Estuarine Habitat and Juvenile Salmon: Current and Historical Linkages in the Lower Columbia River and Estuary

Final Report 2002-2008

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EXECUTIVE SUMMARY

Our studies in the lower 100-km of the Columbia River estuary quantified historical habitat changes and provided new information about contemporary abundance patterns, life histories, and habitat associations of Chinook salmon. The conceptual framework for this research defined salmon performance in the estuary as the product of three factors: habitat opportunity, habitat capacity, and the structure/life histories of source populations. Our 2002-2008 survey results provided empirical data to support this framework by quantifying Chinook salmon performance in terms of temporal abundance, life history and stock-group diversity, foraging success, and growth, as well as by quantifying the relationships between stock groups and one or more of these factors.

In Part I, we detail our reconstruction of historical habitat opportunities and changes in the estuary as influenced by the tide, river flows, and temperature. In Part II, we depict contemporary habitat opportunities based on present-day patterns of salmon distribution and abundance and upon various physical factors that influence fish access to shallow-water rearing areas. In Part III, we compare the capacity of different wetland and nearshore habitats in supporting juvenile Chinook salmon as indicated by variations in prey availability, salmon diet, and rates of consumption. Finally, in Part IV, we examine the effects of upriver population structure and life histories on estuary rearing behavior and performance, including the genetic sources of individual Chinook salmon found within particular habitats and stock-specific patterns of residency and growth.

These surveys provided new information about the present estuarine habitat associations of juvenile salmon. They also provided data for estimates of historical change in habitat conditions, estimates of historical change in salmon life histories, and analyses of food webs. Below we summarize major conclusions drawn from these evaluations.

Historical Change in Estuarine Habitat Opportunity

1. *Extensive wetland loss has substantially decreased the quantity and quality of wetland habitats that support salmonid food webs and provide off-channel rearing areas for subyearling migrants with estuary-resident life histories*

We estimated that diking, filling, and other development activities have decreased the total area of tidal wetlands by more than 50% in the lower Columbia River estuary, a result that was consistent with previous estimates. The effects of these losses are compounded by flow regulation, which influences river stage and the extent of habitat

inundation. Diking has increased the river elevations required to flood many shallow-water habitats, while the effects of flow-management on river stage and habitat inundation varied among estuary reaches.

In the Eagle Cliff-to-Kalama reach, for example, diking alone accounted for the loss of about three-quarters of shallow-water habitat of the type thought to be optimal for use by subyearling Chinook (0.1-2.0 m deep). Our model results implied that habitat in this reach might be restored by removing dikes without significantly modifying the present flow-management regime. In contrast, in the Skamokawa-to-Beaver reach, the combined effects of diking and flow regulation were significantly greater than the sum of their individual effects.

Flow management may be a particularly important driver of habitat opportunity in the upper reaches of the estuary because the flood plain in these reaches is relatively narrow, tidal influence is weaker, and the hydropower system exerts significant control over daily and seasonal water levels.

2. *Together, the loss of rearing opportunities and the decline of historical populations upriver have reduced life history diversity and late-season abundance of Chinook salmon in the estuary.*

Scale patterns from juveniles collected during a survey in 1914-1916 defined at least six juvenile life history types in the Columbia River estuary, five of which were subyearling migrant types. A more recent analysis of estuary beach-seine and tagging data indicated that just three basic migrant types are now represented: emergent fry, yearling, and a single subyearling type composed of spring and early-summer fluvial migrants. Comparisons of past and recent surveys have led to the hypothesis that life history diversity in Columbia River Chinook salmon populations has declined.

Results from our field survey during 2002-2008 revealed a somewhat greater variety of subyearling behaviors than suggested by this hypothesis, including significant numbers of fry and fingerling migrants that rear in the estuary for extended periods. Nonetheless, if results from the survey of 1914-1916 are representative of estuary rearing and migration behaviors a century ago, then our findings did support the hypothesis that Chinook salmon life histories have been simplified.

During 2002-2008, abundance of juvenile Chinook salmon in the Columbia River estuary peaked sharply in spring and declined rapidly thereafter. This differed from the broad temporal distribution observed early in the twentieth century, when large pulses of new recruits appeared in the estuary in July, September, and October. Peak estuary abundances now occur in May, and roughly 90% of the juvenile salmon migration is

complete by the end of August. While surveys during 1914-1916 had indicated that Chinook salmon used the Columbia River estuary as a summer/fall rearing area, our results suggested that this function has diminished. Compared with the protracted period of estuary use described for these early surveys, far fewer juvenile migrants now enter or remain in the estuary from summer to fall.

Multiple factors likely account for this relatively low abundance of subyearling Chinook salmon after mid-summer. Extensive wetland diking and filling in the lower estuary has eliminated habitat for fry and fingerling migrants and has likely reduced the expression of some estuary-resident life histories. Substantial losses of historical wetlands in the lower estuary have reduced rearing opportunities for fry and fingerling migrants that tend to remain in the estuary for the longest periods.

Seasonal patterns of stock composition in the lower Columbia River estuary generally reflected the broad spatial structure of some major population groups. For example, lower Columbia River stock groups dominated stock composition in spring, while significant numbers of upper Columbia River summer/fall stocks appeared later, in summer and fall. The large numbers of juveniles observed entering the estuary late in the rearing season in 1915 and 1916 likely included many migrants from mid- and upper-basin populations that are now depleted or extinct.

Further research is needed to characterize contemporary life history variations among mid- and upper-basin stocks, particularly their patterns of habitat use and residency in the upper estuary (i.e. rkm 100 to Bonneville Dam).

River temperatures that have risen above historical levels during summer and fall are also likely to be limiting estuary rearing opportunities for salmon. We frequently measured water temperatures above 19°C in the main-stem estuary and in shallow wetland channels during late summer and fall. These temperatures appeared to exacerbate the effects of wetland habitat loss and limit salmon rearing in the tidal fluvial zone. Juvenile salmon vacated shallow wetland habitats, and abundance throughout the lower estuary declined by mid-summer, when surface-water temperatures reached and exceeded 19°C. Chinook salmon abundance consistently declined during this period. Chinook salmon were present at some near-shore beach-seine sites in temperatures as high as 24°C, but abundance nonetheless declined rapidly after July. Temperature conditions therefore likely limited the expression of estuary-resident life histories late in the rearing season, reducing the overall capacity of the estuary to support juvenile salmon.

Our analyses of historical temperature using a regression model indicated that under a virgin-flow, cool-temperature scenario (1890-1926), temperatures remained below 19°C in all months of the year; however, under a modern temperature scenario (1976-2002), average temperature exceeded >19°C from July through September. Because cool ocean water does not intrude far upstream in the Columbia River, river temperature is a major driver of habitat conditions across much of the estuary. We attributed more than half of the estimated 2-3°C increase above historical May-December water temperatures to the effects of reservoir storage behind Columbia River dams.

At Russian Island and Lord Island marshes in 2007 and 2008, much if not most shallow wetland habitat that was otherwise accessible to salmon (depth of at least 0.5 m) became marginal for rearing after June because of high water temperatures. Within the Columbia River basin, alluvial habitats affected by groundwater upwelling (i.e., deep pools, low-velocity backwaters, and spring brooks isolated from main channel flows) afford cool-water refugia in many lower elevation reaches and desert streams, where salmonids otherwise might not persist. Although similar thermal refugia may exist within the estuary, their locations and attributes have not been defined. As tidal-freshwater reaches approach stressful temperatures, some juveniles may move downstream into cooler brackish areas or exit the estuary altogether. Additional warming of the Columbia River Basin through climate change could further restrict estuary rearing opportunities and life history expression by juvenile salmon.

To realize the full benefits of estuary restoration, additional action is needed to ameliorate stressful water temperatures in the main-stem river and to identify and protect existing cool-water refugia (cold-water seeps, groundwater upwelling sites, etc.) within the estuary.

Habitat Opportunity: Estuarine Fish Species and Salmon Habitat Use

3. *Salmon habitat use and residence times vary with fish size, but all wetland habitat types in the lower estuary are utilized by the smallest subyearling size classes, which tend to remain in the estuary for the longest periods.*

Our beach-seine and wetland-survey results supported the hypothesis that estuary habitat use by juvenile salmon is size-related, with many small subyearling migrants rearing in shallow wetland channels. Although a wide range of size classes was found at near-shore beach seining sites, including some fish larger than 120 mm, the shallow and protected wetland sites were dominated by smaller fry and fingerlings. Individuals sampled from sites in the secondary interior channels of emergent marsh islands (rkm 35)

rarely exceeded 90 mm. Maximum fish sizes were even smaller in the forested and scrub-shrub wetlands located further upriver (above rkm 75), where individuals rarely exceeded 70 mm.

Mean sizes of juvenile Chinook salmon generally increased toward the estuary mouth and laterally from shallow nearshore to deeper offshore areas. High proportions of recently emerged fry were found throughout summer at sites in and above Cathlamet Bay. This suggested that many of the smallest individuals delayed movement to tidal freshwater areas, while larger fish migrated further downstream. Overall, these results suggested size-dependent patterns, both in migration and habitat use among many juveniles: the smallest individuals reared in shallow peripheral channels of all wetland habitat types throughout the lower estuary—emergent, scrub-shrub, forested, and mixed—and gradually moved offshore and toward the estuary mouth as they fed and grew.

Our back-calculations of residence time based on otolith chemistry indicated that during 2003-2005, estuary residence averaged 2-3 months for the smallest fry migrants and 4-6 weeks for large subyearlings (>90 mm). Approximately three-fourths of the subyearlings we sampled at Pt. Adams Beach had a discernible Sr spike on their otolith, indicating a measurable period of saltwater rearing. We estimated that 30-50% of these residents had stayed in the lower estuary more than 30 d prior to capture. These back-calculations represented minimum estimates of residence time because the Sr technique only measures contact with saltwater and does not account for time spent in the extensive tidal freshwater zone of the estuary.

Somewhat longer mean residence times were estimated from stable-isotope methods, which were used to identify food-web sources of salmon in the lower 65 km of the estuary. On average, juvenile Chinook salmon interacted with emergent-marsh food webs for ~65 d, while individual interaction with marsh-derived prey ranged 0-260 d.

Fry dispersed into wetland channels and other shallow estuarine habitats soon after emergence in early spring, and most subyearlings had vacated wetland habitats by August. Few subyearlings entered or remained in wetland channels in the uppermost forested and scrub-shrub wetlands (i.e., near rkm 100) at sizes greater than 70 mm FL or in the secondary channels of Cathlamet Bay emergent wetlands at sizes above 90 mm FL.

However, the back-calculated size at estuary entry for nearly half of the salmon we analyzed from the Point Adams Beach site was less than 60 mm FL. Mean residency estimates for individuals in this size class were 54 d during January-April and 59 d during May-August (2003-2005). Therefore, whereas tagging studies often target individuals with riverine smolt life histories (i.e., extended periods of freshwater rearing, large size at

estuary entry, short estuary residency), the lower-estuary beach-seine collections included a much greater proportion of riverine-estuarine and estuarine smolts (i.e., brief or moderate periods of freshwater rearing, small size at estuary entry, and extended estuary residency).

Because most sampling methods target particular size classes of fish or types of habitat, no single sampling technique is adequate to characterize the full diversity of Chinook life histories in the estuary.

4. *Naturally produced subyearling salmon dominate in shallow wetland channels and may benefit most directly from restoration of wetland habitats.*

Disproportionately high numbers of naturally produced salmon utilized the interior tidal channels of wetland habitats, particularly in the mixed forested and scrub/shrub wetlands above Cathlamet Bay. Of the subyearling Chinook salmon collected at Lord Island for example, less than 2% in 2007 and only about 7% in 2008 were adipose-clipped hatchery fish (hatchery marking rates were approximately 65% in 2007 and 81% in 2008). These results were consistent with the size classes known to frequent lower-estuary wetlands, such as fry and small fingerlings that are smaller than most juveniles released from hatcheries. ***Dike removal or other actions to restore fish access to lower-estuary wetlands will thus tend to target naturally produced juveniles with subyearling-migrant life histories.***

5. *Large releases from hatcheries have replaced diverse, naturally spawning populations with fewer hatchery stocks; these stocks are reared primarily as freshwater phenotypes with short estuary residence times. In the Columbia River estuary, contemporary patterns of abundance, stock composition, habitat use, and residency are largely driven by artificial propagation programs.*

Artificial propagation has largely replaced rather than supplemented production from historical spawning populations. Losses of habitat due to dam construction, irrigation withdrawals, and other development have concentrated salmon habitat use and constrained life history variation. To the extent that this has occurred, the salmon ecosystem may be more vulnerable to environmental fluctuations. ***The resilience of salmon populations to future environmental change requires that opportunities for diverse life history expression are restored.***

Hatcheries have redistributed Columbia River salmon production in time and space by replacing the dispersed distributions and protracted emergence times of historical populations with a few selected phenotypes—primarily subyearling and yearling riverine smolts—that are released in concentrated pulses. Approximately

one-half of the total subyearling hatchery production is released into habitats utilized by the Lower Columbia River Chinook ESU (evolutionarily significant unit). During our 2002-2008 field surveys, annual production of hatchery subyearling Chinook salmon ranged from approximately 56 to 71 million fish.

Because patterns of estuarine habitat use and residency vary with fish size and time of entry, some hatchery practices tend to limit opportunities for life history expression in the estuary. Hatcheries rear fish to relatively large sizes and concentrate the timing of releases, which now occurs primarily from April through July. In contrast, historical populations contributed larger proportions of small fish, which migrated to the estuary over a broader time period. These changes have likely contributed to the reduced representation of late-season migrants in the estuary. Temporal and spatial concentration of juvenile abundance also could limit salmon performance and prevent the productive capacity of the estuary from being fully utilized.

Unmarked, naturally produced fry (i.e., smaller than the sizes generally released from hatcheries) accounted for most juvenile salmon in the estuary from January through March. By late March or April, we observed a sharp increase in mean size and the appearance of a bimodal size distribution in samples from beach and emergent marsh sites; this increase corresponded with the first hatchery arrivals.

Two ESUs represent nearly two-thirds of the total Chinook hatchery production (subyearling and yearling): Lower Columbia River Fall Chinook, which makes up about 44%, and Upper Columbia River Summer/Fall Chinook, which comprises nearly 20%. Fish from these two ESUs also dominated the genetic stock composition of subyearling Chinook salmon from our beach-seine catches (~92%). Hatchery releases similarly accounted for a substantial proportion of yearlings entering the estuary, but most yearlings migrated through deeper channels further from shore and were poorly represented in our beach-seine and trap-net catches from shallow-water habitats.

Most hatcheries favor production of large fingerling and yearling riverine smolts, a preference that was represented in the size distributions of juvenile salmon sampled in the estuary. In 2007 and 2008, when the marking rate for hatchery fish increased, we were able to compare the numbers of marked and unmarked individuals to provide a useful indicator of the relative sizes of hatchery vs. naturally produced salmon: mean fork lengths of hatchery salmon at beach-seine sites in 2007 and wetland sites in 2007-2008 were considerably larger, and their size distributions narrower, than those of naturally produced juveniles. In our beach-seine samples during 2007, more than 55% of the unmarked (naturally produced) salmon were smaller than 60 mm FL, while less than 3% of the marked (hatchery) salmon were below this size.

Size differences between hatchery and naturally produced juveniles directly influenced their respective habitat distributions and mean residence times in the estuary. Shallow wetland channels, particularly the forested and mixed wetland habitats above Cathlamet Bay, contained primarily small, unmarked fry and fewer hatchery-marked subyearlings (only 2-16%). Larger hatchery fish represented a much greater proportion of the salmon sampled in deeper, near-shore beach seining sites (46-66%) in 2007 and 2008, particularly those located near the river mouth in the estuarine mixing and marine zones.

These respective distributions of hatchery and wild fish were consistent with the negative relationship between size at entry and residence time. The dominant, fluvial phenotypes produced by most hatchery programs are more likely to enter the estuary at a larger size, select deeper habitats further from shore, and migrate to the estuary mouth more quickly than many of their smaller, naturally produced cohorts. Hatchery rearing programs thus strongly influence phenotypic expression by salmon in the estuary.

Along with improvement to upriver habitats, hatchery and other management practices must be adjusted to accommodate a greater variety of salmon phenotypes. This will allow salmon stocks to fully benefit from the estuary's diverse habitat opportunities, including habitats that are reestablished through wetland restoration.

6. *The response of the estuarine ecosystem to large subsidies of hatchery fish and estuary interactions between hatchery and naturally produced salmon remain poorly understood. Such interactions may ultimately determine whether estuary restoration is an effective tool for salmon recovery.*

Our research highlighted the effects of phenotypic hatchery selection on salmon habitat use in the estuary, independent of other ecological or genetic influences of hatchery programs. Until recently, the low marking rates of most Columbia River hatcheries have severely limited the ability of investigators to distinguish hatchery from naturally produced juveniles. For all but the final 2 years of our 2002-2008 surveys, marking rates for Columbia River hatcheries were insufficient to draw conclusions about the behaviors or performance of hatchery vs. naturally produced salmon by comparing marked and unmarked groups, respectively. At the current marking rates, it has become feasible to make these comparisons, although large proportions of some hatchery stocks are still not marked.

Because hatchery production could undermine the effectiveness of recovery measures for at-risk populations, we strongly recommend that additional estuary studies be designed and conducted to examine more explicitly the ecological interactions between hatchery and naturally produced Chinook salmon.

Habitat Capacity: Prey Availability, Diet, and Rates of Consumption

7. *Wetland-derived food webs support juvenile salmon throughout the estuary, including larger individuals that do not typically occupy wetland channels.*

Our survey results provided evidence that the food webs of subyearling Chinook salmon remain closely coupled to wetland and other shallow-water habitats, reinforcing concerns that the estuary's historical capacity to support juvenile salmon may have declined. The diet composition of Chinook salmon from lower-estuary beach-seine surveys during 2002-2007 was similar to that reported from beach-seine surveys 30 years ago. In both cases, juvenile salmon throughout the estuary fed directly on insect and amphipod taxa, which are typically produced in wetlands and other shallow-water habitats. Although previous studies have often stressed the importance of amphipods, adult insects generally ranked highest among prey items in the diets of juvenile salmon collected monthly at Pt. Adams Beach, 2002-2007. *Americorophium salmonis* (formerly classified as *Corophium salmonis*) was often a dominant prey item during the 1980 survey, particularly in the upper (tidal freshwater) reaches of the lower estuary.

Small subyearling Chinook salmon that entered emergent, scrub-shrub, and forested wetland channels also consumed invertebrate taxa, which were produced primarily within these habitats; these taxa included large proportions of emergent chironomids, larval chironomids, and other diptera. Although chironomids dominated salmon diets, the largest size classes of subyearlings also frequently consumed *Americorophium* spp. and other epibenthic invertebrates.

Our emergent chironomid surveys revealed a transitional emergent insect community during the period of juvenile Chinook migration and rearing in freshwater tidal channels. Dipteran taxa, particularly Chironomidae, dominated insect emergence and assemblage composition over time and space. For a given date, insect composition and abundance was consistent within each microhabitat. ***Future studies should examine whether Chinook salmon target specific taxa within the chironomid family as evidence for microhabitat selection within tidal channels.***

Wetlands produce and export insect and other prey taxa to other areas of the estuary. All size classes of juvenile salmon were linked to wetland-derived food webs, the products of which were selected in greater proportions than those of other food-web sources that were more readily available. Extensive wetland losses thus undermine a preferred trophic pathway that could limit the estuary's capacity to support juvenile salmon. These results reinforce the need for wetland restoration, which will benefit all ESUs and size classes of salmon that reside, feed, and grow in the estuary before migrating to the ocean.

Performance Metrics: Population Structure, Life History Diversity, and Growth

8. *Different genetic stock groups of Chinook salmon exhibit characteristic patterns of temporal and spatial distribution in the lower estuary.*

Genetic stock groups in the lower estuary were not uniformly distributed but exhibited distinct temporal and spatial patterns. Fall-run juveniles from the West Cascade and Spring Creek Group stocks (both from the Lower Columbia River Chinook Salmon ESU) dominated shallow habitats of the lower estuary. Whereas the Spring Creek Group fall stocks were widely distributed and abundant primarily during spring, proportions of the West Cascade fall stock generally increased with distance from the estuary mouth and were well represented from spring through fall. Representatives from the Upper Columbia River summer/fall stock group were distributed throughout the estuary, but few appeared in our samples until summer and fall. No consistent differences in stock composition were apparent at finer scales (i.e., habitat scales).

To determine restoration priorities for at-risk salmon, further investigation is needed on stock-specific patterns of estuary habitat use, particularly in the poorly studied tidal-fluvial reaches between rkm 100 and Bonneville Dam.

9. *Most Chinook salmon Evolutionarily Significant Units (ESUs) are capable of expressing subyearling life histories. Both lower and upper Columbia River stock groups can produce subyearlings that reside in the estuary for several months.*

Subyearling Chinook salmon from all ESUs occupied shallow-water habitats of the lower estuary, except for spring run groups from the interior Columbia River basin. Otolith collections from Pt. Adams Beach indicated that subyearlings from a diverse subset of these ESUs expressed estuary-resident life histories. Back-calculations using otolith chemical analyses indicated that size at saltwater entry for lower Columbia River fall stocks (West Cascade and Spring Creek Group) averaged near 60 mm FL. Mean residence times in the saltwater portion of the estuary were estimated at between 1 and 2 months. Although the average back-calculated size at entry for Upper Columbia River summer/fall collections (n = 9) was much larger (88 mm FL), the estimated period of estuary residency averaged 2 months or more.

Estimates from otolith analyses indicated positive growth rates for estuary-rearing salmon during all seasons, including maximum mean values of 0.55 mm day^{-1} during late summer (August). However, by this time, water temperatures in much of the estuary had increased to high levels, and salmon abundance had declined substantially. We have no measure of the daily rations or water temperatures that supported our growth rate estimates, which were back-calculated from otolith increment widths for individuals

captured near the river mouth (Pt. Adams Beach). ***Bioenergetic modeling may offer the best tool to investigate further the interactive effects of water temperature, salmon densities, and prey availability on salmon growth potential in the estuary.***

10. The lower Columbia River estuary supports foraging and growth of juvenile migrants and contributes to the life history diversity of Chinook salmon populations.

Despite evidence that multiple factors have constrained juvenile life histories, we found estuary-resident behaviors were more prevalent than expected. Otolith chemistry results indicated that 32-45% of juvenile salmon sampled in shallow-water habitats had entered the Columbia River estuary soon after emergence, and many early entrants grew for weeks or months before being captured at larger sizes at Pt. Adams Beach. Detections from our PIT-tag monitors on Russian Island indicated that even some hatchery-reared individuals lingered in the estuary for weeks or months and occupied off-channel habitats before migrating to the ocean.

Our data from PIT-tag detections also revealed a surprising degree of habitat fidelity by some individuals. Despite having to vacate shallow wetland channels twice daily with each low tide, some juveniles returned repeatedly to the same site and continued to grow during their residency, while others visited the same channel intermittently for weeks, suggesting habitat fidelity at a coarser scale (e.g., the entire wetland-habitat complex) than that represented by the small, secondary channels we monitored with PIT antennas. These results indicated that a significant number of subyearling Chinook salmon reside in the estuary for months, and some individuals spend days or weeks in or around the same habitat or habitat complex. Individuals that remained within the same emergent wetland complex for a week or more benefitted directly from local feeding opportunities, as indicated by instantaneous growth rates that averaged from 0.65 to 0.82 mm d^{-1}.

Diversity of salmon life histories has been described as an evolutionary strategy to spread risk and avoid brood failure in uncertain environments. From the surveys of 1914-1916, the range of sizes, estuary entrance times, and estuary residence patterns depicted a continuum of juvenile phenotypes produced by a diversity of upriver populations (i.e., genotypes). Our results indicated that the lower estuary contributes to this continuum by providing alternative rearing habitats for juvenile growth and development, particularly by subyearlings, prior to ocean entry.

Additional research is needed to determine the contribution of upper-estuary habitats to the life histories and performance (i.e., foraging success, growth, and survival) of Chinook salmon stocks throughout the basin.

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INTRODUCTION

Since the first study of salmon life history in the Columbia River Basin (Rich 1920), biologists have known that many juvenile Chinook salmon *Oncorhynchus tshawytscha* rear in the estuary prior to ocean entry. Subsequent surveys in other coastal watersheds have verified that estuaries provide productive feeding areas for salmon; estuaries produce transitional environments, which allow individuals to acclimate gradually to salt water, and complex habitat features, which may provide refugia from marine predators (Simenstad et al. 1982; Healey 1991; Thorpe 1994). Yet surprisingly few studies have examined the estuarine habitat requirements of Columbia River salmon, despite documentation of severe habitat loss (Thomas 1983) and the addition of 13 Columbia River stocks to the federal list of threatened and endangered species (NOAA 2011).

Diversity and abundance of salmonid populations in the Columbia River Basin have declined rapidly in the last 150 years in response to habitat modification, particularly degradation and loss of spawning and rearing areas, alteration of river flows, and impediments to migration. Changes in the estuary from dredging, filling, and diking have removed or degraded large areas of tidal swamp and marsh. Dam construction has changed the timing and magnitude of river flow, which in turn has affected water depth and velocity, sedimentation rates, and salinity intrusion. Mitigation practices, including hatchery supplementation to recover stocks affected by the many anthropogenic impacts, have been attempted throughout the 20th century, but have failed to restore the diversity and abundance of fish stocks. Current mitigation activities seek to restore critical rearing habitat previously lost through anthropogenic changes in the lower Columbia River and estuary.

Juvenile salmonids originating from upland freshwater spawning grounds in the Columbia River Basin migrate to the ocean through a complex environment of tidal freshwater and saline estuarine habitats. However, use of the lower Columbia River and estuary by diverse salmon stocks is poorly understood, and this lack of information impedes restoration planning. In 2002, a team of researchers organized by the NOAA National Marine Fisheries Service initiated a research program to address specific information needs identified in a review of previous estuary investigations. This team conducted systematic sampling from 2002-2008 to address these information needs.

Conceptual Approach

For the research described in this report, we adopted the broad definition of *estuary*, which encompasses the entire complex of ecosystem gradients, ranging from fluvial to nearshore-ocean ecosystems. This definition was based on tidal variation, rather than on salinity and geomorphology, as controlling hydrological, sedimentological, geochemical, and ecological gradients. According to this definition, the estuary extends 233 km upstream of the Columbia River mouth to the base of Bonneville Dam. We further characterized the estuary as a continuum of tidal freshwater, oligohaline, euryhaline, and near-ocean plume ecosystems. This definition follows the *Columbia River Estuary Ecosystem Classification* (Simenstad et al. 2011), which establishes a biophysical framework for research, monitoring, and management of this large-river estuary. The field studies summarized here encompass tidal freshwater, oligohaline, and euryhaline habitats within the lower 100 km of the estuary.

Our research design was based on the conceptual framework of the report of Bottom et al. (2005b), *Salmon at River's End: The role of the estuary in the decline and recovery of Columbia River salmon*, which is referred to throughout this document and abbreviated as SARE. The SARE report documented historical decreases in estuarine rearing opportunity for young salmon following the widespread loss of tidal wetlands and the construction of main-stem dams. The report offered evidence that these and various changes upriver had reduced life history diversity in juvenile Chinook salmon, based on apparent decreases in estuary residency, the range of salmon sizes that enter the estuary, and the timing of estuary entry. It assumed that the expression of salmon life histories reflected the diversity of habitat opportunities throughout a river basin, including those within the estuary (Bottom et al. 2005b). According to the conceptual framework established and reported in SARE, salmon performance can be viewed as the product of three key elements (adapted from Simenstad and Cordell 2000):

1. *Habitat opportunity*—the ability of juvenile salmon to access a habitat. Opportunity is largely controlled by physical processes that shape habitat structure in the estuary and determine whether environmental conditions such as depth, current velocity, water temperature, etc., are satisfactory for salmon to occupy a site.
2. *Habitat capacity*—the qualities that promote salmon production within estuarine habitats, including conditions for feeding, growth, assimilation efficiency, and predator avoidance. In contrast to the physical variables that largely govern habitat opportunity, habitat capacity is often determined by density-dependent biological interactions or bioenergetic relationships.
3. *Life history/population structure*—genetic and phenotypic factors that influence estuary use among individuals in a population. Population structure and behaviors at other (i.e., marine and fresh water) life stages directly affect the rearing and

migratory patterns of downstream migrants, and in turn, may determine whether juvenile salmon can fully realize the habitat opportunities and capacities in the estuary.

Each of these elements is also influenced by various physical processes that shape habitat conditions within and outside the estuary, including the effects of regional climate, flow regulation by main-stem dams, and changes in estuarine bathymetry.

The individual studies conducted for this project are grouped for presentation into four parts, each of which corresponds to a key element of our conceptual framework. Our research and monitoring program was also designed to fill specific data gaps identified in the SARE report, which reviewed present knowledge about the estuarine ecology of Columbia River salmon (Bottom et al. 2005b). Below we describe the data gaps, or critical uncertainties identified and reported in SARE as they correspond to our research and monitoring activities of salmon and their estuarine habitats from 2002 to 2008.

Historical Change in Habitat Opportunity

Historical habitat changes summarized in SARE were derived from a relatively coarse analysis of nautical charts for the lower estuary, from river kilometer (rkm) 0 to 76 (Thomas 1983). No historical data were available for the large tidal-freshwater region that extends upriver as far as Bonneville Dam (rkm 233). To address this data gap, we created a detailed historical template for analyzing habitat changes throughout the estuary. This template was formed by digitizing in a Geographical Information System (GIS) the entire series of 19th-century topographical and hydrographical survey maps from the river mouth to Bonneville Dam. We updated Thomas' (1983) previous estimates of lower-estuary habitat change by comparing our historical GIS template with habitat distributions classified from recent Landsat satellite imagery. These results are summarized in earlier reports of this research by Bottom et al. (2008) and Roegner et al. (2008).

Salmon access to shallow-water rearing habitat has been affected by hydrological and climatic changes in the estuary that are poorly understood (Bottom et al. 2005b). We synthesized historical data for Columbia River tides, river flows, and water temperatures to determine whether these physical changes have influenced estuarine habitat opportunities or capacities for juvenile salmon. We also expanded a previous analysis of the effect of flow on the amount of shallow-water habitat available in the reach from Skamokawa to Beaver, OR (rkm 55-85; Kukulka and Jay 2003a,b; Bottom et al. 2005b). This analysis was expanded to include the adjacent reach from Eagle Cliff to Kalama, WA (rkm 83-120).

Large-scale changes in climate and tidal processes over the last century have also changed the dynamics and quality of habitats available to salmon. Important climatic influences have included, for example, a long-term increase in temperature, a trend toward lower flows, and flow variations related to large-scale climate indices such as the Pacific Decadal Oscillation (PDO) and El Niño-Southern Oscillation (ENSO). These and other natural variations have interacted with anthropogenic changes to modify habitat-forming processes in the estuary. For example, river flows control estuarine habitat opportunity directly by altering flood stage and indirectly by influencing tidal range.

Changing coastal tides and shifts in the annual flow cycle linked to hydropower development have displaced juvenile salmon habitat in time and space (Kukulka and Jay 2003b). Maximum habitat opportunity for juvenile salmon now occurs earlier in the migration season than it did historically, while reduced spring river levels restrict the extent and distribution of habitat to lower elevations and locations closer to the thalweg. The spring tidal range has increased, and the displaced (i.e., low-elevation) habitat is now more strongly affected by tides (Kukulka and Jay 2003b). Part I details our reconstruction of historical habitat opportunities and changes in the estuary as influenced by the tide, river flows, and temperature.

Habitat Opportunity: Estuarine Fish Species and Salmon Habitat Use

Conclusions about salmon habitat use and life histories reported in SARE (Bottom et al. 2005b) were based on the results of relatively few ecological surveys in the lower Columbia River, supplemented by research findings from other Pacific Coast estuaries. Thus, further validation was required of interpretations about salmon-habitat relationships, food-web linkages, or the effects of estuarine habitat change.

For example, the conclusion that tidal wetlands provide important rearing habitat for salmon with subyearling life histories was drawn primarily from studies of small coastal drainages, where physical conditions differ substantially from those of the river-dominated Columbia River estuary. Despite considerable monitoring of sandy habitats along the estuary shoreline and near the main-stem channel (e.g., Dawley et al. 1986; McCabe et al. 1986), fish assemblages in tidal wetlands and other off-channel habitats had not been systematically surveyed at the time of the SARE report. Moreover, because most Columbia River salmon are now produced in hatcheries, their estuarine rearing behaviors may differ substantially from those of other (i.e., naturally produced) populations and ecosystems.

We addressed these uncertainties by monitoring fish assemblages in a diversity of emergent, forested, and scrub-shrub wetlands and in a series of nearshore beach-seining sites in the lower 100-km of the Columbia River estuary. We sampled shallow near-shore habitats rather than deeper channel areas to target subyearling Chinook, which is considered the most estuary-dependent of salmon species and life-history types (Healey 1991; Bottom et al. 2005b).

Our study design incorporated landscape- and habitat-scale surveys in the lower 100-km of the estuary. At the landscape scale, we sampled salmon monthly along the tidal (salinity) gradient to compare abundance and size distributions of individuals as they moved through the lower estuary and to the river mouth. At a finer scale, we sampled fish in a variety of wetland habitats that represented different stages of ecological succession, from tidal freshwater marsh (including emergent, scrub-shrub, and forested wetland habitats) to mixed wetland types (scrub-shrub and forested). We monitored fish use of each wetland type and investigated physical factors that affect salmon rearing opportunity throughout the spring and summer months. In Part II, we depict contemporary habitat opportunities based on present-day patterns of salmon distribution and abundance and upon various physical factors that influence fish access to shallow-water rearing areas.

Habitat Capacity: Prey Availability, Diet, and Rates of Consumption

An estimated 82% reduction in the historical sources of estuarine wetland and benthic macrodetritus (Sherwood et al. 1990) has raised concern about the status of food webs that support juvenile salmon in the Columbia River estuary. Yet neither food-web responses to carbon-source change nor the specific effects of such change on juvenile salmon have been measured directly. In the Columbia River estuary, most previous studies of invertebrate prey resources and salmon diets have occurred in or near deep channels and main-stem habitats, where large subyearling and stream-type salmon typically predominate. Shallow-water habitats such as the bays, sloughs, emergent marshes, forested wetlands, and flood plains often used by smaller subyearling salmon (Levy and Northcote 1982; Healey 1991; Miller and Simenstad 1997; Miller and Sadro 2003; Bottom et al. 2005a), have been poorly represented in previous studies of the Columbia River estuary (Bottom et al. 1984; McCabe et al. 1986; Bottom and Jones 1990; Simenstad et al. 1990). The feeding ecology of salmon in this large estuary thus remains poorly understood (Bottom et al. 2005b).

The SARE report hypothesized that productive capacity of the estuary has declined in the last century because diking and filling have eliminated large quantities of shallow-water habitat that historically supported salmonid food webs (Bottom et al.

2005b). These effects on production from habitat loss were inferred from changes in the estimated quality and quantity of carbon sources between historical and contemporary conditions in the estuary (Sherwood et al. 1990; Simenstad et al. 1990). However, these inferences highlighted a gap in the data, where empirical measurements of prey availability or salmon stomach contents and performance (i.e., foraging success or growth) were needed.

Therefore, we initiated a series of studies to characterize salmonid food webs in the Columbia River estuary, investigate the importance of shallow-water habitats as food-producing areas, and interpret the effects of historical habitat loss on the performance of estuarine-rearing salmon. The results of stable isotope studies to identify estuarine organic matter sources (Anderson 2006; Maier et al. 2011) and the food-web pathways of juvenile Chinook salmon (Anderson 2006; Maier and Simenstad 2009) are reported elsewhere.

Our study design combined samples from the tidal-gradient and habitat-scale surveys in the lower 100-km of the Columbia River. We sampled a series of beach-seine sites between the river mouth and Lord Island to document spatial and temporal trends in salmon diet across a broad salinity and tidal gradient. At the habitat scale, we quantified the contribution of different wetland types to salmonid food webs by monitoring prey species composition and abundance and Chinook salmon diets and (at a few sites) diel consumption patterns. We compared prey resource availability and salmon prey selection for a variety of tidal conditions, geomorphic features, and wetland successional stages. Finally, we examined food-web interactions between the dominant fish species (threespine stickleback) and subyearling Chinook salmon to investigate potential behavioral and competitive influences on salmon feeding ecology and foraging success.

Performance Metrics: Population Structure, Life History Diversity, and Growth

The estuarine performance of juvenile salmon is not determined solely by conditions within the estuary. Whether salmon can fully realize the habitat opportunities and capacities of the estuary also depends upon factors that influence salmon population structure and the behavior of downstream migrants (Bottom et al. 2005). Time of estuary entry, size at entry, period of residency, and migratory pathways through the estuary may vary substantially among populations of different geographic origin, genetic composition, or fresh-water life history. Thus, various rearing and migration behaviors in the estuary are directly connected to the geography of spawning populations and the timing of prior life history events.

The sparse frequency and distribution of estuarine fish surveys have limited understanding of life history diversity among Columbia River salmon. Since the first Chinook salmon survey in the basin (Rich 1920), no subsequent estuary study has retained scales or otolith samples. These samples would allow reconstruction of the freshwater and estuarine rearing histories of individual juvenile salmon migrants. Without them, year-to-year changes in the relative proportion of various life-history types leaving the basin cannot be quantified. The only contemporary estimates of estuary residency and growth for Chinook salmon have been derived from recoveries of large, tagged hatchery fish (Bottom et al. 2005b; Burke 2005). These individuals may not represent the full range of rearing behaviors in Columbia River populations, particularly those of smaller subyearlings.

In the absence of contemporary life history surveys, the life history composition of juvenile Chinook salmon was hypothesized based on lower-estuary abundance patterns and size distributions and on the estuary residency of various groups of marked hatchery fish (Dawley et al. 1986; Bottom et al. 2005b; Burke 2005). These results suggested that the historical diversity of juvenile salmon life histories in the basin has been simplified and is now concentrated into just three predominant types:

1) A small group of fry migrants that enter the estuary from February through June
2) A dominant group of riverine smolts (including a large proportion of hatchery-reared juveniles) with short estuary residence times (i.e., days or weeks) that peak sharply in June and July but decline rapidly thereafter
3) A group of yearling fish that migrate rapidly through the estuary, predominantly from April to June

Individuals of different sizes, arriving at different times and from different areas of the basin may express distinct rearing behaviors as they traverse the estuary tidal gradient. For example, very different estuary life histories may be exhibited by a population of upper Columbia Basin spring Chinook salmon, composed predominantly of large yearling migrants vs. a population of lower Columbia River fall Chinook salmon composed primarily of emergent fry and fingerling migrants. Therefore, estuary restoration efforts to benefit salmon must account for stock-specific differences in juvenile life history and the habitat opportunities that allow for these variations to be expressed.

The effects of upriver population structure on estuary rearing behavior and habitat requirements are poorly understood, partly because the tools were lacking to distinguish stocks of origin among individuals from numerous source populations that intermingle in the estuary. Until recently, stock affiliations were known only for hatchery fish marked with identifying tags or fin clips. Recent progress in genetic analysis has afforded new

opportunities to discern the stock origins of unmarked Columbia River Chinook salmon. One example is the successful application of a recent microsatellite DNA baseline (Seeb et al. 2007) to analyze the stock affiliations of subyearling Chinook salmon in lower Willamette River tidal habitats (Teel et al. 2009).

We applied this genetic baseline to estimate the stock-group affiliations of individual fish collected throughout the lower 100-km of the estuary. We analyzed the genetic composition of juvenile Chinook salmon sampled from 2002 to 2007 during the tidal gradient and wetland-habitat surveys described in Part II. We summarized the temporal/spatial and size-frequency distributions of all genetic stock groups and compared the proportional representation of each group among estuary regions and wetland habitat types.

In addition, we reconstructed estuary life histories using otolith microchemistry for Chinook salmon from selected sites sampled during 2003-2005. Recent life history studies have verified that otolith chemistry is a sensitive indicator of salmon entry into saline environments. Otolith analysis provides a quantitative method to reconstruct the freshwater and estuarine rearing histories of individual fish (Volk et al. 2010). These analyses were used to quantify salmon residence time in the brackish portion of the Columbia River estuary and to reconstruct time of and size at salt-water entry. We then compared life history attributes among individuals from different genetic stock groups.

To examine rearing and migration behaviors on a finer-scale, we also marked fish with passive integrated transponder (PIT) tags and used detection of these tags to quantify residence time and growth of individual Chinook salmon in selected channels of the Russian Island emergent wetland. Combined, our genetic, otolith, and PIT tag results provide a comparative snapshot of present-day genetic and life history diversity at both habitat and lower-estuary scales. In Part IV, we examine the effects of upriver population structure and life histories on estuary rearing behaviors, including the genetic sources of individual Chinook salmon found within particular habitats and stock-specific patterns of estuary residency and growth.

Below we present the methods and results for each of the evaluations conducted from 2002 to 2008 for Parts I-IV. We then enumerate our overall conclusions based on all of these evaluations, along with our recommendations for future research, restoration, and recovery efforts.

I. HISTORICAL CHANGE IN HABITAT OPPORTUNITY

Historical Datum and Interaction Between Tides and River Flow

Methods

We developed a digital base map in a Geographical Information System (GIS) to depict the pre-development distribution of habitats throughout the Columbia River estuary (river mouth to Bonneville Dam). Our map was based on a comprehensive series of topographic and hydrographic surveys conducted in the late 19th and early 20th centuries. Estimates of historical habitat change using this baseline require accurate and consistent datum levels, which were applied across all surveys. Here we evaluate datum levels for each of 14 historical hydrographic survey sheets prepared for the Columbia River estuary during the late 19th century. Complete data for the hydrographic surveys (H-sheets and T-sheets) are available from the *Historic Columbia River estuary* website (WET 2010).

We then analyzed historical changes in tidal datum levels for all Columbia River tide stations from ca 1940 to the present to investigate the combined effect of tides and river flow and to distinguish anthropogenic from tidal influences. We interpreted the modulation of tides by river flows based on the wavelet tidal analysis described by Kukulka and Jay (2003a). Datum levels were extracted as a function of river flow and other external forcing using multiple linear regression analyses (Kukulka and Jay 2003a). Discrete regression model coefficients provided values for 21 stations at 5-10 year intervals from 1940 to 2005, with many gaps (Appendix Table B6). The resulting coefficients allowed stage calculations from rkm 21 to 230 at 1.6-km increments, with annual changes to the coefficient values from 1942 to 2004. We also applied an improved harmonic analysis method to extract tidal characteristics from noisy tidal records with gaps (Leffler and Jay 2009). Detailed methods for these analyses are presented in Appendix B.

Results

We analyzed datum levels for 14 historical bathymetric surveys, recorded as hydrographic sheets (H-sheets). These surveys were conducted between 1877 and 1901, and we examined each H-sheet for consistency both internally and with other documentary evidence. Historical datum was provided by the National Ocean Service (formerly U.S. Coast and Geodetic Survey) and the U.S. Army Engineers, Portland District, Northwestern Division. We determined the datum level applied to each H-sheet and its apparent relationship to Columbia River Datum (CRD), evaluated the plausibility

of each value, and estimated the effects of any systematic errors. The near-simultaneous occupation of nine stations from Cathlamet to Warrendale, OR, in September 1877 provided the best data set for evaluating the consistency and quality of H-sheets. The general pattern (with the exception of St. Helens, OR) indicated that the established datum was approximately 0.2-0.3 m above CRD, and that the quality of work was quite good. For reasons that are unclear, other (generally higher) datum levels were used in the 1880s. The estimated datum for each H-sheet relative to CRD is presented in Appendix B.

The tidal datum analyses for Vancouver, WA (Figure 1) provided a near-continuous record of mean water level (MWL) since 1902 and of higher high water (HHW) and lower low water (LLW) for 1940-1942 and after 1972. We estimated that at The Dalles since ca 1900, MWL has dropped by ~0.7, 0.8, 1.1, and 1 m for each of four respective flow levels: 2,000, 5000, 7,500 and 12,500 m^3s^{-1}.

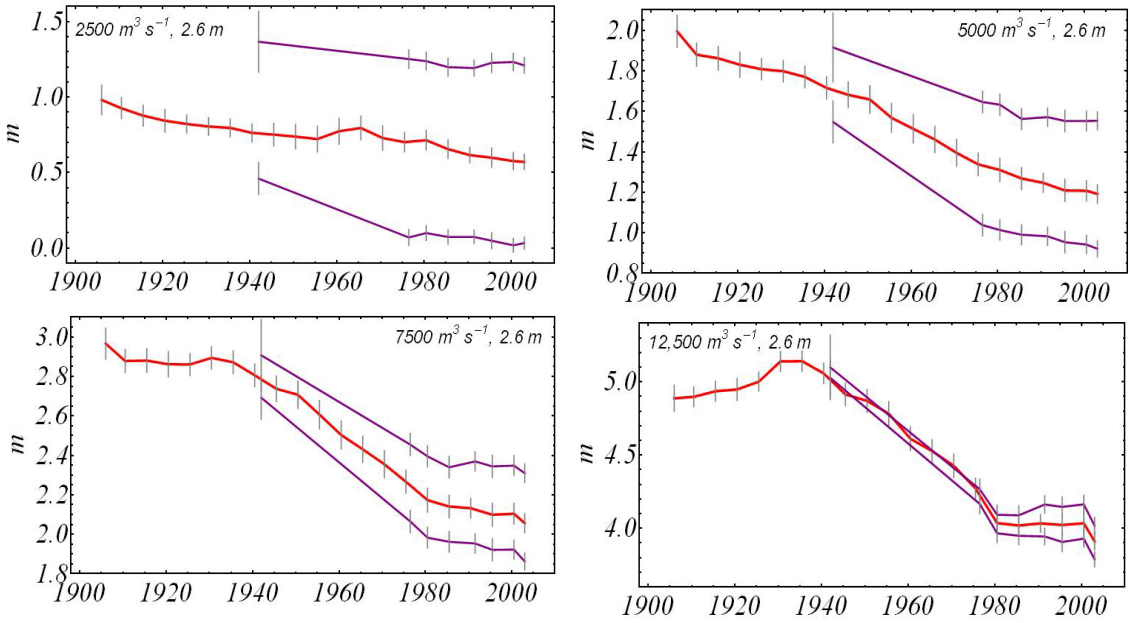


Figure 1. Each panel shows higher high water (upper line), mean water level (center line), and lower low water (lower line) at Vancouver, WA, in 5-year intervals. Water levels at Vancouver reflect flow levels at The Dalles of 2,500, 5,000, 7,500 and 12,500 m^3s^{-1}, all for a tidal range of 2.6 m at Tongue Pt. Gray bars indicate 95% CIs.

At the lowest flow level (2000-2500 m^3s^{-1}), LLW has decreased almost continuously since 1940, but HHW has increased in recent years, reflecting an increase in greater diurnal tidal range (GDTR), the difference between HHW and LLW.

The increase in GDTR was most prominent at flows of 5000 m^3s^{-1}, although GDTR increased for all combinations of flow level and tidal condition at Tongue Pt. The LLW decrease of ~0.4 m since 1940 for flows of 2000-2500 m^3s^{-1} is important to navigation, because low water levels limit the draft of loaded ships. The HHW decrease of ~1.3 m since 1940 at flows of 12,500 m^3s^{-1} has likely reduced shallow-water habitat opportunity for juvenile salmon, because decreases in the HHW level limit inundation of shallow areas.

Regional Tidal Evolution

Methods

Tidal properties are changing rapidly throughout the Columbia River, in part as a symptom of navigational development and sand removal. Tidal evolution also has a regional component that has been evaluated for all long-term tide stations along the eastern Pacific Coast (Jay 2009). To determine the long-term evolution of the major diurnal (K_1) and semidiurnal (M_2) tidal constituents, hourly tidal time series for each station were convolved with filters tuned to the appropriate frequencies, providing an amplitude and phase for each constituent and time period. We used 3-year time windows to isolate the frequencies. The hourly astronomical tidal potential for each station was analyzed in the same manner. Yearly values of admittance (complex ratio of tidal response to astronomical forcing) for the two frequencies were determined and resolved into an amplitude ratio and a phase difference. A trend in admittance amplitude was converted to a trend in amplitude, and the results interpreted.

Results

Analyses of all available Eastern Pacific tidal records for stations with records of ~45 years or more indicated that tidal amplitudes were increasing throughout the Northeast Pacific Ocean, north of about 19°N. San Francisco, Tongue Pt., and Queen Charlotte City all exhibited a rapid increase in tidal range, but the rate of increase in relative tidal range at Astoria (Tongue Pt.) was higher than that at any other station in the region (Figure 2).

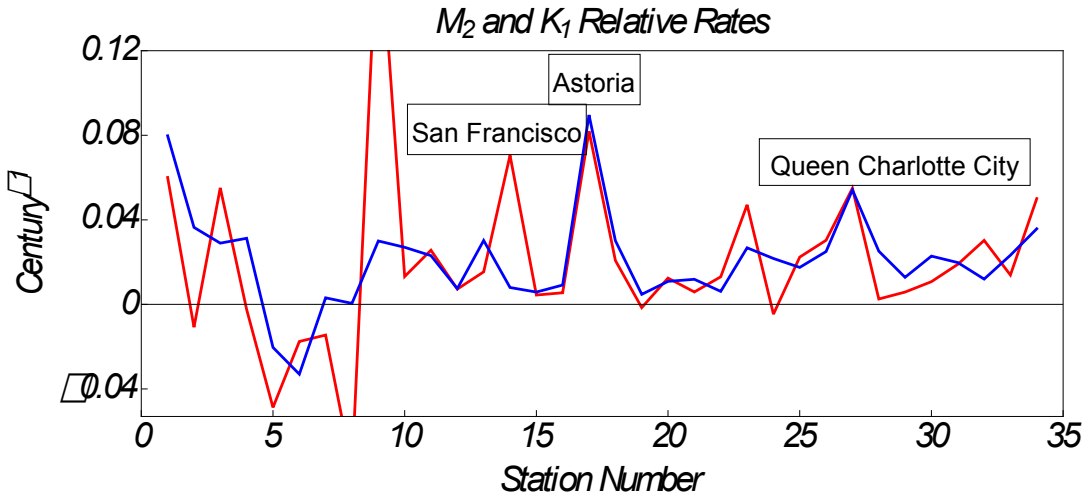


Figure 2. The relative rate of tidal evolution (absolute rate of change in mm century^{-1} divided by amplitude in mm) for the dominant diurnal (K_1, blue) and semidiurnal (M_2, red) constituents from Chile (at left) to Alaska (at right).

The rate of increase in tidal range at Astoria since 1925 has been about 0.3 m (1 ft) or 16% century^{-1} (twice the average of the diurnal and semidiurnal rates). Absolute and relative rates of range increase at Wauna appeared larger than at Astoria, but the record was less complete. The coastline from San Francisco to Astoria represented a hot spot for increasing tidal range. We compared the Astoria data to shorter records for South Beach (Newport Bay) and Charleston (Coos Bay). Results of this comparison indicated that roughly half the rate of change at Astoria was due to local estuarine and fluvial alterations, with the remaining half reflecting change in coastal tides.

Shallow-Water Habitat Area

Methods

We analyzed shallow-water habitat area (SWHA), defined as habitat between 0.1 and 2.0 m above CRD, for a variety of flow and tidal scenarios in a study reach from Eagle Cliff, WA, to just below Kalama, WA (rkm 83-120). This reach encompasses an area just landward of the Skamokawa-to-Beaver reach (rkm 55-85), which was analyzed by Kukulka and Jay (2003a,b). The Eagle Cliff-to-Kalama reach includes narrow entrenched topography as well as broad floodplains (Figure 3). For the analysis, a digital elevation model (10-m resolution) was created from a LiDAR (light distance and ranging) topographic survey flown in 2005 (Puget Sound LiDAR Consortium), and bathymetric data were compiled by the University of Washington. We identified the inundated area, merged the bathymetric and LiDAR data, and interpolated gaps between the bathymetric and LiDAR coverage. Digital ortho-photo quadrangles (DOQ) helped to define shoreline features.

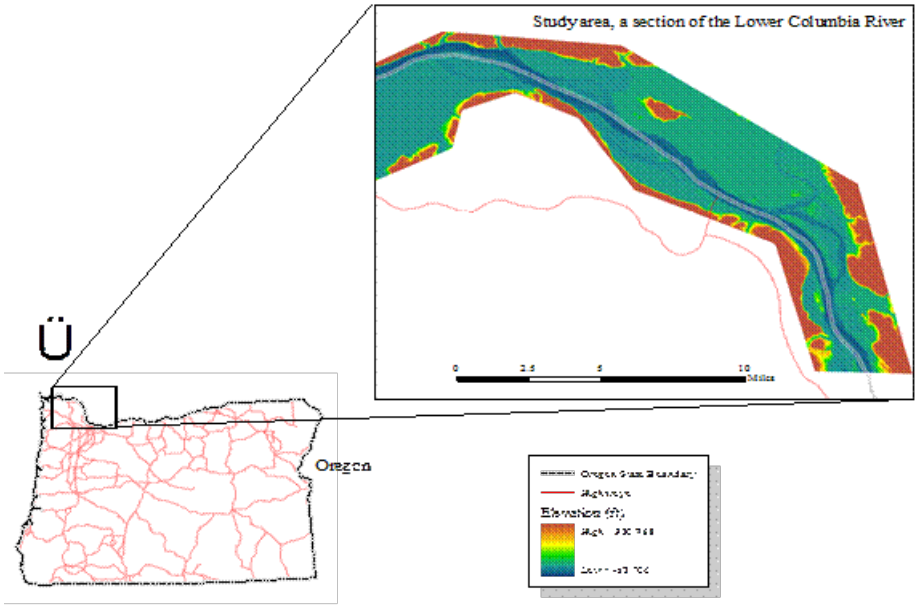


Figure 3. Map of the shallow-water habitat study reach from rkm 83 at Eagle Cliff, WA (at left on expanded area) to rkm 120 near Kalama, WA (lower right of expanded area). State boundary is shown by dotted line.

The 40-km reach was divided into 1.6-km segments. Two sets of shape-files, containing levee line-work and polygons representing flood-protected areas, were used to delineate segments and to locate levee-protected areas. To compare with previous results, we estimated shallow-water habitat area for the same four scenarios applied by Kukulka and Jay (2003a): virgin flow without dikes, virgin flow with dikes, actual flow without dikes, and actual flow with dikes. Virgin flow refers to river flow in the absence of dams and irrigation withdrawals (Naik and Jay 2005). Our analysis differed from that of Kukulka and Jay (2003a,b) in that it used improved LiDAR floodplain topography, accounted for rapid tidal evolution, and analyzed effects on shallow-water habitat area for a longer period (1925-2004 vs. 1974-1998).

We calculated hypsometric curves for each 1.6-km segment to estimate shallow-water habitat area as a function of river stage. These curves, which related river stage to inundated area, were derived for two elevation models, representing diked (modern) and undiked (historical) conditions. This approach allowed more rapid computation of shallow-water habitat area than other analyses based on hydrodynamic equations. However, it did not automatically account for the topographical connectivity of areas behind levees. The calculation of shallow-water habitat area for levee-protected scenarios required manipulation of the original digital elevation model to numerically prevent flooding of the protected areas except during river stage conditions when the levees would be overtopped.

The 1.6-km segments then were aggregated into four larger sub-reaches for each development scenario, each with a different hypsometric curve shape (floodplain vs. entrenched): rkm 82-87, 87-93, 93-106, and 106-120. The defined relationship between river stage and shallow-water habitat area provided a concise numerical depiction of reach flooding.

Results

The response of shallow-water habitat area to river stage with and without diking varied among the four sub-reaches between Eagle Cliff and Kalama (Figure 4). Diking raised the elevation required to inundate a significant amount of shallow-water habitat in all four sub-reaches, but the effects were most pronounced in sub-reaches 1 and 3, and least pronounced in sub-reach 4. The consistent height of dikes in sub-reaches 1 and 3 established an effective floor for inundation of 4 m at Beaver Army Terminal and of 7.5 m in the Longview, Washington area. This presumably reflected the amount of development under protection in these two sub-reaches. Dike heights in the other two sub-reaches were more variable.

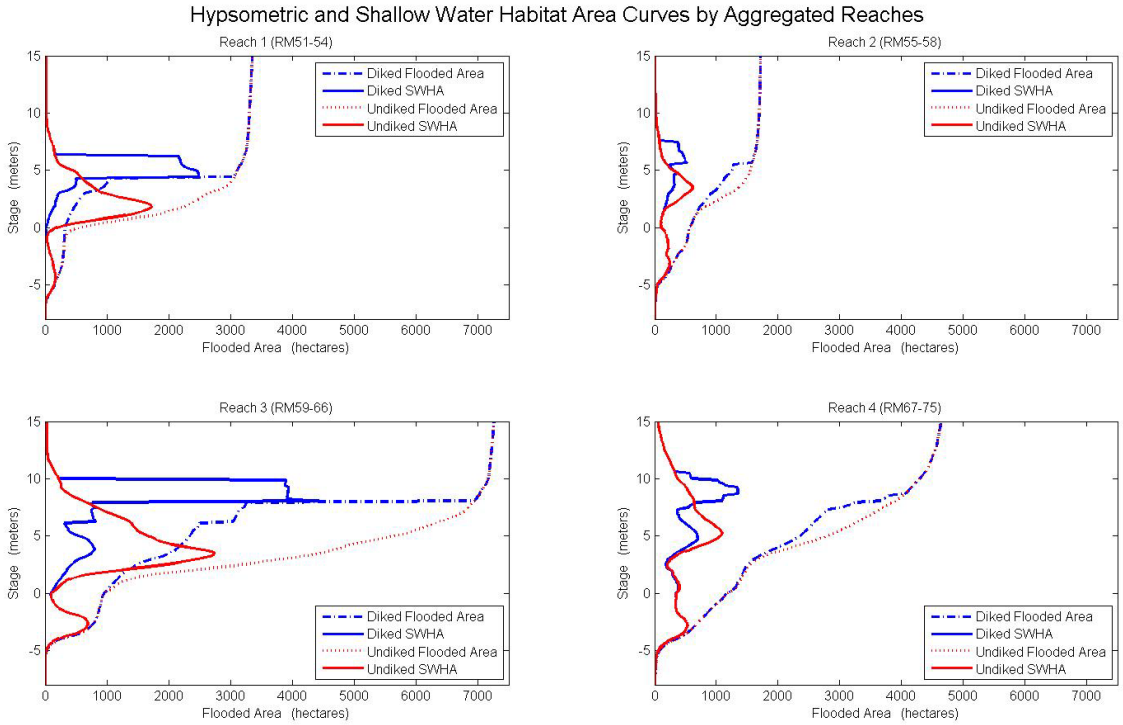


Figure 4. Sub-reach hypsometric and shallow-water habitat area (SWHA) vs. elevation curves for the Eagle Cliff to Kalama reach (rkm 83-120), for diked and undiked topography.

We compared the percent occurrence of varying amounts of shallow-water habitat area during an average non-freshet season (August-April; Figure 5) and average freshet season (May-July; Figure 6). We analyzed three time periods to compare shallow-water habitat area for different climatic and management conditions:

- 1925-1946, a period of warm conditions with a positive signal of the PDO (Pacific Decadal Oscillation) and little flow regulation, moderate flow diversion, and variable (but often low) flows
- 1947-1976, a colder period with negative PDO values, moderate-to-high flow regulation and diversion, and consistently high virgin flows
- 1977-2004, a time of maximum flow regulation and diversion and with variable PDO signals (warm/positive 1977-1995 and cold/negative or mixed from 1996).

Results were compiled using the hindcast maximum shallow-water habitat area (based on estimated daily HHW for each sub-reach) for each day from 1925 to 2004.

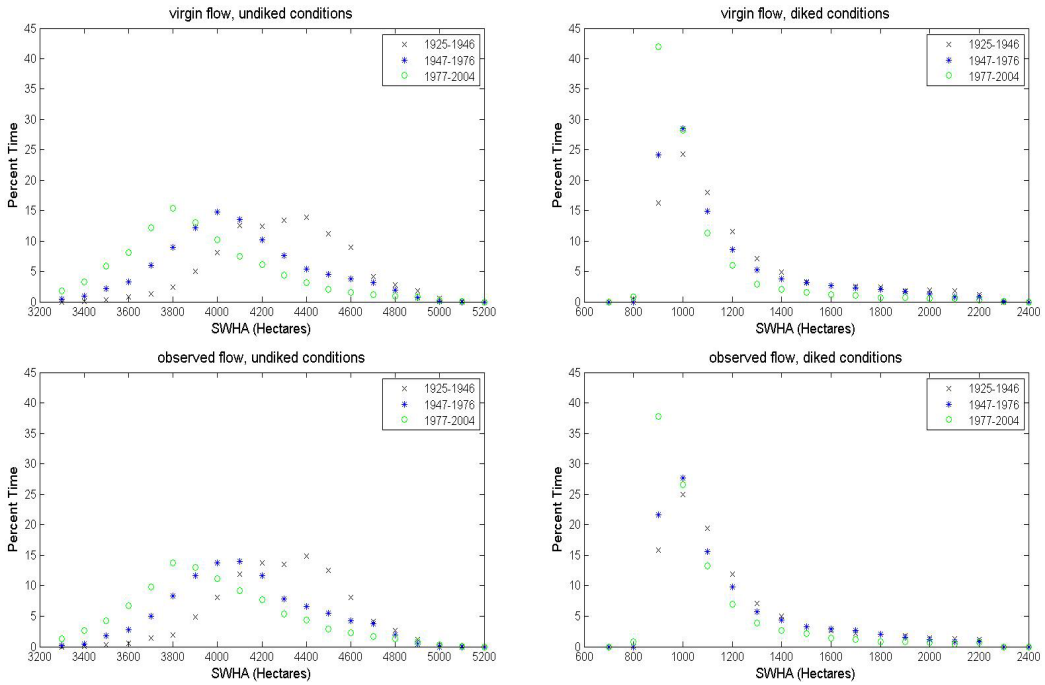


Figure 5. Percentage occurrence of SWHA for three climate periods based on all non-freshet season (August-April) days in 1925-2004, with SWHA summed over all four sub-reaches.

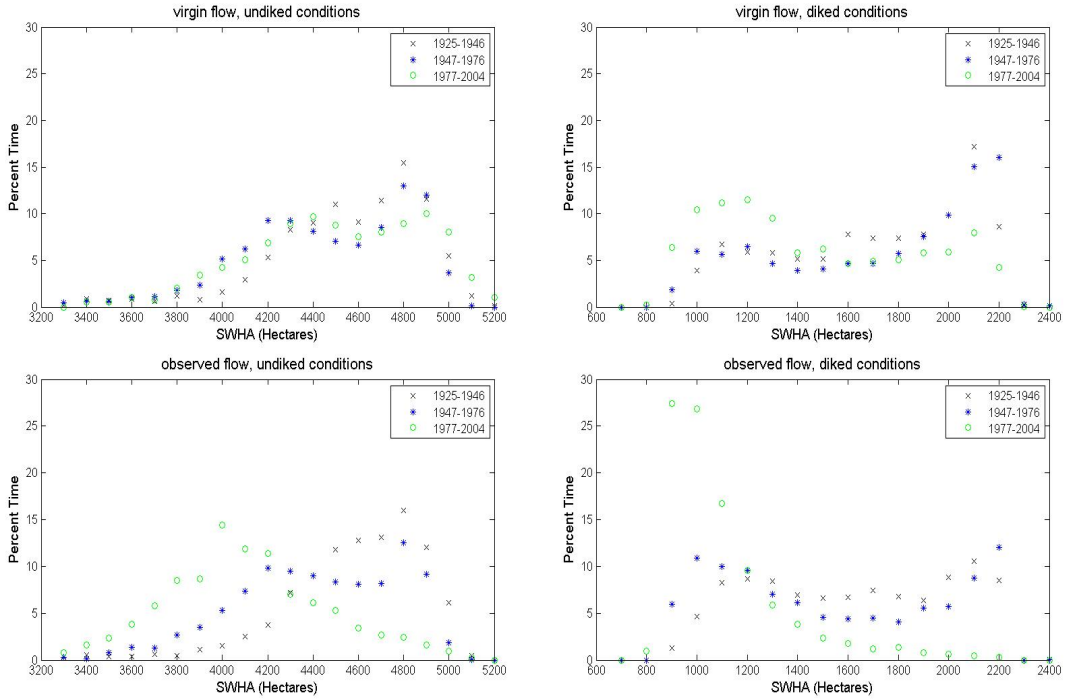


Figure 6. Percentage occurrence of SWHA for three climate periods based on all freshet-season (May-July) days in 1925-2004, with SWHA summed over all four sub-reaches.

These analyses indicated that for all scenarios, high values of shallow-water habitat area occurred least frequently during the latest period (1977-2004; Figures 5-6). These results were consistent with the reduction in flows during this period as documented below. However, even though mean flows were higher, peak values in shallow-water habitat area for undiked conditions (both observed and virgin flows) were lower for 1947-1976 than for 1925-1946, a period with lower mean flows. This resulted from large parts of the flood plain becoming inundated with more than 2 m of water during extremely high flows, thereby exceeding the 0.1 to 2.0-m depth criterion we used to define shallow-water habitat.

Average properties of shallow-water habitat area (as a percentage of 1925-1946) for the virgin flow/undiked scenario are shown in Table 1 for freshet and non-freshet seasons. Under these scenarios, SWHA for the high-flow period of 1947-1976 was slightly less than for the other periods examined. Estimated shallow-water habitat area for the observed flow/undiked scenario of 1925-1946 was slightly greater than that estimated for virgin flows during freshet and non-freshet seasons.

Table 1. Summary of average shallow-water habitat area during three PDO phases for freshet and non-freshet seasons in the study reach from Eagle Cliff to Kalama (rkm 83-120).

	Average available shallow-water habitat area					
	1925-1946		1947-1976		1977-2004	
	Hectares	%	Hectares	%	Hectares	%
Freshet season (May-July)						
Virgin flow, undiked	4,570	100*	4,450	97	4,552	100
Observed flow, undiked	4,634	101	4,454	97	4,146	91
Virgin flow, diked	1,769	39	1,853	41	1,520	33
Observed flow, diked	1,671	37	1,604	35	1,151	25
Non-freshet season (August-April)						
Virgin flow, undiked	4,347	100	4,329	100	3,945	91
Observed flow, undiked	4,350	100	4,166	96	4,014	92
Virgin Flow, diked	1,253	29	1,257	29	1,093	25
Observed Flow, diked	1,240	29	1,244	29	1,118	26

* Shallow-water habitat area for the virgin flow/undiked scenario (1925-1946) is taken as 100%.

The contrast between natural (virgin flow/undiked) and modern (observed flow/diked) scenarios was greater than we might have expected based on similar analyses for the Skamokawa-to-Beaver reach (rkm 55-85; Kukulka and Jay 2003a,b). Of the original shallow-water habitat area in the reach from Eagle Cliff to Kalama, fully 75% has been lost during the freshet season and 74% during the non-freshet season. Much of this loss, relative to that in the reach from Skamokawa to Beaver, was associated with diking. Flow regulation was a smaller issue but caused similar reductions in shallow-water habitat area in both reaches.

One encouraging difference in the reach from Eagle Cliff to Kalama was that little redundancy existed between the effects of diking and those of flow regulation on shallow-water habitat area. Therefore, the total loss of shallow-water habitat area was nearly the same as the sum of the individual components (diking and flow reduction). In contrast, in the Skamokawa-to-Beaver reach, losses from diking and flow regulation were considerably smaller than the total from both. This suggests that some restoration of shallow-water habitat area could occur between Eagle Cliff and Kalama without requiring a substantial change in the flow regime. However, this does not mean that the flood-control effects of diking and flow regulation are entirely independent in the Eagle Cliff-to-Kalama reach. It simply means that with reduced flows, floodplain areas too deeply covered to be classified as shallow-water habitat area under natural conditions are inundated to less than 2 m (i.e., our model criterion for shallow water habitat) under modern flow conditions.

During the non-freshet season in the reach from Eagle Cliff to Kalama, roughly the same amount of shallow-water habitat area was available with or without flow regulation. However, in both the freshet and non-freshet seasons, the shallow-water habitat of modern conditions was much closer to the thalweg and more tidal than that inundated by natural (higher) flows. That is, water elevations were higher in the absence of flow regulation, resulting in a shallow-water habitat area that extended further upward into the floodplain. This occurred because, as flows increased, some shallow-water habitat areas became inundated to depths greater than the 2-m threshold, such that the total increase was fairly small. In this respect, the Skamokawa-to-Beaver and Eagle Cliff-to-Kalama reaches were similar.

Change in Estuarine Water-Temperature Regimes

Methods

Historical changes in the temperature regime of the river were determined using the 1937-2002 Bonneville Dam daily scroll-case temperature record, atmospheric data (temperature and precipitation, 1890-2003), and river flow records. The 1938-1956 portion of the Bonneville record served as the base period for the model to represent conditions when reservoir storage and flow diversion were minimal. After 1956, heating from nuclear reactors at Hanford became an important factor in the fluvial heat budget. Gridded air temperature (T_A) data, available for 1915-2003 and described by Hamlet and Lattenmaier (2005), were obtained from the Surface Water Modeling group at the University of Washington (SWMF 2011). These data have been carefully corrected for changes in the sensor distribution over time.

Monthly and spatially averaged air temperature (T_A) for large parts of the interior sub-basin were highly correlated with monthly average water temperature (T_W) at Bonneville Dam. River flow was an important (but secondary) factor. However, different regression models were necessary for the January-to-June (winter-spring) and July-to-December (summer-fall) periods. These simple models accounted for 95% (winter-spring) and 97.5% (summer-fall) of the base-period (1938-1956) T_W, respectively (Figure 7) with no obvious bias to the residual. Despite poor predictive capability for a few extremely cold winter periods, the model accurately estimated T_W. Only the very high flows of 1894 were outside the range of the 1938-1956 base period, and flow variability was less important than T_A in determining T_W.

We hindcast historical T_W for a variety of climate and flow scenarios back to 1890. For the 1890-1914 period, it was necessary to determine air temperature conditions from a small number of instrumental observations, rather than from the spatially averaged data. To avoid bias associated with the choice of stations, we used a regression model to determine the relationship of the 1915-1927 spatially averaged air temperatures to the values from individual stations for which data were available, 1890-1914. These relationships were used to model the spatially averaged temperatures for 1890-1914, which in turn, were used to drive the T_W model scenarios.

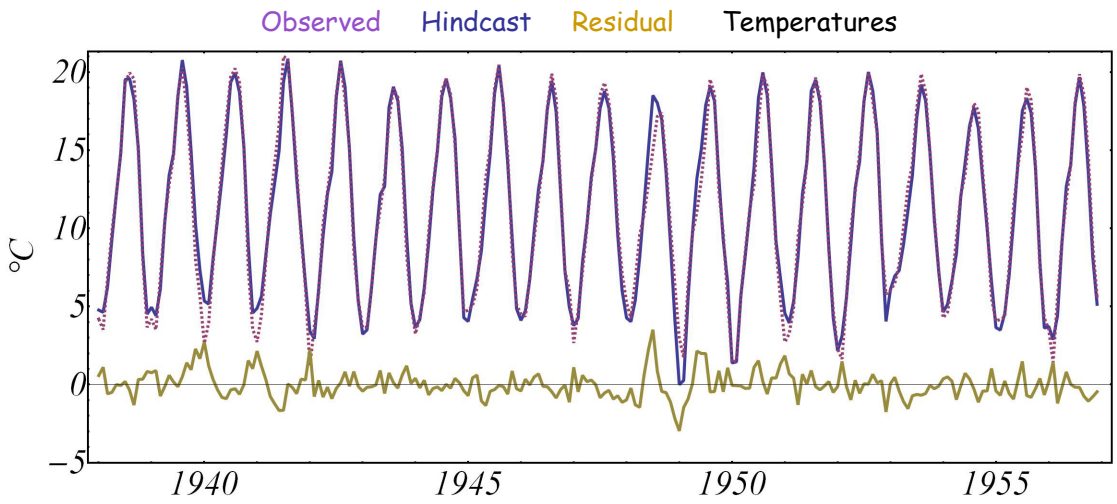


Figure 7. Observed and hindcast water temperatures (T_W)for the 1938-1956 base period, and the residual error.

Results

Water temperature trends at Bonneville Dam were indicative of river basin effects on water temperatures in the lower river. We considered four water temperature (T_W) scenarios:

1. *Observed Conditions*, 1938-2002: This scenario was T_W observed during 1938-2002.

2. *Base Period scenario*, 1938-1956: The T_W regression model for this scenario was based on the flow and air temperature (T_A) conditions during 1938-1956. In the absence of earlier water temperature data, this period was chosen to represent unaltered (pre-reservoir) conditions. However, Grand Coulee, Rock Island, and Bonneville Dams operated for part or all of this period, and together with moderate levels of irrigation diversion, would have modified T_W (Naik and Jay 2005). The regression model was used to hindcast T_W for the entire 1938-2002 period, using the actual (observed) T_A and flows for each year. This scenario closely reproduced Scenario 1 for the base period, 1938-1956.

3. *Virgin Flow scenario*, 1938-2002: This scenario differed from Scenario 2 only in that virgin flows were used (along with the observed T_A for each year) to hindcast T_W for the 1938-2002 period.

4. *Virgin flow cold scenario*, 1890-1926: This scenario differed from Scenarios 2 and 3 in that the T_A data for 1890-1926 were used along with the virgin flow for each year to hindcast T_W for the 1890-1926 period.

We can divide the observed temperature record at Bonneville Dam into four periods between 1938 and 2002:

1. *Base Period*, 1938-1956: This period represented the T_W regime during minimal reservoir manipulations and under moderate effects of irrigation. Warm climate conditions (positive signal of the Pacific Decadal Oscillation) prevailed during the first half of this period (1938-1946), but shifted abruptly to cold conditions (negative PDO) during 1947-1956.

2. *Hanford Period*, 1957-1976: During this period, reservoir manipulation and irrigation diversion increased, and the Hanford reactors added considerably to the heat load. Thus, water temperatures were considerably warmer than during the base period, despite cold climate conditions (negative PDO), which ended in 1976.

3. *Reservoir Period*, 1977-1996: This period occurred after the reduction of Hanford thermal inputs, but the river continued to warm, with positive anomalies of the PDO, warm climate conditions, and low river flows.

4. *Modern Period*, 1997-date: A transition to cold climate conditions (negative PDO) with much higher flows occurred during 1996-1998. At about the same time, cold sub-surface waters from Dworshak Dam were used to cool the Snake River. Although EPA modeling suggested that the effect of the Dworshak outflow disappears by the time the water reaches the Columbia mainstem (M. Soscia, EPA, personal communication), the relationship of T_W to T_A and flow changed after 1996. More data will be required to analyze this period.

By comparing these four temperature scenarios for the above time periods, we were able to evaluate the effects of changing management regimes on mean water temperature (Figures 8 and 9). Model results showed that historical temperatures (Scenario 4, 1890-1926) did not exceed 19°C in a typical year, even in August (Figure 8), whereas modern temperatures (Scenario 1, 1976-2002) averaged >19°C for 3 months (July-September). The 19°C temperature was used as a relative indicator of habitat opportunity, since many juvenile salmon begin to vacate shallow habitats in the estuary as temperatures approach this value (Bottom et al. 2008; Roegner et al. 2008).

Causes of change in the T_W cycle can be deduced from Figure 9. The differences between Scenarios 4 and 1 represented the sum of all historical effects on river temperature (i.e., heating caused by reservoir storage, reduced flows and slower transit of water even without reservoir storage, and climate change).

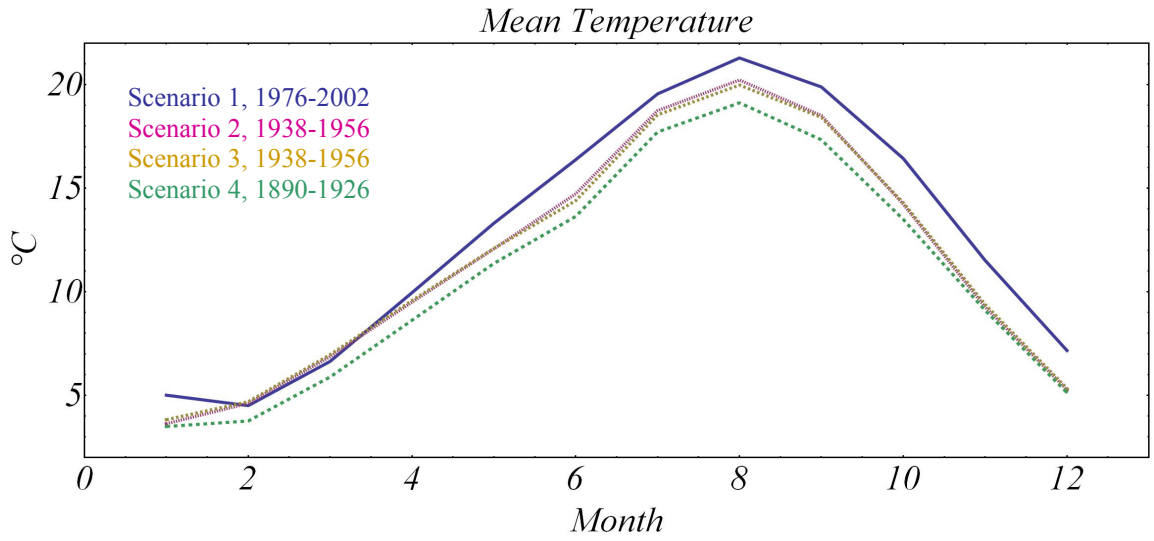


Figure 8. The hindcast average seasonal water temperature (T_W) cycles for January (month 1) to December (month 12). The averaging period over which each scenario is averaged is indicated. See text for details of the scenarios.

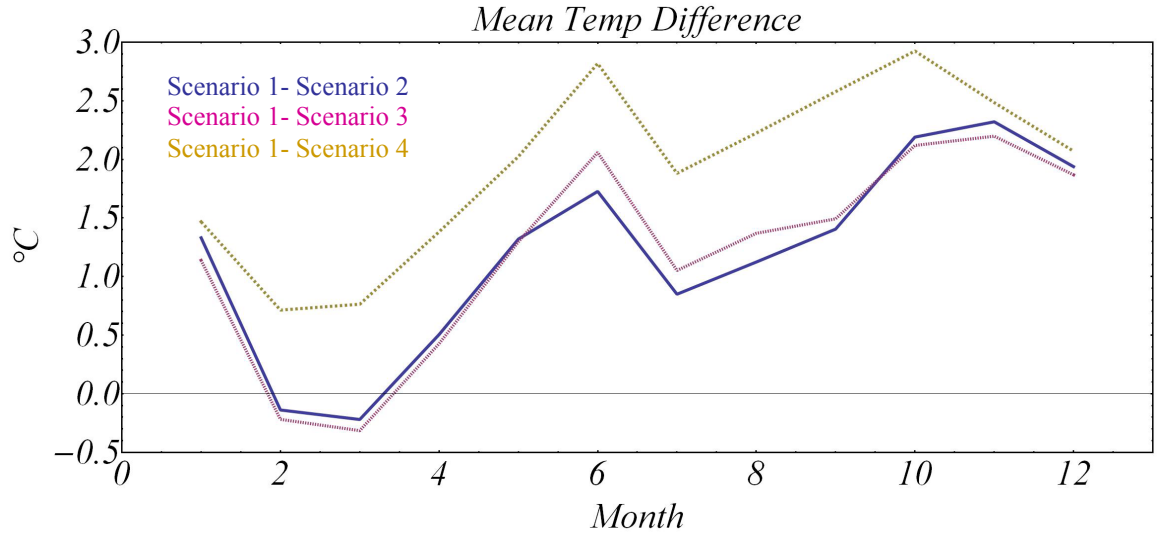


Figure 9. Differences in water temperature (T_W) between the scenarios presented in Figure 8.

The difference between Scenarios 3 and 1 represented the sum of impacts due to reservoir storage and flow reduction. The difference between Scenarios 2 and 1 represented primarily the differences due to reservoir storage. (These descriptions are approximate because a) the 1938-1956 base period still had some reservoir manipulation and flow diversion, and b) the model did not consider non-linear interactions among factors.) Comparison of the scenarios in Figures 8 and the differences between them in Figure 9 suggested that:

- As a result of all factors, water temperature is now 1.8-2.9°C warmer than historically (the difference between Scenarios 1 and 4) from May to December, potentially affecting both outbound juvenile salmonids and returning adults.
- Reservoir manipulations accounted for more than half of the total change in water temperature (difference between Scenarios 1 and 2). These changes vary from 0.8°C in summer to >2°C in October and November, when the river would cool much more rapidly without reservoir storage.
- Climate change accounted for 0.6-1°C of warming from June to October (difference between Scenarios 1 and 3 and difference between Scenarios 1 and 4).
- Water temperature is slightly lower (0.1-0.3°C) in February and March than it would be without reservoir manipulation and flow diversion, but is still warmer than under historical conditions (Scenario 1 − Scenario 3 and Scenario 1 − Scenario 2).

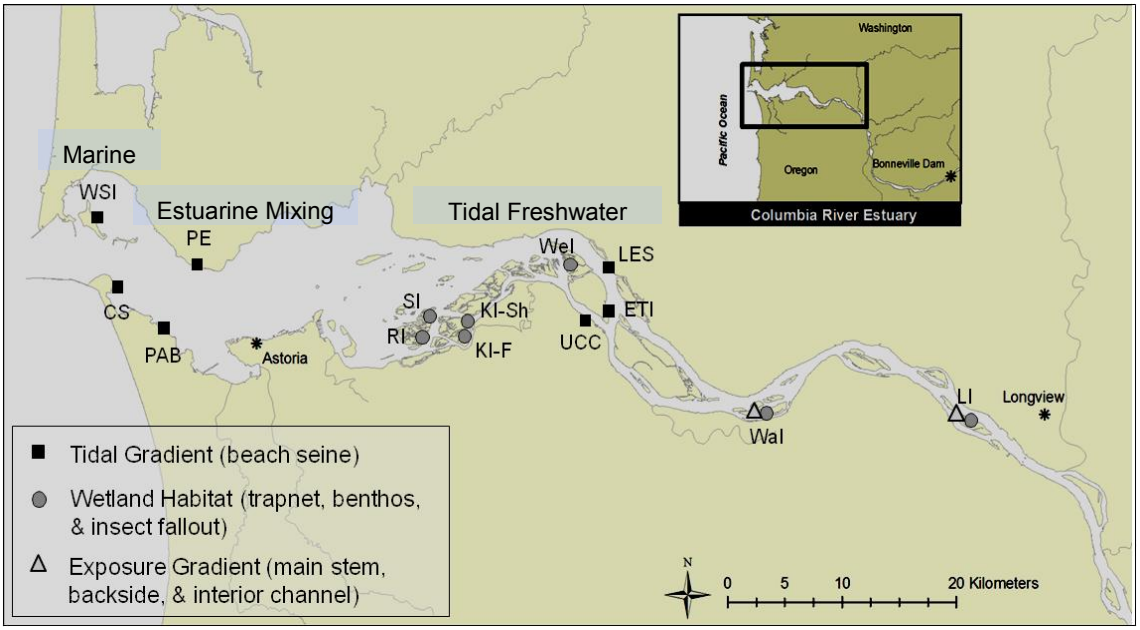
The change in volume of flow (without reservoir manipulation) had a relatively small effect on water temperature (a few tenths of a degree), but this effect was not insignificant, especially in combination with warming from climate change and reservoir manipulation (Scenario 1 − Scenario 2 vs. Scenario 1 − Scenario 3).

II. HABITAT OPPORTUNITY: ESTUARINE FISH SPECIES AND SALMON HABITAT USE

Distribution and Abundance along the Tidal Gradient

Methods

Study Area—We sampled fish communities at seven primary beach-seine sites along a broad salinity gradient, from tidal freshwater habitats near rkm 84 to saline habitats near the river mouth (Figure 10). Stations were paired between the Washington and Oregon sides of the Columbia River and grouped into three spatial zones based on general salinity characteristics. From 2002 to 2005, the tidal freshwater zone was represented by three beach-seine sampling sites upriver from Cathlamet Bay (Lower Elochoman Slough, Upper Clifton Channel, and East Tenasillahe Island). In 2006, we discontinued sampling at East Tenasillahe Island to extend tidal freshwater sampling further upriver to two supplemental sites at Wallace (rkm 77) and Lord (rkm 101) Island.



WSI	West Sand Island	SI-N, SI-S	Seal Island (north, south channels)
CS	Clatsop Spit	RI-N, RI-S	Russian Island (north, south channels)
PE	Point Ellice	KI-Sh	Karlson Island-Shrub
PAB	Point Adams Beach	KI-F	Karlson Island-Forested
LES	Lower Elochoman Slough	Wel-N, Wel-S	Welch Island (north, south channels)
ETI	East Tenasillahe Island	Wal-E, Wal-W	Wallace Island (east, west channels)
UCC	Upper Clifton Channel	LI-E, LI-W	Lord Island (east, west channels)

Figure 10. Sampling sites in the marine, estuarine mixing, and tidal freshwater zones of the Columbia River and estuary used for all experiments (Parts I-IV) from 2002 to 2008.

Throughout the study period, we also sampled two stations in the brackish estuarine mixing zone near the seasonally fluctuating boundary of the salt-freshwater interface (Pt. Ellice and Pt. Adams Beach) and two stations in the marine zone near the river mouth (Clatsop Spit and West Sand Island). Beach-seine surveys continued monthly from January 2002 through September 2007, and were generally conducted within 2 h of low tide. In some years, storms (primarily November to January) prevented monthly sampling in the marine and estuarine mixing zones. In 2008, we sampled only at Pt. Adams Beach and Upper Clifton Channel.

Fish Collection and Sampling—We evaluated fish species composition in shallow-water habitats near shore by sampling with a tapered 3- × 50-m variable-mesh (19.0, 12.7, and 9.5 mm) beach seine. The net was constructed with a knotless web in the bunt to reduce descaling of fish during capture. When deployed, one end of the seine was anchored on the beach while the other was towed by a skiff to enclose a surface area of about 397 m^2 (although the exact area varied with changing hydrodynamic conditions). If the initial set contained less than 10 salmon, a second haul was made adjacent to the first location. For non-salmonids, we measured live weight and fork length (FL) for 30 individuals of each species. Measured fish were then released, and remaining non-salmonids were counted and released.

During 2002-2003 we sacrificed a maximum of 10 individuals of each salmonid species and size class for genetic, otolith, scale, and stomach samples (methods and results described in subsequent sections of this report). We also measured (nearest 1.0 mm FL) and weighed (nearest 0.1 g) an additional 20 individuals of each salmonid species and size class. We retained samples of fin clips, scales, and otoliths for analyses of genetic stock, age, and growth and released these individuals back into the estuary.

During 2004-2007, we retained up to 30 individual salmon for laboratory study, measured and released up to 70 additional individuals, and counted and released the remainder. All sacrificed salmonids and all beach-seined samples measured in the field were examined for indications of hatchery origin, including external marks (i.e., fin clips), coded-wire tags (CWT) or passive integrated transponder (PIT) tags.

Data Analyses—We plotted time series of Chinook and chum *O. nerka* salmon catch per unit effort (CPUE) for each station to examine spatial and temporal variations in abundance. Average CPUE was plotted when the seine was hauled multiple times during a single sampling event. Annual time series of mean salmon FL for each salinity zone were analyzed across the salinity gradient. We examined regressions of mean FL through time to compare trends among years and plotted size-frequency distributions for juvenile salmon at each site to examine size-related variation in habitat use. We also computed the mean monthly percent of salmon fry (individuals < 60 mm) in each

monthly sample, which was plotted as a time series. We used size-at-age to separate subyearling from yearling Chinook salmon (modified from Dawley et al. 1985) using the following regression equation: size (mm) = 0.5 × day of year + 50 (r^2 = 0.99). Finally, we plotted CPUE by temperature and salinity at the time of capture to investigate salmon abundance and size in relation to physical attributes.

We investigated spatiotemporal variability in fish abundance by grouping abundance data by salinity zone (marine, estuarine mixing, tidal freshwater) and by seasonal period. Seasonal periods were determined by river flow and temperature (Figure 11). The winter period (November-February) began during the fall transition to downwelling ocean conditions and was characterized by a decline to low water temperatures (16-4°C) and river flows (4000-5000 m^3 s^{-1}).

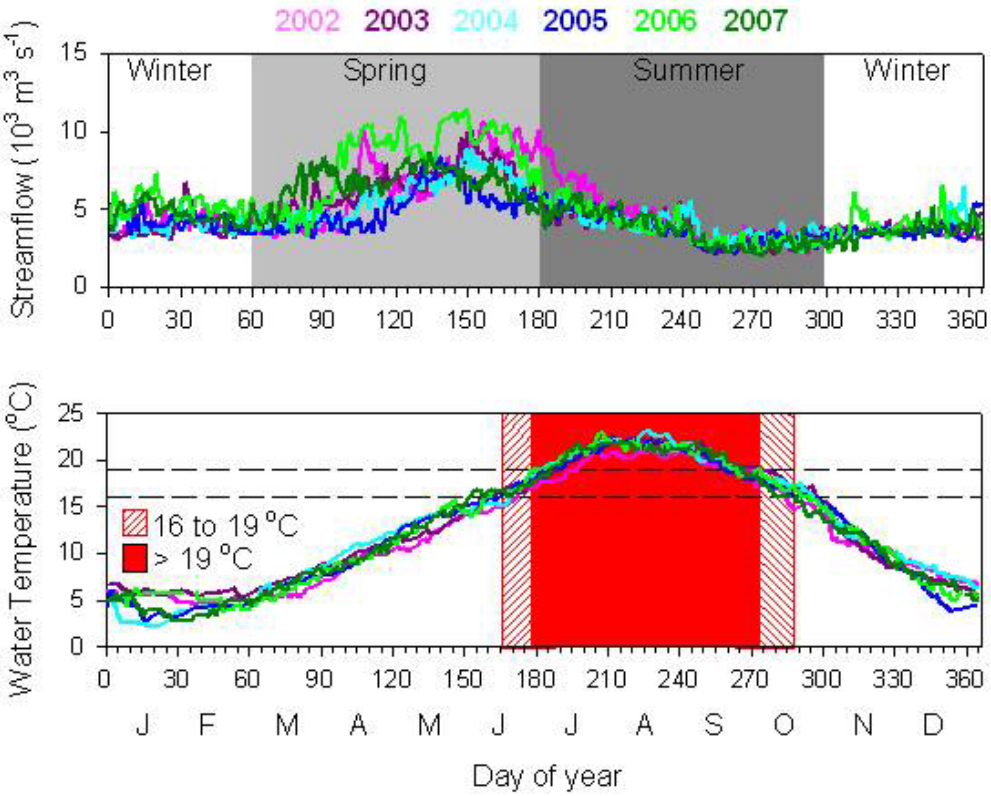


Figure 11. Seasonal and interannual variation in river flow (upper panel) and temperature (lower panel) by day of year at Bonneville Dam 2002-2007. Seasonal divisions are indicated in the upper panel, and temperature regimes in the lower panel. Data courtesy of Columbia River Data Access in Real Time (CRDART 1995).

The spring period (March-June) encompassed the spring transition to upwelling ocean conditions, with increasing river temperatures (4-20°C), and high river flows (including the spring freshet, 4000-12000 m^3 s^{-1}). The summer period (July-October) coincided with coastal upwelling, high river temperature (15-24°C), and declining or low river flow. We calculated proportional species composition and standard diversity indices (Shannon-Weiner and species number) for each zone by season and year.

Results

Fish Assemblages along the Tidal Gradient—Our beach seine catches totaled 273,180 individuals from at least 50 fish species. Seventeen species accounted for over 99% of the total. Of this total, 10 species (20%) were not native to the Columbia River estuary (Appendix C). Introduced banded killifish *Fundulus diaphanous* and American shad *Alosa sapidissima* made up a relatively high proportion of individuals in tidal freshwater environments (Table 2).

At the landscape scale, salinity tolerance was a primary factor determining species presence or absence (Table 2). Abundance was generally greatest in the tidal freshwater zone and lowest in the marine zone, a trend that was driven primarily by the large number of threespine stickleback *Gasterosteus aculeatus*, the dominant species at tidal freshwater sites on all sampling dates. Threespine stickleback comprised 86, 68, and 24% of the total fish population at tidal freshwater, estuarine mixing, and marine sites, respectively.

Table 2. Fish species sampled at landscape zones 2002-2007 (ranked by total abundance). Salmon species designated by shaded cells; non-native species designated by boldface type. Most unidentified fish were juvenile Plueuronectidae.

	Landscape type sampled			Total	
Common name	Marine	Estuarine mixing	Tidal freshwater	n	(%)
Threespine stickleback	10,400	54,659	102,295	167,354	61.37
Surf smelt	22,146	94,35	1	31,582	11.58
Shiner perch	311	29,285	8	29,604	10.86
Chinook salmon	1,487	3,958	7,614	13,059	4.79
English sole	1,541	3,348		4,889	1.79
Starry flounder	292	3,215	1,272	4,779	1.75
Pacific staghorn sculpin	196	4,032	31	4,259	1.56
American shad	84	321	2,867	3,272	1.20
Chum salmon	1,327	1,593	63	2,983	1.09
Peamouth		2	2,504	2,506	0.92
Northern anchovy	202	1,210		1,412	0.52
Pacific sardine	1,378	21		1,399	0.51
Pacific herring	1,179	168		1,347	0.49
Prickly sculpin	2	6	922	930	0.34

Table 2. Continued.

Common name	Landscape type sampled: Marine	Estuarine mixing	Tidal freshwater	Total n	(%)
Sand sole	727	56		783	0.29
Pacific sand lance	672			672	0.25
Banded killifish	11	66	364	441	0.16
Coho salmon	19	65	166	250	0.09
Pacific sanddab	179	41		220	0.08
Largescale sucker			186	186	0.07
Northern pikeminnow			159	159	0.06
Topsmelt	127	19		146	0.05
Yellow perch		2	117	119	0.04
Walleye surfperch	86	1		87	0.03
Pacific tomcod		39		39	0.01
Saddleback gunnel	3	32		35	0.01
Rainbow trout (steelhead)	6	10	17	33	0.01
Cutthroat trout	3	16	5	24	0.01
Bay pipefish	5	10		15	0.01
Largemouth bass	1	3	9	13	0.00
Longfin smelt		10	1	11	0.00
Common carp			9	9	0.00
Snake prickleback	2	7		9	0.00
Black crappie		3	4	7	0.00
Speckled sanddab	2	5		7	0.00
Eulachon		1	5	6	0.00
Sharpnose Sculpin	1	5		6	0.00
Whitebait smelt	3	2		5	0.00
Kelp Greenling	2	1		3	0.00
Redtail surfperch	3			3	0.00
Sand roller			3	3	0.00
Smallmouth bass			3	3	0.00
Sockeye salmon	1		2	3	0.00
Sunfish (Lepomis spp)			4	4	0.00
Golden Shiner			2	2	0.00
Dover sole		1		1	0.00
Pile perch	1			1	0.00
River lamprey	1			1	0.00
Unidentified fish	306	280	30	616	0.23
Total	42,399	111,648	118,635	272,682	100.00

Surf smelt *Hypomesus pretiosus* and shiner perch *Cymatogaster aggregata* dominated the respective communities of the marine and estuarine mixing zones. Juvenile Chinook salmon was the fourth most abundant species (4.8%). The number of species and diversity index were generally lower in tidal freshwater than in estuarine mixing habitats (Figure 12). Seasonal abundance tended to be lowest in winter and highest in summer (Figure 12). Diversity was most variable in the estuarine mixing zone, and oscillated between low winter values and high spring and summer values. Diversity in the tidal freshwater zone was relatively stable, and with one exception, remained below 1.0.

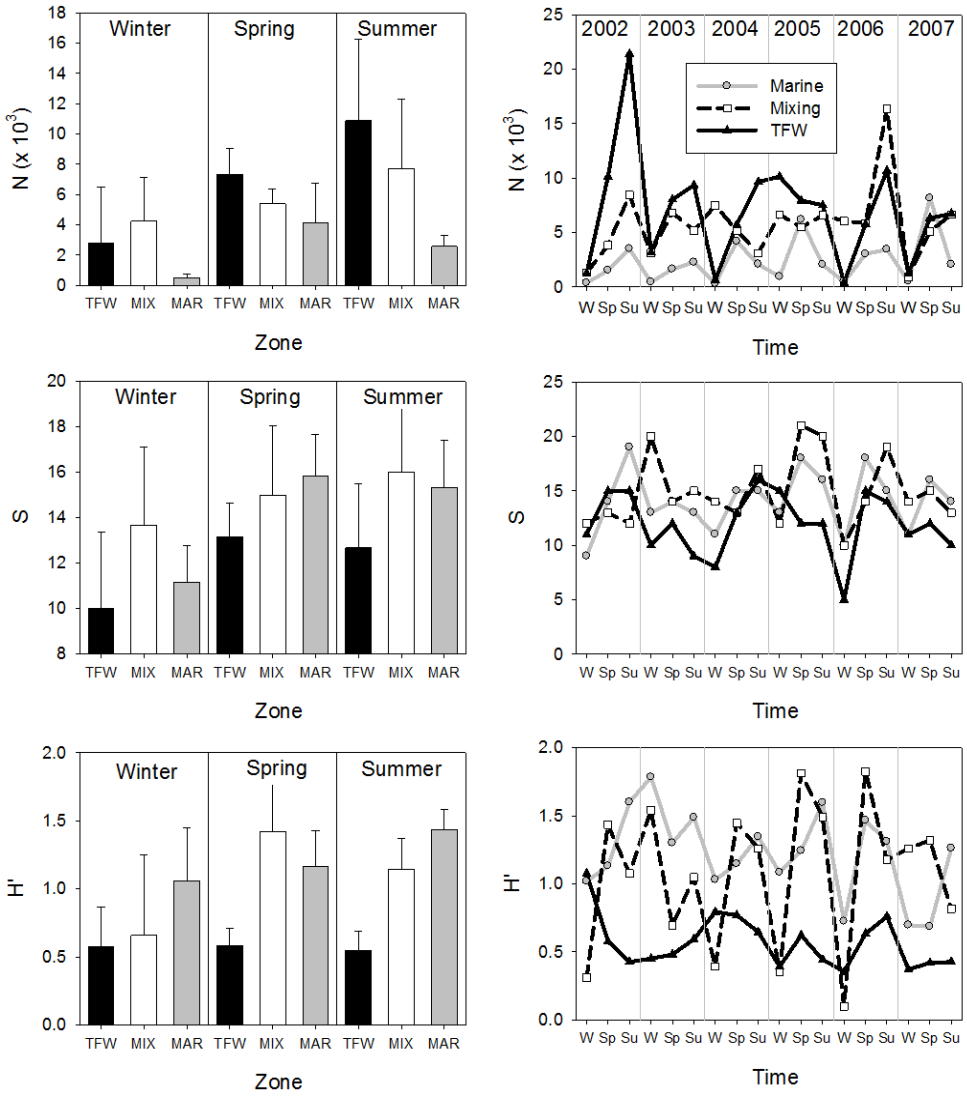


Figure 12. Diversity indices at main beach seine sites 2002-2007. Left panels show mean (+SD) seasonal metrics by zone. Right panels show interannual variation (from top to bottom) in abundance (N in thousands), number of species (S), and Shannon-Weiner diversity index (H').

Salmon Abundance along the Tidal Gradient—At beach-seine sites during 2002-2007, we caught 13,059 Chinook, 2,983 chum, and 250 coho salmon *O. kisutch*, as well as 33 steelhead *O. mykiss*, 24 coastal cutthroat trout *O. clarkii*, and 3 sockeye salmon *O. nerka*. In all years, we found juvenile Chinook salmon in the lower river and estuary during all months of the year (Figure 13). The vast majority of Chinook salmon were subyearlings.

Seasonal patterns of Chinook salmon abundance were relatively consistent: abundance increased steadily from January, reached a peak in late spring and early summer, and declined after July. Chinook salmon often accounted for a relatively high

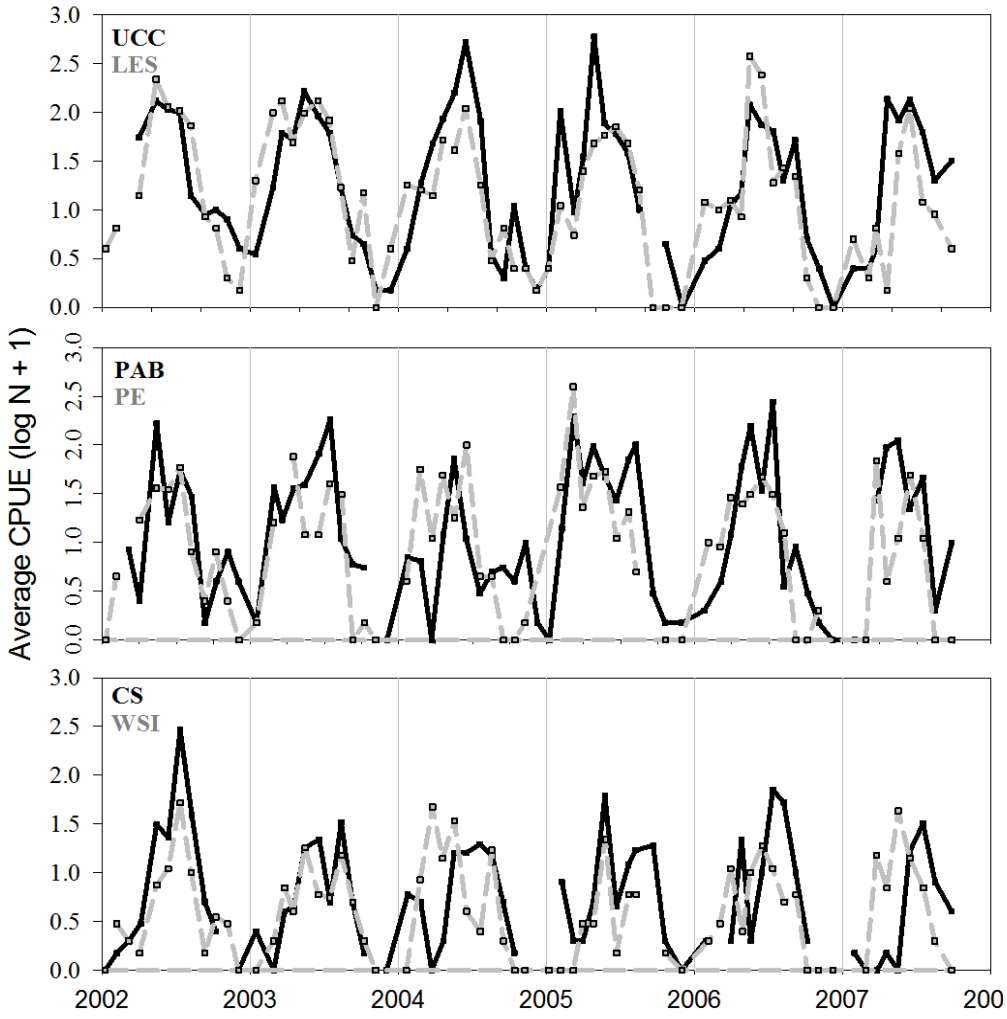


Figure 13. Catch per unit effort for Chinook salmon at tidal freshwater (Upper Clifton Channel and Lower Elochoman Slough), estuarine mixing (Pt. Adams Beach and Pt. Ellice), and marine (Clatsop Spit and West Sand Island) beach-seine stations 2002-2007.

proportion (5-7%) of the total catch. Along the salinity gradient, CPUE at a given date declined longitudinally from the tidal freshwater zone to the estuary mouth. Several factors may have contributed to this pattern, including greater concentration of salmon (and increased sampling efficiencies) in the narrow freshwater sections of the estuary and cumulative losses to the estuary population through mortality and migration

In contrast to Chinook salmon, chum salmon was captured primarily at the four stations closest to the river mouth (1,327 in the marine and 1,640 in the mixing zone, vs. 67 at tidal freshwater sites) and during February-May. Chum salmon abundance usually peaked in April (Figure 14). We encountered few juvenile coho salmon, juvenile steelhead, or yearling Chinook salmon. These fish are thought to migrate swiftly through the system in main-channel environments not sampled by the beach seine.

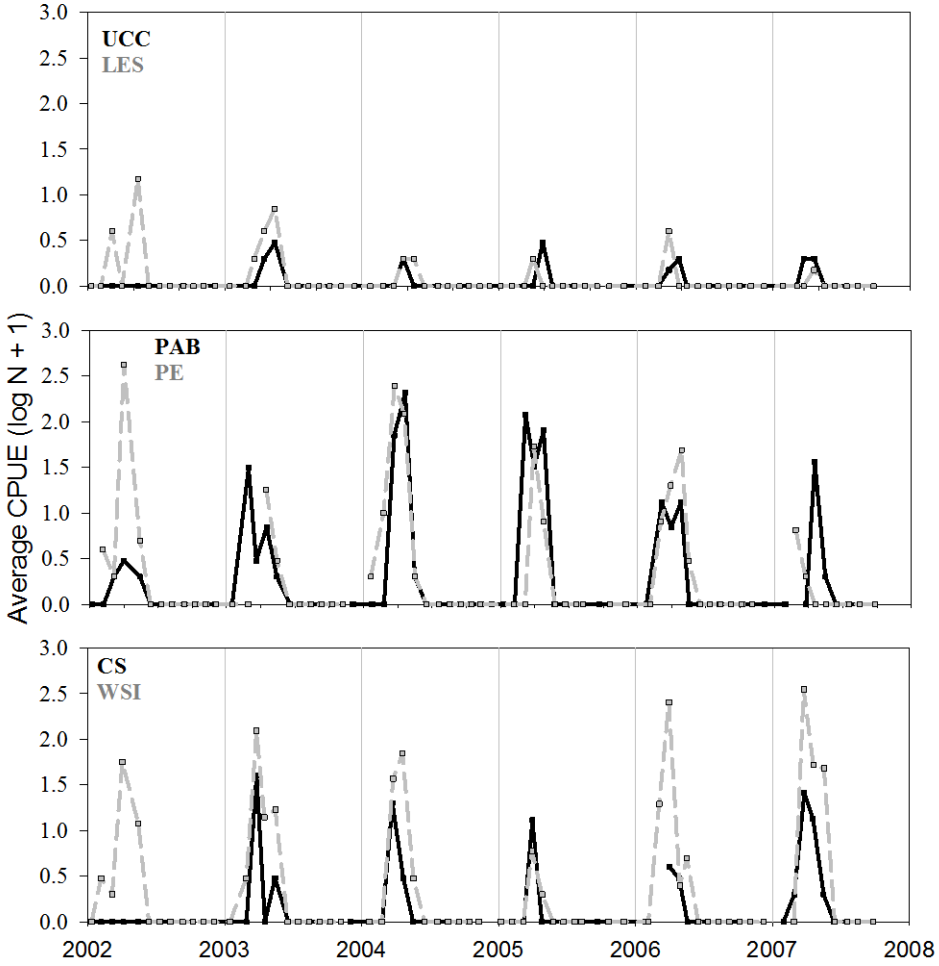


Figure 14. Catch per unit effort of chum salmon at tidal freshwater (Upper Clifton Channel and Lower Elochoman Slough), estuarine mixing (Pt. Adams Beach and Pt. Ellice), and marine (Clatsop Spit and West Sand Island) beach-seine stations 2002-2007.

Size-Related Patterns of Habitat Use—In all years and at all sites, mean FL of Chinook salmon increased rapidly with time (Figure 15). However, within a given period, salmon captured in tidal freshwater habitats were 20-50 mm smaller than those caught further downstream in the estuary. Mean lengths of Chinook salmon often decreased or leveled off from January to April, particularly at tidal freshwater sites. This decrease may have indicated a continued influx of slightly smaller fish during spring and early summer and/or size-selective migration of larger fish to sites further downstream (Figure 15). Mean size increased sharply in March and April, likely a result of hatchery releases of fish larger than 80 mm.

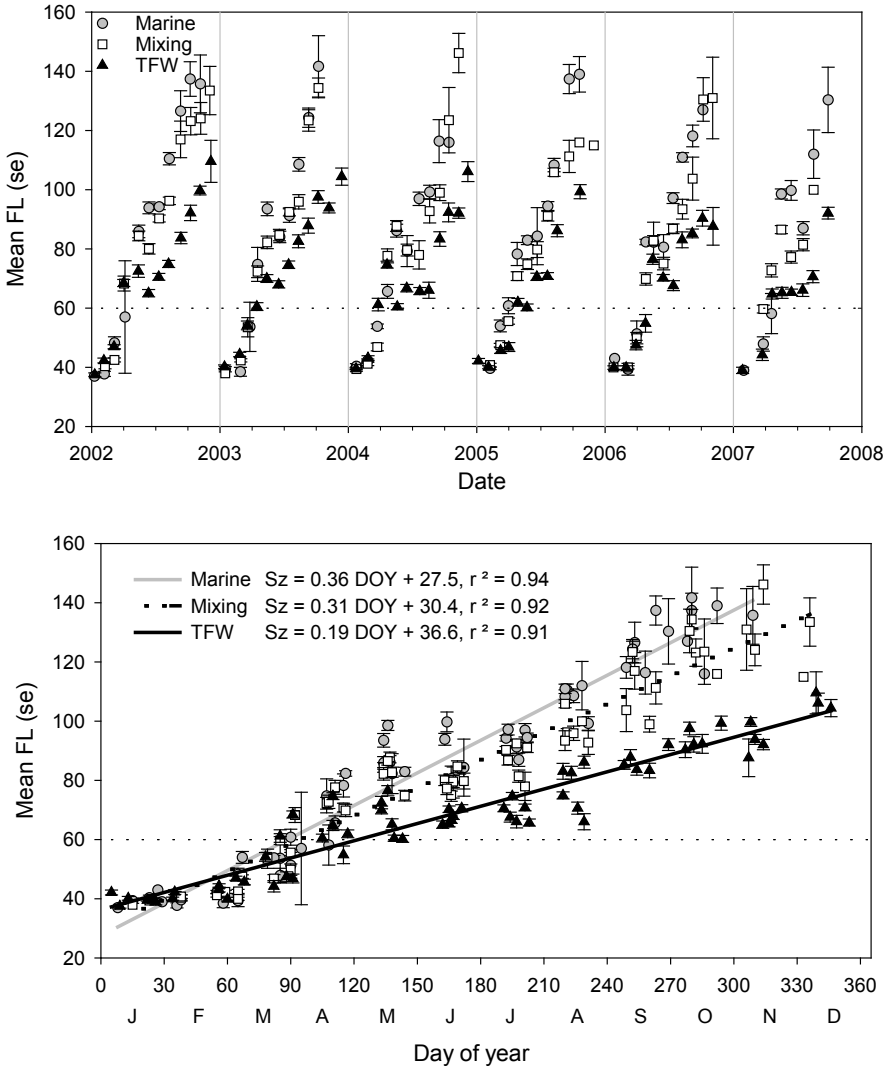


Figure 15. Mean size and standard error for subyearling Chinook salmon at beach-seine sites in tidal freshwater, estuarine mixing, and marine zones for each year (upper graph) and for all years combined (lower graph), 2002-2007. In the lower graph, regression statistics are presented for each zone. Dashed line indicates fry size threshold.

Size-frequency histograms for Chinook salmon show a contrasting pattern between tidal freshwater habitats and the sites located further downstream (Figure 16). In upper Cathlamet Bay and at Lord and Wallace Island, populations had a near-normal distribution, with modes around 70-80 mm. At other beach-seine sites, fish exhibited a bimodal size distribution, with peaks at 40-45 and 90-110 mm. Approximately 30% of all Chinook salmon measured were fry (<60 mm FL; all lower stations plus Lord and Wallace Island mixed sites).

Salmon fry were present in all habitat types, but the relative proportion of fry varied by month and station across all years (Figure 17). Recently emerged fry appeared at all stations simultaneously in early spring and comprised a high percentage of the catch at most stations from January through April. However, fry-sized fish remained a high proportion of the catch at Cathlamet Bay stations as late as August. Together, these abundance and size data supported the premise that small juvenile Chinook salmon reside and accumulate in tidal freshwater sites, while larger fish move further downstream. Nonetheless, even in the marine zone, 13.2% of the Chinook salmon measured across all years were fry-sized animals.

Chum salmon were mostly fry migrants (Figure 18). Most chum salmon were 60 mm or smaller, and 50-70% were smaller than 45 mm, indicating a rapid dispersal to the estuary soon after leaving redds (Figure 19). In each year, mean size increased over time and did not differ significantly between habitat zones as was observed for Chinook salmon. In the Grays River, hatchery chum salmon (52-58 mm) was discernable from naturally produced chum by size range (Roegner et al. 2010). However, fingerling-sized chum salmon were observed at all sites (except Upper Clifton Channel), indicating growth during migration.

Regression statistics for size by day of year suggested a higher average growth rate for chum in the marine zone (Figure 18). Sizes of chum salmon were appreciably larger at West Sand Island than at other sites (up to 90 mm). An unknown proportion of these fish may have originated from the Sea Resources Hatchery in the Chinook River basin or from the Grays River Hatchery further upstream. Although little natural production of chum salmon is thought to occur in Oregon, chum salmon abundance at Oregon sites was relatively high.

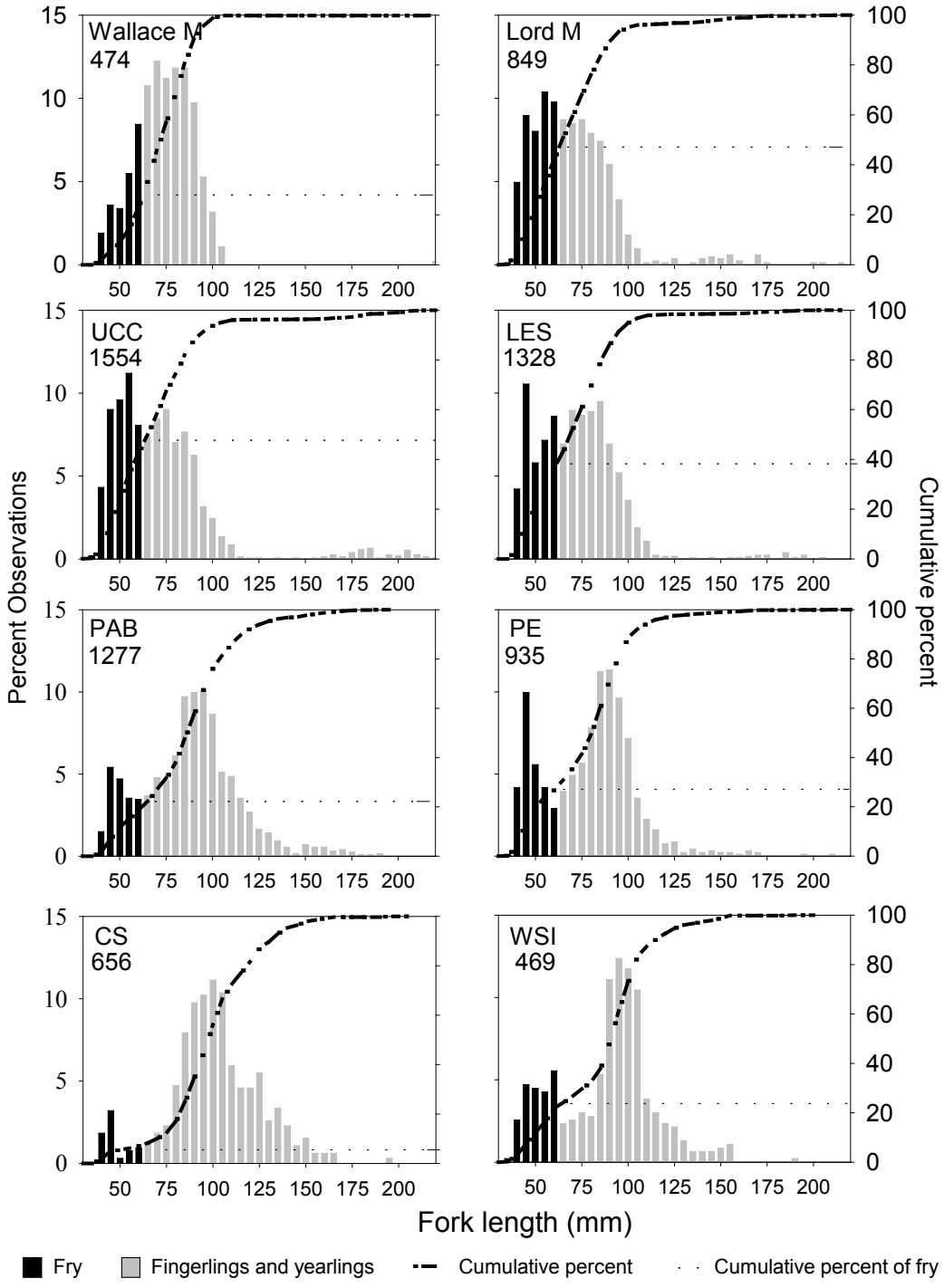


Figure 16. Size-frequency (bars) and cumulative frequency (thick dashed line) plots for Chinook salmon sampled at tidal fresh water (top four graphs), estuarine mixing (Pt. Adams Beach, Pt. Ellice), and marine (Clatsop Spit, West Sand Island) zones. Fry (< 60 mm) are shaded black; thin dashed line designates the total proportion of fry in each collection. The number of fish measured at each site is shown at the upper left of each panel.

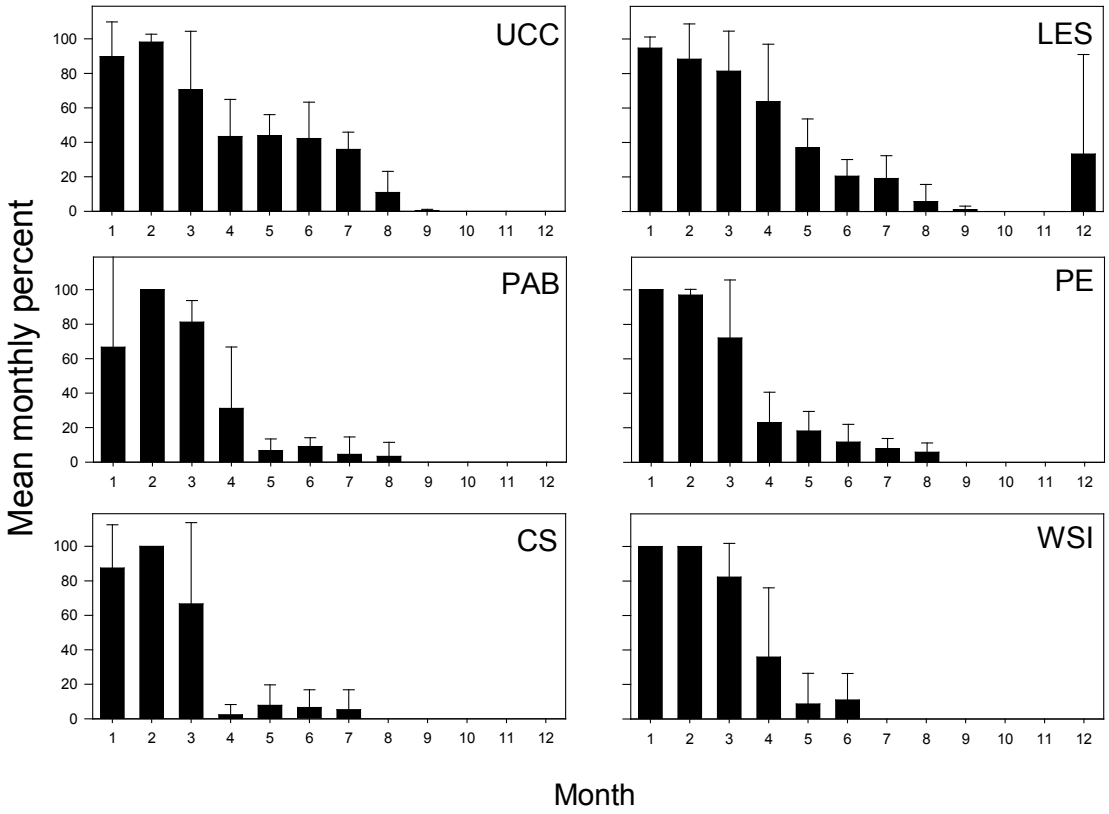


Figure 17. Mean monthly proportion (SD) of fry-sized Chinook salmon at tidal freshwater (Upper Clifton Channel and Lower Elochoman Slough), estuarine mixing (Pt. Adams Beach and Pt. Ellice), and marine (Clatsop Spit and West Sand Island) beach seine stations in the Columbia River estuary, 2002-2007.

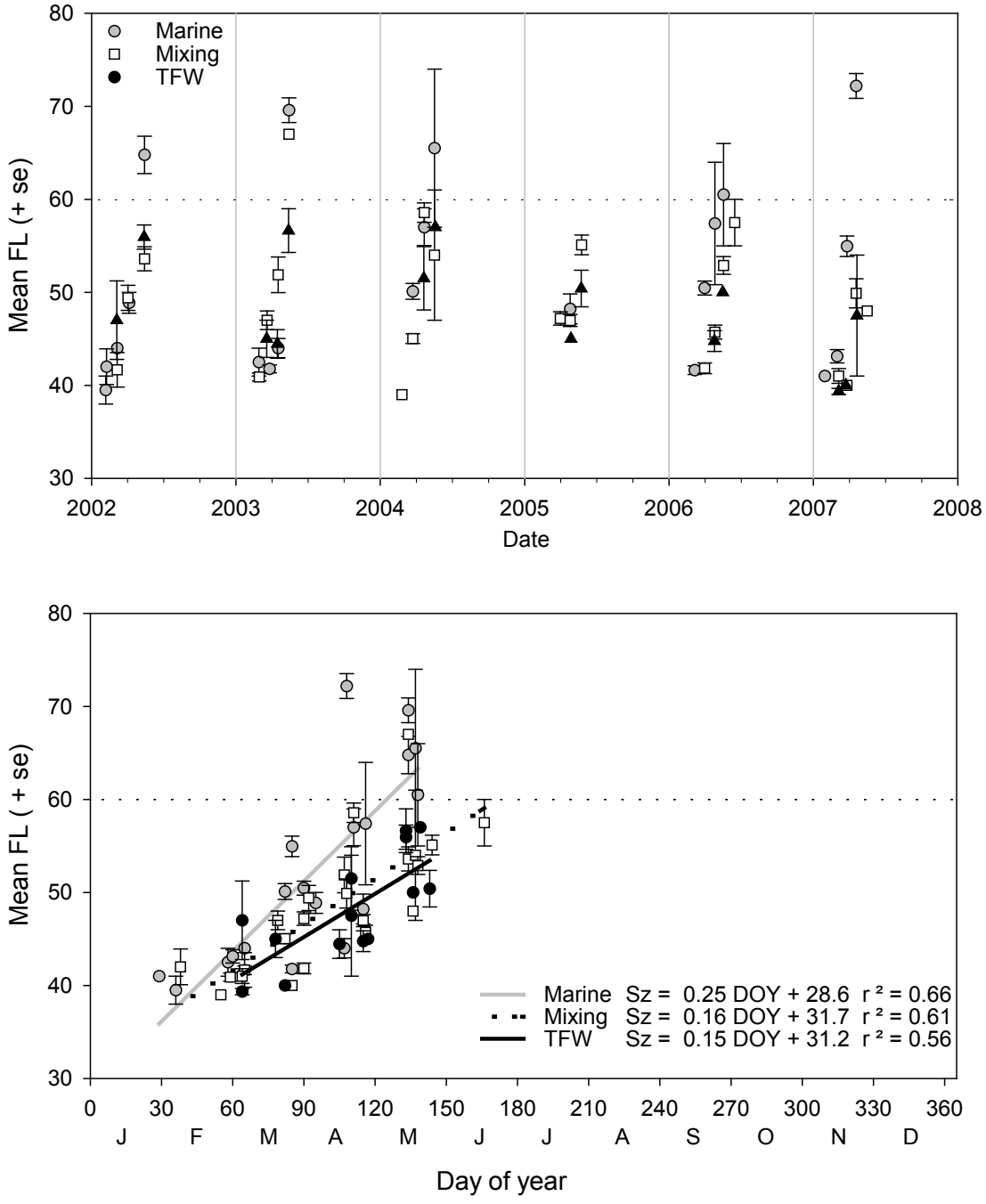


Figure 18. Mean fork length (mm) and standard error for juvenile chum salmon at beach seining sites in tidal freshwater (TFW), estuarine mixing, and marine zones for each year (upper graph) and for all years combined (lower graph), 2002-2007.

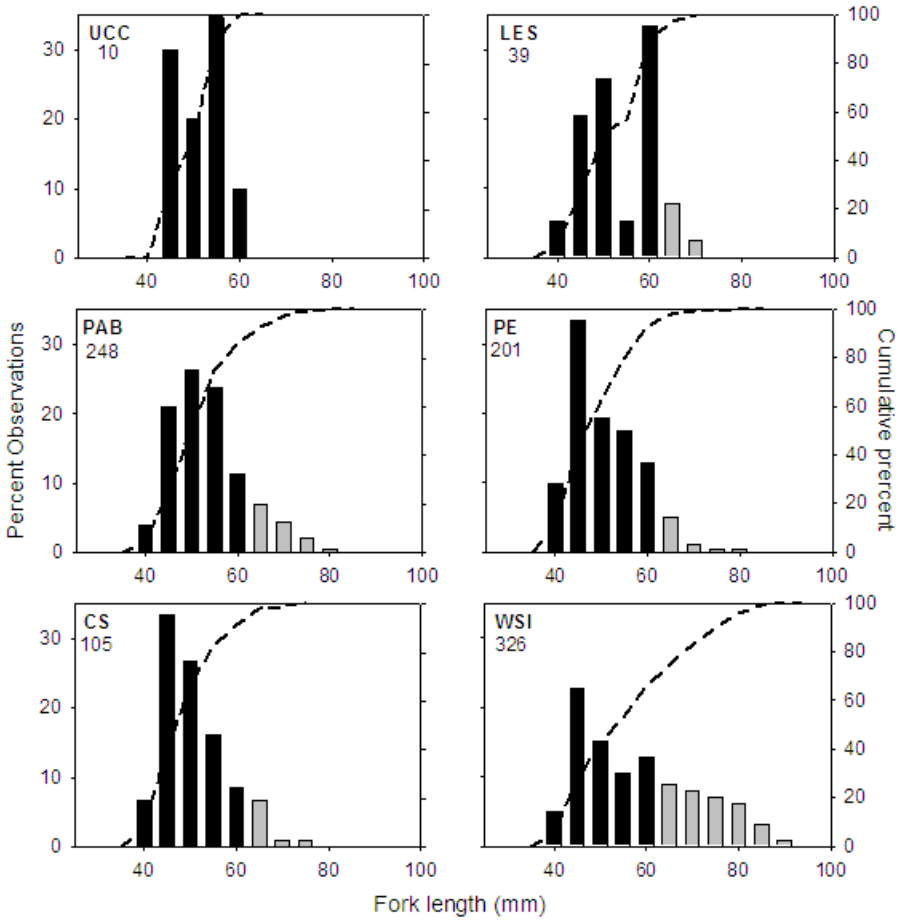


Figure 19. Size frequency (bars) and cumulative frequency (dashed line) plots for chum salmon sampled at estuarine mixing and marine zones. Fry (< 60 mm FL) are shaded black. The number of fish measured at each site is shown at the upper left of each panel.

Proportions of Hatchery Fish along the Tidal Gradient—From 2002 to 2007, 204 of 8,455 Chinook salmon (2.4%) captured at beach-seine sites were tagged with a coded-wire tag (Table 3). We captured only 3 PIT-tagged fish, including one subyearling migrant released from the Snake River 10 km above the confluence with the Clearwater River (Table 4). In beach-seine catches, the percentage of subyearling Chinook salmon with a clipped adipose fin (indicating hatchery origin) increased substantially after 2006 (Table 5). Less than 10% of all subyearlings sampled during 2002-2006 were fin clipped compared with 32% (range 13.2-65.9%) in 2007-2008 (only Pt. Adams Beach and Upper Clifton Channel were sampled in 2008). No corresponding difference was observed in percentages of fin-clipped yearling Chinook salmon between the same periods (71.7 vs. 88.5%).

Table 3. Marked Chinook salmon from main beach seine sites during 2002-2007. Abbreviations: CWT coded wire tag; PIT passive integrated transponder.

Total examined	Adipose fin clip	Coded wire tag	Adipose fin clip & CWT	PIT or fin clip & PIT	Total marked	
					n	(%)
8,455	1,157	52	152	3	1,363	16.1

Table 4. PIT tagged Chinook salmon captured at beach seine sites 2002-2007.

Release			Recapture				Absolute	Specific
Date	Site	Length (mm)	Date	Site	Length (mm)	Travel time (d)	growth rate	growth rate
5/13/2005	BIGC	74	5/24/2005	Clatsop Spit	82	11	0.73	0.93
5/25/2006	SNAKE3	74	6/14/2006	Lower Elochoman Slough	78	20	0.20	0.26
7/9/2007	TOUT	74	7/18/2007	Pt. Ellice	74	9	0.00	0.00

Table 5. Chinook salmon with an adipose fin clip by station, survey period, and life history type. Hatcheries significantly increased the marking rate for subyearling salmon after 2006.

Station	Number caught	Adipose fin clip		Number caught	Adipose fin clip	
		(N)	(%)		(N)	(%)
	Subyearling Chinook salmon					
	2002-2006			2007-2008		
Clatsop Spit	647	65	10.0	55	25	45.5
West Sand Island	433	28	6.5	85	56	65.9
Pt. Adams Beach	1307	133	10.2	520	320	61.5
Pt. Ellice	1043	85	8.2	131	84	64.1
Lower Elochoman Slough	1436	115	8.0	189	99	52.4
Upper Clifton Channel	1664	86	5.2	657	198	30.1
Lord Island—Mixed	796	73	9.2	1802	238	13.2
Wallace Island—Mixed	423	26	6.2	378	205	54.2
Total/mean (+SD)	7749	611	7.9 +1.8	3817	1225	32.1 +19.6
	Yearling Chinook salmon					
	2002-2006			2007-2008		
Clatsop Spit	10	6	60.0	3	2	66.7
West Sand Island	5	4	80.0	3	2	66.7
Pt. Adams Beach	31	21	67.7	0		
Pt. Ellice	25	19	76.0	8	6	75.0
Lower Elochoman Slough	21	16	76.2	1	1	100.0
Upper Clifton Channel	60	39	65.0	10	6	60.0
Lord Island—Mixed	29	24	82.7	48	48	100.0
Wallace Island—Mixed	1	1	100.0	3	3	100.0
Total/mean (+SD)	184	132	76.0 +12.4	78	69	88.5 +18.1

In 2007, about 65% of subyearling Chinook salmon were marked with an adipose fin clip before release (RMPC 1977); this marking rate increased to over 80% in 2008. At this rate, the numbers of marked vs. unmarked fish we sampled were a reasonable index to the size distributions of hatchery vs. naturally produced juveniles in the estuary. For example in 2007, the size distribution of unmarked subyearlings was wide and skewed toward smaller, fry-size fish (Figure 20). In contrast, the size distribution of the marked population was more normal, dominated by fingerling-sized fish (70-90 mm FL), and contained few individuals smaller than 60 mm and none less than 50 mm. In 2008, the relative size distributions for marked and unmarked fish were similar to those in 2007, although we sampled at only a few beach-seine sites in 2008, and therefore our sample size was much smaller.

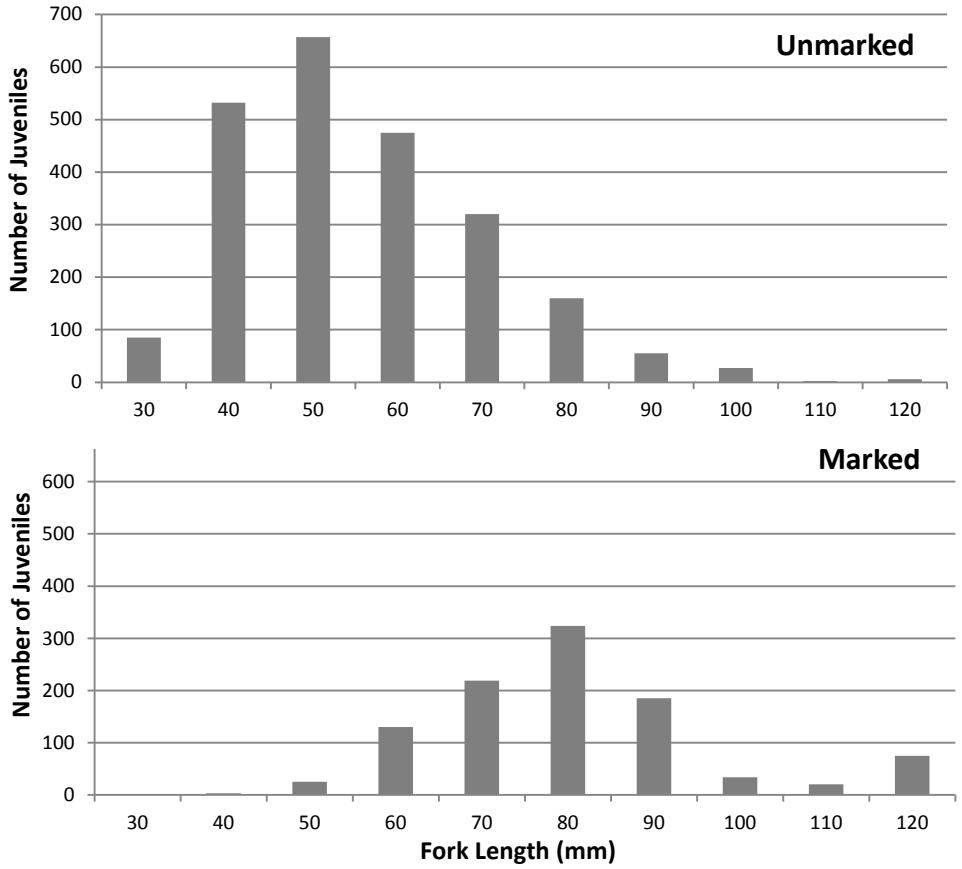


Figure 20. Length distributions of marked (i.e., adipose fin clipped) and unmarked juvenile Chinook salmon sampled with the beach seine in 2007. Measurements are for juveniles collected from each of eight sites (listed in Table 5) distributed from near the estuary mouth to Lord Island at rkm 101.

Salmon and Water Parameters along the Tidal Gradient

Methods

Preceding each beach-seine haul during the tidal gradient surveys of 2003-2006, we profiled the water column with a Sea Bird 19 plus[†] conductivity-temperature-depth (CTD) probe. Four casts were made perpendicular to shore in a transect extending from the beach-seine site (2-5 m depth) out to the channel, 250-300 m from shore. The first two casts were made within the shallow area swept by the beach seine, and the second two were made in deeper channel habitat. For this report, we use salinity and temperature data recorded from the 1.5-m depth within the area sampled by the seine to characterize physical conditions during fish capture. More extensive CTD cast data can be found in Roegner et al. (2008).

Results

Water temperatures at the 1.5-m depth reached levels that were likely stressful to salmonids from late June through October 2003-2006 (Figure 21). Tidal freshwater sites were warmer and unstratified (well mixed) compared to the cooler and vertically stratified sites in the marine and estuarine mixing zones (Figure 10; Roegner et al. 2008). However, substantial numbers of Chinook salmon were sampled during warm conditions. Of the 8,759 Chinook sampled when CTD data were collected (2003-2006), 32.7% occurred where surface-water temperature was above 16°C and 12.3% where surface-water temperature was above 19°C (Table 6). Among the three survey zones, the highest proportion of individuals found at temperatures above 19°C occurred in the estuarine mixing zone (14%). The maximum surface temperature at which we collected Chinook salmon was 24°C. In contrast, chum salmon migration through the estuary was completed before water temperatures reached 16°C.

Table 6. Percentage of salmon caught in beach seine at the 16 and 19°C temperature thresholds.

Zone	Total (N)	>16°C (%)	>19°C (%)
Tidal Fresh	5,208	21.28	11.83
Mixing	2,846	38.95	14.27
Marine	7,05	50.43	7.59
Total	8,759	32.67	12.28

[†] Reference to trade names does not imply endorsement by the National Marine Fisheries Service, NOAA.

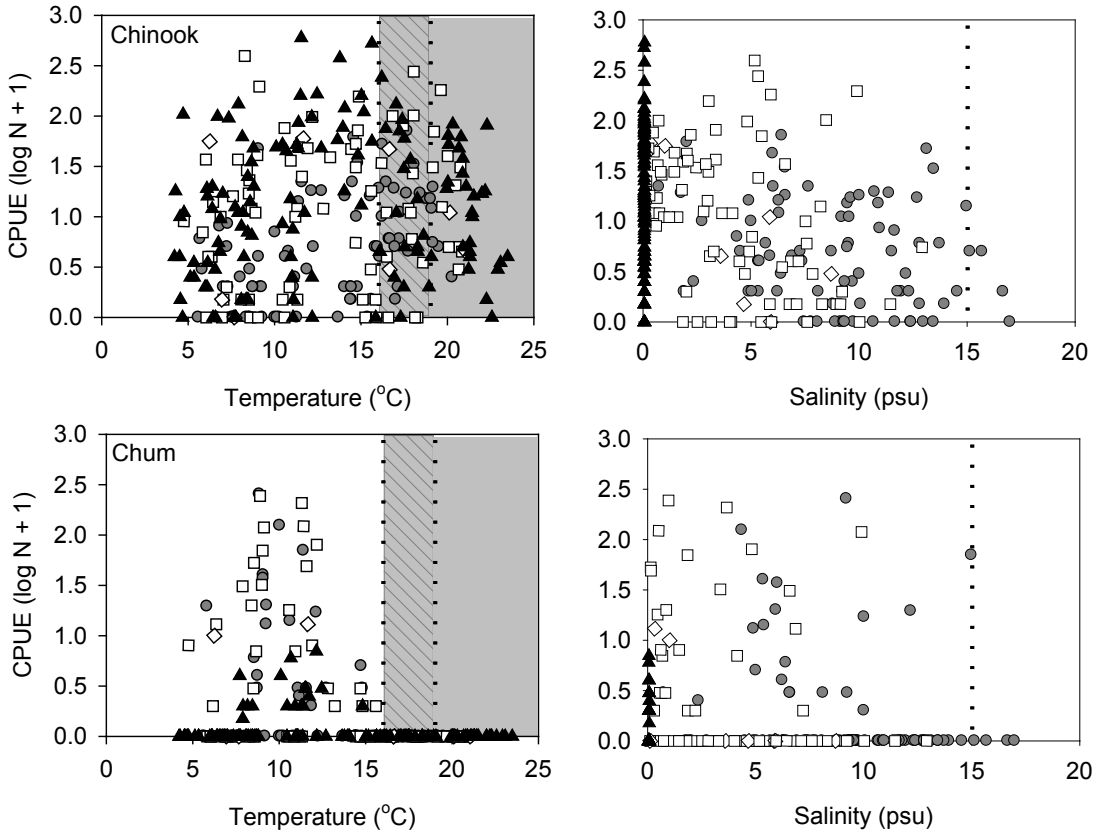


Figure 21. Salmon catch per unit effort as a function of mean surface water temperature and salinity at beach-seine sites in the tidal freshwater (▲), estuarine mixing (□), and marine (●) zones, 2003-2006. Upper row, Chinook salmon; bottom row, chum salmon. Temperatures above 16 and 19°C are shaded; the 15 psu isoosmotic salinity level is also designated.

The rapid decline in Chinook salmon abundance each year after July generally coincided with high surface-water temperatures, particularly at sites upstream from the moderating effects of the Pacific Ocean. These results may indicate that fish leave the estuary or seek deeper, offshore habitats as surface-water temperatures approach stressful levels. Increased mortality after the mid-summer population peak could also be a factor in the observed abundance pattern, but we have no measure of natural mortality trends in the estuary.

All stations in the tidal freshwater zone (Figure 10) were above the upstream limit of salinity intrusion. At brackish stations in the other estuary zones, salinity at the 1.5-m depth ranged from below 1.0 to 16 psu. Salmon were caught throughout this salinity range with no discernable pattern.

Fish Use of Wetland Habitats

Methods

Study Area—At the habitat scale, we sampled juvenile salmon and other fish species within tidally influenced wetland channels at six lower estuary locations between rkm 35 and 101 (Figure 10). All wetland sampling sites were in the tidal freshwater zone of the estuary. Four major wetland habitat types were defined based on vegetative assemblage and channel structure: emergent marsh, scrub-shrub, forested (dominated by evergreens), and a mixed-wetland habitat type, which ranged from emergent marsh/scrub-shrub to forested vegetation. The emergent marsh site at Russian Island was sampled from 2002 to 2008, but collections at other wetland sites shifted every 2-3 years to encompass a wider range of habitat types and locations along the estuary tidal gradient (Table 7).

Table 7. Location, habitat type, and periods of fish sampling (trap net) at wetland survey sites in the Columbia River estuary. Prey resource samples (i.e., fall-out insects and benthos) were collected simultaneously with fish samples at each site except in 2008.

			Years sampled						
Site	rkm	Wetland habitat type	2002	2003	2004	2005	2006	2007	2008
Russian Island—North	35	Emergent marsh	×	×	×	×	×	×	×
Russian Island—South		Emergent marsh	×	×	×	×	×	×	×
Seal Island—North	37	Emergent marsh	×	×					
Seal Island—South		Emergent marsh	×	×					
Karlson Island—Shrub	42	Shrub	×	×	×				
Karlson Island— Forest		Forested	×	×					
Welch Island—North	53	Scrub-shrub			×	×			
Welch Island—South		Scrub-shrub			×	×			
Wallace Island—East	77	Mixed*					×	×	
Wallace Island—West		Mixed*					×	×	
Lord Island—East	101	Mixed*					×	×	×
Lord Island—West		Mixed*					×	×	

* Mixed: transitional marsh from emergent at the entrance to shrub-forested in the interior.

At the Russian and Seal Island emergent marshes, we surveyed replicate secondary channels (north and south) that shared a common entrance channel. The vegetative assemblage at both sites consisted of herbaceous perennials, with Lyngbye's sedge *Carex lyngbyei* dominating in many areas (Elliot 2004). The forested channel at Karlson Island contained vast amounts of large woody debris and a complex vegetative structure, with a mature canopy of Sitka spruce *Picea sitchensis* or other evergreens and shrub and herbaceous underlayers (Lott 2004). In contrast, the scrub-shrub channels at Karlson and Welch Island were lined with deciduous bushes and shrubs and contained smaller pieces of wood. The Wallace and Lord Island channels were fringed by emergent vegetation dominated by reed canary grass *Phalaris arundinacea*, transitioning to a mix of sitka willow *Salix sitchensis*, Pacific willow *S. lasiandra*, and red alder *Alnus rubra* forest (Ramirez 2008).

Whenever possible, we selected wetland survey sites in areas where tidal velocities were low to moderate and channels drained fully at low tide, allowing the use of fixed nets to quantify fish densities. Two exceptions were the west channels at Wallace and Lord Island, where high river flows and a limited tidal range contributed to the retention of significant volumes of water at low tide. During much of the year, fish densities could not be quantified at either west channel site; therefore, we report results only for the smaller east channels at Wallace and Lord Island.

Wetland sampling sites encompassed a broad range of channel sizes and configurations, vegetative assemblages, and tidal and fluvial processes along the tidal gradient from Cathlamet Bay (rkm 35) to Lord Island (rkm 101; Tables 7 and 8). Forested and scrub-shrub channels in the upstream sites generally had much larger drainage areas than those of the emergent wetland channels in Cathlamet Bay (Table 8). On the other hand, the emergent marshes at Russian and Seal Island were more complex and sinuous than the forested swamp habitats further upriver. This complexity was manifested by disproportionately high channel-surface areas, large amounts of edge habitat, and long thalwegs.

Table 8. Physical measurements of wetland channels selected for fish and prey resource studies in the Columbia River estuary. All channel metrics were estimated from digital analysis of aerial photographs in a geographical information system.

Wetland channel site	Drainage area (m^2)	Channel surface area (m^2)	Channel edge length (m)	Channel thalweg length (m)
Russian Island—North	21,159	4,278	1,741	930
Russian Island—South	15,808	4,186	1,452	823
Seal Island—North	14,519	4,320	1,415	771
Karlson Island N (scrub-shrub)	37,495	5,841	1,102	584
Karlson Island (forested)	53,098	6,042	1,649	832
Welch Island—North	2,163	774	304	136
Welch Island—South	44,968	2,436	789	355
Wallace Island—East	138,852	3,070	519	260
Wallace Island—West	179,710	17,558	3,353	1,738
Lord Island—East	129,539	1,795	463	230
Lord Island—West	360,565	79,732	5,326	2,581

Trap Net Surveys—At each wetland channel site, we surveyed fish species abundance and composition with a trap net adapted from the design of Gray et al. (2002). Each trap net consisted of two wings extending from opposite banks of the channel to a centrally located tunnel, which led to a live box (3.2 mm knotless, hexagonal mesh). The tunnel, live box, and a pair of wings were installed at each site during high slack tide. The wings directed all fish into the live box as the tide receded from the study channel.

We sampled each wetland-channel site monthly during spring and summer to coincide with the primary periods of downstream migration and occupation of wetland habitats by juvenile Chinook salmon. Annual sampling usually began in March, but was delayed until April or May in some years because high spring water levels prevented channels from draining fully at low tide. Sampling was discontinued after July because few salmon remained in shallow wetlands in late summer, and high water temperatures during that period increased the risk of handling mortality.

Salmon sampled at each wetland site were identified, enumerated, and retained for laboratory analyses according to the methods described above for beach-seine samples. We also measured the lengths and weights of a maximum of 30 individuals of each non-salmonid species and counted and released the remaining individuals.

We estimated trap-net efficiencies for juvenile Chinook salmon at each wetland location by releasing approximately 25 marked individuals soon after slack high tide into the channel network above each trap net. Efficiencies were measured as river levels and water temperatures permitted, primarily in May and June for at least one of the years each site was sampled. For these estimates we collected juvenile Chinook salmon from nearby habitats with a 3- × 38-m variable-mesh bag seine (10.0- and 6.3-mm wings, 4.8-mm bag). All salmon were marked with a small caudal fin clip and distributed randomly throughout the channel network above each trap net. We estimated trap-net efficiency as the percentage of all marked individuals recaptured in the trap net during the subsequent low tide.

We continuously monitored water levels and temperature in a single wetland channel each at Russian Island and Lord Island, deploying HOBO water-level indicators near the bottom and at the channel entrance near each trap-net location. At all other wetland channels, we monitored water temperature with a Stowaway Tidbit temperature data logger. We did not monitor salinity because all wetland sites for this survey were located in tidal freshwater regions beyond the upper extent of salt-water influence.

Data Analyses—For wetland habitat surveys, we summed trap-net catches for each fish species by site and year. For Chinook salmon only, we adjusted the catch at each site and sampling event to estimate absolute abundance based on the following formula: $C \times 100/E = AC$, where C = total fish catch at a site, E = estimated catch efficiency (%), and AC = adjusted catch.

For time periods when sampling efficiency was not estimated, we applied an average site efficiency to adjust the catch. Finally, we normalized adjusted catch values at each sampling site to catch per unit area based on the estimated total channel area sampled by each trap net (Table 8).

Results

Wetland Fish Assemblages—We sampled fish species for 7 consecutive years at the Russian Island emergent marsh site and for 2-3 years at all other wetland sites during 2002-2007 (Table 9). Including all sites and years, we collected a total of 876,480 freshwater and anadromous fishes representing 22 taxa, including 7 confirmed non-native species (i.e., unidentified taxa not included). Non-native taxa accounted for up to one-third of all species sampled at some wetland sites.

The total number of species sampled in shallow wetland habitats was generally low, ranging 4-6 for most sites and survey dates (Figure 22). Slightly higher species numbers occurred at shrub and mixed wetland sites, where additional non-native taxa occasionally were present.

Threespine stickleback was by far the most abundant species in wetland habitats, accounting for 91-99% of the individuals sampled at emergent marsh, forested, and scrub-shrub sites between rkm 35 and 53 (Table 9). In 7 years of sampling at Russian Island, we estimated that more than 650,000 stickleback were captured from the two small secondary channels sampled by our trap net; this total was 3-4 orders of magnitude larger than that of any other species collected. Relative abundance declined considerably at the Wallace and Lord Island sites upriver, where threespine stickleback accounted for approximately 67 and 49%, respectively, of the entire fish assemblage.

Despite large numbers of threespine stickleback, juvenile Chinook salmon was the second or third most abundant fish species during spring and early summer at all wetland sample sites, regardless of vegetation type or distance upriver (Table 9). Introduced banded killifish also was abundant at emergent marsh (rkm 35-37) and mixed wetland sites (rkm 77-101), whereas peamouth *Mylocheilus caurinus* and prickly sculpin *Cottus asper* were more commonly found at forested and scrub-shrub sites between rkm 42 and 53.

Salmon Abundance in Wetland Habitats—Chinook salmon abundance at most wetland habitat sites followed a characteristic seasonal pattern, increasing after March to a peak in April or May and declining thereafter (Figure 23). However, abundance at Lord Island increased throughout the summer survey period in 2008. Salmon abundance in study channels at Russian Island dropped substantially after the first 2 years of study and remained at low levels thereafter (< 0.002 m^{-2}). Causes for this drop in abundance are unclear, but occasional beach seining revealed greater numbers of fish in adjacent higher-order channels of the Russian Island marsh.

Table 9. Fish species composition and total abundance (N) for each survey year at each wetland site.

EMERGENT

Seal Island

Species	2002	2003	N	%
Threespine stickleback	106,166	53,226	159,392	99.36
Chinook salmon	316	361	677	0.42
Banded killifish*	134	103	237	0.15
Peamouth	9	65	74	0.05
Chum salmon	14	23	37	0.02
American shad*		2	2	<0.01
Coho salmon	3		3	<0.01
Prickly sculpin	1	1	2	<0.01
Total	106,643	53,781	160,424	

Russian Island

Species	2002	2003	2004	2005	2006	2007	2008	N	%
Threespine stickleback	132,834	108,105	118,287	107,554	22,556	42,577	118,759	650,672	99.61
Chinook salmon	256	378	104	147	35	55	41	1016	0.16
Banded killifish*	21	27	245	411	165	67	29	965	0.15
Peamouth	11	332	9	1	3	8	1	365	0.06
Chum salmon	52	32	21	11	15	16	20	167	0.03
American shad*	11	4	2	1		2		20	<0.01
Coho salmon		6		1	1	4		12	<0.01
Pacific staghorn sculpin		3			1		1	5	<0.01
Prickly sculpin		1				1		2	<0.01
Rainbow trout (steelhead)						2		2	<0.01
Yellow perch*					2			2	<0.01
Black crappie*		1						1	<0.01
Common carp*					1			1	<0.01
Starry flounder	1							1	<0.01
Unidentified lamprey	1							1	<0.01
Total	133,187	108,889	118,668	108,126	22,779	42,732	118,851	653,232	

FORESTED

Karlson Island

Species (common name)	2002	2003	N	%
Threespine stickleback	6,762	919	7,681	97.39
Chinook salmon	76	56	132	1.67
Peamouth	13	15	28	0.36
Unidentified sculpin	17	4	21	0.27
Coho salmon	3	5	8	0.10
Prickly sculpin	4	2	6	0.08
Banded killifish*	3		3	0.04
Chum salmon	3		3	0.04
Largemouth bass*		3	3	0.04
Largescale sucker	1		1	0.01
Starry flounder	1		1	0.01
Grand Total	6,883	1,004	7,887	

Table 9. Continued.

SHRUB

	Karlson Island					Welch Island North				Welch Island South			
	2002	2003	2004	N	%	2004	2005	N	%	2004	2005	N	%
Threespine stickleback	8,580	4,738	3,007	16,325	95.50	461	4,589	5,050	91.90	5,267	15,855	21,122	95.70
Prickly sculpin	122	121	53	296	1.73	26	23	49	0.89	65	29	94	0.43
Chinook salmon	152	41	31	224	1.31	89	197	286	5.21	122	263	385	1.74
Peamouth	48	29	6	83	0.49	2	2	4	0.07	6	9	15	0.07
Unidentified sculpin	42	22	1	65	0.38	35	20	55	1.00	25	60	85	0.39
Unidentified fish			17	17	0.10	9		9	0.16	46		46	0.21
Banded killifish*	2	1	13	16	0.09	8	15	23	0.42	12	300	312	1.41
Chum salmon	10	1	2	13	0.08	18		18	0.33	4	3	7	0.03
Largescale sucker	9	2	2	13	0.08					1		1	<0.01
Unidentified Salmonid	12			12	0.07								
Coho salmon	8	3		11	0.06	1		1	0.02	1		1	<0.01
Starry flounder	8	1	1	10	0.06						1	1	<0.01
Rainbow trout (steelhead)	3			3	0.02								
American shad*	1	1		2	0.01								
Sunfish*	2			2	0.07								
Cutthroat trout	1			1	<0.01								
Pacific lamprey			1	1	<0.01								
Sockeye salmon			1	1	<0.01								
Black crappie*											1	1	<0.01
Total	9,000	4,960	3,135	17,095		649	4,846	5,495		5,549	16,521	22,070	

MIXED

	Wallace Island Upper Channel				Lord Island Upper Channel				
Species (common name)	2006	2007	N	%	2006	2007	2008	N	%
Threespine stickleback	2,919	705	3,624	67.47	1,176	1,178	56	2,410	49.12
Chinook salmon	469	487	956	17.80	372	445	532	1,349	27.50
Banded killifish*	676	9	685	12.75	562	341	61	964	19.65
Coho salmon	1	11	12	0.22	2	65	2	69	1.41
Chum salmon	10	46	56	1.04	5	22	9	36	0.73
Unidentified sculpin	2	2	4	0.07	6	19	3	28	0.57
Prickly sculpin	13		13	0.24	10	9	6	25	0.51
Unidentified crappie*	2		2	0.04	11			11	0.22
Common carp*					6			6	0.12
Peamouth	11	1	12	0.22	2			2	0.04
Unidentified fish						1	1	2	0.04
Black crappie*						1		1	0.02
Cutthroat trout						1		1	0.02
Largescale sucker	5	1	6	0.11			1	1	0.02
Unidentified centrarchid*						1		1	0.02
Northern pikeminnow	1		1	0.02					
Total	4,109	1,262	5,371		2,152	2,083	671	4,906	

* denotes species introduced to the Columbia River.

Chinook salmon densities in wetland habitats generally increased with distance upriver, reaching values of 0.05-0.10 fish m^{-2} or greater at sites above rkm 50 (i.e., Welch, Wallace, and Lord Islands; Figure 23). Similar peak densities (0.01-0.16 fish m^{-2}) were reported for Chinook salmon in emergent wetland channels of the Salmon River estuary on the central Oregon Coast (Hering 2009).

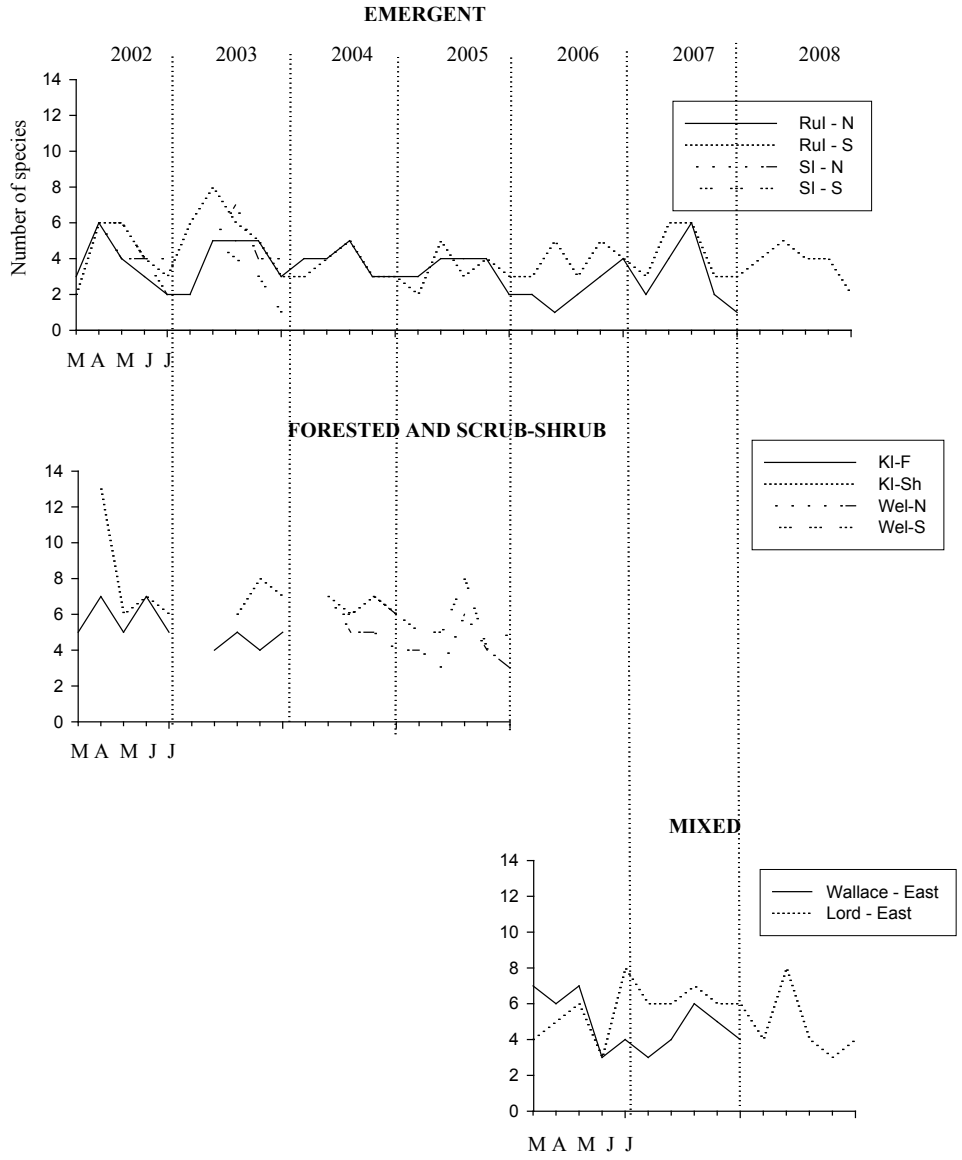


Figure 22. Total number of species for each survey date at each wetland site. Site abbreviations are shown in Figure 10. Data for Wallace and Lord Island are from the east channel site only.

Chum salmon was also found at most wetland sites (Table 9), although numbers were quite variable among years and abundance usually dropped sharply after a brief spring peak. Coho salmon occurred infrequently in wetland channels, and total abundance remained relatively low at all sites except for the mixed wetland site at Lord Island in 2007 (Table 9).

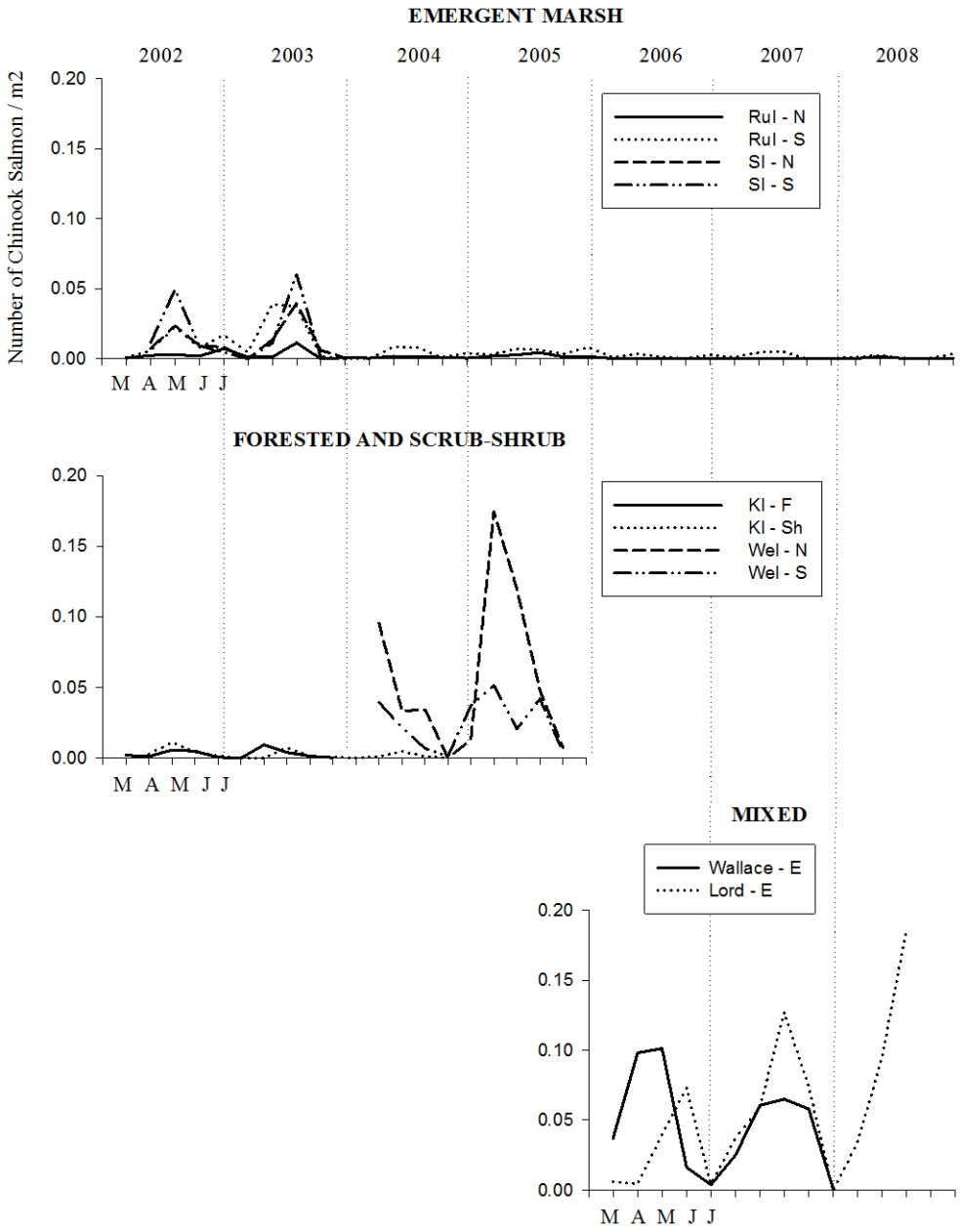


Figure 23. Estimated abundance of Chinook salmon per m^2 of channel habitat for spring and summer survey periods at each wetland survey site. Site abbreviations are shown in Figure 10. Data for Wallace Island and Lord Island are from their respective east channel sites only.

Size-Related Wetland Habitat Use—Mean fork length increased throughout the rearing season for juvenile Chinook salmon at all wetland survey sites (Figure 24). Recently emerged Chinook salmon fry (< 40 mm FL) appeared in most wetland channels in March. The presence of small size classes suggested that fry continue to enter most wetlands until June in most years. In 2008, fry entered the Russian and Lord Island wetlands throughout the sampling season until at least July.

Fish captured at wetland sites in Cathlamet Bay (Russian Island, and to a lesser extent, Karlson Island) exhibited a bimodal size distribution (Figure 25). This bimodal pattern was similar to that observed in catch from the lower-estuary beach-seine sites of the tidal gradient study (Figure 16). However, in samples from Cathlamet Bay wetlands, the secondary peak in the distribution was appreciably smaller—approximately 80 mm—than the 90-110 mm peak from lower estuary sites. Fish from tidal freshwater wetlands above Cathlamet Bay (e.g., Welch, Wallace, and Lord Island) had a near-normal size distribution, with a single peak around 40-45 mm.

Mean sizes of salmon were similar among years within each wetland habitat type, but size classes in the upper range were generally smaller in fish from upriver wetland sites. For example, by the end of the rearing season, maximum mean lengths exceeded 90 mm at Russian Island. In contrast, few juveniles larger than 70-75 mm entered or remained in the wetland channels at Wallace or Lord Island (Figures 24 and 25). Among all salmon collected, the total proportion of fry generally increased with distance upriver: respective proportions were 53 and 61% in the emergent marsh sites at Russian and Seal Island, 77 and 71% in the forested and scrub-shrub sites at Karlson Island, and over 93% in the scrub-shrub or mixed forested sites at Welch, Wallace, and Lord Island.

Over all survey years, the apparent size distribution of juvenile Chinook salmon varied with relative degree of exposure (i.e., depth, water velocity, and amount of cover) in the three habitat types sampled at Wallace and Lord Islands (Figure 26). At each island, sizes were largest near the exposed main-stem channel; large or slightly smaller on the protected back shore; and smallest within shallow, blind wetland channels that extended into each island's interior. The total proportion of fry in each sample decreased along the same habitat gradient at both islands, ranging from 38-67% along the main-stem side and 75-76% along the back side to greater than 93% in the interior wetland channels (Figure 26). The general shape of the size distribution was similar among similar habitats at each island, with a relatively broad range of sizes along the main stem and a truncated distribution that peaked near 40-45 mm within the interior wetland channels. It is possible that the somewhat finer mesh of the gear used to sample interior wetlands led to increased capture efficiency for emergent fry (≤ 40 mm) in interior wetland habitats relative to those in main-stem and back-side habitats.

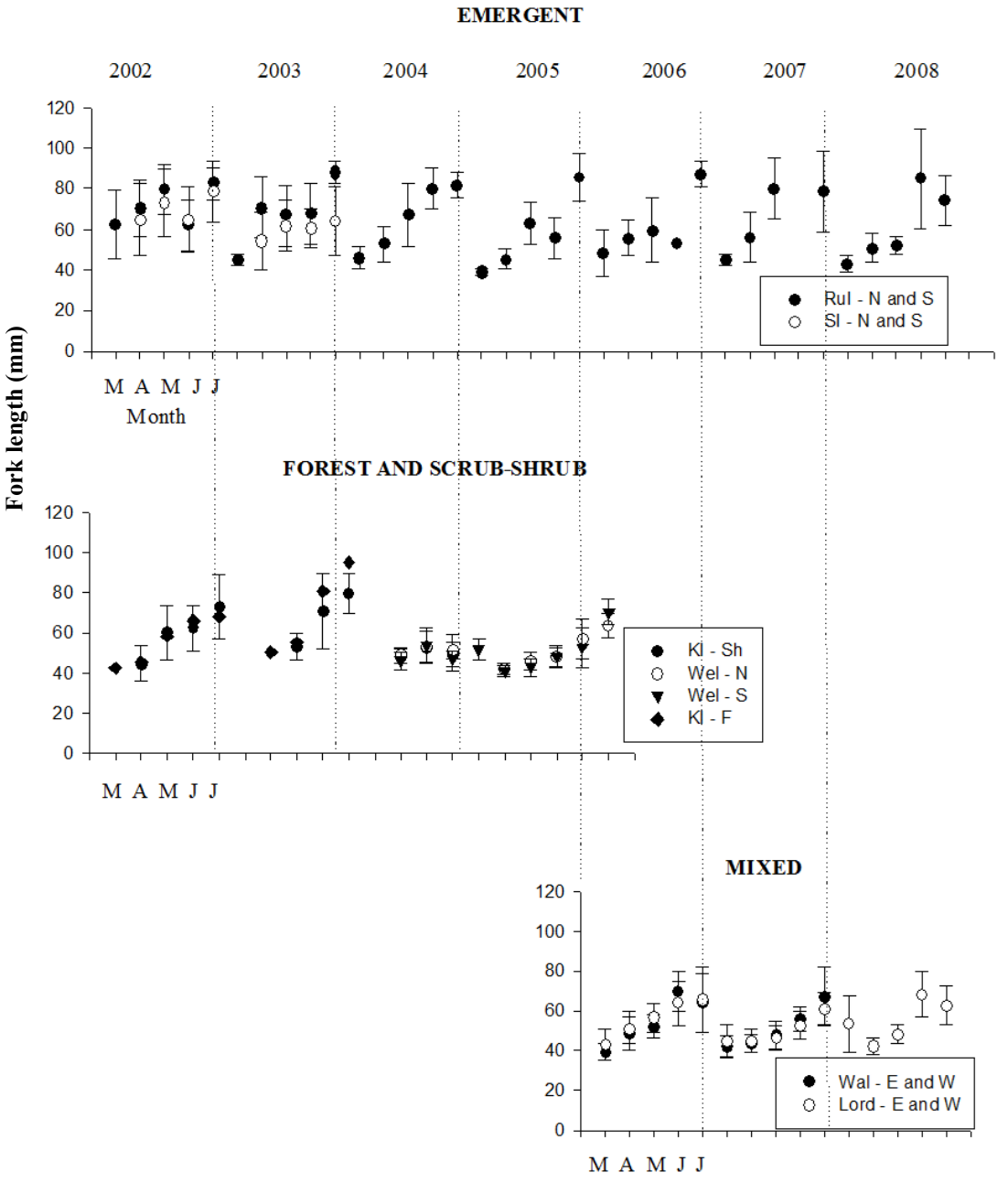


Figure 24. Mean FL (mm) and standard deviation of juvenile Chinook salmon for spring and summer survey periods at each wetland site. Site abbreviations are shown in Figure 10.

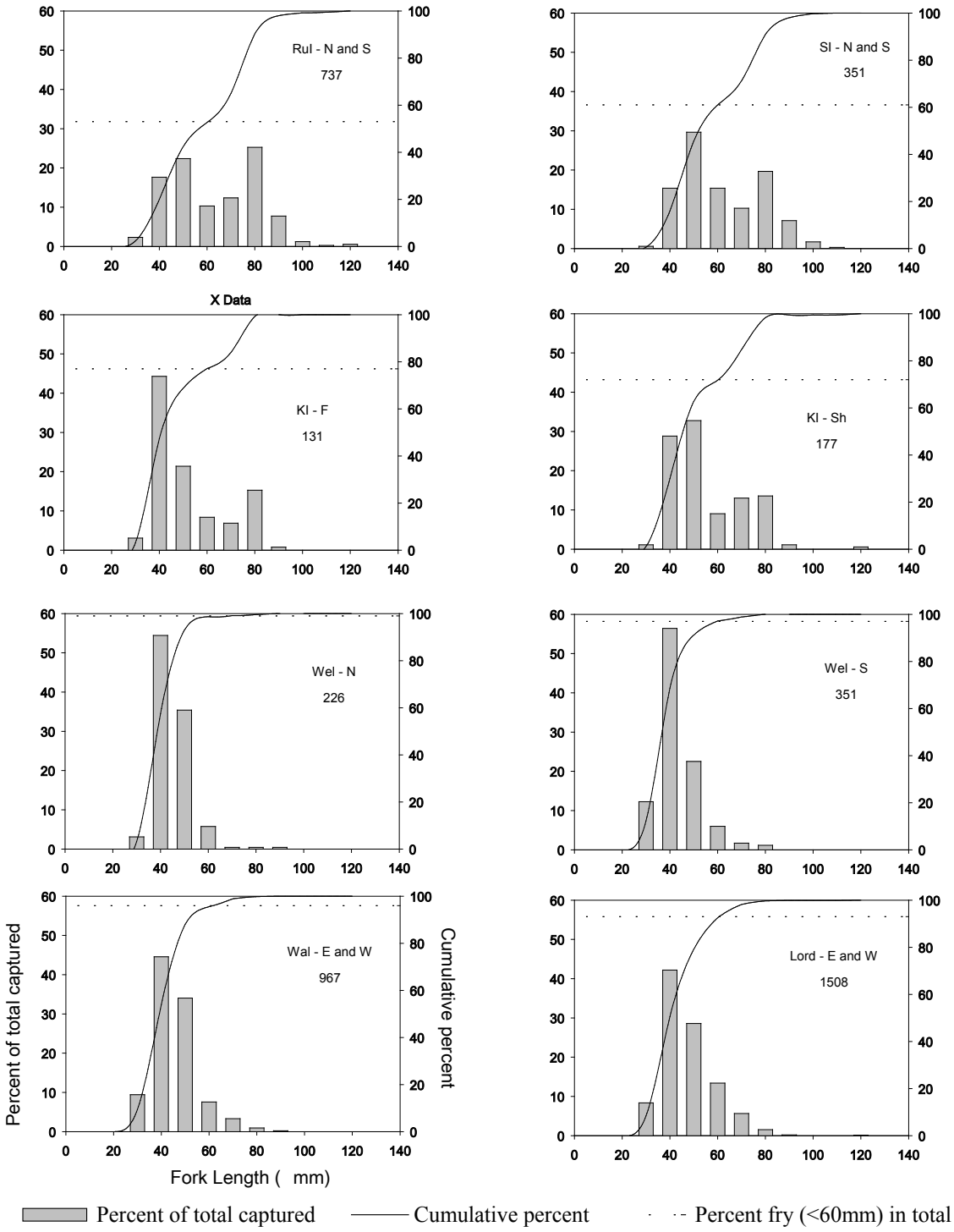


Figure 25. Size-frequency (bars) and cumulative frequency (solid line) plots for Chinook salmon surveys at each wetland channel. Survey periods for each site are listed in Table 7. Dashed lines are labeled with the total proportion of fry (< 60 mm FL) in each sample. Number of fish measured is shown at upper right on each graph. Site abbreviations shown in Figure 10.

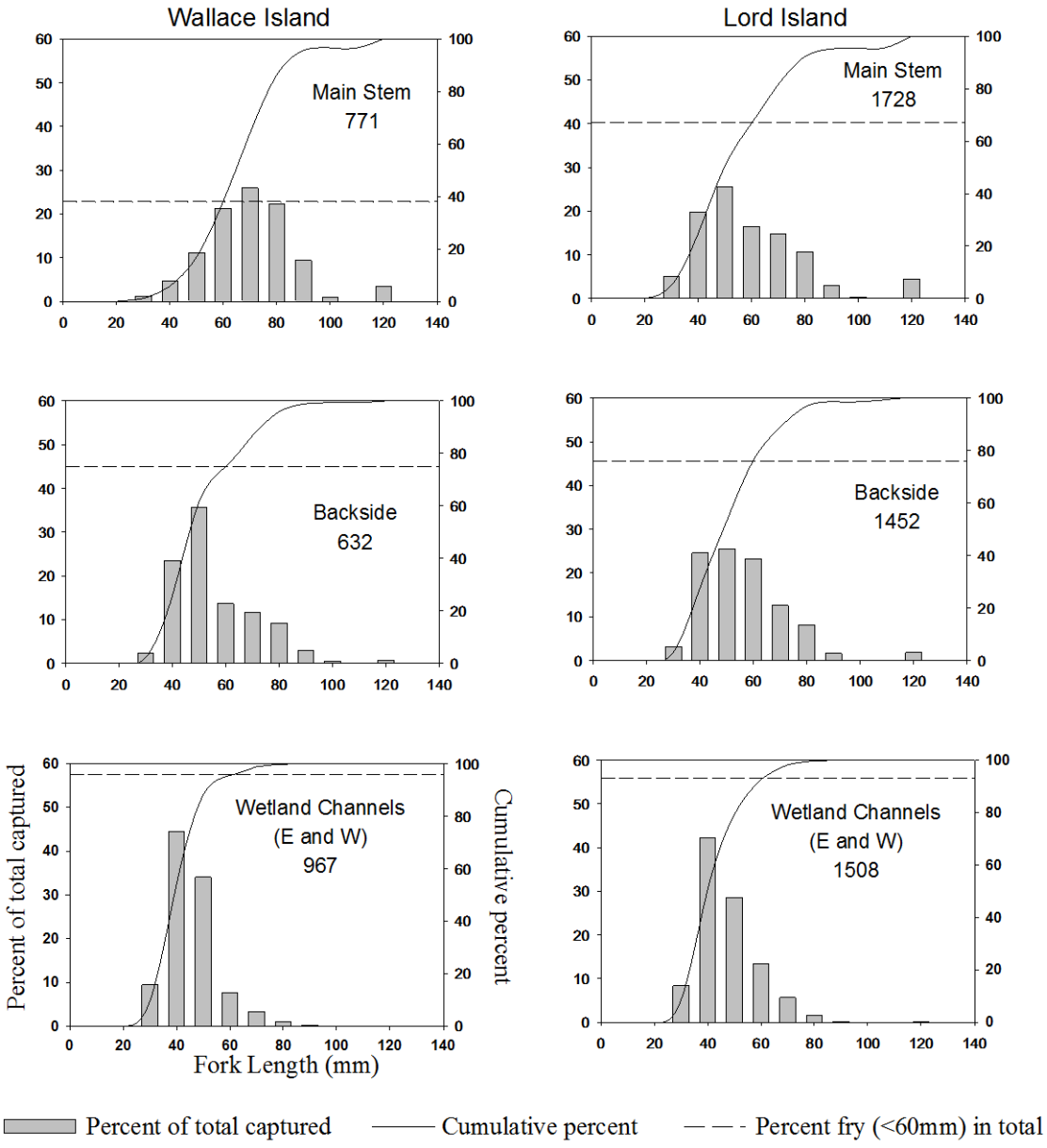


Figure 26. Plots showing length-frequency (bars) and cumulative frequency (solid line) for juvenile Chinook salmon over all survey dates combined at Wallace and Lord Islands. Survey periods for each channel are listed in Table 7. At each island, salmon fork lengths are compared along a decreasing habitat exposure gradient represented by each of three habitat types: main-stem, shoreward or back-side channel, and interior wetland channel. Dashed lines are labeled with the total proportion of fry (< 60 mm) in each sample. Number of fish measured is shown at upper right on each graph.

Proportions of Hatchery Fish in Wetland Habitats—Of the 5,273 Chinook salmon collected in wetland channels from 2002 to 2008, we counted only 148 (2.8%) with identifiable hatchery fin clips (Table 10). During the first 4 years of study, only 13 fin-clipped individuals (<1%) were identified among all salmon sampled at wetland trap-net sites.

The total proportion of marked fish at the Russian Island emergent-wetland site increased from less than 1% in 2002-2005 to 23, 51, and 32%, respectively during 2006, 2007, and 2008 (Table 10). In contrast, proportions of marked fish at the mixed-wetland sites at Wallace and Lord islands remained low (1-2%) in 2006 and 2007, increasing to 7% at Lord Island in 2008. With the exception of Russian Island after 2005, the proportions of marked hatchery fish observed were much lower at wetland (Table 10) than tidal gradient (beach seine) sites (Table 5).

The increased proportions of marked hatchery fish at wetland (Table 10) and beach-seine sites (Table 5) in recent years likely reflected a congressional mandate to increase the proportion of marked fish in Columbia River hatchery releases. Excluding releases of interior spring Chinook stocks, the proportion of fin-clipped subyearlings released from hatcheries increased from approximately 11-14% in 2002-2004 to 37.5% in 2005 and 63% in 2006 (Appendix Table G). For subyearling Chinook salmon, the hatchery marking rate increased to about 65 and 80% in 2007 and 2008, respectively (RMPC 1977).

Table 10. Total numbers of juvenile Chinook salmon captured at each wetland site with/without (first number/second number) identifiable hatchery fin clips, 2003-2007.

Site	2002	2003	2004	2005	2006	2007	2008
Emergent Marsh							
Russian*	3/256	0/196	0/103	0/126	8/35	28/55	13/41
Seal*	3/255	0/208					
Forested & Shrub							
Karlson Forested	2/75	0/34					
Karlson Shrub	2/107	0/41	0/31				
Welch North			0/67	3/137			
Welch South			0/85	0/249			
Mixed							
Wallace*					10/679	11/674	
Lord*					14/562	12/725	39/532

* Data for replicate channels combined

The relatively low proportions of marked hatchery fish at wetland channel sites could reflect the tendency of smaller, unmarked wild fish to select these shallow protected habitats. This interpretation was supported by the consistently higher proportion of marked hatchery fish observed in samples from along the main-stem channel relative to those from the interior wetland channels of Wallace and Lord Islands (Table 11). The percentage of marked hatchery fish often decreased along the habitat-exposure gradient from main-stem shore to back-side shore to interior wetland channel.

Table 11. The total number of juvenile Chinook salmon captured with/without (first number/second number) identifiable hatchery fin clips and the total percentage of all marked salmon (in parentheses) in main-stem, back-side, and interior wetland-channel habitats of Wallace and Lord Islands, 2006-2008.

	2006		2007		2008	
Site	With/without fin clip	Total marked (%)	With/without fin clip	Total marked (%)	With/without fin clip	Total marked (%)
Wallace Island						
Main stem	91/417	21.8	230/408	56.4		
Back-side	9/270	3.3	54/556	9.7		
Channel	10/679	1.5	11/674	16.3		
Lord Island						
Main stem	87/1,012	8.6	132/1,129	11.7	119/536	22.2
Back-side	45/541	8.3	73/702	10.4	81/737	11.0
Channel	14/562	2.5	12/725	1.7	39/532	7.3

Seasonal Availability of Wetland Habitats

Methods

During the March-August or September sampling periods in 2007 and 2008, we estimated habitat opportunity for juvenile Chinook salmon at Russian and Lord Island, the lower- and uppermost wetland monitoring sites, respectively. Habitat opportunity was defined as the percentage of time during each survey month when habitat was classified as available, not available, or marginal. For the purposes of this analysis, we considered wetland channel habitat to be available if both of the following criteria were met:

1. Water depth of at least 0.50 m, the minimum depth at which juvenile salmon entered and exited wetland channels (see Part IV of this report), and
2. Water temperatures ≤ 19°C, the temperature threshold for wetland use by most juvenile Chinook salmon (Bottom et al. 2008; Roegner et al. 2008).

When water levels were below the minimum depth threshold, we classified the habitat as not available. If water depth was satisfactory but temperatures were >19°C (i.e., less than optimal but below lethal levels), we classified habitat opportunity as marginal. Estimated percentages were calculated for each of these habitat-opportunity categories in each survey month based on continuous water level and temperature measurements in Russian and Lord Island wetland channels during 2007 and 2008.

Results

Shallow wetland habitat opportunity for juvenile Chinook salmon was influenced by tide and river flow, factors that determined variations in water level and temperature. Yearly flow conditions measured at Bonneville Dam varied seasonally, with peaks during spring and early summer and rapid declines during fall and winter (Figure 27). Flows exhibited relatively sharp peaks in spring 2006 and 2008, but freshet flows were dampened in 2005 and 2007. Main-stem water temperatures at Bonneville Dam also exhibited considerable seasonality, but lagged behind flows, with peak temperatures exceeding 20°C for extended periods during summer and early fall (Figure 27).

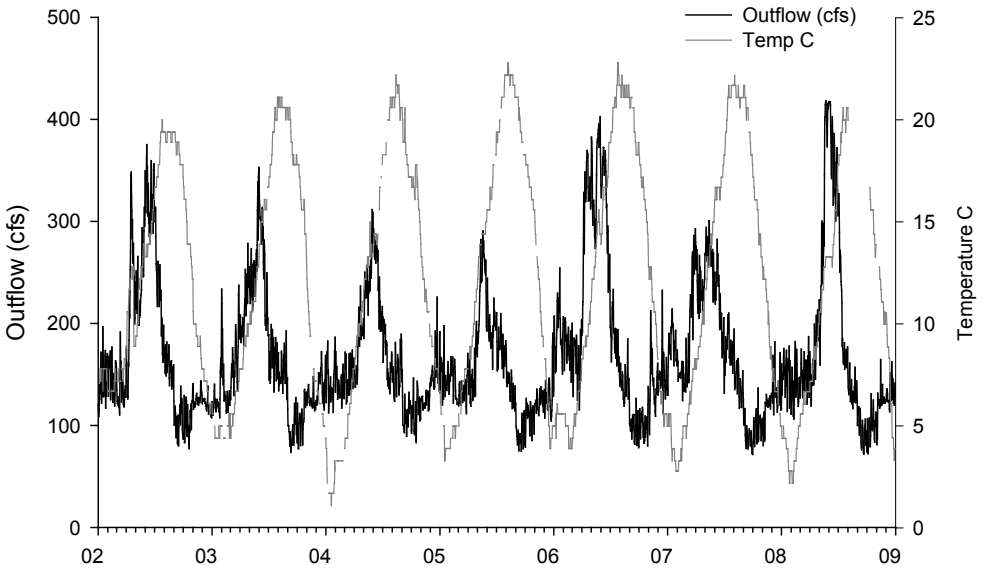


Figure 27. River discharge and temperature at Bonneville Dam, 2002-2008. Data from US Geological Survey (USGS 2006).

At Russian Island (rkm 35), the estimated availability of emergent wetland channels to juvenile salmon was relatively stable during rearing March-August or September, with little seasonal variation in habitat opportunity based on our depth and temperature criteria (Figure 28). During all survey months, emergent wetland channels should have been accessible to juvenile salmon about 60% of the time.

This pattern of accessibility likely reflected the consistent tidal influence on habitat in the lower estuary, as intertidal wetlands filled and drained twice daily. In contrast, at Lord Island (rkm 101), where the tidal range is much smaller, habitat opportunity was much more variable between years. This variation may have been more closely linked to river flow than to tidal influence. For example, during relatively low flows in 2007, Lord Island habitat was available about 75% of the time in March but less than 30% of the time in August. During higher river flows in 2008, the Lord Island wetland channel retained water at a sufficient depth for salmon more than 90% of the time from March through August.

Rearing conditions at both sites were classified as marginal during some periods in June, July, and August, when temperatures exceeded 19°C, but conditions otherwise were satisfactory for juvenile salmon to access the channel (Figure 28). Interestingly, marginal temperatures occurred most frequently at the Russian Island site in 2007. Greater tidal variation to expose channel bottoms and the lack of vegetative cover to provide shade likely explain the greater degree of warming at Russian Island relative to the Lord Island forested site.

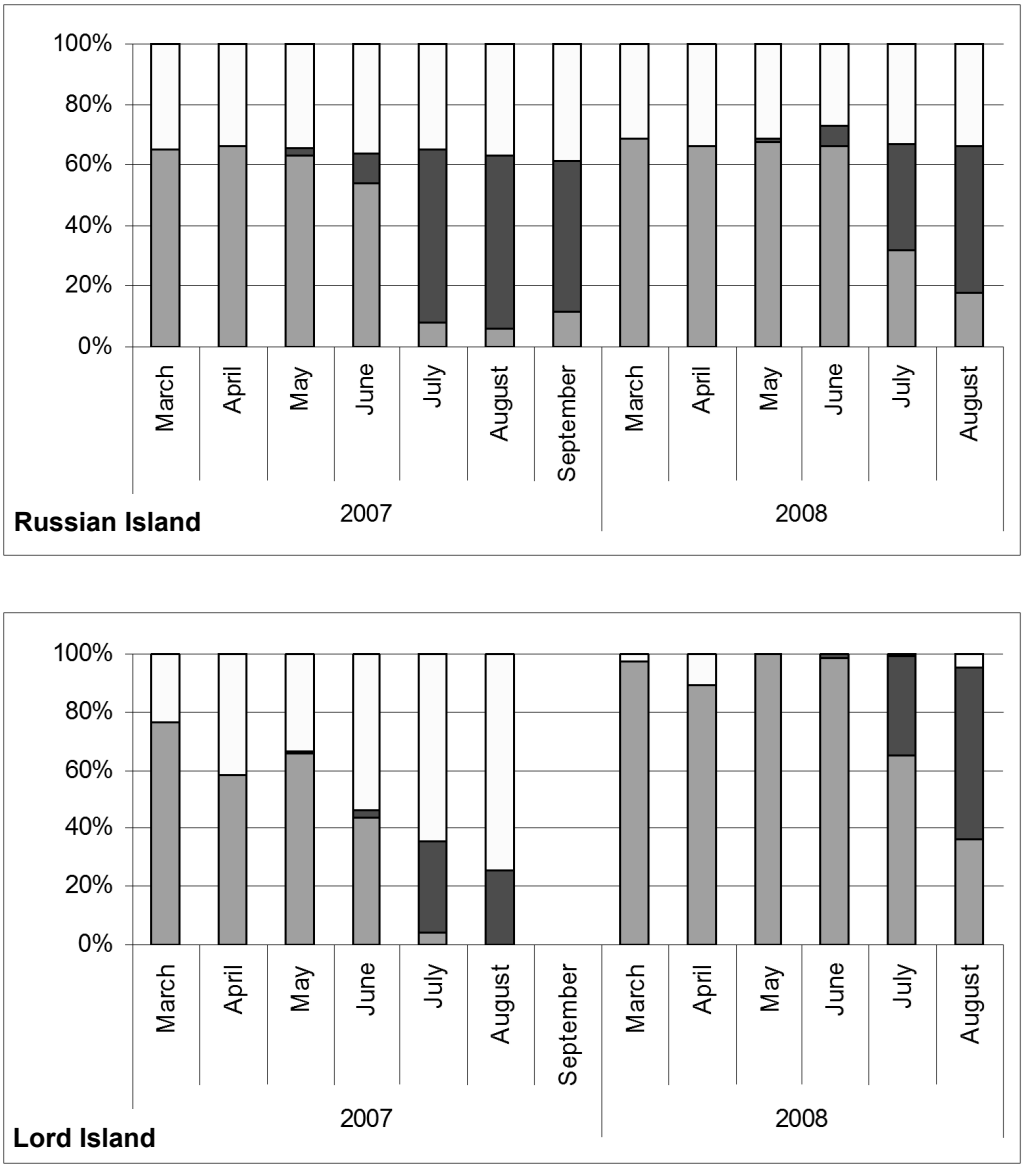


Figure 28. Habitat opportunity for juvenile Chinook salmon at Russian and Lord Islands as classified by water temperature and depth criteria. Each bar represents the percentage of time each month that salmon had access to channel habitat (i.e., depth > 0.5 m and water temperature was 19°C or lower). Channel habitat was rated not available, regardless of water temperature, when depths were < 0.5 m.

Salmon Distribution along the Exposure Gradient within Wetland Habitats

Methods

During the March-July wetland surveys at Wallace Island in 2006-2007 and at Lord Island in 2006-2008, we compared Chinook salmon size distributions and the percentage of hatchery-marked individuals along a shallow habitat gradient characterized by different levels of exposure to water depth, current, and amount of cover. With the 50-m beach seine, we collected juvenile salmon along both sides of each island: the relatively exposed main stem side and the more protected back-side margin. We compared these results with those from trap-net samples collected in shallow wetland channels in the interior of each island (described above), the least exposed of the three habitat types. Despite differences in sampling gear and efficiencies, the mesh sizes of the beach seine and trap net retained juvenile salmon ≥ 40 mm.

Results

The mean size of hatchery-marked individuals was larger than that of unmarked salmon within each of the three habitat types that we sampled along the exposure gradient (Figure 29). Average size of unmarked salmon generally decreased with declining habitat depths and exposure levels: mean sizes were generally largest along the main-stem side of each island; intermediate along the back-shore side; and smallest within the shallow, interior wetland channels. Despite such differences, unmarked fry appeared in all three habitat types throughout the sampling season (March-July).

In contrast to the trend observed for unmarked juveniles, we found no consistent trend in mean sizes for hatchery Chinook salmon along the exposure gradient at Wallace and Lord Island (Figure 29). However, the average size of hatchery fish sampled from interior wetland channels was generally smaller than that of those captured from wetland channels along either side of each island.

Mean sizes of Chinook salmon may have been biased somewhat in these comparisons because of the finer mesh of the trap-net used to sample interior wetland channels: the trap net may have retained a greater proportion of the smallest size class (≤ 40 mm FL) than the beach seine, which was used to sample main-stem and back-side habitats.

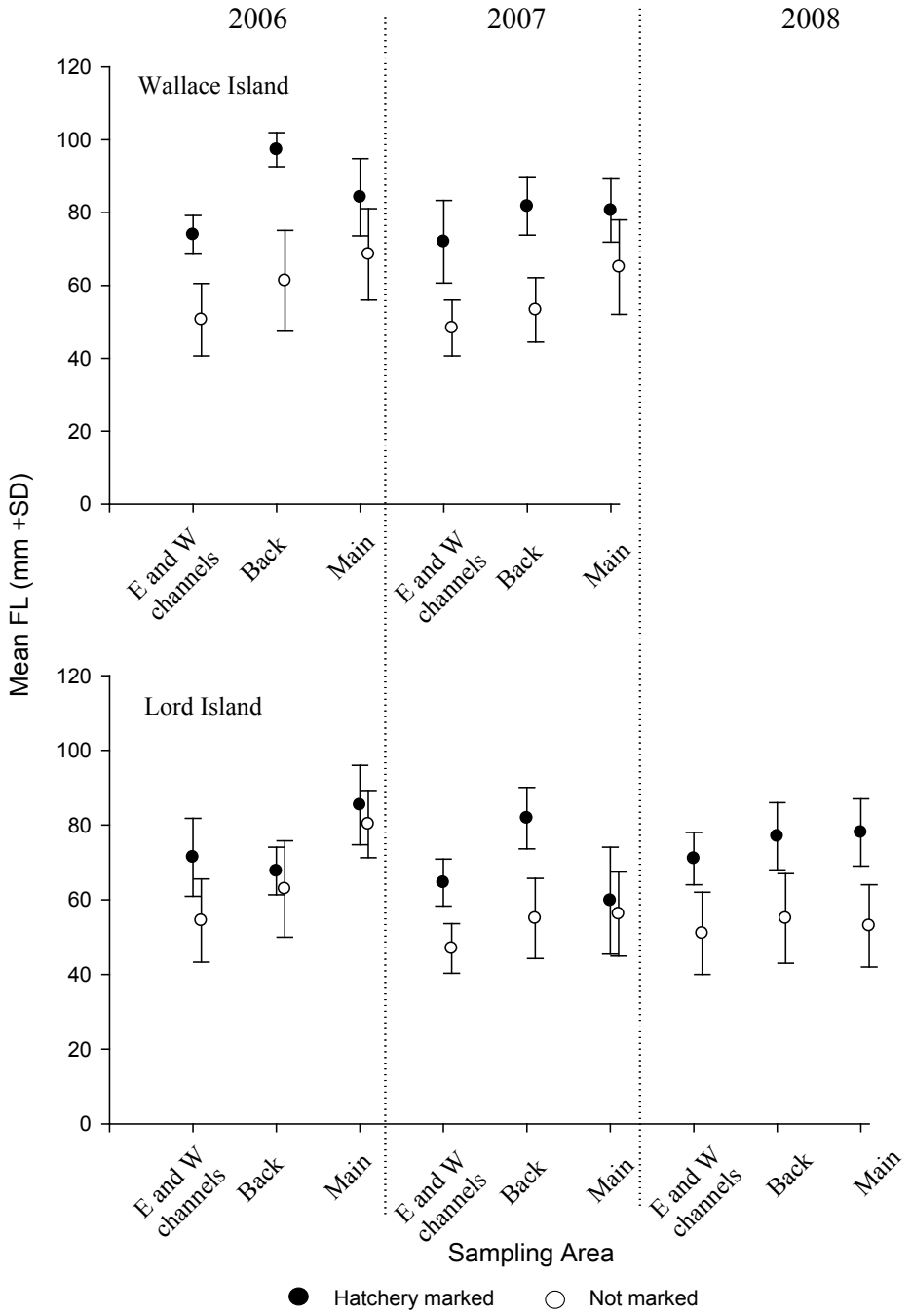


Figure 29. Mean FL (mm ± SD) for marked and unmarked juvenile Chinook salmon captured in Wallace and Lord Island wetland channels from three exposure levels: main stem, back-side, and interior wetland channels. All marked individuals were hatchery-reared, while unmarked fish may have been wild, hatchery-reared, or both.

However, these small differences in the minimum sampling size of gear type did not account for the more substantial differences observed among habitats at the upper end of the size distribution. For example, although we frequently collected subyearling Chinook salmon of 60-90 mm FL along the main-stem and back-side shores of Lord Island during 2007 and 2008, few individuals 60 mm or larger were found in the interior wetland channel (Figure 30). Therefore, it seems unlikely that the variation observed in mean FL along the exposure gradient (Figure 29) was simply an artifact of gear differences.

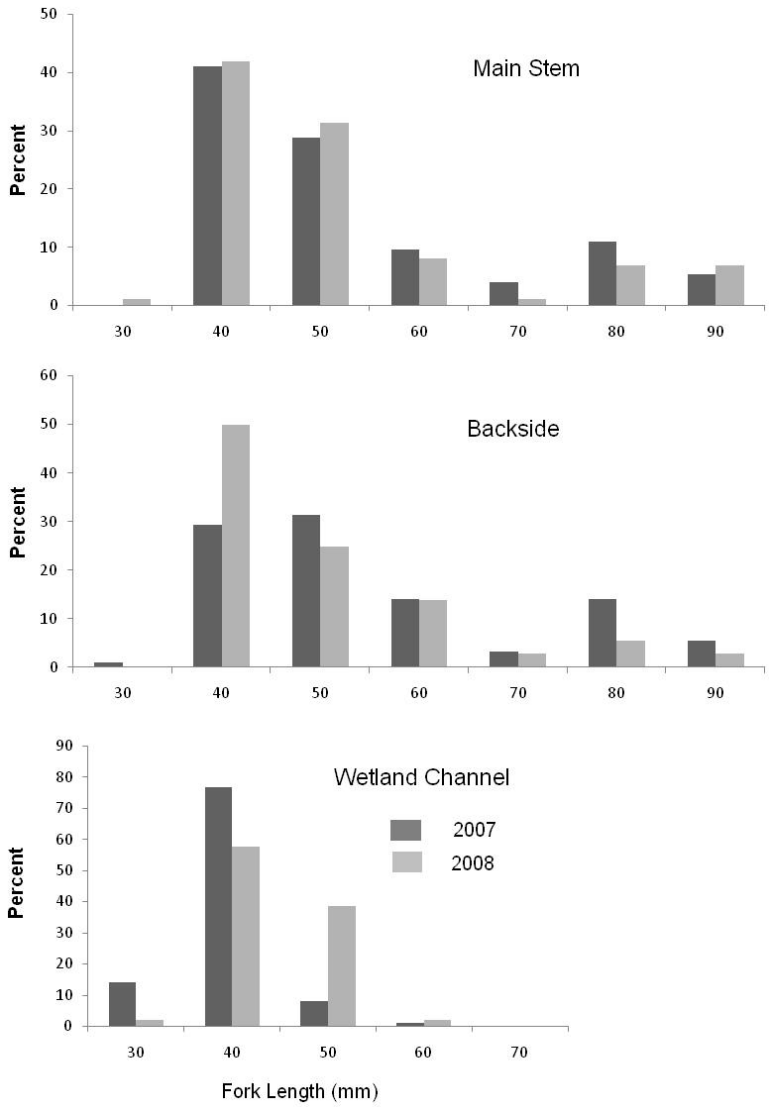


Figure 30. Size distribution of subyearling Chinook salmon captured in main-stem, back-side, and interior wetland-channel habitats at Lord Island, May 2007 and 2008.

III. HABITAT CAPACITY: PREY AVAILABILITY, DIET, AND RATES OF CONSUMPTION

Wetland Prey Assemblages

Insects

Methods—We estimated the composition and abundance of insect prey at each channel where we deployed a trap net to sample wetland fish assemblages (Figure 10) (Appendix Tables E1 and E2). Insect sampling coincided with the sites and dates of fish sampling, from March or April through July, 2002-2007 (see Tables 7 and 8).

During each survey, we distributed five replicate insect fallout traps along each study channel for a grand total of 655 individual collections across all study locations and years. Each fallout trap consisted of a 26.5-L clear plastic tub (58.4 × 40.6 × 15.2-cm) bordered by four PVC poles (2.5-cm diameter × 1.5-m tall), which held the tub in place while allowing it to rise and fall with the tide. Each tub was filled approximately one-quarter full with soapy water and left to sample for two tidal cycles.

We set the replicate fallout traps randomly along the edge of each study channel in the proximity of each trap-net site (Figure 10). After ~30 h of sampling, water in each tub was drained through a 106-μm sieve, and the contents were preserved in 70% isopropyl alcohol. In the laboratory, samples were examined under a dissection microscope. Individuals were counted and identified to the finest taxonomic level possible. Data are presented as counts m^{-2}.

Results—We sampled a total of 128 taxa in the fallout trap (Appendix Table E2), primarily insects of the orders Diptera (26%), Coleoptera (15%), Hymenoptera (15%) and Hemiptera (15%). Annual mean density of individuals at the 12 wetland sites ranged from 551 to 4,365 m^{-2} (Figure 31a). In all years, the numerical composition of fallout samples was dominated by three principal taxonomic groups: Chironomid midges (39%), collembolans (springtails, 28%), and other dipterans (flies and midges, 22%). Fallout insect densities increased from March to July except in 2005, when densities peaked in May (Figure 31b). While in most years total insect density increased throughout the season, mean chironomid abundance typically peaked in May. Between 2002 and 2007, mean chironomid density ranged from 335 to 995 individuals m^{-2}, with the highest densities observed in 2005.

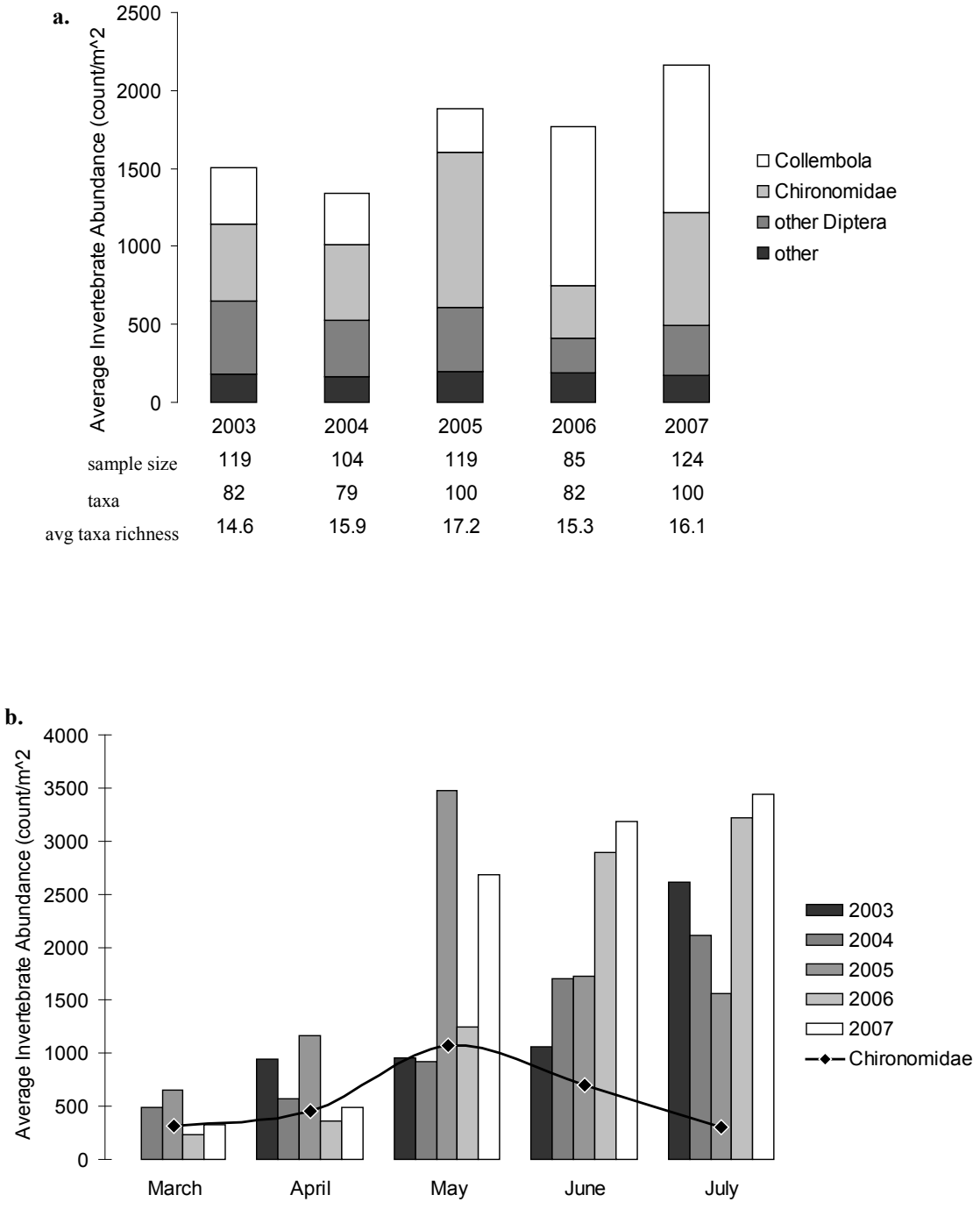


Figure 31. Upper panel (a) shows mean annual insect density, and lower panel (b) mean monthly insect density in insect fallout traps pooled over twelve estuarine wetland sites, Columbia River estuary, 2003-2007.

Benthic Macroinvertebrates

Methods—We collected five macrobenthic core samples from each wetland channel site (Figure 10) during each trap-net and fallout insect survey in 2002-2007. A single core was collected from each channel-bottom site adjacent to each replicate insect fallout traps. Each macroinvertebrate sample was collected by inserting a coring device (5.2 cm inner diameter) into the channel bottom to a depth of 10 cm. Samples were sieved at 500 μm and fixed in a 10% buffered formalin solution stained with Rose Bengal. In the laboratory, samples were examined under a dissection microscope, and taxa were identified and counted to the finest taxonomic level possible. Particular attention was given to the identification of taxa found in corresponding salmonid diets. All data are presented as counts m^{-2}.

Results—We processed 587 macrobenthic core samples from a diversity of wetland habitat types in the estuary during 2002-2007 (Appendix Table E1). A total of 64 taxa were present in core samples, including annelids (14%), gammarid amphipods and other crustaceans (30%), and insects (42%) represented by collembolans and other predominately immature taxa (Appendix Table E2). Annual mean density ranged 14,429-52,593 invertebrates m^{-2}. In all years, the numerical composition of benthic samples was dominated by three primary taxa: oligochaetes (49%), nematodes (27%), and Chironomid larva (8%). Benthic invertebrate densities were relatively consistent within years but varied between years, with average densities slightly higher in 2003 and 2004 than in later years (Figure 32). In all years from March to July, mean oligochaete density ranged 9,003-18,705 individuals m^{-2}, mean nematode density ranged 3,386-10,750 individuals m^{-2}, and Chironomid larva mean density ranged 2,972-4,807 individuals m^{-2}.

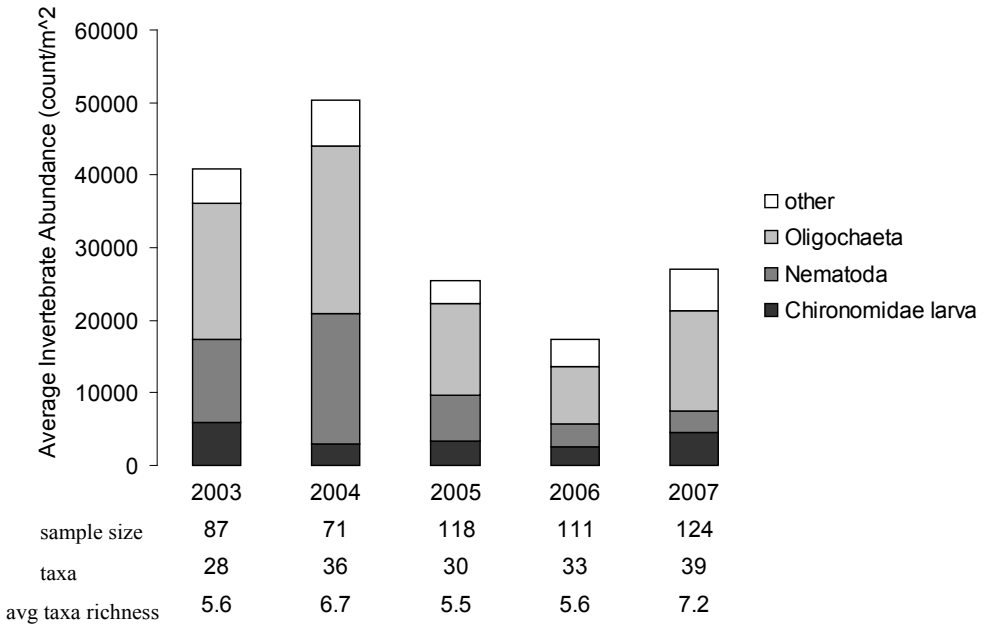


Figure 32. Mean annual densities (number/m^2) of benthic macroinvertebrates from benthic core samples pooled over twelve estuarine wetland sites, Columbia River estuary, 2003-2007.

Emergent Chironomids

Methods—In previous wetland surveys of the lower estuary, Lott (2004) identified emergent chironomids as a major constituent of juvenile Chinook salmon diet. However, chironomid habitat associations and seasonal patterns of abundance were difficult to interpret because taxonomic specificity for this group is generally lacking. To address this problem, we initiated additional studies in 2006 and 2007 to determine seasonality and microhabitat distribution of an emergent insect community in a freshwater tidal marsh, with emphasis on the chironomid family.

We sampled chironomids during emergence at three dendritic tidal channels located on the interior, southeast side of Russian Island (Figure 33). Distinct zones or microhabitats, referred to as the channel bottom, low bench, and high marsh, were sampled at three locations within each channel. We defined microhabitats subjectively as visually distinct habitat units along the channel, characterized by a slight change in elevation (range of about 1 m). Each microhabitat position varied in frequency and duration of flooding, associated water velocity, and vegetative community.

Insects were collected semi-monthly from April to June, and monthly in July and August 2006. At each event, traps were deployed for approximately 48 h. The emergent trap consisted of a truncated cone constructed from clear plastic sheeting with a bulb (modified water bottle) attached to the top to hold emerging insects. The trap was weighted and placed on the ground to capture all insects that emerged within the enclosed 0.6 m^2 basal area. When possible, we identified chironomids to the level of genus following Wiederholm (1989).

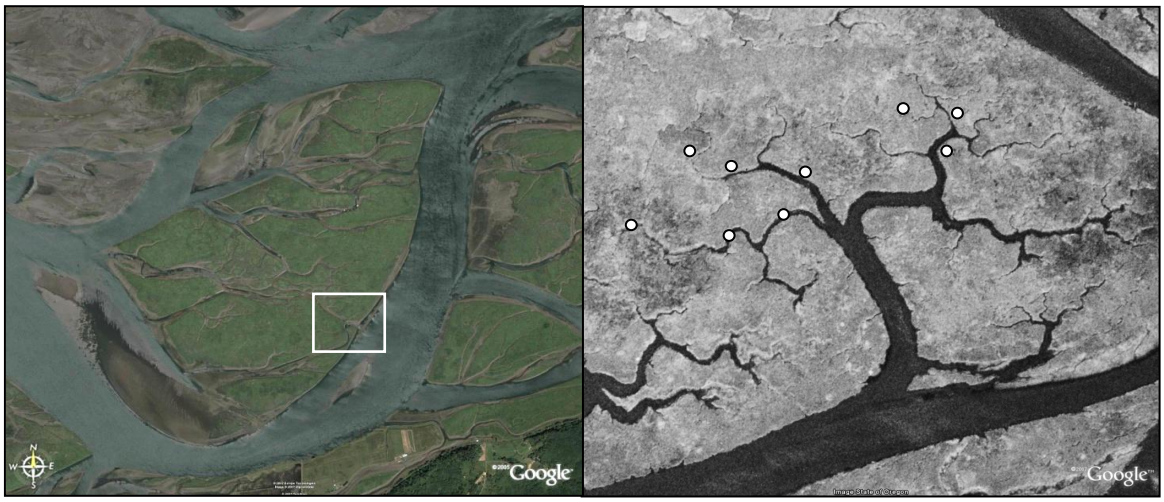


Figure 33. Emergent chironomid sampling sites at Russian Island study area, lower Columbia River estuary (see Figure 10). Left photo shows Russian Island, box outline locating study channels; Right photo shows three study channels; the two on the left were also the north and south study channels of the trap-net survey (Table 7). Microhabitats (channel bottom, low bench, and high marsh) were sampled at each of the nine sampling locations marked by white circles.

Results—In 2006 and 2007, we documented 37 insect families in emergent traps from the Russian Island tidal channels. Five families dominated the counts, representing approximately 78% of total numerical abundance. Chironomidae was typically the most abundant family and accounted for most of the spring emergence, reaching a peak in mid-June. In contrast, abundance of the total insect community increased through August. High densities of Dolichopodidae in July and August contributed to higher insect counts after chironomid numbers declined.

Three of the 11 chironomid subfamilies were present over the study period, and of these three, Orthocladiinae was most abundant (63.2%), followed by Chironominae (33.2%) and Tanypodiane (1.5%). Orthocladiinae dominated during spring periods of rising water temperature, while Chironominae were most abundant during summer, coinciding with maximum water temperatures for the year. Seventeen chironomid genera and an additional 6 morphospecies were identified, resulting in a minimum estimate of 23 genera at Russian Island. Temporal patterns of the family were derived from the specialized timing of individual taxa, particularly of the six major genera, which together accounted for 83.5% of the total abundance (Figure 34). Three dominant genera (Orthocladius, Thalassosmittia, and Paratendipes) constituted 68.4% of the total abundance. The timing of emergence peaks varied markedly among the three groups.

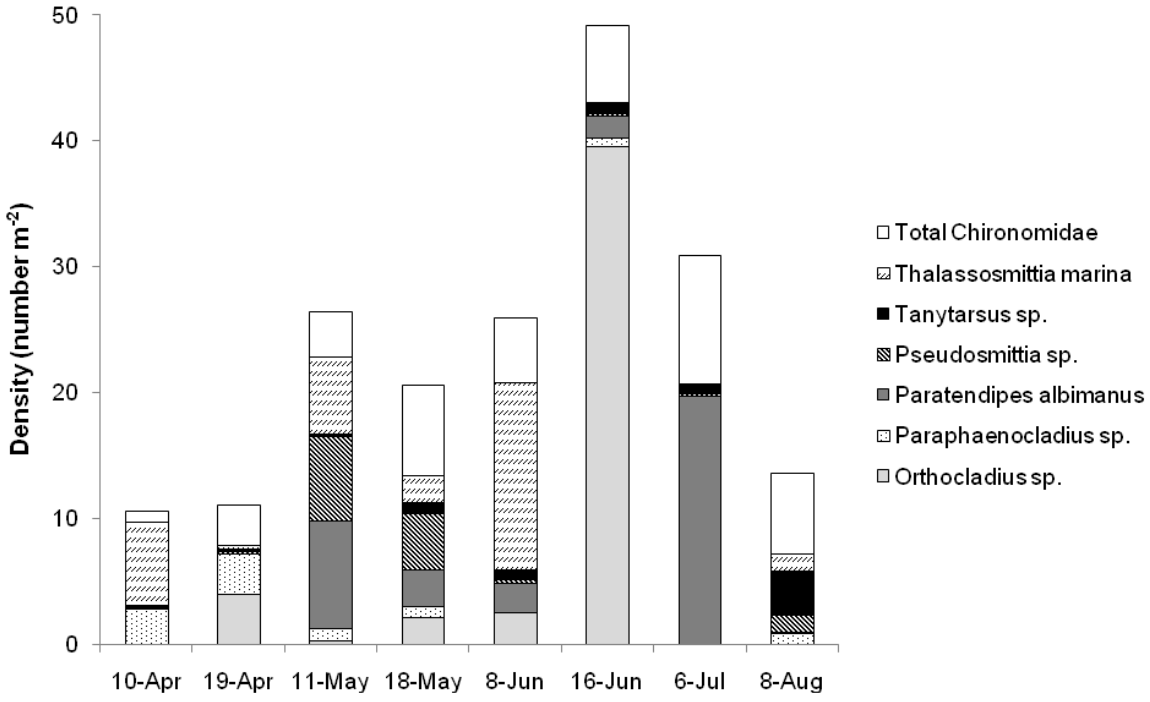
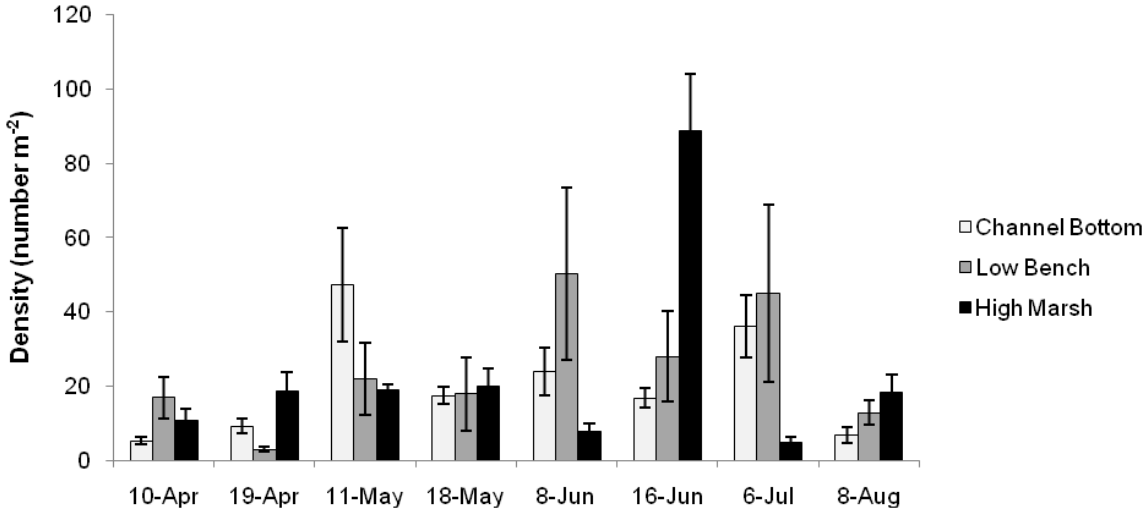
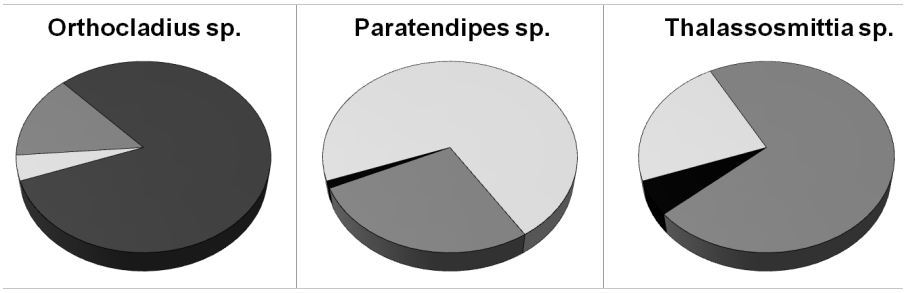


Figure 34. Temporal distribution of adult densities for the six major genera of emergent chironomids observed during 48-h sampling periods of emergent marsh habitat at Russian Island, 2006.

Chironomids, the most abundant family in this survey, occurred consistently over all microhabitats (Figure 35a). Distribution patterns that had been obscured when the more specialized component species of Chironomids were grouped together became apparent at a finer taxonomic resolution. When abundance data were analyzed at the generic level, a two-way crossed ANOSIM (to account for variation across sample dates), revealed significant differences in chironomid assemblages among microhabitats ($R = 0.457$, $P = 0.001$). Three genera were responsible for most assemblage distinctions, each characterizing a particular microhabitat (Figure 35b). The group's habitat associations and differences in emergence timing were reflected in the variations in peak chironomid abundance across microhabitats and through the season.



a. Total chironomids



b. Dominant genera

Figure 35. a) Mean emergence density (± 1 se) of adult chironomids during 48-h sampling periods for each microhabitat type; b) Proportional emergence by microhabitat type for the three dominant chironomid genera, Russian Island emergent marsh, 2006.

Juvenile Salmon Diets

Methods

For diet analyses, we collected juvenile Chinook salmon between 35 and 205 mm FL from each of the nine beach-seine sites sampled monthly in the lower 100 km of the estuary (Figure 10). In 2002-2003, we retained a minimum of 10 randomly selected individuals of each size class, and in 2004-2007, we expanded this number to 30. Each fish was euthanized, stored on ice in the field, and then frozen at -80°C until the stomach was removed for analysis. Stomachs were preserved in 10% buffered formalin for at least 24 h before properties were measured and contents enumerated.

Whole stomachs were rinsed three times in fresh water before blotting dry and weighing. Contents were dissected in the laboratory and sorted to the lowest identifiable taxonomic category. The number and wet mass of items in each category were recorded. Parasites found in the stomach (e.g. nematodes, trematodes) were analyzed separately and results will be reported elsewhere. After laboratory processing, each stomach and its contents were returned to a 70% ethanol solution for archival storage.

In the laboratory, we measured the following parameters for each individual:

- Percent stomach fullness (rated 0-100% based on visually estimated volume)
- Digestive rank (1-6, with 1 = completely digested/nothing identifiable and 6 = all items identifiable)
- Wet mass of all stomach contents, not including the stomach (g)
- Count of all stomach items for each taxon
- Wet mass of all items in each taxon (g)

We calculated the index of relative importance (Pinkas et al. 1971; Bowen 1983) for each prey item in each stomach as follows:

$$IRI_{stomach\ item} = F\,(N + G)$$

where

F = Frequency of occurrence (%) of a prey taxon of the total contents,
N = Numerical composition (%) of a prey taxon of the total contents, and
G = Gravimentric composition (%) of the total contents.

We expressed results as the percent contribution of any particular prey item to the sum of all IRI in a set of stomachs:

$$\% \ Total \ IRI = IRI_i / sum \ IRI_{i,1\text{-}x}$$

where

i = individual prey taxon of x total taxa.

The IRI can vary between 0% (none of an item occurred in the sample set) and 100% (all contents in a sample set were composed of a particular item).

We also estimated instantaneous ration (IR) for each individual, calculated as the total wet weight of stomach contents as a percentage of total body weight.

For the diet analysis, two primary comparisons within the total sample set were investigated. First, we compared sampling zones along the tidal gradient (marine, estuarine mixing, tidal freshwater/lower estuary, tidal freshwater/mid-estuary) to determine whether salmon diets in the estuary varied spatially. Second, we analyzed monthly samples from two stations—Pt. Adams Beach (estuarine mixing zone) and Lower Elochoman Slough (tidal freshwater/lower estuary)—to compare temporal variations in salmon diets.

Results

We examined the stomach contents of 2,145 Chinook salmon captured by beach seine within the following four estuarine zones: a mid-estuary tidal freshwater zone, a lower estuary tidal freshwater zone, an estuarine mixing zone, and a marine zone (Table 12). Each zone was characterized by its relative distance from Bonneville Dam, tidal range, and salinity levels. Tables listing all items found in stomachs from each estuary zone in each year are provided in Appendix D.

Table 12. Total number of juvenile Chinook stomachs examined by zone and year. Paired beach seining sites sampled along the main-stem side of Lord and Wallace Islands are designated UM for upper main stem and LM for lower main stem.

		Juvenile Chinook stomachs (n)						
Zone	Station	2002	2003	2004	2005	2006	2007	Total
Tidal freshwater mid-estuary	Lord Island—LM	-	-	-	-	48	12	60
	Lord Island—UM	-	-	-	-	19	7	26
	Wallace Island—LM	-	-	-	-	21	28	49
	Wallace Island—UM	-	-	-	-	36	55	91
Tidal freshwater lower estuary	E Tennasilahee Island	20	28	12	11	-	-	71
	Lower Elochoman Slough	98	189	168	119	121	-	695
	Upper Clifton Channel	33	43	60	36	17	-	189
Estuarine mixing	Pt. Adams Beach	55	145	113	129	108	-	550
	Pt. Ellice	22	22	24	33	31	-	132
Marine	Clatsop Spit	22	53	41	38	11	-	165
	West Sand Island	23	18	33	22	21	-	117
Total		273	498	451	388	433	102	2,145

We analyzed the stomach contents of 872 juvenile Chinook salmon collected at six wetland survey sites, constituting four discrete habitat types between rkm 35 and 101 (Table 7; Figure 10): emergent, forested, scrub-shrub, and mixed. We processed another 359 juvenile Chinook salmon collected at South Russian Island (emergent marsh) and lower Lord Island (mixed) during the residence time/consumption rate experiments described below. All individuals sacrificed for the wetland diet analysis were placed on ice in the field and then frozen at −80°C for later analysis. Samples were processed in the laboratory with a dissection microscope. The methods and parameters for wetland dietary analysis were the same as those described above for the tidal gradient samples.

Diet Composition along the Tidal Gradient—Most stomach contents were significantly digested. Mean digestive rank was 2.29, indicating that ~25% of stomach items were identifiable to some taxonomic level. Stomach fullness and ration data indicated that juvenile Chinook salmon captured in all zones were actively foraging. Empty stomachs were rare; only 1% of stomachs examined were less than 5% full by volume. Mean stomach fullness by volume was greater than 60% in all zones and greater than 75% in 15 of 17 year-zone combinations (Table 13).

Table 13. Mean stomach fullness as a percentage of total volume by zone and year.

	Volume stomach fullness (%)					
Zone	2002	2003	2004	2005	2006	2007
Tidal freshwater, mid-estuary	-	-	-	-	95.5	93.0
Tidal freshwater, lower estuary	87.6	88.0	90.3	95.7	94.2	-
Estuarine mixing	76.4	77.2	92.3	91.1	91.4	-
Marine	66.2	75.0	88.2	84.2	72.0	-

Estimated instantaneous ration for juvenile Chinook salmon varied between 0.01 and 14.7%. Mean stomach ration varied across zones, with highest rations observed in the marine zone (1.37-4.66%) and lowest in the estuarine mixing zone (0.63-0.70%; Table 14).

Table 14. Mean stomach ration, as a percentage of wet body mass, by zone and year.

	Stomach ration proportion of wet body mass (%)					
Zone	2002	2003	2004	2005	2006	2007
Tidal freshwater, mid-estuary	-	-	-	-	0.63	0.31
Tidal freshwater, lower estuary	1.26	1.26	1.28	1.41	1.53	-
Estuarine mixing	0.63	0.68	0.66	0.70	0.70	-
Marine	4.66	3.37	3.25	3.77	1.37	-

As indicated by total IRI, adult dipterans (flies & midges) and the benthic amphipod *Americorophium salmonis* comprised the two most important diet items in fish captured from most zones (Figure 36). Dipterans composed between 11 and 74.2% of total IRI; dipterans made the highest IRI contribution in fish captured from estuarine mixing zone. *A. salmonis* composed between 2 and 44.9% of total IRI across zones and made the largest dietary contribution among fish captured from tidal freshwater in the

mid-estuary. Freshwater cladocerans (water fleas) were the next most important prey group (34.4% of total IRI) among fish captured from mid-estuary sites at Lord and Wallace Islands.

In samples from the marine zone, the amphipod *A. salmonis* was replaced with the congener *A. spinicorne*, a species known to prefer more saline environments. Plant debris was an important component of stomach contents in the marine environment (16.32% of total IRI). Fish may derive some nutritional benefit either directly from plant material or from the bacterial films that cover it. A complete list of items by IRI contribution from juvenile Chinook salmon stomachs can be found in Appendix D.

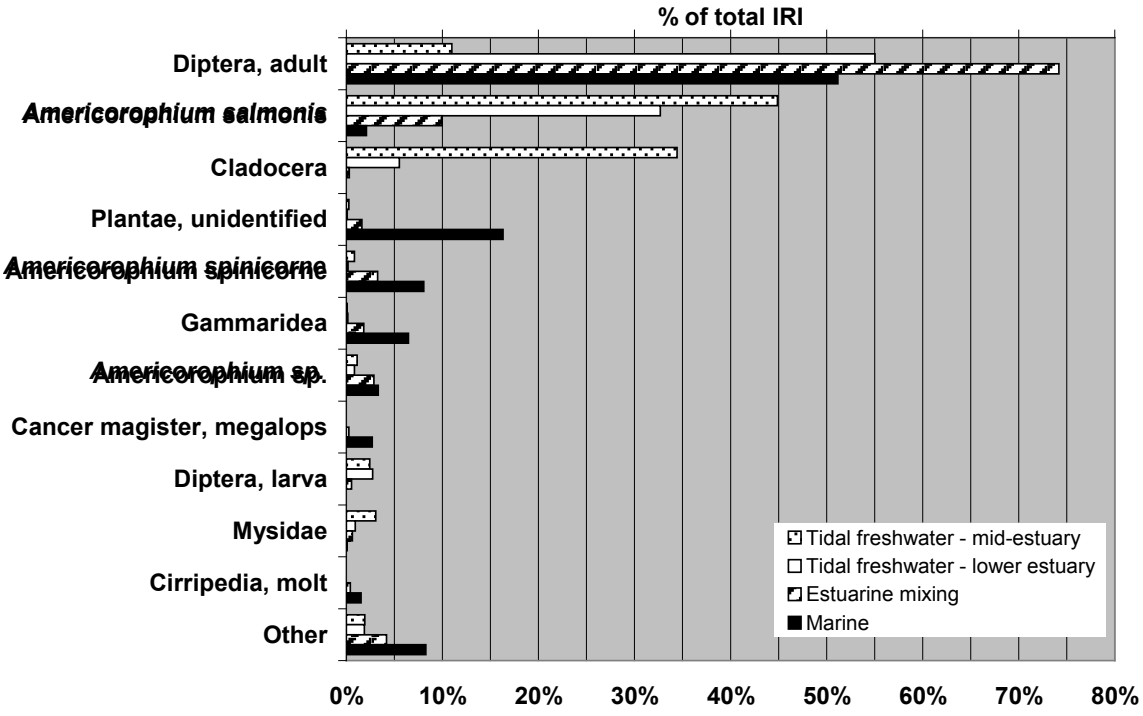


Figure 36. Percent contribution to total index of relative importance (IRI) of the 11 most important stomach contents for juvenile Chinook salmon, by estuary zone.

Seasonal Trends in Diet Composition—Monthly sample sizes were sufficient at Lower Elochoman Slough and Pt. Adams Beach to examine seasonal trends in the importance of each taxon in the diets of juvenile Chinook salmon (Table 15).

In the tidal freshwater area of the lower estuary (Lower Elochoman Slough), both dipterans and the benthic amphipod *A. salmonis* were consumed throughout the year (ranges 7.9-85 and 5-71.3%, respectively), except in August, when freshwater cladocerans became an important component of the diet (65.8% of total IRI; Figure 37). Consumption of *A. salmonis* reached peaks between March and May and again in October and November. Dipteran consumption peaked in June and July and again in September. Other taxa contributed much less to diet composition than these three principals. A complete list of all items contributing to total IRI at Lower Elochoman Slough is provided in Appendix Table D6.

Table 15. Total number of juvenile Chinook salmon stomachs examined by month and sample site. LES, Lower Elochoman Slough; PAB, Pt. Adams Beach.

	Stomachs examined (n)											
Station	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Totals
LES tidal freshwater, mid-estuary	47	59	56	117	113	99	80	56	38	27	3	695
PAB—estuarine mixing	6	33	55	76	109	79	75	43	31	21	22	550

In the estuarine mixing zone (Pt. Adams Beach), stomach contents were dominated by adult dipterans throughout the year (35-86.6% of total IRI; Figure 38). Mysids (all identifiable specimens were *Neomysis mercedis*) were important in January and March (62.2 and 28.5% of total IRI, respectively). *A. salmonis* was important in March through May, the same period during which amphipod contributions peaked at Lower Elochoman Slough. Other taxa contributed smaller amounts to stomach contents. A complete list of all items contributing to total IRI at Pt. Adams Beach is provided in Appendix Table D7.

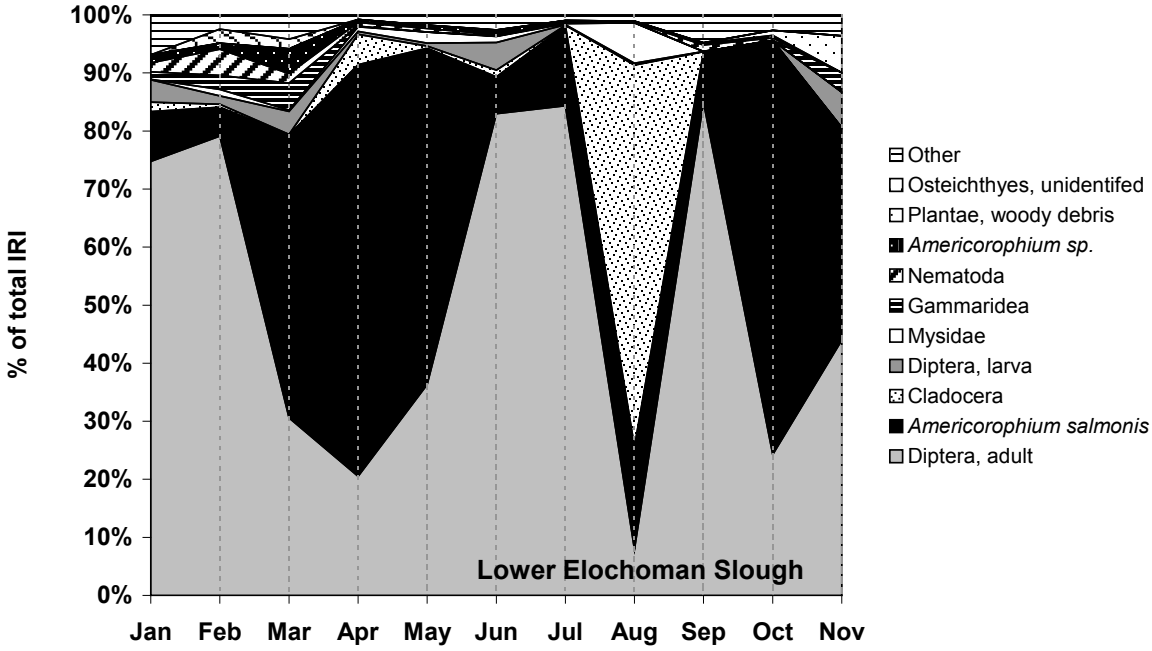


Figure 37. Percent contribution to total index of relative importance (IRI) by month of the ten most important stomach contents for juvenile Chinook salmon sampled from the tidal freshwater zone at Lower Elochoman Slough.

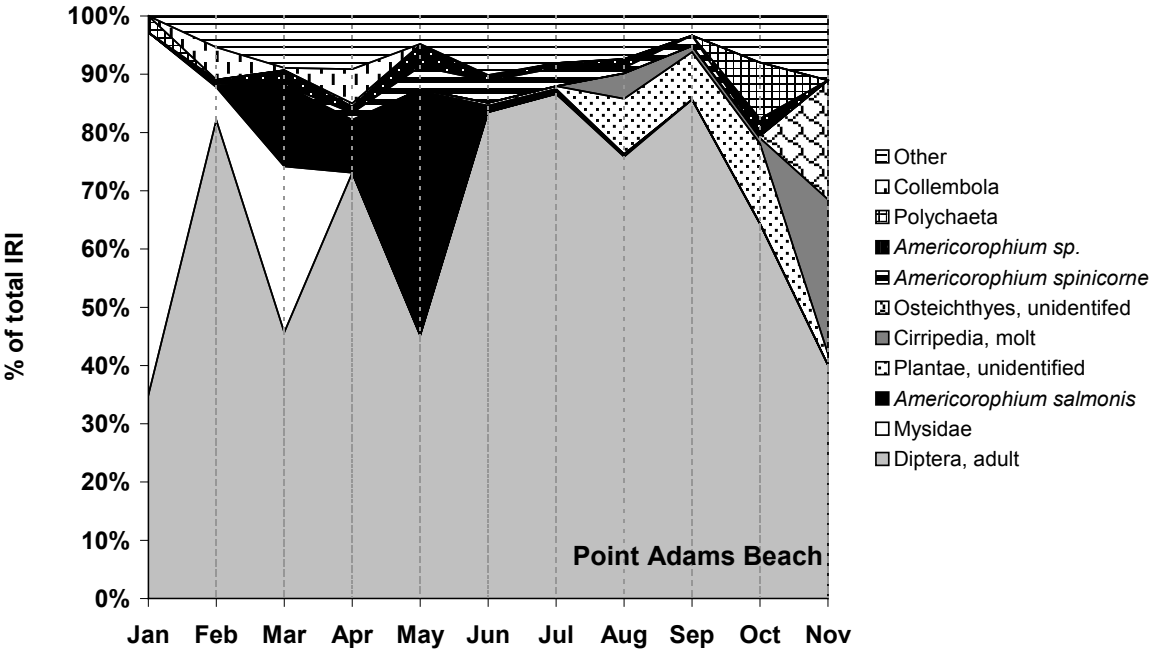


Figure 38. Percent contribution to total index of relative importance (IRI) by month of the ten most important stomach contents for juvenile Chinook salmon sampled from the estuarine mixing zone at Pt. Adams Beach.

Diet Composition in Wetland Habitats—Approximately 110 discrete prey taxa were identified from the stomachs of juvenile Chinook salmon sampled in tidal wetland habitats (Appendix Table E2). Only 2.3% percent of the stomachs were empty. The vast majority of prey taxa occurred rarely and in low abundance or gravimetric contribution. To simplify presentation, we report results for all prey that constituted >5% of the total IRI in any single prey category.

Twelve prey taxa dominated the diet, including 7 insects, 3 amphipods, 1 polychaete annelid, and various types of plant matter (seeds; Figure 39). Except for salmon from scrub-shrub habitats in the 80-98 mm FL size interval, emerging chironomid dipterans (midges) dominated the diet of all fish 30-98 mm and usually constituted over 50% of the total IRI (Figure 39). Other life history stages of chironomids (pupa, larva), and other (unidentified) dipteran pupa usually accounted for another 10% of the total IRI.

Other than different life stages of chironomids, only the amphipods, *Americorophium* spp. (juveniles and adults), approached 10% of the IRI for the two smallest size intervals of juvenile Chinook salmon (30-59 and 60-79 mm FL; Figure 39). Usually gammarid amphipods occurred most frequently in larger salmon (80-98 mm FL). Exceptions to this rule were other insects and juvenile and adult mites (Araneae) found among 60-79 mm fish in the tidal forested (Sitka spruce) habitat and adult water boatmen (Corixidae) and plant matter found in 80-98 mm fish from the scrub-shrub habitat. The contribution of emerging chironomids (i.e., Chironomidae pupa-adult) and other insects generally decreased with increasing fish size, with a relatively abrupt transition to equivalent or higher contributions of other prey among fish >80-98 mm. This was most evident in the emergent marsh among Chinook salmon >100 mm, where juvenile and adult amphipods (*Americorophium* spp.; Anisogammaridae; Gammaridae) and juvenile and adult polychaete annelids were dominant.

One significant exception to the above trend was in the mixed habitat (scrub-shrub and black cottonwood forested), where foraging on emerging chironomids increased, and foraging on chironomid larvae decreased, with increasing fish size (Figure 39). These results were consistent with recent stable isotope data (Maier and Simenstad 2009) indicating that small juvenile Chinook in the lower estuary are primarily connected to food-web pathways from wetland macrophyte production. Detritivorous chironomids, which are consumed as emerging adults by juvenile salmon on or near the water surface, appeared to provide a primary link in this pathway. Supplemental prey, also detritivores found in and around the emergent marshes, included epibenthic (gammarid) amphipods and polychaetes.

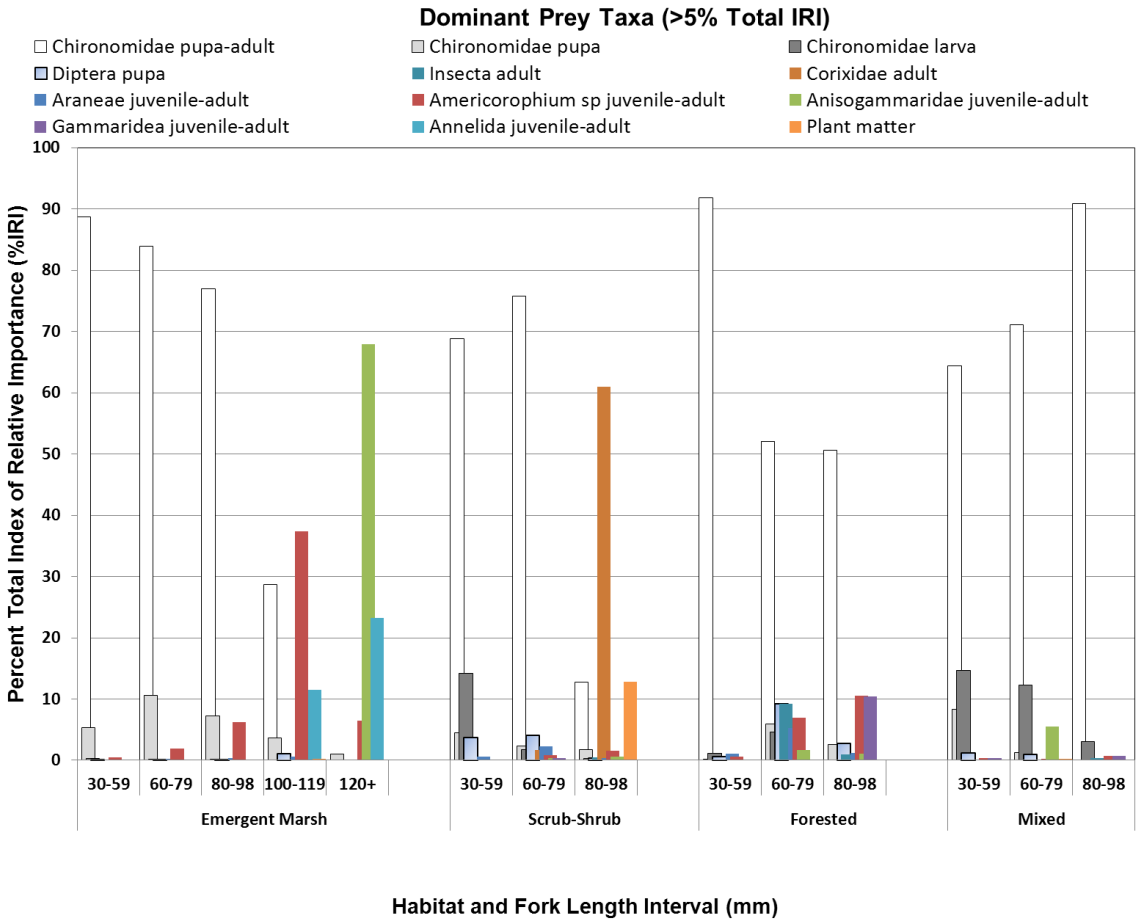


Figure 39. Prey composition (Index of Relative Importance, IRI) of dominant (>5% Total IRI) of juvenile Chinook salmon of up to five size intervals in four estuarine wetland habitats, Columbia River estuary, 2003-2007.

In general, dipteran insects, and specifically emerging chironomids, dominated the diet of juvenile Chinook, regardless of the shallow-water habitat type they occupied (e.g., emergent marsh at Russian Island, and fringing emergent marsh, scrub-shrub and forested wetlands at Wallace and Lord Islands; Figure 40). Exceptions to chironomid dominance typically involved increasing contributions from other dipterans and prey taxa, including the amphipod *Americorophium* spp., with increasing fish size and during later months of the rearing season.

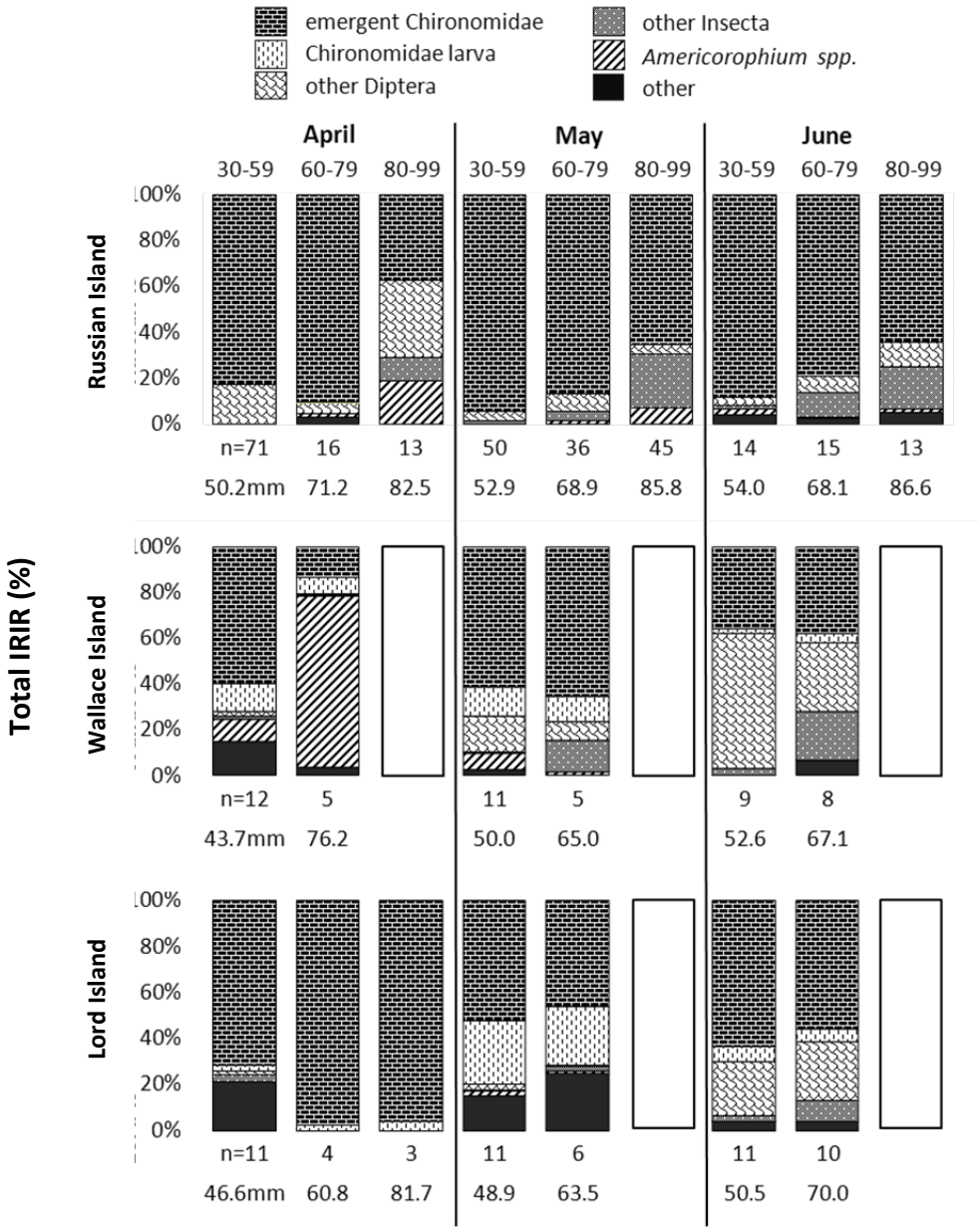


Figure 40. Primary diet composition of three different size classes of juvenile Chinook salmon at each of three wetland sites (Russian, Wallace, and Lord Islands) during three months of peak abundance in the lower Columbia River estuary, 2002-2007 (combined). Diets were pooled into six dominant prey categories based on percent total IRI. Sample size (n) and mean fork length (mm) are listed immediately below each histogram.

This pattern was typical over the 6 years that we sampled the emergent marshes at Russian Island (Figure 41), although sample sizes of May and June collections and of larger fish size classes in April were insufficient for diet characterization in some years (particularly 2004-2006).

The dietary contributions of *Americorophium* spp. and Diptera and other insects generally increased with increasing salmon size class, although *Americorophium* spp. occurrence varied from year to year. We did not have sufficient benthic macroinvertebrate data to verify whether the relative abundance of *Americorophium* spp. varied interannually or whether the dietary shift reflected a feeding preference of juvenile Chinook salmon.

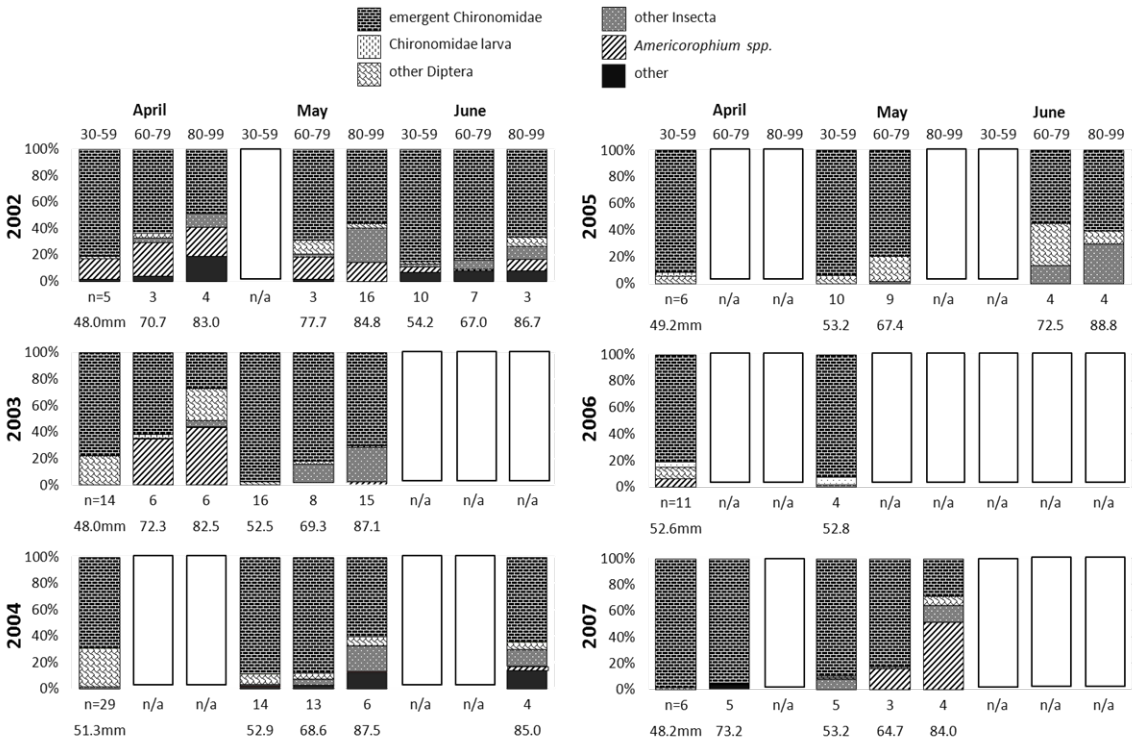


Figure 41. Annual diet composition of three different size classes of juvenile Chinook salmon during three months of peak abundance in the emergent marsh at Russian Island, lower Columbia River estuary, 2002-2007. Diets were pooled into six dominant prey categories based on percent total IRI. Sample size (n) and mean fork length (mm) are listed immediately below each histogram.

Dietary Interaction between Co-occurring Salmon and Threespine Stickleback

Large numbers of threespine stickleback consistently co-occurred with juvenile salmon in lower estuary emergent wetlands and tidal channels. During 2006-2007, we conducted studies to determine 1) whether diet composition of the two species overlapped within wetland habitats, and 2) whether competition for a limited prey resource influenced the performance (e.g., individual growth) of juvenile Chinook salmon (Spilseth 2008; Spilseth and Simenstad 2011).

Methods

In 2006, we examined diet overlap between co-occurring threespine stickleback and ocean-type juvenile Chinook salmon in three estuarine emergent marshes. To examine resource partitioning across multiple temporal scales, we sampled diets monthly, as well as over one complete diel cycle. Monthly and diel experiments were conducted using similar field-sampling and laboratory techniques. However, monthly samples were collected from smaller channels to facilitate capture and to accurately estimate fish density, while diel sampling was conducted in deeper, broader channels to allow capture of fish at the same location throughout the tidal cycle.

In 2007 we conducted a field manipulation (enclosure) experiment in the Chinook River estuary, a tributary of the lower Columbia River, in order to test salmon response to different density ratios of stickleback to salmon under the same relative levels of prey availability. We completed three successive week-long field manipulation experiments in March, April, and May. Each experiment used a geometric design to scale the ratio of fish densities in three 9-m^2 enclosures: threespine stickleback densities varied from low (no individuals) to medium (75 individuals) or high (150 individuals) relative to a constant Chinook salmon density (10 individuals).

Field enclosures were placed adjacent to one another in the same habitat (depth, bottom substrate). We assumed that prey resources associated with the benthos (i.e., emergent chironomid insects) were comparable among experimental treatments. Benthic fauna samples (5-cm core sieved to 500 μm) were acquired at the beginning and end of each experiment to document potential changes in prey density.

At the end of each experiment, all fish from each enclosure were collected and counted. Each juvenile salmon was weighed and its stomach was retained for diet analysis. We estimated the percent dry weight (minus the stomach) of each fish after drying at 60°C for 7 d.

Results

In 2006, the greatest overlap in foraging microhabitat between juvenile Chinook salmon and threespine stickleback occurred in early spring, before Chinook salmon began to consume high proportions of surface prey (Figure 42). Threespine stickleback consumed primarily benthic prey throughout spring and summer, and its diet included a broader range of taxa than that of juvenile Chinook salmon.

Overall, Chinook salmon consumed greater proportions of benthic and epibenthic prey in March, and transitioned to mid-water/surface prey in April-May (e.g., emerging chironomid flies) and surface-prey by July. Seasonal shifts in prey by Chinook salmon have been observed in other studies (Sambrook 1990; Lott 2004), and were likely associated with the increasing availability of aquatic and terrestrial drift insects.

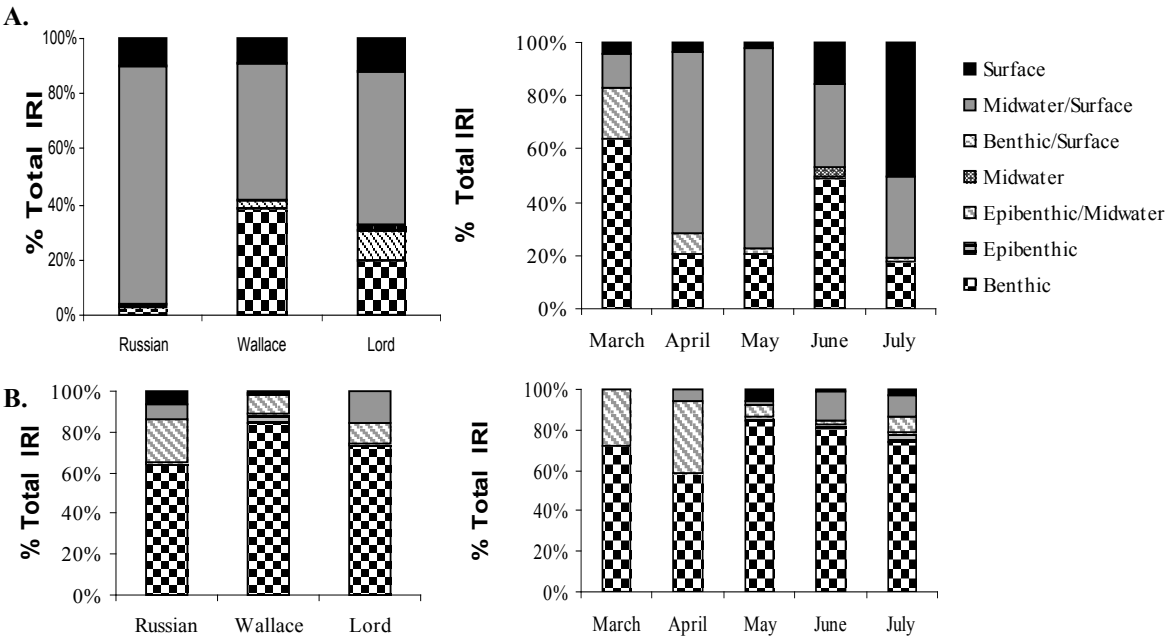


Figure 42. Mean proportional contribution of various prey types (% total index of relative importance) consumed by A) juvenile Chinook salmon, and B) threespine stickleback at each site and month in the lower Columbia River estuary, 2006. Each prey type is categorized by the microhabitat in which it was likely consumed (Appendix D, Spilseth 2008).

During experimental manipulations, threespine stickleback density did not significantly affect juvenile Chinook salmon performance during March or May. However, in April, mean growth and percent dry weight of Chinook salmon decreased with increasing density of threespine stickleback, although the decrease in growth was the only statistically significant change (Figure 43). Relative stomach fullness of threespine stickleback did not differ between treatments in any of the experiments. Also during April, Chinook salmon in the low-density treatments had a greater proportion of epibenthic prey in their diets, while fish in the high-density treatments had a greater proportion of surface prey.

Specific conditions of prey availability during April, perhaps characterized by a low abundance of terrestrial drift prey at the surface, caused threespine stickleback to displace juvenile Chinook salmon foraging on epibenthic prey, resulting in lower salmon growth rates. These results suggest that any competitive interaction between threespine stickleback and juvenile Chinook salmon would most likely occur in early spring, before terrestrial prey become abundant.

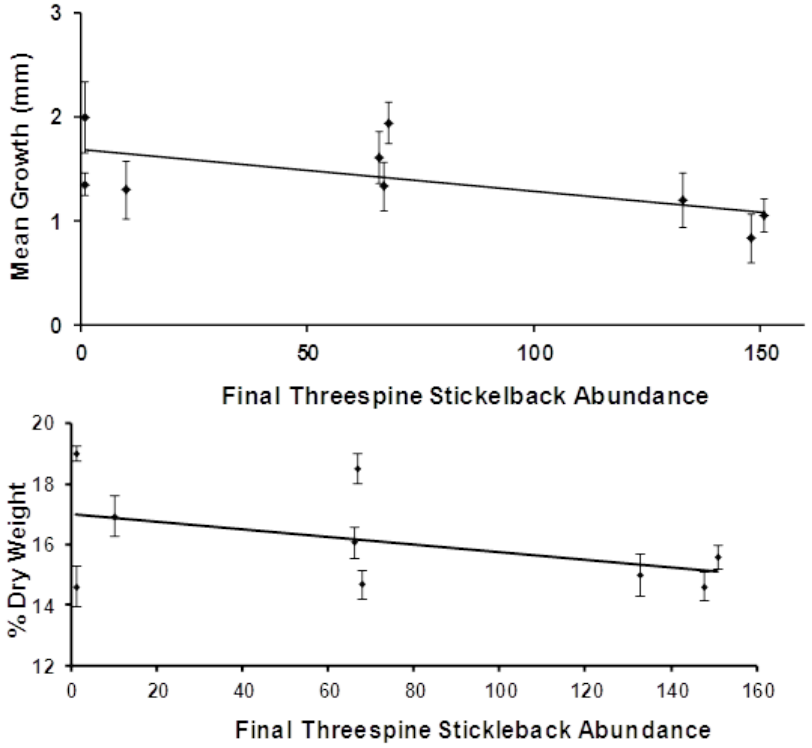


Figure 43. Upper panel shows regression of mean growth (± SE) of juvenile Chinook per enclosure and final number of threespine stickleback ($R^2 = 0.40$, $P = 0.07$). Lower panel shows percent dry weight of juvenile Chinook salmon per enclosure and final number of threespine stickleback ($R^2 = 0.21$, $P = 0.22$).

Diel Consumption Rates of Juvenile Chinook Salmon

Methods

For studies of juvenile Chinook salmon diel consumption rate, we collected a series of samples from two wetland sites. At Russian Island, we collected a total of 299 samples during 19-20 April 2005, 1-2 June 2006, and 12-13 April 2007. All juvenile Chinook salmon samples were collected by seine within a ~0.5 km^2 area at the southeast end of Russian Island. The seining site was located in a large distributary channel just downstream from the chironomid sampling site (Figure 33). This area was chosen because it remained submerged at low tide, allowing fish to be collected throughout the tidal cycle. At the lower channel study site at Lord Island, we also collected 61 juvenile Chinook salmon during 14-15 April 2007 to compare diel consumption rates in a forested swamp habitat with those from an emergent wetland.

For each diel survey, we collected up to 5 unmarked Chinook salmon of each available size class with a 38-m beach (bag) seine. The seine was fished every 3 h over a 30-h period, where each seining period started at the designated time and continued for 1.5 h or until 5 salmon of each size class had been collected. Fish were euthanized using MS-222, and fork lengths and weights were recorded. Individuals were immediately fixed in a 10% buffered formalin solution. Temperature data loggers (Stowaway Tidbit) recorded water temperature at the center of the sampling area every 10 seconds.

For individuals collected for diel consumption studies, we calculated % IRI and instantaneous ration (IR) using the same methods described above for juvenile salmon diets. Evacuation rate was calculated from mean instantaneous ration in fish collected between 2130-2300 and 0030-0200 PDT (the periods when stomach content weight was observed to decrease most rapidly) as:

$$S_t = S_0^{-rt}$$

where S_t is the mean instantaneous ration at time t, S_0 is the initial mean instantaneous ration, r is the instantaneous rate of gastric evacuation, and t is the time elapsed (Doble and Eggers 1978).

Daily meal (F) or the grams per day consumed was calculated as:

$$F = 24Sr$$

where S is mean stomach fullness over a 24-h period.

Daily ration (*D*), or the total amount of food consumed per day as a percentage of fish body weight, was calculated as:

$$D = 100\ F/\text{average fish weight}.$$

Results

Juvenile Chinook salmon sampled during diel consumption studies exhibited a crepuscular feeding pattern, with a peak in feeding activity early in the morning and second peak before dark (Figure 44A-B). Mean instantaneous ration was consistently lowest during 0330-0500 PDT, at 0.01 for all years combined (mean stomach content weight 0.035 g). Mean instantaneous ration was highest during 0930-1100 and 1830-2000, with respective mean IRs of 0.014 and 0.017 and respective mean weight of stomach contents at 0.087 and 0.072 g. Only two empty stomachs were recorded for all years.

While stomach content weight varied throughout the day (Figure 44A), diet composition was similar between seining periods and years, consisting primarily of chironomid larva and gammarid amphipods of the genus *Americorophium*. Diet composition was similar among different size classes of Chinook salmon, although larger fish typically consumed greater proportions of *Americorophium* spp., a larger prey item. At Russian Island in 2007, annelids and nereid polychaetes represented a significant proportion of the total prey biomass consumed by 80-99 mm Chinook salmon.

Because both Americorophium and nereid polychaetes are known to migrate into the water column at night, their prominence in the diets of larger fish suggested nocturnal foraging outside the tidal channels. The incidence of *Americorophium* spp. in the diets of Chinook salmon collected during diel sampling was considerably higher than that of salmon collected during monthly trap-net surveys (Figure 44C). This may reflect some microhabitat differences in invertebrate distributions. Although *Americorophium* spp. rarely occurred in benthic samples collected at the trap-net site, dense colonies of the amphipod could be seen in the larger adjacent tidal channels where the diel studies were conducted.

Estimated evacuation rates from 2005-2007 ranged from 0.30 to 0.41. Evacuation rates among different size classes of juvenile Chinook salmon were similar but consistently higher among salmon collected at Russian Island (0.37-0.41) than at Lord Island (0.30-0.34).

Estimated daily meal was smallest for the smallest size classes of salmon (30-59 mm), at 0.16-0.17 g d^{-1}, and greatest for the largest size classes (80-99 and 100-119 mm), at 1.33-174 g d^{-1} (Figure 44). Daily rations for foraging Chinook salmon were estimated at between 6.0 and 17.0% of body weight per day. These rates were equal to or considerably higher than those reported for juvenile Chinook salmon in riverine habitats (e.g., Sagar and Glova 1988).

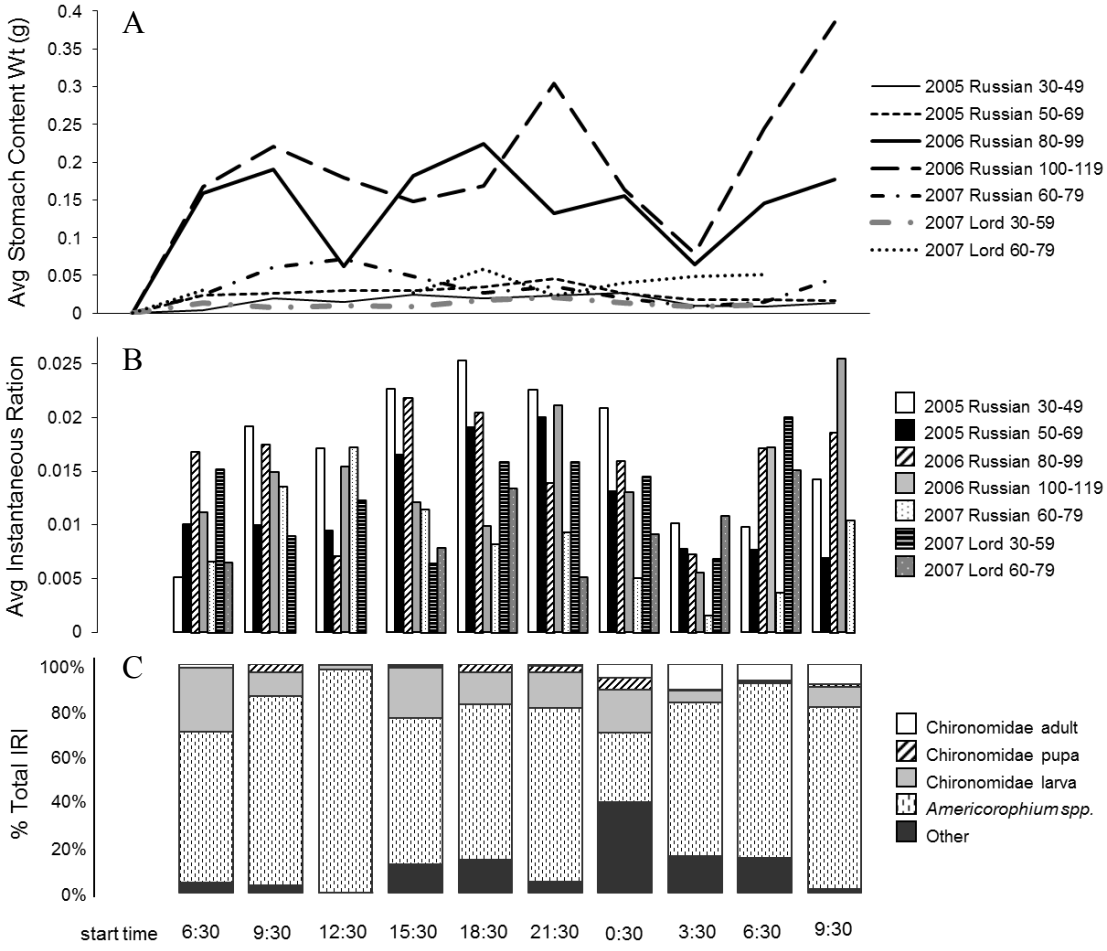


Figure 44. Summary of diel feeding results for juvenile Chinook salmon during consumption studies at Russian Island (2005-2007) and Lord Island (2007). Evacuation rates, daily meal, and daily ration are summarized by sampling date and salmon size class at the top. Figure panels compare diel changes in mean stomach content weights (A) and instantaneous rations (B) by size class. Diel diet composition as % total IRI (C) is shown for 50-69 mm Chinook salmon collected at Russian Island in 2005.

IV. PERFORMANCE METRICS: POPULATION STRUCTURE, LIFE HISTORY DIVERSITY, AND GROWTH

Genetic Stock Composition

Methods

We used microsatellite DNA loci standardized among several West Coast genetics laboratories (Seeb et al. 2007) to estimate the stock origins of Columbia River Basin and coastal populations of Chinook salmon. Fin tissues for DNA analyses were sampled from Chinook salmon captured at beach-seine and wetland sites from 2002-2006. Tissue storage and data collection methods are described in Teel et al. (2009). Proportional stock composition of mixed-stock samples was estimated using the likelihood model of Rannala and Mountain (1997) as implemented by the genetic stock identification program ONCOR (Kalinowski et al. 2007).

We compiled population baseline data from a multi-laboratory standardized genetic database for Chinook salmon (Seeb et al. 2007). Information and data sources for the 45 populations in our baseline are given in Appendix F. Allocations to individual baseline populations were summed to estimate the contributions of 11 regional stock groups (Appendix F). Regional stock groupings were based on the stock identification analysis described by Seeb et al. (2007) and on previous genetic studies of Chinook salmon in the Columbia River Basin (e.g., Waples et al. 2004; Myers et al. 2006).

For Chinook salmon from the interior Columbia River, groups identified using the microsatellite baseline included Snake River spring/summer and fall Chinook stocks and Middle and Upper Columbia River spring stocks from east of the Cascade Mountains. Also from the interior Columbia River were the Upper Columbia River summer/fall stock and a fall stock from the Deschutes River. The Upper Columbia River summer/fall stock group included summer-run populations in the upper Columbia River and upriver bright fall populations, including those in the Hanford Reach area.

Artificial propagation programs have significantly influenced the composition and spatial distribution of genetic stock groups in the Columbia River basin. For example, Hanford Reach fall stock was used to develop the run at Priest Rapids Hatchery. Priest Rapids stock is also used at several other hatcheries that release fish in upper Columbia River locations, as well as in lower river locations such as at Bonneville Hatchery (Appendix G). Other likely influences of artificial propagation include upriver bright fall Chinook spawning in several Columbia Gorge tributaries and in main-stem areas just below Bonneville Dam (Myers et al. 2006).

The Spring Creek Group is a tule fall stock originating from the Spring Creek National Fish Hatchery in the Columbia River Gorge area. This stock has been widely propagated throughout the lower Columbia River (Myers et al. 2006). Also in the lower Columbia River are the Willamette River spring and West Cascade spring and fall stock groups, which are comprised of fish originating from several tributaries and hatcheries (Myers et al. 2006).

Chinook salmon in the Columbia River also include individuals from the Rogue River stock, which was introduced to the Columbia River from southern Oregon in the 1980s as part of a continuing effort to enhance fisheries in off-channel areas (North et al. 2006). We therefore included Rogue River stock in our baseline dataset to identify fish descended from those stock transfers. We also included north Oregon and south Washington coastal stocks in our baseline data to estimate straying of coastal stocks from outside the Columbia River Basin into lower estuary beach-seine habitats.

Precision of the stock composition results was estimated by bootstrapping baseline and mixed-stock data from samples (100 times) as implemented by ONCOR (Kalinowski et al. 2007). We also used ONCOR to estimate the most likely stock group of origin of individual fish. These estimates were used in plots depicting stock-specific size-frequency distributions of fish grouped spatially and seasonally. We included only fish that had been assigned to a stock group with a minimum probability of 0.90 relative to other groups. A power analyses conducted on the baseline data had indicated that by excluding fish assigned with lower probabilities (approximately 25% of our estuary samples), we improve the overall accuracy of assignment and introduced only a small amount of assignment bias (data not shown).

Results

Stock Composition of Tidal Gradient Samples—Genetic analysis showed that fall run fish from the West Cascade fall and Spring Creek Group fall stocks dominated the beach-seine samples, contributing an estimated 51 and 34%, respectively (Table 16). In addition, estimates of stock composition indicated that all genetic stock groups contributed to our samples except for the Middle and Upper Columbia spring-run stocks. In estimates for nine of the stock groups, the lower bound of the confidence interval was greater than zero.

Table 16. Estimated proportional stock composition and 95% confidence intervals for 2,207 juvenile Chinook salmon sampled at beach seine sites 2002-2006. Confidence intervals are from 100 bootstrap resamplings of baseline and mixture genotypes.

Stock group	Stock code	Proportion estimated	95% CI
Snake Spring/Summer	Snake Sp	0.001	0.000-0.003
Snake Fall	Snake F	0.007	0.003-0.017
Middle/Upper Columbia River Spring	M/UCR Sp	0.000	0.000-0.003
Upper Columbia River Summer/Fall	UCR Su/F	0.061	0.054-0.080
Deschutes Fall	Deschutes F	0.004	0.001-0.013
Spring Creek Group Fall	SCG F	0.337	0.275-0.331
Willamette Spring	WR Sp	0.013	0.008-0.018
West Cascade Spring	WC Sp	0.027	0.031-0.065
West Cascade Fall	WC F	0.512	0.468-0.542
Rogue	Rogue	0.025	0.016-0.032
Coastal Fall	Coastal	0.014	0.009-0.023

We used stock assignments of individual fish to investigate the stock-specific size distributions of Chinook salmon sampled at beach-seine sites (Figure 45; Appendix Table F2). All size classes in the overall set of samples were represented among genetic assignments. Although some size classes of fry were underrepresented in the genetic sample, size distributions were similar between the genetic and total fish samples. Spatial and temporal patterns of the stock distributions are depicted in Figures 46 and 47.

Overall, the diversity of stocks increased over the salinity gradient from tidal freshwater sites to marine sites. Tidal freshwater sites were dominated by the West Cascade Fall group, while estuarine mixing and marine habitats had larger proportions of the Spring Creek Group Fall stock. Near-equal proportions of these two stocks were present in spring samples, whereas Spring Creek Group Fall fish were nearly absent in summer/fall samples. Although few samples from the winter period were analyzed, most were small fish from both stock groups. Other stock groups comprised smaller proportions of the samples and showed differing patterns of distribution.

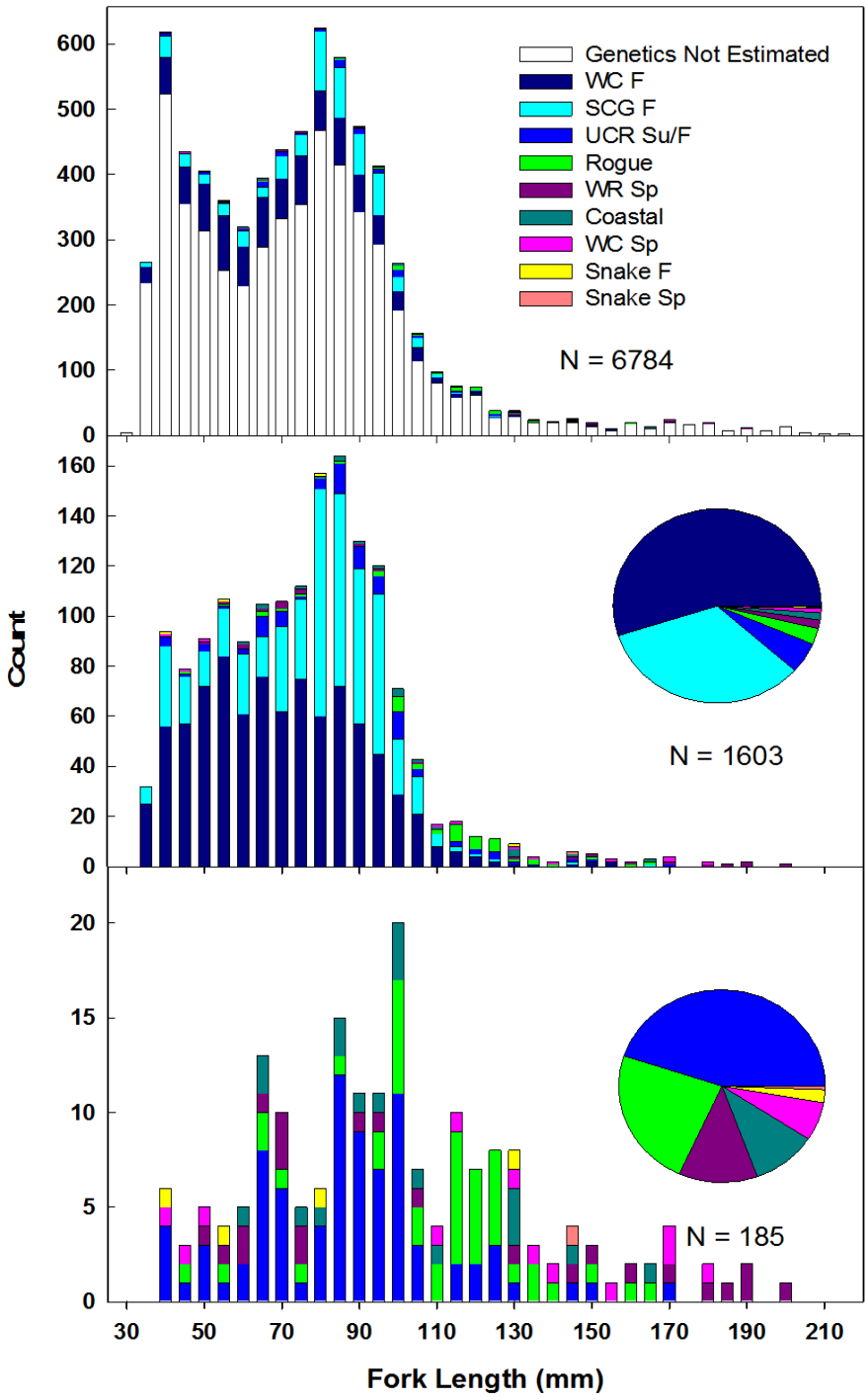


Figure 45. Genetic stock-of-origin of Chinook salmon sampled from tidal gradient (beach-seine) samples during 2002-2006. Bar graphs show length frequency distribution; pie graphs show proportional distribution. Upper panel shows all fish sampled, including those for which no stock origin was estimated. Middle panel shows major contributors, and lower panel minor contributors. Stock codes listed in Table 16.

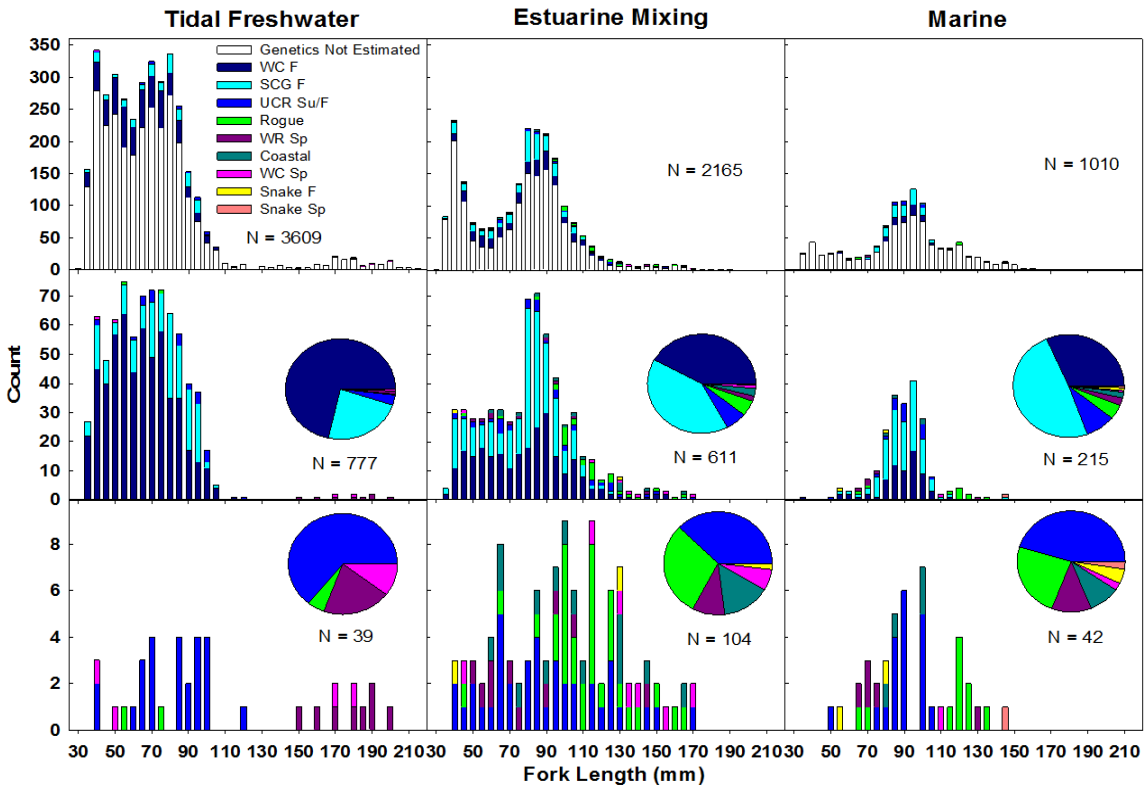


Figure 46. Genetic stock-of-origin of Chinook salmon sampled from beach seine sites at tidal freshwater, estuarine mixing and marine zones during 2002-2006. Bar graphs show length-frequency distribution; pie graphs show proportional distribution. Upper panel shows all fish sampled, including those for which no stock origin was estimated. Middle panel shows major contributors and lower panel minor contributors. Stock codes listed in Table 16.

The Upper Columbia summer/fall group was present in all spatial strata and was primarily estimated in samples from the summer/fall period. Willamette and West Cascade spring-run juveniles were captured mostly in spring and included both subyearling and yearling sized individuals. Small numbers of Rogue fish were found in the tidal freshwater, mixing, and marine habitats and throughout our sampling seasons. The abundance of Rogue stock increased in the mixing and marine zones, closer to the release location for the select off-channel fishery. Nearly all fish of Rogue origin were sampled at sites on the Oregon shoreline (data not shown). In collections from estuarine mixing and marine habitats, we also identified a small number of Chinook salmon juveniles from coastal sources outside of the Columbia River Basin.

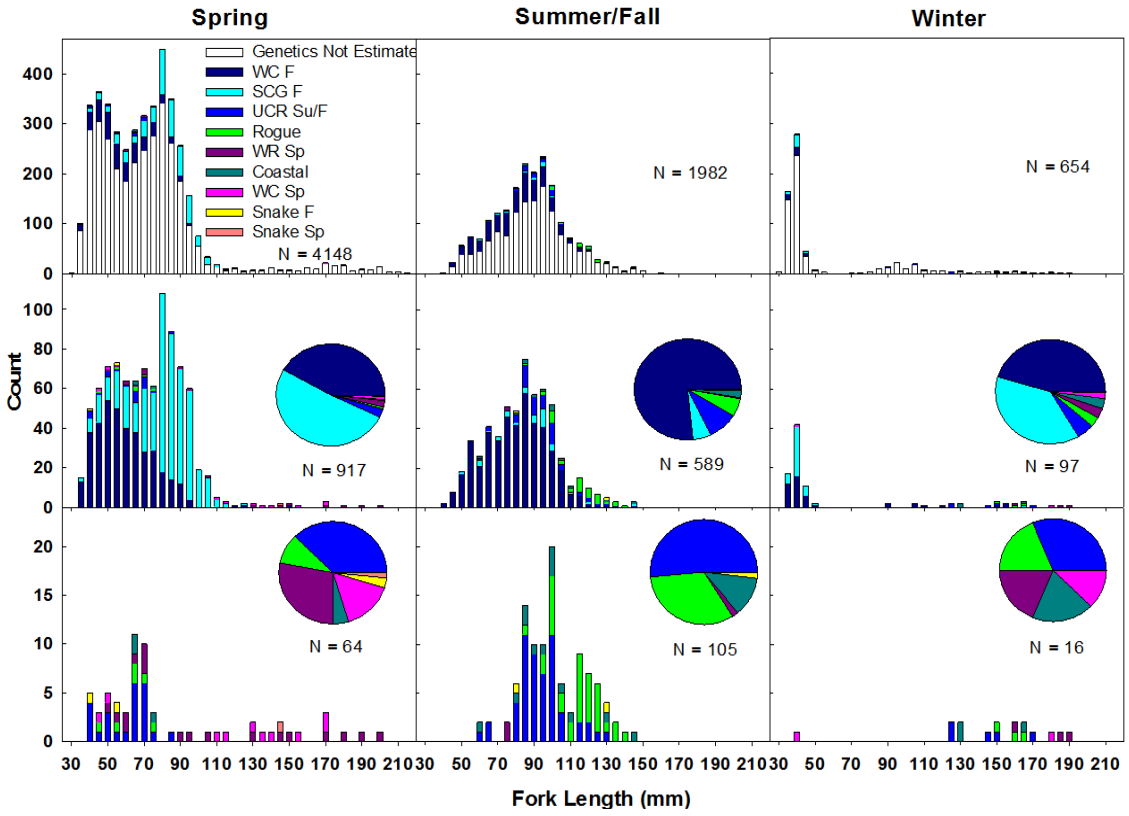


Figure 47. Genetic stock of origin in Chinook salmon sampled from beach-seine sites during spring, summer/fall, and winter. Bar graphs show length-frequency distribution; pie graphs show proportional distribution. Upper panel: all fish sampled, including those for which no stock origin was estimated. Middle panel: Major contributors. Bottom panel: minor contributors.

Stock Composition in Wetland Habitat Samples—Most Chinook salmon sampled in all wetland habitats were from the West Cascade fall stock group (55-78%; Table 17, Figure 48). The Spring Creek Group Fall stock was the second largest contributor to samples from all wetland sites (18-39%) except in samples from Lord Island, where the proportion of Upper Columbia River Summer/Fall stock was 12%. Fish from the Upper Columbia River Summer/Fall group were also present in samples from each of the other sites (3-6%). Small proportions of West Cascade Spring stock were estimated for six of the sites, representing all three habitat types (1-7%). Snake Fall, Deschutes Fall, and Willamette River Spring stocks also were represented in small proportions in some of the sample mixtures.

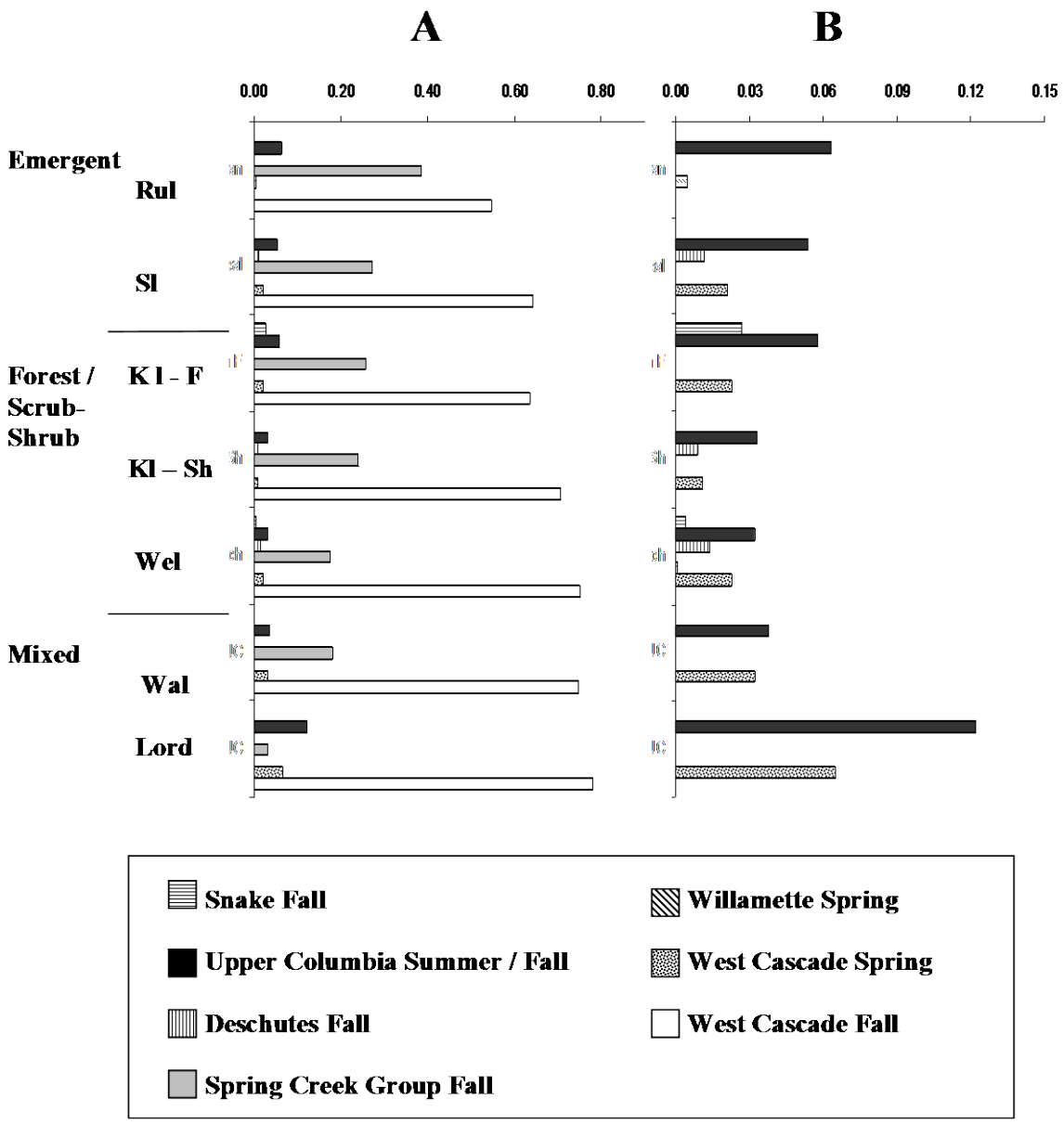


Figure 48. Estimated proportional stock composition of Chinook salmon sampled at seven estuarine wetland sites in 2002-2006. Estimates for seven stock groups are presented showing stock composition both with (panel A) and without (panel B) Spring Creek Group and West Cascade fall groups. Sample sizes and confidence intervals are given in Table 17.

Table 17. Estimated proportional stock composition and 95% confidence intervals for juvenile Chinook salmon sampled at estuarine wetland sites 2002-2006. The sample size (n) at each site is given. Confidence intervals (in parenthesis) are from 100 bootstrap resamplings of baseline and mixture genotypes.

	Emergent marsh		Forested and shrub			Mixed (emergent to forest)	
	Russian	Seal	Karlson Forested	Karlson Shrub	Welch Shrub	Wallace	Lord
Stock group	n = 474	n = 175	n = 75	n = 99	n = 337	n = 43	n = 55
Snake Spring/Summer	0.000 (0.000–0.005)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.006)	0.000 (0.000–0.029)	0.000 (0.000–0.000)
Snake Fall	0.000 (0.000–0.013)	0.000 (0.000–0.030)	0.027 (0.000–0.073)	0.000 (0.000–0.029)	0.004 (0.000–0.024)	0.000 (0.000–0.016)	0.000 (0.000–0.000)
Mid and Upper Columbia River Spring	0.000 (0.000–0.006)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.045)	0.000 (0.000–0.000)
Upper Columbia River Summer/Fall	0.063 (0.041–0.101)	0.054 (0.022–0.096)	0.058 (0.000–0.124)	0.033 (0.000–0.089)	0.032 (0.015–0.069)	0.038 (0.000–0.143)	0.122 (0.044–0.231)
Deschutes Fall	0.000 (0.000–0.014)	0.012 (0.000–0.025)	0.000 (0.000–0.028)	0.009 (0.000–0.031)	0.014 (0.000–0.032)	0.000 (0.000–0.075)	0.000 (0.000–0.038)
Spring Creek Group Fall	0.386 (0.293–0.409)	0.271 (0.190–0.329)	0.257 (0.135–0.337)	0.240 (0.124–0.313)	0.176 (0.086–0.184)	0.182 (0.062–0.297)	0.033 (0.000–0.098)
Willamette Spring	0.005 (0.000–0.011)	0.000 (0.000–0.010)	0.000 (0.000–0.000)	0.000 (0.000–0.013)	0.001 (0.000–0.007)	0.000 (0.000–0.021)	0.000 (0.000–0.000)
West Cascade Spring	0.000 (0.000–0.047)	0.021 (0.000–0.081)	0.023 (0.000–0.130)	0.011 (0.000–0.103)	0.023 (0.017–0.086)	0.032 (0.000–0.135)	0.065 (0.000–0.202)
West Cascade Fall	0.547 (0.482–0.606)	0.642 (0.539–0.725)	0.635 (0.496–0.746)	0.707 (0.570–0.798)	0.750 (0.666–0.805)	0.747 (0.494–0.826)	0.780 (0.558–0.893)
Rogue	0.000 (0.000–0.006)	0.000 (0.000–0.012)	0.000 (0.000–0.034)	0.000 (0.000–0.011)	0.000 (0.000–0.009)	0.000 (0.000–0.000)	0.000 (0.000–0.000)

The stocks estimated to comprise most of the samples from interior wetland channel sites at Lord and Wallace Islands were the same as those found at beach-seine sites on the main-stem sides of the islands (Table 18, Figure 49). The West Cascade Fall, Spring Creek Group Fall, and Upper Columbia River Summer/Fall stocks were found in channel and main-stem sites at both islands. In addition, West Cascade Spring fish were found in channel sites on both Lord (7%) and Wallace (2%) Islands.

Table 18. Estimated proportional stock composition (and 95% confidence interval) for juvenile Chinook salmon sampled at main-stem sites and interior-wetland channels on Lord and Wallace Islands from April through July 2006. Sample size (n) at each site is given; confidence intervals are from 100 bootstrap resamplings of baseline and mixture genotypes.

	Estimated stock composition			
	Lord Island		Wallace Island	
Stock group	Main stem n = 165	Channel n = 75	Main stem n = 107	Channel n = 55
Snake Spring/Summer	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.025)
Snake Fall	0.000 (0.000–0.009)	0.000 (0.000–0.027)	0.045 (0.000–0.110)	0.000 (0.000–0.038)
Mid and Upper Columbia River Spring	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.018)
Upper Columbia River Summer/Fall	0.058 (0.031–0.120)	0.155 (0.071–0.252)	0.061 (0.011–0.122)	0.063 (0.000–0.163)
Deschutes Fall	0.006 (0.000–0.022)	0.000 (0.000–0.036)	0.000 (0.000–0.034)	0.000 (0.000–0.052)
Spring Creek Group Fall	0.143 (0.074–0.195)	0.070 (0.013–0.179)	0.062 (0.014–0.118)	0.179 (0.056–0.347)
Willamette Spring	0.018 (0.000–0.043)	0.000 (0.000–0.000)	0.000 (0.000–0.000)	0.000 (0.000–0.000)
West Cascade Spring	0.009 (0.000–0.114)	0.069 (0.019–0.190)	0.000 (0.000–0.075)	0.020 (0.000–0.152)
West Cascade Fall	0.767 (0.628–0.796)	0.706 (0.501–0.805)	0.828 (0.688–0.858)	0.739 (0.476–0.822)
Rogue	0.000 (0.000–0.014)	0.000 (0.000–0.024)	0.004 (0.000–0.028)	0.000 (0.000–0.000)

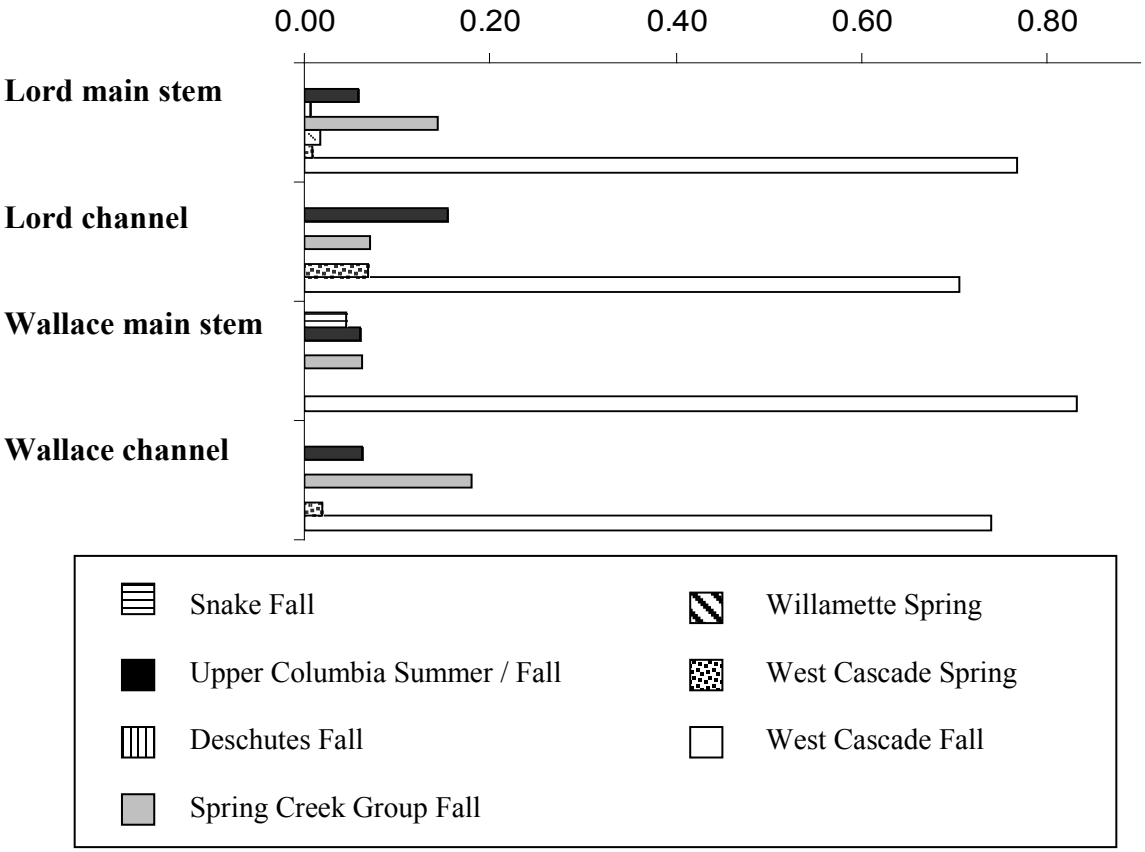


Figure 49. Estimated proportional stock composition of Chinook salmon sampled at estuary main stem and interior-wetland channels on Lord and Wallace Islands, April-July 2006. Sample sizes and confidence intervals are given in Table 18.

Otolith Chemistry and Daily Growth Increments

Methods

We analyzed otolith chemistry (n = 411) and growth increments (n = 91) to quantify estuary residency, time and size at estuary entry, and growth of juvenile Chinook salmon in the lower Columbia River estuary in 2003-2005. All otoliths were collected from beach-seine samples taken for the tidal gradient study (see Part II, methods). Samples were obtained from the saline portion of the lower estuary, primarily from Pt. Adams Beach, and from a single tidal freshwater site at Lower Elochoman Slough (see Figure 10). During periods of low beach-seine catch at Pt. Adams Beach, we augmented collections from the saline portion of the estuary with samples from Pt. Ellice, Clatsop Spit, and West Sand Island.

Otoliths were prepared for chemical and growth-increment analyses by thin-sectioning of the sagittal plane according to the preparation methods described in Campbell (2010). Chemical analyses for all otolith samples were completed at Oregon State University using a New Wave[†] DUV 193-nm ArF laser linked to a Thermal Elemental PQ Excell quadropole inductively coupled plasma mass spectrometer (LA-ICPMS).

Raw counts of Sr and Ca were plotted, and points of interest were recorded for laser transect start and end points and for the Sr inflection point, defined visually as the region immediately prior to a rapid increase in Sr. The inflection point was assumed to correspond to contact with salinity, since no such Sr increases have been found in other Columbia River otolith samples obtained from tidal freshwater sites (Campbell 2010). This interpretation was consistent with results from published laboratory (Zimmerman 2005) and field experiments (Volk et al. 2010).

Points of interest (POI) in chemical output were related to location on the otolith by the equation (similar to Brenkman et al. 2006, Volk et al. 2010):

$$otolith_{POI}\ (\mu m) = \left(\frac{laser_{POI}\ (ms) - laser_{start}\ (ms)}{1000}\right) * 5\frac{\mu m}{sec}$$

[†] Reference to trade names does not imply endorsement by the National Marine Fisheries Service, NOAA.

We used Campana's (1990) proportional biological intercept (BI) method to back-calculate fish size at a given otolith size based on the formula:

$$L_a = L_c + (O_a - O_c)(L_c - L_o)(O_c - O_o)^{-1}$$

where L_c and O_c are the size of the fish and otolith at the time of capture; L_o and O_o are the biological intercept for fish length and otolith size, respectively; and L_a and O_a are the size of the fish and otolith at a particular point of interest, such as Sr inflection.

All estimates of estuary entry and residency using Sr as an indicator were applied to the saline portion of the estuary only and did not account for residency in the extended tidal freshwater reaches of the estuary. We also could not account for any additional time a fish might have remained in the lower estuary had it not been captured and sacrificed for otolith analysis. Thus, all estuary residence times reported here are minimum values.

We measured and counted otolith daily growth increments (DGI) from the otolith edge parallel to the LA-ICPMS chemical transect using light microscopy as reported in Campbell (2010). Where daily growth increments were not discernable due to otolith preparation or clarity, an average increment width of 2.58 µm (mean increment width of all measurable otoliths in 2003) was used to estimate residence time. Growth in the estuary was estimated by measuring otolith daily growth increments for the previous 30 d of estuary residence and then averaged for selected time periods (January-April, May-August, and September-December) to produce a mean increment width (MIW) by season and year.

Results

Estuary residency—We analyzed approximately 100 otolith samples per year from Pt. Adams Beach and surrounding brackish water sites, and approximately 50 per year from the freshwater site at Lower Elochoman Slough (Table 19). The percentage of samples with an obvious inflection and elevated Sr levels, indicating contact with salt water, varied at Pt. Adams Beach and was generally lower during months of peak migration. In samples collected at Lower Elochoman Slough, we rarely found a marker for elevated Sr levels.

Most otoliths of juvenile Chinook salmon collected at Pt. Adams Beach or at nearby lower estuary sites showed evidence of a rapid increase in Sr, indicating contact with salinity. Proportions of otoliths that showed an increase in Sr were 72% in 2003, 68% in 2004, and 86% in 2005. Estimated residence duration for all juvenile Chinook salmon collected in the lower estuary ranged 0-176 d. Residence duration (count data) was normalized by a square-root transformation, and a one-way analysis of variance

(ANOVA) indicated a significant difference between years ($F_{2,215} = 22.6$, $P < 0.001$). Generally residence times were greater in 2004 (mean ± SD, 67.3 ±43.5 d) than in 2003 (53.6 ±41.0 d; $P = 0.021$) or 2005 (29.7 ±23.7 d; $P < 0.001$; Holm-Sidak multiple comparison test). Residence times in 2003 were greater than those in 2005 ($P < 0.001$). Proportions of juvenile Chinook salmon with more than 30 d of estuary residence were 55% in 2003, 51% in 2004, and 30% in 2005.

Table 19. Otoliths examined at Point Adams Beach (PAB) and Lower Elochoman Slough (LES) for elevated Sr indicative of entrance into the saline portion of the estuary.

		2003				2004				2005			
		PAB *		LES		PAB		LES		PAB *		LES	
Month		N	% with Sr signal	N	% with Sr signal	N	% with Sr signal	N	% with Sr signal	N	% with Sr signal	N	% with Sr signal
January	**5**					4	0					1	0
February	**23**					9	44			8	100	6	0
March	**22**									17	94	5	0
April	**52**	6	100			11	100	9	0	20	75	6	0
May	**90**	31	77	8	0	16	31	10	0	20	70	5	0
June	**35**	10	60			10	90	10	0			5	0
July	**73**	29	62	12	0	3	100	10	0	14	79	5	0
August	**40**	3	100	1	0	11	73	4	0	16	88	5	0
September	**30**	9	89			9	89	8	0	4	100		
October	**22**	6	100	9	0	6	100	1	0				
November	**19**					14	86	5	0				
December	**0**												
N	**411**	**94**		**30**		**93**		**57**		**99**		**38**	

* In 2003 and 2005, samples from other lower estuary sites were included with those from PAB.

Size at estuary entry—For all Chinook salmon sampled for otolith microchemistry from 2003-2005, back-calculated sizes at estuary entrance ranged 34-178 mm. We estimated that nearly half of all samples collected in 2004 and 2005 were composed of individuals that had entered the estuary at sizes below 60 mm (Table 20). This total included fry that had recently entered the estuary (and therefore showed no Sr signal), as well as earlier migrants with back-calculated sizes at entry of less than 60 mm. Larger fish in the size ranges of 61-90 mm and over 90 mm made up a smaller, but significant proportion of all individuals that entered the estuary in 2004 and 2005 (Table 20). A larger proportion of fish 61-90 mm was estimated in the 2003 sample collection relative to the other years. However, because the 2003 otolith samples were selected to represent individuals with and without scale checks as part of a related scale chemistry study (Campbell 2010), the 2003 results were likely biased for larger fish. Scale formation does not occur until fish are at least ~38-42 mm, and in 2003, all individuals chosen for scale and otolith microchemistry comparisons were 50 mm or larger.

Table 20. Proportion of juvenile Chinook salmon by size class entering the Columbia River estuary each year. Size at entry in 2003 could be biased toward larger individuals because otolith samples were chosen for a scale chemistry study that targeted fish of 50 mm or larger.

Size at estuary entrance	2003	2004	2005
	(n = 101)	(n = 93)	(n = 98)
<60	0.17	0.41	0.53
61-90	0.64	0.37	0.23
>91	0.19	0.23	0.23

Approximately 32 and 45% of the Chinook salmon collected in the beach seine in 2004 and 2005, respectively, entered the saline portion of the estuary at sizes smaller than 45 mm (i.e., 0-3 weeks post emergence). Despite the sampling bias in 2003, we still estimated that 13% of the individuals in the sample had entered at sizes smaller than 45 mm. Fry with unabsorbed yolk were present occasionally, indicating that some individuals moved into the estuary immediately after emergence. Pooling all sampling years and estimating back-calculated size and season of estuary entrance indicated that progressively larger fish entered and resided in the estuary during the rearing season (Table 21). Small migrants (< 60 mm) that dominated early in the year were composed primarily of newly emerged fry, indicating that these smallest individuals survived and contributed to larger sizes classes (Table 21).

Table 21. Estimated mean residency in days (± 1 SD) of juvenile Chinook salmon for all years combined (2003-2005) by estimated size and season at first entry into the estuary (N = 218). Bold indicates the dominate size class by season. The < 45 mm size class is a sub-group of the < 60 mm.

	Jan-April			May-Aug			Sept-Dec		
	Residency (days)	N	%	Residency (days)	N	%	Residency (days)	N	%
< 45 mm	52 (36.2)	60	21	79 (39.1)	7	2			0
< 60 mm	**54 (34.7)**	**75**	**26**	59 (44.6)	14	5			0
61-90 mm	50 (35.5)	28	10	**46 (45.8)**	**56**	**19**	33 (n/a)	1	0
> 90 mm	31 (13.6)	11	4	45 (44.2)	34	12	**20 (16.1)**	**4**	**1**

A significant negative relationship between mean FL and estuary residence time was apparent for spring and summer months (May-August, in all years) when a wide range of size classes and estuary residence times were represented in the estuary

population (Figure 50). During May-August, FL at estuary entrance explained 46% of the variation in residency in 2003 ($P < 0.001$), 47% in 2004 ($P < 0.001$), and 35% in 2005 ($P < 0.001$). Except in 2003, no significant relationship between size and residency was evident during January-April and September-December time periods ($P > 0.05$), when the size range of the estuary population was relatively narrow.

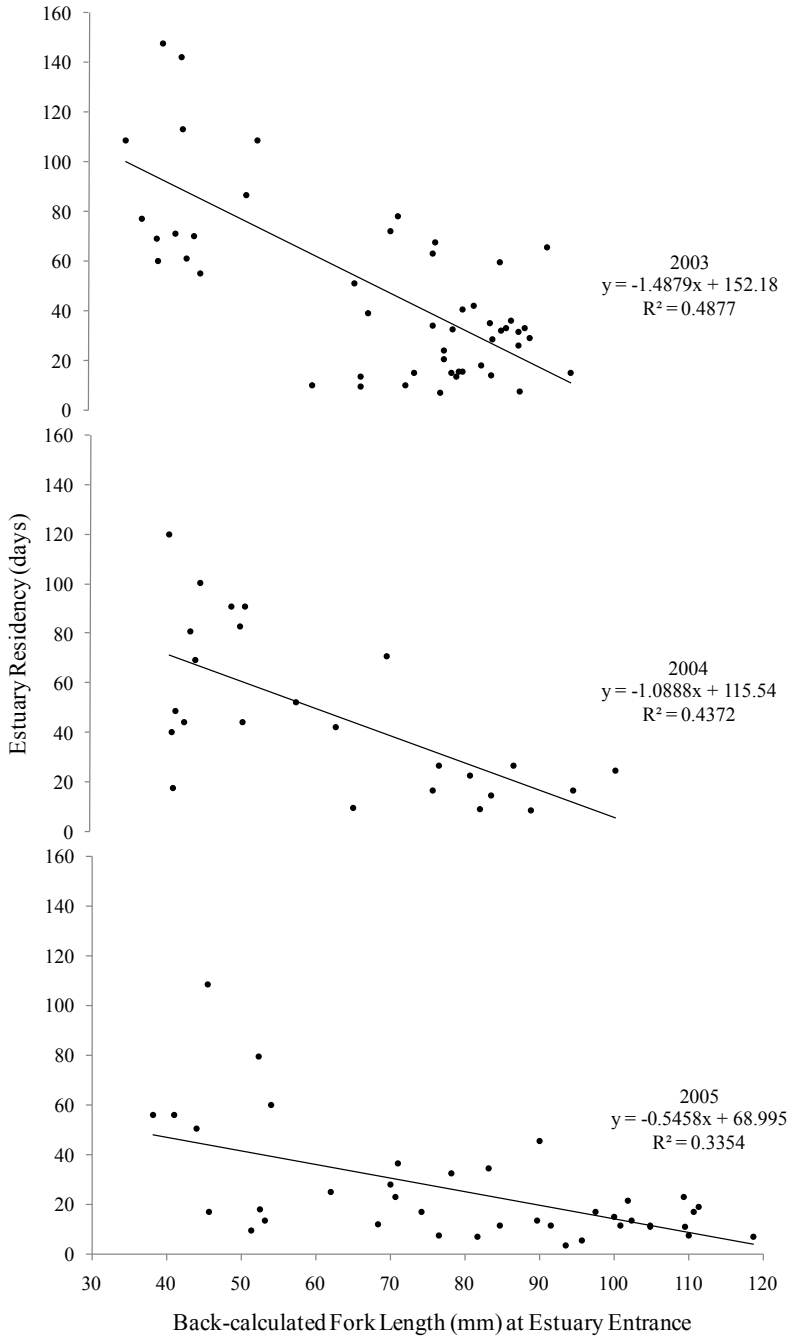


Figure 50. Relationship between back-calculated FL at estuary entrance and the residence time of juvenile Chinook salmon in the Columbia River estuary during May-August, 2003-2005.

Estuary Growth—Estimated mean increment widths (MIW) of Chinook salmon otoliths did not differ significantly among seasons within each year in a one-way ANOVA ($P > 0.05$). Comparison of seasonal growth periods between years revealed a significant difference only between 2003 and 2005 (May-August, $F_{2, 39} = 5.47$, $P = 0.008$; Table 22). However, a two-way ANOVA indicated a significant effect of season (January-April, May-August, and September-December) on growth (MIW) when accounting for year ($F_{2,82} = 8.0$, $P < 0.001$). In general, fish that entered the estuary in May-August (3.08 µm ± 0.46 µm) grew more rapidly than fish entering in January-April (2.36 µm ± 0.39 µm; $P < 0.001$; Holm-Sidak multiple comparison test). No effects of year on growth ($F_{2,82} = 2.2$, $P = 0.116$) or of season and year on growth ($F_{4,82} = 0.584$, $P = 0.675$) were detected.

Table 22. Mean otolith increment widths of juvenile Chinook salmon by year and time period and estimated growth rates in the Columbia River estuary, 2003-2005.

Year	Time of Capture	N	Mean increment width	SD	Growth rate (mm d^{-1})	SD	Statistically significant with:
2003	Jan-April	3	2.13	0.17	0.35	0.06	none
	May-Aug	22	2.57	0.52	0.36	0.1	2005 May-Aug
	Sept-Dec	7	2.57	0.72	0.39	0.07	none
2004	Jan-April	5	2.47	0.16	0.48	0.05	none
	May-Aug	7	3.07	0.56	0.4	0.1	none
	Sept-Dec	7	2.67	0.49	0.43	0.09	none
2005	Jan-April	26	2.37	0.43	0.36	0.12	2005 May-Aug
	May-Aug	13	3.09	0.43	0.49	0.09	2003 May-Aug
	Sept-Dec	1	2.78	n/a	0.43	n/a	none

The estimated mean growth rate of juvenile Chinook salmon during estuary residence was approximately 0.4 mm d^{-1} and ranged from 0.11 to 0.67 mm d^{-1}. Growth rates generally increased from January (0.19 mm d^{-1}) and February (0.36 mm d^{-1}) through the spring and summer, reaching a peak in August (0.55 mm d^{-1}). These results indicated that estuary growth rate varied among seasons, and to a lesser extent among years.

Genetic Composition of Estuary Residents—We determined the genetic composition of all Chinook salmon that were classified as estuary residents from otolith samples collected at Pt. Adams Beach in 2004 and 2005 (see Table 19). In both years, the Spring Creek Group Fall and West Cascades Fall groups (combined) accounted for ~80% of estuary-resident salmon (Figure 51). Upper Columbia River Summer/Fall and Rogue stock groups comprised significant proportions of the estuary-resident fish

collected in 2004 and 2005, respectively. In all but a few stock groups with very low sample sizes, mean estuary residence time ranged from one month to several months. Spring Creek Group Fall and West Cascade Fall groups—both lower Columbia River Fall Chinook stocks—entered the estuary at mean sizes ~60 mm FL or less, appreciably smaller than the average size at entry of fish from other stock groups .

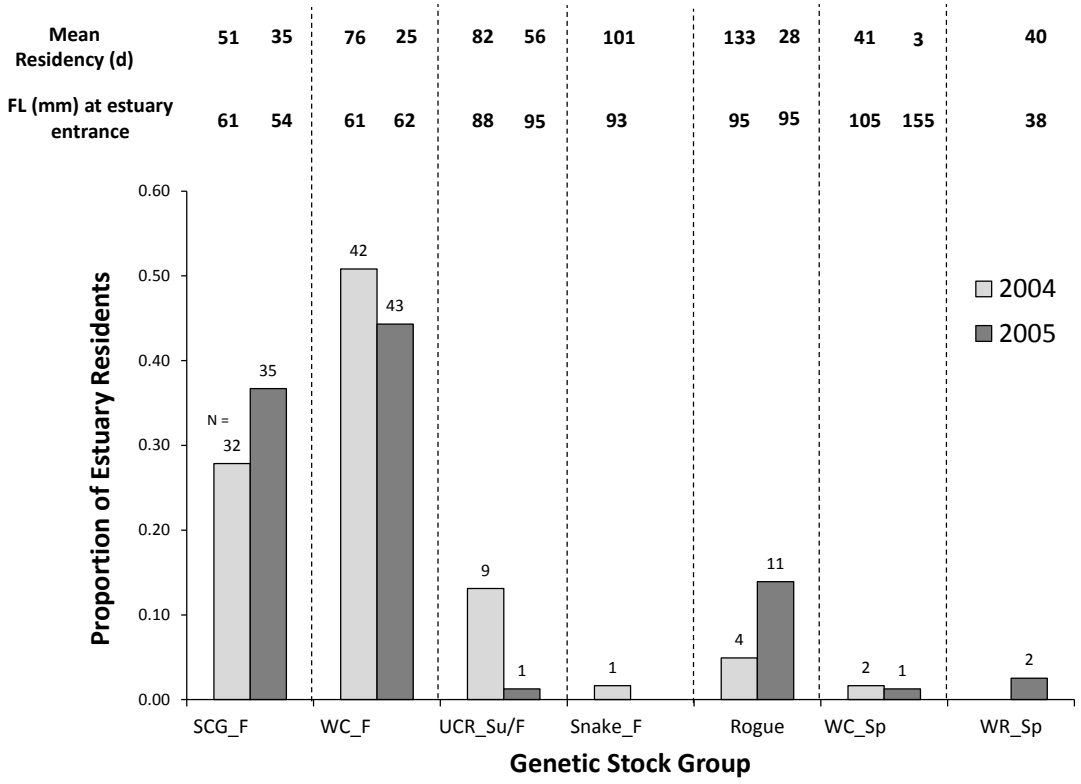


Figure 51. Life history attributes and genetic affiliations of estuary-resident Chinook salmon collected at Point Adams Beach, 2004-2005. The genetic composition of all salmon that were classified as estuary residents is plotted for each annual otolith sample. The mean residence times (d) and mean FL (mm) at estuary entrance of all estuary-resident salmon are listed above by genetic stock group and year (n = 90, 2004; n = 92, 2005).

Residency and Performance in an Emergent Wetland Habitat

Methods

We estimated juvenile Chinook salmon residence times and growth rates within selected channels of the Russian Island emergent marsh during two study periods: 3 April-25 May 2006 and 4 May-18 August 2008. Juvenile Chinook salmon were collected with a 3 × 38-m variable-mesh bag seine (10.0- and 6.3-mm wings, 4.8-mm bag) in three areas of Russian Island (Figure 52). Areas designated as A and B in Figure 52 were sampled in 2006, and area C was sampled in 2008. The study began with a tagging phase using passive integrated transponder (PIT) tags. The tagging phase lasted 8 days in 2006 and 6 days in 2008, and was followed by a recapture phase in both years.

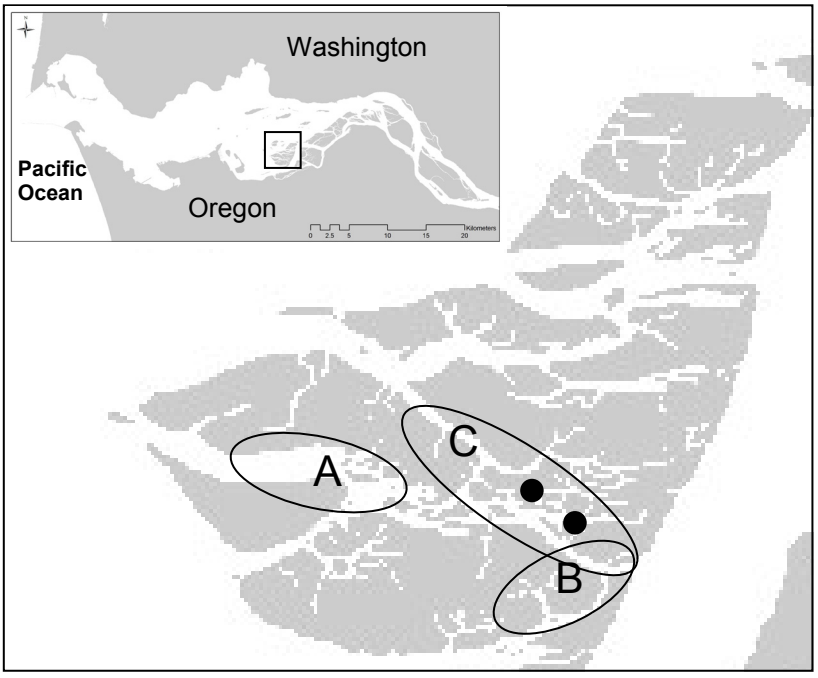


Figure 52. Map of Russian Island sample areas. Black dots indicate locations of PIT tag antenna arrays deployed in 2008.

During the tagging phase, each captured fish was inspected to ensure that no prior marks or tags were present on any individuals used in the residency study. In 2006, we marked all Chinook salmon less than 55 mm FL with a distinct color corresponding to the day of marking and release. Color marks of acrylic paint were applied to the base of the caudal fin with a jet inoculator. We also clipped the upper or lower lobe of the caudal fin

to provide an additional external mark. For Chinook salmon 55 mm or larger, we injected a 12-mm full-duplex PIT tag into the body cavity, providing an individual code for each fish. In 2008, we used PIT tags exclusively to mark individual fish, and we tagged only Chinook salmon of at least 60 mm FL. All fish were measured and weighed after marking or tagging, held for recovery, and released near the area of initial capture. We retained a tissue sample for genetic analysis from a subsample of fish collected each day of the marking phase.

When a PIT-tagged fish was recaptured during the marking phase, the tag code was recorded, and the fish was measured and weighed, held for recovery, and released. If a batch-marked individual (i.e., marked with acrylic paint) was recaptured during the marking phase, the batch code was recorded, and the fish was marked with a new batch code for the current day, measured and weighed, held for recovery, and released.

During the recapture phase of the study in 2006 and 2008, we collected juvenile Chinook salmon with a 38-m bag seine in the same areas of Russian Island that were sampled during the marking phase. Juvenile Chinook salmon were inspected for marks and tags, measured and weighed, held for recovery, and released. Sampling continued for two consecutive sampling days after the last marked or tagged fish was recaptured.

In 2008, we added a passive recapture element to the study design by deploying PIT tag detection arrays within each of two small secondary channels (Figure 52, black dots in area C). Each array consisted of two parallel groups of three antennas (approximately 4 m apart) that spanned the entire width of each channel. These parallel arrays provided information about the directional movement of tagged individuals entering or leaving each study channel. Each six-antenna array in each channel was connected to a single multiplex transceiver (Destron Fearing FS1001M). We monitored PIT-tagged fish with these antenna arrays from 4 May to 18 August. During the marking phase, detection efficiency of the antenna arrays was measured by releasing 20 PIT-tagged Chinook salmon upstream from the arrays in each channel on each of the 6 days of marking.

The PIT-tag detection arrays allowed us to measure fine-scale metrics, including residence time within each secondary channel and patterns of entry and exit relative to water level or other variables at the time of detection. Residence time was measured by comparing the latest recapture or detection date to the date of initial release. These measurements were conservative because we did not know how long each fish had remained in the area prior to marking or tagging.

We estimated instantaneous growth rates for PIT-tagged fish that were recaptured with the bag seine from the equation:

$$G = [(\log_e FL_2 - \log_e FL_1)/(t_2 - t_1)] \times 100;$$

where t_1 is the starting time, t_2 is the ending time, and FL_1 and FL_2 are the respective fork lengths at those times. We restricted these calculations to fish that resided in the channel for more than one day to account for precision errors in length measurements. We also looked for correlations between growth rate and residence time.

Results

In 2006 we collected nearly 900 juvenile Chinook salmon during the Russian Island residency study. Genetic analysis revealed that the overwhelming majority of individuals were Spring Creek group fall stock (Appendix Table F2). We batch-marked 324 (mean FL 47 mm) and PIT tagged 574 individual Chinook salmon (mean FL 81 mm). Twenty-two percent of the marked/tagged fish were recaptured during subsequent sampling (Table 23).

In 2008 we PIT tagged 688 juvenile Chinook salmon (mean FL 92 mm), comprised primarily of the Spring Creek Group Fall stock. Thirty percent of the fish marked in 2008 were either recaptured with the bag seine or detected by antenna arrays in one of the secondary channels (detection efficiency ~96%; Table 23).

Table 23. Summary of Chinook salmon batch-marked/tagged and recaptured for Wetland-channel residency study.

Year	Mark or tag method	Number marked or tagged	Average fork length (mm)	Number recaptured	Recovery rate (%)
2006	Paint/clip (<55 mm FL)	324	47	71	21.9
2006	PIT tag (≥55 mm FL)	574	81	128	22.3
2008	PIT tag (≥60 mm FL)	688	92	207	30.1

Despite differences in the sampling areas and mark-recapture methods during the 2006 and 2008 studies, residence time estimates for juvenile salmon in the Russian Island emergent wetland were similar in the two years. In 2006, average residence time was 7 d (median 5 d) for juvenile Chinook salmon in areas A and B (Figure 53), with no difference in residence time between batch-marked and PIT-tagged fish. In 2008, average residence time of all fish sampled in area C (i.e., captured in the beach seine or monitored at one of the antenna arrays) was 5 d (median 2 d). Maximum residence time was 27 d for PIT-tagged fish and 34 d for batch-marked fish in 2006 compared with 26 d for PIT-tagged fish in 2008. Of all juvenile Chinook salmon recaptured in both years, 37% resided for at least 1 week, 14% resided for at least 2 weeks, and 5% resided for at least 3 weeks (Figure 53). We found no correlation between initial fork length and residence time.

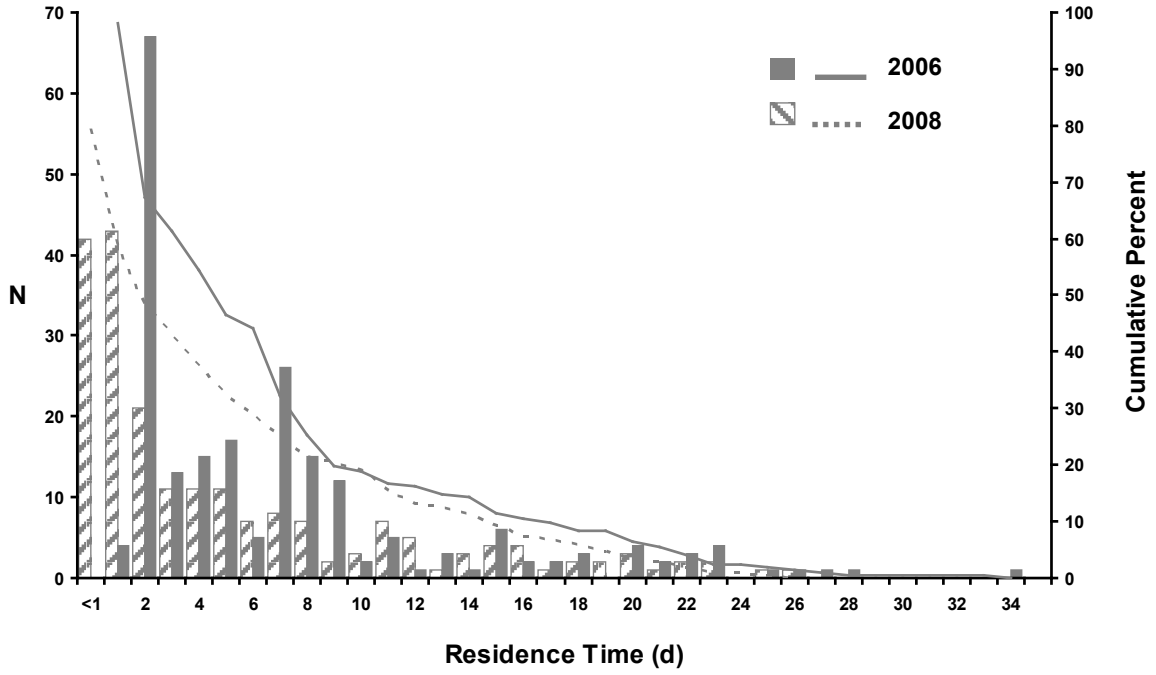


Figure 53. Histogram of residence time in days of subyearling Chinook salmon and cumulative percent of recaptured fish for 2006 (solid bar and line) and 2008 (dashed bar and line).

Detection data from the antenna arrays and water level measurements from sensors placed in the secondary channels enabled assessment of juvenile Chinook salmon behavior and fine-scale movements. The minimum depth at which Chinook salmon were detected in the channels was 0.5 m. This measurement was consistent for entry to and exit from the secondary channels. The vast majority of juvenile Chinook salmon were

detected on more than one antenna. Only 17 of 251 detections were unique. The sequence of detections for most fish indicated that juvenile Chinook salmon entered secondary channels at the mouth, moved upstream past both parallel arrays, remained in the channel for a length of time, and then passed both arrays during exit through the channel mouth.

On average, juvenile Chinook salmon remained in secondary channels for 2.5 h. The maximum time spent in a channel was 18.5 h, far longer than the usual tidal cycle should allow. Long residence times coincided with neap tides when sufficient water was ponded in the study channels for fish to remain even after the tide receded (Figure 54).

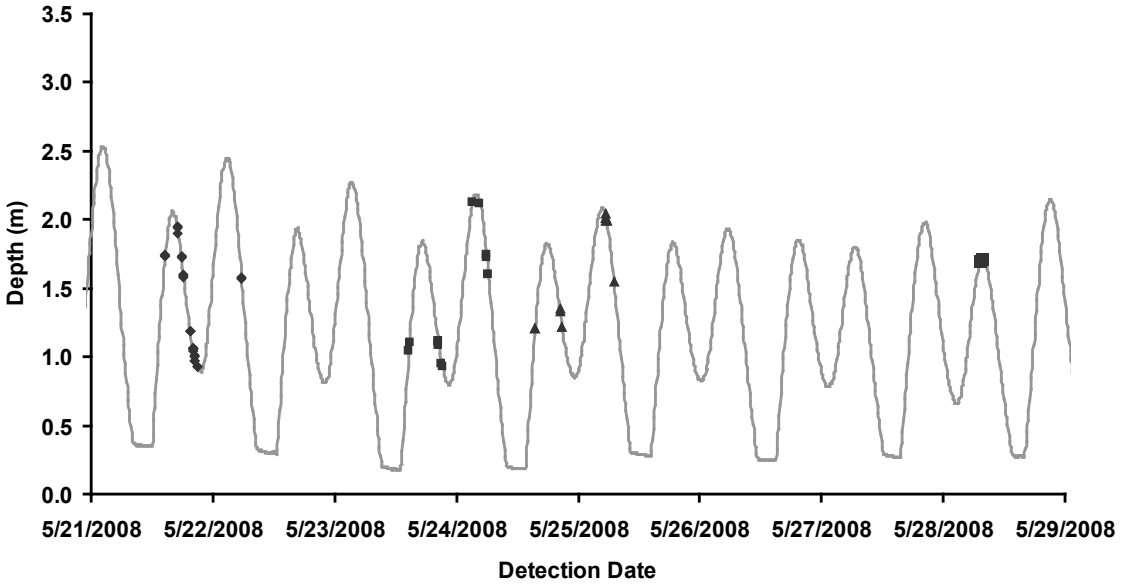


Figure 54. Example of detection data and water level data where an individual Chinook salmon remained in the secondary channel for three higher low tide cycles.

Figure 54 demonstrates intermittent use of the secondary channel during four separate detection events of an individual fish. This pattern may indicate salmon fidelity to a habitat complex larger than the scale of the local habitats that we monitored with nets and stationary PIT detection arrays. Further evidence of such fidelity was shown in 2009, when a subyearling Chinook salmon released from Bonneville Dam was detected at our Russian Island antenna arrays 10 times during a 51-d interval. These results depict a significant period of estuary residency and considerable fidelity to the large emergent-wetland habitat complex at Russian Island.

Tagged individuals recaptured in Russian Island marsh grew at similar rates during the two study periods. Average instantaneous growth rates of Chinook salmon were 0.67 mm d^{-1} in 2006 and 0.60 mm d^{-1} in 2008. Combining data for both years, fish with longer residence times tended to have higher growth rates, although the two variables were not significantly correlated. Individuals that resided for more than one but less than 15 d had an average instantaneous growth rate of 0.65 mm d^{-1}, whereas individuals that resided longer than 15 d had an average instantaneous growth rate of 0.82 mm d^{-1}.

Juvenile salmon benefitted from marsh residency as indicated by their positive growth. Fish residing in the estuary more than 1 but less than 2 weeks grew an average of 5.1 mm FL; individuals that resided more than 2 weeks grew an average of 13.8 mm. Over the entire study period, there was a positive correlation between juvenile salmon residence time in the marsh and increased fork length ($R^2 = 0.80$, $P < 0.0001$, Figure 55).

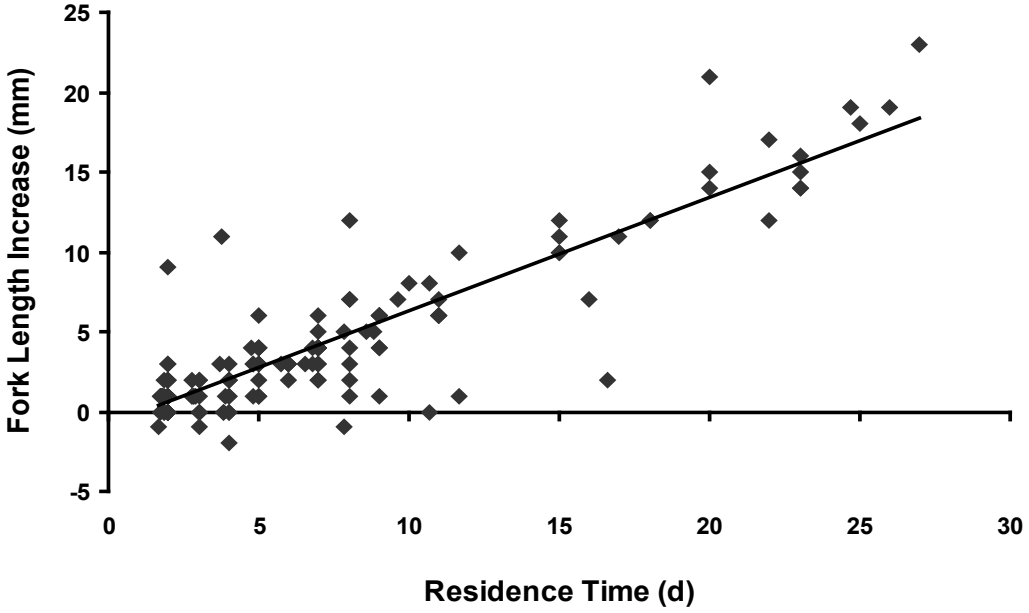


Figure 55. Change in fork length vs. residence time. Juvenile Chinook salmon remaining in marsh habitat continued to experience growth throughout their residency.

In addition to recapturing individual fish tagged for this study, we recaptured a total of 26 Chinook salmon that had been tagged during other Columbia River studies, either as beach-seine catch or as detections on PIT antennas. Nine of these fish had been released downstream from Bonneville Dam as part of an estuary avian predation study. The other 17 fish had been released at Bonneville Dam or Spring Creek National Fish Hatchery (upstream from Bonneville Dam). For individuals released at Bonneville Dam, travel time to our antenna arrays averaged 41 d and ranged 24-61 d. For individuals released at Spring Creek NFH, travel times averaged 37 d and ranged 21-79 d. These results indicated that Chinook salmon detected at the Russian Island antenna arrays moved slowly through the estuary before entering the ocean.

Beyond the final reporting period (2002-2008) for this contract research, we have continued to monitor upriver PIT tags at Russian Island. For example, in 2009 we detected 46 individual fish tagged and released from 14 different upriver locations. Distance from release ranged from rkm 47 (Blind Slough) to rkm 1,389 (Squaw Creek, Salmon River, ID). Figure 56 summarizes detections by release location/project in 2009.

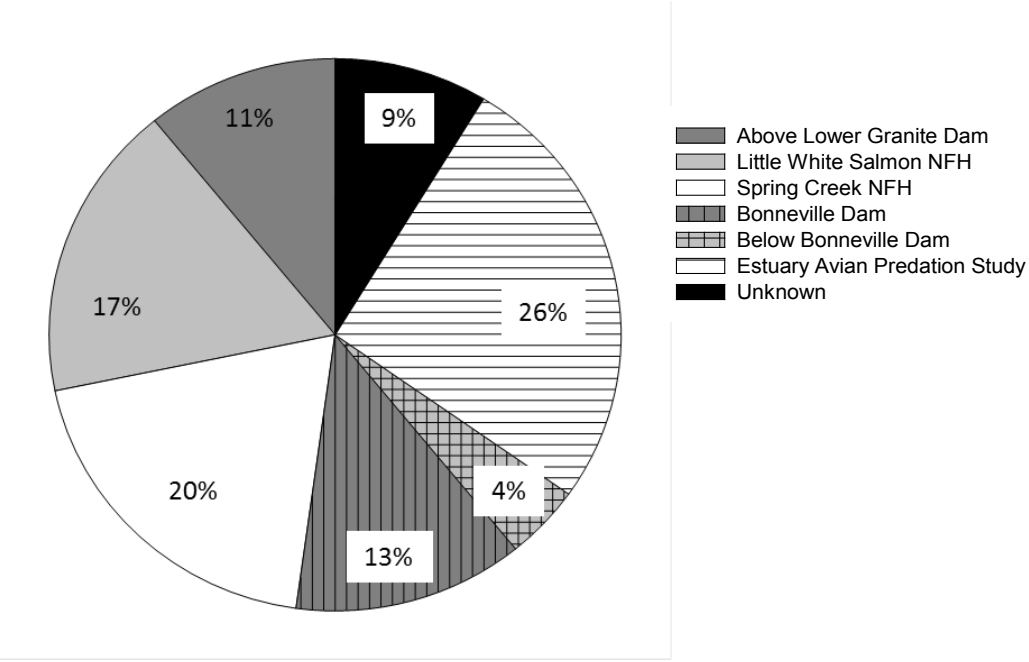


Figure 56. Proportion by release area or project of PIT-tagged fish detected at Russian Island antenna arrays in 2009.

CONCLUSIONS AND RECOMMENDATIONS

Our studies in the lower 100 km of the Columbia River estuary quantified historical habitat changes and provided new information about contemporary abundance patterns, life histories, and habitat associations of Chinook salmon. Habitat use and residence times of juvenile salmon varied with fish size, and shallow emergent marsh, scrub-shrub wetlands, and forested wetlands were important rearing areas for the smallest size classes. Subyearlings from all Evolutionarily Significant Units (ESUs) except spring-run groups from the interior Columbia River basin were represented in the lower estuary. However, two subyearling genetic stock groups dominated the lower-estuary trap-net and beach-seine collections: these were the Spring Creek Group Fall and West Cascade Fall stocks, both from the Lower Columbia River Chinook Salmon ESU.

The conceptual framework for this research defined salmon performance in the estuary as the product of three factors: habitat opportunity, habitat capacity, and the structure/life histories of source populations (Simenstad and Cordell 2000; Bottom et al. 2005b). Our 2002-2008 survey results provided empirical data to support this framework by quantifying Chinook salmon performance in terms of temporal abundance, life history and stock-group diversity, foraging success, and growth, as well as by quantifying the relationships between stock groups and one or more of these factors. The surveys provided new information about the estuarine habitat associations of juvenile salmon, and historical data were analyzed to estimate changes in habitat conditions, salmon life histories, and food webs. Below we enumerate the major conclusions drawn from these surveys and analyses.

Historical Change in Estuarine Habitat Opportunity

1. Extensive wetland loss has substantially decreased the quantity and quality of wetland habitats that support salmonid food webs and provide off-channel rearing areas for subyearling migrants with estuary-resident life histories.

Rearing opportunities for juvenile salmon have decreased in the estuary over the last century. We concluded that diking, filling, and other development activities have decreased the total area of tidal wetlands by more than 50% in the lower Columbia River estuary (Bottom et al. 2008), consistent with the previous estimates of Thomas (1983). The effects of these losses have been compounded by flow regulation, which influences river stage and the extent of habitat inundation. Whereas dikes have increased the river elevations required to flood many shallow-water habitats, the effects of flow management on river stage and habitat inundation has varied among estuary reaches.

In the Eagle Cliff-to-Kalama reach, for example, diking alone accounted for the loss of about three-quarters of the shallow-water habitat within the depth range reported as optimal for use by subyearling Chinook (0.1-2.0 m; Bottom et al. 2005b). Our model results implied that habitat in this reach might be restored by removing dikes without significantly modifying the present flow-management regime. In contrast, the combined effects of diking and flow regulation in the Skamokawa-to-Beaver reach were significantly greater than the sum of their individual contributions to the loss of shallow-water habitat (Kukulka and Jay 2003a,b).

Flow management may be a particularly important driver of habitat opportunity in upper estuary reaches because the flood plain is relatively narrow, tidal influence is reduced, and the hydrosystem exerts significant control over daily and seasonal water levels.

2a. Together, the loss of rearing opportunities in the estuary and the decline of historical populations upriver have reduced life history diversity and late-season abundance of Chinook salmon in the estuary.

Changes in the geographic structure of spawning populations throughout the Columbia River basin most likely have contributed to shifts in juvenile abundance patterns and arrival times in the estuary (Bottom et al. 2005b). Scale patterns from juveniles collected during a survey in 1914-1916 (Rich 1920) defined at least six juvenile life histories, including one yearling-migrant and five subyearling-migrant types (Bottom et al. 2005b; Burke 2005). Rich (1920) observed subyearling Chinook salmon migrating downstream throughout most of the year, which he attributed to the sequential migrations of juveniles from tributaries located progressively further upriver. A similar pattern has been shown for juvenile fall Chinook salmon in Oregon's Salmon River basin, where median travel and arrival times to the estuary are proportional to the distance from main-stem and tributary spawning areas (Bottom et al. 2005a).

A comparison of past and recent salmon surveys hypothesized that life history diversity in Columbia River Chinook salmon populations has declined within the last century (Bottom et al. 2005b; Burke 2005). Burke (2005) analyzed recent estuary beach-seine and tagging data and proposed that just three basic migrant types are now represented: emergent fry, yearling, and a single subyearling type composed of spring and early-summer fluvial migrants with short estuary residence times.

We documented a higher proportion of subyearling Chinook salmon with estuary resident life histories than expected from previous reports (Dawley et al. 1986; Bottom et al. 2005b; Burke 2005). Results from our field surveys during 2002-2008 revealed a somewhat greater variety of subyearling behaviors, including significant numbers of fry

and fingerling migrants that rear in the estuary for extended periods. Nonetheless, if Rich's (1920) results are representative of estuary rearing and migration behaviors a century ago, then our findings support the hypothesis that Chinook salmon life histories have been simplified.

In 2002-2008 abundances of juvenile Chinook salmon in the Columbia River estuary peaked sharply in spring and declined rapidly thereafter. This differed from the broad temporal distribution observed early in the twentieth century (Rich 1920), when large pulses of new recruits appeared in the estuary in July, September, and October (Rich 1920; Burke 2005; Campbell 2010). Peak estuary abundances now occur in May, and roughly 90% of the juvenile salmon migration is complete by the end of August (Campbell 2010). A similar early juvenile migration occurs in many British Columbia estuaries, where relatively small subyearlings (~70 mm) leave rivers by mid-summer to rear in sheltered habitats along the British Columbia coast (Healey 1991). Citing the limited availability of protected habitat along the open coast from Washington to California, Healey (1991) speculated that estuaries in this region may be important summer-fall rearing areas for subyearling Chinook salmon, provided estuary water temperatures are satisfactory.

This is true of many small Oregon coastal estuaries where juvenile Chinook salmon reside until September or later (Reimers 1973; Myers and Horton 1980; Bottom et al. 2005a). In the Sixes River (Oregon) Reimers (1973) estimated that ~90% of the returning fall Chinook salmon from one brood year had migrated to the estuary as subyearlings during mid- or late summer before entering the ocean at the largest mean sizes later in the fall. Although Rich's (1920) surveys during 1914-1916 indicate that Chinook salmon also formerly used the Columbia River estuary as a summer-fall rearing area, our results suggest that this function has diminished. Far fewer juvenile migrants now enter or remain in the estuary from summer to fall compared with the protracted period of estuary use described by Rich (1920).

Multiple factors likely account for the relatively low abundance of subyearling Chinook salmon in the Columbia River estuary after mid-summer. Extensive wetland diking and filling in the lower estuary has eliminated habitat for fry and fingerling migrants and has likely reduced expression of some estuary-resident life histories. More than 50% of historical wetlands in the lower estuary have been removed. These losses have reduced rearing opportunities for fry and fingerling migrants that tend to remain in the estuary for the longest periods.

The importance of tidal wetlands to phenotypic diversity is illustrated in the Salmon River basin (Oregon), where Chinook salmon estuary-resident life histories have expanded since dike removal has restored access to most tidal marshes. With the

increased rearing opportunities, fry and fingerling salmon now appear in the estuary earlier in the year, and smolts leave for the ocean over a wider range of sizes and times (Bottom et al. 2005a; Volk et al. 2010).

Seasonal patterns of stock composition in the lower Columbia River estuary generally reflect the broad spatial structure of some major population groups. For example, lower Columbia River stock groups (Spring Creek Group fall Chinook and West Cascade fall Chinook) dominate the estuary stock composition in spring, and significant numbers of upper Columbia River summer/fall stocks appear later in the summer and fall. The large numbers of juveniles that entered the estuary late in the rearing season in 1915 and 1916 (Rich 1920) likely included many migrants from mid- and upper-basin populations that are now depleted or extinct.

Gustafson et al. (2007) identified 78 populations that have been extirpated from mid- and upper regions of the Columbia River, lower and upper regions of the Snake River, and from Columbia River headwaters. Construction of the hydropower system blocked anadromous fish passage to over 30% of historically accessible stream miles and 55% of formerly accessible drainage area in the Columbia River Basin (NPCC 2003). Inundation of alluvial habitats by dams removed as much as 80-90% of the main-stem spawning habitat for fall Chinook salmon (Dauble et al. 2003), eliminating many fall spawners with subyearling-migrant life histories that otherwise may have reared in the estuary prior to ocean entry. Spring and summer Chinook populations also likely produced subyearling migrants in the middle reaches of some Columbia River subbasins, where growth opportunities were enhanced by warm temperatures (Liss et al. 2006).

Water temperatures now reach stressful or lethal levels in the lower sections of many of these subbasins—for example, the Yakima, Tucannon, Umatilla, Grande Ronde, Okanagan rivers—limiting the expression of subyearling life histories that once depended on juvenile migration during summer and fall (Liss et al. 2006; Stanford et al. 2006). Although spring-summer Chinook salmon in the Pahsimeroi River (Snake River Basin) produce subyearling migrants, no evidence exists that they return as adults, raising concerns that this life history could be eliminated from the population (Copeland and Venditti 2009). These or other changes in life history expression and survival upriver thus influence the migration times, sizes, and residency of juvenile Chinook salmon in the estuary.

Further research is needed to characterize contemporary life history variations among mid- and upper basin stocks, particularly their patterns of habitat use and residency in the upper estuary (i.e. rkm 100 to Bonneville Dam).

2b. Historical increases in seasonal water temperatures have further reduced estuary rearing opportunities during summer and fall months, exacerbating the effects of wetland habitat loss (Figures 8-9).

The excessive seasonal water temperatures that limit salmon rearing in some upper-basin tributaries and main-stem areas also extend into the estuary's tidal fluvial zone. Juvenile salmon vacate shallow wetland habitats, and salmon abundance throughout the lower estuary declines by mid-summer, when surface-water temperatures reach and exceed 19°C. Temperature conditions therefore likely limit the expression of estuary-resident life histories late in the rearing season, reducing the overall capacity of the estuary to support juvenile salmon.

During summer and fall, we frequently measured water temperatures above 19°C in the main-stem estuary and in shallow wetland channels. Our regression model indicated that historical temperatures remained below this level in all months of the year under the virgin-flow, cool-temperature scenario (1890-1926) but averaged above 19°C from July through September under the modern (1976-2002) temperature scenario. Because cool ocean water does not intrude far upstream in the Columbia River, the temperature of river water is a major driver of habitat conditions across much of the estuary. We attributed more than half of the estimated 2-3°C increase above historical May-December water temperatures to the effects of reservoir storage behind Columbia River dams.

Water temperatures in much of the estuary now exceed optimal levels for juvenile salmon and could limit rearing opportunities and salmon performance (i.e., growth and survival) after June or July. Chinook salmon abundance consistently declined during this period, when water temperatures typically exceeded 19-20°C (Roegner et al. 2008). Chinook salmon were present at some near-shore beach-seining sites in temperatures as high as 24°C, but abundance nonetheless declined rapidly after July. In freshwater studies, the distributional limits of many salmonids, including Chinook, coincided with mean daily water temperatures of 20°C or maxima of 22-24°C (McCullough 1999).

At Russian and Lord Island wetland marshes in 2007 and 2008, much if not most of the shallow-water habitat that was otherwise accessible to salmon (> 0.5 m deep) became marginal for rearing after June because of high water temperatures (Figure 29). In riverine environments, salmonids move to thermal refugia, such as cold-water seeps, the mouths of cool tributaries, or areas with intragravel flow, when temperatures climb to stressful levels even for short periods (Hokanson et al. 1977; Bilby 1984; McCullough 1999). Alluvial habitats affected by groundwater upwelling include deep pools, low-velocity backwaters, and spring brooks isolated from main-channel flows, and these afford cool-water refugia in many lower-elevation and desert-stream reaches of the

Columbia River basin, where salmonids otherwise might not persist (Stanford et al. 2006). Although similar thermal refugia may exist within the estuary, their locations and attributes have not been defined. As tidal-freshwater reaches approach stressful temperatures, some juveniles may move downstream into cooler, brackish areas or may exit the estuary altogether. Additional warming of the Columbia River Basin through climate change could further restrict estuary rearing opportunities and life history expression by juvenile salmon.

To realize the full benefits of estuary restoration, additional actions are needed to ameliorate stressful water temperatures in the main-stem river and to identify and protect existing cool-water refugia (cold-water seeps, groundwater upwelling sites, etc.) within the estuary.

II. Estuarine Fish Species and Salmon Habitat Use

3. Salmon habitat use and residence times vary with fish size. All wetland habitat types in the lower estuary are utilized by the smallest subyearling size classes, which tend to remain in the estuary for the longest periods.

Our beach-seine and wetland-survey results supported the hypothesis that estuary habitat use by juvenile salmon is size-related and that many small subyearling migrants rear in shallow wetland channels (Bottom et al. 2005b). A wide range of size classes occurred at near-shore beach-seine sites, including some fish larger than 120 mm; however, the shallow and protected wetland sites were dominated by smaller fry and fingerlings. For example, individuals rarely exceeded 90 mm in the secondary interior channels of the Russian and Seal Island emergent marsh (rkm 35). Some larger juveniles (including yearlings) entered the Russian Island complex, but these fish generally occurred in the largest distributary channels. Maximum fish sizes were even smaller in the forested and scrub-shrub wetlands located further upriver (above rkm 75), where individuals rarely exceeded 70 mm.

As described in other Northwest estuaries (Myers and Horton 1980; Healey 1982, 1991), mean sizes of juvenile Chinook salmon generally increased toward the estuary mouth and laterally from shallow nearshore to deeper offshore areas. The high proportions of recently emerged fry found throughout summer at sites in and above Cathlamet Bay suggested that many of the smallest individuals delayed their movements to enter tidal freshwater areas, while larger fish migrated further downstream. Overall, these results suggest a size-dependent pattern of migration and habitat use among many juveniles: the smallest individuals reared in shallow peripheral channels of all wetland

habitat types—emergent, scrub-shrub, forested, and mixed—throughout the lower estuary and gradually moved offshore and toward the estuary mouth as they fed and grew.

Effects of fish size on rearing behavior likely explain the apparent discrepancies between the estuary residence times we observed and those reported previously for Columbia River Chinook salmon. Previous tagging studies in the lower estuary reported residence times of approximately 1 week (Dawley et al. 1986). Similar residence time was reported in several recent estimates that quantified total estuary transit time from Bonneville Dam to near the river mouth (Schreck et al. 2006; McComas et al. 2008). In contrast, our back-calculations of residence time based on otolith chemistry indicated that during 2003-2005, estuary residence averaged 2-3 months for the smallest fry migrants captured at Pt. Adams Beach and 4-6 weeks for large subyearlings (>90 mm).

Approximately three-quarters of the subyearlings we sampled at Point Adams Beach had a discernible Sr spike on their otoliths, indicating a measurable period of salt-water rearing. We estimated that 30-50% of these residents had stayed in the lower estuary more than 30 d prior to capture (Campbell 2010). These back-calculations represented minimum estimates of estuary residency time because the Sr technique only measures contact with salt water and does not account for time spent in the estuary's extensive tidal freshwater zone.

Anderson (2006) measured somewhat longer mean estuary residence times by estimating the periods that Chinook salmon had interacted with emergent-marsh food webs in the lower 65 km of the estuary. Average "interaction times" with marsh-derived prey were determined from stable isotope ratios of carbon ($\delta^{13}C$), nitrogen ($\delta^{15}N$), and sulfur ($\delta^{34}S$) in salmon muscle and liver tissues collected during our 2003 and 2004 estuary surveys. On average, small subyearling Chinook salmon interacted with marsh food webs for ~65 d. Individual interaction times ranged 0-260 d (Anderson 2006).

Previous estimates of residency time for salmon in the Columbia River estuary have most often relied on tagging studies, which require large subyearling or yearling hatchery fish. Estimates that rely on these larger size classes do not represent the life histories of many smaller juveniles that enter the estuary as fry or fingerlings. Short residency values reported from tagging experiments are not surprising, since PIT and acoustic tags are too large for the smallest size classes of juvenile salmon.

The minimum size class suitable for tagging with the commonly used (12 mm) full-duplex PIT tag is ~60 mm, and the minimum size class suitable for the latest-generation acoustic tag applied in the Columbia River is even larger (> 90 mm FL). The relationship we observed between length at estuary entrance and residence time (Figure 52) suggests that an individual fish entering the lower estuary at 110 mm FL—the

average size used for recent acoustic tagging experiments (McComas et al. 2008)—would have resided in the lower estuary only about 0-9 d prior to capture near the river mouth.

Fry dispersed into wetland channels and other shallow estuarine habitats soon after emergence in early spring, and most subyearlings had vacated these habitats by August. Few subyearlings entered or remained in wetland channels of the uppermost forested and scrub-shrub wetlands (near rkm 100) at sizes larger than 70 mm FL; few entered the secondary channels of emergent wetlands on Russian and Seal Island at sizes larger than 90 mm FL. The back-calculated size at estuary entry for nearly half of the salmon we analyzed from the Pt. Adams Beach site was less than 60 mm FL.

Mean residency estimates for individuals in this size class were 54 d during January-April and 59 d during May-August (2003-2005). Therefore, whereas tagging studies often target individuals with riverine smolt life histories (i.e., extended periods of freshwater rearing, large size at estuary entry, short estuary residency), the lower-estuary beach-seine collections included a much greater proportion of smolts with riverine-estuarine and estuarine life histories (i.e., brief or moderate periods of freshwater rearing, small size at estuary entry, and extended estuary residency).

Because most sampling methods target a particular size-class of fish or type of habitat, no single technique is satisfactory to characterize the full diversity of juvenile Chinook life histories in the estuary.

4. Naturally produced subyearling salmon dominate in shallow wetland channels and may benefit most directly from restoration of wetland habitats.

Disproportionately high numbers of naturally produced salmon utilized the shallow interior tidal channels of wetland habitats, particularly in the mixed forested and scrub/shrub wetlands above Cathlamet Bay. For example, hatchery marking rates were fairly high in 2007 and 2008 (65 and 81%, respectively); yet of the subyearling Chinook salmon collected at Lord Island, less than 2% in 2007 and only about 7% in 2008 were adipose-clipped hatchery fish. These results were consistent with the size classes that frequented lower-estuary wetlands, which included many fry and small fingerlings that were smaller than most juveniles released from hatcheries.

Dike removal or other actions to restore fish access to lower-estuary wetlands will tend to target naturally produced juveniles with subyearling-migrant life histories.

5. Large releases from Columbia River hatcheries have replaced diverse naturally spawning populations with fewer hatchery stocks; these stocks are reared primarily as freshwater phenotypes (i.e., subyearling and yearling riverine smolts) that migrate relatively rapidly through the estuary and do not fully utilize its diverse habitat opportunities or capacity. Thus, contemporary patterns in the estuary of juvenile salmon temporal abundance, stock composition, habitat use, and residency are driven to a large extent by artificial propagation programs.

Hatchery programs to replace production losses from naturally spawning populations now account for as much as 80% of the remaining adult runs of salmon and steelhead in the Columbia River basin (Lichatowich et al. 2006). Approximately 146 million juvenile salmon and steelhead are released annually from hatcheries, including the most intensively propagated species, Chinook salmon (CRDART 1995). The Hatchery Scientific Review Group estimated that nearly 106 million juvenile summer, spring, and fall Chinook salmon are produced by 72 artificial propagation programs throughout the basin (Table 24; HSRG 2009).

Table 24. Total hatchery releases of juvenile Chinook salmon for each Columbia River ESU as summarized by the Hatchery Scientific Review Group (HSRG 2009).

Evolutionarily Significant Unit (ESU)	Hatchery releases (millions)	Number of hatchery programs	Percent of total hatchery production
Lower Columbia R (Fall)	46.9	10	44.2
Lower Columbia R (Spring)	6.9	9	6.5
Upper Willamette R Spring	5.6	6	5.3
Middle Columbia R Spring	4.4	7	4.1
Deschutes R Summer/Fall	0	0	0
Upper Columbia R Spring	3.3	7	3.1
Upper Columbia R Summer/Fall	20.9	13	19.7
Snake R Fall	5.8	1	5.5
Snake R Spring/Summer	12.3	20	11.6
Total	106.1	72	100

Artificial propagation has largely replaced rather than supplemented production from historical spawning populations and habitats lost to dam construction, irrigation withdrawals, and other development. To the extent these changes have concentrated salmon habitat use and constrained life history variation, the Columbia River salmon ecosystem may be more vulnerable to environmental fluctuations than it has been in the past. The combined effects of habitat loss and hatchery replacement on salmon resilience

have been implicated in the collapse of West Coast salmon fisheries, including those for Oregon coastal coho salmon (Nickelson 1986; Lawson 1993; Lichatowich 1999) and Sacramento River Chinook salmon (Lindley et al. 2009).

The resilience of salmon populations to future environmental change requires that opportunities for diverse life history expression are restored.

Hatcheries have redistributed Columbia River salmon production in time and space by replacing historical populations, which express dispersed distributions and emergence times, with a few selected phenotypes—primarily subyearling and yearling riverine smolts—which are released in concentrated pulses.

Approximately one-half of the total subyearling hatchery production is released into habitats utilized by the Lower Columbia River ESU. Annual hatchery production of subyearling Chinook salmon during our 2002-2008 field surveys ranged from approximately 56 to 71 million fish (Appendix G).

Because patterns of estuarine habitat use and residency vary with fish size and time of entry, phenotypic selection by hatcheries limits opportunities for life history expression in the estuary. The majority of hatchery releases now occur from April through July, whereas historical populations contributed estuary migrants over a broader period. These changes have likely contributed to reduced representation of late-season (summer and fall) migrants in the estuary (Rich 1920; Bottom et al. 2005b; Burke 2005).

Temporal and spatial concentration of juvenile abundance also could limit salmon performance through inter- or intra-specific interactions, thus preventing full utilization of the estuary's productive capacity. Unmarked, naturally produced fry (i.e., smaller than the sizes generally released from hatcheries) accounted for most juvenile salmon in the estuary from January through March (Figure 18). By late March or April, we observed the first hatchery arrivals, which contributed to a sharp increase in mean size and the appearance of a bimodal size distribution at beach-seine sites (Figure 16) and at several wetland sites in Cathlamet Bay (Figure 25; see also Roegner et al. 2008).

The dominant influence of hatchery rearing and release strategies on abundance patterns and stock composition in the estuary is further illustrated by trends in subyearling Chinook salmon passage over Bonneville Dam. During 2007 and 2008, for example, Spring Creek National Fish Hatchery (~38 km above the dam) released some 15 million subyearling Chinook salmon, representing about 23% of all hatchery subyearlings produced in the basin (Table 25). All hatchery fish were released simultaneously during one of three large monthly pulses, beginning early March and ending early May. Since releases from other hatcheries above Bonneville occurred later,

the signal of Spring Creek Hatchery on the arrival in the estuary of subyearling Chinook salmon was apparent (Figure 58). In 2007 and 2008, entry into the upper estuary (i.e., passage over Bonneville Dam) peaked immediately after each of the three Spring Creek Hatchery releases.

In 2009, when Spring Creek Hatchery fish were released in two batches only (April and May), the passage index at Bonneville responded similarly: two (rather than three) peaks occurred, each within 1-2 d of hatchery release. The largest peak in the passage index occurred in April 2009, corresponding with a Spring Creek Hatchery release nearly twice as large as all other single batches released during 2007-2009. In contrast to these brief, early spikes, a more protracted migration pattern during June and July in each of the 3 years (Figure 58) overlapped with the cumulative period of most other hatchery releases (CRDART 1995; smolt index composite report).

Table 25. Timing of subyearling Chinook salmon releases from Spring Creek National Fish Hatchery and subsequent (pre-June) peaks in the smolt passage index at Bonneville Dam. Subyearling releases from other hatchery programs above Bonneville occurred after mid-May, and very few fish were released after July. Data courtesy of Fish Passage Center and Columbia River DART Smolt Index Composite Report (CRDART 1995).

	Subyearling Chinook salmon released (millions)			
Year	All Columbia River hatcheries	Spring Creek National Fish Hatchery	Release dates (Spring Creek Hatchery)	Pre-June peaks in subyearling passage index at Bonneville Dam
2007	68.2	7.8	March 5 and 9	March 7-9
		4.2	April 12	April 14-15
		3.5	May 1	May 2-4
2008	64.6	7.4	March 5-6	March 8-10
		4.0	April 10	April 14-15
		3.5	May 2	May 4-6
2009	142.6	13.0	April 13	April 14-16
		9.5	May 1	May 3-4

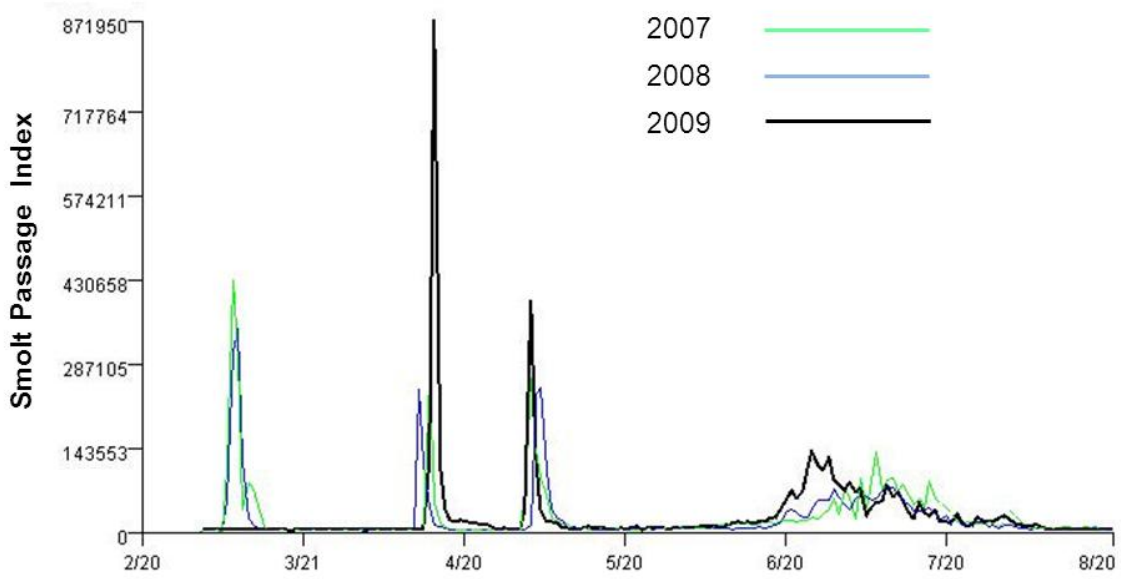


Figure 58. Passage index for subyearling Chinook salmon at Bonneville Dam, 2007-2009. Data courtesy of Fish Passage Center and Columbia River Data Access in Real Time, Smolt Index Composite Report (CRDART 1995).

The passage index at Bonneville Dam does not account for the many lower Snake River fish barged around the hydrosystem or the many individuals naturally produced or released from hatcheries below Bonneville Dam. Nonetheless, brief spikes appear in the passage index immediately following individual releases from Spring Creek Hatchery. These spikes demonstrate the direct link between hatchery release strategies and patterns of salmon abundance and stock composition within the estuary.

Of the total Chinook hatchery production (subyearling and yearling), nearly two-thirds is comprised of just two ESUs: Lower Columbia River Fall Chinook, which makes up about 44%, and Upper Columbia River Summer/Fall Chinook, which comprises nearly 20% (Table 24). Fish from these two ESUs also dominated the genetic stock composition of subyearling Chinook salmon from our beach-seine catches (~92%). Hatchery releases similarly accounted for a substantial proportion of the yearling Chinook salmon entering the estuary, but most yearlings migrated through deeper channels further from shore and were poorly represented in our beach-seine and trap-net catches from shallow-water habitats.

Most hatcheries favor production of large fingerling and yearling riverine smolts, a preference that was represented in the size distributions of juvenile salmon in the estuary. Campbell (2010) estimated that of the juveniles released from 52 Columbia River hatcheries in 2004 and 2005, 85-90% averaged more than 75 mm FL at the time of

release. In 2007 and 2008, when the marking rate for hatchery fish increased, we were able to compare the numbers of marked and unmarked individuals to provide a useful indicator of the relative sizes of hatchery vs. naturally produced salmon. Mean fork lengths of hatchery salmon at beach-seine sites in 2007 (Figure 20) and wetland sites in 2007-2008 (Figure 29) were considerably larger, and their size distributions narrower, than those of naturally produced juveniles. In our beach-seine samples during 2007, more than 55% of the unmarked salmon (primarily naturally produced) were smaller than 60 mm FL, but less than 3% of the marked salmon (hatchery) were below this size. The peak size class in the frequency distribution for marked juveniles was ~80 mm FL (Figure 20).

Size differences between hatchery and naturally produced juveniles directly influenced their respective habitat distributions and mean residence times in the estuary. Shallow wetland channels, particularly the forested and mixed wetland habitats above Cathlamet Bay (Wallace Island and Lord Island), contained primarily small, unmarked fry and fewer hatchery-marked subyearlings (only 2-16%; Table 11). Larger hatchery fish represented a much greater proportion of the salmon sampled in deeper, near-shore beach-seine sites (46-66%) in 2007 and 2008, particularly those located near the river mouth in the estuarine mixing and marine zones at Clatsop Spit, West Sand Island, Pt. Adams Beach, and Pt. Ellice (Table 5). Still higher proportions of hatchery subyearlings (86%) and yearlings (94%) have been recorded during recent purse-seine surveys of deep pelagic habitats in the lower estuary (L. Weitkamp, Northwest Fisheries Science Center, personal communication).

These respective distributions of hatchery and wild fish were consistent with the negative relationship we observed between size at estuary entry and estuary residence time (Figure 50). The dominant, fluvial phenotypes produced by most hatchery programs were more likely to enter the estuary at a larger size, select deeper habitats further from shore, and migrate to the estuary mouth more quickly than many of their smaller, naturally produced cohorts. Hatchery rearing programs thus strongly influence phenotypic expression by salmon in the estuary.

Improvements to upriver habitats and adjustment of hatchery and other management programs are needed to accommodate a greater variety of salmon phenotypes that can fully benefit from the estuary's diverse habitat opportunities, including those that are re-established through wetland restoration.

6. The estuarine ecosystem response to large subsidies of hatchery fish and interactions in the estuary between hatchery and naturally produced salmon remain poorly understood. Such interactions may ultimately determine whether estuary restoration is an effective tool for salmon recovery.

Our research highlighted the effects of hatchery selection on salmon use of estuary habitat, independent of other ecological effects or genetic influences of hatchery programs. However, studies of multiple salmon species have provided evidence that hatchery programs can undermine the reproductive success of natural populations, regardless of the length of exposure to hatchery fish (Nickelson 2003; Araki et al. 2007, 2008; Buhle et al. 2009; Chilcote et al. 2011). The specific mechanisms for these effects are uncertain; nonetheless, the results raise additional concerns about the interactions between hatchery and naturally produced populations.

Intensive hatchery production could undermine the effectiveness of recovery measures for at-risk populations.

Very little is known about ecological response in the estuary to hatchery releases, for example, whether significant competitive interactions exist between hatchery and naturally produced juveniles, or whether large hatchery releases influence estuarine predator populations. Until recently, the low marking rates of most Columbia River hatcheries have severely limited the ability of investigators to distinguish hatchery from naturally produced juveniles. Indeed, for all but the final 2 years of our 2002-2008 surveys, marking rates for Columbia River hatcheries were insufficient to draw conclusions about the behavior or performance of hatchery vs. naturally produced salmon by comparing marked and unmarked groups, respectively. At the current marking rates, it has become more feasible to make these comparisons, although sizeable proportions of some hatchery stocks (e.g., upper Columbia River summer/fall) are still left unmarked.

We strongly recommend that additional estuary studies be designed and conducted to examine more explicitly the ecological interactions between hatchery and naturally produced Chinook salmon.

III. Habitat Capacity: Prey Availability, Diet, and Rates of Consumption

7. Wetland-derived food webs support juvenile salmon throughout the estuary, including larger individuals that do not typically occupy wetland channels.

Previous analyses have suggested that wetland habitat loss may have reduced the rearing capacity of the Columbia River estuary for juvenile salmon (Bottom et al. 2005b). This interpretation was based on an apparent shift in the carbon budget, concurrent with the loss of wetland habitat (Sherwood et al. 1990) and characterized by a reduction in emergent plant and benthic macrodetritus production. Sherwood et al. (1990) estimated that an increase in riverine detritus from phytoplankton production in main-stem storage reservoirs had more than made up for the carbon loss associated with reduced estuarine plant production. However, the ecosystem response to a qualitative shift in carbon sources (i.e., from benthic macrodetritus to pelagic microdetritus), including effects on salmonid food webs, remains poorly understood (Bottom et al. 2005b).

Our survey results provided evidence that the food webs of subyearling Chinook salmon remain closely coupled to wetland and other shallow-water habitats, reinforcing concerns that the historical capacity of the estuary to support juvenile salmon may have declined.

In Chinook salmon from lower-estuary beach-seine sites in 2002-2007, diet composition was similar to that reported from beach-seine surveys 30 years ago (Bottom et al. 1984; McCabe et al. 1986; Bottom and Jones 1990). In both cases, juvenile salmon, including larger fish that rarely enter wetland channels, fed directly on insect and amphipod taxa that are typically produced in wetlands and other shallow-water habitats. Although previous studies have often stressed the importance of amphipods, adult diptera generally ranked highest in importance among prey items in the diets of juvenile salmon collected monthly at Pt. Adams Beach during 2002-2007 (Figure 38). Insects did not rank as highly in diet samples collected at intertidal sites below Tongue Point in 1980 except during April and May (McCabe et al. 1986). *Americorophium salmonis* (formerly classified as *Corophium salmonis*) was often a dominant prey item during the 1980 survey, particularly in the upper (tidal freshwater) reaches of the lower estuary.

Small subyearling Chinook salmon that entered emergent, scrub-shrub, and forested wetland channels also consumed invertebrate taxa, which are produced primarily within these habitats. These included large proportions of emergent and larval chironomids, as well as other diptera. Although chironomids dominated salmon diets, the largest size classes of subyearlings also frequently consumed *Americorophium* spp. and other epibenthic invertebrates (Lott 2004). Juvenile Chinook salmon diets in the tidal wetlands of Grays River, a tributary of the lower Columbia River estuary, were

composed of similar prey taxa (Eaton 2010; McNatt et al. 2010). Chironomid pupae and adult insects also were the dominant prey items in 7 of 12 tidal marsh channel sites surveyed in the Fraser River estuary in May 1979 (Levy and Northcote 1981).

Our emergent chironomid surveys revealed a transitional emergent insect community during the period of juvenile Chinook migration and rearing in freshwater tidal channels. Dipteran taxa, particularly Chironomidae, dominated insect emergence and assemblage composition over time and space. For a given date, insect composition and abundance was consistent within each microhabitat (Figure 36), supporting the notion that this scale was appropriate for examining spatial patterns in aquatic insect distribution.

Chironomids and other macroinvertebrates perform basic life history functions (feed, find shelter, and move about) at this scale, and thereby localize insect emergence (Davies 1984; Baxter et al. 2005). Such distinctions could drive fine-scale decisions regarding prey selection and habitat use by juvenile salmon (McIvor and Odum 1988).

Future studies should examine whether Chinook salmon target specific taxa within the chironomid family as evidence for microhabitat selection within tidal channels.

Results of recent neuston surveys in the Columbia River estuary indicate that the prey taxa produced in tidal wetlands provide food for juvenile salmon both inside and outside these habitats. Ramirez (2008) documented both retention and export of Chironomid pupae, larvae, and adults produced in emergent and forested wetlands in the lower estuary. The propensity for wetlands to function as invertebrate sources or sinks may depend on the structure and hydrology of each site: locally produced prey are retained in higher proportions in wetlands with sinuous channels than in those with simple, linear creeks (Hood 2002; Ramirez 2008).

In the Grays River, a tributary to the lower estuary, tidal wetlands exported more drift insects than they imported, although transport efficiencies varied among taxa (Eaton 2010). In turn, the proportion of terrestrial insects in salmon diets increased in areas of the main stem Grays River that were directly subsidized by wetland-produced prey. Recent stable-isotope analyses of organic carbon sources and salmon in the estuary have demonstrated that energy flow to juvenile salmon remains closely linked to wetland detritus (Maier and Simenstad 2009), despite considerable loss of historical wetland habitat.

Sherwood et al. (1990) estimated that at present, riverine detrital input to the estuary is up to 50 times greater than wetland detrital input. Even so, the invertebrate prey taxa favored by juvenile salmon continued to reflect a selective reliance on marsh detritus. The isotopic signatures of juvenile salmon, in turn, indicated a disproportionate reliance on wetland-sourced organic matter, presumably through consumption of wetland-produced invertebrates. These results are consistent with the hypothesis that historical wetland losses have reduced the estuary's capacity to support some juvenile salmon food webs (Maier and Simenstad 2009).

Bioenergetic efficiencies also influence the capacity of the estuary to support salmon, but these influences vary with changes in both water temperature and prey availability. For example, optimum temperature for juvenile Chinook salmon growth was estimated at 19°C when food rations are unlimited, but only 14.8°C when food rations are at 60% of maximum daily consumption, with the latter temperature value considered more typical of field conditions (Brett et al. 1982). Groot et al. (1995) documented an optimal temperature range for rearing Chinook salmon between 12 and 17°C, with the ideal temperature ~15°C. Although we estimated relatively high daily rations for juvenile salmon collected in wetland channels (6-17% body weight per day), all consumption studies were conducted during spring months, before temperatures had reached stressful levels.

Estimates from otolith analyses indicated positive growth rates for estuary-rearing salmon during all seasons, including maximum mean values of 0.55 mm day^{-1} during late summer (August). By this time, water temperatures in much of the estuary had increased to high levels, but salmon abundance had declined substantially. We had no measure of the daily rations or water temperatures corresponding to individual growth rates, which were back-calculated from otolith increment widths for individuals captured near the river mouth (Pt. Adams Beach).

Bioenergetic modeling may offer the best tool to investigate further the interactive effects of water temperature, salmon densities, and prey availability on salmon growth potential in the estuary.

In conclusion, wetlands produce and export insect and other prey taxa to other areas of the estuary. All size classes of juvenile salmon are linked to wetland-derived food webs, and wetland-derived prey are selected in greater proportion than other food-web sources that are more readily available. Extensive wetland losses thus undermine a preferred trophic pathway, whose deterioration could limit the estuary's capacity to support juvenile salmon.

These results reinforce the need for wetland restoration, which will benefit all ESUs and size classes of salmon that reside, feed, and grow in the estuary before migrating to the ocean.

IV. Performance Metrics: Population Structure, Life History Diversity, and Growth

8. Different genetic stock groups of Chinook salmon exhibit characteristic patterns of temporal and spatial distribution in the lower estuary.

Genetic stock groups in the lower estuary were not uniformly distributed, but exhibited distinct temporal and spatial patterns. Fall-run juveniles from the West Cascade and Spring Creek Group stocks (both from the Lower Columbia River Chinook Salmon ESU) dominated shallow habitats of the lower estuary. Whereas the Spring Creek Group fall stocks were widely distributed and abundant primarily during spring, proportions of West Cascade fall stock generally increased with distance from the estuary mouth, and these stocks were well represented from spring through fall. Representatives from the Upper Columbia River summer/fall stock group were distributed throughout the estuary, but few appeared in collections until summer and fall. No consistent difference in stock composition was apparent at finer (i.e., habitat) scales.

Other studies (e.g., Teel et al. 2009) in the upper estuary have documented greater proportions of interior and Willamette River stocks than were represented in our lower-estuary surveys. In 2010 we initiated a series of synoptic genetic surveys across all upper-estuary reaches (from rkm 100 to Bonneville Dam) to compare estuary-wide stock distributions.

Additional surveys are needed to discern stock-specific patterns of habitat use in the upper estuary and to guide the selection of restoration sites that will most benefit at-risk salmon populations.

9. Most Chinook salmon Evolutionarily Significant Units are capable of expressing subyearling life histories. Both lower and upper Columbia River stock groups can produce subyearlings that reside in the estuary for several months.

Subyearling Chinook salmon from all ESUs occupied shallow-water habitats of the lower estuary, with the exception of spring run groups from the interior Columbia Basin. Otolith collections from Pt. Adams Beach indicated that subyearlings from a diverse subset of these ESUs expressed estuary-resident life histories. Back-calculations using otolith chemical analyses indicated that size at salt-water entry for lower Columbia River fall stocks (West Cascade and Spring Creek Group) averaged near 60 mm FL. Mean residence times in the saltwater portion of the estuary were estimated at between 1 and 2 months. Although the average back-calculated size at entry for Upper Columbia River summer/fall collections (n = 9) was much larger (88 mm FL), the estimated period of estuary residency averaged 2 months or more.

Additional otolith analyses will be needed to quantify estuarine life histories for a greater variety of stock groups, including interior basin and Willamette River stocks, which were poorly represented in collections from Pt. Adams Beach.

10. The lower Columbia River estuary supports foraging and growth of juvenile migrants and contributes to the life history diversity of Chinook salmon populations.

Despite evidence that multiple factors have constrained juvenile life histories, we found estuary-resident behaviors were more prevalent than suggested by previous studies (Bottom et al. 1984; Dawley et al. 1986; Bottom et al. 2005b; Burke 2005). Otolith chemistry results indicated that 32-45% of juvenile salmon sampled in shallow-water habitats had entered the Columbia River estuary soon after emergence. Many of these early entrants fed and grew in the estuary for weeks or months before being captured at larger sizes near the estuary mouth. Estuary residency by fry and fingerling migrants was poorly resolved by other surveys, which did not sample shallow, off-channel areas or which inferred estuary residence times from the behaviors of large hatchery juveniles (Dawley et al. 1986; McCabe et al. 1986; Schreck et al. 2006; McComas et al. 2008). Data from PIT-tag detections on Russian Island indicated that even some large hatchery-reared individuals lingered in the estuary for weeks or months and occupied off-channel habitats before migrating to the ocean.

These detection data also revealed a surprising degree of habitat fidelity by some individuals. Despite having to vacate shallow wetland channels twice daily with each low tide, some juveniles returned repeatedly to the same site and continued to grow during their residency (Figures 54 and 55). Others visited the same channel intermittently for weeks (Figure 55; See also Hering 2009), suggesting that wetland fidelity may be greater at a coarser scale (e.g., the entire wetland-habitat complex) than that represented by the small, secondary channels we monitored with PIT antennas. These results showed that a significant number of subyearling Chinook salmon reside in the estuary for months, and some individuals spend days or weeks in or around the same habitat or habitat complex. Individuals that remained within the same emergent wetland complex for one week or more benefitted directly from local feeding opportunities, as indicated by average instantaneous growth rates of 0.65-0.82 mm d^{-1}.

Because we sampled in the lower estuary only, we could not discount the possibility that some Columbia River stocks had reared primarily in the upper reaches of the estuary between rkm 100 and 233 (Bonneville Dam). Genetic surveys have shown higher proportions of upper Columbia River stock groups than we observed in tidal floodplain wetlands of the lower Willamette River (near and above rkm 140; Teel et al. 2009). These results suggest that estuary habitat selection and distribution may be stock-specific, a hypothesis further supported by results of a recent series of bimonthly

genetic surveys to determine Chinook stock-group distributions between rkm 100 and 233 (D. Teel, Northwest Fisheries Science Center, unpublished data).

Diversity of salmon life histories has been described as an evolutionary strategy to spread risk and avoid brood failure in uncertain environments (Healey 1991; 2009). Rather than a series of discrete life history "types," diversity in Chinook salmon is represented by a continuum of juvenile residency patterns and adult spawning times that reflect spatial and temporal gradients in temperature during incubation and rearing (Brannon et al. 2004). Yearling and subyearling migrants are commonly produced by fall, spring, and summer runs of Chinook salmon across the wide range of temperatures and elevations in the Columbia River Basin (Brannon et al. 2004; Copeland and Venditti 2009; Teel et al. 2009).

The range of sizes, estuary entrance times, and estuary residence patterns described by Rich (1920) depict a continuum of juvenile phenotypes produced by a diversity of upriver populations (i.e., genotypes). Our results indicate that the lower estuary contributes to this continuum by providing alternative rearing habitats for juvenile growth and development, particularly by subyearlings, prior to ocean entry.

Additional research is needed to determine the contribution of diverse habitats in the upper estuary to the life histories and performance (e.g., foraging success, growth, and survival) of Chinook salmon stocks throughout the basin.

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APPENDIX A

Related Publications of the Estuary Research Team

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APPENDIX B

Methods for Evaluating Historical Change in Estuary Habitat Opportunity

Historical Changes in Flow and Sediment Input to the Estuary

Interruption of sediment supplies and the dampening of peak flows by main-stem Columbia River dams could substantially modify habitat structure and salmon rearing opportunities within the estuary. Such changes have created an imbalance in the sand budget, wherein more sediment is now exported by dredging and disposal (at sea and on land) and offshore transport (during high flows) than is supplied from upriver. Fine sediments (silt and clay) are also important for their role in water quality; toxic contaminants bind to organic fines, and may affect dissolved oxygen levels in both the water column and bed. Fines also serve as detrital sources that support estuarine food webs, maintain the estuarine turbidity maximum (ETM), and contribute to habitat construction.

Methods—We analyzed flow and sediment-transport data collected by the USGS, Environment Canada, and the U.S. Army Corps of Engineers to determine seasonality and amount of river flow, to quantify the supply of fine and coarse material to the estuary, and to describe long-term changes in flow and sediment transport. These analyses were completed in the following five stages:

1) Historical flow data were compiled from The Dalles and for western sub-basin tributaries. Missing flow data for the Willamette River at Albany, Oregon, for 1878-1893 were hindcast using multi-lag correlations between flows and precipitation from the records available before 1900.
2) Willamette River flows from Albany and Salem, Oregon, were routed to Portland using the formulation of Orem (1968).
3) Flows were routed to Beaver, Oregon, from The Dalles, Washington, the Willamette River at Portland, Oregon, and from the Cowlitz, Lewis, East Fork Lewis, and Kalama Rivers using the formulation of Orem (1968).
4) Virgin flows were estimated for The Dalles, the Willamette River at Portland, Oregon, and Beaver from 1878 to date. The methodology and irrigation corrections for these estimates followed those of the U.S. Bureau of Reclamation (USBR 1999).
5) Rating curves were development and applied from USGS flow and sediment-transport data for the Columbia River at Vancouver, Washington, and the Willamette River at Portland, Oregon. Separate curves were developed for sand, fines, and total sediment load under observed, adjusted, and virgin flows.

Recently, all flow and sediment supply estimates were updated to 2004 (from 1999). The virgin flow methodology is described in Naik and Jay (2005). Less detailed estimates are available for Beaver, Oregon, and for the Cowlitz, Lewis, East Fork Lewis, and Kalama Rivers (west side tributaries).

Sediment transport hindcasts employed simple power-law rating curves, based on data collected during the following periods and locations: 1962-1970 in the Columbia River at Vancouver, Washington; 1962-1965 in the Willamette River at Portland, Oregon; 1968-1970 at Beaver, Oregon; and various times in the West-side tributaries.

All historical hindcasts assumed that sediment yield has not changed with changes in land use. Effects of the Mt. St. Helens eruption were excluded in all analyses because most material added to the system by Mt. St. Helens had been removed. For Vancouver and Portland, hindcasts were conducted separately for sand transport, fine sediment transport, and total load for observed flow, adjusted flow, and virgin flow as defined by Bottom et al. (2005b) and Naik and Jay (2005). Due to the diverse data collection methodologies employed by the U.S. Geological Survey at different times and places, hindcasts were not possible for all stations. For the west side tributaries only observations of fine sediment were available, whereas for Beaver, only total load was reported.

Results—Comparison of observed, adjusted (observed flow corrected for reservoir manipulations and evaporation), and virgin flows for three time periods (1878-1899, 1945-2004, and 1970-2004) facilitates definition of historical flow, the present or modern flow (in terms of climate), and the modern flow (in terms of management), respectively (Appendix Table B1). Results are presented in m^3s^{-1} and as percent of the 1879-1899 virgin flow for each location. Appendix Tables B2, B3, and B4 present similar results for sediment transport (10^3 metric ton day^{-1}) for the Columbia River at Vancouver, Washington, and Beaver, Oregon; the Willamette River at Portland, Oregon; and the Cowlitz, Lewis, East Fork Lewis, and Kalama Rivers (west side tributaries). Results distinguish between sand (including gravel, if any), fine sediment, and total load, except for Beaver, where only total load could be estimated, and the west side tributaries, where only fine sediment transport could be estimated because no sand transport data were available except in the immediate aftermath of the 1980 Mt. St. Helens eruptions.

Results presented here expand upon the findings of SARE (Bottom et al. 2005b) in several respects:

- For the present period, we extended calculations 5 years beyond 1999, the end date of the SARE analysis. Thus, the present climatic regime is now defined as 1945-2004, and the present management regime as 1970-2004. These distinct averaging periods are needed to distinguish the two aspects of the present flow regime: a relatively long period (60-70 years) is required to cover a representative range of PDO and ENSO conditions, whereas the present management regime (in terms of flow regulation and diversion) was not established until ca 1970.
- Estimated adjusted and virgin flows are now available for The Columbia at Beaver and the Willamette at Portland, allowing corresponding sediment transport estimates.
- Sediment transport (sand, fines, and total load) is now available for the Willamette River at Portland, OR, and for the Columbia River at Vancouver, WA (previously reported in SARE).
- Sand transport at Vancouver was estimated from a rating curve based on actual 1962-1963 observations. Previously, it was based on an estimate of percent sand in the total load using Haushild et al. (1966), an approach that did not always provide reasonable results.

Appendix Table B1. Columbia and Willamette River average observed, adjusted and virgin flows, by period.

	Estimated river flows					
Location and	1879-1899		1945-2004		1970-2004	
flow scenario	m^3s^{-1}	(%)	$m3s^{-1}$	(%)	m^3s^{-1}	(%)
Columbia River						
The Dalles						
Observed	6,272	99.1	5,273	83.3	5,118	80.9
Adjusted	6,272	99.1	5,382	85.1	5,242	82.9
Virgin	6,327	100.0	5,828	92.1	5,719	90.4
Beaver						
Observed	8,074	99.4	7,016	86.4	6,779	83.5
Adjusted	8,074	99.4	7,122	87.7	6,906	85.0
Virgin	8,122	100.0	7,585	93.4	7,404	91.2
Willamette River						
Portland						
Observed	1,074	100.0	1,000	93.1	963	89.6
Adjusted	1,074	100.0	1,000	93.1	965	89.9
Virgin	1,074	100.0	1,009	93.9	977	91.0

Appendix Table B2. Columbia and Willamette River sand transport (in metric tons) for average observed, adjusted, and virgin flows by period.

Location and flow scenario	Sand transport 1877-1899 10^3 t d^{-1}	1877-1899 10^6 t yr^{-1}	1877-1899 (%)	1945-2004 10^3 t d^{-1}	1945-2004 10^6 t yr^{-1}	1945-2004 (%)	1970-2004 10^3 t d^{-1}	1970-2004 10^6 t yr^{-1}	1970-2004 (%)
Vancouver									
Observed flow	37.3	13.6	100	10.7	3.9	29	5.6	2.0	15
Adjusted flow	35.8	13.1	96	25.4	9.3	68	22	8.0	59
Virgin flow	37.4	13.7	100	34.7	12.7	93	29.8	10.9	80
Portland									
Observed flow	0.67	0.2	100	0.51	0.2	76	0.4	0.1	52
Adjusted flow	0.67	0.2	100	0.75	0.3	112	0.7	0.3	92
Virgin flow	0.67	0.2	100	0.75	0.3	112	0.7	0.3	92
Vancouver plus Portland									
Observed flow	38.0	13.9	100	11.2	4.1	29	6.0	2.2	16
Adjusted flow	36.5	13.3	96	26.2	9.6	69	22.7	8.3	60
Virgin flow	38.1	13.9	100	35.5	12.9	93	30.5	11.1	80

Appendix Table B3. Columbia and Willamette River fine sediment transport (in metric tons) for average observed, adjusted, and virgin flows by period.

Location and flow scenario	Fine sediment transport 1877-1899 10^3 t d^{-1}	1877-1899 10^6 t yr^{-1}	1877-1899 (%)	1945-2004 10^3 t d^{-1}	1945-2004 10^6 t yr^{-1}	1945-2004 (%)	1970-2004 10^3 t d^{-1}	1970-2004 10^6 t yr^{-1}	1970-2004 (%)
Vancouver									
Observed flow	19	6.9	103	17.9	6.5	97	18	6.6	97
Adjusted flow	18.4	6.7	99	15.3	5.6	83	15.1	5.5	82
Virgin flow	18.5	6.8	100	16.6	6.1	90	16.8	6.1	91
Portland									
Observed flow	8.8	3.2	100	7.8	2.8	89	6.3	2.3	71
Adjusted flow	8.8	3.2	100	9.4	3.4	107	8.8	3.2	101
Virgin flow	8.8	3.2	100	9.4	3.4	107	8.8	3.2	101
Vancouver plus Portland									
Observed flow	27.8	10.1	102	25.7	9.4	94	24.3	8.9	89
Adjusted flow	27.2	9.9	100	24.7	9.0	91	23.9	8.7	88
Virgin flow	27.3	10.0	100	26.0	9.5	95	25.6	9.4	94
West-Side[1] w/o Mt. St. Helens									
Observed flow	6.6×10^{-3}	2.4×10^{-3}	100	6.6×10^{-3}	2.4×10^{-3}	100	$6. \times 10^{-3}$	2.2×10^{-3}	92
Adjusted flow	6.6×10^{-3}	2.4×10^{-3}	100	6.8×10^{-3}	2.5×10^{-3}	104	6.4×10^{-3}	2.3×10^{-3}	96
Virgin flow	6.6×10^{-3}	2.4×10^{-3}	100	6.8×10^{-3}	2.5×10^{-3}	104	6.4×10^{-3}	2.3×10^{-3}	96
West-Side[2] with Mt St Helens									
Observed flow	6.6×10^{-3}	2.4×10^{-3}	100	9.6×10^{-3}	3.5×10^{-3}	144	11.1×10^{-3}	4.1×10^{-3}	167
Adjusted flow	6.6×10^{-3}	2.4×10^{-3}	100	9.6×10^{-3}	3.5×10^{-3}	144	11.1×10^{-3}	4.1×10^{-3}	167
Virgin flow	6.6×10^{-3}	2.4×10^{-3}	100	9.6×10^{-3}	3.5×10^{-3}	144	11.1×10^{-3}	4.1×10^{-3}	167

[1.] Transports estimated from rating curves developed without using 1980-1988 Cowlitz River data.
[2.] Observed transports related to the May 1980 eruption of Mt St Helens are included for the years 1980-1988.

Appendix Table B4. Columbia and Willamette River total sediment load (in metric tons) for average observed, adjusted, and virgin flows by period.

Location and flow scenario	Total sediment load								
	1877-1899			1945-2004			1970-2004		
	10^3 t d^{-1}	10^6 t yr^{-1}	%	10^3 t d^{-1}	10^6 t yr^{-1}	%	10^3 t d^{-1}	10^6 t yr^{-1}	%
Vancouver									
Observed	56.3	20.6	101	28.7	10.5	51	23.6	8.6	42
Adjusted	54.2	19.8	97	40.7	14.9	73	37.1	13.6	66
Virgin	56	20.5	100	51.3	18.7	92	46.7	17.1	83
Portland									
Observed	9.45	3.5	100	8.3	3.0	88	6.7	2.4	70
Adjusted	9.45	3.5	100	10.2	3.7	108	9.5	3.5	101
Virgin	9.45	3.5	100	10.2	3.7	108	9.5	3.5	101
Vancouver plus Portland									
Observed	65.8	24.0	100	37.0	13.5	57	30.3	11.0	46
Adjusted	63.7	23.2	97	50.9	18.6	78	46.6	17.0	71
Virgin	65.5	23.9	100	61.5	22.5	94	56.2	20.5	86
Beaver									
Observed	72.5	26.5	96	25.5	9.3	34	18.05	6.6	24
Adjusted	72.4	26.4	96	50.9	18.6	67	43.9	16.0	58
Virgin	75.6	27.6	100	68.1	24.9	90	57.3	20.9	76

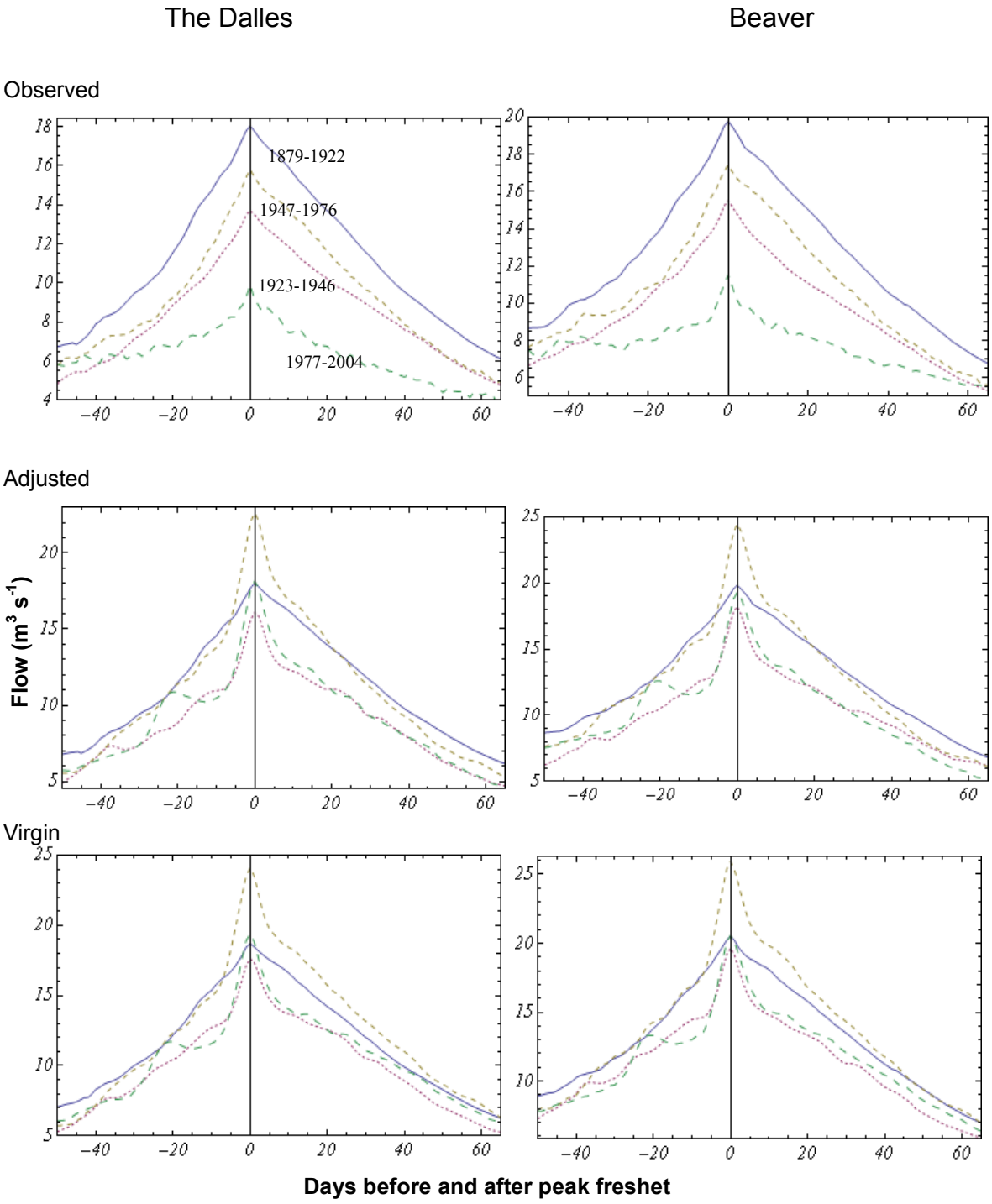
The inclusion of data from 2000-2004 in our analysis reduced average flow estimates for both the present climate and present management periods relative to those reported in SARE (Bottom et al. 2005b) because 2000-2004 was a relatively dry period. For example, observed flow at The Dalles was approximately 19% lower in 1970-2004 than in 1878-1899, whereas the previous estimate was 15-16% lower. Climate has caused a ~9.6% decrease and irrigation a 7.5% decrease in flow at The Dalles, again a slightly larger decrease in the present management period than previously estimated.

Irrigation diversion was smaller (~2%) in the Willamette River basin than in the interior sub-basin landward of The Dalles, but climate impacts were very similar (~9% decrease since ca 1900). Flows at Beaver reflect the influence of both the interior and coastal sub-basins. Because flow in the interior sub-basin is much larger than that in the coastal sub-basin, percentages for Beaver were similar to those at The Dalles. The interior sub-basin virgin flow was ~78% of the total flow at Beaver before 1900. This percentage has decreased to ~77% at present (1970-2004).

Climate change and human modifications of the Columbia River basin have both altered the shape of the annual spring freshet (Appendix Figure B1). We used four time periods for this analysis:

a) 1879-1922 a cold climate but mixed PDO and relatively unaltered flows
b) 1923-1946 a warm PDO and moderately altered flows
c) 1947-1976 a cold PDO, but a substantially altered flow cycle
d) 1977-2004 a contemporary management regime with strongly altered flows and mixed, mostly warm, PDO

At The Dalles before 1923, spring freshet flows above 6,000 m^3s^{-1} persisted throughout a 115-d period (centered on the freshet peak) used for analysis (observed flows, Figure 10). Freshet flows were somewhat less during the 1923-1946 warm PDO period than in the following cold PDO period (1947-1976), but still considerably larger than in the present management regime. Nonetheless, the shape of the observed freshet showed little change until the modern (1977-2004) period. Comparison of observed flows at The Dalles with those at Beaver showed that the western sub-basin freshet caused the flow pattern at Beaver to differ from that at The Dalles. This resulted from peak flow in the western sub-basin typically occurring before that at The Dalles.



Appendix Figure B1. Average shape of observed, adjusted, and virgin flows as a function of time before and after the peak freshet at The Dalles (L) and Beaver (R). For each scenario, time periods are indicated by dark blue solid (1897-1922), maroon dotted (1923-1946), gold dashed (1947-1976), or green dashed (1977-2004) lines. Each average was compiled from values 50 d before and 65 d after the freshet peak each year.

Before 1923, flows increased almost linearly before and decreased almost linearly after the peak at both The Dalles and Beaver. The freshet peak has become much more pronounced over time at both locations, with the highest flows now concentrated in a period of less than 20 d. Without flow regulation and diversion, very high flows would have been more common during 1947-1976 than before 1923, and the average freshet peak considerably higher. After 1923, peaks between the Snake River and the mainstem became separated in time, with Snake River flow peaking 20-25 d before the mainstem freshet peak. In Appendix Figure B1, this is shown by the smaller peak to the left of the central peak, which represents the mainstem freshet.

Given the prominent changes to virgin flow, it appears that climate change, presumably including a more rapid spring snow melt, was the primary cause of observed changes. However, deforestation may also have contributed. To the extent that high flows benefit downstream migration, the concentration of the highest flows in a period of only a few weeks narrows the window of opportunity for wild and hatchery migrants. This could thereby limit the range of life history expression by juvenile salmon. Thus, flow regulation could actually benefit juvenile salmonids by lengthening a freshet that otherwise would be quite brief and intense.

Average sand transport in the Willamette River was small relative to that in the mainstem Columbia River (Appendix Table B2). In contrast, fine sediment transport in the Willamette River was one-quarter to one-third of the total amount for Portland plus Vancouver (Appendix Table B3). Sand transport has been greatly reduced in the mainstem Columbia River. We estimated that at Vancouver, WA, sand transported during the modern (1979-2004) period has been reduced to only 15% of its historical amount (prior to 1900). This reduction was far greater than the 30-40% previously estimated (Bottom et al. 2005b). The new estimate reflects differences in both the time period and methodology. Willamette River sand transport, while quite small, has been much less affected. We could not separate sand transport at Beaver from total sediment load results (Appendix Table B4). Only fine sediment data have been collected in the west-side tributaries.

West-side fine sediment transport is relatively small, possibly because of sediment trapping in reservoirs. Observations by the USGS began after Mt. St. Helens erupted in May 1980 and continued through September 1984. The USGS data indicate that about 65×10^6 metric tons of sand (and coarser material) and 110×10^6 metric tons of total sediment load (including all size classes) reached the Columbia River, despite considerable sand trapping in the Cowlitz and Toutle Rivers. Much of this material was used as fill and did not have any long-term impact on the system, aside from the area filled.

The total load calculation for Beaver was poorly constrained for high flows, because there were no very high flows for the 1968-1970 observation period. The highest observed flow was about 16,000 m^3s^{-1}. Estimated transports for Beaver depended strongly, therefore, on the form of curve used to fit the data. Results presented in Appendix Table B4 use a power law fit for low flows and a cubic polynomial fit at high flows, which gives higher transport estimates for high flows than the power law. Nonetheless, for modern observed flows, Beaver transports were substantially lower than the sum of the Vancouver plus Portland transports. This may reflect storage in the channel between Vancouver and Beaver, or it may indicate that the Beaver transport model was not totally satisfactory. Additional results have been described by Naik and Jay (2011a,b) and Jay and Naik (2011).

Historical Datum Levels

The near-simultaneous occupation of nine stations from Cathlamet to Warrendale, Oregon in September 1877 provided the best data set for evaluating the consistency and quality of the 14 hydrographic sheets (H-sheets) used for 19th century habitat analyses (Appendix Table B5). The general pattern (with the exception of St. Helens, Oregon) suggests that the established datum was ~0.2-0.3 m above the Columbia River Datum (CRD).

The most conclusive result was shown for Vancouver, where a river gauge has been maintained since 1876. No datum could be established at Warrendale despite a month of observations in 1877 because the tide was too small and irregular, presumably due to fluctuating river flows. The first major dredging and pile dike construction in the lower river began on the St. Helens bar in 1877. Therefore, the hydraulics of this reach may have changed considerably between 1877 and establishment of Columbia River Datum in 1912. This could explain the discordant results at St. Helens (Appendix Table B5).

In deriving the results in Appendix Table B5, corrections were made for the effects of the strong 1877 El Niño at stations below Rainier, Oregon based on U.S. National Ocean Service records for the old Astoria gauge (1853 to the 1880s), climate indices, and estimated El Niño effects on contemporary tides. The correction was -0.15 m at Cathlamet and decreased to 0.0 at Rinearson, Oregon, and points landward. The correction assumed that ocean effects on datum levels decrease in a landward direction, as the base level of the river rises with distance from the ocean. If this is incorrect, then the 1877 datum would have risen relative to CRD toward the river mouth. This seems unlikely, since water level fluctuations with river flow are smaller at seaward stations. This alternative interpretation would have yielded datum levels closer to CRD at seaward stations.

Overall, our analysis suggests that the survey results were consistent and accurate, an impressive feat considering the difficult working conditions and limited data collection at the time.

Appendix Table B5. Datum analysis based on September 1877. The hydrographic sheet (H-sheet) designation indicates the sheet for which the datum was used: in some cases, this was 10-25 years after the 1877 datum determination. A positive value in the Columbia River Datum (CRD) column indicates that the datum level is above CRD. A negative correction in the El Niño factor column indicates that the datum report the CRD column was lowered by the indicated amount.

Station	H-sheet	Year	Estimate datum on CRD (m)	El Niño factor (m)
Cathlamet	H-1335	1877	0.23	-0.15
Eagle Cliff	H-1336	1877	0.28	-0.1
Oak Point	H-1336, H1368	1877	0.28	-0.1
Rinearson's Slough	H-1368	1877	0.29	-0.05
Walker Is.	H-1724	1885	0.49	0
Rainier	H-1369a	1877	0.25	0
Kalama	H-1369b	1877	0.25	0
Martin Slough	H-1524	1881	0.59	0
St Helens	H1524, H-1711	1877	0.14	0
Willow Bar	H-1671	1885	1.21	0
Pearcy's Island	H-1673	1885	1.41	0
Vancouver	CL 102-5	1877	0.24	0
Warrendale	H-2574	1901	none	NA

Historical Tides and River Flows

We used the following methods of tidal analysis to investigate the interactive effects of tides and river flow and to distinguish anthropogenic influences from changes in coastal tides.

Wavelet tidal analysis (Jay and Flinchem 1997; Flinchem and Jay 2000)—As described by Kukulka and Jay (2003), a wavelet filter bank was tuned to the diurnal and semidiurnal tidal frequencies and applied to the data, producing a time series of tidal amplitudes and phases with results every 36-72 h. This approach was used to interpret the modulation of tides by river flow.

Datum level extraction—Datum levels (Lower Low Water or LLW, Mean Water level or MWL, and or Higher High Water HHW) and Greater Diurnal Tidal Range (GDTR) were extracted as a function of river flow and other external forcing using multiple linear regression analyses. Depending on the data available for each station, we extracted models of tidal datum levels and tidal range for each 5-10 year period to assess historical trends. The limited dynamic range of river flow over short time periods precluded the use of intervals shorter than 5-10yrs. The analysis was further constrained because hydropower development since 1970 has limited the number of high-flow events, greatly reducing the dynamic flow range during the subsequent decades. We applied two types of regression models. The first described as accurately as possible (for each time period) the variations of each datum level with flow. These were used for shallow-water habitat analyses. The second was used to compare between time periods and was designed to minimize adverse effects on the analysis of changes in the dynamic range of river flow.

The regression equations for the first model were based on those developed by Kukulka and Jay (2003a,b) :

$$LLW = a_{0k} + a_{1k}Q^{1/2} + a_{2k}Q^{3/2} + a_{3k}\ AMWL\ + a_{4k}\left(\frac{R^2}{Q^{1/2}}\right) \tag{1}$$

$$HHW = b_{0k} + b_{1k}Q^{1/2} + b_{2k}Q^{3/2} + b_{3k}\ AMWL + b_{4k}\left(\frac{R^2}{Q^{1/2}}\right) \tag{2}$$

$$MWL = c_{0k} + c_{1k}Q + c_{2k}\,AMWL + c_{3k}\left(\frac{R^2}{Q^{1/3}}\right) \quad (3)$$

$$GDTR = d_{0k} + d_{1k}Q^{1/2} + d_{2k}Q^{3/2} + d_{3k}\,R + d_{4k}\left(\frac{R^2}{Q^{1/2}}\right) \quad (4)$$

Where:

Q = River flow at Beaver Army Terminal ($m^3\ sec^{-1} \times 1000$)
R = Astoria greater diurnal tidal range (m)
$AMWL$ = Astoria mean water level (m)
$GDTR$ = Greater diurnal range (m)
a_{0k}-a_{4k} to d_{0k}-d_{4k} = Model parameters for each station
k = Index for time periods, k = 1, n

The discrete regression model coefficients (in Equations 1-4) provided values at 21 stations at 5- to10-year intervals from 1940 to 2005, with many gaps (Appendix Table B6). A variety of spatial and temporal interpolation methods were used to estimate missing values. The resulting coefficients allowed stage calculations from river kilometer (rkm) 21 to 230 at 1.6-km increments with annual changes to the coefficient values from 1942 to 2004. Using these coefficients, historical tidal data at Astoria, and river flow at Beaver Army Terminal, we calculated daily tidal properties for rkm 21 to 230 from 1940 to 2004 from Equations 1-4. Because the Astoria (Tongue Pt.) tidal data were continuous since 1925, and it was desirable to consider the warm-PDO period from ca. 1926-1946 in the analysis, we used the lower-estuary data series to extrapolate tidal properties at upriver stations for the period 1925 to 1939.

The models in Equations 1-4 corresponded closely to the dynamics of the tide-river interaction and were effective for hindcasting elevations. They were less useful for making historical comparisons for two reasons. First, the appearance of Q_R in non-linear form (as $Q^{½}$ and $Q^{3/2}$) in the analysis interacts significantly with the variable dynamic range of Q_R between time periods. Also, each added variable in the analysis, while improving dynamic realism, broadened the confidence limits of all parameters. A second, simpler type of regression model (in which Q_R and tidal range were the only independent variables) therefore, was chosen for historical comparisons.

Appendix Table B6. Inventory of modeled tidal data

Nominal period	1940-1943	1970s	1980s	1990s	2000s	
Central date	1941.9	1976.5	1985.5	1995.5	2004	rkm
Hammond			×			12.9
Ft Stevens	×					13
Astoria (Tongue Pt.)	×	×	×	×	×	20.9
Altoona	×		×			38.6
Skamokawa	×		×	×	×	54.2
Cathlamet	×					60
Wauna	×	×	×	×	×	66.9
Eagle Cliff	×					82.2
Beaver			×	×	×	87
Stella	×					90.4
Longview	×		×	×	×	106.7
Rainer			×			108.6
Kalama	×					121.3
Columbia City		×	×			135.2
St Helens	×		×	×	×	138.6
Willowbar	×					152.9
Kelly Pt.	×					162.5
Vancouver	×		×	×	×	171
Ellsworth	×					180.4
Washougal	×		×			198.1
Warrendale	×		×			229.5

Robust harmonic analysis (Leffler and Jay 2009)—We used an improved harmonic analysis method to extract tidal characteristics from noisy tidal records with gaps. Specifically, the traditional least-squares minimization used in harmonic analysis was changed to a more flexible (robust) method that minimized noisy parts of the record (Huber 1996). This type of analysis was used to detect poor quality data and to evaluate the effects of record length on the accuracy of derived tidal parameters.

We also merged an elevation data set (1 point per day) for Vancouver for 1902-1971 with hourly data available since late 1972 to reconstruct historical changes in MWL from 1902 to the present.

APPENDIX C

Fish Taxa Identified in the Columbia River Estuary, 2002-2008

Appendix Table C. Scientific and common names of all fish taxa sampled in the Columbia River estuary during the tidal gradient and wetland habitat surveys, 2002-2008. Non-native species designated by asterisk and shaded cell.

Scientific name			Tidal gradient	Wetland habitat
Family	*Genus species*	Common name	study	study
Petromyzontidae				
	Lampetra ayresii	river lamprey	x	x
	Lampetra spp.	unidentified lamprey		x
Engraulidae				
	Engraulis mordax	northern anchovy	x	
Clupeidae				
	Alosa sapidissima	American shad*	x	x
	Clupea pallasii	Pacific herring	x	
	Sardinops sagax	Pacific sardine	x	
Cyprinidae				
	Cyprinus carpio	common carp*	x	x
	Mylocheilus caurinus	peamouth	x	x
	Ptychocheilus oregonensis	northern pikeminnow	x	x
	Notemigonus crysoleucas	golden shiner	x	
	Carassius auratus	goldfish*	x	
Osmeridae	*Hypomesus pretiosus*	surf smelt	x	
	Spirinchus thaleichthys	longfin smelt	x	
	Allosmerus elongatus	whitebait smelt	x	
	Thaleichthys pacificus	eulachon	x	
Catostomidae	*Catostomus macrocheilus*	largescale sucker	x	x
Salmonidae	*Oncorhynchus keta*	chum salmon	x	x
	Oncorhynchus kisutch	coho salmon	x	x
	Oncorhynchus tshawytscha	chinook salmon	x	x
	Oncorhynchus nerka	sockeye salmon	x	x
	Oncorhynchus mykiss	rainbow trout (steelhead)	x	x
	Oncorhynchus clarkii	cutthroat trout	x	x
	Oncorhynchus spp.	unidentified salmon		x
Percopsidae	*Percopsis transmontana*	sand roller	x	
Gadidae	*Microgadus proximus*	Pacific tomcod	x	
Atherinopsidae	*Atherinopsis affinis*	topsmelt	x	
Fundilidae	*Fundulus diaphanus*	banded killifish*	x	x

Appendix Table C. Continued.

Scientific name			Tidal gradient study	Wetland habitat study
Family	*Genus species*	Common name		
Fundilidae	*Fundulus diaphanus*	banded killifish*	x	x
Gasterosteidae	*Gasterosteus aculeatus*	threespine stickleback	x	x
Syngnathidae	*Syngnathus leptorhynchus*	bay pipefish	x	
Hexagrammidae	*Hexagrammos decagrammus*	kelp greenling	x	
Cottidae	*Cottus asper*	prickly sculpin	x	x
	Leptocottus armatus	Pacific staghorn sculpin	x	x
	Clinocottus acuticeps	Sharpnose sculpin	x	
		unidentified sculpin		x
Centrarchidae	*Micropterus salmoides*	largemouth bass*	x	x
	Micropterus dolomieu	smallmouth bass*	x	
	Lepomis gibbosus	pumpkinseed*	x	
	Pomoxis nigromaculatus	black crappie*	x	x
		unidentified sunfish*	x	x
Percidae	*Perca flavescens*	yellow perch*	x	x
Emiotocidae	*Cymatogaster aggregata*	shiner perch	x	
Stichaeidae	*Lumpenus sagitta*	snake prickleback	x	
Pholidae	*Pholis ornata*	saddleback gunnel	x	
Ammodytidae	*Ammodytes hexapterus*	Pacific sand lance	x	
Paralichthyidae	*Citharichthys sordidus*	Pacific sanddab	x	
	Citharichthys stigmaeus	speckled sanddab	x	
Pleuronictidae	*Parphrys vetulus*	English sole	x	
	Psettichthys melanostictus	sand sole	x	
	Platichthys stellatus	starry flounder	x	x
	Microstomus pacificus	dover sole	x	
Embiotocidae	*Hyperprosopon argenteum*	walleye surfperch	x	
	Amphistichus rhodoterus	redtail surfperch	x	
	Rhacochilus vacca	pile perch	x	
Unidentified fish			x	

APPENDIX D

Juvenile Chinook Salmon Stomach Contents

Appendix Tables D1-D4 list all items found in stomach contents of juvenile Chinook salmon. Taxa are listed in phylogenetic order according to the latest Integrated Taxonomic Information System (ITIS) taxonomic codes (www.itis.gov). Items for which no code was available are listed at the end of the table.

Items found in stomach contents were identified to the lowest possible taxonomic level, given the state of digestion of the specimens. In the case of partially digested food items which still had recognizable body parts, such as flies or amphipods, counts were derived by counting body parts that represented a single individual (e.g. heads or thoraxes). Numbers in the table represent total counts of individual stomach items, except for the category "Unidentified stomach material," which, if it occurred in a stomach, was assigned a count of "1" regardless of how much material was found.

Stomach contents are summarized by each of three estuary zones sampled along the estuary tidal gradient: tidal freshwater, estuarine mixing, and marine. The tidal freshwater zone is further divided into lower and mid-sections. (The upper tidal freshwater zone, which extends upriver to Bonneville Dam, was not surveyed for this study.) The mid-estuary portion of the tidal freshwater zone shown in Appendix Table D1 was defined as the region encompassing hydrogeomorphic reaches C-E of the Columbia River estuary (Simenstad et al. 2011) and includes our sampling sites at Wallace and Lord Islands. The lower estuary portion of the tidal freshwater zone shown in Appendix Table D2 extends downstream from the junction of hydrogeomorphic reaches B and C (Simenstad et al. 2011).

Appendix Table D5 lists, by estuary zone, the IRI values for all items found in juvenile Chinook stomachs. Digested, unidentifiable stomach material was found in 93-98% of all stomachs, and comprised between one-third and two-thirds of the wet mass of stomach contents. However, it was not possible to scale the numerical contribution of such material relative to other stomach contents; therefore, this material was not included as a separate category in IRI calculations, nor was its gravimetric contribution included in IRI calculations for other items.

Contents are listed in descending order of mean IRI contribution over all zones, which is equivalent to listing items in descending order of overall importance in the diet. The mid-estuary portion of the tidal freshwater zone listed in Appendix Table D5 is defined as the region encompassing hydrogeomorphic reaches C-E (Simenstad et al.

2011), which includes our sampling sites at Wallace and Lord Islands. The lower estuary portion of the tidal freshwater zone extends downriver from the junction of reaches B and C (Simenstad et al. 2011).

Appendix Tables D6 and D7 list IRI values for all items found in juvenile Chinook stomachs by month. Salmon were rarely caught in December; therefore, we have no stomach samples available from that month.

Digested, unidentifiable stomach material was found in 93-98% of all stomachs, and comprised between one-third and two-thirds of the wet mass of stomach contents. However, it was not possible to scale the numerical contribution of such material relative to other stomach contents; therefore, this material was not included as a separate category in IRI calculations, nor was its gravimetric contribution included in IRI calculations for other items.

Contents are listed in descending order of IRI contribution summed over all months, which is equivalent to listing stomach contents in descending order of overall importance in the diet.

Appendix Table D1. Tidal freshwater and mid-estuary zone.

	Tidal freshwater and mid-estuary zone			
	2006	2007	Total	
Taxon	n = 124	n = 102	n	(%)
Turbellaria	81		81	0.56
Trematoda	6	7	13	0.09
Nematoda	26	82	108	0.74
Polychaeta		1	1	0.01
Bivalvia		5	5	0.03
Arachnida	10	10	20	0.14
Acarina	25	2	27	0.19
Cladocera	4,941	4,416	9,357	64.53
Copepoda	14	72	86	0.59
Mysidae	267	2	269	1.86
Cumacea	56		56	0.39
Isopoda	1		1	0.01
Gammaridea	23	5	28	0.19
Americorophium sp.	52	87	139	0.96
Americorophium salmonis	788	526	1,314	9.06
Americorophium spinicorne	51	53	1,04	0.72
Decapoda, unidentified		1	1	0.01
Insecta, larva	12	6	18	0.12
Insecta, pupa	5		5	0.03
Insecta, unidentified	67	16	83	0.57
Collembola	6	3	9	0.06
Ephemoroptera, adult		2	2	0.01
Ephemoroptera, nymph		9	9	0.06
Odonata, nymph		5	5	0.03
Plecoptera, nymph		2	2	0.01
Hemiptera	5		5	0.03
Cicadellidae	21	22	43	0.30
Aphididae	3	6	9	0.06
Coleoptera	13	1	14	0.10
Diptera, adult	1,535	162	1,697	11.70
Diptera, larva	98	254	352	2.43
Diptera, pupa	56	85	141	0.97
Hymenoptera, wasp	19		19	0.13
Formicidae	9	6	15	0.10
Osteichthyes, bones		1	1	0.01
Plantae, unidentified	28	41	69	0.48
Plantae, woody debris	3	22	25	0.17
Animalia, egg - unidentified	2	66	68	0.47
Animalia, larva - unidentified	1		1	0.01
Eogammarus confervicolus	1	1	2	0.01
Rock/sand	64	14	78	0.54
Unidentified stomach material	118	100	218	1.50
Total stomach items	8,407	6,093	14,500	100.00

Appendix Table D2. Tidal freshwater—lower estuary zone

	Tidal freshwater—lower estuary zone						
	2002	2003	2004	2005	2006	Total	
Taxon	n = 151	n = 260	n = 240	n = 166	n = 138	n	(%)
Platyhelminthes		6				6	0.02
Turbellaria	2	7	14	15	27	65	0.18
Trematoda	1				41	42	0.12
Nematoda	68	109	76	69	73	395	1.12
Polychaeta	1		2			3	0.01
Oligochaeta	1	2				3	0.01
Bivalvia	1		2			3	0.01
Arachnida	31	34	39	17	48	169	0.48
Acarina	53	34	54	31	62	234	0.66
Cladocera	1,125	917	634	148	8,598	11,422	32.29
Copepoda	42	10	32	8	39	131	0.37
Mysidae	11	54	50	73	45	233	0.66
Cumacea				2		2	0.01
Isopoda	1	2	1	1	1	6	0.02
Amphipoda				1	1	2	0.01
Gammaridea	18	20	14	18	29	99	0.28
Americorophium sp.	59	65	44	58	64	290	0.82
Americorophium salmonis	448	686	845	437	470	2,886	8.16
Americorophium spinicorne	17	27	16	26	39	125	0.35
Decapoda, furcilia			1		2	3	0.01
Decapoda, unidentified	1	1				2	0.01
Insecta, larva	2		3	4	4	13	0.04
Insecta, nymph				2		2	0.01
Insecta, pupa	8		12	2	7	29	0.08
Insecta, unidentified	13	6	4	26	99	148	0.42
Collembola	23	29	8	44	143	247	0.70
Ephemoroptera, adult				3	1	4	0.01
Ephemoroptera, nymph	2	1	3			6	0.02
Odonata, nymph			1	4	1	6	0.02
Dermaptera		1				1	0.00
Plecoptera, nymph		7				7	0.02
Hemiptera	2	1	1	1	20	25	0.07
Cicadellidae	40	8	2	12	23	85	0.24
Aphididae	1	1		1		3	0.01
Coleoptera	27	9	3	11	71	121	0.34
Trichoptera, adult			1	1		2	0.01
Tricoptera, larva			1			1	0.00

Appendix Table D2. Continued.

	Tidal freshwater—lower estuary zone (continued)						
	2002	2003	2004	2005	2006	Total	
Taxon	n = 151	n = 260	n = 240	n = 166	n = 138	n	(%)
Diptera, adult	2,658	1,764	2,361	1,501	4,566	12,850	36.33
Diptera, larva	29	83	2,170	103	476	2,861	8.09
Diptera, pupa	8	24	55	33	101	221	0.62
Ceratopagonidae, larva		4				4	0.01
Chironomidae, adult	2			49		51	0.14
Chironomidae, larva	4	14	1	233	1	253	0.72
Chironomidae, pupa	12	15		3		30	0.08
Hymenoptera, unidentified	53	14	14	1		82	0.23
Hymenoptera, wasp	15	1		3	15	34	0.10
Formicidae	13	2			25	40	0.11
Osteichthyes, larva	65			1		66	0.19
Osteichthyes, scales		2		2		4	0.01
Osteichthyes, unidentifed			2			2	0.01
Plantae, unidentified	45	48	26	11	21	151	0.43
Plantae, woody debris	29	9	22	5	4	69	0.20
Animalia, egg - unidentified	8	2			753	763	2.16
Eogammarus confervicolus		30	4	5		39	0.11
Plastic			1			1	0.00
Rock/sand	14	13	216	12	13	268	0.76
Unidentified stomach material	136	158	171	161	136	762	2.15
Total stomach items	5,089	4,220	6,906	31,38	16,019	35,372	100.00

Appendix Table D3. Estuarine mixing zone.

	Estuarine mixing zone						
	2002	2003	2004	2005	2006	Total	
Taxon	n = 77	n = 167	n = 137	n = 162	n = 139	n	(%)
Turbellaria	120	18	2	37		177	1.20
Trematoda		1			35	36	0.24
Nematoda	111	91	10	21	178	411	2.78
Polychaeta	5	3	3	2	4	17	0.11
Nereis limnicola		6				6	0.04
Oligochaeta		3				3	0.02
Bivalvia			1			1	0.01
Arachnida	3	7	24	2	5	41	0.28
Acarina	6		8	2	16	32	0.22
Cladocera	61	57		43	210	371	2.51
Copepoda	1	4			100	105	0.71
Cirripedia, cyprid				1		1	0.01
Cirripedia, molt	17	20	104	1	14	156	1.05
Mysidae	1	5	25	117	71	219	1.48
Neomysis mercedis					1	1	0.01
Cumacea		2		2	4	8	0.05
Isopoda	2	5	5	7	11	30	0.20
Amphipoda	1		1	4		6	0.04
Gammaridea	15	23	32	61	73	204	1.38
Americorophium sp.	39	68	38	99	190	434	2.93
Americorophium salmonis	32	50	47	260	650	1,039	7.02
Americorophium spinicorne	34	119	132	54	119	458	3.10
Hyperiidea		1				1	0.01
Decapoda, furcilia			1		9	10	0.07
Decapoda, megalops	1		6			7	0.05
Decapoda, unidentified			3			3	0.02
Brachyura, megalops		1				1	0.01
Cancer magister, megalops		5	7	12	3	27	0.18
Insecta, larva		7	22	21	2	52	0.35
Insecta, nymph		2				2	0.01
Insecta, pupa			1	0		1	0.01
Insecta, unidentified	1	1	3	2	2	9	0.06
Collembola	1	67	15	88	26	197	1.33
Entomobryidae		4				4	0.03
Ephemoroptera, nymph			2			2	0.01
Odonata, nymph					1	1	0.01
Hemiptera	2	1	2			5	0.03
Cicadellidae			10	7	6	23	0.16
Aphididae					2	2	0.01
Coleoptera	5	23	13	13	5	59	0.40
Trichoptera, adult	2					2	0.01
Diptera, adult	738	1,931	3,017	900	1,438	8,024	54.24
Diptera, larva	108	16	223	39	58	444	3.00
Diptera, pupa	3	4	30	4	12	53	0.36
Ceratopagonidae, larva		1				1	0.01

Appendix Table D3. Continued.

	Estuarine mixing zone (continued)						
	2002	2003	2004	2005	2006	Total	
Taxon	n = 77	n = 167	n = 137	n = 162	n = 139	n	(%)
Chironomidae, adult	1					1	0.01
Chironomidae, larva	9	1		1	2	13	0.09
Chironomidae, pupa		3		2		5	0.03
Hymenoptera, unidentified	2	2	12			16	0.11
Formicidae	1	2	4	3	12	22	0.15
Oikopleura	12					12	0.08
Osteichthyes, bones	1	3	20	28	21	73	0.49
Osteichthyes, juvenile	1					1	0.01
Osteichthyes, larva		4			4	8	0.05
Osteichthyes, scales	2		16	14	46	78	0.53
Osteichthyes, unidentifed	1	1	9	2		13	0.09
Ammodytes hexapterus			1			1	0.01
Plantae, unidentified	50	88	96	26	50	310	2.10
Plantae, woody debris	21	24	9	23	15	92	0.62
Animalia, egg - unidentified	385	73	107	30	46	641	4.33
Animalia, larva - unidentified				3		3	0.02
Potamopyrgus antipodarum			2	6		8	0.05
Eogammarus confervicolus	1	4	21	27		53	0.36
Traskorchestia traskiana		1				1	0.01
Pseudoscorpiones					1	1	0.01
Plastic	3	5	1	11	2	22	0.15
Rock/sand	1		1	119		121	0.82
Unidentified stomach material	75	108	135	159	135	612	4.14
Total stomach items	1,875	2,865	4,221	2,253	3,579	14,793	100.00

Appendix Table D4. Marine zone.

	Marine zone						
	2002	2003	2004	2005	2006	Total	
Taxon	n = 45	n = 71	n = 74	n = 60	n = 32	n	(%)
Turbellaria	1		1	51		53	0.97
Trematoda		2				2	0.04
Nematoda	6	33	3	14	3	59	1.08
Polychaeta	2	3				5	0.09
Bivalvia			1			1	0.02
Arachnida	1	7	10	13	1	32	0.59
Acarina		2			5	7	0.13
Cladocera	1	5				6	0.11
Copepoda				8		8	0.15
Cirripedia, cyprid			36			36	0.66
Cirripedia, molt		57	47	7		111	2.04
Mysidae	1	8	1			10	0.18
Cumacea	9	72	17	32	55	185	3.40
Isopoda	2	5	3	4	6	20	0.37
Amphipoda			1			1	0.02
Gammaridea	6	37	44	51	12	150	2.75
Americorophium sp.	8	38	23	71	22	162	2.98
Americorophium salmonis	50	26	7	11	7	101	1.85
Americorophium spinicorne	31	51	28	41	5	156	2.87
Hyperiidea		1				1	0.02
Decapoda, furcilia			1	2	4	7	0.13
Decapoda, unidentified	1					1	0.02
Crangon alaskensis		1				1	0.02
Brachyura, megalops	1					1	0.02
Cancer magister, megalops	4	1	3	20	1	29	0.53
Insecta, larva				2		2	0.04
Insecta, nymph		2				2	0.04
Insecta, pupa	3	1	10			14	0.26
Insecta, unidentified	1		10	2		13	0.24
Collembola		1	3	7		11	0.20
Odonata, nymph		2	1			3	0.06
Cicadellidae		10	6	3		19	0.35
Aphididae	1	1		4		6	0.11
Coleoptera		17	9	11	2	39	0.72
Diptera, adult	79	957	618	451	202	2,307	42.37
Diptera, larva	4	1		2	2	9	0.17
Diptera, pupa	6	4	3	5	26	44	0.81
Chironomidae, larva				1		1	0.02
Chironomidae, pupa		1				1	0.02
Hymenoptera, unidentified	3	18	12			33	0.61
Hymenoptera, wasp			2			2	0.04
Formicidae		1	13	4	6	24	0.44

Appendix Table D4. Continued.

	Marine zone (continued)						
	2002	2003	2004	2005	2006	Total	
Taxon	n = 45	n = 71	n = 74	n = 60	n = 32	n	(%)
Osteichthyes, bones	10	13	66	54	48	191	3.51
Osteichthyes, juvenile	2				1	3	0.06
Osteichthyes, larva	1	2				3	0.06
Osteichthyes, scales		4	42	6		52	0.96
Hexagrammidae	1					1	0.02
Ammodytes hexapterus	2					2	0.04
Plantae, unidentified	18	67	795	38	21	939	17.25
Plantae, woody debris	21	9	39	8		77	1.41
Animalia, egg - unidentified			75	5		80	1.47
Potamopyrgus antipodarum			1			1	0.02
Eogammarus confervicolus	10	1	14			25	0.46
Plastic			7	1		8	0.15
Rock/sand			3	116	14	133	2.44
Unidentified stomach material	37	60	66	60	32	255	4.68
Total stomach items	323	1,521	2,021	1,105	475	5,445	100.00

Appendix Table D5. Stomach contents vs. total IRI (%) by zone.

Taxon	Stomach contents (%)				Mean total IRI (%) (all zones)
	Tidal freshwater		Estuarine mixing	Marine	
	Mid-estuary	Lower estuary			
Diptera, adult	10.99	55.02	74.18	51.17	47.84
Americorophium salmonis	44.87	32.69	9.98	2.08	22.41
Cladocera	34.44	5.51	0.29	0.00	10.06
Plantae, unidentified	0.25	0.09	1.62	16.32	4.57
Americorophium spinicorne	0.83	0.18	3.25	8.07	3.08
Gammaridea	0.07	0.14	1.80	6.47	2.12
Americorophium sp.	1.12	0.84	2.87	3.32	2.04
Cancer magister, megalops	-	-	0.25	2.69	1.47
Diptera, larva	2.44	2.75	0.54	0.00	1.43
Mysidae	3.07	0.92	0.62	0.07	1.17
Cirripedia, molt	-	-	0.42	1.52	0.97
Nematoda***	0.30	0.39	0.95	0.44	0.52
Osteichthyes, juvenile	-	-	0.00	0.87	0.44
Coleoptera	0.02	0.21	0.18	0.88	0.33
Eogammarus confervicolus	0.00	0.02	0.25	0.84	0.28
Isopoda	0.00	0.00	0.06	0.89	0.24
Plantae, woody debris	0.07	0.03	0.14	0.71	0.24
Osteichthyes, bones	0.00	-	0.05	0.65	0.23
Diptera, pupa	0.53	0.26	0.04	0.08	0.23
Osteichthyes, unidentifed	-	0.01	0.44	-	0.23
Polychaeta	0.00	0.00	0.43	0.37	0.20
Cumacea	0.00	0.00	0.00	0.80	0.20
Decapoda, furcilia	-	0.00	0.06	0.41	0.16
Ammodytes hexapterus	-	-	0.00	0.30	0.15
Animalia, egg - unidentified	0.01	0.04	0.50	0.02	0.14
Arachnida	0.05	0.21	0.08	0.20	0.14
Collembola	0.00	0.12	0.37	0.00	0.12
Insecta, unidentified	0.37	0.08	0.02	0.01	0.12
Cicadellidae	0.16	0.08	0.01	0.17	0.10
Rock/sand***	0.21	0.06	0.02	0.11	0.10
Hymenoptera, unidentified	-	0.16	0.01	0.10	0.09
Insecta, larva	0.04	0.00	0.25	0.00	0.07
Osteichthyes, larva	-	0.03	0.08	0.09	0.06
Osteichthyes, scales	-	0.00	0.04	0.09	0.04
Hexagrammidae	-	-	-	0.03	0.03
Turbellaria	0.02	0.01	0.08	0.02	0.03
Cirripedia, cyprid	-	-	0.00	0.06	0.03
Acarina	0.01	0.08	0.01	0.00	0.02
Formicidae	0.03	0.01	0.02	0.02	0.02
Copepoda	0.05	0.01	0.02	0.00	0.02
Crangon alaskensis	-	-	-	0.02	0.02
Odonata, nymph	0.00	0.00	0.00	0.05	0.01

Appendix Table D5. Continued.

Taxon	Stomach contents (%)				Mean total IRI (%) (all zones)
	Tidal freshwater		Estuarine mixing	Marine	
	Mid-estuary	Lower estuary			
Chironomidae, larva	-	0.03	0.00	0.00	0.01
Plastic*	-	0.00	0.01	0.02	0.01
Decapoda, megalops	-	-	0.01	-	0.01
Nereis limnicola	-	-	0.01	-	0.01
Hymenoptera, wasp	0.01	0.01	-	< 0.01	0.01
Decapoda, unidentified	0.00	< 0.01	0.01	< 0.01	0.01
Ephemoroptera, nymph	0.01	< 0.01	0.00	-	0.00
Trematoda	< 0.01	< 0.01	0.01	< 0.01	0.01
Hemiptera	< 0.01	< 0.01	0.00	-	< 0.01
Plecoptera, nymph	< 0.01	< 0.01	-	-	< 0.01
Aphididae	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
Insecta, pupa	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
Chironomidae, pupa	-	< 0.01	< 0.01	< 0.01	< 0.01
Bivalvia	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
Amphipoda	-	< 0.01	< 0.01	< 0.01	< 0.01
Chironomidae, adult	-	< 0.01	< 0.01	-	< 0.01
Potamopyrgus antipodarum	-	-	< 0.01	< 0.01	< 0.01
Insecta, nymph	-	< 0.01	< 0.01	< 0.01	< 0.01
Oikopleura	-	-	< 0.01	-	< 0.01
Ephemoroptera, adult	< 0.01	< 0.01	-	-	< 0.01
Neomysis mercedis	-	-	< 0.01	-	< 0.01
Hyperiidea	-	-	< 0.01	< 0.01	< 0.01
Animalia, larva - unidentified	< 0.01	-	< 0.01	-	< 0.01
Brachyura, megalops	-	-	< 0.01	< 0.01	< 0.01
Trichoptera, adult	-	< 0.01	< 0.01	-	< 0.01
Entomobryidae	-	-	< 0.01	-	< 0.01
Oligochaeta	-	< 0.01	< 0.01	-	< 0.01
Traskorchestia traskiana	-	-	< 0.01	-	< 0.01
Platyhelminthes	-	< 0.01	-	-	< 0.01
Pseudoscorpiones	-	-	< 0.01	-	< 0.01
Ceratopagonidae, larva	-	< 0.01	< 0.01	-	< 0.01
Tricoptera, larva	-	< 0.01	-	-	< 0.01
Dermaptera	-	< 0.01	-	-	< 0.01

* Not a food item

Appendix Table D6. Diet of Chinook salmon sampled from Lower Elochoman Slough, a tidal freshwater, mid-estuary station. Prey items are listed in descending order of IRI contribution summed over all months, indicating stomach contents in descending order of overall importance in the diet.

	Index of relative importance (%)										
	Lower Elochoman Slough										
Taxon	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Diptera, adult	74.74	79.11	30.57	20.51	36.30	82.99	84.36	7.94	84.96	24.28	43.97
Americorophium salmonis	8.61	4.98	48.77	70.91	57.95	6.33	13.78	17.65	8.68	71.31	36.61
Cladocera	1.59	0.49	0.09	5.21	0.36	1.16	0.05	65.77	-	-	-
Diptera, larva	3.88	1.47	3.95	0.53	0.52	4.75	0.12	0.31	0.04	0.01	6.11
Mysidae	0.07	1.09	0.20	0.79	1.90	1.05	0.19	6.98	0.01	0.01	-
Gammaridea	1.29	2.43	4.66	0.09	<0.01	0.06	0.09	-	0.01	-	3.36
Nematoda*	1.56	4.51	1.61	0.67	0.62	0.24	0.12	0.11	1.10	0.41	-
Americorophium sp.	1.36	1.01	4.29	0.49	0.55	0.74	0.34	0.16	0.33	0.43	-
Plantae, woody debris	-	0.02	-	0.00	0.03	0.07	<0.01	0.01	0.06	0.85	6.33
Osteichthyes, unidentifed	-	2.44	1.65	-	-	-	-	-	-	-	-
Plantae, unidentified	0.06	-	1.35	0.05	0.05	0.16	0.01	0.02	0.04	0.43	1.11
Hymenoptera, unidentified	-	-	-	-	0.04	0.03	0.03	<0.01	2.75	0.20	-
Eogammarus confervicolus	2.90	-	0.03	0.01	<0.01	-	<0.01	-	0.01	-	-
Arachnida	0.10	-	0.02	0.01	0.44	0.62	0.11	<0.01	0.14	1.25	-
Collembola	1.48	0.72	0.06	0.11	0.10	0.12	0.03	0.03	0.01	0.01	-
Americorophium spinicorne	0.79	-	0.84	0.41	0.13	0.06	0.01	0.01	0.02	-	-
Insecta, unidentified	0.01	0.07	-	0.02	0.16	0.32	0.05	<0.01	<0.01	0.01	1.56
Diptera, pupa	0.88	-	-	0.01	0.12	0.17	0.02	-	0.95	-	-
Plecoptera, nymph	-	-	1.44	-	-	-	-	-	-	-	-
Acarina	0.16	-	0.02	<0.01	0.08	0.26	0.31	0.01	0.60	-	-
Coleoptera	-	-	-	0.06	0.42	0.53	0.14	-	0.09	-	-
Polychaeta	0.03	1.01	-	-	-	-	-	-	-	-	-
Cicadellidae	-	0.11	0.22	0.05	0.02	0.10	0.09	0.00	0.11	0.34	-
Plastic*	-	-	-	-	-	-	-	-	-	-	0.95
Animalia, egg - unidentified	0.02	-	-	0.01	-	-	0.01	0.88	-	-	-
Rock/sand*	0.10	0.05	-	0.01	0.04	0.07	0.00	0.02	0.05	0.37	-

Appendix Table D6. Continued.

	Index of relative importance (%) **Lower Elochoman Slough (continued)**										
Taxon	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Copepoda	-	-	0.16	<0.01	-	0.02	-	0.06	<0.01	-	-
Decapoda, unidentified	0.20	-	-	-	-	-	-	<0.01	-	-	-
Osteichthyes, larva	-	0.19	-	-	-	-	-	-	-	-	-
Chironomidae, larva	0.03	-	0.02	-	0.06	<0.01	0.02	-	-	-	-
Isopoda	-	0.07	0.05	-	-	<0.01	-	-	<0.01	-	-
Osteichthyes, scales	0.05	0.07	-	-	-	-	-	-	-	-	-
Hymenoptera, wasp	0.02	-	-	0.00	0.00	0.01	0.03	-	0.03	-	-
Hemiptera	-	-	-	-	<0.01	0.02	0.02	<0.01	<0.01	0.06	-
Insecta, larva	0.03	0.03	-	<0.01	0.01	0.00	-	-	<0.01	0.01	-
Oligochaeta	-	0.07	-	<0.01	-	-	-	-	-	-	-
Ephemoroptera, adult	-	0.06	-	<0.01	-	-	-	-	-	-	-
Turbellaria	-	-	-	<0.01	0.01	0.02	0.01	0.01	0.00	-	-
Insecta, pupa	-	-	-	-	0.04	0.00	<0.01	-	-	-	-
Trematoda	-	-	-	-	0.01	0.03	<0.01	0.00	-	-	-
Odonata, nymph	-	-	-	0.03	-	<0.01	0.01	-	-	-	-
Chironomidae, adult	-	-	-	-	-	<0.01	0.04	-	-	-	-
Formicidae	-	-	-	-	0.00	0.02	-	-	-	0.01	-
Decapoda, furcilia	-	-	-	-	0.03	-	<0.01	-	-	-	-
Chironomidae, pupa	-	-	-	-	-	0.01	0.01	-	-	-	-
Amphipoda	0.01	-	-	-	-	-	-	-	-	-	-
Trichoptera, adult	-	-	-	-	-	<0.01	-	0.01	-	-	-
Ephemoroptera, nymph	-	-	-	-	-	0.01	-	-	-	-	-
Insecta, nymph	-	-	-	0.01	-	-	-	-	-	-	-
Cumacea	-	-	-	-	<0.01	-	-	-	-	-	-
Ceratopagonidae, larva	-	-	-	-	-	<0.01	-	-	-	-	-

* Not a food item

Appendix Table D7. Diet of Chinook salmon sampled from Pt. Adams Beach, an estuarine mixing station. Prey items are listed in descending order of IRI contribution summed over all months, indicating stomach contents in descending order of overall importance in the diet.

	Index of relative importance (%) **Pt. Adams Beach**										
Taxon	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Diptera, adult	34.92	82.15	45.69	73.08	45.15	83.47	86.60	75.84	85.64	64.31	40.06
Mysidae	62.27	5.63	28.48	0.02	0.01	<0.01	-	0.44	-	-	-
Americorophium salmonis	-	0.23	13.91	8.76	41.97	1.15	0.89	0.02	0.03	0.08	0.06
Plantae, unidentified	-	0.06	0.01	0.08	0.29	0.47	0.47	9.48	8.13	13.68	1.88
Cirripedia, molt	-	-	-	-	0.01	0.03	-	4.36	1.04	0.99	26.53
Osteichthyes, unidentifed	-	-	-	0.23	-	-	-	-	-	0.36	20.38
Americorophium spinicorne	-	0.39	0.03	1.21	3.90	3.76	3.12	0.99	1.80	-	0.05
Americorophium SP.	-	0.60	2.57	1.08	3.60	0.76	0.83	1.40	0.02	2.95	-
Polychaeta	2.81	-	0.01	0.55	0.12	0.00	-	0.05	-	9.64	-
Collembola	-	5.58	0.40	5.85	0.17	0.22	0.02	0.03	-	0.03	-
Gammaridea	-	0.24	4.88	1.33	0.65	2.02	0.62	0.25	0.36	1.08	-
Nematoda***	-	0.56	0.05	0.04	0.08	1.89	4.60	0.02	0.44	2.66	0.46
Turbellaria	-	-	-	0.01	-	0.03	0.01	3.18	0.01	0.60	5.75
Animalia, egg - unidentified	-	0.15	0.93	1.21	0.28	3.85	-	-	-	0.25	-
Diptera, larva	-	2.96	0.60	0.43	0.42	0.91	0.49	-	-	0.72	-
Insecta, larva	-	1.17	1.54	0.04	0.01	0.01	<0.01	0.01	1.55	-	-
Cladocera	-	0.08	<0.01	2.93	0.23	0.28	-	-	-	-	-
Osteichthyes, scales	-	-	-	-	0.01	-	1.41	0.96	0.26	-	-
Arachnida	-	-	0.05	0.04	0.06	0.09	0.03	-	0.01	0.23	2.00
Cancer magister, megalops	-	-	-	1.80	0.28	0.09	0.02	0.19	0.01	-	-
Osteichthyes, bones	-	-	-	-	0.00	-	0.38	1.72	0.09	0.02	-
Eogammarus confervicolus	-	0.02	0.02	0.03	0.97	0.17	-	0.17	0.01	0.20	0.06
Coleoptera	-	-	<0.01	0.09	0.13	0.20	0.02	0.09	0.13	0.78	0.05

Appendix Table D7. Continued.

	Index of relative importance (%) **Pt. Adams Beach (continued)**										
Taxon	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Plantae, woody debris	-	-	0.02	0.04	0.75	0.03	0.10	0.05	0.10	0.11	0.05
Osteichthyes, larva	-	0.10	-	0.24	0.17	-	-	0.01	-	0.60	-
Isopoda	-	-	0.01	-	0.01	0.05	<0.01	0.54	0.23	0.05	0.10
Decapoda, furcilia	-	-	-	0.09	0.01	0.01	-	-	-	-	0.84
Decapoda, unidentified	-	-	-	-	-	-	-	-	-	-	0.79
Diptera, pupa	-	-	-	-	0.33	0.08	0.06	0.02	-	0.24	-
Copepoda	-	0.02	0.66	-	-	-	-	-	-	-	-
Hemiptera	-	-	-	-	-	-	-	0.01	-	-	0.64
Rock/sand***	-	-	-	0.37	0.15	-	-	-	-	-	0.05
Nereis limnicola	-	-	-	0.36	-	-	-	-	-	-	-
Plastic***	-	-	-	-	0.04	<0.01	<0.01	0.07	0.03	0.20	-
Formicidae	-	-	-	-	-	0.09	-	-	0.03	0.16	-
Insecta, unidentified	-	-	-	-	0.01	0.20	-	0.04	-	-	-
Trematoda	-	-	0.01	0.01	-	0.01	0.18	-	-	-	-
Cicadellidae	-	0.03	0.12	-	-	<0.01	<0.01	-	0.03	-	-
Hymenoptera, unidentified	-	-	-	0.01	-	0.01	0.02	-	0.02	0.03	0.08
Trichoptera, adult	-	-	-	-	-	-	-	-	-	-	0.12
Decapoda, megalops	-	-	-	-	0.11	-	-	-	-	-	-
Acarina	-	0.01	-	-	-	0.07	0.02	-	-	-	-
Amphipoda	-	-	-	-	-	<0.01	-	0.02	0.03	0.03	-
Ammodytes hexapterus	-	-	-	0.07	-	-	-	-	-	-	-
Chironomidae, larva	-	-	-	-	0.02	-	0.03	-	<0.01	-	-
Bivalvia	-	-	-	-	-	-	-	-	-	-	0.05
Oikopleura	-	-	-	-	-	-	0.04	-	-	-	-
Potamopyrgus antipodarum	-	-	-	-	0.01	-	-	0.02	-	-	-

Appendix Table D7. Continued.

	Index of relative importance (%) **Pt. Adams Beach (continued)**										
Taxon	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Osteichthyes, juvenile	-	-	-	-	0.03	-	-	-	-	-	-
Oligochaeta	-	0.02	-	-	-	-	-	-	-	-	-
Animalia, larva - unidentified	-	-	0.02	-	-	-	-	-	-	-	-
Cumacea	-	-	-	-	0.01	0.01	<0.01	-	-	-	-
Insecta, pupa	-	-	-	-	-	-	-	-	0.02	-	-
Neomysis mercedis	-	-	-	-	-	0.01	-	-	-	-	-
Chironomidae, pupa	-	-	-	-	-	-	0.01	-	-	-	-
Ceratopagonidae, larva	-	-	-	-	-	-	-	<0.01	-	-	-
Ephemoroptera, nymph	-	-	-	-	-	0.01	-	-	-	-	-
Odonata, nymph	-	-	-	<0.01	-	-	-	-	-	-	-
Aphididae	-	-	-	-	-	<0.01	<0.01	-	-	-	-
Insecta, nymph	-	-	-	<0.01	-	-	-	-	-	-	-
Pseudoscorpiones	-	-	-	-	-	-	<0.01	-	-	-	-
Brachyura, megalops	-	-	-	-	<0.01	-	-	-	-	-	-
Traskorchestia traskiana	-	-	-	-	<0.01	-	-	-	-	-	-
Cirripedia, cyprid	-	-	-	-	<0.01	-	-	-	-	-	-
Entomobryidae	-	-	-	-	<0.01	-	-	-	-	-	-
Chironomidae, adult	-	-	-	-	<0.01	-	-	-	-	-	-
Hyperiidea	-	-	-	-	-	<0.01	-	-	-	-	-

*** = not a food item

APPENDIX E

Wetland Diet and Prey Resource Data, 2002-2007

Appendix Table E1. Comprehensive summary of samples collected in Columbia River estuary wetlands for juvenile Chinook salmon diets, consumption rates and diets, and prey resource availability (insect fall-out and benthic), 2002-2007.

Site		Sample Type	2002	2003	2004	2005	2006	2007	Total
Russian Is.	North	Diet	19	25	16		2	4	66
		Insect Fallout	15	20	21	30			86
		Benthic	10	13	14	30			67
	South	Diet	66	47	65	45	32	26	281
		Insect Fallout	30	20	24	30	12	25	141
		Benthic	30	15	15	30	24	25	139
		Consumption Diet				92	106	96	294
Seal Is.	North	Diet	3	50					53
		Insect Fallout	14	20					34
		Benthic	10	15					25
	South	Diet	3	35					38
		Insect Fallout	15	20					35
		Benthic	10	14					24
Karlson Is.	Shrub	Diet	22	17	31				70
		Insect Fallout	15	19	20				54
		Benthic	8	15	15				38
	Forest	Diet	31	27					58
		Insect Fallout	15	20					35
		Benthic	8	15					23
Welch Is.	North	Diet			63	30			93
		Insect Fallout			20	29			49
		Benthic			14	30			44
	South	Diet			62				62
		Insect Fallout			19	30			49
		Benthic			13	28			41
Wallace Is.	Lower	Diet					15		15
		Insect Fallout					24	25	49
		Benthic					24	24	48
	Upper	Diet					28	25	53
		Insect Fallout					14	25	39
		Benthic					25	25	50
Lord Is.	Lower	Diet					31	1	32
		Insect Fallout					20	24	44
		Benthic					18	25	43
		Consumption Diet						65	65
	Upper	Diet					28	30	58
		Insect Fallout					15	25	40
		Benthic					20	25	45
		Total	324	407	412	404	438	495	2480

Appendix Table E2. Comprehensive list of all taxa present in the diets of juvenile Chinook salmon and prey resource samples (insect fallout trap and macrobenthic core) collected in the Columbia River estuary wetlands, 2002-2007. See text for details on methodology.

Phylum Class Order Family *Genus species*	Life History or Parts	Diet Chinook n=872	Benthic 500µm 587	Fallout 106µm 655
Annelida	juv-adult	X	X	X
Clitellata (subclass Hirudinea)	juv-adult		X	
Oligochaeta	juv-adult	X	X	X
	egg		X	
	egg case		X	
Polychaeta	juv-adult	X	X	
Capitella spp.	juv-adult		X	
Hobsonia spp.	juv-adult		X	
Manayunkia spp.	juv-adult		X	
Nereididae	juv-adult	X	X	
Spionidae	larva		X	
Arthropoda		X		
Arachnida				
Acari	juv-adult	X	X	X
Araneae	juv-adult	X		X
Opiliones	juv-adult			X
Psuedoscorpiones	juv-adult	X		X
Arthropoda (subphylum Crustacea)				
Branchiopoda				
suborder Cladocera	adult	X	X	
Bosmina spp.	adult	X		
Daphnia spp.	adult	X		
Malacostraca				
Amphipoda (suborder Gammaridea)	juv-adult	X	X	
Americorophium salmonis	juv-adult	X	X	
Americorophium spinicorne	juv-adult	X	X	
Americorophium spp.	juv-adult	X	X	
Crangonyx spp.	juv-adult		X	
Allorchestes spp.	juv-adult	X	X	
Hyalella azteca	juv-adult	X	X	
Anisogammaridae	juv-adult	X	X	
Cumacea	juv-adult		X	
Decapoda (infraorder Astacura)	juv-adult		X	
Isopoda	juv-adult	X	X	
Caecidotea spp.	juv-adult	X	X	
Gnorimosphaeroma oregonense	juv-adult		X	
suborder Oniscoidea	juv-adult			X
Mysidacea	juv-adult	X		
Neomysis mercedis	juv-adult	X		
Maxillopoda (subclass Copepoda)	adult	X	X	
Calanoida	adult	X	X	
Cyclopoida	adult	X	X	
Harpacticoida	adult	X	X	
Ostracoda	adult	X	X	

Appendix Table E2. Continued.

Phylum Class Order Family *Genus species*	Life History or Parts	Diet Chinook n=872	Benthic 500µm 587	Fallout 106µm 655
Arthropoda (subphylum Hexapoda)				
Insecta	larva	X	X	X
	pupa	X	X	X
	nymph	X		
	adult	X		
Coleoptera	larva	X	X	X
	adult	X	X	X
Apionidae	adult			2
Bruchidae	adult			X
Cantharidae	adult	X		X
Carabidae	adult	X		X
Chrysomelidae	adult			X
Cleridae	adult			X
Coccinellidae	adult			X
Dytiscidae	larva		X	
Elateridae	adult			X
Elmidae	larva		X	
Heteroceridae	adult	X		X
Hydrophilidae	adult			X
Lampyridae	adult			X
Lathrididae	adult			X
Mordellidae	adult			X
Pselaphidae	adult			X
Ptilidae	adult			X
Scarabaeidae	adult			X
Staphylinidae	adult	X		X
superfamily Curculionoidea	adult			X
Curculionidae	adult			X
Collembola	adult	X	X	X
Entomobryidae	adult			X
Hypogastruridae	adult		X	X
Isotomidae	adult	X	X	X
Onychiuridae	adult		X	
Sminthuridae	adult	X	X	X
Diptera	larva	X	X	X
	pupa	X	X	X
	adult	X		X
Agromyzidae	adult			X
Anthomyiidae	adult			X
Anthomyzidae	adult			X
Bibionidae	adult			X
Calliphoridae	adult			X
Cecidomyiidae	adult	X		X
Ceratopogonidae	larva	X	X	
	pupa	X	X	X
	adult	X	X	X

Appendix Table E2. Continued.

Phylum / Class / Order / Family / *Genus species*	Life History or Parts	Diet Chinook n=872	Benthic 500µm 587	Fallout 106µm 655
Diptera (continued)				
Chionomidae	larva	X	X	X
	pupa	X	X	X
	adult	X	X	X
Chloropidae	adult			X
Dolichopodidae	adult	X		X
	larva	X	X	
Drosophilidae	adult	X		X
Empididae	adult	X		X
Ephydridae	adult	X		X
Lonchaeidae	adult			X
Lonchopteridae	adult			X
Micropezidae	adult			X
Muscidae	adult	X		X
Mycetophilidae	adult	X		X
Otitidae	adult			X
Phoridae	adult	X		X
Psychodidae	larva	X		X
	pupa	X		
	adult	X		X
Ptychopteridae	adult	X		X
Rhagionidae	adult			X
Sarcophagidae	adult			X
Scathophagidae	adult			X
Sciaridae	adult	X		X
Sciomyzidae	adult			X
Sphaeroceridae	adult			X
Syrphidae	adult			X
Tabanidae	adult			X
Tachinidae	adult			X
Tipulidae	larva		X	
	adult	X		X
Ephemeroptera	nymph	X	X	
	subimago			X
	adult	X		X
Caenidae	subimago			X
Ephemeridae				
Hexagenia spp.	nymph	X	X	
Heptageneiidae	nymph		X	
Hemiptera	nymph	X	X	X
	adult	X		X
Aleyrodidae	adult			X
Anthocoridae	adult			X
Aphididae	adult	X		X
Cercopidae	adult	X		X

Appendix Table E2. Continued.

Phylum Class Order Family *Genus species*	Life History or Parts	Diet Chinook n=872	Benthic 500µm 587	Fallout 106µm 655
Hemiptera (continued)				
Cicadellidae	adult	X		X
	nymph	X		X
	nymph-adult			X
Corixidae	adult	X		X
	nymph		X	X
Delphacidae	adult	X		X
	nymph	X		X
	nymph-adult	X		X
Gerridae	adult			X
	nymph	X		
Lygaeidae	adult			X
Miridae	adult			X
Nabidae	adult			X
Psyllidae	adult	X		X
	nymph	X		X
	nymph-adult			X
Saldidae	adult	X		X
Tingidae	adult			X
superfamily Aphidoidea	nymph	X		X
	adult	X		X
superfamily Coccoidea	adult	X		X
Hymenoptera	adult	X		X
	larva			X
	pupa		X	
superfamily Apoidea	adult			X
Apidae	adult			X
Megachilidae	adult			X
superfamily Ceraphronoidea				
Ceraphronidae	adult			X
Megaspilidae	adult			X
superfamily Chalcidoidea	adult	X		X
Mymaridae	adult			X
superfamily Cynipoidea	adult			X
Cynipidae	adult			X
superfamily Ichneumonoidea	adult	X		X
Braconidae	adult	X	X	X
Ichneumonidae	adult	X		X
superfamily Proctotrupoidea	adult	X		X
Diapriidae	adult			X
superfamily Sphecoidea	adult			2
Sphecidae	adult			X
superfamily Tenthredinoidea				
Tenthredinidae	adult			X
suprefamily Vespoidea				
Formicidae	adult	X		X

Appendix Table E2. Continued.

Phylum Class Order Family *Genus species*	Life History or Parts	Diet Chinook n=872	Benthic 500µm 587	Fallout 106µm 655
Lepidoptera	larva	X	X	X
	adult			X
suborder Microlepidoptera	adult	X		X
Neuroptera	larva		X	X
	adult			X
Coniopterygidae	adult			X
Sialidae	larva	X	X	X
	pupa	X		
	adult	X		X
Odonata	nymph	X	X	
suborder Zygoptera	adult			X
Coenagrionidae	adult	X		X
Orthoptera				
Tetrigidae	adult			X
Plecoptera	nymph	X		
	adult			X
Nemouridae	adult			X
Psocoptera	nymph			X
	adult	X		X
Siphonaptera	adult			X
Thysanoptera	larva	X		X
	adult	X		X
Phlaeothripidae	larva			X
	adult	X		X
Thripidae	adult			X
Trichoptera	larva	X	X	X
	pupa		X	
	adult	X		X
Calamoceratidae	adult			X
Hydroptilidae	adult	X		X
Lepidostomatidae	adult	X		X
Limnephilidae	adult			X
Arthropoda (subphylum Myriapoda)				
Diplopoda	adult			X
Polyxenida	adult			X
Chordata				
superclass Osteichthyes	larva-juv-adult	X		
Cnidaria				
Hydrozoa				
Hydroida	adult		X	
Mollusca	juv-adult		X	
Bivalvia	juv-adult	X	X	
Anodonta oregonesis	juv-adult		X	
Corbicula fluminea	juv-adult		X	
Gastropoda	juv-adult		X	
Pulmonata	juv-adult			X

Appendix Table E2. Continued.

Phylum Class Order Family *Genus species*	Life History or Parts	Diet Chinook n=872	Benthic 500µm 587	Fallout 106µm 655
Nematoda	adult	X	X	
Platyhelminthes Turbellaria	adult		X	
other Fish scale		X		
Plant matter		X		
Exuvia		X		
Inorganic		X		
Unidentified egg		X	X	

APPENDIX F

Chinook Salmon Genetic Baseline and Stock Assignments

Appendix Table F1. Chinook salmon populations used as baseline data in genetic stock identification analysis in this study. Genetic stock group, source, run time, and sample size are given. Genetic data are from Seeb et al. (2007) except where noted.

Genetic stock group	Source	Run time	Sample size
Snake River Spring/Summer	Imnaha River	Summer	144
	Minam River	Spring	144
	Rapid River Hatchery	Spring	144
	Secech River	Summer	144
	Tucannon River3	Spring	136
	Tucannon Hatchery	Spring	42
	Newsome Creek4	Spring	95
	West Fork Yankee Creek4	Spring	60
Snake River Fall	Lyons Ferry Hatchery	Fall	186
Mid and Upper Columbia River Spring	Carson Hatchery	Spring	144
	John Day River	Spring	143
	Upper Yakima River	Spring	199
	Warm Springs Hatchery	Spring	143
	Wenatchee River	Spring	62
	Wenatchee Hatchery3	Spring	49
Upper Columbia River Summer/Fall	Hanford Reach	Fall	284
	Methow River	Summer	143
	Wells Hatchery	Summer	144
	Wenatchee River3	Summer	135
Deschutes River Fall	Lower Deschutes River	Fall	144
	Upper Deschutes River2	Fall	144
Spring Creek Group Tule Fall	Spring Creek Hatchery	Fall	144
	Big Creek Hatchery1	Fall	99
	Elochoman River1	Fall	95
	Willamette River1	Fall	46
Willamette River Spring	North Fork Clackamas River1	Spring	80
	North Santiam Hatchery	Spring	143
	North Santiam River1	Spring	96
	McKenzie Hatchery	Spring	142
	McKenzie River1	Spring	98

Appendix Table F1. Continued.

Genetic stock group	Source	Run time	Sample size
West Cascade Tributary Spring	Cowlitz Hatchery	Spring	140
	Kalama Hatchery	Spring	144
	Lewis Hatchery	Spring	144
West Cascade Tributary Fall	Cowlitz Hatchery	Fall	140
	Lewis River	Fall	93
	Sandy River	Fall	124
Rogue River	Cole Rivers Hatchery	Spring	142
	Applegate River	Fall	143
Washington and Oregon coastal	Forks Creek Hatchery3	Fall	142
	Humptulips Hatchery3	Fall	83
	Necanicum River5	Fall	77
	Nehalem River5	Fall	151
	Kilchis River5	Fall	58
	Wilson River5	Fall	139
	Trask River5	Fall	162

[1] Northwest Fisheries Science Center, unpublished data.

[2] Columbia River Inter-Tribal Fish Commission, unpublished data.

[3] Washington Department of Fish and Wildlife, unpublished data.

[4] Narum et al. (2007)

[5] Oregon State University, unpublished data

Appendix Table F2. Genetic stock assignments of juvenile Chinook salmon sampled at Columbia River estuary beach seine sites, 2002-2006. Only fish with assignment probabilities of ≥0.90 are given.

	Marine		Mixing			Tidal Freshwater lower estuary			
Stock group	Spring	Summer/ fall	Spring	Summer/ fall	Winter	Spring	Summer/ fall	Winter	Total
Snake Spring/Summer	1								1
Snake Fall	1	1	1	1					4
Upper CR Summer/Fall	2	17	14	20	5	8	17		83
Spring Cr Group Fall	98	7	221	11	18	150	15	19	539
Willamette Spring	4	1	9	1		5		3	23
West Cascade Spring	1		7			2		2	12
West Cascade Fall	12	56	111	133	13	261	262	31	879
Rogue	2	8	2	26	3	2			43
Coastal		4	3	9	3				19
Total	121	94	368	201	42	428	294	55	1,603

APPENDIX G

Chinook Salmon Hatchery Releases, 2002-2006

Appendix Table G. Numbers of Chinook salmon juveniles released in the Columbia River Basin and fraction marked each year, 2002-2006. Fish were marked with adipose fin clips, coded-wire tags, or both. Data from Regional Mark Processing Center (RMPC 1977). Stocks are grouped based on descriptions provided by NOAA Fisheries (NMFS 2003). Releases of subyearling interior Columbia River Basin spring Chinook salmon are not included.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2002				
Snake Fall				
F Snake R	Oxbow	Upper Snake	0.172	1.000
Lyons Ferry H	Lyons Ferry	Clearwater	0.495	0.395
Lyons Ferry H	Lyons Ferry	Snake General	0.998	0.368
Lyons Ferry H	Lyons Ferry	Upper Snake	0.399	0.503
Snake R-Lower	Lyons Ferry	Lower Snake	0.025	0.000
Snake R-Lower	Lyons Ferry	Snake below Grande Ronde	0.224	0.870
Tucannon R	Lyons Ferry	Lower Snake	0.021	1.000
Total			2.333	0.493
Upper Columbia Summer				
Wells H	Turtle Rock	Columbia at Turtle Rock	0.695	0.598
Wells H	Wells	Columbia near Wells	0.376	1.000
Wells H	Wells	Lake Chelan	0.056	0.000
Total			1.127	0.702
Upriver Bright Fall				
Bonneville Pool	Ringold Springs	CR McNary to Priest Rapids	2.283	0.189
Columbia Upriver	Bonneville	Tanner Creek	5.205	0.021
Columbia Upriver	Little White Salmon	Little White Salmon	2.074	0.096
Columbia Upriver	Little White Salmon	Yakima	1.698	0.118
Hanford Reach Stock	NA	Columbia at Hanford Reach	0.128	1.000
Priest Rapids	Klickitat (YKFP)	Klickitat	2.173	0.093
Priest Rapids	Priest Rapids	Columbia at Priest Rapids	6.779	0.032
Washington Brights	Umatilla	Umatilla	0.620	1.000
Total			20.960	0.101
Deschutes Fall	NA	Deschutes	0.011	1.000
Spring Creek Group Fall				
Big Cr H	Big Creek	Big Creek	5.766	0.039
Elochoman R	Elochoman	Elochoman	1.715	0.054
Spring Creek	Spring Creek	Spring Creek	19.126	0.023
Total			26.607	0.029

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2002 (continued)				
West Cascade Fall				
Cowlitz R	Cowlitz	Cowlitz	5.827	0.036
Kalama R	Fallert Creek	Kalama	2.434	0.037
Kalama R	Kalama Falls	Kalama	2.565	0.035
Lewis R	NA	Lewis	0.099	1.000
Toutle R	North Toutle	Cowlitz	1.361	0.033
Washougal R	Elochoman	Elochoman	0.303	0.000
Washougal R	Washougal	Washougal	4.145	0.024
Total			16.733	0.038
West Cascade Spring				
Cowlitz	Cowlitz	Upper Cowlitz	0.498	0.000
Kalama	Kalama Falls	Kalama	0.006	1.000
Lewis	Speelyai	Cowlitz Lake Merwin	0.060	0.000
Total			0.564	0.011
Willamette Spring				
Clackamas Early	Clackamas	Clackamas	0.645	0.978
McKenzie	McKenzie	McKenzie	0.356	0.994
Santiam SF	South Santiam	Santiam	0.298	0.995
Willamette MF	Dexter Ponds	MF Willamette	0.318	0.983
Total			1.618	0.985
Rogue River Fall				
Cole Rivers H	Big Creek	N FK Klaskanine	0.621	0.993
Cole Rivers H	CEDC Youngs Bay Net	Youngs River and Bay	0.467	0.989
Total			1.088	0.991
Unknown / other				
Bonneville Ladder	Nat Prod Tagging	Lower Columbia	0.003	1.000
Total			0.003	1.000
		2002 Total	71.044	0.115
Release Year 2003				
Snake Fall				
Lyons Ferry H	Lyons Ferry	Clearwater	0.621	0.487
Lyons Ferry H	Lyons Ferry	Snake General	0.809	0.473
Lyons Ferry H	Lyons Ferry	Upper Snake	0.705	0.707
Lyons Ferry H	Nez Perce	Clearwater	0.480	0.902
Snake R	Oxbow	Upper Snake	0.209	1.000
Snake R-Lower	Lyons Ferry	Snake below Grande Ronde	0.334	0.900
Total			3.158	0.673
Upper Columbia Summer				
Wells H	Turtle Rock	Columbia at Turtle Rock	1.021	0.396
Wells H	Wells	Columbia near Wells	0.473	1.000
Total			1.494	0.587

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2003 (continued)				
Upriver Bright Fall				
Bonneville Pool	Ringold Springs	Columbia general	3.323	0.059
Columbia Upriver	Bonneville	Tanner Creek	4.999	0.023
Columbia Upriver	Little White Salmon	Little White Salmon	2.084	0.096
Hanford Reach Stock	NA	Columbia at Hanford Reach	0.259	1.000
Little White Salmon	Little White Salmon	Yakima	1.771	0.041
Priest Rapids	Klickitat (YKFP)	Klickitat	3.391	0.063
Priest Rapids	Priest Rapids	Columbia at Priest Rapids	6.778	0.052
Umatilla R	Umatilla	Umatilla	0.625	1.000
Total			23.230	0.088
Deschutes Fall	NA	Deschutes	0.042	1.000
Spring Creek Group Fall				
Big Creek H	Big Creek	Big Creek	5.765	0.038
Elochoman R	Elochoman	Elochoman	2.055	0.045
Total			7.820	0.040
West Cascade Fall				
Cowlitz R	Cowlitz	Cowlitz	5.216	0.039
Kalama R	Fallert Creek	Kalama	2.382	0.038
Kalama R	Kalama Falls	Kalama	2.546	0.033
Lewis R NF	NA	Lewis	0.098	1.000
Toutle R	North Toutle	Cowlitz	2.593	0.035
Washougal R	Washougal	Washougal	3.948	0.025
Total			16.783	0.040
West Cascade Spring				
Cowlitz	Cowlitz	Upper Cowlitz	0.943	0.000
Total			0.943	0.000
Willamette Spring				
McKenzie	McKenzie	McKenzie	0.352	0.989
Santiam SF	South Santiam	Santiam	0.300	0.988
Santiam SF	Willamette	Molalla	0.035	1.000
Willamette MF	Dexter Ponds	MF Willamette	0.318	0.974
Willamette MF	Willamette	MF Willamette	0.094	1.000
Total			1.386	0.985
Rogue River Fall				
Cole Rivers H	Big Creek	N FK Klaskanine	0.702	0.077
Cole Rivers H	CEDC Youngs Bay Net	Youngs River and Bay	0.780	0.506
Total			1.483	0.303
Unknown / other				
Bonneville Ladder	Nat. Prod. Tagging	Lower Columbia	0.010	1.000
Chinook R	Sea Resources	Chinook	0.020	1.000
Total			0.030	1.000
		2003 Total	56.369	0.140

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2004				
Snake Fall				
Lyons Ferry H	Lyons Ferry	Clearwater	0.201	1.000
Lyons Ferry H	Lyons Ferry	Snake General	0.202	1.000
Lyons Ferry H	Nez Perce	Clearwater	0.170	1.000
Snake R	Oxbow	Upper Snake	0.175	1.000
Snake R-Lower	Lyons Ferry	Lyons Ferry	0.202	1.000
Total			0.950	1.000
Upper Columbia Summer				
Methow R	NA	Methow	0.005	0.405
Wells H	Turtle Rock	Columbia general	0.781	0.508
Wells H	Wells	Columbia near Methow	0.215	1.000
Wells H	Wells	Columbia general	0.211	1.000
Wells H	Wells	Lake Chelan	0.050	0.000
Total			1.261	0.653
Upriver Bright Fall				
Bonneville Pool	Ringold Springs	Columbia general	3.007	0.071
Columbia R Upriver	Bonneville	Tanner Creek	0.701	1.000
Columbia R Upriver	Little White Salmon	Little White Salmon	2.032	0.099
Hanford Reach Stock	NA	Columbia at Hanford Reach	0.208	1.000
Priest Rapids	Klickitat (YKFP)	Klickitat	4.226	0.153
Priest Rapids	Priest Rapids	Columbia general	6.815	0.059
Washington Brights	Umatilla	Umatilla	0.611	1.000
Total			17.600	0.169
Deschutes Fall	NA	Deschutes	0.088	1.000
Spring Creek Group Fall				
Big Creek H	Big Creek	Big Creek	5.888	0.039
Elochoman R	Elochoman	Elochoman	1.910	0.047
Spring Creek	Spring Creek	Spring Creek	14.654	0.032
Total			22.451	0.035
West Cascade Fall				
Cowlitz R	Cowlitz	Cowlitz	5.016	0.041
Kalama R	Fallert Creek	Kalama	2.226	0.041
Kalama R	Kalama Falls	Kalama	2.503	0.036
Lewis R NF	NA	Lewis	0.064	1.000
Toutle R	North Toutle	Cowlitz	2.083	0.044
Washougal R	Washougal	Washougal	3.685	0.025
Total			15.577	0.041
West Cascade Spring				
Cowlitz	Cowlitz	Upper Cowlitz	0.289	0.000
Total			0.289	0.000
Willamette Spring				
Clackamas Late	Clackamas	Clackamas	0.304	0.982
McKenzie	McKenzie	McKenzie	0.355	0.995
Santiam SF	South Santiam	Santiam	0.055	1.000
Willamette MF	Willamette	MF Willamette	0.312	1.000
Total			1.025	0.993

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2004 (continued)				
Rogue River Fall				
Cole Rivers H	Big Creek	N FK Klaskanine	0.679	0.081
Cole Rivers H	CEDC Youngs Bay Net	Youngs River and Bay	0.520	0.098
Cole Rivers H	Klaskanine S FK Pond	S FK Klaskanine	0.054	0.000
Total			1.253	0.084
Unknown / other				
	Natural Production			
Bonneville Ladder	Tagging	Lower Columbia	0.025	1.000
Chinook R	Sea Resources	Chinook	0.015	1.000
Total			0.040	1.000
		2004 Total	60.535	0.123
Release Year 2005				
Snake Fall				
Lyons Ferry H	Lyons Ferry	Clearwater	0.200	1.000
Lyons Ferry H	Lyons Ferry	Snake General	0.406	1.000
Lyons Ferry H	Nez Perce	Clearwater	0.869	0.624
Snake R	Oxbow	Upper Snake	0.189	1.000
Snake R	Umatilla	Upper Snake	0.378	1.000
Snake R-Lower	Lyons Ferry	Grande Ronde	0.482	0.416
Snake R-Lower	Lyons Ferry	Lyons Ferry	0.200	1.000
Snake R-Lower	Lyons Ferry	Snake below Grande Ronde	0.234	0.000
Total			2.959	0.715
Upper Columbia Summer				
Methow & Okanogan	Methow	Methow	0.355	1.000
Wells H	Turtle Rock	Columbia general	0.776	0.524
Wells H	Wells	Columbia general	0.471	1.000
Wells H	Wells	Lake Chelan	0.018	0.000
Total			1.620	0.761
Upriver Bright Fall				
Bonneville Pool	Ringold Springs	Columbia general	2.800	0.079
Columbia Upriver	Bonneville	Tanner Creek	4.428	0.025
Columbia Upriver	Little White Salmon	Little White Salmon	1.460	0.995
Hanford Reach Stock	NA	Columbia at Hanford Reach	0.165	1.000
Priest Rapids	Klickitat (YKFP)	Klickitat	4.091	0.160
Priest Rapids	Priest Rapids	Columbia general	6.600	0.031
Umatilla R	Umatilla	Umatilla	0.603	0.998
Willard (L Wht Salm)	Willard	Little White Salmon	0.443	1.000
Total			20.590	0.187
Spring Creek Group Fall				
Big Creek H	Astoria HS	Youngs River and Bay	0.010	1.000
Big Creek H	Big Creek	Big Creek	6.090	0.037
Elochoman R	Elochoman	Elochoman	1.379	0.065
Spring Creek	Spring Creek	Spring Creek	14.533	1.000
Total			22.013	0.675

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2005 (continued)				
West Cascade Fall				
Cowlitz R	Cowlitz	Cowlitz	4.610	0.039
Cowlitz R	NA	Cowlitz	0.002	1.000
Kalama R	Fallert Creek	Kalama	2.626	0.034
Kalama R	Kalama Falls	Kalama	2.581	0.035
Lewis R	NA	Lewis	0.048	1.000
Lewis R NF	NA	Lewis	0.102	1.000
Toutle R	North Toutle	Cowlitz	2.086	0.044
Washougal R	Washougal	Washougal	4.249	0.024
Total			16.304	0.043
West Cascade Spring				
Cowlitz	Cowlitz	Upper Cowlitz	0.266	0.000
Total			0.266	0.000
Willamette Spring				
Clackamas Early	Clackamas	Clackamas	0.205	1.000
McKenzie	McKenzie	McKenzie	0.353	0.991
Santiam NF	Marion Forks	Santiam (Detroit Res)	0.152	0.990
Santiam SF	Klaskanine S FK Pond	Klaskanine	0.566	0.999
Santiam SF	South Santiam	Santiam	0.232	1.000
Santiam SF	Willamette	Santiam	0.100	1.000
Willamette MF	Willamette	MF Willamette	0.323	1.000
Total			1.932	0.997
Rogue River Fall				
Cole Rivers H	Big Creek	N FK Klaskanine	0.735	0.079
Cole Rivers H	CEDC Youngs Bay Net	Youngs River and Bay	0.161	0.311
Cole Rivers H	Klaskanine	N FK Klaskanine	0.202	1.000
Cole Rivers H	Klaskanine S FK Pond	S FK Klaskanine	0.045	0.661
Total			1.144	0.298
Unknown / other				
Bonneville Ladder	Natural Production Tagging	Lower Columbia	0.033	1.000
Total			0.033	1.000
		2005 Total	66.860	0.375
Release Year 2006				
Snake Fall				
Lyons Ferry H	Lyons Ferry	Snake General	0.507	0.395
Lyons Ferry H	Lyons Ferry	Upper Snake	0.397	0.471
Lyons Ferry H	Nez Perce	Clearwater	0.483	0.720
Snake R	Oxbow	Upper Snake	0.191	1.000
Snake R	Umatilla	Upper Snake	0.249	1.000
Snake R-Lower	Lyons Ferry	Grande Ronde	0.409	0.490
Snake R-Lower	Lyons Ferry	Snake below Grande Ronde	0.686	0.894
Total			2.922	0.681

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2006 (continued)				
Upper Columbia Summer				
Wells H	Grant County PUD	Columbia general	0.002	0.998
Wells H	NA	Columbia general	1.378	0.603
Wells H	Wells	Lake Chelan	0.018	0.000
Total			1.398	0.595
Upriver Bright Fall				
Bonneville Pool	Ringold Springs	Columbia general	0.070	0.065
Columbia Upriver	Bonneville	Tanner Creek	9.724	0.506
Columbia Upriver	Little White Salmon	Little White Salmon	1.802	1.000
Hanford Reach Stock	NA	Columbia at Hanford Reach	0.204	1.000
Priest Rapids	Klickitat (YKFP)	Klickitat	4.548	1.000
Priest Rapids	Priest Rapids	Columbia general	6.876	0.266
Washington Brights	Umatilla	Umatilla	0.610	0.997
Total			23.835	0.584
Spring Creek Group Fall				
Big Creek H	Astoria HS	Youngs River and Bay	0.001	0.000
Big Creek H	Big Creek	Big Creek	5.850	0.039
Elochoman R	Elochoman	Elochoman	1.785	0.977
Spring Creek	Spring Creek	Spring Creek	15.239	1.000
Total			22.876	0.752
West Cascade Fall				
Cowlitz R	Cowlitz	Cowlitz	0.205	1.000
Cowlitz R	NA	Cowlitz	0.019	1.000
Kalama R	Fallert Creek	Kalama	2.017	0.971
Kalama R	Kalama Falls	Kalama	2.012	0.993
Lewis R	NA	Lewis	0.052	1.000
Toutle R	NA	Cowlitz	0.090	1.000
Toutle R	North Toutle	Cowlitz	1.578	0.000
Washougal R	Washougal	Washougal	4.109	0.022
Total			10.081	0.438
West Cascade Spring				
Cowlitz	Cowlitz	Upper Cowlitz	0.293	0.000
Total			0.293	0.000
Willamette Spring				
Clackamas Early	Clackamas	Clackamas	0.322	0.989
McKenzie	McKenzie	McKenzie	0.356	0.995
Santiam NF	Marion Forks	Santiam (Detroit Res)	0.109	0.997
Santiam SF	South Santiam	Santiam	0.289	1.000
Willamette MF	Dexter Ponds	MFWillamette	0.324	1.000
Willamette MF	McKenzie	McKenzie	0.115	1.000
Total			1.514	0.996

Appendix Table G. Continued.

Stock group/stock	Hatchery	Release location	No. released (million)	Fraction marked
Release year 2006 (continued)				
Rogue River Fall				
Cole Rivers H	Big Creek	Youngs River and Bay	0.384	1.000
Cole Rivers H	CEDC Youngs Bay Net	Youngs River and Bay	0.093	0.237
Cole Rivers H	Klaskanine S FK Pond	S FK Klaskanine	0.629	0.080
Total			1.105	0.412
Unknown / other				
Bonneville Ladder	Natural Production Tagging	Lower Columbia	0.004	1.000
Total			0.004	1.000
		2006 Total	64.027	0.630
		2002–2006 Total	318.835	0.279