Coquitlam-Buntzen Water Use Plan

Lower Coquitlam River Fish Productivity Index

Implementation Year 10

Reference: COQMON #7

Study Period: 2000-2015

Jody Schick
Generation Maintenance
Living Resources Environmental

September 1, 2016

Final

Coquitlam River Fish Monitoring Program: 2000-2015 Results

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BC Hydro

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September 2016

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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. Fish production under Treatment 2 (release flows from 1.1-6.1 cms) will be monitored for up to 9 years. The Coquitlam River Monitoring Program (CRMP) focuses on four anadromous species: Steelhead Trout and Coho, Chum and Pink Salmon, and includes adult escapement and smolt outmigration monitoring for each species. Higher returns during 2007-2014 allowed Chinook 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 during Treatment 1 (2000-2008) and the first five years of Treatment 2 (2009-2015) for the four major components of the CRMP: adult salmon escapement surveys, Steelhead redd counts, juvenile standing stock surveys, and smolt trapping. The primary emphasis of this report is on 2014 fall salmon escapement estimates and outmigration, fall standing stock and Steelhead escapement in 2015. 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 2002-2014 (880 to 13,290 adults; 36 to 519 females/km) likely exceeded that necessary to seed available juvenile habitat based on preliminary stock-recruitment analysis. The 2013 escapement estimate of 4,957 was based on 17 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 2014 late summer standing stock fry estimate was 36,101 (±31%) based on night snorkel surveys. Similar to 2014, this marks a considerable drop from the relatively high abundance during the previous 5-years. During 2015, 11,854 (± 4%) Coho smolts outmigrated past the lowermost trapping site. 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 12,624 smolts, respectively; 2-tailed t-test p=0.91) but does approach a significant difference for mainstem origin smolts (mean: 6,173 and 7,876, respectively; 2-tailed t-test p=0.06), when excluding the 2009 transition year.

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 2015 estimate of 502 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 of Steelhead fry, age 1+ and 2+ parr for 2015 was 32,227 (± 28%), 5,889 (± 23%), and 1,822 (± 46%), respectively. Smolt yield upstream of the lowermost trap was 3,696 (± 16%). Steelhead smolt production in reach 4 immediately below Coquitlam Dam has generally increased over the period of record, from less than 400 smolts in 1996 (prior to the start of Treatment 1) to over 2,000 smolts in recent years. In all but one year since 2009, reach 4 abundance has been at least 2-fold higher than any during Treatment 1 and significantly higher during Treatment 2 (898 and 2,374 smolts, respectively; 2-tailed t-test, $p < 0.01$). At the river-wide level, mean smolt production has been similar between Treatments 1 and 2 (3,848 and 4,348 smolts, respectively; 2-tailed t-test, $p=0.33$) suggesting that flow treatments have not changed juvenile carrying capacity so far but that some reaches have become more productive while others decreased.

Chum escapement for 2014 was the 2nd lowest since 2002 (8,491 adults), only 2010 escapement was lower (6,931 adults). This was based on 11 evenly spaced surveys under relatively good survey conditions. Considering this, the 2014 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 2015, 2.0 million (± 14%) Chum fry outmigrated past the lowermost trap. This is likely a product of the low escapement in 2014. Egg-to-fry survival ranged from 3.7% to 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 raised fry production compared with Treatment 1 (different y-intercepts) but not that the fry-per-spawner relationship differed between treatments (slope of fry-per-spawner were similar for both treatments). These findings could change as we further refine the Chum escapement model. Chum salmon returns to Coquitlam River were greatly improved in 2002-2015 compared to escapements in years prior to the implementation of the Treatment 1 flow regime in 1997.

No Pink escapement estimates were generated for 2014 or outmigration estimates for 2015 as spawning occurs during odd years for the Coquitlam River. Escapement during the most recent run year, 2013 was 3-fold higher than the previous two run-years, which were also the next high since 2001 (34,280 adults). This is consistent with region wide population trends over recent years. The 2013 estimate was based on 8 surveys that included early-, peak – and late run resulting in a relatively reliable estimate. The 2014 fry yield estimate was 6.03 million fish (± 15%). This is a 10- to 20-fold increase over Treatment 1 fry yield and also mirrors the significant increase in the pink fry yield in the Cheakamus River since 2006 (Lingard et al. 2016) Egg-to-fry survival ranged from 5.1% to 9.7%, excluding the biased high 2011 brood year result (48%), which was comparable to reported values for other streams. With only two run-years under Treatment 2 conditions, between-treatment comparisons are premature. 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 2015 was 572 adults, the 2nd lowest since commencing estimates in 2007. 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-7 Status of Objectives, Management Questions and Hypothesis after Year 13

Primary Objective	Management Question	Management Hypothesis	Year 13 (2015) 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 Insufficient power to detect change. Requires larger Treatment 2 sample size. Section 6.2
		H_{01} – Coho smolt production does not differ between Treatments 1 and 2	H_{01} – not rejected for mainstem coho smolts. Insufficient power to detect change. Requires larger Treatment 2 sample size. 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. Stock-recruitment relationships have: a)similar slope b)similar y-intercept	H_{03a} – not rejected H_{03b} –rejected. Higher productivity levels during Treatment 2 at all escapement levels. . Section 6.3
		H_{04} –Each adult Pink produced the same fry yield during Treatments 1 and 2. Stock-recruitment relationships have: a)similar slope b)similar y-intercept	H_{04a} – not rejected H_{04b} – 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

As part of the LB1 WUP, The Coquitlam-Buntzen Water Use Plan Consultative Committee (COQWUPCC) 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. 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 COQWUPCC 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 (through knife valve installed 2008) prescribed by COQWUPCC (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 Coquitlam-Buntzen Water Use Plan completed in 2003 (LB1 WUP; BC Hydro 2003a), flows in the lower Coquitlam River are regulated through the Coquitlam Dam's low-level outlet gates that release flows from Coquitlam Reservoir. The Coquitlam Reservoir also supplies drinking water for the Greater Vancouver Regional District (Metro Vancouver) and water for power generation 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 simply 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 adjacent gravel pit operations adjacent to Coquitlam River also contribute large amounts of fine sediment directly to the stream. 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 0.8 to 1.4 cms, depending on the time of year. This represents the Treatment 1 flow 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 ones expected (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. 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 (*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, COQWUPCC 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. To determine if habitat predictions would translate into increased fish abundance, COQWUPCC took an empirical approach by implementing the Coquitlam River Monitoring Program (CRMP), a 16-year stock assessment program that focused on several life history stages for several species. The Treatment 1 flow regime was evaluated for 8 years (2000-2008; monitoring did not occur in 2001). The Treatment 2 flow regime will be evaluated for up to 9 years (2009-2017).

The CRMP focuses on four species: Coho Salmon *Oncorhynchus kisutch*, Steelhead Trout *Oncorhynchus mykiss*, 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, see Section 1.2.1), or are considered to be of lower economic, recreational, or cultural importance. Adult escapement and smolt 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 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 Coquitlam River determine the quantity and quality of available spawning substrate and incubation conditions for eggs. For these species, smolt production and egg-to-smolt survival are the most important indicators of freshwater production. Figure 1.1 provides a periodicity chart for different life stages of anadromous salmonids in Coquitlam River.

The CRMP is focused on the effects of dam releases on fish productivity in mainstem habitat in reaches 2a, 2b, 3 and 4, of 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 three 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 the CRMP, 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 five parts (Sections 2-6). The first four parts (Sections 2-5) address methods and results for the four monitoring components of the CRMP: 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 CRMP 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 6), 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 the Coquitlam River Monitoring Program 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 2007-2012, 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 2002-2013, 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 on escapement results for returns up to the 2013 spawning period. Results from spawning during the fall of 2014 will be included in the 2000-2015 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 the Coquitlam River Monitoring Program 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), 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 COQWUPCC 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 Coquitlam River (Decker et al. 2007). Juvenile standing stocks were assessed during 2006-2013 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 2013 juvenile standing stock survey, and summarizes preliminary population estimates for 2006-2014.

1.2.4 Smolt outmigrant trapping

Smolt trapping has occurred in 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 2000-2014, 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 2014 smolt trapping program and summarizes population estimates for all species and reaches for 2000-2014.

1. Introduction

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 2002-2014 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 2007-2014 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 for these years in this report. In this report, we have included escapement results for all four species for 2002-2014. 2014 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, 2002-2015; 2nd crew member: Gord Lewis 2002-2006; Kris Kehler 2007, 2015; Thibault Doix 2008-2014). 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 4- to 6-fold higher numbers of Coho than shore-based observers under Treatment 2. The effect of this shift in protocol with regard to estimating Coho escapements is discussed in Section 2.2.2. Field crew did not report an obvious difference in the detectability of other salmon species between Treatments 1 and 2, and there was some support for this based on similar mark-recapture derived estimates of observer efficiency for Chum salmon under the two treatments (see Section 2.2.2).

2.1.2 Partial surveys, observer efficiency, and survey life

Frequent high flow events and associated high turbidity during the fall and winter spawning period contribute substantially to the uncertainty of salmon escapement estimates in Coquitlam River (Decker et al. 2008). During 2002-2014 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. The CRMP 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:

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

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 2006-2014, we conducted 19 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 2014 due to the frequency of high flow 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. 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 2002-2014, 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.

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

The beta distribution is reparameterized 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 nonrandom 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 τ.o (i.e., ε_t~dnorm(0, τ.o), where σ.o=τ.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,

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

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

2.1.3.3 Parameter Estimation and Assessing Model Fit

Posterior probability distributions of model parameters were estimated using a Monte Carlo Markov Chain (MCMC) algorithm as implemented in WinBUGS (Spiegelhalter et al. 1999). We called WinBUGS from the R2WinBUGS (Sturtz et al. 2005) library from R (R Development

Core Team 2009). We used the mean of the posterior to represent the expected value for the parameter, and the ratio of the standard deviation of the posterior to the mean as a measure of relative parameter uncertainty. The 95% credible intervals were determined from the lower 2.5 and upper 97.5 percentiles of the posterior distribution. Posterior distributions were based on a total of 2,000 MCMC samples. These samples were obtained by drawing every 2nd sample from a total of 5,000 simulations after excluding the first 1,000 samples to remove the effects of initial values. This strategy was sufficient to achieve convergence in all cases. Model convergence was evaluated by visually inspecting the MCMC chains for evidence of non-stationarity and poor mixing.

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

2.2 Results and Discussion

2.2.1 Survey conditions and run timing

Unadjusted survey counts from all surveys during 2002-2014 are shown for Chum, Coho, 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. In some 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 average 1.4 m among surveyed sites in 2014 (Table 2.1). During 2014, survey conditions were moderate allowing for 19 surveys generally 5-10 days apart. As has been the case during previous years, visibility during 2014 was highest during September (>3m) and variable but typically lower during the remaining surveys (0.8-1.4m). Visibility in the lower river below the gravel mines was sufficient to not precluded surveys at index sites A and B during 2014. Once Chum spawning was complete surveys excluded Site A since Coho counts from this section are typically less than 1% of the total for any survey (see Table 2.3).

2.2.2 Observer efficiency and survey life

2.2.2.1 Observer efficiency

During 2006-2014, 19 estimates of observer efficiency were obtained for all species combined. None of these were conducted during 2014 due to several high flow events. In some of the 19 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 to collect additional mark-recapture data under Treatment 2 during the remaining two survey years. 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 nine mark-recapture experiments during 2006-2013 (range: 33%-69%; Table 2.2); with similar means for Treatment 1 and Treatment 2 (50% and 48%, respectively; Appendix 2.1).

For Pink, five 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 2006-2013, 11 of 19 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 2006–2013 provided limited information about survey life for each species. Also due to high flow events, no estimates of survey life were conducted during 2014. Obtaining estimates of survey life requires conducting multiple (minimum of three) consecutive surveys 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 15 estimates of survey life were obtained, six for Chum, three for Coho, four 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. Observations of 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 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=9; R^2=0.52; Figure 2.4). 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 third of the variation in mark-recapture derived estimates of observer efficiency among surveys for Pink (n=5; R^2=0.28; Figure 2.4). However, this relationship is highly uncertain, being based on only three 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 60% to 85%. Unfortunately, the guesstimates were negatively related to the mark-recapture data (linear regression, n=3; R^2=0.49; Figure 2.4), but it provides little information about the accuracy of observer guesstimates since only a narrow range of mark-recapture observer efficiency estimates were available for the comparison. For Chinook, there were only two mark-recapture estimates of observer efficiency available (Table 2.2), which provides little information with respect to 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=19; R^2=0.18; Figure 2.4). Mean observer efficiency (based on mark-recapture) across all species was 59% compared to observed means of 70% and 60% for Coho and Chinook, respectively (Table 2.2). For Chum several more, and for Pink, Coho and Chinook a tripling of the number of mark-recapture experiments will need to be conducted in future before reliable species-specific regression models can be developed. 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.

Point estimates of escapement for all species in all years are summarized in Table 2.4. Among years, estimated escapements ranged from 7,000-57,000 for Chum; 900-13,000 for Coho; 3,000-34,000 for Pink; and 1,000-8,000 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 4-fold for Pink and Chinook, 3-fold for Coho and 2-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 2002-2014 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.

2.2.4 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 Chum, Pink, Coho and Chinook during 2002-2013 are highly correlated with the mean counts of annual surveys (correlation coefficient, n=11; R=0.75; n=5, R=0.96; n=11, R=0.96 and n=5, R=0.91; respectively. 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 but whereas the HBM, which accounts for this with run timing priors, estimated the largest escapement to date, the mean count indicated it was one of the lowest. The use of mean count requires further evaluation.

We previously proposed to use the peak count as an index for escapement rather than continue with the HBM approach for Coho and Chinook (Schick et al 2014) 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. 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, since there is no additional cost as surveys continue after peak run anyways, mean count or peak count could be used instead of HBM. 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.

2.2.5 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 62% of adult Chum spawning in index sites A-C during 2002-2014; 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 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 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 (55%-71%, Treatment 1; 59-76%, Treatment 2; 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 (82%-90%, Treatment 1; 69-78%, Treatment 2; 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 2002-2014; Table 2.3). The trend of low natural or enhanced off-channel habitat use during Treatment 2 continued in 2014 with record low usage (8%, Table 2.5). The combined natural and enhanced off-channel habitat use dropped from 20%-73% during 2002-2007 to 8%-16% during 2009-2014 (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.

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.

2.2.6 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 2014 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 weakens the reliability of inferences drawn from this data. Unlike for Coho, Chum 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 assume that the adult abundance (x axis) is without error, or at least of much less error than juvenile abundance (Zar, 1999). While stock-recruitment analyses rarely satisfy this assumption, knowing the precision of our estimates would allow us to exclude very imprecise estimates, which may increases 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.

2. Adult Salmon Escapement

3.0 ADULT STEELHEAD ESCAPEMENT

3.1 Methods

During 2005-2015, 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 – 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, Jacob *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. For five of the seven surveys during 2014 that satisfied the 2-week interval criteria, we estimated the number of redds simply as the sum of new redds (x_i) counted during n surveys (Equation 3.1). For the remaining two surveys, we used 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.

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

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:

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

Where x_i is the cumulative number of new redds summed across n surveys and 1.2 is a constant representing the number of redds per individual female spawner. In the absence of fecundity data for Coquitlam River Steelhead, we substituted average fecundity for winter Steelhead 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 2005-2015, 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 2009-2015, increased discharge from Coquitlam Dam under Treatment 2 resulted in higher base flows compared to previous years. Mean daily flow exceeded 10 cms on 23-34% of days during the major spawning period in Treatment 2, versus 4-24% in previous years, Figure 3.3), while average discharge during the Treatment 2 spawning periods (8.75-9.6 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 (6.7 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 7 surveys in 2015; across the entire survey period, the length of time between surveys ranged from 9 - 14 days, and averaged 11 days (Table 3.1). In previous years, the length of time between surveys has ranged from 6 to 37 days (Table 3.1).

In 2015, the first survey was conducted on March 17. Similar to past years, live adult Steelhead were observed (41 fish, Table 3.1), along with considerable spawning (21 redds were counted, Table 3.1). 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. In 2015, 82% of new redds were counted on surveys conducted from April 6 to May 24. This was a similar pattern to previous years when 80-90% of new redds were observed during a six-week period spanning early April to mid-May (Table 3.1, Figure 3.4). Different from past years was the relatively large number of redds observed during the initial March 3rd survey (27 redds counted)

Spawning Steelhead preferred mainstem habitat as compared to natural side channel and constructed off-channel habitat by a large margin during all survey years. For example, of the total number of redds observed in 2014, 83% were in the mainstem. 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 2009-2015 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 have been generally higher than during Treatment 1 (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 (Figure 3.5). However, values shown in Figure 3.5 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), as is the case on the Coquitlam River (correlation coefficient, n=10, R=0.52).

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 2015, the period between surveys was typically sufficiently short (≤ 14 days) to assume that only a small number of redds became obscured from one survey to the next based on the evaluation of redd survey life during 2005 - 2015. To confirm this, we used the redd life model using 2015 redd loss information to estimate the number of redds constructed but then obscured between any two surveys. We estimated that no redds became obscured between surveys (Table 3.1). From 2005-2015, 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 2015 was 251 females (Table 3.2), an above average value for the 2005-2015 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 28 redds/km in 2015, and ranged from 13-48 redds/km during 2005-2015 (Table 3.2). Among reaches and years, redd density ranged from 6-71 redds/km (Table 3.2). Spawning distribution during 2015 was balanced between the lower river, reaches 2a and 2b, and the upper river, reaches 3 and 4 (46% and 54%, 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 2005-2015.

4.0 JUVENILE SALMONID STANDING STOCK

In 2006 the COQWUP 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 2007-2015 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 2006-2015 (see *Section 4.1.5*); this model replaces a bootstrap model used in previous years (Decker et al. 2012).

During 2007-2015 we also conducted a separate electrofishing survey (with input and assistance from Ron Ptolemy, MOE stock assessment). As per the CRMP 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 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 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 2014-2015 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 first stage or process error (spatial variation in fish abundance among sampling sites) and second stage or 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 ≈ 0.85 km (Figure 4.1; Reach 4 was not sampled in 2006). The additional 12 sites added in 2014 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 (2009-2015). 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 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.

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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.

4.1.6 Day electrofishing survey

In 2015 we resurveyed four shoreline electrofishing sites previously sampled during 2007-2014. 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 2009-2015 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 2015, the night snorkeling survey was completed during August 29-31 and September 3, 4 and 6 at a flow of 2.7 – 7.2 cms; station 08MH002, Port Coquitlam). Previous surveys were conducted at flows of 0.8-2 cms during Treatment 1, and 2-6 cms during Treatment 2. Water temperatures ranged from 18°C-21°C during 2015 similar to previous years. In 2015, 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 2015. 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

In 2015, Coho fry abundance in Coquitlam River mainstem was low compared with previous years (3,231 vs. 2,061-10,214 fish/km, Table 4.3), and a substantial drop from 2011-2013 (7,305-10,214 fish/km). Total standing stocks of Coho varied among years from 19,000 to 105,000 fry (Table 4.3). Coho fry density was positively correlated with distance from the stream mouth during 2010, 2012 and 2013 (R=0.92, 0.88, 0.67, respectively, P<0.01; Figure 4.3), but not during 2006-2009, 2011, 2014 and 2015 (R= 0.30, 0.59, 0.25, -0.19, 0.58, 0.40 and 0.18, respectively, P > 0.05 for all cases). Averaged by treatment, Coho density generally increased with distance upstream for both Treatments 1 and 2, the increase was more pronounced during Treatment 2, particularly in the upper 2km (Figure 4.3).

The 2015 Steelhead fry density of 1,683 fish/km was low compared to previous years 2006-2014 (2,674-13,833 fish/km). We consider the 2015 fry estimate credible (unbiased) since the assumptions underlying the mark-recapture methodology were largely satisfied, particularly that the minimum forklength was greater than the range included in mark-recapture experiments, see Schick *et al*. 2012. Relative precision also increased substantially in 2015 (±28%) compared to year prior to adding the additional 12 index sampling sites in 2014 (±40%-70%). Annual Steelhead fry standing stocks ranged from 22,000-138,000 (Table 4.3). During Treatment 1, Steelhead fry densities were substantially higher in the middle portion of the study area (km 11-13) compared to upper and lower reaches (Figure 4.3), whereas in 2008-2015, density was far less variable and with no clear trend.

In 2015 the density of age-1+ Steelhead parr had the lowest density since commencing snorkel surveys (510 fish/km, 2015 vs 605-1,664 fish/km, 2006-2014, Table 4.3). Total standing stock of age-1+ parr varied from 5,889-13,456 among years (Table 4.3). Mean density of age-2+ Steelhead parr has been consistently higher since 2009 (199-372 fish/km during 2009-2015; Table 4.3) than that during 2006-2008 (112-177 fish/km). This trend corresponds well with the flow treatments. The age-2+ parr estimates from the fall of 2009 onward represent parr under Treatment 2 conditions for at least a full year. However, it is difficult to draw conclusions about flow treatment effects from age-2+ parr abundance since it represents those that had survived to that age and did not smolt during the preceding spring. There was no strong longitudinal pattern in Steelhead parr density among sites in 2006 or 2009-2013 (Figure 4.3), whereas in 2007 and 2008, Steelhead parr densities were highest at sites located within a 3 km long section immediately downstream of Or Creek (reach 3 and the upper portion of reach 2b; Figures 4.1, 4.3).

The low abundance for all species and age-classes survey during 2015 could be in part due to the relatively low water levels during the spring and summer of 2015. Discharge during this period of 2015 may share more similarities with Treatment 1than Treatment 2 or considered sufficiently anomalous to be excluded from Treatment 2 when comparing treatment effects.

4.3.1 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 2012-2013 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.

4.3.2 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 2006-2015 (16-59 fish/100 m^2) were 3-10 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 2006-2015 were 1.5- to 5-fold higher (15-53 fish/100 m^2). Steelhead parr densities were 6-19 times higher during 2006-2015 (3.2-9.2 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 2006-2015 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, 2011- 2013 (53, 31, 30 and 40 fish/100 m^2, respectively) was relatively high compared to published values for other streams, while densities in 2007-2010, and 2014 (15-20 fish/100 m^2) were average to low. 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). However, the relatively high 2015 escapement resulted in relatively low fall fry abundance

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 (17-60 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.

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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.

4.3.3 Fish densities in 'optimal' habitats based on electrofishing

In comparison to fish density estimates derived from the electrofishing data, the snorkeling data for 2006-2015 suggests 1.3- to 22-fold higher densities of Coho, depending on the year; generally lower densities of Steelhead fry and age-1 parr; and much higher densities of age-2+ parr (0-0.4 parr/100 m^2 based on electrofishing versus 0.8-2.8 parr/100 m^2 based on snorkeling; Tables 4.3 and 4.4). 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. The annual estimates of Steelhead fry were highly correlated for the two methods (R=0.90 excluding 2011) and during recent years, so to for Coho (R=0.73, 2008-2014) but not for age-1+ or age-2+ Steelhead parr.

For Coho, the electrofishing data suggested that abundance remained consistently low during 2006-2011 (1.2-13.3 fish/100 m^2), whereas the snorkeling data indicated an increase from 19 to 60 fish/100 m^2 (Table 4.3). Electrofishing was ineffective for age-2+ Steelhead in all years; a total of only 12 age-2+ parr were captured during electrofishing at 30 sites during 2006-2015 (Table 4.4), whereas snorkelers detected an average of 4 age-2+ Steelhead per site over this period

Electrofishing surveys in Coquitlam River during 2007-2015 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 (2009-2015), 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 2006-2012 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 2006-2015 failed to include 'optimal' sites where maximum Steelhead biomass would be expected.

4.4 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 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 2006-2015 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 PRODUCTION

5.1 Methods

5.1.1 Coho and Steelhead smolt enumeration

In 2015, 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 2006-2015, 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 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

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

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).

The Archery Pond weir, used to enumerate smolts prior to 2008, was again used in 2015 due to the difficulty of marking sufficient Coho fry to use mark-recapture by minnow trapping to estimate pre-smolt abundance. Archery Pond was excluded from downstream trapping during 2009-2012 because, of the four, this site has historically produced the fewest number 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 2015, 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 fry.

The off-channel weirs and the mainstem RSTs were operated continuously from mid-March until mid-June (Table 5.1). 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 – 2015, 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:

$$\pm 95\%\ CI\left(N_{reach\ 2}\right) = \pm 1.96\sqrt{Var\left(N_{RST2}\right) + Var\left(N_{RST3}\right)} \qquad (5.8)$$

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:

$$\pm 95\%\ CI\left(N_{reach\ 4}\right) = \pm 1.96\sqrt{Var\left(N_{RST4}\right)} \qquad (5.9)$$

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

$$\pm 95\%\ CI\left(N_{mainstem}\right) = \pm 1.96\sqrt{Var\left(N_{RST2}\right)} \qquad (5.10)$$

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

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

with a 95% confidence interval of:

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

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

Only Chum salmon fry were present in the Coquitlam River during spring 2015.
To estimate Chum 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 2015, daily catches of Coho and Steelhead 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 (Figures 5.3 and 5.4). Therefore, we assumed that population closure was largely met, and that captures at the weirs accurately represented total smolt output. Observed mortality was < 1% for all target species at the off-channel weirs with the exception of coho at Overland Channel with 3% mortality. No incidents of weir failure or fish leakage were apparent at the off-channel sites during 2015.

An aggregate total of 3,620 Coho 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). Coho abundance in Archery Pond, where a new pond was constructed in 2013, increased by 63% to 456 smolts from 2014 (279 smolts), which was already a substantial increase from prior to construction (19 – 191 smolts, 2011-2013). Total Steelhead smolt production for the Overland, Or Creek, Archery Ponds and Grant's Tomb off-channel sites was 112 smolts (Table 5.2). Mean weighted density of Coho smolts in the off-channel sites was 17.1smolts/100 m^2 while Steelhead smolt density was 0.5 smolts/100 m^2 (Table 5.2).

5.2.2 Coquitlam River mainstem

During 2015, discharge in the Coquitlam River during the spring trapping period was very stable, with daily mean flows exceeding 20 cms on only 2 occasions during Chum fry migration and none during Steelhead and Coho migration, (Figure 5.2). The lack of spring precipitation and low winter snowpack led to lower discharge, particularly for the April-June period (mean 3.5cms near Port Coquitlam), compared to other Treatment 2 years (~8-10 cms) and showed greater similarity to conditions during Treatment 1 (~3-6 cms). Overall, observed mortality at the RSTs was 0.3% for Coho and 0.4% for Steelhead smolts, 1.0% for Chum fry and 3.0% for Chinook smolts. Only 5 Oncorhynchus nerka smolts were captured in 2015. For Steelhead, 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 indication of early season downstream movement of Coho as there was in 2014.

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 (71% and 73%, respectively, Fisher's exact test, P=0.5, Table 5.3, Figure 5.5). The combined mark group produced a similar estimate as the mainstem mark group (difference < 1%) and slightly lower precision (95% CI: ± 34 both, ± 31 mainstem), therefore we used only the mainstem mark group to generate a population estimate of 1,148 Coho smolts (95% CI: ± 31 smolts) for reach 4 (Table 5.2).

At RST3, CE were not significantly different for marked off-channel and mainstem Coho smolts (14% and 13%, respectively, Fisher's exact test, P<0.14; Table 5.3, Figure 5.5). The difference between estimates produced using the mainstem and combined mark groups differed by 535 smolts whereas the SE of the mainstem estimate was 1,073. Therefore, we combined mark groups to generate a population estimate of 4,265 smolts (± 681 smolts, Table 5.2) in reach 3, after smolt numbers from reach 4 were subtracted.

At RST2, CE was not different for the mainstem mark and off-channel mark groups (36% and 38%, respectively, P <0.07; Table 5.3, Figure 5.5). As with RST 4, the combined mark group produced an estimate 10% lower than using the mainstem mark group. The difference between estimates using only mainstem marks and the combined mark groups was also larger than the SE of the mainstem estimate (SE: 476 smolts), a condition for rejecting the use of combined mark groups. Considering this, we used only the mainstem mark group for the reach 2 population estimate of 2,821 ± 830 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 2015 was 8,234 ± 467 (11,854 ± 467 smolts including those from the four off-channel sites, Table 5.2). Average Coho smolt density in the Coquitlam River was 5.5 smolts/100 m^2 (6.9 smolts/100 m^2 including the off-channel sites, Table 5.2). Areal Coho density was higher in reach 3 (9.1 smolts/100m^2, Table 5.2), than in reach 2 or 4 (3.4 and 5.5 smolts /100 $m^{2;}$ respectively). Precision ranged from ± 2.7% for the estimate for reach 4, to ± 29% for the smolt estimate for reach 2 (Table 5.2).

5.2.2.2 Steelhead

At RST4, CE was not significantly different between marked off-channel Steelhead smolts (from Grant's Tomb) and mainstem smolts that were captured at RST4, marked and released upstream (41 % and 30%, respectively, Fisher's exact test, P=0.14; Table 5.3, Figure 5.5). There was also no improved precision when using the combining mark groups (95% CI: ±15% for each), therefore we used only the mainstem mark group to generate a population estimate of 1,808 Steelhead smolts (± 270) for reach 4 (Table 5.2).

At RST3, CE was borderline significantly different for the off-channel and mainstem mark groups (16% and 9%, respectively, P = 0.06). Therefore, we used only the mainstem mark group to estimate that 3,309 mainstem smolts (±1,166) passed RST3 (Appendix 5.1). This resulted in a population estimate for reach 3 of 1,730 smolts (± 1,197 smolts, Table 5.2) for reach 3, after smolt numbers from reach 4, and unmarked smolts from off-channel site were subtracted.

At RST2, we used the mainstem mark group since CE was significantly different for the two groups (21% off-channel and 12% mainstem, $P = 0.01$; Table 5.3). The resultant estimate for reach 2 was 1,428 Steelhead smolts (± 1,284 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 4,966 ± 537 (5,078 ± 537 smolts when off-channel sites were included, Table 5.2). Average Steelhead density in the Coquitlam River mainstem was 3.3 smolts/100 m^2 (2.9 smolts/100 m^2 in the Coquitlam River including the off-channel sites, Table 5.2). Areal Steelhead smolt density was over 2.6-fold higher in reach 4 (9.4 smolts/100m^2, Table 5.2) than in reaches 3 and 2 (3.2 and 1.7 smolts/100m^2, respectively). The precision of the abundance estimates ranged from ± 9% for the estimate smolt abundance the Coquitlam River including off-channel areas, to ± 90% 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 2005-2015 (Appendix 5.4). Scale analysis of 452 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 173 mm to 219 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 2005-2012 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 2007-2015 smolt populations. The proportion of age-2 smolts ranged from 50%- 67% among years. From these estimates, we derived yields of 4,261; 5,225; 5,254; 3,843; 4,029; 3,871; 3,447 and 5,078 Steelhead smolts for the 2005 - 2012 broods, respectively (see Table 6.1). 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 < 120m) will outmigrate after one or two additional winters in the Coquitlam River, though there are a small number that do exhibit the physical smolt characteristics and could smolt during the current year. During 2015, 361 age-1+ Steelhead without a smolt like appearance were captured at the RST 2 trapping location and has ranged from 162-361 fish during 2012-2015. If a portion of age-1+ Steelhead undergo an early outmigration from the study reaches then this would represent unaccounted for productivity.

5.2.2.3 Chum and Pink

Only chum salmon fry were present in the Coquitlam River during spring 2015. Chum were trapped continuously from March 13 to June 10 (Chum) at the RST2 location in reach 2. Chum were batch-marked on nine separate occasions, respectively (Appendix 5.3). For Chum, capture efficiency varied from 1.2%-9.4% (Appendix 5.3), and averaged 6.0% (all strata pooled Appendix 5.1).

During 2015, an estimated 2.0 million Chum fry (± 0.29 million, Table 5.2) migrated past the RST2 trapping site. This equates to a Chum density for the mainstem of the Coquitlam River of 268 thousand fry/km or 1,343 fry/100 m^2 (Table 5.2).

5.2.2.4 Oncorhynchus nerka

In 2015, Only 5 *Oncorhynchus nerka* smolts were captured in the Coquitlam River mainstem. *O. nerka* 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.

5.3 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 1996-2015 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 a similar marking approach as for smolts (distinct mark for capture locations) would provide information about the proportion of parr that move below the study reaches.

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 CRMP 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 power (β) decreases significantly over a range of increasing observation error ($\sigma_{sm,o}$ in their paper) for estimates of smolt abundance from about 0.1 to 0.5 $\sigma_{sm,o}$ (Figure 5, p. 18 in their paper). Expressed as a 95% confidence interval, values for $\sigma_{sm,o}$ of 0.1 to 0.5 are equivalent to levels of precision of ±20% to ±110% of the estimate.

The precision of the 2015 Coho smolt abundance estimate in the Coquitlam River mainstem was high (95% confidence interval: ±6%) compared with estimates during 2000 - 2014 (95% confidence interval: ±6% to ±14%) and was much better than the theoretical optimal value of $\sigma_{sm,o} \approx 0.1$ (±20%). Precision of the 2015 mainstem Steelhead smolt estimate was the highest (95% confidence interval: ±11%) compared with mainstem estimates since 2000 (95% confidence interval: ±14% to ±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.

The precision of fry population estimates for Chum salmon at the RST2 in 2015 is moderate compared with previous years using rotary screw traps (95% confidence interval: ±14% in 2015, and ±7%-18% during 2008-2014) and much higher than years using incline plane traps (±19% to ±25%). This was the product of the relatively high capture efficiency throughout (6.0%) but particularly during the periods of high outmigration.

See Section 6 for the results of hypothesis testing using outmigration data.

6.0 FISH PRODUCTIVITY DURING TREATMENTS 1 AND 2

6.1 Coho

During 2000-2015 Coho smolt yield 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 (mean: 7,244 smolts; Table 6.1a). Annual Coho smolt numbers for mainstem and constructed off-channel habitat combined, were, on average, double that for the mainstem alone, with less variation from year to year (mean: 14,200 smolts, range: 8,400-24,500; Table 6.1a). Excluding 2009, which represents a region-wide recruitment anomaly, mean abundance for Treatment 1and 2 were not statistically different for the mainstem and off-channel smolts combined (t-test, p = 0.90, Table 6.2) but was nearly significant for mainstem smolts alone with a 28% increase from Treatment 1 to Treatment 2 (2 – tailed t-test, p = 0.06). Smolt yield in 2009, the first year affected by Treatment 2, was nearly two-fold higher than smolt yields produced from comparable spawner returns during Treatment 1 (Figure 6.1). However, Coho smolt yield in the nearby Alouette River was also two-fold higher in 2009 compared to other years (Figure 6.2; Cope 2011), suggesting that 2009 represents a recruitment anomaly caused by some factor acting at a larger regional scale. Omitting 2009, the escapement-to-smolt stock-recruitment relationship is indicative of a highly productive system, which reaches carrying capacity at relatively low escapement levels and beyond this threshold is relatively insensitive to additional spawners (Figure 6.1).

In reach 4, where annual downstream trapping has occurred over a longer time period (1997-present), there is 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 6.3). Late summer snorkeling surveys suggest that densities of Coho fry in reaches 2 and 3 were several-fold higher during 2006-2015 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 33% to 77% of the overwintering Coho smolt population during 2000-2015. 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-

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

channels and ponds in other Pacific Northwest streams (67 and 69 smolts/100m^2, respectively; Koning and Keeley 1997).

Although the accuracy of Coho escapement estimates remains highly uncertain, owing to a lack of observer efficiency and survey life data, and different survey methods in Treatment 2 (see Section 2), these estimates nevertheless suggest that in most cases Coho escapements were more than adequate to seed available juvenile habitat during Treatments 1 and 2. Estimated Coho spawner densities during 2002-2014 ranged from 70 to 1038 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).

Mean size of Coho smolts in reach 4 was slightly greater during 1999-2014 (mean: 95 mm), under Treatments 1 and 2, compared to the period preceding Treatment 1 (1996-1998; mean: 89 mm; t-test, p=0.07; Figure 6.4). No size data exist for reaches 2 and 3 prior to 1999. During 1999-2014, Coho smolts were consistently larger in reach 4 than in reaches 2 and 3, and larger in mainstem versus off-channel habitat (Figure 6.4). There were also some consistent among-reach differences in Coho smolt densities. During 2000-2014 areal densities of Coho smolts were generally greatest in reach 4, exhibiting a downstream decline from reach 4 to reach 2. Late summer snorkeling surveys suggested a similar trend (see Section 4.3.2).

For the purpose of comparing Coho productivity 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.), was 1,812 smolts/km (range: 1,118-3,261 smolts/km, 2000-2015). This suggests that Coho smolt productivity in the Coquitlam River study area is comparable to the average for streams at this latitude.

During Treatments 1and 2, Coho smolt production in the Coquitlam River mainstem upstream of RST2 was relatively stable (8,400-14,700 smolts; Figure 6.1), despite a three-fold variation in spawner abundance (799-11,400 spawners), which is expected if spawner abundance exceeds that required for full seeding. Annual Coho smolt yield in the Coquitlam River during Treatment 1, Treatment 2 and overall was positively correlated (R = 0.86, 0.71 and 0.51, respectively; Figure 6.2, Cope 2011) with that in the dam-regulated Alouette River suggests that variation in smolt production in the Coquitlam River during the period of study was governed, to a large degree, by region-wide freshwater rearing conditions. The higher smolt estimates on the Alouette River since 2009 coincide with repositioning the trapping site further upstream to avoid tidal-driven backwatering of the trap and would account for the higher with treatment periods than between them. This limits the ability to use it as a control in a BACI analysis of treatment effects. The strong linear relationship between escapement and fall fry yield reported in prior reports did not continue when incorporating results from recent years. Both linear and Beverton-Holt stock-recruitment models had similarly moderate fit (R^2 = 0.45 and R^2 = 0.43, respectively, Figure 6.1) suggesting similar support for either model. The use of fall fry abundance to compare treatment effects is a weak approach considering that the corresponding brood escapement for all Treatment 1 fry estimates were lower than all but one during Treatment 2. When the range of escapement is unequal between treatments then the comparison may be overly influenced by differences in escapement, especially when very low escapement is over-represented in one of the comparison groups.

Coho egg-to-smolt survival remained consistently low (0.2-1.1%) for the 2002-2013 brood years (Table 6.1b), with the highest values associated with the lowest escapements. These values were not different for Treatment 1 and 2 (mean: both 1%, 2-tailed t-test p=0.78), but we would be cautious about any between-treatment evaluations reliant on adult escapement as survey method is confounded with flow treatments. By comparison, the average egg-to-smolt survival rate for Coho populations in nine other Pacific coastal streams was 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 2000-2015 the estimated number of Steelhead smolts outmigrating from the 7.5 km long section of the Coquitlam River upstream of the RST2 trapping site ranged from 2,300 to 5,600, and averaged 3,848 smolts during Treatment 1 and 4,498 during Treatment 2 (Table 6.1a). Mean smolt yields were not statistically different between Treatment 1 and Treatment 2 (3,848 smolts and 4,498 smolts respectively, 2-tailed t-test $p = 0.20$). Smolt yield from reach 4 has increased substantially since 1996 (R^2=0.71, $p < 0.01$; 1996-2015; Figure 6.3). Mean smolt abundance in this reach has increased over 2-fold from Treatment 1 to 2 (898 smolts and 2,306 smolts respectively; 2-tailed t-test $p < 0.01$). This is likely a product of the higher Treatment 2 base flows combined with the relatively narrow channel width in reach 4 that resulted in a shift from a low to higher energy flow environment more favorable to juvenile Steelhead. However, the increased abundance in reach 4 has been offset by reduced productivity elsewhere in the system.

There was no significant difference in the size of spring migrant Steelhead parr (age-1+) in reach 4 during Treatments 1 and 2 (1999-2015 mean: 98 mm; Figure 6.4) in comparison to earlier years (1996-1998; mean: 90 mm; t-test, $p = 0.21$), when dam releases were lower. In most years, age-1+ spring migrant parr in reach 4 were also larger than those in reaches 2 and 3 and in the off-channel sites (Figure 6.4). On average, forklength of Steelhead smolts was 6-7 mm less during Treatment than Treatment 2 for off-channel, reach 4 and reach 2&3 combined (2-tailed t-test $p = 0.4$, 0.01 and 0.01, respectively, Figure 6.4).

There was little correlation between annual Steelhead smolt yield in the Coquitlam River and that in the Alouette River ($R = 0.37$; Figure 6.2; Cope 2011), a nearby regulated stream, suggesting that variation in annual Steelhead smolt production in the two streams was influenced to a greater degree by local watershed conditions than by broader regional or climatic factors**.** Nearly all Steelhead smolts (≈99%) originated from the Coquitlam River mainstem as opposed to the constructed off-channel sites in the study area. During 2000-2015, 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 and 2010, 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).

Snorkeling surveys indicated that during 2006-2015, Steelhead fry density in the Coquitlam River in late summer averaged 28.0 fish/100m^2, while parr density averaged 6.9 fish /100m^2. In general, these values are fairly typical 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 2006-2015 that 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 1+ parr was similar between Treatments 1 and 2 (age 1+ parr 8,812 and 9,165; respectively; t-test $p = 0.85$; Table 6.3) yet age 2 + parr were only half as abundant during Treatment 1 than during Treatment 2 (age 2+ parr 1,538 and 3,122; respectively; t-test $p < 0.01$; Table 6.2). 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 (2010-2015) compared with Treatment 1 (4,654 smolts, 3,848 smolts; respectively; t-test $p = 0.33$, 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.

During 2005-2015, Steelhead spawner densities in the Coquitlam River ranged from 24 to 80 fish/km (mean: 38 fish/km). Comparisons of Steelhead spawner densities in the Coquitlam River, relative to those in other streams are 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 2002-2015 (mean: 38 fish/km), but were not correlated with Coquitlam River escapements ($R^2 = 0.05$; Figure 6.2). As part of the ongoing Georgia Basin Steelhead Recovery Program (GBSRP; http://www.bccf.com /Steelhead), uncalibrated snorkeling counts of adult Steelhead were conducted in numerous Lower Mainland streams up to 2006, but results have not been reported since 2002. Data from a province-wide mail-out creel survey suggests that total effort and catch in the Steelhead sport fishery in the Coquitlam River was down marginally in 1997-2002 compared 1969-1996, but did not show the precipitous declines that occurred for many Georgia Basin streams.

6. Fish Productivity During Treatments 1 and 2

Estimated Steelhead egg deposition in the Coquitlam River during 2005-2014 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 (≈50% 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 2005-2015, taking into account that egg deposition per unit area exceeded the threshold value for the Keogh River by 3- 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 6.5) 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 5.3% to 9.1% (mean: 6.3%, Table 6.1b) during 2006-2015. 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 0.8%-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 2005-2011 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 (0.3%-1.3%) for Steelhead egg-to-smolt survival in the Keogh River. Fry-to-age-1+ parr survival for the 2006-2012 fry cohorts ranged from 10%-37% (Table 6.1b). Age-1+ parr-to-smolt survival ranged from 33%-69% for the 2006-2012 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 450 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. For the period of record, current estimates suggest that adult returns of Chum salmon to Coquitlam River (including reach 1) have ranged from 12,000-57,000 (Table 6.1a), while fry production upstream of RST2 has ranged from 0.8 to 8.6 million. The high 2014 fry yield was likely a product of the high 2013 escapement combined with sufficient spawning habitat and the favorable freshwater conditions considering that escapement above RST 2 was almost half of that for the 2012 brood yet it produced a third more fry. 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 (two tailed t-test $p < 0.01$), ranging from 18.1% to 40.0% (mean: 24.7%, 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, are 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 smolts, which in 2009, 2011 and 2014 were relatively precise (95% CI: ±7 and 14%, respectively), with no evidence of serious violations of mark-recapture assumptions. Until the HBM used for Chum escapement estimates is able to form measures of precision, we have few objectives means to identify which adult estimates are poorly defined. Given the weaknesses of adult salmon escapement estimates we view them as an index of abundance rather than accurate measures of escapement.

During 2003-2015, there were similar or stronger linear escapement-to-fry relationships within Treatments 1 and 2 ($R^2 = 0.36$ and 0.60, respectively, Figure 6.6) than across both treatment periods ($R^2 = 0.38$), suggesting that flow treatments may play a role in juvenile production but that other factors still play a substantial role in determining smolt yield (i.e., spawning habitat was fully utilized and egg-to-fry-survival was strongly density-dependent). During 2002-2015**,** both Chum escapement and fry yield in the Coquitlam River were only minimally to moderately correlated with that in the Alouette River (escapement: R = 0.55, smolt yield: R=0.03; Figure 6.2; Cope 2011), which reduces the viability of using the Alouette River as a control of region-wide factors influencing Chum productivity. Chum escapement and fry yield are moderately correlated with the Cheakamus River (R = 0.51 and R = 0.54, Fell *et al.* 2013), which may provide some ability to control for region-wide factors.

Overall, Chum salmon returns to Coquitlam River were markedly improved in 2002-2015 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 and 2013 brood egg-to-fry survival far exceeded this range (48% and 27%, respectively), even when incorporating the 95% confidence limits of the 2012 and 2014 fry estimate, 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.

During Treatment 1, Pink fry yield in the Coquitlam River was positively but weakly correlated with escapement (R = 0.36) and strongly and positively correlated during Treatment 2 (R = 0.7, Figure 6.6). When examined over the entire time period, the correlation between fry yield and escapement is also positive and weak (2003-2013, R = 0.84). The strength of the correlations changed substantially with the addition of the past two brood years highlighting one of the limitations of comparing treatments with small sample sizes. Pink escapement was moderately correlated with that and in the Alouette River (R=0.16, Figure 6.2). As with Chum, there was little correlation between Pink fry yield in the Coquitlam River and that in the Alouette River (R^2 = 0.10, Figure 6.2) but a strong correlation exists with the Cheakamus River fry production (R = 0.98, Figure 6.2).

Our ability to distinguish treatment effects from region-wide abundance trends is low given that escapement so far 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

The CRMP 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 data collected during Treatment 1 because, for some life stages, 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. 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 add 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. It should be noted that this does not incorporate the uncertainty of each estimate used to calculate the mean for each treatment. 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 (Figure 6.1). 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 salmon, 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 2003-2014. 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. It is worth highlighting that with the addition of a single Cohort (2013), the results changed from significant values only for the Treatment effect but not for Escapement to significant values for both effects to illustrate that conclusions should be drawn once results are relatively stable to additions or omissions of a single year-class. This analysis will be further developed in future years to incorporate the uncertainty of individual fry and escapement estimates into slope and offset 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 2014 the planned end to smolt monitoring on the Alouette River, analysis will remain relatively unchanged from what is currently available.

The inclusion of the Alouette River, Cheakamus River or other 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 (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.). With a BACI design, a type 1 error would be evident if, for example, fish numbers increased by a similar magnitude in both the Coquitlam and Alouette rivers during Treatment 2. Conversely, all other factors remaining equal, if fish numbers remained unchanged in the Coquitlam River during Treatment 2, but numbers declined substantially in the Alouette River, increased flows in the Coquitlam River may have been responsible for offsetting some other environmental factor that negatively affected both streams in the post-treatment period.

6. Fish Productivity During Treatments 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 focussed on Chum and Pink.

3. Continue reconnaissance surveys at the beginning of the arrival of Pink salmon and following the completion of Coho spawning, if continuing the HBM based escapement estimates, in order to confirm the absence of these species from the study area. 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. During even years, reconnaissance surveys for early arriving Chinook should commence in early September, if accurate HBM based estimates of Chinook escapement are of interest.

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 STANDING STOCK

6. As much as possible, continue sampling the original 12 snorkeling sites and the 12 new sites added in 2014 each year.

7. In addition to the existing 24 index snorkeling sites, add as many index sites as resources will allow. This would improve precision for both species and all age classes. Adding sites in addition the 24 sampled in 2014 and 2015 is secondary to adult salmon mark-recapture experiments.

8. 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

9. 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.

10. 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.

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8. References

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

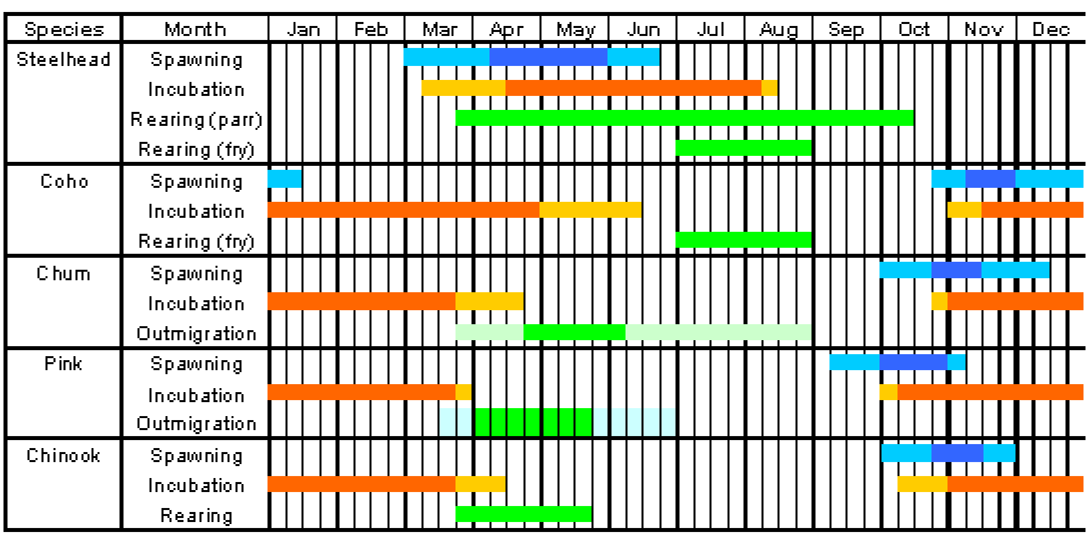

Figure 1.1 Life stage periodicity chart for anadromous salmonids in Coquitlam River.

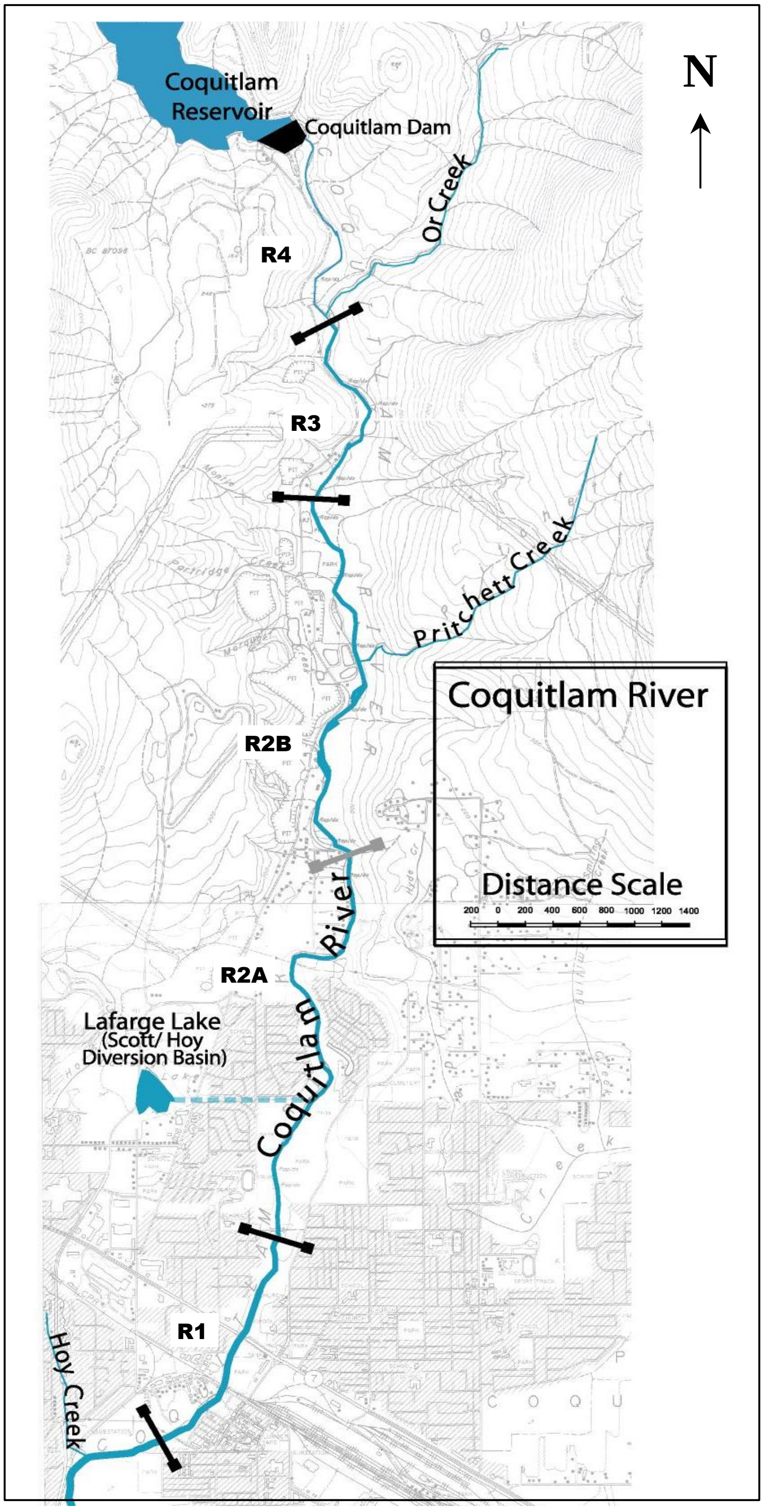

Figure 1.2 Map of lower Coquitlam River study area with stream reaches defined by the Coquitlam-Buntzen Water Use Plan Consultative Committee.

9. Figures, Tables and Appendices

9.2 Figures, Tables and Appendices for Chapter 2

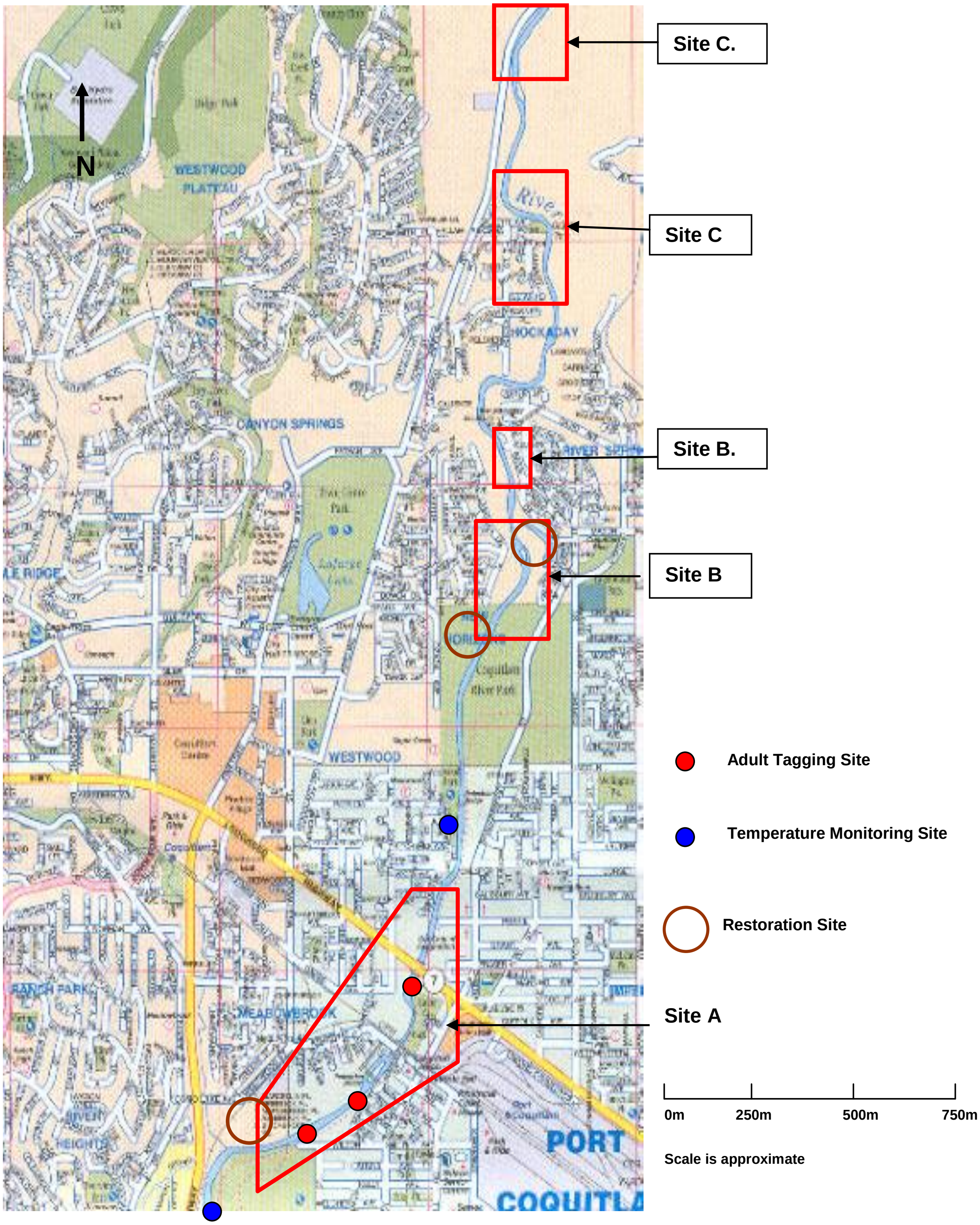

Figure 2.1 Map showing adult spawning index sites A-C in the lower portion of Coquitlam River study area (reaches 1, 2a)

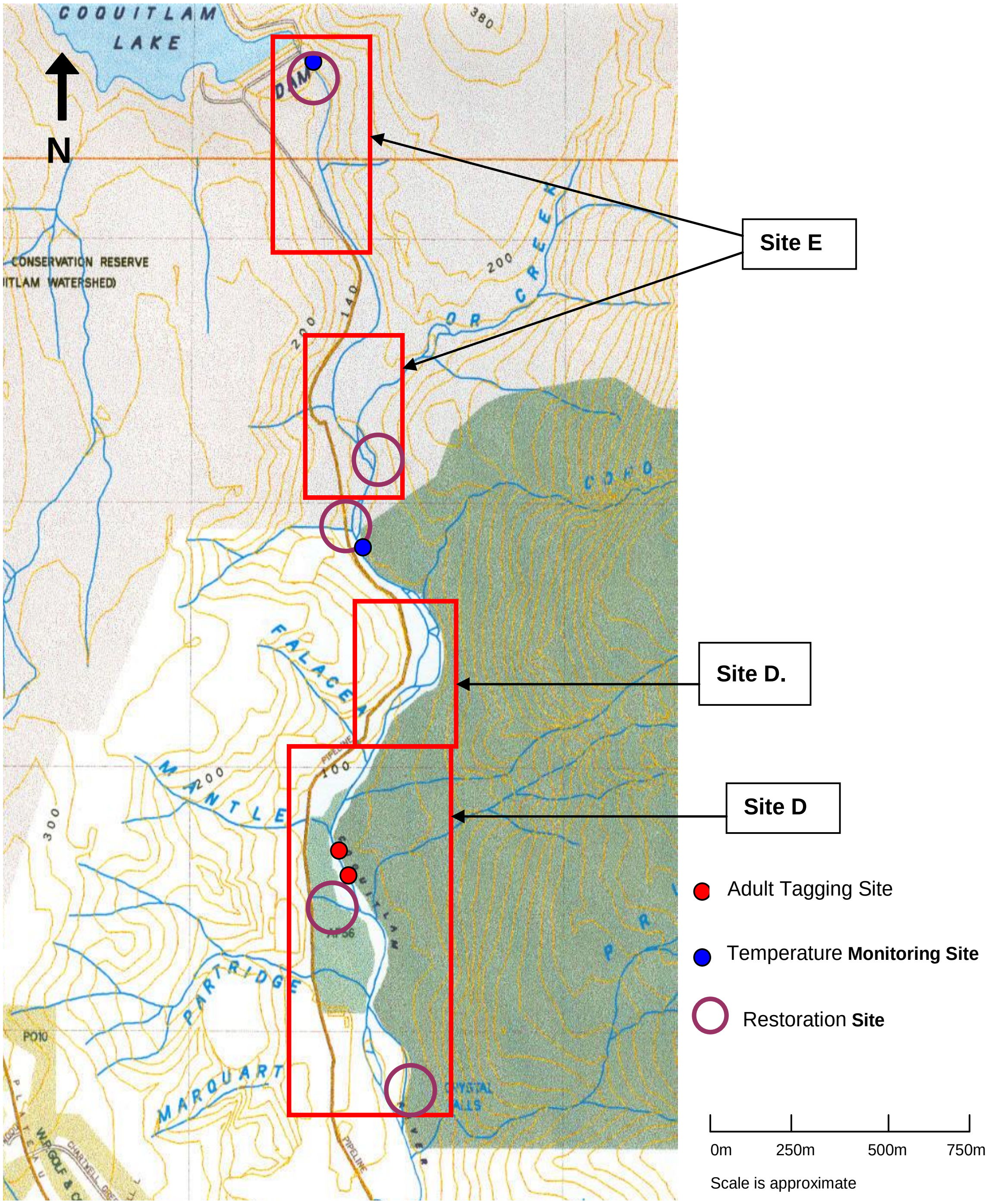

Figure 2.2 Map showing adult spawning index sites D and E, in the upper portion of Coquitlam River study area (reaches 2b, 3 and 4).

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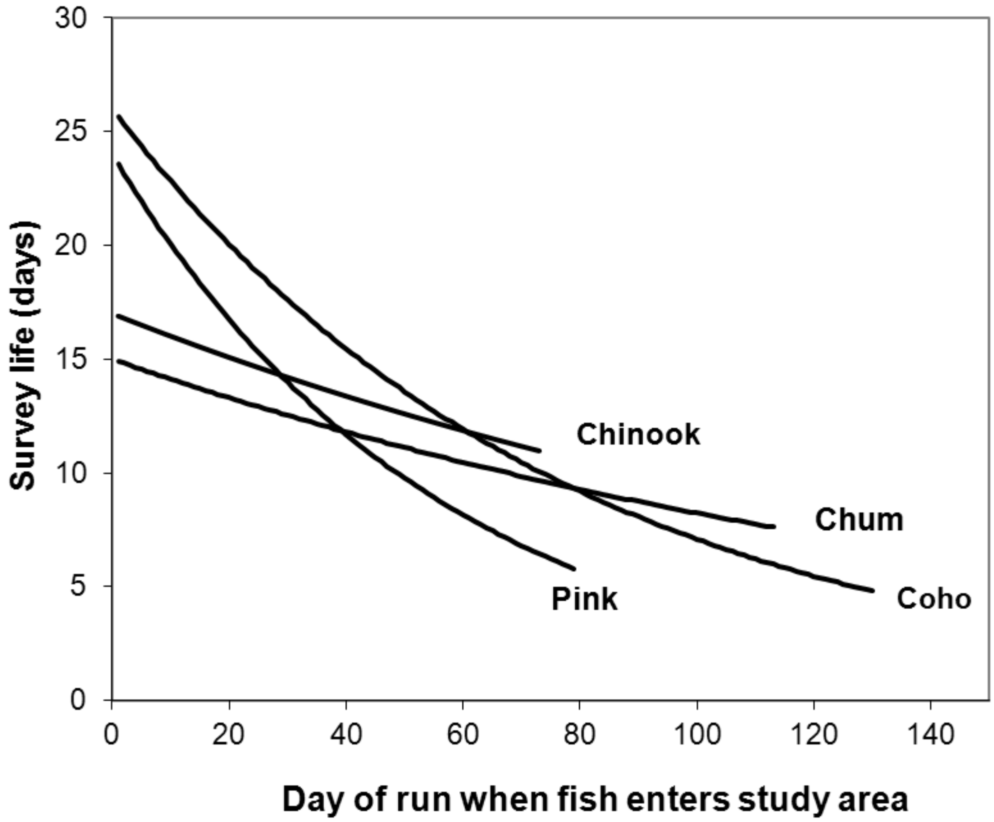

Figure 2.3 Modeled relationship between mean survey life and day of arrival in the study area for Chum, Pink, Coho, and Chinook salmon in the Coquitlam River based on empirical data from other streams.

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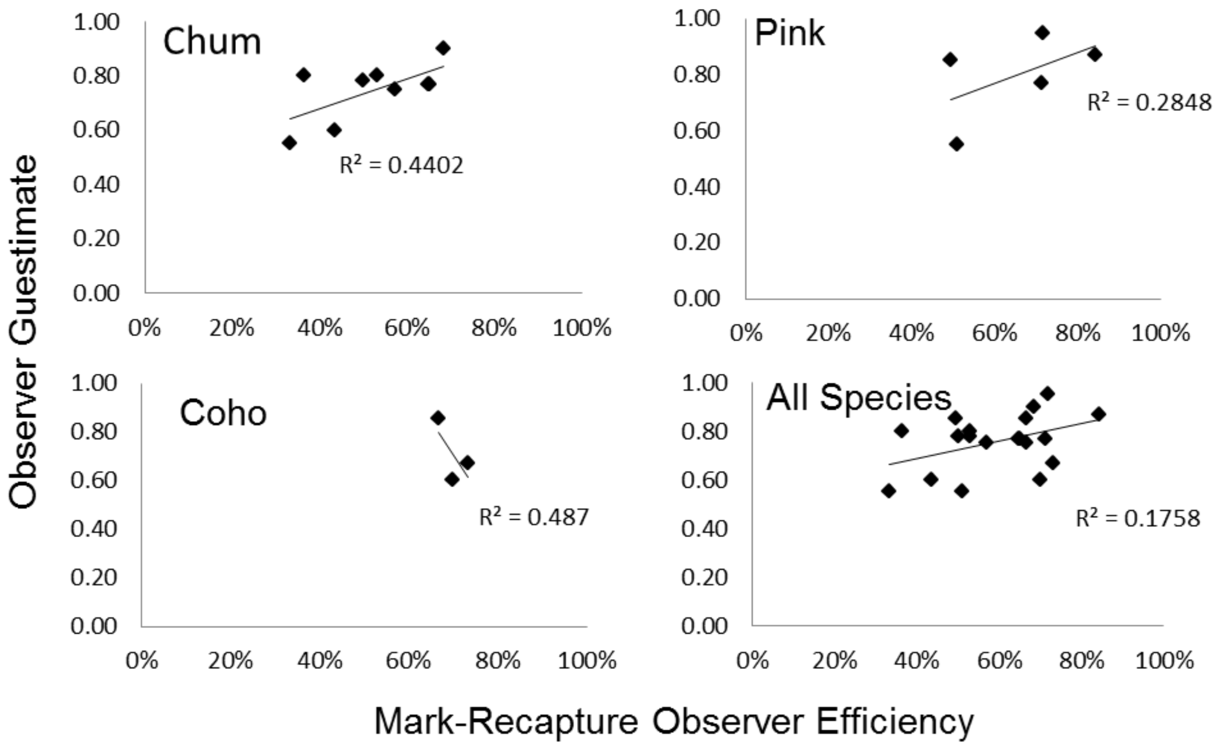


Figure 2.4 Relationship between the surveyor's 'guesstimate' of observer efficiency and observer efficiency estimated from mark recapture experiments for Chum, Pink and all species combined conducted opportunistically since 2006 in the Coquitlam River.

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 2014 brood year.

		Estimated water column visibility (m)					
Escapement Year	Date	site A	site B	site C	site D	site E	non-index
2014	17-Sep	>3	>3	>3	>3	>3	
2014	26-Sep	>3	>3	>3	>3	>3	>3
2014	07-Oct	1.2	1.2	1.2	1.2	1.2	
2014	14-Oct				0.7	0.7	
2014	18-Oct	0.8	0.8	0.9	1.0	1.1	1
2014	23-Oct			0.9	1.0	1	
2014	30-Oct			0.9	1	1	
2014	02-Nov	1.2	1.3	1.4	1.4	1.6	1.4
2014	13-Nov	1.2	1.2	1.2	1.3	1.4	
2014	18-Nov	1.2	1.3	1.4	1.4	1.6	1.3
2014	29-Nov			0.8	0.9	1.1	
2014	03-Dec	1.2	1.2	1.2	1.3	1.4	
2014	08-Dec	1.1	1.1	1.1	1.3	1.4	1.3
2014	20-Dec		0.9	1.0	1.1	1.2	
2014	29-Dec		1.1	1.2	1.3	1.3	
2014	Jan. 4		1.0	1.1	1.2	1.2	
2014	Jan. 9		1.0	1.1	1.2	1.2	
2014	Jan. 18		1.0	1.1	1.2	1.4	
2014	Jan. 23		1.0	1.0	1.3	1.4	

Surveyor "guesstimates" of observer efficiency (0.0-1.0): (chum salmon example)							
2014	17-Sep	0.95	0.95	0.95	0.95	0.95	
2014	26-Sep	0.95	0.95	0.95	0.95	0.95	0.95
2014	07-Oct	0.75	0.75	0.75	0.75	0.75	
2014	14-Oct				0.40	0.5	
2014	18-Oct	0.6	0.55	0.6	0.65	0.70	0.65
2014	23-Oct			0.6	0.60	0.65	
2014	30-Oct			0.6	0.60	0.65	
2014	02-Nov	0.7	0.7	0.8	0.80	0.85	0.80
2014	13-Nov	0.7	0.7	0.75	0.75	0.80	
2014	18-Nov	0.7	0.7	0.80	0.8	0.85	0.75
2014	29-Nov			0.60	0.65	0.75	
2014	03-Dec	0.70	0.70	0.80	0.80	0.85	
2014	08-Dec	0.65	0.6	0.65	0.75	0.80	0.75
2014	20-Dec		0.55	0.6	0.65	0.75	
2014	29-Dec		0.65	0.70	0.75	0.80	
2014	Jan. 4		0.60	0.7	0.70	0.75	
2014	Jan. 9		0.60	0.65	0.70	0.75	
2014	Jan. 18		0.60	0.60	0.70	0.8	
2014	Jan. 23			0.60	0.75	0.8	

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	9	5	3	2	19
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	0.80	0.71	0.77	0.76
minimum	0.55	0.55	0.60	0.75	0.55
maximum	0.90	0.95	0.85	0.78	0.95
Survey life (days)					
Number of estimates	6	4	3	2	
mean of estimates	7.5	13.0	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 2002-2014.

Species	Site	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
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
	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
	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
	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
	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
	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
Pink	A	-	0.16	-	0.17	-	0.21	-	0.13	-	0.17	-	0.21	-
	B	-	0.10	-	0.05	-	0.03	-	0.06	-	0.02	-	0.05	-
	C	-	0.20	-	0.11	-	0.08	-	0.12	-	0.06	-	0.13	-
	D	-	0.21	-	0.20	-	0.24	-	0.25	-	0.19	-	0.22	-
	E	-	0.24	-	0.42	-	0.33	-	0.36	-	0.53	-	0.32	-
	NI	-	0.10	-	0.05	-	0.11	-	0.08	-	0.04	-	0.07	-
Coho	A	0.00	<0.001	0.02	0.09	<0.001	0.01	0.05	<0.001	<0.01	0.01	0.00	0.00	0.01
	B	0.01	0.06	0.03	0.02	0.04	0.07	0.03	0.02	0.04	0.02	0.03	0.02	0.03
	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
	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
	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
	NI	<0.001	<0.001	0.06	0.02	0.05	<0.001	0.01	<0.001	0.07	0.05	0.05	0.06	0.09
Chinook	A						0.02	0.02	0.06	0.04	0.03	0.02	0.03	0.01
	B						0.00	0.01	0.03	0.01	0.02	0.00	0.02	0.00
	C						0.10	0.05	0.08	0.07	0.07	0.00	0.07	0.02
	D						0.06	0.11	0.09	0.18	0.22	0.08	0.22	0.10
	E						0.64	0.76	0.70	0.60	0.61	0.84	0.61	0.86
	NI						0.18	0.05	0.04	0.10	0.06	0.06	0.06	0.00

Table 2.4 Annual escapement estimates for Chum, Pink, Coho and Chinook salmon for the years 2002-2014.

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	
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

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	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	0.76
	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	0.15
	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	0.09
	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	0.24
Pink	M/S	0.55		0.65		0.71	0.64		0.76		0.59		0.77		0.71
	NOC	0.19		0.22		0.20	0.20		0.12		0.22		0.12		0.15
	OCR	0.26		0.13		0.09	0.16		0.12		0.19		0.11		0.14
	OC	0.45		0.35		0.29	0.36		0.24		0.41		0.23		0.29
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	0.88
	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	0.01
	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	0.11
	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	0.12
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	0.95
	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	0.02
	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
	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	0.05

Appendix 2.1. Results of the 2006-2013 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	3	2013	A/D	2	Oct 28	Oct 29	1	92	60	65%	0.77	48%	33	2	0	23	2
chum	3	2013	A/D	2	Oct 29	Nov 6	9	92	22	24%	0.77	48%	9	2	1	7	3

9. Figures, Tables and Appendices

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	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
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
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 2003-2013.

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
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	4-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	2-Nov	60	6	0	0	0	0	0	0
2003	7-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	5-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	1-Nov	59	1	-	-	-	-	0	-
2005	9-Nov	67	2	-	0	-	0	-	-
2005	16-Nov	74	6	0	0	0	0	0	0
2007	4-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	3-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	6-Nov	64	5	1	0	0	0	0	-
2007	29-Nov	87	5	0	0	0	0	0	-
2009	3-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	7-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	4-Nov	65	5	0	0	0	0	0	-

Appendix 2.2 continued (Pink)

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	4-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	1-Nov	62	6	3	1	0	0	41	1
2011	6-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	5-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

Appendix 2.3 Unadjusted live counts of Chum salmon during 2002-2014.

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

Appendix 2.3 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)

			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
2011	04-Sep	2	5	0	0	0	0	0	-
2011	10-Sep	8	5	1	0	0	0	0	-
2011	17-Sep	15	5	0	0	1	0	0	-
2011	24-Sep	22	5	1	0	0	0	1	-
2011	10-Oct	38	6	238	51	63	36	7	9
2011	17-Oct	45	6	790	66	144	229	32	14
2011	25-Oct	53	6	3056	224	557	849	434	495
2011	01-Nov	60	6	6757	575	702	794	183	493
2011	06-Nov	65	6	3785	240	475	446	162	257
2011	15-Nov	74	5	692	132	185	230	42	-
2011	21-Nov	80	5	238	21	114	67	6	-
2011	01-Dec	90	6	23	5	33	8	0	0
2011	07-Dec	96	5	-	0	5	0	0	0
2011	19-Dec	108	5	-	0	0	0	0	0
2012	10-Sep	8	5	1	0	0	0	0	-
2012	17-Sep	15	5	3	0	0	1	0	-
2012	24-Sep	22	5	1	0	0	0	5	-
2012	30-Sep	28	5	81	2	4	20	31	-
2012	8-Oct	36	6	1349	93	747	1475	361	477
2012	14-Oct	42	3	-		928	1808	403	-
2012	15-Nov	74	6	224	214	108	273	65	27
2012	18-Nov	77	5	154	72	89	203	35	-
2012	25-Nov	84	6	25	7	25	11	0	13
2012	3-Dec	92	4	3	0	-	6	0	-
2012	9-Dec	98	6	0	0	1	0	0	0
2013	16-Sep	14	5	2	0	1	5	4	-
2013	27-Sep	25	6	14	10	20	10	0	0
2013	05-Oct	33	6	73	41	40	105	66	10
2013	11-Oct	39	6	570	57	89	207	159	68
2013	18-Oct	46	6	1928	127	490	1003	447	537
2013	23-Oct	51	6	3073	527	1020	1849	804	693
2013	29-Oct	57	6	4273	767	1288	3353	1136	681
2013	05-Nov	64	6	5212	534	1014	2110	980	605
2013	15-Nov	74	5	1682	88	353	885	380	-
2013	21-Nov	80	5	114	45	115	155	9	-
2013	27-Nov	86	5	27	5	33	8	0	-
2013	04-Dec	93	6	0	0	0	0	1	0
2013	11-Dec	100	5	0	2	0	0	1	-
2014	17-Sep	15	5	0	0	0	0	3	-
2014	26-Sep	24	6	4	0	0	1	2	0
2014	07-Oct	35	5	51	3	15	29	16	-
2014	14-Oct	42	2	-	-	-	27	62	-
2014	18-Oct	46	6	292	0	72	100	112	72
2014	23-Oct	51	3	-	-	76	174	148	-
2014	30-Oct	58	3	-	-	119	477	208	-
2014	02-Nov	61	6	1656	201	127	513	272	85
2014	13-Nov	72	5	160	37	43	80	28	-
2014	18-Nov	77	6	52	12	23	16	21	36
2014	29-Nov	88	3	-	-	11	1	0	-

Appendix 2.4 Unadjusted live counts of Coho salmon during 2002-2014.

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

Appendix 2.4 continued (Coho)

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

Appendix 2.4 continued (Coho)

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

Appendix 2.4 continued (Coho)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present: site A	site B	site C	site D	site E	non-index
2013	11-Oct	22	6	0	0	0	0	2	0
2013	18-Oct	29	6	0	0	4	18	11	0
2013	29-Oct	40	6	0	0	13	18	69	0
2013	05-Nov	47	6	3	27	144	126	378	-
2013	15-Nov	57	5	19	47	204	148	449	-
2013	21-Nov	63	5	14	48	243	161	619	-
2013	27-Nov	69	5	13	54	265	262	708	-
2013	04-Dec	76	6	4	43	117	268	1044	131
2013	11-Dec	83	5	0	30	82	318	1060	-
2013	18-Dec	90	6	0	23	128	224	919	54
2013	29-Dec	101	4	-	90	54	129	582	-
2013	5-Jan	108	4	-	70	42	77	365	-
2013	14-Jan	117	4	-	29	10	36	123	-
2013	25-Jan	128	4	-	3	2	13	28	-
2014	07-Oct	18	5	0	0	0	0	9	-
2014	14-Oct	25	3	-	-	0	0	16	-
2014	18-Oct	29	3	-	-	0	2	21	3
2014	23-Oct	34	4	4		9	11	43	-
2014	30-Oct	41	4	-	4	11	13	56	-
2014	02-Nov	44	6	2	0	9	23	106	31
2014	13-Nov	55	5	12	36	134	170	482	-
2014	18-Nov	60	6	6	34	105	252	538	94
2014	29-Nov	71	3	9	23	67	141	495	-
2014	03-Dec	75	5	2	20	47	115	352	-
2014	08-Dec	80	6	0	18	21	89	221	16
2014	20-Dec	92	4	0	12	7	32	123	-
2014	29-Dec	101	4	0	7	4	8	67	-
2014	04-Jan	107	4	0	6	2	8	44	-
2014	09-Jan	112	4	0	3	0	8	25	-
2014	18-Jan	121	4	0	0	0	4	16	-
2014	23-Jan	126	3	0	-	0	1	6	-

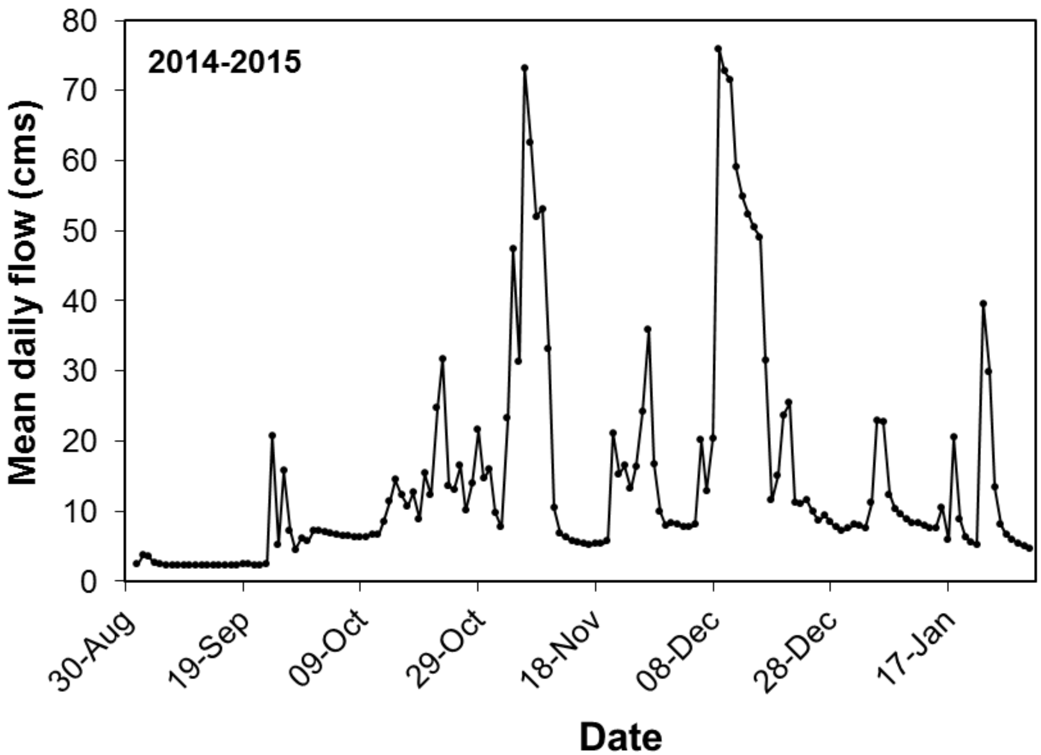
Appendix 2.5 Unadjusted live counts of Chinook salmon during 2007-2014.

			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

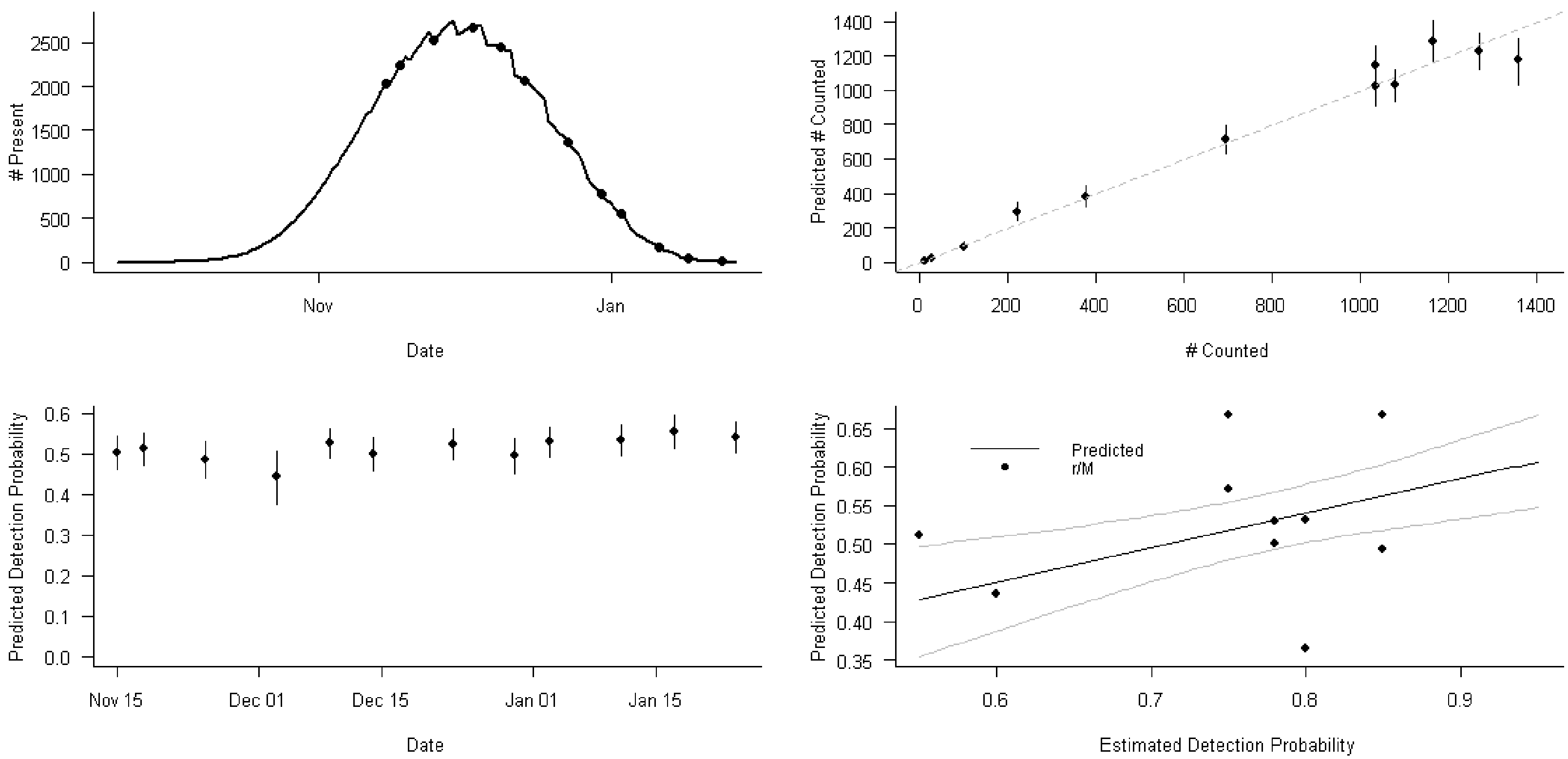
Appendix 2.5 continued (Chinook)

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

Appendix 2.6 Mean daily flows in Coquitlam River at Port Coquitlam during the fall and winter spawning period in 2014-2015 (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

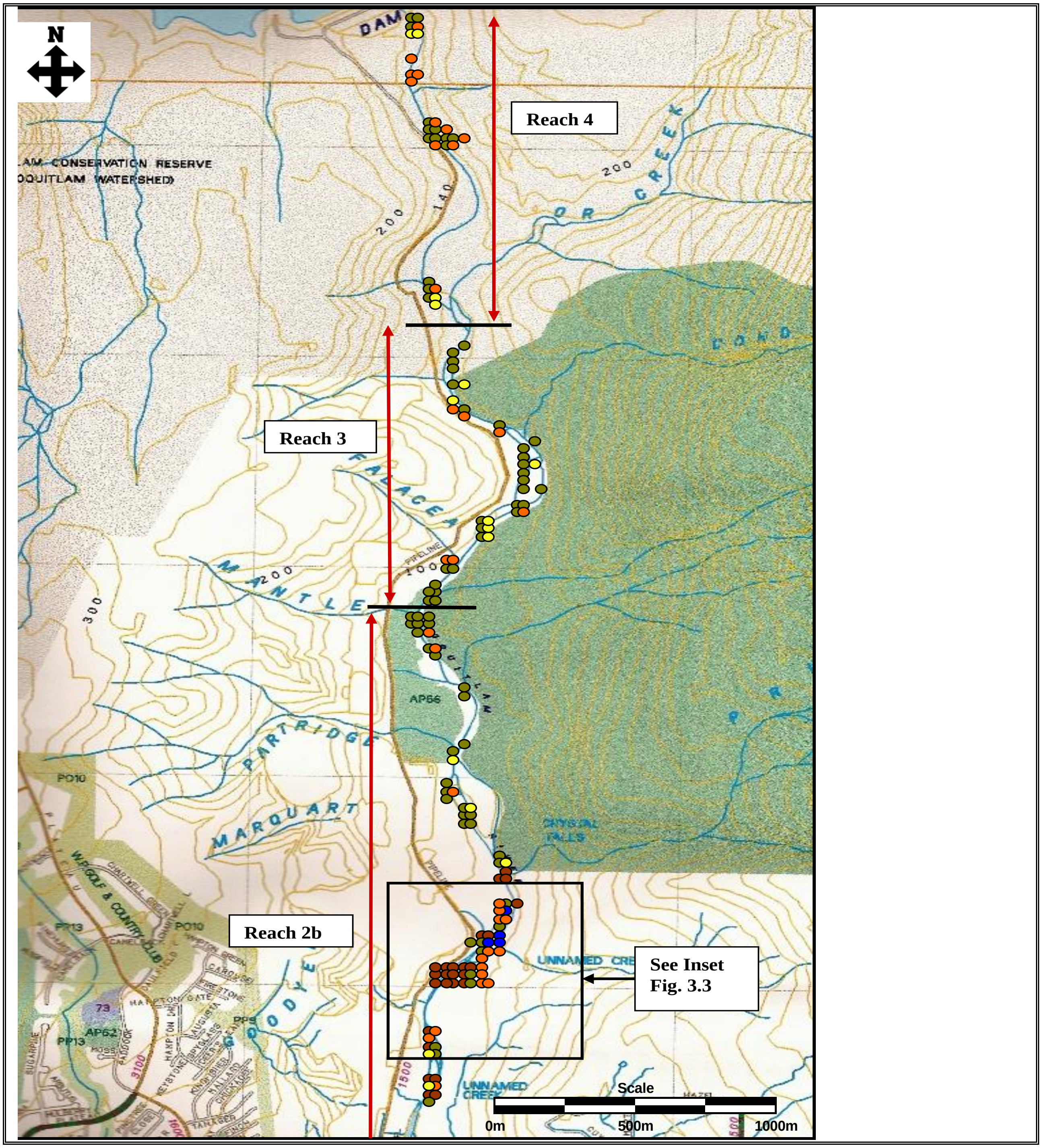


Figure 3.1 Steelhead redd locations in reaches 2b-4 in Coquitlam River in 2006, which was the highest escapement year during 2005-2013. Coquitlam Dam is the upstream boundary of the survey area. See Figure 3.2 for redd symbol legend.

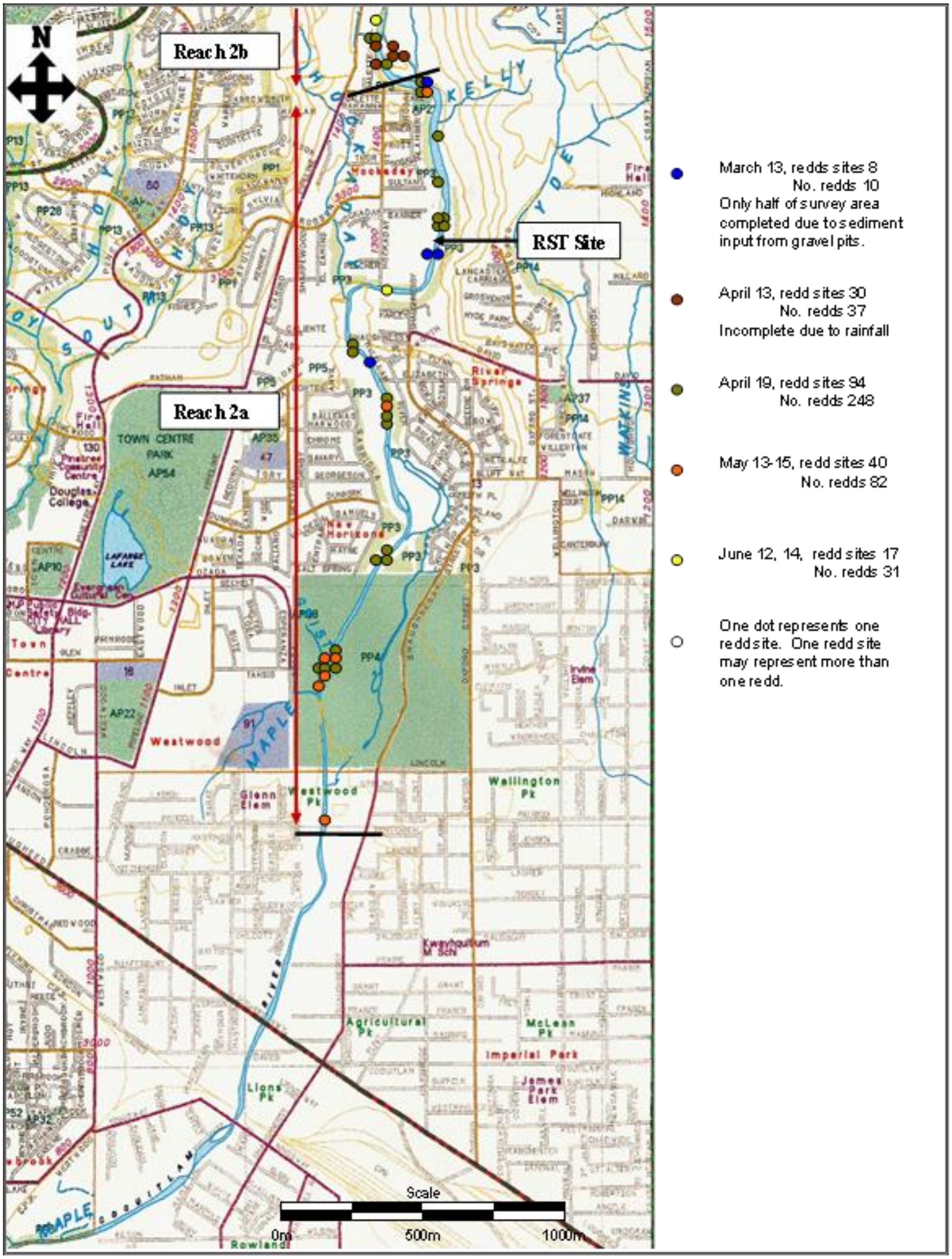


Figure 3.2 Steelhead redd locations in reaches 2a-2b in Coquitlam River in 2006. The downstream boundary of reach 2a is also the survey area boundary.

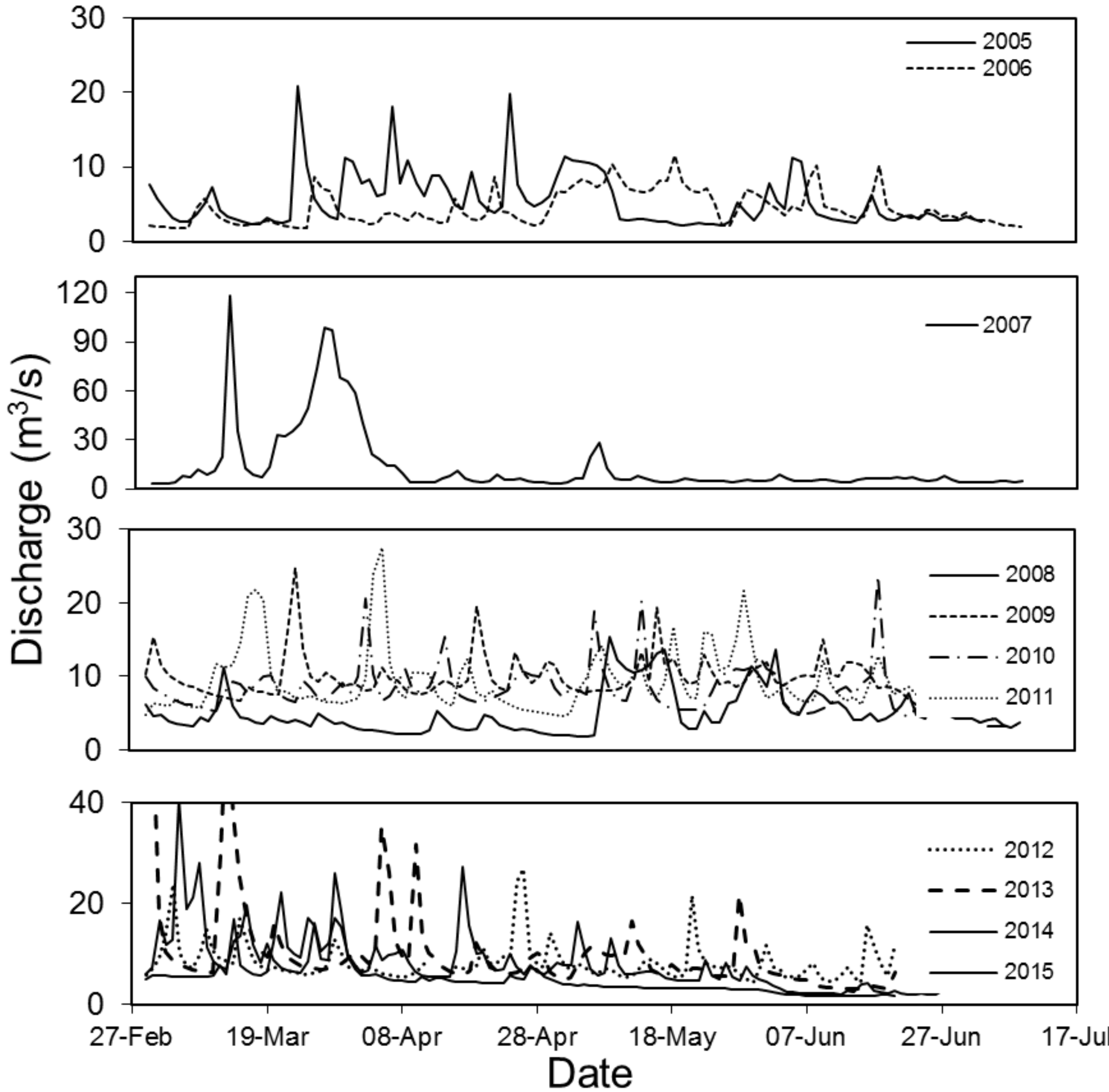


Figure 3.3 Discharge (cms) in Coquitlam River at Port Coquitlam during Steelhead spawning period in 2005 – 2015 (Water Survey of Canada station 08MH002).

9. Figures, Tables and Appendices

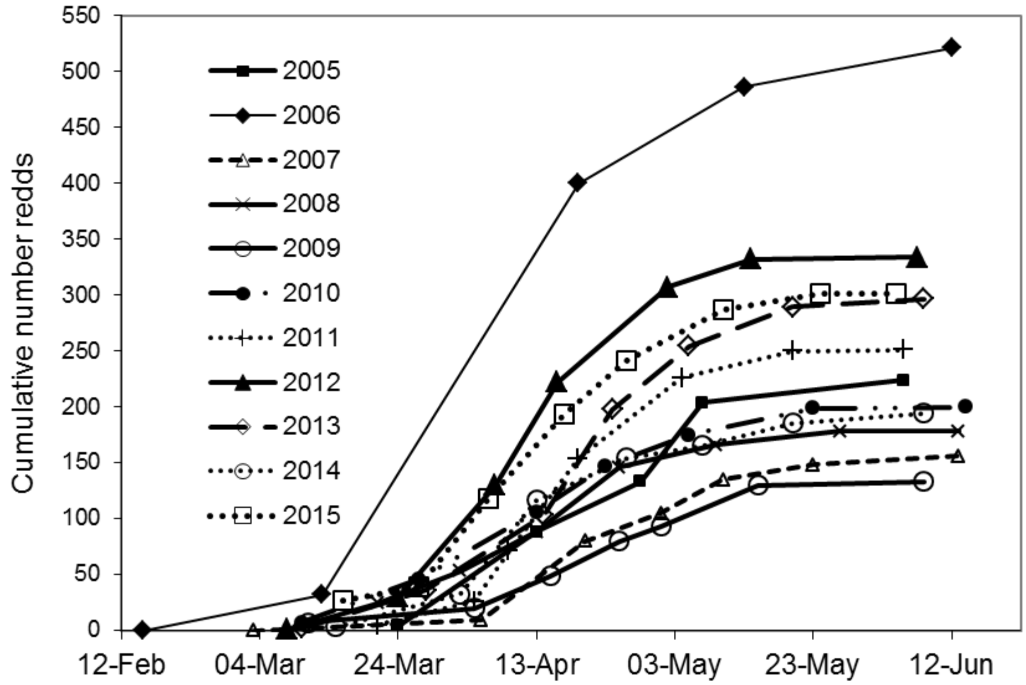


Figure 3.4 Cumulative proportion of the total Steelhead redd count observed over time during 2005 -2014.

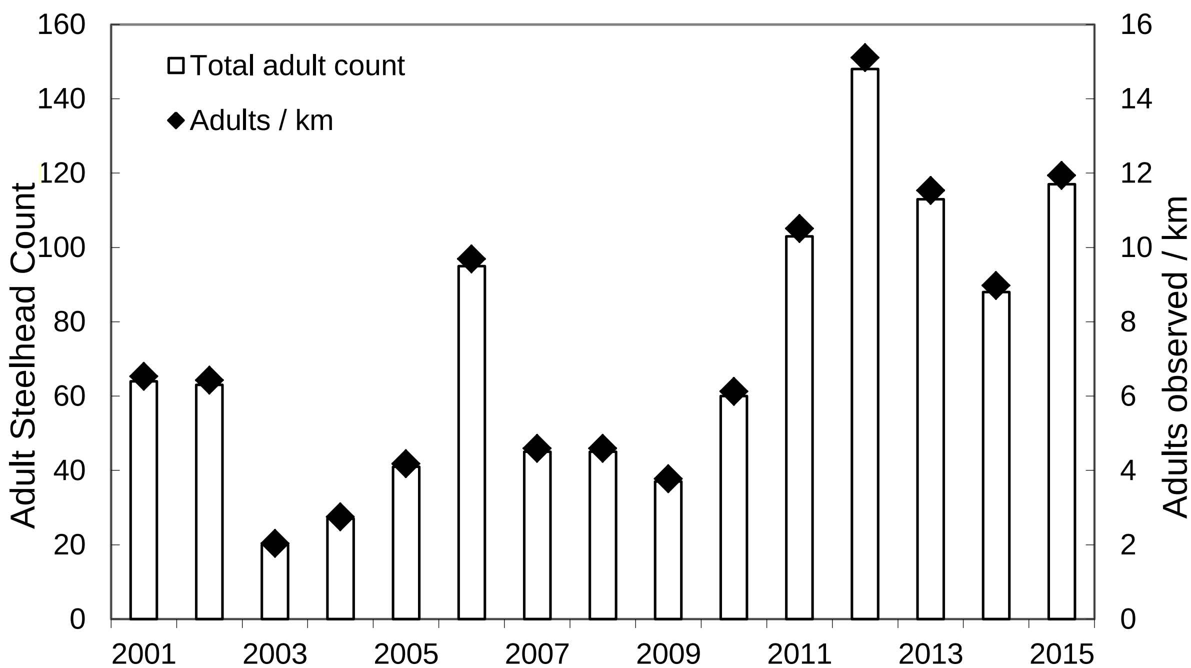


Figure 3.5 Peak counts and densities (fish/km) of adult Steelhead during snorkeling surveys in 2001-2015 (only data for complete surveys of the study area are shown). Data for 2001-2004 were collected as part of a separate study (BCCF, Lower Mainland Branch, data on file).

9. Figures, Tables and Appendices

Table 3.1 Survey dates with raw counts of Steelhead redds, estimated new redds, and live adult counts for all surveys during 2005-2014. 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	7-May	9	71	71	22 [2]
2005	5-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	2-Mar	-	0	0	20
2007	4-Apr	32	5	9	45
2007	19-Apr	15	68	71	43
2007	30-Apr	11	25	25	33
2007	9-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	2-Apr	12	29	29	37
2008	13-Apr	11	35	35	24
2008	25-Apr	12	58	58	45
2008	9-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	4-Apr	24	13	13	25
2009	15-Apr	11	29	29	23
2009	25-Apr	10	31	31	37
2009	1-May	6	13	13	20
2009	15-May	14	37	37	24
2009	8-Jun	24	3	3	4
2009	**Total**		**135**	**135**	**peak = 37**
2010	9-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	5-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**

Table 3.1 continued

Year	Survey date	Days since previous survey	Raw count of new redds	Estimated # new redds	# Live adults observed
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**
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**

[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 2005-2014 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

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

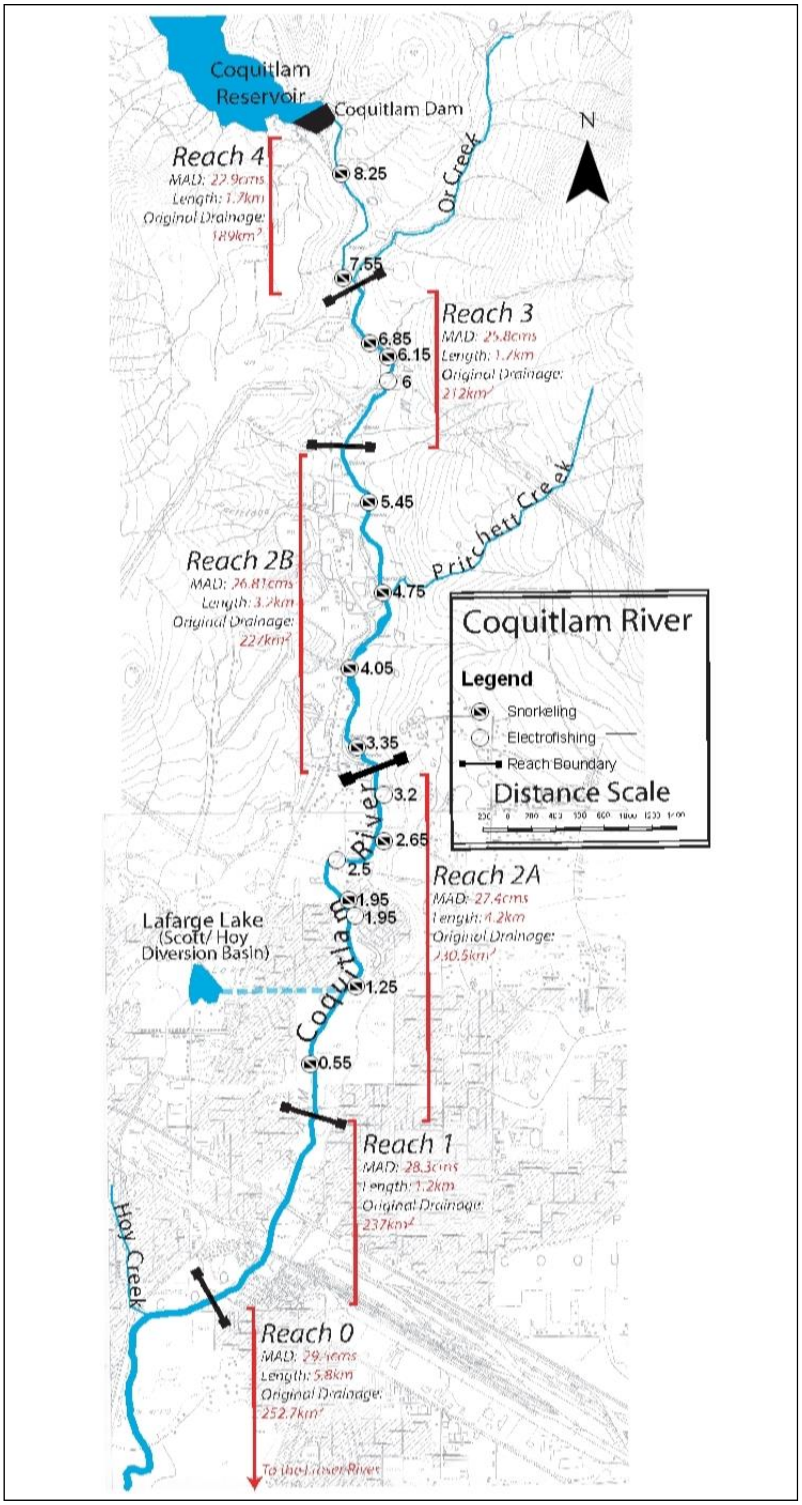


Figure 4.1 Map of Coquitlam River showing juvenile standing stock study area, reach breaks and original 12 sampling sites.

9. Figures, Tables and Appendices

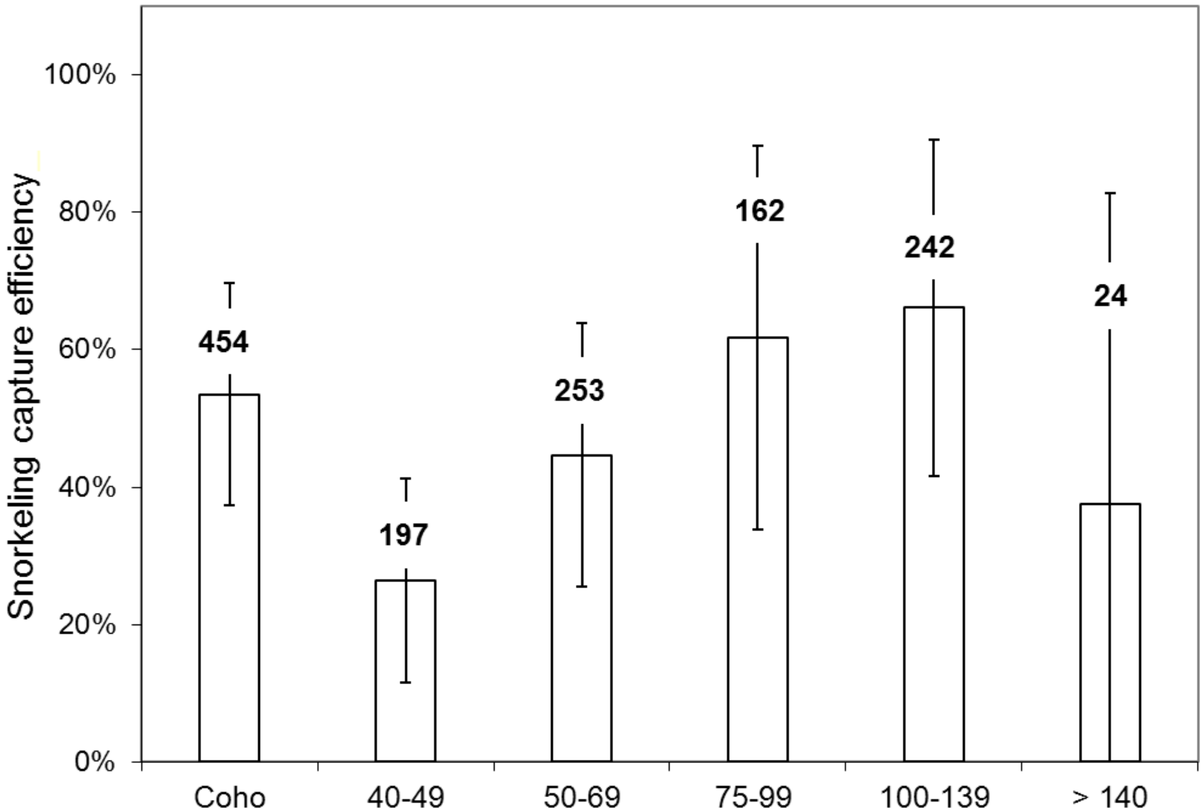


Figure 4.2 Maximum likelihood estimates of mean snorkeling detection probability for juvenile Coho and Steelhead by forklength class (Steelhead only) at 16 sites in the Coquitlam River during 2007-2013. Errors bars represent ±1 standard deviation of the mean. Values above bars are total numbers of marked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

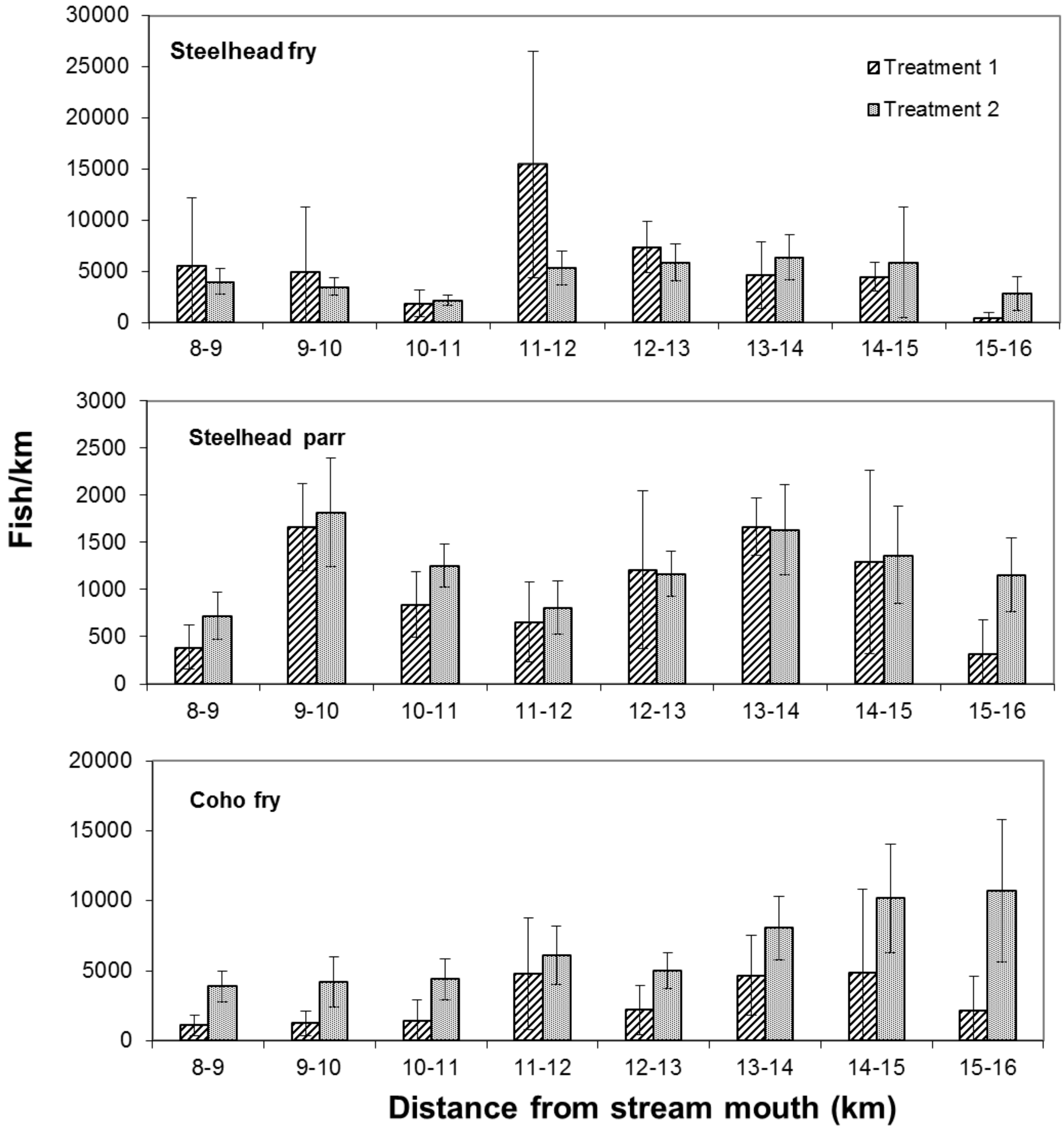


Figure 4.3 Linear distribution of juvenile salmonids in the Coquitlam River during Treatment 1(2006-2008) and Treatment 2 (2009-2015). Bars represent mean abundance estimates and 95% confidence intervals for years under flow Treatments 1 and 2. Estimates are based on calibrated snorkeling counts at 10-12 sampling sites 2006-2013 and 24 sites 2014-2015.

9. Figures, Tables and Appendices

Table 4.1 Summary of habitat data for night snorkeling and day electrofishing sites in Coquitlam River in 2015.

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	538.3	25	23	0.38	0.30	1.80	40	30	20	10	2015
snorkeling	0.90	8.60	408.3	25	15	0.56	0.30	1.80	30	40	20	10	2015
snorkeling	1.25	8.95	446.7	25	18	0.34	0.44	1.65	40	35	20	5	2015
snorkeling	1.60	9.30	737.5	25	31	0.29	0.26	1.30	35	35	25	5	2015
snorkeling	1.95	9.65	427.5	25	19	0.44	0.48	3.05	55	30	10	5	2015
snorkeling	2.65	10.35	416.7	25	17	0.61	0.26	1.90	50	30	10	10	2015
snorkeling	3.00	10.70	750.0	25	24	0.47	0.37	2.80	60	30	10	0	2015
snorkeling	3.35	11.05	569.2	25	15	0.35	0.37	1.20	10	45	35	10	2015
snorkeling	3.70	11.40	559.2	25	23	0.48	0.23	1.80	30	45	20	5	2015
snorkeling	4.05	11.75	900.0	25	35	0.24	0.28	1.30	25	50	20	5	2015
snorkeling	4.40	12.10	834.2	25	30	0.37	0.21	4.00	55	25	15	5	2015
snorkeling	4.75	12.45	446.7	25	18	0.41	0.37	1.70	35	40	20	5	2015
snorkeling	5.00	12.70	447.5	25	16	0.36	0.20	1.45	25	45	20	10	2015
snorkeling	5.20	12.90	616.7	25	22	0.24	0.25	1.10	35	45	15	5	2015
snorkeling	5.45	13.15	448.3	25	18	0.34	0.20	1.40	40	33	23	5	2015
snorkeling	5.60	13.30	451.7	25	19	0.28	0.34	2.50	35	35	25	5	2015
snorkeling	5.70	13.40	622.5	25	22	0.24	0.24	2.60	55	30	10	5	2015
snorkeling	5.80	13.50	481.7	25	18	0.35	0.24	1.60	45	35	15	5	2015
snorkeling	6.15	13.85	342.5	25	12	0.30	0.31	2.50	30	40	20	10	2015
snorkeling	6.85	14.55	472.5	25	12	0.68	0.12	1.70	40	30	20	10	2015
snorkeling	7.20	14.90	303.3	25	11	0.45	0.30	2.10	50	25	20	5	2015
snorkeling	7.55	15.25	452.5	25	16	0.34	0.20	1.65	43	40	13	5	2015
snorkeling	7.90	15.60	458.3	25	16	0.38	0.21	2.20	45	40	15	5	2015
snorkeling	8.25	15.95	420.8	25	14	0.22	0.27	1.45	48	43	8	3	2015
electrofishing	1.95	10.2	114.0	18	6	0.36	0.44	2.30	55	25	10	10	2015
electrofishing	2.50	10.7	133.8	19	7	0.36	0.41	1.40	35	40	20	5	2015
electrofishing	3.20	11.5	136.6	19	7	0.17	0.15	1.90	40	35	20	5	2015
electrofishing	6.00	14.5	126.8	20	7	0.34	0.33	1.50	30	40	20	10	2015

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

9. Figures, Tables and Appendices

Table 4.3 Estimates of juvenile fish density, standing stock, and 95% confidence intervals by species and age class in Coquitlam River during 2006-2015. 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	2,688	14.6	27,111	15,539	840,592	1522%
coho (0+)	2007	1,825	13.1	18,405	12,500	52,431	108%
coho (0+)	2008	4,632	28.5	46,719	32,478	89,047	61%
coho (0+)	2009	5,227	27.7	52,794	39,854	95,912	53%
coho (0+)	2010	5,934	32.1	62,178	41,717	130,782	72%
coho (0+)	2011	10,702	57.1	91,367	61,568	1,961,075	1039%
coho (0+)	2012	7,332	42.9	73,846	52,204	128,002	51%
coho (0+)	2013	7,018	39.2	70,279	53,293	127,942	53%
coho (0+)	2014	4,412	22.8	44,507	36,058	65,971	31%
coho (0+)	2015	3,231	15.7	36,101	28,520	45,660	31%
steelhead (0+)	2006	6,156	28.9	138,132	108,971	257,522	54%
steelhead (0+)	2007	1,440	9.4	32,251	22,193	139,860	182%
steelhead (0+)	2008	2,030	10.6	42,506	32,185	660,106	739%
steelhead (0+)	2009	1,768	9.5	37,047	29,002	1,355,054	1790%
steelhead (0+)	2010	1,891	9.5	39,657	29,627	151,626	154%
steelhead (0+)[1]	2011	2,131	11.2	21,949	-	-	-
steelhead (0+)	2012	2,641	14.5	55,232	40,520	81,398	37%
steelhead (0+)	2013	3,615	19.9	66,017	51,319	107,519	43%
steelhead (0+)	2014	1,574	7.9	32,746	26,499	44,724	28%
steelhead (0+)	2015	1,683	8.9	32,277	26,270	44,291	28%
steelhead (1+)	2006	576	2.9	5,976	3,532	22,859	162%
steelhead (1+)	2007	986	6.6	10,237	7,036	17,771	52%
steelhead (1+)	2008	919	5.8	10,222	7,446	20,770	65%
steelhead (1+)	2009	937	5.1	10,876	8,229	16,041	36%
steelhead (1+)	2010	701	3.9	8,106	6,556	10,710	26%
steelhead (1+)	2011	985	5.6	8,791	6,425	14,701	47%
steelhead (1+)	2012	1,026	6.0	10,668	8,002	17,462	44%
steelhead (1+)	2013	1,237	6.9	13,456	10,129	21,470	42%
steelhead (1+)	2014	614	3.2	6,369	5,115	8,669	28%
steelhead (1+)	2015	510	2.4	5,889	4,869	7,546	23%
steelhead (2+)	2006	200	1.1	1,841	933	3,569	72%
steelhead (2+)	2007	206	1.3	1,978	1,145	3,950	71%
steelhead (2+)	2008	203	1.3	1,255	694	2,598	76%
steelhead (2+)	2009	461	2.6	3,196	1,963	6,402	69%
steelhead (2+)	2010	390	2.2	2,690	1,630	5,331	69%
steelhead (2+)	2011	464	2.6	3,862	2,443	7,266	62%
steelhead (2+)	2012	329	1.8	3,160	1,961	5,666	59%
steelhead (2+)	2013	283	1.6	2,625	1,582	4,713	60%
steelhead (2+)	2014	398	2.1	3,831	2,756	6,634	51%
steelhead (2+)	2015	199	1.0	2,561	1,822	4,181	46%
steelhead (all parr)	2006	726	3.7	7,817	-	-	-
steelhead (all parr)	2007	1,147	7.6	12,215	-	-	-
steelhead (all parr)	2008	1,077	6.8	11,477	-	-	-
steelhead (all parr)	2009	1,278	7.0	14,072	-	-	-
steelhead (all parr)	2010	988	5.5	10,796	-	-	-
steelhead (all parr)	2011	1,328	7.5	12,653	-	-	-
steelhead (all parr)	2012	1,275	7.3	13,828	-	-	-
steelhead (all parr)	2013	1,456	8.1	16,081	-	-	-
steelhead (all parr)	2014	846	4.4	10,200	-	-	-
steelhead (all parr)	2015	659	3.2	8,450	-	-	-

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

Table 4.4 Summary of day electrofishing results at four one-shoreline sites in the Coquitlam River in 2015. Mean fish density estimates are also shown for 2006-2015. The electrofishing survey was conducted at the same four sites during 2007-2015, 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				
2015	1.95	6	0	0	6	6	6	5	625
2015	2.50	6	5	3	17	6	28	14	1,889
2015	3.20	14	4	3	21	19	23	17	2,609
2015	6.00	26	5	3	34	32	36	28	3,931
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
					Steelhead fry				
2015	1.95	19	9	4	34	29	39	30	1,771
2015	2.50	12	7	2	22	18	26	18	1,222
2015	3.20	11	5	3	20	16	24	16	1,242
2015	6.00	14	8	5	31	21	41	25	1,792
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	4179
2012	all sites							23	1704
2013	all sites							36	2418
2014	all sites							34	2364
2015	all sites							22	1507
					Steelhead parr (1+)				
2015	1.95	5	2	0	7	6	8	6.2	365
2015	2.50	3	3	1	7	4	10	5.7	389
2015	3.20	6	3	2	11	8	14	8.8	683
2015	6.00	5	2	0	7	6	8	5.7	405
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
					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

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

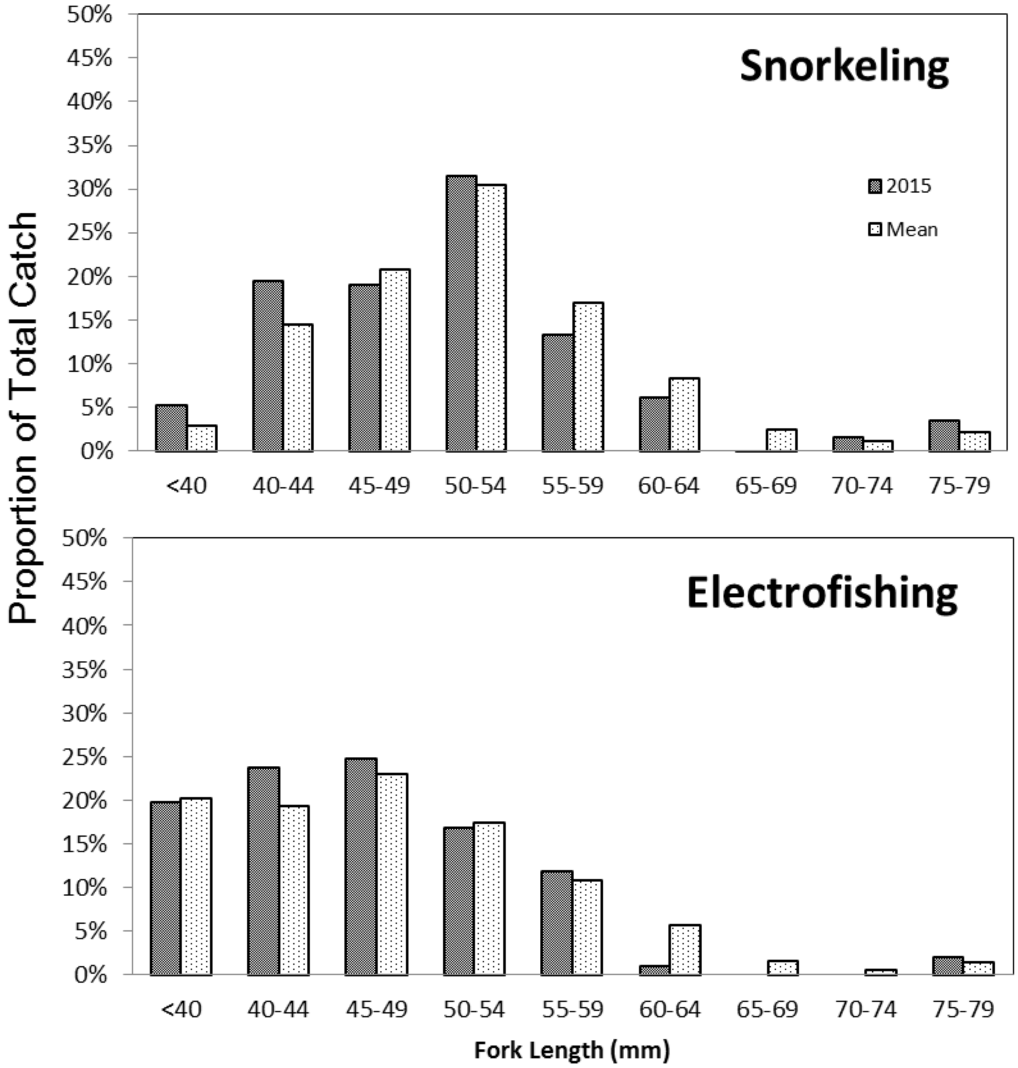
(4.10)	$\mu_{\theta,g} \sim dunif\,(0,1)$ $\sigma_{\theta,g} \sim dhcauchy(0,0.5)$
(4.11)	$\tau_{\theta,g} = \sigma_{\theta,g}^{-2}$ $\alpha_g = \mu_{\theta,g}\tau_{\theta,g}$ $\beta_g = (1-\mu_{\theta,g})\tau_{\theta,g}$
(4.12)	$\mu_\lambda \sim dnorm(0,0.1.0E-6)$ $\sigma_\lambda \sim dhcauchy(0,0.5)$
(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 substituted for counts) for Steelhead fry captured by electrofishing and counted during snorkeling in the Coquitlam River for the mean of 2008-2014 and 2015 (data pooled for all sites).



9.5 Figures, Tables and Appendices for Chapter 5

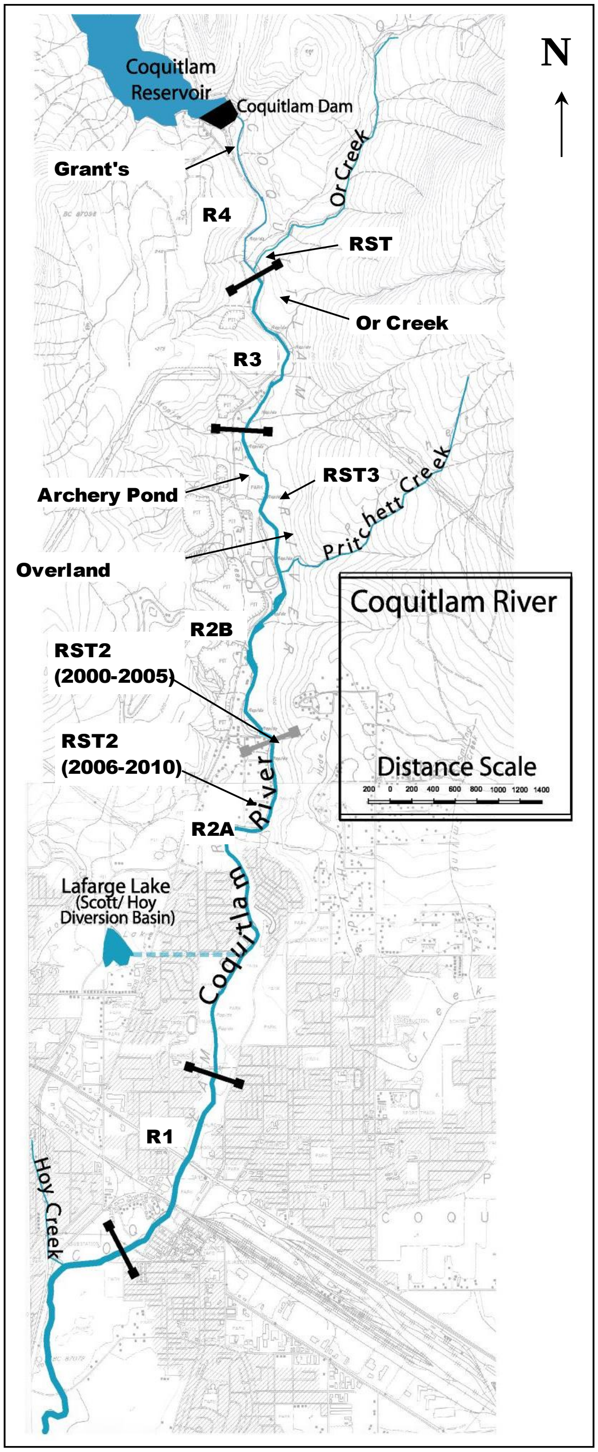


Figure 5.1 Map of the Coquitlam River showing constructed off-channel habitat sites, mainstem reach breaks and the locations of mainstem rotary screw traps (RSTs).

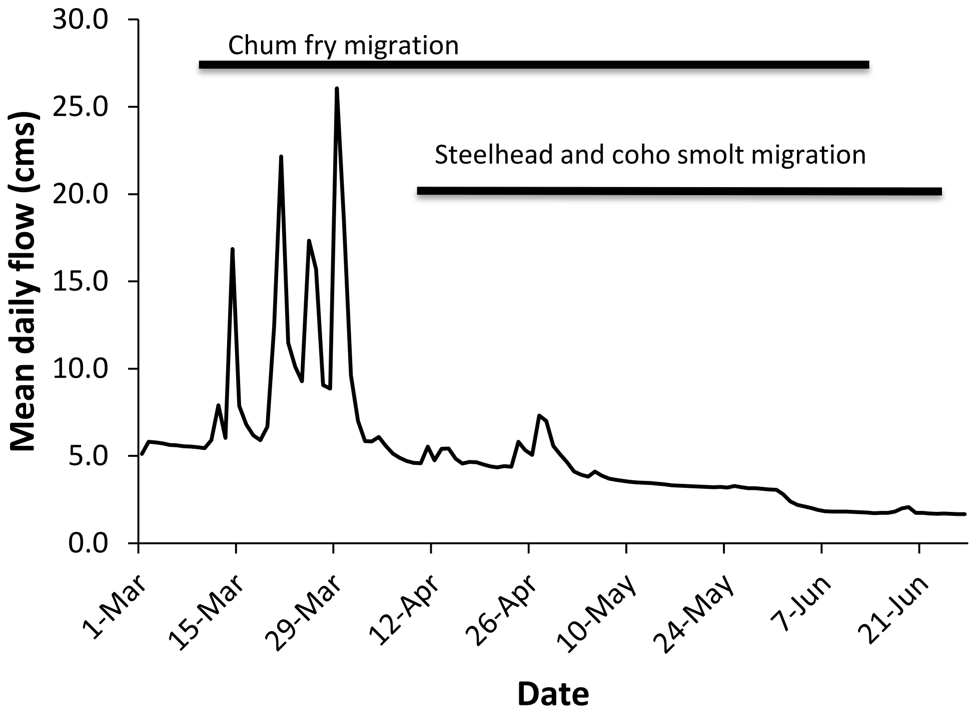


Figure 5.2 Mean daily flows in Coquitlam River at Port Coquitlam during the smolt trapping period in 2015. (Water Survey of Canada, stn. 08MH141). Approximate start times of Chum fry and Steelhead and Coho smolt migration based on captures at all trapping locations.

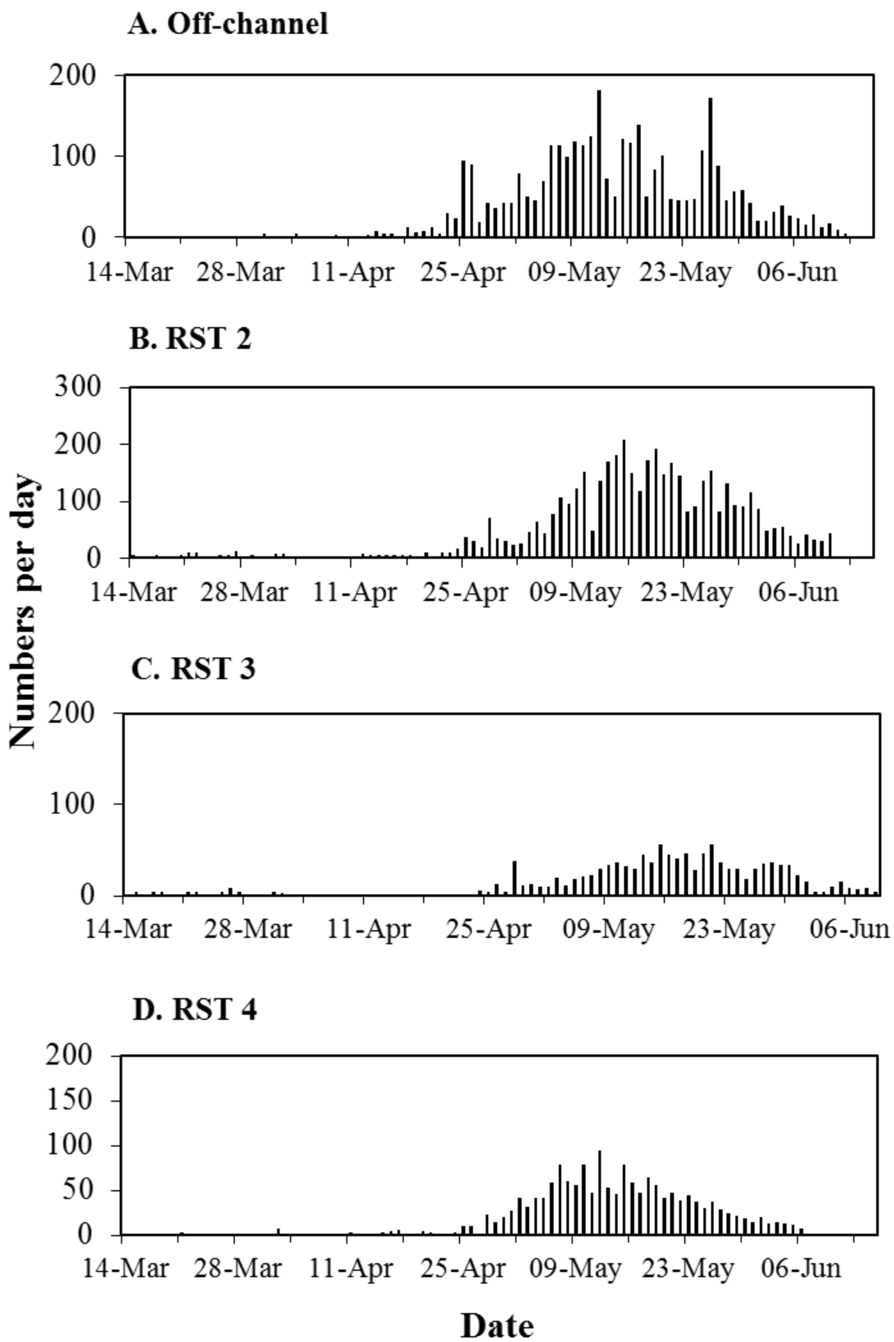


Figure 5.3 Daily catches of Coho smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2015. See Table 5.1 for start and end dates for individual trapping sites.

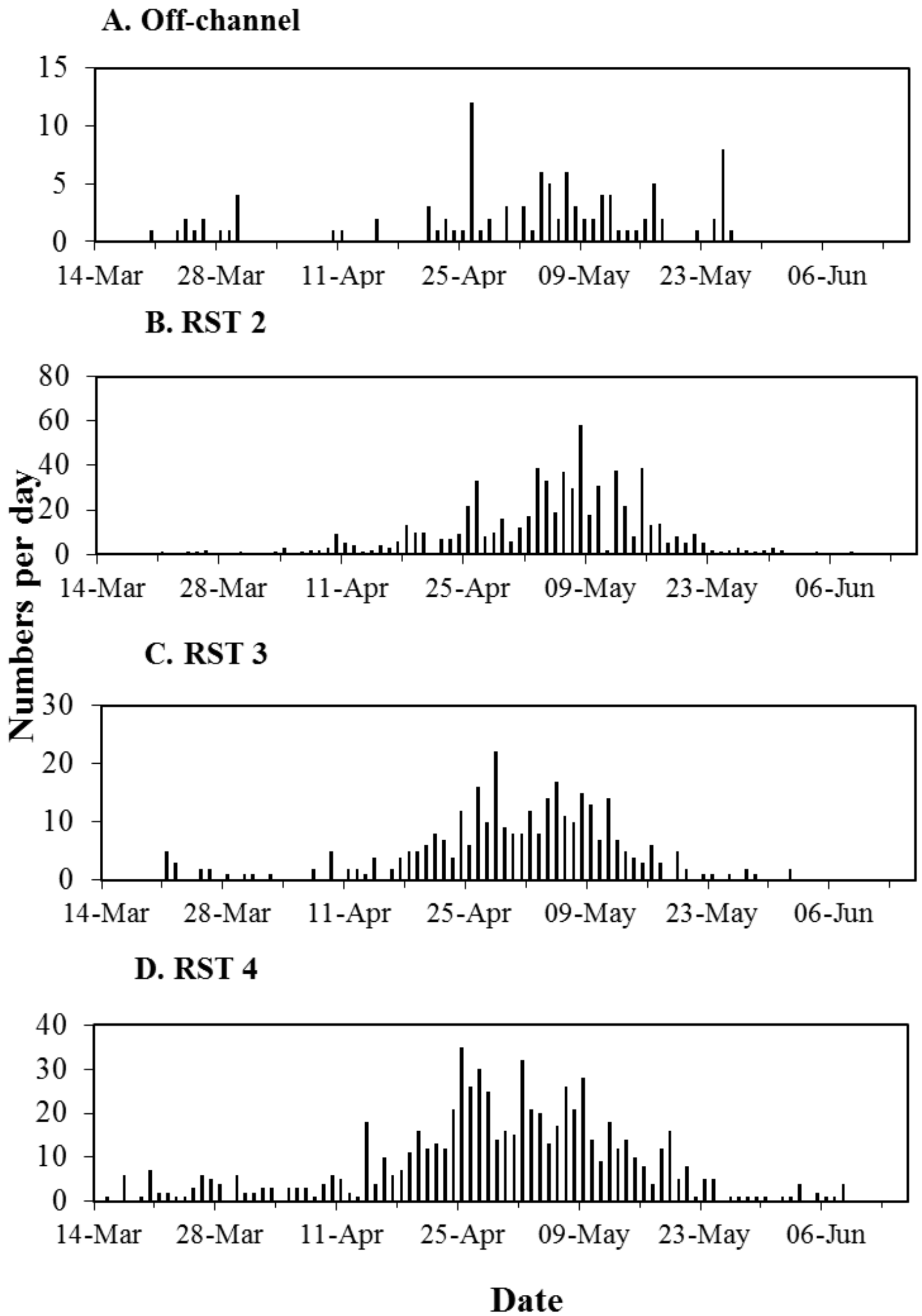


Figure 5.4 Daily catches of Steelhead smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2015. See Table 5.1 for start and end dates for individual trapping sites.

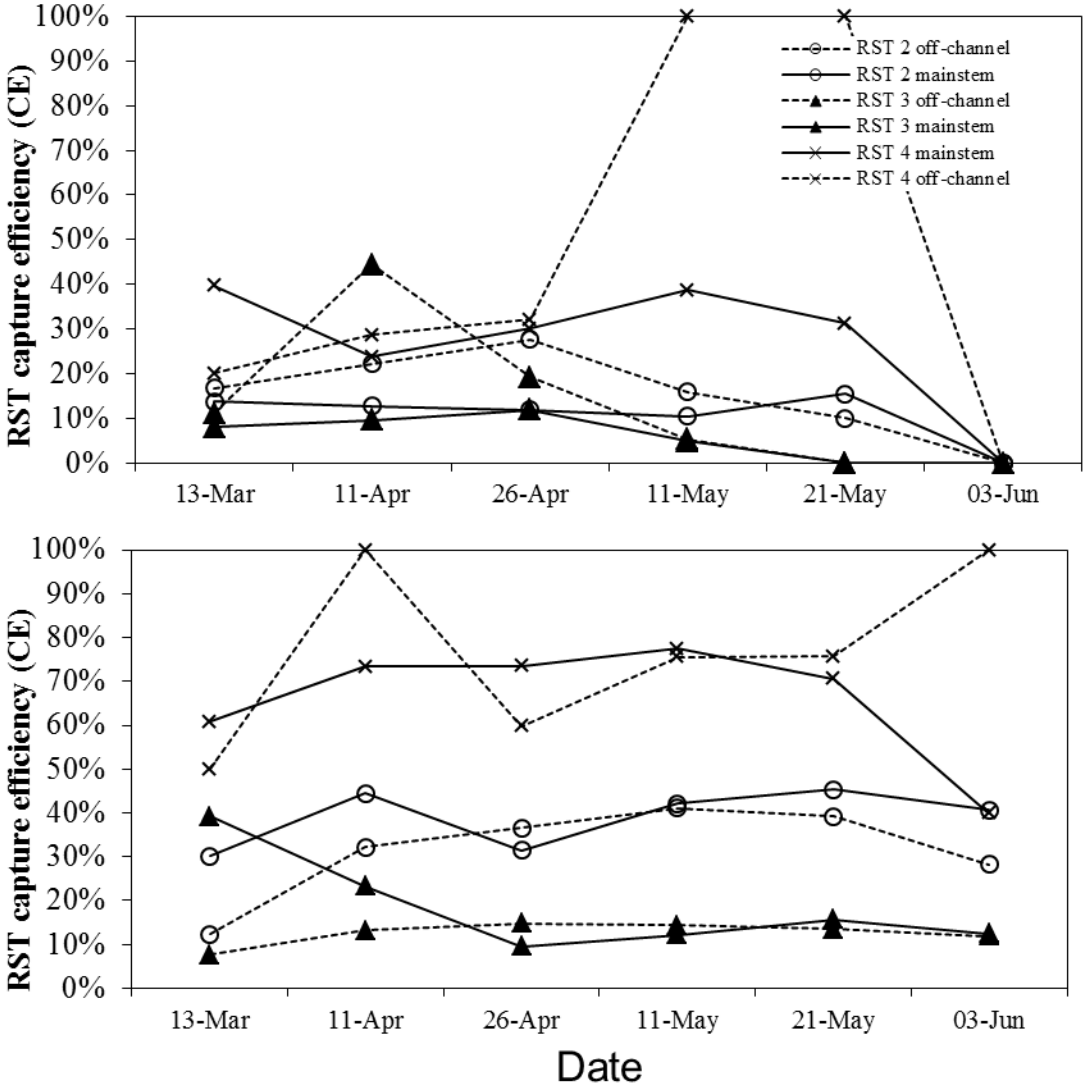


Figure 5.5 Estimated capture efficiencies (across six marking periods) at three rotary screw traps (RSTs) in the Coquitlam River for mark groups of Coho and Steelhead smolts from off-channel (dotted lines) and mainstem (solid lines) habitats in 2015. Dates on the horizontal axis indicate the start point for each marking period.

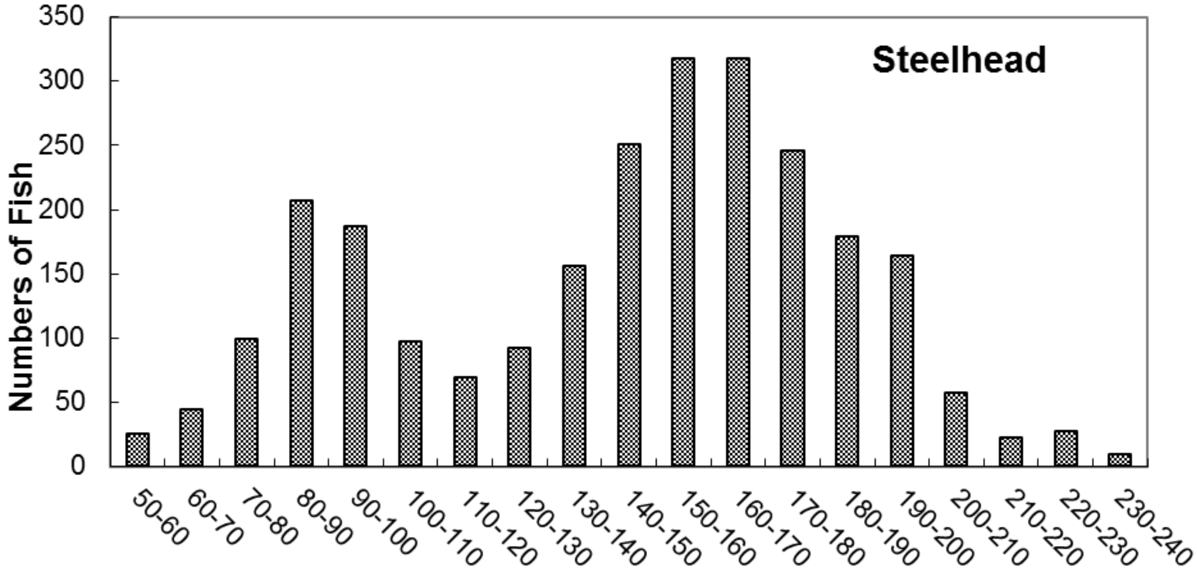


Figure 5.6 Length-frequency histogram for Steelhead captured in the Coquitlam River in 2015 (data pooled for all trap sites).

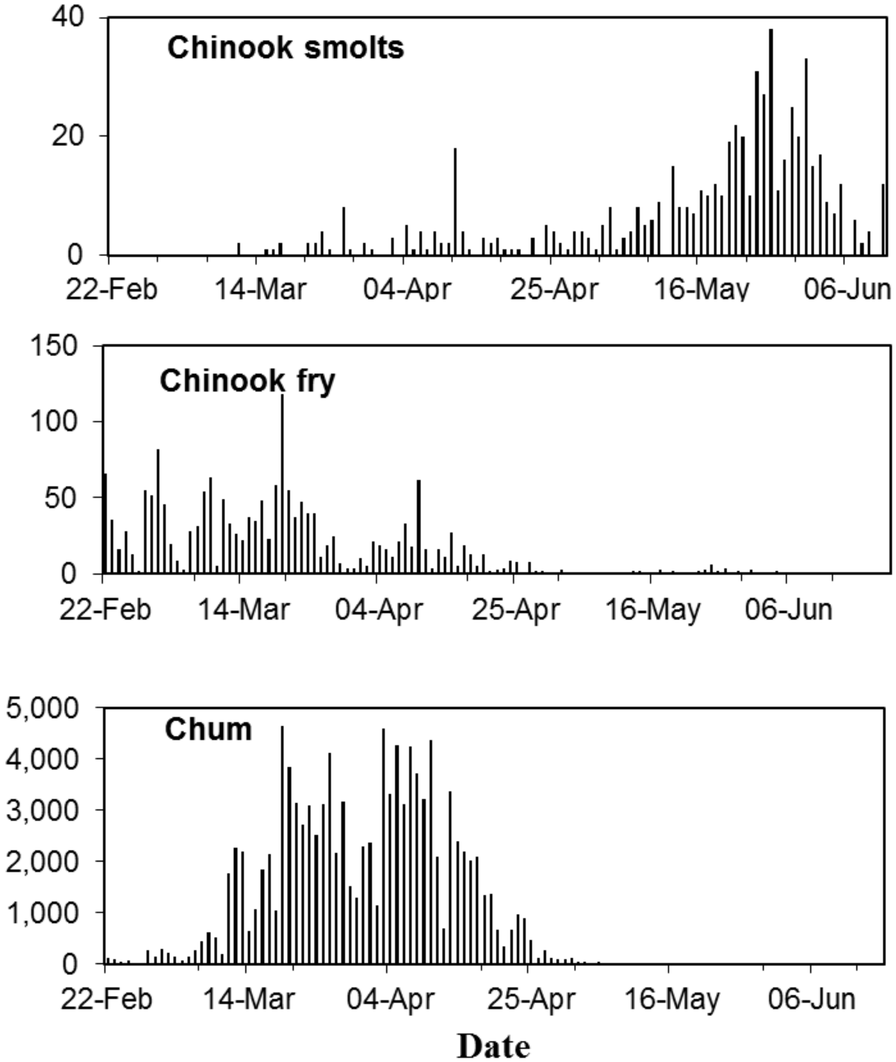


Figure 5.7 Daily catches of Chinook fry and smolts, and Chum fry at the RST2 trapping site in reach 2 in the Coquitlam River in 2015. See Table 5.1 for start and end dates of downstream trapping.

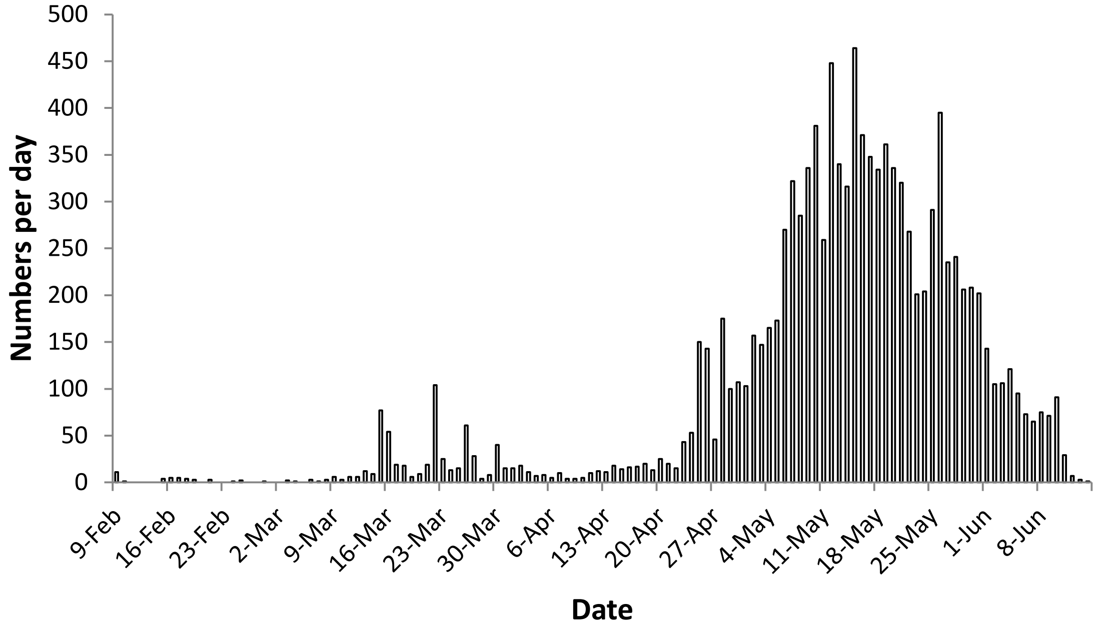


Figure 5.8 Daily catches of juvenile Coho combined for all trapping sites in the Coquitlam River in 2015. RST3 commenced operation early February, two months prior to full trap deployment, to monitor early season fish movement.

Table 5.1 Description of the stratification of fish marking by location and period for Coho and Steelhead smolts in the Coquitlam River in 2015. The start date for each temporal marking period at each RST trap site is also shown. Installation and removal dates are also given.

Downstream RST trapping site	Mark type by location	Mark stratification by period									Traps removed
		1	2	3	4	5	6	7	8	9	
Reach 2 (RST2.2, chum)	mark E	3-13	3-18	3-24	3-29	4-2	4-7	4-9	4-13	4-25	6-10
Reach 2 (RST2.4, coho, steelhead)	mark E	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-10
Reach 2 (RST2.5, coho, steelhead)	mark E	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-10
Reach 3 (RST3, coho, steelhead)	mark D	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-10
Reach 4 (RST4, coho, steelhead)	mark B	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-12
Archery Pond	mark A	3-13	4-11	4-26	5-11	6-3	-				6-15
Overland Ponds	mark A	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-10
Or Creek Ponds	mark A	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-15
Grants Tomb Pond	mark A	3-13	4-11	4-26	5-11	6-3	-	-	-	-	6-13

Table 5.2 Summary of estimated smolt numbers and densities by species in 2015 for three off-channel sites, reaches 2-4 of the Coquitlam River mainstem and the total Coquitlam River mainstem including and excluding the off-channel sites.

Site	Length (km)	Area (m^2)	N smolts	CI (+/-)	CI (%)	Density (no./100m^2)	Density (no./km)
Coho							
Off-channel sites							
Grant's Tomb	-	3,300	344	-	-	10.4	-
Or Creek	-	13,336	1,735	-	-	13.0	-
Overland Channel	-	4,500	1,085	-	-	24.1	-
Archery Pond	-	8,700	456	-	-	5.2	-
Total	-	21,136	3,620	-	-	17.1	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	2,821	830	29%	3.4	882
Reach 3, Coquitlam River	2.7	46,920	4,265	681	16%	9.1	1,580
Reach 4, Coquitlam River	1.6	19,200	1,148	31	2.7%	6.0	718
Total	7.5	149,898	8,234	476	6%	5.5	1,098
Coquitlam R.incl. off-channel	7.5	171,034	11,854	476	4%	6.9	1,581
Steelhead							
Off-channel sites							
Grant's Tomb	-	3,300	47	-	-	1.4	-
Or Creek site	-	13,336	47	-	-	0.4	-
Overland Channel	-	4,500	0	-	-	0.0	-
Archery Pond	-	8,700	18	-	-	0.2	-
Total	-	21,136	112	-	-	0.5	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	1,428	1,284	90%	1.7	446
Reach 3, Coquitlam River	2.7	46,920	1,730	1,197	69%	3.7	641
Reach 4, Coquitlam River	1.6	19,200	1,808	270	15%	9.4	1,130
Total	7.5	149,898	4,966	537	11%	3.3	662
Coquitlam R.incl. off-channel	7.5	171,034	5,078	453	9%	3.0	677
Chinook							
Off-channel sites							
Grant's Tomb	-	3,300	-	-	-	-	-
Or Creek site	-	13,336	5	-	-	0.04	-
Archery Pond	-	5,800	4	-	-	0.07	-
Overland Channel	-	4,500	-	-	-	-	-
Total	-	23,636	9	-	-	0.038	-
Mainstem	data too sparse to generate estimates, 584 captured at RST2						
Chum							
Coquitlam R.incl. off-channel	7.5	171,034	2,012,503	286,605	14%	1,177	268,334

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 2015. 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	0.38	0.36	0.07
Coho	RST 3	0.13	0.14	0.37
Coho	RST 4	0.73	0.71	0.50
Steelhead	RST 2	0.12	0.21	0.01
Steelhead	RST 3	0.09	0.16	0.06
Steelhead	RST 4	0.30	0.37	0.40

Appendix 5.1 Summary of estimated numbers of Coho, Steelhead and Chum smolts passing three RST trapping locations (not reach estimates) in the Coquitlam River mainstem in 2015. Mark group indicates the location where fish were initially captured and marked. Also shown are numbers of marked (M) and recaptured (R) smolts, estimated capture efficiencies (R/M), 95% confidence intervals, and percent relative errors.

Species	Site	Mark group(s)	M	R	U	Capture efficiency	N smolts	CI (+/-)	CI (%)
Coho	RST 2	mainstem	1,543	591	2,614	38%	8,234	476	5.8%
	RST 3	all	3,153	429	773	14%	5,948	2,103	35.4%
	RST 4	mainstem	801	587	848	73%	1,148	31	2.7%
Steelhead	RST 2	mainstem	1,242	150	512	12%	4,728	453	9.6%
	RST 3	mainstem	521	49	261	9%	3,309	1,119	33.8%
	RST 4	mainstem	512	155	540	30%	1,808	270	15.0%
Chum	RST 2	RST 2	17,150	1,023	96,337	6%	2,012,503	286,605	14.2%

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 2015.

Site	Mark group	Pooling	
Coho			
RST 2	mainstem (RST 3-4)	none	Darroch ML
RST 3	all	recovery 1-3	Darroch ML
RST 4	mainstem (RST 4)	recovery 1-2	Darroch ML
Steelhead			
RST 2	mainstem (RST2-4)	none	Darroch ML
RST 3	mainstem (RST 4)	dropped release and recovery 5	Darroch ML
RST 4	mainstem (RST 4)	recovery 1-2	Darroch ML
Chum			
RST 2	RST 2	none	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 2015. 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	6	Capture efficiency
1	73	10	1	3	5	2	1	30%
2	45	0	8	12	0	0	0	44%
3	572	0	0	104	66	9	1	31%
4	503	0	0	0	120	89	3	42%
5	318	0	0	0	0	117	27	45%
6	32	0	0	0	0	0	13	41%
Untagged Fish		86	94	557	934	751	192	

Recovery site: RST 3

All mark groups — Recovery strata

Release strata	Marks	1	2	3	4	5	6	Capture efficiency
1	49	2	0	3	2	3	1	22%
2	83	0	5	7	0	1	1	17%
3	999	0	0	59	65	6	0	13%
4	1083	0	0	0	87	63	0	14%
5	880	0	0	0	0	106	16	14%
6	108	0	0	0	0	0	13	12%
Untagged Fish		52	15	222	253	204	27	

Recovery site: RST 4

Mainstem mark group — Recovery strata

Release strata	Marks	1	2	3	4	5	6	Capture efficiency
1	23	5	1	4	2	1	1	61%
2	30	0	13	9	0	0	0	73%
3	351	0	0	194	62	2	0	74%
4	253	0	0	0	142	54	0	77%
5	129	0	0	0	0	74	17	71%
6	15	0	0	0	0	0	6	40%
Untagged Fish		23	31	357	260	150	27	

Appendix 5.3 continued

Steelhead

Recovery site: RST 2

Mainstem mark group		Recovery strata						
Release strata	Marks	1	2	3	4	5	6	Capture efficiency
1	102	6	4	2	2	0	0	14%
2	283	0	19	16	1	0	0	13%
3	613	0	0	58	15	0	0	12%
4	201	0	0	0	19	2	0	10%
5	39	0	0	0	0	6	0	15%
6	4	0	0	0	0	0	0	0
Untagged Fish		21	79	276	112	24	0	

Recovery site: RST 3

Mainstem mark group		Recovery strata						
Release strata	Marks	1	2	3	4	5	6	Capture efficiency
1	63	2	1	2	0	0	0	8%
2	147	0	5	9	0	0	0	10%
3	228	0	0	19	8	0	0	12%
4	63	0	0	0	2	1	0	5%
5	16	0	0	0	0	0	0	0%
6	4	0	0	0	0	0	0	0%
Untagged Fish		22	60	136	36	7	0	

Recovery site: RST 4

Mainstem mark group		Recovery strata						
Release strata	Marks	1	2	3	4	5	6	Capture efficiency
1	63	10	4	8	2	1	0	40%
2	147	0	19	16	0	0	0	24%
3	220	0	0	51	15	0	0	30%
4	62	0	0	0	17	7	0	39%
5	16	0	0	0	0	5	0	31%
6	4	0	0	0	0	0	0	0%
Untagged Fish		64	150	234	69	23	0	

Appendix 5.3 continued

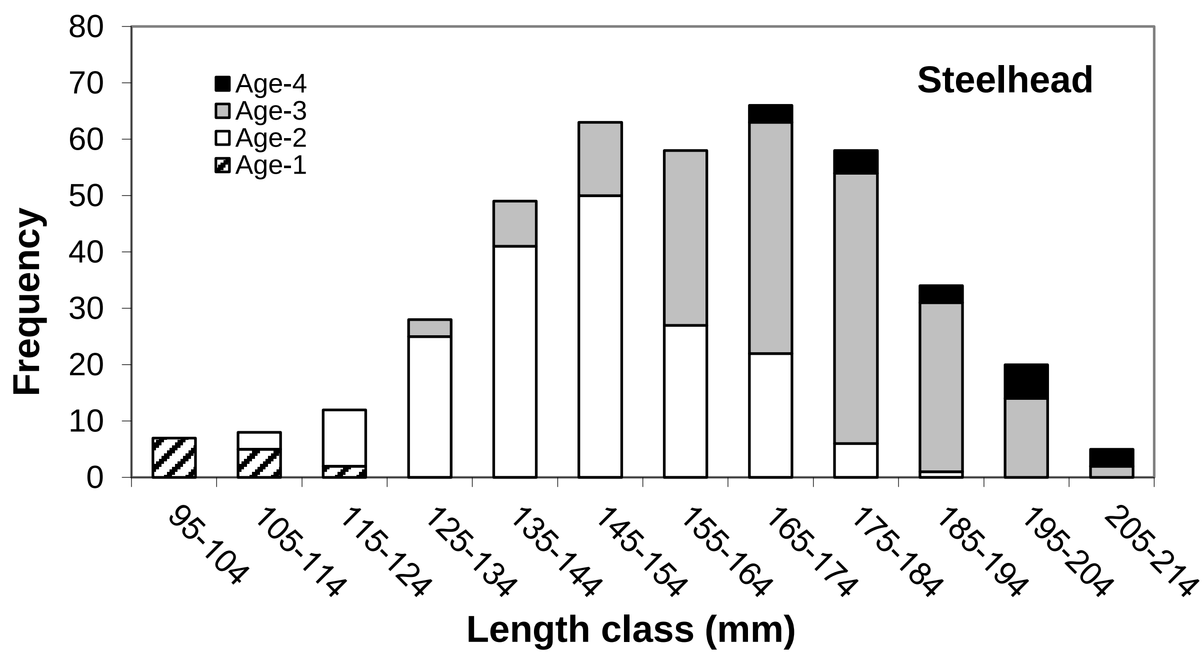
Chum

Recovery site: RST 2.2

All mark groups Recovery strata

Release strata	Marks	1	2	3	4	5	6	7	8	9	Capture efficiency
1	1998	124	0	0	0	0	0	0	0	0	6.2%
2	1881	0	135	0	0	0	0	0	0	0	7.2%
3	1997	0	0	85	0	0	0	0	0	0	4.3%
4	1999	0	0	0	23	0	0	0	0	0	1.2%
5	2000	0	0	0	0	137	0	0	0	0	6.9%
6	2000	0	0	0	0	0	188	0	0	0	9.4%
7	2002	0	0	0	0	0	0	172	0	0	8.6%
8	2000	0	0	0	0	0	0	0	113	0	5.7%
9	1273	0	0	0	0	0	0	0	0	46	3.6%
Untagged Fish		7285	15298	13321	7690	14456	19449	11729	5068	2041	

Appendix 5.4 Age-forklength relationships for Steelhead parr and smolts in the Coquitlam River during 2005-2015 derived from scale-aging analysis.



9.6 Figures and Tables for Chapter 6

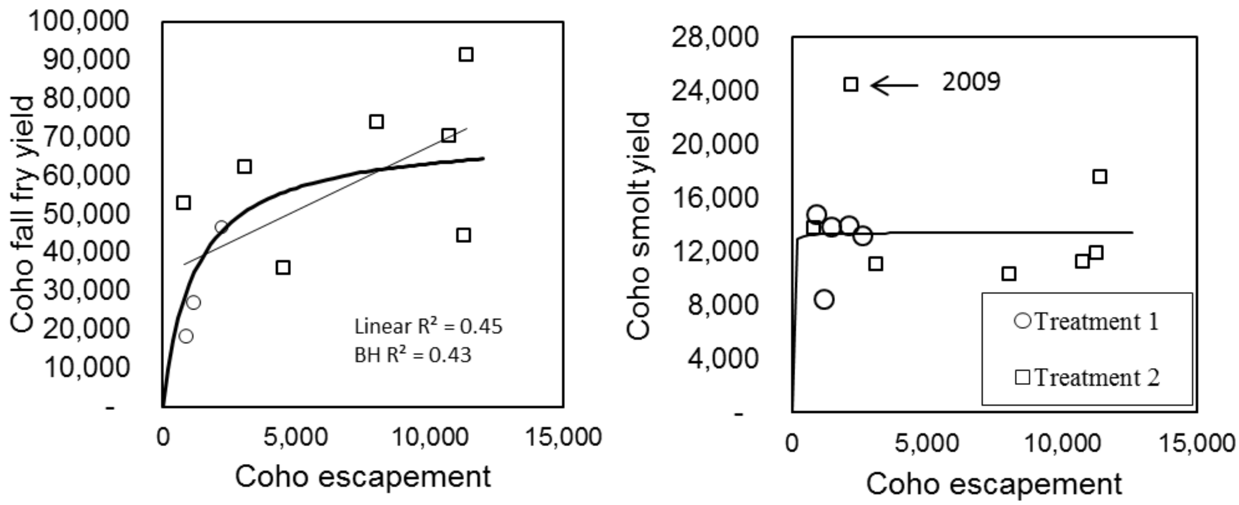


Figure 6.1 Preliminary linear and Beverton-Holt stock-recruitment relationship between Coho escapement and fall fry yield (2006-2015 fry years) and Beverton-Holt stock-recruitment relationship between Coho escapement (upstream of RST2) and total smolt yield in the Coquitlam River during Treatment 1 (2004-2008 smolt years), and during the first 7 years of Treatment 2 (2009-2015 smolt years).

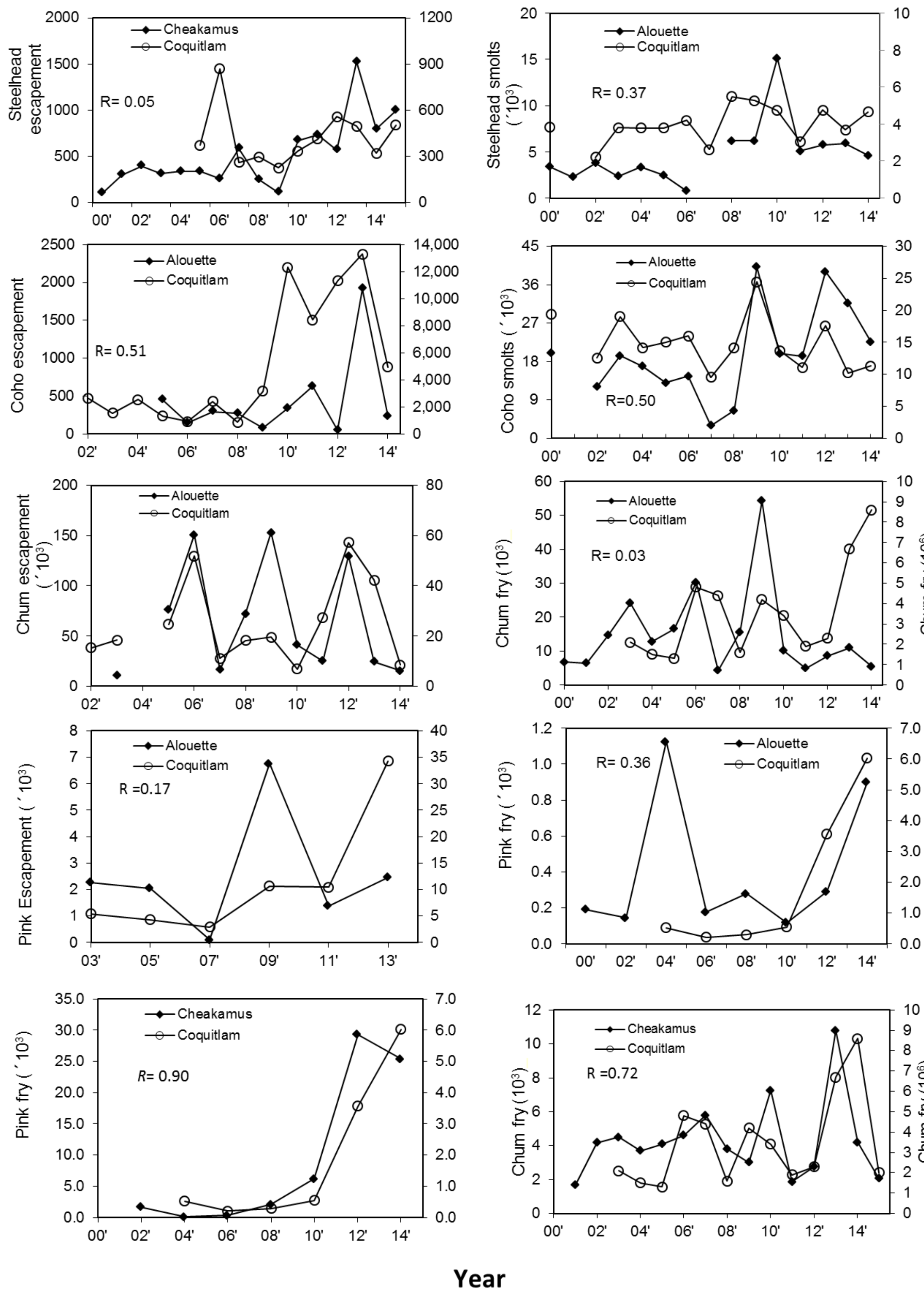


Figure 6.2 Scatterplots of escapement and smolt yield in the Coquitlam River versus that in the Cheakamus and Alouette rivers during 2000-2015. Values for the Coquitlam River are given on the right-hand axis, and values for other streams are given on the left-hand axis.

9. Figures, Tables and Appendices

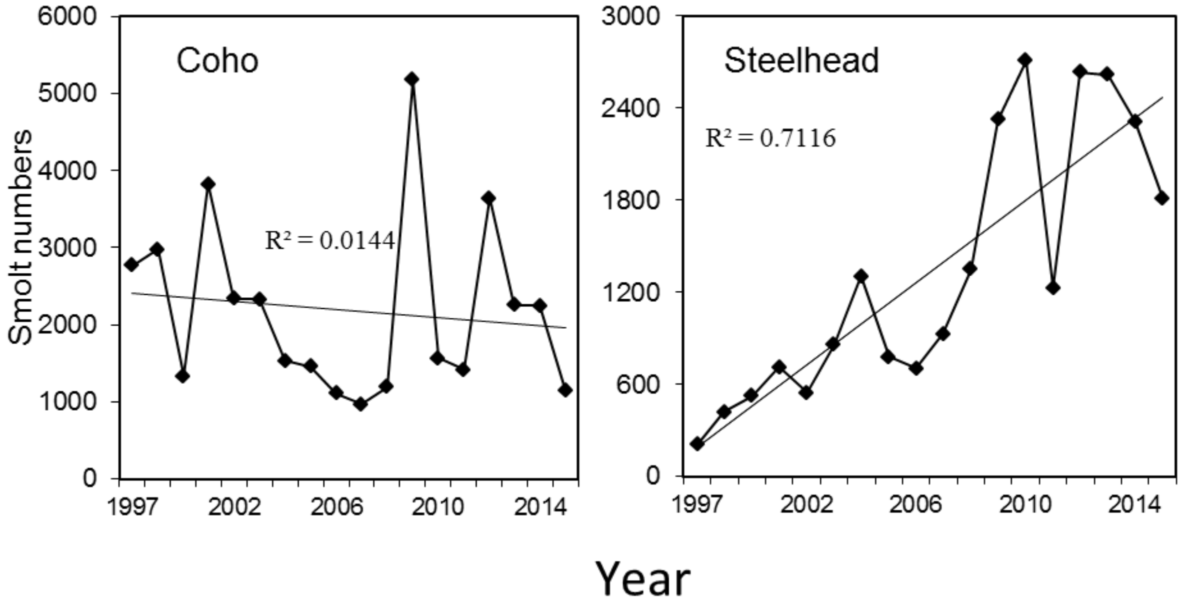


Figure 6.3 Annual numbers of Coho and Steelhead smolts in reach 4 of Coquitlam River during 1997-2015.

9. Figures, Tables and Appendices

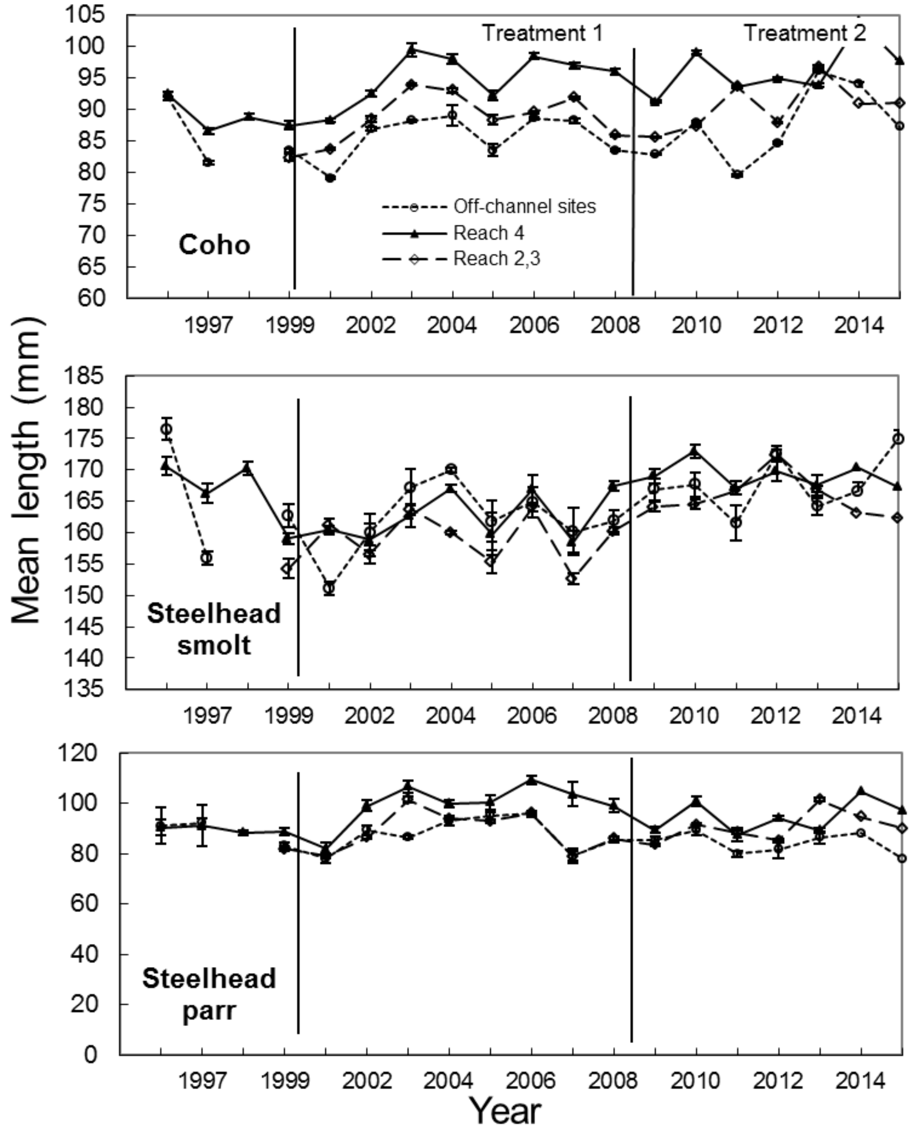


Figure 6.4 Mean annual forklengths for Coho smolts and Steelhead smolts (age 2+ and 3+ combined) and parr in different habitats in the Coquitlam River, 1996-2015. Error bars represent ± 1 standard error.

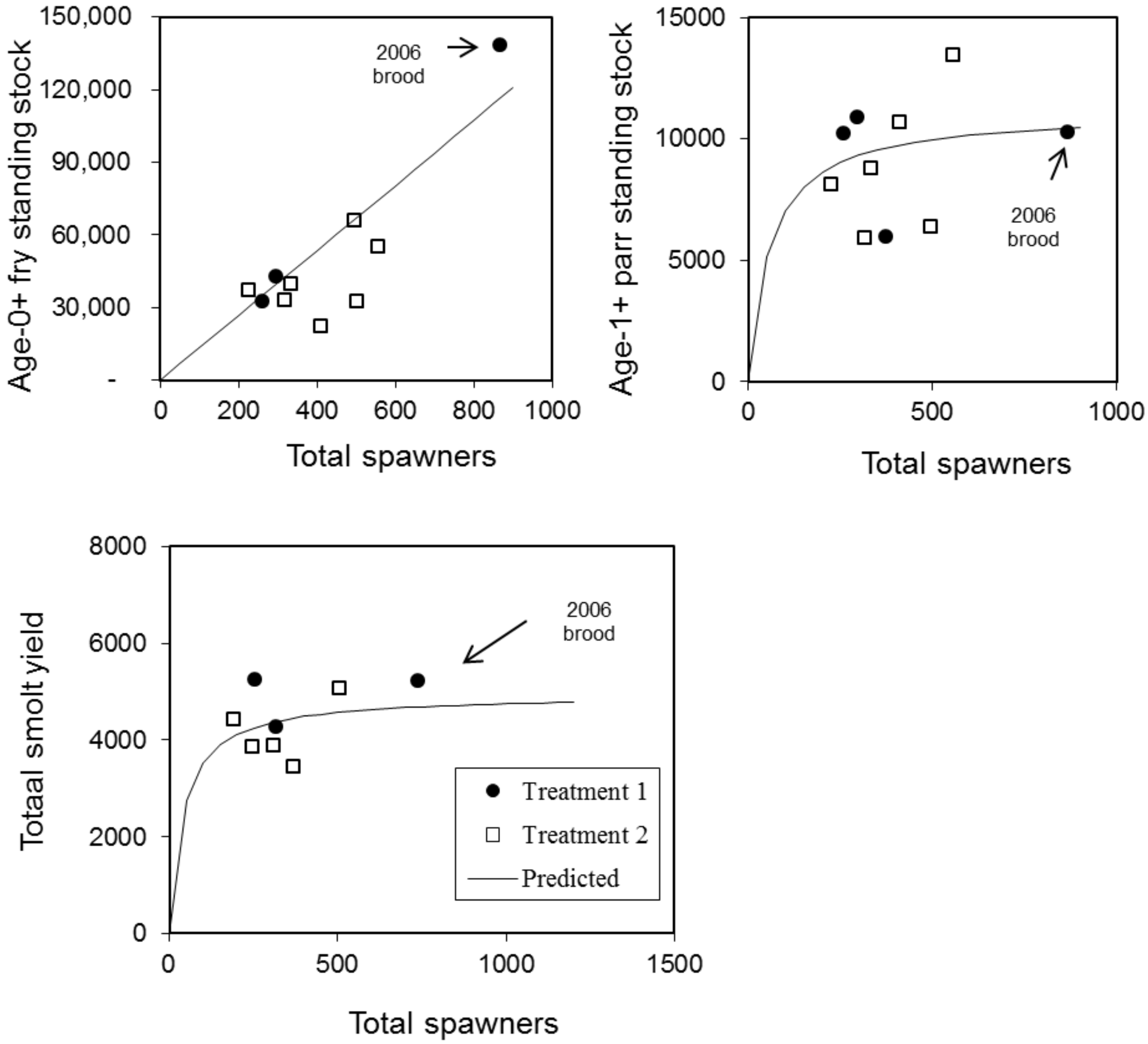


Figure 6.5 Preliminary stock-recruitment relationship for late summer juvenile Steelhead standing stocks and spring smolt yield (2005-2015) versus brood escapements in the Coquitlam River (data points corresponding to peak escapement in 2006 are shown).

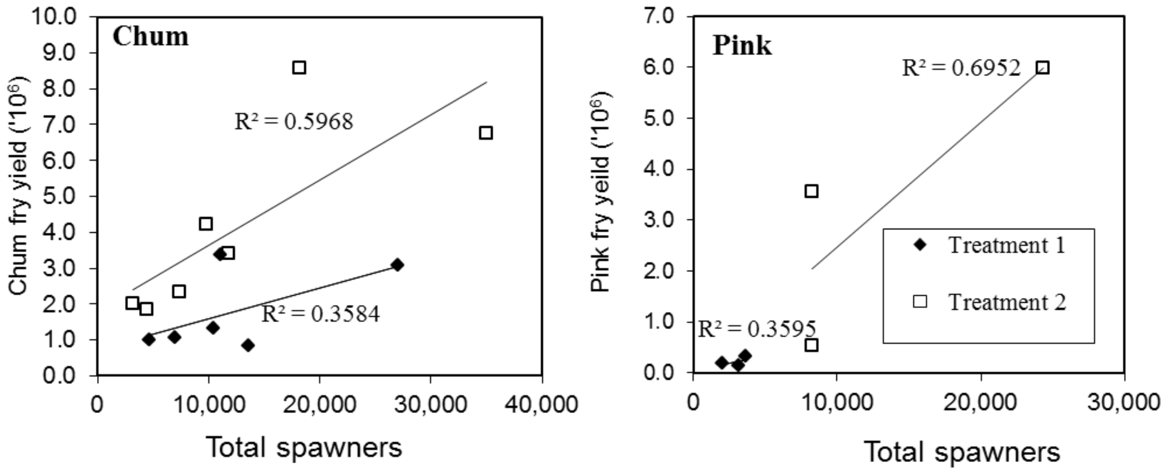


Figure 6.6 Preliminary stock-recruitment relationships for outmigrating Chum (2002-2015) and Pink (2002-2013) fry versus brood escapements in the Coquitlam River.

9. Figures, Tables and Appendices

Table 6.1a Summary of all population estimates for all life stages and species in Coquitlam River, 2000-2015. 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
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	
escapement	pink	-	-	5,418	-	4,406	-	2,876	-	10,698	-	10,427	-	34,280		
	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	
	Chinook	-	-	<300	<100	<100	<100	438	952	1,529	8,018	4,918	363	2,413		
	steelhead (female)	-	-	-	-	187	434	130	148	113	167	206	278	248	158	251
	steelhead (total)	-	-	-	-	373	868	260	297	225	333	412	557	495	317	502
Fall standing	coho	-	-	-	-	-	27,111	18,405	46,719	52,794	62,178	91,367	73,846	70,279	44,507	36,101
stock	0+ steelhead fry	-	-	-	-	-	138,132	32,251	42,506	37,047	39,657	21,949	55,232	66,017	32,746	32,277
	1+ steelhead parr	-	-	-	-	-	5,976	10,237	10,222	10,876	8,106	8,791	10,668	13,456	6,369	5,889
	2+ steelhead parr	-	-	-	-	-	1,841	1,978	1,255	3,196	2,690	3,862	3,160	2,625	3,831	2,561
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
	pink (total - millions)	-	-		0.32	-	0.15	-	0.18	-	0.55	-	3.56	-	6.03	-
	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
	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	10,275	11,265	11,854
	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
	steelhead (2+)	-	-	-	-	-	-	1,412	2,795	2,968	2,588	1,848	2,177	1,927	3,134	3,134
	steelhead (3+)	-	-	-	-	-	-	-	2,849	2,430	2,286	1,256	2,581	1,695	1,520	1,944

Table 6.1b Summary of survival estimates across all life stages and species for 2000-2014 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	Survival by life stage	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
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%	
coho	Egg-to-smolt [1]	-	0.34%	0.63%	0.39%	0.48%	1.12%	0.74%	1.14%	0.24%	0.10%	0.09%	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%
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%	
steelhead	Egg-to-smolt [1,2]	-	-	-	-	0.7%	0.4%	1.1%	0.8%	1.3%	0.7%	0.5%	0.5%			
steelhead	Fry to age-1+ parr	-	-	-	-	-	7.4%	31.7%	25.6%	21.9%	22.2%	48.6%	24.4%	9.6%	18.0%	
steelhead	Fry to age-2+ parr	-	-	-	-	-	0.9%	9.9%	6.3%	10.4%	8.0%	12.0%	6.9%	3.9%		
steelhead	Age 1+ parr to smolt [2]	-	-	-	-	-	68.6%	40.4%	44.1%	35%	55%	44%	32%			
steelhead	Age 2+ parr to smolt [2]	-	-	-	-	-	68.2%	144.0%	193.6%	71.5%	46.7%	66.8%	53.6%			
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%	29.6%	40.0%	
pink	Egg-to-fry [1]	-	-	9.6%	-	5.1%	-	9.7%	-	7.4%	-	48.0%	-	27.4%	-	

[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 (2000-2008) and Treatment 2 (2009-2015) in the Coquitlam River including the p-values for the two-tailed t tests.

	Treatment 1		Treatment 2		t test	Null Hypothesis
Smolt yeild	Mean	N	Mean	N	p value	
Coho (Mainstem)	6,173	8	7,876	7	0.06	reject
Coho (Total)	12,949	8	12,624	7	0.91	do not reject
Steelhead	3,848	8	4,348	7	0.20	do not reject

Table 6.3 Preliminary ANCOVA results for Chum 2003-2014 brood years to examine the significance of flow treatment on fry yield during Treatment 1 (2000-2008) and Treatment 2 (2009-2014) 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	11.5	0.001	reject
Treatment	7.7	0.02	reject
Escapement x Treatment	0.87	0.37	do not reject

9. Figures, Tables and Appendices