Juvenile Salmon Ecology in Tidal Freshwater Wetlands of the Lower Columbia River and Estuary: Synthesis of the Ecosystem Monitoring Program, 2005–2010

April 2013

Prepared by the Lower Columbia Estuary Partnership
For Bonneville Power Administration

Lower Columbia
Estuary
Partnership

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Suggested Citation:
Sagar, J.P., A. B. Borde, L.L. Johnson, C. A. Corbett, J. L. Morace, K. H. Macneale, W.B. Temple , J. Mason, R.M Kaufmann, V.I. Cullinan, S. A. Zimmerman, R. M. Thom, C.L. Wright, P.M. Chittaro, O. P. Olson, S. Y. Sol, D. J. Teel, G. M. Ylitalo, , N.D. Jahns, 2013. "Juvenile Salmon Ecology in Tidal Freshwater Wetlands of the Lower Columbia River and Estuary: Synthesis of the Ecosystem Monitoring Program, 2005–2010", Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration, April, 2013, Available from the Lower Columbia Estuary Partnership, Portland, OR.

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Acknowledgements

This study could not have been completed without the help of our partners. We are grateful to the Northwest Power Planning Council and the Bonneville Power Administration for funding through the Columbia Basin Fish and Wildlife Program. We also thank the land owners and managers who have allowed us to conduct research on their lands including: Alex Chmielewski (Ridgefield National Wildlife Refuge), Jim Clapp (Franz Lake National Wildlife Refuge), Mark Nebeker (Sauvie Island Wildlife Area). Keith Marcoe (Estuary Partnership) provided assistance with site selection and field support. Nathan Jahns, Taya MacLean, and Malini Roberts (SWCA) assisted with the multidiscipline analysis and technical edits to the draft. Matthew Schwartz (Estuary Partnership) provided peer review and assistance with numerous sections of the draft report. Finally, the Estuary Partnership's Science Work Group provided valuable input throughout the process and peer review on final drafts. The Science Work Group is composed of over 60 members, is integral in ensuring the Estuary Partnership represents the best available science. The authors wish to acknowledge the following Science Work Group members for having regularly provided technical review and input for the Ecosystem Monitoring Program in particular:

- Cindy Studebaker (US Army Corps of Engineers, Portland District)
- Russell Scranton (BPA)
- Gary Johnson, Heida Diefenderfer, Nikki Sather (PNNL)
- Bernadette Graham Hudson and Pat Frazier (Lower Columbia Fish Recovery Board)
- Tom Murtagh and Jim Brick (ODFW)
- Alex Uber, Steve Vigg, Donna Bighouse (WDFW)
- Regan McKnatt, Curtis Roegner (NMFS)
- Rudy Salakory (Cowlitz Indian Tribe)
- Kathy Roberts (USFWS)
- Si Simenstad (UW)
- Ian Waite and Tim Counihan (USGS)
- Nancy Leonard, Patty O'Toole (NPCC)
- April Silva, April Cameron (CREST)

Executive Summary

Introduction

The Lower Columbia Estuary Partnership's (Estuary Partnership's) Ecosystem Monitoring Program (EMP) is an integrated status and trends program for the Lower Columbia River and Estuary (LCRE). The study area extends from the mouth of the estuary to the Bonneville Dam. The program is designed to track trends in the overall condition of the LCRE to reduce uncertainties, provide a suite of reference sites for use as end points in the region's habitat restoration actions, and place findings from the program into context with the larger ecosystem.

As part of the National Estuary Program, the Estuary Partnership works with its regional partners to develop and implement a Comprehensive Conservation and Management Plan (CCMP). Ecosystem monitoring is a key element of the Estuary Partnership's CCMP. The CCMP specifically calls for sustained long-term monitoring to understand conditions in the river and to evaluate the trends and impacts of management actions over time. The EMP is funded by the Northwest Power and Conservation Council/Bonneville Power Administration (NPCC/BPA). A primary goal of this program is to collect key information on ecological conditions for a range of habitats in the lower river characteristic of those used by out-migrating juvenile salmon and provide information toward the recovery of threatened and endangered salmonids.

When the EMP was created in 2004, most previous research in the LCRE had occurred in the lower estuary, closest to the river mouth in Reaches A and B. There was a considerable lack of research and monitoring within the tidal freshwater section of the LCRE, resulting in little basic understanding of habitats, fish use and food web dynamics in this region. The EMP and partners developed a list of questions, and a subsequent monitoring design, for which there was little current information and which were fundamental to understanding how estuarine resources occur and interact in the LCRE. Specific questions are defined in each section of this report. Based on the knowledge gaps identified in the LCRE and the Estuary Partnership's and the regional partner's goals, the EMP goals for the 2005–2010 monitoring design were to:

> Track the status and trends of ecosystem conditions to inform decisions for the purpose of conserving and restoring the LCRE through:
>
> 1. A comprehensive assessment of status (spatial variation) and trends (temporal variation) of habitat, fish, food web, and abiotic conditions in the lower river, focusing on relatively undisturbed shallow-water and vegetated habitats used extensively by juvenile salmonids for rearing and refugia;
> 2. A coordinated effort to gather baseline data about estuarine resources (Johnson et al. 2004);
> 3. A determination of the variety of salmon life histories currently expressed in the estuary and habitats that support them (from Bottomet al. 2005); and
> 4. A better understanding of salmon habitat associations to improve predictions of habitat opportunity in order to improve restoration strategies (from Bottom et al. 2005).

Methods

The EMP is a collaborative effort between the Estuary Partnership, Pacific Northwest National Laboratories (PNNL), National Oceanic and Atmospheric Administration, National Marine Fisheries Service (NOAA NMFS) and U.S. Geological Survey (USGS). This report is a synthesis of the data collected through the EMP from 2005 to 2010. To address the EMP goals, the EMP partnership has collected data on habitat structure (vegetation community, water surface elevation, channel morphology, sediment grain size and total organic content [TOC], sediment accretion, and site profiles; by PNNL), fish use (fish community, salmon metrics and diet), macroinvertebrate prey availability, and water temperature at the time of fish sampling (by NOAA NMFS), and abiotic site conditions (water temperature, specific conductance, pH, dissolved oxygen, and depth; by USGS).

Sample sites were minimally-disturbed, tidally influenced freshwater emergent wetlands, with backwater sloughs, representative of the eight hydrogeomorphic reaches across the study area. The Estuary Partnership and its monitoring partners have focused on providing an inventory of salmon habitats (or “status”) across the lower river stratifying by hydrogeomorphic reach (A–H) and including a growing number of fixed sites for interannual variability (or “trends”). The focus of the EMP has been on minimally disturbed tidally influenced emergent wetland sites. Each year, three to four “status” sites, in a previously unsampled reach, were selected along with the continued sampling of a growing number of “trend sites.” Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2010), Whites Island (2009, 2010), and Franz Lake (2008, 2009) are the three trend sites. Data collected for the EMP after 2010 are not included since they were not available at the time data compilation for this synthesis was begun. Sampling in Reaches A and B began in 2011 and therefore only Reaches C through H are discussed in this synthesis. Habitat structure and hydrology data began to be collected in 2005, fish data collection began in 2007, fish prey data collection began in 2008, and water quality data collection has been conducted at a small number of sites, dependent on funding levels.

Habitat Structure and Hydrology

To compare freshwater emergent habitats throughout the LCRE, the habitat structure/hydrology analysis combined data sets from multiple programs (total of 39 sites). For this synthesis, habitat data between 2005 and 2010 were analyzed from relatively undisturbed emergent marshes between Reaches A and H throughout the LCRE as part of the EMP program, the Reference Site Study (Borde et al. 2011), the Cumulative Ecosystem Response to Restoration Study (Johnson et al. 2011) and the Tidal Freshwater Monitoring program (Johnson et al. 2011).

Fish Use

To assess spatial patterns in fish habitat occurrence in various reaches of the LCRE, a total of 12 sites were sampled by beach seine between 2007 and 2010, co-located with the habitat sampling sites—six in Reach C, one in Reach E, one in Reach F, and four in Reach H. Macroinvertebrate data collection was concurrent with fish sampling beginning in 2008 and totaled 10 sites between reaches C and H.

Abiotic Conditions

For three years, the USGS deployed a continuous water quality monitor at Campbell Slough in the Ridgefield National Wildlife Refuge (2008–2010). This site in Reach F has been sampled for habitat structure/hydrology since 2005 and for fish since 2007 and offers preliminary information on temporal variability in abiotic site conditions.

Interdisciplinary Analysis

The multidiscipline analysis evaluates fish diversity, Chinook salmon abundance, and Chinook salmon health (using lipid levels) in relation to other environmental and biological variables. A total of 14 sites (including multiple years of the same site) between 2008 and 2010 had overlapping data sets and were used in the multidiscipline analysis. Trend and lipid analysis used a subset of the 14 sites. To explore relationships between fish and other variables within this study, individual data sets (vegetation, hydrology, fish, macroinvertebrate, and water quality) were examined together using correlation coefficients and subsequent multiple regression analyses.

Summary of Findings

Habitat Structure and Hydrology

Temporal and spatial variability in the hydrology and habitat structure of emergent wetlands affects their capacity for storing carbon, providing habitat for salmon, and contributing to the food web of the greater LCRE. As such, quantifying the expected ranges and variability in these systems can start to reduce uncertainties and inform research areas targets to improve the capacity of the LCRE and provide for these important functions. These findings address goals 1, 2, and 4 listed above.

Wetland structure and process at minimally-disturbed tidally influenced wetland sites-- specifically sediment, hydrology, elevation, and vegetation—fell into distinct spatial patterns across the LCRE. TOC at study sites did not vary greatly across the LCRE and was comparatively low for tidal wetlands, potentially due to a combination of vegetation type, landscape position, and marsh age. The sediment grain size was extremely consistent across sites and was predominantly silt, very fine, and fine sand. Marsh sediment accretion rates fell within a narrow range in our study area, but were variable in time and space throughout the estuary. The most notable spatial patterns in the hydrology data lend themselves to distinct hydrologic zones throughout the LCRE (described in more detail below). Seven taxa dominated plant cover across the LCRE (68% of the cumulative vegetative cover; reed canarygrass, *Phalaris arundinacea*; common spikerush, *Eleocharis palustris*; Wapato, *Sagittaria latifolia*; Lyngby sedge, *Carex lyngbyei*; Canada waterweed, *Elodea Canadensis*; false loosestrife, *Ludwigia palustris*; and slough sedge, *Carex obnupta*). Along the estuarine gradient from the mouth to the dam, the number of vegetation species present and highest percent cover were generally greatest in the lower-middle portion of the LCRE (River Kilometer, rkm, 53–89), although percent cover was high up to rkm 154.These spatial patterns help establish a set of baseline conditions that can be used for regional trend analysis and improving restoration practices in similar habitats.

This study establishes four distinct hydrologic zones in the LCRE that are demarcated by the dramatic variation in hydrology and resulting inundation patterns along the estuarine gradient. The zones indicate high inundation and seasonal variability in the fluvial-dominated upper estuary and lower inundation and daily variability in the tidal-dominated lower estuary. In the mesohaline zone (5 to 18 parts per thousand

[ppt]; ~0 to 15 rkm) near the mouth of the estuary the vegetation cover is high; however, the number of species is limited by salinity. Few non-native species are found in this zone. In the oligohaline zone (0.5 to 5 ppt; ~16 to 40 rkm), species diversity starts to increase as there emerges an overlap in the number of species that can tolerate brackish and freshwater conditions. The highest species diversity occurs in the portion of the river that is tidal freshwater, but not affected by the high seasonal inundation associated with the spring freshet (~41 to 135 rkm). In the fluvial-dominated tidal freshwater zone (above 135 rkm) vegetation cover and species diversity appear to be variable depending on the timing and magnitude of the spring freshet.

The elevations of wetlands in this study exist within a narrow range (90% of the quadrats sampled were between 0.8 meter [m] and 2.6 m, relative to the Columbia River Datum [CRD]). The highest species diversity occurred between the elevations of approximately 1.5 m and 2.5 m CRD, consistent with other studies that have shown increased species diversity in high versus low marshes (Elliot 2004; Leck et al. 2009). This narrow elevation range indicates that these habitats, important to juvenile salmon for feeding, rearing and migration, are vulnerable to potential hydrologic changes such as hydropower alterations in flow and climate change.

The ubiquitous non-native invasive species, *Phalaris arundinacea* (reed canarygrass) decreases plant species diversity and has the potential to affect food web dynamics in the LCRE. Reed canarygrass occurred in 52% of the quadrats and accounted for 28% of the cover at all emergent marsh sites. In the mesohaline zone of the study area salinity precludes the presence of reed canarygrass except for infrequent occurrences at the highest marsh elevations. Beyond this zone, the lower elevation range in this study, where other species may be able to out-compete reed canarygrass, was from approximately 1.2 m to 1.6 m CRD. This range increases to approximately 1.4 to 1.8 m CRD in the fluvial-dominated portion of the estuary as seasonal inundation increases and likely limits the lower elevation range. Another trend observed in this analysis was the interannual variability of reed canarygrass cover due to varying water levels; however, reductions were not persistent between years. Our understanding of the spatial extent, resiliency and prevalence of this species provides information that can be used to improve management of the species.

Vegetation cover and composition is directly related to patterns in inundation, with cover decreasing when inundation increases. This interannual variability associated with varying water levels was documented in our trends analysis at the three upriver trend sites (Cunningham Lake, Campbell Slough and Franz Lake located at 145, 149, and 221 rkm respectively); however, the same patterns were not as discernible at the trend site located at 72 rkm (Whites Island). The boundaries between the major species at the trend sites were generally stable over time even with varying water levels. In the highest water year we did observe an increase in the lower elevation of all species at the lowest elevation site, indicating the potential for an effect on the elevation ranges from this level of hydrological variability. The implications of this type of change include a potential loss of wetland area and a reduction in biomass production.

The potential for fish access at tidally influenced freshwater wetlands in the LCRE is important for understanding the contribution of these habitats for refuge, feeding, and cover. Salmon access to the channel mouth and bank varied with the site's location in the river and time of year. In general, salmon access to the channel mouth (with 50 cm of water) was between 50% and 80% of the time in most areas of the LCRE. In the lower, tidally dominated part of the estuary, this frequency is consistent during the peak salmon migration period and the total year. In contrast, in the upper estuary the amount of time the channel is accessible decreases during the total year due to the low water period that occurs in the fall. In the lower estuary (below rkm 60), the channel bank (with 10 cm of water) was accessible 20% to 30% of the time during the peak salmon migration and total year periods. Above rkm 60 the frequency increased to about 30% to 60% of the time. The elevations of these channel characteristics can be useful for informing restoration projects to ensure that salmon access is maximized at the site.

Fish Use

Overall, findings suggest that tidal freshwater habitats in the LCRE are used for migration, feeding, and rearing by several species of salmonids, and by Chinook salmon stocks from both the lower Columbia River and the interior Columbia Basin. The various reaches show distinctive patterns in terms of fish communities, prey assemblages, and salmonid stocks. Although these sites were relatively undisturbed, there was still evidence of disturbance, including high summer water temperatures, chemical contaminants, and non-native fish species. At the trend sites, the year-to-year consistency in fish communities, patterns of salmon occurrence, and indicators of salmon fitness suggests low interannual variability in these characteristics. The level of disturbance was generally the highest at the Reach E and Reach F sites, which were closer to and downstream of major urban centers. These findings address all four goals listed above.

Fish community composition showed distinctive patterns that varied by reach. Species richness and diversity tended to be lower at the Reach C sites, but these sites also had a higher proportion of native fish species and fewer salmonid predators than sampling sites in the other reaches. Species richness, diversity, and proportion of non-native species in the catch were highest at Campbell Slough, and tended to decline somewhat with increasing rkm.

A variety of salmon species are present at the EMP tidal freshwater emergent marsh sampling sites. While Chinook were most abundant at these sites, coho, chum, and steelhead trout were also observed, suggesting these areas are important habitat for multiple juvenile salmon species. Chinook salmon were the dominant salmonid species in Reaches C through F.

Sites with especially high densities of salmon included Bradwood Slough in Reach C and Pierce Island in Reach H. Relatively high densities of unmarked Chinook salmon were also found at several other Reach C sites. The term “unmarked” is used rather than of “natural origin or wild” since some hatchery fish are not fin clipped. These productive sites tended to have relatively cool water temperatures in summer, and low proportions of non-native fish species and piscivorous predators.

Chemical contaminants were present at concentrations above toxic injury thresholds in a significant proportion of juvenile Chinook salmon from the sampling sites. Contaminants were found in the salmon and/or salmon diets, but they were not measured in the water at the site Exposure to polycyclic aromatic hydrocarbons (PAHs) was a potential concern throughout the study area. DDTs were also widespread but typically at concentrations below estimated levels that would result in deleterious health effects. Exposure to polychlorinated biphenyls (PCBs) occurred in juvenile Chinook salmon at Campbell Slough and Sandy Island; and exposure to polybrominated diphenyl ethers (PBDEs) occurred in juvenile Chinook salmon at Sandy Island and the Reach C sites. Concentrations of PCBs and DDTs tended to decline in juvenile Chinook salmon from Campbell Slough between 2007 and 2010.

Water temperatures were above the range preferred by juvenile salmon during the summer months at many sites, with especially high temperatures in 2009. High summer water temperatures and low salmon densities during these months suggest that water temperature may be a limiting factor in full utilization of some of these habitats.

For marked Chinook and coho salmon, fish size ranges were relatively narrow and consistent among sites, but unmarked fish showed considerable variation in size class. Among coho, both larger, smolt-size fish and smaller, possibly subyearling fish were observed. Chinook salmon were present in size classes ranging from fry at 40 to 60 millimeter (mm) to yearlings of >130 mm. At all sites where they were captured, unmarked coho and Chinook salmon showed increases in length and weight over the sampling season. This pattern, indicating that salmon caught later in the season were larger than those caught earlier in the season, was much less evident for marked fish.

Mean growth rates from otoliths of Chinook salmon ranged from 0.46 to 0.61 mm/day, which is within the typical range reported for Columbia River subyearling Chinook. However, there were distinct patterns in growth rates among sites, stocks, and between marked and unmarked fish. Growth rates were significantly lower in unmarked fish than in marked fish, in fish from Reach C than in fish from other reaches, and in West Cascades fall Chinook than in Chinook from certain other stocks, including Spring Creek Group fall Chinook and Upper Willamette spring Chinook.

Mean lipid content of marked and unmarked juvenile Chinook salmon was similar (1.2–1.4%). However, in marked Chinook, lipid content tended to decline as rkm decreased, so the highest values were in Reach H Chinook and lowest values were in Reach C Chinook, whereas no such relationship was found for unmarked fish.

Fish Prey

A diverse range of prey was available at the sites, and the assemblages showed some differences among reaches. The highest densities of prey were found in emergent vegetation tows, highlighting the importance of vegetated nearshore habitat in providing food for juvenile salmonids.

Regardless of the prey composition in the aquatic environment, juvenile Chinook salmon showed a strong preference for Dipterans in the diet. This was consistent across sites, reaches, and sampling times. Where available, amphipods were also a preferred prey item.

Abiotic Conditions

Although water quality data were limited to two years at one site, some seasonal water quality patterns were apparent. Data collected from multiple sites since 2010 will help us to better assess spatial and temporal trends in abiotic conditions for future analyses.

In 2009 and 2010, water quality conditions in Campbell Slough failed to meet Washington water quality standards set to protect salmonid spawning, rearing, and migration, during portions of the monitoring period (March-August). The weekly maximum water temperature standard was exceeded temporarily in May and from June through the end of the monitoring period during both years. The daily maximum standard for pH was exceeded in late June through August in 2009, and in April through May in 2010. The daily minimum standard for pH, however, was met throughout the monitoring period. The daily minimum dissolved oxygen standard was not met briefly in May (2009) and during much of June through the end of the monitoring period in both years. High water temperatures, low dissolved oxygen concentrations, and non-neutral pH are a potential source of physiological stress to cold-water fish, including salmonids at this site, highlighting the potential for similar conditions at other sites.

Interdisciplinary Analysis

It is important emphasize that regression modeling results should be treated with caution due to small sample sizes and a limited set of environmental and biological variables in the analysis. Emphasis in this synthesis report is therefore not placed on determining statistically significant relationships between variables but rather on exploring potential relationships.

Overall, fish diversity at freshwater tidal wetlands was positively related to prey diversity and coverage of common spikerush (*E. palustris*). Fish diversity had a weaker, negative, correlation with the richness and diversity of vegetation species. The correlation between vegetation variables and fish diversity, however, may be due to the site's location in the river (rkm) rather than site-specific variables. Reaches E, F, and H had higher fish diversity and these reaches, in general, had lower plant species diversity and higher cover of the three dominant species, including common spikerush. Higher fish diversity may be more a function of landscape level influences such as proximity to disturbed sites and higher boat traffic (i.e. due to ballast water and fishing introductions) and reservoirs that promote introduction of non-native species and warmer summer temperatures supporting warm-water acclimated species. Future analysis would benefit from the inclusion of other landscape level variables.

Juvenile Chinook abundance was positively related to species richness of all vegetation species, and negatively related to invertebrate prey species diversity in emergent vegetation and the distance of the sample location from the mouth of the Columbia River. Decreased Chinook abundance with increasing prey species diversity could indicate that where prey species diversity is high, preferred prey, such as Diptera, may be lower. Given the small sample sizes for these analyses, interpretation of results should be treated with caution. Although salmon abundance is potentially related to characteristics of the tidal freshwater wetlands sampled -- deeper tidal channels, higher plant diversity, and prey diversity –the relationship may be more a function of location in the river, rather than site-specific variables. Over 30 plant species were found in Reach C sites, where salmon abundances were greatest, and less than 20 species were found in upriver reaches where salmon abundances were lower. Low numbers of piscivorous predators and lower summer water temperatures in reaches with high juvenile Chinook abundance are also possible explanations for higher abundance and should be included in future analyses. A decrease in juvenile Chinook abundance with distance from the mouth of the Columbia River is likely a proxy for other factors that affect salmonid occurrence and would require additional research.

At the trend sites, Chinook salmon abundance was related positively to the density of Diptera species in May sampling in emergent vegetation, the diversity of non-native vegetation, and greater cover of reed canarygrass. The highest densities of prey were found in emergent vegetation tows, potentially highlighting the link between vegetated nearshore habitat and juvenile salmonids. The positive correlation between reed canarygrass and Chinook salmon abundance is likely driven by patterns in hydrology and inundation, in which higher water results in lower coverage of reed canarygrass and lower abundances of juvenile Chinook salmon (potentially due to the difficulty in sampling for fish by beach seine).

Variables used in this multidiscipline analysis vary not only with factors like river kilometer, distance from the main channel, elevation, etc. but also seasonally and not in a linear fashion. This variability, in addition to high variability in fish sampling, may make it especially difficult to detect clear relationships with correlation analyses. Future analyses would benefit from the inclusion of other landscape level variables, important abiotic site characteristics such as water temperature, water level, and chemistry coincident with fish sampling and the exploration of relationships between other variables, such as vegetation and fish prey.

Conclusions

This report represents the Estuary Partnership's first synthesis of data collected under the EMP to assess juvenile salmon ecology in minimally disturbed tidal freshwater wetlands of the LCRE. Habitat structure and hydrology, fish, macroinvertebrates, and abiotic conditions data analyzed in this study indicate that undisturbed emergent wetlands in the LCRE system are important to fish in general and specifically to multiple salmon species and stocks. The relationships between fish species and physical, biological, and environmental characteristics, coupled with the findings that lower Columbia wetlands exist within a narrow elevation range and are vulnerable to hydrologic changes, suggest a complex system in which disruption could have significant ecosystem-wide impacts. Maintaining or improving the quality of these habitats by reducing this disruption can occur by taking steps to 1) preserve/restore nearshore emergent vegetation and hydrograph supporting the vegetation community, 2) establish habitat structure to support preferred salmon prey, 2) moderate summer temperatures, 3) reduce the spread of non-native species, and 4) reduce chemical contamination.

In addition to addressing the current goals and objectives of the EMP between 2005 and 2010, this synthesis informs current and future program design. Ultimately, one of the goals of this program is to assess ecosystem condition by creating a suite of indicators that can be used to track changes in the LCRE. Indicators will need to distinguish between variability associated with natural conditions, and variability that may result from human impacts. Results from this report begin to document the range in variability for each sampled metric, critical for designing a more statistically rigorous program. For example, habitat analysis revealed distinct hydrologic zones for vegetation that establishes a pattern in spatial variability throughout the LCRE. At trend sites, vegetation elevation ranges remained fairly consistent between years indicating that drastic changes to these baseline conditions could signify a decline in condition in these wetlands. Although fish communities are highly variable throughout the LCRE, trends analysis in this study revealed a surprising site consistency that may aid in detecting changes over time. Additionally, the synthesis of EMP data allows us to evaluate each metric to streamline data collection (see recommendations section) and select the most appropriate forward-looking indicators. Beginning in 2011, current monitoring design includes a suite of primary and secondary production metrics, more intensive abiotic conditions monitoring, and establishes sites in Reaches A and B that will begin to integrate long-term trends in salmon food web dynamics.

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1.0 Introduction and Background

1.1 Summary

The Lower Columbia River Ecosystem Monitoring Program (EMP) encompasses the study area of the Lower Columbia Estuary Partnership (Estuary Partnership) and includes all tidally influenced areas of the mainstem and tributaries from Bonneville Dam to the plume. Tidal influence is defined as historical tidal influence, relative to post-dam construction in the 1930s. The Columbia River historically supported diverse and abundant populations of fish and wildlife and is thought to have been one of the largest historical producers of Pacific salmonids in the world (Netboy 1980). Anthropogenic changes since the 1860s and the construction of the hydropower system have significantly reduced the quantity and quality of habitat available to fish and wildlife species. Contributing factors include altered timing, magnitude, duration, frequency, and rate of change in river flows; degraded water quality and increased toxic, chemical contaminants; introduction of invasive exotic species; and altered food web dynamics. Ecosystem-based monitoring of the Lower Columbia River and Estuary (LCRE) has become a regional priority to aid in the recovery of the historical productivity and diversity of fish and wildlife.

The lower Columbia River is designated an “estuary of national significance” by the U.S. Environmental Protection Agency (USEPA), making it one of 28 National Estuary Programs (NEPs) under Section 320 of the Clean Water Act. The Estuary Partnership was created in 1995 by the governors of Washington and Oregon and the USEPA to coordinate regional partners in protecting and restoring the LCRE ecosystem. Each NEP works with regional stakeholders, such as local, state, tribal and federal governments, industry, citizens, non-profit organizations and academia, to 1) identify issues facing the ecosystem of that estuary, 2) to determine goals and quantifiable objectives to address the issue, 3) create a Comprehensive Conservation and Management Plan (CCMP) outlining these as well as specific steps to reach the goals and objectives, 4) develop a long-term monitoring plan to track progress and ecosystem condition and 5) maintain a “management conference” of stakeholders. Each NEP relies heavily on its partners to implement the actions and meet the goals within the shared CCMP. The major roles of the NEP staff are to ensure coordination and use of best available science, provide a central clearinghouse of information on the estuary, and identify gaps in implementation and work to find ways to fill them.

Effective ecosystem management requires knowledge of changes (particularly detrimental changes) that occur in the ecosystem, and of the factors that lead to those changes. The ultimate goal of status and trends monitoring is to track the status of a resource (e.g., river stage at a given point, salmon escapement in a specific tributary, plankton composition and biomass in a lake or slough) over time but also to allow researchers and managers the ability to distinguish between variability associated with natural conditions from any changes or variability that may result from human intervention. The creation and maintenance of long-term data sets have irreplaceable value for documenting the history of change (long-term trends) within important resource populations, for evaluating the potential significance of human activities on natural resources, and for visualizing and formulating testable hypotheses about the interactions among species, between species and their environment, and the mechanisms for these interactions.

The EMP is an integrated status and trends program for the LCRE. The overall objectives of this program are to track trends in the overall condition of the lower river, provide a suite of reference sites for use as end points in our restoration actions, and place results of findings into the context of the larger ecosystem.

The EMP program is funded by the Northwest Power and Conservation Council/Bonneville Power Administration (NPCC/BPA) under the Fish and Wildlife Program. A primary goal for their funding is to collect key information on ecological conditions for a range of habitats in the lower river characteristic of those used by outmigrating juvenile salmon and provide information toward implementation of the 2008 Federal Columbia River Power System (FCRPS) Biological Opinion (BiOp; NMFS 2008).

This report is a synthesis of the data collected through the EMP from 2005 to 2010. Information collected describes synoptic conditions and changes over time in vegetated floodplain habitats and the opportunity, capacity, and realized function (Simenstad and Cordell 2000) for juvenile salmonids. These habitats are the targets of regional restoration efforts, and this program provides integral information for understanding the success of the regional habitat restoration program. The results of this program provide information on ambient environmental conditions and insight into the cumulative effects of existing and new management actions and anthropogenic impacts as they occur.

1.2 Description of Program Area

1.2.1 Historical Changes to the Lower Columbia River

Since the 1880s anthropogenic impacts to the lower river include diking and conversion of habitat for agriculture, industry, and urban development. Several studies (Thomas 1983; Allen 1999; Garano 2003; Estuary Partnership 2012) noted losses of approximately 70% of vegetated tidal wetlands and 55% of forested uplands for the project area since this era. Other important anthropogenic impacts to the Columbia Basin include the construction of >30 dams and dozens of smaller flow control structures on the mainstem and tributaries for hydropower, flood control, irrigation, and transportation. Fresh water is also diverted to irrigate arid lands in eastern Washington and Oregon for large-scale agricultural production. Water management through dams and maintenance of the navigation channel through dredging and pile dike construction allow deep-water ports to exist as far inland as Lewiston, Idaho.

River flow, a primary factor affecting habitat patterns in the estuary and plume, has been significantly modified by operations of this hydropower system. Changes include a reduction in the mean annual flow, reduced magnitude of spring freshets, an almost complete elimination of overbank flows, and altered timing of ecologically important flow events as well as habitat forming processes (Bottomet al. 2005; Fresh et al. 2005). These hydrological changes, along with floodplain diking, conversion of habitats, and navigation channel maintenance, represent a fundamental shift in the physical state of the lower Columbia River ecosystem, and have resulted not only in a loss of vegetated and shallow-water habitats but also a change in the size, seasonality, and behavior of the river plume (Bottom et al. 2005; Fresh et al. 2005). Kukulka and Jay (2003) suggested that the annual Columbia River flow cycle has been dampened and spring freshet flows has been reduced by >40% due to flow regulation by the hydropower system, water withdrawal for agriculture and climate change, and that during the spring freshet, floodplain diking and flow alteration together reduced average shallow water habitat within their study area (rkm-50 to rkm-90) by 62%. They hypothesized that taken individually, floodplain diking has reduced average shallow water habitat coverage during the spring freshet by 52% and flow alterations by 29% (Kukulka and Jay 2003).

The historic spring freshet aided in juvenile salmon migrations and transported large quantities of sediments, nutrients, cold water and associated organic matter downstream (Naiman et al. 2012). Large-scale floodplain diking has severed the historical connection of habitat with the river, eliminating any direct use ("habitat opportunity") and reducing indirect (e.g., export of organic matter for food webs) benefits to aquatic species, which over time acclimated to the historical conditions (Fresh et al. 2005). Low velocity, peripheral bay habitats, and the mid-estuary estuarine turbidity maximum are locations in the lower river where organic matter is concentrated and where invertebrate prey production and fish and macroinvertebrate feeding are higher than in many other locations (Bottom and Jones 1990; Jones et al. 1990; Simenstad et al. 1990). Researchers hypothesize that the loss of these historical wetlands and macro-algal habitats (e.g., mud and sand flats) through diking and conversions may have shifted estuarine food chains from macrodetrital to microdetrital sources (Sherwood et al. 1990). Such a shift would likely benefit food chains supporting pelagic-feeding fishes such as American shad (*Alosa sapidissima*) with corresponding loss of food webs supporting epibenthic-feeding fishes such as juvenile salmon (Bottom et al. 2005).

Introduction and wide-spread expansion of non-native, invasive species such as the noxious reed canary grass (*Phalaris arundinacea*) or American shad results in altered food webs and increased competition for limited resources by native plant, fish and wildlife species. Similar problems result from wide-spread Columbia Basin hatchery releases. Approximately 130-150 million hatchery salmonids are added to the river annually, significantly impacting the capacity of the lower river to sustain both these artificially produced as well as native fishes over time (Naiman et al. 2012).

Additionally, toxic contaminants from industry, agriculture, and urban development have been introduced throughout the Columbia Basin, and these contaminants have been well documented to pose a threat to fish and wildlife species. Exposure to waterborne and sediment-associated chemical contaminants has the potential to affect survival and productivity of all anadromous fish species as well as predator species that prey upon them in the lower river (Fresh et al. 2005; Johnson et al. 2007; Estuary Partnership 2007). Despite improvements since the 1990s, U.S. Fish and Wildlife Service (USFWS) researchers are still finding lower nesting success in bald eagles in the lower Columbia River than elsewhere in Washington and Oregon, a result of dichlorodiphenyltrichloroethane/dichlorodiphenyldichloroethylene (DDT/DDE), polychlorinated biphenyls (PCBs), and dioxins in this region (cited in NPCC 2004). Additionally, the U.S. Geological Survey (USGS) has consistently found DDE and other chemical contaminants (cited in NPCC 2004) in osprey and their food web along the mainstem lower Columbia River. The type and extent of exposure may vary with timing and length of use. For those organisms that move through the estuary quickly, short-term exposure to waterborne contaminants such as current use pesticides and dissolved metals may be the greatest threat, as these chemicals can disrupt olfactory function and interfere with behavior such as capturing prey, avoiding predators, imprinting, and homing (for stream-type evolutionary significant units [ESUs]) (Fresh et al. 2005). Organisms that use the estuary more extensively (e.g., ocean-type salmonids) may be exposed to these types of contaminants as well as persistent, bioaccumulative toxicants such as PCBs, polybrominated diphenyl ethers (PBDEs), and DDTs that they may absorb through feeding and rearing (in the case of ocean-type salmonids) in the estuary (Estuary Partnership 2007; Fresh et al. 2005; Johnson et al. 2007; Sloan et al. 2010). Chronic exposure to and accumulation of these chemicals in tissues can lead to effects such as reduced growth, immune dysfunction, and metabolic disorders that may lessen their chance of survival (for salmonids, see Arkoosh et al. 2001; Arkoosh et al. 2010; Arkoosh and Collier 2002; and Meador et al. 2002).

Chemical contaminants in salmon habitats can have a significant impact on survival and recovery of endangered salmon stocks and can contribute to salmon mortality, prey base reduction and sublethal health effects. Multiple studies suggest salmon are exposed to toxic contaminants in the lower Columbia River and are experiencing sublethal health effects from this exposure (e.g., Estuary Partnership 2007, Johnson et al. 2005; Johnson et al. 2007). Substantial proportions of specimens caught in the lower river have exposure levels to one or more contaminants, such as organic contaminants, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethanes (DDTs) and polybrominated diphenyl ethers (PBDEs), exceeding values thought to cause health risks (Estuary Partnership 2007; Johnson et al. 2007, 2012; Sloan et al. 2010; Yanagida et al. 2012). Concentrations of PAH metabolites were above estimated effect thresholds (Meador et al. 2008) in over 40% of juvenile Chinook salmon (*Oncorhynchus tshawytscha)* bile samples from the lower Columbia River (Yanagida et al. 2012). Moreover, ~50% of subyearling fall Chinook samples from tidal freshwater sites (Johnson et al. 2012) and ~66% of Chinook smolts from the lower estuary (Johnson et al. 2007) had PCB concentrations exceeding the 2400 ng/g lipid threshold estimated by Meador et al. (2002). Maximum concentrations of PCBs, DDTs, and PBDEs in juvenile salmon from the lower Columbia were all within the upper range of juvenile salmon sampled in the Pacific Northwest, and the condition and lipid content of a number of these fish, especially smolts, was also reduced. Body lipid content can influence an organisms' tolerance of bioaccumulative contaminants, with individuals with lower lipid content typically showing a greater toxic response to comparable exposure (Lassiter and Hallam 1990). Consequently, Johnson et al. (2007, 2012) and Arkoosh et al. (2010) suspect the decline in lipid content described above could increase the sensitivity of fish to the effects of bioaccumulative contaminants, such as PCBs, DDTs and PBDEs. The health of juvenile salmon may also be affected by exposure to other classes of contaminants present in the lower Columbia River, including pharmaceuticals and personal care products in wastewater (Estuary Partnership 2007; Morace et al. 2012); current use pesticides (NMFS 2008) and toxic metals such as copper (Hecht et al. 2007).

Finally, warming water temperatures and changing precipitation patterns resulting from climate change are expected to have a deleterious impact on Pacific salmonid populations throughout the Columbia Basin, including approximately 40% salmon habitat loss in Oregon and Idaho and 22% loss in Washington by 2090 (ISAB 2007). Several studies predict sea level rise within the lowest downstream areas of the river, inundating present floodplain wetland habitats and causing coastal migration inland (NWF 2007; Ducks Unlimited *In Review*). There are additional concerns regarding climate changes specific to the estuary and plume regions. Low oxygen conditions that occur deep in the continental shelves of Oregon and Washington during sustained periods of coastal upwelling are increasing. When combined with low river discharges, those conditions may also lead to oxygen depletion in the Columbia River estuary (Roegner et al. 2011). Upwelled waters have a direct impact on the Columbia River estuary through tidal exchange and entrainment by estuarine circulation. Implications of these low oxygen conditions are significant as these hypoxic episodes may lead to displacement or death by suffocation of marine organisms. Recent research has also documented increased acidification in upwelling waters along the coast of the Pacific Northwest (Feely et al. 2008) and decreasing pH levels in the Columbia River estuary at Beaver Army Terminal (J. Morace, pers. comm. USGS). The low dissolved oxygen levels, increasing acidification and increasing water temperatures that accompany climate change have the potential to alter fish behavior and survival (Roegner at al. 2011) and have significant deleterious impacts on the estuarine food web (Feely et al. 2008).

1.2.2 Importance of the LCRE to Salmonids

The Columbia River Basin is thought to have been the largest historical producer of Chinook salmon and steelhead (*O. mykiss*) in the world (Netboy 1980). Researchers estimate that 8 to 16 million wild Pacific salmon migrated up the Columbia River system each year to spawn in the mid-1870s (Cone 1995; Netboy 1980). In comparison, total current returns of wild fish number less than 1 million annually. All anadromous salmon and steelhead populations within the Columbia River Basin use the estuary as a critical migration corridor. The estuary is thought to offer three advantages to juvenile salmon in their transition from freshwater to saltwater environments: 1) a productive feeding area capable of sustaining increased growth rates; 2) a temporary refuge from marine predators; and 3) a physiological transition zone where fish can gradually acclimate to salt water (Simenstad et al. 1982; Thorpe 1994).

Recent research has documented that Chinook salmon, especially subyearlings, and other salmon such as chum (*O. keta*) and lower Columbia coho (*O. kisutch*), to a lesser degree, can rear extensively in shallow water and vegetated habitats within the estuary, including tidal channels, tributary confluences, and nearshore areas (e.g., Bottom et al. 2005; Casillas 2009; Fresh et al. 2005; Fresh et al. 2006; Good et al. 2005; Roegner et al. 2008). Subyearling migrants that enter the estuary as fry or fingerlings, or "ocean-type" salmon, exhibit a wide range of residence periods depending on the species, from days to weeks (chum) to several months (Chinook) (Thorpe 1994). Juvenile salmon may occur in the estuary all year, as different species, size classes, and life history types continually move downstream and enter tidal waters from multiple upstream sources (Bottom et al. 2005). Peak estuarine migration periods vary among and within species, suggesting that different life history strategies may provide a mechanism for partitioning limited estuarine habitats (Myers and Horton 1982 as cited in Bottom et al. 2005). In the Columbia River estuary, subyearling Chinook salmon are most abundant from May through September but are present all year (Rich 1920 and McCabe et al. 1986 as cited in Bottom et al. 2005). The recent United States Army Corps of Engineers (USACE) Columbia Estuary Ecosystem Restoration Program (CEERP) Synthesis Memo (Thom et al. 2012) provides a recent synopsis of our current understanding of salmonid migratory and habitat use patterns:

1. Six species of salmonids use shallow-water and wetland habitats within the lower river, including peripheral bays and backwater sloughs: Chinook salmon, coho salmon, chum salmon, sockeye salmon (*O. nerka*), steelhead and coastal cutthroat trout (*O. clarkii*) with Chinook, chum and coho found in higher abundances.
2. The various (ESUs) display variations in juvenile life history characteristics, including in the timing and pathways of their migrations.
3. Chinook and coho salmon exhibit yearling and subyearling life-history types, while chum are primarily captured as fry migrants.
4. Yearling Chinook and coho salmon and steelhead primarily use main channel migratory pathways during spring (as cited in Thom et al. 2012: Dawley et al. 1986; Magie et al. 2008; Weitkamp et al. 2012), and larger smolted subyearling Chinook salmon also tend to migrate rapidly through the lower river (as cited in Thom et al. 2012: Dawley et al. 1986; Harnish et al. 2012). However, a portion of these larger fish are also found in shallow-water habitats (as cited in Thom et al. 2012: Poirier et al. 2009; Bottom et al. 2011; Sather et al. 2011; Roegner et al. 2012).

5. Smaller subyearling Chinook and chum salmon make substantial use of shallow tidal habitats, and subyearling coho are often abundant in the lower sections of tributaries (as cited in Thom et al. 2012: Poirier et al. 2009a, b; Roegner et al. 2010; Sagar et al. 2011; Sagar et al. 2012).

National Marine Fisheries Service (NMFS) recommends that the LCRE contributes to the viability and persistence of all anadromous salmonid populations within the Columbia River Basin in the following ways: 1) the amount of estuarine habitat that is accessible affects the abundance and productivity of a population; 2) the distribution, connectivity, number, sizes, and shapes of estuarine habitats affect both the life history diversity and the spatial structure of a population; and 3) attributes of estuarine habitats (e.g., temperature and salinity regimes, food web interactions) affect diversity and productivity of populations (Fresh et al. 2005). Diverse habitats and the expression of life history strategies based on use of these habitats are directly linked to salmon population viability (i.e., persistence) over long time scales (McElhany et al. 2000). Hence, changes to the estuarine ecosystem such as degradation and loss of estuarine habitat, can directly alter salmonid population viability.

Because of the increasing awareness of the importance of the Columbia River estuary for salmonid life cycles, protection and restoration of important salmonid habitats within the estuary has been identified as a priority for salmon recovery. In addition, in life stage risk and sensitivity modeling, Kareiva et al. (2000) and McClure et al. (2003) found that to recover salmonid populations in the Columbia River Basin additional actions above and beyond passage improvements at the Federal Hydropower System dams were needed and that the life stages for rearing in the river, estuary, and ocean were sensitive to disturbances (cited in Fresh et al. 2005). Kareiva et al. (2000) concluded that the maximum potential to contribute to anadromous salmonid recovery was associated with these life stages but could not distinguish between these life stages nor determine how much of a change in survival was possible (Fresh et al. 2005).

Many of the endangered ESUs in the Columbia River Basin have been found in marsh and forested wetland habitats in the estuary (Fresh et al. 2005). The existence of sufficient amounts and connectivity of appropriate habitat subtypes allow salmon and steelhead to express the appropriate spatial structure and diversity of life history strategies demanded by the challenging environmental conditions faced by the juvenile salmon. The member/vagrant hypothesis states that the pattern and richness (diversity) of populations are determined by the number and location of geographic features where a species' life cycle can be fulfilled (Sinclair 1988 as cited in Fresh et al. 2005). The diverse life history strategies of salmon within the Columbia River include variations in timing and size of outmigrating fish and length of estuary residence. Migration timing, fish size, and rearing periods are linked to fine-scale features within each aquatic environment where salmon acclimate, seek refuge, and feed.

1.3 LCRE Ecosystem Knowledge Gaps

In addition to addressing LCRE ecosystem questions, the Estuary Partnership and regional partners are interested in identifying what ecosystem factors affect juvenile salmonid performance. The Columbia River Data Development Program (CREDDP) completed an interdisciplinary analysis of the Columbia River estuary in the 1980s, which provided valuable information related to fundamental estuarine processes, effects of historical changes on the estuarine ecosystem, and a comprehensive analysis of the recovery of marked juvenile salmon in the estuary (Bottom et al. 2005). Despite the CREDDP research, fundamental studies evaluating the links between juvenile salmon and estuarine conditions in the Columbia River are generally lacking. Key information gaps related to salmonids include:

1. Specific (especially shallow-water) habitats used by salmon during rearing and outmigration through the estuary;
2. Effects of physiochemical and biological conditions on estuarine residence times, growth, or survival of juvenile salmon;
3. Food-chain relationships (feeding and predation) among juvenile salmon, invertebrate prey, and vertebrate predators; and
4. Differences in estuarine habitat needs and ecological relationships among salmon species, life history types, and source populations (Bottom et al. 2005).

Additionally, at the time this work began, there was a lack of research and monitoring within the tidal freshwater section of the LCRE, resulting in little basic understanding of food web dynamics in the region. Historically, most research had occurred in the lower estuary, Reaches A through C (e.g., Bottom et al. 2005; Roegner et al. 2010). Recent and ongoing research is continuing to expand our basic understanding and fill these knowledge gaps, including the CEERP synthesis (i.e., Thom et al. 2012). However, these are significant questions that will take time to address and will continue to hinder salmon recovery programs in the basin. The Estuary Partnership with regional partners identified a lack of sustained status and trends monitoring in the estuary, which decreases our ability to evaluate management actions related to improving estuarine conditions for salmonids and other organisms. The Estuary Partnership's EMP was developed to provide a data set of current and changing conditions in the LCRE.

1.4 Ecosystem Monitoring Program

1.4.1 Background

As an NEP, the Estuary Partnership works with regional partners (local, state, federal, and tribal governments, industry, citizens, not-for-profits, and academia) to develop and implement a Comprehensive Conservation and Management Plan (CCMP). Ecosystem monitoring is a key element of the Estuary Partnership's CCMP. Action 28 of our CCMP calls for the Estuary Partnership, with its partners, to implement sustained long-term monitoring to understand conditions in the river and to evaluate the trends and impacts of management actions over time (Estuary Partnership 1999). This monitoring program was also intended to address Reasonable and Prudent Alternatives (RPAs) 161, 163, and 198 of the 2000 Biological Opinion for the Federal Columbia River Power System, and RPAs 58, 59, 60, and 61 of the 2008 Biological Opinion.

When the EMP began in 2004, the Pacific Northwest National Laboratory (PNNL) and the Estuary Partnership outlined a two-phase process for developing a sampling design for LCRE monitoring efforts (Estuary Partnership 2004). Phase I would inventory habitats in the LCRE. During Phase I, habitat monitoring would describe estuarine habitat types and provide field verification of remote sensing–based project components like the Columbia River Estuary Ecosystem Classification System (Classification) (see Simenstad et al. 2011), and measure variability in estuarine habitats so that an appropriate population could be identified for sampling in subsequent efforts. Phase II Long-Term Monitoring was to combine the habitat inventory with the Classification in order to refine the sampling design. Both phases were to incorporate a probabilistic (e.g. stratified rotational) sampling design based on the Classification and include fixed and randomly selected monitoring sites.

Originally, the geographic scope of the habitat monitoring plan was the entire estuary as defined by the highest uncontrolled flood elevation. The scope of the sampling design was shallow-water aquatic habitats (e.g., marshes), riparian fringe habitats, and adjacent small tidal channels in undiked areas of the estuary between the mouth and Bonneville Dam (Estuary Partnership 2004). The navigable waterway and mainstem channel were excluded from the statistical population under consideration. As part of the initial stages of the Classification, using a geographic information system (GIS) platform and bathymetry and topography data sets, the LCRE was divided into eight hydrogeomorphic reaches. The 2004 plan then outlined the following sampling strategy for each reach: "… 20 points will be sampled, providing a total population of 160. We [the Estuary Partnership] will initiate a stratified rotational sampling design, which will utilize both fixed and randomly selected sites. Fixed sites will represent those areas closest to a pristine condition in each stratum and are intended to be carried through to Phase II of the project. These sites may already be recognized as having ecological value and should be included in the initial stage to maximize data for the long-term component of the plan" (Estuary Partnership 2004).

As a result of cost considerations, the Estuary Partnership originally proposed a scaled-down version of the 2004 design to implement Phase II monitoring. This included: 1) assess the current status of primary and secondary productivity in three or more strata of instream channel habitats (e.g., deep main channel, shallow channel margins, lateral side-channel) of the lower river; 2) implement a status and trends monitoring program that fully describes the key ecological conditions within estuarine shallow and inter-tidal habitats of the LCRE using data from vegetation, water quality, sediment cores, and prey monitoring; and 3) conduct systematic monitoring of the water and sediment quality and vegetation structure of the habitats where juvenile salmonids are present. This proposed monitoring would have focused on more than one habitat type within the LCRE and status and trends monitoring. The Estuary Partnership proposed a panel design, including eight fixed sites (one in each hydrogeomorphic reach) and randomly distributed probabilistic sites in each reach. Together, the fixed and randomly distributed sites would have yielded information on the spatial and temporal variation of habitat conditions using the statistical design proposed by the Estuary Partnership (2004).

The proposed project was not fully funded, however, resulting in the removal of primary and secondary productivity from the work effort and a narrowed monitoring focus of one reach per year of sampling and one specific habitat type (emergent wetlands that are undisturbed, or "closest to pristine condition"; Estuary Partnership 2004. As a result of the preferential site selection criteria (see section 2.2 below) necessitated by limited funds; the sampling effort continues Phase I Inventory activities and begins to implement Phase II Long-Term Monitoring activities. While this sampling plan does not constitute a rotational panel design as originally envisioned by the Estuary Partnership (2004), the current approach is appropriate for an observational study characterizing the condition of selected habitats in the LCRE. Recognizing this limitation, the goals and objectives of the project have been refined to reflect this focus on undisturbed emergent wetlands and their role as salmonid habitat. Additionally, since this monitoring effort concentrates on a specific habitat type and is not based on a probabilistic design, results cannot be inferred to all tidally influenced wetlands within a reach or at the estuary scale. Although there are no randomly selected sites in this program, selected sites can be re-sampled over time to provide information on trends in undisturbed emergent wetlands.

1.4.2 Goals and Objectives

To address the identified knowledge gaps in the LCRE and the Estuary Partnership's and regional partner's goals, the EMP objectives between 2005 and 2010 were to track the status and trends of ecosystem condition to inform decisions for the purpose of conserving and restoring the LCRE through:

1. A comprehensive assessment of status (spatial variation) and trends (temporal variation) of habitat, fish, food web, and abiotic conditions in the lower river, focusing on shallow-water and vegetated habitats used extensively by juvenile salmonids for rearing and refugia;
2. A coordinated effort to gather baseline data about estuarine resources (from Johnson et al. 2004);
3. A determination of the variety of salmon life histories currently expressed in the estuary and habitats that support them (from Bottom et al. 2005); and
4. A better understanding of salmon habitat associations to improve predictions of habitat opportunity restoration strategies (from Bottom et al. 2005).

To address these goals, the EMP partnership has collected data on habitat structure (vegetation community, water surface elevation, channel morphology, sediment grain size and total organic content [TOC], sediment accretion, and site profiles; by PNNL), fish use (fish community, salmon metrics and diet; by National Oceanic and Atmospheric Administration [NOAA], NMFS), abiotic site conditions (water temperature, specific conductance, pH, dissolved oxygen, and depth); by USGS),macroinvertebrate prey availability and water temperature at the time of fish sampling (by NOAA, National Marine Fisheries Service or NMFS).

2.0 Methods

2.1 Sampling Design

The LCRE extends from the plume of the Columbia River upstream to the Bonneville Dam at river kilometer (rkm) 235. The Estuary Partnership and monitoring partners use a multi-scaled stratification sampling design for the habitat monitoring component of the EMP using the Classification. The LCRE is divided by major hydrogeomorphic transitions, yielding eight reaches, each with unique characteristics and physical processes (Figure 1; Simenstad et al. 2011). Reach boundaries are based on the USEPA's Level IV Ecoregions, which were modified to include important parameters such as salinity intrusion, maximum tide level, upstream extent of current reversal, geology, and major tributaries.

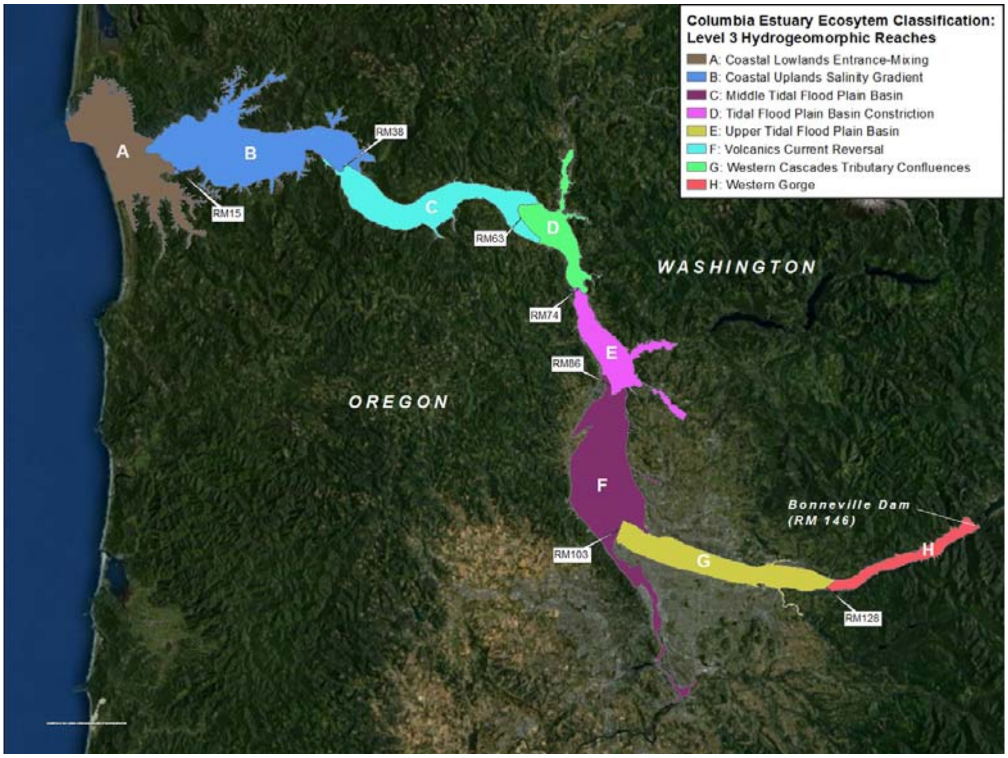


Figure 1. LCRE with hydrogeomorphic reaches (A–H) outlined and specified by color.

The Estuary Partnership and its monitoring partners have focused on providing an inventory of salmon habitats (or "status") across the lower river stratifying by hydrogeomorphic reach (A–H) and including a growing number of fixed sites for interannual variability (or "trends"). Table 1 below shows the specific sites that have been monitored since the beginning of the EMP and the type of sampling associated with the site. Data collected for the EMP after 2010 are not included since they were not available at the time data compilation for this synthesis was begun. Sampling in Reaches A and B began in 2011 and therefore only Reaches C through H are discussed in this synthesis. Habitat structure and hydrology data began to be collected in 2005, fish data collection began in 2007, fish prey data collection began in 2008, and water quality data collection has been conducted at a small number of sites, dependent on funding levels.

The focus of the EMP has been on minimally disturbed tidally influenced emergent wetland sites. Each year, three to four "status" sites, in a previously unsampled reach, were selected along with the continued sampling of a growing number of "fixed sites." Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2010), White Island (2009, 2010), and Franz Lake (2008, 2009) are the three fixed sites. Cunningham Lake is an additional site used to collect trend information on habitat structure and hydrology as supplemental, comparison data for the Campbell Slough site, due to the irregular presence of cows at Campbell Slough that may influence vegetation biomass.

Table 1. Summary of sampling effort by site and year(s) for sites between 2005 and 2010.

Reach	Type of Site	Site	Habitat Structure and Hydrology	Fish and Prey	Water Quality
C	Status	Ryan Island	2009	2009	
	Status	Lord-Walker Island 1	2009	2009	
	Trend	**Whites Island**	2009-2010	2009–2010	2009
	Status	Jackson Island	2010	2010	
	Status	Wallace Island	2010	2010	
	Status	Bradwood Landing	No access permission	2010	
D	Status	Cottonwood Island small slough	2005		
	Status	Cottonwood Island large slough	2005		
	Status	Dibble Slough	2005		2005
E	Status	Sandy Island 1, 2	2007	2007	
	Status	Lewis River Mouth	2007		
	Status	Martin Island	2007		
F	Status	Sauvie Cove	2005		
	Status	Hogan Ranch	2005		
	Trend	**Cunningham Lake**	2005–2010		
	Trend	**Campbell Slough**	2005–2010	2007–2010	2008– 2010
G	Status	Water Resources Center	2006		
	Status	McGuire Island	2006		
	Status	Old Channel Sandy River	2006		
	Status	Chattam Island	2006		
H	Trend	**Franz Lake**	2008–2009	2008–2009	
	Status	Sand Island	2008	2008	2008
	Status	Beacon Rock	2008	2008	
	Status	Hardy Slough	2008	2008	

2.2 Site Selection

As part of the sampling design for the EMP, the Estuary Partnership, University of Washington (UW), and USGS developed a hydrogeomorphic classification scheme for the LCRE. Based on classification schemes developed for other estuarine ecosystems and concepts of ecosystem geography (Bailey 1996), the LCRE Classification has six hierarchical levels (see Simenstad et al. 2011):

1. Ecosystem Province (based on USEPA Ecoregion Level II)
2. Ecoregion (based on USEPA Ecoregion Level III)
3. Hydrogeomorphic Reach (based on modified USEPA Ecoregion Levels III and IV)
4. Ecosystem Complex (based on Primary Cover Class and geomorphic setting within each hydrogeomorphic reach)
5. Geomorphic Catenae
6. Primary Cover Class (based on cover data from Landsat or other remote sensing data sets)

Levels 1 and 2 were taken directly from the USEPA Ecoregion data set, and required no additional mapping. Eight distinct reaches were defined, representing the intersection of broad-scale geologic processes and events over the last 50 million years with more modern or recent geologic and hydrologic processes of the Holocene period. The major hydrologic processes influencing reach boundaries include locations of current reversal, salinity intrusion, confluences of major tributaries, and maximum tide levels. The eight reaches are illustrated in the study area map above (Figure 1).

To characterize juvenile salmon habitat across the spatial extent of the LCRE, the EMP sampled one new reach per year. Prior to a site visit, the potential areas were evaluated using GIS layers including current imagery, Light Detection and Ranging (LiDAR) digital elevation models, and historical maps from the late 1800s. Using these sources of information, the potential sites were narrowed to those that appeared relatively undisturbed and hydrologically connected to the mainstem of the Columbia River. A set of potential sites was then assessed in the field, and a final set of sites was agreed upon by all monitoring partners (PNNL, NOAA, USGS, and the Estuary Partnership). The final habitat criteria used to select monitoring sites were:

1. The site's wetland vegetation is classified as "emergent" in the National Wetland Inventory (NWI; available at http://www.fws.gov/wetlands/index.html /).
2. The site has tidal connectivity with the mainstem Columbia River.
3. The site's wetland is minimally disturbed (e.g., no diking, active grazing, tide-gates, or modifying flow regime present at the site).
4. The area of wetland is greater than 5 acres.
5. Wetlands at the site are shallow water.
6. The site is mainstem fringing or off-channel habitat.
7. The site is not located near immediate stressors or disturbances like industry, grazers, or recreational use.
8. Site sediments are generally smaller particle sizes, which are characteristic of lower-energy systems and more likely to support emergent marsh habitats than habitats with larger particle sizes.

Additional logistical criteria included:

1. Slough channels are present at the site to facilitate the collection of cross-section and fish data.
2. The site is fishable by beach seine or similar gear type.
3. The site is accessible for sampling purposes with landowner permission.

The final criteria for site selection were based on funding levels, the desire for data comparability with previously collected data, and other reasons outlined above.

2.3 Habitat Structure and Hydrology

One of the primary goals of the EMP is to evaluate the status and trends of wetland ecosystems in the LCRE. This study uses standard monitoring protocols developed for the LCRE (Roegner et al. 2009) for this task. Five metrics are included in this part of the monitoring program. These metrics have been determined to represent important structural components that can be used to determine habitat functions. The rationale for choosing these metrics is discussed below.

Elevation, hydrology, and substrate are the primary factors that control wetland vegetation composition, abundance, and cover. Knowing the elevation, soil, and hydrology required by native tidal wetland vegetation is critical to designing and evaluating the effectiveness of restoration projects (Kentula et al. 1992). Sediment accretion is important for maintaining wetland elevation. Accretion rates can vary substantially between natural and restored systems (Diefenderfer et al. 2008); therefore, baseline information on rates is important for understanding potential evolution of a reference or restoration site. Evaluating vegetation composition and species cover provides an indication of the many functions provided by wetland vegetation. These functions include the production of organic matter (macrodetritus), food web support, habitat for many fish and wildlife species including salmon, and contributing to overall biodiversity of the Columbia River estuarine ecosystem. Likewise, collection of vegetation biomass is being conducted at the core sites to begin to quantify the contribution of organic matter from these wetlands to the ecosystem.

Assessment of channel cross-sections and channel networks provides information on the potential for many important estuarine functions including fish access (Simenstad and Cordell 2000) and export of prey, organic matter, and nutrients. This information is also necessary to develop the relationship between cross-section dimensions and marsh size, which aids in understanding the channel dimensions necessary for a self-maintaining restored area (Diefenderfer and Montgomery 2009). The primary objective associated with the channel data collection effort is to determine how unmodified channels may differ between reaches within the region with regard to habitat opportunity.

2.3.1 Status Site Analysis

Each year the Estuary Partnership evaluates the status of a number of sites, characterizing their habitat structure. The Estuary Partnership has begun to analyze data collected over the past seven years to evaluate trends. The analysis of trends over time was initiated in 2010 (Sagar et al. 2012) and continued in 2012 with the addition of a spatial pattern analysis (i.e., status sites). The primary questions on which the spatial pattern analysis is focused are:

1. Are there differences in the wetland vegetation species assemblages along the estuarine gradient?
2. Are there patterns in the elevation ranges of vegetation communities throughout the estuary?
3. Is there a discernible lower elevation limit of reed canarygrass (*Phalaris arundinacea*) in the estuary?
4. What are the patterns of hydrologic inundation and how do they vary in the estuary?
5. Are the inundation patterns in the estuary related to variations in vegetation communities?
6. How do wetland channels vary along the estuarine gradient?

To compare marsh habitats throughout the estuary, we combined the data sets on structural habitat metrics from multiple programs. Structural habitat data from relatively undisturbed emergent marshes between Reaches C and H have been collected as part of this study since 2005. Additionally, similar data from multiple undisturbed habitat types, including marshes, were collected throughout the LCRE as part of the Reference Site Study (Borde et al. 2011) and the Cumulative Ecosystem Response to Restoration Study (Johnson et al. 2011) between 2005 and 2010. Finally, data from three marsh sites in Reach G that were part of the Tidal Freshwater Monitoring program (Johnson et al. 2011) were also included in the analysis, bringing the total number of sites analyzed to 39.

The term "historical class" (Table 2) refers to two categories that can be applied to the marsh sites to provide context for their ages and disturbance histories; they are defined as follows:

Historical:	Sites present, in some form, on historical maps from the late 1800s.
Created:	Sites that were not present on the historical maps and have been created from modified sediment and hydrologic processes and placement of dredge material.

2.3.2 Trend Site Analysis

A multiyear analysis was conducted as part of this program in 2010 and included data from two sites (Cunningham Lake and Campbell Slough) that had been monitored from 2005 to 2009 and one site (Franz Lake) that had been monitored in 2008 and 2009 (Sagar et al. 2012). This analysis included the calculation of inundation (sum exceedance value [SEV]) and vegetation cover. We updated these calculations using data from 2010 and 2011 and added an additional site (Whites Island) that had been monitored from 2009 to 2011. The SEV was calculated using the method established in the previous analysis (Sagar et al. 2012), where data from the nearest long-term water level station (in the mainstem of the river) was modified using regression analysis with the on-site data. In the updated analysis we also evaluated interannual changes in channel morphology in locations where channel cross-sections had been measured in multiple years and sediment accretion rates were available.

In the 2012 trend analysis, we also included a similarity analysis of the vegetation species assemblage and cover. Similarity analyses, using the Bray-Curtis similarity coefficient (S') as a measure of distance between years (described in Clarke and Warwick 2001), were performed on percent cover data from each site by using Primer™. Percent cover data were arc-sin, square-root transformed, but were not standardized, prior to analyses. The similarity matrix was converted to a dendrogram by using the hierarchical, unweighted pair-group mean-averaging method of clustering. Clusters combined at greater linkage distance are more dissimilar than those combined at smaller linkage distances.

Table 2. Marsh study sites included in spatial patterns analysis.

Site Name	**Site Code**	**Distance from Columbia River mouth (river kilometer [rkm])**	**Reach**	**Monitoring Year**	**Historical Class**
Chinook River, mouth*	CHM	12	A	2009	historical
Lewis & Clark River, mouth	LCM	20	A	2009	historical
Cooperage Slough	CSM	tributary	A	2007	historical
Grant Island	GIM	tributary	A	2009	historical
Secret River*	SRM	37	B	2008	historical
Miller Sands	MSC	39	B	2009	created
Welch Island	WIM	53	B	2008	historical
Ryan Island	RIM	61	C	2009	historical
Jackson Island	JIC	71	C	2009	created
Whites Island, cut-off slough	WHC	72	C	2009/10	created
Wallace Island West	WAC	77	C	2010	created
Clatskanie River (near Anunde Island)	CRM	80	C	2009	historical
Gull Island	GUC	89	C	2009	created
Lord-Walker Island	LI1	99	C	2009	created
Lord Island	LI2	100	C	2009	created
Dibblee Slough	DSC	104	C	2005	created
Cottonwood Island, large slough	CI1	113	D	2005/10	created
Cottonwood Island, small slough	CI2	114	D	2005/10	created
Sandy Island, north slough	SI1	121	E	2007	created
Sandy Island, south slough	SI2	123	E	2007	created
Martin Island	MIM	129	E	2007	historical
Goat Island Slough	GIC	131	E	2009	created
No-name Island (7.5 rkm downstream of the Lewis River mouth)	DMI	136	E	2007	created
Scappoose Bay, mouth of McNulty Cr.	SBM	143	F	2010	historical
Cunningham Lake	CLM	145	F	2005–10	historical
Campbell Slough	CS1	149	F	2005–10	historical
Sauvie Island, east slough (inside Willow Bar)	SSC	154	F	2005	created
Water Resources Center	WRC	175	G	2006	historical
McGuire Island	MIC	190	G	2006	created
Washougal River mouth	WRM	195	G	2010	historical
Sandy River, historical channel	OSR	196	G	2006	historical
Sandy River, historical mouth	OSM	198	G	2007	historical
Gary Island	GAM	200	G		historical
Chattham Island	CIC	201	G	2006	created
Reed Island	RIC	201	G		created
Sand Island (near Rooster Rock State Park)	SIM	211	H	2008	historical
Franz Lake (near mouth)	FLM	221	H	2008	historical
Pierce Island	PIM	228	H	2008	historical
Hardy Creek	HCM	230	H	2008	historical

*CHM and SRM had two vegetation sample plots at each site to capture high and low marsh vegetation communities; they are referred to as CHM-L, CHM-H, SRM-L, and SRM-H.

2.4 Fish

Between 2007 and 2010, NOAA-Fisheries conducted surveys to monitor prey availability and juvenile Chinook salmon habitat occurrence at Hardy Slough, Franz Lake, Pierce Island, and Sand Island in Reach H; Campbell Slough, near Ridgefield National Wildlife Refuge, in Reach F; Sandy Island in Reach E; and Wallace Island, Ryan Island, Lord-Walker Island, Whites Island, and Bradwood Slough in Reach C (Figure 1). The location and year that each of the sites were sampled are shown in Table 2. Franz Lake was sampled in 2008 and 2009, while Campbell Slough was sampled in 2007, 2008, 2009, and 2010.

The primary questions on which the spatial pattern and trend analysis are focused were:

1. What are the fish community characteristics in the LCRE and how do they change spatially and temporally throughout the LCRE?
2. When and where are salmon species present?
3. How much variability in salmon use/species is there between years and between sites in the LCRE?
4. How do size class, condition, and contaminants of juvenile salmon differ both spatially and temporally in the LCRE?
5. Which sites are used more by marked versus unmarked Chinook salmon?
6. How does the Chinook salmon stock usage of sites in the LCRE change spatially or temporally?

2.4.1 Fish Monitoring and Sample Collection Methods

Monitoring for fish and prey was generally initiated in April and continued on a monthly basis through August or September; exact sampling times for each site and year are shown in Table 2. Fish were collected with a Puget Sound beach seine (PSBS; 37 × 2.4 meter [m], 10-millimeter [mm] mesh size) and a "baby" beach seine (BBS; 10 × 1.5 m, 5-mm mesh size) at shallow-water sites where boat deployment was not possible. Up to three sets were performed at each site at each sampling time, as site conditions and sampling permit limitations allowed. All fish in each set were identified to species and counted. Salmonids were examined for fin clips and coded wire tags (CWTs) to determine the proportions of marked fish (of known hatchery origin) and unmarked fish (potentially wild). Subsets of up to 30 juvenile Chinook, coho, and chum salmon from each set were measured (to nearest mm) and weighed (to nearest 0.1 gram [g]). Additionally, from Chinook salmon, the following samples were collected: stomach contents for prey taxonomy; whole bodies for measurement of lipid content and classes; otoliths for estimation of age and growth rates; and fin clips for genetic stock identification (GSI). As time and fish availability permitted, the following samples were also collected: bile for measurement of metabolites of polycyclic aromatic hydrocarbons (PAHs); stomach contents for measurement of PAHs and other persistent organic pollutants (POPs), including PCBs, DDTs and organochlorine pesticides, and PBDEs; and whole bodies for measurement of bioaccumulative POPs. Water temperature, dissolved oxygen, and tide condition were measured and recorded at each sampling time as well. Samples for chemical analysis were held on dry ice and transported to the Northwest Fisheries Science Center (NWFSC) laboratory, where they were stored frozen at 40°C (degrees Celsius) until analyses were performed. Stomach contents samples for taxonomic analysis were preserved in ethanol.

2.4.2 Fish Sample Analyses

Multiple regression was used to examine the effects of fish type (unmarked versus marked), site of capture, month of capture, and year of capture on length, weight, and condition factor. Analyses were conducted with the JMP statistical package.

Lipid determination: For lipid and chemical analyses, individual Chinook salmon bodies (carcass plus all internal organs except for the stomachs) were combined to produce composite samples consisting of three to five fish each from the same site, sampling time, genetic stock, and origin (unmarked versus marked). The amount of total, nonvolatile, extractable lipid (reported as percent lipid) and lipid classes in the body composites were determined using thin-layer chromatography–flame ionization detection (TLC–FID) with Iatroscan analysis as described by Ylitalo et al. (2005).

Chemical contaminants in stomach contents and body samples: Body composite and stomach content samples were analyzed by gas chromatography–mass spectrometry (GC–MS) for PCB congeners, PBDE congeners, DDTs, DDT isomers, and other organochlorine pesticides (hexachlorocyclohexanes [HCHs], hexachlorobenzene [HCB], chlordanes, aldrin, dieldrin, mirex, and endosulfans) as described by Sloan et al. (2005, 2010). In addition to PBDEs, PCBs, and pesticides, stomach content samples were also analyzed for low (2–3 ring) and high (4–6 ring) molecular weight aromatic hydrocarbons using capillary column GC–MS (Sloan et al. 2004, 2006). Summed low-molecular-weight aromatic hydrocarbons (∑LAHs) were determined by adding the concentrations of biphenyl, naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, 2,6-dimethylnapthalene, acenaphthene, fluorene, phenanthrene (PHN), 1-methylphenanthrene, and anthracene. Summed high-molecular-weight aromatic hydrocarbons (∑HAHs) were calculated by adding the concentrations of fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[a]pyrene (BaP), benzo[e]pyrene, perylene, dibenz[a,h]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, indenopyrene, and benzo[ghi]perylene. Summed total aromatic hydrocarbons (∑TAHs) were calculated by adding ∑HAHs and ∑LAHs. Aromatic hydrocarbons were not measured in body samples because fish metabolize these compounds and accumulation in tissues is very limited.

Polycyclic aromatic hydrocarbon metabolites in Chinook salmon bile: Due to the relatively small volume of bile that can be collected from individual subyearling Chinook salmon, bile samples were composited from up to 30 individual fish per site and sampling time to provide an adequate sample volume (25 microliter [µL]) for analyses. Bile samples were analyzed for metabolites of PAHs using a high-performance liquid chromatography (HPLC)–fluorescence detection method described by Krahn et al. (1986). Chromatograms were recorded at the following wavelength pairs: (1) 260–380 nm, where several 3- to 4- ring compounds (e.g., PHN) fluoresce, and (2) 380–430 nm, where 4- to 5-ring compounds (e.g., BaP) fluoresce. The concentrations of fluorescent PAHs in the bile samples were determined using PHN and BaP as external standards and converting the fluorescence response of bile to PHN (ng [nanograms] PHN equivalents/g bile) and BaP (ngBaP equivalents/g bile) equivalents. Biliary protein was measured according to the method described by Lowry et al. (1951). Biliary fluorescence values were normalized to protein content, which is an indication of feeding state and water content of the bile. Fish that have not eaten for several days exhibit higher biliary fluorescent aromatic compound (FAC) values and higher protein content than fish that are feeding constantly and excreting bile more frequently (Collier and Varanasi 1991).

2.4.3 Genetic Stock Identification

GSI techniques (see Manel et al. 2005) were used to investigate the origins of juvenile Chinook salmon using the Mirror Lake Complex sites, as described by Teel et al. (2009) and Roegner et al. (2010). The stock composition of juveniles was estimated with a regional microsatellite DNA data set (Seeb et al. 2007) that includes baseline data for spawning populations from throughout the Columbia River Basin (described in Teel et al. 2009). The overall proportional stock composition of Mirror Lake samples was estimated with the GSI computer program ONCOR (Kalinowski et al. 2007), which implemented the likelihood model of Rannala and Mountain (1997). Probability of origin was estimated for the following regional genetic stock groups (Seeb et al. 2007; Teel et al. 2009): Deschutes River fall Chinook; West Cascades fall Chinook; West Cascades spring Chinook; Middle and Upper Columbia spring Chinook; Spring Creek Group fall Chinook; Snake River fall Chinook; Snake River spring Chinook; Upper Columbia River summer/fall Chinook; and Upper Willamette River spring Chinook. West Cascades and Spring Creek Group Chinook are Lower Columbia River stocks.

2.4.4 Fish Community Characteristics, Catch per Unit Effort, and Fish Condition Calculations

Fish species diversity was calculated using the Shannon-Weiner diversity index (Shannon and Weaver 1949):

$$H' = -\sum_{i=1}^{S}(p_i \ln p_i)$$

Where

i = the number of individuals in species i; the abundance of species i.

S = the number of species. Also called species richness.

p_i = the relative abundance of each species, calculated as the proportion of individuals of a given species to the total number of individuals in the community.

Catch per unit effort (CPUE) was calculated as described by Roegner et al. (2009), with fish density reported in numbers per 1,000 m^2.

For all salmonid species, Fulton's condition factor (K) (Fulton 1902; Ricker 1975) was calculated as an indicator of fish health and fitness, using the formula:

$$K = [\text{weight (g)}/\text{fork length (cm)}^3] \times 100.$$

2.4.5 Growth Rate Estimates from Otolith Analysis

Fall Chinook salmon were collected and their otoliths extracted, at each of 21 sites from May to June in 2005 and 2007 to 2010. Otoliths from fish ranging in size from 51 to 90 mm (fork length, mean of 71.9 mm, standard deviation of 9.5 mm) were then processed for microstructural analysis of recent growth.

Specifically, left sagittal otoliths were embedded in Crystal Bond© and polished in a sagittal plane using slurries (Buehler©'s 600 grit silicon carbide, 5.0 alumina oxide, and 1.0 micropolish) and a grinding wheel with Buehler© 1500 micropolishing pads. Polishing ceased when the core of the otolith was exposed and daily increments were visible under a light microscope. Using Image Pro Plus© (version 5.1), with a media cybernetics (evolution MP color) digital camera, and operating at a magnification of 20×, the average fish daily growth rates (i.e., mm of fish length/day) were determined for three intervals of time: a) the last seven days of their life, b) the last 14 days of their life, and c) the last 21 days of their life. A total of 326 otoliths were analyzed. Daily growth rate (DG, mm/day) was determined using the Fraser-Lee equation:

$$La = d + [(Lc - d/Oc)*Oa]$$

$$DG = (Lc - La)/a$$

where La and Oa represent fish length and otolith radius at time a (i.e., last seven, 14, or 21 days), respectively, d is the intercept (12.74 mm) of the regression between fish length and otolith radius, and Lc and Oc are the fish length and otolith radius at capture, respectively. An analysis of variance (ANOVA) was used to determine whether growth rates differed among sites. At the two fixed sites that had collections made in multiple years (Campbell Slough and Franz Lake), the temporal variation of growth rates was assessed (using an ANOVA). Finally, using information about genetic stock and presence or absence of hatchery marking, fish were also analyzed (using an ANOVA) to assess whether somatic growth rate differed among stocks and between marked hatchery and unmarked, presumably wild, fish.

2.5 Fish Prey

The purpose of analyzing juvenile Chinook salmon diets and identifying prey taxa in salmon habitats is to quantify the abundance and composition of prey resources for juvenile salmonids and evaluate the potential influence of prey availability on juvenile salmonid occurrence in these habitats. We focus here on the patterns of prey abundance in open-water habitat and the bank-water margin where emergent vegetation is often present, in order to determine which types of prey juvenile salmon consume at different sites and times of the year. A related objective is to use these data to identify potential sources of contaminants affecting fish in the LCRE.

The primary questions on which the spatial pattern and trend analysis are focused were:

1. Which macroinvertebrates are found in the LCRE and how do their abundances change both spatially and temporally?
2. Where are more macroinvertebrates found (emergent versus open water)?
3. Are Chinook salmon selecting for a prey type and are the preferred prey items similar among sites/reaches?
4. Where are Chinook salmon–preferred prey items found (emergent versus open water)?
5. Which age or life cycle stage of macroinvertebrate is preferred?

2.5.1 Fish Prey Sampling

For the invertebrate prey sampling, the objective was to collect aquatic and terrestrial invertebrate samples and identify the taxonomic composition and abundance of salmonid prey available at sites when juvenile salmonids were collected. These data could then be compared with the taxonomic composition of prey found in stomach contents of fish collected concurrently.

A neuston net was used to sample invertebrates in the water column and on the surface in two types of habitat: open-water habitat and the bank-water margin where emergent vegetation is often present. These two types of habitat are often close to each other, and indeed the fish are sampled from across these habitats as the seine is deployed in the open water and pulled up and through the emergent vegetation. We sampled both habitats in an effort to adequately represent the total prey available to fish in these habitats and to assess how prey availability and composition may vary across this fine habitat scale. Each emergent vegetation tow was collected along a 10-m transect adjacent to the area sampled with the beach seine, and each open-water tow was collected across 50 to 100 m (depending on the year; 2008 and 2009 were over 50 m, 2010 samples were collected over 100 m). Two to three tows of each type were collected per site, per sampling period. Invertebrates were also sampled in terrestrial vegetation using sweep nets in 2007 and 2008, but samples were archived because of limited time and funds for analyses and because terrestrial prey were rare in the diets analyzed thus far.

2.5.2 Fish Prey Sample Analyses

Ivlev's prey electivity values (Lechowicz 1982) were calculated for the most abundant taxa in both the diets and the tows to determine if juvenile Chinook salmon preferred or avoided particular prey taxa based on their relative abundance in the environment. Positive values indicate more of these taxa were consumed than would be expected based on their availability in the environment; negative values indicate fewer of these taxa were consumed than would be expected. Values >0.2 and <−0.2 are considered to indicate strong selection or avoidance, respectively, of prey taxa. It should be noted that these values are based on comparing the proportion of prey in the diets to those in the environment, and here we assume the combined means of taxa in the emergent vegetation and open-water tows represent the prey available to salmon. Because juvenile Chinook salmon feed primarily from the surface and mid-water column, and rarely feed from the benthos, samples like the neuston tows that capture prey in the mid- and upper-water column likely best represent the prey available to salmon at the time the fish and invertebrate samples were collected. The tows do not necessarily represent where the greatest abundance of potential prey reside or where production may be greatest; rather, they are meant to quantify prey available in the habitats where fish are caught.

2.6 Abiotic Site Conditions

For three years, USGS deployed a continuous water quality monitor at Campbell Slough in the Roth Unit of the Ridgefield National Wildlife Refuge (2008–2010). This site in Reach F has been sampled for vegetation since 2005 and for fish since 2007. The monitor deployed was a Yellow Springs Instruments (YSI) model 6600EDS equipped with water temperature, specific conductance, pH, dissolved oxygen, and depth probes. The deployment period for this monitor was designed to characterize water quality conditions while juvenile salmonids were present, during the period of time when they migrated away from the sites, and shortly thereafter, in order to assess limiting factors.

The primary questions on which the spatial pattern and trend analysis are focused were:

1. What are the dissolved oxygen, pH, and water temperature at sites and how do they differ both spatially and temporally?
2. How do the daily and weekly average maximum temperatures and minimum dissolved oxygen concentrations at the site compare to state water quality criteria?

2.7 Interdisciplinary Relationships

To investigate how fish communities and juvenile salmon abundance and health are related to other aspects of the larger food web and environment we conducted multivariate analyses. Specifically, each individual (vegetation/hydrology, fish, macroinvertebrate, and water quality) data set was examined together to investigate possible relationships.

The primary questions on which the status and trend analysis are focused were:

1. How is fish diversity related to environmental and biological variables?
2. How is Chinook salmon abundance (Chinook CPUE) related to environmental and biological variables?
3. How is Chinook salmon health (i.e., lipid content) related to environmental and biological variables?

All interdisciplinary data that were collected in the previous sections were assessed to determine their suitability in multidiscipline statistical analyses. Although habitat structure data have been collected since 2005, fish and fish prey data collection was not concurrent until 2008. Due to the lack of overlap between disciplines at the beginning of the program, only data collected at the individual sites from 2008 to 2010 (Table 3) were used in the interdisciplinary analyses. Although Pierce Island was sampled for fish in 2008, the site was subsequently removed from the data set due to a lack of prey data. Bradwood slough was sampled for fish but access was not permitted for vegetation structure and hydrology sampling and was therefore not used in this analysis. Preliminary water quality data were limited in scope and duration and were also not used in the analyses. The amount of usable data limits the sample size and reduces the inferences that can be drawn from the statistical analyses. As a result, emphasis in this report is not placed on determining statistically significant relationships between variables but rather on exploring possible relationships.

Correlation coefficients and subsequent multiple regression analyses were run on numerous combinations of data sets and variables to determine the presence and strength of interdisciplinary relationships. The specific independent, or predictive, variables used in each analysis were selected because they logically could have an effect on the dependent variable in question.

Each independent variable was graphed against the dependent variable, and the resulting scatterplots were examined to determine the nature of their relationship. Pearson’s product-moment correlation coefficients (*r*) were calculated for each graph to quantitatively assess the strength of the relationships between the variables. Values 0.9 to 1.0 were considered very strong, 0.7 to 0.9 were strong/high, 0.4 to 0.7 were moderate, and <0.4 were weak correlations (J. Mason, personal communication). The value 0.4 was used as a threshold since it was considered both inclusive and parsimonious given the quality of the data. Regression modeling was used in tandem with the correlation coefficients to assess the overall relative strength of the relationship between variables.

Due to a small sample size (14 sites maximum), emphasis was not placed on the significance level of the models (<0.05) but on assessing the adjusted R-squared values generated when using two variables in the regression models. Models with higher adjusted R-squared values were deemed to be better than models with lower adjusted R-squared values. The small sample size also limited the number of variables included in each model to two, in order to conserve the degrees of freedom in the model and prevent overfitting of the model to the data.

All assumptions for linear regression models were met for validity before and after regressions were conducted. Regression results were checked using the standard graphical assessments: 1) plot of residuals against fitted values, 2) normal quantile-quantile plot of the residuals, 3) plot of the square root of the standardized residuals against the fitted values, and 4) plot showing Cook's distances of the observed values for the dependent variable. Models not meeting the assumptions were removed from the analysis. Post-hoc diagnostic tests were run on each regression model to assess the normality of the residuals, presence of outliers, and leverage (influence) of individual data points. Data sets not passing this test were removed from the model and the regression was run again. Models were generated for the following analyses and compared for each set of variables. Only the models with the greatest adjusted R-squared values (i.e., the strongest models) and those passing the post-hoc diagnostic tests are reported.

Three fixed sites, Campbell Slough, Franz Lake, and Whites Island, were sampled in multiple years in order to investigate temporal trends. Rather than combining the individual yearly data sets for the interdisciplinary analyses, years and sites were kept separate to maintain a larger sample size. Since this is an exploratory multivariate analysis, it was more important to have a larger sample size to elucidate correlations between variables rather than test for differences between trends at sites.

Table 3. Data collected and produced in the independent disciplines used in the interdisciplinary analyses.

Discipline	Variable	Description
Vegetation[a]	All Vegetation Diversity	Shannon-Weiner species diversity values for native and non-native vegetation
	Native Vegetation Diversity	Shannon-Weiner species diversity values for native vegetation
	Non-native Vegetation Diversity	Shannon-Weiner species diversity values for non-native vegetation
	Reed Canarygrass Coverage	Percent cover of reed canarygrass (most abundant plant species sampled)
	Common Spikerush Coverage	Percent cover of common spikerush (second-most-abundant plant species sampled)
	Wapato Coverage	Percent cover of wapato (*Sagittaria latifolia*; third-most-abundant plant species sampled)
	All Vegetation Richness	Species richness of all native and non-native vegetation species
	Native Vegetation Richness	Species richness of native vegetation species
	Non-native Vegetation Richness	Species richness of non-native vegetation species
Fish	All Fish Species Diversity	Shannon-Weiner fish species diversity values for various sites and years
	Native Fish Species Diversity	Shannon-Weiner diversity values for native fish species for various sites and years
	Non-native Fish Species Diversity	Shannon-Weiner diversity values for non-native fish species for various sites and years
	Chinook Abundance	CPUE of Chinook salmon
	All Chinook Lipid Content	Percent gravimetric lipid content collected from a subset of all Chinook caught
	Unmarked[b] Chinook Lipid Content	Percent gravimetric lipid content collected from a subset of unmarked Chinook caught
Prey	All Prey Species Richness	Species richness of all prey species collected in May (month of highest Chinook salmon abundance)
	Emergent Prey Species Diversity	Shannon-Weiner prey species diversity in emergent vegetation invertebrate tows
	Open Water Prey Species Diversity	Shannon-Weiner prey species diversity in open water invertebrate tows
	Diptera Density in May Emergent Vegetation Tows	Mean density per meter of Diptera in May emergent vegetation tows (month of highest Chinook salmon abundance)
	Abundance of All May Prey	Total abundance of all prey species in May (month of highest Chinook salmon abundance)
Physical	River Kilometer	Distance of site from mouth of Columbia River
	Distance From Mainstem	Distance of site from the mainstem of the Columbia River
	Elevation[c]	Mean elevation of the vegetation sample sites

[a] Data were recorded yearly during late July–early August, and the assumption was made that this time period is peak biomass of the vegetation at each site for use in cross-site comparison.

[b] The term “unmarked” is used rather than “of natural origin or wild” since some hatchery fish are not fin clipped.

[c] Data were collected for more than 50 individual quadrats per site. The mean of these data was assumed to represent the site as a whole.

Table 4. Sites and years used in the multidiscipline analyses.

Site	Year
Campbell Slough	2008
Campbell Slough	2009
Campbell Slough	2010
Franz Lake	2008
Franz Lake	2009
Hardy Creek	2008
Jackson Island	2010
Lord-Walker Island	2009
Pierce Island	2009
Ryan Island	2009
Sand Island	2008
Wallace Island West	2010
Whites Island	2009
Whites Island	2010

2.7.1 Status Sites

Fish Diversity

The relationship of environmental and biological variables to all fish species diversity and native fish species diversity were examined using linear regression modeling. Sites and data used in this analysis are shown in Table 5 and Table 6.

Table 5. Site data and variables used to assess relationship of environmental variables to All Fish Species Diversity.

Sites Used	Variables Used
Campbell Slough 2008	All Vegetation Diversity
Campbell Slough 2009	Native Vegetation Diversity
Campbell Slough 2010	Non-Native Vegetation Diversity
Franz Lake 2008	Reed Canarygrass Coverage
Franz Lake 2009	Common Spikerush Coverage
Hardy Creek 2008	Wapato Coverage
Jackson Island 2010	All Vegetation Richness
Lord-Walker Island 2009	Native Vegetation Richness
Ryan Island 2009	Non-Native Vegetation Richness
Sand Island 2008	All Prey Species Diversity
Wallace Island West 2010	Emergent Prey Species Diversity
Whites Island 2009	Open Water Prey Species Diversity
Whites Island 2010	River Kilometer

Table 5. Site data and variables used to assess relationship of environmental variables to All Fish Species Diversity.

Sites Used	Variables Used
	Distance From Mainstem
	Elevation

Table 6. Site data and variables used to assess relationship of environmental variables to Native Fish Species Diversity.

Sites Used	Variables Used
Campbell Slough 2008	All Vegetation Diversity
Campbell Slough 2009	Native Vegetation Diversity
Campbell Slough 2010	Non-Native Vegetation Diversity
Franz Lake 2008	Reed Canarygrass Coverage
Franz Lake 2009	Common Spikerush Coverage
Hardy Creek 2008	Wapato Coverage
Jackson Island 2010	All Vegetation Richness
Lord-Walker Island 2009	Native Vegetation Richness
Ryan Island 2009	Non-Native Vegetation Richness
Sand Island 2008	All Prey Species Diversity
Wallace Island West 2010	Emergent Prey Species Diversity
Whites Island 2009	Open Water Prey Species Diversity
Whites Island 2010	River Kilometer
	Distance From Mainstem
	Elevation

Chinook Salmon Abundance

Chinook salmon are of particular interest as expanding research has documented that this species, especially Chinook subyearlings, can rear extensively in shallow water and vegetated habitats within the estuary (Thom et al. 2012). Additionally, this species was the most abundant salmon species and was captured in most months and at all sites sampled. For these reasons, Chinook salmon were a focus of the interdisciplinary analysis.

At sites where fish were sampled, the sampling occurred about once a month from April to August. Chinook Abundance was standardized by calculating the CPUE, which was calculated by dividing the number of fish caught in each seine net cast by the area sampled during the cast. CPUE was then standardized for a sampling unit of 1,000 square meters (m^2). See section 2.4.1 for additional sampling methods.

The influence of environmental variables on Chinook Salmon Abundance was examined using linear regression modeling. Sites and data used in this analysis are shown in Table 7. Chinook Abundance data were modeled for the entire season to better describe the full salmon migratory season. Keeping the aforementioned data sets in the analysis would have excluded the use of the prey and vegetation variables in the multivariate analysis. It was more important to retain the prey and vegetation variables for modeling than retaining the two sites as Chinook Salmon Abundance data points.

Table 7. Site data and variables used to assess relationship of environmental variables to Chinook Salmon Abundance.

Sites Used	**Variables Used**
Campbell Slough 2008	All Vegetation Diversity
Campbell Slough 2009	Native Vegetation Diversity
Campbell Slough 2010	Non-native Vegetation Diversity
Franz Lake 2008	Reed Canarygrass Coverage
Franz Lake 2009	Common Spikerush Coverage
Hardy Creek 2008	Wapato Coverage
Jackson Island 2010	All Vegetation Richness
Lord-Walker Island 2009	Native Vegetation Richness
Ryan Island 2009	Non-native Vegetation Richness
Sand Island 2008	All Fish Species Diversity
Wallace Island West 2010	Native Fish Species Diversity
Whites Island 2009	Non-native Fish Species Diversity
Whites Island 2010	All Prey Species Richness
	Emergent Prey Species Diversity
	Open Water Prey Species Diversity
	River Kilometer
	Distance From Mainstem
	Elevation

Chinook Salmon Health

Lipid content can be a useful indicator of salmon health (Biro et al. 2004), and also affects contaminant uptake and toxicity (Elskus et al. 2005). Studies show that the tissue concentration of a lipophilic chemical that causes a toxic response is directly related to the amount of lipid in an organism (Lassiter and Hallam, 1990; van Wezel et al., 1995); in animals with a high lipid content, a higher proportion of the hydrophobic compound is associated with the lipid and unavailable to cause toxicity. See section 2.4.2 for lipid collection and analysis methods.

The influence of environmental variables on Chinook Lipid Content was examined using linear regression modeling. Sites and data used in this analysis are shown in Table 9. The influence of environmental variables on Unmarked Chinook Lipid Content was examined using linear regression modeling. Sites and data used in this analysis are shown in Table 8.

Table 8. Site data and variables used to assess influence of environmental variables on All Chinook Lipid Content.

Sites Used	Variables Used
Campbell Slough 2008	All Vegetation Diversity
Campbell Slough 2009	Native Vegetation Diversity
Franz Lake 2009	Non-native Vegetation Diversity
Lord- Walker Island 2009	Reed Canarygrass Coverage
Pierce Island 2009	Common Spikerush Coverage
Ryan Island 2009	Wapato Coverage
Whites Island 2010	All Fish Species Diversity
	Native Fish Species Diversity
	Non-native Fish Species Diversity
	Emergent Prey Species Diversity
	Open Water Prey Species Diversity
	Diptera Density in May Emergent Vegetation Tows
	Abundance of All May Prey
	Elevation

Table 9. Site data and variables used to assess influence of environmental variables on Unmarked Chinook Lipid Content.

Sites Used	Variables Used
Campbell Slough 2008	All Vegetation Diversity
Campbell Slough 2009	Native Vegetation Diversity
Franz Lake 2009	Non-native Vegetation Diversity
Lord-Walker Island 2009	Reed Canarygrass Coverage
Pierce Island 2009	Common Spikerush Coverage
Ryan Island 2009	Wapato Coverage
Whites Island 2009	All Fish Species Diversity
	Native Fish Species Diversity
	Non-native Fish Species Diversity
	Emergent Prey Species Diversity
	Open Water Prey Species Diversity
	Diptera Density in May Emergent Vegetation Tows
	Abundance of All May Prey
	Elevation

2.7.2 Trend Sites

Chinook Salmon Abundance

The Chinook salmon abundance at sites with repeated sampling were examined for this analysis, using regression modeling, because they provide the opportunity for an analysis of temporal trends. Three sites, Campbell Slough, Franz Lake and Whites Island, had multiple years of sampling and were used in this analysis (Table 10).

Table 10. Site data and variables used to assess influence of environmental variables on Chinook Abundance at sites sampled in multiple years.

Sites Used	Variables Used
Campbell Slough 2008	All Vegetation Diversity
Campbell Slough 2009	Native Vegetation Diversity
Campbell Slough 2010	Non-native Vegetation Diversity
Franz Lake 2008	Reed Canarygrass Coverage
Franz Lake 2009	Common Spikerush Coverage
Whites Island 2009	Wapato Coverage
Whites Island 2010	All Vegetation Richness
	Native Vegetation Richness
	Non-native Vegetation Richness
	All Fish Species Diversity
	Native Fish Species Diversity
	Non-native Fish Species Diversity
	All Prey Species Richness
	Emergent Prey Species Diversity
	Open Water Prey Species Diversity
	Diptera Density in May Emergent Vegetation Tows
	Abundance of All May Prey
	River Kilometer
	Distance From Mainstem
	Elevation

3.0 Results

3.1 Status Sites

3.1.1 Vegetation

The results presented here describe patterns observed in habitat structure along the longitudinal gradient of the LCRE. The data are from the 39 sites listed in Table 2 (marsh study sites included in spatial patterns analysis), which were sampled under multiple research studies, including the EMP, as described in Section 1.2.2. Site codes used in the analysis are defined in Table 2.

Sediment Composition

TOC content of the sediment samples does not seem to vary longitudinally along the estuarine gradient; however, signatures of vegetation strata and habitat types are evident in some cases (Figure 2). Typically, TOC was higher in the marsh (MA or specific vegetation code) samples than in the channel (CH) samples, with the highest TOC in areas dominated by grass (reed canarygrass, *P. arundinacea* [PHAR]). One exception is the CH sample from Jackson Island, which may be due to high amounts of submerged aquatic vegetation (SAV) at this site. Also notable is that TOC levels were generally higher at historical sites than created sites.

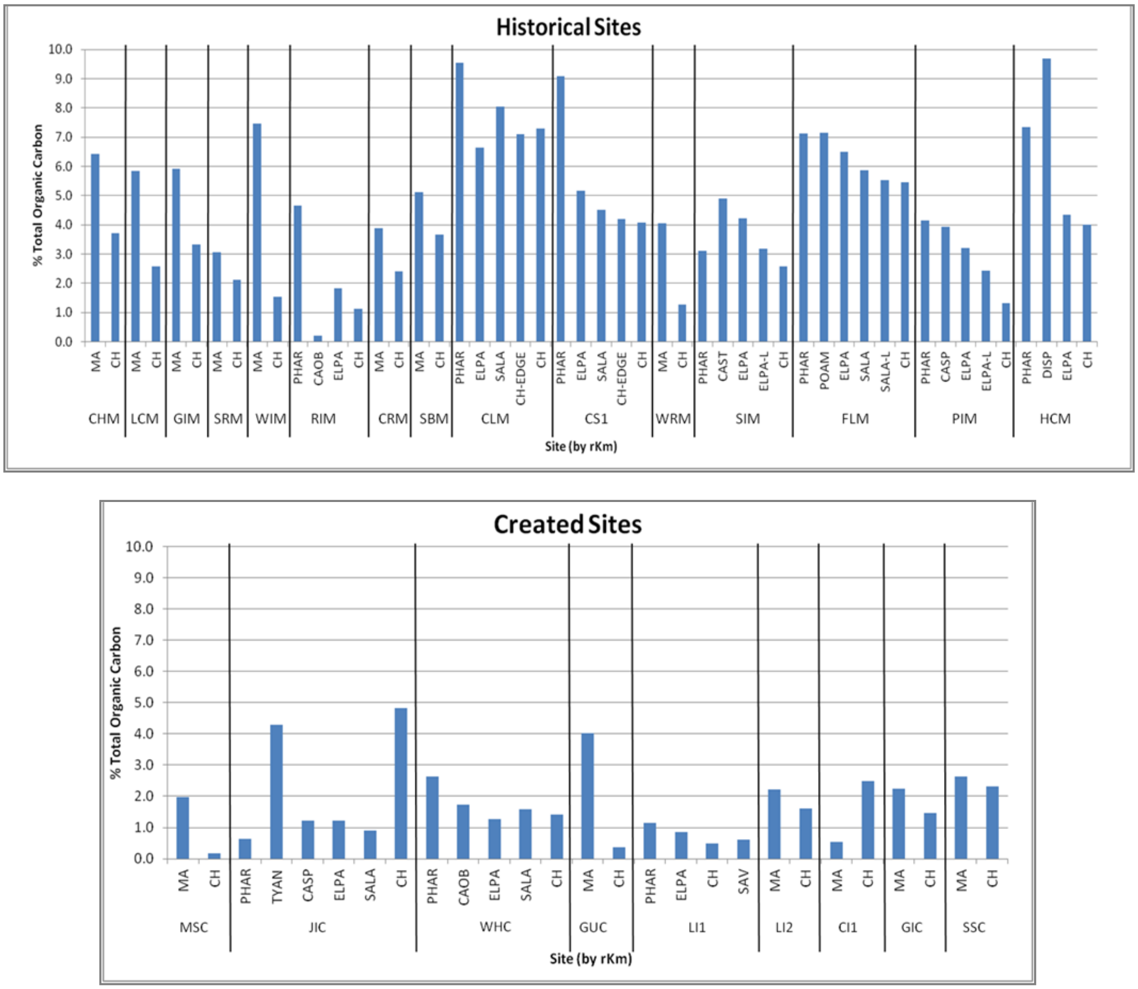
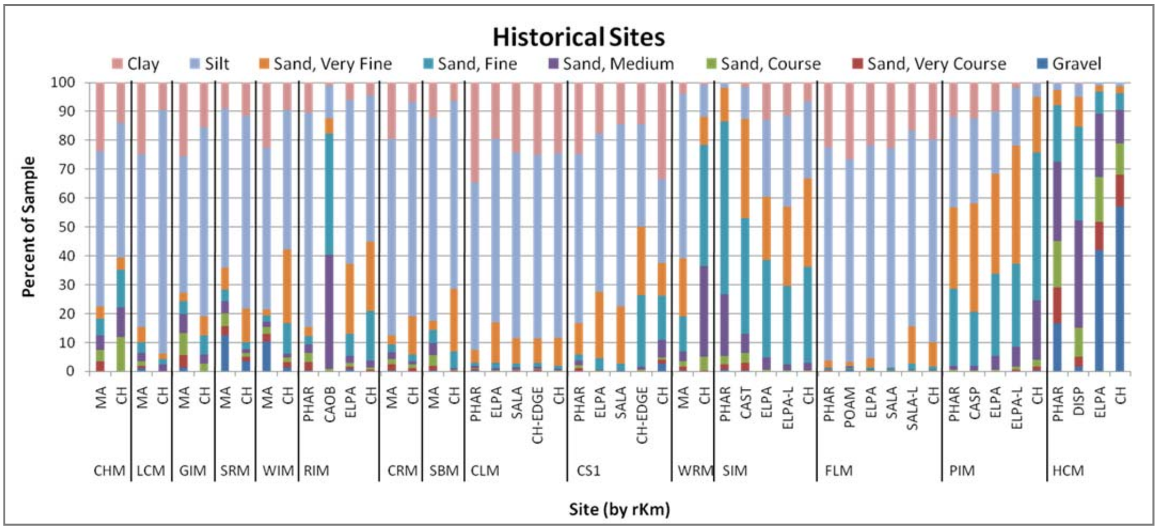


Figure 2. Percent TOC by strata at historical (top) and created (bottom) marsh sites. Sites are ordered by distance from the LCRE mouth, starting at left.

The sediment grain size at the marsh sites analyzed in this study is predominantly silt, very fine, and fine sand (Figure 3). This pattern was true for both historical and created marsh sites, with a few exceptions. Three sites had more sand than silt (Sand Island, Pierce Island, and Lord-Walker Island) and one site

(Hardy Creek) was dominated by coarse sand and gravel. Historical sites had a greater percentage of clay (10%–20%) than the created sites (<10%). Differences between strata within sites were generally not distinctive, except for a few cases where the CH had more coarse sediment than the MA and one case where slough sedge (*C. obnupta*) had more coarse sediments in an area near the outer boundary of the site.



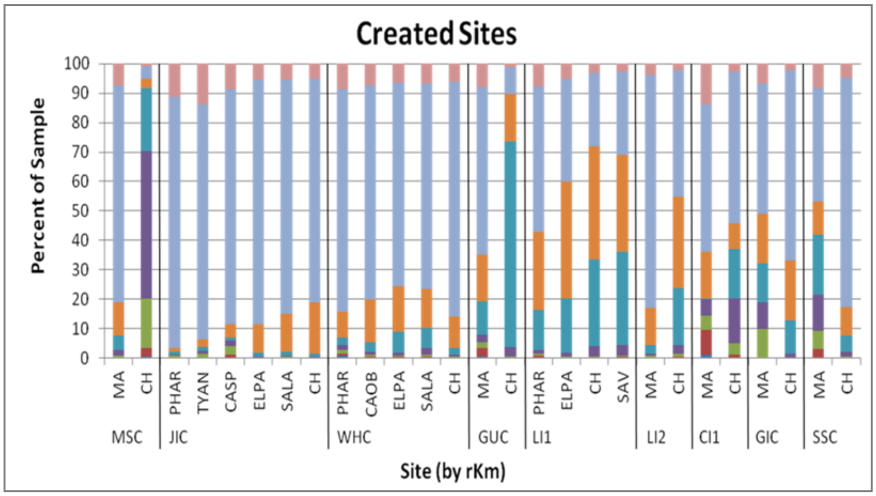


Figure 3. Sediment grain size composition of historical and created marshes, distinguished by strata within the sites. Samples were either taken within the MA and CH strata or separated further into vegetation strata.

Accretion Rates

Annual accretion rates for 22 marshes are shown in Figure 4. Generally, sediment either accreted (positive values) or eroded (negative values) at rates between 2.0 and −2.0 centimeter (cm)/year. The average accretion rates for the created marshes is 0.78 cm/year, and the rate for historical marshes is −0.44 cm/year, including the −7.8 cm/year extreme measured at Sand Island marsh. Without Sand Island marsh, the rate for historical marshes is 0.22 cm/year. In either case, the difference between rates at historical and created sites is not statistically significant.

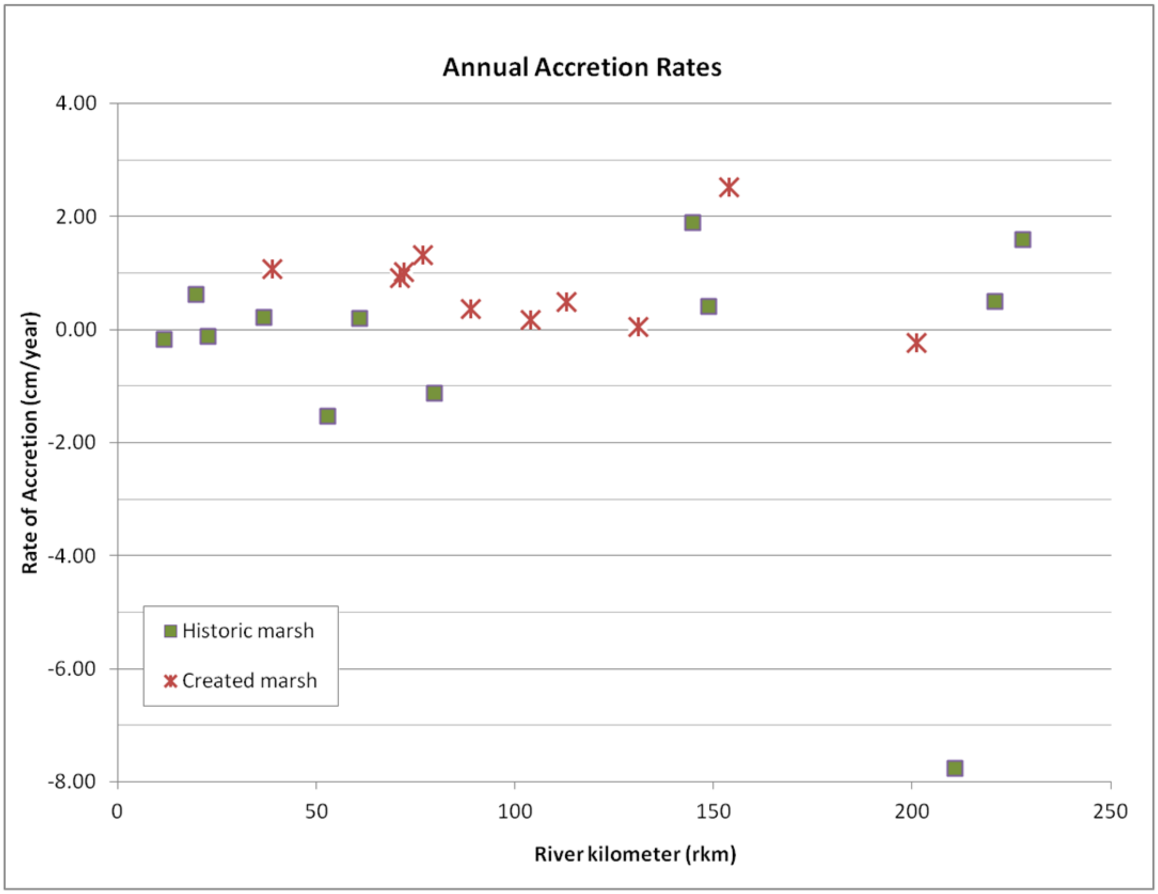


Figure 4. Annual sediment accretion and erosion rates for marshes along the river gradient.

Elevations of emergent marshes in the LCRE fall within a very narrow range (Figure 5). We convert elevations from the North American Vertical Datum of 1988 (NAVD88), the datum in which they are surveyed, to the Columbia River Datum (CRD) in order to make comparisons across the study area. The latter is low water datum developed in the Columbia River as opposed to NAVD88, which is a datum based on a single point at mean sea level in Father Point/Rimouski, Quebec, Canada. The difference in site elevations between datums is apparent in Figure 5. In this study, the average elevations of the sites were generally between 1.0 m and 2.0 m elevation, relative to CRD, with no significant trend relative to rkm. Marsh elevations near the mouth of the Columbia River (0-3- rkm) are slightly higher in elevation likely because the daily low tides limit the lower range of the marsh elevations in this zone. The site with the highest elevation at rkm 230 is Hardy Creek; a limited-connectivity site where the hydrology is driven by flooding from the CR and by the perennial creek flowing through the site and is therefore different than the sites that are driven solely by the hydrologic fluctuations of the CR.

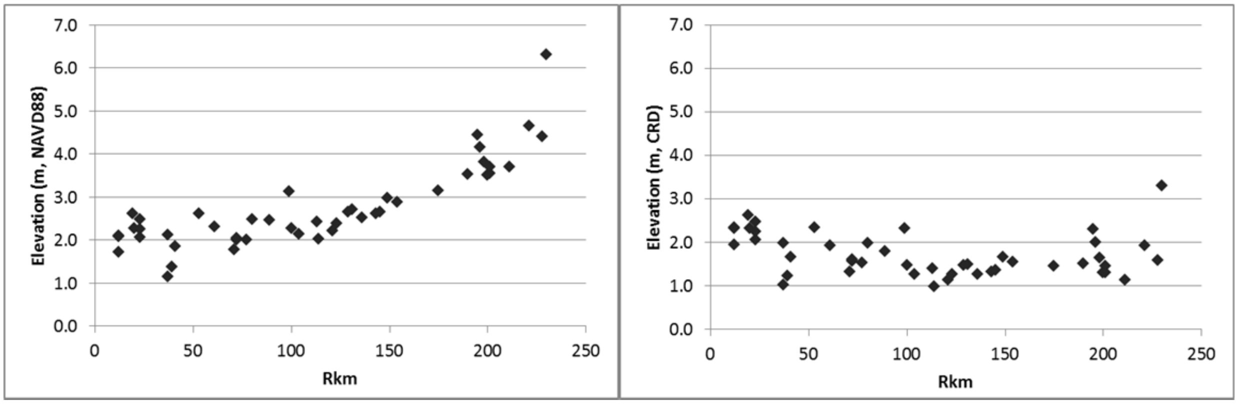
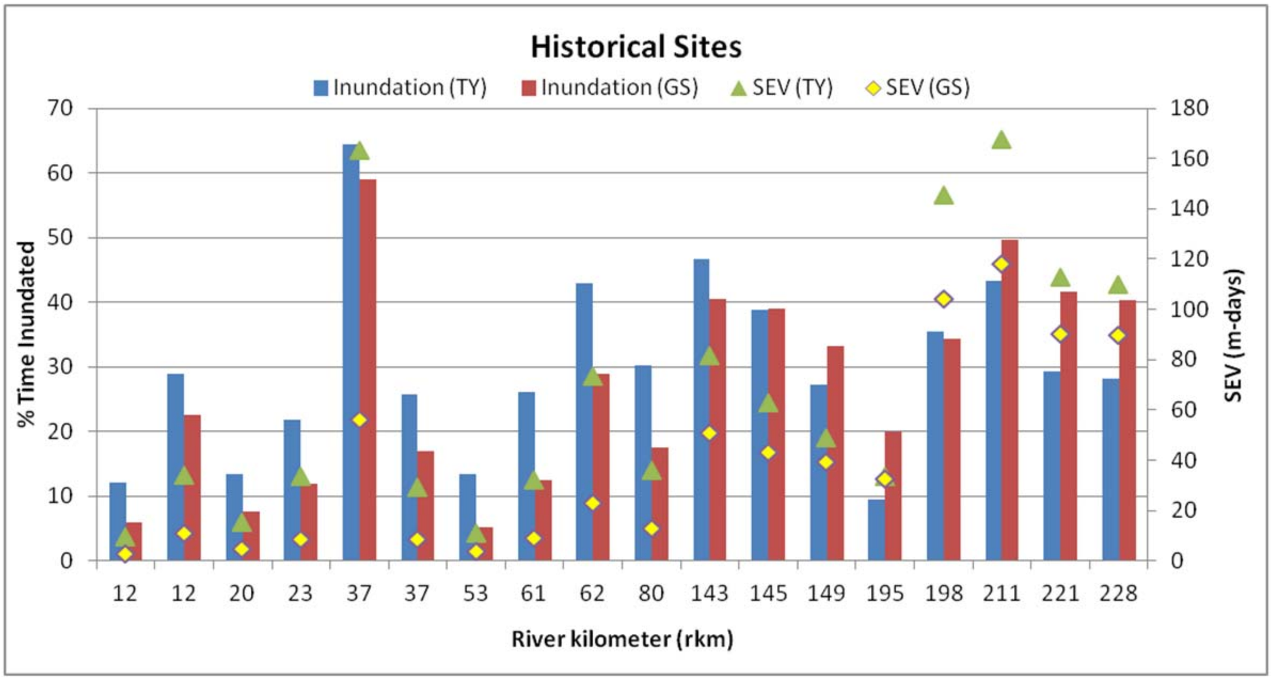


Figure 5. Average elevations of emergent marsh sites evaluated in this study. Elevations are relative to NAVD88 (left) and CRD (right).

Hydrology

Using the hydrology data and elevation data collected at the sites, inundation times and SEVs were calculated for average marsh elevations. These were separated into historical and created sites and graphed along the estuarine gradient (Figure 6; the site names and rkm can be determined from Table 2). Most of the data are from 2009 and 2010, years that were neither anomalously high nor low. The most notable trend in these data is the general increase in inundation time and magnitude in the upper portions of the river. Exceptions include a very low elevation site at rkm 37 (the low marsh portion of the Secret River marsh) and higher elevation sites at rkm 99 and 195 (Lord-Walker Island and Washougal River marsh, respectively). Also notable is the switch from a higher percent inundation in the total year (TY) below approximately rkm 130 to 145 to a higher percent inundation during the growing season. Also, above rkm 190, the SEV, an indication of the magnitude of inundation, was considerably higher than the percent of time inundated. The SEV for the growing season graphed against elevation shows little relationship when the entire estuary is graphed together (Figure 7); however, when the estuary is divided into hydrologic zones the relationship is more apparent (Figure 8). The zones used for this analysis are based on Jay et al. (in review) and Borde et al. (2012).



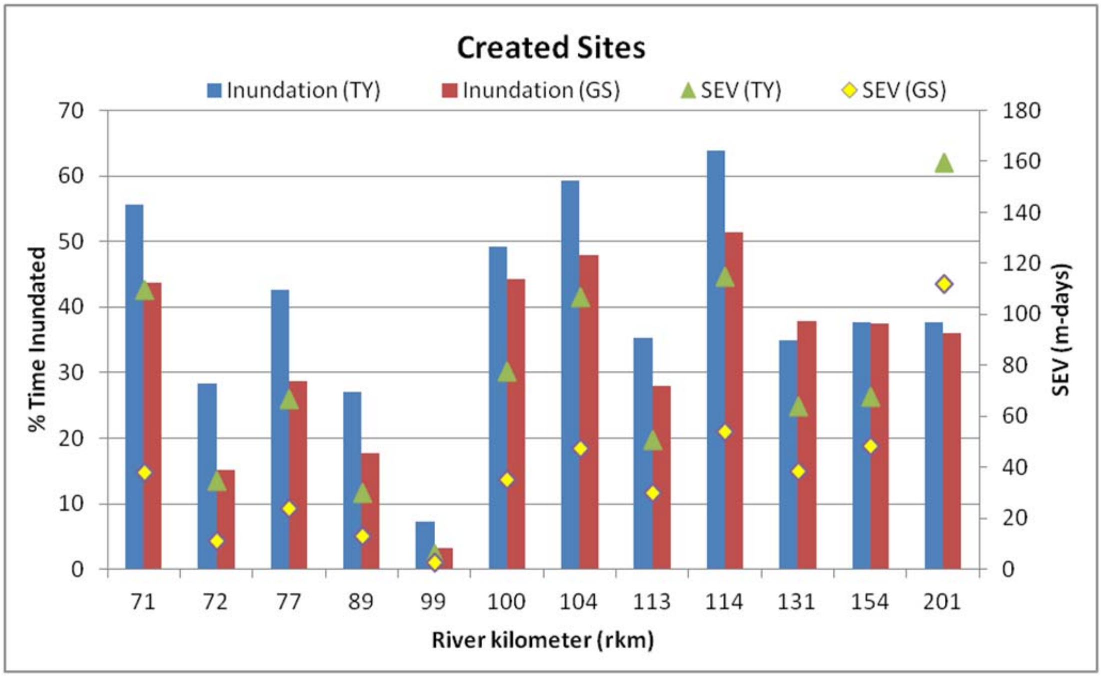


Figure 6. Inundation frequency and SEV of the average marsh elevation during the TY and the growing season (GS) at historical sites (top) and created sites (bottom).

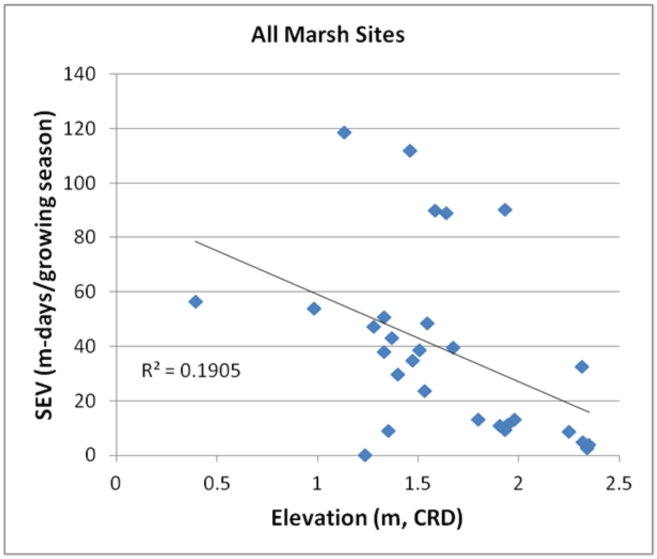


Figure 7. SEV related to elevation for all created and historical marsh sites in the analysis.

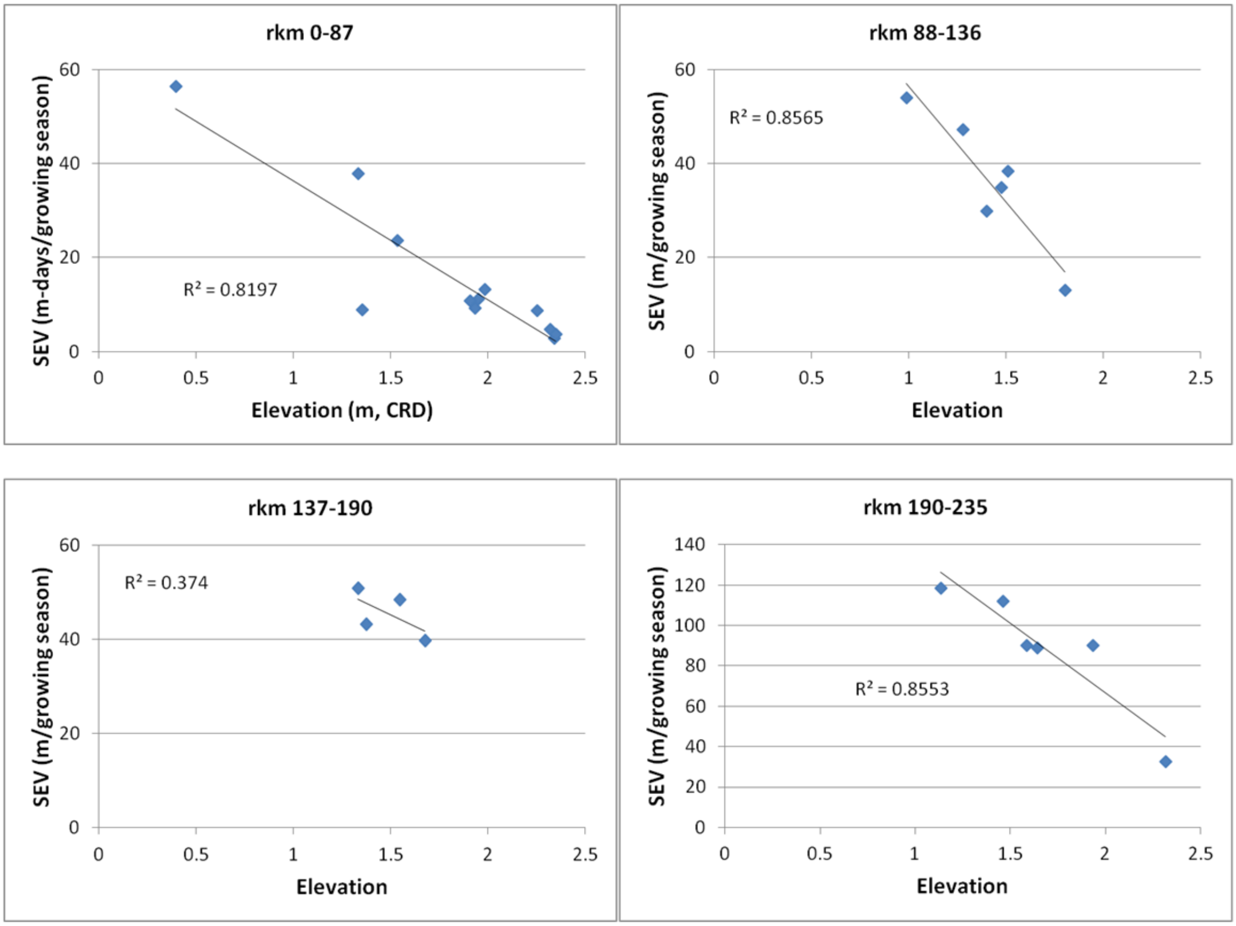


Figure 8. SEV for the growing season related to elevation for created and historical marsh sites within hydrologic zones of the Columbia River estuary.

Vegetation Assemblage Structure

This analysis focused on broad patterns of vegetation distribution along the estuarine gradient and the relationship of those patterns to elevation. We focus much of our analysis on reed canarygrass, a highly invasive species, to increase understanding regarding the spatial extent and prevalence of the species and to provide information that may ultimately help improve management of the species. In this analysis, 172 taxa were observed: 115 in created marshes and 139 in historic marshes. Seven taxa made up 68% of the cumulative cover as shown in Table 11. Reed canarygrass occurred in 52% of the quadrats and accounted for 28% of the cover at all emergent marsh sites.

Table 11. Dominant taxa in terms of percent cover in 37 historical and created marsh sites in the LCRE.

Scientific Name	Common Name	Species Code	Percent Cover	Cumulative Cover
Phalaris arundinacea	Reed canarygrass	PHAR	28%	28%
Eleocharis palustris	Common spikerush	ELPA	21%	49%
Sagittaria latifolia	Wapato	SALA	10%	59%
Carex lyngbyei	Lyngby sedge	CALY	3%	62%
Elodea canadensis	Canada waterweed	ELCA	2%	64%
Ludwigia palustris	False loosestrife	LUPA	2%	66%
Carex obnupta	Slough sedge	CAOB	2%	68%

Along the estuarine gradient from the mouth to the dam, the number of species present at a site is generally greatest in the lower-middle portion of the LCRE (rkm 53–89; Figure 9). Likewise, the greatest number and percent cover of non-native species were generally found in this portion of the estuary, although percent cover was high up to rkm 154. High non-native cover is primarily reed canarygrass. Lower species diversity at the lower and upper reaches of the LCRE is likely due to the effects of higher physical disturbance at these extremes. In the lower portions, a limited number of species can tolerate the hydrologic patterns resulting from daily tidal fluctuations. Likewise, at the upper end of the estuary, high fluvial dominance results in high inundation for part of the growing season, limiting the species that can tolerate this extreme condition.

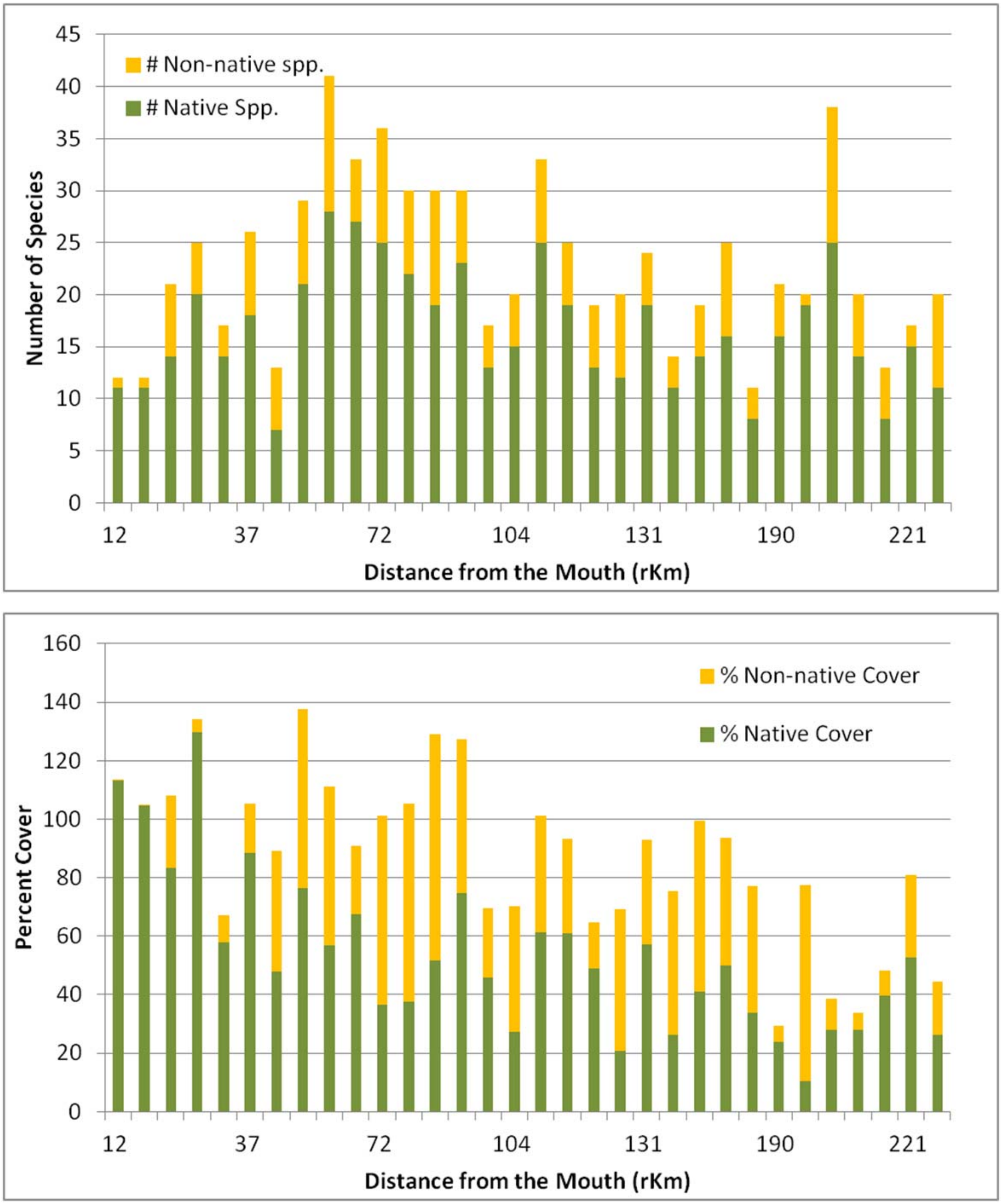


Figure 9. Number of species at each site (top) and average percent cover (bottom) of native and non-native species at all marsh sites in the study area.

Vegetation Elevation Interactions

The elevations of emergent marshes in the LCRE cover a narrow range. The sample areas at our sites were generally representative of this range, with 90% of the quadrats between 0.8 m and 2.6 m at historical sites and between 0.8 m and 2.0 m at created sites (Figure 10). Within this elevation range, the number of wetland species varies, with the greatest number observed between 1.5 m and 2.5 m (Figure 11). The elevation of the maximum number of species was slightly higher in historical sites than in created sites.

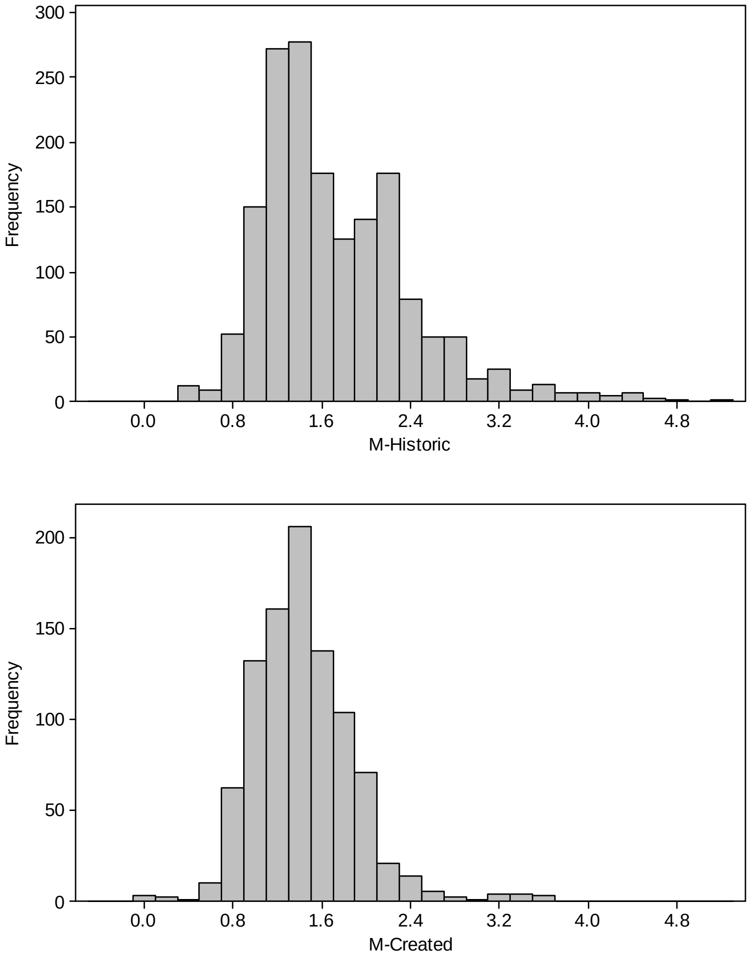


Figure 10. Frequency of vegetation quadrats at elevations (m, CRD) found in historical and created marshes in this analysis.

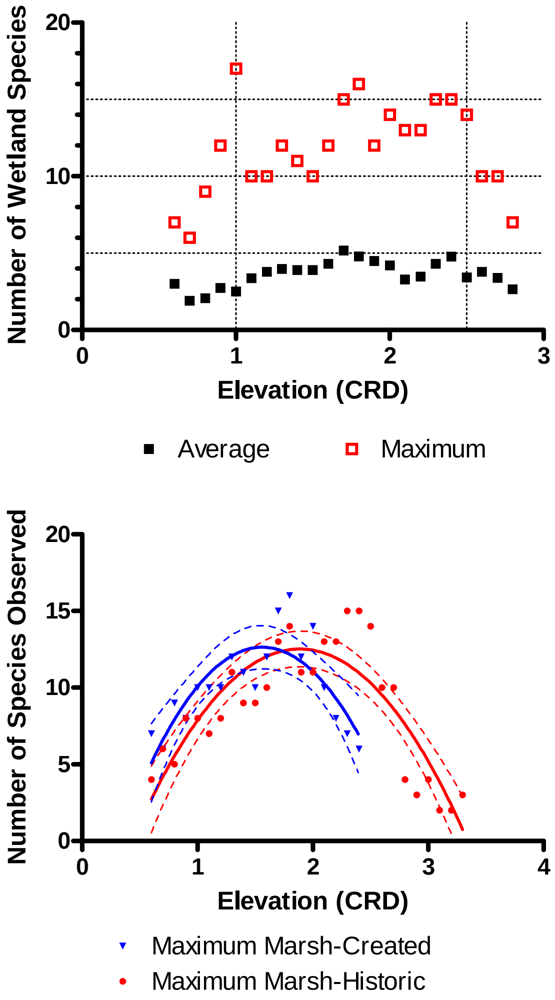


Figure 11. Average and maximum number of species observed at all marsh sites in the study area (top) and maximum number of species at historical and created marsh sites (bottom) by elevation (m, CRD).

The distribution of the common emergent marsh species within the elevation gradient is shown in Figure 12. The three most common species, reed canarygrass (*P. arundinacea*), common spikerush (*E. palustris*), and Wapato (*S. latifolia*), cover distinct elevation ranges. The minimum elevation for reed canarygrass was evaluated more closely in an effort to identify areas where other species may be able to out-compete this invasive species. Figure 13 indicates that in historical sites the lowest elevation where reed canarygrass has the maximum cover in a quadrat is approximately 1.6 m. At created sites, this elevation varies more along the estuarine gradient, from approximately 1.4 to 1.8 m.

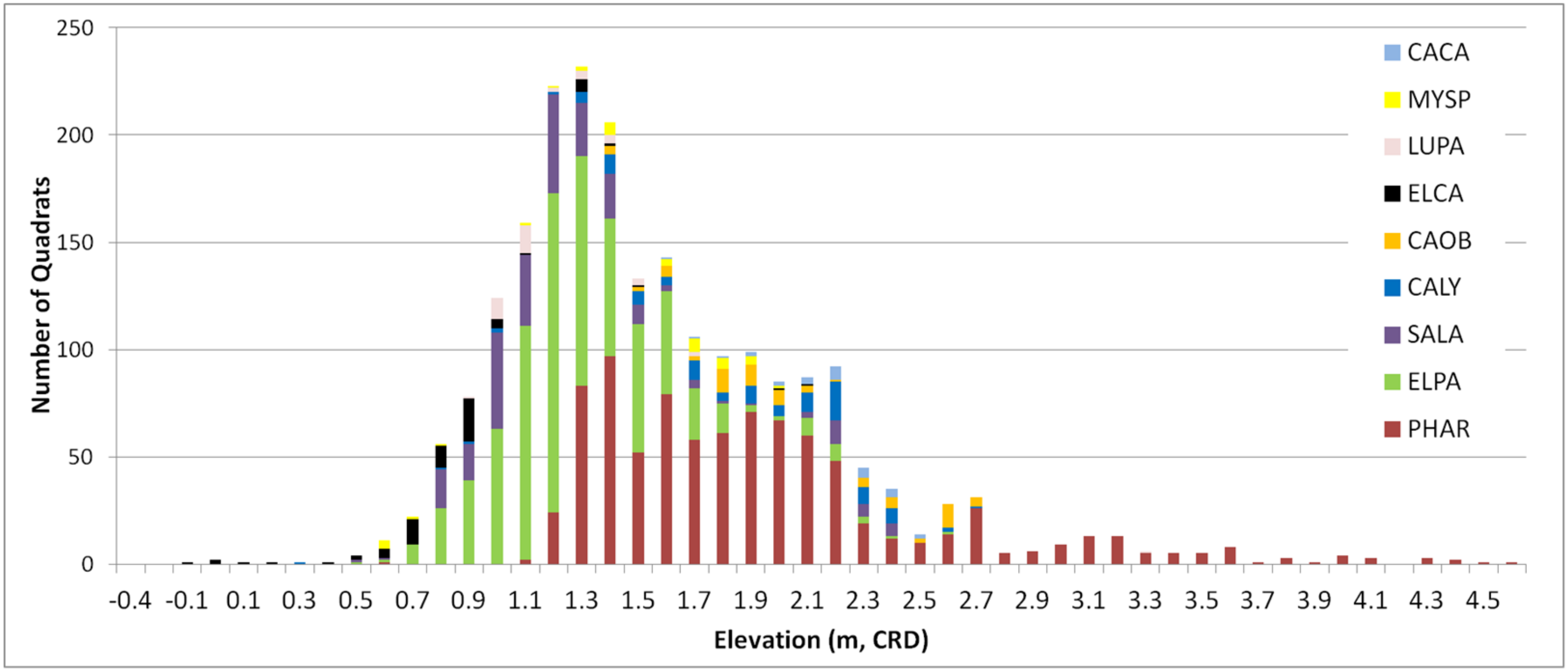


Figure 12. Number of quadrats at each elevation where the common marsh species were observed to have greater than 20% cover (elevation in m relative to CRD).

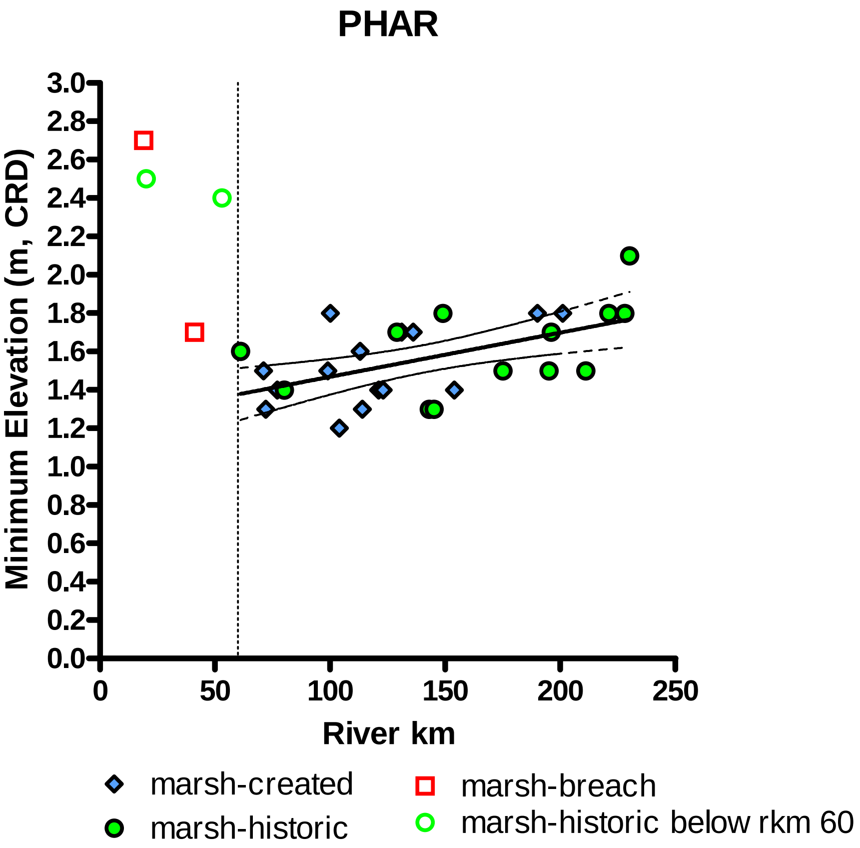


Figure 13. Minimum elevation (m, CRD) of reed canarygrass at sites along the longitudinal gradient of the river. Regression line and 95% confidence limits (dashed lines) are shown for created and historical marshes above rkm 60.

Elevation, Inundation, and Vegetation Interactions

Hydrology is one of the primary factors controlling the presence and distribution of wetland vegetation (Mitsch and Gosslink 2000). The variation in inundation patterns observed throughout the LCRE is directly related to patterns in total wetland vegetation cover (Figure 14). The SEV during the growing season varies from low values in the tidally dominated portion of the estuary to high values in the fluvial-dominated portion. During the time at which we sample (late July to early August), the vegetation cover is high in the tidal portion and low in the fluvial portion. This relationship is caused by the high inundation from the spring freshet stressing the plants and reducing their productivity. The timing, duration, and magnitude of the freshet likely determine the extent of the stress and the timing of recovery. Perhaps the plants in the fluvial-dominated regions have higher cover later in the growing season but whether the cover is comparable to that seen in the more tidally dominated areas has not been determined due to the limit of our sampling period (late July to early August).

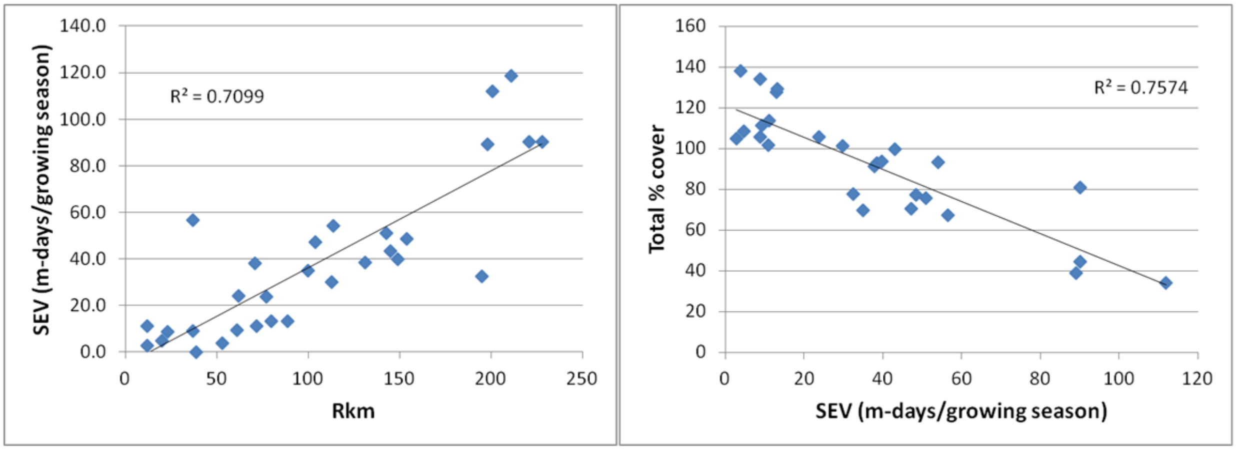


Figure 14. SEV as related to longitudinal gradient of the river (left panel) and total percent vegetation cover related to SEV (right panel).

Channel Morphology and Inundation

Channel morphology and inundation are important factors in the evaluation of potential for fish to access tidal wetlands areas for feeding, rearing, and cover. We have seen that inundation patterns vary along the estuarine gradient (Figure 14); likewise, the channel morphology changes as well. Figure 15 shows a general change in average channel depth, with deeper channels in the lower estuary becoming shallower in the upper estuary. Exceptions to this trend are found at the created sites, where most of the created sites tend to be shallower than the historical sites. However, two of the created sites (Lord-Walker Island and Goat Island Slough) have some of the deepest channels. These conditions can be explained by considering the history of the sites. The generally shallow nature of the channels at created sites could be due to the limited time the sites have been exposed to the channel-forming processes compared to the historically present sites. Alternatively, lower bank elevations of the created sites could also explain the shallower depth. At the created sites with greater depth, the explanation is perhaps that the method of dredge material placement produced higher channel banks.

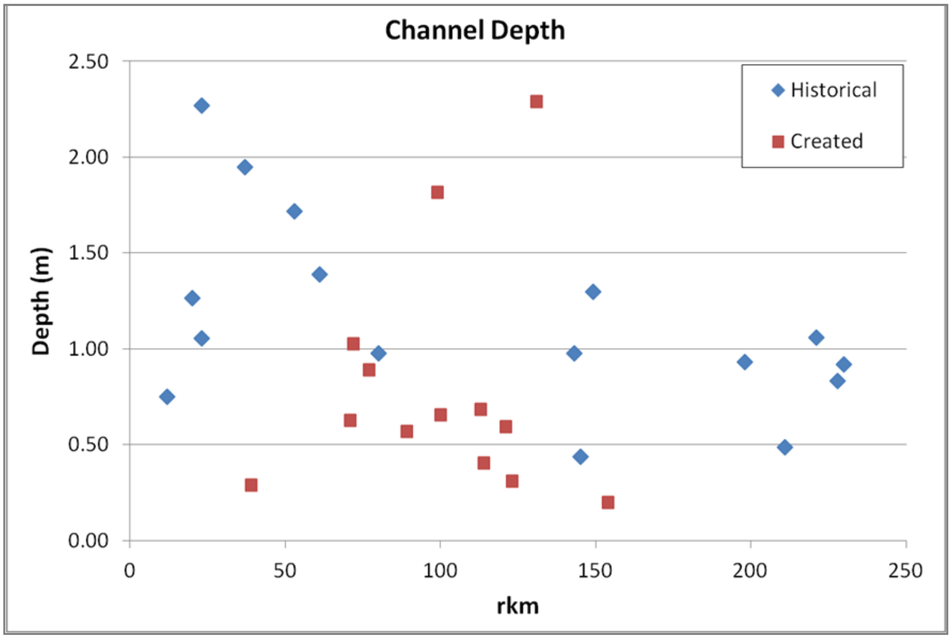


Figure 15. Average channel depth for the primary marsh channel at historical and created sites included in this analysis.

Frequency of inundation at the channel mouth is an indication of the opportunity for juvenile fish access to the tidal channel and the marsh. The frequency was calculated using the thalweg elevation with 50 cm of water as an estimate of the amount of water needed for fish to enter the channel. In general, the frequency of inundation provides access to the channel between 50% and 80% of the time in most areas of the LCRE (Figure 16). In the lower, tidally dominated part of the estuary, this frequency is consistent during the peak salmon migration period (FW) and the TY. In contrast, in the upper estuary the amount of time the channel is accessible decreases during the TY due to the low water period that occurs in the fall. A few sites are inundated greater than 90% of the time, caused either by deep channels at the mouth (Welch Island, Lord-Walker Island, Cottonwood Island- large slough) or the presence of perennial streams (Scappoose Bay-mouth of McNulty Creek, Hardy Creek). Cunningham Lake is not included because no data were collected at the channel mouth for that site, and Franz Lake is not included because beaver dams affected the water level at the sensor so the data were not representative of conditions at the mouth. In the latter case, a sensor would need to be placed below the beaver dam to accurately determine the inundation frequency at this location.

Frequency of inundation at the channel bank is a useful metric for evaluating the amount of time for which juvenile salmon would have access to the marsh interface for feeding and cover. In the lower estuary (below rkm 60), the channel bank (with 10 cm of water) was accessible 20% to 30% of the time during the TY and the FW periods. Above Ryan Island (61 rkm) the frequency increased to about 30% to 60% of the time. Inundation frequencies in the middle reaches of the estuary (~rkm 61–140) were calculated primarily at the created sites, which had greater inundation frequencies in part due to lower bank elevations. The greater inundation above Scappoose Bay-mouth of McNulty Creek (rkm 143) during the FW was primarily due to the influence of higher water levels during the spring freshet. The lowest inundation frequencies were observed at Goat Island Slough and Hardy Creek, both of which have high channel bank elevations. Greatest overall frequency of inundation during the FW period was at three created sites (i.e., Cottonwood Island-large slough, Cottonwood Island-small slough, and Sauvie Island-east slough) providing frequent access to the channel (100%, 79%, and 78%, respectively) and to the bank (71%, 86%, and 68%, respectively).

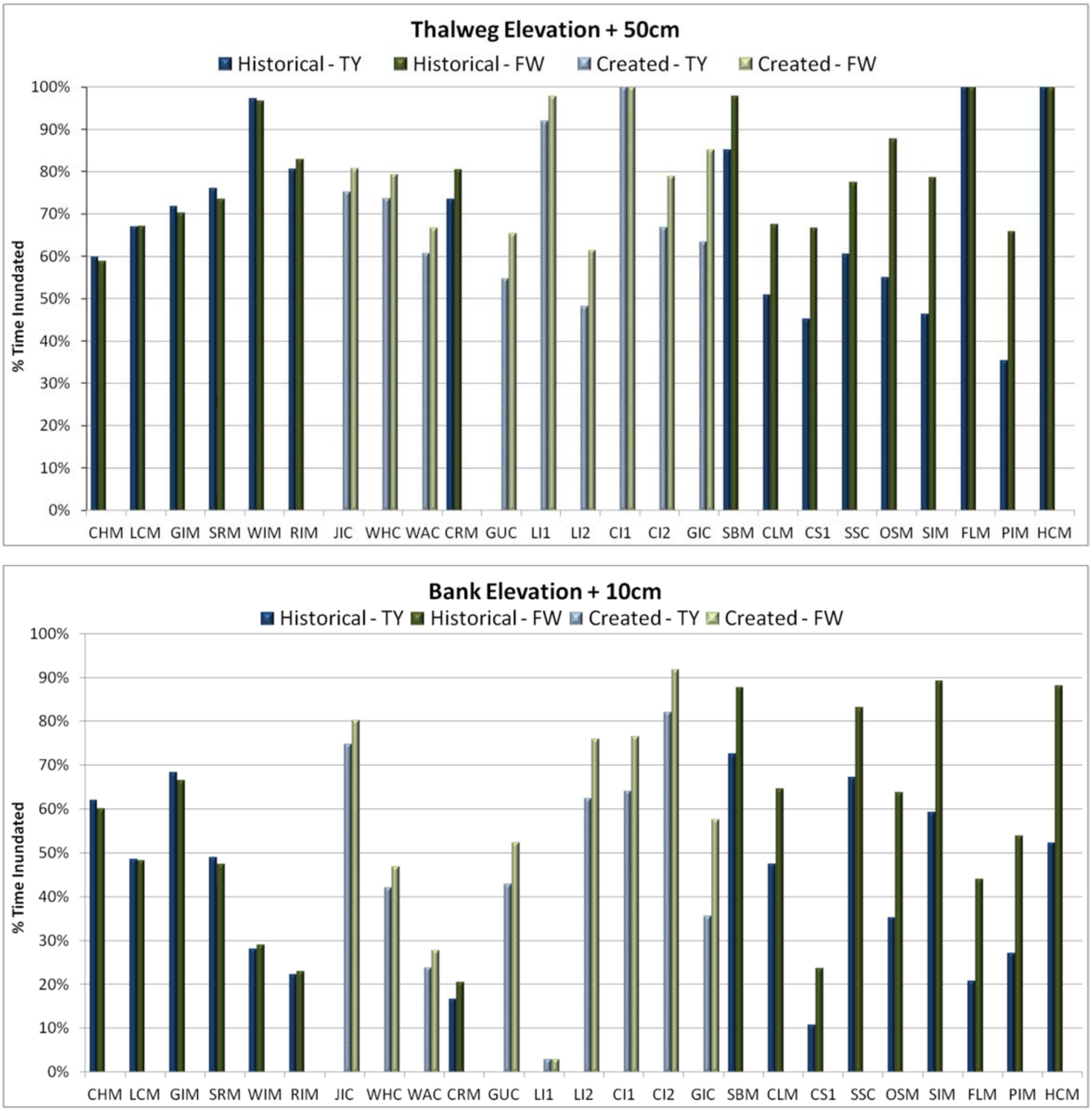


Figure 16. Inundation frequency at the channel mouth cross-section for the thalweg elevation (+50 cm) and for the average channel bank elevation (+10 cm) during TY and FW.

3.1.2 Fish

Water Temperature

Water temperatures followed a similar pattern at all sites, increasing from a low of 5°C to 10°C in April to a maximum of 20°C to 25°C in July and August (Figure 17). Two-way ANOVA including Site and Month ($R^2 = 0.7964$ and $p<0.0001$) indicated that both had significant effects on water temperature. Month of sampling had the greatest influence on water temperature, explaining 69% of its variation (ANOVA, $p<0.0001$). Among the sites, water temperatures tended to be cooler than the overall average at Bradwood Slough and Jackson Island in Reach C, and Pierce Island and Hardy Slough in Reach H, while warmer temperatures were observed at Franz Lake, Campbell Slough, Ryan Island, and Sandy Island ($0.0001<p<0.0189$).

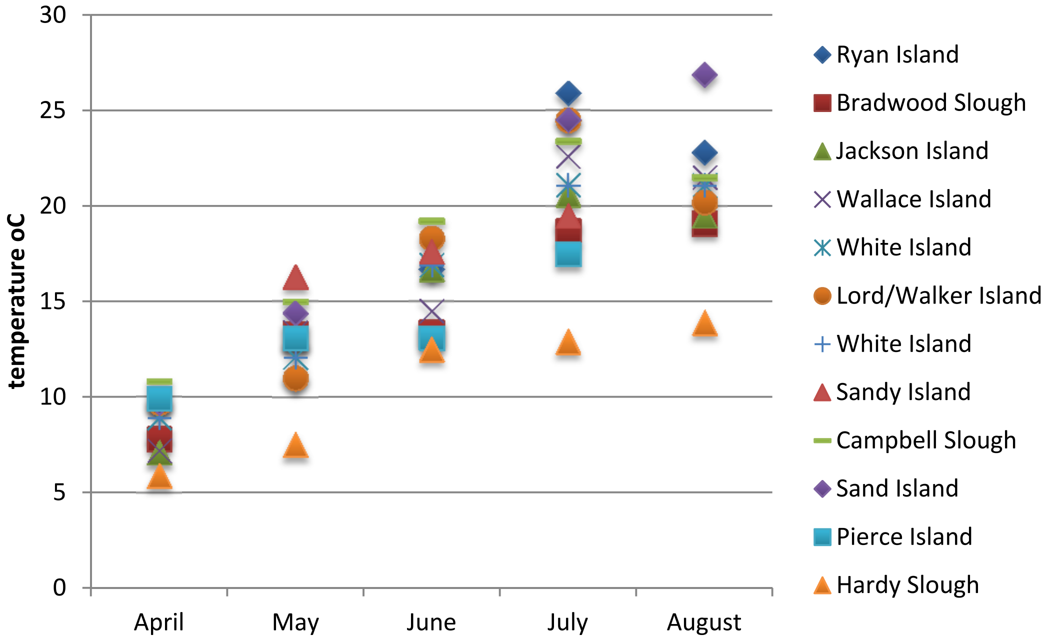


Figure 17. Average water temperature by month at the EMP sampling sites, collected at time of fish sampling.

Fish Habitat Occurrence

To assess spatial patterns in fish habitat occurrence in various reaches of the estuary, a total of 12 sites were sampled between 2007 and 2010—six in Reach C, one in Reach E, one in Reach F, and four in Reach H (Table 12; Figure 1).

Table 12. Site coordinates, rkm, distance from mainstem, and years sampled for 2007–2010 EMP fish sampling sites. Rkm 0 is the mouth of the Columbia River.

Site Name	Reach	Latitude	Longitude	River Kilometer	Distance From Mainstem (m)	Years Sampled
Ryan Island	C	46.206600	123.414817	61	0	2009
Bradwood	C	46.203183	123.447733	62	0	2010

Table 12. Site coordinates, rkm, distance from mainstem, and years sampled for 2007–2010 EMP fish sampling sites. Rkm 0 is the mouth of the Columbia River.

Site Name	Reach	Latitude	Longitude	River Kilometer	Distance From Mainstem (m)	Years Sampled
Slough						
Jackson Island	C	46.169417	123.350600	71	532	2010
Whites Island	C	46.159350	123.340133	72	742	2009, 2010
Wallace Island West	C	46.140467	123.283100	77	248	2009
Lord/Walker Island	C	46.137216	123.040278	99	230	2009
Sandy Island	E	46.015000	122.868333	121	0	2007
Campbell Slough	F	45.783867	122.754850	149	2109	2007, 2008, 2009, 2010
Sand Island	H	45.553350	122.211117	211	0	2008
Franz Lake	H	45.600583°	122.103067	221	305	2008, 2009
Pierce Island	H	45.620967	122.010800	228	0	2008
Hardy Slough	H	45.628217	122.012150	230	1725	2008

Fish Community Characteristics

The EMP sampling sites supported a diverse range of fish species. Fish community composition tended to vary from reach to reach. At the Reach C sites (Ryan Island, Bradwood Slough, Jackson Island, Wallace Island, and Lord/Walker Island), three-spine stickleback (*Gasterosteus aculeatus*) were by far the most abundant species, making up 81% to 96% of the total catch. Salmonid species accounted for 1% to 5% of the catch. Other species observed include banded killifish (*Fundulus diaphanous*), chiselmouth (*Acrocheilus alutaceus)*, and peamouth (*Mylocheilus caurinus*) (Figure 18). At Sandy Island in Reach E, three-spine stickleback were still a dominant species, making up 65% of the total catch, but other species were found in greater abundance. More common species in this reach included banded killifish peamouth, and largescale sucker (*Catostomus macrocheilus*). Salmonid species made up about 9% of the total catch. At Campbell Slough in Reach F, three-spine stickleback made up 34% and carp (*Cyprinidae* sp.) made up 37% of the total catch. Banded killifish and yellow perch (*Perca flavescens*) were also relatively common. Salmonids made up 4% of the catch at Campbell Slough in Reach F.

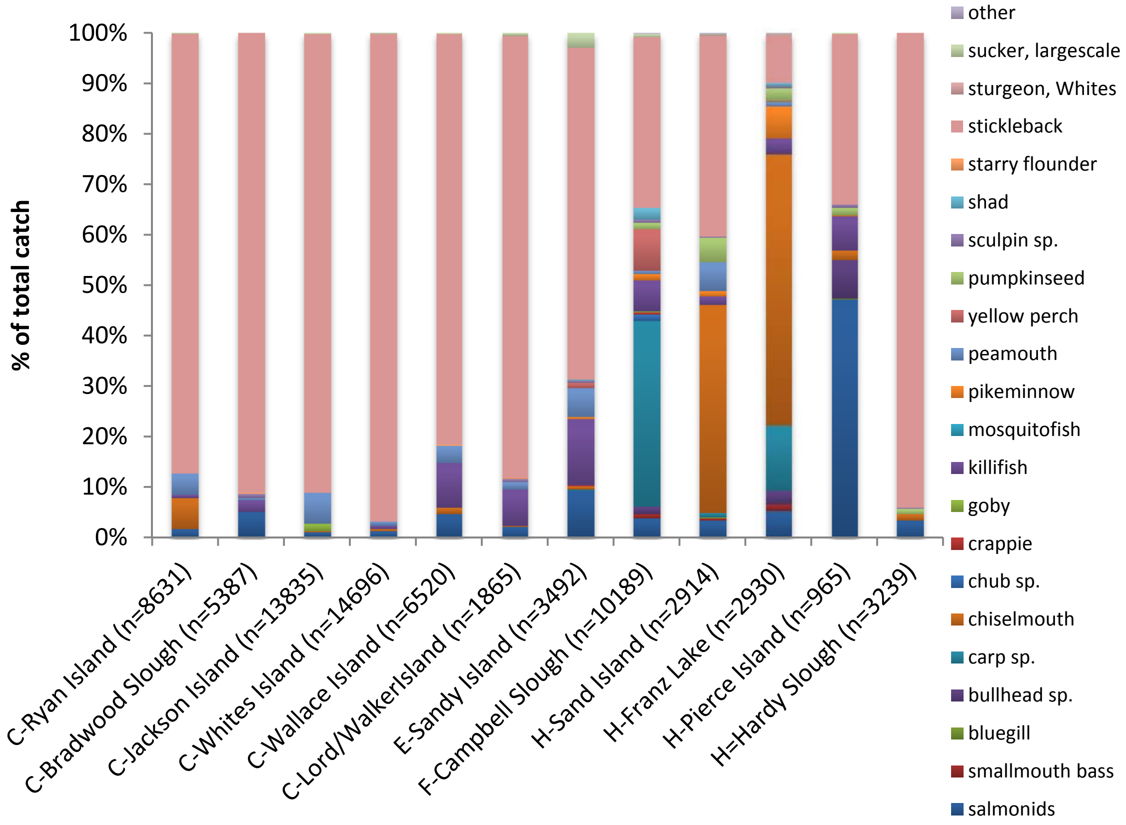


Figure 18. Species composition at EMP sites sampled between 2007 and 2010. Figure represents combined values for all years sampled.

Among the Reach H sites (Sand Island, Franz Lake, Pierce Island, and Hardy Slough), species composition was variable. Chiselmouth were dominant at Sand Island and Franz Lake, accounting for 41% and 53% of the catches at these sites, respectively. Salmonids were the most abundant species at Pierce Island, making up 47% of the total catch. At other sites in Reach H, salmonids accounted for 3% to 5% of the total catch. Hardy Slough was somewhat unusual among the Reach H sites in that it had a very high proportion (94%) of three-spine stickleback in the catch, similar to the sites in Reach C. At the other sites in Reach H, three-spine stickleback were generally less abundant, making up 10% to 40% of the catch. Estuarine species such as starry flounder (*Platichthys stellatus*) were found primarily at the Reach C sites. Several species of fish were found only above Reach C. These included smallmouth bass (*Micropterus dolomieu*), bullhead species (*Ameiurus* sp.), tui chub (*Gila bicolor*), crappie (*Pomoxis* sp.), yellow perch, and American shad.

Species diversity and species richness (Figure 19) both tended to be lower at the Reach C sites, with the total number of species collected ranging from seven to 13 species. At Sandy Island, in Reach E, species richness and diversity were slightly higher. The total number of species captured was highest at Campbell Slough, where 26 different species were observed. The number of species at the Reach H sites ranged from 10 to 22, with the lowest species richness at Pierce Island and Hardy Slough.

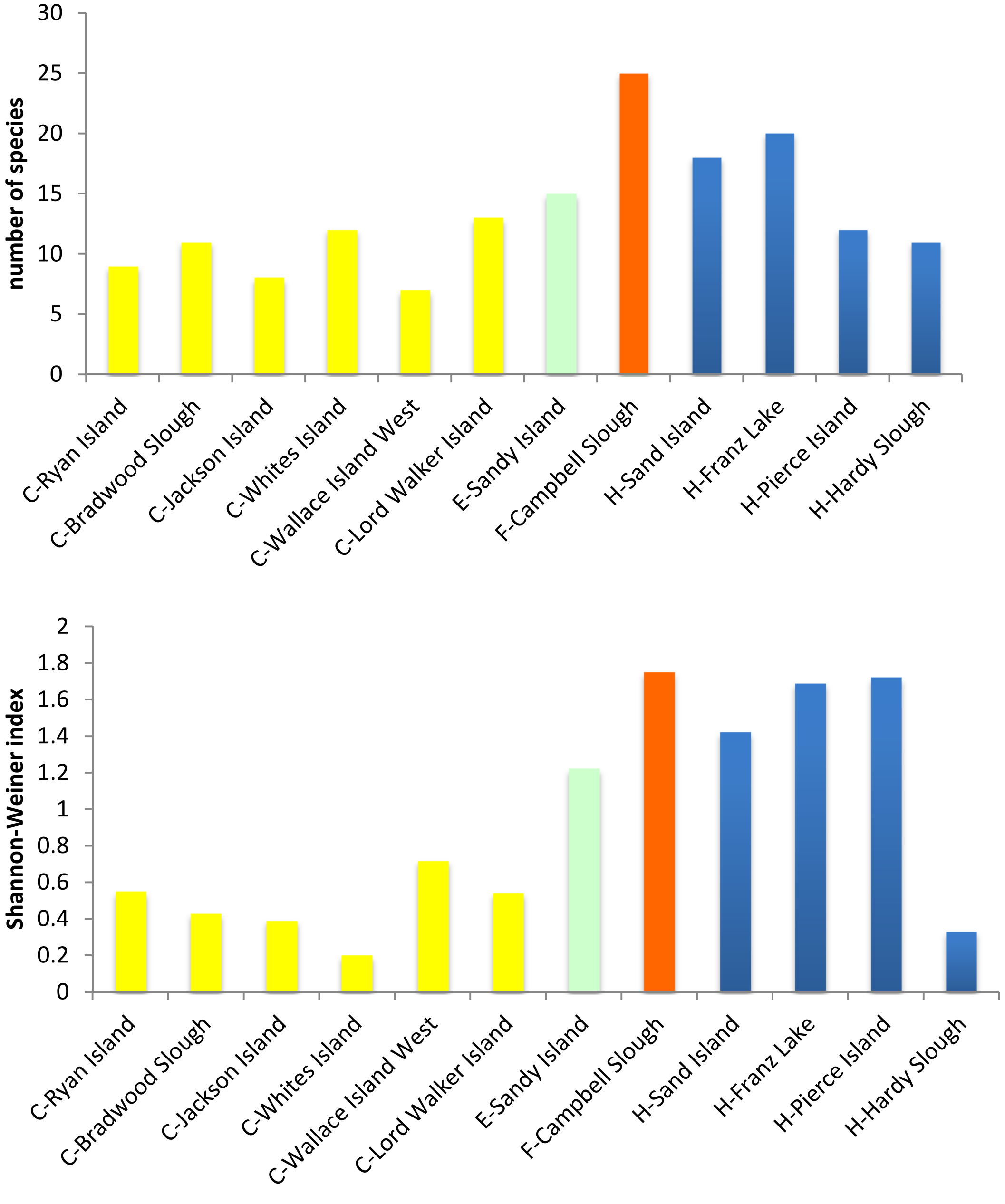


Figure 19. Total number of species and species diversity (calculated using the Shannon-Weiner Diversity Index) at EMP sites. Reach C sites are shown in yellow, Reach E in green, Reach F in orange, and Reach H in blue.

The percentage of non-native species in the catch (Figure 20a) was lowest in Reach C, ranging from 9% at Hardy Slough and Whites Island to 22% to 23% at Ryan and Lord-Walker Islands. It was somewhat higher at Sandy Island (40%) and reached a maximum at Campbell Slough, where the percentage of non-native species was 58%. In Reach H, the percentage of non-native species ranged from 44% at Sand Island to 20% at Hardy Slough, declining at sites farther upriver.

Piscivorous predators (e.g., smallmouth bass, largemouth bass [*M. salmoides*], and northern pikeminnow *Ptychocheilus oregonensis*]) were absent from the Reach C sites (Figure 20b). However, they were present at Sandy Island, Campbell Slough, and several Reach H sites, including Sand Island, Franz Lake, and Pierce Island. At these sites they made up from 0.2% to 7.5% of the total catch, and were most abundant at Franz Lake.

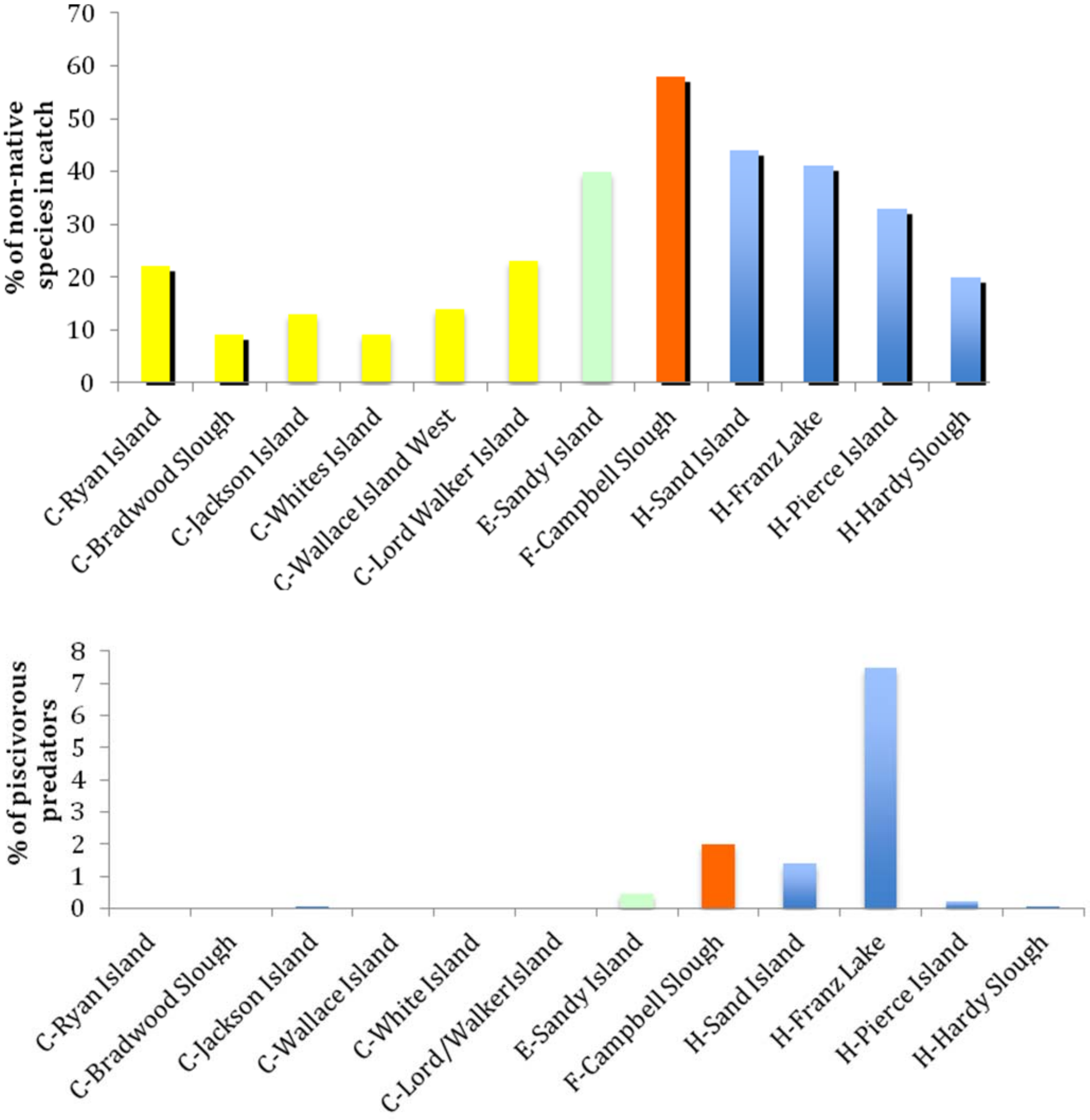


Figure 20. Percentages of a) non-native species and b) piscivorous predators in the catch at EMP sites. Reach C sites are shown in yellow, Reach E in green, Reach F in orange, and Reach H in blue.

Salmonid Species Composition

Juvenile salmon were feeding and rearing at all of the sampling sites from 2007 to 2010, with the total number of salmonids collected ranging from 88 fish at Wallace Island to 451 fish at Pierce Island (Figure 21). At the sites within Reach C ($88 \leq n \leq 273$), as well as at Sandy Island in Reach E ($n = 325$) and Campbell Slough in Reach F ($n = 393$), Chinook salmon were the dominant salmonid species present, accounting for 71% to 100% of the salmonid catch. At the Reach H sites, Chinook were still the most common species at Franz Lake (*n* = 158) and Pierce Island (*n* = 451), accounting for 51% and 71% of salmonid catches at these sites. However, at Sand Island (*n* = 99) and Hardy Slough (*n* = 112) they were less common, accounting for 16% to 21% of the salmonid catch. Coho salmon made up 18% of the salmonid catch at Bradwood Slough (*n* = 273), but otherwise were rarely found at the Reach C sites or at Sandy Island or Campbell Slough. They were found more frequently in Reach H, especially at Sand Island and Hardy Slough, where they accounted for 76% to 80% of the salmonid catch. At Franz Lake and Pierce Island they made up 43% and 21% of the salmonid catch. Chum salmon were found at sites in all reaches, typically accounting for 1% to 5% of the salmonid catch. They were especially abundant at Ryan Island, where they made up 29% of the salmonid catch. Chum were also found somewhat more frequently at Pierce Island and Lord/Walker Island (*n* = 144), where they made up about 8% of the salmonid catch. Steelhead trout were relatively rare, observed only at Bradwood Slough (0.4% of the salmonid catch) and Franz Lake (1.9% of the salmonid catch).

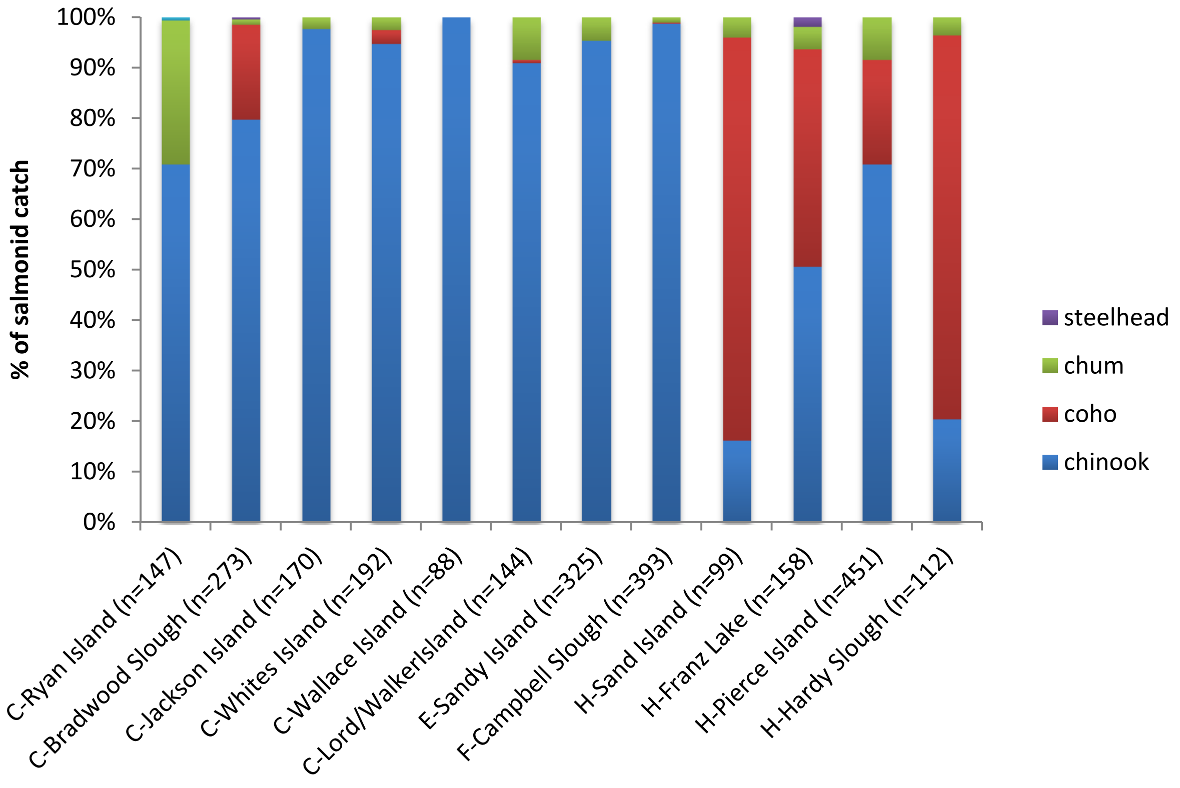


Figure 21. Proportions of salmonid species at EMP sites. The letter before site names indicates the hydrogeomorphic reach in which the site is located.

All chum and steelhead collected were unmarked fish. However, Chinook (Figure 22a) and coho catches (Figure 22b) included both marked fish and unmarked fish. At the Reach C sites, 82% to 97% of Chinook captured (82<*n*< 218) were unmarked. Unmarked fish also predominated at Pierce Island (*n* = 320) and Hardy Slough (*n* = 23) in Reach H, where they made up 87% and 74% of Chinook captured. However, at

Franz Lake (*n* = 80) and Sand Island (*n* = 16) in Reach H, only 25% to 30% of the Chinook captured were unmarked. At Sandy Island and at Campbell Slough in 2007, marking was consistently noted only in a subset of fish that were collected for necropsy, so the marked or unmarked status of Chinook salmon collected in that year (100% of Chinook salmon collected from Sandy Island and 42% of salmon collected from Campbell Slough) was designated as undetermined. Of the 233 Chinook salmon collected at Campbell Slough between 2008 and 2010, 39% were unmarked, and 61% were marked. Of the subset of Chinook salmon necropsied at Campbell Slough and Sandy Island in 2007 (not shown in Figure 22), 52% of Campbell Slough Chinook were unmarked and 48% were marked, while at Sandy Island 64% of were unmarked and 36% were marked. However, because there may have been some bias in selecting fish for necropsy, it is uncertain if these percentages would be representative of the juvenile Chinook population at those sites. Of the small number of coho salmon collected at sites below Reach H, all were unmarked. At the Reach H sites, the proportion of unmarked coho varied from 1% at Sand Island to 78% at Hardy Slough.

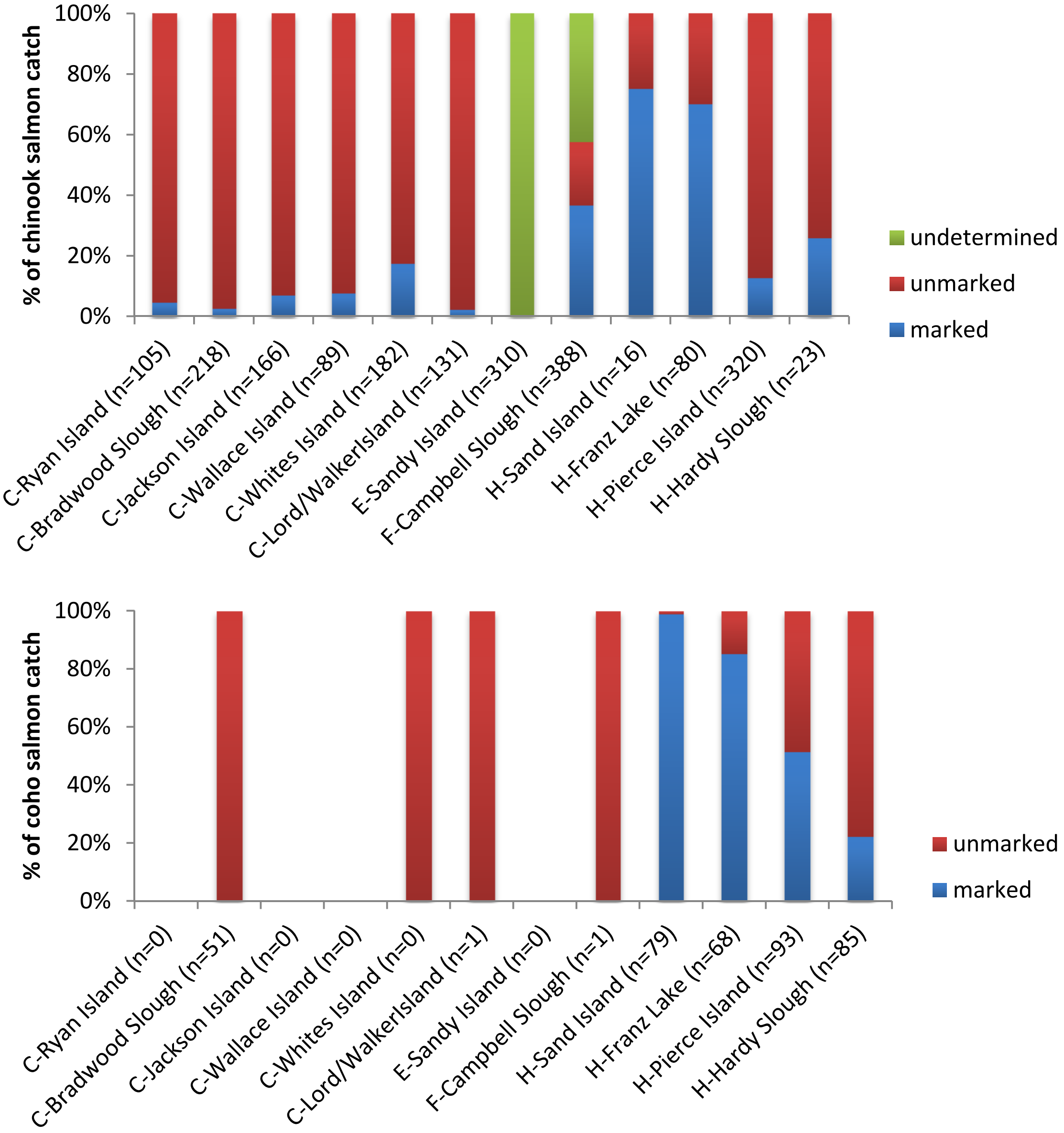


Figure 22. Proportions of a) marked, unmarked, and undetermined Chinook salmon and b) marked and unmarked coho salmon at the EMP sites. The letter before site names indicates the hydrogeomorphic reach in which the site is located. In 2007, fin clips were noted only on necropsied fish from Campbell Slough and Sandy Island, so the category of fish sampled that year could not be determined.

Genetic Stock Identification of Juvenile Chinook Salmon Stock Composition by Site

Juvenile Chinook salmon from stock originating throughout the Columbia Basin (Figure 23) were present at EMP sites.

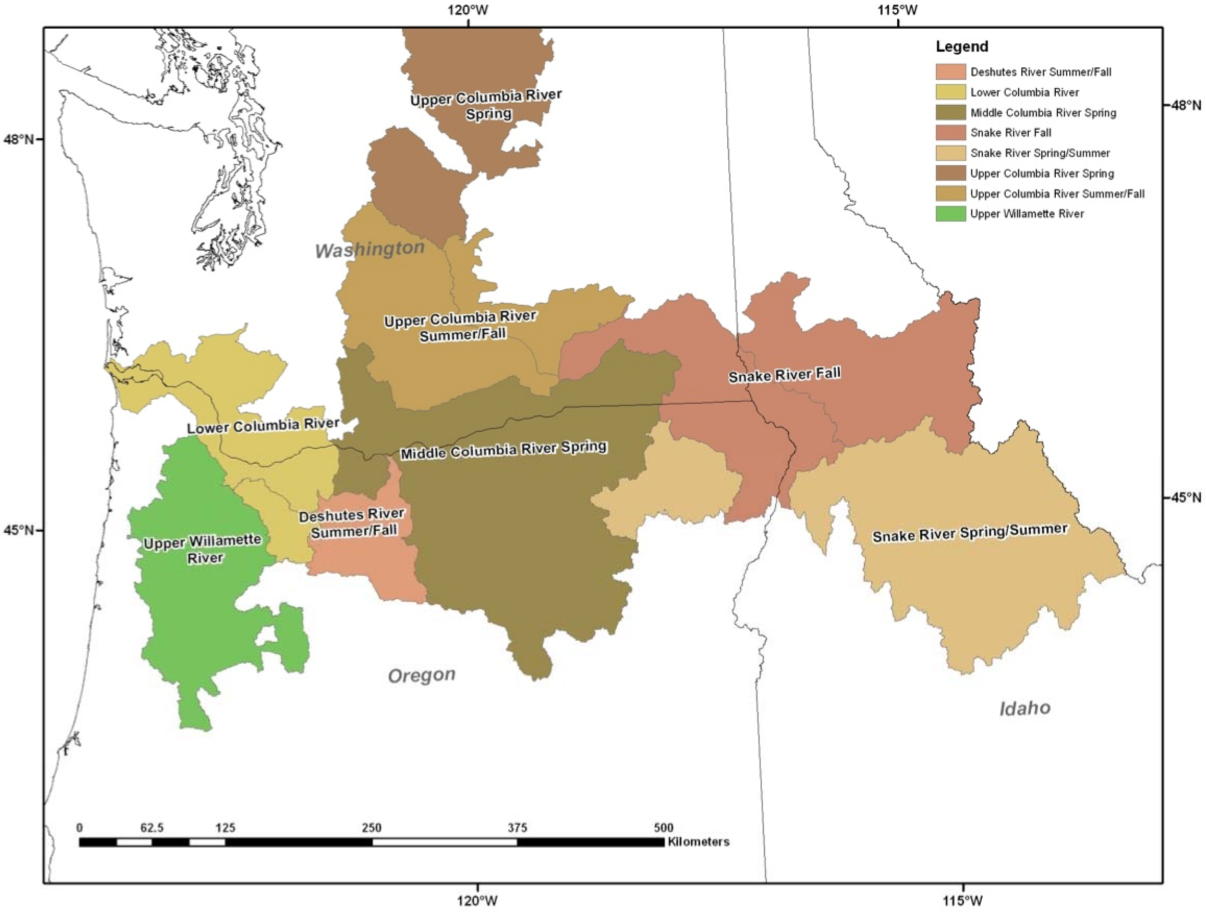


Figure 23. Geographic ranges of Columbia River Chinook salmon stocks.

Unmarked fish. Unmarked Chinook salmon sampled from the Reach C sites (Ryan Island [*n* = 105], Bradwood Slough [*n* = 218], Jackson Island [*n* = 166], Wallace Island [*n* = 89], Whites Island [*n* = 182], and Lord/Walker Island [*n* = 13]) were primarily Lower Columbia River ESU West Cascades fall Chinook, with this stock accounting for 53% to 80% of fish analyzed per site (Figure 24a). An additional 10% to 20% of fish at each site were Lower Columbia River ESU Spring Creek Group fall Chinook, and, at Ryan Island, Jackson Island, and Whites Island, 4% to 5% were West Cascades spring Chinook. Thus, 70% to 98% of fish from these sites were from lower Columbia River stocks. Upper Columbia summer/fall Chinook were the most common interior Columbia fall stock found in Reach C, accounting for from 2% of fish at Jackson Island to 31% of fish at Lord/Walker Island. Small numbers of fish from other stocks, including Upper Willamette spring Chinook, Snake River fall Chinook, and Deschutes River fall Chinook, were also found at the Reach C sites. These fish accounted for about 2% to 5% of fish examined at the sites where they were collected.

West Cascades fall Chinook were also the most common stock at Sandy Island in Reach E (*n* = 310), making up 72% of sampled fish (Figure 24a). Upper Columbia summer/fall Chinook accounted for 17% of fish analyzed. Smaller numbers of Deschutes River fall and Upper Willamette spring Chinook were also found at this site.

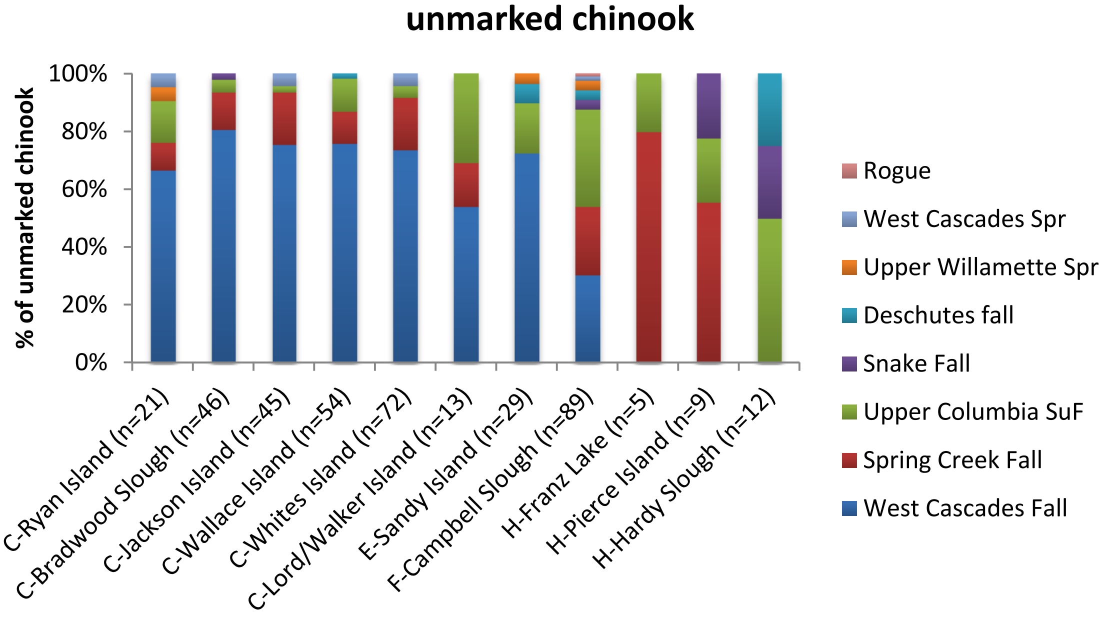
At Campbell Slough in Reach F (*n* = 388), only 30% of fish were from the West Cascades fall group, and 3% were from the West Cascades spring group (Figure 24a). The Spring Creek fall group made up 24% of sampled fish, and the Upper Columbia summer/fall Chinook stock an additional 34% of fish. Additionally, small numbers of fish from a wide range of other stocks were found at Campbell Slough, including Snake River fall Chinook, Deschutes River fall Chinook, Upper Willamette spring Chinook, West Cascades spring Chinook, and Rogue River Chinook. These fish accounted for 1% to 3% of the fish sampled.

In Reach H, West Cascades fall Chinook were not found at any of the sites, and Spring Creek Group fall Chinook became more common (Figure 24a). Interior Columbia stocks were also found in higher proportions in Reach H. At Franz Lake (*n* = 5), 80% of fish were Spring Creek fall Chinook and 20% were Upper Columbia fall Chinook. At Pierce Island (*n* = 9), 56% of fish were Spring Creek fall Chinook, 22% Upper Columbia fall Chinook, and 22% Snake River fall Chinook. At Hardy Slough (*n* = 12), all fish were from interior Columbia stocks; 50% were Upper Columbia fall Chinook, 25% were Snake River fall Chinook, and 25% were Deschutes River fall Chinook.

Marked fish. At the Reach C sites, marked Chinook belonged primarily to the West Cascades fall stock (74% of fish analyzed from sites in this reach; Figure 24b). Spring Creek Group fall Chinook were also fairly common, making up 19% of fish analyzed from Reach C. At Ryan Island (*n* = 4), Wallace Island (*n* = 6), and Whites Island (*n* = 23), West Cascades fall Chinook were the most common stock, accounting for 67% to 100% of marked Chinook salmon sampled. Only one marked fish was collected at Jackson Island, a Spring Creek Group fall Chinook. At Lord/Walker Island two of the four marked fish collected were Spring Creek Group fall Chinook, and the others were West Cascades fall Chinook and West Cascades spring Chinook. Small numbers of marked fish from other stocks (West Cascades spring, Upper Willamette spring, and Upper Columbia fall Chinook) were also observed at some of the Reach C sites.

At Sandy Island in Reach E (*n* = 29), 100% of the 29 fish analyzed were from the West Cascades fall Chinook group (Figure 24b). At Campbell Slough in Reach F, Spring Creek fall Chinook were the most common stock, making up 75% of marked Chinook salmon sampled (Figure 24b). About 18% of sampled marked Chinook from Campbell Slough were from the West Cascades fall stock. Small numbers of fish from a variety of other stocks were also collected at Campbell Slough, including Upper Columbia fall Chinook, Snake River fall Chinook, Upper Willamette spring Chinook, and West Cascades spring Chinook.

At the Reach H sites, the majority of marked Chinook salmon were Spring Creek Group fall Chinook—74% of marked Chinook analyzed from Franz Lake (*n* = 43) and 100% of marked Chinook analyzed from Sand Island (*n* = 14; Figure 24b) were from this stock. Other stocks observed at Franz Lake included West Cascades fall Chinook and Upper Willamette spring Chinook. Only one marked, hatchery Chinook salmon was found at Hardy Slough, and this fish was from the Upper Columbia summer/fall stock.



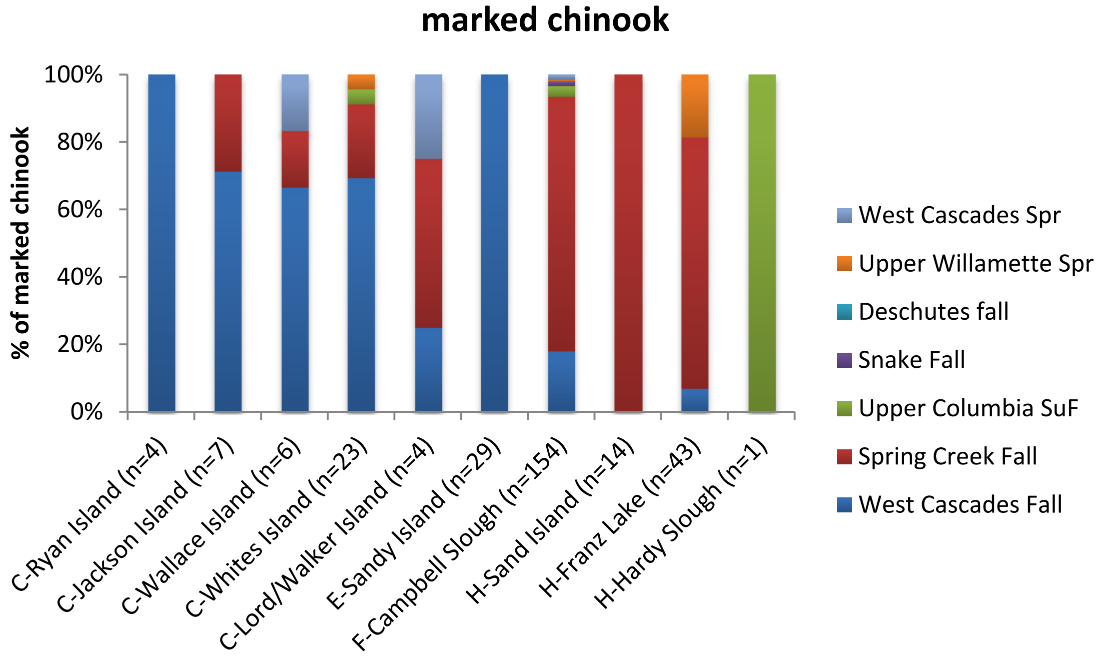


Figure 24. Genetic stock assignments for a) unmarked and b) marked juvenile Chinook salmon collected from EMP sites.

Stock Composition by Month

Unmarked fish. For unmarked Chinook, the proportion of West Cascades fall Chinook remained fairly constant at about 55% to 65% of fish sampled from April through August (Figure 25a). However, the proportion of Chinook belonging to the Spring Creek Group declined from 32% and 22% in April (*n* = 73) and May (*n* = 129) to 7% to 11% in June (*n* = 155), July (*n* = 27), and August (*n* = 9), while the proportion of Chinook belonging to the Upper Columbia summer/fall group increased from 7% in April to 22% to 24% in June through August. Other stocks were relatively rare, but did show some seasonal patterns. Snake River fall Chinook were found from April through June, while Deschutes River fall Chinook were found in June and July. Upper Willamette spring and West Cascades spring Chinook were found only in April and May, and Rogue River Chinook were found only in April.

Marked fish. For marked Chinook, the proportion of Spring Creek Group fall Chinook declined from 83% to 85% in April (*n* = 53) and May (*n* = 142) to 0% in August (*n* = 13), while the proportion of West Cascades fall Chinook increased from 4% in April to 92% in August (Figure 25b). Other stocks were not common, but Upper Columbia fall Chinook were found primarily in June (*n* = 55), while Snake River fall Chinook were found primarily in July (*n* = 29). Upper Willamette spring Chinook were found in April and May only, but West Cascades spring Chinook were found throughout the sampling season, with the highest proportion of fish found in July and August.

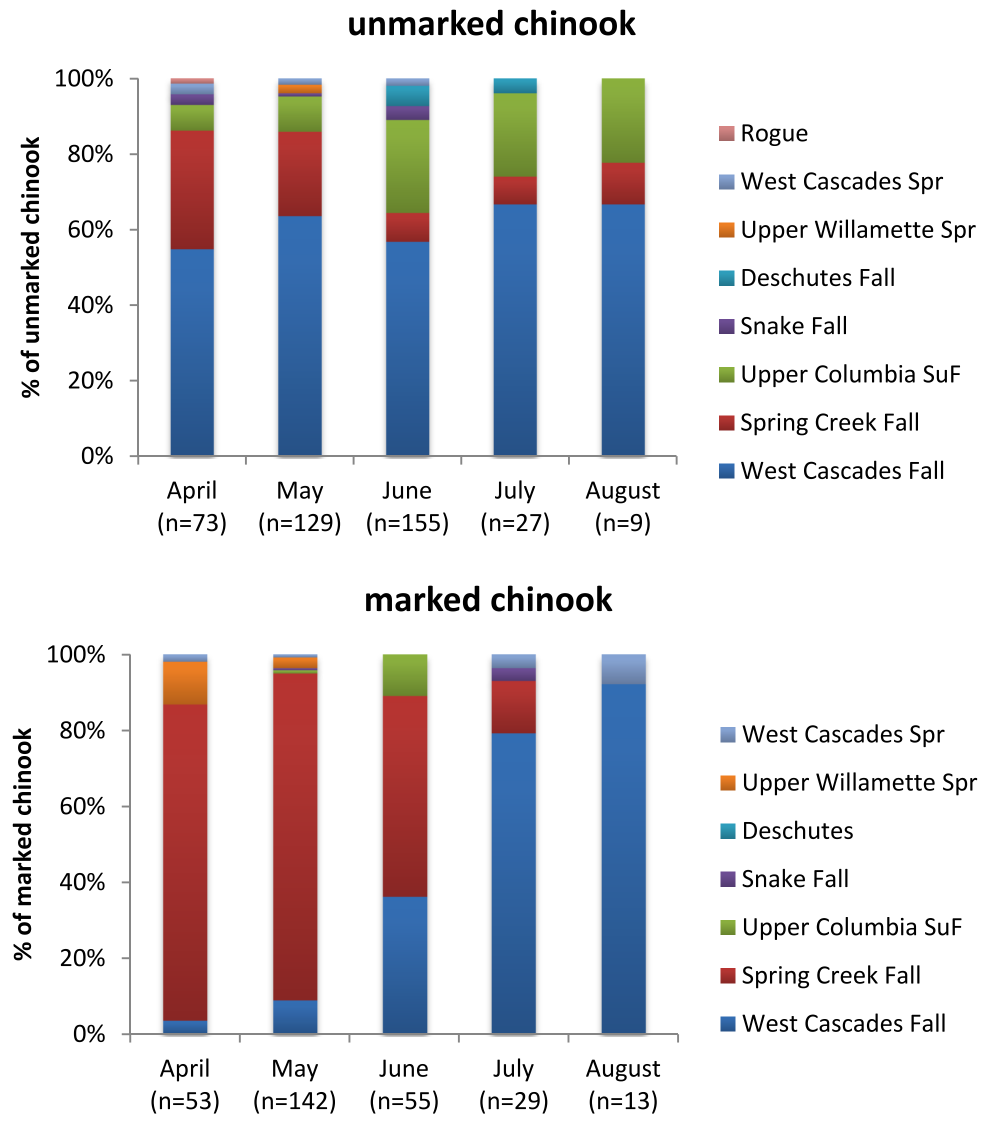


Figure 25. Proportions of different Chinook salmon stocks observed by month in a) unmarked and b) marked Chinook salmon from the EMP sites.

Salmonid Seasonal Habitat Occurrence

Overall, Chinook salmon were the most abundant salmon found at EMP sites, with an average density over the sampling season of 24 unmarked fish and six marked fish per 1,000 m^2 (Figure 26). Coho salmon, especially unmarked coho, were generally less abundant than Chinook, with an average density over the sampling season of 5.8 unmarked fish and 4.5 marked fish per 1,000 m^2 (Figure 26). Densities of chum and steelhead were much lower (Figure 26), with an average of two fish per 1,000 m^2 for chum and 0.6 fish per 1,000 m^2 for steelhead. All chum and steelhead captured were unmarked.

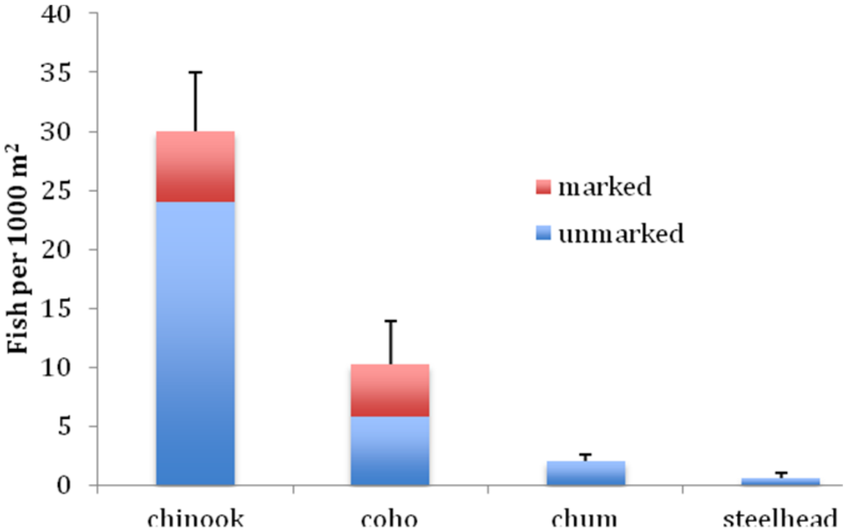


Figure 26. Average density over the sampling season of Chinook salmon, coho salmon, chum salmon, and steelhead trout.

Chinook Salmon. Chinook salmon were present at most sites from April through July or August (Figure 27). Densities were highest in May, declining to very low numbers by August. The same general pattern was observed for both marked and unmarked Chinook (Figure 27), but overall densities were higher for unmarked fish. Also, for unmarked Chinook, densities were significantly different (p<0.0001) over the sampling season, with highest levels in May and June and lowest levels in July and August. For marked Chinook, densities were not significantly different (p = 0.1376) although they tended to follow the same pattern as unmarked fish.

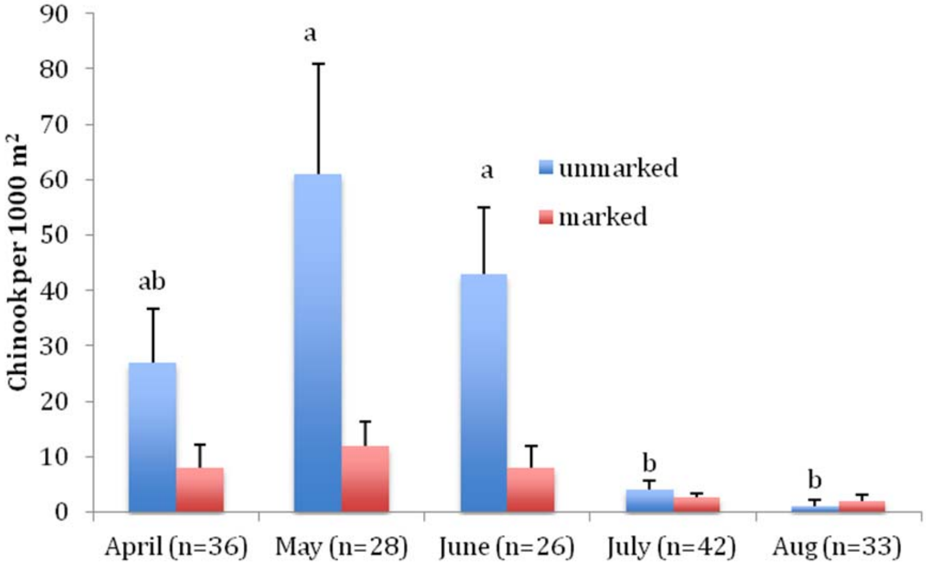


Figure 27. Mean density (Standard Deviation [SD]) by month of marked and unmarked Chinook salmon.

Highest average densities of unmarked Chinook salmon over the whole sampling season were found at Pierce Island in Reach H (143 fish per 1,000 m^2) and at Bradwood Slough in Reach C (90 fish per 1,000 m^2; Figure 28). Relatively high densities of unmarked Chinook were also found at the rest of the Reach C sites, ranging from 17 to 54 fish per 1,000 m^2. Densities at the other sites ranged from 0.5 to 7 fish per 1,000 m^2. Marked Chinook (Figure 28) were most abundant at Pierce Island, where the overall density was 21 fish per 1,000 m^2, and at Campbell Slough and Franz Lake, with densities of 10 and 11 fish per 1,000 m^2. Densities of marked Chinook at other sites ranged from one to four fish per 1,000 m^2.

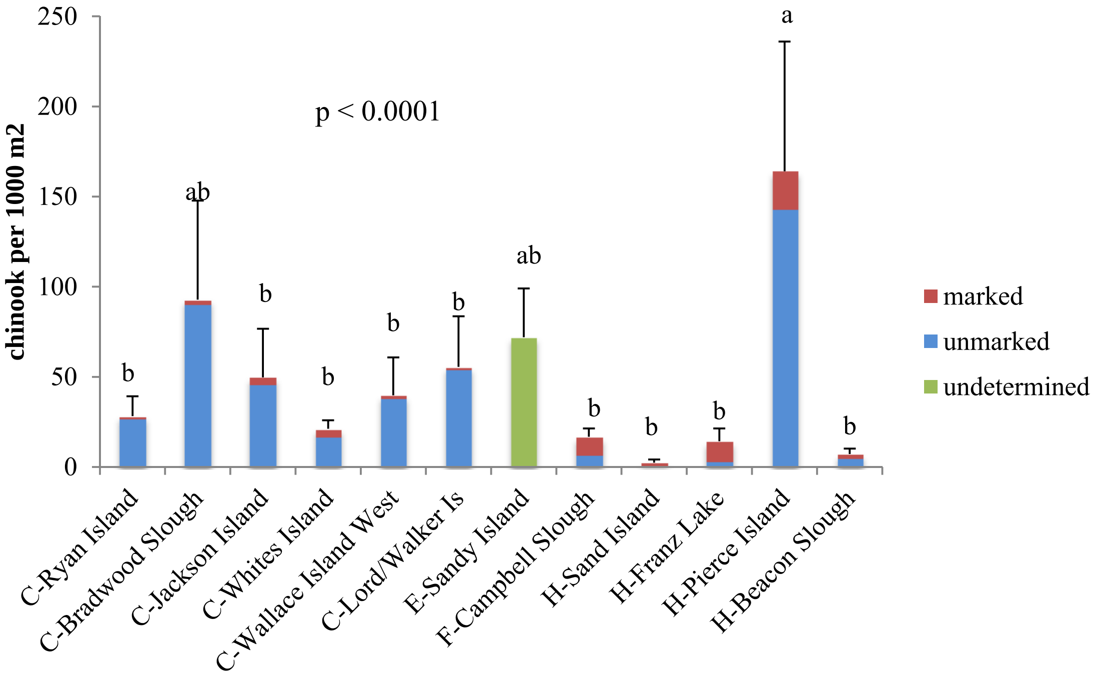
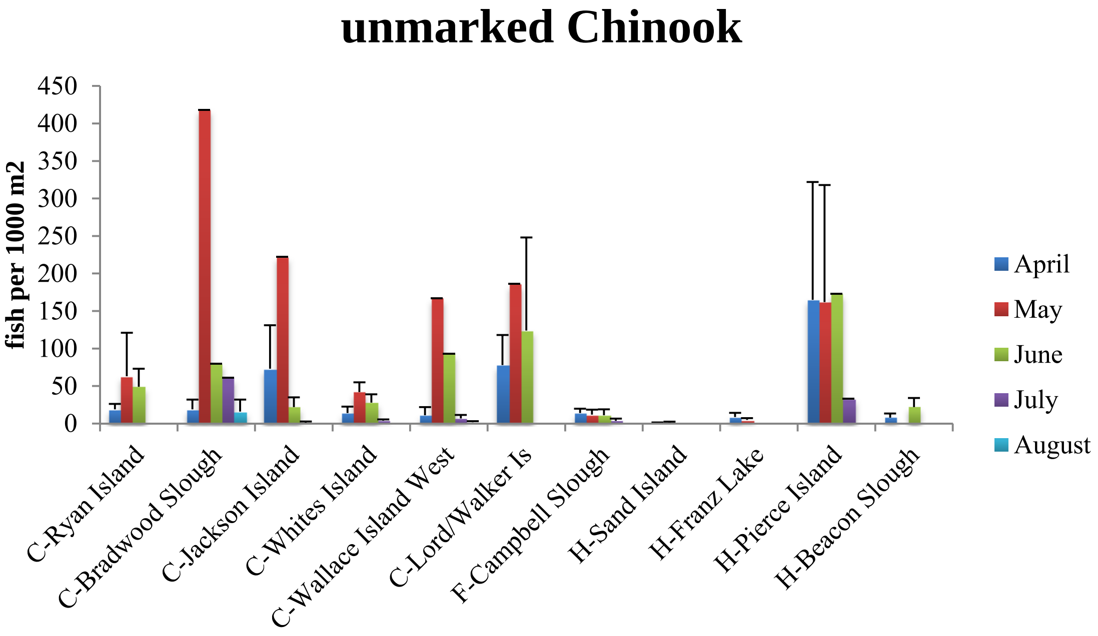


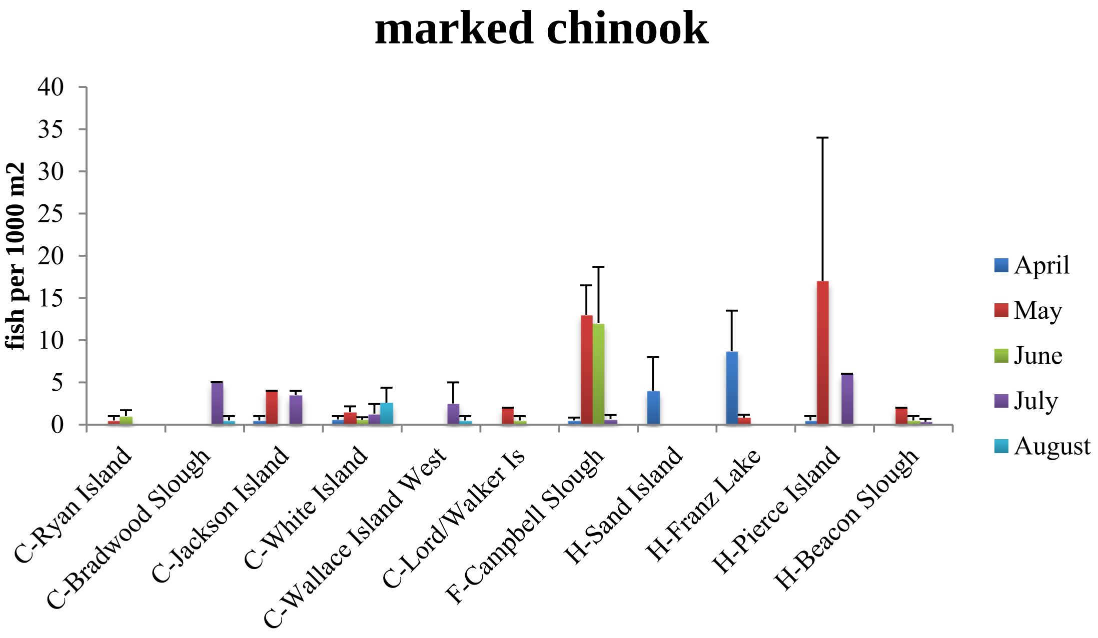
Figure 28. Mean density (SD) by site of Chinook salmon at the EMP sampling sites. Values with different letter designations are significantly different (1-way ANOVA and Tukey's least significant difference test, $p<0.05$).

Rkm did not have a clear relationship with density of either marked or unmarked Chinook, as densities tended to be highest at sites both nearest to and farthest from the mouth of the Columbia River. In multiple regression analyses, after adjusting for month of sampling, rkm was not a significant predictor of density for either marked or unmarked Chinook ($0.1338<p<0.2443$). Densities of unmarked Chinook tended to be lower at sites farther from the main channel. In a multiple regression analysis, after adjusting for the effect of sampling month, unmarked Chinook density declined significantly as the distance of the site from the main channel increased ($p = 0.0106$). However, distance from the main channel did not have a significant effect on density of marked Chinook ($p = 0.3691$).

Densities of unmarked Chinook salmon showed similar seasonal patterns at most of the sampling sites, with highest densities observed in May (Figure 29). An exception was the Pierce Island site, where unmarked Chinook densities were very similar in April, May, and June. Seasonal patterns of density were less consistent from site to site for marked Chinook salmon (Figure 29). At those sites with highest densities of marked Chinook (Campbell Slough and Pierce Island), peak densities were in May and June. Franz Lake, which also had a relatively high density of marked coho, showed a different pattern, with the highest density in April.



unmarked Chinook



marked chinook

Figure 29. Average density of marked and unmarked Chinook salmon by site and month at the EMP sampling sites.

Coho Salmon. The highest density of unmarked coho salmon was found in August, primarily because of a very large number of coho being caught at that time at Hardy Slough, in Reach H, followed by April and May (Figure 30). However, densities of unmarked coho did not vary significantly with sampling month. Unlike unmarked coho, marked coho (Figure 30) showed significant seasonal differences in density, with the highest density of fish observed in May (p = 0.0091).

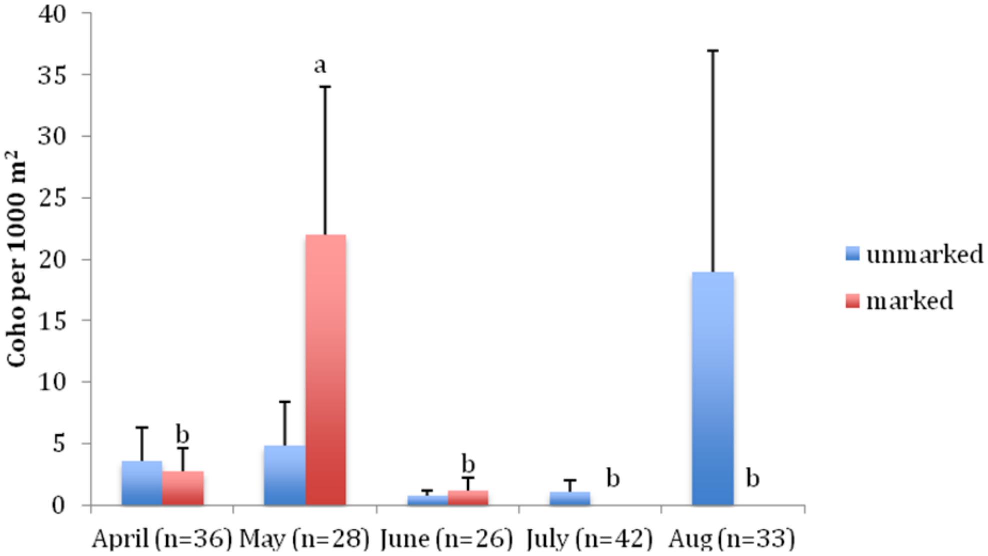


Figure 30. Mean density (SD) by month of marked and unmarked coho salmon. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, $p<0.05$).

Among sampling sites (Figure 31), highest mean densities of unmarked coho over the entire sampling period were found at Hardy Slough (50 fish per 1,000 m^2), Bradwood Slough (22 fish per 1,000 m^2), and Pierce Island (18 fish per 1,000 m^2). At other sampling sites, coho densities were low, ranging from 0 to 1.5 fish per 1,000 m^2). Marked coho were found only in Reach H, with highest mean densities at Pierce Island (25 fish per 1,000 m^2) and Sand Island (33 fish per 1,000 m^2) (Figure 31).

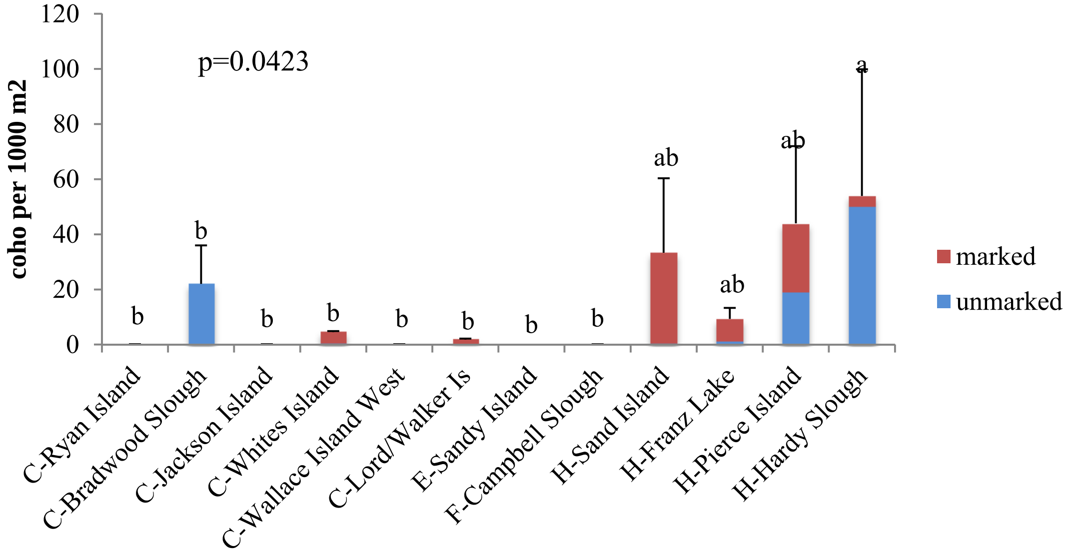
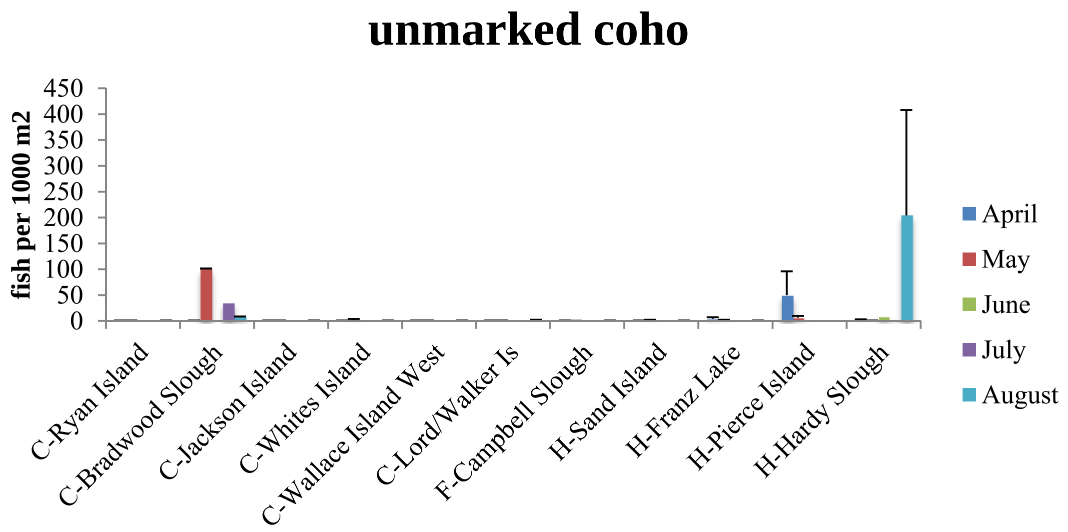


Figure 31. Mean density (SD) of coho salmon at the EMP sampling sites. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, *p*<0.05).

At sites where unmarked coho were present, there was little consistency in their seasonal occurrence (Figure 32). The maximum coho density reported was at Hardy Slough in August (204 fish per 1,000 m^2). High numbers of fish were also collected at Bradwood Slough in May (101 fish per 1,000 m^2) and Pierce Island in April (50 fish per 1,000 m^2). With the exception of Bradwood Slough in Reach C, unmarked coho were found at very low numbers in all sites outside of Reach H. Marked coho were observed in May at both sites, Sand Island and Pierce Island, where they were found in significant numbers.

The density of unmarked coho did not show a significant relationship with rkm (p = 0.1463), as high densities were found at selected sites in Reach C as well as Reach H. Nor was there any significant relationship between distance from the main channel and coho density (p = 0.5075). However, the density of marked coho increased significantly as rkm increased (p = 0.0048), and also decreased significantly as the distance of the sampling site from the main channel increased (p = 0.0302).



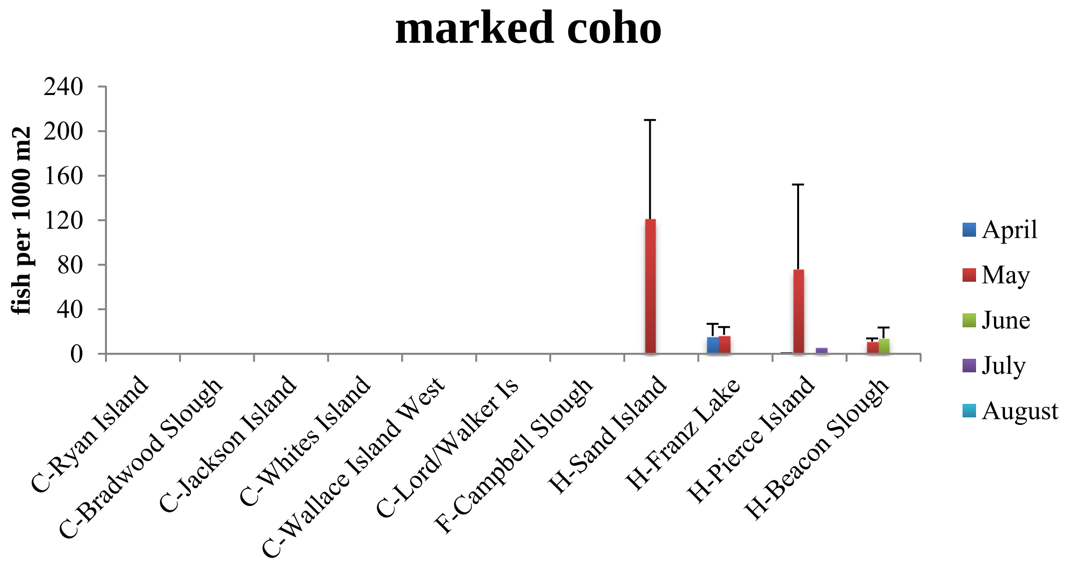


Figure 32. Average density of marked and unmarked coho salmon by sites and month.

Chum Salmon. Chum salmon were found at highest densities in April, and were also present in lower numbers in May, but were absent later in the season (Figure 33). Among the sampling sites, chum were found at the highest density at Pierce Island in Reach H (15 fish per 1,000 m^2), followed by Ryan Island in Reach C (8 fish per 1,000 m^2) (Figure 34). Seasonal patterns of occurrence were fairly similar at all sites (Figure 35), although chum were more likely to be observed in May at downstream sites, including Sandy Island, Ryan Island, and Jackson Island. Chum density did not show a clear correlation with rkm. In the multiple regression analysis, after adjusting for the effect of sampling month, rkm was not significant (*p* = 0.8736). However, chum density tended to decline with increasing distance from the mainstem of the river (*p* = 0.0474).

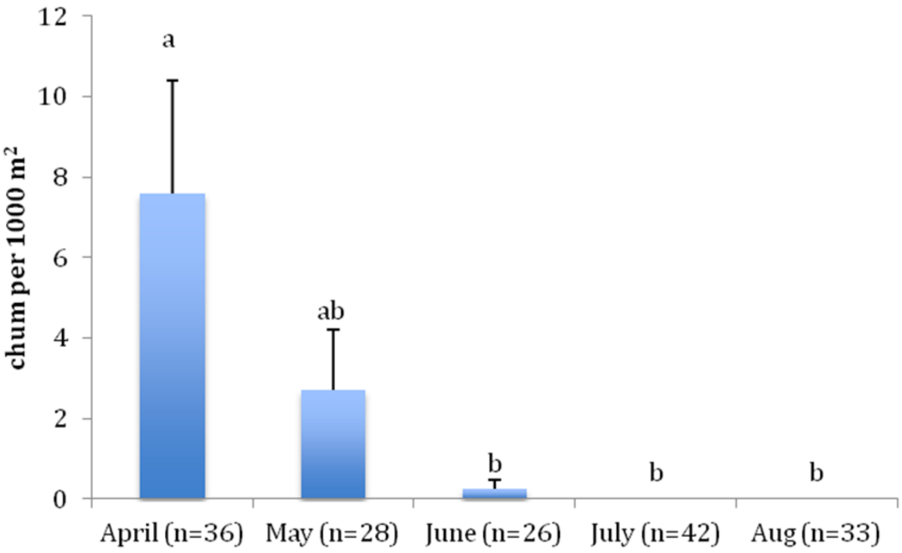


Figure 33. Mean density (SD) by of chum salmon by month. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, *p*<0.05).

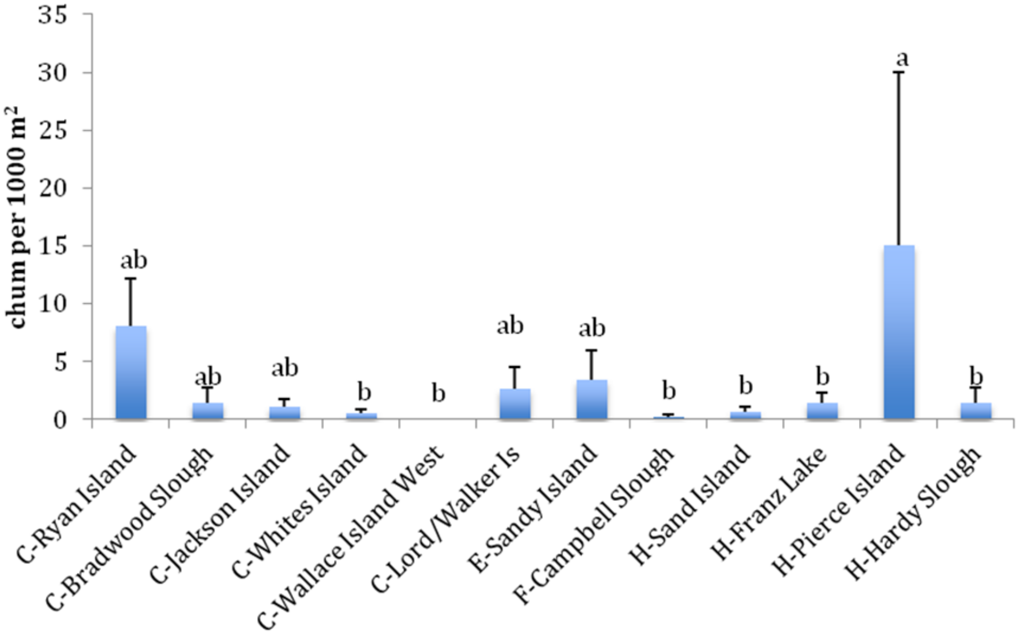


Figure 34. Mean density (SD) of chum salmon at EMP sampling sites. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, *p*<0.05).

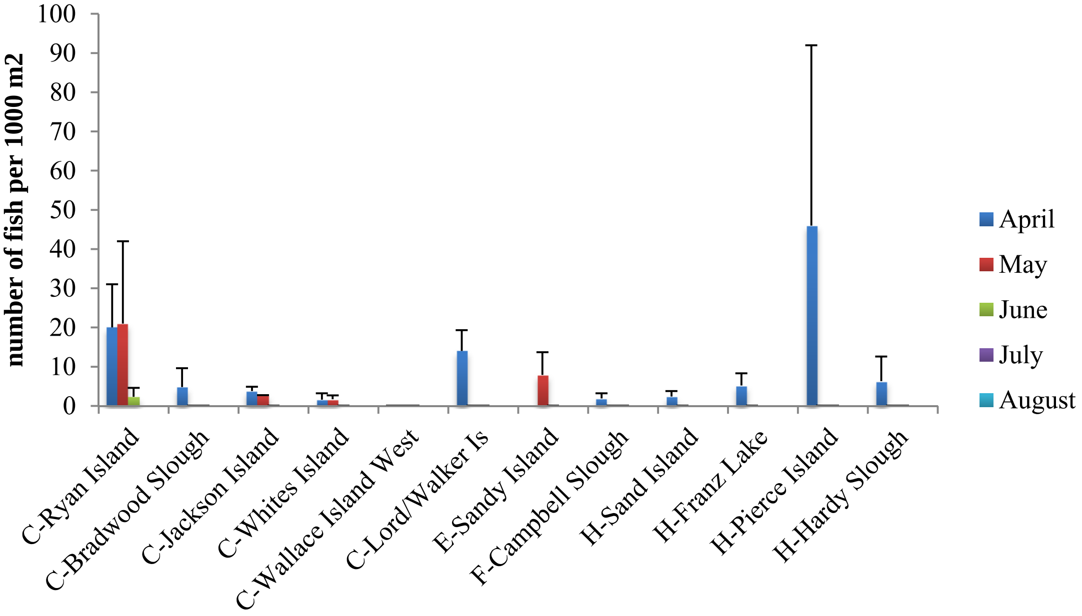


Figure 35. Mean density (SE) of chum salmon by site and month.

Steelhead Trout. Steelhead trout were found at highest densities in July, and were also present in lower numbers in April and June (Figure 36). Among the sampling sites, steelhead were found at the highest density at Franz Lake in Reach H (4.8 fish per 1,000 m^2; Figure 37). Small numbers of steelhead were also observed at Bradwood Slough and Campbell Slough. Seasonal patterns of occurrence were not consistent among sites (Figure 38). At Campbell Slough and Bradwood Slough, steelhead were caught only in June, but at Franz Lake, where they were most abundant, they were found in April and July.

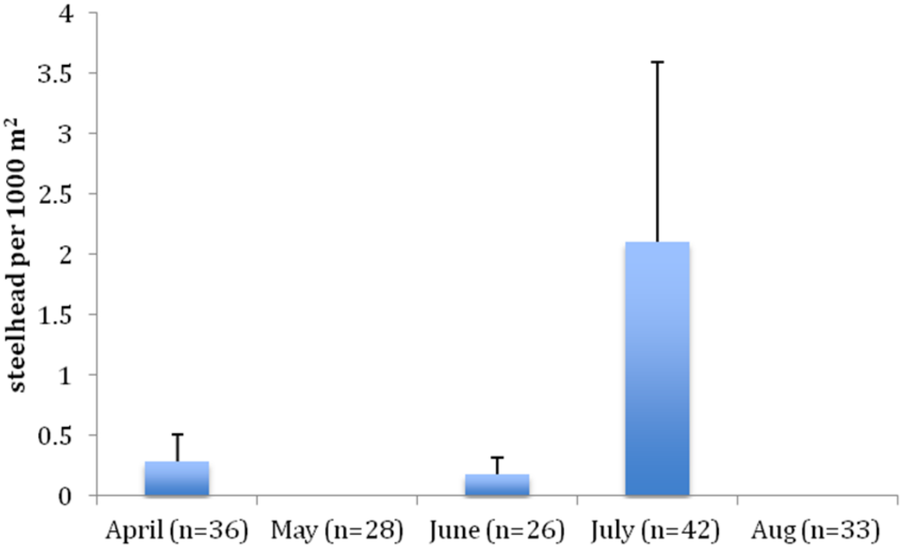


Figure 36. Mean density (SD) of steelhead trout by month.

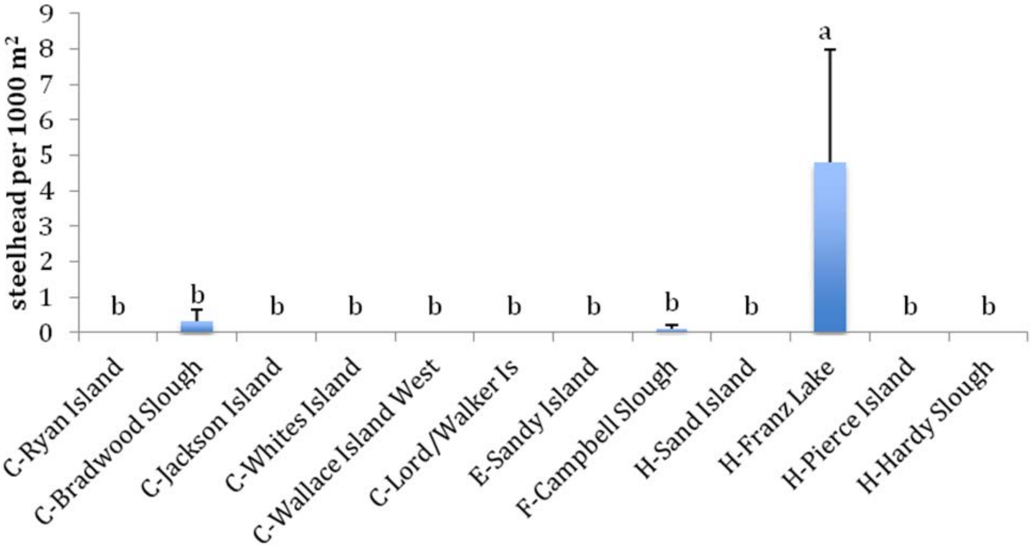


Figure 37. Mean density (SD) of steelhead trout at the EMP sampling sites.

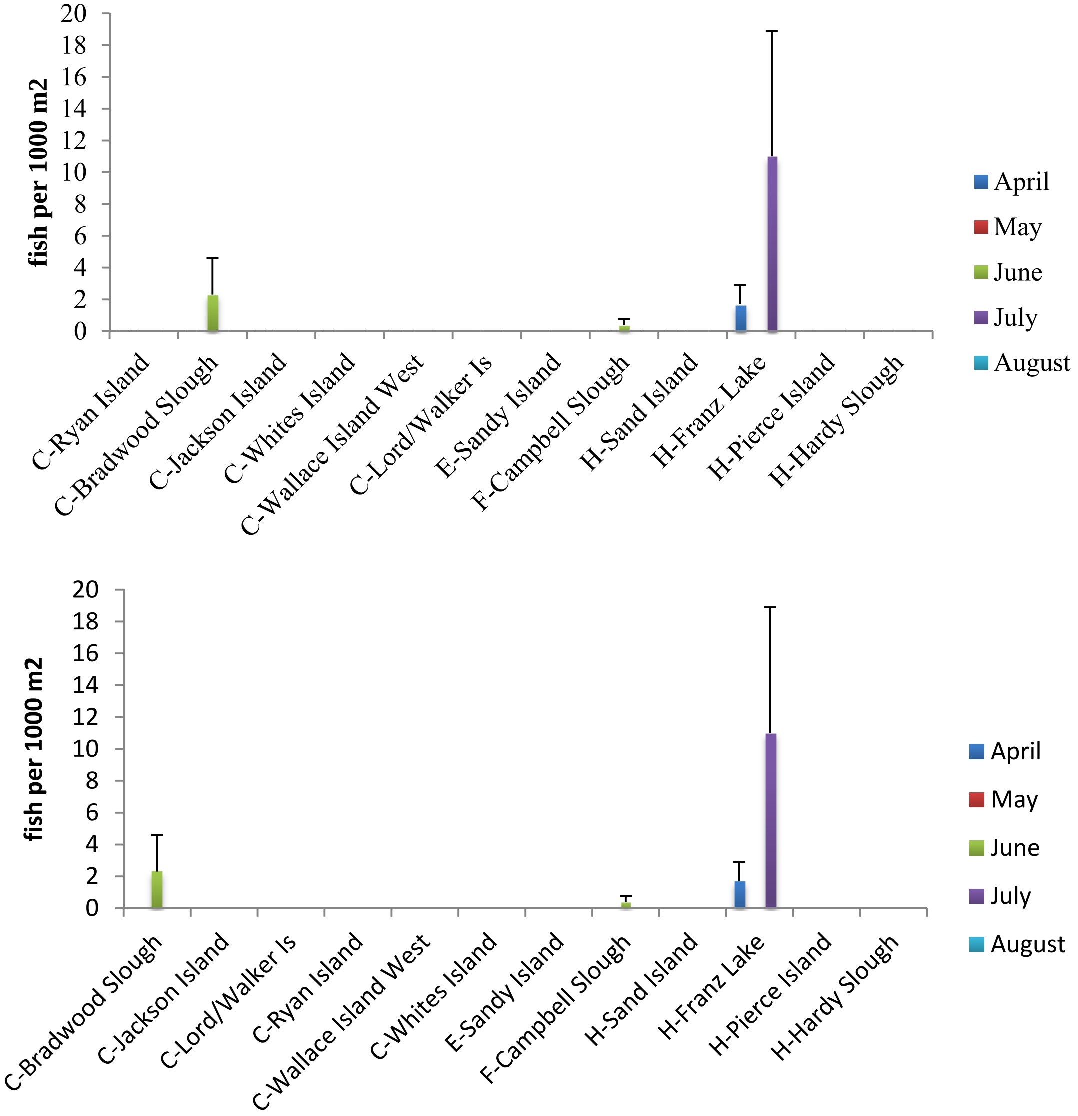


Figure 38. Mean density (SD) of steelhead trout by site and month.

Size Distribution, Length, Weight, and Condition Factor

Chinook Salmon. Size distributions, length, weight, and K by month for marked and unmarked Chinook salmon for sites in Reaches C, E, F, and H are shown in Figure 39 and Figure 40.

For unmarked Chinook salmon, size ranged from 36 to 96 mm fork length, corresponding to subyearling fry <60 mm and fingerlings (60–100 mm; Fresh et al. 2005), and generally followed a normal distribution (Figure 39). However, for marked Chinook salmon, sizes followed a bimodal distribution, with the majority of fish within a size range of 54 to 95 mm, corresponding to subyearling fry and fingerlings, and a second group with sizes ranging from 130 to 177 mm, a size range characteristic of yearling Chinook (Figure 39). Proportions of fry in unmarked Chinook catches varied from 21% at Campbell Slough to 80% or more at Lord/Walker Island, Sand Island, and Franz Lake (Figure 40). The great majority of marked Chinook were within the fingerling size range (Figure 40) but some fry were found at Bradwood Slough and Pierce Island. Chinook within larger yearling size cohort almost all came from Hardy Slough in Reach H, but smaller numbers were from Whites Island and Jackson Island in Reach C. Chinook within this larger size cohort were all captured in April and May. For these yearling fish, length was significantly greater for those captured in April than in May (168 mm ± 6 [$n = 3$] versus 139 mm ± 3 [$n = 13$], $p = 0.0011$), and so was weight (46 g ± 13 [$n = 3$] versus 27 g ± 2 [$n = 13$], $p = 0.0020$). However, there was no significant difference in condition (0.94 ± .05 [$n = 3$] versus 0.99± .02 [$n = 13$], $p = 0.406$). There were also significant differences among sites, with both length and weight decreasing with increasing rkm. However, there was no difference in K ($p = 0.8541$). Distance from the main channel had no relationship to length, weight, or condition for the yearlings ($0.3891<p<0.7136$).

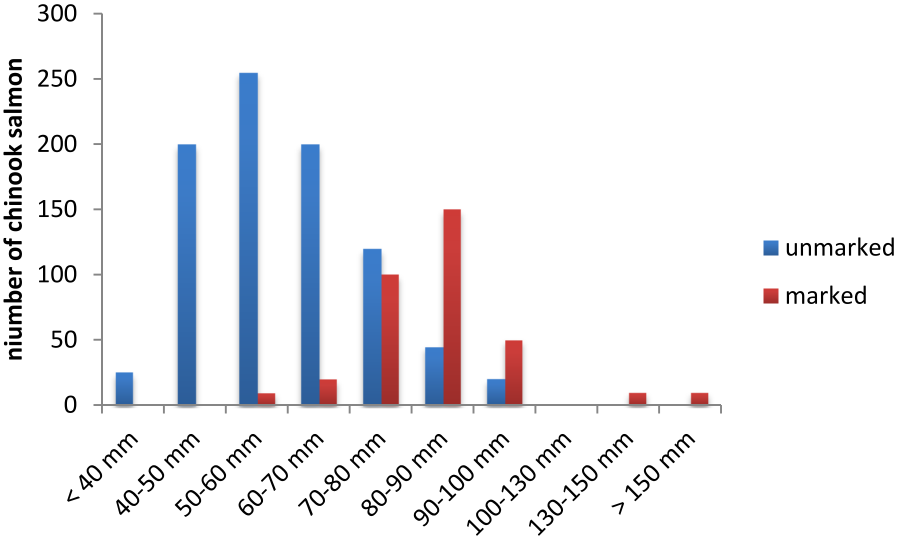
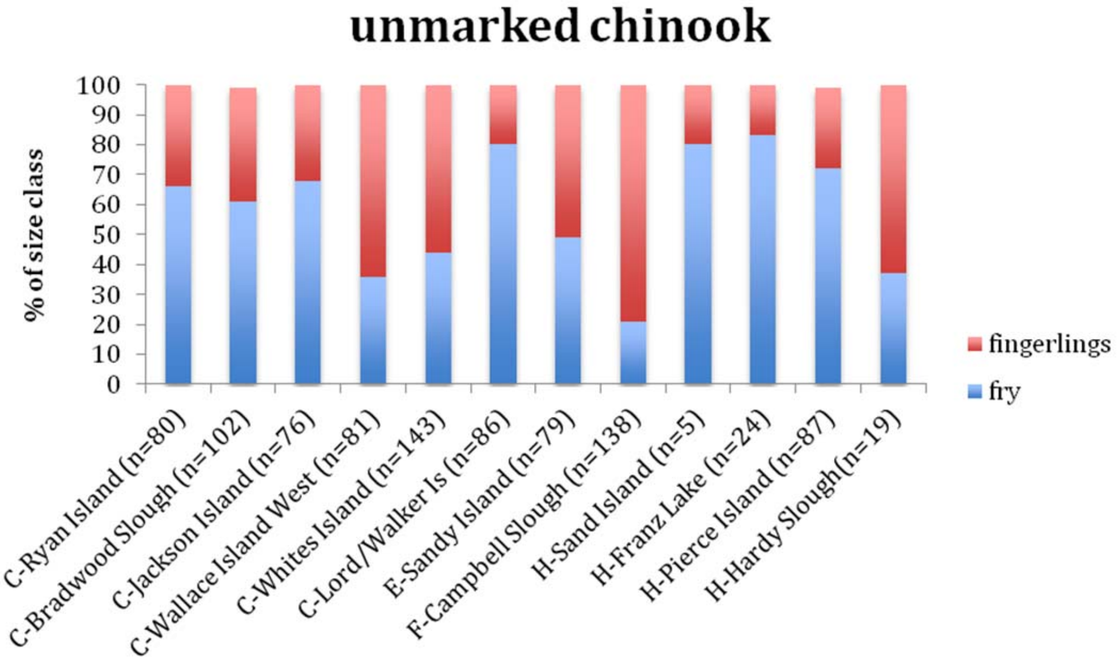


Figure 39. Size distributions (fork length in mm) for unmarked and marked Chinook salmon from the EMP monitoring sites.



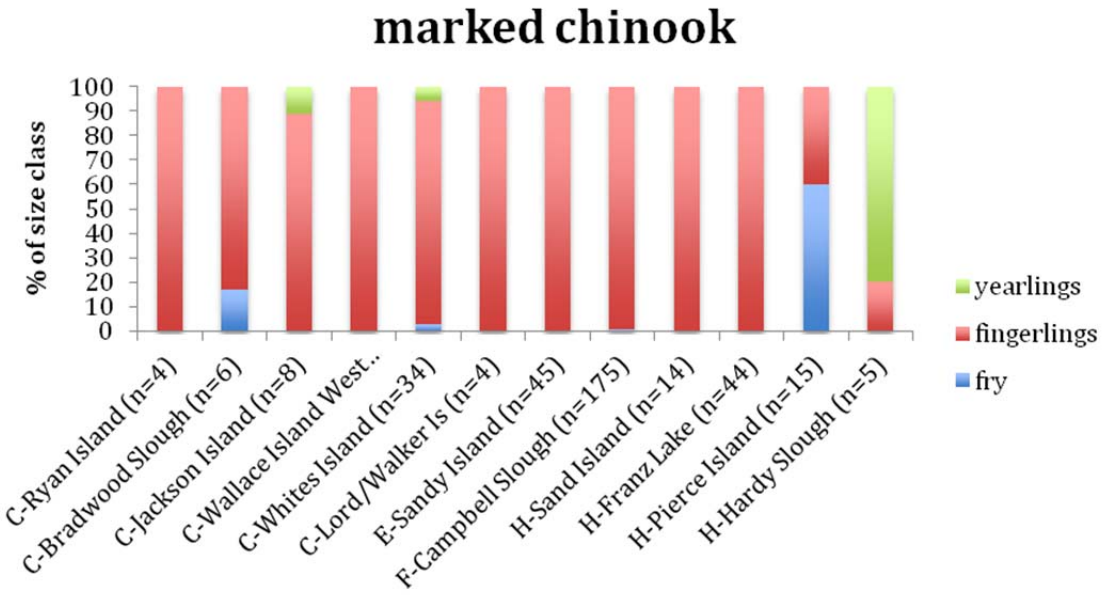
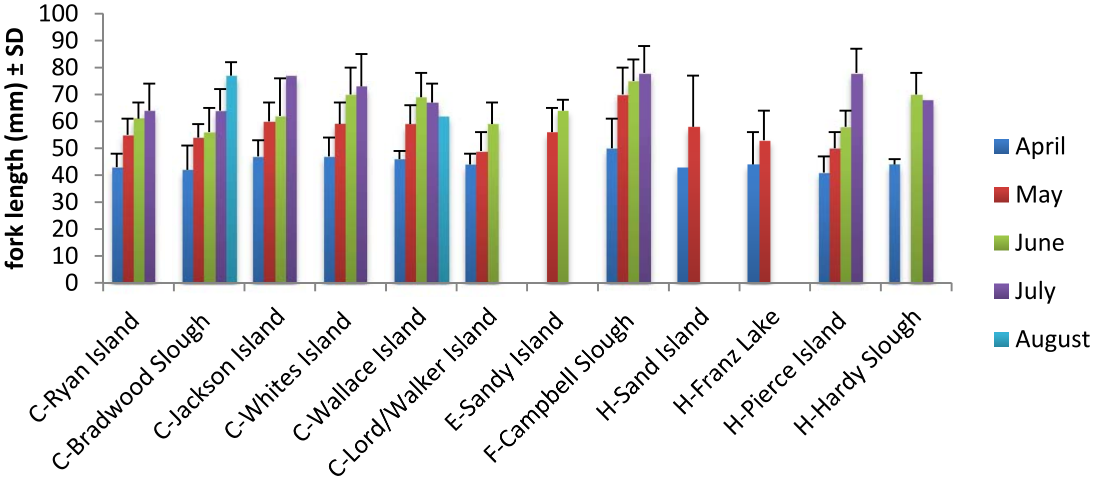


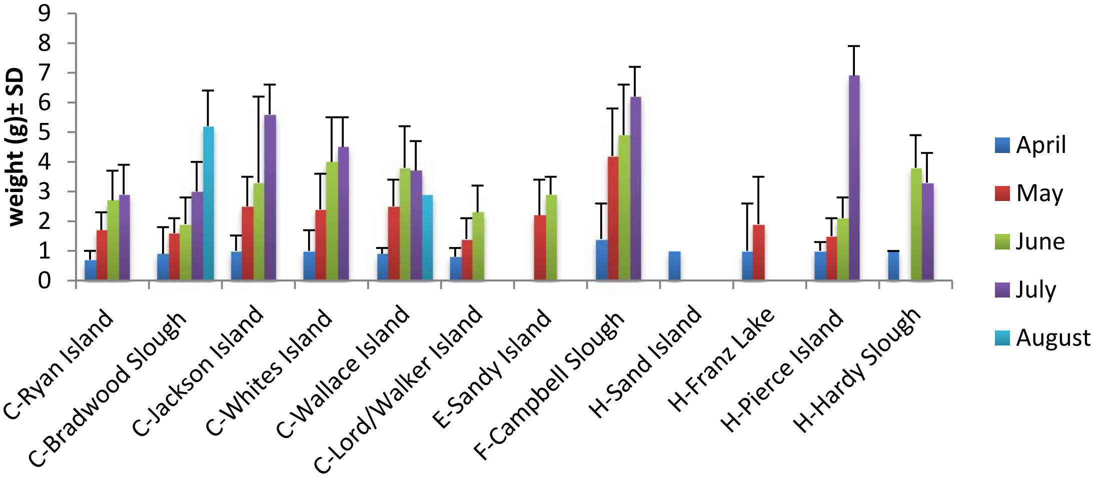
Figure 40. Proportions of unmarked and marked juvenile Chinook salmon of different life history stage size classes at the EMP sampling sites. Fry are <60 mm; fingerlings are 60 to 100 mm; yearlings are >130 mm (Fresh et al. 2005).

For the unmarked, subyearling-sized fish, length increased significantly ($p<0.0001$) over the sampling season, from an overall mean of 45 mm in April to 75 mm in August (Figure 41). Multiple regression analysis indicated that after adjusting for the effect of sampling month, there were significant differences in length among the sampling sites ($p<0.0001$). Fish from Bradwood Slough, Lord/Walker Island, and

Ryan Island in Reach C, and from Pierce Island in Reach H, were significantly smaller than the overall average, while fish from Sandy Island were significantly larger. A similar relationship was seen with fish weight (Figure 41). It also increased significantly over the sampling season (*p*<0.0001), from an overall mean of 0.97 g in April to 4.9 g in August. As with length, after adjusting for the effect of sampling month, significant differences in fish weight among the sampling sites were observed (*p*<0.0001). Fish weight was significantly lower than the overall average at Bradwood Slough, Lord/Walker Island, and Ryan Island in Reach H, and higher than the overall average at Campbell Slough in Reach F and Sandy Island in Reach E. Fish condition (K) also increased significantly over the sampling season (*p*<0.0001) from an overall mean of 0.97 in April to 1.15 in July with no change in August (

Figure **42**). After adjusting for the effect of sampling month, site still had a significant effect on K (*p*<0.0001). K was significantly lower than the overall average at Bradwood Slough, Jackson Island, Lord-Walker Island, and Ryan Island in Reach C, and at Franz Lake in Reach H. It was higher than the overall average at Pierce Island in Reach H.





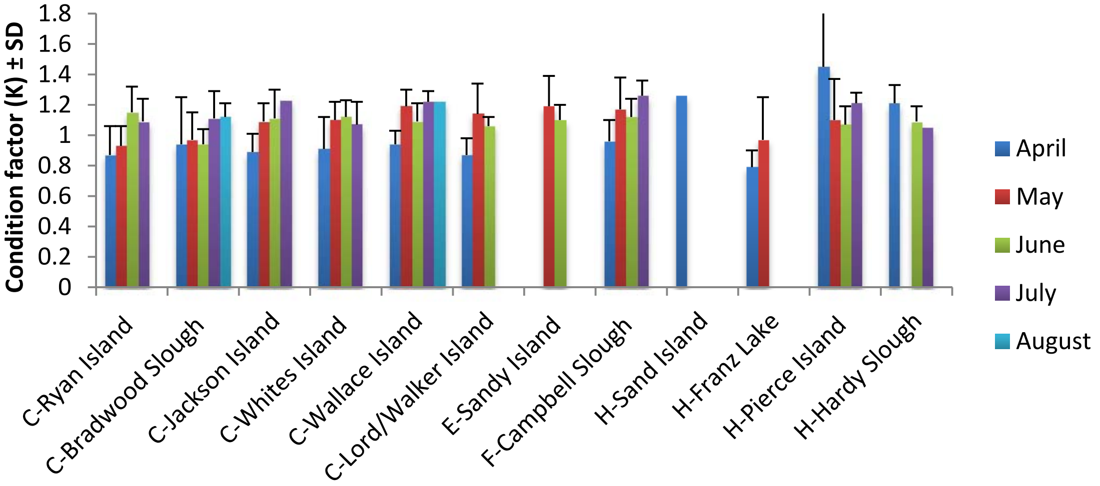
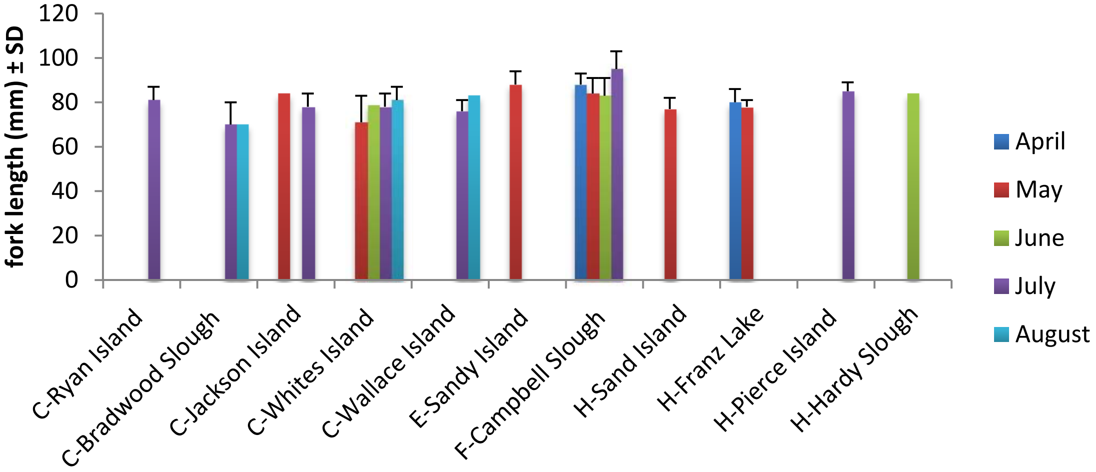


Figure 41. Mean length, weight, and K by month and site for unmarked Chinook salmon

For unmarked Chinook, after adjusting for effect of sampling month, length tended to increase as rkm decreased (*p* = 0.054) and increase as distance from the channel increased (*p*<0.0001). Weight showed no trend with rkm, (*p* = 0.4746) but did tend to increase with distance from the main channel (*p* = 0.0001). Condition (K) increased as rkm increased (*p*<0.0001), but was not related to distance from the main channel (*p* = 0.3339).



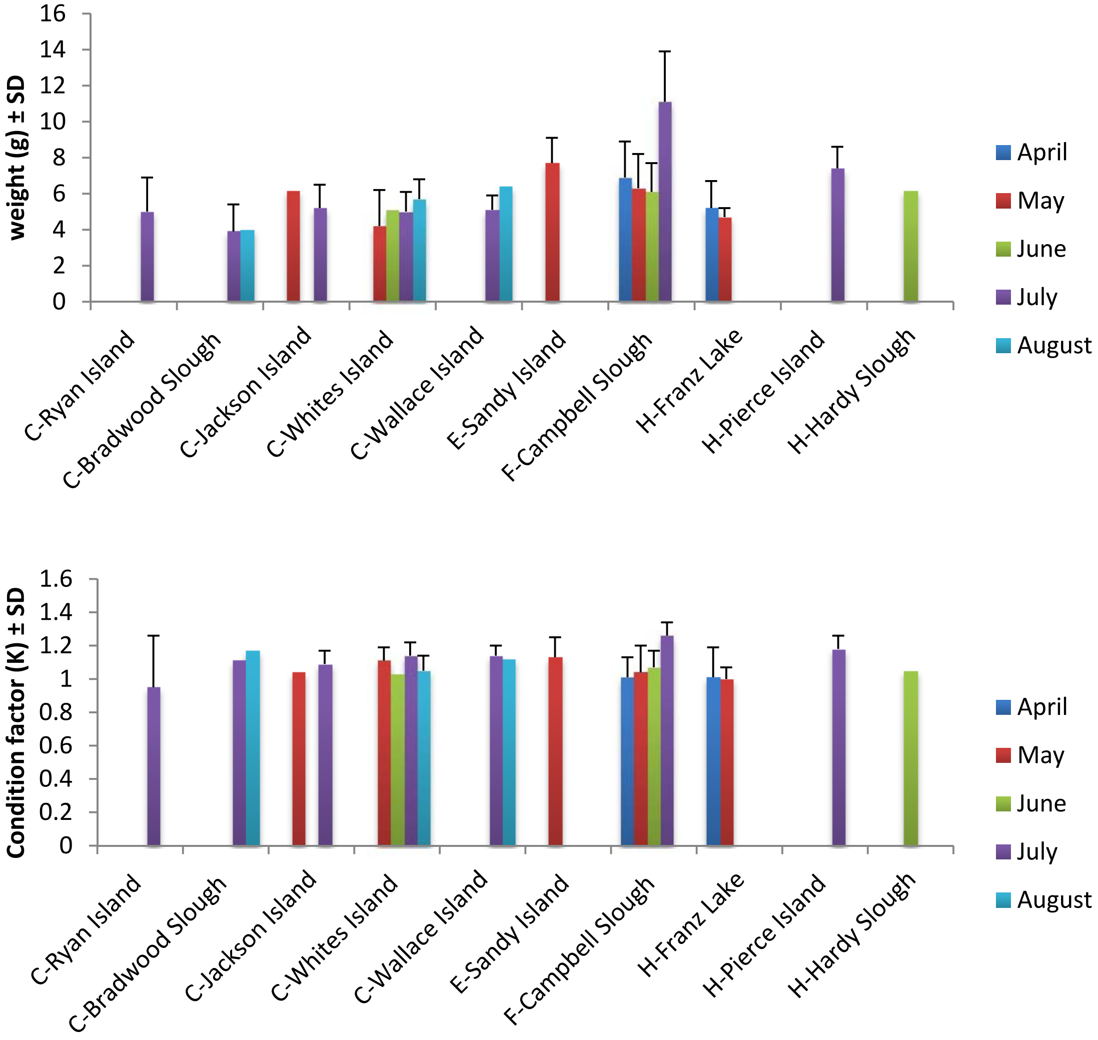


Figure 42. Mean length, weight, and K for marked Chinook salmon by site and month. Yearling size Chinook are not included in these figures.

Seasonal patterns of length and weight were quite different for the marked Chinook (

Figure **42**). Overall, length was highest in April, with an overall mean length of 87 mm, and lowest in June with an overall mean length of 77 mm. This pattern was due to the presence of a number of yearling Chinook, with length >130 mm, collected in May and June. When these larger fish were excluded from the analysis, there was no significant difference over the sampling season in fish length for marked fish, and mean lengths for the sampling months ranged from 77 to 81 mm (

Figure **42**). The same pattern was seen for fish weight (

Figure **42**), which ranged from 5.3 to 5.9 g over the sampling season with no significant differences among months ($p = 0.3694$). Unlike length and weight, condition did change significantly over the sampling season for marked fish that were not yearlings (

Figure **42**).

For marked Chinook salmon, after adjusting for effect of month of capture, length tended to increase as rkm increased (*p* = 0.0001), but there was no significant effect of distance from the main channel (*p* = 0.7504). Weight also showed tended to increase as rkm increased but not significantly (*p* = 0.0853), and like length, was not affected by distance from the main channel (*p* = 0.6821). K was unrelated to rkm (*p* = 0.8910), but tended to increase as distance to the main channel decreased (*p* = 0.0329).

Coho Salmon. Size distribution and length, weight, and K by month for marked and unmarked coho salmon for sites in Reaches C, E, F, and H are shown in Figure 43, Figure 44, Figure 45, and Figure 46.

For unmarked coho, two different size cohorts were observed (Figure 43), one consisting of large fish 110 to 150 mm in length and 16 to 31 g in weight, which were found only in May, and another of smaller fish 37 to 103 mm in length and 0.3 to 12.6 g in weight that were found from April through August. The larger coho were found only at the Reach H sites (Hardy Slough, Franz Lake, and Sand Island). For coho within this larger cohort, there was no significant difference in length, weight, or condition among the sites (Figure 44). Nor was there any relationship between any of these parameters and rkm (0.118<*p*<0.9445) or distance from the main channel (0.5553<*p*<0.6980).

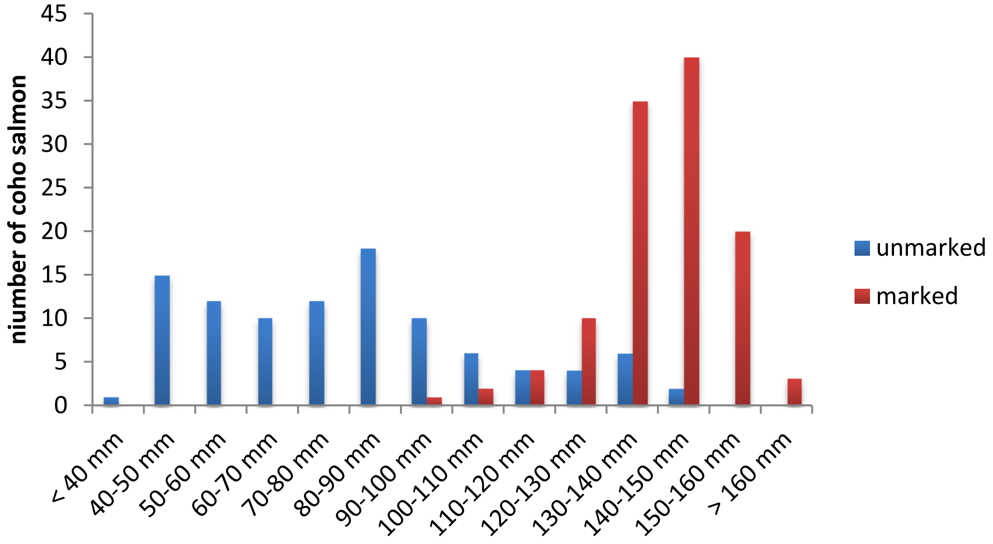


Figure 43. Size distributions (fork length in mm) of unmarked and marked coho salmon.

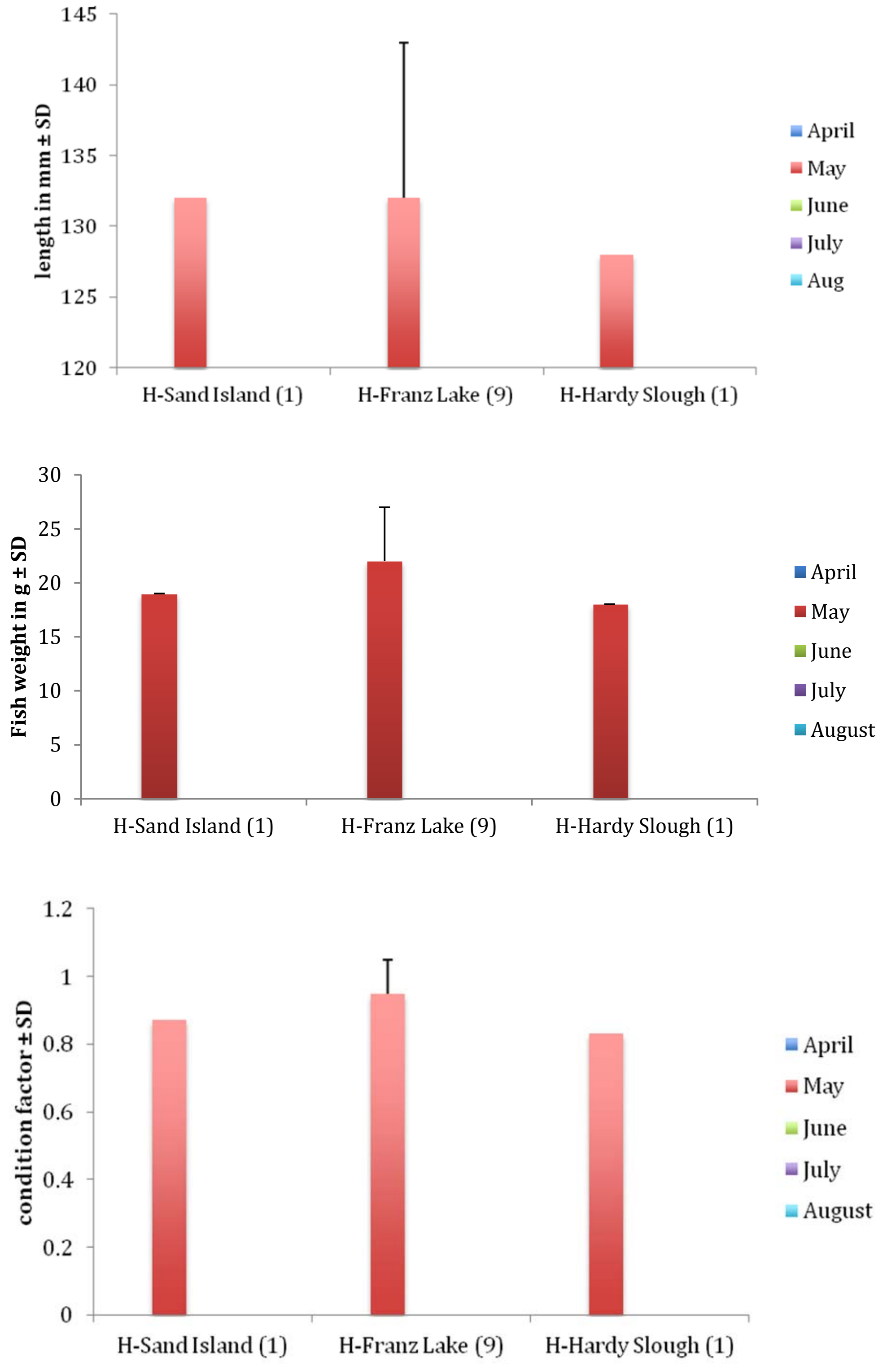


Figure 44. Mean length, weight, and K by site for larger unmarked coho salmon. These fish were found in May only. The number in the parentheses is the number of fish sampled.

For coho within the smaller cohort, length increased significantly over the sampling season from April through July or August, with overall mean size ranging from 42 mm in April to 83 mm in August (Figure 45). Multiple regression analysis indicated that after adjusting for the effect of sampling month, there were significant differences in length among the sampling sites ($p<0.0001$). Fish from Hardy Slough ($p = 0.0341$) and Pierce Island ($p = 0.0317$) were larger and fish from Lord/Walker Island ($p = 0.0095$) were smaller than the overall average. Distance from the main channel and rkm did not have significant effects on fish length ($p = 0.3759$ and $p = 0.1176$).

Coho weight (Figure 45) showed a similar relationship, with both month and site having significant effects ($p<0.0001$ and $p = 0.0008$). Adjusting for month, weight was significantly higher at Hardy Slough ($p = 0.0240$) and Wallace Island West ($p = 0.0463$) and significantly lower at Lord/Walker Island ($p = 0.0157$) than the overall average for all sites. However, as with length, distance from the main channel and rkm were not significantly related to coho weight ($p = 0.1586$ and $p = 0.2973$). Coho condition (K) (Fig. 30c) was significantly affected by both sites and month ($p = 0.0012$ for both). K was lower in coho from Franz Lake than from other sites ($p = 0.0001$). Unlike length and weight, K was affected by rkm and distance from the main channel. Coho condition tended to increase with decreasing rkm ($p = 0.0368$) and increase with increasing distance from the main channel ($p = 0.0018$).

Seasonal patterns of length and weight were quite different for the marked coho than for the smaller unmarked coho (Figure 46). Marked coho were only present in May and June, and length, weight, and condition were not significantly different for fish collected in these two months ($0.2480<p<0.5793$). The overall mean length was 138±12 mm for May ($n = 90$) and 136±6 mm for June ($n = 7$). For weight, the overall mean was 27±7 for May ($n = 90$) and 24+2 g ($n = 7$) for June. Overall mean values for K were 0.9940±0.822 for May and 0.9578±0.0465 for June.

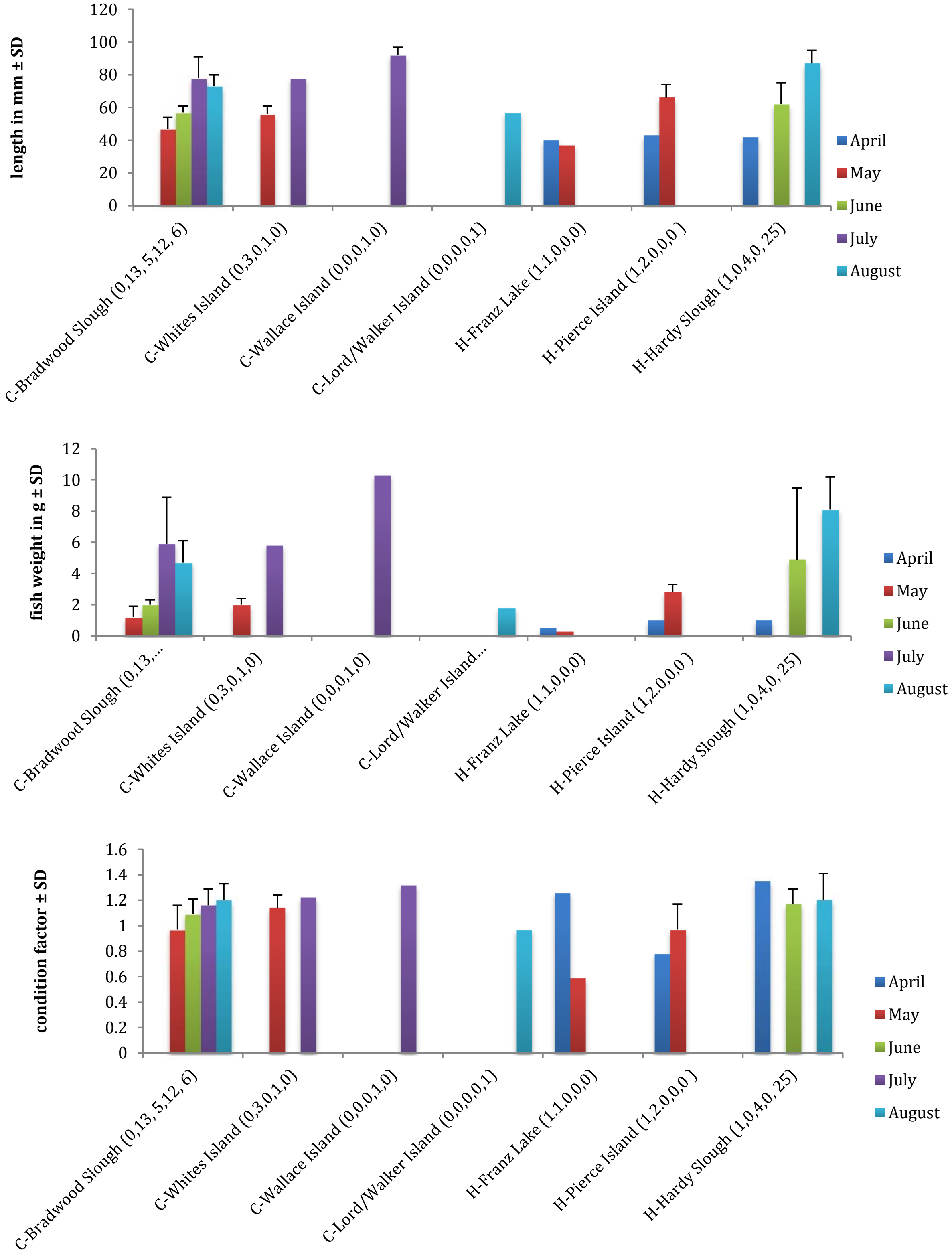


Figure 45. Mean length, weight, and K by month and site for smaller unmarked coho salmon.

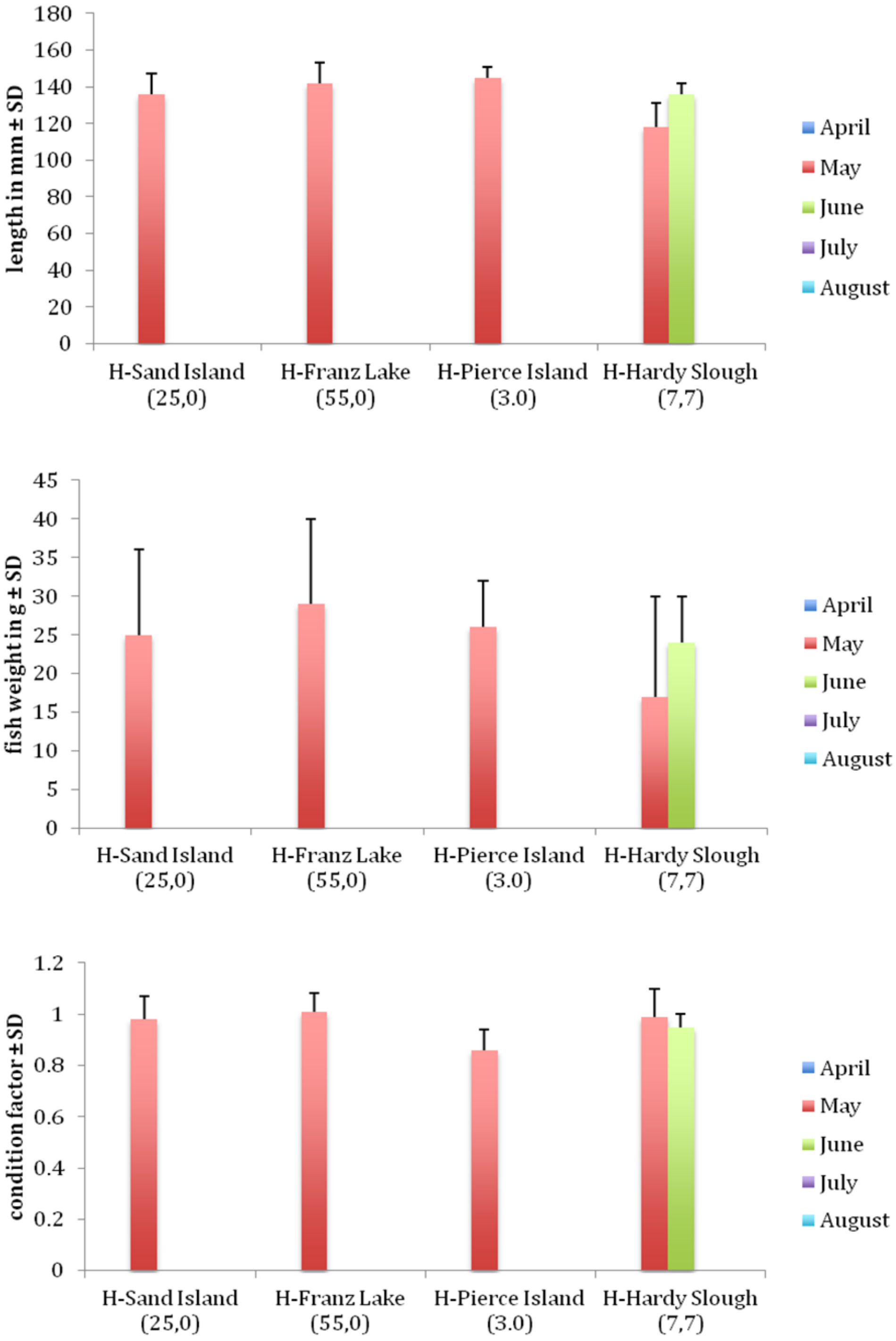


Figure 46. Mean length, weight, and K by month and site for marked coho salmon. Fish were only present in May and June.

For marked coho, after adjusting for effect of sampling month, length tended to increase as rkm increased (*p* = 0.0001), and to decrease as length from the main channel increased (*p* = 0.0007). Weight showed a similar relationship for distance from main channel ($p = 0.0001$) and rkm ($p = 0.0024$). Condition was unrelated to rkm ($p = 0.6260$) or distance to the main channel ($p = 0.4258$).

Chum Salmon. Length, weight, and K by month for unmarked chum salmon for sites in Reaches C, E, F, and H are shown in Figure 47 and

Figure **48**.

Chum salmon did not exhibit the bimodal distribution seen in the marked coho and Chinook, though a small group of somewhat larger fish was evident (Figure 47). Length ranged from 37 to 63 mm, and increased significantly ($p<0.0001$) over the short season when chum were present, from a mean of 44 mm in April to 55 mm in May (

Figure **48**). Multiple regression analysis indicated that after adjusting for the effect of sampling month, there were no significant differences in length among the sampling sites ($p = 0.1267$), though chum from Ryan Island tended to be larger ($p = 0.0528$) and those from Bradwood Slough smaller ($p = 0.0861$;

Figure **48**). Chum weight also increased significantly from April to May ($p<0.0001$), from a mean of 0.74 g in April to 1.2 g in May (

Figure **48**). In contrast to length, after adjusting for the effect of sampling month, significant differences in chum weight among the sampling sites were observed ($p = 0.0450$). Chum weight was significantly lower than the overall average at Lord/Walker Island ($p = 0.0046$). For chum, after adjusting for effect of sampling month, neither rkm nor distance from the main channel had a significant relationship to fish length ($0.1137<p<0.3489$), although fish tended to be smaller at the Reach H sites. The same was true for weight ($0.285<p<0.5347$).

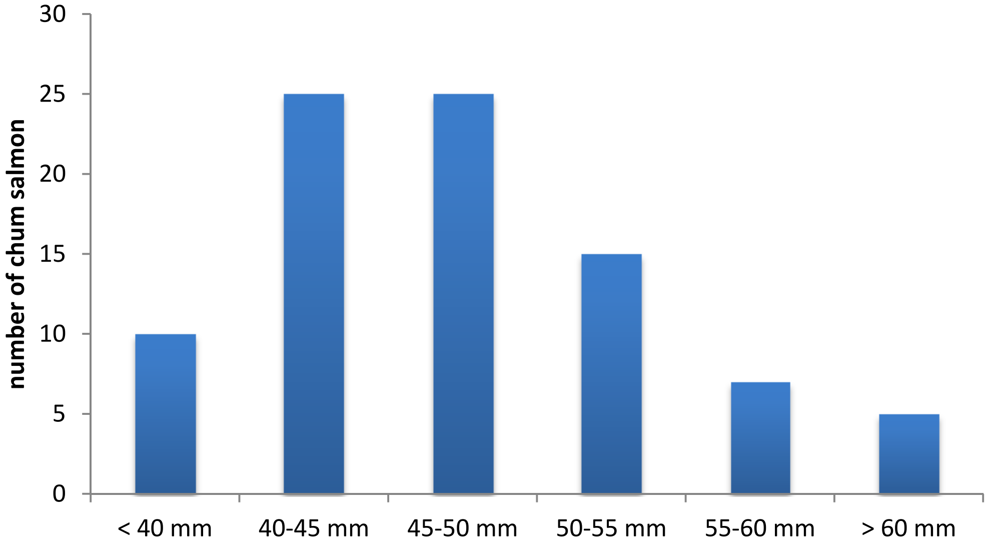


Figure 47. Size distribution for chum salmon at the EMP sampling sites.

Chum condition (

Figure **48**) decreased from an average value of 0.85 in April to 0.69 in May (p = 0.03732). After adjusting for the effect of sampling month, site still had a significant effect on K (p<0.0001). Chum condition was significantly lower than the overall average at Campbell Slough in Reach F, and LordWalker Island and Ryan Island in Reach C. It was higher than the overall average at Hardy Slough, Franz Lake, and Sand Island in Reach H. Condition increased as rkm increased (p = 0.0001), but was not related to distance from the main channel (p = 0.6431)

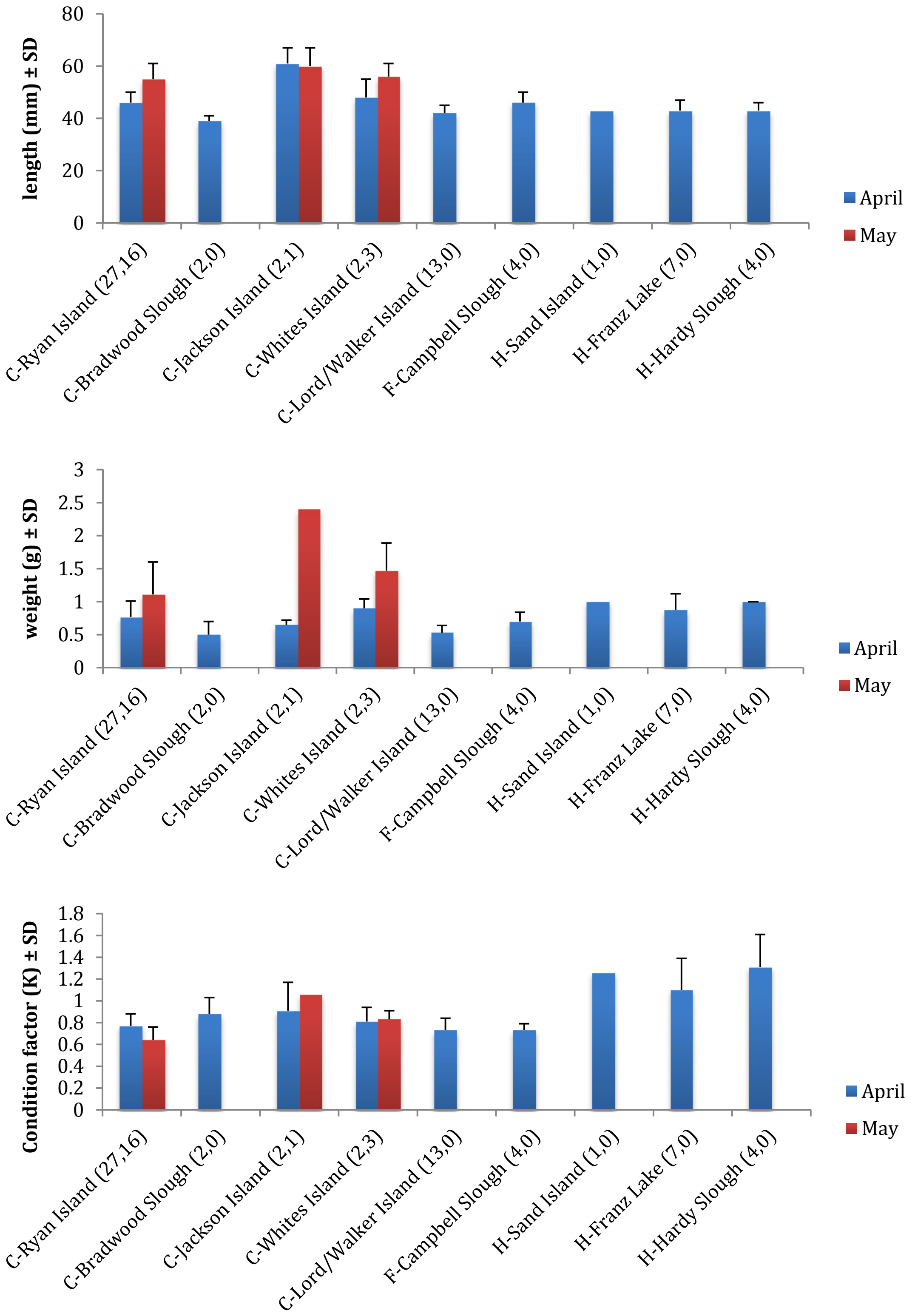


Figure 48. Mean length, weight, and K by month and site for chum salmon.

Steelhead Trout. There were too few steelhead captured to perform an analysis of effects of site, month, rkm, or distance from the main channel on length, weight, or K.

Otolith Analyses for Growth Rate Determination

As part of the EMP salmon sampling, otoliths were collected from juvenile fall Chinook salmon from Bradwood Slough, Ryan Island, Jackson Island, Whites Island, Wallace Island West, and Lord/Walker Island in Reach C, from Campbell Slough in Reach F, and from Sand Island, Franz Lake, Hardy Slough, and Pierce Island in Reach H, for estimation of average daily growth rates (Figure 49). Growth rates were similar at Franz Lake and Campbell Slough (0.56–0.61 mm per day for the last seven days before sampling), but significantly lower at Bradwood Slough, Ryan Island, Whites Island, and Wallace Island West (0.46–0.48 mm per day for the last seven days before sampling; *p*<0.05). The growth rate at Lord/Walker Island was similar to the other Reach C sites (47 mm per day) but not statistically different from Campbell Slough or Franz Lake (*p*>0.05), presumably due to smaller sample size. At Reach H sites, growth rates tended to be slightly lower, but not significantly different (*p*>0.05), than those in fish from Franz Lake, ranging from 48 mm per day in fish from Hardy Slough to 56 mm per day in fish from Pierce Island. The size range of fish analyzed was similar at all sites, so would not bias the analyses.

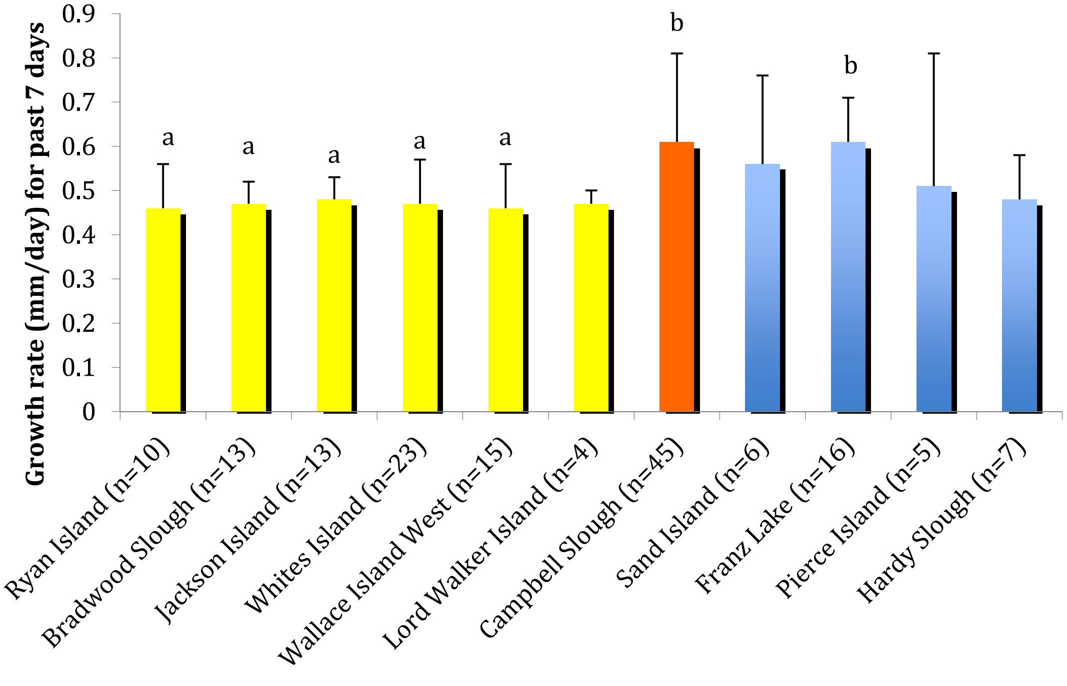


Figure 49. Growth rates in mm per day for the last seven days growth before sampling for Chinook salmon from EMP sites. Growth rates were significantly lower at five of the six Reach C sites than at Campbell Slough or Franz Lake.

Significant differences were also observed among fish from different genetic stocks (Figure 50). These differences were present regardless of the time interval tested (last seven days, F6, 249=3.6, *p*<0.05; last 14 days, F6, 249=3.7, *p*<0.05; last 21 days, F6, 249=4.0, *p*<0.001). Bonferroni post-hoc tests determined that growth rates of Spring Creek Group fall Chinook were significantly greater than West Cascades fall

Chinook for all intervals (i.e., last seven, 14, and 21 days of growth), and Willamette River spring Chinook growth rates were significantly greater than West Cascades fall Chinook, although only for the last seven and 14 days.

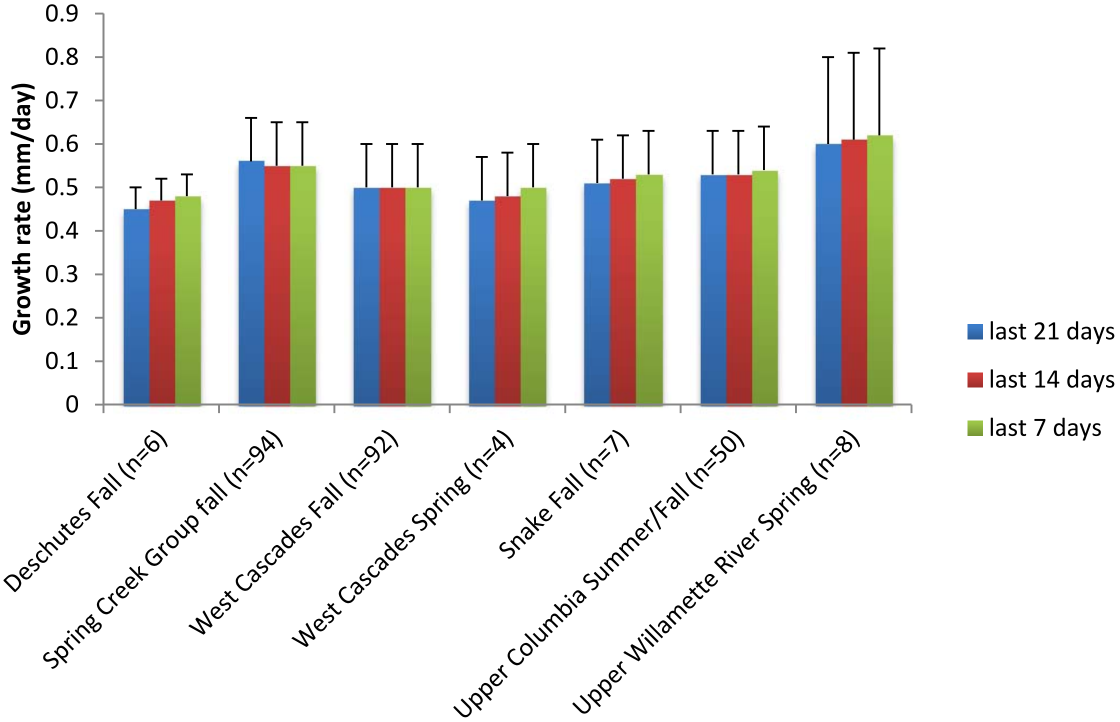


Figure 50. Mean growth rate (mm/day) (last seven, 14, and 21 days) for each of seven genetic stocks. Bonferroni post-hoc tests determined that growth rates of Spring Creek Group fall Chinook were significantly greater than West Cascades fall Chinook for all intervals (i.e., last seven, 14, and 21 days of growth), and growth rates of Upper Willamette River spring Chinook were significantly greater than West Cascades fall Chinook for the last 14 and seven days. Error bars represent standard deviation.

Finally, marked fish grew significantly faster than unmarked fish for all intervals (last seven days, F1, 254 = 48.6, *p*<0.001; last 14 days, F1, 254 = 59.6, *p*<0.001; last 21 days, F1, 254=64.4, *p*<0.001; [Figure 51]).

These results suggest that the low growth rates of juvenile Chinook salmon from Reach C were due to their genetic stock, which was primarily West Cascades fall Chinook, and their origin, which was primarily unmarked, presumably wild fish.

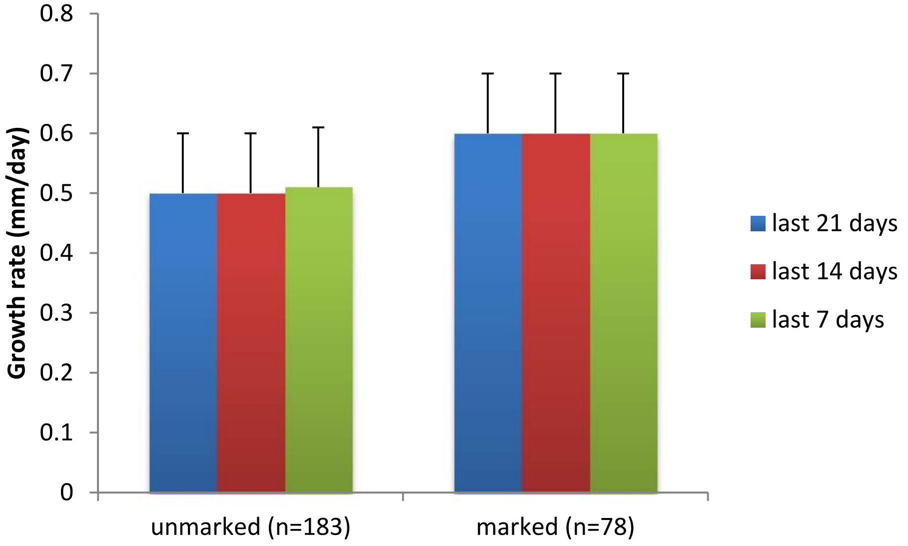


Figure 51. Mean growth rate (mm/day) for unmarked (*n* = 183) and marked (*n* = 78) fish, for each interval (last seven, 14, and 21 days of growth), when all sites are pooled. Error bars represent standard deviation. Bonferroni post-hoc tests determined that growth rates of marked fish were significantly greater ($p<0.05$) than unmarked fish for all intervals (i.e., last seven, 14, and 21 days of growth).

Chinook Salmon Lipid Content and Classes

Overall, the lipid content was not significantly different between marked and unmarked Chinook, although marked fish tended to have higher lipid content. The mean lipid content was 1.4% in marked fish ($n = 87$) and 1.2% in unmarked fish ($n = 98$), ANOVA, $p = 0.06$. However, the two groups of fish exhibited different patterns among seasons and sites. The unmarked Chinook (Figure 52), showed no clear pattern in lipid content with rkm or reach. Mean lipid content at the sampling sites ranged from 0.87% to 1.4%, with the highest lipid content in Chinook from Campbell Slough and the lowest in Chinook from Franz Lake. However, the only statistically significant differences detected were between Campbell Slough and Bradwood Slough. There was no relationship between rkm and lipid content ($p = 0.7857$, $R^2 = 0.0226$, $n = 36$). However, lipid content in unmarked fish was positively correlated with distance from the main channel ($p \leq 0.0001$, $R^2 = 0.38$, $n = 36$).

For marked Chinook (Figure 52), lipid content ranged from 0.87% at Whites Island to 2.1% at Franz Lake. Lipid content tended to decline from Reach H to Reach C, with lipid levels that had significantly lower *p* value in fish from Whites Island than in fish from Franz Lake. There was a significant positive correlation between lipid content and rkm for marked fish ($p \leq 0.0001$, $R^2 = 0.22$, $n = 63$). Also, lipid content was negatively correlated with distance from the main channel, so as distance from the main channel decreased, lipid content increased ($p \leq 0.0001$, $R^2 = 0.22$, $n = 63$).

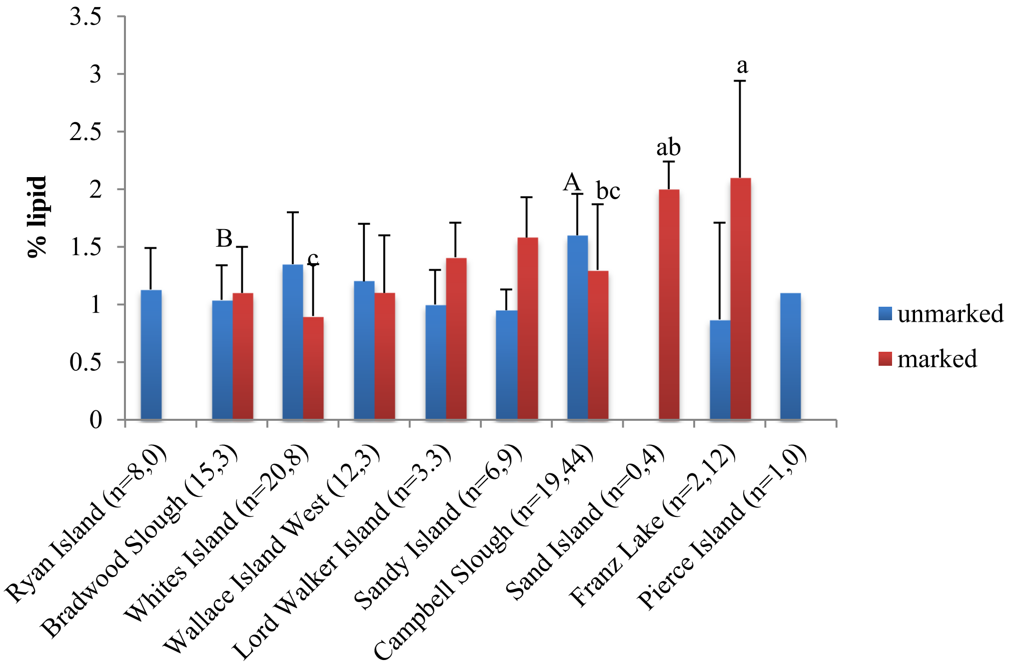


Figure 52. Mean lipid content (±SD) of marked and unmarked subyearling Chinook salmon from the EMP monitoring sites. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, *p*<0.05); upper case letters indicate comparisons among unmarked fish by site, while lower case letters indicate comparisons among marked fish by site.

Among unmarked fish, there was no seasonal trend in lipid content, with values between 1.1% and 1.4% over the whole sampling season (Figure 53). However, among the marked fish, there was a clear seasonal trend in lipid content, declining from a high of 2.3% in April to 0.75% in August (Figure 53).

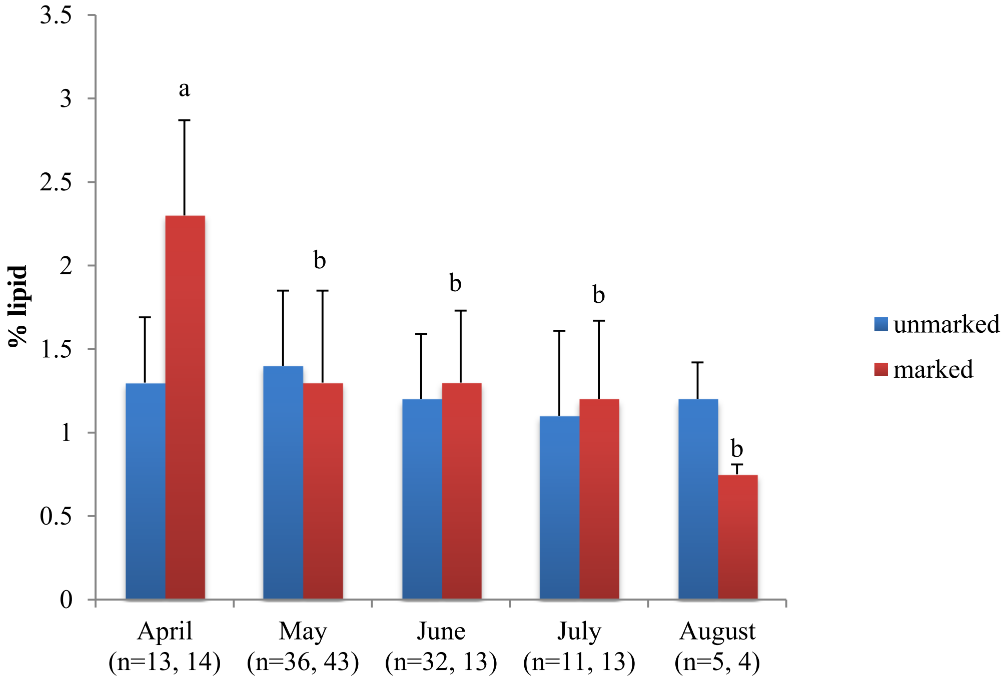


Figure 53. Mean lipid content (±SD) of marked and unmarked subyearling Chinook salmon by month of capture. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, $p<0.05$). Among marked salmon, lipid content of fish collected in April was significantly higher than for any those collected in other months, but among unmarked salmon, there were no significant differences by month.

Contaminants in Chinook Salmon

POPs in Bodies. The major contaminants in salmon from the Reach H sites (Sand Island, Franz Lake, and Pierce Island) were DDTs, although low levels of PBDEs and PCBs were also detected (Figure 54). Of all the salmon, those from the Reach H sites had the lowest concentrations of all three classes of contaminants. Concentrations of DDTs ranged from 720 to 1,100 ng/g lipid, PBDEs from 360 to 390 ng/g lipid, and PCBs from 41 to 150 ng/g lipid. Concentrations of DDTs in salmon from Reach H sites were comparable to those found in most samples collected in Reach C, but concentrations of PCBs and PBDEs were generally significantly lower than levels observed in fish from other sites (Tukey's multiple range test, $p<0.05$).

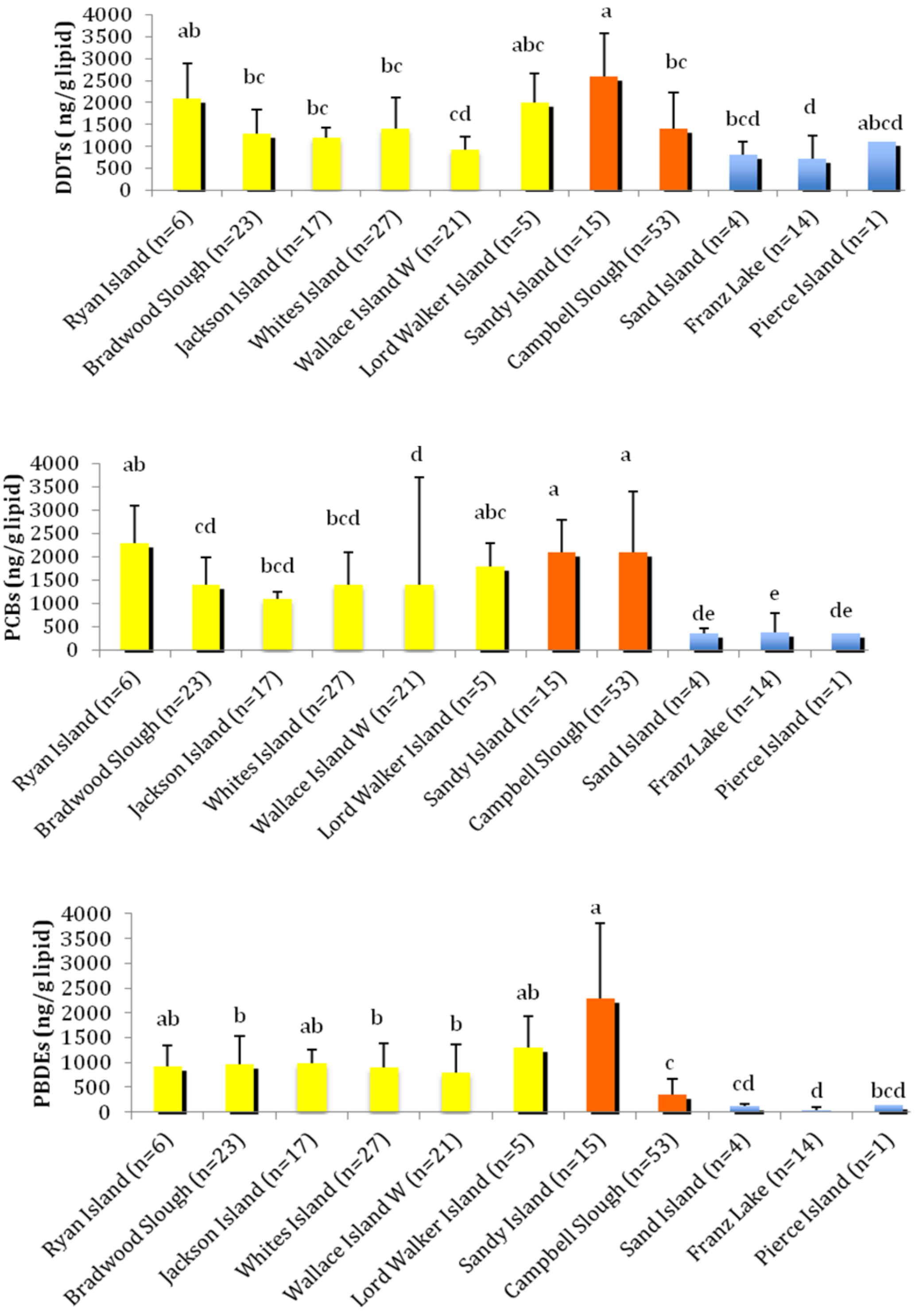


Figure 54. Concentrations of DDTs, PCBs, and PBDEs (ng/g lipid) in bodies of juvenile Chinook salmon from the EMP sampling sites. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, *p*<0.05).

In salmon from Campbell Slough, Sandy Island, and the Reach C sites, mean concentrations of DDTs ranged from 920 ng/g lipid at Wallace Island West to 2,600 ng/g lipid at Sandy Island. Concentrations of DDTs were significantly higher at Sandy Island than at Campbell Slough and all Reach C sites with the exceptions of Ryan Island and LordWalker Island (Tukey's multiple range test, $p<0.05$).

Mean concentrations of PCBs ranged from 1,100 ng/g lipid at Jackson Island to 2,100 ng/g lipid at Sandy Island and Campbell Slough. Concentrations of PCBs were significantly higher at Campbell Slough and Sandy Island than at the Reach C sites, again with the exceptions of Ryan and Lord-Walker Islands (Tukey's multiple range test, $p<0.05$).

PBDEs were found at the highest concentrations at Sandy Island, where the mean PBDE level was 2,300 ng/g lipid. The mean concentration in Chinook from Campbell Slough was quite low (360 ng/g lipid), comparable to levels seen in Reach H Chinook. At the Reach C sites, mean concentrations of PBDEs ranged from 800 ng/g lipid at Wallace Island West to 1,300 ng/g lipid at Lord-Walker Island. Concentrations of PBDEs at all Reach C sites and Sandy Island were significantly higher than those at Campbell Slough (Tukey's multiple range test, $p<0.05$).

In Chinook from all sites, DDTs in all samples were below the estimated effect threshold of 6,000 ng/g lipid (Beckvar et al. 2005 as modified for lipid content in Johnson et al. 2007). The highest concentration of DDTs observed was 4,700 ng/g lipid in a sample collected from Sandy Island in 2007. However, in 15% of samples, concentrations of PCBs were above the estimated effect threshold of 2,400 ng/g lipid proposed by Meador et al. (2002), and 28% of samples were above the 940 ng/g lipid level associated with immunosuppression in juvenile salmon in a study by Arkoosh et al. (2010; Figure 55). Samples with elevated PCB concentrations were most prevalent at Campbell Slough, Sandy Island, and Lord-Walker Island, sites between rkm 149 and 99, but a fairly high proportion of samples with elevated PCBs was also found at Ryan Island at rkm 61. Samples with elevated concentrations of PBDEs were found at Sandy Island and throughout Reach C.

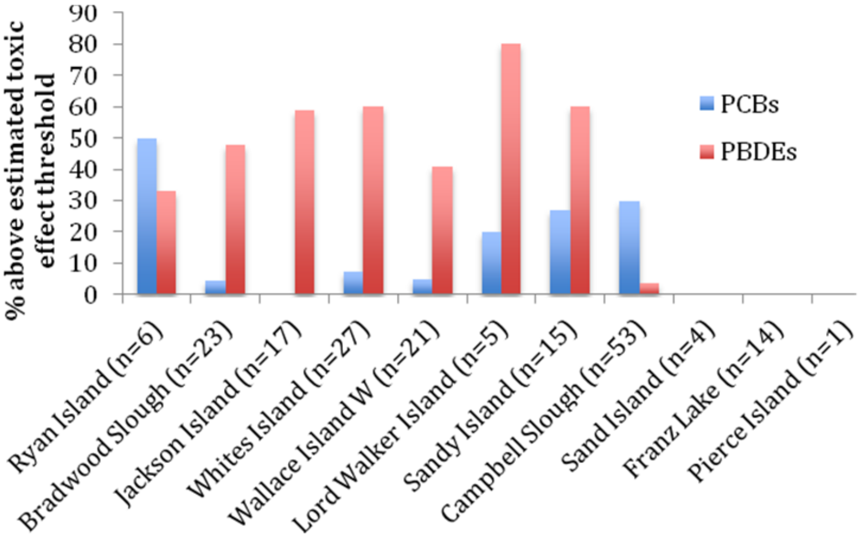
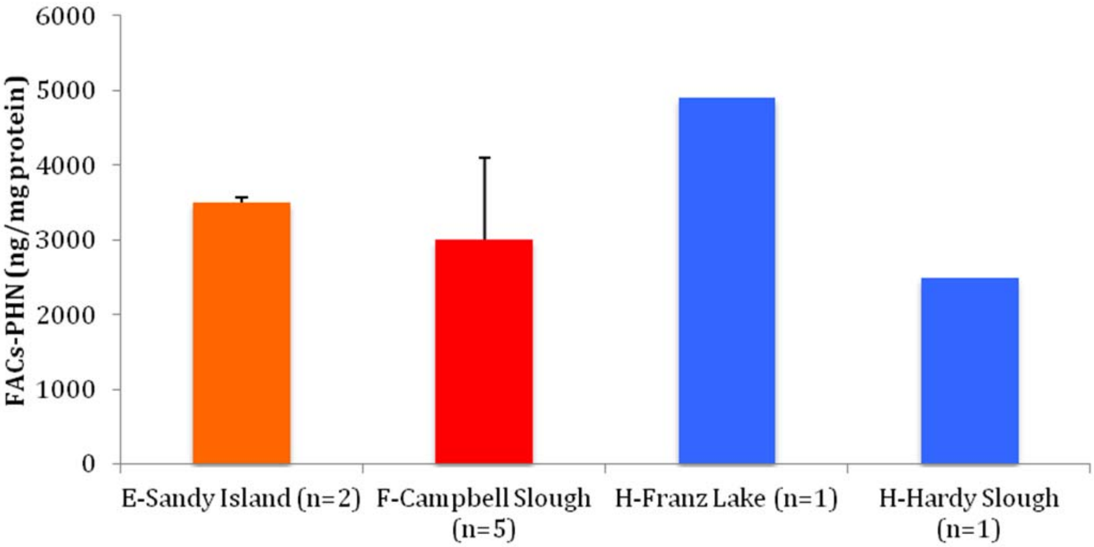


Figure 55. Percentage of Chinook body samples with PCB and PBDE concentrations exceeding estimated toxic effect thresholds. No samples with DDT concentrations above estimated health threshold were found.

Metabolites of PAHs in Bile. Only a limited number of bile samples were available for analysis of PAH exposure from the EMP sites because of permit limitations and difficulty collecting sufficient numbers of juvenile salmon for composite samples. Available data for Hardy Slough and Franz Lake in Reach H, Campbell Slough in Reach F, and Sandy Island in Reach E are shown in Figure 56. No samples could be collected from the Reach C sites.



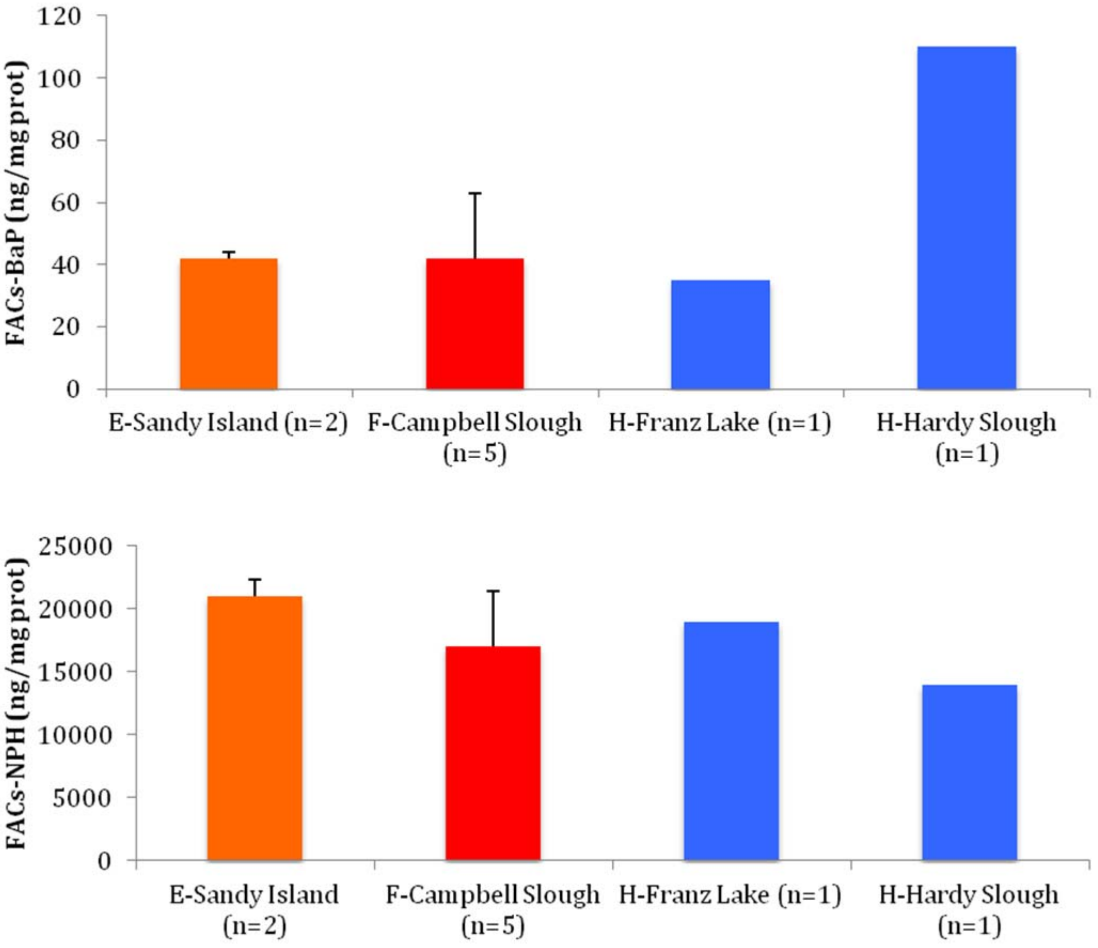


Figure 56. Concentrations (ng/mg bile protein) of PAH metabolites in bile of juvenile Chinook salmon, measured as aromatic compounds fluorescing at phenanthrene (FACs-PHN), benzo[a]pyrene (FACs-BaP), and naphthalene (FACs-NPH) wavelengths. *n* = number of composite samples, each containing bile from 10 to 20 fish.

Levels of FACs-NPH in bile samples were similar at all four sites. However, levels of FACs-BaP were several times higher in fish from Hardy Slough than in fish from the other three sites (110 ng/mg protein versus 35 to 42 ng/mg protein), while the level of FACs-PHN in bile from Franz Lake fish was 1.5 to 2 times as high as the level of FACs-PHN in bile samples from Hardy Slough, Campbell Slough, and Sandy Island (4,900 ng/mg protein versus 2,500–3,500 ng/mg protein). However, in part because of the small number of samples available for analysis, no statistically significant differences were observed in bile metabolite levels by sampling year or site.

3.1.3 Fish Prey

Salmon Diet Samples. 16,799 individual prey items were identified and counted in 325 Chinook salmon stomachs collected from many of the monitoring sites over several years (Table 13 and Table 14). On average, a typical juvenile Chinook salmon had 51.7 prey items in its stomach at the time of capture. The number of prey items ranged from zero (two fish stomachs were empty) to 495 invertebrate prey items. There were significant differences in the mean number of prey items per stomach between fish caught during individual sampling events across the sites (site/month/year, *n* = 26, PERMANOVA *p* = 0.001), but no significant differences when analyzed by site with year nested (PERMANOVA *p* for site and year both >0.05), reflecting that the variation was often as great within a site during a sampling period (month and year) as it was across sites and sampling periods (Figure 57).

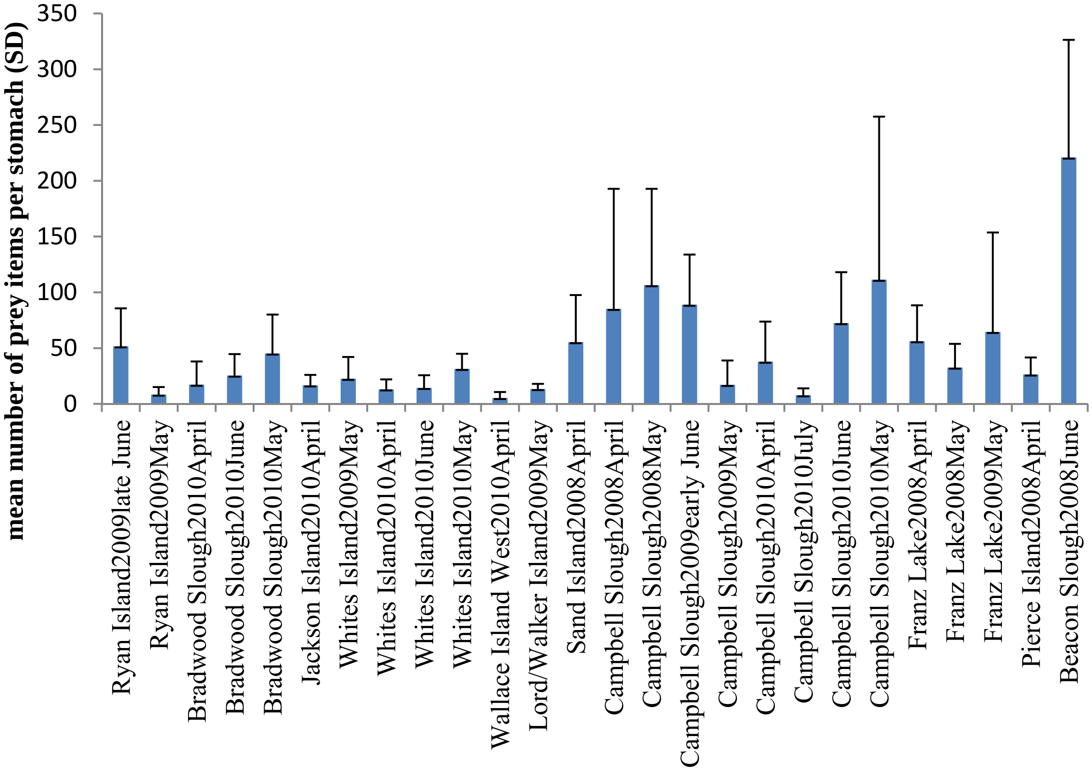


Figure 57. The mean (SD) number of prey items per juvenile Chinook stomach per sampling event. See Table 13 for sample sizes.

Table 13. Number of Chinook diet samples collected and processed from each site and year.*

Site	2008	2009	2010*	Total
Ryan Island		19		19
Bradwood Slough			37	37
Jackson Island			20	20
Whites Island		10	49	59
Wallace Island West			7	7
Lord-Walker Island		6		6
Campbell Slough	25	19	69	113
Sand Island	12			12
Franz Lake	22	8		30

Table 13. Number of Chinook diet samples collected and processed from each site and year.*

Site	2008	2009	2010*	Total
Pierce Island	9			9
Hardy Slough	13			13
Grand Total	81	62	182	325

*There are an additional 95 stomach samples from 2010 and 117 from 2011 that are currently being processed.

While we identified 27 different orders of invertebrates and fish across all of the diets that we examined, the most striking finding was the consistent dominance of aquatic Diptera in the diets of juvenile Chinook salmon. This was true for individual fish as well as for all fish sampled on a given date at each site; for instance, 96% of all the juvenile Chinook salmon diets examined had at least one fly (Diptera) larvae or pupae in their stomachs. Dipterans were also typically the most abundant prey taxa in the diets across sites and sampling periods. When including all fish averaged by site and year, the mean proportion of Dipterans by count in the diets of juvenile Chinook salmon was 78%, ranging from a low of 57% in Chinook caught at Whites Island in 2010 to a high of 98% in Chinook caught at Hardy Slough in 2008 (Figure 58). This pattern was remarkably consistent across sites and within sites across sampling periods (Figure 58).

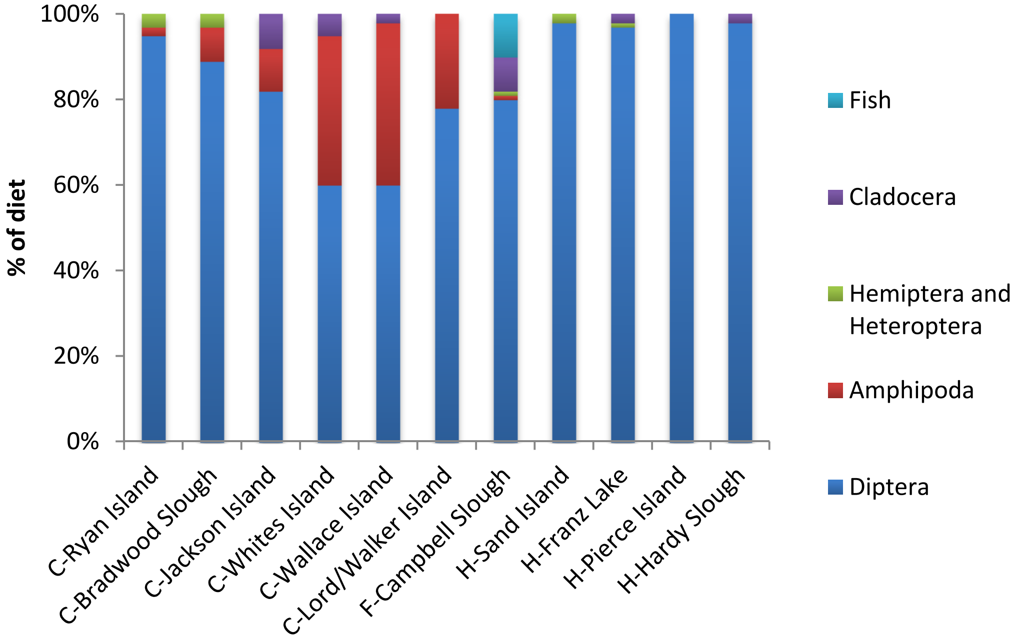


Figure 58. The mean proportion of prey taxa consumed by juvenile Chinook salmon, averaged over sampling periods and by site. For those sites sampled more than one year (Campbell Slough, Franz Lake, and Whites Island) data are a composite of all years of sampling. On average, these five prey taxa compose more than 90% of all prey consumed. Data have been scaled here to total 100%.

Table 14. Number of juvenile Chinook diet samples analyzed from sites sampled in 2008, 2009, and 2010.

	2008			2009			2010			
Site	**April**	**May**	**June**	**May**	**Early June**	**Late June**	**April**	**May**	**June**	**July**
Ryan Island				9		10				
Bradwood Slough							10	17	10	
Jackson Island							20			
Whites Island				10			16	14	19	
Wallace Island West							7			
Lord/Walker Island				6						
Sand Island	12									
Campbell Slough	6	19		10	9		12	24	18	15
Franz Lake	15	7		8						
Pierce Island	9									
Hardy Slough			13							

Figure 59 illustrates how the diets at one site, Campbell Slough, were consistently dominated by Dipterans, even though the mean number of total prey items per stomach varied widely within and across sampling periods. The mean number of prey items consumed over these sampling periods ranged from 7.7 to 111.3, and often the standard deviation was greater than the mean (Figure 59). Dipterans typically made up more than 70% of the prey consumed; exceptions to this include May 2010 when some fish consumed large numbers of Cladocerans and in July 2010 when 14 of the 15 Chinook sampled had consumed three-spine stickleback fish and relatively few invertebrates. The vast majority of the Dipterans at Campbell Slough as well as at other sites were aquatic midge (Chironomidae) larvae and pupae.

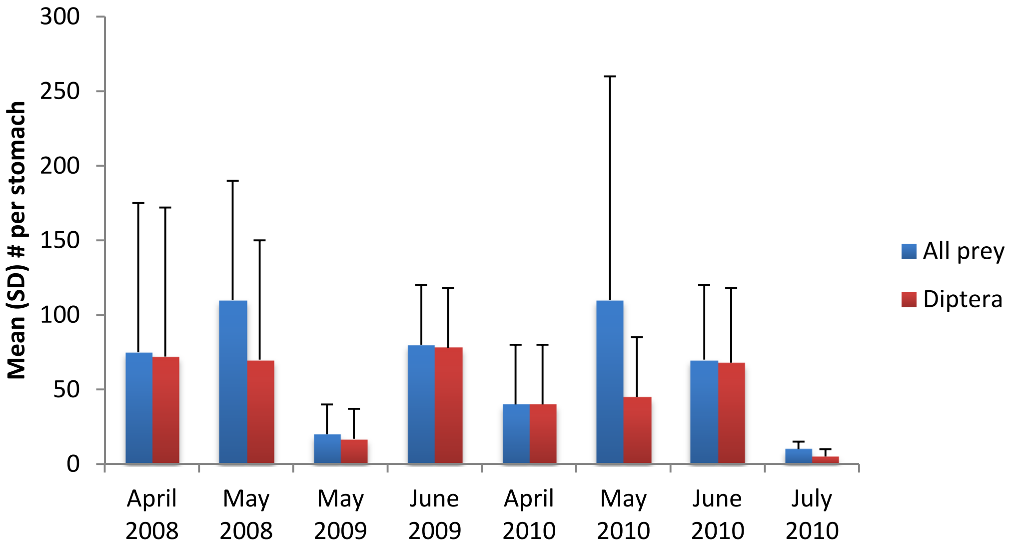


Figure 59. The mean (SD) number of all prey and Diptera consumed by Chinook salmon at Campbell Slough across sampling periods.

Dipterans were consistently abundant in Chinook diets across sites and sampling periods, but Amphipods, and specifically *Americorophium* sp. were also often present in diets from sites in Reach C. In samples from Whites Island, Jackson Island, Wallace Island West, and Lord-Walker Island, *Americorophium* sp. composed more than 10% of the mean number of diet items. Preliminary estimates indicate that when these amphipods are available, they may be quite important energetically because of their relatively large size. Individual *Americorophium* sp. were on average approximately 10 times the biomass of an average Chironomidae larva or pupa and 33 times the biomass of the average Cladoceran (preliminary blotted wet weight estimates).

Prey Availability. Invertebrate samples were collected at all sites when Chinook salmon were caught and stomachs sampled (Table 13). The most common invertebrate taxa observed in both open-water and emergent vegetation tows were Cyclopoid and Calanoid copepods, Cladocerans, and Dipterans (Figure 60). In certain tows and at certain sites, Coleoptera, Hemiptera, and Heteroptera were also abundant. The copepods were most consistently found as a high proportion of prey at the Reach H sites and Campbell Slough, but were not a prominent part of the Reach C samples. In Reach C, Dipterans were often most abundant, but terrestrial and aquatic Heteroptera and Hemiptera were also often abundant. The relative abundances of invertebrate taxa varied across sites and across years (Figure 60), but generally the composition was more similar within sites and across years than across sites overall. For example, the relative proportions of the dominant taxa at Campbell Slough and Franz Lake were similar across years when these sites were sampled (Figure 60). Samples were collected from April through July in some years (Table 14), but typically the majority of fish were caught in May and June and thus, those months have the largest sample sizes for invertebrates.

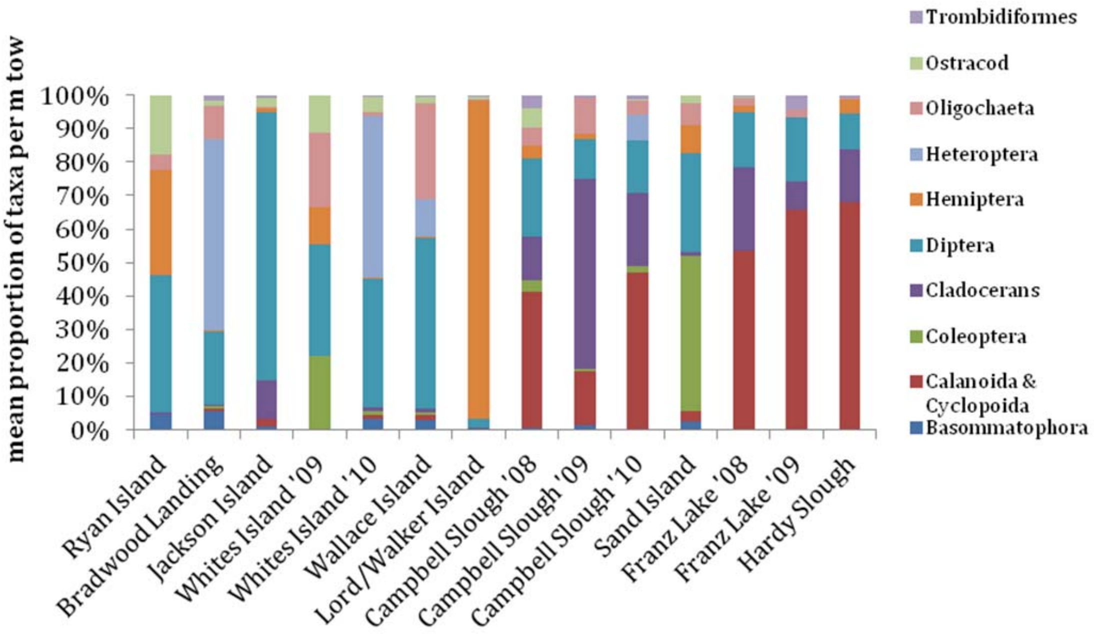


Figure 60. The mean proportions of invertebrate taxa in neuston tows collected from both emergent vegetation habitats and open-water habitats at sites and over multiple years.

The abundance of prey in the emergent vegetation and open-water habitats was highly variable within and across sites, and over the various sampling periods (Figure 61, Figure 62, and Figure 63). On average (and excluding the exceptionally high counts found in the Hardy Slough samples), there were 20.0 invertebrates per meter of tow when averaging both emergent vegetation and open-water tows (Figure 61). The most interesting and reliable pattern we found with the tow samples was that of all the potential invertebrate prey caught in the tows, approximately 94% of the invertebrates were caught in the emergent vegetation tows (e.g., 18.9 invertebrates per meter were from the emergent vegetation tows, while only 1.1 invertebrates per meter were from the open-water tows). This pattern was very consistent within and across sites and across sampling periods; for example, invertebrates from emergent vegetation tows were generally 47 times more abundant per meter in emergent vegetation tows compared to open-water tows (range was from 1.1 times at Franz Lake in 2008 to 181 times at Hardy Slough in 2008). Figure 63 illustrates the differences in the abundance of invertebrates per meter in the emergent vegetation tows compared to the open-water tows. It also illustrates that the open-water tow densities within each habitat were generally much more variable within and across sites and sampling periods as compared to the emergent vegetation tows. Although there were increasing densities in the emergent vegetation tows with time at some sites, there were no clear trends in abundance over all sites across these months.

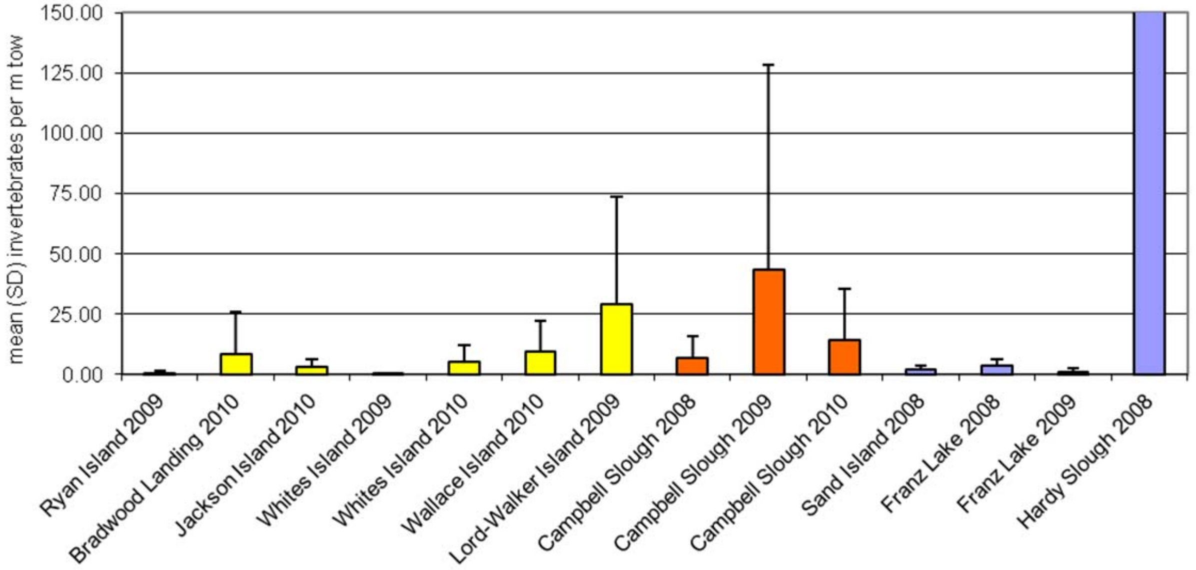


Figure 61. The mean (SD) number of invertebrates per meter tow, including both tows in the margin habitats with emergent vegetation as well as the open-water habitats. The Hardy Slough mean (SD) number of invertebrates per meter was 446.95 (717.18), and was not shown to scale here. Sites are colored by reach (yellow = Reach C; orange = Reach F; blue = Reach H) and ordered by river distance.

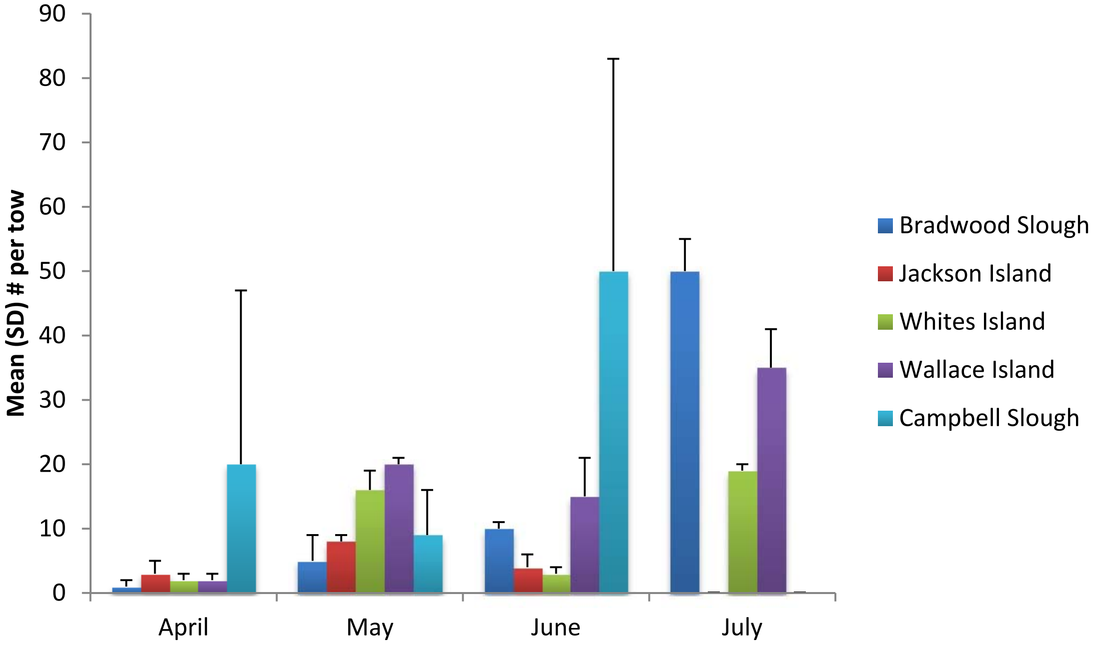


Figure 62. The mean (SD) abundance of invertebrates in the tows collected from margin habitats with emergent vegetation from Bradwood Slough, Jackson Island, Whites Island, Wallace Island, and Campbell Slough across various months in 2010. Note that no emergent vegetation tows were collected from Jackson Island or Campbell Slough in July and Ryan Island was not sampled in 2010.

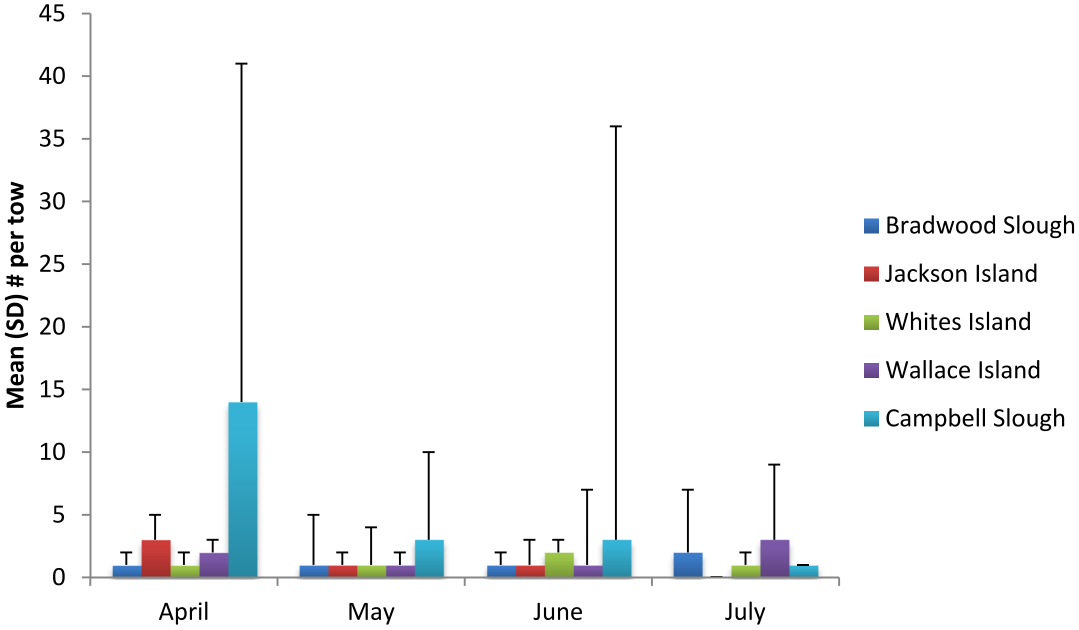


Figure 63. The mean (SD) abundance of invertebrates in the tows collected from open-water habitat from Bradwood Landing, Jackson Island, Whites Island, Wallace Island, and Campbell Slough across various months in 2010. Note that no open-water tows were collected from Jackson Island in July and Ryan Island was not sampled in 2010.

Selectivity Analysis. Ivlev's prey electivity values, which were calculated for the most abundant taxa in both the diets (Diptera, Amphipoda, Ephemeroptera) and the tows (Diptera, Calanoid and Cyclopoid copepods, Cladocerans, Oligochaetes, and Hemiptera/Heteroptera) indicate a strong and consistent preference for Dipteran prey by juvenile Chinook salmon (Figure 64). Strong preference for amphipods was also observed at most sites. Although amphipods were rarely abundant in either the tows or the diets, the disproportionate selection of them when they were available indicates they may be a particularly important prey resource. Mayflies (Ephemeroptera) were also selected at a higher rate than expected at some sites and avoided at others (Figure 64). Many of the invertebrates that were highly abundant were not consumed at all or were consumed at a lower rate than expected given their abundance. These include the Calanoid and Cyclopoid copepods, Cladocerans, Oligochaetes, Hemiptera and Heteroptera. The small relative size of the copepods and Cladocerans probably contributed to their low electivity values. Prey selectivity was also calculated with the Strauss Selectivity Index and the Relativized Electivity Index (Lechowicz 1982) for comparison, and all analyses were similar in demonstrating a strong and consistent juvenile Chinook preference for Diptera and Amphipods and an avoidance of other taxa across sites and sampling periods.

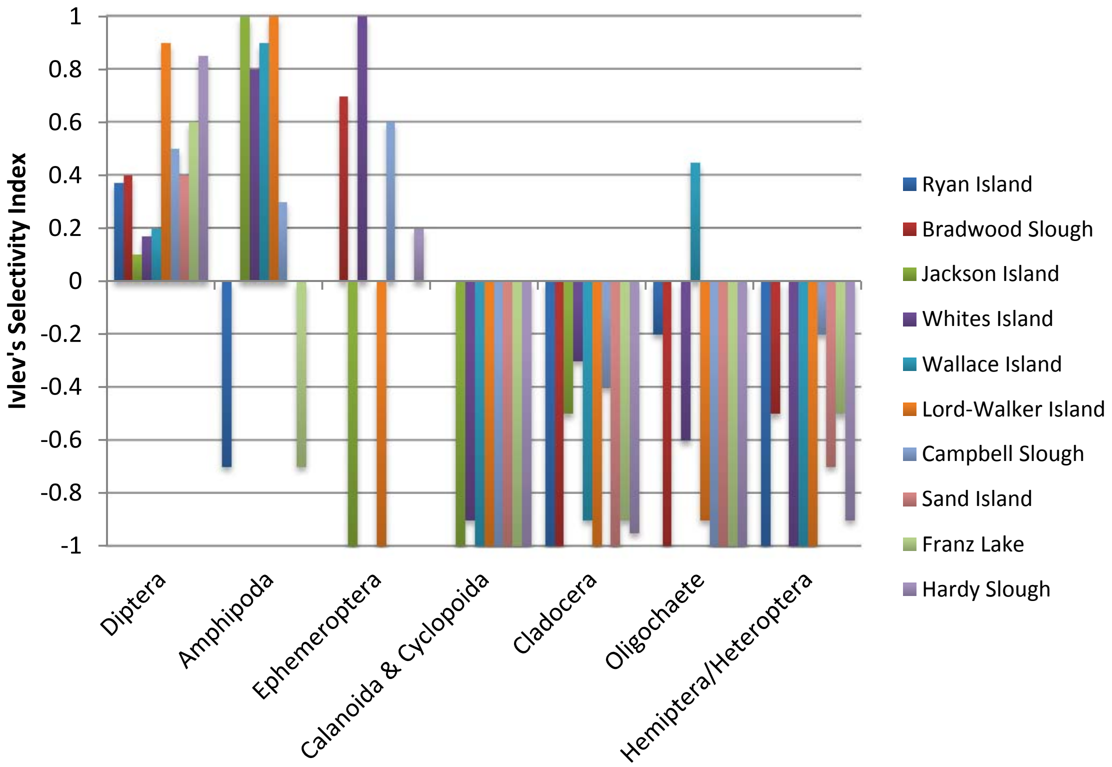


Figure 64. Mean Ivlev's prey electivity values calculated for each site and for seven groups of taxa. Values for each discrete sampling period (e.g., June 2010) were calculated for each site and sampling period; the values presented are the means across each site for all sampling periods.

Although the abundance and composition of the invertebrate prey community varied across sites and sampling periods, juvenile Chinook salmon were remarkably predictable in their preference and avoidance of particular prey taxa across the sites and sampling periods covered in these analyses (Table 15). Dipterans were selected at a greater proportion than they were available during all sampling periods except two (Whites Island in April and June 2010), and copepods were never preferred.

Table 15. Mean prey electivity values (Ivlev's electivity index) for seven most abundant invertebrate taxa for individual sampling periods at all sites for which there are corresponding diet and tow samples.

Site	Year	Month	Diptera	Amphipoda	Ephemeroptera	Calanoid and Cyclopoid copepods	Cladocerans	Oligochaete	Hemiptera/ Heteroptera
Ryan Island	2009	May	0.36	–0.70			–1.00	–0.17	–1.00
Bradwood Landing	2010	April	0.64	0.76	–0.21			–1.00	–0.29
	2010	May	0.22	1.00	1.00			–1.00	–0.44
	2010	June	0.31	1.00	1.00		–1.00	–1.00	–0.74

Table 15. Mean prey electivity values (Ivlev's electivity index) for seven most abundant invertebrate taxa for individual sampling periods at all sites for which there are corresponding diet and tow samples.

Site	Year	Month	Diptera	Amphipoda	Ephemeroptera	Calanoid and Cyclopoid copepods	Cladocerans	Oligochaete	Hemiptera/ Heteroptera
Jackson Island	2010	April	0.08	1.00	–1.00	–1.00	–0.47	–0.05	
Whites Island	2009	May	0.52	0.62				–0.84	–1.00
	2010	April	–0.04	1.00		–0.82	0.93	–1.00	–1.00
	2010	May	0.22	0.97	1.00	–1.00	–0.96	–0.19	–1.00
	2010	June	–0.02	0.97	1.00	–1.00	–1.00	–0.36	–1.00
Wallace Island West	2010	April	0.19	0.95		–1.00	–0.89		–1.00
Lord-Walker Island	2009	May	0.92	0.99	–1.00	–1.00	–1.00	0.43	–1.00
Campbell Slough	2008	April	0.58			–1.00	–0.35	–1.00	
	2008	May	0.50	–0.94	0.81	–1.00	–0.03	–1.00	0.21
	2009	May	0.38	1.00		–1.00	–1.00	–1.00	1.00
	2009	June	0.53			–1.00	–0.98	–1.00	–0.73
	2010	April	0.73		–1.00	–1.00	–1.00	–1.00	–0.01
	2010	May	0.22	0.71	0.45	–1.00	0.51	–0.87	–0.84
	2010	June	0.71	–1.00	–0.88	–1.00	–1.00	–1.00	–0.68
	2010	July	0.31			–1.00	–1.00		0.48
Sand Island	2008	April	0.45			–1.00	–1.00	–1.00	–0.67
Franz Lake	2008	April	0.35	–0.12		–1.00	–1.00	–1.00	1.00
	2008	May	0.78	–1.00		–1.00	–0.82	–1.00	–0.93
	2009	May	0.63			–1.00	–1.00	–0.89	1.00
Hardy Slough	2008	June	0.88		0.21	–1.00	–0.97	–1.00	–0.88

Note: Squares with no value indicate that the particular invertebrate taxa was not present in both the tows and the diets during that sampling period.

3.2 Trend Sites

3.2.1 Vegetation

A temporal analysis of vegetation species composition, vegetation cover, and inundation patterns was previously conducted at three of the trend sites: Cunningham Lake (rkm 145) and Campbell Slough (rkm 149) for the years 2005 to 2009 and Franz Lake (rkm 221) for 2008 to 2009 (Sagar et al. 2012). Herein, this analysis is extended to include data from the most recent monitoring years of 2010 and 2011 and to include the relevant data from an additional trend site, Whites Island (rkm 72) from 2009 to 2011. In addition, accretion rates and channel morphology were compared at the four sites for the years monitored between 2008 and 2011.

Hydrology Patterns

The Columbia River Basin is primarily a snow-melt runoff watershed and as such is subject to interannual hydrologic variability. Figure 65 shows the variation in the timing and magnitude of outflow at Cascade Island, just below Bonneville Dam for the years 2005 to 2011 and is provided here to give an overview of the hydrological patterns for the estuary during this study. In general, 2006, 2008, and 2011 were high flow years, 2005 and 2007 were lower flow years, and 2009 and 2010 were average flow years as compared to the 10-year average flow (2000–2009).

Sediment Accretion Rate

Sediment accretion rates at the four trend sites vary between −1.2 cm/year (erosion) and 3.0 cm/year (accretion) (Table 16). These differences are likely due to a combination of factors at the sites including vegetation, elevation, and flooding impacts. The variability in the rates at Whites Island (WHC) between 2008 and 2009 and 2009 and 2010 may be in part due to a change in vegetation at the location of the stakes. Observations at the time of measurements indicate that in 2009 a 6-cm-thick mat of forget-me-not (*Myosotis scorpioides*) may have caused difficulties in determining the true sediment surface. Over the years, the vegetation has shifted to include more reed canarygrass. The highest rate was measured at Franz Lake (FLM) in 2011, which would be expected following the extended high water period at this site. Likewise, rates greater than 1.0 cm/year were also observed at Campbell Slough (CS1) and Cunningham Lake (CLM) where the high water likely increased sedimentation rates also.

Table 16. Sediment accretion rates at the trend sites between 2008 and 2011.

	Whites Island	**Campbell Slough**	**Cunningham Lake**	**Franz Lake**
	Rate (cm/year)			
2008–2009	−1.2	ND	ND	0.5
2009–2010	1.0	0.4	1.9	ND
2010–2011	0.1	1.7	1.6	3.0

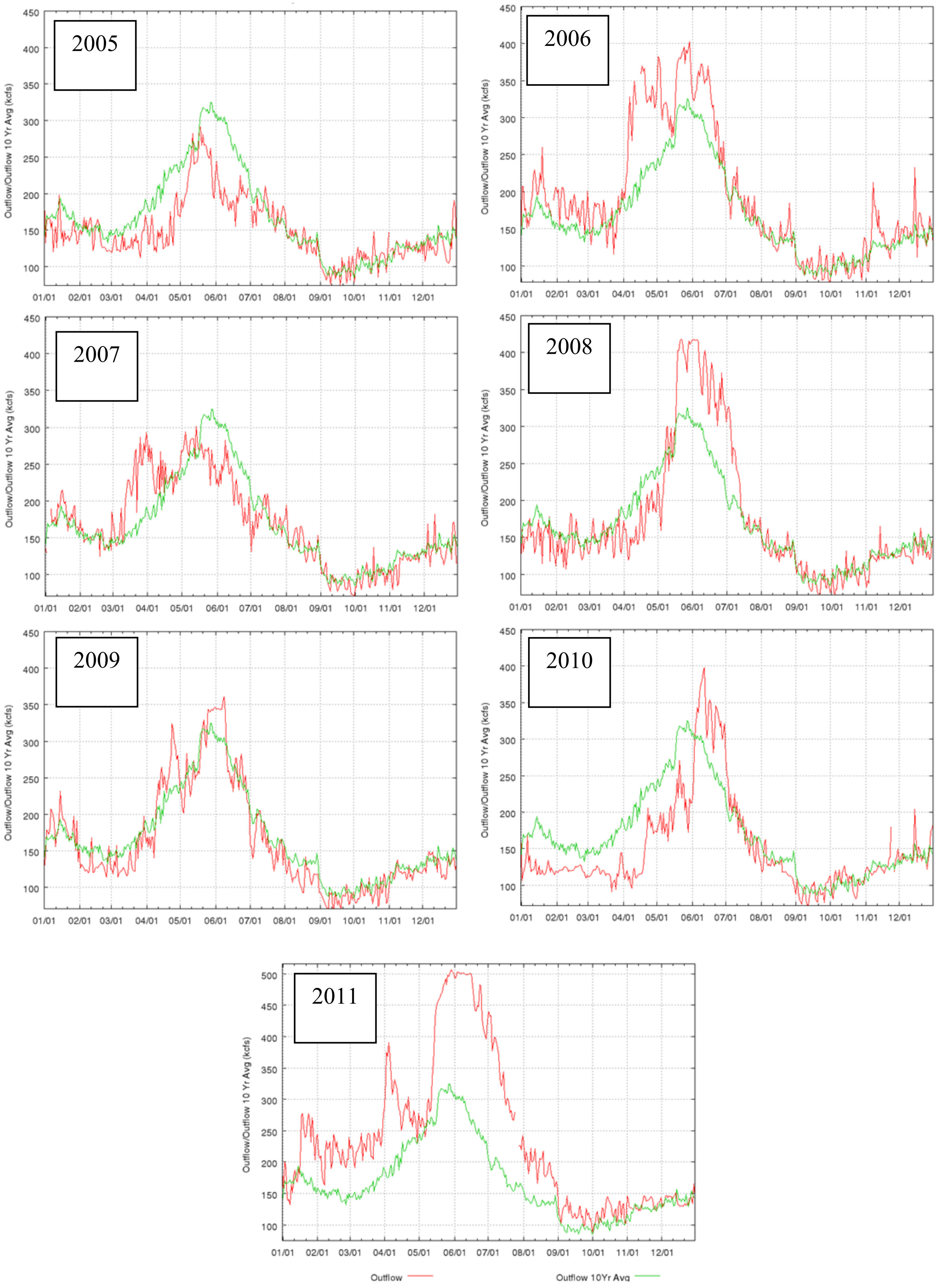


Figure 65. Daily mean outflow (red line) at Bonneville Dam compared to the 10-year average (2000–2009; green line) for the years 2005 to 2011. Note the slightly larger scale for 2011 (Columbia River Data Access in Real Time [DART] 2012).

Vegetation and Elevation

Overall, the vegetation composition at the four sites was similar in all monitoring years (Figure 66); however, a weighted similarity index comparing all years at each site shows there is some variability between years. One way to evaluate similarity over time is to compare the original year of monitoring to the following years. Whites Island became less similar to its original monitored state (2009) over the three years, although a longer record will be necessary to determine if this trend continues. At Cunningham Lake, the trend over time was consistent in that the high water years (2008 and 2011) were the least similar to the low-flow starting year (2005). At the Campbell Slough site, the trend is not as clear. In 2007, cows were present at the Campbell Slough site, resulting in grazing and trampling of some of the vegetation and as expected, this year is the least similar to the original year. Likewise, 2007 was most similar to the high water years at this site, which also had a disturbance to the vegetation. Similarity at Franz Lake was consistent over the three years monitored at the site.

Whites Island

	2009	**2010**
2010	74.4	
2011	64.0	73.8

Cunningham Lake

	2005	**2006**	**2007**	**2008**	**2009**	**2010**
2006	74.7					
2007	72.6	69.7				
2008	68.2	71.4	83.2			
2009	72.1	63.9	80.9	74.4		
2010	72.5	71.5	77.3	76.9	79.0	
2011	53.1	69.1	57.0	63.9	57.5	61.1

Campbell Slough

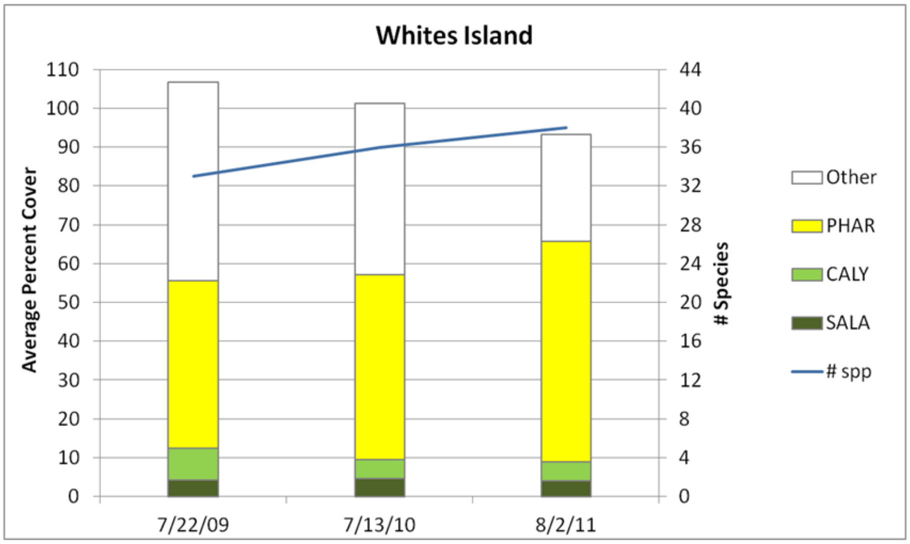
	2005	**2006**	**2007**	**2008**	**2009**	**2010**
2006	75.8					
2007	62.5	64.7				
2008	68.4	71.3	77.9			
2009	72.5	64.6	60.9	66.9		
2010	69.6	62.7	66.7	65.6	82.1	
2011	71.8	73.1	71.6	76.3	68.9	70.3

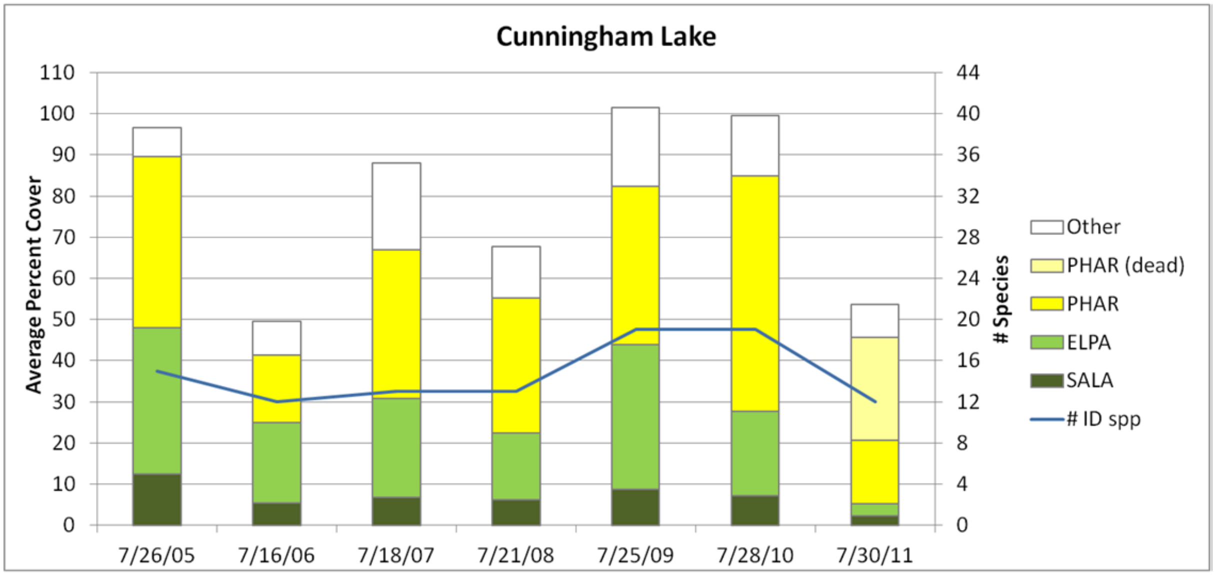
Franz Lake

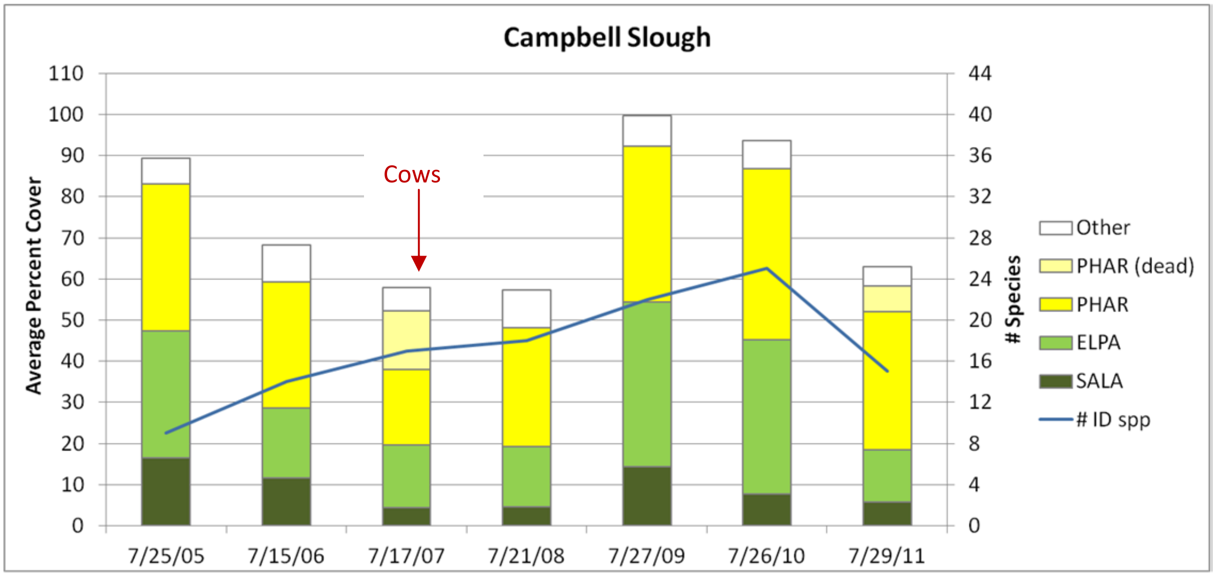
	2008	**2009**
2009	71.0	
2011	71.5	74.1

Figure 66. Similarity analysis results between years for the four trend monitoring sites.

Vegetation cover and the number of species present are the two primary factors contributing to the similarity variability observed at the sites over time. Figure 67 shows the average percent cover of the dominant species, the number of species, and the variation between years. Whites Island is the farthest down-river site and therefore the least affected by hydrologic variability. The site is the most diverse, with the number of species ranging from 33 to 38 between years. Overall cover at Whites Island decreased by 14 percent while reed canarygrass increased by 10% over the three-year period. The trends at Cunningham Lake and Campbell Slough are similar to each other, with the exception of 2007 at Campbell Slough, where lower cover occurred during the higher water years (2006, 2008, and 2011). Cunningham Lake had a lower cover in 2011 than Campbell Slough, perhaps due to the higher elevation of the Campbell Slough site being exposed earlier than the Cunningham Lake site. The trend is similar at Franz Lake, with lower cover in the two high water years. It is likely the cover would have been even lower in 2011 if the site was sampled late; however, the water was still too high to sample at that time and was sampled one month later. To further evaluate the effect of hydrology on vegetation cover, we compared the vegetation to the hydrologic patterns in each year as described below.







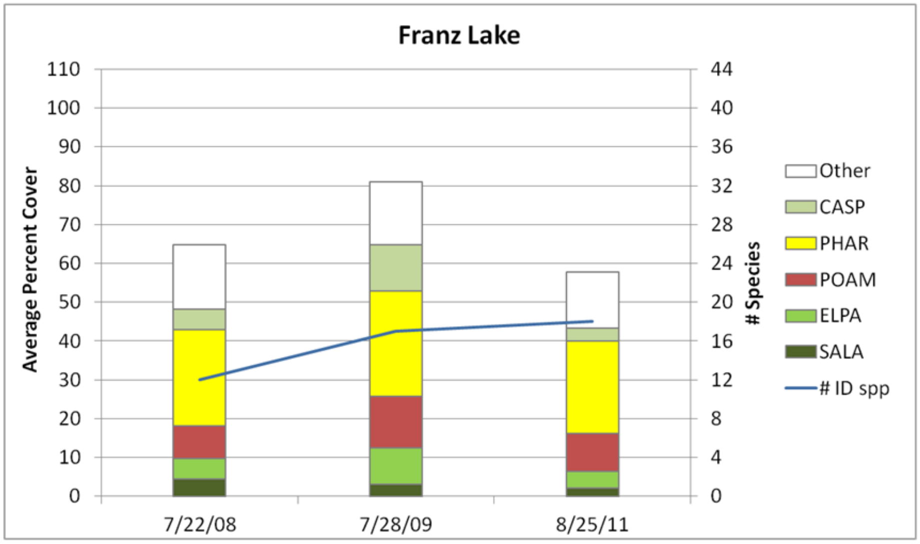
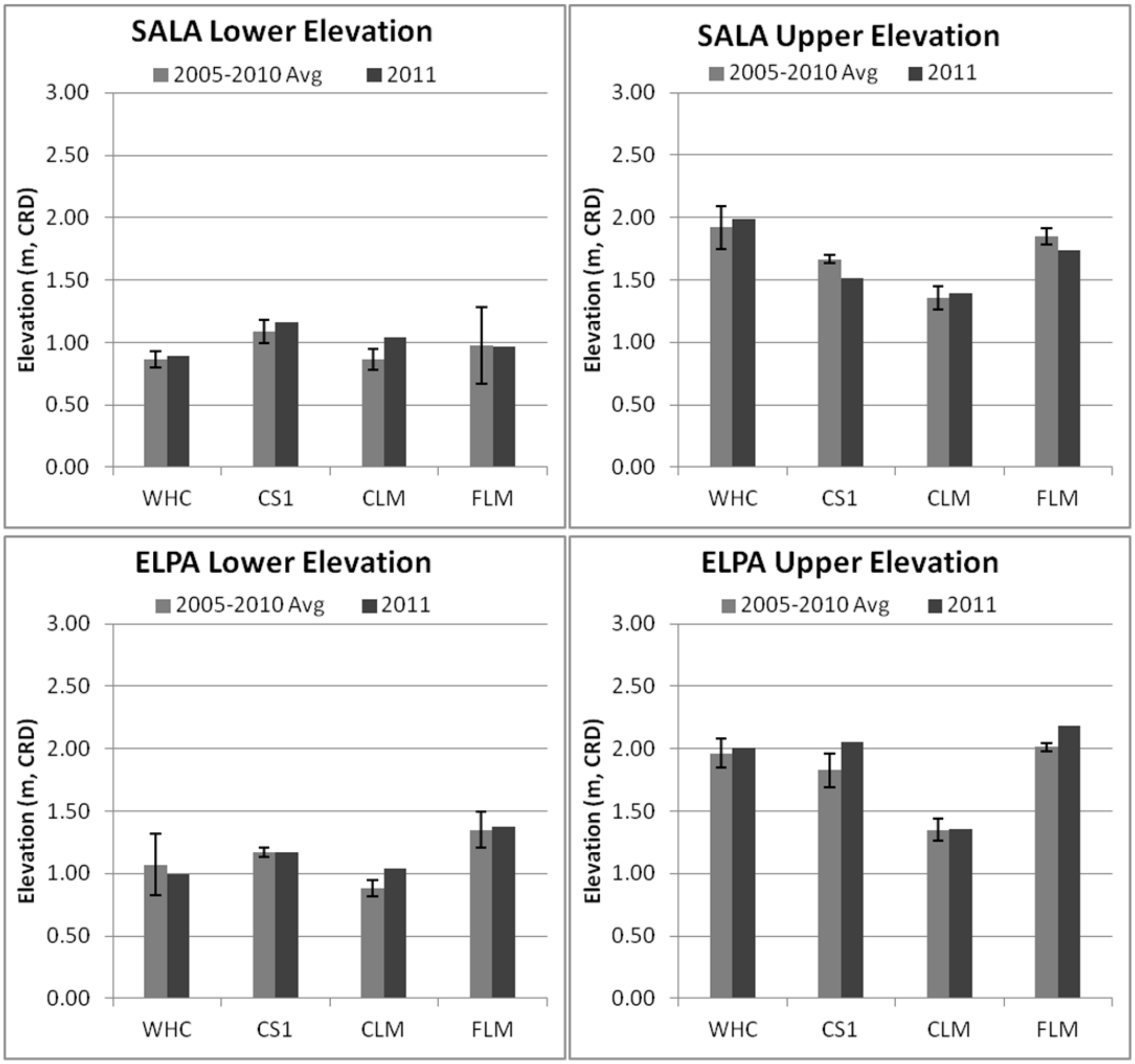


Figure 67. Average percent cover and number of identified species at the trend sites for all years monitored.

The upper and lower limits of the dominant species were evaluated in the previous trends analysis at Campbell Slough and Cunningham Lake. The results indicated that the elevation boundaries were stable, with little variation between years. To evaluate this finding further, we extended the analysis to include recent years and the Whites Island and Franz Lake sites (Figure 68). We compared the 2011 elevations to the average elevation of the lower and upper boundaries for each species to determine if the hydrologic increase caused a change in the boundaries outside the range seen in the previous years. Higher variability occurred at Whites Island, likely due to the greater heterogeneity of elevations at the site due to steep cut-banks. The variability associated with the lower elevation of wapato at Franz Lake is likely due to the increased inundation from the beaver dam in 2009 and 2011 compared to that in 2008 when wapato was able to grow at a lower elevation. In general, the boundaries did not vary more in 2011 than they did in the previous years, with a few exceptions. There was a slight increase in the lower elevations for all three species at Cunningham Lake, but not the upper limits, indicating that indeed the plants were likely affected by the higher water.



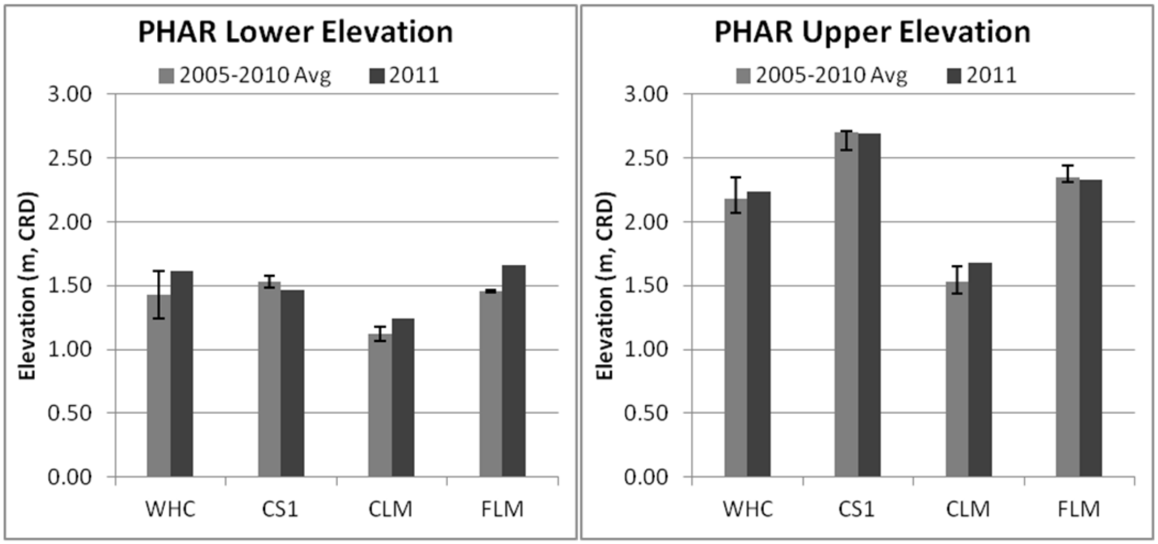


Figure 68. Lower and upper elevations for the dominant species at the trend monitoring sites. Error bars on the 2005 to 2010 average bars represent ± standard deviation. SALA=wapato, ELPA=common spikerush and PHAR=reed canarygrass.

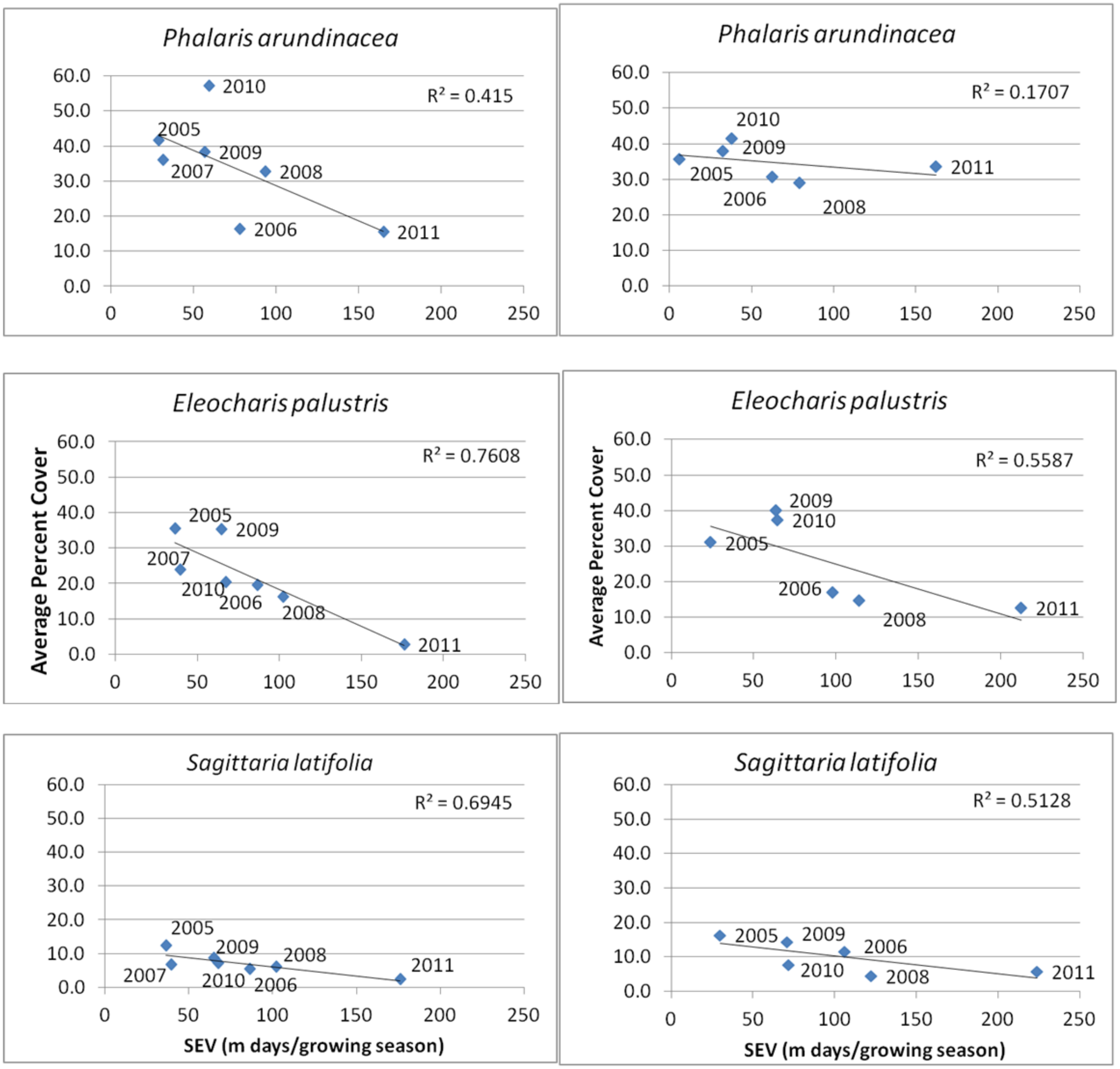
Vegetation Cover and Inundation

Three study sites (Franz Lake, Cunningham Lake and Campbell Slough) were evaluated for interannual variability in vegetation cover and inundation. Whites Island was not included in this analysis because the hydrologic patterns do not vary at this site in the same way they do at the up-river sites (Figure 69). Three species constitute most of the vegetation cover at the study sites (five at Franz Lake), so the SEV was calculated for the average elevation of each vegetation community, using the water levels measured each year during the growing season (Figure 69). The overall pattern at both sites is decreased vegetation cover with increased inundation as measured by the SEV. Interestingly, although there are differences in the elevations of some of the strata between sites, the SEV is very similar. For example, the average elevation of the reed canarygrass strata is 1.3 m and 2.0 m at Cunningham Lake and Campbell Slough, respectively, and the SEV in 2011 was 165 and 163 at Cunningham Lake and Campbell Slough, respectively.

While the relationship between cover and inundation held true for the most part, some exceptions require closer inspection. For example, the cover of reed canarygrass at Campbell Slough was higher than would have been expected in 2011 given the inundation levels that year. One possible explanation could be the higher elevation at this site compared to the others and the timing of the inundation relative to our sampling (e.g., areas of the strata exposed more recently). Also, the cover of wapato at Franz Lake was actually higher in one of the high inundation years (2008) compared to the lower inundation year (2009), which could possibly be attributed to the beaver activity observed at the site causing more ponding and inundation in 2009 at the low elevations where wapato grows.

Channel Morphology

Interannual variability of cross-section morphology is low as seen by the comparison of single cross-section locations from the trend sites (Figure 71). Elevation differences at Whites Island between 2010 and 2011 are likely due to slightly different survey intervals, although some erosion at the channel thalweg is also possible. This site has much more tidal flow than the other up-river trend sites, which could potentially result in more channel scouring.



a) Cunningham Lake b) Campbell Slough

Figure 69. Annual average percent vegetation cover of the dominant species at a) Cunningham Lake and b) Campbell Slough as related to annual growing season SEV calculated at the average elevation for each species.

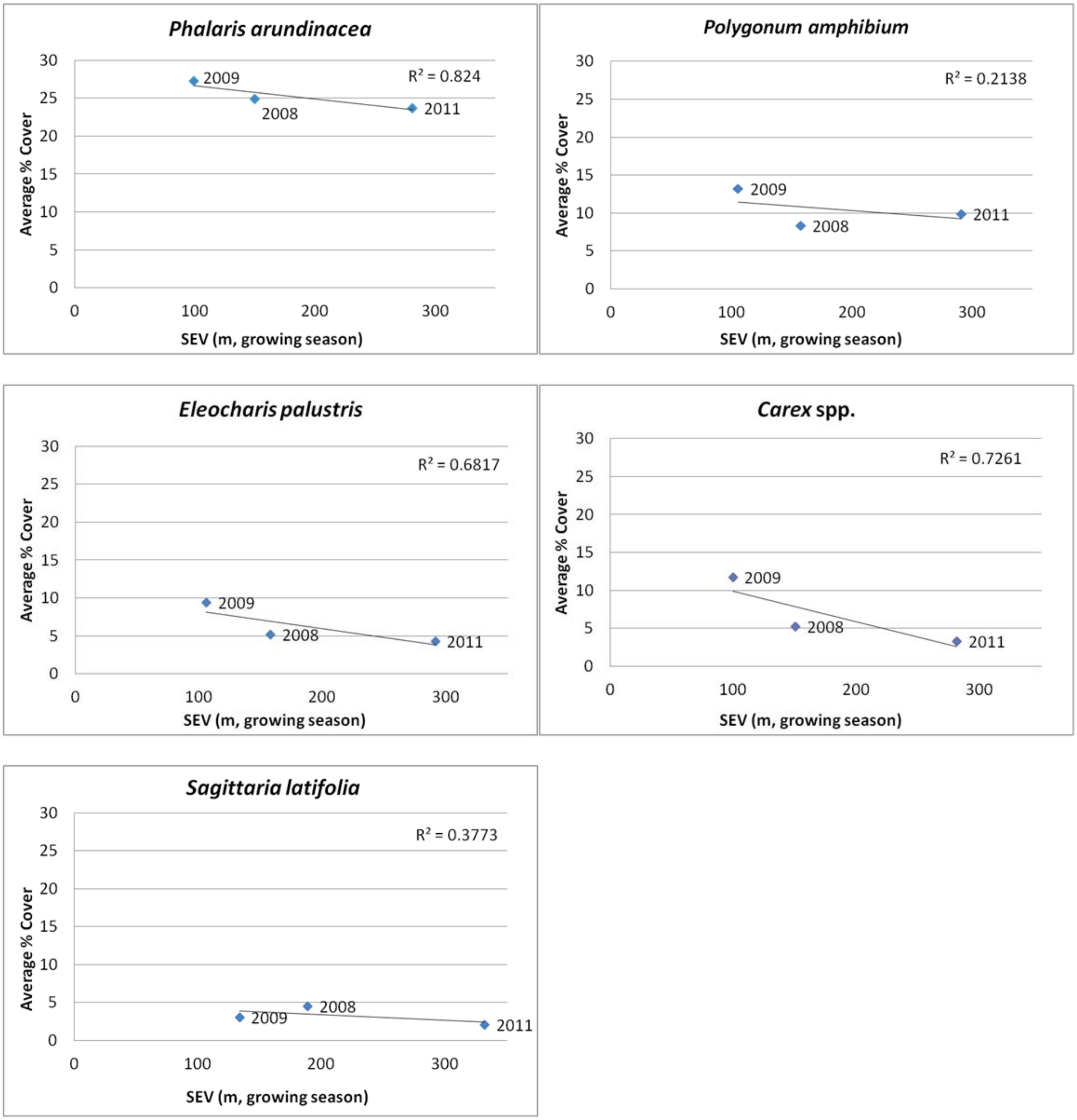


Figure 70. Annual average percent vegetation cover of the dominant species at Franz Lake as related to annual growing season SEV calculated at the average elevation for each species.

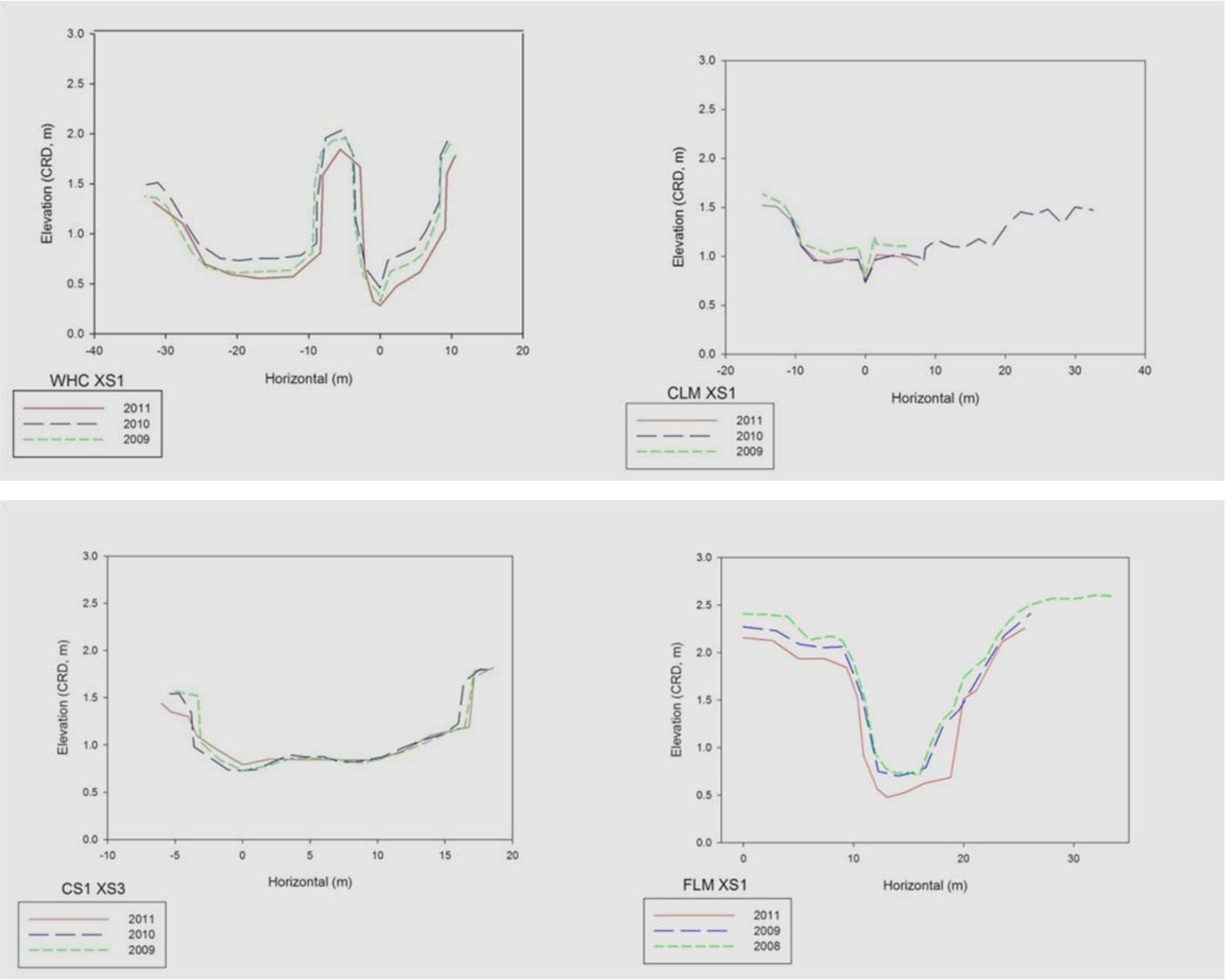


Figure 71. Elevations of the channel cross-sections for the trend sites showing multiple years.

3.2.2 Fish

The EMP fixed sampling sites included Whites Island in Reach C, sampled in 2009 and 2010; Campbell Slough in Reach F, sampled from 2007 to 2010; and Franz Lake in Reach H, sampled in 2008 and 2009 (Table 2).

Water Temperature

At all three fixed sites, in all sampling years, water temperatures increased steadily throughout the sampling season from 9°C to 10°C in April to over 20°C to 25°C by July and August (Figure 72). At Whites Island, temperature ranges in 2009 and 2010 were similar until August, when temperatures reached 25°C in 2009, but only 20°C in 2010. At Campbell Slough, the temperature ranges at the site were similar in 2007, 2008, and 2010, although the site was sampled only from May through July in 2007. In 2009, higher maximum temperatures were reached, with temperatures of 25°C to 28°C in July and August. At Franz Lake, as at the other two sites, summer temperatures were somewhat higher in 2009 than in 2008, with a maximum temperature in 2009 of 28°C, as compared to 25°C in 2008. While no increasing or decreasing trends in temperature were observed, at all sites, highest summer temperatures were observed during the 2009 sampling year.

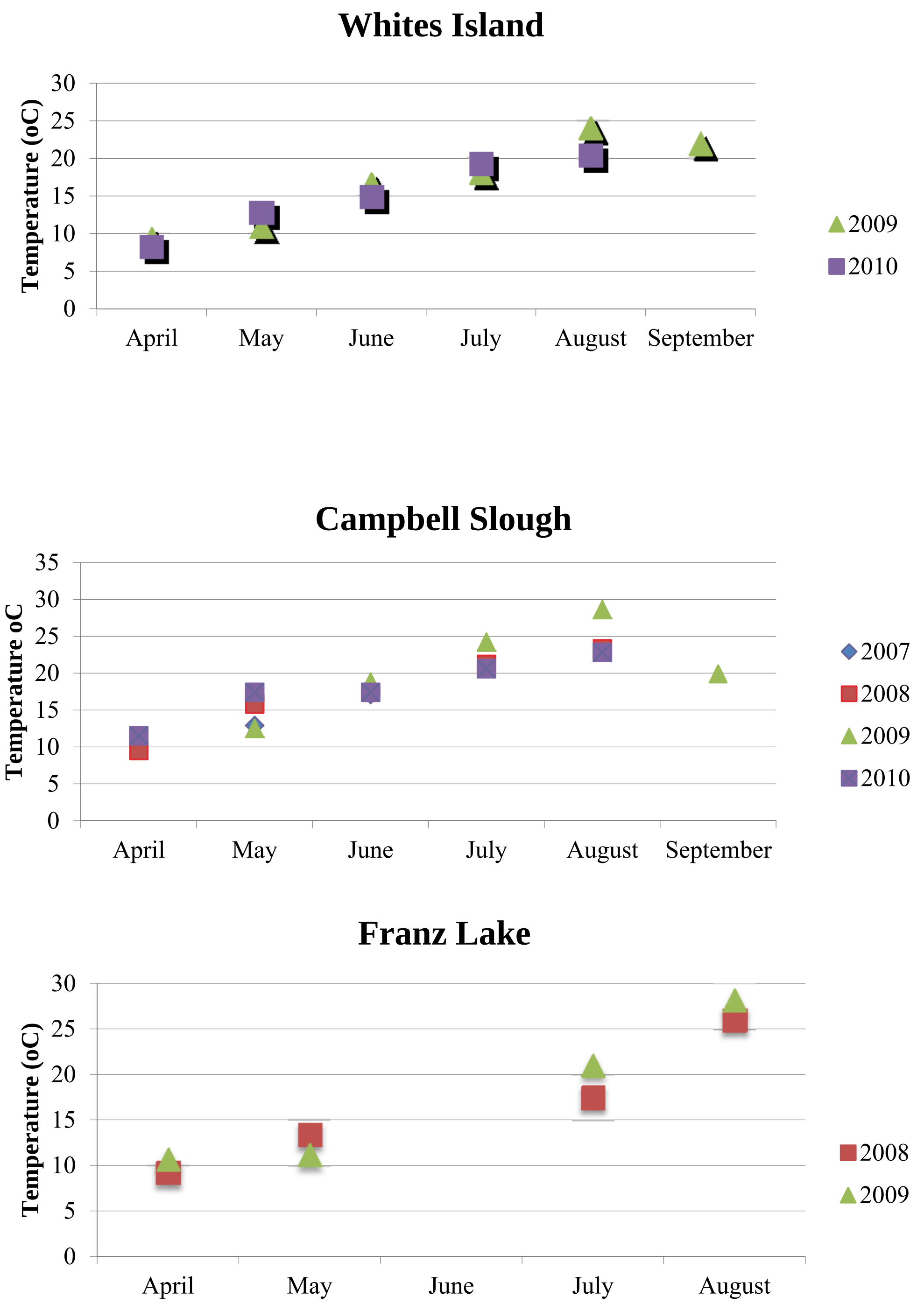


Figure 72. Seasonal water temperatures by year at Whites Island, Campbell Slough, and Franz Lake.

Fish Community Characteristics

Whites Island. Juvenile salmon, as well as juveniles of other fish species, were feeding and rearing at Whites Island in 2009 and 2010 (Figure 73). Three-spine stickleback was by far the most abundant species in both years, making up 93% to 98% of the total catch. Juvenile Chinook and chum salmon were captured in 2009, whereas Chinook, coho, and chum were captured in 2010. Chinook salmon made up 0.5% of the total catch in 2009 and 2.6% of the total catch in 2010. Chum and coho made up 0.01% to 0.07% of the total catch in both years (Figure 73).

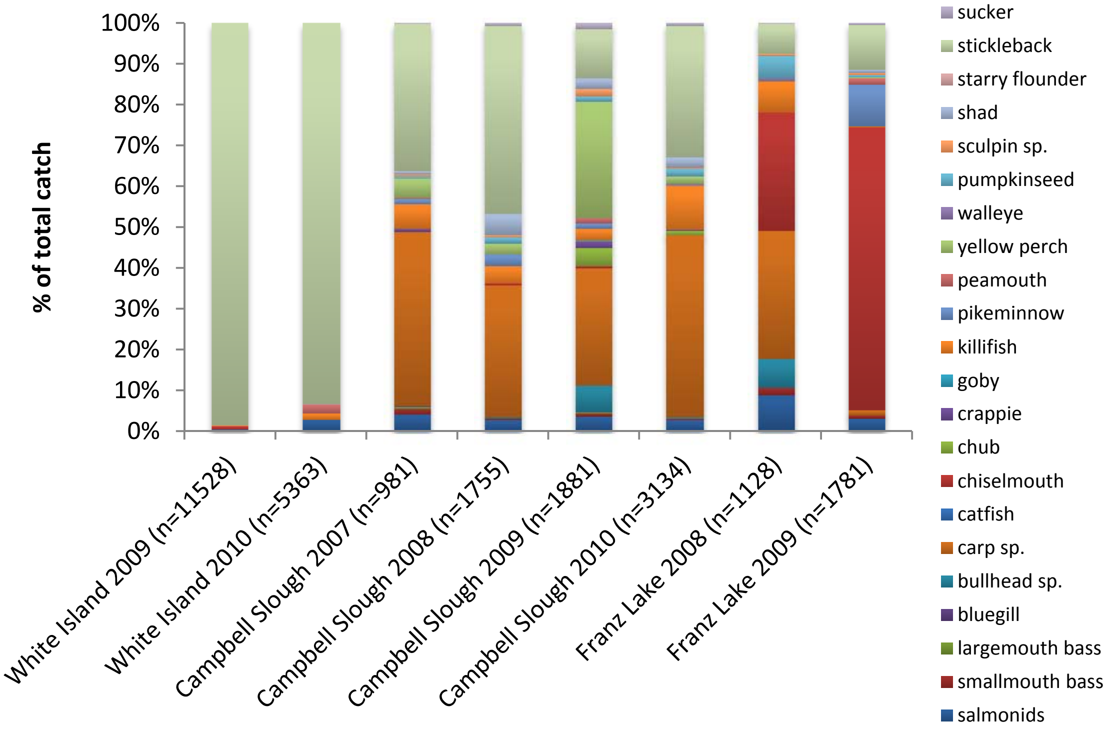


Figure 73. Fish community composition by year at fixed sampling sites.

The total number of fish species collected at Whites Island was consistent over time—nine in both 2009 and 2010 (Figure 74). The Shannon-Weiner species diversity index ranged from 0.33 in 2010 to 0.86 in 2009 (Figure 74). Non-native species accounted for 0.1% and 1.6% of the number of total number of species caught at Whites Island in 2009 and 2010, respectively (Figure 75). No piscivorous predators were found in either sampling year (Figure 75).

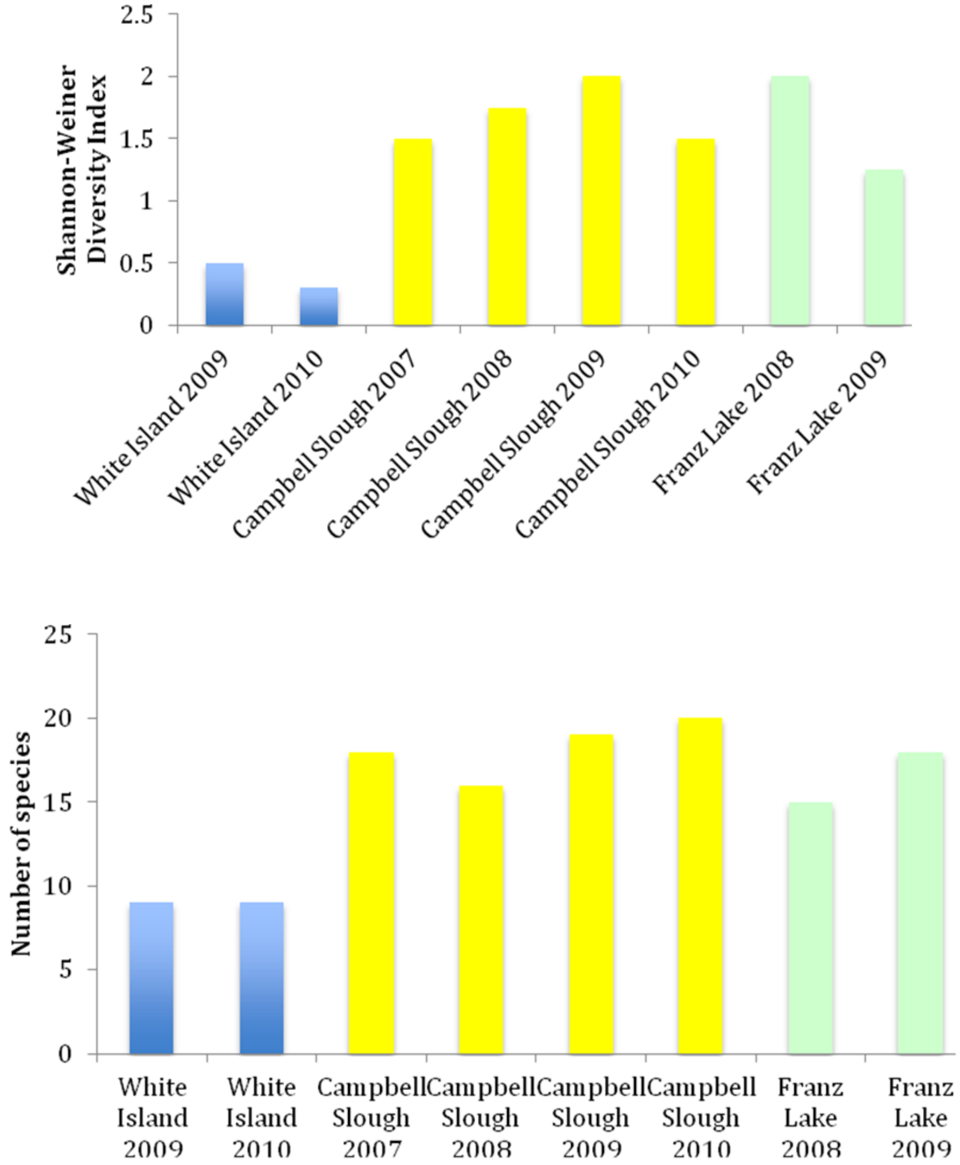
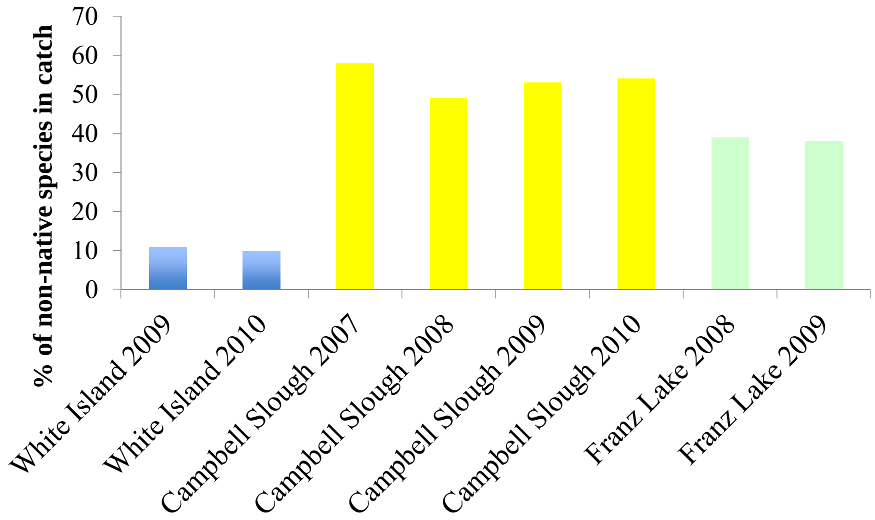


Figure 74. Species diversity and numbers of species by sampling year at Whites Island, Franz Lake, and Campbell Slough.



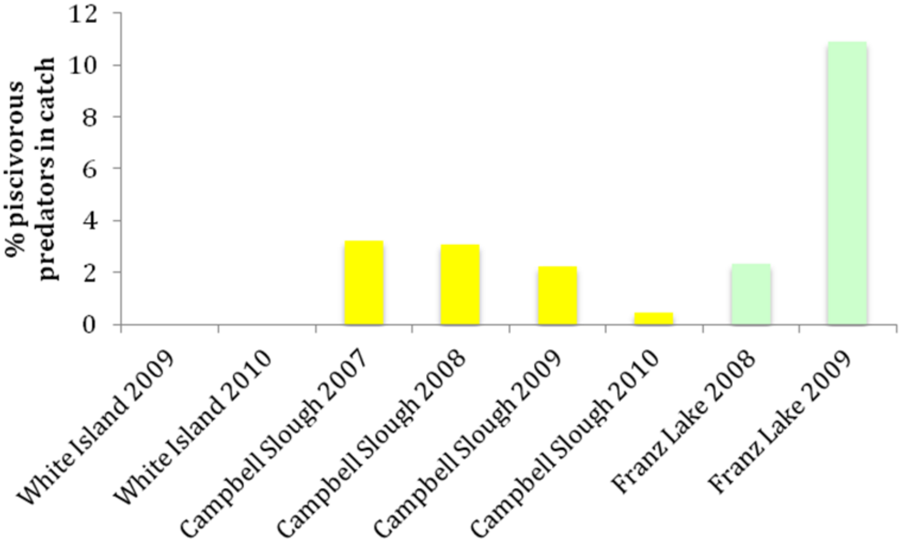


Figure 75. Percentage of species captured that were non-natives, and percentage of piscivorous predators in the catch by sampling year at Whites Island, Franz Lake, and Campbell Slough.

Campbell Slough. In spite of difficulties with site access at certain months in certain years, our sampling showed that juvenile salmon and juveniles of other fish species were feeding and rearing at the Campbell Slough site in 2007, 2008, 2009, and 2010 (Figure 73). Three-spine stickleback and juvenile carp were the most abundant species in 2007, 2008, and 2010, together making up 77% to 79% of the total catch. In 2009, the most abundant species were juvenile carp and yellow perch, accounting for 48% of the total catch. Three-spine stickleback accounted for only 8.4% of the total catch in 2009, as compared to 32% to 46% in other years. Juvenile Chinook salmon were captured in all four years, with the percentage of total catch from 3% to 4% (Figure 73).

The total number of fish species collected at Campbell Slough was fairly consistent over time—18 in 2007, 16 in 2008, 19 in 2009, and 20 in 2010 (Figure 74). The Shannon-Weiner species diversity index varied, but showed no clear trends, increasing from 1.5 in 2007 to 2.3 in 2009, but then declining again in 2010 to 1.5 (Figure 74). Non-native species accounted for 50% to 70% of the total number of species caught at Campbell Slough from 2007 to 2010 (Figure 75). Piscivorous predators made up a small but consistent proportion of the catch, ranging from 0.5% in 2010 to 3.2% in 2007.

Franz Lake. Sampling indicated that juvenile salmon and juveniles of other fish species were feeding and rearing at the Franz Lake site in 2008 and 2009 (Figure 73). Chiselmouth and juvenile carp were the most abundant species in 2008, accounting for 29% and 31% of the total catch, while chiselmouth was the most abundant species in 2009, accounting for 69% of the total catch. Juvenile Chinook were captured in both years; the percentage of the total catch was 5.3% in 2008 and 1.1% in 2009.

The total number of fish species collected at Franz Lake was 15 in 2008 and 18 in 2009 (Figure 74). However, the Shannon-Weiner species diversity index per unit effort decreased from 2 in 2008 to 1 in 2009 (Figure 74). Non-native species accounted for about 42% of the total number of species caught at Franz Lake in both 2008 and 2009 (Figure 75). The percentage of piscivorous predators in the catch was quite variable, ranging from 2.3% in 2008 to nearly 11% in 2009 (Figure 75).

Salmonid Species Composition

Whites Island. In 2009 and 2010, Chinook salmon made up 97.6% and 95% of the juvenile salmonid catch at Whites Island (Figure 76). Chum salmon accounted for 2.4% and 2.7% of the catch in 2009 and 2010. No coho were captured in 2009, but in 2010 unmarked coho accounted for 2.7% of the total catch. No steelhead, cutthroat, or rainbow trout were caught at Whites Island in either sampling year. In 2009 and 2010, both marked and unmarked Chinook salmon were found at the site, but unmarked Chinook accounted for the majority of the catch—83% in 2009 and 74% in 2010. Of the Chinook captured, 85% were unmarked in 2009 and 78% were unmarked in 2010.

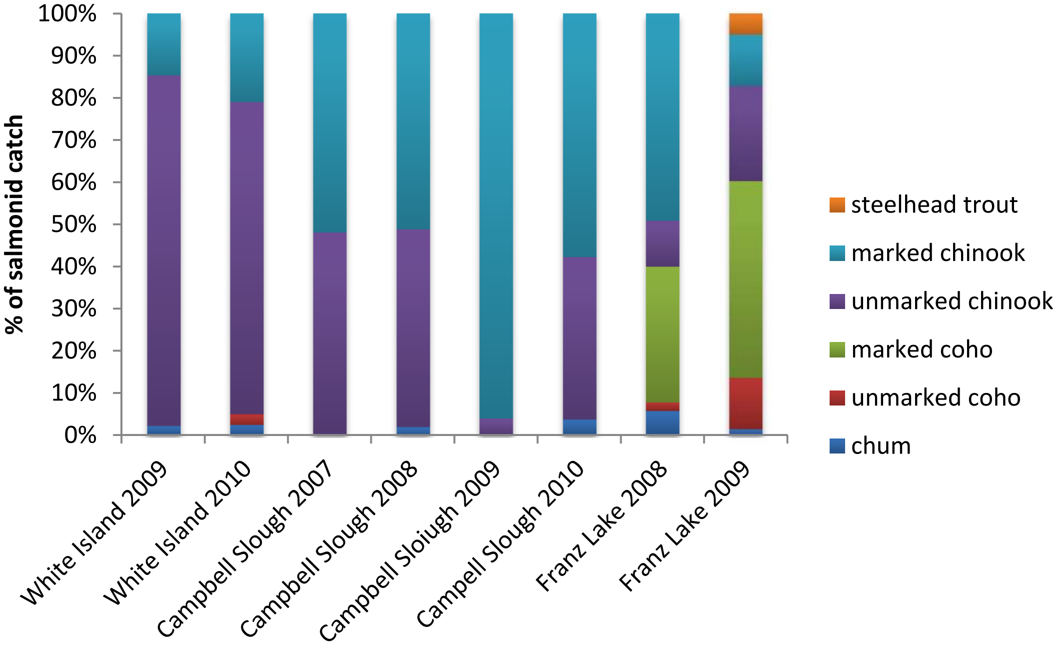


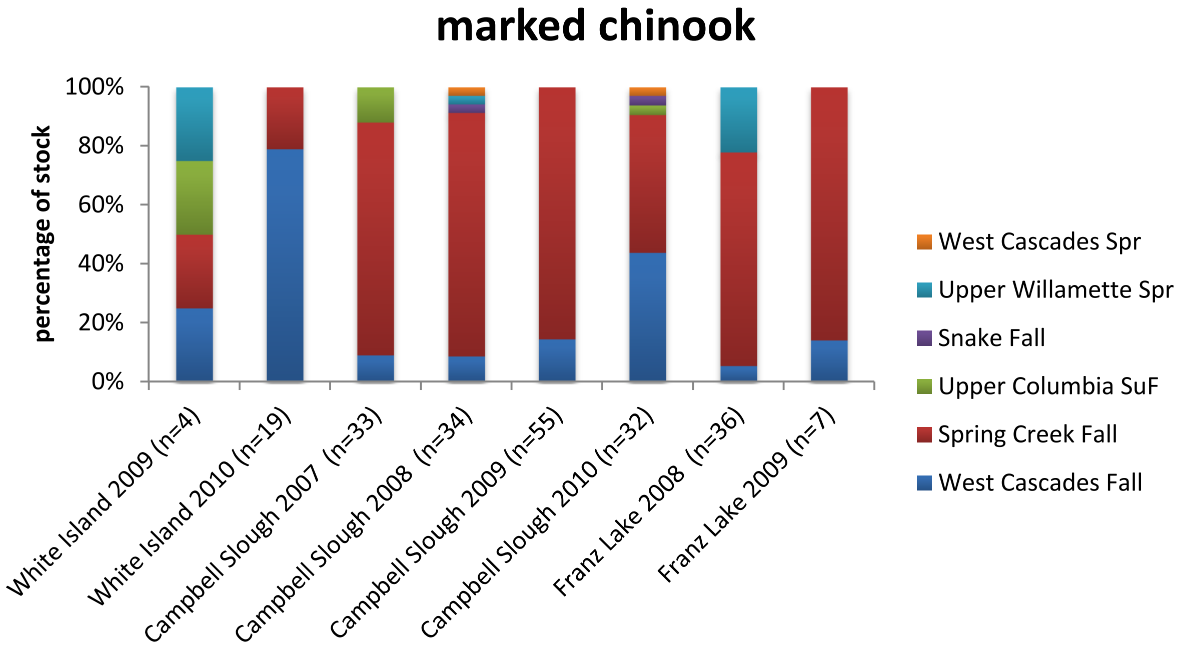
Figure 76. Percentages of different salmonid species in salmonid catches at White Island, Campbell Slough, and Franz Lake by sampling year.

Campbell Slough. In 2007 through 2010, Chinook salmon made up 99%, 98%, 100%, and 95%, respectively, of the juvenile salmonid catch at Campbell Slough (Figure 76). In 2007, no chum salmon and only one coho salmon were collected; in 2008, the opposite occurred as only one chum salmon and no coho salmon were collected. In 2009, neither coho nor chum salmon were collected, whereas in 2010, Chinook and chum were collected. In 2007, 2008, and 2010, both marked and unmarked Chinook salmon were found at the site in similar proportions. Marked fish accounted for 52% of the catch in 2007, 51% in 2008, and 58% in 2010 (Figure 76). Note that the proportions for 2007 are determined from necropsied fish only, so may not accurately reflect true proportions of marked and unmarked fish in the habitat. In contrast, in 2009, 96% of Chinook captured were marked.

Franz Lake. In 2008 and 2009, Chinook salmon made up 60%, and 35.1%, respectively, of the juvenile salmonid catch at Franz Lake (Figure 76). In both years, chum and coho salmon were also collected at this site. Chum made up 6% of the salmonid catch in 2008 and 1.8% of the salmon catch in 2009, while coho made up 34% of the salmon catch in 2008 and 61.8% of the salmonid catch in 2009. In 2009, in addition to salmon species, steelhead trout were also caught, which made up 3.5% of the salmonid catch. All chum salmon caught at Franz Lake were unmarked. However, significant proportions of both Chinook and coho salmon at the site were marked (Figure 76). The proportion of marked Chinook salmon found at the site varied considerably from year to year. Marked fish accounted for 80% of the catch in 2008 but only 35% in 2009. The majority of coho salmon collected at Franz Lake in both 2008 and 2009 were marked. Marked fish accounted for 94% of the coho catch in 2008 and 79% of the coho catch in 2009. The distribution of salmonid species at the two sites was statistically different (Contingency Table, Chi-square analysis, $p<0.0001$).

Genetic Stock Identification of Juvenile Chinook Salmon

Whites Island. In 2009, only four marked juvenile Chinook salmon from Whites Island were analyzed for GSI. Two of these fish belonged to Lower Columbia ESU stocks (Spring Creek Group fall and West Cascades fall Chinook), while the other two fish were assigned to the Upper Willamette spring and Upper Columbia summer/fall stocks. In 2010, when a larger number of marked Chinook were analyzed, nearly all were from Lower Columbia ESU stocks, with 79% from the West Cascades fall Group and 21% from the Spring Creek fall group (Figure 77).



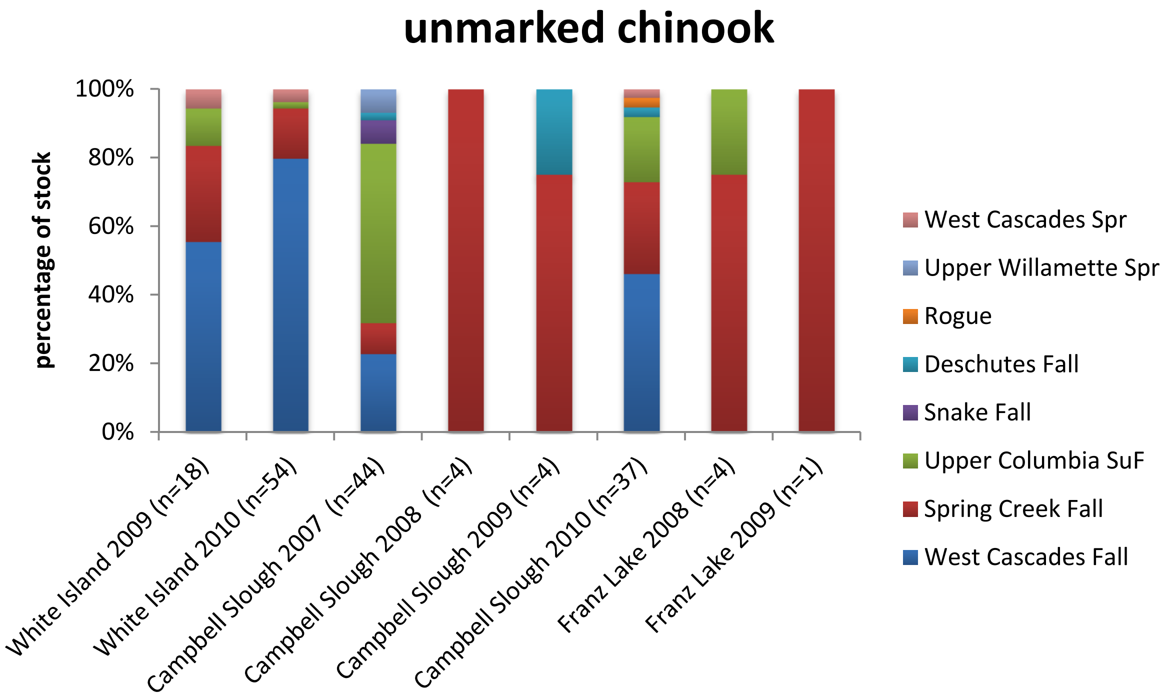


Figure 77. Genetic stock assignments for marked and unmarked Chinook salmon from Whites Island, Campbell Slough, and Franz Lake.

In both 2009 and 2010, unmarked Chinook salmon from Whites Island were primarily from the Lower Columbia River ESU, with fish from this group making up 89% to 98% of Chinook salmon analyzed. Of the Lower Columbia ESU stocks represented (Spring Creek Group fall, West Cascades fall, and West Cascades spring Chinook), the West Cascades fall Chinook stock made up the highest percentage of fish in both years (Figure 77). Small numbers of unmarked Upper Columbia summer/fall Chinook were also observed in both 2009 and 2010.

Campbell Slough. In 2007 through 2010, the majority of marked, juvenile Chinook salmon from Campbell Slough were from stocks included in the Lower Columbia ESU (West Cascades fall, West Cascades spring, and Spring Creek Group fall Chinook; Figure 77). From 2007 to 2009, the majority of the Lower Columbia ESU fish (~90%) were from the Spring Creek fall Group, but in 2010, approximately equal proportions of fish were from the West Cascades fall and the Spring Creek Group fall stocks. Small numbers of marked Chinook from other Columbia River ESUs, including Upper Columbia summer/fall Chinook, Upper Willamette spring Chinook, and Snake River fall Chinook, were also observed in 2007, 2008, and 2010.

In 2007 and 2010, when a relatively large number of unmarked juvenile Chinook were analyzed for GSI, the fish at Campbell Slough came from a diverse array of stocks (Figure 77). In both 2007 and 2010, the most commonly found stocks were Upper Columbia summer/fall Chinook, West Cascades fall Chinook, and Spring Creek Group fall Chinook, but other stocks were also represented, including Upper Willamette spring, West Cascades spring, Snake River fall, Deschutes River fall, and Rogue River Chinook. In 2008 and 2009, when only a small number of unmarked Chinook salmon (four per year) were analyzed, the majority of fish were identified as Spring Creek Group fall Chinook (Figure 77).

Franz Lake. In both 2008 and 2009, marked Chinook salmon sampled from Franz Lake were primarily Spring Creek fall Chinook from the Lower Columbia ESU (Figure 77). A smaller proportion of West Cascades fall Chinook were also present both years. Additionally, Upper Willamette spring Chinook were identified in 2008, but not in 2009.

Unmarked Chinook were also primarily from the Lower Columbia River ESU, with 75% of fish examined from the Spring Creek fall Chinook group (Figure 77). An additional 25% of fish were from the Upper Columbia summer/fall Chinook group. However, the number of unmarked Chinook captured in 2008 was very small, and may not be sufficient to characterize unmarked Chinook from this site. In 2009, genetic information was collected on only one unmarked fish, which was identified as a Spring Creek Group fall Chinook.

Salmonid Density and Seasonal Habitat Occurrence

Whites Island. At Whites Island, marked and unmarked Chinook salmon and chum salmon were caught in both 2009 and 2010, and unmarked coho salmon in 2010 only. Chinook salmon densities were consistently higher for both years than densities of coho or chum salmon, and densities of unmarked Chinook salmon were consistently higher than densities of marked Chinook salmon (Figure 78). For both marked and unmarked Chinook salmon, the mean density (expressed as fish per 1,000 m^2) calculated over the entire sampling year, increased significantly from 2009 to 2010 (1-way ANOVA, $0.035<p<0.051$). The mean density for chum and unmarked coho salmon also tended to be higher in 2010 than in 2009, but the differences were not statistically significant ($p>0.05$).

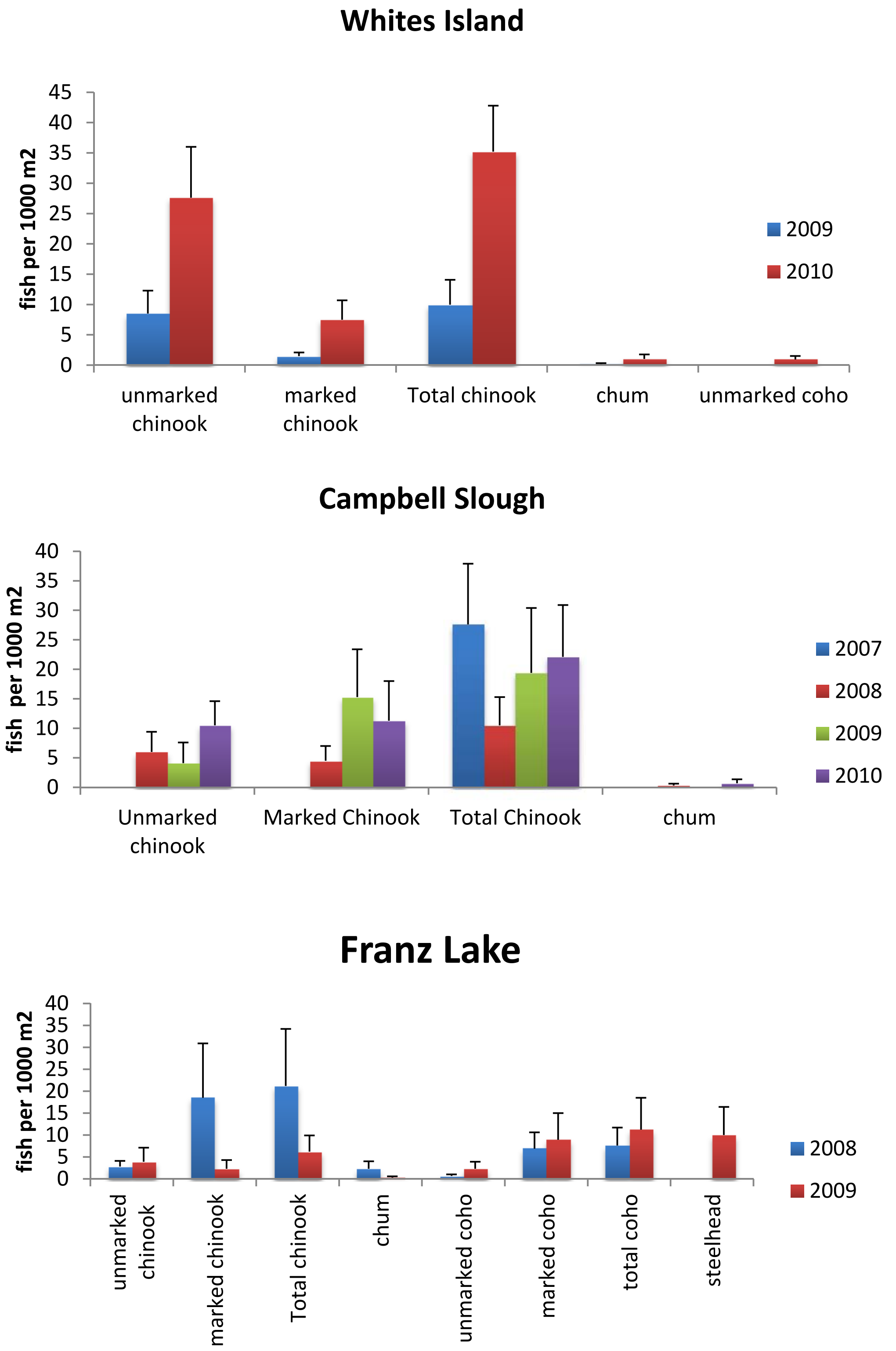


Figure 78. Chinook, chum, coho, and steelhead density (fish per 1,000 m^2) by year at Franz Lake, Campbell Slough, and White Island. Only total Chinook are indicated at Campbell Slough in 2007 because marking was not noted for all fish captured in tows.

For unmarked Chinook salmon, seasonal patterns of occurrence were similar in 2009 and 2010. In both years, unmarked juvenile Chinook were present at Whites Island from the start of the sampling season in April through July, and in both years, fish densities were higher in May and June (Figure 79). However, for all months, densities were consistently higher in 2010 than in 2009. In contrast, marked Chinook salmon at Whites Island showed different seasonal patterns of occurrence in 2009 and 2010. In 2009, marked Chinook were present at the site from April through June, and were not found in July or August. However, in 2010, marked Chinook were present from April through August, with the exception of the month of June, when no marked fish were present in catches. Moreover, while in 2009 the peak density for marked Chinook occurred in June, in 2010 the peak density occurred in August. As coho salmon (Figure 80) were found in 2010 only at Whites Island, seasonal patterns could not be compared by year.

For chum salmon, seasonal patterns of occurrence were also somewhat different in 2009 and 2010 (Figure 81). In 2009, chum were found only in May, while in 2010, they were found in both April and May, with the peak density occurring in April. Both April and May chum densities in 2010 were higher than the May density in 2009. No chum salmon were found at Whites Island in June, July, or August in either 2009 or 2010. Steelhead (Figure 81) were not found at Whites Island in either 2009 or 2010.

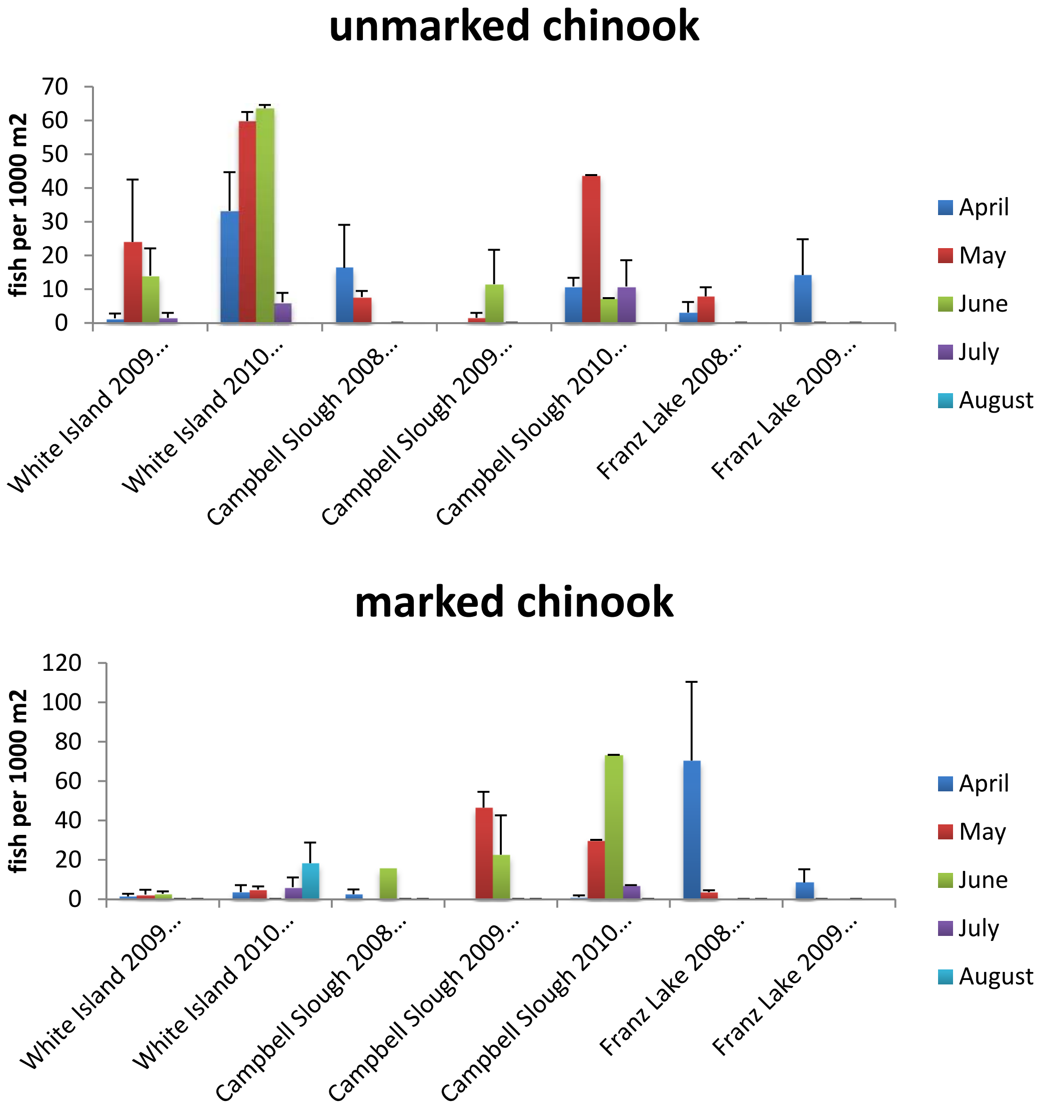


Figure 79. Density of marked and unmarked Chinook salmon by month and year at Whites Island, Campbell Slough, and Franz Lake. The number of tows per month and year are indicated in parentheses.

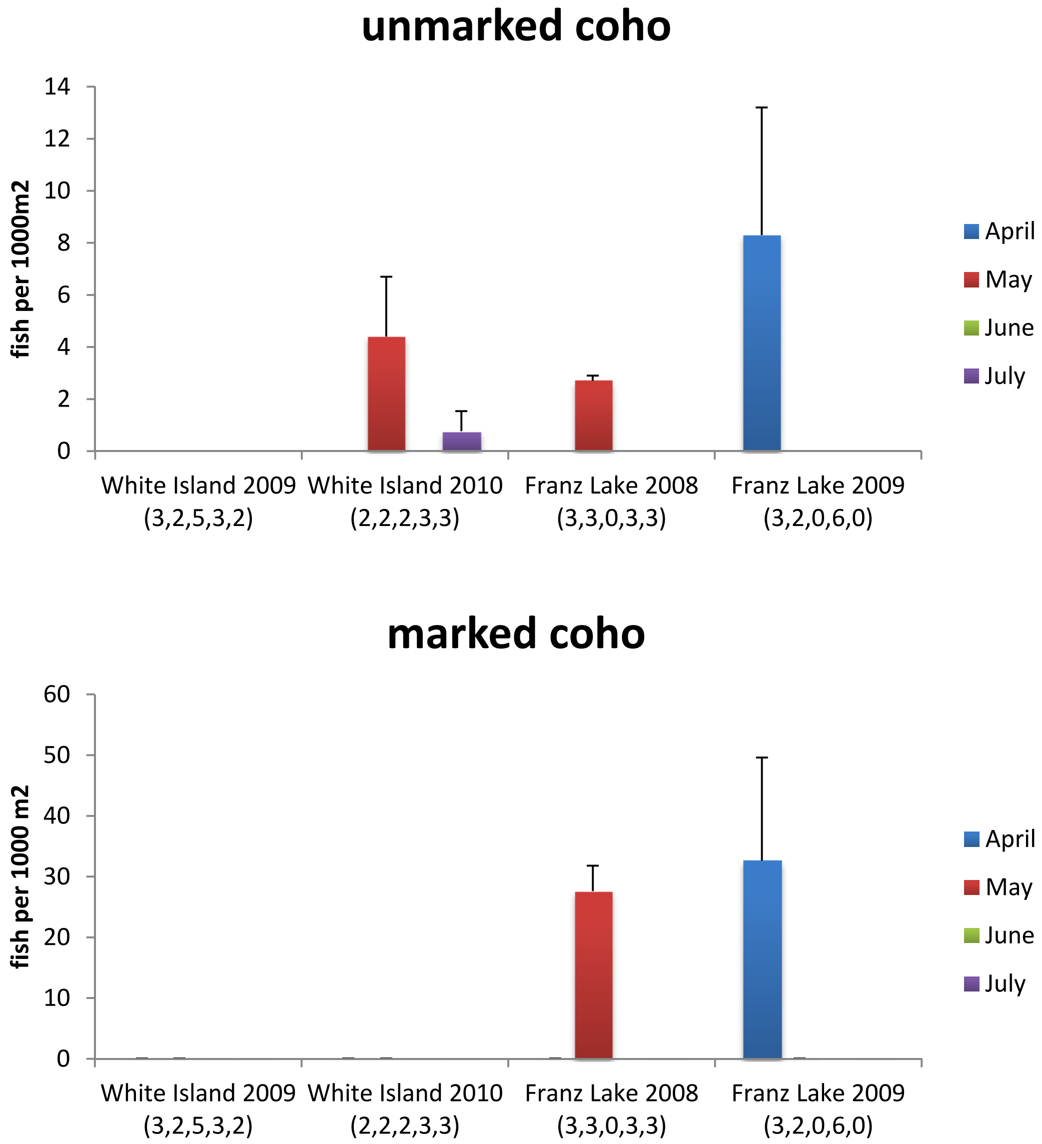


Figure 80. Density of marked and unmarked coho salmon by month and year at Whites Island, Campbell Slough, and Franz Lake. The number of tows per month and year are indicated in parentheses.

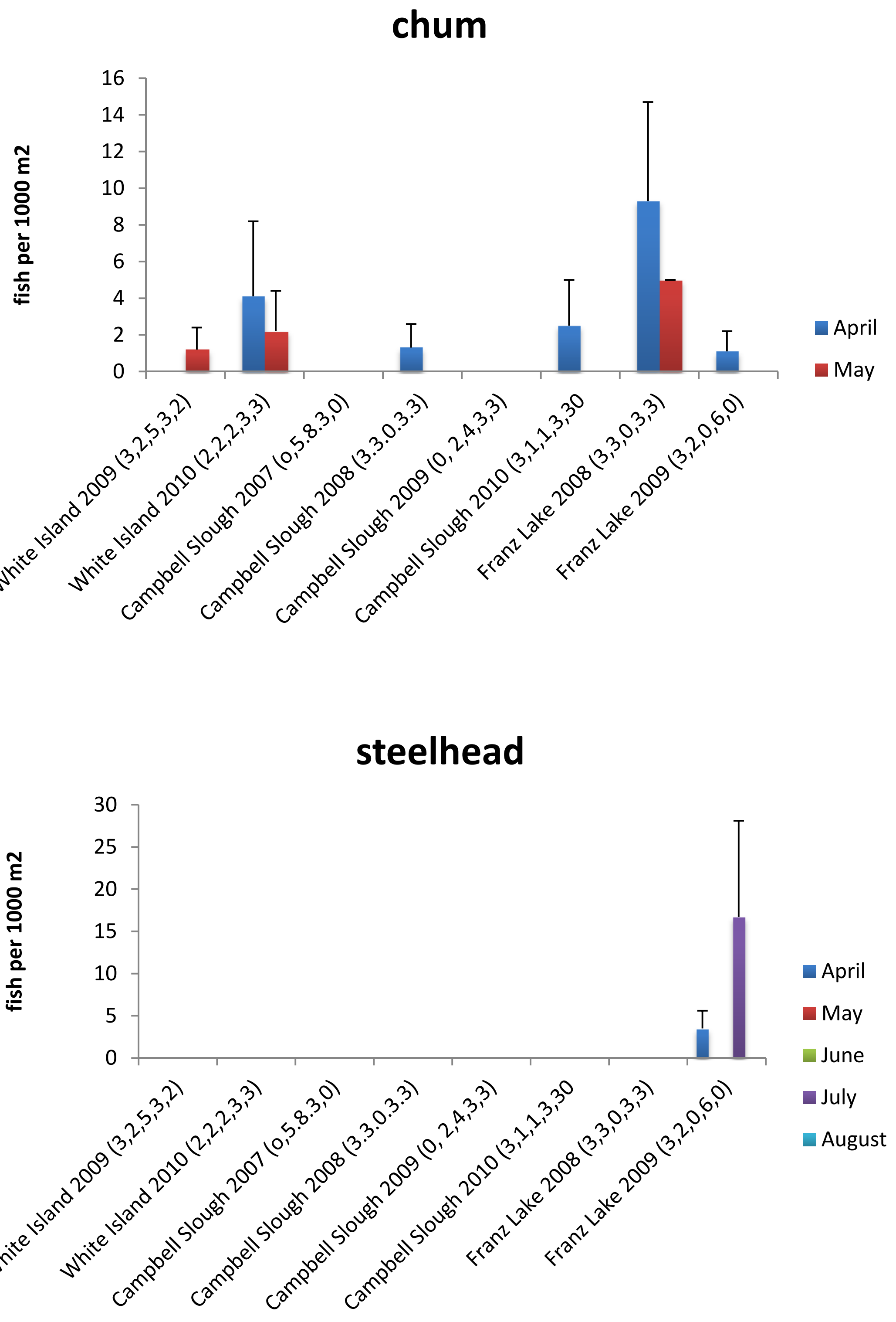


Figure 81. Chum salmon and steelhead trout densities by month and year at Whites Island, Campbell Slough, and Franz Lake. The number of tows per month and year are indicated in parentheses.

Campbell Slough. At Campbell Slough, Chinook salmon were caught in all sampling years, from 2007 to 2010; chum salmon were caught in 2008 and 2010; and one coho salmon was caught, in 2007. While both marked and unmarked Chinook were caught in all sampling years, marking was not recorded consistently for all salmon in 2007 catches, so Chinook for this sampling year are represented as total Chinook only. No statistically significant interannual differences were found in densities of unmarked, marked, or total Chinook salmon densities at Campbell Slough ($0.4553<p<0.6178$; Figure 78). Also densities of marked and unmarked Chinook at Campbell Slough were generally within the same ranges. Like Chinook salmon densities, chum salmon densities did not differ significantly from 2007 to 2010 ($p = 0.4345$). Chinook densities were consistently higher than chum densities in all sampling years.

Seasonal patterns of occurrence for Chinook salmon (Figure 79) were generally similar from 2007 to 2010, although comparisons are somewhat limited due to our inability to fish the site because of access problems in April of 2007 and 2009, and June of 2008. In 2007, 2008, and 2009, both marked and unmarked juvenile Chinook were present at the site from the beginning of the sampling season in April or May through June, but were absent after that time. In 2010, however, both marked and unmarked juvenile Chinook salmon were present from April through July. In all sampling years, peak densities of both marked and unmarked Chinook salmon were found in May or June.

Although chum salmon were found only in 2008 and 2010, in both of those years they were seen only in April, suggesting some consistency in seasonal occurrence (Figure 81). Steelhead trout (Figure 81) were not found at Campbell Slough in any sampling year.

Franz Lake. At Franz Lake, unmarked and marked Chinook salmon, unmarked and marked coho salmon, and chum salmon were caught in both 2008 and 2009, and steelhead trout were caught in 2009 (Figure 78). For unmarked Chinook, mean density over the entire sampling period was nearly the same in 2008 and 2009 (2.8 and 3.9 fish per 1,000 m^2). The density of marked Chinook declined during the same period, from 18.5 fish per 1,000 m^2 to 2.3 fish per 1,000 m^2, although this change was not statistically significant ($p = 0.2333$). For coho salmon, density increased slightly for both marked and unmarked fish, increasing from 0.67 to 2.3 fish per 1,000 m^2 for unmarked coho, and from 6.9 to 8.9 fish per 1,000 m^2 for marked coho, although neither of these differences were statistically significant ($0.3303<p<0.7791$). For chum salmon, density also tended to decline from 2.3 fish per 1,000 m^2 in 2008 to 0.29 fish per 1,000 m^2 in 2009, though again the change was not statistically significant ($p = 0.2628$). Steelhead were found at Franz Lake in 2009 only. With the exception of high densities of marked Chinook in 2008, densities of different groups of salmonids were fairly similar at Franz Lake.

In both 2008 and 2009, unmarked juvenile Chinook were present at Franz Lake from the start of sampling in April through May, though in April of 2008 only one unmarked Chinook (2.9% of the Chinook catch) was captured (Figure 79). The pattern was similar for marked Chinook (Figure 79)—fish were present in April of 2008 and both April and May of 2009, but were not observed after that. Both marked and unmarked Chinook may also have been utilizing this site in June, but no data are available because it was not possible to sample in either year due to high water levels. No juvenile Chinook salmon were encountered in July or August in either sampling year.

Both marked and unmarked coho salmon were collected in May only in 2008, and in April only in 2009 (Figure 80). Densities of marked coho were consistently higher than those of unmarked coho in both months and both years. Chum salmon were found at Franz Lake in both April and May in 2008, and in

May only in 2009 (Figure 81). They were somewhat more abundant in 2008 than in 2009, with an average density of 2.3 fish per 1,000 m^2 in 2008 as compared to 0.29 fish per 1,000 m^2 in 2009. The small numbers of steelhead trout collected at Franz Lake were all found in 2008, in April and July, with the peak density in July (Figure 81).

Length, Weight, and Condition Factor

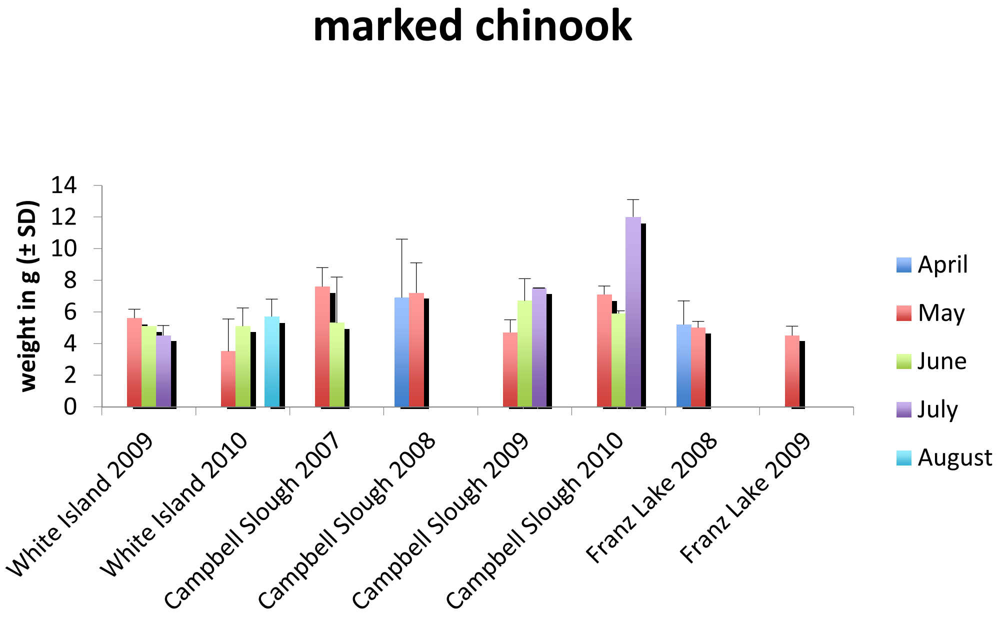
Chinook length, weight, and K by month and year for Whites Island, Campbell Slough, and Franz Lake are shown in Figures 60 to 62. A multiple regression analysis including month of capture, fish origin (marked versus unmarked), and sampling year was conducted for each of the fixed sites to determine which factors had the most effect on salmon length, weight, and K (Table 17). Large yearling Chinook (150–175 mm) captured early in the sampling season were not included in the analysis.

At the three fixed sites, length, weight, and K were generally affected by month of capture and fish origin (marked versus unmarked) (Table 17). At all three sites, both length and weight increased significantly over the sampling season. The same tended to be true for K, but the difference was not significant at Franz Lake. Marked fish tended to be larger and heavier and have higher K than unmarked fish, although these differences were not statistically significant at all three sites (Table 17).

Length, weight, and K varied significantly with year of capture at all three sites (Table 17; Figure 82, Figure 83, and Figure 84). At Whites Island, fish captured in 2009 were significantly larger and heavier than those captured in 2010, and condition was lower in fish captured in 2009 than in fish captured in 2010. At Campbell Slough, length was greater in fish captured in 2007 and 2008 than in those captured in 2009 and 2010, while weight was significantly higher in 2007 and lower in 2009 than in the other sampling years. Similarly, K was significantly higher in fish captured in 2007, and lower in fish captured in 2009, compared with other sampling years. At Franz Lake, fish captured in 2008 were significantly larger and had higher values of K than those captured in 2009. These analyses suggest differences in Chinook salmon size and condition from year to year at all three sampling sites, but no increasing or decreasing trends in weight, length, or condition over the sampling period. Chinook salmon size and condition tended to be lower in 2009 than in other sampling years at all three sites.

Table 17. Factors affecting Chinook salmon length, weight, and K at Whites Island, Campbell Slough, and Franz Lake.

	Month of Capture	Fish Origin	Year of Capture	Model R^2
Whites Island				
Length	$p<0.0001$	$p = 0.0010$	$p = 0.0001$	0.58, $p<0.0001$
Weight	$p<0.0001$	$p = 0.0010$	$p = 0.0001$	0.54, $p<0.0001$
K	$p<0.0001$	Ns	$p = 0.0017$	0.23, $p<0.0001$
Campbell Slough				
Length	$p<0.0001$	$p<0.0001$	$p = 0.0003$	0.57, $p<0.0001$
Weight	$p<0.0001$	$p<0.0001$	$p<0.0001$	0.47, $p<0.0001$
K	$p<0.0001$	$p = 0.0123$	$p<0.0001$	0.29, $p<0.0001$
Franz Lake				
Length	$p<0.0001$	Ns	$p = 0.0154$	0.77, $p<0.0001$
Weight	$p<0.0001$	$p = 0.0010$	$p = 0.0307$	0.64, $p<0.0001$
K	Ns	$p = 0.0424$	$p<0.0424$	0.18, $p = 0.0046$



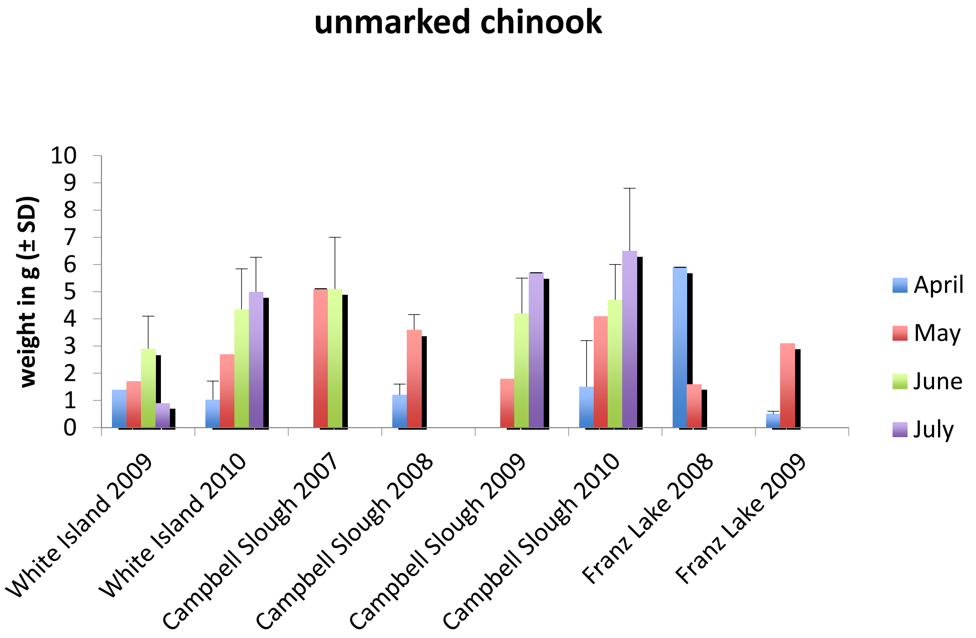
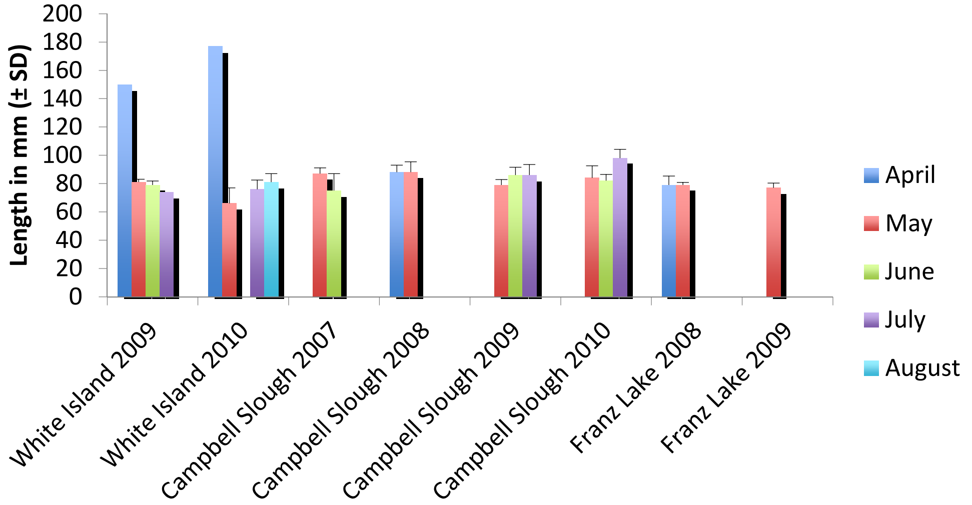
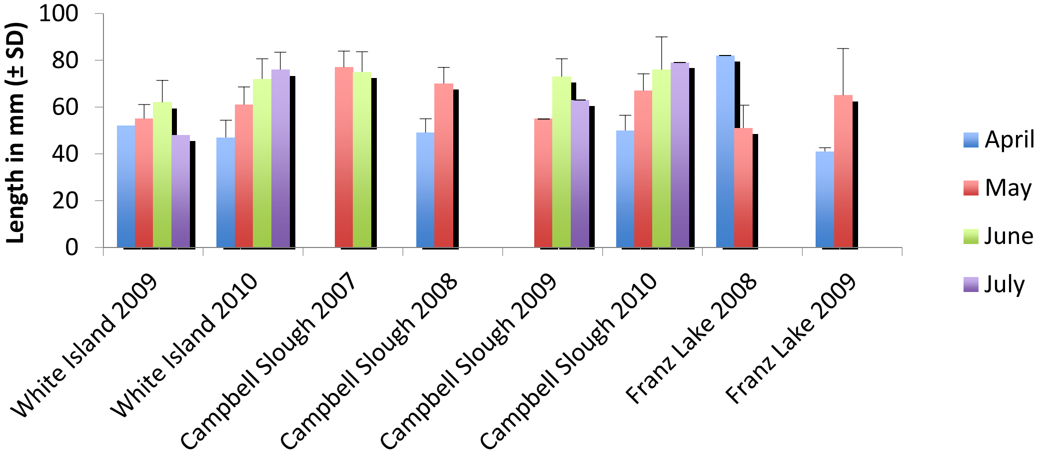


Figure 82. Weight in g of marked and unmarked Chinook from Franz Lake and Campbell Slough over time. No unmarked Chinook were caught in August at any of the fixed sites.

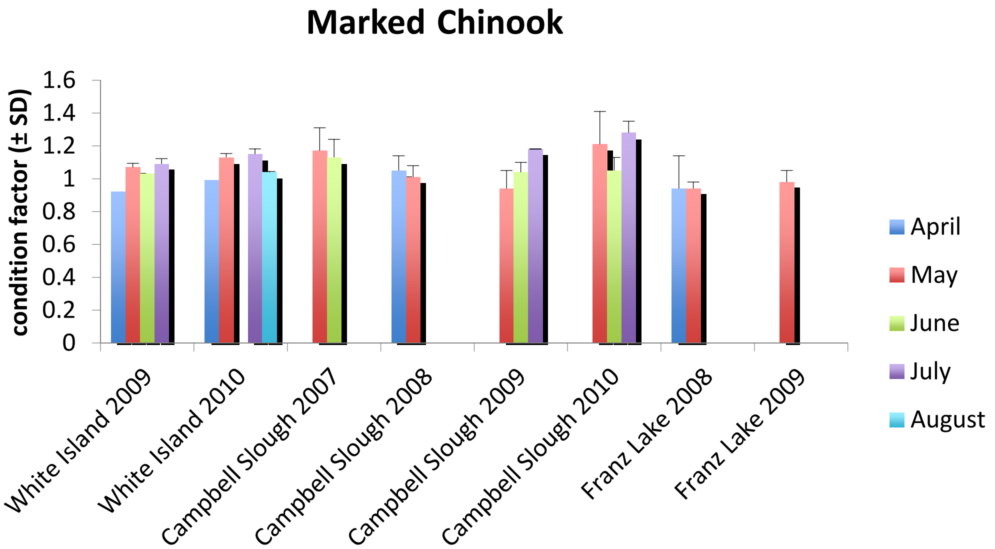


Marked Chinook



Unmarked Chinook

Figure 83. Lengths in mm of marked and unmarked Chinook from Franz Lake and Campbell Slough over time. No unmarked Chinook were caught in August at any of the fixed sites.



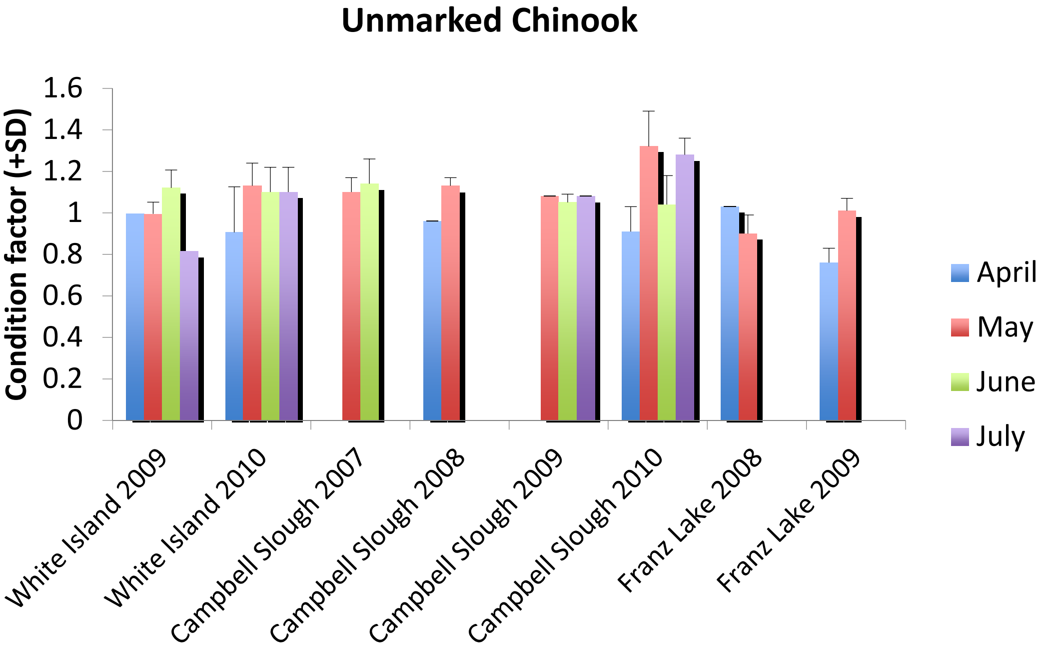


Figure 84. K of marked and unmarked Chinook from Franz Lake and Campbell Slough over time. No unmarked Chinook were caught in August at any of the fixed sites.

Otolith Analyses for Growth Rate Determination

Chinook salmon from Campbell Slough showed no significant differences in growth among the sampling years (Figure 85). Franz Lake fish grew faster in 2008 than 2009, but significant differences were only detected for the last 21 days of growth. Data for Whites Island 2010 are not yet available for evaluation of temporal trends.

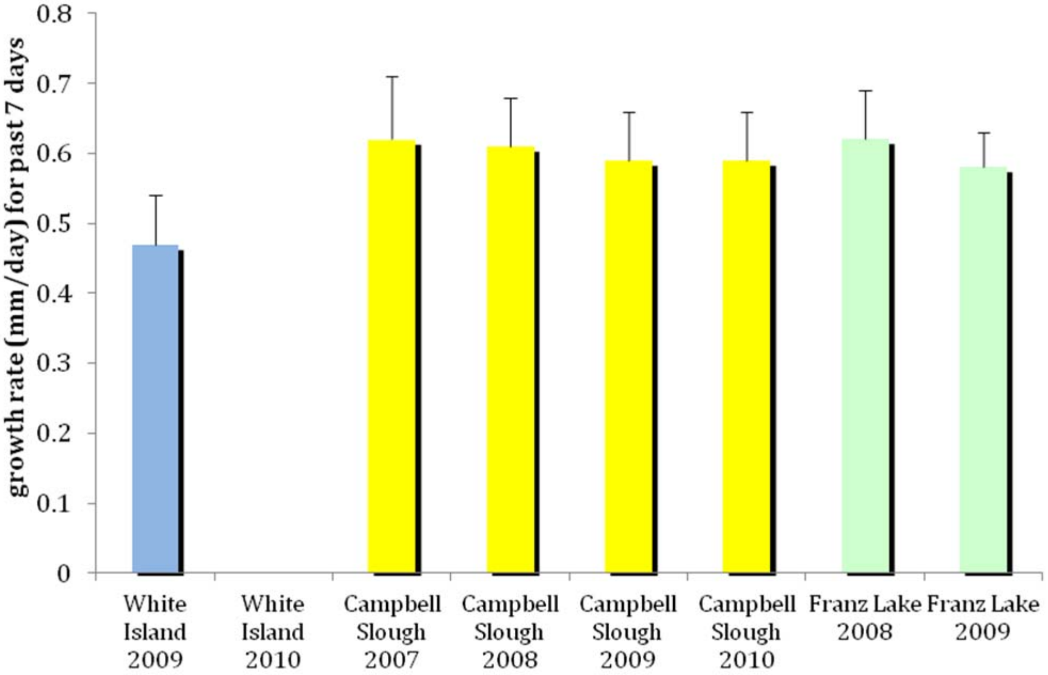
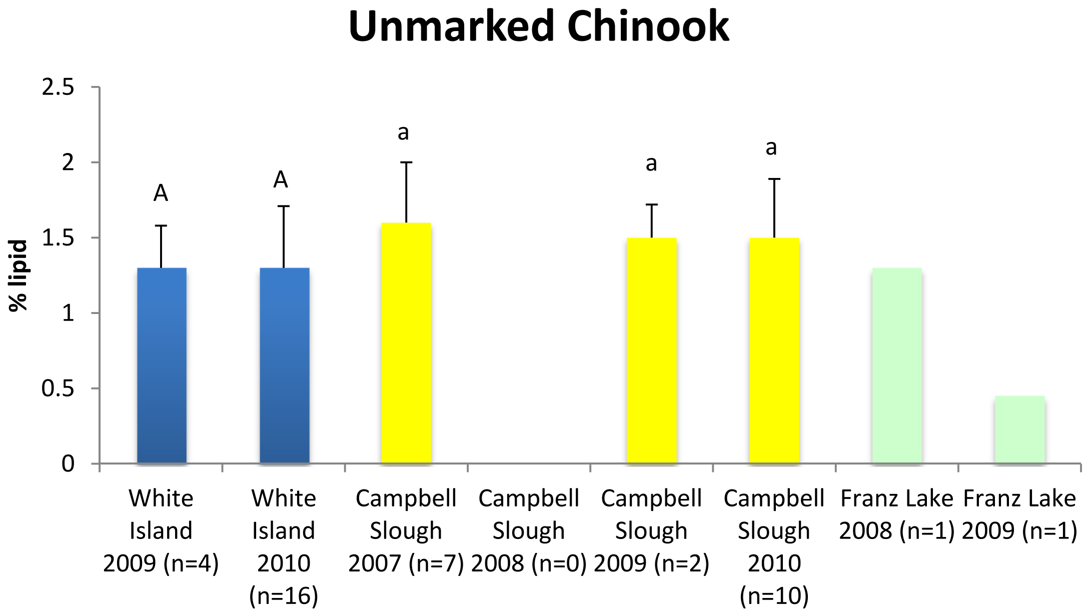


Figure 85. Average daily growth rates for seven-day periods estimated from otoliths for juvenile fall Chinook salmon from Franz Lake and Campbell Slough.

Chinook Salmon Lipid Content

At this point we have data on lipid content for Chinook salmon samples collected from Whites Island in 2009 and 2010, Franz Lake in 2008 and 2009, and from Campbell Slough in 2007 through 2010 (Figure 86). Analyses of whole bodies for lipid classes are still in progress for the subyearling Chinook salmon collected at Campbell Slough and Whites Island in 2010. For this reason, only total lipid content will be presented.

Because lipid content showed different patterns in marked and unmarked Chinook, trends for the two groups at the fixed sites were analyzed separately. Among unmarked fish, no significant year-to-year differences were observed at any of the sites, although the number of samples from unmarked fish was quite low for some sites and years. Among marked fish at Franz Lake, lipid content was significantly lower in 2009 than in 2008, and at Campbell Slough, lipid content was significantly lower in 2007 and 2009 than in 2008 and 2010. At Whites Island, there was no significant difference in lipid content of marked fish in 2009 and 2010.



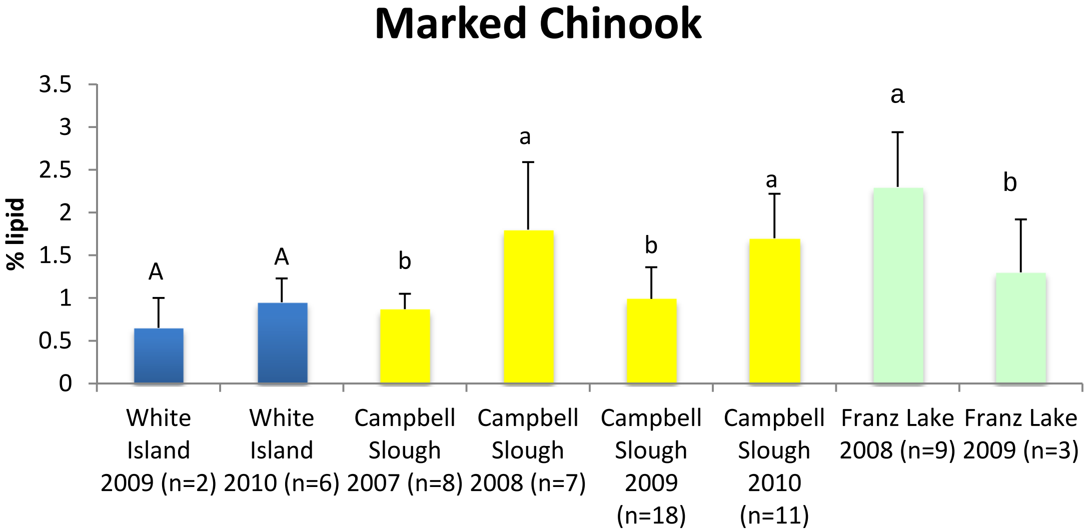
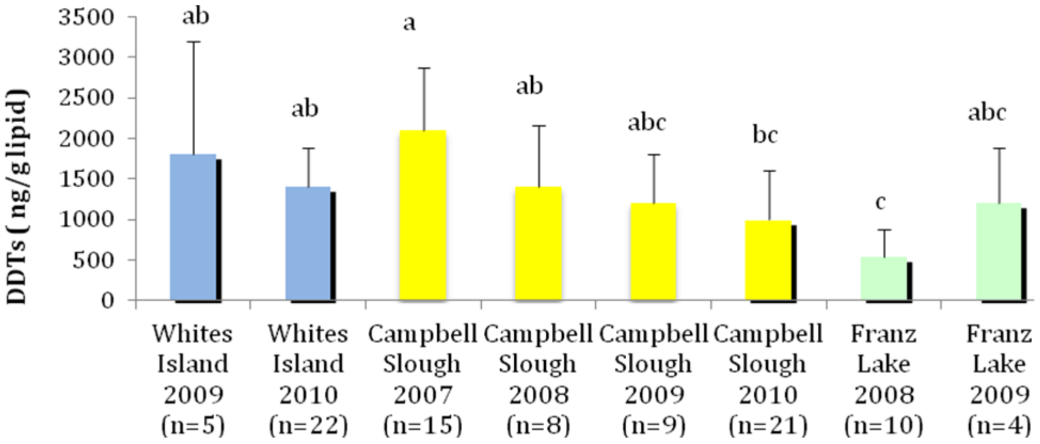
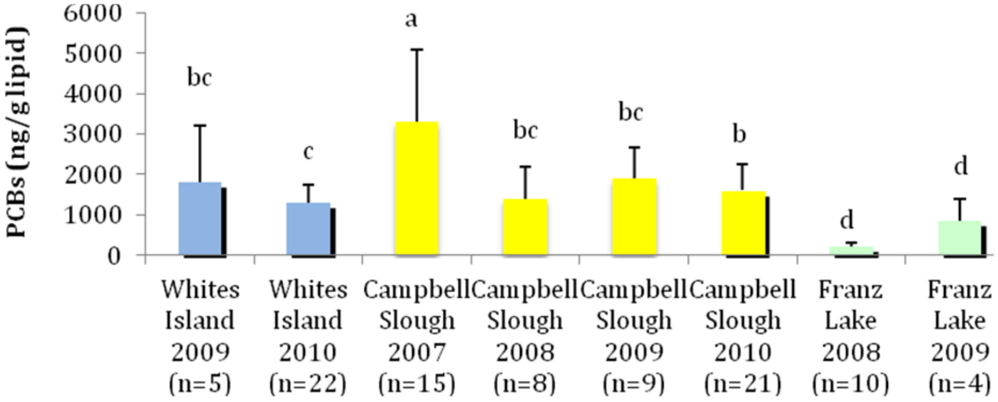


Figure 86. Lipid content of juvenile Chinook salmon from Franz Lake and Campbell Slough. Mean values were compared by year for each site. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, $p<0.05$). No statistical analysis was conducted for Franz Lake samples from unmarked Chinook because only one composite sample was analyzed for each year.

Contaminants in Chinook Salmon

POPs in Chinook. The major contaminants Chinook at Franz Lake were DDTs, although low levels of PBDEs and PCBs were also detected (Figure 87). Chinook from Franz Lake had the lowest concentrations of all three classes of contaminants compared with Campbell Slough or Whites Island. Contaminant concentrations did not change significantly in Chinook from Franz Lake between 2008 and 2009.





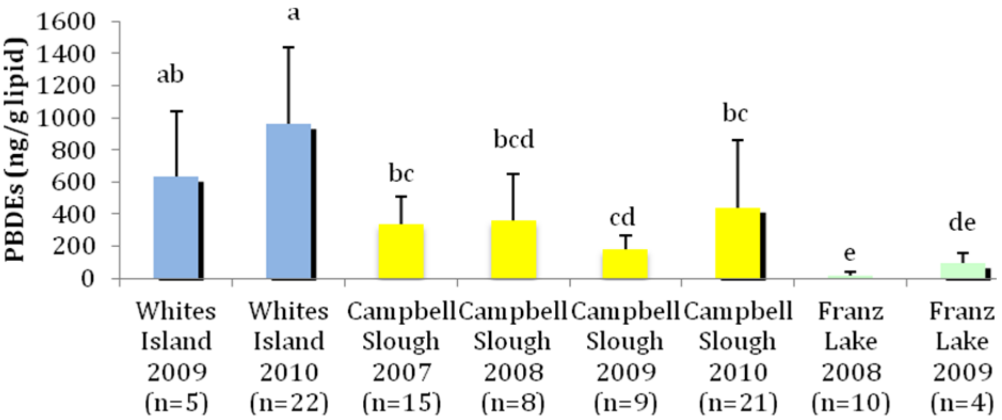


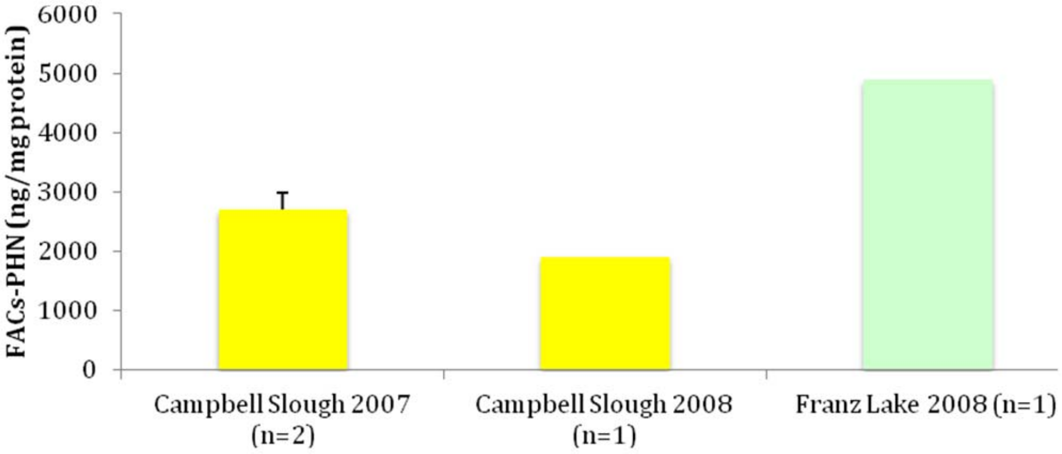
Figure 87. POPs (DDTs, PCBs, and PBDEs) in ng/g wet weight in juvenile Chinook salmon bodies from Whites Island, Franz Lake, and Campbell Slough. Different letters above columns indicate statistically significant differences among values (1-way ANOVA, Tukey's least significant difference test, $p<0.05$).

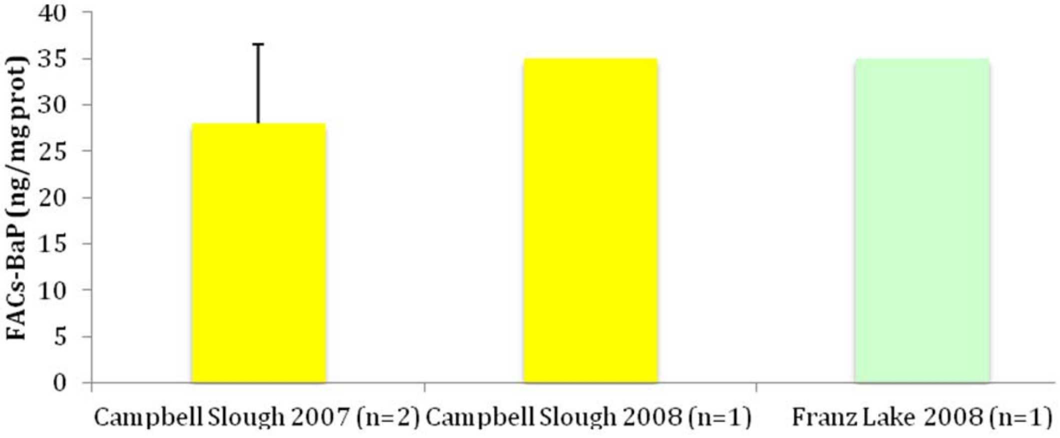
Chinook collected from Campbell Slough in 2007 had the highest concentrations of DDTs and PCBs observed at this group of sites during the sampling period. Concentrations of DDTs declined significantly between 2007 and 2010. Concentrations of PCBs also declined significantly between 2007 and 2008, and did not change significantly in the following years. Concentrations of PBDEs did not change significantly between 2007 and 2010.

In Chinook sampled from Whites Island in 2009 and 2010, concentrations of DDTs were comparable to those observed sampled from Campbell Slough in the same year. The same was true for PCBs in 2009, but concentrations at Whites Island were lower than those at Campbell Slough in 2010. Concentrations of both PCBs and DDTs tended to be lower than those observed in Chinook from Franz Lake, but differences were statistically significant only for PCBs. Whites Island Chinook had significantly higher PBDE concentrations than Chinook from Campbell Slough and Franz Lake in the same years. Concentrations of DDTs, PCBs, and PBDEs did not change significantly at Whites Island between 2009 and 2010, although PBDE concentrations were somewhat higher in 2010 than in 2009.

In Chinook from all sites, DDTs in all samples from all years were below the estimated effect threshold of 6,000 ng/g lipid, proposed by Beckvar et al. (2005), modified for lipid content as suggested in Johnson et al. (2007). At Campbell Slough one of the eight samples analyzed in 2008 (12.5%) and one of 21 samples analyzed in 2010 (4.8%) had PBDE concentrations above the 940 ng/g lipid level associated with immunosuppression in juvenile salmon (Arkoosh et al. 2010). However, 67% of samples collected from Campbell Slough in 2007 had PCB concentrations above the estimated effect threshold for juvenile salmon (Meador et al. 2002). The percentage of samples above this threshold declined to 12.5% in 2008, 33% in 2009, and 9.5% in 2010, representing a significant decline in the percentage of affected fish (Heterogeneity G-statistic, $p<0.05$). At Whites Island, 20% of samples collected in 2009 and 4.6% of samples collected in 2010 had PCB concentrations above 2,400 ng/g lipid, and 20% of 2009 samples and 45% of 2010 samples had PBDE concentrations above the 940 ng/g lipid level associated with immunosuppression.

Metabolites of PAHs in Bile. Only a limited number of bile samples were available for analysis of PAH exposure from the EMP fixed sites. Available data for Franz Lake and Campbell Slough are shown in Figure 66. No samples could be collected from Whites Island. Levels of FACs-NPH and FACs-BaP in bile samples from Franz Lake and Campbell Slough were similar (Figure 88), but the FACs-PHN level in bile from Franz Lake was about twice the level of FACs-PHN in bile samples from Campbell Slough (4,900 ng/mg protein versus 1,900–2,700 ng/mg protein). In fish from Campbell Slough, levels of FACs-PHN and FACs-NPH tended to be lower in 2008 than in 2007, but levels of FACs-BaP were about the same. However, in part because of the small number of samples available for analysis, no statistically significant differences were observed in bile metabolite levels by sampling year or site.





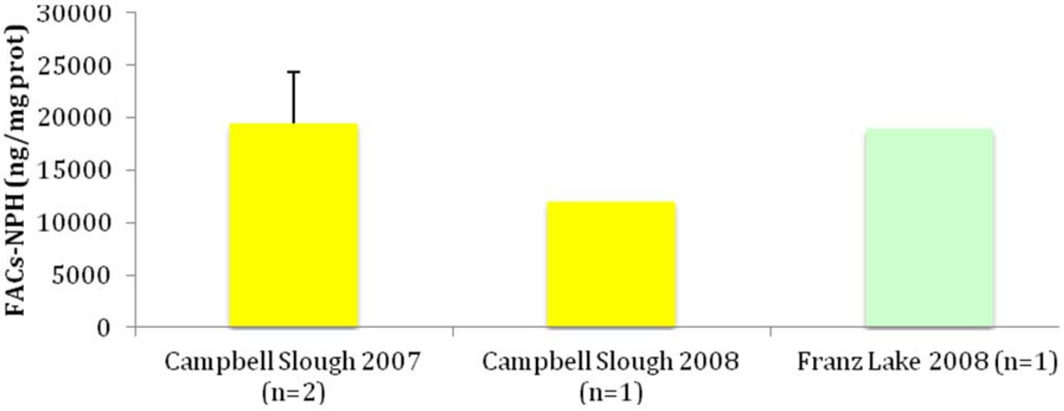


Figure 88. Concentrations of FACs measured at PHN, BaP, and NPH wavelengths (FACs-PHN, FACs-BaP, and FACs-NPH) in bile of juvenile Chinook salmon from Franz Lake and Campbell Slough. No significant differences were found between bile metabolite levels among sites or sampling years.

3.2.3 Abiotic Site Conditions

The melting of the large snowpack in the basin in 2008 caused extremely high water levels in mid-May and into June. This led to delays in the deployment of the monitor at Campbell Slough because access to the site was hindered, and the deployment design had to be modified to accommodate these high water levels. The modified deployment apparatus presented issues once the water levels dropped as well, causing the monitor to be left “high and dry.” During the July salmonid sampling, NOAA-Fisheries did not find any salmonids and decided to conclude their sampling at the site for the year. Therefore, the monitor was removed from the site rather than adjusting the deployment design to accommodate the lower water levels. This resulted in a deployment duration of roughly one month in 2008, but with only about 12 days of acceptable data. For this reason, this analysis is focused on 2009 and 2010. In 2009, the monitors were deployed from May 7 through August 21, and in 2010, from April 1 through July 30. Daily average values for each water quality parameter during each month are shown in Table 18 (2009) and Table 19 (2010). It is important to note that the monitor at Campbell Slough is placed at a fixed location and does not change position in the water column as the water level rises and falls. Water quality standards used for comparisons are from the Washington Department of Ecology (Washington Department of Ecology 2011) and the Oregon Department of Environmental Quality (DEQ; Oregon DEQ 2010).

Table 18. Average daily minimum, mean, median, and maximum water quality values by month in Campbell Slough, May 7 to August 21, 2009.

2009		April	May	June	July	August
Temperature (°C)	Daily min	-	13.2	17.2	20.2	20.7
	Daily mean	-	15.2	18.8	23.6	23.2
	Daily median	-	15.0	18.8	23.5	22.9
	Daily max	-	17.7	20.9	27.4	26.5
pH (standard units)	Daily min	-	7.8	7.4	8.3	7.7
	Daily mean	-	8.0	7.7	8.9	8.2

	Daily median	-	8.0	7.7	8.9	8.2
	Daily max	-	8.2	8.1	9.4	8.9
Dissolved oxygen (mg/L)	Daily min	-	9.4	7.2	7.5	5.1
	Daily mean	-	11.1	9.2	10.6	8.3
	Daily median	-	11.1	9.2	10.4	8.0
	Daily max	-	12.5	10.7	13.5	11.6
Specific conductance (µS/cm)	Daily min	-	147	130	126	147
	Daily mean	-	151	138	135	152
	Daily median	-	151	138	136	153
	Daily max	-	157	146	145	158

°C, degrees Celsius; mg/L, milligrams per liter; µS/cm, microSiemens per centimeter

Table 19. Average daily minimum, mean, median, and maximum water quality values by month in Campbell Slough, April 1 to July 30, 2010.

2010		April	May	June	July	August
Temperature (°C)	Daily min	10.0	12.9	15.6	18.9	-
	Daily mean	11.9	15.2	16.5	20.3	-
	Daily median	11.9	15.4	16.5	20.1	-
	Daily max	14.2	17.5	17.6	22.4	-
pH (standard units)	Daily min	7.9	7.8	7.2	7.2	-
	Daily mean	8.3	8.2	7.3	7.4	-
	Daily median	8.3	8.2	7.3	7.4	-
	Daily max	8.7	8.6	7.5	7.7	-
Dissolved oxygen (mg/L)	Daily min	11.2	9.6	6.3	4.7	-
	Daily mean	13.1	11.5	8.3	6.2	-
	Daily median	13.1	11.4	8.3	6.2	-
	Daily max	14.9	13.5	9.9	7.4	-
Specific Conductance (µS/cm)	Daily min	173	159	142	140	-
	Daily mean	178	166	147	146	-
	Daily median	177	166	147	144	-
	Daily max	183	171	152	158	-

°C, degrees Celsius; mg/L, milligrams per liter; µS/cm, microSiemens per centimeter

Temperature

2009. In-stream temperature at Campbell Slough ranged from 10.5°C to 34.4°C during the 2009 monitoring period. Continuous temperature data are shown in Figure 89. The seven-day maximum temperature ranged from 15.6°C to 31.9°C, averaging 23.3°C. The Washington seven-day maximum standard of 17.5°C was exceeded for the entire time period, except for seven days in May (Figure 90).

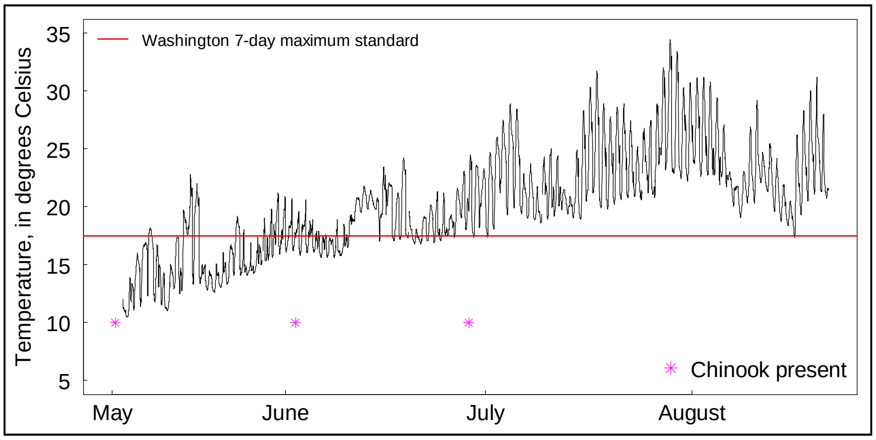


Figure 89. Graph of continuous temperature data from Campbell Slough, May 7 to August 21, 2009. The Washington weekly maximum temperature standard is shown in red. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

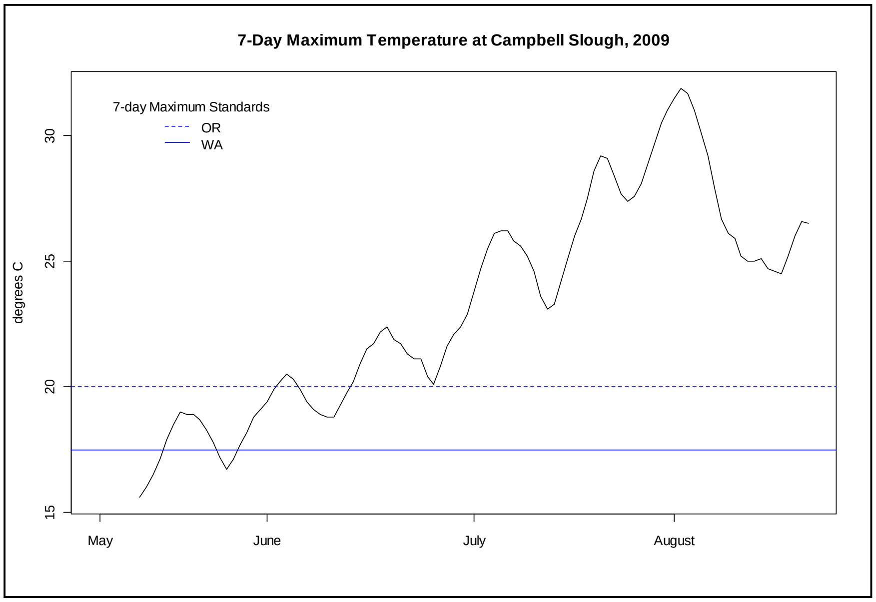


Figure 90. Graph of weekly maximum temperature data from Campbell Slough, May 7 to August 21, 2009. Oregon and Washington weekly maximum temperature standards are shown in blue.

2010. Water temperature ranged from 7.8°C to 25.6°C during the 2010 monitoring period (Figure 91). It increased throughout the period, exceeding the Washington seven-day maximum temperature standard of 17.5°C in mid-May and in late June through July (Figure 92). Nevertheless, Chinook salmon were found at the site on July 6.

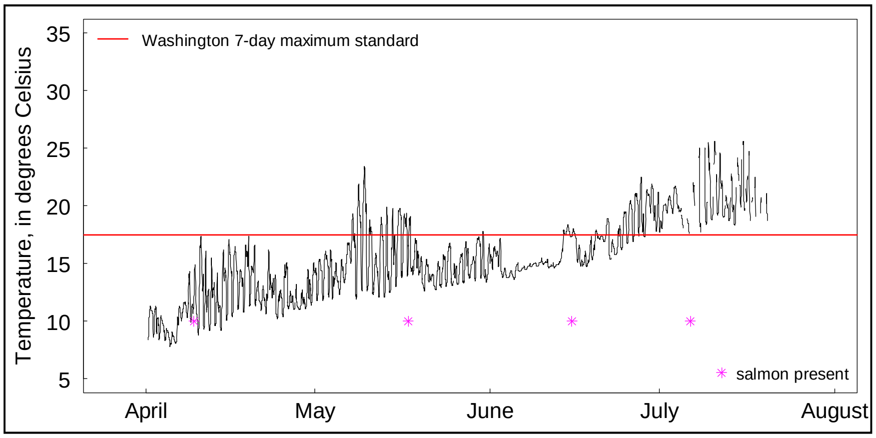


Figure 91. Graph of continuous temperature data from Campbell Slough, April 1 to July 30, 2010. The Washington weekly maximum temperature standard is shown in red. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

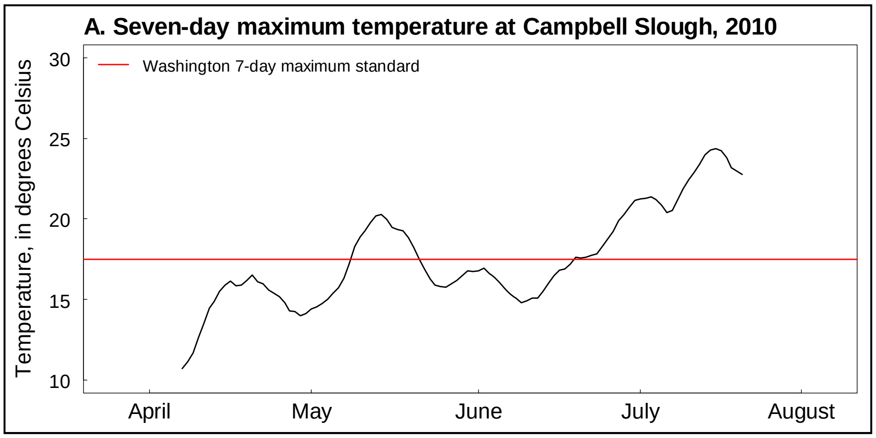


Figure 92. Graph of weekly maximum temperature data from Campbell Slough, April 1 to July 30, 2010. The Washington weekly maximum temperature standard is shown in red.

2009 and 2010 Comparison. The average daily median temperature in 2010 was within one degree of the 2009 daily median temperature in May, about two degrees lower during June, and three degrees lower during July. Differences in average daily maximum temperature between the two years spanned from 0.2 degrees (May) to five degrees (July). In 2010, 40% of days with data available during May to July (*n* = 81) had seven-day maximum temperatures meeting the state standard, compared with 9% in 2009 (*n* = 80).

pH

2009. In 2009, pH ranged from 6.9 to 10.0 standard units, averaging 8.2 (Figure 93). The daily minimum pH was below Oregon's standard of 7.0 on three days during the summer (Figure 94); Washington's minimum standard of 6.5 was never violated. However, 50 days (most of July and August) had daily maximums exceeding both states' maximum standard of 8.5.

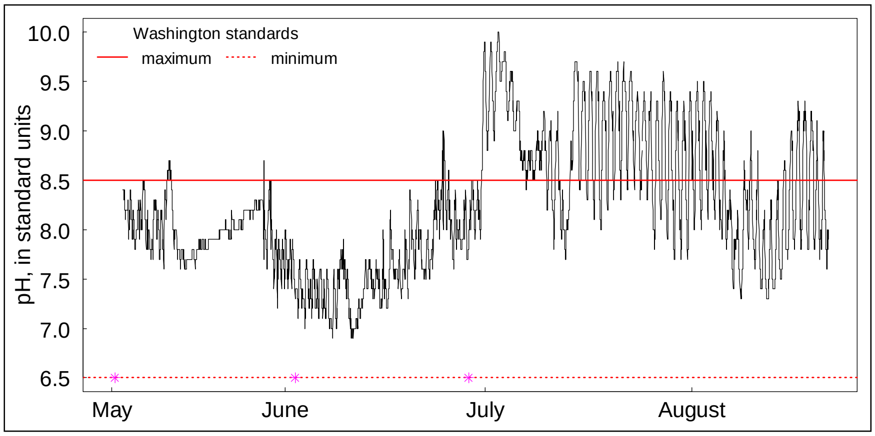


Figure 93. Graph of continuous pH data from Campbell Slough, May 7 to August 21, 2009. The Washington minimum and maximum pH standards are shown in red. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

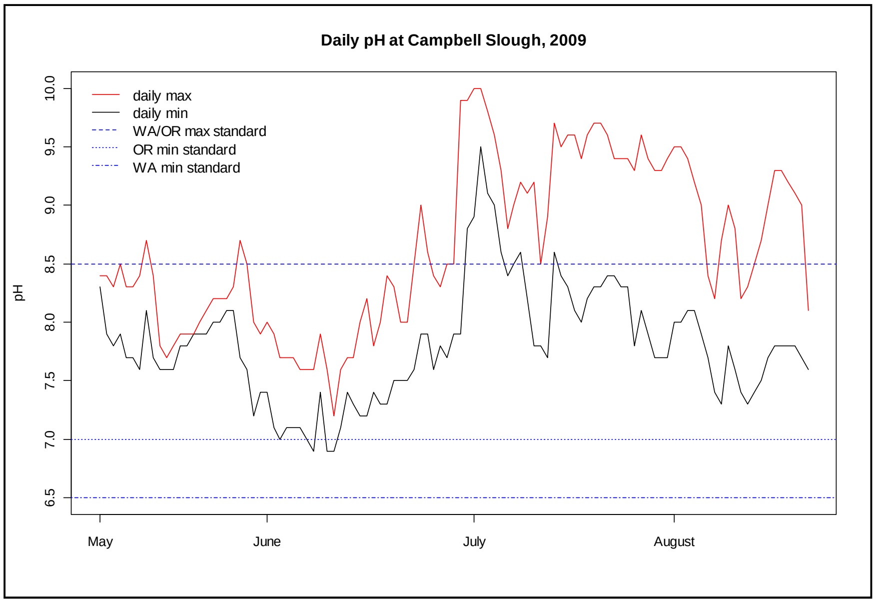


Figure 94. Graph of daily minimum and maximum pH data from Campbell Slough, May 7 to August 21, 2009. Oregon and Washington pH standards are shown in blue.

2010. In 2010, pH ranged from 6.8 to 9.6 standard units, averaging 7.2 (Figure 95). The daily maximum pH exceeded the Washington water quality standard of 8.5 (Washington Department of Ecology 2011) during mid-April through mid-May. After peaking in April and May, pH decreased from mid-May through June and rose through early July (Figure 96). Washington's minimum pH standard was not violated during the 2010 monitoring period.

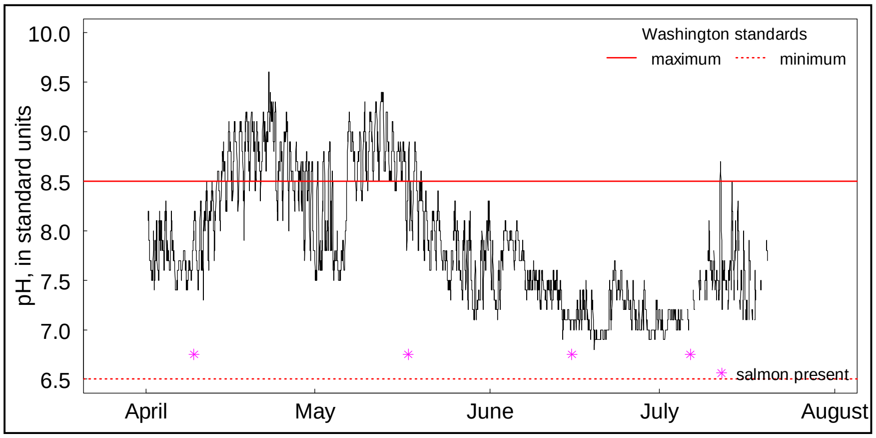


Figure 95. Graph of continuous pH data from Campbell Slough, April 1 to July 30, 2010. The Washington minimum and maximum pH standards are shown in red. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

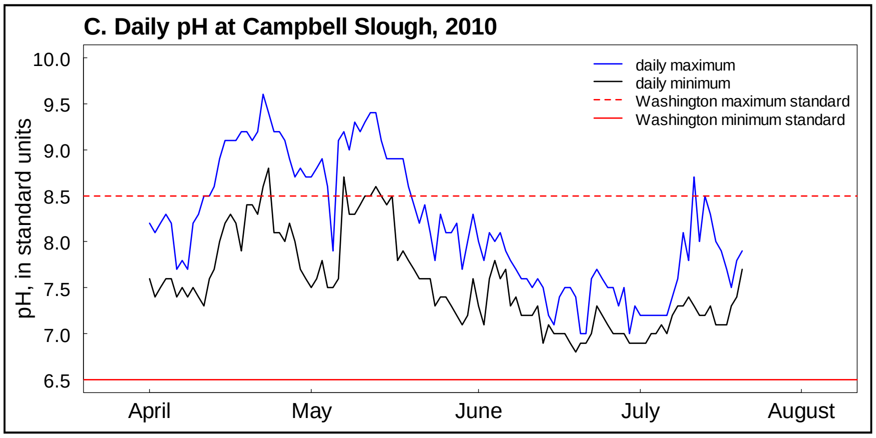


Figure 96. Graph of daily minimum and maximum pH data from Campbell Slough, April 1 to July 30, 2010. Washington pH standards are shown in red.

2009 and 2010 Comparison. The monitoring periods in 2009 and 2010 had opposite trends in pH—in 2009, pH was lower in the spring, rose through June, and peaked in July, whereas in 2010, it peaked in the spring, fell through June, and increased somewhat in July. Differences in minimum, median, and maximum daily averages were largest in July. Warmer temperatures in July 2009 compared to 2010 could have spurred more productivity, resulting in these differences in July pH.

Dissolved Oxygen

2009. Dissolved oxygen ranged from 2.9 to 16.6 mg/L, averaging 9.8 mg/L in 2009 (Figure 97). Washington's daily minimum standard of 8.0 mg/L was not met on 58% of days, primarily throughout July and August, but also during mid-May and early June (Figure 98).

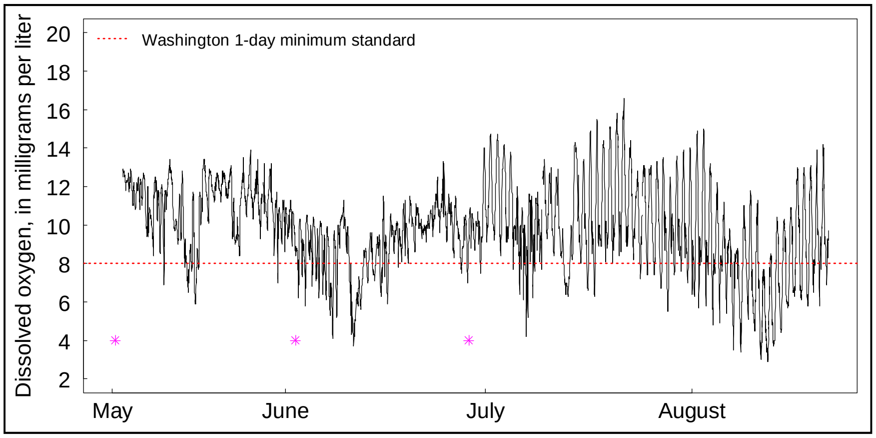


Figure 97. Graph of continuous dissolved oxygen data from Campbell Slough, May 7 to August 21, 2009. The Washington daily minimum standard is shown in red. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

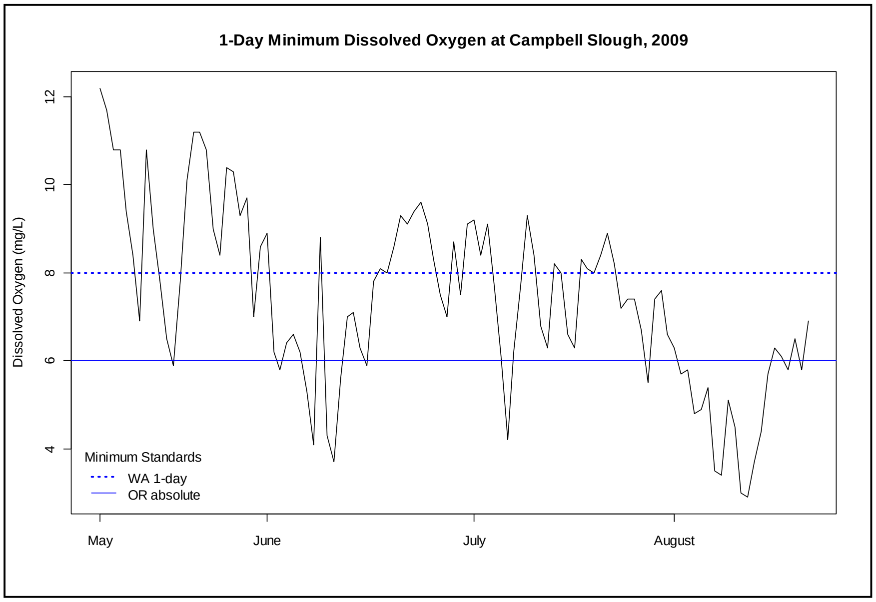


Figure 98. Graph of daily minimum dissolved oxygen data from Campbell Slough, May 7 to August 21, 2009. Oregon and Washington dissolved oxygen standards are shown in blue.

2010. Dissolved oxygen spiked in mid-April and mid-May 2010, decreasing through June and rising again in July, although at a much lower concentration than in the spring (Figure 99). It ranged from 1.8 to 19.5 mg/L from April to July, averaging 10.5 mg/L. The Washington daily minimum dissolved oxygen standard of 8.0 mg/L was consistently not met from mid-June through July (Figure 100).

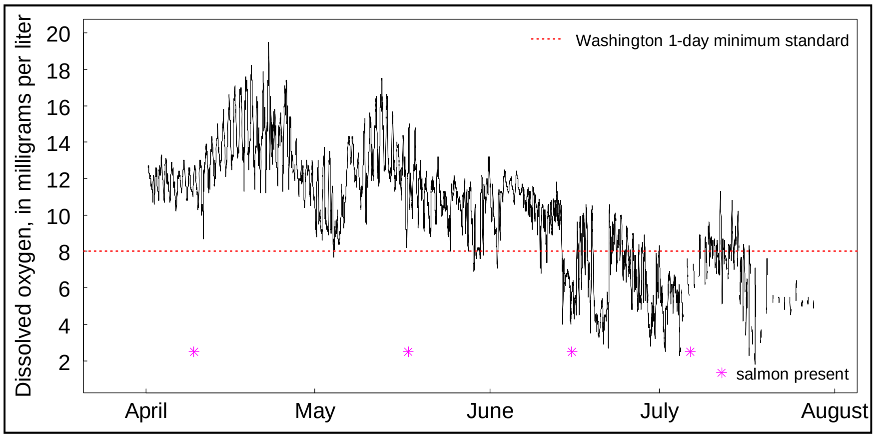


Figure 99. Graph of continuous dissolved oxygen data from Campbell Slough, April 1 to July 30, 2010. The Washington daily minimum standard is shown in red. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

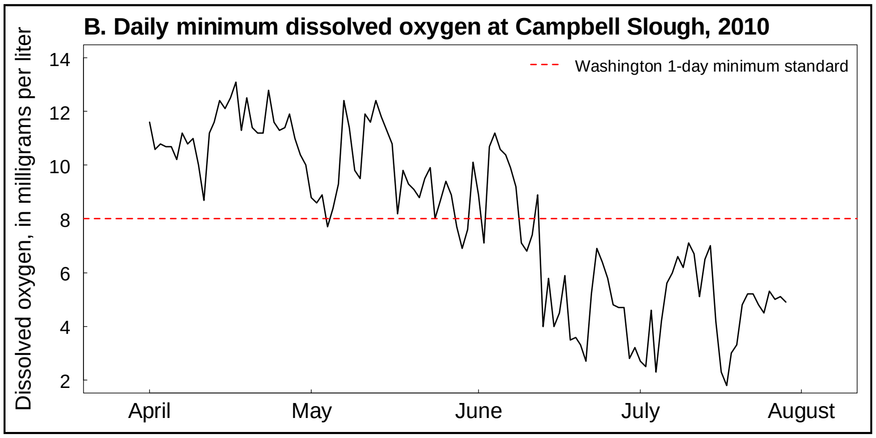


Figure 100. Graph of daily minimum dissolved oxygen data from Campbell Slough, April 1 to July 30, 2010. The Washington daily minimum standard is shown in red.

2009 and 2010 Comparison. In 2010, average daily median dissolved oxygen concentrations were equivalent (May) or less than 2009 values by 1 mg/L (June) to 4 mg/L (July). The average daily minimum dissolved oxygen concentrations were lower for June and July 2010 than for the same period in 2009. This difference could be due to higher temperatures in June and July 2009 resulting in greater primary production than during those months in 2010. The higher pH observed in June and July 2009 compared to 2010 supports this assessment. Additionally, pH and dissolved oxygen observations in May indicate that a peak in primary production occurred during that month in 2010, but not 2009. Therefore, there may have been more biomass available in June and July 2010 to decompose, which would have decreased the dissolved oxygen concentration in the water. Because the sensor was in a fixed position relative to the channel bottom (rather than relative to the water surface), higher water levels in June 2010 meant that the sensor measurements were taken from lower in the water column compared to June 2009. In slow-moving backwaters, lower dissolved oxygen concentrations would be expected at greater depths, especially when there is more biomass decomposing lower in the water column.

Specific Conductance

2009. Specific conductance ranged from 95 to 187 microSiemens per cm (µS/cm), averaging 143 µS/cm (Figure 101). The daily median specific conductance ranged from 136 to 153 µS/cm. State water quality standards do not exist for specific conductance.

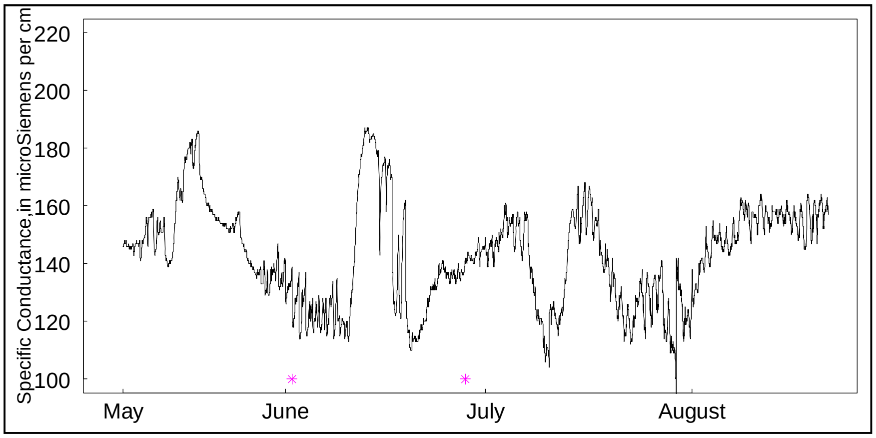


Figure 101. Graph of continuous specific conductance data from Campbell Slough, May 7 to August 21, 2009. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

2010. During 2010 monitoring, specific conductance ranged from 121 to 216 µS/cm and averaged 161 µS/cm (Figure 102). Average daily median specific conductance ranged from 144 to 177 µS/cm. Although it fluctuated during the monitoring period, it generally rose through April, and then declined through June.

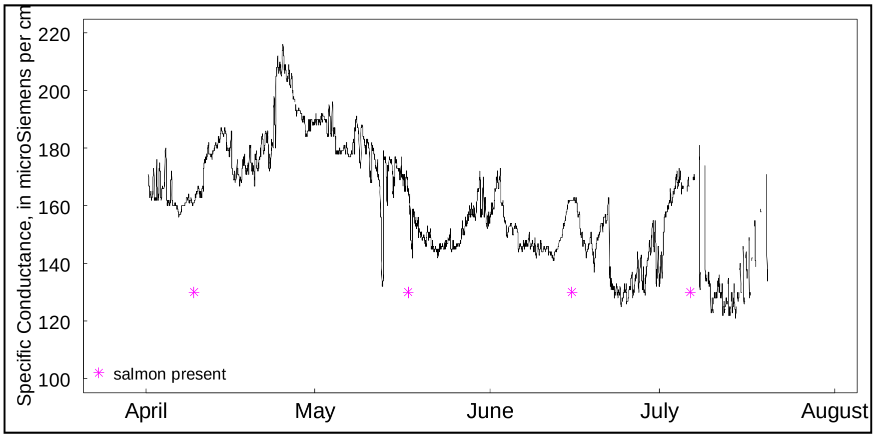


Figure 102. Graph of continuous specific conductance data from Campbell Slough, April 1 to July 30, 2010. Asterisks indicate when juvenile Chinook were present during NOAA-Fisheries' monthly fish sampling events.

2009 and 2010 Comparison. Specific conductance fluctuated during both years, perhaps due to irregular inputs and flushing at the site. The general trend from May through July was flat in 2009 and decreasing in 2010. Higher average and peak values were measured in 2010 than in 2009.

3.3 Interdisciplinary Relationships

3.3.1 Status Site Analyses

All Fish Species Diversity

Correlations were run between environmental variables and All Fish Species Diversity. Pearson's product-moment analysis indicated high correlation coefficients for Common Spikerush Coverage (*r* = .845; Table 20). Ten additional variables indicated moderate correlation coefficients and could possibly be associated with All Fish Species Diversity. Weak correlation coefficient values were produced for the remaining four variables.

Table 20. Correlation coefficients for All Fish Species Diversity and variables.

Variable	Correlation Coefficient
Common Spikerush Coverage	0.845
All Vegetation Diversity	−0.526
Native Vegetation Richness	−0.513
Open Water Prey Species Diversity	0.500
All Vegetation Richness	−0.492
Non-native Vegetation Diversity	−0.475
Wapato Coverage	0.464
River Kilometer	0.446
Native Vegetation Diversity	−0.444
Distance from Mainstem	0.438
Emergent Prey Species Diversity	0.408
Reed Canarygrass Coverage	−0.306
Non-native Vegetation Richness	−0.244
Elevation	0.196
All Prey Species Diversity	−0.038

Multiple regression models were generated between variables with an $r>0.4$ and All Fish Species Diversity. The two best multiple regression models were retained for presentation. Campbell Slough 2010 was found to be an outlier for the first model in the post-hoc diagnostic. The data were removed and, upon running the model again, the adjusted R-squared value improved.

Model Results. The first model indicated that All Fish Species Diversity is significantly and positively predicted by Common Spikerush Coverage ($\beta = 0.040768$, $p = .000428$) and Open Water Prey Species Diversity ($\beta = 0.466483$, $p = .041844$). These two predictors explain 78.4% of the variance ($R^2 = .82$, $F[2,9] = 21.0$, $p = .0004075$) in All Fish Species Diversity. Specifically, the diversity of all fish caught increases as the percent cover of common spikerush and species diversity of open-water invertebrate tows increases.

The second model indicated that All Fish Species Diversity is positively predicted by Common Spikerush Coverage ($\beta = 0.036148$, $p = .000267$) and Emergent Prey Species Diversity ($\beta = 0.101061$, $p = .68184$); however, the first relationship is significant while the latter is not. These two predictors explain 62.5% of the variance ($R^2 = .69$, $F[2,9] = 11$, $p = .002982$) in All Fish Species Diversity. Specifically, the diversity of all fish caught increases as the percent cover of common spikerush and, possibly, prey species diversity for emergent vegetation invertebrate tows increase.

Data from Campbell Slough 2009 indicated that this site had the highest All Fish Species Diversity and the second-highest abundance of Diptera sp. (Figure 59) and all macroinvertebrate availability combined (Figure 60). In general, Campbell Slough was consistently within the top five All Fish Species Diversity sites and the top 40% of prey and Diptera sp. abundance sites each year.

Native Fish Species Diversity

Correlations were run between environmental and biological variables and Native Fish Species Diversity. Pearson's product-moment analysis indicated high correlation coefficients for Common Spikerush Coverage ($r = .703$, Table 21). Seven additional variables indicated moderate correlation coefficients and could possibly be associated with Native Fish Species Diversity. Weak values were produced for the remaining seven variables.

Table 21. Correlation coefficients for Native Fish Species Diversity and variables.

Variable	Correlation Coefficient
Common Spikerush Coverage	0.703
Non-native Vegetation Richness	−0.657
River Kilometer	0.638
All Vegetation Richness	−0.602
Non-native Vegetation Diversity	−0.572
Native Vegetation Richness	−0.544
Reed Canarygrass Coverage	−0.531
All Vegetation Diversity	−0.480
Open Water Prey Species Diversity	0.381
Elevation	0.372
Native Vegetation Diversity	−0.333
Emergent Prey Species Diversity	0.319
Distance from Mainstem	−0.125
All Prey Species Richness	0.041
Wapato Coverage	−0.019

Multiple regression models were generated between variables with an $r>0.4$ and Native Fish Species Diversity. The best multiple regression model produced is presented below. Campbell Slough 2009 was removed from the model due to diagnostic problems. The model exhibits problems with the distance diagnostics as there are still two observations that are pulling the regression line in their direction. However, removal of these data points results in models that are not significant. Therefore, the two data points were retained in the analysis.

Model Results. The model indicated that Native Fish Species Diversity is positively predicted by Common Spikerush Coverage ($\beta = 0.014702$, $p = .0512$) and negatively predicted by All Vegetation Richness ($\beta = -0.014410$, $p = .1016$); however, the first relationship is significant while the latter is not. The two predictors explain 78.4% of the variance ($R^2 = .82$, $F[2,9] = 21$, $p = .0004075$) in Native Fish Species Diversity. Specifically, diversity of native fish caught increases as the percent cover of common spikerush increases and, possibly, as species richness for all native and non-native vegetation species decreases.

The highest Native Fish Species Diversity and the second-highest abundance of Diptera sp. (Figure 59) and all macroinvertebrate availability combined (Figure 60) at all sites were found at Campbell Slough 2009. In general, all Campbell Slough sites were consistently within the top 40% prey and Diptera sp. abundance sites each year.

Chinook Salmon Abundance

Correlations were run between environmental variables and Chinook Salmon Abundance (i.e., Chinook Salmon CPUE). Pearson’s product-moment analysis indicated moderate correlation coefficients with nine variables that could possibly be associated with Chinook Abundance (Table 22). Weak values were produced for the remaining nine variables.

Table 22. Correlation coefficients for Chinook Abundance and variables.

Variable	Correlation Coefficient
Emergent Prey Species Diversity	−0.645
All Vegetation Richness	0.636
Native Vegetation Richness	0.628
River Kilometer	−0.619
Elevation	−0.551
All Vegetation Diversity	0.498
Native Vegetation Diversity	0.495
Non-native Vegetation Richness	0.420
Native Fish Species Diversity	−0.412
All Fish Species Diversity	−0.329
Non-native Vegetation Diversity	0.292
Wapato Coverage	0.279
Common Spikerush Coverage	−0.271
Distance From Mainstem	−0.230
Reed Canarygrass Coverage	−0.205
Open Water Prey Species Diversity	0.190
All Prey Species Richness	0.146
Non-native Fish Species Diversity	−0.124

Model Results. Multiple regression models were generated between variables with an $r>0.4$ and Chinook Salmon Abundance. The two best multiple regression models were retained for presentation. The first model indicated that Chinook Salmon Abundance is significantly and positively predicted by All Vegetation Richness ($\beta = 0.7824$, $p = .0173$) and is significantly and negatively predicted by Emergent Prey Species Diversity ($\beta = -14.6088$, $p = .0157$). The two predictors explain 61.3% of the variance ($R^2 = .67$, F[2,10] = 10.49, $p = .003506$) in Chinook Abundance. Specifically, Chinook CPUE increases as the number of vegetation species increases and the diversity of prey species in emergent vegetation invertebrate tows decreases.

The second model indicated that Chinook Abundance is significantly and negatively predicted by River Kilometer ($\beta = -0.11423$, $p = .0225$) and Emergent Prey Species Diversity ($\beta = -14.7582$, $p = .0168$). The two predictors explain 59.4% of the variance ($R^2 = .66$, F[2,10] = 9.765, $p<.004453$) in Chinook Abundance. Specifically, Chinook CPUE decreased as the distance from the mouth of the Columbia River and the diversity of prey species in emergent vegetation invertebrate tows increases.

Chinook Salmon Health (Lipid Levels)

Correlations were run between environmental variables and All Chinook Lipid Content. Pearson’s product-moment analysis indicated moderate correlation coefficients with six variables that could possibly be associated with All Chinook Lipid Content (Table 23). Weak values were produced for the remaining six variables.

Table 23. Correlation coefficients for All Chinook Lipid Content and variables.

Variable	Correlation Coefficient
Elevation	0.656
Non-native Vegetation Diversity	−0.487
All Vegetation Diversity	−0.466
Non-native Fish Species Diversity	0.423
Native Vegetation Diversity	−0.422
Open Water Prey Species	0.413
Native Fish Species Diversity	0.388
Common Spikerush Coverage	−0.299
Diptera Density in May Emergent Vegetation Tows	0.194
Wapato Coverage	0.129
Reed Canarygrass Coverage	−0.096
All Fish Species Diversity	0.091

Multiple All Chinook Lipid Content regression models were generated but none provided improved results over using elevation alone. All adjusted R-squared values were near 0.15, indicating a very weak relationship between All Chinook Lipid Content and the predictor. A small sample size is likely the cause of these weak models.

Unmarked Chinook Salmon Health (Lipid Levels)

Correlations were run between environmental variables and Unmarked Chinook Lipid Content. Pearson's product-moment analysis indicated moderate correlation coefficients with six variables that could possibly be associated with Unmarked Chinook Lipid Content (Table 24).Weak values were produced for the remaining eight variables.

Table 24. Correlation coefficients for Unmarked Chinook Lipid Content and variables.

Variable	Correlation Coefficient
Elevation	−0.637
Emergent Prey Species Diversity	0.512
Non-native Fish Species Diversity	0.503
Reed Canarygrass Coverage	0.469
Wapato Coverage	0.440
Native Vegetation Diversity	−0.431
Diptera Density in May Emergent Vegetation Tows	0.365
Common Spikerush Coverage	0.365
All Vegetation Diversity	−0.269
Open Water Prey Species	0.260
All Fish Species Diversity	−0.187
Native Fish Species Diversity	−0.187
Non-native Vegetation Diversity	0.163
Abundance of All May Prey	0.002

Model Results. Multiple regression models were generated between variables with r>0.4 and Unmarked Chinook Lipid Content. The best multiple regression model was retained for presentation. This model indicated that Unmarked Chinook Lipid Content is negatively predicted by Elevation (β = 0.2938, p = .2250) and is positively predicted by Emergent Prey Species Diversity (β = 0.3603, p = .3395); however, neither relationship is significant. The two predictors explain 30.6% of the variance (R^2 = .58, F[2,3] = 2.103, p = .2686) in Unmarked Chinook Lipid Content. Specifically, the percent gravimetric lipid content collected from a subset of unmarked Chinook may increase as elevation decreases and as prey species diversity for emergent vegetation invertebrate tows increases.

Multiple data points were found to be outliers in the post-hoc diagnostic. This could not be corrected due to the small sample size. The variables in the models may still have an influence on unmarked Chinook lipid levels but interpretation should be used with caution.

3.3.2 Trend Site Analyses

Chinook Salmon Abundance

Correlations were run between environmental variables and Chinook Salmon Abundance at sites sampled in multiple years. Pearson's product-moment analysis indicated moderate correlation coefficients with seven variables that could possibly be associated with Chinook Abundance (Table 25). Weak values were produced for the remaining 13 variables.

Table 25. Correlation coefficients for Chinook Abundance and variables.

Variable	Correlation Coefficient
Reed Canarygrass Coverage	0.599
Diptera Density in May Emergent Vegetation Tows	0.575
Non-native Vegetation Richness	0.530
Non-native Vegetation Diversity	0.476
All Prey Species Richness	0.462
Open Water Prey Species Diversity	0.459
Abundance of All May Prey	0.431
Native Vegetation Diversity	−0.309
All Vegetation Richness	0.301
Wapato Coverage	0.292
River Kilometer	−0.215
Non-native Fish Species Diversity	0.189
Native Fish Species Diversity	0.160
Native Vegetation Richness	0.151
Elevation	−0.113
All Fish Species Diversity	−0.099
Native Vegetation Diversity	−0.032
Common Spikerush Coverage	−0.032
Emergent Prey Species Diversity	−0.007
Distance From Mainstem	0.001

Model Results. Multiple regression models were generated between variables with an r>0.4 and Chinook Salmon Abundance at sites sampled in multiple years. The two best multiple regression models were retained for presentation. The first model indicated that Chinook Salmon Abundance at sites sampled in multiple years is positively predicted by Diptera sp. density in May emergent vegetation tows (β = 0.2554, p = .4135) and Non-native Vegetation Diversity (β = 12.1117, p = .0227); however, the first relationship is not significant while the latter is. The two predictors explain 94.6% of the variance (R^2=.97, F[2,4] = 35.72, p = .02723) in Chinook Salmon Abundance. Specifically, Chinook CPUE at sites sampled in multiple years increases as the diversity values for non-native vegetation species for various sites and years increases and, possibly, as the mean density per meter of Diptera in May emergent vegetation tows increases.

The second model indicated that Chinook Salmon Abundance at sites sampled in multiple years is positively and significantly predicted by Diptera Density in May Emergent Vegetation Tows (β = 1.5604, p = .0334) and Reed Canarygrass Coverage (β = 0.8392, p = .0497). The two predictors explain 88.2% of the variance (R^2 = .94, F[2,4] = 16.01, p = .0588) in Chinook Abundance. Specifically, Chinook CPUE at sites sampled in multiple years increases as the mean density per meter of Diptera in May emergent vegetation tows and the percent cover of reed canarygrass increases.

It is important to note that although the models indicate strong R-squared values, the results should be treated with caution due to small sample sizes and are purely exploratory.

4.0 Discussion

This synthesis presents status and trends of juvenile salmon ecology in minimally disturbed freshwater emergent wetland sites in the LCRE and potential correlations between disciplines. Although these results cannot be applied to all tidally influenced wetlands within a reach or at the estuary scale, many conclusions can be drawn regarding undisturbed emergent wetlands and their role as salmonid habitat.

4.1 Habitat Structure and Hydrology

In this report, we begin to document the ranges and variation in hydrology and habitat structure of emergent marshes in the LCRE. Temporal and spatial variability in these systems affect the vegetation communities and their capacity for providing habitat for salmon, and contributing to the food web of the greater LCRE. As such, quantifying the expected ranges and variability can start to reduce uncertainties and inform research focus areas to improve the capacity of the LCRE to provide these important functions.

Spatial patterns we have been able to discern with the existing data set fall into the primary categories contributing to wetland structure and process, specifically sediment, hydrology (elevation), and vegetation. Sediment TOC is a means of measuring the organic content in the sediments and varies over time and space depending on inundation, vegetation communities present, age of the marsh, and other sediment constituents such as grain size. Given this complexity, the factors contributing to the variability in sediment TOC at our study sites is difficult to ascertain. All samples from the study area had values less than 10% TOC, with the highest values in the high marsh areas, which is a pattern consistent with measurements elsewhere (Odum et al. 1984). However, the values measured can generally be considered low for tidal wetlands, with overall lower TOC at known created sites. While little data have been collected on organic content in tidal freshwater and brackish marshes in the northwest, one study in a tidal freshwater marsh in the region found TOC between 16 and 26 percent (Thom et al. 2001) while Craft (2007) has documented that tidal freshwater marsh sediments often have higher organic content than salt marshes. One study in the LCRE has documented TOC levels ranging from 13 to 30 and attributes the variation to marsh age and landscape position (Elliot 2004). Studies in other areas have seen patterns of higher organic content in high marshes and lower in low marshes (Odum et al. 1984); we have noted similar but limited patterns in our data as well. While we cannot conclude the factors contributing to low TOC levels at our study sites at this time, we can hypothesize that a combination of vegetation type, landscape position, and marsh age may be factors contributing to the lower than expected levels. Further analysis of marsh age through evaluation of historical records will hopefully inform this theory.

Sediment grain size in this study was extremely consistent across sites and was predominantly silt, very fine, and fine sand. The sediment grain size pattern in the LCRE may be partially explained by proximity to the main channel of the river or the mainstem of a tributary. The hypothesis regarding this landscape pattern is that finer sediments would be present in more backwater settings, away from the higher flows associated with the river. Sherwood et al. (1984) found similar results, with finer sediments found in the peripheral bays as compared to the main channel. This hypothesis does not completely explain the observed patterns, however. Additional factors such as elevation and history of dredge material placement may also be factors. We will continue to evaluate these patterns as more data become available.

Marsh sediment accretion rates fell within a narrow range in our study area, but were variable in time and space throughout the LCRE. Sediment accretion is largely dependent on the sediment load of the contributing watershed, which is variable but estimated to average approximately 10 million metric tons annually in the Columbia (Sherwood et al. 1984). However, sediment transport has changed dramatically in the LCRE and has been reduced an estimated 61 percent from historical levels (Bottom et al. 2005). Altered sediment budgets, variable transport patterns, and historical changes due to dredging and entrapment by the reservoirs interact to create a complex sediment transport environment. For comparison, salt marsh sediment accretion rates measured in the region fell within a similar range between 0.2 to 1.7cm/year (Jefferson 1975; Thom 1992). In the Fraser Estuary, sediment deposition was most often associated with the occurrence of the spring freshet with deposits of 5 cm/year common (Seliskar and Gallagher 1983). Rates can be also be affected by local site factors including elevation, plant density, landscape position, and sediment type. More data on accretion rates over a longer period of record and throughout the LCRE will help to expand our understanding of sedimentation and erosion patterns on multiple scales.

The narrow range of elevations at the emergent marsh sites evaluated in this study is notable considering the varied hydrologic patterns that drive the location of these ecosystems. The inter-annual and spatial variability observed within these wetlands in cover, species diversity, and productivity is indicative of the hydrologic variability in space and time. However, the average elevation of the emergent marshes in the study area is likely a result of the hydrologic patterns over a longer period of time. This raises the question of whether the overall elevation of these sites has changed since the time when hydrologic patterns were less altered. Likewise, the question of future changes is also an important question to consider. Given the narrow range of elevations of these sites, changes in hydrologic patterns resulting from sea level rise, run-off patterns from climate change, and water management alterations could result in elevational shifts of these wetland ecosystems. Further research on the potential changes resulting from these changes is warranted to help inform restoration and conservation planning.

The hydrologic variability and the resulting inundation of the marshes varies dramatically along the estuarine gradient, with high inundation and seasonal variability in the fluvial-dominated upper estuary and lower inundation and daily variability in the tidal-dominated lower estuary. In the mesohaline zone (5 to 18 parts per thousand [ppt]; ~0 to 15 rkm) near the mouth of the estuary the vegetation cover is high; however, the number of species is limited by salinity. Few non-native species are found in this zone. In the oligohaline zone (0.5 to 5 ppt; ~16 to 40 rkm), species diversity starts to increase as there emerges an overlap in the number of species that can tolerate brackish and freshwater conditions. The highest species diversity occurs in the portion of the river that is tidal fresh water, but is not affected by the high seasonal inundation associated with the spring freshet (~41 to 135 rkm). In the fluvial-dominated tidal freshwater zone (above 135 rkm) vegetation cover and species diversity appear to be variable depending on the timing and magnitude of the spring freshet.

Vegetation was also evaluated as a function of elevation and indirectly inundation. As we have shown, inundation is correlated with elevation when compared in hydrologically similar portions of the LCRE. The highest species diversity occurs between the elevations of approximately 1.5 m CRD and 2.5 m CRD, consistent with other studies that have shown increased species diversity in high versus low marshes (Elliot 2004; Leck et al. 2009). Of particular interest in this analysis is the determination of the lower elevation limit of reed canarygrass throughout the LCRE. This aggressive non-native invasive species lowers species diversity and has the potential to affect the food web by reducing invertebrate prey diversity as well (Spyreas et al. 2010). As such, information regarding the limiting factors for growth and success are important to determining management actions. Elevation and inundation appear to be such limiting factors. The lower depth limit varied along the estuarine gradient, affected by salinity in the oligohaline portion of the estuary; therefore, reed canarygrass was only present at higher elevations where the sediments are often fresh (Seliskar and Gallagher 1983). In the tidally dominated freshwater portion of the estuary, the lower elevation ranges from approximately 1.2 m to 1.6 m CRD. This range increases to approximately 1.4 to 1.8 m CRD in the fluvial-dominated portion of the estuary as seasonal inundation increases and likely limits the lower elevation range.

We have found that the hydrologic variability observed between years is a primary factor driving variability in vegetation cover, composition, and biomass. This interannual variability associated with varying water levels was documented in our trends analysis at the three up-river trend sites (located at 145, 149, and 221 rkm); however, the same patterns were not as discernible at the trend site located at 72 rkm. The boundaries between the major species at the trend sites were generally stable over time even with varying water levels. In the highest water year we did observe an increase in the lower elevation of all species at Cunningham Lake, the lowest elevation site, indicating the potential for an effect on the elevation ranges from this level of hydrological variability. The implications of this kind of change include a potential loss of wetland area and a reduction in biomass production (discussed below).

Another trend we observed in this analysis was the interannual variability of reed canarygrass cover due to varying water levels; however, reductions were not persistent between years. At the lower-river trend site (Whites Island), where interannual hydrologic variability does not appear to be a primary controlling factor, the trend over three years has been a gradual increase in cover of reed canarygrass and a decrease in the cover of all other species. This trend could be attributed to the invasive nature of the species or could be due to interannual variability; additional data will provide a better understanding. The slight increase seen in the number of species over time was likely the result of an increase in the number of quadrats each year in an attempt to adequately represent the diverse site.

Inundation of the marsh channel mouths varies longitudinally and as expected between sites with varying channel elevations and morphologies. This affects the potential for fish access and is important for understanding the contribution of these marshes for refuge, feeding, and cover. Most channels were accessible for at least 60% of the time and most channel banks accessible for at least 40% of the estimated peak juvenile salmonid migration period. These elevations can be useful for informing restoration projects to ensure that salmon access is maximized at the site.

In general, the emergent marshes of the LCRE that were evaluated in this study are diverse, productive systems with channels that are providing the opportunity for juvenile salmonid access throughout the LCRE. Additional research evaluating the capacity differences between these emergent wetlands will further reduce the uncertainties regarding the quality of these systems for juvenile salmon. Further research on TOC in the sediment, biomass export, site history, sedimentation rates, and non-native species will help to better understand other ecosystem processes and functions such succession, carbon storage, and food web support

4.2 Fish and Fish Prey

As discussed in Bottom, Simenstad, et al. (2005), the LCRE has historically been viewed as a migration corridor for outmigrant salmon, and not as an important rearing habitat. However, subsequent studies have established the importance of the saltwater portion of the estuary as critical rearing habitat for juvenile salmon, particularly ocean-type Chinook (Bottom, Simenstad, et al. 2005; Bottom, Jones, et al. 2005; Roegner et al 2010). Much less is known about patterns of habitat use for juvenile salmonids in tidal freshwater habitats in the lower Columbia River, and research on their occurrence in this portion of the river has begun only recently (Johnson et al. 2011; Sather et al. 2009).

Over the past several years, the EMP has monitored fish community characteristics, salmonid habitat occurrence, salmon size, growth, and condition, salmon diet, prey availability, and contaminant exposure at representative tidal freshwater marsh sites from four reaches (C, E, F, and H) of the Columbia River between rkm 61 and 230. Results show that several species of juvenile salmon occur in emergent marsh habitats throughout the tidal freshwater portion of the Columbia River, and are present at these sites over an extended period, although the residence time of individual fish is unknown. The data also reveal a number of interesting spatial and temporal patterns, as well as differences in characteristics and habitat occurrence patterns of marked hatchery Chinook and coho salmon as compared to unmarked fish that are presumed to be naturally produced.

Fish Community Composition

The sites within each of the sampled reaches showed distinctive patterns of fish community composition. At the Reach C sites, catches were overwhelmingly dominated by three-spine stickleback, and the number of species present, species richness, and species diversity tended to be lower than at most sites in the other reaches. The percentage of non-native species was also generally low at the Reach C sites. The number of species, species diversity, and percentage of non-native species all tended to increase with increasing rkm between Reach C (rkm 61 to 99) and Campbell Slough (rkm 149), and there was also a gradual shift from dominance of three-spine stickleback to a community dominated by non-native species, including carp, chiselmouth, banded killifish, yellow perch, and peamouth. Above Campbell Slough, the number of species, species diversity, and percentage of non-native species tended to decrease with increasing rkm, although both species richness and diversity remained higher at most of the Reach H sites than at sites in Reach C. We also observed differences among reaches in the proportion of piscivorous salmon predators found in catches. The major piscivores preying on juvenile salmon in the Columbia River are northern pikeminnow, smallmouth bass, walleye (*Stizostedion vitreum*), and channel catfish (*Ictalurus punctatus*) (Petersen and Kitchell 2001; Rieman et al. 1991; Tabor et al. 1993; Vigg et al. 1991), with only northern pikeminnow being native to the Columbia River Basin (Peterson and Kitchell 2001). These species were rarely found at the Reach C Sites, but were present more often at Campbell Slough and Reach H sites such as Franz Lake and Sand Island, where they made up 1.5% to 7.5% of catch. While frequent sampling probably made some contribution to the high number of species present at Campbell Slough, in our multiyear fixed site sampling, this site consistently had high species richness in comparison with Whites Island in Reach C, and to a lesser extent in comparison with Franz Lake in Reach H.

This dominance of three-spine stickleback and generally low species diversity is similar to patterns observed at sites in Reaches A through C of the river by Rogener et al. (2008) and Bottom et al. (2008). A relatively low prevalence of non-native species (14%) was reported by Roegner et al. (2008) at sites in reaches A through C. Similarly, the patterns of occurrence of native and non-native species that we observed at Campbell Slough and in Reach H are much like those reported by Sather et al. (2009) and Johnson et al. (2011) at the Sandy River Delta. The most common species observed by Johnson et al. (2011) were three-spine stickleback, banded killifish, peamouth, and pikeminnow, and 53% of species collected were non-native.

The consistent pattern we observed of high species diversity and higher proportions of non-native species and piscivorous predators at Campbell Slough and some of the Reach H sites may be reflective of proximity to disturbed sites and areas with higher boat traffic (i.e. indicating potential areas of ballast water and fishing boat introductions; Cohen and Carlton 1998). Sanderson et al. (2009) also noted high numbers of non-native species in the heavily developed areas of the Columbia and Willamette Rivers, and mentioned dams and reservoirs as having a potential role in establishing non-native species in the Columbia River, which could contribute to higher numbers of non-native species in Reach H, near the Bonneville Dam. Also, the EMP sites with higher proportions of non-native species tended to have warm summer water temperatures, which these species favor. For example, it has been found that high summer water temperatures increases the potential for predation on juvenile salmon by species such as northern pikeminnow, smallmouth bass, and walleye (Petersen and Kitchell 2001).

Juvenile Salmonid Occurrence

In the tidal freshwater habitats sampled as part of the EMP, four salmonid species were observed: Chinook salmon, coho salmon, chum salmon, and steelhead trout. Chinook salmon were the dominant salmonid species at all of the Reach C sites as well as at Sandy Island and Campbell Slough. At the Reach H sites, Chinook were also present, but coho salmon were more abundant, making up as much as 75% to 80% of catches at Sand Island and Hardy Slough. Chum salmon were found throughout our sampling area, typically making up 1% to 4% of the salmonid catch. Steelhead were rarely found, but were seen most often in Reach H. Overall, the Reach H sites tended to have the greatest diversity of salmonid species, with significant numbers of unmarked and marked coho, chum, and a few steelhead, as well as Chinook.

These general patterns of salmon occurrence are similar to those reported in other studies of juvenile salmon in the lower Columbia River and other Pacific Northwest estuaries. Subyearling Chinook salmon are known to make extensive use of shallow, nearshore habitats (Bottom et al. 2005; Fresh et al. 2005), and Roegner et al. (2008), Bottom et al. (2008), Sather et al. (2009), and Johnson et al. (2011) report them as a dominant species at lower Columbia River sites, with chum and coho salmon and steelhead trout occurring in lower proportions. However, we did observe higher proportions of marked Chinook salmon and both marked and unmarked coho salmon in Reach H than Johnson et al. (2011) report for their sampling in the nearby Sandy River Delta. The reasons for this are unclear, but could be related to the proximity of the sites to hatchery releases or to the duration of the sampling period. Johnson et al. (2011) sampled more extensively in fall, winter, and early spring than we did in the EMP.

Juvenile Chinook salmon occurred at all of our sampling sites from April, when sampling began, until June or July, with a few sites continuing to support Chinook salmon in August. Peak Chinook densities for both marked and unmarked Chinook were found in May and June. Similar seasonal patterns of occurrence for juvenile Chinook salmon are reported by Roegner et al. (2008), Bottom et al. (2008), Sather et al. (2009), and Johnson et al. (2011). Although the majority of Chinook salmon we collected at our sampling sites were within the subyearling size range, we did capture some larger yearling-size fish as well. All of the larger, yearling-size Chinook were marked fish of hatchery origin. These Chinook "yearlings" were found in April and May, likely coinciding with hatchery releases of this life stage (Columbia River DART: http://www.cbr.washington.edu/dart/hatch.html). Some yearling-size Chinook salmon were also reported in catches of Bottom et al. (2008) and Johnson et al. (2011).

In our study, peak densities of Chinook in Reach C ranged from .08 fish/m^2 at Whites Island to 0.42 fish/m^2 at Bradwood Slough, and in Reach H from .02 fish/m^2 at Sand Island to 0.43 fish/m^2 at Pierce Island. These ranges are very similar to those observed by other researchers. Johnson et al. (2011) reported peak seasonal densities for unmarked Chinook in the 0.05 to 0.25 fish/m^2 range in the Sandy River Delta, while, in tidal wetland sites in Reaches B to C Bottom et al. (2008) reported seasonal peak densities for subyearling Chinook salmon ranging from <0.01 to 0.17 fish/m^2. In the Salmon River and Oregon Coast estuaries, the reported range for peak seasonal juvenile Chinook salmon densities is 0.02 to 0.11 fish/m^2 (Bottom et al. 2005; Cornwell et al. 2001).

The relatively high densities of unmarked juvenile Chinook salmon in Reach C suggest this may be an important area for natural production. This is the reach in which the fewest piscivorous predators and non-native species were located. Also, the Reach C sites tended to have lower summer water temperatures than most of the sites in other reaches. Both factors could make this reach especially favorable habitat for juvenile salmon. Marked Chinook, on the other hand, were present at highest densities at Campbell Slough and Reach H. There are several hatcheries releasing Chinook in these areas including the Spring Creek, Little White Salmon, and Cascade hatcheries in the Columbia Gorge, and the Washougal and Bonneville hatcheries, which could be a source of marked fish in Reach H and possibly Campbell Slough as well. Several hatcheries are also located on the Lewis River near the Campbell Slough site.

In our study, chum salmon were found almost exclusively in April and early May at all sampling sites where they occurred, consistent with their typical season of outmigration (Myers 1980; Johnson et al. 1997) and with other recent reports on chum occurrence in the LCRE (Roegner et al. 2008; Johnson et al. 2011). The size range of the chum we collected (37–63 mm) is typical of the size range at which juvenile chum outmigrate to the ocean (Salo 1991). Chum salmon were found at highest densities in Reaches C and H, with peak densities of 0.008 to 0.015 fish/m^2. This is consistent with the fact that the two extant spawning populations of chum salmon are in the Columbia Gorge and Grays River Estuary (Good et al. 2005). Columbia Gorge subpopulations are found in Hamilton Creek and Hardy Creek (Good et al. 2005), so they would be in close proximity to the Reach H sites, including Pierce Island where the highest density of chum was found.

Coho salmon were found at fewer sites, and showed a less consistent seasonal pattern than Chinook or chum salmon, although densities of larger and marked coho were generally highest in May. The larger unmarked coho were presumably outmigrant smolts, as they were found during the established time for coho smolt migration in the Columbia River (Weitkamp et al. 1995) and were within the normal size range reported for coho smolts (Weitkamp et al. 1995). Smaller unmarked coho were found throughout the sampling season at low densities, and were likely subyearlings that typically remain in fresh water until outmigration the following spring (Johnson et al. 1997). These smaller fish showed an increase in size over the sampling season that would be consistent with their undergoing a period of rearing and freshwater growth prior to outmigration. Marked coho were also within the size range for smolts, and were found in May, which corresponds with the timing of hatchery releases for this species, which usually occur in April and May (Columbia River DART: http://www.cbr.washington.edu/dart/hatch.html).

Coho salmon were found primarily in Reach H and at Bradwood Slough in Reach C. Unmarked coho had highest densities at Bradwood Slough in Reach C and Pierce Island in Reach H, with levels of 0.018 to 0.050 fish/m^2. While it is uncertain why especially high numbers of coho salmon were found at these sites, there are natural coho populations in the Cowlitz and Lewis Rivers and Mill, Abernathy, and Germany Creeks that could be a source of fish for Reach C, as well as natural populations in the upper Columbia Gorge that could be a source of fish for Reach H (Good et al. 2005). Also, it is notable that Bradwood Slough and Pierce Island were among the sites with the lowest summer water temperatures, which could make them more favorable for coho salmon. All marked coho were found at sites within Reach H. Several hatcheries that release coho salmon are located near this area, including the Willard and Little White Salmon hatcheries operated by the USFWS, and the Oxbow and Cascades hatcheries operated by the Oregon Department of Fish and Wildlife (ODFW) (Columbia River DART: http://www.cbr.washington.edu/dart/hatch.html), and are a likely source for these fish.

Chinook Salmon Genetic Stock Composition

Chinook salmon from multiple stocks were observed at our sampling sites. While the majority of fish were Spring Creek Group fall Chinook and West Cascade fall Chinook, a significant proportion of fish were from interior Columbia River stocks (i.e., Upper Columbia, Snake River, and Deschutes River fall Chinook), and some Upper Willamette spring and West Cascades spring Chinook were also present. We also observed a seasonal pattern in stock composition, with interior Columbia River stocks increasing in proportion as the sampling season progressed. This could be a reflection of the extended migration these stocks must undertake to reach the LCRE. Additionally, we observed differences in stock composition of marked and unmarked fish. The majority of marked fish were Spring Creek Group fall Chinook, whereas the unmarked fish were much more diverse.

Generally, our findings are similar to those of other researchers. For our sampling sites in Reach H, Chinook salmon genetic stock composition was comparable to that reported for marked and unmarked juvenile Chinook by Sather et al. (2009) and Johnson et al. (2011) in the Sandy River Delta. Sather et al. (2009) and Johnson et al. (2011) also found that marked Chinook were primarily Spring Creek Group fall Chinook, whereas unmarked fish were predominantly Upper Columbia summer/fall Chinook and Spring Creek Group fall Chinook, with a variety of other stocks present in smaller proportions. Johnson et al. (2011) also reported the same temporal patterns in stock composition as those we observed, with interior Columbia stocks increasing and Spring Creek Group fall Chinook declining in proportion as the sampling season progressed. At our Reach C sites, genetic stock composition was very similar to that reported by Bottom et al. (2008), with West Cascades fall Chinook being the most prevalent stock.

Salmon Length, Weight, Growth, and Condition

Chum and coho salmon were caught in low numbers and at a limited number of sites, so spatial patterns of length, weight, growth, and condition are difficult to evaluate. Chum salmon showed little difference among sites in length, but had somewhat lower condition at Campbell Slough and some of the Reach C sites as compared to sites in Reach H, perhaps because of the energy demands of migration and smoltification (Wedemeyer et al. 1980). Large outmigrant coho, which were found only in Reach H, exhibited no intersite differences in fish size or condition. The smaller coho showed some intersite differences, but no consistent patterns by reach.

Conversely, Chinook salmon were found in larger numbers throughout the EMP sampling area, and some patterns in size by reach were evident, as well as differences between marked and unmarked Chinook. The marked fish tended to be larger than unmarked fish, all fish were within a limited size range, and their size tended to be more uniform over the sampling season. In contrast, unmarked fish were present in a much more diverse range of size classes, including fry of <60 mm and fingerlings or subyearling smolts of 60 to 100 mm (Fresh et al. 2005). High life history diversity in unmarked juvenile Chinook salmon has also been reported by Bottom et al. (2005, 2008) and Roegner et al. (2008, 2010) in the saltwater portion of the estuary, and by Sather et al. (2009) and Johnson et al. (2011) at tidal freshwater sites. Also, among unmarked fish there was a clear increase in fish length and weight as the sampling season progressed, which was much less evident in marked fish. Similar patterns were also observed by Bottom et al. (2008) and Johnson et al. (2011).

Although the trend of increasing size over the sampling season was evident in unmarked Chinook salmon at all sites, there were some differences among sites in fish size ranges. In general, juvenile Chinook from Reach C tended to be smaller in size than Chinook in other reaches, with catches including a higher proportion of small unmarked Chinook fry (< 50 mm in length). Length and weight also tended to decrease with distance from the main channel, which would be consistent with previous observations that the smaller fry and fingerlings are likely to use off-channel habitats for feeding and rearing prior to outmigration (Fresh et al. 2005). The size ranges of juvenile Chinook salmon we found in Reaches C and H were generally comparable to those reported by Bottom et al. (2008) and Johnson et al. (2011) for juvenile Chinook salmon from sites in or near those reaches of the Columbia River.

Our measures of fish fitness (growth rate, lipid content, and K) collected for fall Chinook were all within the normal range reported for subyearling Chinook (Barnam and Baxter 1998; Biro et al. 2004). For example, Bottom et al. (2008) reported mean growth rates of 0.06 to 1.06 mm/day in juvenile Chinook salmon from Columbia River reaches A through C, and studies in other Pacific Northwest estuaries report growth rates of 0.21 to 0.62 mm/day for juvenile Chinook salmon (Healey 1991; Korman et al. 1997; Levy and Northcote 1982).

Marked and unmarked Chinook salmon, and Chinook salmon from different sites and stocks, showed significant differences in growth rate, which may not have been reported elsewhere. Growth rates were significantly higher over the last seven to 21 days of life prior to capture in marked fish as compared to unmarked fish. Some stock differences in growth rate were also observed, with highest growth rates occurring in Spring Creek Group fall Chinook and lowest growth rates occurring in West Cascades fall Chinook. Additionally, growth rates were lower in juvenile Chinook from the Reach C sites. This could be partly because the majority of Chinook from these sites were unmarked Chinook belonging to the West

Cascades fall stock, although other site-specific factors, such as the relatively high densities of juvenile Chinook salmon at these sites, might also contribute to lower growth rates.

Overall, there was no significant difference in lipid content or K between marked and unmarked fish, although lipid content tended to be slightly higher in marked fish. However, marked and unmarked fish showed some different patterns of lipid content by reach. In marked fish, lipid content tended to increase with rkm (highest in Reach H and lowest in Reach C). If many of the marked fish are entering the river from sites upstream of Reach C, such a loss of lipid content during downstream outmigration might be expected. Lipid loss with downstream migration in marked fish has been observed in other studies (Arkoosh et al. 2011; Weitkamp 2008). For example, in spring Chinook released from Snake River Basin hatcheries, Arkoosh et al. (2011) observed a decline in lipid content from 3% to 5% to less than 1% during outmigration from the hatcheries and Bonneville Dam. There was no such relationship between lipid and rkm for unmarked fish in this study, suggesting that more of these fish may be entering the sampling sites from rivers and streams within the local watershed, and are resident at the sites for some period of time, feeding and rearing. MacFarlane (2010) also reported little change in lipid content of naturally produced juvenile Chinook salmon from the Central Valley of California.

Somewhat surprisingly, we did not see a consistent correlation between lipid content and K in juvenile Chinook. In unmarked Chinook, K tended to increase with increasing rkm, but showed no significant relationship with rkm in marked fish. Additionally, in both marked and unmarked Chinook, K tended to increase as the sampling season progressed, with greatest differences occurring between fish sampled early in the season, in April, and those sampled later. However, as mentioned above, lipid content changed little over the sampling season in unmarked Chinook, while in marked Chinook it tended to decline. The lack of consistent correlation between lipid content and K was also reported by MacFarlane (2010) for juvenile Chinook salmon in the Central Valley of California. However, MacFarlane (2010) observed a decline in K over the sampling season, which he attributed to a change in the length-weight relationship occurring as a result of body shape changes associated with juvenile salmon development. Our somewhat inconsistent results may be due to some bias in K for small fish because of difficulty in accurately measuring their weight. Values of K were significantly lower in juvenile Chinook below 50 mm than in those of larger size classes. Such a bias in K for small fish would explain the low values of K for fish caught early in the sampling season, as well as the low K values for fish from Reach C, where the proportion of Chinook of small size classes tended to be high.

Contaminants and Water Quality

Our sampling suggests that summer water temperatures may limit salmon use of many of the EMP sampling sites. The preferred water temperature for juvenile Chinook salmon is between 10°C and 16°C (Marine and Chech 1998; McCollough 1999). At all sites except for Hardy Slough, where maximum temperatures were 12°C to 13°C, the water temperatures in July and August were above the preferred range for juvenile salmon, exceeding 25°C at some sites. These findings are consistent with many reports of water quality impairment due to elevated summer temperatures in the lower Columbia River (ODEQ 2010a). Interestingly, those sites with the lowest summer temperatures (Hardy Slough, Bradwood Slough, and Pierce Island) tended to have the highest densities of salmon.

Chemical contaminant exposure was also evident throughout much of the EMP sampling area, in spite of the fact that the EMP is focused on relatively undisturbed sites. Concentrations of POPs (i.e., PCBs, DDTs, and PBDEs) in juvenile Chinook salmon bodies were consistently lower in Chinook salmon from the Reach H sites than in fish from Campbell Slough, Sandy Island, or the Reach C sites. Concentrations of PCBs and DDTs were fairly similar at Campbell Slough, Sandy Island, and the Reach C sites, but concentrations of PBDEs were quite low in fish from Campbell Slough, and higher in fish from Sandy Island and the Reach C sites. There were no consistent differences in contaminant concentrations between unmarked and marked Chinook; mean levels of PCBs, DDTs, and PBDEs were similar in both groups. The patterns of contaminant accumulations appears to reflect the influence of industry and urban development in the Portland and Vancouver areas upstream of Campbell Slough, Sandy Island, and Reach C, as well as industrial development and sewage and industrial outfall present in the vicinity of St. Helens and Columbia City, which would be expected to affect Sandy Island and the sites downstream. The presence of PBDEs in Chinook salmon in Reach C indicates the potential for exposure to other wastewater compounds, including pharmaceutical and personal care products. The fact that some sections of the Columbia River or its tributaries in Reach C are listed as impaired water bodies for bacteria (ODEQ 2010a) provides further evidence that these types of chemicals may be present in Reach C. Although contaminant concentrations at the EMP sites were generally low in comparison to maximum levels found in fish from the Portland/Vancouver area (Estuary Partnership 2007), lipid adjusted concentrations of PCBs and PBDEs in some samples from Sandy Island, Campbell Slough, and the Reach C sites were above the estimated threshold for toxicant-related injury (Arkoosh et al. 2010; Meador et al. 2002).

Although our sampling was quite limited, we also saw evidence of exposure to PAHs in salmon from Sandy Island, Campbell Slough, and Franz Lake and Hardy Slough in Reach H at concentrations at or above levels considered to be associated with negative effects on juvenile salmon (Meador et al. 2008). Unlike DDTs, PCBs, and PBDEs, PAH metabolite levels were not lowest in Reach H sites. In fact, some of the highest levels of PAH metabolites in bile were found in samples from Franz Lake and Hardy Slough in Reach H. Exposure to PAHs in salmon from Campbell Slough and Sandy Island is not surprising, because of their proximity to industrial centers such as Portland, Vancouver, Columbia City, and St. Helens, where PAH contamination has been identified (Estuary Partnership 2007; ODEQ 2010a). It is less clear why PAH metabolite levels are so high in fish from the Reach H sites. Both Reach H sites are close to railways and highways, which could potentially cause PAH contamination. Also, PAHs in natural products, such as decomposing wood and organic matter, could be present at these sites and may contribute to high levels of PAH metabolites in bile (Johnson et al. 2009). At this point, however, the number of samples is too small to realistically evaluate risks associated with PAHs in fish from these sites. Overall, our findings highlight the pervasive presence of chemical contaminants, even at sites that are not in urban environments and are considered relatively undisturbed.

Prey and Diets

Our prey sampling revealed some differences in prey types and abundance at the EMP sites in different reaches. The most common prey species available to salmon at the EMP sampling sites in Reaches E though H were Cyclopoid and Calanoid copepods, Cladocerans, and Dipterans. The prey species available at the Reach C sites were somewhat different—the most common taxa were Dipterans, and terrestrial and aquatic Heteroptera and Hemiptera were also often abundant, but copepods were not a prominent part of Reach C samples. At sites in Reaches A through C, Bottom et al. (2008) also reported Dipterans as the most abundant prey, as well as amphipods, mysids, and Cladocerans. Generally, the types and proportions of prey in the tow samples were broadly comparable throughout the sampling season, although it should be noted that seasonal comparisons are somewhat limited as the majority of samples were collected in May and June, when Chinook salmon were present at highest densities.

Prey densities were quite variable among sites, sampling times, and between the two types of tows, but on average, there were 20 invertebrates per meter of tow when averaging both emergent vegetation and open-water tows. This is comparable to the findings of Johnson et al. (2011), who report peak densities of Dipterans in drift of about 20 to 60 individuals per m^3. Prey densities were consistently higher in emergent vegetation tows than in open-water tows, suggesting that these nearshore, vegetated environments may be especially important in providing food for juvenile Chinook salmon.

One of the most interesting observations from the prey and diet findings was the prevalence of Diptera, primarily Chironomid larvae and pupa, in stomach content samples of juvenile Chinook salmon from all sites. Selectivity indices showed a strong preference for Dipterans that was consistent across sites. Amphipods, which were generally only available at the Reach C sites, were another preferred prey item. Our findings regarding the consistent consumption of aquatic Dipteran larvae and pupae and the likely importance of larger amphipods when available are consistent with other studies in the region (Bottom et al. 2008; Johnson et al. 2011; Sather et al. 2009; Spilseth and Simenstad 2011). While Johnson et al. (2011) did not see a clear preference for Dipterans over other prey in their sampling of the Sandy River Delta, selectivity for Dipterans particularly Chironomids, has been reported for juvenile Chinook salmon in the Great Lakes (Principe 2005), estuarine wetlands in Puget Sound (Shreffler et al. 1992), and Reaches A through C of the LCRE (Lott 2004). Thus, although Chinook salmon are thought of as opportunistic feeders (Beamish et al. 2003; Duffy et al. 2010) our data indicate they do have preferences for certain taxa, especially larger bodied taxa, and these preferences appear to similar across habitats and early summer months.

Multiyear Trends at Fixed Sites

Over the two to four years that we sampled our fixed sites (Whites Island in Reach C, Campbell Slough in Reach F, and Franz Lake in Reach H), fish community characteristics, patterns of juvenile salmonid habitat occurrence, juvenile Chinook salmon diets, and prey types present were generally consistent within each site over the sampling period.

In a few cases, substantial variation among years was observed. For example, at Franz Lake, the proportion of piscivorous predators was quite a bit higher in 2009 than in 2008 (11% versus 2% of the total catch, respectively). The high proportion of piscivorous predators in 2009 is not characteristic of other Reach H sites, where they only accounted for 1% to 2% of the catch, and appeared to be due to the large number of northern pikeminnow caught at Franz Lake in 2009. Additional sampling may provide a more accurate assessment of the prevalence of piscivorous predators at this site.

We also found that the proportion of marked versus unmarked Chinook salmon varied substantially at Campbell Slough from year to year. The proportion of marked Chinook salmon was about 50% in 2007, 2009, and 2010, but 96% in 2008. It is not clear why such a high proportion of marked fish was caught in 2008, but this may have occurred in part because our 2008 sampling was confined to May and June when most hatchery releases occur (Columbia River DART: http://www.cbr.washington.edu/dart/hatch.html), and when we observed the highest densities of marked Chinook salmon in all sampling years. Genetic stock composition of Chinook at Campbell Slough showed similar changes—there was high stock diversity for unmarked Chinook salmon in 2007 and 2010, but much less in 2008 and 2009. This is probably related to the very small number of unmarked Chinook salmon analyzed for genetics in 2008 and 2009. In the years when genetic data were collected from a more substantial number of unmarked Chinook, several different stocks were present.

While most of the parameters we measured showed no clear increasing or decreasing trends, we did find some evidence of temporal trends in contaminant levels in juvenile Chinook salmon. At Campbell Slough, the site that had been sampled over the longest time period, concentrations of both DDTs and PCBs tended to decrease between 2007 and 2010, and the proportion of juvenile salmon with body PCB concentrations above the estimated toxic effect threshold (Meador et al. 2002) declined from 67% in 2007 to 9.5% in 2010. This is consistent with declining trends in the same contaminants that have been observed in resident fish (Hinck et al. 2006) osprey eggs (Henny et al. 2008, 2010), and bald eagle eggs (Buck et al. 2005). In contrast, concentrations of PBDEs in Chinook salmon from Campbell Slough showed little change over the sampling period. Only a limited number of bile samples could be collected for assessment of PAH exposure in salmon. Similar to PCBs and DDTs, there appeared to be some decline is FACs-PHN and FACs-NPH at Campbell Slough between 2007 and 2008, but there are too few data to consider this a trend.

One of our most interesting findings was the temporal trend in length, weight, and condition at Whites Island, Campbell Slough, and Franz Lake. Length, weight, and K were low in fish from all three sites in 2009. The trends we saw in fish size and K are consistent with trends in growth rate and lipid content in fish from Franz Lake and Campbell Slough. (Growth rate and lipid data from Whites Island are currently available for 2009 only, so temporal trends at this site for lipids cannot be evaluated.) Consistent with size and condition, growth rates in juvenile Chinook salmon from Campbell Slough and Franz Lake tended to be low in 2009. Lipid content showed a similar pattern; lipid levels in both marked and unmarked Chinook salmon from Franz Lake and from marked Chinook salmon at Campbell Slough in 2009 were significantly lower than samples collected in other years. We also found that, at Whites Island, the density of unmarked Chinook salmon was significantly lower in 2009 than in 2010. In some other cases, densities tended to be lower in 2009 than in other sampling years, but differences were not statistically significant.

These consistently low measures of salmon size, growth, and fitness appeared to be associated with very warm summer temperatures in 2009, which may have made conditions unsuitable for juvenile Chinook salmon. Our sampling suggests that summer water temperatures may limit salmon use of all three of these sampling sites. At all three fixed sites, in all sampling years, the water temperature in July and August was above the favorable range for juvenile salmonids (Marine and Chech 1998; McCullough 1999) and salmon were either not present or at very low densities. Temperatures were especially high in 2009. There was evidence of poorer fish health and condition in 2009 (a year with higher water temperatures), as indicated by lipid levels, growth rates, and K, suggesting that high water temperatures may affect fish fitness as well as habitat access.

4.3 Abiotic Conditions

Although water quality data were only available for two years at one site, some conclusions can be drawn based on the limited data. Habitat suitability for salmonids, in terms of water quality, varied over the spring–summer monitoring season and from year to year depending on weather and hydrologic conditions. Fish sampling data indicate that use of tidal wetlands by juvenile salmonids may be limited by water quality conditions, especially temperature and dissolved oxygen concentrations. However, as of 2010, the residence time and movement in and out of the sites by juvenile salmonids was unknown. At Campbell Slough, even when daily and weekly water quality standards were not met, during some periods, tidal and diurnal variability caused conditions to be suitable for juvenile salmonids during smaller time scales (i.e., for some period within a day). Fish sampling data indicate that juvenile Chinook were present in Campbell Slough even when water quality conditions appeared to be stressful (high temperature and/or low dissolved oxygen), such as in late June 2009 and early July 2010. However, the spatial variability of water quality parameters within the site is not known, so suitable conditions may have been present in certain areas of the site.

4.4 Multidiscipline Analyses

It is important to emphasize that regression modeling results should be treated with caution due to small sample sizes and a limited set of environmental and biological variables. Emphasis in this synthesis report is therefore not placed on determining statistically significant relationships between variables but rather on exploring possible relationships for future study.

Fish Diversity

Overall, fish diversity at freshwater tidal wetlands was positively related to prey diversity and coverage of common spikerush (*E. palustris*). Fish diversity had a weaker, negative, correlation with the richness and diversity of vegetation species. The correlation between vegetation variables and fish diversity, however, may be due to the site's location in the river rather than site-specific variables. Reaches E, F, and H had higher fish diversity and these reaches, in general, had lower plant species diversity and higher cover of the three dominant species, including common spikerush. Higher fish diversity may be more a function of landscape level influences such as proximity to disturbed sites and higher boat traffic (i.e. due to ballast water and fishing introductions) and reservoirs that promote introduction of non-native species and warmer summer temperatures supporting warm-water acclimated species. Future analysis would benefit from the inclusion of other landscape level variables.

Variables used in this multidiscipline analysis vary not only with factors like river mile, distance from the main channel, elevation, etc. but also seasonally and in a non-linear fashion. This variability, in addition to high variability in fish sampling, may make it especially difficult to detect clear relationships with correlation analyses. Future analyses would benefit from the inclusion of other landscape level variables, important abiotic site characteristics such as water temperature and chemistry coincident with fish sampling and the exploration of relationships between other variables, such as vegetation and fish prey.

It is notable that: 1) Campbell Slough 2009 displayed the highest fish diversity and the second-highest abundance of Diptera and all macroinvertebrates combined and 2) all Campbell Slough sites were within the top five highest fish diversity sites and the top 40% prey and Diptera abundance sites. This trend may indicate that abundant prey is an important component to heightened fish diversity.

Juvenile Chinook Salmon

Juvenile Chinook abundance was positively related to species richness of all vegetation species, and negatively related to invertebrate prey species diversity in emergent vegetation and the distance of the sample location from the mouth of the Columbia River. Decreased Chinook abundance with increasing prey species diversity could indicate that where prey species is high, preferred prey, such as diptera, may be lower. Although higher Chinook salmon abundances are potentially related to characteristics of the tidal freshwater wetlands sampled -- deeper tidal channels, higher plant diversity, and prey diversity -- the relationship may be more a function of river location, rather than site-specific variables. Over 30 plant species were found in Reach C sites, where salmon abundances were greatest, and less than twenty species were found in upriver reaches where salmon abundances were lower. Low numbers of piscivorous predators and lower summer water temperatures in reaches with high juvenile Chinook abundance are also possible explanations for higher abundance and should be included in future analyses. A decrease in juvenile Chinook abundance with distance from the mouth of the Columbia River is likely a proxy for other factors that affect salmonid occurrence and would require additional research. Given the small sample sizes for these analyses, interpretation of results should be treated with caution.

Biological and physical variables affecting juvenile Chinook salmon abundance differed entirely when the analysis was limited only to sites sampled over multiple years (Campbell Slough, Franz Lake, and Whites Island). At the trend sites, Chinook salmon abundance was related positively to the density of Diptera species in May in emergent vegetation, the diversity of non-native vegetation, and greater cover of reed canarygrass. The highest densities of prey were found in emergent vegetation tows, highlighting the link between vegetated nearshore habitat and juvenile salmonids. The positive correlation between reed canarygrass and Chinook salmon abundance is likely driven by patterns in hydrology and inundation, in which higher water results in lower coverage of reed canarygrass and lower abundances of juvenile Chinook salmon. Low Chinook salmon abundances could potentially be due to an inability to beach seine for fish during some high water conditions. Variables affecting abundance of Chinook captured at all sites appear to be quite different than those affecting abundance at sites sampled in multiple years and underscores the importance of obtaining trend data.

Few conclusions can be drawn from multiple regression analyses investigating the relationship between environmental variables on all Chinook lipid levels due to low sample sizes. Correlations, however, suggest an association of increasing lipid levels with decreasing site elevation. Decreased site elevation may allow greater access to sites resulting in heightened feeding and greater refuge. When the analysis was limited to only unmarked Chinook, the diversity of prey from emergent vegetation and elevation had more of a relationship with juvenile Chinook lipid levels than other biological and physical variables. Increased prey diversity could lead to greater choice in prey, some of which may have a high caloric content, leading to higher fish lipid levels.

A negative relationship between unmarked Chinook and marsh elevation, however, is contradictory to the positive relationship of elevation with all Chinook lipid levels. Although it would seem likely that elevation serves as a proxy for rkm, marsh elevations relative to CRD don’t increase with increasing rkm. The CRD is relative to an imaginary low water level in the Columbia River rather than relative to a stationary point such as mean sea level at the river mouth. The relationship between marked Chinook and rkm (i.e. lipid levels decrease with decreasing rkm), may be driving these contradictory findings at the site specific scale, indicative of the small sample size. In future analyses a relationship between elevation and rkm at the site specific scale should be controlled for. It should also be noted that lipids were measured in composite salmon samples, so might not represent the true variability in lipid content of individual salmon.

Variables used in this multidiscipline analysis vary not only with factors like river mile, distance from the main channel, elevation, etc. but also seasonally and in a non-linear fashion. This variability, in addition to high variability in fish sampling, may make it especially difficult to detect clear relationships with correlation analyses. Future analyses would benefit from the inclusion of other landscape level variables, important abiotic site characteristics such as water temperature and chemistry coincident with fish sampling and the exploration of relationships between other variables, such as vegetation and fish prey.

5.0 Conclusions and Recommendations

This report represents the Estuary Partnership's first synthesis of data collected under the EMP to assess juvenile salmon ecology in minimally disturbed tidal freshwater wetlands of the LCRE. Habitat structure and hydrology, fish, macroinvertebrates, and abiotic conditions data analyzed in this study indicate that undisturbed emergent wetlands in the LCRE system are important to fish in general and specifically to multiple salmon species and stocks. The relationships between fish species and physical, biological, and environmental characteristics, coupled with the findings that lower Columbia wetlands exist within a narrow elevation range and are vulnerable to hydrologic changes, suggest a complex system in which disruption could have significant ecosystem-wide impacts. Maintaining or improving the quality of these habitats by reducing this disruption can occur by taking steps to 1) preserve/restore nearshore emergent vegetation and hydrograph supporting the vegetation community, 2) establish habitat structure to support preferred salmon prey, 2) moderate summer temperatures, 3) reduce the spread of non-native species, and 4) reduce chemical contamination.

In addition to addressing the current goals and objectives of the EMP between 2005 and 2010, this synthesis informs current and future program design. Ultimately, one of the goals of this program is to assess ecosystem condition by creating a suite of indicators that can be used to track changes in the LCRE. Indicators will need to distinguish between variability associated with natural conditions, and variability that may result from human impacts. Results from this report begin to document the range in variability for each sampled metric, critical for designing a more statistically rigorous program. For example, habitat analysis revealed distinct hydrologic zones for vegetation that establishes a pattern in spatial variability throughout the LCRE. In trend sites, vegetation cover and elevation ranges remained fairly consistent between years indicating that drastic changes to these baseline conditions could signify a decline in condition in these wetlands. Although fish communities are highly variable throughout the LCRE, trends analysis in this study revealed a surprising site consistency that may aid in detecting changes over time. Additionally, the synthesis of EMP data allows us to evaluate each metric to streamline data collection (see recommendations below) and select the most appropriate forward-looking indicators. Beginning in 2011, current monitoring design includes a suite of primary and secondary production metrics, more intensive abiotic conditions monitoring, and establishes sites in Reaches A and B that will begin to integrate long-term trends in salmon food web dynamics.

5.1 Recommendations

- This study underscores the need for randomly selected sample sites for greater statistical inference, increased sample size, and continued co-location of sampling sites in future program design. Randomly selected sites, however, will need to take into account site accessibility for fish sampling (beach seining).
- Due to difficulties in interpreting the apparent variation among some of the fish and invertebrates measures across sites, we recommend expanding the scope of project to include several disturbed sites as well as tributary sites. We expect that the apparent, albeit weak, differences among some of the biotic measures may be rather insignificant when compared to those from more disturbed sites. These types of comparisons would help put the EMP sites in context of the larger river and the mosaic of habitats that vary in their disturbance histories and current conditions. A greater diversity of site disturbance would allow us to assess resiliency in habitat and food web dynamics. For instance, including sites in the Lower Willamette River and around Portland would increase our understanding of how local conditions may affect fish growth and condition, prey availability, and the accumulation of contaminants. In addition, comparing the EMP sites to more disturbed sites would help clarify the relative importance of vegetation in marsh habitats for production of salmonid prey along the river continuum and access to rearing habitats for fish. The addition of tributary sites would help fill gaps in our spatial understanding of juvenile salmon use of off-channel tributary habitats.
- Climate change has the potential to have a large impact on these low elevation marsh habitats. Future study should incorporate measures of climate change that are not already being assessed (i.e. measures of acidification, hypoxia)
- The study has pointed to several interesting questions about the role of temperature and how it may affect fish habitat use and fish growth and condition. In particular, it would be useful to have temperature loggers deployed continuously at each site. These data would be helpful in modeling the bioenergetics of fish in these sites, and may help explain how fish are able to persist in some sites despite extreme temperatures.
- Although permitting and budget limitations will continue to limit the number of fish that can be collected and necropsied, targeted collections of additional fish at particular sites may help resolve some of the patterns seen in the growth rates and lipid concentrations across sites. If additional fish are caught at some sites, it may be necessary to reduce the lethal take of fish from other sites during certain months.
- To further explore potential effects of vegetation and invertebrate prey on Chinook abundance, the program could try to quantify the links between vegetation and invertebrate prey. Namely, with additional sampling of vegetation and invertebrates in the near-shore zone, we could assess how the extent, type and production of vegetation affect the production and structure of the invertebrate communities at these sites. By both increasing the number of invertebrate tows per sampling event and concurrently quantifying relevant measures of vegetation, we would improve our estimates of prey availability and our understanding of factors that affect their availability.

- In future years, there should be increased support for measuring contaminants in the fish and prey samples. Currently, these analyses are not funded through the EMP and results are provided through leveraged funding by NMFS. To explore trends in contaminants in the biota, we should continue to sample fish and prey at the trend sites and we should revisit a larger suite of sites that were sampled in 2005.
- Surface elevation tables (SET) could be installed at some or all of the core monitoring locations to evaluate accuracy of the current method for measuring wetland accretion or erosion and to allow for better characterization of overall elevation changes due to sediment dynamics and shallow subsidence (Rybczyk and Cahoon 2002). In addition, multiple sediment accretion stakes could be placed at core sites to better understand site-scale patterns of sediment dynamics.
- Timing of sensor deployment should be altered so the entire growing season is recorded in one year (e.g., deploy and retrieve in late October).
- Vegetation mapping efforts could be reduced at trend sites unless obvious change is observed, perhaps every 3-5 years.
- Only single cross-sections at the channel mouth could be measured at the trend sites to evaluate change, with the whole channel being surveyed less frequently. Changes in the channel morphology would likely be detected by measurements at the channel mouth. If change were to be observed at the mouth then a full survey should be completed in the following year. Otherwise, the channel could be surveyed at a regular interval such as every five years. In addition, at trend sites the channel cross-sections need to be surveyed at exactly the same start points and at consistent intervals to be able to evaluate change over time. Initial surveys of the status sites should still have the full channel surveyed as part of the characterization of the site.
- Finally, a synthesis of results, such as this, should be undertaken at a minimum every 3-5 years in order to assess new findings, improve on methods and disseminate results to resource managers in the region.

6.0 References

Allen, T.H. 1999. Areal Distribution, Change and Restoration Potential of Wetlands within the Lower Columbia River Riparian Zone, 1949 – 1991. Oregon State University. Corvallis, OR.

Arkoosh, M.R., D. Boylen, J. Dietrich, B.F. Anulacion, G. Ylitalo, C.F. Bravo, L.L. Johnson, F.J. Loge, and T.K. Collier. 2010. Disease susceptibility of salmon exposed to polybrominated diphenyl ethers (PBDEs). *Aquatic Toxicology* 98(1):51–59.

Arkoosh, M.R., D. Boylen, J. Dietrich, B.F. Anulacion, G. Ylitalo, C.F. Bravo, L.L. Johnson, F.J. Loge, and T.K. Collier. 2010. Disease susceptibility of salmon exposed to polybrominated diphenyl ethers (PBDEs). *Aquatic Toxicology* 98(1):51-59.

Arkoosh, M.R., E. Clemons, P. Huffman, A.N. Kagley, E. Casillas, N. Adams, H.R. Sanborn, T.K. Collier, and J.E. Stein. 2001. Increased susceptibility of juvenile Chinook salmon to vibriosis after exposure to chlorinated and aromatic compounds found in contaminated urban estuaries. *Journal of Aquatic Animal Health* 13:257–268.

Arkoosh, M.R., and T.K. Collier. 2002. Ecological risk assessment paradigm for salmon: analyzing immune function to evaluate risk. *Human and Ecological Risk Assessment* 8(2):265–276.

Arkoosh, M.R., S. Strickland, A. Van Gaest, G.M. Ylitalo, L. Johnson, G.K. Yanagida, T.K. Collier, and Dietrich JP. 2011. Trends in organic pollutants and lipids in juvenile Snake River spring Chinook salmon with different outmigrating histories through the Lower Snake and Middle Columbia Rivers. Science of the Total Environment 409(23):5086–5100.

Bailey, R.G. 1996. *Ecosystem Geography*. New York: Springer.

Barnam, C., and A. Baxter. 1998. *Condition Factor, K, for Salmonid Fish.* Fisheries Notes FN005. Melbourne, Australia: State of Victoria Department of Primary Industries.

Beamish, R.J., I.A. Pearsall, and M.C. Healey. 2003. A history of the research on the early marine life of Pacific salmon off Canada's Pacific coast. *North Pacific Anadromous Fish Commission Bulletin* 3:1–40.

Beckvar, N., T.M. Dillon, and L.B. Read. 2005. Approaches for linking whole-body fish residues of mercury or DDT to biological effects thresholds. *Environmental Toxicology and Chemistry* 24:2094–2105.

Berg, M.E., E.S. Rosenberg, and D.R. Tripp. 1975. Vegetation associations and primary productivity of the Nisqually salt marsh on Southern Puget Sound, Washington. Pp. 109-144 in Contributions to the natural history of the Nisqually salt marsh on Southern Puget Sound, Washington. (S.G. Herman and A.M. Wiedeman, eds.). The Evergreen State College, Olympia, Washington.

Biro, P.A., A.E. Morton, J.R. Post, and E.A. Parkinson. 2004. Over-winter lipid depletion and mortality of age-0 rainbow trout (*Oncorhynchus mykiss*). *Canadian Journal of Fisheries and Aquatic Sciences* 61:1513–1519.

Borde, A.B., S.A. Zimmerman, V.I. Cullinan, J. Sagar, H.L. Diefenderfer, K.E. Buenau, R.M. Thom, C.A. Corbett, R.M. Kaufmann. 2012. Lower Columbia River and Estuary Ecosystem Restoration Program Reference Site Study: 2011 Restoration Analysis, Prepared for Lower Columbia River Estuary Partnership by Pacific Northwest National Laboratory. Sequim, Washington: Battelle Marine Sciences Laboratory.

Borde, A.B., S.A. Zimmerman, R.M. Kaufmann, H.L. Diefenderfer, N.K. Sather, and R.M. Thom. 2011. *Lower Columbia River and Estuary Restoration Reference Site Study: 2010 Final Report and Site Summaries.* PNWD-4262. Prepared for the Lower Columbia Estuary Partnership. Sequim, Washington: Battelle Marine Sciences Laboratory.

Bottom, D., G. Anderson, A. Baptista, J. Burke, M. Burla, M. Bhuthimethee, L. Campbell, E. Casillas, S. Hinton, K. Jacobson, D. Jay, R. McNatt, P. Moran, C. Roegner, C. Simenstad, V. Stamatiou, D. Teel, and J. Zamon. 2008. Salmon Life Histories, Habitat, and Food Webs in the Columbia River Estuary: An Overview of Research Results, 2002–2006. Prepared for the U.S. Army Corps of Engineers and Bonneville Power Administration, Portland, Oregon. National Marine Fisheries Service Northwest Fisheries Science Center.

Bottom, D. L. and K. K. Jones. 1990. Species composition, distribution, and invertebrate prey of fish assemblages in the Columbia River estuary. *Progress in Oceanography* 25:243-270.

Bottom, D.L., C.A. Simenstad, J. Burke, A.M. Baptista, D.A. Jay, K.K. Jones, E. Casillas, and M.H. Schiewe. 2005. *Salmon at River's End: The Role of the Estuary in the Decline and Recovery of Columbia River Salmon.* NOAA Technical Memorandum NMFS-NWFSC-68. U.S. Department of Commerce.

Buck, J.A., R.G. Anthony, C.A. Schuler, F.B. Isaacs, and D.E. Tillitt. 2005. Changes in productivity and contaminants in bald eagles nesting along the lower Columbia River, USA. *Environmental Toxicology and Chemistry* 24(7):1779–1792.

Casillas, E. 2009. Role of tidal saltwater habitats for juvenile salmonids (myths vs. reality in the Columbia River estuary). Presentation to the Northwest Power Conservation Council, September 2009, Astoria, Oregon.

Clarke, K.R., and R.M. Warwick. 2001. *Change in Marine Communities: An Approach to Statistical Analysis and Interpretation*. 2nd ed. Plymouth, United Kingdom: PRIMER-E.

Cohen, A.N., and J.T. Carlton. 1998. Accelerating invasion rate in a highly invaded estuary. *Science* 279:555–558.

Collier, T.K., and U. Varanasi. 1991. Hepatic activities of xenobiotic metabolizing enzymes and biliary levels of xenobiotics in English sole (*Parophrys vetulus*) exposed to environmental contaminants. *Archives of Environmental Contamination and Toxicology* 20:462–473.

Columbia River DART. 2012. DART river environment graphics and text. Available at: http://www.cbr.washington.edu/dart/river.html. Accessed October 2012.

Cone, J. 1995. *A Common Fate: Endangered Salmon and the People of the Pacific Northwest.* New York: Henry Holt and Company.

Cornwell, T.J., D.L. Bottom, and K.K. Jones. 2001. *Rearing of juvenile salmon in recovering wetlands of the Salmon River estuary.* Oregon Department of Fish and Wildlife. Information Reports 2001-2005, Portland, OR.

Craft, C.B. 2007. Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and U.S. tidal marshes. *Limnology and Oceanography* 52:1220–1230.

Dawley E., R.D. Ledgerwood, T.H. Blahm, C.W. Sims, J.T. Durkin, R.A. Kirn, A.E. Rankis, G.E. Monan, and F.J. Ossiander. 1986. *Migrational Characteristics, Biological Observations, and Relative Survival of Juvenile Salmonids Entering the Columbia River Estuary, 1966-1983.* National Marine Fisheries Service and National Oceanic and Atmospheric Administration, prepared for Bonneville Power Administration and U.S. Department of Energy Division of Fish and Wildlife, Portland, Oregon.

Diefenderfer, H.L., A.M. Coleman, A.B. Borde, and I.A. Sinks. 2008. Hydraulic geometry and microtopography of tidal freshwater forested wetlands and implications for restoration, Columbia River, U.S.A. *Ecohydrology and Hydrobiology* 8:339–361.

Diefenderfer, H.L. and D.R. Montgomery. 2009. Pool spacing, channel morphology, and the restoration of tidal forested wetlands of the Columbia River, U.S.A. *Restoration Ecology* 17:158–168.

Duffy, E.J., D.A. Beauchamp, R.M. Sweeting, R.J. Beamish, and J.S. Brennan. 2010. Ontogenetic diet shifts of juvenile Chinook salmon in nearshore and offshore habitats of Puget Sound. *Transactions of the American Fisheries Society* 139:803–823.

Elliot, C.M. 2004. Environmental and historical factors driving vegetation communities on Russian Island, Columbia River Estuary. Unpublished M.S. thesis, University of Washington, Seattle.

Feely, R. A., C. L. Sabine, J. M. Hernandez-Ayon, D. Ianson and B. Hales. 2008. Evidence for Upwelling of Corrosive "Acidified" Water onto the Continental Shelf, *Science* 13 June 2008: Vol. 320. no. 5882.

Fresh, K.L., E. Casillas, L.L. Johnson, and D.L. Bottom. 2005. *Role of the Estuary in the Recovery of Columbia River Basin Salmon and Steelhead: An Evaluation of the Effects of Selected Factors on Salmonid Population Viability.* NOAA Technical Memorandum NMFS-NWFSC-69. U.S. Department of Commerce.

Fresh, K.L., D.J. Small, H. Kim, C. Waldbillig, M. Mizell, M.I. Carr, and L. Stamatiou. 2006. *Juvenile Salmon Use of Sinclair Inlet, Washington, in 2001 and 2002.* Report FRT-05-06. Olympia: Washington State Department of Fish and Wildlife.

Fulton, T. 1902. *Rate of Growth of Sea Fishes.* Scientific Investigations Report 20. Glasgow: Fishery Board of Scotland.

Garono, R.J., B.D. Anderson, R. Robinson. 2003. Change in Land Cover along the Lower Columbia River Estuary as Determined from LandSAT TM Imagery. Available from Lower Columbia River Estuary Partnership, Portland, OR.

Good, T.P., R.S. Waples, and P.B. Adams. 2005. *Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead.* NOAA Technical Memorandum NMFS-NWFSC-66. U.S. Department of Commerce.

Harnish R.A., G.E. Johnson, G.A. McMichael, M.S. Hughes, and B.D. Ebberts. 2012. “Effect of Migration Pathway on Travel Time and Survival of Acoustic-Tagged Juvenile Salmonids in the Columbia River Estuary.” *Transactions of the American Fisheries Society* 141:507–519.

Healey, M.C. 1991. Life history of Chinook salmon (*Oncorhynchus tshawytscha*). In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis, pp. 312–393. Vancouver, Canada: UBC Press.

Hecht, S. A., D. H. Baldwin, C. A. Mebane, T. Hawkes, S. J. Gross, N. L. Scholz. 2007. *An overview of sensory effects on juvenile salmonids exposed to dissolved copper: Applying a benchmark concentration approach to evaluate sublethal neurobehavioral toxicity.* NOAA Tech. Memo., NMFS-NWFSC-83.

Henny, C.J., R.A. Grove, and J.L. Kaiser. 2008. Osprey distribution, abundance, reproductive success and contaminant burdens along the lower Columbia River, 1997/1998 versus 2004. *Archives of Environmental Contamination and Toxicology* 54:525–534.

Henny, C.J., R.A. Grove, J.L. Kaiser, B.L. Johnson, C.V. Furi, and R.J. Letcher. 2010. Wastewater dilution index partially explains observed polybrominateddiphenyl ether flame retardant concentrations in osprey eggs from Columbia River Basin, 2008–2009. *Ecotoxicology* 20(4):682–697.

Hinck, J., C.J. Schmitt, V.S. Blazer, N.D. Denslow, T.M. Bartish, P.J. Anderson, J.J. Coyle, G.M. Dethloff, and D.E. Tillitt. 2006. Environmental contaminants and biomarker responses in fish from the Columbia River and its tributaries: Spatial and temporal trends. *Science of the Total Environment* 366(2–3):549–578.

Independent Scientific Advisory Board (ISAB). 2007. *Climate Change Impacts on Columbia River Basin Fish and Wildlife,* ISAB 2007-2.

Jay, D.A., K. Leffler, H.L. Diefenderfer, and A.B. Borde. In review 2011. Tidal-fluvial and estuarine processes in the Lower Columbia River: I. Along-channel water level variations, Pacific Ocean to Bonneville Dam. *Estuaries and Coasts*.

Jefferson, C.A. 1975. Plant communities and succession in Oregon coastal salt marshes. Unpublished Ph.D. dissertation, Oregon State University, Corvallis.

Johnson, G.E. (ed.). 2007. *Evaluating Cumulative Ecosystem Response to Restoration Projects in the Columbia River Estuary, Annual Report 2006.* PNNL-16561. Prepared for the U.S. Army Corps of Engineers, Portland District. Richland, Washington: Pacific Northwest National Laboratory.

Johnson, L. L., B. F. Anulacion, M. R. Arkoosh, O. P. Olson, C. A. Sloan, S. Y. Sol, J. A. Spromberg, D. J. Teel, G. K. Yanagida, G. M. Ylitalo. 2012. Persistent Organic Pollutants in Juvenile Chinook Salmon in the Columbia Basin: Implications for Stock Recovery. *Transactions of the American Fisheries Society (in press).*

Johnson, G.E., H.L. Diefenderfer, T.J. Berquam, B.D. Ebberts, C. Tortorici, and J.D. Wilcox. 2004. *Plan for Research, Monitoring, and Evaluation of Salmon in the Columbia River Estuary.* PNNL - 14632. Richland, Washington: Pacific Northwest National Laboratory.

Johnson, G.E., N.K. Sather, A.J. Storch, D.J. Teel, J.R. Skalski, E.M. Dawley, A.J. Bryson, G.R. Ploskey, C. Mallette, T.A. Jones, A.B. Borde, S.A. Zimmerman, E.S. Van Dyke, D.R. Kuligowski, and K.L. Sobocinski. 2011. *Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats of the Lower Columbia River, 2007-2010*. PNNL-20083, Pacific Northwest National Laboratory, Richland, Washinton.

Johnson, L.L., G.M. Ylitalo, M.R. Arkoosh, A.N. Kagley, C.L. Stafford, J.L. Bolton, J. Buzitis, B.F. Anulacion, and T.K. Collier. 2007. Contaminant exposure in outmigrant juvenile salmon from Pacific Northwest estuaries. *Environmental Monitoring and Assessment* 124:167–194.

Johnson, O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. *Status Review of Chum Salmon from Washington, Idaho, Oregon, and California.* NOAA Technical Memorandum NMFS-NWFSC-32. U.S. Department of Commerce.

Johnson, L. L., G. M. Ylitalo, M. S. Myers, B. F. Anulacion, J. Buzitis, W. L. Reichert, T. K. Collier. 2009. *Polycyclic aromatic hydrocarbons and fish health indicators in the marine ecosystem in Kitimat, British Columbia.* U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-98, 123 p.

Johnson, R., and D. Wichern. 2002. *Applied Multivariate Statistical Analysis.* 5th ed. Upper Saddle River, New Jersey: Prentice Hall.

Kalinowski, S.T., K.R. Manlove, and M.L. Taper. 2007. ONCOR a computer program for genetic stock identification. Montana State University, Department of Ecology, Bozeman. Available at: montana.edu/kalinowski/Software/ONCOR.htm.

Kareiva, P., M. Marvier, and M. McClure. 2000. Recovery and management options for spring/summer Chinook salmon in the Columbia River basin. *Science* 290:977–979.

Kentula, M., R. Brooks, S. Gwin, C. Holland, A. Sherman, and J. Sifneos. 1992. *An Approach to Improving Decision Making in Wetland Restoration and Creation.* EPA/600/R-92/150. Washington, D.C.: U.S. Environmental Protection Agency.

Korman, J., P.F. Bravender, and C.D. Levings. 1997. Utilization of the Campbell River estuary by juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in 1994. *Canadian Technical Report of Fisheries and Aquatic Sciences* No. 2169. Fisheries and Oceans Canada.

Krahn, M.M., L.K. Moore, and W.D. MacLeod, Jr. 1986. *Standard Analytical Procedures of the NOAA National Analytical Facility, 1986: Metabolites of Aromatic Compounds in Fish Bile.* NOAA Technical Memorandum NMFS-NWC-102. U.S. Department of Commerce.

Kukulka, T. and D.A. Jay. 2003. *Impacts of Columbia River Discharge on Salmonid Habitat: A Nonstationary Fluvial Tide Model*. Department of Environmental Science and Technology, Oregon Health and Science University, Beaverton, OR.

Lechowicz, M.J. 1982. The sampling characteristics of electivity indices. *Oecologia* 52:22–30.

Leck, M.A., A.H. Baldwin, V.T. Parker, L. Schile, and D.F. Whigham. 2009. Plant communities of tidal freshwater wetlands of the continental USA and southeastern Canada. In *Tidal Freshwater Wetlands*, edited by A. Barendregt, D. Whigham, and A. Baldwin, pp. 41–58. Leiden, Netherlands: Backhuys Publishers.

Levy, D.A., and T.G. Northcote. 1982. Juvenile salmon residency in a marsh area of the Fraser River estuary. *Canadian Journal of Fisheries and Aquatic Sciences* 39(2):270–276.

Lott, M.A. 2004. Habitat-specific feeding ecology of ocean-type juvenile Chinook salmon in the lower Columbia River estuary. Unpublished M.S. thesis, University of Washington, Seattle.

Lower Columbia Estuary Partnership (Estuary Partnership). 1999. *Comprehensive Conservation and Management Plan*. Portland, Oregon: Lower Columbia Estuary Partnership.

———. 2004. *Columbia River Estuary Habitat Monitoring Plan.* Portland, Oregon: Lower Columbia Estuary Partnership.

———. 2007. *Lower Columbia River Estuary Ecosystem Monitoring: Water Quality and Salmon Sampling Report*. Lower Columbia Estuary Partnership.

———. 2012. *A Guide to the Lower Columbia River Ecosystem Restoration Program, Second Technical Review Draft.* December 14, 2012. Portland, Oregon: Lower Columbia Estuary Partnership.

Lowry, O.H., N.J. Rosebrough, A.L. Farr, and R.J. Randall. 1951. Protein measurement with the Folin-Phenol reagents. *Journal of Biological Chemistry* 193:265–275.

Magie R., A. Cameron, R. Ledgerwood, B. Sandford, and G. Matthews. 2008. *Detection of PIT-Tagged Juvenile Salmonids in the Columbia River. Estuary using Pair-Trawls,* 2006. Report of research by Fish Ecology Division, Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, Washington.

MacFarlane, R.B. 2010. Energy dynamics and growth of Chinook salmon (*Oncorhynchus tshawytscha*) from the Central Valley of California during the estuarine phase and first ocean year. *Canadian Journal of Fisheries and Aquatic Sciences* 67:1549–1565.

Manel, S., O.E. Gaggiotti, and R.S. Waples. 2005. Assignment methods: matching biological questions with appropriate techniques. *Trends in Ecology and Evolution* 20:136–142.

Marine, K.R. and J.J. Chech. 1998. Effects of elevated water temperature on some aspects of the physiological and ecological performance of juvenile Chinook salmon, *Oncorhynchus tshawytscha*: implications for management of California's Chinook salmon stocks. Stream Temperature Monitoring and Assessment Workshop, January 1998. Sacramento, California, Forest Science Project. Humboldt State University, Arcata, California.

McCabe, G.T., R.L. Emmett, W.D. Muir and T.H. Blahm. 1986. Utilization of the Columbia River estuary by subyearling Chinook salmon. *Northwest Science* 60:113–124.

McClure, M.M., E.E. Holmes, B.L. Sanderson, and C.E. Jordan. 2003. A large-scale, multi-species status assessment: Anadromous salmonids in the Columbia River. *Ecological Applications* 13:964–989.

McCullough, D.A. 1999. *A Review and Synthesis of Effects of Alterations to the Water Temperature Regime on Freshwater Life Stages of Salmonids with Special Reference to Chinook Salmon.* EPA 910-R-88-010. Seattle, Washington: U.S. Environmental Protection Agency, Region 10.

McElhany, P., M.H. Ruckelshaus, M.J. Ford, T.C. Wainwright, and E.P. Bjorkstedt. 2000. *Viable Salmon Populations and the Recovery of Evolutionarily Significant Units.* NOAA Technical Memorandum NMFS-NWFSC-42. U.S. Department of Commerce.

Meador, J.P., J. Buzitis, and C.F. Bravo. 2008. Using fluorescent aromatic compounds (FACs) in bile from juvenile salmonids to predict exposure to polycyclic aromatic hydrocarbons. *Environmental Toxicology and Chemistry* 27(4):845–853.

Meador J.P., T.K. Collier, and J.E. Stein. 2002. Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juvenile salmonids listed under the U.S. Endangered Species Act. *Aquatic Conservation: Marine and Freshwater Ecosystems* 12:493–516.

Mitsch, W.J. and J.G. Gosselink. 2000. *Wetlands*. 3rd ed. New York: John Wiley and Sons.

Morace, J.L. 2012. *Reconnaissance of contaminants in selected wastewater-treatment-plant effluent and stormwater runoff entering the Columbia River, Columbia River Basin, Washington and Oregon, 2008–10*. U.S. Geological Survey Scientific Investigations Report 2012–5068.

Myers, K.W.W. 1980. *An investigation of the utilization of four study areas in Yaquina Bay, Oregon, by hatchery and wild juvenile salmonids.*M.S. thesis, Oregon State University, Corvallis, 234 p

Myers, K.W., and H.F. Horton. 1982. Temporal use of an Oregon estuary by hatchery and wild juvenile salmon. In *Estuarine Comparisons*, edited by V.S. Kennedy, pp. 377–392. New York: Academic Press.

Naiman, Robert J., J. Richard Alldredge, David A. Beauchamp, Peter A. Bisson, James Congleton, Charles J. Henny, Nancy Huntly, Roland Lamberson, Colin Levings, Erik N. Merrill, William G. Pearcy, Bruce E. Rieman, Gregory T. Ruggerone, Dennis Scarnecchia, Peter E. Smouse and Chris C. Wood. 2012. Developing a broader scientific foundation for river restoration: Columbia River food webs, *Proceedings from the National Academy of Sciences*, December 26, 2012, vol. 109 no**. 52 21201-21207.**

National Wildlife Federation (NWF). 2007. *Sea-level Rise and Coastal Habitats in the Pacific Northwest: An Analysis for Puget Sound, Southwestern Washington, and Northwestern Oregon, July 2007* Prepared by Patty Glick, Jonathan Clough, and Brad Nunley, National Wildlife Federation.

Netboy, A. 1980. The *Columbia River Salmon and Steelhead Trout, Their Fight for Survival.* Seattle: University of Washington Press.

Neter, J., M. Kutner, C. Nachtsheim, and W. Wasserman. 1996. *Applied Linear Statistical Models.* 4th ed. San Francisco: WCB/McGraw-Hill.

National Marine Fisheries Service (NMFS). 2008. *Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation;* Consultation on Remand for Operation of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(I)(A) Permit for Juvenile Fish Transportation Program (Revised and reissued pursuant to court order, NWF v. NMFS, Civ. No. CV 01-640-RE (D. Oregon).

Northwest Power and Conservation Council (NPCC). 2004. *Mainstem Lower Columbia River and Columbia River Estuary Subbasin Plan*. May 2004. Prepared for Northwest Power and Conservation Council. Portland, Oregon: Lower Columbia Estuary Partnership; Longview, Washington, Lower Columbia Fish Recovery Board.

Odum, W.E. 1988. Comparative ecology of tidal freshwater and salt marshes. *Annual Review of Ecology and Systematics* 19:147–176.

Odum, W.E., T.J. Smith III, J.K. Hoover, and C.C. McIvor. 1984. *The Ecology of Tidal Freshwater Marshes of the United States East Coast: A Community Profile.* FWS/OBS-83/17. U.S. Fish Wildlife Service.

Oregon Department of Environmental Quality (ODEQ). 2010a. Oregon 2010 integrated water quality report and 303(d) listing. Available at: http://www.deq.state.or.us/wq/assessment/rpt2010/search.asp.

———. 2010b. Water quality standards. Available at: http://www.deq.state.or.us/wq/standards/standards.htm. Accessed March 22, 2010.

Petersen, J.H., and J.F. Kitchell. 2001. Climate regimes and water temperature changes in the Columbia River: Bioenergetic implications for predators of juvenile salmon. *Canadian Journal of Fisheries and Aquatic Sciences* 58:1831–1841.

Poirier J., S. Lohr, T.A. Whitesel, and J. Johnson. 2009a. *Assessment of Fishes, Habitats, and Fish Passage at Tide Gates on Deer Island Slough and Lower Tide Creek,* 2009 Project Report. U.S. Fish and Wildlife Service, Vancouver, Washington.

Poirier, J., J.Johnson, J. Jolley, G. Silver, M. Hudson, S. Lohr and T. A. Whitesel, 2009b. *Presence, distribution and movement of select aquatic species in Tide Creek, Merrill Creek and Deer Island Slough, Columbia County, Oregon* 2009 Progress Report, U.S. Fish and Wildlife Service, Vancouver, Washington.

Principe, N.D. 2005. The feeding dynamics of out-migrated age-0 chinook salmon (*Oncorhynchus tshawytscha*) in Lake Ontario. Unpublished M.S. thesis, Graduate School of Cornell University, Ithaca, New York.

Rannala, B., and J.L. Mountain. 1997. Detecting immigration by using multilocus genotypes. *Proceedings of the National Academy of Sciences* 94: 9197–9201.

Rich, W.H. 1920. Early history and seaward migration of Chinook salmon in the Columbia and Sacramento Rivers. *Bulletin of the Bureau of Fisheries* 37:1–74.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. *Bulletin of the Fisheries Research Board of Canada* 191:1–382.

Rieman, B.E., R.C. Beamesderfer, S. Vigg, and T.P. Poe. 1991. Estimated loss of juvenile salmonids to predation by northern squawfish, walleyes, and smallmouth bass in John Day Reservoir, Columbia River. *Transactions of the American Fisheries Society* 120:448–458.

Roegner, G.C., A. Baptista, D.L. Bottom, J. Burke, L. Campbell, C. Elliot, S. Hinton, D. Jay, M. Lott, T. Lundrigan, R. McNatt, P. Moran, C.A. Simenstad, D. Teel, E. Volk, J. Zamon, and E. Casillas. 2008. *Estuarine Habitat and Juvenile Salmon: Current and Historical Linkages in the Lower Columbia River and Estuary, 2002-2004.* Prepared for United States Army Corps of Engineers, Seattle, Washington: Northwest Fisheries Science Center, National Marine Fisheries Service.

Roegner, G.C., E.W. Dawley, M. Russell, A. Whiting, and D.J. Teel. 2010. Juvenile salmonid use of reconnected tidal freshwater wetlands in Grays River, lower Columbia River basin. *Transactions of the American Fisheries Society* 139:1211–1232.

Roegner, G.C., H.L. Diefenderfer, A.B. Borde, R.M. Thom, E.M. Dawley, A.H. Whiting, S.A. Zimmerman, and G.E. Johnson. 2009. *Protocols for Monitoring Habitat Restoration Projects in the Lower Columbia River and Estuary.* NOAA Technical Memorandum NMFS-NWFSC-97. U.S. Department of Commerce.

Roegner G.C., J.A. Needoba and A.M. Baptista. 2011. "Coastal Upwelling Supplies Oxygen-Depleted Water to the Columbia River Estuary." *Plos One* 6(4):e18672.

Roegner, G.C., and D.J. Teel. Submitted 2012. Morphometric condition of juvenile Chinook salmon in the lower Columbia River and estuary in relation to water temperature and genetic stock of origin. *Transactions of the American Fisheries Society*

Rybczyk, J.M., and D.R. Cahoon. 2002. Estimating the potential for submergence for two subsiding wetlands in the Mississippi river delta. *Estuaries* 25:985–998.

Sagar, J.P., K.E. Marcoe, A.B. Borde, L.L. Johnson, J.L. Morace, T. Peterson, K.H. Macneale, R.M. Kaufmann, V.I. Cullinan, S.A. Zimmerman, R.M. Thom, C.L. Wright, P.M. Chittaro, O.P. Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, D. Lomax, W.B. Temple, A. Silva, C.A. Simenstad, M.F. Ramirez, J.E. O'Connor, C. Cannon, and M. Schwartz. 2012. *Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 7 (September 1, 2010 to December 31, 2011).* Prepared for the Bonneville Power Administration. Portland, Oregon: Lower Columbia Estuary Partnership.

Sagar, J.P., K. E. Marcoe, C. A. Simenstad, M. F. Ramirez, J. L. Burke, J. E. O'Connor, T. D. Counihan, I. R. Waite, A. B. Borde, S. A. Zimmerman, N. K. Sather, R. M. Thom, J. L. Morace, L. L. Johnson, P.M. Chittaro, K. H. Macneale, O. P. Olson, S. Y. Sol, D. J. Teal, G. M. Ylitalo and L. K. Johnson. 2011. *Lower Columbia River Ecosystem Monitoring Project Annual Report for Year 6 (September 2009 to November 2010).* Prepared by Lower Columbia River Estuary Partnership for Bonneville Power Administration.

Sagar, J. P., S. Y. Sol, O. P.Olson, K. H. Macneale, P. M. Chittaro, L.L. Johnson, G.L. Kral, M.A. Rowe Soll, J. M. St. Pierre, R. M. Beaston, S. Holman, A. S. Cameron, M. Russell and A. Silva. 2011b. *Action Effectiveness Monitoring for the Columbia River Estuary Habitat Restoration Program: Annual Report for September 15, 2009 – December 31, 2010.* Prepared by Lower Columbia River Estuary Partnership for Bonneville Power Administration.

Salo, E.O. 1991. Life history of chum salmon, *Oncorhynchus keta*. In *Pacific Salmon Life Histories*, edited by C. Groot and L. Margolis, pp. 231–309. Vancouver, Canada: UBC Press.

Sanderson, B.L., K.A. Barnas, and M. Rub. 2009. Non-indigenous species of the Pacific Northwest: an overlooked risk to endangered salmon? *BioScience* 59:245–256.

Sather N.K., G.E. Johnson, A.J. Storch, D.J. Teel, J.R. Skalski, T.A. Jones, E.M. Dawley, S.A .Zimmerman, A.B. Borde, C. Mallette, and R. Farr. 2009. *Ecology of Juvenile Salmon in*

Shallow Tidal Freshwater Habitats in the Vicinity of the Sandy River Delta, Lower Columbia River, 2008. PNNL-18450, Pacific Northwest National Laboratory, Richland, Washington.

Sather, N.K., D.J. Teal, A.J. Storch, G.E. Johnson, E.S. Van Dyke, E.M. Dawley, D.R. Kuligowski, T.A. Jones, A.J. Bryson, and K.L. Sobocinski. 2011. Juvenile salmon and fish community characteristics. In *Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats of the Lower Columbia River, 2007–2010*, pp. 2.1–4.35. PNNL-20083. Richland, Washington: Pacific Northwest National Laboratory.

Seeb, L.W., A. Antonovich, M.A. Banks, T.D. Beacham, M.R. Bellinger, S.M. Blankenship, M.R. Campbell, N.A. Decovich, J.C. Garza, C.M. Guthrie III, T.A. Lundrigan, P. Moran, S.R. Narum, J.J. Stephenson, K.T. Supernault, D.J. Teel, W.D. Templin, J.K. Wenburg, S.F. Young, and C.T. Smith. 2007. Development of a standardized DNA database for Chinook salmon. *Fisheries* 32:540–552.

Seliskar, D.M., and J.L. Gallagher. 1983. *The Ecology of Tidal Marshes of the Pacific Northwest Coast: A Community Profile*. FWS/OBS-82/32. Washington, D.C.: U.S. Fish and Wildlife Service, Division of Biological Services.

Shannon, C.E., and W. Weaver. 1949. *The Mathematical Theory of Communication*. Urbana: University of Illinois Press.

Sherwood, C.R., J.S. Creager, E.H. Roy, G. Gelfenbaum, and T. Dempsey. 1984. *Sedimentary Processes and Environments in the Columbia River Estuary*. Prepared for the Columbia River Estuary Data Development Program and the Columbia River Estuary Task Force.

Sherwood, C.R., D.A. Jay, R.B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical changes in the Columbia River estuary. *Progress in Oceanography* 25:299–357.

Shreffler, D.K., C.A. Simenstad, and R.M. Thom. 1992. Foraging by juvenile salmon in a restored estuarine wetland. *Estuaries* 15:204–213.

Simenstad, C.A., J.L. Burke, J.E. O'Connor, C. Cannon, D.W. Heatwole, M.F. Ramirez, I.R. Waite, T.D. Counihan, and K.L. Jones. 2011. *Columbia River Estuary Ecosystem Classification—Concept and Application*. Open-File Report 2011-1228. Seattle, Washington: U.S. Geological Survey.

Simenstad, C.A., and J.R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. *Ecological Engineering* 15(3–4):283–302.

Simenstad, C.A., K.L. Fresh, and E.O. Salo. 1982. *The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function.* In Estuarine Comparisons, edited by V.S. Kennedy, pp. 343–364. New York: Academic Press.

Sinclair, M. 1988. *Marine Populations: An Essay on Population Regulation and Speciation.* Seattle: Washington Sea Grant Program.

Sloan, C.A., B.F. Anulacion, J.L. Bolton, D. Boyd, O.P. Olson, S.Y. Sol, G.M. Ylitalo, and L.L. Johnson. 2010. Polybrominated diphenyl ethers in outmigrant juvenile Chinook salmon from the lower Columbia River and estuary and Puget Sound, Washington. *Archives of Environmental Contamination and Toxicology* 58(2):403–414.

Sloan, C.A., D.W. Brown, R.W. Pearce, R.H. Boyer, J.L. Bolton, D.G. Burrows, D.P. Herman, and M.M. Krahn. 2004. *Extraction, Cleanup, and Gas Chromatography/Mass Spectrometry Analysis of Sediments and Tissues for Organic Contaminants.* NOAA Technical Memorandum NMFS-NWFSC-59. U.S. Department of Commerce.

———. 2005. Determining aromatic hydrocarbons and chlorinated hydrocarbons in sediments and tissues using accelerated solvent extraction and gas chromatography/mass spectrometry. In *Techniques in Aquatic Toxicology*, vol. 2, edited by G.K. Ostrander, pp. 631–652. Boca Raton, Florida: CRC Press.

Sloan, C.A., D.W. Brown, G.M. Ylitalo, J. Buzitis, D.P. Herman, D.G. Burrows, G.K. Yanagida, R.W. Pearce, J.L. Bolton, R.H. Boyer, and M.M. Krahn. 2006. *Quality Assurance Plan for Analyses of Environmental Samples for Polycyclic Aromatic Compounds, Persistent Organic Pollutants, Fatty Acids, Stable Isotope Ratios, Lipid Classes, and Metabolites of Polycyclic Aromatic Compounds.* NOAA Technical Memorandum NMFS-NWFSC-77. U.S. Department of Commerce.

Small, L.F., C.D. McIntire, K.B. Macdonald, J.R. Lara-Lara, B.E. Frey, M.C. Amspoker, and T. Winfield. 1990. Primary production, plant and detrital biomass, and particle transport in the Columbia River estuary. *Progress in Oceanography* 25:175–210.

Spilseth, S.A., and C.A. Simenstad. 2011. Seasonal, diel, and landscape effects on resource partitioning between juvenile Chinook salmon (*Oncorhynchus tshawytscha*) and threespine stickleback (*Gasterosteus aculeatus*) in the Columbia River estuary. *Estuaries and Coasts* 34:159–171.

Spyreas, G., B.W. Wilm, A.E. Plocher, D.M. Ketzner, J.W. Matthews, J.L. Ellis, and E.J. Heske. 2010. Biological consequences of invasion by reed canary grass (*Phalaris arundinacea*). *Biological Invasions* 12:1253–1267.

Tabor, R.A., R.S. Shively, and T.P. Poe. 1993. Predation on juvenile salmonids by smallmouth bass and northern squawfish in the Columbia River near Richland, Washington. *North American Journal of Fisheries Management* 13:4, 831–838.

Teel, D.J., C. Baker, D.R. Kuligowski, T.A. Friesen, and B. Shields. 2009. Genetic stock composition of subyearling Chinook salmon in seasonal floodplain wetlands of the lower Willamette River. *Transactions of the American Fisheries Society* 138:211–217.

Thom, R.M. 1992. Accretion rates of low intertidal salt marshes in the Pacific Northwest. *Wetlands* 12(3):147–156.

Thom, R.M., S.L. Blanton, D.L. Woodruff, G.D. Williams, and A.B. Borde. 2001. Carbon sinks in nearshore marine vegetated ecosystems. In *Proceedings of the First National Conference on Carbon Sequestration*. May 14–17, 2001. National Energy Technology Laboratory. Available at www.netl.doe.gov/publications/proceedings/01/carbon_seq/5c5.pdf..

Thom, R., N. Sather, G.C. Roegner, and D.L. Bottom. 2012. *Columbia Estuary Ecosystem Restoration Program. 2012 Synthesis Memorandum.* Prepared for the Portland District Army Corps of Engineers. Richland, Washington: Pacific Northwest National Laboratory; NOAA Fisheries.

Thomas, D.W. 1983. *Changes in Columbia River Estuary Habitat Types over the Past Century*. Astoria, Oregon: Columbia River Estuary Data Development Program.

Thorpe, J.E. 1994. Salmonid fishes and the estuarine environment. *Estuaries* 17:76–93

Vigg, S., T.P. Poe, L.A. Prendergast, and H.C. Hansel. 1991. Rates of consumption of juvenile salmonids and alternative prey fish by northern squawfish, walleyes, smallmouth bass, and channel catfish in John Day Reservoir, Columbia River. *Transactions of the American Fisheries Society* 120:421–438.

Washington Department of Ecology. 2011. Surface water criteria. Available at: http://www.ecy.wa.gov/programs/wq/swqs/criteria.html. Accessed November 25, 2011.

Weitkamp, L.A. 2008. Buoyancy regulation by hatchery and wild coho salmon during the transition from freshwater to marine environments. *Transactions of the American Fisheries Society.* 137:860-868

Weitkamp, L.A., T.C. Wainwright, G.J. Bryant, G.B. Milner, D.J. Teel, R.G. Kope, and R.S. Waples. 1995. *Status Review of Coho Salmon from Washington, Oregon, and California.* NOAA Technical Memorandum NMFS-NWFSC-24. U.S. Department of Commerce.

Weitkamp LA, PJ Bentley, and MN Litz. 2012. "Seasonal and interannual variation in juvenile salmonids and associated fish assemblage in open waters of the lower Columbia River estuary, U.S.A." *Fisheries Bulletin*.

Wedemeyer, G.A., R.L. Saunders, and W.C. Clarke. 1980. Environmental factors affecting smoltification and early marine survival of anadromous salmonids. *Marine Fisheries Review* 6:1–14.

Yanagida, G. K., B. F. Anulacion, J. L. Bolton, D. Boyd, D. P. Lomax, O. P. Olson, S. Sol, M. J. Willis, G. M. Ylitalo, L. L. Johnson. 2012. Polycyclic aromatic hydrocarbons and risk to threatened and endangered Chinook salmon in the Lower Columbia River estuary. *Archives of Environmental Contamination and Toxicology* 62(2):282-295.

Ylitalo, G.M., G.K. Yanagida, L.C. Hufnagle, Jr., and M.M. Krahn. 2005. Determination of lipid classes and lipid content in tissues of aquatic organisms using a thin layer chromatography/flame ionization detection (TLC/FID) microlipid method. In *Techniques in Aquatic Toxicology*, vol. 2, edited by G.K. Ostrander, pp. 227–237. Boca Raton, Florida: CRC Press.