Utilization of the Campbell River Estuary by Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) in 1994

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UTILIZATION OF THE CAMPBELL RIVER ESTUARY BY JUVENILE CHINOOK SALMON (*ONCORHYNCHUS TSHAWYTSCHA*) IN 1994

by

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ABSTRACT

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Juvenile salmon population growth and abundance data collected in the Campbell River estuary in 1994 were analyzed to describe chinook habitat use, residency timing, growth, and potential competitive interactions between wild chinook fry, hatchery chinook, and other salmon species. Wild chinook fry densities were highest in estuarine zone sites while hatchery chinook densities were generally higher than wild densities in transition zone sites. Habitat type significantly affected density of wild chinook in the estuary where their densities were greatest at riparian and intertidal island sites. Hatchery and wild chinook juveniles showed different patterns in their seaward emigration timing. The timing of peak abundance of hatchery chinook in the estuary coincided with the peak abundance of wild fry; this was considered a likely period of strong competitive interaction between hatchery and wild chinook salmon. Wild and hatchery chinook juveniles were generally larger at transition zone sites compared to those from the estuarine zone. Growth rates of wild chinook tended to be slightly higher than growth rates of hatchery chinook. The inverse relationship between wild chinook fry size and total salmon biomass, assessed in mid-May, was similar to that established with earlier data, supporting the conclusion that growth of wild chinook in the Campbell River estuary may be density dependent. Close to half the estuarine habitat of the estuary has been degraded due to industrial development since the early 1900s. Recovery of degraded estuarine habitat would improve rearing conditions for wild chinook fry. These measures should be integrated with freshwater habitat improvement.

RÉSUMÉ

Korman, J., B. Bravender, and C.D. Levings. 1997. Utilization of the Campbell River estuary by juvenile chinook salmon (*Oncorhynchus tshawytscha*) in 1994. Can. Tech. Rep. Fish. Aquat. Sci. 2169: 45 p.

Nous avons analysé les données recueillies en 1994 dans l'estuaire de la rivière Campbell sur la croissance et l'abondance de la population de saumons juvéniles pour décrire l'utilisation de l'habitat par les quinnats, leur durée de séjour, leur croissance et les interactions compétitives potentielles entre les alevins sauvages de quinnat, les quinnats issus des écloseries et les autres espèces de saumons. Les densités des alevins sauvages de quinnat étaient les plus élevées aux sites de la zone estuarienne, tandis que celles des quinnats d'élevage étaient dans l'ensemble plus hautes que celles des alevins sauvages dans les zones de transition. Le type d'habitat avait un

effet notable sur la densité des quinnats sauvages dans l'estuaire, où leurs densités étaient au plus haut aux sites du littoral et des îles intertidales. Le calendrier de migration vers la mer était différent chez les quinnats juvéniles selon qu'ils étaient sauvages ou d'élevage. Le moment du pic d'abondance des quinnats d'élevage coincidait avec celui des alevins sauvages; c'était probablement une période où l'interaction compétitive était forte entre les deux groupes de juvéniles. Les juvéniles des deux groupes étaient dans l'ensemble plus gros aux sites de la zone de transition que dans la zone estuarienne. Les taux de croissance des quinnats sauvages étaient dans l'ensemble plus hauts que ceux des quinnats d' élevage. La relation inverse entre la taille des juvéniles sauvages et la biomasse totale de saumon, évaluée à la mi-mai, était semblable à celle qui avait été établie avec des données antérieures, ce qui confirme que la croissance des quinnats sauvages dans l'estuaire de la Campbell peut être dépendant de la densité. Près de la moitié de l'habitat estuarien de la zone a été dégradée par le développement industriel survenu depuis le début du siècle. Le rétablissement de l'habitat estuarien dégradé améliorerait les conditions de croissance des juvéniles sauvages de quinnat. Ces mesures devraient être intégrées aux efforts d'amélioration de l'habitat dulcicole.

1.0. INTRODUCTION

To partially compensate for lost salmon habitat resulting from the creation of a dryland log sort in the Campbell River estuary in the early 1980's, B.C. Forest Products Ltd. constructed four intertidal islands. During 1982-1986, a sampling program monitored juvenile salmonid populations in the estuary and surrounding area, focusing on populations near the intertidal islands. The Quinsam River Hatchery releases 12 million juvenile salmon each year (1989-1993 average) into the Campbell River system. Concern has been raised that hatchery fish could negatively affect wild chinook through increased competition for food resources in the estuary. Levings et al. (1986) used data from the early years of the sampling program to examine differences in the use and residency of hatchery and wild chinook fry and juveniles in the estuary and adjacent coastal habitats. McAllister and Brown (unpubl. manuscr.) examined these same issues using a longer time series (1982-1986 and 1989). They concluded that the hatchery juveniles affect the growth and size of wild chinook fry in the estuary and that increases in rearing capacity resulting from the intertidal islands may be reducing competitive effects on wild chinook growth. They also concluded that in years of high total salmon biomass in the estuary, wild chinook fry appeared to move seaward more rapidly than in low biomass years, potentially exposing smaller chinook to larger predators outside the estuary.

The Salmon Enhancement Program (SEP) and B.C. Hydro have further plans for habitat management and enhancement initiatives for juvenile salmon in the Campbell River and estuary, particularly for chinook. The estuary was therefore sampled in 1994 to update the database on wild and hatchery chinook habitat use, residency timing, growth, and potential competitive interaction. This report summarizes the 1994 beach seine survey data and compares the results with previous information on habitat use, residence timing, and growth of wild and hatchery chinook. The analyses are presented in five parts:

1. the use of a discriminant function to predict the origin of chinook juveniles (hatchery or wild) for fish whose origin could not be determined in the field;
2. estimation of the density and biomass of juvenile salmon in the estuary and a comparison of habitat use and residence timing;
3. calculation of growth rates for wild and hatchery chinook;
4. prediction of wild chinook fry weight based on total salmon biomass; and
5. estimation of the carrying capacity of the estuary for juvenile chinook, coho, and all salmon species combined based on mark-recapture and escapement-biostandard methods.

2.0. METHODS

2.1. Field and Laboratory Procedures

Ten sampling trips were completed, beginning May 12-13, 1994 and extending to August 10-11, 1994 (Table 1, Fig. 1). A full description of the sampling methodology used in the 1994 Campbell River estuary beach seine survey will be available in a later report. To briefly summarize, on the majority of these trips, 25 sampling sites were sampled. Twenty-one of these sites were located in the lower river and estuary area and are considered part of the estuarine zone. Five of the sites were characterized by higher salinities than the estuarine stations and were classified as transition sites. Sampling sites were also classified based on habitat types (marsh, gravel, eelgrass, marsh on artificial islands, gravel on artificial islands, riprap, and riparian) described in Levings and Macdonald 1991. Duplicate beach seines were done at most sites except for six grooves in the islands, where only one set was done (43 site-date combinations with one set, 174 with two sets). When sites were sampled with duplicate beach seines, fish sampled during the first set were released close to the sample area before the second set was taken, but generally these fish were dispersed by tide and river currents. This methodology was consistent with the sampling procedures used in the 1980's. Where large numbers of fish were caught in a set, a subsample was counted and the total for the set estimated based on the proportion of the subsample to the total sample. A small sample of each group of chinook (wild, marked and unmarked hatchery fish) was preserved in 10% formalin and returned to the laboratory for length/weight analysis. Samples of other species of salmon were also obtained. Fork lengths were measured to the nearest mm and weight to the nearest 0.01 gram.

2.2. Determination of the Origin of Chinook Juveniles

There are three groups of chinook juveniles in the estuary; wild fish produced from natural spawning, unmarked fish suspected to be of hatchery origin based on their larger size compared to wild chinook, and marked hatchery fish. Because many hatchery fish are not marked by an adipose clip, a procedure was required to separate wild and hatchery fish. Two methods were used in the field to identify the origin of chinook salmon during the 1994 field season. Early in the season, wild and hatchery fish were categorized by size (hereafter referred to as the field-ID method) with the larger fish being classified as hatchery in origin. As the season progressed and size distributions of wild and hatchery fish increasingly overlapped, it became inappropriate to use size for determining origin. Consequently, a substantial portion of the catch was classified as 'unmarked', meaning that the fish were likely hatchery in origin but that the true origin (i.e. hatchery or wild) could not be definitively determined in the field. Of the total catch of chinook in 1994 (34,049 fish), 3,747 were designated as marked (hatchery origin), 25,858 as unmarked (unknown origin), and 4,444 as wild. A new technique based on analysis of otolith rings (hereafter referred to as the otolith method) was used to categorize a small subsample (148) of fish. A previous study has shown

that the otolith technique can determine the origin of chinook smolts as hatchery or wild with an accuracy of approximately 85% (Zhang et al. 1995).

The accuracy of the field-ID method was not quantified. Presumably it was relatively accurate early in the season, but became less certain as the season progressed. In contrast, the accuracy of the otolith technique was known. However, of the total chinook catch, only 148 categorizations based on otoliths were made. Thus, the otolith method, while more accurate than the field-ID method, identified the origin of only a small proportion of the entire 'unmarked' part of the chinook catch.

To analyze catch data, we used a simple (two-way) discriminant function to categorize wild and hatchery fish. The discriminant function used the descriptors: sampling date, weight, and fork length to predict the hatchery or wild origin of fish based on otolith data. The otolith data set was used to compute the function based on the assumption that the categorization of fish using the otolith method represents their true origin. The function was applied to the entire length-weight data set for chinook and predictions of origin based on the discriminant function were compared to origin determinations made using the field-ID method.

2.3. Density and Biomass of Juvenile Salmon

Density (numbers 100 m^{-2}) and biomass (g wet wt 100 m^{-2}) estimates were computed for all species of salmon sampled in the Campbell River estuary. When more than one seine set for each sampling site-date combination was taken, all length-weight and catch data were combined. Density was computed by dividing the total catch by the estimated area swept by the beach seine.

Calculating the average density of wild and hatchery chinook salmon for each site and sampling trip required several additional steps :

1. length-weight data and sampling date were used to predict the origin (hatchery or wild) for individual fish which were categorized as unmarked in the field, using the discriminant function;
2. the average number of wild and hatchery fish within the unmarked fish samples was determined by multiplying the catch of unmarked fish by the proportions of hatchery and wild fish calculated from step 1; and
3. the average number of fish designated as wild or marked in the field were added to the average catch of wild and hatchery fish computed from step 2. These combined values gave the average catch (across 1 or 2 seine sets) of wild and hatchery chinook for each sampling site-trip combination.

Biomass of salmon juveniles at each sampling site-date combination was computed by multiplying the average weight (for each sampling site-date combination) of each species by the average density. For chinook, this calculation was done

separately for wild- and hatchery-origin fish using average weights calculated by the method described in section 2.4 below. Results were expressed as numbers or biomass (g wet wt) 100 m^{-2}, the area swept by the beach seine at most sites. Assuming seine efficiency is equal among all sites, the following correction factors were used to standardize the areas for sampling sites where the area swept was not 100 m^2:

Site	Correction Factor
47	2.0
1	1.33
11	0.67
Island 3 (14-17, 141, 151)	6.67

Analysis of variance (ANOVA) was used to test for the effects of site location (estuarine or transition zone) and sampling date on wild and hatchery chinook densities. ANOVA was also used to test for the effect of habitat type (marsh, eelgrass, intertidal island-gravel, intertidal island-marsh, gravel, riparian, riprap) on wild and hatchery chinook density and biomass. For statistical tests, these data were $\log_e$-transformed ($\log_e(X + 1)$) to compensate for non-normality and to reduce heteroscedasticity.

2.4. Growth Rates for Chinook Salmon

The average fork length and weight of hatchery and wild chinook for each sampling site-trip combination was computed as follows:

1. all length-weight data for each site-date combination were pooled across seine sets;
2. the length-weight data and sampling date were used to predict the origin of fish categorized as 'unmarked' in the field using the discriminant function; and
3. the average length and weight for each site-trip combination was computed using data pooled across seine sets and dates within a trip, including the data for fish originally designated as 'wild' or 'marked' (hatchery) in the field.

ANOVA was used to test for the effect of site location (estuarine and transition zone) on wild and hatchery chinook fork length. Growth rates were computed by regressing the average fork length and weight (across all sampling stations) on the median sampling date for each trip. Weight data were $\log_e$-transformed. Analysis of covariance was used to test for differences in growth rates between hatchery and wild fish.

2.5. Density Dependent Growth and Evidence for Changes in Carrying Capacity

McAllister and Brown (unpubl. manuscr.) developed a relationship between wild chinook fry weight and total salmon biomass using data collected on May 21 from

1982-1986. They also included a 1989 data point in the graph of their results, but did not use it to estimate their regression. To compare the 1994 data point with their relationship, wild fry weight and total salmon biomass (sum of biomass of all salmonid species computed based on methods described in Section 2.3) were computed as averages across all sampling stations and sets from May 17-19, the trip closest to May 21. While there were non-salmonid species collected in the estuary in 1994 and in previous sampling seasons, these species were not included in the total biomass values used in the density dependent relationship with chinook fry weight. To estimate the total salmon biomass average for 1994, all Island #3 stations (Fig. 1) were treated as a single site by using the mean of catches at Island #3 sites sampled per trip. This specific treatment of data from the Island #3 sites was done for this analysis to make the 1994 data point comparable with McAllister and Brown's results. The rationale for this was to avoid bias and over-representation of these sites in the averages. Data from Fig. 9 of McAllister and Brown (unpubl. manuscr.) were used to re-estimate the regression, using only the 1982-1986 data. A semi-log relationship between wild fry weight and total salmon biomass was also fitted to all the data excluding the 1989 data point.

2.6. Carrying Capacity for Juvenile Salmon

2.6.1. Population estimates based on mark-recapture calculations

Very approximate estimates of the carrying capacity of the Campbell River estuary for chinook, coho, or all salmon species combined was derived: 1) by estimating total population sizes using 1994 data; and 2) assuming that estuary habitat in 1994 was fully saturated. Estimates of population size in the estuary were made using Chapman's adjusted Peterson estimate (Ricker 1975, Eqn. 3.7):

$$[1] \qquad N = ((M + 1) * (C + 1)) / (R + 1),$$

where N is the estimated population size of wild and hatchery chinook combined, M is the total number of tagged chinook released from the Quinsam River Hatchery and seapens (treated as the marked individuals in this mark-recapture estimate), C is the total catch of wild and hatchery chinook across all sets and stations per sampling trip and R is the number of recaptured chinook that were marked (across all sets and stations per sampling trip). These calculations were repeated to get an estimate of the hatchery chinook population size (C = hatchery chinook only), coho population size (M = total coho tagged releases, C = total coho catch, R = total recaptured coho that were marked) and population size of all salmon species combined (C = total salmon catch, M = total chinook tagged releases and R = total recaptured chinook that were marked). The method assumes that all estuary and transition zone sampling sites are within a closed system, without emigration or immigration. The 95% confidence limits for the population size estimates were obtained using Pearson's approximation (Ricker 1975, Appendix II):

[2] $$R = R + 1.92 \pm 1.96 * (R + 1)^{.5}$$

and then substituting the confidence limit of R into Eqn. 1. The size of the wild chinook population in the estuary was calculated by subtracting the estimated total number of hatchery chinook released in 1994 from the adjusted Peterson estimates of total chinook population size (wild and hatchery combined).

2.6.2. Population estimates based on adult escapement and biostandards

To provide an alternate method for estimating the total number of wild chinook fry in the estuary, we used SEP biostandards (Lill and Tautz 1983) in conjunction with the combined natural escapement to the Campbell and Quinsam rivers in 1993. The total escapement was multiplied by 0.5 based on an assumed sex ratio of 50:50. The number of females was multiplied by a fecundity of 6000 eggs/female (Jim Van Tine, Manager, Quinsam River Hatchery, pers. comm.) to estimate the total egg deposition. The resulting number of fry was calculated by multiplying the egg deposition by either a minimum assumed egg-fry survival rate of 0.15 (Lill and Tautz 1983) or a maximum rate of 0.40 (Bradford 1995).

3.0. RESULTS

3.1. Determination of the Origin of Chinook Juveniles

The discriminant function explained 63.3% of the variance between hatchery and wild fish groups in the otolith data set based on sampling date, fork length and weight (Table 2). Fork length and weight were the dominant descriptors in the discriminant function as shown by their large canonical loadings. The discriminant function was highly significant (Wilks' $\lambda = 0.599$, $p < 0.001$). Of the 148 observations in the otolith data set (Table 3a), the discriminant function categorized 126 fish (85% of 148 observations) into the correct hatchery (104) or wild category (22).

The discriminant function accurately predicted field-ID origin; 98% of the 1141 fish categorized as hatchery or wild by the field crew were categorized correctly by the discriminant function (Table 3b). Of the 752 chinook classified as unmarked by the field crew where weight and length measurements were taken (bottom row, Table 3b), the discriminant function classified 98 fish (13%) as wild in origin and 654 (87%) as hatchery in origin.

The performance of the discriminant function justified its use to classify fish designated as 'unmarked' during the survey into hatchery - or wild - origin categories. However, it should be recognized that classification based on the discriminant function is subject to uncertainty as is the field-ID method. Fig. 2 shows the probability of each fish in the otolith data set being designated as wild in origin using the discriminant function. The symbols denote the 'true' origin of the fishes based on otolith analysis. Fish with probabilities >0.5 would have a discriminant-predicted wild origin, while fish

with probabilities <0.5 would be designated as originating from the hatchery. Hence triangles above the 0.5 line denote hatchery fish which have been incorrectly classified as wild, and filled circles below the line denote the reverse. Uncertainty for discriminant-based classification is greatest for samples near the 0.5 line, and decreases at increasing distance from the line. Fork lengths for all fish incorrectly classified ranged from 72-99 mm, with a mean length of 85 mm. The bulk of fish designated as unmarked had fork lengths ranging from 75-100 mm, a similar range to the one where uncertainty in predictions of the discriminant function is highest. There is no way to quantify how uncertain the discriminant predictions are for the unmarked group of chinook, since the true origins of these fish are not known.

3.2. Spatial and Temporal Patterns in Density and Biomass of Juvenile Salmon

3.2.1. All salmon species

At both estuarine and transition zone sites, chinook and coho salmon were the dominant species while chum, pink and sockeye salmon were of secondary importance (Fig. 3a). Coho were most abundant at eelgrass and island gravel sites (Fig. 3b). Biomass of chinook and coho were similar for most habitat types, with greater chinook biomass at gravel, riparian and riprap sites (Fig. 3b). In the estuarine zone, coho were the dominant species at all habitat types except island marsh, riparian and riprap sites where chinook dominated (Fig. 3c). In the transition zone, coho dominated eelgrass sites while chinook was the dominant species at gravel sites (Fig. 3d).

Chinook showed peak biomass levels on May 25-26, while peak coho biomass occurred on June 8 (Fig. 4a-b). Chum biomass was relatively consistent throughout the summer. Pink and sockeye biomass declined to very low levels after mid-June and mid-May, respectively. Based on habitat use, residency timing and biomass, coho and chinook salmon clearly could have the greatest competitive impact on wild chinook juveniles relative to the other salmon species.

3.2.2. Chinook salmon

ANOVA results showed that wild chinook fry density was significantly higher in the estuarine zone than in the transition zone, while the opposite pattern occurred for hatchery chinook (Table 4). Sampling date was also a significant factor on both wild and hatchery chinook densities. The interaction effect of sampling location - date (zone* date row in Table 4) was significant for wild chinook densities but not for hatchery chinook. Thus, differences in densities between transition and estuarine zones was dependent on sampling date for wild chinook only. Hatchery chinook density was higher in the transition zone than in the estuarine zone on all 10 sampling trips, while wild chinook densities were higher in the estuarine zone on 8 trips.

Habitat type was a statistically significant determinant on the density and biomass of hatchery chinook at the estuarine zone sites, while only density was

significantly affected by habitat type for wild chinook (Table 5). For wild chinook, density and biomass were greatest in riparian, island marsh and island gravel sites. For hatchery chinook, biomass was greatest at eelgrass and island marsh sites, while densities were greatest at these habitat types as well as at island gravel sites.

To examine the potential for competitive interactions between seapen-reared chinook and wild chinook in the Campbell River estuary, we used coded wire tag (CWT) data to estimate the spatial distribution of seapen-reared hatchery chinook released into Discovery Passage from a marina several kilometers south of the river mouth. Of the 494 hatchery fish whose CWTs were read in 1994, only 26 originated from the seapens (Table 6a). The majority of these fish were found in the transition zone sites, thus most seapen fish apparently did not migrate inshore to the estuary. Fish with CWTs originating from the seapens found in the transition zone were much larger (almost 3-fold) than hatchery origin fish in either zone, or seapen-reared fish in the estuarine zone (Table 6b). Thus, larger fish from the seapens may remain further offshore following release than smaller ones. These results should be viewed very cautiously as they are based on a sample of only 0.008% of the total seapen releases of 320,750 fish released on May 3 and 6 (0.1% of the 25,515 CWT marked seapen fish released).

When data were averaged across both estuarine and transition zone stations, temporal patterns in densities of hatchery and wild chinook were similar at the beginning of the season, but differed as the season progressed (Fig. 5a). Both hatchery and wild chinook reached peak densities on May 25-26, but densities of wild fish dropped off rapidly after this date, while hatchery densities declined only slightly following the peak before declining sharply by the end of June.

Hatchery and wild chinook showed different patterns in their emigration timing from estuarine sites (Fig 5b). Wild fry density reached its peak at estuarine stations on May 25-26, and remained at lower but relatively constant values for much of the summer (June 1 to July 15). By the last sampling trips (late July, mid August), most of the wild fry were no longer found at the estuarine stations, corresponding to an estuarine residence time of about 3 months. Hatchery chinook reached their peak density in the estuarine zone on May 17-19, the first sampling trip following the final release date from the Quinsam River Hatchery. Their densities in subsequent sampling trips decreased much faster than for wild fry, and by mid June, most hatchery fish had migrated out of the estuarine zone. Average densities of hatchery fish in the estuarine zone decreased 88% (41.2 → 4.8 fish 100 m^{-2}) from peak values in slightly more than 20 days. Hatchery chinook occurred in low numbers on the July 13-15 sampling trip and were found in only very low densities on trips after this date. Based on these observations, estuarine zone residence for hatchery chinook was estimated to be 1.5 months.

Temporal patterns of densities of wild and hatchery chinook in the transition zone were very different from patterns in the estuarine zone (Fig 5c). Wild fry were

virtually absent in the transition zone in early and mid May, and their peak densities on May 25-28 in the transition zone coincided with the peak in the estuarine zone. In the transition zone, wild fry density was variable between trips throughout the summer, and virtually no wild fish were found in the transition zone after mid July. Based on these data, residency for wild chinook in the transition zone spans from late May to mid-July, or about 50 days. Hatchery chinook density was consistently high in the transition zone from early May to late June and did not increase following the peak abundance in the estuary. The relatively low numbers of hatchery chinook on the June 8 sampling trip may have been related to the small number of sites sampled during this trip (only 6 out of 26 sites). By late July, densities of hatchery fish in the transition zone were reduced by 95% relative to earlier dates, presumably because fish moved offshore.

3.3. Size and Growth Rates of Juvenile Chinook Salmon

Size of both wild and hatchery chinook juveniles was generally larger at transition zone stations than at estuarine zone stations. Fork length was significantly higher at transition zone sites for wild- and hatchery-origin chinook juveniles when averaged across sampling trips (Table 7). Increases in fork length and weight for hatchery and wild juvenile chinook are presented in Fig. 6. Three different data sets were used: 1) all weight-length data which include fish designated as 'unmarked' during the sampling which were subsequently categorized as hatchery or wild in origin using the discriminant function, as well as data for fish whose origins were determined in the field based on size (n = 1893); 2) only data where the origin of fish had been determined in the field based on size (n = 1141); and 3) only data where the origin of the fish had been determined based on the otolith technique (n=148). The comparison between wild and hatchery chinook growth was done using all three data sets to determine if errors in the classification of chinook origin would effect our conclusions on growth rate differences between hatchery and wild fish. Weight and fork length of wild and hatchery chinook increased over time in all 3 data sets and all size-at-time regressions had highly significant positive slopes. Regressions were not calculated using the otolith data due to limited degrees of freedom.

Growth rates of wild chinook tended to be higher than hatchery growth rates (Fig 6). Analysis of covariance demonstrated that growth rates of wild fish were statistically significantly higher than hatchery fish for all regressions except the regression based on the field-ID data (Table 8). The wild and hatchery mean fork lengths (i.e. intercepts) were significantly different for the field-ID regression ($p < 0.001$), but intercept differences for the other regressions could not be tested using ANCOVA because the slopes were significantly different (i.e. failure of homogeneity of slopes test).

3.4. Density Dependent Growth and Evidence for Changes in Carrying Capacity

To place the 1994 results within the context of McAllister and Brown's (unpubl. manuscr.) density dependent relationship (Fig. 7), two estimates of the average 1994 chinook wild fry weight between May 17-19 were calculated. The first estimate, 1.45

grams, was based on data for chinook which were identified as wild fry in the field (145 fish) during this sampling period. The second estimate, 1.54 grams, included data for these same fish as well as from 5 additional fish which were designated as unmarked during collection, but which the discriminant function categorized as wild (150 fish total). The two means are very close because they are estimated from virtually the same population of weight data, thus potential error in the wild fry weight estimate on the May 17-19 sampling trip due to uncertainty in the origin of chinook will have little effect on our results. The 1994 data point, using the field-ID data only, lies reasonably close but slightly above the relationship developed by McAllister and Brown (Fig. 7).

3.5. Carrying Capacity for Juvenile Salmon

3.5.1. Population estimates based on mark-recapture calculations

Figures 8a-8c show estimates of total juvenile population sizes in the Campbell River estuary for chinook, coho and all salmon species for each sampling trip. These estimates represent the population size at the time of marking, and not the population size at the time of each sampling trip. Marked chinook were released from the Quinsam River Hatchery or seapens in Discovery Harbour and Hidden Harbour marinas between May 3 and May 18, thus all marked chinook had been released by the end of trip 2 (May 17-19). Estimates of population size for chinook and all salmon species combined (which are based on the number of marked chinook) in mid to late May should have stabilized after this date assuming that the timing of marked and unmarked fish leaving the sampled area of the estuary is similar. Coho were released between May 27 and June 1, thus total coho population size estimates should be stable after this date.

A total of 43,742 fish were tagged of the estimated 1,128,886 coho released from the hatchery in 1994. The estimated population size of coho at the time of release (May 27 - June 1) ranged from 0.4 to 1.1 million fish with an average across release dates after June 1 of 0.9 million (Fig. 8b). The mean value was close to the true number of releases, but also includes wild fish. The calculated population size therefore underestimates the true size. Total 'population' size of all salmon species combined in early-mid May (Fig. 8c) ranged from 3.1 to 10.4 million when estimated from data from different trips, with an average of 6.9 million fish.

A total of 288,644 fish were tagged of the estimated 2,121,683 chinook released from the hatchery and seapens in 1994. The adjusted Peterson estimate represents the actual population size of chinook hatchery fish at the time of marking, thus N estimated from Eqn. 1 should be equal to the total number of marked fish released. This provides a check of the accuracy of the adjusted Peterson estimate to calculate population size for hatchery chinook. Hatchery chinook population estimates beyond May 19 ranged from 1.6 to 3.3 million fish when estimated from data from different trips, with an average of 2.0 million fish. These values are reasonably close to the actual number of chinook released of 2.1 million fish. The total population of all chinook (wild

and hatchery combined) ranged from 2.1 to 10.0 million fish, with an average of 4.8 million fish (Fig 8a). Unfortunately, there is no way to test how accurate this total estimate is, but there is some comfort in the correspondence between the estimated and true hatchery population sizes. The estimated population size of wild chinook fry, calculated by subtracting the known number of hatchery releases (2.1 million) from the mean total chinook population size from May 19 to August 11 (4.8 million fish), was 2.7 million fish.

3.5.2. Population estimates based on adult escapement and biostandards

Wild chinook fry production from the Quinsam and Campbell rivers in 1994 was estimated at 0.5 million fry based on the 1993 escapement of 1,071 adults, a 15% egg-fry survival rate, 6,000 eggs/female and an assumed sex ratio of 50:50, about one-fifth the Peterson-based population estimate for wild chinook fry in the Campbell River estuary. When a maximum egg-fry survival rate of 40% (Bradford 1995) was used in the biostandard calculations, the wild chinook fry estimate increased to 1.3 million fish. Assuming a 15% egg-fry survival rate is correct, the natural escapement to the Campbell and Quinsam rivers would have had to be 6,000 fish to produce the numbers of wild fry estimated from the mark-recapture analysis (2.7 million). If a 40% egg-fry survival rate is more accurate, only 2,200 spawners would be required to generate the same number of wild fry estimated from the mark-recapture method.

Chinook escapements to the Quinsam and Campbell rivers combined, ranged from 1,071 to 10,028 adults between 1973 and 1993, with an average of 3,836 adults (Fig. 9). The increase in returns to the Quinsam River clearly mirrors the decline in escapement to the Campbell River and may reflect the homing of hatchery chinook to their natal water from the Quinsam River. When the above biostandards were applied to the 1973 to 1993 range in combined escapements to the Campbell and Quinsam rivers, estimated wild fry production ranged from 0.5 - 4.5 million fish with an average of 1.7 million based on the 15% egg-fry survival rate, and 1.3 - 12.0 million with an average of 4.6 million based on the 40% egg-fry survival rate.

4.0. DISCUSSION

4.1. Determination of the Origin of Chinook Juveniles

McAllister and Brown (unpubl. manuscr.) used an alternative method developed by Brown et al. (1987) to estimate the total number of hatchery and wild chinook caught on each trip. Their method was based on principles of a Peterson population estimate. The number of marked chinook in each sample was divided by the marked: unmarked ratio for chinook released from the Quinsam River Hatchery which gave an estimate of hatchery fish in the catch. If this estimate was greater than the total number of chinook in a seine haul, then the hatchery estimate was set equal to the total number of chinook. The estimated number of wild chinook was the difference between total chinook and hatchery chinook.

The excellent predictive ability of the discriminant function to estimate the origin of chinook juveniles justifies its use in this analysis. However, there is uncertainty both in the field-ID and discriminant methods for classifying fish as hatchery or wild in origin. The discriminant function accurately predicted both the field-ID and otolith-based classifications, but uncertainty in discriminant predictions was greatest for fish in the unmarked category (i.e. those fish whose origins could not be definitively determined in the field). The alternative to using the discriminant function to categorize unmarked fish was to eliminate all fish designated as unmarked from the analysis. However, unmarked fish represented 40% of the available weight-length measurements for chinook and 76% of the total catch. Eliminating these data from the analysis represented a huge loss of information and would likely introduce serious biases into subsequent analyses.

4.2. Spatial and Temporal Patterns in Density and Biomass of Juvenile Salmon

Chinook and coho salmon were the dominant species at both estuarine and transition zone sites. Based on habitat use, residency timing and biomass, coho and hatchery chinook salmon clearly could have the greatest competitive impact on wild chinook juveniles. Levings et al. (1986) concluded that the potential for interaction in 1982-1983 between hatchery and wild chinook in the Campbell River estuary was greatest in the transition zone, where hatchery fish were most abundant and because hatchery releases in those 2 years occurred when catches of wild chinook were highest in this area. Data from 1994 show the same trend in the distribution of hatchery chinook, but releases occurred in early to mid-May, probably slightly before most of the wild fish migrated to the transition zone. An alternate interpretation of these data is that the potential for wild-hatchery chinook interaction is greatest in the estuarine zone where wild chinook were most abundant, especially in May when hatchery chinook were being released and made extensive use of the estuarine area. This later interpretation supports McAllister and Brown's (unpubl. manuscr.) speculation that density dependent competitive effects were more likely in the estuary than to seaward. The timing of peak abundance of hatchery chinook in the estuarine zone coincided with the peak abundance of wild chinook fry and may have been a period of strong competitive interaction between hatchery and wild chinook.

There was no lag between the peak chinook densities in the estuary and the peak densities in the transition zone indicating migration from estuarine to transition zones as there was in 1982 and 1983 (Levings et al. 1986). This difference between 1994 and earlier years might be explained if the 1994 sampling trips missed peak densities in the estuarine and transition zones. However this was not likely as the estuary was sampled weekly from May 12 to June 15 in 1994, and a similar sampling frequency was used in previous years. An alternate explanation is that if hatchery chinook entrained wild chinook in their migration to the transition zone, the earlier 1994 release dates compared to previous years would have resulted in an earlier migration of wild fry to the transition zone. Hatchery chinook density was consistently high in the

transition zone from early May to late June and did not increase following the peak abundance in the estuary. This agrees with Levings et al.'s (1986) results who found that hatchery chinook were abundant in both estuarine and transition zones by early May in 1982 and 1983 following releases on May 5, a date close to the release dates in 1994 (May 12-18). The high density of hatchery chinook in the transition zone during the first sampling trip was surprising because the high catches occurred at about the same time (May 12-13) as the earliest production releases from the Quinsam River Hatchery. Based on analysis of CWT data, most of the hatchery chinook caught during early and mid May in the transition zone originated from the Quinsam River Hatchery and not from the seapens. Thus, some Quinsam River Hatchery fish migrate through the entire estuary to the transition zone in about 1 day.

4.3. Size and Growth Rates of Juvenile Chinook Salmon

Size of juvenile chinook salmon released from the Quinsam River Hatchery has been shown to be an important determinant of survival (Morley et al. 1996) and therefore is of direct relevance to our study. Size of both wild and hatchery chinook juveniles was generally larger at transition zone stations than in the estuarine zone. It is not possible to determine whether the differences in size between fish in transition and estuarine zones arose from size-dependent migration (i.e., larger fish migrating from the estuarine to transition zone), due to actual differences in growth rates, or due to better catchability of larger fish in transition zone habitats.

Wild chinook fry growth rates tended to be higher than those of hatchery chinook. Growth rates of wild (0.49 and 0.55 mm day^{-1} for field-ID and total data sets, respectively) fish in 1994 were very similar to those estimated by Levings et al. (1986) in 1982-1983 (0.46 to 0.55 mm day^{-1}). Growth rates of hatchery fish (0.40 and 0.36 mm day^{-1} for field-ID and total data sets, respectively) in 1994 were generally lower and had a narrower range than the 1982-1983 Levings et al. (1986) estimates (0.26 to 0.70 mm day^{-1}). However, the latter values were derived from various experimental release groups, where differences in growth rates were expected.

The magnitude of growth rate differences between hatchery and wild fish seemed data set-dependent. Wild growth rates clearly appear higher than hatchery ones when all the data are used, but this difference was much less pronounced when only the field-ID data is used. There are three possible explanations for this result:

1. As the season progressed, an increasing proportion of fish may have been categorized as unmarked chinook. Excluding these larger fish from the calculation of the mean size for each sampling trip when using the field-ID data set would result in a negative bias which would be magnified as the season progressed.

2. Misclassification of fish originating from the hatchery as wild by the discriminant function would have positively biased the size-at-time means since hatchery fish were released at a larger size. As the season progressed and more fish were

designated as unmarked, the predictions of the discriminant function would have had an increasing effect on mean size, resulting in greater positive bias in the means.

3. Size dependent migration of hatchery fish out of the transition zone would have negatively biased the observed hatchery growth rates based on the remaining slower growing fish. This bias would have increased as the season progressed as a greater proportion of fish left the estuary.

If hypothesis 1) is correct, the data show that wild chinook growth rates are higher than hatchery rates so that by August, wild fish were almost the same size as hatchery fish. If hypotheses 2) or 3) are correct, differences in growth rates between wild and hatchery fish are an artifact, created by bias introduced by the discriminant function, or by size-dependent migration, respectively.

The higher growth rate of wild fish compared to hatchery fish was marginally apparent in both the field-ID- and otolith-based relationships supporting the first hypothesis that differences in growth rates between wild and hatchery fish are real. In a comparison of various experimental hatchery chinook release groups in 1982-1983 in the Campbell River estuary, Levings et al. (1986) found higher growth rates of fish with earlier release dates and smaller mean sizes. This is consistent with the findings in 1994, which showed higher growth rates for wild fish which begin the estuarine growth period at smaller sizes relative to hatchery fish.

4.4. Density Dependent Growth and Evidence for Changes in Carrying Capacity

McAllister and Brown (unpubl. manuscr.) fitted their wild chinook fry weight-total salmon biomass relationship using data from 1982-1986 only, and then compared the 1989 data point with this relationship. While acknowledging the weak data for 1989, they concluded that the unexpectedly high fry weight in this year given the very high total salmon biomass, "suggests that increases in food production accompanying the continued development of the planted marshes, and the natural colonization of new substrate by sea grasses has increased the rearing capacity of the estuary, and reduced the impacts of hatchery outputs on the growth of wild chinook fry." The conclusions could be further substantiated with more data since habitat restoration in the estuary to date has only recovered 20% (19 ha) of the entire estuarine area (Levings and Macdonald 1991). More years of high total salmon biomass could be added to confirm the relationship between chinook fry weight and total salmon biomass. The range of total salmon biomass from 1982-1986 on which the regression is based is relatively small, the largest values being less than one-half of the 1989 biomass. The regression is based on only 5 data points and is not statistically significant at $a = 0.05$ ($r^2 = 0.68$, $p = 0.09$). If habitat restoration had significantly increased the carrying capacity of the estuary, the 1994 wild fry weight value should have been well above the fry weight predicted from McAllister and Brown's relationship, especially given the additional five years for colonization of the man-made intertidal islands. However, fry

weight in 1994, while higher than that estimated by the regression, was quite similar to the 1989 value. It is possible however, that habitat restoration only mitigates density-dependent growth effects at very high total salmon biomass levels, which were not reached in 1994.

Two key issues in assessing the effects of competition between wild and hatchery chinook in the estuary are the interannual variability in the duration of the interaction time between these groups, and the biomass of hatchery chinook released prior to the index date (May 21). Assuming that wild chinook fry enter the estuary prior to hatchery releases in all years, interannual variability in the duration of competition may largely be driven by the timing of releases from the Quinsam River Hatchery. The release dates and total biomass produced from the hatchery each year has varied considerably since the hatchery has been operating and suggests that potential competitive effects of hatchery fish on wild chinook growth has also varied considerably.

A semi-log relationship predicting fry weight from total salmon biomass is theoretically more sound than a linear function, especially at high total salmon biomass values. The rationale for this model form is that size declines rapidly as biomass increases initially, but there is an asymptotic lower limit. Semi-log functions have been used in analyses of density-dependent salmonid growth in freshwater (e.g. P.G. Amiro, Fisheries and Oceans Canada, Halifax, N.S. unpublished data used in Korman et al. 1994). A semi-log fit to all the data except 1989 (Fig. 7 - dashed line) seems plausible, but was not statistically significant (r^2=0.4, p=0.18, N=6). Regardless of the model form, the 1994 wild chinook fry weight-total salmon biomass data point was in-line with McAllister and Brown's (unpubl. manuscr.) 1982-1986 relationship and supports their observation of density dependent growth of chinook fry in the Campbell River estuary.

4.5. Carrying Capacity for Juvenile Salmon and Implications for Habitat Restoration

Approximate estimates of the carrying capacity of the Campbell River estuary to support juvenile chinook, coho and all salmon species combined were made by estimating total population sizes in 1994 based on a mark-recapture methodology, and assuming that the habitat was fully saturated. Populations of wild chinook, total chinook (wild and hatchery combined), coho and all salmon species combined were 2.7, 4.8, 0.9 and 6.9 million fish, respectively. The analysis should be repeated using data collected in previous years when the system may have been closer to saturation. Estimates of total population size should be viewed as very approximate because assumptions of the adjusted Peterson mark-recapture method have not been tested in the Campbell River estuary. The assumption of a closed system, for example, was likely not applicable to the estuary. However, if marked and unmarked fish left the system at the same rate, the ratio of total catch (marked and unmarked) to recaptured marked fish should have been the same, resulting in stable estimates over time. We further assumed that marked fish became randomly mixed with unmarked ones yet data

presented in Section 3.2 (Fig. 5) clearly show differences in the distribution of wild and hatchery chinook. Finally, we assume that the distribution of fishing effort is proportional to the number of fish present in various sectors of the estuary. The 1994 data show that a substantial portion of the marked chinook were located in the transition zone, yet only 25% of the sampling sites are found in this area. Wild chinook population estimates for 1994 based on the biostandards approach were 0.5 million fry assuming a 15% egg-fry survival rate, and 1.3 million fry assuming a 40% egg-fry survival rate. These population estimates are considerably less than those based on the mark-recapture approach. The 0.5 million fry estimate is almost 6-fold less than the mark-recapture estimate for wild chinook, and it is not likely that chinook escapement to these rivers has been underestimated to this extent. This either brings into question the validity of the mark-recapture estimate for the wild fry population, or suggests that a 15% egg-fry survival rate in the Campbell and Quinsam rivers is too low.

The large biomass of chinook released from the Quinsam River Hatchery and especially the combined biomass of all species released, certainly has the potential to influence wild chinook growth. Growth may be density dependent, and reduced growth rate might provide an indicator that the carrying capacity of the estuary has been exceeded (Fig. 7). McAllister and Brown's (unpubl. manuscr.) use of a consistent index growth date against which the response of growth to salmon biomass can be measured may be useful since this procedure standardizes the length of time wild fish are exposed to density dependent processes in the estuary. The standardized index date does not control for interannual differences in the magnitude of competition exerted by hatchery chinook or other species in the estuary. Assuming that the migration patterns of all wild salmon species in the estuary are relatively consistent between years, the majority of this variability will be driven by Quinsam River Hatchery release dates. It would be possible to recompute each annual total salmon biomass value using a weighting factor which accounts for the length of time hatchery chinook and other species are released into the estuary relative to the index growth date. Such an analysis would attempt to control the interannual variability in biomass and timing of hatchery releases, and might provide a clearer indication of the effect of hatchery fish on wild chinook growth, and the effectiveness of the habitat enhancement activities in mitigating density dependent effects. It should be noted that growth rate effects can be assessed over longer time periods rather than through the use of a single index date as done in this analysis. McAllister and Brown (unpubl. manuscr.) showed that the residuals of wild chinook growth rates over the spring and summer were affected by total salmon biomass, and this may be a more powerful method for assessing competitive effects.

Historically, before the estuary was heavily industrialized, before chinook harvest levels increased and prior to opening of the Quinsam River Hatchery, the estuary must have supported the progeny from much larger broods of wild chinook. A wide margin of safety must be provided for the wild Campbell River chinook stock, given that the majority of the estuary habitats are still degraded (Bell and Thompson 1977), the fact that the hatchery fish are overwhelmingly the most significant users of the system (e.g.

present study), and that there is good evidence of density effects on wild chinook (Levings et al. 1986; McAllister and Brown (unpubl. manuscr.); present study). Before the Quinsam River Hatchery began operating, the maximum recorded escapement to the Campbell River was 8,000 chinook (Serbic 1991). These adults would have produced approximately 3.6 (15% egg-fry survival rate) to 9.6 million (40% egg dry survival rate) fry using the biostandards approach. If all the estuary habitats (est. 73 ha; Levings and Macdonald 1991) were available to this population, densities would have been approximately 4.9 fish m^{-2} (15% egg-fry survival) to 13.1 fish m^{-2} (40% egg-fry survival). For planning purposes this range could be considered the carrying capacity of the Campbell River estuary for wild fry in pristine conditions, but since the estuary was already degraded in 1965 (e.g. Figure 12.1 from 1945 given in Bell and Thompson 1977) this range must be considered a minimum value. The total present area of good quality habitat is not known but it is clearly much reduced from historical levels. In 1977, 62.6 ha of the estuary was leased for industrial purposes (Bell and Thompson 1977, Table 10.1) but that was before the major rehabilitation project in the eastern sector (Brownlee et al. 1984).

To recover the historical productive capacity, we strongly recommend that habitat restoration and/or habitat management initiatives in the estuary focus on rehabilitation of habitats in the heavily-degraded western sector. Habitats should be restored at a 2:1 ratio based on area (i.e. 2 ha restored for every one ha lost), or possibly greater using historical conditions as baseline. Density-dependent growth for wild chinook fry in the estuary as described in this analysis may still be evident because the western half of the estuary is severely degraded and cannot be considered prime rearing habitat for chinook fry relative to historical conditions. Some recent escapements to the Campbell River system (e.g. 1989, Fig. 9) have produced fry populations equal to or exceeding historical levels. If only 37.5 ha of good quality habitat are available, this implies fry densities in prime habitat may have doubled relative to historical conditions. Indeed, the middle and eastern portion of the estuary where 18.8 ha has been rehabilitated from log storage activities is currently heavily used by wild chinook fry suggesting that when good quality habitat is provided it is used immediately (Brownlee et al. 1984; Levings and Macdonald 1991; McAllister and Brown (unpubl. manuscr.).

Habitat restoration in freshwater must proceed in concert with cleanup measures in the estuary, but in order to develop an effective watershed-estuary plan and evaluate its success, coordinated monitoring is required. Compared to the other stocks on the east coast of Vancouver Island (e.g., Big Qualicum - Lister and Walker 1966, Lister and Genoe 1970; Nanaimo - Healey 1980; Cowichan - Nagtegaal et al. 1994) the freshwater life history of Campbell River chinook is poorly documented in the literature. More data are needed on freshwater survival since this determines how many fry enter the estuary and their pattern of use. Further monitoring is also needed of wild fry survival in the estuary and ocean habitats since it is only with long term data that proper evaluations will be possible.

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Table 1. Sampling schedule for the 1994 Campbell River estuary beach seine survey. Adapted from Anderson and Bravender (in prep.).

Sampling Trip	Calendar Date	Number of Stations Sampled
1	May 12-13	18
2	May 17-19	26
3	May 25-26	6
4	June 1-3	25
5	June 8	6
6	June 15-16	25
7	June 28-29	25
8	July 13-15	23
9	July 27-28	25
10	August 10-11	25

Table 2. Results of discriminant function analysis applied to 148 observations in the otolith data set. The classification coefficients and constants comprise the Fisher discriminant functions (F_{wild} or $F_{hatchery}$) for classifying the raw data. If $F_{wild} > F_{hatchery}$ a fish is designated as wild in origin, if $F_{hatchery} > F_{wild}$ a fish is designated as hatchery in origin.

Descriptor	Mean	Canonical Loadings	Classification Function Coefficients	
			Wild	Hatchery
Sampling Date	53.7	0.024	0.314	0.246
Fork Length (mm)	92.6	-0.823	2.583	2.853
Weight (g)	10.8	-0.628	-6.011	-6.365
Constant			-91.299	-106.929

Table 3. Cross-tabulation showing the number of observations categorized into hatchery- or wild-origin groups based on the otolith (a) or field-ID method (b), and predicted using the discriminant function based on sampling date, fish fork length and weight.

a)

Otolith	Discriminant Function		
	Wild	Hatchery	Total
Wild	22	16	38
Hatchery	6	104	110
Total	28	120	148

b)

Field-ID	Discriminant Function		
	Wild	Hatchery	Total
Wild	638	9	647
Hatchery	12	482	494
Total	650	491	1141
Unmarked	98	654	752

Table 4. ANOVA results testing for the effects of sampling date and sampling site location (estuary zone vs. transition zone) on the abundance of hatchery and wild chinook. Means are in densities (number of fish 100 m^{-2} set^{-1}), but the ANOVA was performed using $\log_e(x + 1)$ transformed data. The means shown below are the least-square means predicted by the ANOVA model and are not equal to the mean values by date or zone due to the unbalanced design of the ANOVA. The least-squares means are the values used in the statistical comparison.

		Wild			Hatchery		
Strata	**N**	**Mean**	**SE**	**P**	**Mean**	**SE**	**P**
all zones-dates	154	21.8	9.1		23.9	4.3	
Zone				<0.001			<0.001
estuary	113	55.2	12.5		11.0	5.5	
transition	41	5.1	16.5		55.7	7.3	
Date				0.001			<0.001
May 12-13	15	13.5	33.4		32.5	14.7	
May 17-19	21	32.9	26.5		62.3	11.7	
May 25-26	6	218.6	42.2		75.3	18.7	
June 1-3	22	9.7	26.3		67.8	11.6	
June 8	6	5.2	44.8		16.1	19.8	
June 15-16	22	7.1	26.3		44.6	11.6	
June 28-29	18	3.6	27.2		27.9	12.0	
July 13-15	14	7.9	30.6		4.0	13.5	
July 27-28	15	1.1	33.4		1.6	14.7	
Aug. 10-11	15	1.9	30.2		1.5	13.3	
Zone * Date				0.024			0.095

Table 5. ANOVA results testing the effect of habitat type on the a) density and b) biomass of hatchery and wild chinook across all sampling trips for estuary zone sites only. Means and standard errors of density (number of fish 100 m^{-2} set^{-1}) and biomass (g wet wt 100 m^{-2}) are in untransformed units, but the ANOVAs were performed using $\log_e(x + 1)$ transformed data. The means shown below are the least-square means predicted by the ANOVA model and are not equal to the mean values by date or zone due to the unbalanced design of the ANOVA. The least-squares means are the values used in the statistical comparison.

a) Density

	Wild				Hatchery		
	N	**Mean**	**SE**	**P**	**Mean**	**SE**	**P**
Habitat Type				0.004			0.049
Eelgrass	10	5.5	39.7		14.1	10.6	
Gravel	21	9.6	27.4		10.4	7.3	
Isl. Gravel	11	22.3	37.8		14.3	10.1	
Isl. Marsh	20	19.5	28.1		16.7	7.5	
Marsh	34	5.0	21.5		13.1	5.8	
Riparian	13	161.2	34.8		0.0	9.3	
Riprap	4	1.2	62.8		0.4	16.8	

b) Biomass

	Wild				Hatchery		
	N	**Mean**	**SE**	**P**	**N**	**Mean**	**SE**	**P**
Habitat Type				0.30				0.003
Eelgrass	7	26.8	68.6		10	134.0	75.5	
Gravel	19	28.8	41.7		21	86.0	52.1	
Isl. Gravel	11	52.6	54.8		11	81.3	72.0	
Isl. Marsh	20	34.4	40.6		20	113.9	53.4	
Marsh	29	19.2	33.7		34	104.7	41.0	
Riparian	12	185.8	52.4		13	0.4	66.2	
Riprap	4	10.6	90.8					

Table 6. Cross-tabulation of the numbers (a) and average weight (b) of hatchery fish with coded wire tags originating from the Quinsam River Hatchery or Discovery Passage seapens, found in either transition or estuarine zone stations.

a) Numbers

Origin	Zone CWT Collected		Total
	Transition	Estuary	
Quinsam River	287	181	468
Seapens	24	2	26
Total	311	183	494

b) Weight (g)

Origin	Zone CWT Collected		Average
	Transition	Estuary	
Quinsam River	9.1	7.2	8.3
Seapens	18.9	7.2	18.0
Average	9.8	7.2	8.8

Table 7. ANOVA results testing for the effects of sampling site location (estuary zone vs. transition zone) on the average fork length of hatchery and wild chinook across all sampling trips.

	Wild				Hatchery			
Strata	**N**	**Mean**	**SE**	**P**	**N**	**Mean**	**SE**	**P**
Zone				0.034				0.023
estuary	96	64.3	1.6		72	90.4	1.5	
transition	26	71.8	3.0		40	96.3	2.0	

Table 8. Comparison of wild and hatchery growth rates between May and August. Regressions of size vs. time were calculated using all length-weight data (including unmarked fish), and only data where the field ID method was used to distinguish between wild and hatchery fish. Weight vs. time regressions were computed using $\log_e$ transformed weight data, but growth rates presented here have been transformed back to the original units. Probability of type I error for slopes and intercepts test the significance for differences between hatchery and wild growth rates (slopes) and size-at-time (intercepts). Statistical differences in size-at-time could not be tested (denoted by '-') if slopes were not homogenous ($p<0.05$).

Dependent Variable	Data Used	Origin	r^2	Slope (growth rate)	Probability of Type I Error	
					Slopes	Intercepts
				mm day^{-1}		
Fork Length	All	Wild	0.98	0.55	0.002	-
		Hatchery	0.91	0.36		
	Field-ID	Wild	0.99	0.49	0.19	<0.001
		Hatchery	0.87	0.40		
				(g day^{-1})		
Weight	All	Wild	0.98	0.115	<0.001	-
		Hatchery	0.90	0.036		
	Field-ID	Wild	0.99	0.088	0.015	-
		Hatchery	0.92	0.051		

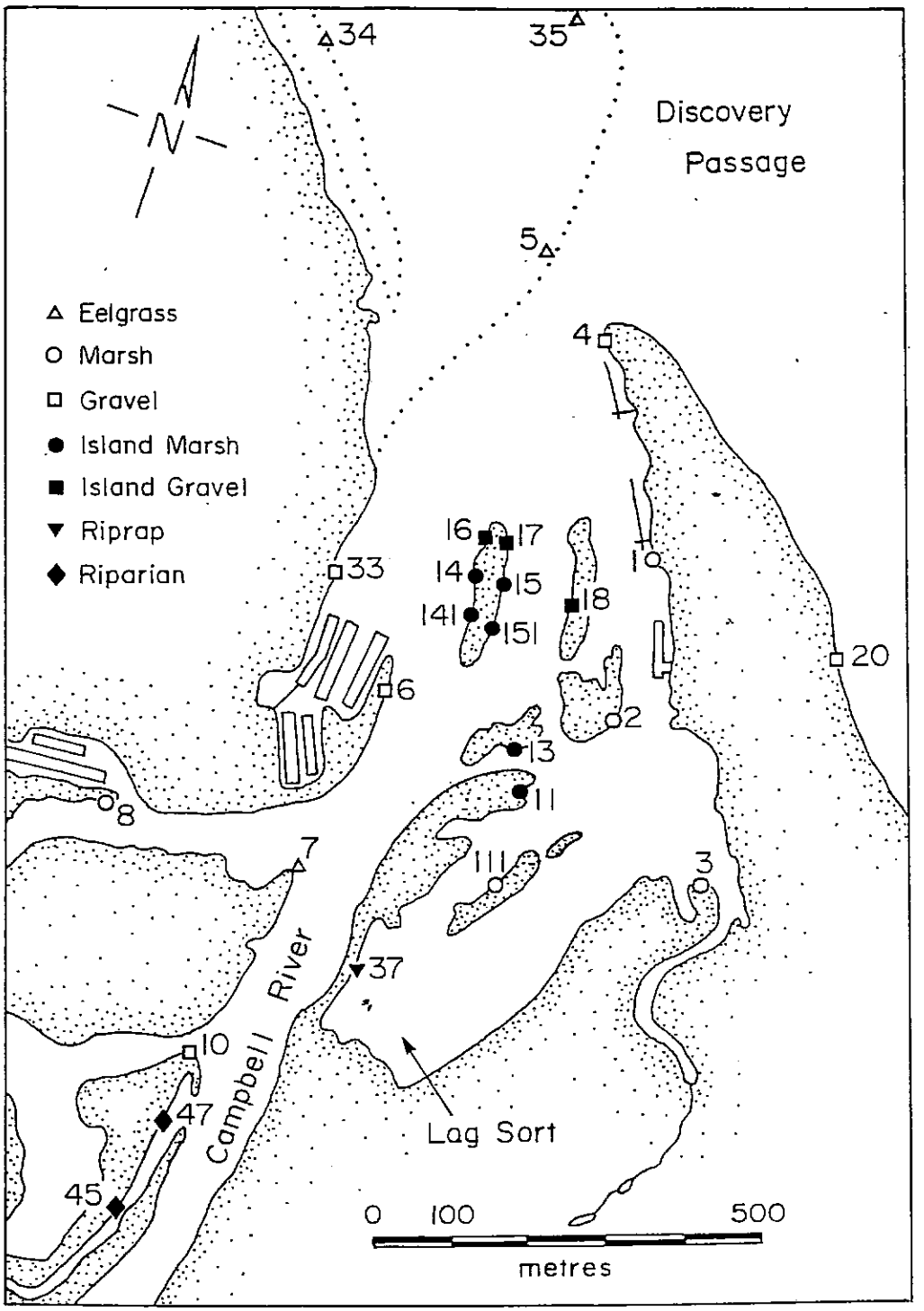

Figure 1. Campbell River study area showing sampling locations, the four intertidal islands (stippled), and the 1 m tide level (broken line). Habitat types at each station are also shown. Island 3 is the northern-most island (stippled) containing 6 sampling stations. All sites shown are in the estuarine zone with the exceptions of the most seaward sites (34, 35, 5, 4, and 20) which are in the transition zone. Adapted from Levings et al. (1986).

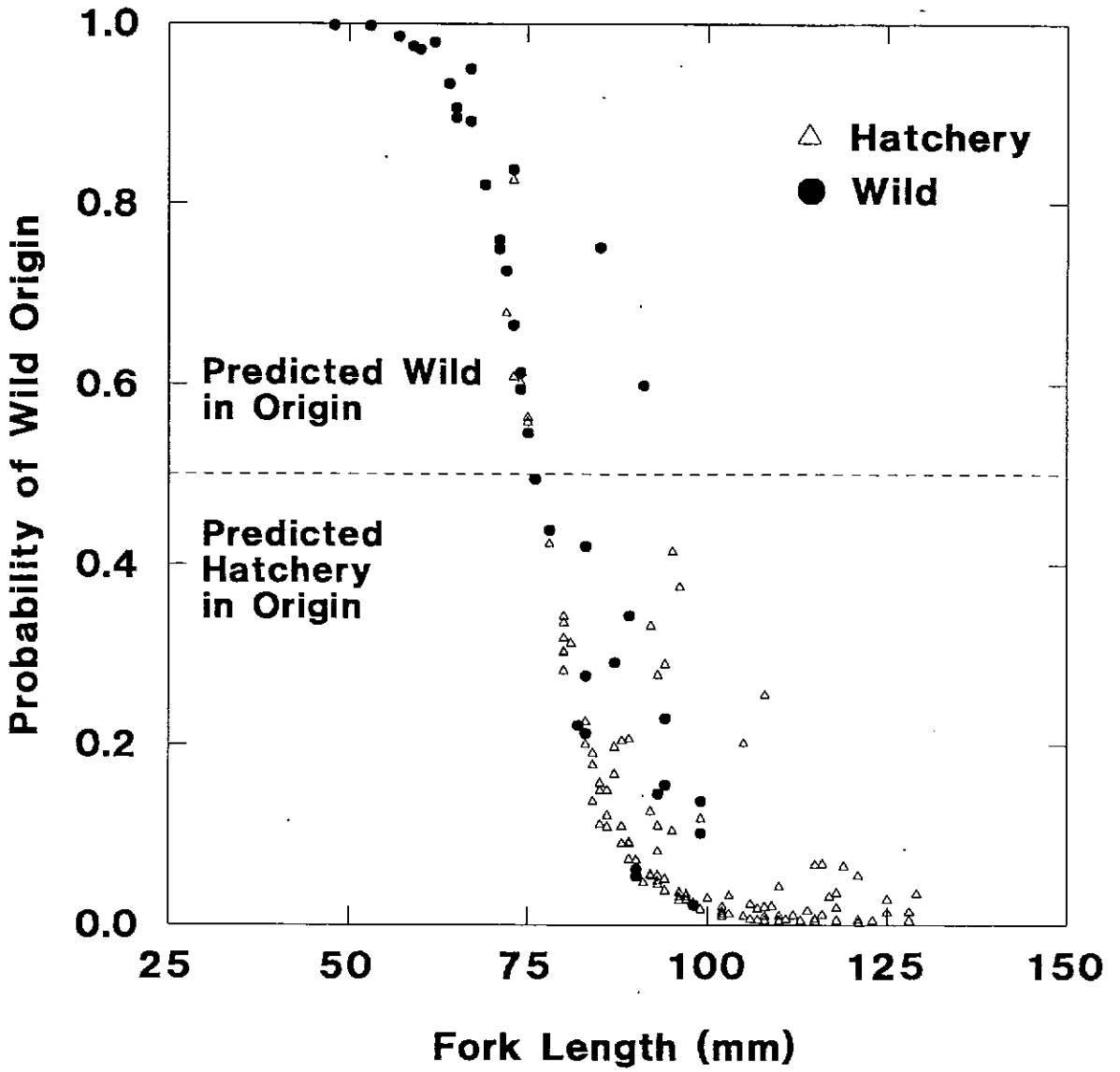

Figure 2. Probability of each fish in the otolith data set being designated as wild in origin using the discriminant function as a function of fork length. The symbols denote the 'true' origin of the fish based on otolith analysis. Fish with probabilities >0.5 would be of predicted wild origin, while fish with probabilities <0.5 would be designated as originating from the Quinsam River Hatchery.

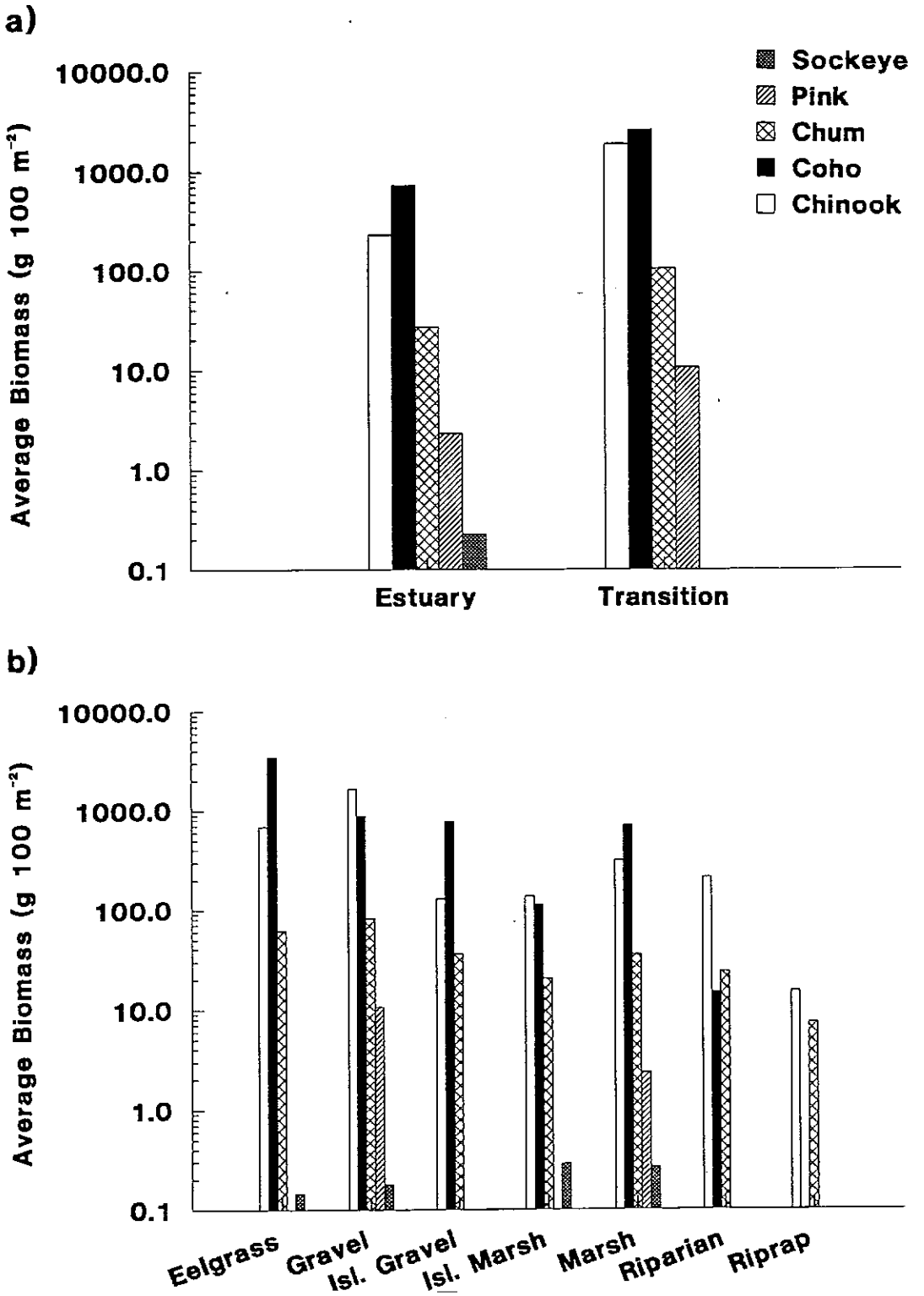

Figure 3. Biomass of juvenile salmon species per seine set (g wet wt 100 m^{-2}) averaged across all sampling trips and stations in estuarine and transition zone sites (a), within different habitat types combining estuarine and transition zone sites (b), and within different habitat types in estuarine (c) and transition (d) zone sites only. Note $\log_{10}$ scale of y-axis.

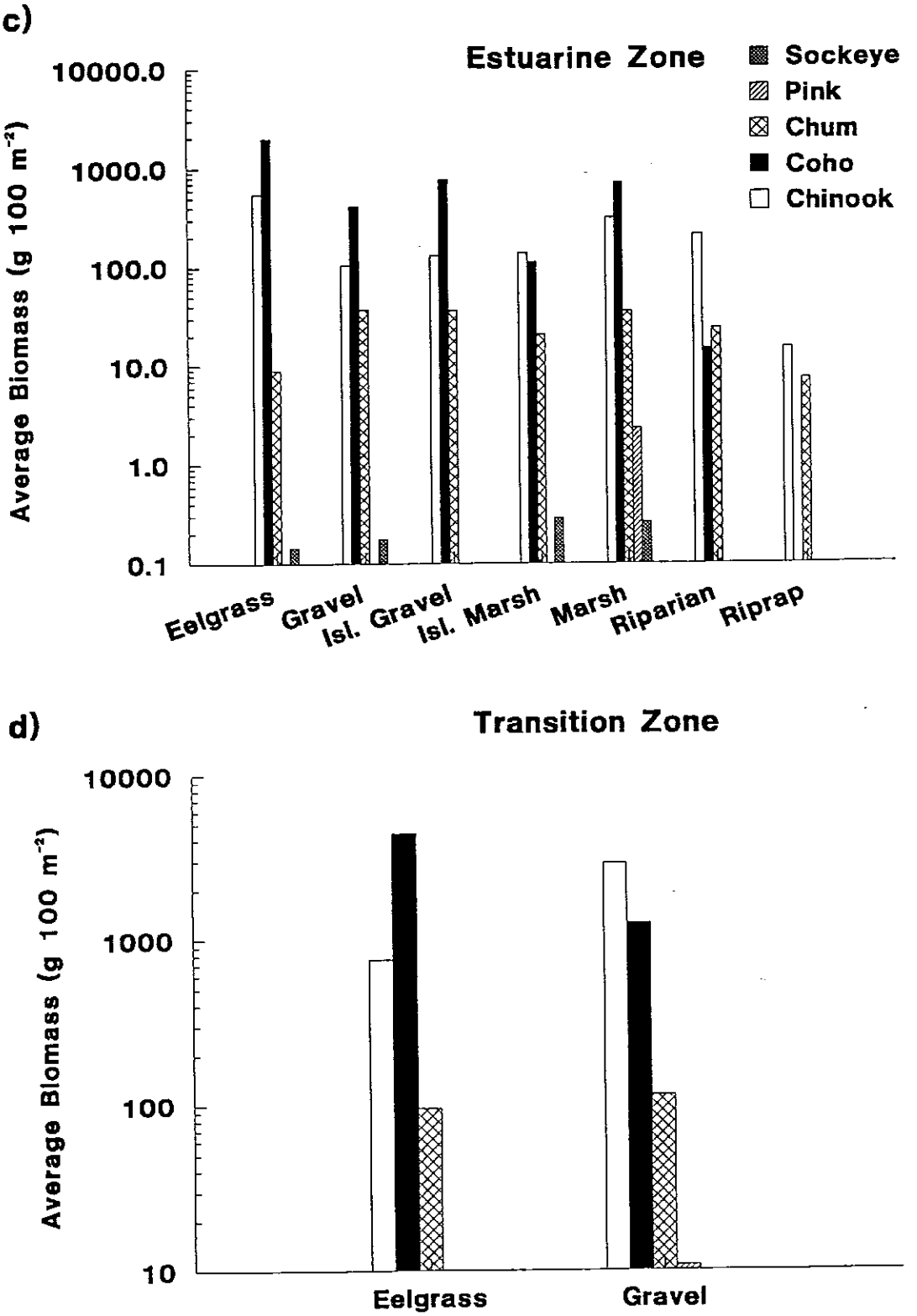

Figure 3. (cont'd)

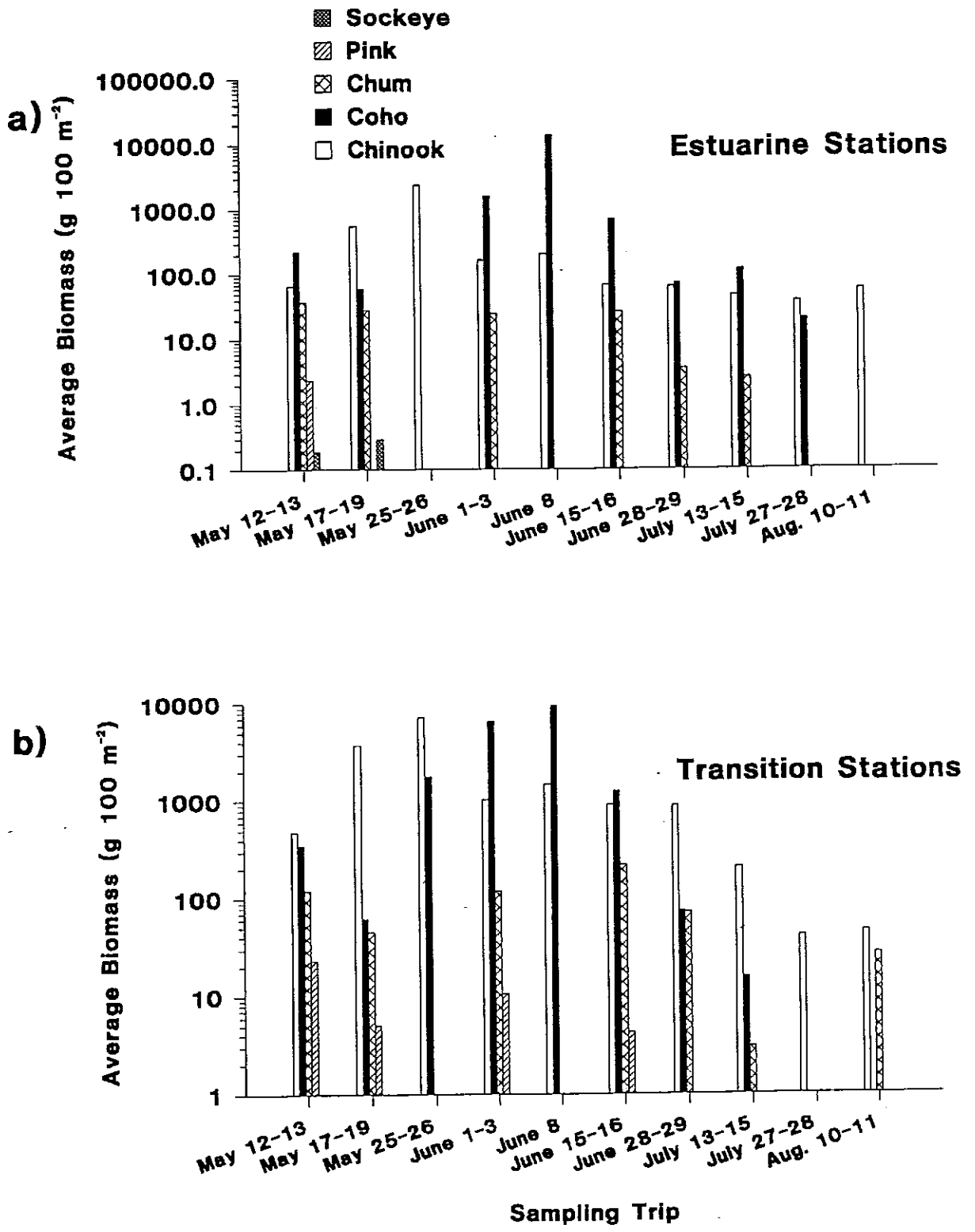

Figure 4. Biomass of juvenile salmon species per seine set (g 100 m^{-2}) in estuarine (a), and transition (b) zone sites by sampling date. Note log_{10} scale of y-axis.

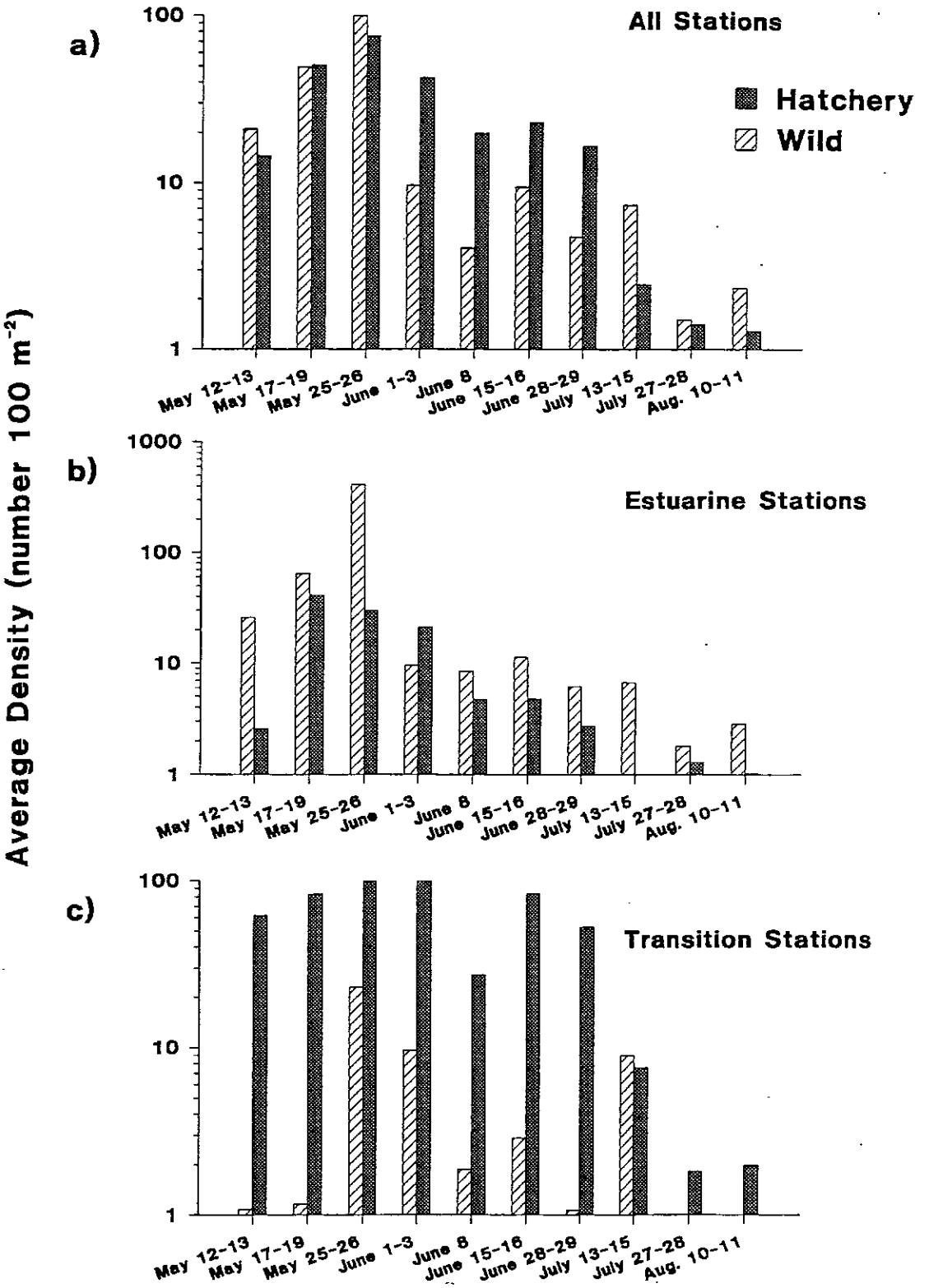

Figure 5. Density (catch per seine set in numbers 100 m^{-2}) of hatchery and wild chinook juveniles for each sampling trip averaged across all sampling stations (a), and in estuarine (b), and transition (c) zones only. Note $\log_{10}$ scale of y-axes.

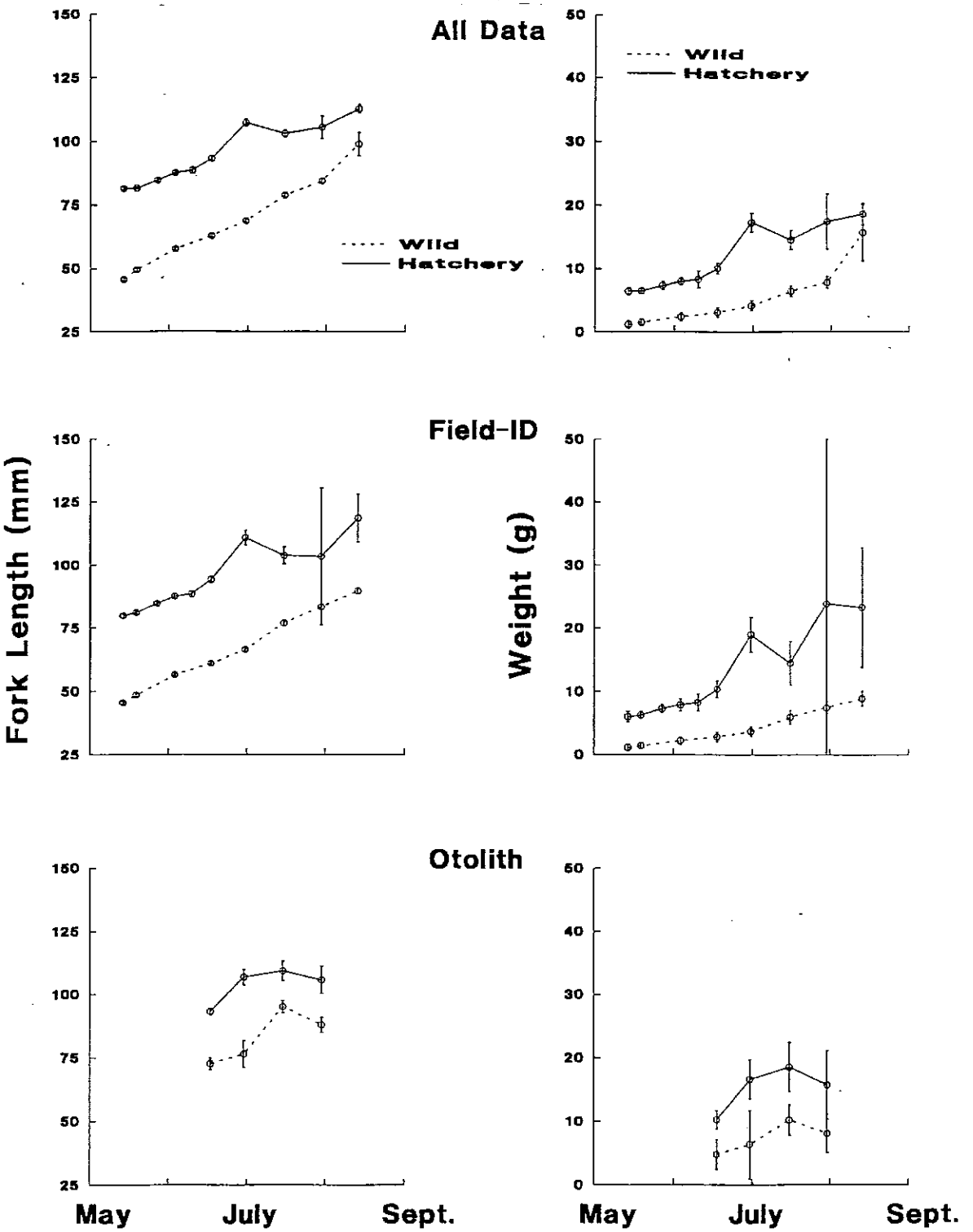

Figure 6. Change in fork length and weight of wild and hatchery juvenile chinook salmon over 10 sampling trips in 1994. Each size estimate for a trip is the mean across all stations and error bars denote one standard error. Graphs depict size-at-time using: 1) all length-weight data (discriminant function method), 2) data where the origin of fish as identified based on size in the field only, and 3) data where origin was determined using the otolith technique.

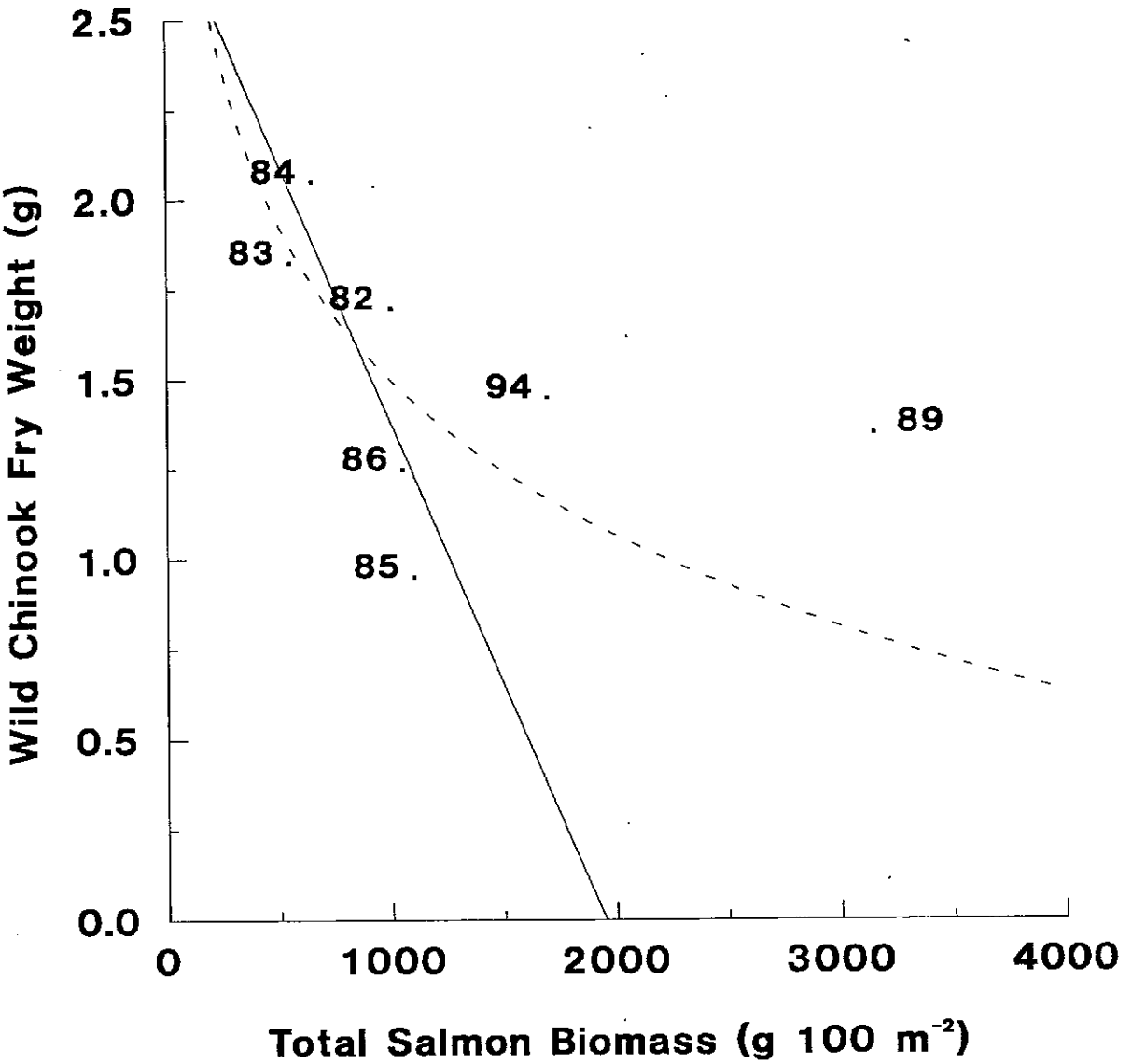

Figure 7. Wild chinook fry weight in late May predicted as a function of total salmon biomass at the same date. The solid line is the regression fitted by McAllister and Brown (unpubl. manuscr.) using 1982-1986 data only (r^2 = 0.68, p = 0.09) and the dashed line is a semi-log fit to 1982-1986 data and the 1994 point (r^2 = 0.4 , p = 0.18). Figure partially adapted from McAllister and Brown (unpubl. manuscr.).

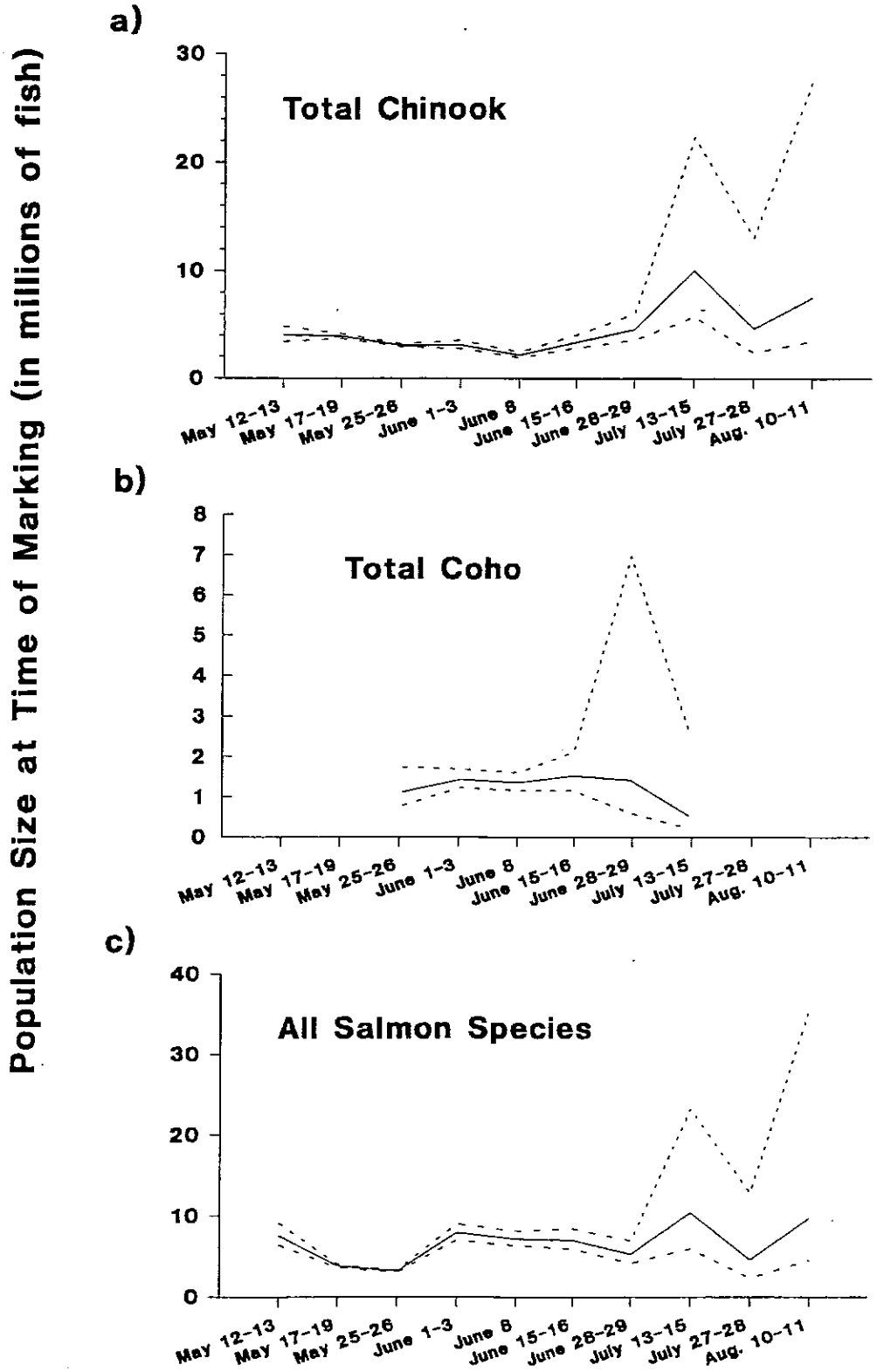

Figure 8. Adjusted Peterson estimates of total juvenile population size of wild and hatchery chinook (a), wild and hatchery coho (b) , and all salmon species combined (c) in the Campbell River estuary based on marked:unmarked ratios at the time of hatchery releases. Dashed lines in all graphs represent upper and lower 95% confidence limits.

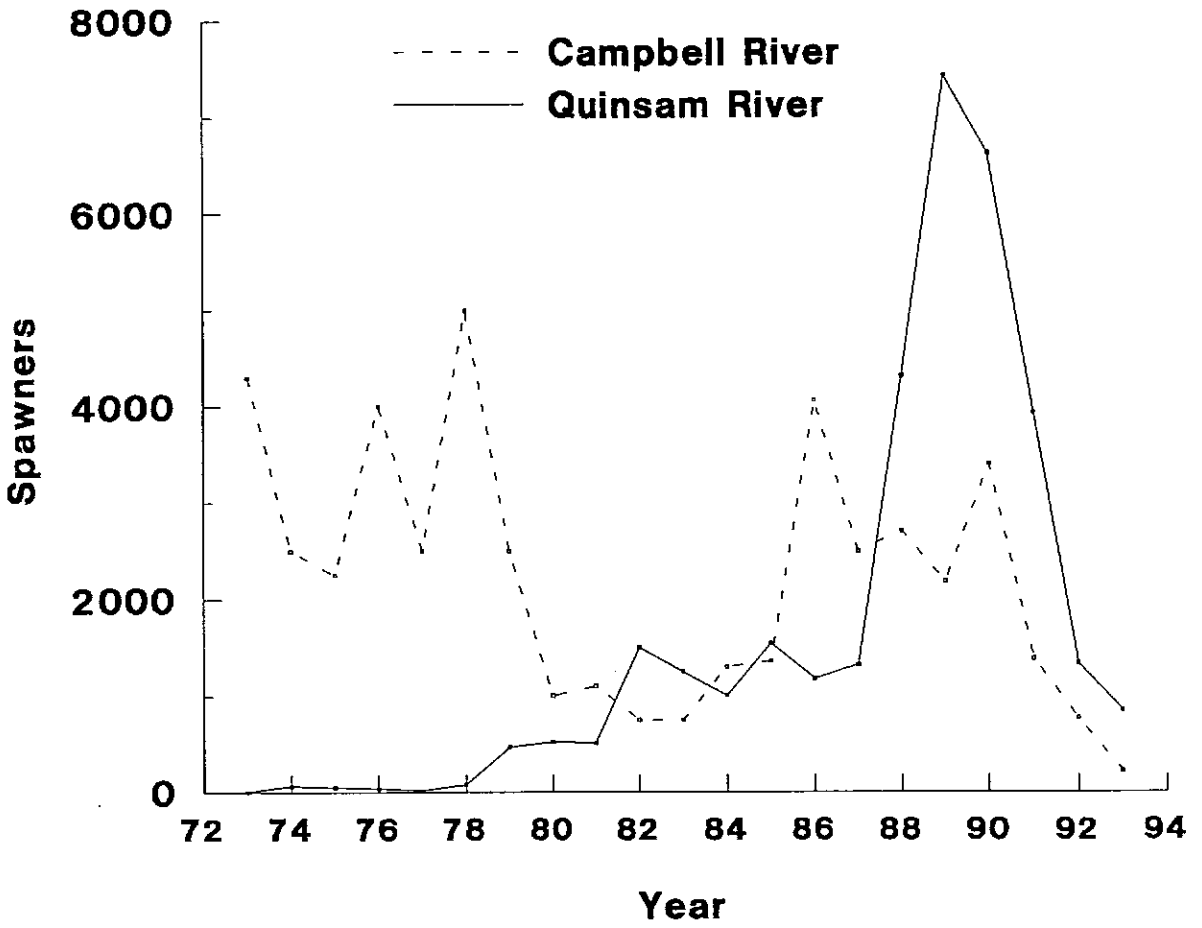

Figure 9. Chinook escapement to the Quinsam and Campbell rivers estimated by visual surveys between 1973 and 1993.