

A COMPARATIVE STUDY OF THE DEMOGRAPHIC TRAITS AND EXPLOITATION  
PATTERNS OF COHO SALMON STOCKS FROM  
S.E. VANCOUVER ISLAND, B.C.

by

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## ABSTRACT

An investigation was initiated in southern British Columbia during 1984 to determine the degree of similarity between populations of coho salmon, in terms of their demographic traits and exploitation patterns. Fourteen stocks of wild or hatchery origin were subject to coded-wire tagging in nine different streams located within a 150 km region of southeast Vancouver Island during 1985, 1986, and 1987. Escapement enumeration and tag recovery were conducted during the 1985-1988 period in all streams by means of counting fences and stream surveys. A new mark-recapture model was formulated specifically for estimating escapement levels in natural streams where only a partial enumeration of spawners is possible. Estimates of the number of tags recovered in various sport and commercial fisheries were generated primarily on the basis of catch and sampling records extracted from the Mark-Recovery Program database, located at the Pacific Biological Station of the Canadian Department of Fisheries and Oceans. Estimates of sampling rates in the Strait of Georgia sport fishery were determined from simulation studies based on catch and sampling statistics associated with the Georgia Strait Creel Survey and Head Recovery programs.

Populations were contrasted in terms of their juvenile migration patterns, smolt-to-adult survival, catch distributions, straying rates, escapement patterns, run composition, adult sizes, and exploitation rates. Considerable variation in smolt size and juvenile migration time was detected among populations each year. The most pronounced difference was the consistently shorter migration period of smolts released from public hatcheries. Estimates of ocean survival was highly variable across years and streams, and even among stocks within the same stream. No particular stock or stock type had consistently higher survival, but hatchery fish from the Big Qualicum River exhibited consistently lower survival. Considerable variation was observed in the duration and timing of various runs within the study area. On average, the contribution of strays to each spawning population was relatively small (< 2%), but strays could account for as much as 50% of the escapement to a given stream. Average exploitation rates were in the neighborhood of 80% each year, but were as high as 96% for some stocks. Substantial differences in exploitation rate were detected between stocks from the same stream, and between stocks from adjacent streams, but exploitation rates were not consistently higher or lower for any particular stock or stock type.

The influence of specific factors upon straying rates, survival rates and exploitation patterns was estimated by means of log-linear models. Stock contributions to various fisheries appeared to be related to the release location, and two stock assemblages were identified within the study area based on the level of similarity among stocks in fishery contribution. Attempts to quantify the level of co-variation among stocks in survival and exploitation rates were hampered by the lack of sufficiently long time series of data, but preliminary results gave no indication of a high level of similarity among stocks or stock types. Still, it was possible to identify stocks which could be used as indicators of the general status of all stocks in the study area in terms of escapement trends, smolt-to-adult survival and exploitation rates. Based on the level of similarity observed, indicator stocks are considered to be useful stock-assessment tools, and can provide useful information for management purposes.

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## 1.0 INTRODUCTION

To more effectively manage stocks of Pacific salmon (*Oncorhynchus spp.*), it is essential that the escapement levels, fishery contributions, and stock-recruit relationships of various stocks be determined. There are about 1000 populations of coho salmon (*Oncorhynchus kisutch*) in British Columbia, which are harvested jointly at many locations on the coast (Aro and Shepard 1967). Monitoring each population on a continuing basis to obtain the required statistics represents an expensive and impractical task. This situation created a need for an alternative, cost-effective method of stock assessment. One simplifying approach has been to partition a population or an assemblage of populations into several units, and use one of these as a representative of the larger set. This is essentially the rationale that promoted the use of 'indexing' techniques for management purposes (ODFW 1981, Cousens *et al.* 1982, Symons and Waldichuk 1984, Walters 1984). For instance, reference is often made to an 'indicator or index stock' when drawing inferences about the demographic traits of salmon populations from a certain geographical region. The following example typifies its current application:

"We selected four indicator stocks for which adequate coded-wire tag recovery information is available, and which we judged to be representative of the stocks of concern in terms of geographic distribution, age, maturity characteristics and exploitation patterns. All indicator stocks coded wire recovery data were from hatchery releases. These data are considered the best scientific information for describing both wild and hatchery exploitation and contribution." (Anon. 1983).

In a context such as the one cited above, it is usually assumed that populations in close proximity to each other behave similarly in terms of their natural variability and susceptibility to exploitation. However, prior to this study, comparable statistics on the patterns of exploitation of coho populations from a regional cluster of streams were not available, so the validity of this assumption was largely a matter of conjecture. The same situation prevails with regards to other species of Pacific salmon, which led Walters (1984) to conclude that; "There is no good empirical evidence for saying much about whether indexing should provide a more accurate way to assess either general stock trends or production parameters". Still, based on an analysis of the 1948-1980 escapement records for all salmon species in the south coast management region of British Columbia, Walters (1984) found that wild stocks which spawned in close proximity (streams surrounding a single inlet) showed strong correlations in escapement levels. These correlations were particularly high for coho and chum stocks. In addition, Walters (1984) found much similarity in the productivity parameters of various stocks within geographical regions in southern B.C. Such results support the notion that geographically close

populations of wild coho can exhibit substantial similarity for some biological traits. Nevertheless, after conducting an extensive literature review, I hypothesized that substantial differences in survival and exploitation patterns within local assemblages of wild and hatchery stocks<sup>1</sup> could still exist. The following paragraphs reviews the evidence in support of this hypothesis.

Fishing mortality exerts selective pressures on populations. This selection is influenced by the distribution of the fish in relation to the fishermen, the behavioral response of the fish to the gear, and the mechanical selection of the gear for morphological characteristics (Parrish 1963). Such factors, respectively referred to as availability, vulnerability and gear selectivity, influence the probability of capture among members of each population. Each factor characterizes a given fish population as it relates to the biological properties of its members, which implies that populations exhibiting different behavioral and morphological attributes may be subject to different exploitation patterns.

Numerous investigations have documented the biological adaptations of Pacific salmonids to local conditions (Helle 1981). Adaptations such as body size, run timing and geographical distribution obviously influence the exploitation of each population (MacDonald 1981). Recent investigations have suggested that less obvious factors, such as smolt size and smolt migration patterns, may also influence exploitation rates and contribution to selected fisheries (Novotny 1978, Bilton *et al.* 1980, Fedorenko and Perry 1984). Additional studies have shown differences between hatchery and wild strains of salmonids which could potentially affect their susceptibility to exploitation. Domesticated and wild strains of fresh water salmonids have been observed to exhibit differences in survival (Keller and Plosila 1981, Fraser 1981, Mason *et al.* 1967, Reisenbichler and McIntyre 1967, Miller 1951, Schuck 1948), growth rate (Vincent 1960, Reisenbichler and McIntyre 1967), feeding behavior (Rowe 1984), distribution (Vincent 1960), catchability (Flick and Webster 1963), and availability to anglers (Cordone and Nicola 1970). Behavioral differences have also been detected between hatchery and wild strains of Atlantic salmon (Dickson and MacCrimmon 1982, Sosiak *et al.* 1979). With regard to coho salmon, juveniles of both types have been shown to differ in behavior (Glova 1978), distribution and feeding habits (Myers 1980), and phenotypic traits (Hjort and Schreck 1982).

Differences between hatchery and wild strains are induced by genetic factors and by hatchery experience, but the contribution of each factor is usually not known with certainty. In some cases, hatchery experience has been observed to have little effect (MacCrimmon and Hawkins 1976, Webster

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<sup>1</sup> In the present report, the term stock will refer to a distinct group of coho which departed as smolts from a stream in a given year, and which is assumed to differ (phenotypically or genetically) from another group in the same or adjacent stream. Thus the coho population of a given stream can contain members of several stocks, such as fish of hatchery origin (released as fry or smolts) and of non-hatchery origin (progeny of wild spawners).

and Flick 1981), but in others the effects are so pronounced that the fish behave as different species after release (Dickson and MacCrimmon 1982). Specific hatchery practices are known to influence subsequent exploitation of domesticated stocks (Johnson 1970, Lander and Henry 1973, Hager and Noble 1976, Bilton 1978, Novotny 1982, Bilton *et al.* 1982, 1984).

The information presented above suggests that genetic factors, hatchery practices and early life experience can influence survival rates and exploitation patterns. However, it remains to be demonstrated that the cumulative effects of these factors, operating in concert within a geographically close assemblage of hatchery and wild populations, can induce substantial differences among these populations in terms of escapement trends, ocean survival and exploitation patterns. To address this issue, and provide an assessment of the potential merits of the 'indicator stock' approach, an empirical investigation was initiated in 1984. The primary objective of this study was to assess the level of similarity between nearby coho populations in terms of ocean survival and exploitation rates. This involves testing two hypotheses; (1) that there exists no significant differences among stocks in ocean survival, and (2); that there exists no significant differences among stocks in their susceptibility to exploitation. With regards to the latter hypothesis, all stocks must meet two conditions to prevent rejection; the stocks must contribute to the same fisheries, and must be subject to the same overall exploitation rate. Thus, the second hypothesis would be rejected if significant differences are detected among stocks in catch distribution or exploitation rates.

A secondary objective of the study was to identify population attributes that have a major influence on exploitation rates and ocean survival. Not all population attributes can be monitored during such a study, so monitoring efforts were to focus on attributes which have been previously hypothesized to have some influence on such traits. These include smolt size and time of ocean entry, stream location, run timing, size and age at return, year of return, stream flow patterns, hatchery conditioning, hatchery rearing practices, and genetic differences<sup>2</sup>. Another secondary objective of the study was to provide a basis for the selection of 'indicator stocks'. It is important to make a distinction between two different categories of indicator stocks. Such stocks could be representative of the year to year variation in survival or exploitation which characterize a particular stock assemblage, and could be identified by reference to the level of co-variation among stocks for such traits. Alternatively, indicator stocks could be considered to be representative of the average survival or exploitation rate operating on

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<sup>2</sup> In the present study, coho populations which reared in different streams will be considered to be genetically distinct irrespective of the distance separating the streams. An exception will be made when a population in one stream consists of the progeny of spawners from a different stream, such as when brood stock is collected elsewhere for enhancement purposes. In such cases, it will be assumed that the progeny (or population) is genetically similar to the population in the parent stream.

the assemblage selected, and could be identified by reference to the level of similarity between each stock and the mean annual rates.

To meet these objectives, reliable information on the biological traits and fishery contributions of several neighboring coho populations must be obtained. Walters (1984) suggested that regional groups of populations exploited in the same mixed-stock fisheries be used for indexing studies. Based on a series of criteria, Walters (1984) identified populations in Statistical Areas<sup>3</sup> 13 and 14 as good candidates for conducting such studies. Several salmon hatcheries are located in those regions, and production and enumeration facilities are already in place. For some stocks, historical and current estimates of escapement and exploitation patterns are available from tagging studies and on-going investigations. These stocks are also subject to intense exploitation and controversial management actions are likely to occur in this area, which creates a need for additional monitoring efforts.

In view of these recommendations, nine streams on the east side of Vancouver Island (Fig. 1.0) were selected on the basis of their accessibility, coho abundance levels, hydrological regimes, availability of logistic facilities, and exposure to enhancement activities. The streams consisted of the Quinsam River, Black Creek, Puntledge River, Trent River, Rosewall Creek, Big Qualicum River, Little Qualicum River, French Creek, and the Millstone River. Some of these streams were used by several types of coho which could be treated as separate stocks. These included fish which were not subject to hatchery rearing (wild stock), or which were hatchery reared up to the smolt stage (production stock), or which were hatchery reared and transplanted as fry into the headwaters (colonization stock). As a result, a total of 14 stocks were available to test a variety of hypotheses about the level of similarity among stocks from the same stream, from adjacent streams, and from the entire region. One particularly useful feature of this experimental set-up lies in the fact that the fish released from the Rosewall Creek hatchery were to consist of the progeny of spawners collected at Black Creek, Trent River, and Little Qualicum River. The effects of hatchery conditioning and release location could thus be assessed from comparisons between the Rosewall Creek stocks and their natural counterparts. Insight into the influence of genetic factors could also be obtained by conducting comparisons among stocks released from Rosewall Creek.

To generate reliable information on their survival and exploitation patterns, members of each stock were to be tagged prior to their ocean migration. Coded-wire tags (CWT, Jefferts *et al.* 1963; Jewell and Hager 1972) were used for this purpose since this technology has been shown to be a cost-

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<sup>3</sup>As identified by the Canada's Department of Fisheries and Oceans (DFO).

effective for large scale juvenile tagging programs, and because all fisheries in Canada and the U.S are routinely sampled for CWT recovery. Escapement estimates for all stocks subject to tagging were to be obtained primarily by means of direct enumeration of adults intercepted at counting fences located on each stream. Information on tag recoveries in various fisheries was to be obtained from the large database maintained for such purposes by DFO personnel at the Pacific Biological Station in Nanaimo, B.C.

Given the time limits associated with this dissertation, the study was originally designed to be completed by 1988. Coded wire tagging was to be conducted on as many stocks as possible during the spring of 1985, 1986, and 1987, and escapement enumeration was scheduled to be conducted each fall during the 1985-1988 period. Given the amount of time required for catch sampling and data entry, analysis of the catch statistics was to be delayed until 1989, and be completed by 1990. By and large the study proceeded as planned, and all objectives of the study were met.

## 2.0 DESCRIPTION OF STREAM POPULATIONS AND DATA SOURCES

The streams selected for this study were characterized by the presence or absence of permanent enhancement facilities. The absence of such facilities at some locations dictated the need to conduct extensive field operations, and rely on sampling methods which differed substantially from those employed at other locations. Because of this distinction, both types of systems are described separately.

### 2.1 Systems with major enhancement facilities

#### Quinsam River

The Quinsam River is located at the northwestern edge of the town of Campbell River, about 200 km north of Nanaimo. The river is approximately 40 km long, and flows in a northerly direction to connect with the Campbell River, which flows for another three km until it joins the Strait of Georgia at 125° 16' W., 50° 02' N. This river drains an extensive network of tributaries and lakes in its headwaters, but only the lower 29 km of stream habitat is accessible to coho salmon (Lukyn *et al.* 1985). Escapement records indicate that historically, all five species of Pacific salmon spawned in this river, which was renowned for its chinook sport fishery (Hancock and Marshall 1985).

In 1974, a hatchery was constructed to enhance the salmon resources. The hatchery is located approximately one km upstream from the river mouth, just above the Cold Creek junction. About 500 m upstream of the hatchery complex, a large, permanent steel fence was built across the river. This fence has been used to monitor juvenile migrations and to divert a portion of the adult escapement to holding ponds for brood stock collection.

In 1975, juvenile trapping operations were conducted to assess the productivity of the river prior to enhancement activities. An estimated 2,730,000 pink salmon fry, 56,000 chum fry, 400,000 coho fry, 40,000 coho smolts and 4,600 fry emigrated from Quinsam River above the new fence (Reinhardt and MacKinnon 1979). The median dates of the coho fry and smolt migrations were May 15 and May 24 respectively. Currently, a large portion of the coho smolt population emigrating from this river is of hatchery origin (Table 2.1).

English *et al.* (1990) gave a detailed account of the hatchery production and sampling programs conducted in this river by hatchery personnel. Coho are reared to the smolt stage (production group) in holding ponds and channels adjacent to the hatchery. Several weeks prior to release, a fraction of the fry population in each section is coded-wire tagged. The total population in each channel is then determined by mark-recapture methods. Dead fry recovered subsequently are deducted from the mark-recapture estimate to provide an estimate of the remaining fry population. English *et al.* (1990) noted that these 'book estimates' are usually positively biased, since undetected losses due to predation and decomposition are not included.

Throughout the rearing period, the fry in each rearing basin are sampled regularly to obtain length and weight measurements. The last measurements are usually taken several weeks prior to the release date, and average sizes at release are estimated by hatchery personnel based on expected growth that occurs during the interval prior to release. Coho smolts are usually given access to the river when they appear fully smolted and agitated (R. Kraft<sup>4</sup>, pers. comm.), rather than at times recommended for obtaining maximum biomass at return (Bilton *et al.* 1984). Once access is given, smolts evacuate the raceways over a five day period, with 50-90% of them leaving within 48 h. It is estimated that the smolts reach the estuary within 48 h after entering the river (R. Kraft, pers. comm.).

Each year, some of the fry produced at the hatchery are given a characteristic fin clip (adipose and left ventral or right ventral fin excised), coded wire tagged, and released during their first summer into several lakes in the headwaters (colonization group). A substantial number of smolts are also produced each year from spawning events in the accessible portion of the river (wild group). During the 1985-1988 period, smolt traps were installed at the main fence, and used to conduct a partial enumeration of the migration of smolts of wild and colonization origin. Trap catchability tests were used to estimate the daily smolt output for each group. During 1986 and 1987, sampling and coded-wire tagging of wild smolts was also conducted on site throughout the migration period. Smolts that were injured or exhibited abnormalities were released untagged. Approximately 20,000 smolts were tagged in 1986, and 24,000 in 1987, which represented about 50% of the wild smolts captured each year (Table 2.1). Each season, three tag codes were used to tag each group, and approximately equal numbers of smolts were tagged with each code.

Random samples of ~100 smolts were obtained from the wild and colonization groups in 1986-1987, as well as from the production group in 1986. Smolts from the first two groups were collected

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<sup>4</sup> Assistant Manager, Quinsam River hatchery, Campbell River, B.C.

during the peak migration period, while those of the later group were collected just prior to the release date. All smolts were frozen, and subsequently examined and measured at the University of British Columbia (UBC). No smolt size data on the 1985-1988 production releases were obtained from the hatchery records.

Coho escapement to this river has exceeded 20,000 since 1983 (Table 2.2). Most of the coho returning to this river converge towards the hatchery, and access the holding channels via the fishway entrance located near the fence. These fish are sexed, aged, measured, examined for marks, and processed for sales and brood stock acquisition. Coho which do not enter the fishway, and seek to move upstream are usually enumerated at the fence, and given access to the upper sections of the river. Each year, a substantial number of adults spawn in the river below the fence. In addition, floods occasionally allow some adults to escape to the headwaters undetected. As a result, it is estimated that up to 10% of the coho escapement to the river is not accounted for or examined for marks (R. Kraft, pers. comm.). Total counts by age category, and estimates of the number of jacks<sup>5</sup> and adults that did not enter the hatchery channels were obtained through interviews with the hatchery manager (Table 2.2). Daily fence counts of adult and jack coho, tagged proportions for each age group, and size measurements were obtained from the hatchery records for the 1985-1988 escapement period. Field observations conducted by hatchery personnel indicated that the migration pattern of adults of hatchery origin is similar to that of colonization and wild adults returning to the headwaters (R. Kraft, pers. comm.).

### Puntledge River

This river is located approximately 120 km north of Nanaimo, near the town of Courtenay. It is approximately 14 km long, 59 m wide at mid-section, and flows from Comox lake to the Courtenay River, which in turn connects to the Strait of Georgia at 124° 59' W, 49° 41' N. The Puntledge River has three tributaries: the Tsolum River, Morisson Creek, and the Browns River, which respectively join it two km, three km, and six km upstream of the estuary. Currently, only the lower 10 km section of the river is accessible to adult salmon owing to the presence of two hydroelectric dams located 10 km and 15 km upstream from the river mouth. Both dams serve to control the flow of the river throughout the year, which still varies substantially during the fall period.

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<sup>5</sup> Small, mature coho, which return to spawn after a relatively short period of marine life (< 9 months). In this report, they are also referred to as age 2+ fish, to distinguish them from adults (age 3+).

Historically, all species of Pacific salmon were found in the Puntledge River, and it was renowned by sport fisherman for its vigorous races of fall and summer chinook (Hancock and Marshall 1985). In 1965, a spawning channel was constructed as part of a program to protect the fishing resource. In 1972, a hatchery was built by DFO primarily for chinook production. The facility currently produces coho, chum and chinook salmon, as well as steelhead trout. The main hatchery building, chinook rearing channels, the aeration tower, and the adult diversion fence are located approximately one km upstream from the river mouth. Coho are kept in semi-natural rearing channels located at the upper rearing site about 10 km upstream from the river mouth. Currently, most of the coho production in this river is of hatchery origin (Table 2.1). Escapement of adult coho to the river has declined considerably during the last few years, to about 5600 adults in 1988 (Table 2.2).

English *et al.* (1990) gave a detailed account of the hatchery production and sampling programs conducted in this river by hatchery personnel. Coho are reared to the smolt stage (production group) in four separate sections of the upper channels. Several months prior to release, a fraction of the fry population in each section is coded-wire tagged. The total population in each section is then determined by mark-recapture methods. Throughout the remaining rearing period, each group of fry is monitored periodically, and all dead fry recovered are enumerated to produce a 'book estimate' of the remaining population (English *et al.* 1987).

Throughout the rearing period, fry are sampled regularly to obtain length and weight measurements. The last measurements are usually taken several weeks prior to the release date. The average sizes at release are estimated by hatchery personnel on the basis of the expected growth that occurs during the interval prior to release. Random samples (300-400) of length and weight measurements taken by hatchery personnel during the last sampling event in 1985 and 1986 were obtained for comparative purposes. In addition, a random sample of 100 smolts was obtained from the rearing channel in 1987 just prior to the release period. These specimens were frozen, and subsequently examined and measured at UBC.

Coho smolts are given access to the river when they appear to be fully smolted, and exhibit increased activity (H. Genoe<sup>6</sup>, pers. comm.). The smolts in each channel are released sequentially over a period of several days. Usually, about 10% of the smolts leave the channel as soon as access is given, but it may take as much as two weeks for the 90% of the smolts to leave the channel (C. Biggs<sup>7</sup>, pers. comm.). Field observations by hatchery personnel indicated that about 50% of the smolts leave over the

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<sup>6</sup>Manager, Puntledge River hatchery, Courtenay, B.C.

<sup>7</sup>Assistant Manager, Puntledge River hatchery, Courtenay, B.C.

three day period just prior the end of the release period reported. Once the smolts have entered the river, it is estimated that they require two to seven days to reach the estuary (C. Biggs, pers. comm.).

Each year, a fraction of the fry production is used to enhance other streams in the area. Substantial numbers of fry are coded-wire tagged and released during their first summer into the upper reaches of the Puntledge River (colonization group). Sampling conducted during 1983 indicated that smolts from colonization releases averaged about 10 g in weight, and migrated to sea throughout the month of May. However, juvenile trapping programs are not conducted routinely to determine the smolt production resulting from various releases. Based on a few experimental releases, it is estimated that 60-70% of the smolts produced die as they pass through the turbines of the power generating station. Juvenile trapping operations conducted above the dam during 1989 showed that smolt output could reach 114,000. This figure indicated that fry-to-smolt survival was about 20.3%, and that approximately 13% of the colonization fry reached the ocean as smolts. In the absence of additional information, it was assumed that these survival rates also applied to the fry released between 1984 and 1986 (Table 2.1).

Smolts captured in 1988 averaged 96 mm and 8.1 g. The smolts migrated past the dam between May 3 and June 23, but the majority migrated between May 22 and June 15. The median migration date was June 1, which appeared to be about one week later than was observed in previous years (H. Genoe, pers. comm.). Smolts sampled in 1988 appeared to be similar in size to those examined in 1986 and 1987 (C. Biggs, pers. comm.). Therefore, for comparative purposes, it was assumed that colonization smolts migrating during 1985-1987 period had the same average size and coefficient of variation as those measured during 1988.

Smolts released for colonization purposes supplement the natural fry production in the river, which consists of the progeny of wild coho returning to Morisson Creek, and the progeny of hatchery fish that spawn in the river. Total smolt production from the lower section of the river is also not known, but is estimated to be < 30,000 (H. Genoe, pers. comm.). No information is available on the migration pattern of these smolts.

Each year, most of the adult coho and chinook returning to the river are directed from the counting fence to the holding channels. Some adults usually bypass the fence during large floods, and escape undetected to the upper sections of the river (H. Genoe, pers. comm.). In addition, a substantial number of coho returning to the hatchery each year do not enter the holding channels, but remain in the river to spawn and die. As a result, it is estimated that 25-35% of the escapement of hatchery coho each year is not examined by hatchery personnel (H. Genoe, pers. comm.). Estimates of the fraction of the

total escapement that remained in the river, and the number of coho sampled each year for CWT recovery were obtained through interviews with the hatchery manager (Table 2.2). Daily fence counts of adult and jack coho, tagged proportions among each age group, and a random sample of size measurements were obtained from the hatchery records for the 1985-1988 escapement period.

### Rosewall Creek

This creek is located approximately 80 km north of Nanaimo, near the town of Bowser. The creek is about 20 km long, 14 m wide at mid-section, and flows in a northeasterly direction until reaching the Strait of Georgia at 49° 28' N, 124° 48' W. Only the lower section of the creek is accessible to salmon due to the presence of falls located four km upstream from the mouth. Historical records of escapement estimates obtained by fishery officers indicate that coho, chum, and a relatively small number of steelhead spawn in this creek. Between 1948 and 1975, escapement levels ranged from 125 to 7500 for chum salmon, and from 75 to 750 for coho salmon (Hancock and Marshall 1985).

In 1968, a hatchery was built next to the creek by the Fisheries Research Board of Canada for experimental purposes. This facility consisted of a series of concrete raceways (20 m x 2 m), laboratory and storage facilities, and water pumps for supplying ground water to the facility during the winter months. An electrical fence (Burrows 1957) was constructed in 1974 to intercept all adult coho returning from previous releases (see Bilton *et al.* 1982). The electrical field generated by this fence prevents salmon from migrating upstream, by temporarily paralyzing them while they are within the field. Coho released from the hatchery tend to return directly to the raceways via the spillway located 200 m downstream from the fence. Field observations indicate that most coho hold in the lower 500 m of the creek three to five days before entering the holding channels, but some adults have been observed to hold there for up to three weeks, until increasing discharges from the spillway or the river induce them to enter the holding channels (R. Humphreys<sup>8</sup>, pers. comm.). Field observations indicate that in years of high escapement resulting from previous releases, a large portion of the coho escapement enters the hatchery channels (Table 2.2). In the absence of fry releases from the hatchery, wild smolt production is considered to be negligible (Table 2.1), given the small number of spawners detected, and the limited amount of spawning habitat available to them (R. Humphreys, pers. comm.) .

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<sup>8</sup>Manager, Rosewall Creek hatchery, Bowser, B.C.

In 1984, this site was selected for experimental releases of juvenile coho for the present study, to determine if the exploitation pattern of natural coho populations could be altered by hatchery rearing practices and changes in rearing location. Brood stock was obtained from Black Creek, Trent River and Little Qualicum River in order to release fish of comparable genetic make-up as the natural stocks used in this study.

During the early, mid and late part of November 1984 and 1985, 20-30 ripe adult coho were collected unsystematically at Black Creek, Trent River, and the Little Qualicum River. All adults were killed, and transported on the same day to the Rosewall Creek hatchery. Eggs were then removed from each female, and fertilized with a mixture of milt collected from up to four males. All eggs were incubated at the hatchery, and the fry from each stock were reared separately to the smolt stage. By and large, the rearing and feeding practices used at Rosewall conformed to those used at other public hatcheries operated by DFO personnel. Efforts were made to regulate both temperature and feeding regimes during the rearing period to produce smolts that were similar in size to their natural counterparts.

During February 1986, and April 1987, each group of fry was counted, measured, fin clipped and coded-wire tagged within a 36-72 h period. Fry that were injured or exhibited abnormalities were removed, and not tagged. In most cases, three distinct tag codes were used for each group. Tag retention tests were conducted on each group approximately 24 h after tagging was complete. All fry mortalities that occurred between the tagging and release period were deleted from the total count to produce 'book' estimates of the number of smolts released.

On May 1, 1987, all groups were found to be suffering from 'cold water disease' which was caused by the myxobacterium, *Cytophaga psychrophila*. This condition was presumably induced by a combination of low water temperatures and limited feed availability. Diseased fry showed signs of severe skin erosion, particularly at the anterior edge of the upper jaw. Attempts were made to control the outbreak by flushing the raceways with Chloramine T, and increasing the feed ration, but considerable losses of smolts in each group occurred prior to the release date. These smolts were released earlier than planned since exposure to salt water tends to eliminate the infection (D. Kieser<sup>9</sup>, pers. comm).

Two days prior to the release dates, random samples of 100-200 smolts were obtained from each group for comparative purposes. All specimens were frozen on the same day, and subsequently measured at UBC. All smolts were released simultaneously at dusk on May 20, 1986, and May 14,

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<sup>9</sup>Fish disease section, Pacific Biological Station, Nanaimo, B.C.

1987. Field observations indicated that over 50% of the smolts released had migrated to sea by the following day, and over 90% of them left the creek within 72 h. The final number of smolts released ranged from approximately 10,000 to 24,000 per group.

During the fall of 1986, 1987, and 1988, the electric fence was activated to prevent returning salmon from reaching the middle and upper reaches of the creek. The vast majority of coho converged directly to the hatchery spillway located below the fence. Each day, coho which entered the holding channel above the spillway were counted, measured, sexed, examined for missing fins, and beheaded if the adipose fin was missing. All trout caught in the channel were recorded, and released in the creek. Stream surveys were also conducted periodically throughout the lower reaches of the creek to determine the number of carcasses present, and to recover tagged coho. During early December each year, attempts were made to capture any coho still holding in the lower pools using electrofishing gear. The electrical fence was kept in operation until mid January each year, but less than 2% of the run was caught after Dec. 20. Steam surveys were conducted occasionally upstream from the fence, but no live or dead coho were ever found. All heads recovered were frozen, and eventually sent to DFO's Head Recovery Laboratory for coded-wire tag identification.

#### Little Qualicum River

This river is located near the town of Qualicum Beach, approximately 40 km north of Nanaimo. The river is approximately 20 km long, 30 m wide at mid-section, and connects to the Strait of Georgia at 124° 30' W, 49° 22' N. Only the lower 13 km section is accessible to salmon due to a series of large falls located in the Provincial Park. The river has two major tributaries which connect to the accessible portion of the river; Kinkade Creek and Whiskey Creek. Historical records of escapement from fishery officer estimates indicate that all five species of Pacific salmon spawn in the river, but populations of coho and chum are the most abundant, and reached up to 7,500 and 100,000 adults respectively prior to 1970 (Hancock and Marshall 1985).

During 1979, a salmon enhancement facility was constructed by SEP in an area adjacent to the river, approximately 2.5 km upstream of the estuary. The facility consists of a diversion fence and fishway, laboratory facilities, incubation chambers, chinook rearing ponds, and a large chum spawning channel. This channel is 4.2 km long, 7.5 m wide, and is designed to accommodate up to 50,000 chum spawners. The channel is sectioned at 800 m intervals by diffuser structures which allow control of the distribution of spawners in the channel. Water flow through the channel is controlled manually

throughout the year by adjusting the water intake from the river. As a result, channel flow and spillway outflow remain fairly constant throughout the year, irrespective of fluctuations in river discharge.

The main purpose of this facility is to produce chum fry. However, some chinook fry rear in side channels and are released as smolts each year. A substantial number of wild coho smolts are also produced each year (Table 2.1). These consist of the progeny of adults spawning under natural conditions in the river, as well as of adults which are given access to the spawning channel each year. Since the coho, chinook and chum fry rear in the channel under natural conditions, and are not manually fed or sorted, these can be considered as wild populations. Each year, some coho and chinook fry emigrate from the channel during the chum fry migration, but a substantial number of coho fry remain in the channel until the smolt stage. Field observations have shown that the majority of coho smolts remain in the channel until the end of the chum fry migration period (J. Hargrave<sup>10</sup>, pers. comm.).

Each year, the total fry and smolt output from the channel is estimated for each species by means of a mechanical sampling device installed at the exit of the channel. This device allows for a partial sampling of the small chum fry leaving the channel, and also traps the larger coho smolts for subsequent sampling and tagging. For the purposes of this study, juvenile sampling and coded wire tagging operations were conducted throughout the coho smolt migration periods of 1985, 1986 and 1987. The number of coho smolts intercepted at the exit of the channel each year ranged from about 17,000 to 24,000 (Table 2.1). All smolts captured which showed no physical deformities were anesthetized, fin clipped and tagged in a laboratory located adjacent to the main fence. After a brief recovery period, all smolts were given access to the river. Three tag codes were used each season, and approximately equal numbers of smolts were tagged with each code. Over 95% of the smolts captured during 1985 and 1986 were tagged, but only 87% were tagged in 1987 due to logistic problems that occurred at the end of the migration period. Each year, approximately 100 smolts were collected at random during the migration period. All specimens were frozen, and subsequently examined and measured at UBC.

Adult and jack coho escaping to this system spawn mainly in the channel, and less frequently at various locations throughout the river (Table 2.2). Adult salmon are sorted as they access the channel via the fishway. A set number of chinook and steelhead are removed for egg takes, but the remaining fish are given access to the main channel to spawn freely. Estimates of the number and type of salmon in the channel are obtained primarily through deadpitch counts conducted at regular intervals during the

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<sup>10</sup>Manager, Little Qualicum River spawning channel, Qualicum Beach, B.C.

spawning period. All carcasses processed are examined for external marks and missing fins. Coho that appear to be tagged are sexed, measured and beheaded for CWT identification.

Little spawning activity has been observed in the two tributaries, presumably due to the scarcity of suitable spawning substrate (J. Hargrave, pers. comm.). Estimates of the number of spawners above and below the fence were obtained from visual surveys conducted by hatchery personnel. Tagged proportions among adults spawning above the fence are usually about 10% of those observed in the channel, and in areas below the fence (J. Hargrave, pers. comm.). Based on this observation, it is believed that coho produced in the channel spawn mainly in the channel and in the river below the diversion fence. Carcasses found in the river during stream surveys are also sampled for tag recoveries. The tagged proportions among adults and jacks escaping to the river were estimated separately for each age group from the tagged proportions in each section weighted by the number of spawners in each section. Daily fence counts for adults and jacks were obtained from the hatchery records for the 1985-1988 escapement period. Samples of size measurements collected randomly throughout the same periods were provided by hatchery personnel.

### Big Qualicum River

This river is located approximately 50 km north of Nanaimo. Most of the 150 km<sup>2</sup> watershed of the Big Qualicum River is channelled by the upper Qualicum River into Home Lake. This lake flows into the Big Qualicum River, which flows 11 km before reaching the Strait of Georgia at 49° 24' N, 124° 36' W. Hunt's Creek joins the river about two km above the estuary, and is the only major tributary of the river between Home lake and the ocean. Escapement records from fishery officers indicate that all five species of Pacific salmon were historically found in the river, but populations of chinook, coho and chum were the most abundant (Hancock and Marshall 1985).

Fraser *et al.* (1983) compiled much of the information available on demographic traits of the coho population prior to enhancement activities. During 1947 to 1958, coho escapement averaged about 3300 fish. Most of the spawners entered the river between mid-September and late December. The median date of the migration during this period was estimated to be October 28 d  $\pm$  9 d. Typically, the run was composed mainly of age 3+ adults (80%), but jacks accounted for up to 50% of the run in some years. Age 4+ adults (4<sub>2</sub>) accounted for less than 2% of the escapement. The average period of stream residency was estimated to be about 33 d.

The average annual coho fry output during 1959-1962 was estimated at 347,000, and the average coho smolt output was 29,000. Coho smolts migrated to sea primarily between early April and mid-June. The migration period ranged from 47 d to 162 d, and averaged 119 d. From 1960 to 1967, the median date of juvenile coho migration occurred within the May 3-21 period. Coho smolts averaged 98.5 mm in fork length, 11.2 g in weight, and 0.011 in condition factor (Fraser *et al.* 1983).

In 1963, a dam was constructed to control the lake discharge and flow to the one km long chum spawning channel located adjacent to the river. In 1967, rearing channels, incubation facilities, a permanent counting fence, and ancillary facilities were built by DFO. The fence is located about 0.5 km upstream from the river mouth, and can intercept all adults returning to the river. About 500 m upstream from the main counting fence is an adult diversion fence, which controls access to the spawning channel, and to the holding channels.

English *et al.* (1990) gave a detailed account of the hatchery production and sampling programs conducted in this river by hatchery personnel. Currently, over one million coho fry are reared and released as smolts from the main channels each year. An experimental rearing channel located four km upstream (Mundie's channel) is also used to produce an additional 250,000 coho smolts each year. These 'production' releases make up the bulk of the smolt output from this system (Table 2.1).

Several months prior to release, a fraction of the fry population in each channel is coded-wire tagged. The total population in each channel is then determined by mark-recapture methods. Throughout the remaining rearing period, each population is monitored periodically, all dead smolts recovered are removed, and 'book' estimates of population sizes are generated. Throughout the rearing period, fry in each rearing channel are sampled regularly by hatchery personnel to obtain length and weight measurements. The last samples are usually taken within one month from the release date, and the size at release is estimated based on the expected growth that occurs during the interval prior to release. Each year, 200-400 length and weight measurements collected by hatchery personnel during the last sampling events were obtained for comparative purposes. Random samples of 50 smolts were also obtained from the rearing channels just prior to the release date in 1986 and 1987 to provide additional data on smolt sizes. These specimens were frozen, and subsequently examined and measured at UBC.

Smolts from the production group are usually given access to the river each year over several days to minimize predation losses. Once the gates are opened, the smolts leave the channels rapidly. Based on studies conducted during 1974 and 1979-1981, Mace (1983) reported that migration rate decreased exponentially during the first eight days, and linearly during the subsequent four days, so that

over 95% of the smolts reached the estuary within 10 d after release. Smolts released during the 1985-1988 period were assumed to follow the same migration pattern as reported by Mace (1983), since the rearing methods and release procedures have not been modified (G. Ladouceur<sup>11</sup>, pers. comm.).

Each year, a fraction of the total fry production is used to enhance other streams in the area. Substantial numbers of coho fry are released during their first summer in Home lake (colonization group). These fry supplement the wild fry production of the river, which consists mainly of the progeny of coho that spawn in the river and the chum spawning channel. Fry traps installed at the main fence are used to intercept a fraction of the chum fry and coho smolts leaving the spawning channel, river and lake each year. Total fry and smolt outputs from these areas, and information on their migration pattern are obtained from this partial enumeration. Examination of the hatchery records indicated that since 1980, coho fry output from the river has ranged from 500,000 to over 1.2 million, with total smolt output exceeding 50,000. Data collected between 1979 and 1984 (1985+ not available) indicated that smolts emigrated predominantly between early May and mid-June, which overlapped with the release period for production groups. On average, peak migration occurred around May 25, with over 50% of the smolts leaving between May 20 and June 4. These smolts averaged about 105 mm in size, and 12 g in weight. Smolt output associated with recent colonization releases has exceeded 500,000, but none of the fry released during 1984-1988 for colonization purposes were coded-wire tagged (Table 2.1).

Currently, over 200,000 adult salmonids return to the Big Qualicum River each year to spawn, but coho make-up only a small fraction of this (Table 2.2). Due to flow controls, nearly all salmon returning to the river can be intercepted at the fence. Only a partial enumeration of the chum salmon is conducted, but most coho and chinook are diverted to the holding channels. Coho are enumerated and aged while being processed for sale, but only a fraction of these are measured, sexed, and sampled for CWT recovery. Each year, some coho remain below the fence to spawn and die, but this group makes up only a negligible fraction of the total run (<1%, G. Ladouceur, pers. comm.). Daily fence counts of adult and jack coho, estimates of tagged proportions for each age group, and a random sample of size measurements were obtained from the hatchery records for the 1985-1988 escapement period.

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<sup>11</sup>Manager, Big Qualicum hatchery, Home Lake, B.C.

### Millstone River

This river passes through the town of Nanaimo, and connects with Departure Bay in the Strait of Georgia at 123° 58' W, 49° 12' N. The river is 13 km long and 15 m wide at mid section. Several lakes are found in its watershed, the largest of which is Brannen lake (109 ha.). A fall located about two km from the river mouth prevents adult salmon from accessing the middle and upper reaches. Historical escapement estimates by fishery officers indicate that only a small number of chum and coho salmon spawn in this river. The magnitude of wild smolt production is not known with certainty, but is believed to be low (< 1000) given the limited amount of accessible habitat (R. Hurst<sup>12</sup>, pers. comm.).

In 1981, a coho colonization program was initiated by personnel of the Nanaimo River hatchery, in the hope of capitalizing on the rearing potential of the habitat in the upper reaches. Since 1981, small fry (1 - 10 g) have been released each year during the summer and fall period in the river and the lake (Table 2.3). In 1985, two juvenile counting fences were installed to assist in the evaluation of coho colonization methods. One permanent fence was installed at the outlet of Brannen lake mainly to monitor fry movements in and out of the lake. A temporary smolt trap was installed in the river about 0.5 km from the estuary, to monitor migration patterns and overall survival rates associated with various fry releases (Hurst and Blackman 1988).

A substantial portion of the fry used for colonizing various sections of the river during 1985 and 1986 were coded wire tagged prior to release. Juvenile trapping conducted at the lower fence during 1986 and 1987 provided information on the growth rates, survival rate and migration pattern of each group. Coho catches during each year amounted to 15,775 and 14,440 smolts respectively, of which 57.8% and 56.4% had been tagged previously (Table 2.1). Estimates of the number of smolts tagged with a given code were obtained each year based on the recapture rates of fin clipped smolts released with specific tag groups. Throughout both migration periods, a variable number of smolts were collected, measured, weighted, and released while processing the catch each day. Estimates of daily smolt catches by fin clipped group, as well as the measurements collected each year, were obtained from R. Hurst.

In the absence of a counting facility, stream surveys are conducted during the spawning season by personnel from the Nanaimo hatchery for purposes of brood stock acquisition, escapement estimation and coded-wire tag recovery. Escapements of adult and jack coho are crudely estimated on the basis of

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<sup>12</sup>Special Projects Division, SEP/DFO. Nanaimo, B.C.

foot survey counts obtained during the peak period of spawner abundance. Escapement estimates, as well as tagged proportions by age group, were obtained from the Nanaimo River hatchery records (Table 2.2). Accurate data on the migration pattern and the size of males and females is not collected routinely by hatchery personnel during field surveys. However, field observations indicate that the escapement pattern is very similar to that observed at French Creek, with runs occurring mostly during the late October, early November period. Adult sizes are usually larger than those observed at the French Creek, and tend to be similar to those observed at the Little Qualicum River (R. Hurst, pers. comm.). Therefore, run timing and spawner sizes were assumed to conform to the patterns observed at the corresponding locations.

## 2.2 Systems lacking major enhancement facilities

### Black Creek

Black Creek is located approximately 30 km north of Courtenay. The creek begins in the higher elevations of Mount Washington, and flows northeast to join the Strait of Georgia at 125° 07' W., 49° 51' N. The stream is about 26 km long, and has a width of 5-6 m at mid-section. It varies from a steep gradient mountain stream at higher elevations, to a sluggish meandering creek flowing through agricultural land and swamps, with dense overgrowth of wild rose, salmonberry and trees. In the lower reaches, the creek includes small lakes, swampy depressions, beaver pounds and several tributaries (Millar, Kelland, Surgenor and Sayer Creeks). Bank erosion, undercutting, scouring and silting are nearly absent along most of the middle and upper reaches, but are quite common in the lower sections. Throughout the creek, the streambed is composed mostly of a mixture of sand, silt and clay. Peak discharge is known to reach  $60 \text{ m}^3 \cdot \text{s}^{-1}$  during fall and winter floods after a heavy rainfall. During summer, stream flow may decrease to negligible levels and cause some sections in the lower reaches to be completely dry.

Physical features of the stream and particular attributes of its salmon population have previously been described in detail by Hamilton (1978), Hancock and Marshall (1985), and Bocking *et al.* (1988). The stream supports resident and anadromous populations of cutthroat and rainbow trout, as well as a considerably larger population of coho salmon. Prior to 1967, escapement estimates were based on visual surveys conducted by fishery officers, and ranged from 1500 to 15,000. A fence was first constructed near the mouth of the river in 1968, and a partial escapement enumeration indicated that at least 4656 coho entered the creek. This fence was refurbished in 1972, and operated intermittently by

personnel from the B.C. Fish & Wildlife Branch. A larger fence was installed in 1975, and fence counts showed escapement levels of 9492 coho, of which 20.5% were jacks. Partial counts in 1976 showed escapement levels of at least 3,510 coho, and 199 trout. Escapement enumeration was also conducted by DFO personnel from 1978 to 1980. Escapement levels during this period ranged from 800 to 7600 adults, and from 550 to 1200 jacks (Clark and Irvine 1989).

Hamilton (1978) noted that upstream migration at Black Creek is governed by the arrival of fall rains. In some years, insufficient flow delayed upstream migration considerably, inducing substantial fish mortality at the mouth in tidewaters (250 in 1974), and prevented access to spawning grounds in headwaters. Between 1969 and 1976, the upstream migration occurred as early as October 3, and as late as November 9. Hamilton (1978) reported a considerable loss of spawning habitat in the stream has occurred since 1960. Logging activities, habitat loss due to land development, and low summer flows induced by the seasonal lack of rainfall and increased agricultural water demands, were assumed to seriously affect the rearing capacity of the system. During the late 1980's, coho were still spawning throughout the creek, but most of the spawning activity was in Millard Creek, in areas adjacent to the Kelland and Sturgess road bridges, and in the upper reaches of the stream near the Duncan Main logging road (Jack Trent<sup>13</sup>, pers. comm.).

Smolt traps operated during 1978 and 1979 indicated outputs of approximately 48,000 coho smolts each year (Clark and Irvine 1989). Distinct tag codes were used each year to identify smolts captured prior to and after the peak migration period. Scale samples showed that over 97% of the outmigrants were of age 1+. No attempt has been made to conduct substantial enhancement activities, in part because the escapement levels are still considered adequate. However, in recent years there has been a series of small fry releases into the stream to determine the survival rate of coho fry in certain habitats (Table 2.3). Given the suspected low survival rate associated with previous fry releases, and the relative magnitude of the wild fry population, the vast majority of smolts outmigrating are considered to be of wild origin.

### Trent River

The Trent River is located near the town of Royston, south of Courtenay. The river begins in the Beaufort mountain range, and flows northeast to join the Strait of Georgia at 124° 56' W., 49° 39'

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<sup>13</sup> Fishery officer, Dept. of Fisheries & Oceans, South Coast Division, Campbell River, B.C.

N. The total length of the system, including several tributaries in the upper reaches, is approximately 30 km. Seven kilometers from the sea, Bloedel Creek connects to the mainstem, but this tributary is inaccessible to adult salmon because of a series of falls and cascades in its lower section. Several falls, located one km upstream from this junction, prevent fish from reaching the upper sections of the river, including its two tributaries: Tremain Creek and Idle Creek. The intertidal section of the river is about one km long and 30 m wide, with sand flats, rocky outcrops, some logging debris and aquatic vegetation. This area is located at the periphery of the Comox harbor, a two-layered estuarine system (Morris *et al.* 1979), which receives the waters of the Puntledge and Tsolum Rivers.

Steep and high banks with very little overgrowth are found along the main section of the river. Cover is mainly in the form of large log jams ( $20 \text{ m}^2 - 400 \text{ m}^2$ ), scattered mainly throughout the lower two km section. The river bed is composed mostly of bedrock with rocky outcrops and boulders. Along the middle reaches of the river, spawning substrate is scarce, and usually consists of shale material. In the lower reaches, spawning substrate is more abundant and consists of gravel, cobble and shale deposits. As a result, the water is generally clear, and adult densities in pools are easily assessed visually. Seasonal fluctuations in water levels are pronounced, and large floods are common particularly during the winter period. Peak discharge can reach  $150 \text{ m}^3 \cdot \text{s}^{-1}$  during fall and winter months, causing extensive gravel movement, log jam shifts, and scouring in the lower reaches. During late summer, flows usually decrease to negligible levels, and several sections in the lower reaches dry up completely.

The physical features of this river, and particular attributes of its salmon population have previously been described by Hancock and Marshall (1985), and Bocking *et al.* (1988). In addition to coho, the river supports anadromous populations of rainbow trout, chum salmon, and pink salmon. Because of the lack of cover, spawning substrate, and rearing habitat, the river is not recognized as a good coho producer. Prior to this study, escapements were estimated by fishery officers from visual surveys. During the 1962-1984 period, coho escapements ranged from 75 to 1000 adults, and rarely exceeded 350 fish (max. 1000). The river also served as spawning grounds for a small number of pink salmon (3-150), up to 3000 chum salmon, and an unknown number of rainbow trout. Coho spawning activity has been reported throughout the system, but the major spawning grounds are located in the lower two km of the river, and in the lower section of China Creek. Chum salmon spawn mainly in the lower section of the river, between the mouth of the river and the railroad bridge.

Attempts have been made to enhance coho and rainbow trout by stocking the mainstem, as well as Bloedel Creek and Bradley Lake. To facilitate local enhancement efforts, a small hatchery was built near Courtenay in 1979. The hatchery is operated on a volunteer basis mainly by members of the

Courtenay Fish and Game Club, and funded by the DFO's Salmonid Enhancement Program (SEP). The water source of this hatchery consists of ground water provided mainly by Morrison Creek and Bevon Creek, two tributaries of the Puntledge River. Since 1981, volunteers and SEP personnel have collected brood stock, reared the fry to various stages, and released them in several sections of the Trent River. The available records show that a variety of brood stocks have been used for enhancement purposes (Table 2.3). Marked fry were released mainly for habitat assessment, while unmarked fry were released for enhancement purposes. Given the lack of characteristic fin clips, the majority of these outplanted fry are not readily distinguishable from wild fry in samples collected during the juvenile tagging. In the absence of reliable information on the historical abundance of wild smolts, and on the fry survival rates associated with most releases, it is not possible to accurately assess the success of the enhancement program prior to this study. Furthermore, since the smolts emigrating from this river consist of a mixture of wild and colonization smolts, tagged smolts released from this stream were considered as an 'enhanced' group in order to distinguish them from a typical wild, production, or colonization group.

### French Creek

French Creek is located adjacent to the town of Parksville, approximately 70 km north of Nanaimo. The creek begins in the eastern slopes of Mount Arrowsmith, and flows northeast to join the Strait of Georgia at 124° 22' W., 49° 21' N. The creek is approximately 20 km long and 12 m wide at mid-section. The intertidal section is approximately one km long, 100 m wide, and characterized by the presence of several small channels meandering through mud flats, with gravel and algal growth at the periphery. A large marina and several stores are located adjacent to the mouth of the creek. Several small falls exist in the mainstem section some 7-13 km upstream from the estuary. A rudimentary fishway was constructed in 1961 to facilitate the upstream migration, but insufficient discharge and debris build-up often prevent the fish from using it. Above the fishway, 12.5 km from the estuary, the creek bifurcates into two major sections. Most of the West fork is inaccessible to adult salmon because of 12 m high falls located 400 m upstream from the confluence. The East fork meanders for a few kilometers, and then splits into numerous small tributaries scattered among the logging zones.

This creek is characterized by an abundant vegetation cover, a relatively low gradient, many pools, some erosion and bank undercutting. Large seasonal fluctuations in water levels are common, and peak discharges of up to  $125 \text{ m}^3 \cdot \text{s}^{-1}$  have been recorded during winter months. During late summer, flows usually decrease to negligible levels, and several sections in the lower reaches become completely

dry. Extensive gravel movement has been observed in lower reaches, which are characterized by relatively soft banks and a streambed of bedrock, boulders and cobble.

The physical features of the system, and particular attributes of its salmon population have been described by Hancock and Marshall (1985), Johnston *et al.* (1987), and Bocking *et al.* (1988). Prior to this study, escapement estimates were based on visual surveys conducted by fishery officers. The creek is used by anadromous populations of cutthroat and rainbow trout, as well as by four species of Pacific salmon. Historical records indicate coho and chum escapements were the most abundant, reaching up to 7500 spawners per species. The combined escapements of chinook, pink and trout amounted to less than 1000 fish. Coho escapement levels have remained below 3500 fish since 1969. Coho spawn throughout the system, but the major spawning grounds are located near the fork junctions, and adjacent to Winchester Road in the upper reaches of the system.

During the last decade, intensive logging near the headwaters, urbanization in the watershed, and reduced summer flows due to high demands are suspected to have had detrimental effects on the coho population (R. Hurst, pers. comm.). Since 1986, substantial fry mortality has been observed each summer, induced presumably by the low flow conditions. Attempts to enhance coho and rainbow populations were initiated in 1982 with the construction of a small hatchery on Miller Road. The hatchery is operated mainly on a volunteer basis by members of the Qualicum-Parksville Fish & Game club, with some financial assistance and logistic support provided by SEP. Since 1985, club volunteers have been attempting to increase the survival of the fry during the winter period by providing food (Oregon Moist Pellets) periodically to the fry populations rearing in pools throughout the lower reaches. In addition, club members have collected brood stock from the stream, and have raised the fry for subsequent release in headwaters. The available records show that only adults captured in French Creek were used as brood stock for enhancement purposes (Table 2.3). Marked fry were released mainly for habitat assessment, while unmarked fry were released for enhancement purposes. Given the lack of characteristic fin clips, the majority of these outplanted fry are not readily distinguishable from wild fry in samples collected during juvenile tagging. Therefore, the tagged smolts released from this stream were also considered as an 'enhanced' group.

### 3.0 SMOLT MIGRATION PATTERNS AND CWT RELEASE STATISTICS

Since there were no existing enumeration facilities at Black Creek, Trent River and French Creek, juvenile trapping operations were conducted each spring during 1985, 1986 and 1987 to enumerate and tag coho smolts. Only a partial enumeration of each smolt population was achieved each year, so total smolt outputs were estimated based on tagged proportions in subsequent escapements. Information on CWT releases statistics and smolt migration patterns at the remaining locations were obtained from DFO sources (Section 2.1), and were used for cross-stock and cross-year comparisons of smolt migration characteristics.

#### 3.1 Smolt trapping and tagging methods

Downstream smolt traps (Conlin and Tutty 1979) have been shown to be effective in capturing migrating smolts, and were used in the present study for enumeration and tagging purposes. Specifications concerning fence construction and trapping methods followed those outlined in Conlin and Tutty (1979), although some modifications were required to strengthen the structure, increase trapping efficiency, and adapt the fence to local conditions. In the following sections, a brief description is given of the fence designs, trapping methods, tagging procedures, and results obtained at each location.

##### Fence and trap design

Each fence consisted of several wood frame panels (1 m x 2.5 m) covered by galvanized wire mesh screen with one cm<sup>2</sup> openings. All fence panels were joined to form a 'V' or 'W' configuration across the stream. Such configurations allow each 'wing' of the fence to block the current at an angle of less than 35° to minimize hydraulic pressure on each panel and increase trapping efficiency. At each joint, panels were supported by steel rods driven into the substrate, with wooden back braces for added structural strength. Flood gates were built into several sections of the fence. These consisted of hinged panels which could be opened manually at critical times. Flood gates also facilitated cleaning operations, and could be opened to allow passage of adult trout if necessary.

At the downstream apex(es) of each fence, a trap was used to collect and hold captured smolts. Each trap was composed of rectangular boxes (2 m x 1 m x 1 m) made of plywood sheets braced externally by angle sheet metal. Internal baffles were placed inside each box to minimize exposure of trapped smolts to inflow turbulence. Several windows, covered externally by galvanized wire mesh screen and internally by soft nylon netting, were built on the sides of each box to allow sufficient water circulation. Usually, several boxes (2-4) were joined together to form a complete trap. The size and configuration of the passage linking the boxes were adjusted to separate the large trout from the smaller smolts, in order to reduce the number of injuries, the amount of scale loss, and the level of stress on the smolts.

A complete trap was connected to the apex(es) of the fence by means of a semi-rigid plastic pipe some 20 cm in diameter. Smolts concentrating at the downstream end of the fence usually seek holes to escape, and easily funnel into the pipe intake leading to the trap. Adjustments for variation in flows were done mainly by controlling the vertical angle of the pipe as it connects to the trap. Captured smolts were prevented from returning upstream by positioning the pipe at least 20 cm above the waterline in the trap, and by partially blocking the outflow with nylon screen. At each trapping site, up to six holding boxes were installed near the smolt trap to contain all fish captured.

#### Location of trapping sites

Preliminary field surveys identified potential sites for fence locations on the basis of relative flow rates, stream width, geomorphological characteristics and accessibility. Final decisions regarding fence location were made in conjunction with local authorities and DFO fishery officers.

At Black Creek, the site chosen was approximately 0.5 km from the mouth of the creek, within the Miracle Beach Park, adjacent to the road leading from the Highway 19 to the Miracle Beach Resort. The smolt trap was positioned about 40 m downstream from the bridge, with the holding boxes and tagging platform located adjacent to the old concrete abutments. This site is located beyond tidal reaches, has a stream width of 16 m, and a streambed composed of small rocks, cobble and gravel. This site was also used in previous juvenile trapping operations (Clark and Irvine 1989).

At Trent River, fences and traps were constructed approximately 400 m upstream from the Trent River bridge on Highway 19. At this location, the mainstem splits into several channels around two small islands, which lie beyond tidal reaches. The presence of numerous channels required the

construction of two fences, each operated independently as a function of the flow rates in each channel. At these locations, channel widths ranged from 20 to 50 m, with a streambed characterized by an unstable and uneven mixture of rocks, cobble and fine gravel. Substantial efforts were made to prepare each site to accommodate the fence and stabilize the streambed. The site chosen at French Creek was behind the private hatchery on Miller Road, approximately 1.5 km from the river mouth. This site lies beyond tidal reaches, has a stream width of approximately 20 m, and a streambed is composed of gravel, cobble and small rocks.

#### Installation and operation of smolt traps

Prior to the actual installation of the fence, the streambed along the path of each wing was flattened, graded, and covered with a one meter wide sheet of polyethylene, held in place by a row of sandbags. This eliminated gaps under the base of the panels, and helped to reduce hydraulic scouring of the streambed. The panels were then positioned, and nailed to one another in a straight line to distribute the water pressure evenly across each wing. As the wings were assembled, a continuous row of sandbags was placed along the base of each panel. All gaps between the sandbags were filled with gravel, and then covered with rocks. This prevented the rapid erosion of the sandbags, and ensured that smolts did not try to escape through holes in the burlap. Once the fence and trap were in position, holding boxes were installed downstream from the fence, and anchored with steel rods. Usually, all traps were in operation by late April.

Once installation was complete, each fence was monitored and maintained on a daily basis by at least one worker for the duration of the migration period. Field crews worked on site during most daylight hours. During peak migration periods, times of flooding and high water levels, field crews maintained surveillance periodically throughout the night. During the enumeration, the fence was inspected for damage and if necessary, repairs were made immediately. Predators and injured fish (if any) were removed and noted. Passage of adult salmonids and other species through the fence was facilitated and recorded. Debris build-up was adjusted if necessary to improve the trapping efficiency and minimize fish injury. Smolts caught in the trap were transported to holding boxes if excessive densities were apparent. Field crews also monitored environmental conditions and provided assistance to other ongoing DFO investigations.

Crude tests of trapping efficiency were performed sporadically at each site during the trapping periods. Each test involved the release of 50 live, coded-wire tagged smolts about 0.5 km upstream from

the fence. Throughout the subsequent trapping period, captured smolts lacking an adipose fin were passed through a tag detector (QCD), and the number of recaptures were recorded. Subsequent tests were conducted in the same fashion, but slightly different combinations of fin marks were used on each test group.

### Tagging and sampling

Coded-wire tags were applied exclusively to juvenile coho salmon exhibiting smolting characteristics (Wedemeyer *et al.* 1980), and exceeding seven cm in fork length. Tags were implanted with a Mark II injector unit from Northwest Marine Technology. Tag retention was verified by means of a Quality Control Device (QCD) supplied by the same company. Up to four distinct tag codes were used to identify each smolt population in a given year. These were alternated periodically throughout the tagging process so that approximately equal numbers of smolts were tagged with each code. The number of smolts tagged each day in a given stream was determined by the catch levels, the available manpower, predetermined tagging quotas, and the necessity to spread the tagging effort throughout the season as much as possible.

Tagging and fin clipping was performed by an experienced tagger and three assistants, using a semi-permanent marking platform installed near the trapping site. This platform supported the tagging unit, QCD, anaesthetic and recovery basins. The basins consisted of 10 l plastic containers, covered with soft nylon net to minimize scale loss and facilitate handling. Tricane Methane Sulfonate (MS 222) was used in May 1985 to anesthetize the fish, but was replaced later by 2-Phenoxyethanol, which appeared to be less toxic to the fish and easier to regulate.

Captured coho smolts were held in live boxes until sufficient numbers ( $\geq 300$ ) had accumulated to allow tagging to proceed. Captured smolts were always tagged within 36 h after capture. Tagging began by transporting smolts to the marking platform in a 20 l bucket. Small numbers of smolts ( $\leq 30$ ) were transferred from the bucket to the anaesthesia basins. All fish other than coho smolts were removed, identified, recorded, and released immediately downstream from the trap. Coho smolts exhibiting excessive scale loss ( $>10\%$ ), disease, or injuries were recorded and released without tags.

At the beginning of each tagging session, appropriate head molds and needle depths were determined. The first few smolts tagged were dissected, and if tag emplacement corresponded to that of the preferred depth, tagging proceeded. As soon as smolts exhibited disorientation (roll-over) and

reduced activity, the adipose fin was removed. Each smolt was then oriented into the head mold, tagged, funnelled through the QCD, and diverted to a separate container for recovery.

During the tagging period, the water in each basin was changed every 15 min to maintain oxygen levels, replace the anaesthetic solution, and remove mucus, scales, debris and fin pieces. Additional oxygenation or aeration was not judged to be necessary. Throughout the tagging period, adjustments were made to the tagging unit if the smolts exceeded the size range associated with the head mold/needle depth combination used. Once all the smolts of one tag group had been tagged, the untagged fish rejected by the QCD were examined and tagged again if necessary. All tagged smolts were held 6-12 h to recover, and then allowed to passively leave the recovery boxes through an opening after sunset on the day of tagging. After each tagging session, a random sample of 50 to 200 tagged smolts was kept overnight to check for the tag retention rate. Although it is recommended that tag loss be measured approximately four weeks after tagging (Blankenship 1988), all tag retention checks in the field were done within 24 h. The constraints imposed by holding juveniles at the smolt stage, under field conditions, at relatively high water temperatures and high densities, dictated this approach.

To minimize mortality levels, no attempt was made to collect smolt samples throughout the migration for subsequent lab studies. Instead, all specimens retained for this purpose consisted of smolts that died during the trapping, tagging and holding periods. As a result, samples were collected unsystematically throughout the migration period, but there was no evidence that these smolts differed in any respect from the live smolts released. All smolts collected were frozen as soon as possible, and were subsequently examined for size, weight, age, morphometric attributes, parasitic infections and scale loss. All smolts examined were weighed separately on a Mettler balance to the nearest 0.1 g. Fork lengths (FL) were measured by caliper to the nearest mm. For each smolt, five to ten scales were removed from the preferred location (Scamecchia 1979), and placed on labelled glass slides for examination.

All ageing was done by means of a Tri-Simplex scale reader, using 80x magnification. Age estimates were based mainly on the number of distinct bands of circuli observed. Smolts which showed only one large band of circuli were classified as age 1+ smolts (1 year < fresh water rearing period < 2 years), while those which showed two distinct bands were classified as age 2+. None of the smolts captured appeared to be of age 0+ or 3+. Initially, age estimates were compared to those obtained by other scale readers<sup>14</sup> in order to verify the ageing technique used. It should be emphasized that the

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<sup>14</sup> Steve Cox-Roger (International Pacific Salmon Commission), and Gary Birch (Dept. of Zoology, UBC).

ageing method used is largely a subjective process with an undetermined error component. Bilton (1984) recognized this problem, and noted that this ageing technique had not been validated for coho salmon.

### 3.2 Smolt trapping and tagging statistics

#### Black Creek

During the 1985 to 1987 juvenile trapping periods, water levels were adequate for conducting trapping operations, and full counts were obtained throughout each season. As a result, the daily catch records (Fig. 3.1) are representative of the actual migration patterns for this population. In 1985, smolt enumeration began on May 1, but during the following years, favorable conditions for fence installation allowed the enumeration to begin earlier. This revealed that migration could start as early as April 8, but on average, substantial catches ( $> 200\text{-day}^{-1}$ ) began within a two week period centered around May 1. Typically, daily catches would increase to at least 4000 smolts within the next two to three weeks. The peak migration period occurred between May 10 and May 24, with daily catches reaching up to 8800 smolts in 1986. Peak catches always occurred over a relatively short period of time ( $< 5$  d), and were not consistently high within such periods. After this period, catches decreased progressively over the next three weeks. Trapping ended between May 26 and June 4, because water levels were too low ( $< 5$  cm) to operate the smolt trap. On average, the major migration period lasted approximately 25 d. Field observations also indicated that the catch records are representative of the main migration 'pulse' each year.

Smolts were captured mostly at night until early May, and caught in equal numbers during daytime afterwards. Throughout the migration period, low discharge often required the installation of plastic sheets on the panels to maintain sufficient water levels for trapping operations. Visual observations suggested that daily smolt catches were influenced by the manual adjustment of water levels near the trap, as well as smolt densities and the abundance of large trout ( $> 30$  cm) in upstream pools. The combined influence of these factors undoubtedly contributed to the day to day variability in catches.

The observed seasonal smolt outputs were 50,208 (1985), 38,212 (1986), and 60,909 (1987). Fence calibration tests, conducted sporadically in 1986 and 1987, showed trapping efficiencies of 80% and 96% respectively. Usually, at least 50% of the test fish released 350 m upstream of the fence were recaptured within four days from release, but the remaining ones took as much as 10 d before re-

entering the trap. Based on a daily inspection of the fence condition, the smolt trapping efficiencies during each season were thought to be consistently greater than the tests indicated (~98%).

Each year during fence installation and disassembly, smolts were occasionally seen moving downstream across the shallow reaches of the creek. After fence removal, several hundred smolts were observed holding in various pools further upstream, and presumably migrated to sea later on. Since some migration occurred prior to, and after the trapping period, the smolt catches obtained undoubtedly underestimated the actual smolt output. Based on the number of smolts observed in the stream after fence removal, it appeared that the actual output could exceed the total catch by at least 15% each year.

Tagging was conducted mostly during the early part of the migration period in 1985, and during the early and middle parts in 1986 and 1987. Smolts captured at the end of the trapping period were never tagged due to the excessive stress induced by high water temperatures and low flows. A highly variable portion of the smolts caught each day were tagged. During 1985, various logistic constraints and low tagging quotas allowed for only 47.6% of the total catch to be tagged. In 1986, smolt output was substantially lower, and 82.8% of the smolts caught were tagged. Additional tagging effort in 1987 was compensated by higher smolts output, and only 58.5% of the smolts caught were tagged.

The average migration time of the smolts captured was nearly identical to that of the smolts tagged (Table 3.1). However, tagging smolts predominantly during the peak migration period induced differences in the migration period between the groups. The mean angular deviation<sup>15</sup> of migration time for the tagged group was always one or two days less than that of the captured group. As a result, statistical differences in migration time were detected between groups each season (Kuiper's test,  $P < 0.001$  each year). Therefore, in a strict sense, tagged groups were not entirely representative of their respective populations in terms of migration time.

Tag retention was consistently about 99.0%. Smolt mortality due to trapping and tagging was negligible each season. Total mortality due to trapping, marking and holding was estimated at less than 0.5%. Mortality due to predation was not quantified, but was noticeable. Minks, otters and kingfishers were commonly observed chasing and catching smolts in pools above the fence. In addition, minks, lampreys and large trout occasionally entered the traps at night, and caused considerable smolt injuries and mortalities. Approximately 10% of the untagged fish released each day showed predator injuries or deformities.

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<sup>15</sup> This measure ( $s$ ) is analogous to the standard deviation of linear data, and ranges from 0 to 81.03 (Zar 1984).

The number of smolts collected for sampling purposes during each season increased from 115 in 1985, to 257 in 1987. Preliminary examination of smolt size data revealed that some temporal variation in body length occurs during the migration (Fig. 3.4). During each season, smolts captured within distinct time periods exhibited significant differences in mean fork length (Kruskal-Wallis test,  $P < 0.01$ ). Each year, the mean length was highest early in the season, near average around May 10 (calendar day 130), and smallest towards the end of the trapping season. As the season progressed, the average length of the smolts decreased by 17.5% over a 20 d period in 1985, and by 13.5% and 14.2% over a 30 d period in 1986 and 1987 respectively.

Given the temporal variation in body length, and the non-proportional sampling regime used during the study, weighted samples of length measurements were used to generate estimates of mean body size for each cohort. The estimation procedure was based on a stratified random sampling design, with each time interval considered as a distinct stratum. As suggested by Cochran (1977), stratified estimates of the mean fork length ( $FL_{str}$ ) for each population were calculated:

$$(Eq. 3.1) \quad FL_{str} = \sum_{h=1}^H \frac{\text{catch}_h}{t.\text{catch}} \cdot FL_h$$

with the associated variance estimate:

$$(Eq. 3.2) \quad V(FL_{str}) = \sum_{h=1}^H \left( \frac{\text{catch}_h}{t.\text{catch}} \right)^2 \cdot V(FL_h)$$

where:  $\text{catch}_h$  = Smolt catch obtained during time interval h;  
 $t.\text{catch}$  = Total smolt catch obtained during the season;  
 $FL_h$  = Mean fork length of the smolts sampled during time interval h;  
 $V(FL_{str})$  = Variance of stratified mean fork length.

Weighted estimates of mean fork length (Table 3.2) were lower than the unweighted means obtained by averaging the mean sizes over all time periods (Fig. 3.4). The mean fork length of the 1985 cohort was found to be significantly smaller than that of the 1986 and 1987 cohorts (Tukey Test,  $P < 0.001$ ). Examination of the residuals from linear regressions of smolt weight against fork length indicated a curvilinear relationship between these variables. As noted by Ricker (1975), the weight-length relation is usually expressed as a power function of the type  $W = aL^b$ . Given weight (W) measurements in g, and fork length (FL) measurements in cm, the coefficients a and b were estimated

from the simple linear regression of logarithms of weights against lengths (Table 3.4). Significant differences were detected among years (Analysis of covariance,  $P < 0.001$ ). Significant difference was not detected between the slopes of the regressions (Tukey test,  $P > 0.05$ ), but was detected between the elevations of the regression (1985 < 1986 and 1987, Tukey test,  $P < 0.05$ ), indicating that smolts caught in 1985 were leaner than those caught in 1986 and 1987.

In 1985, only 30 smolts were aged from scale samples, but this number increased to 153 in 1986, and 206 in 1987 (Table 3.6). Scale examination indicated that Black Creek outmigrants consist mostly of age 1+ smolts. The contribution of age 2+ smolts to each sample was 3% in 1985, 8% in 1986, and 5% in 1987. Omitting the 1985 sample (too few cases), age 2+ smolts migrated, on average, approximately three days before age 1+ smolts, but the difference was not statistically significant (Watson  $U^2$  test,  $P > 0.05$  for both years). The mean length and weight of age 2+ smolts was consistently larger than that of age 1+ smolts, but the difference was statistically significant only for the 1986 sample (Mann-Whitney test,  $P = 0.001$ ). The considerable overlap in length and weight of age 1+ and 2+ smolts indicates that the groups are not distinguishable on the basis of size alone.

The results obtained were very similar to those reported by Clark and Irvine (1988), and Fielden *et al.* (1989). The 1978, 1979 and 1988 smolt migrations observed during those studies occurred predominantly during the late April to early June period, and peaked within the May 10 to May 24 period. Mean smolt sizes and weights during 1978, 1979, and 1988 were estimated at 115, 111 and 113 mm, and 15.6, 14.8 and 14.5 g respectively. During 1978 and 1979, age 1+ smolts comprised over 96% of the smolts aged each year, had an average fork length of 100 to 115 mm, and an average weight of 15.6 to 14.5 g. Age 2+ smolts had an average length of 123 to 142 mm, and a weight of 17 to 28 g. Early migrants were significantly larger and heavier than late migrants in 1978 and 1988, but not in 1979. Smolts lacking an adipose fin at capture accounted for 0.02% to 0.03% of the smolts examined in 1978 and 1979. Such results indicate that the timing of the migration, as well as the condition and composition of the smolt population are fairly consistent from year to year in this stream. However, the average length and weight of the smolts in 1985 still remains well below the overall average, and both size measurements are lower than the corresponding means for other seasons.

### Trent River

During the 1985-1987 smolt enumeration periods, water levels were highly variable, and flood conditions occurred at least once every season (Fig. 3.2). As a result, light to moderate fence damage

was experienced each season, and only partial smolt counts were obtained. In 1985, smolt enumeration began on May 1, but relatively high water levels prevented field crews from completing the installation of the fence before May 26. However, field observations suggested that a near complete enumeration was obtained on May 5, 6, 7, 16, 19 and 20, while the fence was being modified. More favorable conditions during the following seasons allowed for a complete enumeration to begin by April 25. This was sufficient for tagging purposes since substantial catches ( $>100\text{-day}^{-1}$ ) never occurred before May 1.

During 1986-87, substantial catches were obtained for a continuous period of 30 d, indicating a fairly extended migration period for this population. Catches during the migration period were highly variable, and often fluctuated between high and low levels on a day-to-day basis. Peak catches never exceeded  $1700\text{-day}^{-1}$ , and always occurred between May 12 and May 25. Peak migration dates were not always clearly identifiable, due to the presence of several periods of high catches, and the occurrence of floods which prevented complete enumeration. Trapping was terminated between June 7 and June 15 because stream discharge was too low to operate the trap. Each year, after the fence was removed, field observations revealed the presence of hundreds of smolts remaining in pools upstream from the trapping site. The proportion of the remaining smolts that eventually migrated to sea could not be determined on the basis of subsequent visual surveys.

During the migration period, smolts were caught throughout the day and night, with the largest catches occurring just before dawn. Plastic sheets were required to maintain sufficient water levels only during the last three weeks of each season. Visual observations suggested that daily smolt catches were not affected by the manual adjustment of water levels near the traps. Perhaps the absence of holding pools in the vicinity of the fences contributed to this, since smolts committed to move downstream over shallow reaches had no place to seek refuge under conditions of relatively low water levels.

The observed seasonal smolt outputs were 12,952 (1985), 11,388 (1986), and 17,635 (1987). Fence calibration tests were only conducted in 1986 and 1987, and showed trapping efficiencies of 92-100%. Usually, at least 50% of the marked smolts released 300 m upstream from the fence were recaptured within four days, but some smolts took up to 16 d to re-enter the trap. Based on daily inspections of the fences, the efficiency estimates obtained from the tests were thought to be representative of the actual performance of the trap, except during periods of flooding. Given that some smolts escaped undetected during floods, as well as during the periods before and after the enumeration, the observed smolt outputs were lower than the actual outputs. A subjective assessment based on field observations indicated that the actual output exceeded the observed output by at least 25% in 1985 and

1986, and 20% in 1987. Field observations also suggested that the catch statistics are representative of the main migration 'pulse' each year, at least in terms of the temporal patterns of migration.

Due to logistic difficulties, tagging was conducted mostly during the middle and late part of the migration in 1985, and only 69% of the smolts caught were tagged. During 1986 and 1987, tagging was conducted throughout the entire enumeration period, and approximately 90% of the trapped smolts were tagged. Late migrants were always tagged since the water temperature never reached stressful levels ( $>16^{\circ}$  C.), and holding densities were always low.

Tagged smolts migrated at a different time than their respective populations during 1985 and 1987, but the difference in the median date of smolt migration was less than three days (Table 3.1). Differences in the variability of migration time of both groups, as measured by the mean angular deviation, was always less than one day, with the variation being always slightly larger for the enumerated group. Such minor differences were sufficient to induce statistical differences in migration time between the groups within each season (Kuiper's test,  $P < 0.001$  each year).

The mean tag retention each season ranged from 99.0% to 99.9%. Estimates of overall smolt mortality due to trapping, marking and holding were always negligible ( $< 0.5\%$ ). Mortality due to predation in the vicinity of the trap was not quantified, but was less noticeable than at Black Creek. Minks and kingfishers were rarely seen, and only a few mergansers were occasionally observed feeding in tidal waters. Less than 5% of the fish released untagged each day showed distinct predator injuries. However, the majority of the smolts released untagged exhibited physical abnormalities (backbone deformity, abnormal jaws, white pupils, and scale loss).

As noted in Section 2.2, only a portion of the hatchery fry released in the headwaters in 1985 and 1986 were fin clipped. The relatively low number of marks applied served as the basis for a crude assessment of enhancement activities in this system. Fin clipped smolts made up 34% of the total smolt catch in 1986, and 28% of the total catch in 1987, indicating a substantial contribution of hatchery releases to the total smolt output (Table 3.1). On average, about 92% of the fin clipped smolts captured in 1986 and 1987 consisted of fish released in Bradley Lake. For this group, the survival rate from fry to smolt, averaged over both years, was 16.5%. The survival rates of fry released in headwaters (1985), and in Bloedel Creek (1986) were much lower, with both rates between 5% and 6%. It should be emphasized that the actual survival rates of fin clipped fry are probably higher than the rates reported above, since the smolt catches obtained in this study underestimate the true smolt output each year.

The contribution of the 1984 fry release to the 1985 smolt catches is unknown given the absence of characteristic fin clips and the uncertainty associated with the total smolt output. Fry released in 1984 were about half the size of those released in 1985. Assuming that the 1984 releases survived at half the rate as their corresponding 1985 releases, then the 1984 fry release produced 5160 smolts, which represents 40% of the total smolt output of 1985. Assuming that the fry released into Bloedel Creek during 1985 survived at the same rate as those released there in 1986, then this group made up about 58% of the total smolt output in 1986. Assuming that no mortality operated between fry release and subsequent fin clipping at Bloedel Creek in 1986, then the fry released in 1986 made up 47% of the total smolt output in 1987. These figures suggest that the fry released by the Fish & Game club accounted for about half of the total output each year. However, it must be emphasized that these crude estimates are based upon a large number of questionable assumptions.

Significant differences between the migration time of clipped and unclipped smolts were detected for both seasons (Kuiper's test,  $P < 0.001$ ). During 1986 and 1987, clipped smolts showed a tendency to leave the stream a few days later than unclipped smolts (Fig. 3.8). The difference was most pronounced in 1986, when the median date of departure of the unclipped group was seven days earlier than that of both fin clipped groups. There was greater similarity between the migration time of LV and RV<sup>16</sup> smolts than between clipped and unclipped smolts (Kuiper's test).

The number of smolts collected for sampling purposes during each season increased from 88 in 1985, to 184 in 1987. In addition, extensive sampling of unclipped and clipped smolts was conducted in 1986 on a bi-weekly basis by R. Hurst, during which an additional 3712 smolts were measured. Samples from both surveys in 1986 were combined to produce a more accurate description of the physical attributes of the population. Examination of the length data revealed that the size of outmigrants varied considerably during the migration period. Unclipped smolts captured within successive 10 d periods exhibited a small but significant decrease in mean fork length during 1986 and 1987 (Kruskal-Wallis test,  $P = 0.0002$ ,  $P = 0.008$ ). As the season progressed, the average length of the smolts decreased by 5% over a 50 d period in 1986, and by 7% over a 30 d period in 1987. By contrast, smolts caught in 1985 exhibited a small, but non-significant increase in average body length as the season progressed. Fin clipped smolts did not exhibit the same decrease in body length as the season progressed (Fig. 3.6). During 1986, fin clipped smolts captured within successive time periods showed no significant decrease in mean fork length (Kruskal-Wallis test,  $P > 0.10$  for LV and RV groups). Smaller sample sizes obtained in 1987 showed similar patterns for LV smolts (Kruskal-Wallis test,  $P >$

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<sup>16</sup> LV and RV refers to marked fish lacking a left ventral or right ventral fin. Such marks were only applied to some of the fry released into the stream for enhancement or experimental purposes.

0.50). Given that a portion of the unclipped smolts captured each year were probably outplanted fry, the decrease in body size of unclipped smolts observed during 1986 and 1987 may be less pronounced than that exhibited by smolts of natural origin.

Samples of length measurements for each group in a given year were aggregated according to Eq. 3.1 to compensate for non-proportional sampling. The effects of stratification upon the estimates of mean fork length were negligible. Therefore, unweighted estimates of mean fork length were used to facilitate comparisons between groups (Table 3.3). During 1986 and 1987, LV smolts were larger than unclipped smolts, and RV smolts captured in 1986 were the smallest. The mean size of unclipped smolts was largest in 1986, near average in 1987, and lowest in 1985.

During 1985 and 1987, fin clipped and unclipped smolts were sampled roughly in proportion to their abundance in the catch, but R. Hurst sampled predominantly fin clipped smolts during 1986. To obtain a representative sample of length measurements for the 1986 smolt population, random subsamples of length measurements from each set of samples for a given group were obtained in proportion to the group's contribution to the 1986 catch. A comparison of the within season variation in smolt size exhibited by each cohort showed no consistent reduction in smolt size during each season (Fig. 3.7). Weighted estimates of mean length for each cohort were generated according to Eq. 3.1 (Table 3.2). For the 1986 cohort, weighted estimates of mean length were nearly identical to unweighted estimates. The average smolt size was highest in 1986, lowest in 1985, and near average in 1987. The maximum year to year change was observed between 1985 and 1986, during which the average length increased by 16%.

Only a fraction of the fish measured in 1986 by R. Hurst were weighed. These weight measurements were combined with those collected during this study to estimate weight to length relationships for the unclipped, LV and RV groups (Table 3.5). Significant differences were detected among the groups in 1986 (Analysis of covariance,  $P < 0.001$ ). Significant differences were not detected between the slopes of the regressions (Tukey test,  $P > 0.05$ ), but were detected between the elevations of the regression for the unclipped group and those of the LV and RV groups (Tukey Test,  $P < 0.05$ ). This indicates that clipped and unclipped smolts of a given length differed in weight, with the former groups being significantly leaner than the unclipped group. Weight to length relationships were also determined for LV and unclipped smolts captured in 1987 (Table 3.5). Significant differences in elevation and slope were detected between these regressions (t-test for slopes and elevations,  $P < 0.05$ ). The average condition factor of LV smolts was significantly lower than that of the unmarked smolts (Mann-Whitney test,  $P < 0.001$ ), with the difference being about 6.5%.

To best describe the outmigrating smolt population each year, random sub-samples of length and weight measurements were obtained from each set of measurements, for a given group, in proportion to the group's contribution to the total catch that year. The resulting sample of measurements was used to determine the weight to length relationship for each cohort (Table 3.4). Significant differences were detected among the groups (Analysis of covariance,  $P < 0.001$ ). Significant differences were not detected among the regression slopes (Tukey test,  $P > 0.05$ ), but were detected among the regression elevations (Tukey Test,  $P < 0.05$ ), which indicates that smolts of a given length exhibit year to year differences in weight.

In 1985, only 22 smolts were aged from scale samples, but this number increased to 122 in 1986, and 183 in 1987. Only one smolt captured in 1986 was determined to be of age 2+, while all others examined were of age 1+. This smolt measured 102.6 mm and weighed 11.8 g, which is very similar to the average smolt size observed during 1986. These results suggest that age 2+ smolts are rare in this river, and probably account for less than 1% of the smolts emigrating each year.

Fielden *et al.* (1989) determined that the 1988 Trent River outmigrants averaged 103.9 mm in fork length, and 11.76 g in weight. Both estimates are slightly larger than the largest average size observed during this study, with the average size and weight of the 1985 outmigrants remaining well below the overall seasonal average. The average fork length of the outmigrants each year was correlated with the average weight of the fry outplanted in the previous year (Spearman's rank correlation,  $r_s = 1.0$ ), indicating that the condition of the fry released in the Trent River may influence the condition of the smolt population. As was observed during this study, the peak migration period during the 1988 season occurred between the second and third week of May.

### French Creek

Smolt enumeration was not conducted during 1985 at French Creek because of concerns raised by DFO staff about the proposed trapping program. However, trapping and tagging operations were conducted as planned in 1986 and 1987. During both seasons, water levels were adequate for enumeration purposes, and full counts were obtained each day. As a result, the catch patterns presented (Fig. 3.3) are representative of the actual migration patterns for this population.

During 1986 and 1987, smolt enumeration began around mid-April. By then, the migration was already in progress, as indicated by the small catch rates obtained. Smolt catches increased progressively thereafter, and reached  $100\text{-day}^{-1}$  during the week centered around May 1 when the water temperature reached  $10^{\circ}\text{C}$ . Peak catches of  $\approx 2000\text{-day}^{-1}$  occurred only twice each season within the May 15-27 period, but catches  $>1000\text{-day}^{-1}$  were obtained frequently within the three week period centered around the peak period. During the two weeks following this peak, catches usually decreased progressively as did the water levels. Each season, trapping was terminated between June 6-12 because stream discharge was too low to operate the trap. The duration of the main migration period was about the same each year, or approximately 40 d. The median date of migration during 1987 was eight days earlier than in 1986 (Table 3.3).

Throughout the migration period each year, smolts were captured almost exclusively at night and at dawn. Daytime catches were obtained only occasionally during the last part of each season, when water temperatures were  $\geq 13^{\circ}\text{C}$ . Plastic sheets were rarely used in 1986 to maintain water levels in the vicinity of the trap, but were required throughout 1987 because of unseasonably low water levels. Visual observations indicated that manual adjustments of water levels near the trap did not appear to influence the migration pattern of the smolts. Smolts seemed to be more responsive to natural increases in stream flow, the smolt densities in upstream pools, and the presence of predators near the trapping area.

Over the two year period, smolt output increased slightly from 29,155 to 33,917 (Table 3.3). A single fence calibration test was conducted during each season, with both tests indicating a trapping efficiency of 100%. During 1986, it took 18 d to recapture all 25 marked smolts released upstream, compared to nine days in 1987. During both tests, 50% of the marked fish were recaptured within three days after release. Based on a daily inspection of the fence, the efficiency estimates obtained from the tests were thought to be representative of the actual performance of the trap. However, each year, smolts undoubtedly escaped undetected prior to completing the fence installation, as well as after the removal of the fence. A subjective assessment based on visual surveys indicated that the actual output exceeded the observed output by at least 10% in both years.

During both seasons, tagging was conducted throughout the migration period, and was centered around the peak migration period. Smolts caught during the first week were not tagged due to logistic constraints, while those caught at the end of the season were released untagged due to insufficient flows through the holding boxes. Approximately 24,000 smolts were tagged each season, representing about 82% of the 1986 catch, and 72% of the 1987 catch. The median dates of migration of the enumerated

and tagged groups were identical each year. Given that smolts captured at the tail ends of the migration period were not tagged, the migration period of the tagged group was always shorter than that of the enumerated group. This was sufficient to induce significant differences in migration time between groups within each season (Kuiper's test,  $P < 0.001$ ), meaning that in a strict sense, marked groups are not entirely representative of their respective populations.

Seasonal averages in tag retention rates exceeded 99% each year. Estimates of overall smolt mortality due to trapping, marking and holding were always negligible ( $< 0.5\%$ ). Losses due to predation in the vicinity of the trap was not quantified, but did not appear to be a serious problem. Muskrats, kingfishers, and mergansers were occasionally seen catching fish in the various pools 100 to 200 m downstream from the traps, but rarely in the vicinity of the fence. Less than 5% of the fish released untagged each day showed distinct predator injuries, but nearly all of them exhibited physical abnormalities (crooked backbone, white pupils, deformed eyeballs, scale loss, jaw injury, etc.).

As noted in Section 2.2, a small portion of the fry released by the Fish & Game club each year were fin clipped. Information on the catches of fin clipped smolts was used for a preliminary assessment of the success of enhancement activities in this stream. Fin clipped smolts made up 2.6% of the total smolt catch in 1986, and 2.7% of the 1987 catch. For Dudley marsh releases, delaying the release date from May to October, and increasing the size of the fry from one to six grams was accompanied by an increase in survival rate from 7.1% to 9.9%. It is hypothesized that the survival rates of the 1985 releases were slightly lower than the norm, given that unusually high mortality was detected during the freezing period following the transplant (R. Hurst, pers. comm.).

The conditions in Dudley marsh differ substantially from those in the main section of the stream, and the survival rates of fry released in each area are probably not comparable. Given the absence of a characteristic fin clip, and the lack of comparable releases of marked fry in the past, it is impossible to estimate the survival rates of the fry released into the mainstem, or their contribution to the total smolt outputs. In view of the year to year variability in survival rate of Dudley marsh releases, no attempt was made at estimating the survival rates and contribution of the mainstem releases.

During both seasons, smolts released in Dudley marsh outmigrated much earlier than those from the mainstem (Fig. 3.9). The median date of migration of the groups was markedly different, with the Dudley marsh releases leaving 19 d earlier in 1986, and 16-17 d earlier in 1987. Each year, by the time the migration of the unclipped group was half complete, over 90% of the smolts from Dudley marsh had migrated past the fence. Within the Dudley marsh group, one gram fry released in May began their

migration earlier than those released as six gram fry in October. Many smolts from the first group were captured as soon as the trap was operational, which raises the possibility that a significant portion of this group migrated to sea before the fence installation was complete.

The number of smolts collected for sampling purposes increased from 163 in 1986, to 250 in 1987. Examination of length data revealed that the average size of smolts captured changed during the migration each year (Fig. 3.5). Unclipped smolts captured within successive 10 d periods exhibited significant differences in average fork length during both years (Kruskal-Wallis test,  $P < 0.001$  for 1986 and 1987). As the season progressed, the average length decreased by 5% over 30 d in 1986, and by 17% over 40 d in 1987. It is not known if fin clipped smolts exhibited similar trends, as the samples were too irregularly spaced, and were not sufficiently large for comparative purposes.

Samples of length measurements taken each year were aggregated according to Eq. 3.1 to compensate for the non-proportional sampling regime used (Table 3.2). The mean fork length of the 1987 cohort was only 2.5% greater than that of the 1986 cohort, but the difference was significant (t-test,  $P < 0.05$ ). Fin clipped and unclipped smolts could not be statistically compared for size differences since fin clipped smolts were not sampled in 1986, and only a single fin clipped smolt was measured in 1987. Weight to length relationships were determined for the 1986 and 1987 cohorts (Table 3.4), but could not be determined specifically for fin clipped groups because of insufficient samples. Significant differences were detected between the slopes and elevations of the regressions describing the cohorts (t-test,  $P < 0.05$ ), indicating year to year changes in smolt condition.

All smolts sampled in 1986 were aged, as well as 237 smolts collected in 1987. The vast majority of the smolts examined were age 1+. The proportions of age 2+ smolts were 0.6% and 2.5% respectively. Age 2+ smolts were usually larger and heavier than age 1+ smolts, but could not be identified on the basis of size alone, because both age groups overlapped considerably in length and weight (Table 3.6, 1987 season).

Fielden *et al.* (1989) determined that the 1988 French Creek outmigrants averaged 101.1 mm in fork length, and 10.2 g in weight. Significant differences in fork length were detected between cohorts (ANOVA,  $P < 0.01$ ), with the 1988 outmigrants being significantly larger than the 1986 and 1987 outmigrants (Tukey test,  $P < 0.05$ ). No correlation was found between the average length of the outmigrants, and the size of the fry released in the stream during the preceding year. However, it should be noted that the time of release varies considerably from year to year, which makes it difficult to determine the exact effects of size at release on the condition of the smolts produced. Peak migration in

1988 occurred during the third week of May, which is in the middle of the peak period for 1986 and 1987.

### 3.3 Estimation of actual smolt outputs

Since a complete enumeration of all outmigrants was never achieved during this study, the actual smolt outputs of each stream must be estimated to assess the actual trapping efficiency, and the level of similarity between the migration patterns of the tagged and untagged smolts. Actual smolt outputs are often estimated on the basis of trap efficiency tests (De Hrussochy-Wirth 1979). However, it should be emphasized that such tests provide only crude estimates of the minimum proportion of all smolts present that moved downstream during the test period. Marked smolts released upstream may be preyed upon, may be reluctant to re-enter the trap, may exhibit abnormal behavior and delay their migration due to stress. During past studies in the Quinsam River (Quinsam Coal Project), some marked fish released for testing purposes were captured at the same site 12 months after release (B. Lister, pers. comm.). Similar results have been obtained with coho smolts released above the fence on the Keogh River, and at Black Creek during 1989 (Dr. J. Irvine, pers. comm.<sup>17</sup>). These facts highlight the need for an alternative indicator of trapping efficiency.

In the present study, actual outputs were determined by making use of the proportion of tagged fish in the subsequent escapement ( $p(\text{tagged})_{esc}$ ), after accounting for the presence of strays in the escapement and differential mortality of tagged and untagged fish induced by the trapping and tagging operations (see Section 5.2):

$$(Eq. 3.3) \quad \text{Smolt output} = \frac{\text{Tagged smolts released}}{p(\text{tagged})_{esc}}$$

A comparison of the tagged proportions among juveniles and adults revealed that the former was usually greater than the latter (Table 3.7). Major discrepancies are apparent for the 1985 migration at Trent River and Black Creek, suggesting that only 16.7% and 43.5% of the respective smolt outputs were enumerated at each location. This suggests that relatively high outputs would have occurred in these streams during 1985, which may have been the cause for the significantly lower mean size and condition factors observed, presumably induced by limited resources and increased competition in freshwater. Interestingly, smolt output at the Little Qualicum River was also unusually high in 1985,

<sup>17</sup> Department of Fisheries & Oceans, Pacific Biological Station, Nanaimo, B.C.

which supports the notion that relatively high smolt outputs may have occurred in several natural systems on the east side of Vancouver Island that year. In the absence of any evidence to discredit this hypothesis, it must be assumed that the smolt output figures for 1985 are correct.

However, the nature of the factor(s) responsible for the major discrepancies between the observed and estimated outputs could not be identified with certainty. The 1985 trapping operations at Black Creek and Trent River were conducted under conditions similar to those of 1986 and 1987, during which an apparently larger and more constant fraction of the smolt population was enumerated. The only major difference in operation between 1985 and the other years concerned the use of MS222 as the main anaesthetic at both locations in 1985 (instead of 2-Phenoxyethanol), which may have induced a greater mortality among the smolts tagged and led to an overestimation of total output. However, juvenile trapping and tagging at Black Creek during 1985 were conducted under the same conditions, by the same tagger (D. Clark<sup>18</sup>), with the same anaesthetic (MS222), as in 1978 and 1979, without similar consequences. The possibility remains that during 1985, a large portion of the smolt population at Black Creek remained upstream after the fences were removed, and migrated to sea during the following freshets. This would indicate that the tagged smolts released at Black Creek in 1985 are more representative of the early migrants than the middle and late migrants. Assuming that approximately 55% of the smolt populations migrated to sea after May 23 suggests that the actual median migration date for this population was around May 24 that year.

At Trent River, given that floods occurred during the main migration period when trapping was conducted, and that trapping was terminated before the end of the migration period, it is assumed that most undetected smolts migrated to sea during floods and during freshets following the fence removal. This would indicate that the tagged smolts released at Trent River were more representative of the late migrants in 1985, but were representative of the main outmigration pulse in 1986 and 1987. Based on the observed migration pattern and the periodicity of the floods, it is estimated that the median date of smolt migration during 1985 was approximately May 21.

With regard to French Creek, no floods occurred during the trapping period so it is assumed that the large number of smolts that migrated to sea undetected during 1986 moved downstream before and after the trapping period. This would suggest that the median date of migration of the smolt population was probably identical to the date estimated previously, and that the tagged group was mainly representative of the middle segment of the smolt population.

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<sup>18</sup> Harbor City Trade School, Box 113, Nanaimo, B.C.

The estimated output figures indicate that smolt production in these streams can be quite variable from one year to the next, even if no enhancement is conducted. In the absence of accurate escapement records prior to 1984, and information on the exact contribution of the enhancement programs, smolt output per female spawner cannot be estimated with certainty for French Creek and Trent River. For Black Creek, the estimates based on the 1984-1987 spawner abundance figures (Table 4.15), and the associated 1986-1988 smolt output figures, indicate that production ranged from 16.3 to 33.6 smolts per female (1986, 1988). However, using unpublished estimates of smolt output for 1989 (R. Bocking<sup>19</sup>, pers. comm.), an estimate of 76.5 smolts per female was obtained. This figure is substantially greater than those obtained during the present study, and may indicate a potential smolt productivity level that can be achieved under conditions of low fry densities, brought about by the lowest escapement levels observed since 1984. It should be noted that none of the above estimates was associated with relatively high escapement levels as reported for years previous to this study.

Estimates of total smolt output and stream size were used to provide preliminary estimates of the rearing potential of each stream on a per kilometer basis. For Black Creek, smolt output per km per year ranged from 4440 in 1985, to 1587 in 1986, and averaged 2552 across all years. At Trent River, the mean was 1456 with a range of 963 to 2590, and at French Creek, the mean was 1927 with a range of 1331 to 2798. These figures should be interpreted with caution, since the area of habitat available is not known with certainty. Although systematic surveys were not conducted for this purpose, field observations revealed substantial changes in the availability of rearing habitats in all streams during this study. During the very cold period in the fall of 1985, entire stream sections normally used as spawning grounds were frozen for a few weeks. Small pools and shallow reaches at French Creek normally used as overwintering grounds were frozen, and large numbers of dead fry were found in such areas after the ice had thawed (R. Hurst, pers. comm.). This weather anomaly was also accompanied by unusually dry summers in 1985 and 1986 when entire sections of stream dried up totally.

Marshall and Britton (1990) reported an average carrying capacity of 1782-1894 smolts per km for streams 20-30 km in length, from Washington and southern British Columbia. The mean smolt output estimated for Black Creek and French Creek was above the average reported for streams of similar sizes. It remains to be determined if such high smolt output reflects the quality of the habitat, particular attributes of the resident stock, the effects of enhancement procedures, or a combination of these factors. By contrast, the smolt output per km observed at Trent River is below the estimate

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<sup>19</sup> LGL Ltd., 9768 Second street, Sidney, B.C.

reported by Marshall and Britton (1990) for streams of similar length. The scarcity of good rearing and overwintering habitat in the main section undoubtedly has some major influence, and may always hamper attempts to enhance the coho population by fry outplants, as opposed to habitat alterations.

### 3.4 Comparisons of CWT release patterns across all systems

Coded-wire tag release records, smolt migration patterns and size distributions of the tagged groups used in the present analysis are summarized in Tables 3.8 - 3.10. Examination of the migration patterns of each tagged group (Fig. 3.9) shows that each year, hatchery production groups entered the ocean over a shorter time period (< 2 weeks) than colonization, enhanced and wild groups (>2 weeks). This was attributed to the fact that production groups are usually forced out of the raceways at an advanced stage of maturity, as opposed to the other groups which migrate to sea according to their physiological readiness and environmental conditions. Among the production groups, those from the Puntledge hatchery entered the ocean over a longer period of time than those from other hatcheries, perhaps due to the greater distance smolts must travel to reach the ocean. Production groups from the Big Qualicum hatchery always entered the ocean earliest, and those from the Quinsam River were the latest. The largest difference between the median date of ocean entry for production groups was about 10 d during 1987.

By contrast, the non-production groups overlapped considerably with each other, and tended to have similar median dates of entry within a given year. Colonization groups tended to enter the ocean over a slightly longer time period than enhanced and wild groups, mainly because these latter groups were rarely tagged throughout the entire smolt migration period. The largest discrepancy in median entry date observed among these groups was between the Trent River and Black Creek smolts in 1985 (16 d).

The apparent year to year consistency migration patterns of colonization fish from the Puntledge River (Fig. 3.9) is because each year's pattern was assumed to correspond to the pattern observed in 1989. Given the large uncertainty associated with the exact pattern of migration, this group was not used for further comparisons. Among the remaining non-production groups, statistical difference in the median date of ocean entry was detected in 1985, but not during 1986 and 1987 (Median test,  $\alpha = 0.05$ ). Such results are to a large extent attributed to the fact that tags were not applied proportionally throughout the entire 1985 migration period (Section 3.2), but were applied in this fashion during 1986 and 1987. On an overall basis, the median date of entry of the non-production groups was identical in

1985 and 1986 (May 25), but was considerably earlier in 1987 (May 15). Production groups also followed the same pattern, with the median dates of ocean entry being May 22, 25 and 16.

Sample sizes for length and weight measurements obtained from various smolt populations ranged from 100 to 2000, but were generally at the low end of this range. When necessary, length-weight data sets were randomly subsampled in order to have equal size samples of 100 measurements from each population for statistical comparisons. The majority of the frequency distributions of lengths and weights, obtained prior to and after subsampling, were not normally distributed (Lilliefors test,  $P < 0.05$ ). Normal distributions were not obtained even after log transformation of the data, mainly because the distributions were generally not unimodal. Therefore, non-parametric methods were relied upon for contrasting smolt sizes among tagged populations. Significant differences in average length, and weights, were detected among populations each year (Kruskal-Wallis test,  $P < 0.0001$  for both traits each year). Tukey type multiple comparison tests (Zar 1984, p. 199), were used to group populations that were not statistically different in mean length or weight in a given year (Fig. 3.10, 3.11). Smolts from hatchery production groups tended to be larger and heavier than non-production groups each year, and those of enhanced and wild groups tended to have the smallest mean lengths and weights. Smolts from the Rosewall Creek hatchery were significantly smaller than those from other hatcheries, because feeding regimes were controlled to ensure that their size at release would approximate that of their wild counterparts. Pronounced year to year changes in size were only noticeable at Black Creek. As a result, smolts from this stream were classified as being amongst the smallest in 1985, but were grouped with the largest smolts from production groups in 1987.

As suggested by Ricker (1975), Fulton's condition factor ( $100 W \cdot L^{-3}$ ) was determined for each population from the linear regression, without a constant, of weights (in g) against the cube of the fork lengths (in cm). Regression coefficients were contrasted by means of Tukey type tests for multiple slope comparisons (Zar 1984, p. 302). Regression coefficients that were not significantly different were grouped (Fig. 3.12). Wild smolts from the Little Qualicum River were usually heavier for a given size than most other groups, while colonization groups tended to have the smallest CF each year. Except for the Big Qualicum production group, no substantial changes in CF were observed between 1986 and 1987 ( $r^2 = 0.63$ ). Condition factors in 1985 tended to exhibit greater variability than their 1986-1987 counterparts. For the Quinsam production groups, this was attributed to the fact that there was a greater difference in size between the various groups released in 1985 than between the groups released in 1986-1987 (Tables 3.8 to 3.10). This factor may also account for the large variability in CF of the colonization groups since these fry are initially reared with production groups.

The nature of the factor responsible for the relatively large variation of CF for the Black Creek, Trent River, and Little Qualicum River smolts in 1985 could not be determined with certainty. Given that this phenomenon was observed in several streams, it is hypothesized that environmental conditions may have been responsible for inducing large variation in CF reported. It should be noted that the average smolt size at Black Creek and Trent River was smallest in 1985, and that smolt output for each stream was higher than in 1986 and 1987. Thus it is hypothesized that higher rearing densities may have induced the large variation in CF within these populations.

## 4.0 RUN PATTERNS AND ESCAPEMENT ESTIMATION

To calculate exploitation rates and determine the influence of run timing on exploitation patterns, accurate information is required on the upstream migration patterns, escapement levels, and total tag returns to each system. In the absence of permanent adult enumeration facilities at Black Creek, Trent River and French Creek, escapement enumeration and CWT recovery were conducted by means of semi-permanent adult fences, mark-recapture operations and stream surveys. Escapement estimates and CWT recovery date for the remaining systems were obtained from DFO sources. Comparisons of run patterns and run composition were then made across all stocks and years.

### 4.1 Escapement enumeration

#### Fence design

The fences used were designed by SEP engineer R. Finnigan of DFO's Special Projects Division. Prototypes of the fence had previously been tested at Kanaka Creek, Lang Creek and Sliammon River, and had proven to be relatively inexpensive and reliable. The main feature of this fence type is that it can be collapsed onto the streambed during floods. Specifications for fence construction and installation were obtained from SEP (blueprints # 33-26-8 and 33-26-9). A brief summary of the main characteristics and operation of the fences are given below.

The fence consists of a series of interconnected wooden frame panels with aluminum tube insets. The panel bases are hinged on a foundation composed of wooden sills imbedded across the river. The top end of each panel is attached to a supporting beam which spans the whole width of the stream. This beam allows all panels to be lowered or raised simultaneously up to a 50° angle against the current. The supporting beam can be moved into position through the use of pulleys, winches and cables stretched across the stream. Wooden abutments located on each stream bank hold the winches and the wooden towers used to support the main cable and pulleys.

Small debris passes through the fence, but large leaves and branches are retained. These must be periodically brushed off the panels by field crews, who use the fence as a walkway during cleaning periods. Since the fence is wedged against the current in an inclined position and does not project greatly above the water, large floating debris is easily raked over the fence. Occasionally, the cables attached to

the downstream edge of the fence can catch logs and branches. When substantial pressure is applied, these cables slide off the hooks connecting them to the beam, causing a partial collapse of the fence, and allowing the debris to float downstream. Cables can be hooked subsequently under appropriate hydrological conditions, and the panels raised again.

Most of the large logs and branches move downstream during floods. At times of flooding and excessive debris accumulation, the fence can be lowered partly to let large debris float over it. This allows some fish to move upstream, but greatly reduces the possibility of damage. The fence remains particularly susceptible to damage by logs and stumps which scour the streambed and catch the base of the panels. When this type of damage occurs, field crews must wait for water levels to recede before repairing the fence. Because of the modular design of the structure, fence components separate from each other during breakdowns, and get dispersed on the streambed further downstream. The fence can then be reassembled within a few hours under appropriate hydrological conditions.

During normal fence operations, all upstream migrants are funnelled into a large (1.5 m x 3 m) trap made of wooden dowls. This trap is located adjacent to the abutment, on the upstream side of the fence. Salmon moving upstream are generally attracted to the turbulence created below the trap entrance on the downstream side of the fence. From there, they enter a wooden tunnel that extends through the sill, and leads into the trap. The configuration of this channel forces fish to swim under the fence panels. As a result, fish can access the trap even during periods of low water levels.

### Fence locations and installation

Adult fences were installed in the lower section of each stream during 1985. Preliminary field surveys identified potential sites for fence locations, based on considerations of flow rates, stream widths, geomorphological characteristics and accessibility. Final decision regarding fence location was made in conjunction with local authorities, DFO fishery officers and SEP engineers. Most of the specifications concerning the fence installation were outlined in the SEP blueprints. Only slight modifications were made to adapt each structure to local conditions.

At Black Creek, the site chosen was approximately 0.5 km from the mouth of the river, inside the Miracle Beach Park, adjacent to the road leading from Highway 19 to the Miracle Beach resort. This location is approximately 10 m downstream from the juvenile trapping site, and is beyond tidal influence. One of the main reasons for selecting this site was the existence of a well preserved

concrete structure used for the old escapement fence which was operated in 1979-1980. The old structure consisted of two concrete abutments, some 16 m apart on each bank, connected by a one meter wide concrete sill. The existing abutments were extended from 2.5 m to 4.0 m, and encased in a wooden structure to provide sufficient area for the winches and towers. A small channel (15 cm x 30 cm) was cut into the concrete sill, and a rectangular wooden tunnel (15 cm x 1 m x 2 m) was installed on top of the cement sill to serve as a trap entrance. The aluminum panels normally hinged on the sill were positioned on top of this tunnel.

At Trent River, the site chosen lies some 300 m downstream from the Highway 19 bridge over the river, or approximately 600 m downstream from the juvenile trapping site. At this location, stream width is approximately 24 m, and the streambed is characterized by an unstable and uneven mixture of rocks, gravel and sand. This area is strongly influenced by tides, which alone can account for daily variations of 1.5 m in water levels. This location was selected by fishery officers mainly for reasons of convenience. At French Creek, the site chosen lies behind the French Creek hatchery on Miller Road, approximately 1.5 km upstream of the estuary. At this location, the stream is about 20 m wide, and the streambed is composed of gravel, cobble and small rocks. This site lies beyond tidal reaches, and is approximately 100 m above the smolt trapping site.

At Trent River and French Creek, the front portions of the traps were strengthened by using iron bars instead of wooden dowling, to make the traps more resistant to damage from floating logs. Additional braces were used on the towers of the Trent River fence to support the heavy load of a longer fence. Finally, at all locations, the main cables were secured to large trees, as opposed to buried logs, to reduce costs and provide stronger anchoring.

#### Fence operation and enumeration procedure

Each fence was assembled prior to the first fall freshet, while stream flows were negligible. Once the installation was complete, each fence was checked periodically until the upstream migration began. Throughout the run, fences were monitored each day by at least one worker during all daylight hours. The usual daily schedule of activities is described below.

Field crews usually arrived at daylight. All debris which had accumulated overnight on the fence panels was brushed off and released downstream. Fence components, towers and the trap were inspected for damage, and if necessary, repairs were done immediately and recorded. Fence inclination

was then adjusted on the basis of prevailing water levels to maximize trapping efficiency. If water conditions were too low to provide a good flow through the trap entrance, plastic sheets were placed over some panels to concentrate the flow through the trap. Fish captured overnight and throughout the day were dipnetted out of the trap, marked and released upstream. While processing the catch, each fish was identified, aged (jack or adult), examined for injuries and missing fins (unclipped, LV, RV). Fish lacking an adipose fin were usually not killed at the fence, in order to allow most fish to spawn. Tagged coho recovered as carcasses below the fence, or seriously injured at capture, were measured, sexed and beheaded for CWT recovery. Each head recovered was stored in a freezer until shipment to DFO's Head Recovery Laboratory for identification.

At the end of the day, the angle of the fence was adjusted in relation to anticipated flow rates. During 1985, adult enumeration was not conducted at night given the limited manpower and equipment available, and lack of adequate means of communication. During the first week of migration in 1985, all fence traps remained open throughout the night at the request of local DFO authorities, who were concerned about excessive accumulations of fish below the fence. Complete enumeration was allowed in subsequent years, and adult enumeration was conducted throughout the night during the peak migration periods, as additional resources were available.

At the request of the local Fish & Game clubs, some adult coho were retained at the French Creek and Trent River fences to supply the local hatcheries with brood stock. Efforts were made to retain at least 10% of the catch for this purpose. No attempt was made to select adults on the basis of their sex or size. Instead, efforts were made to select primarily adult coho in an intermediate or advanced stage of maturation, since silver fish were less tolerant of handling and transportation. Efforts were also made to collect the fish required throughout the period of migration, rather than on a single occasion.

### Marking procedures

Some of the fish escaping to each stream were marked at the fence during the enumeration to determine fence efficiencies and escapement levels. Preliminary field surveys in 1984 indicated that a considerable amount of effort would be required to capture more than 100-200 coho near the spawning grounds given the adult densities observed. Therefore, it was judged necessary to maximize the marking effort relative to the recovery effort in order to compensate for the anticipated low recovery rates. The initial objectives were to mark and re-sample at least 40% and 10% of the escapement respectively. According to Robson and Regier (1964), this effort allocation should provide Petersen estimates within

10% of the true value, 19 times out of 20, under ideal conditions. These authors also noted that under such conditions, the resulting bias associated with the abundance estimate can be considered negligible (< 2%).

As a result of concerns expressed by local DFO officials over the potential effects of the marking plans, some restrictions were imposed in 1985. The use of external, numbered tags was not allowed for general marking purposes. In addition, marking could only be conducted when there was no apparent build-up of fish below the fence. When build-ups occurred, marking had to be temporarily stopped until the build-up had been eliminated. Because of these restrictions, a substantial portion of the fish intercepted at the fences in 1985 and 1986 had to be released without marks. These restrictions were relaxed during 1987 and 1988 to facilitate the study.

Coho were marked by means of a hand held paper punch. During 1985 and 1986, an 8.5 mm wide hole was placed in either the upper, middle or lower lobe of the caudal fin, depending on the time of passage at the fence. This procedure was used in the hope of determining the migration rate, distribution and stream residency of different timing components of the run. Following a suggestion by Dr. J. R. Irvine, tail marks were replaced by opercular marks in 1987, which proved to be less susceptible to fraying, and more readily discernible on carcasses. Irrespective of the body area used, marked fish did not appear to be more visible than unmarked ones, and both groups were considered to be equally detectable. Coho exhibiting serious injuries were usually released upstream without marks.

Attempts were made to mark as many coho as possible throughout the run, but owing to various constraints, marks were not applied to a fixed portion of the daily catch, nor were they applied proportionally throughout the run. Individually numbered external tags were also applied by other investigators to a limited number of coho during 1987 and 1988 to estimate the period of stream residency (Bocking *et al.* 1988). During 1987, numbered Petersen disks were used to tag coho at all locations. At Black Creek, disks were applied to 68 adult coho on November 17. At Trent River, 43 adult coho were captured and tagged in the headwaters on November 9, and 12 more were captured and tagged one km above the fence site. No tags were applied at French Creek due to the low numbers of adults caught (Bocking *et al.* 1988). During 1988, orange, blue and green spaghetti tags were used at all locations. At Black Creek, 100 adults were tagged on November 1, and again on November 4. At Trent River, 30 adults and 20 jacks were tagged on October 25. At French Creek, 50 adults were tagged on November 1, and 20 adults on November 3 (R. Bocking, pers. comm).

### Trapping efficiency

The frequent occurrence of floods at Trent River, and the unanticipated debris build-up at night, caused considerable damage to this fence on many occasions. Large tree stumps, scouring the streambed while being displaced downstream, hooked the base of the panels at least once a year, causing the fence to break apart. The supporting beam was broken many times, and had to be braced or rebuilt entirely. The main cable tore apart during 1987, and had to be replaced. However, the traps, towers and panels were never damaged, and the fence could usually be reassembled within 48 h after flood waters had receded.

When water levels were  $< 0.5$  m, all fences seemed to intercept over 95% of all adults. Small adults ( $< 35$  cm) and a variable portion of the jacks could swim between the aluminum bars of the panels, as well as between the dowls of the trap. All attempts to modify the panels and traps so as to prevent the passage of small coho caused additional debris build-up, and led to excessive stress loads on the supporting cables. Due to the location of the fence at Trent River, water levels often exceeded 0.5 m; ideally, a location above tidal influence should be selected in the future for operating this type of fence.

Field observations in 1985 indicated that the shape of the streambed below the fence seemed to influence trapping efficiency during the early part of the adult migration. Under low water levels, fish moving upstream often found refuge in the surrounding pools after a brief attempt to pass through the fence. When small pools in the vicinity of the French Creek fence were filled with boulders, coho appeared to search longer for an opening, and were caught more readily. During 1986, 1987, and 1988, a 100 m section of streambed below the French Creek fence was leveled with a backhoe, and then slightly channelled towards the fence entrance. This allowed coho to swim directly into the trap while water levels were still low, and led to improved trapping efficiency during the early part of the run.

## **4.2 Run patterns and composition**

### Black Creek

Each year, coho began aggregating at the stream mouth during late September. Prior to the first freshet, water levels were usually too low for coho to access the lower pools. Substantial rainfall was required each fall to fill Northy Lake and provide sufficient outflow to raise the water levels in the lower

reaches. Coho arrived at the fence as soon as the water levels across the cement sill reached 10 cm. During 1985, the migration began relatively early in the year, and most coho examined were silver and had some ectoparasites (sea lice). Some adults caught in late November also had this appearance, suggesting that coho were still entering the stream by this date.

From 1986 to 1988, delays in the arrival of fall rains forced all early migrants to hold near the estuary. Each year during this period, between five and 50 dead coho were recovered in intertidal waters along the bank. Minor catches reported during 1986 prior to the first major migration pulse consisted of such recoveries (Fig. 4.1). During 1986 and 1987, when the first freshet occurred relatively late in the season, most of the fish examined at the fence had matured to a considerable extent; approximately 20% were silver, 60% had some coloration, and 20% were dark brown. From 10 to 15% of the fish were judged to be ripe, based on the amount of pressure required to extrude gonadal products, and four adults released eggs or milt while being processed. During 1986 and 1987, about 10 spawning events were observed between the fence and the mouth of the river.

Each year, some coho aggregated in the pool located 30 m downstream from the fence, particularly during periods of low flow. During 1985, unseasonably cold temperatures caused the surface waters to freeze between Nov. 10 and Dec. 2. A solid layer of ice covered all surface waters adjacent to the fence, thus preventing further migration. On several occasions, fence crews broke the ice, seined out all coho from the lower pools, and released them in pools above the fence. Catches reported on November 13, 21, 22 and 25 correspond to the numbers of fish handled by the crew while the stream was frozen. Seining was also required during late November 1986 and 1987 to ensure that the fish holding below the fence could be examined, marked and released prior to the next flood.

Floods interrupted counting operations at least once every year. During floods, very little debris in the form of logs and branches was transported downstream. This allowed the crew to maintain the fence upright under considerable flow, except under conditions of extreme water levels ( $> 65$  cm), water velocity ( $> 170$  cm·s<sup>-1</sup>), and discharge rates ( $> 22$  m<sup>3</sup>·s<sup>-1</sup>). During this study, the duration of floods per season increased from six d in 1986, to 18 d in 1988, in accordance with the mean seasonal discharge (Table 4.1). This caused an increase in the frequency of incomplete counts obtained each year.

Fence counts and visual observations conducted at the fence suggested that over 80% of the returning coho enter the stream during the two week period following the first freshet each year. Small pulses of migrants were detected as late as December 8 (1985), but usually the vast majority of adults had moved upstream by the end of November. It is doubtful that significant numbers of adults enter the

river during December. This observation is supported by numerous census results which showed that silver, immature coho are rarely captured late in the season (Section 4.4). Adult counts were highest in 1986 (4408), and lowest in 1987 (750), indicating a sixfold variation in the number of adults intercepted at the fence during this study (Table 4.1). Jack counts were considerably less variable, ranging from 191 in 1986, to 528 in 1987. Field observations suggested that about 20% of the jacks present at the fence each day swam through the panels without being enumerated. None of the adults or jacks examined at the fence lacked a ventral fin, indicating that few hatchery fish stray to this stream<sup>20</sup>.

During each season, a variable portion of the fish examined each day was marked prior to release. The proportion of adults marked increased from 58% in 1985, to 82.1% in 1988, while the corresponding proportions among jacks increased from 12.4% to over 95% during the same period. None of the marked fish were ever reported to have been recovered in neighboring streams where escapement levels were being monitored by DFO personnel. Therefore, it was assumed that adults and jacks from Black Creek did not emigrate to other streams once they had moved beyond the fence.

### Trent River

Each year, coho began aggregating in the lower sections of the river in early October, prior to the first freshet. At the beginning of each season, negligible stream flows prevented these early migrants from moving beyond the tidal waters. Early migrants could occasionally be seen from the stream bank, moving upstream with the flow, holding below the highway bridge during high water, and retreating back to sea with the ebb. Installing the fence disrupted the normal movement patterns, and forced the fish to hold below the fence at high water. Field observations suggested that decreasing barometric pressures, tidal changes and increasing water levels induced coho to move upstream. Prior to the first freshet, coho were often caught during periods of receding tides and low stream flows. All fish captured, processed, and released under these conditions would aggregate below the highway bridge, and hold until there was at least 12 cm of water flowing through the channels connecting various pools. Even during the unseasonably dry periods of 1986, there was sufficient flow for the migration to begin about two weeks prior to the first major freshet (Fig. 4.2).

With the arrival of the first freshet, large amounts of leaves and forest debris were transported downstream in this river. Water levels usually increased abruptly following periods of heavy rain, and a

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<sup>20</sup> Approximately 95% of the fry and smolts released during 1984 and 1985 by the public hatcheries around the Strait of Georgia were fin clipped (l.v or r.v) for assessment of harvest rates during 'Expo' (1986).

0.5 m increase in water level over four hours was common. As a result, it was not usually possible to maintain the fence upright during the first freshet. Even if six persons were available to clean the fence continuously, the fence generally had to be collapsed a few hours after the flood began. In the absence of debris, the fence could not be kept upright under conditions of extreme water levels ( $> 87$  cm), water velocity ( $> 2$  m·s<sup>-1</sup>), and stream discharge ( $> 37$  m<sup>3</sup>·s<sup>-1</sup>). During 1985, slippage in the cable anchors necessitated a partial collapse of the fence during periods of moderate to heavy flows. During the following seasons, both the seasonal discharge rates and the frequency of floods increased substantially (Table 4.2). These conditions required that the fence be collapsed intermittently for at least two weeks each season. Each year, fence crews observed some coho ( $< 100$ ) moving upstream while the fence was collapsed, and on some occasions, a few coho being passively transported out to sea. As a result, only a partial escapement enumeration was possible each year.

During 1985, surface waters in areas adjacent to the fence were frozen between November 10 and December 2. The catches reported on December 3 (Fig. 4.2) were obtained by seining coho held below the fence. Seining was also conducted during 1986 on November 8, 14 and 15, to obtain brood stock for the hatchery, and to ensure that the remaining fish could be examined, marked and released before the anticipated flood. Seining was also conducted below the fence during November in 1987 and 1988 for the same purposes.

Fence counts and visual observations conducted at the fence suggest that this population has a protracted run. The adult migration occurred over a four to five week period in 1985 and 1986, and over a two to three week period in 1986 and 1987. Each year, coho showed a tendency to enter the river at irregular intervals prior to and during the first flood. Coho captured after November 23 were usually those that had held below the fence, and rarely included silver coho that had recently arrived. Thus, most of the migration occurred each year before the last week of November. It is doubtful that significant numbers of coho entered the river during December. This observation is supported by numerous census results which showed that silver, immature coho are rarely captured late in the season (Section 4.4).

Adult counts were highest in 1986 (1076) and lowest in 1987 (194), indicating a fivefold variation in the number of adults intercepted at the fence (Table 4.2). Jack counts were even more variable, ranging from 12 (1987) to 115 (1988). Field observations indicated that about 35% of the jacks present managed to swim through the trap and fence panels. Fence observations, and snorkel counts conducted further upstream by other investigators during 1988, revealed that an unusually large number of jacks ( $\approx 80$ ) escaped undetected early in the season while the fence was in operation (R. Bocking, pers. comm.), representing about 400% of the total trap count up to that date.

During 1986, adult strays from neighboring hatcheries were identified by their missing ventral fin, since most of these had been fin clipped prior to release. Adults lacking a left ventral fin made up over 47.6% of the total fence count. Without the presence of a characteristic fin clip, the contribution of hatchery strays to the total escapement during 1985, 1987 and 1988 could only be determined from the identification of the coded-wire tags recovered (Section 5.2). The proportion of jacks lacking a fin (*lv*, *rv*, or *ad*) was usually smaller than the corresponding proportion among their adult siblings (Table 4.2). Such discrepancies might be attributed in part to the small number of jacks examined, and the confounding effect of the relatively large number of hatchery strays among the adult sample.

A substantial number of adults caught at the fence were retained each year to supply the local hatcheries with brood stock. The proportion of the adult fence catch used for this purpose was lowest in 1986 and 1988 (8%, 11%), but higher in 1987 and 1985 (16%, 20%). A few injured adults were retained each year for CWT identification. As a result, only a fraction of the adults intercepted each year were available for mark-recovery operations. During 1985, about 1/3 of the adults and jacks were released without marks, but nearly all adults and jacks were marked prior to release in subsequent years (Table 4.2).

Not all marked fish remained in the river. During 1986, 1987 and 1988, field crews at the fence observed a few coho moving downstream during the first major flood, and marked fish were occasionally found in neighboring streams after the flood. On November 25, 1986, 143 marked adults were recovered at the Puntledge hatchery, and two marked adults were caught in Roy Creek. On November 30, 1987, six marked adult coho were recovered at the Puntledge hatchery. In addition, one unmarked adult that had been tagged with an external Peterson disk in the headwaters of the Trent River, was recovered in the headwaters of the Tsolum River by Puntledge River hatchery staff. During 1988, seven marked adults and one marked jack were recovered at the Puntledge River hatchery on November 4. A few days later, two marked adults were recovered in Roy Creek, and one marked adult was found in a small adjacent creek. Each year, several other systems in the region were surveyed by DFO personnel, but no other marked fish were reported.

The number of marked fish recovered at the Puntledge River hatchery was expanded by approximately 25% to account for the sampling regime used at the hatchery (Section 2.1). The number of marked fish that emigrated from the Trent River during the 1986, 1987 and 1988 seasons was estimated to be 181, eight and twelve adults respectively. This represents 16.8%, 4.1%, and 3.3% of the adults counted at the Trent River fence. Based on the time of recovery and the occurrence of floods

on the Trent River, it was assumed that the marked fish left the stream on Nov. 18, 1986, Nov. 11, 1987, and Nov. 1, 1988.

Fence counts were adjusted to account for the emigration of marked fish from the Trent River each year. For 1986, 82% of the emigrants were assumed to lack a left ventral fin, which represents the fin clipped fraction among the smolts released at the Puntledge hatchery in 1985. It was also assumed that 6.4% of the fin clipped adults that emigrated during 1986 lacked an adipose, as this represents the tagged proportion among fin clipped adults enumerated at the Trent River fence that year. Fence statistics on the number of fish released, marked, and the number of fish lacking a particular fin were reduced accordingly. The proportion in each fin clipped category was recalculated to provide preliminary estimates of the size and composition of the remaining group (Table 4.2). Similar adjustments were made to the 1987 and 1988 counts, except that the proportion in each fin clipped category was not recalculated, given the relatively small number of emigrants observed.

Chum salmon arrived at the fence mostly around mid-November each year. Marks were applied to all chums captured at the fence, but this was usually an insignificant portion of the total run since the vast majority of adults either moved upstream during floods, or remained below the fence where they spawned and died. Some spawning events were observed throughout the 600 m section below the fence, but most spawners used the section between the old trestle and the railroad bridge. Given this situation, escapement levels were based exclusively on visual survey counts (Section 4.5).

### French Creek

Prior to the first freshet, some 10 to 50 coho were seen each year near the stream mouth. During 1986 and 1987, delays in the arrival of fall rains forced these fish to hold in the pool under the highway bridge. During 1985 and 1988, sufficient stream flows allowed most early migrants to move into several fresh water pools above tidal reaches. Each year, a few adult coho (< 12) died while holding in the lower sections prior to the first major freshet.

Approximately 12 to 16 cm of water over the sill was sufficient for coho to swim up to the fence. As a result, several hundred coho were able to migrate upstream in 1985, although freshets never occurred during the season (Fig. 4.3). During the following seasons, water levels increased more abruptly, and over 85% of the coho enumerated were captured within five days from the first period of high water. Fence counts reported after this period usually consisted of fish which had been holding or

spawning in the lower pools, and rarely included newly arrived fish. During 1985, some late migrants remained downstream under the ice from Nov. 23 to Dec. 5. Coho were captured with seines in the lower pools only during December 1985, after the ice had thawed, to collect brood stock for the hatchery. Surveys conducted in the lower sections indicated that nearly all coho had moved upstream by the end of November. It is doubtful that significant numbers of adults entered the river during December.

Substantial amounts of leaf material were transported downstream even when stream flows were moderate. Up to six workers were required to continuously clean the fence panels during the first freshet. If much debris was present, the fence had to be collapsed when discharge rates exceeded  $5 \text{ m}^3\text{-s}^{-1}$ . By contrast, under clear conditions the fence could be maintained upright until discharge rates reached  $18 \text{ m}^3\text{-s}^{-1}$ . Floods conditions occurred at least once a year, so only a partial enumeration of each run was conducted. Partial counts were only obtained on a few occasions during 1985 and 1986, but were nearly three times more frequent during 1987 and 1988. Fence counts of adult coho ranged from 76 (1987) to 937 (1988), indicating a twelve-fold variation in the number of adults intercepted at the fence (Table 4.1). Jack counts were considerably less variable, ranging from 26 (1986) to 109 (1988). Field observations suggested that about 25% of the jacks present at the fence moved through the panels without being enumerated.

The absence of jacks lacking a ventral fin in the 1985 sample indicated that hatchery fish did not stray in large numbers to this stream. This was confirmed during 1986, as only four of the 625 adults examined at the fence lacked their adipose fin. Assuming that these fish did not naturally lack their adipose fin, then hatchery strays made up less than 1% of the adult escapement.

The fraction of the catch that was retained to supply the local hatchery with brood stock each year ranged from 9% to 25%. No other live adults were retained for CWT recovery. Because of this fact, and other logistic constraints, less than 1/4 of all adults and jacks released were marked during 1985. However, during subsequent years, nearly all adults and jacks released from the fence were marked. None of the marked fish were ever reported to have been recovered in neighboring streams where escapement levels were being monitored by DFO personnel. Finally, only a few chum salmon were caught at the fence each year (Table 4.3). Field surveys indicated that most chum spawned in the lower one km section of the stream. Each year, some 10 to 30 chum spawners were counted in that section.

### 4.3 Stream survey design and sampling methods

Sampling of spawners was conducted periodically in the three streams throughout the spawning periods to determine the proportion of adults and jacks that were marked, recover coded-wire tags, and obtain additional information on the demographic traits of each population. Preliminary field surveys were conducted in 1984 to determine the location of spawning grounds and holding pools, evaluate the accessibility of various areas, identify potential sampling sites, and obtain information on the relative distribution coho in each stream. Field observations revealed a considerable amount of spatial and temporal variation in coho densities, but high densities were observed at specific sites previously identified by local DFO biologists, which indicated that the spawners aggregated in certain locations year after year.

Howard (1948) and Schaefer (1951) noted that non-uniform distribution of marks in the population can influence the mark-recapture estimates. Howard (1948) suggested that sampling for marks should be distributed over time and areas to obtain a representative sample of the whole population. Many statistical methods require that samples be taken at random in space and time to eliminate bias. In the present case, sampling at randomly selected times and areas was considered to be impractical due to the inaccessibility of certain areas and the difficulty of sampling under certain conditions. However, Hurlbert (1984) noted that for ecological studies, interspersed samples is often a more practical approach than randomization, and generally serves the same purpose. With this general approach in mind, one sampling area was selected in each of the upper, middle and lower sections of each system. Within each area, specific sampling sites were identified based on the apparent availability of fish, accessibility, and suitability for electroshocking. The sites chosen consisted mainly of shallow holding areas, narrow pools and reaches under stream banks, scattered adjacent to and away from the spawning grounds. Efforts were made to conduct surveys each week during the spawning period. During each survey, efforts were made to patrol all areas, and sample in at least one site per area.

For purposes of consistency, a single crew conducted nearly all surveys done in each stream during this study. Live coho were captured mainly by means of an electroshocker using the minimum electrical output necessary to temporarily stun the fish. On some occasions, large seines were also used to collect coho holding in deep pools. Samples obtained by these two methods were combined when possible in order to minimize the potential selectivity of the sampling gear. Irrespective of the sampling gear used, some of the fish present in the section sampled usually escaped without being examined. As a result, catch per unit effort statistics based on the sampling data are not always indicative of the relative abundances in each area. During each sampling event, a distance of about 0.5 km was covered within

each area. At times when densities were low, longer distance were covered in an attempt to recapture at least four marked fish per sample. Ricker (1975) noted that when this condition is met the 95% confidence intervals associated with the number of recoveries do not include zero.

Each coho captured was measured, sexed, and examined for marks and missing fins. On some occasions, ripe females were kept to supply the local hatchery with brood stock, but in most cases, all fish examined were given a secondary mark on the operculum by means of a hand-held paper punch, and released after a brief recovery period. All carcasses recovered were examined for marks, given a secondary mark, and left in the stream. Live and dead coho which were displaced downstream and accumulated on the fence panels were also caught, examined, marked, and released. Heads were only removed from coho that had spawned or died, and that lacked an adipose fin. All heads recovered were labelled, placed in a plastic bag, and frozen at the end of the day. Surveys were usually terminated between Dec. 15 and Dec. 25 when fish densities appeared to be too low to justify further sampling. Field observations suggested that over 95% of the spawners had died and disappeared by December 25 each year.

#### **4.4 Mark incidence and spawner distribution**

##### **Black Creek**

The majority of the surveys conducted during 1986-1988 were done in the middle and upper reaches of the creek, mainly because the lower reaches could not be surveyed under high water conditions. The opposite situation prevailed in 1985, and no samples were collected in the headwaters, because the unusually low water conditions prevented adults from reaching the upper spawning grounds. Approximately eight surveys were conducted each year, but substantial numbers of coho were only captured on certain occasions, as determined by the period of stream residency and escapement levels.

The number of adults examined for marks each year ranged from 41 to 169, while the number of jacks examined ranged from five to 21 (Table 4.4). The proportion of marked adults recaptured each year ranged from 2% to 5% of the marks released, while the proportion of marked jacks recaptured ranged from 1% to 4%. Recapture rates of jacks and adults are not directly comparable, since the number of marked adults and jacks released each year differed considerably. The incidence of marks among adult samples (mark proportion) ranged from 27.9% to 90.2%, while the incidence of marks in

jack samples was about 30% to 50% of the mark proportion among adults. Considerable variation in the mark proportions was observed among samples each year. No evidence was found to indicate that significant and consistent differences existed between the mark proportions in samples from various locations, or to indicate that specific segments of the run aggregated or spawned predominantly at certain locations. Marked and unmarked fish appeared to distribute themselves throughout the stream in a fashion that was not affected by their time of arrival at the fence. Mark proportions generally decreased as the season progressed, partly in response to immigration of unmarked fish during the floods late in the season. The slight increase in marked proportions observed during the early part of 1985 was attributed to the substantial releases of marked fish late in the season. No significant differences in the marked-to-unmarked ratios were detected between males and females in samples collected during surveys (Table 4.7).

Sufficient numbers of adults and jacks were examined each year during stream surveys to obtain estimates of the proportions of adults and jacks lacking an adipose (i.e. tagged), left ventral or right ventral fin ( $p(ad)$ ,  $p(lv)$ ,  $p(rv)$ ). Reference to Tables 4.1 and 4.4 shows some discrepancy between the proportions of adults and jacks lacking an adipose or left ventral fin based on stream surveys, and the corresponding proportions based on fence counts. Data from both sources were entered in  $2 \times 2$  contingency tables, and analyzed by means of a chi-square test corrected for continuity (Table 4.8). No significant differences were found between the relative abundance of tagged jacks (no adipose) from fence counts and stream surveys. Significant differences were not detected between the tagged ratios among adults sampled at each location during the 1985-1987 seasons, but were detected during the 1988 season.

The nature of the factor accounting for differences in tagged ratios could not be determined with certainty. No evidence was found to indicate the presence of tagged adults from other systems at the fence, or of emigration of tagged fish during floods. It was hypothesized that there might have been a higher proportion of tagged adults in the early part of the run than during the flood when adults moved upstream undetected, but the estimates of  $p(ad)$  obtained at the fence or during surveys showed no apparent increase or decrease through time. In the absence of evidence to indicate that the stream survey samples were not representative of the marked proportions in the population, sampling statistics from both sources were pooled within each season to estimate minimum escapement levels, and the proportions that were tagged or fin clipped. These estimates were calculated separately for each age group as follows (using lv clips as an example):

$$(Eq. 4.1) \quad m.esc_{lv} = \sum count_{f,lv} + \sum count_{s,u,lv} - \sum emi_{lv}$$

$$(Eq. 4.2) \quad p(lv)_{esc} = \frac{m.esc_{lv}}{m.esc}$$

where:  $m.esc_{lv}$  = Minimum escapement of fish lacking a left ventral fin;

$count_{f,lv}$  = Daily count of fish released at the fence without a left ventral fin;

$count_{s,u,lv}$  = Stream survey counts of unmarked fish, lacking a left ventral fin, which were not previously counted at the fence<sup>21</sup>;

$emi_{lv}$  = Estimate of the number of marked emigrants, lacking a left ventral fin, that were recovered in other streams after the floods;

$p(lv)_{esc}$  = Proportion of adults or jacks in the escapement that lacked a left ventral fin.

Given the relatively small number of fish captured during stream surveys, and the absence of emigration of marked fish to other streams, the estimates based on pooled data (Table 4.9) do not differ substantially from the estimates based on fence counts.

Stream surveys revealed considerable year to year variation in the distribution of spawners, and in the time lapse between stream entry and spawning period. During 1985, spawning was mostly confined to the stream section between Northy Lake and the fence, since low water levels and ice coverage prevented access to most of the upper reaches. Spawning activity was not noticed until 10 d after the first day of adult migration. By contrast, spawning activity was detected as soon as the fish entered the stream during 1986 and 1987, and to a lesser extent in 1988. During the 1986 and 1987, spawning activity was mainly confined to the areas adjacent to the Duncan Main Rd., in the lower end of Millar Cr., in the 500 m section below the bridge at the end of Endall Rd., and throughout the one km section above the fence. During 1984 and 1988, a larger portion of the spawners were found in the headwaters. Very little spawning activity was detected in the small tributaries near Dzini Rd., except during 1986. Each year, spawning activity peaked during late November and early December. Nearly all adults examined after December 10 were spent. During 1987, the unusually large jack to adult ratio detected at the fence was also noticeable during the spawning period, as most spawning females were

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<sup>21</sup> This was set to the number of fish, in each category, lacking a primary and secondary mark, if the number of fish released at the fence without marks was relative small compared to the total marks applied (<10%). When this condition was met (in most cases), the possibility that a fish gets counted twice was considered to be negligible. In cases where this condition was not met (for instance Black Cr., '85, '86, '87), the number of fish in this group was set to 0.

paired with jacks. During other years, jacks were occasionally observed in the vicinity of females, but were rarely observed to be paired with one during a spawning event.

Some of the Petersen and spaghetti tags applied at the fence during 1987 and 1988 were recovered each year during both stream surveys and the instream indexing surveys. Tags recovered during the indexing surveys were used to generate tag depletion curves. Estimates of the period of stream residency were then obtained by integrating each curve and dividing it by the number of tags applied (AUC method, Bocking *et al.* 1988). The estimated average period of stream residency of adults was 10.3 d in 1987 (Bocking *et al.* 1988), and 11.2 d in 1988 (R. Bocking, pers. comm.).

### Trent River

Considerable effort was required to collect coho with an electroshocker due to the presence of large log jams, deep pools, and wide channels which provided shelter to the fish. As a result, about 10 surveys were conducted each year, but substantial catches were not obtained on each survey. On several occasions, samples obtained at various sites on a given day had to be combined in order to estimate the proportions required (Table 4.5). Large samples were usually obtained by seining various pools in the headwaters, or by electrofishing near the spawning grounds.

The number of adults sampled during the 1986 stream surveys represented a relatively small percentage (6.3%) of the total fence count, but this percentage was higher in other years, ranging from 33% in 1985, to 70% in 1987. The recovery rate of adults marked at the fence ranged from 5% to 16% of the fish marked, while the recovery rate of jacks ranged from 0% to 20%. The recovery rate of marked adults was not consistently greater than for jacks, but the factor(s) responsible for this pattern could not be determined with certainty given the large differences between the number of marks released. The average mark proportion among adult recapture samples was relatively high in 1986 (53.7%), but substantially lower in other years, ranging from 12% to 28%. The average mark proportion among jack recapture samples was considerably more variable, ranging from 0 to 50%. Each season, there was considerable variation in the mark proportions observed during the surveys, and a general tendency for them to decrease after the first flood (Fig. 4.2). Marked fish released in the early, middle or late stages of the run were subsequently recovered at sampling sites throughout the river. Statistical comparisons of the mark proportions obtained at different times and sampling sites was not possible due to small sample sizes. However, no obvious differences were noticed between the spatial

distributions of marked or unmarked fish within the river. No significant differences were detected between the marked-to-unmarked ratios in males and females each year (Table 4.7).

Discrepancies were observed between the proportions of jacks or adults lacking a particular fin in samples taken at the fence and during stream surveys (Tables 4.2, 4.5). Chi-square tests revealed significant differences between the relative abundance of fin clipped adults (*lv*) in samples from these sources during 1986, even after adjusting the fence counts for emigration losses (Table 4.8). Significant differences in the relative abundance of fin clipped adults were also detected between samples from the upper and lower sections of the river ( $\chi^2 = 5.5$ ,  $P = 0.02$ ). Stream survey samples from sections upstream of China Creek contained relatively fewer fin clipped adults than samples from lower sections, indicating that hatchery strays tended to remain in the lower reaches of the river. Thus, the discrepancies in the relative abundance of fin clipped fish between fence samples and survey samples could have been induced by sampling predominantly in the headwaters. There was also a reduction in the relative abundance of fin clipped adults in samples taken before and after the flood, which is to be expected from the departure of fin clipped hatchery fish returning to the Puntledge River hatchery (as was observed during this study). This reduction was not found to be statistically significant ( $P = 0.175$ ), so all survey samples were combined with the adjusted fence counts to estimate the overall fraction of the population that was fin clipped.

The proportion of adults and jacks lacking an adipose fin (*p(ad)*) in stream survey samples was larger than observed at the fence during 1986, but the difference was not statistically significant ( $P = 0.898$ ). Estimates of *p(ad)* from samples taken before the flood were not significantly different from those taken after the flood ( $P = 0.930$ ). Such results could be expected irrespective of the contribution of hatchery fish, if the tagged proportions among hatchery and non-hatchery adults were similar. Accordingly, tagged proportions among adults of Trent River origin were comparable to those among the adults returning to the Puntledge River (12% and 14%, Tables 3.7, 5.9).

Similar results were obtained in 1987. Significant differences in the relative abundance of tagged fish (no adipose) were detected between the samples from both sources (fence, stream), even after adjusting the fence counts for emigration losses (Table 4.8). Significant differences in the relative abundance of tagged adults were detected between samples taken above and below China Creek ( $\chi^2 = 5.659$ ,  $P = 0.017$ ), with the headwater samples containing larger proportions of tagged adults. It is thus hypothesized that most of the adults in the headwaters were of Trent River origin, of which a relatively high proportion were tagged (36%, Table 3.7). By contrast, adults of hatchery origin with lower tagged proportions ( $\approx 6\%$  for Puntledge R., Table 5.9) tended to remain in the lower reaches of the river, where

they caused a localized reduction in tagged proportions. Thus, the significant differences obtained when comparing stream and fence samples might have been induced by sampling predominantly in headwaters which misrepresented the overall contribution of hatchery strays in the river. With regards to 1988, the relative abundance of tagged adults was also higher in stream survey samples than was observed at the fence, but the difference was not statistically significant (Table 4.8). Fin clipped proportions among samples from both sources were almost identical, which was expected since fin clipped adults returning that year were not of hatchery origin (as in 1986). In view of the above facts, minimum escapement levels, fin clipped and tagged proportions among adults were calculated according to Eq. 4.1 and 4.2 for all years, using fence counts adjusted for emigration losses (Table 4.9).

Similar trends were observed for jacks. Tagged proportions among jacks were always greater in population survey samples than in samples taken at the fence, but the difference was statistically significant only in 1987 (Table 4.8). It should be noted that jack emigration from the Tent River to the Puntledge River was only detected during 1987. Tagged proportions among jacks examined at the Puntledge hatchery during 1987 were about 12% (Table 5.9), while the tagged proportions among jacks of Trent River origin was estimated to be  $\approx 55\%$  (Table 3.7). The presence of hatchery jacks with proportionally fewer tagged individuals could account for the low tagged proportions observed at the fence. Presumably, jacks of hatchery origin also occupied lower sections of the river (as was observed for adults) which would account for the differences in their relative abundance at the fence and in areas upstream. The available samples were not sufficiently large to test this hypothesis. Minimum escapement levels, fin clipped and tagged proportions among jacks were calculated by the same procedure used for adults (Table 4.9).

Population surveys revealed that marked fish released at the fence distributed themselves throughout the lower three km of the stream within 24 h after release, but usually required at least 48 h to reach the middle reaches (China Cr. to Bloedel Cr. junctions), and 72 h to reach the headwaters. Each year, a substantial number of coho were observed to aggregate in the lower reaches of the stream, in the middle reaches near China Creek, and in the upper reaches beyond the Elm trail near the Bloedel Creek junction. It should be noted that adults have been observed in the mainstem, at least one km beyond the Bloedel Creek junction, but were never seen in the lower section of Bloedel Creek.

On November 19, 1987, nine days after the initial migration, 43 adults were seined from the pools below the Bloedel Creek junction, marked with Petersen disks, and released (Bocking *et al.* 1988). One tagged adult was subsequently observed spawning in China Creek a few days later, and another tagged adult was recovered in the headwaters of the Tsolum River some 15 km away. The

remaining fish were subsequently recovered in areas adjacent to the location of tagging. Assuming that the extensive migration to the Tsolum River was not induced by the tagging, it appears that adult strays can migrate several kilometers upstream before deciding to leave the system. A fraction of the external tags applied in the headwaters during 1987, and at the fence during 1988, were recovered during the stream surveys and the instream indexing surveys. Based on the tag recovery pattern observed during the later surveys, the average period of stream residency for adults was estimated at 11.0 d in 1987 (Bocking *et al.* 1988), and 8.9 d in 1988 (R. Bocking, pers. comm.).

Field surveys conducted during 1986 revealed that a considerable number of hatchery fish (identified by their missing ventral fin) that had strayed into the Trent River, remained to spawn and die. To determine if hatchery and non-hatchery fish spawned together, efforts were made to capture adults that were paired together near redds, as well as adults that were in the process of spawning. Examination of the fish captured revealed that hatchery adults were spawning with other adults of hatchery and non-hatchery origin (assuming unclipped fish were not of hatchery origin). This suggested that there is a considerable amount of hybridization occurring in the Trent River coho population. No attempt was made to dig out the redds to determine the extent of egg fertilization.

Field surveys indicated no apparent changes in the distribution of spawners from year to year. Spawning activity was observed mainly throughout the 1.5 km section above the old tressel, in the lower reaches of China Creek, and areas adjacent to the Bloedel Creek junction. In addition, isolated spawning events were observed each year throughout the system wherever spawning substrate was available. Spawning activity started around late October, usually peaked during the first week of December, and was observed as late as December 24. Field observations suggested that over 95% of the stream population had spawned and died by the third week of December. In some years, a few (<3) silver bright coho with ectoparasites were captured in late December, indicating that some coho enter this river late in the season.

Chum salmon were observed to spawn mainly between the railroad track and the estuary. Field observations and examination of the spawners suggested that the run occurred over a relatively short period of time, with the vast majority of spawners arriving in one large pulse. Deadpitching was not conducted, mainly for aesthetic reasons, given the proximity of residential areas. Instead, visual counts were made of the number of whole carcasses and live fish present throughout this area during the period of peak abundance. The counts obtained were then expanded subjectively to account for the number of partial carcasses, and the number of fish removed by predators or tides prior to the count. This enumeration method indicated that escapement levels were lowest during 1985 and 1988, with spawning

populations of 500 and 450 adults respectively. Escapement levels were substantially higher in 1986 and 1987, with estimates of 1100 and 900 adults respectively.

### French Creek

The majority of the surveys conducted between 1985 and 1988 were done in the upper and middle reaches of the stream, mainly because the holding locations in the lower sections were too difficult to sample properly. The temporal distribution of surveys and the catches reported reflect the period of spawner residency (Table 4.6). Relatively large catches were obtained in 1985 and 1986, when fence counts indicated intermediate levels of escapement. The relatively low fence counts obtained in 1987 were accompanied by unusually low densities of adults in the stream and low catches upstream. Large catches were obtained in 1988, due in part to the substantial number of carcasses recovered at the fence.

The proportion of marked adults recaptured each year ranged from 9.3% to 13.8% of the fish marked, which was remarkably stable given the large year to year differences in the number of marks applied. The corresponding proportion for jacks was always lower than among adults each year, and ranged from 0% to 11.1%. During 1987, nearly twice as many marked jacks were released into the stream than marked adults, but the recapture rates of marked adults were nearly three times greater, which suggests that the catchability of jacks was much lower than that of adults.

Within a given season, there was considerable variation in the mark proportions observed (Table 4.6). Samples were too irregularly spaced to be contrasted statistically for spatial trends, but field observations provided no evidence that marked fish distributed themselves in a particular fashion in the stream. No significant differences in mark proportions were detected between males and females from survey samples (Table 4.7). Mark proportions generally decreased during the season, due in part to the influx of unmarked fish during floods late in the season (Fig. 4.3). Mark proportions obtained during 1985 did not exhibit this trend because floods did not occur that year, and unmarked fish entered mostly during the brief period of partial enumeration. On a seasonal basis, marked proportions among adults in the recapture samples increased from 8.9% to 81.8% during the 1985-1988 period, while the jack proportions increased from 6% to 41%. This was mainly attributed to increased effort levels at the fence during adult enumeration and tagging, rather than improvements in trapping efficiencies.

Over the four year period, the number of fish captured during stream surveys accounted for 15% to 26% of the adult fence counts, and 4% to 35% of the jack fence counts. Stream survey samples provided much additional information on the proportion of adults and jacks in each fin clipped category. Reference to Tables 4.3 and 4.6 reveals some discrepancies between the proportions of fish lacking a particular fin as observed at the fence and during stream surveys. No significant differences were detected between the tagged proportions among males and females sampled during surveys (Table 4.7), or between the relative abundance of fin clipped adults or jacks (ad or lv) sampled at the fence and during surveys within any given season (Table 4.8). Therefore, minimum escapement levels, and estimates of the proportion of fish in each category were calculated according to Eq. 4.1 and 4.2 (Table 4.9). Some of the spaghetti tags applied at the fence in 1988 were recovered during the population surveys and the instream indexing surveys. Based on the tag recovery pattern observed during the indexing surveys, the average period of stream residency for adults was estimated at 13 d (R. Bocking, pers. comm.).

Stream surveys indicated that coho usually distributed themselves throughout the lower and middle sections within 24 h after passing the fence, but at least 48-72 h was usually required to reach the upper sections. However, low water levels and debris build-up at the fishway occasionally delayed access to the upper sections for several days. During 1985, coho moved into the lower sections relatively early in the season, where they remained in the deep pools and under log jams for approximately two weeks before moving further upstream. Between November 15 and December 15, low water levels and ice cover interfered considerably with movements within the middle sections of the stream, preventing coho from reaching the headwaters until late December. During the 1986-1988 period, coho entered the stream later in the season, and moved into the spawning areas more rapidly.

Most of the observed spawning activity took place in the upper reaches along Grafton and Winchester Roads near the junction of the East and West fork, throughout the one km section adjacent to the powerline crossing, and in areas between the fence and the river mouth. Pronounced year to year differences were observed in the degree of utilization of the spawning areas. Most of the spawning activity occurred in headwater areas during 1984, 1985 and 1988, in areas adjacent to the junction and below the fishway during 1986, and in the vicinity of the powerline crossing during 1987. Isolated spawning incidents were also witnessed sporadically throughout the accessible areas each year.

Year to year variation in spawning times was also observed. Based on the appearance and condition of the adults examined, peak spawning activity appeared to occur at an earlier date each year. Spawning activity peaked around December 17 during 1984 and 1985, around December 8 during 1986

and 1987, and in late November during 1988. It should be noted that water temperatures also increased progressively during this period, from an average of 4.2°C in 1985, to 7.5°C in 1988. Chum spawning activity was mostly confined to the lower reaches of the stream, extending from intertidal waters to the fence itself. On some occasions chum spawners were captured further upstream near the powerline. Crude estimates of chum escapement based on the densities observed indicated that less than 100 chum spawned in this stream each year.

#### 4.5 Escapement estimation models

##### Model selection

Methods for estimating animal abundance by means of mark-recapture techniques have developed considerably over the past 40 years, and are covered extensively in the literature (Cormack 1968, Ricker 1975, Seber 1982, Burnham *et al.* 1987). Several deterministic models have been proposed for estimating salmon escapement levels (Howard 1948, Chapman 1948, Schaefer 1951), and have been used in several investigations (Eames *et al.* 1983, Simpson 1984), including some on coho salmon (Pritchard and Neave 1942, Salo and Noble 1953, Eames and Hino 1981, Eames *et al.* 1981). By far, the most commonly used model for estimating salmon escapement from mark-recapture data is the Petersen model. Simpson (1984) assessed the accuracy and precision of Petersen estimates of salmon escapement. In most cases where such estimates were evaluated against known counts of fish, Simpson (1984) noted that overestimation and large variation between replicates was observed. The apparent unsuitability of this model is not surprising, since the model does not account for immigration, emigration and death which typically occur during the spawning period when censuses are being conducted.

Various modifications of the Petersen model have been proposed to cope with marking and sampling difficulties. Bailey (1951) proposed slight modifications to the simple Petersen formulae for situations where small samples are available. Schaefer (1951) recognized the difficulties associated with obtaining reliable mark-recapture estimates of salmon escapement, and devised a stratified mark-recapture model applicable to situations where fish are marked at one point along their migration route, and recovered later at different locations. Ricker (1975) noted that even under conditions where the Schaefer model is applicable, random marking or recovery would allow the simpler Petersen model to provide unbiased and consistent estimates of abundance. Accordingly, Eames *et al.* (1981) noted that in

cases where the Schaefer method was applied, the escapement estimates obtained were not significantly different than those obtained through simple Petersen models.

More sophisticated deterministic and stochastic models based on data from multiple censuses have also been proposed by Schnabel (1938), Schumacher and Eschmeyer (1943), Chapman and Junge (1956), Darroch (1961), Jolly (1965) and Seber (1965) to account for the immigration, emigration and death that can occur during the survey period. However, comparatively little use has been made of such models for escapement estimation, since the data requirements of these models often exceed what is typically obtained during a field season. In the present study, serially numbered tags were not routinely used for tagging purposes, which precluded the use such models for escapement estimation. Even in the absence of such a constraint, Minta and Mangel (1989) concluded that there is a definite need for simple, inexpensive, alternative estimators for accommodating the variety of experimental conditions. Minta and Mangel (1989) indicated that traditional probability models should be incorporated into a simulation framework for estimating population sizes from mark-recapture data.

With this approach in mind, an 'open' population model was developed specifically for estimating escapement levels in situations where salmon are marked at a fence as they enter the river, and are sampled subsequently at various times throughout the spawning period. The model is structured to account for the potential immigration of fish during floods, as well as death during the subsequent sampling period. A distinguishing feature of this model is that information on the successive recapture histories is not essential for estimation purposes. In the following section, the main attributes of the model and the estimation process are described. To provide some point of reference for judging the relative performance of the estimation procedure described, corresponding estimates were also produced by means of 'closed' population models based on pooled Petersen estimators.

#### Description of the 'open' population model

In the context of escapement enumeration, the objective of the mark-recapture operation is to calibrate the fence counts to account for fish that entered the stream undetected, by making use of marking and recovery statistics. During a typical season of field work, information is also obtained on the magnitude of the build-up of fish below the fence, the pattern of migration based on fence counts, the temporal changes in marked proportions and indices of abundances within the stream, and the salmon die-off pattern in the stream. This dataset is often too deficient to accurately estimate the parameters required by traditional models, or conduct rigorous statistical tests to detect violation of

assumptions. However, given an appropriate algorithm, this information can be used to draw inferences about the likelihood of certain events, abundance levels, and life history parameters.

The model was designed to simulate the upstream migration and the associated mark-recapture operation. As a first step, the build-up of the stream population and its subsequent decline (spawning, death) are simulated. Based on the time series of abundance generated, the proportions of marked adults expected to be found on given dates are predicted. Discrepancies between the expected (i.e. theoretical) and observed mark proportions observed on successive sampling periods are determined. Iterations are then conducted over the whole range of plausible values associated with the hypothesized escapement levels, migration patterns, and stream residency parameters. Each hypothesized combination of parameters is evaluated in terms of how well the predicted trajectory of mark proportions fits the actual pattern, and if the predicted time series of abundance conforms to the patterns observed during the season.

The sequence of calculations is best visualized by first considering all adults or jacks entering the stream on a given day ( $d$ ) as marked ( $m_d$ ) or unmarked ( $u_d$ ). Members of the latter category include all fish released without marks ( $uc_d$ ) plus those that escaped undetected ( $uu_d$ ), so that  $u_d = uu_d + uc_d$ . Given these definitions, the total fence count at the end of the season (after  $D$  days) and the total escapement to the stream are given by:

$$(Eq. 4.3) \quad Fence\ count = \sum_{d=1}^D m_d + uc_d$$

$$(Eq. 4.4) \quad Escapement = \sum_{d=1}^D m_d + uc_d + uu_d = \sum_{d=1}^D m_d + u_d$$

Assuming that the fence intercepts all upstream migrants while in operation, then all undetected fish must enter the stream during floods while the fence is collapsed. Thus, if floods occurred on days 23 to 26, then the following relationship will hold;

$$(Eq. 4.5) \quad Escapement - fence\ count = \sum_{d=23}^{26} uu_d$$

Equation 4.5 may appear to be overly simplistic at first glance, but it best describes what is essentially the initial stage in the numerical reconstruction of the run: the identification of all time periods

where undetected fish could have moved upstream, and the separate tabulation of the numbers of marked fish and unmarked fish released at the fence on each day. In the absence of a complete enumeration of all fish entering the stream, the upstream migration pattern followed by the run cannot be reconstructed with certainty. Even if the total escapement was known, it must be assumed that the undetected fish entered the stream according to some pattern if there were two or more days of flooding during the season. The migration pattern of the undetected group is best inferred on the basis of field observations. In some instance, it may appear that an equal portion of the undetected group entered the stream each day during the period of flooding (uniform upmigration pattern), while in others, field observations may indicate that they all entered at once after fence was collapsed. To facilitate the following discussion, let's assume that the undetected fish entered the stream over a four day period according to the uniform upmigration pattern. Given a hypothesized escapement level, the number of fish passing the fence on any day can now be reconstructed based on fence counts, and the hypothesized migration pattern of the undetected fish (estimated from Eq. 4.5).

Irrespective of the migration pattern followed by the fish when entering the stream, both marked and unmarked fish are considered to be subject to natural mortality as soon as they enter the stream. Preliminary estimates of the average time that adult coho remained alive in the three streams ranged from 9.0 to 13.0 d (Bocking *et al.* 1988, R. Bocking, pers. comm.). Recovery patterns of marked fish released at the fence during the 1985-1988 period indicated that adult coho could survive for as little as one day, to as much as six weeks. Such figures are comparable to those obtained by other investigators (Willis 1954, Johnston *et al.* 1986, Crone and Bond 1976), who also reported positively skewed distributions of stream residency period.

There are a number of skewed distributions (Gamma, Weibull, Exponential, etc.) which may mimic the distribution of survival times, and the appropriate model is often selected on the basis of how the instantaneous rate of death at a given time is expected to behave given that the fish has survived up to that time, i.e., from the hazard function (see Neilson *et al.* 1989). The exact shape of this function could not be determined with certainty given the lack of sufficient tag recoveries. However, the limited information obtained during the present study indicated that the instantaneous rate of mortality definitely increased up to a point during the period following the entry date, and appeared to decrease thereafter. This was explained by the fact that newly arrived fish often held in pools and under banks for a period of time before moving onto the shallow spawning grounds where they are easily captured by predators. Thus, mortality is relatively low initially, reaches its maximum during the spawning period, and tapers off thereafter as the fish remain sheltered until death.

The hazard function of the log-normal distribution does have a humpback shape, increasing to a point and decreasing thereafter. Thus, in the absence of a more suitable alternative, the probability that a fish dies after a certain period of stream residency can be predicted by reference to the log-normal probability density function:

$$(Eq. 4.6) \quad f(x) = \frac{1}{x \sigma (2\pi)^{1/2}} \exp \left[ \frac{-[\log(x/\phi)]^2}{2\sigma^2} \right]$$

where:  $f(x)$  = Probability of dying during the  $x$ th day after entering the stream;  
 $x$  = Number of days since entering the stream;  
 $\phi$  = Median stream residency period (in days);  
 $\sigma$  = standard deviation of  $\log(x)$ .

By using the following substitution,  $W = e^{\sigma^2}$ , the shape parameters  $\phi$  and  $\sigma$  are related to the more commonly used measures of central tendency and dispersion:

$$(Eq. 4.7) \quad \text{Mean} = \phi \cdot \exp 0.5\sigma^2$$

$$(Eq. 4.8) \quad \text{Standard dev.} = \sqrt{\phi (W^2 - W)}$$

For each hypothesized combination of  $\phi$  and  $\sigma$ , a survival curve can be generated from the cumulative distribution function (CDF) obtained from Eq. 4.6 over a 70 d period (Fig. 4.4, 4.5). The fraction of the population which survives after  $x$  days in the stream can be obtained directly from the survivorship curve, and is equal to 1.0 minus the integral of Eq. 4.6 from 0 to  $x$ . The number of fish still available for sampling on day ( $t$ ), which entered the stream on day ( $d$ ) is given by:

$$(Eq. 4.9) \quad m_{t,x} = m_d \cdot S_x$$

$$(Eq. 4.10) \quad u_{t,x} = u_d \cdot S_x$$

where:  $m_{t,x}$  = number of marked fish alive on day  $t$ , which arrived  $x (= t-d)$  days ago;  
 $m_d$  = number of marked fish that entered the stream on a given date ( $d$ );  
 $S_x$  = expected survival rate of fish residing in the stream for  $x (= t-d)$  days.

The number of survivors from each daily pulse is then used to estimate the total number of marked fish ( $m_t$ ) and the total population ( $n_t$ ) present in the stream on a certain day ( $X$  days after the migration began):

$$(Eq. 4.11) \quad m_t = \sum_{x=0}^X m_{t,x}$$

$$(Eq. 4.12) \quad n_t = \sum_{x=0}^X u_{t,x} + m_{t,x}$$

For each date on which stream surveys were conducted, expected mark proportions ( $m_t/n_t$ ) can be compared with the actual proportions observed (Fig. 4.6). Assuming that the stream survey samples are independent of each other, and that the distribution of recaptures conforms to the binomial, the likelihood function of the observations is given by:

$$(Eq. 4.13) \quad L(N_i, \phi_j, \sigma_k | R_1, R_2, \dots, R_t) = \prod_{t=1}^T \binom{C_t}{R_t} \left( \frac{m_t}{n_{i,t}} \right)^{R_t} \left( 1 - \frac{m_t}{n_{i,t}} \right)^{C_t - R_t}$$

- where:
- $N_i$  = Hypothesized escapement of level selected within the plausible range (the  $i$  discrete levels selected must be  $> \Sigma$  fence count,  $< \infty$ );
  - $n_{i,t}$  = Number of fish from  $N_i$  alive in the stream during the  $t$  sampling date;
  - $m_t$  = Number of marked fish from  $M$  alive in the stream during the  $t$  sampling date;
  - $\phi_j$  = Median stream life selected within permissible range ( $j = 4 - 34$ );
  - $\sigma_k$  = Standard error selected within permissible range ( $k = 0.1 - \max$ );
  - $C_t$  = Number of fish examined during the  $t$  sampling date;
  - $R_t$  = Number of fish with primary marks recaptured during the  $t$  sampling date;
  - $T$  = Total number of distinct sampling periods;

The relative likelihoods of various hypothesized parameter combinations ( $H_n$ ) can be determined by evaluating Eq. 4.13 over the grid of population sizes and parameter combinations. One particular hypothesis ( $H'$ ) will be characterized by the greatest likelihood. In the present context, mortality attributes mainly serve as nuisance parameters in the estimation of population size, and no weight is given to a particular combination of stream life parameters. Thus, the likelihood of distinct population sizes can be obtained by summing up the likelihoods over the range of  $\phi_j$  and  $\sigma_k$  values:

$$(Eq. 4.14) \quad L'(N_i) = \sum_j \sum_k L(N_i, \phi_j, \sigma_k)$$

The population size characterized by the greatest likelihood is considered as the maximum likelihood estimate of population size (MLE). The relative plausibility of alternative levels of abundance ( $N_i$ ) can be evaluated by reference to the likelihood ratio. Alternative levels of abundance were considered unlikely, if their likelihood was less than 10% of that associated with the maximum likelihood estimate<sup>22</sup>. In log-likelihood terminology,  $N_i$ 's had to meet the condition in Eq. 4.15 to be considered plausible, and  $N_i$ 's at the upper and lower limit of the permissible range were considered as the bounds of the credibility regions:

$$(Eq. 4.15) \quad \text{Log}_e \left( \frac{L'(N_i)}{MLE} \right) \geq -2.3$$

It should be noted that there exist upper and lower limits to the combinations of population attributes that can be considered as potential candidates for evaluation. All hypotheses are characterized by a series of expected values for  $m_t$  and  $n_t$ , which can be compared to the field survey results. Hypotheses that predict lower  $n_t$  or  $m_t$  than the number of fish or marked fish observed on a given date may be considered as unrealistic or invalid hypotheses. If reliable estimates of stream residency are available from tagging studies, constraints may also be imposed upon the range of  $\phi$  and  $\sigma$  values considered. In addition, if reliable time series of abundance of live or dead salmon are available from other sources (index or snorkel counts), constraints can also be imposed so that the only hypotheses evaluated are those for which the predicted trajectories of live fish ( $n_t$ ) or carcasses do not deviate excessively from the actual pattern observed in the field. This filtering process further restricts the number of hypotheses tested and improves the precision of the estimated population size. Specific examples of this process are presented in the following Section.

### Description of 'closed' population models

Given that coho were released after examination (recapture rate sampled with replacement), Bailey's (1951) binomial model was used to provide maximum likelihood estimates of escapement. Using aggregated sampling data, this estimate is equivalent to the pooled Petersen estimate, and is given by (Seber 1982):

<sup>22</sup> This procedure is analogous to the rejection method based on a two standard deviation spread in parametric testing (Edwards 1984).

$$(Eq. 4.16) \quad N = \frac{M(C + 1)}{R + 1}$$

with the variance  $V(N)$  given by:

$$(Eq. 4.17) \quad V(N) = \frac{M^2(C + 1)(C - R)}{(R + 1)^2(R + 2)}$$

and 95% confidence limits given approximately by:

$$(Eq. 4.18) \quad N \pm 1.96\sqrt{V(N)}$$

where:  $N$  = Estimate of population size;

$M$  = Total number of marked coho released at the fence;

$C$  = Total number of coho examined for marks;

$R$  = Total number of marked coho in the samples.

Escapements were also estimated by means of a sequential Bayesian algorithm recently proposed by Gazey and Staley (1986) for closed populations. According to the authors, the estimates generated by this model are comparable to those obtained by traditional mark-recapture models (Schnabel 1938, Delury 1951) when sample sizes are sufficiently large, but the estimates are more reliable under conditions of small sample sizes and low recovery rates. Although this model was not designed specifically for escapement estimation purposes (W. Gazey, pers. comm.), Bayesian models have been used in estimating population sizes (Gazey and Staley 1986, Smith 1988), including fish populations (Schmitt 1969). A complete description of the model and the estimation process is given by Gazey and Staley (1986). For reference purposes, only the mathematical structure of the maximum likelihood estimator is presented. The uncertainty associated with any hypothesized level of abundance ( $N_i$ ) given the stream survey data on hand is described by the posterior distribution of  $N$  given the sequence of recoveries ( $R_1, R_2, \dots, R_k$ ):

$$(Eq. 4.19) \quad P(N_i | R_1, R_2, \dots, R_k) = \frac{\prod_{t=1}^T \left(\frac{1}{N_i}\right)^{R_t} \left(1 - \frac{M_t}{N_i}\right)^{C_t - R_t}}{\sum_{i=1}^K \prod_{t=1}^T \left(\frac{1}{N_i}\right)^{R_t} \left(1 - \frac{M_t}{N_i}\right)^{C_t - R_t}}$$

where:  $N_i$  = Hypothesized escapement level ( $\geq \Sigma$  fence counts);  
 $M_t$  = Total number of primary marks released before the  $t$  sampling date;  
 $C_t$  = Number of coho examined during the  $t$  sampling date;  
 $R_t$  = Number of primary marks recaptured during the  $t$  sampling date.

The prior probability  $P(N_i)$  is assumed to be uniform over the hypothesized range of discrete escapement levels ( $i$ ) selected. By solving Eq. 4.18 over the range of  $N_i$ 's after each sampling event, a new posterior distribution is generated. This posterior distribution can be generated successively after each sampling event, in order to determine if the abundance estimates are stabilizing (Fig. 4.7). The final posterior distribution serves to identify the maximum likelihood estimate and the associated credibility region. This region is defined as the interval between  $a$  and  $b$  such that  $P(a \leq N \leq b) = 1 - \alpha$ , where  $\alpha = 0.05$ . Values for  $a$  and  $b$  were determined from the highest probability density region (HPD) such that  $b - a$  was minimal.

#### Model assumptions

There are fundamental differences between the underlying assumptions of Bayesian and non-Bayesian inferences. It is beyond the scope of this study to review these in detail. Further discussion will only focus on the underlying assumptions of open and closed population models as they related to populations sampled.

The major difference between the two categories of models concerns the issue of closure; that is whether or not mortality and migration are operating in the population sampled. In the present context, geographic closure is assumed to prevail since emigration was rare, but the assumption of demographic closure was definitely not met. The open population model is particularly well suited for dealing with death and immigration, but Petersen type estimators are particularly sensitive to violations of this assumption. The assumption of no natural mortality can be relaxed to a certain extent when applying closed models, if mortality is assumed to operate equally among marked and unmarked fish. Under such conditions, Chapman and Junge (1956) showed that Petersen type estimators are not affected significantly. Immigration is not always easily quantifiable given the low recovery efforts (as in the present cases), and can only be indirectly dealt with by relying only upon samples collected after the migration period for estimation purposes.

With regards to emigration, none of the models presented are structured to account for the departure of marked and unmarked fish from the stream during floods. As a result the estimates generated by either model will most likely be biased to a certain extent, unless the magnitude of the emigration can be quantified. With Petersen type estimators, emigration can be indirectly accounted for by making adjustments to the number of fish enumerated and marked, and estimating abundance on the basis of stream survey data collected after the emigration has occurred. If the open population model is used, it is preferable to adjust all values of  $n_t$  and  $m_t$  after the flood to reflect the departure of the fish at a given time.

Petersen models and the Bayesian estimator used also assume that all fish have the same probability of being caught in the first sample (i.e., at the fence). This assumption is nearly always violated to some extent, since it is mainly the fish arriving early that get marked, as late fish often escape undetected during the floods. The open model is not affected by this violation, but Petersen estimators will be affected to some extent depending on the timing and magnitude of the undetected migration. Finally, it should be noted that there are a few assumptions which must be satisfied by both types of models described above, namely:

- a. Marking does not affect the subsequent catchability or the distribution of the fish;
- b. Sampling during stream surveys is a random process, and all fish are equally susceptible to capture.  
Since jacks are less susceptible than adults to being caught at the fence and during subsequent sampling, jacks and adult abundances were estimated separately;
- c. Fish do not lose their marks prior to sampling for recoveries;
- d. All marks observed during stream surveys are reported;
- e. All coho are distributed at random throughout each stream;
- f. Marked and unmarked coho, as well as members of each sex, have similar rates of natural mortality during their period of stream residency.

#### Estimation procedure

As an initial step in the estimation process, it is necessary to compile an appropriate set of statistics for vectors  $M$ ,  $C$ , and  $R$ , in order to compensate for some sampling deficiencies and to ensure that the model requirements are met. For this purpose, the following adjustments were made to the stream survey data before estimating population sizes.

Adjustments were made to the total number of marked fish released to reflect the potential number of marks available for sampling at various locations. For each sampling event, the total number of marked fish available for sampling ( $M$ , or  $M_t$ ) was set to the sum of the primary marks released up to the day earlier for samples taken in the lower reaches, two days earlier for samples taken in the middle sections, and three days earlier for samples taken in the headwaters. This adjustment provided an allowance for the time required by marked coho to distribute themselves throughout the various sampling sites after being released at the fence.

In some instances, census data obtained from various sites within a 48 h period were combined to produce a more representative estimate of the mark proportion in the population within that time period, and to minimize the possibility of bias induced by using small samples<sup>23</sup>. Individual samples with less than four fish or three recoveries were combined with the sample taken during the nearest time period ( $< 4$  d). When combining samples of unequal size, the sampling date of the largest sample was considered as the actual sampling date.

Finally, the number of fish examined, and the number of marked fish recovered during each stream survey, was adjusted to account for the condition of the fish sampled. For the closed population models, there is no theoretical justification for omitting fish recovered as carcasses from the sampling data. In fact, the likelihood of violating the assumption of closure is minimized by including all identifiable carcasses in the sample dataset. This has the additional advantage of making both estimates of  $C$  and  $R$  larger, thus minimizing the chances of bias. For the open population model, the samples only include fish captured live, since dead fish do not serve to determine the marked proportions in the stream at various times. As a result, two distinct recapture datasets were generated for each stream/year combination ( $L$  and  $L+D$  categories, Tables 4.10 to 4.12).

Bayesian estimates of abundance were generated according to the procedure described by Gazey and Staley (1986). The suitable range of population sizes for the prior distribution was usually determined from repeated estimation trials. The lowest escapement hypothesized ( $N_{\min}$ ) was set to the minimum escapement estimate (Table 4.9). The largest hypothesized escapement ( $N_{\max}$ ) was initially set to five times the suspected level of abundance.  $N_{\max}$  was then adjusted if necessary after a cursory examination of the posterior distribution curve, so that the probability levels associated with  $N_{\max}$  and

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<sup>23</sup> As noted by Ricker (1975), in situations where  $M/N$  is small, the probability that the confidence intervals of  $R$  include 0 is less than 5% if the observed  $R \geq 3$ . Pooling samples can ensure that this condition is met, and reduce the possibility of bias.

$N_{\min}$  were negligible compared to that of the mode. For each simulation, 400 hypothetical population sizes were evaluated within the final range selected.

Census data obtained prior to major pulses of immigration or emigration were not used for estimation with the closed population models, to minimize violation of the closure assumption. Such pulses were detected or confirmed by fence counts (immigration), the recovery of marked fish in other streams (emigration), or as suggested Gazey & Staley (1986), from the visual examination of the successive posterior distributions generated (Fig. 4.7). Chi-square testing of census data was also conducted to test for changes in abundances, but these proved to be of limited use, due to the small or unequal sample sizes available. Omitting some earlier samples was necessary only in few instances (Tables 4.10, 4.11), since the migration pulses usually occurred early in the season, prior to the major sampling period.

Estimation by means of the open population model is a more complex and interactive process. Initially, a migration pattern must be selected to mimic the pattern followed by the undetected fish which enter the stream while the fence is collapsed. Based on the field observations, it was assumed that all undetected fish entered the stream prior to Dec. 15, and followed either one of two hypothesized patterns. In the first pattern, a decreasing fraction of the undetected population entered the stream each day, such that at least 50% of the fish move upstream during the first two floods. This resembles the migration pattern observed at French Creek and Black Creek, when high flows precede a period of fish build-up below the fence. For the second pattern, equal numbers of fish were assumed to move upstream each day, such that the entire undetected population has entered by the end of the last flood. This corresponds roughly to the pattern observed at Trent River during periods of high flow rates when the fence was still intercepting fish. Simulations were conducted using both upmigration patterns, and the pattern resulting in the greatest likelihood was considered to be representative of the actual migration pattern.

During a typical estimation trial with the open model, 30 increments in population sizes were tested between the lowest and highest abundance levels hypothesized. Realistic bounds for the shape parameters  $\phi$  and  $\sigma$  were chosen so that less than 1.5% of the fish entering the stream on any given day remained alive longer than 70 d. To facilitate computations, the parameters were only allowed to take discrete values ranging from 4 d to 34 d, and 0.1 to 1.0 respectively. Each  $\phi$  value had an associated range of permissible  $\sigma$ 's; the entire range of  $\sigma$  values was used for low  $\phi$  values (< 6 d), the range was decreased progressively for intermediate  $\phi$  values, and was lowest (0.1 to 0.36) for high  $\phi$  values (> 30). For each population size tested,  $\phi$  was increased by one unit from 4 to 34, and for each  $\phi$  value,  $\sigma$

was increased by at least 0.025 units, from 0.1 to the maximum allowable value. In some instances, the increments and limits selected were to a large extent dictated by the amount of contrast in the output generated. Examination of the contrast between the likelihood contours generated indicated if the level of incrementation used for increasing the parameter values was adequate for proper identification of the best fitting hypothesis.

Initial simulation trials revealed that limits on population size at the end of the sampling surveys were required in some cases to eliminate hypothesized sets of parameters producing unrealistic outcomes. Field observations suggested that less than 5% of the population remained alive by Dec. 30 each year, irrespective of the timing of the run. Thus, for a hypothesis to be acceptable, the number of live fish remaining in the stream by December 30 had to be less than 5% of the Bayesian estimate of abundance. In addition, it was necessary to select an appropriate time limit beyond which no substantial migration of undetected fish occurred even if the fence was collapsed. Based on field observations, it was assumed that the migration generally ended during the first week of December. In some instances, the periodicity of floods and/or field observations dictated that slightly shorter or longer time limits be selected for the termination of the migration date. It should be noted that in the majority of cases, imposing such limits had little effect upon the estimates.

Additional data obtained from instream indexing censuses (Bocking *et al.* 1988) were also used for escapement estimation if sufficient censuses had been conducted. Since standard effort levels were used throughout these surveys, it was assumed that the time series of live counts obtained by the surveyors were indicative of the relative changes in abundance through time. Thus, the ratio of the live counts obtained on successive dates by the surveyors were used to determine relative changes in abundance between the survey periods. All hypotheses which predicted changes in abundances levels exceeding the calculated ratios by  $\pm 15\%$  were rejected. This figure was selected on the basis of a subjective assessment of the reliability of the indices of abundance.

#### 4.6 Estimates of population size

##### Stream population estimates from open population model

An examination of the best fitting parameter values obtained for each case reveals little intra-stock or inter-stock consistency in stream residency estimates for the 1985-1988 period (Table 4.13). No evidence was obtained to indicate that the period of stream residency was consistently greater for one

stock, stream, or age group. No strong relationship was found between the point estimates of  $\phi$  and  $\sigma$  across all years for any given population. There was a tendency for the median period of stream residency of the adult populations at French Creek and Trent River to follow a similar trend over the years. It should be emphasized that the stream residency parameters estimated by this method are characterized by a substantial amount of uncertainty. In several cases, the range of the mean period of stream residency associated with the good fitting hypotheses was very broad (4 - 35 d). Using additional data from snorkel surveys and index counts to filter out hypotheses which predicted substantially different time series of abundance usually resulted in a narrower range of the mean period of stream residency acceptable (Pop. size limits = Yes, Table 4.13).

The range of the estimates of mean residency period for each case occasionally overlapped, and usually exceeded the estimates of average stream residency obtained by means of spaghetti tags during 1988 (9 - 13 d), but were comparable to some estimates obtained by means of opercular punches at the Little Qualicum spawning channel during 1987 (15-28 d, unpublished data). The possibility exists that the figures obtained by means of external tags underestimated the average stream life owing to the physical effects of spaghetti tags, and the uncertainty associated with the estimation process based on depletion curves (see Bocking *et al.* 1988). Alternatively, both the point estimates, as well as the range of means obtained from the open model, may not accurately represent the actual values. The range could be narrowed considerably, and the point estimates improved, if accurate empirical estimates of the variation in stream residency were available for various components of the run. In the absence of any additional information to account for the discrepancies between various estimates of stream residency, and to justify the use of a particular range of values, the likelihood of escapement levels was inferred based on the procedure outlined previously.

The MLE of abundance was similar to the abundance level associated with the best fitting hypothesis ( $H'$ ), and the discrepancies between both estimates was <25% of the largest estimate. The discrepancies were attributed to the fact that the level of abundance associated with the best fitting parameter combination is not always the most frequent level of abundance among acceptable hypotheses, as well as the level of resolution used in the grid calculations. Simulations conducted with constraints generally produced less discrepancy between the two levels of abundance, owing in part to the elimination of hypotheses with substantially different trajectories of abundance. The estimates of abundance obtained when constraints were imposed also tended to have comparatively narrower bounds. This may be an indication of the 'consistency' of the estimation procedure; as more information is used in the estimation process, the likelihood curve becomes progressively narrower without shifting substantially along the axis or converging towards different values.

### Stream population estimates from closed population model

Bayesian estimates of adult abundance were generally similar to the adjusted Petersen estimates (Fig. 4.8). Bayesian estimates of jack abundance always exceeded those obtained from Bailey's model (data not presented for purposes of brevity). No obvious explanation was found to account for this observation. However, it was noted that in all cases where major discrepancies were observed between the two closed population estimates of jack abundance, the number of marked jacks recovered was very small ( $\leq 5$ ). Theoretically, the Bayesian estimates should be more reliable under such conditions (Gazey and Staley 1986), and relatively large discrepancies should be observed.

### Comparison of population estimates from various models

On an overall basis, Bayesian estimates of adult abundance were only marginally closer to the open model estimates. The later estimates were usually lower than the Bayesian estimates, with the ratio of these two ranging from 0.45 to 1.13. The discrepancies between estimates were smallest at Black Creek where marking was generally conducted throughout the run, and substantially larger and less consistent at Trent River, where fence operations were plagued by frequent and lengthy flood periods. For both age groups, the lower and upper bounds of the Bayesian estimates were usually wider than those of the open model estimates (Fig. 4.9). Additional simulations revealed that in some cases (Black Cr. 1985, Trent R. 1986), abundance levels at the upper bound would only be possible if the populations were characterized by combinations of stream residency parameters that were unrealistic or that changed substantially within the season. In addition, under such scenarios, the number of fish dying at given times would not conform to any pattern observed during the stream surveys. This suggests that the credibility regions associated with Bayesian estimates were in some instances unrealistically wide.

Assuming that the estimates generated by the open model are correct, then it is possible to obtain crude estimates of the fraction of the escapement enumerated, marked and examined for marks each year. This fraction enumerated ranged from 12% to 95% for adults, 30% to 73% for jacks, and averaged approximately 48% for both age groups across all cases. The proportion marked was usually slightly less than the proportion enumerated, and the fraction examined for marks was generally less than 10% of the total escapement.

Simple linear and multiple regressions revealed no strong relationship ( $r^2 < 0.3$ ) between the natural logarithm of the ratio of estimates (MLE with or without constraints / Bayesian), and the

following log transformed variables: the total count, the number of marks applied, the number of fish examined for marks, and the number of marks recovered, expressed as actual figures or as fractions of the total escapement. However, the average discrepancy between estimates was greatest at Trent River where, on average, 38% of all adults were enumerated (range: 27-49%), smallest at Black Creek where 70% of adults were enumerated (range: 48-90%), and intermediate at French Creek where 62% were enumerated (range: 52-84). This indicates that when most of the migration occurs within a relatively short period of time, and where marks are applied throughout the run, closed population models can provide estimates that are most similar to those obtained by the open model.

In the absence of complete counts of all upstream migrants, and of better open population models for generating comparable estimates of abundance, the relative performance of the open population model can only be inferred on the basis of theoretical grounds, and by reference to the other estimates of abundance. First, the open population model accounts for mortality and immigration, which are known to occur prior to and during the sampling period, and to have complex patterns during this period. Second, Petersen type estimators generally overestimate salmon escapement levels (Simpson 1984), and therefore it is likely that the closed population models used in this study also overestimated the actual escapement for both age groups. As a third consideration, it must be noted that the majority of escapement estimates derived from index surveys conducted in these streams since 1986 were lower than the mark-recapture estimates based on the Bayesian model (Johnston *et al.* 1987, Bocking *et al.* 1988, Bocking, pers. comm.). Finally, in each case where a high proportion of the run was enumerated and marked (Black Cr. 1986-1987, French Cr. 1988), MLE's were closer to the observed escapement than the Bayesian estimates. Thus, based on the circumstantial evidence presented, it is hypothesized that the open population model provides estimates that are more reliable than those obtained with closed population models.

#### Estimates of jack abundance based on jack-to-adult ratio in samples

To determine the level of jack abundance in cases where an insufficient number of recoveries was available to use the previous models, estimates were generated by reference to the jack-to-adult ratio in the stream. The relative abundance of jacks in each stream was first estimated as follows:

$$(Eq. 4.20) \quad (J/A)_{esc} = \left( \frac{\sum \text{count}_{jacks,f} + \sum \text{count}_{jacks,u,s}}{\sum \text{count}_{adults,f} + \sum \text{count}_{adults,u,s}} \right)$$

where:  $\text{count jacks}_f$  = Daily count of jacks released the fence adjusted for emigration losses;  
 $\text{count jacks}_{u,s}$  = Count of unmarked jacks obtained during each survey;  
 $(J / A)_{esc}$  = Jack-to-adult ratio in the escapement.

Estimates of jack abundance were obtained by multiplying each ratio with the associated MLE's of adult abundance. The jack-to-adult ratios were assume to follow a binomial sampling distribution. The 95% confidence intervals associated with each ratio were extracted from the appropriate tables and considered as the upper and lower bounds of the ratios. Estimates of the lower and upper bounds of jack abundance were obtained by multiplying the bounds of each ratio with the associated MLE's of adult abundance.

Reliable estimates of jack abundance can be obtained by this method if jacks and adults are equally susceptible to capture during the sampling process. Given the apparent lower detectability of jacks at the fence, and their lower catchability during stream surveys, the estimates generated by this model are expected to be negatively biased. Accordingly, a comparison of the figures in Tables 4.13 and 4.14 reveals that the estimates obtained from Eq. 4.20 were always lower than their corresponding mark-recapture estimates. In addition, estimates of the minimum number of jacks in the stream (Table 4.8) occasionally exceeded the estimated lower bound. Therefore, the estimates obtained by this method were only relied upon in the absence of alternative estimates of population size (Trent River 1986, 1987, and French Creek 1986). For such cases, no attempt was made at correcting the estimates to account for potential bias, given the lack of information on the magnitude of the potential differences in detectability.

#### Estimates of spawning populations and stream escapements

Total escapement to each stream was determined for each age group by adding the number of fish retained at the fence to the estimated stream populations. Spawning population estimates were obtained by deducting the total number of fish collected in the stream from the estimated stream population (Table 4.15). Figures obtained by this method may be slightly biased in some instances since the exact number adults collected in each stream by club members was not always known. No attempt was made to estimate the number of fish that died before spawning, but field observations suggested that at least 5% of the females died before spawning due to predation, injuries and physiological disorder.

Crude estimates of the escapements to each stream during 1984 were made on the basis of the relative abundance of fish on the spawning grounds as observed during the stream surveys. At Black Creek and French Creek, adult densities at all sites sampled during the spawning season appeared to be similar to those observed in 1985, and definitely greater than those observed in subsequent years. At Trent River, adult coho densities in 1984 appeared to be slightly less than those observed in 1985 and 1986. Thus, the 1984 escapements to each stream were estimated to be similar to the 1985 levels. Jack abundance in each stream during 1984 was also estimated to be similar to the 1985 levels.

A cursory examination of the figures presented reveals some trends in adult escapement patterns from year to year. During the course of this investigation, escapement levels at Black Creek and French Creek showed a pronounced drop during 1987, followed by a partial recovery in 1988. By contrast, escapement levels to the Trent River remained fairly stable during the 1985-1988 period. To determine the significance of the trends observed, information on the contribution of strays to each run, and the previous smolt outputs must be examined (Section 5).

#### Discussion of the results obtained

Field observations suggested that selectivity during the marking stage was probably the most important factor affecting the performance of closed population models. In anticipation of floods, a large number of marks were usually applied during the early part of the run when water levels were low and fish were readily available. Large pulses of unmarked fish entered the streams during floods and diluted the mark ratios. Because of the time delays between the entry of the enumerated group and the undetected group, marked fish appeared to die sooner than unmarked fish which entered predominantly later in the season while the fence was collapsed. The reduction in the mark ratios is therefore a function of both processes operating simultaneously (immigration and differential mortality). However, in the absence of information on the latter process, the reduction in mark ratio is attributed solely to immigration, which leads to an overestimation of abundance.

The magnitude of emigration as observed at Trent River must also be considered as a major factor affecting the performance of any mark-recapture model. Emigration of adult coho after marking has been reported in other coastal streams where mark-recapture operations were conducted (N. Schubert<sup>24</sup>, pers. comm.). Whether or not the departure of some marked coho was induced by the

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handling process is not known with certainty, but given that salmon can 'visit' various streams before spawning ('proving' behavior, Ricker 1972), it is doubtful that the marking process itself induced all the emigration reported. It could be hypothesized based on their location of capture that these emigrants were mainly strays from the Puntledge River. As noted previously, strays from this stream tended to remain in the lower section of the Trent River. Interestingly, this is the only system which had a fence located in intertidal waters. If the fence had been located further upstream as in other systems, the magnitude of the emigration might have been lower than reported.

The emigration of marked individuals is difficult to quantify since the vast majority of small coastal streams in this region are not surveyed for this purpose. Thus the number of emigrants reported most likely underestimates the actual loss of marks from the system, resulting in the overestimation of the number of fish present, unless there is a greater tendency for unmarked fish to emigrate. Field observations also revealed that the majority of emigrants were among the earliest fish marked, and emigrated during the first flood. Thus, potential losses of marked coho could be reduced by releasing the early migrants without prior marking. This may also help improve the overall survival rate of marked fish, since early migrants are most susceptible to handling and often die while holding in intertidal waters prior to the first period of high water (Hamilton 1978). This point highlights the major uncertainties associated with the estimation process, namely, the assumptions about marked and unmarked fish having identical rates of natural mortality and stream residency periods. The validity of these assumptions are questionable given the above facts, but this approach was necessary given the lack of empirical data on the magnitude of differential mortality between various segments of the run.

Additional simulations have shown that the range of stream residency parameters used can have a profound influence upon the escapement estimates generated by the open model particularly in situations when stream surveys cannot be conducted in sufficient numbers and at regular intervals throughout the season to reveal clear trends in mark proportions. Ironically, it is exactly under this type of condition that stream residency is most difficult to estimate from census data, since the die-off pattern cannot be estimated with certainty from typical depletion curves. This problem is further compounded by the fact that preliminary results from ongoing indexing studies conducted in these streams suggest that stream residency may vary considerably between streams in a given year (Dr. J. Irvine, pers. comm.). Such results are in agreement with the those obtained during this study, which implies that estimates of stream residency of neighboring populations cannot be relied upon for estimating escapement levels in streams which could not be sampled adequately. When reliable information on mortality rates is lacking, abundance levels can be estimated by the method proposed, but ideally, empirically derived hazard functions should be relied upon instead of the theoretical die-off pattern used. This point emphasizes the

need to obtain further information on the stream residency patterns of this species, as well as on the differences exhibited by different segments of the run.

#### 4.7 Comparison of run characteristics across all systems

A cursory examination of escapement levels to all streams (Table 5.6) revealed that adult escapements to systems with and without major enhancement facilities varied by as much as an order of magnitude during the course of this study. However, adult escapements did not appear to vary from year to year in a synchronized fashion across all streams or across streams in each category. This is not surprising given that escapement levels are influenced by a multitude of factors including smolt output levels, jacking rates, straying rates, ocean survival and fishery exploitation. Thus, a meaningful assessment of the co-variation in escapement levels among stocks must account for these factors, and should ideally be based on CWT recovery statistics. Therefore, this analysis was conducted in Section 5.4, and only variation in run timing and spawner size are covered here.

##### Run Timing

The number of adult coho that escaped undetected to Black Creek, Trent River and French Creek was estimated from the difference between the total escapement estimates and the fence counts. The actual migration pattern of each run was reconstructed, by assuming that the undetected fish entered each stream according to the hypothesized migration pattern which provided the best fit for the open population model (uniform or decreasing, Table 4.13). The reconstructed migration patterns of these populations were then compared with those observed in other systems where escapement enumeration was conducted by means of permanent counting facilities. Substantial variability in upstream migration patterns between years and populations is apparent for the 1985-1988 period (Fig. 4.10). For comparative purposes, the calendar date on which 5% of the run had entered the stream was considered as the date on which the run started. The median date of migration (day on which 50% of the run was achieved) was considered as a measure of run timing. The number of weeks required for 95% of the run to be complete (2.5% - 97.5%) was considered as the duration of the upstream migration (Table 4.16).

The upstream migration started, on average, earlier in 1985, and latest in 1986-1987, with the average year to year difference ranging from one to three weeks. A two factor ANOVA test with no replication showed significant differences in starting dates among years and among populations ( $F_{\text{year}} = 8.8$ ,  $F_{\text{pop.}} = 14.7$ , both  $P < 0.001$ ). On average, the migration in large rivers (Quinsam, Puntledge, B.

Qualicum, L. Qualicum) started two to five weeks earlier than in smaller streams. No significant differences in starting dates were detected among years or among populations in comparisons involving only large rivers. The same results were obtained in comparisons involving only smaller streams. These results suggest that coho must have been present in nearshore waters each year prior to the start of the upstream migration, which began as soon as water levels were sufficiently high for stream access. At Quinsam River, Puntledge River, Little Qualicum River and Big Qualicum River, sufficient flows exist early in the season due to flow controls, but the remaining streams lack flow control, and stream discharge was affected to a large extent by the amount of precipitation. Field observations indicated that amount of precipitation at various locations often differed on a given day as a result of localized weather patterns, but large frontal systems affected all streams simultaneously. Information on the amount of precipitation and stream discharge was not available for all locations, which precluded further analysis of stream discharge patterns. However, the starting dates of the migrations at Black Creek, Trent River, Rosewall Creek, French Creek, and Millstone River for the 1985-1988 period were found to be highly correlated (range of  $r$  values in correlation matrix: 0.58 - 0.94).

Similar trends were observed with respect to the median date of upstream migration (i.e. run timing). The median date of migration occurred, on average, three weeks earlier in 1985 (calendar day 297) than in 1986-1987 (calendar day 316). The median date of upstream migration, averaged across all years, was earliest at Big Qualicum, Puntledge and Quinsam Rivers (range: 302 to 306), and latest at Black Creek, Rosewall Creek, and French Creek (range: 312 to 320). Significant differences in run timing were not detected among populations ( $F = 1.748$ ,  $P = 0.143$ ), but were detected among years for streams lacking flow control ( $F = 15.8$ ,  $P < 0.001$ ). This result can be explained by the fact that in streams lacking flow control, the median date of upstream migration usually coincides with the first freshet, which can vary substantially from year to year (Section 4.1).

On average, the duration of the upstream migration period was shortest in 1986 and 1987 (6 weeks), longest in 1985 and 1988 (7 - 8 weeks), and tended to be shorter for small systems such as Black and French Creek (mean = < 4.5 weeks) than large systems such as the Quinsam, Puntledge and Big Qualicum (means = 8.8 - 9.5 weeks). Both factors (year and population) were found to have a significant effect upon the duration of the runs ( $F = 4.4$  and  $16.96$  respectively, both  $P < 0.01$ ). Differences in run duration are mainly induced by differences in starting dates, since most runs usually ended at approximately the same time each year (Fig. 4.10).

### Spawner size

Samples of length measurements (post-orbital hypural) obtained by hatchery personnel at each facility were used for comparative purposes. In the majority of cases, the length data obtained were not associated with particular tag codes, so wild, colonization and production groups could not be distinguished. However, some length data obtained at the Quinsam hatchery in 1986 and 1987 were associated with particular tag codes. The average size of adults from colonization groups was approximately three cm greater than that of adults from production groups in both years. The average size of wild adults in 1987 was about one cm larger than production fish, and two cm less than that of colonization fish. Since the sample sizes of length data for colonization and wild groups were generally small, length data for production, colonization and wild fish returning to each hatchery were combined for comparative purposes. The sizes of adults and jacks in these systems could be compared to those obtained during population surveys conducted at the remaining locations (Fig 4.11). Mean lengths for jacks ranged from 21 to 26.9 mm, while the mean lengths for adult males and females ranged from 40.0 to 52.4 mm, and 43.9 to 54.1 mm respectively. Within each population, the average size of males was equal to or smaller than that of females, with the ratio of the two mean sizes ranging from 0.89 to 1.02 across all populations and seasons.

During the course of this study, the average size of jacks increased by approximately four cm. Regressions of the natural logarithms of mean jack size on mean smolt size were generated for the 1985 to 1987 period, since smolt size data for 1988 were not available for all populations. A significant relationship was found between jack and smolt sizes for the 1986 jack return year ( $r^2 = 0.55$ ,  $F = 11.0$ ,  $P = 0.01$ ), but not over all seasons ( $r^2 = 0.12$ ,  $F = 0.55$ ,  $P = 0.38$ ). Since no samples were obtained at Rosewall Creek in 1985 and 1988, these groups were omitted from further analyses. Significant differences in average jack size were detected between seasons and populations (Two-way ANOVA,  $F$  values for main and interaction effects = 101.1, 6.9 and 14.6, both  $P < 0.001$ ). No particular population or group was found to be consistently larger or smaller than others, and no particular group of jacks returning to Rosewall Creek in 1986 and 1987 was consistently larger or smaller than average in mean size. However, jacks returning to the Rosewall Creek hatchery tended to be smaller than average in mean size, while those escaping to three large public hatcheries were generally above average.

Adult males and females exhibited a substantial reduction in average size during the 1985-1988 period. Adult male and female measurements from the same escapement were combined. On average, the mean size of adult males and females decreased by about 5.0 cm during the 1985 to 1987 period, and increased slightly in 1988. Linear regressions of the logarithms of mean adult size on mean smolt size

were generated for the 1986-1988 period, since smolt size data for 1984 were not available for all populations. A significant relationship was found between smolts and adult sizes for the 1988 return year ( $r^2 = 0.6$ ,  $F = 14.0$ ,  $P = 0.005$ ), but not over the entire 1986-1988 period ( $r^2 = 0.05$ ,  $F = 1.46$ ,  $P = 0.238$ ). Since no adult samples were obtained at Rosewall Creek during 1985 and 1986, these groups were omitted from further analysis. No particular population or group was consistently below or above average in mean size. However, Black Creek adults were often above average, while adults returning to Rosewall Creek in 1987 and 1988 tended to be smaller than average. No particular group of adults returning to Rosewall Creek in 1987 and 1988 was consistently larger or smaller than average in mean size. A three-way fixed effect Anova test was conducted to assess the effects of sex, year and stream on adult size. All the main factors were found to have significant effects on the average size (Table 4.17). Scheffe's multiple contrast tests revealed that adults from Black Creek had the largest overall mean size, which was significantly different from the average adult size of the remaining streams (at  $\alpha = 0.05$ ). All interaction effects were also significant, and indicated that stream effects were influenced by the sex and the season. The year\*sex interaction was only marginally significant, indicating that both sexes responded similarly to the season effects.

Ricker (1981) noted that the average weight of coho caught by commercial fisherman in the Strait of Georgia did not change substantially over the 1951-1975 period. He hypothesized that selective effects of the fishery may have been masked by the production of larger hatchery fish. Such selection pressures would cause the average size of spawners to decrease since the larger fish reach legal sizes sooner and are exploited over longer periods of time. Presumably, greater fishing efforts would induce a greater reduction in average size. The trends in average size observed during this study appear to be related to trends in exploitation rates (Section 7.2) which increased from 1986 to 1987, and decreased from 1987 to 1988, as did the average size of adult coho.

## 5.0 CWT RECOVERY PATTERNS IN ESCAPEMENTS

To measure exploitation rates, survival rates and straying patterns, estimates of total escapement of tagged fish must be obtained for each coded wire tag release group. In obtaining these estimates, consideration must be given to the sampling regimes used in each stream monitored, the identity of the tag codes recovered, tag rejection rates, and sources of error associated each of these factors. In this Section, the various sources of information used to obtain the estimates are described, and a detailed account is given of the methods used to estimate the tag recoveries in the escapement.

### 5.1 Description of data sources

Statistical records relating to the release and recovery of coded-wire tagged salmon in Canada and the U.S have been collected since 1970 through DFO's Mark-Recovery Program (MRP). These records are stored on a large database at DFO's Pacific Biological Station in Nanaimo. This database (MRP Database) currently serves as the main source of information to fishery managers and researchers conducting investigations on Pacific salmon. The structure and contents of the database, and the computer programs available for data extraction, have been described in detail (Holmes and Hamer 1988, Kuhn 1988, Kuhn *et al.* 1988). Information contained in this database serves as the main source of analytical data for the present study.

Currently, the MRP database includes three complementary datasets, which contain biological and statistical data associated with the release, recovery and sampling processes. All statistics associated with coded-wire tag releases, and tag recoveries in the escapement are supplied by hatchery managers and various investigators conducting field surveys. Tag recoveries in various fisheries are obtained by sampling the commercial catches on a coast wide basis, and from fish heads returned voluntarily by sport fishermen. Tagged heads recovered from each source are usually sent to the Head Recovery Laboratory for decoding, along with information on the location and date of recovery, the sampling rates (if available), and associated biological measurements. Tag code identities and the associated data are then submitted to DFO personnel for verification and data entry. Each source of recovery data has an associated level of stratification (spatial/temporal) and data types used for reporting. The terminology proposed by Kuhn *et al.* (1988) will be used to describe the sampling and reporting of CWT data.

The term 'observed recoveries' applies to the actual number of tags of a given code recovered from a sample. Usually, a fraction of the fish which appear to be tagged in a given sample are not identified. These 'non-tags' are grouped into three categories. A head which contains a tag that is lost before being decoded is referred to as a 'lost pin'. A head which was dissected but contained no tag is considered as a 'no pin'. These include fish that rejected their tags following tag implantation, that were sampled erroneously, or that were misidentified because they naturally lacked an adipose fin. The number of 'no data' refers to the number of fish in a sample that lacked an adipose fin, but no attempt was made to extract the tag. When the number of fish naturally lacking an adipose fin is negligible, and there is no sampling error, the number of tagged fish present in the sample is estimated from:

$$\text{(Eq. 5.1)} \quad \text{Tagged fish} = K + ND + LP + NP$$

where: LP = Number of 'lost pin' present in the sample;  
 NP = Number of 'no pin' present in the sample;  
 ND = Number of 'no data' present in the sample;  
 K = Number of identified tags present in the sample ( $\Sigma$  obs. recov.).

The number of 'non-tags' (LP+NP+ND) in escapement samples is generally substantial, and must be taken into account when estimating the total number of tags of each code. The specific method used to account for the 'non-tags' depends on the tag recovery source, and the type of estimate required (Kuhn *et al.* 1988). Usually, only information on the tag composition of a sample is used for this purpose. In some instances, ancillary information on the particular attributes of the tagged fish, or estimates of tag loss associated with certain codes, can help to associate 'non-tags' with specific codes.

Once the number of 'adjusted recoveries' has been determined, the total number of tags of each code recovered in a given stratum can be estimated based on the sampling regime. Tagged heads are often submitted to the Head Recovery Laboratory without information on the total catch or escapement in that stratum ('select' samples). In such instances, the number of observed recoveries in that stratum is considered as an estimate of the minimum number of tags present. Alternatively, the sampling rate may not be known with certainty, as in sport fisheries where heads are recovered through a voluntary program, and ancillary information must be used to estimate the sampling rate and the associated number of recoveries. When sampling rates are known with certainty ('random' samples), as in most escapement and commercial fishery strata, the total number of recoveries can be estimated directly. Irrespective of the procedure used, expanded estimates of the tags recovered in a stratum are referred to as 'estimated recoveries'.

Thus, in order to properly estimate the number of tagged fish in various strata, consideration must be given to the particular attributes of the recovery data on hand, and the characteristics of the sampling regime used. In the present study, the estimated recoveries associated with each tag code were derived primarily on the basis of CWT recovery and sampling statistics contained in the MRP database. Given that considerable time is required to enter data from ongoing studies into this database, and that some inconsistencies can occur during the data entry and reporting phases (Kuhn *et al.* 1988), all MRP records were verified by comparing the MRP database statistics to those obtained from the juvenile and adult samples obtained at each site, the actual hatchery records, and the information obtained from interviews with hatchery managers. All discrepancies observed were reported to C. Cross<sup>25</sup> for additional verification. If the discrepancies could not be resolved, efforts were made to identify the most reliable source of data for subsequent analysis. In some cases, information recently obtained from ongoing investigations was also incorporated into the analysis, and used to make further adjustments to the MRP statistics. As a result, all CWT release and recovery statistics presented here consist of the corrected MRP statistics, which may differ from the actual figures contained in the MRP database.

## 5.2 Estimation of tag recoveries in escapements

### Observed recoveries, sampling statistics, and sources of error

The identities of all tagged heads recovered at the Quinsam River, Puntledge River, and Big Qualicum River hatcheries were obtained from the MRP records (Tables 5.1, 5.2). The identities of tags recovered in other streams involved in this study were obtained directly from the Head Recovery Lab (Tables 5.3, 5.4). Information on 'select recoveries' obtained during field surveys conducted by DFO personnel at other locations within the study area were also obtained from the Head Recovery Lab (Table 5.5). Information on the number of non-tags in each category, sample sizes, and escapement estimates for both age groups were obtained for each stream where random samples were collected (Table 5.6). These figures show that a large portion of the tagged fish present in streams lacking public hatcheries could not be identified each year ( $ND > K+LP+NP$ ), in part due to the sampling methods used at these sites. In some cases the samples available were not sufficiently large to determine the contribution of each tag code to the total recoveries, and to categorize the 'non-tags' obtained. For such cases, it was

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necessary to estimate tag code contribution on the basis of ancillary information, and to assume that sampling error was negligible.

At Black Creek, all positively identified tags recovered during the 1985 to 1988 period had been applied at Black Creek (Table 5.3). No evidence indicated that the Black Creek escapement contained adults which reared at sea for two years or more (age 4<sub>1</sub> or 5<sub>2</sub>). Given the absence of fish lacking a ventral fin (of hatchery origin), and of tags applied at neighboring hatcheries, there was no evidence to indicate that coho from neighboring streams strayed to Black Creek during the period of study. Since no tags were applied in 1984 in this stream, the 'no pin' recovered in 1985 was assumed to consist of a fish which naturally lacked an adipose fin. No tagged jacks were sampled in 1986 for CWT identification due to their relatively low abundance. Given the absence of strays in the 1987 adult escapement samples, all tagged jacks in the 1986 escapement were assumed to be of Black Creek origin. In this case, tagged jacks were assumed to be distributed equally among the four tag codes used at release, since equal numbers of smolts were tagged with each code. All remaining 'no pin' recoveries in 1986 and 1987 were assumed to consist of tagged fish which rejected their tags, since the relative abundance of untagged smolts which naturally lacked an adipose was negligible relative to the tag rejection rates at release during the previous season (Section 3.2).

At Trent River, tag code identification revealed that a substantial number of tagged jacks and adults recovered each season were not of Trent River origin (Table 5.3). All strays recovered were of hatchery origin, and released from facilities located as far as 100 km from the Trent River. On an overall basis, the majority of strays were from the Puntledge River hatchery, which is the closest contributing hatchery. No evidence indicated that the Trent River escapement contained adults of age 4<sub>1</sub> or 5<sub>2</sub>. Since no tagging was conducted at Trent River in 1984, and since the number of untagged smolts released which naturally lacked an adipose fin was negligible (Section 3.2), all adult 'no pin' recoveries during 1985 were assumed to consist of hatchery fish which rejected their tags. All adult heads recovered in 1986 lacking tags were fin clipped (LV), and were assumed to consist of fin clipped hatchery fish released for 'Expo 86' which rejected their tags. These 'no pin' recoveries were mainly associated with tag codes 022946 and 022915, since these were the two major contributing codes in which all tagged fish were fin clipped (lv, Table 5.3). For this special case, the number of observed recoveries for these two codes was set to 23 and seven respectively, and the number of 'no pin' was set to 0. All the remaining adult 'no pin' recoveries for 1987-1988 were considered to be tagged fish which rejected their tags, and could not be associated with any particular tagged group. Both 'no pin' jacks recovered at Trent River in 1986 were assumed to consist of tagged fish of Trent River origin which rejected their

tags. During 1987, no tagged jacks were sampled for CWT recovery, so all tagged jacks were assumed to be of Trent River origin.

At French Creek, no juvenile tagging was conducted there in 1985, but all adults captured in 1985 and 1986 were examined for missing adipose fin, and none were detected (Table 5.3). All tags recovered from adults during 1987 and 1988 were of French Creek origin. No evidence indicated that age 4<sub>1</sub> or 5<sub>2</sub> adults escaped to French Creek. Thus, the contribution of hatchery strays to the adult escapement at French Creek was assumed to be negligible. This is supported by the fact that the relative abundance of fin clipped adults ('Expo' hatchery releases) in the 1986 escapement was < 0.8% (Table 4.9). The contribution of strays to the jack escapement at French Creek was also assumed to be negligible given the absence of recoveries of tags from other streams (Table 5.4). Thus, all 'no pin' recoveries were assumed to consist of tagged fish of French Creek origin which rejected their tags.

For Black Creek, Trent River and French Creek, the observed recoveries were corrected as described above. For the Quinsam River, Puntledge River and Big Qualicum River, sample sizes were always sufficiently large to properly categorize the recoveries obtained (Table 5.6). However, evidence reported by DFO investigators suggested that sampling errors commonly occur while large numbers of coho are being processed at the hatchery. Systematic surveys conducted during 1988 and 1989 at three public hatcheries in the Fraser Valley revealed that 2-16% of the tagged coho examined were misidentified as untagged fish, and comparatively fewer untagged fish were misidentified as being tagged (K. Wilson, DFO Memorandum, March 20, 1989). The surveys also suggested that the misidentification rate at each hatchery varied significantly between years. Accurate estimates of the magnitude of this error were not determined for the Quinsam River, Puntledge River and Big Qualicum River hatcheries for the 1985-1987 period, but additional investigations conducted by DFO personnel revealed that misreporting errors of such magnitude probably occurred at these facilities during the period (Carol Cross, pers. comm.). In the absence of actual figures on the misreporting rates, it was assumed that during the 1985-1988 period, 8% of the tagged fish were not detected during processing, and 1% of the untagged fish sampled were categorized as tagged fish. Both types of errors must be corrected for since they lead to biased estimates of total recoveries.

#### Estimation of tag loss and tag contributions

To properly associate all 'non-tag' recoveries to particular tag groups, and determine the number of adjusted recoveries in escapement samples, it is necessary to determine the contribution of strays to

the escapement of tagged fish, as well as the magnitude of tag loss associated with the tag codes recovered. The contribution of tagged strays among tagged fish of each age group in escapement samples was estimated from the number of observed recoveries identified as strays divided by the total observed recoveries. These estimates (Table 5.7) indicate that a major portion of the tags recovered in some years at Trent River, Big Qualicum River, and Puntledge River had been previously released in other streams. The majority of strays recovered at the Puntledge River hatchery consisted of fry which were reared at this hatchery and subsequently released in the Tsolum River for colonization purposes. Strays recovered at Big Qualicum River and Trent River were of various origins, but consisted mainly of fish released in neighboring streams.

When the contribution of tagged strays to the escapement of tagged fish is minimal, estimates of tag loss among tagged fish escaping to their stream of origin can be estimated from the recovery statistics. Tag loss for each age group was estimated from  $NP / (NP+LP+K)$  for all cases at Black Creek, Rosewall Creek, Little Qualicum River, French Creek and Millstone River where sufficient recoveries were available (Table 5.6). For these cases, tag loss among adults was usually larger than among jacks of the same cohort, but linear regressions of tag loss among jacks against that of adults from the same cohort revealed no significant relationship between these variables ( $r^2 = 0.0011$ ). In cases where tag loss could be estimated for both age groups of the same cohort, weighted estimates of tag loss were obtained by pooling sampling statistics for jacks and adults. Where weighted estimates could not be determined, tag loss among adults was used as the tag loss estimate because adult samples were generally larger. Estimates of tag loss for these populations ranged from 0% to 15%, averaged 8%, and were usually larger than the corresponding estimates of tag loss conducted 24 h after tagging (Table 5.8). The difference between tag loss at return and at release for smolts released in 1985, 1986 and 1987 averaged 5.7% , 6.8% and 9.9% respectively. These results agree with those reported by Blankenship (1990), who showed that tag loss continues to occur in coho for up to 29 d after tag implantation. In Blankenship's experiment, tagging was conducted under hatchery conditions, and maximum tag loss after 293 d of retention was 5.3%, which is slightly less than the losses obtained under field conditions reported for the present study.

Linear regression of the logarithms of tag loss at release against tag loss at return showed no significant relationship between these variables ( $r^2 = 0.12$ ), indicating that tag loss among adults could not be accurately predicted from the release statistics, even in the absence of significant numbers of strays. Therefore, if tag loss at return could not be estimated for a given cohort due to the lack of sufficient recoveries, the average difference in tag loss observed that year was added to the tag loss at release, and considered as the best estimate of tag loss among fish escaping to their stream of origin.

This approach was also used for estimating tag loss among adults and jacks escaping to the Trent River, which were released from this stream in 1985 and 1986.

For the Quinsam River, Puntledge River and Big Qualicum River, weighted estimates of tag loss based on MRP statistics, uncorrected for sampling errors, ranged from 4.9% to 36.9%, and averaged 16.8%. These estimates are generally larger than the corresponding estimates from other streams. Even in the absence of strays, the figures were not readily comparable with estimates of tag loss at release due to variation in retention rates among the codes used (Table 3.8 to 3.10). However, it should be noted that retention tests for production and colonization releases were usually conducted after a long retention period (> 4 weeks), which is theoretically sufficient to account for all tag loss (Blankenship 1988). Therefore, estimates of tag loss at release for these groups should be indicative of the expected tag loss at return. Since the majority of tagged groups released each year were held for long periods of time (> 4 weeks), crude estimates of tag loss at return among tagged fish from a given cohort escaping to their home stream were obtained from weighted estimates of tag loss at release for each group (i.e. hatchery - brood year combination):

$$(Eq. 5.2) \quad Tag\ loss_{ret, g} = \frac{\sum_C Tags\ applied_{c, g} \cdot tag\ loss_{rel, c, g}}{\sum_C Tags\ applied_{c, g}}$$

where:  $tag\ loss_{rel, c, g}$  = Tag loss at release for fish with code  $c$  in group  $g$ , (Tables 3.8 - 3.10);

$C$  = all tag codes that were used on a given release group, which were tested after a retention period  $\geq 28$  d, or the longest period if none were held that long.

Tag loss estimates obtained from Eq. 5.2 were much smaller than weighted estimates of tag loss based on escapement samples (Table 5.8). Even in cases where strays did not contribute substantially to the escapements of tagged fish, the differences observed far exceeded those obtained in Blankenship's (1988) study, as well those observed in other streams in the present study. Three factors could potentially account for the unusually large discrepancies observed: (1) misidentification of untagged fish, (2) the presence of large numbers of fish which naturally lacked an adipose fin, and (3) unusually high rates of tag loss for certain codes.

With regards to point (1), the effects of misidentifying 1% of the untagged fish examined at each hatchery were assessed through a numerical reconstruction of the sampling process. The extra tagged fish obtained as a result of misidentification can contribute to the NP and ND categories. As a result, NP =

$NP' + NP''$ , in which  $NP'$  represents the number of 'no pin' obtained when no misidentification occurs, and  $NP''$  represents the extra number of 'no pin' obtained as a result of misidentification. The same distinction can be made for ND, so that  $ND = ND' + ND''$ . When partial sampling is conducted, a fraction of all extra tagged fish obtained as a result of misidentification ( $ND'' + NP''$ ) contribute to the ND category. This fraction is equal to the proportion of all tagged fish that were not sampled for CWT recovery ( $p(ND) = ND/(ND+LP+NP+K)$ ). The remaining fraction ( $1-p(ND)$ ) of the extra tagged fish make up the  $NP''$ , which implies that  $ND'' = NP'' \cdot p(ND) / 1-p(ND)$ . Given these relations, the total number of extra tagged fish obtained as a result of the misidentification of 1% of the untagged fish is given by:

$$(Eq. 5.3) \quad NP'' + ND'' = 0.01 \cdot (S - (K + LP + NP' + ND'))$$

Since  $ND' = ND - ND''$ , Eq. 5.3 transforms to:

$$(Eq. 5.4) \quad \frac{NP''}{0.01} + 99 ND'' = S - K - LP - NP' - ND$$

Substituting  $ND''$  for its  $NP''$  equivalent results in:

$$(Eq. 5.5) \quad S - K - LP - ND - \frac{NP''}{0.01} - 99 NP'' \cdot \frac{p(ND)}{1 - p(ND)} = NP'$$

Since  $NP = NP' + NP''$ , Eq. 5.5 can be transformed to:

$$(Eq. 5.6) \quad NP = NP'' + S - K - LP - ND - \frac{NP''}{0.01} - 99 NP'' \cdot \frac{p(ND)}{1 - p(ND)}$$

The number of extra no pins obtained in each escapement as a result of the misidentification of untagged fish can be estimated by finding the values of  $NP''$  which solve Eq. 5.6. After deleting  $NP''$  and the corresponding  $ND''$  from the hatchery recovery statistics, estimates of tag loss that would have been obtained in the absence of misidentification were recalculated for each age group and cohort. The results (Table 5.8, bottom section) show that at Quinsam River and Puntledge River, differences between the estimates of tag loss at release and at return were reduced considerably after corrections were made. The maximum differences observed at these sites were well within the range of differences observed at other locations. In some cases, misidentifying 1% of the untagged fish translated into excessively large numbers for  $NP''$  ( $NP'' > NP$ ). This was the case for jacks in some years, as well for

both age groups at Big Qualicum River each year. Such results suggests that misreporting error may differ for each age group, and that a misreporting error of 1% may have slightly exceeded the actual misreporting error at Big Qualicum River, and at Quinsam River and Puntledge River in 1985.

With regards to point (2), Blankenship (1990) reported that fish with naturally missing adipose fins accounted for 0.5% of the adult coho escaping to four Puget Sound hatcheries, which was about ten times larger than the fraction observed in neighboring streams. Blankenship (1990) showed that estimates of tag loss at return are severely biased when the tagged proportions at release are low, and 0.5% of the untagged fish released lacked their adipose fin. Estimates of the relative abundance of fish with naturally missing adipose fins among the hatchery production groups are not routinely obtained. Thus, no attempt was made to estimate the potential effects of their presence. However, it should be noted that misidentifying untagged fish in escapement samples can also have the same effect upon tag loss estimates as the presence of fish with naturally missing adipose fins. Thus, a combination of low tagged proportions at release and a 0.5% misidentification rate of untagged fish should also result in severely biased estimates of tag loss.

With regards to point (3), no evidence was found to support the hypothesis that there is a high rejection rate associated with certain tagged groups. Colonization and production groups were often tagged at the same stage, by experienced tagging teams. Retention rates were usually similar among tagged groups irrespective of the retention period (Table 3.8 to 3.10). Tagged groups which were released after a short retention period usually contributed a small fraction of the total releases. As a result, the retention rates for these groups would have to be disproportionately large (> 30%) to compensate for the high retention rates among other groups, which was considered unlikely.

Examination of the estimates of tag loss at return based on hatchery escapement samples revealed that tag losses among jacks and adults of the same cohort were similar. On average, tag loss among adults was about 1.33% higher than among jacks, and the figures for both age groups were found to be correlated ( $r^2 = 0.64$ ). Relatively high rates of tag loss were observed among the 1987 jack and 1988 adult returns to the Quinsam River (36% and 38%, Table 5.6), as well as among the 1986 jack and 1987 adult returns at Puntledge River. Although there is no reason to suspect that untagged jacks and adults of the same cohort would be particularly susceptible to being misidentified at a high rate, it should be emphasized that the overall tagged proportions at release for both groups were less than 5% (Table 5.8). As noted above, under such conditions, positively biased tag loss estimates would be expected for both age groups if untagged fish are misidentified even at a low rate. Thus, similarity in tag

loss estimates among siblings is viewed as further evidence in support of the hypothesis that untagged fish were misidentified to some extent during processing.

Since the contribution of three potential factors responsible for the discrepancies between tag loss estimates at release and return could not be quantified with certainty, estimates of tag loss at return were based exclusively on the tag retention rates at release. Tagged adults and jacks which were held at least 28 d for tag retention tests were assumed to have a tag loss equal to the estimated tag loss at release. Weighted estimates of tag loss obtained from Eq. 5.2 were considered as representative of tag loss at return for all remaining tagged groups of the same cohort which were checked after shorter retention periods. The number of 'no pin' and 'no data' resulting for the hypothesized sources of error were accounted for before estimating the number of adjusted recoveries. For each age group, estimates of the actual number of 'no pin' in the escapement samples at Quinsam River, Puntledge River and Big Qualicum River in the absence of misidentification of untagged fish were calculated as:

$$(Eq. 5.7) \quad NP' = \sum_C T_{obs,c} \cdot \frac{tag\ loss_c}{1 - tag\ loss_c}$$

where:  $T_{obs,c}$  = Observed recoveries of code c (subset of C);

Tag loss<sub>c</sub> = Estimate of tag loss at return for code c.

The number of 'no pin' and 'no data' obtained as a result of the hypothesized sources of error are then estimated from  $NP'' = NP - NP'$ , and  $ND'' = NP'' \cdot p(ND)/(1-p(ND))$ . Deducting  $NP''$  and  $ND''$  from the associated recovery statistics results in the elimination of all recoveries which cannot be accounted for by the estimated tag loss. The corrected statistics (Table 5.9) showed slight reductions in the 'no data' recoveries at Quinsam River and Big Qualicum River, as well as considerable reductions in the number of 'no pin' recoveries at Quinsam River.

#### Estimates of total escapement recoveries adjusted for unidentified tags and sampling rates

The standard method proposed by Kuhn *et al.* (1988) for estimating the number of adjusted recoveries for each code ( $T_{adj,c}$ ) consists of allocating 'non-tag' recoveries to various tag codes in proportion to the contribution of each code to the total number of identified tags in a sample:

$$(Eq. 5.8) \quad T_{adj,c} = T_{obs,c} \cdot \left( 1 + \frac{LP}{K} + \frac{ND \cdot (K + LP)}{K \cdot (K + LP + NP)} \right)$$

In the above model, the number of 'no pin' recoveries is only used to allocate the number of 'no data' in the sample. The 'no pin' recoveries are not distributed among the contributing tag codes, and do not contribute to the adjusted recoveries. This approach is preferable when tag loss at return is not known, and can differ substantially between contributing tag codes. The above model is most applicable to situations where the number of 'no pin' recoveries is relatively small, and where tag identities represent the only source of information available for the allocation 'non-tag' recoveries.

Such conditions are not representative of the present situation. The NP category often made up a major portion of the total number of heads recovered, particularly in the small samples obtained during stream surveys (Table 5.6). For streams lacking public hatcheries, the 'no pin' recoveries could be associated with particular tagged groups because of the presence of distinct external marks in most cases. For these streams, differences in tag loss at return among contributing codes were usually considered to be negligible due to the tagging procedures used and the relatively small contribution of tagged strays. For streams with public hatcheries, estimates of tag loss at return were available for most of the contributing codes, allowing 'no pin' recoveries to be allocated in relation to tag loss and tag contributions. Therefore, the number of adjusted recoveries in escapement samples were estimated using different procedures depending on the stream type (with or without major public hatchery).

Estimates of the number of adjusted recoveries in escapement samples at Quinsam River, Puntledge River and Big Qualicum River were generated for each each age class as follows:

$$(Eq. 5.9) \quad T_{corr,c} = T_{obs,c} \cdot \left( 1 + \frac{LP}{K} \right) \cdot \left( 1 + \frac{tag\ loss}{1 - tag\ loss} \right)$$

$$(Eq. 5.10) \quad K_{corr} = \sum_C T_{corr,c}$$

$$(Eq. 5.11) \quad T_{adj,c} = T_{corr,c} \cdot \left( 1 + \frac{ND}{K_{corr}} \right)$$

Estimates of the number of total recoveries were then obtained by accounting for the hypothesized non-detection of tagged fish (8%) during sorting operations at the hatchery, as well as the sampling rate used:

$$(Eq. 5.12) \quad T_{tot,c} = \frac{T_{adj,c}}{.92} \cdot \frac{escapement}{sample}$$

For the remaining streams, as noted previously, there was no justification for estimating the number of adjusted recoveries in the remaining systems based on potential tag loss among contributing codes. Therefore, all 'non-tag' recoveries still unaccounted for after the corrections proposed at the beginning of Section 5.2 were allocated in proportion to the contribution of each code:

$$(Eq. 5.13) \quad T_{adj,c} = T_{obs,c} \cdot \left(1 + \frac{NP + LP + ND}{K}\right)$$

$$(Eq. 5.14) \quad T_{tot,c} = \frac{T_{adj,c} \cdot Esc}{Sample}$$

Estimates obtained by Eq. 5.14 represent the total number of tagged fish from each code that escaped to each stream sampled during this study. Information on tags recovered in streams that were surveyed occasionally but not systematically sampled ('select' recoveries) was also accounted for to estimate the total escapements of tagged fish. The number of tagged fish of each stock/age/return year combination (or group  $g$ ) that escaped to all streams surveyed was estimated by pooling the total recoveries across all codes associated with each group:

$$(Eq. 5.15) \quad Tag\ esc_g = \sum_{st} \sum_c T_{tot,st,c}$$

where:  $Tag\ esc_g$  = total escapement of tagged fish of a given group ( $g$ );

$T_{tot, st, c}$  = total recoveries of fish of a given code ( $c$ ) in a stream ( $st$ ).

The results indicate that for most groups a high proportion of the tagged fish escaped to their home stream (Tables 5.10 and 5.11). Among adults, straying (defined as strays/total escapement) was lowest for production releases (Table 5.12). On average, adults from colonization releases strayed slightly more than those of production releases, but no significant differences in straying were detected between these types for 1985, 1987 and 1988. Adults from wild populations strayed slightly more than the two former types, and adults from enhanced populations strayed more than those of wild populations. Adults from experimental releases conducted at the Rosewall Creek hatchery exhibited unusually high levels of straying during 1987, but straying in 1988 was only slightly greater than

average. Chi-square tests of the stray-to-non stray ratios revealed significant differences among stock types each year, but the differences were not consistent across all years (Table 5.12). On average, adults exhibited more straying in 1987 than in other years. On an overall basis, adult straying averaged 1.5%, but if the experimental releases from Rosewall Creek were omitted, the average was 1.0%. Adult strays escaped to streams located two to 159 km from their home stream, but the average straying distance was 15.7 km, and over 50% of the strays escaped to streams located less than seven km from their home stream.

Straying rates among jacks ranged from 0 to 50%. As for adults, production and colonization releases from large hatchery systems exhibited very little straying, while those from experimental releases at Rosewall, and colonization releases at the Millstone River, strayed the most. On average, straying appeared to be greatest during 1988 and 1986. No relationship was found between the level of straying among jacks in a given year and their adult siblings during the following year. On an overall basis, jack straying averaged 0.43%, and if the experimental releases from Rosewall Creek were omitted, the average was 0.38%, which is substantially less than was observed for adults. Jack strays escaped to streams located two to 91 km from their home stream, but the average straying distance was 26.3 km, and over 50% of the strays escaped to streams located less than seven km from their home stream.

Some of the Trent River strays were recovered in the Puntledge River, which serves as the water source for the colonization fry reared at the Courtenay Fish & Game hatchery. In addition, some adults of Trent River brood stock released from the Rosewall Creek facility were recovered at the Trent River. However, in the majority of cases where adults and jacks strayed from their location of release, they were recovered in streams which were neither their ancestral stream or the stream where they reared for a portion of their pre-smolt life. This suggests that some factor(s) may affect the ability of fish to recognize the stream where they reared. Insight into this matter may be gained from further examination of the rearing conditions of stocks exhibiting the greatest straying. Rosewall Creek releases consisted of the progeny from distant populations, which were hatchery reared, and released from a stream which was not their parental one. This hatchery uses mainly surface water from Rosewall Creek for rearing purposes, but relies heavily on groundwater during the summer months when surface flows are insufficient. Fry released in the Millstone River and Trent River for enhancement purposes were also reared at hatcheries located at some distance from their stream of origin, using non-natal water sources. Thus, it is hypothesized that the process of exposing fry to groundwater or foreign (non-natal) water sources for a prolonged period during the rearing stage increases the amount of straying exhibited by the returning fish.

### Stock composition of escapements

To calculate stock specific exploitation rates, the contribution of strays in the escapements of each stream must be determined. To do so, requires that estimates of the total tag recoveries of each code of foreign origin be expanded to account for the untagged smolts associated with each tagged group at release. For each tag code ( $c$ ) released at Quinsam River, Puntledge River and Big Qualicum River, the tagged proportions at release ( $p(\textit{tagged})_{rel}$ ) were estimated from the MRP database records as follows:

$$(Eq. 5.16) \quad p(\textit{tagged})_{rel,c} = \frac{\textit{Tag Rel}_c}{\textit{Total Rel}_c}$$

where:  $\textit{Tags Rel}_c$  = Total number of smolts released that were tagged with code  $c$ ,  
adjusted for mortalities and tag loss prior to release;  
 $\textit{Total Rel}_c$  = Total number of smolts released represented by code  $c$ , adjusted  
for mortalities prior to release.

The number of tagged fish associated with each code that strayed to a stream ( $\textit{Stray}_c$ ), and the contribution of strays associated with all foreign tag codes ( $C = \sum c$ ) to each escapement (expressed as the proportion:  $p(\textit{stray})_{esc}$ ) were estimated as:

$$(Eq. 5.17) \quad \textit{Stray}_c = \frac{T_{tot,c}}{p(\textit{tagged})_{rel,c}}$$

$$(Eq. 5.18) \quad p(\textit{stray})_{esc} = \frac{\sum C \textit{Stray}_c}{Esc}$$

This estimation procedure does not account for the contribution of strays from unassociated<sup>26</sup> hatchery releases, given the large uncertainty involved in the association process. However, such groups represented a relatively minor portion of the total production from large public hatcheries during the 1984-1988 period.

For all tag codes released from Black Creek, French Creek and the Millstone River, given the absence of strays in the escapements, it was assumed that the tagged proportions at return were

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<sup>26</sup>A group of fry released from a hatchery, or transplanted to a neighboring stream, which does not contain tagged individuals, and which cannot be readily associated with another tagged group.

indicative of the tagged proportions at release. Theoretically, both proportions would be equal if: (1) the outmigrating fry displaced by residents do not survive until maturity and escape to the stream, and (2) tagged and untagged smolts are subject to the same natural mortality rates, maturation rates and migratory behavior. No evidence was obtained during this study to reject assumption (1). With regards to (2), Blankenship and Hanratty (1990) observed from experimental studies that coho smolt-to-adult survival of a test group subject to trapping and coded-wire tagging during their smolt migration was 84% of that of the control group released below the fence. The authors suggested that an expansion factor of 1.19 be applied to the number of recoveries of tagged fish for proper estimation of survival rates. Their results also indicate that this expansion factor should be applied to the number of tag recoveries in the escapement to obtain an estimate of the actual tagged proportion at release. Therefore, tagged proportions at release were obtained from the weighted average of the tagged proportions among adults and jacks in the corresponding escapements, adjusted for the hypothesized differential mortality:

$$(Eq. 5.19) \quad p(\text{tagged})_{rel,c} = \frac{(Tag_{esc\ adult,c} + Tag_{esc\ jack,c}) \cdot 1.19}{Esc_{adult} + Esc_{jack}}$$

where:  $Tag_{esc\ adult,c}$  = Estimates of total adult recoveries with code c in the escapement;

$Esc_{adult}$  = Estimate of adult escapement.

The relative stray contributions from these three streams to other streams was then estimated according to Eq. 5.18. For the Trent River, the total number of strays from other streams was first deducted from the escapement to estimate the escapement of adults and jacks of Trent River origin. Estimates of the tagged proportions at release for Trent River fish were then calculated according to Eq. 5.19. The same procedure was used to estimate the tagged proportions at release for the Little Qualicum groups. Estimates of the contribution of strays from each of these populations to other streams were then calculated according to Eq. 5.18.

The results indicate that in the majority of cases, strays contributed less than 1% of the escapements of adults and jacks (Table 5.13). Stray contribution, averaged across all streams, amounted to approximately 3% of the jack escapement, and 4.7% of the adult escapements. Such high average contribution estimates were largely induced by the relatively large contribution of strays to the Trent River, which in some cases accounted for more than half of the fish escaping to this river. Adults and jacks tended to stray more to the Puntledge River than to other large systems such as the Big Qualicum River and Quinsam River. Strays in the Puntledge River consisted mainly of fish reared at the Puntledge River hatchery and released in the headwaters of the Tsolum River for colonization purposes. This

suggests that the relatively large contribution of strays to this system is mainly due to enhancement practices rather than peculiar attributes of the stream itself.

Information on the relative abundance of fin clipped adults observed in some escapements during 1986 ('Expo' fish) was used to assess the accuracy of stray contribution estimates based on tag recovery data. The proportion of fin clipped adults in the 1986 escapement at Trent River was previously estimated to be about 40.1% (Table 4.9). Given that about 82% of the smolts released from the Puntledge River hatchery for Expo were fin clipped, the proportion of fin clipped adults in the Trent River was expanded by 1.22 to account for the unclipped smolts released. This resulted in an estimated contribution of Puntledge River hatchery adults to the 1986 Trent River adult escapement of about 49.0%, which is almost identical to the estimate based on the expansion of tag recoveries (50.2%). At French Creek, fin clipped adults accounted for less than 0.8% of the escapement, which is comparable to the figure obtained by expansion of tag recoveries (0%). Such results indicate the contribution of strays can be reliably estimated by expansion of tag recovery data.

### 5.3 Effects of various factors on straying rates

The influence of various factors hypothesized to have an effect upon the level of homing (and straying) was assessed by means of log-linear models. Green and Macdonald (1987) suggested that for analysis of coded wire tag recovery patterns, this class of models is more suitable than conventional analysis of variance because the effects of various factors are often combined multiplicatively, and because the interactions of categorical, interval and continuous variables can be estimated. Fienberg (1980) argued that the effects of particular factors upon certain variables are best quantified by converting log linear models into logit or linear logistic models, which are the categorical response analogs to regression models for continuous response variates. Therefore, a linear logistic model was used to describe the structural relationship between the tag recovery patterns and various factors monitored during this study. For modelling purposes, the recovery locations in Tables 5.10 and 5.11 were collapsed into two strata (recoveries in the stream of origin vs. all streams), and the cell frequencies were combined accordingly for each group. The relative cell frequencies (homing proportions) were considered as the response variables of interest. The population attributes and environmental factors tested for their effects upon homing were:

- x<sub>1</sub>. The calendar year of escapement (1 - 4, representing 1985 to 1988);
- x<sub>2</sub>. The genetic make-up of the group released. Eight genetic categories were used for classification:  
Quinsam (1), Black Creek (2), Puntledge (3), Trent (4), B. Qualicum (5), L. Qualicum (6), French

Cr. (7), Millstone (8). Wild, colonization and production groups released in the same stream were considered as members of the same category. Rosewall Creek releases from different brood stocks were categorized according to the parent stock used (2, 4, and 6);

- x3. The experimental treatment (i.e. level of hatchery conditioning). The number of months each stock was reared in a hatchery environment was used as a measure of conditioning;
- x4. Smolt size at release (smolt weight selected for consistency with other studies);
- x5. The median date of smolt migration (calendar day);
- x6. The release location, expressed as a distance (in km) to the north of Nanaimo;
- x7. The starting date of the upstream spawner migration (calendar date from Section 4.2);
- x8. The median date of the upstream spawner migration (calendar date from Section 4.2);
- x9. The age of the spawners, in terms of ocean residency period (jacks=1, adults =2);
- x10. Exposure to different water sources during the rearing stage (Y/N);
- x11. The total number of escapement recoveries for a given age class (Sum of tag recoveries in Tables 5.10 and 5.11);
- x12. The existence of flow control in the stream of release. The two categories used were (1) some control resulting in sustained flows during the fall, and (2) no flow control.

In the following analysis, the objective is to predict the probability of straying for different types of fish, characterized by particular combinations of population attributes and environmental factors. The logistic model describing this relationship is:

$$(Eq. 5.20) \quad E \left( \frac{s}{e} \right) = \frac{\exp^{(b_0 + b_1 x_1 + b_2 x_2 + \dots)}}{1 + \exp^{(b_0 + b_1 x_1 + b_2 x_2 + \dots)}}$$

- where:  $s$  = estimated number of strays for a particular group;  
 $e$  = estimate of the total escapement for a particular group;  
 $b_n$  = parameters to be estimated;  
 $x_n$  = population attribute or environmental factor ( $x_1$ =year,  $x_2$ =genetic, etc.).

The stepwise logistic regression program (LR) of the BMDP statistical package (Dixon 1988) was used to fit the logistic model to the recovery data. Initially, the program generates design variables for the categorical variables and their interactions. The independent predictor variables, consisting of a continuous variable or one set of design variables, are then entered into the model in a stepwise manner. At each step, the set of coefficients for the included terms are estimated iteratively as the values which

maximize the likelihood function. A decision is then made whether to enter or remove any term on the basis of its log-likelihood ratio test statistic ( $G$  values or chi-square equivalent). The tail area probabilities associated with these values are computed, and the term with the largest probability is removed if it exceeds certain rejection limits ( $\geq 0.05$ ). If no term has a probability value larger than this limit, the term with the smallest probability value is entered if it is less than the probability level required for entry ( $< 0.05$ ). The whole process is repeated until no variable satisfies the conditions for entry or removal.

The LR program allows higher order interactions to be incorporated into the model if lower order interactions and main effects are already in the model. However, this option can be omitted when it is desired to obtain a more parsimonious model, or when one wishes to increase the number of distinct factors tested simultaneously (whose upper limit is set by the number of cases). Thus, given the relatively low number of distinct observations (i.e. homing proportions) in relation to the potentially large number of factors that could be incorporated into the model, the presence of lower order interactions was not required for incorporation of higher order interactions.

For the same reason, limits were also imposed upon the type of interactions tested simultaneously. Only the main factors and their first order interactions were allowed into the model since higher order interactions are often difficult to interpret. Most (but not all) first order interactions were tested for simultaneously. The number and type of first order interactions tested was limited to the combinations of factors which were hypothesized to have an influence on straying through some plausible mechanism. Interactions including genetic factors were limited because such interactions involve a large number of coefficients. For instance, incorporating an interaction such as time\*genetic involves the inclusion of seven dummy coefficients into the model, which greatly limits the number of additional factors that can be included into the model (parameters  $<$  observations). To deal with this constraint, and assess all first order interactions involving genetic factors, only one such interaction was evaluated during the fitting stage. This interaction was then substituted with another interaction, and the process was repeated until the influence of all interactions involving genetic factors upon homing were evaluated. The interaction which had the largest influence upon homing was considered as the only such interaction that could be incorporated into the model.

As suggested by Green and Macdonald (1987), the emphasis during the model fitting stage is not so much to find the best fitting model, but rather to determine which sources contribute most to an improvement in fit. This is accomplished by examining the reduction in "deviance" obtained by the inclusion of additional factors in the model. This deviance, a general term describing the discrepancy in

fit, is equivalent to the likelihood ratio G statistic, and is analogous to the residual sum of squares obtained from linear regressions. Green and Macdonald (1987) suggest that additional parameters should be included until the discrepancy between the observed and expected values is small enough that the fitted model can be used for predictive purposes. However, the authors acknowledge that given such an objective, selection of an appropriate model is to a large extent a function of the management goals rather than statistical merits. Since the purpose of the present analysis was simply to identify the major determinants of homing, a model was considered adequate if it documented which factors are systematically associated with homing. Therefore, the inclusion of additional factors in the model was considered acceptable, until the point was reached where the reduction in deviance associated with the additional factors was  $\leq 5\%$  of the largest reduction in deviance obtained during previous inclusions. This procedure ensured that factors which had a relatively small contribution to the improvement in fit were not incorporated in the model, even if their contribution was determined to be statistically significant. By omitting factors with marginal contributions, the resulting model was more parsimonious, yet could still be used for 'predictive' purposes since it accounted for the effects of the main factors.

Before attempting to determine which factors influence homing, a correlation matrix of the variables was constructed to determine if any of these were correlated. Substantial correlation ( $r^2 > 0.50$ ) was found between smolt weight and treatment ( $x_3, x_4$ ), as well as between the start of the upmigration period ( $x_7$ ) and the median date of upmigration ( $x_8$ ) and flow control ( $x_{12}$ ). Therefore, smolt weight and run starting date were omitted from the set of variables tested.

The results obtained from this fitting procedure indicated that not all variables listed above contributed to a substantial improvement in fit (Table 5.14). The treatment factor ( $x_3$ ) was not required in any of the models, indicating that hatchery conditioning had no influence on homing rates. All remaining variables were useful in predicting homing proportions, with some of them being incorporated into the model only as interaction factors. Based on the selection criteria described above, model 9 was judged to be an appropriate model for describing homing proportions. More complex models incorporated mainly interactions of factors already included in model 9, and failed to provide a substantial improvement in fit as shown by the trend in the deviance reduction. This indicates that information on return year, exposure to water sources, genetic make-up, run timing and flow control was sufficient for predicting homing rates, and that the importance of information on aspects of the juvenile migration (smolt migration date, smolt size, release location) was relatively minor.

The relative importance of each factor in model 9 was determined by performing goodness-of-fit tests using different versions of the model characterized by the presence or absence of each factor (Table 5.15). The magnitude of the difference in chi-square obtained in the presence and absence of a particular factor does not reflect the magnitude of its effect upon the response variable, but is indicative of the relative importance of this factor on the performance of the model. The inclusion of the genetic\*exposure interaction was found to have been a major determinant of homing proportions, and its presence indicated that exposure to foreign water sources can influence the homing ability of various stocks to a different extent. Run timing and flow controls were also important predictors of homing, but their influence upon homing varied from year to year. Run timing was also found to influence the homing rates of various age classes and stocks to a different extent.

A sensitivity analysis was performed on model 9 to assess the direction and magnitude of the changes in homing resulting from variation in the levels of each factor. Estimates of the regression coefficients generated by the BMDP program were incorporated into the logistic model, which was then used to predict homing proportions associated with particular combinations of parameter values arbitrarily selected within a realistic range. As expected, predicted homing rates were found to be negatively affected by the exposure to non-natal water sources during rearing stages. Predicted homing rates also improved in the presence of flow control, but the magnitude of the improvement differed from stock to stock. Predicted homing rates were affected by the age at return, with a reduction in the age at return resulting in an increase in the predicted homing. This supports the previous observation that jacks tended to stray less than adults.

A one week delay in run timing caused a 6% reduction in the predicted homing rate of adults returning to the Trent River in 1986, but only a 0.5% reduction in the predicted homing rates of Trent River jacks returning that year. This example demonstrates the influence of the age \* run time interaction factor, and suggests that jacks are less likely to stray due to delays in fall freshets. The magnitude of the reduction in the predicted homing rates of Trent River adults was found to differ slightly in other years for comparable delays in run timing as suggested by the year \* run time interaction effect detected. In addition, comparable delays in run timing were found to influence the predicted homing rates of other stocks to a different extent as indicated by the genetic \* run time interaction effect. Of particular interest is the finding that delays in run timing led to slight increase the predicted homing rates of stocks returning to streams characterized by flow controls. This type of effect would have been detected more readily if higher order interactions had been allowed for in the model (genetic\*run time\*flow control). This pattern differs from the trends predicted for other types of systems, and may simply reflect the fact that early runs are more likely to stray due to relatively low flows.

#### 5.4 Co-variation in CWT release-to-escapement ratio

In theory, if natural mortality, exploitation rates and jack-to-adult ratios were similar for all stocks in a given year, then the total escapements of tagged fish from each group, expressed as fractions of CWT releases, would be similar or equal across all stocks each year. If these conditions were met, then one member of the assemblage could be used as an indicator of the average trend in relative escapement levels for the entire assemblage. To assess the merits of this approach, escapement-to-release ratios were determined for all stock/year combinations (i.e. groups) based on CWT recovery rates. Estimates of relative escapement levels for each group ( $g$ ) were expressed as the fraction of the total number of tagged fish released of a particular group that escaped to the various streams as adults:

$$(Eq. 5.21) \quad \text{Relative escapement}_g = \frac{\text{Tag}_{esc\ adult, g}}{\text{Tag}_{rel\ g}}$$

On average, adult escapements accounted for approximately 2.1% of the smolts released in 1985, 1.1% of the 1986 releases, and 1.7% of the 1987 releases (Table 5.16). Relative escapement levels, averaged across all years, were highest for the Quinsam colonization fish (4.7%), and lowest for the Big Qualicum production fish (0.2%). However, no stock or particular type of fish (colonization, wild, etc.) exhibited consistently higher or lower relative escapement levels across all years. Statistical comparisons of the escapement-to-release ratios by means of log-likelihood ratio tests indicated significant differences among stocks each year ( $P < 0.0001$ ). Only a few stocks exhibited escapement-to-release ratios which were not statistically different from one another. For the 1987 adult return year, these consisted of the Puntledge production and two Rosewall Creek stocks (LQ and BC). For the 1988 adult return year, these included both Puntledge stocks and the same two Rosewall Creek stocks. These results indicate that there are substantial differences among stocks within a geographical region in terms of their capacity to contribute to the adult spawning populations. Stocks characterized by high survival and low exploitation have high escapement-to-release ratios, and contribute proportionally more to the spawning populations than stocks characterized by low survival, high exploitation, which leads to low escapement-to-release ratios.

An assessment of the co-variation in escapement levels was made based on the year to year trends observed during this study. Changes in relative escapement levels were calculated for each stock from the difference in relative escapements levels observed between two consecutive years (1986-1987, and 1987-1988). Each stock monitored for the three year period was then characterized by two point estimates, which were used as co-ordinates in a two dimensional plot (Fig. 5.1). Using this procedure,

stocks which exhibited comparable year to year changes in relative escapements for both periods would cluster together.

The results suggest the existence of two clusters; the first cluster included production fish from Big Qualicum River and Quinsam River, and colonization fish from Puntledge River. The second cluster included enhanced and wild fish from the Trent River and Little Qualicum River. The remaining stocks were scattered and failed to show much similarity. Unfortunately, many stocks were not monitored for the three year period, and could not be incorporated directly into the plot. Their location along the single axis associated with the monitoring period serves to indicate which stocks they might have clustered with if they had been monitored for the entire period. The possibility exists that wild fish from the Quinsam River, and all three stocks from Rosewall Creek would have been included in the first cluster, which would have resulted in a cluster containing at most seven out of fourteen stocks. However, the results obtained indicate that at least half of the stocks monitored exhibited no co-variation in relative escapement levels. In addition, stocks from the same or adjacent streams do not necessarily exhibit the same year to year trends in escapement levels. The same conclusion applies to stocks subject to the same treatments since wild, enhanced and colonization stocks did not always cluster together. Thus, no single stock would be representative of the year to year variation in escapement levels among the majority of stocks within the assemblage, or among stocks of the same type. Given this situation, there appears to be little incentive in identifying one stock as an indicator of the year to year variation in escapement levels among stocks in the assemblage.

However, even in the absence of substantial co-variation among stocks in this regard, one stock might be selected as an indicator of the average escapement levels (relative levels averaged across all stocks each year). Estimates of the level of similarity between a given stock and the mean relative escapement level were calculated as follows:

$$(Eq. 5.22) \quad MSD_s = \frac{\sum_y (RE_{s,y} - RE_{a,y})^2}{Y}$$

where: *MSD* = Mean squared deviation from the relative escapement of the assemblage;  
*RE* = Relative escapement of a stock/year (s,y) or assemblage/year (a,y) combination;  
*Y* = Total number of years where the stock was monitored ( $\leq 3$ ).

The estimates obtained from Eq. 5.22 indicate that stocks from Rosewall Creek or the Millstone River conformed most closely to the annual levels (Fig. 5.2). However, selection of an indicator stock should ideally be based upon stocks which were monitored for the entire three year period. Among those which meet this condition, colonization and production stocks from the Puntledge River appear to be the best choices as indicators of relative escapement levels for the assemblage.

## 5.5 Discussion of straying rates and escapement patterns

Little quantitative information has been published on the straying patterns of coho salmon. A similar situation exists with respect to other species of Pacific salmon, which led Quinn and Fresh (1984) to state that "reliable estimates of the proportion of salmon that home are virtually non-existent". Nevertheless, the results obtained during this study are in agreement with the circumstantial and factual information obtained from other field surveys, and inferences from genetic studies.

Taft and Shapovalov (1938) reported that 14.9% of the adult coho returning to Waddell Creek in California strayed to Scott Creek, a coastal stream located eight km north of it. Hatchery reared coho released at Scott Creek also strayed to Waddell, but the magnitude of straying in this group could not be quantified. Donaldson and Allen (1958) observed that about 1% of the adult coho returning to Lake Washington strayed to another tributary further upstream. Jacobs (1988) reported the results of an extensive survey of escapements in Oregon coastal streams aimed at determining the magnitude of straying by hatchery coho that were subject to accelerated growth conditions and released as pre-yearlings after a brief exposure to salt-water. Based on an analysis of the recovery pattern of coded-wire tagged fish, Jacobs estimated that these fish exhibited a considerable amount of straying as adults (range: 1-30% of escapement), and strayed predominantly to a variety of basins within a 40 km radius from their point of release. Hatchery reared coho transported offshore prior to release were found to be twice as likely to stray as those released from onshore facilities. Jacobs also reported that strays from these hatcheries tended to spawn earlier than wild coho and coho from public hatcheries (reared full term and released on site).

Using information on patterns of genetic variation among coho populations from southern Vancouver Island and the lower coastal mainland, Wehrhahn and Powell (1987) inferred that straying rates from each breeding population were about six successful spawning individuals per generation. For populations with about 1000 breeding adults, this translates into straying rates of approximately 0.5%. Straying rates among the five natural populations (wild and enhanced) monitored during this study

which had at least 1000 breeding adults ranged from 0% to 11.1%, and averaged 3.7%. It should be noted that this average rate was to a large extent influenced by the unusually high straying rates of Trent River adults. Assuming that this stock is characterized by an abnormally high rate of straying, and is not considered as representative of natural populations, then the average rate becomes 1.3%. This figure should be considered as a maximum rate of straying, since it is not known if all strays recovered at other locations would have remained in the stream and spawned. Still, this figure is substantially higher than the estimate of Wehrhahn and Powell (1987). This can be explained by the fact that the straying rates estimated during the present study apply to local populations, and are expected to be higher than those based across all stocks covering a wide geographical area.

Wehrhahn and Powell (1987) estimated that the straying rate between the lower coastal mainland and southern Vancouver Island populations averaged 50 fish per year. During the present study, only two strays were detected from the mainland, but it is not known if these individuals would have spawned under natural conditions. Because the number of streams monitored during this study represents only a small fraction of all the streams in southern Vancouver Island, the straying rates hypothesized by Wehrhahn and Powell (1987) cannot be rejected based on the straying patterns observed during this study.

With regard to other species of Pacific salmon, Quinn and Fresh (1984) estimated straying rates among chinook salmon released from the Cowlitz River hatchery in the Lower Columbia, based on CWT recovery patterns. They found that the adults did not stray substantially ( $< 1.5\%$ ), and strays were generally recovered in streams within 30 km from the Cowlitz River. They reported a four-fold difference in the amount of straying between successive year classes, indicating that there can be pronounced year to year changes in the level of straying, as was also observed during this study. They found that older age classes tended to stray more than younger ones, which also agrees with the results of the present analysis. The authors also noted that high homing proportions were associated with large returns, and speculated that social factors strengthen the motivation to home. During the present study, no relationship was detected between homing rates and the number of recoveries to a given stream, but perhaps this is a consequence of the relatively narrow range of recoveries observed during the study.

Bams (1976) conducted a series of experimental releases of pink salmon fry in the Tsolum River (near Courtenay, B.C.), which consisted of the progeny of fish from the Tsolum River, and the progeny of a hybrid stock (Tsolum R. x Kakweiken R. cross). Differences in treatment effects between the various groups released were minimal. Based on the return rates observed, Bams concluded that adults from the transplanted hybrid stock homed less than those of the first group. It should be noted that Bams

compared homing rates based on the relative number of recoveries in the stream of origin, and not on the basis of the relative fractions of the total escapement recovered in other streams (as in the present study). In fact, an examination of Bams results shows that the only marked adults recovered in adjacent streams were of Tsolum brood stock. Thus, if the data had been analyzed in accordance with the procedure used in this study, the author would have reached the opposite conclusion. Nevertheless, the results of Bams study still support the notion that homing is affected by the genetic make-up of the fish released. An alternative interpretation is that homing accuracy is affected by human manipulation. The present analysis indicates that both factors, as well as run timing can influence homing rates, and efforts should be made to account for their separate effects in future experimental studies.

The results obtained during this investigation revealed differences in straying rates between populations. The lack of significant numbers of hatchery strays in the escapements of French Creek and Black Creek indicates that the level of inter-breeding with hatchery fish from neighboring public hatcheries is not large in all natural systems located in proximity to these facilities. However, the extent of the contribution of hatchery strays to the Trent River escapement, coupled with the apparent hybridization observed, indicates that some populations are strongly affected, and that there can be considerable year-to-year variability in the stock composition of that spawning population.

It is puzzling that Trent River differs markedly from other streams in terms of its 'attractiveness' to strays. Presumably, the proximity of the Trent River to the Puntledge River is partly responsible for the large number of Puntledge River strays found in the Trent River. However, the geographical location of the Trent River does not explain the relatively large number of strays from distant locations, such as across the Strait of Georgia (Porpoise Bay) and the Lower Fraser (Capilano River). By comparison to other systems lacking public hatcheries, the Trent River has a larger and more continuous discharge, and a larger estuary. Such facts suggest that strays may be attracted more readily to larger rivers than to small streams. This hypothesis is further supported by the fact that a relatively large number of hatchery strays were also found during 1986 in the Tsable River (30% of the fish examined, R. Hurst, pers. comm.), which is physically similar to the Trent River and located just 10 km south of it. Large numbers of strays were also detected in larger streams such as the Big Qualicum River and the Puntledge River. Although the number of strays detected at the Quinsam River was negligible, it should be noted that the detection of strays is partly a function of the tagging effort used in neighboring streams. Since this river is located at the upper edge of the study area, and adjacent natural stocks are not routinely tagged, the absence of many strays in this system is not surprising.

The large emigration of adults from the Trent River in 1986, coupled with their return to the Puntledge River hatchery, suggests that many of the fish entering the Trent River are simply exhibiting 'proving' behavior (Ricker 1972) and may not be committed to spawning there. Whether or not the number of fish proving a stream is related to a particular attribute of the stream or the size of the stray populations is not known. No strays were ever detected at French Creek, Black Creek, and Millstone River, although they are located in the proximity of public hatcheries (at Big Qualicum River, Quinsam River and Nanaimo River) which had substantial runs. It may be that these small streams have a more characteristic odor (Hasler and Scholz 1983) that allows coho to distinguish them more easily than the Trent River, which has unusually clear water and comparatively little organic debris. Perhaps this type of water does not provide strong olfactory cues, which may mislead fish that enter this stream. Increases in discharge rates may provide stronger cues, allowing fish to distinguish this system from their own, and induce them to move out as was observed during this study when floods occurred.

Finally, it should be noted that although the overall level of straying observed during the present study tended to be relatively small compared to the level of homing, some stocks exhibited a considerable amount of straying in some years. In such cases, not accounting for the strays would certainly cause bias in the estimates of total escapement, which would translate into positively biased estimates of exploitation rates. Strays accounted for a large fraction of the total escapement in some streams. Not accounting for this contribution would lead investigators to overestimate survival rates and underestimate the exploitation rates of the resident stock. This emphasizes the need to determine both the magnitude of straying and the overall contribution of strays to escapements before assessing survival and exploitation rates.

## 6.0 CWT RECOVERIES IN FISHERIES

To estimate the total catch of tagged fish associated with various CWT releases, information must be available on catch levels in various fisheries where tagged fish are intercepted, the sampling regimes, the catch reporting and estimation procedures, the identity of the tag codes recovered, and the sources of error associated each of these factors. In this Section, the information sources are described, and a detailed account is given of the procedures used to estimate the total recoveries in the catch.

### 6.1 Description of data sources

Information on the number of tags recovered in various fisheries is obtained through an extensive coast-wide sampling program of commercial and sport fisheries in U.S and Canada. In British Columbia, all major fisheries are sampled in such a fashion as to cover all seasonal periods and Statistical Areas. Commercial catch estimates are based mainly on records of fish sales by fishermen to processors. Commercial catches are examined and sampled at various landing sites along the coast throughout the fishing season in order to estimate the catch of tagged fish and to recover tagged heads. Sampling effort is allocated across areas and fisheries to ensure that approximately 20% of the weekly catch is sampled, as dictated by international agreements. This is accomplished by sampling all fish from 20% of vessels, trucks or packers (Kuhn *et al.* 1988). Sport catches are estimated from overflight<sup>27</sup> counts, interviews with anglers, and a coast wide program of voluntary reporting in which sport fisherman submit tagged fish heads to one of 232 'Head Depots' with information on the location of capture. Sampling of sport catches to determine the number of tagged fish caught is restricted to the Strait of Georgia where the vast majority of sport fishing activity occurs.

Kuhn *et al.* (1988) noted that commercial fishing vessels often catch fish in several adjacent Statistical Areas. This creates sampling difficulties at the landing sites, since the catch from different areas is often mixed. To minimize uncertainty, catch and sampling statistics are usually associated with catch regions, which are combinations of several adjacent Statistical Areas, fishing gears and time strata. Currently, over 40 catch regions are defined in the MRP database, and are routinely used for data

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<sup>27</sup> An overflight consists of an aerial survey conducted at a specific time period over the region surveyed. Typically, the entire Strait of Georgia is surveyed within a few hours by two technicians in a small plane. The flight leaves from Sidney, flies in a northerly direction along the Vancouver Island coast up to Quadra Island, and then returns south along the mainland coast. During each trip, the number of sport boats on each side of the plane is recorded along with the location and survey time.

reporting. Some catch regions consist of sport fisheries, others consist of troll, gillnet and seine fisheries, but all catch regions are representative of the current management boundaries. Therefore, catch regions represent a suitable stratum choice for contrasting the exploitation patterns of several stocks, particularly if the analysis has management implications (i.e. requires that distinction be made between the major fisheries).

Catch and sampling statistics can be extracted from the MRP database on an annual, monthly or statistical week basis depending on the type of catch region selected, so the level of stratification is not fixed. Since fishermen can catch fish in several regions during a single trip, and may fish during several time periods used for data reporting, it is preferable to use a level of resolution lower than the maximum allowable so as to increase the reliability of catch and sampling statistics. In terms of spatial resolution, the minimum level selected should distinguish between the major fisheries characterized by specific season lengths, gear types, sampling characteristics and geographic locations. These objective were met by first selecting seventeen commercial catch regions and seven sport catch regions for data extraction. The major catch regions where the stocks of interest were intercepted are abbreviated below by gear category from north to south, along with their corresponding Statistical Areas. Miscellaneous fisheries (enclosed within brackets) were also included in some regions to minimize the number of abbreviations used:

- A.TR: Southeast Alaska troll.
- N.TR: Northern troll. Areas 1, 2e, 2w, 3, 4, 5.
- NC.TR: North Central troll. Areas 6, 7, 8, 9, and 30.
- SC.TR: South Central troll. Areas 10, 11, 12.
- NW.TR: Northwest Vancouver Island troll. Areas 25, 26, 27.
- SW.TR: Southwest Vancouver Island troll. Areas 21, 22, 23, 24.
- GS.TR: Strait of Georgia troll. Areas 13, 14, 15, 16, 17, 18, and all sub-areas in 29.
- WA.TR: Washington troll. Combination of all troll fishing areas along Washington coast.
  
- A.N: Southeast Alaska net.
- N.N: Northern net. Areas 1, 2e, 2w, 3, 4, 5.
- C.N: Central net. Areas 6, 7, 8, 9, 10, 11.
- NWV.N: Northwest Vancouver Island net. Areas 25, 26, 27.
- SWV.N: South West Vancouver Island net. Areas 21, 22, 23, 24.
- JS.N: Johnstone Strait net. Areas 12, 13.
- GS.N: Georgia Strait net. Areas 14, 15, 16, 17, 18, (Area 29 Fraser River net incl.).

- JF.N: Juan de Fuca Strait net. Area 20.
- WA.N: Washington net.
  
- A.SP: Alaska sport.
- N.SP: Northern sport. Areas 1, 3, 5, 2e, 2w.
- C.SP: Central Sport. Central sport. Areas 6, 7, 8, 9, 10, 11, 12, 30.
- W.SP: West coast Vancouver Island sport. Areas 21, 22, 23, 24, 25, 26, 27.
- GS.SP (north): Northern Georgia Strait sport. Areas 13, 14, 15, 16.  
(associated fresh water sport catches included).
- GS.SP (south): Southern Georgia Strait sport. Areas 17, 18, 19, 20, 28, 29.  
(associated fresh water sport catches included).
- WA.SP: Combination of all sport fishing areas along Washington coast.

To determine the appropriate level of temporal stratification required for data extraction, and if additional pooling across strata is required for data analysis, consideration must be given to the specific objectives and requirements of the analysis, as well as the accuracy of the catch and sampling statistics on hand. Since commercial and sport catch and sampling statistics differ substantially in availability and reliability, the appropriate level of resolution used for data extraction must be determined separately for each type of fishery. The method and rationale used for selection of stratification levels used for data extraction will be addressed in Section 6.2, prior to estimating the number of recoveries in various fisheries.

## 6.2 Selection of stratification level for extraction of catch statistics

### Stratification of Strait of Georgia sport fishery data

Most of the sport fishing activity in B.C. is centered in the Strait of Georgia (Argue *et al.* 1983). Catch and effort statistics associated with this fishery are currently collected through the Georgia Strait Creel Survey Program (English *et al.* 1986). The program consists of two separate survey components: periodic interviews of anglers at marinas and boat launching sites, and a series of aerial surveys. Angler interviews are conducted in 30 coastal locations selected primarily on the basis of boat traffic volume. These interviews are conducted in various locations at certain times throughout the day. During an interview, information is obtained on the catch composition, and the duration, location and timing of fishing activity associated with each boat trip. The interview results are used to estimate the mean and

variance of the catch per boat trip ( $cpue$ ), and the proportion of the boats engaged in fishing during a given hour of the day ( $p(b)$ ), for each Statistical Area/month stratum. Aerial surveys are used to determine the number of sport boats ( $b$ ) in each area during specific time periods. Data from both sources are combined to produce monthly estimates the total effort ( $e = b \cdot p(b)^{-1}$ ), and total catch ( $catch = e \cdot cpue$ ) for each stratum.

Estimates of the proportion of coho caught that are tagged ( $p(tagged)_{catch}$ ) are also obtained during interviews from the ratio of the number of tagged coho observed ( $tags_o$ ) to the total number of coho examined ( $fish_o$ ) in each stratum. This proportion serves to estimate the catch of tagged fish in each stratum ( $catch_{tag} = catch \cdot p(tagged)_{catch}$ ). The stock composition within each stratum is based on the identification of the tags in heads returned voluntarily by anglers. Each year, only a fraction of the tagged fish caught by anglers are returned to DFO's Head Recovery depots located near the landing sites. This fraction, referred to as the 'awareness factor' (Kimura 1976), is estimated from the ratio of the number of heads returned ( $hr$ ) to the total catch of tagged fish in a given stratum ( $af = hr \cdot catch_{tag}^{-1}$ )<sup>28</sup>. The awareness factor represents the fraction of the total catch of tagged fish in a stratum that was sampled for CWT identification, and serves to estimate the total catch of tagged fish (i.e. estimated recoveries) associated with each contributing tag code ( $c$ ) in a given stratum ( $catch_{tag,c} = hr_c \cdot af^{-1}$ ). It should be noted that in the MRP database documentation, the term awareness factor does not have the same definition ( $hr \cdot tags_o^{-1}$ ), but the expression is used differently to provide an equivalent estimate of the catch of tagged fish.

A preliminary analysis conducted by DFO officials of trends in awareness factor indicated a substantial amount of variation, which could affect the results of this study. Therefore, a careful analysis of trends in awareness factors for the Strait of Georgia fishery was conducted as part of this study. Much of the remainder of this Section describes the results of this study.

Awareness factors were calculated for each month in nine Statistical Areas during the 1980-1988 period. Catch and effort statistics were obtained from the the Georgia Strait Creel Survey database (Tom Shardlow<sup>29</sup>, pers. comm.), and observed tag recovery statistics were obtained from the Head Recovery Program database (Vic Palermo<sup>30</sup>, pers. comm.). Minor adjustments were made to account for a small discrepancy between the geographical boundaries of the Statistical Areas used for data reporting in each program. Catch and effort data from Statistical Areas 19A and 19B used by the Creel Survey Program

<sup>28</sup> This expression is essentially equivalent to Kimura's (1976) formulae; ( $P_A = r_v / (1-f) M^V$ , when  $f=0$ ).

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<sup>30</sup> Mark recovery biologist, DFO Fisheries Operations Branch, Biological Services Division, Vancouver, B.C.

were combined, and considered as Area 19A-B+. Tag recoveries from Statistical Areas 19 and 20 used by the Head Recovery Program were combined and considered as Area 19A-B+.

The estimates obtained show a great deal of variation in awareness factor between strata (Tables 6.1, 6.2). Awareness factors could not be calculated for at least 50% of the strata each year, either because of the lack of surveys, the absence of tagged fish among those examined, or the lack of heads submitted by fishermen. Unrealistic estimates ( $> 1.0$ ) were also obtained in some instances each year, and can occur when the estimated catch of tagged fish in a stratum is less than the number of heads submitted. These results suggest that the sampling regime used for estimating sport catches in the Strait of Georgia is not suitable for producing reliable estimates of awareness factors at the Statistical Area by month level of stratification given past levels of participation to the Head Recovery program. This situation dictates that a different level of stratification be used for appropriate expansion of observed tag recoveries.

Examination of the awareness factor estimates for 1987 (arbitrarily selected) shows that the inter-stratum variation declines with increasing catches (Fig. 6.1). Since only a fraction of the catch is examined each year, this variability is also related to the number of coho and marked coho observed in each stratum. As a result, the awareness factors stabilize considerably when catches exceed 10,000 coho per stratum, and the number of coho and marked coho examined per stratum exceeds 500 and 50 respectively. In cases where these conditions were met, the awareness factors obtained during 1987 ranged from 0.01 to 0.41, with the mean, variance, and coefficient of variation being 0.18, 0.79, and 0.59 respectively.

The level of stratification was reduced for the estimation of awareness factors by aggregating statistics from adjacent area and time strata. For pooled strata, the awareness factor was estimated from:

$$(Eq. 6.1) \quad af = \frac{\sum_T \sum_A hr}{\sum_T \sum_A catch \cdot \frac{\sum_T \sum_A tags_o}{\sum_T \sum_A fish_o}}$$

in which:  $T$  = subset of the monthly time periods;  
 $A$  = subset of Statistical Areas;

The number of months and areas combined into each new stratum was chosen so as to meet two objectives. The first objective was to maintain some spatial and temporal resolution to capture any latitudinal and seasonal trends in awareness factors. The second objective was to ensure that the majority of the strata would have catch levels exceeding 10,000 fish and sample sizes exceeding 500. These two objectives were met by separating the fishing period into three seasons and four geographical regions: winter/spring (January-June), summer (July-August), fall/winter (Sept.-Dec.), north (Areas 13+15), central north (Areas 14+16), central south (Areas 17+29+28), and south (Areas 18+19A-B+). Pooling catches and samples into 12 strata resulted in nearly all estimated awareness factors being within realistic bounds (Table 6.3). Substantially less inter-stratum variability was apparent, and excessively large values were only obtained for one stratum (CS, spring 1982). Additional pooling was conducted to obtain yearly estimates of awareness factors for the Strait of Georgia. The figures obtained ranged from 0.15 to 0.31 during the 1980 to 1988 period, and have been below 0.2 since 1986 (bold values, Tables 6.1, 6.2).

Before conjecturing on the significance of the trends observed, numerical simulations were conducted to examine the potential bias associated with awareness factor estimates given the census methods used and the level of program participation. As a first step, a descriptive model was constructed to represent the surveying, sampling, and reporting processes in the sport fishery, as well as the procedure used to estimate the catch and awareness factors. Monte Carlo simulations were then conducted on this descriptive model to assess the magnitude and distribution of the discrepancies between the estimates generated and the corresponding parameter values based on the Creel Survey statistics reported in Table 6.4. Details of the model and the simulation procedures are given in the Appendix, and only the main results are presented here.

The simulation results indicated that the awareness factor estimates associated with catch levels ranging from 1000 to 60,000 fish per stratum almost always exceeded the actual values, irrespective of the survey year (Fig 6.3). Since there were no noticeable year to year differences in the patterns observed, all statistics for the 1985-1988 period were aggregated. The results show that the magnitude of the error decreased with increasing catch levels (Fig. 6.4). For the 1985-1988 seasons, the magnitude of the error was generally ~25% when catches were ~25,000, and ~10% when catches were ~40,000. Therefore, all  $af$ 's in the 1983-1988 dataset which were associated with region/season strata having catch levels smaller than 60,000 adults were corrected for potential errors. A linear regression of the natural logarithm of  $(1 + \text{error})$  against catch per stratum ( $r^2 = 0.59$ ) was used to calculate the correction factor for various catch levels.

Corrected  $af$ 's (Table 6.5) were transformed ( $\arcsin (af^{0.5})$ ), and submitted to a factorial Anova, with year, seasons, and regions considered as the main factors. Both area and year effects were found to be statistically significant (both  $F > 6.6$ ,  $P < 0.001$ ), but seasonal and interaction effects were found to be negligible (all  $F < 1.6$ ,  $P > 0.13$ ). It should be noted that the same test conducted on uncorrected  $af$ 's also produced the same results, which is attributed to the fact that the corrections tended to be relatively minor in the majority of cases.

Since seasonal effects were apparently minor, awareness factors were generated for each region and year by pooling statistics across all seasons (Fig. 6.5). Awareness factors tended to be highest in the central regions, smallest in the south, and have been decreasing steadily since 1983. It is likely that the apparent trend reflects a general reduction of participation in the Head Recovery program by anglers who object to recently introduced fishing regulations. During the 1988 summer season, fishermen from the Victoria region actually campaigned against the program. Such localized activity could account for the pronounced regional differences in  $af$ 's observed in some years (region S, 1988). The absence of seasonal differences suggests that the level of participation does not change as a result of the influx of tourists during the summer months. Such results are in agreement with those of Kimura (1976), who detected statistically significant differences in awareness factors between areas in the Puget Sound sport fishery during 1974, but observed no significant differences between months during that year.

The results indicate that expanding tag recoveries without consideration for the potential errors associated with the awareness factor estimates at low catch levels would underestimate the catches of tagged fish, as well as the resulting survival and harvest rates. This deficiency could be rectified in part by aggregating the data into coarser strata. In view of the significant differences in awareness factors between regions, pooling catch and sampling statistics across many Statistical Areas could produce estimates of awareness factors which are not representative of the regional conditions. Although significant differences in awareness factors could not be detected across seasons, such differences are still noticeable and may still reflect actual trends. If particular populations have distinct migration patterns, and are intercepted primarily in certain regions, expanding the associated observed recoveries by means of an unrepresentative awareness factor would lead to biased estimates of total recoveries. For these reasons, it might be advisable to restrict pooling as much as possible. The four region by three season level of stratification used in the remainder of this chapter appears to offer an acceptable alternative between the maximum and minimum stratification levels.

### Stratification of other sport fishery data

By comparison to the Strait of Georgia sport fishery, sport catches in the West, North and Central Coast regions (W.SP, N.SP and C.SP) are considered to be negligible, but tagged heads are occasionally recovered from these areas as well. Sport catch figures for these areas consist mainly of crude estimates provided by fishery officers based on interviews conducted at remote logging camps and fishing lodges. Additional information on the distribution of sport fishing activity has also been obtained from small creel surveys. The largest survey was conducted during 1986 in Barkley Sound where most of the sport fishing activity along the West coast of Vancouver Island takes place. The survey results indicated that most of the fishing activity occurred between June and September, and that the awareness factor during this period was close to 1.0 (DFO internal report). Further inquiries revealed that in such remote locations, the level of participation in the Head Recovery Program is usually quite high throughout the year, and is generally not representative of the level of participation observed in the Strait of Georgia (Margaret Birch<sup>31</sup>, pers. comm.). Therefore the number of observed recoveries in sport fisheries located outside Georgia Strait were extracted from the MRP database using a catch region by year level of resolution, and an awareness factor of 1.0 was assumed to apply for each stratum. Since it is doubtful that fishermen report all catches of tagged fish, the estimated catch surely underestimates the actual catch, but the difference is considered to have a negligible effect on the overall sport catch estimate (Table 6.6).

Catch estimates for the Alaska and Washington sport fisheries were provided to DFO staff by U.S. representatives, along with corresponding estimates of total CWT recoveries (by tag code) for each catch region by year stratum. Since the U.S. sampling programs differ considerably from those used in Canadian waters, no attempt was made to obtain additional information on awareness factors and potential sources of error.

### Stratification of commercial fishery data

For commercial fisheries in B.C., the observed recovery and sampling statistics are believed to be statistically reliable for combinations of catch regions and recovery month (Kuhn *et al.* 1988). However, for some commercial fisheries, recovery data are only available from the MRP database on a catch region by year basis. Therefore catch and sampling statistics for all commercial fisheries were

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extracted on a year/region basis to standardize the data extraction procedure across all fisheries and minimize the effects of misreporting errors. Estimates of commercial catches in the Alaska and Washington net and troll fisheries, and estimates of total CWT recoveries (by tag code) were provided to DFO staff by U.S. representatives for each catch region by year stratum. No attempt was made to determine the potential sources of error associated with the U.S. catch estimation process.

### 6.3 Estimation of CWT recoveries in sport and commercial fisheries

For each tag code listed in Tables 3.8 to 3.10, the number of observed recoveries in each Statistical Area by month stratum was determined for each age class separately. The number of 'non-tag' recoveries in a given stratum associated with various tag codes could not be determined solely on the basis of the relative contribution of the tag codes recovered in that stratum. However, Kuhn *et al.* (1988) noted that 'non-tag' recoveries in catch samples are relatively rare compared to escapement samples, because the number of 'no data' is usually negligible relative to the numbers of 'lost pin' and 'no pin' (most heads submitted are decoded). In addition, the number of 'lost pin' is generally negligible compared to the total number of tags successfully decoded (see Table 5.9). Therefore, it was assumed that the numbers of 'no data' and 'lost pin' associated with the number of observed recoveries in each stratum were negligible. With regard to the number of 'no pin', it was assumed that tagged fish rejected their tags prior to capture according to the corresponding estimates of tag loss at return (Section 5.2). The number of adjusted recoveries in the Strait of Georgia Sport fishery was estimated by accounting for this potential loss of tags:

$$(Eq. 6.2) \quad T_{adj,c,a,t} = T_{obs,c,a,t} \cdot \left( 1 + \frac{tag\ loss_c}{1 - tag\ loss_c} \right)$$

where: c = tag code;  
a = Statistical Area in the Strait of Georgia;  
t = time period (month):

For each group, the number of estimated recoveries in each region/season stratum was calculated based on the corrected awareness factor for that stratum:

$$(Eq. 6.3) \quad T_{est,g,r,s} = \frac{\sum_c \sum_a \sum_t T_{adj,c,a,t}}{af_{r,s}}$$

where: g = group (stock/year/age) represented by specific tag codes (c);  
 r = region including assemblage of Statistical Areas (a);  
 s = season associated with the range of months (t);

An estimate of the total number of tagged fish of each group recovered in the Strait of Georgia sport fishery was obtained by pooling the estimated recoveries of that group across all region/season strata:

$$(Eq. 6.4) \quad T_{tot, g} = \sum_r \sum_s T_{est, g, r, s}$$

For the remaining sport fisheries, estimates of total recoveries of tagged fish were obtained by adjusting the estimated recoveries in each region/year stratum for tag loss only since the awareness factors were assumed to be ~1.0:

$$(Eq. 6.5) \quad T_{tot, g} = \sum_c T_{est, g, c} \cdot \left( 1 + \frac{tag\ loss_c}{1 - tag\ loss_c} \right)$$

Estimated recoveries in Canadian commercial fisheries were extracted from the MRP database for each catch region by year stratum using the standard procedure described by Kuhn *et al.* (1988), in which the observed recoveries are expanded according to the sampling rates used in each fishery. Total recoveries by group in Canadian and American commercial fisheries were then estimated from Eq. 6.4.

The estimates obtained indicate that the vast majority of age 3+ adults were caught in southern B.C., where they contributed mainly to the Georgia Strait sport fishery, the South Coast and Georgia Strait troll fisheries, and net fisheries in Georgia Strait and Johnstone Strait (Table 6.6). Recoveries in net, sport, and troll fisheries of Alaska and the North coast were non-existent, and were negligible for some Central coast fisheries. A comparison of estimates in Tables 5.10 and 6.6 shows that for each group, the total number of age 3+ adults recovered in commercial and sport catches far exceeded the number recovered in the escapements. By contrast, jack catches accounted for less than 3% of the total jack return<sup>32</sup> associated with each group. Recoveries of age 4+ adults (ocean rearing > 2 years) accounted for up to 2.5% of the total recoveries for some groups, but on average, represented < 0.2% of the total catches of age 3+ adults from all groups. Catches of jacks and age 4+ adults were not

<sup>32</sup>Sum of catches and escapement of tagged fish from the same group.

considered sufficiently large for contrasting exploitation patterns among age classes, and their catch distribution was not tabulated.

To reduce the number of catch regions with negligible catches, and to facilitate the statistical comparisons of recovery patterns of the various groups, catches of some regions were pooled with those of adjacent and/or spatially overlapping catch regions. Pooling was selected so as to minimize loss of information about latitudinal gradients in catch distribution, which would be indicative of the migration range of each stock. The new fishing regions created for analytical purposes, and the catch regions included in each zone, were as follows:

- Northern fishery = North = A.SP + N.SP + A.TR + N.TR + A.N + N.N.
- Central fishery = Central = C.SP + NC.TR + SC.TR + C.N.
- Interception fishery = JS.N.
- Outside fishery (north) = WVI (north) = NW.TR + NWV.N.
- Inside troll fishery = GS.TR.
- Inside sport fishery (north) = GS.SP (north).
- Inside sport fishery (south) = GS.SP (south).
- Outside fishery (south) = WVI (south) = SW.TR + SWV.N + W.SP.
- Southern fishery = South = WA.SP + WA.N + WA.TR + JF.N.
- Inside net fishery = GS.N.

The condensed catch distribution of age 3+ adults from each stock was tabulated along with estimates of total return for jacks and age 4+ adults (Table 6.7). Statistical comparisons of the distribution of recoveries across stocks and seasons are presented in Section 7.1, along with an assessment of the effects of fishing effort and demographic traits. Only information on the relative rates of return by age class is presented here, as required for further analysis of smolt-to-adult survival rates.

The contribution of jacks to the total return of all age groups combined ranged from zero to 25.3%. Production groups from hatcheries at Quinsam River, Puntledge River and Big Qualicum River had the largest proportions of jacks in total returns ( $\bar{x} = 15\%$ ), while enhanced and colonization groups tended to have the lowest ( $\bar{x} < 5\%$ ). In some populations, the proportion of jacks in total returns varied considerably from year to year, but the magnitude and direction of the changes from year to year was not consistent across the populations exhibiting such variability. Within each season, significant correlations were found between the average weight of the smolts at release and the proportion of jacks in total returns ( $0.54 < r^2 < 0.74$ ). Such results are in agreement with those of Bilton *et al.* (1984) who found a

strong positive relationship between juvenile weight at release and the proportion of jacks in total returns. No significant relationship was found between the median date of juvenile migration and the proportion of jacks in total returns. No relationship was found between the number of smolts released, the average size at release, or the median date of juvenile migration and the proportion of age 4+ adults in total returns. On average, the Millstone River, Big Qualicum River and Puntledge River stocks tended to have higher proportions of age 4+ adults ( $\bar{x} \approx 1\%$ ), but no stock or stock type had a consistently larger proportion age 4+ adults in total returns.

#### 6.4 Estimates of smolt-to-adult survival rates

##### Temporal and geographical trends in survival rates

Estimates of ocean survival (smolt-to-adult) for each stock were obtained from the ratio of the total returns for the three age classes combined to the number of tagged fish released. As noted in Section 5.2, Blankenship and Hanratty (1990) suggested that survival estimates for groups which were tagged during their smolt migration stage be expanded by a factor of 1.19 to account for the detrimental effects of trapping and tagging, which do not affect other groups tagged at the hatchery several months prior to release. Correction factors were therefore applied to the survival rates of the selected groups to facilitate comparisons between stocks. Survival estimates ranged from as low as 0.5% for the 1985 Big Qualicum River production release, to 23.1% for the 1985 Quinsam River colonization release (Fig. 6.6). No particular stock or group exhibited consistently greater ocean survival, but the Big Qualicum River production releases were always characterized by the lowest survival. Survival rates for the Big Qualicum River stock over the 1986-1988 period contrasted sharply with those of the 1975-1982 period, which ranged from 15.5% to 41.6% for 16-26 g smolts released between May 1 and June 6 (Mathews 1984). Salt-water challenge tests performed on Big Qualicum River smolts prior to release during this study suggested that the three cohorts released were in relatively poor condition (G. Ladouceur, pers. comm.). Hatchery personnel hypothesized that the smolts suffered from a disease transmitted through contact with the rearing channel substrate, but all attempts to identify the disease failed (G. Ladouceur, pers. comm.). In the absence of any evidence to support this hypothesis, or evidence that this condition was endemic, the low survival rate of Big Qualicum River smolts cannot be considered as an anomaly. Still parasitic infections and various pathogens are known to have a major influence on survival in Pacific salmon (Wood 1979).

Stock specific estimates of survival averaged across all years ranged from 1% (Big Qualicum River), to 17.8% (Quinsam River colonization). Average survival rates were > 13% for Black Creek, Trent River and Millstone River stocks, and were < 6.5% for wild and enhanced stocks from French Creek and Quinsam River. Estimates of the mean ocean survival for each adult return year (averaged over all stocks), ranged from 8.3% in 1986 to 10.1% in 1988. Smolts from various brood stocks released from the Rosewall Creek hatchery exhibited similar survival rates, and none of the groups exhibited consistently higher or lower survival rates. However, all three stocks released from Rosewall Creek were characterized by lower survival rates than their natural counterparts for both years. The differences in survival rates ranged from 1.7 to 11.9 percentage points, and averaged about 6.2 percentage points for the six stock-year comparisons. The differences tended to be greatest for Black Creek broods, and lowest for Little Qualicum River broods, suggesting an apparent relationship between survival and the distance from the donor stream to the release location. Although efforts were made to minimize the differences in time and size at release between the Rosewall Creek releases and their natural counterparts, smolts from the same brood stock released at each location still differed in terms of time and size at release, release location and exposure to hatchery conditioning. These factors have been hypothesized to affect ocean survival, and must be accounted for to properly evaluate the influence of hatchery conditioning and outplanting on survival rates (next Section).

In theory, fish stocks subject to relatively high exploitation rates should exhibit higher survival rates than those subject to lesser rates of exploitation, since the fish are caught before natural mortality has taken its toll on the population. Linear regressions of survival rates against estimates of exploitation rates (from Section 7.2) showed no significant relationship between these variables within each year. These results may reflect the fact that ocean mortality rates are highest during early marine life when coho are not yet subject to exploitation. Potential losses due to natural mortality may also be replaced by equal rates of 'shaker mortality' associated with hook and line fisheries.

#### Effects of various factors on survival rates

The specific effects of various factors could not be tested by factorial Anova tests (too many missing values, and no replication), but such tests were attempted by means of linear logistic models. The environmental and biological factors tested were:

x<sub>1</sub>. The calendar year of adult escapement (1986 to 1988);

- x2. The genetic make-up of the stock released. Eight genetic categories were used for classification: Quinsam (1), Black Cr. (2), Puntledge (3), Trent (4), B. Qualicum (5), L. Qualicum (6), French Cr. (7), and Millstone (8). Wild, colonization and production fish released within the same system were considered to be members of the same category. Rosewall Creek releases from different brood stocks were categorized according to the parent stock used (2, 4, and 6);
- x3. The release location, expressed as a distance (in km) from Nanaimo (stream latitude);
- x4. The experimental treatment (i.e. level of hatchery conditioning). The number of months each stock was reared in a hatchery environment was used as a measure of conditioning;
- x5. Smolt size at release (smolt weight selected for consistency with other studies);
- x6. The median date of smolt migration (calendar day);
- x7. The proportion of adults that reared outside of the Strait of Georgia. The index used was the ratio of the total catch of age 3+ adults in all non-Georgia Strait fisheries to the total catch of age 3+ adults in all fisheries;
- x8. The relative contribution of jacks to total returns for all age groups;
- x9. The starting date of the upstream migration of adults (calendar date from Section 4.2);
- x10. The estimated total return of tagged jacks and adults from the same brood year.

For the present analysis, the ratio of total returns to smolts released for each stock was considered as the response variable of interest (smolt-to-adult survival rate). Before attempting to determine which factors influence survival, a correlation matrix of the variables was constructed to determine if any of these were correlated. Substantial correlation ( $r^2 = 0.65$ ) was found between size at release and the relative abundance of jacks in the returns ( $x_5$ ,  $x_8$ ), so the latter variable was omitted from the set of variables tested. Still, given the number of observations (35), the effects of all remaining variables and their first order interactions could not be assessed simultaneously since the total number of predictor variables would exceed the number of observations. Therefore, the number of first order interactions tested was limited to those combination of factors which were hypothesized to have an influence upon survival rates through some plausible mechanism. Since time and size at release have been shown to have an influence upon survival rates and exploitation patterns in previous studies, the interaction of these variables was considered as a factor. In addition, the interaction of each of these variables with other factors such as year, genetic make-up and stream location were considered as factors. No other interaction was considered as a potential factor in the analysis, and the effects of each factor were assessed by the procedure described in Section 5.3.

The results indicate that all factors listed contributed to a significant improvement in fit (Table 6.8). Model 10 was considered to be adequate for describing survival, and provided a very good fit to

the data ( $r^2 = 0.95$ ). Thus, the inclusion of the run timing factor was not necessary to adequately describe survival, although it could be argued that this factor had a significant influence on the response. This could be interpreted as evidence that pre-spawning mortality, which is often hypothesized to occur when the migration is delayed by a prolonged period of drought, is not a major determinant of survival. A sensitivity analysis showed that the inclusion of factors such as genetic make-up, return year, and smolt migration date had a large influence on the performance of the model, while others such as treatment, stream latitude and the date\**size* interaction had a relatively smaller influence (Table 6.9).

Information on the relative magnitude and direction of change in the predicted response was obtained by examining how it was affected by variation in the levels of each factor. Survival was found to be positively related to the proportions rearing in outside waters. Survival was also positively related to the total number of recoveries, but doubling the total returns translated into non-significant increases in predicted survival ( $\approx 0.3\%$ ) for all stocks. This suggested that the survival estimates for stocks with relatively small returns were not more biased than those with large returns. However, the existence of this relationship indicates that the survival estimates may be negatively biased to a small extent due to the relatively low number of tagged fish released in each group. Further attempts to correlate estimates of ocean survival with the total number of tags released also showed no significant relationship between these variables within each year or across all years.

Survival was found to be inversely related to the latitude of the stream, with smolts released from northern locations surviving at a slightly lower rate than their southern counterparts. Survival was also found to be inversely related to the duration of hatchery rearing. For instance, doubling the rearing time of colonization fry released into the Puntledge River while maintaining the same size at release is predicted to decrease survival from 5.9% to 5.3%. Increasing rearing time is usually accompanied by a larger size at release, which was also found to be positively related to predicted survival in most cases. However, the magnitude and direction of the change in predicted survival associated with comparable increases in weight at release varied slightly from stock to stock (genetic\**weight* interaction effect). For instance, increasing size at release by 10% would cause a 3 percentage point reduction in predicted survival among colonization fish from the Puntledge River. The effect of smolt size was also found to be influenced by the time of ocean entry (weight\**date* interaction). For instance a 10% increase in smolt size was predicted to cause a 9 percentage point reduction in survival among Trent River smolts released in 1986, but only a 7 percentage point reduction in survival if the same fish had migrated to sea 10 d later. These results suggest that the influence of smolt size upon survival is greatest when the peak migration date occurs relatively early in the season.

Delays in time at release were predicted to always result in a relatively large increase in survival, but the increase associated with a given difference in migration date differed between years (date\*year interaction). A one week delay in the ocean entry date of the colonization fish from the Puntledge River would have caused the predicted survival rate to increase by 25 percentage points in one year and 29 percentage points in the other year. The magnitude of the change in predicted survival associated with delays in release date indicate that relatively small changes in time of ocean entry can have a much greater influence upon survival than substantial changes in smolt size at release and treatment effects. These results agree with those of Bilton *et al.* (1982) who showed that the ocean survival of coho reared at the Rosewall Creek hatchery was affected by smolt size and time of release, and release size had less effect on survival than release date.

The nature of the factor(s) accounting for the differences in survival between Rosewall Creek releases and their natural counterparts was assessed with the model. This was accomplished by first predicting the survival rates that would have been obtained for the different groups if the smolts had been released at the same time and size as their natural counterparts. Under such conditions, the differences in predicted survival rates between the hatchery reared and naturally reared smolts of the same brood stock ranged from 0.1 to 5.7 percentage points, and average 2.5 percentage points for the 6 comparisons possible during the 1987 and 1988 return years. Thus at least half of the discrepancy in survival rates observed between the corresponding wild and hatchery releases were attributed to differences in time and size at release, which were occasionally considerable despite an attempt to minimize them (for instance 13.5 g for Black Cr. released at Rosewall in 1986 vs. 18.5 g for the natural group). Having accounted for the effects of time and size at release, the remaining discrepancies in predicted survival between wild and hatchery releases could be attributed to differences in rearing conditions and release locations. The effects of hatchery conditioning were assessed by predicting the survival rates of the various Rosewall Creek releases that would have been obtained in the absence of hatchery conditioning under identical time and size at release. The predicted difference in survival rates between the hatchery reared and naturally reared smolts of the same brood stock ranged from -0.2 to 3.9 percentage points and averaged approximately 0.5 percentage points. Thus hatchery conditioning accounted for almost all of the remaining differences in survival rates between the corresponding releases. These results indicate that releasing smolts at locations other than their parental stream does not necessarily have a major influence upon the survival rates of the smolts released.

The influence of genetic differences among the Rosewall Creek releases was also assessed with the model. The survival rates of the three distinct groups released each year were predicted after standardizing the smolt weights and total recoveries across all groups within each year. The predicted

survival rates for the 1986 and 1987 releases were 5.0% and 5.4% for Black Creek fish, 9.7% and 10.4% for Trent River fish, and 8.1% and 8.7% for Little Qualicum River fish. Thus if all smolts had been released at identical weights each year, smolts from Trent River brood would have exhibited consistently greater survival, while those of Black Creek brood would have exhibited the lowest survival. These results support the hypothesis that genetic factors have a major influence upon survival. This conclusion would not have been reached if inferences had been made based on a simple examination of the estimated survival rates associated with each genetic group (Fig. 6.6). This emphasizes the need to standardize time and size at release when conducting experimental releases to assess treatment and genetic effects on survival rates.

#### Discussion of the survival pattern results

Mathews and Ishida (1989) reported that the effects of release size on survival were not consistent in their experiments. Bilton *et al.* (1982) showed that factors such as size at release have *systematic* effects upon survival; survival increases with smolt weight up to a certain point, and then decreases afterwards with further increase in weight. The results of the present study are in agreement with these previous studies. Increasing size at release did not always translate into increases in survival rate, and in cases where a negative relationship between both variables was predicted, it is hypothesized that smolt weight at release exceeded the optimum size for survival.

Mathews and Ishida (1989) reported that higher survival was always associated with later release dates. Bilton *et al.* (1984) conducted a series of smolt releases from the Quinsam River hatchery, and found that when smolts of similar weights were released at different intervals between late April and mid-June, survival increased up to late May and decreased thereafter. The results of the present study are also consistent with those obtained by these investigators. Survival was always positively related to release date, because all median dates of smolt migration (i.e. release date) observed during this study occurred during May. Presumably, time at release would have been shown to have a systematic effect in the present study as well if some of the smolt populations used had entered the ocean predominantly in June.

Bilton *et al.* (1982) suggested the existence of 'optimum release windows', which provide optimum conditions for survival through the abundance of forage organisms. Mathews and Ishida (1989) found little evidence that fish released earlier encountered poor feeding conditions, but suggested that predation was the major determinant of time-varying mortality. Although the major mortality agents

are not known with certainty, the existence of so-called 'optimum release windows' is a generally accepted concept. Bilton *et al.* (1982) noted that the response surfaces (relation between adult returns, release time, and release size) generated for the Rosewall Creek releases did not appear to be applicable to other hatcheries, and suggested that there may be a different optimum release window associated with each location. The finding that the genetic\*size interaction is a major determinant of survival certainly supports this hypothesis for it suggests that different populations are affected differently by changes in size at release.

Mathews and Ishida (1989) suggested that the optimum release date may in fact be an 'inter-annually moving target' dependent upon a variable receiving environment, and/or a function of a smolting response that depends on variable factors in the fresh water rearing environment. If the first hypothesis is correct, then one could expect year to year changes in survival rates for comparable releases from the Rosewall Creek facility. Bilton *et al.* (1982) showed that the highest proportion of adult returns ( $\approx 43\%$ ) was obtained at Rosewall Creek by releasing 25 g smolts during early summer (June 22, calendar day 143). Given the approximate smolt weight and time of release used during the present study at Rosewall Creek in 1986 and 1987, anticipated returns of 20% and 18% should have been obtained based on the response surface generated by Bilton *et al.* (1982) from the 1975-1976 returns to Rosewall Creek. Instead, adult returns amounted to approximately 8% of the smolts released from Rosewall Creek each year, which was less than the lower bound associated with Bilton's predicted return rate ( $\approx 12\%$ ). It could be hypothesized that the results were induced by the release of smolts from foreign brood stocks not adapted to rearing conditions near Rosewall Creek, but foreign brood stocks (Big Qualicum R.) were also used in experiments conducted by Bilton *et al.* (1982). Therefore, the results are interpreted as support for the inter-annually moving target hypothesis. If the optimum release date does change from year to year, then one would expect that the year\*date interaction would have a major influence upon survival, as was shown in the present study.

The significance of the effects of the outside rearing proportions on survival rates was also anticipated. Mathews and Buckley (1976) noted that among Puget Sound coho stocks, those which reared in the sound during their ocean life (the so-called resident populations) showed more pronounced decline in abundance than the ocean-going populations, and suggested that more favorable ocean conditions existed in outside waters than in Puget Sound. Coho populations from tributaries to the Strait of Georgia are also known to migrate to outside waters in different proportions (Argue *et al.* 1983). Ricker (1972) interpreted the consistent difference in mean size of coho taken in Puget Sound and the Strait of Georgia, as compared to outside waters, as resulting from environmental effects. After further analysis, Ricker (1981) concluded that coho stocks which migrate to outside waters grow faster than

those remaining inside. Presumably, conditions favorable for growth are also conducive to greater survival, which explains why stocks migrating to outside waters generally survive better than those remaining inside.

Nickelson (1986) showed evidence that sea surface temperatures off the coast of Oregon were correlated with the marine survival of hatchery coho, with higher temperatures resulting in lower survival. However, he found no significant correlation between survival of wild coho and ocean temperature, which suggested that hatchery and wild populations responded differently to ocean conditions. In the present study, exposure to hatchery conditioning was found to influence survival, although the mechanism by which survival is reduced cannot be determined.

It should be stressed that simply classifying stocks into hatchery and non-hatchery categories as done by Nickelson (1986) tends to hide the fact that the stocks compared may also be genetically distinct. Genetic factors alone could very well have a greater influence upon survival than hatchery conditioning *per se*. The results of the present study suggest that certain stocks can exhibit substantially higher survival rates than others, and that the effects of genetic make-up far outweigh those of the hatchery treatment factor. There is considerable evidence of the extensive role of hereditary factors in characterizing life history traits of various populations (Ricker 1972). Hatchery managers are well aware of the fact that certain coho strains perform better than others under hatchery conditions. Gowan and McNeil (1984) gave evidence that coho smolts obtained from particular hybrid stocks can have substantially higher ocean survival rates than other hybrid and non-hybrid coastal stocks. These results indicate that when the performances of hatchery and non-hatchery fish are compared, fish from the same parent stock should be used.

#### Co-variation in survival rates and selection of an indicator stock

Statistical comparisons of survival rates among stocks are hampered by the lack of replicates for all stocks within the same year, and the existence of distributional uncertainties associated with catch and escapement statistics. In view of these facts, survival rates were contrasted statistically by means of non-parametric tests. As recommended by Zar (1984), testing of the differences among proportions (i.e. survival rates) is best done by expressing them as ratios, which are then subject to contingency table analysis. Therefore, survival rates for each stock were expressed as the ratio of total return to total releases minus total return (survivor : dead). The contingency tables generated were then analyzed by means of log-likelihood ratio tests, which are preferred over chi-square tests (Zar 1984). In all cases

where statistical comparisons were made between adjacent populations within a given year, significant differences in survival rates were detected, except for the comparison between two stocks released from Rosewall Creek in 1987 (Trent R. and Little Qualicum R.,  $P = 0.07$ ).

Given the uncertainty over the suitability of the chi-square distribution as the appropriate distribution for the assessment of differences in survival rates (as done when such tests are used for contingency table analysis), the finding that neighboring stocks differ significantly from one another should be interpreted with caution. Greater emphasis should be placed on knowledge of the magnitude of the differences in survival rates between stocks from the same stream, and from adjacent streams. The differences in survival rates between stocks from the same stream ranged from 0.2% for Rosewall Creek smolts released in 1987, to 14% for smolts released from the Quinsam River in 1985, and averaged 4.4% over all possible comparisons. The differences in survival rates between stocks from the adjacent streams ranged from 0.48% between Little Qualicum River and French Creek smolts released in 1987, to 15.6% between wild smolts from Black Creek and Quinsam River released in 1987, and averaged 7.7% over all possible comparisons.

An assessment of the co-variation in survival rates was made on the basis of the survival rates observed during the present study. No attempt was made to adjust the estimates of survival rate for Black Creek and Trent River to account for potential differences in migration dates between the tagged group and the actual smolt population (Section 3.3), because of the unknown relationships between this factor and other factors affecting survival rates (% outside, total returns, run timing, etc). As described in Section 5.4, estimates of year to year changes in survival rates were calculated for each stock from the difference in survival rates between two consecutive years (1986-1987, and 1987-1988).

The results show that four stocks clustered together; production fish from Big Qualicum River and Quinsam River, colonization fish from Quinsam River, and wild fish from Little Qualicum River (Fig. 6.7). The remaining stocks were scattered in three other quadrants, and failed to show much similarity. Stocks which were monitored for a two year period could not be incorporated directly into the plot, but could be lined up along the axis at levels corresponding to the associated change in exploitation. The Quinsam River wild and the two Rosewall Creek stocks might have clustered with the first four stocks if they had been monitored during all three years, which would have produced a cluster containing a maximum of seven out of fourteen stocks. These results indicate that different stocks from the same stream do not always exhibit the same year to year trends in survival rates. This conclusion also applies to stocks subject to the same treatments, since wild, enhanced, and colonization stocks did not cluster together. In fact, the composition of the only apparent assemblage clearly indicates that stocks

exhibiting similar trends in survival rates are not necessarily from adjacent streams, and may include stocks of wild, production and colonization origin. Thus, no single stock would be representative of the year to year variation in survival among the majority of stocks within the assemblage, or among stocks of a particular type.

The level of similarity in survival rate between a given stock and the average annual survival rate was quantified according to Eq. 6.6 to determine which stock would best describe the average survival rate for the entire stock assemblage:

$$(Eq. 6.6) \quad MSD_s = \frac{\sum_y (SR_{s,y} - SR_{a,y})^2}{Y}$$

when: *MSD* = Mean squared deviation from the average survival rate for the assemblage;  
*SR* = Survival rate for a stock/year (s,y) or assemblage/year (a,y) combination;  
*Y* = Total number of years for which survival rates were determined ( $\leq 3$ ).

The results indicate that the Quinsam River production stock would have been the best candidate since the Rosewall Creek stocks were only monitored for two of the three years (Fig. 6.8). The survival rates of all other stocks still differed considerably from that of the Quinsam River production stock, with the differences ranging from -14.0 to +10.9 percentage points. It should be noted that this stock is from a stream located at the northern edge of the study area. This indicates that the stock selected for monitoring the average exploitation rate does not have to be from a stream located at the center of the study area, as is often hypothesized by local fisheries biologists.

## 7.0 COMPARISON OF EXPLOITATION PATTERNS

### 7.1 Catch distribution patterns

#### Temporal and geographical trends in catch distribution

The various stocks used in this study exhibited similar patterns in geographic distributions of catches (Fig. 7.1). With a few noticeable exceptions, the dominant catch component was the Georgia Strait sport fishery. Examination of the fishery contribution patterns of each stock revealed an apparent relationship between the catch region and the latitudinal location of the stream of release and recovery. There was a tendency, particularly during 1985, for northern stocks to be caught in greater numbers in the Central troll and Johnstone Strait net fisheries, and for southern stocks to contribute more to the southern troll and the sport fisheries. This distinction became less pronounced during 1987 and 1988 with the increasing contribution of each stock to the sport fishery in the northern section of the Strait of Georgia. The pronounced change in catch distribution from 1986 to 1988 was probably induced by a 14% increase of the sport fishing effort in the Strait of Georgia during this period, which in turn was accompanied by a 90% increase in sport catch (Table 6.4). Declines in chinook stocks may have also contributed to the increase in coho catch through a potential re-direction of fishing effort. Sport fishing regulations during this period did not change substantially, and consisted mainly of a 30 cm size limit, and the closure of approximately 30 small areas for certain periods of the year (spot closures).

Net fishery openings in Johnstone Strait (JS) and Juan de Fuca (JF) occur intermittently during the migration periods of major salmon runs of southern B.C. and Puget Sound. Coho tend to be caught incidentally in these fisheries which target primarily on chum, pink and sockeye stocks. The ratio of catches in the net fisheries of Juan de Fuca to catches in the Johnstone Strait was relatively higher for southern stocks every year (Fig. 7.2). Regressions of the catch ratios against the latitude of the stream of release (in km from Nanaimo) indicated a significant relationship between both variables for 1986 and 1987 ( $F = 23.4, 16.2$ , all  $P < 0.003$ ). This relationship was also evident during 1988, but was found to be significant only if the effect of an apparent outlier (Millstone R.) was removed ( $r^2 = 0.44$ ,  $P = 0.019$ ). Given the year to year consistency in the trends, these patterns are considered to reflect relative differences between stocks in the migration route followed on their return to inside (Georgia Strait) waters. The progressive decrease in average catch ratios from 1986 to 1988 appears to indicate a general reduction in the proportion of each stock returning through the southern route, but this trend is not clear given the year to year variation in total effort and the distribution of fishing effort within each fishing

region (Table 7.1). No attempt was made to quantify the effects of these factors, in view of the lack of sufficient time series for proper assessment. Further conjecture on the nature of the factors responsible for the year to year variation in the average JF/JS catch ratios is not warranted.

Coho stocks monitored during this study do not contribute significantly to the terminal net fisheries operating near the mouth of the Fraser River, but contribute substantially to the inside net fisheries in Area 14 which target on large runs of chum salmon returning mainly to the Big Qualicum River hatchery during November. Occasionally, the openings for this net fishery are adjusted to coincide with the coho escapement period if substantial coho returns to the hatchery are anticipated (Wilf Luedke<sup>33</sup>, pers. comm.). This net fishery intercepted mainly stocks located in the center of the study area. The coho catch in this fishery usually accounted for a small fraction of the total catch of age 3+ adults (mean  $\approx$  2%, range: 0-13% ), and tended to be largest for stocks escaping to streams in the vicinity of Baynes Sound. Although the relative contribution of coho stocks to this fishery was small, this fishery can have a pronounced impact upon the potential spawners of each stock, which have managed to escape the hook and line fisheries and the interception net fisheries. Estimates of exploitation rates on this segment of each stock were obtained from the ratio of net catches to the sum of escapement and net catches. The exploitation rates of the inside net fishery during the 1986-1988 period were always less than 3% on stocks from streams located north of the Puntledge River. Stocks from Rosewall Creek were always exploited at the highest rates, which ranged from 47% to 65% in 1987, and 20% to 23% in 1988. Stocks from streams located between Rosewall Creek and the Puntledge River, and from streams located to the south of Rosewall Creek were subject to inconsistent rates of exploitation which were relatively low in 1988 (mean = 6%), and intermediate in 1986 and 1987 (means of 21% and 23% respectively).

#### Identification of stock assemblages from catch distribution patterns

Kolmogorov-Smirnov tests were used to assess the overall level of similarity between stocks in terms of fishery contribution. As recommended for this type of test, the number of *observed* recoveries in each of the ten regions was used for comparative purposes instead of the number of estimated recoveries. For stocks with relatively large tag returns, the number of recoveries in each region was reduced proportionally so that the total number of recoveries was equal across all stocks within each year. Pair-wise comparisons of recovery patterns were conducted to reveal differences between specific

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stocks, and significant differences were detected between many pairs each year (Table 7.2). Nearly identical results were obtained when additional tests were performed on the total estimated recoveries, which indicated that the distribution of observed recoveries was not substantially different from the estimated recoveries.

Using the probability levels associated with  $D_{\max}$  as a measure of similarity, it is possible to distinguish assemblages of stocks with similar catch distributions. Interpretation of the statistical results obtained is best done in conjunction with an examination of plots of the cumulative distribution of catches (Fig. 7.5 to 7.7). For 1986, two distinct assemblages emerged. The first assemblage included production fish from the Puntledge River and all stocks to the north of this river. The second assemblage consisted of all remaining stocks. These two assemblages appeared to differ mainly with regards to the contribution of their stock to northern fisheries. It should be noted that the catch distribution of colonization fish from the Puntledge River was similar to the southern stocks, and was significantly different ( $P = 0.005$ ) from that of the production fish from the same river.

Similar results were obtained for 1987. The stock composition of the first assemblage was identical to that of 1986. All stocks south of the Puntledge River were included in the second assemblage, and contributed substantially less to northern fisheries than members of the first assemblage. Within the second assemblage, significant differences were detected only between two stocks (Trent River and Little Qualicum). Examination of the recovery patterns in Fig. 7.1 shows that the relative contribution of the latter stock to the southern fisheries was slightly larger. This may reflect an actual difference, or may consist of a typical Type I error (Zar 1984) which tends to occur when a two sample testing procedure is used in a multi-sample context (as in the present case). It should also be noted that once again, the catch distribution of the colonization fish from the Puntledge River was significantly different from that of the production group.

For 1988, similar results were obtained, but the distinction between assemblages was less clear. The first assemblage was reduced to Black Creek and all stocks from more northern locations. The production and colonization groups from Puntledge River did not exhibit significant differences in catch distribution, and were both in the second assemblage which included all stocks from streams between Puntledge River and Big Qualicum River. It should be noted that the distinction between the first and second assemblage is mainly induced by the influence of the Puntledge River stocks. If the colonization and production groups of this river had not been included in the comparisons, then the first assemblage would have included all stocks from streams north of Little Qualicum River. This would represent a

marked departure from the patterns observed in previous years, but in the absence of evidence to indicate that these stocks exhibited abnormal behavior, the composition of each assemblage was not altered.

Stocks from Little Qualicum River, French Creek and the Millstone River exhibited peculiar trends in catch distribution during 1988, and could not always be readily categorized. Examination of the recovery pattern in Fig. 7.1 shows that an unusually large fraction of the French Creek fish were caught in the inside sport fishery, while fish from the Millstone River had the most uniform recovery pattern of all stocks. However, the catch distributions of these stocks did not always differ significantly from each of those in the second assemblage, since pair wise comparisons revealed substantial similarity between them in several cases. Furthermore, the same tests conducted on the number of *estimated* recoveries showed slightly fewer significant differences between stocks, which suggests that the differences observed might have been influenced by CWT sampling regimes. Given the lack of consistency, and the fact that these stocks were previously associated with those of the second assemblage, these three stocks could still be considered as members of the second assemblage if little weight is given to the anomalies observed during 1988.

To assess the effects of inside net fisheries upon the similarity patterns in catch distribution, the same tests were repeated using recovery data for only nine regions (all inside net catches omitted). Nearly identical results were obtained, and the number and type of pair-wise comparisons which resulted in significant differences remained the same. This indicates that inside net fisheries are not a major determinant of the overall catch distributions, and the stock composition of each assemblage. The test results obtained are undoubtedly influenced by the stratification level selected for comparisons, but further comparisons of catch distributions based on a reduced spatial level of stratification were not done. No alternative combination of regions and fisheries was judged to be more suitable for comparative purposes. Conducting further tests based on pooled data was not considered to be warranted since at least 10 categories should be used when conducting goodness of fit tests on two distributions (Watson 1957).

The above results indicate that the geographical location of a stream has some influence on the catch distribution of the resident stocks. The consistent similarities observed between certain stocks over the three seasons indicate that coho populations from streams located between Courtenay and Campbell River have similar catch distributions, which differ from those of streams located between Courtenay and Nanaimo. Within each assemblage, and even within a single river, significant differences in catch distributions can occur, presumably as a result of inter-stock differences in biological attributes.

For the present study, all stocks considered equally susceptible to exploitation must satisfy two conditions: they must have similar catch distributions, and must be subject to similar exploitation rates. Given that significant differences in fishery contributions were detected among the stocks monitored, the null hypothesis that all stocks within the study site are equally susceptible to exploitation is rejected. However, stocks from streams located between Black Creek and Quinsam River always satisfied the first condition, as well as those from streams located between Puntledge River and Big Qualicum River. In some years, additional stocks located near the edge of each geographical region could also be included in these assemblages. Thus, if comparisons are limited to the stocks within each assemblage, then the first condition would be met in most cases.

The first assemblage included stocks of wild, colonization and hatchery origin from a geographical region covering 23 km of coastline. Since no stocks located beyond the Quinsam River were monitored for this study, it is impossible to properly define the northern limit of the first assemblage. Furthermore, the results suggest that the southern boundary of the first assemblage could oscillate in latitude from year to year between Courtenay and Black Creek. Thus, the geographical area covered by the first assemblage should be considered as a minimum range. The second assemblage also included stocks of various origins, and covered a coastal region of at least 40 km, and as much as 81 km if all stocks from streams between Courtenay and Nanaimo are included. Although the boundaries of each assemblage were not shown to be fixed, the fact that certain stocks were consistently included in each assemblage suggests that there is some year to year stability in the regions covered by each assemblage, provided that no major changes are made to the existing structure in fishery composition and season duration.

#### Genetic and rearing effects on catch distribution patterns

An assessment of the influence of genetic factors on catch distributions was made by first contrasting the distributions of the three groups released from the Rosewall Creek hatchery. Adults of Black Creek brood stock released from Rosewall Creek tended to contribute more to the northern fisheries (North, Central, JS.N, WVI(n) and GS.TR) than those of the two other broods, particularly during the 1987 return year (Fig. 7.1). An examination of the fishery contribution patterns of their three natural counterparts (Fig. 7.1, 7.5 - 7.7) shows that the Black Creek stock also contributed proportionally more to the northern fisheries than the Trent River and Little Qualicum River stocks during 1986, 1987 and 1988. Thus, it would appear that genetic factors have some influence upon catch distribution patterns, which is maintained to some extent despite hatchery rearing and being released

from a different location. However, Kolmogorov-Smirnov tests indicated no significant differences in catch distributions between the three different stocks released from Rosewall Creek in 1987 and 1988 (Table 7.2). Thus, it could not be demonstrated that the small differences in catch distributions between the three stocks released from the hatchery were induced by genetic factors.

The effects of hatchery rearing and transplanting distance can be assessed by contrasting the catch distribution patterns of the three stocks released from Rosewall Creek, to those of their natural counterparts which emigrated from each corresponding donor stream. Pair-wise comparisons by means of Kolmogorov-Smirnov tests indicated no significant differences between the two stocks of Black Creek origin, or between those of Trent River origin, or between those of Little Qualicum River origin (Table 7.2). No relationship was found between the distance separating Rosewall Creek from the donor stream and the level of similarity in catch distribution between the corresponding stocks. Thus, rearing and releasing smolts at locations other than their parental stream appears to have no major impact on the subsequent catch distribution of adults, if the smolt populations released from both locations are comparable in time and size at release, and the distance between both release locations is <45 km.

Hatchery rearing effects were assessed by comparing the catch distributions of fish released from the same location, but exposed to different levels of hatchery conditioning. During 1987 and 1988, the colonization stock from Quinsam River contributed slightly more to the northern fisheries than the production stock, but no significant differences in catch distribution were found between the production, colonization and wild stocks from this river during the 1986, 1987 and 1988 return years (Table 7.2). For Puntledge River stocks, significant differences in catch distribution were detected between the production and colonization stocks only in 1986 (not 1987 and 1988). Thus, it could not be conclusively demonstrated that hatchery rearing influenced the subsequent catch distribution of adults from a particular stream. It should be noted that production stocks from the Puntledge River tended to contribute more to the northern fisheries than colonization stocks, which is the opposite of the pattern observed for Quinsam River stocks. Colonization smolts migrated to sea earlier than production smolts at Quinsam River, but the opposite situation generally prevailed at Puntledge River. This suggests that within a stream, smolts migrating relatively early in the season may exhibit a greater tendency to migrate north. This observation agrees with the findings of Irvine and Ward (1989) who showed that coho smolts which left the Keogh River early in the smolt run generally had a more northerly marine catch distribution than later migrating smolts.

In the above comparisons, hatchery rearing appeared to have a greater influence upon catch distributions when comparisons involved fish released from the same stream, than when comparisons

involved different release locations. Since the process of rearing and releasing smolts in a non-parental stream could not be shown to influence the catch distributions, one is led to hypothesize that hatchery conditioning effects were influenced by other factors. As noted previously, attempts were made to minimize the differences in time and size at release between the Rosewall Creek releases and their natural counterparts. As a result, the smolt populations at Black Creek, Trent River and the Little Qualicum River migrated to sea at similar sizes and times as their Rosewall Creek counterparts in both years where releases were made. By contrast, wild, colonization and production groups from the same stream (Puntledge R., or Quinsam R.) exhibited much greater differences in time and size at release among themselves (Fig. 3.9 and 3.10). Such facts suggest that the effects of hatchery conditioning are best assessed from the results of comparisons involving Rosewall Creek releases than from the comparisons involving Quinsam River or Puntledge River releases. If so, hatchery rearing would not appear to be a major determinant of adult catch distribution.

## 7.2 Exploitation rates

### Temporal and geographical trends in exploitation rates

Exploitation estimates of adults (age 3+ and 4+ combined) for each stock were obtained from the ratio of total catch to total return (catch + escapement) for these age groups (Fig. 7.3). The ranges of estimates were 71-89% for 1986, 71-97% for 1987, and 61-91% for 1988. Estimates of mean annual exploitation rate, obtained by averaging stock specific estimates each year (not weighted by stock sizes) were 77.9% for 1986, 84.8% for 1987 and 80.3% for 1988. The influence of inside net fisheries in the Strait of Georgia on the stock-specific exploitation rates were assessed by adding inside net fishery catches to escapements for each stock (i.e. assume no inside net fishery), and recalculating the exploitation rates (Fig. 7.4). In the absence inside net fisheries, the average exploitation rates would have decreased by 3.5% in 1986, 4.7% in 1987, and 1.4% in 1988.

Inside net fisheries had their greatest impact upon the Little Qualicum stock in 1986 where they accounted for 12% of the exploitation rate, and the Rosewall Creek and Little Qualicum River stocks in 1987 where they respectively accounted for 10% and 8% of the exploitation rate. The location of greatest impact is not surprising since the late openings for the net fishery generally occur in the vicinity of the Big Qualicum River. The relatively low impact of the inside net fishery in 1988 was mainly due to the shorter season (Table 7.1), which occurred because test fishing results indicated weak runs of Big Qualicum River coho (Wilf Luedke, pers. comm.). By and large, patterns in exploitation rates observed

in the absence of net catches (Fig. 7.3) were similar to those described previously (Fig. 7.2), which indicates that inside net fisheries did not have a pronounced influence upon relative trends in exploitation rates.

Negative relationships were found between mean adult size (in escapements) and the proportion of the total catch in the inside net fishery during each year. The relationships were strongest in years of substantial inside net catches (1986 and 1987), but linear regressions showed that the relationship was only significant for 1987 ( $F = 4.83$ ,  $P = 0.04$ ). This observation can be interpreted in different ways. Small adult size may reflect a tendency to rear predominantly in the Strait of Georgia, which may make a stock more susceptible to being caught in net fisheries. An alternative hypothesis is that the net fisheries tend to select large coho because smaller ones can escape through the mesh openings (a gear selectivity hypothesis). Support for the first hypothesis would be obtained if a negative relationship was also found between adult size and the proportion of the total catch in the inside sport and troll fisheries. No relationship between these variables was found to be statistically significant in any year, so the latter hypothesis was considered to be the most credible.

No stock or stock type was found to be subject to consistently lower or higher exploitation rates in all years. Exploitation rates of colonization and production stocks from the same river (Quinsam and Puntledge) never differed by more than 5%, and were usually within 2% of one another. However, exploitation rates on wild fish from the Quinsam River were always higher (7-11%) than for production and colonization fish from this river. Cumulative distributions of relative catches by fishery were generated to show where major discrepancies in fishery contribution exist between stocks with different exploitation rates (Fig. 7.5 to 7.7). The main difference between wild and production fish concerned the extent of the contribution to the GS.SP-n and WVI-n fisheries, with wild fish always contributing more to both fisheries than production groups. The difference in exploitation could not be attributed solely to any single factor characterizing the corresponding smolt populations, since wild and production smolts differed in migration patterns (Fig. 3.9), smolt size (Fig. 3.10 and 3.11), and exposure to hatchery conditioning.

Exploitation rates on the three stocks released from the Rosewall Creek hatchery never differed by more than 4%, and none of the stocks was characterized by a consistently larger or smaller rate of exploitation. Such results suggests that the exploitation rates of the fish produced at a hatchery can be similar even if brood stock is obtained from a variety of streams located within 45 km from the rearing and release location. Exploitation rates on all three stocks released from the Rosewall Creek facility were always between 5% and 35% higher than those of their natural counterparts. Such results indicate that

brood stock reared and released from a hatchery located on a non-parental stream can exhibit exploitation rates which differ substantially from those of the donor stream stock, even if efforts are made to release the fish at similar times and sizes as their natural counterparts.

Examination of the variation in exploitation patterns across stocks revealed some geographic trends that appeared to be consistent from year to year. Low exploitation rates for northern stocks increased progressively to reach their maximum levels at Rosewall Creek, then decreased rapidly up to the Big Qualicum - Little Qualicum area, and increased again as one proceeded southward from the Little Qualicum region (Fig. 7.3). The pronounced reduction in exploitation rate south of Rosewall Creek was not associated with a particular stock since exploitation rates on Big Qualicum River fish were not lower than those on Little Qualicum River fish in all years. The main factor accounting for the low exploitation rate on Big Qualicum River fish in 1987 was relatively low contribution to fisheries in the central coast and adjacent areas (Fig. 7.6). This situation reversed in 1988, as the contribution of this stock to all fisheries conformed to the general pattern, and the exploitation rate was near average. The fishery contribution patterns of French Creek and Little Qualicum River stocks in 1988 were similar to that of the Big Qualicum River stock in 1987 in terms of the relatively low contributions to the central coast, northern Vancouver Island and Johnstone Strait net fisheries. This also caused their exploitation rates to drop to the lowest levels as it did for the Big Qualicum River stock in 1987. Thus, the relatively low exploitation rates of stocks located within the area bounded by the Little Qualicum River and Rosewall Creek was caused mainly by their failure to contribute to the fisheries north of the Strait of Georgia.

Failure to contribute to the northern fisheries could not be attributed to any single biological trait monitored during this study, including time and size at release, smolt condition factor, adult size, and run timing. However, the above comparisons suggest that for stocks within the study area, an apparent relationship exists between exploitation rates and the number of fisheries contributed to. As a measure of the extent of stock contribution to all fisheries, the Shannon-Weaver diversity index (Zar 1984) was used:

$$(Eq. 7.1) \quad D = - \sum_{f=1}^k p_f \cdot \log(p_f)$$

when:

D = Diversity index ranging from 0 to 1.0;

k = Total number of fisheries (10 in this case);

p = Proportion of the total catch of adults in a given fishery (f).

Estimates of  $D$  were generated for each year/stock combination. Linear regressions of diversity indices against the number of tags released, and against the number of tags recovered, revealed no significant relationship in any year, suggesting that the catch distribution patterns observed were not a function of the number of tags applied and recovered. Linear regressions revealed no significant relationship between the diversity index and exploitation rates in any year, and no stock exhibited consistently higher or lower indices of diversity across all years. However, fish released from Rosewall Creek, which were subject to the highest average exploitation rates, exhibited consistently higher diversity indices than stocks from other streams ( $0.89 < \text{Mean } D\text{'s} > 0.91$ ). As noted previously, fish released from this hatchery also strayed more than other stocks. Such facts are indicative of the greater tendency of these fish to explore new marine and fresh water environments. Interestingly, fish from the Millstone River and Trent River exhibited the next highest diversity indices ( $0.85 < \text{Mean } D\text{'s} > 0.91$ ). As noted in Sections 5.2 and 5.3, fish from these three streams were exposed to foreign water sources during the rearing stages. Such facts could be considered as evidence in support of the hypothesis that hatchery practices can have a large influence on the subsequent behavior and distribution of the fish released.

Linear regressions revealed no significant relationship between the diversity index and the latitudinal location of a stream in 1987 and 1988, but one was found 1986. Within each year, there was a trend for stocks with later median dates of smolt migration to have lower diversity indices, but linear regressions failed to reveal significant relationships between these variables. Still, this last observation agrees with the results obtained by Mathews and Ishida (1989), who observed a geographical compression of the catch distribution with later release date among coho stocks adjacent to the Columbia River.

#### Effects of various factors on exploitation rates

The lack of significant relationships observed among the previous comparisons is not surprising since several factors can potentially have a simultaneous influence upon exploitation patterns. An assessment of the combined influence of various biological and physical factors on exploitation rates was made by means of linear logistic models. The environmental and biological factors tested were:

- $x_1$ . The return year for age 3+ adults;
- $x_2$ . The genetic make-up of each stock. Eight genetic categories were used for classification; Quinsam (1), Black Cr. (2), Puntledge (3), Trent (4), B. Qualicum (5), L. Qualicum (6), French Cr. (7),

Millstone (8). Wild, colonization and production fish released within the same system were considered to be members of the same category. Rosewall Creek releases from different brood stocks were categorized according to the genetic make-up of the parents (2, 4, and 6).

- x3. The experimental treatment (i.e. level of hatchery conditioning). The number of months each stock was reared in a hatchery environment was used as a measure of conditioning;
- x4. Smolt weight at release (in g);
- x5. The median date of smolt migration (calendar day);
- x6. The smolt migration period (Tables 3.8 to 3.10);
- x7. The latitude of the release location, expressed as a distance (in km) from Nanaimo;
- x8. The starting date of the run, expressed as a calendar day (Table 4.16);
- x9. The duration of the run, in weeks (Table 4.16);
- x10. The mean size of adults (Post-orbital length in cm) in the spawning population (Fig. 4.11).

For the present analysis, the ratio of catch to total return was used as a measure of exploitation rate, and was considered as the response variable. Before attempting to determine which factors influence exploitation rates, a correlation matrix of the ten variables was constructed to determine if any of the variables were correlated. Substantial correlations ( $r^2 > 0.5$ ) were found between  $x_3$  and  $x_6$ , as well as between  $x_8$  and  $x_9$ . Therefore variables  $x_6$  and  $x_9$  were omitted from the set of variables tested, which left a total of 14 potential predictor variables in the model containing only main effects. Given the number of observations (35), it is impossible to assess the effects of all first order interactions simultaneously since the total number of predictor variables would exceed the number of observations. Therefore, the number of first order interactions tested was limited to those combinations of factors which were hypothesized to have an influence upon the exploitation rate through some plausible mechanism. The following hypotheses were proposed:

- a. Adult exploitation is influenced by the distribution of adults, which is a function of the combination of time and size at release ( $x_4 * x_5$ );
- b. Delays in time of ocean entry may have a more pronounced impact on the exploitation of certain populations than others ( $x_2 * x_5$ );
- c. The effects of smolt size or migration time may be more pronounced at certain locations than others ( $x_7 * x_4$ ,  $x_7 * x_5$ );
- d. Delays in run timing affect the exploitation rates, but the effects may be more pronounced at some locations than at others ( $x_7 * x_8$ );
- e. Certain combinations of hatchery treatment and size at release may have unusually large influence upon the subsequent exploitation rates of adults. For instance, large smolts released from a hatchery

may exhibit a totally different behavior than small smolts, either by not interacting with wild smolts from the same river or by returning as jacks and failing to contribute to the distant fisheries ( $x_3 \times x_4$ ).

It is acknowledged that many other interactions could potentially influence exploitation rates through different mechanisms. The possibility also exists that the hypothesized interaction effects are found to be significant, but the mechanism by which exploitation rates are affected differs from the one proposed above. Still the set of interactions tested encompasses the mechanisms that are often proposed by investigators attempting to account for trends in exploitation rates.

Main and interaction effects were tested simultaneously using the step-wise procedure described in Section 5.3. The results obtained indicate that not all factors and interactions were helpful in describing exploitation rates (Table 7.3). The fit was improved mainly by including adult size, adult run timing, treatment and genetic make-up, as well as by the interaction of latitude and smolt size. Further improvements in fit associated with models more complex than models 9 to 11 were considered negligible judging by the trajectory of the reduction in deviance. Model 9 was judged to be sufficiently complete for the present purposes, but since model 11 provided a better fit to the data for the same degrees of freedom, it was selected as the most suitable model. The predicted values were highly correlated with the observations ( $r^2 = 0.94$ ), and no pattern was detected in the residuals. Sensitivity analysis performed on this model indicated that the performance of the model was mainly influenced by the inclusion of genetic factors, run timing, stream latitude, and the interaction between genetic factors and smolt migration date. The inclusion of stream latitude and its interaction with run timing had a larger influence than the inclusion of year and smolt weight. By contrast, treatment, adult size, smolt weight and return year tended to have a relatively small influence on the performance of the model.

An assessment was made of the direction and relative magnitude of changes in exploitation rate induced by variation in the levels of each factor. Estimates of the regression coefficients generated by the BMDP program (Table 7.5) were incorporated into the logistic model, which was then used to estimate the exploitation rates associated with particular combinations of time and size at release for Big Qualicum River and Little Qualicum River stocks in 1987 (Fig. 7.8). For both stocks, increasing the release date induced an increase in exploitation rates. The increase was much more pronounced for the Little Qualicum River stock than the Big Qualicum River for comparable increases in time (genetic\*date interaction effect). A reduction in size at release also translated into comparable increases in exploitation rates for both stocks. Such results indicate that neighboring stocks of wild and hatchery origin can respond in a similar fashion to variation in size at release, but the magnitude of the response may differ from stream to stream. These results also suggest that the relatively low exploitation rates of the Big

Qualicum River stock observed during 1987 could very well have been induced by hatchery practices. Smolts were released from the Big Qualicum River hatchery relatively early during the 1987 season, and entered the ocean earlier than smolts from adjacent streams (Fig. 3.9). If these smolts had been released 12 d later at a slightly smaller size (15 g), then the exploitation rate would have been 81%, which is close to the average exploitation rate for that year.

Effects of variation in treatment and run timing were also assessed using the Big Qualicum River stock. Reducing the duration of hatchery exposure from 18 months to 6 months (analogous to treating these groups as colonization releases) would have reduced the predicted exploitation rate from 73% to 68% for similar time and size at release. However, headwater rearing environments are usually less conducive to growth than hatchery rearing ponds, so colonization smolts tend to migrate to sea at a smaller size and at later date than production smolts from the same system. Thus, the overall effects of changes in hatchery exposure could be more realistically assessed by also incorporating the associated changes in time and size at release. Using a hatchery rearing period of 6 months, a median smolt migration date of 140 ( $\approx$  1 week later than the actual date in 1987), and a release size of 15 g, the predicted exploitation rate would have been only slightly lower than the rate observed in 1987. This example shows how the influence of hatchery conditioning *per se* is counteracted by the influence of associated changes in time and size at release, and highlights the difficulty of detecting the effects of different levels of exposure to hatchery conditioning.

Delays in the upstream migration of adults (run timing) resulted in higher exploitation rates, with a 10 d delay in run timing causing the predicted exploitation rate to increase from 73% to 83%. The mechanism by which run timing influences exploitation rates is unclear. Delays in the first flooding period may induce fish holding near the stream mouth to stray, increasing the likelihood of not being accounted for during escapement surveys. Fish holding near the stream mouth may be subject to high mortality rates due to physiological stress, and/or be subject to high levels of predation by seals and otters which aggregate near the stream mouth during the escapement period. Both of these processes would result in apparently higher exploitation rates. Alternatively, fish that enter the stream early in the season may be less susceptible to capture in the inside net fisheries and by salt-water anglers fishing near the stream mouths. In view of the limited data available, no attempt was made at identifying the exact nature of the mechanism(s) accounting for the run timing effects. However, it should be noted that during 1987, the run at Big Qualicum River began five days earlier than at Little Qualicum River, and occurred earlier than the runs in all streams south of Courtenay. Presumably, this fact also contributed to the lower exploitation rate of the Big Qualicum River stock that year.

By contrast to 1987, the exploitation rate on the Big Qualicum River stock in 1988 far exceeded that of the Little Qualicum River stock (81% vs. 61%). In terms of time and size at release, adults returning to the Big Qualicum River had migrated to sea 9 d earlier, and were approximately 6 g heavier than their Little Qualicum River counterparts. The response surfaces of the 1988 exploitation rates in relation to time and size at release were generated for both stocks (Fig. 7.9). According to those surfaces, a reduction in smolt weight of 6 g would have caused the exploitation rate on Big Qualicum River stock to increase from 81% to 85%, while a 9 d delay in time of release would only have increased the exploitation rate from 81% to 82.5%. Thus, differences in time and size at release alone do not account for the major discrepancies in exploitation rates between these stocks. In terms of run timing, the Big Qualicum River run started about two weeks later than at Little Qualicum River. A two week delay in run timing would have caused the exploitation rate to decrease from 81% to 66%, which accounts to a large extent for the differences in exploitation rates between these stocks. Thus, it is hypothesized that the differences in exploitation rates between these stocks are to a large extent due to the differences in run timing. It should be noted that the Little Qualicum River stock exhibited the earliest run in 1988, and also had an exploitation rate lower than all other stocks monitored.

Adults from French Creek were also subject to lower than average exploitation rates in 1988. This stock was characterized by an average run timing and smolt migration period, and a smaller than average smolt size. In light of the results presented above, this combination of attributes should have translated into an exploitation rate equal to or greater than average. For this stock, the exploitation rate was predicted to decrease slightly when the migration date was delayed, which is the opposite trend to that observed for Big Qualicum River and Little Qualicum River stocks. This indicates that populations from various streams do not necessarily respond in similar fashions to changes in time of release. Higher exploitation rates could have resulted from delays in run timing, or a reduction in the size at release. As noted in Section 2.1, the smolts leaving this stream had been reared in a hatchery for a few months, and had their diet supplemented during the winter. Presumably, in the absence of such enhancement, smolts would leave this stream at a much smaller size, which would be predicted to cause an increase in the rate of exploitation.

The nature of the factor(s) accounting for the differences in exploitation rate between the Rosewall Creek releases and their natural counterparts was assessed with the model. This was accomplished by first predicting the exploitation rates that would have been obtained if the smolts had been released from the hatchery at the same time and size as their natural counterpart. The predicted exploitation rates of Little Qualicum River fish returning to Rosewall Creek in 1987 did not change by adjusting release size, but increased from 94% to 96.7% when release date was adjusted. Having

accounted for the effects of time and size at release, the remaining discrepancies in exploitation rates between the pairs must be attributed to differences in run timing, rearing conditions, and release locations. Adjusting for run timing differences had a large effect upon the predicted exploitation of Little Qualicum River fish returning to Rosewall Creek, reducing it from 96.7% to 91.5%. Assuming this group had not been exposed to hatchery conditioning further reduced the predicted exploitation rates to 87.7%. The remaining difference in exploitation rates between this group and its natural counterpart was reduced to about 3.7% (84% for L. Qualicum R. wild, vs. 87.7%), which can be attributed to the effect of release location. These results suggest that the differences in exploitation rates between the natural and hatchery reared fish of Little Qualicum River brood were induced mainly by differences in run timing, release location and hatchery conditioning, under similar conditions of time and size at release.

The influence of the genetic differences among the Rosewall Creek releases were also assessed with the model. The exploitation rates on the three brood stocks released each year were predicted after standardizing the smolt weights across all stocks within each year. The predicted exploitation rates for the 1986 and 1987 releases were 94.2% and 89.4% for Black Creek broods, 90.8% and 89.1% for Trent River broods, and 94.8% and 90.9% for Little Qualicum River broods. Thus, adults from Little Qualicum origin would have been exploited at a consistently higher rate, while those of Trent River origin would have been subject to the lowest exploitation rate. The magnitude of the predicted differences are not sufficiently large to be considered as convincing evidence of differential vulnerability among these three groups.

#### Similarity in stock exploitation rates within each assemblage

Statistical comparisons of exploitation rates among stocks are hampered by the lack of replicates for all stocks within the same year. Theoretical estimates of the variance associated with tag recoveries have been proposed for some commercial fisheries in Alaska (Clark and Bernard 1987), but there is no consensus on the applicability of such estimates to commercial and sport fisheries in British Columbia. As a result, there is still considerable uncertainty about the statistical distribution of tag recovery (and escapement) statistics. In view of these facts, exploitation rates were contrasted statistically by means of non-parametric tests. As recommended by Zar (1984), testing of the differences among proportions (i.e. exploitation rates) is best done by expressing the proportions as ratios, which are then subject to contingency table analysis. Therefore, exploitation rates for each stock were expressed as catch-to-escapement ratios. The contingency tables generated were then analyzed by means of log-likelihood ratio tests (or G tests), which are preferred over chi-square tests (Zar 1984).

For 1986, no significant differences in catch-to-escapement ratios were detected between the Quinsam River and Black Creek stocks, as well as between colonization and production stocks of the Puntledge River. For 1987, no significant differences were detected between the stocks from Black Creek, Puntledge River and Trent River. During 1988, no significant differences were detected between Puntledge River and Trent River stocks. However, significant differences were found in all other comparisons between stocks from adjacent streams. Significant differences were also detected each year among the different groups released from the Rosewall Creek hatchery. In all cases where significant differences in catch-to-escapement ratios were detected, the exploitation rates of the stocks contrasted differed by more than 3%. Stocks from neighboring streams were found to differ in exploitation rates by as much as 20%, and differences in exploitation rate between stocks from adjacent streams largely exceeded the differences between stocks within the same stream (max. 5%).

The above results indicate that stocks from the same or adjacent streams can have different catch distribution patterns yet still exhibit no differences in their overall rate of exploitation. This could happen when fishing pressures are similar in both areas occupied by these stocks, or because of a combination of chance events, such as when the inside net fishery intercepts particular stocks which had been previously subject to lower exploitation rates. By contrast, the results also show that stocks which exhibit no significant differences in catch distributions (members of the same assemblage) can be subject to significantly different exploitation rates. In fact, significant differences in exploitation rates were detected each year among stocks within each of the two assemblages identified in Section 7.1. Within the northern assemblage of stocks, exploitation rates differed by as much as 7% in 1986, and 14% in 1987 and 1988. Within the second assemblage, exploitation rates differed by as much as 13% in 1986, 23% in 1987, and 30% in 1988. Such results clearly demonstrate that the exploitation rate on a given stock may differ considerably from that of a neighboring stock. Therefore, the null hypothesis (no differences between the stocks in their susceptibility to exploitation) is rejected, since the two necessary conditions (same catch distribution and same exploitation rates) are not met in most cases. Even within assemblages of stocks with similar catch distribution patterns, the second condition was not met in the vast majority of cases where stocks from adjacent streams were compared. Furthermore, none of the stocks found to be equally susceptible to exploitation in some years (Black Cr. and Quinsam R. 1986, Puntledge R. and Trent R. in 1987 and 1988), exhibited consistent similarities during the three years.

### Co-variation in exploitation rates and selection of indicator stocks

An assessment of co-variation in exploitation rates among the stocks monitored was conducted according to the procedure described in Section 5.4. The results show that three stocks, namely the Quinsam production, Puntledge colonization and Puntledge production stocks clustered in the upper left hand portion of the graph, which is indicative of a slight increase in exploitation from 1986 to 1987, followed by a reduction from 1987 to 1988 (Fig. 7.10). The remaining stocks were scattered in three quadrants, and failed to show much similarity. Stocks which were monitored for a two year period could not be incorporated directly into the plot, but could be lined up along the axis at levels corresponding to the associated change in exploitation. Their location along the single axis associated with the monitoring period serves to indicate which stocks they might have clustered with if they had been monitored for the entire period. The largest cluster would have been obtained by having all Rosewall Creek stocks and both stocks from the Quinsam River fall near the line connecting the Trent River and the Big Qualicum River. The resulting cluster would have consisted of half of the stocks monitored, encompassing a coastal region of some 81 km in length. Still, the patterns obtained show that stocks from the same stream do not always cluster together (as seen for Quinsam River stocks), and that stocks from geographically close streams do not necessarily cluster together. Wild, enhanced and colonization stocks did not show any particular tendency to cluster together. Thus, no single stock would be representative of the year to year variation in exploitation rates among the majority of stocks within the assemblage, or among stocks of the same type. Given this situation, there appears to be little incentive in identifying one stock as an indicator of year to year variation in exploitation rates among stocks in this assemblage.

The level of similarity in exploitation rate between a given stock and the average annual rate was quantified according to Eq. 7.2 to determine which stock would best represent the average exploitation rate in each of the two stock assemblages identified from catch distribution patterns:

$$(Eq. 7.2) \quad MSD_s = \frac{\sum_y (ER_{s,y} - ER_{a,y})^2}{Y}$$

when:  $MSD$  = Mean squared deviation from the average exploitation rate for the assemblage;  
 $ER$  = Exploitation rate for a stock/year (s,y) or assemblage/year (a,y) combination;  
 $Y$  = Total number of years for which exploitation rates were determined ( $\leq 3$ ).

The estimates obtained indicate that for the northern assemblage, the production stock for the Quinsam River was the most representative of the average exploitation rate (Fig. 7.11). For the southern assemblage, the enhanced stock from the Trent River was the most representative. Thus each of these stocks could be used as a yardstick for assessing the overall exploitation rate on their respective assemblages. If one considers the Trent River stock as an indicator of exploitation rates on members of the second assemblage, then it can be seen that all stocks within this assemblage were subject to exploitation rates which were within -10% to +20% points of those on the indicator stock, and in the majority of cases (14 / 19), were within  $\pm 10\%$  points of those on the indicator stock.

MSD values were recalculated for each stock on the basis that there was only one assemblage comprising all stocks monitored. Under such conditions, the production stock from the Puntledge River was found to be the best indicator of the average exploitation rate on the assemblage monitored. It should be noted that this stock was not found to be the best indicator stock for monitoring survival rates (Section 6.4). This suggests that indicator stocks determined to be suitable for one purpose, might not be suitable for another purpose. Deviations from this stock in terms of exploitation rates ranged from -22% to +12% points, which is a slightly greater than the range obtained by using the Trent River population as an indicator stock. This shows that selection of an indicator stock for the purposes of monitoring exploitation rates should ideally be based on consideration of fishery contribution patterns in order to minimize potential differences in exploitation rates between the indicator stock and other stocks.

## 8.0 DISCUSSION

Previous attempts at using one population as an indicator of the status of others in neighboring streams have been conducted mainly for the purpose of assessing general trends in either smolt production or escapement levels among populations (see for instance Holtby *et al.* 1984, and comments by Waldichuck 1984). The major deficiencies of this approach have been the lack of information on juvenile production and catch levels to account for the resulting variation in escapement levels, and the lack of information on escapement levels to account for the subsequent variation in juvenile production. Furthermore, the exclusion of hatchery stocks from the stock assemblages selected greatly limits the applicability of the results obtained. In the present study, the indicator stock concept was evaluated based on juvenile production data, catch statistics and escapement levels of tagged groups within an assemblage composed of wild, hatchery and enhanced stocks. Thus the present study was the first to use an integrated approach to evaluate the indicator stock concept, and assess the level of similarity in demographic traits within a regional stock assemblage. This approach allowed me to contribute to the biology of coho salmon, and to assess the usefulness of indicator stocks for assessment of escapement levels, smolt-to-adult survival, and exploitation rates of geographically close coho populations. In the present section, the major findings of this study are reviewed, their management implications are discussed, unresolved issues are addressed and recommendations are made for future research.

### 8.1 Overview of major findings

Information was obtained on the migration patterns and smolt productivity levels of populations subject to various levels of enhancement activity. Natural streams within the study area were found to differ considerably in terms of their smolt output levels despite attempts to stabilize the smolt production by means of colonization releases. The stream subject to the lowest degree of enhancement (Black Creek) exhibited smolt productivity levels (output per km) which exceeded the highest levels reported by other investigators for streams of comparable size. Populations subject to moderate levels of enhancement (French Creek and Trent River) were shown to have smolt sub-populations of wild and colonization origin which exhibited significant differences in body size and migration patterns. Streams with large public hatcheries were also shown to have distinct smolt sub-populations characterized by different body size and migration patterns. As a result, considerable variability in migration patterns and smolt size was detected among stocks and seasons. The most pronounced difference between the

various types of stocks monitored concerned the consistently shorter migration period of production releases, presumably induced by hatchery practices.

The difficulties associated with escapement enumeration under natural conditions were described. Recently designed counting fences were used in an attempt to overcome such difficulties and enumerate a larger portion of the run. A new mark-recapture model was also developed specifically for estimating escapement levels of salmon populations subject to immigration, natural mortality and irregular sampling. A characteristic feature of this model lies in its underlying simulation framework and unconventional types of data (fence counts, stream residency, visual survey indices, relative densities) to assess the plausibility of potential events. Although the model has not been rigorously tested yet, the available evidence suggests that the model is more accurate and more appropriate than other conventional models for escapement estimation purposes. The escapement enumeration method combined with the estimation procedures used served to provide escapement estimates as reliable as those from streams with permanent enumeration facilities, as well as detailed information on the run characteristics.

Considerable variation was observed in the duration and timing of various runs within the study area. Runs to large rivers with permanent enhancement facilities and flow controls always started earlier than those at other locations, but year to year variation in adult escapements in both types of systems was comparable and not synchronized. Run timing and run duration were similar among natural streams lacking flow controls, and among larger streams with flow controls. Flow controls thus had a pronounced influence on the adult upstream migration, which in turn had an effect on exploitation rate estimates. Ideally, a distinction should be made between both types of systems in future escapement indexing programs.

Adult escapements to the Trent River remained relatively stable from year to year, partly as a result of the substantial and highly variable contribution of strays to this stream. This finding is particularly important as it stresses the need to assess the magnitude and direction of straying among streams selected for an escapement indexing program. The straying rate estimates presented here stemmed from the most comprehensible and thorough assessment of straying rates so far conducted on coho salmon, and provide the first set of empirical estimates of straying among coastal stocks in British Columbia. Relatively low levels of adult straying were detected in the majority of stocks surveyed during this study, but in some instances, as much as 50% of the potential adult spawning population strayed to other streams. This finding highlights the need to account for levels of straying to minimize the bias of exploitation estimates. Substantial straying should be suspected in all cases where extensive

human intervention impacts on natural populations, and particularly in cases where enhancement measures include flow controls, selective breeding, and exposure of fry to various water sources.

A comparison of catch distribution patterns revealed that stock contribution to various fisheries was influenced by the geographic location of the stream of origin. Stock assemblages defined on the basis of similarity in fishery contributions covered coastal regions of up to 80 km in length. However, even within this region, stocks from the same stream or from adjacent streams were found to occasionally exhibit substantial differences in catch distribution patterns, which shows that there are exceptions to the above principle.

Estimates of survival rates were highly variable but were not found to be strongly affected by the geographical location of the stream. No stock or stock type exhibited consistently higher survival, but the production fish from the Big Qualicum River hatchery exhibited consistently lower survival than other stocks. The latter observation is important since it highlights the fact that hatchery fish may exhibit abnormal trends in survival, and are not necessarily representative of average survival rates.

Exploitation rates were not consistently higher or lower for any particular stock or stock type, but exploitation rates on these stocks averaged about 80%, and in some instances reached up to 96%. Based on the stock-recruitment data, the maximum sustainable exploitation rate on natural coho stocks in Oregon was estimated to be 73% (ODFW 1981). Wong (1982) analyzed catch and escapement statistics obtained during the 1950-1975 period for stock assemblages from several coastal regions in British Columbia, and estimated that the optimum harvest rate on natural coho stocks from the Strait of Georgia was approximately 71%. Based on these results, the 1986-1988 exploitation rates are considered to be relatively high for natural stocks. If sustained on a long term basis, such high rates of exploitation would reduce the abundance of natural stocks to levels lower than required for maximum yield, and perhaps to levels which are insufficient for maintaining genetic viability.

Previous studies aimed at identifying the effects of environmental and/or biological factors upon ocean survival of coho salmon have focused on the effects of one factor, and occasionally on the effects of two factors. In the present study, the relative influence of several factors operating simultaneously was assessed by means of recently developed quantitative techniques. Genetic factors were found to have some influence on relative escapements, survival rates, and exploitation rates. The overwhelming influence of genetic factors is not unexpected. In the process of adapting themselves to various local environments, populations will exhibit some dissimilarity in life history traits. Some traits are inherited and persist under hatchery rearing or even geographic shifts in rearing location. When comparisons

involve a large number of stocks of various origin (as in the present case), the peculiar attributes of some stocks will emerge, revealing genetically distinct traits.

Various environmental factors were shown to have some influence on specific traits as well. Smolt migration time, smolt weight, and seasons had a large influence on survival rates. Flow controls, age at return, and exposure to non-natal water sources had a large influence on straying rates. Run timing, stream latitude and hatchery conditioning were found to have a large influence on exploitation rates. While hatchery conditioning *per se* was not found to be a major determinant of straying and survival rates, all of these traits were affected to some extent either directly or indirectly by hatchery practices and the associated flow controls. This shows that establishing a permanent enhancement facility on a stream can create a new environment which may also create distinct selective pressures. As was shown in the present study, production and wild stocks from the same stream can end up exhibiting considerable phenotypic differences.

Attempts to quantify the level of co-variation in survival rates, exploitation rates and capacity to contribute to adult spawning populations were hampered by the unavailability of estimates for some stock/year combinations (non-orthogonal data set), and short time series of data. Longer time series would obviously be helpful in determining if the stock composition of the clusters identified will remain unchanged. Still, the available data showed no indication of a high level of similarity among all stocks, or among stock types, or among neighboring stocks with regards to the above attributes. Given the nature of the major factors influencing exploitation and survival rates, the relatively low level of co-variation could be anticipated. Estimates of exploitation rates for instance were shown to be influenced by hatchery practices and run timing. Significant differences in run timing were detected in some years even among streams lacking flow control because their stream flows responded differently to weather conditions early in the season. Hatchery practices were not uniform across all hatchery systems, and probably never will be, owing to the effects of disease outbreaks, broodstock availability, selection practices and logistic problems. Thus, one would expect some unsynchronized year to year variation in run timing, time and size at release, and hatchery conditioning. Added to this are the effects of inside net fisheries which appear to have variable impacts upon specific stocks. The end result is a situation characterized by much stochasticity in terms of escapement patterns, survival rates, and exploitation rates, brought about by particular combinations of factors and events. Thus, it is hypothesized that longer time series of data would not reveal a high level of co-variation in survival and exploitation rates among stocks, at least within the study area selected.

In the absence of substantial co-variation among stocks within the assemblage studied, there appeared to be no justification for selecting one stock as an indicator of the year to year variation among stocks of a particular type. However, certain stocks could still be used as indicators of the average exploitation rate or survival rate for the assemblage. Such stocks could thus be used as a yardstick to provide information on the general status of other neighboring stocks for certain traits. It was shown that the stock identified as the best indicator for one trait was not necessarily the best indicator of other traits.

## 8.2 Management implications

As noted in the introduction, monitoring all streams used by coho on a continuing basis to assess the status of the resource is an impractical task. As a result, current assessments are largely based upon trends observed in 'index streams'. Such streams are usually large systems where escapement enumeration and coded-wire tagging has been conducted for several years in a more or less consistent manner. Hatchery and wild coho populations in neighboring streams are often assumed to exhibit similar exploitation patterns and escapement trends as observed for index stream populations. A recent assessment of the current status of coho salmon stocks in southern British Columbia conducted by DFO biologists indicated that escapements to index streams (natural streams in this case) have declined by 60 to 95 percent throughout the Strait of Georgia since the 1970s (Anon. 1990). Catches of wild coho salmon in major fisheries have also declined considerably during this period. Excessive exploitation rates are believed to be largely responsible for the reduction in catches and escapements of wild coho stocks. Exploitation rates on stocks from selected Strait of Georgia hatcheries were estimated to average approximately 80% during the 1985-1988 period, and were considered to be about 10-15 percentage points higher than necessary for maintaining the long term catch from wild stocks. In order to preserve natural stocks, the current management plan calls for a reduction in fishing pressures (Anon. 1990).

In view of the above context, the major findings of this study have important management implications. The lack of substantial co-variation among stocks in terms of escapement trends, survival rates, and exploitation rates indicates that stock assemblages do not show strongly correlated responses to changes in marine conditions and fishing effort, particularly if such assemblages cover a large area and include stocks of various origin (production, wild, colonization, etc.). Since particular stock types also failed to show strongly correlated responses, there appears to be little need to consider hatchery or non-hatchery stocks as separate entities, at least from an indexing point of view. However, it can be safely assumed that over a relatively small coastal area (< 100 km), in the absence of drastic changes in hatchery rearing methods, hatchery and non-hatchery stocks will respond similarly to substantial

reductions in fishing pressures. Monitoring the general response of stock assemblages could be accomplished through the use of indicator stocks. However, based on the results obtained in the present study, it appears that an indicator stock will only provide a crude estimate of the general status of nearby stocks in terms of escapement trends, survival rates, and exploitation rates.

Despite such limitations, indicator stocks are considered to be useful stock-assessment tools, and can definitely provide information for management purposes. Let's assume for instance that a fishery manager wishes to regulate coho harvest rates to maximize the catch and minimize the likelihood of eliminating natural stocks in the area selected for this study. To reach this goal, the manager may choose to manage the fishery so that exploitation rates on natural stocks do not exceed the target rate of 65%. Let's further assume that the Puntledge River production stock is selected as the indicator. As was shown in Section 7.2, deviations from this stock in terms of exploitation rates ranged from -22 to +12 percentage points over the three year period. In the absence of drastic changes in the distribution of fishing effort in the near future, it might be assumed that such deviations are typical. Based on this knowledge, the manager could regulate fishing effort to ensure that the exploitation rate on the indicator stock does not exceed 53%, in order to ensure that the highest exploitation rate on the most vulnerable stock in the assemblage does not exceed 65%. Alternatively, the manager may wish to regulate harvest on stocks in the first and second assemblage separately, and use the Trent River and Quinsam River stocks as indicators. This approach would provide an indication of the average harvest rate on each stock assemblage, which would be most appropriate in situations where substantial changes in fishing effort are expected in northern, southern or outside waters. This may sound like an overly conservative approach, but it illustrates how the technique could be applied to meet some management goals, while reducing the monitoring costs associated with coded-wire tagging and escapement enumeration at all sites.

A similar approach could be used to monitor escapement trends in the region. As was shown in Section 5.4, one of several stocks could be used for this purpose. Ideally, one might wish to select a population that is not known to exhibit unusually high straying rates, or one that is characterized by occasionally large influx of strays from other streams, especially when no coded-wire tag information is available to identify members of various stocks. Under such conditions, it might seem preferable not to use Rosewall Creek fish as an indicator (which was previously identified as such based on statistical merits). Instead, it might be tempting to use a stock which is already being monitored routinely such as the Puntledge River production stock. As mentioned previously, hatchery practices can have a pronounced influence upon survival rates and adult behavior. Furthermore, escapement patterns to large rivers with flow controls are not always representative of those in small creeks. Therefore, if hatchery

stocks are to be used as an indicators, one should select stocks from hatcheries with minimum flow controls, relatively constant hatchery practices, and a low incidence of disease outbreaks.

Assuming that the Puntledge River production stock is selected, deviations in terms of relative escapements (fraction of total release that survives to escape the fishery, 1986-1988 average = 2%) calculated in Section 5.4 ranged from -2.1 to +3.6 percentage points over the three year period. Stock-assessment biologists might consider that escapements for wild stocks should be increased to say 3% of total smolt output, and fishing effort could be regulated to ensure that the relative escapement of the indicator stock is approximately 5%.

### 8.3 Unresolved issues and recommendations for further research

As noted by Waldichuk (1984), if the indexing approach proves to be a useful management tool, then the next logical step would be to select index streams in other parts of the province, preferably in different biogeoclimatic zones. The results of the present study indicate that the level of similarity among stocks in terms of escapement trends, survival and exploitation rates is related to the composition and size of assemblages. However, little is known about level of similarity that will prevail in other regions. Further speculation can be made on the nature of the factors which affect the catch distribution patterns, and ultimately the level of similarity among stocks within a region.

Assuming that the fishery contributions patterns observed during this investigation accurately reflect the marine distribution of adults, then it would appear that stocks north and south of Courtenay follow different migration patterns. Little is known about the factors which control migration patterns of coho, but it is believed that adult distribution is linked to the juvenile migration patterns followed during early marine stages (Hartt 1980, Pearcy and Fisher 1988). Pearcy and Fisher (1988) gave evidence that juvenile coho salmon usually remain in surface waters (0-20 m), and that their distribution pattern may be influenced to a large extent by surface currents during the first few weeks of marine life. Healey (1980) analyzed the results of juvenile salmon surveys in the Strait of Georgia, and noted that coho smolts disperse rapidly throughout the Strait, and tend to remain in nearshore waters for the first few weeks. Milne (1950) was one of the first to hypothesize that some coho from tributaries to the Strait of Georgia remained in inside waters during their whole life, while the remaining fish migrated to outside waters. Healey (1980) noted that the proportion of each population remaining inside may vary from year to year, perhaps due to oceanographic conditions, and hypothesized that coho which leave the Strait to rear in the open ocean might emigrate soon after ocean entry.

There is considerable information about the physical oceanography of the Strait of Georgia, but relatively little is known about the surface currents in various parts of the Strait (Leblond 1983). Thomson (1981) provided a summary of the surface circulation inside the Strait of Georgia during spring and summer based on the results of drift bottle experiments, current meter records, and studies with drift drogues conducted since the 1920s. Thomson suggested the existence of a counterclockwise circulation pattern in the area between Comox and Discovery passage, with a southward drift on the Vancouver Island side. The central portion of the Strait (Vancouver area to Texada Island) is characterized by a moderately strong tidal stream, the influence of the Fraser River runoff, and another counterclockwise circulation pattern. This second counterclockwise gyre consists of water masses moving northward along the mainland coast which get diverted southward near the southern tip of Lasqueti Island and northern tip of Texada island. The southward portion of the current would tend to move down the Island coast from Comox to Nanaimo. Water from this second gyre eventually mixes with waters from the southern portion of the Strait (Vancouver to Juan de Fuca) just southeast of Gabriola Island. Waters in this southern section are characterized by strong tidal currents, which carry surface waters to the Pacific Ocean (Thomson 1981).

Depending on the location of a stream in relation to Courtenay, and the migration characteristics of its coho smolt population, a variable portion of the smolts leaving each stream could get entrained into the currents of different gyres. Smolts from streams located to the north of Courtenay might be transported to the north where they come in contact with the surface water outflow moving through Johnstone Strait. Smolts entering the ocean south of Courtenay might be transported along the coast of Vancouver Island to the Gulf Islands, and eventually through the Strait of Juan de Fuca. As these fish grow older, they achieve greater control over their distribution, and as shown in several tagging studies since that of Milne (1957), can disperse themselves throughout coastal waters ranging from Washington to Alaska. Godfrey *et al.* (1975) noted that coho from Washington and southern British Columbia do not generally undertake long ocean migrations, and once they have entered certain rich feeding grounds, they remain there until the onset of maturity. Once this stage is reached, a major portion of the harvest has been achieved, so fishery contributions are to a large extent determined by the distribution of the fish prior to the spawning migration.

The evidence presented above leads me to hypothesize that circulation patterns inside the Strait of Georgia have some influence on the geographic boundaries of the assemblages, and the level of similarity in fishery contribution among populations in each assemblage. Coastal oceanographic conditions are not uniform throughout the coast of B.C., but can be considered as such within smaller

regions, which could form the boundaries of assemblages. The number and types of fisheries contributed to by stocks in each of these regions would also differ to a greater or lesser extent, which would also affect the apparent level of similarity among stocks in exploitation patterns. This suggests that in other coastal regions, it may be possible to define stock assemblages with similar fishery contribution patterns mainly on the basis of geographical landmarks and the respective locations of the stocks selected.

Large public hatcheries are also less common in other coastal regions, and there are relatively fewer streams subject to small scale enhancement work on a permanent basis. The populations within such regions should therefore be more homogeneous, and might respond in a more synchronous fashion to local atmospheric conditions, marine conditions and fishing pressures. It is doubtful that stock assemblages in other coastal regions of comparable size would exhibit more dissimilarity among stocks in survival and exploitation rates than was observed during this study. It is hypothesized that the indicator stock approach could be more successfully applied in other coastal regions. Coho stocks from the central coast region, or from the northwest part of Vancouver Island could very well exhibit a substantial amount of similarity in escapement trends, survival rates and exploitation rates, and be considered as large stock assemblage. On the other hand, stocks from various tributaries of the Fraser River may exhibit peculiar trends in terms of similarity due to the geographical attributes of this system. This dictates the need for additional comparative studies in other parts of the province to determine the composition and size of stock assemblages. In the meantime, one might assume that natural stocks from streams located within a 100 km area will exhibit similar trends in abundance and productivity if no terminal fisheries target on particular stocks and if the entire area is under the influence of the same atmospheric and oceanographic conditions. In the absence of empirical assessments to determine which stock is the most reliable indicator based on statistical merits, the stock selected for indexing purposes should be from a typical stream for that area. The results obtained during this study indicate that the maximum discrepancies observed during 1986-1988 between any two stocks within such an area in terms of relative escapements, survival rates, and exploitation rates were 5.8, 22.3 and 30.0 percentage points respectively, and averaged 1.5, 6.0 and 8.0 percentage points. Presumably, the discrepancies observed between natural stocks in other areas will be lower than these figures. The findings of this study should be reassessed in light of additional data from previous studies, and data from monitoring programs currently conducted by DFO personnel. This additional knowledge would allow for a more rigorous analysis of the patterns of co-variation in survival and exploitation rates, the stability of the assemblages identified, and determine how reliance on indicator stocks data can affect the accuracy of various types of predictions.

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## APPENDIX

Description of the awareness factor simulation procedure

The Creel Survey statistics reported in Table 6.4 were used to produce the following set of baseline parameters for the model (represented by non-italic lower case abbreviations):

- the expected distribution of coho catches by strata ( $catch_{a,t}$ );
- the overall tagged proportion in the catch ( $p(\text{tagged})_{\text{catch}}$ );
- the number of interviews conducted in each stratum;
- the mean sampling rates at boat ramps;
- the proportion of the tagged heads returned.

For simulation purposes, these parameters were considered as being representative of the actual conditions in the sport fishery. Information on sampling error and random variation as reported by English *et al.* (1986) were used in conjunction with the above parameter values to generate estimates of the number of fish and tagged fish examined, the tagged proportions observed, the fraction of all anglers fishing at a given time, and the catch per unit effort for each stratum. These estimates were then used to calculate the awareness factor for each stratum. The figures and trends obtained from the model were compared to those reported from the Creel Survey program, and sampling regimes were adjusted to ensure that the model mimicked the actual patterns reported.

To best describe the components and structure of the descriptive model, reference will be made to the 1987 Creel Survey statistics (Table 6.4). The total adult coho catch for all strata was set at 641,500 coho. The catch distribution reported in the Creel Survey statistics was adjusted slightly in order to provide a more representative distribution of the actual catches by ensuring that some fish were caught in all strata. Using the 9 x 12 array of catches ( $catch_{a,t}$ ) produced from the Creel Survey, the expected catches in each area ( $a$ ) by month ( $t$ ) stratum were calculated as follows:

$$(Eq. A1) \quad catch_{a,t} = \frac{\sum_A catch_{a,t} \cdot \sum_T catch_{a,t}}{\sum_{AT} catch_{a,t}}$$

The expected catches were considered as the actual catches for the simulations. The expected number of tagged coho in each stratum was obtained by multiplying the expected catch by the tagged proportion as observed during 1987 ( $\approx 7\%$ ). This proportion was assumed to be binomially distributed about a mean of seven for every lot of 100 fish. As a result, the tagged proportions generated were fairly uniform across all strata with large catches, but varied noticeably only when catches were less than 2000 coho.

Examination of the sampling success by strata achieved during 1987 indicates that the fraction of the catch examined during the interviews was always less than 40%, with a mean of 4.5%, and a variance of 0.7%. This fraction tended to increase as the catch per stratum decreased, which was assumed to be induced by greater variation in sampling fractions and catch estimates under conditions of low catches. Since the actual sampling rates are achieved through a crude allocation of sampling effort in relation to catch levels (K. English<sup>34</sup>, pers. comm.), sampling was assumed to be a negative binomial process with a success rate of 4.5% for each lot of 4000 fish examined. This sampling regime was used to generate estimates of the number of fish observed ( $fish_o$ ), and tagged fish observed ( $tags_o$ ) in each stratum. Since it is possible to examine a certain percentage of the fish caught, and a different percentage of the tagged fish caught during each sampling event, catch samples and tagged fish samples were generated independently for each stratum. Estimates of the observed tagged proportions ( $p(tagged)_{catch}$ ) for each stratum were calculated from the ratio ( $fish_o \cdot tags_o^{-1}$ ).

Stratum catch estimates in the simulations were based on the assumption that errors in boat counts were negligible compared to those of  $cpue$  and  $p(b)$  estimates, since overflights are usually not conducted under conditions of poor visibility, and fairly accurate counts of the number of fishing boats can be obtained even under conditions of high fishing effort. Estimates of  $cpue$  and  $p(b)$  were generated on the basis of the number of interviews conducted at the boat ramps, and the underlying distribution of each variable. In the absence of data on the exact number of interviews conducted in each stratum, this figure was calculated from the ratio  $fish_o \cdot cpue^{-1}$ , with the denominator representing the rate reported in the Creel Survey, averaged across all areas for a given month. English *et al.* (1986) assumed that the distribution of  $p(b)$ 's was binomial, and observed that under such conditions, the distribution of  $cpue$ 's within a given stratum was similar to the negative binomial. Thus, these distributions were used to generate one estimate of  $cpue$  and  $p(b)$  for each interview conducted. Random variates of  $cpue$  were generated by assuming a success rate equal to the mean monthly  $cpue$  value reported from the Creel Survey. Random variates of  $p(b)$  were generated assuming mean value of 0.5 since overflight timing

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<sup>34</sup>LGL Ltd. 9768 2nd street, Sidney, B.C.

corresponds to periods when approximately 50% of the fisherman are active. Estimates of the mean *cpue* and *p(b)* for each stratum were obtained by averaging the random variates over the number of interviews conducted.

The actual boat counts obtained during the Creel Survey overflights were not provided in time for this analysis. Therefore estimates of total catch and catches of tagged fish in each stratum were obtained by reference to the actual catch and the bias associated with estimates of *p(b)* and *cpue*;

$$(Eq. A2) \quad catch = catch \cdot \frac{p(b)}{p(b)} \cdot \frac{cpue}{cpue}$$

$$(Eq. A3) \quad catch_{tag} = catch \cdot p(tagged)_{catch}$$

When the average monthly *cpue* figure in the Creel Survey statistics exceeded 1.0, both ratios *cpue/cpue* and *p(b)/p(b)* were close to 1.0, since the number of successful interviews conducted during such periods was generally large. Under such conditions, differences between the actual and estimated catches in strata with relatively large catch levels were mainly attributed to the bias associated with the estimated tagged proportions.

The number of heads submitted by fisherman during 1987 was approximately 20% of the actual catch of tagged fish in each stratum. The number of heads submitted by fisherman (*hr*) was assumed to be a negative binomial process, with a probability of success of 0.2 for every 100 tagged fish captured. The success rate was assumed to be similar across all strata, irrespective of the catch level, fishing area or month. It was also assumed that complete and accurate information was supplied with each head returned, and no errors were made during the decoding and reporting process. Thus, only random variation in fisherman behavior with respect to the proportion of heads submitted was allowed for.

Estimates of awareness factor for each stratum were based on the simulated ratio of the number of heads submitted to the simulated catch of tagged coho. The output generated from the descriptive model shows that the relationship between catch levels and sampling rates, and catch levels and awareness factors were similar to the patterns observed in the Creel Survey statistics for 1987 (Fig. 6.2, 6.1). Slightly different *af*'s for given catch levels were obtained during additional simulations using different seed numbers for random number generation, but the general trends, as well as the magnitude of the dependent variables for given catch levels, were still similar to the reported patterns. These results suggest that a combination of low tagged proportions, low sampling rates, and random variation could

account for all the variation in awareness factors observed between strata even if the level of participation to the program was high (20%), relatively uniform across all strata, and carried out without any misreporting. Thus, real variation in awareness factor due to external factors would be difficult to detect at this level of stratification, particularly between strata with low catches.

The sampling, reporting and estimation processes reported for 1985, 1986 and 1988 were also described with the same model using appropriate parameter values for *cpue*, *p(tagged)*, and sampling rates. Monte Carlo simulations were then conducted using each version of the model to determine the potential magnitude of bias associated with *af* estimates under various catch levels. All estimated parameter values were allowed to fluctuate randomly according to their hypothesized distribution patterns. After each iteration, relative error was estimated from  $(af - af) \cdot af^{-1}$ , and the average error for a given catch level was calculated after 50 iterations.

**TABLES**

Table 2.1. Smolt output by category at selected streams from 1985 to 1987. Wild smolt output figures for the Puntledge, Big Qualicum and Quinsam Rivers were obtained from hatchery records. Estimates of colonization smolt outputs at Puntledge River based on a 13% survival rate from fry to smolt stage. Tagged smolt outputs not corrected for tag loss. Millstone River output estimates obtained from Hurst and Blackman (1988), and R. E. Hurst (pers. comm.).

Stream name	Smolt outm. yr.	Prod. smolts	Number tagged	Colon. smolts	Number tagged	Wild smolts	Number tagged
Quinsam R.	85	1853852	220929	>21819	>21819	> 35000	0
"	86	1201640	77380	29167	29167	> 45000	19524
"	87	1284360	42182	>32000	0	> 50000	24505
Puntledge R.	85	733336	98197	381512	14162	< 30000	0
"	86	950865	39124	419433	13633	< 30000	0
"	87	358807	58145	118839	8088	< 30000	0
L. Qual. R.	85	0	0	0	0	> 20000	19596
"	86	0	0	0	0	>17000	16356
"	87	0	0	0	0	> 25000	20550
Rosewall Cr.	85	0	0	0	0	< 2000	0
"	86	51385	51385	0	0	< 2000	0
"	87	59068	57768	0	0	< 2000	0
B. Qual. R.	85	3522034	191620	>500,000	0	> 70000	0
"	86	2658239	152273	>500,000	0	> 70000	0
"	87	1472413	119424	>500,000	0	> 70000	0
Millstone R.	85	0	0	8000	0	>1000	0
"	86	0	0	15775	9115	0	0
"	87	0	0	13974	8158	470	0

Table 2.2. Coho escapement estimates and sample sizes at selected streams from 1985 to 1988. For systems with hatcheries, fence counts represent actual hatchery escapement count. Estimates of spawners above and below the counting fences at the Quinsam, Puntledge and Big Qualicum and Little Qualicum rivers were provided by hatchery managers. Escapement figures for the Millstone River are field survey estimates provided by R. E. Hurst, and personnel from the Nanaimo River hatchery.

Stream name	Escap. year	Jacks fence count	Jacks above fence	Jacks below fence	Total jack escap.	Total jacks exam.	Adults fence count	Adults above fence	Adults below fence	Total adult escap.	Total adults exam.
Quinsam	1985	14289	0	75	14364	13536	28662	0	750	29412	20837
	1986	18052	0	90	18142	16919	35901	0	750	36651	26980
	1987	39330	0	200	39530	37571	24012	0	2100	26112	18253
	1988	30338	0	150	30488	30252	33190	0	2000	35190	31974
Puntl. R.	1985	11637	5000	1650	18287	11611	39266	0	20000	59266	37011
	1986	2235	800	200	3235	3189	11068	2500	2500	16068	10114
	1987	1236	400	100	1736	1159	4811	1000	2000	7811	4890
	1988	4034	500	500	5034	3906	4609	500	500	5609	4654
L. Qual.	1985	270	0	0	270	240	1960	400	1200	3560	1914
	1986	87	0	0	87	87	497	170	280	947	579
	1987	439	1	0	440	440	725	210	340	1275	800
	1988	81	0	0	81	81	950	300	500	1750	801
Rosewall	1985	0	0	0	0	0	0	0	0	0	0
	1986	124	0	0	124	124	5	0	0	5	5
	1987	104	0	0	104	104	182	0	0	182	182
	1988	147	0	0	147	147	502	0	0	502	502
B. Qual.	1985	7077	0	negl.	7077	7023	22182	0	negl.	22182	19572
	1986	4006	0	negl.	4006	3449	18950	0	negl.	18950	15669
	1987	9862	0	negl.	9862	9393	7793	0	negl.	7793	6496
	1988	5572	0	negl.	5572	5210	7038	0	negl.	7038	4290
Millstone	1985	0	-	-	n/a	0	0	-	-	n/a	0
	1986	200	-	-	200	89	0	-	-	0	0
	1987	25	-	-	25	13	57	-	-	57	44
	1988	2	-	-	2	2	275	-	-	275	69

Table 2.3. Colonization fry releases at Black Creek, Trent River and French Creek, and the Millstone River, 1982-87. [\*] Approximately 7.4% of the total release (or 4636 fry) were recaptured in the field as 1.5 g fry and fin clipped. Estimates from Hurst and Blackman (1988), and R. Hurst (pers. comm.).

Release Location	Release date	Release site	Species released	Brood stock	Fry released	Fry Size (g)	Mark used
Black Cr.	Jul. 85	Cranberry	coho	Black Cr.	10249	1.7	R. ventral
"	Jul. 85	mainstem	coho	Black Cr.	10600	1.7	none
"	Jul. 86	headwaters	coho	Black Cr.	16500	4.5	R. ventral
"	Aug. 86	headwaters	coho	Black Cr.	6998	3.0	L. ventral
"	Aug. 86	headwaters	coho	Black Cr.	25284	2.6	none
Trent R.	May 82	Bloedel Cr.	coho	Puntledge	65000	<1.0	none
"	May 83	Bloedel Cr.	coho	Puntledge	90000	<1.0	none
"	Jun. 83	headwaters	coho	Puntledge	90000	<1.0	none
"	Jul. 83	mainstem	coho	Puntledge	149000	5.6	none
"	May. 84	Bloedel Cr.	coho	Big Qual.	100000	<1.0	none
"	Jun. 84	headwaters	coho	Big Qual.	80000	1.2	none
"	Jun. 85	Bloedel Cr.	coho	Trent	45000	2	none
"	Jun. 85	Bradley L.	coho	Trent	18185	2.4	L. ventral
"	Jul. 85	headwaters	coho	Trent	7500	2.4	R. ventral
"	May 86	Bloedel Cr.	coho	Trent	62645	0.8	R. ventr *
"	Jun. 86	Bradley L.	coho	Trent	34461	1.5	L. ventral
"	Jun. 87	China Cr.	coho	Trent	5300	2.3	none
"	Jun. 87	China Cr.	coho	Trent	5300	2.3	none
"	Jun. 87	headwaters	coho	Trent	5400	2.3	none
"	Jun. 87	Bradley L.	coho	Trent	30000	2.5	none
"	Jun. 87	Bloedel Cr.	coho	Trent	30000	2.5	none
French Cr.	Jun. 85	mainstem	coho	French Cr.	20000	1.5	none
"	Sep. 85	Dudley m.	coho	French Cr.	9855	5.1	L. ventral
"	Sep. 85	Dudley m.	rainbow	French Cr.	829	n/a	R. ventral
"	May. 86	Dudley m.	coho	French Cr.	10558	1	L. ventral
"	Jun. 86	mainstem	coho	French Cr.	80000	1.5	none
"	Oct. 86	Dudley m.	coho	French Cr.	1508	6	R. ventral
"	May. 87	Dudley m.	coho	French Cr.	10400	2.0	L. ventral
"	May. 87	mainstem	coho	French Cr.	8500	1.8	none
"	Jun. 87	mainstem	coho	French Cr.	44000	2.5	none
Millstone R.	May 82	mainstem	coho	Millstone R.	50000	0.7	none
"	Aug. 83	mainstem	coho	Millstone R.	34000	6.0	none
"	Jun. 84	mainstem	coho	Millstone R.	34399	1.8	none
"	Oct. 84	Brannen L.	coho	Millstone R.	9349	7	L. ventral
"	Oct. 84	Mainstem	coho	Millstone R.	11193	7	R. ventral
"	Jul. 85	mainstem	coho	Millstone R.	29447	3.6	R.v R.m
"	Jul. 85	Brannen L.	coho	Millstone R.	26221	3.6	L.v R.m
"	Sep. 85	Brannen L.	coho	Millstone R.	11300	9.9	R.v L.m
"	Sep. 85	Brannen L.	coho	Millstone R.	37106	6.7	none
"	May 86	mainstem	coho	Millstone R.	24869	0.75	L. ventral
"	Jul. 86	mainstem	coho	Millstone R.	24750	3.5	R.ventral
"	Jul. 86	mainstem	coho	Nanaimo R.	47582	3.6	adip.
"	Jul. 86	Brannen L.	coho	Nanaimo R.	48041	3.6	adip.

Table 3.1. Juvenile tagging summary statistics for Black Creek, Trent River, and French Creek, 1985-1987. %T and %S indicate the fraction of total smolt catch, and fraction of total release.

Season Operation	1985		1986		1987	
	Trapping	Tagging	Trapping	Tagging	Trapping	Tagging
<b>Black Cr.</b>						
Start Date	May 1	May 1	Apr 15	May 7	Apr 8	Apr 29
End Date	May 26	May 21	Jun 4	May 31	May 23	May 18
Peak period	May 19	May 16	May 24	May 28	May 10	May 10
Median date	May 18	May 15	May 23	May 23	May 12	May 12
Angular deviation	4 d	3 d	6 d	5 d	7 d	5 d
Total Coho	50208	% T % S	38212	% T % S	60909	% T % S
No L. ventral	0	0.0 ---	10	0.0 ---	283	0.5 4.0
No R. ventral	0	0.0 ---	1	0.0 0.0	297	0.5 1.8
No adipose	24	0.0 ---	458	1.2 ---	48	0.1 ---
Juv. rainbow trout	37	--- ---	144	--- ---	930	--- ---
Juv. cutthroat trout	60	--- ---	99	--- ---	281	--- ---
Sculpins	n/a	--- ---	858	--- ---	605	--- ---
Lampreys	5	--- ---	21	--- ---	9	--- ---
Tags released	24134		31648		35640	
% Total Tagged	48.1		82.8		58.5	
% no tag & no adip.	0.3		2.1		0.7	
Est. Undetected (%)	15.0		15.0		15.0	
<b>Trent River</b>						
Start date	May 1	May 22	Apr 25	May 7	Apr 25	May 4
End date	Jun 14	Jun 13	Jun 18	May 31	Jun 8	Jun 8
Peak period	May 25	May 30	May 20	May 20	May 10	May 10
Median date	May 29	June 1	May 26	May 26	May 18	May 19
Angular deviation	6 d	5 d	10 d	9 d	8 d	8 d
Total Coho	12952	% T % S	11388	% T % S	17635	% T % S
No L. ventral	0	0.0 ---	3526	31.0 19.4	4646	26.3 13.5
No R. ventral	0	0.0 ---	406	3.6 5.4	273	1.5 5.9
No adipose	1	0.0 ---	0	0.0 ---	46	0.3 ---
Juv. rainbow trout	397	--- ---	1274	--- ---	2004	--- ---
Juv. cutthroat trout	0	--- ---	8	--- ---	35	--- ---
Sculpins	880	--- ---	984	--- ---	675	--- ---
Lampreys	5	--- ---	8	--- ---	4	--- ---
Tags released	8975		10531		15691	
% Total Tagged	69.3		92.5		89.0	
% no tag & no adip.	0.07		0.28		0.9	
Est. Undetected (%)	25.0		25.0		20.0	
<b>French Cr.</b>						
Start date	none	none	Apr 18	May 9	Apr 11	Apr 11
End date			Jun 12	Jun 6	Jun 5	May 27
Peak period			May 27	May 27	May 15	May 15
Median date			May 26	May 26	May 18	May 16
Angular deviation			9 d	7 d	9 d	6 d
Total Coho			29155	% T % S	33917	% T % S
No L. ventral			760	2.6 7.7	754	2.2 7.1
No R. ventral			0	0.0 ---	149	0.4 9.9
No adipose			6	0.0 ---	25	0.1 ---
Juv. rainbow trout			2042	--- ---	2591	--- ---
Juv. cutthroat trout			644	--- ---	730	--- ---
Sculpins			198	--- ---	423	--- ---
Lampreys			15	--- ---	28	--- ---
Tags released			23838		24354	
% Total Tagged			81.8		71.8	
% no tag & no adip.			0.6		0.8	
Est. Undetected (%)			10		10	

Table 3.2. Weighted estimates of mean fork length (mm) and sample statistics for smolt populations at Black Creek, Trent River and French Creek, 1985 - 1987.

Survey Year	Black Creek			Trent River			French Cr.		
	1985	1986	1987	1985	1986	1987	1985	1986	1987
Sample size	115	153	257	88	1935	184	n/a	163	250
St. Mean F.L.	96.2	117.0	116.9	86.7	100.6	96.5	n/a	91.2	94.5
SE	1.48	1.85	0.91	0.90	0.32	0.82	n/a	0.73	0.75
SD	15.90	22.80	14.50	8.49	13.90	11.23	n/a	9.32	11.81
Coef. Var.	0.17	0.19	0.12	0.10	0.14	0.12	n/a	0.10	0.13
Length range	90-169	83-170	78-172	67-111	67-155	74-150	n/a	68-119	75-168

Table 3.3. Mean fork length (mm) and sample statistics for marked and unmarked smolts from the Trent River, 1985 - 1987.

Group	l.v	1985		l.v	1986		l.v	1987	
		r.v	no mark		r.v	no mark		r.v	no mark
Sample size	n/a	n/a	88	1367	365	1257	40	n/a	144
Mean F.L.	"	"	86.4	105.7	93.6	98.6	105.1	"	94.6
SE	"	"	0.90	0.37	0.41	0.31	2.34	"	0.72
SD	"	"	8.41	13.72	7.81	11.0	14.8	"	8.67
Length range	"	"	67-111	75-155	69-124	67-148	83-150	"	74-122

Table 3.4. Weight to length relationship ( $W = aL^b$ ) coefficients, and sample statistics for smolt populations at Black Creek, Trent River and French Creek, 1985 - 1987.

Survey Year	Black Creek			Trent River			French Creek		
	1985	1986	1987	1985	1986	1987	1985	1986	1987
Sample size	115	153	257	88	278	184	n/a	163	250
a	.0170	.0139	.0184	.0182	.0194	.0305	n/a	.0259	.0103
b	2.800	2.921	2.812	2.789	2.712	2.533	n/a	2.617	3.021
r <sup>2</sup>	0.921	0.967	0.944	0.883	0.911	0.889	n/a	0.913	0.955

Table 3.5. Weight to length relationship ( $W = aL^b$ ) coefficients, and sample statistics for marked and unmarked smolts at Black Creek, Trent River and French Creek, 1986 - 1987.

1986 sample	Black Creek			Trent River			French Creek		
	l.v	r.v	no mark	l.v	r.v	no mark	l.v	r.v	no mark
a	n/a	n/a	153	178	33	190	n/a	n/a	163
b	"	"	.0139	.0181	.0188	.0178	"	"	0.026
r <sup>2</sup>	"	"	2.921	2.732	2.703	2.758	"	"	2.617
1987 sample	"	"	0.967	0.937	0.916	0.914	"	"	0.913
a	n/a	n/a	257	40	n/a	143	n/a	n/a	250
b	"	"	.0184	.0306	"	.0257	"	"	.0103
r <sup>2</sup>	"	"	2.812	2.523	"	2.612	"	"	3.021
	"	"	0.944	0.925	"	0.925	"	"	0.955

Table 3.6. Body size and sample statistics for age 1+ and 2+ smolts from Black Creek (above), and French Creek (below). Mean lengths (mm) and weights (g) for age 1+ smolts are weighted estimates. Estimates of mean length and weight for age 2+ smolts were not weighted due to insufficient sample sizes.

Survey Year Smolt Age	1985		1986		1987	
	1+	2+	1+	2+	1+	2+
Mean F.L.	92.9	155.0	116.9	143.0	115.5	121.6
SD	13.7	-	22.0	14.2	14.9	12.8
# Obs.	30	1	140	13	196	10
Range	78-172	-	83-170	115-158	93-169	106-151
Mean Wt.	9.1	37.5	18.8	33.2	18.4	21.2
SD	5.1	-	10.6	8.7	6.7	6.4
# Obs.	30	1	140	13	196	10
Range	4.7-58.7	-	5.6-49.6	17.5-43.3	9.0-54.1	14.0 - 36.7
Mean F.L.	n/a	n/a	91.2	101.8	94.1	125.7
SD	"	"	9.3	-	11.6	30.21
# Obs.	"	"	162	1	231	6
Range	"	"	68 - 119	-	75 - 140	88-168
Mean Wt.	n/a	n/a	8.83	10.7	9.25	26.6
SD	"	"	2.72	-	3.56	18.95
# Obs.	"	"	162	1	231	6
Range	"	"	4.3 - 20.5	-	4.3 - 26.5	7.5 - 54.0

Table 3.7. Estimates of smolt output for streams lacking major hatchery production facilities. % Tagged represents the tagged proportions among jacks and adults escaping to their stream of origin, adjusted for differential mortality of tagged and untagged fish (Section 5.2). Weighted average obtained by pooling sample statistics for adult and jack siblings. Figures for 1978, 1979, and 1988 are from Clark and Irvine (1989), and Fielden *et al.* (1989). Tagged proportions among adults in 1989 provided by R. Bocking (pers. comm.). \* Estimated from tagged proportions at return based on on fin clipped fractions in escapement (Table 6.3).

Stream name	Release year	% Tagged at release (smolts)	% Tagged in escap. (jacks)	% Tagged in escap. (adults)	% Tagged in escap. (w. average)	Observed output	Estimated output (max.)	%smolt output counted
Black Cr.	1978	<64.5	50.7	69.9	55.0	46,405	55,247	84.0
	1979	<84.6	91.4	81.6	85.9	46,566	46,580	99.9
	1985	<48.1	30.0	20.4	21.0	50,208	115,436	43.5
	1986	<82.8	66.1	84.6	77.5	38,212	41,263	92.6
	1987	<58.5	67.7	58.8	59.1	60,909	60,913	99.9
	1988	<98.0	80.0	94.9	93.1	76,404	80,816	94.5
Trent R.	1985	<69.3	-	11.6 *	11.6	12,952	77,714	16.7
	1986	<92.5	51.0	36.2	36.6	11,388	28,894	39.4
	1987	<89.0	55.3	53.0	53.2	17,635	29,805	59.2
	1988	<98.4	35.9	57.6	48.1	18,704	38,411	48.7
French Cr.	1985	-	-	-	-	-	-	-
	1986	<81.8	23.8	48.2	42.9	29,155	55,961	52.1
	1987	<71.8	99.0	69.1	73.0	33,917	33,920	100.0
	1988	<96.2	100.0	100.0	100.0	27,568	27,568	100.0
L. Qual. R.	1985	<98.2	24.7	12.4	15.1	19,596	126,888	15.4
	1986	<96.3	71.1	37.2	40.6	16,987	40,245	42.2
	1987	<88.0	70.9	45.4	50.8	23,354	40,485	57.7
	1988	-	-	-	-	-	-	-
Millst. R.	1985	-	-	-	-	-	-	-
	1986	<57.8	4.2	56.4	15.7	15,775	57,875	27.3
	1987	<56.5	57.1	18.6	21.8	11,444	37,378	30.6
	1988	-	n/a	n/a	n/a	-	-	-

Table 3.8. CWT groups migrating from each stream during 1985. Time held is the period retained (d) for CWT retention tests. Ocean entry date corresponds to the estimated time period when 50% of the tagged fish entered the ocean. Migration period is the time period (d) during which 95% of the tagged smolts entered the ocean. Average length and weights are in cm and g.

Stream name	Stock type	Tag Code	Tags applied	Weight tagging	Tag Ret. (%)	Time held	Tagged smolts	Median entry date	Migr. period	Mean weight	Mean length
Quins. R.	Colon.	22963	25191	6.4	99.6	12	5430	23-May	24	16.2	122
	Colon.	22962	25155	6.4	99.2	13	5452	23-May	24	16.2	122
	Colon.	23001	25361	6.4	99.6	8	5452	23-May	24	16.2	122
	Colon.	23002	25085	6.4	99.6	6	5452	23-May	24	16.2	122
Quins. R.	Prod.	22916	9520	12	98.2	79	9348	30-May	8	18.8	119
	Prod.	22917	8057	12	98.2	79	7911	30-May	8	18.8	119
	Prod.	22918	11040	12	98.2	79	10841	30-May	8	18.8	119
	Prod.	22919	10202	16	96.7	77	9865	30-May	8	22.5	125
	Prod.	22920	8873	16	96.7	77	8580	30-May	8	22.5	125
	Prod.	22921	10072	16	96.7	77	9739	30-May	8	22.5	125
	Prod.	22922	9973	16	96.7	50	9643	30-May	8	22.5	125
	Prod.	22923	10016	16	96.7	50	9685	30-May	8	22.5	125
	Prod.	22949	14430	18	99.7	16	14430	24-May	8	23.0	125
	Prod.	22950	14896	18	99.7	21	14896	24-May	8	23.0	125
	Prod.	22951	13270	18	99.7	n/a	13270	24-May	8	23.0	125
	Prod.	82313	50519	18	96.7	73	48851	30-May	8	24.9	127
	Prod.	82314	50061	13	100	59	50061	30-May	8	20.6	122
	Black Cr.	Wild	23119	8066	9.6	99.5	1	8026	15-May	15	9.6
Wild		23120	8095	9.6	99.5	1	8054	15-May	15	9.6	96
Wild		23121	8095	9.6	99.5	1	8054	15-May	15	9.6	96
Puntl. R.	Colon.	22762	36769	3.9	99.4	2	4742	25-May	41	8.2	96
	Colon.	22763	35439	3.9	91.7	2	4225	25-May	41	8.2	96
	Colon.	22801	36728	3.9	91.3	2	4359	25-May	41	8.2	96
Puntl. R.	Prod.	22912	9782	4.0	94.3	52	9224	25-May	8	19.8	127
	Prod.	22913	10262	4.0	95.7	53	9820	25-May	8	19.1	125
	Prod.	22914	9502	4.0	91.3	68	8675	25-May	8	20.7	129
	Prod.	22915	9732	4.0	91.3	68	8885	25-May	8	20.4	128
	Prod.	22943	14673	4.0	94.3	52	13836	25-May	8	19.8	127
	Prod.	22944	15394	4.0	95.7	58	14732	25-May	8	19.0	125
	Prod.	22945	14254	4.0	91.3	68	13013	25-May	8	20.7	129
	Prod.	22946	14598	4.0	91.3	68	13327	25-May	8	20.4	128
Trent R.	Enhanc.	23122	4473	7.5	99.9	1	4468	1-Jun	20	7.5	87
	Enhanc.	23123	4512	7.5	99.9	1	4507	1-Jun	20	7.5	87
L. Qual.	Wild	22937	3397	14.8	99	1	3363	29-May	12	14.8	108
	Wild	22937	3103	14.8	97	1	3010	29-May	12	14.8	108
	Wild	22938	6415	14.8	99	1	6351	29-May	12	14.8	108
	Wild	22939	6358	14.8	99	1	6294	29-May	12	14.8	108
B. Qual.	Prod.	22957	17014	10	97	10	16503	17-May	6	16.7	117
	Prod.	22958	20542	10	97	10	19925	17-May	6	16.7	117
	Prod.	22959	21304	10	97	10	20664	17-May	6	16.7	117
	Prod.	22960	21868	10	97	10	21211	17-May	6	16.7	117
	Prod.	82251	70002	10	97.4	219	68181	17-May	6	16.7	117
	Prod.	82252	40890	10	97.6	219	39908	17-May	6	16.7	117

Table 3.9. CWT groups migrating from each stream during 1986. Time held is the period retained (d) for CWT retention tests. Ocean entry date corresponds to the estimated time period when 50% of the tagged fish entered the ocean. Migration period is the time period (d) during which 95% of the tagged smolts entered the ocean. Average length and weights are in cm and g.

Stream (Type)	Stock type	Tag Code	Tags Applied	Weight tagging	Tag Ret. (%)	Time held	Tagged smolts	Median entry date	Migr. period	Mean weight	Mean length
Quins. R.	Wild	23834	6352	9.5	99.9	1	6345	21-May	31	8.9	94
	Wild	23835	6353	9.5	99.9	1	6346	21-May	31	8.9	94
	Wild	23836	6353	9.5	99.9	1	6346	21-May	31	8.9	94
Quins. R.	Colon.	23205	24712	5.4	99.2	8	7291	25-May	31	15.4	114
	Colon.	23207	24650	5.4	99.2	8	7291	25-May	31	15.4	114
	Colon.	23201	25653	5.4	100	13	7291	25-May	31	15.4	114
Quins. R.	Colon.	23202	24857	5.4	100	13	7291	25-May	31	15.4	114
	Prod.	23435	9514	18	94.9	85	9028	29-May	8	30.2	135
	Prod.	23436	9277	18	94.9	85	8803	29-May	8	30.2	135
	Prod.	23437	9622	18	98	88	9429	29-May	8	30.2	135
	Prod.	23438	9636	18	98	88	9443	29-May	8	30.2	135
	Prod.	23439	9823	18	99.1	86	9734	29-May	8	30.2	135
	Prod.	23440	9984	18	99.1	78	9894	29-May	8	30.2	135
	Prod.	23441	9766	18	99.5	78	9717	29-May	8	30.2	135
	Prod.	23442	9758	18	99.5	78	9709	29-May	8	30.2	135
Black Cr.	Wild	23841	8000	18.3	98.9	1	7912	23-May	21	18.3	117
	Wild	23825	8000	18.3	98.9	1	7912	23-May	21	18.3	117
	Wild	23823	8000	18.3	98.9	1	7912	23-May	21	18.3	117
	Wild	23824	8000	18.3	98.9	1	7912	23-May	21	18.3	117
Puntl. R.	Colon.	23231	32069	3.5	96.5	2	4023	25-May	40	8.1	96
	Colon.	23232	36672	3.5	98.5	2	4696	25-May	40	8.1	96
	Colon.	23233	36131	3.5	97.5	2	4579	25-May	40	8.1	96
	Prod.	23443	9942	4.0	98.7	37	6233	30-May	13	24.9	132
	Prod.	23444	9790	4.0	97.7	37	9564	30-May	13	26	133
	Prod.	23445	10166	4.0	100	37	10166	30-May	13	22.8	126
Trent R.	Prod.	23446	9226	4.0	92	36	8487	30-May	13	22.6	126
	Enhanc.	23827	3450	10	99.7	1	3439	26-May	34	10	101
	Enhanc.	23826	3450	10	99.7	1	3439	26-May	34	10	101
Rosew.	Enhanc.	23833	3664	10	99.7	1	3653	26-May	34	10	101
	B. Cr (p)	23125	7726	17.6	100	1	7726	21-May	3	17.6	114
	B. Cr (p)	23126	7497	17.6	100	1	7497	21-May	3	17.6	114
	B. Cr (p)	23127	7640	17.6	100	1	7640	21-May	3	17.6	114
	T. R. (p)	23432	6296	15.7	99	1	6296	21-May	3	15.7	111
	T. R. (p)	23433	6292	15.7	100	1	6292	21-May	3	15.7	111
	T. R. (p)	23434	6086	15.7	100	1	6086	21-May	3	15.7	111
	L. Q. (p)	23124	5106	18.2	99	1	5021	21-May	3	18.2	115
	L. Q. (p)	23130	4742	18.2	99	1	4694	21-May	3	18.2	115
B. Qual.	Prod.	23712	49185	13	96.6	65	47512	14-May	6	19.9	127
	Prod.	82406	64908	13	97.2	6	63090	14-May	6	19.9	127
	Prod.	82407	38180	13	97.9	4	37378	14-May	6	19.9	127
L. Qual.	Wild	23828	5468	18.4	99	1	5413	30-May	21	18.4	115
	Wild	23829	5488	18.4	99	1	5433	30-May	21	18.4	115
	Wild	23830	5400	18.4	99	1	5346	30-May	21	18.4	115
French Cr	Enhanc.	23831	8000	8.4	99.3	1	7946	26-May	26	8.4	91
	Enhanc.	23832	8000	8.4	99.3	1	7946	26-May	26	8.4	91
	Enhanc.	23837	8000	8.4	99.3	1	7946	26-May	26	8.4	91
Mills. R.	Colon.	81602*1	29447	3.6	n/a	1	5224	23-May	35	21.4	128
	Colon.	81603*1	26221	3.6	n/a	1	1775	23-May	35	15.2	117
	Colon.	81604*1	12632	9.9	n/a	1	2116	23-May	35	15.2	117

Table 3.10. CWT groups migrating from each stream during 1987. Time held is the period retained (d) for CWT retention tests. Ocean entry date corresponds to the estimated time period when 50% of the tagged fish entered the ocean. Migration period is the time period (d) during which 95% of the tagged smolts entered the ocean. Average length and weights are in cm and g.

Stream name	Stock type	Tag Code	Tags applied	Weight tagging	Tag Ret. (%)	Time held	Tagged smolts	Median entry date	Migr. period	Mean weight	Mean length
Quins. R.	Wild	82420	8183	8.9	99.9	1	8124	14-May	29	8.9	94
	Wild	24135	8067	8.9	99.9	1	8059	14-May	29	8.9	94
	Wild	24136	8089	8.9	99.9	1	8124	14-May	29	8.9	94
Quins. R.	Prod.	24138	13155	12	95.5	10	12563	28-May	5	25	127.5
	Prod.	24139	14495	12	99.7	10	14451	28-May	5	25	127.5
	Prod.	24140	14532	12	99.9	10	14517	28-May	5	25	127.5
Black Cr.	Wild	82421	9000	18.5	99	1	8910	12-May	18	18.5	117
	Wild	82422	9000	18.5	99	1	8910	12-May	18	18.5	117
	Wild	82423	9000	18.5	99	1	8910	12-May	18	18.5	117
	Wild	82424	9000	18.5	99	1	8910	12-May	18	18.5	117
Puntl. R.	Colon.	23655	29205	4	98	20	3721	25-May	41	8.1	96
	Colon.	24058	29687	3.7	99	21	3820	25-May	41	8.1	96
	Colon.	24060	33321	3.7	96	16	4157	25-May	41	8.1	96
Puntl. R.	Prod.	24149	19358	4	97	78	18622	14-May	14	19.7	123
	Prod.	24150	19892	4	97	68	19295	14-May	14	20.5	125
	Prod.	24151	18895	4	95	65	17950	14-May	14	22.5	126
Trent R.	Enhanc.	24127	5043	9.6	99	1	4993	19-May	30	9.6	97
	Enhanc.	24128	5100	9.6	99	1	5049	19-May	30	9.6	97
	Enhanc.	24129	5706	9.6	99	1	5649	19-May	30	9.6	97
Rosew.	B. C. (p)	24441	6475	13.5	97	1	6280	15-May	3	13.5	107
	B. C. (p)	24442	6475	13.5	97	1	6280	15-May	3	13.5	107
	B. C. (p)	24443	6475	13.5	97	1	6280	15-May	3	13.5	107
	L.Q. (p)	24126	7861	12.5	97.6	1	7672	15-May	3	12.5	105
	L.Q. (p)	24124	7861	12.5	97.6	1	7672	15-May	3	12.5	105
	L.Q. (p)	24125	7861	12.5	97.6	1	7672	15-May	3	12.5	105
	T. R. (p)	24440	5140	15	99	1	5088	15-May	3	15	110
	T. R. (p)	24131	5450	15	99	1	5396	15-May	3	15	110
	T. R. (p)	24130	5460	15	99	1	5405	15-May	3	15	110
L. Qual.	Wild	82417	6824	14.9	99	1	6756	17-May	21	14.9	107
	Wild	82418	6894	14.9	99	1	6825	17-May	21	14.9	107
	Wild	82419	6794	14.9	99	1	6726	17-May	21	14.9	107
B. Qual.	Prod.	24144	14105	15	92.2	59	13004	8-May	6	21.9	122
	Prod.	24145	14237	15	92.2	59	13126	8-May	6	21.9	122
	Prod.	24146	14347	15	92.2	59	13227	8-May	6	21.9	122
	Prod.	82410	45230	15	93.4	297	42244	8-May	6	21.9	122
	Prod.	82411	31505	15	93.4	297	29425	8-May	6	21.9	122
French Cr	Enhanc.	82425	8200	10	99	1	8118	16-May	24	10	95
	Enhanc.	82426	8200	10	99	1	8118	16-May	24	10	95
	Enhanc.	82427	8200	10	99	1	8118	16-May	24	10	95
Mills. R.	Colon.	23918	23138	3.6	n/a	n/a	3170	11-May	25	17.2	120
	Colon.	23919	24444	3.6	n/a	n/a	3349	11-May	25	17.2	120
	Colon.	23920	25930	3.6	n/a	n/a	885	11-May	25	17.2	120
	Colon.	23921	22111	3.6	n/a	n/a	754	11-May	25	17.2	120

Table 4.1. Summary of escapement enumeration statistics for Black Creek, 1985 - 1988. Run times correspond to the dates of first and last catches of adult coho at the fence. Figures on hydrological conditions are estimated from the first period of high water to the end of the season.

Black Creek	1985	1986	1987	1988
Start run	Oct. 21	Oct. 27	Nov. 11	Oct. 31
End run	Dec. 09	Nov. 24	Nov. 29	Nov. 18
Catch period	50 d	29 d	18 d	19 d
Partial counts	13	6	12	18
First freshet	Oct. 21	Nov. 18	Nov. 11	Nov. 1
Mean Level (cm)	36.6	59.2	82.8	68.5
SD Level	16.5	20.0	38.2	23.5
Mean Temp.(°C)	3.3	4.1	6.4	6.4
SD Temp.	3.20	1.92	1.48	1.96
Mean Disch. (m <sup>3</sup> /s)	1.5	3.2	16.6	10.1
SD Discharge	1.63	6.53	18.55	10.97
Peak Discharge	7.20	31.4	60.0	41.6
# Measurements	75	73	34	58
Adult count	2892	4408	750	2122
% no adipose	0.1	17.1	71.2	49.4
% no L.V	0	0	1.3	0
% no R.V	0	0	1.3	0
# released	2833	4386	714	2122
marks applied	1643	2710	713	1743
marks/released	58.0	61.8	99.9	82.1
Jack count	259	191	528	215
% no adipose	25.1	54.3	57.4	67.0
% no L.V	0	0	0.6	0
% no R.V	0	0	0	0
# released	194	188	525	162
marks applied	24	96	510	155
marks/released	12.4	51.1	97.1	95.7
Chum count	0	0	0	1
Chinook count	0	0	0	1
Pink count	0	0	0	0
Cutthroat count	8	5	0	6

Table 4.2. Summary of escapement enumeration statistics for Trent River, 1985 - 1988. Run times correspond to the dates of first and last catches of adult coho at the fence. Figures on hydrological conditions are estimated from the first period of high water to the end of the season. In cases where two figures are given, the second figures relate to the stream population remaining after accounting for emigration of marked fish recovered in adjacent streams.

Trent River	1985	1986	1987	1988
Start run	Oct. 3	Oct. 9	Oct. 14	Oct. 1
End run	Dec. 12	Nov. 18	Nov. 30	Dec. 4
Catch period	75 d	41 d	48 d	65 d
Partial counts	29	14	18	21
First freshet	Oct. 7	Oct. 26	Nov. 10	Oct. 16
Mean Level (cm)	49.1	30.8	35.0	43.0
SD Level	31.60	38.1	46.78	38.89
Mean Temp.(°C)	5.3	7.3	8.75	7.6
SD Temp.	4.33	3.73	2.75	3.00
Mean Disch. (m <sup>3</sup> /s)	7.6	16.2	14.5	15.2
SD Discharge	12.15	41.54	26.75	26.00
Peak Discharge	71.3	230	133	153
# Measurements	75	73	69	71
Adult count	459	1076 - 895	194 - 186	367 - 355
% no adipose	1.7	14.7 - 17.6	20.1	41.7
% no L.V	0	47.6 - 40.7	7.7	16.6
% no R.V	0	0.1 - 0.1	0.5	1.1
# released	323	959 - 778	144 - 136	324 - 310
marks applied	231	956 - 775	143 - 135	321 - 307
marks/released	71.5	99.7	99.3	99.1
Jack count	42	29	12 - 11	115
% no adipose	9.5	17.2	0.8	9.3
% no L.V	0	6.9	0.8	0
% no R.V	0	3.4	0	0
# released	38	26	11 - 10	115
marks applied	26	26	10 - 09	112
marks/released	68.4	100	90.9	97.4
Chum count	117	20	93	15
Chinook count	0	0	0	0
Pink count	1	0	2	3
Cutthroat count	0	0	0	0

Table 4.3. Summary of escapement enumeration statistics for French Creek, 1985 - 1988. Run periods correspond to the dates of first and last catches of adult coho at the fence. Figures on hydrological conditions are estimated from the first period of high water to the end of the season.

French Creek	1985	1986	1987	1988
Start run	Oct. 12	Nov. 18	Nov. 22	Oct. 17
End run	Dec. 13	Dec. 10	Nov. 30	Nov. 9
Catch period	64 d	23 d	9 d	24 d
Partial counts	5	6	17	17
First freshet	Oct. 19	Nov. 18	Nov. 22	Oct. 31
Mean Level (cm)	10.2	32.6	48.7	36.0
SD Level	3.78	11.5	18.63	19.30
Mean Temp.(°C)	4.2	5.2	6.4	7.5
SD Temp.	3.80	1.28	1.03	1.91
Mean Disch. (m <sup>3</sup> /s)	0.6	4.7	14.1	6.1
SD Discharge	0.62	4.04	12.8	5.86
Peak Discharge	2.9	16.2	41	24.0
# Measurements	61	28	21	54
Adult count	661	625	76	937
% no adipose	0	0	42.1	58.5
% no L.V	0	0.6	5.3	0.1
% no R.V	0	0.2	0	0
# released	571	558	57	853
marks applied	126	508	46	850
marks/released	22.1	91.0	80.1	99.6
Jack count	46	26	89	109
% no adipose	0	23.1	85.3	87.2
% no L.V	0	15.4	7.9	1.8
% no R.V	0	0	0	0
# released	46	24	89	109
marks applied	9	24	85	108
marks/released	19.6	100	95.5	99.1
Chum count	7	5	3	3
Chinook count	0	0	0	0
Pink count	0	0	0	0
Cutthroat count	0	0	0	3

Table 4.4. Timing, location and results of population surveys conducted at Black Creek, from 1985 to 1988. Marked coho had primary marks only. Unmarked coho lacked primary and secondary marks.

Survey date	Stream section	Adults caught	Adults marked	Adults no ad.	Adults no l.v	Adults no r.v	Jacks caught	Jack marked	Jacks no ad.	Jacks no l.v	Jacks no r.v
31.10.85	Low	12	2	0	0	0	7	1	1	0	0
12.11.85	Mid	14	4	0	0	0	2	0	1	0	0
19.11.85	Mid	15	6	0	0	0	1	0	1	0	0
10.12.85	Mid	3	2	0	0	0	0	0	0	0	0
12.12.85	M, L	21	6	0	0	0	1	0	0	0	0
22.12.85	M, L	46	11	0	0	0	0	0	0	0	0
Total		111	31	0	0	0	11	1	3	0	0
% Total			27.9	0	0	0		9.1	27.3	0	0
Unmarked		79		0	0	0	10		3	0	0
28.11.86	High	22	16	5	0	0	n/a	n/a	n/a	n/a	n/a
03.12.86	High	47	23	7	0	0	5	1	5	0	0
05.12.86	Mid	14	5	4	0	0	0	0	0	0	0
07.12.86	Fence	3	1	1	0	0	0	0	0	0	0
10.12.86	High	17	9	4	0	0	0	0	0	0	0
10.12.86	High	14	7	1	0	0	0	0	0	0	0
10.12.86	Mid	15	9	7	0	0	0	0	0	0	0
Total		132	70	29	0	0	5	1	5	0	0
% Total			53.3	22.0	0	0		20.0	100	0	0
Unmarked		63		11			4		4	0	0
29.11.87	Mid	5	5	3	0	0	3	1	2	0	0
29.11.87	Mid	11	11	5	0	0	15	8	7	0	0
12.12.87	High	25	21	19	0	0	3	1	2	0	0
Total		41	37	27	0	0	21	10	11	0	0
% Total			90.2	65.8	0	0		47.6	52.4	0	0
Unmarked		4		2	0	0	11		5	0	0
08.11.88	High	7	5	1	0	0	0	0	0	0	0
10.11.88	High	14	6	4	0	0	0	0	0	0	0
16.11.88	Mid	8	4	5	0	0	4	0	3	0	0
16.11.88	High	22	9	8	1	0	5	3	4	0	0
16.11.88	High	29	15	11	2	0	2	0	2	0	0
19.11.88	Fence	7	7	2	0	0	1	0	0	0	0
27.11.88	Mid	7	1	5	0	0	1	0	0	0	0
29.11.88	High	71	25	27	1	0	2	1	2	0	0
01.12.88	Fence	4	0	0	0	0	1	0	0	0	0
Total		169	72	63	4	0	16	4	11	0	0
% Total			42.6	37.2	2.4	0		25.0	68.8	0	0
Unmarked		92		24	3	0	12		8	0	0

Table 4.5. Timing, location and results of population surveys conducted at Trent River, from 1985 to 1988. Marked coho had primary marks only. Unmarked coho lacked primary and secondary marks.

Survey date	Stream section	Adults caught	Adults marked	Adults no ad.	Adults no l.v	Adults no r.v	Jacks caught	Jack marked	Jacks no ad.	Jacks no l.v	Jacks no r.v
02.11.85	Fence	3	2	0	0	0	1	1	0	0	0
08.11.85	H, L	26	6	0	0	0	1	0	0	0	0
16.11.85	H, F	59	3	0	0	0	2	2	0	0	0
20.11.85	Low	23	8	0	0	0	1	0	0	0	0
11.12.85	Low	4	0	0	0	0	0	0	0	0	0
15.12.85	High	19	5	0	0	0	2	0	1	0	0
21.12.85	High	15	1	0	0	0	0	0	0	0	0
24.12.85	Low	6	3	0	0	0	1	1	0	0	0
Total		155	28	0	0	0	8	4	1	0	0
% Total			18.1	0	0	0		50.0	12.5	0	0
Unmarked		127		0	0	0	4		1	0	0
08.11.86	Low	8	7	2	4	0	2	0	0	0	0
14.11.86	High	5	4	1	1	0	1	0	0	0	0
30.11.86	High	10	3	3	0	0	0	0	0	0	0
04.12.86	High	4	3	1	0	0	2	0	0	0	0
05.12.86	Low	11	5	2	4	0	1	0	1	0	0
09.12.86	High	15	9	3	2	0	0	0	0	0	0
12.12.86	High	14	5	1	3	0	0	0	0	0	0
Total		67	36	13	14	0	6	0	1	0	0
% Total			53.7	19.4	20.9	0		0	16.7	0	0
Unmarked		31		3	7	0	6		1	0	0
14.11.87	Mid	15	2	5	2	0	5	0	2	0	0
19.11.87	H,M,L	72	6	26	8	0	3	1	1	0	0
28.11.87	M,L	38	6	12	6	0	10	0	8	1	0
30.11.87	High	5	1	0	1	0	0	0	0	0	0
13.12.87	Mid	5	1	2	0	0	2	1	0	0	0
Total		135	16	45	17	0	20	2	11	1	0
% Total			11.8	33.3	12.6	0		10.0	55.0	5.0	0
Unmarked		109		38	16	0	10		9	1	0
23.10.88	H,L	8	7	6	1	0	8	2	1	0	0
14.11.88	H,M,L	33	7	12	2	0	8	2	4	0	0
18.11.88	High	32	7	19	2	2	11	1	6	0	0
19.11.88	Mid	16	6	7	1	0	2	0	0	0	0
25.11.88	Mid	30	9	20	13	0	1	0	1	0	0
26.11.88	Low	9	3	3	2	0	5	1	0	0	0
01.12.88	Fence	11	6	2	1	0	0	0	0	0	0
04.12.88	M,L	24	1	13	3	0	1	0	0	0	0
07.12.88	Fence	14	4	9	1	0	2	1	0	0	0
14.12.88	Mid	2	0	1	0	0	1	0	1	0	0
Total		179	50	92	26	2	39	7	13	0	0
% Total			27.9	51.3	14.5	0		17.9	33.3	0	0
Unmarked		129		57	22	0	30		11	0	0

Table 4.6. Timing, location and results of population surveys conducted at French Creek, from 1985 to 1988. Marked coho had primary marks only. Unmarked coho lacked primary and secondary marks.

Survey date	Stream section	Adults caught	Adults marked	Adults no ad.	Adults no l.v	Adults no r.v	Jacks caught	Jack marked	Jacks no ad.	Jacks no l.v	Jacks no r.v
05.11.85	M,M	44	2	0	0	0	5	1	0	0	0
14.11.85	Mid	57	4	0	0	0	9	0	0	0	0
11.12.85	Mid	23	2	0	0	0	2	0	0	0	0
23.12.85	High	45	7	0	0	0	0	0	0	0	0
Total		169	15	0	0	0	16	1	0	0	0
% Total			8.9	0	0	0		6.3	0	0	0
Unmarked		154		0	0	0	15		0	0	0
02.12.86	High	11	9	0	0	0	0	0	0	0	0
02.12.86	High	9	8	0	0	0	0	0	0	0	0
08.12.86	Mid	12	9	0	0	0	0	0	0	0	0
08.12.86	Mid	25	7	0	0	0	1	0	1	0	0
11.12.86	Mid	19	5	0	0	0	0	0	0	0	0
11.12.86	High	10	5	0	0	0	0	0	0	0	0
17.12.86	High	8	4	0	0	0	0	0	0	0	0
Total		94	47	0	0	0	1	0	1	0	0
% Total			50.0	0	0	0		0	100	0	0
Unmarked		45		0	0	0	1		1	0	0
26.11.87	Mid	3	2	1	1	0	6	2	5	0	0
27.11.87	Mid	3	2	1	0	0	2	1	1	0	0
08.12.87	High	5	1	3	0	0	2	1	2	0	0
14.12.87	Mid	5	1	0	0	0	0	0	0	0	0
Total		16	6	5	1	0	10	4	8	0	0
% Total			37.5	31.3	6.3	0		40	80.0	0	0
Unmarked		10		3	0	0	6		6	0	0
08.11.88	High	3	3	3	0	0	0	0	0	0	0
10.11.88	Mid	5	4	5	0	0	1	0	1	0	0
10.11.88	Mid	16	15	9	0	0	4	2	4	0	0
10.11.88	High	10	10	5	0	0	0	0	0	0	0
15.11.88	Fence	14	12	9	0	0	1	0	0	0	0
17.11.88	Mid	29	24	15	0	0	4	3	4	0	0
19.11.88	Fence	3	3	2	0	0	0	0	0	0	0
27.11.88	Mid	17	11	10	1	0	1	1	1	0	0
28.11.88	Mid	16	13	12	0	0	3	1	2	0	0
01.12.88	Fence	7	6	5	0	0	0	0	0	0	0
06.12.88	High	11	9	7	0	0	2	0	2	0	0
06.12.88	Mid	4	3	2	0	0	1	1	1	0	0
06.12.88	Fence	8	4	4	0	0	0	0	0	0	0
Total		143	117	88	1	0	17	8	15	0	0
% Total			81.8	61.5	0.7	0		47.1	88.2	0	0
Unmarked		25		11	1	0	8		7	0	0

Table 4.7. Comparison of mark ratios between male and female adult coho, obtained during stream surveys. No figures were provided for 1985 due to the absence on information on sex in the mark-recapture dataset.

Survey Area	Survey Year	Adult Sex	With Marks	No Marks	$\chi^2$	Prob.	Total Adult	% Marked
Black Cr.	86	F	32	22	3.377	>0.06	54	59.3
		M	21	32			53	39.6
	87	F	16	2	0.134	>0.7	18	88.9
		M	19	2			21	90.5
	88	F	27	42	0.222	>0.6	69	39.0
		M	44	56			100	44.0
Trent R.	86	F	9	19	3.089	>0.07	28	32.1
		M	15	10			25	60.0
	87	F	8	15	0.682	>0.4	23	34.8
		M	5	20			25	25.0
	88	F	23	67	0.226	>0.6	90	25.6
		M	26	61			87	29.9
French Cr.	86	F	17	22	0.594	>0.4	39	43.6
		M	25	21			46	54.3
	87	F	5	7	0.001	>0.9	12	41.7
		M	1	3			4	25.0
	88	F	65	14	0.003	>0.9	79	82.3
		M	52	12			64	81.2

Table 4.8. Comparison of fin clipped ratios among adults between samples collected at the fence and during stream surveys. Statistical significance at .05 level (\*) and <.01 level (\*\*). Trent River figures adjusted for emigration losses. Chi-squares are all adjusted for continuity.

Stream	Year	Site	no adip.	Adults normal	$\chi^2$	no L.V	Adults normal	$\chi^2$	no adip.	Jacks normal	$\chi^2$
<b>Black Cr.</b>											
1985	Fence	0	111	2.55	-	-	-	-	65	194	0.04
	River	2	2890		-	-	-	-	3	8	
1986	Fence	755	3653	1.07	-	-	-	-	107	84	n/a
	River	29	110		-	-	-	-	n/a	n/a	
1987	Fence	534	216	0.31	-	-	-	-	303	225	0.05
	River	27	14		-	-	-	-	11	10	
1988	Fence	1049	1073	8.78 **	-	-	-	-	144	71	0.17
	River	63	106		-	-	-	-	11	5	
<b>Trent R.</b>											
1985	Fence	8	451	1.55	0	459	-	-	4	38	1.49
	River	0	155		0	155	-	-	1	7	
1986	Fence	149	746	0.17	365	530	9.51 *	-	5	24	0.32
	River	13	54		14	53		-	1	5	
1987	Fence	38	148	6.13 *	15	171	1.32	-	1	11	5.12 *
	River	45	90		17	118		-	11	9	
1988	Fence	153	204	3.17	61	294	0.44	-	34	81	0.06
	River	92	87		26	153		-	13	26	
<b>French Cr.</b>											
1985	Fence	0	661	-	0	661	-	-	0	46	-
	River	0	169		0	169		-	0	16	
1986	Fence	0	625	-	4	621	0.00	-	6	20	-
	River	0	94		0	94		-	1	0	
1987	Fence	32	44	0.28	4	72	0.20	-	76	13	0.00
	River	5	11		1	15		-	8	2	
1988	Fence	548	389	0.36	9	928	0.03	-	95	14	0.07
	River	88	55		1	142		-	15	2	

Table 4.9. Minimum estimates of escapement levels, spawner abundance and fin clipped fish by age group, from 1985 to 1988. The fractions presented correspond to the figure above it expressed as a percentage of the total count. If no figures are available, the proportion is obtained from the stream survey data only.

Stream Year	Adult Total	Adult river	Adult no ad.	Adult no l.v	Adult no r.v	Jack Total	Jack river	Jack no ad.	Jack no l.v	Jack no r.v
<b>Back Cr.</b>										
1985	2892	2833 0.953	2 0.001	0 0	0 0	259	194 0.749	65 0.251	0 0	0 0
1986	4408	4386 0.995	755 0.171	0 0	0 0	191	188 0.979	107 0.56	0 0	0 0
1987	754	718 0.952	536 0.711	1 0.001	1 0.1	539	536 0.994	308 0.571	3 0.006	0 0
1988	2122	2122 1.00	1049 0.494	4 0.002	0 0	227	174 0.767	152 0.670	0 0	0 0
<b>Trent R.</b>										
1985	459	323 0.703	8 0.017	0 0	0 0	42	38 0.904	4 0.095	0 0	0 0
1986	926	809 0.874	152 0.164	372 0.401	1 0.001	32	32 1.00	6 0.188	0 0	0 0
1987	295	245 0.830	76 0.258	31 0.105	0 0.000	21	20 0.952	10 0.476	2 0.095	0 0
1988	484	441 0.911	210 0.433	83 0.171	4 0.008	145	145 1.00	45 0.310	1 0.007	0 0
<b>French Cr.</b>										
1985	661	571 0.86	0 0	0 0	0 0	61	61 1.0	0 0	0 0	0 0
1986	625	558 0.89	0 0	4 0.006	1 0.001	26	24 0.923	6 0.23	4 0.148	0 0
1987	86	67 0.779	35 0.407	4 0.047	0 0	95	95 1.0	82 0.863	7 0.074	0 0
1988	962	878 0.912	559 0.581	10 0.01	1 0.001	117	117 1.0	102 0.872	2 0.017	0 0

Table 4.10. Adjusted stream survey data on adults collected at Black Creek, 1985-1988. Survey data are categorized as live fish only (L), or combined live and dead fish (L+D). Primary and secondary marks (P. and S.). Stars (\*) indicate samples not used for Petersen-type estimators.

Census date	Calendar day	Stream section sampled	Primary marks released	Fish exam. (L)	P. marks recov. (L)	S. marks applied (L)	S. marks recov. (L)	Fish exam. (L+D)	P. mark rec. (L+D)	S. marks rec. (L+D)
31/10/85	304	Lower	620	9	1	9	-	12 *	3 *	-
12/11/85	306	Middle	694	14	4	14	0	14 *	4 *	0
19/11/85	323	Middle	854	14	6	14	0	15 *	7 *	0
10/12/85	344	Middle	1643	3	2	3	0	3	2	0
12/12/85	346	Middle	1643	19	5	19	0	21	8	0
22/12/85	356	M,L	1643	44	9	42	0	46	11	0
28/11/86	332	Upper	2710	22	16	22	-	22	16	-
3/12/86	337	Upper	2710	47	23	46	0	47	23	0
5/12/86	339	Middle	2710	14	5	14	0	17	6	0
10/12/86	344	U, M	2710	46	25	46	0	46	25	0
29/11/87	333	Middle	709	16	16	16	-	16	16	-
12/12/87	346	Upper	713	25	21	13	0	25	21	0
8/11/88	312	Upper	1734	7	5	7	-	7	5	-
9/11/88	313	Upper	1734	14	6	14	0	14	6	0
16/11/88	320	U, M, L	1740	59	28	58	2	66	35	2
29/11/88	333	U, M, L	1743	71	23	60	4	82	26	4

Table 4.11. Adjusted stream survey data on adults collected at Trent River, 1985-1988. Survey data are categorized as live fish only (L), or combined live and dead fish (L+D). Primary and secondary marks (P. and S.). Stars (\*) indicate samples not used for Petersen-type estimators.

Census date	Calendar day (t)	Stream section sampled	Primary marks released	Fish exam. (L)	P. mark recov. (L)	S. mark applied (L)	S. mark recov. (L)	Fish exam. (L+D)	P. mark recov. (L+D)	S. mark recov. (L+D)
8/11/85	312	U, L	188	26	6	22	-	29	8	-
16/11/85	320	U, L	202	56	2	53	0	59	3	0
20/11/85	324	Lower	205	22	7	18	0	23	8	0
13/12/85	347	U, L	231	19	5	19	0	23	5	0
23/12/85	357	U, L	231	21	4	21	0	21	4	0
8/11/86	312	Lower	412	8	7	8	-	8 *	7 *	-
14/11/86	318	Upper	532	5	4	5	0	5 *	4 *	0
30/11/86	334	Upper	755	10	3	10	0	10	3	0
5/12/86	339	U, L	755	15	8	9	0	15	8	0
9/12/86	343	Upper	755	15	9	15	0	15	9	0
12/12/86	346	Upper	755	14	15	14	0	14	5	0
14/11/87	318	Middle	42	15	2	15	-	15 *	2 *	-
19/11/87	323	U, M, L	79	70	6	66	0	72	6	0
29/11/87	333	M, L	134	38	6	36	5	38	6	5
13/12/87	347	Middle	135	5	1	5	0	5	1	0
24/10/88	297	U, L	64	8	7	8	-	8 *	7 *	-
14/11/88	318	U, M, L	304	33	7	32	0	33	7	0
19/11/88	323	U, M	308	48	13	45	2	48	13	2
26/11/88	330	M, L	308	38	12	32	5	39	12	5
1/12/88	335	Lower	308	0	0	0	0	11	6	2
6/12/88	340	U, M, L	308	24	1	4	5	40	5	6

Table 4.12. Adjusted stream survey data on adults collected at French Creek, 1985-1988. Survey data are categorized as live fish only (L), or combined live and dead fish (L+D). Primary and secondary marks (P. and S.). Stars (\*) indicate samples not used for Petersen-type estimators.

Census date	Calendar day	Stream section sampled	Primary marks released	Fish exam. (L)	P. mark recov. (L)	S. mark appl. (L)	S. mark recov. (L)	Fish exam. (L+D)	P. mark recov. (L+D)	S. mark recov. (L+D)
5/11/85	309	M, M	120	44	2	44	-	44	2	-
14/11/85	318	Middle	121	57	4	57	0	57	4	0
11/12/85	345	Middle	125	23	2	17	0	23	2	0
23/12/85	357	Upper	126	39	4	39	0	45	7	0
2/12/86	336	Upper	505	20	17	20	-	20	17	-
8/12/86	342	Middle	508	37	16	37	0	37	16	0
11/12/86	345	U, M	508	29	10	29	0	29	10	0
17/12/86	351	Upper	508	8	4	8	0	8	4	0
27/11/87	331	Middle	46	6	4	6	-	6	4	-
8/12/87	342	Upper	46	5	1	4	0	5	1	0
14/12/87	348	Middle	46	5	1	6	0	5	1	0
10/11/88	314	U, M	849	34	32	31	-	34	32	-
15/11/88	319	Lower	850	0	0	0	0	14	12	0
18/11/88	322	M, L	850	29	24	23	1	32	27	1
28/11/88	332	Middle	850	33	24	27	1	33	24	1
1/12/88	335	Lower	850	0	0	0	0	7	6	0
6/12/88	340	U,M,L	850	17	13	12	2	23	16	2

Table 4.13. Estimated abundances of adults (above), and jacks (below). Estimates generated from the open population model are; the best fitting combination of parameter ( $N'$ ), MLE and associated bounds, and the range of average stream residency periods associated with the plausible hypotheses ( $H_n$ 's). Pop. size limits & pattern indicates if constraints were imposed on the population size remaining on Dec. 15 (Yes - No), and the migration pattern which provided the best fit (U = uniform, D= decreasing).

Survey year	Total count	( $N'$ ) Open model ( $N, \phi, \sigma$ )	Pop. size limits & Pattern	Open model N, LB-UB	Residency period range ( $\bar{x}$ )	Bayesian model (N, LB-UB)	Bailey model (N, LB-UB)
B.C '85	2833	5433, 15, .40	No - U	5933, 4583-7483	4 - 24	6049, 4305-9074	5302, 7102-3501
B.C '86	4386	4596, 5, .64	No - U	4796, 4386-5536	5 - 32	5108, 4450-6015	5076, 4275-5876
B.C '87	718	789, 22, .15	No - U	749, 739-879	5 - 25	790, 727-910	788, 711-864
B.C '88	2122	3922, 26, .3	No - D	3222, 2622-4022	7 - 31	4083, 3491-4965	4059, 3360-4757
"	"	3222, 16, .5	Yes - D	3122, 2750-3600	6 - 19	"	"
T.R '85	323	773, 14, .15	No - U	973, 698-1073	10 - 18	1154, 825-1672	1242, 840-1643
T.R '86	809	1028, 20, .25	No - D	1078, 903-1428	6 - 34	1631, 1250-2233	1597, 1159-2034
T.R '87	245	995, 24, .25	No - D	995, 550-1750	4 - 16	880, 538-1584	1118, 587-1648
"	"	995, 12, .35	Yes - D	995, 720-1820	11 - 16	"	"
T.R '88	441	1112, 31, .15	No - U	962, 812-1250	31 - 33	1224, 945-1616	1204, 900-1507
"	"	912, 31, .15	Yes - U	862, 710-1062	25 - 32	"	"
F.C '85	571	1271, 14, .7	No - D	1021, 796-1496	18 - 32	1381, 879-2401	1338, 732-1943
F.C '86	558	668, 14, .3	No - D	768, 668-870	11 - 22	1016, 852-1289	1005, 807-1203
"	"	808, 20, .23	Yes - D	808, 738-898	19 - 22	"	"
F.C '87	67	97, 8, .55	No - D	107, 70-210	6 - 24	123, 67-324	111, 52-170
F.C '88	878	943, 14, .45	No - D	973, 920-1063	6 - 33	1038, 965-1131	1037, 958-1116
"	"	993, 24, .4	Yes - D	993, 935-1053	24 - 27	"	"
B.C '87	536	880, 23, .15	No - D	730, 620-1500	6 - 33	1072, 713-1924	1020, 612-1428
B.C '88	174	360, 6, .95	No - D	410, 250-1160	6 - 34	622, 282-1981	527, 172-881
"	"	343, 6, .95	Yes - D	343, 220-770	6 - 21	"	"
T.R '88	145	427, 5, .85	No - D	477, 300-1277	6 - 33	670, 345-1962	578, 193-963
"	"	427, 5, .85	Yes - D	477, 300-1270	8 - 30	"	"
F.C '87	95	216, 26, .15	No - D	216, 126-566	5 - 26	212, 110-712	187, 76-297
F.C '88	117	217, 33, .15	No - D	192, 145-317	6 - 34	230, 143-478	216, 121-311

Table 4.14. Estimates of the stream population of jacks based on jack-to-adult ratio at Black Creek, Trent River and French Creek, 1985-1988. Jack and adult figures are minimum escapement counts from Table 4.8. Estimates of adult abundance and associated bounds were obtained from Table 5.5.

Stream/Yr	# Jacks in stream	# Adults in stream	Jack / Adult ratio	J/A ratio 95% C.I	Adult Pop. (N <sup>o</sup> )	Jack Pop. (N)	Jack Pop. (LB - UB)
BC '85	194	2833	0.068	.067 - .079	5933	403	313 - 595
BC '86	188	4386	0.043	.042 - .050	4796	206	199 - 277
BC '87	536	718	0.747	.745 - .775	749	560	550 - 681
BC '88	174	2122	0.082	.081 - .095	3122	256	225 - 342
TR '85	38	323	0.118	.094 - .152	973	115	65 - 108
TR '86	32	809	0.396	.300 - .550	1078	41	26 - 78
TR '87	20	245	0.833	.560 - 1.260	995	83	42 - 230
TR '88	145	441	0.328	.299 - .367	862	281	212 - 390
FC '85	61	571	0.107	.087 - .127	1021	109	69 - 190
FC '86	24	558	0.043	.027 - .063	808	35	20 - 56
FC '87	95	67	1.418	1.234 - 1.652	107	123	116 - 259
FC '88	117	878	0.133	.118 - .154	993	131	110 - 162

Table 4.15. Estimates of total escapement and spawning population sizes for adults (above), and jacks (below) at Black Creek, Trent River and French Creek, 1984-1988. # Collected represent the number of fish removed at the fence or in the stream during the season.

Stream/yr.	Pop. size estimate	# Collected Fence	Estimated Escapement	Esc. LB - UB	# Collected Stream	Estimated Spawners	Spawners LB - UB
BC '84	Visual	0	5990	n/a	30	5060	n/a
BC '85	O. model	59	5992	4742-7542	0	5933	4583-7483
BC '86	O. model	22	4818	4658-5558	0	4796	4636-5536
BC '87	O. model	36	785	775-915	0	749	739-879
BC '88	O. model	0	3122	2750-3600	18	3104	2732-3582
TR '84	Visual	0	1109	n/a	20	1089	n/a
TR '85	O. model	136	1109	834-1209	57	916	641-1016
TR '86	O. model	117	1195	1020-1545	10	1068	893-1418
TR '87	O. model	50	1045	730-1430	4	991	676-1376
TR '88	O. model	43	905	753-1105	5	857	705-1057
FC '84	Visual	0	1111	n/a	20	1091	n/a
FC '85	O. model	90	1111	886-1586	10	1011	786-1486
FC '86	O. model	67	875	805-965	0	808	738-898
FC '87	O. model	19	126	89-229	0	107	70-210
FC '88	O. model	81	1074	1016-1134	0	993	935-1053
BC '84	Visual	0	350	n/a	0	350	n/a
BC '85	C. model	65	349	259-2812	0	284	194-2747
BC '86	C. model	3	483	195-2770	0	480	188-2767
BC '87	O. model	3	733	623-1503	0	730	620-1500
BC '88	O. model	53	396	273-823	0	343	220-770
TR '84	Visual	0	52	n/a	0	52	n/a
TR '85	C. model	4	52	42-222	0	48	38-218
TR '86	J/A ratio	3	44	36-81	0	41	26-78
TR '87	J/A ratio	1	84	43-231	0	83	42-230
TR '88	O. model	0	477	300-1270	0	477	300-1270
FC '84	Visual	0	143	n/a	0	143	n/a
FC '85	C. model	0	143	61-2231	0	143	61-2231
FC '86	J/A ratio	2	35	23-59	0	33	20-56
FC '87	O. model	0	216	126-566	0	216	126-566
FC '88	O. model	1	193	146-318	0	192	145-317

Table 4.16. Timing and duration of the adult coho runs at each location monitored during the 1985-1988 period. Duration figures represent the number of weeks during which 95% of the run occurred. Start and median dates represent the calendar day on which 5%, and 50% of the run had entered the stream.

Escapement location	Duration				Start date				Median date			
	1985	1986	1987	1988	1985	1986	1987	1988	1985	1986	1987	1988
Quinsam R.	9	10	9	9	271	288	286	285	288	324	309	306
Black Cr.	6	3	4	5	294	322	315	305	298	322	327	308
Puntl. R.	8	8	9	10	273	274	282	272	292	300	316	299
Trent R.	10	7	4	6	289	298	314	289	304	312	315	306
Rosew. Cr.	-	-	5	7	-	-	315	305	-	-	325	314
B. Qual. R.	10	10	10	9	285	280	281	291	300	299	303	311
L. Qual. R.	10	9	7	8	280	283	286	276	302	322	315	305
French Cr.	6	4	3	5	292	322	326	304	294	323	327	306
Millst. R.	5	3	3	4	292	322	326	304	294	323	327	307

Table 4.17. Results of a fixed effect factorial ANOVA test of adult body size (Post-orbital hypural length in cm). The adult returns year were 1985 to 1988. The eight streams tested consisted of all those used in the experiment except Rosewall Creek. The two sexes were adult males and females (no jacks).

Source	df	Sum of Squares	Mean Square	F-Value	P-Value
Year	3	9495.817	3165.272	116.870	.0001
Stream	7	2329.223	332.746	12.286	.0001
Sex	1	4486.957	4486.957	165.671	.0001
Year * Stream	21	5960.355	283.826	10.480	.0001
Year * Sex	3	237.605	79.202	2.924	.0326
Stream * Sex	7	811.475	115.925	4.280	.0001
Year * Stream * Sex	21	1218.826	58.039	2.143	.0018
Residual	4153	112478.282	27.084		

Table 5.1. Identities of tag codes recovered from adult coho sampled at major hatcheries on the Big Qualicum, Puntledge and Quinsam rivers, 1985-1988.

Recov. Loca.	Rec. year	Tag Code	Obs. rec.	Release Loca.	Recov. Loca.	Rec. year	Tag Code	Obs. rec.	Release Loca.	Recov. Loca.	Rec. year	Tag Code	Obs. rec.	Release Loca.
B. Qual.	1985	22615	60	B. Qual.	Puntl. R.	1986	22912	125	Puntl.	Quins. R.	1985	82232	135	Quins.
B. Qual.	1985	22616	80	B. Qual.	Puntl. R.	1986	22913	129	Puntl.	Quins. R.	1985	82233	115	Quins.
B. Qual.	1986	22945	3	Puntl.	Puntl. R.	1986	22914	141	Puntl.	Quins. R.	1985	82234	125	Quins.
B. Qual.	1986	22946	1	Puntl.	Puntl. R.	1986	22915	101	Puntl.	Quins. R.	1985	82235	145	Quins.
B. Qual.	1986	22957	32	B. Qual.	Puntl. R.	1986	22916	1	Quins.	Quins. R.	1985	82236	120	Quins.
B. Qual.	1986	22958	22	B. Qual.	Puntl. R.	1986	22934	1	Capil.	Quins. R.	1985	82237	132	Quins.
B. Qual.	1986	22959	54	B. Qual.	Puntl. R.	1986	22943	161	Puntl.	Quins. R.	1985	82238	186	Quins.
B. Qual.	1986	22960	45	B. Qual.	Puntl. R.	1986	22944	184	Puntl.	Quins. R.	1985	82239	201	Quins.
B. Qual.	1986	23123	1	Trent	Puntl. R.	1986	22945	175	Puntl.	Quins. R.	1985	82240	155	Quins.
B. Qual.	1986	82251	1	B. Qual.	Puntl. R.	1986	22946	142	Puntl.	Quins. R.	1985	82241	203	Quins.
B. Qual.	1986	82252	42	B. Qual.	Puntl. R.	1986	22962	1	Quins.	Quins. R.	1985	82242	185	Quins.
B. Qual.	1987	23124	1	Rosew.	Puntl. R.	1986	23002	1	Quins.	Quins. R.	1985	82243	136	Quins.
B. Qual.	1987	23125	1	Rosew.	Puntl. R.	1986	23122	2	Trent	Quins. R.	1985	82244	142	Quins.
B. Qual.	1987	23126	2	Rosew.	Puntl. R.	1986	23123	1	Trent	Quins. R.	1985	82245	145	Quins.
B. Qual.	1987	23127	7	Rosew.	Puntl. R.	1987	22906	1	Tsolum	Quins. R.	1985	82246	117	Quins.
B. Qual.	1987	23130	1	Rosew.	Puntl. R.	1987	23127	1	Rosew.	Quins. R.	1986	22413	1	Quins.
B. Qual.	1987	23432	3	Rosew.	Puntl. R.	1987	23152	14	Tsolum	Quins. R.	1986	22916	247	Quins.
B. Qual.	1987	23433	5	Rosew.	Puntl. R.	1987	23153	24	Tsolum	Quins. R.	1986	22917	198	Quins.
B. Qual.	1987	23434	4	Rosew.	Puntl. R.	1987	23154	17	Tsolum	Quins. R.	1986	22918	276	Quins.
B. Qual.	1987	23453	1	Capil.	Puntl. R.	1987	23155	8	Tsolum	Quins. R.	1986	22919	200	Quins.
B. Qual.	1987	23712	55	B. Qual.	Puntl. R.	1987	23156	10	Tsolum	Quins. R.	1986	22920	175	Quins.
B. Qual.	1987	23827	3	Trent	Puntl. R.	1987	23231	7	Puntl.	Quins. R.	1986	22921	205	Quins.
B. Qual.	1987	23829	1	L. Qual.	Puntl. R.	1987	23232	12	Puntl.	Quins. R.	1986	22922	150	Quins.
B. Qual.	1987	23830	1	L. Qual.	Puntl. R.	1987	23233	44	Puntl.	Quins. R.	1986	22923	153	Quins.
B. Qual.	1987	23832	1	French	Puntl. R.	1987	23443	2	Puntl.	Quins. R.	1986	22949	177	Quins.
B. Qual.	1987	23833	1	Trent	Puntl. R.	1987	23444	13	Puntl.	Quins. R.	1986	22950	182	Quins.
B. Qual.	1987	23837	2	French	Puntl. R.	1987	23445	35	Puntl.	Quins. R.	1986	22951	165	Quins.
B. Qual.	1987	81604	1	Millst.	Puntl. R.	1987	23446	46	Puntl.	Quins. R.	1986	22962	277	Quins.
B. Qual.	1987	82407	24	B. Qual.	Puntl. R.	1987	23815	31	Tsolum	Quins. R.	1986	22963	285	Quins.
B. Qual.	1988	23533	1	Tsolum	Puntl. R.	1987	23826	1	Trent	Quins. R.	1986	23001	152	Quins.
B. Qual.	1988	24124	1	Rosew.	Puntl. R.	1987	23833	2	Trent	Quins. R.	1986	23002	145	Quins.
B. Qual.	1988	24125	1	Rosew.	Puntl. R.	1988	23152	2	Tsolum	Quins. R.	1986	23119	1	Black
B. Qual.	1988	24126	1	Rosew.	Puntl. R.	1988	23154	2	Tsolum	Quins. R.	1986	82313	609	Quins.
B. Qual.	1988	24131	1	Rosew.	Puntl. R.	1988	23530	7	Tsolum	Quins. R.	1986	82314	815	Quins.
B. Qual.	1988	24144	31	B. Qual.	Puntl. R.	1988	23532	17	Tsolum	Quins. R.	1987	22962	1	Quins.
B. Qual.	1988	24145	25	B. Qual.	Puntl. R.	1988	23533	2	Tsolum	Quins. R.	1987	23001	2	Quins.
B. Qual.	1988	24146	28	B. Qual.	Puntl. R.	1988	23534	9	Tsolum	Quins. R.	1987	23002	4	Quins.
B. Qual.	1988	24443	1	Rosew.	Puntl. R.	1988	23655	23	Tsolum	Quins. R.	1987	23201	110	Quins.
B. Qual.	1988	24443	1	Rosew.	Puntl. R.	1988	24055	1	Tsolum	Quins. R.	1987	23202	148	Quins.
B. Qual.	1988	82410	3	B. Qual.	Puntl. R.	1988	24058	60	Puntl.	Quins. R.	1987	23205	147	Quins.
B. Qual.	1988	82411	52	B. Qual.	Puntl. R.	1988	24060	17	Puntl.	Quins. R.	1987	23207	145	Quins.
Puntl. R.	1985	22327	1	Puntl.	Puntl. R.	1988	24126	1	Rosew.	Quins. R.	1987	23435	133	Quins.
Puntl. R.	1985	22362	1	Puntl.	Puntl. R.	1988	24127	8	Trent	Quins. R.	1987	23436	113	Quins.
Puntl. R.	1985	22447	28	Puntl.	Puntl. R.	1988	24128	8	Trent	Quins. R.	1987	23437	102	Quins.
Puntl. R.	1985	22603	26	Puntl.	Puntl. R.	1988	24129	14	Trent	Quins. R.	1987	23438	112	Quins.
Puntl. R.	1985	22604	25	Puntl.	Puntl. R.	1988	24149	78	Puntl.	Quins. R.	1987	23439	117	Quins.
Puntl. R.	1985	22643	367	Puntl.	Puntl. R.	1988	24150	106	Puntl.	Quins. R.	1987	23440	150	Quins.
Puntl. R.	1985	22644	430	Puntl.	Puntl. R.	1988	24151	94	Puntl.	Quins. R.	1987	23441	107	Quins.
Puntl. R.	1985	22645	449	Puntl.	Quins. R.	1985	22349	198	Quins.	Quins. R.	1987	23442	125	Quins.
Puntl. R.	1985	22723	483	Puntl.	Quins. R.	1985	22413	367	Quins.	Quins. R.	1987	23834	25	Quins.
Puntl. R.	1986	22762	15	Puntl.	Quins. R.	1985	22448	262	Quins.	Quins. R.	1987	23835	28	Quins.
Puntl. R.	1986	22763	32	Puntl.	Quins. R.	1985	22548	565	Quins.	Quins. R.	1987	23836	13	Quins.
Puntl. R.	1986	22801	41	Puntl.	Quins. R.	1985	22549	296	Quins.	Quins. R.	1988	24135	27	Quins.
Puntl. R.	1986	22902	6	Tsolum	Quins. R.	1985	22550	301	Quins.	Quins. R.	1988	24136	12	Quins.
Puntl. R.	1986	22903	28	Tsolum	Quins. R.	1985	22619	1	Chilli	Quins. R.	1988	24138	168	Quins.
Puntl. R.	1986	22904	1	Tsolum	Quins. R.	1985	82229	234	Quins.	Quins. R.	1988	24139	226	Quins.
Puntl. R.	1986	22905	11	Tsolum	Quins. R.	1985	82230	204	Quins.	Quins. R.	1988	24140	253	Quins.
Puntl. R.	1986	22906	32	Tsolum	Quins. R.	1985	82231	167	Quins.	Quins. R.	1988	82420	30	Quins.

Table 5.2. Identities of tag codes recovered from jack coho sampled at major hatcheries on the Big Qualicum, Puntledge and Quinsam rivers, 1985-1988.

Recov. Loca.	Rec. Year	Tag Code	Obs. Rec.	Release Loca.	Recov. Loca.	Rec. Year	Tag Code	Obs. Rec.	Release Loca.
B. Qual	1985	22951	1	Quins.	Puntl. R.	1987	23530	1	Tsolum
B. Qual	1985	22957	13	B. Qual.	Puntl. R.	1987	24058	1	Puntl.
B. Qual	1985	22958	14	B. Qual.	Puntl. R.	1987	24060	2	Puntl.
B. Qual	1985	22959	15	B. Qual.	Puntl. R.	1987	24128	1	Trent
B. Qual	1985	22960	20	B. Qual.	Puntl. R.	1987	24149	22	Puntl.
B. Qual	1985	82252	46	B. Qual.	Puntl. R.	1987	24150	35	Puntl.
B. Qual	1986	23712	40	B. Qual.	Puntl. R.	1987	24151	70	Puntl.
B. Qual	1986	82407	20	B. Qual.	Puntl. R.	1988	24628	47	Puntl.
B. Qual	1987	24124	1	Rosew.	Puntl. R.	1988	24629	36	Puntl.
B. Qual	1987	24125	1	Rosew.	Puntl. R.	1988	24630	36	Puntl.
B. Qual	1987	24127	1	Trent	Puntl. R.	1988	24631	25	Puntl.
B. Qual	1987	24131	2	Rosew.	Puntl. R.	1988	24717	3	Puntl.
B. Qual	1987	24144	82	B. Qual.	Puntl. R.	1988	24719	8	Puntl.
B. Qual	1987	24145	91	B. Qual.	Puntl. R.	1988	24721	1	Puntl.
B. Qual	1987	24146	65	B. Qual.	Puntl. R.	1988	82438	1	Trent
B. Qual	1987	24440	2	Rosew.	Quins. R.	1985	22916	50	Quins.
B. Qual	1987	82411	155	B. Qual.	Quins. R.	1985	22917	41	Quins.
B. Qual	1987	82419	1	L. Qual.	Quins. R.	1985	22918	52	Quins.
B. Qual	1987	82423	1	Black	Quins. R.	1985	22919	95	Quins.
B. Qual	1987	82425	2	French	Quins. R.	1985	22920	99	Quins.
B. Qual	1987	82426	4	French	Quins. R.	1985	22921	139	Quins.
B. Qual	1987	82427	1	French	Quins. R.	1985	22922	70	Quins.
B. Qual	1988	25102	60	B. Qual.	Quins. R.	1985	22923	71	Quins.
B. Qual	1988	25111	24	B. Qual.	Quins. R.	1985	22949	151	Quins.
B. Qual	1988	25112	5	B. Qual.	Quins. R.	1985	22950	104	Quins.
B. Qual	1988	25130	31	B. Qual.	Quins. R.	1985	22951	72	Quins.
B. Qual	1988	25131	22	B. Qual.	Quins. R.	1985	22962	62	Quins.
B. Qual	1988	25132	14	B. Qual.	Quins. R.	1985	22963	78	Quins.
B. Qual	1988	25133	24	B. Qual.	Quins. R.	1985	23001	62	Quins.
B. Qual	1988	25134	34	B. Qual.	Quins. R.	1985	23002	54	Quins.
B. Qual	1988	25135	16	B. Qual.	Quins. R.	1985	82313	186	Quins.
B. Qual	1988	82435	1	French	Quins. R.	1985	82314	280	Quins.
B. Qual	1988	82436	2	French	Quins. R.	1986	23035	2	Chehal.
B. Qual	1988	82441	1	Black	Quins. R.	1986	23201	31	Quins.
B. Qual	1988	82443	1	Black	Quins. R.	1986	23202	32	Quins.
B. Qual	1988	82444	2	Quins.	Quins. R.	1986	23205	54	Quins.
B. Qual	1988	82450	2	Black	Quins. R.	1986	23207	60	Quins.
B. Qual	1988	82451	1	Black	Quins. R.	1986	23435	123	Quins.
Puntl. R.	1985	22762	3	Puntl.	Quins. R.	1986	23436	112	Quins.
Puntl. R.	1985	22763	2	Puntl.	Quins. R.	1986	23437	93	Quins.
Puntl. R.	1985	22801	3	Puntl.	Quins. R.	1986	23438	111	Quins.
Puntl. R.	1985	22912	116	Puntl.	Quins. R.	1986	23439	116	Quins.
Puntl. R.	1985	22913	192	Puntl.	Quins. R.	1986	23440	114	Quins.
Puntl. R.	1985	22914	247	Puntl.	Quins. R.	1986	23441	142	Quins.
Puntl. R.	1985	22915	189	Puntl.	Quins. R.	1986	23442	127	Quins.
Puntl. R.	1985	22943	135	Puntl.	Quins. R.	1986	23836	1	Quins.
Puntl. R.	1985	22944	212	Puntl.	Quins. R.	1987	24135	5	Quins.
Puntl. R.	1985	22945	328	Puntl.	Quins. R.	1987	24136	5	Quins.
Puntl. R.	1985	22946	252	Puntl.	Quins. R.	1987	24138	250	Quins.
Puntl. R.	1986	23155	1	Tsolum	Quins. R.	1987	24139	290	Quins.
Puntl. R.	1986	23231	1	Puntl.	Quins. R.	1987	24140	308	Quins.
Puntl. R.	1986	23233	2	Puntl.	Quins. R.	1987	82420	6	Quins.
Puntl. R.	1986	23443	4	Puntl.	Quins. R.	1988	24505	287	Quins.
Puntl. R.	1986	23444	9	Puntl.	Quins. R.	1988	24506	281	Quins.
Puntl. R.	1986	23445	22	Puntl.	Quins. R.	1988	24507	272	Quins.
Puntl. R.	1986	23446	41	Puntl.	Quins. R.	1988	82444	1042	Quins.
Puntl. R.	1986	23815	11	Tsolum	Quins. R.	1988	82445	894	Quins.
Puntl. R.	1986	81602	1	Millst.	Nanaimo R.	1986	081602	1	Millst. R.

Table 5.3. Identities of tag codes recovered from adult coho sampled in streams without major hatchery production facilities, 1985-1988. Tagged fish lacking a left ventral fin that were recovered during 1986 are indicated by (lv) mark.

Recov. Location	Recov. Year	Tag Code	Obs. Rec.	Release Location	Recov. Location	Recov. Year	Tag Code	Obs. Rec.	Release Location
Black Cr.	1986	23119	4	Black	Rosew.	1987	23432	23	Rosew.
Black Cr.	1986	23120	15	Black	Rosew.	1987	23433	29	Rosew.
Black Cr.	1986	23121	9	Black	Rosew.	1987	23434	24	Rosew.
Black Cr.	1987	23823	16	Black	Rosew.	1988	24124	37	Rosew.
Black Cr.	1987	23824	5	Black	Rosew.	1988	24125	66	Rosew.
Black Cr.	1987	23825	11	Black	Rosew.	1988	24126	59	Rosew.
Black Cr.	1987	23841	7	Black	Rosew.	1988	24129	1	Trent
Black Cr.	1988	82421	12	Black	Rosew.	1988	24130	47	Rosew.
Black Cr.	1988	82422	8	Black	Rosew.	1988	24131	44	Rosew.
Black Cr.	1988	82423	7	Black	Rosew.	1988	24149	1	Puntl.
Black Cr.	1988	82424	7	Black	Rosew.	1988	24440	49	Rosew.
Trent R.	1985	22643	2	Puntl.	Rosew.	1988	24441	54	Rosew.
Trent R.	1985	22644	2	Puntl.	Rosew.	1988	24442	34	Rosew.
Trent R.	1985	22645	4	Puntl.	Rosew.	1988	24443	40	Rosew.
Trent R.	1985	22723	1	Puntl.	L. Qual.	1986	22937	5	L. Qual.
Trent R.	1986	22912	1	Puntl.	L. Qual.	1986	22938	21	L. Qual.
Trent R.	1986	22913	1	Puntl.	L. Qual.	1986	22939	20	L. Qual.
Trent R.	1986	22914	2	Puntl.	L. Qual.	1987	23434	1	Rosew.
Trent R.	1986	22915	6	Puntl. (lv)	L. Qual.	1987	23828	43	L. Qual.
Trent R.	1986	22923	1	Quins. (lv)	L. Qual.	1987	23829	55	L. Qual.
Trent R.	1986	22944	2	Puntl.	L. Qual.	1987	23830	46	L. Qual.
Trent R.	1986	22946	20	Puntl. (lv)	L. Qual.	1988	82418	83	L. Qual.
Trent R.	1986	23122	9	Trent	L. Qual.	1988	82419	66	L. Qual.
Trent R.	1986	23123	9	Trent	L. Qual.	1988	82417	89	L. Qual.
Trent R.	1987	23125	1	Rosew.	L. Qual.	1988	24441	2	Rosew.
Trent R.	1987	23152	1	Puntl.	L. Qual.	1988	82426	1	French
Trent R.	1987	23432	2	Rosew.	L. Qual.	1988	24145	1	B. Qual.
Trent R.	1987	23434	2	Rosew.	L. Qual.	1988	82430	1	Cowich.
Trent R.	1987	23445	1	Puntl.	French Cr.	1987	23831	5	French
Trent R.	1987	23826	2	Trent	French Cr.	1987	23832	3	French
Trent R.	1987	23827	4	Trent	French Cr.	1987	23837	3	French
Trent R.	1987	23833	4	Trent	French Cr.	1988	82425	29	French
Trent R.	1988	24127	19	Trent	French Cr.	1988	82426	19	French
Trent R.	1988	24128	18	Trent	French Cr.	1988	82427	31	French
Trent R.	1988	24129	20	Trent	Millst. R.	1987	81602	12	Millst.
Trent R.	1988	24151	1	Puntl.	Millst. R.	1987	81603	3	Millst.
Rosew.	1987	23124	15	Rosew.	Millst. R.	1987	81604	4	Millst.
Rosew.	1987	23125	20	Rosew.	Millst. R.	1988	23918	18	Millst.
Rosew.	1987	23126	16	Rosew.	Millst. R.	1988	23919	16	Millst.
Rosew.	1987	23127	19	Rosew.	Millst. R.	1988	23920	5	Millst.
Rosew.	1987	23130	17	Rosew.	Millst. R.	1988	23921	4	Millst.

Table 5.4. Identities of tag codes recovered from jack coho sampled in streams without major hatchery production facilities, 1985-1988.

Recov. Location	Recov. Year	Tag Code	Obs. Rec.	Release Location	Recov. Location	Recov. Year	Tag Code	Obs. Rec.	Release Location
Black Cr.	1985	23119	19	Black	Rosew. Cr.	1987	24124	23	Rosew.
Black Cr.	1985	23120	26	Black	Rosew. Cr.	1987	24125	13	Rosew.
Black Cr.	1985	23121	17	Black	Rosew. Cr.	1987	24126	20	Rosew.
Black Cr.	1987	82421	4	Black	Rosew. Cr.	1987	24130	4	Rosew.
Black Cr.	1987	82423	1	Black	Rosew. Cr.	1987	24131	3	Rosew.
Black Cr.	1988	82440	2	Black	Rosew. Cr.	1987	24440	4	Rosew.
Black Cr.	1988	82441	5	Black	Rosew. Cr.	1987	24441	9	Rosew.
Black Cr.	1988	82442	2	Black	Rosew. Cr.	1987	24442	8	Rosew.
Black Cr.	1988	82443	1	Black	Rosew. Cr.	1987	24443	5	Rosew.
Black Cr.	1988	82446	1	Black	Rosew. Cr.	1987	82426	1	French
Black Cr.	1988	82447	1	Black	L. Qual.	1985	22937	2	L.Qual.
Black Cr.	1988	82449	10	Black	L. Qual.	1985	22938	26	L.Qual.
Black Cr.	1988	82450	19	Black	L. Qual.	1985	22939	18	L.Qual.
Black Cr.	1988	82451	15	Black	L. Qual.	1986	23828	17	L.Qual.
Black Cr.	1988	82461	1	Black	L. Qual.	1986	23829	11	L.Qual.
Trent R.	1985	22846	1	Porp. B.	L. Qual.	1986	23830	19	L.Qual.
Trent R.	1985	22915	1	Puntl.	L. Qual.	1987	82417	92	L.Qual.
Trent R.	1985	22938	1	L. Qual.	L. Qual.	1987	82418	80	L.Qual.
Trent R.	1985	82249	1	Capil.	L. Qual.	1987	82419	80	L.Qual.
Trent R.	1986	23437	1	Quins.	L. Qual.	1987	82426	1	French
Trent R.	1988	82438	2	Trent	French Cr.	1986	23832	2	French
Trent R.	1988	82439	1	Trent	French Cr.	1987	82427	3	French
Rosew. Cr.	1986	23124	18	Rosew.	French Cr.	1988	82435	4	French
Rosew. Cr.	1986	23125	21	Rosew.	French Cr.	1988	82436	3	French
Rosew. Cr.	1986	23126	20	Rosew.	French Cr.	1988	82437	1	French
Rosew. Cr.	1986	23127	22	Rosew.	Millst. R.	1986	81602	3	Millst.
Rosew. Cr.	1986	23130	8	Rosew.	Millst. R.	1987	23918	4	Millst.
Rosew. Cr.	1986	23432	9	Rosew.	Millst. R.	1987	23919	2	Millst.
Rosew. Cr.	1986	23433	9	Rosew.	Millst. R.	1988	23915	1	Millst.
Rosew. Cr.	1986	23434	11	Rosew.	Millst. R.	1988	23916	1	Millst.

Table 5.5. Identities of tag codes recovered from coho sampled in adjacent streams where escapements levels were not known (select samples), 1985-1988.

Recovery Location	Recovery Year	Survey method	Age group	Tag Code	Observed Recoveries	Release Location
Chase R.	1988	visual	Jack	24638	1	Millst.
Chef Cr.	1986	electroshock	Adults	22944	1	Puntl.
Chef Cr.	1986	electroshock	Adults	22945	2	Puntl.
Chef Cr.	1987	electroshock	Adults	23436	1	Quins.
Chef Cr.	1987	electroshock	Adults	23127	1	Rosew.
Chef Cr.	1987	electroshock	Adults	23434	1	Rosew.
Coal Cr.	1987	electroshock	Adults	23124	1	Rosew.
L. River	1986	electroshock	Adults	23119	1	Black
L. River	1986	electroshock	Adults	23120	1	Black
Oyster R.	1987	fence count	Adults	23823	5	Black
Oyster R.	1987	fence count	Adults	23824	3	Black
Oyster R.	1987	fence count	Adults	23825	6	Black
Oyster R.	1987	fence count	Adults	23841	5	Black
Tsable R.	1986	visual	Adults	22906	1	Puntl.
Waterloo R.	1988	visual	Adults	24443	1	Rosew.

Table 5.6. Estimated escapements, sampling statistics, and number of recoveries by category for each stream where random samples were obtained, 1985-1988.

Escapement Location	Rec. year	Age group	Fish exam.	Estim. escap.	Number no pin	Number lost pin	Number no data	Known tags	Tag loss (by age)	Tag loss (cohort)
Quins. R.	85	A	20843	29412	361	68	1	4837	0.07	-
Quins. R.	85	J	13531	14364	64	24	0	1666	0.04	0.049
Quins. R.	86	A	26980	36651	255	27	1	4413	0.05	-
Quins. R.	86	J	16919	18142	115	11	0	1118	0.09	0.108
Quins. R.	87	A	18253	26112	215	9	323	1582	0.12	-
Quins. R.	87	J	37571	39530	480	2	43	864	0.36	0.369
Quins. R.	88	A	31974	35190	447	4	152	716	0.38	-
Quins. R.	88	J	30252	30488	270	0	0	2776	0.09	-
Puntl. R.	85	A	37011	59266	127	11	0	1810	0.07	-
Puntl. R.	85	J	11611	18287	192	14	0	1679	0.10	0.114
Puntl. R.	86	A	10114	16068	198	4	0	1327	0.13	-
Puntl. R.	86	J	3189	3235	36	0	0	92	0.28	0.234
Puntl. R.	87	A	4890	7811	74	1	0	268	0.22	-
Puntl. R.	87	J	1159	1736	11	1	0	132	0.08	0.114
Puntl. R.	88	A	4654	5609	64	1	3	447	0.13	-
Puntl. R.	88	J	3906	5034	57	1	1	157	0.27	-
B. Qual. R.	85	A	19572	22182	24	1	209	140	0.15	-
B. Qual. R.	85	J	7023	7077	17	1	83	109	0.13	0.182
B. Qual. R.	86	A	15669	18950	52	0	40	201	0.21	-
B. Qual. R.	86	J	3449	4006	12	0	1	60	0.17	0.215
B. Qual. R.	87	A	6496	7793	36	1	80	114	0.24	-
B. Qual. R.	87	J	9393	9862	62	2	7	409	0.13	0.125
B. Qual. R.	88	A	4290	7038	18	1	4	146	0.11	-
B. Qual. R.	88	J	5210	5572	37	4	0	240	0.13	-
L. Qual. R.	85	A	1914	3560	0	0	0	0	-	-
L. Qual. R.	85	J	240	270	4	0	0	46	0.08	0.155
L. Qual. R.	86	A	947	947	13	1	38	46	0.22	-
L. Qual. R.	86	J	87	87	5	0	0	47	0.10	0.077
L. Qual. R.	87	A	1275	1275	11	0	91	145	0.07	-
L. Qual. R.	87	J	440	440	8	0	0	253	0.03	0.083
L. Qual. R.	88	A	801	1750	37	0	14	243	0.13	-
L. Qual. R.	88	J	81	81	0	0	0	0	-	-
Rosew. Cr.	86	A	5	5	1	0	0	0	1.00	-
Rosew. Cr.	86	J	124	124	6	0	0	118	0.05	0.044
Rosew. Cr.	87	A	182	182	7	1	0	163	0.04	-
Rosew. Cr.	87	J	104	104	12	0	0	90	0.12	0.069
Rosew. Cr.	88	A	502	502	27	1	9	432	0.06	-
Rosew. Cr.	88	J	147	147	0	0	0	0	-	-
Black Cr.	85	A	2892	5992	1	0	1	0	-	-
Black Cr.	85	J	259	349	2	0	1	62	0.03	0.022
Black Cr.	86	A	4408	4818	0	0	725	28	0.00	-
Black Cr.	86	J	191	483	0	0	106	0	-	-
Black Cr.	87	A	754	785	2	0	495	39	0.05	-
Black Cr.	87	J	539	733	0	0	302	5	0.00	0.152
Black Cr.	88	A	2122	3122	7	0	1007	34	0.17	-
Black Cr.	88	J	227	396	0	0	95	57	0.00	-
Trent R.	85	A	604	1109	2	0	0	9	-	-
Trent R.	85	J	46	52	0	0	0	4	0.00	-
Trent R.	86	A	926	1195	4	0	96	51	-	-
Trent R.	86	J	32	44	2	0	3	1	-	-
Trent R.	87	A	295	1045	5	0	54	17	-	-
Trent R.	87	J	21	84	0	0	10	0	-	-
Trent R.	88	A	484	905	8	0	143	58	0.12	-
Trent R.	88	J	145	477	0	0	41	3	-	-
French Cr.	85	A	661	1111	0	0	0	0	0.00	-
French Cr.	85	J	61	143	0	0	0	0	0.00	-
French Cr.	86	A	625	875	0	0	0	0	0.00	-
French Cr.	86	J	26	35	0	0	3	2	0.00	0.133
French Cr.	87	A	86	126	2	0	22	11	0.15	-
French Cr.	87	J	95	216	0	0	78	3	0.00	0.079
French Cr.	88	A	962	1074	7	0	472	79	0.08	-
French Cr.	88	J	117	193	1	0	93	8	0.11	-
Millstone	86	J	89	200	0	0	0	3	0.00	0.083
Millstone	87	A	44	57	2	0	0	19	0.10	-
Millstone	87	J	13	25	0	0	0	6	0.00	0.000
Millstone	88	A	69	275	0	0	0	43	0.00	-
Millstone	88	J	n/a	n/a	0	0	0	2	0.00	0.000
Nanaimo R.	86	J	26	157	5	0	0	1	0.83	0.83

Table 5.7. Estimates of the proportion of tagged fish escaping to each stream that were strays.

Stream	Jacks				Adults			
	1985	1986	1987	1988	1985	1986	1987	1988
Quins. R.	0	0	0	0	0.02	0.02	0	0
Black Cr.	0	0	0	0	0	0	0	0
Puntl. R.	0	14.1	1.5	0.6	0	6.4	40.7	21
Trent R.	100	33	n/a	0	100	67.2	52.9	1.7
Rosew. Cr.	-	0	1.1	0	-	n/a	0	0.5
B. Qual. R.	0.9	0	3.9	4.2	0	2.5	30.7	4.8
L. Qual. R.	0	0	0.4	0	0	0	0.7	5.3
French Cr.	0	0	0	0	0	0	0	0
Millstone. R.	-	0	0	0	-	0	0	0

Table 5.8. Estimates of tag loss at release and at return among fish released from each stream. Time periods correspond to the year that smolts migrated to sea. Estimates within brackets are not computed from escapement samples (see Section 5). Figures in the lower section are estimates based on recovery statistics corrected for the potential misidentification of untagged fish in the escapement.

Stream	Release			Return			Difference		
	1985	1986	1987	1985	1986	1987	1985	1986	1987
Quinsam R.	0.018	0.021	0.015	0.049	0.108	0.369	0.031	0.087	0.354
Black Cr.	0.005	0.021	0.001	0.022	0.05	0.152	0.017	0.029	0.151
Puntl. R.	0.068	0.028	0.039	0.114	0.234	0.114	0.046	0.206	0.075
Trent R.	0.001	0.003	0.001	(0.058)	(0.071)	0.120	-	-	0.119
Rosew. Cr.	-	0.005	0.002	-	0.044	0.069	-	0.039	0.067
L. Qual. R.	0.002	0.001	0.001	0.155	0.077	0.083	0.153	0.076	0.082
B. Qual. R.	0.027	0.034	0.072	0.182	0.215	0.125	0.155	0.181	0.053
French Cr.	-	0.007	0.001	-	0.133	0.079	-	0.126	0.078
Millstone R.	-	-	-	-	0.083	0	-	-	-
Quinsam R.	0.018	0.021	0.015	0.007	0.039	0.147	-0.011	0.017	0.132
Puntl. R.	0.068	0.028	0.039	0.063	0.084	0.039	-0.005	0.056	0.000
B. Qual. R.	0.027	0.034	0.072	n/a	.000	n.a	-	0.034	-

Table 5.9. Estimated recoveries by category, corrected for hypothesized sources of error.

Escapement location	Rec. year	Age group	Fish exam.	Estim. escap.	Number no pin	Number lost pin	Number no data	Known tags
Quins. R.	85	A	20843	29412	361	68	1	4837
Quins. R.	85	J	13531	14364	36	24	0	1666
Quins. R.	86	A	26980	36651	90	27	1	4413
Quins. R.	86	J	16919	18142	24	11	0	1118
Quins. R.	87	A	18253	26112	35	9	291	1582
Quins. R.	87	J	37571	39530	13	2	28	864
Quins. R.	88	A	31974	35190	10	4	95	716
Quins. R.	88	J	30252	30488	28	0	0	2776
Puntl. R.	85	A	37011	59266	31	11	0	1810
Puntl. R.	85	J	11611	18287	131	14	0	1679
Puntl. R.	86	A	10114	16068	97	4	0	1327
Puntl. R.	86	J	3189	3235	4	0	0	92
Puntl. R.	87	A	4890	7811	10	1	0	268
Puntl. R.	87	J	1159	1736	6	1	0	132
Puntl. R.	88	A	4654	5609	20	1	3	447
Puntl. R.	88	J	3906	5034	2	1	1	157
B. Qual. R.	85	A	19572	22182	7	1	188	140
B. Qual. R.	85	J	7023	7077	3	1	74	109
B. Qual. R.	86	A	15669	18950	6	0	33	201
B. Qual. R.	86	J	3449	4006	3	0	1	60
B. Qual. R.	87	A	6496	7793	5	1	64	114
B. Qual. R.	87	J	9393	9862	33	2	7	409
B. Qual. R.	88	A	4290	7038	12	1	4	146
B. Qual. R.	88	J	5210	5572	11	4	0	240
Nanaimo R.	86	J	26	157	0	0	0	1

Table 5.10. Estimates of the number of tagged adults from each population that escaped to selected streams from 1985 (top) 1988 (bottom). Release locations and stock types (if >1) are listed in first column. Recovery locations, and their approximate distances (in km) from the Chase R. are in columns 2-18. Figures accompanied by > sign are minimum estimates of tag escapements obtained from select samples. % Homing indicates the proportion of the total tag escapement that escaped to their stream of origin.

Rec. loc. km	Quatsese 300	Granite B. 143	Quins. 141	Oyster 120	Black 118	Little 101	Puntl. 99	Trent 93	Tsable 80	Coal 75	Waterloo 74	Rosewall 73	Chef 71	B. Qual. 59	L. Qual. 50	French 41	Millst. 8	Chase 0	Sum	% Homing
Quins. (p)			7472																7472	100.0
Quins. (c)			614																614	100.0
Puntl. (p)							3079	19											3098	99.4
Puntl. (c)							144												144	100.0
B. Qual. (p)														414					414	100.0
Quins. (p)			5388				2	4											5394	99.9
Quins. (c)			1301				4												1305	99.7
Black Cr.			2		824	>2													828	99.5
Puntl. (p)							2152	128					>3	7					2290	94.0
Puntl. (c)							164												164	100.0
Trent R.							6	64											72	88.9
B. Qual.														307					307	100.0
L. Qual.															99				99	100.0
Quins. (p)			1715										>2						1717	99.9
Quins. (c)			996																996	100.0
Quins. (w)	>10		118																128	92.2
Black Cr.				>19	558														577	96.7
Puntl. (p)							175	16											191	91.6
Puntl. (c)							113												113	100.0
Trent R.							6	158											173	91.3
Rosew. (b.c)							2	16				58	>2	21					99	58.6
Rosew. (t.r)												79	>2	25	2				172	45.9
Rosew. (l.q)										>1		34		4					39	87.2
B. Qual.														165					165	100.0
L. Qual. R.														4	245				249	98.4
French Cr.																51			58	87.9
Millstone																	27		29	93.1
Quins. (p)			892																892	100.0
Quins. (w)			95																95	100.0
Black Cr.					1541														1541	100.0
Puntl. (p)							382	7				1							390	97.9
Puntl. (c)							104												104	100.0
Trent R.							45	384											430	89.3
Rosew. (b.c)											>1	139		4	5				149	93.3
Rosew. (t.r)														2					154	98.7
Rosew. (l.q)							1					176		6					183	96.2
B. Qual.														276					279	98.9
L. Qual.															628				628	100.0
French Cr.																623			626	99.5
Millstone																	172		172	100.0

Table 5.11. Estimates of the number of tagged jacks from each group that escaped to selected streams from 1985 (top) to 1988 (bottom). Release locations and stock types are listed in first column. Recovery locations, and their approximate distances (in km) from the Chase R. are in columns 2-18. Figures accompanied by > sign are minimum estimates of tag escapements obtained from select samples. (\*) represent recoveries at Nanaimo River hatchery, adjacent to Chase River.

Rec. Loc. (km)	Quatsese 300	Granite B. 143	Quins. 141	Oyster 120	Black 118	Little 101	Puntl. 99	Trent 93	Tsable 80	Coal 75	Waterloo 74	Rosewall 73	Chef 71	B. Qual. 59	L. Qual. 50	French 41	Millst. 8	Chase 0	Sum	% Homing
Quins. (p)		>4	1687											2					1693	99.6
Quins. (c)		>2	305																307	99.3
Black Cr..					88														88	100.0
Puntl. (p)		>2					3109	1											3112	99.9
Puntl. (c)							15												15	100.0
Trent R.																			0	-
B. Qual.														203					203	100.0
L. Qual.								1							56				57	98.2
Quins. (p)			1128					2											1130	99.8
Quins. (c)			213																213	100.0
Quins. (w)			1																1	100.0
Black Cr.					268														268	100.0
Puntl. (p)							88												88	100.0
Puntl. (c)							3												3	100.0
Trent R.								6											7	85.7
Rosew. (b.c)												66							66	100.0
Rosew. (t.r)												30							30	100.0
Rosew. (l.q)												27							27	100.0
B. Qual.														80					80	100.0
L. Qual.															52				52	100.0
French Cr.																7			7	100.0
Millstone							1										7	6*	14	50.0
Quins. (p)			1018																1018	100.0
Quins. (w)			19																19	100.0
Black Cr.					417									1					418	99.8
Puntl. (p)							218												218	100.0
Puntl. (c)							5												5	100.0
Trent R.							2	39											42	92.9
Rosew. (b.c)												25							25	100.0
Rosew. (t.r)												13							18	72.2
Rosew. (l.q)												64							67	95.5
B. Qual.														494					494	100.0
L. Qual.														1	261				262	99.6
French Cr.												1			1	184			195	94.4
Millstone																	12		12	100.0
Quins. (p)			3072											2					3074	99.9
Black Cr.					266									6					272	97.8
Puntl. (p)							207												207	100.0
Puntl. (c)							17												17	100.0
Trent R.							2	144											146	98.6
B. Qual.														284					284	100.0
French Cr.														4		168			172	97.7
Millstone																	>2	>1	3	66.7

Table 5.12. Proportion (x100) of total tag escapement of each population that strayed to other streams (Top section), and average straying distances (in km, bottom section). All groups are abbreviated; production (p), colonization (c), wild (w), enhanced (e). Abbreviations for the Rosewall Creek production releases refer to the brood stocks used; Black Creek (bc), Trent River (tr), Little Qualicum river (lq). BY figures represent differences in straying between the two corresponding age classes (jacks - adults).

	Q.R.(p)	Q.R.(c)	Q.R.(w)	B.C.(w)	P.R.(p)	P.R.(c)	T.R.(e)	R.C.(bc)	R.C.(tr)	R.C.(lq)	B.Q.(p)	L.Q.(w)	F.C.(e)	M.R.(c)	Mean
<b>Adults</b>															
1985	0	0			0.6	0					0				0.1
1986	0.1	0.3		0.5	6	0	11.1				0	0			2.2
1987	0.1	0	7.8	3.3	8.4	0	8.7	41.4	54.1	12.8	0	1.6	12.1	6.9	11.2
1988	0		0	0	2.1	0	10.7	6.7	1.3	3.8	1.1	0	0.5	0	2.0
Mean	0.0	0.1	3.9	1.3	4.3	0.0	10.2	24.1	27.7	8.3	0.3	0.5	6.3	3.5	6.5
<b>Jacks</b>															
1985	0.4	0.7		0	0.1	0					0	1.8			0.4
1986	0.2	0	0	0	0	0	14.3	0	0	0	0	0	0	50	4.6
1987	0		0	0.2	0	0	7.1	0	27.8	4.5	0	0.4	5.6	0	3.5
1988	0.1			2.2	0	0	1.4				0		2.3	33.3	4.9
Mean	0.2	0.4	0.0	0.6	0.0	0.0	7.6	0.0	13.9	2.3	0.0	0.7	2.6	27.8	
<b>BY</b>															
85 rel.	0.3	0.4		-0.5	-5.9	0					0	1.8			-0.6
86 rel.	0.1	0	-7.8	-3.3	-8.4	0	5.6	-41.4	-54.1	-12.8	0	-1.6	-12.1	5.6	-9.3
87 rel.	0		0	0.2	-2.1	0	-3.6	-6.7	26.5	0.7	-1.1	0.4	5.1	0	1.5
Mean	0.1	0.2	-3.9	-1.2	-5.5	0.0	1.0	-24.1	-13.8	-6.0	-0.4	0.2	-3.5	2.8	
<b>Adults</b>															
1985					6										6.0
1986	46	42		20	8		13								25.8
1987	70		159	2	6		23	16	18	12		9	18	51	34.9
1988					9		6	17	14	16	9		9		11.4
Mean	58	42	159	11	7.25		14	16.5	16	14	9	9	13.5	51	
<b>Jacks</b>															
1985	29	2			31							43			26.3
1986	48						34							19.9	34.0
1987				59			15		14	14		4	19		20.8
1988	82			59			6						19	8	34.8
Mean	53.0	2.0		59.0	31.0		18.3		14.0	14.0		23.5	19.0	14.0	

Table 5.13. Estimates of the contribution of strays to each escapement based on expansion of CWT recoveries. The contributions are expressed as a fraction of the total escapement x 100.

Stream	Jacks				Adults			
	1985	1986	1987	1988	1985	1986	1987	1988
Quins. R.	0	0.03	0	0	0.03	0.02	0	0
Black Cr.	0	0	0	0	0	0	0	0
Puntl. R.	0	2.6	0.3	0.1	0	7.6	8.6	9.1
Trent R.	29.2	69.1	0	0	25.3	44.8	50.2	4.6
Rosew. Cr.	-	0	1.5	0	-	0	0	1.6
B. Qual. R.	1.0	0	0.3	0.3	0	0.9	1.6	0.2
L. Qual. R.	0	0	0.4	0	0	0	0.2	5.9
French Cr.	0	0	0	0	0	0	0	0
Millstone. R.	-	0	0	0	-	0	0	0

Table 5.14. Composition of various logistic models describing the homing proportion of coho stocks during the 1985-1988 period. The main factor and interactions included in each model (1-15) are listed with the corresponding degrees of freedom (DF) and deviance (G). The factors are year ( $x_1$ ), genetic ( $x_2$ ), treatment ( $x_3$ ), smolt weight ( $x_4$ ), smolt migration date ( $x_5$ ), stream latitude ( $x_6$ ), starting date of spawning run ( $x_7$ ), median date of spawning run ( $x_8$ ), spawner age ( $x_9$ ), exposure to foreign water sources ( $x_{10}$ ), total number of escapement recoveries ( $x_{11}$ ), and flow control ( $x_{12}$ ).

MODEL	MAIN FACTORS	INTERACTIONS	DF	G
0	Constant (k)	-	81	2202
1	k, $x_{10}$	-	80	1317
2	k, $x_{10}$ , $x_{11}$	-	79	1212
3	k, $x_{10}$ , $x_{11}$	$x_8 \times x_9$	78	1077
4	k, $x_{10}$ , $x_{11}$	$x_8 \times x_9$ , $x_1 \times x_{12}$	75	832
5	k, $x_{10}$ , $x_{11}$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$	68	503
6	k, $x_{10}$ , $x_{11}$ , $x_9$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_8$	67	463
7	k, $x_9$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$	62	397
8	k, $x_9$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$	59	369
9	k, $x_9$ , $x_1$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$	56	321
10	k, $x_9$ , $x_1$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$ , $x_6 \times x_8$	55	302
11	k, $x_9$ , $x_1$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$ , $x_6 \times x_8$ , $x_9 \times x_{12}$	54	290
12	k, $x_9$ , $x_1$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$ , $x_6 \times x_8$ , $x_9 \times x_{12}$ , $x_9 \times x_{10}$	53	274
13	k, $x_9$ , $x_1$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$ , $x_6 \times x_8$ , $x_9 \times x_{12}$ , $x_9 \times x_{10}$ , $x_{10} \times x_{11}$	52	263
14	k, $x_9$ , $x_1$	$x_8 \times x_9$ , $x_1 \times x_{12}$ , $x_2 \times x_8$ , $x_2 \times x_{10}$ , $x_1 \times x_8$ , $x_6 \times x_8$ , $x_9 \times x_{12}$ , $x_9 \times x_{10}$ , $x_{10} \times x_{11}$ , $x_5 \times x_6$	51	252

Table 5.15. Relative influence of various factors on homing rates predicted from model 9. Goodness-of-fit tests were performed in the presence and absence of each factor. The factor removed precedes the minus (-) sign. The difference in chi-square obtained after the removal of the factor is given under the Difference column. () = Degrees of freedom of factor omitted.

Model	Factor omitted	Log-likelihood	Chi-square	Difference	D.F	Prob.
Main + interac.	-	- 1992.80	320.911	-	56	0.0000
" " - x <sub>1</sub>	Year	- 2016.96	369.231	48.320	(3)	0.0000
" " - x <sub>9</sub>	Age	- 2001.77	338.851	17.940	(1)	0.0000
" " - x <sub>2</sub> *x <sub>10</sub>	Genetic * exposure	- 2088.74	512.801	191.89	(7)	0.0000
" " - x <sub>1</sub> *x <sub>8</sub>	year * run time	- 2017.56	370.441	49.53	(3)	0.0000
" " - x <sub>2</sub> *x <sub>8</sub>	genetic * run time	- 2015.38	366.071	45.16	(7)	0.0000
" " - x <sub>1</sub> *x <sub>12</sub>	year * flow control	- 2006.19	347.701	26.79	(3)	0.0000
" " - x <sub>8</sub> *x <sub>9</sub>	run time * age	- 2002.98	341.271	20.36	(1)	0.0000

Table 5.16. Fraction of the CWT releases associated with each stock that escaped as age 3+ adults during the 1986-1988 period. The year represents the adult return year.

Year	Qp	Qc	Qw	Bw	Pp	Pc	Te	R(B)	R(L)	R(T)	BQp	Lw	Fe	Mc	Mean
1986	.0244	.0599		.0341	.0233	.0123	.0080				.0016	.0051			.0211
1987	.0222	.0342	.0067	.0180	.0049	.0085	.0164	.0043	.0040	.0092	.0011	.0152	.0024	.0032	.0107
1988	.0211		.0039	.0428	.0067	.0089	.0271	.0077	.0078	.0096	.0023	.0306	.0254	.0211	.0165
Mean	.0226	.0471	.0053	.0316	.0116	.0099	.0172	.0060	.0059	.0094	.0017	.0170	.0139	.0122	

Table 6.1. Monthly estimates of awareness factors by statistical area, 1980-1984.  
 Bold values are overall estimate of awareness factors for the Strait of Georgia.

Year	area	Month												mean	
		1	2	3	4	5	6	7	8	9	10	11	12		
1980	13							0.156	0.163	0.193	0.183				0.237
1980	14							0.244	0.465	0.308	0.626				0.627
1980	15							0.239	0.183	0.025					0.246
1980	16							0.104	0.265	0.103	0.075				0.263
1980	17							0.211	0.443	0.095	0.153				1.079
1980	18														
1980	28							0.238	0.289	0.090	0.038				0.223
1980	29							0.012	0.032	0.070					0.024
1980	19A-B+ mean							0.164	0.211	0.167	0.325				<b>0.310</b>
1981	13				0.060	0.263	0.250	0.192	0.452						0.274
1981	14				0.195	0.385	0.326	0.253	0.219						0.314
1981	15					0.330	0.128	0.123	0.289						0.154
1981	16				0.059	0.149	0.167	0.260	0.131						0.184
1981	17				1.193	0.555	0.320	0.219	0.315						0.426
1981	18					0.060	0.015								0.300
1981	28						0.754	0.541	0.360						0.536
1981	29							0.058	0.050						0.061
1981	19A-B+ mean				0.216	0.326	0.246	0.232	0.220						<b>0.265</b>
1982	13					0.099	0.358	0.161	0.357	0.567					0.244
1982	14					0.233	0.638	0.287	0.433	0.271		0.061			0.356
1982	15						1.132	0.252	0.092						0.453
1982	16					0.403	0.733	0.599	0.162						0.420
1982	17					0.828	0.951	0.394	0.118	0.103					0.449
1982	18							0.381		0.139	0.129	0.250			0.398
1982	28						0.293	0.198	0.381						0.345
1982	29							0.007	0.037	0.098					0.086
1982	19A-B+ mean				0.016	0.308	0.551	0.254	0.263	0.107	0.223	0.133			0.252
					0.278	0.541	0.239	0.240	0.382	0.445	0.390				<b>0.311</b>
1983	13					0.153	0.215	0.120	0.202	0.168	0.277				0.171
1983	14				0.150	0.982	0.667	0.403	0.751	0.863	0.346				0.524
1983	15							0.336	0.698						0.624
1983	16				0.425	0.073	0.121	0.423	0.331						0.181
1983	17			0.447		3.462	0.795	0.496	1.012	0.127	0.082				0.764
1983	18									0.022					0.681
1983	28					0.205	0.267	0.519	0.312	0.958					0.380
1983	29							0.055	0.069	0.556					0.185
1983	19A-B+ mean	0.889	1.766	0.537	0.605		0.164	0.272	0.314	0.343	0.083	0.203			0.263
		0.349	2.523	0.470	1.364	0.252	0.272	0.324	0.366	0.279	0.373	8.725			<b>0.312</b>
1984	13					0.132	0.178	0.222	0.128	0.280					0.194
1984	14	0.028		0.258	0.081	0.782	0.459	0.409	0.240	0.680	1.014				0.423
1984	15					0.256	1.095								2.215
1984	16				0.286	0.275	0.321	0.247	0.547	0.839					0.330
1984	17	0.240		0.245	0.213	0.188	0.331	0.302	0.198	0.040					0.211
1984	18										0.209				1.883
1984	28							0.417	0.523	0.187					0.345
1984	29							0.007	0.000	0.001	0.009				0.053
1984	19A-B+ mean	0.808		0.397	0.343	0.047	0.207	0.270	0.205	1.020	0.147				0.264
		0.076		0.344	0.175	0.309	0.291	0.292	0.210	0.196	1.973				<b>0.270</b>

Table 6.2. Monthly estimates of awareness factors by statistical area, 1985-1988. Bold values are overall estimates of awareness factor for the Strait of Georgia.

Year	area	Month												mean		
		1	2	3	4	5	6	7	8	9	10	11	12			
1985	13					0.176	0.241	0.213	0.128	0.163	0.102					0.188
1985	14	0.355	0.203	0.313	0.288	0.402	0.377	0.302	0.235	0.125	0.552					0.304
1985	15						1.562	0.516	0.586							0.761
1985	16				2.777	0.429	0.531	0.511	0.362	0.392	0.374					0.476
1985	17			0.382	0.160	0.281	0.481	0.555	0.281	0.232						0.289
1985	18															
1985	28					0.662	1.249	0.430	0.284	0.415	0.109					0.361
1985	29						0.048	0.019	0.005	0.019						0.050
1985	19A-B+	0.267	0.275	0.175	0.215		0.379	0.352	0.279	0.132	0.181					0.254
	mean	0.469	0.265	0.246	0.214	0.293	0.334	0.298	0.177	0.167	0.365	4.000				<b>0.262</b>
1986	13					0.179	0.271	0.159	0.170	4.620	0.122					0.198
1986	14			0.104	0.185	0.184	0.413	0.326	0.136	0.136	0.975					0.241
1986	15						0.725	0.514	1.101	0.058						0.431
1986	16				0.643	0.239	0.224	0.238	0.152		0.417					0.225
1986	17	0.102		0.054	0.143	0.310	0.234	0.288	0.261	0.055						0.210
1986	18					0.088	0.215			0.003						0.014
1986	28					0.081	0.509	0.216	0.292	0.223	0.165					0.252
1986	29							0.104	0.021	0.004						0.055
1986	19A-B+	0.097	1.177	0.123	0.059	0.132	0.385	0.223	0.298	0.217	0.232	0.299	0.147			0.249
	mean	0.298	3.333	0.095	0.170	0.208	0.324	0.224	0.169	0.105	0.276	4.787				<b>0.211</b>
1987	13					0.141	0.012	0.154	0.112	0.093	0.152					0.129
1987	14	0.239	0.297	0.907	0.359	0.150	0.388	0.219	0.201	0.391	1.920	0.061				0.265
1987	15						0.546	0.633	1.264							0.753
1987	16				0.467	0.074	0.136	0.156	0.186	0.271	1.935					0.168
1987	17		0.137	0.125	0.210	0.140	0.418	0.192	0.184	0.178	0.192					0.206
1987	18															
1987	28					0.417		0.459	0.215	0.502						0.300
1987	29					0.003	0.133	0.008	0.007	0.020	1.162					0.061
1987	19A-B+	0.094	0.046	0.035	0.095		0.172	0.183	0.124	0.137	0.162		0.722			0.149
	mean	0.184	0.213	0.258	0.256	0.136	0.268	0.195	0.161	0.151	0.364	1.245				<b>0.197</b>
1988	13				0.157	0.124	0.147	0.124	0.093	0.095	0.224					0.127
1988	14	0.023	0.082	0.150	0.185	0.184	0.246	0.119	0.175	0.384	0.443		0.178			0.184
1988	15						0.247	0.171								0.262
1988	16			1.139	0.418	0.302	0.161	0.098	0.128	0.056						0.148
1988	17			0.071	0.188	0.143	0.183	0.153	0.130	0.295	0.032					0.156
1988	18															
1988	28					0.242	0.409	0.149	0.125	0.676						0.182
1988	29					0.004	0.002	0.028	0.003	0.226						0.041
1988	19A-B+	0.016	0.041	0.005	0.006		0.021	0.060	0.091	0.029	0.028					0.033
	mean	0.027	0.073	0.090	0.161	0.160	0.191	0.118	0.114	0.132	0.248	1.241				<b>0.149</b>

Table 6.3. Estimates of awareness factor by region/season/year stratum, 1980 - 1988.

Survey year	catch region	spring	Catch summer	fall	spring	A. Factor summer	fall	A. Factor (region)	Catch (region)
80	N	-	126000	30900	-	0.155	0.141	0.222	156900
80	CN	-	113800	20500	-	0.228	0.262	0.467	134300
80	CS	-	51800	13000	-	0.151	0.098	0.259	64800
80	S	-	26300	11450	-	-	-	-	37500
81	N	35900	43838	-	0.244	0.228	-	0.258	79738
81	CN	72500	67388	-	0.275	0.285	-	0.302	139888
81	CS	20700	39353	-	0.444	0.235	-	0.323	60053
81	S	14200	23212	-	0.026	-	-	0.060	37412
82	N	30099	90808	15894	0.353	0.196	0.642	0.250	136801
82	CN	94221	90388	12290	0.470	0.362	0.229	0.396	196899
82	CS	8810	29030	8479	1.704	0.138	0.543	0.285	46319
82	S	1740	11830	18097	0.261	0.323	0.244	0.334	31667
83	N	49527	60503	18568	0.189	0.162	0.210	0.177	128598
83	CN	108560	51360	21728	0.303	0.624	0.531	0.374	181648
83	CS	20936	14036	9478	0.874	0.320	0.389	0.493	44450
83	S	15631	27894	5810	0.314	0.350	0.126	0.311	49335
84	N	45425	92146	22251	0.147	0.209	0.315	0.190	159822
84	CN	73054	71067	14359	0.424	0.325	0.696	0.381	158480
84	CS	44907	21334	9554	0.218	0.250	0.132	0.209	75795
84	S	17081	26189	6223	0.219	0.259	0.361	0.251	49493
85	N	102735	147057	25164	0.252	0.186	0.162	0.208	274956
85	CN	137435	128550	28781	0.409	0.323	0.196	0.350	294766
85	CS	73256	40354	8738	0.265	0.187	0.276	0.228	122348
85	S	8544	18034	9549	0.247	0.254	0.145	0.226	36127
86	N	57387	148097	21561	0.262	0.184	0.322	0.214	227045
86	CN	105526	93057	16804	0.264	0.220	0.289	0.246	215387
86	CS	45810	31695	11487	0.208	0.176	0.095	0.170	88992
86	S	9283	14579	16694	0.288	0.199	0.183	0.231	40556
87	N	60366	129738	19339	0.146	0.169	0.102	0.160	209443
87	CN	121325	128883	12476	0.262	0.197	0.434	0.236	262684
87	CS	43983	46551	12225	0.225	0.144	0.161	0.182	102759
87	S	10332	26801	29553	0.107	0.154	0.145	0.144	66686
88	N	155546	156799	17113	0.150	0.121	0.116	0.133	329458
88	CN	293928	186474	14438	0.206	0.123	0.218	0.175	494840
88	CS	121268	44277	5674	0.147	0.071	0.496	0.128	171219
88	S	35818	38230	15225	0.019	0.080	0.032	0.043	89273

Table 6.4. Summary of catch and sampling statistics for the coho sport fishery in the Strait of Georgia, 1980 - 1988.

Survey Year	Tagged fish sampled	Total fish sampled	Tagged proportion (sample)	Fraction catch sampled	Total effort	Total catch	Total tags caught	Total heads submitted	Awareness factor
1980	1208	20484	0.059	0.052	510150	393750	23221	6665	0.287
1981	1305	15136	0.086	0.048	494604	317091	27339	7233	0.265
1982	351	10737	0.033	0.026	559395	411686	13458	4657	0.346
1983	563	13300	0.042	0.033	574257	404031	17103	5702	0.333
1984	671	19443	0.035	0.044	651090	443590	15309	4152	0.271
1985	2259	38192	0.059	0.052	628513	728197	43072	11682	0.271
1986	1526	27646	0.055	0.048	582946	571980	31572	6949	0.220
1987	1949	28757	0.068	0.045	589731	641572	43482	8752	0.201
1988	2238	46553	0.048	0.043	664517	1084790	52150	7489	0.144

Table 6.5. Corrected estimates of awareness factor by region/season/year stratum, 1983-1988.

Survey year	Catch region	spring	Catch summer	fall	spring	A. Factor summer	fall	A. Factor (region)	Catch (region)
83	N	49527	60503	18568	0.185	0.162	0.205	0.174	128598
83	CN	108560	51360	21728	0.303	0.612	0.518	0.374	181648
83	CS	20936	14036	9478	0.853	0.312	0.378	0.483	44450
83	S	15631	27894	5810	0.306	0.342	0.123	0.304	49335
84	N	45425	92146	22251	0.144	0.209	0.307	0.190	159822
84	CN	73054	71067	14359	0.424	0.325	0.678	0.375	158480
84	CS	44907	21334	9554	0.213	0.244	0.128	0.209	75795
84	S	17081	26189	6223	0.213	0.253	0.350	0.246	49493
85	N	102735	147057	25164	0.252	0.186	0.158	0.208	274956
85	CN	137435	128550	28781	0.409	0.323	0.191	0.350	294766
85	CS	73256	40354	8738	0.265	0.182	0.268	0.228	122348
85	S	8544	18034	9549	0.240	0.248	0.141	0.221	36127
86	N	57387	148097	21561	0.257	0.184	0.314	0.214	227045
86	CN	105526	93057	16804	0.264	0.220	0.282	0.246	215387
86	CS	45810	31695	11487	0.204	0.172	0.092	0.170	88992
86	S	9283	14579	16694	0.280	0.194	0.178	0.226	40556
87	N	60366	129738	19339	0.146	0.169	0.099	0.160	209443
87	CN	121325	128883	12476	0.262	0.197	0.422	0.236	262684
87	CS	43983	46551	12225	0.220	0.141	0.157	0.182	102759
87	S	10332	26801	29553	0.104	0.150	0.142	0.144	66686
88	N	155546	156799	17113	0.150	0.121	0.113	0.133	329458
88	CN	293928	186474	14438	0.206	0.123	0.212	0.175	494840
88	CS	121268	44277	5674	0.147	0.069	0.481	0.128	171219
88	S	35818	38230	15225	0.019	0.079	0.031	0.043	89273

Table 6.6. Estimates of the total number of tagged coho of each group recovered as 3 year old adults in various commercial and sport fisheries each year. GS.TR, GS.SPT and GS.net include catches in north and south sections.

Year	Group	A. SPT	N. SPT	C. SPT	W. SPT	GS. SPT	WA. SPT	A. TR	N. TR	NC. TR	SC. TR	NW. TR	SW. TR	GS. TR	WA. TR	A. Net	N. Net	C. Net	NWV Net	SWV Net	JS Net	GS Net	JF. Net	WA. Net
1986	Quins. C.	0	0	10	0	850	9	11	22	11	1036	278	134	159	4	0	4	26	0	0	732	16	33	40
	Quins. P.	0	0	64	2	5185	49	10	37	55	2829	626	471	1007	46	0	0	82	12	0	2348	48	118	89
	Black Cr.	0	0	7	1	469	9	2	16	24	664	313	164	131	3	0	0	15	0	0	323	6	30	15
	Puntl. C.	0	0	0	0	223	6	0	0	6	73	25	79	64	0	0	0	3	0	0	36	23	18	3
	Puntl. P.	0	0	16	1	1772	85	0	29	62	1963	590	685	779	16	0	14	15	8	2	1100	553	128	118
	Trent R.	0	0	0	0	242	4	0	0	0	53	31	35	64	0	0	0	0	0	0	30	17	5	3
	B. Qual.	0	0	0	0	327	25	0	0	0	71	57	151	122	10	0	0	0	0	0	61	45	38	43
	L.Qual	0	0	0	0	319	0	0	0	0	67	37	146	71	0	0	0	2	0	0	27	108	28	14
1987	Quins. W.	0	0	2	0	326	0	5	0	0	58	60	16	36	5	0	0	5	0	0	51	0	3	0
	Quins. C.	0	0	5	1	1106	4	8	73	34	329	224	93	230	0	0	0	18	0	0	286	0	13	52
	Quins. P.	0	0	10	2	3043	17	0	25	42	377	265	166	857	0	0	6	20	0	0	376	11	48	47
	Black Cr.	0	0	2	1	1293	11	9	98	26	381	391	136	477	0	0	9	17	0	0	277	19	23	40
	Puntl. C.	0	0	0	0	288	0	0	3	7	22	61	61	106	0	0	0	4	0	0	40	45	12	11
	Puntl. P.	0	0	2	2	390	13	0	42	0	105	98	57	155	0	0	3	0	0	0	104	61	18	8
	Trent R.	0	0	0	1	458	9	0	5	0	69	45	53	196	0	0	4	5	0	0	54	79	6	6
	Rosew. BC	0	0	4	0	679	0	0	56	22	166	137	37	196	0	0	3	0	0	0	121	172	16	22
	Rosew. LQ	0	0	1	0	310	0	0	10	0	15	29	42	99	0	0	3	0	0	0	40	37	0	7
	Rosew. TR	0	0	2	1	841	18	2	5	0	82	70	47	185	0	0	0	13	0	0	81	217	27	4
	B. Qual.	0	0	2	0	201	0	0	0	0	7	32	54	83	2	0	0	0	0	0	28	20	19	7
	L. Qual.	0	0	0	0	504	25	0	5	4	25	49	124	134	4	0	0	0	0	0	24	91	16	25
	French Cr.	0	0	0	0	164	3	0	0	0	2	37	20	40	2	0	3	0	0	0	45	16	33	13
	Millstone	0	0	3	0	347	0	0	9	0	50	118	219	65	0	0	7	4	0	0	53	8	31	0
1988	Quins. W.	0	0	2	0	283	0	0	0	0	42	59	22	57	0	0	0	0	0	0	91	0	3	0
	Quins. P.	0	0	8	0	1416	0	0	18	19	229	288	48	207	0	0	0	26	0	0	392	6	9	0
	Black Cr.	0	0	8	0	1595	0	6	42	47	443	597	164	394	0	6	0	33	0	0	375	7	4	0
	Puntl. C.	0	0	0	0	358	0	0	0	0	33	39	10	80	0	0	0	0	0	0	39	3	2	0
	Puntl. P.	0	0	6	0	987	0	0	4	4	91	181	90	317	0	0	0	19	0	0	182	7	4	4
	Trent R.	0	0	8	0	1071	0	0	5	5	97	159	60	308	0	0	0	0	0	0	265	14	6	0
	Rosew. BC	0	0	0	0	580	0	0	5	3	95	158	90	60	0	0	0	6	0	2	132	45	6	0
	Rosew. LQ	0	0	10	0	1042	0	0	8	4	114	108	68	215	0	0	6	3	0	1	264	46	3	0
	Rosew. TR	0	0	4	0	670	0	0	5	2	68	117	52	112	0	5	0	8	0	0	151	45	3	0
	B. Qual.	0	0	2	0	667	0	0	0	2	80	56	31	149	0	4	0	5	0	0	127	44	10	0
	L. Qual.	0	0	2	0	683	0	0	0	5	31	57	25	81	0	0	0	4	0	1	48	34	7	0
	French Cr	0	0	0	0	891	0	0	0	3	21	26	92	189	0	0	0	5	0	1	55	25	7	0
	Millstone	0	0	3	1	471	0	0	5	0	53	180	85	74	0	0	0	4	0	1	85	32	3	0

Table 6.7. Recovery patterns of tagged coho from each population. Number of tags released not corrected for tag loss. Numbers in 2+ and 4+ categories represent the total return of tagged jacks and age 4+ adults. Numbers in the remaining categories represent the total number of recoveries of age 3+ adults in various fishing regions and escapements (Esc.). Northern and southern sections within catch regions are represented by (n) and (s).

Return year Stock	Tag rel.	2+	North	Centr.	JS.N	WVI (n)	GS.TR	GS.SP (n)	GS.SP (s)	WVI (s)	South	GS.N	Esc.	4+
1986														
Quins. C.	21786	320	37	1083	732	278	159	810	40	134	86	16	1305	34
Quins. P.	220929	1715	47	3030	2348	638	1007	4911	274	473	302	48	5394	5
Black Cr.	24256	99	18	710	323	313	131	423	46	165	57	6	828	11
Puntl. C.	13326	21	0	82	36	25	64	206	17	79	27	23	164	3
Puntl. P.	98197	3133	43	2056	1100	598	779	1632	140	688	347	553	2290	0
Trent R.	8985	0	0	53	30	31	64	215	27	35	12	17	72	0
B. Qual.	191620	209	0	71	61	57	122	295	32	151	116	45	307	6
L. Qual.	19273	57	0	69	27	37	71	287	32	146	42	108	99	0
1987														
Quins. W.	19058	1	5	65	51	60	36	321	5	15	8	0	128	0
Quins. C.	29164	214	81	386	286	224	230	1049	57	94	69	0	996	0
Quins. P.	77380	1143	31	449	376	265	857	2894	149	168	112	11	1717	8
Black Cr.	32000	308	116	426	277	391	477	1262	31	137	74	19	577	5
Puntl. C.	13298	3	3	33	40	61	106	279	9	61	23	45	113	17
Puntl. P.	39124	88	45	107	104	98	155	342	48	59	39	61	191	7
Trent R.	10564	11	9	74	54	45	196	433	25	54	21	79	173	2
Rosew. BC	22863	66	59	192	121	137	196	601	78	37	39	172	99	0
Rosew. LQ	9848	27	13	16	40	29	99	283	27	42	7	37	39	0
Rosew. TR	18674	41	7	97	81	70	185	747	94	48	49	217	172	0
B. Qual.	152273	80	0	9	28	32	183	182	19	54	28	20	165	7
L. Qual.	16356	52	5	29	24	49	134	417	87	124	70	91	249	0
French Cr.	24000	7	3	2	45	37	40	110	54	20	51	16	58	0
Millstone	9115	14	16	57	53	118	65	202	145	219	31	8	29	9
1988														
Quins. W.	24339	19	0	44	91	59	57	275	7	22	3	0	95	0
Quins. P.	42182	1034	18	282	392	288	207	1355	61	48	9	6	892	0
Black Cr.	36000	471	54	531	375	597	394	1497	98	164	4	7	1541	0
Puntl. C.	11698	5	0	33	39	39	80	344	14	10	2	3	104	0
Puntl. P.	58145	220	4	120	182	181	317	905	82	90	8	7	390	0
Trent R.	15849	51	5	110	265	159	308	992	79	60	6	14	430	0
Rosew. BC	19425	37	5	104	132	158	60	500	80	92	6	45	149	0
Rosew. LQ	23583	67	14	131	264	108	215	970	72	69	3	46	183	7
Rosew. TR	16050	30	10	82	151	117	112	604	66	52	3	45	154	0
B. Qual.	119424	502	4	89	127	56	149	584	83	31	10	44	279	28
L. Qual.	20512	267	0	42	48	58	81	613	70	25	7	34	628	0
French Cr	24600	195	0	29	55	26	189	733	158	93	7	25	626	0
Millstone	8158	32	5	60	85	180	74	382	89	87	3	32	172	5

Table 6.8. Composition of logistic models describing the survival from smolt-to-spawner for the 1986-1988 adult return years. The main factor and interactions included in each model are listed with the corresponding degrees of freedom (DF) and deviance (G). The factors are year ( $x_1$ ), genetic ( $x_2$ ), stream latitude ( $x_3$ ), treatment ( $x_4$ ), smolt weight ( $x_5$ ), smolt migration date ( $x_6$ ), fraction of catch in outside waters ( $x_7$ ), relative contribution of jacks to total returns ( $x_8$ ), starting date of the adult run ( $x_9$ ), total returns of tagged fish ( $x_{10}$ ).

MODEL	MAIN FACTORS	INTERACTIONS	DF	G
0	Constant (k)	-	34	75879
1	k, $x_3$	-	33	54345
2	k, $x_3$ , $x_4$	-	32	44521
3	k, $x_3$ , $x_4$ , $x_{10}$	-	31	39097
4	k, $x_3$ , $x_4$ , $x_{10}$ , $x_2$	-	24	15185
5	k, $x_3$ , $x_4$ , $x_{10}$ , $x_2$ , $x_5$	-	23	14148
6	k, $x_3$ , $x_4$ , $x_{10}$ , $x_2$ , $x_5$ , $x_7$	-	22	11574
7	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$	$x_5 * x_6$	22	10620
8	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$	$x_5 * x_6$ , $x_2 * x_5$	15	6035
9	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$	$x_5 * x_6$ , $x_2 * x_5$ , $x_1 * x_6$	13	4028
10	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$ , $x_1$	$x_5 * x_6$ , $x_2 * x_5$ , $x_1 * x_6$	11	2800
11	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$ , $x_1$ , $x_{10}$	$x_5 * x_6$ , $x_2 * x_5$ , $x_1 * x_6$	10	1176
12	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$ , $x_1$ , $x_{10}$	$x_5 * x_6$ , $x_2 * x_5$ , $x_1 * x_6$ , $x_3 * x_5$	9	530
13	k, $x_3$ , $x_4$ , $x_2$ , $x_5$ , $x_7$ , $x_1$ , $x_{10}$ , $x_9$	$x_5 * x_6$ , $x_2 * x_5$ , $x_1 * x_6$ , $x_3 * x_5$	8	422

Table 6.9. Relative influence of various factors in model 11 upon the goodness of fit. Goodness-of-fit tests were performed in the presence and absence of each factor. The factor removed precedes the minus (-) sign. The difference in chi-square obtained after the removal of the factor is given under the Difference column. () = Degrees of freedom associated with factor omitted.

Model	Factor omitted	Log-likelihood	Chi-square	Difference	D.F	Prob.
Main + interac.	-	- 345827.81	1176.656	-	10	0.0000
" " - $x_2$	Genetic	- 350350.00	10330.036	9153.380	(7)	0.0000
" " - $x_1$	Year	- 347163.00	3956.036	2779.380	(2)	0.0000
" " - $x_{10}$	Total returns	- 346635.63	2901.286	1724.630	(1)	0.0000
" " - $x_5$	Smolt weight	- 346392.25	2414.536	1237.880	(1)	0.0000
" " - $x_7$	Proportion outside	- 346155.75	1941.536	764.880	(1)	0.0000
" " - $x_3$	Stream latitude	- 345958.13	1546.286	369.630	(1)	0.0000
" " - $x_4$	Treatment	- 345856.88	1343.786	167.130	(1)	0.0000
" " - $x_1 * x_6$	Year * migr. date	- 349524.81	8679.656	7503.000	(2)	0.0000
" " - $x_2 * x_5$	Genetic * weight	- 346106.44	1842.906	666.250	(7)	0.0000
" " - $x_5 * x_6$	Weight * mig. date	- 345827.81	1285.656	109.000	(1)	0.0000

Table 7.1. Fishing periods and estimates of total fishing effort in the major B.C. commercial fisheries harvesting coho salmon. Gill net and seine net statistics represented by GN and SN. GS.N\* represents net fishery in area 14 only. Data source for the Juan de Fuca and Johnstone Strait net fisheries: DFO Fisheries Operations Branch, South Coast Division, Nanaimo, B.C. Data source for the troll fisheries: DFO's ISCMP and HSCDS databases (Wong 1983).

Catch region	Fishing season	First opening wk-m	Last opening wk-m	Fishing days (GN)	Fishing days (SN)	Boat days (GN)	Boat days (SN)	Combined boat days (GN+SN)	Coho catch (GN+SN)
JS.N	1985	3-Jul	2-Oct	54	42	2155	1864	4019	145690
"	1986	1-Aug	4-Oct	40	26	2677	2366	5043	123059
"	1987	1-Aug	4-Sep	26	16	1868	1600	3468	57916
"	1988	5-Jul	4-Oct	28	20	2576	1655	4231	82663
JF.N	1985	4-Jul	2-Sep	16	16	664	1046	1710	224735
"	1986	1-Aug	1-Sep	8	8	723	636	1359	202501
"	1987	1-Aug	1-Oct	7	7	438	563	1001	216400
"	1988	2-Aug	4-Oct	8	7	413	405	828	56715
GS.N*	1985	2-Oct	3-Nov	10	1	1425	160	1585	22128
"	1986	2-Oct	1-Nov	11	0.4	1256	260	1516	14015
"	1987	3-Oct	1-Nov	4	4	1715	668	2383	7147
"	1988	3-Oct	4-Oct	2.5	0	1022	0	1022	2228

Catch region	Fishing season	First opening wk-m	Last opening wk-m	Fishing days	Total effort (deliveries)	Coho catch
N.TR	1986	3-Jun	1-Sep	78	26291	1089486
"	1987	1-Jul	2-Sep	70	30555	595735
"	1988	1-Jul	2-Sep	60	23327	348038
NC.TR	1986	3-Jun	1-Sep	78	5571	163295
"	1987	1-Jul	2-Sep	70	4423	73441
"	1988	1-Jul	2-Sep	60	3027	38493
SC.TR	1986	3-Jun	1-Sep	78	8904	429891
"	1987	1-Jul	2-Sep	70	5200	141050
"	1988	1-Jul	2-Sep	69	4901	145362
NW.TR	1986	3-Jun	5-Aug	72	24981	610503
"	1987	1-Jul	3-Aug	54	13149	525107
"	1988	1-Jul	2-Sep	69	17993	555914
SW.TR	1986	3-Jun	5-Aug	72	28326	1553560
"	1987	1-Jul	3-Aug	54	22047	1295914
"	1988	1-Jul	2-Sep	69	28962	1039729
GS.TR	1986	3-Jun	2-Oct	113	7802	181421
"	1987	3-Jun	5-Sep	92	8415	217537
"	1988	1-Jul	5-Sep	92	7962	256481

Table 7.2. Results of comparisons of catch distributions using Kolmogorov-Smirnov two sample test. All figures correspond to the probability levels associated with  $D_{\max}$  values obtained from pair-wise comparisons of cumulative frequency distributions of contribution estimates. Boxes identify stock assemblages with similar catch distributions.

1986	Quins. (p)	Quins. (c)	Black (w)	Puntl. (p)	Puntl. (c)	Trent (e)	B. Qual. (p)	L. Qual. (w)
Quins. (p)	-							
Quins. (c)	0.116	-						
Black (w)	0.986	0.986	-					
Puntl. (p)	0.111	0.240	0.240	-				
Puntl. (c)	0.000	0.000	0.016	0.016	-			
Trent (e)	0.000	0.000	0.001	0.988	0.988	-		
B. Qual. (p)	0.000	0.000	0.002	0.468	0.200	0.200	-	
L. Qual. (w)	0.000	0.000	0.000	0.263	0.099	0.805	0.805	-

1987	Quins. (p)	Quins. (c)	Quins. (w)	Black (w)	Puntl. (p)	Puntl. (c)	Trent (e)	Rosew. (bc)	Rosew. (lq)	Rosew. (tr)	B. Qual. (p)	L. Qual. (w)	French (c)
Quins. (p)	-												
Quins. (c)	0.238	-											
Quins. (w)	0.722	0.722	-										
Black Cr.	0.946	0.840	0.840	-									
Puntl. (p)	0.224	0.082	0.292	0.292	-								
Puntl. (c)	0.005	0.032	0.052	0.285	0.285	-							
Trent R.	0.001	0.031	0.011	0.163	1.000	1.000	-						
Rosew. (bc)	0.181	0.069	0.267	0.818	0.813	0.404	0.404	-					
Rosew. (lq)	0.001	0.038	0.015	0.113	1.000	1.000	0.468	0.468	-				
Rosew. (tr)	0.000	0.002	0.000	0.005	0.623	0.626	0.175	0.923	0.923	-			
B. Qual.	0.000	0.007	0.006	0.104	1.000	1.000	0.279	0.998	0.555	0.555	-		
L. Qual.	0.000	0.000	0.000	0.002	0.368	0.321	0.026	0.272	0.988	0.379	0.379	-	
French Cr.	0.005	0.000	0.000	0.144	0.312	0.196	0.173	0.173	0.581	0.632	0.581	0.581	-
Millst. R.	0.275	0.275	0.000	0.118	0.264	0.163	0.141	0.142	0.153	0.560	0.077	0.275	0.275

1988	Quins. (p)	Quins. (w)	Black (w)	Puntl. (p)	Puntl. (c)	Trent (e)	Rosew. (bc)	Rosew. (lq)	Rosew. (tr)	B. Qual. (p)	L. Qual. (w)	French (c)
Quins. (p)	-											
Quins. (w)	1.000	-										
Black Cr.	0.988	0.988	-									
Puntl. (p)	0.182	0.037	0.037	-								
Puntl. (c)	0.006	0.001	0.766	0.766	-							
Trent R.	0.415	0.110	1.000	0.510	0.510	-						
Rosew. (bc)	0.367	0.120	0.650	0.054	0.691	0.691	-					
Rosew. (lq)	0.754	0.274	0.593	0.173	0.970	0.899	0.899	-				
Rosew. (tr)	0.692	0.179	0.981	0.267	0.974	1.000	1.000	1.000	-			
B. Qual.	0.323	0.078	0.935	0.441	0.996	0.859	1.000	1.000	1.000	-		
L. Qual.	0.005	0.000	0.057	0.755	0.039	0.044	0.082	0.179	0.281	0.281	-	
French Cr.	0.000	0.000	0.040	0.485	0.017	0.000	0.004	0.005	0.028	0.749	0.749	-
Millst. R.	0.208	0.295	0.529	0.037	0.460	1.000	0.689	0.994	0.754	0.030	0.000	1.000

Table 7.3. Composition of various logistic models describing the exploitation levels of age 3+ adults from all stocks for the 1986-1988 period. The main factor and interactions included in each model (1-11) are given with the corresponding degrees of freedom (DF) and deviance (G). The factors are year (a), genetic (b), treatment (c), smolt weight (d), smolt migration date (e), stream latitude (g), starting date of adult run (h), and adult length (j).

MODEL	MAIN FACTORS	INTERACTIONS	DF	G
0	Constant (k)	-	34	3180
1	k, x <sub>10</sub>	-	33	2357
2	k, x <sub>10</sub>	x <sub>7</sub> *x <sub>4</sub>	32	2016
3	k, x <sub>10</sub> , x <sub>8</sub>	x <sub>7</sub> *x <sub>4</sub>	31	1850
4	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub>	x <sub>7</sub> *x <sub>4</sub>	30	1626
5	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub> , x <sub>2</sub>	x <sub>7</sub> *x <sub>4</sub>	23	802
6	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub> , x <sub>2</sub>	x <sub>7</sub> *x <sub>4</sub> , x <sub>7</sub> *x <sub>8</sub>	22	680
7	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub> , x <sub>2</sub> , x <sub>7</sub>	x <sub>7</sub> *x <sub>4</sub> , x <sub>7</sub> *x <sub>8</sub>	21	499
8	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub> , x <sub>2</sub> , x <sub>7</sub>	x <sub>7</sub> *x <sub>4</sub> , x <sub>7</sub> *x <sub>8</sub> , x <sub>2</sub> *x <sub>5</sub>	14	213
9	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub> , x <sub>2</sub> , x <sub>7</sub> , x <sub>4</sub>	x <sub>7</sub> *x <sub>4</sub> , x <sub>7</sub> *x <sub>8</sub> , x <sub>2</sub> *x <sub>5</sub>	13	152
10	k, x <sub>10</sub> , x <sub>8</sub> , x <sub>3</sub> , x <sub>2</sub> , x <sub>7</sub> , x <sub>4</sub> , x <sub>1</sub>	x <sub>7</sub> *x <sub>4</sub> , x <sub>7</sub> *x <sub>8</sub> , x <sub>2</sub> *x <sub>5</sub>	12	135
11	k, x <sub>8</sub> , x <sub>4</sub> , x <sub>2</sub> , x <sub>7</sub> , x <sub>1</sub>	x <sub>7</sub> *x <sub>4</sub> , x <sub>7</sub> *x <sub>8</sub> , x <sub>2</sub> *x <sub>5</sub> , x <sub>3</sub> *x <sub>4</sub>	13	127

Table 7.4. Relative influence of various factors in model 11 upon the goodness of fit. Goodness-of-fit tests were performed in the presence and absence of each factor. The factor removed precedes the minus (-) sign. The difference in chi-square obtained after the removal of the factor is given under the Difference column. () = Degrees of freedom associated with factor omitted.

Model	Factor omitted	Log-likelihood	Chi-square	Difference	D.F	Prob.
Main + interac.	-	- 47466.90	127.400	-	-	0.0000
" " - x <sub>2</sub>	Genetic	- 47718.63	631.800	504.400	(7)	0.0000
" " - x <sub>8</sub>	Run timing	- 47689.29	573.120	445.720	(1)	0.0000
" " - x <sub>7</sub>	Latitude	- 47597.10	388.740	261.340	(1)	0.0000
" " - x <sub>4</sub>	Smolt weight	- 47498.97	192.470	65.070	(1)	0.0000
" " - x <sub>1</sub>	Year	- 47475.61	145.740	18.340	(1)	0.0000
" " - x <sub>2</sub> *x <sub>5</sub>	genet. * migr. date	- 47724.55	643.630	516.230	(7)	0.0000
" " - x <sub>7</sub> *x <sub>8</sub>	lat. * run timing	- 47619.11	432.760	305.360	(1)	0.0000
" " - x <sub>7</sub> *x <sub>4</sub>	latitude * s. weight	- 47486.04	166.620	39.220	(1)	0.0000
" " - x <sub>3</sub> *x <sub>4</sub>	treatm. * s. weight	- 47485.61	165.760	38.360	(1)	0.0000

Table 7.5. Estimated regressions coefficients associated with each factor included in the logistic model of exploitation (Table 7.4). Values within parentheses represent the rank of the coefficients associated with the dummy variables (genetic factor).

Factor	Coefficient	Standard error	Coeff / S.E	exp(coeff)
x <sub>1</sub>	- 0.0673	0.01572	- 4.29	0.9359
x <sub>2</sub> (1)	12.3830	0.7085	17.48	238709
x <sub>2</sub> (2)	- 1.0971	0.7133	- 1.54	0.3338
x <sub>2</sub> (3)	0.3469	1.0110	0.34	1.4150
x <sub>2</sub> (4)	0.9798	1.5190	0.64	2.6640
x <sub>2</sub> (5)	9.2062	0.8704	10.58	9958
x <sub>2</sub> (6)	- 5.9696	1.8210	- 3.28	0.0026
x <sub>2</sub> (7)	- 8.5183	1.9710	- 4.32	0.0002
x <sub>4</sub>	0.1152	0.0144	7.99	1.1220
x <sub>7</sub>	- 0.3128	0.0195	- 16.08	0.7314
x <sub>8</sub>	- 0.1163	0.0056	- 20.78	0.8902
x <sub>5</sub> *x <sub>2</sub> (1)	- 0.0933	0.0053	-17.72	0.9108
x <sub>5</sub> *x <sub>2</sub> (2)	0.0058	0.0049	1.18	1.0060
x <sub>5</sub> *x <sub>2</sub> (3)	- 0.0040	0.0072	- 0.56	0.9960
x <sub>5</sub> *x <sub>2</sub> (4)	0.0121	0.0114	- 1.06	0.9879
x <sub>5</sub> *x <sub>2</sub> (5)	- 0.0711	0.0063	- 11.35	0.9313
x <sub>5</sub> *x <sub>2</sub> (6)	0.0544	0.0133	4.08	1.0560
x <sub>5</sub> *x <sub>2</sub> (7)	0.0660	0.0148	4.45	1.0680
x <sub>7</sub> *x <sub>8</sub>	0.0011	0.0000	17.37	1.0010
x <sub>7</sub> *x <sub>4</sub>	- 0.0007	0.0001	- 6.21	0.9993
x <sub>3</sub> *x <sub>4</sub>	- 0.0012	0.0002	- 6.14	0.9988
K	31.7320	1.7080	18.57	6.0E+13

**FIGURES**

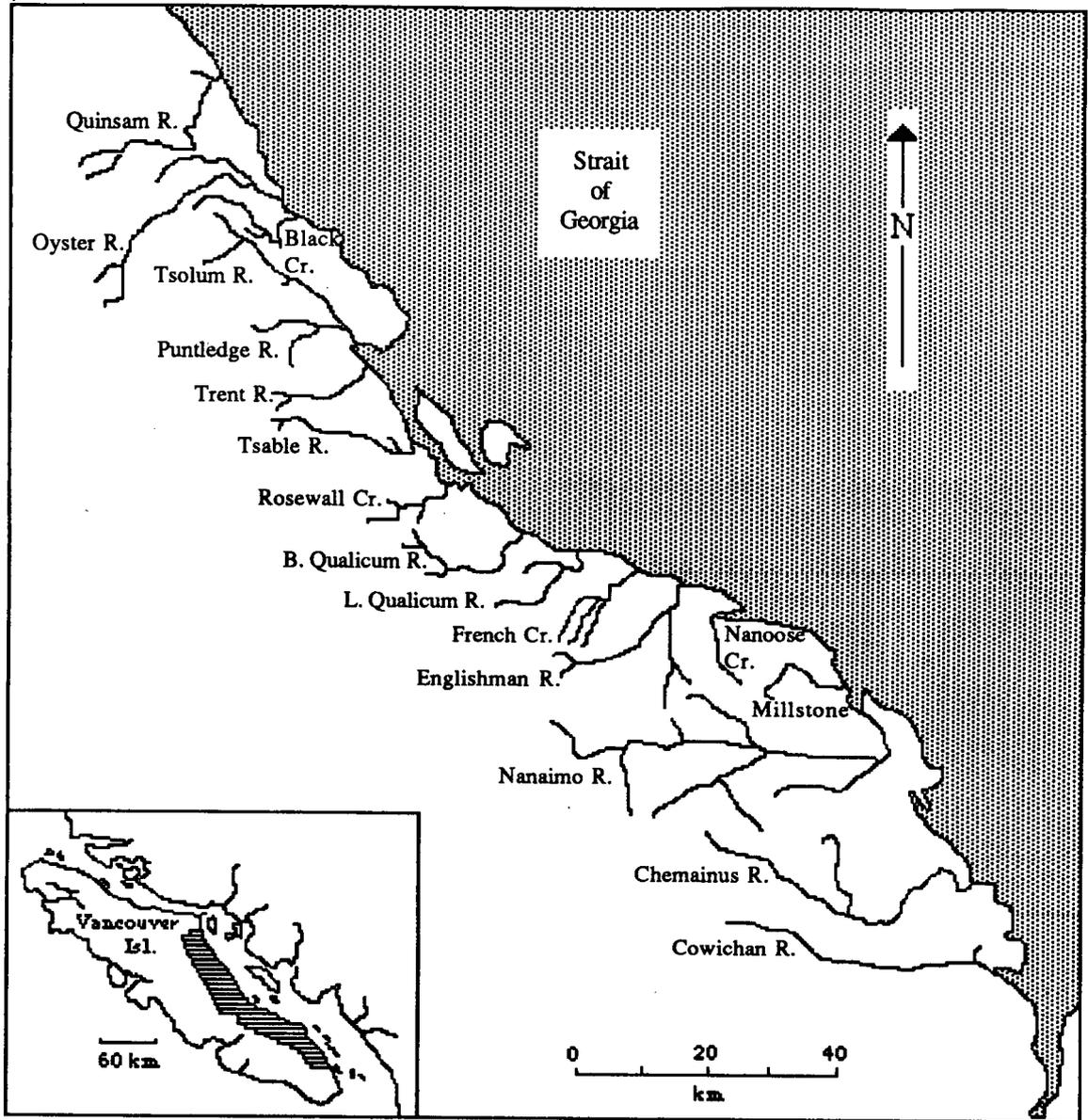


Figure 1.1. Geographical location of study site

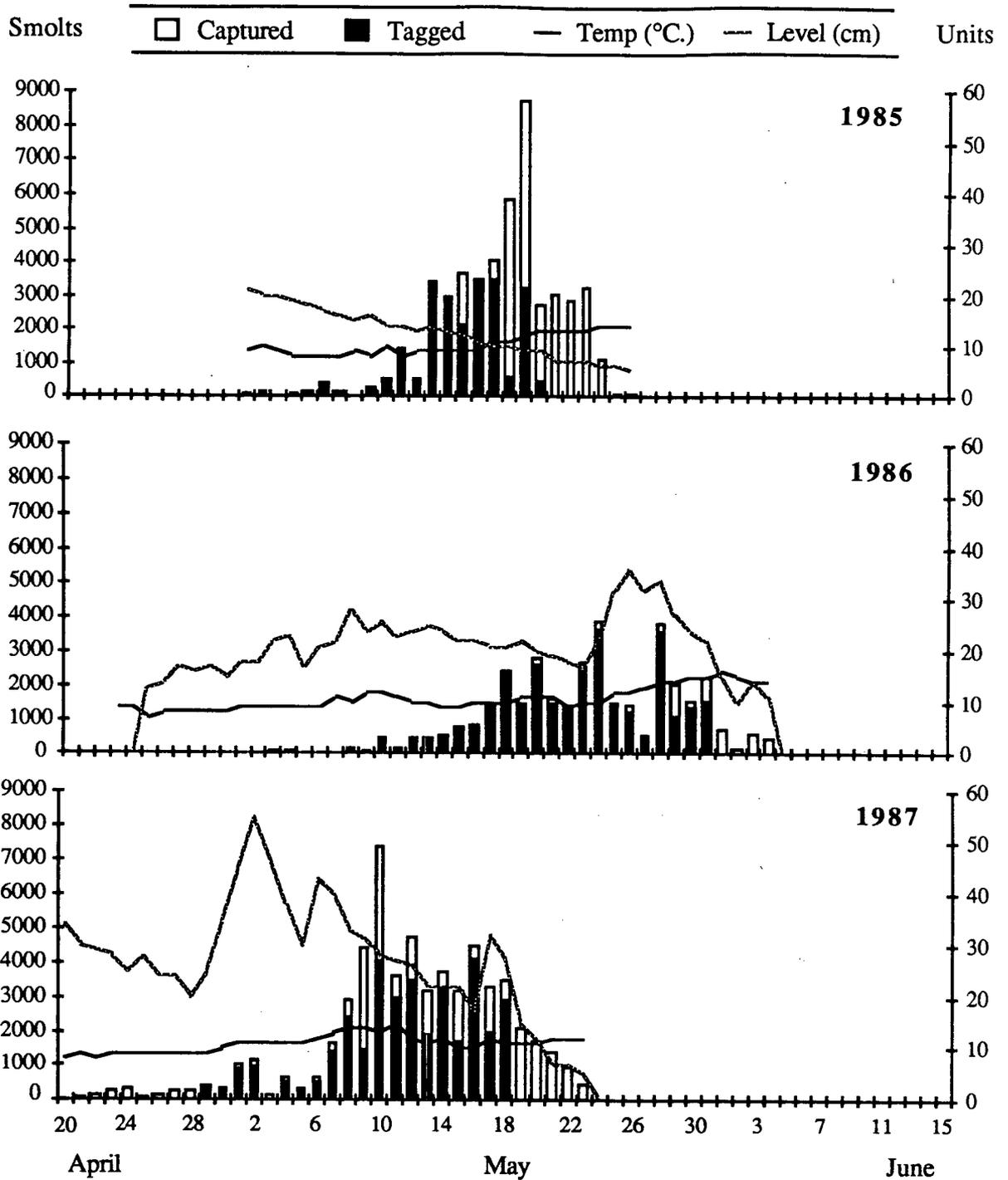


Figure 3.1. Smolt outmigration patterns and hydrological conditions at Black Creek, 1985 to 1987. The clear region on some vertical bars correspond to the untagged portion of the daily catch.

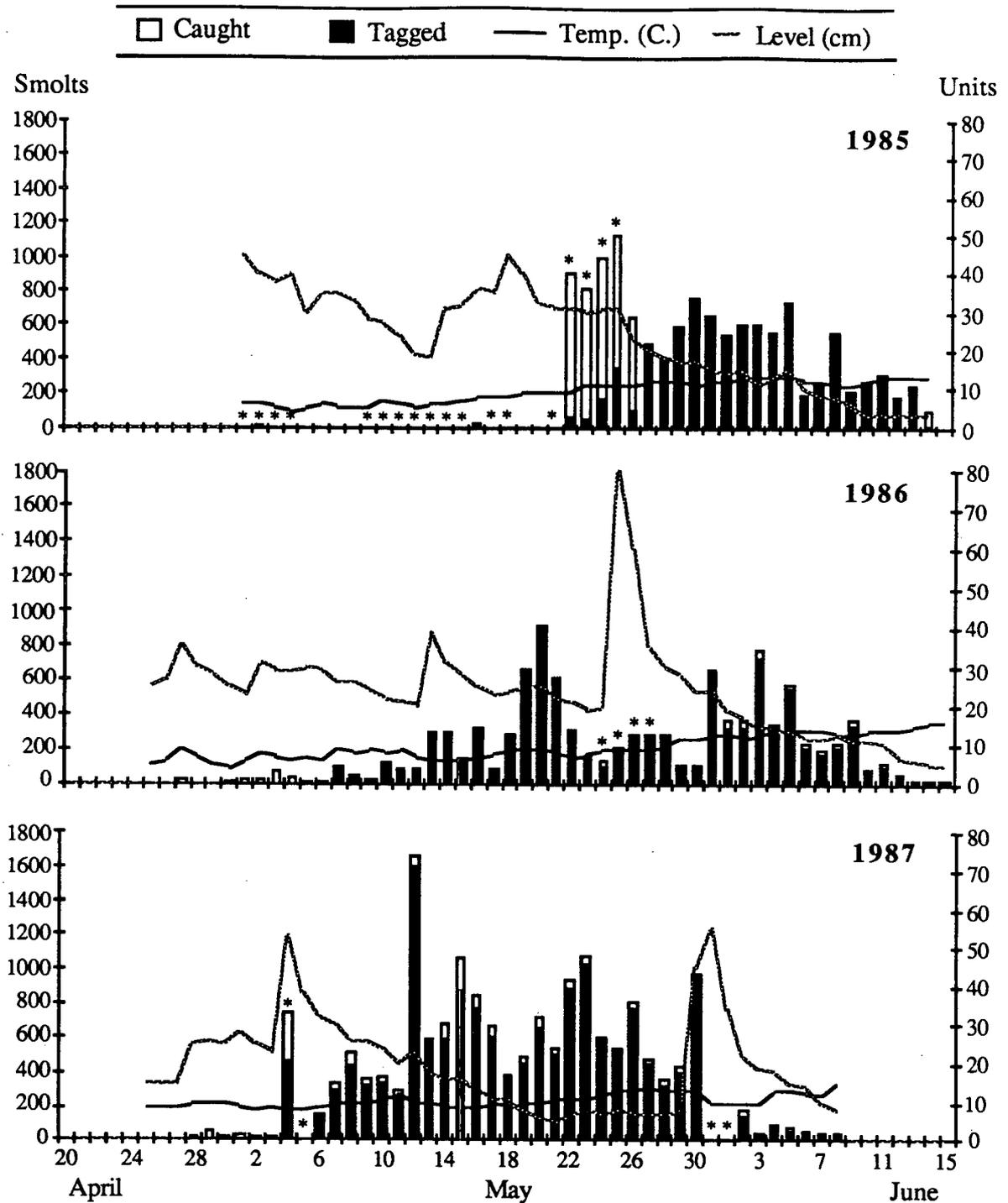


Figure 3.2. Smolt outmigration patterns and hydrological conditions at Trent River 1985 - 1987. The clear region on some vertical bars correspond to the untagged portion of the daily catch. [\*] Denotes incomplete daily counts.

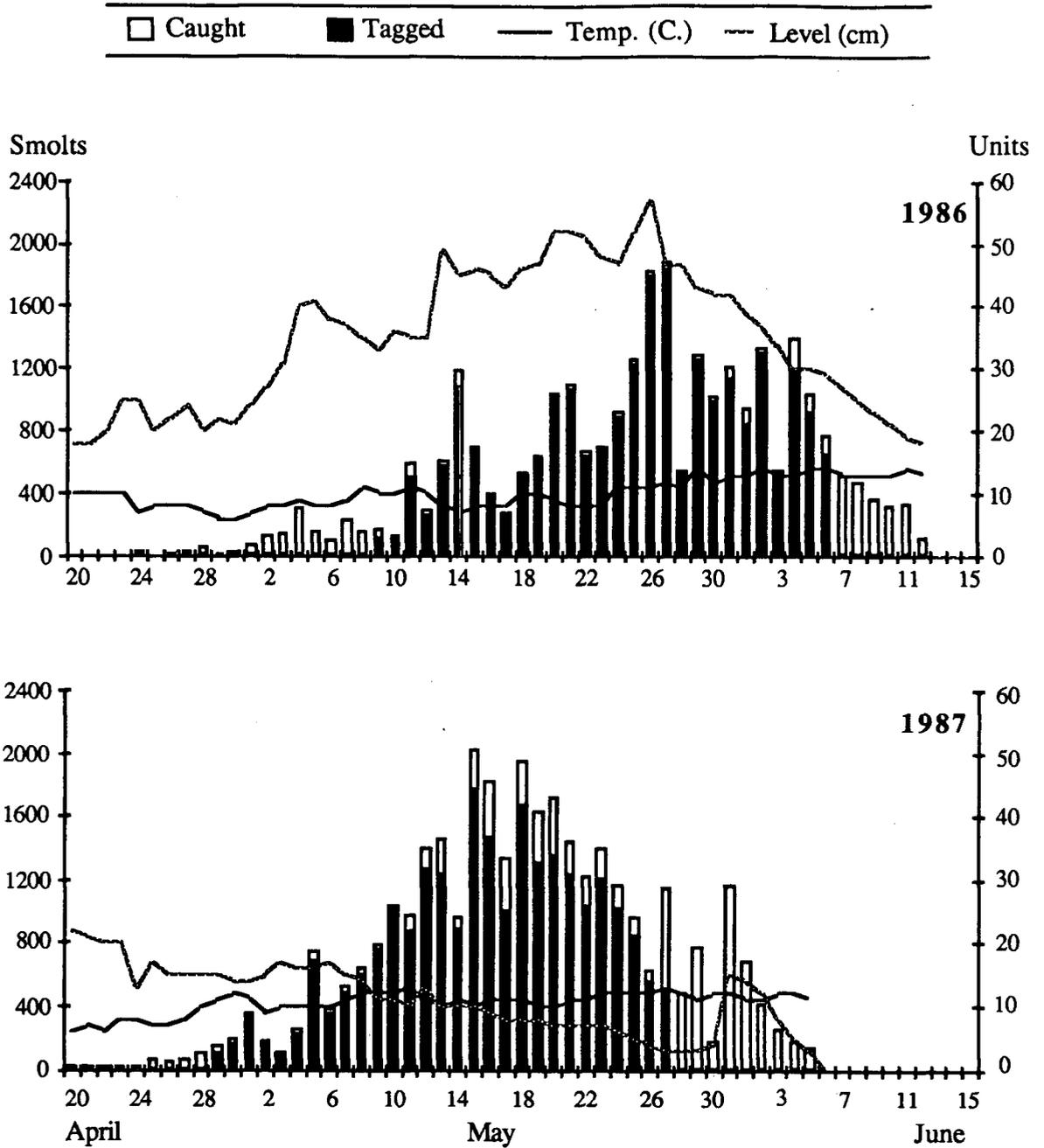


Figure 3.3. Smolt outmigration patterns and hydrological conditions at French Creek 1986 - 1987. The clear region on some vertical bars correspond to the untagged portion of the daily catch.

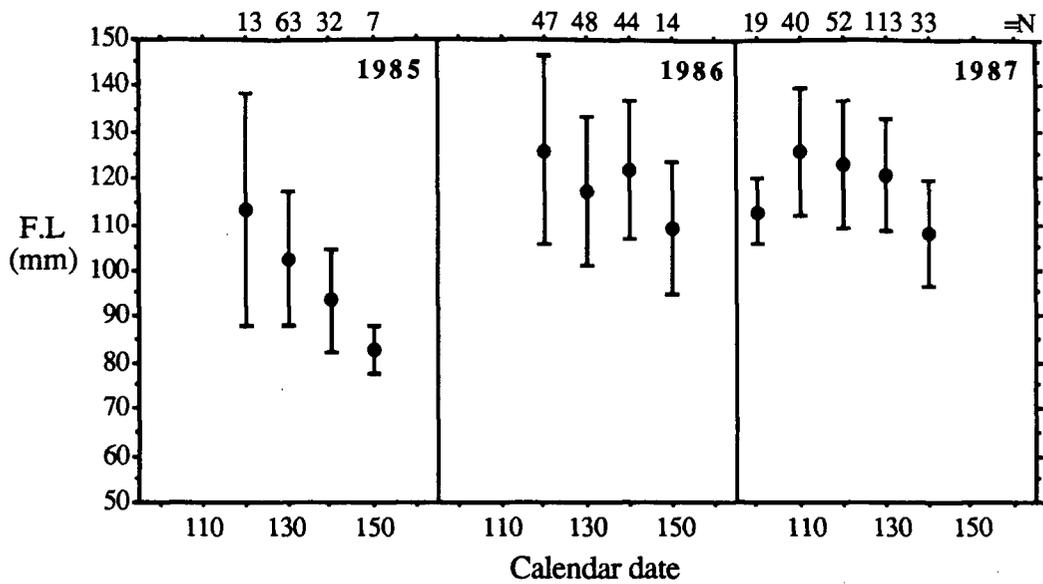


Figure 3.4. Black Creek smolt size during outmigration periods, 1985-1987. Time periods are in calendar days. For each 10 d interval, the mean fork length (mid dot)  $\pm$  one standard deviation, and the corresponding sample size are given.

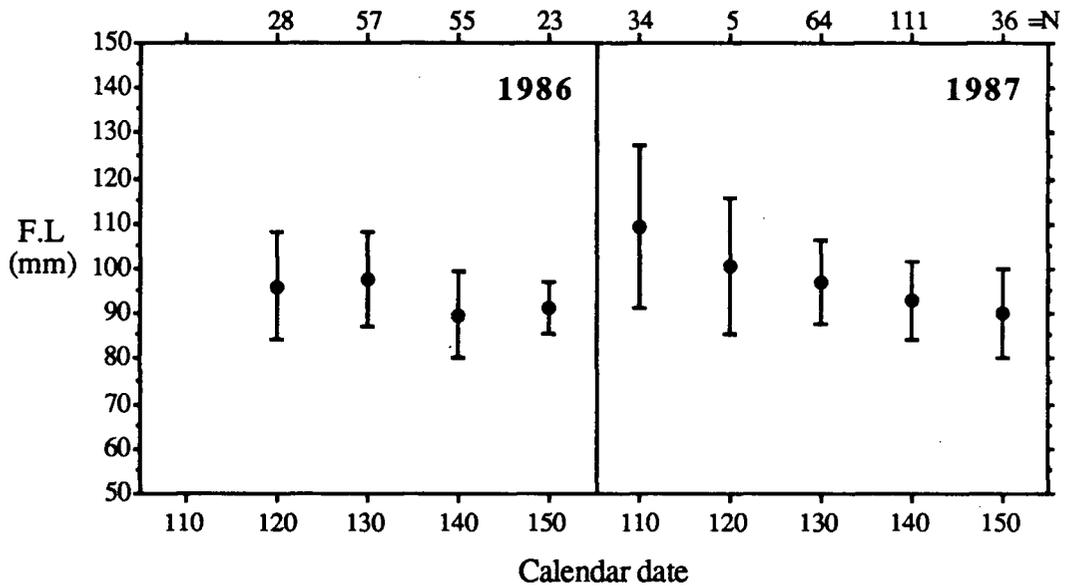


Figure 3.5. French Creek smolt size during outmigration periods, 1986-1987. Marked and unmarked smolts sizes were combined. For each 10 d interval, the mean fork length (mid dot)  $\pm$  one standard deviation, and the corresponding sample size (above) are given.

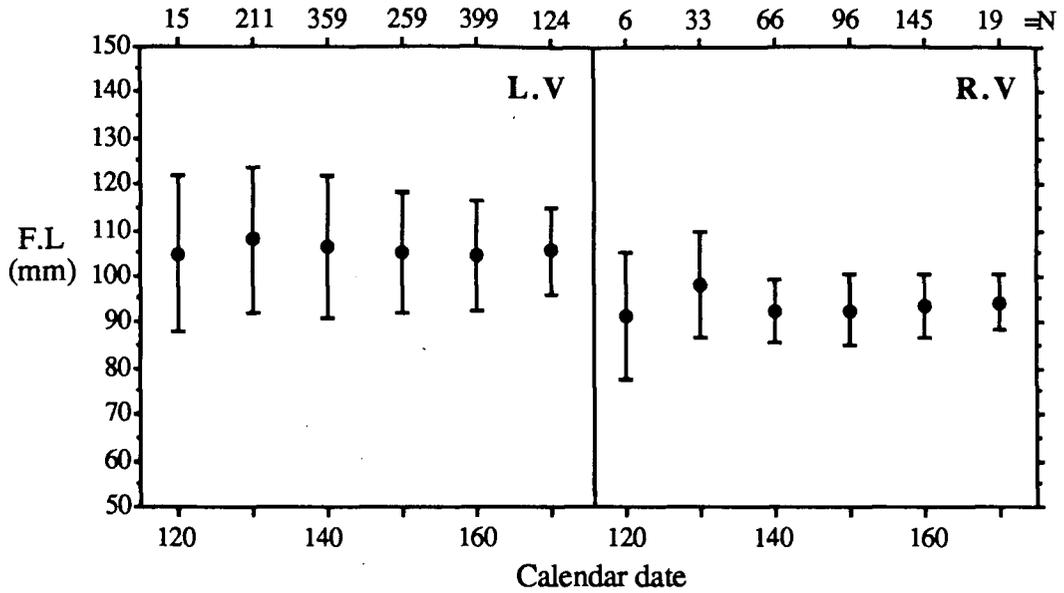


Figure 3.6. Trent River smolt size during 1986 outmigration. Sizes are for fin clipped smolts only. For each 10 d interval, the mean fork length (mid dot)  $\pm$  one standard deviation, and the corresponding sample size are given.

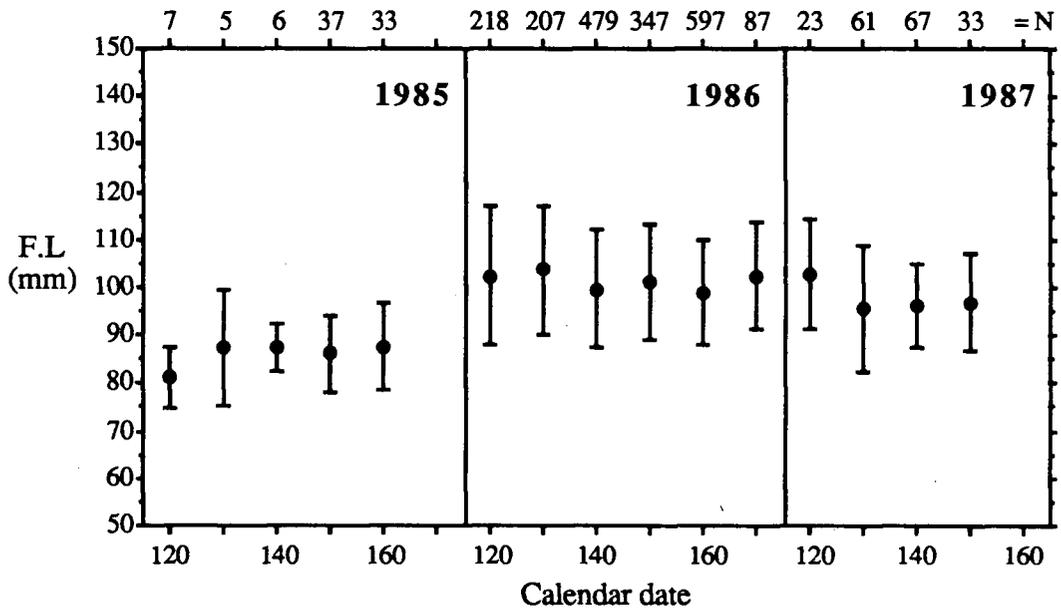


Figure 3.7. Trent River smolt size during outmigration periods, 1985-1987. Marked and unmarked smolts sizes were combined. For each 10 d interval, the mean fork length (mid dot)  $\pm$  one standard deviation, and the corresponding sample size (above) are given.

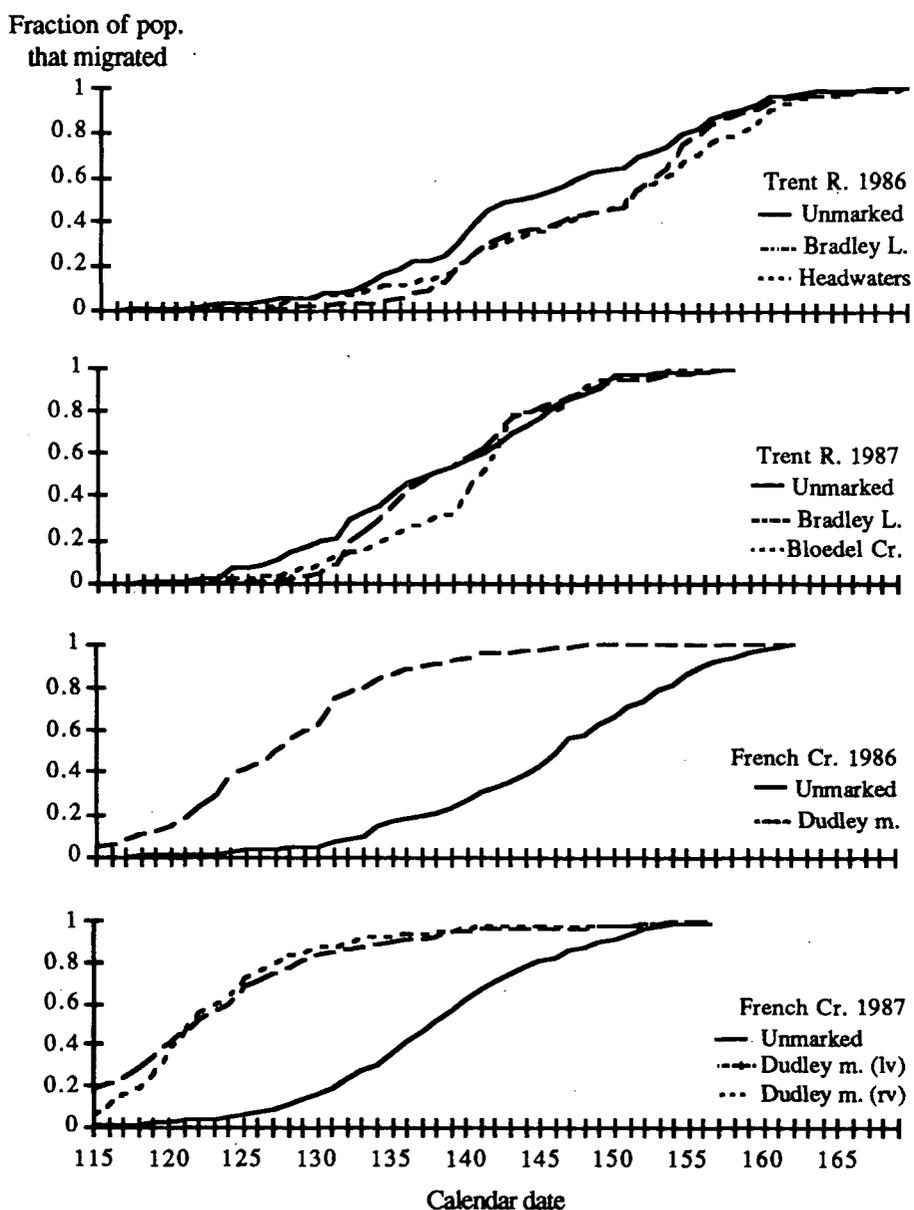


Figure 3.8. Smolt migration patterns of marked groups released at Trent River (above), and French Creek (below), during 1986 and 1987. The cumulative fractions of the total catch of each group are presented separately. The marks used at Trent River were; Bradley Lake (L.V), Bloedel Creek (R.V), and Headwaters (R.V). The marks used at French Creek were; 1986 Dudley marsh (LV), 1987 Dudley marsh small (LV), and 1987 Dudley marsh large (RV).

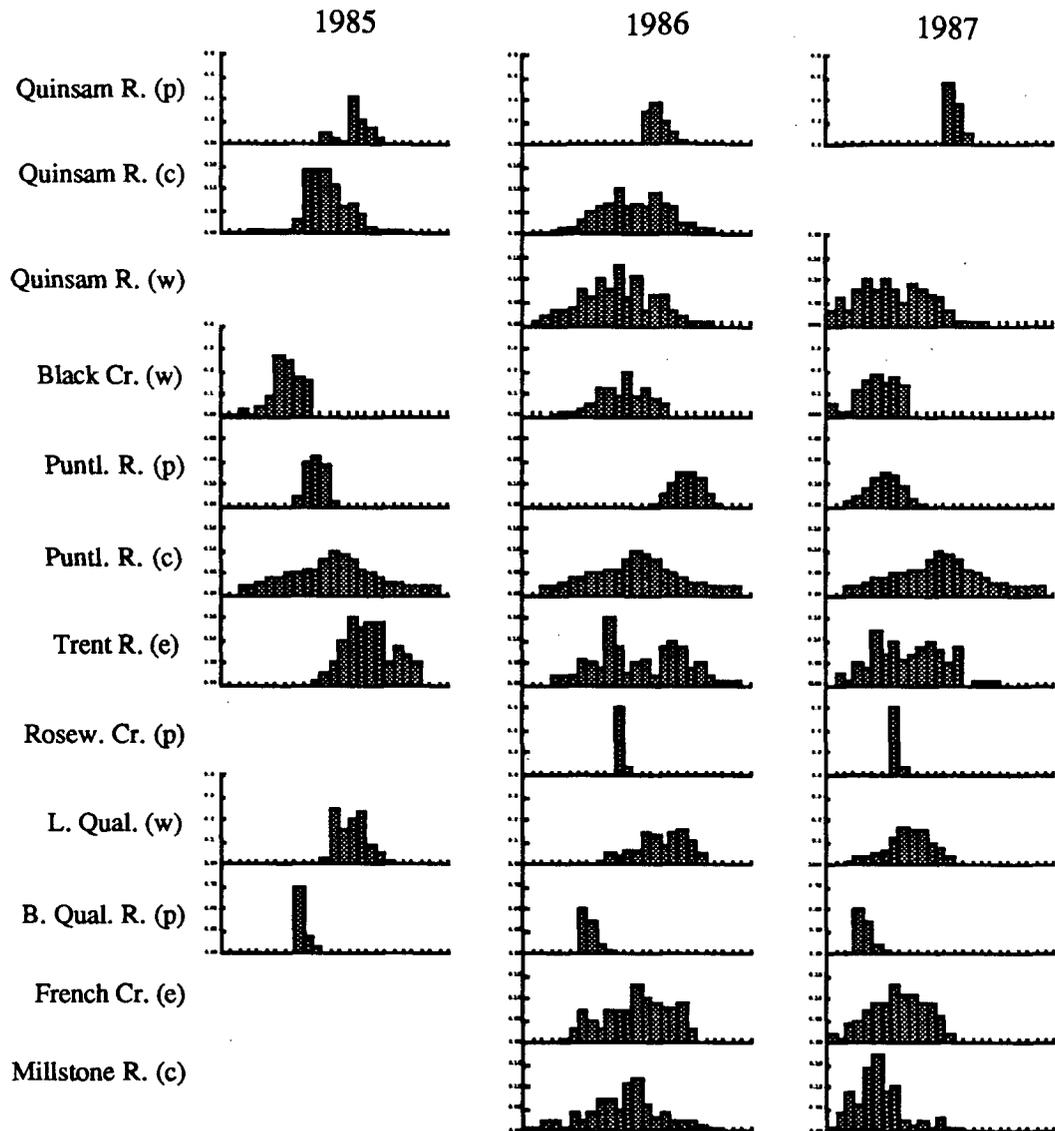


Figure 3.9. Migration patterns of tagged smolts released during 1985-1987. Each graphs shows the fraction of the total release entering the ocean during each successive 2 d interval between May 1 and June 16.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

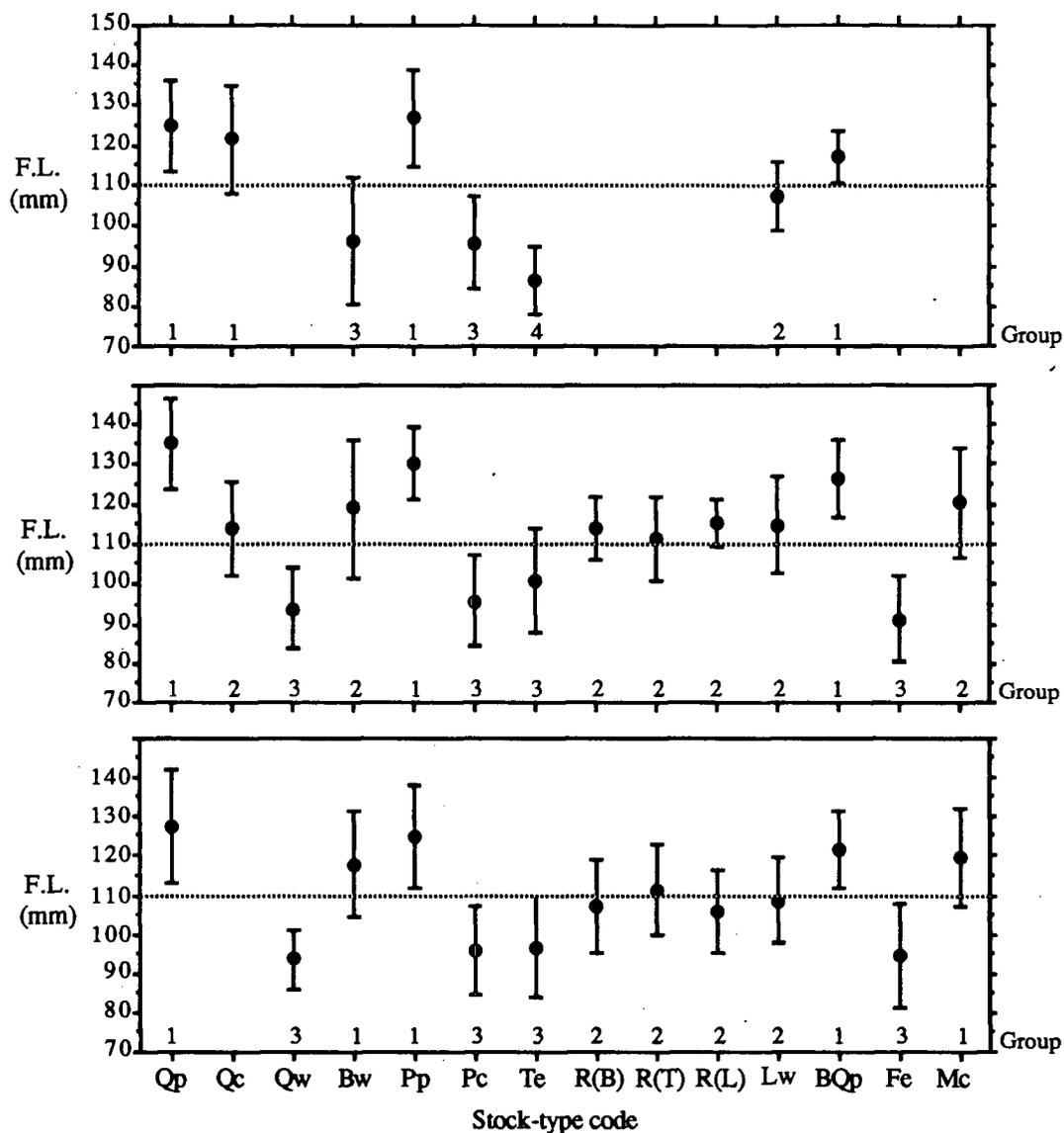


Figure 3.10. Mean fork length in each tagged smolt population that migrated during 1985 (above), 1986 (middle) and 1987 (below). Each vertical bar consists of the mean (mid dot)  $\pm$  one standard deviation. All sample sizes = 100. Statistically similar means are grouped from highest to lowest.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

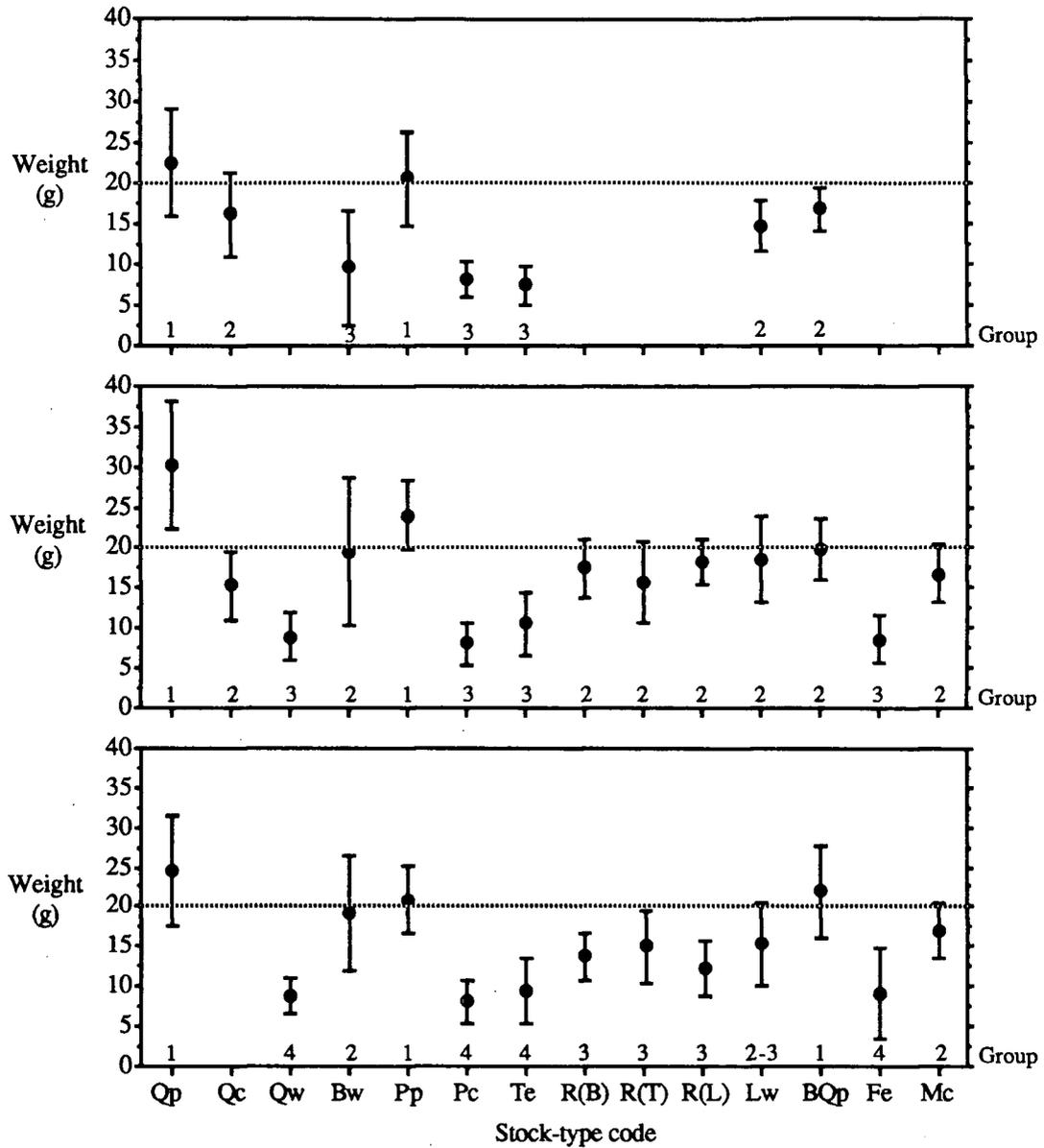


Figure 3.11. Mean weight in each tagged smolt population that migrated during 1985 (above), 1986 (middle) and 1987 (below). Each vertical bar consists of the mean (mid dot)  $\pm$  one standard deviation. All sample sizes = 100. Statistically similar means are grouped from highest to lowest.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

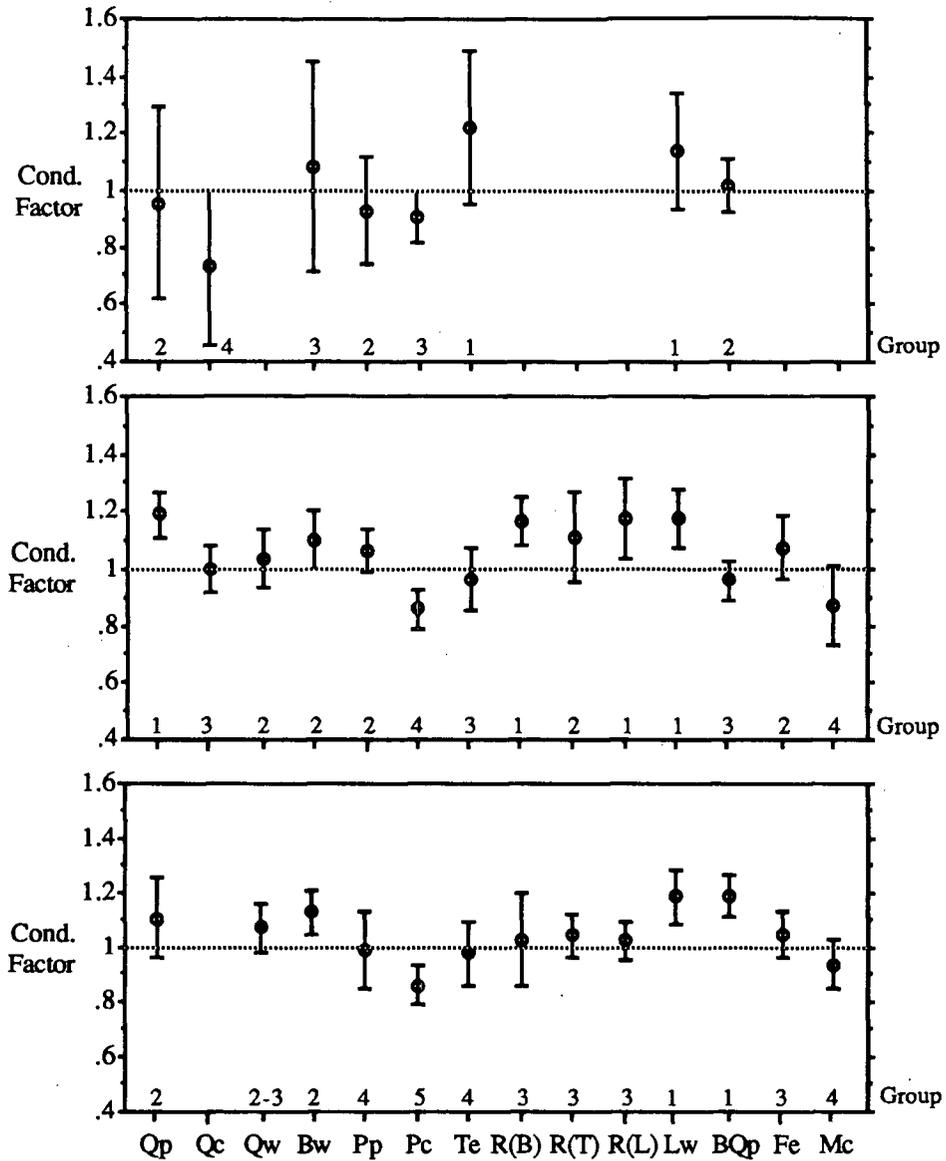


Figure 3.12. Mean condition factor ( $100 W \cdot FL^{-3}$ ) of each tagged smolt population that migrated during 1985 (above), 1986 (middle) and 1987 (below). Each vertical bar consists of the mean (mid dot)  $\pm$  one standard deviation. All sample sizes = 100. Statistically similar means are grouped from highest to lowest.

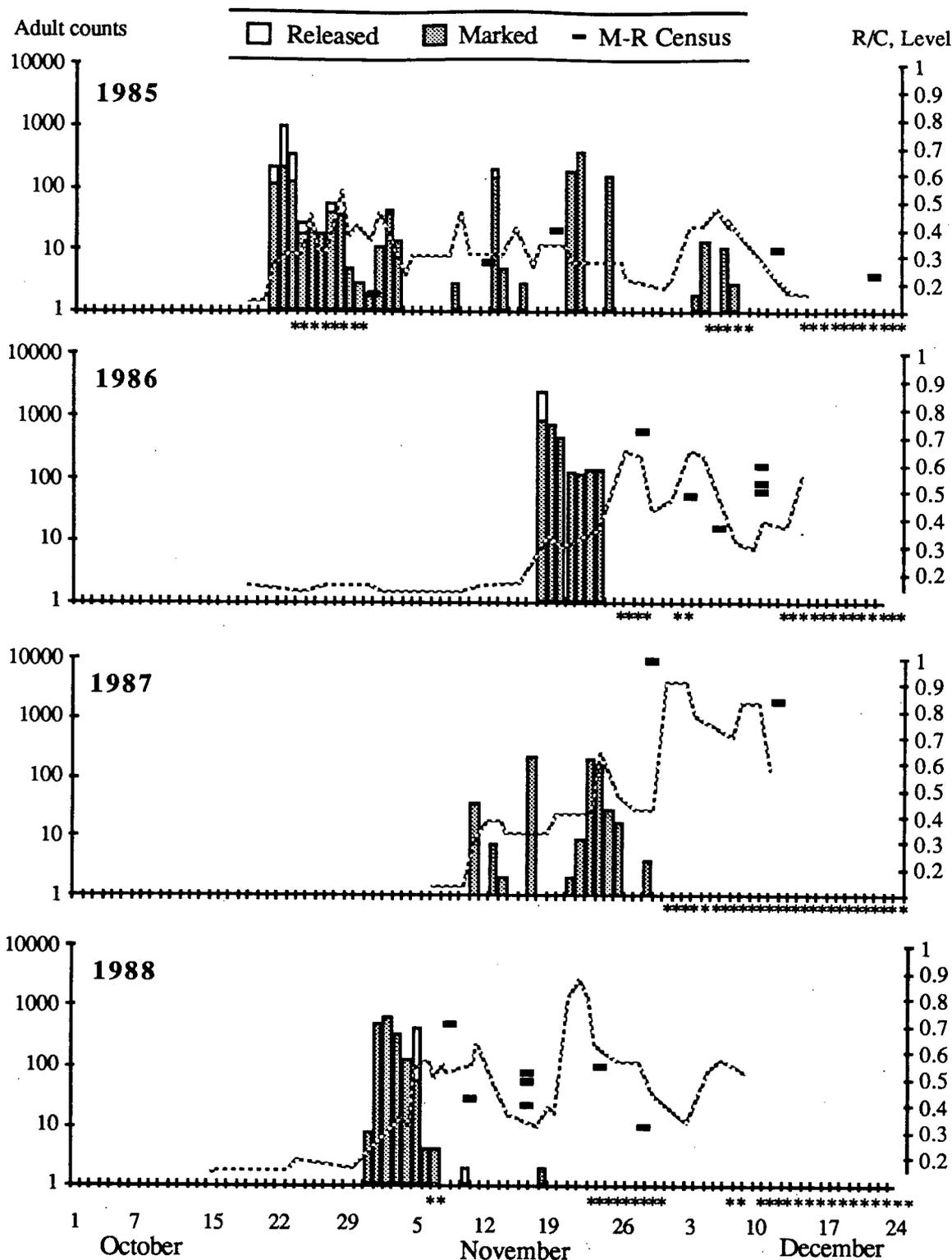


Fig. 4.1. Daily fence counts (vertical bars), water levels (dotted line), and distribution of population surveys (horizontal bars =  $R_t/C_d$ ) for adults at Black Creek. Stars (\*) indicate incomplete fence counts. Water levels correspond to scale units x 1.6 m.

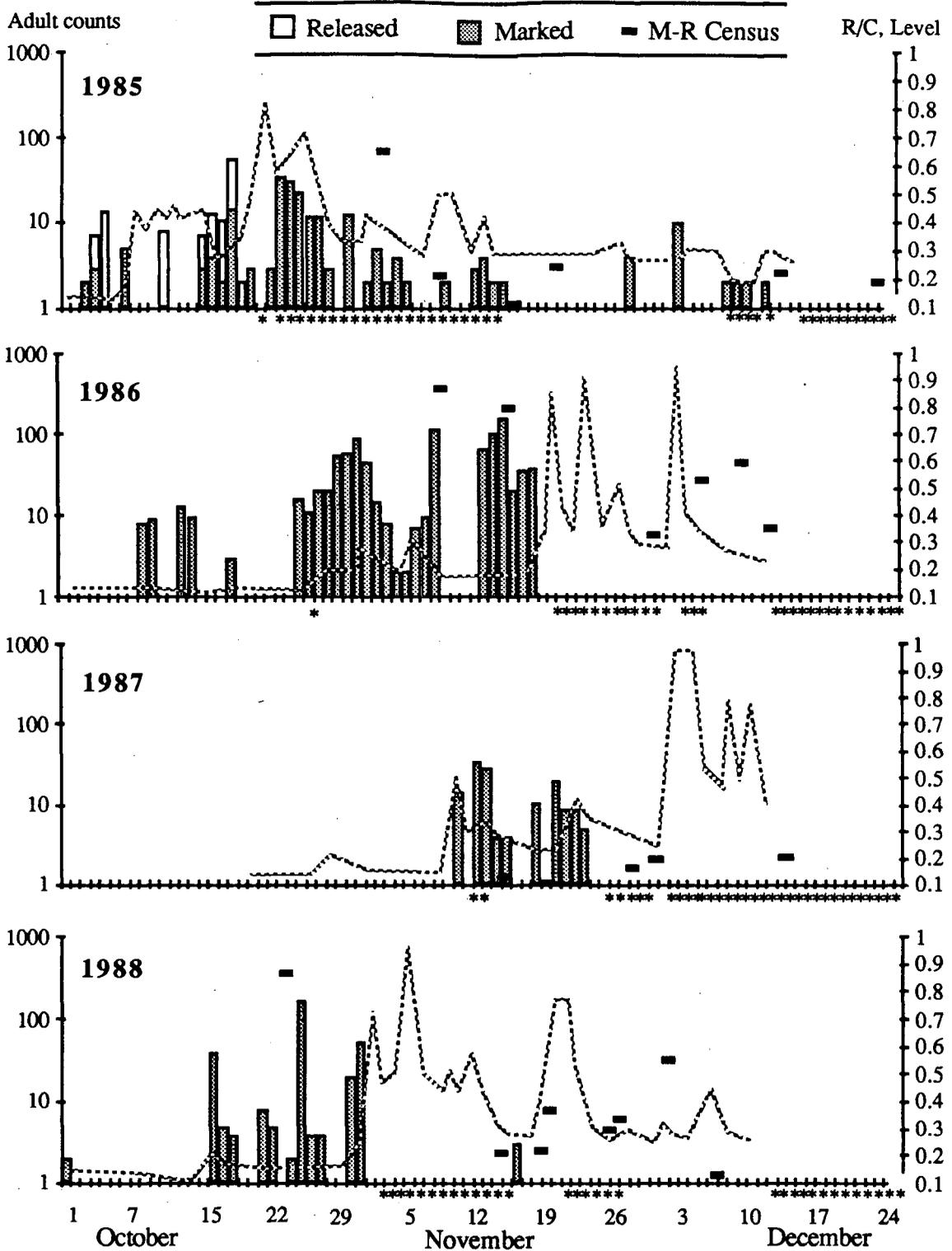


Fig. 4.2. Daily fence counts (vertical bars), water levels (dotted line), and distribution of population surveys (horizontal bars =  $R_i/C_i$ ) for adult coho at Trent River. Stars (\*) indicate incomplete fence counts. Water levels correspond to scale units x 2.0 m.

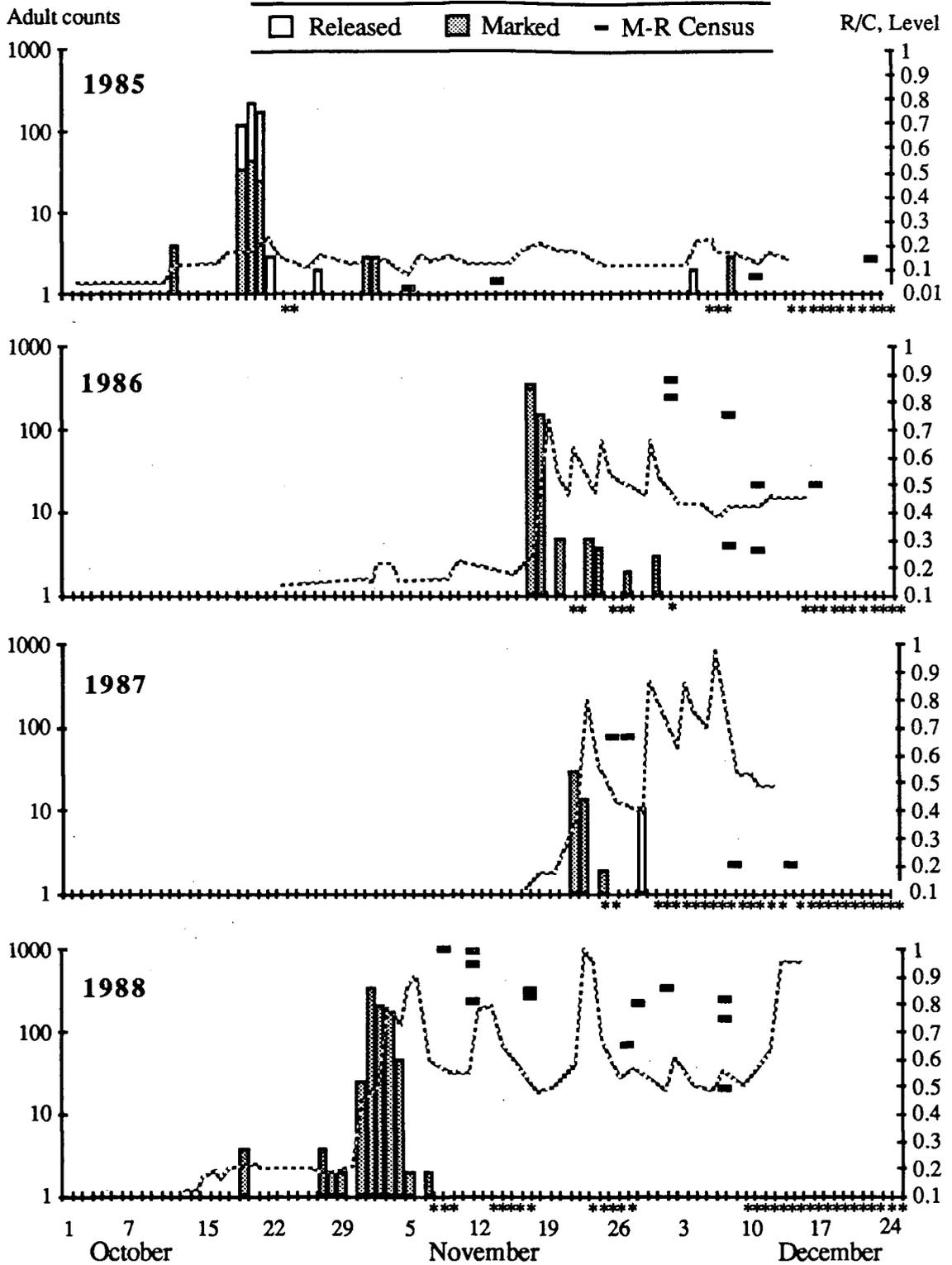


Fig. 4.3. Daily fence counts (vertical bars), water levels (dotted line), and distribution of population surveys (horizontal bars =  $R_i/C_i$ ) for adult coho at French Creek. Stars (\*) indicate incomplete fence counts. Water levels correspond to scale units x 0.8 m.

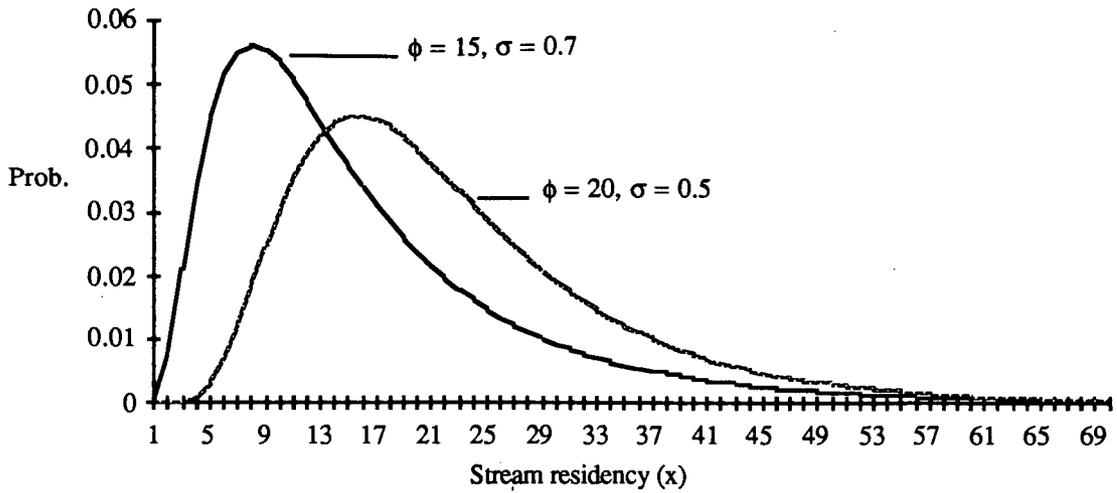


Fig. 4.4. Probability density functions of stream residency for two hypothetical populations with different stream residency parameters  $\phi$  and  $\sigma$ . The shape parameters translate into a mean ( $\bar{x}$ ) and standard deviation of 19 d and 13 d (top curve), and 23 d and 10 d (bottom curve).

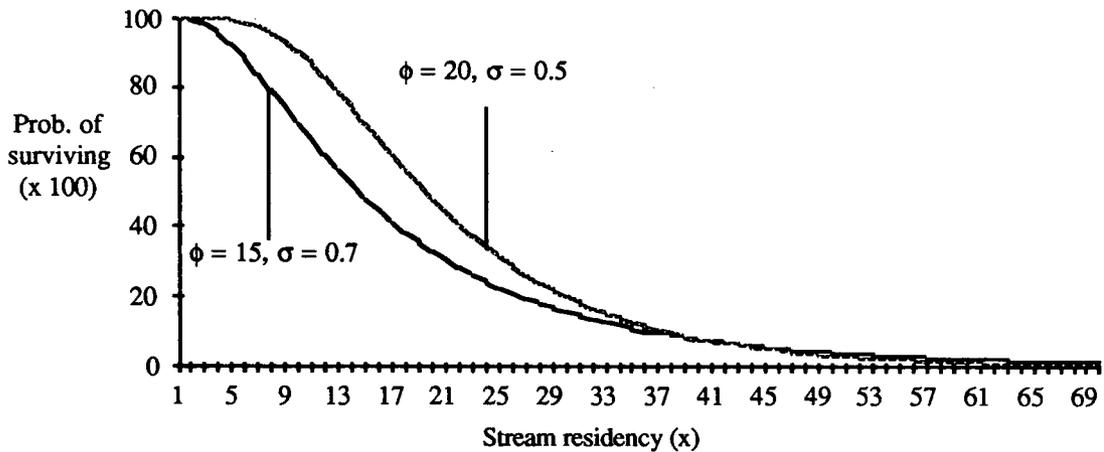


Fig. 4.5. Survival curve of two hypothetical populations with different stream residency parameters.

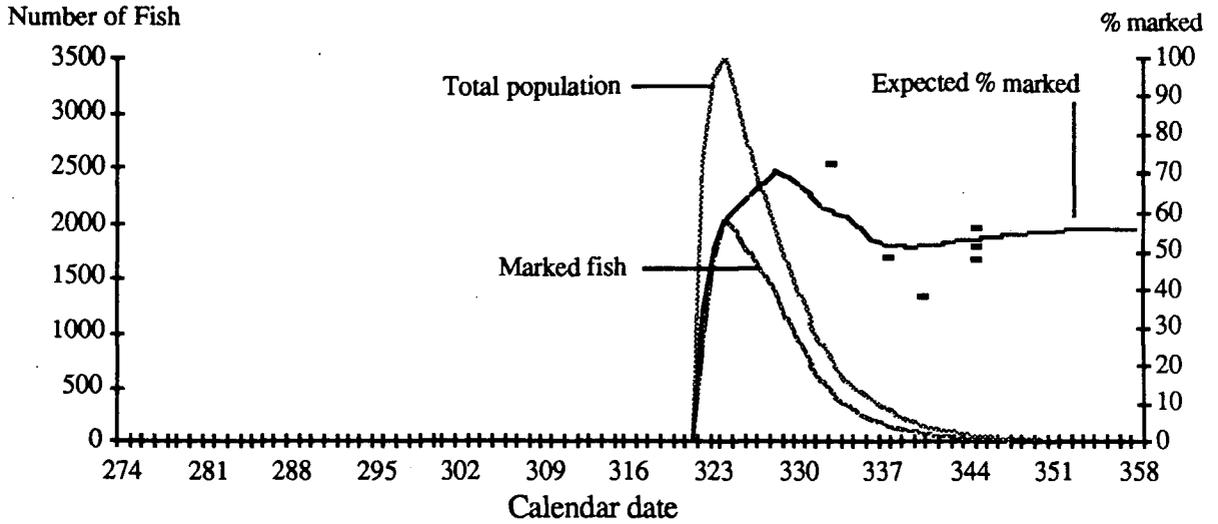


Fig. 4.6. Simulated build-up and decline of fish population showing hypothesized trajectory of expected mark proportions in relation to actual mark proportions observed during stream surveys at Black Creek, 1986.

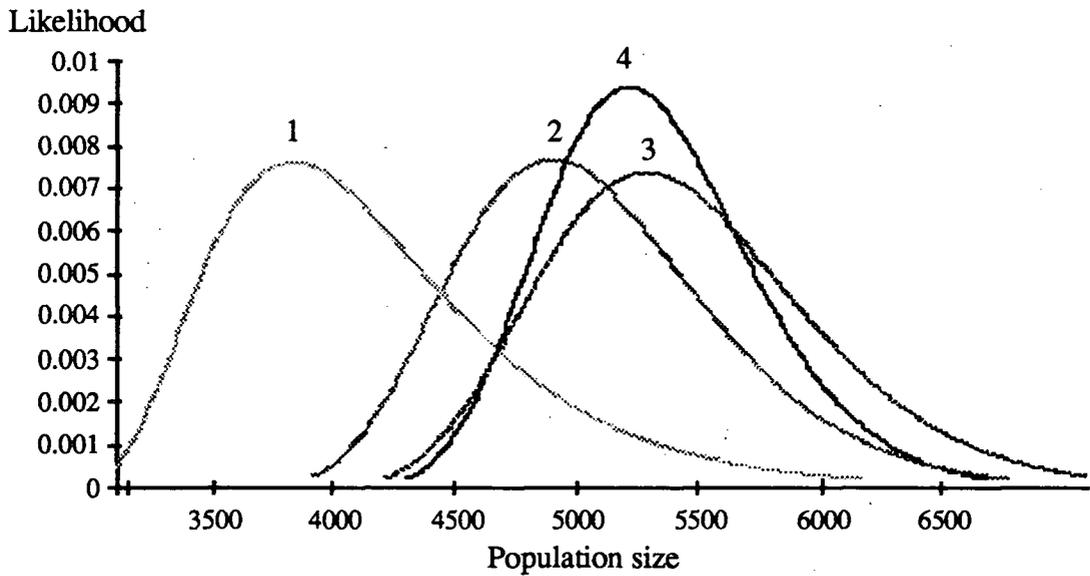


Fig. 4.7. Successive posterior distributions generated from the Bayesian model based on census data collected while immigration was occurring (Black Creek 1986).

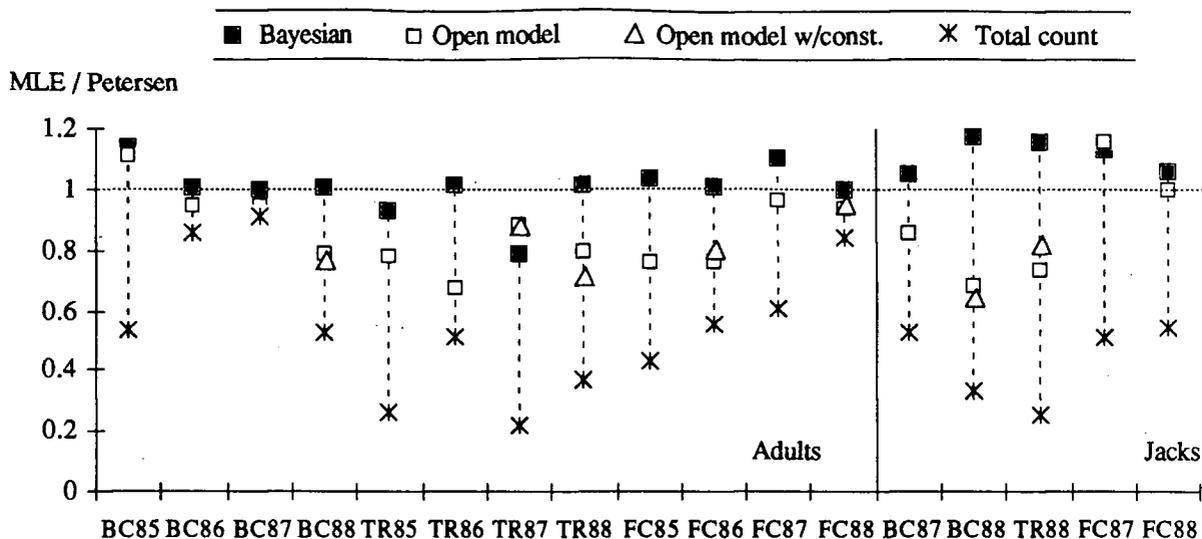


Fig. 4.8. Comparison of estimates obtained for each age group/stream/year combination. MLE's generated from the Bayesian, and Open population models (with and without constraints) are presented as ratios to the adjusted Petersen estimate (Bailey's model).

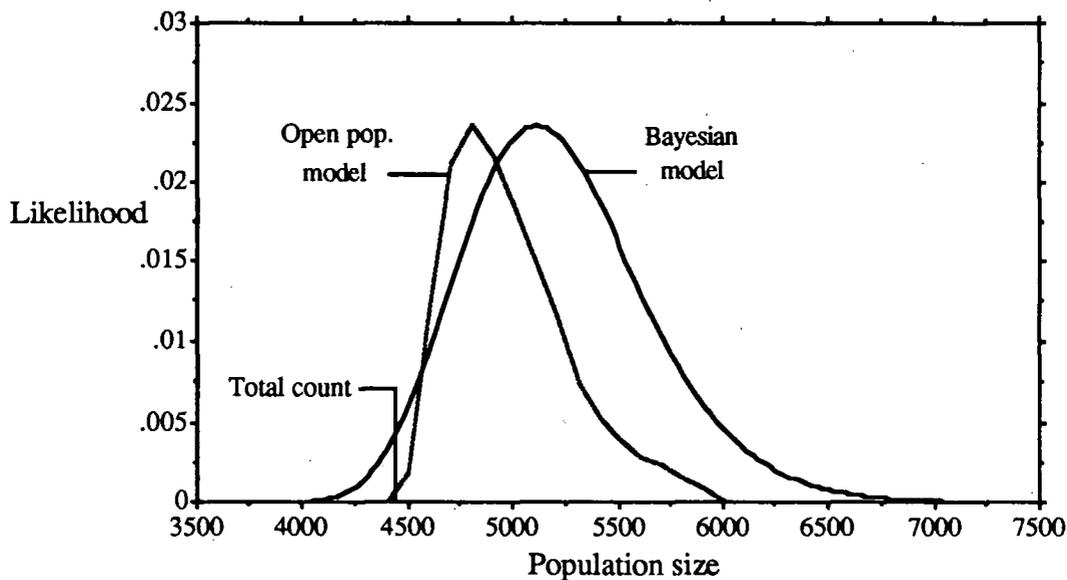


Fig. 4.9. Comparison of likelihood curves generated by both models, using adult census data from Black Creek, 1986. The total count indicates the minimum escapement.

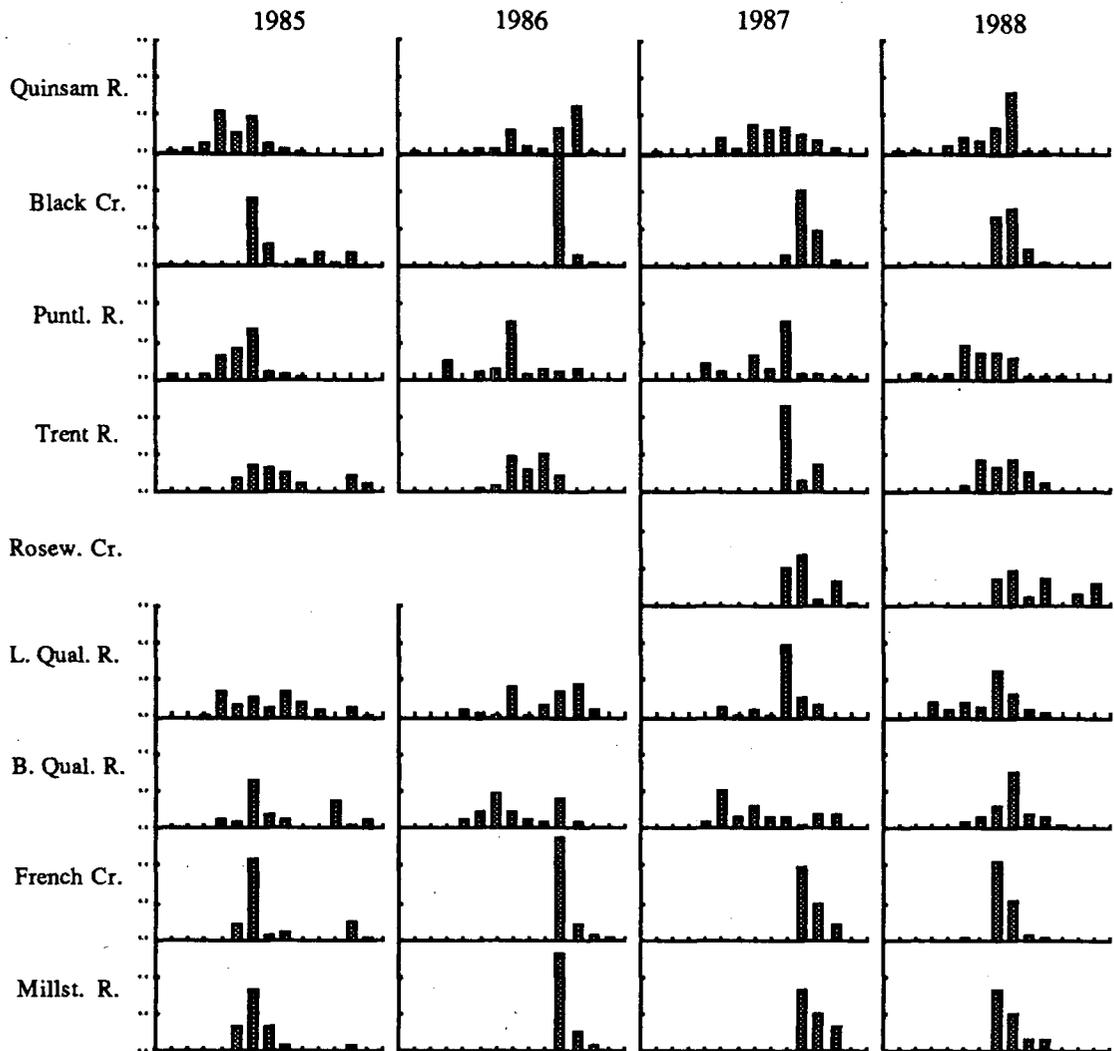


Figure 4.10. Migration patterns of adults escaping to the various streams in the study area. Each graph shows the estimated fraction of the total run that arrived the fence each week between September 15 and December 15. Each weekly interval is delineated by tick marks along the abscissa.

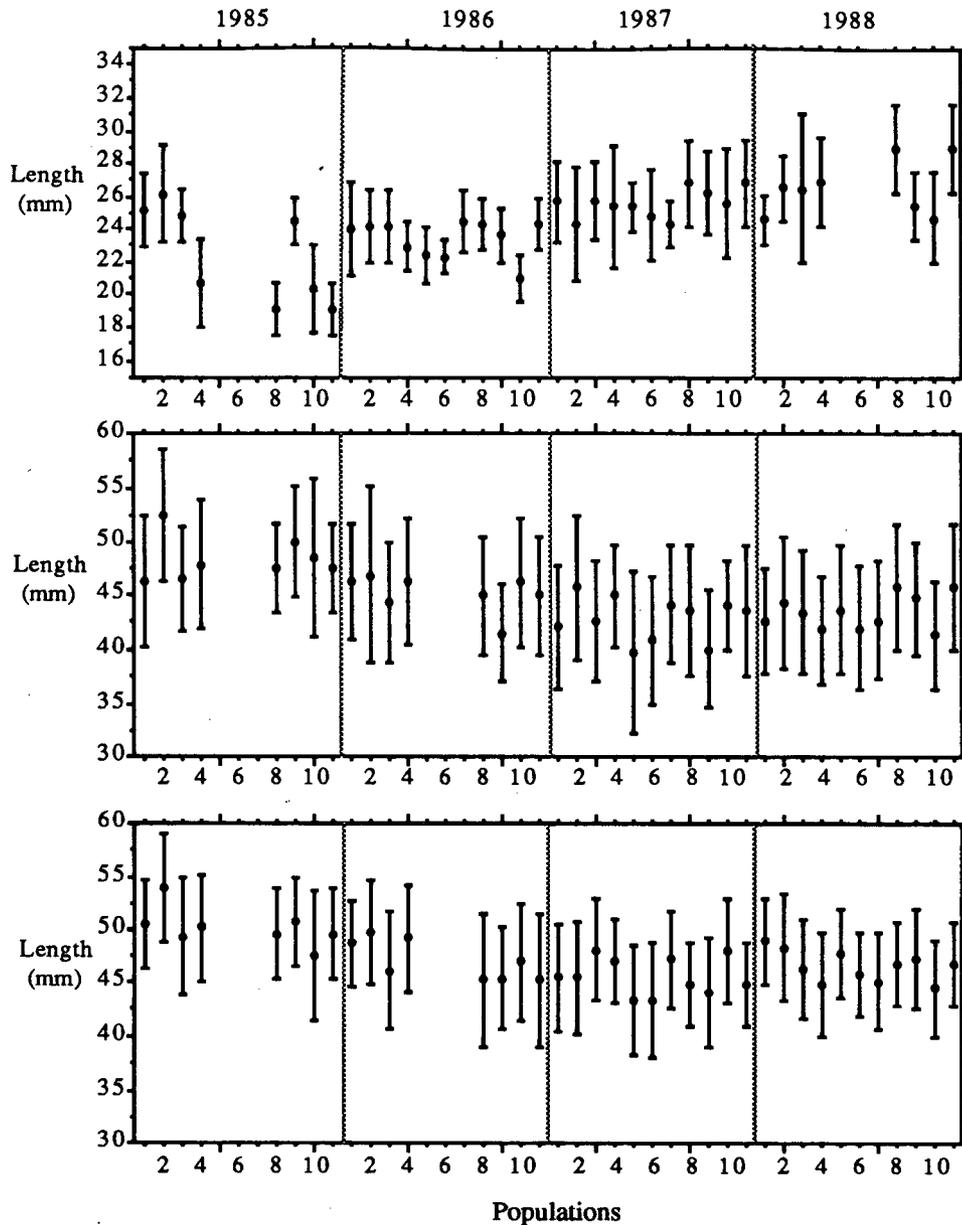


Figure 4.11. Post-orbital lengths of jacks (above), adult males (middle), and females (below) returning to each stream from 1985 to 1988. Each range represents the mean (mid dot)  $\pm$  one standard deviation. Population codes are Quinsam (1), Black Creek (2), Puntledge R. (3), Trent R. (4), Rosewall Cr./Black Cr. (5), Rosewall Cr./Trent R. (6), Rosewall Cr./L. Qual. R. (7), L. Qual. R. (8), B. Qual. R. (9), French Cr. (10), Millst. R. (11). Quinsam R., Puntledge R., and B. Qualicum sizes based on pooled returns of production, colonization, and wild fish.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

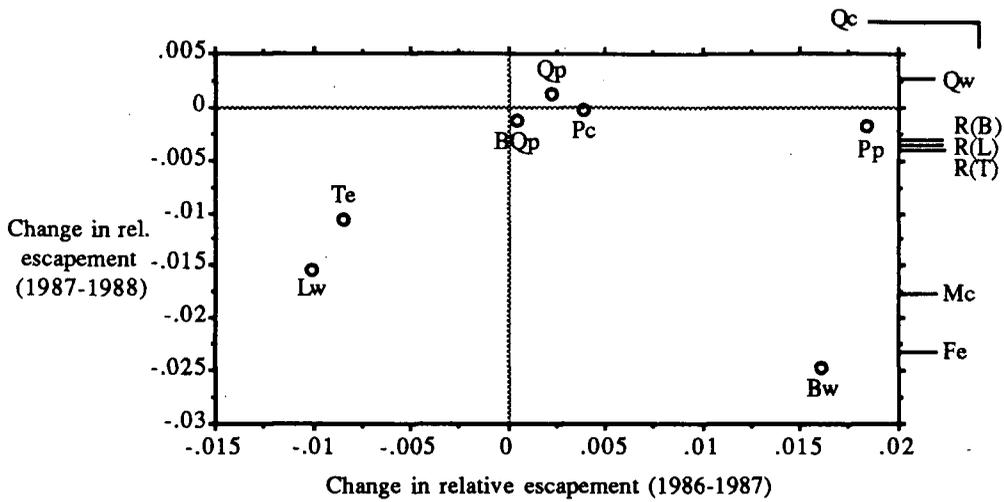


Figure 5.1. Pattern of co-variation in relative escapement (CWT escapement / CWT release). The coordinates of each stock monitored for three consecutive years are expressed in terms of the changes in relative escapement levels between the two successive periods. Stocks which were monitored for only two years are positioned outside along each axis next to their associated change in relative escapement.

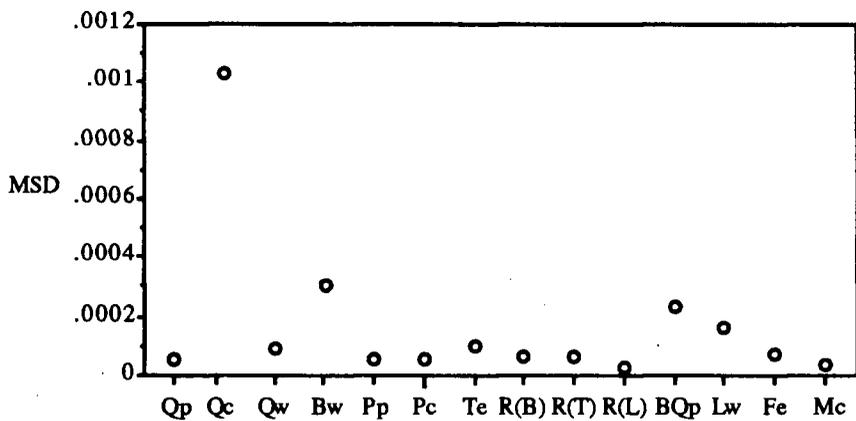


Figure 5.2. Estimates of the mean squared deviations (MSD) of relative escapement. Stocks with relative escapements closest to the annual mean value have the lowest MSD score.

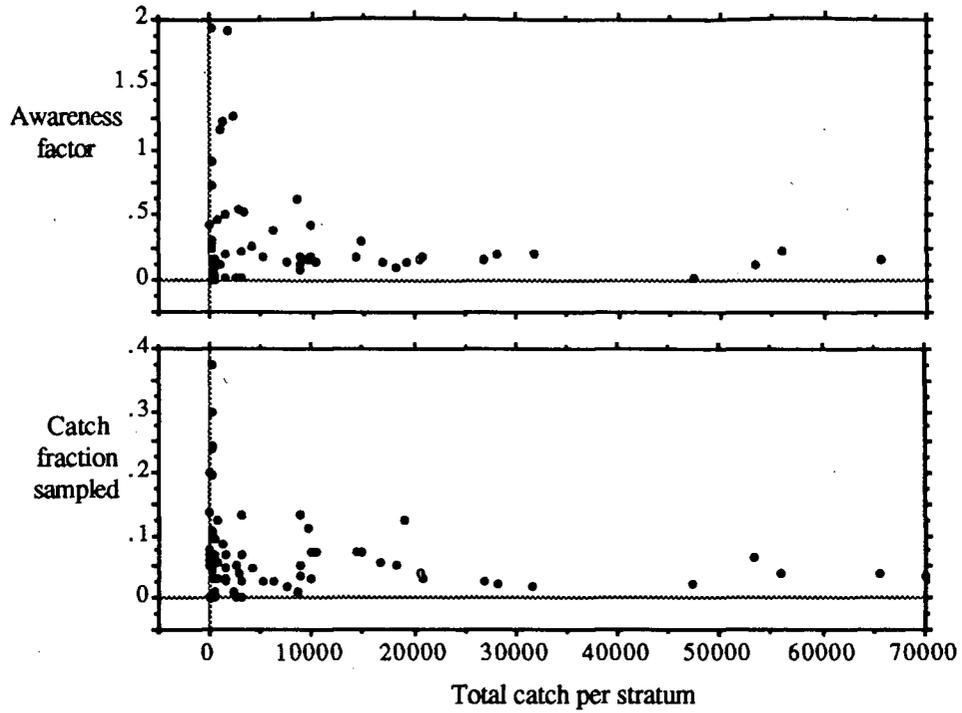


Figure 6.1. Awareness factors and sampling fractions versus stratum catches in the Strait of Georgia Strait during 1987 (source: Creel Survey statistics).

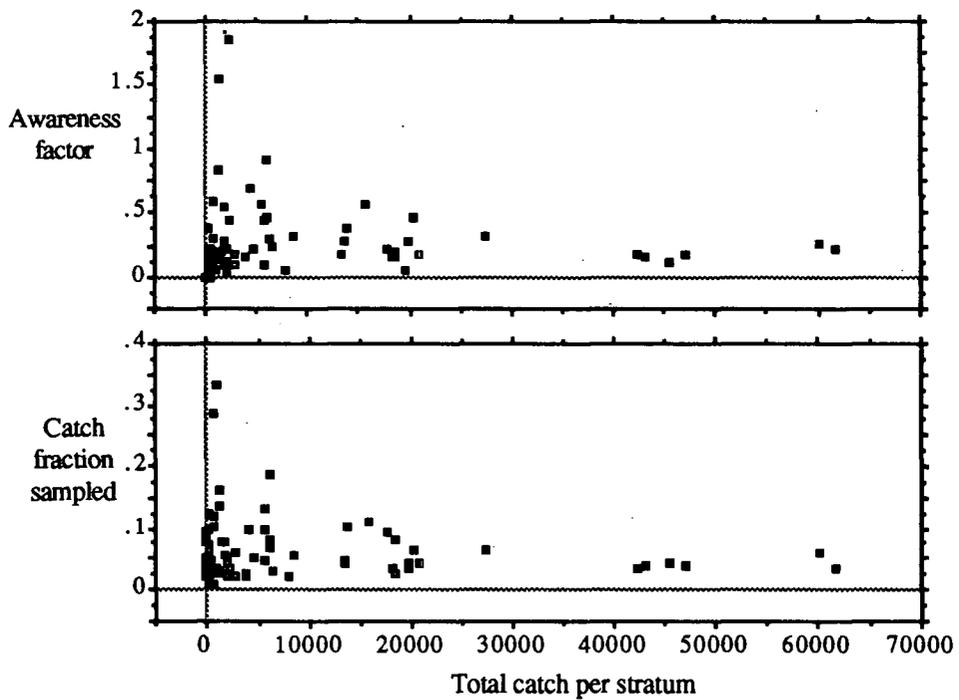


Figure 6.2. Descriptive model output of awareness factors and sampling fractions versus stratum catches in the Strait of Georgia during 1987.

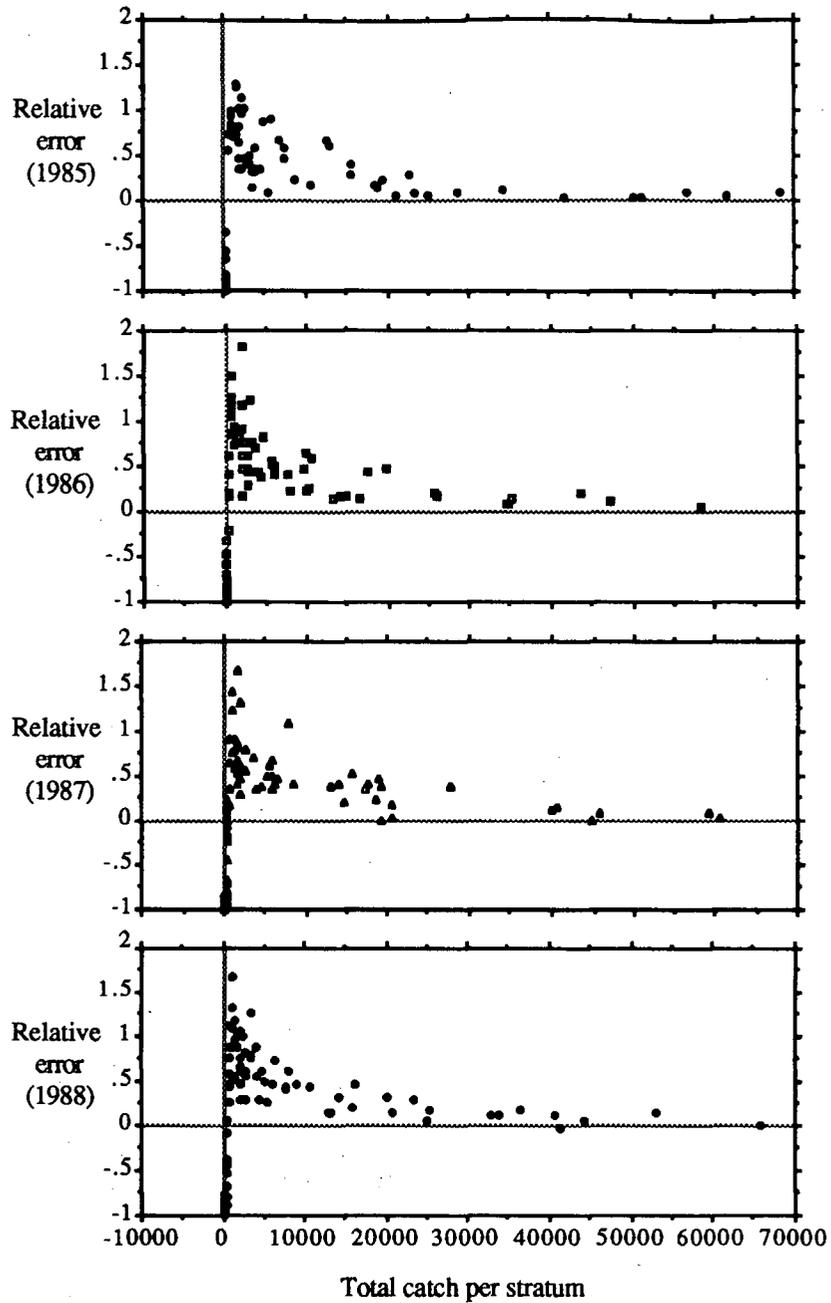


Figure 6.3. Estimates of relative error associated with awareness factor estimates in relation to stratum catch during 1985 - 1988. Each dot represent the average error for a given catch level.

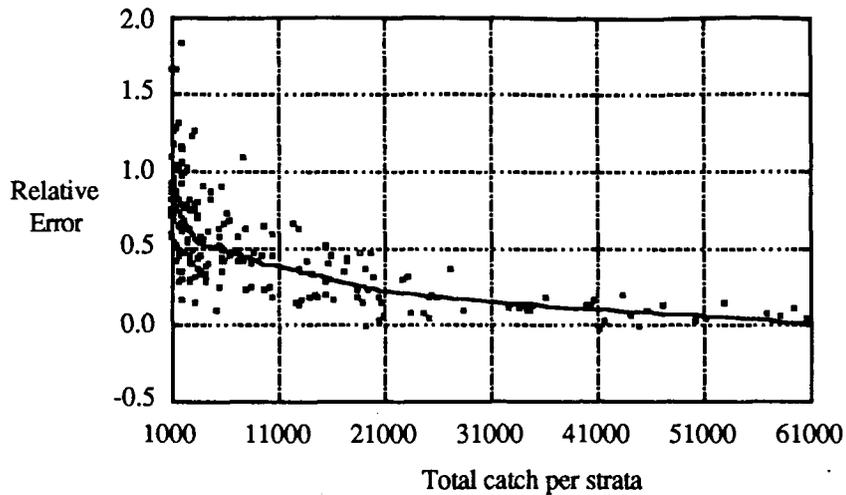


Figure 6.4. Relationship between the relative error in awareness factor estimates and stratum catch based on pooled data from 1985-1988. The thin line corresponds to the average error for a given catch level (Lowess smoothing function).

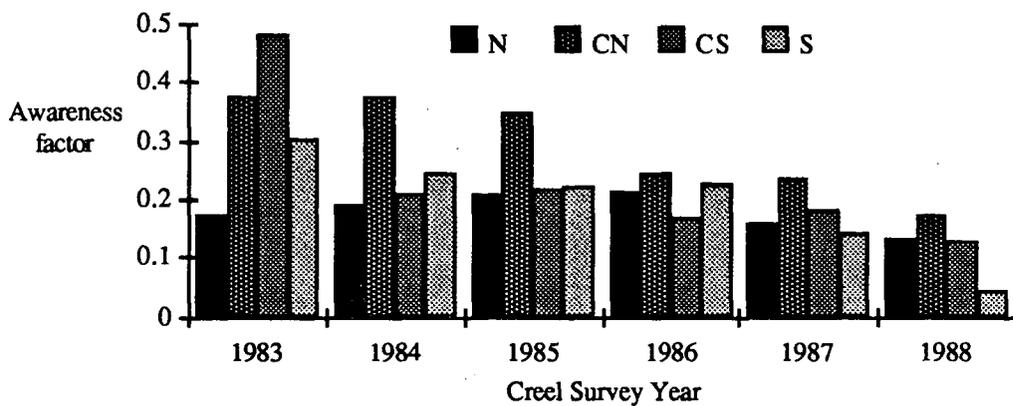


Figure 6.5. Estimates of awareness factors by geographical region in the Strait of Georgia, 1983-1988. The 4 regions are north (N = Areas 13, 15), central north (CN = Areas 14, 16), central south (CS = Areas 17, 28, 29), and south (S = Areas 18, 19, 20). All figures are corrected for estimation errors.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

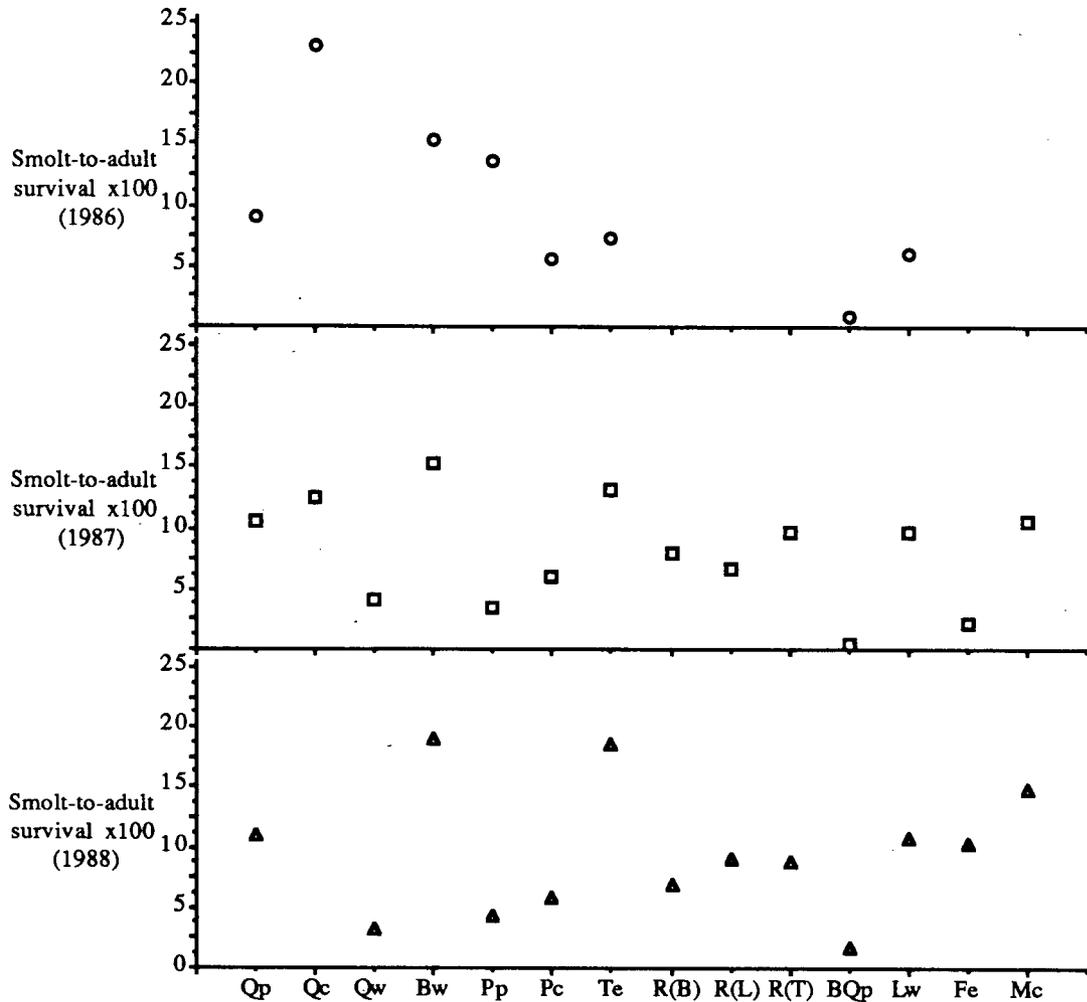


Figure 6.6. Estimates of survival (%S x 100) from smolt to adult, uncorrected for trapping and tagging effects. Each year corresponds to the adult (age 3+) return year.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	p = production
			w = wild
			c = colonization

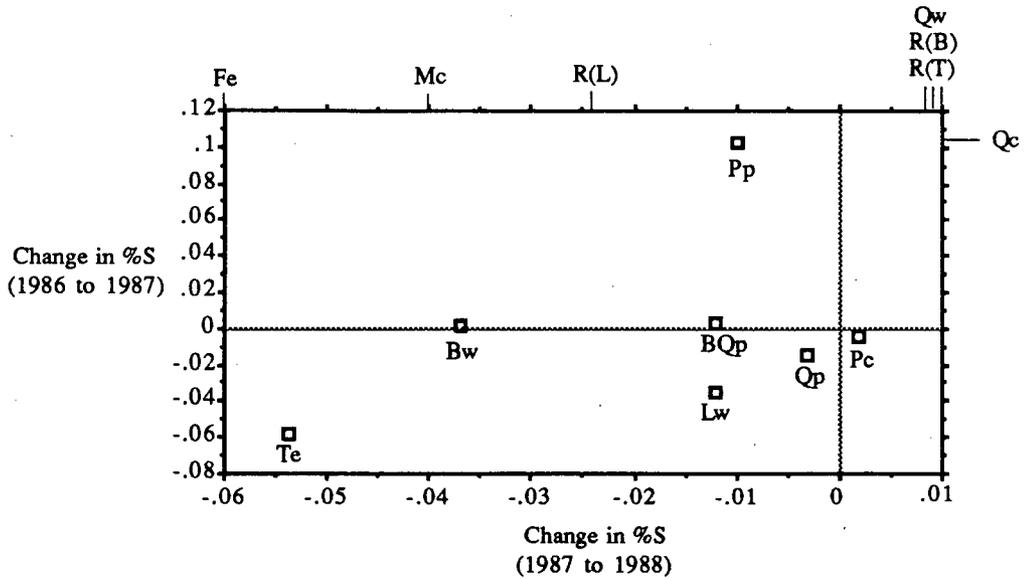


Figure 6.7. Pattern of co-variation in survival rates (%S). The co-ordinates of each stock monitored for three consecutive years is expressed in terms of the changes in survival rates between the two successive periods. Stocks which were monitored for two years are positioned outside along each axis next to their associated change in survival rate.

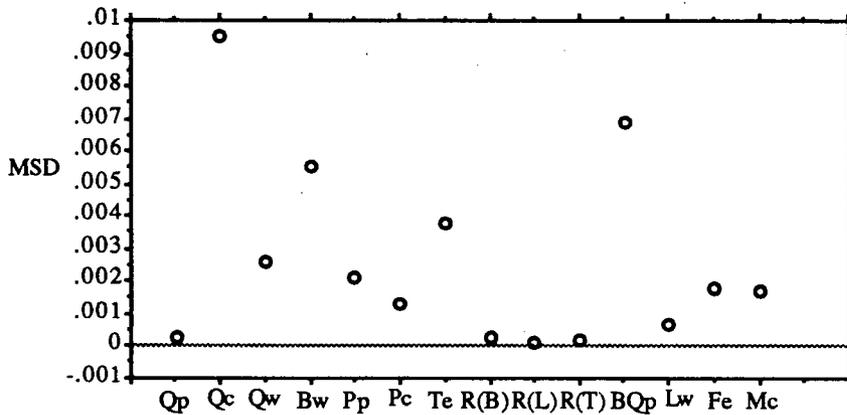


Figure 6.8. Estimates of the mean squared deviations (MSD) from the mean annual survival rates. Stocks with survival rates closest to the overall mean rate have the lowest MSD score.

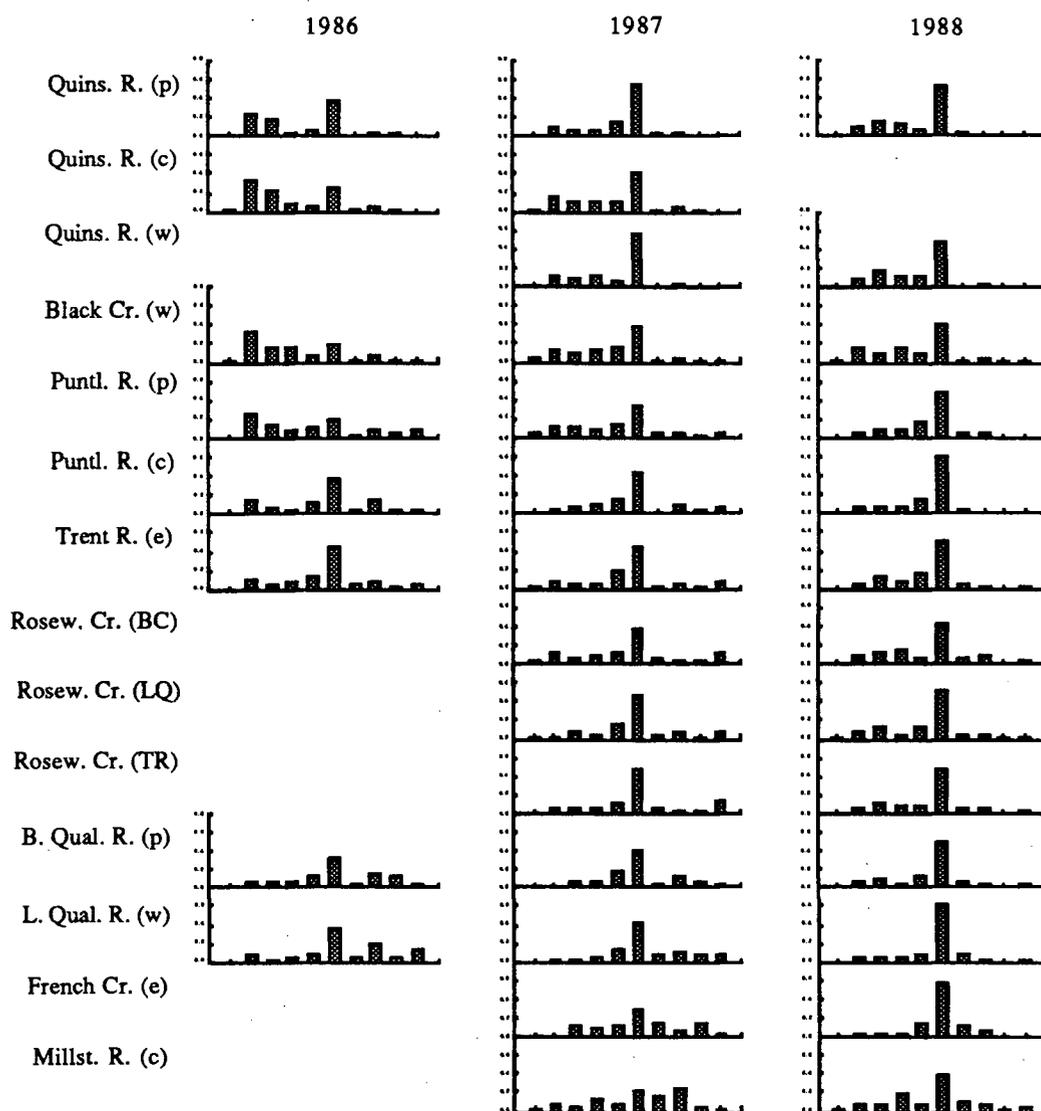


Figure 7.1. Relative contribution of age 3+ adults from each stock to various fishing regions by adult return year. The categories, from left to right, correspond to; North, Central, JS.N, WVI(n), GS.TR, GS.SP(n), GS.SP(s), WVI(s), South, and GS.N.

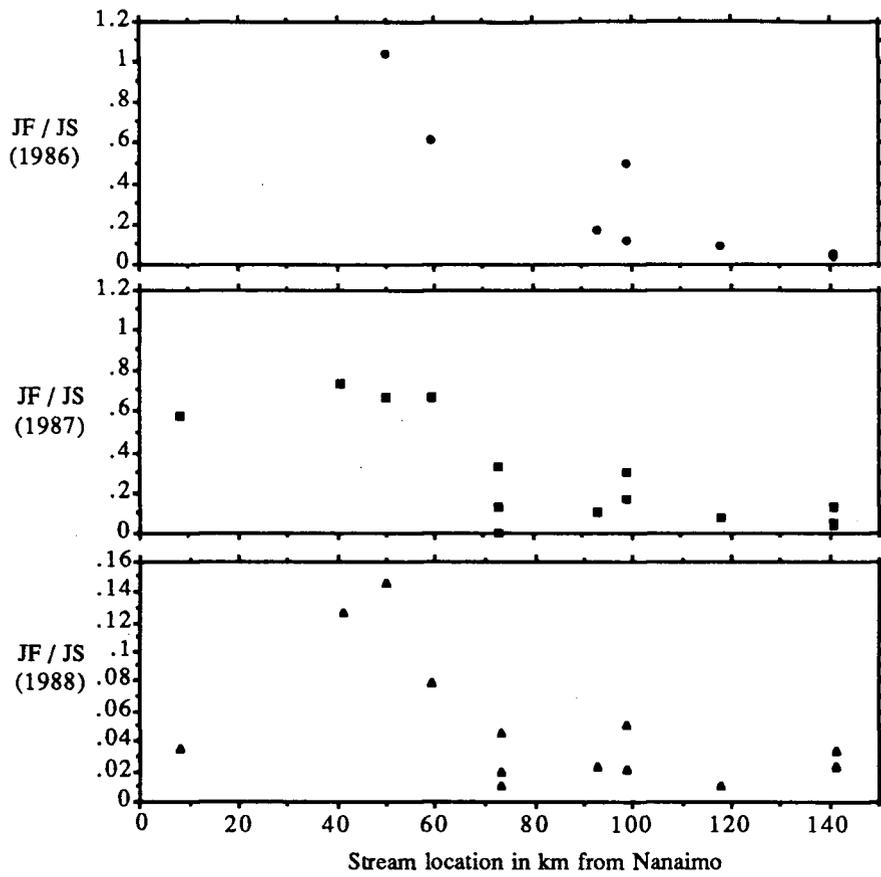


Figure 7.2. Relative contribution of various stocks to Juan de Fuca (JF) and the Johnstone Strait (JS) net fisheries in relation to the latitude of the release location (expressed as a distance to Nanaimo). Note changes to the scale of the 1988 plot.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

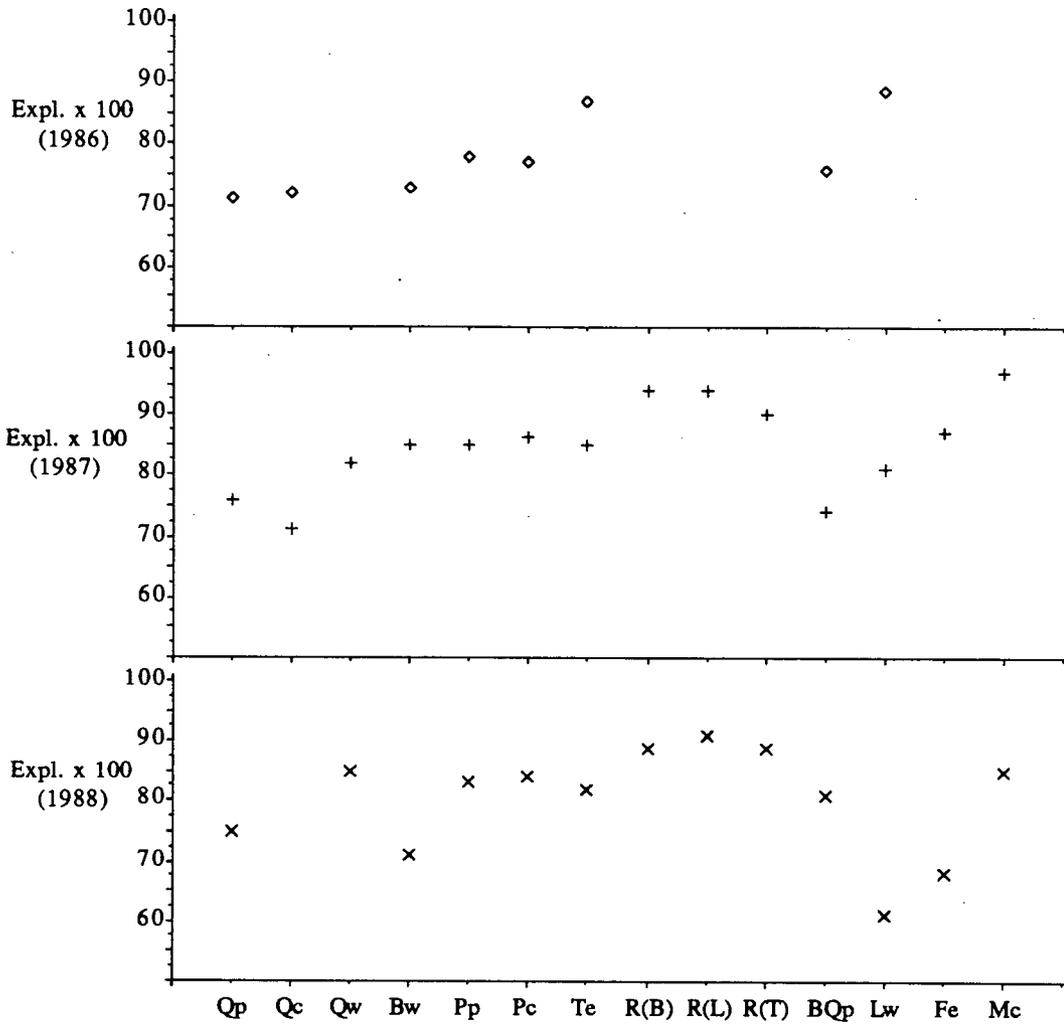


Figure 7.3. Estimates of exploitation rates (x 100) for adults from each stock. The year corresponds to the age 3+ adult return year.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

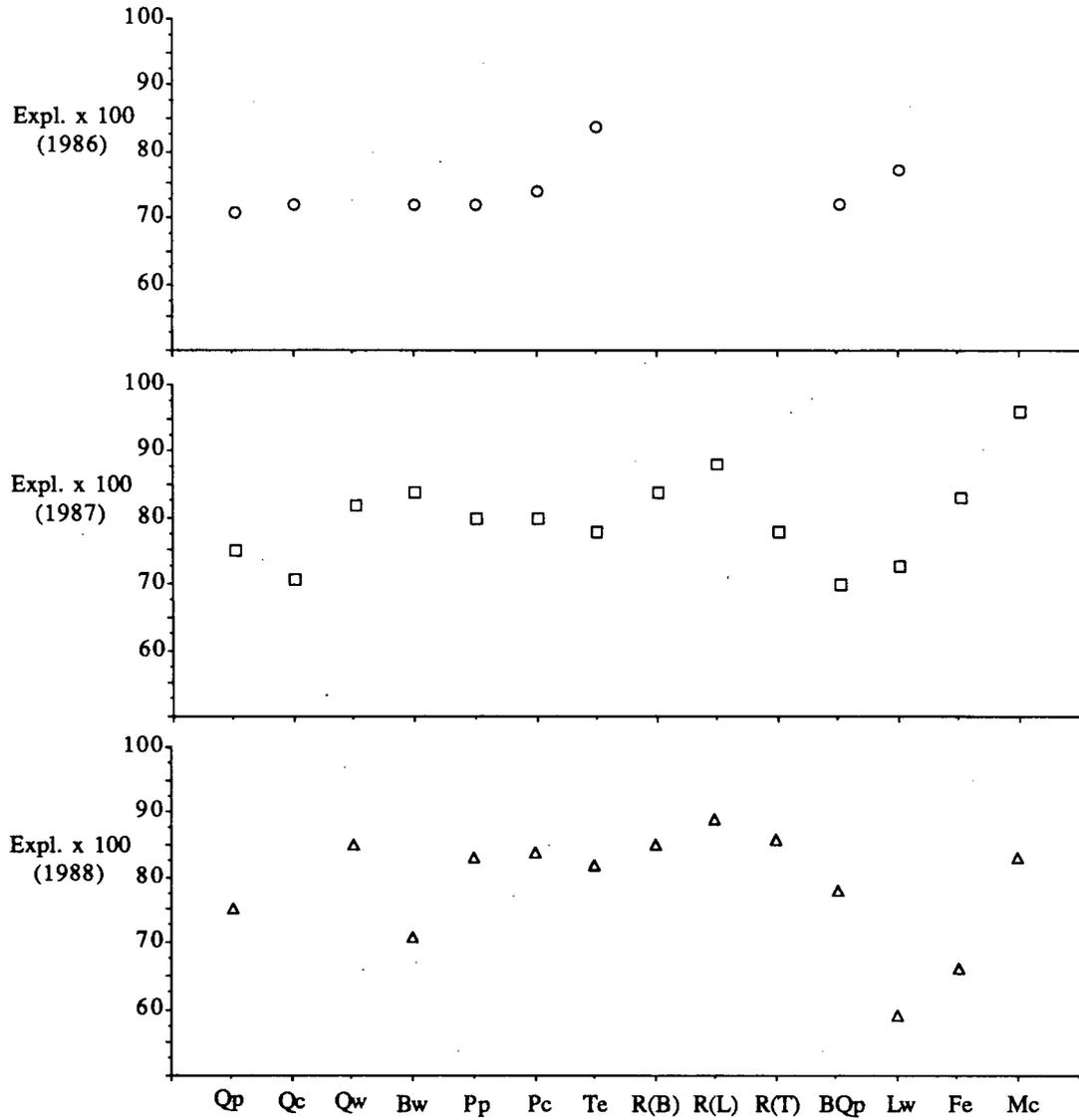


Figure 7.4. Predicted estimates of exploitation rates in the absence of terminal net fisheries in the Strait of Georgia. The year corresponds to the age 3+ adult return year.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

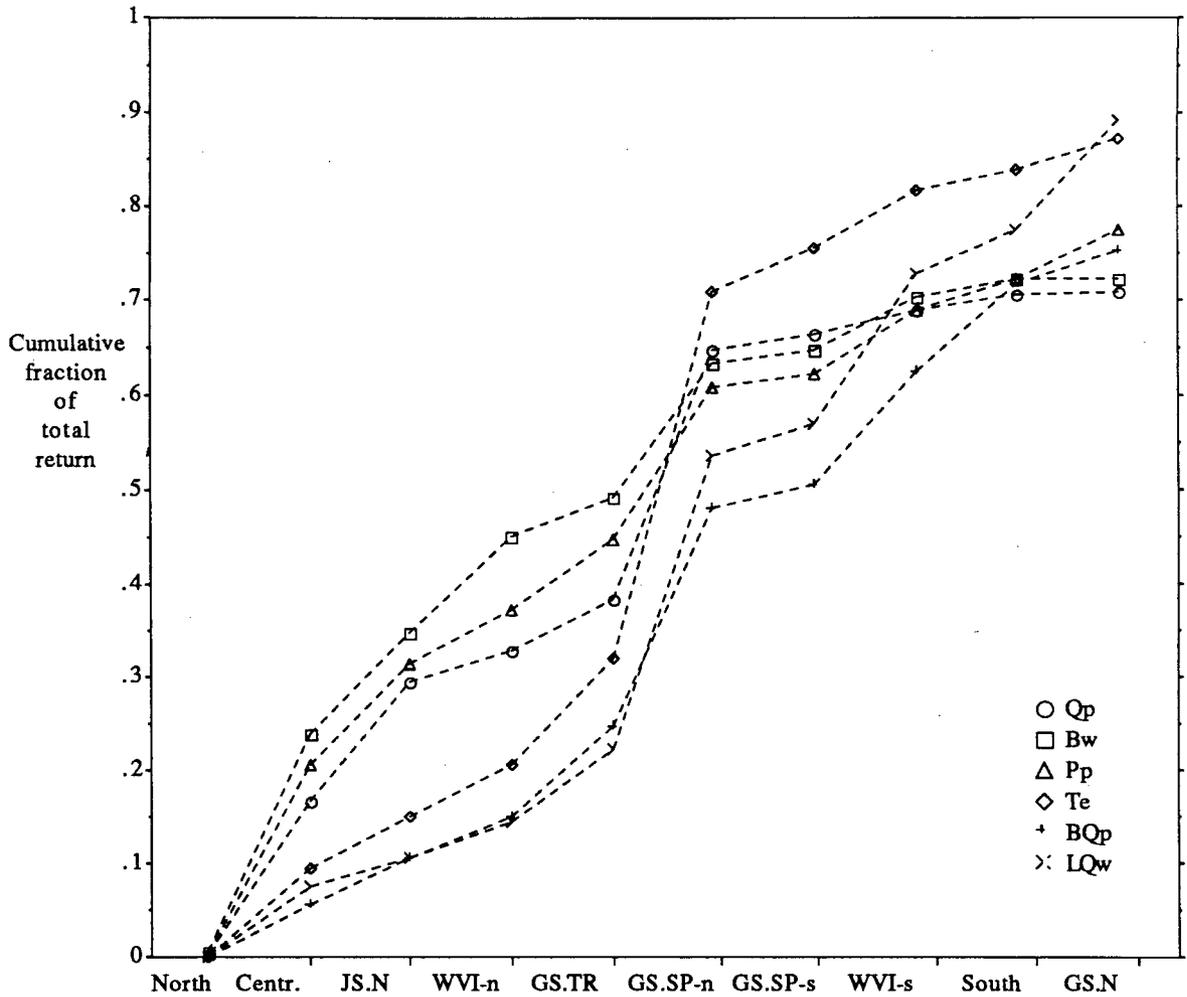


Figure 7.5. Cumulative distribution of catches by fishery of age 3+ adults from each stock expressed as a fraction of the total return for 1986. Stocks from the same stream with nearly identical distributions were considered as one stock purposes of clarity. In this case, catches of colonization and production stocks from the Quinsam River and Puntledge River were pooled separately.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

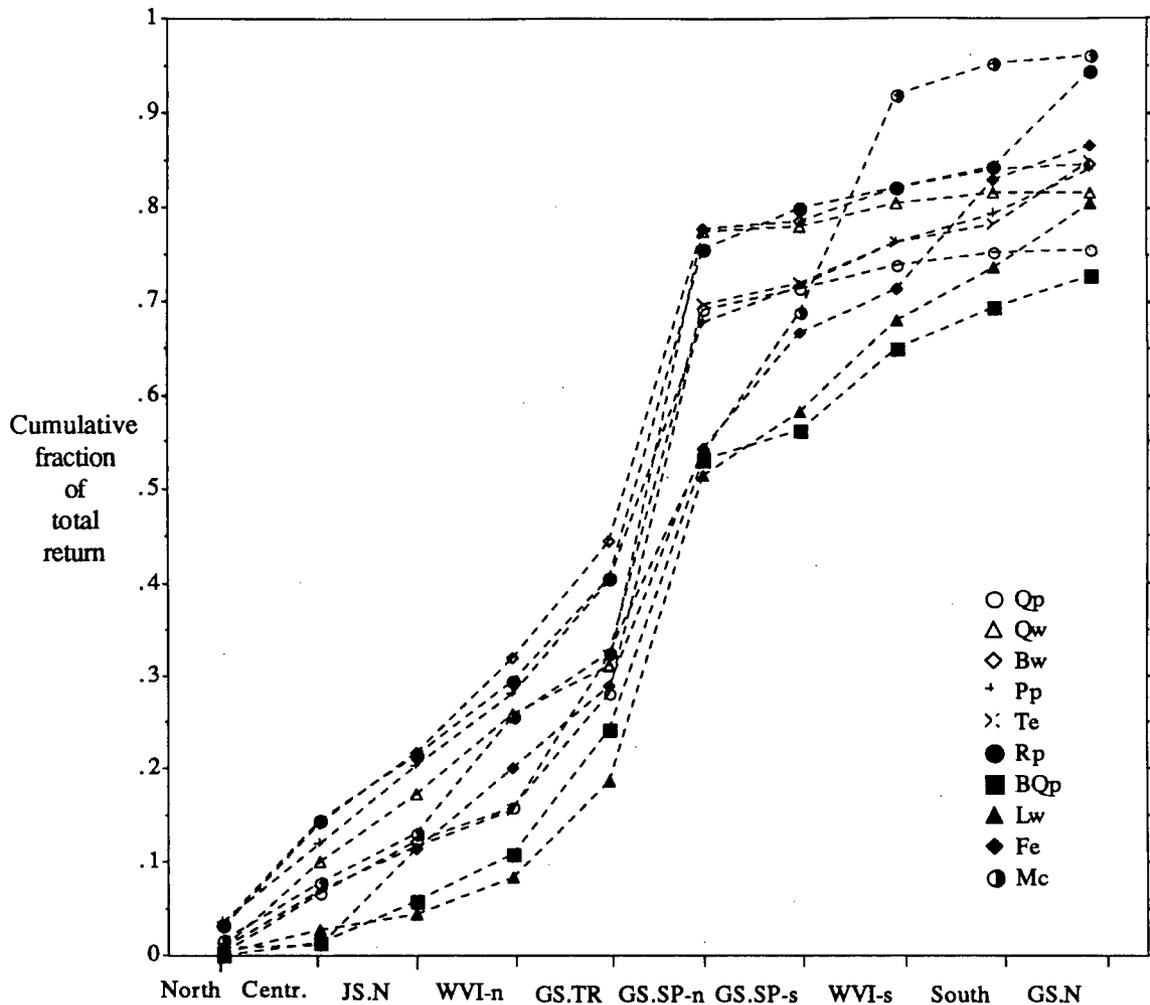


Figure 7.6. Cumulative distribution of catches by fishery of age 3+ adults from each stock expressed as a fraction of the total return for 1987. Stocks from the same stream with nearly identical distributions were considered as one stock for purposes of clarity. In this case, catches of the three stocks from Rosewall Creek were combined, as well as those of the colonization and production stocks from Quinsam River.

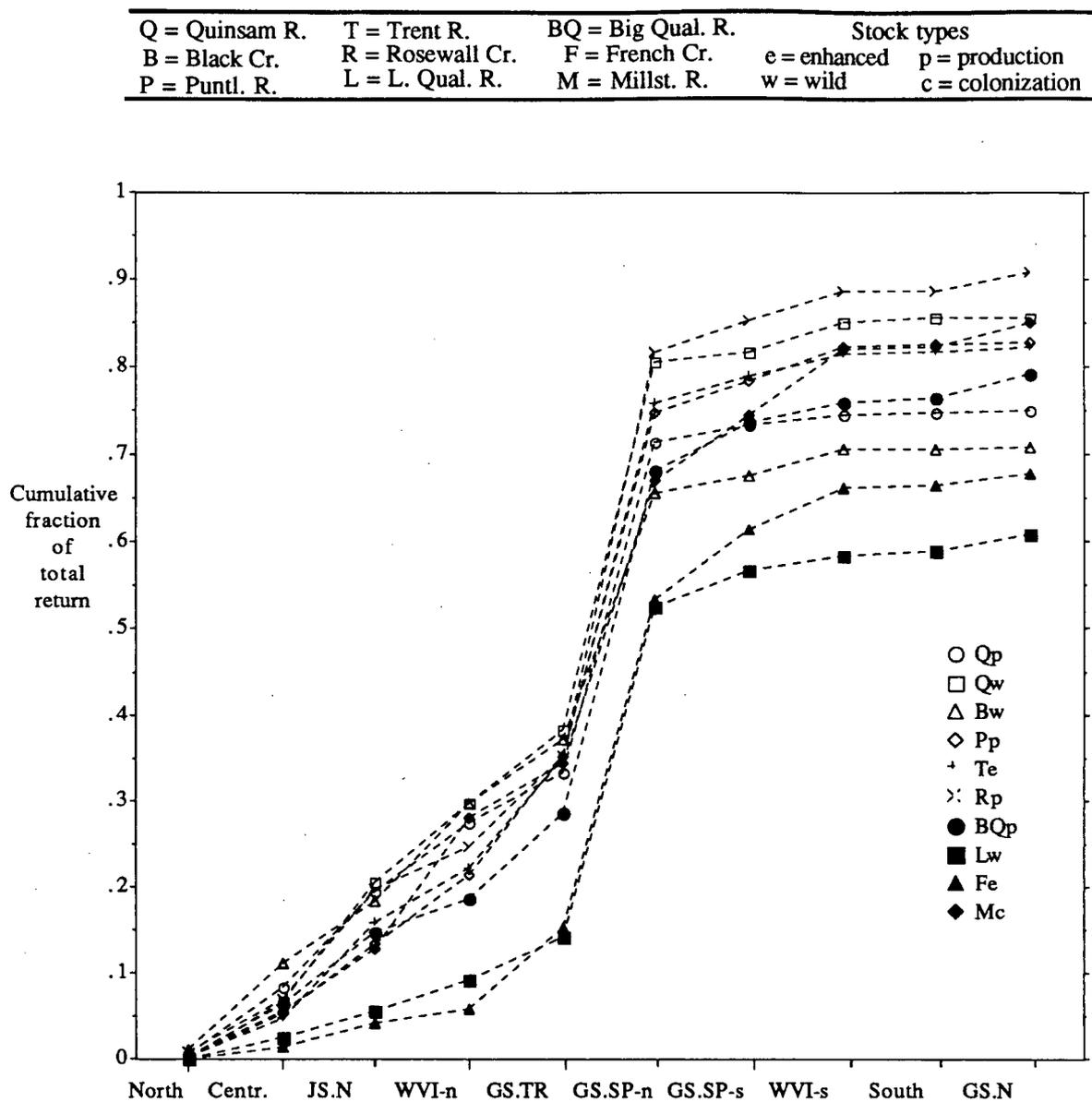


Figure 7.7. Cumulative distribution of catches by fishery of age 3+ adults from each stock expressed as a fraction of the total return for 1988. Stocks from the same stream with nearly identical distributions were considered as one stock for purposes of clarity. In this case, catches of the three stocks from Rosewall Creek were combined, as well as those of the colonization and production stocks from Puntledge River.

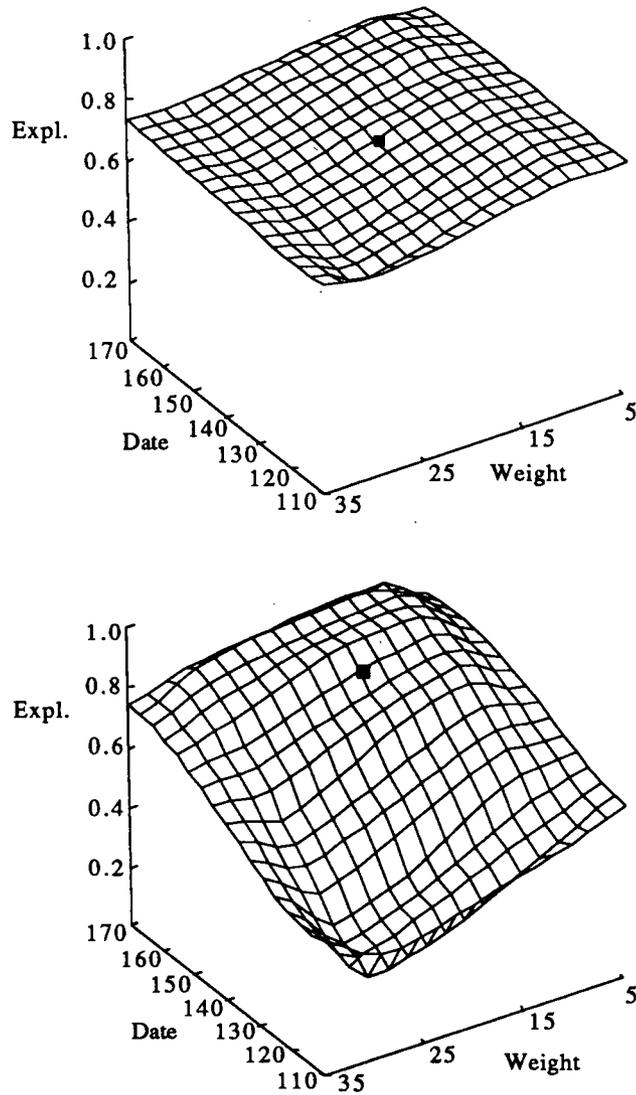


Figure 7.8. Response surface of 1987 exploitation rates in relation to smolt migration date and smolt weight for the Big Qualicum (above) and Little Qualicum (below) stocks. The date is the median date of smolt migration, expressed as a calendar date. The weight is the average weight (in g). The dot indicates the actual time and size of release in 1987. Computations were made over a range of time and size values of 105-165, and 5-35 g. The response surface covers values exceeding this range in order to accentuate the 3-D profile.

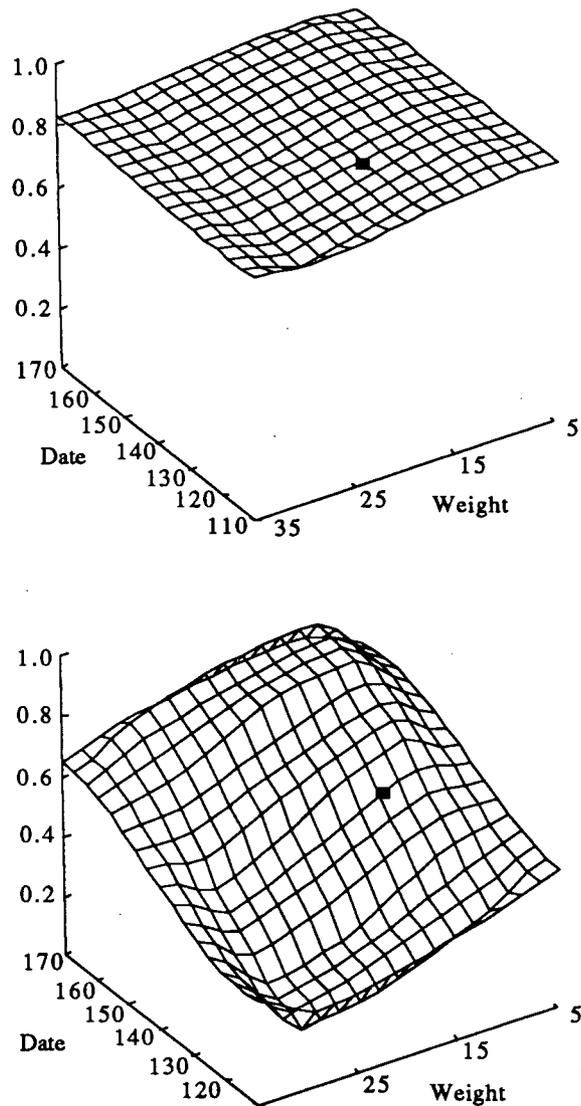


Figure 7.9. Response surface of 1988 exploitation rates in relation to smolt migration date and smolt weight for Big Qualicum (above) and Little Qualicum (below) stocks. The date is the median date of smolt migration, expressed as a calendar date. The weight is the average weight (g). The dot indicates the actual time and size of release in 1988. Computations were made over a range of time and size values of 105-165, and 5-35 g. The response surface covers values exceeding the viable range in order to accentuate the 3-D profile.

Q = Quinsam R.	T = Trent R.	BQ = Big Qual. R.	Stock types	
B = Black Cr.	R = Rosewall Cr.	F = French Cr.	e = enhanced	p = production
P = Puntl. R.	L = L. Qual. R.	M = Millst. R.	w = wild	c = colonization

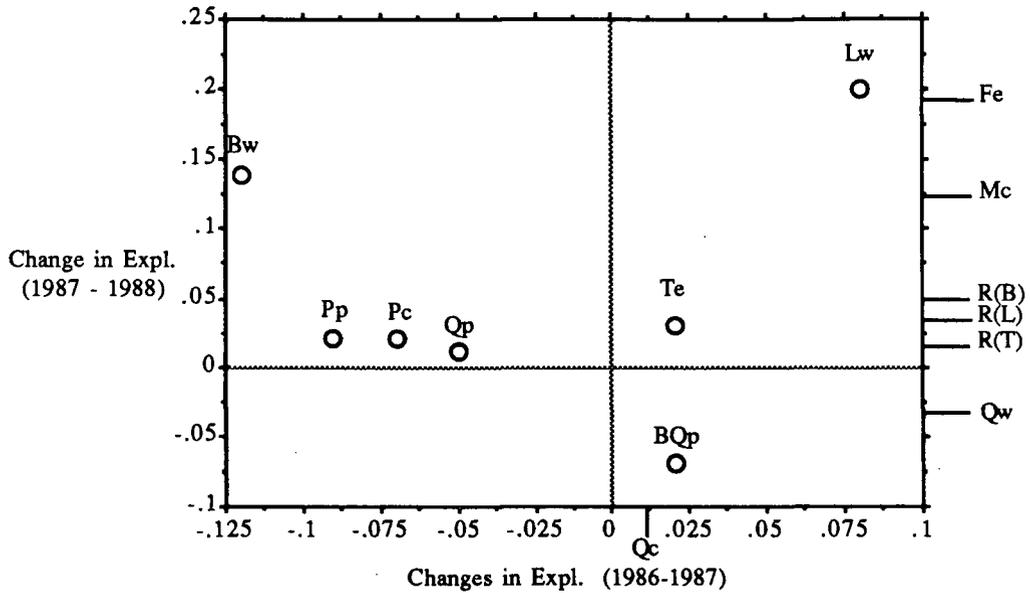


Figure 7.10. Pattern of co-variation in exploitation rates. The co-ordinates of each stock monitored for three consecutive years is expressed in terms of the changes in exploitation rates between the two successive periods. Stocks which were monitored for two years are positioned outside along each axis next to their associated change in exploitation rate.

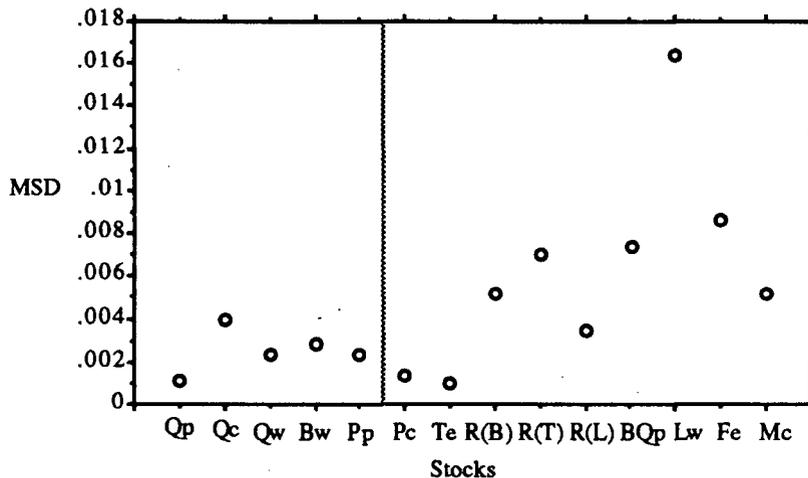


Figure 7.11. Estimates of the mean squared deviations (MSD) from the average exploitation rates on each stock assemblage (delineated by dotted line). Stocks with exploitation rates closest to the average annual level for the assemblage have the lowest MSD value.