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Lower Columbia River Ecosystem Monitoring Project
Annual Report for Year 6 (September 1, 2009 to November 15, 2010)

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Executive Summary

Our ability to understand the relationships between sensitive organisms, such as salmonids, and the lower Columbia River estuary (LCRE) ecosystem has been hindered by major data gaps and poor access to existing data. The Lower Columbia River Estuary Partnership (Estuary Partnership) implements elements of its Aquatic Ecosystem Monitoring Strategy (LCREP, 1998) to address needs for habitat and toxic contaminant monitoring and data management through its Ecosystem Monitoring Project (EMP). Efforts for the EMP include an ecosystem classification system and on-the-ground monitoring of vegetation, habitat, juvenile salmon, and water quality. This monitoring was 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 addresses RPAs 58, 59, 60, and 61 of the 2008 Biological Opinion. The Estuary Partnership executes the EMP by engaging regional experts at the University of Washington (UW), Battelle-Pacific Northwest National Laboratory (PNNL), National Oceanic and Atmospheric Administration Fisheries (NOAA-Fisheries), and United States Geological Survey (USGS). Financial support for the EMP comes from the Bonneville Power Administration (BPA) and Northwest Power and Conservation Council (NPCC).

This report describes EMP accomplishments during September 1, 2009 to November 15, 2010, or Year 6 of this on-going project. During this period, the Estuary Partnership and monitoring partners:

- Further refined The Columbia River Estuary Ecosystem Classification (Classification) to include: refined mapping of draft Classification Levels 4-6 for Reaches E-H and updated draft report describing the Classification's conceptual basis, methods, and applications as part of ongoing peer-review.
- Collected bathymetry data, filling all remaining medium and high priority data gaps that were identified in the 2007 workshop and are needed for completing the Classification.
- Initiated a high-resolution land cover mapping effort to support the Classification as well as an overall regional need for current estuarine land cover data.
- Facilitated 2009-2010 monitoring efforts by providing GIS support for site selection, coordinating discussions and site field trips, acquiring special use permits for site access, assisting sampling crews, creating a geodatabase of monitoring activities, and managing partner subcontracts (Estuary Partnership).
- Collected datasets (such as vegetation, habitat, prey, and salmonids) at 3 new sites in Reach C, 2 previously sampled sites in Reach F, and 1 previously sampled site in Reach C to characterize habitat, fish, and prey at all sites and assess year-to-year trends at previously sampled sites (PNNL, NOAA-Fisheries, and USGS).
- Compiled Classification and monitoring report contributions from partners into this annual report document (Estuary Partnership).
- Developed scopes of work for the 2010-2011 monitoring efforts (Estuary Partnership, UW, PNNL, USGS, and NOAA-Fisheries).
- Participated in regional monitoring coordination efforts, like Pacific Northwest Aquatic Monitoring Partnership (PNAMP) and Estuary Partnership's Science Work Group (Estuary Partnership).

2010 Results Summary

Tidal Forested Freshwater Wetlands Results

- The tidal freshwater forested wetlands in the lower Columbia River appear to fall into two groups, according to analyses of faunal and floral assemblages: lower estuarine forested wetlands; and mid- and upper estuarine forested wetlands.
- Tides are the dominant hydrological regime affecting these sites on a daily basis, and the hydrological differences between the lower and upper estuarine sites may be a determining factor in the biota present at these sites.

- The mid- and upper estuarine tidal freshwater forested wetlands have more diverse vegetation zones, with the forested zone dominated by deciduous trees, primarily black cottonwood, Oregon ash, and Pacific willow.
- All of the vegetation zones at the mid- and uppder sites have lower species richness than the lower estuarine zones, and in some cases zones are monotypic in composition.
- Detailed information about the community ecology of the freshwater tidal forested wetlands of the Columbia River estuary from this study will be useful to both restoration efforts and the Classification and will be made available on the Estuary Partnerships website.

Habitat Results

Monitoring data collected resulted in the further characterization of Reach C sites, which have greater tidal influence and more complex tidal channels than upstream reaches. The evaluation of the marsh and channel elevations coupled with hydrology data increases our ability to better characterize the drivers for the vegetation communities and potential for fish access and rearing at these sites.

- The correlation between the SEV (sum exceedance value; i.e., magnitude and duration of inundation) and elevation is very strong ($r^2 = 0.75$) indicating that elevation explains much of the variation in SEV. These results are remarkable considering the extremely different hydrologic drivers and geomorphic settings of these sites.
- Vegetation assemblages within Reach C had higher species richness and higher cover than those encountered in other river reaches to date. We suspect that this is due in part to lower hydrological disturbances because tidal action and flood extremes are muted in this reach.
- Inundation time of the channels at the monitoring sites ranged from 56 to 100% of the year, while inundation at the marsh edge of these channels varied from 20 to 55% of the year. These values indicate that these marsh-channel systems are providing significant opportunities for fish to access and feed.

Water Quality Results (Campbell Slough)

Campbell Slough in 2010, similar to previous years, experienced periods of “poor” water quality with respect to conditions for salmon health.

- Water temperatures were greater than 20 degrees Celsius in summer months, low dissolved oxygen levels were found during periods of high water temperature, and pH was often above 8.5, creating stressful conditions for salmon.

Fish and Macroinvertebrate Results

Sampling in 2010, as in 2009, found that unmarked juvenile Chinook, coho, and chum salmon are feeding and rearing in representative tidal freshwater sites in Reach C of the LCRE.

- Chinook salmon were the most abundant juvenile salmon species overall, representing 90% of all salmon captured, as well as the most abundant salmon species at all sites.
- All of the sites had a relatively low species diversity and richness in comparison to the sites we have sampled in other reaches, and were dominated by stickleback. However, they also supported multiple salmon species, including chum, Chinook, and coho salmon.
- Chum salmon were present in April and May only. In 2010, Chinook and coho salmon were present at some sites from April through August.
- High water temperatures, low DO and other conditions may have limited salmon use of some sites in July and August of 2009, as fish were present for a longer period in 2010.
- When reaches were compared, the Reach C sites generally had higher densities (based on CPUE) and higher proportions of unmarked juvenile Chinook salmon than Campbell Slough or the Reach H sites. They also had higher proportions of chum salmon in catches than either the Campbell Slough or Reach H sites.
- Condition factor values showed some variation among sites and years, but were generally within a healthy range (1.0-1.2).
- Juvenile Chinook are often described as opportunistic feeders, but prey selectivity results suggest that they select Dipteran larvae and pupae at greater rates than would be expected given their modest availability.

Fixed Site Synthesis Results: Franz Lake and Campbell Slough

- Juvenile salmon were utilizing both sites from April, when sampling began, until June. Salmon were not found in late July and August sampling events.
- Water temperature and dissolved oxygen levels were prohibitive in July and August at Campbell Slough. This relationship was not tested at Franz Lake due to budget considerations.
- Although wild salmonids were present at both sites, hatchery salmonids made up substantial proportions of the catch.
- The Franz Lake site had a greater diversity of salmonids, with significant numbers of coho, chum, and Chinook, while Chinook predominated at Campbell Slough.
- Fish community characteristics (number of species, species richness and diversity) were similar between sites, but the percentage of non-native species tended to be higher at Campbell Slough.
- Most of Campbell Slough and Franz Lake salmonids are LCR stocks with smaller proportions of upriver stocks.
- Fish length, weight, and condition factor were similar between sites, after differences in proportions of unmarked and hatchery fish and sampling dates are taken into account.
- Contaminant concentrations tended to be highest in juvenile Chinook salmon from Campbell Slough.
- A wide range of prey availability and richness were observed for both sites but fairly similar in terms of abundance of prey collected.
- Across sites and time, prey were more abundant in samples collected nearshore and associated with emergent vegetation relative to samples collected in deeper, open water.
- Juvenile Chinook salmon consumed primarily aquatic fly larva and pupa (Diptera) at both sites.
- Results from a mean selectively analysis, which compares available prey versus consumed prey indicate a high selection of dipterans versus Cyclopoida and Cladocerans, even though these latter macroinvertebrates were more abundant.
- At both sites, the dominant vegetative species were reed canary grass, spike rush and wapato; boundaries between vegetative communities was comparable at both sites and didn’t change between years; and the cover within the communities changed between years (likely due to difference in water levels and cattle grazing at Campbell Slough in 2010).
- Overall, emergent vegetation cover at the sites was comprised of approximately 60% native and 40% non-native species, which did not change significantly between years.
- Variation in vegetation species richness and composition occurred between years, as did inundation patterns.
- Differences in water levels between years likely affected vegetation composition and potential for fish access and feeding.

1.0 Project Background

In September 2003, the Bonneville Power Administration (BPA) and the Northwest Power and Conservation Council (NPCC) awarded a three-year contract to the Lower Columbia River Estuary Partnership (Estuary Partnership) for its Ecosystem Monitoring Project (EMP) focused on the lower Columbia River estuary (LCRE). Prior to this date, the Estuary Partnership's Science Work Group designed some project elements, including toxic contaminant and habitat monitoring. Once funding was secured, BPA project managers finalized the project with the Science Work Group. Plans were made to monitor conventional and toxic pollutants using a multi-species approach (including salmon, eagles, and osprey), and develop a data management strategy. The Estuary Partnership coordinates monitoring and data analysis, resolves problems, develops projects, provides project oversight, and administers the EMP with technical guidance from the Science Work Group.

Although fieldwork was scheduled for late 2003, BPA notified the Estuary Partnership that the project required further refinement and subsequent review by the Independent Scientific Review Panel (ISRP). Specifically, the pollutant monitoring should focus on salmon and the effects of toxic and conventional pollutants in the LCRE on salmon. Furthermore, BPA requested that monitoring for fecal coliform and mercury and data management be removed from the proposal. While the habitat monitoring portion of the project was in relatively good condition, no work could proceed until the pollutant monitoring portion was revised. After the Estuary Partnership, United States Geological Survey (USGS), and National Oceanic and Atmospheric Administration Fisheries (NOAA-Fisheries) revised and re-submitted the toxic contaminant portion, the full monitoring plan was reviewed by the ISRP in April 2004. The ISRP had a favorable review of the toxic contaminant monitoring portion, and given minor revisions, this monitoring could move forward. The habitat monitoring portion, however, did not receive favorable reviews. Thus, the Columbia River Estuary Habitat Monitoring Plan (LCREP, 2004) was drafted to address comments by more clearly defining the goals and methods of the habitat monitoring portion of the EMP.

Following the ISRP's review of the Columbia River Estuary Habitat Monitoring Plan, the Estuary Partnership, Battelle-Pacific Northwest National Laboratory (PNNL), USGS, and University of Washington (UW) worked in Year 2 (September 1, 2004 to August 31, 2005) of the EMP to develop a sampling plan for the LCRE. The Estuary Partnership and monitoring partners use this sampling plan to monitor the status and trends of habitat types in the LCRE. The sampling plan is informed by the draft Columbia River Estuary Ecosystem Classification (Classification) in development by UW and USGS (Simenstad et al., 2007; Simenstad et al., In review) for the EMP. This Classification is based on classified LANDSAT TM imagery, as well as bathymetric, geologic, and various ecological datasets, and was used to identify specific LCRE reaches for sampling during summer 2005. During these 2005 surveys in Reaches D and F (Figure 1 in Study Area), PNNL collected data on habitat conditions including salinity, water depth, temperature, dissolved oxygen, and vegetative cover and derived water elevation estimates. Results of this sampling were summarized in the Columbia River Estuary Habitat Monitoring Pilot Field Study and Remote Sensing Analysis (Sobocinski et al., 2006a).

Additionally, during 2004-2005, NOAA-Fisheries and USGS implemented toxic contaminant monitoring to assess contaminant accumulation in sensitive habitat areas, trends over time, and impacts on salmon. NOAA-Fisheries convened a workshop with managers of other fish, habitat, and water quality monitoring projects in the LCRE (River miles 0-146) to develop a conceptual model for tracking toxic contaminant sources, pathways, and effects on salmon populations (Dietrich et al., 2005). NOAA-Fisheries used this conceptual model to then develop quantitative models describing contaminant uptake and bioaccumulation by juvenile salmon in the LCRE, and ecological risk models linking contaminant body burdens in salmon to health risks such as impaired immune systems, decreased growth rates, and reduced survival rates (Loge et al., 2005; Spromberg and Meador, 2005). The ecological risk models also examine the impacts of these health risks on the survival and productivity of federally listed salmonids. Lastly, in 2004-2005, NOAA-Fisheries sampled fish from April 2005 through September 2005 while USGS conducted fixed station water quality monitoring and installed semipermeable membrane devices (SPMDs) to provide data on conventional and toxics pollutants near the fish sampling sites.

During Year 3 (September 1, 2005 to August 31, 2006) of the EMP, habitat work elements concentrated on vegetation surveys and refinement of the Classification and supporting bathymetric data. In July 2006, PNNL surveyed vegetation at 4 tidally influenced wetlands in Reach G (Figure 1) and re-sampled 2 of the Year 2 Reach F sites in order to assess interannual variability in vegetation cover and composition (Sobocinski et al., 2006b). UW revised the Classification, developed a new Classification level (Geomorphic Catena), created ancillary datasets to refine the classified Landsat TM 2000b classified imagery, finalized stage one of the Landsat TM 2000b refinement, and presented the Classification at several Columbia River and estuary meetings. USGS collected bathymetric data and expended funds to identify additional bathymetric datasets for filling critical data gaps in secondary channels and shallows in priority reaches.

Contaminant work elements of the EMP during 2005-2006 involved analyzing contaminants in juvenile salmon samples, revising contaminant models, and assessing contaminants in the water column. NOAA-Fisheries completed analyses of juvenile salmonid samples (including whole bodies for chlorinated hydrocarbons, stomach contents for chlorinated and aromatic hydrocarbons, bile for metabolites of aromatic hydrocarbons, fin samples for genetic stock determination, and blood for vitellogenin, an indicator of exposure to environmental estrogens) collected in 2004-2005. NOAA also expanded a population model to incorporate population-specific contaminant effects on salmon stocks within the Lower Columbia River Evolutionary Significant Unit (ESU). Models were updated with fish exposure data, water quality, sediment, and salmonid prey information generated from 2005 sampling by NOAA-Fisheries and USGS. Moreover, NOAA-Fisheries incorporated new information on biological effects of contaminants on salmonids into the ecological risks models and explored options for modeling contaminant uptake by juvenile salmonids in the Columbia (e.g., Trophic Trace steady state uptake models). NOAA-Fisheries developed a non-equilibrium model, which may more effectively capture contaminant uptake in salmonids that move quickly through portions of the Columbia River Estuary. USGS retrieved Semipermeable Membrane Devices (SPMDs) from 1 site in the Willamette River and 3 sites in Columbia River, and analyzed samples for polyaromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), organochlorine pesticides (OCs), and polybrominated diphenyl ethers (PBDEs).

In Year 3b (September 1, 2006 to August 31, 2007) of the EMP, the Estuary Partnership and monitoring partners compiled and synthesized the results from past toxic contaminant monitoring efforts (described above). Data describing toxic contaminants in the water column, sediments, and juvenile salmonids (collected by USGS and NOAA-Fisheries, respectively, in Years 2-3) were analyzed and presented in a final report, “The Lower Columbia River and Estuary Ecosystem Monitoring: Water Quality and Salmon Sampling Report” (LCREP, 2007; available in Pisces and on the Estuary Partnership’s website). This report integrates the results of water quality and salmon sampling to document the presence and effects of toxic contaminants on juvenile salmon, including stocks listed under the Endangered Species Act, in the LCRE. NOAA-Fisheries used the information in this report to update the contaminant transport and ecological risk models.

The Estuary Partnership and monitoring partners created tools and built datasets to support comprehensive status and trends monitoring of habitat types in the LCRE. Habitat monitoring work elements for 2006-2007 included refinements to the Classification, identification of bathymetric data gaps, initial designs of a scientifically-sound sampling design, and development of fundamental vegetation datasets. UW and USGS refined the Classification (Simenstad et al., 2007) using completed LiDAR and available bathymetric datasets. USGS used the Classification to begin developing a sampling design strategy intended for use in Years 5-7 of this Project for selecting sampling locations. PNNL continued building fundamental datasets describing wetland vegetation patterns along elevation gradients in the LCRE. Their 2007 surveys expanded vegetation and elevation datasets to include 4 sites in Reach E and included re-sampling of 2 sites in Reach F (Figure 1 in Study Area).

NOAA-Fisheries sampled juvenile salmon at 2 tidal freshwater sites (1 in Reach E and 1 in Reach F), and found that wild juvenile salmon, especially Chinook (*Oncorhynchus tshawytscha*), are feeding and rearing at these sites primarily from early May through July. These sites appear to function as nursery habitat for other fish species as well. NOAA-Fisheries also reported on analyses of previously collected samples. They found that salmon collected in 2005 grew at significantly different rates among sites for each of the 3 time periods tested. Fish from Columbia City had the lowest growth rates, possibly due to their chemical contaminant load. Fish from this area had especially high concentrations of PAHs in their prey and showed uptake of PCBs, DDTs, and PBDEs. Salmon fed on a variety of prey items, including aquatic and terrestrial invertebrates. Chemical testing of salmon found that fish from several sites had elevated vitellogenin levels, indicating that exposure to environmental estrogens may be more widespread than expected. Additionally, salmon from several sites had higher vitellogenin levels in May than in June, which suggests a possible temporal variation to estrogenic compound exposure.

Although contaminant concentrations in juvenile salmon from some sampling sites were relatively high, sediment contaminant levels were uniformly low. When compared to other urban sites in the Pacific Northwest, contaminant levels in the lower Columbia River sediments were low. This suggests that bed sediments may not be the primary source of exposure for juvenile salmon. Instead, contaminants in the food web, on suspended particles, and in the water column may be important sources of exposure. Comparison of contaminant burdens in juvenile Chinook salmon and three-spine sticklebacks (*Gasterosteus aculeatus*, a resident fish species), found that overall, concentrations were higher and less variable in sticklebacks. However, concentrations of PCBs were an exception to this trend, indicating that other factors are influencing salmon body burdens, such as accumulation of contaminants upstream of the sampling site.

Additionally in 2006-2007, analyses of filtered water, suspended sediment, and extracts from SPMDs detected pesticides, pesticide degradation products, pharmaceuticals, personal care products, and other contaminants at nearly all sampling sites. Although the compounds detected were present at levels that are low relative to laboratory reporting limits, their detection in systems as large as the Columbia and Willamette Rivers indicates that they are likely widespread throughout the basin and concentrations may be considerable higher near their sources. These data also indicate that the Willamette River is an important source of contaminants to the estuary.

In Year 4 (September 1, 2007 to August 31, 2008) of the EMP, UW continued their efforts on the Classification, including a revision to the Level 3 hydrogeomorphic boundary between Reaches F and G, inclusion of hydrologic processes and geomorphic structures in the delineation of Level 4 complexes, and development of two ancillary datasets (dikes/levees, dredge material). Additionally, the Estuary Partnership and UW prioritized bathymetric data gaps in the LCRE and developed a data collection strategy for acquisition starting in 2008-2009. The Estuary Partnership, USGS, NOAA-Fisheries, and PNNL formalized the monitoring program’s goal and objectives, examined other sampling design considerations, and assessed the potential of a probabilistic survey design for the EMP at current project funding levels.

With respect to monitoring efforts PNNL collected vegetation and sediment data at 3 new Reach C sites, 2 previously sampled sites in Reach F, and one previously sampled site in Reach C to characterize vegetation and sediment conditions at all sites and to assess year-to-year trends in vegetation at previously sampled sites. NOAA-Fisheries sampled juvenile salmon and their prey at 3 new sites in Reach C, 1 previously sampled site in Reach C and one previously sampled site in Reach F to characterize juvenile salmon occurrence, condition, and prey at all sites and year-to-year trends at the repeated sites. USGS monitored water depth, temperature, dissolved oxygen, pH, turbidity, and conductivity at 1Reach F site to provide water depth and basic chemistry data for integration with results from the vegetation and salmon sampling efforts. Lastly, UW characterized habitat conditions and biological communities at 6 forested tidal freshwater wetlands.

In Year 5 (September 1, 2008 to August 31, 2009) of the EMP, UW along with USGS continued their efforts on the Classification, including delineation of draft Classification Levels 4-6 for Reach F and development of ancillary datasets (dikes/levees, dredge material). Using Level 4, UW compared historical versus current conditions in Reach F. The Estuary Partnership, UW, and USGS developed a report describing the conceptual basis, methods, and applications of the Classification (Simenstad et al., In review). As of November 2009, this document is being peer-reviewed via the USGS publication process. Upon review and revision, the document will be published and made available via USGS and the Estuary Partnership. Completing the Classification to Level 6 for the entire LCRE requires improved bathymetry and landcover data. The Estuary Partnership contracted for bathymetry data collection, and in 2009 filled 12,600 acres of 14,235 acres identified as high and medium priority data gaps at the 2007 workshop. We also hosted a landcover workshop to discuss data gaps and collection strategies and released a Request for Proposals to select a contractor for acquiring these data in 2010-2011. Finally, we processed existing LiDAR data to fill data gaps in riparian topography for use by monitoring and restoration partners in the LCRE.

In Year 6 (September 1, 2009 to November 15, 2010) of the EMP, UW along with USGS continued their efforts to update and expand the Classification. USGS has taken on the role of mapping terrestrial complexes and catena, while UW continued mapping of aquatic complexes as new bathymetric data becomes available. UW completed draft deep water Complexes for the entire LCRE, while USGS continued to refine the classification scheme, and develop Levels 4 and 5 for Reaches D-H. Bathymetric data collection which began in Year 5 was completed in Year 6, with an additional 7000 acres of data gaps filled. All data gaps identified as medium or high priority at the 2007 workshop have been filled, in addition to several thousand additional acres of lower priority areas. A contractor was selected to generate an updated, high resolution land cover data set, and the first stages of image classification, as well as supporting field data collection, have been completed. This work is scheduled to be completed in early 2011.

2.0 EMP Efforts by the Estuary Partnership in 2009-2010

Funding for the EMP supports the Estuary Partnership's Monitoring Coordinator. As part of 2009-2010 EMP efforts, the Monitoring Coordinator (now titled Research Scientist):

- Coordinated development of the Classification and work timelines
- Facilitated discussions and planning for 2009-2010 monitoring efforts
- Coordinated site field trips
- Acquired special use permits for accessing monitoring sites
- Provided field support for EMP monitoring partners
- Coordinated Science Work Group meetings dedicated to the ecosystem monitoring efforts
- Managed EMP subcontracts with UW, PNNL, USGS, and NOAA-Fisheries
- Compiled report contributions from EMP subcontractors into this annual report to BPA
- Developed new scopes of work with EMP subcontractors for the 2010-2011 EMP activities
- Prepared and presented materials for several meetings with BPA, NOAA Fisheries, PNNL, and other regional monitoring partners to determine scope of EMP activities for 2010-2011

EMP funds also support the Research Scientist's work on the Estuary Partnership's Action Effectiveness Monitoring (AEM) program funded by BPA. For this program, the Research Scientist:

- Refined site monitoring plans for 2009-2010 AEM efforts
- Coordinated a Science Work Group meeting dedicated to AEM
- Developed and managed AEM subcontracts with NOAA-Fisheries, Columbia River Estuary Study Taskforce (CREST), Scappoose Bay Watershed Council, and Ash Creek Forest Management for 2009-2010
- Developed new scopes of work with AEM subcontractors for 2010-2011
- Organized and facilitated site trips with subcontractors to discuss AEM methods and challenges and ensure data comparability between sites
- Compiled AEM reports from subcontractors for the Restoration Program's 2009-2010 annual report to BPA

In addition to the work described above for the EMP and AEM programs, the Research Scientist contributed to regional monitoring efforts, such as:

- Coordination and communication amongst parties by staying abreast of RME activities in the LCRE and sharing this information and principal contacts
- Coordination with Pacific Northwest Aquatic Monitoring Partnership (PNAMP) workgroups related to the estuary, Action Effectiveness Monitoring, and Integrated Status and Trends Monitoring
- Development of an inventory of on-going effectiveness monitoring at restoration sites
- Refinements to standardized protocols for restoration effectiveness monitoring

Funding for the EMP also provides partial support for the Estuary Partnership's GIS/Data Management Specialist. For the 2009-2010 EMP efforts, the GIS/Data Management Specialist:

- Managed bathymetry data collection in the LCRE by subcontractor David Evans and Associates.
- Processed final bathymetry raster grids provided by David Evans and Associates in order to maximize funding available on-the-ground bathymetry data collection.
- Developed process to evaluate land cover data development RFPs and select a contractor.
- Managed high resolution land cover data development project contracted to Sanborn Map Co and SWCA, and supported by NOAA C-CAP program.
- Analyzed various GIS datasets to support the 2009-2010 site selection process for on-the-ground monitoring in Reach A,D,E,F and updated geodatabase inventory of EMP monitoring efforts.
- Provided field support for PNNL sampling crews during the 2010 field season.
- Delivered updates on the bathymetry and landcover data collection at SWG meetings.
- Coordinated data sharing efforts in order to disseminate datasets, including those generated by the EMP, to public and private entities engaged in natural resource protection and restoration activities in the LCRE.
- Coordinated development of website mapping functionality to access EMP monitoring data and information online.
- Coordinated Columbia River Ecosystem Classification System (CREEC) development efforts between LCREP, USGS and UW.
- Initiated project to map diked/tidally influenced areas of the LCRE, in support of (CREEC).
- Hired student intern to assist with the diked/tidally influenced mapping efforts.

In addition to the work described above for the EMP program, the GIS/Data Management Specialist contributed to the following regional monitoring efforts:

- Coordinated with the Corps of Engineers to develop a seamless terrain model for the LCRE based on recently aquired bathymetric and topographic data. Model was completed in Fall, 2010.
- Distributed terrain model and new elevation data to several research affiliates.

3.0 EMP Coverage of RPAs in the 2008 Biological Opinion

Work implemented under the Ecosystem Monitoring Project addresses Reasonable and Prudent Alternatives (RPAs) 58, 59, 60, and 61 of the 2008 Biological Opinion for the Federal Columbia River Power System (FCRPS; NMFS 2008). From May – July 2009, the Estuary Partnership presented overviews describing RPA coverage by the EMP to the Bonneville Power Administration, US Army Corps of Engineers, and NOAA-Fisheries. This section summarizes the EMP coverage of RPAs presented to the Action Agencies.

The EMP is the <u>only</u> estuary project covering RPAs 59.1 and 59.2 calling for collecting bathymetry and developing a hierarchical classification system, respectively. See the Bathymetry and Classification sections of this report for more information. The EMP supports RPAs 60.1 and 60.2 as the EMP coordinates with the Reference Site Study (funded by the Estuary Partnership/BPA under the Habitat Restoration contract) to collect data at undistributed sites throughout the LCRE. Coordination between these two projects maximizes efficiency and yields a greater number of reference sites. The EMP supports RPAs 61.1 and 61.3 by collecting data on juvenile salmonid usage of the estuary and Chinook genetic stocks. See Table 1 and Table 2 for coverage summaries of RPAs 58.3 and 59.5, respectively.

Table 1. EMP Coverage of RPA 58.3.

Estuary Projects Covering RPA	Spatial Overlap	# of Sites	Data Collected
7 including EMP	Minimal EMP covers tidal Freshwater portion (Reaches C-H)	19 fish & prey sites ('04-'10) • 11 EMP emergent wetland sites • 3 EMP emergent wetland fixed sites • 9 mainstem sites • 6 prey sites at EMP forested wetlands ('08-'09)	Growth rates • Prey samples in juvenile salmonid stomach contents, emergent vegetation, open water, & benthic cores

Table 2. EMP Coverage of RPA 59.5.

Estuary Projects Covering RPA	Spatial Overlap	# of Sites	Data Collected
8 including EMP	Minimal	32 total ('04-'10) • 23 EMP emergent wetland sites • 4 EMP emergent wetland fixed sites • 6 EMP forested wetland sites ('08-'09)	• Vegetation % cover • Community structure • Topography • Channel cross-sections
		8 total ('04-'09) water quality • 7 EMP emergent wetland sites • 1 EMP emergent wetland fixed site	Vary by site • Temperature, dissolved oxygen, salinity, turbidity, conductivity
		Sites planned for 2011 for primary and secondary productivity	• None

Originally, RPA 59.5 was developed based on the rotational panel design proposed in LCREP (2004). The design called for a synoptic sampling of 160 sites throughout the lower river to inventory the types of habitat and their conditions. Subsequent monitoring would collect data at 8 fixed sites and 12 randomly distributed sites annually rotating around the lower river. This design would allow an understanding of baseline conditions (i.e., status) and changes in those conditions over time (i.e., trends) in a cost effective manner. Table 3 summarizes the proposed data collection design. To date, this design has not been fully implemented in the LCRE. Monitoring is limited spatially to 4-6 sites per year and 1-2 habitat types and limited coverage in the parameters sampled. Monitoring by the EMP largely focuses on undisturbed emergent wetlands with some freshwater wetland sampling, limiting results to these habitat types alone (i.e., data cannot be extrapolated to other habitats). Additionally, water quality and sediment data collection are limited to a few sites and primary and secondary productivity are not monitored. Due to the limited implementation of the rotational panel design, the data collected to date support minimal status and trends analyses.

Table 3. Summary of rotational panel design proposed for estuary monitoring in LCREP (2004).

Phase	# of Sites	Monitoring Parameters	Status
1 Inventory	20 sites per reach (160 sites total sampled in 1 year)	• Landscape features, hydrology, sediment, basic WQ, macroinvertebrates and vegetation at all 160 sites	• Limited implementation • Sample veg. & macroinverts at 3-4 sites per reach each summer • Sample hydrology & WQ at subset • Sample sediment qualitatively

			Landscape features not assessed
2 Long-term Monitoring	8 fixed sites in 8 reaches; 12 randomly distributed sites in 1-2 reaches (20 sites total/year)	Same as above	• Limited implementation • 1 fixed site in Reach F for veg, fish & macroinverts & WQ ('07-'10) • 1 fixed site in Reach F for veg ('05-'10) • 1 fixed site in Reach H for veg., fish & macroinverts ('08-'10) • 1 fixed site in Reach C for veg., fish, & macroinverts ('10)

Overall, all tasks within the EMP address multiple RPAs and implement the 2008 FCRPS Biological Opinion. This data collection provides juvenile salmonid stock data in understudied reaches, feeds into development of regional restoration strategies, provides key data on habitat, prey resources, and juvenile salmonid usage of wetland habitats, and yields reference site data for implementation and evaluation of restoration actions.

4.0 Study Area

The lower Columbia River estuary (LCRE) is designated an "Estuary of National Significance" and as such is part of the National Estuary Program, established in Section 320 of the Clean Water Act. The Ecosystem Monitoring Project's (EMP) study area encompasses all tidally influenced waters of the LCRE, extending from the plume of the Columbia at river mile (RM) 0 upstream to the Bonneville Dam at RM 146. The LCRE extends from the plume of the Columbia River at river mile (RM) 0 upstream to the Bonneville Dam at RM 146. The Estuary Partnership and monitoring partners collect data for the EMP on habitats supporting juvenile salmonids, including shallow emergent wetlands, undiked tidally influenced sloughs adjacent to the Columbia River, scrub/shrub forested wetlands, and mud/sand flats.

The Estuary Partnership and monitoring partners use a multi-scaled stratification sampling design for the emergent wetland component of the EMP based on the Classification. Level 3 of the Classification divides the LCRE into major hydrogeomorphic transitions, yielding 8 reaches, each with unique characteristics and physical processes (Figure 1). Mapping of these Level 3 Reaches was completed in 2007. The Reach boundaries are based on the Environmental Protection Agency's (EPA) Level IV Ecoregions modified to include important parameters such as salinity intrusion, maximum tide level, upstream extent of current reversal, geology, and major tributaries. Previous habitat monitoring efforts for the EMP have concentrated on Reaches D and F (2004-2005), G and F (2005-2006), E and F (2006-2007), H and F (2008-2009), C, F and H (2008-2009). In 2009-2010, the Estuary Partnership and partners monitored emergent wetland habitats in Reaches C, and F.

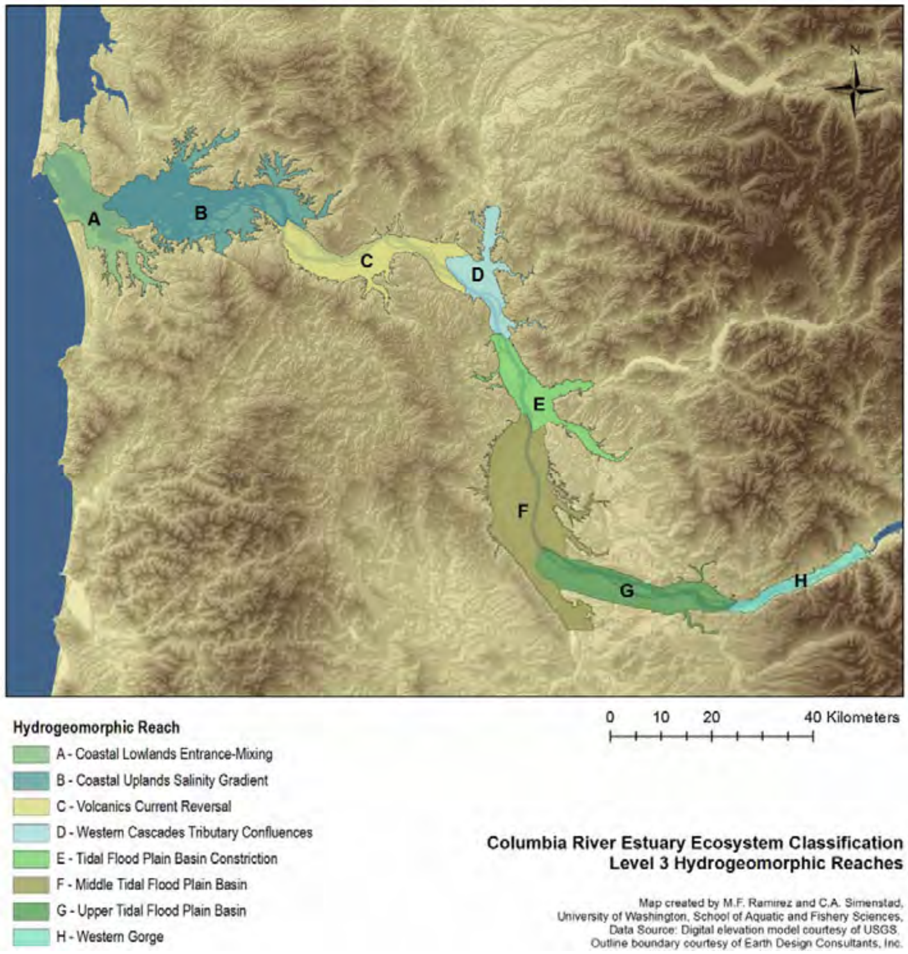


Figure 1. Lower Columbia River and estuary (LCRE) with hydrogeomorphic reaches (A-H) outlined and specified by color (2009 version of hydrogeomorphic reaches).

5.0 Columbia River Estuary Ecosystem Classification (Classification)

The 2009-2010 project period is the seventh year developing and refining the Classification. The Classification is a hierarchical framework that will allow delineation of the diverse ecosystems and component habitats across different scales in the LCRE. The primary purpose of this Classification is to enable systematic monitoring of diverse, scale-dependent, and scale-independent ecosystem attributes. The Classification, however, also provides a more utilitarian framework for understanding the underlying ecosystem processes that create the dynamic structure of the LCRE. As such, it aims to provide the broader community of scientists and managers with a larger scale perspective in order to better study, manage, and restore LCRE ecosystems. Hence, the Classification should also provide an important framework for habitat restoration and protection strategies.

Levels 4-6 are the remaining levels of the Classification to be completed. Completion of these levels for the entire LCRE is dependent on the availability of recent and high quality bathymetric and landcover remote sensing data, and much of this year's work effort was focused on acquisition of these 2 data sets; however, in 2009-2010, we have also worked to refine the Level 4-5 classification scheme to better reflect the underlying geomorphic and hydrologic processes which act as habitat forming processes in the LCRE. A need for this refinement was realized as the mapping efforts from the previous year's Reach F work was reviewed and extended to additional upstream reaches (D, E, G, H). The restructuring has resulted in more complete and systematic mapping and organization of the terrestrial habitats. USGS has been focused on mapping of terrestrial Complexes (Level 4) and Catena (Level 5), and has completed draft maps for Reaches D-H. They have also mapped land-use features that have impacted the landscape (i.e. railroad, ditches, wastewater treatment ponds) , thereby increasing the potential applications of the resulting map products. With the availability of recently acquired bathymetric data (see Section 6.1), UW has completed draft aquatic complexes (including deep water and permanently flooded shallow Complexes) for the entire estuary. Review and refinement of these draft products, and extension of mapping to the remaining Reaches, continues in Year 7.

5.1 Background

Based on classification schemes developed for other estuarine ecosystems and concepts of ecosystem geography (Bailey, 1996), UW and USGS developed a classification scheme for the LCRE that has 6 hierarchical levels:

1) Ecosystem Province (based on EPA Ecoregion Level II)
2) Ecoregion (based on EPA Ecoregion Level III)
3) Hydrogeomorphic Reach (based on modified EPA Ecoregion Levels III and IV)
4) Ecosystem Complex (based on Primary Cover Class and geomorphic setting within each hydrogeomorphic reach)
5) Geomorphic Catena (based on Stanford et al., 2005)
6) Primary Cover Class (based on cover data from LANDSAT or other remote sensing datasets)

For more background information on the Classification, see Leary et al. (2006).

5.2 Classification Level 4: Ecosystem Complexes

Ecosystem complexes involve biophysical patches that reflect both antecedent processes that establish long-term geomorphic templates in the estuary and its floodplain but also reflect continuous processes and changing landscapes. Thus, they include the overlapping of the massive Holocene disturbances (e.g., landslide and volcanic sediment pulses, large floods and storm surges, and tectonic movement) with shorter-term biophysical processes (e.g., more localized flooding, sediment accretion, vegetation succession, local extinction and recruitment events) as well as the reflections of anthropogenic modifications on the landscape such as diking and filling, channel hardening, and urban and suburban development on the floodplain.

Delineation of Ecosystem Complexes in 2009-2010 focused on Hydrogeomorphic Reaches D,E,G, and H, as well as modifications to the draft Reach F layer generated in 2008-2009. USGS assumed responsibility for all terrestrial mapping at the Complex and Catena level. As mapping expanded to these additional Reaches, the need arose for refinement of the classification scheme in order to capture new features that were not previously encountered in the Reach F mapping. Thus, a portion of the 2009-2010 work effort became devoted to refining the scheme, and applying it to the mapping of the new Reaches. The major changes to the applied scheme with respect to Level 4 Complexes include the addition of a 'developed floodplain' complex, as well as the separating out of other complexes that owe to fundamentally distinct geomorphic and hydrologic processes (such as 'floodplain bar & scroll' and 'floodplain backswamp'). In addition, USGS elected to map at the higher Catena (Level 5) level initially. This was believed to be a more efficient process, as the Level 4 Complexes can be quickly and easily derived from the Level 5 information. At the end of the contract period, terrestrial Complexes had not yet been derived from the Catena maps.

The foundation of the Ecosystem Complex level was the isolation of major hydrologic features of the estuary, which are derived from bathymetric data. UW continued work on delineating these aquatic Complexes, as the recently acquired bathymetry became available. A ‘deep-water channel’ Complex was

assigned to depths falling within a maximum 20% quantile. This value was found to be most effective at representing the deep water navigation channel, while at the same time maintaining a relatively continuous segment. Using mean low water estimates relative to Columbia River Datum, a 'permanently flooded' Complex was generated. Draft layers for both the 'deep water' and 'permanently flooded' Complexes were completed for the entire estuary (Figure 2). The final aquatic Complex to be completed is the 'intermittently flooded' Complex, which represents a transition zone between terrestrial and permanently flooded features. Generation of this Complex should be straightforward and will result from the merging of the terrestrial and existing aquatic Complexes.
Figure 2 illustrates the draft aquatic Complexes for the entire estuary, with the exception of the 'intermittently exposed' Complex which has not yet been generated.

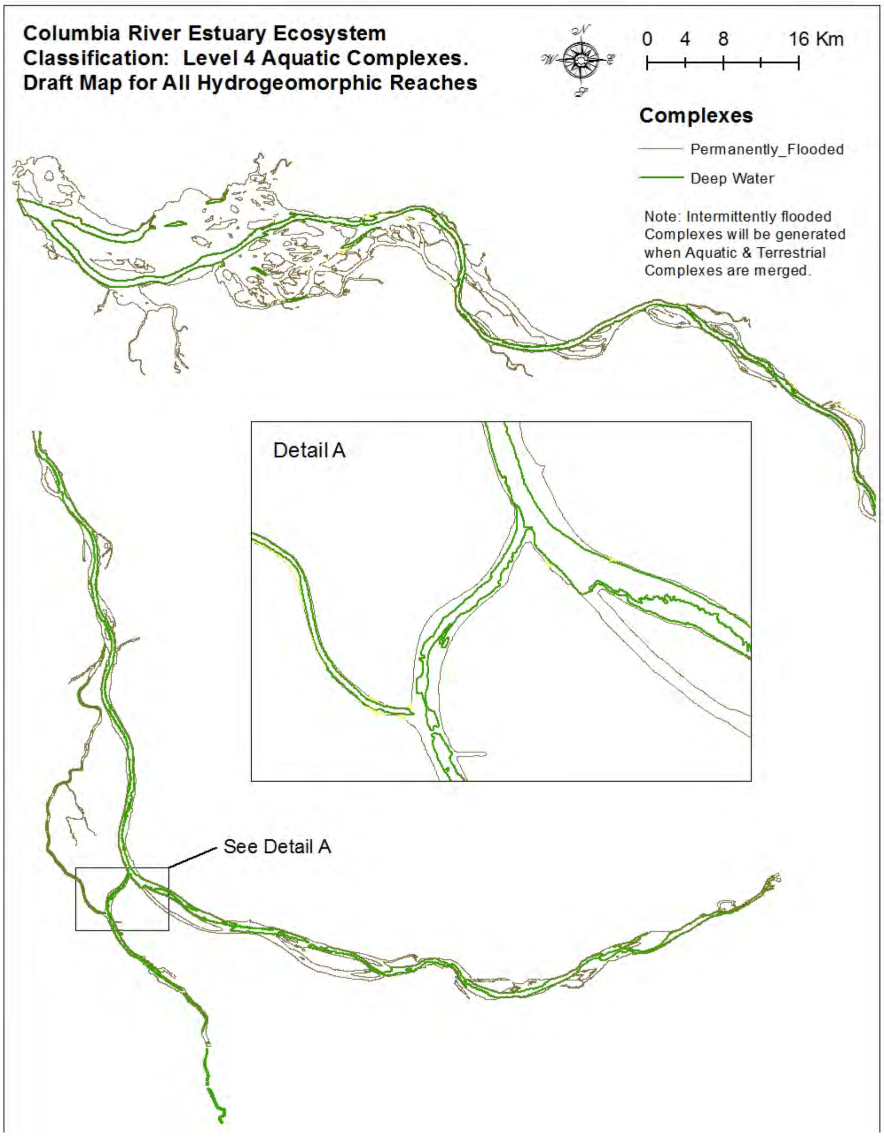


Figure 2. Classification Level 4 Draft Aquatic Ecosystem Complexes illustrated for entire LCRE.

5.3 Classification Level 5: Geomorphic Catena

Geomorphic catena form the mosaic of features nested within ecosystem complexes. Because they vary and change over space and time as a function of both natural ecosystem processes and intrinsic, moderate or minor disturbances, the catena constitute a 3-dimensional shifting mosaic of ecosystems along the river-ocean continuum (Stanford et al., 2005).

Delineation of Ecosystem Catena in 2009-2010 focused primarily on Hydrogeomorphic Reaches D, E,G, and H, as well as modifications to the draft Reach F layer generated in 2008-2009. USGS assumed responsibility for all terrestrial mapping at the Complex and Catena level. Geomorphic catena are classified and delineated in two steps: (1) Use of multiple mapping criteria and sources to distinguish water body and geologic and geomorphic floodplain and adjoining terrestrial features (units) occurring within each complex; and, (2) Application of Level 6—Primary Cover Class data in conjunction with other geospatial data (e.g., LiDAR) to delineate discrete biological associations with the geologic/geomorphic units delineated in step (1). In addition to bathymetry, the primary data sources for the first step included: (1) aerial photography; (2) topography maps; (3) soils maps; (4) geology maps; the primary sources for the second step included the LiDAR and land cover data. Because the acquisition of new land cover data was in progress during the contract year, USGS work efforts were focused exclusively on step 1of this process. Step 2, integration of land cover, will be accomplished in 2010-2011, when recent land cover data becomes available.

Figure 3 illustrates the 25 classes of geomorphic catena identified within Reaches D, E, G, and H, prior to incorporation of land cover information. The Catena level has been mapped at finer resolution than previously, and is focused on individual landforms. This distinction between Catena (landform) and Complex (process domain) provides for more systematic mapping criteria, and a more objective approach that can be better applied by other operators and for other river systems. At this level, the mapped polygons have a median area of 9700 square meters (2.4 acres), thereby providing a high resolution framework for future analysis. A significant change from the 2008-2009 draft Reach F layer is the separation of anthropogenic features from naturally occurring processes, where possible. These anthropogenic features have been assigned to a secondary 'Cultural Features' Catena layer. In areas which would be non-distinguishable in the absence of anthropogenic features, these features have remained in the primary Catena layer. This includes artificial water bodies, as well as some of the dredge fill areas. Figure 4 illustrates the draft 'Cultural Features ' Catena layer for Reaches D,E,G, and H.

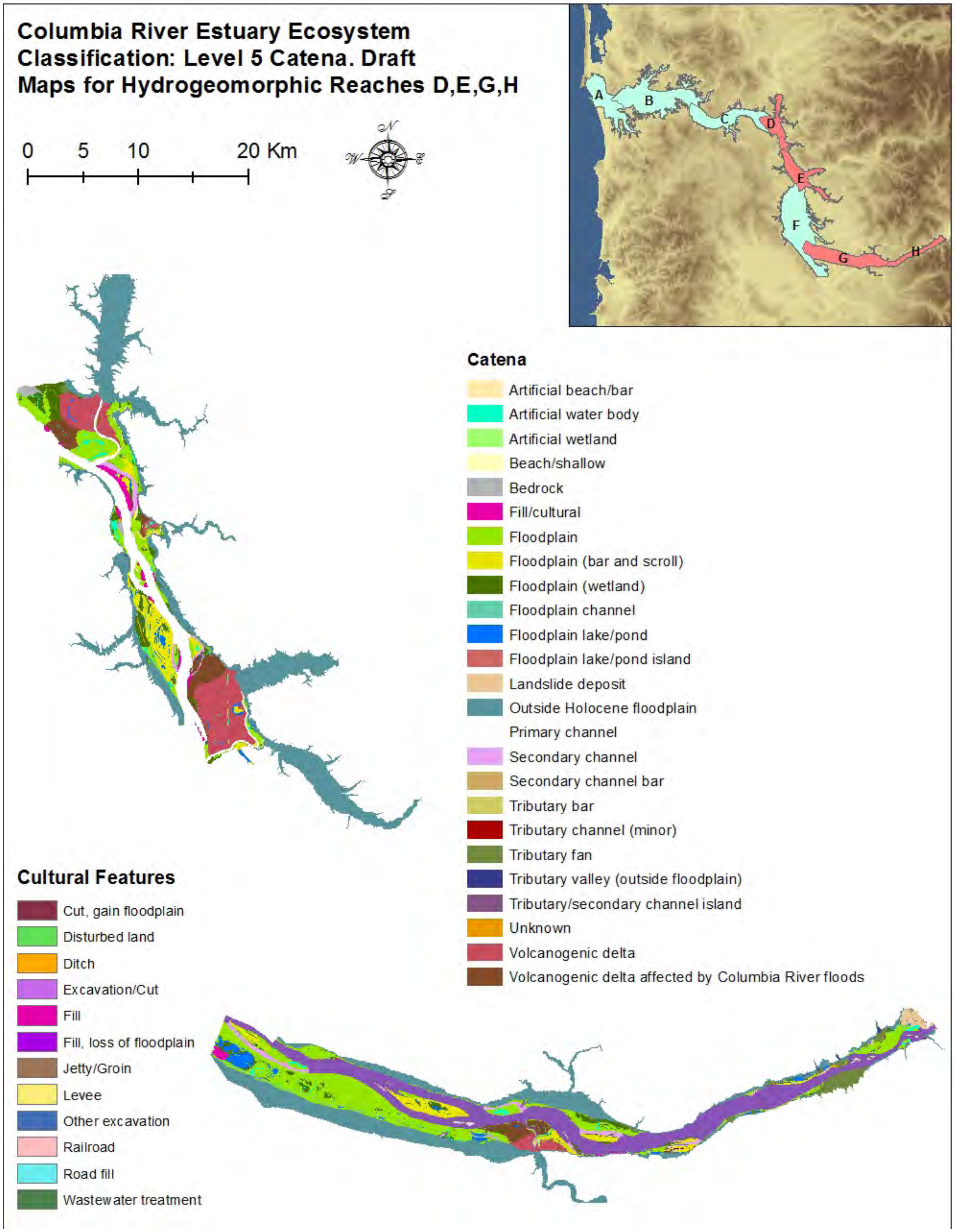


Figure 3. Classification Level 5 (Geomorphic Catena) illustrated for Hydrogeomorphic Reaches D,E,G,H

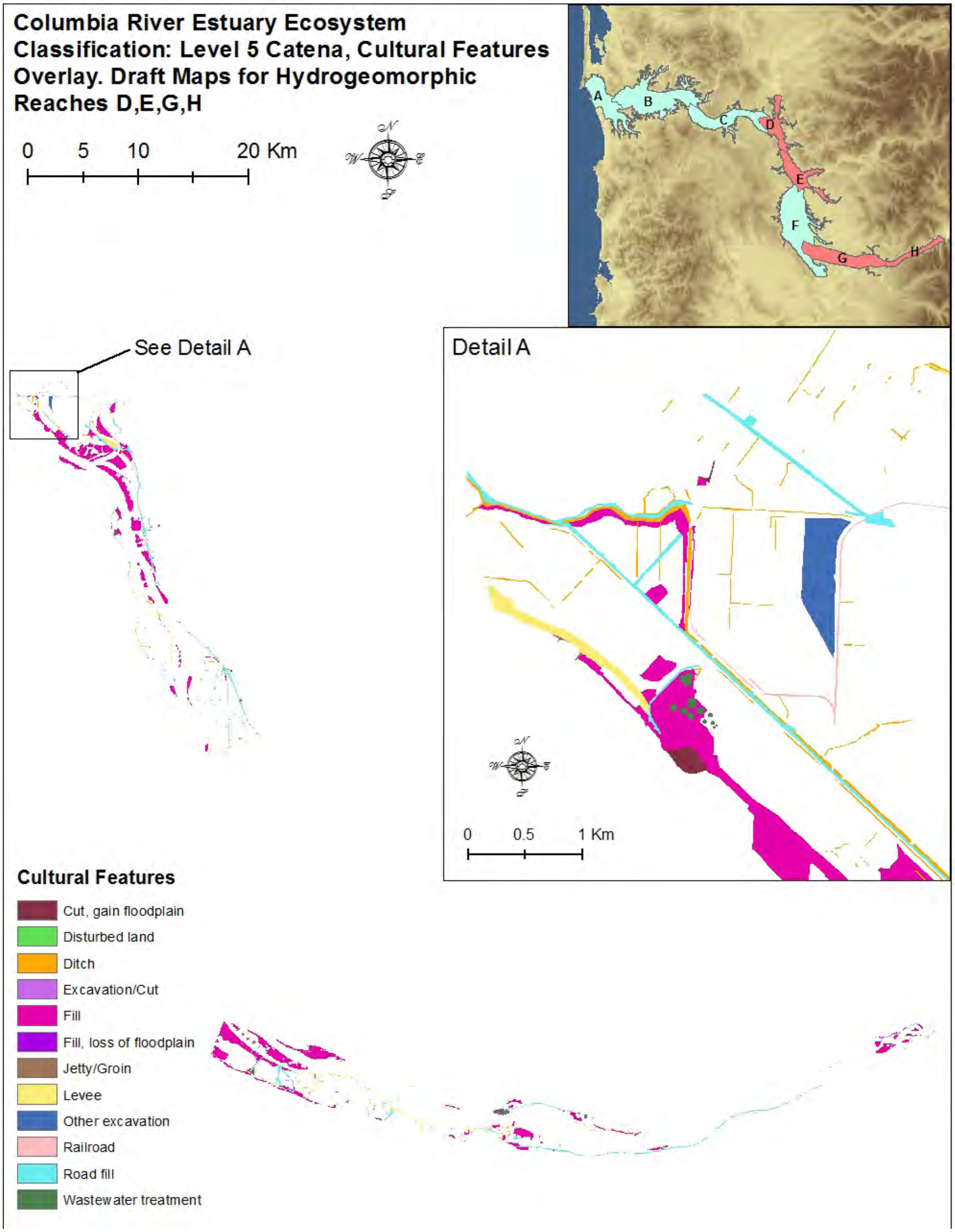


Figure 4. Classification Level 5, Cultural Features (Geomorphic Catena) illustrated for Hydrogeomorphic Reaches D,E,G,H

5.4 Classification Work Efforts Planned for 2010-2011

Classification activities planned for the project period September 1, 2010 – August 31, 2011 are as follows: 1) Refinements to draft Level 4 Complexes and Level 5 Catena for Reaches D-H, based on review of draft versions. 2) Delivery of land cover data (Level 6), expected in early 2011. 3) Further refinement of Level 5 Catena through incorporation of land cover information. 4) Completion of final Level 4 Aquatic Complexes for the entire estuary. 4) Draft and refinement of Level 4 Complexes and Level 5 Catena for Reaches A-C. 5) Merging of draft terrestrial and aquatic Complexes/Catena for the entire LCRE. In addition, final edits to the Classification, based on peer review results, are expected to be completed in 2011 (Simenstad et al., In review).

6.0 Datasets Needed to Complete the Classification

Completion of the Classification Levels 1 to 6 for the entire LCRE requires up-to-date bathymetry and landcover data. Bathymetry supports delineation of Levels 4 (Ecosystem Complexes) and 5 (Geomorphic Catena). Landcover supports delineation of Levels 5 (Geomorphic Catena) and 6 (Primary Cover Class). Thus, in 2009 and continuing in 2010-2011, the Estuary Partnership has been coordinating efforts to fill these data gaps and provide these datasets to UW and USGS to facilitate Classification completion.

6.1 Bathymetry

In October 2007, the Estuary Partnership, UW, and USGS convened a workshop to discuss bathymetry gaps in the LCRE. At this workshop, resource managers prioritized areas of bathymetric data gaps for collection (Figure 5, Figure 6). Following the workshop, the Estuary Partnership and UW, developed a strategy for data bathymetry collection based on the gap priority rankings. This strategy is needed because bathymetry collection in the LCRE has historically been implemented for navigation purposes and shipping channel maintenance, leaving many data gaps distributed throughout the LCRE. In addition to its use in the Classification, bathymetry can inform site selection for monitoring and restoration efforts in tidally influenced emergent wetlands, See Jones et al. (2008) for additional information on the bathymetry workshop, collection strategy, and data gap characteristics.

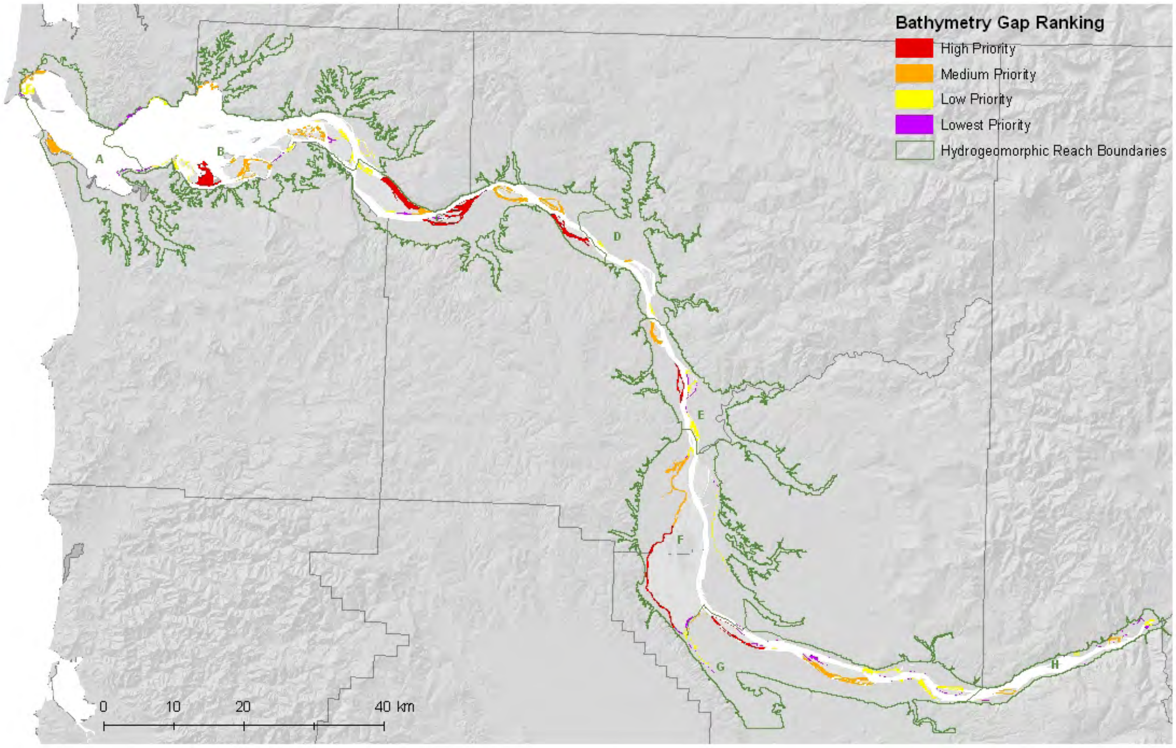


Figure 5. Existing bathymetric gaps ranked by priority for data collection at 2007 workshop.

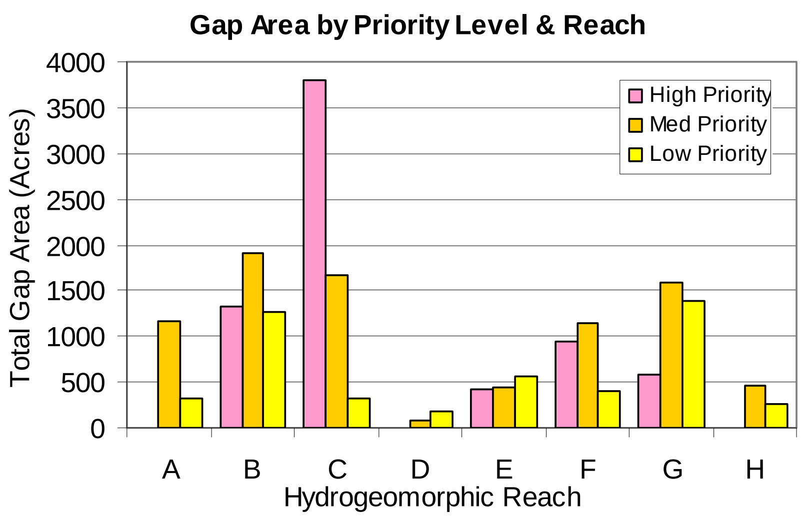


Figure 6. Bathymetry gap area by reach and priority, as identified for 2007 workshop.

In February 2010, the Estuary Partnership contracting with David Evans & Associates Marine Services (DEA) continued its bathymetric data collection effort which began in 2009 (see year 5 report for additional details related to the data collection effort). During 23 days of surveying, a total of 7000 acres was covered, including all remaining medium and high priority gaps identified in 2007. As in 2009, data collection was scheduled around the highest possible water levels to maximize data collection and facilitate eventual integration with LiDAR data into a seamless dataset. DEA was again able to utilize jet skis to access extreme shallow water areas. Figure 7 shows the survey plan for 2010. A summary of data collection efforts for 2009 and 2010 (Years 5 & 6) is provided in Table 4 while Figure 8 shows the final extent of coverage for the two-year effort

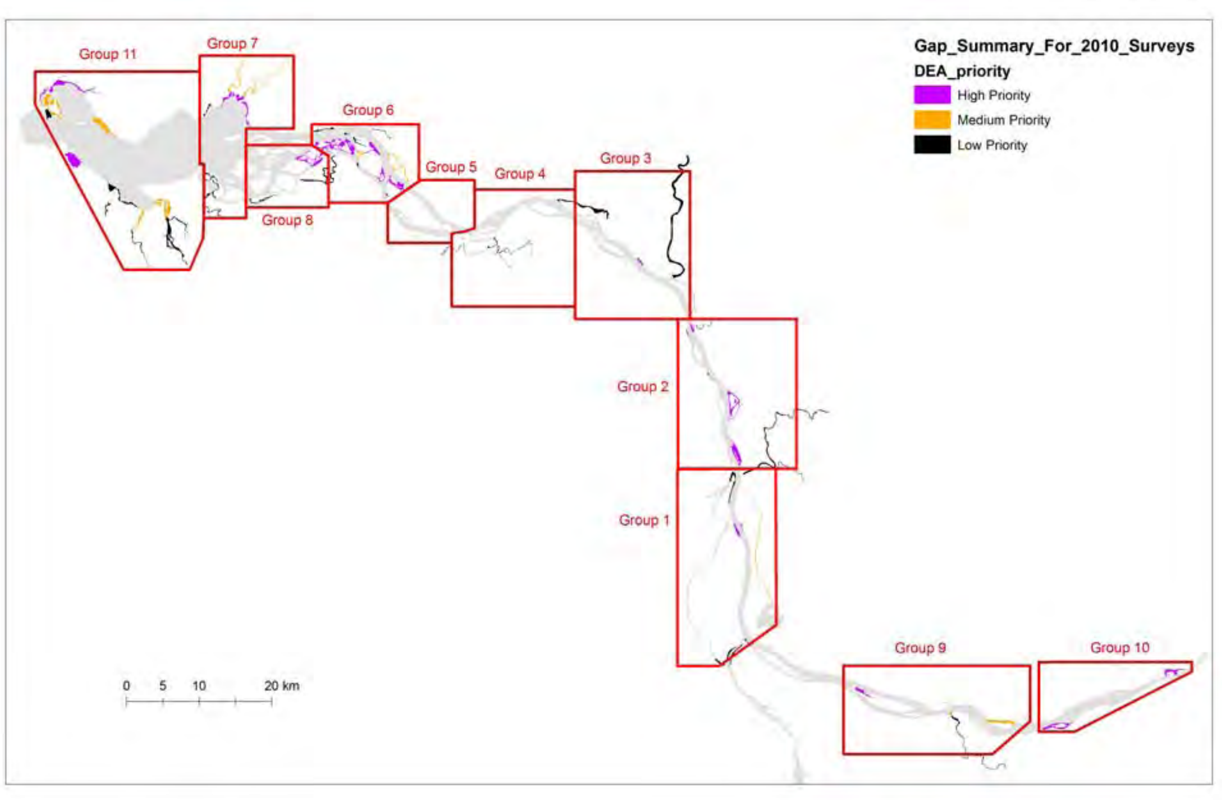


Figure 7. Bathymetry survey plan showing LCRE divided into 11 data collection groups. In 2010, gaps in Groups 6,10 & 11 were targeted, in addition to gaps in remaining groups which were missed in 2009.

Table 4. Summary of Year 5 and Year 6 bathymetric survey results

Contract Year	**Survey Days**	**Targeted Reaches**	**Targeted Acres**	**Acquired Acres**	Notes
5 (2008-2009)	30	C-G	11,830 of med/high priority gaps	10,595 (med/high priority) 2045 (lower priority)	
6 (2009-2010)	23	A,H and remaining gaps in C-G	3200 remaining med/high priority	3200 (med/high priority) 3790 (lower priority or newly identified)	Completed all med/high priority gaps as identified in 2007 workshop

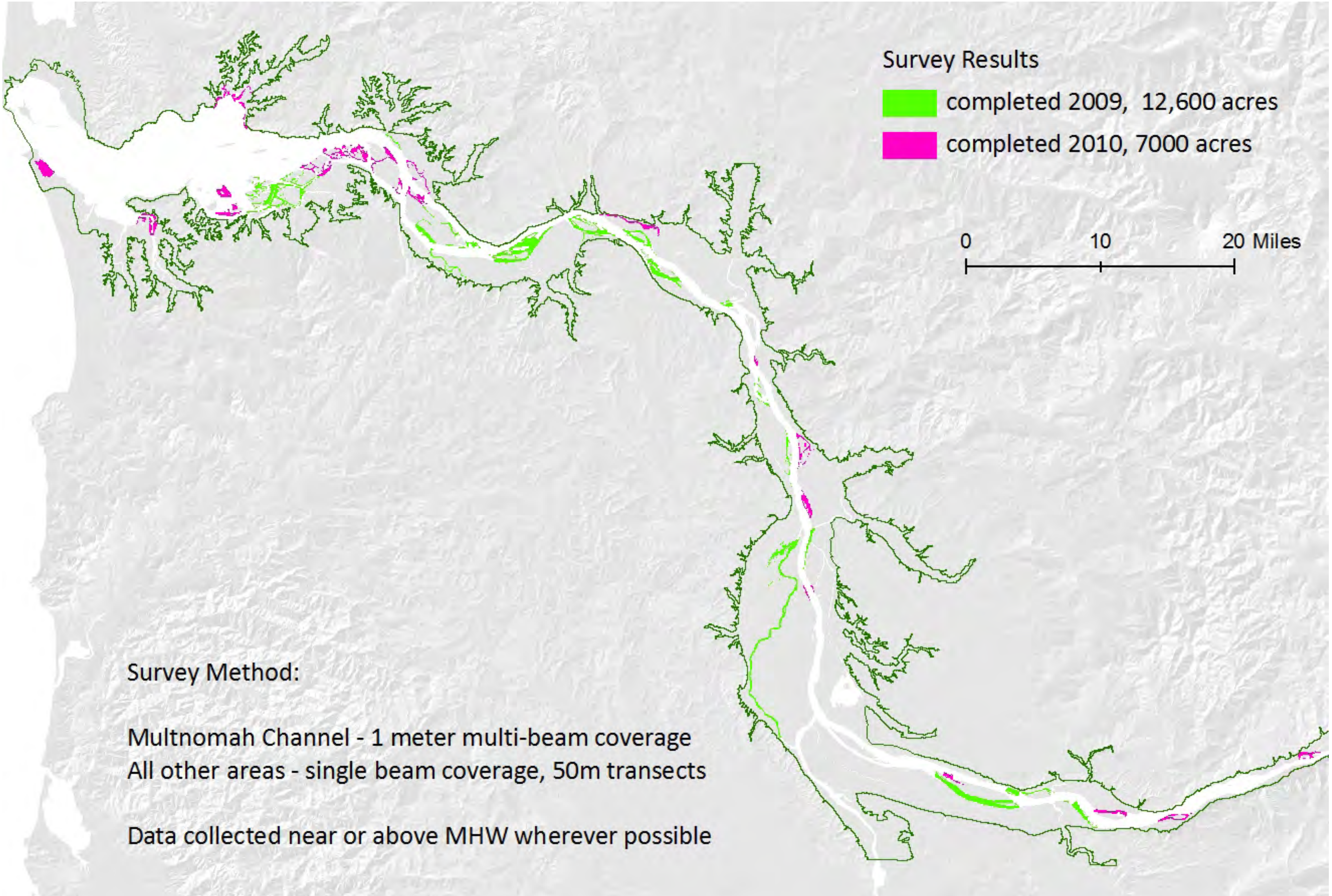


Figure 8. Map of LCRE showing bathymetry data collected in 2009 (green), and 2010 (pink)

The Estuary Partnership's bathymetric data collection efforts coincided with a related Corps of Engineers (COE) project to develop a seamless elevation model for the LCRE, using recently acquired bathymetric and topographic LIDAR data sets. This resulted in an efficient working relationship wherein the Estuary Partnership was able to contribute a significant amount of source data, which was then processed and incorporated by the COE into its Terrain model. This new product represents the most up to date, comprehensive, and highest resolution elevation data set that has been generated for the LCRE to date. The two groups will work together moving forward to keep the dataset current through the addition/replacement of data as more becomes available.

The Terrain model has been provided to both USGS and UW, who now have the elevation data necessary for completion of the Classification.

6.2 Land Cover

Specifically, landcover assists in the delineation of Levels 5 (Geomorphic Catena) and serves as a standalone Layer 6 (Primary Cover Class). The existing 2000 LANDSAT classification is nearly 10 years old and is functionally limited with regard to the Classification. For instance, the 2000-landcover data does not differentiate well between tidal and non-tidal wetlands, uplands and wetlands, and forest classes like mixed, coniferous, and deciduous forests.

To address this data gap, the Estuary Partnership convened a landcover workshop in May 2009 to investigate options for acquiring a more-recent landcover dataset. Details regarding the workshop results, and ensuing RFP process are provided in the BPA annual report for Year 5. In December of 2010, after a rigorous evaluation process, the Estuary Partnership contracted with Sanborn Mapping Company to generate the new land cover dataset. Sanborn has done extensive work of similar nature for the NOAA Coastal Change Analysis Program (C-CAP), and has developed an innovative, polygon based approach to high resolution land cover mapping that offers some advantages relative to more traditional techniques. Their work was highly regarded in conversations with C-CAP personnel, who have a mutual interest in mapping of this region and have been collaborating on the project. Sanborn subcontracted with SWCA, a local consulting firm, to complete the field sampling portion of this project, while they themselves are handling the image classification tasks.

Table 4 shows the schedule for the land cover effort which was developed during the RFP process. Due to delays in executing our contract with BPA, the schedule was shifted backward by approximately 1 month, with the last 2 tasks (completion of 2nd round of on the ground data collection, and delivery of QA/QC field data and final sampling report) being moved to Phase 2 (2011). Sanborn was able to meet this revised schedule, and successfully completed the Phase 1 work on time. Deliverables for 2010 included the following: 1) Memorandum on ancillary data reviews. 2) Field Data Sampling Design. 3) Field data database schema. 4) Review of supporting datasets for classification. 5) Delivery of QA/QC sampling data from the first field campaign (leaf off effort in late March/early April). 6) Summary report for Phase 1 work completed. In preparation for the 2nd field campaign, Sanborn was able to generate a draft map product, which could be taken into the field for verification purposes. Figure 9 provides a snapshot of that map, to illustrate what is expected from the final product. The Classification scheme that was chosen closely resembles that used for the Estuary Partnership's previous LandSAT TM2000 land cover product.

In 2011, the Estuary Partnership will renew their contract with Sanborn to complete Phase 2 of the Land cover data collection effort. To date, the project is on schedule, with delivery of the final product land cover map product expected in late January, 2011. This will be delivered to UW and USGS, and will fill the last critical data gap necessary for completing the Classification.

Table 5. Outline of landcover effort based on a January 1, 2010 start date. Schedule may change based

on contracting with identified vendor and execution of LCREP-BPA contract.

Approx. Due Date	Description
Phase 1 – Sampling methodology and collection of training and ground truth data	
Jan 1, 2010	Start Phase I
Jan 15, 2009	Complete review of existing classification and available training data sources
Feb 20, 2009	Complete sampling designs for newly acquired training and ground truth data
Mar 1, 2010	Complete database schema for training and ground truth data
Mar-Apr 2010	Complete 1st round of on the ground data collection (leaf-off)
Apr 1, 2010	Complete review of supporting datasets for classification
Jun-Aug 2010	Complete 2nd round of on the ground data collection (leaf-on)
Aug 31, 2010	Deliver report and database of QA/QC field data
Phase 2 – Classification of RS imagery based on training data	
Sep 1, 2010	Start Phase 2
Oct 1, 2010	Complete selection of imagery for landcover classification
Jan 10, 2011	Complete classification and accuracy assessment for cover classes
Jan 31, 2011	Deliver final report
Aug 31, 2011	Deliver draft of peer reviewed publication on work effort and analysis

Figure 9. Draft land cover sample from map generated for the 2nd field effort.

7.0 Characterization of Emergent Wetlands in the LCRE

The on-going objective of the Ecosystem Monitoring Project is to characterize tidal freshwater habitats and monitor salmon occurrence and health in those habitats in the LCRE. Based on funding levels, the EMP has largely concentrated on characterizing relatively undisturbed emergent wetlands and tidal forested wetlands that provide important rearing habitat for juvenile salmonids. Since 2007, we have co-located vegetation, fish, fish prey, and additional habitat monitoring sites as much as possible in emergent wetlands in order to have the same datasets for multiple sites throughout the LCRE.

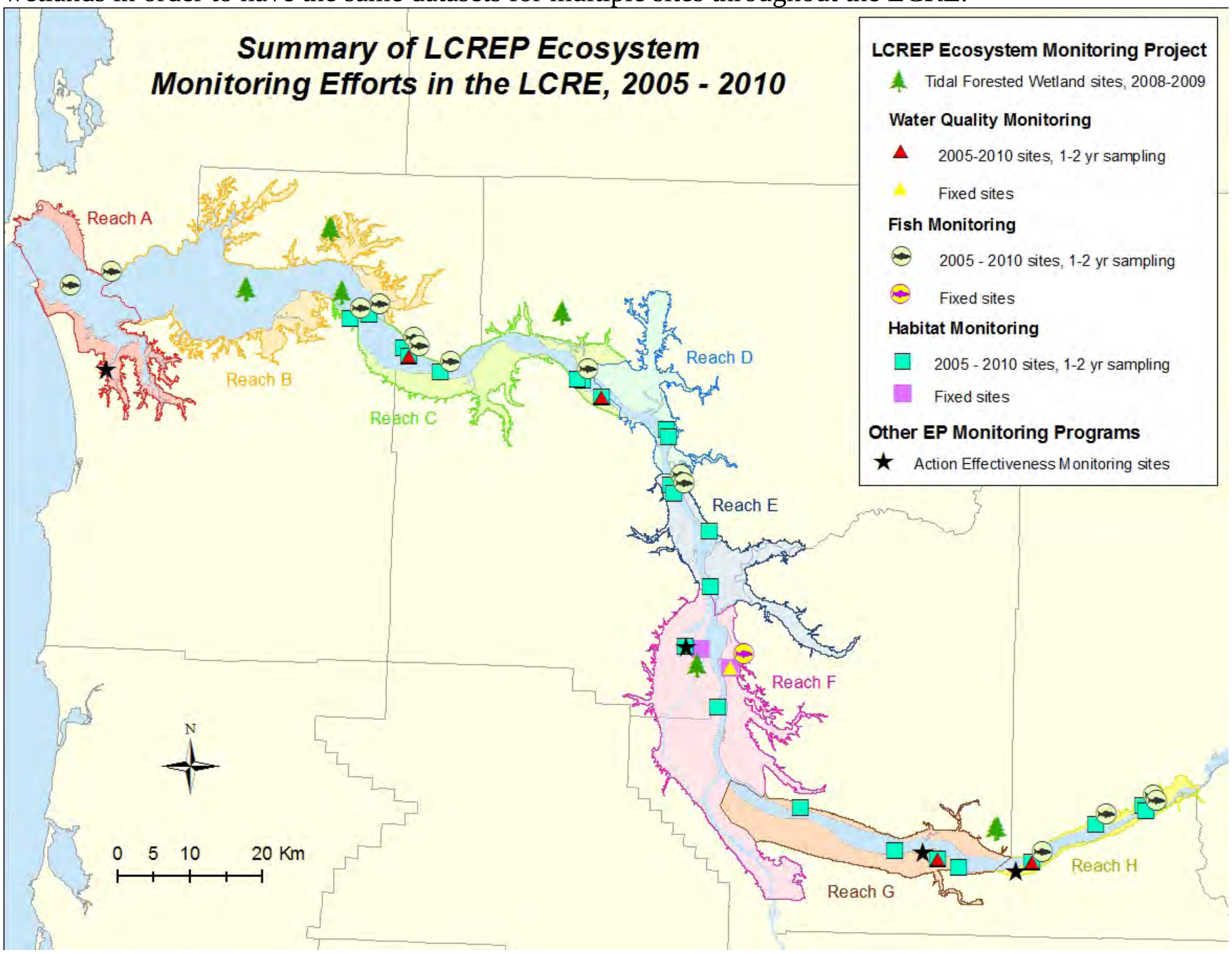


Figure 10 shows the locations of EMP sampling sites. Data collected at these sites support multiple RPAs in the 2008 Biological Opinion, provide reference site and salmonid genetic stock information for regional restoration programs, and contribute to our understanding of salmonid occurrence and habitat usage in the LCRE.

As of November 2010, the EMP has collected:

- One to two-year vegetation and habitat data at 24 emergent wetlands between 2005 and 2010 ("status sites;" denoted by blue squares)
- Multiple summers of vegetation and habitat data at 2 additional long-term emergent wetland sites between 2005 and 2010 ("year-to-year trend site;" denoted by purple squares)
- Salmon and prey data over one to two sampling seasons (approximately March/April – August) at 14 emergent wetlands between 2005 and 2010 ("status sites;" sites denoted by green/black fish)

- Multiple sampling seasons of salmon and prey data at 1 additional long term emergent wetland sites between 2005 and 2010 ("year-to-year trend sites;" denoted by yellow/purple fish)
- Basic water quality and depth over one sampling season (varies by year) at 4 emergent wetlands between 2006 and 2009 ("status sites;" denoted by red triangles)
- Basic water quality and depth over multiple sampling seasons at 1 additional long term emergent wetland site in 2006, 2008-2009 ("year-to-year trend site;" denoted by purple triangle)
- Community data at 6 forested wetlands from 2008-2009 (sites denoted by "trees")

Co-located datasets collected by the EMP include:

- Vegetation, habitat, salmon, and prey at 10 emergent wetlands between 2007 and 2010
- Vegetation, habitat, salmon, prey, and basic water quality parameters relevant to salmonids at a subset of 3 emergent wetlands between 2008 and 2009

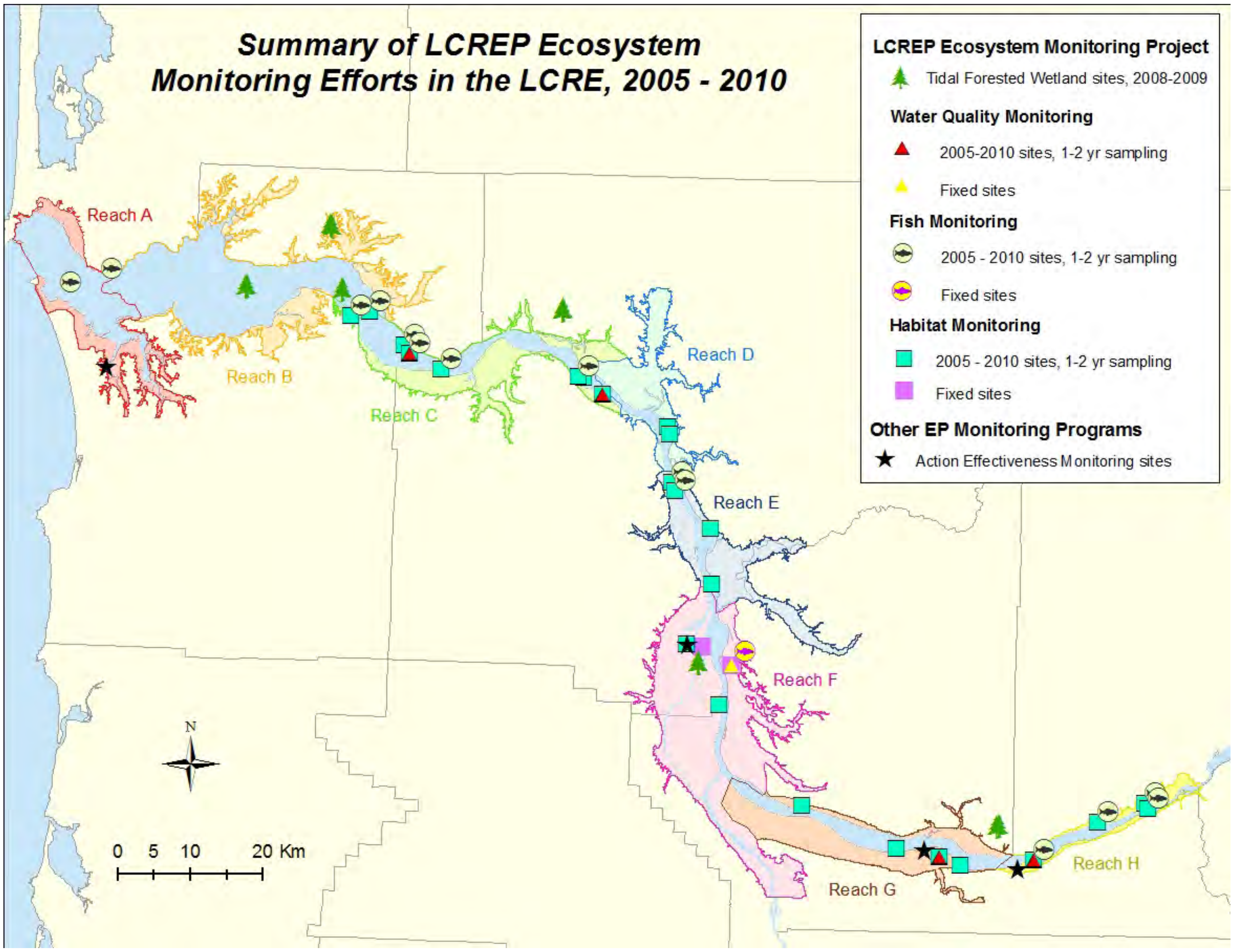


Figure 10. Map of EMP sites throughout the LCRE by year and monitoring type.

7.1 Sites

7.1.1 Selection

For the 2010 data collection efforts, the Estuary Partnership used the National Wetland Inventory (NWI, available at http://www.fws.gov/nwi/) for Reach C (Figure 11 to generate a list of potential sampling sites. This initial list was filtered using the following criteria applied in previous years to select the vegetation monitoring sites:

1. The site's wetland vegetation is classified as "emergent" in the NWI layer.
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-gate modifying flow regime present at the site).
4. The area of wetland is greater than 5 acres.

During this process, Ecosystem Monitoring Project's partners determined that a random sampling design was not appropriate for current monitoring efforts because:

1. Monitoring was focused on a specific habitat type (undisturbed emergent wetland) and reach.
2. A limited number of emergent wetlands occur on the landscape due to past land use activities.
3. Sampling was only possible at a limited number of sites due to reduced funding.
4. Data collected in 2010 should be consistent and comparable with data collected from 2006 to 2009.

In spring 2010, the Estuary Partnership, NOAA-Fisheries, PNNL, and USGS visited the potential sampling sites during a reconnaissance trip. In the end, the final habitat criteria used to select the 2010 monitoring sites were:

1. The site's wetland vegetation is classified as "emergent" in the NWI layer.
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-gate 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 disturbance 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.

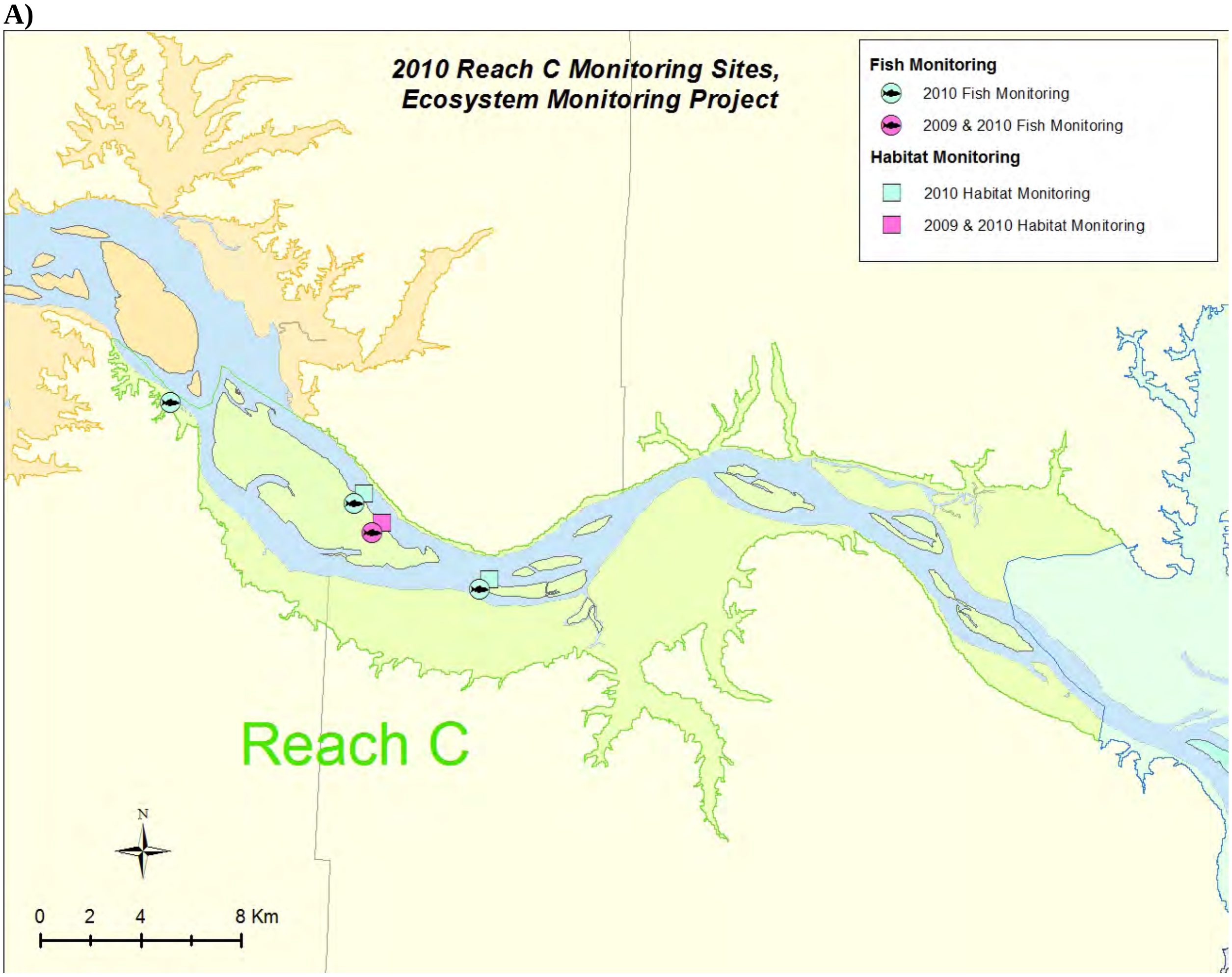
Additional logistical criteria included:

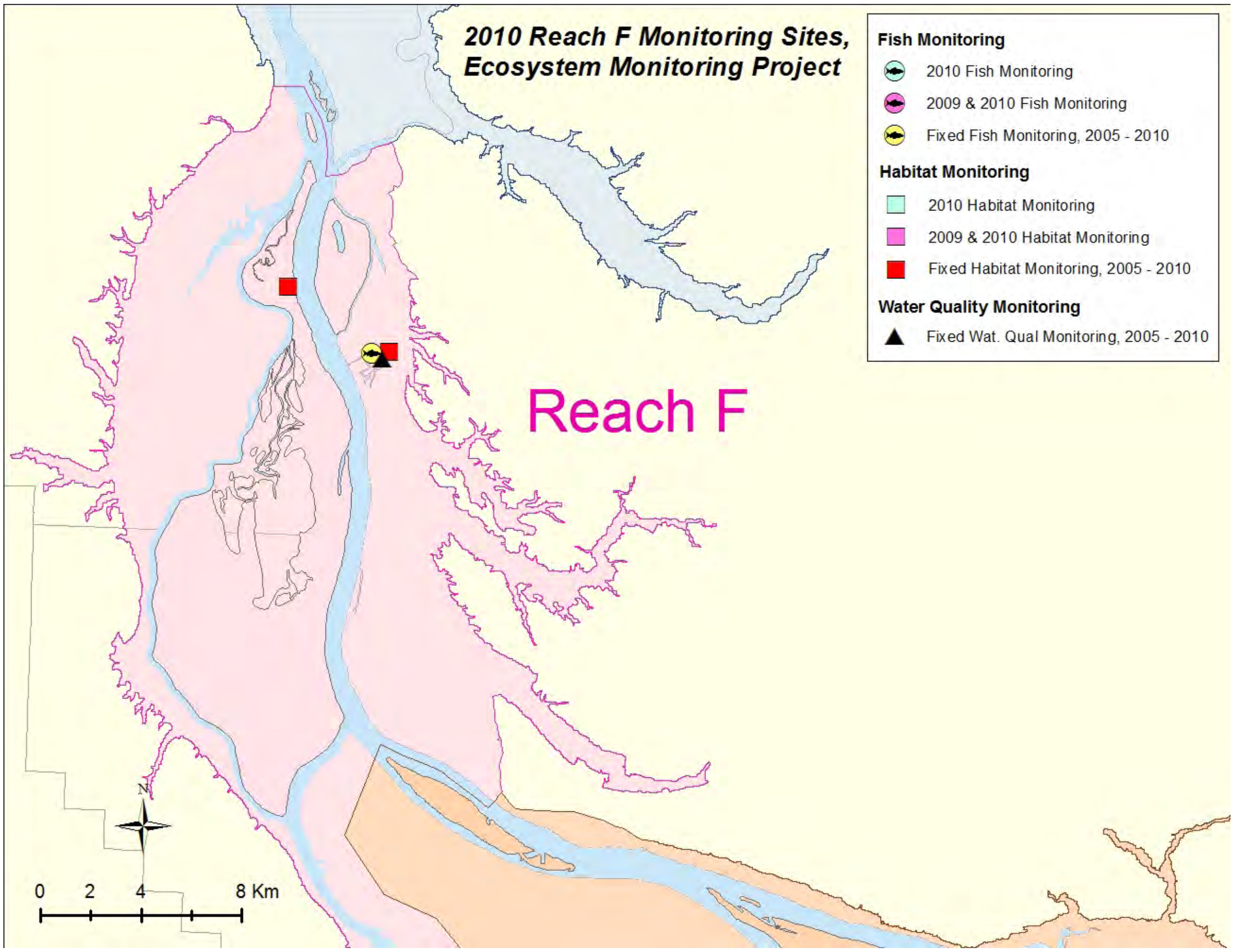
1. Stream channels are present at the site to facilitate the collection of cross-section and fish data.
2. The site is fishable by beach seine or similar gear-type.
3. The site is accessible for sampling purposes and with landowner permission.

The final criteria for 2010-site selection were selected based on funding levels, the desire for data comparability with previously collected data, and reasons outlined above. This strategy focused the monitoring effort and facilitated the collection of data comparable with previous efforts. This strategy, however, does not meet the original goal of the monitoring submitted for the FY 2007-2009, because current monitoring can only focus on 1 habitat type (undisturbed emergent wetlands) and not multiple habitat strata with current funding levels. At this time, data collected by the EMP will not support an assessment of ecosystem condition nor overall wetland condition within individual reaches due to its limited scope. The strategy does not support the collection of data that represents variation within and between different wetland types across the entire reach (es) being sampled or at an estuary-wide scale. At this time, it is not feasible to collect data facilitating the extrapolation of sampling results to the reach scale and considerations of statistical issues like the optimal size of the sampling unit, sources of error, and measures of variation. Instead, data collected in 2010 characterize a subpopulation of Reach C's wetlands (undisturbed emergent wetland), which are likely important habitat for juvenile salmon. The remaining wetland types in Reach C may have less salmon and lower abundances of marsh vegetation and wider ranges in sediment particle size and other physical attributes. While the 2010 effort provides initial information useful for understanding habitat conditions and salmonid use of undisturbed emergent wetlands in Reach C, sampling at a larger number of sites and habitat types throughout the 8 reaches is necessary to extend results to the estuary at large, assess system-wide ecosystem "health," and obtain the adequate statistical power needed for such analyses.

In 2010, the EMP partners selected 3 sites in Reach C for status monitoring. Reach C status monitoring sites were Wallace Island, Jackson Island and Bradwood Landing (Table 6; Figure 11). Partners re-sampled 3 sites (Campbell Slough and Cunningham Lake in Reach F and Whites Island in Reach C) where data were previously collected (Table 6; Figure 11).

A)





B)

Figure 11. Maps showing 2010 sampling sites in: A) Reach C and B) Reaches F to H.

Table 6 . Summary of sampling effort by site and year(s) for sites where data were collected in 2010. **(Lord-Walker Island 2 was sampled by the EMP in conjunction with the Reference Site Study; thus, only vegetation and habitat data were collected at Lord-Walker 2.

Reach	Site	Vegetation & Habitat	Fish & Prey	Water Quality & Depth
C	Ryan Island	2009	2009	
	Lord-Walker Island 1	2009	2009	
	Lord-Walker Island 2**	2009		
	White Island	2009, 2010	2009, 2010	2009
	Jackson Island	2010	2010	
	Wallace Island	2010	2010	
	Bradwood Landing		2010	
F	Cunningham Lake	2005-2010	2007-2009	
	Campbell Slough	2005-2010	2007-2010	2008- 2010
H	Franz Lake	2008-2009	2008-2009	

7.1.2 Descriptions

Whites Island is located on the southern (upstream) end of Puget Island, near Cathlamet, Washington. A portion of the island is owned by Washington Department of Fish and Wildlife (WDFW) and is maintained as Columbia white-tailed deer habitat. Whites Island is not present on the historical maps from the 1880s (Figure 12). The monitoring site, located at the confluence of a large tidal channel and an extensive slough system (Figure 13), is approximately 0.7 km from the Cathlamet Channel. The site is characterized by primarily high marsh and a few willows, with numerous small tidal channels.

Jackson Island is located approximately 1 km downstream of Whites Island, adjacent to Puget Island. Ownership is unknown at this time. Jackson Island was also not present on the historic maps from the 1880s and has likely been created with dredge material over the years (Figure 12). The site we focused on is a very shallow-water slough with depths less than a meter (Figure 13) located approximately 0.3 km from Cathlamet Channel. The emergent vegetation, a mix of low marsh, high marsh, and reed canary grass, is located on both sides of the slough, grading up to willows, shrubs, and trees.

Wallace Island is upstream of Puget Island. While most of Wallace Island was present on the historic maps (from the 1880s), the portion of the island we monitored appears to have been created adjacent to the main island with a shallow channel/slough between Wallace Island and the monitoring area (Figure 12). The monitoring site is along the north side of the channel and is characterized by a slight depressional area formed from a small tidal channel (Figure 13). Much of the vegetation in the depression was flattened, likely due to recent high tides prior to monitoring.

Upstream of Reach C, the remaining two sites (Cunningham Lake and Campbell Slough) are in Reach F (Figure 13 D and E). These sites have been surveyed annually since the original 2005 monitoring. In the absence of a true rotational-panel sampling design, these two sites have been included with each annual survey to better understand inter-annual variability in vegetation patterns. Cunningham Lake is located on Sauvie Island in the Oregon DFW Wildlife Area at the end of Cunningham Slough approximately 6.4 km from the mainstem of the Columbia River. The site is a fringing emergent marsh bordering the extremely shallow “lake” (Figure 13) that in some years is covered with wapato (*Sagittaria latifolia*). The second site, Campbell Slough, is located on the Ridgefield National Wildlife Refuge in Washington. The monitoring site is an emergent marsh adjacent to the slough approximately 1.4 km from the mainstem of the Columbia River. The site grades from wapato up to reed canary grass and is adjacent to fenced in pasture land. Extensive grazing occurred at the site in 2007 but has been recovering since then. In 2010 slight evidence of grazing was again observed.

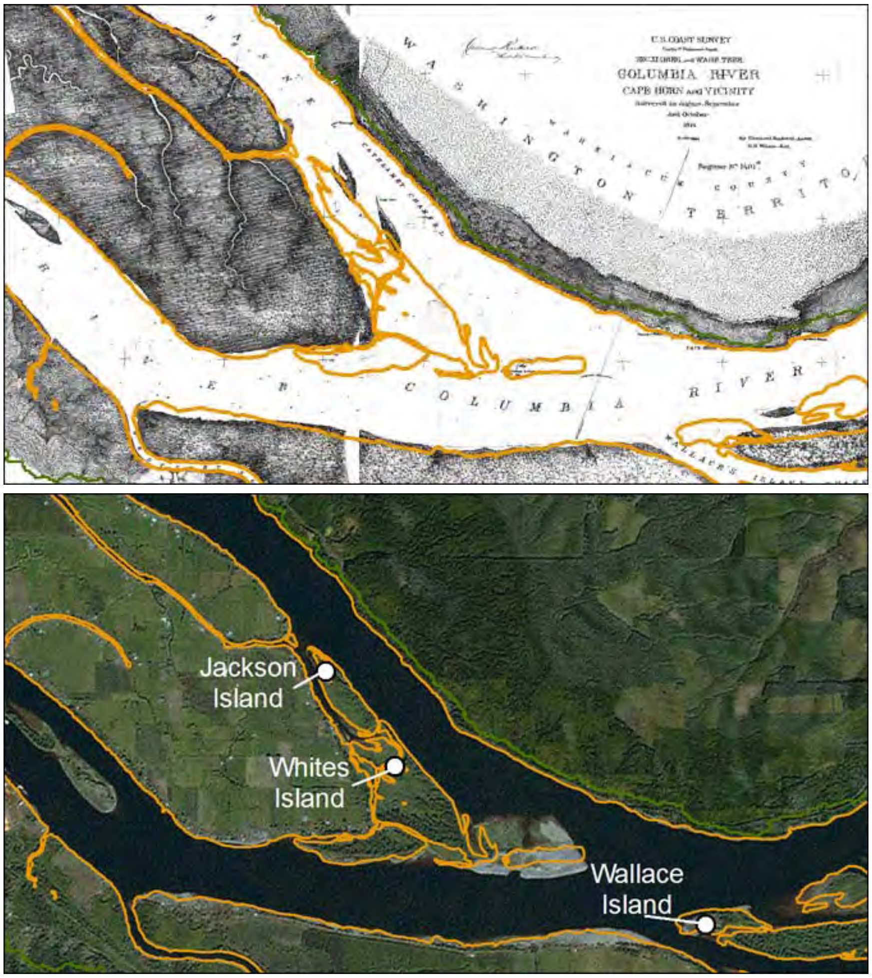


Figure 12. Historic (1880s) map of the Reach C monitoring area (above) and present day map (below). The most recent shoreline delineation is shown in orange on both maps.

A)  **B)** 

C)  **D)** 

E) 

Figure 13. Photos of Reach C sites: Reach C sites: (A) Whites Island, (B) Jackson Island, and (C) Wallace Island and Reach F sites: (D) Campbell Slough and (E) Cunningham Lake

7.2 Water Year

The water level fluctuations throughout the year, due to the variability in flows of the Columbia River, can affect the vegetation communities at the monitoring sites. One means of characterizing the variability is to evaluate the outflow at Bonneville Dam relative to the 10-year mean (Figure 14). This information, provided by the University of Washington data access in real time (DART) program, allows an evaluation of the magnitude and the timing of the spring freshet. In 2010, outflow was below the average during the spring, then above average for 4 weeks in June during the spring freshet. During this time outflow was above average by approximately 50-100 thousand cubic feet per second (kcfs). In comparison, in 2009 the freshet occurred in late April to late May (three weeks total) and was generally 50 kcfs above the 10-year average. In 2008 the spring freshet flows were considerably above average by approximately 100 cfs for eight weeks from mid May to mid-July.

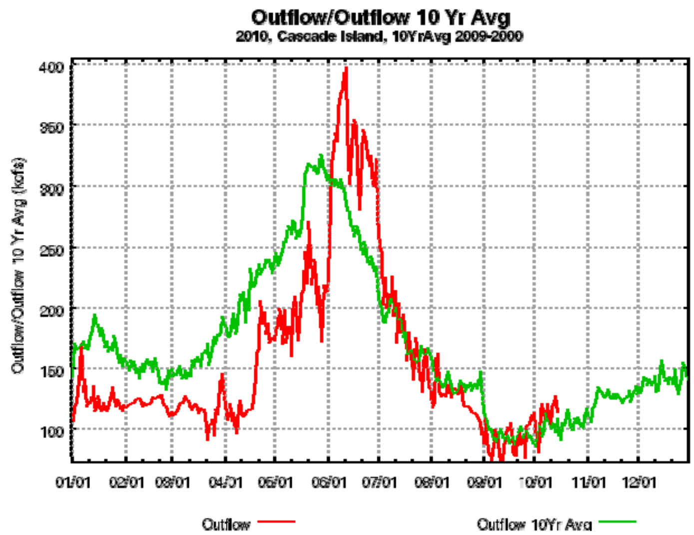


Figure 14.Outflow at Bonneville Dam, comparing outflow in 2010 (red) to 10-year average (green). Data from Columbia River DART website **(http://www.cbr.washington.edu/dart/river.html**).

7.3 Vegetation and Habitat Monitoring

The goal of the program is to conduct emergent wetland monitoring aimed at characterizing salmonid habitats in the lower Columbia River and from previously understudied portions of the estuary from the mouth of the estuary to Bonneville Dam. This is an ecosystem based monitoring program focused on improving the survival of juvenile salmonids through the lower Columbia River and estuary. This project comprehensively assesses 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. The information will be used to guide management actions associated with species recovery, particularly as it relates to threatened and endangered salmonids. PNNL’s role in this multi-year study is to monitor the habitat structure (e.g., vegetation, topography, channel morphology, and sediment type) as well as hydrologic patterns.

In 2010, PNNL collected field data on vegetation and habitat conditions at three study sites in Reach C and two in Reach F (Figure 15). Two of the sites in Reach C are new sample sites added this year and the other three are previously monitored sites, which are monitored to evaluate interannual trends. The sites in Reach F have been monitored previously in 2005-2009 and the third Reach C site was monitored in 2009. To date, 24 sites have been sampled in this program and we anticipate sampling an additional 3 sites per year in future years, while re-sampling a set number of core sites. This report summarizes the 2010 field effort and provides the results for multi-year data analysis for three repeat sample sites.

In March 2009, a site selection field trip was made with PNNL, NOAA-Fisheries, USGS, and Estuary Partnership staff to evaluate potential sites in Reach C; six potential sites were chosen in 2009 for monitoring in 2009 and 2010. Water levels, which can be high in the spring, were low enough during the site visits to permit an estimate of the suitability for most areas explored. In selecting sites, the research team sought consistency with sites surveyed in Reaches D, E, F, G, and H during previous study years, meaning desired sites were relatively undisturbed shallow water wetlands mainstem fringing or off-channel with characteristic emergent marsh vegetation and typically fine sediments.

Vegetation monitoring occurred from July 13-16 and 26-27, 2010. A total of five sites were sampled, three in Reach C and two in Reach F (Figure 15). The sites within Reach C included Jackson Island, Whites Island, and Wallace Island. The Reach F sites are the long-term monitoring sites at Campbell Slough and Cunningham Lake. Maps of the sites are presented in Appendix B.

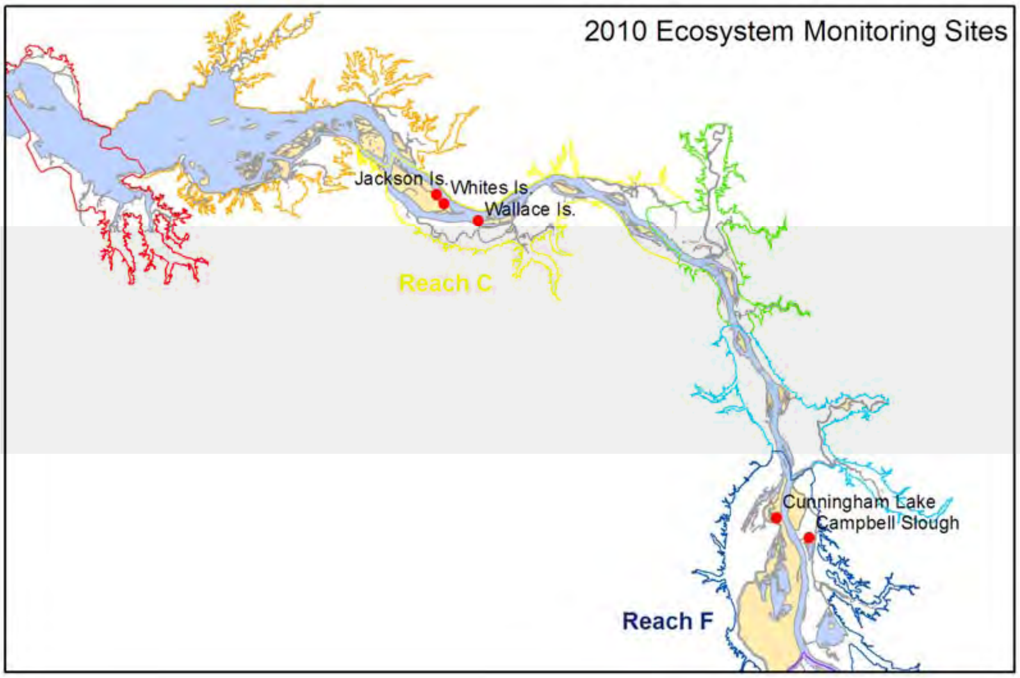


Figure 15. Map of Reach A to F, showing the location of the 2010 sampling sites.

7.3.1 Methods

As in previous years (i.e., 2005-2009), we surveyed sites for elevation, determined percent cover of vegetation along transects, and mapped prominent vegetation communities within the marsh. Since 2009, we have also measured channel cross sections, installed sediment accretion stakes at all the sites, and collected sediment samples at new sites. A photo point was also designated at each site from which photographs were taken to document the 360-degree view. Methods generally follow the restoration monitoring protocols developed by Roegner et al. (2009) for the Lower Columbia River and Estuary.

Transect Surveys

Upon arrival at a given site, the optimum location of transects was established such that most major plant communities from the lower emergent vegetation edge to the upland area would be included in the survey. Two to seven transects were established at a site, depending on the diversity of vegetation (Tiner 1999). At repeated monitoring sites, we re-sampled the same transects as previous years. A species area curve calculated for Whites Island determined that the 2009 level of sampling may have not captured all the species at the site (Figure 16). Consequently, at Whites Island we added two additional transects and sampled at similar level at the other two Reach C sites. At all sites, transects were located to encompass the elevation gradient at the site from the channel up to high marsh or trees.

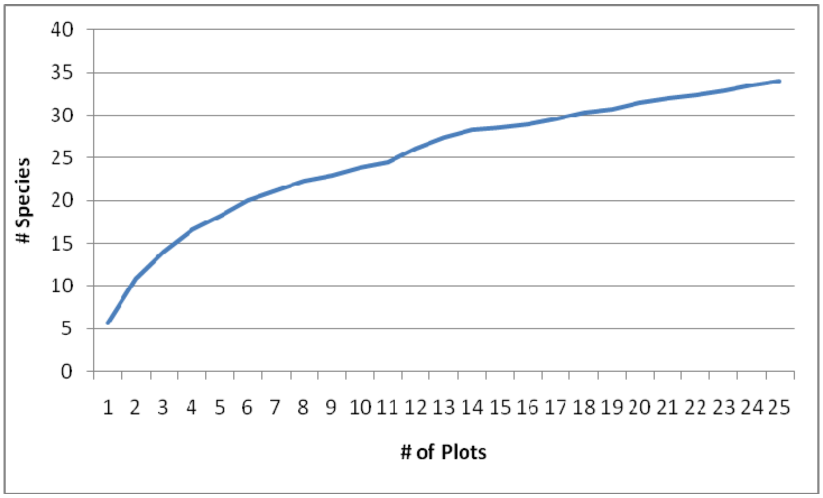


Figure 16. Species area curve for Whites Island from 2009.

Along each transect, vegetative percent cover was evaluated at 2-5 meter intervals. Interval length was based on the transect length and/or the vegetation homogeneity. At each interval on the transect tape, a 1-m2 quadrat was placed on the substrate and percent cover was estimated by observers in 5% increments. If two observers were collecting data then they would work together initially to ensure their observations were "calibrated." Species were recorded by their four letter code (1st two letters of genus and 1st two letters of species, with a number added if the code has already been used, eg. LYAM2 is *Lycopus americanus*). In addition to vegetative cover, features such as bare ground, open water, wood, and drift wrack were also recorded. When plant identification could not be determined in the field, a specimen was collected for identification using taxonomic keys or manuals at the laboratory. If an accurate identification was not resolved, the plant remained "unidentified" within the database. Where visibility through the water column allowed, the degree of submerged aquatic vegetation coverage was estimated to the extent possible by the observers.

Elevation at all sites was surveyed using a Trimble real time kinematic (RTK) global positioning system (GPS) with survey-grade accuracy. All surveying was referenced to the NAVD88 vertical datum; horizontal position was referenced to NAD83. Data collected from the base receiver were processed using the automated Online Positioning User Service (OPUS) provided by the National Geodetic Survey. OPUS provides a Root Mean Squared (RMS) value for each set of static data collected by the base receiver, which is an estimate of error. A local surveyed benchmark was located whenever possible and measured with the RTK to provide a comparison between the local benchmark and OPUS derived elevations.

Trimble Geomatics Office (TGO) was used to process the data. Each survey was imported and overviewed. Benchmark information was entered into TGO and rover antenna heights were corrected for disc sink (measured at each survey point to the nearest half inch) at each point. The survey was then recomputed within TGO and exported in a GIS shapefile format. Surveys were visually checked within TGO and GIS software for validity. Elevations were then converted from NAVD88 to the Columbia River Datum (CRD) based on conversions developed by the USACE (unpublished).

All initial data assessments were recorded on data sheets during site visits, and subsequently transferred into Microsoft Excel at the laboratory. Quality assurance checks were performed on 100% of the data entered. Elevations from the RTK survey were entered into the Excel spreadsheet to correspond to the appropriate transect and quadrat location.

In 2011, we will begin sampling above ground biomass of the emergent and submerged aquatic vegetation (SAV) as part of a larger study on primary productivity. In previous studies we have sampled above ground emergent vegetation (Johnson and Diefenderfer 2009), however we have not sampled SAV. As a pre-curser to the 2011 sampling effort, a pilot study at Campbell Slough was undertaken to compare two sampling methods for SAV sampling. A transect was placed across the Slough in line with one of the emergent vegetation sampling transects. Three 1-meter square sampling plots were placed at equal intervals apart using a random start point. Within the meter square plot a 0.1 m2 quadrat was placed in a randomly selected corner and all the SAV that was rooted in the quadrat was clipped at the sediment surface and placed in bags. In a different randomly selected corner of each 1-m square plot, a 35.6 cm thatching rake was placed on the sediment surface and turned 360 degrees, sampling the vegetation within a 0.1 m2 area. The rake broke the SAV stems at the sediment surface while the rake was twisting. The sample was then removed from the rake, any root material was removed, and the sample placed in bags. The six samples, collected at three plots, were kept cold and returned to the lab for drying and weighing.

Mapping

Using a Trimble GeoXT handheld GPS unit, a representative portion of each site (using reasonable natural boundaries) was mapped and major vegetation communities were delineated within the site. Additionally, features of importance to the field survey (e.g., transect start/end points, depth sensor location, and photo-point) were also mapped. All data were input to a GIS and maps of each site showing major communities and features were created.

Channel Metrics

In addition to the elevation surveys conducted along the vegetative transects, channel cross-sections were surveyed at sites containing channel networks. This metric lends itself to further understanding the relationship between cross-section dimensions, marsh size, and opportunity for fish access and is currently being developed for wetlands elsewhere in the Columbia River estuary. This effort will aid in understanding the channel dimensions necessary to maintaining a marsh ecosystem via restoration efforts in similar habitats. The primary objective associated with this data collection effort is to determine how unmodified channels may differ between reaches, as well as to document similarities within the region with regard to fish access. When possible, we collected five channel cross-sections from the mouth of the main marsh distributary channel to the headwaters of this channel. Intermediate cross-section surveys were done at the confluence of major secondary channels or equidistant along the channel, as appropriate.

Sediment

Sediment samples were collected within each major vegetation community strata at Jackson Island and Wallace Island. Unfortunately, the samples from Wallace Island were lost in a boating accident prior to shipment to the lab. Sediment samples were collected in 2008 at Campbell Slough and Cunningham Lake, and at Whites Island in 2009 and were therefore not recollected this year. Four 10 cm cores were collected within each strata and homogenized in a large metal bowl, placed in a clean plastic bag, and kept in a cooler until shipment to the analyzing lab. Samples were analyzed by Columbia Analytical Services in Kelso, Washington for total organic carbon (TOC) following the ASTM D4129-82M method and grain size following PSEP (1986) methods. Samples were analyzed within 28 days from the time of collection.

Simple sedimentation stakes were placed at each of the 2010 sites in 2009 to measure sediment accretion over a one year period. At each site, PVC stakes separated by one meter were driven into the sediment and leveled. The distance from the plane at the top of the stakes to the sediment surface is measured as accurately as possible every 10 cm along the one meter distance. The stakes are measured at deployment and again, one year later at recovery. The accretion or erosion rate is calculated by averaging the 10 measurements from each year and comparing the difference.

Hydrology

In 2009, pressure transducers (HOBO Water Level Data Loggers, Onset Computer Corporation) were deployed at a number of sites as a means of logging in situ water level data for one year. These sensors were placed at two of the Reach C sites that had been monitored in 2009 (Whites Island and Ryan Island) and three potential 2010 Reach C sites (Bradwood Slough, Jackson Island, and Wallace Island). In addition, a sensor was placed in the shallow channel at the Cunningham Lake site to get an indication of water levels at the site even though the sensor was expected to be exposed a portion of the time. These sensors were downloaded in 2010. The sensors at Campbell Slough and Whites Island were downloaded in 2010 and re-deployed for another year. In addition, a sensor was deployed at Franz Lake (a core site that will be monitored in 2011).

The data from the sensors was used to calculate inundation metrics from the marsh and channel elevations collected at those sites. The elevation data for Ryan Island were previously collected in 2009; the data for the other sites were collected in 2010. At Bradwood Slough, we were not permitted to access the site for monitoring, so we used a single elevation collected at the sediment accretion stakes as a representative elevation for the site. At Jackson Island, the t-post to which the sensor was attached was moved out of the slough during the year making the data un-usable. Fortunately, data comparability between the Jackson Island sensor and the Whites Island sensor (off by 3cm on average over the year) allowed use of the Whites Island data to calculate the inundation metrics at Jackson Island.

The percent of time each marsh was inundated was calculated for the entire year and for the growing season. The growing season, was based on the number of frost-free days for the region as determined by the Natural Resource Conservation Service (NRCS) in the wetland determination (WETS) table for Clark County, WA (NRCS 2002). The start of the growing season was determined to be April 12 and the end was October 12. The Clark County growing season is used for all the sites in the estuary so that the inundation calculations are standardized to one period. The frequency of inundation during the growing season was also limited to daylight hours (between 0900 and 1700).

In order to better assess hydrologic patterns and to make sites comparable over time and space, we needed a single measurement that would incorporate magnitude, timing, and duration of surface water flooding. Following work conducted in the US and Europe (Gowing et al., 2002; Simon et al., 2007; Araya et al., 2010) we calculated the sum exceedance value (SEV) using the following equation:

$$\mathrm{SEV} = \sum_{i=1}^{n} (d_{\mathrm{elev}})$$

where n is the number of days present in the growing season, d_{elev} is the daily average water level elevation above the average marsh elevation.

7.3.2 Results

Sediment Composition

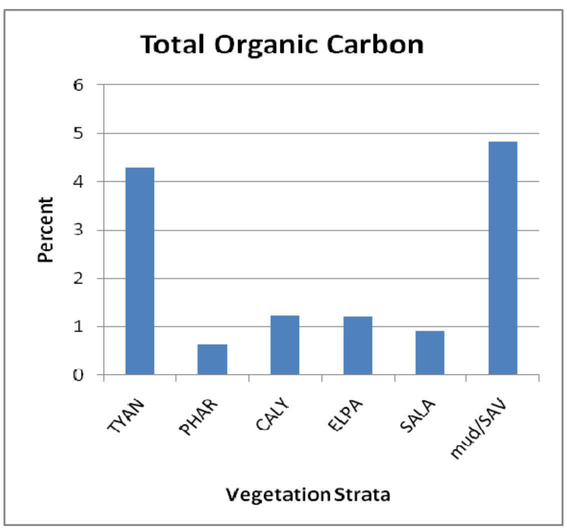
Sediment samples were only collected from Jackson Island in 2010 for reasons discussed in the Methods section. The vegetation strata at which the samples were collected in provided in Table 7. The samples from this site are similar to samples collected from this Reach in 2009 in that the range of the percent total organic carbon (TOC) is similar. However, the only percent TOC in 2009 greater than 2.6 was 4.7 percent from the high marsh at Ryan Island. This year the sample in TYAN (*Typha angustifolia*) in the high marsh had 4.3 percent TOC and likewise the bare mud/small SAV sample had 4.8 percent TOC (Table 7).

Overall, the TOC in the sediment samples is indicative of mineral soil, with organic soils generally having TOC greater than 12 percent (Mitsch and Gosselink 2000). Peat is not common in the soils at these sites, which makes them unusual in comparison with other wetland sites. We are curious why this is the case in the CRE, and will delve into an explanation when we compare these sites with others in the system.

Table 7. Vegetation strata associated with sediment samples at the 2009 monitoring sites in Reach C.

Site	Sample	Vegetation Strata
	TYAN	*Typha angustifolia*
Jackson	PHAR	*Phalaris arundinacea*
Island	CAREX	*Carex lyngbyei*
	ELPA	*Eleocharis palustris*
	SALA	*Sagittaria latifolia*
	SM.SAV	Mud/small submerged aquatic vegetation (SAV)

The samples from Jackson Island are also similar in grain size content to the samples collected in 2009 from Whites Island and Ryan, with silt the dominant component mixed with clay and some smaller sand particles (Figure 17). The upper marsh samples from Jackson (TYAN, PHAR, and CALY) are more similar to the high marsh sample from Ryan Island, while the lower marsh and channel samples (ELPA, SALA, mud/SAV) is more like the channel sample from Whites Island. Jackson Island is a very shallow slough and appears to be a depositional area, perhaps explaining the high silt content in most of the samples.

a) 

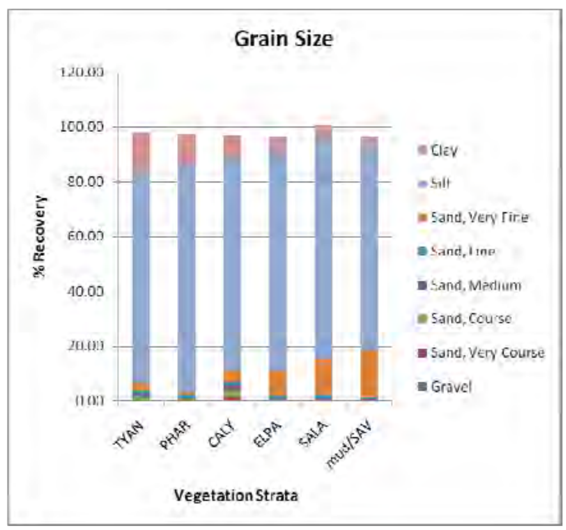
b) 

Figure 17. (a) Total organic carbon (TOC) and (b) grain size in the sediment samples from Jackson Island.

Accretion Rates

Sediment accretion stakes were installed at many of the study sites. The accretion or erosion rates varied from site to site, with the 8 of the 11 rates falling between 0.0 – 2.0 cm of accretion/year (Table 8). Outliers are Bradwood Slough and Sand Island sites, which measured erosion rates of 0.8 and 7.8 cm/year, respectively. Conversely, the Lord Island accretion rate was 3.6 cm/year. The reasons for the differences are not readily clear. One explanation for the erosion at Sand Island could be related to the timing of the initial measurement in 2008, immediately following a large flood event. This event could have deposited fine sediment at the site which subsequently was flushed from the site over the following year. This was not the case at the other Reach H sites, therefore the difference may be related to particular conditions at Sand Island (e.g., the large adjacent, unvegetated sand bluff).

Table 8. Sediment accretion/erosion rates for sites from 2008-2010

Reach	Site	Rkm	Elevation (m, CRD)	Year	Accretion/Erosion Rate (cm/year)
C	Ryan Island	61	1.89	09-10	0.2
	Bradwood Slough	62	1.59	09-10	-0.8
	Jackson Island	71	1.98	09-10	0.9
	Whites Island	72	1.96	09-10	1.0
	Wallace Island	77	1.36	09-10	1.3
	Lord Island	99	1.70	09-10	3.6
F	Cunningham Lake	145	1.45	09-10	1.9
	Campbell Slough	149	1.51	09-10	0.4
H	Sand Island	211	na	08-09	-7.8
	Franz Lake	221	1.84	08-09	0.5
	Pierce Island	228	na	08-09	1.6

na – not available

Vegetation Assemblage Structure

In general, species diversity is higher at Reach C sites than sites sampled in Reaches D-H. Within Reach C, the specific vegetation patterns differed somewhat among the 2010 sites (Table 9). Elevation and percent cover of species observed during 2010 sampling are shown in Figure 18. The elevations are relative to the Columbia River Datum (CRD), which alleviates elevation differences due to increasing elevation of the river bed. In general, the elevations of the emergent vegetation communities fall within a narrow range of 1-2 m relative to CRD, with the exception of Whites Island and Campbell Slough that extend just above this range. The upland border at all sites, which was not part of the sample area, was comprised of willows (*Salix* spp.), cottonwood (*Populus balsimifera*), and ash (*Fraxinus latifolia*). Maps of vegetation distributions at each site illustrate vegetation patterns and the spatial distribution of each major species communities relative to tidal channels at each site (Appendix B).

At the Reach C sites, reed-canary grass (*Phalaris arundinacea*; PHAR) was present ubiquitously, however was only dominant at Whites Island (48% of all vegetation cover). This site was also characterized by a diverse mix of eight other high marsh species (Figure 18). Due to the shallow gradient at Jackson Island, a higher portion of samples was in the SAV zone than sites with a steeper channel gradient. Consequently, seven SAV species were observed at this site. The emergent marsh was dominated by common spikerush (*Eleocharis palustris*; ELPA), Lyngby sedge (*Carex lyngbyei*; CALY), dominated lower elevations at Jackson Island, while the mid-elevations were comprised of a higher number and more diverse mix of wetland species. Narrowleaf cattail (*Typha angustifolia*; TYAN) was the most common species at Wallace Island followed by *P. arundinacea*. Most of the “overstory” cover (76%) was comprised of just four species, with Canada waterweed (*Elodea Canadensis*; ELCA) present in the “understory.” The latter was present to some degree at all the Reach C sites, and although it is a submerged aquatic species, it was often found in small depressions throughout the marsh.

Species composition at Campbell Slough and Cunningham Lake was similar to previous years (Table 10). Reed-canary grass (*Phalaris arundinacea*; PHAR), common spikerush (*Eleocharis palustris*; ELPA) and wapato (*Sagittaria latifolia*; SALA) were the most commonly occurring species at Campbell Slough and Cunningham Lake. However, there was a greater number of species at both sites in 2009 and 2010 compared to previous years. This could be explained by a variety of factors including the high water year in 2008, which not only caused a disturbance of sorts, but also could have brought in additional seed sources. The vegetation at the sites in 2008 was stunted and likely had lower species diversity during the July sample period due to the recent high water levels in that year. Additionally, Campbell Slough could be recovering from the disturbance of cattle grazing in previous years. Evidence of cow grazing was again noted at the site in 2010. The 15% litter cover was primarily dead *P. arundinacea* that may have been a result of cow trampling and grazing or possibly, from the late-spring high water occurring after the grass was already well established.

Table 9. Species lists by code for 2010 Reach C sites and the 2009 list for Whites Island, number of species is provided at the bottom of the table (see Appendix A for species names).

Whites Island 2009	Whites Island 2010	Jackson Island	Wallace Island
AGEX	AGEX	ALTR	ALTR
ALTR	ALTR	BICE	BICE
BICE	BICE	CAHE	CAHE
CAHE	CAHE	CALY	CALY
CALY	CALY	CAPA	COPA
ELAC	CAPA	CEDE	ELCA
ELCA	ELCA	ELAC	ELPA
ELPA	ELPA	ELCA	EPCI
EPCI	EPCI	ELPA	GATR
EQFL	EQFL	EPCI	GREB
GATR	GATR	GLGR	JUOX
GLEL	GATR3	GREB	LEOR
GLGR	GLGR	IRPS	LIAQ
IMSP	IMSP	JUOX	LIOC
IRPS	IRPS	LIAQ	LOCO
JUEF	JUOX	LIOC	LUPA
JUOX	LIAQ	MEAR	LYAM2
LIOC	LIOC	MIGU	LYSA
LOCO	LOCO	MYSC	MIGU
LUPA	LYSA	MYSI	MYSC
MIGU	MIGU	OESA	OESA
MYSC	MYSC	PHAR	PHAR
MYSP2	MYSP2	POCR	POCR
OESA	OESA	POPE	POHY
PHAR	PHAR	PORI	RARE
POAN	POCR	POHY	SALA
POHY	POPE	POZO	SCAM
POPE	POHY	SALA	SCTA
POZO	POZO	SCAM	SISU
SALA	RUMA	SCTA	TYAN
SISU	SALA	SISU	
TYAN	SCAM	SYSU	
VEAM	SISU	VEAM	
	SODU		
	TYAN		
	VEAM		
33	36	33	30

Table 10. Species lists by code for Campbell Slough and Cunningham Lake over six sampling years (see Appendix A for species names).

Campbell Slough						Cunningham Lake					
2005	2006	2007	2008	2009	2010	2005	2006	2007	2008	2009	2010
ELPA	AMFR	CAHE	CAOB	CASP	AGST	ALTR	CAHE	ALTR	ELPA	CACO	CAHE
EQFL	CAHE	ELPA	ELCA	ELCA	ALTR	CAHE	ELPA	CAHE	ELPAR	ELAC	ELAC
JUAC	ELCA	EQSP	ELPA	ELPA	AMFR	ELPA	EQFL	ELPA	EPCI	ELPA	ELPA
LYNU	ELPA	IRPS	EQSP	EQSP	CAHE	ELPAR	IRPS	EQFL	EQSP	EQSP	EQFL
PHAR	EQFL	LIAQ	HEAU	HEAU	CASP	EQFL	PHAR	LUPA	IMNO	IMSP	GATR
PONA	FRLA	LOCO	LUPA	JUOX	ELCA	JUAC	POHY	PHAR	LUPA	IRPS	IRPS
SALA	LYNU	LUPA	LYNU	LEOR	ELPA	PHAR	PONA	POHY	PHAR	LEOR	LEOR
SALU	MEAR	LYNU	PHAR	LIOC	EQPA	POAM	SALA	POPE	PONA	LUPA	LUPA
VEAM	MYSP	PHAR	PLLA	LOCO	HEAU	POHY	SALU*	SALA	POPE	LYNU	LYNU
	PHAR	PLMA	POCR	LUPA	JUOX	PONA	SPAN	SALU*	SALA	MESP	PHAR
	POAM	POCR	POHY	LYNU	JUTE	SALA	VEAM	SCTA	SALU*	PHAR	PONA
	POBA	POPE	POPE	MEAR	LEOR	SALU*		SPAN	SCMI	POHY	POPE
	SALA	RARE	RUCR	MOSS	LOCO	SCTA			SPAN	PONA	POHY
	SASP	RUSP	RUSP	PHAR	LUPA	SPAN				POPE	RUMA
		SALA	SALA	PLMA	LYNU	VEAM				RUDI	SALA
		SASP	SASP	POCR	PHAR					SALA	SALU*
		VEAM	SPAN	POPE	PLLA					SALU*	SCTA
				POHY	POAM					SCTA	SPAN
				RARE	POCR					SPAN	VEAM
				RUCR	POHY						
				SALA	RARE						
				SASP	RUMA						
				SCAM	SALA						
					SCTA						
					VEAM						
9	14	17	17	23	25	15	11	12	13	19	19

Non-native species were prevalent at all the sites often accounting for greater than 50% of the cover (Figure 11). The highest non-native cover was observed at Wallace and Whites Island and the least at Jackson Island. This could be related to the lower elevation of the Jackson Island site.

Table 11. Species richness and areal cover of native and non-native species at the 2010 monitoring sites.

Site	# Native species	Native species cover	# Non-native species	Non-native species cover
Jackson Island	27	67.5	6	23.4
Whites Island	25	36.5	11	64.8
Wallace Island	22	37.7	8	67.8
Campbell Slough	16	49.9	9	43.7
Cunningham Lake	14	40.8	5	58.6

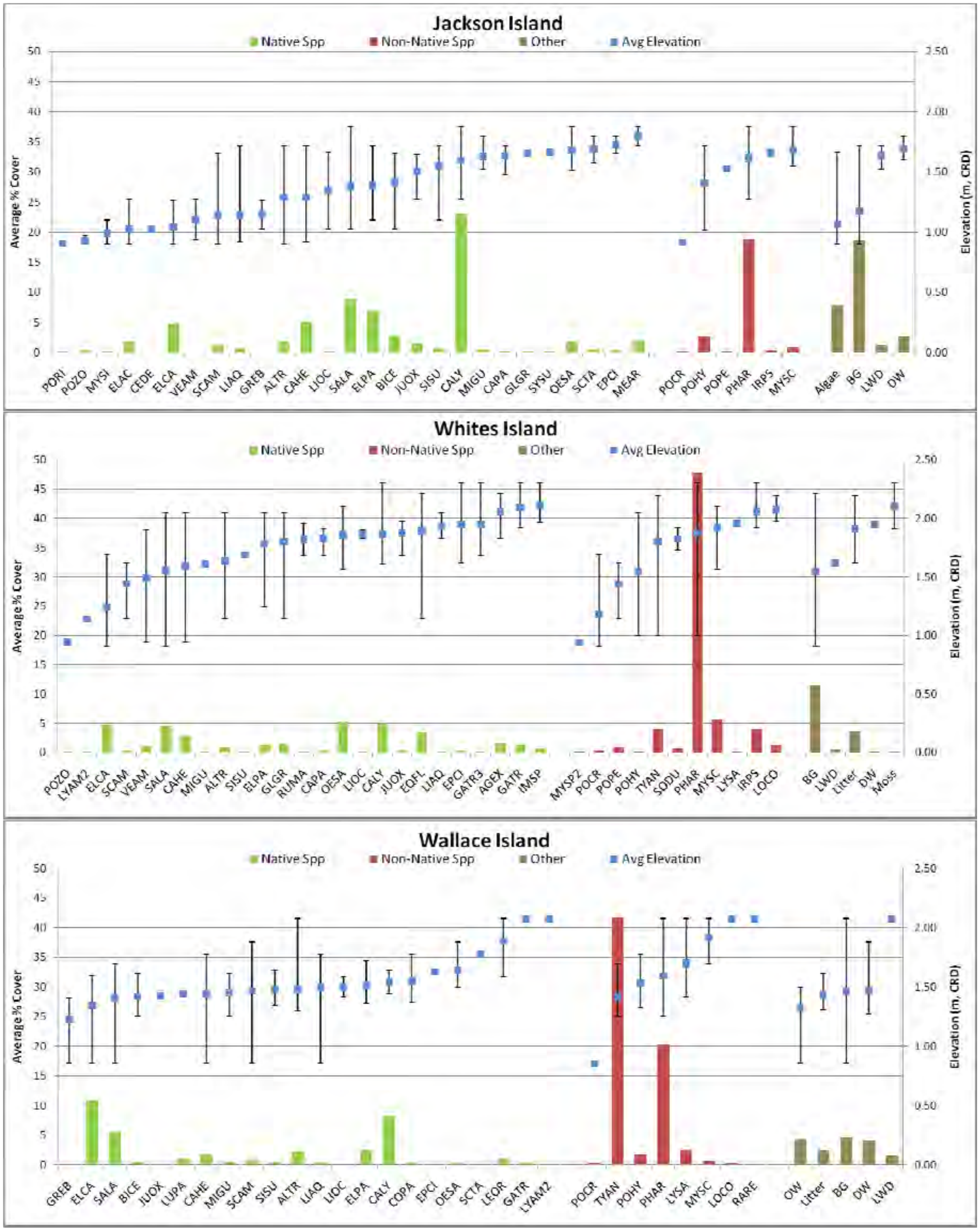


Figure 18. Vegetation species cover and elevations for sites sampled in 2010. Bars represent the minimum and maximum elevations at which the vegetative species occurred within the sample area (See Appendix A for species names associated with codes along the x-axis).

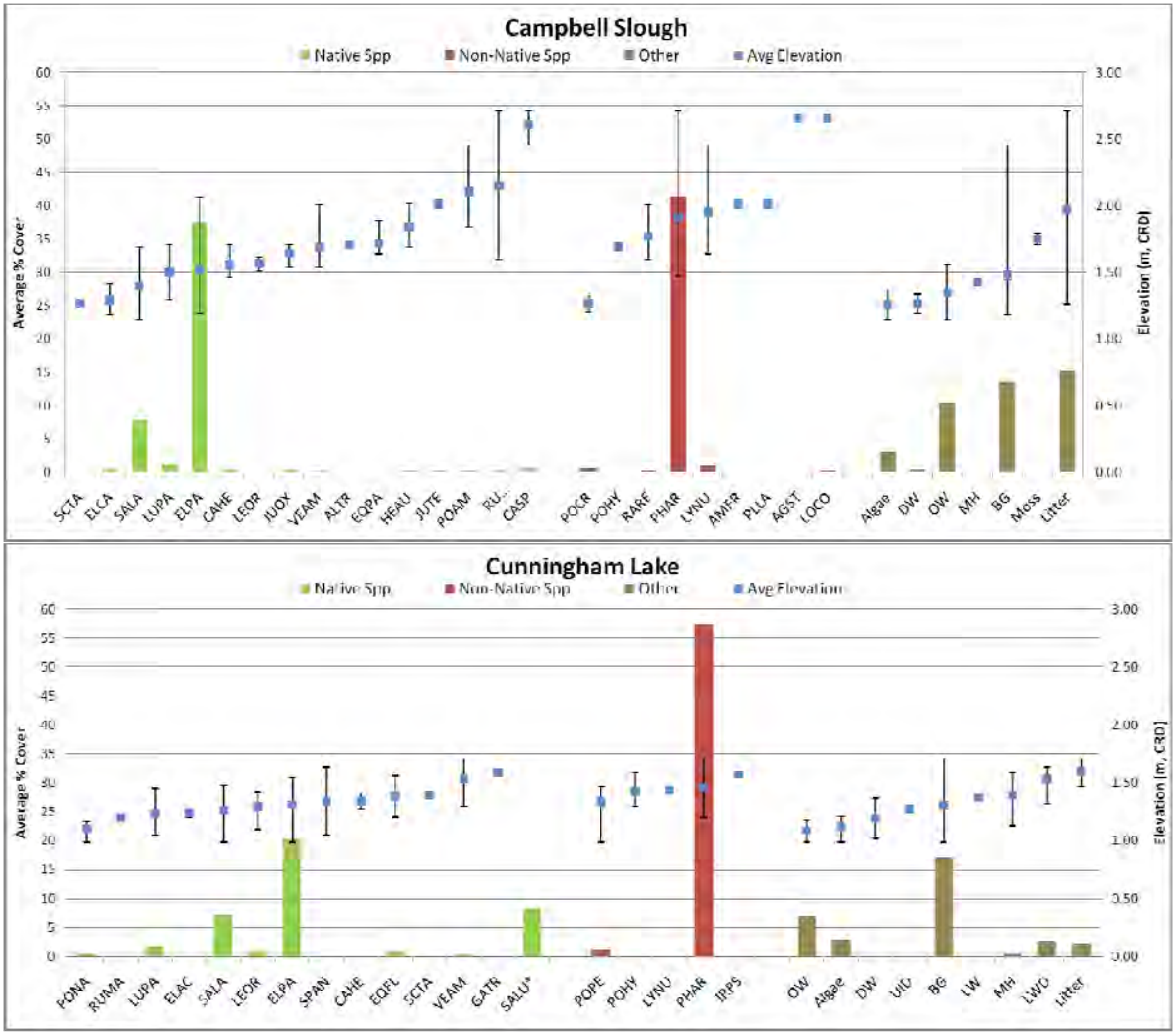


Figure 18. continued.

Submerged Aquatic Vegetation Biomass

We conducted a methods assessment study at Campbell Slough in 2010 to compare two biomass sampling methods, as discussed in the Methods section of this report. The two methods involved either clipping from within a 0.1 m2 quadrat or twisting a rake to remove the vegetation from a 0.1 m2 area. The former is perhaps a more exact method, however it is sometimes less feasible in some environmental conditions. The results, summarized in Table 12, indicate that while the overall dry weight of the samples collected using the rake method was higher, the difference between the sampling methods was not significant ($t=1.32$, $df=4$, $p>0.05$). The average difference between the samples was 2.33 g ± 0.53. This consistent difference between the samples from the different methods and the finding of non-significance implies that the rake method could be used as an alternative to the quadrat method. The higher dry weights from the rake method could be a result of additional root material gathered in this method. Future sampling and processing could include a step to remove any root material before drying to improve the accuracy of the method.

Table 12. Results of submerged aquatic vegetation biomass sampling at Campbell Slough

Sample ID	Quadrat (Dry Wt., g/0.1m2)	Rake (Dry Wt., g/0.1m2)
CSI-SAV-21.5	7.87	10.45
CSI-SAV-5.5	6.67	8.39
CSI-SAV-8.5	3.57	6.25
Average	6.04	8.36
Standard Deviation	2.22	2.10
Variance	4.92	4.41

Elevation, Inundation and Vegetation Assemblages

The elevation at which many of the species are found may be largely controlled by the frequency and duration of inundation at that elevation. In Table 13 we calculated the percent of time that the average elevation of the marsh was inundated during the deployment period (July 2009 to July 2010) and for the growing season (April 22 to Oct 12; 173 days during daylight hours). We did not evaluate spatial differences at the sites, but rather the frequency the water level was greater than the average elevation of the marsh plus 0.15 m and 1.0 m. For these calculations, the channel portions of the study site were not included in the averages (channel inundation is discussed in the Channel Section below). Sites from the 2009 monitoring effort are included in this analysis because the water level data was collected from 2009-2010. The percent of time the average marsh elevation is inundated during the growing season varied between 26 and 55 percent (Figure 19).

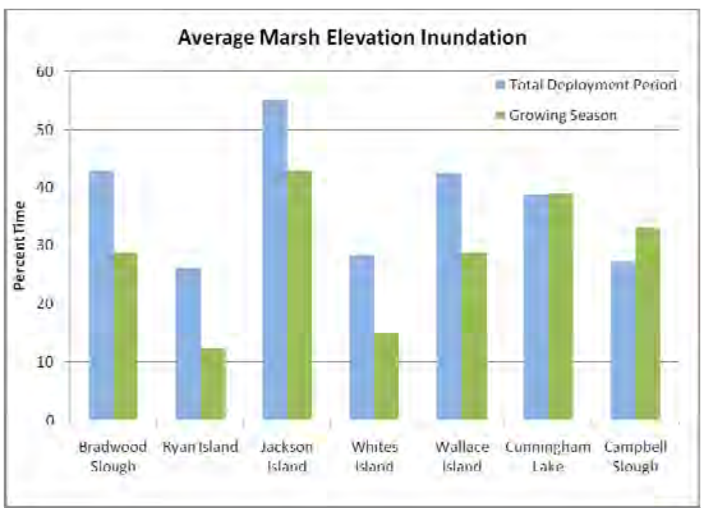


Figure 19. Percent of time the average marsh elevation was inundated at each site during the deployment period and during the growing season.

The difference between the Reach C and Reach F sites is evident by the difference in the amount of time the marsh has greater than 1 m of water over the average elevation. The Reach F sites were inundated with greater than 1 m water level 12 - 13 percent of time, whereas the Reach C sites were inundated to this level less than 1 percent of the time. The exception is Jackson Island, a slightly lower elevation site, which had the highest overall inundation time and was inundated 2.5 percent of the time with greater than 1 meter of water. Another difference between the Reach C and Reach F sites is the difference between the percent of time inundated during the total deployment period and the growing season period. Figure 19 shows that although the percentage of inundation time at the Reach F sites is within the same range as the Reach C sites, there is 12-14 percent higher inundation time during the total period than the growing season for Reach C sites. Conversely, inundation is higher during the growing season at the Reach F sites. This is likely due to the greater fluvial influence at the Reach F during the spring freshet.

Table 13. Inundation time at the average marsh elevation (m, Columbia River Datum [CRD]), 0.15 m above average, and 1.0 m above average for sites where water level data were collected from 2009-2010.

			Total Deployment Period		Growing Season	
		Elevation (m, CRD)	Inundation Time (days)	Percent Time Deployed	Inundation Time (days)	Percent Growing Season
Bradwood Slough	Sensor	0.46	355.8	97.5	55.3	95.3
	Marsh Average	1.59	156.7	42.9	16.8	28.9
	Average + 15cm	1.74	133.4	36.6	13.3	22.9
	Average + 1m	2.59	18.0	4.9	0.1	0.2
Ryan Island	Sensor	0.38	365.0	100.0	58.0	100.0
	Marsh Average	1.93	95.3	26.1	7.2	12.4
	Average + 15cm	2.08	73.4	20.1	4.4	7.5
	Average + 1m	2.93	2.2	0.6	0.0	0.0
Jackson Island*	Sensor	0.64	356.0	100.0	55.0	100.0
	Marsh Average	1.33	195.8	55.0	23.6	43.0
	Average + 15cm	1.48	170.3	47.8	18.9	34.3
	Average + 1m	2.33	37.0	10.4	1.3	2.3
Whites Island	Sensor	0.64	356.0	100.0	55.0	100.0
	Average	1.90	100.8	28.3	8.3	15.1
	Average + 15cm	2.06	78.0	21.9	5.3	9.7
	Average + 1m	2.90	2.8	0.8	0.0	0.0
Wallace Island	Sensor	0.30	358.0	100.0	55.6	100.0
	Marsh Average	1.53	152.6	42.6	16.0	28.8
	Average + 15cm	1.68	125.9	35.2	12.2	21.9
	Average + 1m	2.53	13.1	3.7	0.0	0.1
Cunningham Lake	Sensor	0.76	313.5	85.9	40.3	69.5
	Marsh Average	1.37	142.0	38.9	22.7	39.1
	Average + 15cm	1.52	105.8	29.0	20.0	34.5
	Average + 1m	2.37	23.3	6.4	7.4	12.7
Campbell Slough	Sensor	0.94	359.5	98.7	56.5	97.8
	Marsh Average	1.67	99.1	27.2	19.2	33.2
	Average +	1.82	73.5	20.2	16.3	28.2

15cm					
Average + 1m	2.67	21.8	6.0	7.0	12.0

* Water level data from Jackson Island was compromised due to displacement of the sensor during the year, therefore data from the Whites Island sensor was used after the data was deemed comparable.

Another method for analyzing the hydrologic patterns is to calculate the magnitude and duration of inundation at a given elevation over a set period, or the sum exceedance value (SEV). The SEVs for the average marsh elevations are provided in Table 13. The relationship between the average marsh elevation and the SEV is shown in Figure 20. The correlation between the variables is very strong (r2 = 0.75) indicating that elevation explains much of the variation in SEV. The Campbell Slough data point is a bit of an outlier; removing this point from the regression results in an r2 of 0.94. One possible explanation for the Campbell Slough SEV falling outside of this model is that the rip/rap dike at the mouth (see Channel section) is preventing the channel and marsh from draining as it would without the dike.

These results are remarkable considering the extremely different hydrologic drivers and geomorphic settings of these sites. The Reach C sites are very tidally influenced, with an average tidal range of 1.6 meters, while the average tidal range at Reach F sites is only 0.5 m. In contrast, the fluvial influence is much greater at the Reach F sites, with a pronounced peak in water levels during the spring freshet (most of June this year). The Reach C sites are all relatively close the main stem of the Columbia River (<1 km), whereas the Campbell Slough and Cunningham Lake sites are 1.6 km and 6.4 km from the main stem, respectively.

Table 14. Sum exceedance values for the sites where water level data were collected from 2009-2010.

Site	Average Marsh Elevation	Growing Season SEV (m days)
Bradwood Slough	1.59	36.1
Ryan Island	1.93	13.8
Jackson Island	1.33	52.8
Whites Island	1.90	14.9
Wallace Island	1.52	32.6
Cunningham Lake	1.37	55.7
Campbell Slough	1.67	50.8

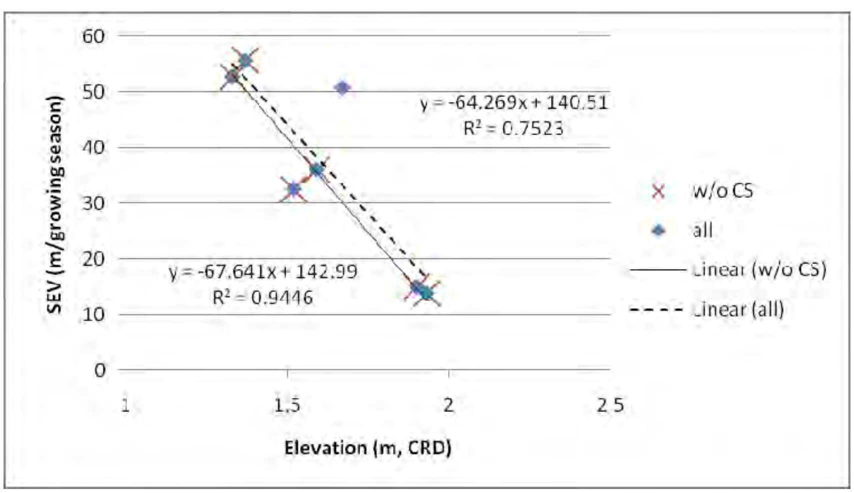


Figure 20. Linear regression results between average marsh elevation (m, CRD) and the SEV for hydrologic data collected in 2009-2010. The dashed line represents the correlation using data from all sites, while the solid line does not include the Campbell Slough data point.

Channel Morphology

The elevations of the cross sections are shown in Figure 21. For all sites, we collected the first cross-section at the mouth of the channel and then collected subsequent cross-sections progressing toward the upper portion of the study area. At some sites, we also collected a "cross section 0," which was located beyond the mouth of the channel (the mouth being defined as the part with vegetated banks). At some locations the cross sections coincided with the locations of the vegetation transects. See Appendix B for site maps showing the location of all cross sections.

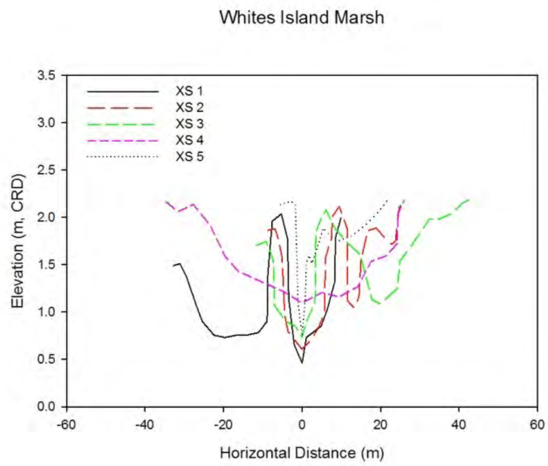
The channel cross sections at the Reach C sites generally follow the pattern of increasing elevation with each subsequent cross section. In other words, there are no sills or blockages to connectivity within the channel. The exception is at Wallace Island where cross section 1 is higher than cross section 0 or 2, indicating that there is a slight sill in the channel. This elevated area of the channel would slightly reduce connectivity of the channel, meaning that at low water levels salmonids would have limited access to the channel.

A single channel cross section was surveyed at Cunningham Lake due to the location of the study area at the uppermost extent of the 6.5 km very shallow and muddy channel. Observations of aerial imagery indicate that the mouth of Cunningham Slough is free from barriers that might influence connectivity or fish access. In 2009, one cross section was surveyed at the Campbell Slough site near the depth sensor. In 2010 we included a survey of the mouth of the slough as well (see map in Appendix B). Cross section 0 was located at the mouth, on the outer side of a low rip/rap dike (the low point is the lowest elev of the dike) and cross section 1 was just up-slough from the dike. It appears that the rip/rap dike acts as a barrier to hydrological connectivity at low water (below approximately 1 m, CRD) and likewise may retain water due to slow or no drainage through the rip/rap.

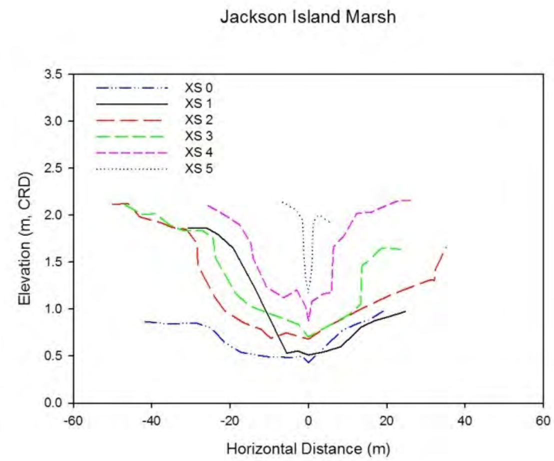
Further analysis of cross-channel data will coincide with the Estuary Partnership's Reference Site Study and will be included in the spatial analysis for the annual monitoring sites as part of the current study.

Channel inundation frequency to a level of 15 cm or more is a measure how often fish would be able to access a channel. This level of channel inundation varied between sites depending on the morphology and elevation of the channels. The mouth of the Reach C channels were generally inundated to a depth greater than 15 cm 100 percent of the time. These sites gradually become less inundated as the channel elevation increases. One of these sites, Wallace Island, had a higher channel elevation near the mouth (XS1), which may limit connectivity during lower water levels. In Reach F, the Campbell Slough channel was inundated to 15 cm depth nearly 100 percent of the time, however the mouth of the slough is only inundated to 15 cm depth 83 percent of the time due to the higher elevation of the rip/rap dike. This reduced inundation is also an indication that the Slough is not connected to the Columbia River at lower water levels. At Cunningham Lake the channel, located at the very upper extent, was inundated to 15 cm 75 percent of the time. The upper extent of all the channels measured were inundated at least 56 percent of the year.

Likewise, the frequency of inundation at the top of the bank is an indication of how often fish can access the marsh edge for feeding and refuge. The inundation at the top of the channel banks at all sites varied between 20 and 55 percent of the time, exclusive of the channel mouth banks where inundation was sometimes higher.



A)



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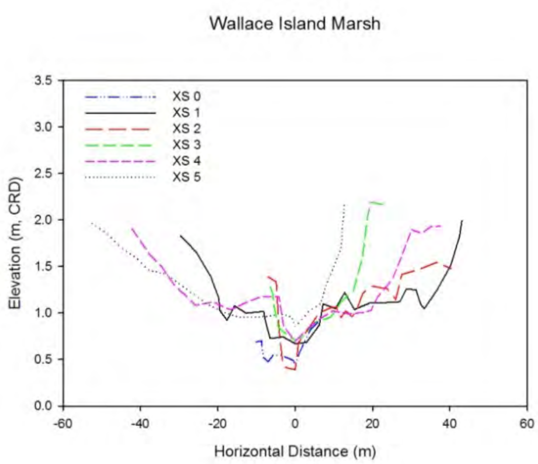

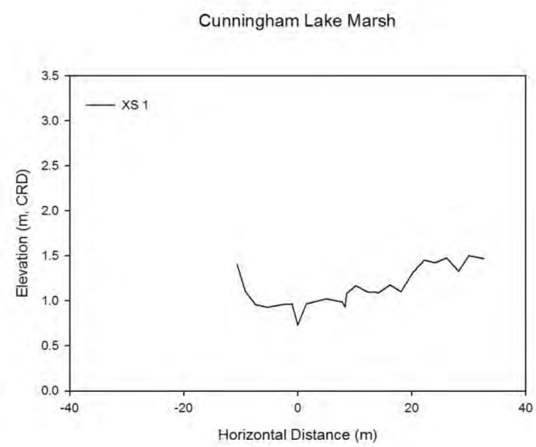

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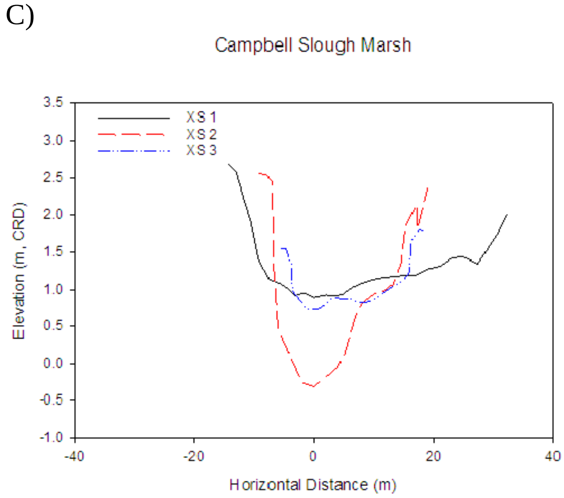


E)

Figure 21. Elevations of the channel cross sections at A) Whites Island; B) Jackson Island; C) Wallace Island; D) Cunningham Lake; and E) Campbell Slough

Table 15. Channel depth and inundation at the cross sections for sites where water level data was collected from 2009-2010. The sensor elevation (in m, CRD) is in parentheses after the site name.

Site	Cross Section Location	Bank Elevation (m, CRD)	Thalweg Elevation (m, CRD)	Channel Depth (m)	% Time WL >15 cm in channel	% Time WL >top channel bank
Bradwood Slough † (0.461)	na	1.711	-0.139	1.850	97*	38
Ryan Island (0.38)	1 (mouth)	1.930	0.214	1.716	100	26
	2	2.015	0.544	1.471	82	23
	3	2.087	0.598	1.489	79	20
	4	2.080	0.762	1.318	72	20
	5	1.972	1.027	0.945	60	25
Jackson Island (0.64)	0	0.839	0.427	0.412	100	79
	1 (mouth)	0.811	0.511	0.300	100	81
	2	1.419	0.678	0.741	80	51
	3	1.458	0.702	0.756	78	49
	4	1.660	0.876	0.784	70	39
	5	1.931	1.172	0.759	56	27
Whites Island (0.64)	1	1.511	0.462	1.049	100	47
	2	1.881	0.607	1.274	86	30
	3	1.746	0.735	1.011	77	36
	4	2.029	1.094	0.935	60	23
	5	1.598	0.748	0.850	77	43
Wallace Island (0.30)	0 (mouth)	0.702	0.441	0.261	100	95
	1	1.830	0.671	1.159	80	28
	2	1.335	0.391	0.944	100	53
	3	1.279	0.707	0.572	78	55
	4	1.897	0.693	1.204	79	25
	5	1.418	0.853	0.565	70	48
Cunningham Lake (0.76)	1	1.168	0.731	0.437	75*	57
Campbell Slough (0.94)	1 (mouth)	2.009	0.891	1.118	83*	13
	2 (mouth)	1.854	-0.308	2.162	99*	19
	3	1.341	0.728	0.613	99*	50

† At Bradwood Slough we estimated the thalweg depth based on our survey at the depth sensor and estimates of the height of the sensor above the thalweg. Bank elevation was only measured at one location.

* The sensor was exposed a percentage of the time at these sites as follows: Bradwood 3%, Cunningham 14%, and Campbell Slough 1% of the total deployment period. At Bradwood Slough, the thalweg +15 cm elevation is lower than the sensor elevation, therefore the inundation calculation is only representative of the time that the sensor was inundated (e.g., the thalweg was probably inundated more than 97% of the time).

7.3.3 Summary

Monitoring data collected in Reach C resulted in the further characterization of these species-rich sites, with greater tidal influence and more complex tidal channels. The evaluation of the marsh and channel elevations coupled with hydrology data is increasing our ability to better characterize the drivers for the vegetation communities and potential for fish access and rearing at the ecosystem monitoring sites. Continued monitoring at the sites in Reach F has allowed us to evaluate the range of conditions between years and to analyze the relationship between the habitat and the variables driving change (as discussed in the next section) and provide a consistent means of comparison between these sites and others throughout the estuary. We are looking forward to future analyses of spatial variability at the 26 sites monitored to date as part of this program.

Specific characteristics of the monitoring sites are described throughout the report and are summarized here. Sediment grain size at sites in Reach C was primarily silt, with some fine sand components. Total organic carbon (TOC) had slight variations within the site, but generally occurs below 5 percent of the sample and is indicative of mineral rather than organic soils. Comparison with other sites will occur in 2011 to put these sediment characterizations in the larger context of wetlands throughout the lower River and estuary. Vegetation assemblages within Reach C had higher species richness and higher cover than those encountered in other Reaches to date. We suspect that this is due to lower hydrological disturbances because tidal action and flood extremes are muted in this Reach. We will explore this hypothesis further in the 2011 site comparison.

Channel morphology in Reach C varied greatly within and between sites, ranging in depth from 0.3 m at Jackson Island to 1.8 m at Bradwood Slough. This range likely results from differing histories of the sites, with the sites present historically (Bradwood and Ryan) having more incised tidal channels than the other more shallow created sites. Inundation time of the channels at the monitoring sites ranged from 56 to 100 percent of the year, while inundation at the marsh edge of these channels varied from 20 to 55 percent of the year. These values indicate that these marsh-channel systems are providing significant opportunities for fish to access and feed. In future years, we will calculate these values based on information relative to the timing of peak fish migration periods.

Findings from this year's research help to fill data gaps regarding wetland habitats in the LCRE and have implications for restoration actions in the future. Information on the timing, frequency, and magnitude of inundation is an important driver determining vegetation species composition and cover and potentially the habitat capacity for supporting juvenile salmon. The frequency that channels and marsh edges are inundated provides an understanding of the habitat opportunity and the frequency that juvenile salmon could potentially access emergent marsh habitats. The elevation and inundation ranges determined through this research are critical for informing restoration design.

7.4 Water Quality Monitoring and Foodweb Resource Assessment

To support characterizations of salmon habitat by PNNL and NOAA-Fisheries, USGS conducted seasonal water-quality monitoring to characterize basic water quality conditions (e.g., temperature and dissolved oxygen) relevant to salmonids (Table 16). While USGS efforts in past years have been concentrated solely on measuring water-quality conditions at selected sites, the efforts in 2010 were focused also on developing and testing methods for assessing food web resources. For 2010, these methods were applied at the Campbell Slough site, with plans to expand these methods to multiple sites in future years. USGS deployed water-quality monitors Although 5 sites were chosen for salmonid sampling, funding was only available to perform water-quality monitoring at 1 site. For the third consecutive year, USGS deployed a continuous water-quality monitor at Campbell Slough in the Roth Unit of the Ridgefield National Wildlife Refuge in 2010 (Figure 22). This site in Reach F has been sampled for vegetation since 2005 (PNNL) and for fish since 2007 (NOAA Fisheries).

Table 16. Site information for locations of water-quality monitors.

Site	Reach	Latitude	Longitude	Deployment Date	Retrieval Date

Campbell Slough	F	45° 47’ 05”	122° 45’ 14.5”	April 1, 2010	July 30, 2010

Figure 22. Campbell Slough, Roth Unit, Ridgefield Wildlife Refuge. Left: Ponded area, yellow arrow shows direction to water-quality monitor. Right: Pipe housing used to deploy water-quality monitor (This picture is from 2008. In 2009, an extra piece of pipe was added so that the monitor was not left out of the water as happened in 2008.)

7.4.1 Methods

Seasonal Water Quality

The monitor deployed was a Yellow Springs Instruments (YSI) model 6600EDS equipped with water temperature, specific conductance, pH, dissolved oxygen, and depth probes. See Table 17 for the specifics on the accuracy and effective ranges for each of these probes. The deployment period for these monitors 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 2010, the monitors were deployed April 1 through July 30, with visits roughly every 4 weeks to exchange the batteries, check the calibration of the variables, and make any adjustments needed. The monitoring period in 2010 was approximately one month earlier than in previous years to capture conditions during months when salmonids were found at the site in recent years.

Table 17. Range, resolution, and accuracy for water-quality monitors deployed by USGS
[ft, feet; m, meters; °C, degrees Celsius; µS/cm, microSiemens per centimeter; mg/L, milligrams per liter]
resolution, and accuracy for water-quality monitors deployed by USGS

Monitoring Metric	Range	Resolution	Accuracy
Water depth	0-30 ft, 0-9 m	0.001 ft, 0.0003 m	±0.06 ft, ±0.02 m
Temperature	-5 to 70 °C	0.01 °C	±0.15 °C
Specific conductance	0-100 µS/cm	0.001-0.1 µS/cm	± µS/cm
ROX optical dissolved oxygen	0-50 mg/L	0.01 mg/L	±0-20 mg/L
pH	0-14 units	0.01 units	±0.2 units

Assessment of food web resources

In 2010, USGS tested methods to assess food web resources supporting juvenile salmonids at the Campbell Slough site in order to refine protocols to be applied at multiple sites in future years. The goal of assessing food web resources is to relate indicator parameters reflecting food web structure and production to higher trophic levels in order to assess the overall biological integrity of the lower Columbia River (LCREP, 1998). Algal production is particularly important because it is at the base of the food chain. Moreover, some evidence suggests that algal production has recently become a more important component of the Columbia River food chain in comparison to a pre-development food chain that was based more on wetland and intertidal production (LCREP, 1998).

The assessment included measurements of:

1. Water-column nutrient concentrations and photosynthetically available radiation (PAR)
2. Phytoplankton and periphyton biomass
3. Phytoplankton and periphyton net productivity
4. Stable isotope ratios of algae, plants, insects, and juvenile salmonids

Brief descriptions of methods used to collect those data are provided below. Refer to Table 18for sample dates of each component.

Table 18. Water-quality monitor and primary-productivity sampling schedule at Campbell Slough, 2010[PAR, photosynthetically available radiation; chla-a, chlorophyll *a*; AFDM, ash-free dry mass; POM, particulate organic matter]

	April 1	**April 21**	**April 27**	**May 7**	**May 25**	**June 22**	**July 8**	**July 30**
Monitor	deploy monitor		service monitor		service monitor	service monitor		retrieve monitor
Nutrient samples		collect nutrient samples	collect nutrient samples	collect nutrient samples	collect nutrient samples	collect nutrient samples	collect nutrient samples collect blank and replicate samples	
PAR				measure water column PAR	measure water column PAR	measure water column PAR	measure water column PAR	
Periphytometers		deploy periphy-tometers		retrieve perphytometers from 4/21	deploy periphytomers	retrieve periphytometers from 5/25; deploy periphytomers	retrieve periphytometers from 6/22	
Phytoplankton biomass (chl-a, AFDM)				collect phytoplankton biomass samples		collect phytoplankton biomass samples	collect phytoplankton biomass samples	

	April 1	April 21	April 27	May 7	May 25	June 22	July 8	July 30
POM stable isotopes				collect POM stable isotope samples		collect POM stable isotope samples	collect POM stable isotope samples	
Other samples for stable isotope analysis					macrophytes	macrophytes	macrophytes	macrophytes
					deploy rocks		collect periphyton from rocks	

Water-column nutrient concentrations and photosynthetically available radiation (PAR) and Phytoplankton and periphyton biomass

Light at wavelengths of 400-700 nanometers (nm) can be absorbed by photosynthetic pigments in algae and plants and used for photosynthesis. Light in this range is called photosynthetically available radiation, or PAR. Nitrogen and phosphorus are the two nutrients that are required in the greatest amounts for algal growth. PAR and concentrations of biologically available forms of nitrogen and phosphorus are therefore important factors that can influence rates of algal growth. Algal biomass can be estimated by measuring the concentration of chlorophyll-*a*, a photosynthetic pigment that is common to all types of algae, or as ash-free dry mass (AFDM), which measures carbon biomass. Biomass of phytoplankton (suspended algae) and periphyton (attached algae) were measured in concert to provide a more complete assessment of algal availability at the site.

Five one-liter water samples were collected from the intersection of the main slough channel and the ponded area. These samples were composited and subsampled for nutrient concentrations on six sample dates and algal biomass on three dates. Subsamples were filtered to collect particulate organic matter (POM) for stable isotope analysis on three sample dates. To test an alternative collection method, additional water samples were collected using syringes and were filtered for stable isotope analysis of POM. In the future, POM samples will be collected only from the composited water sample in order to determine the average site conditions rather than within-site variation. These data will be used as part of the juvenile salmonid food web analysis (part 4, below). On May 25, rocks collected from the mainstem of the Columbia River that had been brushed clean were placed under water to serve as substrate for periphyton growth because no suitable substrate was available at the site. On July 8, periphyton was scraped from those rocks, filtered, and analyzed for chlorophyll-*a* and AFDM. In the future, suitable substrate will be deployed earlier in the season if none is available at the site, so that periphyton biomass can be determined multiple times throughout the season. During every sampling event, a vertical profile of PAR was measured at five-centimeter increments.

Phytoplankton and periphyton net productivity

Estimation of algal productivity is important in the assessment of aquatic food web resources because algae provide the energetic base of the food chain. In order to characterize algal productivity as representatively as possible, both phytoplankton and periphyton productivity were assessed.

^{14}C Uptake Experiment

The uptake of radioactive tracer carbon during photosynthesis can be used to determine the *in-situ* rate of phytoplankton productivity in the environment (Wetzel and Likens, 1991). Using this approach, water samples with a known concentration of dissolved inorganic carbon-12 (DIC) are spiked with a known amount of radioactive tracer carbon-14 (^{14}C) and incubated in bottles in the stream. After a few hours, the amount of ^{14}C incorporated into the algal biomass during photosynthesis is measured. The uptake of ^{14}C

relative to the total ^{14}C that is available is assumed to be equivalent to the proportion of DIC that is incorporated during photosynthesis, relative to the total DIC available, as follows:

$$\frac{14C\ available\ (known\ spike\ concentration)}{14C\ assimilated\ (measured\ at\ end\ of\ experiment)} = \frac{12C\ available\ (measured\ DIC)}{12C\ assimilated\ (calculated)}$$

(modified from Wetzel and Likens, 1991). Therefore, the calculated ^{12}C assimilated value is used to determine the rate of primary production in mass of carbon assimilated/volume/time. A "dry run" was carried out on July 8 to test these procedures, although tracer carbon was not added due to permitting constraints. Therefore, methods were tested, but no data were collected for this experiment in 2010.

Periphytometers
Nutrient-diffusing substrate (NDS) periphytometers can be used to estimate periphyton productivity. Micro-NDS periphytometers, as described by Wise and others (2009), were used to estimate periphyton accrual during a two-week period three times during the monitoring period. However, results from one deployment could not be analyzed because the site was inaccessible due to high water and the filters were too degraded to analyze by the time they could be retrieved. For each deployment, eight 40-milliliter glass vials were filled with each treatment solution: deionized water (control treatment), Sodium nitrate ($NaNO_3$) solution (nitrogen (N) treatment, 350 micromolar (µM) as N), sodium hydrogen phosphate (Na_2HPO_4) solution (phosphorus (P) treatment, 100 µM as P), or $NaNO_3$ plus Na_2HPO_4 solution (N+P treatment, 350 µM as N and 100 µM as P). The control treatment was used to determine the ambient periphyton productivity rate, while the nutrient treatments were used to assess nutrient limitation or co-limitation. Vials were capped with a 0.45-micron nylon barrier membrane and a glass-fiber filter, which served as the artificial substrate for periphyton growth. Half of the replicates of each treatment were covered with fiberglass window screen to reduce the effect of grazers on phytoplankton accrual. Screened and unscreened accrual rates were found not to be significantly different from one another, so these treatments were pooled for analysis.

The ratios of carbon and nitrogen stable isotopes in tissues of consumers reflect the stable isotope ratios of their food sources (Neill 1992; France, 1995), and therefore, can be useful to determine major food sources, provided that the food sources have distinct isotopic ratios. The stable isotope ratios of carbon and nitrogen were measured from juvenile salmonid muscle tissue and several potential food sources to provide information on the food web supporting juvenile salmonids.

Algae
Samples of POM and periphyton from rock scrapings as described above were filtered and analyzed for stable carbon and nitrogen isotopes. Additional replicates of the periphytometer control treatments were also used as substrate for periphyton for stable isotope analysis.

Plants
Samples of *Eleocharis palustris* (creeping spikerush), and dead and live *Phalaris arundinacea* (Reed canarygrass) were collected from throughout the ponded area on May 25, June 22, and July 8. *Sagittaria latifolia* (wapato) was collected from the same area on July 8, the first sample date when it was present at the site. Because it was not available for collection during earlier sample dates and in order to have a larger sample, it was also collected on July 30 when the monitor was taken down for the season. Plants were rinsed at least five times in deionized water to remove external material, such as invertebrates and algae, and were kept on ice until drying.

Insects and Fish
Insect bodies and juvenile salmonid muscle tissue were collected by NOAA Fisheries staff. Insects were collected from two tows in open water (100 meters) and emergent vegetation (10 meters) using a 250-

nanometer neuston net. Wild juvenile salmonids were collected using a seine and skinned muscle tissue samples were collected. Fish and insect samples were frozen for later processing.

Fish tissue, insects, and plant material were dried in an oven at 60°F for 12 hours. Plants of the same type from the same sample date were composited and ground using a clean coffee grinder. Insect bodies collected on the same date from the same source (open water or emergent vegetation) were composited, ground, and subsampled when enough material was available. In the future, insects will be analyzed separately by taxa and only those that are potential salmonid food will be included (those that are too large to be eaten will not analyzed).

7.4.2 Results

Continuous water-quality data for July were somewhat limited due to the water level dropping below the monitor sensor depth. This occurred for some portion of each day from July 4 through July 20, after which the monitor was continuously exposed. Continuous data are shown in Figure 23. These data are available at http://or.water.usgs.gov/cgi-bin/grapher/graph_setup.pl?basin_id=tdg&site_id=454705122451400#step2. Daily or weekly summaries of these data and relevant state water-quality standards are shown in Figure 24. Average daily minimum, mean, median, and maximum values for each parameter by month are shown in Table 19.

Measured water-quality parameters showed daily and seasonal variation. Water temperature ranged from 7.8 to 25.6° C during the 2010 monitoring period. It increased throughout the period, exceeding the Washington 7-day maximum temperature standard of 17.5° C in mid-May and in late June through July. Nevertheless, Chinook salmon were found at the site on July 6. Compared to 2009, the average daily median temperature was within one degree in May, about 2 degrees lower in June, and 3 degrees lower in July 2010. Differences in average daily maximum temperature between the two years spanned from 0.2 degrees (May) to 5 degrees (July). In 2010, 40% of days with data available during May to July (n=81) had 7-day maximum temperatures meeting the state standard, compared with 9% in 2009 (n=80).

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. The Washington daily minimum dissolved-oxygen standard of 8.0 mg/l was violated consistently from mid-June through July. 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.

pH ranged from 6.8 to 9.6 standard units, averaging 7.2. The Washington maximum water-quality standard for pH was violated during mid-April through mid-May, when the daily maximum pH exceeded the state standard of 8.5. After peaking in April and May, pH decreased from mid-May through June and rose through early July. Washington's minimum pH standard was not violated during the 2010 monitoring period. 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; 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 could have spurred more productivity, resulting in these differences in July pH.

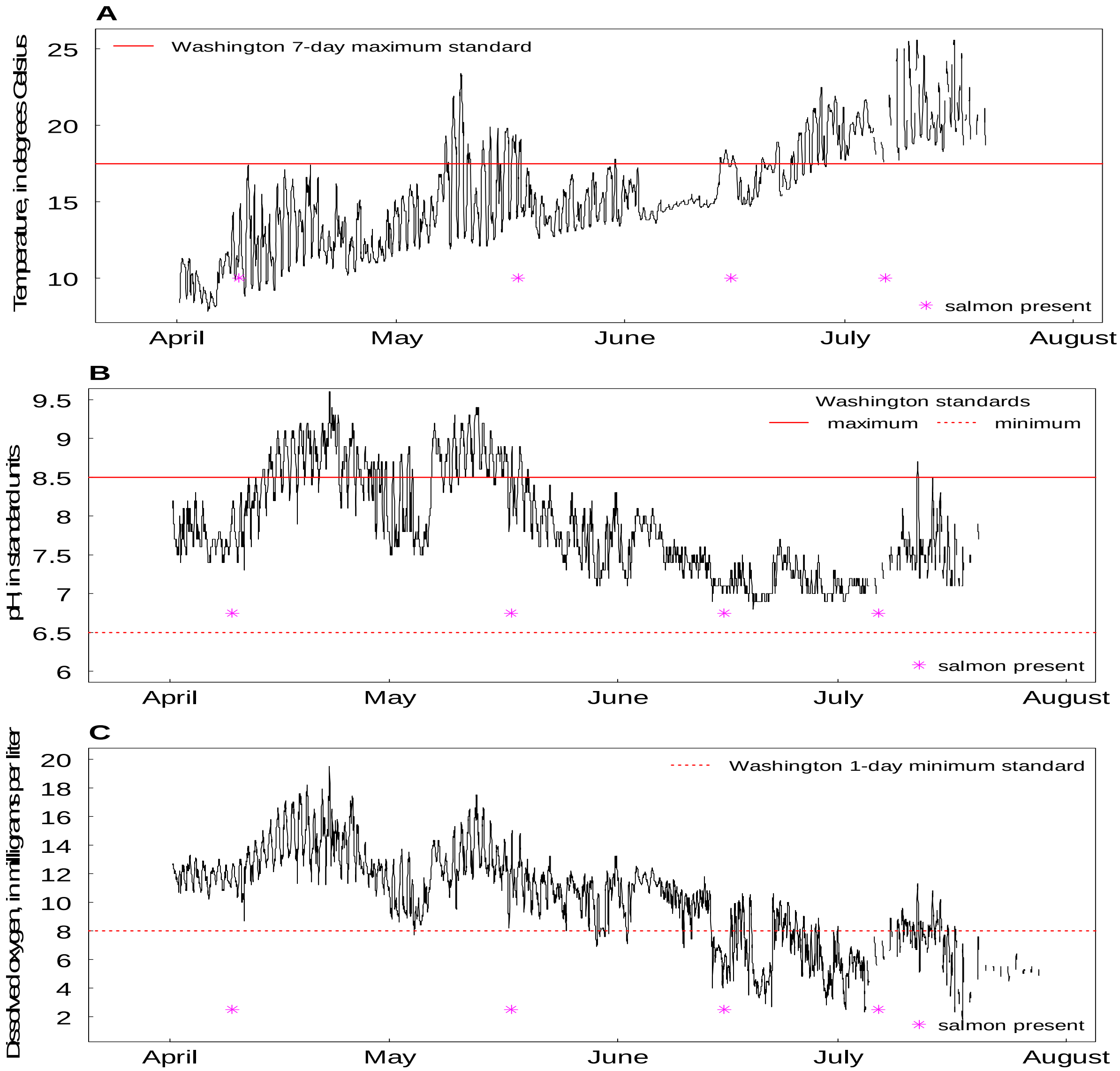
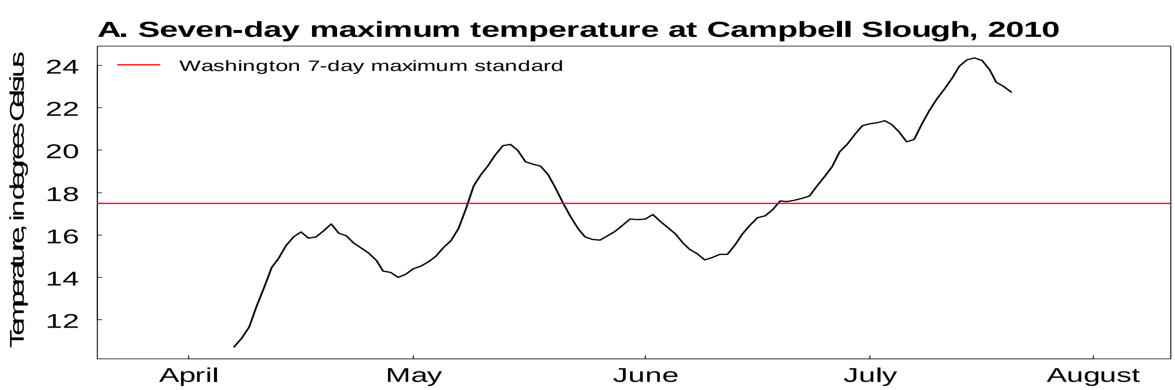


Figure 23. Continuous water temperature (A), dissolved oxygen (B), and pH (C), measured at Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. Measurements were taken every 15 minutes while the monitor was submerged. Data are not available for portions of July, when the water level dropped below the sensor height. °C, degrees Celsius; mg/L, milligrams per liter.



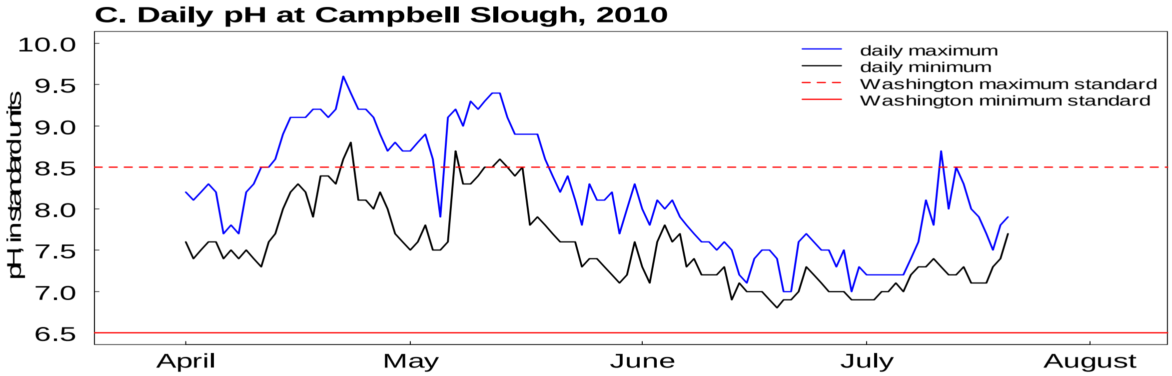


Figure 24.Calculated weekly maximum temperature (A), daily minimum dissolved oxygen (B), and daily minimum and maximum pH (C), at Campbell Slough, Ridgefield, WA, April 1–July 30, 2010, and comparable Washington State water-quality standards.

Table 19. Average daily minimum, mean, median, and maximum water-quality values by month, Campbell Slough, Ridgefield, WA, April 1–July 30, 2010.

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

		April	May	June	July
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

Assess food web resources

Data for this work component were not available for analysis at this time, but will be interpreted next year.

7.4.3 Summary

The sampling location at Campbell Slough is much further away from the mouth and, therefore, the influences of the Columbia River are mitigated. The tidal variations are, however, still noticeable, particularly later in the season when water levels are lower and the factors of snowmelt and dam releases are not as strong. Fluctuations in the specific-conductance values, particularly earlier in the season, indicate that upstream factors may be affecting this site. This observation needs to be explored further.

One of the key reasons for studying these sites is to learn more about their function as off-channel habitat for salmon. Campbell Slough experiences periods of "poor" water quality with respect to conditions for salmon health. Warm water (water temperatures greater than 20 degrees Celsius), low dissolved oxygen (less than 8 milligrams per liter [mg/L]), and high pH (higher than 9) create stressful conditions for salmon.

The influence of algal growth and productivity affected conditions at Campbell Slough. Once water temperatures reached 20 degrees Celsius in late June, pH increased, indicating a period of algal growth. These high pH values (often above 8.5) along with the warm water temperatures can create stressful conditions for salmon. NOAA-Fisheries data indicate that few salmon were observed at this site in July, perhaps because of these stressors. In early August, the pH fell and the dissolved oxygen decreased indicating algal die-off and decomposition. Again, these conditions can be harmful to salmon, but the out-migrating juveniles seem to be on their way to the ocean by August and are no longer using this site.

7.5 Juvenile Salmon and Prey Monitoring

In 2009-2010, NOAA Fisheries focused on the following six work elements:

1. A survey of prey availability and habitat use by salmon and other fishes at three sites in Reach C of the LCRE and data collection on fish habitat use in relation to physical habitat characteristics (monitored by PNNL and USGS). This effort also included re-sampling of the fixed monitoring site at Campbell Slough site in the Ridgefield National Wildlife Refuge (NWR) in Reach F and the 2009 White Island site in Reach C in order to examine year-to-year trends in fish use of these sites.
2. Taxonomic analyses of prey in salmon stomach contents in order to identify preferred prey types at different sites and times, and to compare these with prey identified at the sites.
3. Analyses of otoliths collected from juvenile Chinook salmon at 2009 and 2010 sites for determination of growth rates.
4. Analyses of biochemical measures of growth and condition for juvenile Chinook salmon collected at the 2009 and 2010 sites.
5. Identification of genetic stock for juvenile Chinook salmon collected at 2009 and 2010 sites.
6. Compilation of data and annual report preparation.

In addition to the above work elements, NOAA Fisheries conducted additional research and monitoring activities to build upon information collected between 2005 and 2007. These activities included:

- Chemical analyses of stomach contents, bodies, and bile from juvenile Chinook salmon collected in 2008 from the Reach H sites. Chemical analyses were conducted with NOAA Fisheries funds. Analyses were also done on additional fish collected from sites near the Lower Willamette and Lower Columbia Confluence.
- Completion of reports and manuscripts describing data collected earlier in the Ecosystem Monitoring Project. Manuscripts are intended for publication in peer-reviewed literature using NOAA Fisheries funds.

In spring and summer 2010, we monitored prey availability and habitat use by juvenile Chinook salmon and other fishes at three new tidal freshwater sites in Reach C, Bradwood Slough, Wallace Island West, and Jackson Island (Figure 25). Additionally, we re-sampled fish at the 2007-2008 Ridgefield Wildlife Refuge site (Campbell Slough) in Reach F and the White Island site in Reach C in order to examine year-to-year trends in fish use of the site (Figure 26; Table 20). Our objectives were to collect preliminary information on fish habitat use that may be related to physical habitat characteristics and availability of prey organisms.

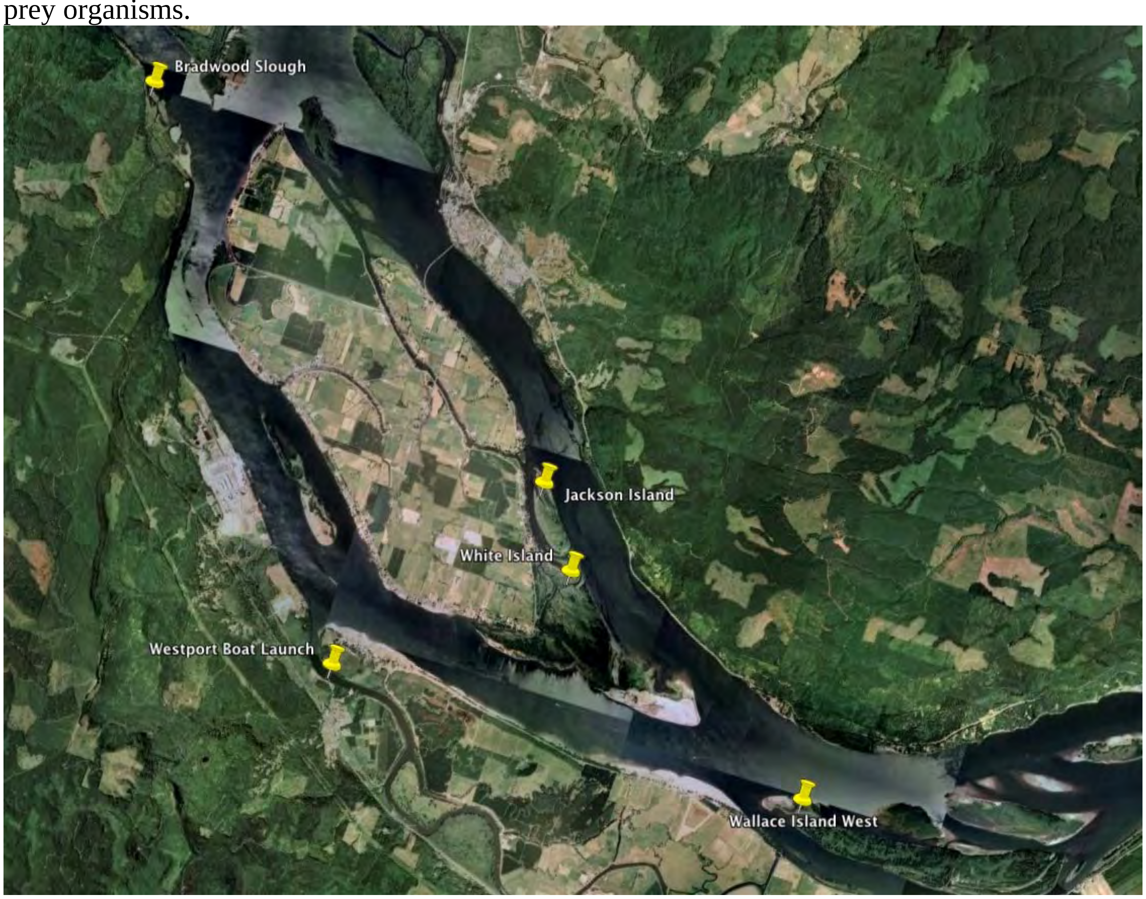


Figure 25. Locations of Ecosystem Monitoring sites in Reach C.

Figure 26. Location of Ridgefield National Wildlife Refuge (NWR) long-term monitoring sites in Reach F of the Lower Columbia River and Estuary.

7.5.1 Methods

Fish Sampling

Fish use of the sites was assessed by analysis of catch data. Fish were collected from April 2010 through August 2010.

Table 20. Coordinates of the sites sampled in 2010.

Site Name	Latitude	Longitude
Bradwood Slough	46° 12.191'N	123° 26.864'W
Campbell Slough	45° 47.032'N	122° 45.291'W
Jackson Island	46° 10.165'N	123° 21.036'W
Wallace Island West	46° 8.428'N	123° 16.986'W
White Island	46° 9.561'N	123° 20.408'W

Fish were collected using a Puget Sound beach seine (PSBS) (37x2.4m, 10mm mesh size). PSBS sets were deployed using a 17 ft Boston Whaler or 9 ft inflatable raft. Up to three sets were performed per sampling time as conditions allowed. Sampled fish were identified to the species level and counted. Salmonid species (up to 30 specimens) were measured (fork length in mm) and weighted (in g) and checked for adipose fin clips and coded wire tags to distinguish between marked hatchery fish and unmarked, presumably wild fish. At each sampling event, the coordinates of the sampling locations, the time of sampling, water temperature, weather, habitat conditions, tide conditions, and vegetation were recorded.

When Chinook salmon were present, up to 30 individual juvenile Chinook were collected for necropsy at each field site at each sampling time. Salmon were measured (to the nearest mm) and weighed (to the nearest 0.1 g), then sacrificed by anesthesia with a lethal dose of MS-222. The following samples were collected from the field-sampled fish: stomach contents for taxonomic analysis of prey; whole bodies (minus stomach contents) for measurement of lipids and persistent organic pollutants (POPs); fin clips for genetic stock identification; otoliths for aging and growth rate determination, and, when sufficient fish were available, bile for measurement of metabolites of polycyclic aromatic hydrocarbons (PAHs); stomach contents for measurement of PAHs and POPs, including dichlorodiphenyltrichloroethanes (DDTs), polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and various organochlorine pesticides. These samples were not collected for coho salmon or other salmonid species because our permits did not authorize this type of sampling for these species.

Samples for chemical analyses were frozen and stored at –80°C until analyses were performed. Samples for taxonomic analyses were preserved in 10% neutral buffered formalin. Fin clips for genetic analyses were collected and preserved in alcohol, following protocols described in (Myers et al., 2006). Otoliths for age and growth determination were also stored in alcohol. The number and type of samples collected at each site and sampling time are listed in Table 21.

Table 21. Samples collected from juvenile Chinook salmon in 2010.

Site	otolith	bile*	stom tax	stom chem	body chem	genetics
Bradwood Slough	55	0	55	0	55	55
Campbell Slough	70	2	70	0	70	70
Jackson Island	45	0	45	0	45	45
Wallace Island West	43	0	43	0	43	45
White Island	81	1	81	0	81	81

*collected as composites at the time of sample collection

Prey Sampling

For the invertebrate prey sampling, the objective was to collect aquatic 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.

In 2010, NOAA Fisheries conducted the following types of invertebrate collections at the monitoring sites:

1) Open water column Neuston tows (2 tows at each site at each sampling time). These tows collect prey available to fish in the water column and on the surface of open water habitats. For each tow, the net was towed for a measured distance of at least 50 m. Invertebrates, detritus, and other material collected in the net were sieved, and invertebrates were removed and transferred to a labeled bottle. The sample was preserved with 95% ethanol.
2) Emergent vegetation Neuston tows (2 tows at each site at each sampling time). These vegetation tows collect prey associated with emergent vegetation and available to fish in shallow areas. For each tow, the net was dragged through water and vegetation at the channel margin where emergent vegetation was present and where the water depth was < 0.5 m deep for a recorded distance of at least 10 m. The samples were then processed and preserved in the same manner as the open water tows.

In addition to the invertebrate sampling along the channel margin, the density and type of emergent vegetation at the sampled sites were noted and photographed. The objective of surveying the % cover of emergent vegetation was to determine if there are correlations between the diversity and abundance of invertebrate prey and the extent of emergent vegetation across sites. To quantify vegetation, a surveyor placed a 0.5x0.5m PVC frame at 5 sites evenly spaced along each 10 m transect. The surveyor then photographed the complete frame and the aquatic area and any vegetation within that frame so that standardized photos could be analyzed later (to ensure analysis is as objective as possible, photos from all sites will be analyzed in random order after code names have been assigned). The surveyor also visually assessed and recorded estimates of % cover and type of vegetation within each frame, and photographed the larger area sampled (upstream and downstream from the transects).

Sample Analyses

Genetic analysis. 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 in 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.

Lipid Determination. As part of our study we determined lipid content in salmon whole bodies. 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.

Prior to analyses, salmon whole body samples from the field were composited by genetic reporting group and date and site of collection into a set of composite samples, each containing 3-5 fish each. In salmon whole bodies composite samples from the total amount of extractable lipid (percent lipid) was determined by Iatroscan and lipid classes were determined by thin layer chromatography with flame ionization detection (TLC/FID), as described in Ylitalo et al. (2005).

Otolith Analyses. Otoliths of juvenile Chinook collected from the 2010 Ecosystem Monitoring sites were extracted and will be processed for microstructural analysis of recent growth in the coming months. Specifically, sagittal otoliths are embedded in Crystal Bond© and polished in a transverse plane using 30-3μm lapping film. Using Image Pro Plus© (version 5.1), with a media cybernetics (evolutionMP color) digital camera operating at a magnification of 20 x, the average fish daily growth rate (i.e., mm of fish length/day) is determined for three time periods: a) the last 7 days of their life, b) the last 14 days of their life, and c) the last 21 days of their life. Average daily growth (DG, mm/day) is calculated using the Fraser-Lee equation:

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

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

where La and Oa represents fish length and otolith radius at time a (i.e., last 7, 14, or 21 days), respectively, d is the intercept (13.563) of the regression between fish length and otolith radius, Lc and Oc are the fish length and otolith radius at capture, respectively.

Chemical Contaminants in Whole Bodies and Stomach Contents. Composite whole body, stomach contents, and feed samples were extracted with dichloromethane using an accelerated solvent extractor. The sample extracts were cleaned up using size exclusion liquid chromatography and analyzed by gas chromatography/mass spectrometry (GC/MS) for PCB congeners, PBDE congeners, and organochlorine (OC) pesticides including DDTs, hexachlorocyclohexanes (HCHs), chlordanes, aldrin, dieldrin, mirex, and endosulfans, as described by Sloan et al. (2004, 2006). Summed PCBs were determined by adding the concentrations of 45 congeners (PCBs 17, 18, 28, 31, 33, 44, 49, 52, 66, 70, 74, 82, 87, 95, 99, 101/90, 105, 110, 118, 128, 138/163/164, 149, 151, 153/132, 156, 158, 170/190, 171, 177, 180, 183, 187, 191, 194, 195, 199, 205, 206, 208, 209). Summed DDT levels (∑DDTs) were calculated by summing the concentrations of *p,p'*-DDT, *p,p'*-DDE, *p,p'*-DDD, *o,p'*-DDD, *o,p'*-DDE and *o,p'*-DDT. Summed chlordanes (∑CHLDs) were determined by adding the concentrations of heptachlor, heptachlor epoxide, g-chlordane, a-chlordane, oxychlordane, *cis*-nonachlor, *trans*-nonachlor and nonachlor III. Summed hexachlorocyclohexanes (∑HCHs) were calculated by adding the concentrations of a-HCH, b-HCH, g-HCH, and lindane.

In addition to POPs, stomach content samples, feed samples, and hatchery body samples were 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; 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, 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.

To adjust for the influence of lipid on toxicity, we normalized whole body contaminant concentrations for lipid, and relied primarily on lipid-normalized data to evaluate potential health effects of toxicants on juvenile salmon. Wet weight data are also presented to facilitate comparison with other studies, and to evaluate risks to predators who consume salmon that have accumulated toxicants.

PAH metabolites in salmon bile. Bile samples were analyzed for metabolites of PAHs using a high-performance liquid chromatography/fluorescence detection (HPLC/fluorescence) method described by Krahn et al. (1986). Briefly, bile was injected directly onto a C-18 reverse-phase column (PhenomenexSynergi Hydro) and eluted with a linear gradient from 100% water (containing a trace amount of acetic acid) to 100% methanol at a flow of 1.0 mL/min. Chromatograms were recorded at the following wavelength pairs: 1) 260/380 nm where several 3-4 ring compounds (e.g., phenanthrene) fluoresce, and 2) 380/430 nm where 4-5 ring compounds (e.g., benzo[a]pyrene) fluoresce. Peaks eluting after 5 minutes were integrated and the areas of these peaks were summed. The concentrations of fluorescent PAHs in the bile samples of juvenile fall Chinook salmon were determined using phenanthrene (PHN) and benzo[a]pyrene (BaP) as external standards and converting the fluorescence response of bile to phenanthrene (ng PHN equivalents/g bile) and benzo(a)pyrene (ng BaP equivalents/g bile) equivalents.

To ensure that the HPLC/fluorescence system was operating properly, a PHN/BaP calibration standard was analyzed at least 5 times, and a relative standard deviation of less than 10% was obtained for each PAC. As part of our laboratory quality assurance (QA) plan, two QA samples [a method blank and a fish bile control sample (bile of Atlantic salmon, *Salmo salar*, exposed to 25 µg/mL of Monterey crude oil for 48 hours)] were analyzed with the fish bile samples (Sloan et al. 2006).

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 FAC values and higher protein content than fish that are feeding constantly and excreting bile more frequently (Collier and Varanasi, 1991).

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

Fish species diversity was calculated using the Shannon-Weiner diversity index (Margaley, 1958):

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

Where

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

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

N = the total number of all individuals

Pi = 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 in Roegner et al. 2009, with fish density reported in number per 1000 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 =[weight (g)/fork length $(cm)^3$] x 100

7.5.2 Results

In 2010, as in other years (Jones et al. 2008, Johnson et al. 2009, 2010) we encountered considerable variation in water level at all of our sampling sites in Reach C and at Campbell Slough (Figure 27).Extreme high water levels and tidal fluctuations, especially in Reach C, made some sites difficult to access at times. We tried to time our Reach C sampling to coincide with high tide, but this was not always possible. Thus, while fish sampling took place every month, it was not always possible to do three fishing sets at all sites each month because of problems with timing and accessibility Table 22.
At all sites, water temperature varied throughout the season, ranging from 7.1 – 11.5°C in April to 19.2 – 22.8°C in August (Figure 28, Table 22). Observed temperatures were similar throughout the season at the Reach C sites (Bradwood Slough, Jackson Island, Wallace Island West, and White Island), with the exception of slightly higher temperatures at Wallace Island West than at the other Reach C sites in July. Temperatures at Campbell Slough tended to be higher than the other sites, especially from April through June. For instance April and May temperatures at Campbell Slough were 11.5 and 17.4°C, respectively, while temperatures at the other sites were in the 7-13 °C range.

Fish Species Composition

Our monitoring efforts in 2010 showed that juvenile salmon and other juvenile fish species were present at all Reach C sites as well as at Campbell Slough in Reach F (Table 23, Figure 29). Salmonid species generally accounted for 5% or less of the total catch (Table 23, Figure 29). Juvenile Chinook were captured at all five sites, with the percentage of total catch for the entire sampling period ranging from 1.2% at lowest site to 4.7% at highest site (Table 23). Coho salmon were captured at two of the five sites (White Island and Bradwood Slough), at percentages ranging from 0.07 to 0.95% of total catch, and chum salmon were captured at four of the five sites, with the percentage of total catch ranging from 0.03 to 0.1%. Of the non-salmonid species, three-spine stickleback and carp were the most abundant (Table 23, Figure 29). The predominant species captured overall was three-spine stickleback, which increased rapidly in numbers from May until August (Table 23). Three-spine stickleback was the dominant species at all of the Reach C sites (Bradwood Slough, Jackson Island, Wallace Island West, and White Island in all months sampled (Table 23, Figure 29), with especially large catches at the Jackson Island site (Figure 30). At Campbell Slough, the catch composition showed more variation over the sampling season. Stickleback accounted for the highest proportion of catch in May and July, and carp accounted for the majority of the catch in August (Table 23). However, in April and June, Chinook salmon and shad, respectively, made up the highest proportions of the catch.

Overall, Campbell Slough had the greatest species richness or total number of species captured (20), with number of species captured at other sites ranging from 7 to 10 (Table 23, Figure 31). Fish assemblages were also analyzed for fish species diversity using the Shannon–Wiener diversity index (Margalev 1958). Campbell Slough had the highest species diversity (Figure 31) while White Island had the lowest. All the Reach C sites had lower species diversity than Campbell Slough (Figure 31); this was reflected in both the larger number and the more equal percentages of different species captured at Campbell Slough in comparison to the Reach C sites (Table 23, Figure 29)

Salmon Occurrence at Ecosystem Monitoring Sites

Chinook salmon were the most abundant juvenile salmon species overall, representing 90% of all salmon captured, as well as the most abundant salmon species at all sites (Figure 32). Chinook represented from 95-100% of the salmonid catch at Jackson Island, Wallace Island West, White Island, and Campbell Slough, while at Bradwood Slough, they accounted for 75% of salmonids captured. Overall, coho salmon made up 7.0% of the total salmonid catch. Coho were most abundant at Bradwood Slough where they made up 23% of the total salmon catch (Figure 32). They were absent from Campbell Slough in Reach F and Jackson Island and Wallace Island West in Reach C, and only made up 2.5% of the salmonid catch at White Island in Reach C (Figure 31). Chum salmon accounted for 2.4% of the salmonid catch. They were most abundant at Jackson Island, where they made up 4.7% of the total salmonid catch; at the other sites, they represented 0 – 3.9% of the catch (Figure 32). In addition to salmon species, one steelhead trout was caught at Bradwood Slough, where it made up 0.7% of the salmonid catch (Figure 32). Trout species were not found at any of the other sampling sites, and accounted for less than 0.2% of the salmonid catch overall. We collected chum salmon mainly in April and May, Chinook from April to August, and coho in May through August. The steelhead trout was found in June.

Overall, CPUE for Chinook salmon was highest at Bradwood Slough (84 fish per 1000 m2) and lowest at Campbell Slough (23 fish per 1000 m2; Figure 10). Bradwood Slough had the highest CPUE for coho salmon as well, as this species was rarely found at any of the other sites (Figure 33). Chum and steelhead CPUE were low at all sites (Figure 33)

At Bradwood Slough, Jackson Island, Wallace Island West, and Campbell Slough, Chinook density (estimated as CPUE) increased from April to a peak in May, then declined again in June (Figure 34). At White Island, the Chinook CPUE was more constant from April through June. In July, an appreciable number of Chinook were still present at Bradwood Slough, but at other sites catches declined to close to zero. At all sites, CPUE was very low by August.

Coho salmon CPUE (Figure 35) was lower than Chinook CPUE at all sites, and was only significant at Bradwood Slough, where a maximum of 100 fish per 1000 m2 were collected. At this site coho CPUE peaked in May, and then showed another smaller peak in July. At other sites, very few coho were captured. For chum salmon (Figure 36), CPUE was still lower (less than 5 fish per 1000 m2). At sites where chum were present, maximum CPUE was in April, and by June no fish were present.

All collected chum and coho salmon, as well as steelhead trout, were unmarked (presumably wild fish), but both marked (hatchery) and unmarked (presumably wild) Chinook salmon were found at all sites (Figure 37). Overall, 19% of Chinook captured were marked hatchery fish. At the Reach C sites, hatchery fish, when present, were less abundant than wild fish, accounting for 6% of the Chinook catch at Bradwood Slough, 7% at Wallace Island West, 11% at Jackson Island, and 22% at White Island (Figure 37). In contrast, at Campbell Slough, 60% of Chinook collected were of hatchery origin (Figure 37).

Figure 38shows the relationship between wild and hatchery Chinook in terms of CPUE. For all of the Reach C sites, the number of wild fish caught per 1000 m2 was much higher than the number of hatchery fish, but at Campbell Slough, the density of marked fish was slightly higher than the density of wild fish.

Genetic Stock Identification of Juvenile Salmon Collected in 2009 and 2010

In 2009, fin clips were collected for genetic analyses from how 132 juvenile Chinook from Campbell Slough, Franz Lake, Lord/Walker Island, Ryan Island, and White Island. Results of these analyses (Figure 39) showed that the fish from the Reach C sites were similar in genetic origin and distinctive from the fish at Franz Lake and Campbell Slough. At all sites, the majority of fish were from the Lower Columbia River ESU (i.e., Spring Creek, West Cascades Fall, or West Cascades Spring stocks). However, at Franz Lake and Campbell Slough, fish from the Spring Creek Group stock dominated the catch, while at the Reach C sites, fish from the West Cascades stock were more abundant. Also, the Reach C sites were being utilized by higher proportions of stocks from other evolutionarily stable units (ESUs), including Willamette River Spring, Upper Columbia Summer/Fall stocks. Genetic samples from 2010 have not yet been analyzed; the report will be updated with the results as soon as they are available.

Salmon Size and Condition

Chinook salmon. Several factors affected the length and weight of sampled Chinook salmon, including origin (i.e., wild vs. hatchery; Table 24and Table 25). Marked, hatchery Chinook salmon were significantly larger than unmarked, presumably wild Chinook. (82 ± 16 mm, n=94 vs. 61 ± 12 mm, n=411, mean ± SD; 6.9 ± 7.1 g, n=94 vs. 2.8 ± 1.8 g, n=411, mean ± SD). Of the unmarked fish, 48% were below 60 mm, in comparison to only 2% of marked fish. For the unmarked fish, length ranged from 35 to 96 mm and weight ranged from 0.2 to 12.3 g. For the marked fish, length ranged from 54 to 177 mm and weight ranged from 1.8 to 55 g. The two fish within this group that were 177 mm in length were likely yearling Chinook, one sampled from Jackson Island and one from White Island in April. Condition factor was slightly higher in hatchery than in wild Chinook (1.07 vs. 1.10) but the difference was not statistically significant (p=0.08).

The mean length and weight of unmarked Chinook (Table 24, Figure 40) differed significantly by site (p = 0.0001). Fish length was significantly higher at Campbell Slough, White Island, and Wallace Island West than at Bradwood Slough or Jackson Island (one-way ANOVA, Tukey's LSD ; < 0.05; Figure 40). Mean length ranged from 56 mm at Jackson Island to 66 mm at Campbell Slough. If the two yearling Chinook capture from Jackson and White Island were excluded from the analysis, there were also significant differences in length among the hatchery fish (p=0.0001; Table 25). Mean length was significantly greater at Campbell Slough than at White Island or Bradwood Slough, with average lengths ranging from 70 mm at Bradwood Slough to 84 mm at Campbell Slough.

The mean weights of unmarked Chinook (Table 24) also differed significantly by site (p = 0.0001). Fish weight was significantly higher at Campbell Slough than at Bradwood Slough, Jackson Island, or Wallace Island, and weights at White Island and Wallace Island West were significantly higher than at Bradwood Island and Jackson Island (one-way ANOVA, Tukey's LSD ; < 0.05). Mean weight ranged from 2.1 g at Jackson Island and Bradwood Slough to 4.1 g at Campbell Slough. Mean weights of the hatchery fish are shown in Table 25. If the two yearling Chinook captured from Jackson and White Island were excluded from the analysis, there were also significant differences in weight among the hatchery fish (p=0.0003).

Mean weight was significantly greater at Campbell Slough than at White Island or Bradwood Slough. Mean weight ranged from 4.0 g at Bradwood Slough to 6.7 g at Campbell Slough

Over the sampling season, the average length of unmarked juvenile Chinook increased steadily each month, from an average of 47 mm in April to 75 mm in August; monthly means differed significantly from April through June, but not from June through August (Table 24, Figure 40). In contrast, the marked hatchery fish showed no such increase in average length (Figure 41). Two large fish (177 mm), probably yearling Chinook, were collected in April, one at Jackson Island and one at White Island, but monthly mean lengths were in the 78-82 mm range at all other months, and not differ significantly from one another (Figure 41).

Multiple regression analysis showed that site, month of capture, and origin (wild vs. hatchery) all had a significant effect on both fish length and fish weight (p=0.0001 for all factors); hatchery fish were larger than wild fish, length increased over the sampling season, and fish from Bradwood Slough were smaller and Campbell Slough larger than fish from the other sites.

In both wild and hatchery Chinook, condition factor varied significantly with month of capture p=0.0001, but did not show a clear seasonal trend (Table 24and Table 25). In wild Chinook, K was significantly lower in April than in any other month, with an average value of 0.91, but then varied between 1.06 and 1.16 from May to August, with the highest value in July and lowest in June. The value in June was significantly lower than in May or July. Similarly, in hatchery Chinook there was significantly variation from month to month p=0.0004. K was lowest in the April (0.96), when the two yearlings were the only coho sampled. From May to August K varied from 1.05 to 1.18, with the highest value in May and lowest in June. K was significantly higher in May than in June or August.

In wild Chinook, K also varied significantly from site to site (p=0.0001; Table 24). Values for Campbell Slough, Wallace Island West, and White Island, which ranged from 1.09 to 1.17, were significantly higher than values for Jackson Island and Bradwood Slough, which ranged from 1.00 to 1.01. However, there were no significant intersite differences in K for hatchery Chinook (p=0.7551; Table 25). Values ranged from 1.06 at Jackson Island to 1.14 at Wallace Island West.

Multiple regression analysis indicated that site (p=0.0001), month (p=0.0001), and origin (hatchery vs. wild; p=0.0016) all had significant effects on K. Condition tended to be higher in wild than in hatchery fish, lower at Bradwood Slough and higher at Ridgefield and Wallace Island West, and lower in April and June and higher in May, July, and August.

Coho salmon. For coho, the fork length of unmarked fish ranged from 40 to 103 mm and weight from 0.4 to 12.5 g (Table 26). The largest coho (92 mm and 10.3 g) was found at Wallace Island West, but that was the only coho salmon captured at that site. At Bradwood Slough and White Island, mean length and weights of coho salmon were very similar (63 and 61 mm and 3.5 and 2.9 g). There were no intersite differences in length p=0.2081, but weight of the fish at Wallace Island was significantly higher than at Bradwood Slough p = 0.05. Over the sampling season, the average length and weight of unmarked coho increased, reaching a peak by July; mean length was significantly higher in July and August than in May and June. Mean weight was significantly higher in July than in May or June and higher in August than in May. Multiple regression results were similar; month had a significant effect on both length and weight, with length and weight tending to increase over the sampling season; site did not have a significant effect on length, but had a borderline significant effects on weight with fish weights at Bradwood Slough tending to be lower and Wallace Island higher. Condition factor increased significantly (p=0.0099) over the sampling season from 1.00 in May to 1.20 in August. The mean values in July and August were significantly higher than in May. However, there were no intersite differences in condition (p =.3076). Values ranged from 1.09 at Bradwood Slough to 1.32 at Wallace Island West.

Chum salmon. The length and weight of chum salmon ranged from 31 to 61 mm and 0.5 to 2.4 g, respectively. Chum tended to increase in size with time, with their average length increasing from 43-54 mm in April to 57 mm in May (p=0.0011) and average weight increasing from 0.68 g to 1.7 g from April to May (p=0.0013; Table 27). However, there were no intersite differences in length p=0.2413 or weight p=0.4200. Average length ranged from 39 mm at Bradwood Slough to 52 mm at White Island and weight ranged from 0.5 g at Bradwood Slough to 1.23 g at Jackson Island. Multiple regression showed month but not site had a significant effect on weight and length of chum. Condition factor did not significantly differ among sites for chum salmon (p=0.2966), with average values ranging from 0.80 at White Island to 0.96 at Jackson Island. Nor did CF vary with month (p=0.5990); the mean value in April was 0.83 and in May was 0.88.

Lipid Content of Juvenile Chinook Salmon

As a biochemical indicator of salmon health and condition, we collected salmon whole bodies for analysis of lipid content and classes. Analyses of whole bodies for lipid content and classes are now in progress for the subyearling juvenile Chinook salmon collected in 2009, and the report will be updated with the results when they are available. The 2010 samples will be analyzed as soon as we have the genetic stock information needed to composite the samples.

Otolith Analysis for Growth Rate Determination

As part of the Ecosystem Monitoring salmon sampling in 2010, otoliths were collected from juvenile fall Chinook salmon from Reach C sites and Campbell Slough. The otoliths have not yet been analyzed; the report will be updated with the results as soon as they are available.

Contaminants in Whole bodies of Chinook Salmon Collected in 2008, 2009, and 2010

Concentrations of persistent organic pollutants are being determined in whole bodies of juvenile Chinook salmon collected in 2008 and 2009 from Campbell Slough, Franz Lake, and other sites in Reaches C and H. At this point, data are available for juvenile Chinook from Franz Lake, Pierce Island, and Sand Island in Reach H, as well as additional samples from Campbell Slough in Reach F (Figure 42). The major contaminants in salmon from all the Reach H sites were DDTs, although low levels of PBDEs and PCBs were also detected. In the fish from Campbell Slough, DDTs were also present, at concentrations similar to those found in the fish from the Reach H sites. Additionally, bodies of these fish contained PBDEs and PCBs at concentrations several times higher than those observed in fish from Reach H (Figure 19). However, in comparison to contaminant concentrations measured in juvenile Chinook as part of the Salmon and Water Quality Study (LCREP, 2007), concentrations of all three classes of contaminants were in salmon from Campbell Slough, Franz Lake, Pierce Island, and Sand Island were relatively low (Figure 42).

The 2010 samples will be analyzed as soon as we have the genetic stock information needed to composite the samples.

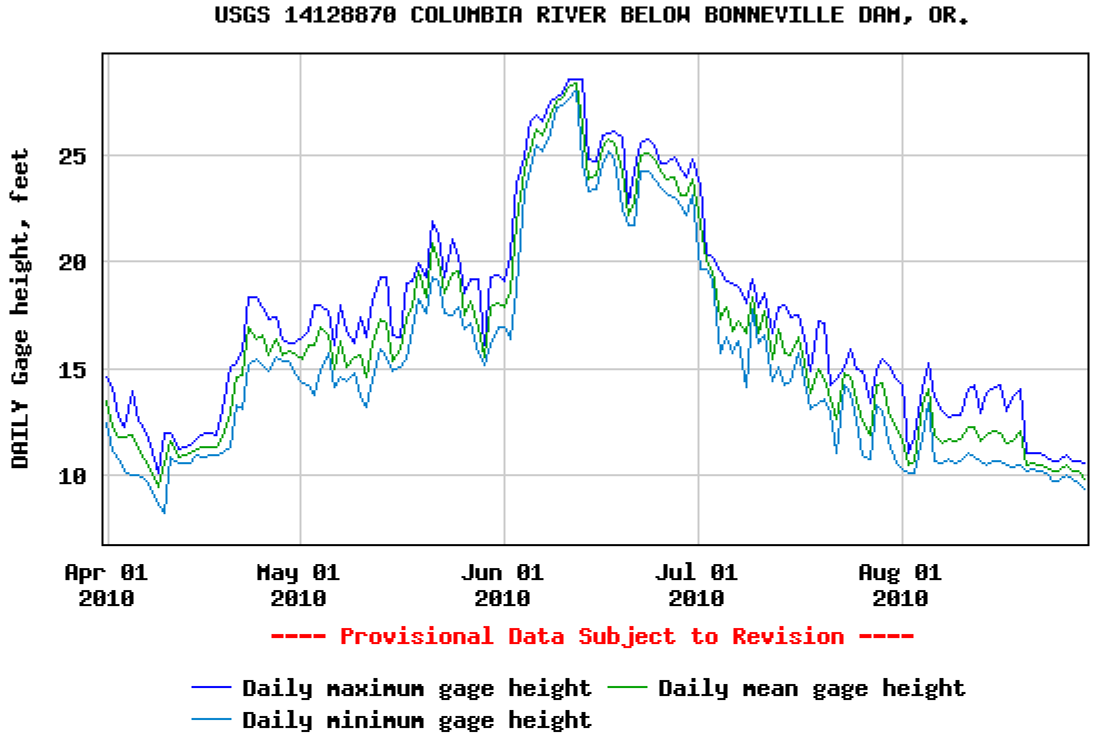


Figure 27. Water depth (ft) below Bonneville Dam (Lat 45° 38'00", long 121° 57'33") over the salmon sampling period. Data provided by USGS.

Table 22. Average water temperature and fishing attempts made at 2010 Ecosystem Monitoring Project fishing sites. The Puget Sound beach seine was used to fish all these sites.

Site Name	Date	Temperature ^{0}C	Fishing attempts
Bradwood Slough	4/7/10	7.8	2
	5/10/10	13.3	1
	6/7/10	13.4	1
	7/8/10	20.6	1
	8/4/10	19.3	2
White Island	4/8/10	8.3	2
	5/11/10	12.8	2
	6/7/10	14.9	2
	7/8/10	19.3	3
	8/4/10	20.5	3
Wallace Island West	4/8/10	7.2	2
	5/10/10	13.4	1
	6/7/10	14.5	1
	7/8/10	22.7	2
	8/4/10	21.5	2
Jackson Island	4/8/10	7.1	3
	5/11/10	13.2	1
	6/7/10	16.7	2
	7/8/10	20.5	2
	8/4/10	19.5	2

Campbell Slough	4/9/10	11.5	3
	5/17/10	17.4	1
	6/15/10	17.4	1
	7/6/10	20.6	3
	8/2/10	22.8	3

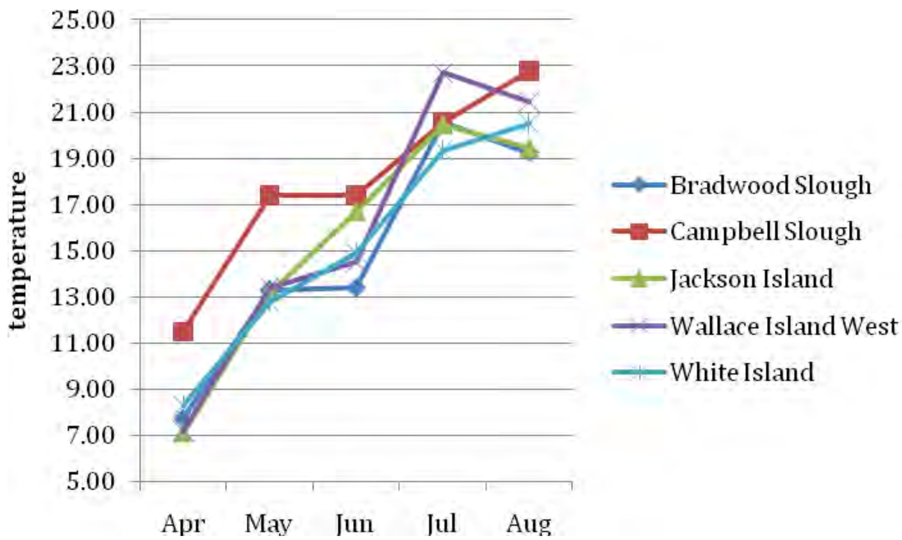


Figure 28. Mean water temperature in degrees centigrade by month at each of the 2010 Ecosystem Monitoring sites.

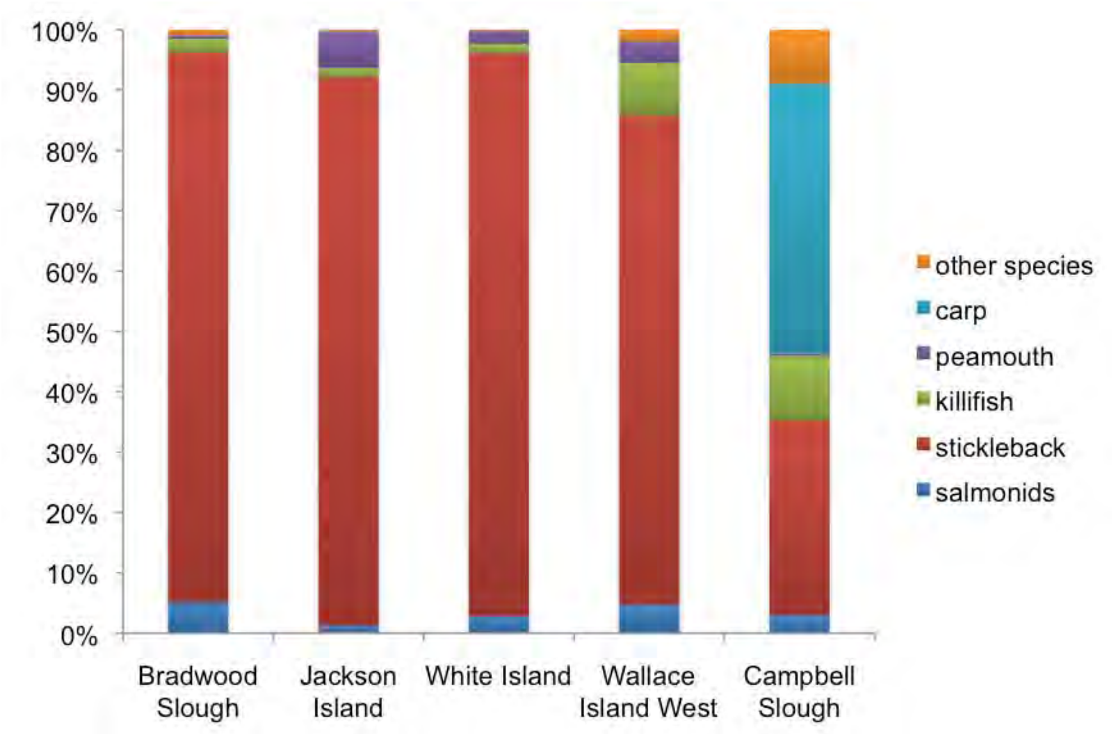


Figure 29. Composition of fish catches by salmonids vs. other species at 2010 Ecosystem Monitoring sites.

Table 23. Total number of each species captured as a percentage of the total number of all individual fish captured.

site name	date	# of species	Bullhead, yellow	bass, small mouth	mosquito fish	carp	chiselmouth	chub	crappie	flounder, starry	Oriental weatherfish	killifish, banded	peamouth	perch, yellow	pike minnow	pumpkinseed	salmon, chum	salmon, Chinook	salmon, coho	sculpin	shad	unknown	trout, steelhead	stickleback, three spine	sucker
Bradwood	04/07/10	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.41	0.00	0.00	0.00	4.23	18.31	0.00	0.00	0.00	0.00	0.00	76.06	0.00
Slough	05/10/09	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.36	0.00	0.00	0.00	0.00	0.00	9.90	2.40	0.00	0.00	0.00	0.00	87.34	0.00
	06/07/10	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.40	0.00	0.75	0.00	0.00	0.13	94.22	0.00
	07/08/10	2	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.07	0.00	0.95	0.00	0.00	0.00	0.00	0.00	1.90	0.88	0.73	0.00	0.00	0.00	95.17	0.00
	08/04/10	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.86	1.69	0.00	0.00	0.00	0.00	0.45	0.34	1.47	0.00	0.00	0.00	90.19	0.00
	Total	10	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.09	0.00	2.27	0.58	0.00	0.00	0.00	0.06	4.05	0.95	0.45	0.00	0.00	0.02	91.14	0.00
Jackson	04/08/10	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.69	0.00	0.00	0.00	2.08	38.89	0.00	0.00	0.00	0.00	0.00	58.33	0.00
Island	05/11/10	5	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.03	2.75	0.00	0.00	0.00	0.00	0.00	97.12	0.00
	06/07/10	7	0.00	0.00	0.00	0.00	0.30	0.00	0.00	0.00	0.00	2.94	56.11	0.00	0.30	0.00	0.00	1.28	0.00	0.00	0.00	0.00	0.00	36.95	2.11
	07/08/10	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.39	1.08	0.00	0.01	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.00	98.42	0.00
	08/04/10	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	7.72	0.80	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	91.48	0.00

site name	date	# of species	Bullhead, yellow	bass, small mouth	mosquito fish	carp	chiselmouth	chub	crappie	flounder, starry	Oriental weatherfish	killifish, banded	peamouth	perch, yellow	pike minnow	pumpkinseed	salmon, chum	salmon, Chinook	salmon, coho	sculpin	shad	unknown	trout, steelhead	stickleback, three spine	sucker
	Total	8	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	1.48	6.08	0.00	0.04	0.00	0.03	1.20	0.00	0.00	0.00	0.00	0.00	90.94	0.20
Wallace Island	04/08/10	6	0.00	0.00	0.00	0.00	10.34	0.00	0.00	1.72	0.00	6.90	0.00	0.00	0.00	0.00	0.00	12.07	0.00	0.00	0.00	0.00	0.00	58.62	0.00
	05/10/10	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	11.48	0.00	0.00	0.00	0.00	57.38	0.00	0.00	0.00	0.00	0.00	24.59	6.56
	06/07/10	5	0.00	0.00	0.00	0.00	5.97	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.00	10.38	0.00	0.00	0.00	0.00	0.00	82.70	0.63
	07/08/10	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	15.76	3.87	0.00	0.00	0.00	0.00	1.09	0.00	0.00	0.00	0.00	0.00	79.29	0.00
	08/04/10	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.29	0.00	0.00	0.00	0.00	0.48	0.00	0.00	0.00	0.00	0.00	95.24	0.00
	Total	7	0.00	0.00	0.00	0.00	1.66	0.00	0.00	0.05	0.00	8.74	3.48	0.00	0.00	0.00	0.00	4.72	0.00	0.00	0.00	0.00	0.00	81.03	0.32
White Island	04/08/10	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.08	38.78	0.00	0.00	0.00	0.00	0.00	57.14	0.00
	05/11/10	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.00	0.22	5.52	0.33	0.00	0.00	0.00	0.00	93.82	0.00
	06/07/10	4	0.00	0.00	0.00	0.00	7.08	0.00	0.00	0.00	0.00	0.88	0.00	0.00	0.00	0.00	0.00	38.05	0.00	0.00	0.00	0.00	0.00	53.98	0.00
	07/08/10	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.48	0.09	0.00	0.00	0.00	0.00	1.39	0.09	0.17	0.00	0.00	0.00	96.79	0.00
	08/04/10	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	2.16	3.53	0.00	0.00	0.00	0.00	0.41	0.00	0.0	0.00	0.00	0.00	93.86	0.0

site name	date	# of species	Bullhead, yellow	bass, small mouth	mosquito fish	carp	chiselmouth	chub	crappie	flounder, starry	Oriental weatherfish	killifish, banded	peamouth	perch, yellow	pike minnow	pumpkinseed	salmon, chum	salmon, Chinook	salmon, coho	sculpin	shad	unknown	trout, steelhead	stickleback, three spine	sucker
																				0					0
	Total	8	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.02	0.00	1.60	2.11	0.00	0.00	0.00	0.07	2.63	0.07	0.04	0.00	0.00	0.00	93.31	0.00
Campbell	04/09/10	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	13.51	0.00	0.00	0.00	2.70	8.11	35.14	0.00	8.11	0.00	2.70	0.00	27.03	2.70
Slough	05/17/10	7	0.00	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.21	0.00	0.85	0.00	0.28	0.00	7.63	0.00	0.28	0.00	0.00	0.00	83.05	1.13
	06/15/10	6	0.00	0.00	0.00	0.00	0.00	0.94	0.00	0.00	0.00	5.66	0.00	0.00	0.00	1.89	0.00	31.13	0.00	0.00	39.62	0.00	1.89	18.87	0.00
	07/06/09	8	0.00	0.13	0.00	0.00	0.00	0.00	0.26	0.00	0.00	10.67	0.00	0.00	0.00	0.13	0.00	1.93	0.00	0.13	3.60	0.00	0.00	83.16	0.00
	08/02/09	14	0.43	0.05	0.05	75.31	0.00	1.88	0.00	0.00	0.16	11.83	0.65	2.96	0.54	2.42	0.00	0.00	0.00	0.54	0.27	0.00	0.00	2.15	0.75
	Total	20	0.26	0.03	0.03	44.67	0.00	1.15	0.06	0.00	0.10	10.72	0.38	1.85	0.32	1.59	0.10	2.81	0.00	0.03	2.39	0.03	0.06	32.26	0.61

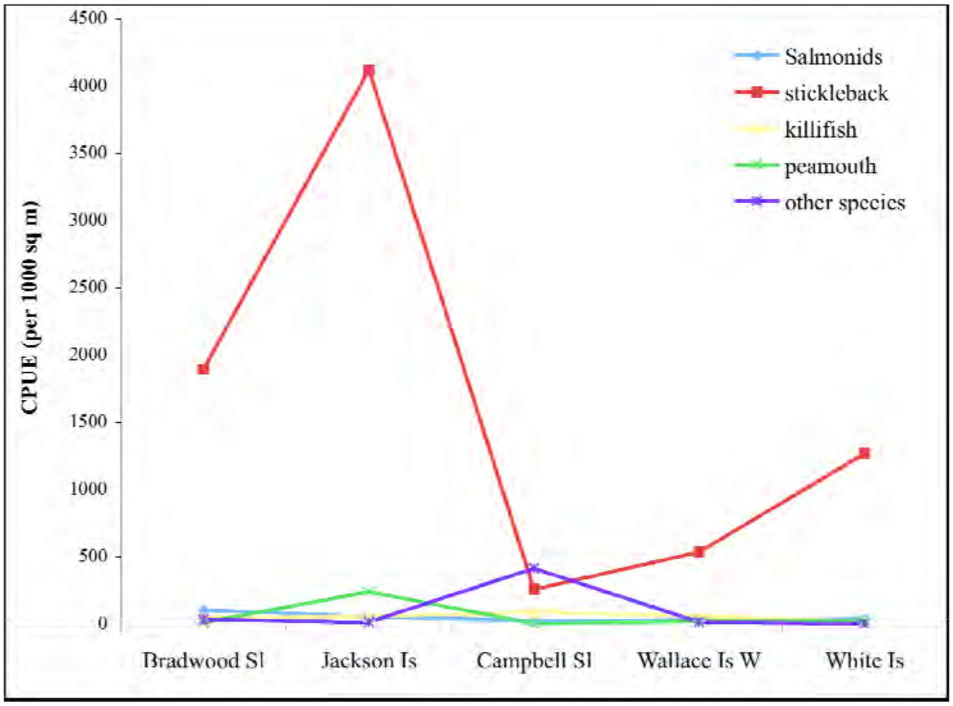


Figure 30. Catch per unit effort for salmonids vs. other species at 2010 Ecosystem Monitoring sites.

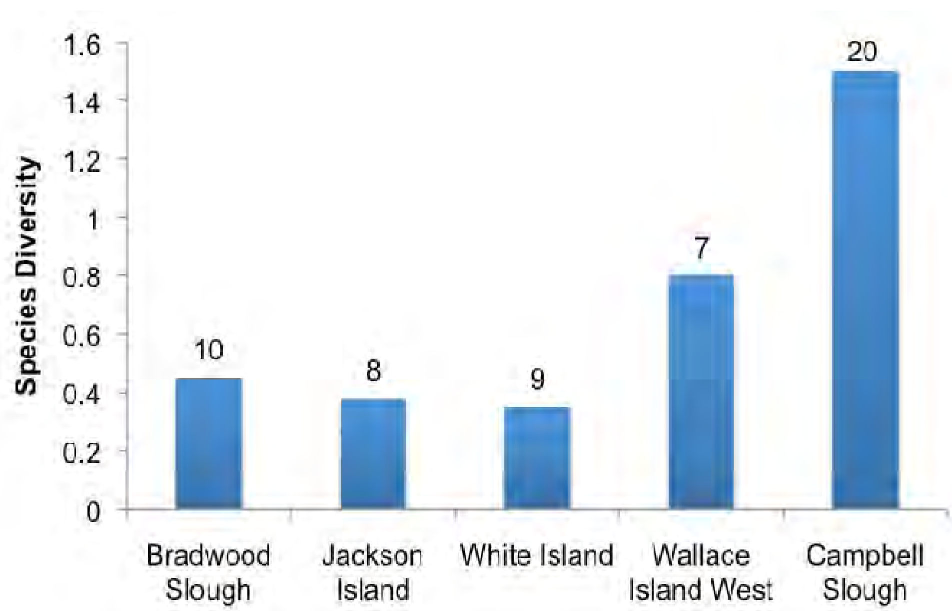


Figure 31. Diversity (Shannon Weiner) and species richness (total number of species captured, number above bar) and 2010 Ecosystem Monitoring sites

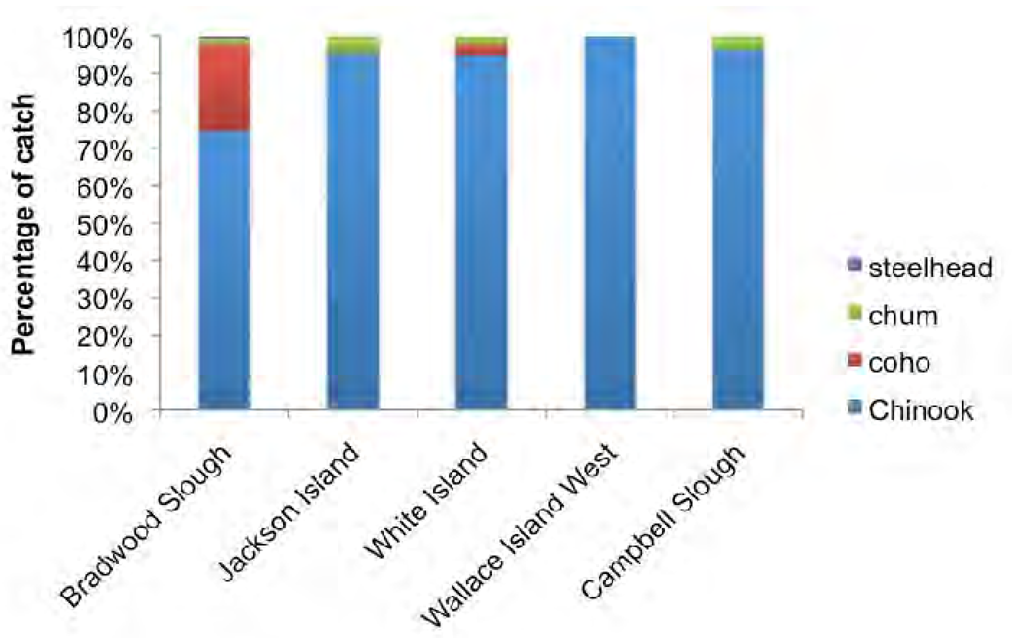


Figure 32. The composition of salmonid catch at 2010 Ecosystem Monitoring sites.

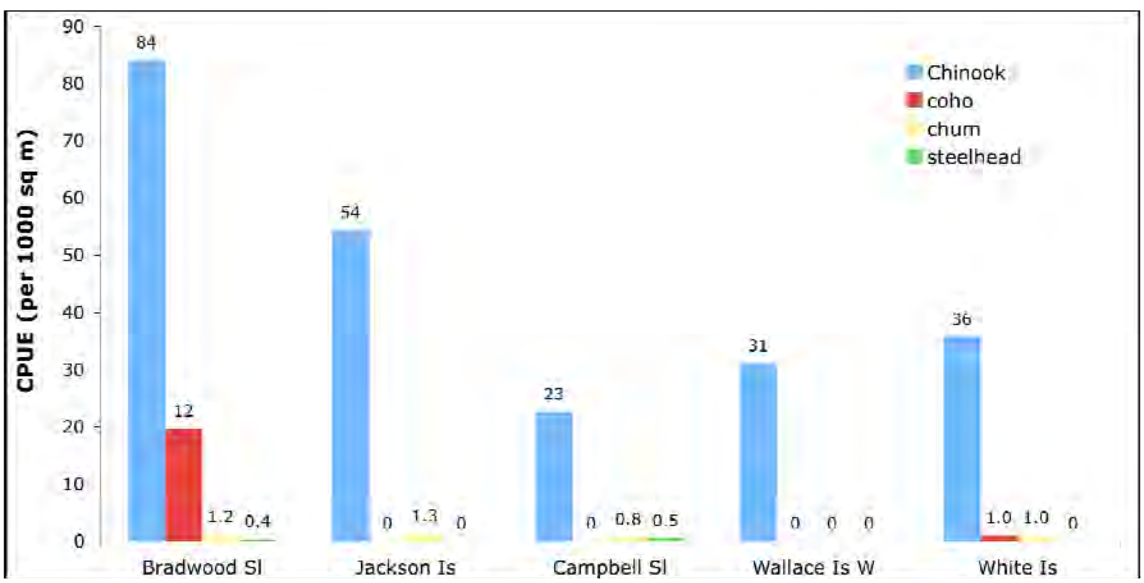


Figure 33. Salmonid catch per unit effort (CPUE) in fish per 1000 sq meters at the 2010 Ecosystem Monitoring sites.

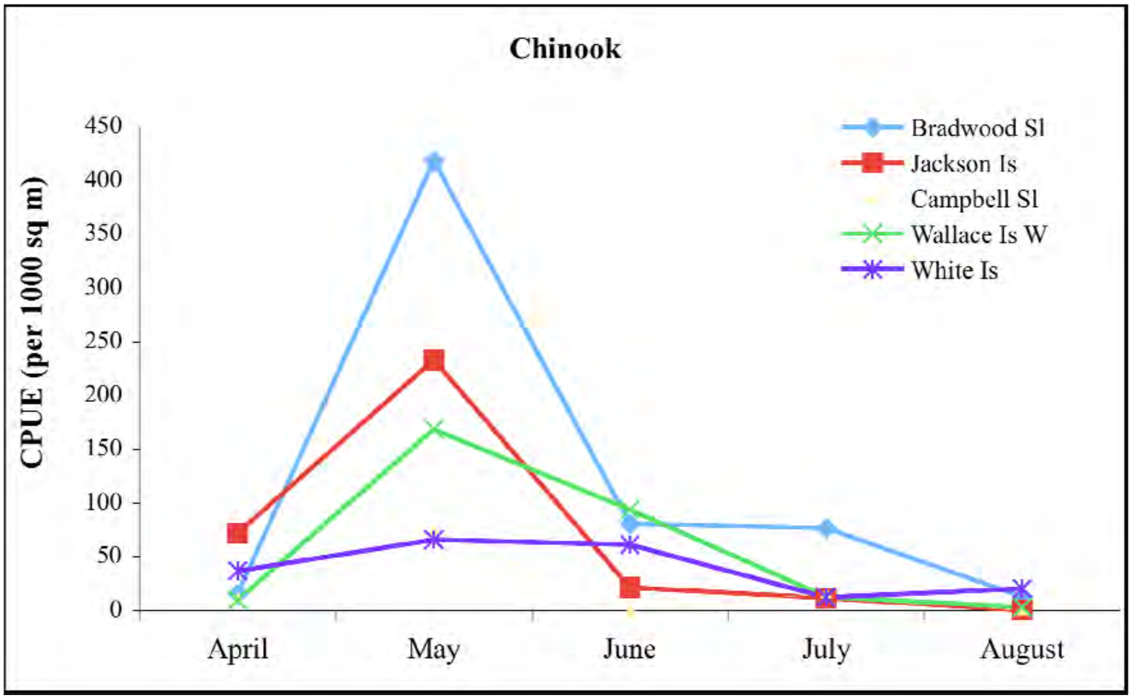


Figure 34. Seasonal trends in the capture of Chinook salmon at 2010 LCREP sites.

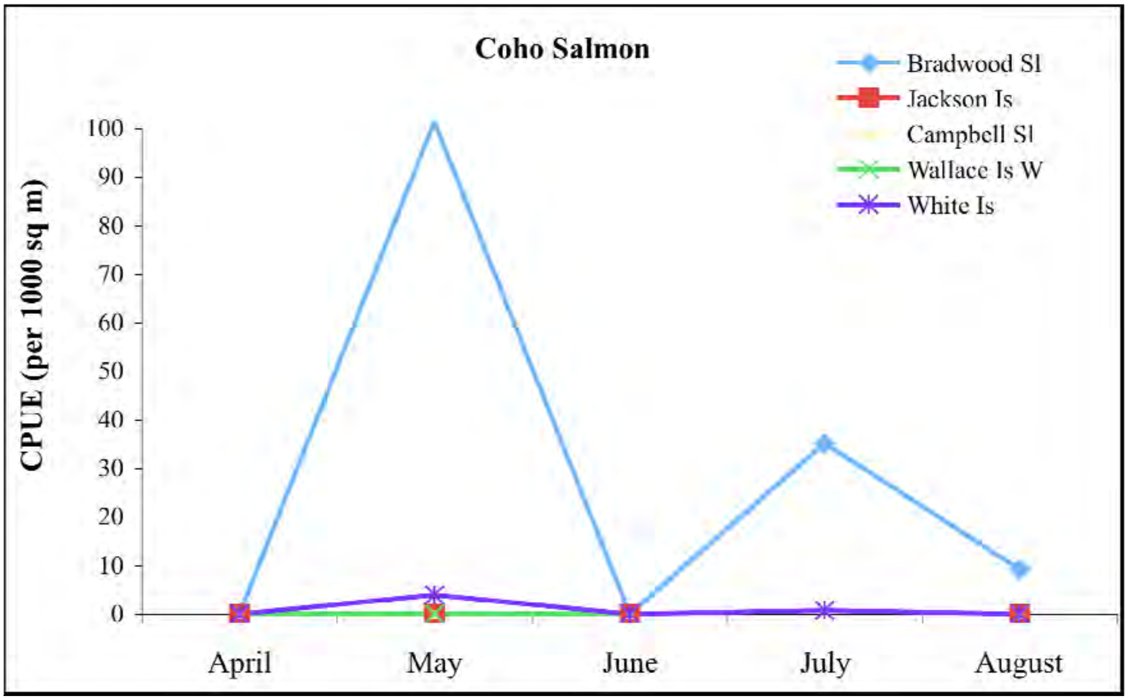


Figure 35. Seasonal trends in the capture of coho at 2010 LCREP sites.

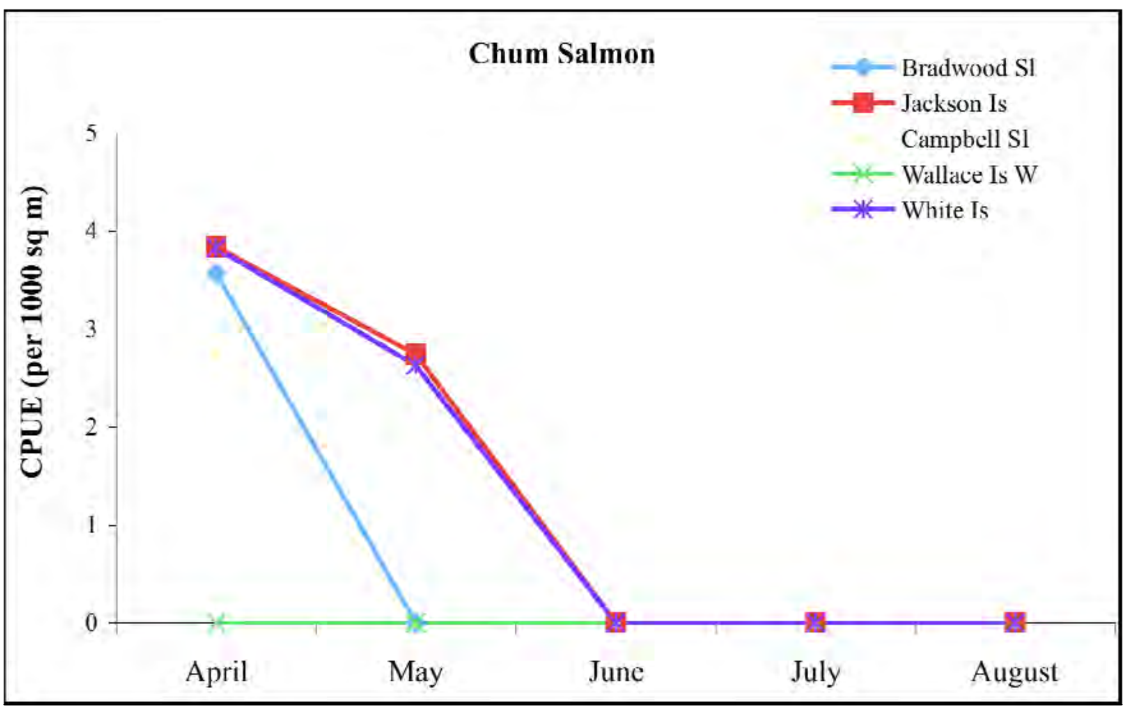


Figure 36. Seasonal trends in the capture of chum salmon at 2010 LCREP sites .

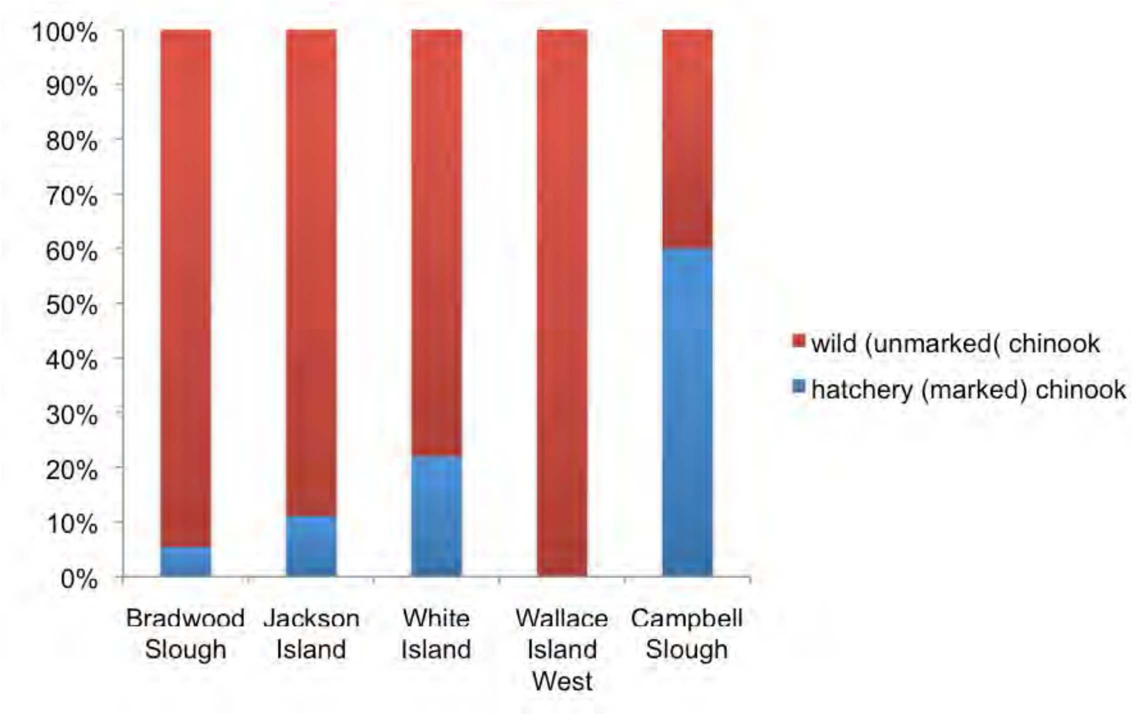


Figure 37. Proportions of wild and hatchery Chinook salmon in catches at the 2010 Ecosystem Monitoring sites.

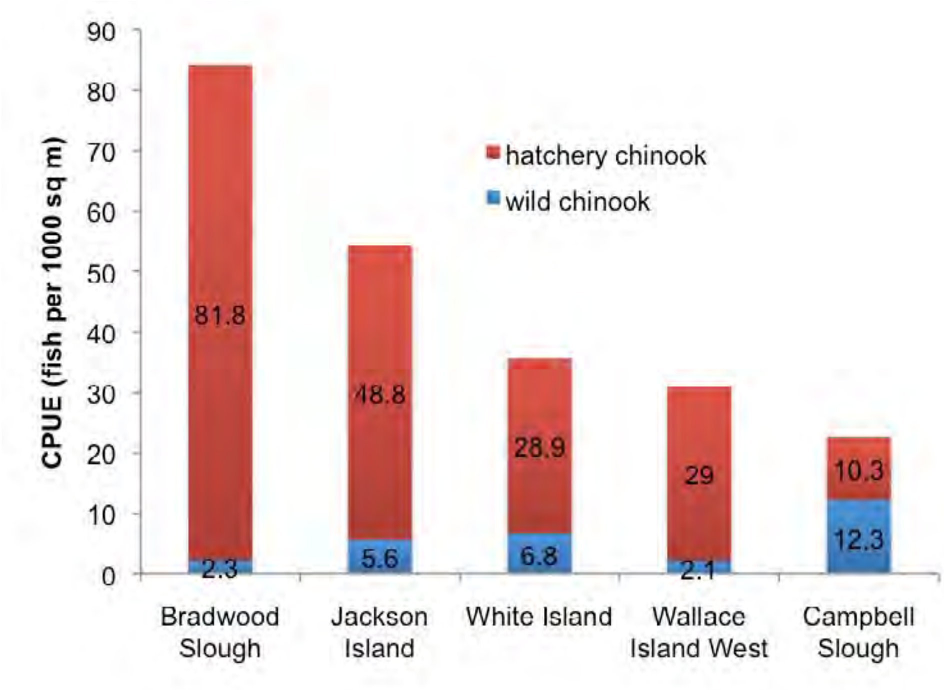


Figure 38. Catch per unit effort of marked (hatchery) versus unmarked Chinook.

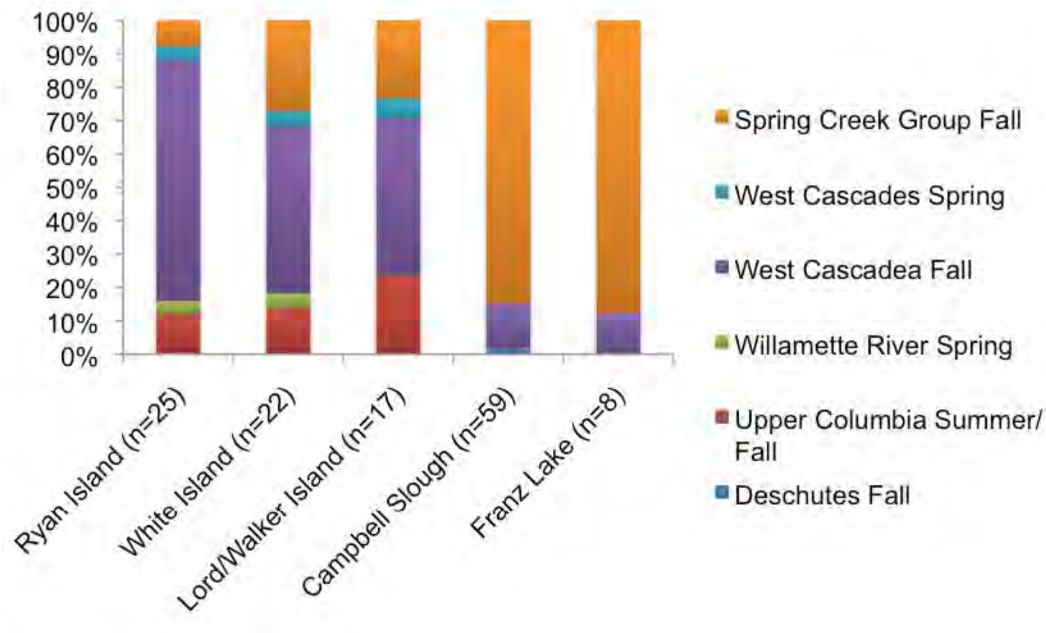


Figure 39. Genetic stock identification of Chinook salmon from 2009 Ecosystem Monitoring sites. Spring Creek Group Fall, West Cascades Spring, and West Cascades Fall stocks are all part of the Lower Columbia River ESU (Evolutionarily Significant Unit).

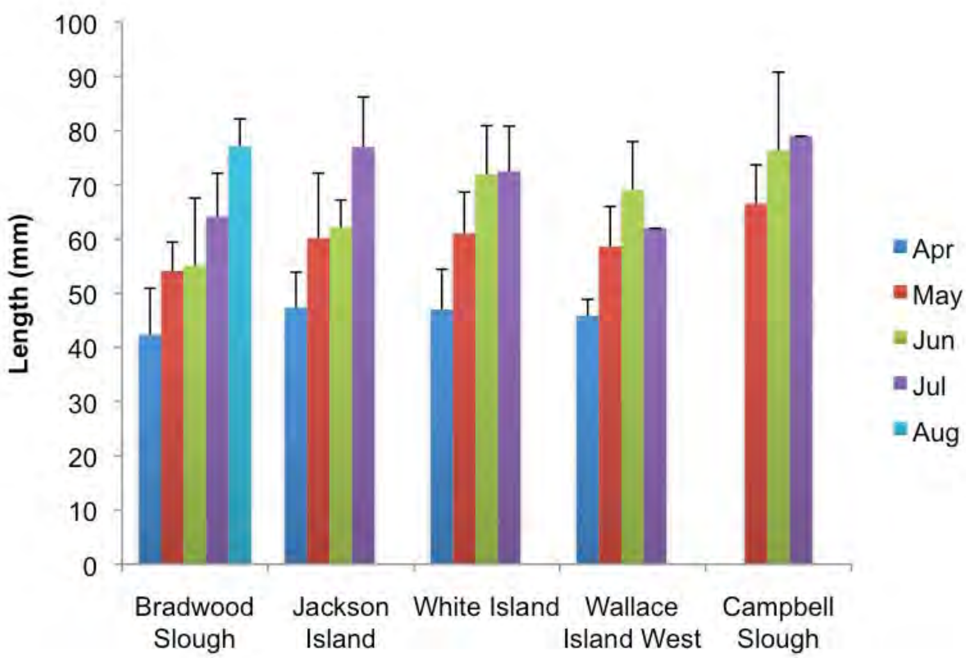


Figure 40. Mean length (± SD) of unmarked (presumably wild) subyearling Chinook salmon over the sampling season at the 2010 Ecosystem monitoring sites.

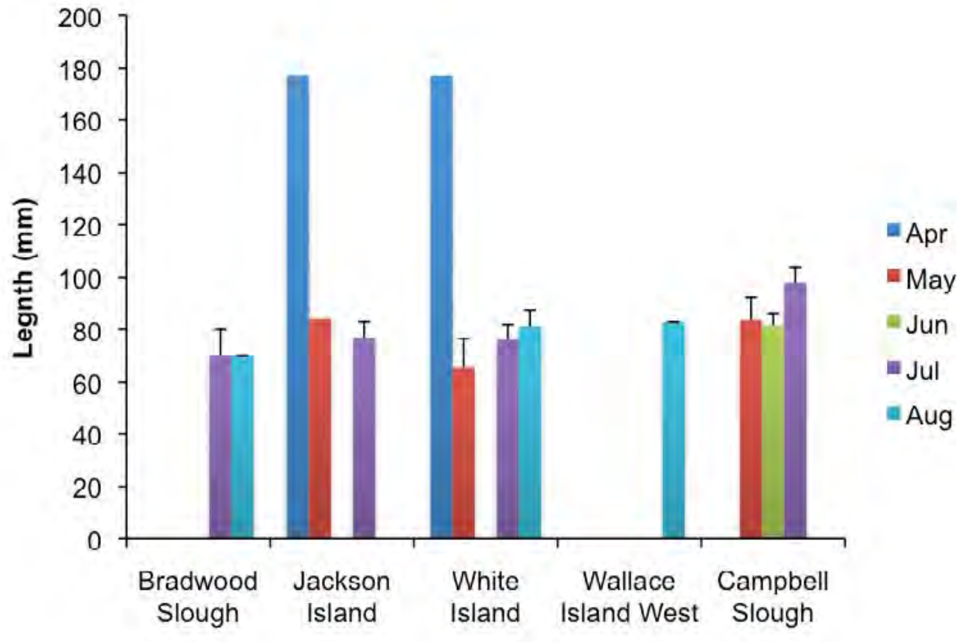


Figure 41. Mean length (± SD) of marked, hatchery Chinook salmon over the sampling season at the 2010 Ecosystem monitoring sites.

Table 24. Mean fork length in mm and weight in g and condition factor (± SD) of unmarked (presumably wild) subyearling Chinook salmon by month at each of the 2010 Ecosystem Monitoring salmon sampling sites. Lower case letters indicate significant differences in length, weight, or condition between months within each site, while upper case letters indicate significant differences in overall length, weight, or condition among sites (Tukey-Kramer multiple comparison test, $p \leq 0.05$).

Site	Date	n	Fork length in mm (±SD)	Weight in g (±SD)	Condition Factor (±SD)
Bradwood	4/7/10	14	42±9d	0.9±0.9	0.94 ± 0.31b
Slough	5/10/10	25	54±5c	1.6±0.5	0.97 ± 0.18b
	6/7/10	35	55±12c	1.9±0.9	0.94 ± 0.10b
	7/8/10	21	64±8b	3.0±1.1	1.11 ± 0.18a
	8/4/10	7	78±5a	12.9±20.8	1.12 ± 0.09a
	Overall	102	57 ± 11B	2.1 ± 1.4C	1.00 ± 0.19B
Jackson	4/8/10	31	47±7b	1.0±0.5	0.89 ± 0.12b
Island	5/11/10	27	60±7a	2.5±0.1.0	1.09 ± 0.12a
	6/7/10	17	62±14a	3.3±2.9	1.11 ± 0.19a
	7/8/10	1	77a	5.6	1.23ab
	8/4/10	0	-	-	-
	Overall	76	56 ± 11B	2.1 ± 1.8C	1.01 ± 0.17B
White	4/8/10	16	47±7c	1.0±0.7	0.91 ± 0.22b
Island	5/11/10	46	61±8b	2.7±1.3	1.14 ± 0.11a
	6/7/10	41	72±9a	4.3±1.6	1.10 ± 0.12a
	7/8/10	7	73±8a	4.5±1.4	1.10 ± 0.12a

	8/4/10	0	-	-	-
	Overall	110	64 ± 12A	3.2 ± 1.8AB	1.09 ± 0.15A
Wallace Island	4/8/10	7	46±3c	0.9±0.3c	0.94 ± 0.09c
	5/10/10	35	59±7b	2.5±0.9b	1.19 ± 0.11b
	6/7/10	33	69±9a	3.8±1.4a	1.09 ± 0.12a
	7/8/10	5	67±7ab	3.7 ± 1.0ab	1.22 ± 0.07ab
	8/4/10	1	62abc	2.9abc	1.22abc
	Overall	81	62 ± 10A	3.0 ± 1.4B	1.13 ± 0.15A
Campbell Slough	4/9/10	12	50 ± 15b	-	0.91 ± 0.12c
	5/17/10	16	67±12a	4.1±1.7	1.32 ± 0.17a
	6/15/10	3	76±5a	4.7±1.3	1.04 ± 0.14bc
	7/6/10	11	79±9a	6.5±2.3	1.28 ± 0.08ab
	8/2/10	0	-	-	-
	Overall	42	66 ± 16A	4.1 ± 2.6A	1.17 ± 0.22A
All Sites	All dates	411	61 ± 12	2.8 ± 1.8	1.07 ± 0.18

Table 25. Mean fork length in mm and weight in g and condition factor (± SD) of marked (hatchery) Chinook salmon by month at each of the 2010 Ecosystem Monitoring salmon sampling sites. Lower case letters indicate significant differences in length, weight, or condition between months within each site, while upper case letters indicate significant differences in overall length, weight, or condition among sites (Tukey-Kramer multiple comparison test, $p \leq 0.05$).

Site	Date	n	Fork length in mm (±SD)	Weight in g (±SD)	Condition Factor (±SD)
Bradwood Slough	4/7/10	0	-	-	-
	5/10/10	0	-	-	-
	6/7/10	0	-		-
	7/8/10	5	70±10a	3.9±1.5a	1.11 ± 0.16a
	8/4/10	1	70a	4.0a	1.17a
	Overall	6	70±9A	4.0±1.4A	1.12 ± 0.14A
Jackson Island	4/8/10	1	177a	50.9a	0.92a
	5/11/10	1	84b	6.2b	1.05a
	6/7/10	0	-	-	-
	7/8/10	7	78 ± 6b	5.2 ± 1.3b	1.09 ± .08a
	8/4/10	0	-	-	-
	Overall	9	89 ± 33A	10.4 ± 15.2A	1.06 ± 0.09A
White Island	4/8/10	1	177a	55a	0.99a
	5/11/10	4	66±11c	3.5±2.1c	1.13 ± 0.09a
	6/7/10	0	-	-	-
	7/8/10	10	76±6bc	5.1±1.0bc	1.15 ± 0.08a
	8/4/10	13	81±6b	5.7±1.1b	1.04 ± 0.09a
	Overall	28	81 ± 21A	6.9 ± 9.5A	1.09 ± 0.10A
Wallace Island	4/8/10	0	-	-	
	5/10/10	0	-	-	
	6/7/10	0	-	-	
	7/8/10	5	76 ± 5a	5.1 ± 0.8	1.14 ± 0.06a
	8/4/10	1	83a	6.4	1.12a
	Overall	6	77 ± 5A	5.3 ± 0.9A	1.14 ± 0.06A
Campbell Slough	4/9/10	0	-	-	-
	5/17/10	11	84±9b	7.1±1.8b	1.21 ± 0.20a
	6/15/10	30	82±5b	5.8±0.9c	1.05 ± 0.08b
	7/6/10	4	98±6a	12.0±2.2a	1.28 ± 0.07a
	8/2/10	0	-	-	
	Overall	45	84 ± 7A	6.7 ± 2.2A	1.11 ± 0.15A
All Sites	All dates	94	82 ± 16	6.9 ± 7.1	1.10 ± 0.12

Table 26. Length, weight, and condition factor of coho sampled in 2010. Lower case letters indicate significant differences in length, weight, or condition between months within each site, while upper case letters indicate significant differences in overall length, weight, or condition among sites (Tukey-Kramer multiple comparison test, $p \leq 0.05$).

Site	Date	n	Length(mm) ±SD	Weight(g) ±SD	Condition factor (K)
Bradwood Slough	05/10/10	13	48±7b	1.2±0.7c	0.97 ± 0.19b
	06/07/10	5	57±4b	2.0±0.3bc	1.09 ± 0.12ab
	07/08/10	12	78±13a	5.9±3.0a	1.16 ± 0.13a
	08/04/10	4	73±7a	4.7±1.4ab	1.20 ± 0.08a
	Overall	36	63±16A	3.5±2.8B	1.09±0.17A
White Island	05/11/10	3	56±5a	2.0±0.4b	1.14 ± 0.10a
	07/08/10	1	78a	5.8a	1.22a
	Overall	4	61±12A	2.9±1.9AB	1.16±0.09A
Wallace Island West	07/08/10	1	92A	10.3A	1.32A

Table 27. Length, weight, and condition factor of chum sampled in 2010. Lower case letters indicate significant differences in length, weight, or condition between months within each site, while upper case letters indicate significant differences in overall length, weight, or condition among sites (Tukey-Kramer multiple comparison test, $p \leq 0.05$).

Site	Date	n	Length (mm) (±SD)	Weight (g) (±SD)	Condition factor (K)
Bradwood Slough	04/07/10	2	39 ± 2	0.5±0.0	0.88 ± 0.15
Jackson Island	04/08/10	3	42 ± 6a	0.7±0.1b	0.91 ± 0.26a
	05/11/10	1	61a	2.4a	1.06a
	Overall	4	48 ± 12	1.2 ± 1.0	0.96 ± 0.21
White Island	04/08/10	2	49 ± 0a	0.9±0.1a	0.81 ± 0.13a
	05/11/10	2	56 ± 6a	1.4±0.6a	0.79 ± 0.06a
	Overall	4	52 ± 6	1.2 ± 0.4	0.80 ± 0.08
Campbell Slough	04/09/10	3	44 ± 4	0.7±0.2	0.76 ± 0.03

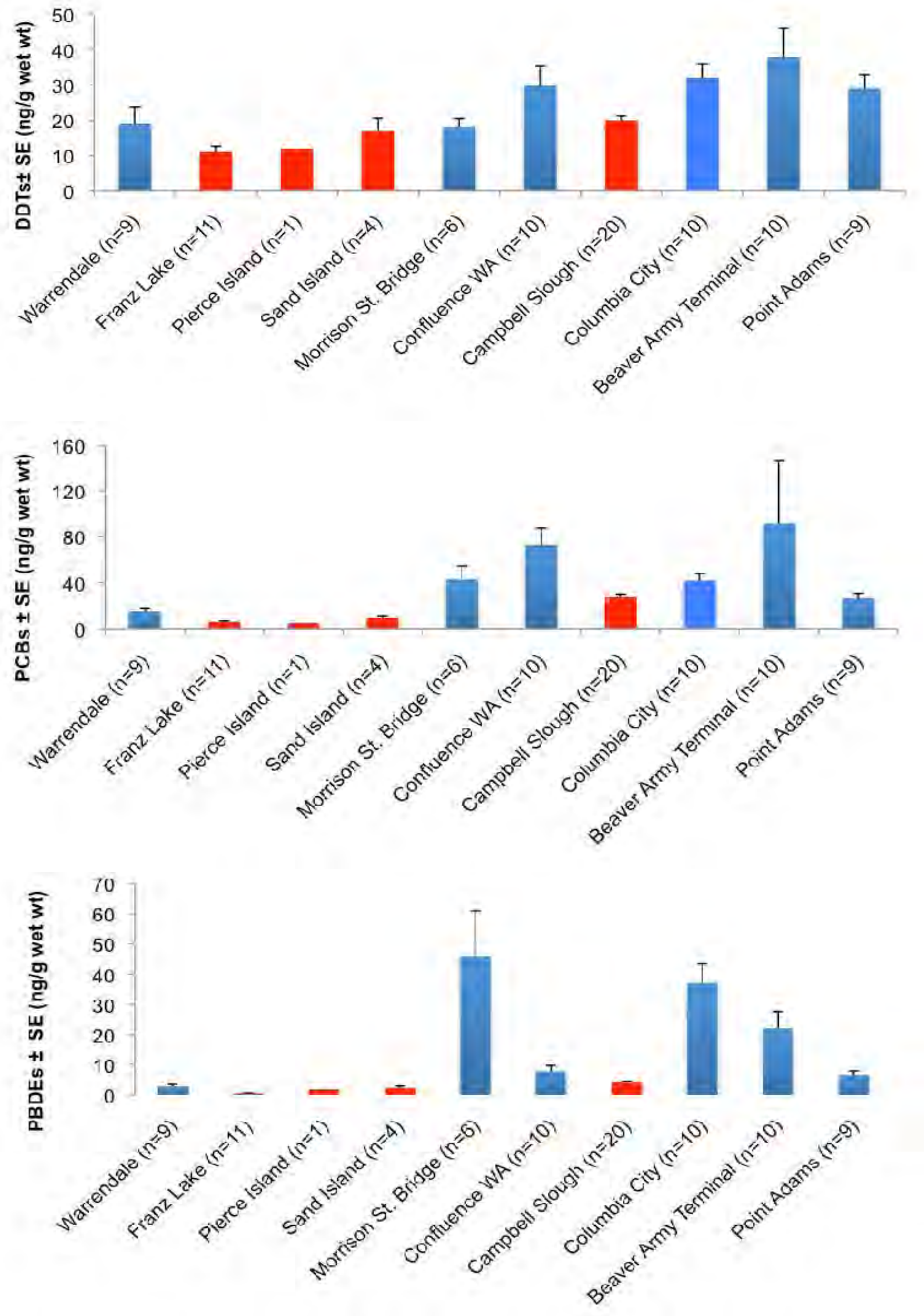


Figure 42. Mean concentrations of DDTs, PCBs, and PBDEs, in ng/g wet wt, in bodies of juvenile Chinook salmon from the 2008 Ecosystem Monitoring sites (red), as compared to sites sampled as part of the Salmon and Water Quality Project (blue; LCREP, 2007). Bars represent standard error.

Salmonid Prey Availability Surveys and Diet Analyses for Juvenile Chinook Salmon

We are analyzing diets of juvenile Chinook salmon and identifying prey species in salmon habitats to understand prey sources for juvenile salmonids and the potential influence of prey availability on juvenile salmonid occurrence in various habitat types. A related objective is to use these data to identify potential sources of contaminants affecting fish in the LCRE. These collections coincided with collections of juvenile salmonids, so that when sufficient numbers of fish were collected the taxonomic composition and abundance of consumed prey can be compared with available prey.

In 2009, we sampled invertebrates at five Columbia River sites (Ryan Island, Lord/Walker Island, and White Island in Reach C, Campbell Slough in Reach F, and Franz Island in Reach H) in an effort to assess the diversity and relative abundance of prey available to juvenile salmonids. Results for Franz Lake and Campbell Slough were presented in more detail in the multiyear report for Campbell Slough and Franz Lake (Johnson et al. 2010). Results for the Reach C sites are shown in Table 28.

Salmonid prey densities were diverse and highly variable across sites (range of means was 0.23 – 43.33 individuals per m towed, Table 28). Among the Reach C sites, the highest prey density was at Lord/Walker Island. Hemiptera made up 93% of the prey items captured at this site; at Ryan Island, the most common prey types were Diptera and Hemiptera, while at White Island, the most common prey items were Diptera and Oligochaetes. In the diet samples (Table 29), Diptera accounted for the majority of prey consumed at all three sites, accounting for 69-92% of the diet. At Lord/Walker and White Island, amphipods were also fairly common prey items, accounting for 21-29% of the diet.

In 2010, 114 emergent vegetation and open water Neuston tow samples were collected over 5 sampling periods at Bradwood Slough, Jackson Island, Campbell Slough, Wallace Island West, and White Island (Table 29). Corresponding diet samples were collected from fish at these sites between May and July (Table 29) from a total of 294 individual Chinook salmon (see Table 20). These samples are currently being processed by the Northwest Fisheries Science Center and by Rhithron Associates.

Table 28. Proportions of samples (by taxa) collected from juvenile Chinook diets and from tow samples. This table includes the mean number of invertebrates found per diet (mean # per stomach) and per tow (mean # per meter sampled) at five sites sampled in 2009. The relative proportions of those samples by taxa are listed below; for example, of the 8 stomachs analyzed from Franz Lake, 93% of the prey items consumed were Diptera. Of the 4 tow samples analyzed from the same dates at Franz Lake, only 19% of the invertebrates caught in tow samples were Diptera. Tow samples represent Neuston tows that were collected from areas with emergent vegetation (each tow 10 m along margin of aquatic habitat) and from open water areas (each tow 50 m through open water habitat), and for each site there were an equal number of each type of tow. The fish sampled were similar in size at the time of collection; mean lengths were 75.5 mm at Franz Lake, 71.8 mm at Lord/Walker Island, 83.9 mm at Campbell Slough, 68.6 mm at Ryan Island, and 61.6 mm at White Island.

	Consumed Prey: juvenile Chinook diets					**Prey availability: Neuston net tows**				
Site	Franz Lake	Lord/Walker Island	Campbell Slough	Ryan Island	White Island	Franz Lake	Lord/Walker Island	Campbell Slough	Ryan Island	White Island
# of samples (stomachs or tows)	8	6	19	19	10	4	4	8	4	4
mean (SD) # of invertebrates per stomach or per m	64.5 (89.11)	13.33 (4.59)	51.16 (49.94)	31.16 (33.07)	22.60 (19.39)	1.17 (1.64)	29.13 (46.48)	43.33 (85.35)	0.64 (0.95)	0.23 (0.41)
Mean proportion by taxa										
Amphipoda		0.21	0.03	0.01	0.29				0.12	0.11
Araneae		0.01	0.01	0.01						
Basommatophora								0.02	0.04	
Bivalvia		0.03								
Cladocera						0.08		0.56	0.01	
Coleoptera			0.01							0.11
Collembola			0.01							
Cyclopoida						0.64		0.16		
Diptera	0.93	0.74	0.86	0.92	0.69	0.19	0.02	0.12	0.36	0.33
Hemiptera	0.03		0.02	0.03			0.93	0.01	0.28	0.11
Hymenoptera			0.02							
Insect egg				0.02						
Isopoda										0.02
Trombidiformes	0.01		0.01			0.04		0.01		
Nematoda				0.01	0.01					
Neotaenioglossa							0.01			
Odonata						0.02		0.01		
Oligochaeta		0.01		0.01	0.01	0.02	0.01	0.11	0.04	0.22
Ostracoda							0.01		0.16	0.11
Thysanoptera			0.01							
Trichoptera	0.02		0.01				0.01			

Table 29. Invertebrate samples collected at Ecosystem Monitoring sites in 2010. The number reflects the total number of samples collected, including open water tows and emergent vegetation tows. An "*" indicates juvenile salmonid stomachs were also collected on that date and at that site.

2010 LCREP sampling sites	April	May	June	July	August	Total
Campbell Slough		4*	4*	4	6	22
Bradwood Slough	6	4*		4	5	19
Jackson Island	6	4*	4	4*	4	26
Wallace Island West	6	4*	4	4	4	24
White Island	5	4*	4*	2	4	23
Total	23	20	16	18	23	114

7.5.3 Summary

Our findings to date indicate that while juvenile salmon are utilizing tidal freshwater habitats in all of the sampled reaches (C, E, F, H) for migration, feeding and rearing, the salmonid species and stocks present, as well as the non-salmonid fish community, show distinctive patterns moving downriver from the Columbia Gorge (Reach H) toward the estuary (Reach C).

In 2010 we expanded our coverage of reach C by adding three new sites (Bradwood Slough, Jackson Island, and Wallace Island West) and also resampled White Island to begin to document long-term trends in salmon occurrence, fish community characteristics, and prey availability in this Reach of the river. In general, fish community characteristics, and patterns of occurrence of salmon species at the new sampling sites were very similar to those we observed in at the Reach C sites we sampled in 2009. The numbers and type of species present were almost the same (7-10 species per site), and, as at the 2009 sites, fish communities at the 2010 sites were dominated by stickleback. Again, unmarked, presumably wild Chinook were the dominant salmon species captured; chum salmon were also observed, as well as coho salmon, which were generally found in higher numbers in 2010 than in 2009. Chinook salmon CPUE values for at the 2010 Reach C sites were generally comparable to 2009 estimates, with seasonal peak values in the 50-250 fish/1000 m2 range. An exception was Bradwood Slough, where peak CPUE was 425 fish/1000 m2, higher than any of the other sites samples. Chinook condition factor (K) for Reach C sites overall was higher in 2010 than in 2009 (1.06 vs. 0.94), and among the 2010 sites, condition was higher at Wallace Island and White Island (1.08-1.12) than at Jackson Island and Bradwood Slough (1.00-1.02).

Our observations at White Island in 2010 were also generally consistent with our observations at this site in 2009. The species present were similar and the total number of species was almost the same (9 in 2010 and 8 in 2009). The proportion of salmonids in the total catch was somewhat higher in 2010 than in 2009 (2.8% in 2010 vs. 0.5% in 2009). In 2010, as in 2009, unmarked, presumably wild Chinook salmon dominated the catch, and chum salmon were also present, accounted for about 2.5% of the salmon catch. However, in 2010, a small number of coho salmon were also captured (about 2.5% of the salmon catch), a species that was not observed in 2009. Chinook CPUE was somewhat lower at White Island in 2010 than in 2009, with peak values of 50 as compared to 150 fish/1000 m^2). However, Chinook salmon condition factor (K) at White Island was higher in 2010 than in 2009 (1.09 vs. 1.04).

Temperatures in 2010 were cooler than in 2009, with a maximum temperature of 21°C at White Island as compared to 25°C in 2009. Perhaps in part because of this, juvenile Chinook were present at the site for a longer period. In 2009, Chinook had left the site by early July, whereas in 2010, they were present from April through August. Seasonal patterns of CPUE at White Island were also slightly different in 2009 and 2010. In 2009 CPUE went from almost zero in April to a peak of 150 fish per 1000 m^2 in May, then dropped to about 40 fish per 1000 m2 in early June, then to zero in July and August. In 2010 CPUE remained more constant throughout the sampling season, with around 50 fish per 1000 m^2 from April through June, then declining to values near zero in July and August.

Our 2010 monitoring results at Campbell Slough were also consistent with earlier findings (Johnson et al 2009, 2010). As in 2009, species richness and diversity were higher at Campbell Slough than at the Reach C sites, and the proportion of non-native fish species was higher. As in previous years, carp and stickleback were the dominant non-salmonid species, together making up about 77% of the total catch. Chinook salmon were again most abundant salmon species at Campbell Slough. About 60% of these were of hatchery origin, as compared to 51-95% in previous years. Chinook CPUE for 2010 was about 23 fish per 1000 m^2, comparable to values observed from 2007 to 2009, but somewhat lower than typical values for the Reach C sites. Chinook salmon condition factor (K) was higher at Campbell Slough in 2010 than in 2009 (1.14 vs. 1.05 for unmarked fish and 1.11 vs., 1.03 for marked fish), though the reasons are unknown.

As yet, the results of salmon diet and prey analyses at the 2010 sites are not yet available, but results of the 2009 analyses showed that salmonid prey densities were diverse and highly variable across sites. The range of means was 0.23 – 43.33 individuals per m towed at the Reach C sites (Lord/Walker Island, White Island, and Ryan Island), but the number and identity of consumed prey was surprisingly similar across sites. Juvenile Chinook are often described as opportunistic feeders, but these results suggest that they select Dipteran larvae and pupae at greater rates than would be expected given their modest availability. Juvenile Chinook did not consume Cladocera or Cyclopoida, despite their high abundances at some sites (e.g., Campbell Slough and Franz Lake, respectively). Likewise, juvenile Chinook appeared to consume fewer Hemipterans than expected given their abundances. The selection of Dipterans (primarily Chironomidae larvae and pupae) was consistent across sites within 2009, and this is similar to patterns observed at other sites in 2008.

In addition to our regular fish and prey monitoring analyses, we were able to complete analyses of contaminant concentrations in juvenile Chinook body samples collected in 2008 from Ecocsystem Monitoring sites in Reach H, as well as Campbell Slough. The results indicate that contaminant concentrations in salmon from Campbell Slough are somewhat higher than in fish from the Reach H sites, especially in the case of PCBs and PBDEs. However, even at Campbell Slough, concentrations of these contaminants were well below those observed at urban sites near Vancouver and Portland in the 2007 Salmon and Water Quality study (LCREP, 2007). It is not clear whether fish are picking up PCBs and PBDEs at Campbell Slough, or through time spent feeding and rearing at more industrialized area upstream of the site. Stomach contents samples should provide some information on this.

In summary, our sampling confirmed our 2009 observations that wild juvenile Chinook, coho, and chum salmon are feeding and rearing in representative tidal freshwater sites in Reach C of the LCRE. Fish community composition, species diversity, species richness, and patterns of salmon occurrence at the 2010 Reach C sites and at the Reach C sites we sampled in 2009, and at White Island in 2009 and 2010. All of the sites had a relatively low species diversity and richness in comparison to the sites we have sampled in other reaches, and were dominated by stickleback. However, they also supported multiple salmon species, including chum, Chinook, and coho salmon. Chum salmon were present in April and May only, but in 2010, Chinook and coho salmon were present at at least some sites from April through

August. High water temperatures may have limited salmon use of some sites in July and August of 2009, as fish were present for a longer period in 2010. When reaches were compared, the Reach C sites generally had higher densities (based on CPUE) and higher proportions of wild juvenile Chinook salmon than Campbell Slough or the Reach H sites. They also had higher proportions of chum salmon in catches than either the Campbell Slough or Reach H sites. Condition factor values showed some variation among sites and years, but were generally within a healthy range (1.0-1.2). Overall, the 2010 sampling results highlight emergent marsh tidal freshwater habitats in Reach C as productive rearing areas for juvenile salmonids.

8.0 Multi-Year Trends in Vegetation at Franz Lake and Campbell Slough

Vegetation

This analysis evaluates five years of monitoring data from the Campbell Slough and Cunningham Lake sites in Reach F as well as two years of monitoring data from the Franz Lake site in Reach H. The primary focus of the analysis is the changes in vegetation species composition, community distribution, and aerial cover of the dominant vegetation species. Additionally, we analyzed elevation and hydrology data to determine if annual variations in cover could be explained by hydrological variability.

Overall, the vegetation composition at the three sites was similar in all monitoring years. A weighted similarity index comparing all years at each site shows the vegetation to be at least 80 percent similar between years. In 2007, cows were present at the Campbell Slough site resulting in grazing and trampling of some of the vegetation. If this year is removed from the analysis then the vegetation is at least 93 percent similar between years at this site and at least 86 percent similar at Cunningham Lake between years. The Franz Lake site had the lowest similarity between 2008 and 2009 of 82 percent.

Average percent cover of the three dominant species varied between years (Figure 43) however, the elevation boundaries (upper and lower range) stayed the same. Figure 44 shows the running median of percent cover compared to 10 cm elevation intervals. This analysis shows that the elevation boundaries of the three dominant species did not change between years at Campbell Slough and Cunningham Lake, however the cover change is notable within those boundaries. The elevation ranges are provided in Table 30. To further evaluate these changes we compared the vegetation to the hydrologic patterns in each year as described next.

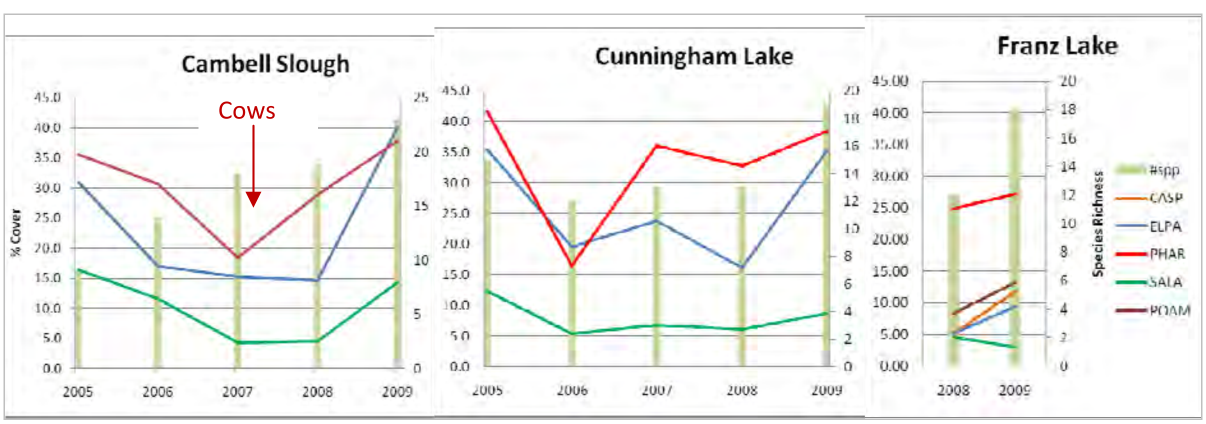


Figure 43. Average percent cover of the dominant species as measured in July of 2005-2009.

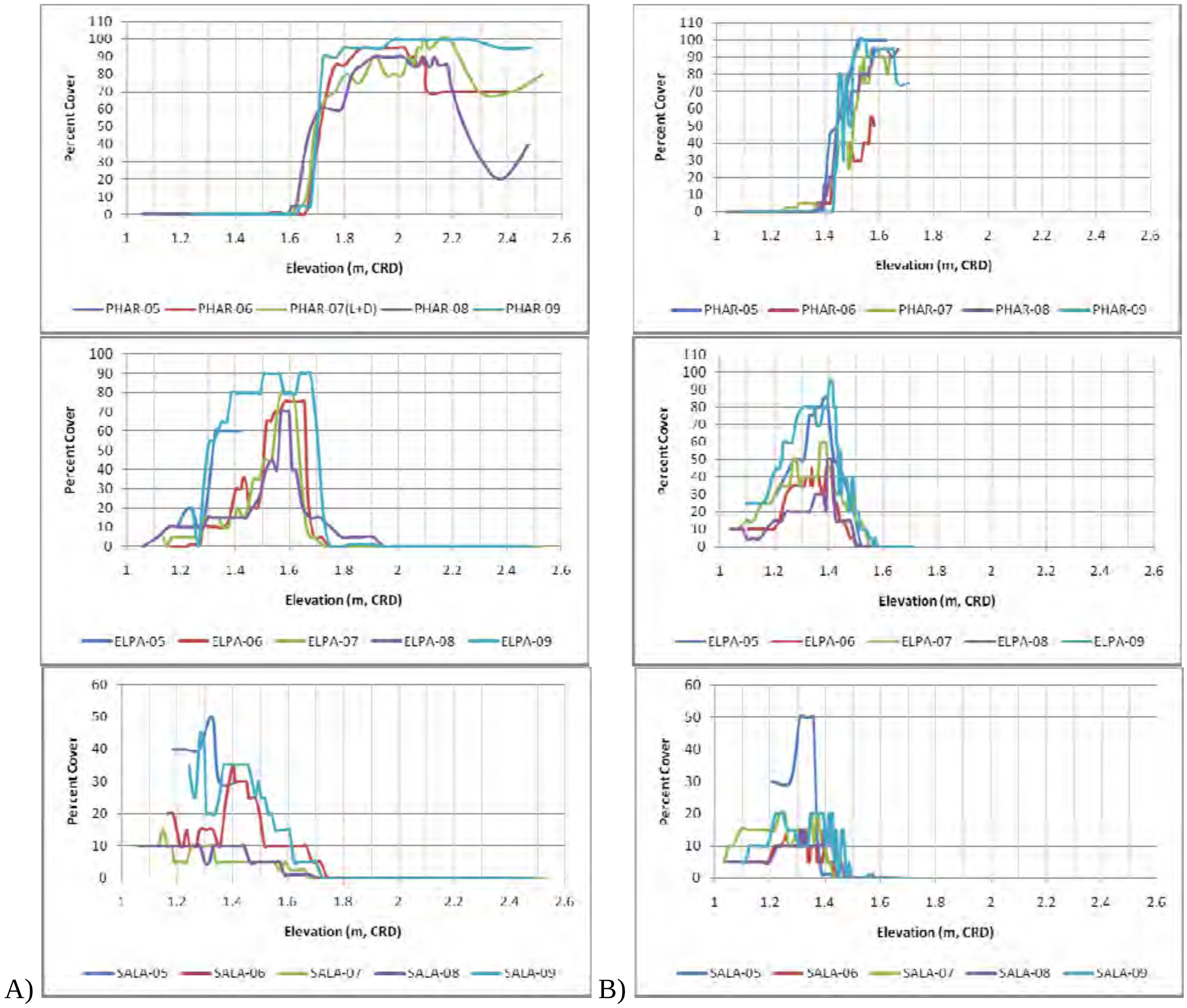


Figure 44. Elevations and the median percent cover between years for the dominant species at A) Campbell Slough and B) Cunningham Lake.

Table 30. Upper and lower elevation ranges for the primary vegetation species found at Cunningham Lake, Campbell Slough, and Franz Lake. Elevations provided in meters relative to the Columbia River Datum (CRD).

	Cunningham Lk. CRD (m)		Campbell Sl. CRD (m)		Franz Lk. CRD (m)	
Sagittaria latifolia (SALA)	**1.01**	**1.41**	**1.10**	**1.40**	**0.78**	**1.78**
Eleocharis palustris (ELPA)	**1.01**	**1.46**	**1.30**	**1.70**	**1.48**	**1.88**

Phalaris arundinacea (PHAR)	**1.21**	**1.71**	**1.70**	**2.70**	**1.78**	**2.08**

Hydrology Patterns

The Columbia River basin is primarily a snow-melt runoff watershed and as such is subject to interannual hydrologic variability. Figure 45 shows the variation in the timing and magnitude of outflow at Cascade Island, just below Bonneville dam for the years 2005-2009 and is provided here to give an overview of the hydrological patterns for the estuary during this study. In general, 2006 and 2008 were high years and 2005, 2007, and 2009 were lower flow years as compared to the 10-year average flow.

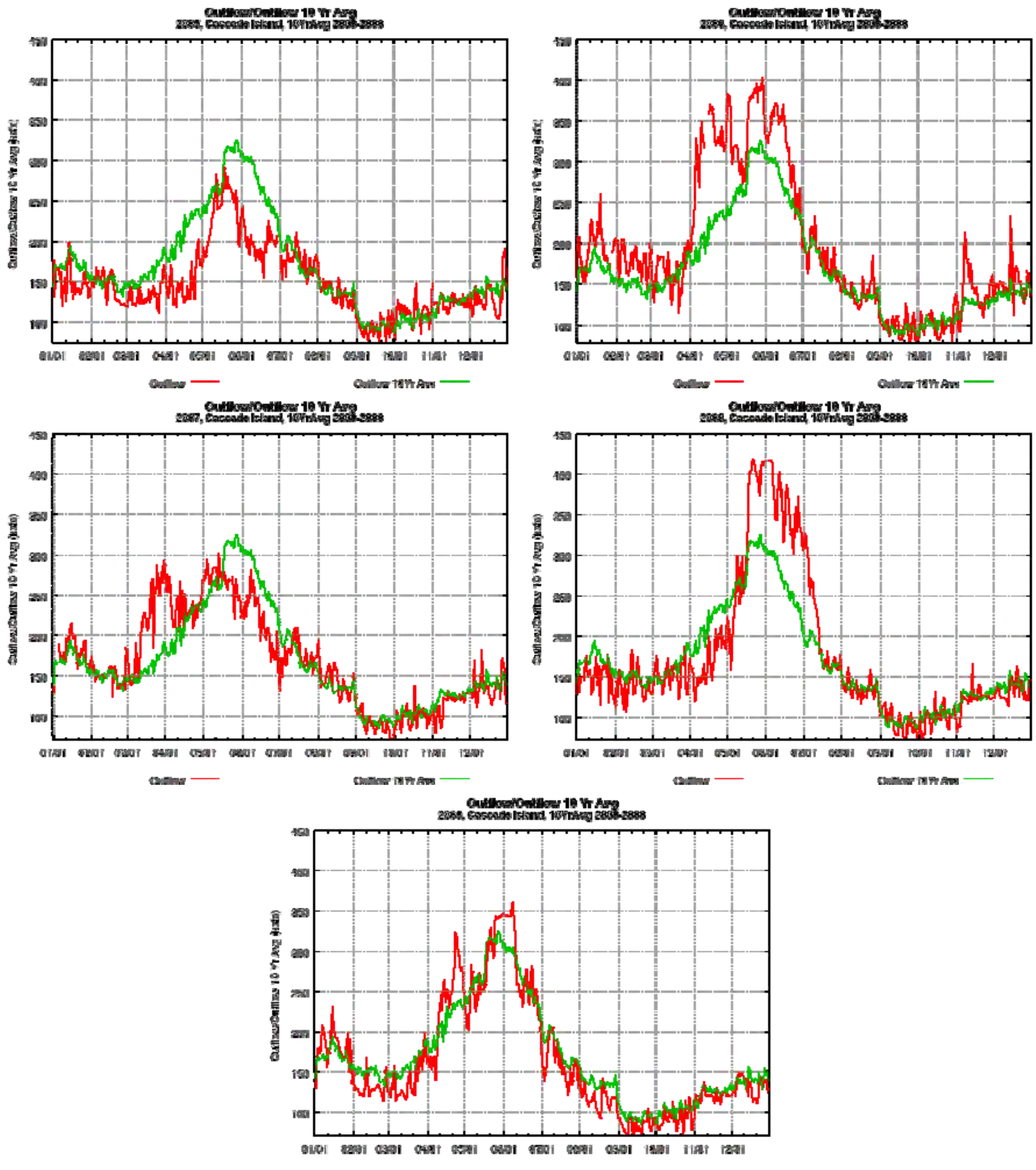
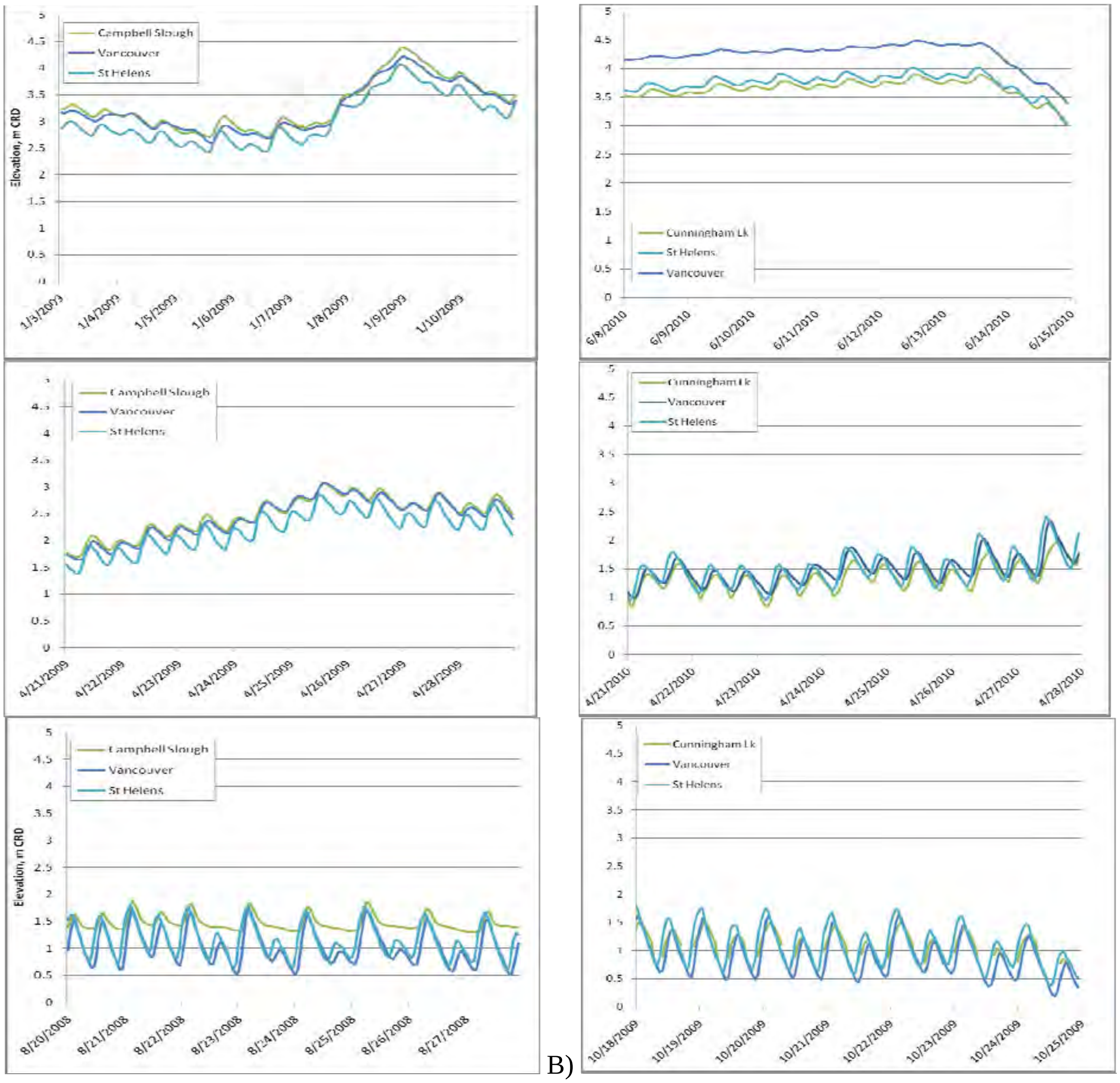


Figure 45. Daily mean outflow (red) at Bonneville dam compared to the 10-year average (green) for

the years 2005-2009. Data from Columbia River DART website (**http://www.cbr.washington.edu/dart/river.html**).

The water level data from in situ water level sensors were compared to data from the closest water level monitoring station in the main stem of the Columbia River to determine whether this longer term data could be used to hindcast water levels at the sites for the monitoring years prior to collection of in situ data. The water elevation data was compared over time periods with varying water levels (Figure 46).



A) B)

Figure 46. Water level comparisons between on-site data and tidal station data during low, moderate, and high water periods at A) Campbell Slough and B) Cunningham Lake.

The water levels at the Campbell Slough site were most similar to the Vancouver tide station and the Cunningham Lake site was most similar to the St. Helens tide station (Figure 46). Likewise, the Franz Lake site was deemed most comparable to the Cascade Island station water levels, just below Bonneville dam. The low water times were the exception to the comparability, due to the higher elevations of the site channels compared to that of the stations. The greatest differences in water levels were the water levels below the elevations used in our analysis and therefore deemed acceptable for use in our analyses.

Vegetation Inundation Relationship

Inundation patterns at the site scale vary from year to year. We evaluated the percent of time that relevant elevations are inundated during the growing season (Figure 47) based on water level data from the tide stations. In general 2006 and 2008 were years with the highest percent of time inundated, 2005 and 2007 the lowest, and 2009 in the middle. The elevations relative to CRD shown in Figure 47 cover the range of elevations found at the sites, with the elevation breaks of the dominant vegetation species provided in Table 30.

The greatest differences in the percent of time inundated (almost 30%) between years are at the elevations of 1.7 m and greater and therefore would primarily affect *P. arundinacea* (reed canary grass) growth. While at the 1.2 m elevation and lower a difference of only 10% of time inundated is apparent between years. Figure 43 and Figure 44 show that in fact the years 2006 and 2008 had lower percent cover of all three dominant species at Cunningham Lake and Campbell Slough at all elevations (ignoring 2007 at Campbell Slough, where the confounding factor of cows affected the cover).

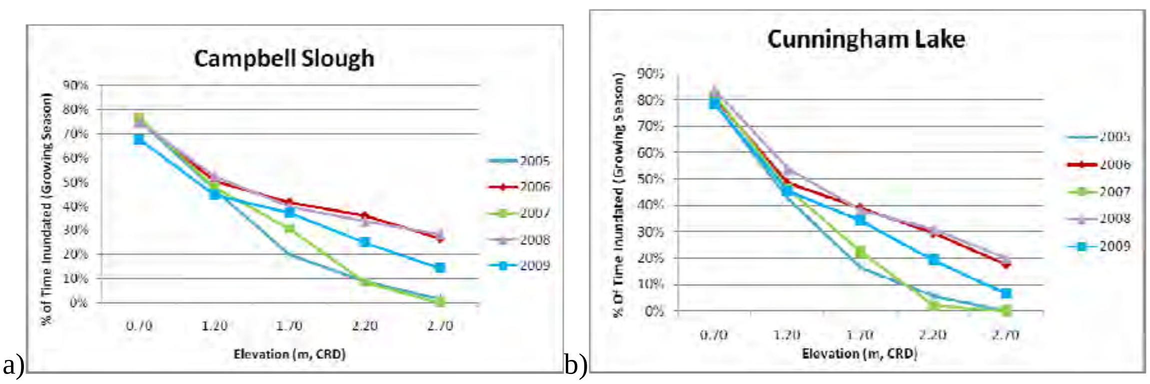


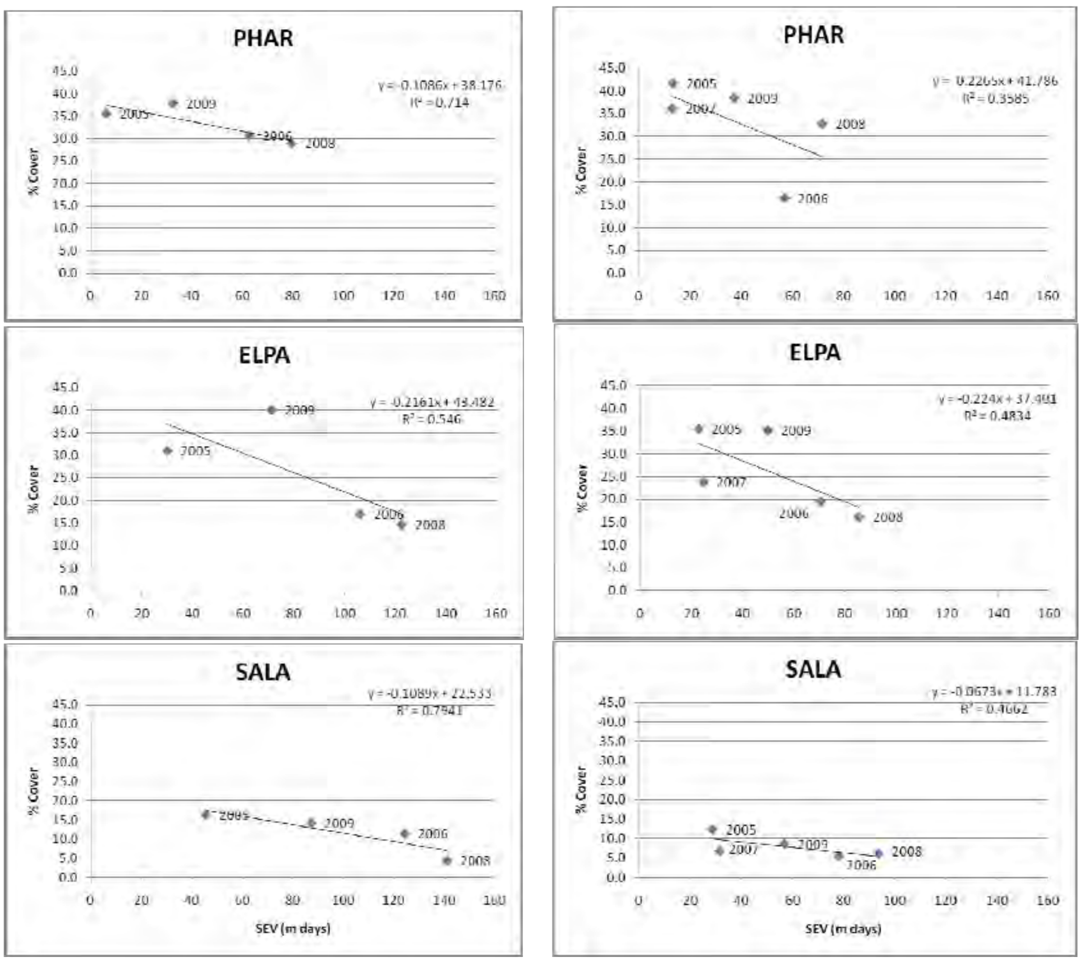
Figure 47. Percent of the time during the growing seasons of 2005-2009 when the elevations within the wetland are inundated at (a) Campbell Slough and (b) Cunningham Lake.

Inundation (SEV)

To evaluate the percent cover variation in the lower elevation species (SALA and ELPA) we evaluated the variation in magnitude of inundation, not just the timing. To do this we calculated the sum exceedance value (SEV) index to determine how much water was present at the average vegetation community elevations during each growing season using the equation presented in the Methods. This also allowed comparisons of the inundation regime between sites and between years.

In general, the variation in SEV is greater at Campbell Slough than at Cunningham Lake, ranging from 6 m/growing season (m/gs) at the average PHAR elevation in 2005 to 142 m/gs for the 2008 SALA elevation. The range at Cunningham Lake in the same vegetation zones and years ranged from 13 to 93 m/gs. This difference is likely caused by the reduced hydrologic range due to the greater distance from the main channel at Cunningham Lake (6.5 km) compared to Campbell Slough (1.4 km). This also could explain the smaller elevation range at Cunningham Lake (Table 30). In contrast, the Franz Lake SEV index was considerably higher, with a low of 105 in 2009 at the PHAR elevation and a high of 256 in 2008 at the lowest elevation. Here the explanation is likely the greater fluvial influence. The narrow geomorphology of the river at this site coupled with the proximity to the dam results in much greater inundation during the spring freshet, however the water drops to similar levels as the other sites after the high water period.

The response of vegetation cover to the variable annual inundation regime, represented by the SEV index, is shown in Figure 48. At Campbell Slough 2007 was not included in the analysis due to the reduced cover caused by cows. At this site, much of the reduction in cover can be explained by increased inundation during the growing season (r^2 between 0.55 and 0.79). The response is not so strong at Cunningham Lake (r^2 between .36 and .48), perhaps due to the reduced hydrologic variability at the site. The same responses in cover were also noted at Franz Lake; however, a regression analysis was not feasible due to the limited 2-year data collection period. These findings provide confirmation of the working hypothesis that vegetation communities in the LCRE respond to interannual hydrologic variability of the river.



A) Campbell Slough **B) Cunningham Lake**

Figure 48. Percent cover of the three dominant species compared to the sum exceedance value (SEV) at the average elevation of those species for 2005-2009 at A) Campbell Slough and B) Cunningham Lake.

Discussion

Overall, our results from the temporal analysis indicated that there is a reduction in the cover of the dominant species in response to increases in inundation. To date, the sites evaluated appear to be resistant to the levels of inundation variation measured during the years of study (2005-2009). One exception may be the cover of wapato (*S. latifolia*) at Cunningham Lake, which covered extensive areas of the site in 2005 and has not since then (Figure 49). One hypothesis is that the plants at this site are at the lower end of their elevation range. The 2005 photo (Figure 49) shows plants in the shallow flats of the “Lake,” whereas the flats are bare in the other years. The plants on the flats in 2005 are below the lowest elevations measured in our surveys. The low elevation coupled with the high inundation in years 2006 and 2008 may have resulted in water levels during the growing season exceeding the inundation or temperature tolerance of the plants and from which they have not yet recovered.

Figure 49. Cunningham Lake photos from 2005-2009 (2006 was not available). Photos all taken during the period between July 18 and July 26.

Changes as described at the Cunningham Lake site have led us to consider the potential effects of changing inundation magnitude and timing throughout the estuary resulting from changes in climate. The Climate Impacts Group at the University of Washington recently published predictions for future hydrologic changes in the Columbia River basin due to the effects of climate change (Elsner et al., 2010). Part of their analysis simulated discharge at The Dalles dam (rkm 304, the next dam upriver from Bonneville dam) for the historic period (1916-2006) and three future periods based on historic and projected precipitation, temperature, solar radiation, wind, and vapor pressure deficit (Figure 50). The simulated future flows show a decrease in peak flow in the late spring and early summer and an increase in cool season flow connected with reduced snowpack (Elsner et al., 2010). Historic records are also available from this station from 1880 to present and provide a means of comparing simulated natural flows to the actual flows that have occurred in the regulated River (USGS website: http://waterdata.usgs.gov/or/nwis/). Figure 50 shows the simulated (A) and historic (B) average monthly flow from 1915-2006 as well as 30 year increments for the past and future time frames. From the historic data, we can see that there has been a steady decline in peak flows and an increase in cool season flows, particularly for the most recent periods since 1970.

A) B)

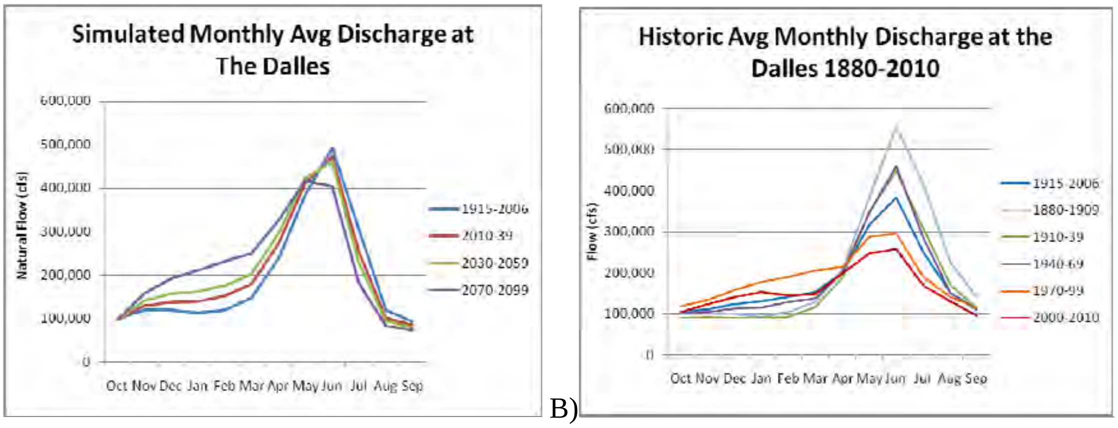


Figure 50. A) Simulated natural flow and B) actual historic flows downstream of The Dalles dam.

Based on the projected changes to the hydrograph and the changes already observed over the past 100 years we estimated what flows might look like in the future (Figure 51) by calculating the percent of change from simulated historic flows to future flows. Notable differences are increases in the cool season flows, decreases in the late spring peak flows, and a decrease in the fall low flows. The effect of these changes on wetland ecosystems in the LCRE is difficult to predict, however the patterns will likely result in increased inundation time during the growing season and potentially a reduction in vegetative cover at existing emergent marsh sites in the reaches of the River that are fluvial dominated.

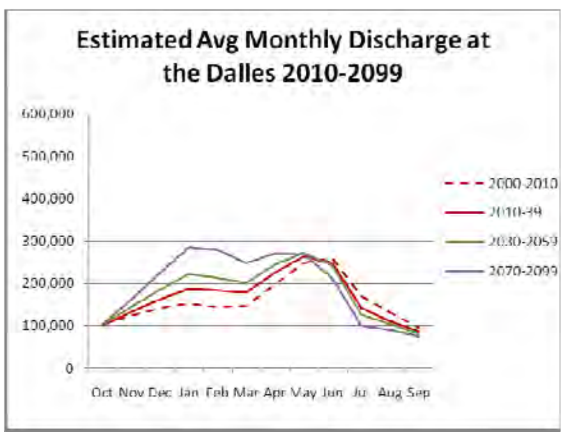


Figure 51. Estimated flows downstream of The Dalles dam based on historic records and climate change projections.

Conclusions and Recommendations

Implications from the multi-year study indicate that perhaps restoration design should account for potential increases in growing season inundation due to changes in the hydrograph resulting from climate change. Likewise, the findings of this research are directly applicable to restoration planning by informing prioritization of restoration sites relative to their position in the landscape and along the longitudinal gradient of the River. Future analyses will stem from the results described in this report and will specifically focus on evaluating spatial patterns in hydrology, structural morphology, and vegetation and how these patterns relate to fish access and use of these ecosystems. Further analysis on the implication of climate change on these shallow water habitats is needed.

9.0 Multi-Year Trends in Water Quality Data at Campbell Slough

For three years, (2007-2010) USGS deployed a continuous water-quality monitor at Campbell Slough in the Roth Unit of the Ridgefield National Wildlife Refuge. This site in Reach F has been sampled for vegetation since 2005 (PNNL) and for fish since 2007 (NOAA Fisheries). 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 these monitors 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. The melting of the large snowpack in the basin in 2008 caused extremely high water levels in mid-May and in to June. This led to delays in the deployment of the monitors 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 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 31(2009) and Table 32(2010).

Table 31. Average daily minimum, mean, median, and maximum water quality values by month, Campbell Slough, Ridgefield, WA, May 7–August 21, 2009.

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

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

Table 32. Average daily minimum, mean, median, and maximum water-quality values by month, Campbell Slough, Ridgefield, WA, April 1–July 30, 2010.

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

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	-

2010		April	May	June	July	August
	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	-

Temperature

In-stream temperature ranged from 10.5 to 34.4 degrees Celsius during the 2009 monitoring period. Continuous temperature data are shown in Figure 52. The seven-day maximum temperature ranged from 15.6 to 31.9° C, averaging 23.3° C. The Washington seven-day maximum standard of 17.5° was exceeded for the entire time period, except seven days in May (Figure 53).

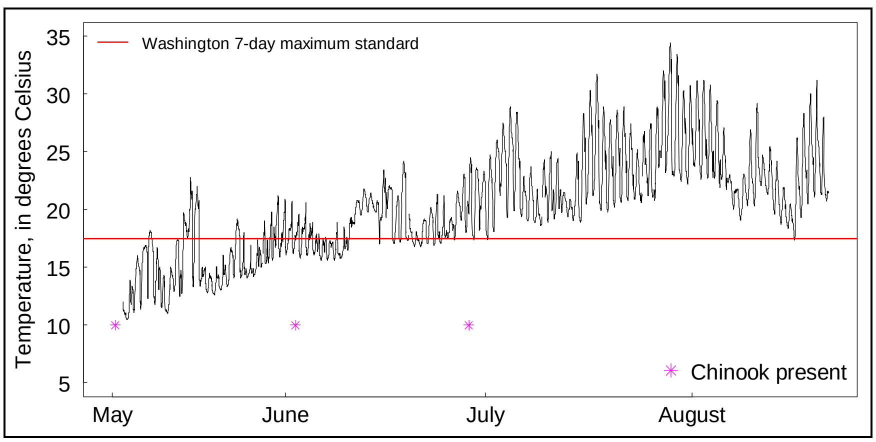


Figure 52. Graph of continuous temperature data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009. The Washington weekly maximum temperature standard is shown in red.

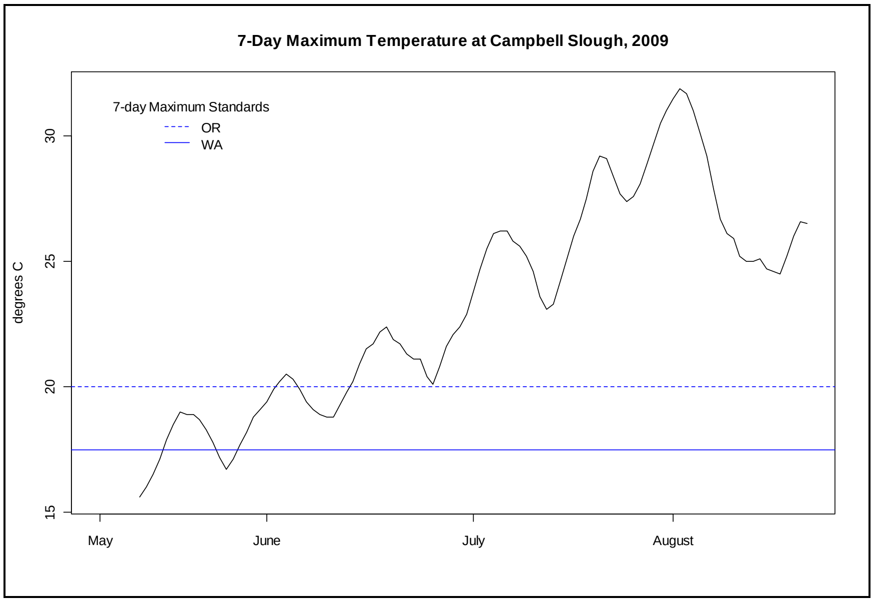


Figure 53. Graph of weekly maximum temperature data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009. Oregon and Washington weekly maximum temperature standards are shown in blue.

Water temperature ranged from 7.8 to 25.6° C during the 2010 monitoring period (Figure 54). It increased throughout the period, exceeding the Washington 7-day maximum temperature standard of 17.5° C in mid-May and in late June through July (Figure 55). Nevertheless, Chinook salmon were found at the site on July 6.

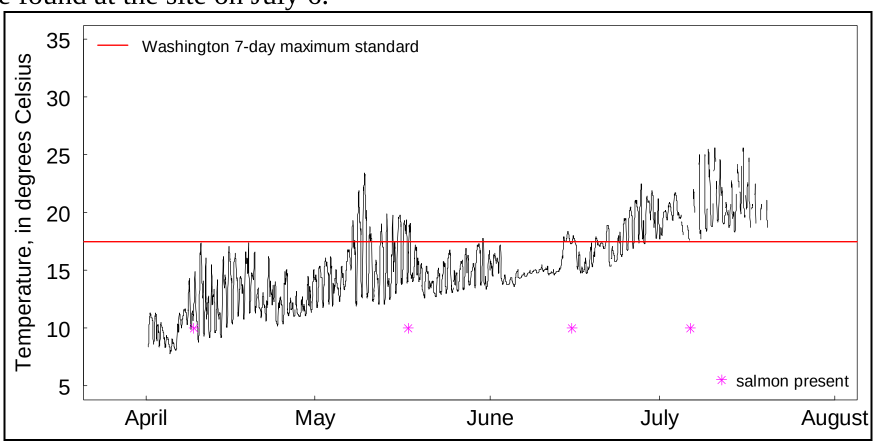


Figure 54. Graph of continuous temperature data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. The Washington weekly maximum temperature standard is shown in red.

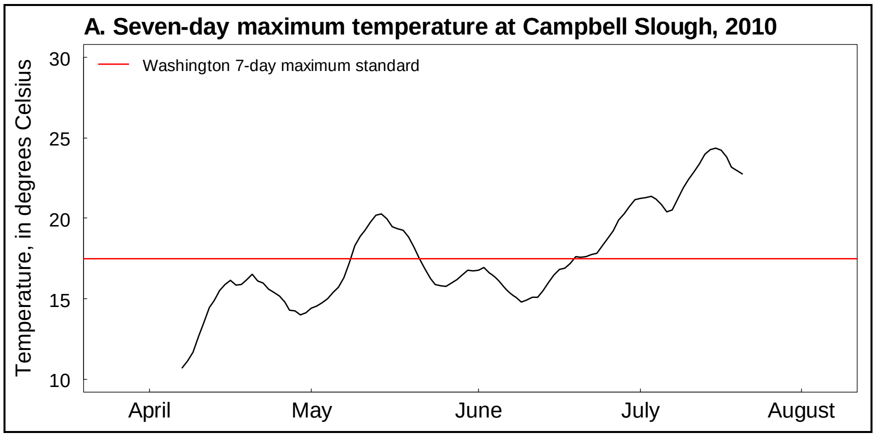


Figure 55. Graph of weekly maximum temperature data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. The Washington weekly maximum temperature standard is shown in red.

2009-2010 Comparison

Compared to 2009, the average daily median temperature in 2010 was within one degree 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 percent of days with data available during May to July (n=81) had 7-day maximum temperatures meeting the state standard, compared with 9 percent in 2009 (n=80).

pH

2009

In 2009, pH ranged from 6.9 to 10.0 standard units, averaging 8.2 (Figure 56). The daily minimum pH was below Oregon's standard of 7.0 on 3 days during the summer Figure 57); Washington's minimum standard of 6.5 was never violated. However, 50 days (most of July and August) had daily maxima exceeding both states' maximum standard of 8.5.

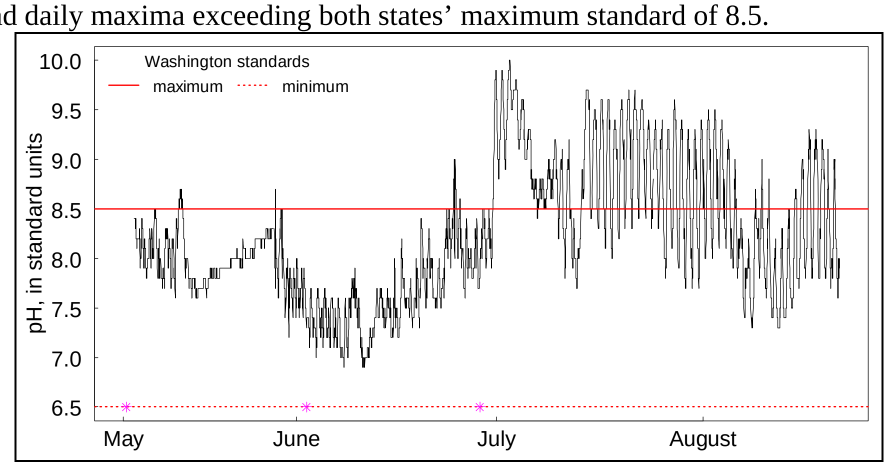


Figure 56. Graph of continuous pH data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009. The Washington minimum and maximum pH standards are shown in red. Salmon presence is shown pink diamonds.

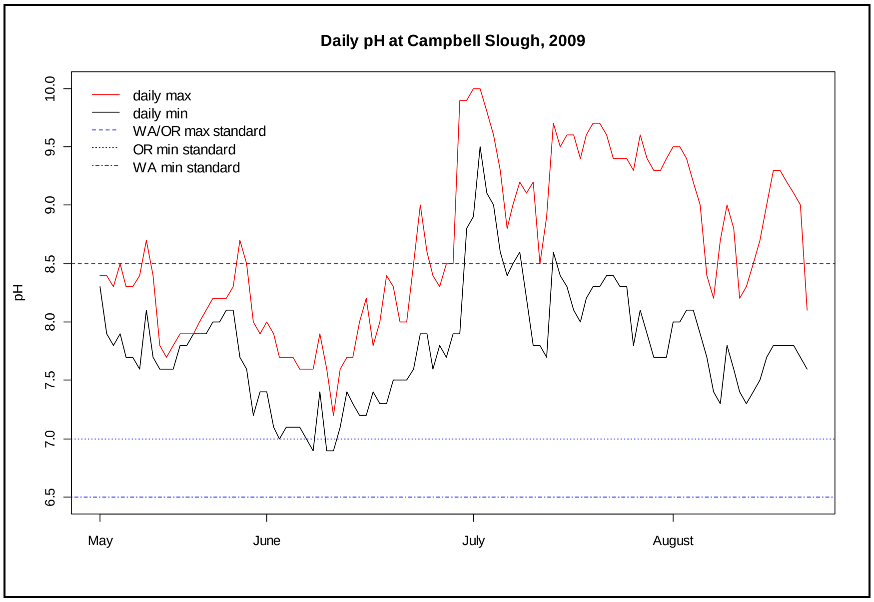


Figure 57**.** Graph of daily minimum and maximum pH data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009. Oregon and Washington pH standards are shown in blue.

2010

In 2010, pH ranged from 6.8 to 9.6 standard units, averaging 7.2 (Figure 58). The Washington maximum water-quality standard for pH was violated during mid-April through mid-May, when the daily maximum pH exceeded the state standard of 8.5. After peaking in April and May, pH decreased from mid-May through June and rose through early July (Figure 59). Washington's minimum pH standard was not violated during the 2010 monitoring period.

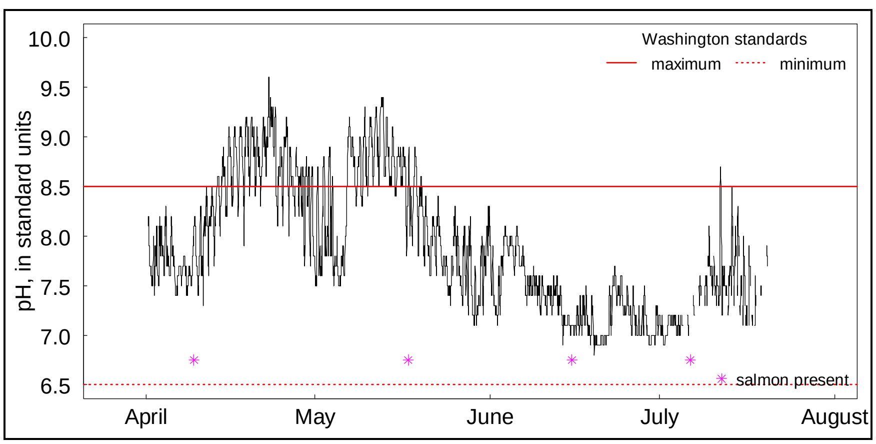


Figure 58. Graph of continuous pH data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. The Washington minimum and maximum pH standards are shown in red.

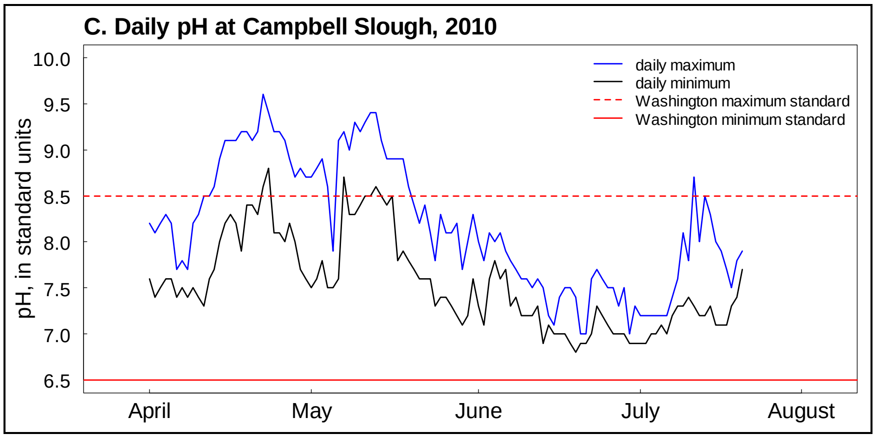


Figure 59. Graph of daily minimum and maximum pH data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. Washington pH standards are shown in red.

2009-2010 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; 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 60). Washington's daily minimum standard of 8.0 mg/L was violated 58% of days, primarily throughout July and August, but also during mid-May and early June (Figure 61).

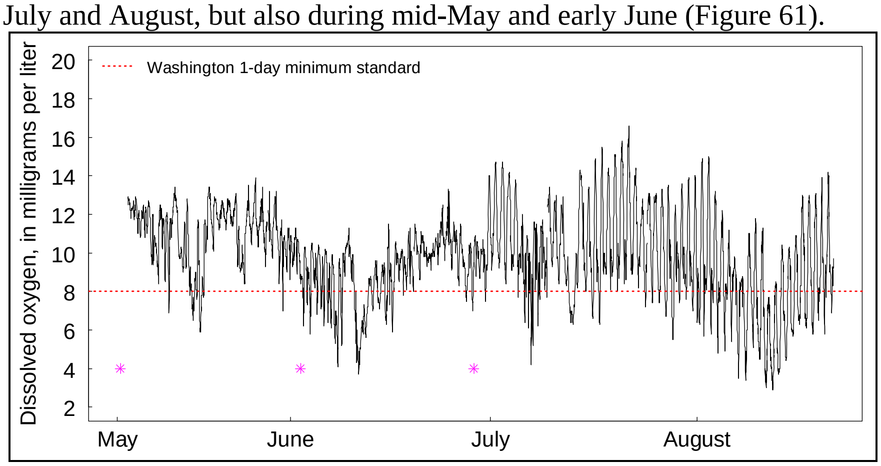


Figure 60. Graph of continuous dissolved oxygen data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009. The Washington daily minimum standard is shown in red. Salmon presence is shown pink diamonds.

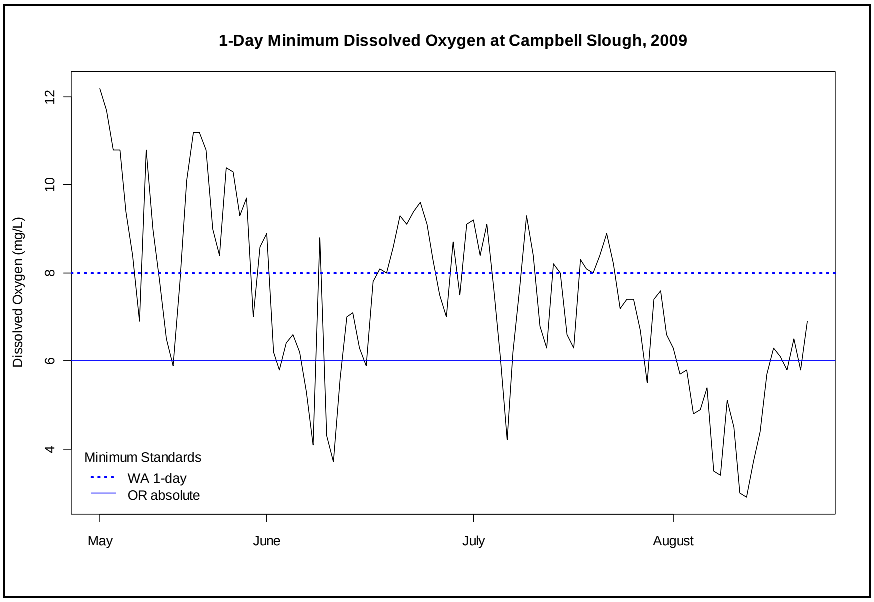


Figure 61. Graph of daily minimum dissolved oxygen data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009. Oregon and Washington dissolved oxygen standards are shown in blue.

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 62). 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 violated consistently from mid-June through July (Figure 63).

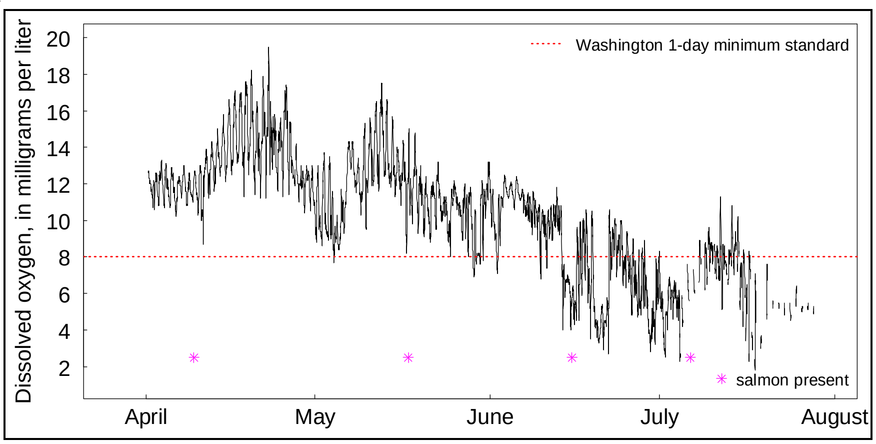


Figure 62. Graph of continuous dissolved oxygen data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. The Washington daily minimum standard is shown in red.

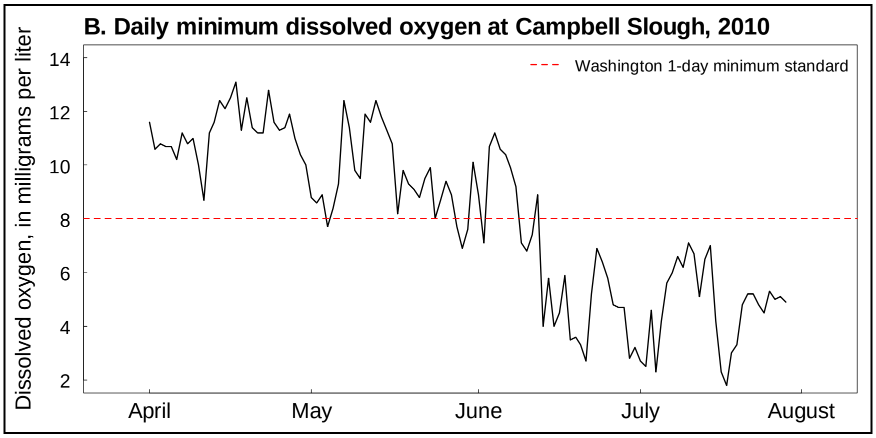


Figure 63. Graph of daily minimum dissolved oxygen data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010. The Washington daily minimum standard is shown in red.

2009-2010 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.

Specific Conductance

2009

Specific conductance ranged from 95 to 187 microSiemens per centimeter (μS/cm), averaging 143 μS/cm (Figure 64). The daily median specific conductance ranged from 136 to 153 μS/cm. State water quality standards do not exist for specific conductance.

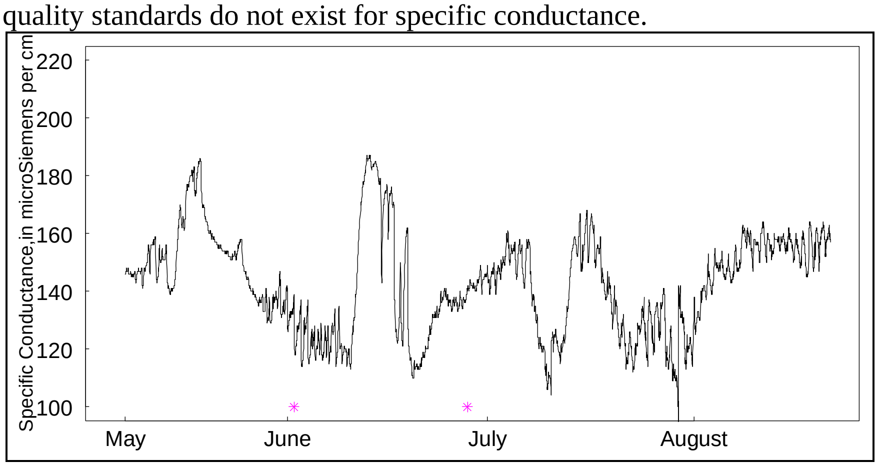


Figure 64. Graph of continuous specific conductance data from Campbell Slough, Ridgefield, WA, May 7–August 21, 2009.

2010

During 2010 monitoring, specific conductance ranged from 121 to 216 μS/cm and averaged 161 μS/cm (Figure 65). Average daily median specific conductance ranged from 144 to 177 μS/cm. Although it fluctuated during the monitoring period, it generally rose through April, then declined through June.

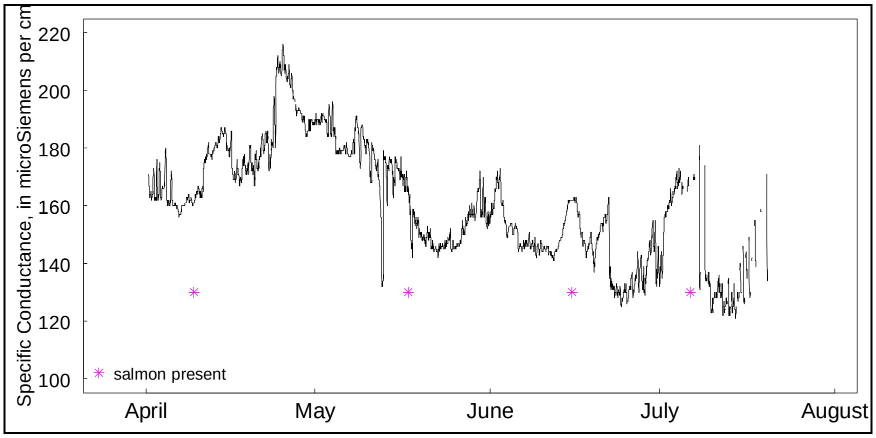


Figure 65**.** Graph of continuous specific conductance data from Campbell Slough, Ridgefield, WA, April 1–July 30, 2010.

2009-2010 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.

10.0 Multi-year Trends in Fish Monitoring at Franz Lake and Campbell Slough

Introduction

A major objective of the Lower Columbia River Estuary Partnership's Ecosystem Monitoring Project is to characterize tidal freshwater habitats and monitor salmon occurrence and health in those habitats in different reaches of the Lower Columbia River and Estuary (LCRE). As part of this project, between 2007 and 2010, NOAA Fisheries, USGS, PNNL, and the Estuary Partnership, with support from the BPA, have conducted multi-year monitoring at two sites, Campbell Slough on the Ridgefield National Wildlife Refuge in Reach F, and Franz Lake near Beacon Rock State Park in Reach H. Two of the goals of this monitoring are to understand differences in habitat characteristics and fish occurrence patterns between these reaches, and to understand temporal variability and year-to-year trends at the sites within each reach. NOAA Fisheries has focused on the following work elements:

- A survey of prey availability and habitat use by salmon and other fishes and data on fish habitat use in relation to physical habitat characteristics (monitored by PNNL and USGS).
- Taxonomic analyses of prey in salmon stomach contents in order to identify prey types consumed at different sites and times and to compare this with prey available in the habitat.
- Analyses of otoliths collected from juvenile Chinook salmon at the sites for determination of growth rates.
- Analyses of biochemical measures of growth and condition (e.g., lipid content) for juvenile Chinook salmon collected at the sites.
- Identification of genetic stock for juvenile Chinook salmon collected at the sites.
- Chemical analyses of stomach contents, bodies and bile from juvenile Chinook salmon collected from the sampling sites, as an additional indicator of habitat quality. This work was performed in addition to work elements specified by LCREP and BPA and was conducted with NOAA Fisheries funds.
- Compilation of data and report preparation.

Methods

Between 2007 and 2010, NOAA Fisheries conducted surveys to monitor prey availability and juvenile Chinook salmon habitat occurrence at Franz Lake in Reach H and Campbell Slough, in Ridgefield Wildlife Refuge, in Reach F (Figure 66). Franz Lake was sampled in 2008 and 2009, while Campbell Slough was sampled in 2007, 2008, 2009, and 2010.

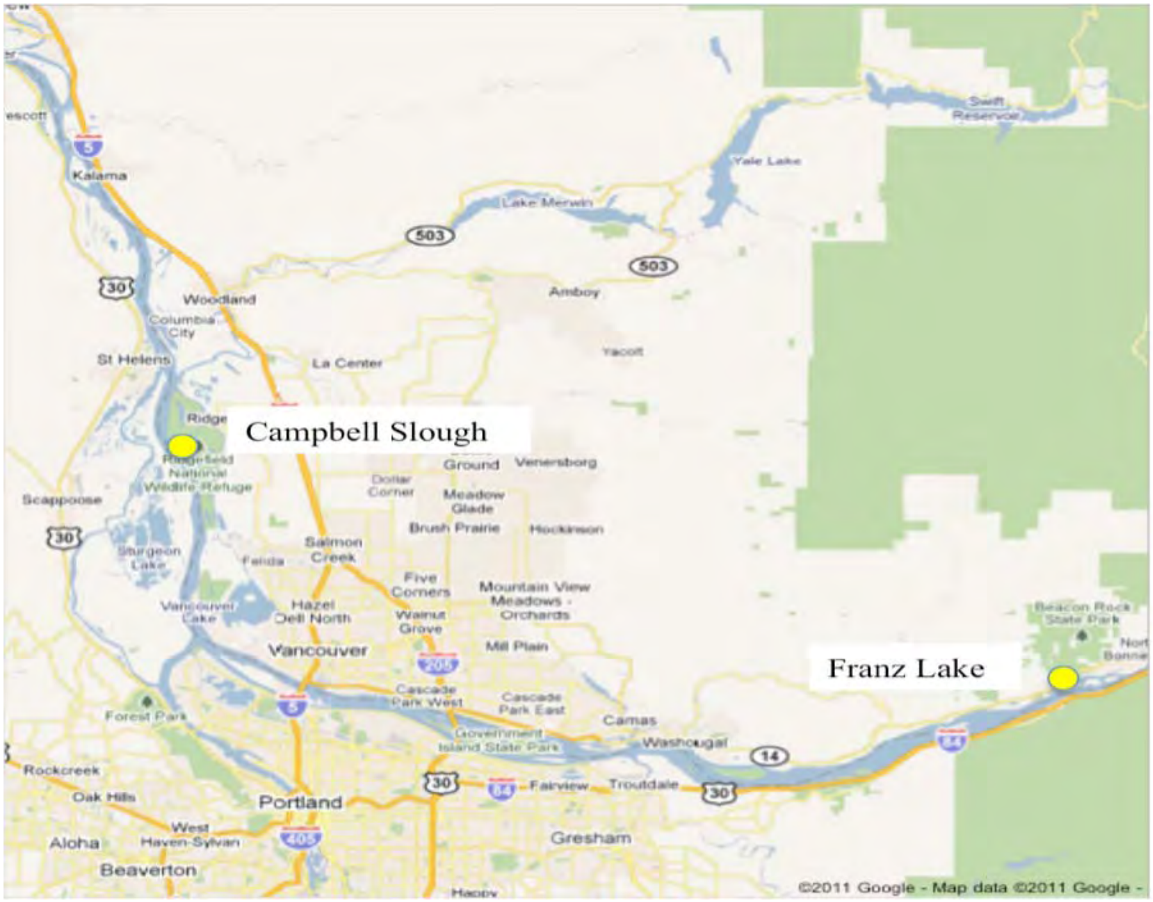


Figure 66. Locations of long-term monitoring sites at a) Campbell Slough in the Ridgefield National Wildlife Refuge in Reach F of the Lower Columbia River and b) Franz Lake in Reach H of the Lower Columbia River:

Site Name	Reach	Latitude	Longitude
Campbell Slough	F	45.783867°	-122.754850°
Franz Lake	H	45.600583°	-122.103067°

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 33. Fish were collected with a Puget Sound beach seine (PSBS) (37 x 2.4 m, 10 mm mesh size) or a baby beach seine (BBS) (10 x 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) in order to determine the proportions of marked fish (of known hatchery origin) and unmarked fish (potentially wild). Subsets of up to 30 juvenile Chinook (*Onchorhynchus tshawytscha*), coho (*Onchorhynchus kisutch*), and chum (*Onchorhynchus keta*) salmon from each set were measured (to the nearest mm) and weighed (to the nearest 0.1 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. 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 and dissolved oxygen were measured and tide condition were recorded at each sampling time as well. Samples for chemical analysis were held on dry ice and transported to NOAA's Northwest Fisheries Science Center laboratory, where they were stored frozen at 40^{o}C until analyses were performed. Stomach contents samples for taxonomic analysis were preserved in ethanol. Table 34 lists the numbers of samples collected from each site at each sampling event.

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.

NOAA Fisheries conducted the following types of invertebrate collections at Franz Lake and Campbell Slough between 2008 and 2010; Table 35 lists the numbers of prey samples collected from each site at each sampling event.

1) Open water column Neuston tows (2-3 tows at each site at each sampling time). These tows collect prey available to fish in the water column and on the surface of open water habitats. For each tow, the net was towed for a measured distance of 100 m. Invertebrates, detritus, and other material collected in the net were sieved, and invertebrates were removed and transferred to a labeled glass jar or Ziploc bag. The jar or bag was then filled with 95% ethanol so that the entire sample was covered.
2) Emergent vegetation: Neuston tows (2-3 tows at each site at each sampling time). These vegetation tows collect prey associated with emergent vegetation and available to fish in shallow areas. For each tow, the net was dragged through water and vegetation at the river margin where emergent vegetation was present and where the water depth was < 0.5 m deep for a recorded distance of 10 m. The samples were then processed and preserved in the same manner as the open water tows.
3) Terrestrial vegetation: Terrestrial sweep netting (3 collections at each site at each sampling time). With this type of sampling, terrestrial invertebrates that are associated with riparian vegetation and may be prey for fish in these habitats were sampled. For these samples, insects were collected using a sweep net along a transect of a recorded distance of at least 5 m along the river margin where vegetation was present. Transects were parallel to the bank and approximately 3 m from the water's edge. The net was swept through the vegetation for the length of the transect and for ~0.5 m on either side once thoroughly. Insects were transferred from the net into labeled plastic bags or jars and preserved as described above in 95% ethanol. Terrestrial vegetation samples were

collected in 2008 only. Preliminary analyses suggested these were less representative of salmon diets than the Neuston tow samples, so samples were archived for future analysis.

Table 33. Fishing attempts made at Franz Lake and Campbell Slough from 2007-2010. The Puget Sound beach seine was used to fish all of the sites except as indicated.

Site Name	Date	Fishing attempts	Comments
Franz Lake 2008	4/16/08	3	
	5/14/08	3	
	6/9/08	0	Site was not fishable due to extremely high water
	7/22/08	3	
	8/4/08	3	Baby beach seine used
Franz Lake 2009	4/9/09	3	
	5/5/09	2	
	6/2/09	0	Site was not fishable due to extremely high water
	7/1/09	3	
	7/28/09	3	
	8/5/09	0	Site was not fishable due to low water level
Campbell Slough 2007	5/4/07	9	High number of tows to collect samples for stomach chemistry and bile
	5/18/07	3	
	6/1/07	5	
	6/13/07	5	
	6/18/07	2	
	7/19/07	3	
Campbell Slough 2008	4/17/08	3	
	5/16/08	3	
	6/9/08	0	Site was not fishable due to extremely high water
	7/21/08	3	
	8/2/08	3	Modified block net used
Campbell Slough 2009	4/9//09	0	Site was not fishable due to extremely high water
	5/5/09	2	
	6/2/09	1	
	6/28/09	3	
	7/27/09	3	
	8/25/09	3	
Campbell Slough 2010	4/9/10	3	
	5/7/10	1	Fishing limited to one set because of take limits
	6/5/10	1	Fishing limited to one set because of take limits
	7/6/10	3	
	8/2/10	3	

Table 34. Samples collected from juvenile salmon at Franz Lake and Campbell Slough 2007-2010.

Site	collection date	Genetics (individual)	Otolith (individuals)	bile	diet	stomach chemistry	body chemistry
Franz Lake	4/16/08	33	33	1*	15	1*	33
	5/14/08	7	7	1*	7	0	7
	5/4/09	8	8	0	8	0	8
Campbell Slough	5/4/07	30	30	2*	0	1*	30
	6/1/07	19	19	2	8	10	88
	6/13/07	0	29	26	10	20	9
	4/17/08	6	6	1*	6	0	6
	5/12/08	4	0	1*	4	0	4
	5/16/08	33	33	1*	15	1*	33
	5/4/09	31	31	1*	10	21	31
	6/1/09	25	25	1*	9	15	25
	6/28/09	3	0	0	0	0	0
	4/8/2010	12	11	0	12	0	12
	5/7/2010	25	25	0	25	0	25
	6/15/2010	18	17	0	18	0	18
	7/6/2010	15	15	0	15	0	15

*Composite samples typically containing material from 10-20 individuals

Table 35. Terrestrial and aquatic prey samples collected at the 2008 Ecosystem Monitoring sites in the Lower Columbia Estuary

	open water						emergent vegetation						terrestrial sweepnet						total
site	April	May	June	July	Aug	total	April	May	June	July	Aug	total	April	May	June	July	Aug	total	
Franz Lake																			
2008	3	3	0	3	3	12	3	3	0	0	0	6	3	0	0	4	3	10	28
2009	3	2	2	2	0	9	3	2	2	3	0	10	0	0	0	0	0	0	19
Campbell Slough																			
2008	3	3	0	3	3	12	0	3	0	0	0	3	3	3	0	3	3	12	27
2009	0	2	4	3	2	11	0	2	4	3	2	11	0	0	0	0	0	0	22
2010	0	2	2	2	3	9	0	2	2	2	3	9	0	0	0	0	0	0	18
Grand Total	9	12	8	13	11	53	6	12	8	8	5	39	6	3	0	7	6	22	114

Sample analyses

Lipid determination. —For lipid and chemical analyses, individual Chinook salmon bodies (carcass plus internal organs) were combined to produce composite samples consisting of 3-5 fish each from the same site, sampling time, genetic stock, and origin (wild vs. hatchery). 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 contents samples were analyzed by gas chromatography–mass spectrometry (GC–MS) for PCB congeners, PBDE congeners, DDTs, DDT isomers, and other organochlorine (OC) 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; 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, 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 these compounds are metabolized by fish and accumulation in tissues is very limited.

Polycyclic aromatic hydrocarbon metabolites in Chinook salmon bile. —Salmon exposure to PAHs was assessed by measurement of PAH metabolites in 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 uL) for analyses. Bile samples were analyzed for metabolites of PAHs using a HPLC–fluorescence detection method described by Krahn et al. (1986). Chromatograms were recorded at the following wavelength pairs: (1) 260–380 nm, where several 3–4- ring compounds (e.g., PHN) fluoresce; and (2) 380–430 nm, where 4–5-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 PHN equivalents/g bile) and BaP (ng BaP 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 values and higher protein content than fish that are feeding constantly and excreting bile more frequently (Collier and Varanasi, 1991).

Genetic stock identification

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 in 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.

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

Fish species diversity was calculated using the Shannon-Weiner diversity index (Margaley, 1958):

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

Where

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

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

Pi = 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 in Roegner et al., 2009, with fish density reported in number per 1000 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:

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

Statistical Analyses. Multiple regression was used to examine the effects of fish type (wild vs. hatchery), site of capture, month of capture, and year of capture on length, weight, and condition factor. Analysis of variance (ANOVA) was used to examine differences among sites and years in variables such as lipid content, growth rate, length, weight, and contaminant concentrations. Chi-square analysis and contingency tables were used to examine differences among sites and years in proportions of fish species present. Analyses were conducted with JMP statistical package.

Results

Water Temperature and other Physical Factors

At Campbell Slough, in all sampling years, water temperature increased steadily throughout the sampling season from 9-17°C in April and May to over 20°C in July and August (Figure 67). In 2007, 2008, and 2010, the temperature ranges at the site were similar, although it was sampled only from May through July in 2007. In 2009, higher maximum temperatures were reached, with temperatures of 25-28°C in July and August. At Franz Lake, the water temperature range was similar to that observed at Campbell Slough, increasing from about 10°C in April to over 25°C in August (Figure 67). Summer temperatures were somewhat higher in 2009 than in 2008, with a maximum temperature in 2009 of 28°C. At both sites, in all sampling years, the water temperature in July and August was above the favorable range for juvenile salmonids (Marine and Chech 1998; McCollough 1999).

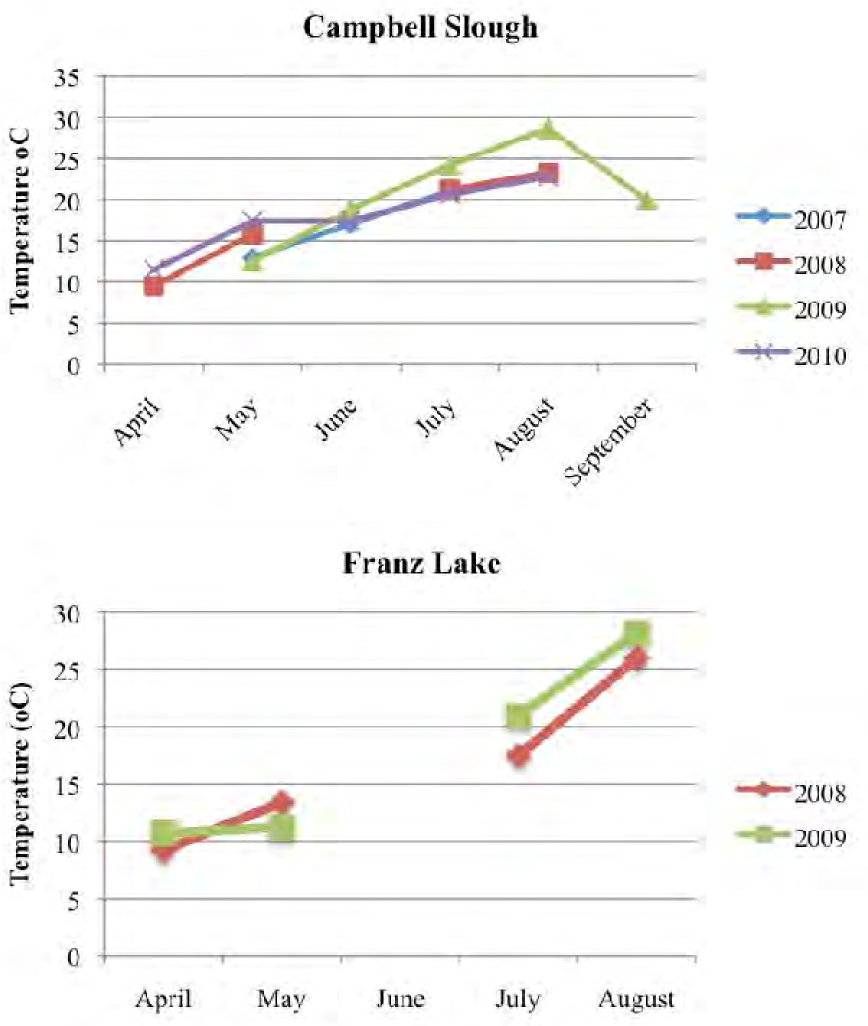


Figure 67. Seasonal water temperatures at Campbell Slough and Franz Lake.

Year-to-Year Trends in Fish Use at Campbell Slough and Franz Lake

To assess long-term trends in fish habitat occurrence in various reaches of the estuary, Campbell Slough in Reach F has been sampled from 2007 through 2010. The 2007 fish sampling was conducted from early May through July whereas the 2008, 2009, and 2010 samplings were conducted from April through August. Due to higher and more variable water levels in 2008 relative to 2007, we could not sample the site in June 2008 and hence do not have fish data available for this month for comparison. The Franz Lake site in Reach H has so far been sampled for fish in 2008 and 2009. Both the 2008 and 2009 samplings were conducted from April through August.

Fish Community Characteristics.

Campbell Slough. In spite of difficulties with site access at certain months in certain years, our sampling overall 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 68). Three-spine stickleback and juvenile carp were the most abundant species in 2007, 2008, and 2010, together making up 77-79% of the total catch. In 2009, the most abundant species were juvenile carp and yellow perch, together accounting for 57% of total catch. Stickleback were still present, but accounted for only 12% of the total catch, as compared to 32-46% in other years. Juvenile Chinook salmon were captured in all three years; the percentage of total catch was 4.1% in 2007, 2.9% in 2008, 3.8% in 2009, and 3.0% in 2010 (Figure 68).

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 69). Non-native species accounted for 50-60% of the number of total number of species caught at Campbell Slough from 2007 to 2010 (Figure 69). 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 70).

Franz Lake. Our sampling overall showed that juvenile salmon and juveniles of other fish species were feeding and rearing at the Franz Lake site in 2008, and 2009 (Figure 68). 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, account for 69% of the total catch. Juvenile Chinook were captured in both years; the percentage of 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, with non-native species accounting for 31-37% of the total number of species caught (Figure 69). The Shannon-Weiner species diversity index decreased from 2 in 2008 to 1 in 2009 (Figure 70).

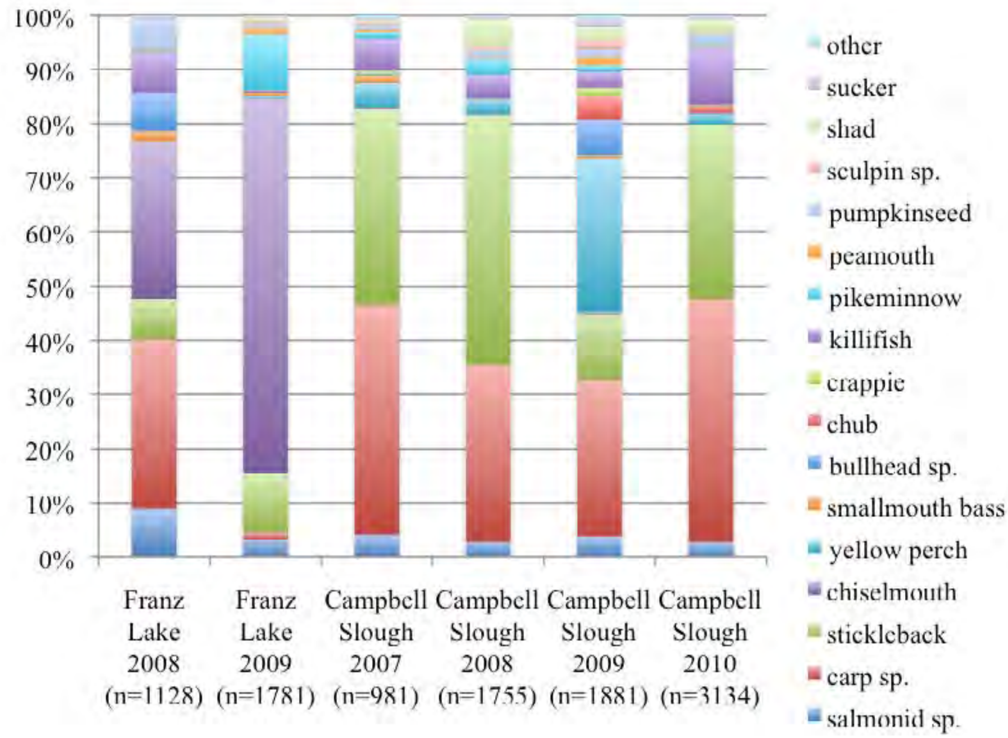


Figure 68. Species composition at Franz Lake and Campbell Slough by sampling year.

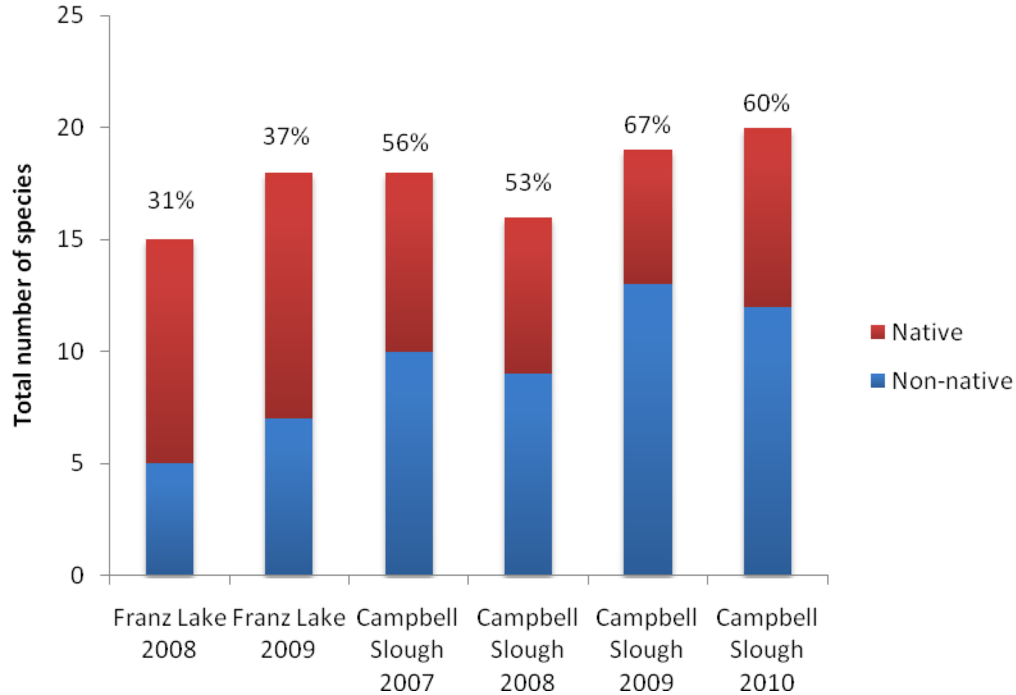


Figure 69. Number of species and proportions of native and non-native species captured at Franz Lake and Campbell Slough. The percentage of non-native species at each site for each year is shown above the bar graphs.

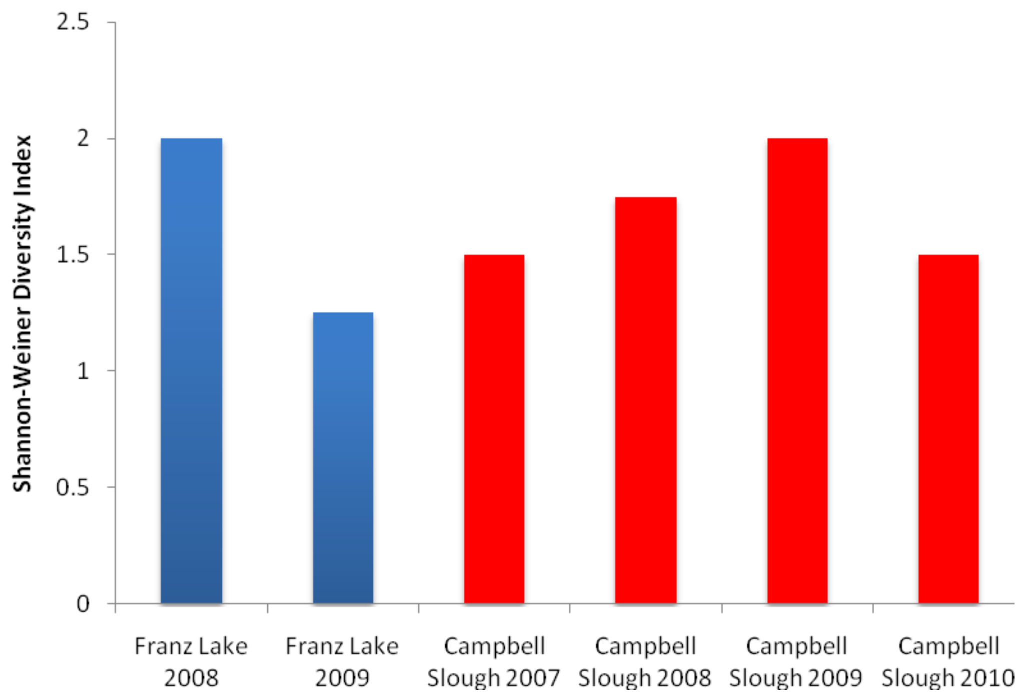


Figure 70. Shannon-Weiner species diversity at Fran Lake and Campbell Slough.

Salmonid species composition

Campbell Slough. In 2007, 2008, 2009, and 2010, Chinook salmon made up 97%, 98%, 100%, and 96%, respectively, of the juvenile salmonid catch at Campbell Slough (Figure 71). In 2007, we collected no chum salmon and only one coho salmon; in 2008, the opposite occurred as we collected only one chum salmon and no coho salmon. In 2009, neither coho nor chum salmon were collected while in 2010, Chinook and chum were collected. In 2007, 2008, and 2010, both hatchery (marked) and presumably wild (unmarked) Chinook salmon were found at the site in similar proportions. Hatchery fish accounted for 52% of the catch in 2007, 51% in 2008, and 58% of the in 2010 (Figure 71). In contrast, in 2009, 96% of Chinook captured were of hatchery origin. It is not clear why such a high proportion of the Chinook salmon captured 2009 were hatchery fish, although this could be related to the fact that all of our sampling took place in May and June, the months when hatchery fish are typically released.

Franz Lake. In 2008 and 2009, Chinook salmon made up 60%, and 34.5%, respectively, of the juvenile salmonid catch at Franz Lake (Figure 71). In both years, we also collected chum and coho salmon at this site. Chum made up 6% of the salmonid catch in 2008 and 1.7% of the salmon catch in 2009, while coho made up made up 34% of the salmon catch in 2008 and 58.7% of the salmonid catch in 2009. In 2009, in addition to salmon species, we also caught steelhead and cutthroat trout, which made up 3.4% and 1.7% of the salmonid catch, respectively. All chum salmon caught at Franz Lake were unmarked, presumably wild fish. However, significant proportions of both Chinook and coho salmon at the site were of hatchery origin. The proportion of hatchery (marked) Chinook salmon found at the site varied considerably from year to year. Hatchery 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 hatchery fish. 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, $p < 0.0001$).

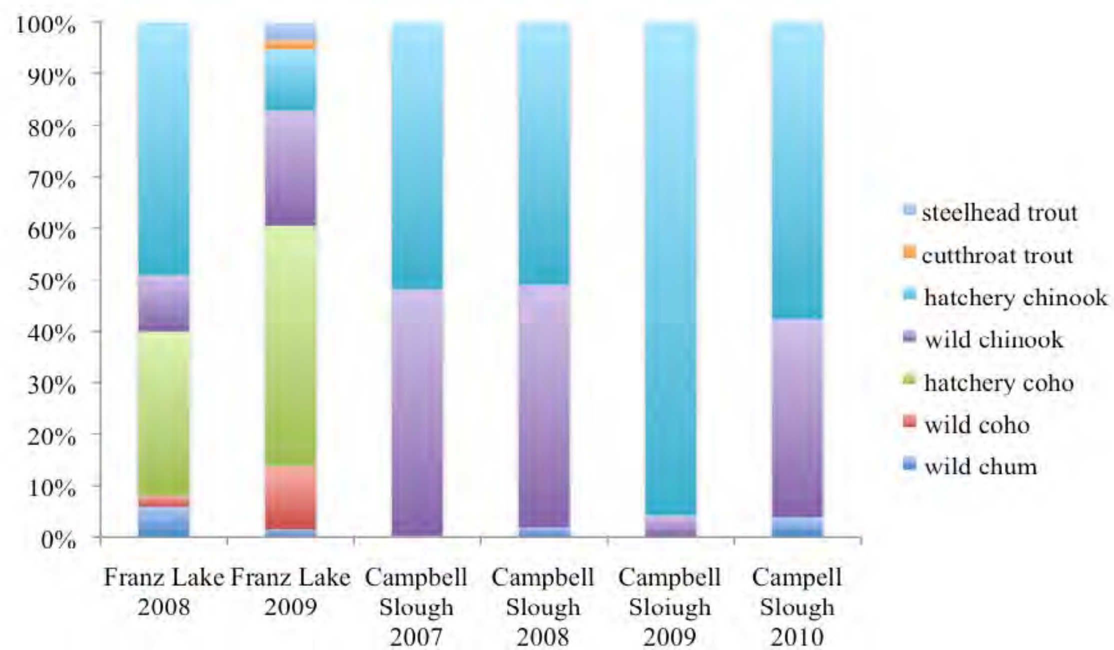


Figure 71. Percentages of salmonid species in catches at Franz Lake and Campbell Slough.

Genetic Stock Identification of Juvenile Chinook salmon

Campbell Slough. In 2007, 2008, and 2009, the majority of marked, hatchery-reared juvenile Chinook salmon from Campbell Slough were from stocks included in the Lower Columbia ESU (Figure 72). In all three years, the majority of the Lower Columbia ESU fish were from the Spring Creek Fall Group (~90%), with a smaller proportion (~10%) belonging to the West Cascades Fall group. In 2007, the remaining fish were from the Upper Columbia Summer/Fall stock, while in 2008, fish were present from the Upper Willamette Spring, West Cascades Fall, and Snake River Fall stocks. In 2009, only fish for Lower Columbia stocks (Spring Creek and West Cascades Fall groups) were collected. In 2007, wild juvenile Chinook at Campbell Slough came from a diverse array or stocks (Figure 72). The majority of fish were from the Upper Columbia Summer Fall stock (52%), and 32% were from Lower Columbia ESU stocks (Spring Creek Group Fall and West Cascades Fall). Other stocks present included Upper Willamette Spring, Snake River Fall, and Deschutes River Fall. In 2008, only a small number of wild fish were captured, and all of these were identified as Spring Creek Group Fall Chinook (Figure 7). In 2009, fish from Spring Creek and Deschutes fall Chinook groups were identified, with the Spring Creek Group making up 75% of the total (Figure 72).

Franz Lake. Hatchery Chinook salmon sampled from Franz Lake in 2008 were primarily Lower Columbia ESU stocks (78% of fish analyzed; Figure 72). The majority (72%) belonged to the Spring Creek Fall Chinook group, with a smaller proportion (6%) from the West Cascades Fall group. Additionally, about 22% of sampled fish were from Willamette River stocks. In 2009, all marked fish were from Lower Columbia stocks. Of these 86% were identified as Spring Creek Group Fall Chinook, and the remainder as West Cascades fall Chinook (Figure 72). Unmarked, presumably wild Chinook were also primarily from the Lower Columbia River ESU, with 75% of fish examined from the Spring Creek Fall Chinook group (Figure 72). An additional 25% of fish were from the Upper Columbia summer/fall Chinook group. However, the number of wild Chinook captured in 2008 was very small, and may not be sufficient to characterize wild 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.

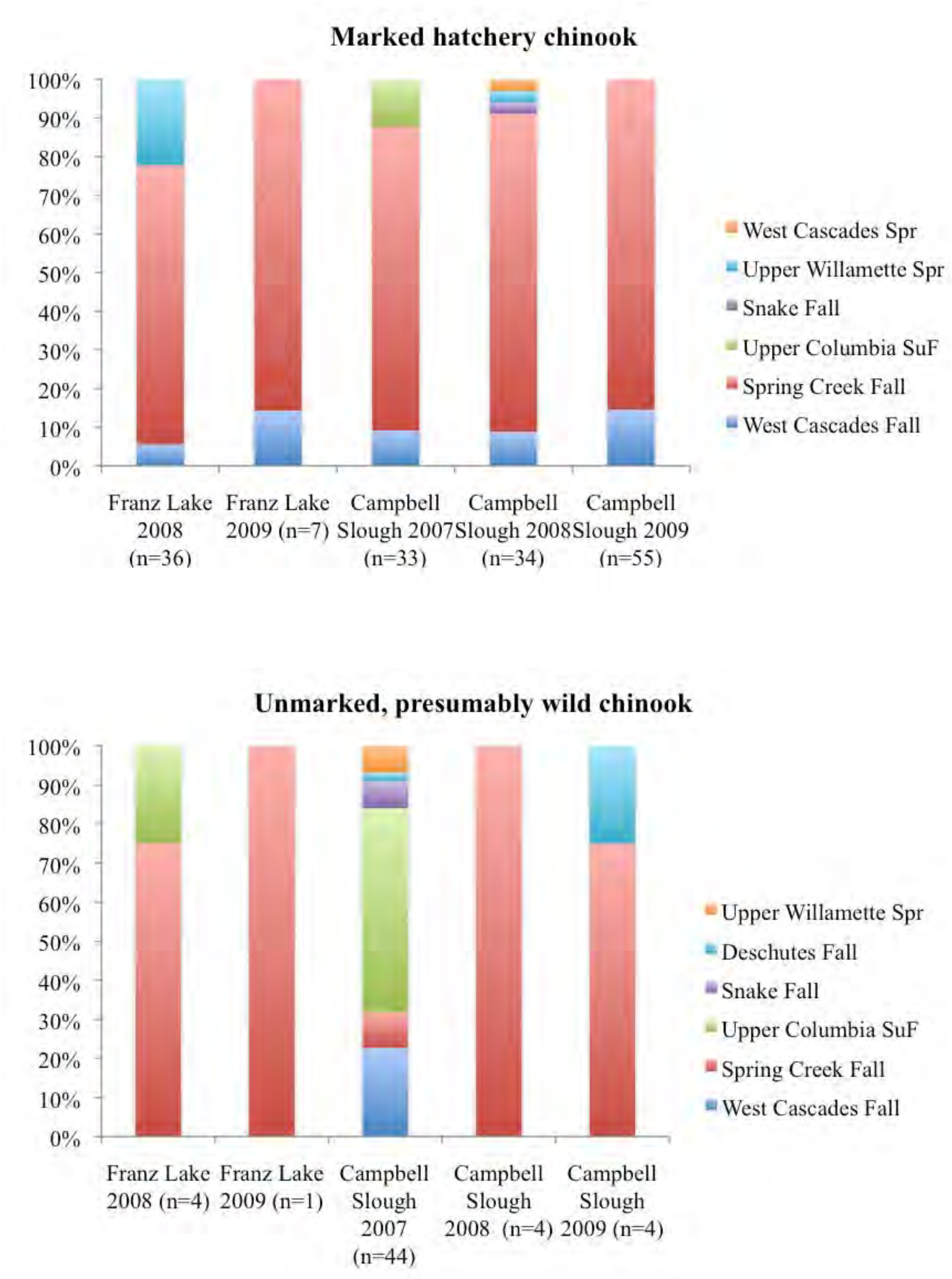


Figure 72. Genetic stock assignments for marked (hatchery) and unmarked (presumably wild) Chinook salmon from Franz Lake and Campbell Slough.

Salmonid seasonal habitat occurrence

Campbell Slough. In 2007, both unmarked, presumably wild juvenile Chinook and marked, hatchery origin Chinook were present from the start of sampling in May through June (Figure 73). In 2008, both wild and hatchery juvenile Chinook were present from the start of sampling in April through May; fish may have also been utilizing this site in June 2008, but no data are available because it was impossible to sample due to the high water levels. In 2009, both wild and hatchery juvenile Chinook were present from May through the beginning of July; they may have also been utilizing the site in April but it could not be sampled due to high water. In 2010, wild juvenile Chinook were present from April through July, while hatchery Chinook were present from May through July. No salmon were observed in later July or August in any of the four sampling years, possibly because of water temperatures exceeding 20°C all four years.

The number of Chinook captured per-unit-effort (CPUE) was generally highest in May. Over the course of the sampling period, CPUE decreased from 23.5 fish per 1000 m^2 in 2007 to 13 in 2008, then increased again in 2009 to 19 and in 2010 to 22.6. Coho CPUE was very low throughout the sampling period, ranging from 0 to 0.25.

Franz Lake. In both 2008 and 2009, unmarked juvenile Chinook were present at Franz Lake from the start of sampling in April through May. The pattern was the same for marked hatchery Chinook; fish were present in both April and May of 2008 and 2009, but were not observed after that. Both marked and unmarked Chinook may have also been utilizing this site in June, but no data are available because is was impossible to sample in either year due to the high water levels. We sampled, but did not encounter any juvenile salmon in July or August in either sampling year.

The number of Chinook captured per-unit-effort (CPUE) was lower in April 2009 than in April 2008, but in May CPUE was similar in both years. In April, CPUE was similar at Franz Lake and Campbell Slough, but in May and June, CPUE was much higher at Campbell Slough than at Franz Lake. Likely factors include high water levels, difficulty fishing the site, and inter-annual variation in populations.

The small numbers of chum salmon, steelhead trout, and rainbow trout that were collected at Franz Lake were all found in April. Of the Coho salmon collected, 97% were capture in May, and the remaining 3% in April.

For both coho and Chinook salmon CPUE declined from 2008 to 2009. For Chinook salmon CPUE was 22 in 2008 and 3 in 2009, while for coho salmon, it was 12.7 in 2008 and 4.9 in 2009. For chum salmon CPUE increased, from 2 in 2008 to 5 in 2009.

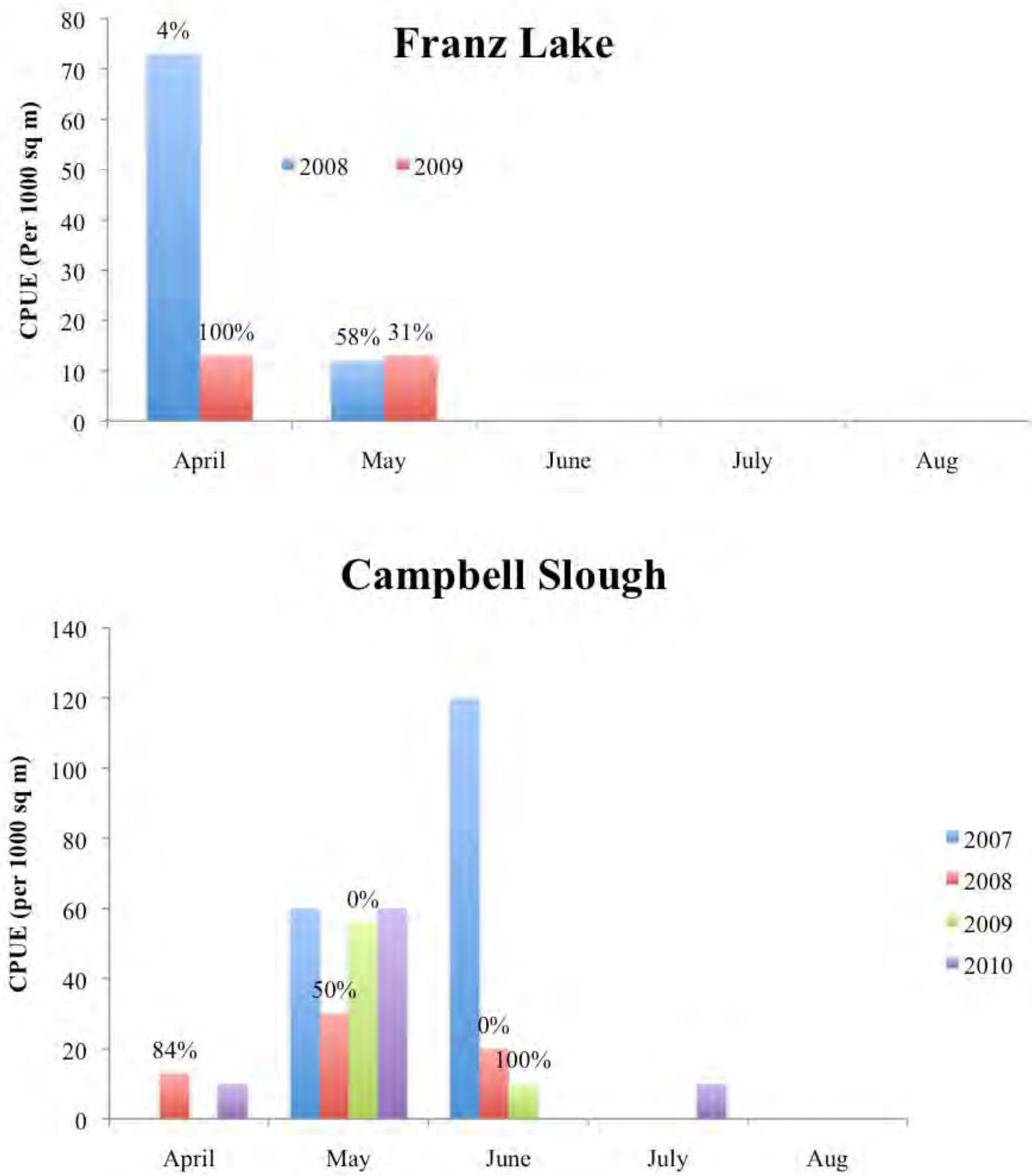


Figure 73. Chinook salmon catch per unit effort (CPUE) by month and year at Franz Lake and Campbell Slough. Percentages of hatchery fish in the catches are indicated above the bars representing CPUE. Percentages of hatchery fish are not indicated at Campbell Slough in 2007 because this was not noted for all fish captured in tows.

Salmon Length. Weight, and Condition Factor

Chinook salmon length, weight, and condition factor by month and year for Campbell Slough and Franz Lake are shown in Figure 73 to Figure 76. A multiple regression analysis including site of capture, month of capture, and fish origin (wild vs. hatchery) was conducted to determine which factors had the most effect on salmon length, weight, and condition. The effect of year of capture was considered separately for the two sites, as Franz Lake was sampled only in 2008 and 2009, whereas Campbell Slough was sampled from 2007 to 2010.

The results indicated that the significant factors affecting fish length and weight were fish origin and month of capture ($p < 0.0001$ for these factors for both length and weight). For length, r2 for the overall model = 0.52, $p < 0.0001$, while for weight, r2 for the overall model = 0.42, $p < 0.0001$. The model results indicated that hatchery fish were larger and weighed more than wild fish. Their mean length and weight were 83 mm and 6.1 g, as compared to 66 mm and 3.8 g for wild fish. Both length and weight tended to increase over the sampling season from April to July, a trend that was most clearly visible in wild fish. For fish condition, only month of capture was significant ($p < 0.0001$) , with fish condition generally increasing from April through June. For the overall model for K, r2 = 0.16, $p < 0.0001$. Length, weight, and condition distributions at Franz Lake and Campbell Slough were not significantly different once the effects of sampling time and fish origin had been taken into account.

At Franz Lake, fish length, weight, and condition were all lower in 2009 than in 2009 ($0.0154 < p < 0.0424$). At Campbell Slough, there were also significant differences in length, weight, and condition among the sampling years ($0.0001 < p < 0.0003$). Length, weight, and condition were all higher in 2007 and lower in 2009 than in other years.

These analyses suggest that fish size and condition followed the same general patterns at both sampling sites, after differences in proportions of wild and hatchery fish and sampling dates had been taken into account.

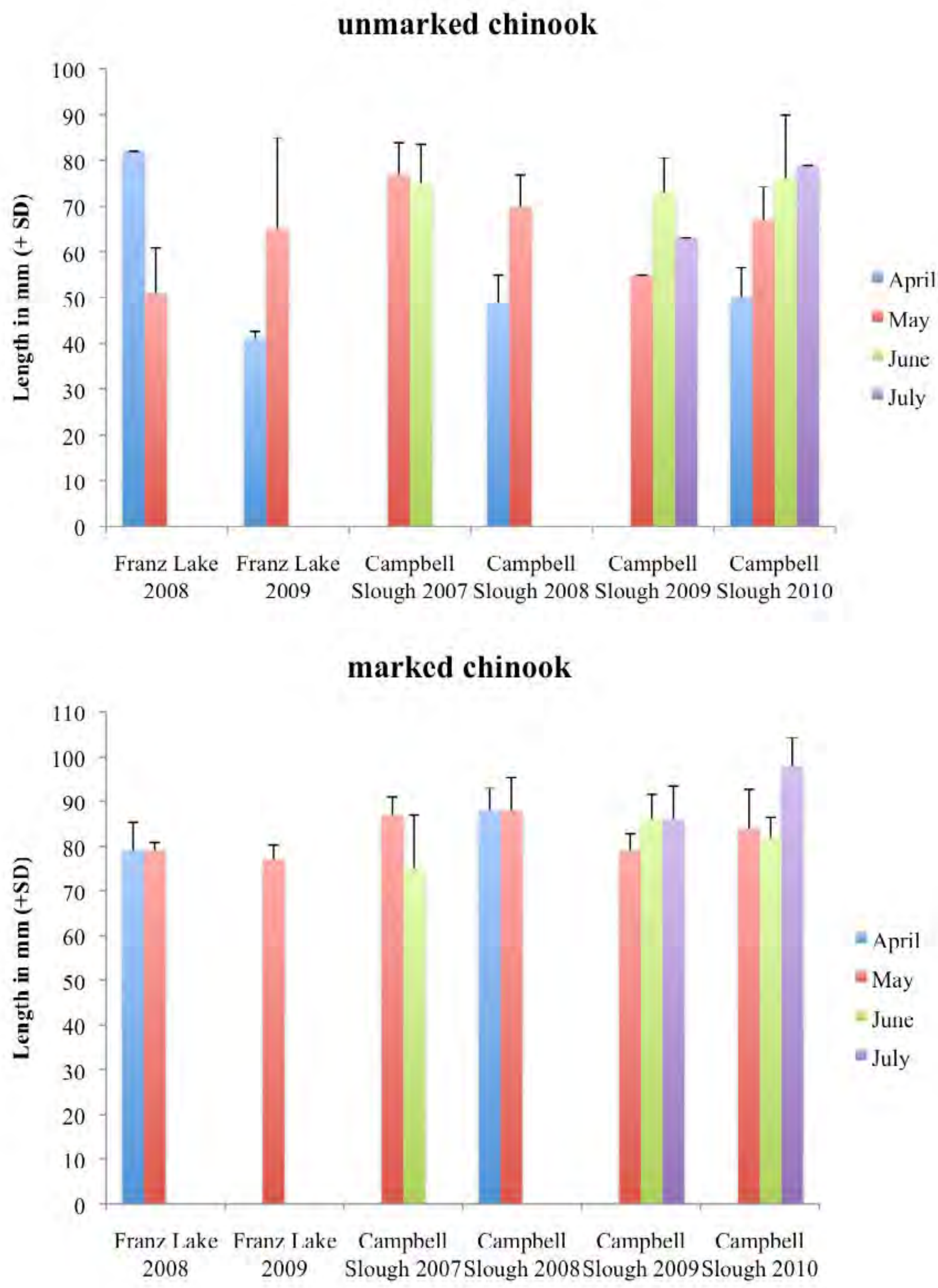


Figure 74. Lengths in mm of marked and unmarked Chinook from Franz Lake and Campbell Slough over time the sampling season.

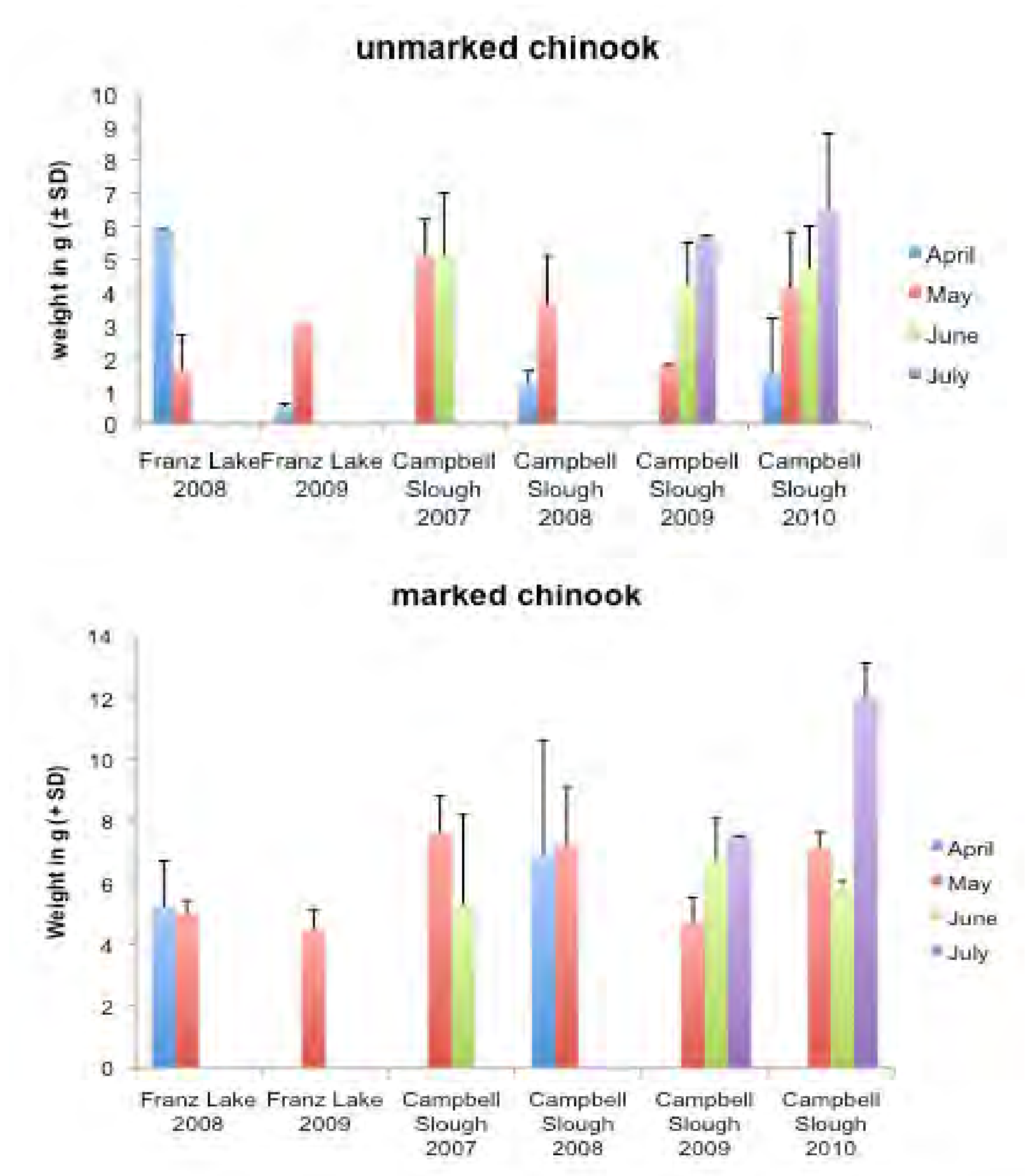
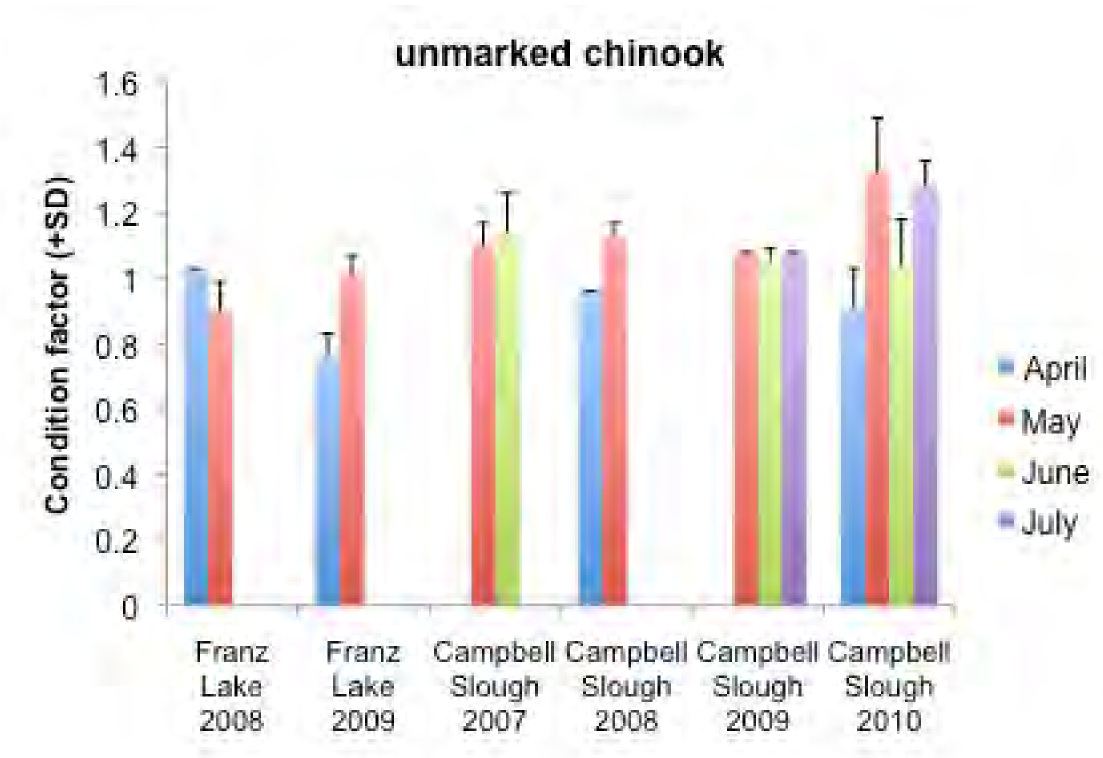


Figure 75. Weights in g of marked and unmarked Chinook from Franz Lake and Campbell Slough over time.



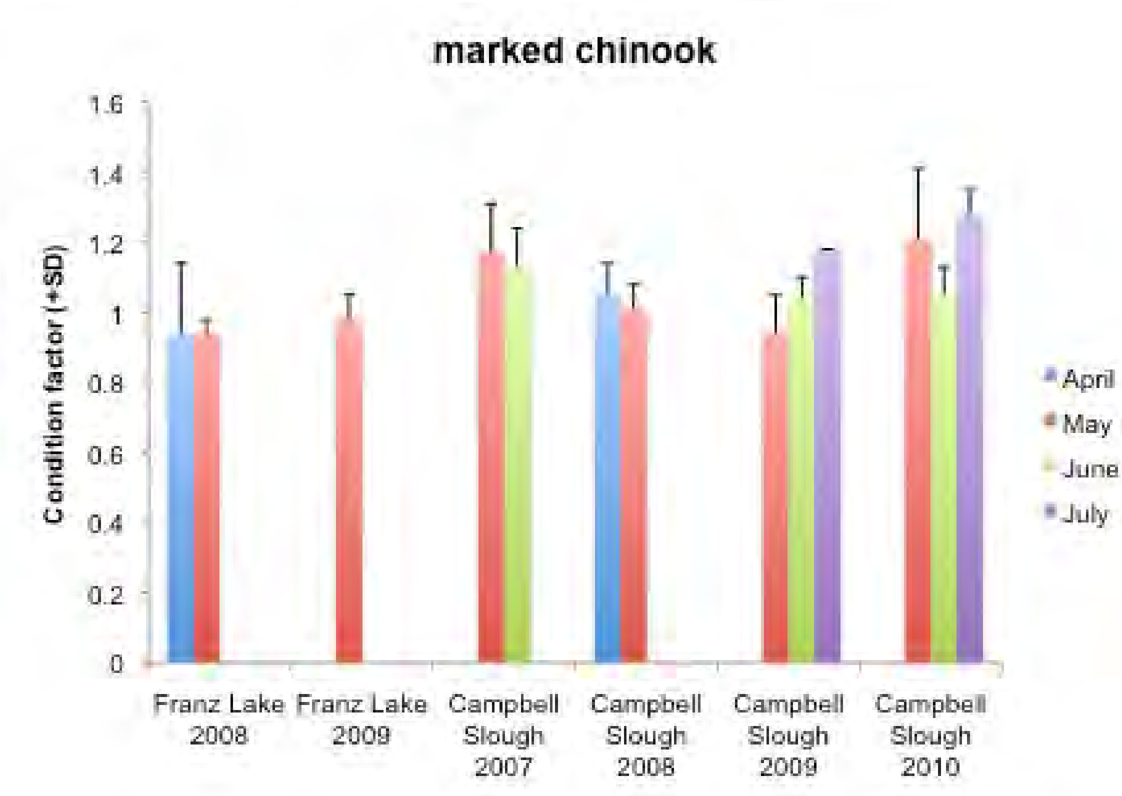


Figure 76. Condition factor (K) of marked and unmarked Chinook from Franz Lake and Campbell Slough over time

Otolith Analyses for Growth Rate Determination

As part of the Ecosystem Monitoring salmon sampling, otoliths were collected from juvenile fall Chinook salmon from Campbell Slough, and Franz Lake for estimation of growth rates average daily growth rates. Growth rates overall were very similar at Franz Lake and Campbell Slough Growth (0.61 mm per day for the last 7 days before sampling at both sites). As for temporal trends (Figure 77), Franz Lake fish grew faster in 2008 than 2009, but significant differences were only detected for the last 21 days of growth. Fish from Campbell Slough showed no significant differences on growth among the sampling years.

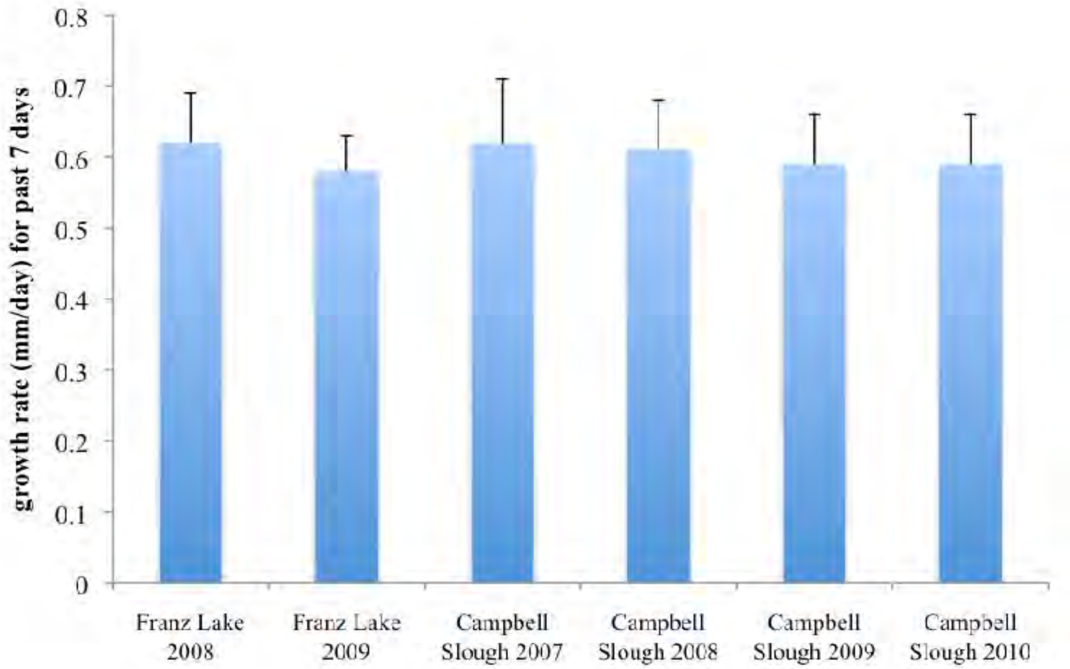


Figure 77. Average daily growth rates for 7-day periods estimated from otolith analyses for juvenile fall Chinook salmon from Franz Lake and Campbell Slough.

Chinook Salmon Lipid Content and Classes

At this point we have lipid data for Chinook salmon samples collected from Franz Lake in 2008 and 2009, and from Campbell Slough in 2007, 2008, and 2009 (Figure 78). Analyses of whole bodies for lipid content and classes are now in progress for the subyearling juvenile Chinook salmon collected at Campbell Slough in 2010.

At both sites, lipid content varied significantly from year to year. At Franz Lake mean lipid content was 2.2% in 2008, but only 1.1% in 2009. At Campbell Slough, lipid content was 1.2% in 2007 and 1.0% in 2009, but 1.7% in 2008. The average lipid content of juvenile Chinook over the entire period sampled was significantly higher (p=0.0012) at Franz Lake than at Campbell Slough (1.9% vs. 1.3%). The distribution of lipid classes differed somewhat between Franz Lake and Campbell Slough. The fish from Campbell Slough tended to have lower proportions of triglycerides than fish from Franz Lake, and in 2008, the Campbell Slough had a significantly higher proportion of cholesterol than Franz Lake fish.

The lipid levels in Franz Lake and Campbell Slough salmon were within the range observed in juvenile salmon sampled as part of the Salmon and Water Quality Study (LCREP, 2007; Figure 79). The lipid content of Franz Lake salmon was comparable to higher lipid values measured in fish from Warrendale and Point Adams, while the lipid content of salmon from Campbell Slough was lower, similar to the level measured in fish from Beaver Army Terminal.

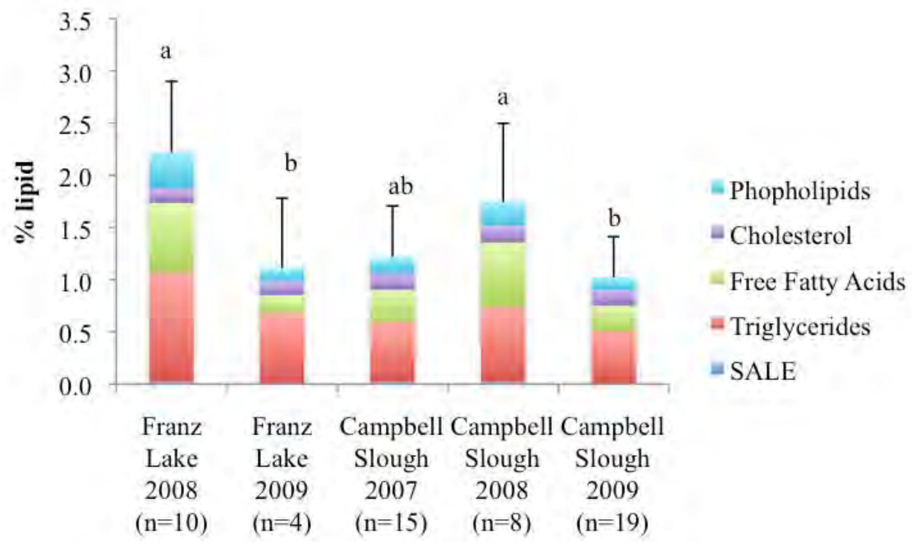


Figure 78. Lipid content and classes of juvenile Chinook salmon from Franz Lake and Campbell Slough.

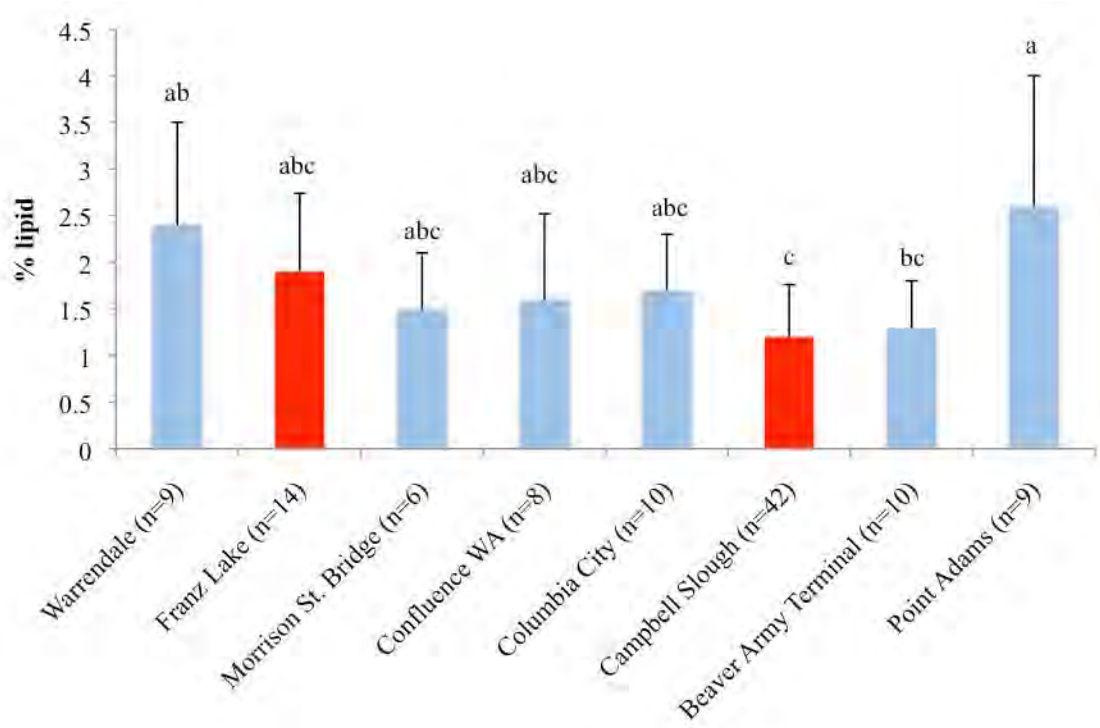


Figure 79. Lipid content and classes of juvenile Chinook salmon from Franz Lake and Campbell Slough as compared to sites sampled in the Salmon ad Water Quality project (LCREP, 2007).

Salmonid Prey Availability Surveys and Diet Analyses for Juvenile Chinook Salmon

We are analyzing diets of juvenile Chinook salmon and identifying prey species in salmon habitats to understand prey sources for juvenile salmonids and the potential influence of prey availability on juvenile salmonid occurrence in various habitat types. A related objective is to use these data to identify potential sources of contaminants affecting fish in the LCRE.

Salmon stomach contents samples. At Campbell Slough, prey species from salmon stomach contents were identified for April 2008, May 2008, May 2009, and June 2009, while for Franz Lake, prey in stomach contents were identified for April 2008, May 2008, and June 2009 (Table 36). At both sites and at all sampling times, Dipteran species, primarily Chironomids, made up the bulk of the diet. The proportion of Dipterans in the diet of juvenile Chinook salmon from Franz Lake ranged from 81-97%, while at Campbell Slough, the proportions of Dipterans ranged from 67-95%.

Prey availability samples. Neuston tows in open water and emergent vegetation were performed and invertebrate samples analyzed to characterize the types of prey available at the study sites (Table 36). Terrestrial vegetation was also sampled using sweep nets in 2007 and 2008, but samples were archived because of limited time and funds for analyses. At Franz Lake, the primary invertebrate prey available to salmonids in both types of samples were Cyclopoid copepods, Cladocerans and Diptera. The copepods were most consistently found as a high proportion of prey, but Cladocera and Diptera contributed significantly proportions of prey at some sampling times. The types and proportions of prey present in the samples were similar in 2008 and 2009.

At Campbell Slough, the primary invertebrate prey available to salmonids in both types of samples were Cyclopoid copepods, Cladocerans and Diptera. The only exception was an open water tow in May 2009, when Cyclopoids were not present. Ostracods made up a relatively high percentage of prey (35%) in the open water tows collected in April 2008, but made up a small percentage of the samples otherwise. The types and proportions of prey present in the samples were similar in 2008 and 2009.

Selectivity analysis. Selectivity values calculated for the three most abundant taxa (Diptera, Cyclopoda, and Cladocera) in emergent vegetation and open water tows are shown in Table 37. 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. In all sampling periods at both sites, selectivity indices showed a strong preference for Dipterans, as indicated by the large positive values.

Table 36. Mean proportion of taxa in Chinook diets and in emergent vegetation tows and open water tows (by count) at Franz Lake and Campbell Slough.

	Franz Lake									Campbell Slough										
	Diets April 2008	EV April 2008	OW April 2008	Diets May 2008	EV May 2008	OW May 2008	Diets May 2009	EV May 2009	OW May 2009	Diets April 2008	OW April 2008	Diets May 2008	EV May 2008	OW May 2008	Diets May 2009	EV May 2009*	OW May 2009*	Diets June 2009	EV June 2009	OW June 2009
Amphipoda	<0.01		<0.01		0.01							<0.01	0.01		0.05					
Araneae				0.02			<0.01					0.01			<0.01			<0.01		
Basommatophora					0.01						<0.01		<0.01						0.01	0.09
Calanoida						<0.01					<0.01		<0.01	<0.01			0.04			
Cladocera		**0.51**	**<0.01**	**0.02**	**0.25**	**0.18**		**0.07**	**0.17**	**0.01**	**0.03**	**0.14**	**0.13**	**0.17**		**0.13**	**0.58**	**<0.01**	**0.56**	**0.28**
Coleoptera	0.01			0.02						0.05		0.03	0.01	0.02	0.02				<0.01	
Collembola												<0.01			0.02					
Copepoda						<0.01														
Cyclopoida		**0.46**	**0.31**		**0.55**	**0.62**		**0.64**	**0.65**		**0.17**		**0.39**	**0.40**		**0.25**			**0.16**	**0.03**
Diptera	**0.97**	**0.01**	**0.66**	**0.81**	**0.07**	**0.11**	**0.93**	**0.19**	**0.17**	**0.93**	**0.25**	**0.67**	**0.22**	**0.15**	**0.78**	**0.38**	**0.13**	**0.95**	**0.11**	**0.48**
Ephemeroptera												0.06	<0.01	<0.01						
Hemiptera	<0.01			<0.01	0.05		0.03					0.05	0.04		0.04			<0.01	0.01	
Hydrozoa											0.07									
Hymenoptera	<0.01			0.02			<0.01					0.01			0.03			<0.01		

Insect egg				0.10								<0.01								
Lepidoptera				<0.01								<0.01								
Nematoda			<0.01										0.01	0.02					<0.01	
Odonata								0.02		<0.01	<0.01	<0.01	0.03		<0.01				0.01	<0.01
Oligochaeta			<0.01		0.06	0.01	<0.01	0.02			0.12		0.04	0.08		0.13	0.08		0.11	0.12
Ostracoda		0.01	0.01			<0.01					0.35	<0.01	0.02	0.13			0.08			
Plecoptera												<0.01								
Poecilostomatoida						0.06					<0.01		0.04	0.02					<0.01	
Psocoptera												<0.01								
Thysanoptera				<0.01								<0.01			0.01					
Trichoptera	0.01					<0.01	0.02					0.01	<0.01		0.02				<0.01	
Trombidiformes			<0.01	0.01	0.01		<0.01	0.05			<0.01	<0.01	0.04	0.02	<0.01	0.13	0.08	0.03	<0.01	<0.01
Turbellaria						<0.01														
Unknown															<0.01					

Table 37. Mean selectivity values for the three most abundant taxa collected in emergent vegetation tows and open water tows at Franz Lake and Campbell Slough in 2008 and 2009. 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.

Site	Month and year	comparison	Diptera	Cyclopoida	Cladocera
Franz Lake	April 2008	Diets vs. emergent vegetation tows	0.95	-0.47	-0.51
		Diets vs. open water tows	0.3	-0.31	0.00
	May 2008	Diets vs. emergent vegetation tows	0.74	-0.55	-0.23
		Diets vs. open water tows	0.7	-0.62	-0.16
	May 2009	Diets vs. emergent vegetation tows	0.74	-0.64	-0.07
		Diets vs. open water tows	0.76	-0.65	-0.17
Campbell Slough	April 2008	Diets vs. emergent vegetation tows	-	-	-
		Diets vs. open water tows	0.69	-0.17	-0.01
	May 2008	Diets vs. emergent vegetation tows	0.45	-0.39	-0.01
		Diets vs. open water tows	0.52	-0.4	-0.03
	June 2009	Diets vs. emergent vegetation tows	0.84	-0.16	-0.56
		Diets vs. open water tows	0.48	-0.03	-0.27

Contaminants in Bodies, Stomach Contents, and Bile of Chinook Salmon from Campbell Slough and Franz Lake

Stomach contents. Currently, stomach contents chemistry data are available only for juvenile Chinook salmon collected from Campbell Slough in 2007. These samples contained moderate concentrations of DDTs (21 ng/g ww) and PCBs (37 ng/g ww) and low levels of PBDEs (2.5 ng/g ww) and PAHs (43 ng/g ww total PAHs), as compared to samples from the Salmon and Water Quality Study (Figure 80 and Figure 81).

Salmon Bodies. At this point, body chemistry data are available for juvenile Chinook collected from Campbell Slough in 2007, 2008, and 2009 and from Franz Lake in 2008 and 2009. The major contaminants in the Franz Lake salmon were DDTs, although low levels of PBDEs and PCBs were also detected (Figure 82). Contaminant concentrations did not change significantly in fish from Franz Lake between 2008 and 2009, although 2009 values tended to be higher.

In the fish from Campbell Slough, DDTs were present at concentrations generally in a similar range to those found in the fish from Franz Lake, although the highest DDT levels reported at Campbell Slough, in 2007, were significantly higher than the lowest levels reported at Franz Lake, in 2008. Additionally, bodies of the Campbell Slough fish contained PBDEs and PCBs at concentrations several times higher than those observed in fish from Franz Lake (Figure 82).

In comparison to contaminant concentrations measured in juvenile Chinook as part of the Salmon and Water Quality Study (LCREP, 2007), concentrations of PCBs and PBDEs in salmon from Campbell Slough and Franz Lake were relatively low, while concentrations of DDTs were similar to those observed in fish from most of the previously sampled sites (Figure 83). Additionally, at Franz Lake, concentrations of DDTs, PCBs, and PBDEs in all samples were below estimated toxic effect concentrations of 6000 ng/g lipid for DDTs, 2400 ng/g lipid for PCBs and 940 ng/g lipid for PBDEs (Meador et al. 2002, Johnson et al. 2007; Beckvar et al. 2005; Arkoosh et al. 2010). In samples from Campbell Slough, DDT concentrations were also below estimated effect thresholds. However, concentrations of PCBs in a number of samples were above estimated toxic effect thresholds for PCBs. These included 10 of the 15 samples collected in 2007, 1 of the 8 samples collected in 2008, and 3 of the 9 samples collected in 2009. Additionally, one sample collected in 2008 was above the estimated PBDE concentrations associated with immunosuppression in juvenile Chinook salmon (Arkoosh et al., 2010).

Multivariate analysis indicated that, of the factors site, year, month, and fish type (wild vs. hatchery), the two significant factors affecting PCB concentration were site and year ($p < 0.0001$). Concentrations of PCBs were significantly higher in fish from Campbell Slough than in fish from Franz Lake, and significantly higher in 2007 than in 2008 or 2009. Month of capture and fish origin (hatchery vs. wild) were not significant. R^2 for the model = 0.85 ($p < 0.0001$). The only significant factor affecting DDT concentration was year ($p < 0.0001$), although DDT concentrations tended to be higher in fish from Campbell Slough than from Fran Lake ($p = 0.09$). Concentrations of DDTs were significantly lower in fish captured in 2009 than in 2007 or 2008. R^2 for the model = 0.62 ($p < 0.0001$). The only significant factor affecting PBDE concentration was site ($p < 0.0001$). Concentrations of PBDEs were significantly higher in fish from Campbell Slough than in fish from Franz Lake. R2 for model 0.4567 ($p = 0.0003$).

Bile. Concentrations of BaP and phenanthrene (PHN) metabolites (measured as fluorescent aromatic compounds detected at BaP and PHN wavelengths (FACs-BaP and FACs-PHN) in bile of Chinook salmon from Franz Lake and Campbell Slough are shown in Figure 84. Because the number of composite samples collected each year was so small, no statistically significant differences were observed among concentrations. However, levels of FACs-PHN were as high or higher at Franz Lake than at Campbell Slough, while levels of FACs-BaP at Franz Lake were in the lower range of values observed at Campbell Slough. Particularly in the case of FACs-BaP, there appeared to be an increasing trend in bile metabolite levels at Campbell Slough between 2007 and 2009.

In comparison to FACs levels in Chinook salmon from the Salmon and Water Quality sites (LCREP, 2007; Figure 85), the FACs-PHN concentration at Franz Lake was among the highest, while FACs-PHN levels at Campbell Slough were moderate. As for FACs-BaP levels, values at Fran Lake and Campbell Slough were relatively low, comparable to concentrations at relatively undisturbed sites such as Warrendale and Point Adams.

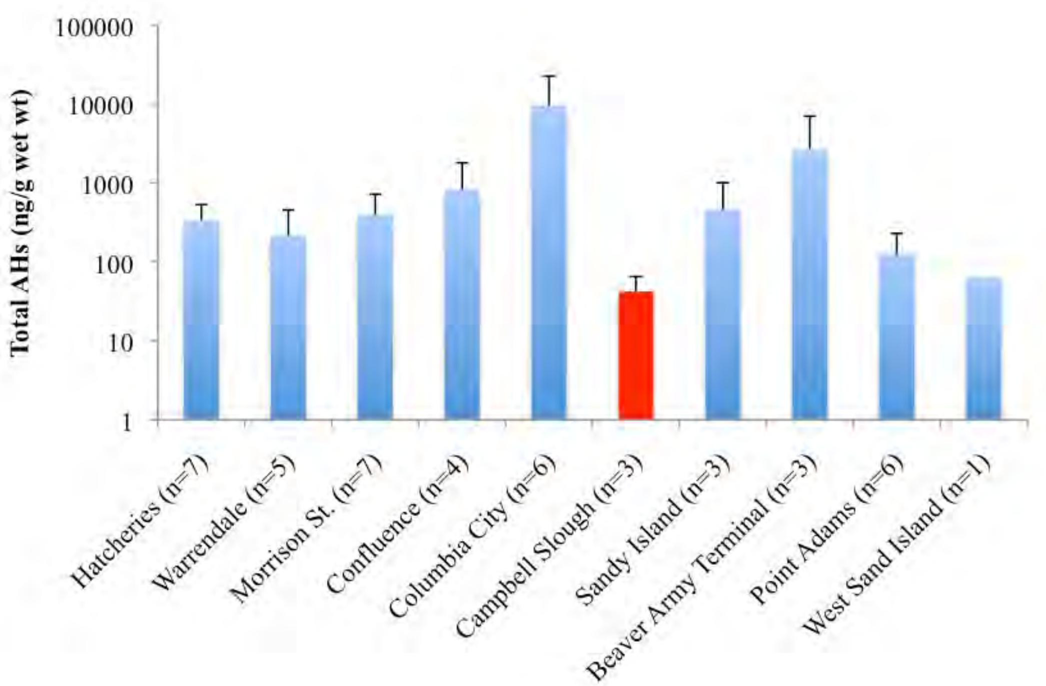


Figure 80. Concentrations of aromatic hydrocarbons (AHs) in stomach contents of juvenile Chinook salmon from Campbell Slough as compared to juvenile Chinook salmon from sites sampled in the Salmon and Water Quality project (LCREP, 2007).

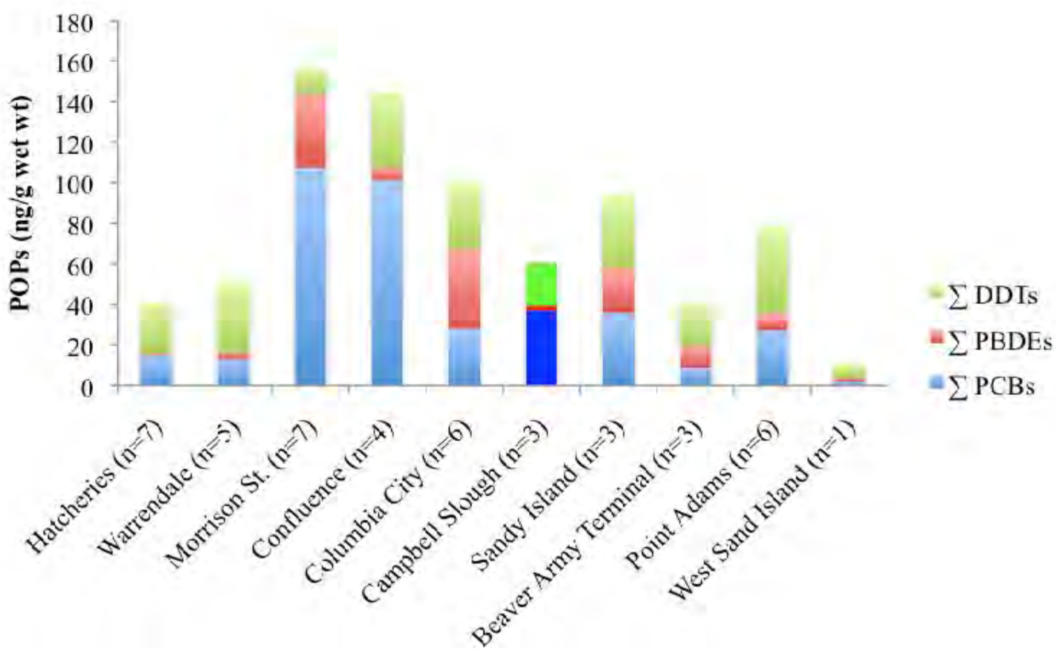


Figure 81. Concentrations of DDTs, PBDEs, and PCBs in stomach contents of juvenile Chinook salmon from Campbell Slough as compared to juvenile Chinook salmon from sites sampled in the Salmon and Water Quality project (LCREP, 2007).

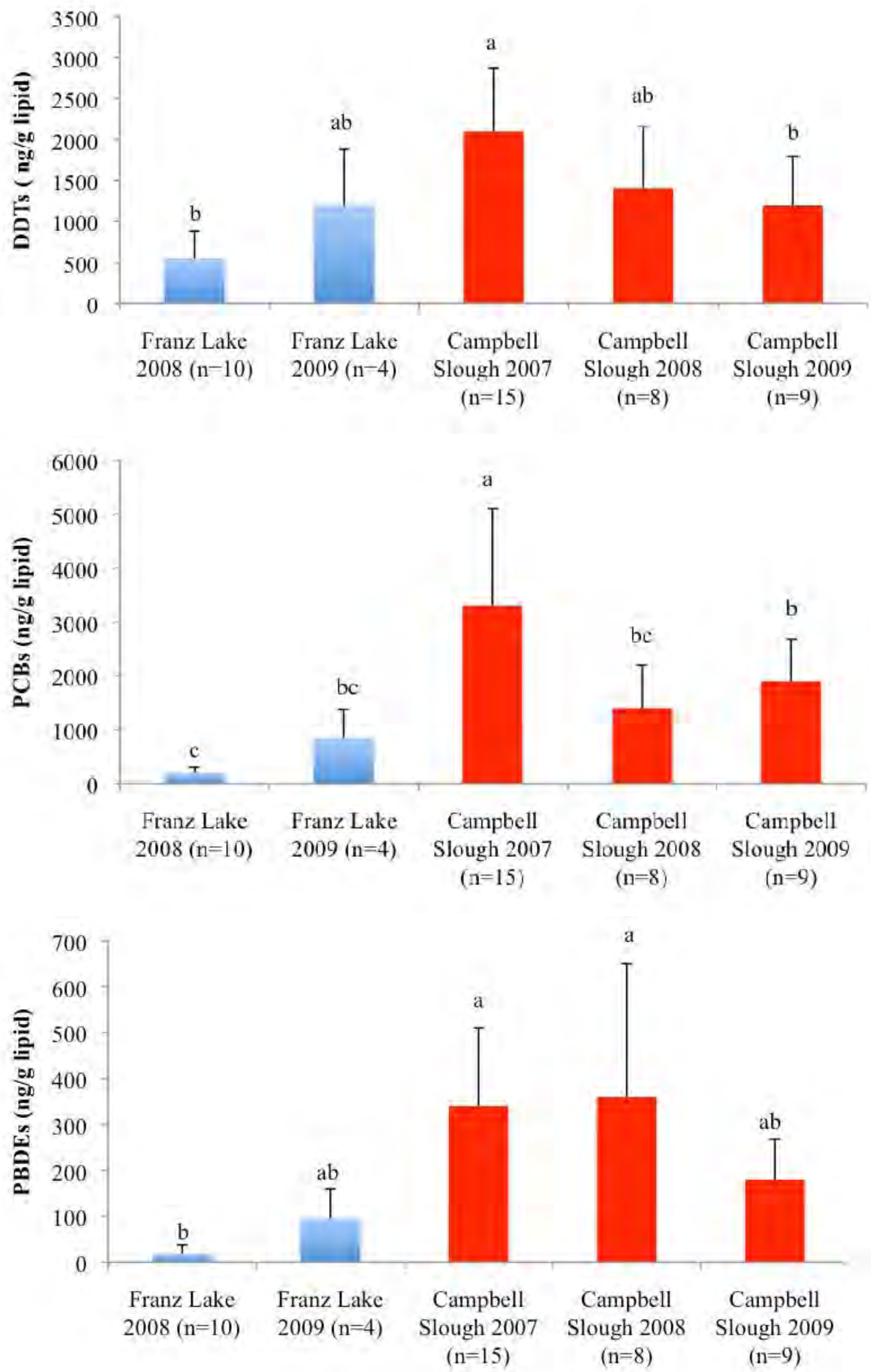


Figure 82. Persistent organic pollutants (DDTs, PCBs, and PBDEs) in body samples of juvenile Chinook salmon from Franz Lake and Campbell Slough.

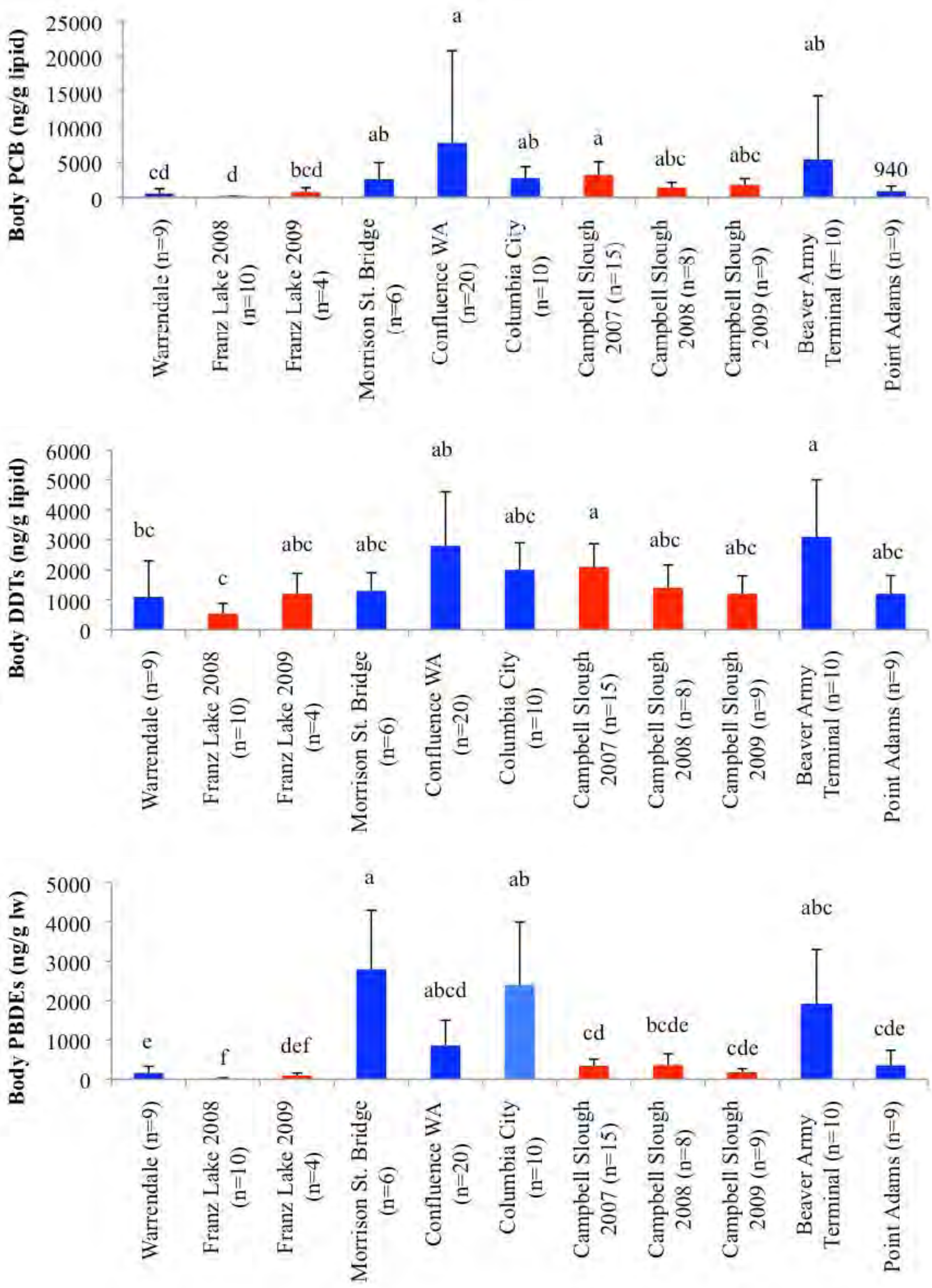
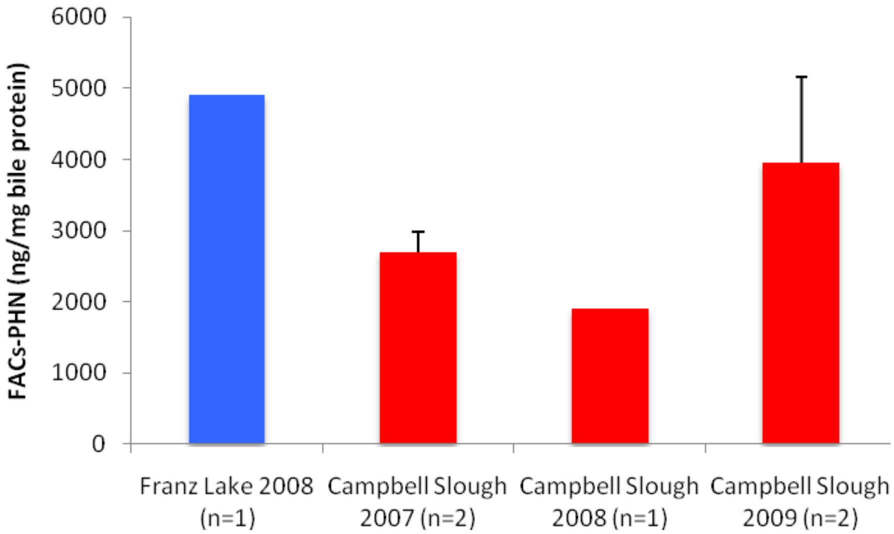

Body PCB (ng/g lipid)
25000
20000
15000
10000
5000
0
cd
d
bcd
ab
a
ab
a
abc
abc
ab
940
Warrendale (n=9)
Franz Lake 2008 (n=10)
Franz Lake 2009 (n=4)
Morrison St. Bridge (n=6)
Confluence WA (n=20)
Columbia City (n=10)
Campbell Slough 2007 (n=15)
Campbell Slough 2008 (n=8)
Campbell Slough 2009 (n=9)
Beaver Army Terminal (n=10)
Point Adams (n=9)
Body DDTs (ng/g lipid)
6000
5000
4000
3000
2000
1000
0
bc
c
abc
abc
ab
abc
a
abc
abc
a
abc
Warrendale (n=9)
Franz Lake 2008 (n=10)
Franz Lake 2009 (n=4)
Morrison St. Bridge (n=6)
Confluence WA (n=20)
Columbia City (n=10)
Campbell Slough 2007 (n=15)
Campbell Slough 2008 (n=8)
Campbell Slough 2009 (n=9)
Beaver Army Terminal (n=10)
Point Adams (n=9)
Body PBDEs (ng/g lw)
5000
4000
3000
2000
1000
0
e
f
def
a
abcd
ab
cd
bcde
cde
abc
cde
Warrendale (n=9)
Franz Lake 2008 (n=10)
Franz Lake 2009 (n=4)
Morrison St. Bridge (n=6)
Confluence WA (n=20)
Columbia City (n=10)
Campbell Slough 2007 (n=15)
Campbell Slough 2008 (n=8)
Campbell Slough 2009 (n=9)
Beaver Army Terminal (n=10)
Point Adams (n=9)

Figure 83. Comparison to Salmon & Water Quality between Campbells Slough and Franz Lake.



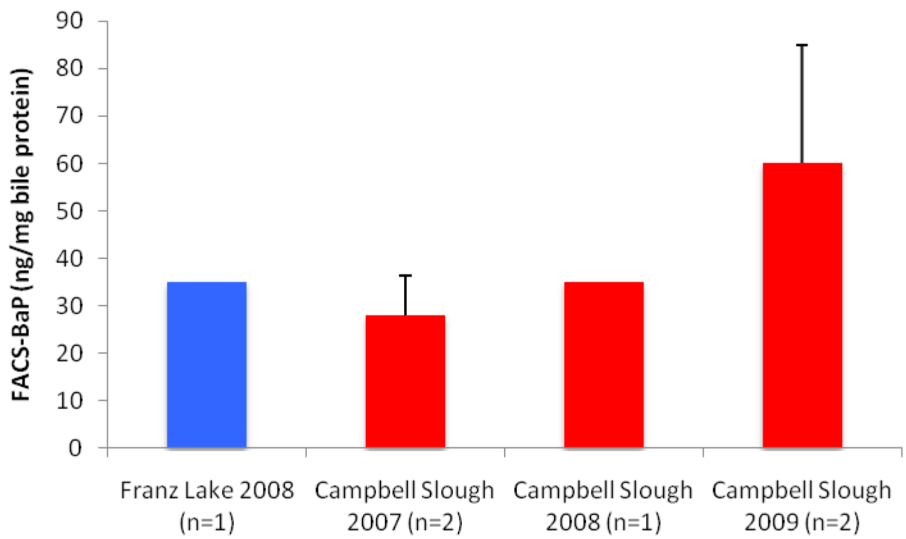
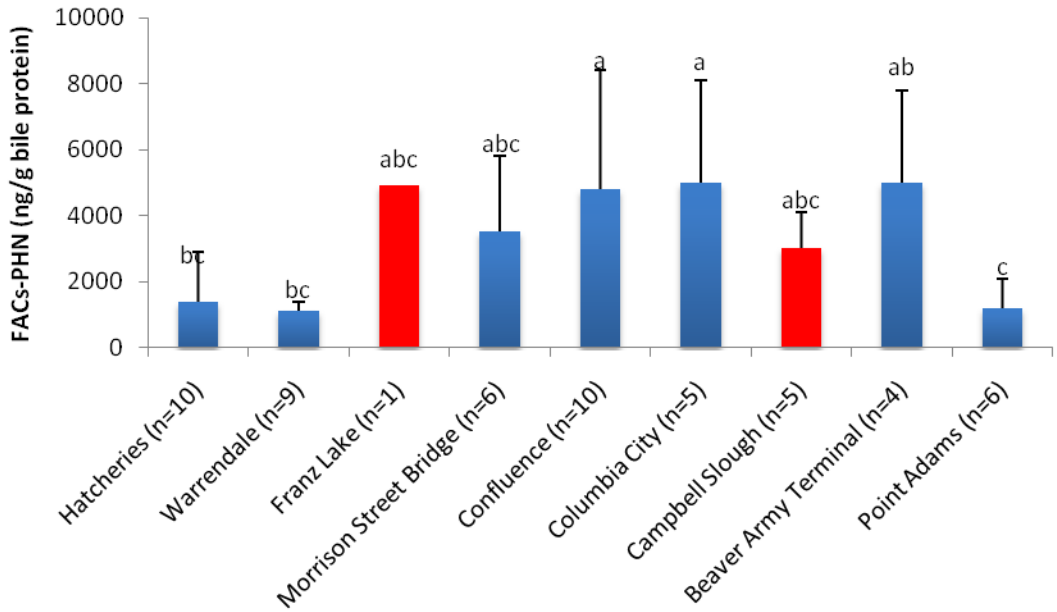


Figure 84. Comparison of years between Campbell Slough and Franz Lake



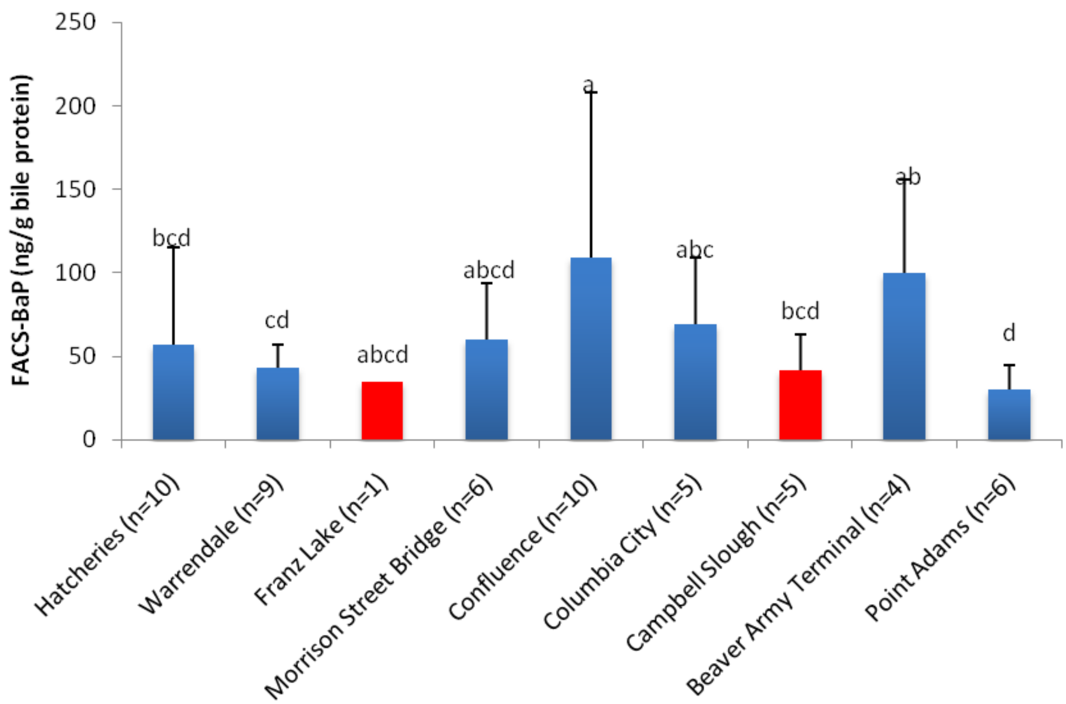


Figure 85. Comparison with Salmon and Water Quality

Discussion

Over two to four years, we have monitored fish community characteristics, salmonid habitat occurrence, various measures of salmon size, growth, and condition, salmon diet, prey availability, and contaminant exposure at Franz Lake and Campbell Slough, two tidal freshwater emergent marsh sites. The Franz Lake site in Reach H of the Lower Columbia River, in the Columbia Gorge, while the Campbell Slough site is in Reach H. Overall, fish community characteristics, patterns of salmonid use, salmonid diets, and prey types present were consistent within each site over the sampling period.

We found juvenile Chinook salmon using both of these sites from April, when sampling begins, until June or July. Although wild salmonids were present at both sites, hatchery salmonids made up substantial proportions of the fish using these sites. The Franz Lake site had a greater diversity of salmonids, with significant numbers of coho and chum as well as Chinook. Franz Lake also had a higher proportion of wild Chinook salmon in small size classes; 25% of Chinook were below 70 mm in length at Franz Lake, in contrast to 13% from Campbell Slough.

Fish community characteristics were somewhat similar at the two sites. The number of species, species richness and diversity were within the same range. However, the exact mix of species varied, with higher numbers of stickleback at Campbell Slough and more chiselmouth at Franz Lake. The percentage of non-native species tended to be higher at Campbell Slough than at Franz Lake, ranging from 53-67% of species captured. At Franz Lake, the percentage of non-native species ranged from 31-37% of species captured.

Fish length, weight, and condition factor (K) followed the same general patterns at both sampling sites, after differences in proportions of wild and hatchery fish and sampling dates had been taken into account. Hatchery fish were consistently larger and heavier than wild fish, and length, weight, and condition all tended to increase over the course of the sampling season, particularly in wild fish. Length, weight and condition all varied from year to year at both sites. At both Campbell Slough and Franz Lake, fish size and condition tended to be low in 2009, perhaps because of the very warm summer temperatures that year. Like size and condition distributions, growth rates in fish from Campbell Slough and Franz Lake were quite similar, and tended to be low in 2009. Lipid content showed a similar pattern; lipid levels from samples collected at both sites in 2009 were significantly lower than samples collected in other years.

Available invertebrate prey species were also similar at both sites. The primary species in both the open-water tows and the emergent vegetation tows were Cladocerans, cyclopoid copepods, and Dipterans. In salmon stomach contents from both sites, the primary prey species were dipterans, primarily Chironimids. Selectivity indices showed a strong preference for Dipterans at both sites.

In general, contaminant concentrations in fish from Franz Lake and Campbell Slough were low to moderate, in comparison to fish sampled in the Lower Columbia as part of the Estuary Partnerships's Water Quality and Salmon Study (LCREP, 2007), which would be consistent with the relatively undisturbed character of these sites. Moreover, concentrations of bioaccumulative contaminants in salmon bodies, especially PCBs and PBDEs, tended to be lower at Franz Lake than at Campbell Slough, a finding that is not surprising considering the location of Campbell Slough downstream of the urban and industrial centers of Portland and Vancouver. In fact, in several samples from Campbell Slough, concentrations of PCBs were at levels associated with an increased risk of toxic injury (Meador et al., 2002). These was particularly true of fish collected in 2007, where 67% of fish collected had PCB concentrations above the injury threshold of 2400 ng/g lipid). Fortunately, our data suggest declining trends in concentrations of PCBs and DDTs, with significantly lower levels of both compounds in fish collected from Campbell Slough in 2009 as compared to 2007. Contaminant concentrations were not significantly different in hatchery and wild fish.

Exposure to PAHs could be a problem in some juvenile salmon from Franz Lake and Campbell Slough, although the evidence for this is mixed. In stomach contents samples from Campbell Slough, PAH levels are fairly low in comparison to some of the more urbanized sites monitored as part of the Salmon and Water Quality Study (LCREP, 2007). Similarly, metabolites of BaP, the carcinogenic high molecular weight PAH present in combustion products and common in urban and industrial samples, were low in bile samples of fish from these sites. On the other hand, there was evidence of exposure to lower weight PAHs, represented by FACs-PHN, especially at Franz Lake. The source of PAHs at Franz Lake is unclear, but it is possible that they may be naturally occurring PAHs found in wood products and decaying organic matter (Venkatesan, 1988; Tavendale et al.,1995). A recent spill or runoff from the nearby highway or railway are other possible sources. With only one sample, it is difficult to know whether our observation of PAH contamination at this site is representative or typical conditions or not. A somewhat disturbing trend in an apparent increase in PAH metabolite levels in bile of salmon from Campbell Slough, a change that could be related to increased population density in the surrounding area.

In summary, our sampling showed that wild juvenile salmon are feeding and rearing at Franz Lake and Campbell Slough, representative tidal freshwater sites in the LCRE. The sites also appear to function as nursery areas for other fish species. The sites showed a number of similarities: salmon diets and prey species present in the environment were comparable, and various measures of salmon performance (e.g., growth rates, lipid content) were within the same range at both sites. However, there were some differences in fish community composition, and the Franz Lake site tended to support a higher number of salmon species and Chinook salmon stocks. The Campbell Slough site showed greater evidence of disturbance, with a higher proportion of non-native fish species at the site, as well higher concentrations of urban and industrial contaminants in fish captured there. High water temperatures may be limiting fish use of both sites in July and August. We also saw some evidence of poorer fish health and condition, as indicated by lipid levels, growth rates, and condition factor, in 2009 as compared to other sampling years, perhaps because of unusually high temperatures that year.

11.0 Emergent Wetland Monitoring Efforts Planned for 2010-2011

In 2010-2011, monitoring partners will collect datasets at 4 new emergent wetlands in a TBD reach of the LCRE and revisit 3 previously sampled sites in Reaches C, F, and H. OHSU will conduct primary and secondary productivity sampling at all fixed sites (3 previously sampled sites in Reaches C, F, and H and one new fixed site in Reach A). Additionally, the Estuary Partnership and partners will compile and synthesize multiple years of data for all sites (vegetation, fish, fish prey and water quality), and conduct interdisciplinary analyses of those datasets in order to make comparisons between metrics. This synthesis task will support on-going efforts to report monitoring results to BPA and regional partners. The synthesis findings will be presented to the Science Work Group in fall 2011 and written up in a technical report form.

12.0 Characterization of Forested Tidal Freshwater Wetlands in the LCRE

Freshwater tidal wetlands are a relatively rare ecosystem, existing only where tidal influences extend beyond the reach of saline water, most prominently in rivers with high discharge and large tidal floodplains ("great rivers"). The freshwater tidal forested wetlands of the Pacific Northwest generally have not been studied comprehensively in any detail, and specifically the Columbia River estuary, despite the fact that they occupy a significant portion of the ~235 km extent of that estuary. They provide essential habitats for juvenile salmonids (*Oncorhynchus* spp.), many of which are listed as threatened or endangered under the United State's Endangered Species Act (Bottom et al., 2005).

Hydroregulation of the Columbia River and urbanization of its watershed and floodplain have likely had a tremendous impact on this estuarine ecosystem. While detailed studies have been conducted on emergent marsh ecosystems and physical processes, only preliminary work has explored a few types of freshwater tidal forested and scrub-shrub wetlands of the Columbia River estuary, and relatively a few comprehensive community structure perspective (Christy and Putera 1992, Diefenderfer 2007, LCREP 1999). Comprehensive ecological characterizations are necessary to build a baseline data set and conceptual model of the ecosystem components and structure that can be used to assess future changes due to anthropogenic or climatic alterations in the Columbia River watershed.

This study was conducted to address the lack of detailed scientific studies of the freshwater tidal forested wetlands worldwide and specifically within the Columbia River estuary. These ecosystems are known to provide valuable ecosystem functions, goods and services and habitat for a variety of faunal groups (Lugo et al., 1990), but have experienced significant declines over the past century as a result of human development of the estuarine floodplain and regulation of the Columbia River flow (Thomas, 1983). Study findings will provide a better understanding of the composition and structure of freshwater tidal forested wetland ecosystems, and their important role in estuarine ecosystems both within the Columbia River system.

A primary goal of the study was to quantitatively characterize variation in the structure of freshwater tidal forested wetlands along the estuarine gradient of the Columbia River. Study sites throughout the freshwater tidal portion of the Columbia River estuary were selected to capture the variability in forested wetlands present along the estuarine gradient. We used a combination of field surveys, laboratory techniques, and multivariate statistics to evaluate patterns in biotic and abiotic characteristics of the forested wetland sites. The specific outcome of the study was designed as a “community profile” that provides a more thorough understanding of the associations among plant and animal species assemblages at freshwater tidal forested wetlands in the Columbia River estuary, and the abiotic factors that determine these relationships.

12.1 Site Descriptions

Site Selection. Candidate sites for field studies were selected by examining current satellite imagery of the Columbia River estuary available on Google Earth® and maps of forested wetland locations present in the late 1970s (U.S. Army Corps of Engineers, 1976). Individual sites were then researched using the internet, personal communications with Si Simenstad, Jennifer Burke, Kathryn Sobocinski, the Lower Columbia River Estuary Partnership (LCREP), and field visits. Sites were selected based upon the presence of relatively unimpacted forested wetlands, representation of different variants of forested wetlands present in the estuary, site accessibility, and the availability of historic vegetation records for comparison purposes.

Three sites, Big Creek, Willow Bar, and Mirror Lake, were selected for the 2008 field season Figure 86; Table 38). 2008 research demonstrated that Big Creek, which is a Sitka spruce tidal swamp, differed dramatically from the black cottonwood-dominated riparian floodplain forests located at Willow Bar and Mirror Lake. Three additional sites, Julia Butler Hansen Wildlife Refuge, Robert W. Little Preserve, and Willow Grove, were selected for data collection in the 2009 field season. All of the 2009 sites were located between Big Creek and Willow Bar in order to capture the transition in forested wetlands that was found to occur between those two sites.

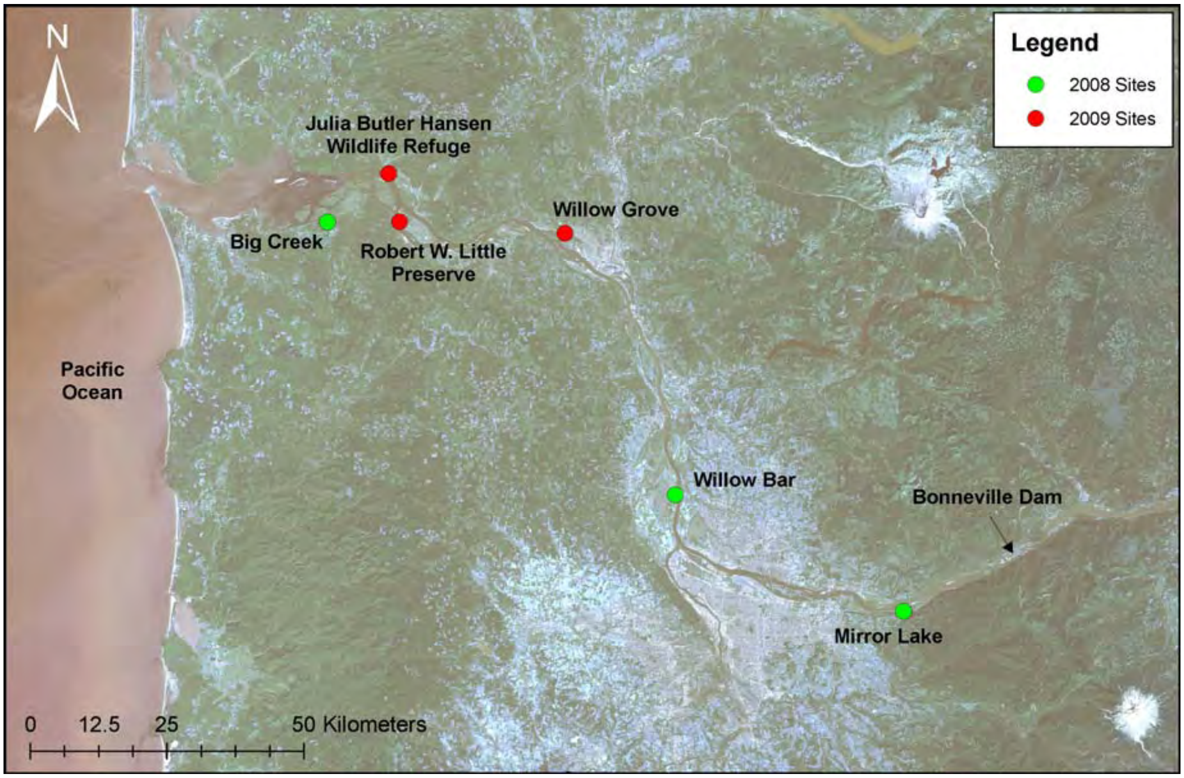


Figure 86. Study site locations for community characterization of tidal forested wetlands of the Columbia River estuary.

Table 38. Coordinates of study sites.

Site Name	Latitude (DD MM.SS)	Longitude (DD MM.SS)
Big Creek	N 46 11.070	W 123 35.61
Julia Butler Hansen	N 46 15.86	W 123 26.91
Robert W. Little	N 46 11.14	W 123 25.38
Willow Grove	N 46 10.06	W 123 01.87
Willow Bar	N 45 44.15	W 122 46.29
Mirror Lake	N 45 32.56	W 122 14.31

Big Creek. The study site closest to the mouth of the Columbia River was located on Big Creek (BC) (near Knappa, OR), near the confluence of the creek and Knappa Slough at approximately RKm 42 of the mainstem Columbia River. The forested wetlands at Big Creek represent Sitka spruce tidal swamps that were once common in the lower Columbia River estuary. A salmon hatchery located approximately 4.8 km upstream of the confluence also depends upon the wetlands to provide habitat for their juvenile salmon releases. Abandoned railroad tracks run along the shore of the Columbia River, crossing Big Creek, and were used to access the study site. Tidal fluctuation in Big Creek is approximately 2.0-2.6 m. The wetlands at Big Creek are dense with scrub-shrub species such as willows, red osier dogwood (*Cornus sericea*), and blackberry (*Rubus* spp.), and large trees such as

Sitka spruce and western red cedar (*Thuja plicata*) provide canopy cover. This site was included in the U.S. Army Corps of Engineers intensive sampling efforts in the 1970s, and was thus selected in part for the availability of data for comparison purposes (U.S. Army Corps of Engineers 1976). The forested wetlands at Big Creek are owned and monitored by The Nature Conservancy (TNC), who encourages the public to enjoy bird watching, canoeing, and kayaking in the preserve.

Julia Butler Hansen Wildlife Refuge. Julia Butler Hansen Wildlife Refuge (JBH) is located at approximately RKm 53, near Cathlamet, Washington. The Refuge was established in 1972 to protect the endangered Columbian white-tailed deer (*Odocoileus virginianus leucurus*), and is managed by the United States Fish and Wildlife Service. The Refuge contains over 24 km^2 including Sitka spruce tidal swamps. The sampling location selected for this study is situated on the mainland portion of the Refuge across from Price Island. The wetland vegetation assemblage at the Refuge is similar to that of Big Creek, consisting primarily of Sitka spruce, Sitka willow, red osier dogwood, and red alder (*Alnus rubra*), with the addition of black cottonwood trees. Tidal fluctuation at this location is approximately 1.9-2.3 m, based on the nearby tidal gauge at Skamokawa, Washington.

Robert W. Little Preserve. The Robert W. Little Preserve (RWL) is positioned at approximately RKm 63 on Puget Island, Washington. The Preserve is owned and managed by the Nature Conservancy, and contains about 0.12 km^2 of native Sitka spruce tidal forested wetlands. Tidal fluctuation at the site is approximately 1.8-2.1 m, based on the nearby tidal station at Wauna, Oregon. The vegetation assemblage at the Preserve is dominated by Sitka spruce, red alder, black cottonwood, red osier dogwood, and Pacific willow.

Willow Grove. Willow Grove (WG), located at approximately RKm 97, was acquired by the Columbia Land Trust for conservation purposes in August 2008. Willow Grove is approximately 1.26 km^2 and includes a variety of wetland habitats including tidal channels, emergent marshes, and tidal forested wetlands. The forested wetland vegetation assemblages that are the focus of this study are composed primarily of black cottonwood, Pacific willow, and Oregon ash. Tidal fluctuation at the site is estimated to be 1.1-1.4 m, based on the nearby tidal station at Longview.

Willow Bar. Willow Bar (WB) is positioned at approximately RKm 153 along the mainstem Columbia River and is connected to Sauvie Island, Oregon, via a land bridge. Between Willow Bar and Sauvie Island is an inlet that contains tidal forested wetlands comprised mainly of willow, black cottonwood and Oregon ash trees. Willow Bar appears to function as a riparian floodplain as well as a wetland area, because at peak river flows in June 2008 the study site was inaccessible due to high water. During times of lower flow, shallow water (less than 0.6 m deep at low tide) is present in the inlet, and tidal fluctuation is approximately 0.3 m. Willow Bar is part of the Sauvie Island Wildlife Area and is managed by the Oregon Department of Fish and Wildlife.

Mirror Lake. Mirror Lake (ML), located at approximately RKm 208, is a wetland area about 32 km downstream of Bonneville Dam, and is connected to the Columbia River by two large culverts underneath Interstate 84. The vegetation present is very similar to that of Willow Bar. Also like Willow Bar, the wetland area functions as a riparian floodplain, and was partly inaccessible during peak river flows in June 2008. During lower flow periods, shallow water is present in the wetland, and tidal fluctuations are minimal. Mirror Lake is part of Rooster Rock State Park managed by the Oregon State Parks department. This site was also included in the U.S. Army Corps of Engineers intensive sampling efforts of riparian habitat in the 1970s (U.S. Army Corps of Engineers 1976).

12.2 Methods

Site Sampling Design. We used a transect and zone based sampling design as the framework for sampling faunal assemblages and corresponding environmental variables at sites. At each site, we established three transects aligned perpendicular to the water portion of the wetland area that extended to the edge of the forested area (Figure 87). The goal of the transect method was threefold: (1) to capture the full range of variation in species present and physical conditions at a given site; (2) to function as replicates for statistical analysis; and, (3) to document changes in species and conditions over the gradient from the wetland area to the forested/uplands area. Transects were positioned at least 100 m apart, because the bird survey literature generally agrees that the audio portion of point count surveys covers a 50-m radius (Ralph et al., 1995). Transects varied in length as the distance between the aquatic and forested portion of the wetlands differed at each transect, but ranged from 7 to 60 meters with a mean length of 21 meters.

Within transects, major vegetation zones were identified for sampling faunal communities (Figure 87). Based on the U.S. Fish and Wildlife Service wetland classification system (Cowardin et al., 1979), the zone designations included: aquatic (A); emergent (E); scrub-shrub (S); and forest (F). The zones were usually easily differentiated from one another by noticeable transitions in vegetation composition from the wetland to the upland portion of the site. In the case that a zone did not exist, WE did not collect samples for that location. For example, the sites studied in 2009 tended to transition from the river or side channel immediately to emergent vegetation zones, resulting in no data for the aquatic zones at those sites since they were not present. All sampling locations were recorded using a handheld Garmin GPSmap 60CSx Global Positioning System.

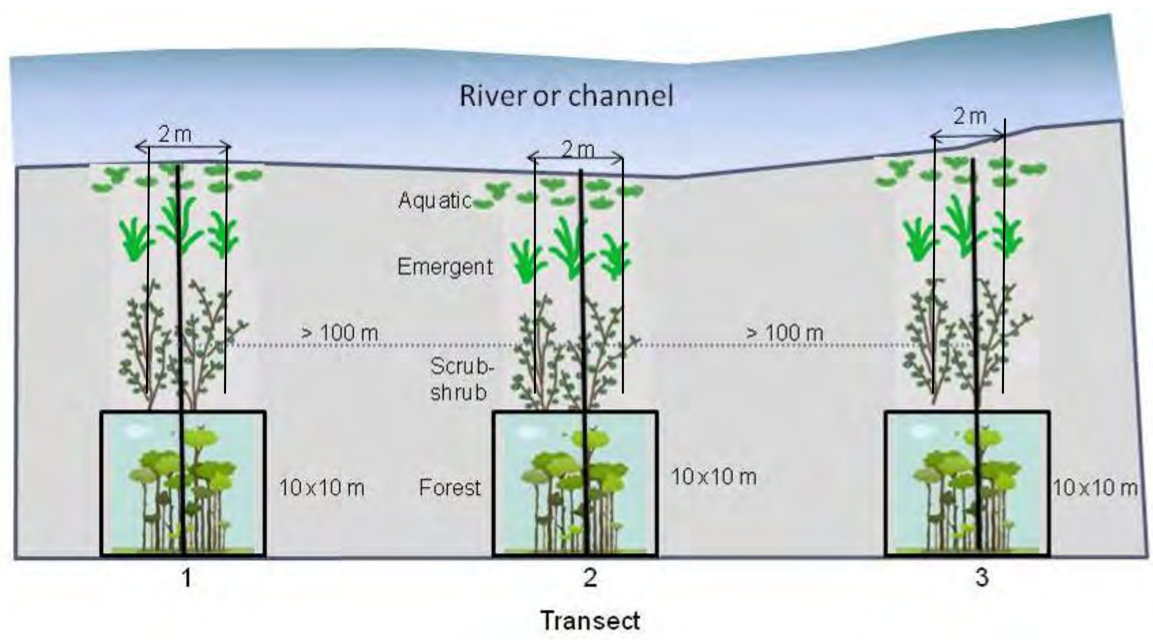


Figure 87. Schematic drawing of sampling layout at forested wetland study sites (not to scale).

Vegetation. A combination of 2-m wide belt transects and 10-m x 10-m plots was used to document the vegetation present at the sites (Kent and Coker, 1992). A 2-m wide belt transect was established at each sampling transect from well within the aquatic vegetation zone to the edge of the forested vegetation zone (Figure 87). We recorded all vegetation species present within each 1-m interval along the length of the transect. In order to adequately capture the full range of tree species and understory present at the sites, we established a 10-m x 10-m plot at the edge of the forested zone (marked by the first tree of stem diameter of 2 cm or more). Within the forested plot, all species present were recorded, the diameter at breast height (DBH) of all trees larger than 2.0 cm was measured and recorded, and the percentage of canopy cover provided by each species of tree within the plot was visually estimated. If no forested zone was present for a given transect, we confined the vegetation survey to only a 2-m belt transect that extended well into the scrub-shrub zone. If the vegetation transitioned immediately from the water to the forested zone for a given transect (this was the case with one transect), we confined the vegetation survey to only a 10-m x 10-m forested plot. Vegetation was identified to species level according to two regional field guides (Pojar and MacKinnon, 2004; Spear Cooke, 1997).

Avifauna. We conducted systematic bird surveys once per season at each site to determine species presence/absence. The spring and autumn surveys occurred approximately during the periods of maximum seasonal migration. Birds present at the sites were surveyed using 10-min point count methods and both visual and audio identification (Ralph et al., 1995). The surveyor stood at a point within each transect where they felt they had a good view of all portions of the wetland, which varied among sites due to topography and vegetation. Binoculars and field identification guides were used to visually identify species during the 10 min of the field observation. Audio identifications were also permitted, and a small recording device was used to record bird calls and songs during the length of the observation. The recording was later analyzed for any bird species not already identified visually or audibly in the field. We repeated the 10-min survey at each transect for a total of three times at a site, giving a total of 90 min of bird surveys during each visit. Sampling was conducted at first light in the morning hours or shortly thereafter, when birds at the study sites were most active and vocal.

Insects. One insect fall-out trap was placed in each transect and zone for a 24-hour period. Insect fall-out traps consist of an approximately 0.24 m^2 plastic tub supported on the bottom by a PVC platform and held in place on the sides by PVC pipes or bamboo poles. The tub was partially filled with water and biodegradable dish soap which acts as a surfactant and prevents insects that have fallen into the bin from escaping. At the end of the 24-hour period, each trap was sieved into a 106-µm sieve, washed, and fixed using a 70% isopropanol solution. The taxa present were later identified in the laboratory according to a taxonomic key (Triplehorn and Johnson, 2005).

Benthic Macroinvertebrates. We acquired one 5-cm dia. (19.6-cm^2) benthic core to 10-cm depth in each zone and along each transect. Samples were sieved and washed over 500-µm sieves. Samples were fixed using a 10% buffered formalin solution, and were later analyzed in the laboratory to identify and enumerate the benthic macroinvertebrate taxa present (Pennak, 1953).

Amphibians. Systematic visual search methods were employed for individual amphibian identification (Bury and Corn, 1991). In general, amphibian surveys were most successful when walking between transects or zones for other sampling purposes, rather than during specific searches. A regional field guide was used to identify amphibians present at the forested wetland sites (Jones et al., 2005).

Mammals. We surveyed mammals present at the sites using visual sightings and track and scat identification. Mammal searches were not limited to transects and zones, since animal ranges often cover the entire site, although it was noted where within the site evidence of mammals was seen (i.e., near Transect 1, or between Transects 1 and 2). Sightings of mammals or evidence of mammals often occurred while walking from the parking area to the study sites, or while walking between sampling locations at a site. Mammals and their tracks and scat were identified using an extensive field guide (Elbroch, 2003).

Environmental Characteristics. Soil cores were collected using a 5-cm diameter (19.6-cm^2) benthic core to 10-cm depth in each zone and along each transect. The samples were later homogenized and split for separate analysis of soil percent organic content and grain size. Soil samples were kept on ice while in the field and placed in a freezer in the laboratory to prevent breakdown of organic matter between the time of collection and analysis.

We determined the percentage of organic material in the soil in each sample by calculating loss-on-ignition. First, soils were weighed, and then placed in drying oven at approximately 30 degrees Celsius (C) for 24 hr. Samples were then weighed and placed in a muffle furnace at approximately 500ºC for six hr (Luczak et al., 1997). The post-burn sample weights were subtracted from pre-burn sample weights to determine the percentage of soil composed of organic material.

We determined the grain size of the soil samples using a Sedigraph 5100 (Bianchi et al., 1999). The Sedigraph uses x-ray beams to calculate the sizes of particles suspended in solution in phi units, which corresponds to sand, silt, and clay grain size classes in the Wentworth scale.

The elevation of each sampling location was determined from Light Detection and Ranging (LiDAR) data. The LiDAR dataset that covers the Columbia River estuary was collected in 2005 and is relatively high-resolution at 1.8 m pixel size. Coordinates from the GPS data collected for each sampling location and rasters derived from the LIDAR data were mapped using ESRI ArcMap and elevations corresponding to GPS points were recorded. Elevations were then converted from North American Vertical Datum 1988 (NAVD88) to Columbia River Datum (CRD), a local vertical datum, in order to compare elevations without the confounding effect of the slope of the river.

12.3 Data Analysis

Univariate Analysis. Descriptive statistics were used to interpret sampling variables that had either low sample size or did not meet criteria for formal statistical analysis. Simple graphical plots were used to interpret several of the variables related to the vegetation communities at tidal forested wetland sites, including average species richness by zone, DBH measurements of trees, and canopy cover by species. Trees species that contributed less than 3% to the total sampled were removed from the DBH boxplot analysis, in order to visualize trends in DBH of dominant trees at sites. Graphs and plots were created using SPSS 17.0 (version 17.0.0) and Microsoft Office Excel 2007.

The sampling methods utilized for amphibians and mammals resulted in relatively low sample numbers, so no statistical analysis was performed on these faunal groups. Instead, the results of the amphibian and mammal surveys are presented in tabular form.

We used SPSS 17.0 (version 17.0.0) to conduct a one-way Analysis of Variance (ANOVA) on the environmental characteristics (site elevation and percent sand, silt, clay and organic content) of vegetation zones within sites. A post-hoc Bonferroni test was performed in order to avoid the problem of multiple comparisons in an ANOVA test (Cabin and Mitchell, 2000). The values for each factor were averaged across all sampling locations within a site, for the purposes of investigating variation in physical conditions along the estuarine gradient. We considered alpha levels of 0.05 to be statistically significant for this test.

Multivariate Analysis. PRIMER 6 (version 6.1.12) was used for all multivariate analyses (Clarke and Warwick, 2001). Vegetation, avifauna, and insect community data was analyzed based on species presence/absence at sampling locations. Benthic macroinvertebrate community data was analyzed based on abundance at sampling locations, because field methods were consistent with techniques used to identify abundance and the small number of taxa observed at the sites was not well suited to analysis based simply on taxa presence and absence.

First, for each community consisting of presence/absence data (vegetation, avifauna, and insect communities) a similarity matrix of sites was calculated based on the Sorensen similarity coefficient

(Sorensen, 1948). The Sorensen similarity coefficient is widely used by ecologists, as it is useful for species presence/absence data. The coefficient excludes double-zeros, which is particularly useful for ecological applications where the unimodal distribution of species distributions along environmental gradients may result in the absence of a species because one site is above and another is below the optimal niche for that species (Legendre and Legendre, 1998).

A similarity matrix for the benthic macroinvertebrate community was calculated using the Bray-Curtis similarity coefficient after applying a log (1+x) transformation to the species abundance data (Bartlett, 1947; Bray and Curtis, 1957). The Bray-Curtis similarity coefficient is mathematically comparable to the Sorensen coefficient described above, but is applied to abundance data rather than presence/absence data.

After assembling the similarity matrices, we performed two-dimensional Non-Metric Multidimensional Scaling (NMDS) on each of the vegetation, avifauna, insect, and benthic macroinvertebrate similarity matrices in order to ordinate sites by similarity of biota (Kruskal, 1964; Shephard, 1962). The advantage of NMDS is that it produces a graphical representation of site similarity, where sampling units located closer to one another in the ordination plot are more similar to one another than those that are further away. NMDS is considered to be an excellent method for analysis of ecological data due to its conceptual simplicity, lack of assumptions about sample data, and preservation of relationships between sampling units in ordination space (Clarke and Warwick, 2001). Stress values in an NMDS plot indicate the distortion between the displayed plot and the similarity rankings of sampling units. Generally, stress values should be less than 0.2 for the 2-dimensional plot to be considered ecologically interpretable (Clarke and Gorley, 2006).

NMDS ordination was also used to examine vegetation assemblages by site and zone. For this analysis, vegetation survey data from all three transects at each site were combined by zone. Then, a NMDS ordination was performed to see if zones within sites were significantly different, and if the same zone at different sites within the estuary were different in terms of vegetation assemblage composition.

We used an Analysis of Similarity (ANOSIM) to test for significant differences in community composition of sites and hydrogeomorphic reaches (Clarke, 1993). ANOSIM provides a p-value that is used to evaluate the significance of the results. For all analyses, a p-value of less than 0.05 was considered statistically significant. We used a one-way ANOSIM test for each community type (vegetation, avifauna, insect, and invertebrate) by site and hydrogeomorphic reach separately.

A Similarity of Percentages (SIMPER) test was performed to determine which species primarily account for differences between tidal forested wetland sites. The SIMPER function in PRIMER is based on Bray-Curtis or Sorensen dissimilarity between two samples, and breaks down the dissimilarities between sample groups by species contributions. The results of the SIMPER analysis allow the ecologist to make conclusions about which species may be good discriminators of two particular sites or sampling units (Clarke and Warwick, 2001).

Finally, we performed a Mantel test, which is a test of the correlation between two matrices, to evaluate the relationship between the vegetation community composition and geographic distance between study sites (Mantel, 1967). For this test, a p-value of less than 0.05 was considered statistically significant.

12.4 Results

Forested Wetland Vegetation Assemblages. A total of 110 plant species were observed at the six tidal forested wetland study sites (see Johnson (2010) for a complete list of plant species identified during field surveys). Wetland vegetation assemblages changed both in assemblage composition and structure along the estuarine gradient. The lower estuarine sites were composed of emergent, scrub-shrub, and forested zones, while the two upper estuarine sites, Willow Bar and Mirror Lake, contained aquatic vegetation zones in addition to the other three zones (Figure 88). Species richness was highest in the lower estuary, decreased along the estuarine gradient to Willow Grove in the mid-estuary, and then increased to the upper estuarine sites of Willow Bar and Mirror Lake. However, the number of trees present in all forested plots at sites increased from the lower to the upper estuary, as did the density of trees in forested plots (Figure 89; Table 39).

Tree diameter at breast height (DBH) measurements are allometrically related to tree biomass and are thus an indicator of forest structure and total biomass (Ketterings et al., 2001). The Columbia River estuary freshwater tidal forested wetland sites displayed a trend of larger DBH measurements at the lower estuarine sites and smaller DBH measurements at upper estuarine sites (Figure 90). The boxplots demonstrate that Western red cedar, Sitka spruce, and red alder account for the majority of trees encountered at the down-estuarine sites (Big Creek, Julia Butler Hansen, and Robert W. Little) while Pacific willow, black cottonwood, and Oregon ash were most frequently encountered in the mid- and upper estuarine sites (Willow Grove, Willow Bar, and Mirror Lake).

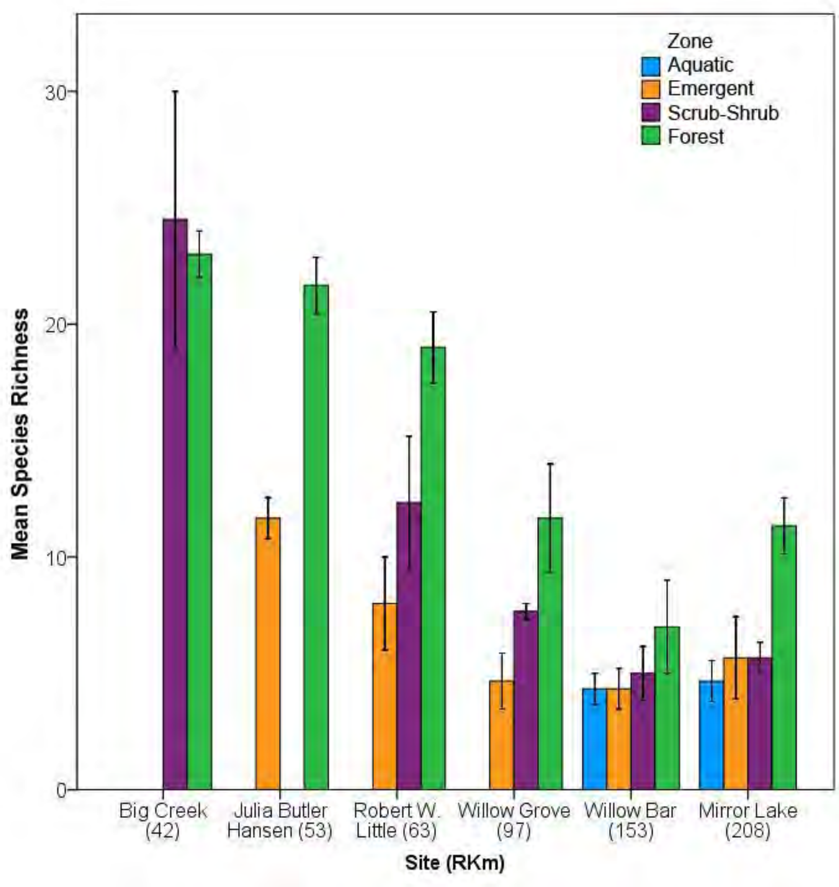


Figure 88. Mean species richness by vegetation zone at tidal forested wetland study sites. Error

bars represent +/- 1 standard error of the mean. Values in parentheses along x-axis indicate site river kilometer (RKm).

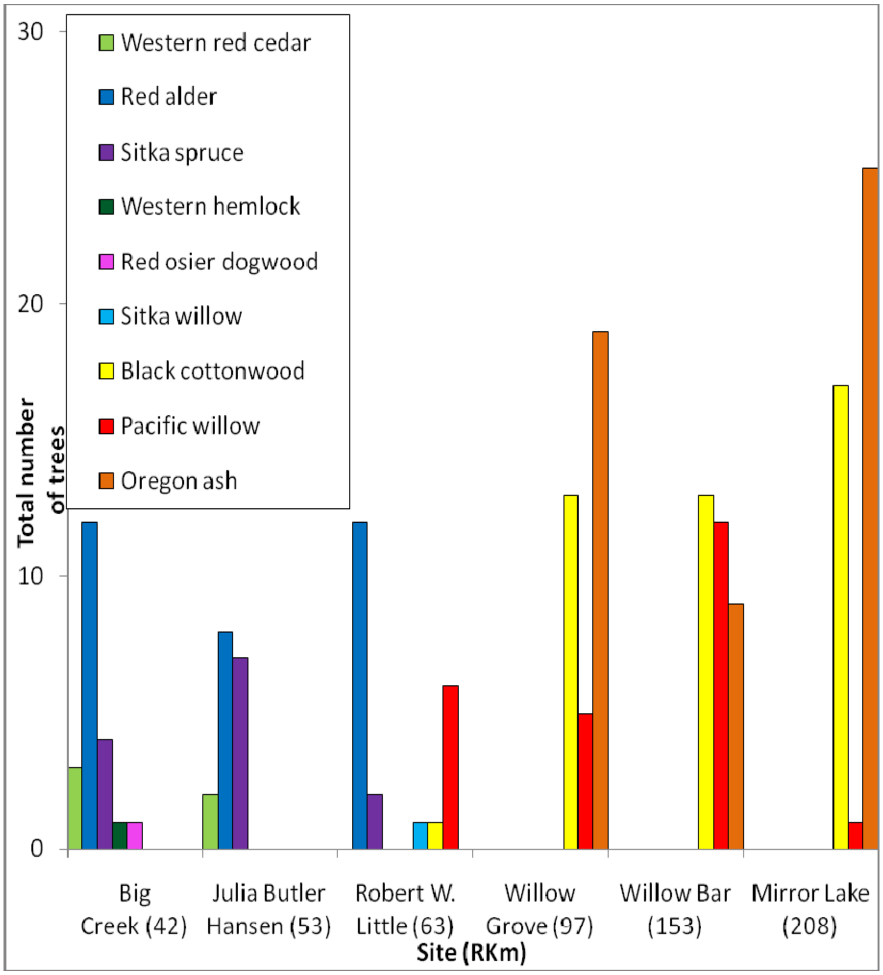


Figure 89. Total number of trees present at forested wetland study sites. Values in parentheses along x-axis indicate site river kilometer (RKm).

Table 39. Mean density of trees in plots at forested wetland study sites.

Site (RKm)	**Mean density of trees present in forested plots** (no. trees per 100 m^2 ± 1 standard error)
Big Creek (42)	10.50 ± 1.50
Julia Butler Hansen (53)	5.67 ± 0.66
Robert W. Little (63)	7.33 ± 2.91
Willow Grove (97)	12.33 ± 2.60
Willow Bar (153)	17.00 ± 4.00
Mirror Lake (208)	14.33 ± 2.60

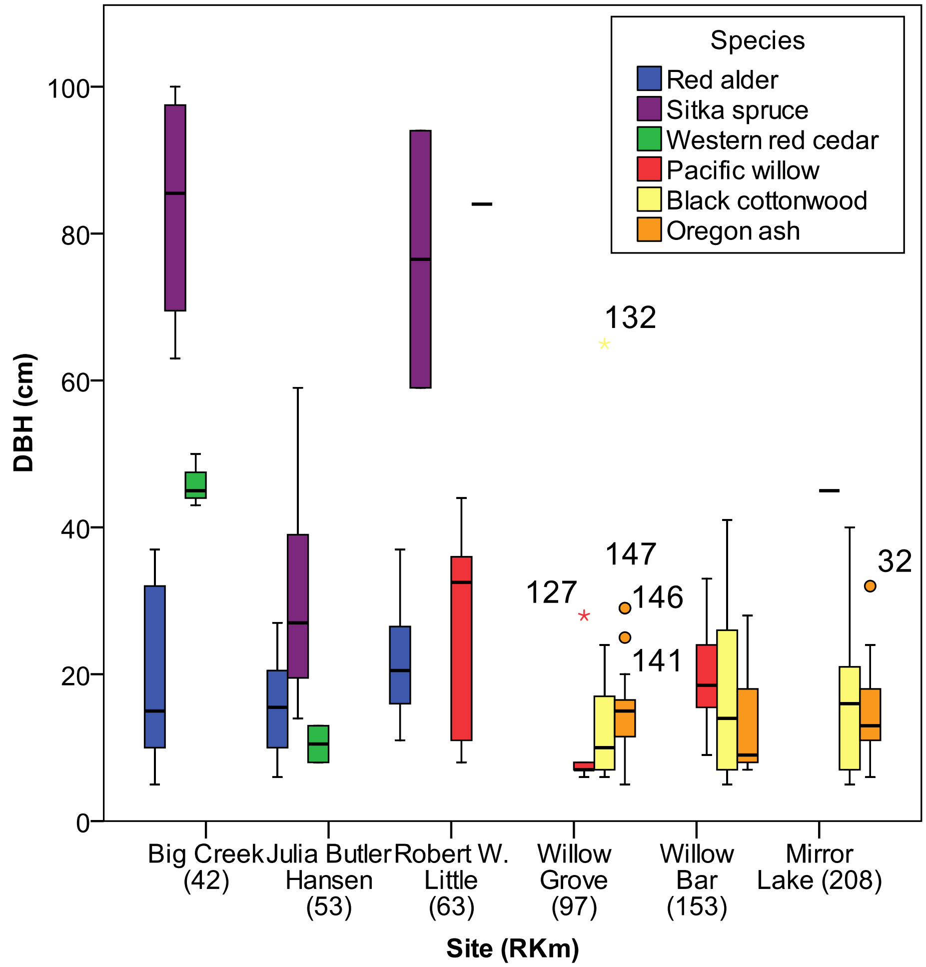


Figure 90. Diameters at breast height (DBH) of tree species present at tidal forested wetland study sites. Numbers on plot indicate outlier tree DBH measurements. Values in parentheses along x-axis indicate site river kilometer (RKm).

Canopy cover showed trends across the forested wetland sites (Figure 91). The percentage of open canopy varied by forested plot within a site, but the composition of tree species contributing to canopy closure in plots demonstrated distinct differences along the estuarine gradient. Lower estuarine sites, including Big Creek, Julia Butler Hansen, and Robert W. Little, had much greater diversity in tree and scrub-shrub species contributing to canopy closure. Coniferous species, including Sitka spruce, western red cedar, and western hemlock (*T. heterophylla*), contributed to canopy closure, but scrub-shrub species such as vine maple (*A. circinatum*), red osier dogwood, Sitka spruce, and European holly (*I. aquifolium*) also accounted for up to 50% of the canopy cover in forested plots. The canopy of mid- to upper estuarine sites (Willow Grove, Willow Bar, and Mirror Lake) consisted of three tree species (Pacific willow, black cottonwood, and Oregon ash) and one scrub-shrub species (red osier dogwood).

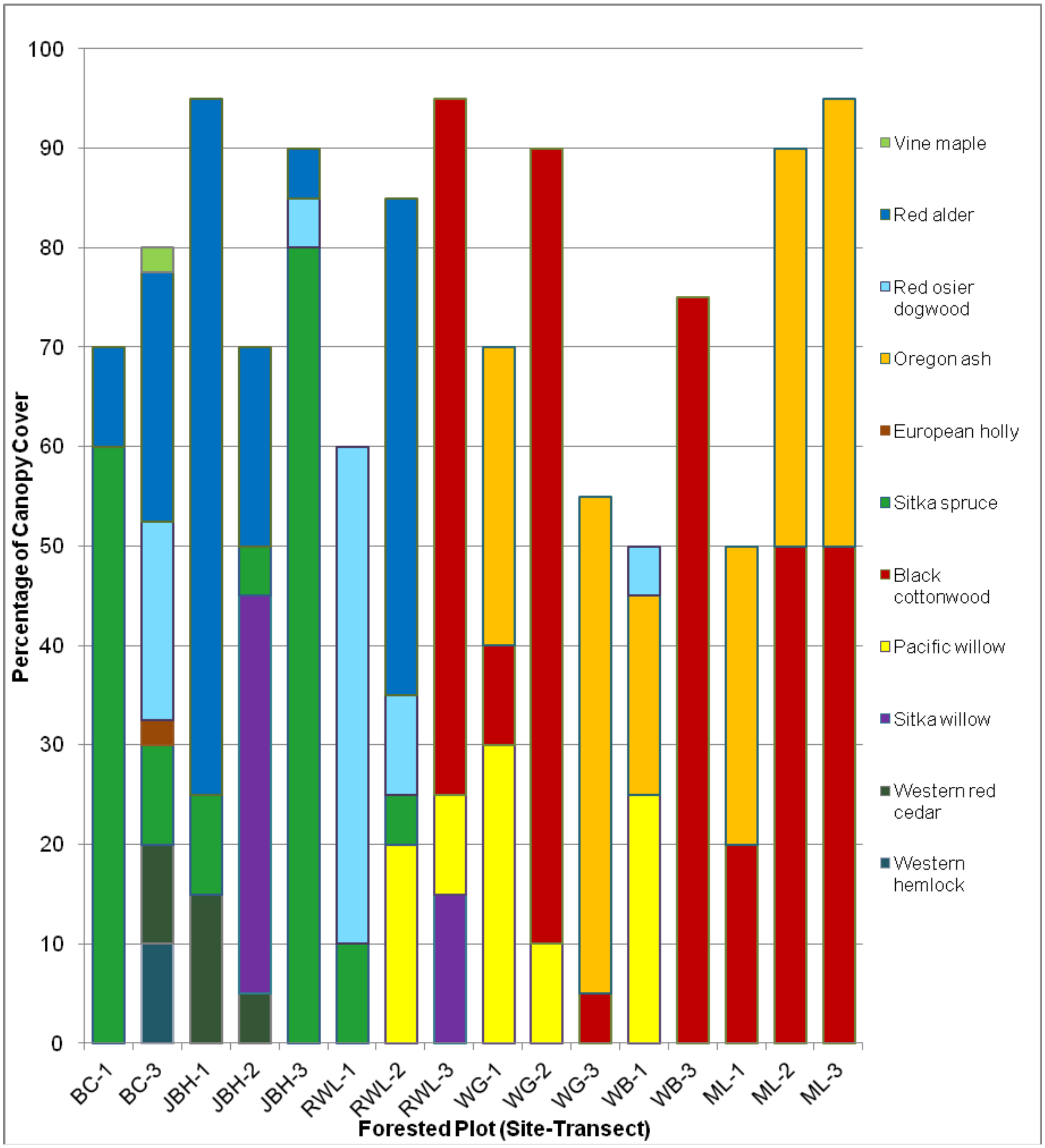


Figure 91. Contributions of individual species to canopy cover in tidal forested wetland survey plots. Note that the percentage of sky visible in plots accounts for the remainder of each column to equal 100%. Site abbreviations are as follows: BC = Big Creek; JBH = Julia Butler Hansen; RWL = Robert W. Little, WG = Willow Grove; WB = Willow Bar; ML = Mirror Lake. Numerals 1, 2, and 3 following site abbreviations indicate the transect number.

The NMDS ordination plot of vegetation presence/absence survey data indicates that transects within sites were similar to one another, because they are located closer together in the plot (Figure 92); thus, among-site variation was consistently higher than within-site variation in vegetation assemblage structure. Forested wetland sites are located farther away from one another, signifying that sites differed from one another based on vegetation species present. The NMDS ordination plot also reveals a trend in sites along the estuarine gradient, with upper estuarine sites located toward the left side of the plot and sites progressively down estuary located along the right side of the plot. Two-dimensional stress in the NMDS ordination plot is 0.12, which is considered ecologically interpretable. The ANOSIM test revealed that the vegetation community composition of sites is significantly different from one another (global R = 0.891, p = 0.001).

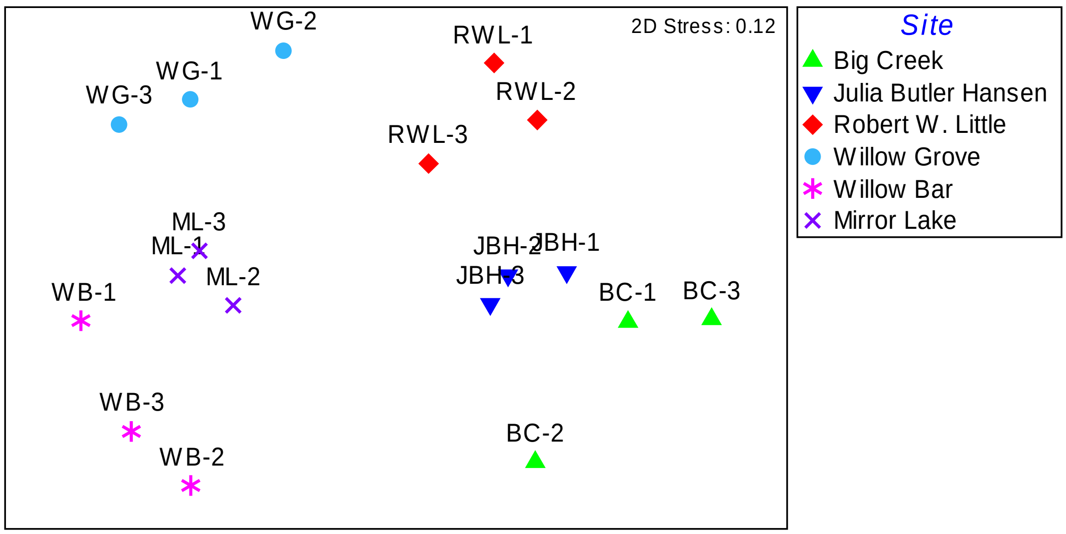


Figure 92. NMDS ordination plot of vegetation presence/absence survey data.

SIMPER analysis of the Sorensen site similarity matrix provided both an average percent similarity of sites studied and the contribution of individual species to site similarities (Table 40). All sites were less than 50% similar to one another in terms of species composition. Generally, sites that are located closer to one another within the estuary are more similar to one another than sites located farther from one another. This finding was confirmed by the use of a Mantel test, which showed a significant correlation between the geographic location of study sites and their vegetation community (p = 0.001).

Table 40. Average percent similarity of forested wetland study sites.

	Big Creek	Julia Butler Hansen	Robert W. Little	Willow Grove	Willow Bar	Mirror Lake
Big Creek						
Julia Butler Hansen	41.08					
Robert W. Little	31.79	38.24				
Willow Grove	9.46	24.90	30.06			
Willow Bar	10.73	22.17	16.56	26.60		
Mirror Lake	18.15	27.22	28.36	41.87	42.54	

Vegetation assemblages were also examined by site and zone, and the resulting ordination plot had a two-dimensional stress of 0.14 (Figure 93). The NMDS ordination plot indicates that lower estuarine sites grouped together (Big Creek, Julia Butler Hansen, and Robert W. Little) while mid- and upper estuarine sites (Willow Grove, Willow Bar, and Mirror Lake) also grouped together. Additionally, within the lower-estuarine sites, the forest zones, scrub-shrub zones, and emergent zones plotted close to one another in ordination space. Similarly, zones within the mid- and upper estuarine group of sites were located near each other in the ordination plot. An ANOSIM test of the vegetation data by site and zone showed that there were significant differences between vegetation assemblages in sites and zones (global R = 0.446, p = 0.003).

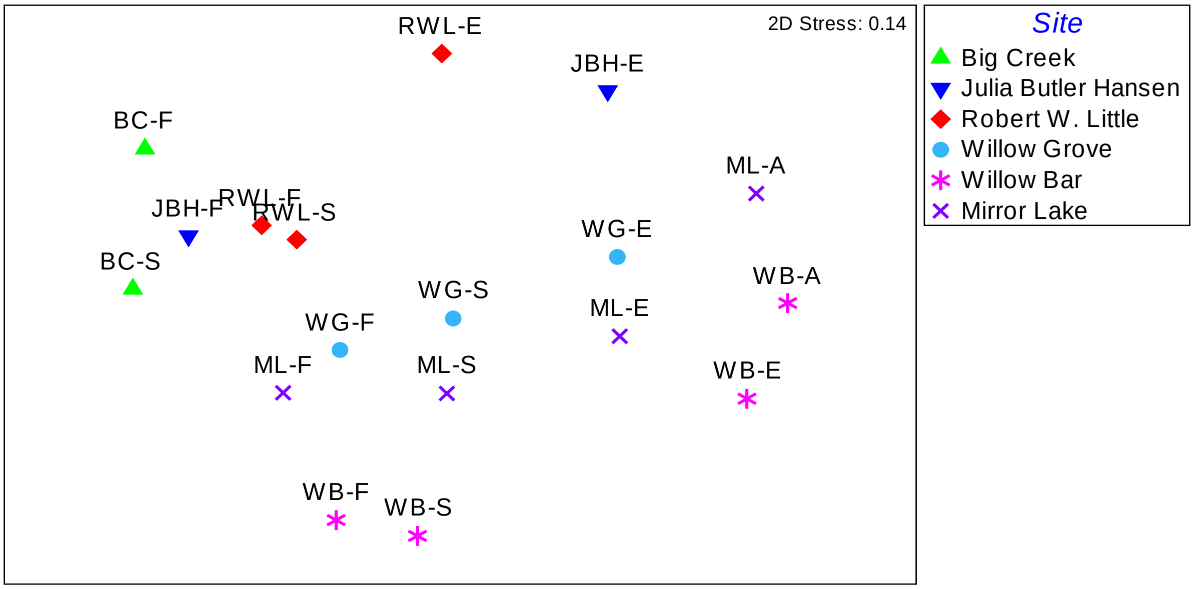


Figure 93. NMDS ordination plot of vegetation assemblages by site and zone and tidal forested wetland sites. Zones are designated as follows: F = Forest; S = Scrub-shrub; E = Emergent; A = Aquatic.

The SIMPER analysis of vegetation assemblages by zone and estuary location (lower estuarine versus upper estuarine) was performed in order to identify the species that contribute the most in each group. Based on the NMDS ordination results and plots of tree composition at sites (Figure 89 through Figure 93), the Lower Estuary group included sites Big Creek, Julia Butler Hansen, and Robert W. Little, while the Upper Estuary Group included Willow Grove, Willow Bar, and Mirror Lake. Zones were as follows: Lower Emergent, Lower Scrub-shrub, Lower Forest, Upper Aquatic, Upper Emergent, Upper Scrub-shrub, and Upper Forest (Table 41).
All vegetation species present at sites were classified as to native or non-native, according to regional field guides (Pojar and MacKinnon, 2004; Spear Cooke, 1997). Non-native vegetation species within the Columbia River estuary comprised 18% of the documented 110 total vegetation species (Table 41).

Table 41. Composition of lower and upper estuarine vegetation zones based on SIMPER analysis. Non-native species are denoted with an asterisk (*) after the common name.

Group (Estuarine Location and Vegetation Zone)	Species	Percent Contribution
Lower Emergent	*Carex obnupta* (slough sedge)	28.36
	Phalaris arundinacea (reed canary grass)*	28.36
	Solanum dulcamara (European bittersweet)	11.36
	Sagittaria latifolia (wapato)	9.54
	Impatiens noli-tangere (yellow touch-me-not)	7.46
	Juncus effusus (common rush)	7.46
Lower Scrub-Shrub	*Cornus sericea* (red osier dogwood)	14.36
	Phalaris arundinacea (reed canary grass)*	14.36
	Rubus ursinus (dewberry)	14.36
	Athyrium filix-femina (lady fern)	6.04
	Galium trifidum (small bedstraw)	6.04
	Salix lucida ssp. Lasiandra (Pacific willow)	6.04
	Spiraea douglasii ssp. Douglasii (hardhack)	6.04
	Carex obnupta (slough sedge)	3.64
	Climacium dendroides (tree moss)	3.64
	Equisetum fluviatale (swamp horsetail)	3.64
	Impatiens noli-tangere (yellow touch-me-not)	3.64
	Polystichum munitum (sword fern)	3.64
	Rosa nutkana (Nootka rose)	3.64
	Rubus discolor (Himalyan blackberry)*	3.64
Lower Forest	*Athyrium filix-femina* (lady fern)	7.45
	*Carex obnupt*a (slough sedge)	7.45
	Climacium dendroides (tree moss)	7.45
	Cornus sericea (red osier dogwood)	7.45
	Rubus discolor (Himalyan blackberry)*	7.45
	Rubus ursinus (dewberry)	7.45
	Ribes lacustre (black swamp gooseberry)	3.92
	Galium trifidum (Small bedstraw)	3.80
	Phalaris arundinacea (reed canary grass)*	3.80
	Adiantum pedatum (maidenhair fern)	3.32
	Alnus rubra (red alder)	3.32
	Lythrum salicaria (purple loosestrife)*	3.32
	Physocarpus capitatus (pacific ninebark)	3.32

	Picea sitchensis (Sitka spruce)	3.32
	Polystichum munitum (sword fern)	3.32
	Vaccinum parvifolium (red huckleberry)	3.32
	Iris pseudacorus (yellow-flag iris)*	1.38
	Rosa pisocarpa (peafruit rose)	1.38
	Angelica genuflexa (kneeling angelica)	1.33
	Heracleum lanatum (cow parsnip)	1.33
	Populus balsamifera ssp. Trichocarpa (black	1.33
	Salix lucida ssp. Lasiandra (Pacific willow)	1.33
	Spiraea douglasii ssp. Douglasii (hardhack)	1.33
	Alectoria sarmentosa (common witch's hair)	1.13
Upper Aquatic	*Eleocharis palustris* (creeping spikerush)	25.00
	Ludwigia palustris (water purslane)	25.00
	Polygonum hydropiper (waterpepper)	25.00
	Sagittaria latifolia (wapato)	25.00
Upper Emergent	*Hippurus vulgaris* (common marestail)	25.00
	Phalaris arundinacea (reed canary grass)*	25.00
	Polygonum hydropiper (waterpepper)	25.00
	Sagittaria latifolia (wapato)	25.00
Upper Scrub-	*Amelanchier alnifolia* (serviceberry)	16.67
	Cornus sericea (red osier dogwood)	16.67
	Equisetum fluviatale (swamp horsetail)	16.67
	Phalaris arundinacea (reed canary grass)*	16.67
	Populus balsamifera ssp. Trichocarpa (black	16.67
	Rubus discolor (Himalyan blackberry)*	16.67
Upper Forest	*Cornus sericea* (red osier dogwood)	14.29
	Fraxinus latifolia (Oregon ash)	14.29
	Phalaris arundinacea (reed canary grass)*	14.29
	Populus balsamifera ssp. Trichocarpa (black	14.29
	Rosa nutkana (Nootka rose)	14.29
	Rubus discolor (Himalyan blackberry)*	14.29
	Salix lucida ssp. Lasiandra (Pacific willow)	14.29

Avifauna. A total of 88 avian species were observed at the study sites during all seasonal observations (see Johnson 2010 for a complete list of avian species identified during field surveys). NMDS ordination of the avian presence/absence survey data showed that avifauna occurrence in transects within sites tended to be similar to one another, while avian assemblages differed among sites (Figure 94). The ordination plot also revealed that the lower to mid-estuarine sites of Julia Butler Hansen, Robert W. Little, and Willow Grove tended to be similar to one another. The two upper estuarine sites, Willow Bar and Mirror Lake, plot close to one another in ordination space. Finally, the lowest estuarine site, Big Creek, appears to be distinct from the all other sites. The ANOSIM test of the avian data showed that there were significant differences in species present among forested wetland sites (global R = 0.895, p = 0.001).

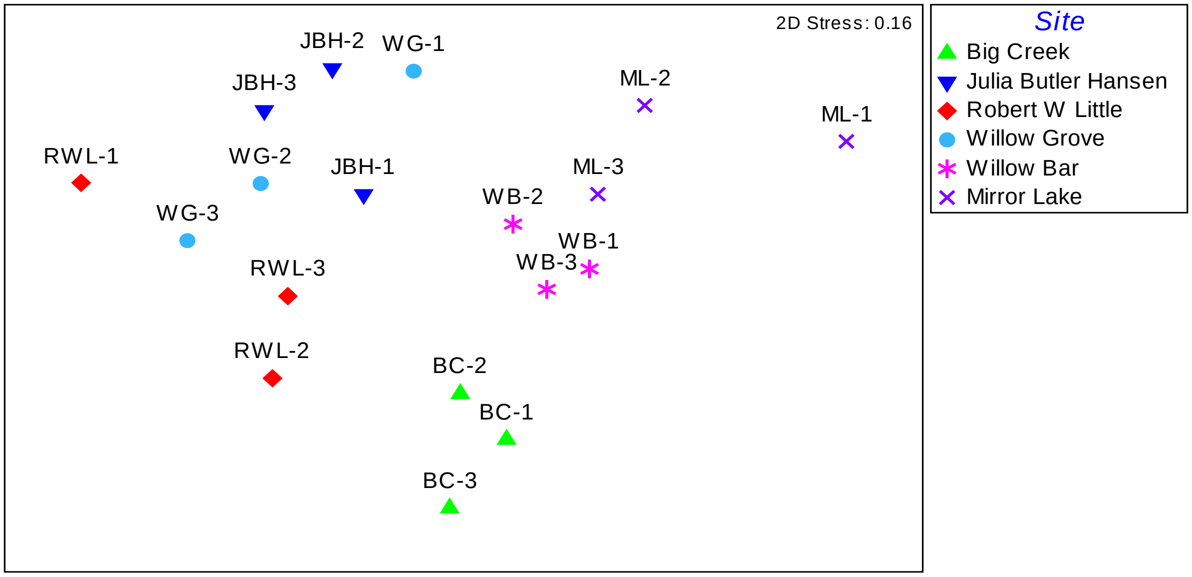


Figure 94. NMDS ordination plot of avifauna assemblage composition at tidal forested wetland sites.

Insects. A total of 87 insect taxa were observed at the forested wetland sites (see Johnson 2010 for a complete list of insect species identified during field surveys). NMDS ordination of the insect presence/absence survey data indicated that insect composition and abundance in transects within sites was generally similar to one another (Figure 95). Sites were somewhat distinct from one another, with a slight trend along the estuarine gradient. The trend is most visible over the lower to mid-estuarine sites, including Big Creek, Julia Butler Hansen, Robert W. Little, and Willow Grove. The two upper estuarine sites, Willow Bar and Mirror Lake, were similar to one another but also appear to be similar to the mid- and lower estuarine sites and transects. The two-dimensional stress for the NMDS ordination plot is 0.19, which is relatively high. A 3-dimensional ordination plot had lower stress at 0.13, but is not shown here due to difficulties in displaying 3-dimensional data. Further testing using the ANOSIM technique revealed that the insect assemblage composition of sites is statistically significant (global R = 0.592, p = 0.001).

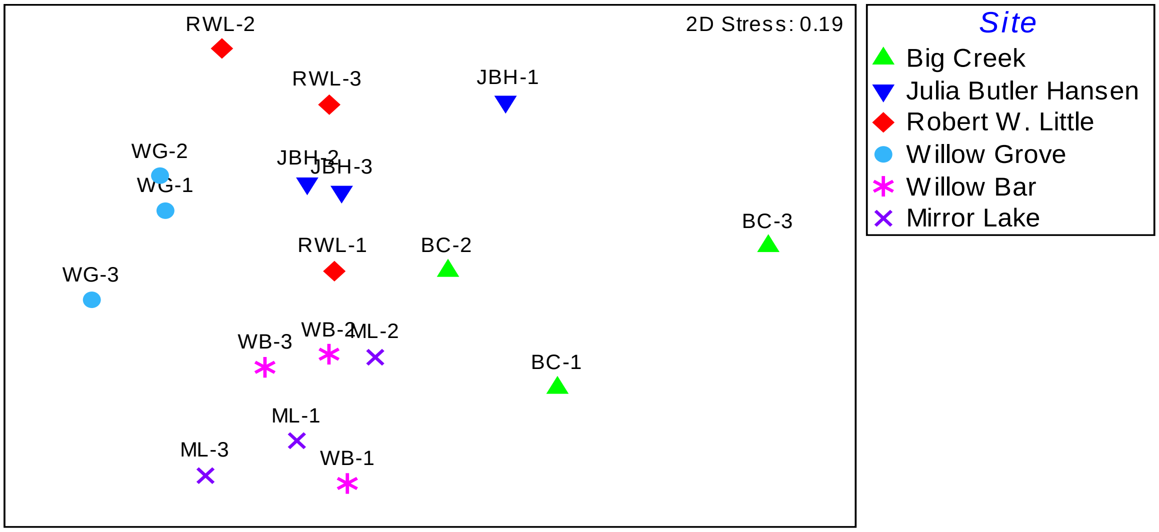


Figure 95. NMDS ordination plot of insect assemblage composition at tidal forested wetland sites.

SIMPER analysis revealed which insect groups constituted the majority (at least 4%) of the assemblages at tidal forested wetland study sites. The diversity of insect assemblage composition by order increased along the estuarine gradient (Table 42). At Big Creek, the majority of the insect assemblage was composed of only two insect orders, while the majority of the insect assemblage contained 10 separate orders of insects at Willow Bar in the upper estuary. Similarly, insect species richness increased along the estuarine gradient (Figure 96). Insect species richness appears to be correlated with habitat complexity, or the presence of additional vegetation zones as seen in the upper estuary (Figure 88, Figure 96).

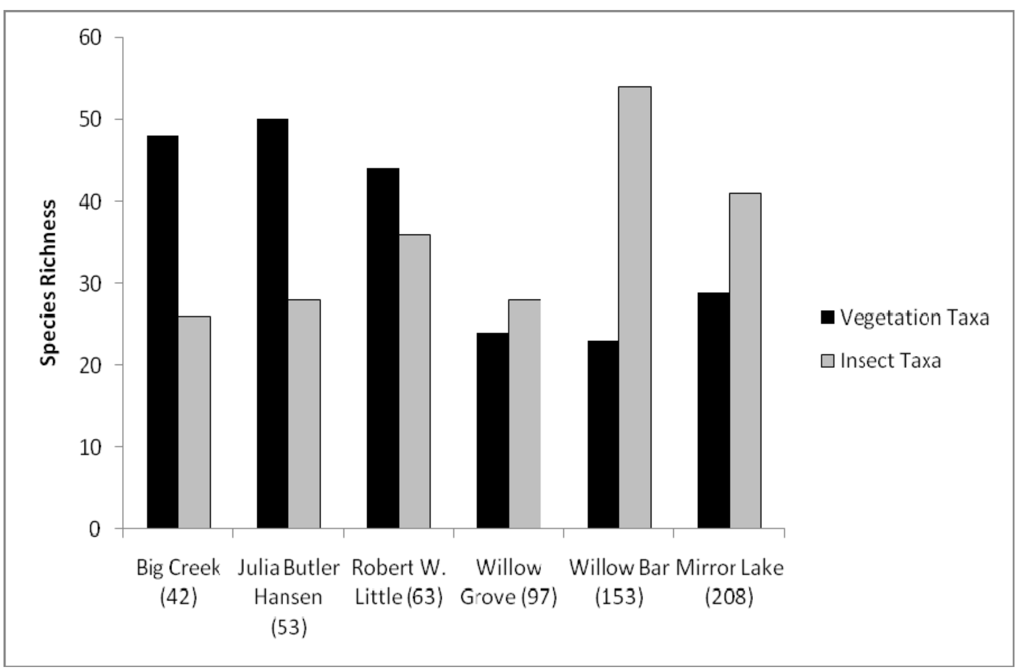


Figure 96. Vegetation and insect species richness at tidal forested wetland sites. Values in parentheses along the x-axis indicate site RKm.

Table 42. Insects composing the majority of the insect assemblages at tidal forested wetland study sites based on SIMPER analysis.

Insect Order and Family	Big Creek	Julia Butler Hansen	Robert W. Little	Willow Grove	Willow Bar	Mirror Lake
Acari		X	X	X	X	X
Araneae			X	X		X
Coleoptera Carabidae					X	
Coleoptera Tenebrionidae					X	
Collembola Entomobryiidae	X	X				X
Collembola Isotomidae	X	X	X	X	X	
Collembola Sminthuridae	X	X	X	X	X	X
Diptera Cecidomyiidae	X					X
Diptera Ceratopogonidae	X	X	X		X	X
Diptera Chironomidae	X	X	X	X	X	
Diptera Culicade					X	X
Diptera Dolichopodidae		X			X	X
Diptera Ephydridae		X			X	X
Diptera Ptychopteridae						X
Diptera Sciaridae					X	
Diptera Sphaeroceridae					X	
Diptera Tipulidae	X				X	X
Hemiptera Miridae					X	
Homoptera Cicadellidae			X	X	X	X
Hymenoptera Chalcidoidea					X	
Hymenoptera Formicidae				X		X
Hymenoptera Tenthredinoidea				X		
Psocoptera			X		X	
Thysanoptera Thripidae		X	X	X		X
Zoroptera					X	

Benthic Macroinvertebrates. Ten benthic macroinvertebrate taxa were found at the tidal forested wetland study sites (see Johnson (2010) for a complete list of benthic macroinvertebrate species identified during field surveys). Of these, two taxa, Oligochaeta and Nematoda, composed the vast majority of all benthic macroinvertebrates collected (96%). An NMDS ordination plot of the benthic macroinvertebrate abundance data indicates that transects within sites tended to be relatively similar to one another, but site similarity did not follow a consistent trend along the estuarine gradient (Figure 97). The two-dimensional stress of the ordination plot was moderate (0.14). An ANOSIM test showed that benthic macroinvertebrate assemblage was significantly different among sites (global R = 0.31, p = 0.011). However, due to the low number of taxa found at the tidal forested wetland sites and the lack of statistical significance in the ANOSIM test, a SIMPER test was not performed on this dataset.

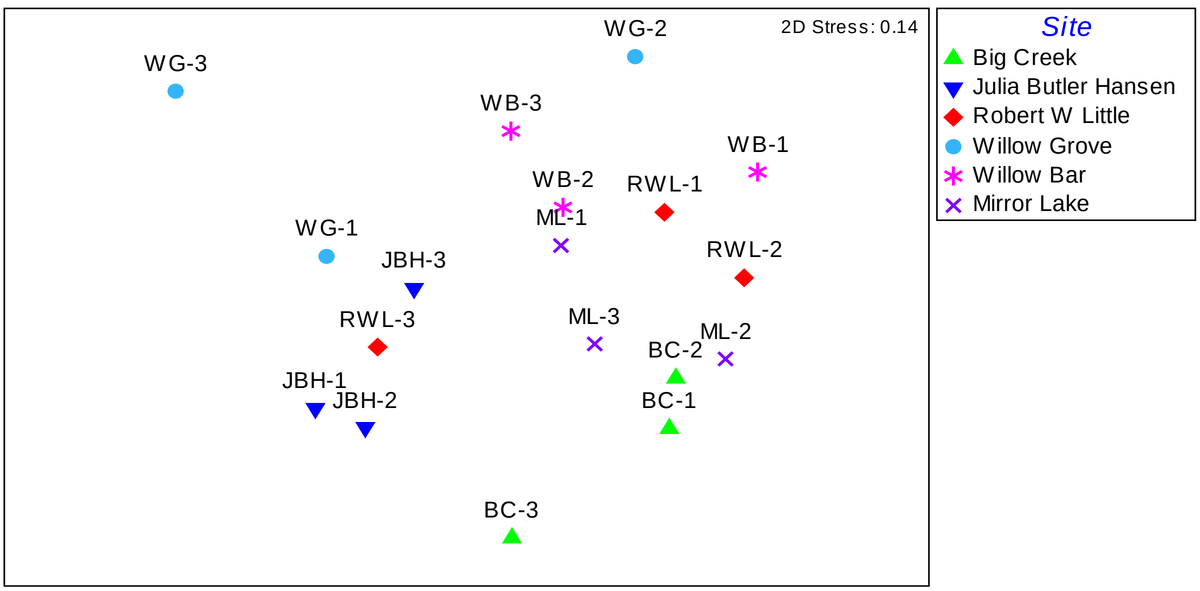


Figure 97. NMDS ordination plot of benthic invertebrate assemblage composition at forested wetland study sites.

Amphibians. A variety of amphibian species were observed at the tidal forested wetland sites, although sample numbers were not sufficient to perform statistical analysis on the dataset. As a result, the taxa observed at the study sites are presented in tabular format in Table 43.

Table 43. Amphibian species observed at forested wetland study sites.

Amphibian species scientific name(common name)	**Big Creek**	**Julia Butler Hansen**	**Willow Grove**	**Willow Bar**	**Mirror Lake**
Taricha granulosa (rough-skinned newt)	X				
Rana aurora (Northern red-legged frog)	X				
Pseudacris regilla (Pacific tree frog)	X		X	X	
Rana luteiventris (Columbia spotted frog)	X	X			
Amybstoma gracile (Northwestern salamander)	X				X
Rana catesbeiana (American bullfrog)					X

Mammals. Similar to the amphibian survey, the mammalian survey resulted in too low of sample numbers to perform statistical analysis on the dataset. As a result, the mammalian taxa observed at the study sites are presented in tabular form in Table 44. The Northern raccoon, *Procyon lotor*, was the most commonly detected species at the tidal forested wetland sites. The Columbian white-tailed deer, which is listed as endangered in the U.S. Endangered Species Act, was seen at two sites in the lower estuary. One of the two sites, the Julia Butler Hansen Wildlife Refuge, was established specifically to conserve this species.

Table 44. Mammalian species observed at tidal forested wetland study sites.

Mammalian species scientific name (common name)	**Big Creek**	**Julia Butler Hansen**	**Robert W. Little**	**Willow Grove**	**Willow Bar**	**Mirror Lake**
Ondatra zibethicus (Muskrat)	X					
Procyon lotor (Northern raccoon)	X	X	X		X	
Lontra canadensis (River otter)	X					X
Odocoileus virginianus leucurus (Columbian white-tailed deer)		X	X			
Castor canadensis (Beaver)				X	X	X
Sylvilagus sp. (Rabbit)				X		
Canis latrans (Coyote)					X	
Odocoileus hemionus columbianus (Black-tailed deer)					X	X
Cervus canadensis (Elk)					X	

Environmental Factors. Based on ANOVA, sites and zones were significantly different in the mean percent sand, silt, and organic content of soils (Table 45). The mean percent clay and mean elevation of zones were not significantly different from one another. A post-hoc Bonferroni pairwise test indicated significant differences in mean sand and mean silt content of soils among some sites (Table 46and Table 47). Although the ANOVA showed percent organic content to be statistically significant, the post-hoc Bonferroni pairwise test did not show any significant differences among specific sites. Significant differences in the mean sand content were present among Willow Bar and Julia Butler Hansen, Robert W. Little, and Willow Grove. The mean sand content at Willow Grove and Mirror Lake was also statistically different from one another. Significant differences in mean silt content were present when comparing Julia Butler Hansen to Willow and Mirror Lake, and when comparing Willow Grove to Willow Bar.

Patterns in soil composition at sites reveal some trends across the estuarine gradient (Figure 98). The soils at the upper estuarine sites (Willow Bar and Mirror Lake) had relatively high mean percent sand, while lower estuary sites tended to have higher mean percent silt and organic content. The mean percent clay at zones within forested wetland sites was relatively low (less than 24% of soil composition) at all sites but fluctuated throughout the estuary. Although the mean elevation of zones within forested wetlands sites were not significantly different from one another, the scrub-shrub and forested zones of Mirror Lake in the upper estuary are higher than all other zones within sites (Figure 99).

Table 45. Results of ANOVA test on environmental characteristics of zones within sites. Statistically significant P-Values are presented in bold text.

Environmental Factor	Source of Variation	Sum of Squares	df	Mean Square	F Value	P Value
Percent Sand	Among Groups	7356.622	5	1471.324	8.203	**0.001**
	Within Groups	2152.279	12	179.357		
	Total	9508.901	17			
Percent Silt	Among Groups	2120.188	5	424.038	9.871	**0.001**
	Within Groups	515.482	12	42.957		
	Total	2635.670	17			
Percent Clay	Among Groups	125.789	5	25.158	0.669	0.654
	Within Groups	451.196	12	37.600		
	Total	576.985	17			
Percent Organic Content	Among Groups	2562.568	5	512.514	3.185	**0.046**
	Within Groups	1931.053	12	160.921		
	Total	4493.621	17			
Elevation	Among Groups	41.379	5	8.276	1.959	0.158
	Within Groups	50.685	12	4.224		
	Total	92.064	17			

Table 46. P-values of ANOVA with post-hoc Bonferroni test for mean sand content of soils at study sites. Highlighted values denote statistically significant values at the 95% confidence level.

	Big Creek	Julia Butler Hansen	Robert W. Little	Willow Grove	Willow Bar	Mirror Lake
Big Creek						
Julia Butler Hansen	1.000					
Robert W. Little	1.000	1.000				
Willow Grove	1.000	1.000	1.000			
Willow Bar	0.237	0.018	0.039	0.002		
Mirror Lake	1.000	0.232	0.717	0.031	1.000	

Table 47. P-values of ANOVA with post-hoc Bonferroni test for mean silt content of soils at study sites. Highlighted values denote statistically significant values at the 95% confidence level.

	Big Creek	Julia Butler Hansen	Robert W. Little	Willow Grove	Willow Bar	Mirror Lake
Big Creek						
Julia Butler Hansen	0.123					
Robert W.	1.000	0.095				

Little						
Willow Grove	1.000	0.260	1.000			
Willow Bar	0.199	0.000	0.068	0.021		
Mirror Lake	1.000	0.006	1.000	0.667	0.960	

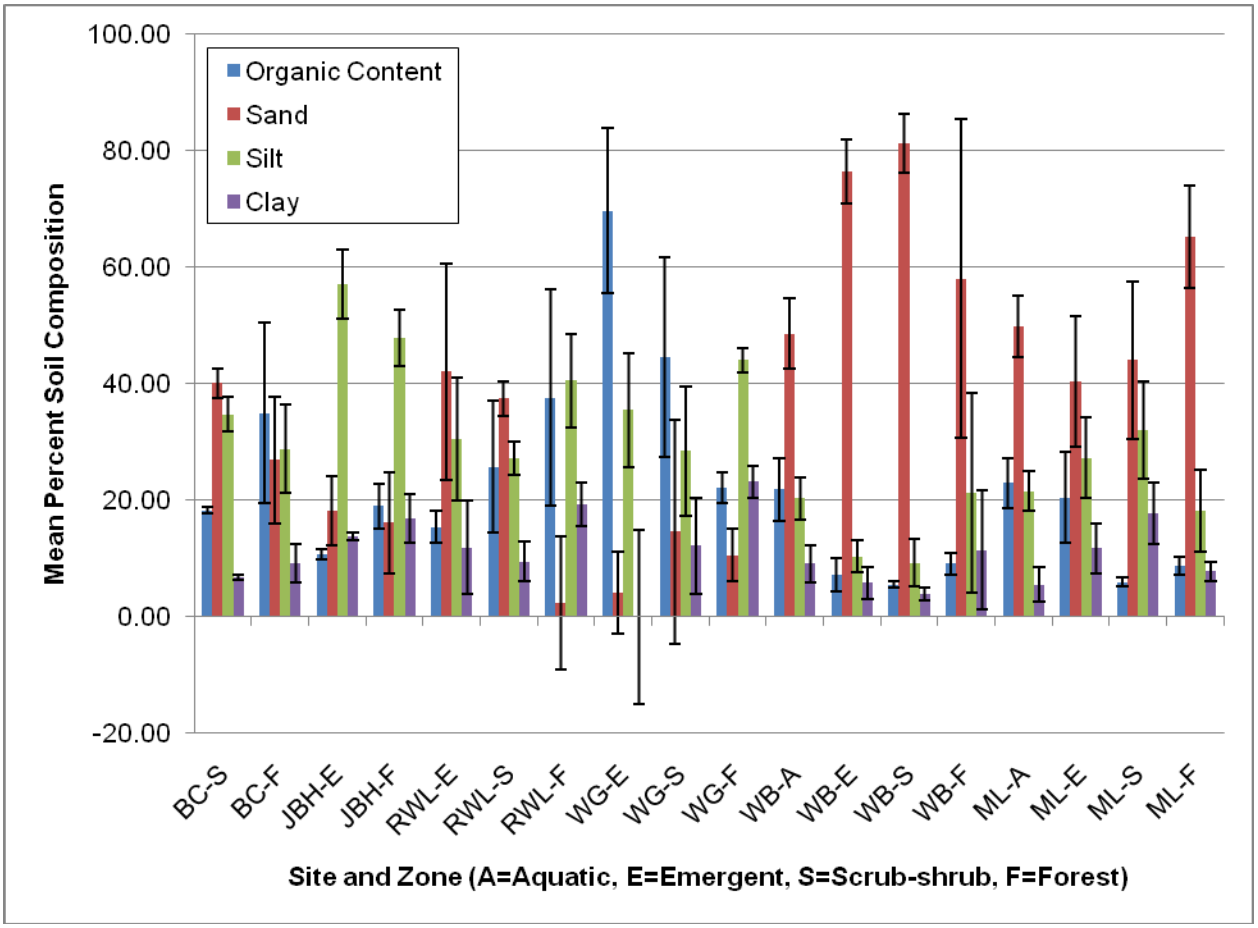


Figure 98. Mean percent organic content, sand, silt, and clay at zones within forested wetland sites. Error bars represent +/- 1 standard error of the mean. Site abbreviations are as follows: BC = Big Creek; JBH = Julia Butler Hansen; RWL = Robert W. Little, WG = Willow Grove; WB = Willow Bar; ML = Mirror Lake. Letters following sites indicate vegetation zones: A=Aquatic, E=Emergent, S=Scrub-Shrub, F=Forest.

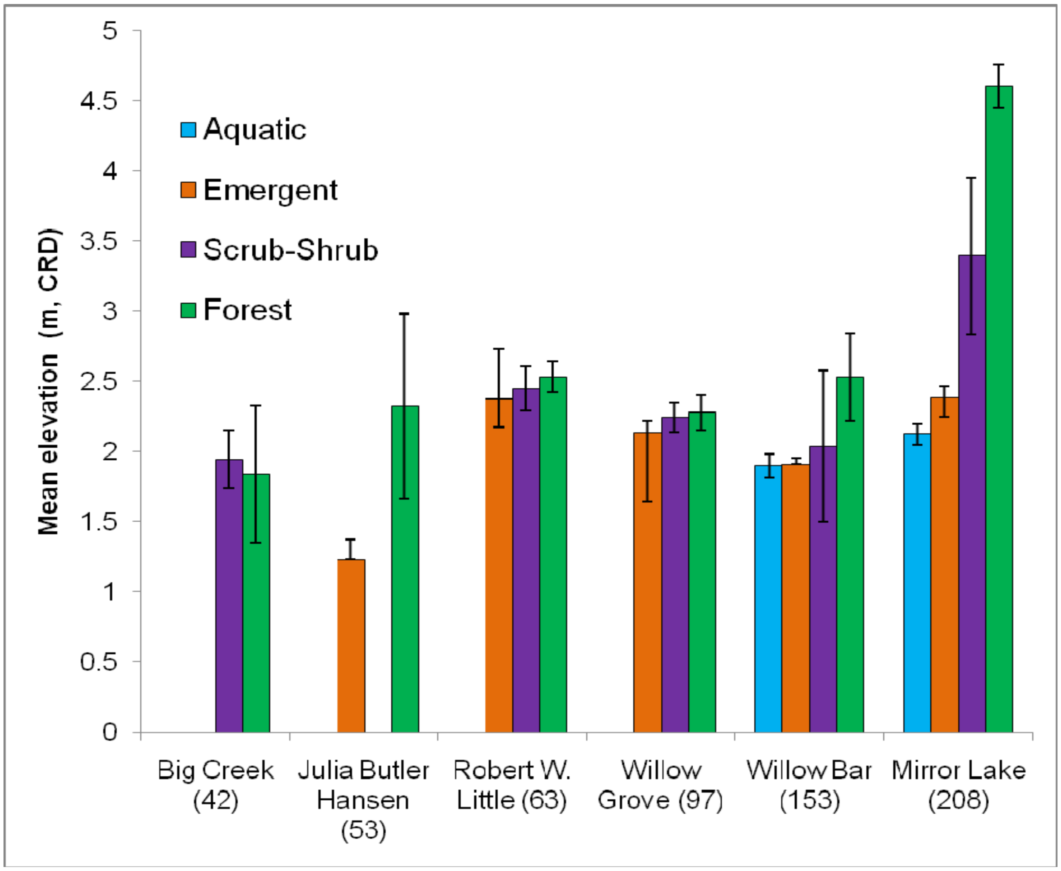


Figure 99. Mean elevation (m relative to CRD) of zones within tidal forested wetland sites. Error bars represent +/- 1 standard error of the mean. Site abbreviations are as follows: BC = Big Creek; JBH = Julia Butler Hansen; RWL = Robert W. Little, WG = Willow Grove; WB = Willow Bar; ML = Mirror Lake. Letters following sites indicate vegetation zones: A=Aquatic, E=Emergent, S=Scrub-Shrub, F=Forest.

12.5 Discussion

Vegetation Assemblages. Freshwater tidal forested wetland vegetation assemblages changed dramatically over the length of the Columbia River estuary, in terms of species present, species richness, and structure. Sites in the lower estuary were characterized by having fewer vegetation zones, primarily emergent, scrub-shrub, and forest zones, and higher species richness within these zones than sites in the mid- and upper estuary (Figure 88). Big Creek, Julia Butler Hansen, and Robert W. Little had complex assemblages of species within each zone, and each meter along the transect tended to have many species present. In contrast, sites in the mid- and upper estuary, including Willow Grove, Willow Bar, and Mirror Lake, had lengthy vegetation zones, particularly emergent zones that included only several species. Two non-native species, reed canary grass and yellow-flag iris, were prevalent at these sites and formed quite dense, sometimes monotypic stands when present. Reed canary grass in particular was present at every site in the estuary, and often in every zone within a site. The broad environmental tolerances of this grass species, which was introduced intentionally within the estuary for cattle grazing (Christy and Putera, 1992) appears to allow it to thrive well in sunny emergent and aquatic zones and as an understory species, although less dominant, in scrub-shrub and forested zones (Hovick and Reinartz, 2007). These results are similar to those of Lavoie et al. (2003), who found that freshwater tidal forested wetlands in the lower St. Lawrence River estuary had lower numbers of exotic species than wetlands further up the river (above the reach of tides). In particular, they noted that purple loosestrife and reed canary grass were extensive in the upper portions of the river, and hypothesized that regular tidal freshwater inundation makes establishment of non-native species much more difficult.

All aspects of vegetation assemblages studied, including species and types of vegetation zones present, composition of canopy cover, tree density, and tree DBH showed trends across the estuarine gradient. The major shift in the structure and composition of vegetation assemblages appears to occur between Robert W. Little and Willow Grove, or between Puget Island, WA and Longview, WA. Interestingly, this is the portion of the estuary where the dominant tree species in the forested portions of the wetlands transitions from Sitka spruce to black cottonwood. At the lower estuarine sites where Sitka spruce is the dominant tree species, a greater variety of scrub-shrub and tree species contribute to the canopy cover than in the mid- and upper estuary, despite the fact that the lower estuarine sites have a greater proportion of large trees in terms of DBH (Figure 90 and Figure 91) . However, the amount of sky visible in the forested plots did not show any discernible trends across the estuary. In the mid- and upper estuary, canopy cover is comprised of fewer species and typically only tree species rather than a mix of trees and scrub-shrubs. The sites in this portion of the river tended to have a greater density of grasses and sedges as the understory in forested plots, while the lower estuarine sites had large scrub-shrubs such as red osier dogwood and vine maple growing in the understory. The lower estuary had up to seven total species contributing to canopy cover in the forested plots, while in the mid- and upper estuary, a maximum of three species contributed to canopy cover. Tree density (number of trees per site and per forested plot) showed striking differences across the estuarine gradient (Figure 89; Table 39). The lower estuarine sites, Big Creek, Julia Butler Hansen, and Robert W. Little, had fewer trees per plot and per site than the mid- and upper estuarine sites. Values ranged from a mean of 5.67 trees per 100 m^2 in the lower estuary (at Julia Butler Hansen) to as high as 17.0 trees per 100 m^2 in the upper estuary (at Willow Bar). In contrast, the proportion of sky visible in forested plots ranged from 5 to 50% but did not show a consistent trend across the estuarine gradient. Therefore, since the proportion of sky visible in the canopy of forested plots did not show much variation across the estuarine gradient (Figure 91), individual trees in the forested plots in the lower estuary likely provide more canopy cover per tree than more numerous, smaller trees in the mid- and upper estuary.

The transition in vegetation communities along the estuarine gradient appears to correspond to hydrogeomorphic reaches described in the Columbia River Estuary Ecosystem Classification (CREEC) (Simenstad et al., In revision). According to the Classification, the Columbia River estuary can be divided into eight hydrogeomorphic reaches that are the result of hydrologic processes and geomorphologic formation of the estuarine floodplain (Figure 100). The transition seen in vegetation community composition and structure along the estuarine gradient is evidence for the strong influence of hydrology and geomorphology on floodplain forests. We identified Willow Grove (RKm 97) as transitional within the Columbia River estuary in terms of forested wetland composition and structure. This site lies in the upper portion of Reach C (Volcanoes Current Reversal), as having a shift in tidal influence over the length of the reach. The upper portion of Reach C, where Willow Grove is located, is the area of the river where tidal influence diminishes and fluvial hydrology plays a larger role in the estuarine floodplain.

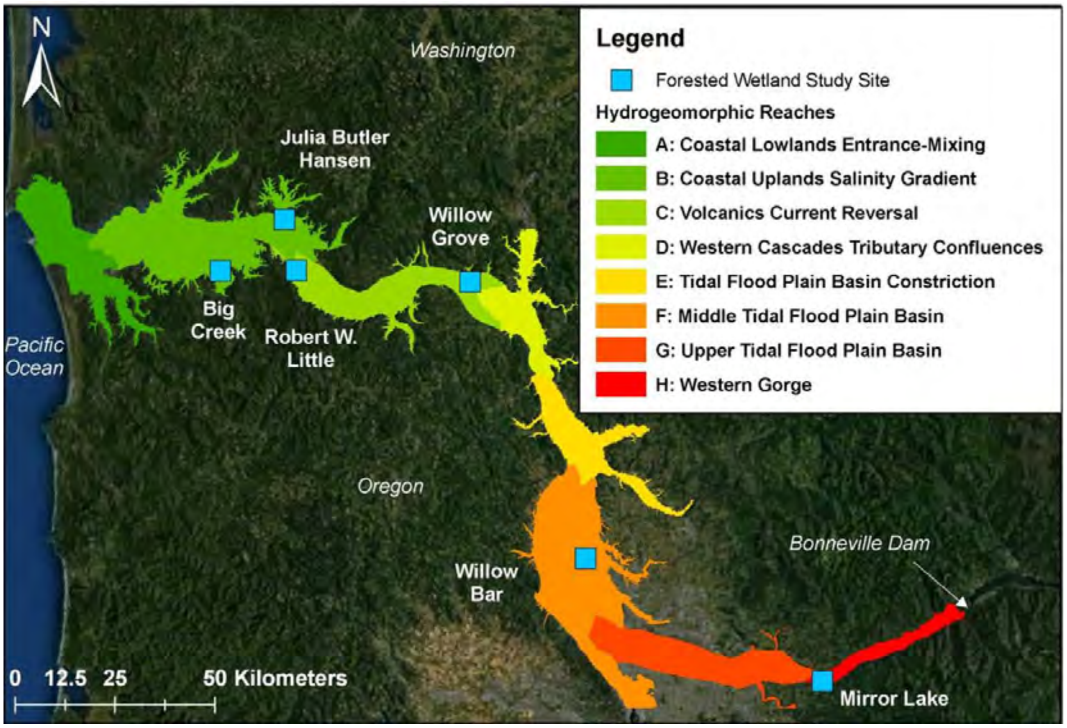


Figure 100. Hydrogeomorphic reaches of the Columbia River estuary (Data courtesy of Jennifer Burke, University of Washington. Imagery is ESRI World Imagery, December 2009.

In summary, the vegetation assemblages in the lower portion of the Columbia River estuary are different in both composition and structure from the assemblages in the mid- and upper portions of the estuary. Most likely, the variations are due to a suite of environmental factors, including flow velocity, site topography, frequency and duration of combined fluvial and tidal flooding inundation, and shifts in overall climate when moving away from the Pacific Ocean. The correlation between hydrogeomorphic reaches and biotic assemblages was confirmed by additional ANOSIM testing. Test results revealed significant differences in vegetation communities according to hydrogeomorphic reach ($p = 0.001$). Similarly, avifauna and insect assemblages are significantly different according to hydrogeomorphic reach ($p = 0.004$ and 0.018, respectively).

Vegetation assemblages in other river systems around the world appear to follow similar patterns of structure along environmental gradients. The Tana River floodplain in Kenya is composed of evergreen forests (*Acacia elatior*) in the lower portion of the river, and the riparian forest transitions to *Populus spp.* in the upper portion of the basin (Hughes, 1990). Flooding regimes, including frequency and duration, were found to control the location of riparian forest community types. Although all of the Tana River floodplain forests have limited tolerances to high frequency and duration of flooding, the *Populus spp.* forests were located in a portion of the river that experienced more flooding than the evergreen trees, since they require flooding for regeneration. The distribution of forest types in the Tana River floodplain thus parallels those of the Columbia River estuary, although the dominant species differ.

Extensive studies conducted on regulated and non-regulated rivers in northern Sweden point to shifts in vegetation that occur as a result of river regulation, including decreased species richness and cover, and shifts in composition of vegetation according to dispersal mechanism (Jansson et al., 2000a; Jansson et al., 2000b; Nilsson et al., 1997). Their finding that alteration to the natural flow regime of boreal rivers dramatically affects the associated riparian vegetation demonstrates the importance of river hydrology as a controlling factor of vegetation community composition and structure.

Thus, studies of river floodplain vegetation communities around the world illustrate the importance of hydrology and flooding regimes in determining the location, composition, and structure of riparian floodplain forests. Therefore, although a variety of environmental factors probably play a role in shaping the estuarine floodplain forests in the Columbia River system, discharge and flooding regimes are probably the most important.

Faunal Assemblages. In general, faunal assemblages did not show as distinct of a trend across the estuarine gradient as vegetation assemblages did. The avian and insect assemblages appeared to differ according to the estuarine gradient more than the benthic macroinvertebrate, amphibian, and mammalian assemblages (Figure 94, Figure 95, and Figure 97; Table 43, Table 44). Interestingly, although the vegetation assemblages at Willow Grove are most similar to the upper estuarine sites Willow Bar and Mirror Lake, the avian assemblages at Willow Grove are statistically most similar to the lower estuarine sites Julia Butler Hansen and Robert W. Little. This is likely a reflection of the shift in hydrogeomorphology at this point in the estuary, as described in the CREEC (Simenstad et al. In revision; Figure 100). Thus, the area around Willow Grove is a transitional area in the ecological community of the Columbia River estuary, with some physical and biological similarities to both the lower estuarine and upper estuarine sites.

The insect assemblages, which are often closely associated with vegetation and physical characteristics of sites (Lawton and Strong, 1981), showed trends in composition across the estuarine gradient in a pattern similar to the vegetation assemblages (Figure 95). Specifically, collembolans and dipterans formed the majority of the insect assemblages at the lower estuarine sites (Table 42). These results are generally in agreement with another recent study of insect emergence in the lower Columbia River estuary (Ramirez, 2008). These insects may have an association or preference for either the vegetation or physical factors (e.g., sediment structure, organic matter) at these sites. Since the tidal range is much greater at the lower estuarine sites than those in the upper estuary, these insect groups may have life history adaptations linked with the tidal inundation of the lower estuary, as other studies have showed (Saigusa and Akiyama, 1995). Diversity of insect groups increased in the upper estuary, and at Willow Bar, the insect assemblages consisted of a wide range of groups including coleopterans, hemipterans, homopterans, hymenopterans, as well as the collembolans and dipterans common in the lower estuary. This increase in insect assemblage diversity may be a result of the increase in types of vegetation zones present or a preference for the plant species at these sites (Figure 96). Additionally, physical factors such as sandy substrate, minimal tidal fluctuation, or climatic dissimilarity may play a role in the higher diversity of insect assemblages at the upper estuarine relative to lower estuarine tidal forested wetland sites.

Sampling of the benthic macroinvertebrate community revealed almost no differences in species found at study sites. This finding suggests that identification of benthic macroinvertebrates to a finer resolution taxonomic level might be necessary to detect trends across the estuary if present. The two most common groups of benthic macroinvertebrates present in samples across all sites were oligochaetes and nematodes, which is unsurprising since these two groups are common in many areas of the country (Pennak, 1953).

The mammalian community showed little variation in composition across the estuary (Table 44). Small sample sizes likely prevented the detection of a trend present across the estuarine gradient; however, it may also be that the same mammalian species inhabit different forested wetland sites. Similarly, the amphibian sampling effort yielded too small a sample size to make conclusions on an estuarine level (Table 43). However, since amphibian species observations were generally unique to one or two sites, this suggests that different forested wetland sites may support different amphibian species. Five amphibian species were observed at Big Creek, whereas only one or two species were seen at any other site within the estuary, which may indicate a more diverse amphibian community inhabits the lower estuary.

Thus, it appears that there are associations between vegetation assemblages of the forested wetland sites and the faunal assemblages present. The associations may be a direct result of faunal preference for particular vegetation assemblages for feeding and other habitat requirements, or the faunal and vegetation assemblages may be independently driven by the physical factors that govern the estuary.

Community Ecology Summary of Freshwater Tidal Forested Wetlands. The freshwater tidal forested wetlands in the Columbia River estuary appear to fall into two groups according to analyses of faunal and floral assemblages: lower estuarine forested wetlands, and mid- and upper estuarine forested wetlands. The lower estuarine forested wetland vegetation assemblages are dominated by coniferous species such as Sitka spruce and western red cedar and have associated scrub-shrub zones that are densely vegetated with a diverse group of large scrub-shrubs. The coniferous-dominated forested wetlands are utilized by multiple faunal assemblages. The avian assemblages includes a broad range of bird groups, including eagles, thrushes, sparrows, wrens, and warblers. Insect assemblages in the lower estuarine forested wetlands consist mainly of collembolans and dipterans, and nematodes and oligochaetes are the primary benthic macroinvertebrates present. Northern raccoons, river otters, Columbian white-tailed deer, and a variety of amphibians utilize these sites. Tides are the dominant hydrological regime affecting these sites on a daily basis, and the hydrological differences between the lower and upper estuarine sites may be a determining factor in the biota present at these sites (Fox et al., 1984).

The mid- and upper estuarine freshwater tidal forested wetlands have more diverse vegetation zones, with the forested zone dominated by deciduous trees, primarily black cottonwood, Oregon ash, and Pacific willow. All of the vegetation zones at these sites have lower species richness than the lower estuarine zones, and in some cases zones are monotypic in composition. The groups of birds present in the upper estuary are similar to those in the lower estuary, but specific species within groups differ significantly. The insect assemblages in the upper estuary have much more variable taxa composition compared to those in the lower estuary and include members of many insect orders. Beavers, black-tailed deer, elk, coyotes, and river otters were observed at forested wetland sites in the upper estuary. Although technically within the reach of tides in the Columbia River estuary, the seasonal and annual variations in river flow are the dominant flow regime affecting these sites (Fox et al., 1984; Kukulka and Jay, 2003). Little if any changes in hydrology at the sites occur on a daily basis, which may determine the biota present in the mid- and upper estuarine forested wetlands.

Implications and Recommendations for Future Research and Monitoring. Detailed information about the community ecology of the freshwater tidal forested wetlands of the Columbia River estuary from this study will likely be useful to both restoration efforts and the ecosystem classification. Recently, restoration of tidal forested wetlands has become a priority in the Columbia River estuary (LCREP, 1999). Reference sites for monitoring and assessing restoration performance are often lacking but are vital to the success of restoration projects (Brophy, 2009). The data gathered at these relatively unimpacted forested wetland sites in the Columbia River estuary may provide valuable information for restoration project managers during the design phase of restoration projects. In addition, the multi-faceted ecosystem classification of the Columbia River estuary currently underway (Simenstad et al. In revision); quantitative characterization of the vegetation along the length of the Columbia River estuarine gradient will add useful detail to the hydrogeomorphic reaches described in the classification system.

Due to hydroregulation of the Columbia River, freshwater tidal forested wetlands located in the estuary are directly affected by river management practices and the altered river flow regime (Simenstad et al., In revision). Ideally, river managers utilize available ecological data to practice adaptive management in order to manage river flow in the most sustainable method possible (Richter et al., 2003). Therefore, detailed ecological data about the ecosystems downstream from river impoundments is crucial for the implementation of adaptive management in a river system. The information provided by this study may be useful to river managers for managing the hydrologic impacts to the remaining freshwater tidal forested wetlands in the Columbia River estuary. For example, in other regulated river systems, managers have altered river flow in order to facilitate establishment and survival of riparian tree species (Rood et al., 2005). Regulating to ensure peak, or even higher/more variable, spring flows continue to occur may help flood-dependent species such as cottonwood and willow thrive in the mid- and upper estuary where fluvial hydrology dominates (Sherwood et al., 1990).

In addition, this study could potentially serve as baseline information for future research projects focusing on specific sites within the estuary, or on particular species or assemblages present at the forested wetland sites. Baseline ecological information is important in future studies for comparison and change analyses, especially given the relatively short period of time since hydroregulation of the Columbia River began (Simenstad et al., 1992). Ecological studies such as this are critical to our understanding of how alterations to ecosystems, ecosystem processes, the climate and natural flow regimes impact specific estuarine ecosystems.

13.0 Planned Ecosystem Monitoring Project Efforts for 2010-2011

For a summary of the activities in 2009-2010, see the Executive Summary. In 2010-2011, UW and USGS will complete the remaining Level 4 and Level 5 mapping, while the Estuary Partnership will coordinate the completion of Level 6 land cover data. On-the-ground data collection in 2010-2011 is anticipated to include vegetation, water chemistry relevant to salmonids, primary productivity, secondary productivity and salmon in TBD Reach(es) of the LCRE. Monitoring partners will continue to work closely to ensure efforts are not duplicated and resources can be shared to maximize the efficiency of the EMP. Monitoring partners will synthesize multi-year datasets for emergent wetland sites (vegetation, fish, fish prey and water quality) in order to characterize undisturbed emergent wetlands as juvenile salmon habitat in the LCRE. OHSU will conduct primary and secondary productivity sampling at all fixed sites (3 previously sampled sites and one new fixed site in Reach A). Water quality sampling (USGS) will expand to include all fixed sites.

14.0 EMP Budget

Table 48: Budget for Estuary Partnership's EMP contract (#45816), including the USGS EMP contract (#44032*).*

BPA Project Number: 2003-007-00
Contract Numbers: Estuary Partnership #45816, USGS #44032
Performance/Budget Period: September 1, 2009 – November 15, 2010

Budget Items	Contract Amount	Funds Received To Date	Contract Balance
I. Direct Costs			
Personnel	$ 117,559.00	$ 98,026.68	$ 19,532.32
Travel	$ 3,088.00	$ 509.25	$2,578.75
Office Supplies	$ 2,618.00	$ 2,618.00	-
Ground Transportation	$ 2,070.00	$ 651.93	$ 1,418.07
Project Supplies/Equipment	$ 17,003.00	$ 17,003.00	-
Rent/Utilities	$ 10,228.00	$ 10,248.81	$ (20.81)
Sub Total	*$ 152,566.00*	*$ 129,057.67*	*$ 23,508.33*
Overhead	$ 30,513.00	$ 25,811.53	$ 4,701.47
Sub Total Direct Costs	*$ 183,079.00*	*$ 154,869.20*	*$ 28,209.80*
II. Sub Contracts			
Battelle	$ 112,362.00	$ 112,362.00	-
Univ. of Washington	$ 82,164.00	$ 80,234.56	$ 1,929.44
NOAA	$ 103,385.00	$ 103,385.00	-
USGS	$ 131,495.00	$ 131,495.00	-
David Evans & Associates	$ 170,865.00	$ 152,867.10	$ 17,997.90
Sanborn Map Company	$ 120,000.00	$ 120,000.00	-
Sub Contracts Sub Total	*$ 720,271.00*	*$ 700,343.66*	*$ 19,927.34*
Project Management	$ 71,650.00	$ 71,650.00	-
Totals	**$ 975,000.00**	**$926,862.86**	**$ 48,137.14**

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Appendix A. Site elevation (in meters, relative to the Columbia River vertical datum CRD) and vegetation species average percent cover. The three dominant cover classes are bolded in red for each site and the invasive species are shaded in yellow (not necessarily non-native species). Species are sorted by their four letter code (1st two letters of genus and 1st two letters of species).

Code	Scientific Name	Common Name	Wetland Status	Native	Jackson Is.	Whites Is.	Wallace Is.	Campbell Slough	Cunningham Lake
					Elevation (m, CRD)				
				Min	0.90	1.56	0.86	1.15	0.99
				Avg	1.33	1.90	1.52	1.67	1.37
				Max	1.88	2.30	2.44	2.71	1.72
Code	**Scientific Name**	**Common Name**	**Wetland Status**	**Native**	**Average Percent Cover**				
AGEX	Agrostis exarata	spike bentgrass	FACW	yes	0.00	1.63	0.00	0.00	0.00
AGST	Agrostis stolonifera L.	creeping bentgrass	FAC	no	0.00	0.00	0.00	0.08	0.00
ALTR	Alisma triviale	northern water plaintain	OBL	yes	1.82	0.87	2.37	0.02	0.00
AMFR	Amorpha fruticosa	indigo bush	FACW	no	0.00	0.00	0.00	0.02	0.00
BICE	Bidens cernua	Nodding beggars-ticks	FACW+	yes	2.94	0.00	0.51	0.00	0.00
CAHE	Callitriche heterophylla	Water starwort	OBL	yes	5.27	2.95	1.78	0.53	0.03
CALY	Carex lyngbyei	Lyngby sedge	OBL	yes	23.03	4.89	8.32	0.00	0.00
CAPA	Caltha palustris	Yellow marsh marigold	OBL	yes	0.27	0.34	0.00	0.00	0.00
CASP	Carex sp.	Carex	mixed	yes	0.00	0.00	0.00	0.65	0.00
CEDE	Ceratophyllum demersum	Coontail	OBL	yes	0.03	0.00	0.00	0.00	0.00
COPA	Comarum palustre	purple marshlocks, marsh cinquefoil	OBL	yes	0.00	0.00	0.39	0.00	0.00
ELAC	Eleocharis acicularis	Needle spikerush	OBL	yes	1.79	0.00	0.00	0.00	0.16

Code	Scientific Name	Common Name	Wetland Status	Native	Jackson Is.	Whites Is.	Wallace Is.	Campbell Slough	Cunningham Lake
ELCA	Elodea canadensis	Canada waterweed	OBL	yes	4.79	4.74	**10.98**	0.37	0.00
ELPA	Eleocharis palustris	Common spikerush	OBL	yes	7.06	1.18	2.44	**37.4**	**20.4**
EPCI	Epilobium ciliatum	Willow herb	FACW-	yes	0.45	0.34	0.05	0.00	0.00
EQFL	Equisetum fluviatile	Water horsetail	OBL	yes	0.00	3.50	0.00	0.00	0.92
EQPA	Equisetum palustre	marsh horsetail	FACW	yes	0.00	0.00	0.00	0.08	0.00
GATR	Galium trifidum var. pacificum	Pacific bedstraw	FACW	yes	0.00	1.26	0.24	0.00	0.08
GATR3	Galium trifidum	small bedstraw	FACW+	yes	0.00	0.24	0.00	0.00	0.00
GLGR	Glyceria grandis	American mannagrass	OBL	yes	0.15	1.58	0.00	0.00	0.00
GREB	Gratiola ebracteata	bractless hedgehyssop	OBL	yes	0.06	0.00	0.20	0.00	0.00
HEAU	Helenium autumnale	common sneezeweed	FACW	yes	0.00	0.00	0.00	0.35	0.00
IMSP	Impatiens capensis,Impatiens noli-tangere	Spotted touch-me-not, Common touch-me-not	FACW	yes	0.00	0.61	0.00	0.00	0.00
IRPS	Iris pseudacorus	Yellow iris	OBL	no	0.45	3.89	0.00	0.00	0.02
JUOX	Juncus oxymeris	Pointed rush	FACW+	yes	1.52	0.42	0.15	0.48	0.00
JUTE	Juncus Tenuis	slender rush, poverty rush	FACW-	yes	0.00	0.00	0.00	0.16	0.00
LEOR	Leersia oryzoides	Rice cutgrass	OBL	yes	0.00	0.00	0.98	0.03	0.94
LIAQ	Limosella aquatica	Water mudwort	OBL	yes	0.76	0.16	0.37	0.00	0.00
LIOC	Lilaeopsis occidentalis	Western lilaeopsis	OBL	yes	0.30	0.16	0.05	0.00	0.00
LOCO	Lotus corniculatus	Birdsfoot trefoil	FAC	no	0.00	1.18	0.37	0.16	0.00
LUPA	Ludwigia palustris	False loosestrife	OBL	yes	0.00	0.00	0.98	1.29	1.89

Code	Scientific Name	Common Name	Wetland Status	Native	Jackson Is.	Whites Is.	Wallace Is.	Campbell Slough	Cunningham Lake
LYAM2	Lycopus americanus	American water horehound	OBL	yes	0.00	0.03	0.12	0.00	0.00
LYNU	Lysimachia nummularia L.	Moneywort, Creeping Jenny	FACW	no	0.00	0.00	0.00	1.05	0.02
LYSA	Lythrum salicaria	Purple loosestrife	FACW+	no	0.00	0.26	2.56	0.00	0.00
MEAR	Mentha arvensis	wild mint	FACW-	yes	2.00	0.00	0.00	0.00	0.00
MIGU	Mimulus guttatus	Yellow monkeyflower	OBL	yes	0.61	0.03	0.44	0.00	0.00
MYSC	Myosotis scorpioides	Common forget-me-not	FACW	no	0.94	**5.61**	0.61	0.00	0.00
MYSI	Myriophyllum sibiricum	northern milfoil, short spike milfoil	OBL	yes	0.21	0.00	0.00	0.00	0.00
MYSP2	Myriophyllum spp.	Milfoil	OBL	mixed	0.00	0.03	0.00	0.00	0.00
OESA	Oenanthe sarmentosa	Water parsley	OBL	yes	1.73	5.29	0.32	0.00	0.00
PHAR	Phalaris arundinacea	Reed canary grass	FACW	no	**18.9**	**47.8**	**20.3**	**41.5**	**57.3**
PLLA	Plantago lanceolata var. lanceolata	Rib plantain	FAC	no	0.00	0.00	0.00	0.02	0.00
POAM	Polygonum amphibium	water ladysthumb, water smartweed	OBL	yes	0.00	0.00	0.00	0.18	0.00
POCR	Potamogeton crispus	Curly leaf pondweed	OBL	no	0.21	0.42	0.37	0.66	0.00
POHY	Polygonum hydropiper, P. hydropiperoides	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	2.70	0.08	1.80	0.03	0.13
PONA	Potamogeton natans	Floating-leaved pondweed	OBL	yes	0.00	0.00	0.00	0.00	0.45

Code	Scientific Name	Common Name	Wetland Status	Native	Jackson Is.	Whites Is.	Wallace Is.	Campbell Slough	Cunningham Lake
POPE	Polygonum persicaria	Spotted ladysthumb	FACW	no	0.15	0.79	0.00	0.00	1.19
PORI	Potamogeton richardsonii	Richardson's pondweed	OBL	yes	0.18	0.00	0.00	0.00	0.00
POZO	Potamogeton zosteriformis	Eelgrass pondweed	OBL	yes	0.48	0.13	0.00	0.00	0.00
RARE	Ranunculus repens	Creeping buttercup	FACW	no	0.00	0.00	0.12	0.18	0.00
RUMA	Rumex maritimus	Golden dock, seaside dock	FACW+	yes	0.00	0.16	0.00	0.24	0.16
SALA	Sagittaria latifolia	Wapato	OBL	yes	**8.97**	4.53	5.56	7.77	7.23
SALU*	Salix lucida	Pacific willow	FACW+	yes	0.00	0.00	0.00	0.00	8.23
SCAM	Schoenoplectus americanus	American bulrush, threesquare bulrush	OBL	yes	1.33	0.29	0.85	0.00	0.00
SCTA	Schoenoplectus tabernaemontani	Softstem bulrush, tule	OBL	Yes	0.61	0.00	0.12	0.02	0.02
SISU	Sium suave	Hemlock waterparsnip	OBL	yes	0.79	0.16	0.46	0.00	0.00
SODU	Solanum dulcamara	Bittersweet nightshade	FAC+	no	0.00	0.66	0.00	0.00	0.00
SPAN	Sparganium angustifolium	Narrowleaf burreed	OBL	yes	0.00	0.00	0.00	0.00	0.10
SYSU	Symphyotrichum subspicatum	Douglas aster	FACW	yes	0.30	0.00	0.00	0.00	0.00
TYAN	Typha angustifolia	Narrowleaf cattail	OBL	no	0.00	4.08	**41.7**	0.00	0.00
VEAM	Veronica americana	American speedwell	OBL	yes	0.06	1.05	0.00	0.27	0.26
		Total Vegetation Cover			**90.91**	**101.29**	**105.44**	**93.60**	**99.47**