BChydro

Coquitlam-Buntzen Water Use Plan

Lower Coquitlam River Fish Productivity Index

Implementation Year 6

Reference: COQMON #7

Final Report: Lower Coquitlam River Fish Productivity Index

Study Period: 2000 - 2011

Living Resources Environmental Services
  #3-108 West 11th Ave., Vancouver B.C. V5Y 1S7
  Ph: 604-862-2323 Email: jacemacnair@yahoo.ca

April 30, 2014

Final Submission

Coquitlam River Fish Monitoring Program:
2000-2011 Results

Prepared for:

BC Hydro

Prepared by:

Jody Schick[1]*, Scott Decker[2], Jason Macnair[3] and Gordon Lewis[4]

April 2013

[1] 529 Gower Point Road, Gibsons, BC., V0N 1V0, jodschick@gmail.com

[2] 1034 Fraser Street, Kamloops B.C., V2C 3H7, decker_scott@hotmail.com
[3] 2919 Ontario St., Vancouver B.C. V5T 2Y5, jacemacnair@yahoo.ca
[4] 1701 Pipeline, Coquitlam, B.C, V3E 2X1, GMaintenance@MidBC.com
* Author to whom correspondence should be addressed

EXECUTIVE SUMMARY

As part of the Coquitlam River Water Use Plan (LB1 WUP), a long-term adaptive management study is being conducted in 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: coho, steelhead, chum and pink, and includes adult escapement and smolt outmigration monitoring for each species. Higher returns during 2007-2011 allowed Chinook escapements to be estimated as well. Since 2006, night snorkelling 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 three years of Treatment 2 (2009-2011) 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 work completed in 2010 (adult salmon escapement) and 2011 (all life stages), but 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 2002-2011 (880 to 12,300 adults; 36 to 480 females/km) likely exceeded that necessary to seed available juvenile habitat. During 2000-2011, annual coho smolt numbers upstream of the lowermost trapping site varied from 8,400 to 24,500 fish. Late summer standing stocks of coho fry ranged from 21,000 to 105,000 fish during 2006-2011. Abundance has increased every year since 2007. Coho egg-to-smolt survival was low (0.3%-1.1%) relative to that in other coastal streams (0.7-3.0%).

Redd counts suggested that steelhead escapements during 2005-2011 (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. Steelhead smolt yield upstream of the lowermost trap ranged from 2,300 to 5,600 fish. During 2006-2011, late summer standing stocks of steelhead fry and parr ranged from 33,000-128,000, and from 7,800-14,800, respectively. Steelhead egg-to-fry survival (5.7%-9.1%), egg-to-parr survival (0.7%-2.2%), and egg-to-smolt survival (0.4%-1.1%) were comparable to 1976-1982 averages for the Keogh River. While no clear trend has been observed for the Coquitlam River as a whole, steelhead smolt production in reach 4 immediately below Coquitlam Dam has increased steadily over the period of record, from less than 400 smolts in 1996 (prior to the start of Treatment 1) to over 1,200 smolts in recent years (exceeding 2,000 smolts in 2009 and 2010).

 Chum escapements ranged from 7,000 to 52,000 adults for the study area as a whole, and yielded 0.8 to 4.2 million smolts upstream of the lowermost trap, with the highest smolt yield occurring in 2009, similar to the case for coho and steelhead. Egg-to-smolt survival ranged from 3.7% to 26.8%, and averaged 12.5% during Treatment 1. Egg-to-smolt survival during the first

two years of Treatment 2 averaged 22.1% and ranged from 18%- 26%., but the lower value observed in 2009 was likely biased-high, possibility as a result of a biased-low escapement estimate.

Pink escapements ranged from 2,900-10,600 adults and yielded 150,000-550,000 smolts. 2009 and 2011 experienced escapements 2- to 4-fold higher than previous years with the highest smolt yield in 2010 (2009 brood year). Egg-to-smolt survival ranged from 5.1% to 9.7%, which was comparable to reported values for other streams. Chum and pink salmon returns to Coquitlam River were greatly improved in 2002-2011 compared to escapements in years prior to the implementation of the Treatment 1 flow regime in 1997.

Chinook escapements ranged from 400-8,000 adults during 2007-2011, and were likely less than 300 adults prior to this period. The highest Chinook escapement occurred in 2010.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

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 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.

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

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

1.1 **Background**

The lower Coquitlam River flows 17 km from the base of 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 outlets that release flows from Coquitlam Reservoir. The Coquitlam Reservoir also

supplies drinking water for the Greater Vancouver Regional District (GVRD) and water for power via a diversion tunnel to Buntzen Lake.

Typical of lotic habitats downstream of dams, spawning and rearing habitat in the lower Coquitlam River (hereafter referred to as 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, flow releases were increased to 0.8 to 1.4 cms, depending on the time of year. This represents the Treatment 1 regime of two fish valves fully open, and is the baseline for this adaptive management study.

The Treatment 2 flow regime (i.e., Coquitlam River CQD LLOG3 knife gate) was initiated on October 22, 2008, with seasonal target flow releases from Coquitlam Dam ranging from 1.1 to 6.1 cms (Table 1.1). After the knife gate was put into operation, BC Hydro personnel conducted measurements of actual flows during the spring and summer of 2009, and these were compared to modeled flows to verify the theoretical discharge rating curve through the various flow ranges. These measurements indicated that actual flows were consistently higher than predicted 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 facility. Once sufficient data was collected, the discharge rating curve was adjusted and brought into service on September 15, 2009. With respect to the flow experiment then, 2009 was not strictly representative of Treatment 2. However, given the planned 9-year duration of Treatment 2, this is not likely to have a significant impact on the comparison of the two treatment periods.

The Coquitlam River historically supported all six Pacific salmon, as well as cutthroat trout (Oncorhynchus clarki), which are still present at low numbers, and Dolly Varden (Salvelinus malma) char, which appear to have been extirpated. Dam construction resulted in the extirpation of an anadromous stock of summer sockeye (O. 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 (*O. kisutch*), steelhead (*O. mykiss*), chum (*O. keta*), and pink (*O. gorbuscha*). Other fish species are either of too low abundance to effectively monitor (this appears to be changing for Chinook salmon, 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

Species	Month	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Steelhead	Spawning												
	Incubation												
	Rearing (parr)												
	Rearing (fry)												
Coho	Spawning												
	Incubation												
	Rearing (fry)												
Chum	Spawning												
	Incubation												
	Outmigration												
Pink	Spawning												
	Incubation												
	Outmigration												
Chinook	Spawning												
	Incubation												
	Rearing												
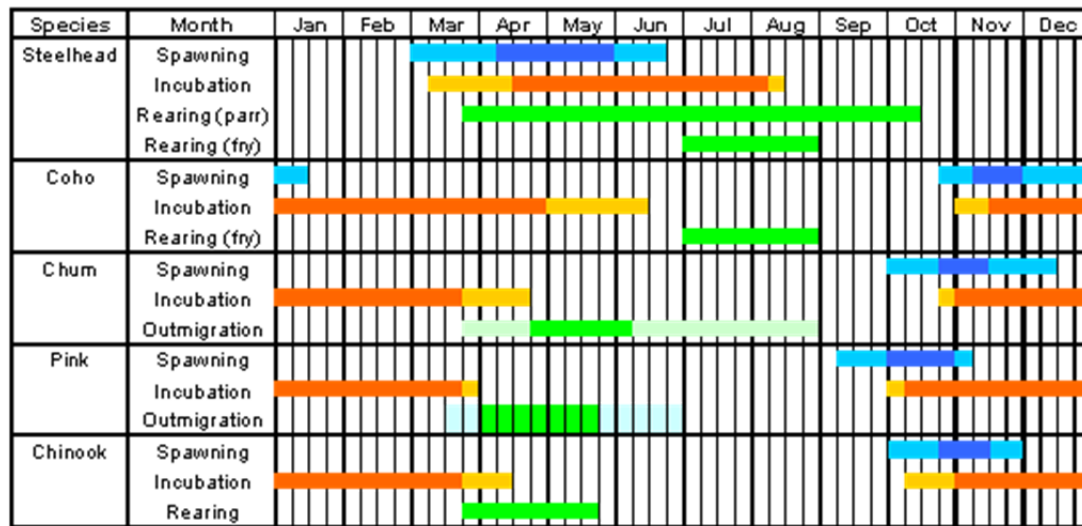
Figure 1.1 Life stage periodicity chart for anadromous salmonids in Coquitlam River.

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 on 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 2007-2011, 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 2002-2011, 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 focuses on escapement results for returns during the 2010 and 2011 spawning periods.

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
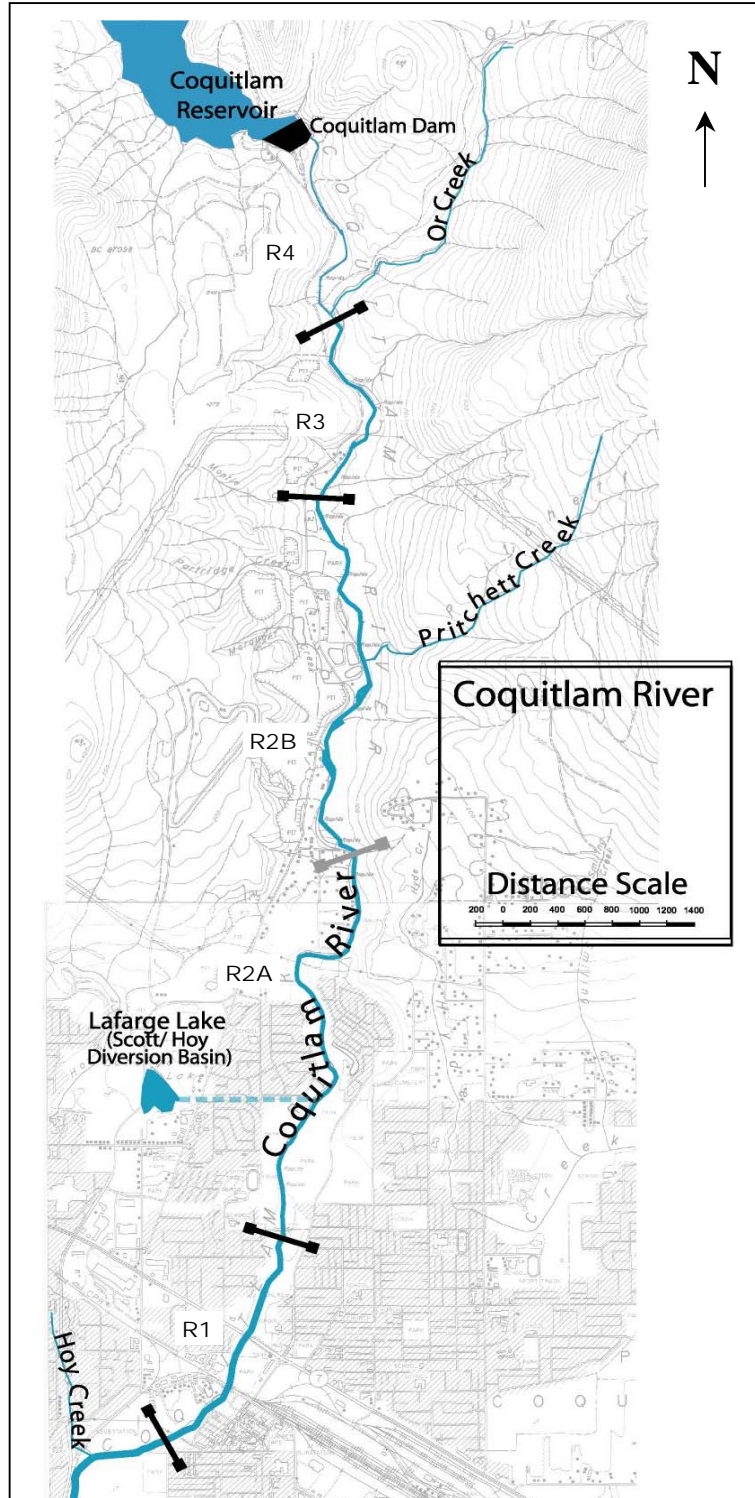
Coquitlam River

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

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 2006-2011 using this methodology; mark-recapture experiments were conducted during 2007-2011 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 2011 juvenile standing stock survey, and summarizes preliminary population estimates for 2006-2011.

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 2000-2011, 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 2011 smolt trapping program and summarizes population estimates for all species and reaches for 2000-2011.

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 2002-2011 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 2007-2011 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 2002-2011. 2010 and 2011 escapements have not been reported previously. Normally 2011 escapements would be reported in the following year (i.e., 2012 report), but they have been included here due to the later completion date for this report.

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).
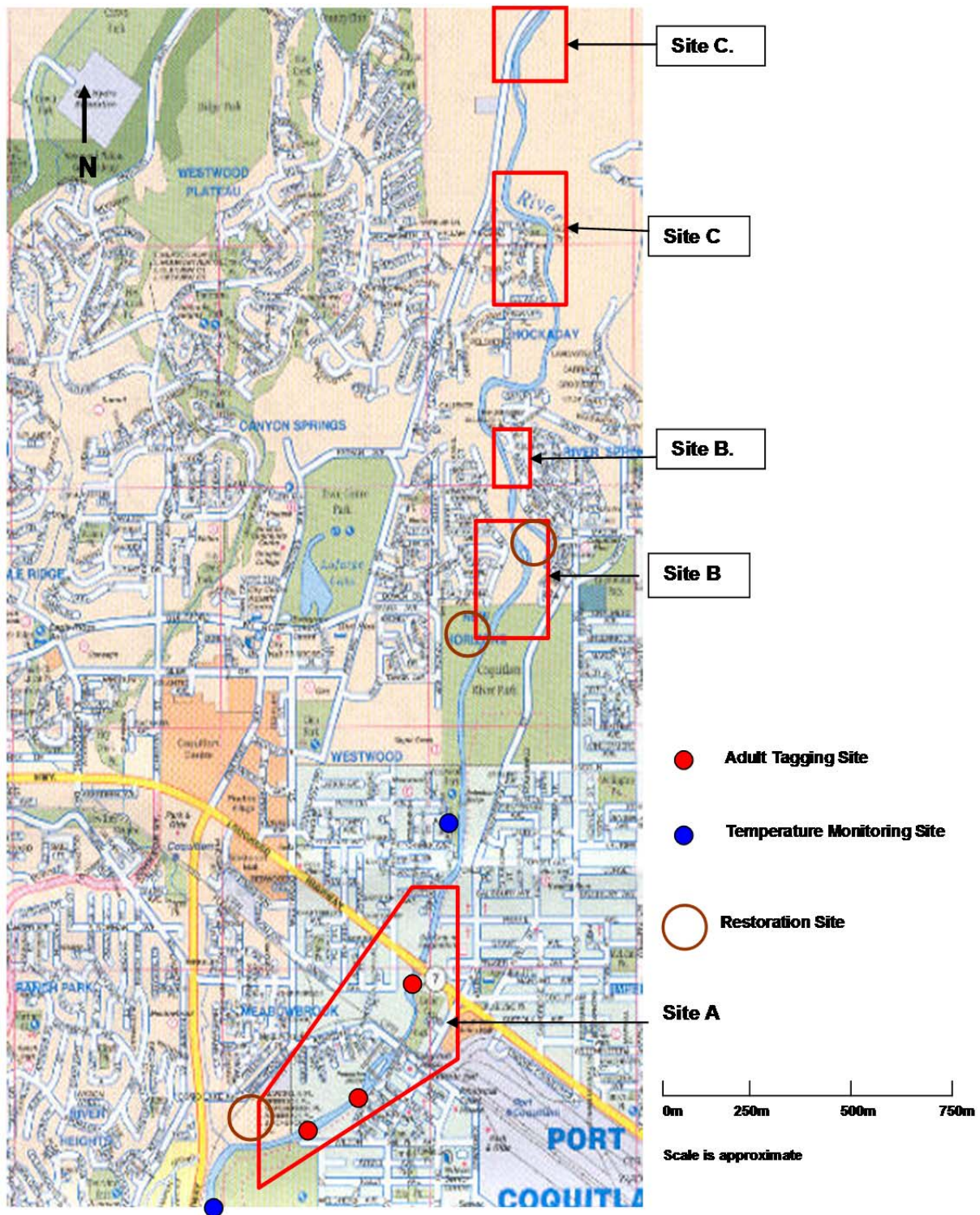
Figure 2.1 Map showing adult spawning index sites A-C in the lower portion of Coquitlam River study area (reaches 1, 2a)
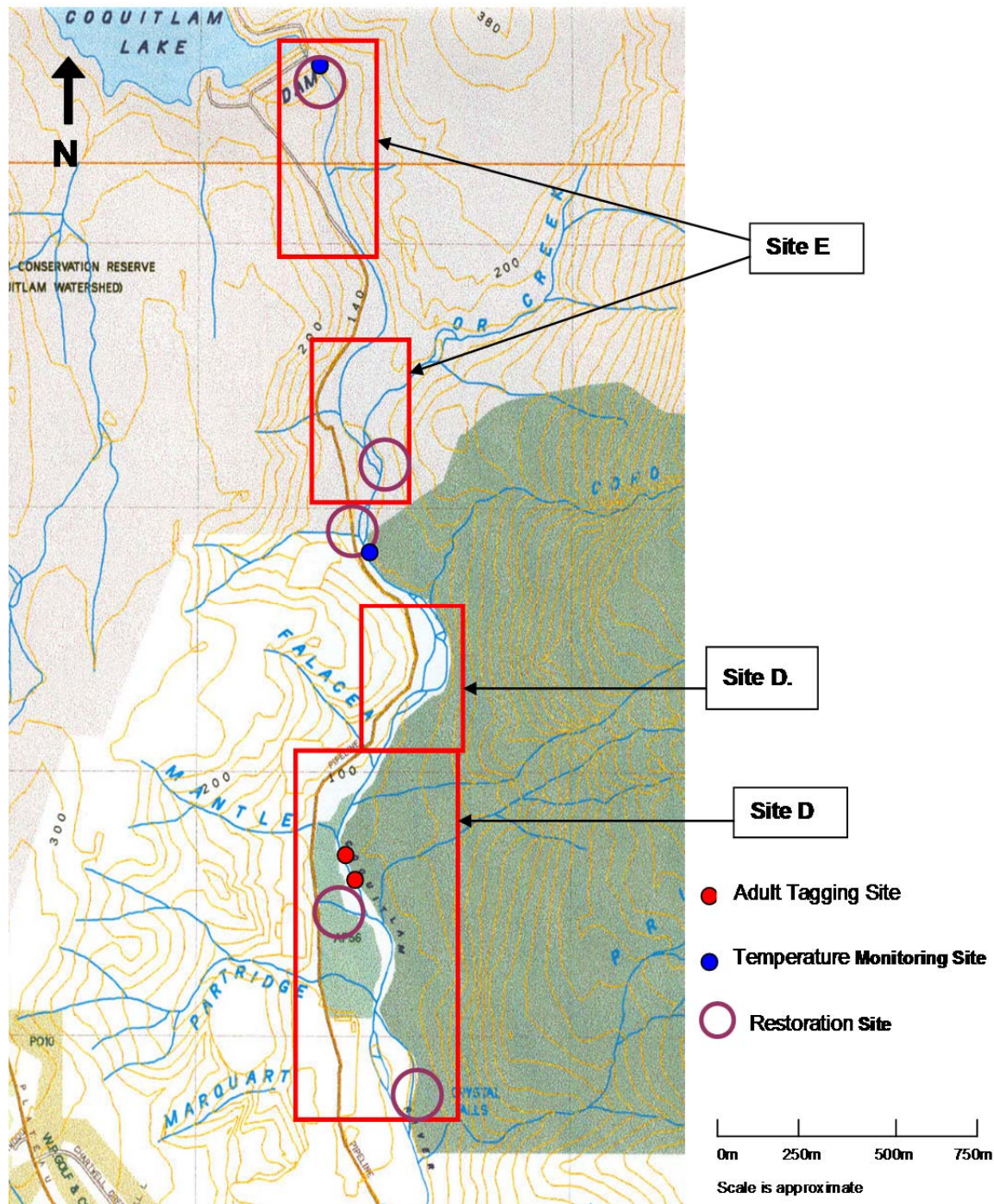
Figure 2.2 Map showing adult spawning index sites D and E, in the upper portion of Coquitlam River study area (reaches 2b, 3 and 4).

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 (Crew leader: one of two

individuals; 2nd crew member: one of three individuals). This greatly 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 2002-2011 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, 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:

Expanded total count = (total count$_{sites A-E}$)/ (1 - 0.15).   (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 2006-2011, we conducted 14 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). 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 2006-2011 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 2002-2011, 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,

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

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.

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:

$$\alpha = \mu * \frac{1}{\sigma^2}$$

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

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,

$$SL_t = \lambda_c e^{-\lambda_s t} \qquad (2.4)$$

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,

$$PD_t = \sum_t PA_t \mid t = D_t \qquad (2.5)$$

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.

$$N_t = E\left[\int_1^t PA - \int_1^t PD\right] \qquad (2.6)$$

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),

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

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,

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

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

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

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

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

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

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 underfits 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 overfits 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 2002-2011 are shown for chum, pink, coho, and Chinook in Appendices 2.2-2.5. The typical period of peak spawning was the first week of October for pink, the last week of October for chum, the second week of December for coho, and the last half of October for Chinook. Run timing curves were generally well defined for each species in 2010 and 2011 (Appendices 2.2-2.5). This was accomplished through frequent and well-timed surveys that captured the beginning and end of the spawning period for each species. 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 and end of the spawning period was generally well defined each year (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), significant numbers of coho were still present during the final survey (Appendix 2.4). 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). For modeling purposes, the maximum length of the spawning period for pink, chum, coho, and Chinook was constrained to 82 days (September 1 – November 21), 113 days (September 9 – December 30), 130 days (September 20 – January 27), and 94 days (September 3 – November 30), respectively.

Water column visibility ranged from 0.5-3.0 m and average 1.4 m and 1.3 m among surveyed sites in 2010 and 2011, respectively (Table 2.1a and 2.1b). Interruptions caused by high flows and/or poor water visibility occurred infrequently during 2010 and 2011, and surveys were completed on a near-weekly basis during the chum spawning period and bi-weekly during the predominantly coho spawning period in December and January (Table 2.1a and 2.1b). There were very few instances where visibility in the lower river below the gravel mines precluded surveys at index sites A and B during 2010 or 2011. 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).

Table 2.1a 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 2010 brood year.

Escapement Year	Date	Estimated water column visibility (m)					
		site A	site B	site C	site D	site E	non-index
2010	3-Sep	>3	>3	>3	>3	>3	-
2010	10-Sep	>3	>3	>3	>3	>3	
2010	21-Sep	0.8	0.8	0.7	>3	>3	-
2010	5-Oct	2.0	1.2	1.2	>3	>3	-
2010	12-Oct	1.0	1.0	1.0	1.2	1.2	1.0
2010	20-Oct	1.0	0.9	0.9	1.5	1.5	1.0
2010	23-Oct	1.0	0.9	0.9	1.5	1.5	1.0
2010	31-Oct	-	0.6	0.6	1.2	1.5	1.0
2010	4-Nov	0.8	0.9	0.9	1.5	1.5	1.0
2010	13-Nov	1.0	-	0.9	1.5	1.5	-
2010	23-Nov	0.6	0.5	0.6	1.5	1.5	-
2010	29-Nov	-	1.0	1.0	1.5	1.5	-
2010	6-Dec	-	0.7	0.7	1.2	1.3	-
2010	19-Dec	-	0.6	0.6	1.5	1.5	-
2010	30-Dec	-	0.8	0.8	1.3	1.5	-
2010	11-Jan	-	0.8	0.8	1.1	1.2	-
2010	18-Jan	-	-	0.8	1.0	1.2	-

Surveyor "guesstimates" of observer efficiency (0.0-1.0): (chum salmon example)

2010	3-Sep	0.95	0.95	0.95	0.95	0.95	-
2010	10-Sep	0.95	0.95	0.95	0.95	0.95	0.95
2010	21-Sep	0.60	0.60	0.55	0.95	0.95	-
2010	5-Oct	0.90	0.80	0.75	0.95	0.95	-
2010	12-Oct	0.75	0.75	0.75	0.80	0.80	0.70
2010	20-Oct	0.75	0.70	0.70	0.80	0.85	0.70
2010	23-Oct	0.75	0.70	0.70	0.80	0.85	0.80
2010	31-Oct	-	0.55	0.50	0.80	0.75	0.70
2010	4-Nov	0.65	0.65	0.70	0.75	0.75	0.70
2010	13-Nov	0.60	-	0.60	0.75	0.75	0.70
2010	23-Nov	0.50	0.40	0.60	0.75	0.75	-
2010	29-Nov	-	0.65	0.65	0.8	0.8	-

Table 2.1b 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 2011 brood year.

Escapement Year	Date	Estimated water column visibility (m)					
		site A	site B	site C	site D	site E	non-index
2011	10-Sep	>3	1.4	1.3	>3	>3	-
2011	17-Sep	>3	1.3	1.5	>3	>3	-
2011	24-Sep	>3	1.3	1.5	>3	>3	-
2011	10-Oct	1.3	1.3	1.3	1.3	1.5	1.4
2011	17-Oct	1.1	1.3	1.2	1.4	1.4	1.3
2011	25-Oct	1.1	1.1	1.1	1.5	1.5	1.2
2011	1-Nov	1.0	1.0	1.0	1.5	1.5	1.1
2011	6-Nov	1.1	1.1	1.1	1.3	1.3	1.1
2011	15-Nov	1.3	1.3	1.3	1.2	1.2	-
2011	21-Nov	1.1	1.1	1.1	1.2	1.3	-
2011	1-Dec	0.75	0.75	0.8	0.8	0.8	0.75
2011	7-Dec	-	1.0	0.75	1.2	1.2	1.0
2011	19-Dec	-	0.50	0.60	0.8	1.0	0.9
2011	27-Dec	-	0.60	0.60	0.8	1.0	0.8
2011	2-Jan	-	0.80	0.75	1.2	1.2	1.0
2011	12-Jan	-	0.80	0.80	0.75	1.1	0.9
2011	21-Jan	-	0.70	0.75	1.1	1.1	1.0

Surveyor "guesstimates" of observer efficiency (0.0-1.0): (chum salmon example)

Escapement Year	Date	site A	site B	site C	site D	site E	non-index
2011	10-Sep	0.95	0.95	0.95	0.95	0.95	-
2011	17-Sep	0.95	0.8	0.8	0.95	0.95	-
2011	24-Sep	0.95	0.8	0.9	0.75	0.95	-
2011	10-Oct	0.95	0.8	0.9	0.95	0.95	0.85
2011	17-Oct	0.8	0.8	0.8	0.8	0.9	0.8
2011	25-Oct	0.7	0.8	0.75	0.85	0.85	0.8
2011	1-Nov	0.7	0.7	0.7	0.9	0.9	0.8
2011	6-Nov	0.7	0.7	0.7	0.9	0.9	0.75
2011	15-Nov	0.7	0.7	0.7	0.8	0.8	-
2011	21-Nov	0.8	0.8	0.8	0.85	0.85	-
2011	1-Dec	0.7	0.7	0.75	0.8	0.85	0.6
2011	7-Dec	0.6	0.6	0.65	0.65	0.65	0.7

2.2.2 Observer efficiency and survey life

2.2.2.1 Observer efficiency

During 2006-2011, only 14 estimates of observer efficiency were obtained for all species combined, with the majority (7) obtained for chum salmon. In some 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 in future 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 seven mark-recapture experiments during 2006-2011 (range: 33%-69%; Table 2.2); with similar means for Treatment 1 and Treatment 2 (50% and 48%, respectively; Appendix 2.1). For pink, three mark-recapture experiments yielded an average observer efficiency estimate of 57% (range: 49%-72%; Table 2.2, Appendix 2.1). For coho and Chinook, two mark-recapture experiments under Treatment 2 provided average observer efficiency estimates of 68% and 60%, respectively. 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 for coho in the Coquitlam River was probably lower than 68% during Treatment 1; it 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. 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. 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.

Another issue is that the spatial distributions of marked populations of chum and pink differed from that of the population as a whole. For example, during 2007, marking occurred exclusively in the lowermost index site (Site A), and most of the tagged chum (63%-84%; Appendix 2.1) and pink (83%) were resighted in Site A, providing little information about observer efficiency in the remainder of the survey area upstream. By contrast, only 45% and 21%, respectively, of unmarked chum and pink were counted across all surveys in Site A during 2007 (Table 2.3). Conversely, the distribution of marked chum originating from a tagging event near downstream boundary of the study area was relatively similar to that of unmarked fish following a tagging event on October 15 in 2008 (see Table 2 in Decker and Macnair 2009).

During 2009-2011, salmon (all species) were tagged at both lower and upper index sites in an attempt to provide more broadly distributed marked populations. 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).

In some cases (mostly during 2006 and 2007), marked populations of chum and pink 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 2006–2011 provided limited information about survey life for each species. 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 10 estimates of survey life were obtained, four for chum, and two each for pink, coho and Chinook (Table 2.2). Based on the area-under-the-tag-depletion curve method (Perrin and Irvine 1990), mean survey life values ranged from 7.7 days for coho, to 11.2 days for pink. 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.
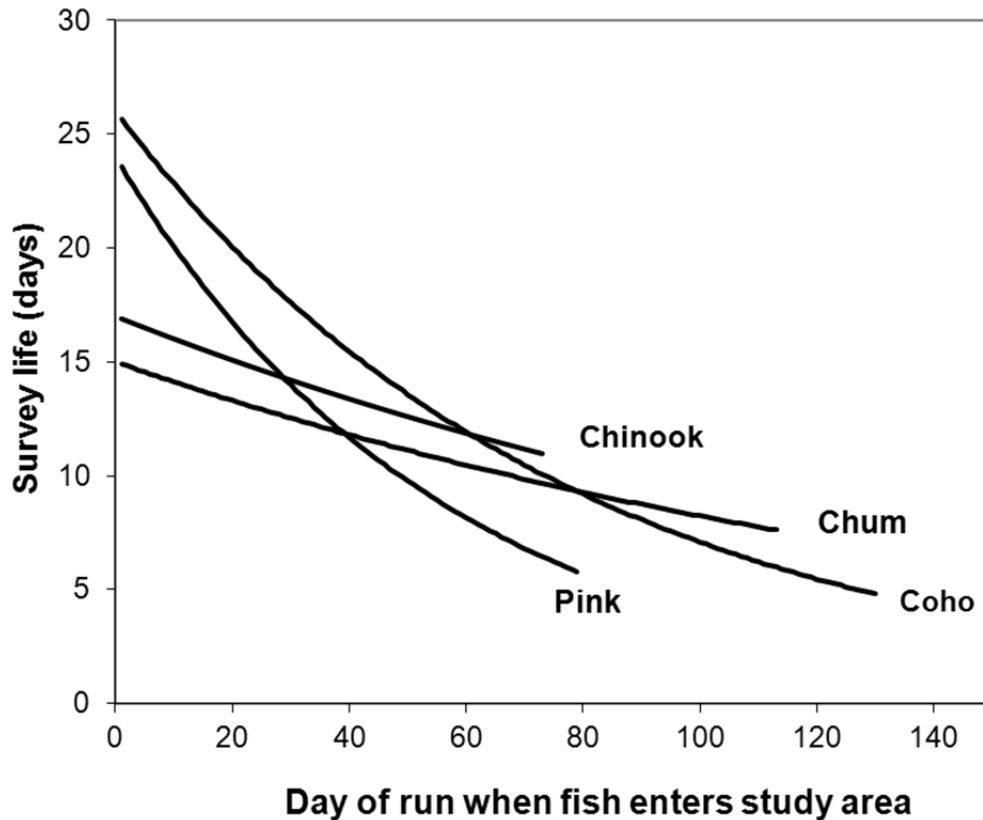
Figure 2.3 Modeled relationship between mean survey life and day of arrival in the study area for chum, pink, coho, and Chinook salmon in the Coquitlam River based on empirical data from other streams.

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 30% to 95%, and average 81% (Table 2.2). When compared to mark-recapture estimates of observer efficiency, the surveyor guesstimates were biased high, but were useful predictors, explaining 52% of the variation in mark-recapture derived estimates of observer efficiency among surveys for chum (linear regression, $n=7$; $R^2=0.52$; Figure 2.4). For pink, surveyor guesstimates ranged from 40% to 95% for three surveys for which mark-recapture data were available, and were reasonably similar to the mark-recapture derived estimates of observer efficiency (Table 2.2). Surveyor guesstimates explained 41% of the variation in mark-recapture derived estimates of observer efficiency among surveys for pink ($n=3$; $R^2=0.41$; 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 and Chinook, there were only two mark-recapture estimates of observer efficiency available in each case (Table 2.2), which provided 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 for these species. To model observer efficiency for coho and Chinook, we regressed surveyor guesstimates against mark-recapture derived estimates of observer efficiency using pooled data for all four species (n=14; R^2=0.21; Figure 2.4). Mean observer efficiency (based on mark-recapture) across species was 55% compared to observed means of 68% and 60% for coho and Chinook, respectively (Table 2.2). For pink, coho and Chinook, additional mark-recapture experiments will need to be conducted in future before reliable species-specific regression models can be developed.

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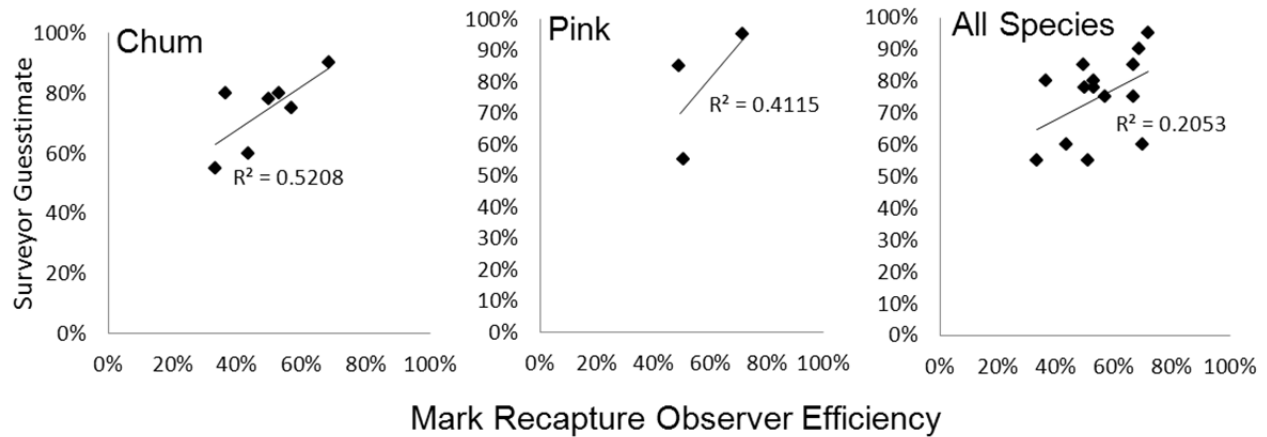
Figure 2.4 Relationship between the surveyor's 'guesstimate' of observer efficiency and observer efficiency estimated from mark recapture experiments for chum, pink and all species combined conducted opportunistically since 2006 in the Coquitlam River.

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. Examples of model fit for 2011 data for each species are provided in Appendix 2.7a-d. 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 7,000-52,000 for chum; 900-12,000 for coho; 3,000-11,000 for pink; and 1,000-8,000 for Chinook. The only clear trend between treatments is for pink salmon where the two near identical escapement estimates during Treatment 2 are approximately double the highest escapements from Treatment 1. For chum, the average escapement for Treatment 2 is considerably lower than Treatment 1 but this is driven primarily by the 2010 estimate, the lowest recorded during this project, while the other two Treatment 2 escapement estimates fall within the typical range of 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).

Escapements estimates generated for 2002-2011 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. Escapements 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. This may allow uncertainty in escapement estimates to be modelled as well at some point.

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
Mark-recapture-derived estimates of observer efficiency				
Number of estimates	7	3	2	2
mean	0.49	0.57	0.68	0.60
minimum	0.33	0.49	0.67	0.53
maximum	0.69	0.72	0.70	0.67
Surveyor guesstimates of observer efficiency				
mean	0.74	0.82	0.73	0.77
minimum	0.55	0.40	0.60	0.75
maximum	0.90	0.95	0.85	0.78
Survey life (days)				
Number of estimates	4	2	2	2
mean of estimates	7.8	11.2	10.6	7.7
range of estimates	7.0 - 9.9	6.8 - 15.6	6.5 - 11.6	6.9 - 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 2002-2011.

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

Table 2.4 Annual escapement estimates and the mean escapement estimates for Treatment 1 and 2 for chum, pink, coho and Chinook salmon for the years 2002-2011.

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	8,428	4,918
Treatment 1 Mean		23,911	4,214	1,761	952
Treatment 2 Mean		17,980	10,563	7,980	4,822

2.2.4 Adult habitat distribution and access to off-channel sites

Chum salmon in particular show a preference for mainstem spawning habitat in the Coquitlam River (Table 2.5). This preference has been noted in many studies for chum salmon in medium-sized rivers (Salo, 1991). In addition, adult chum show a preference for spawning in the lower reaches of the Coquitlam River, (an average of 63% of adult chum spawning in index sites A-C during 2002-2011; 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). Pink salmon also made much greater use of off-channel sites for spawning than did chum (29-45% pink versus 14-24% chum; Table 2.5). 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 (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 2002-2011; Table 2.3). Natural and enhanced off-channel habitat use dropped from 51%-73% during 2002-2005 to 9%-29% during 2006-2011 (Table 2.5). This shift 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. Reopening of the Grant's Tomb site in 2007 did not result in a significant shift back to off-channel habitat for coho.

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.

Species	Habitat	Treatment 1						Transition	Treatment 2			
		2003	2004	2005	2006	2007	mean	2008	2009	2010	2011	mean
Chum	M/S	0.87	0.82	0.87	0.90	0.84	0.86	0.76	0.77	0.73	0.77	0.76
	NOC	0.06	0.06	0.07	0.06	0.10	0.07	0.12	0.13	0.16	0.18	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
	OCC	0.13	0.18	0.13	0.10	0.16	0.14	0.24	0.23	0.27	0.23	0.24
Pink	M/S	0.55		0.65		0.71	0.64		0.76		0.59	0.76
	NOC	0.19		0.22		0.20	0.20		0.12		0.22	0.12
	OCR	0.26		0.13		0.09	0.16		0.12		0.19	0.12
	OCC	0.45		0.35		0.29	0.36		0.24		0.41	0.24
Coho	M/S	0.32	0.27	0.46	0.80	0.70	0.51	0.74	0.84	0.87	0.91	0.84
	NOC	0.15	0.38	0.33	0.07	0.11	0.21	0.04	0.03	0.01	0.01	0.02
	OCR	0.53	0.35	0.21	0.13	0.19	0.28	0.22	0.13	0.12	0.08	0.14
	OCC	0.68	0.73	0.51	0.20	0.29	0.48	0.26	0.16	0.13	0.09	0.16
Chinook	M/S	0.82	0.71	0.84	0.990	0.87	0.85	0.77	0.94	0.94	0.98	0.91
	NOC	0.06	0.02	0.07	0.008	0.04	0.04	0.02	0.02	0.02	0.01	0.02
	OCR	0.12	0.26	0.10	0.000	0.09	0.11	0.21	0.04	0.04	0.01	0.08
	OCC	0.18	0.29	0.16	0.008	0.13	0.15	0.23	0.06	0.06	0.02	0.09

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 OCC = off-channel sites combined.

2.2.5 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 2010 and 2011 spawning and incubation periods.

2.3 Recommendations and future activities

1. 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. Serious consideration should be given to restructuring the adult salmon escapement program so that more effort is made to collect this information. One possibility is to shift resources away from coho and Chinook, to allow greater resources to be applied to conducting mark-recapture experiments for chum and 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 probably be focussed on chum and pink.

2. The mark-recapture study should be continued in 2013 to provide additional estimates of observer efficiency and survey life for chum and pink. As many tagging events as possible should be conducted during 2013 and future years, and these should ideally be distributed throughout the spawning period for each species. Every effort should be made to conduct a complete survey of the study area within one or two days of each tagging event to provide estimates of observer efficiency. To provide information about survey life, additional surveys must follow a tagging event, ideally at 3-day intervals. If tagging exclusively in the lower river results in too few tagged fish moving to upstream sites, tagging should occur in Site D or E as well (using different tags than those applied to fish in the lower river) to provide information about observer efficiency specific to the upper river, where visibility is consistently higher. Data from fish tagged in Site D or E should not be used to estimate survey life.

3. Reconnaissance surveys should be conducted prior to the beginning of the arrival of pink salmon and following the completion of coho spawning 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 estimates of Chinook escapement are of interest.

Appendix 2.1.   Results of the 2006-2011 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	% females	A	B	C	D	E
chum	1	2006	below A	1	Oct 17	Oct 21-22	4.5	11	1	9%	18%	1	0	0	0	0
chum	1	2006	below A	1	Oct 17	Oct 31-Nov 1	14.5	11	0	0%	18%	0	0	0	0	0
chum	1	2006	C	2	Oct 19-20	Oct 21-22	1-3	89	61	69%	33%	0	0	0	49	12
chum	1	2006	C	2	Oct 19-20	Oct 31-Nov 1	11-13	89	1	1%	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%	44%	0	1	2	71	14
chum	1	2006	C	3	Oct 24, 28-30	Nov 30	31.5-36.5	59	0	0%	44%	0	0	0	0	0
chum	1	2007	A	1	Oct 11	Oct 13	2	33	11	33%	45%	7	4	0	0	0
chum	1	2007	A	1	Oct 11	Oct 26-27	15-16	33	3	9%	45%	1	0	0	1	1
chum	1	2007	A	2	Oct-25	Oct 26-27	1-2	62	27	44%	55%	22	4	0	0	1
chum	1	2007	A	2	Oct-25	Oct 31-Nov 1	6-7	62	19	31%	55%	16	1	0	2	0
chum	1	2008	A	1	Oct 15	Oct 16-17	1-2	81	43	53%	37%	27	3	8	5	0
chum	2	2008	A	1	Oct 15	Oct 23-24	7-8	81	18	22%	37%	14	0	2	2	0
chum	2	2008	A	1	Oct 15	Oct 28-29	13-14	81	2	2%	37%	0	0	1	1	0
chum	2	2008	A	1	Oct 15	Nov 4-5	20-21	81	0	0%	37%	0	0	0	0	0
chum	2	2008	A/D	2	Oct 21	Oct 23-24	1-2	93	34	37%	35%	15	1	1	14	3
chum	2	2008	A/D	2	Oct 21	Oct 28-29	7-8	93	37	40%	35%	10	2	3	15	7
chum	2	2008	A/D	2	Oct 21	Nov 4-5	14-15	93	3	3%	35%	0	2	0	0	1
chum	2	2008	A/D	2	Oct 21	Nov 15	24-25	93	0	0%	35%	0	0	0	0	0
chum	2	2010	A/D	1	Oct 19	Oct 20	1	64	32	50%	56%	14	0	3	15	0
chum	2	2010	A/D	1	Oct 19	Oct 23	4	64	22	34%	56%	8	0	0	11	3
chum	2	2010	A/D	1	Oct 19	Oct 31	12	64	4	6%	56%	0	0	0	4	0
chum	2	2010	A/D	1	Oct 19	Nov 4	16	64	0	0%	56%	0	0	0	0	0
chum	2	2011	A/D	1	Oct 24	Oct 25	1	70	40	57%	49%	28	1	1	10	0
chum	2	2011	A/D	1	Oct 24	Nov 1	8	70	12	17%	49%	6	1	2	2	0
chum	2	2011	A/D	1	Oct 24	Nov 7	14	70	0	0%	49%	0	0	0	0	0

Appendix 2.1. continued

Species	Treat-ment	Year	Index site	Tag group	Tagging date	Recovery date	Duration (days)	Marks (M)	Recoveries (R)	R/M	% females	Recoveries by section				
												A	B	C	D	E
pink	1	2007	A	1	Oct 9-11	Oct 13	2-4	45	23	51%	22%	19	4	0	0	0
pink	1	2007	A	1	Oct 9-11	Oct 26-27	17-19	45	0	0%	22%	0	0	0	0	0
pink	2	2009	A/D	1	Sept 22	Sept 23	1	32	23	72%	59%	4	2	7	9	1
pink	2	2009	A/D	1	Sept 22	Oct 7	15	32	6	19%	59%	2	0	1	2	1
pink	2	2009	A/D	1	Sept 22	Oct 12	20	32	3	9%	59%	1	0	0	1	1
pink	2	2009	A/D	1	Sept 22	Oct 28	36	32	0	0%	59%	0	0	0	0	0
pink	2	2009	A/B	2	Oct 6	Oct 7	1	79	39	49%	65%	11	9	1	14	4
pink	2	2009	A/B	2	Oct 6	Oct 12	6	79	41	52%	65%	17	0	11	8	5
pink	2	2009	A/B	2	Oct 6	Oct 28	22	79	0	0%	65%	0	0	0	0	0
chinook	2	2010	A/D	1	Oct 19	Oct 20	1	17	9	53%	41%	1	1	0	4	3
chinook	2	2010	A/D	1	Oct 19	Oct 23	4	17	6	35%	41%	1	0	1	3	1
chinook	2	2010	A/D	1	Oct 19	Oct 31	12	17	3	18%	41%	0	0	0	1	2
chinook	2	2010	A/D	1	Oct 19	Nov 4	16	17	0	0%	41%	0	0	0	0	0
chinook	2	2010	A/D	1	Oct 19	Nov 13	25	17	1	6%	41%	0	0	0	1	0
chinook	2	2010	A/D	1	Oct 19	Nov 23	35	17	0	0%	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%	60%	0	0	0	0	14
coho	2	2010	A/D	1	Dec 4	Dec 19	15	20	9	45%	60%	0	0	0	0	9
coho	2	2010	A/D	1	Dec 4	Dec 31	27	20	6	30%	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

Appendix 2.2 Unadjusted live counts of pink salmon during 2003-2011.

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	-
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	-

Appendix 2.3 Unadjusted live counts of chum salmon during 2002-2011.

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

Appendix 2.3 continued (chum)

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

Appendix 2.3 continued (chum)

Year	Date	Run day	No. sites surveyed	Unadjusted count of the number of adults present					
				site A	site B	site C	site D	site E	non-index
2011	4-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	1-Nov	60	6	6757	575	702	794	183	493
2011	6-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	1-Dec	90	6	23	5	33	8	0	0
2011	7-Dec	96	5	-	0	5	0	0	0
2011	19-Dec	108	5	-	0	0	0	0	0

Appendix 2.4 Unadjusted live counts of coho salmon during 2002-2011.

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

Appendix 2.4 continued (coho)

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

Appendix 2.5 Unadjusted live counts of Chinook salmon during 2007-2011.

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
2007	27-Sep	10	5	0	0	0	0	0	-
2007	3-Oct	16	6	0	0	0	0	0	2
2007	13-Oct	26	5	1	0	0	22	44	-
2007	17-Oct	30	2	-	-	-	0 '	27 '	-
2007	27-Oct	40	6	2	0	34	7	134	5
2007	31-Oct	44	6	3	0	6	0	49	0
2007	6-Nov	50	5	0	0	0	1	22	-
2007	29-Nov	73	5	0	0	0	0	0	-
2008	23-Sep	6	5	5	5	0	0	0	-
2008	29-Sep	12	5	0	5	4	7	90	-
2008	6-Oct	19	5	6	0	1	22	166	-
2008	10-Oct	23	6	11	3	3	13	242	23
2008	17-Oct	30	6	3	1	24	36	190	10
2008	23-Oct	36	6	3	0	24	36	107	3
2008	29-Oct	42	6	0	0	0	9	68	0
2008	5-Nov	49	5	0	0	0	2	9	-
2008	15-Nov	59	5	1	0	0	1	2	-
2008	24-Nov	68	5	0	0	0	0	0	-
2009	3-Sep	1	5	0	0	0	0	0	-
2009	12-Sep	10	5	5	7	6	5	2	-
2009	17-Sep	15	5	2	0	2	10	12	-
2009	23-Sep	21	6	3	6	3	8	107	16
2009	7-Oct	35	6	7	6	9	81	250	35
2009	12-Oct	40	6	89	29	40	84	495	6
2009	20-Oct	48	3	-	-	-	-	263	-
2009	28-Oct	56	6	3	0	41	19	126	0
2009	4-Nov	63	5	0	0	0	0	27	-
2009	12-Nov	71	5	0	0	0	0	8	-
2009	24-Nov	83	3	-	-	0	0	3	0
2009	5-Dec	94	5	0	0	0	0	0	-
2010	3-Sep	1	5	0	0	0	0	0	
2010	10-Sep	8	6	2	0	0	0	2	1
2010	21-Sep	19	5	0	0	0	5	5	
2010	5-Oct	33	5	56	49	159	86	1025	
2010	12-Oct	40	6	52	18	150	250	1036	292
2010	20-Oct	48	6	52	22	97	281	915	114
2010	23-Oct	51	6	69	4	86	343	911	105
2010	31-Oct	59	5		0	43	213	625	69
2010	4-Nov	63	6	25	0	30	101	331	34
2010	13-Nov	72	4	11		8	30	58	
2010	23-Nov	82	5	0	0	0	1	10	
2010	29-Nov	88	4		0	0	0	0	
2011	10-Sep	8	5	1	0	0	0	0	
2011	17-Sep	15	5	3	1	0	0	0	
2011	24-Sep	22	5	10	9	31	38	32	
2011	10-Oct	38	6	17	20	75	268	800	74
2011	17-Oct	45	6	12	5	95	246	730	66
2011	25-Oct	53	6	4	9	38	181	505	33
2011	1-Nov	60	6	53	21	44	103	167	36
2011	6-Nov	65	6	23	7	10	62	159	19
2011	15-Nov	74	5	7	3	11	16	22	
2011	21-Nov	80	5	5	0	4	1	6	
2011	1-Dec	90	6	0	0	0	0	2	0
2011	7-Dec	96	5		0	5	0	0	0

Appendix 2.6 Mean daily flows in Coquitlam River at Port Coquitlam during the fall and winter spawning period in 2010-2011 and 2011-2012 (Water Survey of Canada, stn. 08MH141).
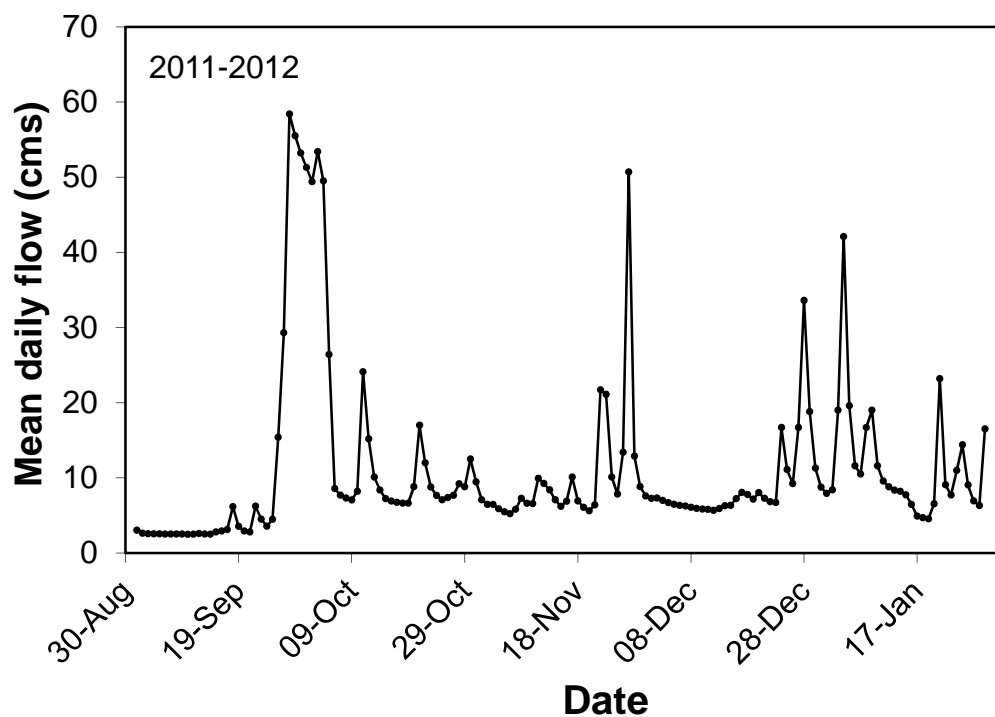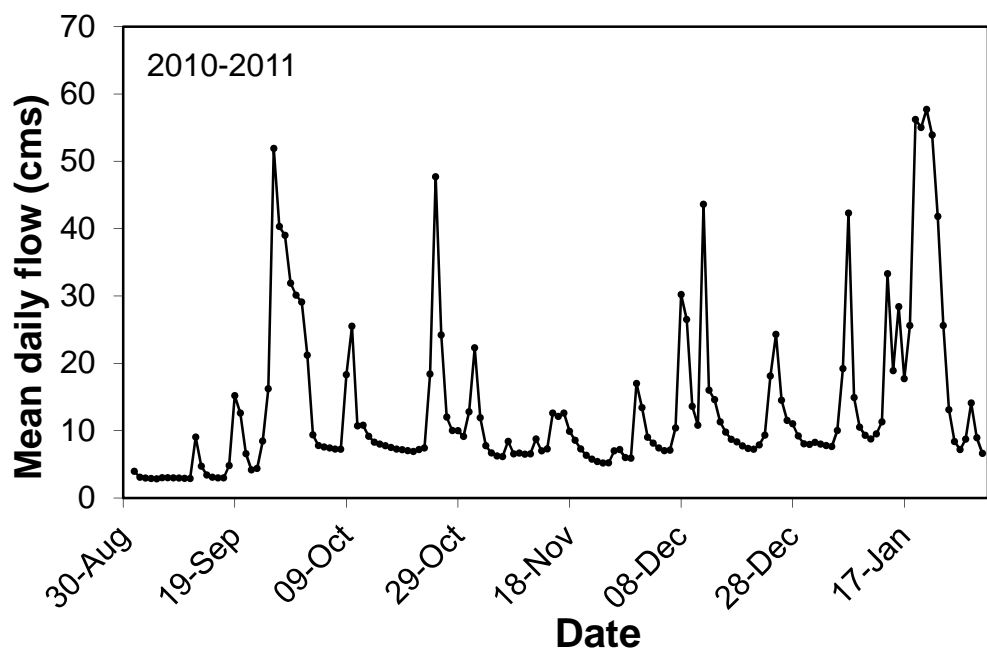
Appendix 2.7a. Diagnostic graphs used to evaluate model fit to the observed data for chum salmon in 2011. Top-left graph shows fit of predicted run timing curve (line) to unadjusted counts of spawners over time. Top-right shows relationship of predicted to observed counts with 95% credible intervals 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.
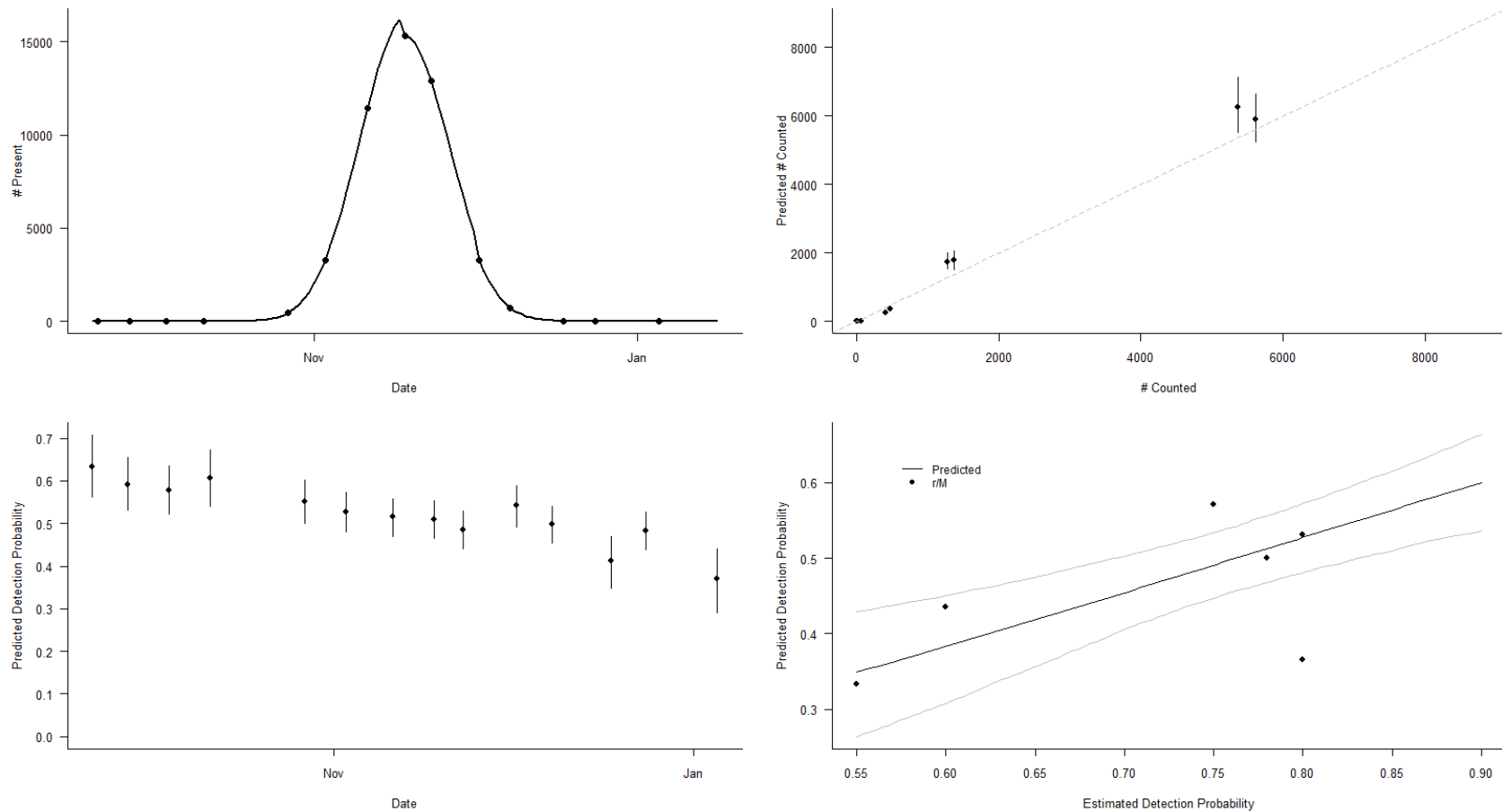
Appendix 2.7b. Diagnostic graphs used to evaluate model fit to the observed data for coho salmon in 2011. Top-left graph shows fit of predicted run timing curve (line) to unadjusted counts of spawners over time. Top-right shows relationship of predicted to observed counts with 95% credible intervals 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.
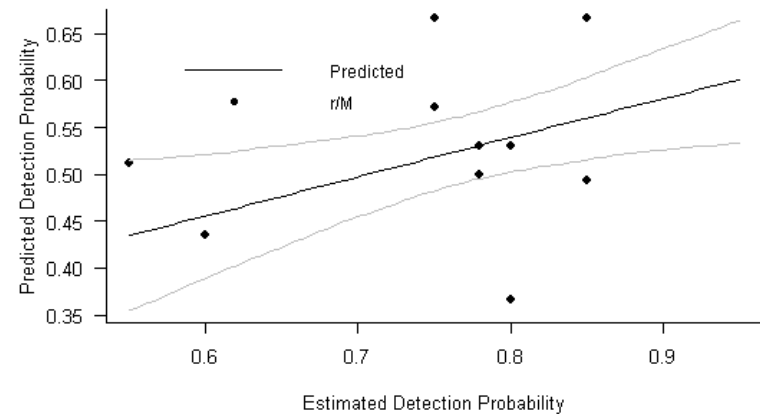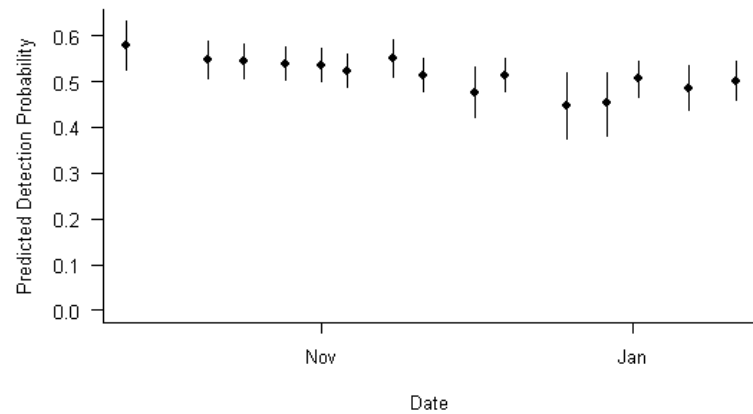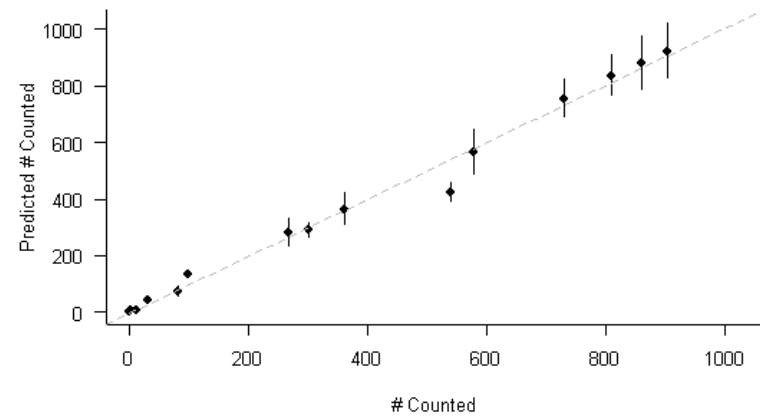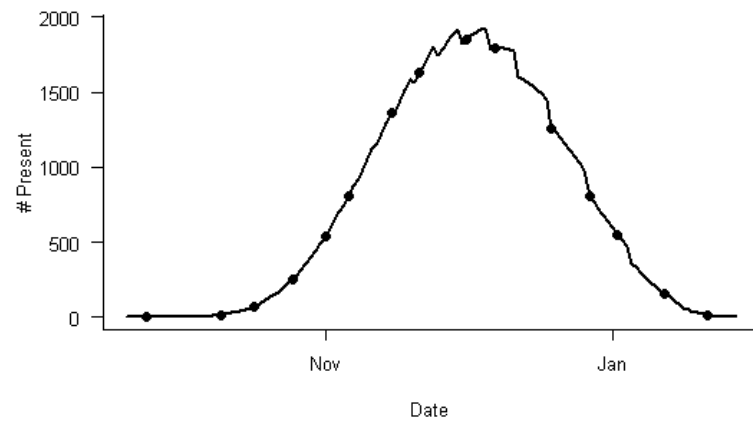
Appendix 2.7c. Diagnostic graphs used to evaluate model fit to the observed data for pink salmon in 2011. Top-left graph shows fit of predicted run timing curve (line) to unadjusted counts of spawners over time. Top-right shows relationship of predicted to observed counts with 95% credible intervals 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.
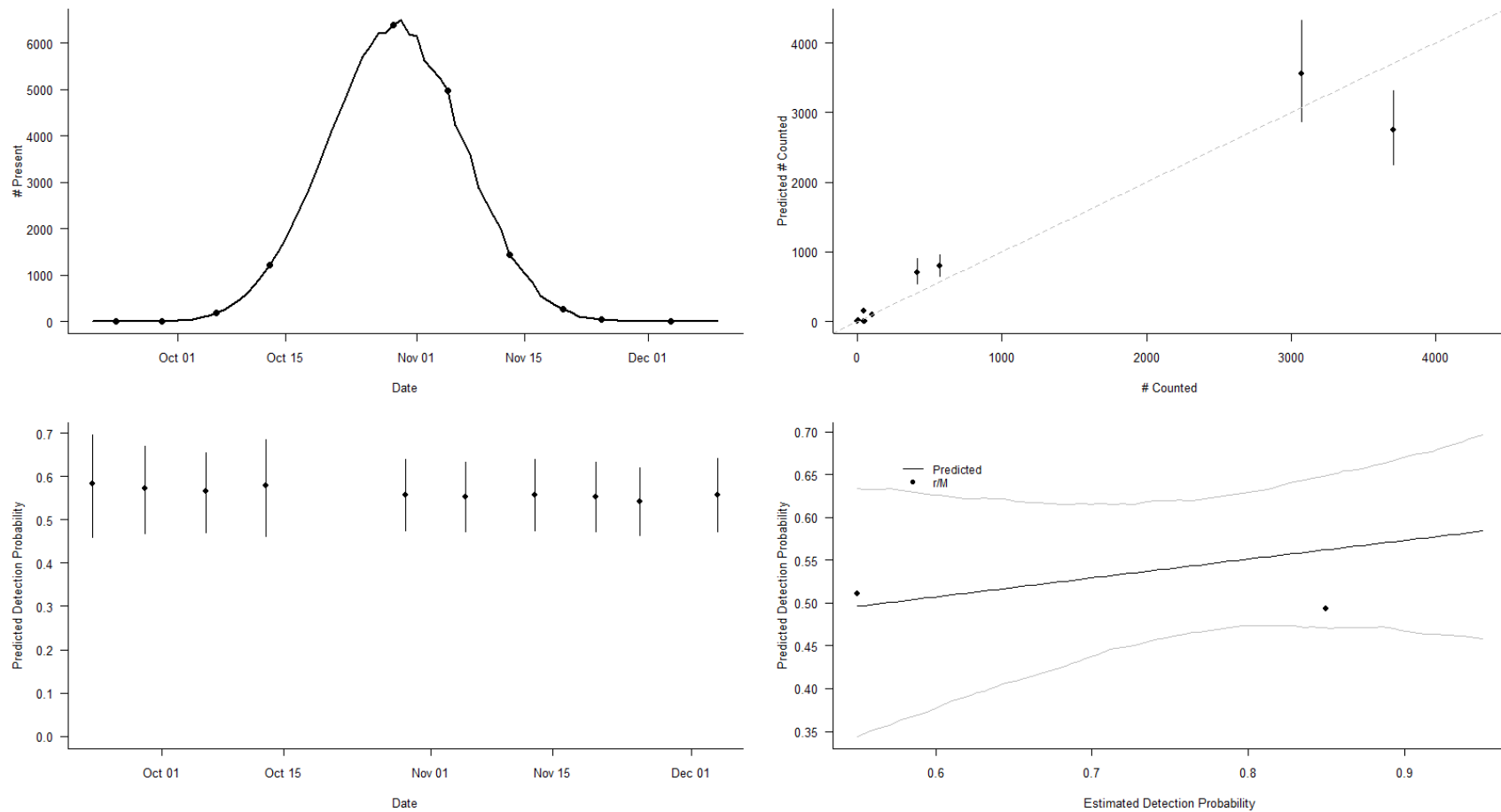
Appendix 2.7d. Diagnostic graphs used to evaluate model fit to the observed data for Chinook salmon in 2011. Top-left graph shows fit of predicted run timing curve (line) to unadjusted counts of spawners over time. Top-right shows relationship of predicted to observed counts with 95% credible intervals 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.
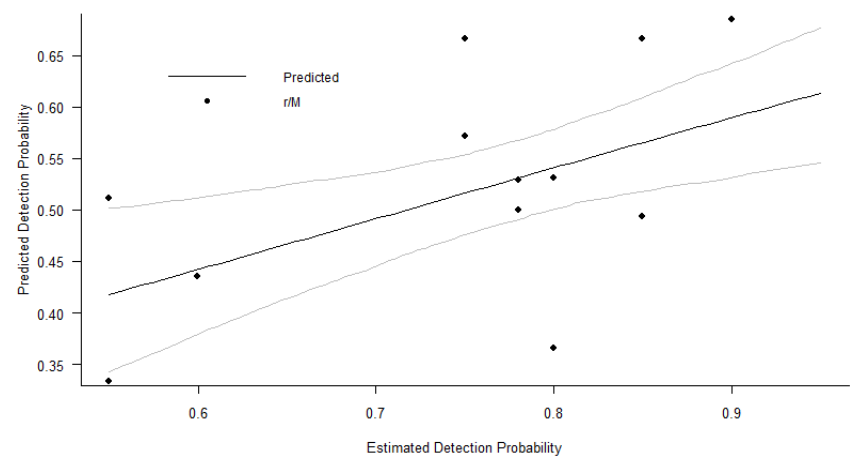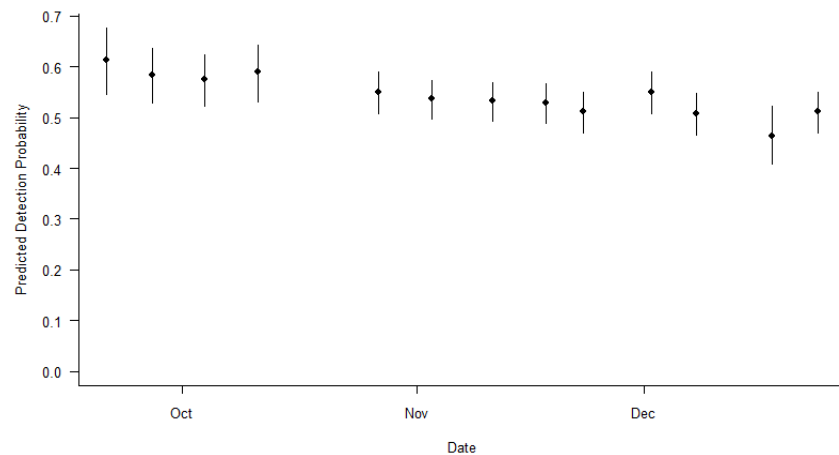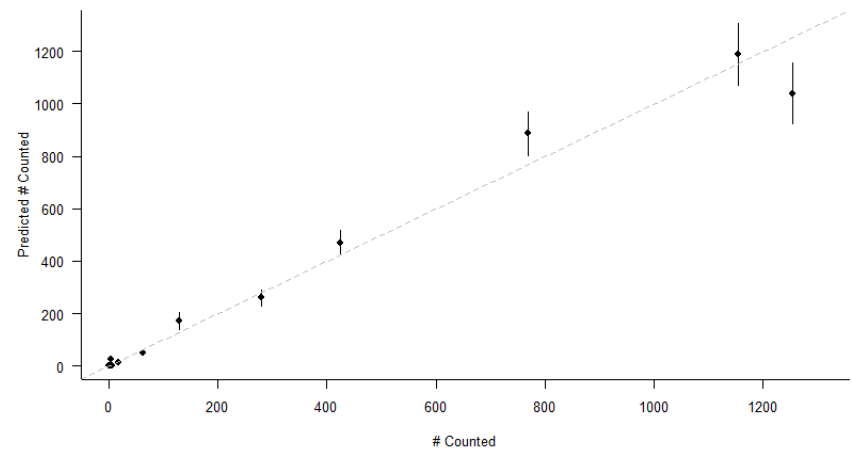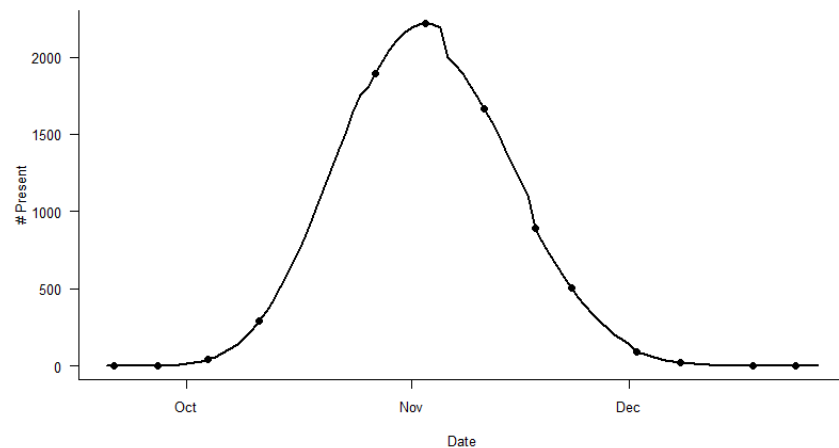
3.0 ADULT STEELHEAD ESCAPEMENT

3.1 Methods

During 2005-2011, 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. Beginning in 2007, our goal was to conduct weekly surveys from mid March through the first half of May, rather than bi-weekly surveys as in previous years. During 2005-2006, it was found that the majority of steelhead spawning occurred during this seven-week period. Analysis of previous years' data suggested that conducting weekly surveys ensures that redd survey life exceeds the survey interval (see Section 3.1.3). For the remainder of the spawning period ($\approx$ March 1-15, and May 15 - June 15), target survey frequency was every two weeks. 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 crewmember 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
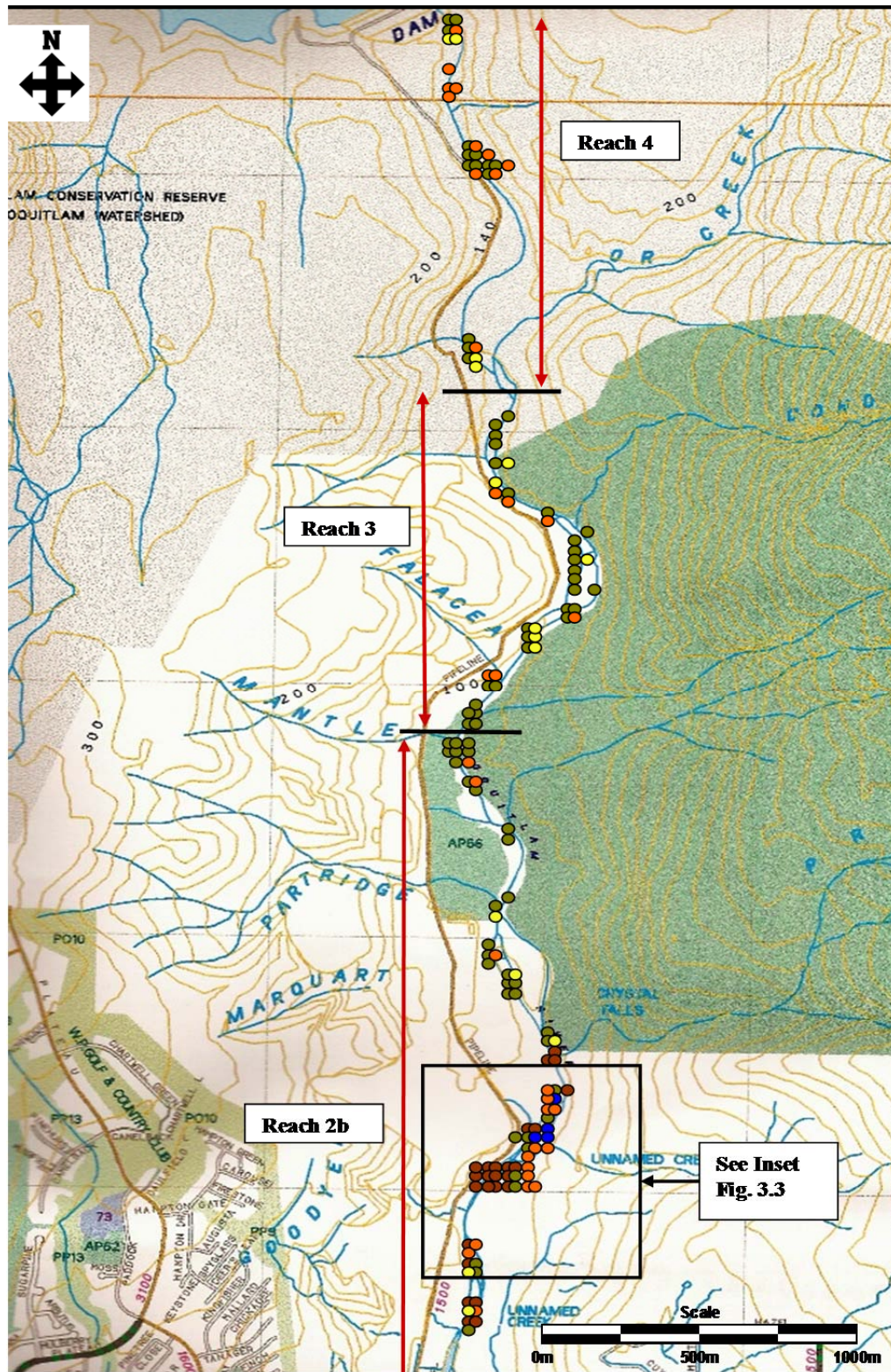
Figure 3.1 Steelhead redd locations in reaches 2b-4 in Coquitlam River in 2006, which was the highest escapement year during 2005-2011. Coquitlam Dam is the upstream boundary of the survey area. See Figure 3.2 for redd symbol legend.
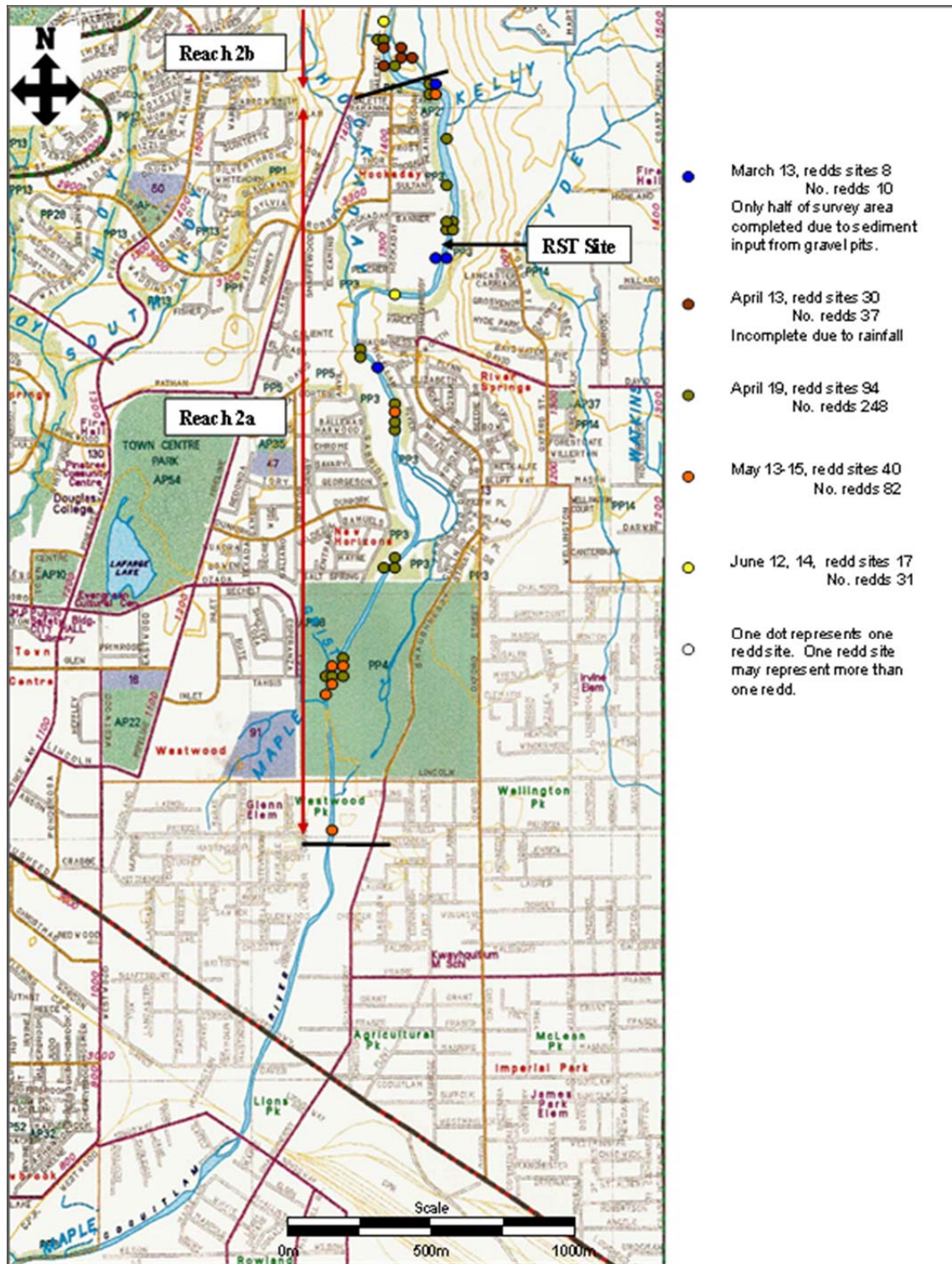
Figure 3.2 Steelhead redd locations in reaches 2a-2b in Coquitlam River in 2006. The downstream boundary of reach 2a is also the survey area boundary.

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 redds 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.

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. Since survey frequency during 2011 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). 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. 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.

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

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. Because redd numbers are a direct measure of spawning effort and egg deposition, 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 an averaged 1.2, with relatively little variably among years for individual streams. We used this value (1.2 redds/female) to convert total redd numbers to female escapement.

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

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

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).
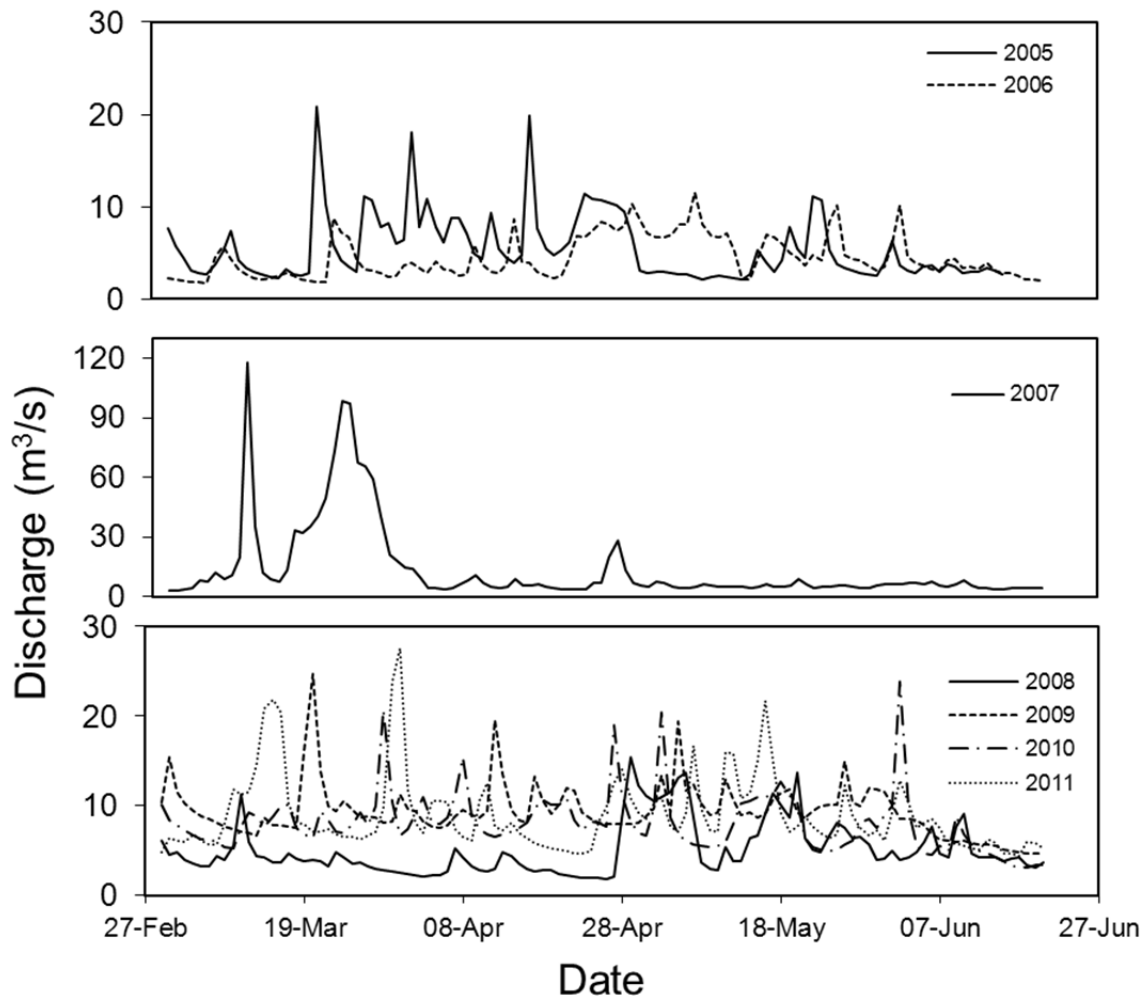
Figure 3.3 Discharge (cms) in Coquitlam River during steelhead spawning period in 2005-2011.

Table 3.1 Survey dates with raw counts of steelhead redds, estimated new redds, and live adult counts for all surveys during 2005-2011. 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**
2011	22-Mar	-	5	5	43
2011	5-Apr	14	21	21	61
2011	10-Apr	5	45	45	97
2011	20-Apr	10	83	83	103
2011	5-May	15	68	68	67
2011	21-May	16	24	24	36
2011	6-Jun	16	1	1	11
2011	**Total**		**247**	**247**	**peak = 103**

[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

3.2 Results and Discussion

During 2005-2011, 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 (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 2009-2011, 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 (7.0 cms) was more than double that on survey days during Treatment 1. 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 weekly surveys during the major portion of the spawning period was not met in 2011; across the entire survey period, the length of time between surveys ranged from 5 - 16 days, and averaged 12 days (Table 3.1). In previous years, the length of time between surveys has ranged from 6 to 37 days (Table 3.1).

In 2011, the first survey was conducted on March 22. Live adult steelhead were observed (33 fish, Table 3.1), and spawning had already begun (5 redds were counted, Table 3.1). Spawning had already begun by the time of the first survey in 2010 (March 15), 2009 (March 11), 2008 (March 21), and 2005 (March 24) as well (Table 3.1). In 2006 and 2007, adult steelhead were present on the first survey February 15 and March 1, respectively), but spawning had not yet begun (Table 3.1). These results suggest that steelhead typically begin spawning in the Coquitlam River in early March. In 2011, 88% of new redds were counted on surveys conducted from April 5 to May 5. 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).

Spawning steelhead preferred mainstem habitat 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 2011, 89% 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 – 2011 reduced the ability of the survey crew to spot adult steelhead compared to that in previous years under Treatment 1. Several sections of the river had increased turbulence that prevented ideal conditions for adult observations, 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 in 2011 (103) was the highest for the period on record (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). Counts of live adult steelhead in the Coquitlam River were as variable within years as peak counts were among years: during 2001-2011, CV for live counts within years (for surveys conducted during the April-mid May major spawning period) averaged 0.56, while the CV for peak count among years was 0.52.

For the first time since 2005, 4 redds were in found during the once-a-year reconnaissance survey of Reach 1. Reach 1 is not included in the annual steelhead redd survey and thus, the 4 redds are not included in estimates of adult escapement for the purposes of continuity with past years.

3.2.1 Redd survey life

In 2011, the period between surveys was sufficiently short ($\leq$ 16 days) to assume that only a negligible number of redds became obscured from one survey to the next based on the previous evaluation of redd survey life that close to 100% of redds remain visible 20-25 days after construction on the Coquitlam River (Decker *et al.* 2010). From 2005-2011, 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.
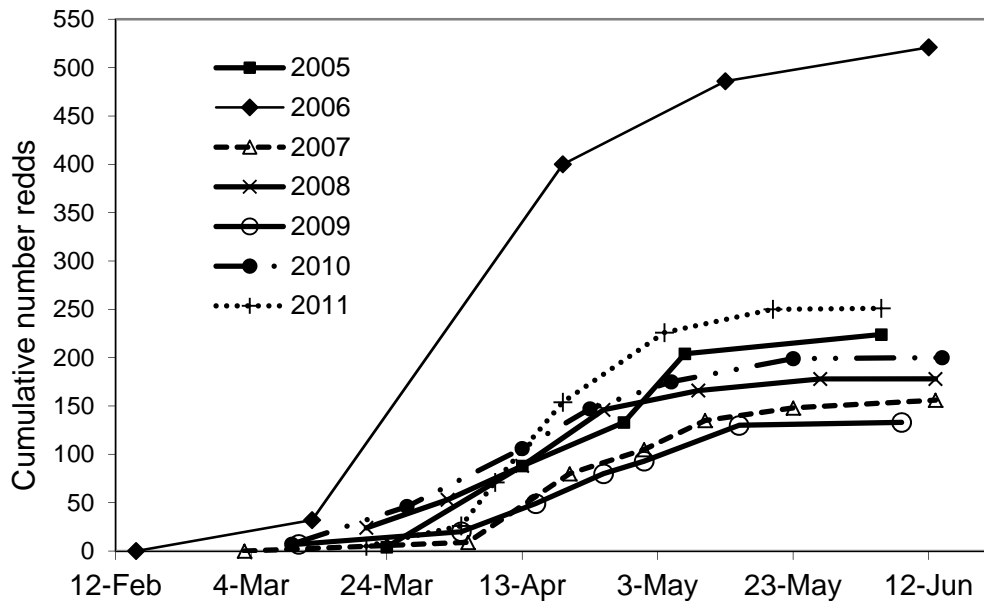
Figure 3.4 Cumulative proportion of the total steelhead redd count observed over time during 2005 and-2011.
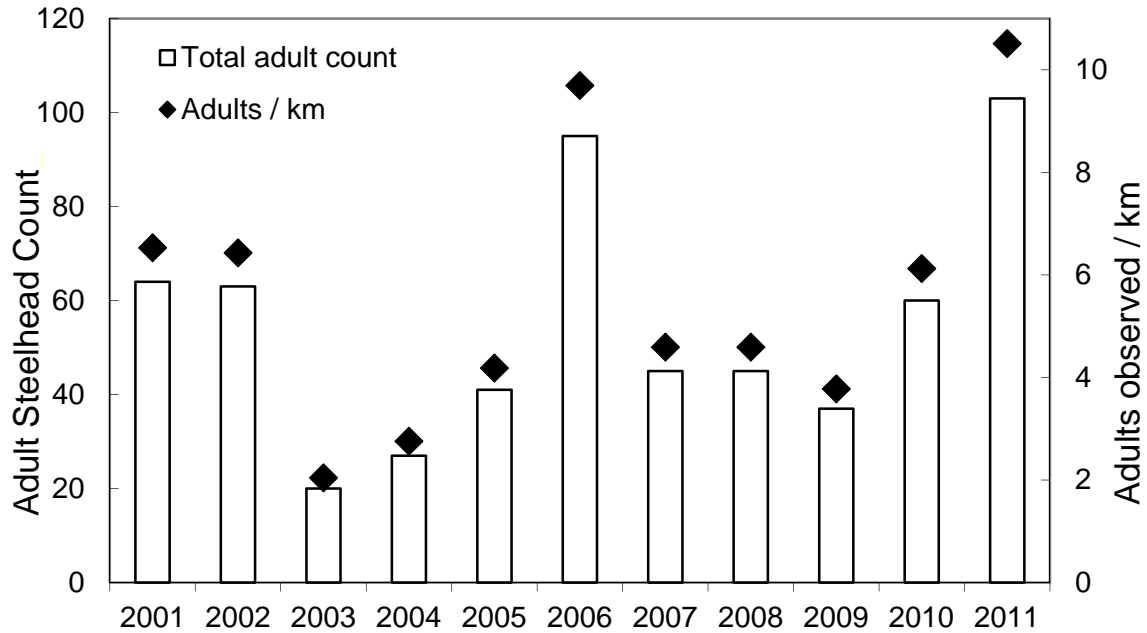
Figure 3.5 Peak counts and densities (fish/km) of adult steelhead during snorkeling surveys in 2001-2011 (only data for complete surveys of the study area are shown). Data for 2001-2004 were collected as part of a separate study (BCCF, Lower Mainland Branch, data on file).

3.2.2 Female escapement and egg deposition

Estimated adult female escapement in 2011 was 206 females (Table 3.2), which represented a median value for the 2005-2011 period of record. 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 23 redds/km in 2011, and ranged from 13-48 redds/km during 2005-2011 (Table 3.2). Among reaches and years, redd density ranged from 6-71 redds/km (Table 3.2). Spawning distribution was fairly similar during 2005-2007 and 2010-2011, with reaches 2a and 2b accounting for about half of the total number of redds, and reaches 3 and 4 accounting for the other half. In 2008 and 2009, the proportion of total redds found in reaches 3 and 4 was reduced to 29% and 38%, respectively. Figures 3.1 and 3.2 illustrate the fine-scale distribution of redds in the study area in 2006 (see Macnair 2011a, Appendix 3 for a redd distribution map for 2011).

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 and redds/female ratios remain constant between treatments.

3.3 Recommendations for the 2012 survey

The Coquitlam River is well suited to conducting steelhead redd surveys. The redd survey life model suggested that the number of redds that went undetected during 2005-2011 as a result of overly long periods between surveys was negligible in all years except 2006. 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 2005-2011. The potential for undercounting of redds and the need for a redd survey life model to correct for this was demonstrated in one instance in 2006 when survey frequency was reduced as a result of poor visibility conditions.

1. 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.

2. We recommend that the first survey occur in early March, and that additional surveys be conducted beyond June 15 if a significant number of new redds are observed at the end of the normal survey period. Every effort should be made to take advantage of periods of clear water conditions during this period, with surveys being conducted less than the scheduled interval if necessary to avoid high water events. Jacobs et al. (2002) recommended conducting steelhead redd surveys in Oregon coastal streams on a 7-10 day recurrence interval. We also recommend the continued use and development of the redd life models described Decker *et al.* 2010 in place of the model proposed in the CRMP-TOR. Finally, where possible, two survey crews should be employed rather than one so that the entire study area can be surveyed in one day, thereby decreasing the risk of a survey being aborted due to the onset of poor conditions. For the most part, incomplete or missed redd surveys were not a large source of uncertainty in 2005-2011, but this may not be the case in future years.

Table 3.2 Summary statistics for steelhead escapement to Coquitlam River during 2005-2011 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	(178-533)	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	(413-1,240)	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	(124-371)	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	(141-424)	27
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	(107-321)	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	(159-476)	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	(198-588)	38

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 = 0.029CSI - 0.002R - 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**

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 2007-2011 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 2006-2011 (see *Section 4.1.5*); this model replaces a bootstrap model used in previous years (Decker et al. 2012).

During 2007-2011 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 that are sampled each year. 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 $_{index\ sites\ 1\text{-}12}$) was estimated based on estimates

of the mean and variance in fish density for the 12 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 2007-2011 the 10 sites originally selected in 2006 were re-sampled, and an additional two sites were added in reach 4 to maintain a uniform sampling interval of $\approx$ 0.85 km (Figure 4.1; Reach 4 was not sampled in 2006). 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 (2009-2011). 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).
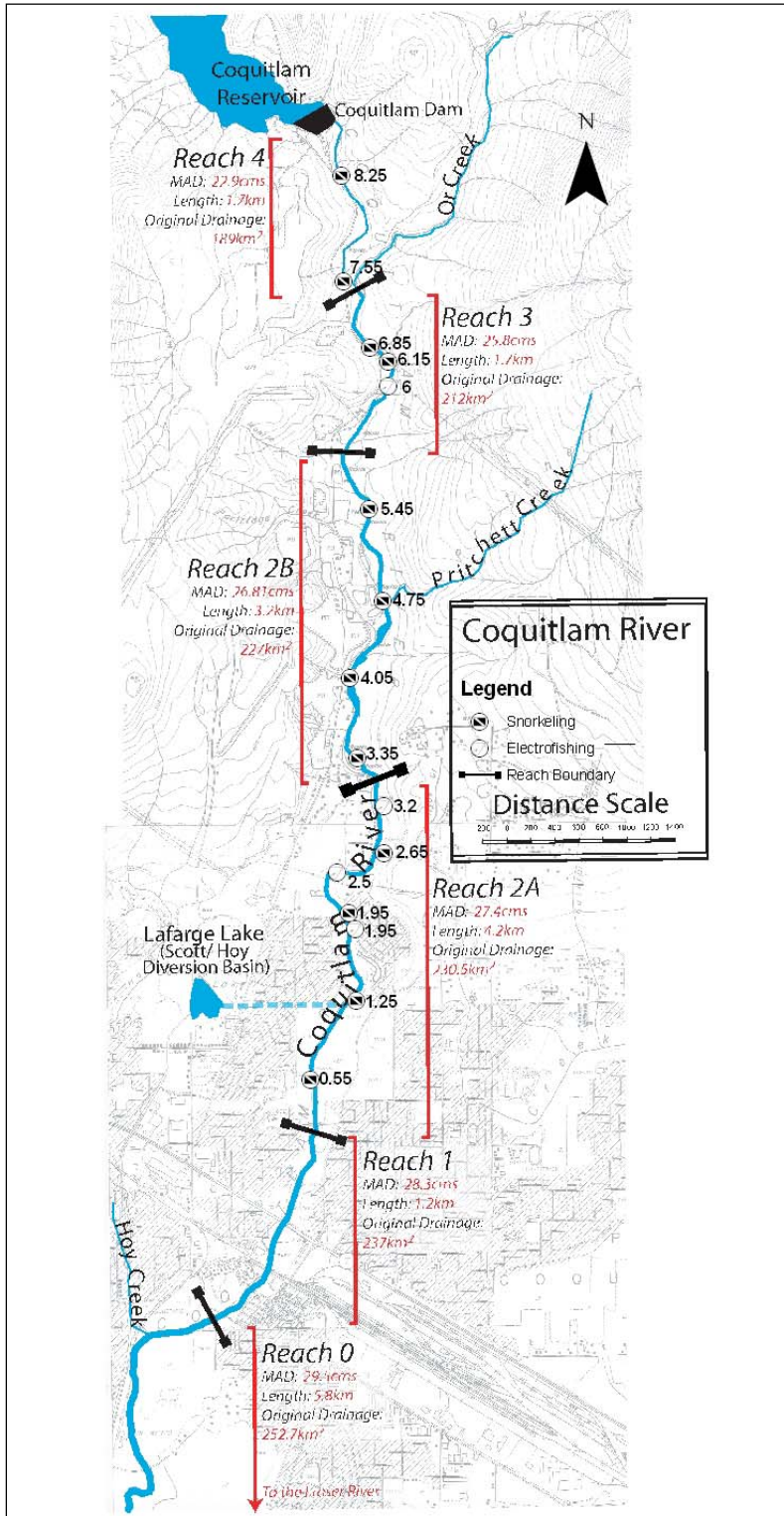
Figure 4.1 Map of Coquitlam River showing juvenile standing stock study area, reach breaks and sampling sites.

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 study design for the juvenile standing stock component 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 mark-recapture experiments at a total of 16 sites during 2007-2011 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.

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 crew member 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 resighting 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 2008). 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 veresus fully integrated two-phased estimation approaches and found the increase in uncertainty under the latter approach was relatively modest (a few %). Based on their results, we adopted the more computationally efficient median approach. Korman et al. (2010) also used computer simulations to evaluate the extent of bias in standing stock estimates and hyper-parameters generated from the Cheakamus River HBM and found that bias to be negligible in all cases.

To describe the precision of the standing stock estimates, in this report we have used percent relative error, which we computed as the average half credible interval (upper 95% credible limit minus the lower credible limit divided by two and then divided by the mean and expressed as a percentage; Krebs 1999). It is important to note that standing stock estimates and confidence intervals reported here will differ in future years' reports as estimates of size-specific snorkelling detection probability are refined by additional mark-recapture experiments, or if 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 significant predictors of snorkeling detection probability.

4.1.6 Day electrofishing survey

In 2011 we resurveyed four shoreline electrofishing sites previously sampled during 2007-2009. 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), 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).

Table 4.1 Summary of habitat data for night snorkeling and day electrofishing sites in Coquitlam River in 2011.

Sampling method	Site no.	Upstream distance (km)	Site area (m²)	Mean length (m)	Mean width (m)	Mean depth (m)	Mean velocity (m)	D90 (m)	Boulder (%)	Cobble (%)	Gravel (%)	Fines (%)
snorkeling	0.55	8.3	463	25	19	0.44	0.35	0.75	30	40	25	5
snorkeling	1.25	9.0	458	25	18	0.36	0.45	0.65	40	30	25	5
snorkeling	1.95	9.8	488	25	20	0.41	0.41	0.75	45	35	15	5
snorkeling	2.65	10.8	440	25	18	0.67	0.23	0.9	25	40	20	15
snorkeling	3.35	11.7	549	25	22	0.28	0.23	0.4	10	45	35	10
snorkeling	4.05	12.5	901	25	36	0.31	0.37	0.6	25	50	20	5
snorkeling	4.75	13.3	444	25	18	0.50	0.34	0.7	40	40	25	5
snorkeling	5.45	14.1	476	25	19	0.33	0.40	0.7	25	40	30	10
snorkeling	6.15	14.9	324	25	13	0.44	0.46	0.7	30	35	25	10
snorkeling	6.85	15.8	400	25	16	0.37	0.29	0.64	15	40	45	10
snorkeling	7.55	16.6	504	25	20	0.38	0.24	0.9	45	35	15	10
snorkeling	8.25	17.4	406	25	16	0.29	0.41	0.7	30	50	15	10
electrofishing	1.95	9.8	120	15	8	0.35	0.34	1.05	40	35	20	5
electrofishing	2.55	10.7	144	19	8	0.30	0.35	0.6	25	40	30	5
electrofishing	3.20	11.5	166	21	8	0.23	0.31	0.8	35	30	25	10
electrofishing	6.00	14.5	117	14	9	0.28	0.29	0.75	30	55	10	5

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 2009-2011 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 2011, the night snorkeling survey was completed during August 26-September 3, at flows of 2.5-4 cms (≈41%-65% post-flow regulation MAD; station 08MH002, Port Coquitlam). Previous surveys were conducted at flows of 0.8-2 cms during Treatment 1, and 5-6 cms during Treatment 2. Water temperatures ranged from 18°C-19°C during 2011, similar to previous years. In 2011, horizontal underwater visibility exceeded 4 meters (2008, 3-4 metres; all other years, >4 metres), which is more than adequate for conducting snorkeling counts (Hagen et al. 2011)

4.2.1.1 Mark-recapture experiments to estimate snorkeling detection probability

During 2007-2011, we marked totals of 303 coho fry, 289 steelhead fry, and 290 steelhead parr at 16 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 56% (Table 4.2, Figure 4.2), whereas for steelhead, detection probability ranged from 28% for the < 50 mm length class of steelhead, to 65% 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 (53%) remains highly uncertain given the limited number of tagged fish for this size class (seventeen 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 not strictly met when mark-recapture were expanded to account for small-scale fish movement. Across the 16 mark-recapture sites, 12 marked coho, 16 marked steelhead fry, and 22 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 21 of 303 marked coho (7.1%), 44 of 289 marked steelhead fry (15.0%), and 35 of 290 marked steelhead parr (age-1+ and 2+ combined: 12.0%), had moved from the original marking site to one of the adjacent sections during the 24-hour interval between the marking and resighting 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.

Table 4.2 Summary of mark-recapture results and snorkeling detection probability estimates for 16 sites in Coquitlam River combined for all years (2007-2011).

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	303	170	0.56	0.18	12	21
Steelhead	<50	127	36	0.28	0.17	7	25
Steelhead	50-69	162	78	0.48	0.20	9	19
Steelhead	70-99	99	64	0.65	0.31	5	8
Steelhead	100-140	174	112	0.64	0.25	15	23
Steelhead	>140	17	9	0.53	0.44	2	4
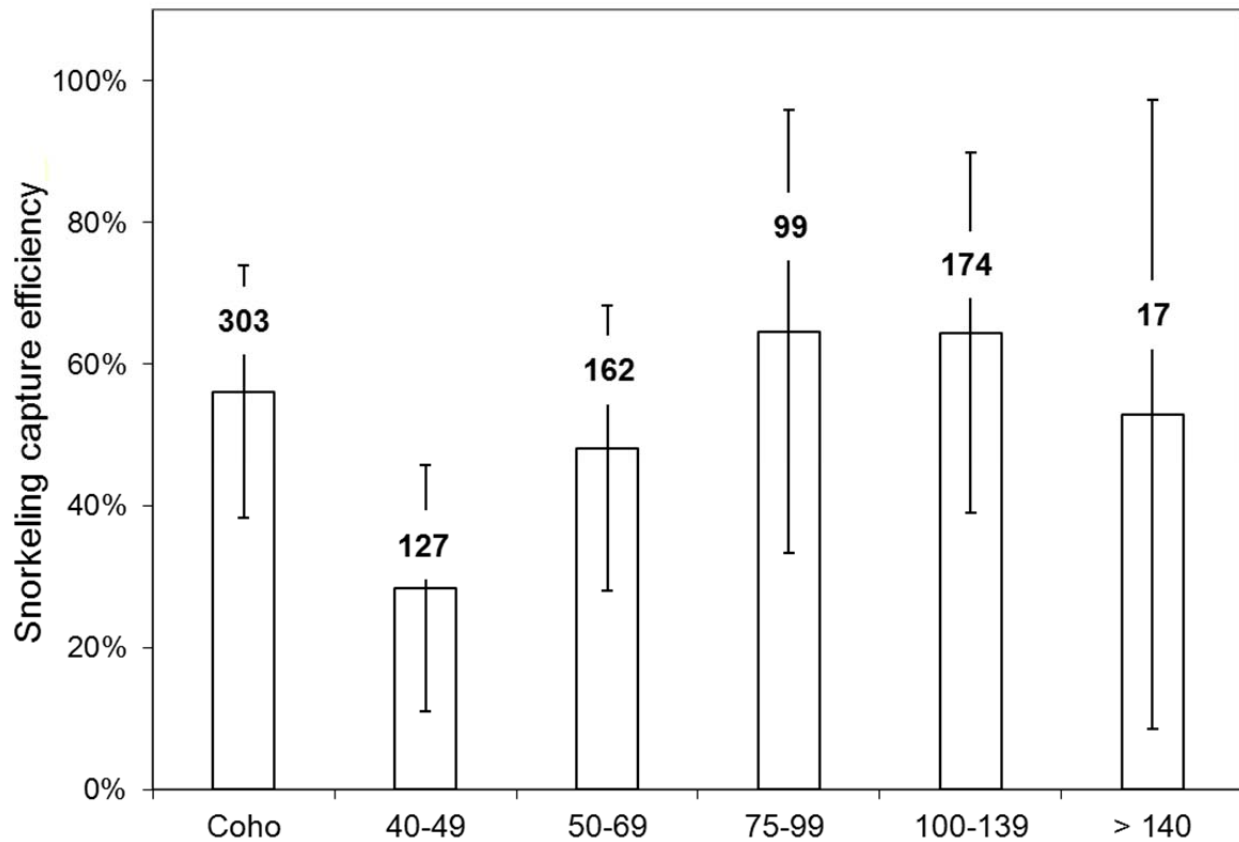
Figure 4.2 Maximum likelihood estimates of mean snorkeling detection probability for juvenile coho and steelhead by forklength class (steelhead only) at 16 sites in the Coquitlam River during 2007-2011. Errors bars represent ±1 standard deviation of the mean.   Values above bars are total numbers of marked fish for each category.

4.2.1.2 Juvenile fish distribution and abundance

In 2011, coho fry abundance in Coquitlam River mainstem was about two-fold higher than the previous three years (56 vs. 26-29 fish/100m^2, respectively, Table 4.3), and about four-fold higher compared to 2006 and 2007 (13 and 12 fish/100m^2, respectively). 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 2007, 2008 and 2010 (R=0.73, 0.62, 0.85, respectively, P<0.01 for all cases; Figure 4.3), but not during 2009 or 2011 (R= -0.07, P>0.05 and R=0.52, P=0.14, respectively).

Steelhead fry density was three-fold higher in 2006 (61 fish/100m^2, Table 4.3) compared to that during 2007-2011 (18-21 fish/100m^2). However, the 2011 snorkeler derived fry estimate is not considered credible as it is likely biased low, a product of an overestimation of the detection probability applied to fry < 40mm forklength. In 2011, fry less than 40mm forklength comprised 13% and 47% of snorkeler and electrofishing counts, respectively, compared with average snorkeler and electrofishing counts of 0% and 14%, respectively, during 2008-2010 (appendix 4.4, See section 4.3.1 for explanation). In this situation, using electrofishing may provide a more informative index of fry density as the two gear types are highly correlated

(R=0.97) on a fry/m^2 basis but is less informative for a river wide abundance estimate since that is based on fry/km, which the two gear types have a weaker correlation (R=0.84). Adjusting the electrofishing density estimate to reflect the relationship between the two gear types (snorkeler estimate = electrofishing x 0.76) suggests a 2011 fry density roughly double that of the mark recapture base snorkeler estimate (39 and 21 fish/100m^2, respectively).

Annual steelhead fry standing stocks ranged from 33,000-128,000 (Table 4.3). During 2006-2007, steelhead fry densities were substantially higher in the middle portion of the study area (reaches 2b and 3) compared to upper and lower reaches (Figure 4.3), whereas in 2008-2011, the only clear pattern was lower fry densities in reach 4 compared to downstream reaches.

Mean density of age-1+ steelhead parr varied from 2.9-8.1 fish/100m^2 during 2006-2011(Table 4.3), with no apparent temporal trend. Total standing stock of age-1+ parr varied from 6,200-12,900 among years (Table 4.3). Age-1+ parr abundance was similar in 2007 and 2008 despite a three-fold difference in cohort fry abundance for the two years (Table 4.3), suggesting survival rates are highly density-dependant. Mean density of age-2+ steelhead parr was higher during the three years of Treatment 2 (1.8-2.1 fish/100m^2 during 2009-2011; Table 4.3) compared to that during Treatment 1 (0.8-1.2 fish/100m^2 during 2006-2008). During 2006-2011, age-1+ parr represented 71% to 87% of the total parr standing stock (age-1+ and 2+ combined). There was no strong longitudinal pattern in steelhead parr density among sites in 2006 or 2009-2011 (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).

Table 4.3 Estimates of juvenile fish density, standing stock, and 95% confidence intervals by species and age class in Coquitlam River during 2006-2011. Estimates were derived exclusively from night snorkeling counts.

Species/age class	Year	Density (fish/km)	Density (fish/100m^2)	Standing stock	Lower 95% CI	Upper 95% CI	± 95% CI
coho (0+)	2006	2,089	13.3	21,520	13,049	36,959	56%
coho (0+)	2007	2,061	11.9	21,225	12,937	47,095	80%
coho (0+)	2008	4,735	27.4	48,775	32,066	82,468	52%
coho (0+)	2009	5,521	27.3	56,870	39,936	102,008	55%
coho (0+)	2010	6,182	28.7	63,675	42,438	115,005	57%
coho (0+)	2011	10,214	55.7	105,200	67,932	224,620	74%
steelhead (0+)	2006	13,833	59.4	142,485	98,438	220,986	43%
steelhead (0+)	2007	2,674	18.4	27,547	18,416	42,462	44%
steelhead (0+)	2008	4,050	23.5	41,715	29,558	72,536	52%
steelhead (0+)	2009	3,696	17.6	38,070	24,099	82,362	77%
steelhead (0+)	2010	3,803	20.5	39,170	28,065	59,574	40%
steelhead (0+)[1]	2011	8,107	51.0	83,502	-	-	-
steelhead (1+)	2006	605	2.9	6,231	3,602	16,883	107%
steelhead (1+)	2007	1,020	6.5	10,510	6,894	17,053	48%
steelhead (1+)	2008	1,038	5.5	10,690	7,195	19,303	57%
steelhead (1+)	2009	1,055	4.8	10,870	7,943	15,662	36%
steelhead (1+)	2010	742	3.8	7,647	6,276	10,422	27%
steelhead (1+)	2011	877	5.2	9,034	6,337	13,805	41%
steelhead (2+)	2006	159	0.8	1,641	887	3,390	76%
steelhead (2+)	2007	177	1.2	1,820	959	3,640	74%
steelhead (2+)	2008	112	1.0	1,153	581	2,198	70%
steelhead (2+)	2009	314	2.1	3,232	1,916	5,621	57%
steelhead (2+)	2010	254	1.8	2,617	1,493	4,714	62%
steelhead (2+)	2011	348	2.1	3,583	2,074	6,340	60%
steelhead (all parr)	2006	764	3.7	7,872	-	-	-
steelhead (all parr)	2007	1,197	7.6	12,330	-	-	-
steelhead (all parr)	2008	1,150	6.5	11,843	-	-	-
steelhead (all parr)	2009	1,369	6.8	14,102	-	-	-
steelhead (all parr)	2010	996	5.5	10,264	-	-	-
steelhead (all parr)	2011	1,225	7.2	12,616	-	-	-

[1]Biased low estimate due to overestimate of age-0+ detection probability (see section 4.3.1 for explanation)
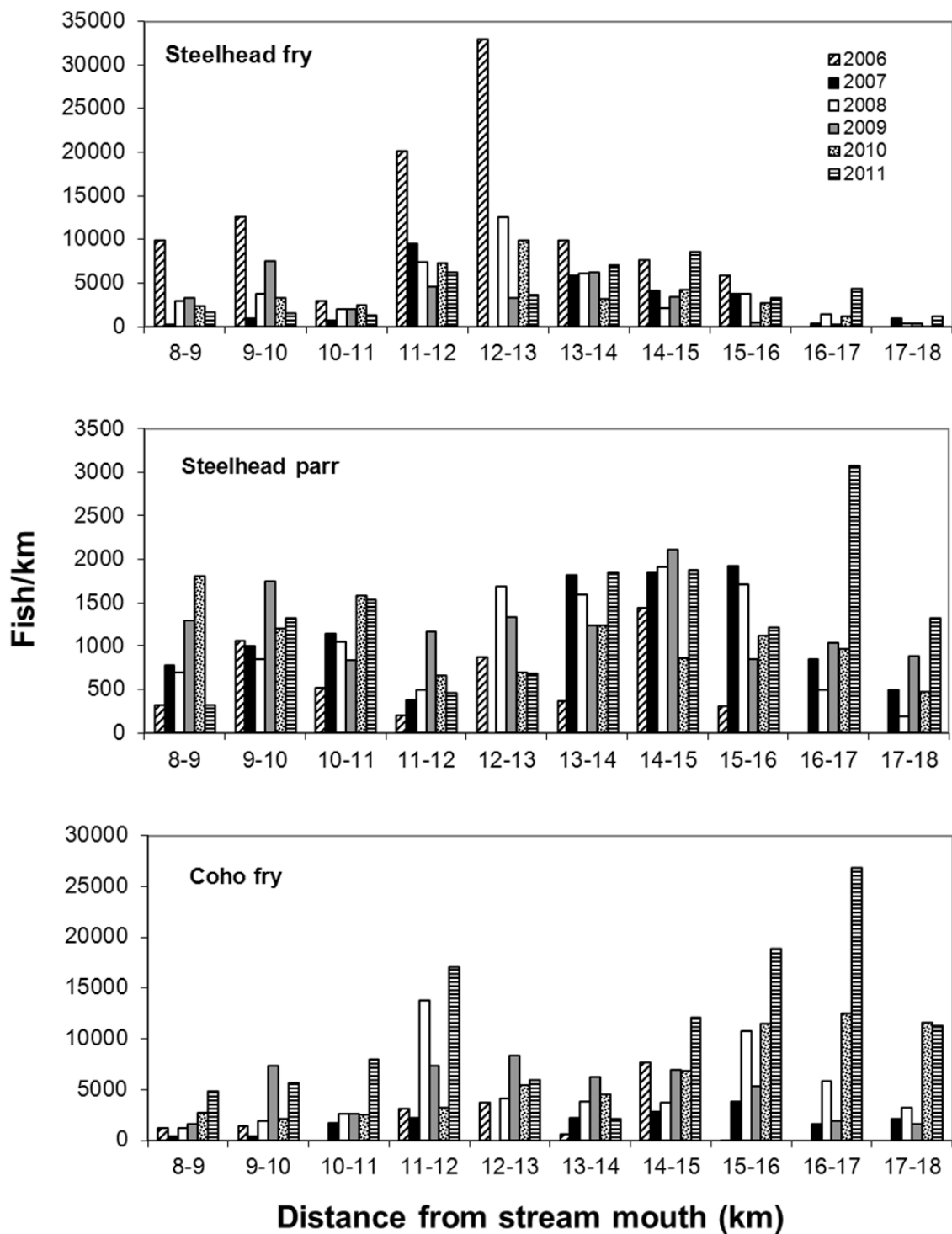
Figure 4.3 Linear distribution of juvenile salmonids in the Coquitlam River during 2006-2011. Estimates are based on calibrated snorkeling counts at 10-12 sampling sites per year. Fish density values are averaged for all sites falling within each 1 km long stream section.

Table 4.4 Summary of day electrofishing results at four one-shoreline sites in the Coquitlam River in 2011. Mean fish density estimates are also shown for 2006-2011. The electrofishing survey was conducted at the same four sites during 2007-2011, 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)	fish/km
					Coho fry				
2011	1.95	8	3	0	11	11	12	9	743
2011	2.50	8	8	4	27	20	45	19	1,436
2011	3.20	13	4	1	18	18	19	11	849
2011	6.00	11	4	1	17	16	17	15	1,259
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
					Steelhead fry				
2011	1.95	19	13	5	42	37	52	35	2,838
2011	2.50	11	12	6	45	29	82	31	2,394
2011	3.20	53	19	4	77	76	80	46	3,632
2011	6.00	59	28	11	106	98	116	91	7,852
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
					Steelhead parr (1+)				
2011	1.95	8	5	1	14	14	16	11.6	946
2011	2.50	5	3	0	8	8	9	5.6	426
2011	3.20	4	3	0	7	7	8	4.2	330
2011	6.00	0	0	0	0	0	0	0.0	0
2006	all sites							3.4	206
2007	all sites							11.0	891
2008	all sites							6.8	493
2009	all sites							6.7	505
2010	all sites							2.7	200
2011	all sites							5.4	425
					Steelhead parr (2+)				
2011	No age-2+ parr captured							0.0	0
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

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 resighting 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 resighting 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 7% (coho fry) to 23% (steelhead 1+ parr) of the total number marked. We included these movers as part of the resighted 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 the size class during mark-recapture experiments match that during index sampling. However, in 2011 the catchable population included steelhead fry smaller than were included in previous mark recapture experiments. Steelhead fry less than 40mm are not included in mark recapture experiments as the minimum tag size is considered too large to satisfy the assumption of equal catchability of marked and unmarked individuals. In the absence of a direct measure of detection probability for fry smaller than 40mm, all those under 50mm were applied the same detection probability even though it was likely much lower as it is inversely related to forklength and can drop to near zero for the smallest fry (Korman et al. 2011). Thus, the snorkeling estimate likely underestimated steelhead fry abundance by as much as 50% considering that the electrofishing estimate, highly correlated with snorkeling (R=0.97), detected a doubling of fry abundance from 2010 to 2011 whereas snorkeling showed almost no difference.

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 2006-2011 (12-55 fish/100 m^2) were 2-10 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 2006-2011 were 1.5- to 5-fold higher (18-61 fish/100 m^2). Steelhead parr densities were 7-19 times higher during 2006-2011 (3.7-9.4 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 2006-2011 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 and 2011 (61 and 51 fish/100 m^2, respectively) was relatively high compared to published values for other streams, while densities in 2007-2010 (18-51 fish/100 m^2) were somewhat 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 and 2011 was associated with a relatively high brood escapement (see Section 3), which is consistent with the positive linear relationship between steelhead escapement and fry abundance that has been observed in other streams (e.g., Keogh River, Ward and Slaney 1993).

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

Coho densities in the Coquitlam River (11-55 fish/100 m^2) were much lower than the range of mean coho densities observed at annual index sites in 15 other Lower Mainland streams (59-455 fish/100 m^2, respectively; DFO, data on file), although these streams were considerably smaller, and were sampled at sites chosen to represent 'good' coho habitat. It is important to note that constructed off-channel habitat contributes about half of coho smolt production in the Coquitlam River, and numbers of coho fry from off-channel areas were not included in our estimates of mean densities and standing stocks in Table 4.3.

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

4.3.3 Fish densities in 'optimal' habitats based on electrofishing

In comparison to fish density estimates derived from the electrofishing data, the snorkeling data for 2006-2011 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 bases 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 age-1+ parr and were highly correlated for the two methods (R=0.90), so too were those for steelhead fry (R=0.97 excluding 2011). The substantial difference in the 2011 fry estimates is likely due to a negative biased snorkeler abundance estimate rather than a positive biased electrofishing estimate (See section 4.3.1 for further explanation) and cannot be explained by sampling error alone as the 95% confidence limits of the 2011 estimates do not overlap (45-61 fish/100 m^2 based on electrofishing versus 13-34 fish/100 m^2 based on snorkeling).

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 a trend of increasing density during 2008-2011 compared to that in 2006-2007 (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 in 2006-2011 (Table 4.4), whereas snorkelers detected an average of 4 age-2+ steelhead per site during 2006-2011.

Electrofishing surveys in Coquitlam River during 2007-2011 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 and 38-121 g/100m^2 during Treatment 1 (2006-2008) and Treatment 2 (2009-2010), 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 2006-2011 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 2006-2011 failed to include 'optimal' sites where maximum steelhead biomass would be expected.

4.4 Recommendations for future surveys

1. As much as possible, the same 12 snorkeling sites should be sampled each year.

2. Snorkeling sites should be 25 m in length, should span the entire wetted width of the stream, and where the stream is braided, all channels within the 25 m length of stream should be surveyed.

3. Electrofishing sites should be a minimum of 20 m in length. To ensure useful estimates, continue to use a minimum of 3 passes at each site, adding additional passes if a satisfactory depletion pattern is not achieved for all age classes and species.

4. Mark-recapture experiments should by conducted each year at a subset of the regular sampling sites ($\approx$ 2-4 sites each year) so that a Coquitlam River-specific model of snorkeling detection probability can continue to be refined. Across years, mark-recapture experiments should be equally distributed among the 12 permanent sampling sites. Mark-recapture methodology should follow that described in Hagen et al. (2011) and Decker and Hagen (2009).

5. Marking age-2+ steelhead should take priority over all other steelhead and coho age classes to increase the precision of the capture efficiency estimate. This should be achieved by adjusting the length of each mark-recapture to satisfy the marking target for age-2+ steelhead (20+ fish).

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

Variable	Description

Data

$r_{i,g}$ Marks detected at snorkeling mark-recapture site i, fish size strata g

$m_{i,g}$ Marks released at mark-recapture site i, strata g

$c_{j,g}$ Fish detected at index site j for strata g

l_j Stream length for index site j

Site-Specific Parameters

$\theta_{i,g}$ Estimated detection probability at mark-recapture site i for fish size strata g

$\theta_{j,g}$ Simulated detection probability for index site j for strata g

λ_j Estimated density (fish/m) at index site j

Hyper-Parameters

$\mu_{\theta,g}$ Mean of beta hyper-distribution for detection probability for strata g

$\tau_{\theta,g}$ Precision of beta hyper-distribution for detection probability for strata g

μ_λ Mean of normal hyper-distribution for log fish density

τ_λ Precision of normal hyper-distribution for log fish density

Derived Variables

$\alpha_{i,g}$ Parameter for beta hyper distribution of detection probability for strata g

$\beta_{i,g}$ Parameter for beta hyper distribution of detection probability for strata g

$N_{j,g}$ Abundance at index site j for strata g

Ns Total abundance across all index sites

Nus_s Total abundance in unsampled stream length

Nt Total abundance in the Coquitlam River study area

Indices and Constants

i Index for snorkeling mark-recapture site

j Index for snorkeling index site

g Index for fish size strata

l_j Shoreline length for index site j

L Total shoreline length for the Coquitlam River study area

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

Detection Model

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

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

Population Model

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

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

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

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

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

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

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

Appendix 4.2 (continued).

Priors and Transformation

(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 2006-2011.

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 at 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 substituted for counts) for steelhead fry captured by electrofishing and counted during snorkeling in the Coquitlam River 2008-2011 (data pooled for all sites).
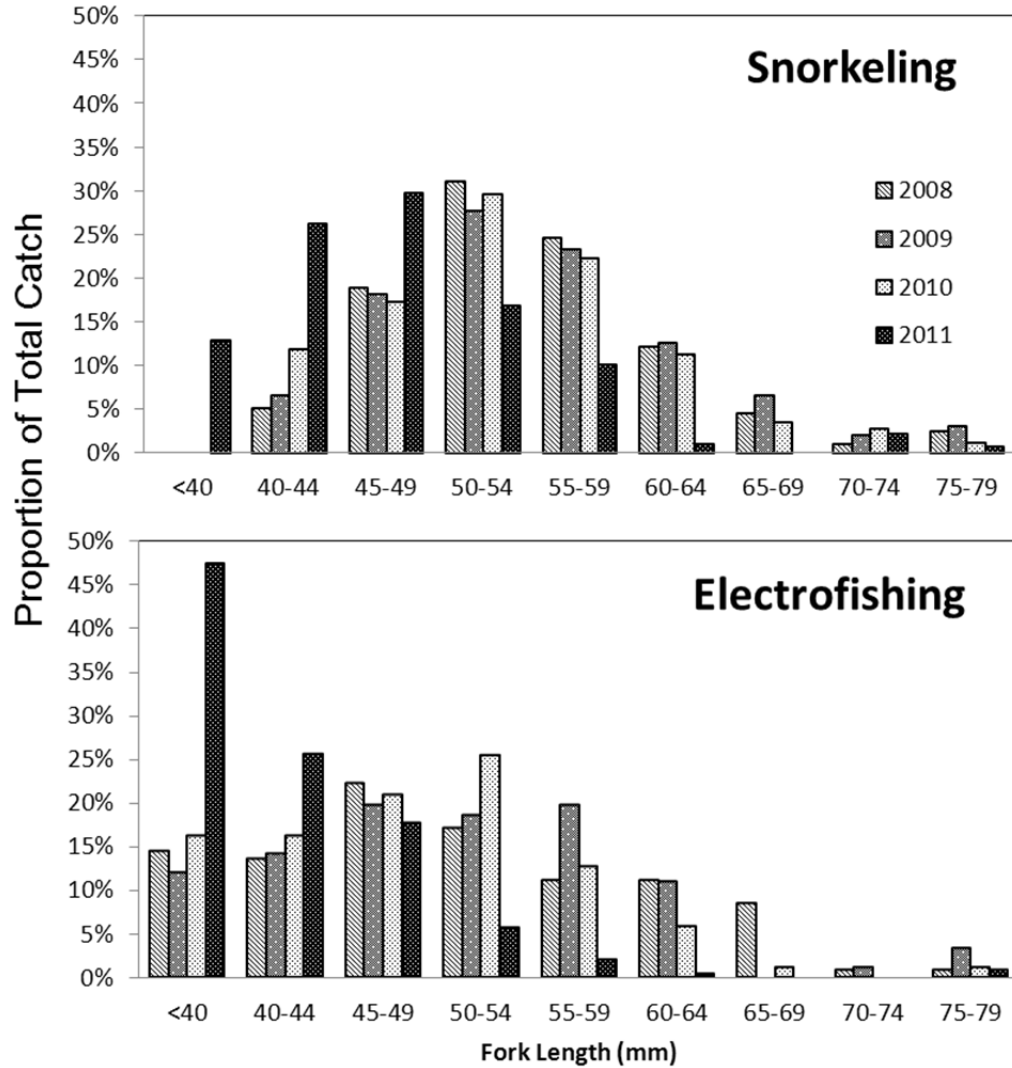
5.0 SMOLT PRODUCTION

5.1 Methods

5.1.1 Coho and steelhead smolt enumeration

In 2011, downstream migrating coho and steelhead smolts were captured at three rotary screw trap (RST) sites in the Coquitlam River mainstem (Figure 5.1), and at weirs at the outlets of three constructed off-channel sites (Figure 5.1); a minnow trapping mark-recapture study was used to estimate coho smolt abundance at a fourth off-channel site. 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 2006-2011, 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 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

---

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

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, totalling 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).

Prior to 2008, a weir was also used to enumerate smolts leaving Archery Pond, the fourth off-channel site, but this weir was not operated during 2008-2011 due to budget constraints. Archery Pond was excluded from downstream trapping because, of the four, this site has historically produced the fewest number of smolts. To approximate the contribution of Archery Pond to total off-channel smolt production during 2008, and to correct for the presence of smolts from this site in smolt captures at downstream trapping locations in the Coquitlam River mainstem, we simply used average smolt numbers for Archery Pond for the previous three years. During 2009-2011, minnow trapping and mark-recapture methods were used to obtain estimates of coho pre-smolt abundance in Archery Pond (too few steelhead were captured to generate mark-recapture estimates). During 2011, coho pre-smolts in Archery Pond were minnow trapped and marked during a series of three mark-recapture experiments conducted on February 23, February 24, and March 2. During each trapping period approximately 100 "Gee" brand wire mesh minnow traps were baited with 2 g of preserved roe and set for 24 hours. All coho were anaesthetized, measured for forklength, and marked by removing a small portion of the caudal fin. Fish were then distributed uniformly throughout the sample site.
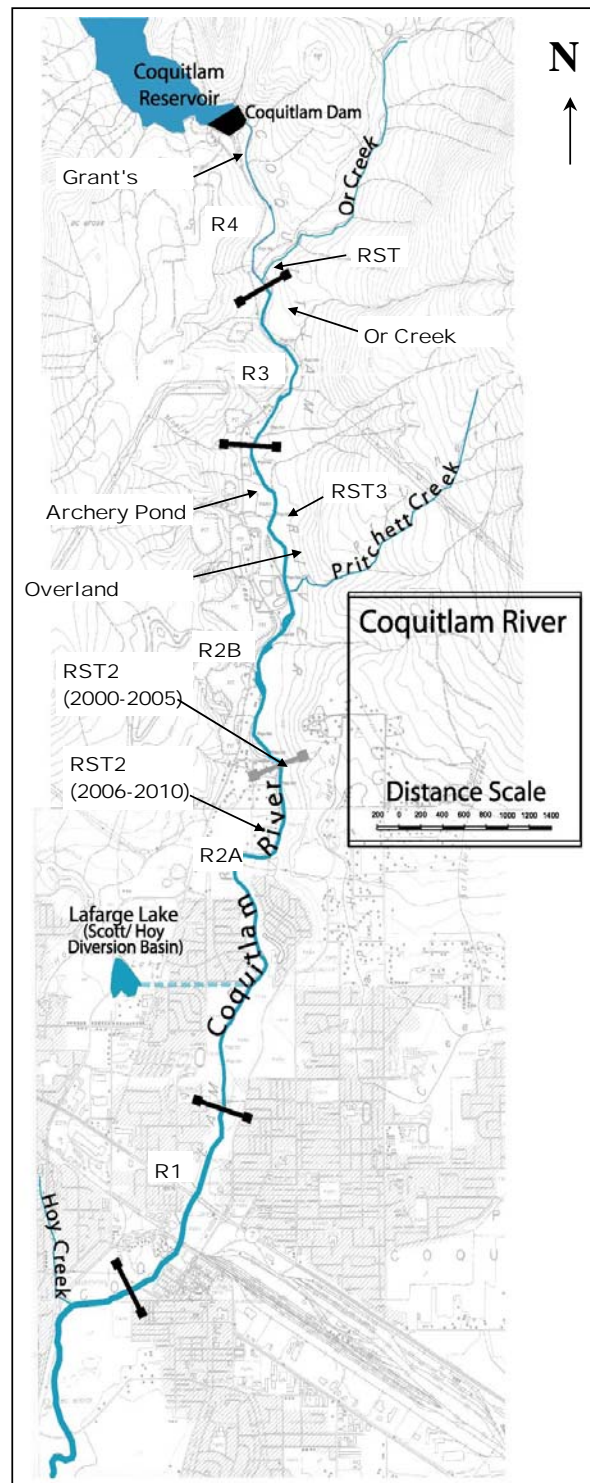
Figure 5.1 Map of the Coquitlam River showing constructed off-channel habitat sites, mainstem reach breaks and the locations of mainstem rotary screw traps (RSTs).

Table 5.1 Description of the stratification of fish marking by location and period for coho and steelhead smolts in the Coquitlam River in 2011. The start date for each temporal marking period at each RST trap site is also shown. RST installation dates correspond to the start dates for mark group 1. RST removal dates are also give

Downstream RST trapping site	Mark type by location	Mark stratification by period								Traps removed
		1	2	3	4	5	6	7	8	
Reach 2 (RST2.2, chum and pink)	mark E	3/23	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/2
Reach 2 (RST2.4, coho and steelhead)	mark E	4/7	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/15
Reach 2 (RST2.5, coho and steelhead)	mark E	3/27	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/15
Reach 3 (RST3, coho and steelhead)	mark D	3/24	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/15
Reach 4 (RST4, coho and steelhead)	mark B	-	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/15
Overland Ponds	mark A	3/23	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/22
Or Creek Ponds	mark A	-	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/20
Grants Tomb Pond	mark A	3/24	4/12	4/27	5/4	5/11	5/18	5/31	5/13	6/15

5.1.1.2 Downstream trap operation

In 2011, 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 (RST2.4 and RST2.5; Figure 5.1) were operated in close proximity to one another in reach 2 (RST2; the stream channel is considerably larger in reach 2). Screening used on all of these RSTS was 12 mm for the drum and 9mm for retention box. An additional 1.3 m diameter RST 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 late March until mid June (Table 5.1). All juvenile fish captured at the weirs and RSTs were identified to species and counted. All smolts were measured for forklength (nearest mm), and 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, 2008, and 2009, 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.

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 of initial capture (eight mark groups; Table 5.1). In addition, unmarked steelhead captured at RST2 were uniquely marked so that they could be released upstream ($\approx$ 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. Although we were not able to test this (see section 5.1.1.5), 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 capture efficiency for marked off-channel smolts.

2. Using Fisher's exact test, we tested whether overall capture efficiency (pooled data for temporal mark groups) differed ($P < 0.05$) for marked smolts from the off-channel and mainstem areas. For example, capture efficiencies (CE) for off-channel and mainstem smolts at RST2 were computed as:

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

where

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

$M_{off-channel,i}$ = number of off-channel smolts marked during marking period i
$R_{RST2,RST3,RST4,I}$ = 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 included 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 pooled data for the mainstem mark groups only.

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. To estimate coho pre-smolt abundance at the Archery Pond site, Seber's variation of Peterson's 2-stage estimator was applied to the minnow trapping data collected during the two marking events and single recapture event (Krebs 1996, p. 20):

$$N = \frac{(M+1)(C+1)}{(R+1)} - 1 \qquad\qquad (2.2)$$

$$SE\left(\frac{R}{C}\right) = \sqrt{\frac{(1-f)pq}{(C-1)} + \frac{1}{2C}} \qquad\qquad (2.3)$$

$$\text{Lower 95\% CL of N} = \left(\frac{1}{p+(1.96)(SE)}\right)(R) \qquad\qquad (2.4)$$

$$\text{Upper 95\% CL of N} = \left(\frac{1}{p-(1.96)(SE)}\right)(R)$$

where

M = number of pre-smolts marked during time 1
C = number of marked and unmarked pre-smolts caught at time 2
R = number of marked pre-smolts recovered at time 2
N = population size or pre-smolts at time 2
p = proportion of C that were marked

q = proportion of C that were not marked
f = proportion of M that were recovered at time 2

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 represented 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). We also used the Peterson estimator if either the test for "equal proportions" or "complete mixing" passed (p > 0.05) or if

the Peterson and Darroch ML estimates were similar (within the standard error of the Darroch estimate).

To estimate smolt abundance for a particular reach (N_{reach}), we computed an estimate for the RST at the downstream end of that reach, and then subtracted, from this estimate, the estimate for the next RST upstream[2]:

$$N_{reach\ 2} = N_{RST2} - N_{RST3} \qquad (5.5)$$

$$N_{reach\ 3} = N_{RST3} - N_{RST4} \qquad (5.6)$$

$$N_{reach\ 4} = N_{RST4} \qquad (5.7)$$

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

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

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:

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

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

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

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:

---

[2] In computing smolt numbers for each reach, smolt numbers for off-channel sites with weirs are not subtracted because all fish from these sites were marked and excluded.

$$\pm\, 95\%\; CI\left(N_{total}\right) = \pm\, 1.96\sqrt{Var\left(N_{RST\,2}\right)} \tag{5.12}$$

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 smolt enumeration

5.1.2.1 Downstream trapping

Only chum salmon smolts were present in the Coquitlam River during spring 2011 (fall returns of adult pink salmon to the Coquitlam River occur only in odd years). To estimate chum smolt outmigrant 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 smolts. Beginning in 2008, a 1.5 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 15 mm).

5.1.2.2 Differential marking over time

To generate temporally stratified mark-recapture estimates, single day catches of chum smolts 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 smolts 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 chum smolts 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. Smolts 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 smolts 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 smolts 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 2011, 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. No incidents of weir failure or fish leakage were apparent at the Grant's Tomb, Or Creek, or Overland Channel sites.

An aggregate total of 3,986 coho were captured at the downstream weirs as they outmigrated from the Overland, Or Creek, and Grant's Tomb off-channel sites (Table 5.2). At Archery Pond, a total of 171 coho pre-smolts were marked during the 2-day marking phase of the experiment. Six days later 63 marked pre-smolts were recaptured during the single mark-recapture event, along with 138 unmarked fish. The resulting population estimate was 313 pre-smolts with a 95% confidence interval of $\pm$ 45 pre-smolts (Table 5.2). Total steelhead smolt production for the Overland, Or Creek, and Grant's Tomb off-channel sites was 52 smolts (Table 5.2). Too few steelhead were captured to generate a population estimate for Archery Pond. Mean weighted density of coho smolts in the off-channel sites was 14.9 smolts/100 m^2, while steelhead smolt density was 0.2 smolts/100 m^2 (Table 5.2).
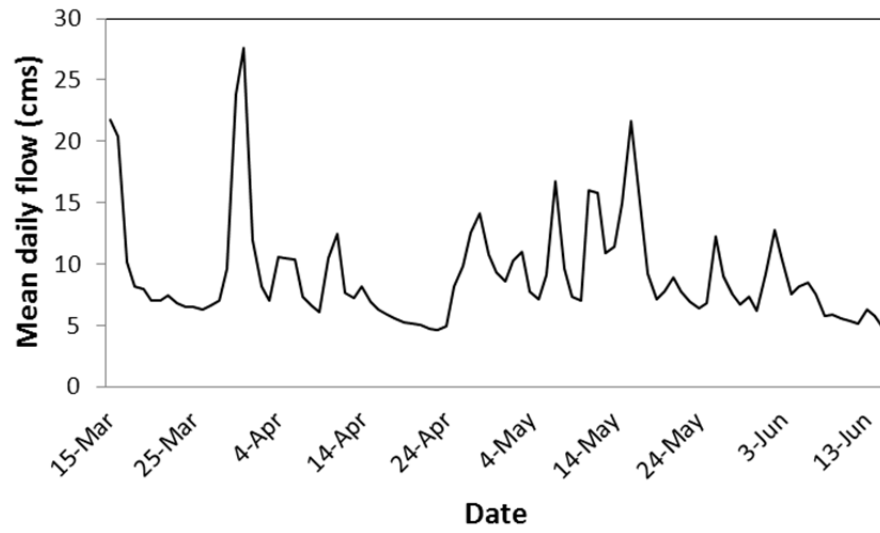
Figure 5.2 Mean daily flows in Coquitlam River at Port Coquitlam during the smolt trapping period in 2011. (Water Survey of Canada, stn. 08MH141).

Table 5.2 Summary of estimated smolt numbers and densities by species in 2011 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 (m²)	N smolts	CI (+/-)	(%)	Density (no./100m²)	(no./km)
Coho							
Off-channel sites							
Grant's Tomb	-	3,300	519	-	-	15.7	-
Or Creek	-	13,336	2,454	-	-	18.4	-
Archery Pond	-	5,800	313	45	14%	5.4	-
Overland Channel	-	4,500	700	-	-	15.6	-
Total	-	26,936	3,986	-	-	14.8	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	2,629	721	27%	3.1	822
Reach 3, Coquitlam River	2.7	46,920	3,037	474	16%	6.5	1,125
Reach 4, Coquitlam River	1.6	19,200	1,420	86	6.1%	7.4	888
Total	7.5	149,898	7,086	554	8%	4.7	945
Coquitlam R.incl. off-channel	7.5	176,834	11,072	554	5%	6.3	1,476
Steelhead							
Off-channel sites							
Grant's Tomb	-	3,300	13	-	-	0.4	-
Or Creek site	-	13,336	30	-	-	0.2	-
Archery Pond	-	-	-	-	-	-	-
Overland Channel	-	4,500	9	-	-	0.2	-
Total	-	21,136	52	-	-	0.2	-
Mainstem							
Reach 2, Coquitlam River	3.2	83,778	903	845	94%	1.1	282
Reach 3, Coquitlam River	2.7	46,920	921	697	76%	2.0	341
Reach 4, Coquitlam River	1.6	19,200	1,228	254	21%	6.4	767
Total	7.5	149,898	3,052	541	18%	2.0	407
Coquitlam R.incl. off-channel	7.5	176,834	3,104	541	17%	1.8	414
Chinook							
Off-channel sites							
Grant's Tomb	-	3,300	0	-	-	0.00	-
Or Creek site	-	13,336	0	-	-	0.00	-
Archery Pond	-	5,800	-	-	-	-	-
Overland Channel	-	4,500	0	-	-	0.00	-
Total	-	23,636	0	-	-	0.000	-
Mainstem	data too sparse to generate estimates						
Chum							
Coquitlam R.incl. off-channel	7.5	176,834	1,855,244	222,831	12%	1,049	247,366
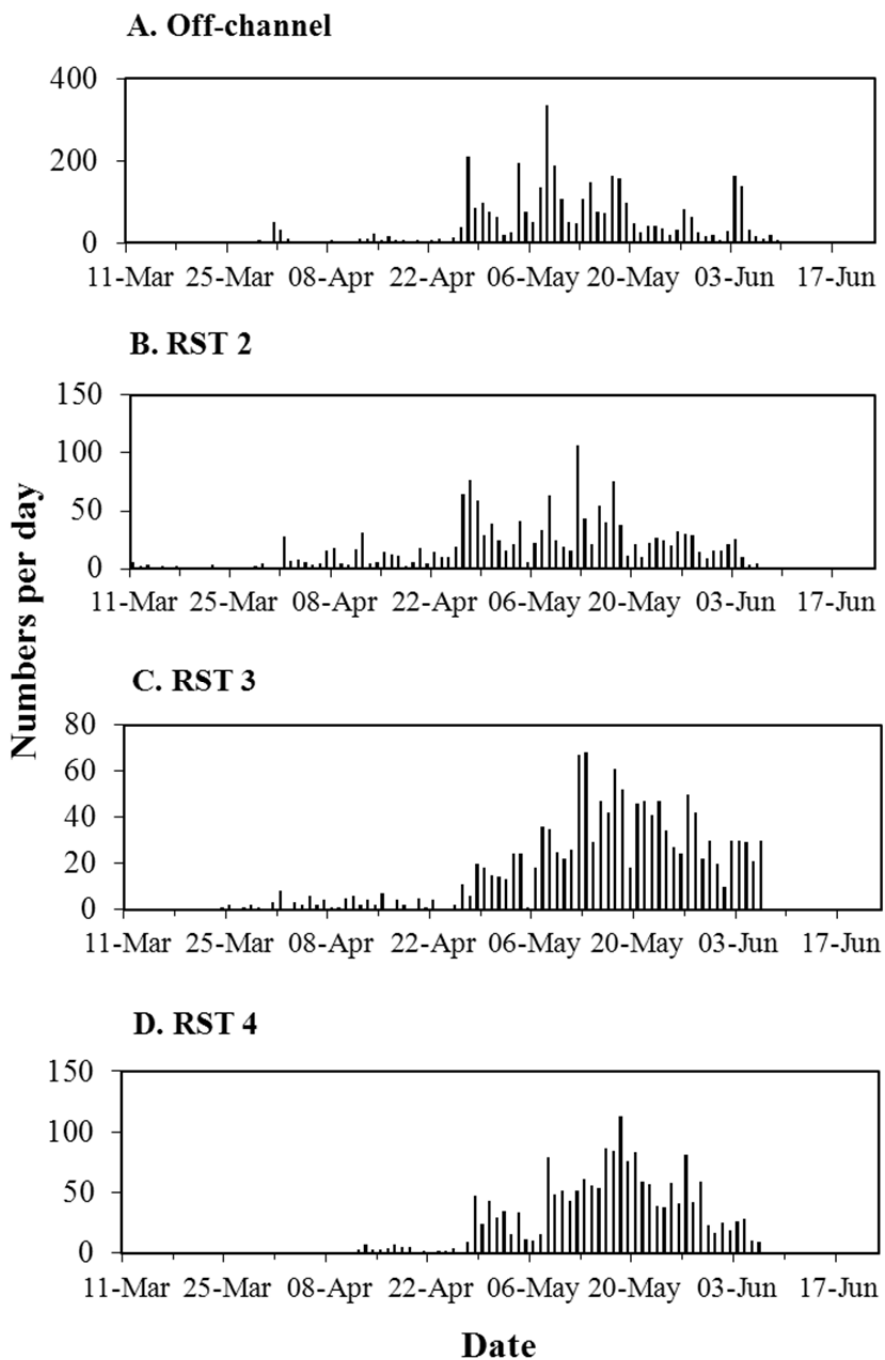
Figure 5.3 Daily catches of coho smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2011. See Table 5.1 for start and end dates for individual trapping sites.
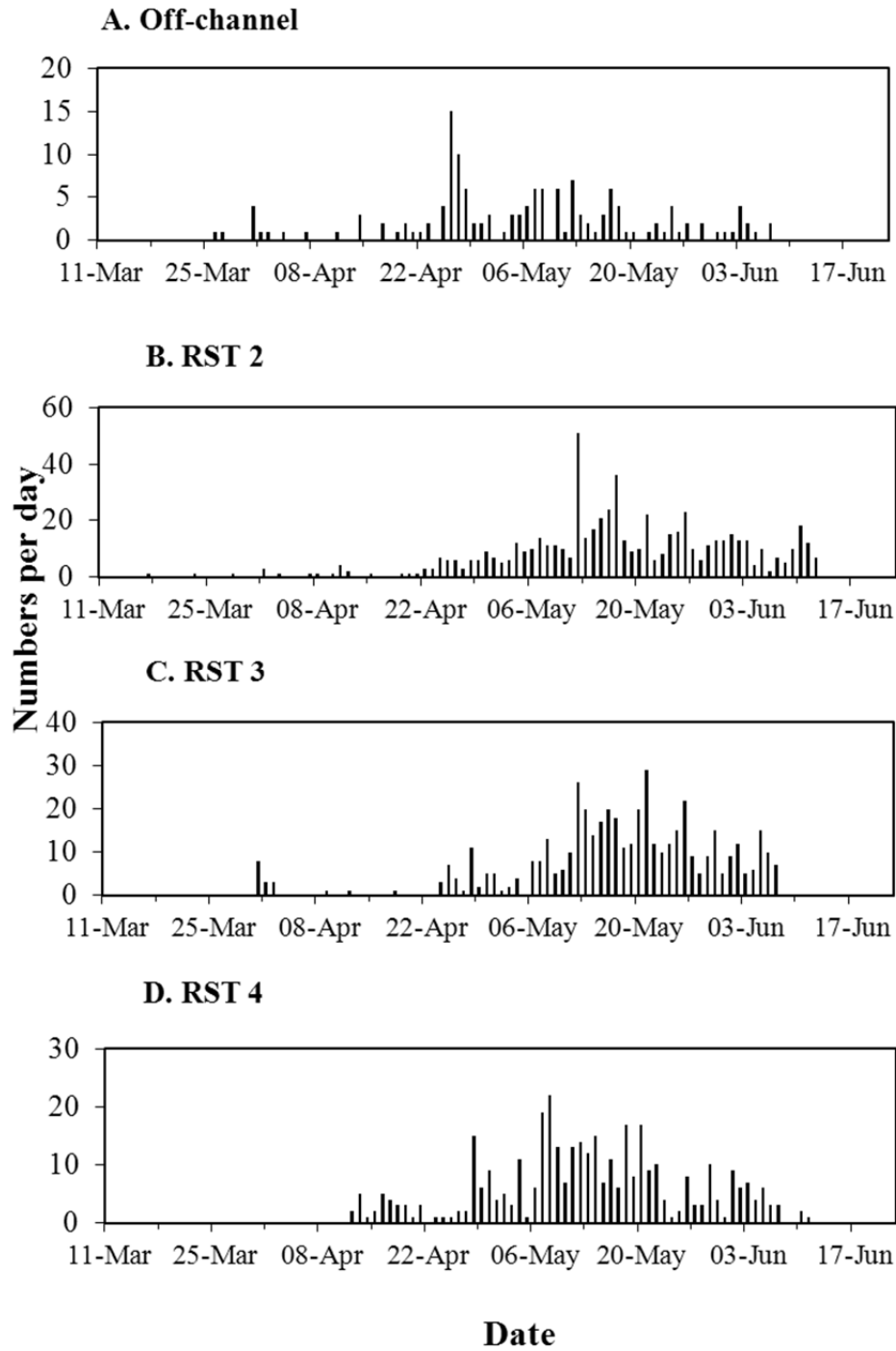
Figure 5.4 Daily catches of steelhead smolts at downstream weirs in three off-channel sites (pooled data) and at three rotary screw trapping locations in the Coquitlam River mainstem in 2011. See Table 5.1 for start and end dates for individual trapping sites.

5.2.2 Coquitlam River mainstem

During 2011, discharge in the Coquitlam River during the spring trapping period was relatively stable, with peak flows exceeding 20 cms on three occasions only (Figure 5.2), but similar to 2010, base flows were higher than in years prior to 2009 (≈8-10 cms versus 3-6 cms at the WSC Gauging Station), owing to larger releases from the dam under Treatment 2. Overall, observed mortality at the RSTs was 1.6% for coho, 1.3% for steelhead smolts, 6.0% for chum and 6.8% for Chinook smolts. 62% for Oncorhynchus nerka smolt captured (142 across all RSTs) were dead or near dead when counted. The unusually high mortality rate is likely influenced by injuries sustained during entrainment, though it is unknown what proportion can be attributed to in-trap versus pre-trap causes. For coho, steelhead, chum and pink smolts, daily catches at the beginning and end of the trapping period were 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.
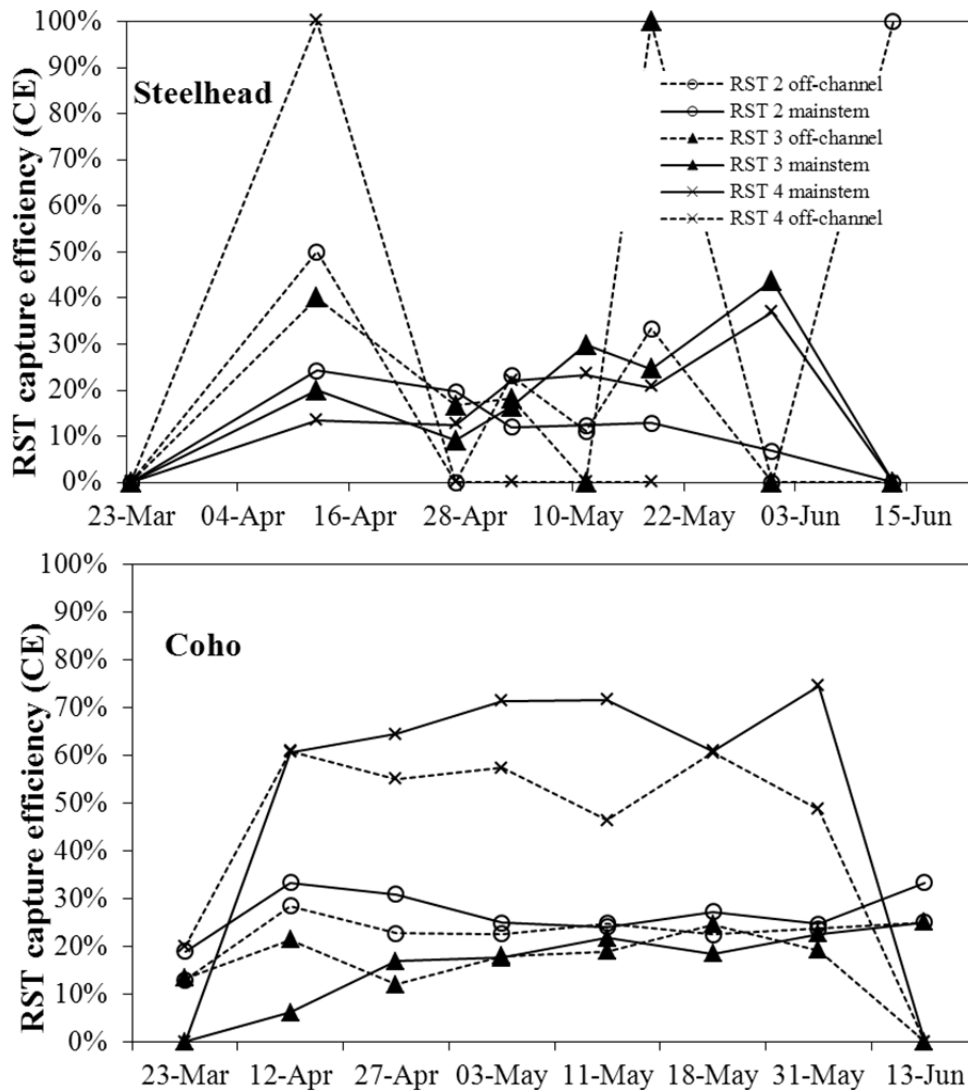
Figure 5.5 Estimated capture efficiencies (across eight marking periods) at three rotary screw traps (RSTs) in the Coquitlam River for mark groups of coho and steelhead smolts from off-channel (dotted lines) and mainstem (solid lines) habitats in 2011. Dates on the horizontal axis indicate the start point for each marking period.

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 modestly, but significantly lower for marked off-channel coho smolts from Grant's Tomb compared to that for mainstem smolts that were captured at RST4, marked and released upstream (54% and 67%, respectively, Fisher's exact test, $P<0.01$, Table 5.3, Figure 5.5). Therefore, we used only the mainstem mark group to generate a population estimate of 1,420 coho smolts (95% CI: ± 86 smolts) for reach 4 (Table 5.2).

At RST3, CE was similar for marked off-channel and mainstem coho smolts (18% and 19%, respectively, Fisher's exact test, $P=0.92$; Table 5.3, Figure 5.5).   Therefore, we combined stratified data for the two mark groups to generate a population estimate of 4,457 smolts (± 461) passing RST3 (Appendix 5.1). This resulted in a population estimate of 3,037 smolts (± 474 smolts, Table 5.2) for reach 3, after smolt numbers from reach 4, and the Grant's Tomb, Or Creek and Archery Pond off-channel sites were subtracted.

At RST2, CE was higher for the Mainstem mark group than for off-channel mark group (23% and 26%, respectively, $P = 0.01$; Table 5.3, Figure 5.5). The resultant estimate for reach 2 was 2,629 ± 721 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 combined data for the off-channel and mainstem mark groups, the estimated number of coho smolts outmigrating from the mainstem of the Coquitlam River upstream of RST2 in 2011 was 7,086 ± 554 (11,072 ± 554 smolts including those from the four off-channel sites, Table 5.2). Average coho smolt density in the Coquitlam River was 4.7 smolts/100 m^2 (6.3 smolts/100 m^2 including the off-channel sites, Table 5.2). Areal coho density was highest in reach 4 (7.4 smolts/100m^2; Table 5.2), followed by reach 3 (6.5 smolts/100m^2), and reach 2 (3.1 smolts /100 m^2). Precision ranged from ± 5% for the estimate for the entire study area including off-channel sites, to ± 27% for the smolt estimate for reach 2 (Table 5.2).

5.2.2.2 Steelhead
At RST4, the mean CE differed between marked off-channel steelhead smolts from Grant's Tomb and mainstem smolts that were captured at RST4, marked and released upstream, (8% and 21%, respectively, Fisher's exact test, $P=0.47$; Table 5.3, Figure 5.5) though this due to small sample size as only 13 off-channel smolts were marked and one recaptured. Therefore, we combined the two mark groups to generate a population estimate of 1,228 steelhead smolts (± 1,033) for reach 4 (Table 5.2).

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 2011. 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.

| Species | Recapture site | Capture efficiency | | Fisher's exact test (P) |
		Mainstem mark group	Off-channel mark group	
Coho	RST 2	0.26	0.23	0.01
Coho	RST 3	0.19	0.18	0.92
Coho	RST 4	0.67	0.54	<0.01
Steelhead	RST 2	0.13	0.17	0.29
Steelhead	RST 3	0.22	0.16	0.54
Steelhead	RST 4	0.21	0.08	0.47

At RST3, CE was again not significantly different for the off-channel and mainstem mark groups (16% and 22%, respectively, $P = 0.54$), and we combined data for the two mark groups to generate a population estimate of 2,149 smolts ($\pm$ 649) passing RST3 (Appendix 5.1). This resulted in an imprecise population estimate of 921 smolts ($\pm$ 697 smolts, Table 5.2) for reach 3, after smolt numbers from reach 4, and the Grant's Tomb, Or Creek and Archery Pond off-channel sites were subtracted.

At RST2, we combined the mark-recapture data for off-channel and mainstem smolts since CE was, again, not significantly different for the two groups (13% and 17%, respectively, $P = 0.29$; Table 5.3, Figure 5.5). The resultant estimate for reach 2 was 903, steelhead smolts which was also highly imprecise ($\pm$ 845 smolts, Table 5.2).

Based on the mainstem mark group data, the estimated number of steelhead smolts outmigrating from the Coquitlam River mainstem upstream of RST2 was 3,052 $\pm$ 541 (3,102 $\pm$ 541 smolts when off-channel sites were included, Table 5.2). Average steelhead density in the Coquitlam River mainstem was 2.0 smolts/100 m^2 (1.8 smolts/100 m^2 in the Coquitlam River including the off-channel sites, Table 5.2). Areal steelhead smolt density was several-fold higher in reach 4 (6.4 smolts/100m^2, Table 5.2) than in reaches 3 and 2 (2.0 and 1.1 smolts/100m^2, respectively). The precision of the abundance estimates ranged from $\pm$ 17% for the estimate for the entire study area including off-channel sites, to $\pm$ 86% for reach 2 (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 2005-2009 (Appendix 5.4). Scale analysis 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 207 mm. To estimate steelhead adult-to-smolt survival for the 2005-2007 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 2007-2011 smolt populations. The proportion of age-2 smolts ranged from 50%- 55% among years. From these estimates, we derived yields of 4,261, 5,225, 5,254, and 3,843 steelhead smolts for the 2005 - 2008 broods, respectively (see Table 6.1). Fish larger than 230 mm had the general appearance of resident rainbow trout (i.e., cryptic colouring, heavily spotted) as opposed to smolts (bright silver), and some were sexually mature.

5.2.2.3 Chum

During 2011, chum were trapped continuously from March 23 to June 2 at the RST2 location in reach 2. Chum were batch-marked on 10 separate occasions (Appendix 5.3). Marked and recaptured chum from the 6th mark group were excluded from the analysis due to non-standard release of marked chum. Among the remaining mark groups, capture efficiency varied from 1%-8% (Appendix 5.3), and averaged 4.5% overall (all strata pooled but excluding chum marked and recaptured in 6th stratum; Appendix 5.1). During 2011 an estimated 1.85 million chum smolts ($\pm$ 0.2 million, Appendix 5.1) migrated past the RST2 trapping site. This equates to a density of 247,366 smolts/km or 1,049 smolts/100 m^2 (Table 5.2).

5.2.2.4 Oncorhynchus nerka

In 2011, 142 *O. nerka* smolts were captured in the Coquitlam River mainstem (Figure 5.7). Of these, 107 were captured at RST4 and represent the minimum number of outmigrants assuming that all originated above the dam. Captured smolts ranged in forklength from 60-100 mm (Figure 5.6), and were likely age-1. Given the limited number of *O. nerka* captured, no attempt was made to mark fish or generate population estimates.

Figure 5.6 Length-frequency histogram for steelhead and Oncorhynchus nerka captured in the Coquitlam River in 2011 (data pooled for all trap sites).
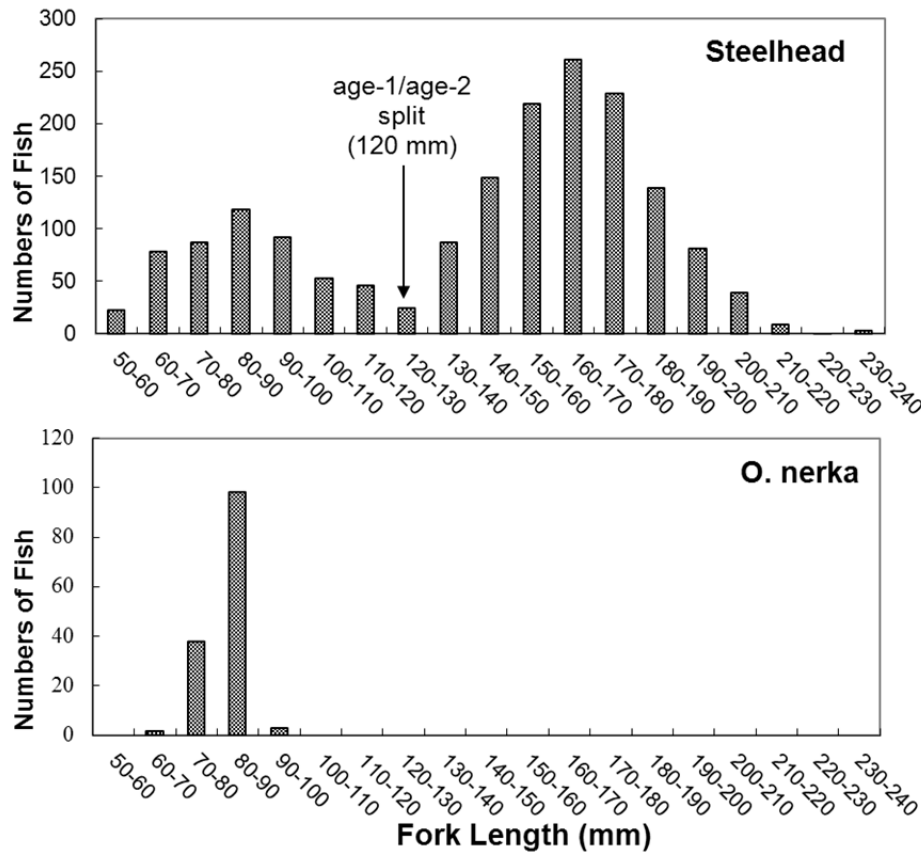
5.3 Discussion

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: we excluded from the analysis 36 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 1996-2011 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).

5.3.2 Reliability of estimates and implications for the flow experiment

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.

During 2000-2011, 95% confidence intervals for annual coho smolt abundance estimates in the Coquitlam River mainstem ranged from ±7% to ±14%, which bettered the theoretical optimal value of $\sigma_{sm,o} \approx 0.1$ (±20%). The exception was 2004 when precision was slightly less than optimal (±25% or $\sigma_{sm,o} \approx 0.13$). For steelhead, precision of the annual smolt estimates during 2000-2011 (±14% to ±37%) approached, and, in some years, bettered the theoretical optimum (The 2010 and 2011 estimates were the most precise to date). The least precise estimate for steelhead occurred in 2009 (±37%, Table 5.2) under the first year of Treatment 2. This was despite relatively high abundance, which generally contributes to better precision through greater numbers of marked and recaptured smolts.

The precision of smolt population estimates for chum salmon at the RST2 was lower in 2011 than in 2009 and 2010 (95% confidence interval: ±12% in 2011 and ±7% in both 2009 and 2010) but still higher than previous years (±18% to ±25%). That the 2011 run size was about half that of 2010 (1.7 million 2011, 3.4 million 2010) contributed to the decreased precision as there were correspondingly fewer chum marked and recaptured. Excluding smolts marked and recaptured during the 6th stratum would have lowered the precision for the same reason. This could have introduced bias into the estimate as we had to assume the capture efficiency during this stratum was the same as the adjacent strata but were unable to evaluate if this was true. However, the similar number of unmarked coho captured during these three strata combined with the similar capture efficiencies during stratum 5 and 7 suggest the bias may have been minimal. The continuation of using a RST in place of the IPT's used in earlier years and more intensive monitoring efforts contributed to the adequate estimate.
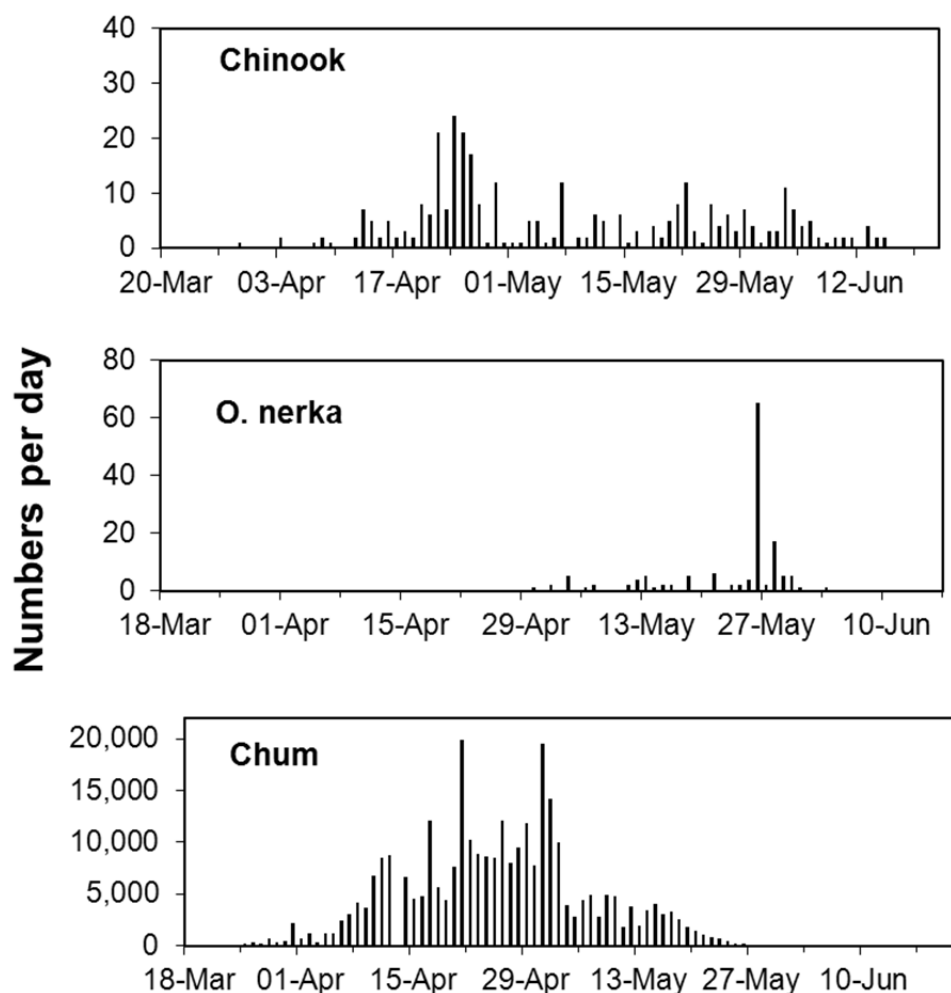
Figure 5.7 Daily catches of chum and Chinook smolts at the RST2 trapping site in reach 2, and Oncorhynchus nerka smolts at RST2, RST3 and RST4 in the Coquitlam River in 2011. See Table 5.1 for start and end dates for downstream trapping.

5.4 Recommendations for smolt trapping in 2012

For the purpose of assessing smolt production and survival rates among life stages under Treatments 1 and 2, we recommend continuing to limit the study area to the portion of the mainstem between Coquitlam Dam and RST2 to avoid bias associated with extrapolating smolt densities upstream of RST2 to unsampled reaches below. The design of the adult salmon and steelhead redd surveys are such that it is possible to obtain discrete estimates of spawner numbers upstream of RST2. 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 abundance for all species to meet CRMP objectives. Top priority should continue to be given to maximizing the number of steelhead recaptures at RST2

by maintaining a high capture efficiency at RST2 and at upstream RSTs so as many mainstem smolts are marked as possible. The length of the trapping period and the trap configurations and locations for coho and steelhead were appropriate in 2011, and a similar approach should be applied in 2012. The use of a larger (2.4 m diameter) RST in reach 4 in 2009-2011 was effective in maintaining adequately high CE during the higher flows prescribed by Treatment 2, and use of this trap should continue in future years. For chum and pink smolts, we recommend the continued use of a single 1.5 m RST (with modified small screening) at the RST2 trapping location.

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 2011. 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	1,588	418	1,565	0.26	7,086	550	7.8%
	RST 3	all	3,820	709	679	0.19	4,457	461	10.3%
	RST 4	mainstem	917	610	988	0.67	1,420	86	6.1%
Steelhead	RST 2	all	939	121	394	0.13	3,052	541	17.7%
	RST 3	all	293	62	409	0.21	2,149	649	30.2%
	RST 4	all	262	54	256	0.21	1,228	254	20.7%
Chum	RST 2	RST 2	14,529	660	81,335	0.05	1,855,244	222,831	12.0%

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 2011.

Site	Mark group	Pooling
Coho		
RST 2	mainstem (RST 3-4)	all
RST 3	all	release 1-2, 7-8, recovery 1-3, 7-8
RST 4	mainstem (RST 4)	none
Steelhead		
RST 2	all	release 1-2, 7-8, recovery 1-3, 7-8
RST 3	all	release 1-2, 7-8, recovery 1-4, 7-8
RST 4	all	all
Chum		
RST 2	RST 2	none

Appendix 5.3 Mark-recapture data for coho, steelhead, chum, and pink at three rotary screw trap sites (RST2, RST3, RST4) in the Coquitlam River mainstem in 2011. 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									Species: coho
Mainstem mark groups				Recovery strata					
Release strata	Marks	1	2	3	4	5	6	7	8	Capture efficiency
1	37	2	1	2	0	0	2	0	0	19%
2	66	0	7	7	0	6	2	0	0	33%
3	165	0	0	20	19	8	2	2	0	31%
4	237	0	0	0	19	36	4	0	0	25%
5	387	0	0	0	0	42	50	1	0	24%
6	552	0	0	0	0	0	113	37	0	27%
7	138	0	0	0	0	0	0	31	3	25%
8	6	0	0	0	0	0	0	0	2	33%
Untagged Fish		106	213	257	205	356	285	136	7	

Recovery site: RST 3									Species: coho
All mark groups				Recovery strata					
Release strata	Marks	1	2	3	4	5	6	7	8	Capture efficiency
1	15	0	0	1	0	0	0	1	0	13%
2	141	0	8	12	0	2	3	0	0	18%
3	578	0	0	32	25	10	5	3	0	13%
4	1111	0	0	0	50	111	24	13	1	18%
5	746	0	0	0	0	54	86	7	0	20%
6	863	0	0	0	0	0	148	40	1	22%
7	371	0	0	0	0	0	0	68	5	20%
8	10	0	0	0	0	0	0	0	1	10%
Untagged Fish		44	34	61	84	159	210	84	3	

Recovery site: RST 4									Species: coho
Mainstem mark group				Recovery strata					
Release strata	Marks	1	2	3	4	5	6	7	8	Capture efficiency
1	15	0	1	0	1	0	0	1	0	20%
2	51	0	11	9	4	2	5	0	0	61%
3	178	0	0	70	22	8	7	2	0	61%
4	316	0	0	0	74	99	27	3	0	64%
5	322	0	0	0	0	97	103	3	0	63%
6	445	0	0	0	0	0	239	31	1	61%
7	96	0	0	0	0	0	0	59	3	65%
8	2	0	0	0	0	0	0	0	0	0%
Untagged Fish		0	35	122	146	227	384	68	6	

Appendix 5.3 continued

Steelhead

Recovery site: RST 2   **Species: steelhead**
All mark groups   Recovery strata

Release strata	Marks	1	2	3	4	5	6	7	8	Capture efficiency
1	29	0	0	0	0	0	0	0	0	0%
2	39	0	2	3	3	1	2	0	0	28%
3	88	0	0	3	8	3	1	0	0	17%
4	172	0	0	0	2	17	3	0	0	13%
5	252	0	0	0	0	11	20	0	0	12%
6	267	0	0	0	0	0	23	12	0	13%
7	88	0	0	0	0	0	0	5	1	7%
8	4	0	0	0	0	0	0	0	1	25%
Untagged Fish		7	12	26	54	101	72	47	2	

Recovery site: RST 3   **Species: steelhead**
All mark groups   Recovery strata

Release strata	Marks	1	2	3	4	5	6	7	8	Capture efficiency
1	8	0	0	0	0	0	0	0	0	0%
2	20	0	0	0	1	2	2	0	0	25%
3	45	0	0	2	2	1	0	0	0	11%
4	84	0	0	0	4	9	0	1	0	17%
5	52	0	0	0	0	8	6	0	0	27%
6	66	0	0	0	0	0	14	3	0	26%
7	16	0	0	0	0	0	0	7	0	44%
8	2	0	0	0	0	0	0	0	0	0%
Untagged Fish		14	10	20	33	93	139	43	2	

Recovery site: RST 4   **Species: steelhead**
All mark groups   Recovery strata

Release strata	Marks	1	2	3	4	5	6	7	8	Capture efficiency
1	8	0	0	0	0	0	0	0	0	0%
2	16	0	1	0	1	1	0	0	0	19%
3	33	0	0	1	3	0	0	0	0	12%
4	74	0	0	0	4	10	2	0	0	22%
5	48	0	0	0	0	6	5	0	0	23%
6	63	0	0	0	0	0	11	2	0	21%
7	19	0	0	0	0	0	0	7	0	37%
8	1	0	0	0	0	0	0	0	0	0%
Untagged Fish		0	15	33	64	57	66	20	1	

Appendix 5.3 continued

Recovery site: RST 2									Species: chum			
All mark groups					Recovery strata							
Release strata	Marks	1	2	3	4	5	6	7	8	9	10	Capture efficiency
1	642	16	0	0	0	0	0	0	0	0	0	2.5%
2	1658	0	16	0	0	0	0	0	0	0	0	1.0%
3	2122	0	0	114	0	0	0	0	0	0	0	5.4%
4	1406	0	0	0	57	0	0	0	0	0	0	4.1%
5	1978	0	0	0	0	156	0	0	0	0	0	7.9%
6	2020	0	0	0	0	0	410	0	0	0	0	20.3%
7	1994	0	0	0	0	0	0	165	0	0	0	8.3%
8	1872	0	0	0	0	0	0	0	72	0	0	3.8%
9	1502	0	0	0	0	0	0	0	0	46	0	3.1%
10	1355	0	0	0	0	0	0	0	0	0	28	2.1%
Untagged Fish		5139	3348	10479	8416	15309	12071	15670	4732	3732	2439	

Appendix 5.4 Age-forklength relationships for steelhead parr and smolts in the Coquitlam River during 2005-2009 derived from scale-aging analysis.
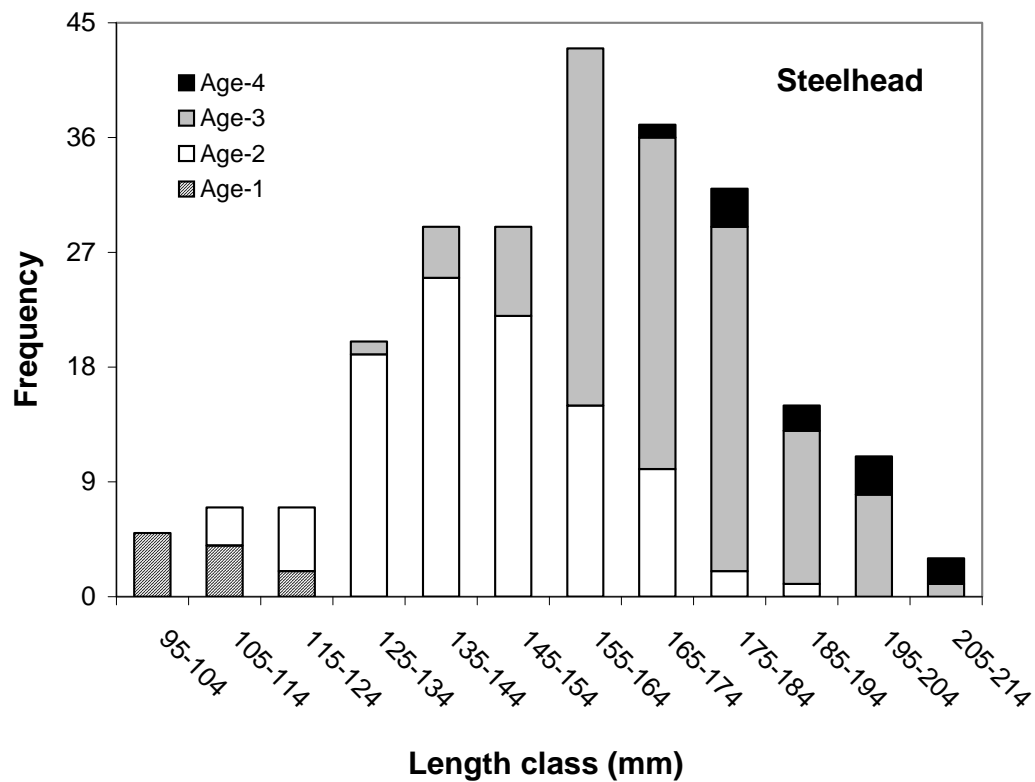
6.0 FISH PRODUCTIVITY DURING TREATMENTS 1 AND 2

6.1 Coho

During 2000-2011 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 no consistent trend over time (mean = 6,990 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: 13,892 smolts, range: 8,400-24,500; Table 6.1a). 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. Smolt yields in Coquitlam River in 2010 and 2011 were comparable to that in years prior to 2009. In reach 4, where annual downstream trapping has occurred over a longer time period (1997-present), smolt yields since 2004 (omitting 2009) have been substantially lower than the previous period (Figure 6.3). While there is no obvious differences in coho productivity between Treatments 1 and 2 thus far, late summer snorkeling surveys suggest that densities of coho fry in reaches 2 and 3 were several-fold higher during 2006-2011 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).
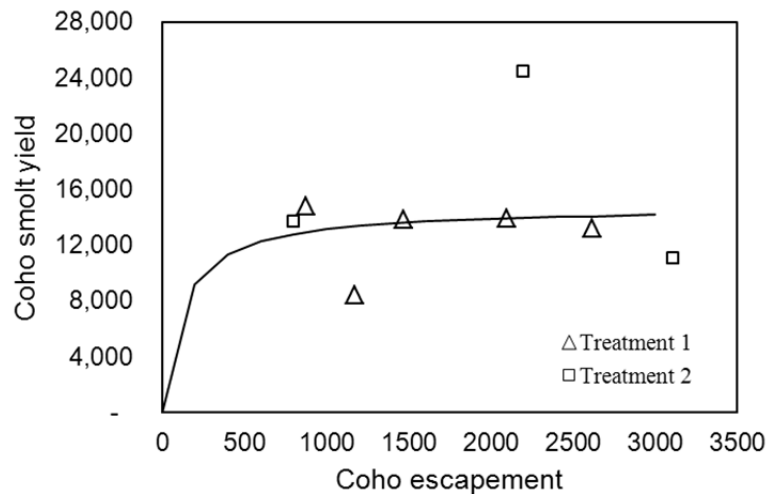

Figure 6.1 Beverton-Holt stock-recruitment relationship between coho escapement (upstream of RST2) and total smolt yield in the Coquitlam River during Treatment 1 (2004-2008 smolt years), and during the first 3 years of Treatment 2 (2009-2011 smolt years).

l population estimates for all life stages and species in Coquitlam River, 2000-2011.   Values shown for the different life stages for a
nd 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.
nt life stage are also not strictly comparable because study areas differ somewhat for the different components of the monitoring

	Treatment 1								Treatment 2		
es	2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
	-	15,378	18,301	27,992	24,559	51,860	11,066	18,224	19,600	6,931	27,410
	-	-	5,418	-	4,406	-	2,876	-	10,698	-	10,427
	-	2,648	1,562	2,562	1,334	939	2,401	878	3,175	12,338	3,835
ok	-	-	<300	<100	<100	<100	0	952	1,529	8,018	4,918
head (female)	-	-	-	-	187	434	130	148	113	167	206
head (total)	-	-	-	-	373	868	260	297	225	333	412
	-	-	-	-	-	21,520	21,225	48,775	56,870	63,675	105,200
eelhead fry	-	-	-	-	-	142,485	27,547	41,715	38,070	39,170	40,921
eelhead parr	-	-	-	-	-	6,231	10,510	10,690	10,870	7,647	9,034
eelhead parr	-	-	-	-	-	1,641	1,820	1,153	3,232	2,617	3,583
(total)	-	-	1,318,883	1,056,218	835,815	3,367,177	3,099,569	1,013,014	4,219,597	3,396,678	1,855,244
total)	-	-		321,700	-	147,938	-	181,403	-	545,067	-
(mainstem)	11,036	4,838	8,195	4,234	3,215	5,979	2,870	9,020	13,844	6,573	7,086
(total)	16,384	9,307	13,849	13,163	13,819	13,891	8,387	14,790	24,457	13,691	11,072
head (total)	4,191	2,308	3,885	3,842	3,966	4,277	2,668	5,644	5,398	4,874	3,104
head (2+)	-	-	-	-	-	-	1,412	2,795	2,968	2,588	1,848
head (3+)	-	-	-	-	-	-	-	2,849	2,430	2,286	1,256

Table 6.1b Summary of survival estimates across all life stages and species for 2000-2010 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	Treatment 1								Treatment 2		
		2000	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
coho	Egg-to-fall fry [1]	-	-	-	-	0.5%	0.8%	0.7%	2.2%	0.7%	0.3%	-
coho	Egg-to-smolt [1]	-	0.3%	0.6%	0.4%	0.5%	1.1%	0.7%	1.1%	0.2%	-	-
steelhead	Egg-to-fall fry [1]	-	-	-	-	-	8.9%	5.7%	7.6%	9.1%	6.3%	11.0%
steelhead	Egg-to-parr [1]	-	-	-	-	0.9%	0.7%	2.2%	2.0%	1.8%	1.5%	-
steelhead	Egg-to-smolt [1,2]	-	-	-	-	0.7%	0.4%	1.1%	0.8%	-	-	-
steelhead	Fry to age-1+ parr	-	-	-	-	-	7.4%	38.8%	26.1%	20.1%	23.1%	-
steelhead	Fry to age-2+ parr	-	-	-	-	-	0.8%	11.7%	6.3%	9.4%	-	-
steelhead	Age 1+ parr to smolt [2]	-	-	-	-	-	68.6%	40.4%	44.1%	-	-	-
steelhead	Age 2+ parr to smolt [2]	-	-	-	-	-	76.6%	156.6%	210.7%	70.7%	48.0%	-
chum	Egg-to-smolt [1]	-	7.9%	9.5%	3.8%	19.0%	7.2%	13.5%	26.8%	18.1%	26.1%	-
pink	Egg-to-smolt [1]	-	-	9.6%	-	5.1%	-	9.7%	-	7.4%	-	-

[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).
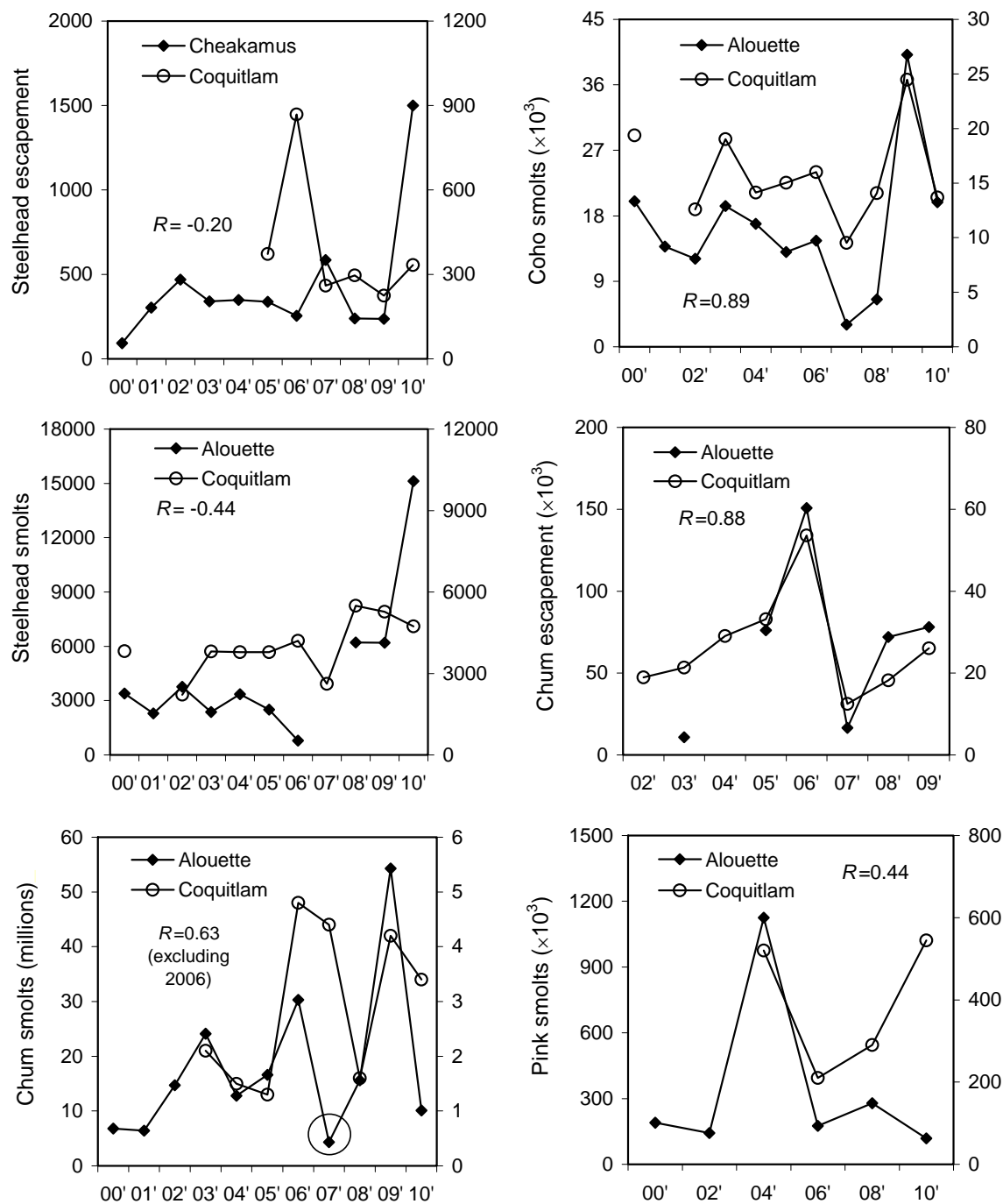
Figure 6.2 Scatterplots of escapement and smolt yield in the Coquitlam River versus that in the Cheakamus and Alouette rivers during 2000-2010. Values for the Coquitlam River are given on the right-hand axis, and values for other streams are given on the left-hand axis.
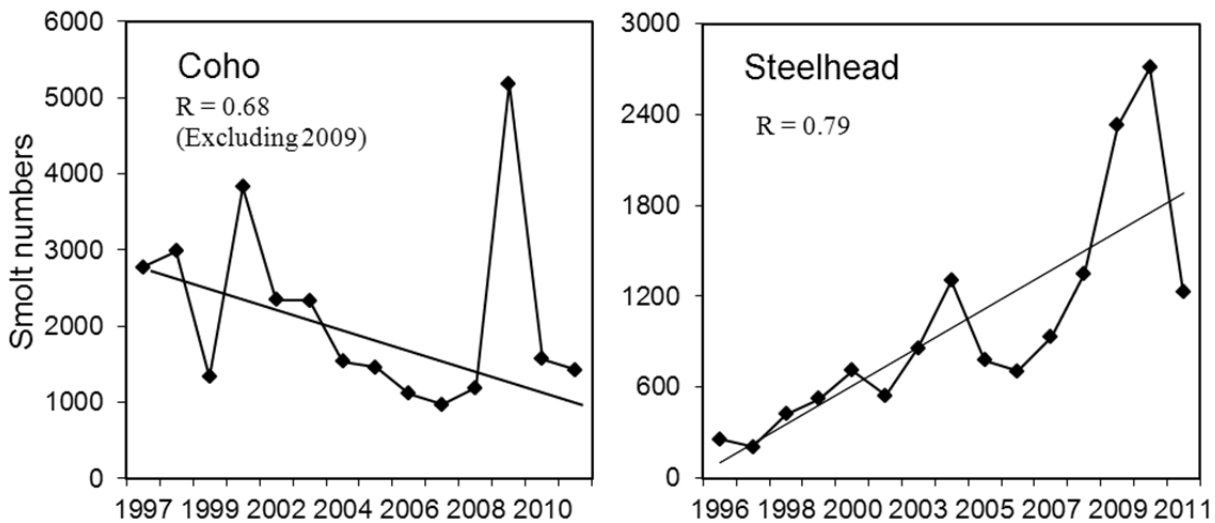
Figure 6.3 Annual numbers of coho and steelhead smolts in reach 4 of Coquitlam River during 1997-2011.

The constructed off-channel habitats included in the study[3], 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 2000-2011. 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 the off-channel sites (19.9 to 44.9 smolts/100m^2). While constructed off-channel habitat may represent relatively productive coho habitat in the Coquitlam River, smolt densities in Coquitlam River off-channel sites were below average densities reported for constructed side-channels and ponds in other Pacific Northwest streams (67 and 69 smolts/100m^2, respectively; Koning and Keeley 1997).

Although the accuracy of estimates of coho escapement is highly uncertain, owing to a lack of observer efficiency and survey life data (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 2002-2011 ranged from 70 to 960 fish/km (Table 2.4), or 34 to 482 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).

[3] 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.
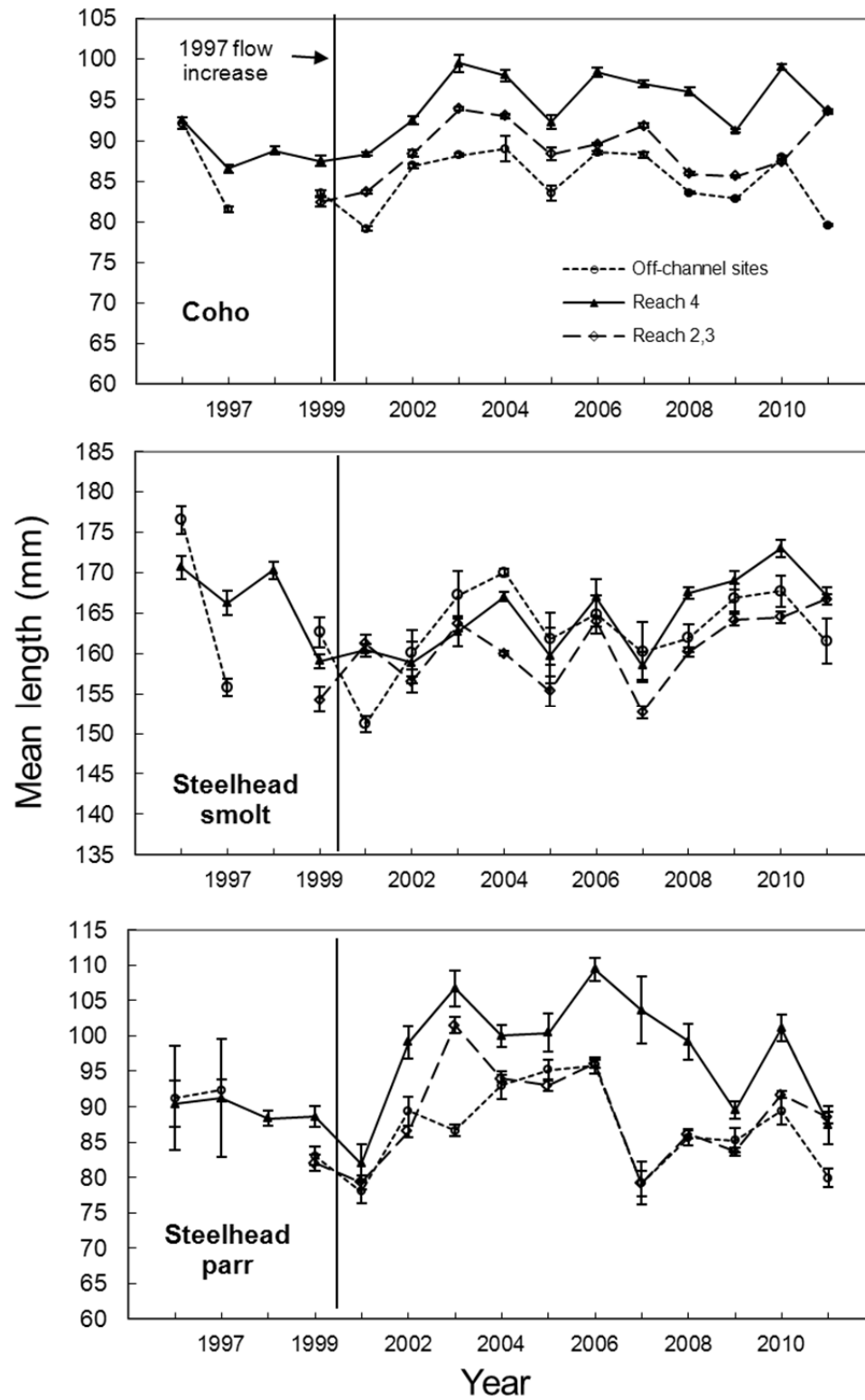
Figure 6.4 Mean annual forklengths for coho smolts and steelhead smolts and parr in different habitats in the Coquitlam River, 1996-2011. Error bars represent ± 1 standard error.

Mean size of coho smolts in reach 4 was slightly greater during 1999-2011 (mean = 94 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 1999-2011, 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 2000-2011 areal densities of coho smolts was 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 1,852 smolts/km (range: 1,118-3,261 smolts/km, 2000-2011). This suggests that coho smolt productivity in the Coquitlam River study area is comparable to the average for streams at this latitude.

During Treatment 1, coho smolt production in the Coquitlam River upstream of RST2 was relatively stable (8,400-14,700 smolts; Figure 6.1), despite three-fold variation in spawner abundance (878-2,900 spawners), which is expected if spawner abundance exceeds that required for full seeding. Moreover, annual coho smolt yield in the Coquitlam River during Treatment 1 was strongly and positively correlated (R = 0.80, n = 11, p <0.01; Figure 6.2) with that in the dam-regulated Alouette River, despite the fact that brood escapements for the two streams were not correlated (R = -0.09, n=5), which also suggests that variation in smolt production in the Coquitlam River during the period of study was governed more by freshwater rearing conditions than by egg deposition.

Coho egg-to-smolt survival remained consistently low (0.2-1.1%) for the 2002-2009 brood years (Table 6.1b), with the highest values associated with the lowest escapements. 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. 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 2000-2011 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 4,014 smolts (Table 6.1a). There is no clear trend of increasing smolt yield over time though the 3 highest estimates occurred since 2008 (Figure 6.2). Mean smolt yields were not statistically different between Treatment 1 and 2 (3,716 smolts and 4,354 smolts

respectively, p = 0.42). Smolt yield from reach 4 has increased substantially (R=0.78, *n* = 15, *p*<0.01; 1996-2011; Figure 6.3) though with considerable interannual variation. 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.

There was no significant difference in the size of spring migrant steelhead parr (age-1) in reach 4 during Treatments 1 and 2 (1999-2011 mean = 98 mm; Figure 6.4) in comparison to earlier years (1996-1998; mean = 90 mm; *t-test*, *p* = 0.08), 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). There was no trend in the mean size of steelhead smolts among years or between treatment periods (Figure 6.4).

In contrast to that for coho, there was no correlation between annual steelhead smolt yield in the Coquitlam River and that in the Alouette River (*R* = -0.39, *n* = 8, *p* = 0.30; 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 2000-2011, steelhead smolt densities for the mainstem study area upstream of RST2 as a whole averaged 2.6 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 were less pronounced. 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 2006-2011, steelhead fry density in the Coquitlam River in late summer averaged 26.7 fish/100m^2, while parr density averaged 6.2 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 2006-2011 that derived from both electrofishing and snorkeling surveys suggest several-fold higher densities of fry and par (see Section 4.3.2).

During 2005-2011, steelhead spawner densities in the Coquitlam River ranged from 24 to 80 fish/km (mean = 37 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 2002-2011 (mean = 35 fish/km), but were not correlated with Coquitlam River escapements (R = -0.16, *n* = 7, *p* = 0.73; 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 mailout 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.
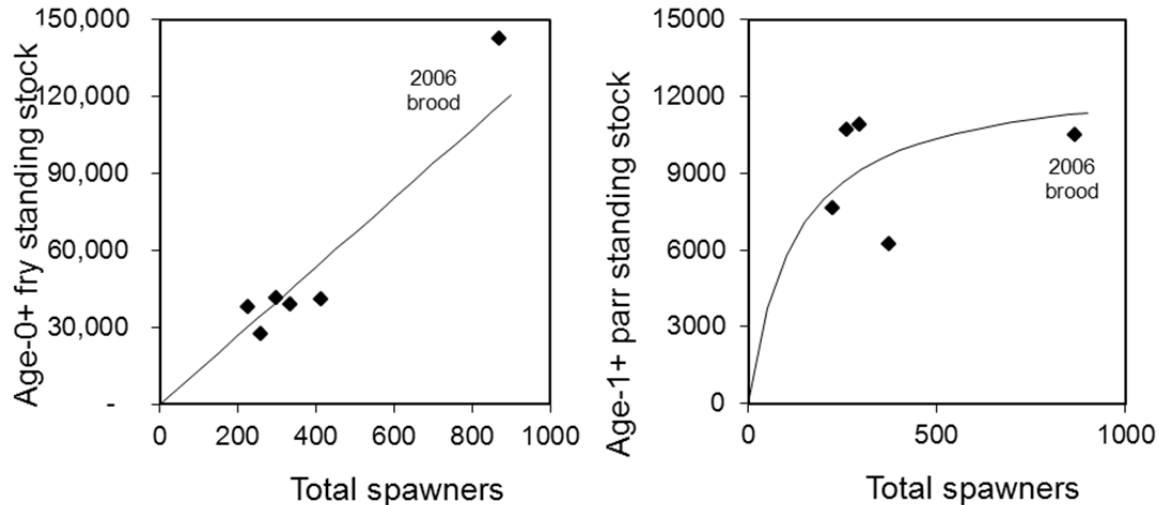

Figure 6.5 Preliminary stock-recruitment relationship for late summer juvenile steelhead standing stocks (2005-2011) versus brood escapements in the Coquitlam River (data points corresponding to peak escapement in 2006 are shown.

Estimated steelhead egg deposition in the Coquitlam River during 2005-2011 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 ($\approx$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 2005-2011, 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 were very consistent during 2006-2011, ranging from 6.7% to 9.1% (Table 6.1b). 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%-2.5% (Table 6.1b), 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% during 2005-2008 (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 2006-2010 fry cohorts ranged from 10%-37% (Table 6.1b).   Age-1+ parr-to-smolt survival ranged from 35%-69% for the 2006-2009 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 ($\approx$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). A limited number of scale samples (250) have been collected to date. As more scale samples are collected in future years, size-at-age and relative proportions of age-2 and age-3 smolts will be reanalyzed.

6.3 Chum and pink

Similar to that for coho, escapement and egg-to-smolt survival estimates for chum and pink 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 12,000-54,000 (Table 6.1a), while smolt production upstream of RST2 has ranged from 0.8 to 4.2 million. Maximum chum production occurred in 2009, the same year in which coho and steelhead production peaked (see Section 6.2 above), and was likely related to favorable region wide conditions. During Treatment 1 (2002-2007 brood years) chum egg-to-smolt survival ranged from 3.7% to 14.1% (mean = 8.6%; Table 6.1b); thus far during Treatment 2, egg-to-smolt survivals have been relatively high (26.8%, 13.6% and 26.1% for the 2008 - 2010 brood years, respectively). Bradford (1995) reported an average egg-to-smolt survival rate of 6.7% ($\pm$1 standard deviation = 3.3%-13.5%) for chum populations in nine other streams. The chum egg-to-smolt survival estimates for the 2008 and 2010 brood years in the Coquitlam River exceeds published values for this species, and are likely 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 and 2011 were relatively precise (95% CI: $\pm$7 and 12%, respectively), with no evidence of serious violations of mark-recapture assumptions.

During 2003-2011, chum smolt yield in the Coquitlam River was not well correlated with escapement (R = 0.35, n = 9, p = 0.35), suggesting that factors other than variation in escapement played a substantial role in determining smolt yield (i.e., spawning habitat was fully utilized and egg-to-smolt-survival was strongly density-dependent). As well, during 2002-2010, both chum escapement and smolt yield in the Coquitlam River were positively correlated[4] with that in the Alouette River (escapement: R = 0.88, n = 6, p = 0.02; smolt yield: R = 0.63, n = 7, p = 0.03;

---

[4] note that the correlation for chum smolt yield was computed with 2006 excluded because of anomalously low egg-to-smolt survival in the Alouette River in that year (see Cope 2011).

Figure 6.2; Cope 2011), suggesting that regional or climatic factors play a significant role in both freshwater and marine productivity for Coquitlam River chum.

   Estimated adult pink salmon returns to Coquitlam River ranged from 2,900-10,600 adults, with peak escapement occurring in 2009 (Table 6.1a). Smolt production upstream of RST2 ranged from 148,000-545,000 (Table 6.1a), with peak numbers in 2010 produced from the peak escapement the previous year. Egg-to-smolt survival for pink ranged from 4.9%-9.9%. These values were comparable to the range reported for pink populations in 18 other streams (mean: 7.4%; ±1 standard deviation: 3.2%-17.0%; Bradford 1995). During 2004-2010, pink smolt yield in the Coquitlam River was positively correlated with escapement (R = 0.94, n = 4, p = 0.06) suggesting that the river did not reach the point of density dependant interactions during those years, although with only four years' data this is still preliminary. Unlike chum, there was no obvious correlation between pink smolt yield in the Coquitlam River and that in the Alouette River (R = 0.44, n = 4, p = 0.58; Figure 6.2).

   Overall, chum and pink salmon returns to Coquitlam River were markedly improved in 2002-2010 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). 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 chum and pink. There is some indication that larger dam releases under Treatment 2 have further improved access to spawning habitats for pink salmon (Macnair 2010b).
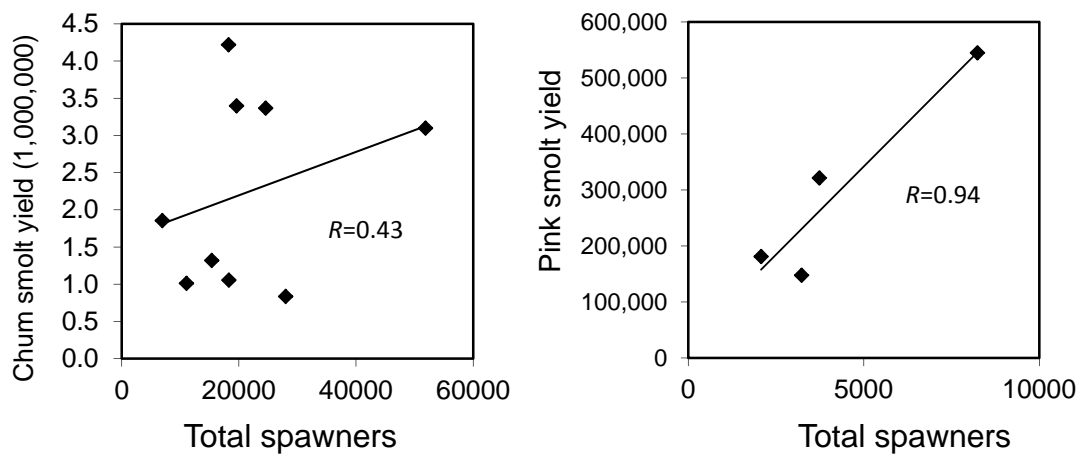

Figure 6.6 Preliminary stock-recruitment relationship for outmigrating chum and pink smolts (2002-2011) versus brood escapements in the Coquitlam River..

6.4 Comparison of fish productivity 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, 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. Other metrics of stream productivity such as smolts per spawner or egg-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. This supports the notion that smolt abundance us the best metric for comparing flow treatments. Moreover, interannual variation in coho and steelhead smolt abundance was relatively low during Treatment 1 (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 this 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 smolts per recruit or egg-to-smolt survival as the performance measure for comparing treatments. For Treatment 1, there will be six datapoints for chum (2002-2007 brood years, Table 6.1b), but only three for pink (2003, 2005, and 2007 brood years).

The strong correlations between the Coquitlam and Alouette rivers for chum spawner and smolt abundance, and coho smolt abundance (Figure 6.2), suggests the possibility of using the Alouette River as a control for the flow experiment in the Coquitlam River. The two streams 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. 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). Moreover, the Alouette River smolt monitoring program is scheduled to continue until the end of 2014, with no changes to the current flow regime (D. Hunter, BC Hydro, pers. comm.).

The inclusion of the Alouette River 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. We recommend that the COQWUPCC consider including the Alouette River in the study as an experimental control.

7.0 REFERENCES

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Cope, S. 2011. Alouette River salmonid smolt migration enumeration: 2010 data report. Unpublished report prepared for the Alouette River Management Committee and BC Hydro by Westslope Fisheries, Cranbrook BC, March 2011. 63 p.

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

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

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

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

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

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

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

Decker, A.S., and J. Hagen. 2007. The distribution and abundance of juvenile Chinook salmon in the lower Thompson River basin in relation to spawner abundance and habitat

characteristics. Unpublished report prepared for the Pacific Salmon Commission, June 2007. 88 pages.

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

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

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

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

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

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

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

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

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

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

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

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

Hagen, J., A. S. Decker, J. Korman, and R. Bison. 2011. Effectiveness of night snorkeling for estimating steelhead parr abundance in a large river basin. North American Journal of Fisheries Management 30:1280-1302.

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

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

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

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

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

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

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

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

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

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

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

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

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

Leake, A.C. 2005. Coquitlam-Buntzen Water Use Plan Terms of Reference: 7COQMON#7 - Lower Coquitlam River Fish Productivity Index. BC Hydro, Water Use Plan Monitoring Report, November 28, 2005.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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