5,653	643	11.4%
	RST 4	all	989	504	926	51%	2,012	161	8.0%
Steelhead	RST 2	mainstem	1,121	106	439	9%	5,142	939	18.3%
	RST 3	all	344	50	489	15%	3,802	1,233	32.4%
	RST 4	mainstem	301	50	360	17%	2,807	1,051	37.4%
Chum	RST 2	RST 2	18,801	1,202	541,444	6%	12,742,642	2,173,848	17.1%

Appendix 5.2 Summary marking and recovery strata pooling used to compute maximum likelihood population estimates for three species at mainstem trapping sites in the Coquitlam River in 2017.

		Pooling		
Site	Mark group	Release	Recovery	
Coho				
RST 2	mainstem (RST 3-4)	none	none	Darroch ML
RST 3	all (RST 4)	none	pooled 1-2	Darroch ML
RST 4	all (RST 4)	none	pooled 1-2	Darroch ML
Steelhead				
RST 2	mainstem (RST 3-4)	none	none	Darroch ML
RST 3	all (RST 4)	none	pooled 1-2	Darroch ML
RST 4	mainstem (RST 4)	none	pooled 1-2	Darroch ML
Chum				
RST 2	RST 2	dropped 5	pooled 5-6	Darroch ML

Appendix 5.3 Mark-recapture data for Coho, Steelhead and Chum at three rotary screw trap sites (RST2, RST3, RST4) in the Coquitlam River mainstem in 2017. Tables include numbers of smolts marked and released, numbers of marked and unmarked smolts recovered, and percentages of marked smolts recovered (capture efficiency) by marking period.

Coho

Recovery site: RST 2

Mainstem mark group		Recovery strata					
Release strata	Marks	1	2	3	4	5	Capture efficiency
1	101	24	1	1	1	1	28%
2	261	0	27	21	2	0	19%
3	965	0	0	211	38	1	26%
4	311	0	0	0	61	12	23%
5	74	0	0	0	0	26	35%
Untagged Fish		214	379	913	273	136	

Recovery site: RST 3

All mark groups		Recovery strata					
Release strata	Marks	1	2	3	4	5	Capture efficiency
1	42	2	2	2	1	0	-
2	572	0	49	38	5	0	17%
3	2334	0	0	519	88	2	27%
4	1352	0	0	0	332	34	22%
5	283	0	0	0	0	78	40%
Untagged Fish		92	157	518	198	72	

Recovery site: RST 4

All mark groups		Recovery strata					
Release strata	Marks	1	2	3	4	5	Capture efficiency
1	20	3	2	5	2	0	-
2	134	0	30	23	3	0	41%
3	633	0	0	260	51	0	47%
4	180	0	0	0	108	7	69%
5	22	0	0	0	0	10	40%
Untagged Fish		15	144	586	163	18	

Appendix 5.3 continued

Steelhead

Recovery site: RST 2
Mainstem mark group

Release strata	Marks	Recovery strata 1	2	3	4	5	Capture efficiency
1	62	5	0	0	0	0	7%
2	204	0	12	11	0	0	12%
3	593	0	0	44	8	0	10%
4	210	0	0	0	19	1	9%
5	52	0	0	0	0	6	13%
Untagged Fish		22	105	236	52	24	

Recovery site: RST 3
All mark groups

Release strata	Marks	Recovery strata 1	2	3	4	5	Capture efficiency
1	28	2	0	3	2	0	25%
2	98	0	5	2	0	0	7%
3	133	0	0	22	2	0	18%
4	68	0	0	0	11	0	16%
5	17	0	0	0	0	1	6%
Untagged Fish		23	48	282	113	23	

Recovery site: RST 4
All mark groups

Release strata	Marks	Recovery strata 1	2	3	4	5	Capture efficiency
1	24	3	1	1	2	1	33%
2	92	0	6	3	0	0	10%
3	124	0	0	10	6	0	13%
4	62	0	0	0	17	0	27%
5	14	0	0	0	0	6	43%
Untagged Fish		23	99	150	70	18	

Appendix 5.3 continued

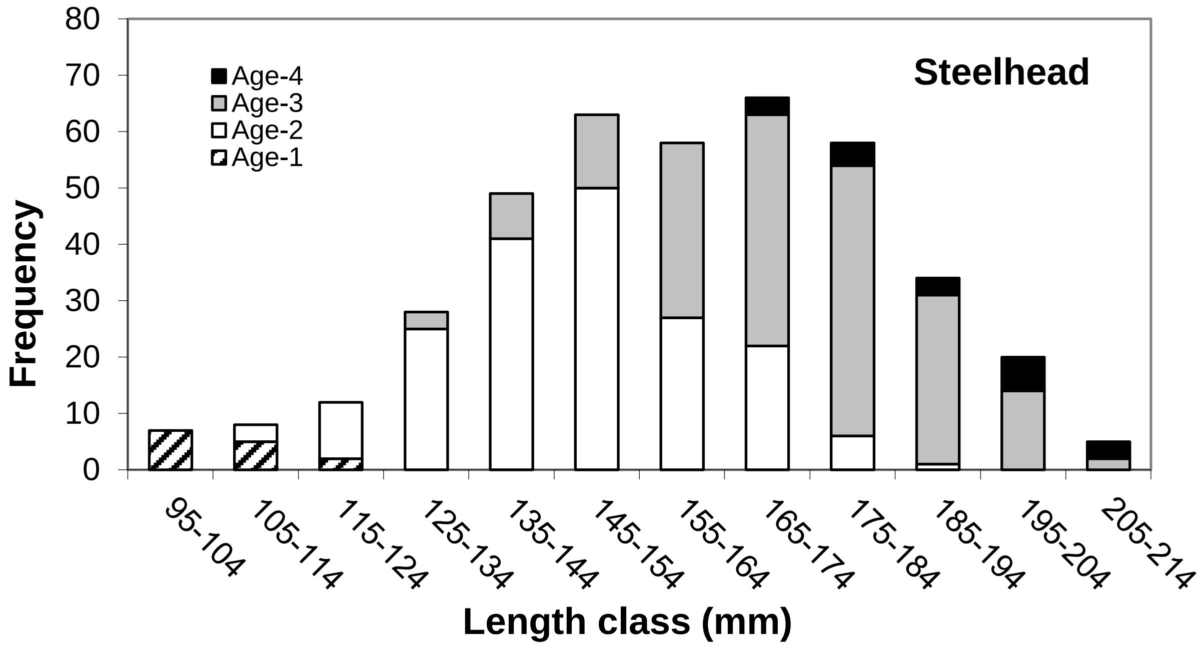
Chum

Recovery site: RST 2.2

All mark groups Recovery strata

Release strata	Marks	1	2	3	4	5	6	7	8	9	10	Capture efficiency
1	1997	219	0	0	0	0	0	0	0	0	0	11.0%
2	1888	0	175	0	0	0	0	0	0	0	0	9.3%
3	1996	0	0	120	0	0	0	0	0	0	0	6.0%
4	1986	0	0	0	174	0	0	0	0	0	0	8.8%
5	1997	0	0	0	0	0	0	0	0	0	0	0.0%
6	1969	0	0	0	0	0	101	0	0	0	0	5.1%
7	2002	0	0	0	0	0	0	27	0	0	0	1.3%
8	1994	0	0	0	0	0	0	0	140	0	0	7.0%
9	1995	0	0	0	0	0	0	0	0	205	0	10.3%
10	1995	0	0	0	0	0	0	0	0	0	41	2.1%
Untagged Fish		31,734	43,180	67,891	78,271	106,852	82,447	73,478	28,241	19,562	9788	

Appendix 5.4 Age-forklength relationships for Steelhead parr and smolts in the Coquitlam River during 2005-2017 derived from scale-aging analysis.



9.6 Figures and Tables for Chapter 6

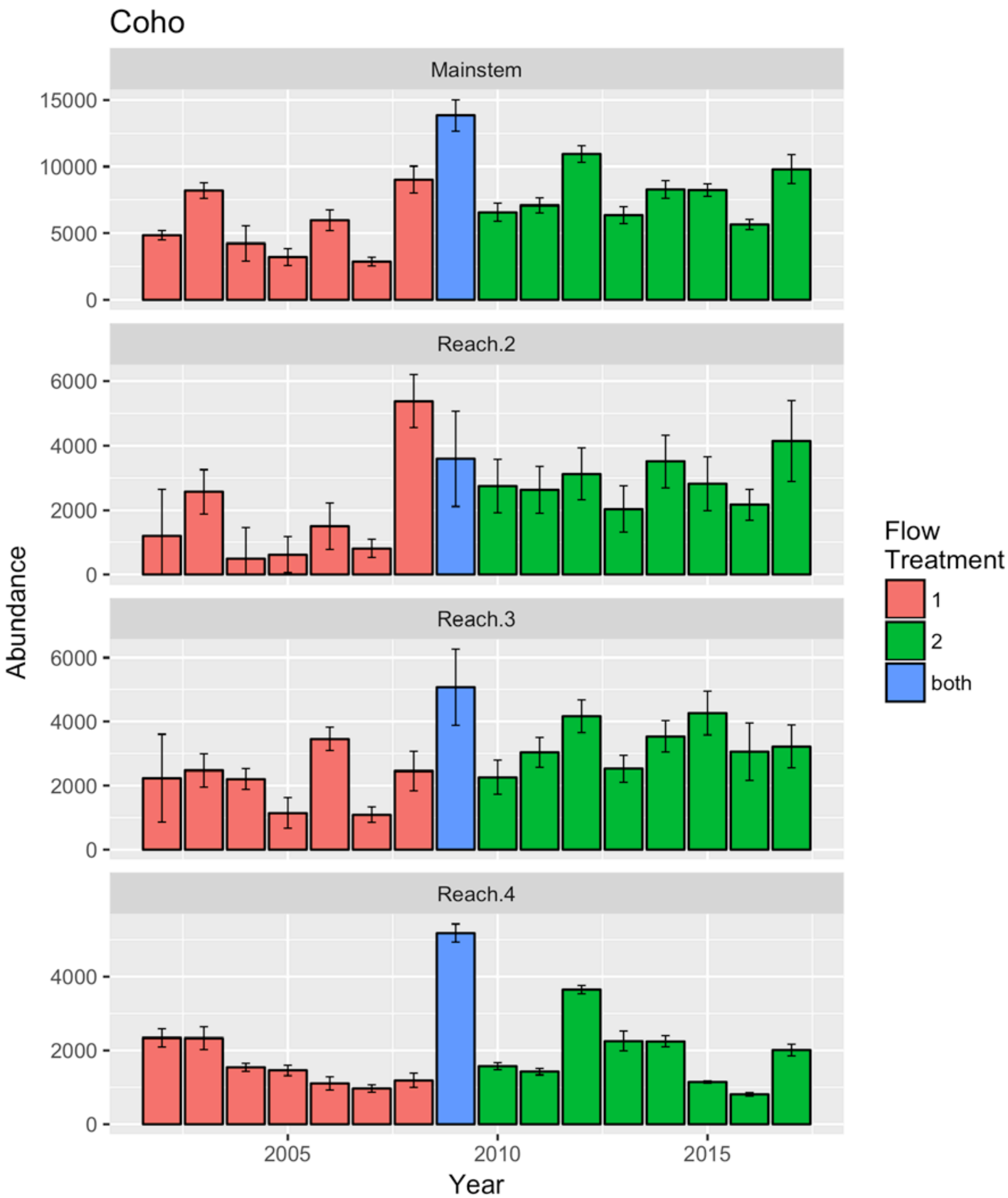


Figure 6.1 Annual Coho smolts yield and 95% confidence intervals for the 7.5km section of the Coquitlam river mainstem as well from individual reaches 2-4. The colours of the bars reflect the flow treatment period of the cohort outmigrating during that year: red – Treatment 1, blue – both treatment conditions, green – Treatment 2.

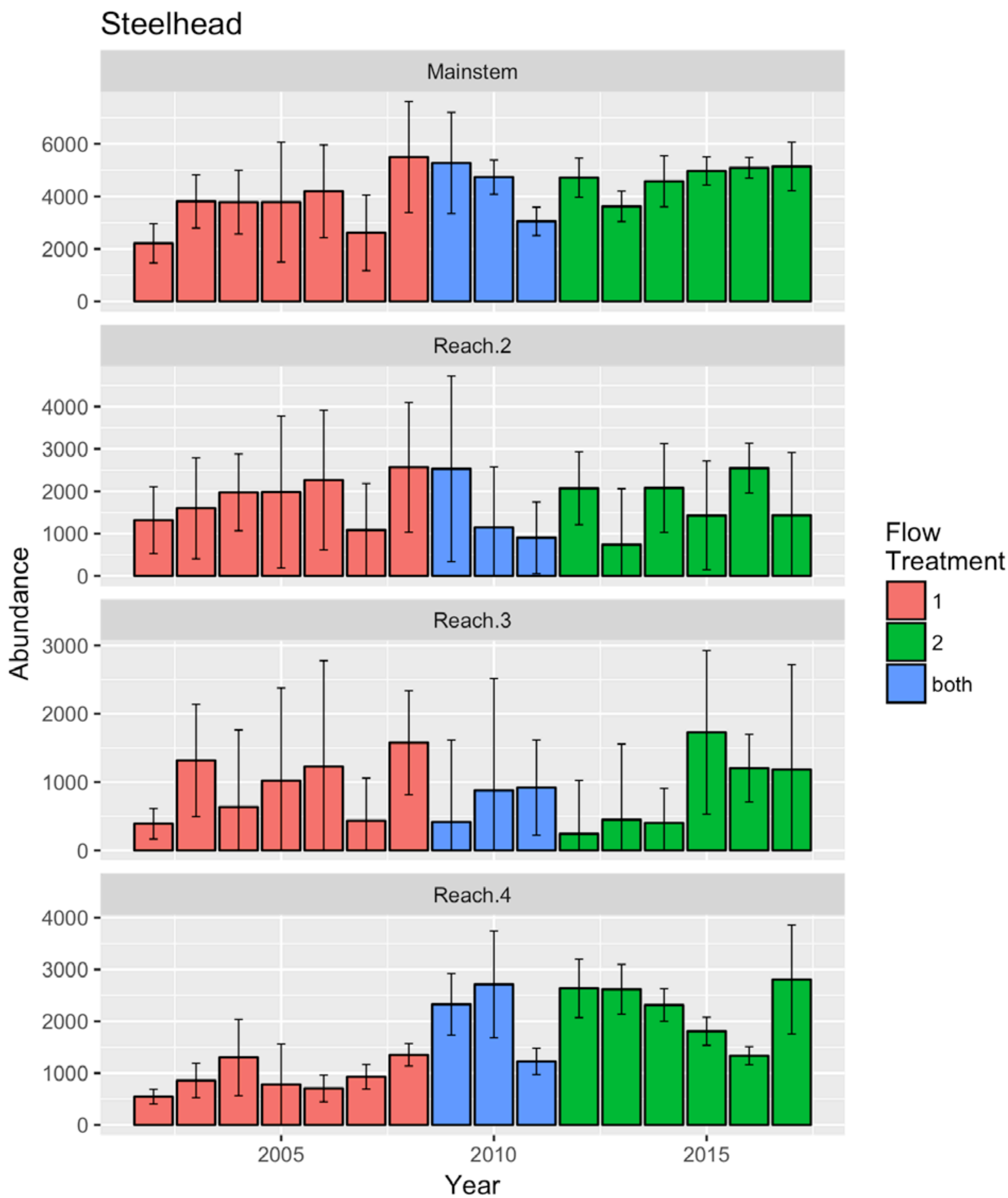


Figure 6.2 Annual Steelhead smolts yield and 95% confidence intervals for the 7.5km section of the Coquitlam river mainstem as well from individual reaches 2-4. The colours of the bars reflect the flow treatment period of the cohort outmigrating during that year: red – Treatment 1, blue – both treatment conditions, green – Treatment 2.

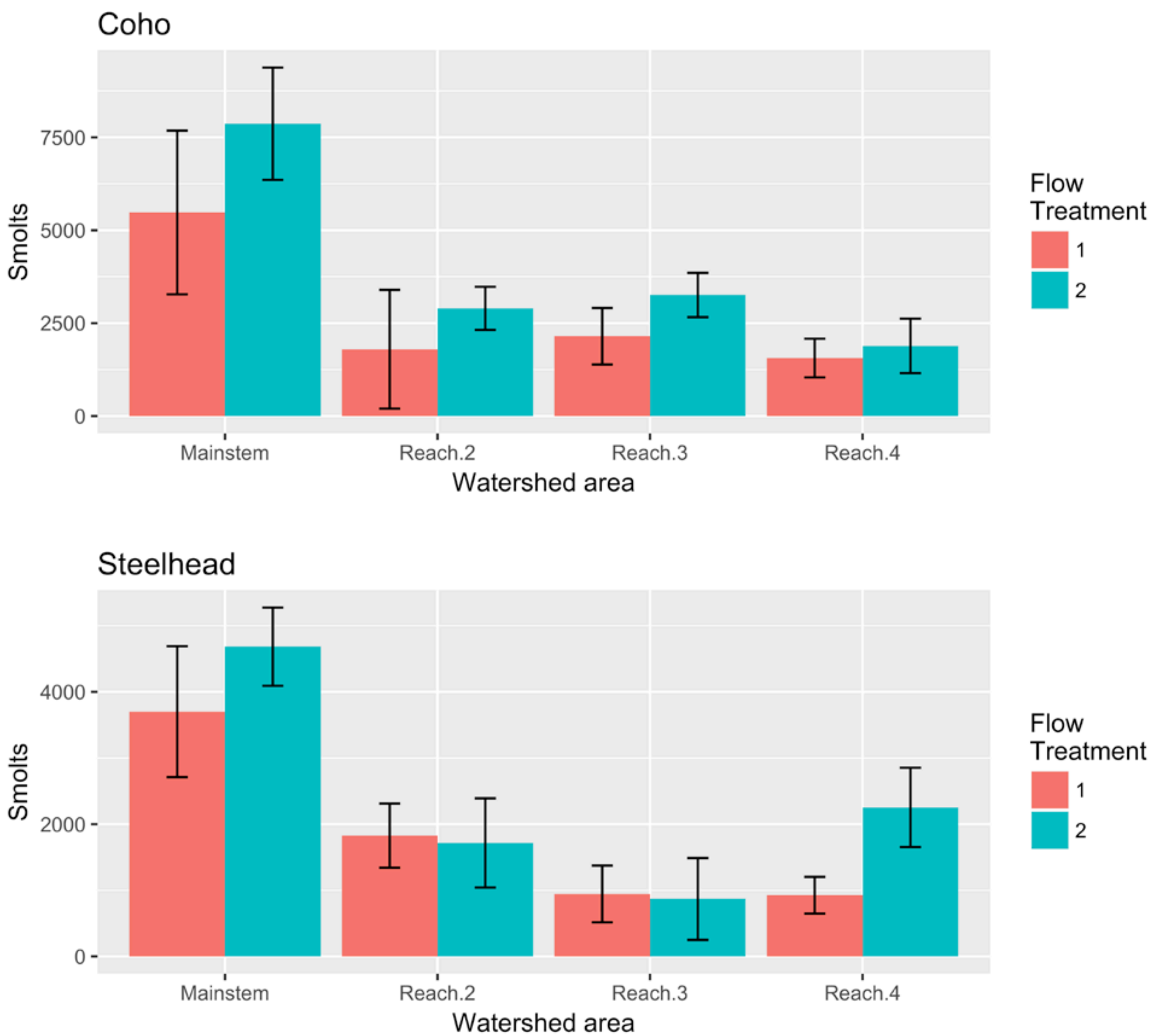


Figure 6.3 Mean Coho and Steelhead smolt yield and 95% confidence intervals for Treatment 1 and Treatment 2 in the 7.5 km of the Coquitlam River mainstem and for reaches 2-4. Only annual estimates for cohorts that reared exclusively under either Treatment 1 or Treatment 2 conditions were included. For Coho, this includes 2002-2008 for Treatment 1 and 2010-2017 for Treatment 2. For Steelhead, this includes 2002-2008 for Treatment 1 and 2012-2017 for Treatment 2.

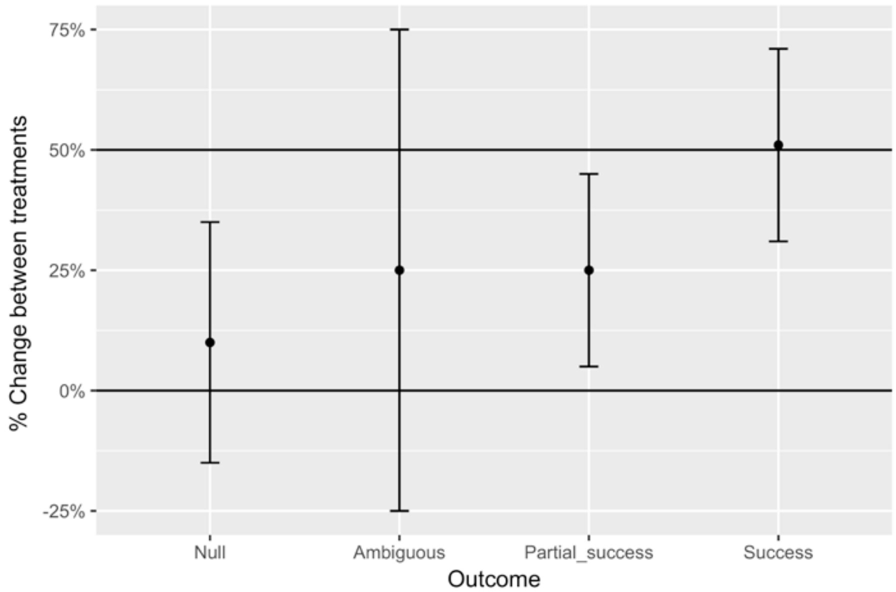


Figure 6.4. Example of how to categorize results based on the confidence intervals of the percent change between two treatments where the expected change was a 50% increase.

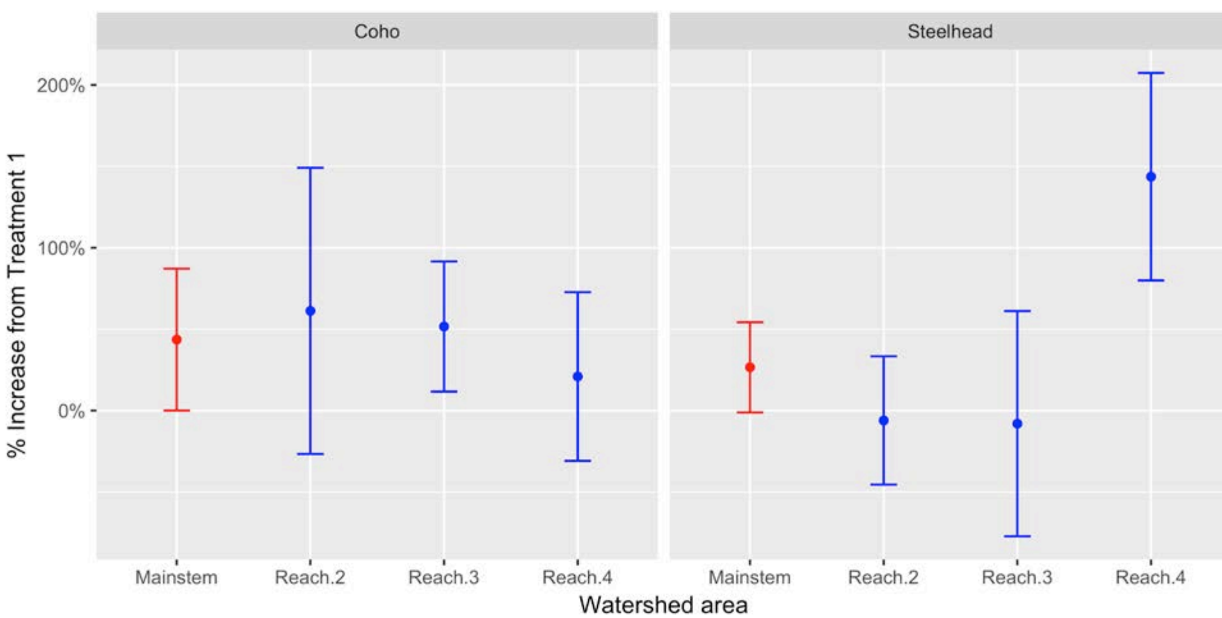


Figure 6.5 Average effects size and 95% confidence intervals of the change in smolt yield from Treatment 1 to 2 for Coho and Steelhead for 7.5km of the Coquitlam River mainstem (red) and for individual reaches 2-4 (blue). Effect size expressed as the relative change in yield from Treatment 1.

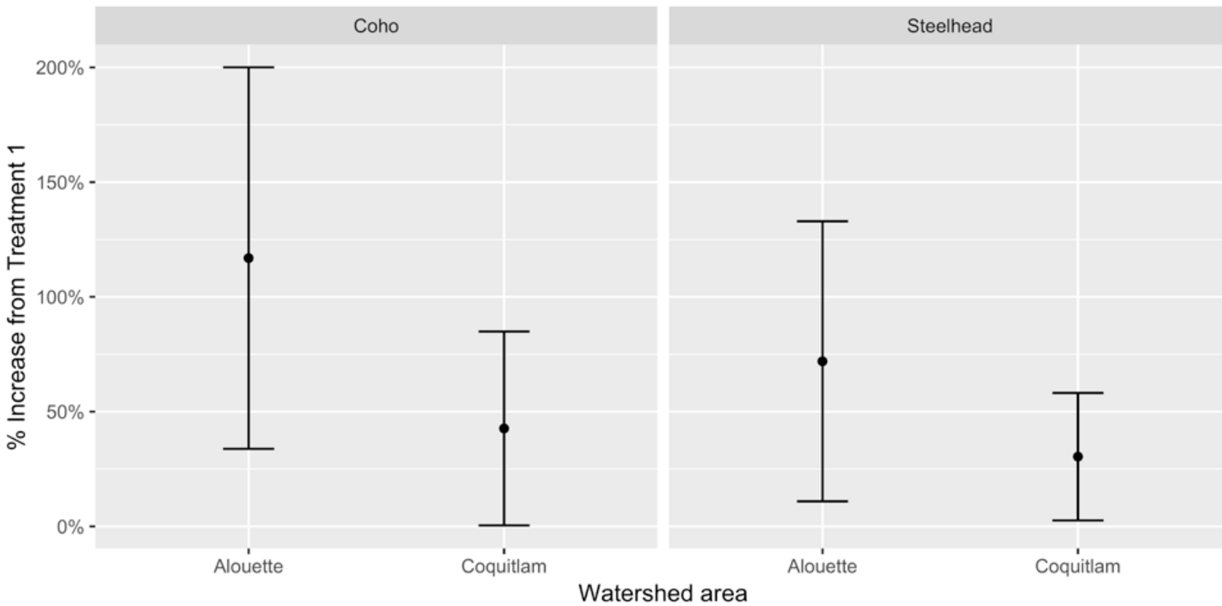


Figure 6.6 Average percent change in smolt yield from Treatment 1 to 2 and 95% confidence intervals for cohorts that reared entirely during Treatment 1(2002-2008) and Treatment 2 for Coho (2010-2014) and Steelhead (2012-2014) from mainstem habitats in the Coquitlam River an d Alouette River.

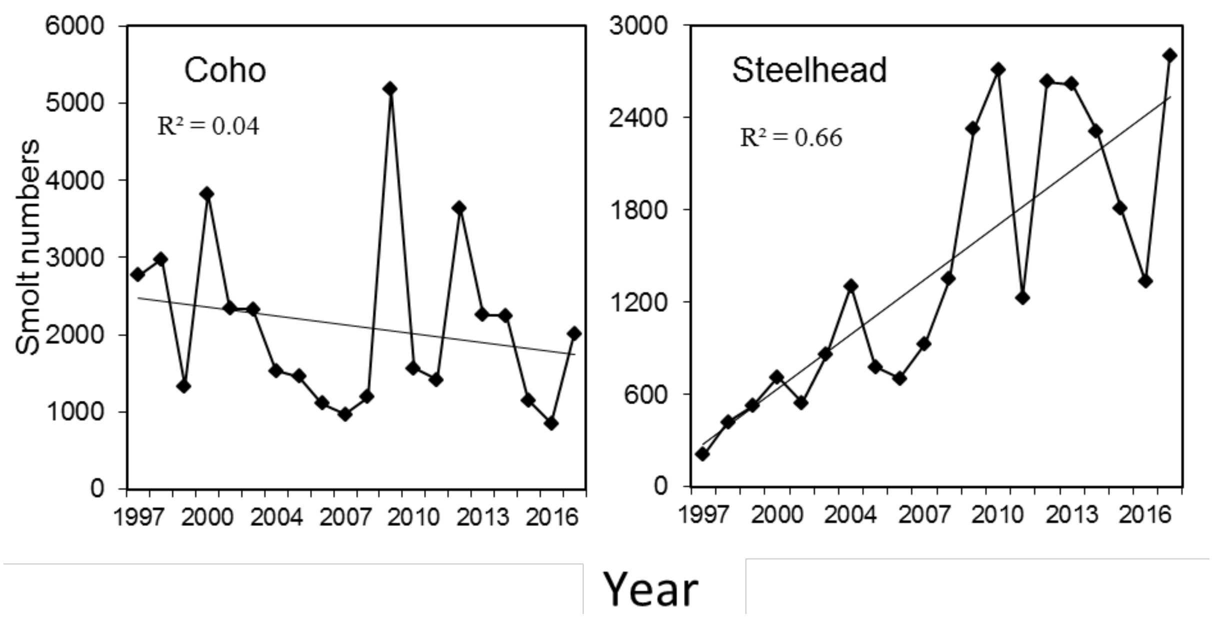


Figure 6.7 Annual numbers of Coho and Steelhead smolts in reach 4 of Coquitlam River during 1997-2017.

9. Figures, Tables and Appendices

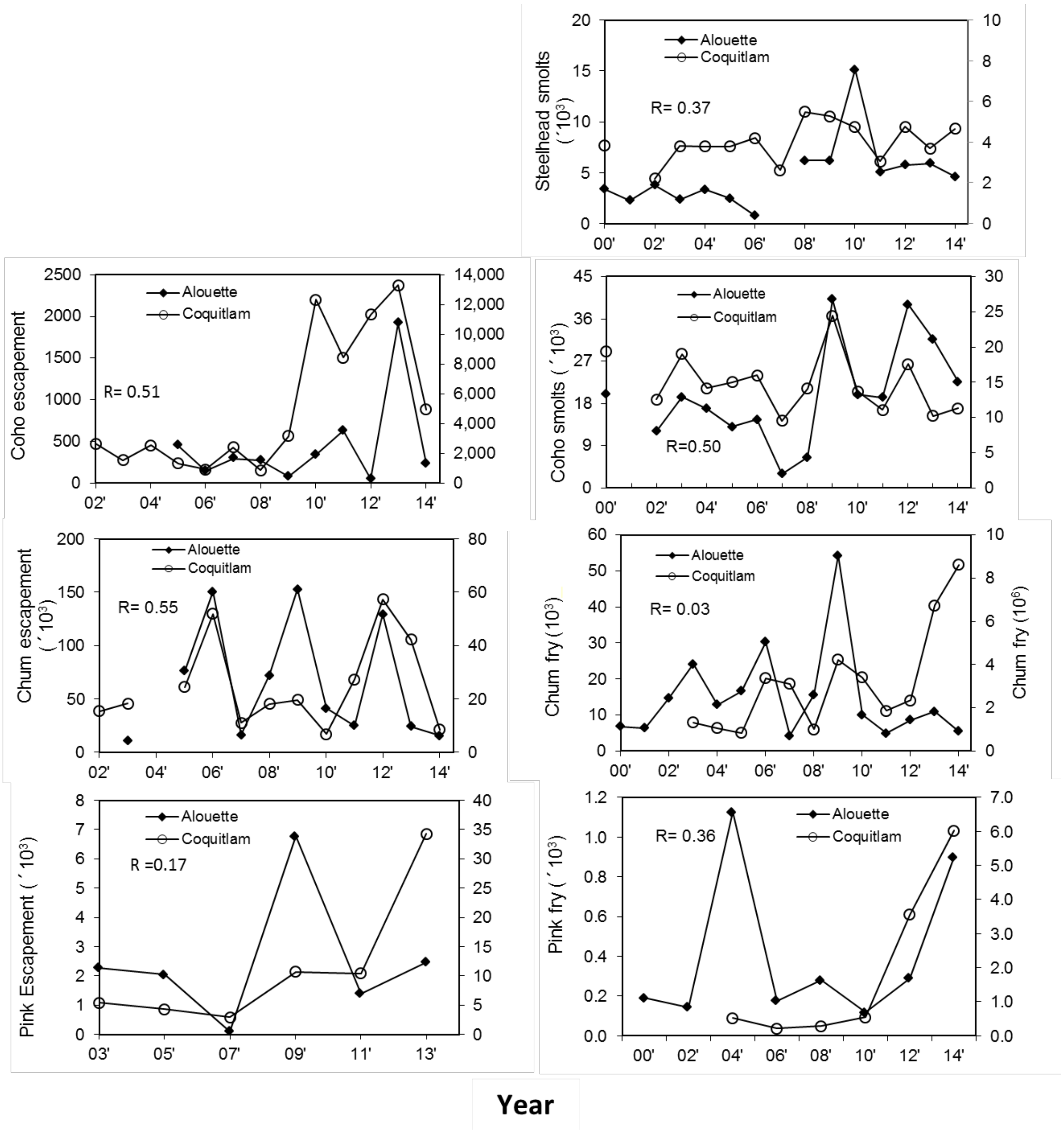


Figure 6.8 Scatterplots of escapement and smolt yield in the Coquitlam River versus that in the Alouette River during 2002-2014. Values for the Coquitlam are given on the right-hand axis, and values for the Alouette are given on the left-hand axis.

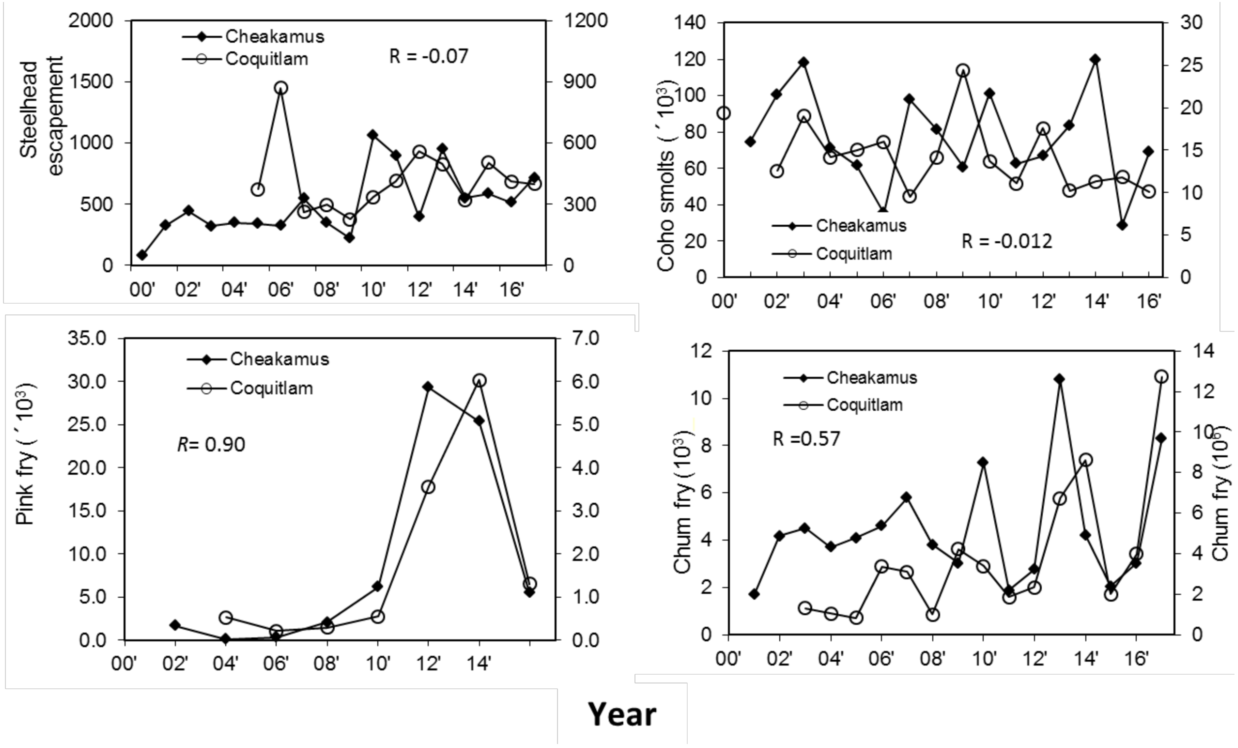


Figure 6.9 Scatterplots of escapement and smolt yield in the Coquitlam River versus that in the Cheakamus River during 2002-2016. Values for the Coquitlam are given on the right-hand axis, and values for the Cheakamus are given on the left-hand axis.

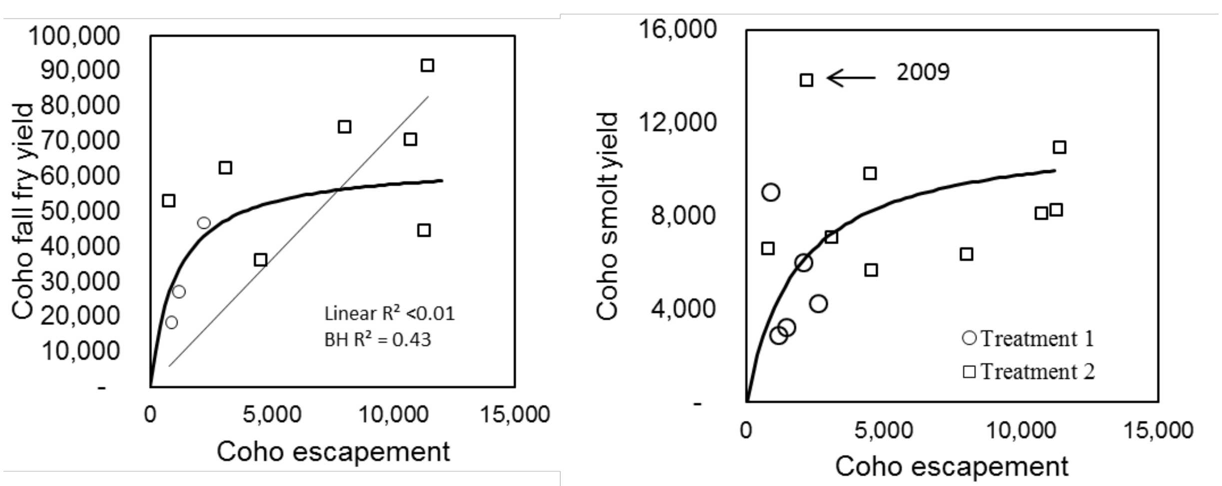


Figure 6.10 Preliminary linear and Beverton-Holt stock-recruitment relationship between Coho escapement and fall fry yield (2006-2017 fry years) and Beverton-Holt stock-recruitment relationship between Coho escapement (upstream of RST2) and total smolt yield in the Coquitlam River during Treatment 1 (2004-2008 smolt years), and during the first 7 years of Treatment 2 (2009-2017 smolt years).

9. Figures, Tables and Appendices

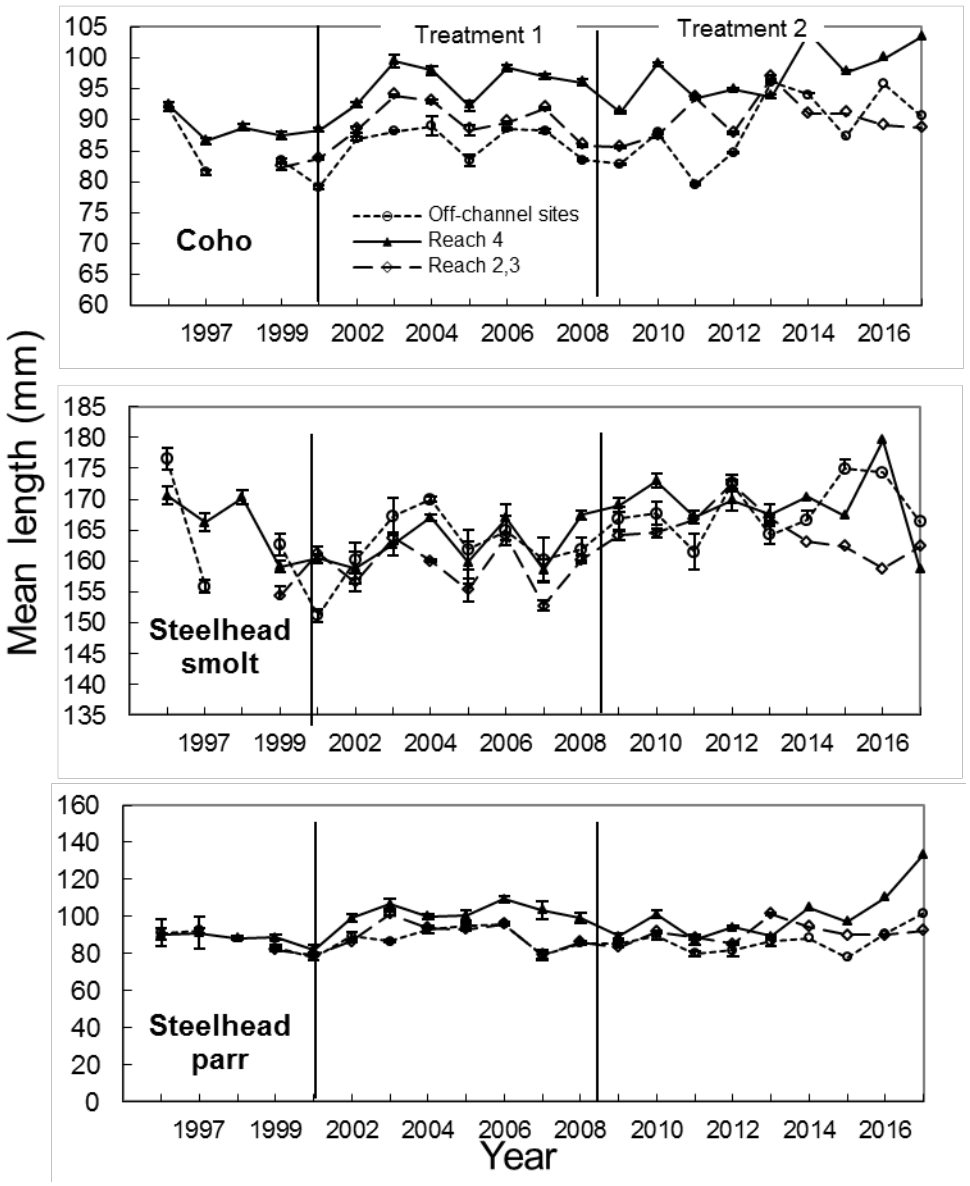


Figure 6.11 Mean annual forklengths for Coho smolts and Steelhead smolts (age 2+ and 3+ combined) and parr in different habitats in the Coquitlam River, 1996-2017. Error bars represent ± 1 standard error.

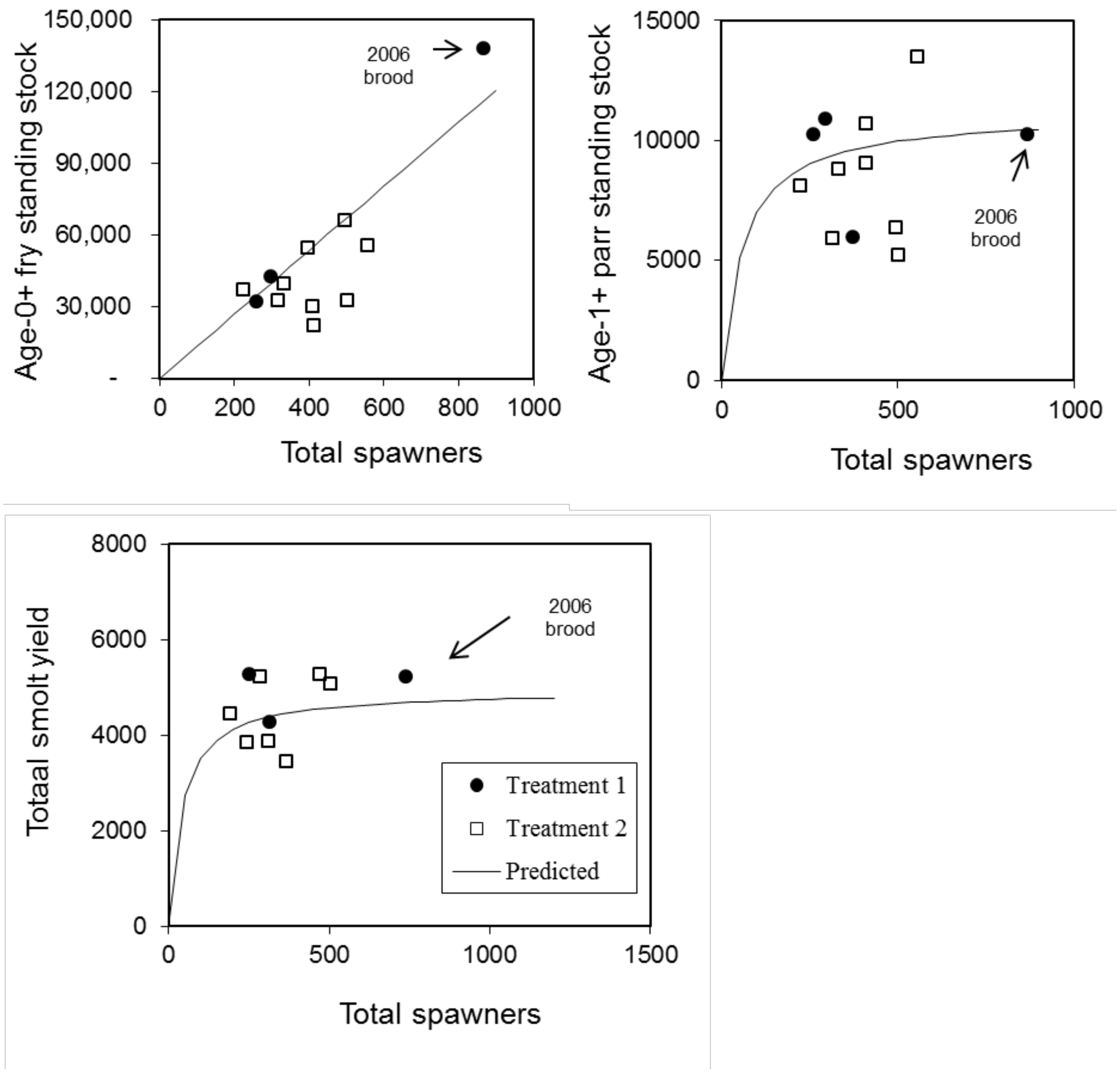


Figure 6.12 Preliminary stock-recruitment relationship for late summer juvenile Steelhead standing stocks and spring smolt yield (2005-2017) versus brood escapements in the Coquitlam River (data points corresponding to peak escapement in 2006 are shown).

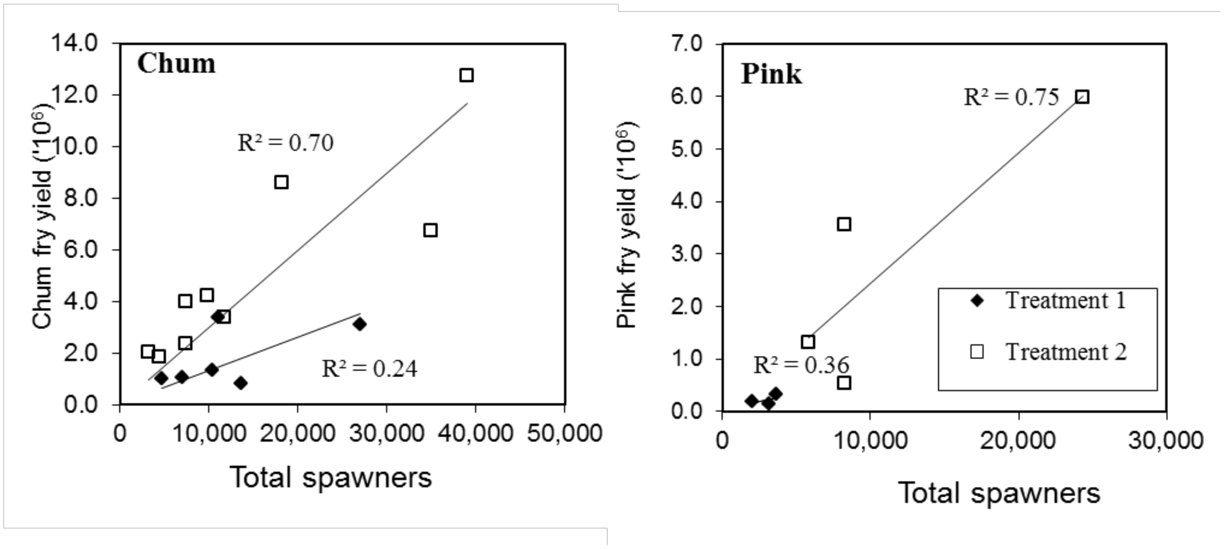


Figure 6.13 Preliminary escapement-to-fry stock-recruitment relationships of Chum and during flow Treatment 1 (2002-2008) and Treatment 2 (2009-2017) from 7.5 km of the Coquitlam River. The best-fit lines intercept the x and y axis at 0 as is typical of stock-recruitment relationships. Note that because Pink Salmon spawn every other year in the lower Fraser River watershed, the number of datapoints is half as for Chum. R^2 values reflect the fit of stock-recruitment.

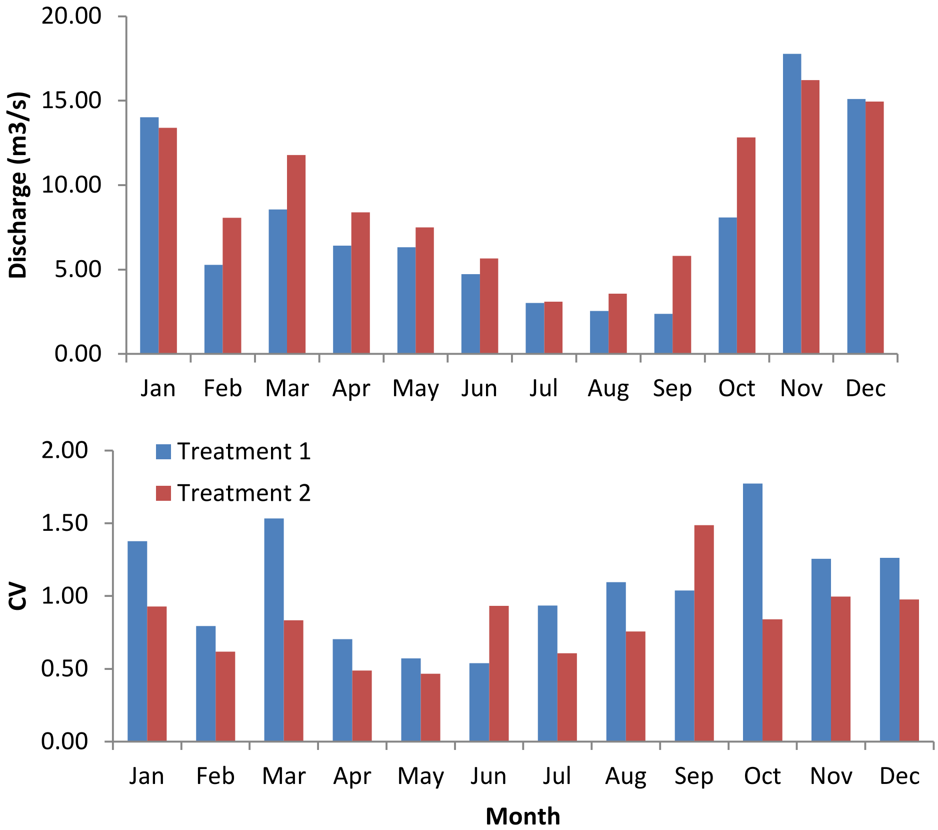


Figure 6.14 Mean monthly discharge and the coefficient of variation in discharge (CV), in Coquitlam River at Port Coquitlam during for Treatment 1 (2000-2008) and Treatment 2 (2009-2017) (Water Survey of Canada, stn. 08MH141). CV is a standardized measure of the variability that allows for comparisons between time periods with different mean discharge.

Table 6.1a Summary of all population estimates for all life stages and species in Coquitlam River, 2000-2017. 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
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	
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	
	Chinook	-	-	<300	<100	<100	<100	438	952	1,529	8,018	4,918	363	2,413	572	123	511	
	steelhead (female)	-	-	-	-	187	434	130	148	113	167	206	278	248	158	251	205	199
	steelhead (total)	-	-	-	-	373	868	260	297	225	333	412	557	495	317	502	410	398
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
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
	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
	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
Smolt yield	chum (total - millions)	-	-	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
	pink (total - millions)	-	-		0.32	-	0.15	-	0.18	-	0.55	-	3.56	-	6.03	-	1.31	-
	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
	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
	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
	steelhead (mainstem)	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
	steelhead (2+)	-	-	-	-	-	-	1,412	2,795	2,968	2,588	1,848	2,177	1,927	3,134	3,134	4,056	3,882
	steelhead (3+)	-	-	-	-	-	-	-	2,849	2,430	2,286	1,256	2,581	1,695	1,520	1,944	1,212	1,343
	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			

Table 6.1b Summary of survival estimates across all life stages and species for 2000-2016 brood escapements 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	life stage	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
coho	Egg-to-fall fry [1]	-	-	-	-	0.68%	0.65%	0.65%	2.00%	0.65%	0.25%	0.29%	0.21%	0.11%	0.24%	0.17%	0.29%	
coho	Fall fry-to-smolt	-	0.00%	0.00%	0.00%	0.00%	2.08%	27.97%	32.57%	17.74%	17.87%	49.82%	11.50%	12.24%	25.15%	17.52%		
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%	
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%
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%	
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%			
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%	
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%		
steelhead	Age 1+ parr to smolt [2]	-	-	-	-	-	68.6%	40.4%	44.1%	35%	55%	44%	32%	38%	68%	92%		
steelhead	Age 2+ parr to smolt [2]	-	-	-	-	-	68.2%	144.0%	193.6%	71.5%	46.7%	66.8%	53.6%	57.9%	50.7%	47.3%	50.8%	
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%	
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).

Table 6.2 Preliminary comparison of mean smolt yield during Treatment 1 and Treatment 2 in the Coquitlam River including the p-values for the two-tailed t tests. 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 2010-2017 for Treatment 2. For Steelhead, this includes 2002-2008 for Treatment 1 and 2012-2017 for Treatment 2.

	Treatment 1		Treatment 2		t test	
Smolt yield	Mean	N	Mean	N	p value	Null Hypothesis of no change (p<0.05)
Coho (Total)	12,458	7	12,507	8	0.971	do not reject
Coho (Mainstem)	5,479	7	7,840	8	0.048	reject
Coho (Reach 2)	1,797	7	2,851	8	0.141	do not reject
Coho (Reach 3)	2,149	7	3,258	8	0.015	reject
Coho (Reach 4)	1,561	7	1,893	8	0.405	do not reject
Steelhead (Total)	3,848	7	4,767	6	0.081	do not reject
Steelhead (Mainstem)	3,716	7	4,684	6	0.069	do not reject
Steelhead (Reach 2)	1,827	7	1,716	6	0.405	do not reject
Steelhead (Reach 3)	944	7	868	6	0.799	do not reject
Steelhead (Reach 4)	925	7	2,253	6	0.000	reject

Table 6.3 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 (2009-2017) 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 Treatments 1 and 2.

Predictive Variable	F value	Significance level probablilty (>F)	Null hypothesis prob < 0.05
Escapement	27.4	<0.001	reject
Treatment	9.1	0.01	do not reject
Escapement x Treatment	2.4	0.14	do not reject

Table 6.4 The sample size for flow Treatment 1 and 2 based on a variety of population metrics useful for evaluation changes in productivity in the Coquitlam River. Only estimates for cohorts that reared entirely under only Treatment 1 or Treatment 2 conditions were included.

		Number of annual esitmates		
Species	Evaluation metric	Treatment 1	Treatment 2	6 years at T1 and T2
Steelhead	smolt abundance	7	6	Yes
	fall fry abundance	3	9	No
	fall age 1 abundance	3	8	No
	fall age 2 abundance	3	7	No
	egg to smolt stock-recruitment	1	6	No
	egg to fall fry	3	8	No
Coho	smolt abundance	7	8	Yes
	fall fry abundance	3	9	No
	egg to smolt stock-recruitment	4	7	No
Pink	fry abundance	3	4	Yes
	recruitment	3	4	Yes
Chum	fry abundance	6	9	Yes
	egg to smolt stock-recruitment	6	9	Yes

9. Figures, Tables and Appendices