Ecosystem Monitoring Program

Annual Report for Year 8

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Lower Columbia River Ecosystem Monitoring Program
Annual Report for Year 8 (October 1, 2011 to September 30, 2012)

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1   Executive Summary

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

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

This monitoring was intended to address Action 28 of the Estuary Partnership's CCMP, and Reasonable and Prudent Alternatives (RPAs) 161, 163, and 198 of the 2000 Biological Opinion for the Federal Columbia River Power System, and RPAs 58, 59, 60, and 61 of the 2008 Biological Opinion. 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 National Marine Fisheries Service (NOAA-Fisheries), United States Geological Survey (USGS), Oregon Health and Sciences University (OHSU) and Columbia River Estuary Study Taskforce (CREST).

Methods
In 2012, the EMP partnership collected co-located data on fish, fish prey, habitat structure and hydrology, abiotic site conditions, abiotic mainstem conditions and food web dynamics. The specific metrics included: habitat structure and hydrology (vegetation community, water surface elevation, channel morphology, sediment grain size and total organic content [TOC], sediment accretion, biomass and site profiles; by PNNL), fish use (fish community, salmon metrics and diet), macroinvertebrate prey availability, and water temperature at the time of fish sampling (by NOAA NMFS), abiotic site conditions and food web (nutrients, primary productivity, water temperature, specific conductance, pH, dissolved oxygen, photosynthetically available radiation, and isotope ratios; by USGS), secondary productivity (zooplankton abundance and taxonomy; by OHSU) and mainstem abiotic conditions (nitrate, conductivity, temperature, dissolved oxygen, turbidity, chlorophyll concentration, Colored Dissolved Organic Matter; by OHSU). Sample sites were minimally-disturbed, tidally influenced freshwater emergent wetlands with backwater sloughs, representative of a subset of the eight hydrogeomorphic reaches across the study area.

In 2012, the EMP partnership monitored three new tidal freshwater sites in Reach G: Government/Lemon Island, Washougal, and Reed Island in order to continue to examine spatial variation in the LCRE. In addition, we re-sampled four "trend" sites: Franz Lake Slough site in Reach H (sampled in 2008, 2009, 2010, and 2011), Campbell Slough in Reach F (sampled from 2007-2011), Whites Island site in Reach C (sampled in 2009, 2010, and 2011), and Ilwaco in Reach A (sampled in 2011), in order to examine temporal trends. In 2012, two new trend sites were added in reach B, Secret River and Welch Island. Also in 2012, the Center for Coastal Margin Observation and Prediction (CMOP) OHSU installed an in situ water quality monitoring platform in the mainstem Columbia River to provide baseline water quality measurements in support of the EMP. The station is located at River Mile 122 (in Reach G) and is physically located on the outer-most floating dock at the Port of Camas-Washougal (45.577°N, -122.379°E).

2012 Summary of Results:

Mainstem abiotic conditions
- The installation of the RM-122 Biogeochemical platform at the Port of Camas-Washougal established a time series of water quality observations of the mainstem Columbia River upstream of the Willamette River. Measurements from RM-122 were compared to measurements from the existing platform at RM-53 (Beaver Army Terminal) to determine changes in water properties of the mainstem that are caused by inputs from major tributaries in the lower Columbia, including the Willamette.
- A summary of RM-53 observations between 2009–2012 is provided, including analysis of water temperature during the summer periods when temperatures reached critical levels for salmon habitat. Nitrogen and organic matter fluxes were calculated from the daily observations of concentration and river discharge, and allow for an assessment of these loadings to the coastal margin.
- A comparison of RM-122 and RM-53 nitrate, turbidity, and CDOM data clearly shows the important role of tributaries in the lower Columbia on altering water quality, particularly during winter storm events. As the data set from RM-122 expands in 2013, more detailed information on primary production in the mainstem river will be feasible.

Off-channel abiotic conditions
- Water temperature at four sites (excludes Reach B sites) exceeded the maximum Washington State standard and dissolved oxygen concentration were less than the minimum standard at all sites by July.
- Among the four water-quality monitoring sites, Cut-Off Slough in Whites Island had the most consistently suitable conditions for juvenile salmon during the monitoring periods in 2011/2012.
- Franz Lake Slough and Campbell Slough had suitable water-quality conditions for much of the monitoring period, but were slower to recover from periods of poor water quality than more frequently flushed sites.

Habitat Structure and Hydrology

Sediment

- Total organic carbon has been measured in the sediment samples from EMP sites since 2008 and in general the amounts recorded in these samples is low compared to values from other marsh systems (Sagar et al. 2013).
- Sediment accretion rates varied from -0.4 cm to 2.7 cm at the sites measured in 2012.

Hydrology

- Hydrographs from the sites where Water Surface Elevation was collected during the 2012 water year illustrate the extremely different hydrologic patterns that occur throughout the estuary, particularly in a high water year such as 2012.

Salinity

- Salinity was measured at the Ilwaco site during 2011-2012. The salinity is highly variable at the site both seasonally and daily.

Vegetation Assemblages

- At most sites in 2012, native vegetative cover was higher than non-native cover. The high water in 2012 may have had more of an effect on non-native than native cover by reducing reed canary grass cover; whereas the cover of native species such as spike rush (*Eleocharis palustris*) was cumulatively higher.
- Due to the sequence of two high water years, we were able to better evaluate the effects of hydrology on vegetation in upriver sites. Through this analysis we documented a decrease in vegetation cover and productivity, a migration in the upper and lower boundaries of vegetation species, and a shift in the dominance of species. These significant effects elucidate the changes that are possible given the hydrologic changes that are possible in the LCRE. Sea level rise, changes in the timing and magnitude of the spring freshet from climate change, and changes in the regulation of flows could all affect the hydrologic patterns that have resulted in the current location and vegetation composition of the wetlands in the LCRE.

Biomass

- In 2011, the primary productivity and the potential macro-detritus export were considerably higher in the lower River sites compared to the upper River sites. We hypothesize that the upriver sites were affected by the high water in 2011; additional biomass sampling is needed to test this hypothesis and provide data on the productivity of the upriver marshes (above rkm 72).

Elevation, Inundation and Vegetation Interactions

- Elevation of emergent wetlands in the LCRE covers a very narrow range (Sagar et al. 2013). Elevations measured in 2012 confirm this finding, with sites measured relative to Columbia River Datum (CRD) throughout the estuary covering a range of only 1.83 meters (m).
- Several patterns are notable in the frequency of inundation results. First, percent of time sites are inundated increases with distance upstream. Second, there is a shift in the timing of high inundation frequency between lower and upper river sites. In the lower river sites (Burke Island and lower) a higher percentage of the inundation occurs in the winter (November-March) whereas above this a higher percentage occurs in the growing season (April-October).

- Regardless of where sites are located in the LCRE, the relationship between inundation (as measured by the Sum Exceedance Value, SEV) and total vegetative cover holds true. That is, with increasing SEV vegetation cover declines. Importantly, water level, especially in extreme water years can produce a measurable response in vegetation cover.

Channel Morphology and Inundation
- In the lower river, during March – July of 2012, the channel thalweg at these sites was accessible to salmon 47-60 percent of the time. Previous studies at the Secret River site in 2008 indicate that the channel was accessible 74 percent of the time between March and July (Sagar et al. 2013). Whereas, the upriver channels starting at Whites Island, were accessible 93 percent of the time or greater during this period.
- The banks of the channels, with at least 10 cm of water, were accessible less of the time than the channel thalwegs. In the lower river, channel banks during the period from March to July were accessible 9-39 percent of the time. At the Secret River site in 2008, the channel bank was accessible 47 percent of the time between March and July (Sagar et al. 2013). At Whites Island, the frequency increases to 76 percent and in the upper River it is greater than 92 percent of the time except at very high elevation sites

Food Web
Phytoplankton
- The phytoplankton assemblages were largely dominated by diatoms, with higher contributions by non-diatom flagellated taxa at Campbell Slough (Reach F) and Franz Lake Slough (Reach H), particularly later in the summer.
- Phytoplankton-based primary production was determined in 2012, but the results are not yet available. Based on the 2011 data for phytoplankton primary production and results from periphyton-based primary production experiments, an inverse relationship between phytoplankton production and periphyton production was observed. There was a trend of increasing phytoplankton abundance and productivity with upstream distance and a decreasing trend of periphyton production with upstream distance.
- Phytoplankton abundance based on chlorophyll a concentrations indicated that there was a greater similarity among sites when water levels were high compared to other times of year.

Zooplankton
- Zooplankton assemblages were either dominated by small rotifers or by crustaceans (copepods or cladocerans). The assemblages differed by site and time of year, with copepods dominating the assemblages at Ilwaco throughout the time series.
- Rotifers dominated the zooplankton assemblages numerically at the downstream freshwater sites (Secret River, Welch Island, Whites Island) throughout the 2012 time series. In contrast, a rotifer-dominated zooplankton community was replaced by a copepod-cladoceran dominated community later in the season at Campbell Slough. Franz Lake Slough showed a shift toward dominance by rotifers then to copepods and back to rotifers in late June-July.

Macroinvertebrates

- The density of invertebrate salmonid prey varied tremendously by site, the month that sites were sampled, and whether samples were collected from the open water habitat versus the near-shore habitat that is often associated with emergent vegetation. Much of the variation was explained by habitat, indicating there is more variation at the local scale within a site than there is systematically among sites or by the times in which they were sampled.
- Overall prey densities and biomass were nearly 20 times greater in samples collected from the near-shore emergent vegetation habitats compared to open water habitats, suggesting these near-shore habitats are very important for prey production and salmonid feeding
- Juvenile Chinook salmon are generally opportunistic but do show preferences for midge larvae and pupa (primarily Chironomidae, Diptera), and when available, Amphipods.
- Sediment-associated (benthic) macroinvertebrate taxa differed according to site, with the seawater-influenced site (Ilwaco) supporting Corophium spp., in contrast to sites further upstream. All sites were dominated by oligochaete and nematode worms. The contribution of Ceratopogomonidae (Order Diptera) was higher at Whites Island than the other sites examined.

Fish

- Our findings for 2012 were consistent with our previous results described in the Ecosystem Monitoring Synthesis document in which tidal freshwater habitats in the LCRE are important for migration, feeding, and rearing for several species of salmonids, and by Chinook salmon stocks from both the lower Columbia River and the interior Columbia Basin.
- The new sites sampled in Reach G in 2012 (Government/Lemon Island, Washougal River, and Reed Island) were most similar to our Reach H sites in terms of species composition and salmon occurrence while the new Reach B sites (Secret River and Welch Island) were most similar to Reach C sites. Ilwaco Slough in Reach A was distinct from the other sites because of the higher proportion of marine species.
- We continued to see clear patterns in fish community characteristics and salmon occurrence from reach to reach. Sites in Reaches A-C typically had low species diversity and richness, lower proportions of non-native species. Seasonal patterns in fish community composition were evident, with warm water, primarily non-native species making up a higher percentage of catches in summer and fall.
- Patterns of salmon occurrence also varied longitudinally along the river. Chinook salmon were the dominant salmon species at most sites, but coho salmon made up relatively high proportions of catches in Reaches G and H. Coho salmon were also found at significant numbers at a few sites in Reaches B-E (Secret River, Bradwood Slough, and Deer Island). Chum salmon were found throughout the estuary, but only made up a large proportion of the catch at Ilwaco Slough in Reach A.
- In general, hatchery Chinook and coho salmon were most abundant in catches at the upriver sites, in Reaches E-H. Hatchery salmon were not especially abundant at most of the new 2012 sites, making up less than 10% of catches at Secret River and Welch Island in Reach B, and

Government/Lemon Island and Reed Island in Reach G. Washougal Wetland, in Reach G, was an exception; here, hatchery salmon made up 45% of the Chinook catch and 99% of the coho catch.

- Extended fall and winter sampling showed Chinook salmon were present at least some sites all year, but densities were extremely low from August through November. Coho salmon were present at low densities throughout the year, while chum salmon were found from February through May only.

- Chinook salmon stock usage of the EMP sampling sites changed longitudinally along the river. Among unmarked fish, West Cascades fall Chinook were more abundant in Reaches C and E, with Spring Creek Group fall Chinook becoming more prevalent in reaches F-H (data on reaches A, B, and G are not yet available). While they were present throughout the Lower Columbia River, Chinook stocks from the Interior Columbia, such Upper Columbia summer/fall and Snake River fall Chinook, tended to be more common in Reach H. The longitudinal pattern for marked fish was fairly similar, with West Cascades fall Chinook being most common in Reach C, and Spring Creek fall Chinook being more common in Reaches E-H.

- Stock distribution changed with sampling time for some stocks of both marked and unmarked Chinook salmon. Among unmarked fish, Spring Creek Group fall Chinook were more abundant early in the sampling season, while Upper Columbia summer/fall Chinook were found in greater numbers later, in June, July, and August. Among marked fish, Spring Creek Group fall Chinook dominated in April and May, and West Cascades fall Chinook became more abundant later in the sampling season.

- Differences were found among sites in condition factor and lipid content, but there were no very clear patterns by reach or longitudinally along the river for unmarked Chinook salmon. However, at the Washougal site in Reach G, CF was significantly lower than the overall average for both marked and unmarked juvenile Chinook, even after accounting for the effect of sampling month. In marked Chinook salmon, on the other hand, lipid content tended to decline downriver from Reach H sites to the Reach B sites, which would be consistent with lipid loss in migrating hatchery fish.

- At those sites that have been sampled in multiple years, patterns of salmon occurrence and fish community composition were fairly consistent. Ilwaco Slough and Whites Island consistently showed low species richness and diversity and a low percentage of non-native species, while Campbell Slough and Franz Lake Slough consistently showed high species richness and diversity and high proportions of non-native species. Patterns of salmon occurrence were also consistent, e.g., few salmon at Ilwaco Slough, unmarked Chinook salmon at Whites Island, a mixture of marked and unmarked Chinook salmon at Campbell Slough, and a mixture of marked and unmarked coho and Chinook salmon at Franz Lake Slough.

- The limited data available on stable isotope ratios in juvenile Chinook salmon bodies suggests that there may be some differences in diet among marked and unmarked fish, and fish of different stocks and from different sites. Most notable trends were lower $\partial^{15}N$ values (delta-N-15 is a measure of the ratio of stable isotopes 15N:14N) in fish from Reach H and from Interior Columbia River stocks (Upper Columbia summer/fall Chinook, Snake River fall Chinook, and Deschutes River fall Chinook). $\partial^{15}N$ values also tended to be lower in unmarked fish than in

marked fish. This would be consistent with the use of hatchery feeds derived from fish meal and fish oils, and also could be reflective of the somewhat larger size of most of the hatchery fish collected from field sites.   The results also suggest that fish from the lower part of the estuary may be feeding on items from higher in the trophic levels than those originating in the interior basin or rearing at sites in the Gorge. Another possibility is that fish from lower in the estuary are exposed to nitrogen from other sources, such as sewage (Hansson et al. 1997), as there is evidence that fish from this area are exposed to wastewater compounds (Sagar et al. 2013).

- New data on polycyclic aromatic hydrocarbon (PAH) concentrations in bodies of juvenile Chinook salmon collected in 2010 and 2011 show higher levels of exposure in fish from Campbell Slough in comparison to other sites sampled in Reaches C and E. Also, PAH exposure increased at Campbell Slough between 2010 and 2011. This was not true at the other trend site sampled (Whites Island).

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2 Introduction

2.1 Background

Ecosystem monitoring is a key element of the Lower Columbia Estuary Partnership's (Estuary Partnership's) Comprehensive Conservation and Management Plan (CCMP) which specifically calls for sustained long-term monitoring to understand conditions in the lower Columbia River and an evaluation of the impact of management actions over time. Efforts for the EMP include the development of an estuarine ecosystem classification system and on-the-ground monitoring of habitat, hydrology, juvenile salmon, food web dynamics, and abiotic conditions in shallow, off-channel tidal habitats in the lower Columbia River. This monitoring was intended to address Action 28 of the Estuary Partnership's CCMP and Reasonable and Prudent Alternatives (RPAs) 161, 163, and 198 of the 2000 Biological Opinion for the Federal Columbia River Power System, and RPAs 58, 59, 60, and 61 of the 2008 Biological Opinion. 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 National Marine Fisheries Service (NOAA-Fisheries), United States Geological Survey (USGS), Oregon Health and Sciences University (OHSU) and Columbia River Estuary Study Taskforce (CREST). Funding for the EMP comes from the Bonneville Power Administration (BPA) and Northwest Power and Conservation Council (NPCC).

The Columbia River historically supported diverse and abundant populations of fish and wildlife and is thought to have been one of the largest historical producers of Pacific salmonids in the world (Netboy 1980). Anthropogenic changes since the 1860s and the construction of the hydropower system have significantly reduced the quantity and quality of habitat available to fish and wildlife species. Contributing factors include altered timing, magnitude, duration, frequency, and rate of change in river flows; degraded water quality and increased toxic, chemical contaminants; introduction of invasive exotic species; and altered food web dynamics. Ecosystem-based monitoring of the LCRE has become a regional priority to aid in the recovery of the historical productivity and diversity of fish and wildlife.

The Estuary Partnership's EMP is an integrated status and trends program for the Lower Columbia River and Estuary (LCRE). The EMP encompasses the study area of the Estuary Partnership and includes all tidally influenced areas of the mainstem and tributaries from Bonneville Dam to the mouth of the river. Tidal influence is defined as historical tidal influence, relative to post-dam construction in the 1930s. The EMP is designed to track trends in the overall condition of the LCRE to reduce uncertainties, provide a suite of reference sites for use as end points in the region's habitat restoration actions, and place findings from the program into context with the larger ecosystem. A primary goal of this program is to collect key information on ecological conditions for a range of habitats in the lower river characteristic of those used by out-migrating juvenile salmon and provide information toward the recovery of threatened and endangered salmonids.

When the EMP was created in 2004, most previous research in the LCRE had occurred in the lower estuary, closest to the river mouth in Reaches A and B. There was a considerable lack of research and monitoring within the tidal freshwater section of the LCRE, resulting in little basic understanding of habitats, fish use and food web dynamics in this region. The EMP and partners developed a list of

questions, and a subsequent monitoring design, for which there was little current information and which were fundamental to understanding how estuarine resources occur and interact in the LCRE. Based on the knowledge gaps identified in the LCRE and the Estuary Partnership's and the regional partner's goals, the EMP goals for the monitoring design were to:

Track the status and trends of ecosystem conditions to inform decisions for the purpose of conserving and restoring the LCRE through:

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

2.2 Activities Performed, Year 8 Contract (October 1, 2011-September 30, 2012)

Funding for the EMP by the NPCC/BPA supports the Estuary Partnership's Research Scientists. As part of 2011-2012 EMP tasks, funding supported the following:
- Coordinated development of the Columbia River Estuary Ecosystem Classification and work timelines.
- Facilitated discussions and planning for 2012-2013 monitoring efforts.
- Coordinated site field trips to select monitoring sites for 2012 field season.
- Acquired special use permits and landowner permission 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, OHSU, CREST, Environment International (EI) and NOAA-NMFS.
- Coordinated meetings, provided technical guidance, compiled results of data analyses (between 2005 and 2010) and edited draft report to complete a five-year synthesis for the EMP program with PNNL, NOAA-Fisheries, EI and USGS.
- Compiled annual data collection summary report contributions from EMP subcontractors into 2011-2012 annual report to BPA.
- Summarized yearly activities and results per individual RPA for BPA in a separate, standardized reporting format.
- Completed preliminary program protocols in MonitoringMethods.org.
- Coordinated discussions on goals, objectives, actions and candidate indicators for an estuarine indicator system.
- Researched other estuarine indicator systems and provided recommendations to the Science Work Group.
- Developed new scopes of work with EMP subcontractors for the 2012-2013 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 2012-2013.

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:
- Identified six new restoration sites and four reference sites for 2012-2013 AEM efforts
- Developed and refined site monitoring plans for 2012-2013 AEM efforts.
- Assisted in the development of a programmatic plan for restoration action effectiveness monitoring and research (AEMR) for the lower Columbia River and Estuary.
- Created a site monitoring matrix to codify elements of AEMR plans for comparison and to ensure all relative monitoring metrics are considered.
- Developed criteria to designate AEMR levels of intensity for restoration sites
- Managed AEMR subcontracts with NOAA-Fisheries, CREST, and Ash Creek Forest Management (ACFM) for 2011-2012.
- Compiled AEMR reports from subcontractors for the Restoration Program's 2011-2012 annual report to BPA.
- Developed new scopes of work with AEMR subcontractors for 2012-2013.
- Coordinated with project sponsors to implement the collection of extensive AEMR data related to the programmatic AEMR plan

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 among interested 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.

Funding for the EMP also provides partial support for the Estuary Partnership's GIS/Data Management Specialist. For the 2011-2012 EMP efforts, the GIS/Data Management Specialist:
- Completed final mapping of all Reaches at the Catena and Complex levels
- Final QA/QC of data including USGS Technical review of data layers and metadata
- Publication of final data set on USGS website: http://water.usgs.gov/GIS/metadata/usgswrd/XML/creec_hydrogeomorphic_reach.xml

In addition to the work described above for the EMP program, the GIS/Data Management Specialist contributed to the following regional monitoring efforts:
- Coordinated Columbia River Ecosystem Classification System (Classification) development efforts between the Estuary Partnership, USGS and UW.
- Provided development support for the Classification, which included QA/QC on the final data sets, and assistance with refinement of the Classification scheme.
- Provided field support for PNNL sampling crew during the 2012 field season.
- 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.

2.3 Study Area

The 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 EMP's study area encompasses all tidally influenced waters of the LCRE, extending from the plume of the Columbia at river kilometer (Rkm) 0 upstream to the Bonneville Dam at Rkm 235. The Estuary Partnership and monitoring partners collect data for the EMP on habitats supporting juvenile salmonids, including shallow emergent wetlands, 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. The sampling has been organized according to Level 3 of the Classification (described below), which divides the LCRE into eight major hydrogeomorphic transitions. 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), C and F (2009-2010) and A, C, H, E and F (2010-2011). In 2011-2012, the Estuary Partnership and partners monitored emergent wetland habitats in Reaches A, B, C, F, G, and H (

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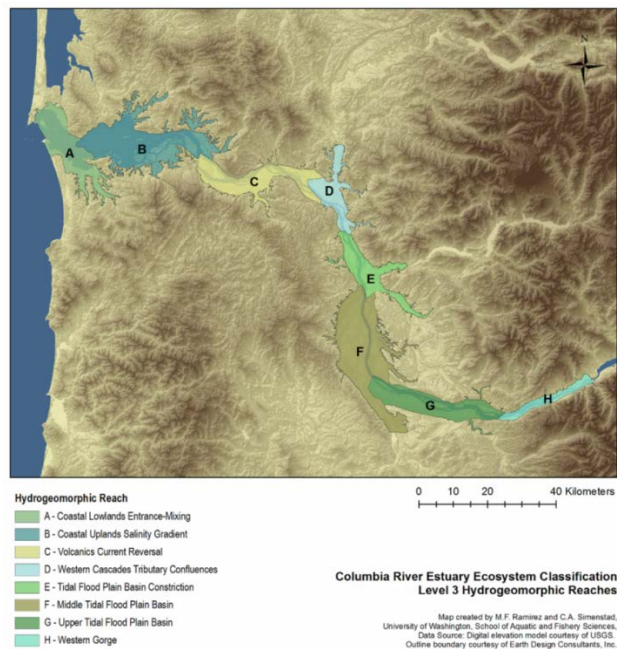
Figure 1. Lower Columbia River and estuary (LCRE) with hydrogeomorphic reaches (A-H) outlined and specified by color.

2.4 Columbia River Estuary Ecosystem Classification (Classification)

The Columbia River Estuary Ecosystem Classification, a GIS based data set, is a six tier hierarchical framework that delineates the diverse ecosystems and component habitats across different scales in the LCRE. The primary purpose of the 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.

2.5   Characterization of Emergent Wetlands in the LCRE

2.5.1 Sampling Effort, 2005-2012

The objective of the EMP is to characterize estuarine and 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 monitoring of vegetation, fish, fish prey, and some basic water quality at emergent wetland sites in order to overlap datasets for multiple sites throughout the LCRE. Starting in 2011, the Estuary Partnership added food web and abiotic site conditions (i.e., conditions influencing productivity such as temperature, water clarity, dissolved oxygen, nutrients) sampling and analysis in both the mainstem Columbia and trend sites to the EMP.

Table 1. Summary of sampling effort by site and year(s) for sites; where data were collected in 2012 site is in bold. *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	Type of Site	Site	Vegetation & Habitat	Fish &Prey	Abiotic Conditions	Food Web	Mainstem abiotic conditions
A	Trend	**Ilwaco**	2011	2011	2011	2011	
B	Trend	**Secret River**	2012	2012	2012	2012	
	Trend	**Welch Island**	2012	2012	2012	2012	
C	Status	Ryan Island	2009	2009			
	Status	Lord-Walker Island 1	2009	2009			
	Status	Lord-Walker Island 2*	2009				
	Trend	**Whites Island**	2009-2011	2009-2011	2009	2011	
	Status	Jackson Island	2010	2010			
	Status	Wallace Island	2010	2010			
	Status	Bradwood Landing	No access permission	2010			
D	Status	Cottonwood Island small slough	2005				
	Status	Cottonwood Island	2005				

		large slough					
	Status	Dibble Slough	2005		2005		
E	Status	Sandy Island 1, 2	2007	2007			
	Status	Lewis River Mouth	2007				
	Status	Martin Island	2007				
F	Status	Sauvie Cove	2005				
	Status	Hogan Ranch	2005				
	Status	Goat Island	2011	2011			
	Status	Deer Island	2011	2011			
	Status	Burke Island	2011	2011			
	Trend	**Cunningham Lake**	2005-2011	2007-2009			
	Trend	**Campbell Slough**	2005-2011	2007-2011	2008-2011	2011	
G	Status	Water Resources Center					
	Status	McGuire Island	2006				
	Status	Old Channel Sandy River	2006			2006	
	Status	Chattam Island	2006				
	Status	**Government/Lemon Island**	2012	2012	2012	2012	
	Status	**Reed Island**	2012	2012	2012	2012	
	Status	**Washougal Wetland**	2012	2012	2012	2012	
	Trend	**RM122**					2012
H	Trend	**Franz Lake (slough)**	2008-2009, 2011	2008-2009, 2011	2011	2011	
	Status	Sand Island	2008	2008	2008		
	Status	Beacon Rock	2008	2008			
	Status	Hardy Slough	2008	2008			

2.5.2 Site Selection

A site selection trip took place on November 3, 2011 to evaluate the feasibility and appropriateness of potential sites in Reaches G and on December 1, 2011 to choose two trend sites in Reach B. Members from each of the monitoring collaborators (PNNL, NOAA, and Estuary Partnership) were present. Prior to the trip, the areas were evaluated using Geographic Information System (GIS) layers including current imagery, LiDAR digital elevation models, and historical maps from the late 1800's. Using these sources of information, the potential sites were narrowed to those that appeared relatively undisturbed and hydrologically connected to the mainstem of the Columbia River. Four potential sites were visited in Reaches G and three were acceptable for monitoring (trip report not included but available upon request from the Estuary Partnership). Three areas were also visited in Reach B, with two sites accepted in the field assessment.
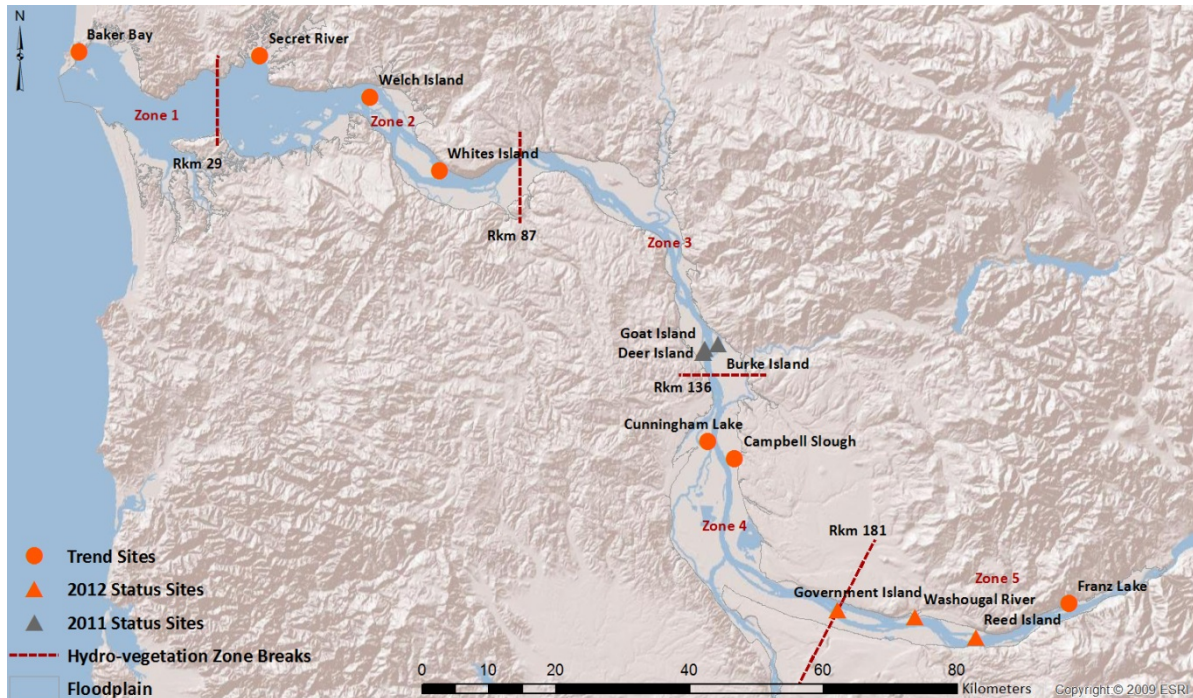
Figure 2. Map of the 2012 monitoring sites with the estimated hydro-vegetation zone breaks as described in Borde et al. 2012

2.5.3 Site Descriptions

The 2012 monitoring sites are described below in order starting at the mouth of the Columbia and moving upstream. Maps of the sites, including vegetation communities, are provided in Appendices.

Ilwaco (Baker Bay). This site is located at River kilometer (rkm) 6, southeast of the entrance of Ilwaco harbor, in Baker Bay. The property is currently owned by Washington Department of Natural Resources. The site has developed in the past century as the Bay has filled in, likely due to changes in circulation from construction of the jetties at the mouth of the Columbia River, the placement of dredge material islands at the mouth of the Bay, and changes in River flows. Selected as a long-term monitoring site in 2011, Baker Bay marsh is dominated by lush fields of Lyngby's sedge (*Carex lyngbyei*) with higher portions occupied by tufted hairgrass (*Deschampsia cespitosa*) and cattail (*Typha angustifolia*) (**Error! Reference source not found.**a). Being so close to the mouth of the Columbia River, the tidal channel is regularly inundated with brackish water.

Secret River. The Secret River marsh, located in Grays Bay at the mouth of Secret Rive at rkm 37, is an extensive marsh owned by the Columbia Land Trust and selected as a long-term monitoring site in 2012. However, the site was also monitored as part of the Reference Site Study in 2008 (Borde et al. 2011). The marsh was present on the historical maps from the late 1880's, however the marsh edge has receded approximately 400 m. The cause of this erosion is unknown at this time. The marsh grades from *C. lyngbyei* and soft stem bulrush (*Schoenoplectus tabernaemontani*) in the low and mid marsh to a diverse mix of species in the upper marsh. The primary tidal channel is a low grade channel with low banks near the mouth, becoming steeper as it cuts through the higher marsh and then in to the tidal

swamp above the marsh. Many smaller tidal channels also cut through the marsh plain. The marsh and the channel have large wood scattered throughout, with an accumulation at the high tide margin.

Welch Island. The monitoring site on Welch Island is located on the northwest (downstream) corner of the island at rkm 53, which is part of the Julia Butler Hanson Wildlife Refuge. The area was selected as a long-term monitoring site in 2012; two other areas of the island were monitored as part of the Reference Sites Study in 2008 and 2009 (Borde et al. 2011). The island was present on the historical late-1800's map, however the island has expanded since then and study site has developed wetland vegetation where there was previously open water. The site is a high marsh dominated by *C. lyngbyei*, but with diverse species assemblage and a scattering of willow trees. Small tidal channels grade up to low marsh depressions within the higher marsh plain.

Whites Island. The Whites Island site is located on Cut-Off Slough at the southern (upstream) end of Puget Island, near Cathlamet, Washington at rkm 72. 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 1880's and was likely created from dredge material placement. A long-term monitoring site since 2009, the site is located at the confluence of a large tidal channel and an extensive slough system, is approximately 0.2 km from an outlet to Cathlamet Channel; however, according to historic photos, this outlet was not present prior to 2006 and the River connection was approximately 0.7 km from the monitoring site. The site is characterized by high marsh and a few willows, with numerous small tidal channels.

Cunningham Lake. Cunningham Lake is a floodplain lake located at rkm 145 on Sauvie Island in the Oregon DFW Wildlife Area. The site is a fringing emergent marsh at the upper extent of the extremely shallow "lake" (**Error! Reference source not found.**f) and at the end of Cunningham Slough which meanders approximately 8.7 km from Multnomah Channel (a side channel of the Columbia River). The mouth of the Slough is located between rkm 142 and 143 close to where Multnomah Channel meets the Columbia River. This long-term monitoring site has been surveyed annually since 2005 site. In some years the "lake" is covered with wapato (*Sagittaria latifolia*), however in all years since 2005, this cover has been sparse or non-existent.

Campbell Slough. The Campbell Slough site is located at rkm 149 on the Ridgefield National Wildlife Refuge in Washington. This long-term monitoring site has been surveyed annually since 2005. The monitoring site is an emergent marsh adjacent to the slough, approximately 1.5 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 vegetation appeared to recover in subsequent years. In 2010 and 2011, slight evidence of grazing was again observed, and in 2012 the site was heavily grazed and trampled by cows.

Government/Lemon Island. The Government/Lemon Island site is located at rkm 181 and is owned by Port of Portland and managed by Oregon State Parks and Recreation. The site is located on a sandy area between Government Island and Lemon Island. The course sediments at the site are likely caused by high flows during the spring freshet. Emergent wetland vegetation is present primarily in the lower elevation channels. In 2012, following recent high water, the site was carpeted with small seedlings that likely do not persist during the low water period.

Washougal River. Located near the mouth of the Washougal River at rkm 195, this site is in the location of a former channel of the Washougal River that was cut off from the main channel by construction of a

road. The site is now a shallow dead end slough grading from wapato up to reed canary grass and willows. A small weir at the mouth of the slough maintains shallow water throughout the low water season. Ownership of the slough is uncertain; however the property is bracketed by land owned by Georgia Pacific Corporation.

Reed Island. The Reed Island site, located at rkm 204, is a recently formed wetland area caused by erosion at the upstream end of Reed Island and subsequent deposition near the site forming a shallow backwater slough. Higher depositional areas support *Carex* species and willows while lower depressional areas are dominated by spikerush. Reed canary grass was not present at this site perhaps due to course sediment, high slope, and/or recent high water.

Franz Lake. The long-term monitoring site located the furthest up river at rkm 221 is Franz Lake, which is part of the Pierce National Wildlife Refuge. The site has an expansive area of emergent marsh extending 2 km from the mouth of the slough to a large, shallow ponded area. Several beaver dams have created a series of ponds along the length of the channel resulting in large areas of shallow-water wetland with fringing banks gradually sloping to an upland ecosystem. The sample site was located approximately 350 m from the channel mouth, spanning an area impacted by a beaver dam. The site is primarily high marsh with scattered willow saplings, fringed by willows, ash, and cottonwood.

a) b) c) d)

Figure 3. 2012 Ecosystem Monitoring sites: (a) Ilwaco; (b) Secret River; (c) Welch Island; (d)Whites Island, Cut-Off Slough; (e) Cunningham Lake; (f) Campbell Slough; (g) Government/Lemon Island; (h) Washougal River; (i) Reed Island; and (j) Franz Lake.

2.5.4 Water Year

The 2012 water year was characterized by slightly above average water surface elevations (WSE) in the fall and winter of 2011 and considerably above average WSE starting in early March continuing through August 2012 (Figure 4). During our usual habitat and hydrology sampling period in the last 2 weeks of July, the high WSE precluded sampling at any 2012 monitoring sites until early August, and delayed sampling at the up-river sites (Rkm 181 - 221) until late August.
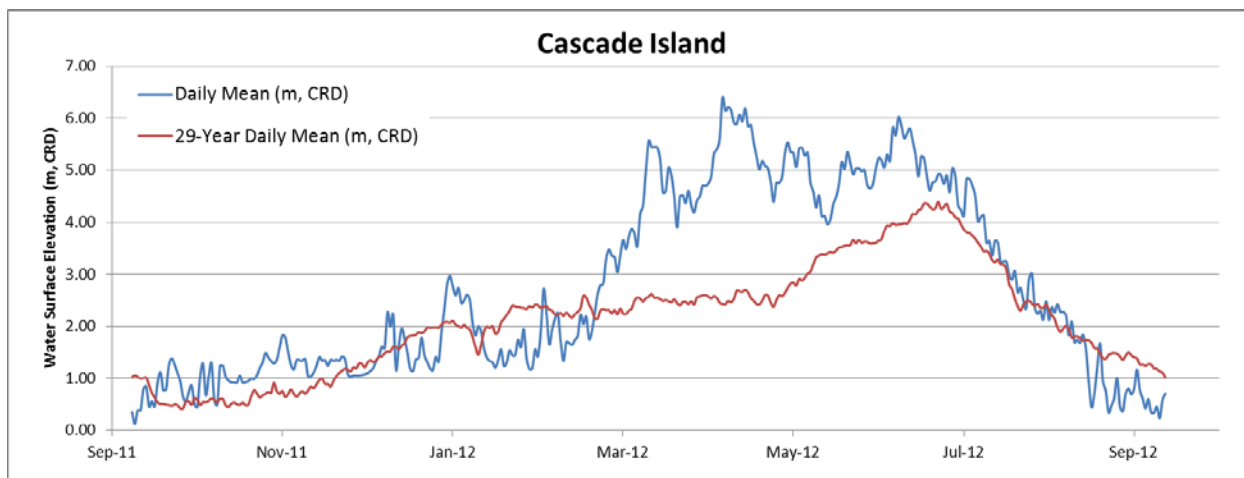
Figure 4. Water surface elevation at Cascade Island, just below Bonneville Dam (rkm 233), from October 2011 to September 2012 compared to the 29-year daily mean water surface elevation (Data from USGS National Water Information System at: http://waterdata.usgs.gov/nwis/).

3 Methods

3.1 Mainstem conditions

Overview

The Center for Coastal Margin Observation and Prediction (CMOP) at the Oregon Health and Science University (OHSU) operates two in situ water quality monitoring platforms in the mainstem Columbia River that provide baseline water quality measurements in support of the Ecosystem Monitoring Program. The first platform, funded by the National Science Foundation, was installed in July 2009 at River Mile 53 (in Reach C) and is physically located on a USGS Dolphin piling (46.182°N, -123.188°E, Figure 6A, B). A second platform, funded by the Bonneville Power Administration (BPA) under the Ecosystem Monitoring Program, was installed in August 2012 at River Mile 122 (in Reach G) and is physically located on the outer-most floating dock at the Port of Camas-Washougal (45.577°N, -122.379°E, Figure 6, C,D).

Each instrument platform consists of a physical structure, sensors, sensor control, power supply and distribution, and wireless communication. Data transmitted from the sensors is available within 1-2 hours of collection. Raw data can be downloaded from a dedicated webpage (http://columbia.loboviz.com/) and also can be accessed as part of the CMOP observation network from the CMOP server (http://www.stccmop.org/datamart/observation_network). In addition to collecting unprecedented spatial and temporal resolution of basic water quality and biogeochemical observations for the mainstem Columbia River, an additional outcome of this effort is to provide daily estimates of other useful parameters for assessing ecosystem conditions and relevant biogeochemical processes in the Columbia River watershed. One such product is flux calculations for various inorganic or organic components such as nitrate or phytoplankton biomass. Knowledge of flux of nutrients and organic matter for a large river is important for a variety of applications, including assessment of pollution, indications of eutrophication, and quantification of loading to the coastal zone, where many important ecological processes may be affected. An additional data analysis product is the assessment of Net

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Ecosystem Metabolism, which provides a daily measure of the gross primary production and aerobic respiration occurring in the river as measured by hourly changes in dissolved oxygen.
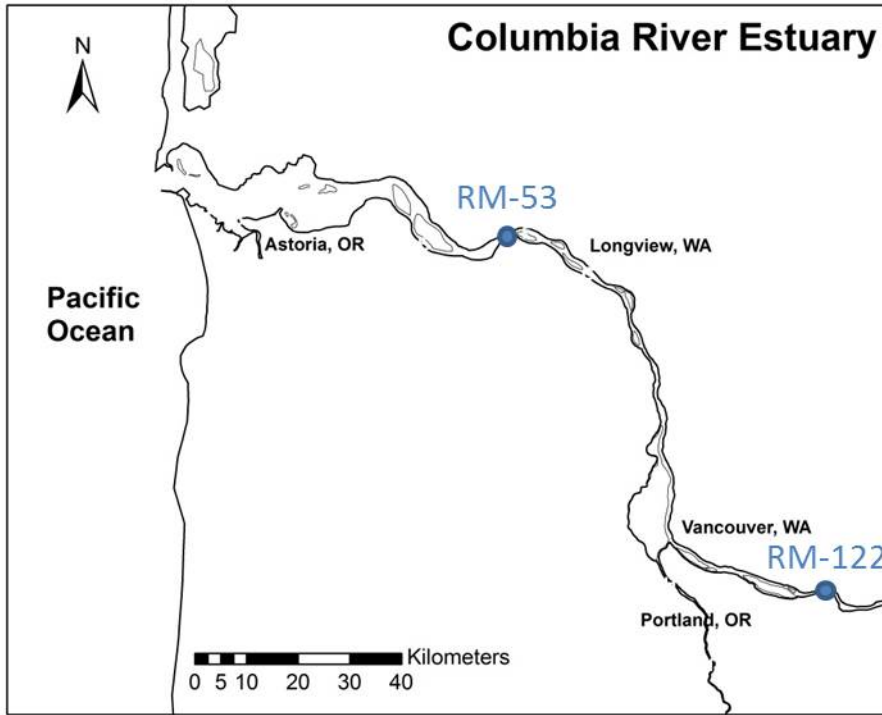
Figure 5. Station locations for the two in situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program.
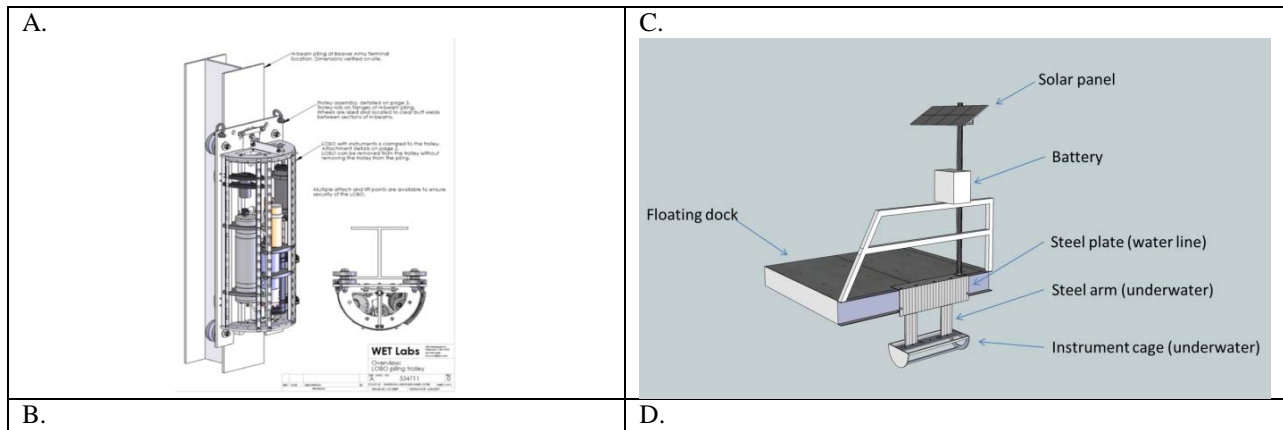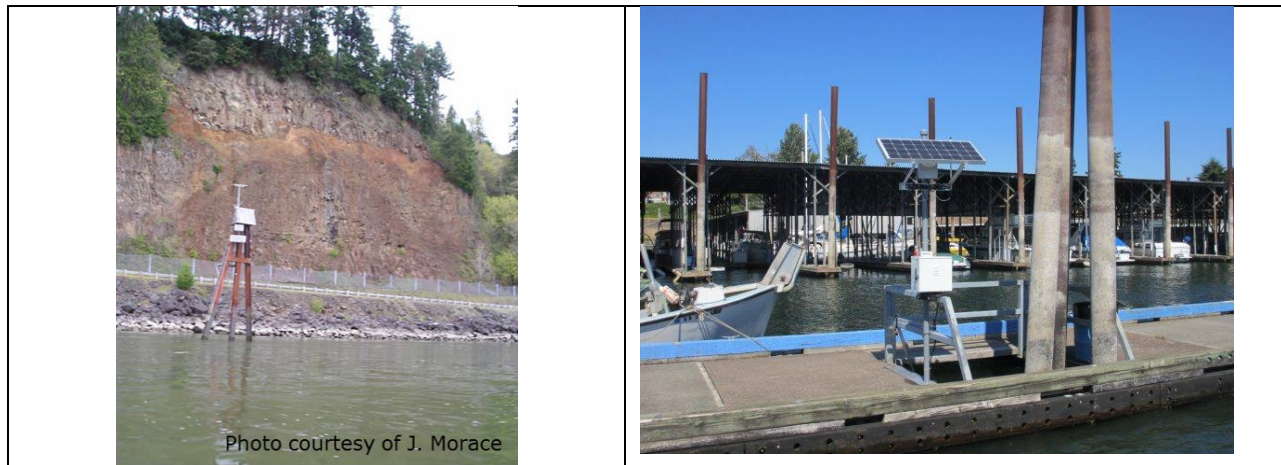
Figure 6. A) Illustration of platform components (left) for the RM-53 platform. B) Photograph of the USGS Dolphin piling at RM-53 that houses the underwater instrument platform. C) Illustration of the RM-122 platform. The steel arm and instrument cage rotate up and outwards to rest on the floating dock for maintenance purposes. D) photograph of the installation on the Port of Camas-Washougal floating dock.

Installation of RM-122 Platform at Port of Camas-Washougal

The instrument platform and sensors were purchased and acquired from Satlantic Inc. (Halifax, Canada) in winter 2012. A lease with the Port of Camas commenced in June 2012 and installation of the instrument platform was completed by faculty and staff from OHSU in August 2012. The platform is located approximately 100 meters from a USGS monitoring station (ID: CWMW) that operates seasonally to measure dissolved gas and water temperature. The physical structure of the platform consists of a power/telemetry system and fabricated steel plates that are secured to the floating dock in a manner that allows for the instruments to be deployed underwater and raised onto the dock for maintenance purposes (Figure 6C). Tampering is discouraged by: 1) deployment on a public dock with visible security signs and a staffed year-round harbormaster, 2) a low profile design (Figure 6D), and Figure 8) robust locks that prevent easy access to any of the components.

Sensor Configuration

Common instruments/sensors to both platforms are provided in Table 2. Sensors are configured to collect a sample and telemeter the data every hour. In addition to the parameters listed in Table 2, the RM-122 station is designed to operate a WET Labs Cycle-PO4 to measure dissolved ortho-phosphate concentration. This measurement is a wet chemistry analysis and therefore this instrument has reagent limitations, which restricts its operation to a reduced schedule (twice daily).

Table 2. Description of the components on the sensor platforms.

Company	Sensor	Parameters
Satlantic	LOBO	Power distribution Sensor control Wireless communication Data management
Satlantic	SUNA Nitrate	Nitrate Concentration
WET Labs	ECO-CDS	Colored Dissolved Organic Matter (CDOM)
WET Labs	WQM Water Quality Monitor	Conductivity, Temperature, Dissolved Oxygen, Turbidity, Chlorophyll Concentration

Sensor Maintenance
The sensors on both platforms are designed to operate autonomously, at high temporal resolution (hourly), and over long periods between maintenance (estimated at 3 months, although sensors are maintained at shorter intervals – see below). This is achieved through a design that maximizes power usage and minimizes bio-fouling. Antifouling is achieved through the use of: sunlight shielding (to prevent algae growth), window wipers, copper instrument surfaces, and bleach injection of the internal pumping chamber. Maintenance of the RM-53 platform is performed in collaboration with the USGS-NAWQA monitoring program that provides boat access to the platform on a regular schedule (approximately monthly). The RM-122 platform can be accessed from shore, thus there are fewer limitations of the frequency of maintenance – currently we maintain the site at the same schedule as RM-53.

Quality Control
Initial sensor calibration was performed by the manufacturer. Each instrument is supplied with a certificate of calibration, and where appropriate, instructions for recalibration. For example, the Satlantic SUNA for nitrate measurements operates with a calibration file determined at the factory under strictly controlled environmental conditions but for which can be periodically checked and modified for sensor drift by performing a "blank" measurement at our OHSU laboratory using deionized water. At longer intervals (every 1-2 years) the sensors are returned to the factory for maintenance and recalibration.

During periodic sensor maintenance samples are collected for additional quality control criteria. At RM-53, nutrients and chlorophyll a samples are returned to our OHSU laboratory and analyzed using established laboratory techniques. In addition, during the same maintenance trips the USGS NAWQA personnel collect samples for nutrients and other water chemistry parameters that can be later compared to the in situ data. An example of this data analysis is shown in Figure 2 for nitrate and chlorophyll a during 2009-2012 at RM-53.

Starting in 2013, the maintenance trips to RM-122 will also include use of an OHSU YSI Sonde (6600 V2) to provide periodic calibrated measurements of conductivity, chlorophyll a, turbidity, dissolved oxygen, and pH by following similar protocols to the USGS procedure at RM-53.
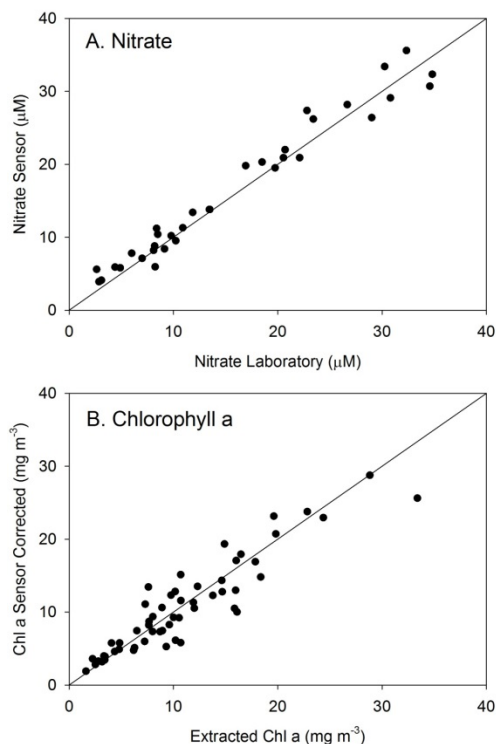
Figure 7. QC data. A) Nitrate concentration measured by the Satlantic SUNA sensor at River Mile 53 (y-axis) and by sample collection and laboratory analysis (x-axis). Also shown is the 1:1 relationship (solid line). B) Chlorophyll a measured by the Wet Labs WQM-FLNTU (y-axis) and by sample collection and laboratory analysis (x-axis). Samples were collected during the period July 2009- August 2012.

Advanced Data Analysis

Measurements of flux of inorganic and organic material can be achieved by multiplying the daily average concentration (computed from the hourly measurements) and the daily river discharge to determine a daily flux. The resulting high resolution flux measurements are useful to observe variations in the system associated with episodic events such as storm runoff and to monitor changes associated with seasonal shifts in climate, and thus are compared to river discharge and other measured parameters to observe correlations in the time series data. Columbia River discharge is measured at RM-53 by the USGS (http://waterdata.usgs.gov/usa/nwis/uv?site_no=14246900) and at Bonneville Dam by BPA. The difference between discharges at the two sites (Figure 8) reflects inputs from the tributaries of the lower Columbia River, including the Willamette, Cowlitz, and many smaller rivers.

Biological processes associated with organic matter production by photosynthesis and organic matter respiration by all organisms in an aquatic ecosystem can be measured using the "open water" technique first described by Odum in the 1950's (Odum et al. 1984). The sum of these metabolic processes over a 24 hour period is termed Net Ecosystem Metabolism (NEM) or Net Community Metabolism (NCM), and can be used to quantify and better understand benthic and pelagic primary production, nutrient biogeochemistry, and carbon export to the coastal margin. The basic premise of NEM measurements is to use high resolution, dissolved oxygen measurements (as collected at RM-53 and RM-122) to quantify oxygen production by photosynthesis and oxygen consumption by aerobic respiration. After accounting for air-water gas diffusion, the daily net change in oxygen provides an estimate of the net production or consumption of organic carbon within the riverine food web. Although the method has some recognized limitations, the holistic approach and ecosystem scale application is appealing for a large river such as

the Columbia, since the system cannot be adequately sampled for these processes using existing resources and methodology. For a detailed description of the methodology used to calculate NEM at RM-53, readers can refer to Needoba et al. 2013.
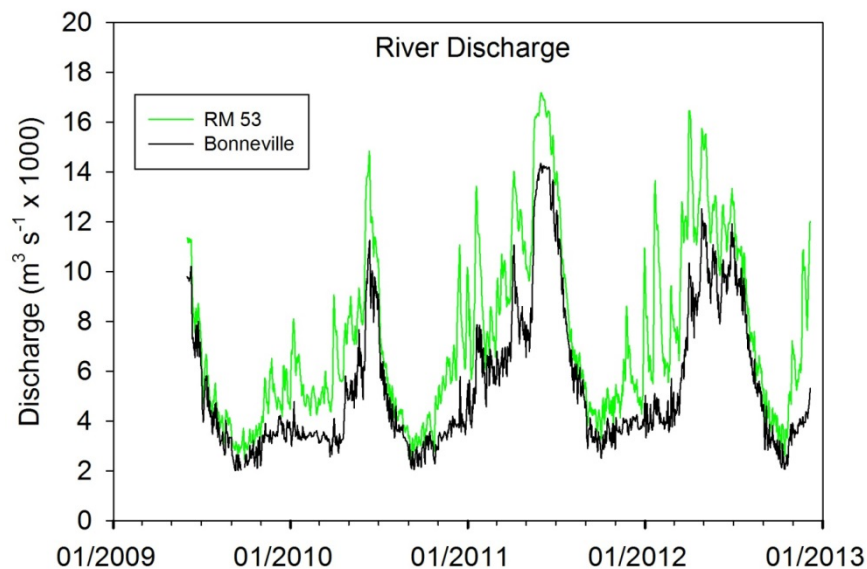
Figure 8. Columbia River daily discharge (m³ d⁻¹) measured at RM-53 (green line) and Bonneville Dam (black line) during the period July 2009-December 2013. The difference in discharge between the two sites represents the input from tributaries in the Lower Columbia River.

3.2 Abiotic site conditions

3.2.1 Continuous water-quality data (temperature, DO, pH, conductivity)

In 2012, USGS monitored water quality and assessed food-web resources at the four of the trend sites: Franz Lake Slough, Campbell Slough, Whites Island, and Ilwaco (Table 3). For the fifth consecutive year, USGS deployed a continuous water-quality monitor at Campbell Slough in the Roth Unit of the Ridgefield National Wildlife Refuge in 2012. This site in Reach F has been sampled for vegetation since 2005 (PNNL) and for fish since 2007 (NOAA Fisheries). USGS also deployed a monitor in a tidal slough in Whites Island in the Columbia River. This site in Reach C was monitored for water quality in 2009 and 2011-2012 and sampled for vegetation (PNNL) and fish (NOAA Fisheries) since 2009. Franz Lake Slough in Franz Lake National Wildlife Refuge in Reach H was monitored for water quality in 2011—2012 and for vegetation (PNNL) and fish (NOAA Fisheries) in 2008-2009 and 2011-2012. Water quality was also monitored in a tidal channel of the Columbia River near Ilwaco marina in Baker Bay, WA. This site in Reach A was monitored by all partners in 2011-2012.

Table 3. Site information for locations of water-quality monitors in 2012. *In order to be consistent with site names used by other monitoring partners, site names used in this report differ from official USGS site names.

Site name*	USGS site number	USGS site name*	Reach	Latitude	Longitude	Monitor deployment date	Monitor retrieval date
Franz Lake Slough	453604122060000	Franz Lake Slough Entrance, Columbia River, WA	H	45° 36' 04"	-122° 06' 00"	March 28	July 31

Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47' 05"	-122° 45' 15"	March 28	July 31
Whites Island	460939123201600	Cut-Off Slough, Whites Island, Columbia River, WA	C	46° 09' 39"	-123° 20' 16"	March 27	July 30
Ilwaco	461802124024400	Columbia R. at Port of Ilwaco Marina at Ilwaco, WA	A	46° 18' 02"	-124° 02' 43"	March 27	July 30

The water-quality monitors deployed were Yellow Springs Instruments (YSI) models 6600EDS and 6920V2 equipped with water temperature, specific conductance, pH, dissolved oxygen, and depth probes. Table 4 provides 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 during the period when juvenile salmonids were present at the sites, so the deployment period includes some time before and after they are expected to use the sites. In 2012, the monitors were deployed during the last week of March through the last week of July, with visits roughly every 4 weeks to change the batteries, clean the instruments, check the calibration of the variables, and make any adjustments needed. Before 2011, the targeted monitoring period was May through August, but was shifted to one month earlier starting in 2011, so as to capture conditions during months when salmonids had been found at the sites in recent years.

Table 4. 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]

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	-5–70 °C	0.01 °C	±0.15 °C
Specific conductance	0–100,000 µS/cm	1 µS/cm	±1 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

In this report, monitored water-quality data are compared to standards for temperature, pH, and dissolved oxygen set by the Washington Department of Ecology to protect salmonid spawning, rearing, and migration. Those standards are available at http://www.ecy.wa.gov/programs/wq/swqs/criteria.html.

3.2.2 Nutrients (N,P)

Nitrogen and phosphorus are the nutrients that are most commonly limited in the environment relative to the amounts required for algal growth. Concentrations of biologically available forms of nitrogen and phosphorus in the water column are therefore important factors that can influence rates of algal growth. To analyze water-column nutrient concentrations, one-liter water grab samples were collected from representative areas within the sites and composited in a plastic churn. Water from the churn was subsampled and nitrogen and phosphorus species were analyzed during six sampling events in 2012.

3.2.3 Photosynthetically Available Radiation (PAR)

Light at wavelengths of 400-700 nanometers can penetrate the water column and be absorbed by photosynthetic pigments in algae and plants and used for photosynthesis (Day et al. 1989). Light in this range is called photosynthetically available radiation, or PAR. During every sampling event, a vertical profile of PAR was measured at ten-centimeter depth increments using a LiCor LI-1400 meter and LI-193SA sensor.

3.3 Habitat Structure

3.3.1 Monitoring Sites

In 2012, PNNL collected field data on vegetation and habitat conditions at 10 sites (Figure 2). Details regarding these sites and other sites visited in 2012 are provided in
Table 5.

Table 5. Site location and sampling dates for each site visited in 2012.

Site Name	Site Code	River kilometer (rkm)	Site Type	Sampling Date	Monitoring Activity
Ilwaco	BBM	6	Trend	8/4/12	Full sampling
Secret River (low marsh)	SRM-L	37	Trend	8/2/12	Full sampling
Secret River (high marsh)	SRM-H	37	Trend	8/3/12	Full sampling
Welch Island	WI2	53	Trend	8/1/12	Full sampling
Whites Island	WHC	72	Trend	7/31/12	Full sampling
Prescott Slough	PSM	115	NA[1]	8/8/13	Sensor retrieval[2]
Burke Island	BIM	131	Status (2011)[3]	8/9/12	Sensor retrieval
Goat Island	GIC	131	Status (2011)	8/9/12	Sensor retrieval
Deer Island	DIC	132	Status (2011)	8/9/12	Sensor retrieval
Cunningham Lake	CLM	145	Trend	8/8/12	Full sampling
Campbell Slough	CS1	149	Trend	8/10/12	Full sampling
Government/Lemon Island	GOM	181	Status	8/28/12	Full sampling
Washougal River	OWR	195	Status	8/27/12	Full sampling
Reed Island	RI2	204	Status	8/29/12	Full sampling
Franz Lake	FLM	221	Trend	8/30/12	Full sampling

1 Prescott Slough was an alternative monitoring site for 2011 so a depth sensor and sediment accretion stakes were deployed in 2010. Retrieval was not possible until 2012.
2 Sensor retrieval includes measurement and retrieval of depth sensor and measurement of sediment accretion stakes.
3 Sensor deployment was not possible at these sites until 2011, therefore retrieval occurred in 2012.
Detailed maps of the 2012 monitoring sites are presented in Appendices.

3.3.2 Metrics Monitored

This study is using standard monitoring protocols developed for the LCRE (Roegner et al. 2009). Five metrics are included in this part of the monitoring program. These metrics have been determined to

represent important structural components, which can be inferred to provide habitat functions. The rationale for choosing these metrics is discussed below.

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

Assessment of channel cross sections and channel networks provides information on the potential for many important estuarine functions including fish access (Simenstad and Cordell 2000) and export of prey, organic matter, and nutrients. This information is also necessary to develop the relationship between cross-section dimensions and marsh size, which aids in understanding the channel dimensions necessary for a self-maintaining restored area (Diefenderfer and Montgomery 2009). The primary objective associated with the channel data collection effort is to determine how unmodified channels may differ between reaches within the region with regard to habitat opportunity (Bottom et al. 2005).

3.3.3 Historical Analysis

Information on the history of the sites monitored as part of the EMP provides context for current conditions and improves hypothesis development regarding sediment composition, vegetation assemblages, and channel development. A GIS analysis was conducted through a collaborative effort of the EMP and the Salmon Benefits project (EST-P-09-1) funded by the USACE evaluating the recently developed land cover classes from the historical T-sheets from the late 1800s (Burke 2010) and current land cover classes (Sanborn Map Company and LCEP 2011). The analysis focused on sites that were thought to have been present historically (i.e., not created from dredged material placement); six sites from that analysis are part of the EMP.
Table 6 summarizes the results for the six sites.

Table 6. Proportion of area covered by historical and current land cover classes, as defined by the historical land cover delineation from the late 1800 T-sheet maps (Burke 2010) and the 2010 land cover analysis (Sanborn Map Company and LCEP 2011).

	Site					
Land Cover Class	**BBM**	**SRM/S**	**RIM**	**BIM**	**CLM**	**CS1**
Historical	proportion of total area					
1 - Water	0.95	0.00	1.00	0.52	0.73	0.87
2 - Flats	0.04	0.01	0.00	0.05	0.00	0.00
3 - Marsh	0.00	0.65	0.00	0.09	0.20	0.00

4 - Shrub Wetland	0.00	0.00	0.00	0.00	0.00	0.00
5 - Forested Wetland	0.00	0.13	0.00	0.00	0.04	0.00
6 - Upland (forest, shrub, grass)	0.02	0.21	0.00	0.34	0.04	0.00
Current	proportion of total area					
1 - Water	0.39	0.33	0.11	0.10	0.68	0.12
2 - Flats	0.01	0.00	0.00	0.00	0.00	0.00
3 - Marsh	0.59	0.33	0.67	0.70	0.17	0.83
4 - Shrub Wetland	0.00	0.03	0.20	0.01	0.01	0.00
5 - Forested Wetland	0.00	0.29	0.02	0.19	0.15	0.05
6 - Upland (forest, shrub, grass)	0.01	0.01	0.00	0.00	0.00	0.00

Four of the sites had very little or no areas classified as marsh on the historical maps: Ilwaco, Ryan Island, Burke Island, and Campbell Slough. However, these four sites all had greater than 50 percent cover of marsh in the current analysis. The reasons for the changes at these sites are varied, but in general can be attributed to changes in hydrologic processes and natural succession. The Cunningham Lake site had very little change in marsh cover, but a decrease in open water and an increase in forested wetland indicating a successional shift in habitat types over this time period. One site, Secret River (which includes the marsh and swamp area of the site), had a decrease of nearly one half as much marsh area from the historical area. The reason for this change is uncertain.

3.3.4 Annual Monitoring

As in previous years (2005-2011), 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, and collected sediment samples. Since the 2011 sampling year, biomass collection was performed at all of the trend sites, excluding Cunningham Lake. 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 LCRE.

The vegetation monitoring schedule was delayed this year by an extended high water period. All vegetation sampling was delayed at the upriver sites until late August. The high water influenced several of the metrics recorded, including vegetation percent cover. At the upper river sites, lower portions of the marshes were inundated during the entire monitoring period. All sampling occurred between 7/31/2012 and 8/30/2012 (
Table 5). Biomass sampling did not occur at Campbell Slough due to cattle grazing at the site.

3.3.4.1 *Sediment Composition*
Sediment samples were collected within each major vegetation community strata at Secret River, Welch Island, Government/Lemon Island, Washougal River, and Reed Island. Sediment samples were collected in 2008 at Campbell Slough and Cunningham Lake sites, at Franz Lake and Whites Island sites in 2009, and at the Ilwaco site in 2011 and therefore were not recollected this year. Five 10-cm cores were collected within each stratum 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. Two additional sediment samplings were added in 2012 to look specifically at TOC associated with a vegetated core. Following the methods of Thom (1992), we took a 12 cm diameter, 25 cm deep, and divided it into 5 cm sections. One core was taken at Secret River and one at the Ilwaco site in similar vegetation communities. Samples were analyzed within 5 days from the time of collection.

3.3.4.2   *Sediment Accretion Rate*
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 stakes, termed sedimentation stakes, are used to determine gross annual rates of sediment accretion or erosion (Roegner et al. 2009). Sedimentation stakes are measured annually at each of the trend sites and were installed and measured at the Burke, Deer Island and Goat Island sites in 2011, where they measured one year later in 2012. Sedimentation stakes were installed and measured at the Secret River, Welch Island, Government/Lemon Island, Washougal River, and Reed sites this year, and will be measured again in 2013. The accretion or erosion rate is calculated by averaging the 11 measurements from each year and comparing the difference.

3.3.4.3   *Hydrology*
In 2010, pressure transducers (HOBO Water Level Data Loggers, Onset Computer Corporation) were deployed at each of the trend sites as a means of logging in situ water level data for one year. Sensors were redeployed at Whites Island, Cunningham Lake, Campbell Slough and Franz Lake in the summer of 2010. During the fall of 2010, a sensor was deployed at the Ilwaco site that turned out to be faulty, and was replaced in April 2011. For the Reach E sites, sensors were deployed at Burke and Goat Islands in July 2011 and were retrieved during the summer of 2012. The sensor at Goat Island was also used for Deer Island. For the Reach G sites, sensors were deployed in November 2011 at Government/Lemon Island, McGuire Island, Washougal River, and Reed Island. The sensors will be retrieved in the fall of 2012.

3.3.4.4   *Salinity*
In order to better assess the influence of salinity on habitat, a conductivity data logger (Onset Computer Corporation) was deployed at the Ilwaco site in August of 2011. The data logger records conductivity and temperature within the slough and derives salinity on-the-fly from those two measurements, based on the Practical Salinity Scale of 1978 (see Dauphinee 1980 for description of conversion). Sensor was cleaned, downloaded, and a verification sample taken in February and August, 2012.

3.3.4.5   *Vegetation Assemblage*
The vegetation sample areas at each site were selected to be near a tidal channel and to be representative of the elevations and vegetation communities present at the site. This was easier in the upper portions of the estuary, where the sites were generally narrower and the entire elevation range could be easily covered in the sample area. In the lower estuary, the sites were broad and covered a larger area, so in some cases multiple sample areas were surveyed if possible to cover different vegetation communities (e.g., low marsh and high marsh).

Along each transect, vegetative percent cover was evaluated at 2-10 meter intervals. Interval and transect length was based on the marsh size and/or the vegetation homogeneity. At each interval on the transect tape, a 1-m^2 quadrat was placed on the substrate and percent cover was estimated by observers in 5% increments. If two observers were collecting data then they worked together initially to ensure their observations were "calibrated." Species were recorded by four letter codes (1st two letters of genus and 1st two letters of species, with a number added if the code had already been used, e.g., LYAM is *Lysichiton americanus* and 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 later identification using taxonomic keys or manuals at the laboratory. If an accurate identification was not resolved, the plant remained "unidentified" within the database.

3.3.4.6   *Vegetation Biomass*

Beginning in 2011, above ground biomass was sampled to estimate the primary productivity at the trend sites. The sampling continued in 2012 to evaluate inter-annual variability and to implement refined sampling methods developed from the first year of monitoring. For the emergent marsh biomass sampling, a 1-m square plot was randomly placed along the established vegetation transect, but off-set 2-m from the transect to ensure that the biomass plots did not intersect the vegetation percent cover plots. In 2012, the biomass was randomly sampled within distinct vegetation strata to 1) more clearly associate the samples with vegetation type and 2) to reduce the variability between samples within a strata.. Within the 1-m square biomass plot, a 0.1 m^2 quadrat was placed in a randomly selected corner and all rooted vegetation, live or dead, was removed using shears. Each sample was placed in a uniquely numbered bag, and held in a cooler for the remainder of the sampling trip. For the submerged aquatic vegetation (SAV) plots, similar methods were employed with the exception of the placement of the plots. Either existing transects were extended past the baseline or new transects were created to reach the main slough. In some instances, an existing transect intersected the slough and an SAV plot was randomly placed along it. Depending on the width of the channel, either one or two SAV plots were randomly placed along each transect. Vegetation species were recorded in field notebooks along with the corresponding biomass sample number. In the laboratory, the biomass samples were stored in a cold room until processing could begin. The samples were then individually rinsed of all non-organic material, and obvious root material was removed. Pre-weighed pieces of tinfoil were used to secure the individual biomass samples, a wet weight was then measured, and the samples were placed in an oven set at 90°C for three to four days. When the samples were deemed completely dry, a second weight was then measured for each sample, and entered either into a datasheet or directly into a spreadsheet software program.

3.3.4.7   *Vegetation Community Mapping*

Using Trimble GeoXT and GeoXH handheld global positioning system (GPS) units, 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 points) were also mapped. All data were input to a GIS, and maps of each site showing major communities and features were created (Appendices). Trend (repeat) sites were only re-mapped if there was an observable change at the site.

3.3.4.8 *Elevation*

At all sites, elevation was measured at each of the following locations: vegetation quadrats, the water level sensor, sediment accretion stakes, vegetation community boundaries, and in the channels. Elevation was surveyed using a Trimble real time kinematic (RTK) GPS with survey-grade accuracy and an auto-level. 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) software was used to process the data. Each survey was imported and overviewed by a scientist. Benchmark information was entered into TGO and rover antenna heights were corrected for disc sink (measured at each survey point to the nearest centimeter) 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). Using the CRD alleviates elevation differences associated with the increasing elevation of the river bed in the landward direction. Sites below RKM 37, the lower limit of the CRD, were converted to mean lower low water (MLLW).

All survey notes 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. All elevations in this report are referenced to CRD unless noted otherwise.

3.3.4.9 *Channel Metrics*

Elevation surveys were conducted for channel cross-sections at all sites. Five channel cross-sections were surveyed at the new sites starting near the mouth of the channel and continuing past the marsh vegetation survey area. For the purposes of the Ecosystem Monitoring program, the channel mouth was generally defined as the location where the vegetated bank begins, and this location is usually designated as channel cross section 1 (XS1). Exceptions based on site configuration necessarily occur. For example, the Cunningham Lake site is approximately 6.5 km from Multnomah Channel and the mouth has not been surveyed as part of this program. At Franz Lake, the cross-section designated as "XS0" is the XS that would typically be designated as XS1, but because the initial survey did not include a XS at the outer edge of the bank vegetation, XS0 was added later to ensure the mouth was surveyed. At repeat sites, a single cross section was re-surveyed; most often this was the mouth cross section, unless the mouth was not part of the study area. When five cross sections were measured, the channel cross-sections were distributed evenly along the channel. Exceptions were made where a major side-channel met with the main channel. In these cases, the cross-section was moved above the confluence. Site maps identify the locations of all cross-sections (Appendix B). Additional notes were made for features of interest located at the cross-section: top and bottom of bank, vegetation edges, and thalweg. Data from the surveys were used to calculated channel depth. The elevation data were combined with hydrology data to calculate inundation times for the channel and bank edge.

3.3.4.10 *Inundation*

The data from the water level sensors were used to calculate inundation metrics from the marsh and channel elevations collected at those sites. Inundations were calculated for only the trend sites, with the exception of Franz Lake, where the sensor could not be found at the time of retrieval because of beaver activity. Due to the faulty sensor at Ilwaco, inundation metrics were only calculated from April 2011 to August 2011.

The percent of time each marsh was inundated was calculated for the entire period of record (approximately one year) and for the growing season, April 22-October 12. The growing season is 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 Clark County growing season is used for all the sites in the estuary so that the inundation calculations are standardized to one period. The inundation frequency during the growing season was only calculated during daylight hours (between 0900 and 1700). This limitation was employed primarily for tidal areas where the timing of the daily high tide can be a factor in the amount of time available for plants to photosynthesize.

The percent of time each channel was inundated was calculated for the thalweg and top-of-bank elevations and for two time periods. In order to estimate habitat opportunity for juvenile salmonids, water depth of 50 cm was added to the thalweg elevation of each cross-section as an indicator of the amount of water adequate for fish use of the channel (Nichole Sather, personal communication). Likewise, a 10 cm water depth was added to the top of bank elevation at each cross-section to represent a minimum amount of water needed for fish to access the vegetation at the edge of the bank (Bottom et al., 2005; Kurt Fresh personal communication). The periods assessed were 1) the deployment period (generally July to July) and 2) the period from March 1 through July 31, which represented the peak juvenile Chinook migration period as determined from data collected as part of this Ecosystem Monitoring Program and other studies (Bottom et al. 2005; Sather et al. 2011).

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 in Europe (Gowing et al. 2002; Simon et al. 1997; Araya et al. 2010) we calculated the sum exceedance value (SEV) using the following equation:

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

where *n* is the number of hours present in the time period evaluated, and d_{elev} is the hourly water surface elevation above the average marsh elevation. This differs from previous LCRE studies (Borde et al. 2011 and Sagar et al. 2011) in which the daily mean water surface elevation was used in the calculation rather than the hourly water level elevation used here. The latter was chosen to ensure we captured daily inundation fluctuations that occur in the more tidally dominated sites. The time periods evaluated were the annual deployment period and the growing season. Both periods were standardized to include the same days in each year, as follows:

Growing season: April 22 to June 21 and August 20 to October 12 (115 days)
Annual deployment period: August 20 to June 21 (of the next year; 306 days)

This standardization was necessary because in the past, the deployment and retrieval dates for sensors varied between June 21 and August 20 and to compare calculations from past and present data required that the same time periods be used.

The SEVs were also calculated at the three up-river trend sites using the method described above. This differs from the previous trend analyses where daily mean water levels were used for the SEV calculation. The hourly method provides higher resolution results, accounts for daily variation in water levels, and also provides consistent results between the SEV analyses reported for the status sites. For the trend analysis, the SEV was calculated for the average elevation of the three to five species comprise most of the vegetation cover at the study sites using the water surface elevations measured each year during the growing season. For the years that water surface elevation data were not collected at the sites, we used data from the NOAA tide station with the greatest similarity in hydrologic magnitude and pattern. For Cunningham Lake this was the St. Helens station, for Campbell Slough it was the Vancouver station, and for Franz Lake it was the USGS Bonneville station. A linear regression model was developed between existing site data and the station data from the same years ($r^2 \geq 0.99$). The model was then applied to the station data to predict the site water surface elevation for missing years. Average water years were used to predict average or low water years an high water years to predict results in higher water years.

3.3.4.11 *Vegetation Similarity Analysis at Trend Sites*

Similarity analyses, using the Bray-Curtis similarity coefficient (S') as a measure of distance between years (described in Clarke and Warwick 2001), were performed on percent cover data from the trend sites by using Primer™. Percent cover data were arc-sin, square-root transformed, but were not standardized, prior to analyses. The similarity matrix was converted to a dendrogram by using the hierarchical, unweighted pair-group mean-averaging method of clustering. Clusters combined at greater linkage distance are more dissimilar than those combined at smaller linkage distances. Each similarity matrix was also transformed to a two-dimensional non-metric multidimensional scaling (nMDS) plot by using Primer™. The program generated each plot by restarting the nMDS algorithm 30 times and selecting the plot that had the lowest stress value (Clarke and Warwick 2001). The Bray-Curtis similarity metric is one commonly used method for assessing these relationships (Clarke and Warwick 2001).

3.4   Food web

3.4.1 Primary Productivity

3.4.1.1   *Phytoplankton/periphyton*

3.4.1.1.1   Abundance

Algal abundance was estimated in three ways: (1) from pigment concentrations, (2) from ash-free dry mass (AFDM) and by direct counts using light microscopy. Algal abundance 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 (Hambrook Berkman and Canova 2007). Abundance of phytoplankton (suspended algae) and periphyton (attached algae) were measured according to pigments and AFDM in concert to provide a more complete assessment of the abundance of primary producers at the sites. Water samples collected from representative locations at the site were composited in a plastic churn. A subsample of water from the churn was filtered onto a 47-millimeter glass-fiber filter (GF/F) for each chlorophyll *a* and AFDM analysis of phytoplankton and kept frozen (-20°C) until analyzed. Periphyton samples were scraped from a measured area of a natural

or artificial substrate (rocks, underwater branches, pipes, wood, etc.), diluted with deionized water, homogenized, and subsampled as described above for phytoplankton.

Algal abundance was also determined by enumeration of individual cells using inverted light microscopy. Duplicate 100 mL whole water samples were collected from each of six trend sites on the dates shown in Table 7. The samples were preserved in 1% Lugol's iodine and examined at 100, 200 and 400x magnification using a Leica DMIL inverted light microscope following concentration achieved through settling 10-25 mL of sample in Utermohl chambers (Utermohl 1958) overnight (~24 h). Cell counts were performed at 200 or 400x magnification, with an additional scan done at 100x magnification to capture rare cells in a broader scan of the slide. The estimated error in abundance measurements was <5% at the class level, and ~10% for genus-level counts. After counting, the concentrated material was then transferred to small (7 ml) sampling vials for archiving and more detailed examination of acid-cleaned material should additional studies of diatom biodiversity be carried out in the future.

Table 7. List of samples collected from six trend sites in the Lower Columbia River in 2012. Those for which phytoplankton samples were analyzed are noted.

Site	Reach	Sample collection date	Sample processed	NOTES
ILWACO MARINA	A	3/27/12		MARINA
`	A	4/12/12	X	MARINA
	A	4/24/12	X	MARINA
	A	5/15/12	X	MONITOR SITE
	A	5/31/12		MONITOR SITE
	A	7/9/12		MARINA
	A	7/9/12		MONITOR SITE
	A	7/30/12		MARINA
WELCH ISLAND	B	5/16/12	X	
	B	6/13/12	X	
	B	7/17/12	X	
SECRET RIVER	B	5/16/12	X	
	B	6/13/12	X	
	B	7/17/12	X	
WHITES ISLAND	C	3/27/12		
	C	4/11/12	X	
	C	4/24/12		
	C	5/17/12	X	
	C	5/29/12		
	C	6/27/12	X	
	C	7/9/12		
	C	7/30/12		
CAMPBELL SLOUGH	F	3/28/12		
	F	4/9/12	X	
	F	4/23/12		
	F	5/13/12	X	
	F	5/31/12		
	F	6/25/12	X	
	F	7/11/12		
	F	7/31/12		
FRANZ LAKE SLOUGH	H	3/28/12		
	H	4/10/12	X	
	H	4/23/12		

H	5/14/12	X
H	5/30/12	
H	6/26/12	X
H	7/10/12	
H	7/31/12	

Phytoplankton species diversity was calculated using the Shannon-Weiner diversity index (Shannon and Weaver 1949), similar to that described below for fish species diversity:

$$H' = \sum p_i \ln p_i$$

where p_i is 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, and $\ln p_i$ is the natural logarithm of the proportional abundance of that species. Species richness, S (the total number of species), and total phytoplankton abundance, N, were also calculated.

3.4.1.1.2   Rates
Estimation of algal productivity is important in the assessment of aquatic food-web resources in order to characterize organic matter production at the base of the food chain. Rates of primary production of phytoplankton (suspended algae) and periphyton (attached algae) were assessed at the four of the trend sites in 2011-2012 in the following experiments.

3.4.1.1.2.1 ^{14}C Uptake Phytoplankton Productivity Experiment (2011)
The uptake of the radio-tracer, carbon-14, in the form of sodium bicarbonate during photosynthesis can be used to determine the in-situ rate of phytoplankton production in the environment (Wetzel and Likens 1991). Using this approach, water samples with a measured concentration of dissolved inorganic carbon-12 (DI^{12}C) are spiked with a known amount of radioactive tracer carbon-14 (^{14}C) and incubated in bottles in the stream. After 2 to 4 hours, the samples are filtered and the amount of ^{14}C incorporated into the algal biomass during the incubation period is measured from the filters. An isotopic correction factor of 1.06 is used in the calculation of ^{14}C assimilated to account for the preferential uptake of the lighter ^{12}C isotope over ^{14}C isotope by phytoplankton (Wetzel and Likens 1991). The uptake of ^{14}C relative to the total ^{14}C that is available is assumed to be equivalent to the proportion of DI^{12}C that is incorporated during photosynthesis, relative to the total DI^{12}C available, as follows:

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

(modified from Wetzel and Likens 1991). Therefore, the calculated DI^{12}C assimilated value is used to determine the rate of primary production in mass of carbon assimilated per volume per time. ^{14}C assimilation by phytoplankton was measured using a liquid scintillation counter at Oregon Health & Science University in 2011 following acidification of the filters to remove inorganic carbon as CO_2. To each sample, 0.25 mL of 1 N trace metal grade HCl was added and left to fume overnight. Two ^{14}C uptake experiments were done at each site, except for Campbell Slough, where limited site access in 2011 only allowed for one experiment. Results of each experiment are reported as the average of three replicates, corrected for measured carbon uptake in dark conditions and retention of ^{14}C on the filters in the absence of phytoplankton.

3.4.1.1.2.2 ^{13}C Uptake Phytoplankton Productivity Experiment (2012)

In 2012, phytoplankton productivity was estimated using the ^{14}C uptake approach described above, except the stable isotope ^{13}C as a tracer in place of ^{14}C (Hama et al. 1983). This method eliminates the safety and environmental impacts and regulatory burdens associated with the use of radioisotopes. Samples were spiked with a solution of ^{13}C-labeled sodium bicarbonate (NaH$^{13}CO_3$) at a concentration of ~8% of the average ambient dissolved inorganic carbon (DIC) across the four sites based on calculations from 2011. This spike rate is in the range of rates used in other studies (Hama et al. 1983; Kanda et al. 1985; Hashimoto et al. 2005). In place of the liquid scintillation counter, an elemental analyzer coupled to an isotope ratio mass spectrometer was used to analyze the filters for ^{13}C incorporated into algal biomass during the incubation period.

3.4.1.1.2.3 Periphytometers: Periphyton Productivity Experiments

Nutrient-diffusing substrate (NDS) periphytometers can be used to estimate periphyton productivity. Micro-NDS periphytometers, as described by Wise et al. (2009), were used to estimate periphyton accrual during a two-week period three times during the monitoring period in 2011 and 2012. For each deployment, eight 40-milliliter glass vials were filled with each treatment solution: deionized water (control treatment), sodium nitrate solution (nitrogen [N] treatment, 350 micromolar [µM] as N), sodium hydrogen phosphate solution (phosphorus [P] treatment, 100 µM as P), or N plus P solution (NP 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 filter and a glass-fiber filter, with the latter serving as the artificial substrate for periphyton growth. Half of the replicates of each treatment were covered with 18 x 14 mesh fiberglass window screen to test for the effect of grazers on phytoplankton accrual. It was expected that if grazing impacted periphyton accrual on the filters, the screened filters would have higher chlorophyll *a* concentrations than the unscreened filters. If periphyton accrual on screened and unscreened filters differed significantly, then only the screened values would be used for the productivity calculations.

3.4.2 Secondary Productivity

3.4.2.1 *Zooplankton*

Secondary productivity (the rate of growth of consumers of primary production) was not measured directly, but was estimated from the abundance of zooplankton. The samples were collected from near the surface of the water (<1 m) using an 80 µm nylon mesh net with a mouth diameter of 0.5 m and a length of 2 m. A list of the collection sites and dates they were sampled is given in Table 8.

Table 8. List of samples collected from six trend sites in the Lower Columbia River in 2012. Those for which phytoplankton samples were analyzed are noted.

Site	Reach	Sample collection date	Sample processed	NOTES
ILWACO MARINA	A	3/27/12	X	MARINA & MONITORING SITE SAMPLES; NET 75% SUBMERGED DURING TOW
`	A	4/12/12		MARINA; QUESTIONABLE FLOW METER DATA
	A	4/24/12	X	MARINA
	A	5/15/12		MONITOR SITE; NET 50%

				SUBMERGED DURING TOW; POOR FIXATION
	A	5/31/12	X	MONITOR SITE
	A	7/9/12	X	MARINA
	A	7/9/12	X	MONITOR SITE; LOTS OF SAND IN TOW MATERIAL
	A	7/30/12	X	MARINA
WELCH ISLAND	B	5/16/12	X	
	B	6/13/12	X	
	B	7/17/12	X	
SECRET RIVER	B	5/16/12	X	
	B	6/13/12	X	
	B	7/17/12	X	
WHITES ISLAND	C	3/27/12		
	C	4/11/12		
	C	4/24/12	X	
	C	5/17/12	X	
	C	5/29/12	X	
	C	6/27/12	X	
	C	7/9/12	X	
	C	7/30/12	X	NET 50% SUBMERGED DURING TOW
CAMPBELL SLOUGH	F	3/28/12	X	FLOW METER DID NOT SPIN CONTINUOUSLY; VOLUME SUSPECT
	F	4/9/12		QUESTIONABLE FLOW METER DATA DUE TO SLOW WATER VELOCITY
	F	4/23/12	X	
	F	5/13/12	X	QUESTIONABLE FLOW METER DATA DUE TO SLOW WATER VELOCITY
	F	5/31/12	X	QUESTIONABLE FLOW METER DATA DUE TO SLOW WATER VELOCITY
	F	6/25/12	X	
	F	7/11/12	X	
	F	7/31/12		NO TOW PERFORMED- LIMITED ACCESS
FRANZ LAKE SLOUGH	H	3/28/12		
	H	4/10/12		NET 90% SUBMERGED DURING TOW
	H	4/23/12	X	QUESTIONABLE FLOW METER DATA
	H	5/14/12	X	
	H	5/30/12	X	NET 90% SUBMERGED DURING TOW
	H	6/26/12	X	
	H	7/10/12	X	
	H	7/31/12	X	NET 90% SUBMERGED DURING TOW

3.4.2.1.1 Abundance

Zooplankton abundances collected via net tow were determined at each of six trend sites (Ilwaco Slough or Ilwaco marina (depending on water level), Secret River, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). When possible, the net was fully submerged under the water and was dragged back and forth (by hand or from a small boat) through the water for ~3-5 min. The samples were preserved in ~1.5% formalin immediately after collection. A flow meter (General Oceanics Inc., Model 2030R) was mounted to the net's bridle to provide an estimate of the volume flowing through the net. The flow meter was not available for sampling of Welch Island or Secret River sites, so the volume sampled was estimated from the product of the distance traveled during the tow and the area through which the net was towed. When the flow meter was used, the volume examined was calculated by determining the volume of water passing through the net by knowledge of the distance of water passing through the net, the velocity of the water passing through the net, and the volume of water passing through the net, as calculated from both the distance traveled and the net diameter, as described in the flow meter manual. The distance covered (in meters) was determined from:

$$Distance = \frac{Difference\ in\ counts \times Rotor\ Constant}{999999} \qquad (1)$$

where the difference in counts refers to the difference between the initial and final counts on the six-digit counter, which registers each revolution of the instrument rotor. The speed is calculated from:

$$Speed = \frac{Distance\ in\ meters \times 100}{Time\ in\ seconds} \qquad (2)$$

The volume is determined as:

$$Volume\ in\ m^3 = \frac{3.14 \times net\ diameter^2 \times Distance}{4} \qquad (3)$$

For each net tow, the volume of material collected in the cod end of the net was recorded. From this, a concentration factor was calculated, and a final estimate of the volume examined (shown in Table 1) was determined by multiplying the concentration factor by the final volume of concentrated sample examined under the microscope.

3.4.2.1.2 Taxonomy

Zooplankton species composition was determined for each net tow sample as follows. The taxa were divided into one of the following groupings: rotifers, crustaceans, annelids, ciliates, and copepods, and 'other'. Eggs of rotifers, cladocerans (crustaceans), and copepods were enumerated separately when observed.

3.4.2.2 *Macroinvertebrate Prey Availability*

3.4.2.2.1 Open water and emergent vegetation

We quantified the density, diversity and size of invertebrate prey available to juvenile Chinook salmon across twelve sites between 2010 and 2012, focusing on months during which salmon were abundant (generally when n>5; Table 9). To assess prey availability across the range of habitats sampled for fish by a beach seine (i.e., from open water to the beach), we collected invertebrates from an open water reach ("open water", OW) and from the water's margin where emergent vegetation is often present

("emergent vegetation", EV). For the open water sample, a Neuston net (250 μm mesh) was deployed from a boat for an average distance of 100 m and was held such that it sampled the top 20 cm of the water column. For the emergent vegetation sample, a 10m transect was positioned parallel to the water's edge and where the water was at least 25 cm deep. A Neuston net was pushed along the transect, through any emergent vegetation, so as to collect invertebrates within the top 20 cm of the water column. We typically collected two emergent vegetation samples and two open water samples per site per month (concurrent with two beach seine fish collections), but occasionally one or three of each were collected depending on field conditions (Table 9). Here we include analyses for samples collected in 2010, 2011 and 2012 because some of the data for 2010 and 2011 were not available for previous reports.

Table 9. The number of juvenile Chinook diet samples and total number of tow samples (in brackets) collected at each site per sampling event during 2010-2012.

	2010					2011			2012					Total
Site	April	May	June	July	August	May	June	July	Feb	March	April	May	June	diets
Secret River									15 (4)		15 (4)		14 (6)	44
Welch Island									16	14 (2)	14 (5)	30 (4)	15 (4)	89
Bradwood Slough	10 (4)	17 (4)	9 (4)	10 (4)	8									54
Jackson Island	19 (4)	15 (4)	8 (4)											42
Whites Island	16 (4)	14 (4)	18 (4)	19 (4)	13	10 (6)	25 (4)	2		13 (2)	11 (6)	11 (4)	15 (4)	167
Wallace Island West	6 (4)	14 (4)	11 (4)	11 (4)										42
Deer Island						10 (4)								10
Goat Island						13 (4)								13
Burke Island						10 (4)		2						12
Campbell Slough	12 (4)	24 (4)	18 (4)	15 (2)		22 (4)						18 (4)	15 (4)	124
Lemon Island										13 (3)	7 (4)	15 (4)	15 (2)	50
Washougal											15 (2)	18 (4)	36 (4)	69
Total diets	63	84	64	55	21	65	25	4	31	40	62	92	110	716

Taxonomists at Rhithron and Associates (Missoula, MT) processed all invertebrate samples from 2011 and 2012. For tow samples, processing included sorting, identifying, counting, measuring and, for 2012 samples only, weighing up to 500 invertebrates per sample. If a sample contained more than 500 individuals, it was subsampled and total counts were estimated based on the proportion that was processed. Invertebrates were identified to the lowest possible taxonomic level (typically species or genus), except for the Chironomidae (Diptera) that were identified to family and Oligochaetes that were identified to subclass. Individual invertebrates were measured (head capsule width and whole body length to nearest 0.1 mm). For samples collected in 2012, all individuals per sample were composited by family and life stage (e.g., Chironomidae larvae) and each composite was weighed (blotted wet weight to nearest 0.0001 g). To assess more general patterns, however, we grouped taxa by order for statistical analyses.

To explore how prey densities varied by habitat (EV vs. OW), site, year and month, we used these factors and all sample data in a stepwise regression. Because the sampling design and distribution of samples was not balanced, we also used paired mean values of prey densities from EV and OW samples collected during the same sampling event to evaluate if there were differences between the densities of prey caught in the two habitat types. In addition, we used these paired samples to determine if there was a strong correlation between prey availability in the EV and OW habitats at a site. A strong correlation may suggest connectivity between habitats within a site (e.g. the EV habitat may be a source of prey for the adjacent OW habitat) and/or that prey availability is determined by site conditions and not by smaller scale habitat conditions. A weak correlation between prey availability in EV and OW habitats may suggest there is little connectivity among adjacent habitats within a site and/or prey availability is

strongly determined by conditions at the habitat scale. Paired samples (n_{pairs}=41) consisted of mean values from the emergent vegetation samples and mean values from the open water samples collected concurrently at a site.

To address whether the local extent of emergent vegetation was correlated with the availability of invertebrate prey, we recorded the presence and estimated the percent cover of bare ground, dead vegetation, live grass, and live "other" vegetation present along the 10m transect. We did this using 5, 0.5x0.5m quadrats placed every 2 meters along each transect, and visually estimated the percent cover of each type. This was done for 71 transects that were sampled between 2010 and 2012. The mean % cover by type for each transect was used in regression analyses.

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 2012, 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).

3.4.2.2.2   Benthic Macroinvertebrates

To characterize the benthic macroinvertebrate assemblage, coring sites were selected to correspond to locations directly adjacent those where the fish community was assessed. The core samples were collected at low tide from exposed sediments. The samples were collected using a 2 inch diameter PVC pipe. One end was inserted approximately 4 inches into the sediment of the channel at or near low tide, and a rubber stopper was placed on the other end of the pipe creating a vacuum suction used to contain

the sample in the pipe while it was removed from the substrate. The samples were rinsed through a 500 µm mesh sieve using deionized water and preserved in individual plastic jars with 95% Ethanol. Rose bengal, an inert stain, was applied to facilitate sorting invertebrates from other debris in the sample.

3.4.3 Salmon Diet

3.4.3.1 *Chinook salmon diets and prey selectivity*

Chinook salmon were usually killed within an hour of collection. If fish were not processed immediately, they were kept on ice until stomachs were extracted later that day. Whole stomachs were preserved in ethanol and, for all 2011-2012 samples, were shipped to Rhithron and Associates for analysis. As with the invertebrate tow samples, prey items were identified, counted, measured and weighed. All identified invertebrates were composited by family and life stage for weights, and unidentifiable pieces of invertebrates were composited and weighed as "unknown" for each stomach.

Ivlev's prey electivity values (Lechowicz 1982) were calculated for the most abundant taxa in both the diets and the tows to determine if juvenile Chinook salmon preferred or avoided particular prey taxa based on their relative abundance in the environment. The index compares the mean proportion of each invertebrate order in the diets and the tow samples collected during each sampling event. Ivlev's electivity index is:

Ivlev's index = (% of order *i* in diet – % of order *i* in tows)/(% of order *i* in diet + % of order *i* in tows)

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

3.4.3.2 *Isotope Ratios*

The ratios of carbon and nitrogen stable isotopes in tissues of consumers reflect the stable isotope ratios of their food sources (Neill and Cornwell 1992; France 1995), and therefore, can be useful to determine major food sources, provided that the food sources have distinct isotopic ratios. Stable isotope analysis of carbon and nitrogen is being used to determine whether algae or wetland plants are more important contributors to the food web supporting juvenile salmonids.

Most carbon atoms have 12 neutrons (^{12}C), but approximately 1% of carbon atoms have 13 neutrons (^{13}C). Likewise, most nitrogen atoms have 14 neutrons (^{14}N), while some have 15 neutrons (^{15}N). Lighter isotopes are metabolized preferentially to heavier isotopes, so consumers at higher trophic levels

(higher in the food web) become enriched in the heavier isotopes. Therefore, the ratios of heavy to light isotopes ($^{15}N/^{14}N$ and $^{13}C/^{12}C$) in the tissues of food sources, plus a small compensation for the metabolic loss of light isotopes ("trophic fractionation"), are reflected in the tissues of consumers. Typically, with each step up in the food chain (i.e., from a plant to an herbivore or an herbivore to a carnivore), the $^{15}N/^{14}N$ ratio increases by 2.2 to 3.4 parts per thousand ($^0/_{00}$), so stable isotope analysis of nitrogen is useful in determining trophic position. The $^{13}C/^{12}C$ ratio usually changes by less than 1 $^0/_{00}$, making stable isotope analysis of carbon useful for determining inputs of primary producers when the different primary producers analyzed have distinct stable isotope 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. Muscle samples were collected from individual juvenile salmonids for stable isotope analysis. Isotopic signatures of more metabolically active tissues such as liver, mucus, or blood turn over more quickly than those of muscle, otoliths, or scales, so they are good media with which to examine relatively recent dietary sources (Phillips and Eldridge 2006; Church et al. 2009; Buchheister and Latour 2010). In 2012, epidermal mucus was collected from a subset of juvenile salmonids from which muscle samples were also collected to test the suitability of mucus for this analysis.

3.4.3.2.1 Algae
Samples of particulate organic matter (POM) and periphyton collected as described above for phytoplankton and periphyton abundance were filtered onto 25 millimeter (mm) glass-fiber GF/F filters, freeze dried, and analyzed for stable carbon and nitrogen isotopes.

3.4.3.2.2 Plants
Samples of dominant emergent vegetation species were collected from representative areas within each site. Plant samples were rinsed at least five times in deionized water to remove external material, such as invertebrates and periphyton, and were kept frozen for later processing.

3.4.3.2.3 Insects and Juvenile Salmonids
Juvenile salmonids were collected by NOAA Fisheries staff using a beach seine. In 2011, skinned muscle tissue samples were collected in the field and frozen for later analysis. In 2012, whole bodies were wrapped in foil and kept on ice until being stored frozen. Aquatic insects were collected by USGS staff in open water and in emergent vegetation at the water's margin using opportunistic sampling. The aquatic midge *Chironomidae* and the amphipod *Corophium salmonis* were selected because they have been found to be preferred food sources for juvenile salmonids in the lower Columbia River (Sagar et al. 2013; Maier and Simenstad 2009). Most insect samples were found attached to submerged portions of vegetation. Those insects were collected by rinsing the exterior of the vegetation with deionized water and manually removing the insects from the rinse water using forceps. Insect samples were then rinsed with deionized water to remove algae or other particulate matter. Salmonid and aquatic insect samples were frozen for later processing. Epidermal mucus was collected from individual juvenile salmonids as described by Church et al. (2009) and composited in order to meet the minimum sample mass requirements for the analysis.

Frozen salmonid tissue, insects, and plant material were freeze-dried using a lyophilizer. Freeze-dried plants of the same species from the same sample date were composited and ground using a clean coffee grinder. Freeze-dried insect bodies of the same taxa were composited, ground using a clean glass mortar and pestle, and subsampled when enough material was present. Otherwise, whole bodies of all individuals of the same taxa from the same site were included composited into a single sample. Skinned muscle tissue was cut from juvenile salmonid bodies after freeze-drying. Skinned muscle tissue samples

from individual juvenile salmonids were analyzed separately; muscle tissue samples were not composited for analysis.

3.5 Fish Use

3.5.1 Fish Community

In 2012, NOAA Fisheries monitored prey availability and habitat use by juvenile Chinook salmon and other fishes at three new tidal freshwater sites in Reach G: Government/Lemon Island, Washougal, and Reed Island. Additionally, we re-sampled fish and prey at the Franz Lake site in Reach H (sampled in 2008, 2009, 2010, and 2011), Campbell Slough in Reach F (sampled from 2007-2011), Whites Island site in Reach C (sampled in 2009, 2010, and 2011), and Ilwaco in Reach A (sampled in 2011), in order to examine year-to-year trends in fish use at the sites. In 2012 two new trend sites were added in reach B, Secret River and Welch Island. Coordinates of the sites are shown in Table 10.

Table 10. Coordinates of the sites sampled in 2012.

Site Name	Latitude	Longitude
Ilwaco Slough	46°18.035'N	124° 2.784'W
Secret River	45° 9.561'N	122° 20.408'W
Welsh Island	45° 47.032'N	122° 45.291'W
Whites Island	45° 9.561'N	122° 20.408'W
Campbell Slough	45° 47.032'N	122° 45.291'W
Government/Lemon Island	45° 47.032'N	122° 45.291'W
Washougal River	45° 58.037'N	122°39.58'W
Reed Island	45°55.529'N	122°29.74'W
Franz Lake	45° 36.035'N	122° 6.184'W

Fish use of the sites was assessed by analysis of catch data. Fish were collected from February 2012 through December 2012; however this report will present catch data through September 2012 only. 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 weighed (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, as conditions allowed, the coordinates of the sampling locations, the time of sampling, water temperature, weather, habitat conditions, tide conditions, salinity, 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), including polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethanes (DDTs), polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and various organochlorine pesticides; 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 POPs, including PAH, DDTs, PCBs, PBDEs, and various organochlorine pesticides.

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.

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

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

Where

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.

3.5.2 Salmon Metrics

3.5.2.1 *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 Lower Columbia River Estuary, 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 Lower Columbia River 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.

3.5.2.2 *Lipid Determination and Condition Factor*

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).

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

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

3.5.2.3 *Otoliths (Growth Rates)*

Otoliths of juvenile Chinook salmon were extracted and 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 (evolution MP 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.

3.5.2.4 *Chemical Contaminants in Chinook salmon*

3.5.2.4.1 Persistent organic pollutants in bodies

Composite body samples, with stomach contents removed, 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; organochlorine (OC) pesticides including DDTs, hexachlorocyclohexanes (HCHs), chlordanes, aldrin, dieldrin, mirex, and endosulfans; and low (2-3 ring) and high (4-6 ring) molecular weight aromatic hydrocarbons 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. 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.

3.5.2.4.2   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).

4 Results

4.1 Mainstem conditions

The instrument Platform at River Mile 53 has operated since July 2009 in collaboration with the USGS Water Science Center and Wet Labs Inc. Data from this station has been used in three student thesis projects and a number of publications are in preparation or in press that discuss the data or use it for supporting information. Here we summarize the data set to provide context for the measurements in 2012. In the following sections we present results for 2012 and for RM-122 since its installation in September 2012.

4.1.1 Biogeochemical observations at RM-53 2009-2011

Biogeochemical measurements from the platform at RM-53 are shown in Figure 9 for the period up until spring 2011 when the sensor system was removed for a major overhaul. This time period encapsulates one complete calendar year (2010) and one complete water year (2010-2011) as well as two partial years 2009 and 2011. There is data from two summer periods (2009 and 2010) and two winter periods (2010 and 2011) as well as three freshet periods (2009, 2010, and 2011). A summary of the parameters is provided here.
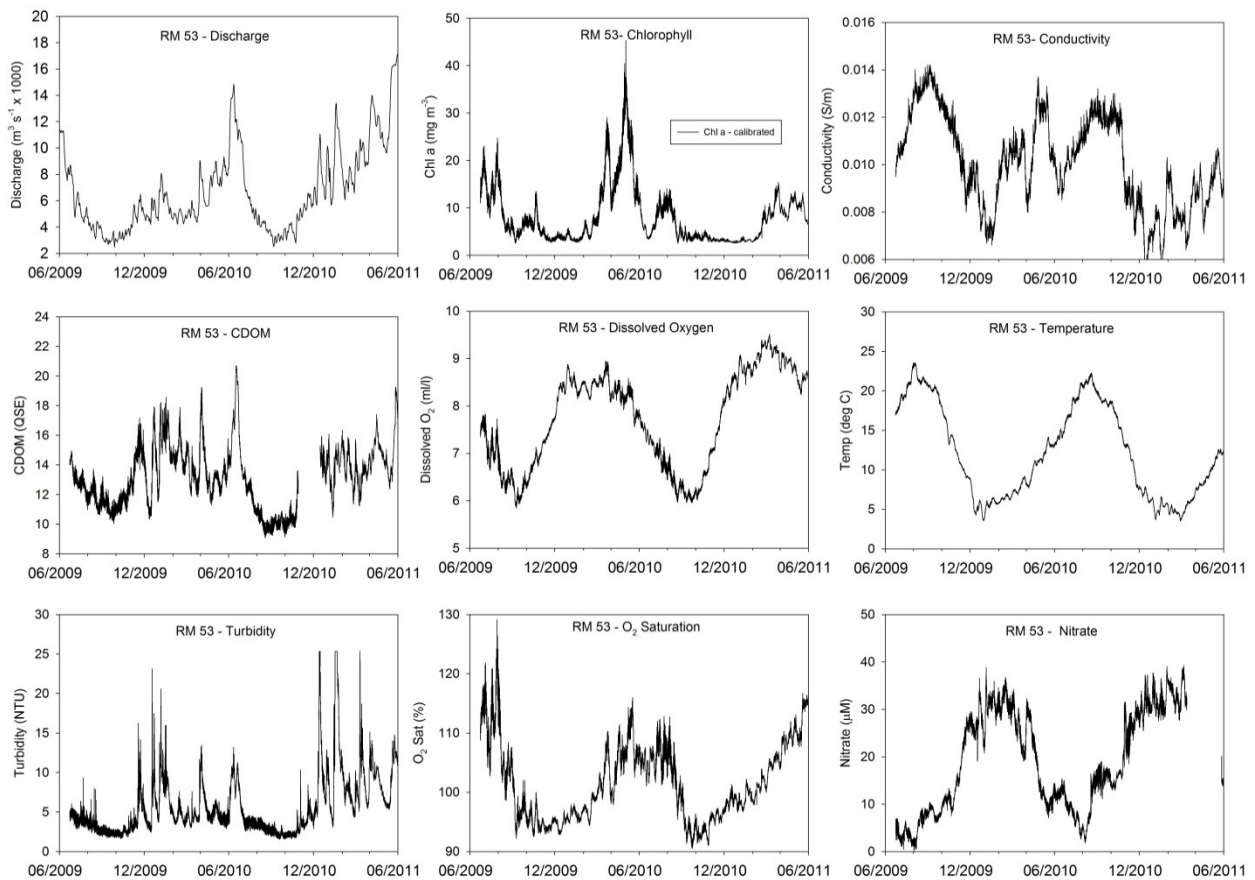
Figure 9. Time series of biogeochemical parameters measured hourly at RM-53 during the two year period between July 2009 - June 2011. Also shown is river discharge at this location.

Columbia River Discharge

Highest discharge in all years occurred during the freshet with flows above 14000 m^3 s^{-1} recorded each year, although significant flows were measured at other times, especially during winter. Lowest flows occurred in summer and dropped below 4000 m3 s^{-1} in late summer/early fall in both 2009 and 2010. During 2011, the freshet extended well into June as a result of a large snowpack.

Temperature

Water temperature followed expected seasonal patterns with lowest temperatures in winter and highest in summer. Since this site is completely mixed throughout the water column, temperature measurements represent the average Columbia River conditions, and thus warmer (in summer) and cooler (in winter) may occur in the shallower fringes of the mainstem. A summary of warm temperatures for each year are given in Table 11. During 2009 the number of days above 19 °C was significantly higher than 2010 or 2012 (we did not have a complete dataset for 2011).

Table 11. Number of days with temperatures > 19°C measured in the Columbia River at RM-53

	2009	2010	2012
Range 19-21 °C	70	49	53
Above > 21° C	11	2	2
Total > 19°C	82	51	55

Dissolved Oxygen

Dissolved oxygen follows the seasonal pattern in temperature, primarily due to the relationship between oxygen solubility and water temperature. Other important processes at work in the Columbia River are air infusion by dam spillage, air-water gas exchange, and biological production and consumption processes. These processes are more easily detected by considering O_2 saturation (also plotted on Figure 9), which is a measure of the relative equilibrium of oxygen between water and the atmosphere. A general pattern emerges where oxygen is higher than equilibrium (> 100% saturation) between approximately March and September as a result of increased dam spillage and primary production by plants and algae. Oxygen is under saturated during winter months when dam spillage is lower and aerobic respiration dominates the biological signal.

Chlorophyll

Chlorophyll a concentration represents phytoplankton biomass in the river. Changes in biomass can result from increased rates of phytoplankton growth or decreased rates of phytoplankton mortality, therefore changes in biomass can have multiple interpretations depending on food web interactions. During 2009-2011 a few patterns in chlorophyll a are evident. First, phytoplankton biomass increases during spring before the onset of the freshet. The freshet coincided with low biomass, similar to winter levels, and then rebounded again during summer months. This likely reflects a combination of factors that limit growth of phytoplankton in the river, most notably low light levels as a result of seasonal light availability and increased turbidity associated with high runoff. During summer, the biomass tends to fluctuate between 10-20 mg m^{-3}. Inter-annual variability is also evident, for example there was a fall biomass increase in 2009, but not in 2010.

Turbidity

Turbidity was strongly correlated to discharge. Episodic increases in turbidity coincided with increases in discharge during winter months, and likely reflect increased sediment input from the watershed. In addition to these events, there was a strong correlation between the freshet and turbidity in 2010, and

an overall increase in turbidity throughout winter and spring 2011 that corresponded to the large discharge that occurred in 2011 during the spring freshet.

CDOM
Fluorescence measured as CDOM originates primarily from dissolved organic matter. The variability in CDOM observed in winter is similar to the turbidity pattern - in that peaks in CDOM follow discharge associated with episodic discharge increases. In addition, CDOM increased along with turbidity during the spring freshet of 2010 and remained relatively high during the large discharge in 2011.

Nitrate
Nitrate concentration has a seasonal pattern of high concentrations (> 30 µM) in winter and low concentrations (< 5 µM) in summer. Nitrate was never observed to be completely depleted. Nitrate concentrations are a result of inputs and thus are associated with discharge, especially during winter.

Fluxes at RM-53 2009-2011
Fluxes of nitrate and phytoplankton particulate organic carbon were determined for the time period 2009-2011 by calculating the average daily concentration from the hourly measurements and multiplying by the daily discharge. Molar concentrations of nitrate were converted to kilograms using the molar mass of N (14).

Figure 10 illustrates the nitrate flux calculated from the concentrations shown in Figure 9 and the discharge at RM-53. Nitrate Flux is highest in winter (Figure 10A) and was typically higher in winter 2010-2011 than in winter 2009-2010. The total nitrate transported at RM-53 for the 12 month period between October 2009 and October 2010 was 47,000 tonnes of nitrogen as nitrate. There is not a complete record for nitrate flux during subsequent years, but the flux was likely higher in 2011 given that concentrations and discharge were both higher throughout the spring of that year. For example, by March 12, 2011 the nitrogen flux was 41,000 tonnes compared to 25,000 tonnes at the same time in 2010.   When compared with discharge (Figure 10B) there are strong correlations between individual episodic discharge events and peaks in nitrate flux. There is also a peak in nitrate flux associated with the 2010 spring freshet.

When nitrate flux is compared to the Bonneville Flow only (Figure 10C) the correlation between nitrate flux and discharge is less noticeable, although there remains a strong correlation during the 2010 freshet and during a large discharge event in late January 2011. However, when the Bonneville discharge is removed from the total discharge (leaving only the discharge associated with tributaries downstream of Bonneville) the correlations with winter episodic events are again obvious.

Phytoplankton particulate organic carbon (PPOC) flux was computed from the chlorophyll a biomass and river discharge (Figure 19). The highest flux occurred during the spring bloom in 2010 with values exceeding 500 tonnes C per day. There was approximately 34,000 tonnes of phytoplankton POC transported at RM-53 during the 12 month period between October 2009 and October 2010.

PPOC was inversely correlated with discharge (Figure 19 B-D), but the largest fluxes are likely associated with favorable light and nutrient conditions rather than specifically tied to runoff.
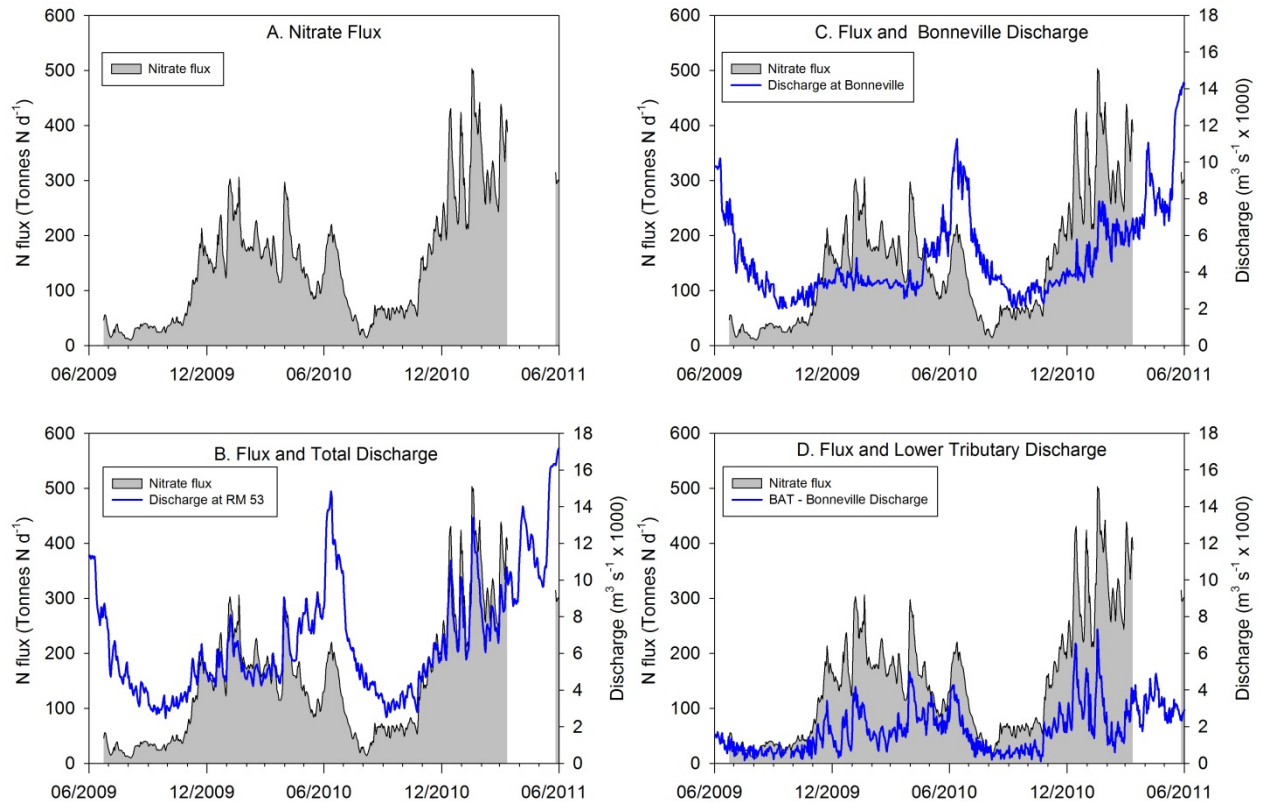
Figure 10. Daily Nitrate Flux calculated at RM-53 for the two year period July 2009- June 2011. A) Nitrate flux in tonnes (i.e. metric ton or 1000 kg) nitrogen per day. B) Comparison of nitrate flux with river discharge at RM-53. C) Comparison of nitrate flux with river discharge at Bonneville Dam. D) Comparison of nitrate flux with river discharge associated with tributaries of the lower Columbia River, as determined from the difference in discharge between RM-53 and Bonneville Dam.
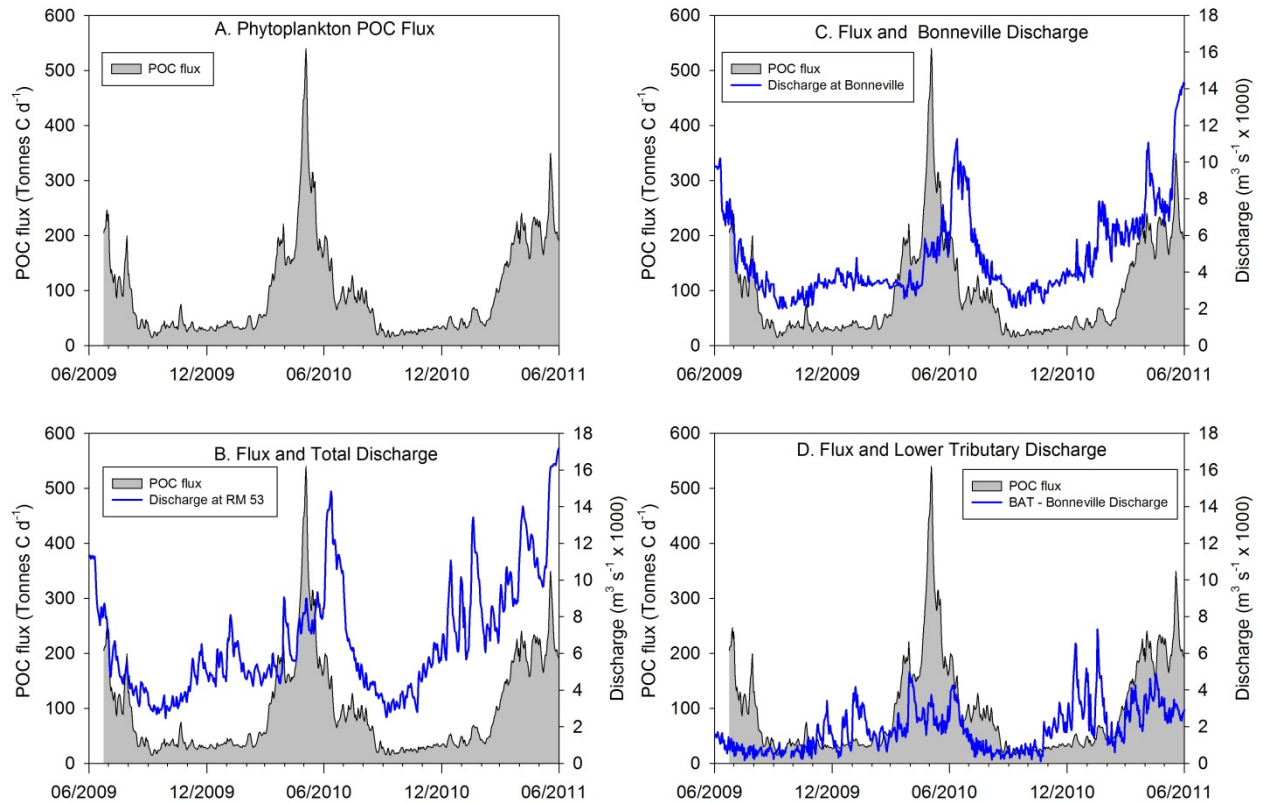
Figure 11. Daily phytoplankton particulate organic carbon flux calculated at RM-53 for the two year period July 2009- June 2011. A) POC flux in tonnes (i.e. metric ton or 1000 kg) carbon per day. B) Comparison of POC flux with river discharge at RM-53. C) Comparison of POC flux with river discharge at Bonneville Dam. D) Comparison of POC flux with river discharge associated with tributaries of the lower Columbia River, as determined from the difference in discharge between RM-53 and Bonneville Dam.

Net Ecosystem Metabolism

Primary production, aerobic respiration, and net ecosystem metabolism estimates were calculated for the 2009-2011 period (Figure 12). Gross primary production and respiration were strongly correlated and were highest when phytoplankton biomass was highest. During winter and the spring freshet, both parameters were lower. The difference between the two parameters (i.e. NEM) followed a seasonal pattern, such that in winter there was a net removal of oxygen (and therefore carbon), suggesting that heterotrophic processes dominated the overall ecosystem metabolism. In spring and summer (except during the freshet) there was a net production of oxygen indicating that autotrophic processes dominated. Therefore, the Columbia River acts as an organic carbon sink in winter, but during summer there is a net production of carbon. In addition, the high rates of primary production in summer of 2010 compared to spring suggest that phytoplankton biomass alone does not predict growth rates, since phytoplankton biomass was typically higher in spring of that year. This can result from either: 1) higher grazing of phytoplankton in summer or 2) higher rates of oxygen production into the water column from attached vegetation not related to chlorophyll a biomass in the water column.
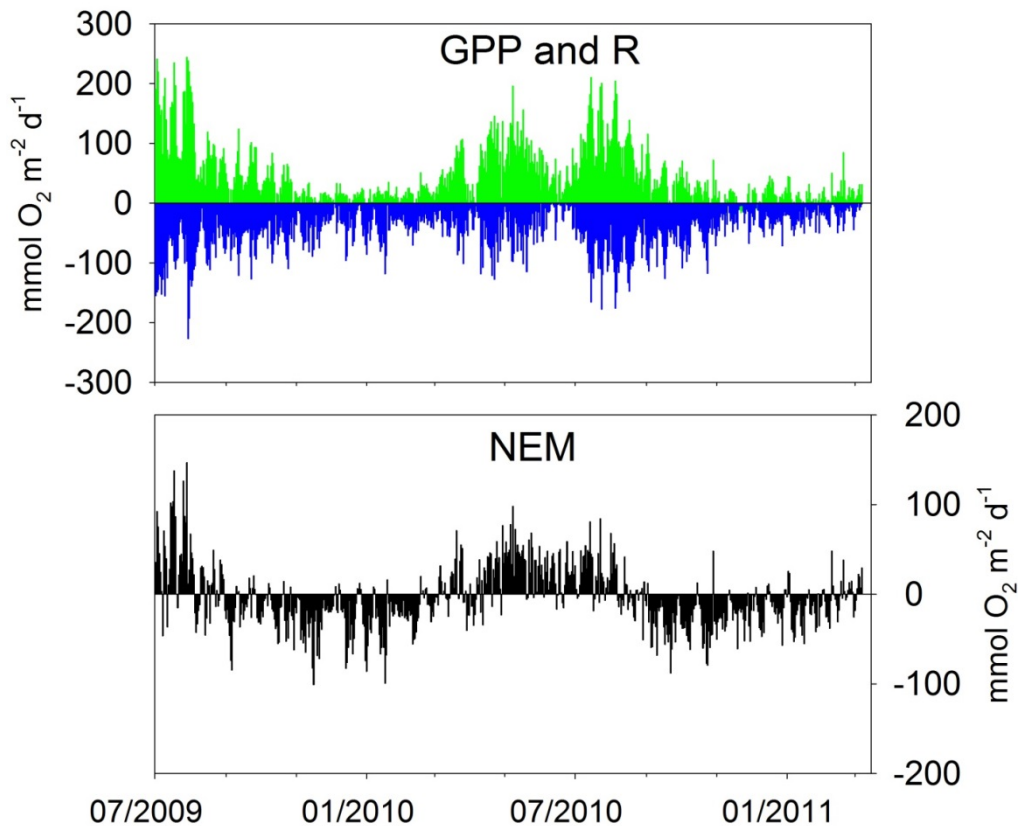
Figure 12. Daily calculated Gross Primary Productivity (GPP), aerobic cellular respiration (R), and net ecosystem metabolism (NEM) calculated at RM-53 for the two year period July 2009- June 2011. Parameters were determined from hourly oxygen measurements, wind speed, and water velocity, as described in the text.

4.1.2 Biogeochemical observations at RM-53 2012

Biogeochemical measurements from the platform at RM-53 are shown in Figure 13 for 2012 when the sensor system was installed after a major overhaul. This time period includes the winter rainy season, the spring freshet, and the summer dry period. The data set overlap with the RM-122 data set starting in September 2012. The following is a description of the results in 2012 with an emphasis on differences or similarities to previous years described in section 4.1.1.
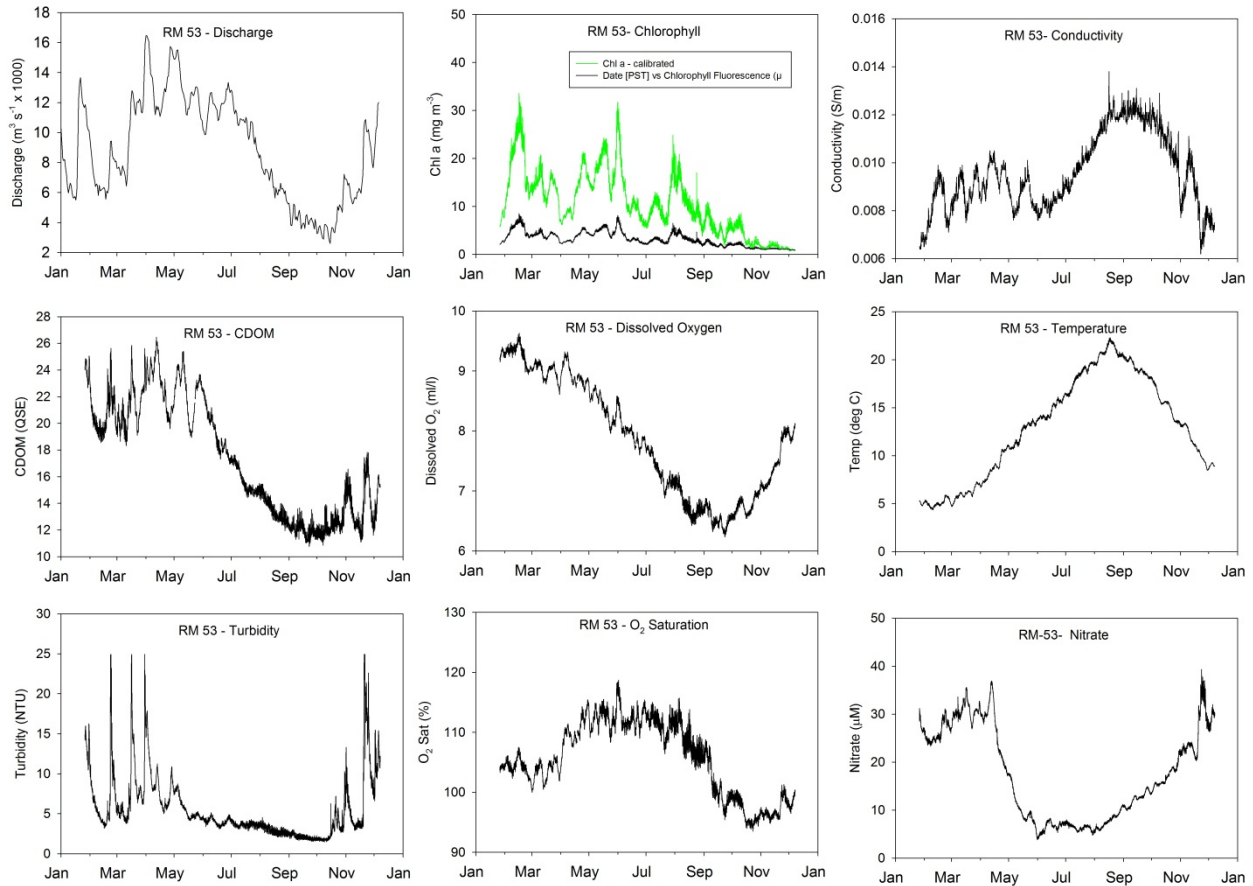
Figure 13. Time series of biogeochemical parameters measured hourly at RM-53 during 2012. Also shown is river discharge at this location (from Figure 7). Chlorophyll a measurements are presented both for the raw signal and the corrected signal based on laboratory calibrations.

Columbia River Discharge

Discharge in 2012 was characterized by periodic runoff events during winter followed by the spring freshet. There were clear differences between Bonneville and RM-53 discharge in winter (Figure 8) that are attributed to episodic storm events and increases in lower Columbia River tributary runoff.

Temperature

Water temperature displayed a seasonal pattern associated with climate, and had warm water characteristics more similar to 2010 than to 2009 (Table 11).

Dissolved oxygen

Dissolved oxygen followed the expected patterns associated with temperature dependent gas solubility, with lowest concentrations associated with the warmest water. Oxygen saturation was generally above 100% over the entire winter, spring and summer period, only decreasing below 100% after September, similar to previous years.

Chlorophyll

Chlorophyll a biomass was relatively high throughout winter 2012. High biomass was present in February onward, but significant variability occurred throughout 2012 with no obvious "spring bloom" as was seen in 2010.

Turbidity

Turbidity displayed high variability during winter 2012 and correlated to discharge events as was seen in previous years. However, there was no obvious high turbidity signal associated with the spring freshet.

CDOM

CDOM displayed a winter pattern similar to turbidity and therefore likely associated with runoff.

Nitrate

Nitrate concentration displayed similar patterns to previous years, with higher values associated with runoff in winter. There was no obvious increase in nitrate associated with the spring freshet, in contrast to 2010 when the freshet was shorter and was associated with an obvious peak in nitrate. Nitrate was not depleted at any time during 2012, with lowest values in summer similar to previous years (approx. 5 μM)

Fluxes

Nitrate flux during 2012 is shown in Figure 14. High fluxes in winter and low fluxes in summer demonstrate a pattern similar to previous years, although distinct differences exist including the absence of a noticeable freshet signal. When compared to discharge (Figure 14B) it is evident that flux is not directly related to discharge. Nitrate flux began to decrease in early April, and reached a steady and relatively low rate by early June. Discharge was still increasing in early April and remained relatively high until early August. This pattern reflects the decrease in nitrate concentration associated with the source waters of the lower Columbia River. This observation is supported by comparing the flow at Bonneville (Figure 10C) and the flow from the lower Columbia river tributaries (Figure 10d), which show strong correlations between increases in winter nitrate fluxes and increases in tributary episodic discharges (Figure 10D). In contrast, the large discharge associated with the spring freshet (Figure 10C) is not well correlated with the patterns in nitrate flux.

Phytoplankton POC fluxes were highest between April and June, and loosely followed the pattern for total discharge indicating that phytoplankton were abundant between Feb - Aug 2012 despite the large flows associated with the spring freshet. The inverse correlation between PPOC flux and discharge in winter is similar to previous observations. When compared to other parameters that correlate with these discharge events (see Figure 16) it is evident that high discharge, high turbidity events are directly inversely correlated to phytoplankton biomass fluxes. After approximately April, PPOC fluxes increased from approximately 200 tonnes C d^{-1} to 400 tonnes C d^{-1} through mid-June. During summer, significant PPOC flux continue through September, after which biomass and flux reach low values characteristic of fall and winter in previous years.
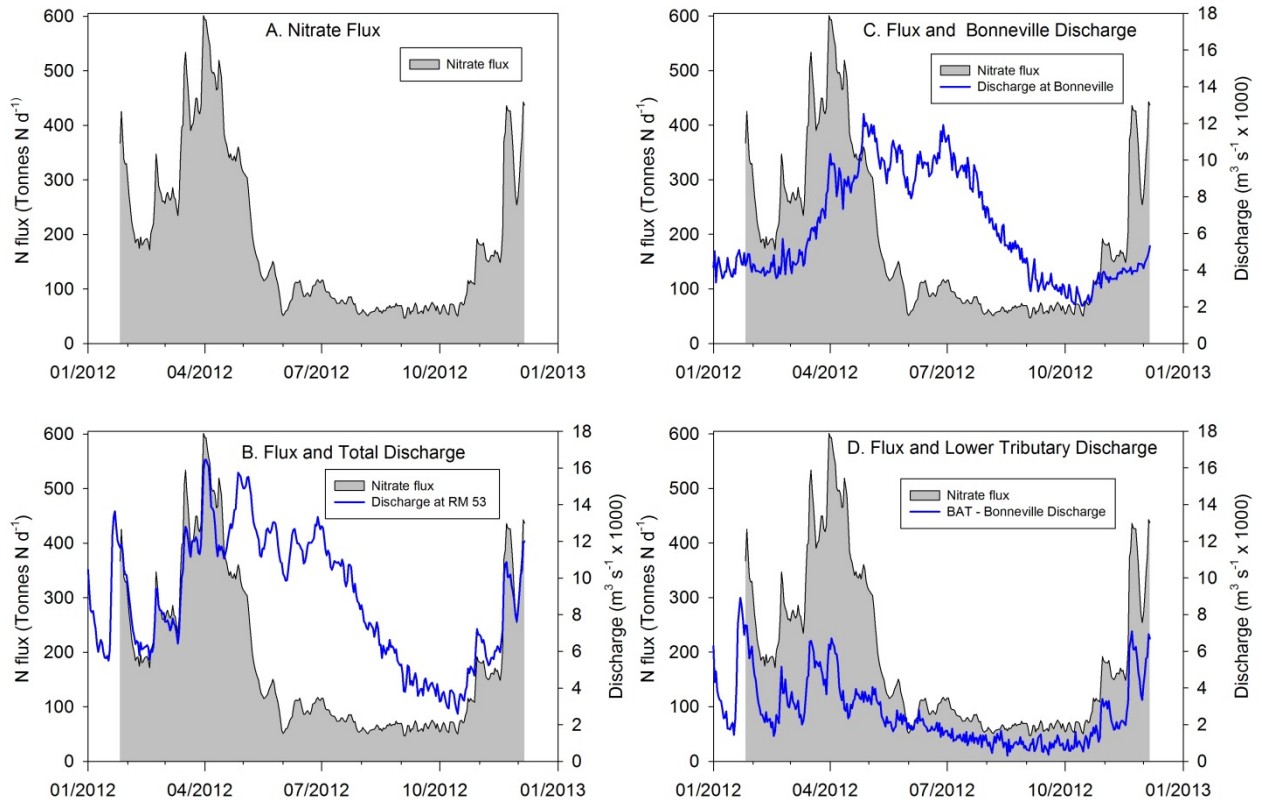
Figure 14. Daily Nitrate Flux calculated at RM-53 during 2012. A) Nitrate flux in tonnes (i.e. metric ton or 1000 kg) nitrogen per day. B) Comparison of nitrate flux with river discharge at RM-53. C) Comparison of nitrate flux with river discharge at Bonneville Dam. D) Comparison of nitrate flux with river discharge associated with tributaries of the lower Columbia River, as inferred from the difference in discharge between RM-53 and Bonneville Dam.
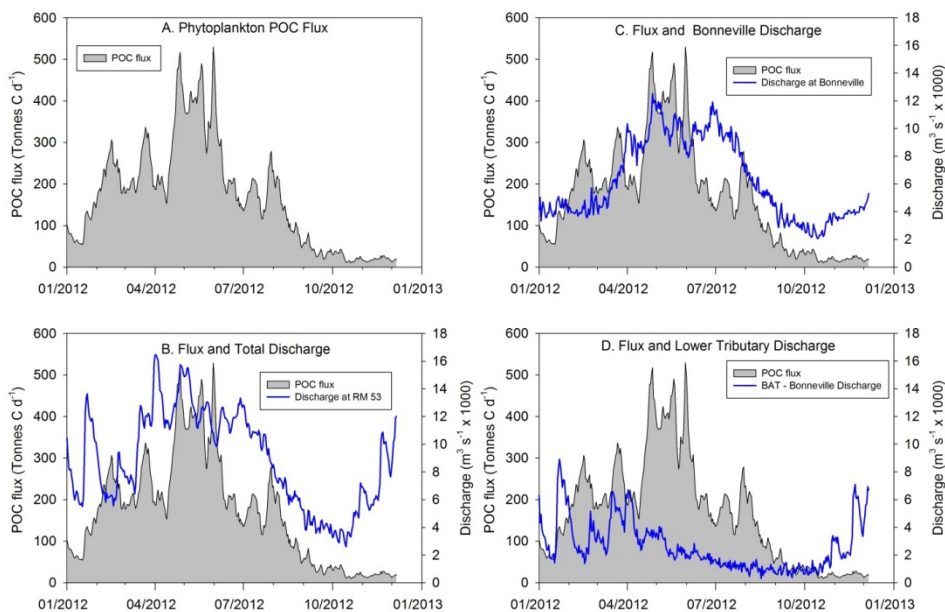
Figure 15. Daily phytoplankton particulate organic carbon flux calculated at RM-53 during 2012. A) POC flux in tonnes (i.e. metric ton or 1000 kg) carbon per day. B) Comparison of POC flux with river discharge at

RM-53. C) Comparison of POC flux with river discharge at Bonneville Dam. D) Comparison of POC flux with river discharge associated with tributaries of the lower Columbia River, as inferred from the difference in discharge between RM-53 and Bonneville Dam.
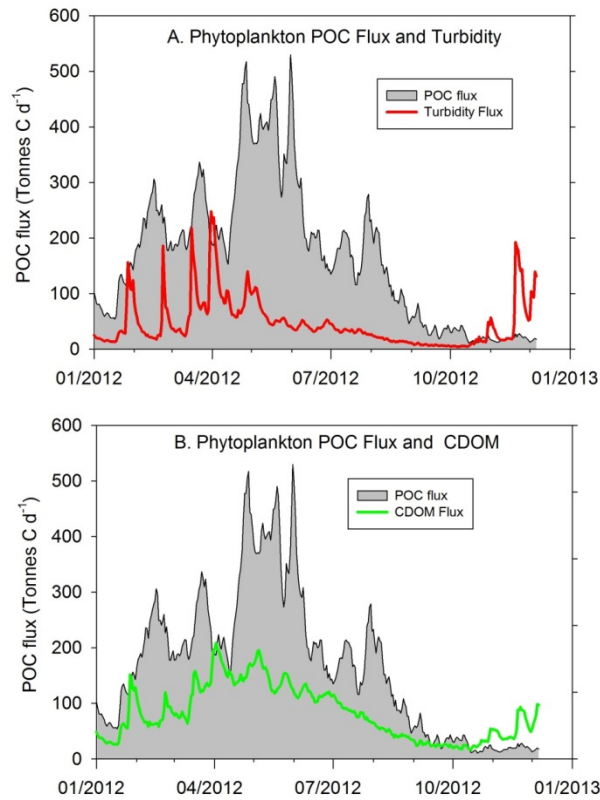
Figure 16. Comparison of daily phytoplankton particulate organic carbon flux calculated at RM-53 during 2012 (from Figure 15) with: A) relative turbidity flux, and B) relative CDOM flux.

4.1.3 Comparison of RM-53 and RM-122 in fall 2012

The installation of RM-122 in late summer 2012 allows for quantitative comparisons between RM-53 and RM-122 show the data for both sites in a format similar to Figure 9 and Figure 13. The time period captures the decline of phytoplankton biomass and the onset of the winter rainy season and increases in river discharge.
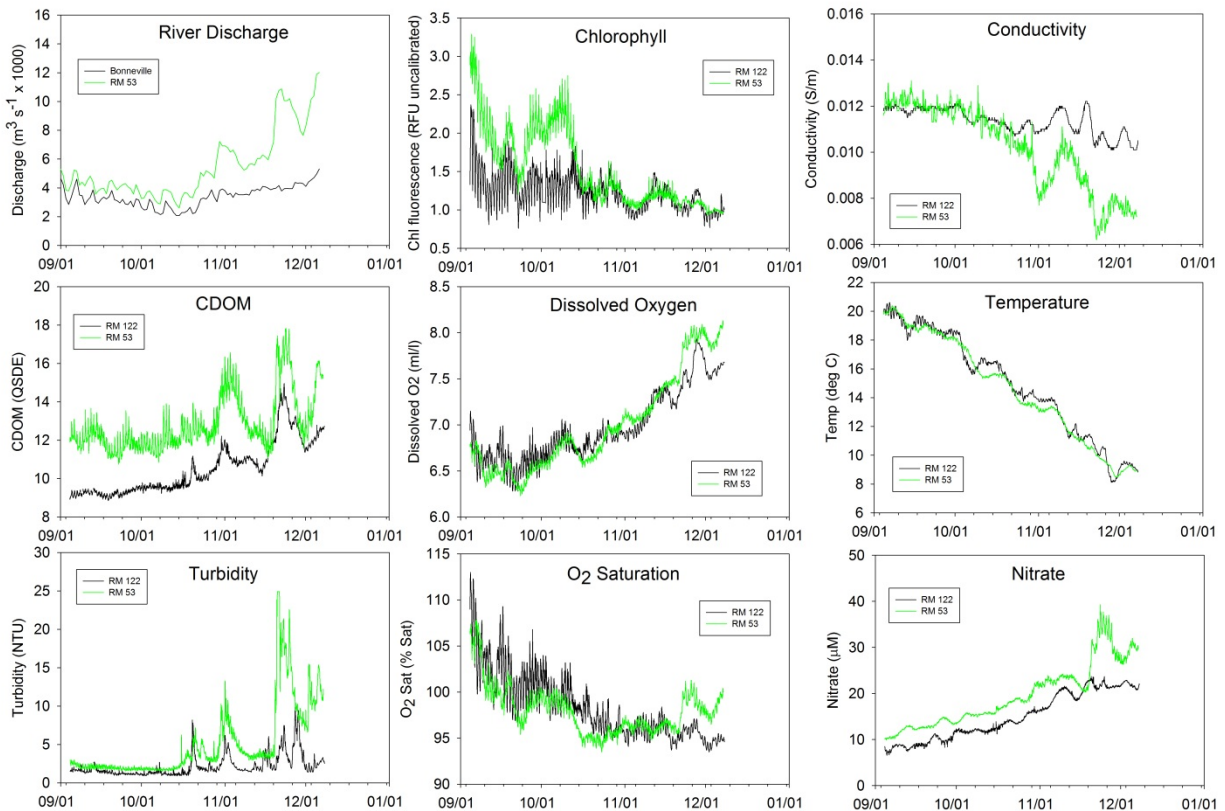
Figure 17. Comparison of time series of biogeochemical parameters measured hourly at RM-53 and RM-122 between September – December 2012. Also shown is river discharge at RM-53 and at Bonneville Dam. For comparison purposes, Chlorophyll a measurements are presented as the raw fluorescence signal without corrections imposed.

Columbia River Discharge

Discharge was similar at both sites in September, indicating that the input from tributaries in the lower Columbia River was low relative to the flow at Bonneville Dam. Differences in discharge began in late October and large increases in discharge at RM-53 occurred throughout the fall. The increase at Bonneville was only slightly higher and did not display the episodic patterns seen at RM-53 during the same periods.

Temperature

The decrease in temperature associated with seasonal changes was observed at both sites and temperatures matched closely. Small differences in the pattern of temperature decreases are noticeable but in general temperature is very similar at both sites despite differences in discharge later in the fall.

Dissolved oxygen

Dissolved oxygen was similar at both sites throughout the measurement period, and reflected temperature changes in the river. Fluctuations in the oxygen measurement at high temporal resolution (hourly-daily) were more evident at RM-122.

Chlorophyll
Chlorophyll biomass has not been corrected for instrument bias at RM-122 and therefore the raw fluorescence is shown for site comparison. RM-53 had higher chlorophyll biomass than RM-122 during early September and October. Both sites decreased in chlorophyll concentration following the onset of higher discharge and the chlorophyll fluorescence converged at the two sites to a low values associated with low biomass.

Turbidity
Turbidity was similar between sites during September and October and increased during the onset of increased discharge. Turbidity increased more at RM-53 than at RM-122, and both sites showed a pattern of variability that strongly correlated with discharge.

CDOM
CDOM was higher at RM-53 than RM-122 during all periods. CDOM increased at RM-53 during major discharge events in both early and late November that were reflected at RM-122 but with a muted response.

Nitrate
Nitrate was higher at RM-53 than RM-122 during all periods. Both sites showed a similar increase with time and a muted response relative to changes in discharge. However, the second large discharge event in late November coincided with a large increase in nitrate that was not detected at RM-122.

Fluxes
Nitrate flux was higher at River Mile 53 (Figure 18) during all periods. This reflects a higher concentration during September and October, and both a high concentration and higher river discharge in November. The two sites allow for a quantitative comparison of the relative contribution of the tributaries to the nitrate flux, since the total flux at RM-53 can be compared to the total flux at RM-122. These results are shown in Figure 13B. The % discharge associated with tributaries of the Lower Columbia River are calculated from the difference between RM-53 and Bonneville Dam, and the % Nitrate Flux is calculated from the fluxes in Figure 18A. The results show that tributaries accounted for approximately 20% of discharge and 40 % of nitrate flux in early September and these values increased to as high as 61% of discharge and 76% of the nitrate flux during the late November high discharge event at RM-53.
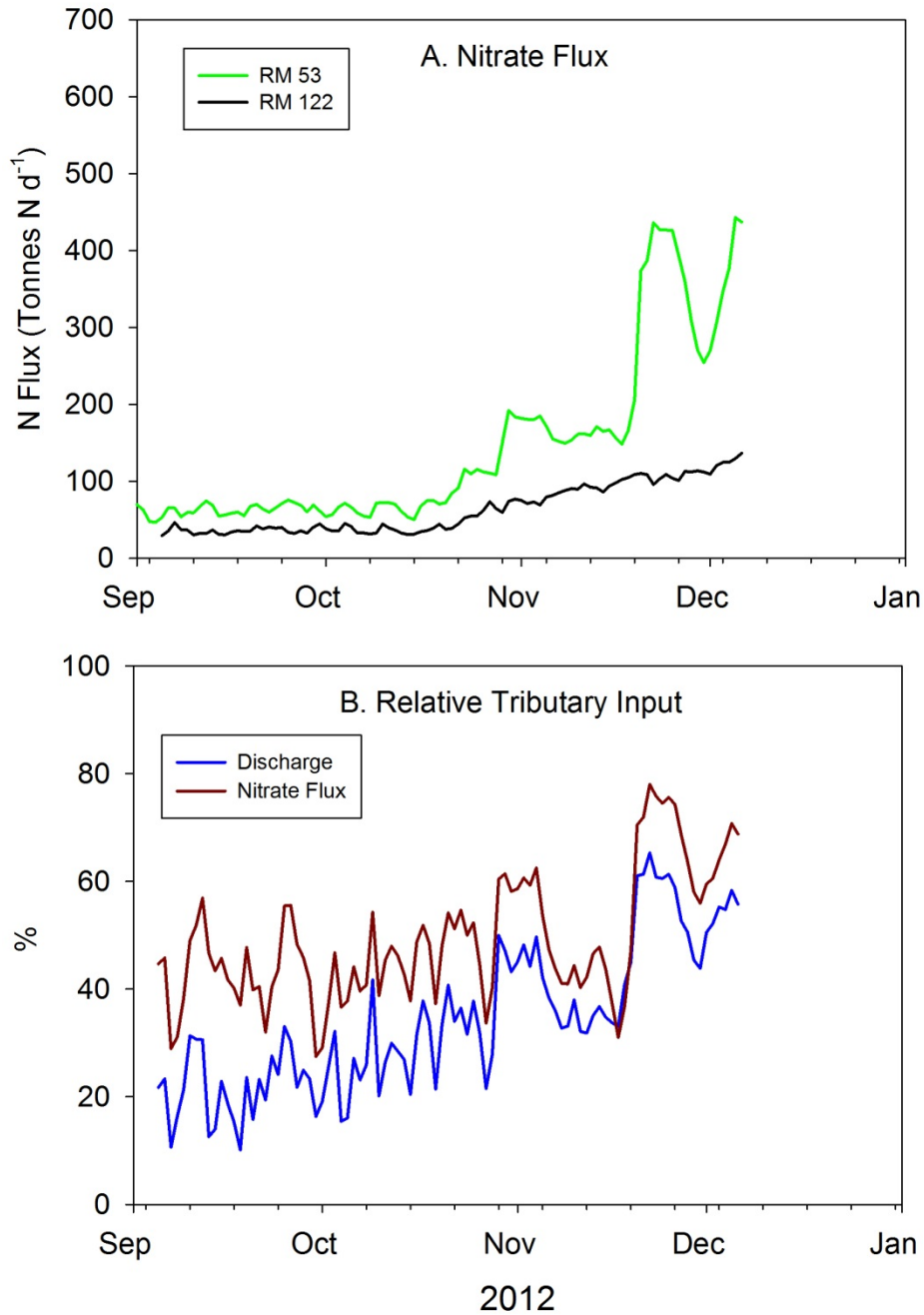
Figure 18. A. Comparison of daily nitrate flux in tonnes per day (i.e. metric ton or 1000 kg) for RM-53 and RM-122 between September – December 2012. B. The relative input by lower Columbia river tributaries (i.e. % of total input at RM-53 from tributaries) as determined by the difference between Bonneville Dam and RM-53 (for discharge) and RM-122 and RM-53 (for nitrate flux).

4.2 Abiotic site conditions

4.2.1 Continuous water-quality

Franz Lake Slough

The water temperature at Franz Lake Slough increased steadily during the 2012 monitoring period, exceeding the weekly maximum temperature standard of 17.5 degrees Celsius (°C) after early July (Figure 19). Dissolved oxygen concentrations decreased over the monitoring period, with the range in daily fluctuations increasing over the season, except during the high water in June (Figure 20). The daily minimum dissolved oxygen concentration briefly dipped below the standard of 8 milligrams per liter (mg/L) in late May, and then was less than the standard more regularly starting in early July. The pH of Franz Lake Slough fluctuated in a similar pattern as dissolved oxygen, although the seasonal decline in pH was less sharp (Figure 21). The maximum pH standard of 8.5 was only exceeded briefly in mid-April and the minimum standard of 6.5 was never violated. Specific conductance ranged from approximately 50-170 microsiemens per centimeter (uS/cm), with the largest fluctuations in April (Figure 22).
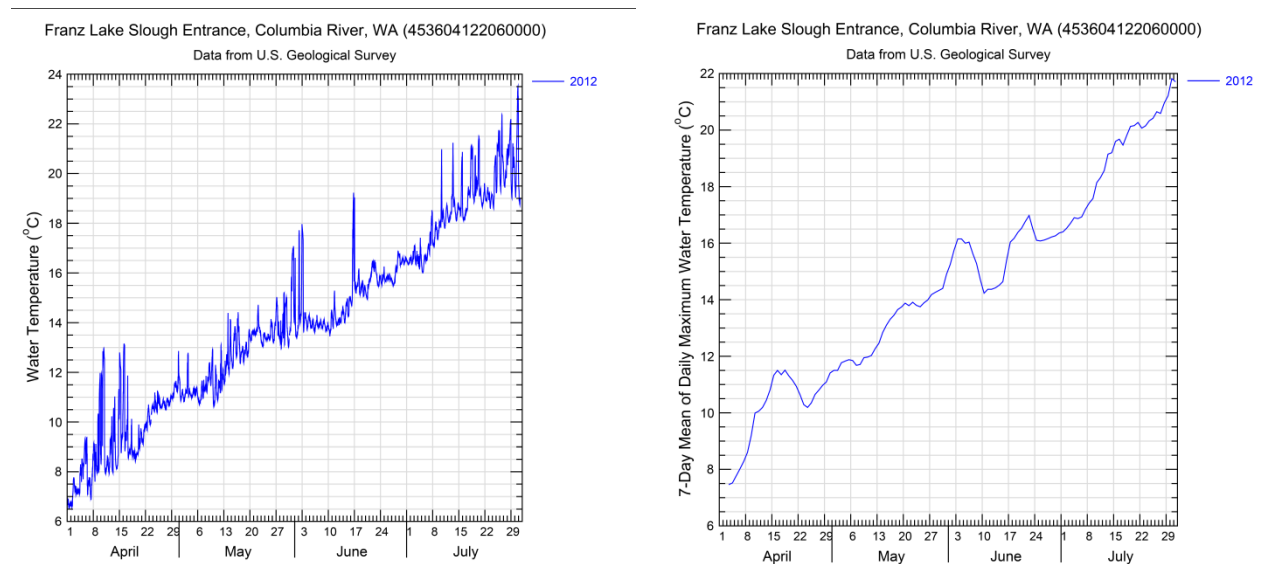
Figure 19. Water temperature at Franz Lake Slough, April–July 2012: a) continuous measurements; b) average weekly maximum. [°C, degrees Celsius]
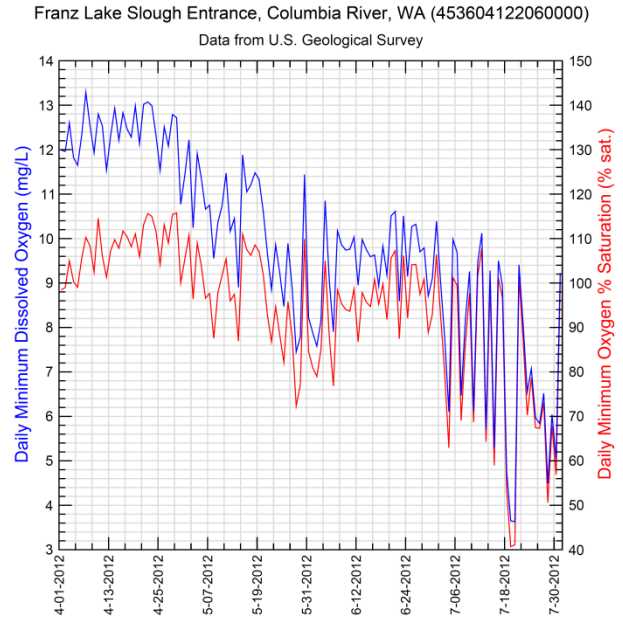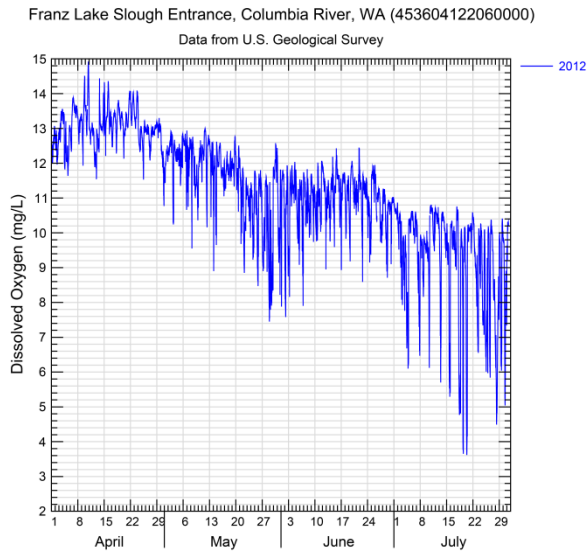
Figure 20. Dissolved oxygen concentrations at Franz Lake Slough, April–July 2012: a) continuous measurements; b) daily minimum concentrations and percent saturation. [mg/L, milligrams per liter]
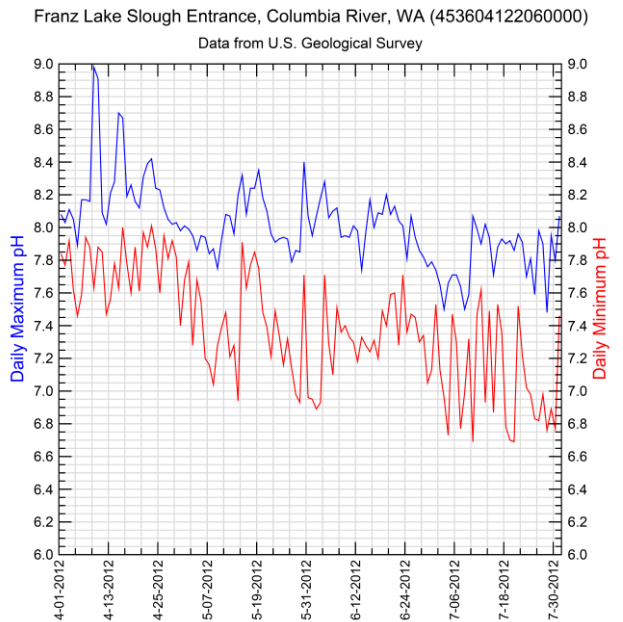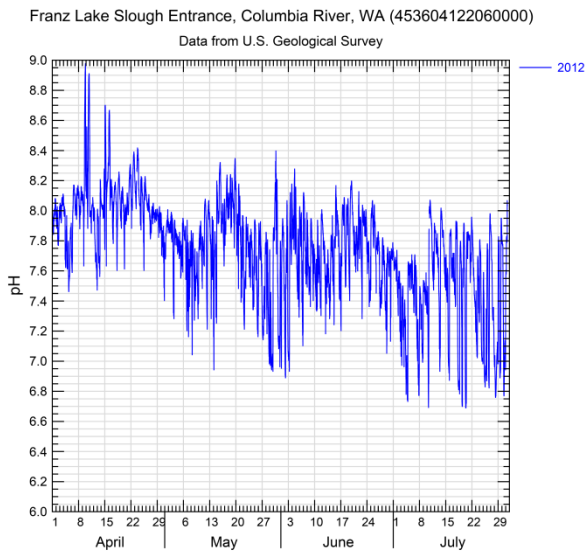
Figure 21. pH at Franz Lake Slough, April–July 2012: a) continuous measurements; b) daily minimum and maximum.

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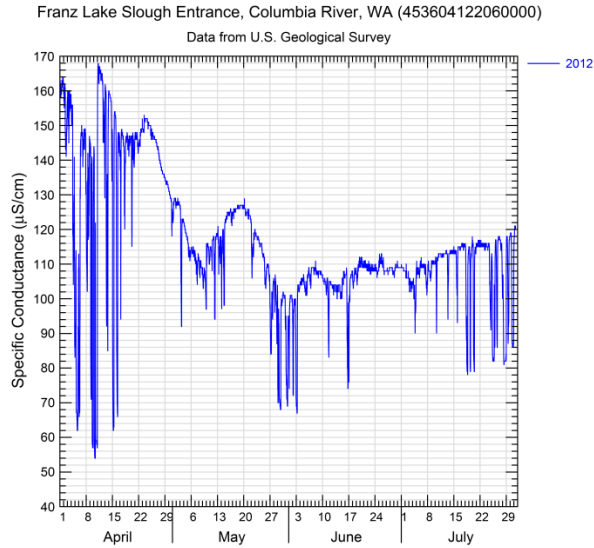
Figure 22. Specific conductance at Franz Lake Slough, measured continuously, April–July 2012. [uS/cm, microsiemens per centimeter]

Campbell Slough

Water temperature at Campbell Slough increased steadily throughout the 2012 monitoring period, with the temperature consistently greater than the threshold of 17.5 °C by the second week of July (Figure 23). Shorter-term temperature peaks earlier in the season also exceeded the threshold, causing the weekly maximum temperature standard to be violated in late May, early and late June, and nearly all of July. Dissolved oxygen concentrations decreased over the season, with on average, larger daily fluctuations as the season progressed (Figure 24). The daily minimum dissolved oxygen concentration dropped to less than the 8 mg/L standard in mid-April, and then intermittently in May and June and for most of July. pH at Campbell Slough peaked roughly monthly through June, then fluctuated with more regularity in July (Figure 21). It was between the standards of 6.5 and 8.5, except briefly during the peak in late April. Specific conductance ranged from approximately 105 to 175 uS/cm during the monitoring period (Figure 22). It decreased through May and gradually increased through July, with large fluctuations starting in May.

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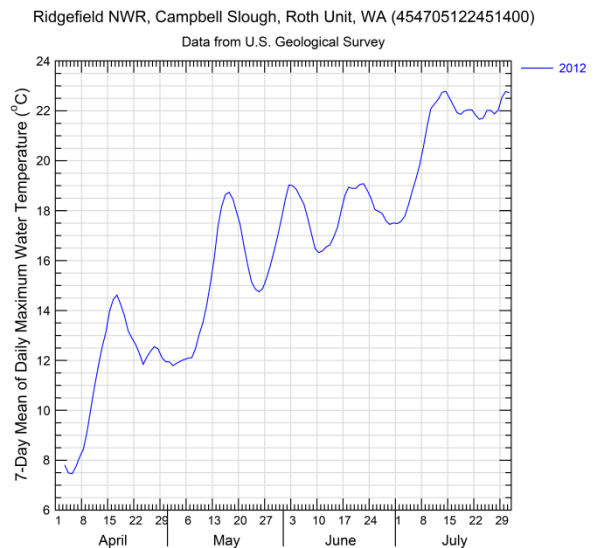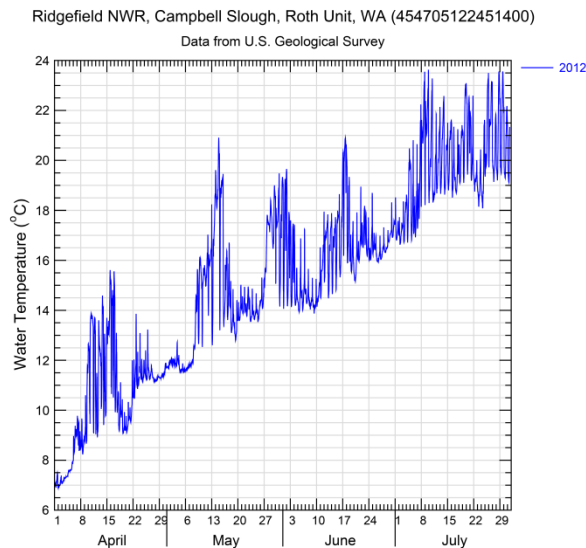
Figure 23. Water temperature at Campbell Slough, April–July 2012: a) continuous measurements; b) average weekly maximum. [°C, degrees Celsius]
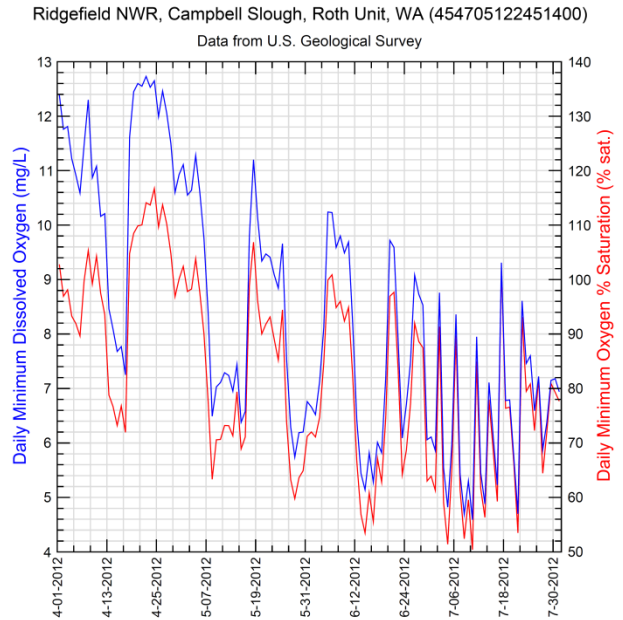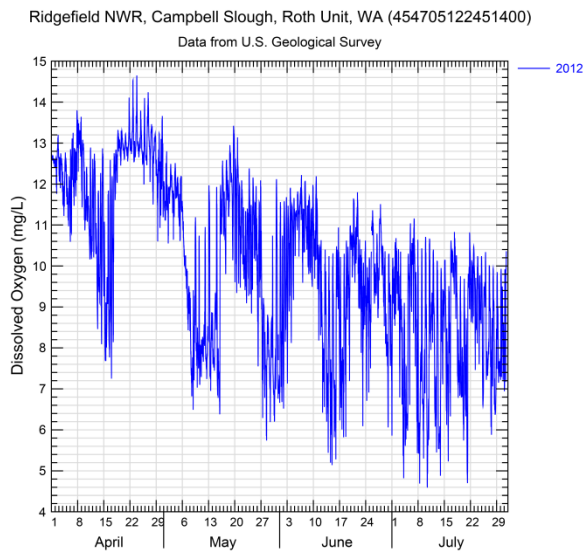
Figure 24. Dissolved oxygen concentrations at Campbell Slough, April–July 2012: a) continuous measurements; b) daily minimum concentration and percent saturation. [mg/L, milligrams per liter]
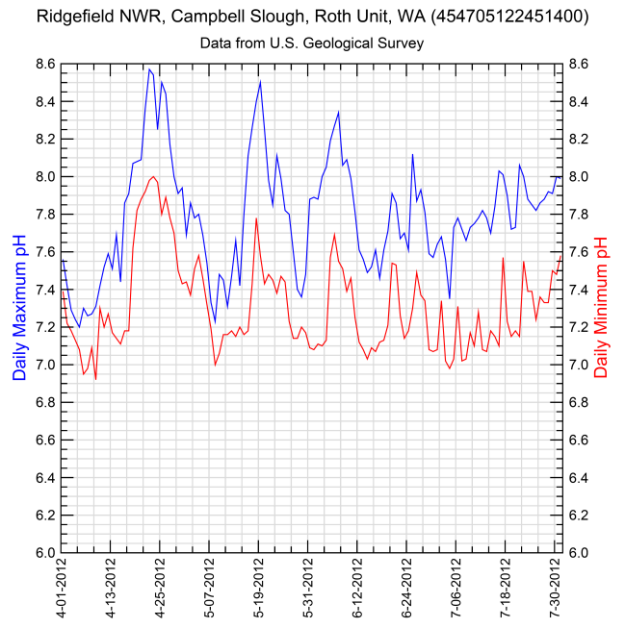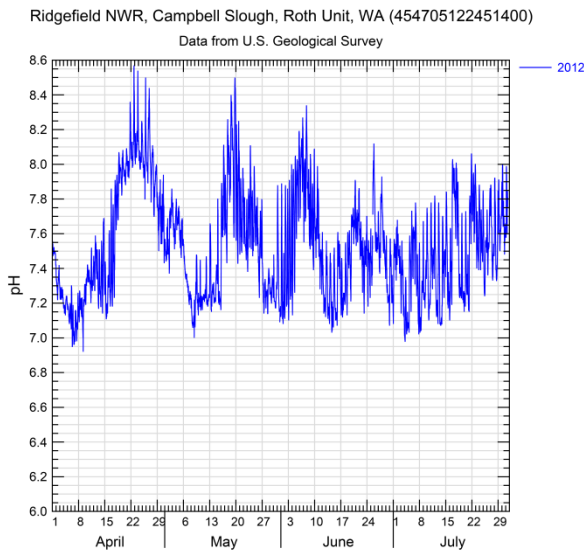
Figure 25. pH at Campbell Slough, April–July 2012: a) continuous measurements; b) daily minimum and maximum.
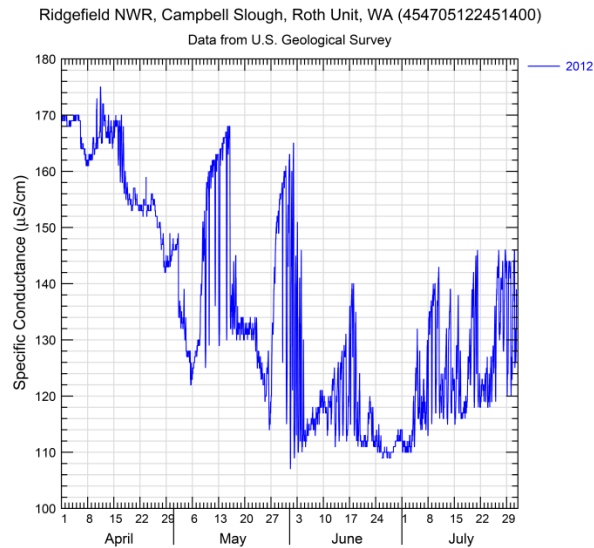
Figure 26. Specific conductance at Campbell Slough, measured continuously, April–July 2012. [uS/cm, microsiemens per centimeter]

Whites Island

Water temperature at Whites Island rose steadily during the 2012 monitoring period, with slightly larger daily fluctuations in May and July compared to April and June (Figure 25). The weekly maximum temperature standard of 17.5 °C was not exceeded until early July, at which point even the daily minima were almost continuously greater than that threshold. Dissolved oxygen concentrations decreased during the monitoring period, varying over larger ranges each day by late June and into July (Figure 28). The daily minimum dissolved oxygen concentration was greater than the standard of 8 mg/L until early July. The daily mean pH was approximately 7.7 to 8.0 throughout the monitoring period (Figure 27). As with dissolved oxygen concentrations, the daily range in pH increased as the season progressed. The maximum pH standard of 8.5 was exceeded briefly in early June and again in early and late July; all measured values were greater than the minimum standard of 6.5. Specific conductance ranged from approximately 100 to 160 uS/cm during the monitoring period (Figure 28). Changes in specific conductance roughly inversely mirror the patterns in Columbia River discharge at the nearby gaging station at Beaver Army Terminal (USGS Station 14246900); increased river flows are generally associated with decreased specific conductance.
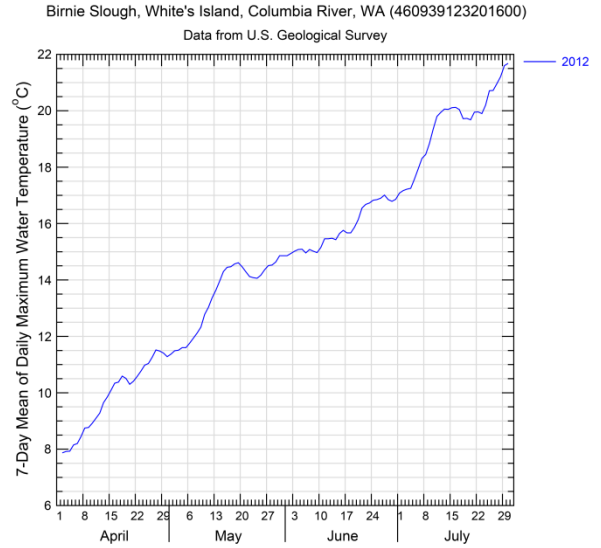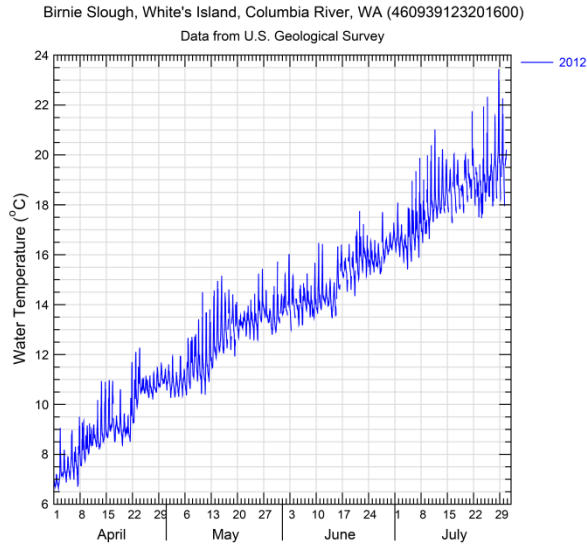
Figure 27. Water temperature at Whites Island, April–July 2012: a) continuous measurements; b) average weekly maximum. [°C, degrees Celsius]
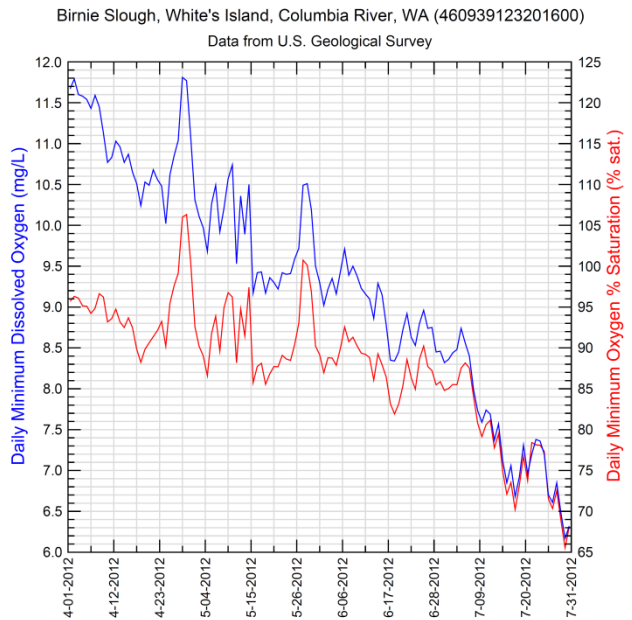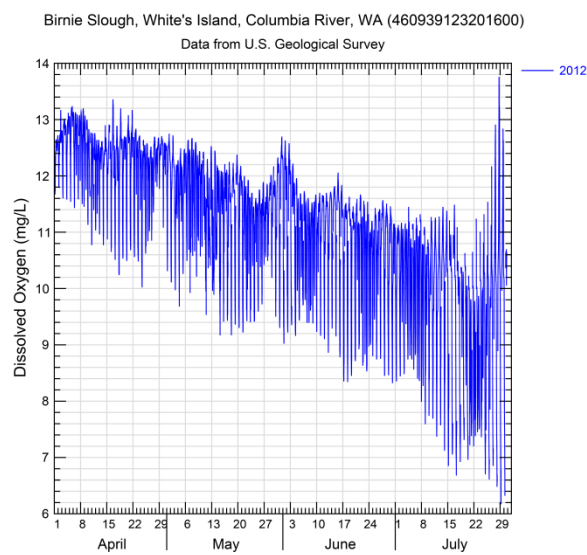
Figure 28. Dissolved oxygen concentrations at Whites Island, April–July 2012: a) continuous measurements; b) daily minimum concentrations and percent saturation. [mg/L, milligrams per liter]
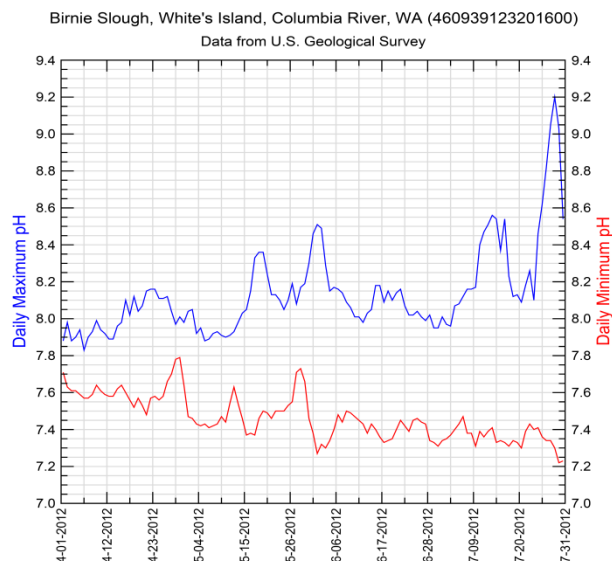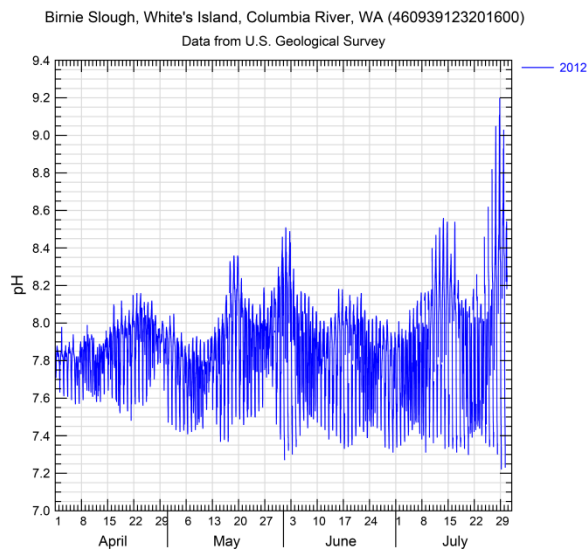
Figure 29. pH at Whites Island, April–July 2012: a) continuous measurements; b) daily minimum and maximum.
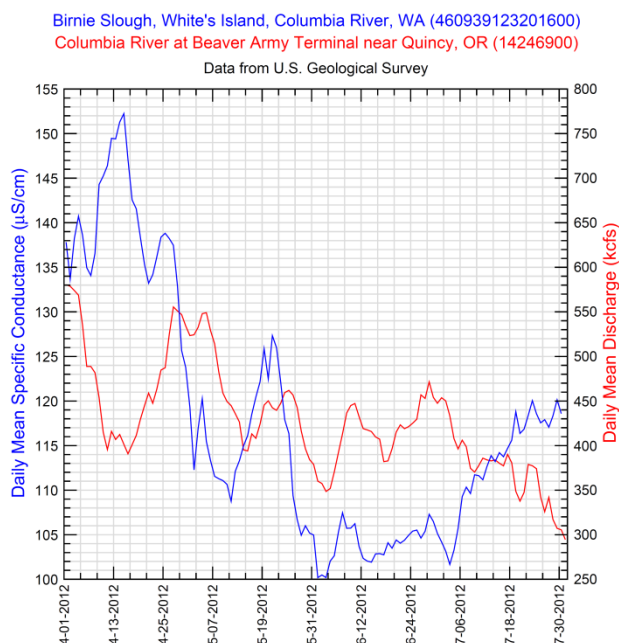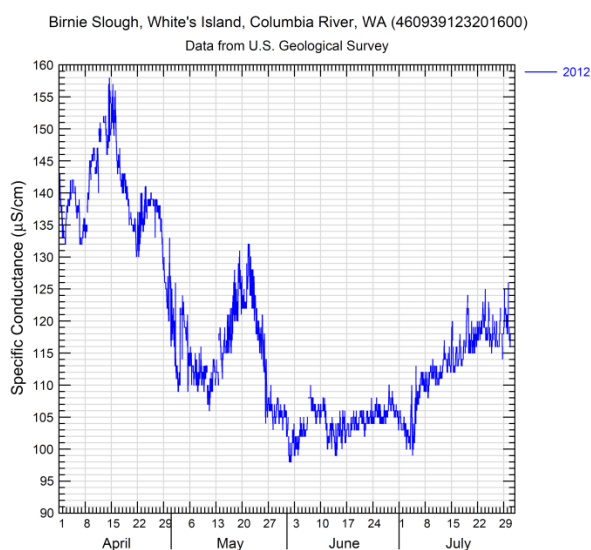
Figure 30. Specific conductance at Whites Island, April–July 2012: a) continuous measurements; b) daily mean specific conductance at Whites Island and daily mean Columbia River stage at Beaver Army Terminal. [uS/cm, microsiemens per centimeter; kcfs, thousand cubic feet per second]

Ilwaco

The water temperature at Ilwaco increased over the 2012 monitoring period and regularly experienced large fluctuations due to the strong tidal influence on this site (Figure 31). The weekly maximum temperature standard of 17.5 °C was exceeded at times in April and May and consistently after late May. However, the daily mean temperature at Ilwaco was less than the 17.5 °C threshold until early July, except briefly in mid-June. Overall, dissolved oxygen concentrations declined during the monitoring period, although less sharply than at the upstream sites (Figure 32). However, dissolved oxygen concentrations varied greatly, with average daily ranges of 9.7 mg/L. The daily minimum dissolved

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oxygen concentration was less than the 8 mg/L standard almost continuously throughout the monitoring period, although daily mean concentrations were greater than that threshold in April and most of May. Average daily minimum pH was generally consistent throughout the monitoring period, while the daily maximum pH showed larger variation over the season (Figure 33). The maximum standard of 8.5 was exceeded intermittently in April and May. The specific conductance at Ilwaco ranged from approximately 2,000 to 18,000 uS/cm during the monitoring period (Figure 34). Although the large specific conductance values indicate a clear marine influence, the pattern during the monitoring period inversely corresponds to Columbia River flows at the nearest USGS gaging station at Beaver Army Terminal (USGS Station 14246900).
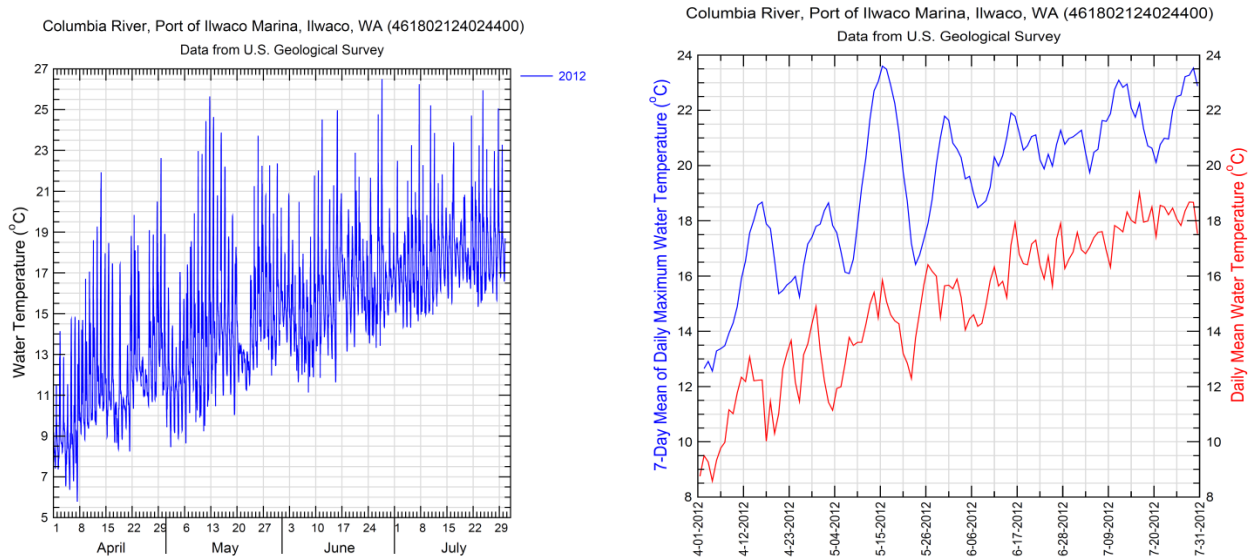
Figure 31. Water temperature at Ilwaco, April–July 2012: a) continuous measurements; b) average weekly maximum and daily mean temperature. [°C, degrees Celsius]
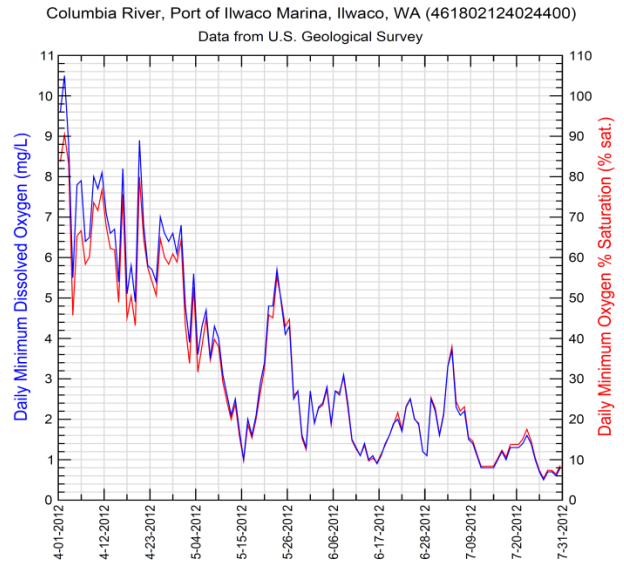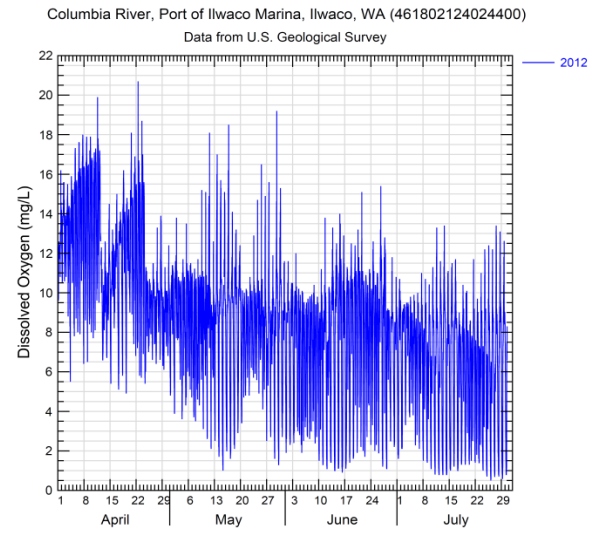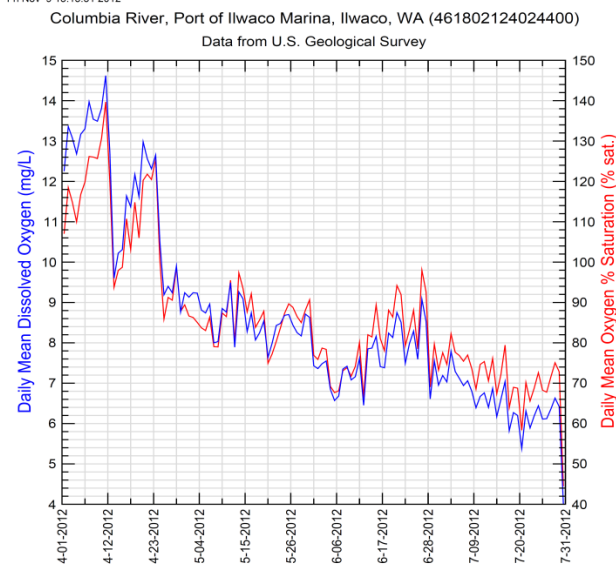
Figure 32. Dissolved oxygen concentrations at Ilwaco, April–July 2012: a) continuous measurements; b) daily minimum concentrations and percent saturation; c) daily mean concentrations and percent saturation. [mg/L, milligrams per liter]
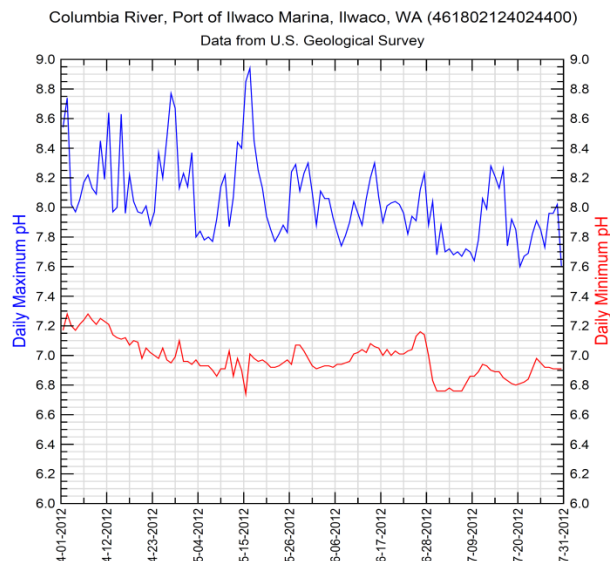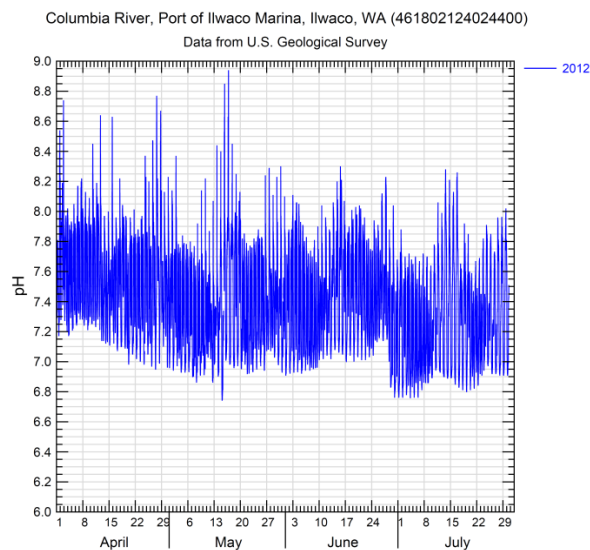
Figure 33. pH at Ilwaco, April–July 2012: a) continuous measurements; b) daily minimum and maximum.
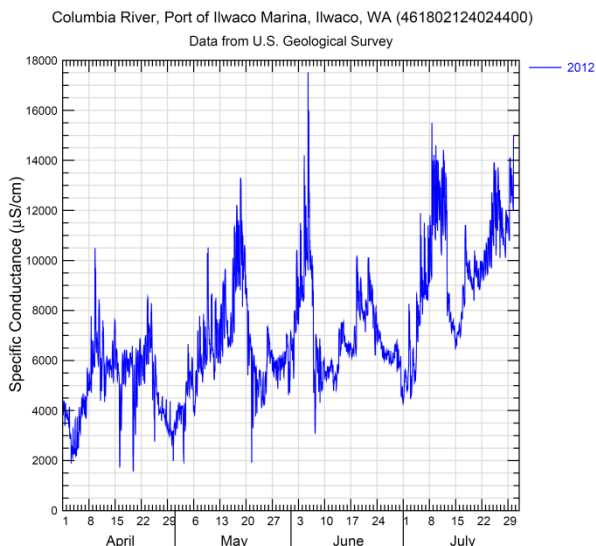
Figure 34. Specific conductance at Ilwaco, measured continuously, April–July 2012. [uS/cm, microsiemens per centimeter]

4.2.2 Nutrients

Franz Lake Slough

During the monitoring period in 2012, total nitrogen concentrations ranged from 0.24 to 0.51 milligrams per liter (mg/L) at Franz Lake Slough. Nearly all of the nitrogen at this site was organic nitrogen, shown as Total Kjeldahl Nitrogen (TKN) on Figure 35. TKN is comprised of organic nitrogen, ammonium (NH_4^+) and ammonia (NH_3). In neutral and high pH waters, ammonia would be more common than ammonium, so in the absence of ammonia, as at Franz Lake Slough in 2012, TKN likely represented primarily organic nitrogen. Total nitrogen patterns were primarily driven by changes in TKN over the monitoring period. The exception to this is in late April, when nitrate, the most bioavailable form of nitrogen, peaked likely due to the freshet. The peak in nitrate did not cause a resultant peak in total

nitrogen due to a dip in TKN. Total nitrogen peaked in mid-May, decreased through the end of June, and then rose in early July. This pattern is inversely correlated with the gage height on the Columbia River below Bonneville Dam (USGS Station ID 14128870), the closest gaging station to Franz Lake Slough. Increasing Columbia River levels corresponded to lower total nitrogen concentrations at Franz Lake Slough.
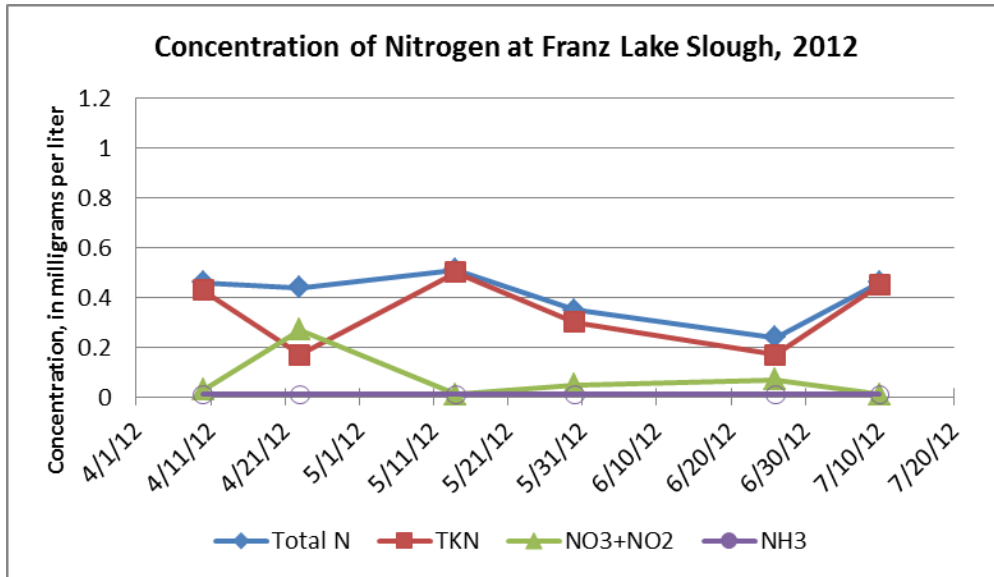
Figure 35. Concentrations of nitrogen species at Franz Lake Slough in 2012. [N, nitrogen; TKN, Total Kjeldahl Nitrogen; NO3+NO2, nitrate plus nitrite; NH3, ammonia]

Total phosphorus concentrations at Franz Lake Slough were also inversely related to the Columbia River stage, although to a lesser extent than nitrogen. Therefore, patterns in total phosphorus mirrored those in total nitrogen. Orthophosphate, the bioavailable inorganic phosphorus species, did not appear to drive the changes in total phosphorus concentrations (Figure 36). However, like total phosphorus, orthophosphate concentrations also peaked in mid-May and reached the minimum in late June.
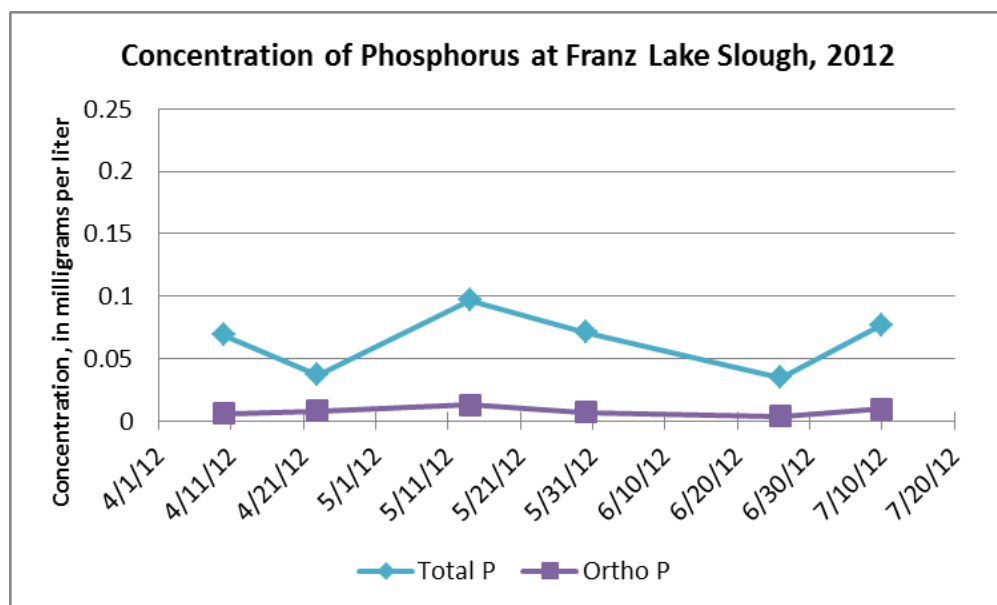
Figure 36. Concentrations of phosphorus species at Franz Lake Slough in 2012. [P, phosphorus; Ortho P, orthophosphate]

Campbell Slough

In 2012, nearly all the total nitrogen is organic nitrogen (shown by TKN on Figure 37). The only exception is in late June, when nitrate was also detectable. Like at Franz Lake Slough, total nitrogen concentrations at Campbell Slough correlated negatively with Columbia River gage heights. The gage heights used to compare to data from Campbell Slough came from the nearest gaging station, the Columbia River at Vancouver (USGS Station 14144700). As at Franz Lake, total nitrogen at Campbell Slough peaked in early May, decreased through late June, and rose in early July.
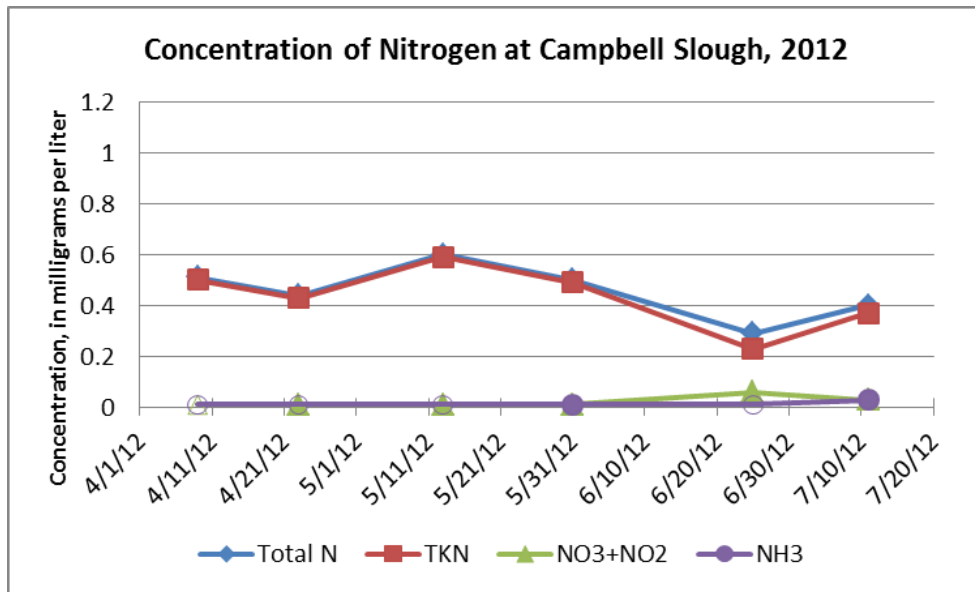
Figure 37. Concentrations of nitrogen species at Campbell Slough in 2012. [N, nitrogen; TKN, Total Kjeldahl Nitrogen; NO3+NO2, nitrate plus nitrite; NH3, ammonia]

Total phosphorus concentrations at Campbell Slough were also inversely related to the Columbia River gage height. The total phosphorus concentrations and pattern over the 2012 monitoring period were similar at Campbell Slough and Franz Lake Slough (Figure 38 and Figure 36). However, the orthophosphate concentrations increased through late May at Campbell Slough and remained higher at that site through the end of the monitoring period.
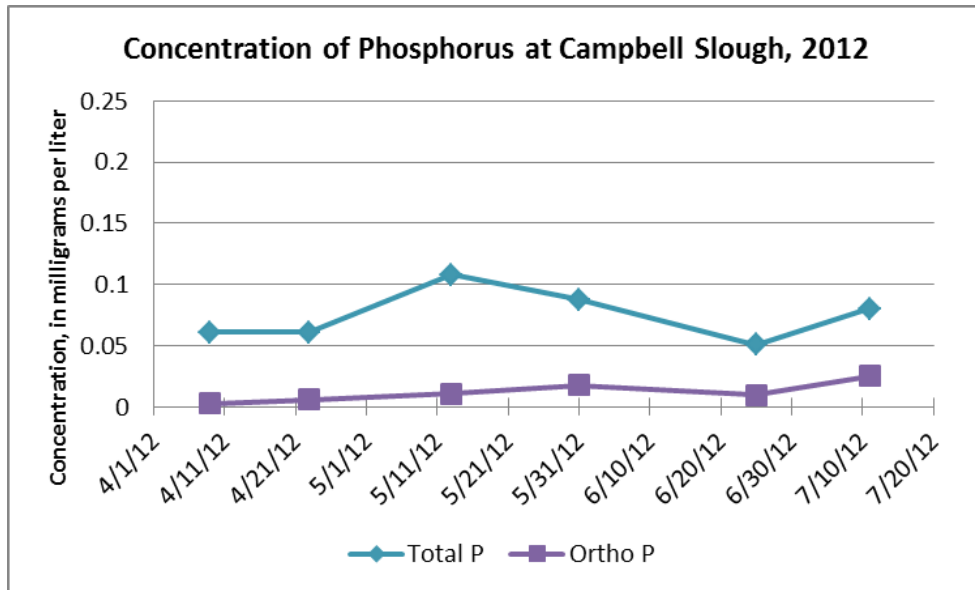
Figure 38. Concentrations of phosphorus species at Campbell Slough in 2012. [P, phosphorus; Ortho P, orthophosphate]

Whites Island

The concentration of total nitrogen in the water column at Whites Island decreased from early April to late May 2012, increased slightly in late June, and decreased in early July (Figure 39). The changes in total nitrogen concentration at this site were driven largely by changes in nitrate plus nitrite, as TKN was relatively consistent throughout the monitoring period.
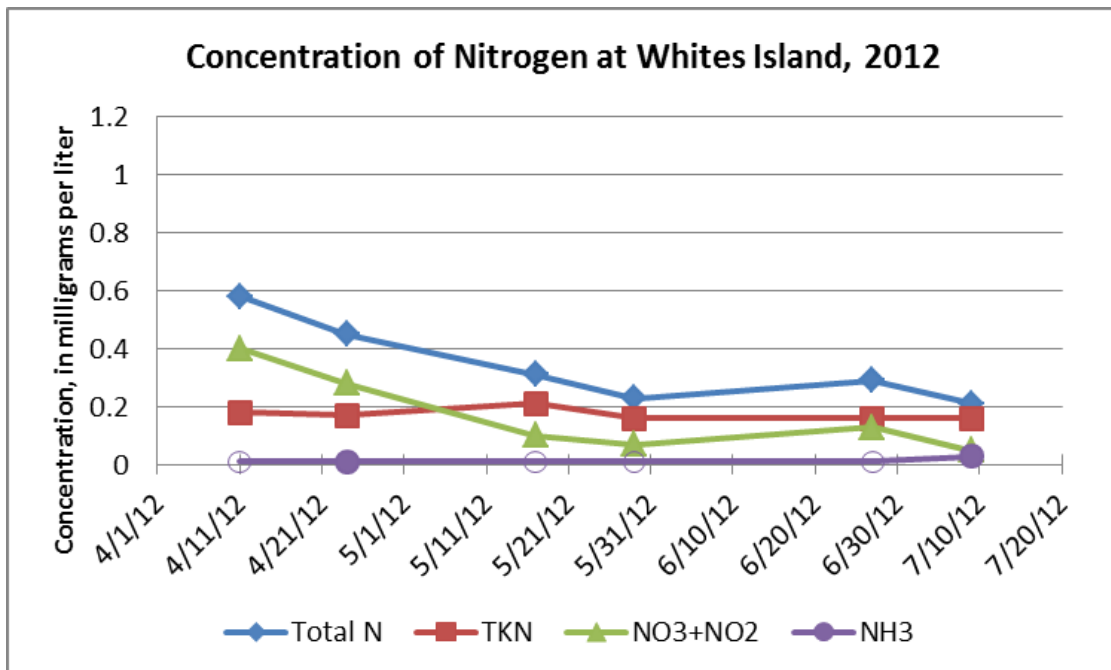
Figure 39. Concentrations of nitrogen species at Whites Island in 2012. [N, nitrogen; TKN, Total Kjeldahl Nitrogen; NO3+NO2, nitrate plus nitrite; NH3, ammonia]

As opposed to the upstream sites, the relationship between total nitrogen and gage height on the main stem of the river was positively correlated; total nitrogen at this site increased with increasing gage height. The Columbia River gaging station used for comparisons at Whites Island is Columbia River at Beaver Army Terminal, OR (USGS Station ID 14246900).

Compared to the other sites, the concentration of total phosphorus at Whites Island was typically lower and more consistent throughout the monitoring period. The concentrations of total phosphorus and orthophosphate decreased slightly in April and May and then were steady through early July (Figure 40).
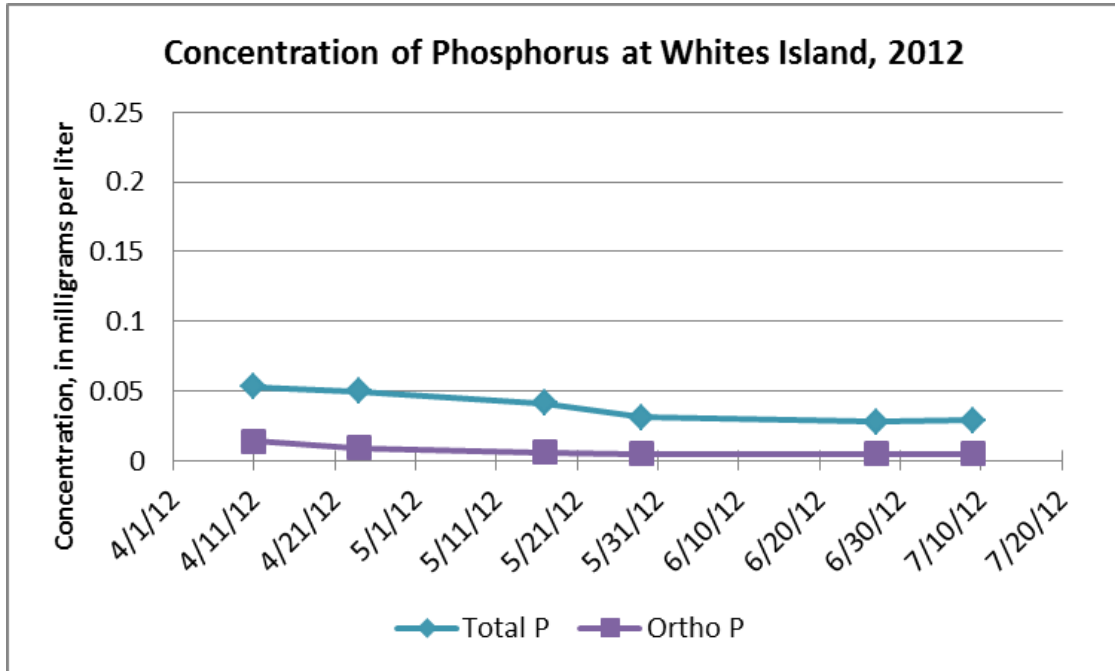
Figure 40. Concentrations of phosphorus species at Whites Island in 2012. [P, phosphorus; Ortho P, orthophosphate]

Ilwaco
Total nitrogen and most nitrogen species at Ilwaco were relatively consistent from mid-May through early July (Figure 41). Concentrations of all nitrogen species were higher in early April than in May and June and peaked in late April. However, those differences may have been a result of the tidal stage during the time of sampling. The later season samples were collected at roughly the same tide height, but the two earlier samples were collected at lower tides (the late April sample was collected at very low tide). Ilwaco is the only site that had measureable ammonia (NH_3) during every sampling event.
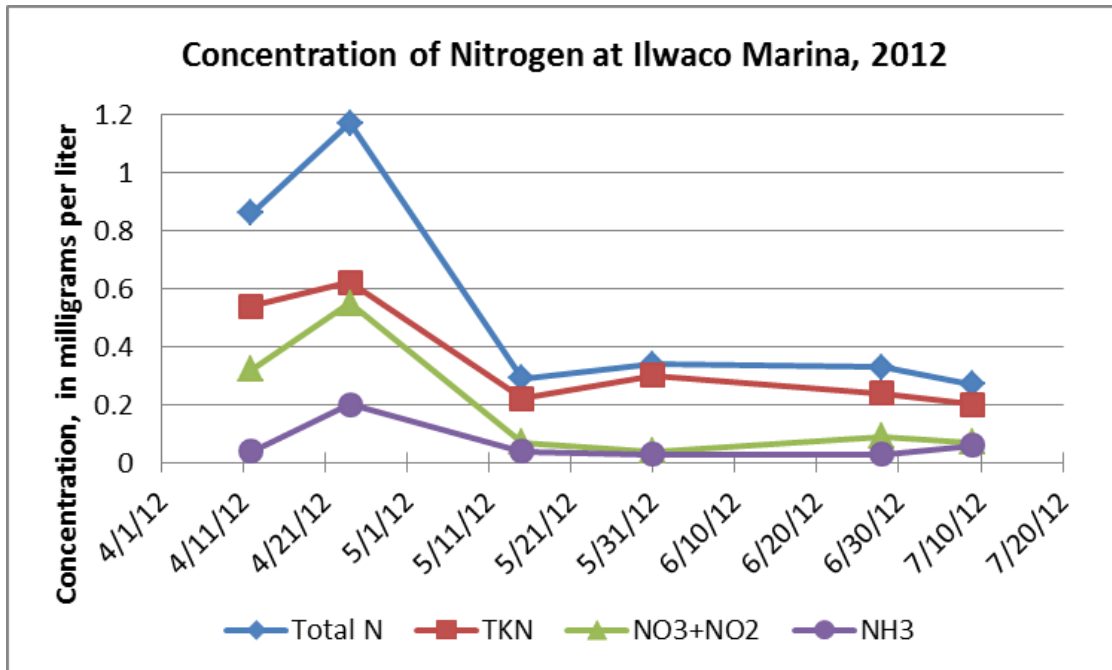
Figure 41. Concentrations of nitrogen species at Ilwaco in 2012. [N, nitrogen; TKN, Total Kjeldahl Nitrogen; NO3+NO2, nitrate plus nitrite; NH3, ammonia]

Like the nitrogen species, orthophosphate in the water column at Ilwaco spiked in late April, then decreased and was fairly consistent through early June (Figure 42). Total phosphorus concentrations fluctuated more strongly than did orthophosphate. Most of the measured total phosphorus concentrations at Ilwaco were much larger than at the upstream sites, reflecting the marine influence at this site
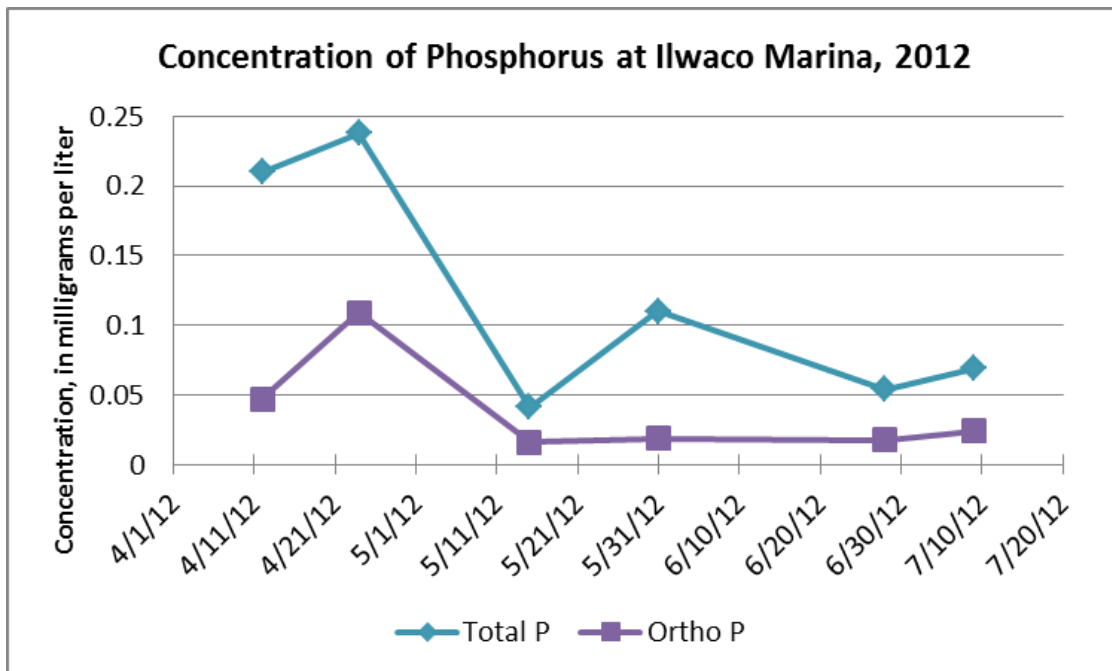
Figure 42. Concentrations of phosphorus species at Ilwaco in 2012. [P, phosphorus; Ortho P, orthophosphate]

4.2.3 PAR

Photosynthetically Available Radiation (PAR) data were collected only during site visits (not measured continuously), so are of limited utility on their own and are not presented in the site-by-site descriptions of abiotic conditions. Instead, those data will be used in a later analysis linking environmental conditions and phytoplankton productivity data.

4.2.4 Synoptic Temperature and Salinity During Fish Sampling

In addition to the continuous water quality monitoring conducted by USGS at the Ilwaco Slough, Whites Island, Campbell Slough, and Franz Lake, synoptic temperature data were collected at the fishing sites during sampling. Salinity measurements were also taken at Ilwaco Slough, the only one of the sampling sites with saltwater influence.

Due to sampling permit issues and extreme water levels described above, not all sites could be sampled each month. At those sites that could be sampled, water temperatures in February and March ranged from 6-9°C. Temperatures began increasing in April, rising steadily to maximum levels of 20-28°C in August, then declining again to 10-15°C by October. All sites followed the same general seasonal pattern in water temperature, but summer water temperatures were most extreme at Campbell Slough, Government/Lemon Island, Washougal, and Franz Lake, reaching 27-28°C. At other sites maximum water temperatures were in the 20-21°C range (Table 12).

Table 12. Average monthly water temperature at 2012 Ecosystem Monitoring Project fishing sites. NS = not sampled

	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct
A-Ilwaco Slough	7	7	10.5	14.4	16.7	20	17	NS	10.5
B-Secret River	7	6	9	14.9	13.3	19.1	21	14	12
B-Welch Island	6	6.5	12	13.8	13.4	19	20	19.7	15
C-Whites Island	6	6	9.5	14.5	14.2	18.9	21	19.9	15
F-Campbell Slough	NS	NS	NS	20.5	16.4	20	27	26	NS
G-Government/Lemon Island	NS	6	9	13.7	16	19.2	27	24	14
G-Washougal	7	6.5	11	14.5	17.5	22.8	28	23	NS
G-Reed Island	8	NS	NS	NS	NS	NS	21	NS	NS
H-Franz Lake	9	6	NS	NS	NS	NS	27	23	13

Ilwaco Slough was the only site sampled with a saltwater influence. In 2011, the salinity of the water did not change greatly throughout the study. It was much more variable in 2012 ranging from a low of 2 ppt in August to a high of 19 ppt in October (Table 13).

Table 13. Average monthly salinity (ppt) at Ilwaco in 2011 and 2012

	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
A-Ilwaco Slough 2011	ND	ND	ND	ND	ND	10	10	10	9	11	9
A-Ilwaco Slough 2012	8	3	4	7	ND	3	2	ND	19	ND	ND

At the trend sampling sites, water temperatures measured in 2012 were generally consistent with ranges observed in previous years (Table 14). At Campbell Slough and Franz Lake, maximum water temperatures in 2012 were relatively high, approaching the maximums measured in 2009.

Table 14. Water temperature trends at EMP trend sites. ND = no data

site	Feb	Mar	Apr	May	June	Jul	Aug	Sept	Oct	Nov	Dec
						month					
A-Ilwaco Slough 2011			8.9	14.7	16.8	19.5	19	16	15.5	10	7
A-Ilwaco Slough 2012	7	7	10.5	14.4	16.7	20	17	NS	10.5		
C-Whites Island 2009			9.4	10.9	16.6	18.1	24.1	22	ND	ND	ND
C-Whites Island 2010			8.3	12.8	14.9	19.3	20.5			ND	ND
C-Whites Island 2011				12.7	16.8	20	21	19	16	10.5	8
C-Whites Island 2012	6	6	9.5	14.5	14.2	18.9	21	19.9	15		
F-Campbell Slough 2007	ND	ND	ND	12.9	17.1	21	ND	ND	ND	ND	ND
F-Campbell	ND	ND	ND	9.5	15.8	ND	21.2	23.3	ND	ND	ND

Site											
F-Campbell Slough 2008				12.5	18.8	24.2	28.6	19.9	ND	ND	ND
F-Campbell Slough 2009			11.5	17.4	17.4	20.6	22.8	ND	ND	ND	ND
F-Campbell Slough 2010											
F-Campbell Slough 2011			12.6			22.5	21.5	22	15		
F-Campbell Slough 2012			20.5	16.4	20	27			3	ND	ND
H-Franz Lake 2008			9.2	13.4		17.5	26				
H-Franz Lake 2009			10.7	11.2		21	28.2				
H-Franz Lake 2011						23	20.5	21.2	13	9	5.5
H-Franz Lake 2012	9	6	NS	NS	NS	NS	27	23	13		

a denotes sites not sampled due to lack of permission to access the site.
b denotes sites not sampled due to extreme water levels

4.3 Habitat Structure

4.3.1 2012 Annual Monitoring

All results are presented by site and figures are ordered by location in the LCRE, starting at the mouth and continuing up-river.

4.3.1.1 *Sediment Composition*

The grain size and total organic carbon (TOC) results from the sites monitored in 2012 are similar to those we have measured at other sites in the LCRE in previous years with some notable exceptions (Figure 43). The Secret River site had a higher percentage of fine sand in the channel and mudflat than the marsh portions of the site, with the percentage increasing at the higher elevation marsh. The TOC patterns are similar to the percent fines; lowest in the flats and highest in the high marsh. The latter pattern also likely corresponds to higher vegetation present in the strata. The Welch Island site is similar, except the low marsh at this site only occurs in channels where the fines and TOC are higher than the low marsh at the Secret River site. The high marsh also had a slightly higher percentage of sands than the Secret River site.

Two of the upriver sites, Government/Lemon Island and Reed Island, had high sand content with very low TOC. The highest elevation strata (ELPA/SASP saplings) had higher silt content perhaps due to proximity to the willow tree zone, where floodwaters slowed by trees may allow more deposition of fines. The third upriver site at the old Washougal River channel is more like other off channel sites we have measured, with higher silt content and TOC.
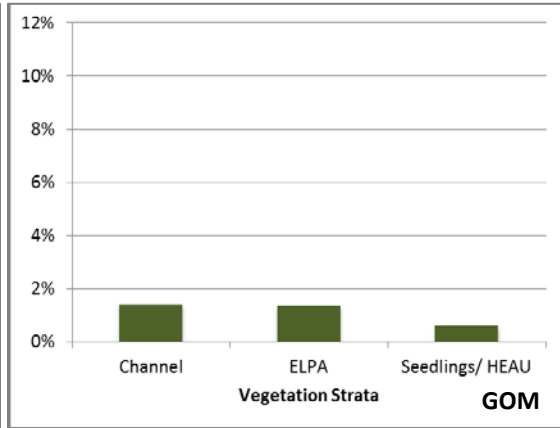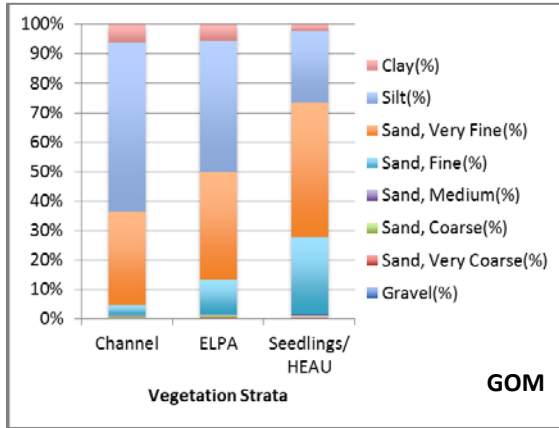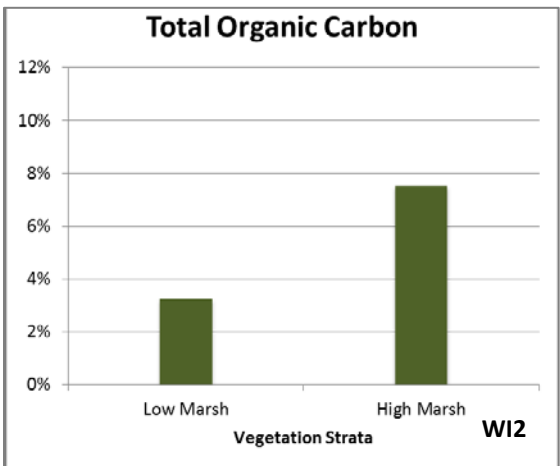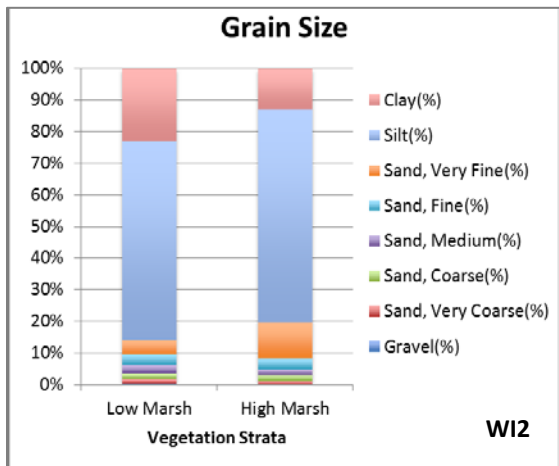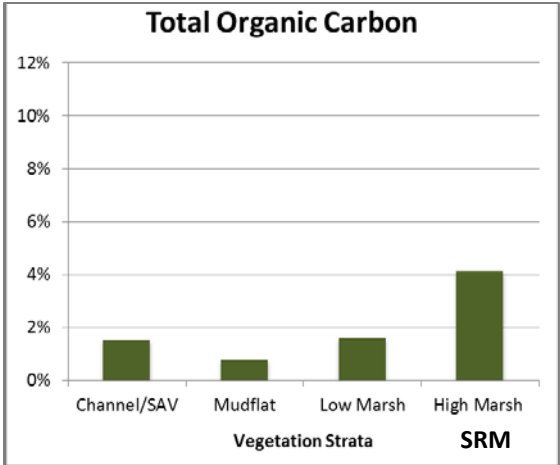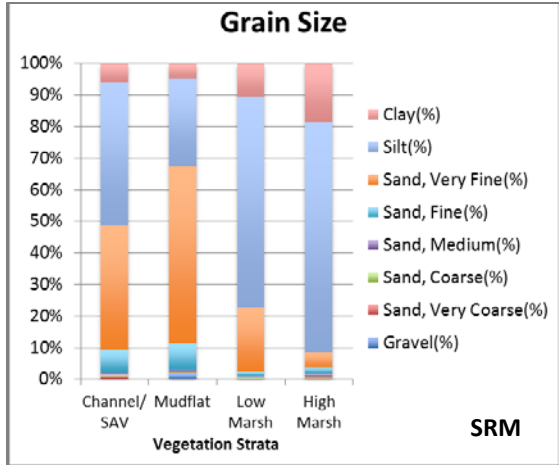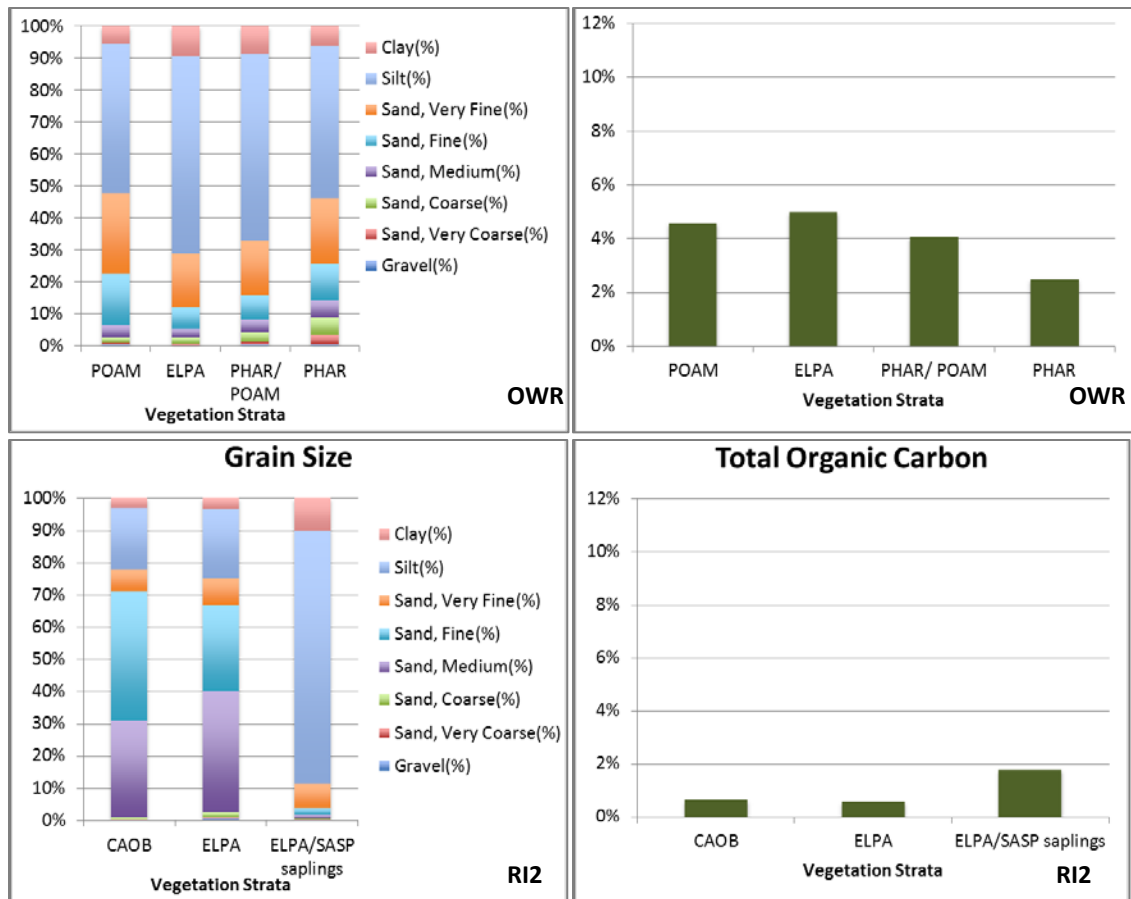
Figure 43. Grain size (on the left) and total organic carbon (TOC; on the right) at the new monitoring sites for 2012.

Total organic carbon has been measured in the sediment samples from EM sites since 2008 and in general the amounts recorded in these samples is low compared to values from other marsh systems (Sagar et al. 2013). An alternative sampling method was employed to determine if our original method might be contributing to the lower TOC amounts. The results indicate that our original sampling method does not appear to be the reason for the low TOC values (Figure 44). The Secret River sample in the top 10 cm of the 25 cm core (0-5 = 3.2%, 5-10 = 4.6%) had percent TOC very similar to the 10 cm core taken from the high marsh (4.1%; Figure 43). Likewise, the results from the 2011 sediment sampling at the Ilwaco marsh (original sampling method) ranged from 5.7% to 8.9% whereas the 2012 sampling resulted in 11.3% TOC in the upper two core segments (0-10 cm total).
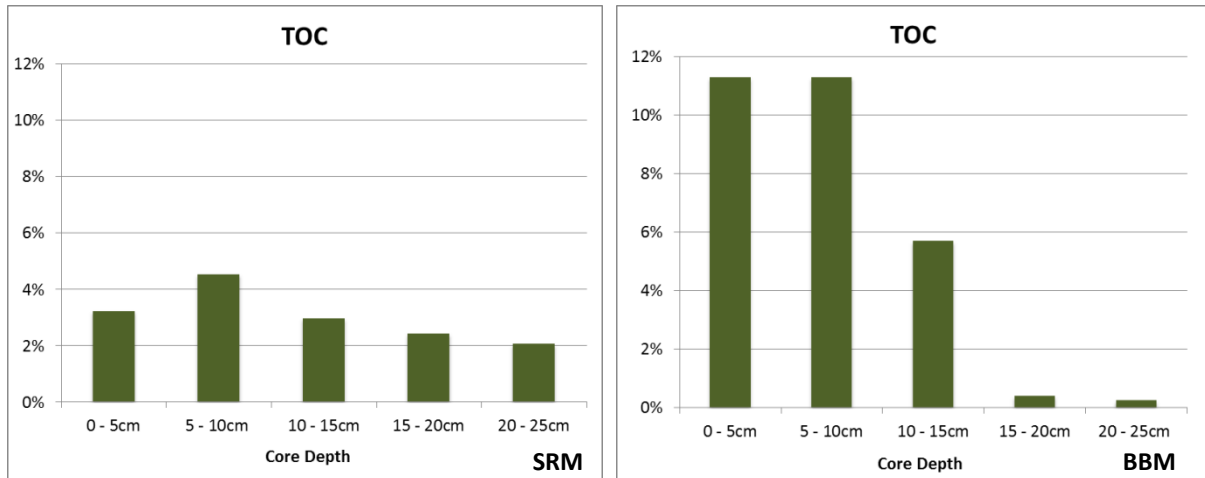
Figure 44. Total organic carbon from 25 cm cores from Secret River high marsh (left) and Ilwaco high marsh (right).

4.3.1.2 *Sediment Accretion Rates*

Accretion rates varied from -0.4 cm to 2.7 cm at the sites measured in 2012. Accretion appears to be partially affected by elevation, with higher accretion occurring at lower elevations. However, other factors such as location in the estuary/river, distance from the main stem of the river, and sediment sources also likely affect sediment accretion rates. Interannual variability related to flows is also a likely cause for variation.

Table 15. Sediment accretion rates measured at sites in 2012.

Site	Rkm	Stake Elevation (m, CRD)	Year	Accretion /Erosion (cm)	Days Deployed	Annual Accretion/ Erosion Rate (cm)
Ilwaco	6	1.81	11-12	0.1	369	0.1
Secret River - low	37	1.06	11-12	ND1	NA	NA
Secret River - high	37	2.02	11-12	ND2	NA	NA
Whites Island	72	2.05	11-12	0.9	364	0.9
Prescott Slough	115	ND	10-12	4.0	628	2.3
Burke Island	131	0.73	11-12	2.8	378	2.73
Goat Island	131	1.67	11-12	0.7	380	0.7
Deer Island	132	1.62	11-12	0.3	379	0.3
Cunningham Lake	145	1.49	11-12	1.4	375	1.4
Campbell Slough	149	1.54	11-12	0.9	378	0.9
Franz Lake	221	1.87	11-12	-0.4	371	-0.4

1 The north accretion stake originally deployed in 2008 appears to have sunk 7 cm between 2011 and 2012, therefore measurement not valid.

2 The south accretion stake originally deployed in 2011 appears to have sunk 2 cm between 2011 and 2012, therefore measurement not valid.

3 The 2011 measurements at Burke Island were taken in 9 cm of water; the effect of water on the measurement is uncertain.

4.3.1.3 *Hydrology*

Hydrographs from the sites where WSE was collected during the 2012 water year are provided Appendices. The following observations were made for these sites:

- The Ilwaco site results indicate that the WSE at this site is not affected by the spring freshet. The WSE is however elevated due to winter storm events.
- The Whites Island site was exposed at low tide for a shorter duration each day during the period of the spring freshet than during the rest of the year.
- The Campbell Slough and Cunningham Lake sites had similar hydrologic patterns except that because the mouth of Cunningham Slough is located near the tip of Sauvie Island, approximately 7 rkm downstream from the mouth of Campbell Slough, the site has a slightly greater tidal range and slightly lower WSE during the freshet. Both sites were inundated for approximately 5 months during the spring freshet.
- The Government/Lemon Island and Franz Lake sites were only inundated during high water events due to the high elevation of the sites 1.95 m and 1.81 m, CRD, respectively.
- The Government/Lemon Island site was inundated for approximately 5 months, 2 of which had greater than 2 meters of water depth. The Old Washougal River and Franz Lake sites were inundated for 5.5 months, with 4 months greater than 2 m water.
- The Reed Island site was inundated for 6 months due to the lower elevation of the site (1.48m, CRD).

These observations serve to illustrate the extremely different hydrologic patterns that occur throughout the estuary, particularly in a high water year such as 2012. The varying effects these patterns can be observed in the vegetation community composition, cover, and productivity as will be discussed in Sections 4.3.1.5, 4.3.1.6, and 4.3.1.7 below and the in the Trends Analysis in Section 4.3.2.

4.3.1.4 *Salinity*

Salinity was measured at the Ilwaco site during 2011-2012. The salinity is highly variable at the site both seasonally and daily. The overall range for the entire deployment period was 23.3 parts per thousand (ppt), with the highest salinity in October 2011 and the lowest (0.1 ppt) in the spring of 2012. This pattern corresponds to low discharge of the River in the fall and high discharge in the spring and summer (Figure 4). In addition, the peak salinity events in the fall and winter correspond to high tide events every two weeks, with the exception of periods in November and January when river discharge increased from high precipitation during winter storms (see hydrographs in Appendices). The daily range varies between 0.5 ppt and 13 ppt per day, with 65 percent of the ranges between 2 ppt and 6 ppt per day. Daily variability in salinity is directly related to the tide level, with high tide bringing higher salinity water to the site, and also stratification of the water column where higher salinity water is on the bottom and fresher water at the top. Deviation from this pattern was observed in the late summer when high temperatures appear to be drive an increase in salinity during low tide and high tide brings in fresher water.

On 16 February 2012 the sensor was inadvertently moved 15 cm higher in the water column due to complications during re-deployment after sensor maintenance. The effect of this small move is considerable; giving further evidence of the high variability of the salinity at this site and the stratification of the water column. The lower salinities observed after this date are in part also due to the increased freshwater in the system from the spring freshet which started near this same time (approximately 23 February 2012; see Figure 4).
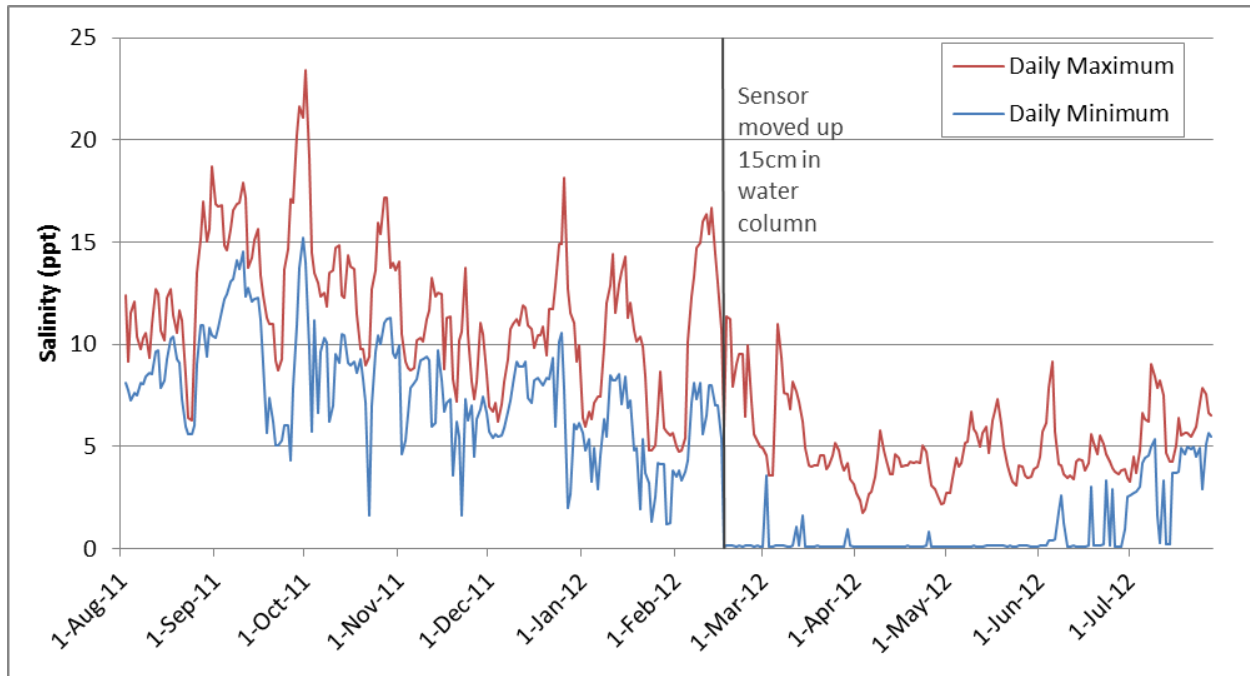
Figure 45. Daily salinity values for the Ilwaco site between August 2011 and August 2012.

4.3.1.5 *Vegetation Assemblage*

The vegetation assemblages observed at the 2012 monitoring sites (
Table 16) can be broadly grouped into three categories associated with the hydro-vegetation zones
(Figure 2) as follows (note that no vegetation data were collected at any sites in Zone 3 in 2012):
1) low species richness/high vegetative cover (Zone 1)
2) high species richness/high vegetative cover (Zone 2)
3) low species richness/low vegetative cover (Zone 4 and 5).

The first category only occurs at the Ilwaco site, with the low number of species attributable to the higher salinity. The second category occurs at the three Zone 2 sites with the number of species ranging from 29 to 40 (the latter is at the Secret River site for the low and high marsh plots combined). The third category occurs in the up-river sites where the lower cover is likely attributable to the duration of high water in 2012 (see Section 4.3.1.7 below for further results and discussion).

At most sites in 2012 native vegetative cover was higher than non-native cover. The two exceptions were at Whites Island where the non-native cover, predominantly reed canary grass (*Phalaris arundinacea*), comprises 57 percent of the total cover and at Cunningham Lake where native and non-native cover is equal at 23 percent each. The high water in 2012 may have had more of an effect on non-native than native cover by reducing reed canary grass cover; whereas the cover of native species such as spike rush (*Eleocharis palustris*) was cumulatively higher (Figure 46).

Characteristics of the vegetation assemblages at the 2012 monitoring sites are discernible in Figure 46. Submerged aquatic vegetation (SAV) species generally occur at the lowest elevations at the sites, although some species also occur in small depressions in the higher elevations, for example horned pondweed (*Zannichellia palustris*) occurs at the Ilwaco site in the tidal channel and in pools in the marsh. At the Secret River low marsh site three SAV species account for 35 percent of the cover, with Canada

waterweed (*Elodea canadensis*) occurring throughout the low marsh in small depressions that hold water at low tide. At all other sites, SAV species account for less than five percent of the cover.

The four sites that occur in Zones 1 and 2 offer an interesting spectrum from sites with high native species cover dominated by Lyngby sedge (*Carex lyngbyei*) (BBM and WI2 at rkm 6 and 53, respectively) to the Whites Island site (at rkm 72) which has high non-native cover dominated by reed canary grass. The Secret River site (rkm 37) is intermediate with lower cover of the sedge and 20 percent cover of reed canary grass. Elevation and salinity are likely the causes for this spectrum of results. Salinity precludes the occurrence of reed canary grass at the Ilwaco site. While salinity is likely much lower at the Secret River and Welch Island sites elevation is a primary driver, with reed canary grass more likely to occur at elevations greater than 1.5 m, CRD (Sagar et al. 2013). At WI2 site elevations ranged from 0.9 m to 1.7 m, CRD, with the reed canary grass occurring only between 1.5 m and 1.7 m, CRD. Conversely, at the SRM high marsh site elevations ranged from 1.9 m to 2.2 m, CRD which is the same elevations reed canary grass occurred. However salinity may be a factor limiting growth at this site. At the WHC site, much of the site is within the elevation range where reed canary grass occurs and salinity is not a factor at this site.

The up-river sites in Zones 4 and 5 have a high percentage of bare ground in 2012 with low overall cover (Figure 46). Spike rush had the highest cover at four of the six sites. At the two Zone 4 sites, species dominance was shared between spike rush and reed canary grass. Reed canary grass was non-existent or present at very low cover at the Zone 5 sites. At the Franz Lake site water smartweed (*D*) had the highest cover. This species is often found in shallow water of lakes, ponds, and streams (Hamel et al. 2001) and seems to thrive in the high water conditions present in 2011 and 2012.

The Government/Lemon Island site was different than any other site monitored as part of this program. The high elevation site is more like a beach than a wetland, with very sandy sediment (Figure 43) that likely experiences high flow velocity during the spring freshet due to the morphology of site and location in the River. The site had no dominant species and most plants were very small and stunted except common sneezeweed (*Helenium autumnale*) and sparse spike rush at lower elevations

Table 16. Species richness and areal cover of native and non-native species at the 2012 monitoring sites.

Site	Rkm	# Native Species	Native Species % Cover	# Non-native Species	Non-native Species % Cover	Total # Species	Total % Cover
Ilwaco	6	16	101.6	1	1.2	17	102.8
Secret River - High	37	22	82.0	7	26.8	29	108.8
Secret River - Low	37	20	89.6	3	3.5	23	93.1
Welch Island	53	30	99.5	9	19.6	39	119.1
Whites Island	72	20	30.3	11	56.7	31	87.0
Campbell Slough	145	14	30.3	6	17.9	20	48.2
Cunningham Lake	149	8	22.9	3	22.9	11	45.8
Government/Lemon Island	181	21	33.8	6	19.7	27	53.5
Washougal River	195	14	30.4	4	8.3	18	38.7
Reed Island	204	17	43.3	6	1.1	23	44.4
Franz Lake	221	14	46.5	2	5.0	16	51.5
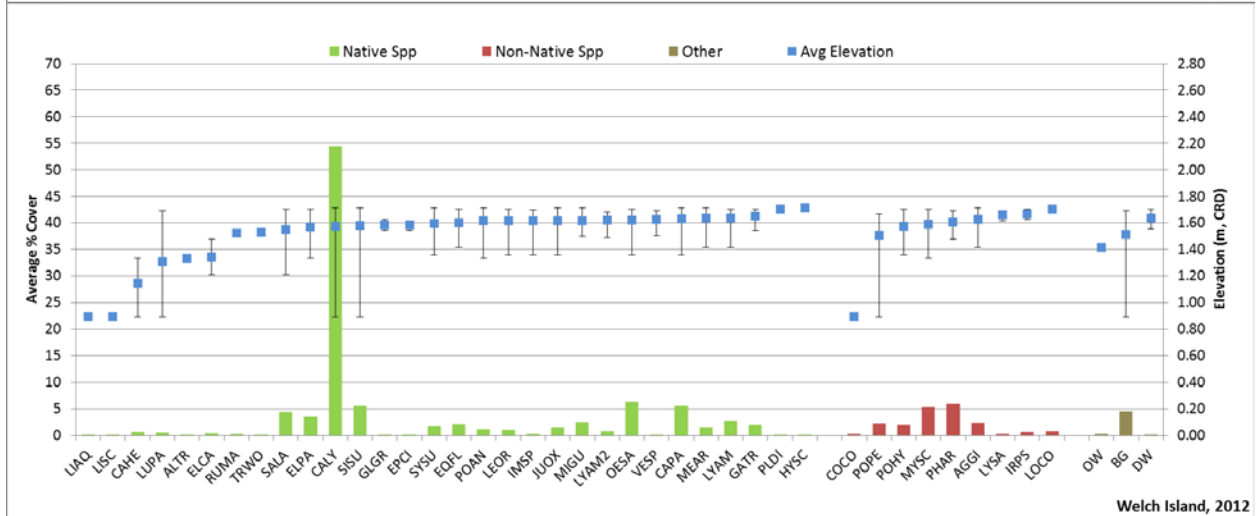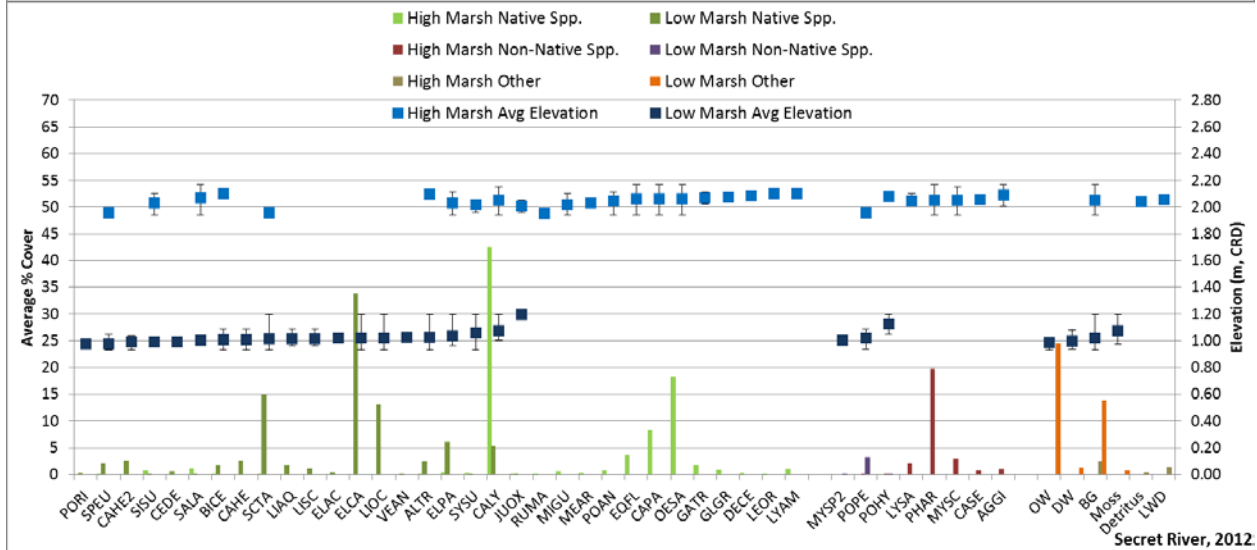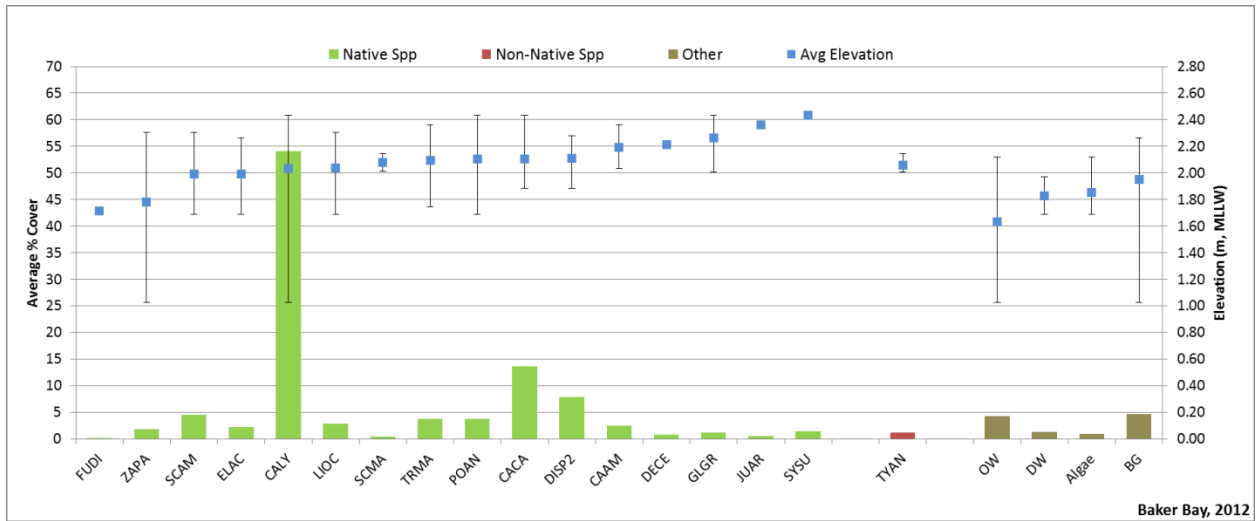
Figure 46. Vegetation species cover and elevations for sites sampled in 2012. Bars represent the minimum and maximum elevations at which the vegetative species occurred within the sample area (See Appendices for species names and percent cover data associated with codes along the x-axis).
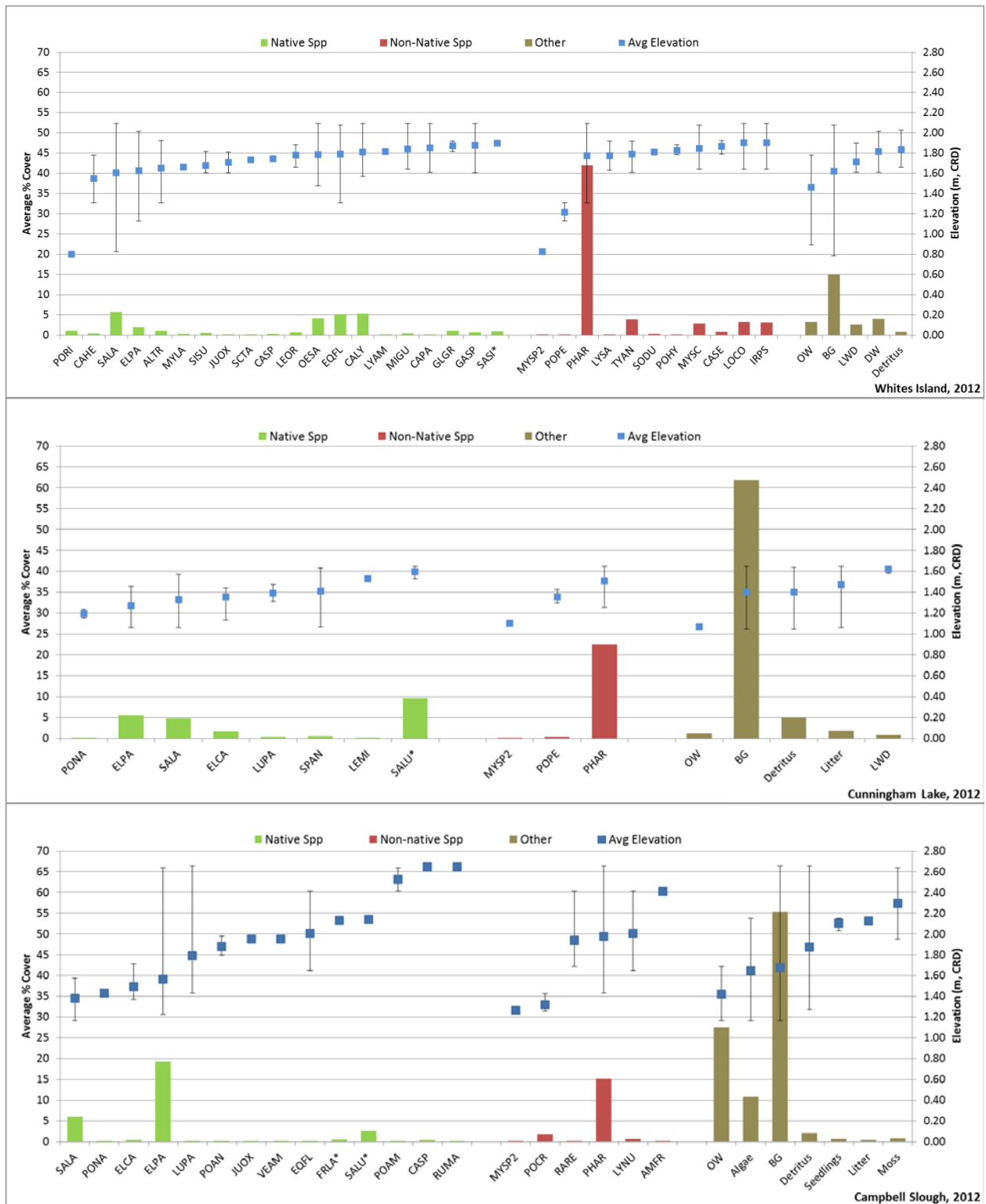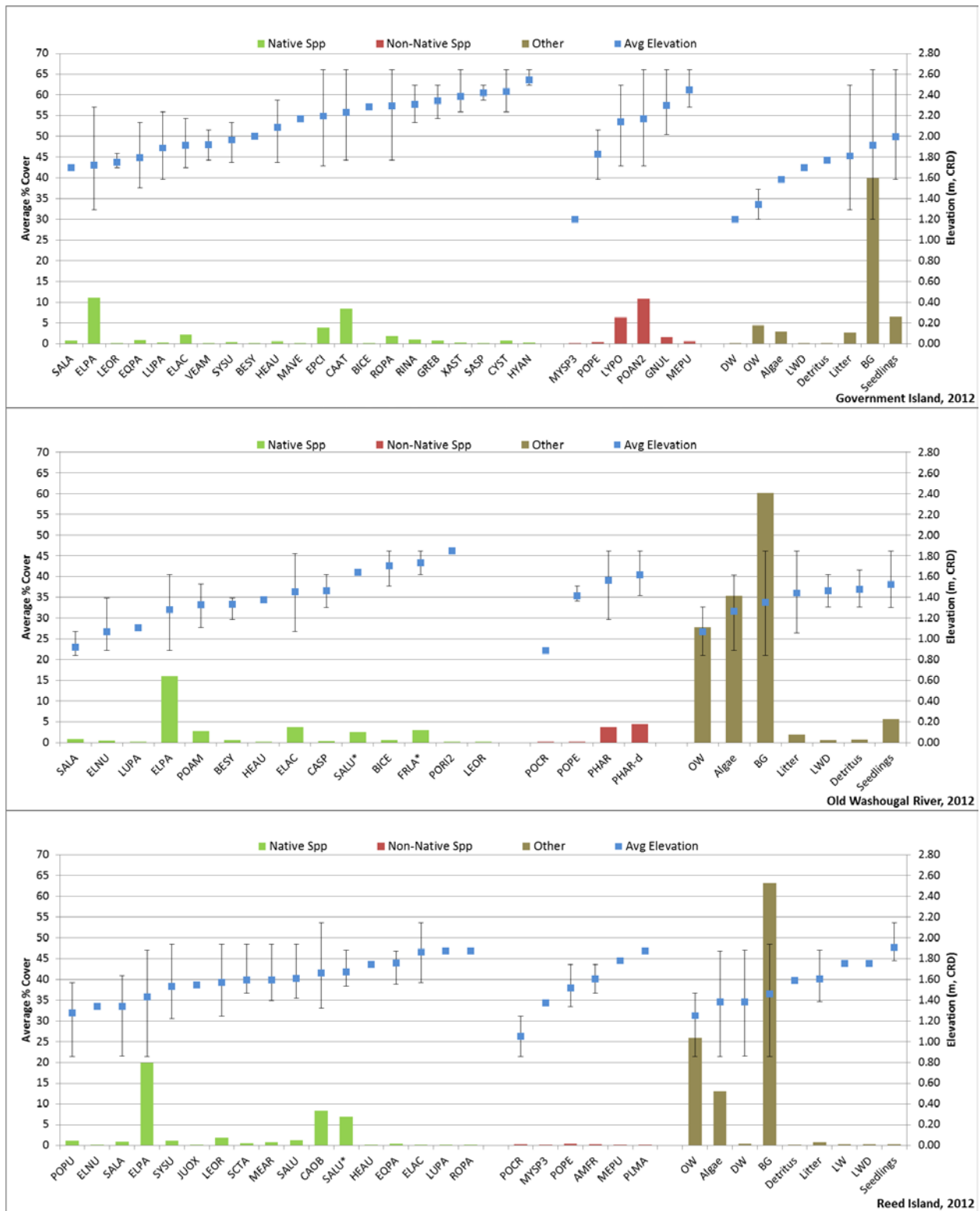
Figure 46. Continued.

Figure 46. Continued.

Franz Lake, 2012
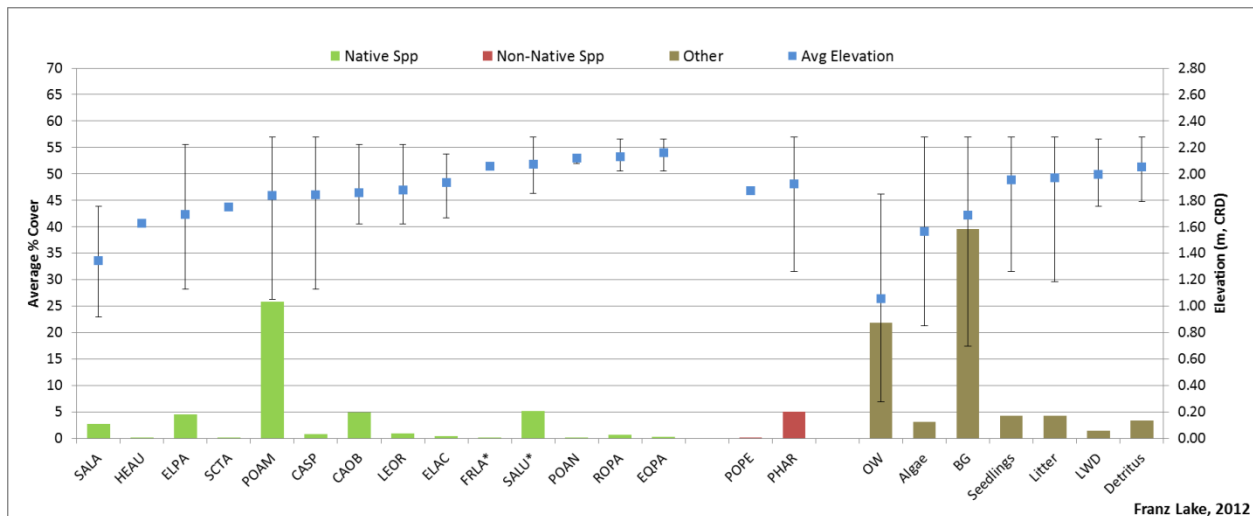
Figure 46. Continued.

4.3.1.6 *Biomass*

The biomass sample design is such that samples are collected in the summer at peak biomass and then again in winter, just prior to the initiation of primary production in the next growing season. In this way, the annual primary production can be estimated as well as the potential amount of biomass, as macro-detritus, exported from the wetland (Figure 47). The timing of vegetation growth and dieback is variable between years and is dependent on the timing of seasonal flooding (spring freshet and winter storms) and winter conditions. Dieback primarily occurs during winter, with the timing of the breakdown of the standing dead depending on temperature and frequency of winter flood events. A portion of the dead plant material is exported from the wetland as macro-detritus and a portion remains and is broken down in the wetland contributing organic matter to the sediment and macro-invertebrates in the wetland ecosystem (Figure 47).

The four trend sites were sampled for summer biomass in August 2011 and winter biomass in February 2012 (Table 17). The two lower estuary sites, Ilwaco and Whites Island, have the highest emergent and submerged aquatic vegetation (SAV) biomass. These results are likely representative of peak biomass at the Ilwaco site and possibly at Whites Island, however, at the latter site the total vegetation cover was slightly lower at the time of sampling in 2011 than it had been in the previous two years (107, 101, and 93 percent in 2009, 2010, and 2011 respectively). Due to high water in the two upriver sites the timing of peak biomass collection was delayed until late August, however, the mild fall weather likely delayed the actual peak season until late fall. The results at Campbell Slough and Franz Lake are indicative of the effect of the high water on biomass production, with lower summer productivity levels compared to the sites in the lower River (Table 17). The summer results, however, are likely not indicative of peak biomass production at the upriver sites due to the combined effects of the spring/summer high water and mild fall conditions. Primary productivity is assumed to have been higher in the falloff 2011 at Franz Lake, the most upriver site, and resulted in greater winter biomass than the stunted summer amounts even after winter dieback occurred. Signs of cow grazing were noted at the Campbell Slough site in the summer of 2011 and again in the winter sampling. This undoubtedly affected the results by reducing the amount of biomass in either or both seasons.
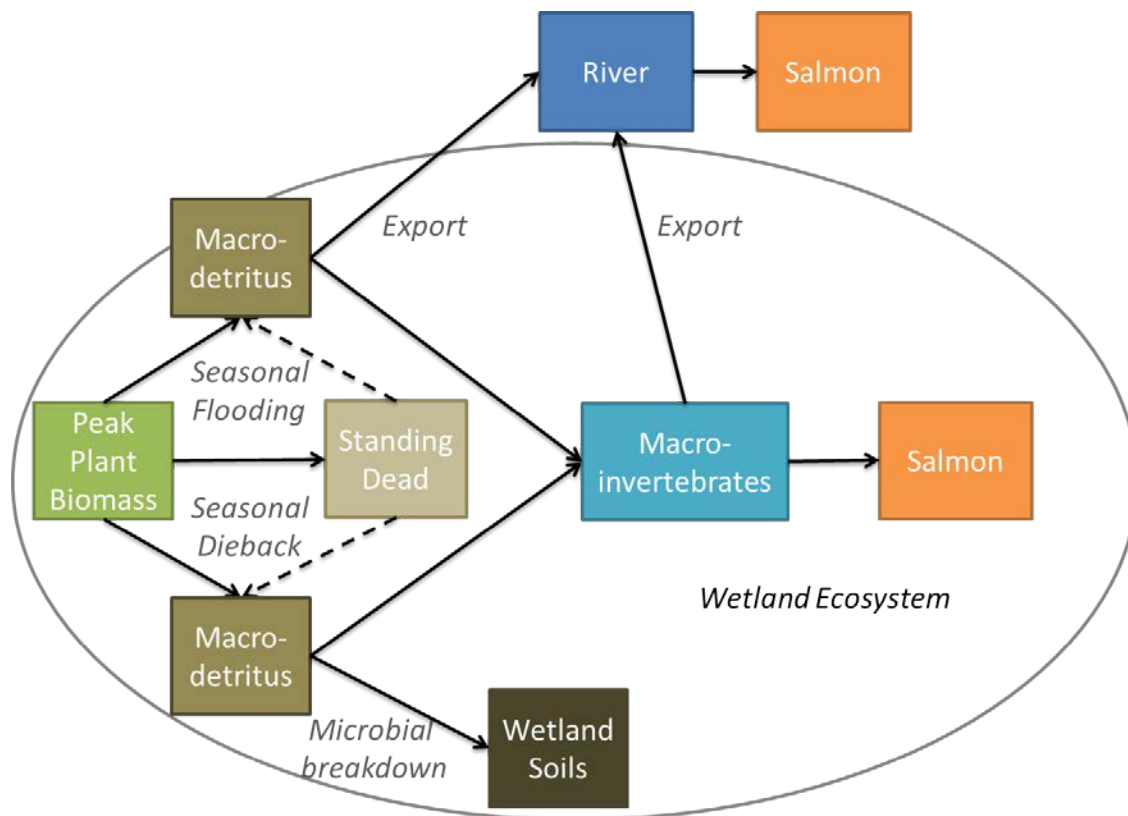
Figure 47. Conceptual food web model for emergent marshes in the LCRE.

Table 17. Emergent and submerged aquatic vegetation biomass per site for the Summer 2011 and Winter 2012 sampling periods.

Site	Vegetation Strata	Summer Peak Biomass (Dry Wt., g/m2)	Winter Biomass (Dry Wt., g/m2)	Export Potential (Dry Wt., g/m2)
Ilwaco (BBM)	Emergent	857.2	336.7	521
	Submerged	81.8	0.0	82
Whites Island (WHC)	Emergent	886.1	325.3	561
	Submerged	49.3	0.0	49
Campbell Slough (CS1)	Emergent	327.3	52.0	275
	Submerged	0.4	0.0	1
Franz Lake (FLM)	Emergent	203.2	234.9	NA1
	Submerged	ND2	ND	ND

1   Due to the high water and mild fall weather, winter biomass was greater than summer biomass at this site therefore export potential cannot be determined.

2   No SAV was observed or collected at the site due to high water.

The macro-detritus export potential from the four trend monitoring sites was calculated by subtracting the winter biomass from the summer peak biomass (Table 17), and is based on the following assumptions:

1) Most of the growing season primary productivity is captured in the single sampling of biomass at the end of the summer.

   Comparison of our data to that collected previously in the lower estuary with more intensive sampling (MacDonald 1984), indicates that this is a valid assumption in the lower estuary, however this does not appear to hold true in the upper estuary during high water years. In these years, there is likely some productivity that occurs prior to the high water that may die back during the spring freshet.

2) The amount of biomass that has died and broken off has all been exported from the wetland between the summer and winter sampling.

   This assumption is not completely true, as some of the biomass stays in the wetland and contributes to the organic material in the sediment (Mitsch and Gosselink 2000), however much of the export of macro-detritus occurs during winter storms and floods (Thom et al. 2012). In addition, some of the vegetative material remains after the winter period as standing dead; this material may provide more of a contribution to the "local" wetland soils because it is not flushed out during the winter storms.

Other studies have used a similar method to estimate potential biomass export from tidal wetlands (MacDonald 1984; Thom et al. 2012) and while there are limitations, especially in anomalous years, the method provides a relatively efficient means of estimating primary productivity and potential export at multiple sites throughout the estuary.

In 2011, the primary productivity and the potential macro-detritus export were considerably higher in the lower River sites compared to the upper River sites. We hypothesize that the upriver sites were affected by the high water in 2011; additional biomass sampling is needed to test this hypothesis and provide data on the productivity of the upriver marshes (above rkm 72). The potential biomass export estimated at the four trend sites begins to provide data that can be used to determine current levels of macro-detritus export from existing wetlands in the LCRE. These estimates can also be used to calculate export potential from restoration actions (Diefenderfer et al. 2012).

4.3.1.7 *Elevation, Inundation, and Vegetation Interactions*

Average elevations of the 2012 vegetation sampling areas, relative to the CRD, and their location in the River are provided in Figure 48 and Table 18. Data for 2011 is also included in Table 18 and in Figure 49 because we were not able to collect water surface elevation (WSE) data for three 2011 sites until 2012 (Burke, Goat, and Deer Islands); their elevations are provided here as are the inundation calculations and the WSE graphs in Appendix B.

The overall range in elevations at the 2012 monitoring sites is 1.83 m, with the lowest elevation occurring at Whites Island (0.83 m, CRD) and the highest at Campbell Slough (2.66 m, CRD). The highest average elevation is at the Secret River high marsh site (2.06 m, CRD), closely followed by the Ilwaco (2.05 m, CRD) and Government/Lemon Island (1.95 m, CRD) sites. The lowest average elevation is at the Secret River low marsh site (1.02 m, CRD). In the lower part of the estuary, below Whites Island (rkm 72), the large size of the sites makes it difficult to sample the entire elevation gradient therefore we sampled two separate plots at Secret River to capture the lower marsh area in this Zone. At Whites

Island and higher in the River, we capture the entire elevation gradient of the emergent marsh vegetation in our sampling, except perhaps the highest elevations of reed canary grass.

Figure 48. Average elevation of vegetation survey area of each site (points) with the range of elevations measured in the vegetated survey area (green bars).

Several patterns are notable in the frequency of inundation results (Figure 49). First, percent of time sites are inundated increases with rkm. Second, there is a shift in the timing of high inundation frequency between lower and upper river sites. In the lower river sites (Burke Island and lower) a higher percentage of the inundation occurs in the winter (Nov-March) whereas above this a higher percentage occurs in the growing season (April-October).
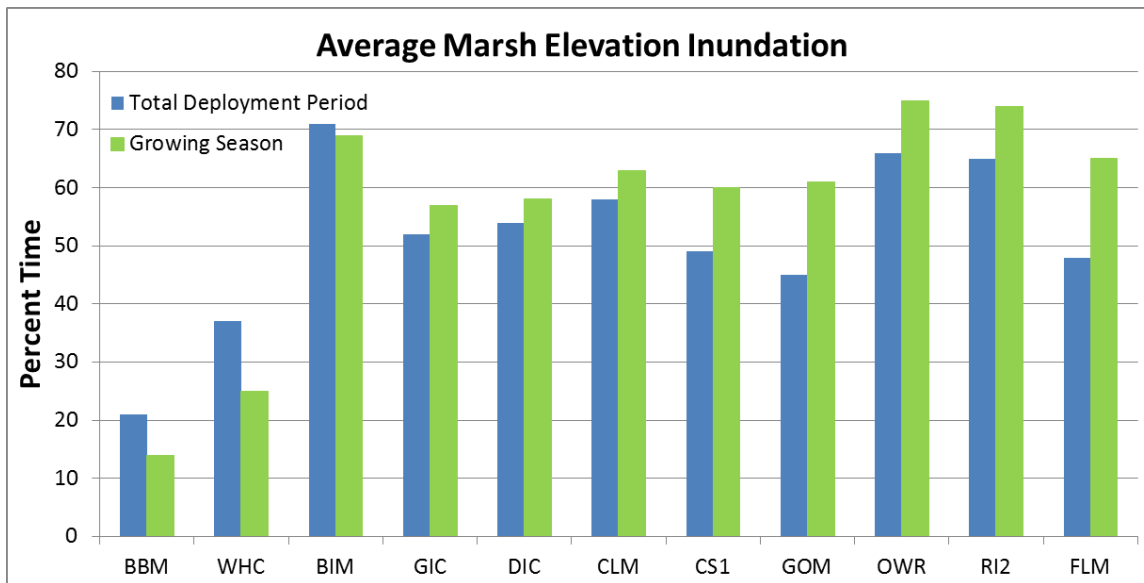
Figure 49. Percent of time the average marsh elevation was inundated at each site in 2012 during the deployment period and during the growing season.

The SEVs measured for 2012 range from a low at the Ilwaco site of 139 m-hours/growing season to a high at the Reed Island site of 4344 m-hours/growing season. The values are primarily related location in the river and the hydrologic drivers: tidal versus fluvial. Elevation is also a secondary variable that

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controls SEV, with higher elevations resulting in lower SEVs. This variation is apparent when the SEV at the Reed Island site (SEV 4344; elevation 1.48 m, CRD) is compared to the SEV at the Franz Lake site (SEV 3679; elevation 1.81 m, CRD). The latter has a lower inundation even though it is located 16 km farther up the estuary because the elevation is higher. Similarly, the low elevation at the Secret River low marsh site results in a much higher inundation value than at the high marsh sites in the same zone.

Table 18. Average elevation of vegetation survey areas (relative to CRD) and inundation as measured by the 2012 Sum Exceedance Value (SEV) calculated at the average marsh elevations.

Reach	Site	Site Code	Rkm	Average Elevation (m, CRD)	Growing Season SEV (m-hours)
A	Ilwaco	BBM	6	2.05	139
B	Secret River – Low	SRM-L	37	1.02	1356*
	Secret River – High	SRM-H	37	2.06	215*
	Welch Island	WI2	53	1.57	ND
C	Whites Island	WHC	72	1.72	440
E	Burke Island	BIM	131	1.18	2286
	Goat Island	GIC	131	1.57	1742
	Deer Island	DIC	132	1.51	1826
F	Cunningham Lake	CLM	145	1.40	2225
	Campbell Slough	CS1	149	1.68	2052
G	Government/Lemon Island	GOM	181	1.95	2533
	Washougal River	OWR	195	1.35	3832
	Reed Island	RI2	204	1.48	4344
H	Franz Lake	FLM	221	1.81	3679

*SEVs for Secret River are calculated using WSE data collected in 2008 because the hydrology data for 2012 was not yet available at the time of this reporting. Due to the location of SRM in the lower estuary, these values are not expected to differ greatly year-to-year.

Regardless of where sites are located in the LCRE, the relationship between inundation (as measured by SEV) and total vegetative cover holds true. That is, with increasing SEV cover declines (Figure 11). Importantly, water level, especially in extreme water years can produce a measurable response in vegetation cover. Because vegetation cover is a function of both the biomass and the species present, water level contributes significantly to both productivity and biodiversity. Understanding this dynamism is critical to interpreting between year variation at trend sites, and within year variation among sites. In addition, the overall control of water level by regional climatic conditions and flow manipulation at the dam contributes to these sources of variation in the ecosystem.
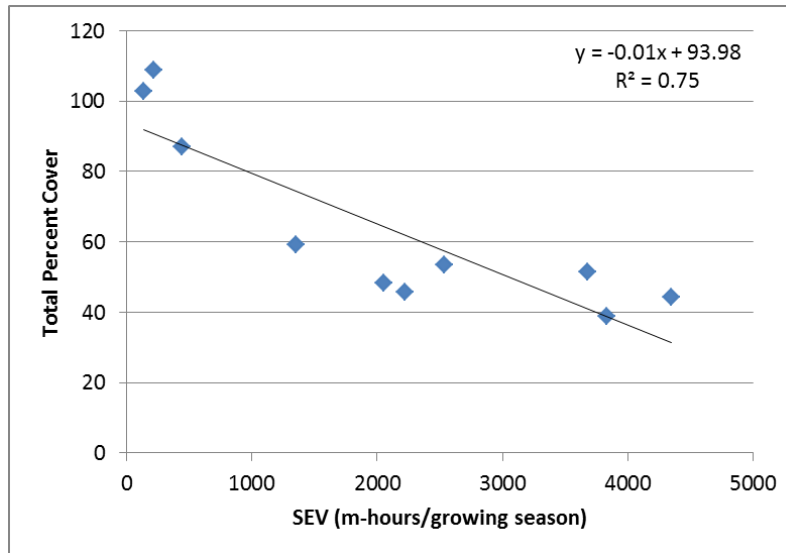
Figure 50. Sum exceedance value (SEV) calculated at the average site elevation compared to the total percent emergent vegetated cover at the sites in 2012.

4.3.1.8 *Channel Morphology and Inundation*

The channel cross sections provide a means of evaluating hydrologic connectivity of a site to the main channel of the River. The channel morphology along with the hydrology dictates the amount of time that the channel, the banks, and the smaller marsh channels are accessible for fish for rearing, protection, and feeding. Channel cross-sectional morphologies are shown in Figure 51 and channel morphometrics and inundation times are provided in Table 19. Channel features that can be discerned from channel cross sections are sills or barriers, elevation gain (channel slope), channel area, and channel depth. All these features are directly related to the hydrologic connectivity of the site as indicated by the inundation times calculated for the channel thalweg (deepest point of the cross section) and the channel banks.

In some cases, a cross-section is shallower than the next cross-section upstream. This sill limits hydrologic connectivity during low water. Examples of sills at the channel mouth are at the Ilwaco (XS1) and Secret River sites (XS1; Figure 51), although these sills are small, approximately 20 cm and 30 cm respectively, and do not effect connectivity greatly. For example, at the Ilwaco site the mouth thalweg has a 50 cm water depth 9 percent less of the time than the next cross section up (Table 19). However, at Reed Island the sill is approximately 77 cm higher than the next cross section up and the connectivity is limited 31 percent of the time for the year as a whole, but only 1 percent during the high water period of March to July. The Campbell Slough and the Washougal River sites have rip rap weirs across the mouths which restricts access once the WSE of the Columbia River is below 0.89 m CRD and 0.94 m CRD, respectively (Table 19; the weir at Campbell Slough was surveyed in 2010 and is discussed in Sagar et al. 2011). The weir at the Washougal River site was approximately 15 cm high, and consequently only reduces connectivity 6 percent during the year as a whole and 2 percent during the peak fish migration period.

Generally the slope of tidal and riverine channels at our sites are not steep, however as a channel reaches the upper extent elevation gain may limit hydrologic connectivity. For example, at the Government/Lemon Island site, the thalweg of the highest cross section is 2.81 m higher than the thalweg at the mouth and as a result it is inundated only 20 percent of the year and 46 percent of the

period from March to July compared to 91 percent and 100 percent, respectively at the mouth (Table 19). Other sites reported here have a lower slope than the Government/Lemon Island site, with elevation changes ranging from less than 0 m at the Secret River site to 1.1 m at the Welch Island site.

The width to depth ration is a means of evaluating channel morphology and is a means of comparing morphologies between sites. The higher the number, the shallower and wider the channel is. At most sites the ratio is larger at the mouth and decreases with distance from the mouth as the channel gets smaller and narrower (Table 19). Exceptions to this pattern are sites that have a small channel mouth that opens into a broad shallow water area such as the Burke Island, Goat Island, and Washougal River sites. In general channels at sites in the lower estuary (below rkm 60) have lower width to depth ratios (<20), with narrower and deeper channels. Tidal hydrology may be a driver in this morphology in the lower River; however similar ratios are also observed in the Franz Lake channel.

At most sites, bank elevation increased with distance from the mouth as the overall channel increased in elevation. The Franz Lake site had a higher bank elevation at the mouth than other sites in this year's analysis (Table 19), although the bank at the Burke Island site may not have been accurately measured due to extremely thick underbrush.

Inundation frequency of the channel thalweg (plus 50 cm of water) and bank (plus 10 cm of water) was calculated as a means of estimating the potential for juvenile salmonid access to the channel and to the marsh edge (at the bank) (Table 19). Patterns of channel inundation are related to the overall hydrologic patterns of the LCRE, with lower inundation frequencies observed in the tidally dominated portion of the River and higher inundation frequencies in the fluvial dominated part of the River. The frequencies were calculated for the entire year and also for the period of peak Chinook salmon migration (March – July). The latter also coincides with the timing of the spring freshet resulting in much higher inundation frequencies during this period. In the lower river, during March – July of 2012, the channel thalweg was accessible 47-60 percent of the time. Previous studies at the Secret River site in 2008 indicate that the channel was accessible 74 percent of the time between March and July (Sagar et al. 2013). Whereas, the upriver channels starting at Whites Island, were accessible 93 percent of the time or greater during this period.

The banks of the channels, with at least 10 cm of water, were accessible less of the time than the channel thalwegs. In the lower river, channel banks during the period from March to July were accessible 9-39 percent of the time (Table 19). At the Secret River site in 2008, the channel bank was accessible 47 percent of the time between March and July (Sager et al. 2013). At Whites Island, the frequency increases to 76 percent and in the upper River it is greater than 92 percent of the time except at very high elevation sites.
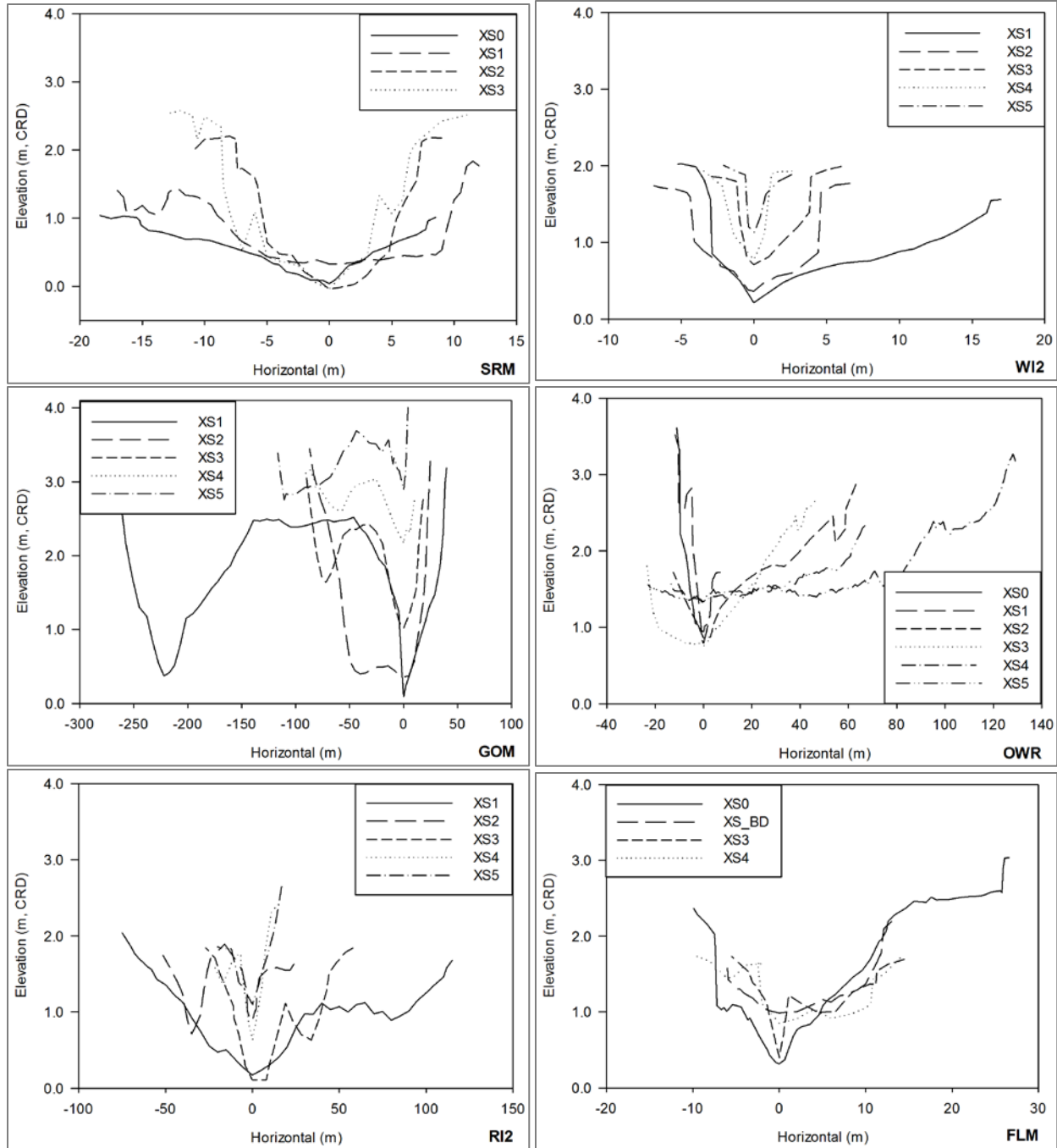
Figure 51. Elevations of channel cross sections for the 2012 sites where multiple cross sections were measured. Cross sections are numbered starting at the mouth. Note the different scale of the x-axes for each of the plots. The trend sites, where cross sections have been measured in multiple years, are plotted elsewhere.

Table 19. Physical channel metrics measured at each site; the year of measurement is provided in parentheses after the site code. Inundation time percentages for 1 year (Fall 2011 - Fall 2012) and between 1 March and 31 July, 2012 (the peak juvenile Chinook salmon migration period). Cross sections are numbered starting at the mouth.

Site (year)	Cross Section	Thalweg Elevation (m, CRD)	Bank Elevation (m, CRD)	Channel Depth (m)	Cross Section Area (m2)	Channel Width (m)	Width:Depth Ratio	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm
		Physical Metrics						**Inundation**			
								Year		**March-July**	
BBM (11)	1	0.90	1.59	0.68	3.5	6.3	9.3	50%	37%	51%	39%
	2	0.70	1.86	1.16	8.9	9.3	8.0	59%	25%	60%	27%
	3	0.90	2.12	1.22	9.7	10.1	8.3	50%	14%	51%	14%
	4	1.01	2.00	0.99	4.3	5.2	5.2	45%	19%	47%	20%
	5	1.17	2.26	1.09	1.6	2.7	2.5	38%	9%	40%	9%
BBM (12)	1	0.93	1.70	0.77	4.3	6.8	8.8	48%	32%	50%	34%
SRM (12)	0	0.04	1.01	0.98	11.9	24.0	24.6	ND*	ND	ND	ND
	1	0.32	1.42	1.09	19.3	22.6	20.6	ND	ND	ND	ND
	2	-0.04	2.13	2.17	22.5	14.9	6.9	ND	ND	ND	ND
	3	-0.03	1.98	2.01	20.7	15.1	7.5	ND	ND	ND	ND
WI2 (12)	1	0.21	1.55	1.34	14.9	19.3	14.4	ND	ND	ND	ND
	2	0.36	1.65	1.29	8.8	9.2	7.1	ND	ND	ND	ND
	3	0.71	1.80	1.09	4.0	5.1	4.7	ND	ND	ND	ND
	4	0.78	1.74	0.96	2.1	3.3	3.4	ND	ND	ND	ND
	5	1.31	1.62	0.31	0.4	1.3	4.3	ND	ND	ND	ND
WHC (12)	1	0.37	1.15	0.78	14.9	36.5	47.1	79%	61%	94%	76%
BIM (11)	1	0.09	0.53	0.44	4.7	17.7	40.1	97%	97%	100%	100%
	2	0.46	1.89	1.43	32.6	28.7	20.0	82%	38%	100%	79%
	3	0.74	1.22	0.48	18.0	77.7	162.2	67%	63%	97%	96%
	4	0.98	1.78	0.80	67.7	104.4	130.2	55%	41%	94%	84%
	5	1.05	1.75	0.70	56.7	95.2	135.6	53%	42%	93%	86%
DIC (11)	0	0.66	1.84	1.18	113.8	68.6	58.1	72%	42%	98%	87%
	1	0.43	2.15	1.72	21.5	15.8	9.2	83%	33%	100%	72%

Site (year)	Cross Section	Physical Metrics						Inundation			
								Year		March-July	
		Thalweg Elevation (m, CRD)	Bank Elevation (m, CRD)	Channel Depth (m)	Cross Section Area (m2)	Channel Width (m)	Width:Depth Ratio	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm
	2	0.63	2.64	2.01	47.0	30.7	15.3	73%	16%	98%	35%
	3	0.79	2.18	1.39	37.2	29.6	21.3	65%	32%	96%	70%
	4	0.48	2.28	1.80	40.3	27.4	15.2	80%	29%	99%	64%
	5	0.51	2.43	1.92	56.7	34.1	17.8	79%	23%	99%	51%
	1	0.11	0.64	0.53	17.4	51.6	97.7	100%	91%	100%	100%
GIC	2	-0.37	0.84	1.21	42.5	83.0	68.6	100%	82%	100%	100%
(11)	3	-0.31	2.11	2.42	76.1	53.5	22.1	100%	34%	100%	75%
	4	0.20	2.29	2.09	197.4	123.8	59.2	94%	29%	100%	63%
CLM (12)	1	0.71	1.13	0.42	4.2	17.6	42.4	70%	69%	98%	98%
CS1 (12)	1	0.77	1.33	0.56	8.2	21.1	37.7	67%	58%	98%	96%
	1	0.09	2.97	2.88	362.7	302.4	104.8	91%	29%	100%	68%
GOM	2	0.36	3.28	2.92	236.8	109.9	37.6	82%	20%	100%	46%
(12)	3	1.02	2.58	1.56	66.7	102.4	65.7	54%	38%	95%	85%
	4	2.17	3.00	0.83	25.2	80.2	96.6	38%	28%	85%	65%
	5	2.90	4.01	1.12	95.7	119.3	107.0	20%	6%	46%	13%
	0	0.94	1.04	0.10	0.2	3.9	39.0	63%	75%	97%	100%
	1	0.79	1.63	0.84	3.4	6.7	8.0	69%	52%	99%	94%
OWR	2	0.86	1.72	0.86	13.6	34.5	40.1	66%	50%	99%	93%
(12)	3	0.76	1.85	1.10	36.2	48.8	44.5	71%	48%	99%	93%
	4	1.34	1.60	0.27	11.1	80.3	298.5	50%	53%	93%	95%
	5	1.31	1.56	0.25	6.7	57.3	231.8	50%	55%	93%	95%
	1	0.17	1.68	1.51	136.7	179.0	118.5	100%	56%	100%	95%
RI2	2	0.88	1.59	0.71	5.0	20.8	29.2	69%	59%	99%	96%
(12)	3	0.11	1.84	1.73	73.5	84.8	49.0	100%	52%	100%	93%
	4	0.64	1.77	1.13	12.2	13.9	12.3	78%	54%	100%	94%

Site (year)	Cross Section	Physical Metrics						Inundation			
								Year		March-July	
		Thalweg Elevation (m, CRD)	Bank Elevation (m, CRD)	Channel Depth (m)	Cross Section Area (m2)	Channel Width (m)	Width:Depth Ratio	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm
	5	1.10	1.83	0.73	9.0	21.9	30.0	61%	53%	97%	94%
FLM (12)	0	0.31	2.03	1.72	19.2	19.7	11.5	100%	44%	100%	92%
	BD	0.99	1.58	0.60	7.0	17.0	28.6	58%	51%	96%	94%
	3	0.40	1.39	0.99	4.2	14.3	14.4	100%	58%	100%	96%
	4	0.85	1.45	0.60	6.2	13.2	22.0	87%	54%	100%	95%

*No Data. Water level data were not yet available at the time of reporting.

4.3.2 Trends Analysis

A temporal analysis of vegetation species composition, vegetation cover, and inundation patterns was conducted at four of the trend sites as follows:

Whites Island (rkm 72)   2009-2012
Cunningham Lake (rkm 145)   2005-2012
Campbell Slough (rkm 149)   2005-2012
Franz Lake (rkm 221)   2008-2009, 2011-2012

In addition, we compared the accretion rates and channel morphology at the four sites for the years monitored between 2008 and 2011.

4.3.2.1 *Hydrology Patterns*

The Columbia River basin is primarily a snow-melt runoff watershed and as such is subject to inter-annual hydrologic variability. Figure 52 shows the variation in the timing and magnitude of outflow at Cascade Island, just below Bonneville dam for the years 2005-2012 and is provided here to give an overview of the hydrological patterns for the estuary during this study. Temporal variability in water level was dramatic between years. Although the greatest water level was centered on June, peaks varied in duration and number: there were two peaks in 2006 and 2012. In general, 2006, 2008, 2011, and 2012 were high flow years, 2005 and 2007 were lower flow years, and 2009 and 2010 average flow years as compared to the 10-year average flow (2000-2009). Note however, that although 2012 was a high water year, the magnitude was not as great as 2011, instead the duration was much longer. Inter-annual hydrologic variability is much less pronounced in the lower estuary where the floodplain expands and tidal flows are the primary driver of hydrologic patterns.

Interestingly, similar SEVs occurs at the average elevations of the Cunningham Lake and the Campbell Slough sites. The average elevation of Campbell Slough is 0.28 m higher; however, the inundation is also higher at the Campbell Slough site due to the location of the slough mouths, at rkm 142 and 149 rkm, respectively, which results in hydrologic differences between the sites. The hydrologic regime at the Cunningham Lake site is more similar to that at the St. Helens tide station (rkm 139) whereas the Campbell Slough site is more similar to the Vancouver station (rkm 169).
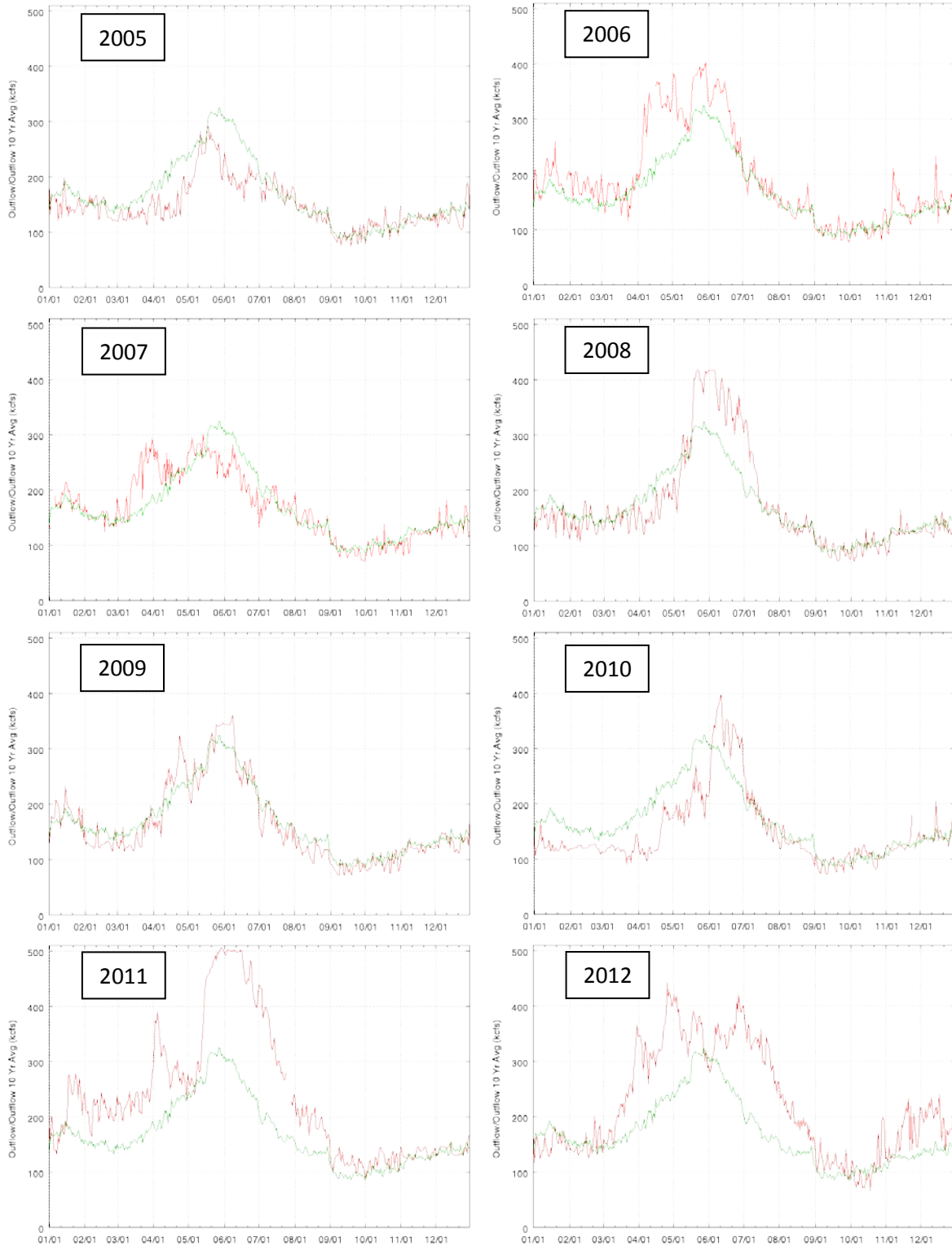
Figure 52. Daily mean outflow (red line) at Bonneville dam compared to the 10-year average (2000-2009; green line) for the years 2005-2012. (Columbia River Data Access in Real Time (DART) 2013).
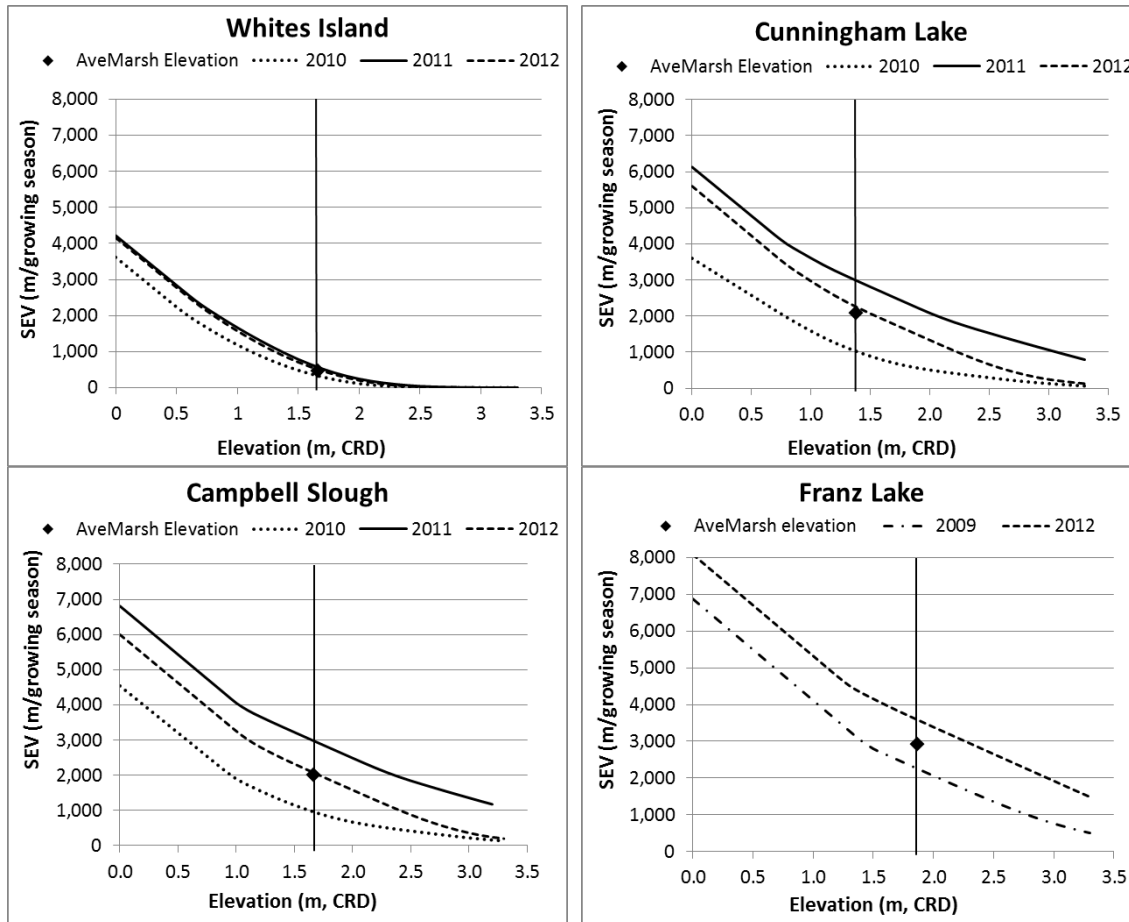
Figure 53. Growing season sum exceedance values (SEVs) for the trend monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the LCRE; the vertical line represents the average elevation at each site.

4.3.2.2 *Sediment Accretion Rate*

Sediment accretion rates at the four trend sites vary between -1.2 cm/year (erosion) and 3.0 cm/year (accretion) (Table 20).

Table 20. Sediment accretion rates at the trend sites between 2008 and 2011.

	BBM	SRM	WHC	CLM	CS1	FLM
	Rate (cm/year)					
08-09	ND1	0.2	-1.2	ND	ND	0.5
09-10	ND	2.8	1.0	1.9	0.4	ND
10-11	ND	0.9	0.1	1.6	1.7	3.0
11-12	0.1	ND2	0.9	1.4	0.9	-0.4

1 ND No data.

2 The north accretion stake originally deployed in 2008 appears to have sunk 6 cm between 2011 and 2012, therefore measurement not valid.

4.3.2.3　*Vegetation and Elevation*

Vegetation cover and the number of species present vary at the trend sites over time. Figure 54 shows the average percent cover of the dominant species, the number of species, and the variation between years. The Ilwaco (Baker Bay) site is the farthest down-river site and therefore the least affected by hydrologic variability. The Whites Island site is the most diverse, with the number of species ranging from 31 to 38 between years. Overall cover at the Whites Island site decreased by 14 percent while *Phalaris arundinacea* increased by 10 percent over the initial three year period then decreased in 2012 perhaps due to the combined effects of two high water years.

The trends at Cunningham Lake and Campbell Slough sites are similar to each other, where lower cover occurred during the higher water years (2006, 2008, 2011, and 2012). The Cunningham Lake site had lower cover in 2011 and 2012 than the Campbell Slough site perhaps due to the higher elevation of the latter site being exposed earlier than the Cunningham Lake site. The trend is similar at FLM, with lower cover in the two high water years. Most notable at this site is the shift in dominant species from *P. arundinacea* to *Polygonum amphibium*. To further evaluate the effect of hydrology on vegetation cover, we compared the vegetation to inundation in each year as described below in Section 4.3.2.4.
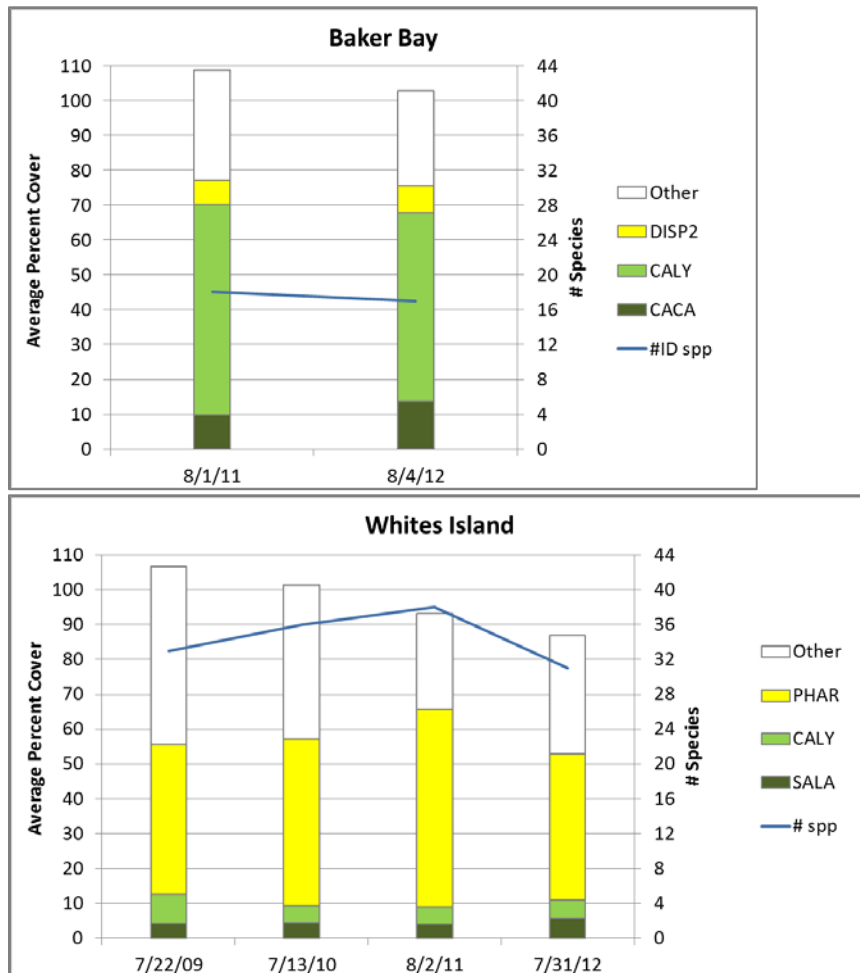
Figure 54. Average percent cover and number of identified species at the trend sites for all years monitored.
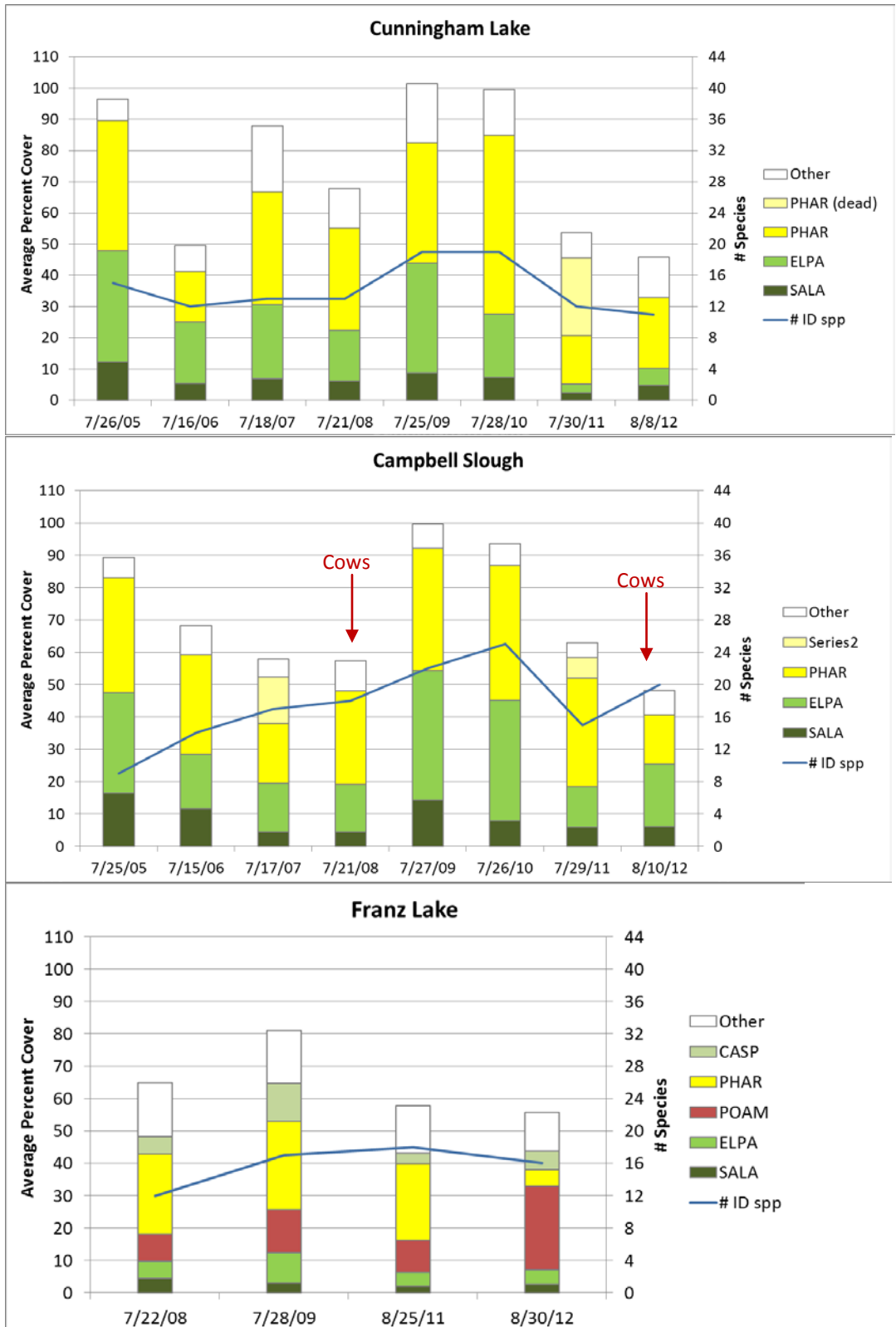
Figure 54. (Continued).

Overall, the vegetation composition at the four sites was similar in all monitoring years; however, a weighted similarity analysis comparing all years at each site shows there to be some variability between years. One way to evaluate similarity over time is to compare the original year of monitoring to the following years. The Ilwaco site was 88.7 percent similar between 2011 and 2012; the highest similarity of any site. The Secret River high marsh site in 2012 was 75.0 percent similar to 2008. The low marsh at the same site was only 62.6 percent similar between years. The similarity results for all the other trend sites are shown in Figure 55. The Whites Island site became less similar to its original monitored state (2009) over the initial three years and 2011and 2012 are the most similar to each other indicating that there is likely an effect of hydrology at this site. Longer term data will determine whether this trend of increasing dissimilarity will increase at this site. At the Cunningham Lake site, the trend over time was consistent in that the high water years (2008 and 2011) were the least similar to the low-flow initial survey year (2005), with 2011 the least similar to all other years. Low similarity between 2012 and previous years could be a result of the effects of two sequential high water years. At the Campbell Slough site, the trend is not as clear. In 2007, cows were present at the CS site resulting in grazing and trampling of some of the vegetation and as expected, this year is the least similar to the original year. Likewise, 2007 was most similar to the high water years at this site, which also cause a disturbance to the vegetation. Similarity at the Franz Lake site got less similar over time, with 2012 the least similar to all other years likely due to the shift in species dominance that occurred at that site (Figure 54).

WHC	2009	2010	2011
2010	77.6		
2011	68.1	78.2	
2012	69.4	73.0	81.1

CLM	2005	2006	2007	2008	2009	2010	2011
2006	74.7						
2007	72.6	69.7					
2008	68.2	71.4	83.2				
2009	72.1	63.9	80.9	75.9			
2010	72.5	71.5	77.3	76.9	79.0		
2011	46.8	59.3	50.3	55.8	51.2	54.1	
2012	56.9	68.6	64.4	71.4	58.1	66.4	65.8

CS1	2005	2006	2007	2008	2009	2010	2011
2006	75.8						
2007	56.9	58.7					
2008	68.4	71.3	70.7				
2009	76.1	64.6	56.6	68.2			
2010	72.4	62.7	61.9	67.2	82.1		
2011	68.8	68.5	73.4	73.0	65.3	66.6	
2012	70.0	70.6	64.9	72.7	66.0	69.1	71.0

FLM	2008	2009	2011
2009	70.6		
2011	67.0	70.6	

| 2012 | 56.5 | 62.0 | 62.7 |

Figure 55. Similarity analysis results between years for the four trend monitoring sites.

The upper and lower limits of the dominant species were evaluated in the previous trends analysis at CS1 and CLM. The results indicated that the elevation boundaries were stable, with little variation between years. To evaluate this finding further, we extended the analysis to include 2011 and 2012 (Figure 56). We compared the 2011 and 2012 elevations to the average elevation of the lower and upper boundaries for each species to determine if the hydrologic increase caused a change in the boundaries outside the range seen in the previous years. Variability in elevation boundaries occurred at the Whites Island site likely due to the greater heterogeneity of elevations at the site due to steep cut-banks. Slight increases in elevation occurred at some sites in 2011 and 2012. There was a slight increase in the lower elevations for all three species at the Cunningham Lake site in 2011, but the upper limits did not increase until 2012. Likewise, elevations did not increase at Campbell Slough until 2012. These results indicate that indeed the plants were likely affected by the higher water however the cumulative effects of two high water years was required to result in migration of elevation boundaries in 2012 . The variability associated with the lower elevation of *Sagittaria latifolia* (SALA) at the Franz Lake site is likely due to the increased inundation from the beaver dam from 2009 to 2012 compared to that in 2008 when *S. latifolia* was able to grow at a lower elevation. Additionally, the elevation boundaries all decreased in 2012 compared to 2011 perhaps due to the overall lower inundation in 2012 or/and because of competition from *P. amphibium* that became dominant at the site in 2012.

4.3.2.4 *Vegetation Cover and Inundation*

Three study sites were evaluated for interannual variability in vegetation cover and inundation: Cunningham Lake, Campbell Slough, and Franz Lake. Whites Island was not included in this analysis because the interannual hydrologic variability is not as large at this site as it is at the up-river sites (see Figure 53). Overall vegetation cover decreased with increased inundation as measured by the SEV. However, specific patterns and exceptions require closer inspection (Figure 57 and Figure 58.

The Cunningham Lake and Campbell Slough sites are comparable in many ways: 1) they are located in the same Zone of the LCRE at rkm 145 and 149, respectively; 2) they have the same dominant vegetation; 3) they are located a distance away from the main stem of the River; and 4) the average elevation of each site has similar SEVs in a given year (Figure 53). However there are some differences between the sites that may be resulting in differences in how inundation affects the vegetation. The distance to the main channel, along the slough, is much greater at Cunningham Lake (8.7 km) than at Campbell Slough (1.5 km). The elevations of the sites are also different, with the average elevation of Cunningham Lake 0.28 m lower than Campbell Slough. Some of the differences observed between the two sites maybe due to differences in the sites location and morphology. For example, at a given elevation (e.g, 1.4 m, CRD) inundation was higher at the Campbell Slough site (2476 m-hours/growing season) than it was at the Cunningham Lake site (2224 m-hours/growing season). The higher elevation and higher inundation results in similar vegetation communities, however the cover and elevation ranges of the species differs between sites depending on the conditions. At Campbell Slough, reed canary grass may not be affected by high water as much as the lower Cunningham Lake site because the higher elevation of this stratum (2.0 m, CRD) results in inundation levels lower than other sites (1583 m-hours/growing season in 2012 at Campbell Slough compared to 2389 m-hours/growing season at Cunningham Lake).
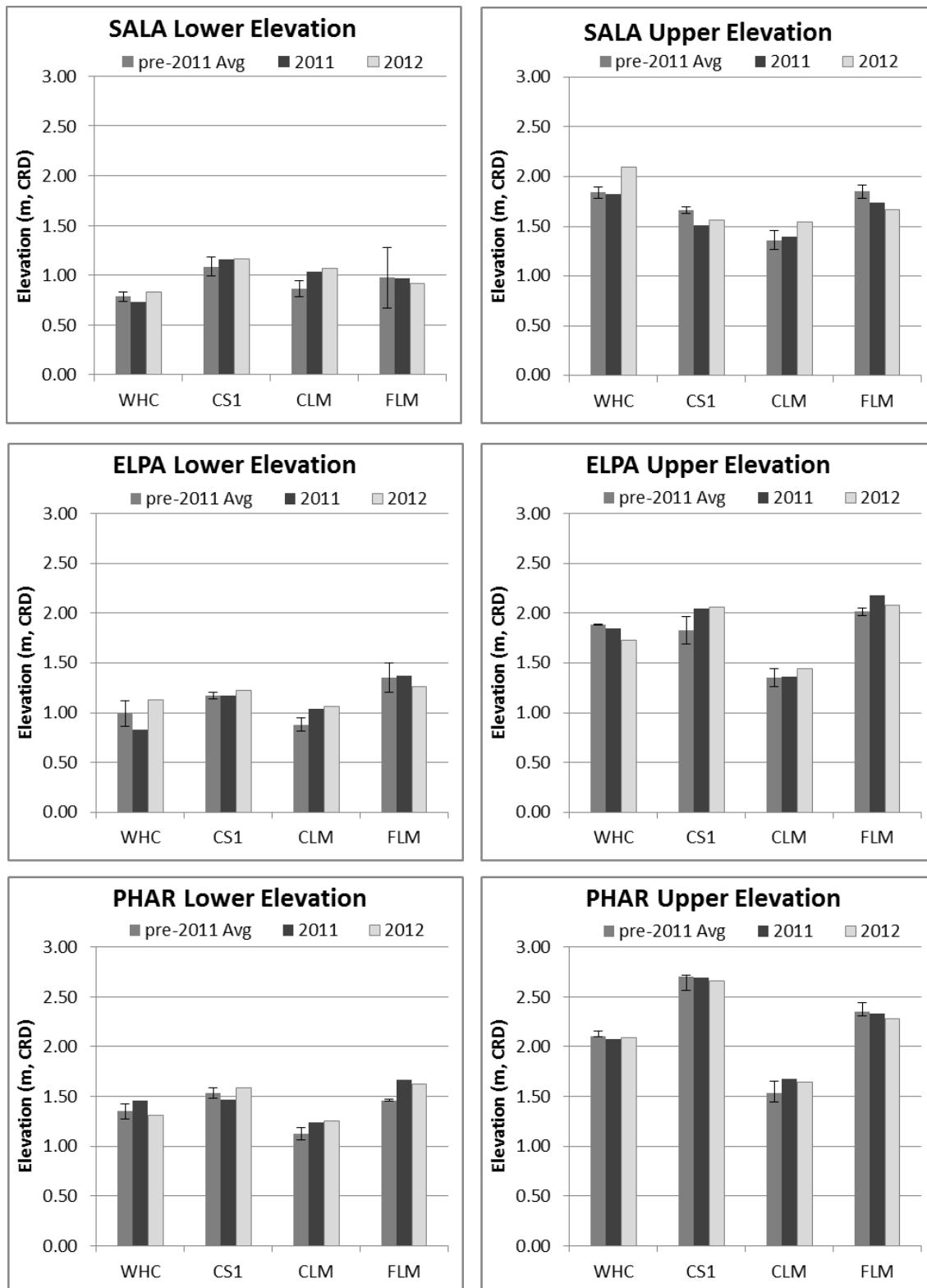
Figure 56. Lower and upper elevations for the dominant species at the trend monitoring sites. Error bars on the pre-2011 average represent ± 1 standard deviation. Elevations for years prior to 2011 are grouped to determine if elevation ranges differed in the following years.

The cover of *P. arundinacea* at Campbell Slough was higher than would have been expected in 2011 given the inundation levels that year (Figure 57). One possible explanation could be due to the higher elevation at this site compared to Cunningham Lake and the timing of the inundation relative to our sampling (e.g. areas of the strata exposed earlier at Campbell Slough). In contrast, *P. arundinacea* cover was lower than expected at the site in 2007 and 2012. The most likely explanation for this pattern is grazing by cows. In 2012, the water level had just recently receded and only the *P. arundinacea* strata had signs of grazing whereas in 2007 all strata were affected. These strata and years were not included in the regression analysis.
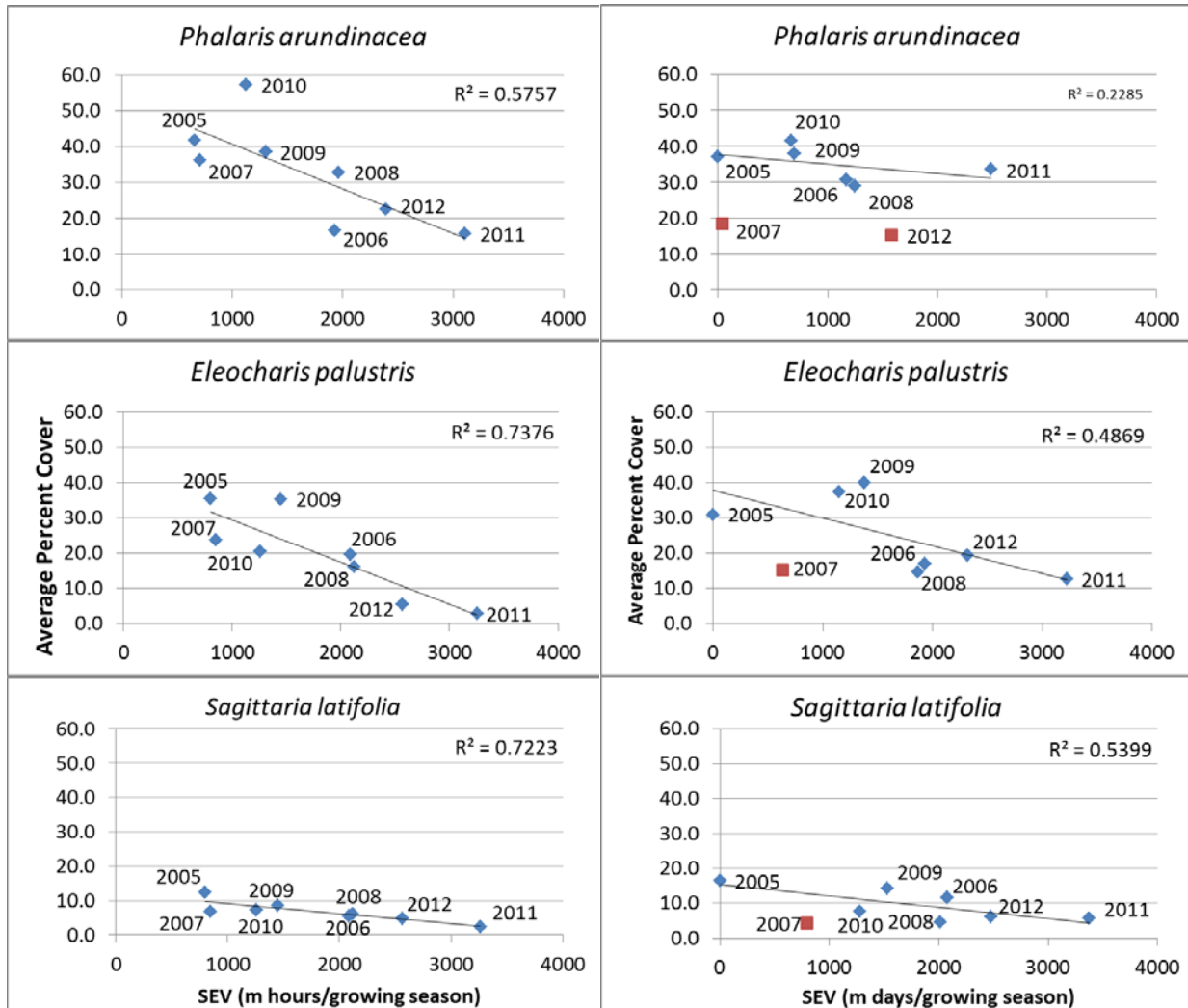
a) Cunningham Lake   b) Campbell Slough

Figure 57. Annual average percent vegetation cover of the dominant species at a) Cunningham Lake and b) Campbell Slough as related to annual growing season sum exceedance values (SEV) calculated at the average elevation for each species. The red square dots on the Campbell Slough plots represent years and strata where cow grazing was observed and were not included in the regression.

At Franz Lake, three of the species, *Carex* spp, *E. palustrus*, and *S. latifolia* followed the pattern of decreased cover with increasing inundation. However, the increase in *Polygonum amphibium* and decrease in *P. arundinacea* in 2012 resulted in a greater increase in the former and a greater decrease in the latter than would be expected. A decrease the cover of *S. latifolia* at was actually higher in one of

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the high inundation years (2008) compared to the lower inundation year (2009), which could possibly be attributed to the beaver activity observed at the site causing more ponding and inundation in subsequent years at the low elevations where *S. latifolia* grows.
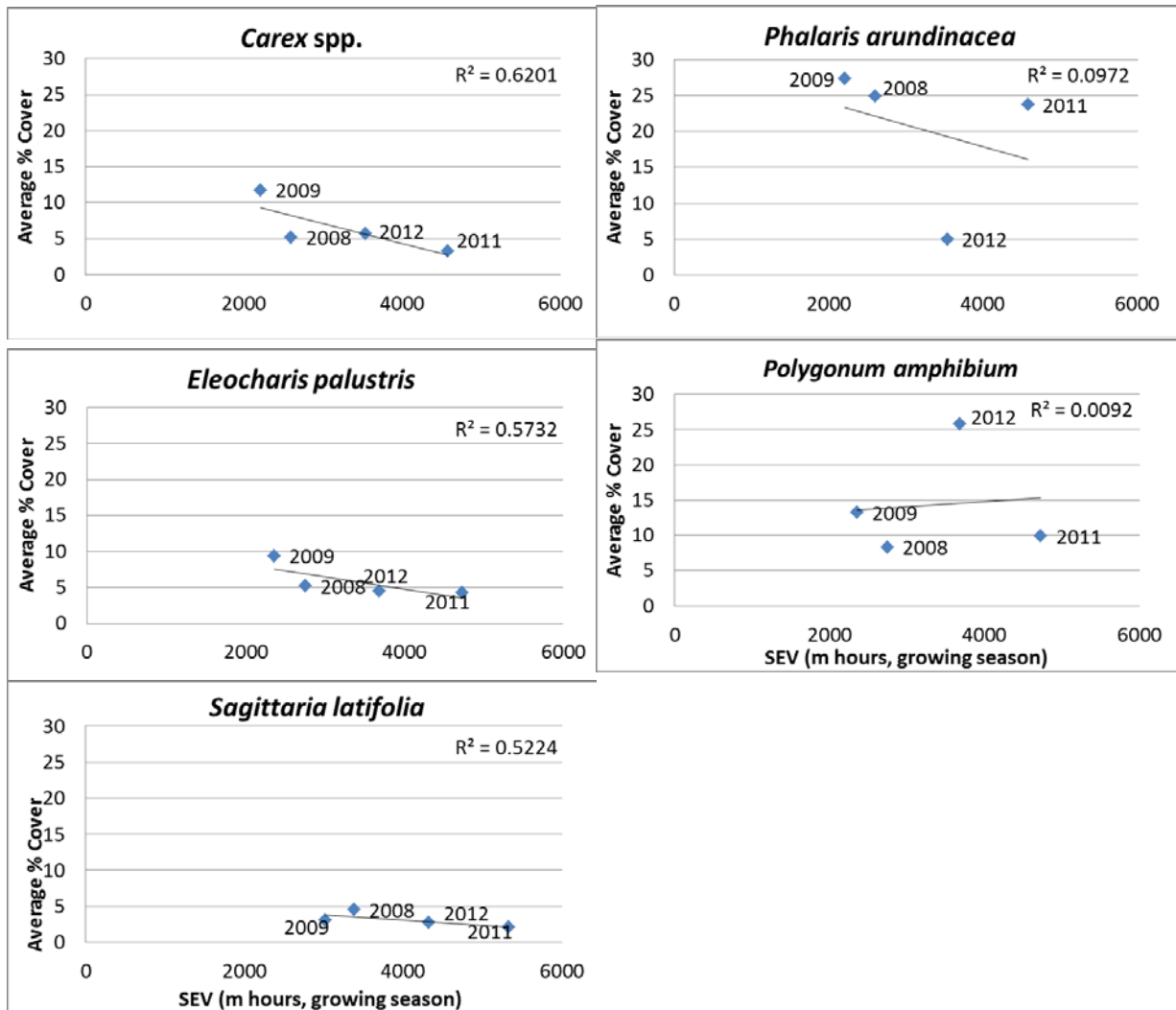
Figure 58. Annual average percent vegetation cover of the dominant species at Franz Lake as related to annual growing season sum exceedance values (SEV) calculated at the average elevation for each species.

4.3.2.5   *Channel Morphology*

Inter-annual variability of cross section morphology is low as seen by the comparison of single cross section locations from the repeat (trend) sites.

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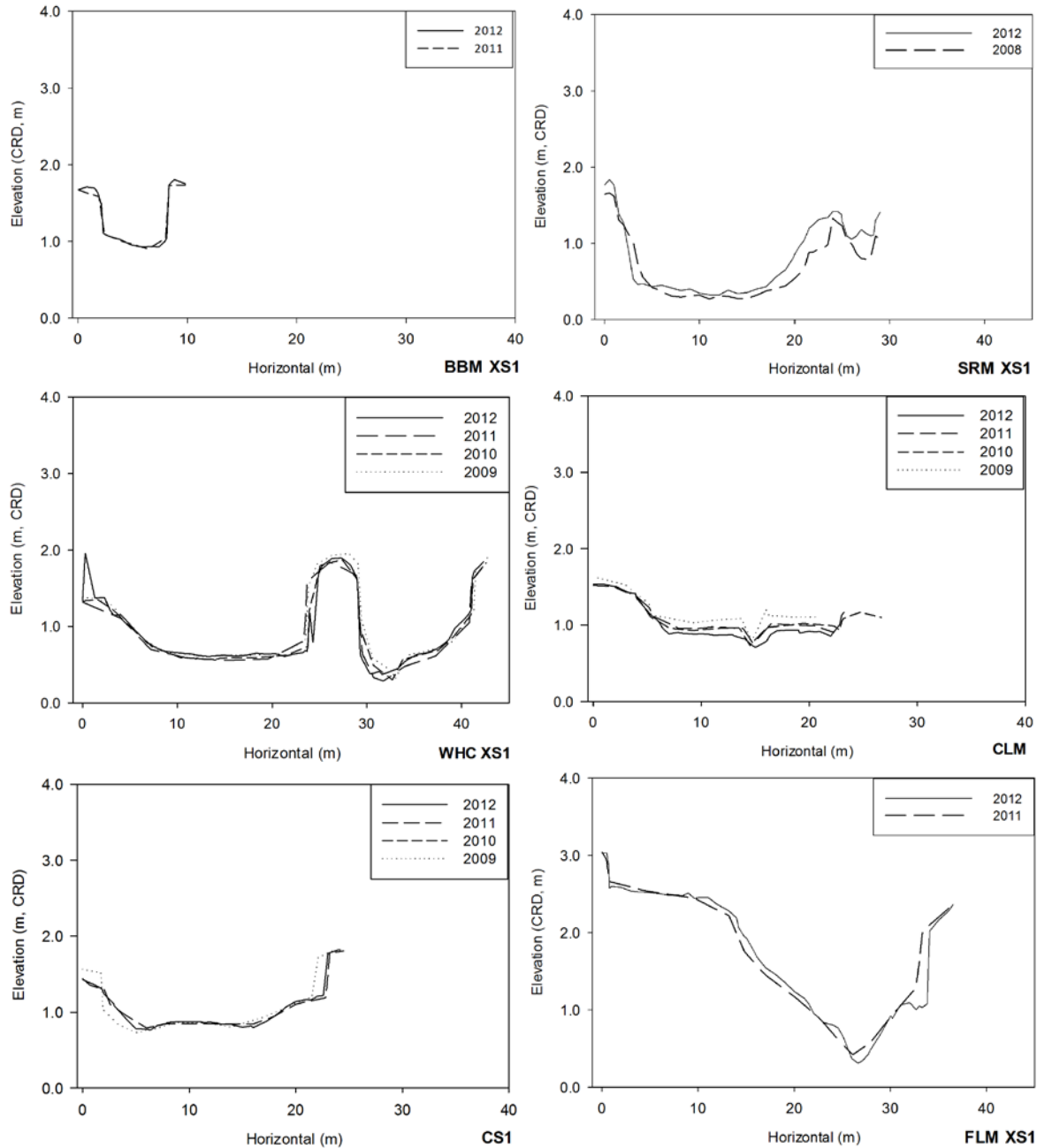
Figure 59. Elevations of the channel cross sections for the trend sites showing multiple years.

4.4 Food web

4.4.1 Phytoplankton and periphyton

4.4.1.1 *Abundance and diversity*

Ambient phytoplankton abundance as measured by chlorophyll *a* concentrations was generally higher at the upstream sites compared to downstream sites in 2012 (Figure 60). In April, there is a clear pattern of higher phytoplankton abundance upstream, whereas in May and June, the highest concentrations

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were measured at Campbell Slough (Reach F). That site is farther off the main stem of the river and experiences the slowest flushing among the sites. At all sites, phytoplankton abundance was higher in April and May than during the spring freshet in June. In April at Ilwaco and in June at Ilwaco and Whites Island, chlorophyll *a* concentrations were undetectable.
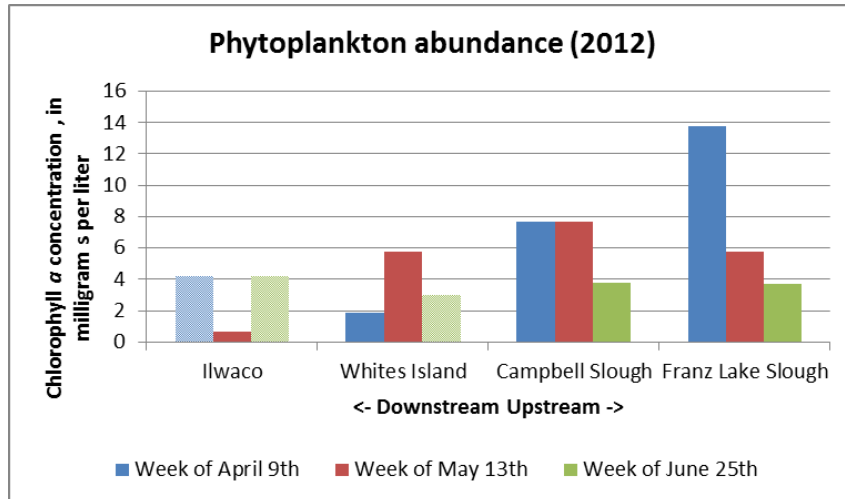
Figure 60. Phytoplankton abundance measured as chlorophyll *a* concentration at all four water-quality monitoring sites in 2012. Columns that are filled with slashed lines represent measured concentrations less than the indicated detection limit.
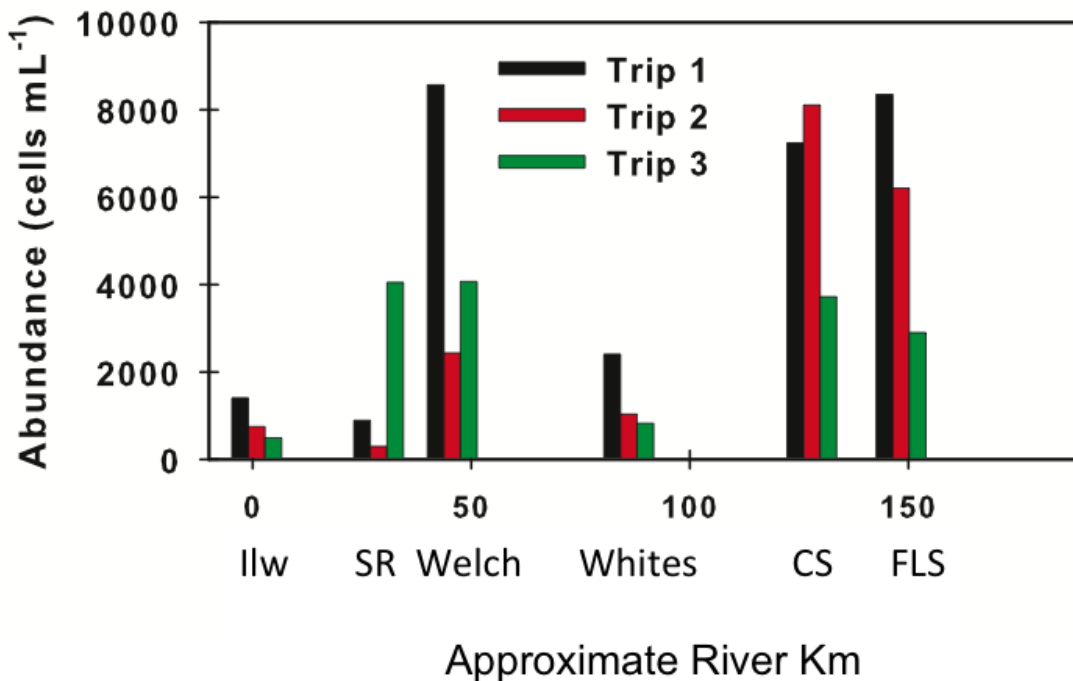
Figure 61. Phytoplankton abundance measured in cells mL-1 at six trend sites in 2012 organized according to approximate river km (Ilw = Ilwaco; SR = Secret River; Welch = Welch Island; Whites = Whites Island; CS = Campbell Slough in Ridgefield National Wildlife Refuge; FLS = Franz Lake Slough).

Since the field collections did occur on the same dates for the suite of sites, they are organized according to trip number, which is chronological. Sampling dates are given in Table 21. The highest phytoplankton abundances were found at Welch Island in May, at Campbell Slough in April and May, and at Franz Lake Slough in April and May.

Table 21. Phytoplankton diversity metrics determined at six trend sites on a variety of dates in 2013. S = species richness (i.e., total number of species observed), N = total phytoplankton abundance, H' is the Shannon-Weaver diversity metric.

Site	Date	Latitude	Longitude	Long dec	Reach	S	N	H'
Ilwaco marina	4/11/12	46° 18' 02"(46.30)	-124° 02' 43"	-124.034	A	25	1406	2.29
	4/24/12	46° 18' 02"	-124° 02' 43"			30	750	2.79
	5/15/12	46° 18' 02"	-124° 02' 43"			24	492	2.22
Welch Island	5/16/12	45° 47.032' (45.78)	-122° 45.291'	-122.75	B	37	8573	2.47
	6/13/12	45° 47.032'	-122° 45.291'			40	2440	2.47
	7/17/12	45° 47.032'	-122° 45.291'			47	4074	2.63
Secret River	5/16/12	45° 9.561' (45.16)	-122° 20.408'	-122.34	B	27	893	2.63
	6/13/12	45° 9.561'	-122° 20.408'			40	582	3.10
	7/17/12	45° 9.561'	-122° 20.408'			33	4051	2.58
Whites Island	4/12/12	46° 09' 39"(46.15)	-123° 20' 16"	-123.33	C	36	2410	2.60
	5/17/12	46° 09' 39"	-123° 20' 16"			30	4983	2.12
	6/27/12	46° 09' 39"	-123° 20' 16"			26	4406	2.26
Campbell Slough	4/9/12	45° 47' 05"45.78)	-122° 45' 15"	-122.75	F	29	7241	2.51
	5/13/12	45° 47' 05"	-122° 45' 15"			21	8107	1.60
	6/25/12	45° 47' 05"	-122° 45' 15"			46	3722	2.13
Franz Lake Slough	4/10/12	45° 36' 04"(45.60)	-122° 06' 00"	-122.10	H	22	8349	1.77
	5/14/12	45° 36' 04"	-122° 06' 00"			28	6206	1.67
	6/26/12	45° 36' 04"	-122° 06' 00"			30	2903	2.41

Phytoplankton cell abundance agreed well with estimates of chlorophyll (Figure 25). Phytoplankton species richness (S) was not correlated with phytoplankton total abundance (p=0.74). There was a significant difference in the total number of phytoplankton cells per volume among the different sites according to a one-way ANOVA performed on the averages of all observations at each site (p=0.026). However, a Student-Keuls post hoc test of pair-wise differences did not reveal any significant differences between pairs of sites, suggesting that the overall between-site differences were well distributed across sites in 2012. Qualitatively, the highest phytoplankton abundances were observed at Campbell Slough, at the Ridgefield National Wildlife Refuge. Franz Lake Slough and Welch Island also had high phytoplankton abundances, particularly prior to the spring freshet. In contrast, Ilwaco marina had the lowest averaged phytoplankton abundances, and Secret River had the absolute lowest abundances in mid-June, during the spring freshet.

According to the Shannon-Weaver diversity index, diversity (H') was highest at Secret River (May, June, July), Whites Island (April), Welch Island (July), and Ilwaco marina (April) (Table 21). The highest diversity

index values tended to be observed when phytoplankton abundances were low. A negative correlation (r=-0.41) between the diversity index value and total cell abundance, however, was not significant (p=0.088 at α=0.05), likely due to the low number of observations, which yielded low statistical power.

The phytoplankton assemblage was generally numerically dominated by diatoms at all sites during the time periods sampled (Figure 62). At both Campbell Slough and Franz Lake Slough, flagellate species (including green algae, chrysophytes, dinoflagellates, and cryptophytes-those taxa that bear flagella for swimming) were numerically abundant, particularly following the freshet. At Ilwaco, cryptophytes in particular were abundant in June 2012.
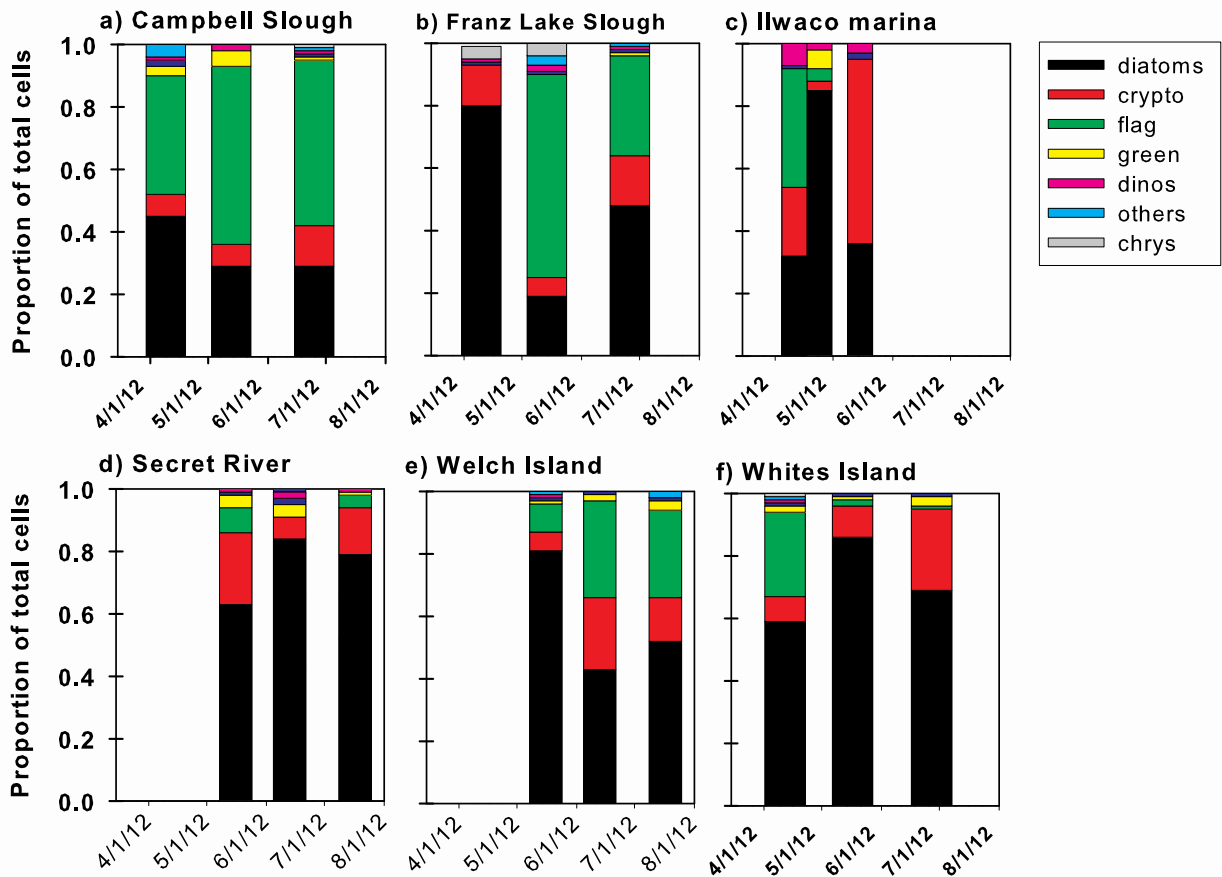
Figure 62. Proportion of the phytoplankton assemblage at each of the six trend sites (a-f) accounted for by each of the following taxonomic groupings: diatoms, cryptophytes (crypto), flagellates (flag), green non-motile algae (green), dinoflagellates (dinos), chrysophytes (chyrs), and 'others'. The latter includes photosynthetic ciliates and euglenoids. Note that these proportions were calculated based on cell abundance, not biomass.

The abundance of flagellate species (and the relative proportion of flagellates to diatoms) was highest at Campbell Slough (Reach F) and Franz Lake Slough (Reach H), particularly in mid-May (Figure 63). The highest observed abundances of diatoms were noted at Franz Lake Slough (mid-April) and at Welch Island (mid-May). The proportion of flagellates to diatoms exceeded 1:1 at Campbell Slough, Franz Lake Slough, and Ilwaco at various times of year (Figure 63); cryptophytes accounted for the high abundance

of flagellates at Ilwaco, while at the other two sites small flagellates (including zoosporic fungi) less than ~5 µm in diameter comprised the flagellate assemblage.
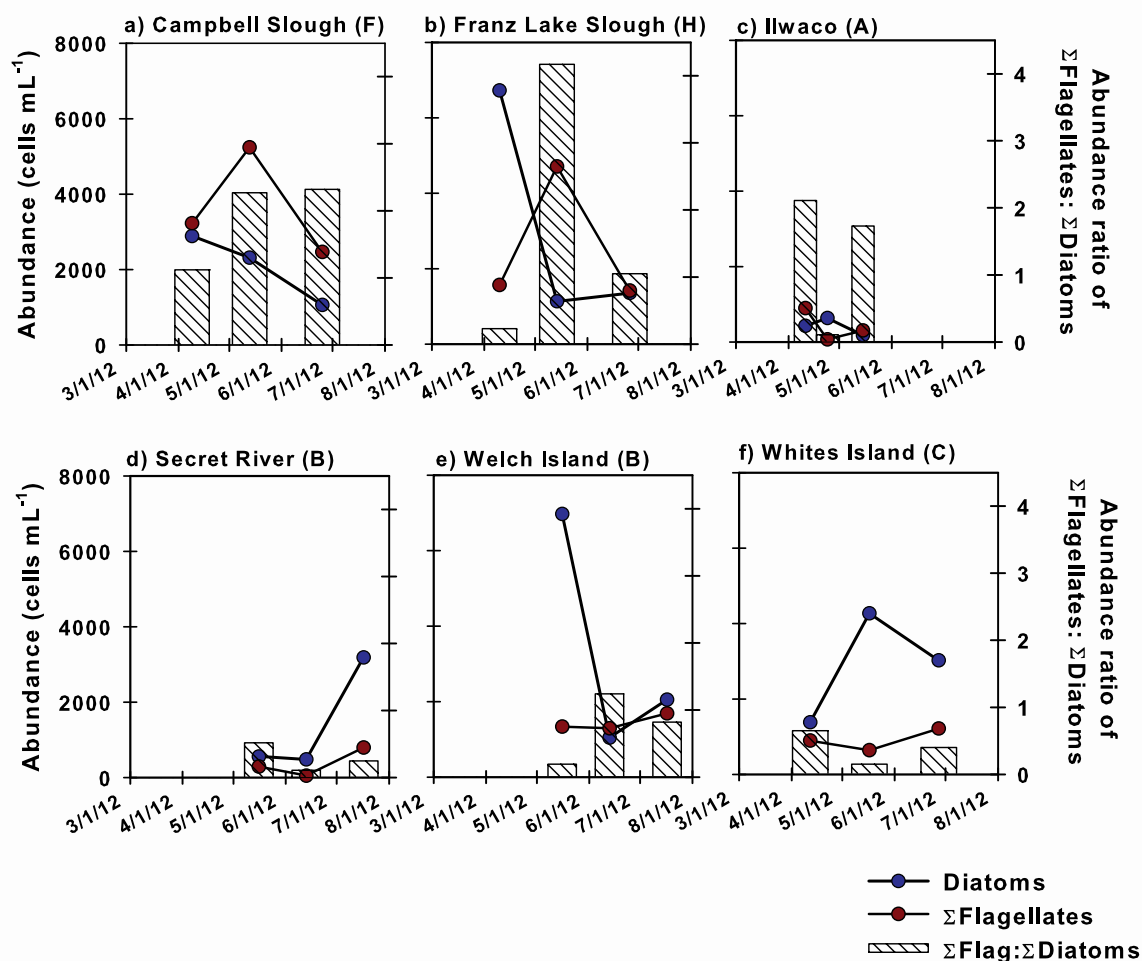
Figure 63. Abundances of diatoms (ΣDiatoms), flagellate species (ΣFlagellates), and the ratio of flagellates to diatoms (ΣFlag: ΣDiatoms) at each of six sites: a) Campbell Slough, b) Franz Lake Slough, c) Ilwaco, d) Secret River, e) Welch Island, and f) Whites Island. The letter describing the particular reach in which the site is found is indicated in brackets after the site name.

In 2012, periphyton abundance as measured by chlorophyll *a* concentrations was highest at Ilwaco, except in early April, when it was highest at Franz Lake Slough (Figure 64). By far, the highest chlorophyll *a* concentrations measured during 2012 were at Ilwaco in May, when filamentous periphyton was very abundant. The absence of filamentous periphyton at the other sites explains the large difference between the May sample at Ilwaco and even the highest concentrations measured at other sites. No samples were collected in April at Whites Island or in April or May at Campbell Slough because of the absence of substrate that could be sampled. Other than the overall higher chlorophyll *a* concentrations of periphyton at Ilwaco, there were no clear temporal or spatial patterns in periphyton abundance among the trend sites in 2012.
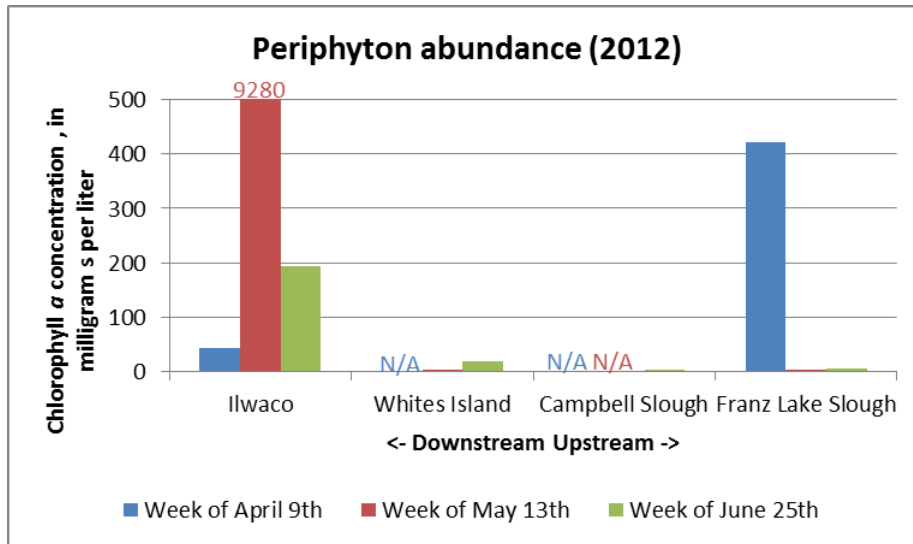
Figure 64. Periphyton abundance measured as chlorophyll *a* concentration at all four water-quality monitoring sites in 2012. [N/A, no sample collected]

4.4.1.2 *Rates*

4.4.1.2.1 Carbon Uptake Phytoplankton Productivity Experiments

Phytoplankton productivity data from 2011 were not available in time for the Year 7 report, so they are presented here. Phytoplankton productivity was estimated as carbon uptake in milligrams of carbon per cubic meter of water per hour using the 14C uptake method during the weeks of May 9 and June 20, 2011.

In May, phytoplankton productivity rates was highest at Franz Lake Slough and decreased at sites moving downstream (Figure 65). In June, phytoplankton productivity was lower than in May at Franz Lake Slough and Whites Island. No experiment was run in June at Campbell Slough. Ilwaco was the only site with a greater phytoplankton productivity in June than in May.
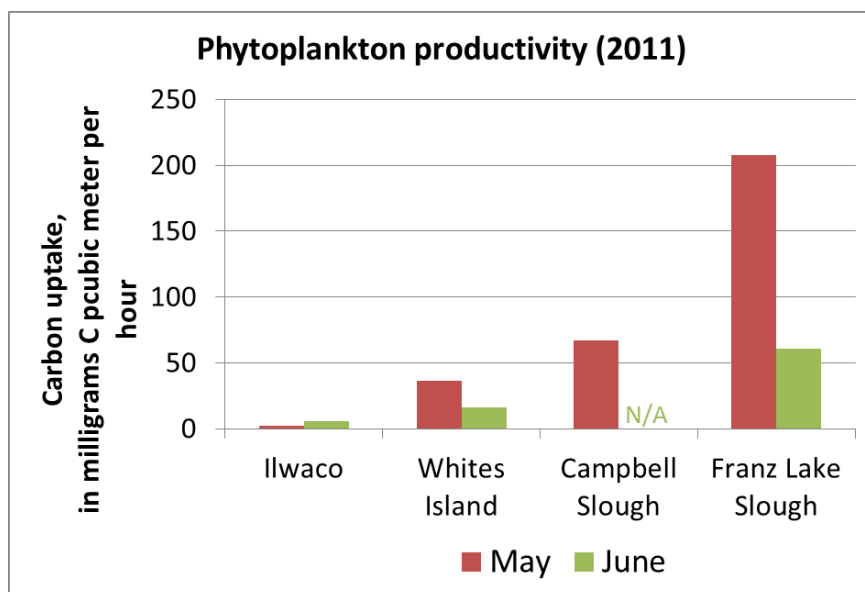
Figure 65. Phytoplankton productivity measured experimentally using the carbon-14 uptake method at the four water-quality monitoring sites in 2011. [N/A, no experiment]

Results from 2012 were not available in time for this report.

4.4.1.2.2 Periphytometers: Periphyton Productivity Experiments

Periphyton productivity was measured as chlorophyll *a* accumulation on 21 mm glass-fiber filters placed underwater for two weeks per experiment. Periphyton productivity experiment results from 2011 were not available in time for the Year 7 BPA report, so those results are presented here. Average periphyton productivity rates for each experiment are reported as the average of four unscreened replicates, in milligrams of chlorophyll *a* per square meter of substrate per day. It was expected that the presence of mesh screen over the four screened replicates would inhibit grazing of periphyton from the filters. Therefore, if grazing impacted periphyton accrual on the filters, the unscreened filters were expected to have less periphyton accumulation compared to the screened filters and the screened filters would be used in the periphyton productivity calculations. However, at the end of the experiments, amphipods were often found on many of the filters underneath the screen. These screened filters typically had less periphyton growth than the unscreened filters, so they were excluded from the analyses in 2011 and 2012 and data from unscreened filters were used instead.

Experiments were run starting the weeks of April 11, May 9, and June 20, 2011. Campbell Slough could not be accessed due to high water in April 2011. In May 2011, periphytometer experiments were deployed at Campbell Slough, but could not be retrieved for four weeks due to high water. Those filters were too degraded to be analyzed, so data from Campbell Slough are only available from the June deployment.

The results of each month's periphytometer experiments in 2011 showed the lowest ambient periphyton productivity rates at Franz Lake Slough and increasing periphyton productivity rates at each site moving downstream, although there were no data for April or May at Campbell Slough (Figure 66). Although there is a consistent spatial pattern during each month's experiments, there was no consistent temporal relationship across the all sites. At Ilwaco and Whites Island, periphyton productivity increased from April to May, and then decreased in late June to early July, after the spring freshet. At Campbell

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Slough, seasonal trends in 2011 cannot be determined because data are only available for June-July. At Franz Lake Slough, periphyton productivity rates were lower in May and June-July than in April 2011.
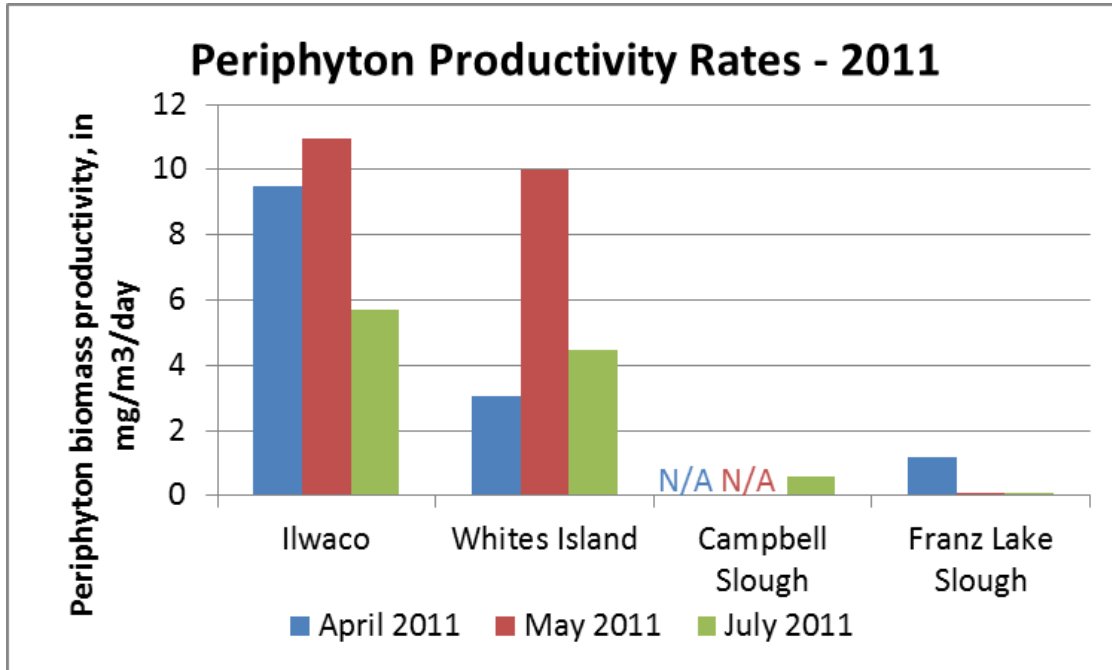
Figure 66. Periphyton productivity rates measured experimentally using micro-NDS periphytometers in 2011. Experiments were run the weeks of April 11–25, May 9–23, and June 20–July 4, 2011. [N/A, no experiment]

Experiments were run starting the weeks of April 9, May 14, and June 25, 2012. As in 2011, there was a distinct pattern of increasing periphyton productivity at each site moving downstream in April and May 2012 (Figure 67). However, in June-July, that pattern did not hold up, as periphyton productivity at Campbell Slough exceeded that at Whites Island. The periphyton productivity rates measured at each site show different seasonal patterns in 2011 and 2012.
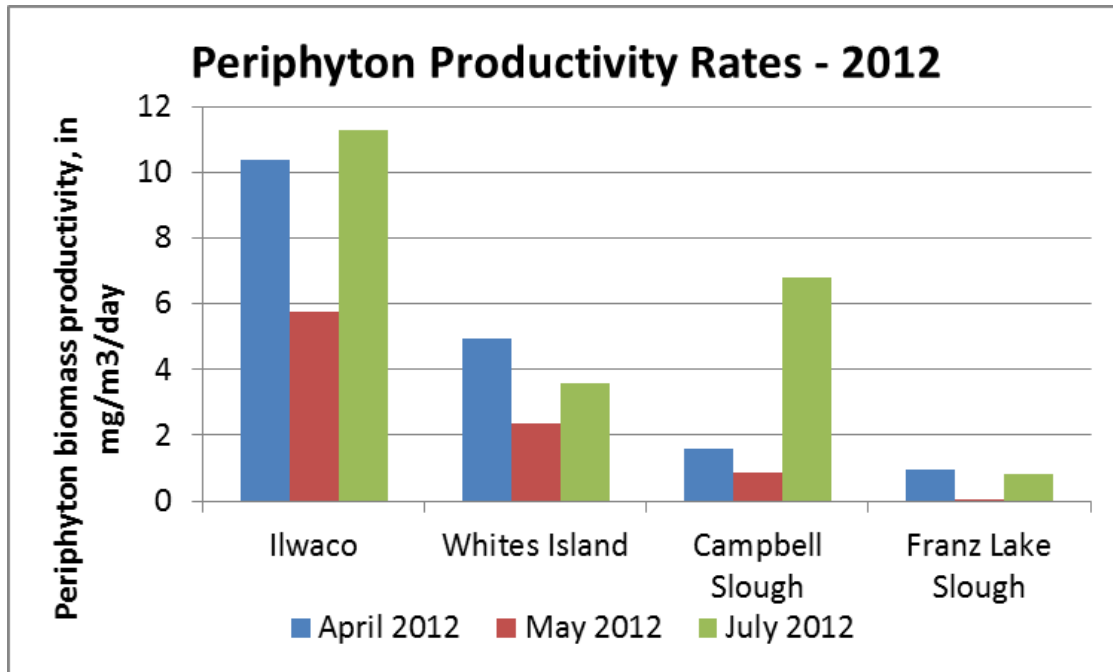
Figure 67. Periphyton productivity rates measured experimentally using micro-NDS periphytometers in 2012. Experiments were run the weeks of April 9–23, May 14–28, and June 25–July 9, 2012.

4.4.1.2.2.1 Periphytometers: Effect of Nutrient Enrichments on Periphyton Productivity Experiments

Additional replicates of the periphytometer experiments were exposed to enriched nutrient solutions in order to assess nutrient limitation or co-limitation. Significantly increased growth of periphyton on filters exposed to enriched nutrient solutions compared to those exposed only to ambient nutrient concentrations (the control treatment) would indicate that periphyton growth is limited by that nutrient. Three enriched nutrient solutions were used: nitrogen (N), phosphorus (P), and nitrogen plus phosphorus (NP).

In 2012, the periphytometer experiments did not show any nutrient limitation in periphyton growth at Ilwaco or Franz Lake Slough. At Whites Island, the growth of periphyton on filters exposed to enriched concentrations of phosphorus was significantly greater than growth on the control filters (Figure 68). At Campbell Slough, the NP treatment had significantly greater growth than the control for all three experiments in 2012, indicating that co-limitation of nitrogen and phosphorus controlled the rate of periphyton growth throughout the spring and early summer at that site (Figure 69—Figure 71).
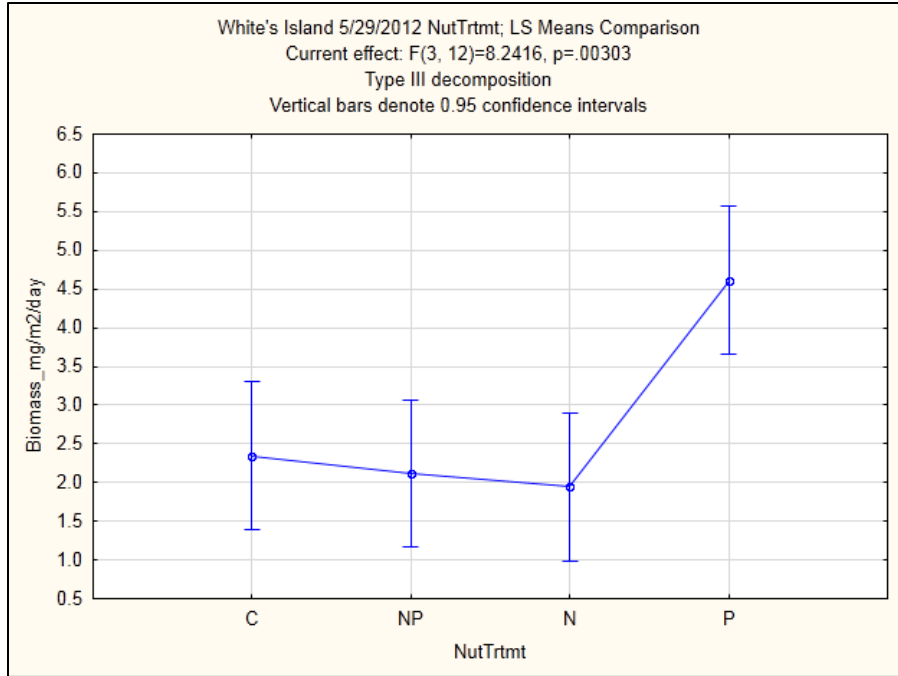
Figure 68. Comparison of mean productivity rates of periphyton exposed to enriched nutrient solutions measured experimentally using micro-NDS periphytometers in at Whites Island in May 2012. Vertical bars show 95% confidence intervals. [mg/m2/day, milligrams per square meter per day; NutTrtmt, nutrient treatment; C, control (ambient); NP, nitrogen plus phosphorus; N, nitrogen; P, phosphorus]
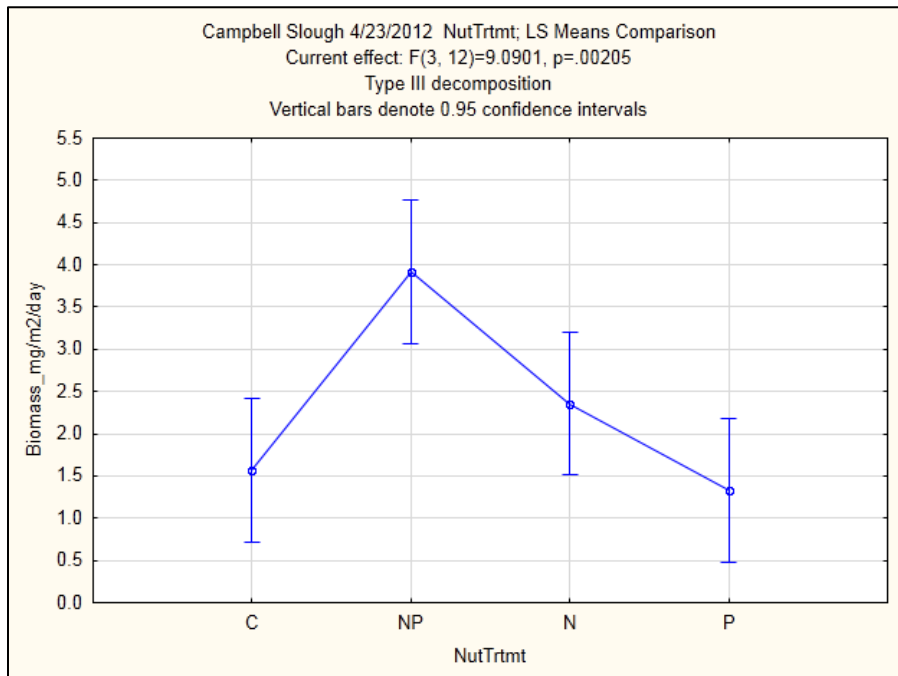
Figure 69. Comparison of mean productivity rates of periphyton exposed to enriched nutrient solutions measured experimentally using micro-NDS periphytometers in at Campbell Slough in April 2012. Vertical bars show 95% confidence intervals.   [mg/m2/day, milligrams per square meter per day; NutTrtmt, nutrient treatment; C, control (ambient); NP, nitrogen plus phosphorus; N, nitrogen; P, phosphorus]
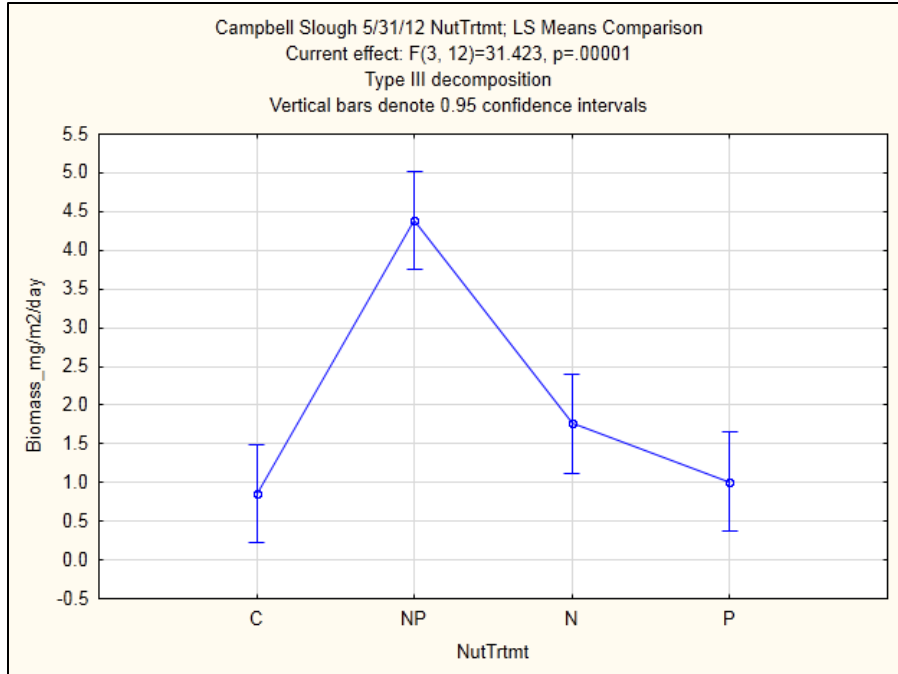
Figure 70. Comparison of mean productivity rates of periphyton exposed to enriched nutrient solutions measured experimentally using micro-NDS periphytometers in at Campbell Slough in May 2012. Vertical bars show 95% confidence intervals. [mg/m2/day, milligrams per square meter per day; NutTrtmt, nutrient; C, control (ambient); NP, nitrogen plus phosphorus; N, nitrogen; P, phosphorus]
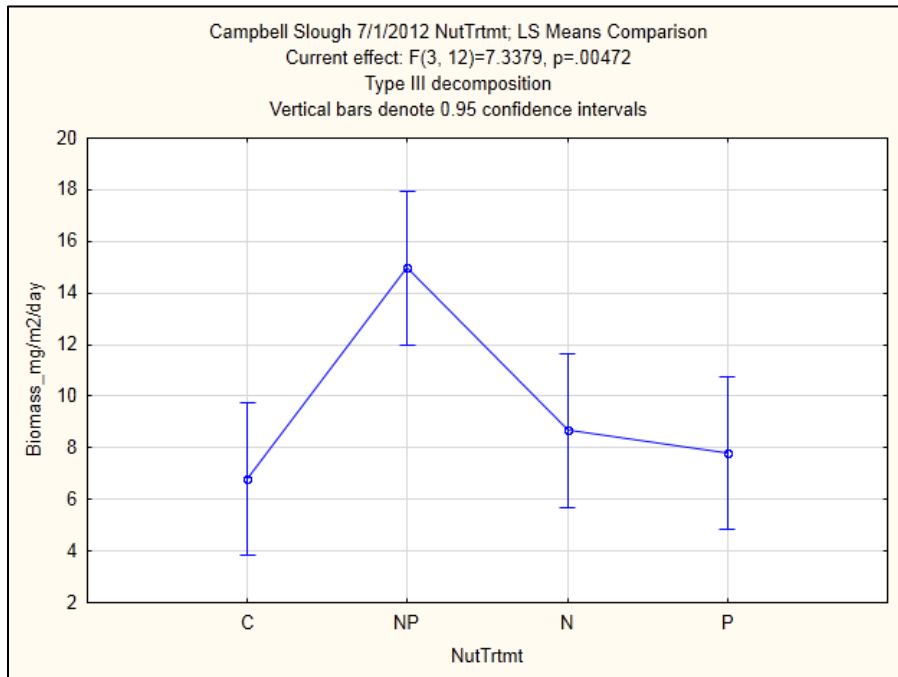
Figure 71. Comparison of mean productivity rates of periphyton exposed to enriched nutrient solutions measured experimentally using micro-NDS periphytometers in at Campbell Slough in June–July 2012. Vertical bars show 95% confidence intervals. [mg/m2/day, milligrams per square meter per day; NutTrtmt, nutrient treatment; C, control (ambient); NP, nitrogen plus phosphorus; N, nitrogen; P, phosphorus]

4.4.2 Zooplankton/secondary productivity

4.4.2.1 *Zooplankton abundance*

Zooplankton abundances were highest at Campbell Slough in Ridgefield National Wildlife Refuge (Figure 37). Total abundances in Reaches B and C were lower than in Reach F (Campbell Slough), Reach H (Franz Lake Slough), and Reach A (Ilwaco). Abundances were highest in mid to late May at all sites. Values were also high in late March at Campbell Slough. Although zooplankton abundances were not determined in late March at most of the sites, total abundances were low at Ilwaco. By late April, values had declined at Campbell Slough, while they had risen at Franz Lake Slough.
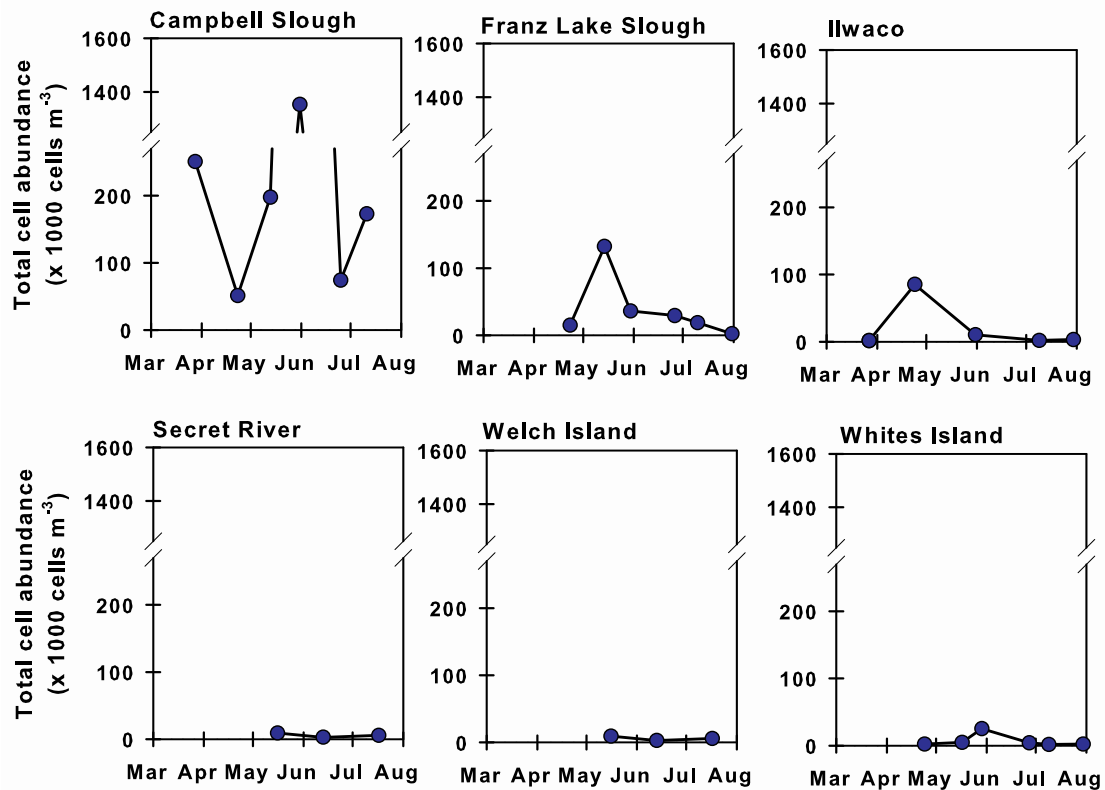
Figure 72. Total abundance of zooplankton at each of the six trend sites in 2012.
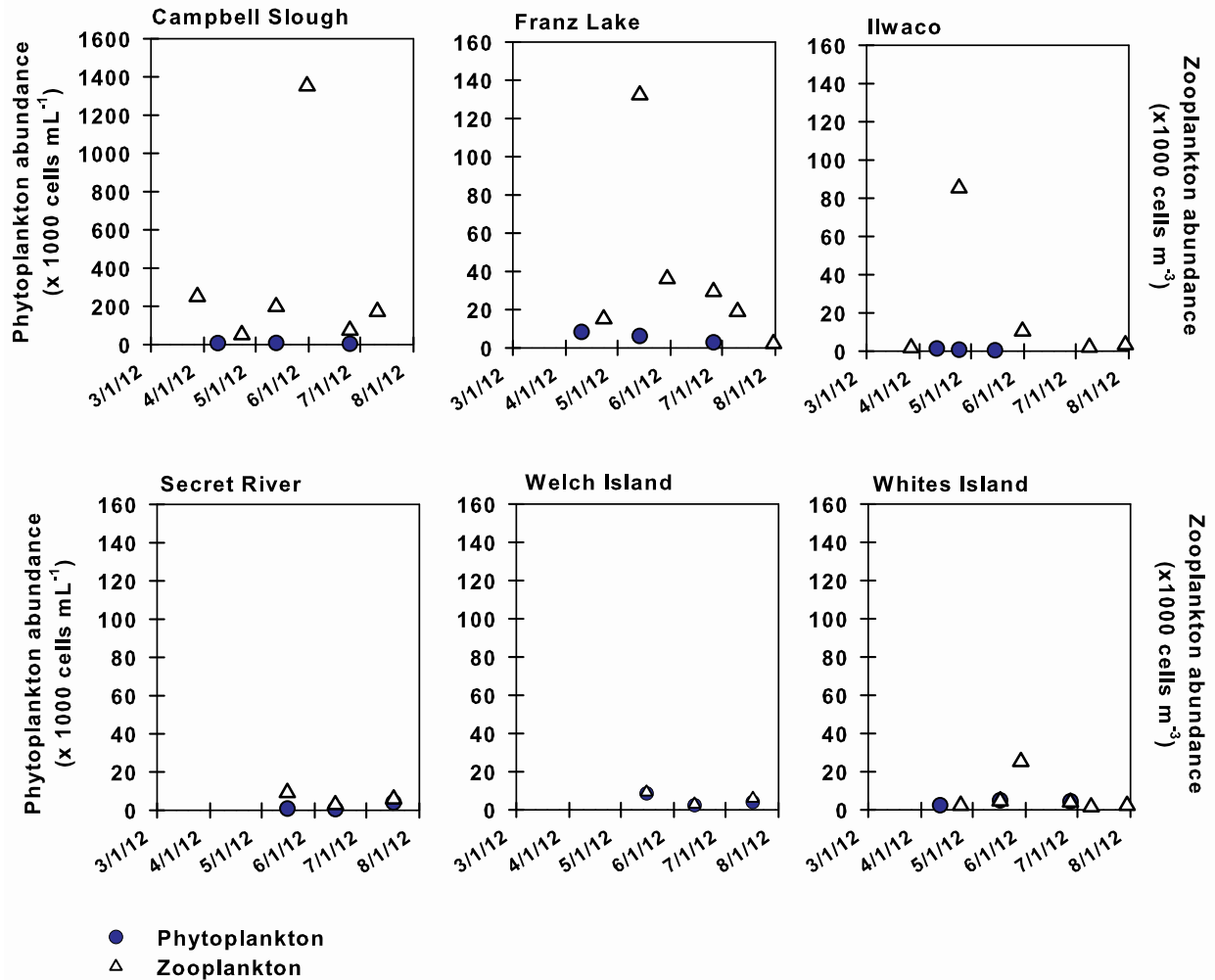
Figure 73. Abundances of phytoplankton (cells mL-1) and zooplankton (cells m-3) at the six trend sites in 2013, plotted together to facilitate visual comparison of total abundances. Note that these are abundance values, and do not reflect variations in biomass explicitly.

Phytoplankton and zooplankton abundances were not significantly correlated ($r=0.415$, $p=0.159$), although the number of observations was small, rendering the statistical power of the test low. A lag between phytoplankton abundance and zooplankton abundance values could have existed, but either temporal aliasing or too few observations could have obscured the relationship. A similar trend in values for both abundances of phytoplankton and zooplankton was noted at Secret River and Welch Island (Figure 73), but with $n=3$ observations, the relationship could not be supported statistically.

4.4.2.2 *Zooplankton diversity*

Zooplankton diversity was lowest at the Ilwaco site, which was dominated by copepods on all dates sampled (one-way ANOVA, $p<0.001$ with Student Newman-Keuls post hoc test). Shannon-Weaver index values did not differ significantly among the other sites. The open water zooplankton assemblage was dominated by copepods at all times sampled at Ilwaco. At the downstream sites (Secret River, Welch Island, and Whites Island), rotifers dominated the assemblage throughout the time series, while at the

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two more upstream sites (Campbell Slough and Franz Lake Slough), rotifers were abundant early in the spring freshet phase, while during the later portion of the freshet and into the summer, cladocerans and copepods dominated at Campbell Slough and Franz Lake Slough, respectively (Figure 74).

Table 22. Shannon-Weaver diversity index values (H') for zooplankton taxa enumerated at the six trend sites: Ilwaco, Welch Island, Secret River, Whites Island, Campbell Slough, and Franz Lake Slough. The corresponding river reach is identified by letter in the table.

Site	Date	Latitude	Longitude	Long dec	Reach	H'
Ilwaco	3/27/12	46° 18' 02"(46.30)	-124° 02' 43"	-124.034	A	1.2
	4/24/12					0.036
	5/31/12					0.6
	7/9/12					0.8
	7/30/12					1.2
Welch Island	5/16/12	45° 47.032' (45.78)	-122° 45.291'	-122.75	B	2.3
	6/13/12					1.9
	7/17/12					2.5
Secret River	5/16/12	45° 9.561' (45.16)	-122° 20.408'	-122.34	B	1.7
	6/13/12					1.9
	7/17/12					2.2
Whites Island	4/24/12	46° 09' 39"(46.15)	-123° 20' 16"	-123.33	C	1.8
	5/17/12					1.9
	5/29/12					2.0
	6/27/12					1.4
	7/9/12					2.1
	7/30/12					1.6
Campbell Slough	3/28/12	45° 47' 05"(45.78)	-122° 45' 15"	-122.75	F	2.0
	4/23/12					2.2
	5/13/12					1.8
	5/31/12					2.1
	6/25/12					1.1
	7/11/12					1.3
Franz Lake Slough	4/23/12	45° 36' 04"(45.60)	-122° 06' 00"	-122.10	H	2.0
	5/14/12					1.8
	6/26/12					2.1
	7/10/12					1.5
	7/31/12					2.3
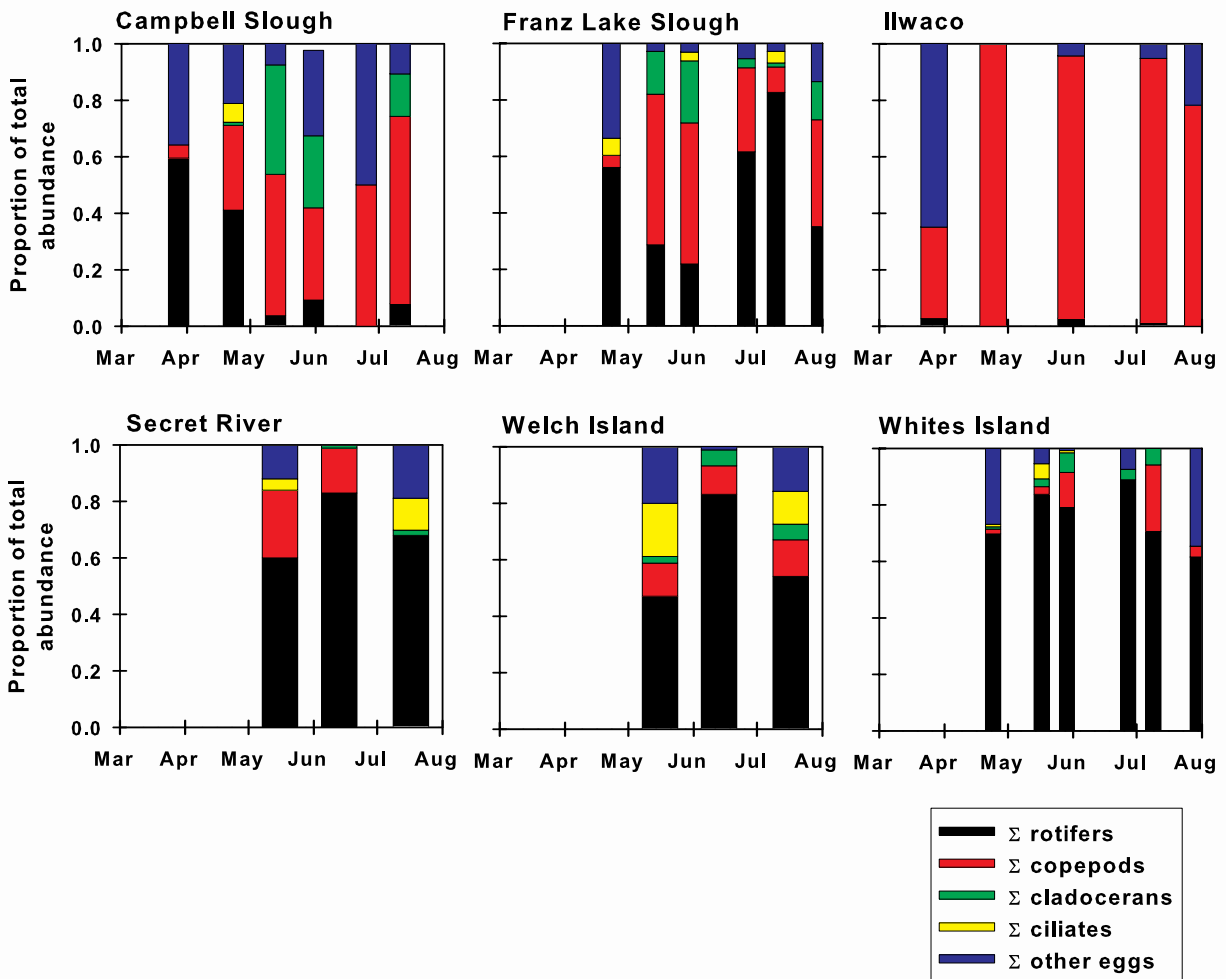
Figure 74. Proportional abundance of different zooplankton taxa. 'Other' eggs include those that could not be associated with a particular group.

Benthic macroinvertebrates

Benthic invertebrate taxa were determined at four sites: Ilwaco Slough, Secret River, Welch Island, and Whites Island. The proportional representation of taxa differed markedly between the brackish Ilwaco site and the other three sites (Figure 75), with the exception of oligochaete worms, which were abundant throughout the sites and over the time series. At Ilwaco, Tipulidae, oligochates, nematodes, and gastropods, and *Corophium* were present at abundances greater than 10%. The species composition was very similar between Secret River and Welch Island, with a numerical dominance of oligochaete worms and nematodes. Chironomids were present at 10-20% total abundance at Secret River during the three sampling time points, while the accounted for 5-10% of total abundance at Welch Island in May and June. At Whites Island there was a greater contribution of Ceratopogomonidae compared to the Secret River and Welch Island sites. The overall numbers of total individuals at each site were similar (data not shown).
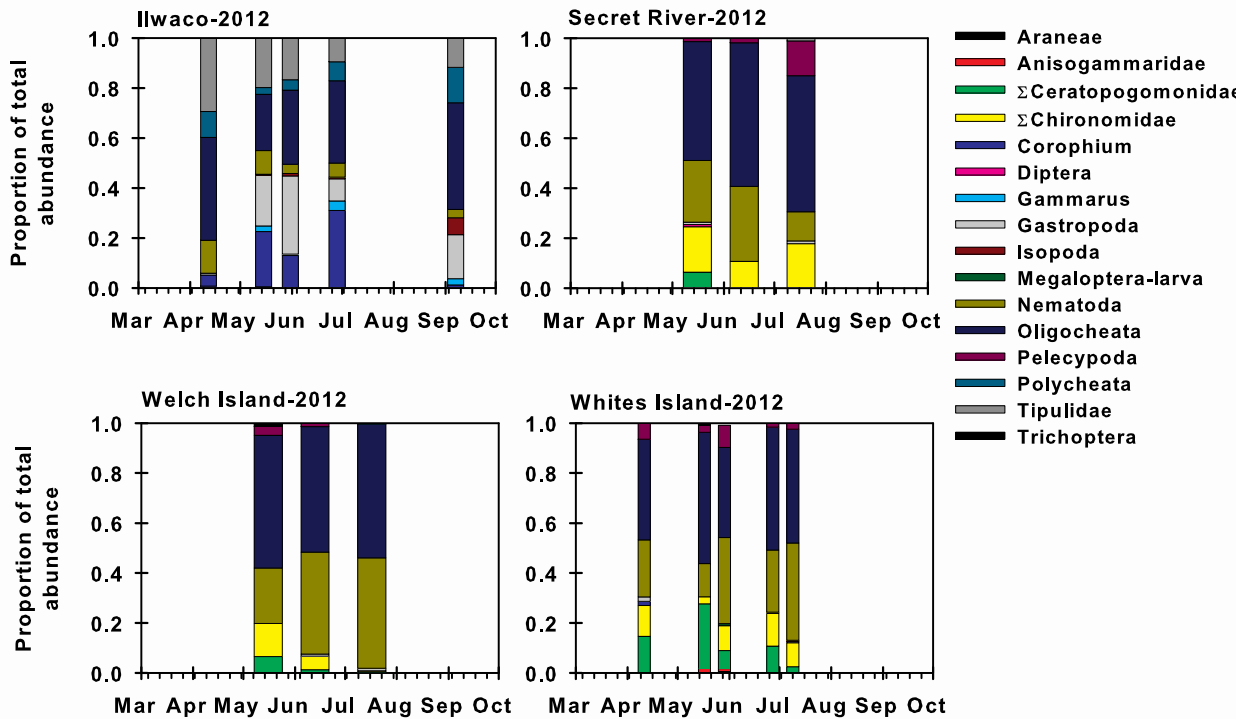
Figure 75. Proportional contribution of various taxa of benthic invertebrates at four of the trend sites: Ilwaco (Ilwaco; Reach A), Secret River (Reach B), Welch Island (Reach B), and Whites Island (Reach C). The sum symbol (S) indicates that adults, larvae, and pupae abundances are included in the category.

4.4.3 Isotope ratios

Stable isotope food web data were not available for this report. However, a subset of juvenile Chinook salmon bodies collected in 2010 were analyzed for stable isotopes. In juvenile salmon bodies ^{13}C ratios ranged from –22.6 to –14.1, while ^{15}N ratios ranged from 7.4 to 14.1 (Figure 76). Ratios were significantly different (ANOVA, $p \leq 0.0001$) in marked and unmarked fish. The mean ∂^{13}C value was -20.7 ± 1.9 in marked fish (n=35) vs. -22.6 ± 1.9 (n=53) in unmarked fish, while the mean ∂^{15}N value was 12.2 ± 1.2 in marked fish vs. 10.7 ± 1.2 in unmarked fish.

Stable isotope ratios by site and stock are shown in Table 23. In both marked and unmarked fish from Campbell Slough, δ^{13}C values were more negative than at the other sites, but no clear spatial trends were evident. In contrast, ∂^{15}N values in unmarked Chinook showed a longitudinal gradient, with lower values in Reach H and higher values at sites in Reach C. This was not the case for marked fish; in these fish, ∂^{15}N values tended to be higher at Whites Island and Wallace Island than at other sites.

∂^{13}C ratios did not vary among stocks, for either marked or unmarked fish. In marked fish, ∂^{15}N were also similar among stocks. However, for unmarked fish, ∂^{15}N values were lower in Snake River fall, Upper Columbia fall, and Deschutes fall Chinook than in the Lower Columbia stocks.
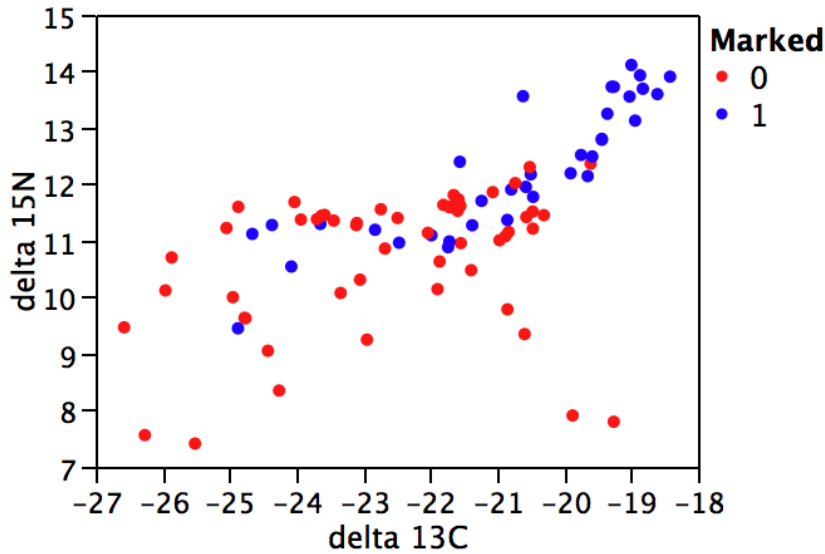
Figure 76. Stable isotope ratios in marked and unmarked juvenile Chinook salmon sampled from the 2010 Ecosystem Monitoring sites.

Table 23. Stable isotope ratios in juvenile Chinook salmon from 2010 Ecosystem Monitoring sites. Two Mirror Lake sites are also included as representative of fish from Reach H.

	$\partial^{13}C$		$\partial^{15}N$	
	Unmarked	marked	Unmarked	marked
C-Bradwood Slough	-20.7 ± 0.2 (n=5)	-19.6 ± 0.8 (n=3)	11.2 ± 0.2 (n=5)	12.7 ± 0.8 (n=3)
C-Jackson Island	-21.4 ± 0.9 (n=4)		11.6 ± 0.2 (n=4)	
C-Whites Island	-23.4 ± 1.2 (n=12)	-20.0 ± 1.6 (n=11)	11.4 ± 0.2 (n=12)	12.9 ± 1.0 (n=11)
C-Wallace Island	-21.6 ± 0.9 (n=10)	-18.8 ± 0.2 (n=3)	11.7 ± 0.3 (n=10)	13.9 ± 0.3 (n=3)
F-Campbell Slough	-24.4 ± 1.2 (n=9)	-22.0 ± 1.6 (n=16)	10.0 ± 0.6 (n=9)	11.4 ± 0.8 (n=16)
H-Mirror Lake Culvert	-22.8 ± 2.6 (n=11)	-19.8 ± 0.11 (n=2)	9.4 ± 1.5 (n=11)	12.4 ± 0.2 (n=2)
H-Mirror Lake Lake	-20.2 ± 0.5 (n=2)		8.6 ± 1.0 (n=2)	
	p=	P=	P=	P=
All sites	-22.6 ± 1.9 (n=53)	-20.8 ± 1.9 (n=35)	10.7 ± 1.2 (n=53)	12.2 ± 1.2 (n=35)

Table 24. Stable isotope ratios in juvenile Chinook salmon from different stocks. All fish were collected in 2010.

	$\partial^{13}C$		$\partial^{15}N$	
	Unmarked	marked	Unmarked	marked
West Cascades fall	-22.5 ± 1.6 (n=30)	-20.7 ± 1.9 (n=21)	11.1 ± 1.0[a] (n=30)	12.4 ± 1.1 (n=21)

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West Cascades spring	-22.7 (n=1)	-19.3 ± 0.5 (n=2)	10.9[ab] (n=1)	13.1 ± 1.4 (n=2)
Spring Creek Group fall	-22.5 ± 1.6 (n=5)	-21.0 ± 1.6 (n=8)	11.5 ± 0.3[ab] (n=5)	12.1 ± 1.1 (n=8)
Upper Columbia summer/fall	-22.2 ± 2.5 (n=13)	-21.5 ± 2.9 (n=3)	9.9 ± 1.4[b] (n=13)	11.4 ± 1.7 (n=3)
Snake River fall	-22.5 (n=1)	-22.8 (n=1)	9.3[ab] (n=1)	11.2 (n=1)
Deschutes fall	-24.5 ± 2.7 (n=3)		9.4 ± 1.6[ab] (n=3)	
	p =	p =	p = 0.0078	p=

4.4.4 Prey availability in emergent vegetation and open water habitats

The diversity of prey available at sites sampled in 2010 - 2012 was similar to patterns seen in previous years, with 39 invertebrate orders represented across the tows and diets from all sites (Figure 77). The overall mean number of orders collected in tow samples per site per month was 7.9, with greater diversity in the shoreline habitats than in the open water habitats (mean number of orders, EV = 8.9, OW = 6.9, p<0.0002). Diptera were abundant, comprising 31% of the invertebrates caught in all tows in 2010-2012. Cladocerans and Copepods were also abundant at specific sites, especially at Deer Island and sites upriver from there (Figure 77). Hemiptera, primarily aquatic Corixidae but also some terrestrial bugs, were also abundant at some of the sites (Figure 77).
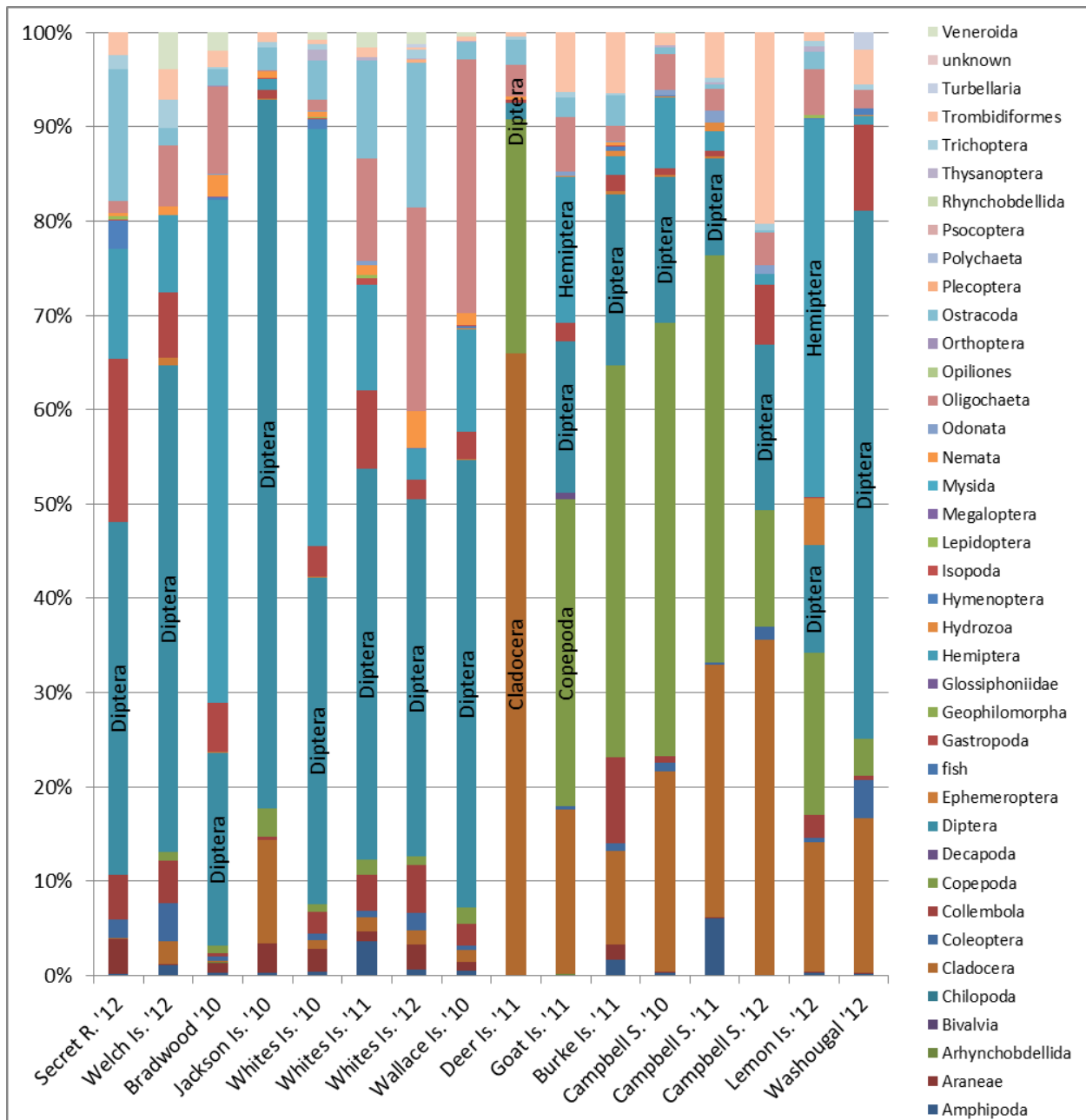
Figure 77. Composition (mean % by order) of invertebrates caught in all tows collected per site in 2010-2012. Some taxa are labeled in each column to help orient the reader.

The abundance of prey varied tremendously by site, month and habitat during these times (Figure 83-Figure 86). Exploratory analyses indicate "habitat" (EV vs. OW) consistently explains more of the variation in the number of invertebrates captured during each sampling event (stepwise regression, "habitat" entered first, R^2=0.47; "site" explained an additional 15% and "month" an additional 4% of the variation). There was little correlation between invertebrate abundance from EV and OW habitats collected during the same sampling event (r=0.29). These patterns indicate there is more variation within a site on a local scale – within the area surveyed by each beach seine – than there is systematically among sites or the times in which they are sampled.

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The densities and biomass of invertebrates were consistently higher in the emergent vegetation (EV) habitats compared to the open water (OW) habitats (Figure 78), and this was especially true for preferred prey items (see prey selection results below). Overall, densities in EVs were 19.8x those of OWs, and for Diptera, Amphipods and Hemiptera, densities in EVs were 24.6x, 16.2x, and 163.9x greater than those in OWs, respectively. Interestingly, even for taxa that are pelagic and typically abundant in open water habitats, densities were generally greater in EVs compared to OWs. For instance, for Copepods and Cladocerans, mean densities in EVs were 7.8x and 24.7x greater than in OWs, respectively. These comparisons are even more dramatic when considering biomass. Comparing wet weights of invertebrates caught in the 2012 tows (the one year for which we have wet weights), invertebrate biomass was 91.3x greater per meter towed in the EV habitats than in OW habitats. For Diptera, Amphipods and Hemiptera, the EV:OW ratios of biomass were 45:1, 15.6:1, and 264:1, respectively. As with densities, biomass of Copepods and Cladocerans was also greater in the EV habitats compared to the OW habitats (8.5X and 130x, respectively).

Generally, prey densities and prey biomass increased in the emergent vegetation habitat throughout the growing season (Figure 78-Figure 81), suggesting there may be a relationship with plant growth and invertebrate abundance. Indeed, the percent cover of live vegetation (grass as well as other vegetation) was significantly related to the density of Diptera across emergent vegetation transects (p=0.03). However, little variation in invertebrate density was explained by this measure of cover (r^2=0.06), and there were no other significant relationships with other measures of cover or for the total mean number of invertebrates. Using a stepwise regression for the densities of Diptera, adding "% live vegetation" to the model after "month" and "site" did not improve the fit of the model (ΔAICc<2). Therefore, although there was a significant and positive relationship between Diptera abundance and % live emergent vegetation, the large amount of scatter suggests the relationship is weak.

Although we sampled only two EMP sites all three years (Campbell Slough and Whites Island), it appears "year" is not as useful in explaining variation compared to site, month or habitat. For the mean abundance of all invertebrates, there was no consistent difference among years during the months we could compare (Figure 82; ANOVA: site p=0.0008, year p=0.72, site*year p=0.43).
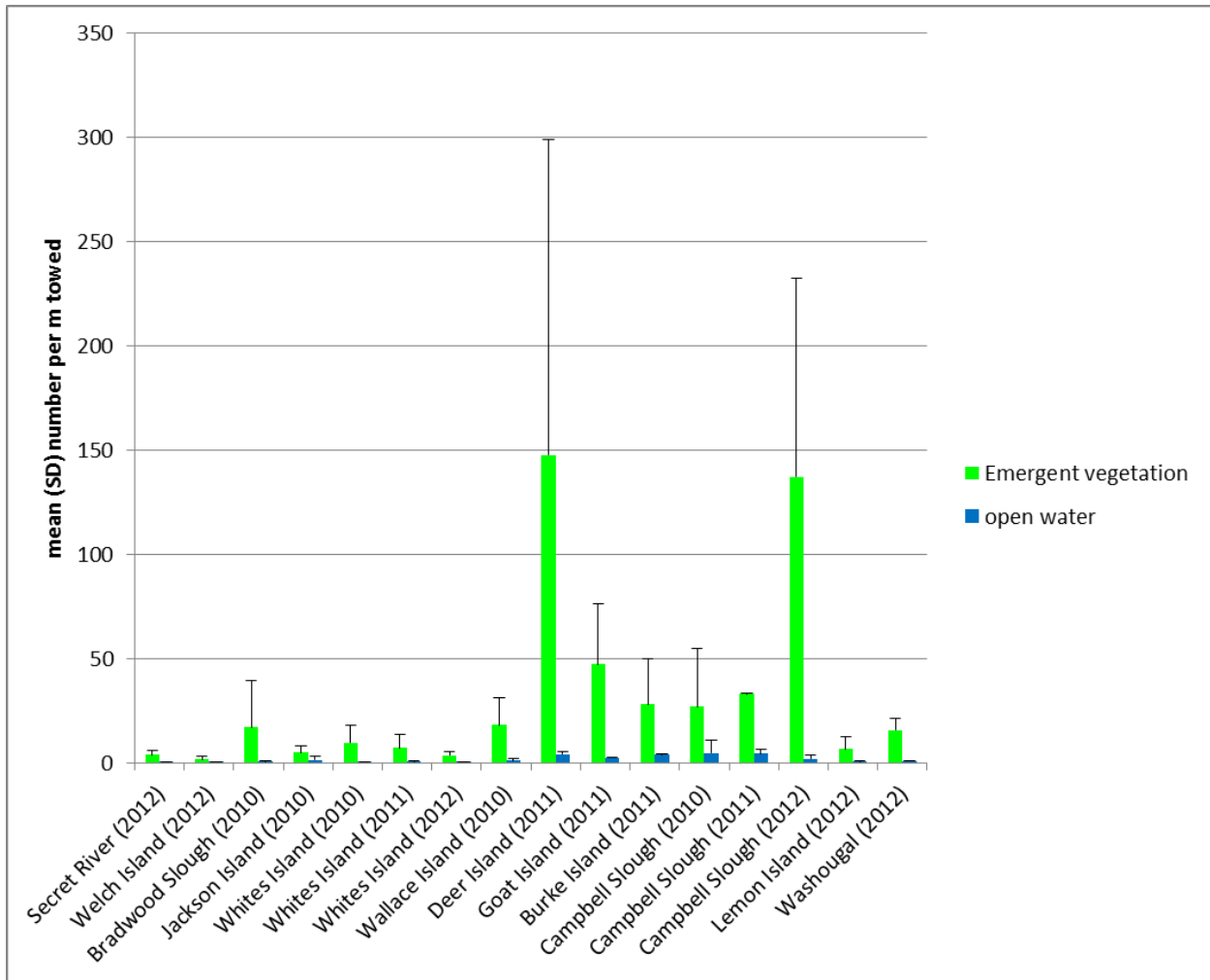
Figure 78. Mean (SD) number of invertebrates captured at each site by year. Note the extreme variation within and among sites and the difference between abundances captured in the emergent vegetation tows and the open water tows.
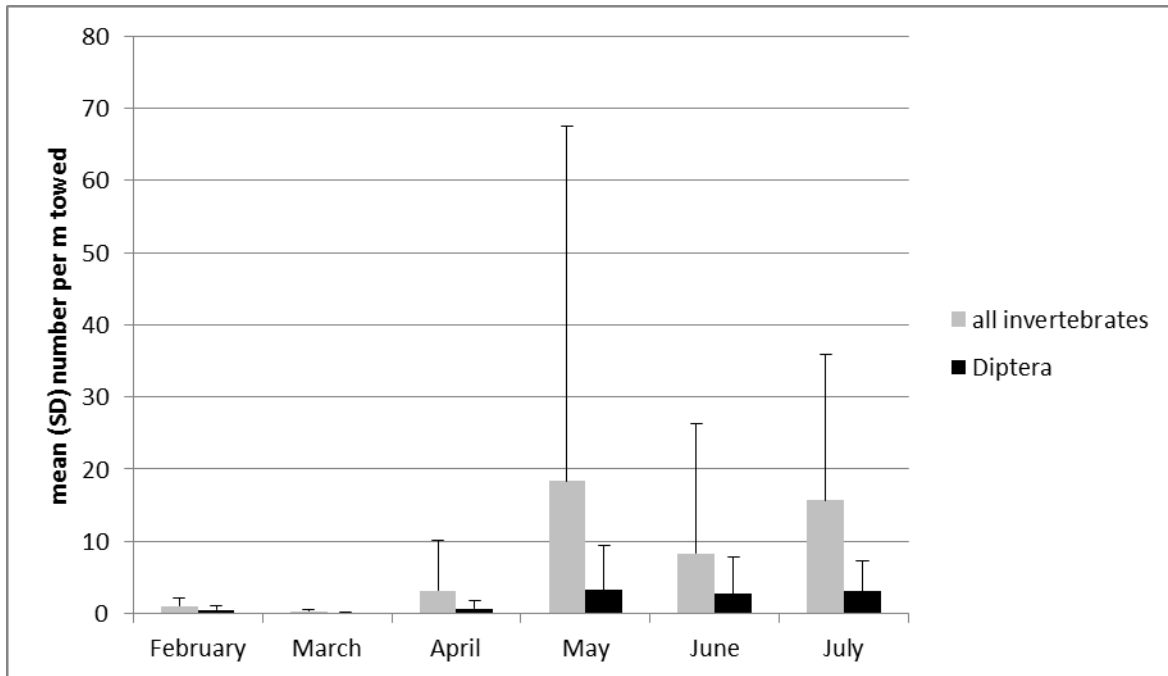
Figure 79. The mean (SD) number of all invertebrates and Diptera by month, for all collections from all sites samples during 2010-2012.
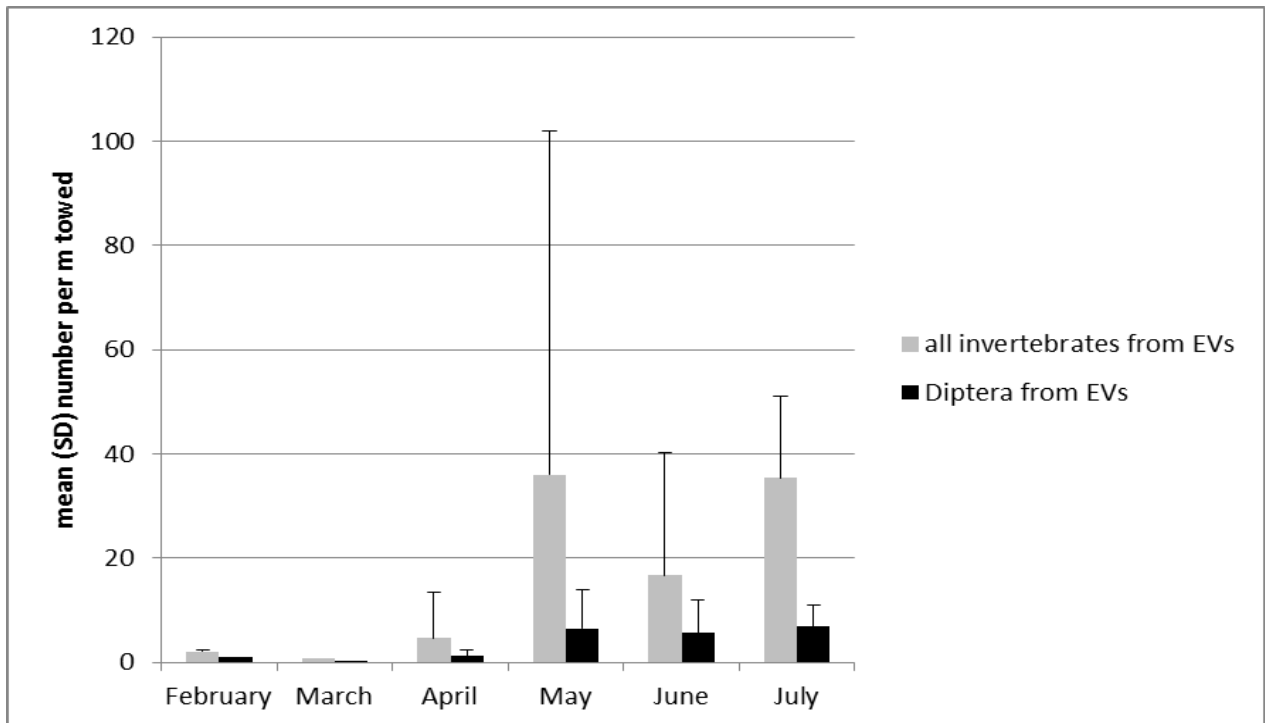
Figure 80. The mean (SD) number of all invertebrates and Diptera by month, for only the emergent vegetation samples collected during 2010-2012.
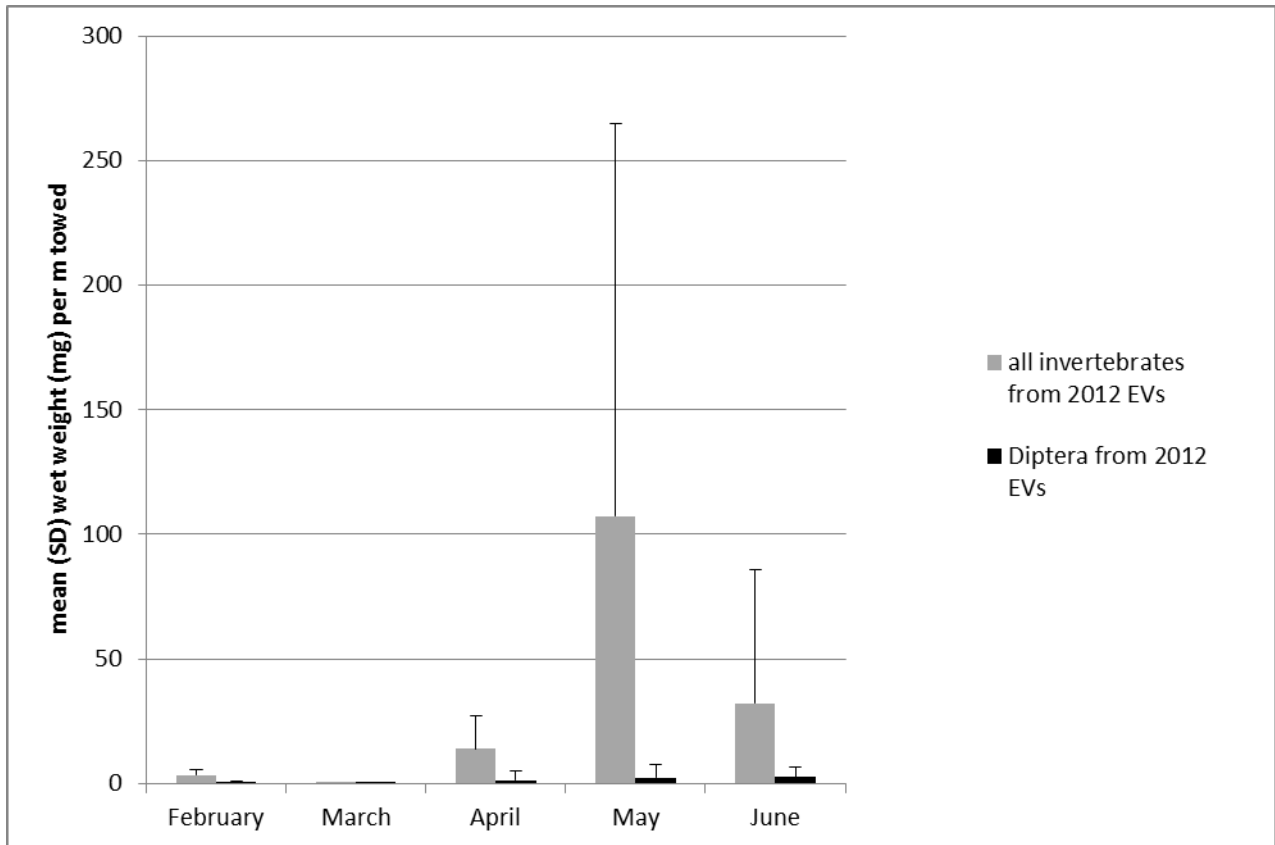
Figure 81. The mean (SD) wet weight of all invertebrates and Diptera by month, for only the emergent vegetation samples collected during 2012.
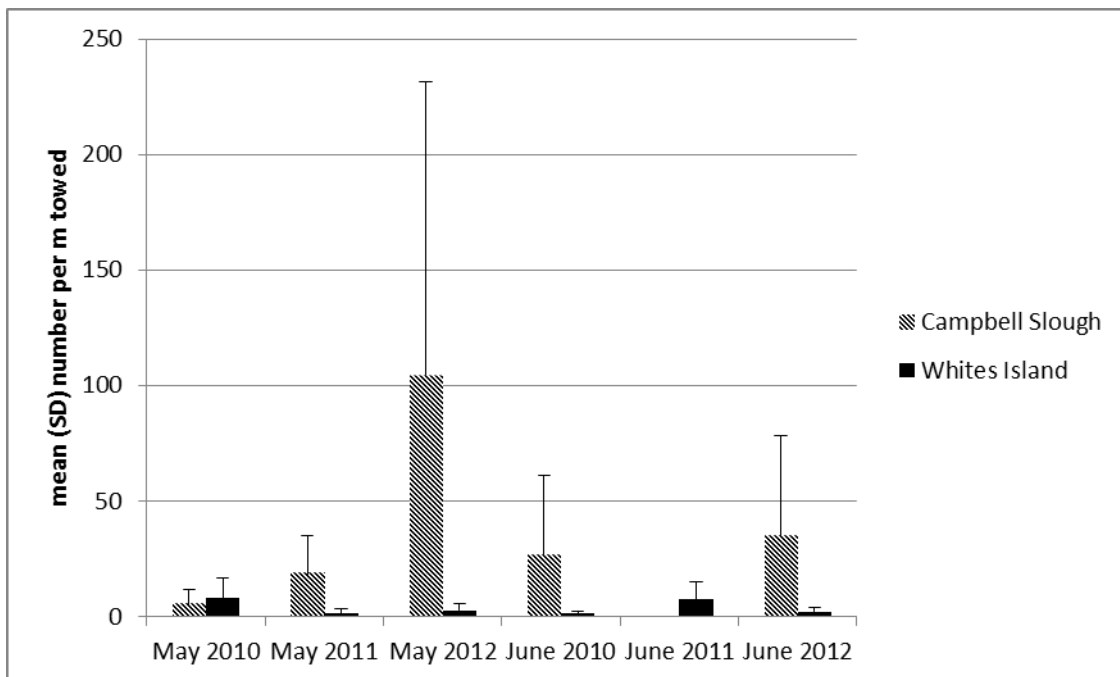
Figure 82. The mean (SD) number of invertebrates captured across both emergent vegetation and open water habitats during May and June of 2010, 2011, and 2012 at Campbell Slough and Whites Island. Note no tow samples were collected at Campbell Slough in June 2011.

4.5 Fish Use

In 2012, as in other years, there was considerable variation in water level at all of the fish sampling sites. Extreme high water levels, especially in Reaches F, G, and H, made some sites difficult to access. Also, the Campbell Slough site was closed to access during some months because of the presence of Endangered Species Act listed birds. Thus, while fish sampling took place every month, it was not always possible to fish all successfully each month. Table 25 summarizes the fishing attempts made at the 2012 sampling sites.

Table 25. Number of fishing attempts made at the 2012 Ecosystem Monitoring Sites each month. PBSB = Puget Sound beach seine; mPBSB = modified Puget Sound beach seine.

Site	Date	method	#of attempts	comment
Ilwaco Slough	2/21/12	PSBS	3	
Ilwaco Slough	3/19/12	PSBS	3	
Ilwaco Slough	4/19/12	PSBS	3	
Ilwaco Slough	5/17/12	PSBS	3	
Ilwaco Slough	6/12/12	PSBS	3	
Ilwaco Slough	7/16/12	PSBS	3	
Ilwaco Slough	8/13/12	PSBS	2	
Ilwaco Slough	9/11/12	PSBS	0	equipment failure
Ilwaco Slough	10/16/12	mPSBS	3	
Secret River	2/21/12	PSBS	1	
Secret River	3/20/12	PSBS	3	
Secret River	4/18/12	PSBS	3	
Secret River	5/16/12	PSBS	3	
Secret River	6/13/12	PSBS	3	
Secret River	7/17/12	PSBS	3	
Secret River	8/13/12	PSBS	3	
Secret River	9/11/12	PSBS	3	
Secret River	10/16/12	PSBS	1	high water/strong current/hangups
Welch Island	2/20/12	PSBS	3	
Welch Island	3/22/12	PSBS	1	
Welch Island	4/18/12	PSBS	3	
Welch Island	5/16/12	PSBS	1	
Welch Island	6/13/12	PSBS	1	
Welch Island	7/17/12	PSBS	1	
Welch Island	8/14/12	PSBS	3	
Welch Island	9/11/12	PSBS	3	
Welch Island	10/16/12	PSBS	3	
Whites Island	2/20/12	PSBS	3	

Whites Island	3/22/12	PSBS	2	
Whites Island	4/18/12	PSBS	3	
Whites Island	5/16/12	PSBS	1	
Whites Island	6/12/12	PSBS	1	
Whites Island	7/16/12	PSBS	3	
Whites Island	8/14/12	PSBS	3	
Whites Island	9/11/12	PSBS	3	
Whites Island	10/16/12	PSBS	3	
Campbell Slough	2/20/12		0	no sampling permit
Campbell Slough	3/22/12		0	no sampling permit
Campbell Slough	4/18/12		0	high water level
Campbell Slough	5/15/12	PSBS	2	
Campbell Slough	6/14/12	PSBS	1	
Campbell Slough	7/19/12	PSBS	3	
Campbell Slough	8/13/12	PSBS	3	
Campbell Slough	9/6/12	PSBS	3	
Campbell Slough	10/17/12		0	no sampling permit
Govt/Lemon Island	2/20/12	PSBS	0	No sampling permit
Govt/Lemon Island	3/21/12	PSBS	2	
Govt/Lemon Island	4/17/12	PSBS	2	
Govt/Lemon Island	5/15/12	PSBS	1	
Govt/Lemon Island	6/13/12	PSBS	1	
Govt/Lemon Island	7/18/12	PSBS	1	
Govt/Lemon Island	8/14/12	PSBS	3	
Govt/Lemon Island	9/5/12	PSBS	1	low water level
Govt/Lemon Island	10/17/12	mPSBS	3	
Washougal Wetland	2/22/12	PSBS	2	
Washougal Wetland	3/21/12	PSBS	2	
Washougal Wetland	4/16/12	PSBS	1	
Washougal Wetland	5/14/12	PSBS	1	
Washougal Wetland	6/11/12	PSBS	1	
Washougal Wetland	7/17/12	PSBS	1	high water level
Washougal Wetland	8/14/12	PSBS	2	
Washougal Wetland	9/5/12	mPSBS	2	
Washougal Wetland	10/17/12		0	low water level/site dried up
Reed Island	2/22/12	PSBS	3	
Reed Island	3/21/12		0	
Reed Island	4/16/12		0	

Reed Island	5/14/12		0	
Reed Island	6/11/12		0	
Reed Island	7/17/12		0	
Reed Island	8/16/12	PSBS	2	
Reed Island	9/5/12		0	low water level
Reed Island	10/17/12		0	low water level
Franz Lake Slough	2/22/12	PSBS	3	
Franz Lake Slough	3/21/12	PSBS	1	
Franz Lake Slough	4/16/12		0	high water level
Franz Lake Slough	5/14/12		0	high water level
Franz Lake Slough	6/11/12		0	high water level
Franz Lake Slough	7/17/12		0	high water level
Franz Lake Slough	8/15/12	mPSBS	3	
Franz Lake Slough	9/5/12	block	2	
Franz Lake Slough	10/17/12	block	3	

4.5.1 Fish Community Composition

Juvenile salmon and a variety of other fish species were present at all sites sampled in 2012 (Table 26). Salmonid species generally accounted for 5% or less of the total catch. Of the non-salmonid species captured, three-spined stickleback (*Gasterosteus aculeatus*) were among the most abundant at nearly all sites. Other species commonly observed in 2012 included chiselmouth (*Acrocheilus alutaceus*), yellow perch (*Perca flavescens*), pumpkinseed (*Lepomis gibbosus*), carp (*Cyprinus carpio*), and at Ilwaco Slough, the euryhaline species shiner perch (*Cymatogaster aggregata)* and staghorn sculpin (*Leptocottus armatus*). Species composition showed some distinctive seasonal patterns in 2012 (Table 26, Figure 83). Salmonids were present from February when sampling began, through July. Sculpin species, especially staghorn sculpin, were also fairly abundant in catches during this time. Later in the sampling season, species such as chiselmouth, shiner perch, and yellow perch became more abundant. During all months, stickleback were one of the dominant species.
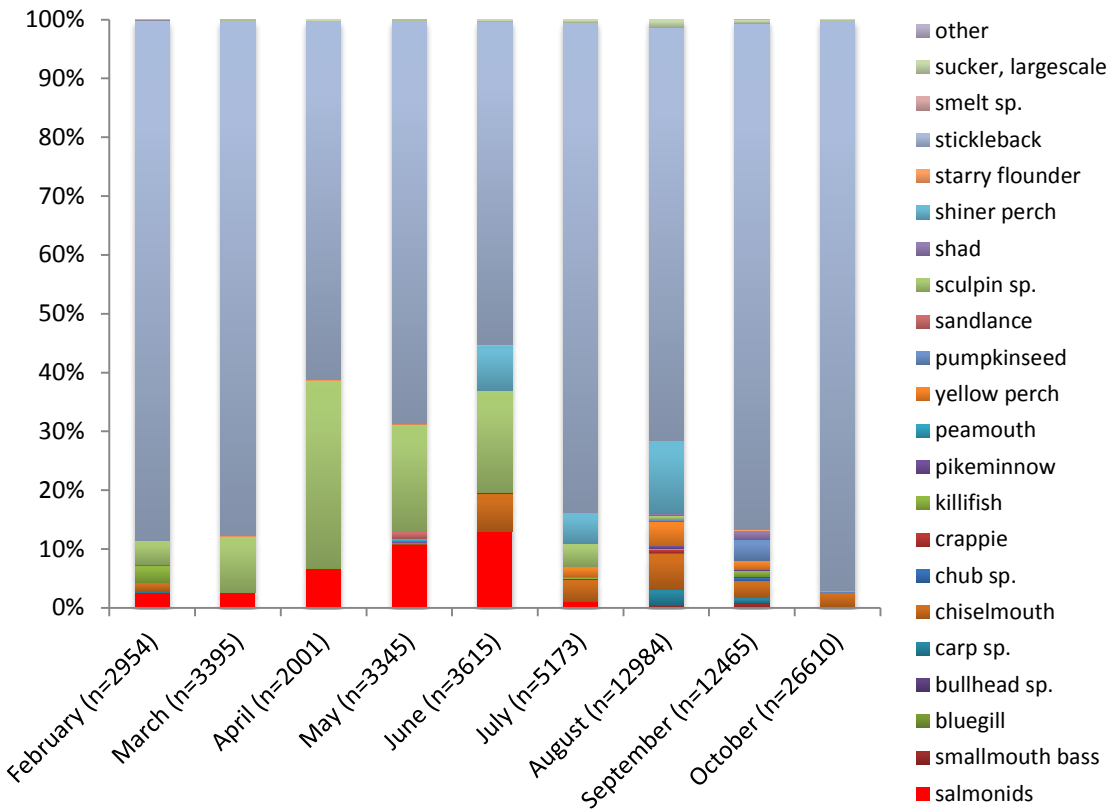
Figure 83. Species composition of 2012 catch by sampling month.

At the Reach B sites (Secret River and Welch Island), sampled for the first time in 2012, catch composition was fairly similar to Reach C (Figure 84). Sticklebacks were the most abundant species (96% and 91% of the total catch, respectively), followed by salmonids at Secret River (2.2% of the catch), and salmonids and chiselmouth at Welch Island (4.9% and 5.8% of the catch, respectively). Other species captured at these sites included yellow perch, killifish, chub, sculpin, starry flounder, largescale sucker, and shad. A total of 11 species were collected at both sites, but species diversity was
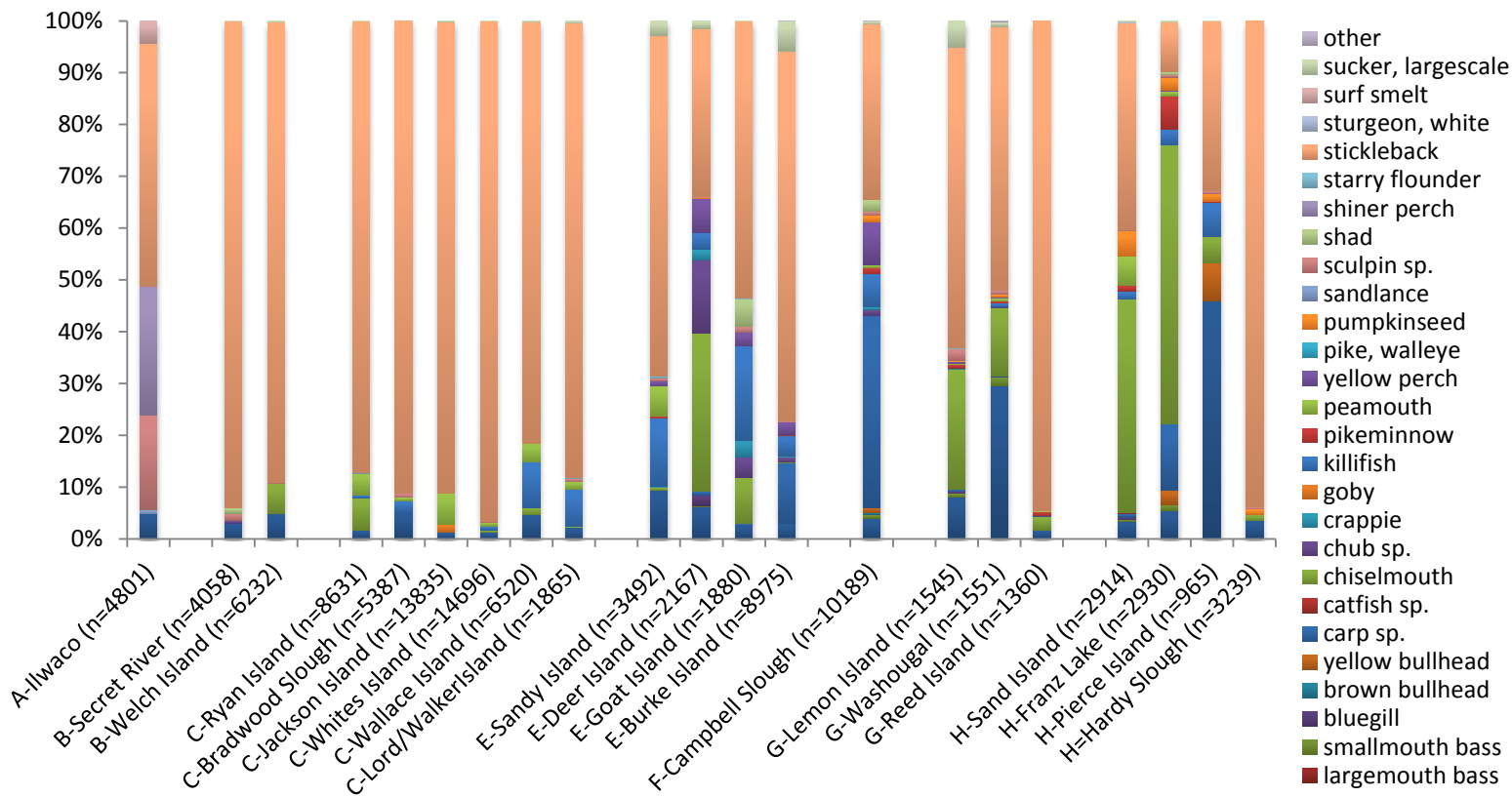
Figure 84. Fish community composition at the 2012 Ecosystem Monitoring sites, as compared to sites sampled in previous years. This figure includes data collected from February through October, pooled over the sampling season. For the sites sampled in multiple years (Ilwaco, Whites Island, Campbell Slough, and Franz Lake) data from all the years sampled are pooled.

Table 26. Species caught at the 2012 Ecosystem Monitoring sites, and percentage of total catch for each species.

date	# of fish	# of species	Chinook unmarked	Chinook marked	coho unmarked	Coho marked	chum	steelhead	trout, cuttthroat	trout, brown	stickleback	banded killifish	sculpin	pacific staghorn	riffle sculpin	pumpkinseed	bluegill	crappie	peamouth	chiselmouth	n. pikeminnow	perch, yellow	carp	sucker	Yel Bullhead	bass, smallmouth	perch shiner	mosquitofish	shad	chub, tui	chub, lake	starry flounder	mountain whitefish	pygmy whitefish	smelt	speckled dace
Ilwaco Slough																																				
2/21/12	310	5	0.7				0.3				36.1	26.1		36.8																						
3/19/12	515	4			0.2		0.2				37.5			62.1																						
4/19/12	1120	4					0.3				43.8	0.2		55.8																						
5/17/12	992	4	0.1								40.0	0.2		59.7																						
6/12/12	1655	5	0.1								47.0	0.1		36.1														16.8								
7/16/12	2842	5									83.9		1.2	5.2									<0.1					9.6									
8/13/12	4846	5									65.6	0.1	0.5	0.9														33.0									
10/16/12	26009	3									99.9		<0.1	<0.1																						
Total	**38289**	**9**	**<0.1**		**<0.1**		**<0.1**				**87.6**	**0.2**	**0.2**	**6.4**									**<0.1**					**5.6**								
Secret River																																				
2/21/12	980	4	2.6				0.3				96.9		0.2																							
3/20/12	241	3	1.2								97.9		0.8																							
4/18/12	250	7	18.0	0.4	1.2	0.4	1.6				78.0		0.4																							
5/16/12	42	7	2.4	2.5	7.1						81.0	2.4	2.4																		2.4					
6/13/12	206	6	12.1		1.9						83.5	0.5	1.5												0.5											
7/17/12	480	9	0.2	0.2	0.4						94.8		1.9							0.2			1.9								0.2	0.2					
8/13/12	2103	4			0.1						97.5			1.7																	0.8						
9/11/12	1078	4									99.6		0.1												0.2								0.1				
10/16/12	135	4									89.6		5.9								0.7				3.7												
Total	**5515**	**14**	**1.8**	**0.1**	**0.2**	**<0.1**	**0.1**				**95.9**	**<0.1**	**0.5**	**0.6**						**<0.1**	**<0.1**		**0.2**		**0.2**						**0.3**	**<0.1**			**<0.1**		
Welch Island																																				
2/20/12	1105	6	3.7	0.1							95.8	0.3											0.1										0.1				

date	# of species	Chinook unmarked	Chinook marked	coho unmarked	Coho marked	chum	steelhead	trout, cuttthroat	trout, brown	stickleback	banded killifish	sculpin	pacific staghorn	riffle sculpin	pumpkinseed	bluegill	crappie	peamouth	chiselmouth	n. pikeminnow	perch, yellow	carp	sucker	Yel Bullhead	bass, smallmouth	perch shiner	mosquitofish	shad	chub, tui	chub, lake	starry flounder	mountain whitefish	pygmy whitefish	smelt	speckled dace
3/22/12	2247	3	2.1								97.9																						<0.1				
4/18/12	296	4	5.8				0.7				93.2													0.3												
5/16/12	161	3	33.5	11.2							55.3																									
6/13/12	278	6	37.8	0.4	0.7						41.7									17.3				2.2												
7/17/12	97	5	16.5	1.0							77.3	2.1								3.1																
8/14/12	2048	8	0.1	0.1							84.1	0.1	0.1							15.0				0.5								0.1				
9/11/12	3288	6			<0.1						93.2	0.6								5.2				0.6								0.4				
10/16/12	429	3									96.0									3.5												0.5				
Total	**9949**	**10**	**2.8**	**0.2**	**<0.1**	**<0.1**					**90.6**	**0.3**	**<0.1**							**5.5**		**<0.1**		**0.4**								**0.2**				
Whites Island																																			
2/20/12	417	2	0.2								99.8																									
3/22/12	338	3	3.6	0.6							95.9																									
4/18/12	63	5	15.9	1.6			1.6				79.4																					1.6				
5/16/12	269	4	9.7								89.6		0.4						0.4																	
6/12/12	780	4	3.5								77.0		0.3							19.3																
7/16/12	83	6	8.4	1.2							85.5								1.2	2.4		1.2														
8/14/12	353	7		0.3							90.7		0.3							6.2	1.1	1.1										0.3				
9/11/12	6006	4									97.1									2.5				0.4								<0.1				
10/16/12	65	2									98.5									1.5																
Total	**8374**	**10**	**1.0**	**0.1**		**<0.1**					**94.5**		**0.1**						**<0.1**	**3.9**	**0.1**	**0.1**		**0.3**								**0.1**				
Campbell Slough																																			
5/15/12	1468	10	1.4	1.2							95.4	0.1	0.9		0.1				0.4			0.4		0.1	0.1											
6/14/12	204	5	9.8	1.5							85.8											1.0		2.0												

date		# of species	Chinook unmarked	Chinook marked	coho unmarked	Coho marked	chum	steelhead	trout, cuttthroat	trout, brown	stickleback	banded killifish	sculpin	pacific staghorn	riffle sculpin	pumpkinseed	bluegill	crappie	peamouth	chiselmouth	n. pikeminnow	perch, yellow	carp	sucker	Yel Bullhead	bass, smallmouth	perch shiner	mosquitofish	shad	chub, tui	chub, lake	starry flounder	mountain whitefish	pygmy whitefish	smelt	speckled dace
7/19/12	523	9	1.0	0.2							79.0	1.7	1.0							1.0		14.9		1.2											0.2	
8/13/12	1808	16									38.6	0.9	0.1			1.2	0.1	2.4	0.1	0.1	0.6	29.3	16.8	5.4	0.1	0.4			3.4	0.8						
9/6/12	756	15									0.7	12.3	0.1			13.5	0.1	1.5		0.7	0.9	24.5	7.2	2.1	2.0	10.8			23.5							0.3
Total	**4759**	**19**	**1.0**	**0.4**							**56.6**	**2.5**	**0.4**			**2.6**	**0.04**	**1.1**	**0.2**	**0.2**	**0.4**	**16.8**	**7.5**	**2.6**	**0.4**	**1.9**			**5.0**	**0.3**					**<0.1**	**<0.1**
Government/Lemon Island																																				
3/21/12	29	5	51.7				3.5				34.5	3.5												6.9												
4/17/12	35	8	8.6	2.9					5.7		40.0		14.3							5.7				20.0												2.9
5/15/12	98	9	30.6	3.1	13.3	2.0	1.0				41.8		5.1							0.00				1.0												2.0
6/13/12	175	6	18.3								53.7		12.0							14.9		0.6														0.6
7/18/12	552	10	2.9	0.5							90.4		0.7			0.2			0.7	1.8		0.4		2.2	0.2											
8/14/12	608	12									31.4					0.2	0.8	0.3		52.3	1.6	0.3	1.2	9.5		1.8				0.3						0.2
9/5/12	23	2										52.2								47.8																
10/17/12	66	10									43.9	3.0	1.5			18.2				22.7		1.5	3.0	1.5					1.5							3.0
Total	**1586**	**21**	**6.1**	**0.4**	**0.8**	**0.1**	**0.1**		**0.1**		**55.4**	**1.0**	**2.3**			**0.9**	**0.3**	**0.1**	**0.3**	**24.1**	**0.7**	**0.4**	**0.6**	**5.1**	**0.1**	**0.7**			**0.1**	**0.1**						**0.4**
Washougal																																				
2/22/12	30	7	3.3		3.3	3.3					76.7	6.7	3.3						3.3																	
3/21/12	23	5	26.1								60.9		4.4											4.4												4.4
4/16/12	237	4	16.0						0.4		81.0		2.5																							
5/14/12	277	7	39.7			22.0					33.6								2.2	1.4	0.7															0.4
6/11/12	317	9	22.4	56.8							17.0								0.3	1.6	0.3	0.3	0.3							1.0						
7/17/12	597	6									68.0									29.8	0.2	0.3	0.2	1.5												
8/14/12	100	9									26.0	8.0	4.0			9.0				17.0	2.0			7.0	1.0	26.0										
9/5/12	469	6									32.4	1.1	1.1			64.0				1.1			0.4													

date	n	# of species	Chinook unmarked	Chinook marked	coho unmarked	Coho marked	chum	steelhead	trout, cuttthroat	trout, brown	stickleback	banded killifish	sculpin	pacific staghorn	riffle sculpin	pumpkinseed	bluegill	crappie	peamouth	chiselmouth	n. pikeminnow	perch, yellow	carp	sucker	Yel Bullhead	bass, smallmouth	perch shiner	mosquitofish	shad	chub, tui	chub, lake	starry flounder	mountain whitefish	pygmy whitefish	smelt	speckled dace
Total	**2050**	**17**	**11.0**	**8. 8**	**0.1**	**3.0**		**0.1**			**46.8**	**0.7**	**0.8**			**15.1**			**0.4**	**10.2**	**0.3**	**0.2**	**0.2**	**0.8**	**0.1**	**1.3**							**0.2**			**0.1**
Reed Island																																				
2/22/12	22	4	4.6								86.4									4.6			4.6													
8/16/12	820	6			2.2						92.0									4.2	1.2	0.1	0.4													
Total	**842**		**0.1**		**2.1**						**91.8**									**4.2**	**1.2**	**0.1**	**0.5**													
Franz Lake Slough																																				
2/22/12	90	7						1.1			38.9	2.2	4.4							51.1													1.1	1.1		
3/21/12	2	1										100.0																								
8/15/12	333	7									52.9		1.5			2.1		3.3		24.3	10.2		5.7													
9/5/12	213	10									60.1		0.5			9.4	1.9	1.9		0.5			18.8	4.2		2.4										0.5
10/17/12	902	8			0.3					0.1	17.7	0.8	3.0			2.8				72.4				2.9												
Total	**1540**	**17**			**0.2**			**0.1**		**0.1**	**32.4**	**0.7**	**2.4**			**3.4**	**0.3**	**1.0**		**50.7**	**2.2**		**3.8**	**2.3**		**0.3**							**0.1**	**0.1**		**0.1**

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slightly higher at Welch Island than at Secret River (0.45 vs. 0.33; Figure 85). Of the species collected at Secret River and Welch Island, 27% and 20%, respectively were non-native species (Figure 86). Non-native species found at Secret River were killifish, yellow perch, and shad, with shad the most abundant. At Welch Island, killifish and yellow perch were present. At Secret River, these species accounted for 1.2% of the total catch, while at Welch Island they accounted for only 0.1% of the total catch.

At the Reach G sites (Government/Lemon Island, Washougal, and Reed Island), also sampled for the first time in 2012, species composition was somewhat variable (Figure 84). At Reed Island, the catch was dominated by sticklebacks (95% of the catch), with chiselmouth and salmonids accounting for an additional 2.6% and 1.5% of the catch. At Government/Lemon Island and Washougal, stickleback were also the most abundant species, accounting for 58% and 51% of the total catch, but other species were also present in significant numbers. At Washougal, salmon species made up 30% and chiselmouth made up 13% of the catch, while at Government/Lemon Island, salmonids made up 8.1% of the catch and chiselmouth made up 23% of the catch. A total of 20 species were collected at Government/Lemon Island, and 18 species at Washougal, but only 9 species at Reed Island (Figure 85). Species diversity was comparable at Government/Lemon Island and Washougal (1.34 and 1.63), values similar to those observed at Campbell Slough and some of the Reach H sites (Figure 85). At Reed Island, species diversity was much lower (0.29), similar to Hardy Slough in Reach H or to the sites in Reaches B and C. Of the species collected at the Reach G sites, 39-40% were non-native at Government/Lemon Island and Washougal, and 33% at Reed Island (Figure 86). A number of non-native species were found at Government/Lemon Island and Washougal, including smallmouth bass, bullheads, bluegill, carp, crappie, killifish, yellow perch, and pumpkinseed. An uncommon species found at Washougal was specked dace. At Reed Island, non-native species included carp, killifish, and yellow perch. At Government/Lemon Island and Washougal, non-native species accounted for 2.2% and 3.5% of the total catch, but at Reed Island they accounted for only 0.4%.

At the trend sites (Figure 87), patterns of fish community composition were quite similar from year to year at Ilwaco Slough and Whites Island. In both 2011 and 2012, catches at Ilwaco Slough were comprised mainly of stickleback, shiner perch, and staghorn sculpin, in fairly equal proportions. At Whites Island, catches were consistently dominated by stickleback. Patterns were more variable at Campbell Slough and Franz Lake Slough. At Campbell Slough, carp made up 30-40% or more of the catch from 2007 to 2010, but were much less abundant in 2011 and 2012. Carp were also abundant at Franz Lake Slough in 2008, but found in much smaller numbers in subsequent years. The proportion of stickleback in the catch also increased at Franz Lake Slough in 2011 and 2012.

Franz Lake Slough and Campbell Slough consistently had the greatest species richness or total number of species captured, generally ranging from 16 to 20 total species (Figure 88). The number of species captured at Franz Lake Slough was unusually low in 2012 (11 species), a year when the sampling period was very limited due to high water. At Ilwaco Slough and Whites Island, the number of species collected was quite stable, ranging from 8-11 species over the multi-year sampling period. Fish species assemblages were also analyzed for fish species diversity using the Shannon–Wiener diversity index (Shannon and Weaver 1949) (Figure 88). Although species diversity varied somewhat from year to year at the sampling sites, but the highest values were consistently found at Franz Lake Slough and Campbell Slough. Species diversity was consistently lowest at Whites Island.

In addition to increased species richness and diversity, Franz Lake Slough and Campbell Slough had higher percentages of non-native fish species than either Ilwaco Slough or Whites Island (Figure 89). At Campbell Slough, from 47-60% of the species collected were non-native, while at Franz Lake Slough,

from 35-55% of species captured were non-natives. At Whites Island, from 11-22% of species collected were non-native, and at Ilwaco Slough, 20-33% of species collected were non-native. At Campbell Slough and Franz Lake Slough, non-native species also made a significant proportion of the catch, ranging from 31-75% of the catch at Campbell Slough and from 3.9 to 54% of the catch at Franz Lake Slough. In contrast, at Whites Island, the proportion of non-native species was consistently very low, ranging from .06 to 1.6% of the total catch.
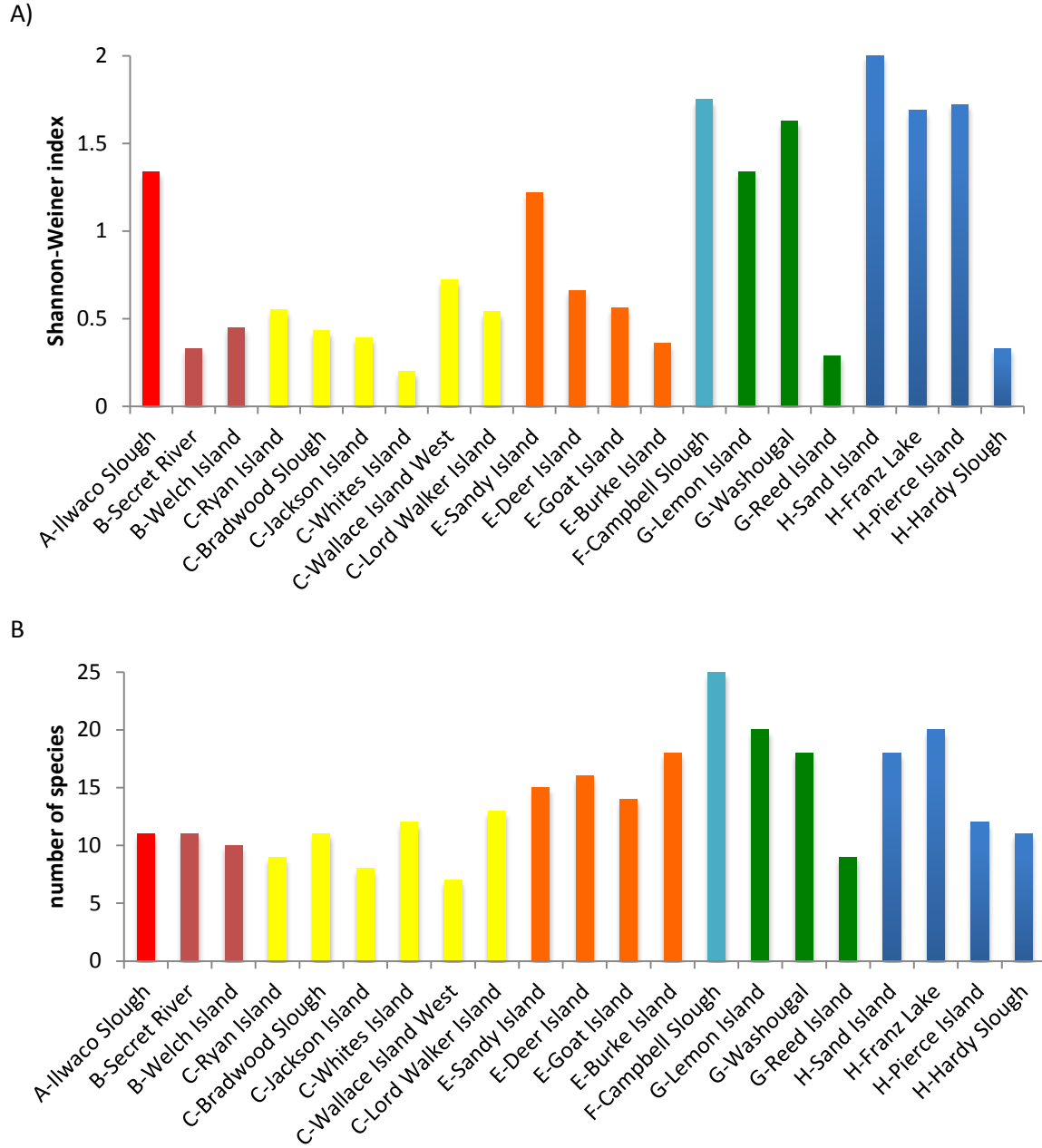
Figure 85. A) Species diversity (Shannon Weiner) and B) species richness (total number of species captured) at 2012 Ecosystem Monitoring sites, as compared to previously sampled sites.

A

B
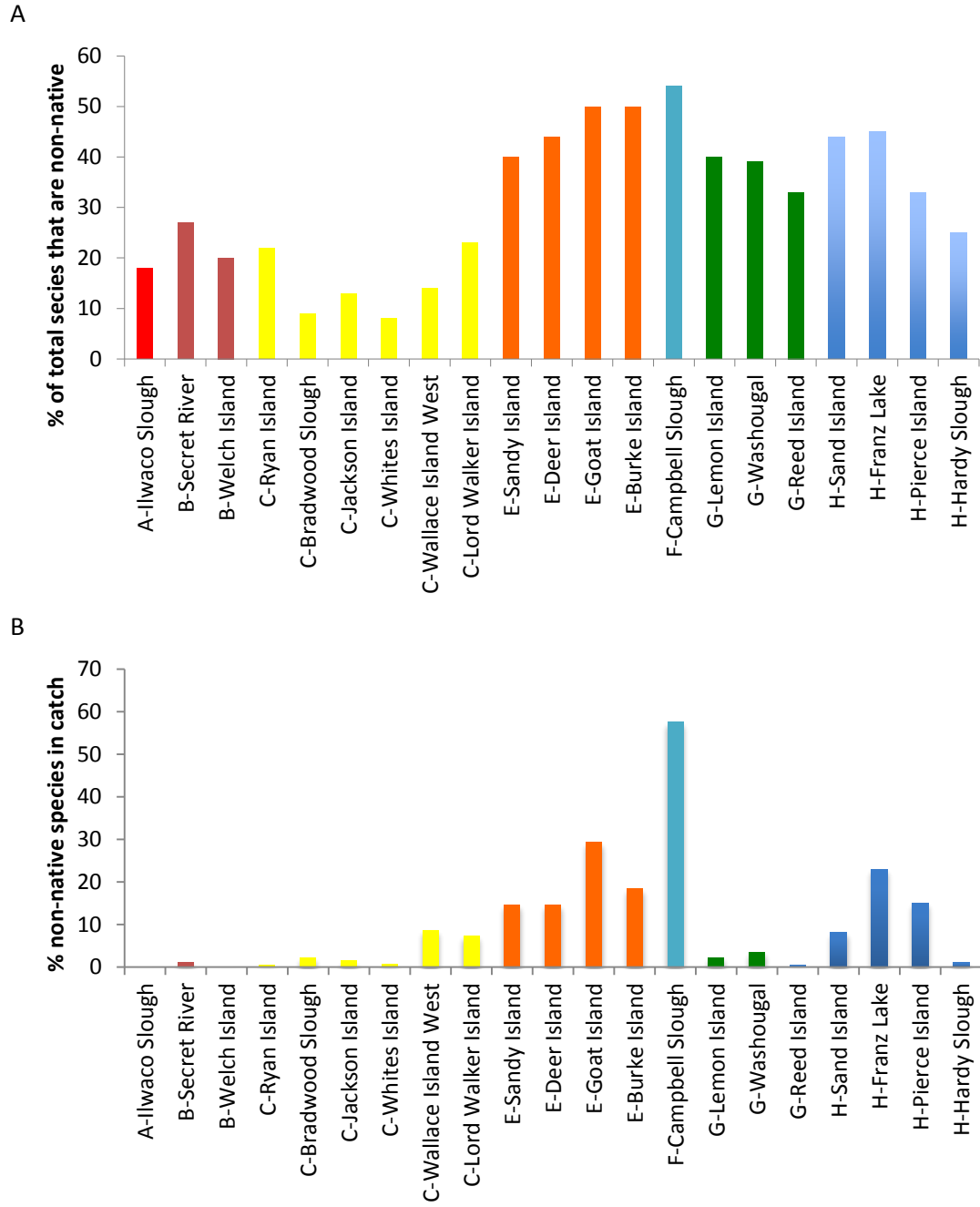
Figure 86. Proportions of non-native species at the 2012 Ecosystem Monitoring sites, as compared to earlier sampling sites, as A) the percentage of species captured and B) as the percentage of total catch.
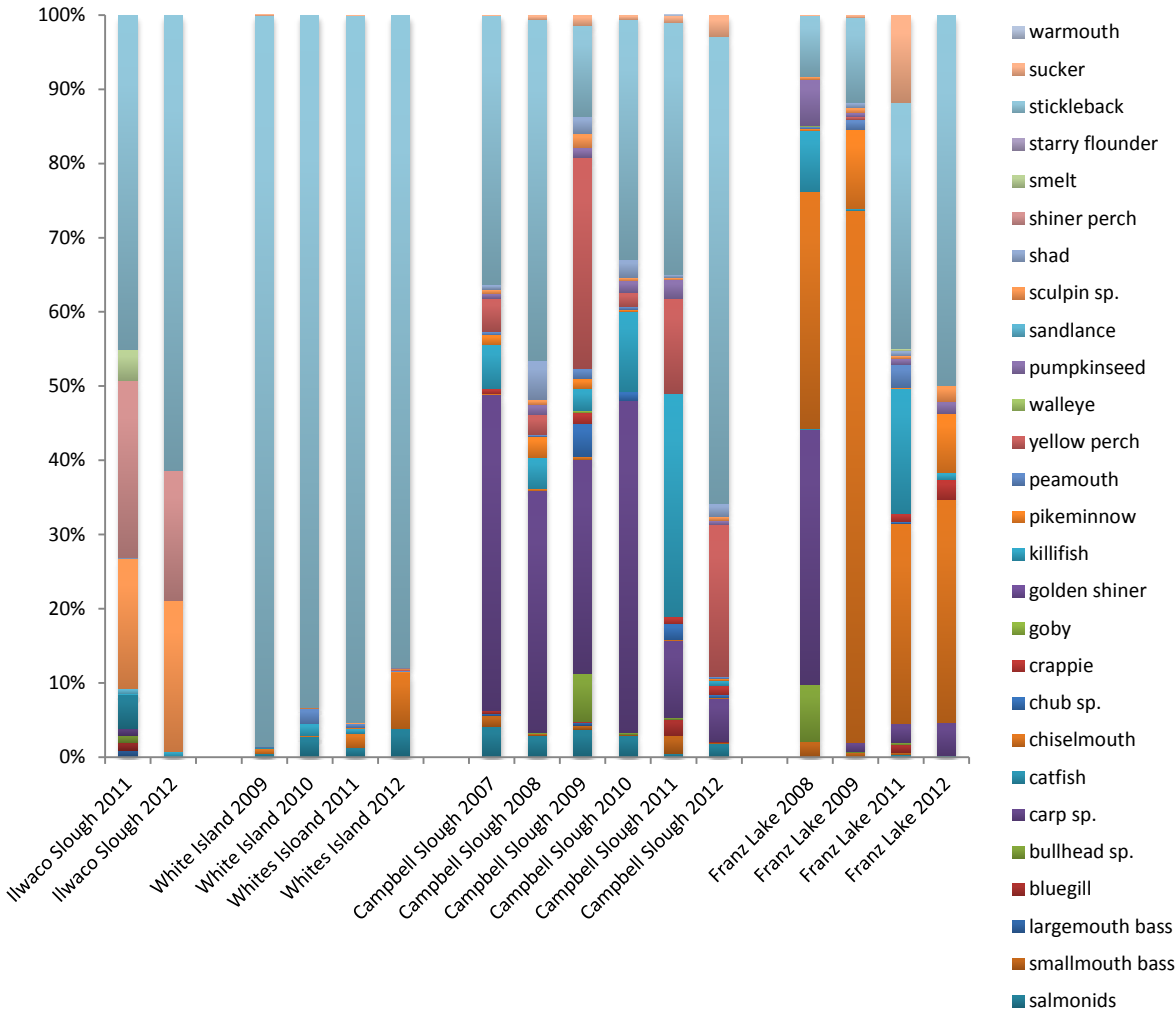
Figure 87. Trends in fish community composition by sampling year at the Ecosystem Monitoring trend sites.
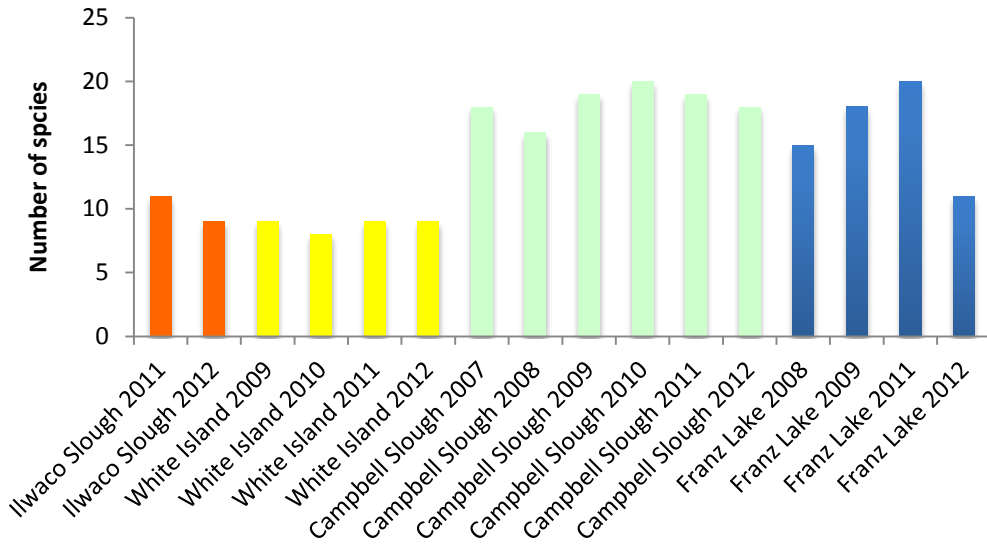
B)
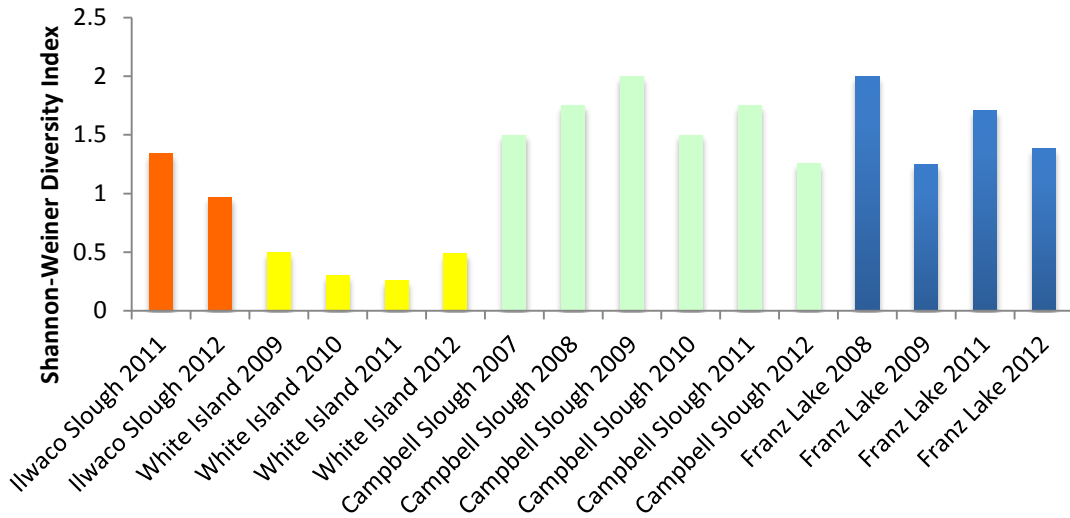
Figure 88. Trends in proportions of non-native species at the Ecosystem Monitoring trend sites, as A) the percentage of species captured and B) as the percentage of total catch. Percentages of non-native fish in catches at Ilwaco Slough and Whites Island were very low, generally < 1%.

A

% non-native species

B

% non-native species in catch
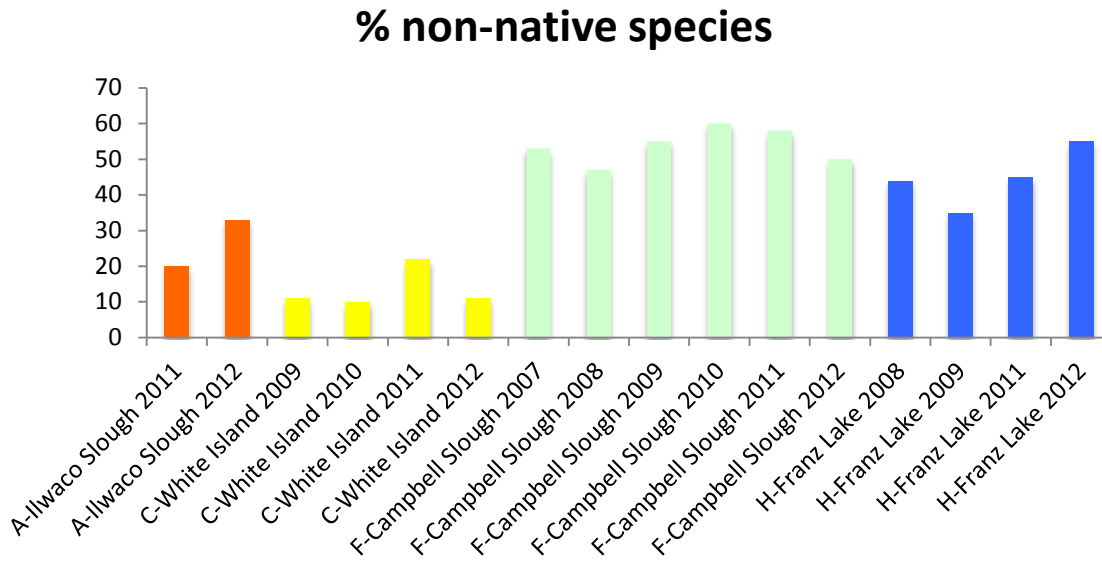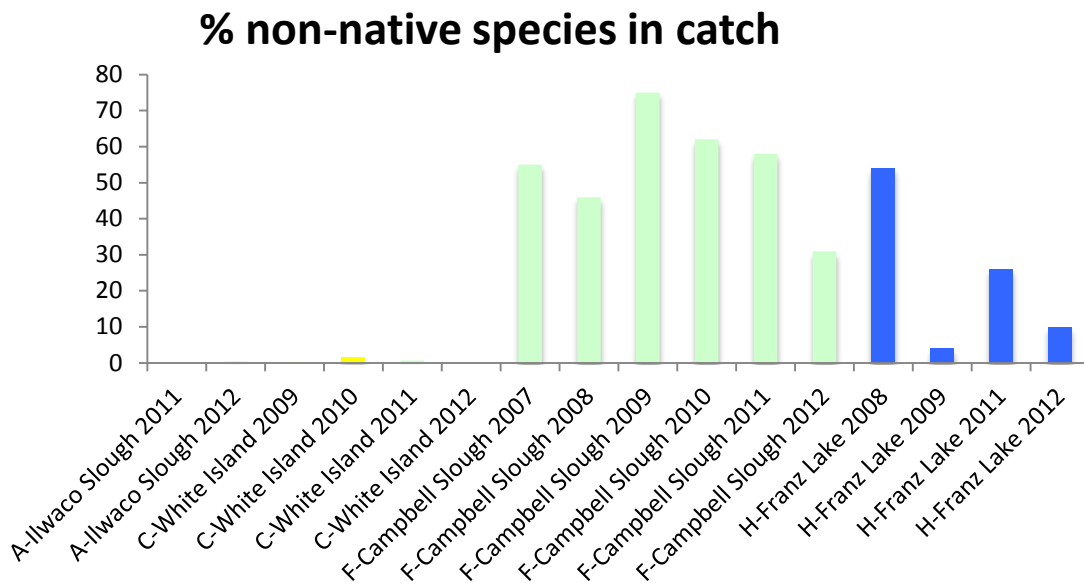
Figure 89. Trends in proportions of non-native species at the Ecosystem Monitoring trend sites, as A) the percentage of species captured and B) as the percentage of total catch. Percentages of non-native fish in catches at Ilwaco Slough and Whites Island were very low, generally < 1%.

4.5.1.1   *Salmon Species Composition*

In 2012, as in other sampling years, Chinook salmon were the most abundant juvenile salmon species overall, representing 89% of all salmonids captured, as well as the most abundant salmon species at the majority of sites sampled (Figure 90). Considering the sites sampled for the first time in 2012, Chinook represented from 83% of the salmonid catch at Secret River and 99% of the salmonid catch at Welch Island. In Reach G, Chinook salmon accounted for 85-86% of the salmonid catch at Government/Lemon Island and Washougal. At Reed Island, however, they accounted for only 5% of the catch. Overall, coho salmon made up 9.6% of the total salmonid catch. Of the sites sampled for the first time in 2012, coho made up the highest proportion of the catch at Reed Island in Reach G (95%).   At Government/Lemon Island and Washougal, they made up 12% and 13% of the salmonid catch. Coho salmon made up 11% of the salmon catch at Secret River, but only 0.7% of the catch at Welch Island. Chum salmon accounted for 1.4% of the salmonid catch overall. They made up 5.6% of the salmonid catch at Secret River, 0.7% of the catch at Welsh Island, and 1.6% of the total salmonid catch at Government/Lemon Island, but were not found at Washougal or Reed Island.

At the trend sites (Figure 90), Chinook salmon consistently dominated catches at Whites Island and Campbell Slough. At Ilwaco, chum made up a substantial proportion of the catch in both 2011 and 2012, but in 2012, Chinook and coho salmon were collected as well as chum. At Franz Lake Slough, catch composition was more variable. Between 2008 and 2011, the proportion of coho salmon in the catch increased from 34% to 85%. In 2012, only one steelhead trout was capture at Franz Lake Slough, likely because the site could not be sampled for most of the season.

Both marked (hatchery) and unmarked (presumably wild) Chinook and coho salmon were found at the 2012 EMP sampling sites (Figure 91). At the Reach B sites, almost all the Chinook salmon collected were unmarked, 97% at Secret River and 92% at Welch Island. In Reach G, the percentage of hatchery Chinook was higher. At Government/Lemon Island, the majority of Chinook salmon (93%) were unmarked, but at Washougal, only 55% were unmarked. The only Chinook salmon found at Reed Island was a marked fish.   As for coho salmon, at Secret River, Welch Island, Government/Lemon Island, and Reed Island, the majority of coho salmon collected (87-100%) were unmarked. However, at Washougal, 61 of the 62 coho salmon collected (98%) were marked fish.

As for trends at the trend sites, at Ilwaco Slough and Whites Island, unmarked Chinook salmon dominated in all sampling years (Figure 92). At Campbell Slough, the proportions of marked and unmarked Chinook salmon was variable, with marked Chinook salmon making up 50% of more of the catch in most years. In 2012, however, unmarked Chinook were more abundant, accounting for 69% of Chinook salmon caught.   Coho salmon were only found consistently at Franz Lake Slough, but the percentage of unmarked Coho increased between 2008 and 2011, from 6% of the catch to 100% of the catch.

Generally, unmarked Chinook salmon were present at the sampling sites from February, when sampling began, through July, with the highest number of fish found in May and June (Figure 93). Marked Chinook were found over the same time-period, but only rarely in February, March, April or July. Almost all marked fish were seen in May and June. Unmarked Coho salmon were found throughout the sampling season. Marked coho were found in February, April and May, but the only large catch was in May, at the Washougal sampling site. Chum salmon from found from February through April, with the largest number of fish found in April. Seasonal patterns of salmon species occurrence were generally

consistent with these overall patterns at the individual sampling sites (Figure 94), although not all sites could be sampled each month, and certain stocks or species were more prevalent at some sites than others, as noted above.

A

B
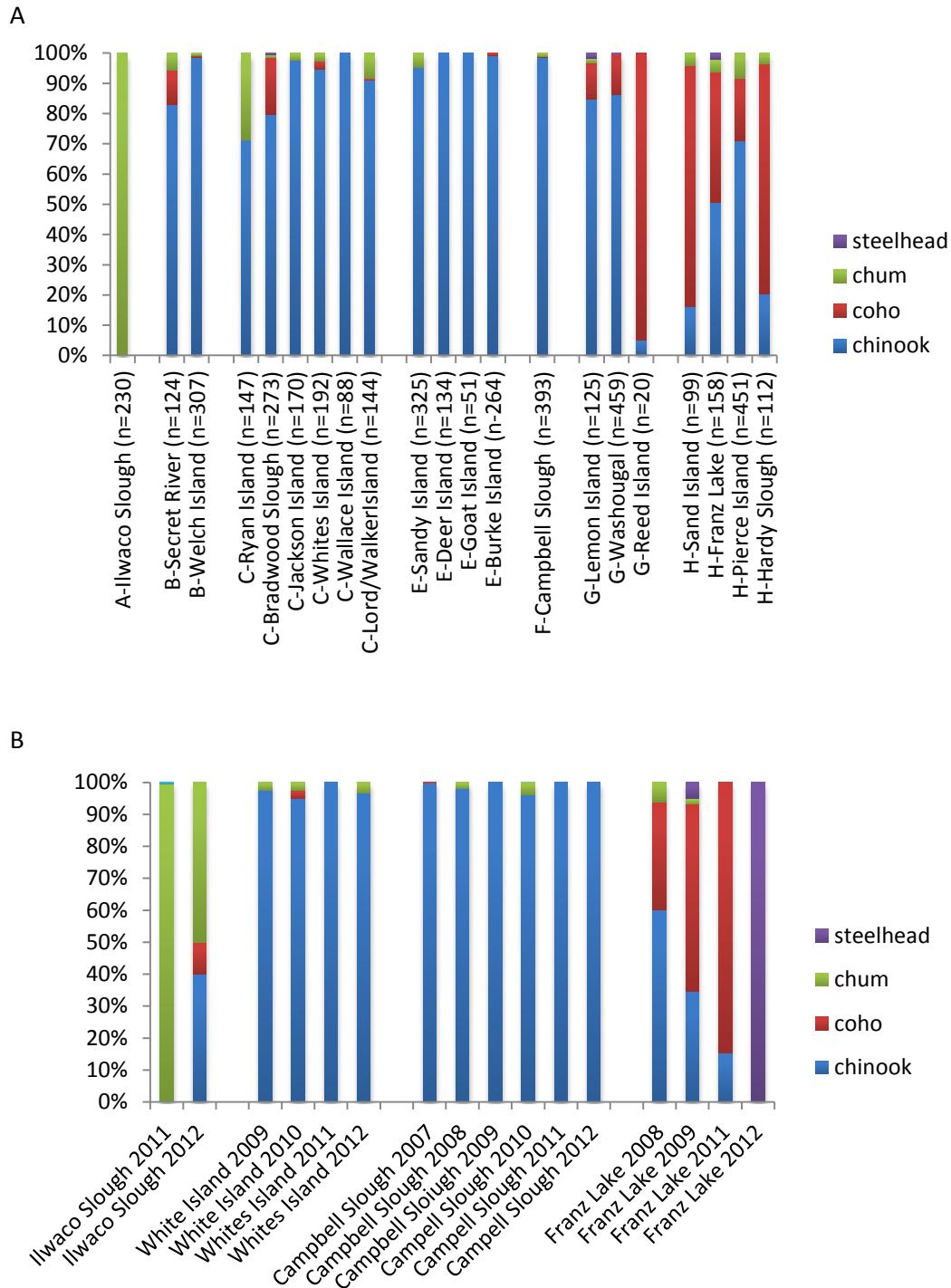
Figure 90. Composition of the salmonid catch at the A) 2012 Ecosystem Monitoring sites, as compared to sites sampled in previous years and B) trend Ecosystem Monitoring sites by year.

A

B
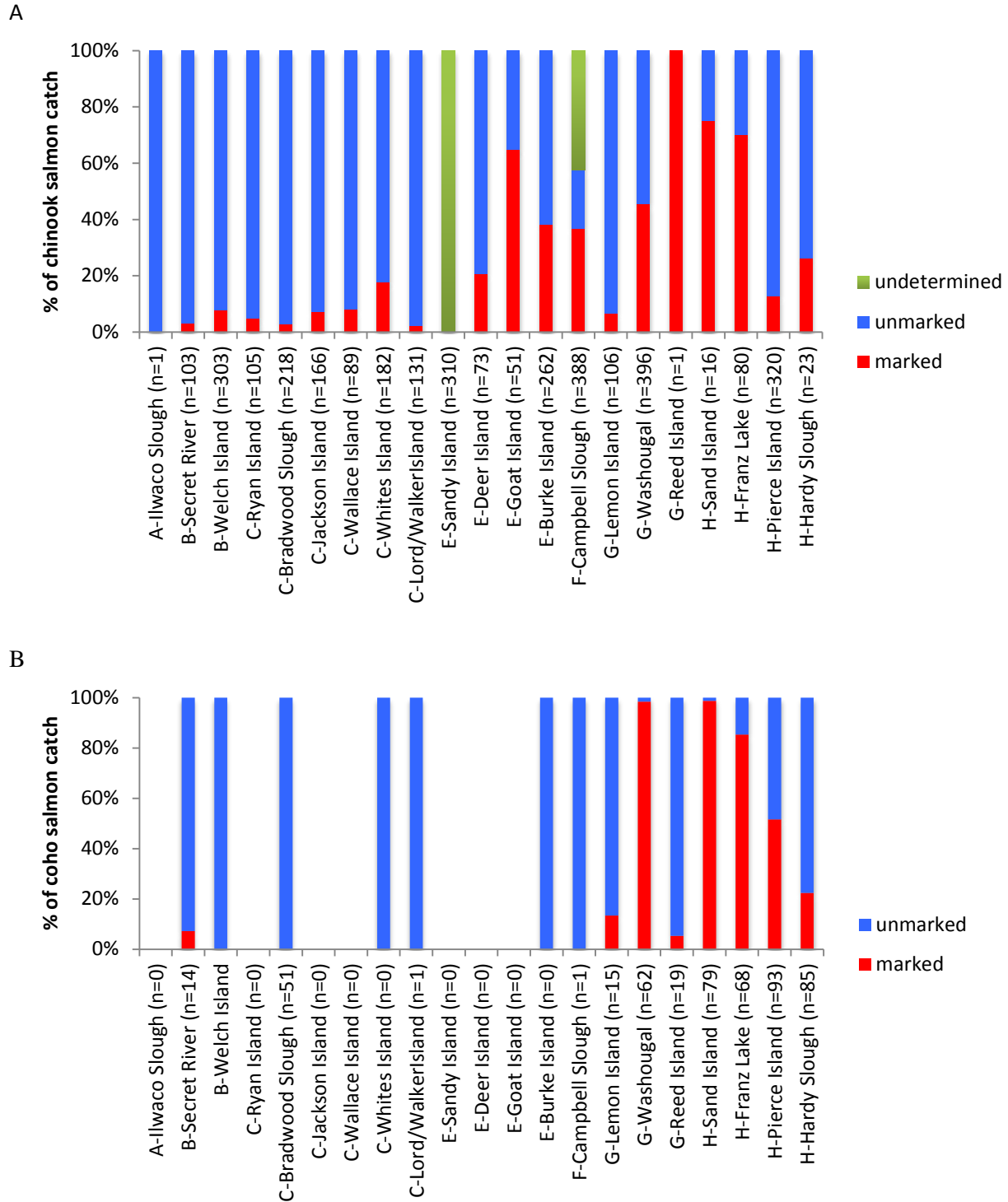
Figure 91. Proportions of unmarked and marked A) Chinook salmon and B) coho salmon in catches at the 2012 Ecosystem Monitoring sites, as compared to sites sampled in previous years.
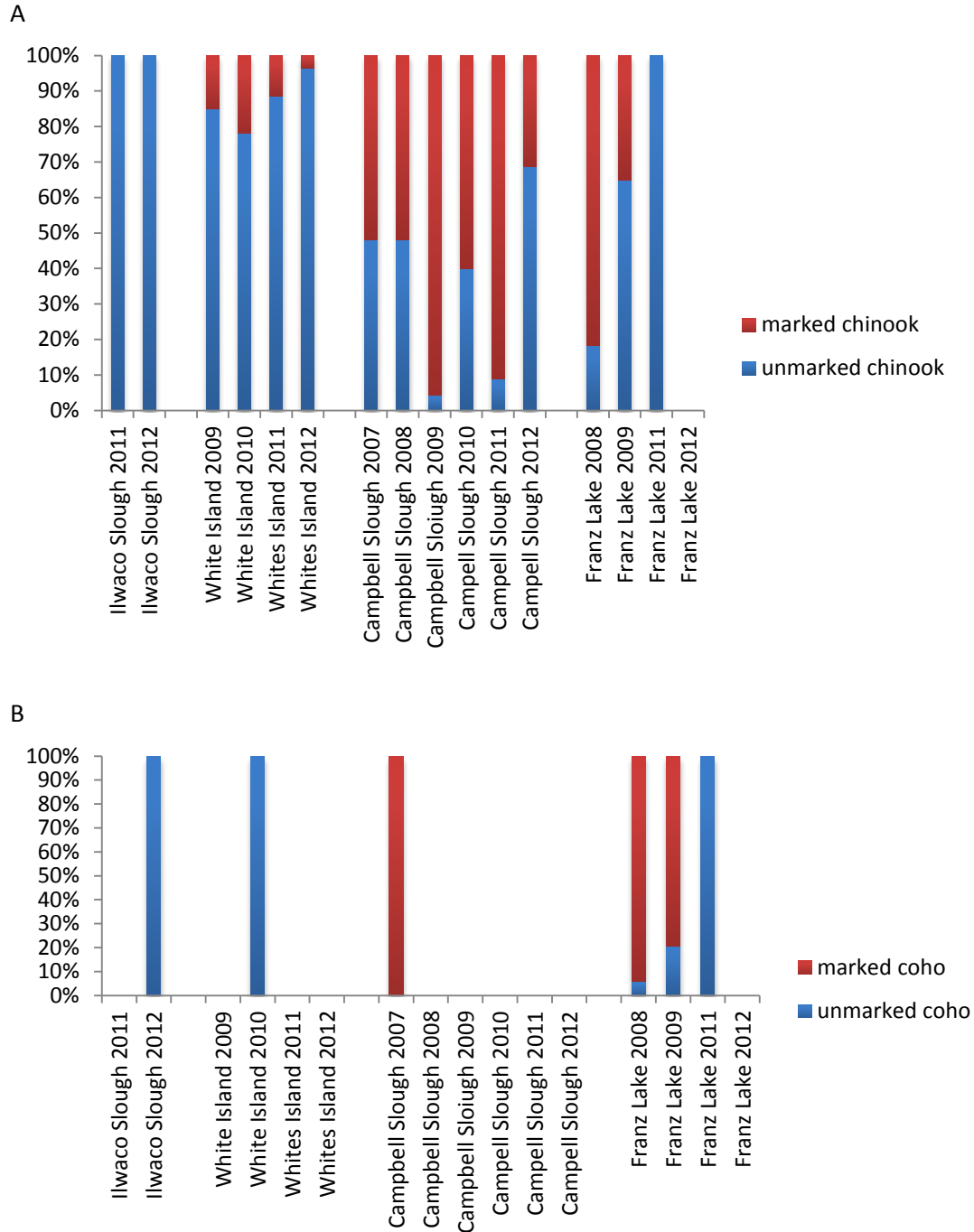
Figure 92. Proportions of unmarked and marked A) Chinook salmon and B) coho salmon in catches at the trend Ecosystem Monitoring sites by year.
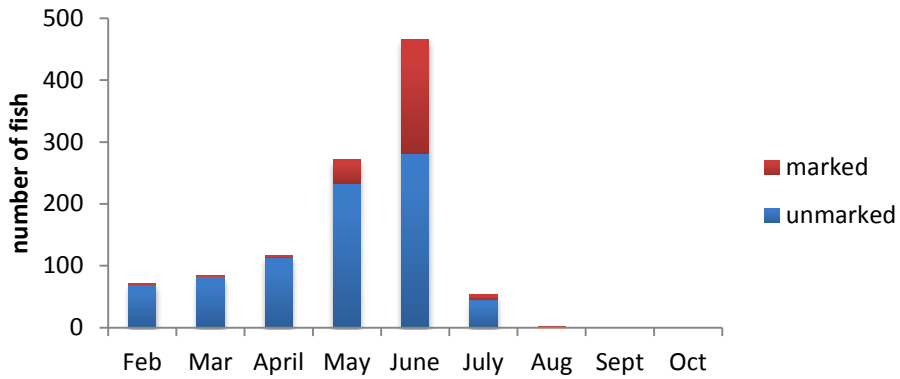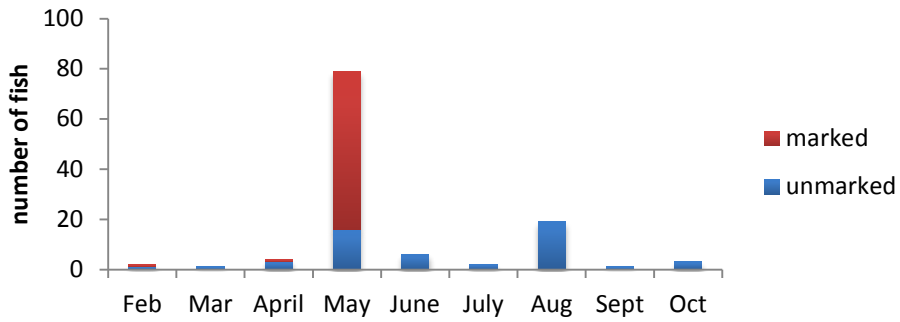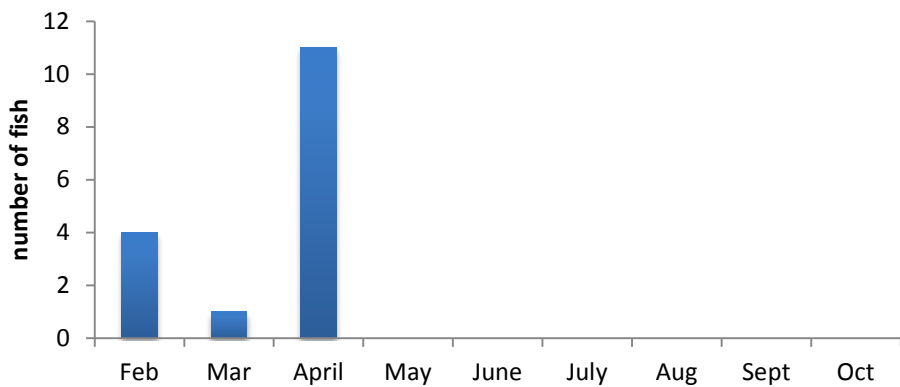
Figure 93. Seasonal occurrence of salmon species at the 2012 Ecosystem Monitoring Sampling sites.
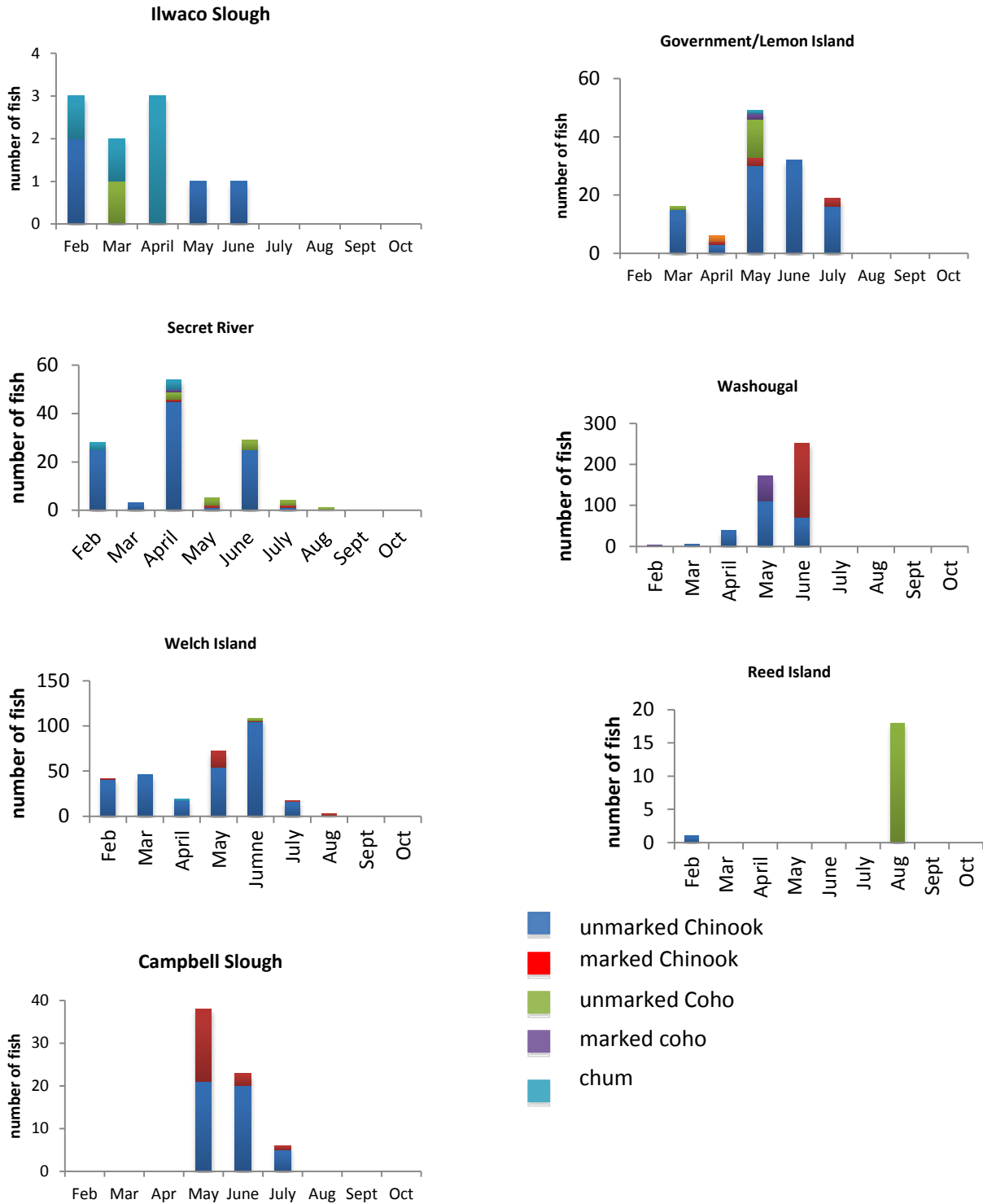
Figure 94. Seasonal distribution of Chinook, coho, and chum salmon at the 2012 Ecosystem Monitoring sampling sites. Note different scales on the y-axis.

4.5.1.2 *Salmon Density*

NOAA NMFS are currently the process of analyzing these data and they are not available for this report.

4.5.2 Salmon Metrics

4.5.2.1 *Genetic Stock Identification*

Genetic stock analyses for Chinook salmon sampled in 2011 (from Whites Island, Deer Island, Burke Island, and Campbell Slough) were completed during 2012, and are shown in Figure 95. At the Reach E sites sampled for the first time in 2011 (Goat Island, Burke Island and Deer Island), West Cascades fall Chinook made up the majority (75% and 83%) of unmarked Chinook at Goat and Burke Islands, while at Deer Island, West Cascades fall Chinook and Spring Creek Group fall Chinook each made up 40% of the Chinook salmon analyzed. Other stocks present included Upper Columbia summer/fall Chinook at Deer Island, Spring Creek Group fall Chinook at Goat Island, and West Cascades spring Chinook at Burke Island. The majority of marked fish from all three sites were Spring Creek Group fall Chinook, but some marked fish from other stocks were present, including West Cascades fall Chinook at Burke Island, and Upper Columbia fall Chinook and Upper Willamette spring Chinook at Deer Island. Overall, the stock composition of unmarked Chinook from the new Reach E sites most closely resembled that of unmarked Chinook from Reach C, while the composition of the marked fish more closely resembled that of marked Chinook from Reach F-H. At Ilwaco Slough only one unmarked and no marked Chinook salmon were collected in 2011; the single unmarked Chinook that was analyzed was a Spring Creek Group fall Chinook.

At the trend sampling sites (Figure 96) stock composition of the 2011 samples from Whites Island was very similar to that observed in earlier years for unmarked Chinook, and very similar to the 2010 results for marked Chinook. In 2009, the samples were more diverse, but the number of marked Chinook sampled was very small (n=4). In 2011, only one unmarked Chinook salmon was analyzed from Campbell Slough, so its similarity to results for earlier years is difficult to evaluate. For marked Chinook, however, results were comparable to other years, with Spring Creek Group fall Chinook making up the majority of the fish sampled. At Franz Lake Slough, no genetics data are available for 2011, as the number of Chinook salmon collected was too small.

For both marked and unmarked Chinook salmon, there were clear seasonal patterns in stock occurrence in the estuary (Figure 97). For unmarked Chinook, West Cascades fall Chinook were present throughout the sampling season, but Spring Creek Group fall Chinook were more prevalent in April and May, while Upper Columbia summer/fall Chinook were found more frequently in June, July, and August. Other stocks were detected too rarely to demonstrate any clear seasonal patterns. For marked Chinook, Spring Creek Group fall Chinook predominated in April and May, then gradually declined, while West Cascades fall Chinook showed the opposite trend, became more abundant later in the sampling season. While number of other stocks were small, they also showed some seasonal patterns; Upper Willamette spring Chinook were observed in April and May, while Upper Columbia summer/fall Chinook were found in June and July.

Among unmarked Chinook, length, weight, and condition did not differ significantly among stocks, although West Cascades fall Chinook tended have lower, and Deschutes River fall and Willamette River spring Chinook have higher lengths and weights than other stocks (Figure 98). Hatchery fish were consistently larger than unmarked fish belonging to the same stock, and did show some significant differences in size among stocks (Figure 98). The length of Spring Creek Group fall Chinook was significantly greater than that of West Cascade fall and spring Chinook, and CF of Willamette River spring Chinook was significantly lower than that of any of the other stocks. Length and weight were particularly variable for Willamette River spring Chinook because this group included yearling and subyearling migrants.

Data on Chinook salmon sampled in 2012 are not yet available.
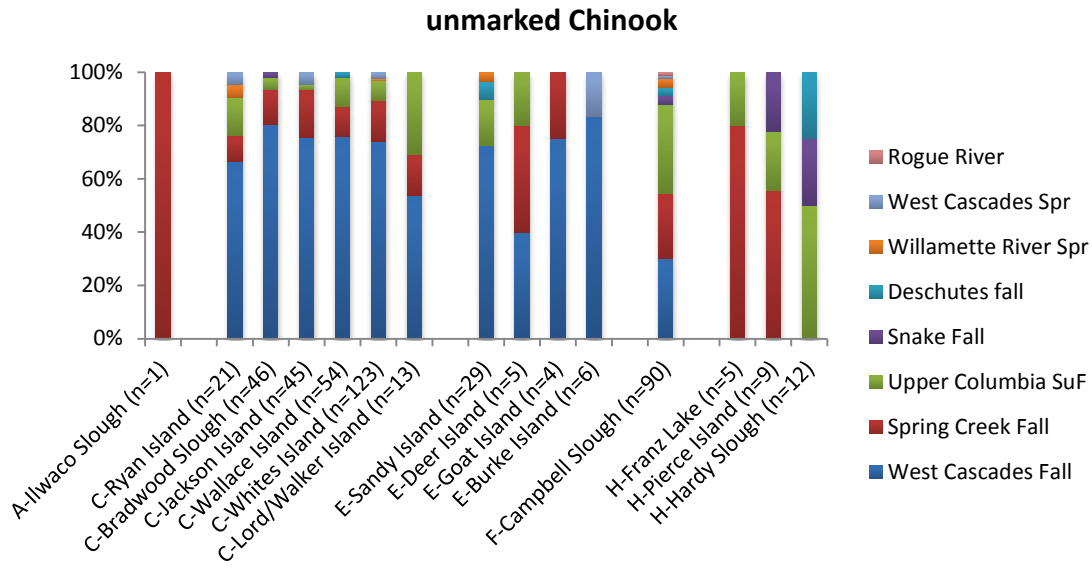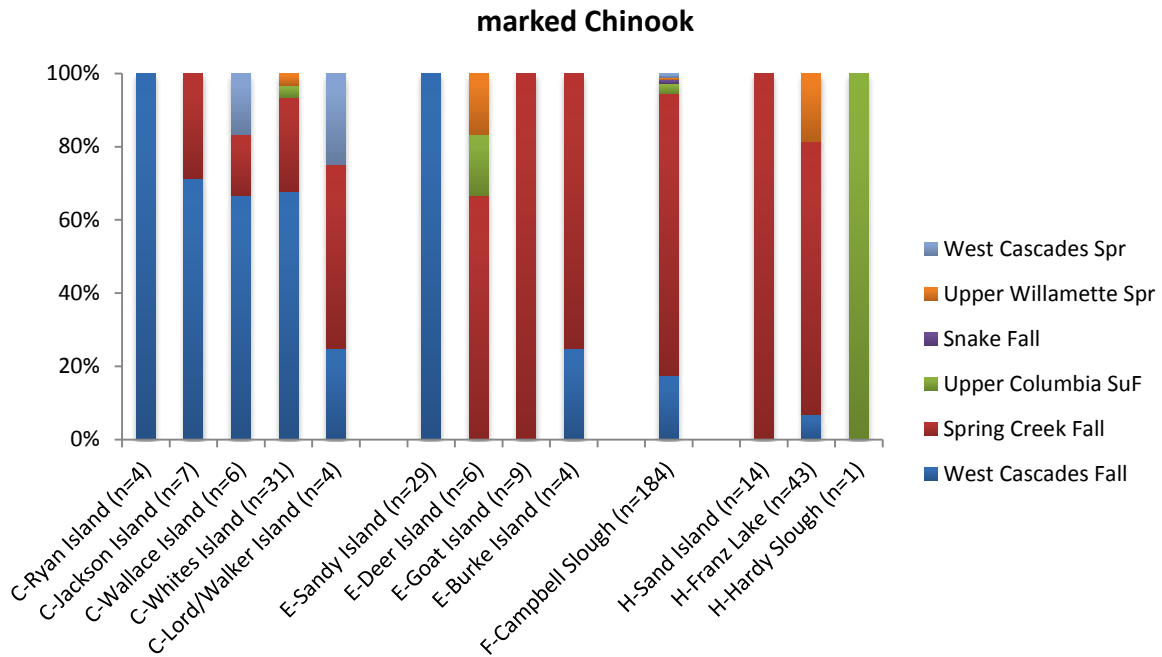
Figure 95. Genetic stock composition of A) unmarked and B) marked juvenile Chinook salmon at the 2012 EMP sampling sites, as compared to sites sampled in previous years.

A

Unmarked Chinook

Legend:
- West Cascades Spr
- Upper Willamette Spr
- Deschutes Fall
- Snake Fall
- Upper Columbia SuF
- Spring Creek Fall
- West Cascades Fall

B

Marked Chinook

Legend:
- West Cascades Spr
- Upper Willamette Spr
- Snake Fall
- Upper Columbia SuF
- Spring Creek Fall
- West Cascades Fall
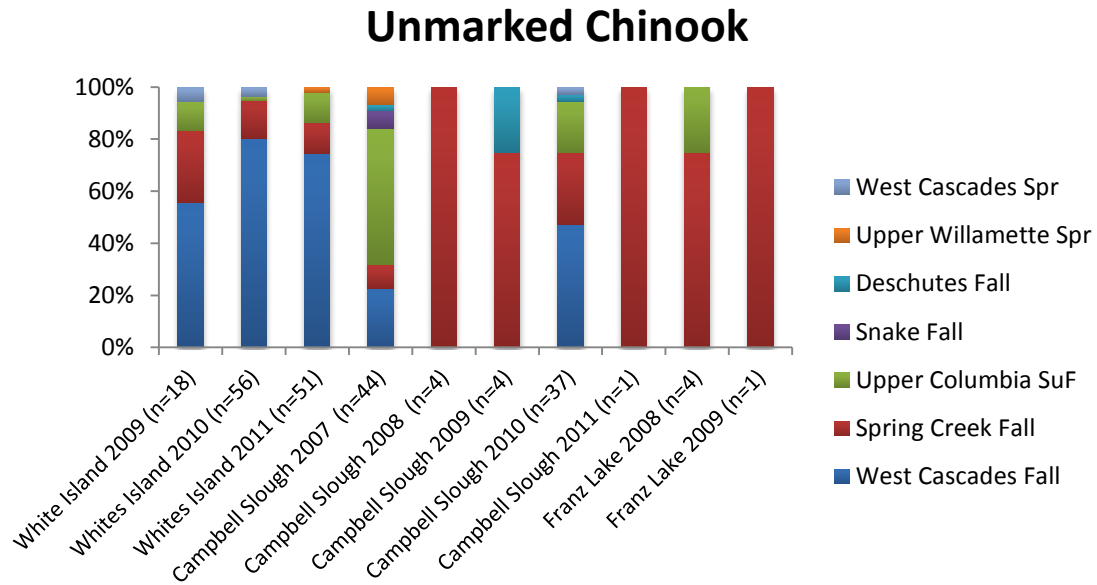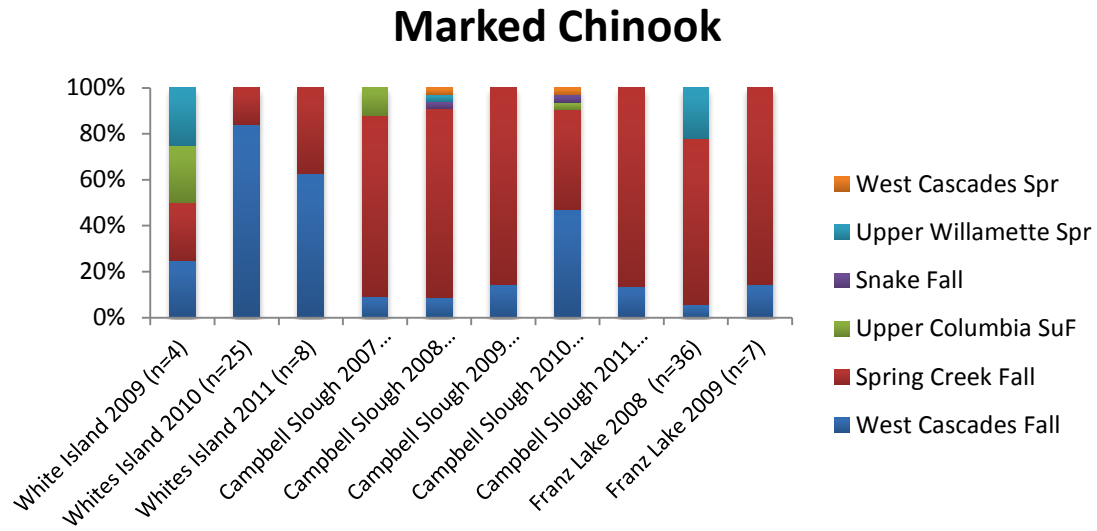
Figure 96. Year to year variation in genetic stock composition of A) unmarked and B) marked juvenile Chinook salmon sampled from the EMP trend sites.
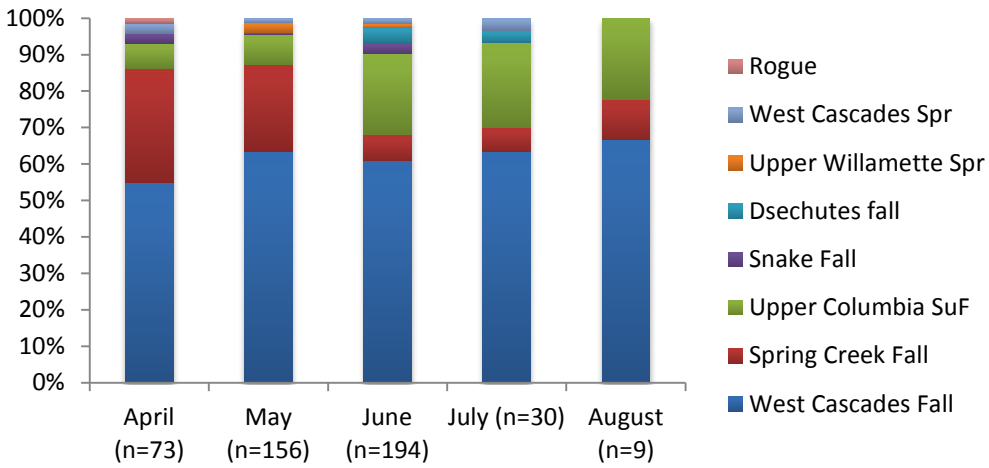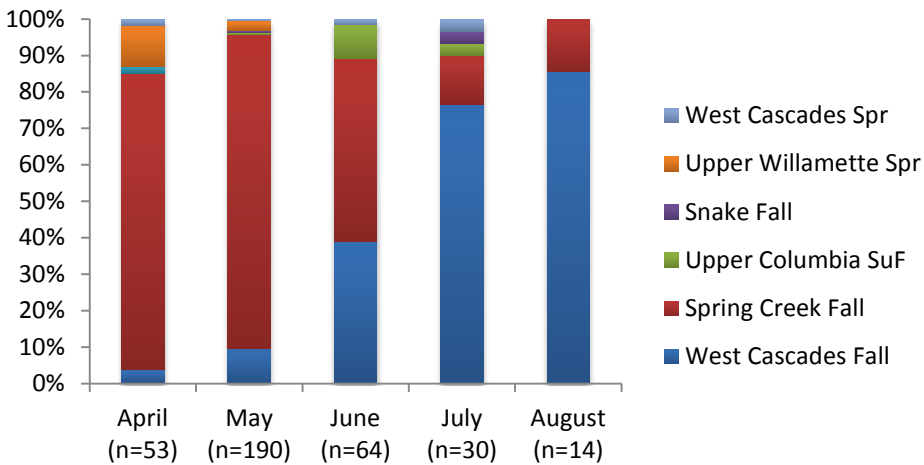
Figure 97. Seasonal patterns of Chinook salmon stock occurrence at the Ecosystem Monitoring sampling sites. All fish sampled from 2008 to 2011 are included in this analysis.
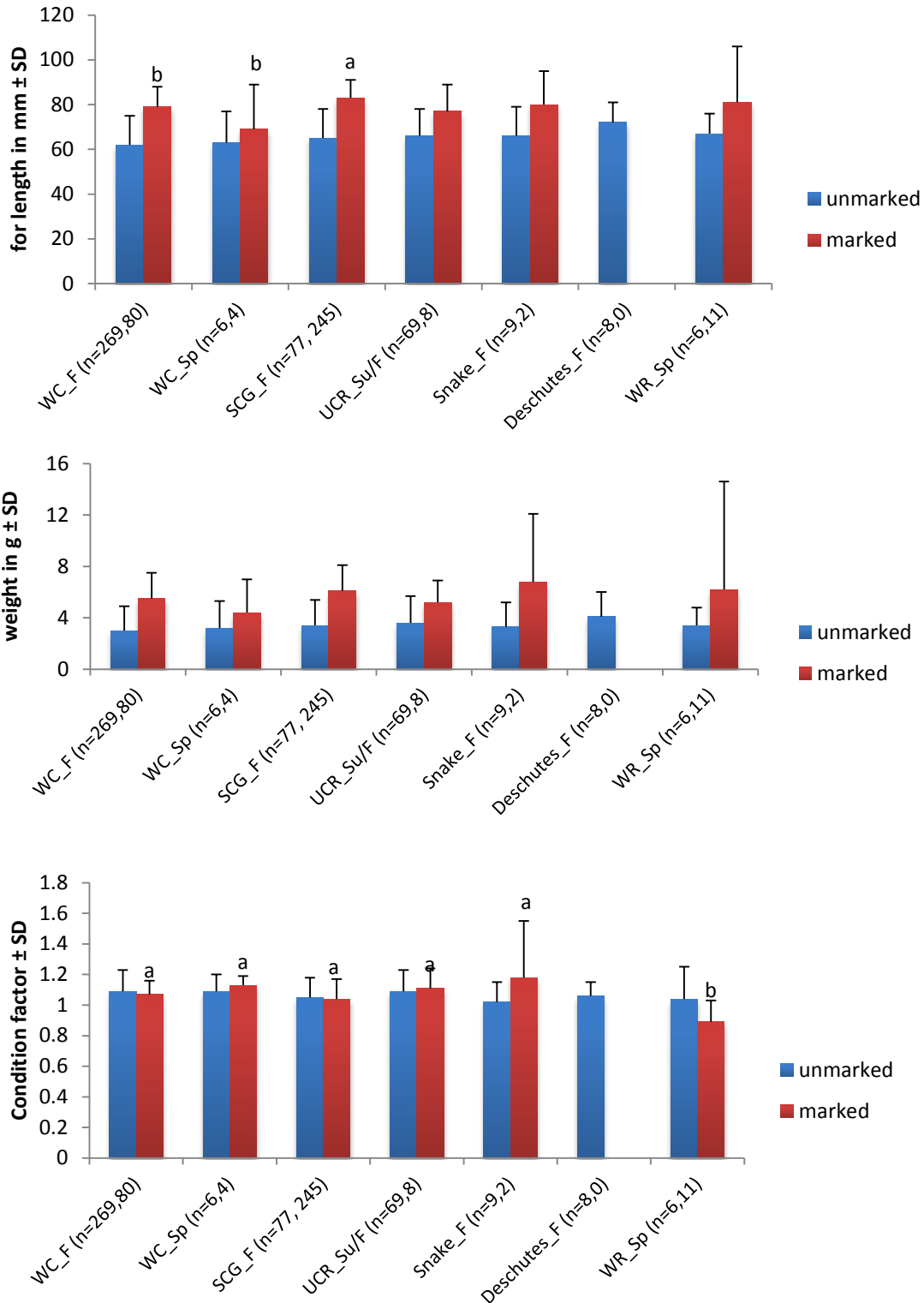
Figure 98. Length, weight and condition factor of juvenile Chinook salmon by genetic stocks. All fish sampled from 2008 to 2011 are included in this analysis. Statistically comparisons were conducted among stocks within unmarked or unmarked groups of fish. Values with different letter superscripts are significantly different (ANOVA, Tukey's multiple range test, $p < 0.05$).

4.5.2.2 *Salmon Size and Condition*

4.5.2.2.1 Chinook salmon

In unmarked subyearling Chinook salmon, size class distribution, length, weight, and condition factor (CF) were all strongly influenced by month of capture (Figure 99 and Figure 100). Length, weight, and CF, as well as the proportion of larger, fingerling Chinook, all increased over the sampling season from February through August. These trends were less evident in marked Chinook salmon; fish weight tended to increase from April through August (Figure 99 and Figure 100), but length and condition remained fairly constant, the almost all marked fish caught were fingerlings, with the exception of some yearling Chinook that were found in April and May. Because the number of fish captured per month varied from year to year and site to site, the influence of sampling time taken into account in statistical comparisons among sites and among years for the trend sites.

Size and condition at 2012 sites

Size class distribution. Marked Chinook at the new 2012 sites (Secret River, Welch Island, Government/Lemon Island, Washougal, and Reed Island) were almost all fingerlings (Figure 101). At Government/Lemon Island, some yearling Chinook were also captured; they accounted for 12.5% of the catch. No yearlings were found at the other new sites. These size class distributions are very similar to those observed at sites sampled in previous years, with the exception of Beacon Slough and Pierce Island, where higher proportions of yearlings were observed.

Unmarked Chinook salmon catches at the new 2012 sampling sites (Figure 101) contained a high proportion of fry (< 60 mm in length). At Secret River, Washougal and Reed Island, fry made up over 98% of the catch, while at Government/Lemon Island, 67% were fry. The proportions of fry at Secret River, Welch Island, Washougal, and Reed Island were among the highest observed at the sites sampled to date. This was partly due to more intensive sampling at the 2012 sites early in the season (February and March) in 2012. However, the new sites also tended to have higher proportions of fry in catches May and June than catches from the EMP sites overall (Figure 102).

Length, weight, and condition factor. Mean length, weight, and CF for marked and unmarked Chinook salmon at the 2012 sampling sites are shown in Table 27. Length, weight and condition factor of subyearling Chinook at the new 2012 sites are compared with the sites sampled previously in Figure 103, Figure 104, and Figure 105.

After adjusting for the effects of variation in sampling time, length, weight, and CF of unmarked Chinook salmon from Secret River were not significantly different the average values for all unmarked subyearling juvenile Chinook ($p < 0.05$). However, length and/or weight of unmarked Chinook from Welch Island, Government/Lemon Island, and Washougal tended to be lower than the overall average. For unmarked Chinook from the Washougal site, CF was also lower than the overall average.

Marked subyearling Chinook from Secret River were comparable in size and CF to the overall average of marked subyearlings. However, fish from Welch Island and Government/Lemon Island tended to be higher in length and/or weight, after taking into account the effects of sampling time. Marked Chinook from Washougal, on the other hand, were significantly smaller and had lower CF than marked Chinook salmon overall.

Trends in size and condition at fixed sites

Trends at the fixed sites incorporating the 2012 data could be evaluated only at Whites Island (Figure 106) and Campbell Slough (Figure 107). At Franz Lake Slough, no Chinook salmon were collected in 2012, while at Ilwaco Slough, 2012 was the first year in which more than one Chinook salmon was found at the site.

At Whites Island and Campbell Slough, the length and weight of unmarked juvenile Chinook salmon in 2012 were within the range of sizes observed in previous years. Differences among years were compared using a multiple regression analysis accounting for the effects of both month of capture and year of capture. At both Campbell Slough and Whites Island, fish collected in 2012 were intermediate in size, with length, weight and CF not differing significantly from the overall mean of fish from this site after the effects of sampling time had been taken into account. Marked Chinook salmon from Campbell Slough in 2012 were also intermediate in size as compared to other years, and did not differ from the overall mean in length, weight, or CF. At Whites Island, the number of marked fish captured was very small, and almost all were caught in May, so it was not possible to evaluate trends over the sampling years while adjusting for month of capture. However, when mean values of length, weight, and CF were compared, there were no significant differences among years ($0.1515 \leq p \leq 0.2534$).

Table 27. Mean length and weight of marked and unmarked Chinook salmon by month at the 2012 Ecosystem Monitoring sampling sites.

Date	Site	Unmarked			Marked		
		Length (mm)	Weight (g)	n	Length (mm)	Weight (g)	n
2/21/12	Ilwaco Slough	40	0.4	1			
4/19/12	Ilwaco Slough	48	0.8	1			
5/17/12	Ilwaco Slough	48	1.6	1			
6/12/12	Ilwaco Slough	60	2.2	1			
2/21/12	Secret River	40±2	0.5±0.2	25			
3/20/12	Secret River	44±7	0.6±0.22	3			
4/18/12	Secret River	46±4	0.8±0.3	29	66	2.8	1
5/16/12	Secret River	47	1.0	1	79	4.4	1
6/13/12	Secret River	61±10	2.5±1.4	25			
7/17/12	Secret River	73±5	4.3±1.2	3	84±1	6.2±1.0	2
2/20/12	Welch Island	41±2	0.6±0.2	32	108	12.6	1
3/22/12	Welch Island	42±3	0.6±0.2	25			
4/18/12	Welch Island	44±4	0.7±0.2	17			
5/16/12	Welch Island	50±4	1.3±0.4	21	83±5	6.3±1.1	7
6/13/12	Welch Island	56±6.	1.9±0.6	24			
7/17/12	Welch Island	64±6	3.3±1.1	15	78	4.8	1
8/14/12	Welch Island	91	8.9	1	99±9	12.1±3.2	2
2/20/12	Whites Island	38	0.3	1			
3/22/12	Whites Island	43±5	0.8±0.3	12			
4/18/12	Whites Island	44±4	0.7±0.3	10	60	2.0	1
5/16/12	Whites Island	50±5	1.2±0.4	26			
6/12/12	Whites Island	55±5	1.7±0.5	27			
7/16/12	Whites Island	66±7	3.3±0.9	7	77	4.3	1
8/14/12	Whites Island				74	4.4	1
5/15/12	Campbell Slough	54±8	1.7±1.0	20	79±7	5.0±1.1	18
6/14/12	Campbell Slough	70±6	4.0±0.9	20	76±5	4.5±1.0	2
7/19/12	Campbell Slough	75	5.5	1	86±6	7.6±2.2	4
3/21/12	Govt/Lemon Island	40±3	0.6±0.2	15			
4/17/12	Govt/Lemon Island	44±4	0.7±0.2	6	114±50	13.8±14.4	2
5/15/12	Govt/Lemon Island	48±9	1.2±0.8	25	130±19	18.4±7.9	3
6/13/12	Govt/Lemon Island	59±6	2.8±0.7	32			
7/18/12	Govt/Lemon Island	77±6	5.2±1.1	16	91±8	8.3±2.2	3
2/22/12	Washougal	40	0.50	1			
3/21/12	Washougal	37±1	0.3±0.1	6			
4/16/12	Washougal	41±3	0.6±0.2	28			
5/14/12	Washougal	42±4	0.7±0.3	25			
6/11/12	Washougal	52±7	1.5±0.6	12	67±3	2.9±0.5	15
7/17/12	Washougal	74	4.6	1	82±5	6.0±1.0	7

| 2/22/12 | Reed Island | 36 | 0.3 | 1 |
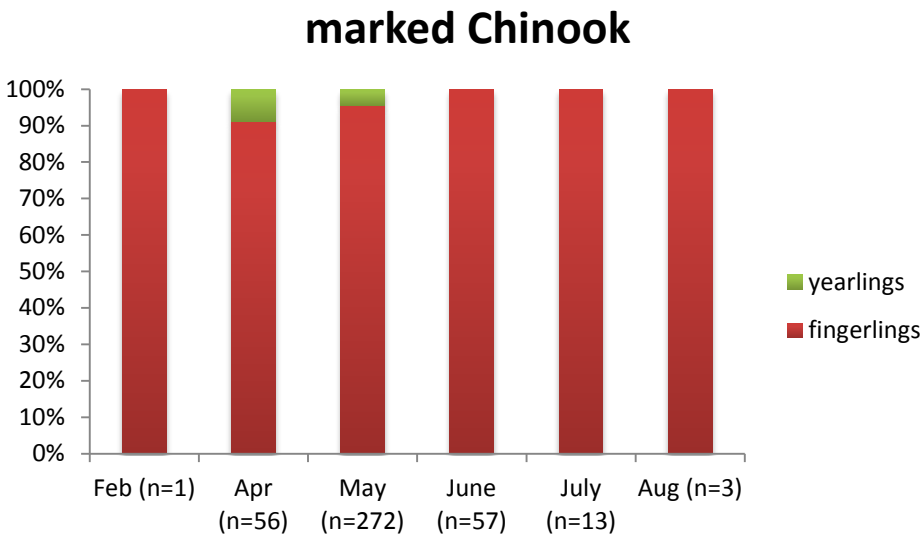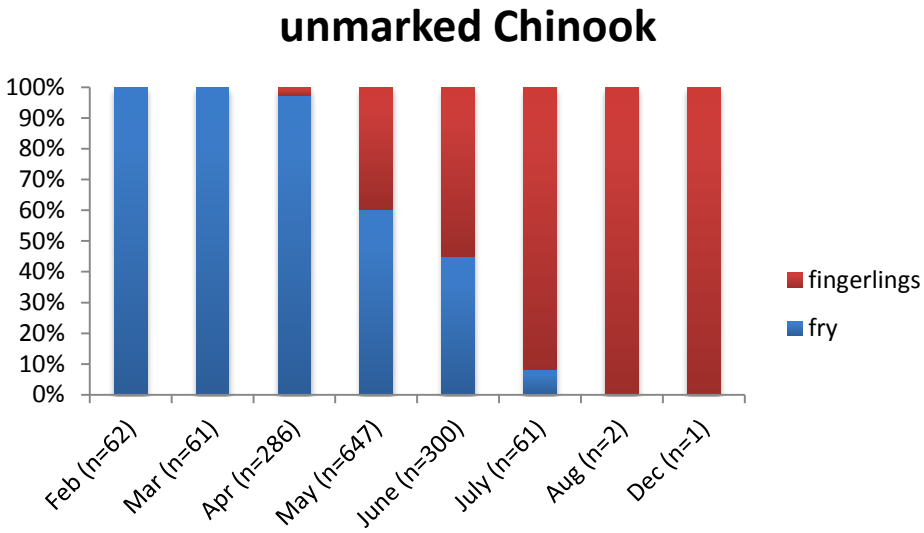
Figure 99. Size class distribution of marked and unmarked juvenile Chinook salmon by sampling month.
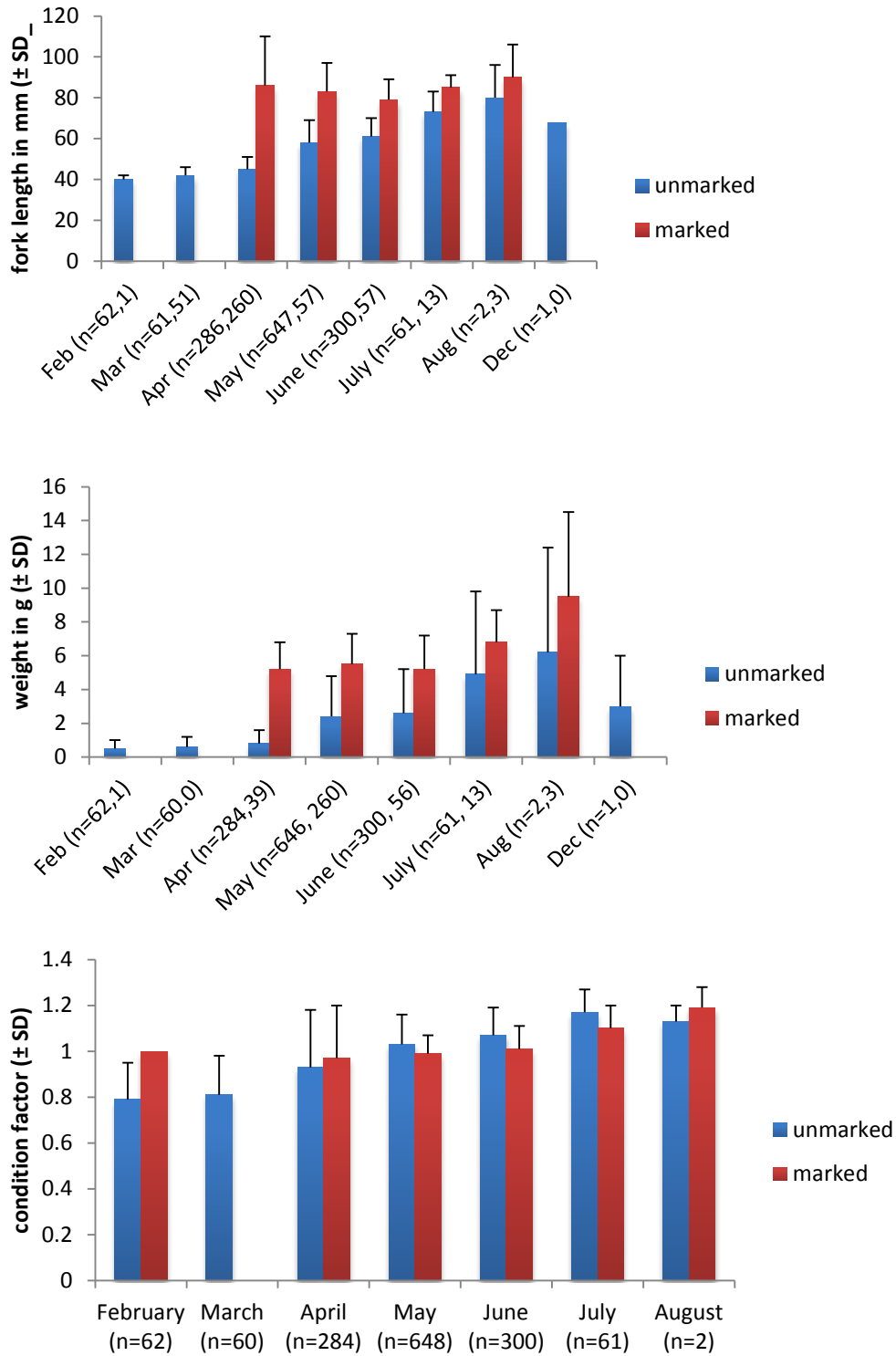
Figure 100. Length, weight, and condition factor in marked and unmarked juvenile Chinook salmon by sampling month.
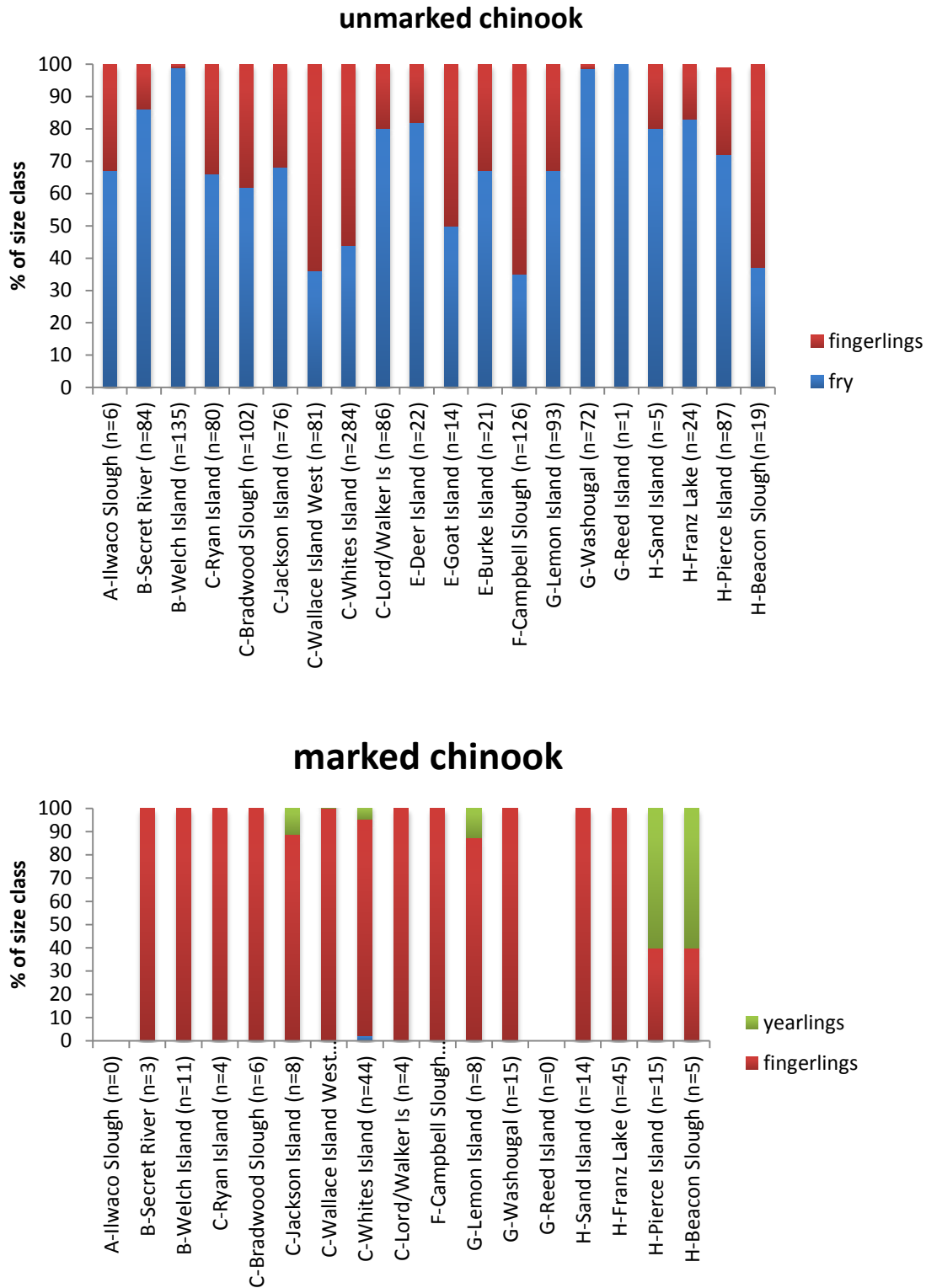
Figure 101. Size class distribution by site in marked and unmarked juvenile Chinook salmon collected at the new 2012 Ecosystem Monitoring sites, as compared to proportions at sites sampled in previous years.
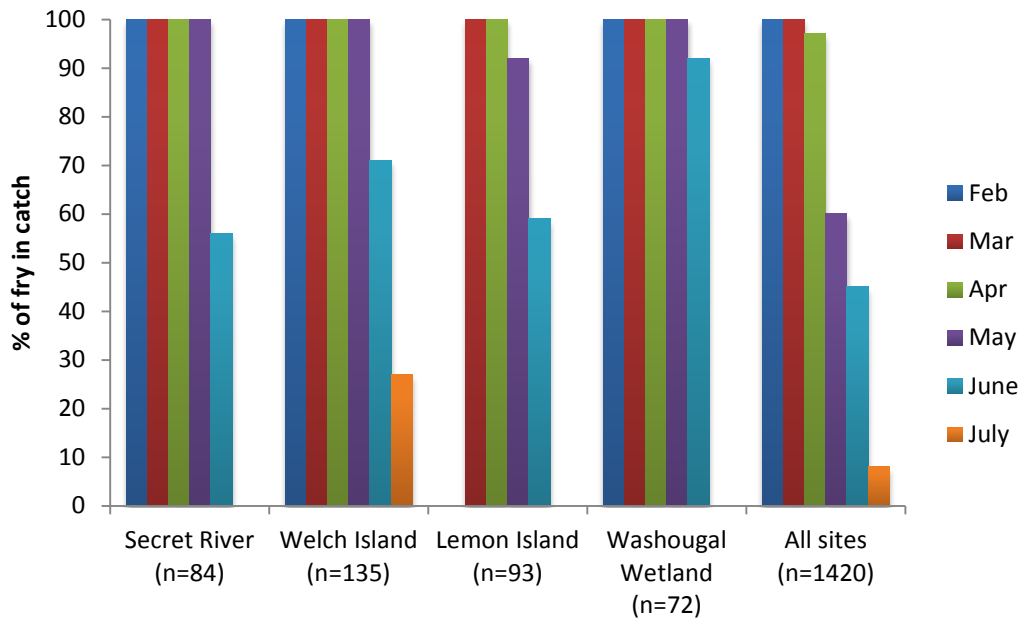
Figure 102. Proportion of fry in catches at the new 2012 Ecosystem Monitoring sampling sites (Secret River, Welch Island, Government/Lemon Island, and Washougal Wetland) by sampling as compared to the proportions for all fish collected during the EMP.
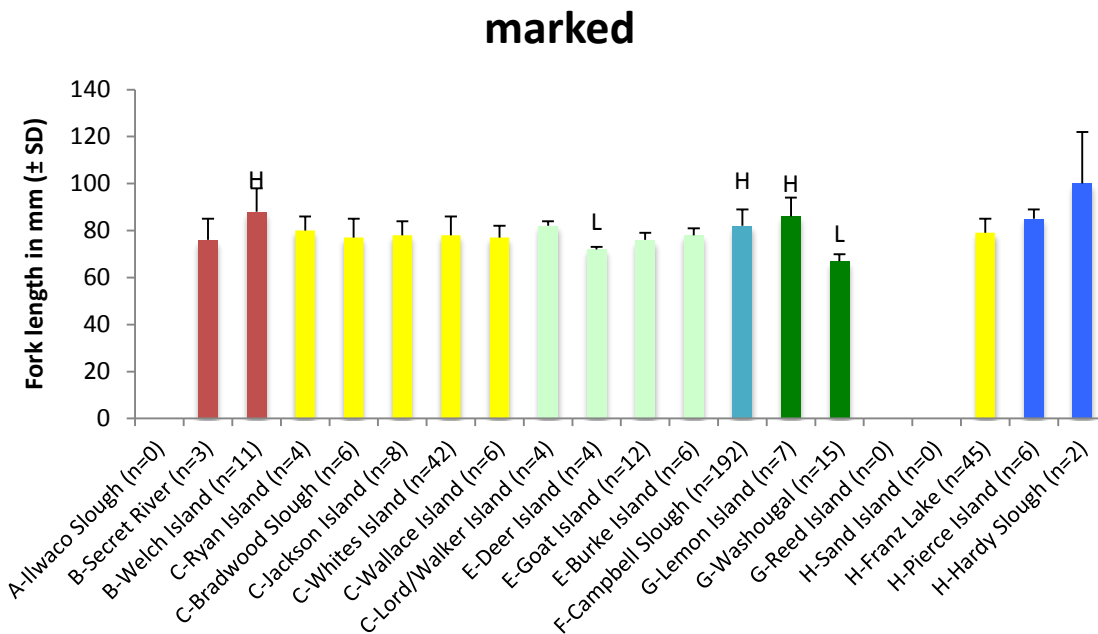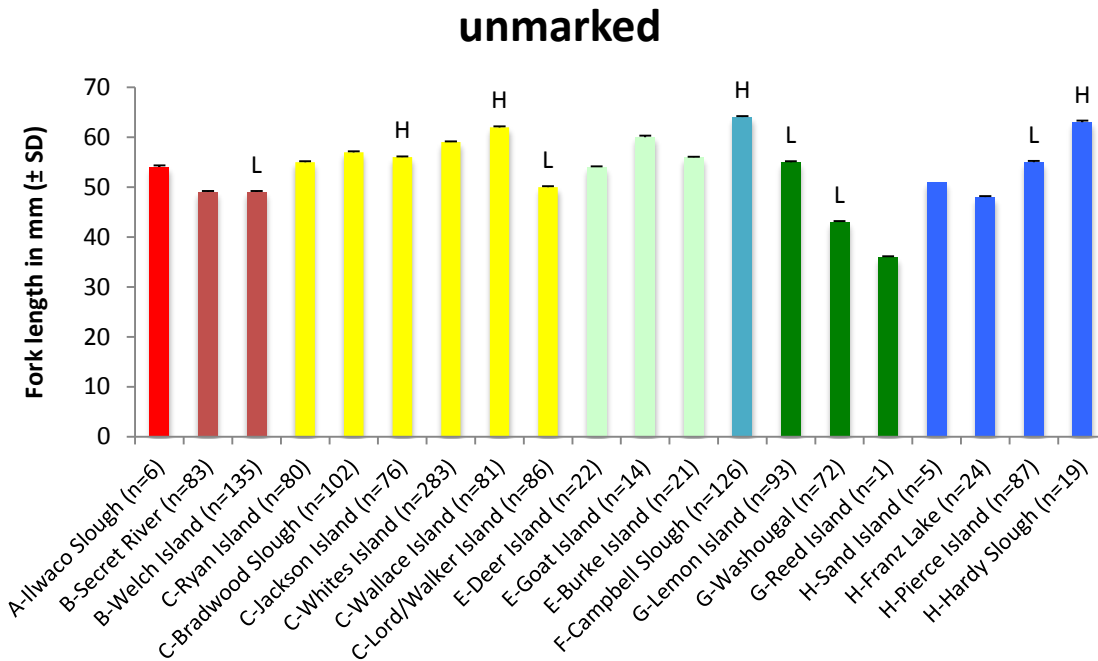
Figure 103. Mean fork length (± SD) of a) unmarked and b) marked juvenile Chinook salmon from the Ecosystem Monitoring sampling sites. H = value is significantly higher and L= value is significantly lower than the overall mean after adjusting for the effects of sampling month.
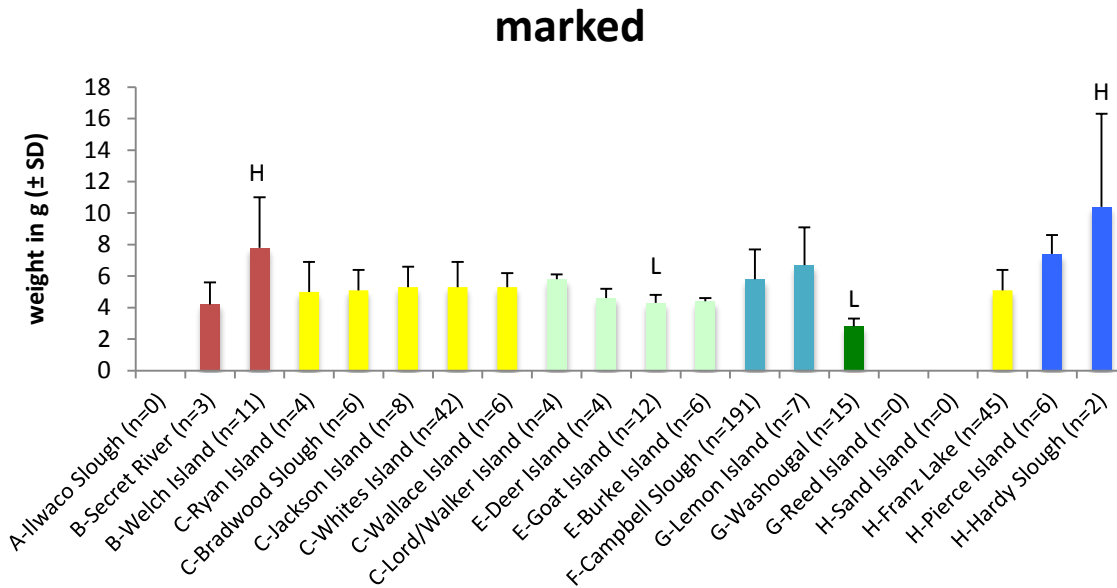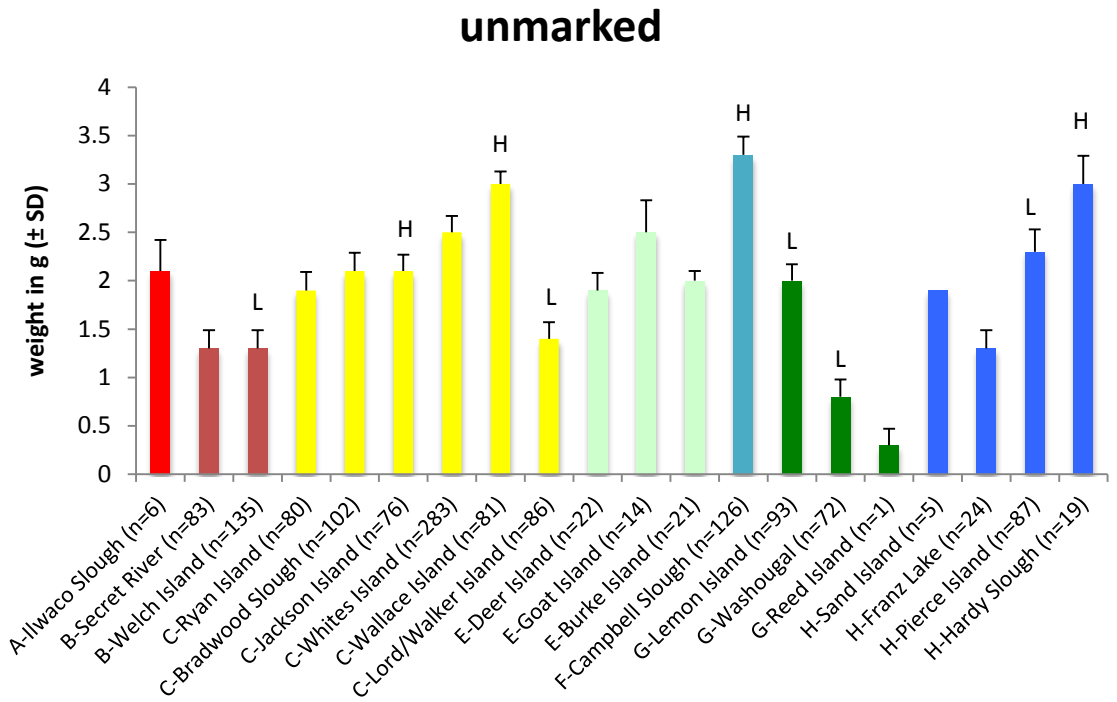
Figure 104. Mean weight (± SD) of a) unmarked and b) marked juvenile Chinook salmon from the Ecosystem Monitoring sampling sites. H = value is significantly higher and L= value is significantly lower than the overall mean after adjusting for the effects of sampling month.
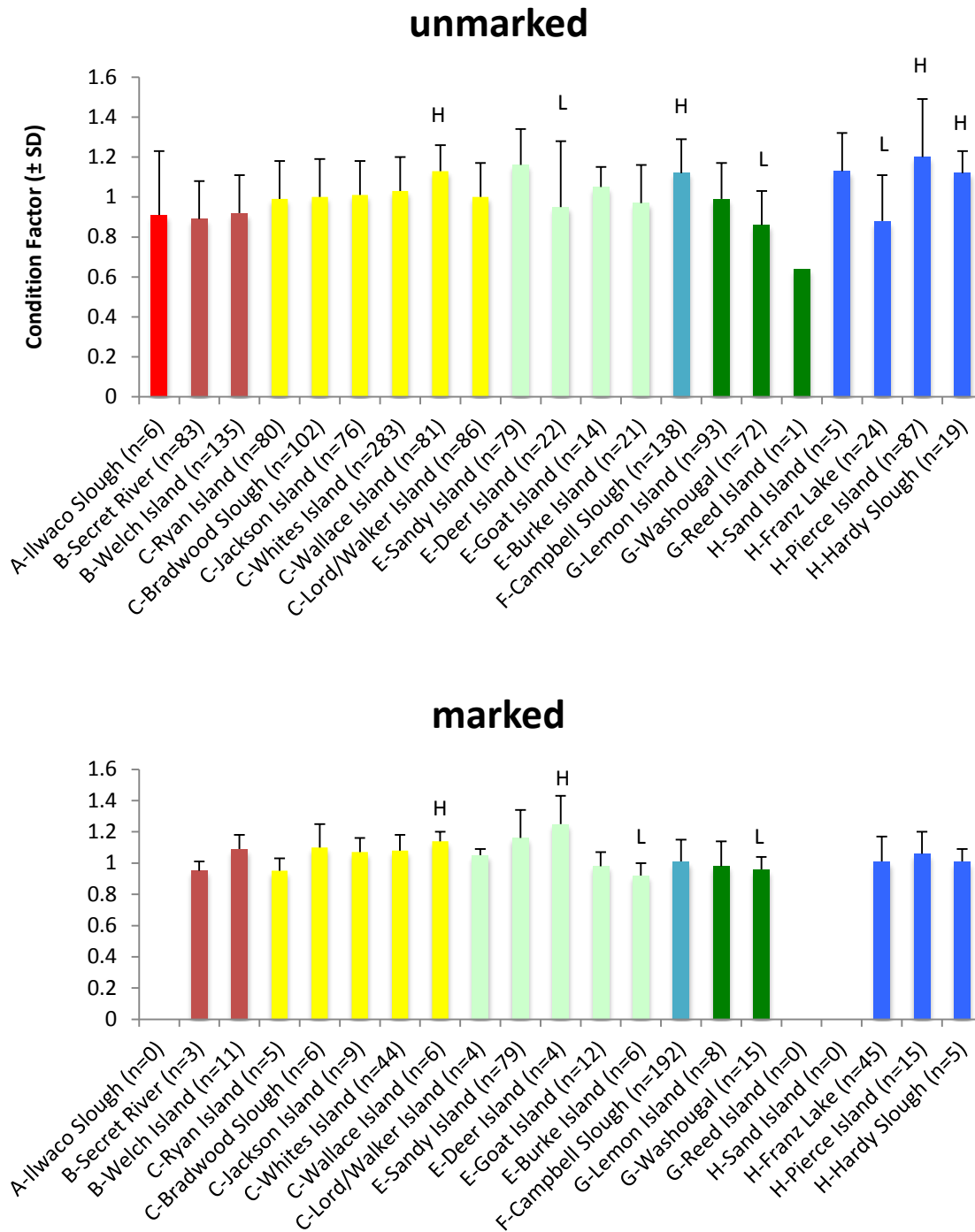
Figure 105. Mean condition factor (± SD) in A) unmarked and B) marked juvenile Chinook salmon from the 2012 Ecosystem Monitoring sites as compared to sites sampled in previous years. H = CF is significantly higher at this site than the overall mean value for all fish, after adjusting for the effect of sampling month. L = Sites with an H designation are significantly higher CF is significantly higher at this site than the overall mean value for all fish, after adjusting for the effect of sampling month.
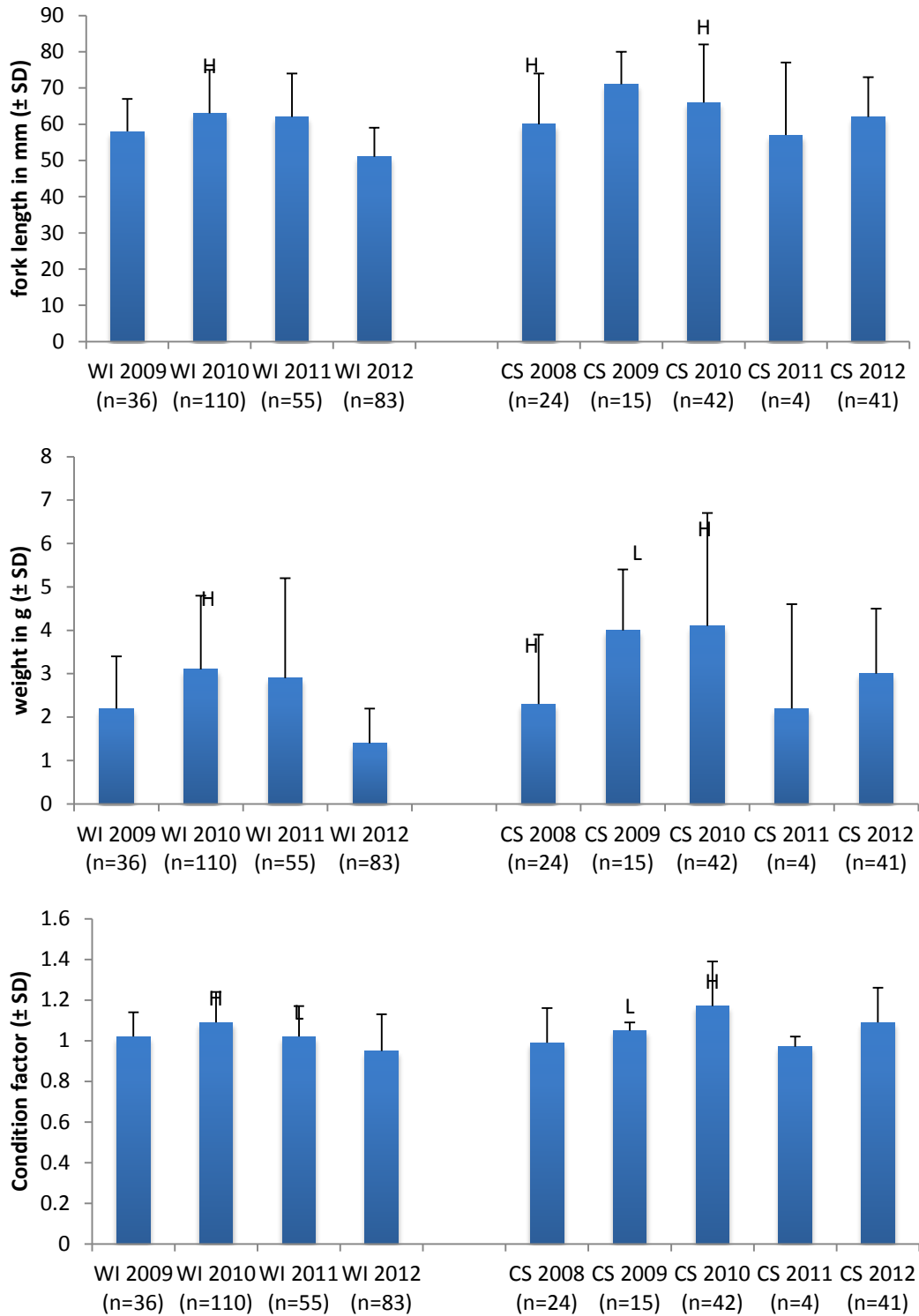
Figure 106. Mean length, weight, and condition factor of unmarked juvenile Chinook salmon from Whites Island and Campbell Slough by sampling year. H or L indicates value for that year was significantly higher or lower than the overall mean after adjusting for the effect of sampling month. No values for 2012 were significant.
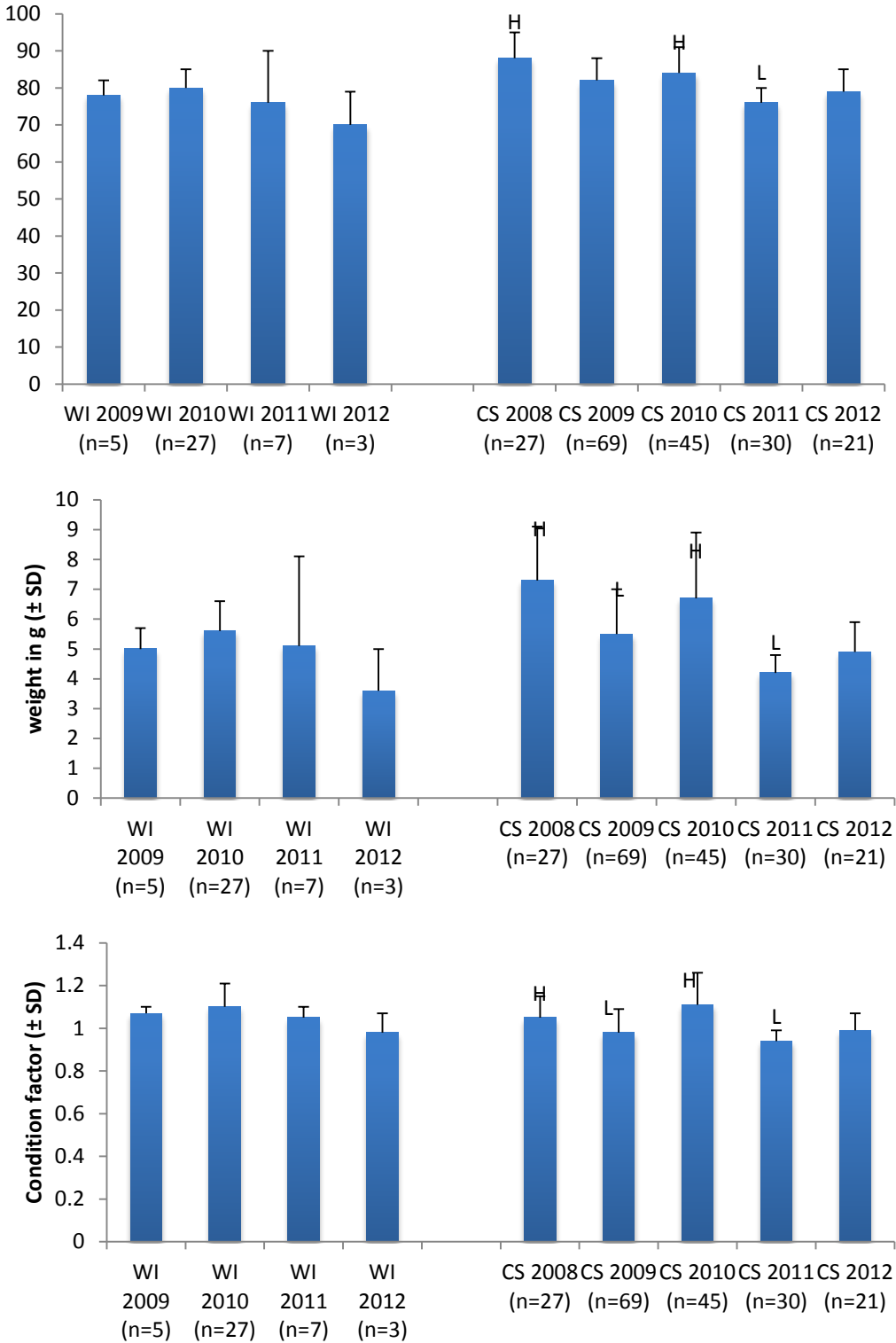
Figure 107. Mean length, weight, and condition factor of marked juvenile Chinook salmon from Whites Island and Campbell Slough by sampling year. H or L indicates value for that year was significantly higher or lower than the overall mean after adjusting for the effect of sampling month. No values for 2012 were significant.

4.5.2.2.2 Coho salmon

Size class distribution, length, weight, and condition factor (CF) of coho salmon were not as strongly related to month of capture as they were for Chinook salmon, probably in part because the number of coho collected overall was small, making it more difficult to detect seasonal trends. Among unmarked coho, the percentage of smaller fish in the fry size range was higher in May and June ($p \leq 0.05$, Likelihood Ratio and Heterogeneity G-test), and lower in August and December, than the overall proportions for all coho collected (Figure 108). Length and weight tended to be lower in May and June, but did not differ significantly by month of capture ($0.1207 < p < 0.2695$;Figure 109). However, condition factor did vary by month, and was significantly lower in May and December than in August (ANOA and Tukey's Multiple range test, $p < 0.0001$; Figure 109).

Among marked coho, the percentage of smaller fish in the fingerling size range was higher in February and September ($p \leq 0.05$, Likelihood Ratio and Heterogeneity G-test). Length also varied significantly by month of capture, with fish being significantly larger in May and June than in February or September (Figure 109). Weight followed a similar pattern to length, but differences in weight among months were not statistically significant (Tukey Multiple Range test, $p \leq 0.05$; Figure 109). Condition factor tended to be higher in the marked fish capture in February and September than in the other months, but again differences among months were not statistically significant (Tukey Multiple range test $p < 0.05$; Figure 109).

Size and condition at new 2012 sites

**Size class distribution**. Marked coho at the new 2012 sites (Secret River, Welch Island, Government/Lemon Island, Washougal, and Reed Island) were primarily yearling-size fish (Figure 110), similar to the coho size ranges found at site sampled previously. The two marked coho caught at Welch Island and Reed Island were both smaller.

Unmarked coho salmon catches at the new 2012 sampling sites (Figure 110) contained a range of size classes. Sites in Reach B and C, including Secret River and Welch Island, tended to include higher proportions of smaller, fry-size juveniles (< 60 mm in length) as compared to sites farther up the river. At Government/Lemon Island and Reed Island in Reach G, unmarked coho salmon were predominantly in the fingerling-size, while the catch at Washougal contained a higher proportion of yearlings.

Length, weight, and condition factor. Mean length, weight, and CF for marked and unmarked Chinook salmon at the 2012 sampling sites are shown in Table 28. Length, weight and condition factor of subyearling Chinook at the new 2012 sites are compared with the sites sampled previously in Figures 69-70. Although there was some influence of sampling month on some of these parameters, too few coho salmon were collected to evaluate the effect of sampling site while adjusting for sampling month. Consequently, statistical comparisons were among sites only.

For unmarked coho (Figure 111), length and weight could be quite variable, because of the range of different size classes found in catches, as described above. In general fish unmarked coho tended to by smaller at the sites in Reaches A-C than in Reaches G and H, though this was not true for all sites. Coho salmon at Government/Lemon Island were among the largest, comparable in length to fish from Franz Lake Slough and Sand Island. Those from Reed Island and Washougal, on the other hand, were intermediate in size. Unmarked coho from Secret River and Welch Island in Reach B were relatively small, comparable in size to those from the sites in Reaches C and the Reach H where the smallest fish

were found (Pierce Island and Hardy Slough). Fish weights followed a similar pattern, with relatively high weights in fish from Government/Lemon Island, and moderate to low weights at Secret River, Welch Island, Washougal, and Reed Island. Condition factor, on the other hand, was relatively high is unmarked coho salmon from Reed Island and Secret River, and relative low in fish from Welch Island and Government/Lemon Island, and intermediate at Washougal.

At Secret River, Welch Island, and Reed Island, only one marked coho was found per site, and these tended to be smaller fish of lower weight (Figure 112). At Government/Lemon Island marked coho were relatively large in both length and weight, while those from Washougal were intermediate (Figure 112). Condition factor in marked coho was showed no significant differences among the sampling sites (Figure 112).

Trends in size and condition at fixed sites

Trends at the fixed sites incorporating the 2012 data could be evaluated only for unmarked coho salmon at Franz Lake Slough. At Whites Island and Campbell Slough, no coho salmon were caught in 2012, while at Ilwaco Slough, they were caught for the first time in 2012.

At Franz Lake Slough (Figure 113), the length and weight of unmarked juvenile coho salmon in 2012 tended to be lower than in previous years, although the differences were not statistically significant because of the small number of fish collected per year and the large variation in both length and weight. Condition factor varied little from year to year. At least some of the variation in length, weight, and condition by year could be attributed to the season when the fish were sampled. The coho salmon sampled in 2012 were collected mainly in October, whereas those collected in 2008 and 2009 were collected mainly in May, and those collected in 2011 were collected mostly in November and December.

Table 28. Coho length and weight at the EMP sites in 2012.

Date	Site	Unmarked			Marked		
		Length (mm)	Weight (g)	n	Length (mm)	Weight (g)	n
3/19/12	Ilwaco Slough	104.00	11.40	1			
4/18/12	Secret River	134.33±16.92	26.20±5.89	3	133.00	22.10	1
5/16/12	Secret River	54.33±7.09	1.90±0.66	3			
6/13/12	Secret River	53.50±8.06	2.03±1.07	4			
7/17/12	Secret River	69.33±4.04	4.20±0.82	3			
8/13/12	Secret River	67.00	3.60	1			
6/13/12	Welch Island	48.00	0.70	1			
9/11/12	Welch Island				105.00	11.30	1
5/15/12	Govt/Lemon Island	130.33±19.40	18.37±7.91	3	147.5±9.43	29.36±6.85	12
2/22/12	Washougal	94.00	8.50	1			
3/21/12	Washougal						
4/16/12	Washougal						
5/14/12	Washougal				130.50±7.31	22.49±5.23	16
2/22/12	Reed Island	100.00	10.20	1			
8/16/12	Reed Island	98.84±7.46	11.62±2.73	18			
10/17/12	Franz Lake Slough	84.67±13.05	5.80±2.55	3			
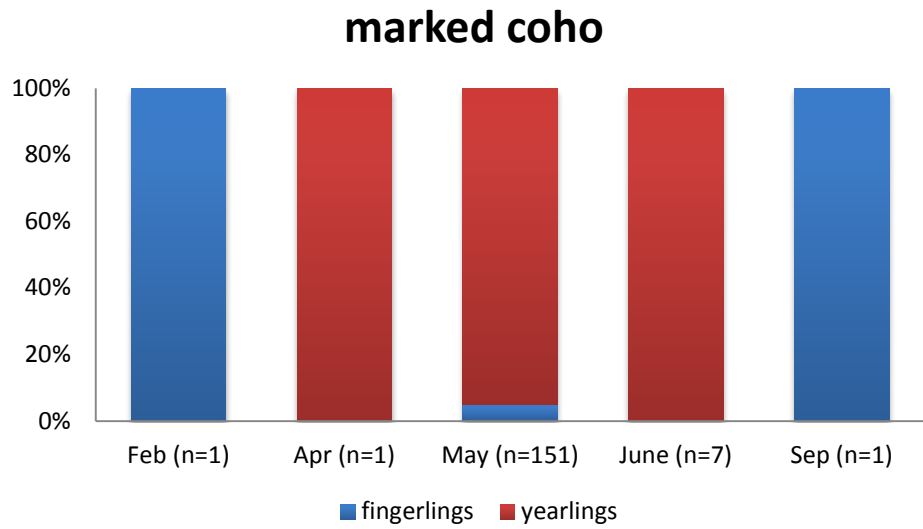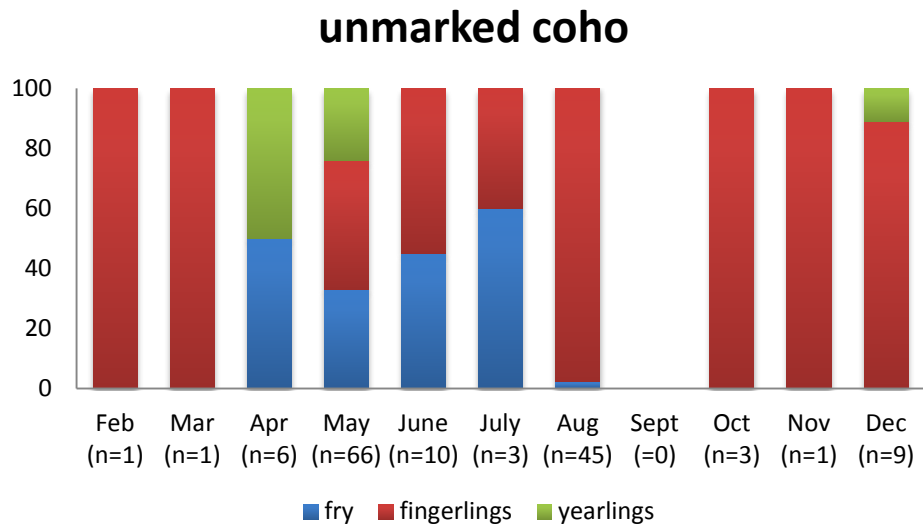
Figure 108. Size class distribution of unmarked and marked coho salmon by month.
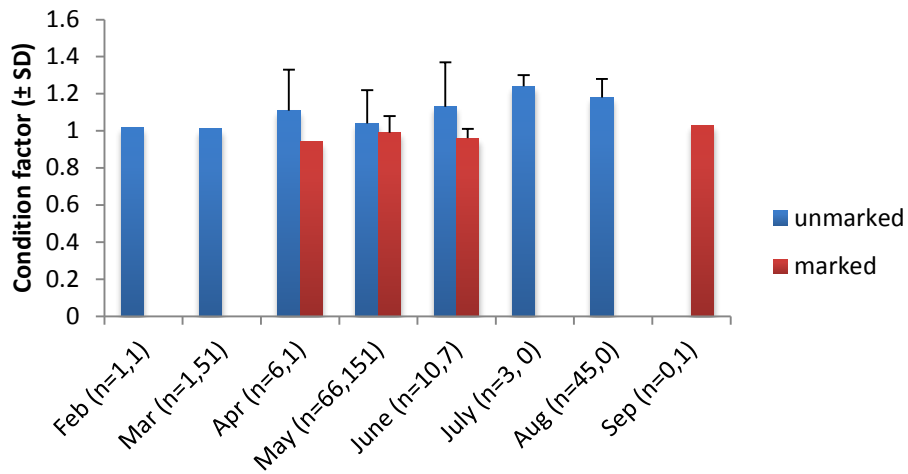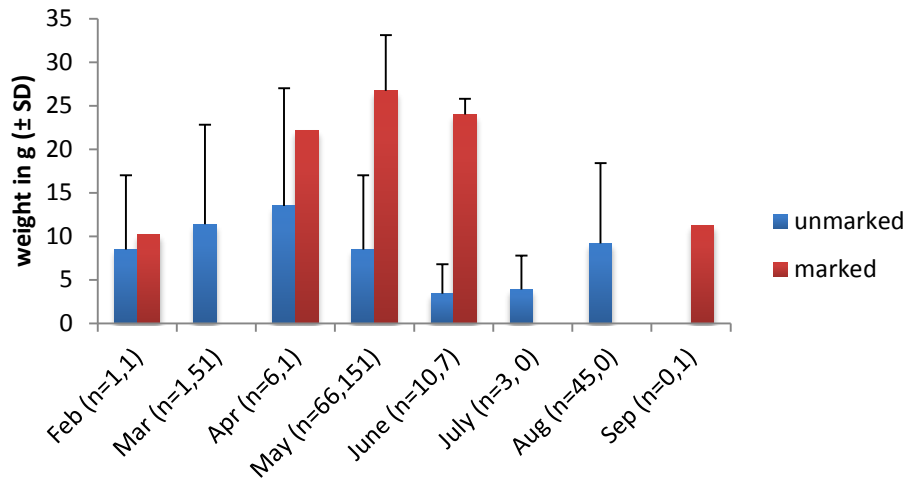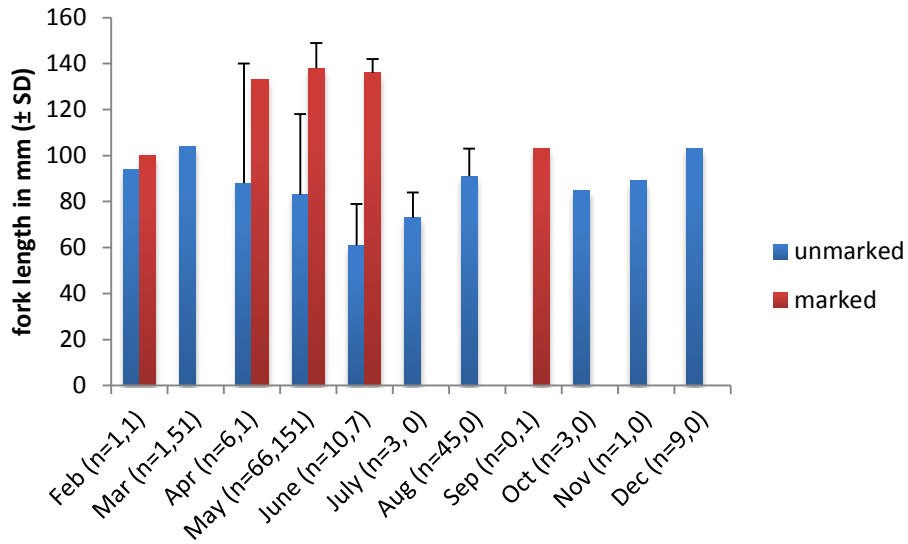
Figure 109. Length, weight, and CF in marked and unmarked coho salmon by sampling month.

unmarked coho
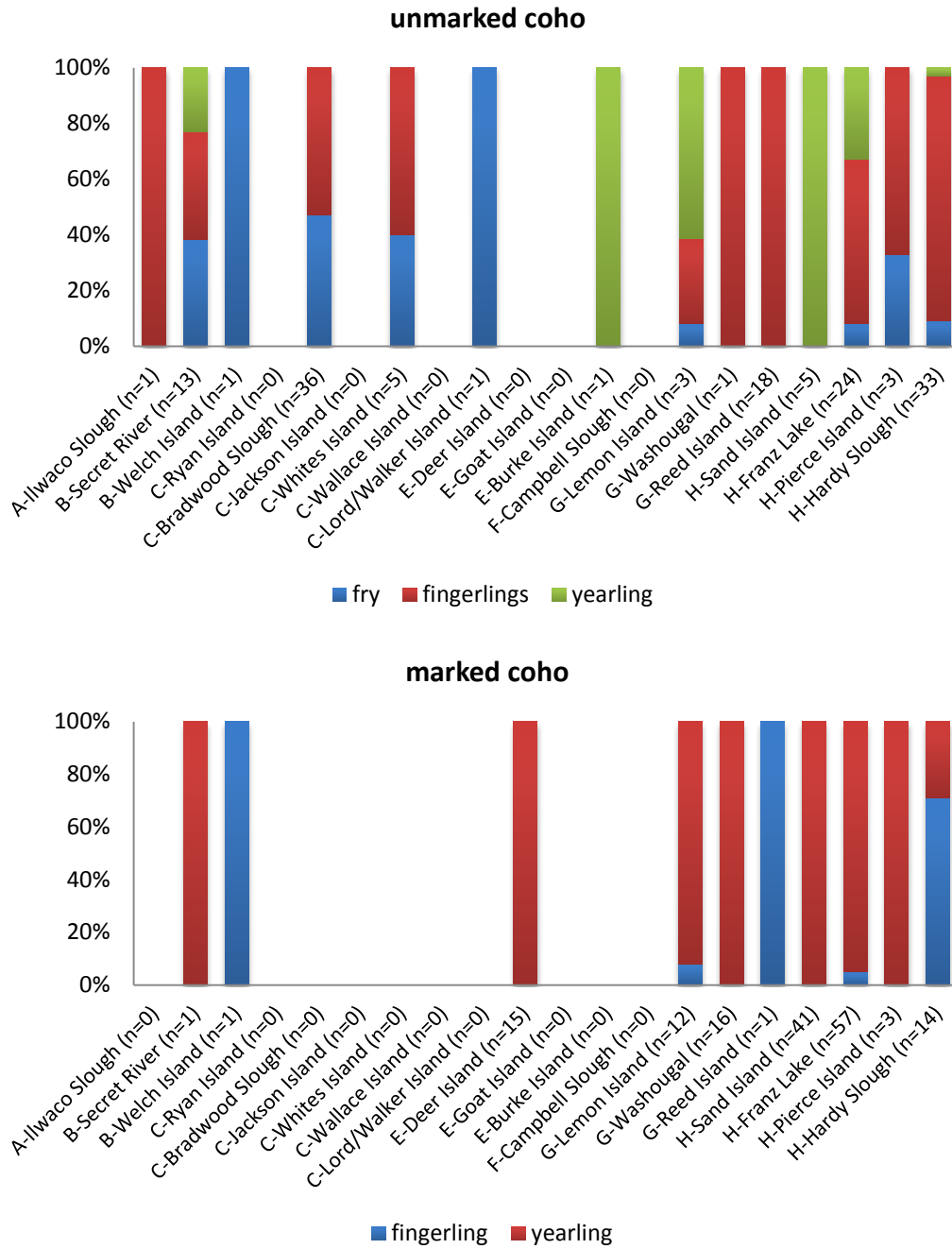
marked coho

Figure 110. Size class distribution of marked and unmarked coho salmon at 2012 Ecosystem Monitoring sampling sites as compared to sites sampled previously.
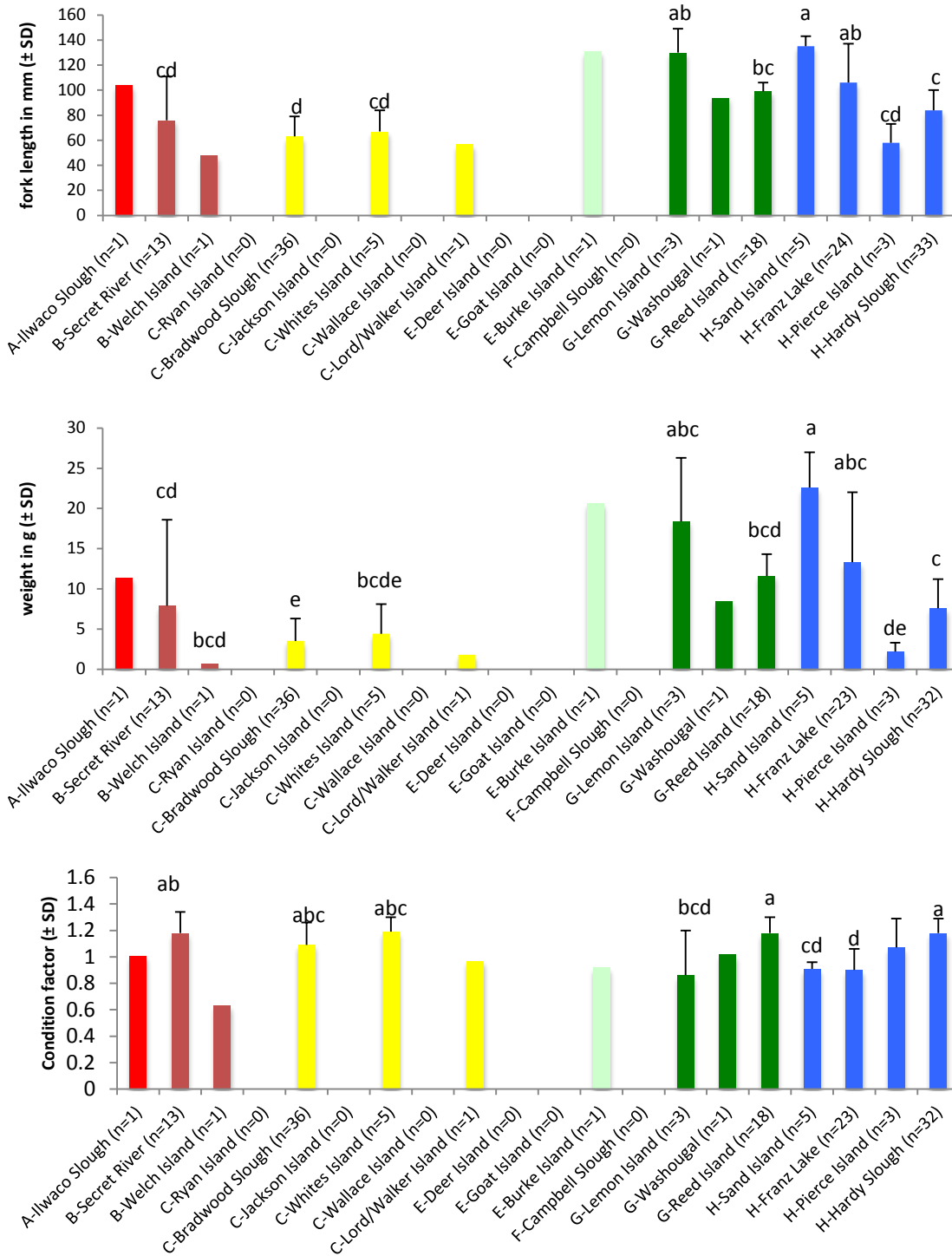
Figure 111. Length, weight and condition factor at 2012 Ecosystem Monitoring sites as compare to sites sampled previously. Sites with different letter superscripts are significantly different (ANOVA and Tukey's multiple range test, p ≤ 0.05).

Figure 112. Length, weight and condition factor of marked coho salmon at the 2012 Ecosystem Monitoring sites as compare to sites sampled previously. Sites with different letter superscripts are significantly different (ANOVA and Tukey's multiple range test, p ≤ 0.05).
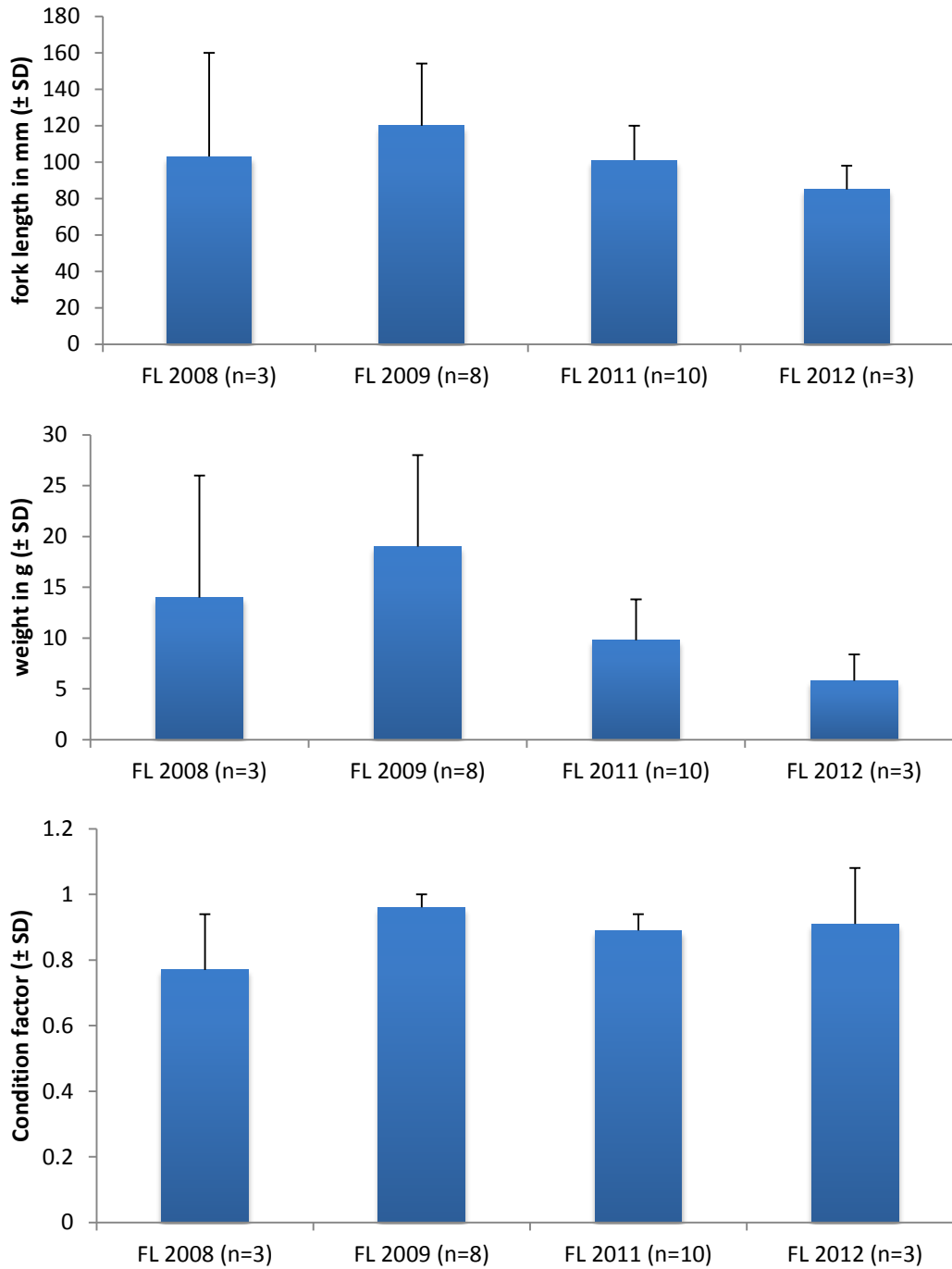
Figure 113. Mean length, weight, and condition by sampling year in juvenile chum salmon from Franz Lake Slough (FL). No significant differences among sampling years were observed. ($p \leq 0.05$) from those for other year within that site.

4.5.2.2.3   Chum salmon

Length, weight, and condition factor of chum salmon all tended to increase over the period when they were captured, from February to May (Figure 114). In the case of length and weight these differences were statistically significant, but in the case of condition factor, they were not.

Mean length, weight, and CF for chum salmon at the 2012 sampling sites are shown in Table 29. Length, weight and condition factor of chum at the new 2012 sites are compared with the sites sampled previously in Figure 115. Because of the influence of sampling month on these parameters, site means were compared statistically after adjusting for the influence of sampling month.

Table 29. Chum salmon length and weight at the 2012 EMP sites.

Date	Site	Length (mm)	Weight (g)	n
2/21/12	Ilwaco Slough	42	0.5	1
3/19/12	Ilwaco Slough	55	0.8	1
4/19/12	Ilwaco Slough	47±4	0.8±0.3	3
2/21/12	Secret River	37±1	0.3±0.1	3
4/18/12	Secret River	42±1	0.4±0.1	4
4/18/12	Welch Island	42±1	0.5±0.2	2
3/22/12	Whites Island	38 ±1	0.4±0.1	2
4/18/12	Whites Island	40	0.4	1
5/15/12	Govt/Lemon Island	42	1.2	1

Chum salmon length was fairly uniform among most of the sampling sites, typically in the 40-50 mm range. However, fish were significantly larger than the overall average at Ilwaco Slough and Ryan Island after adjusting for the effect of sampling month (Figure 115). Weights were generally in the 0.5-1.5 g range. After adjusting for the influence of sampling time, mean weight was significantly lower in fish from Secret River and Lord-Walker Island, and significantly higher at Jackson Island, than the overall average for all juvenile chum sampled (Figure 115). Condition factor showed a clearer longitudinal relationship over the sampling area than either length or weight, tending to be higher in fish collected from Reach H than in fish from the downstream reaches. After adjusting for the effect of sampling month, CF was significantly lower at Ilwaco Slough, Secret River, Ryan Island, Lord/Walker Island and Campbell Slough, and significantly higher at Franz Lake Slough and Hardy Slough, than the overall average for all chum salmon sampled (Figure 115).
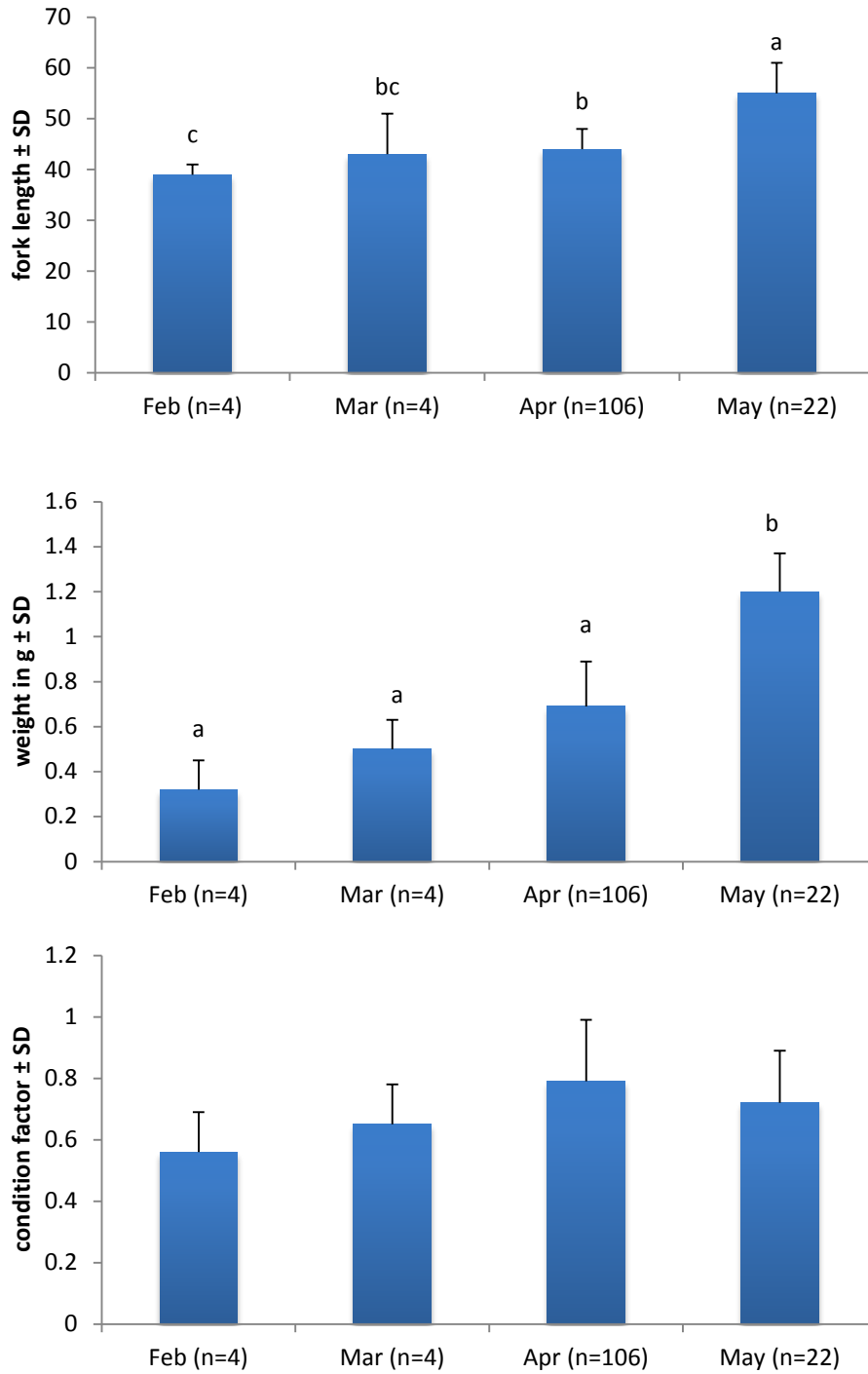
Figure 114. Length, weight, and condition factor by month of juvenile chum salmon collected at the Ecosystem Monitoring sites. Values with different letter superscripts and significantly different (ANOVA and Tukey's multiple range test, $p < 0.05$).
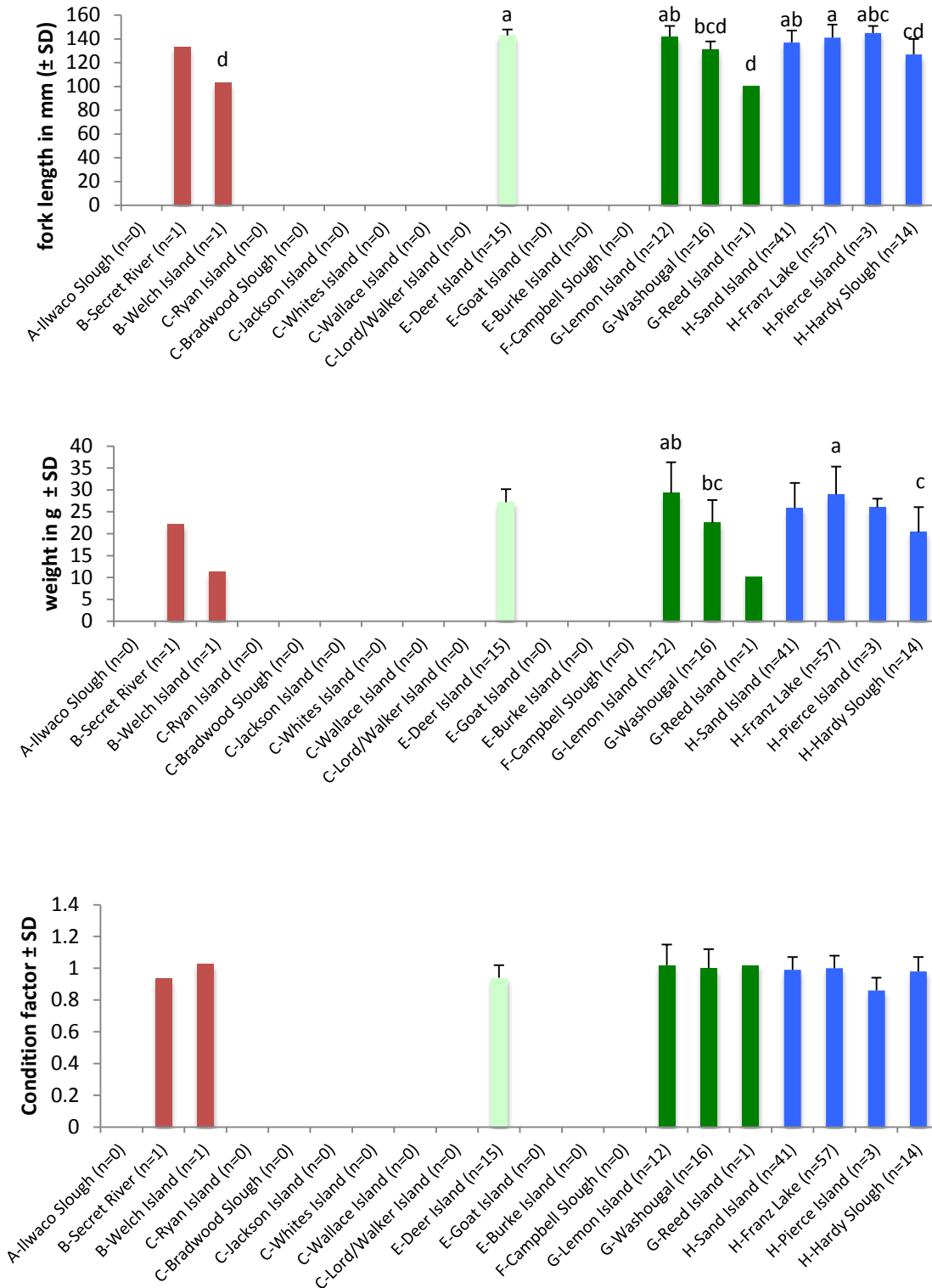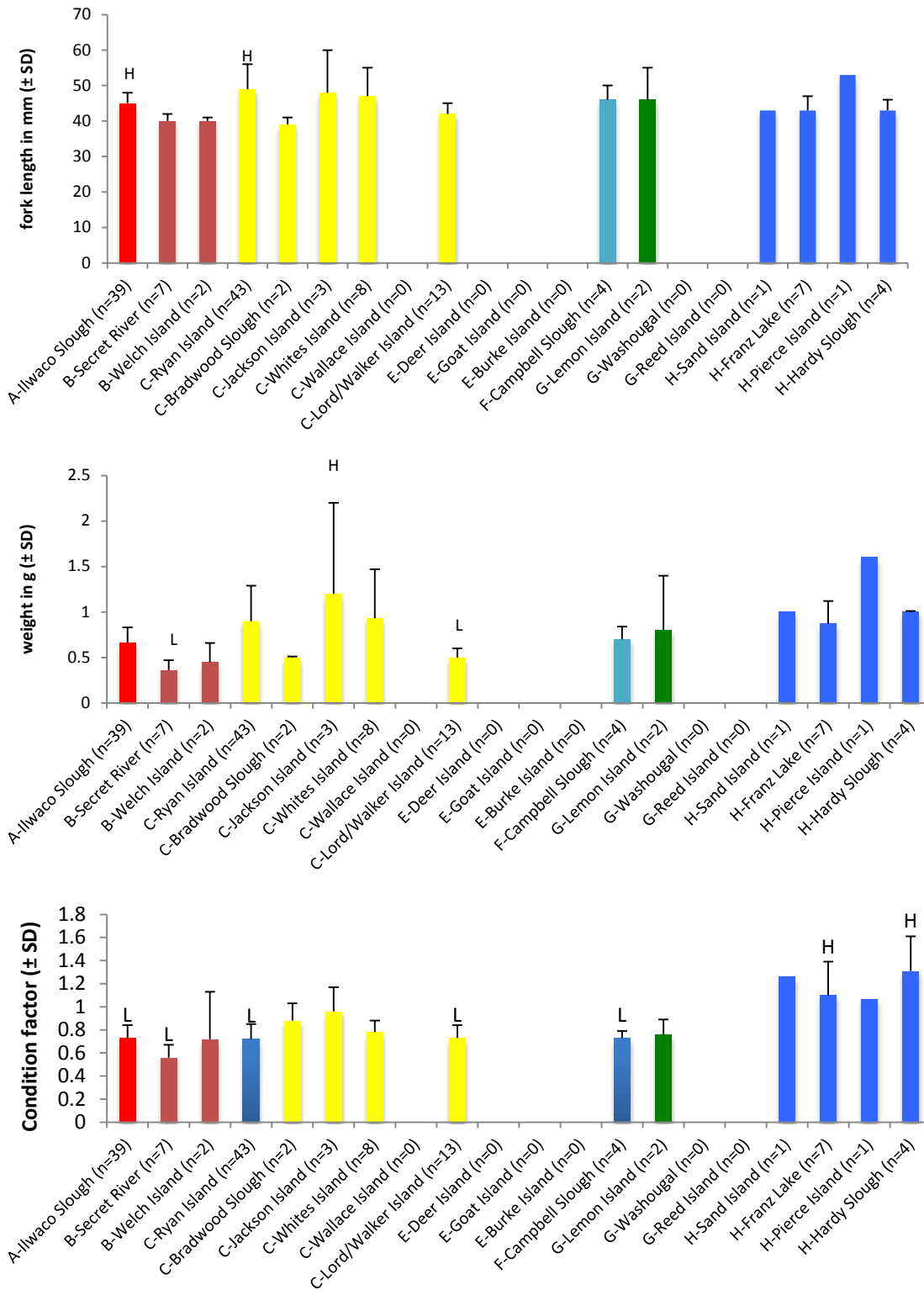
Figure 115. Length, weight and condition factor of juvenile chum salmon from the 2012 Ecosystem Monitoring sites as compared to sites sampled previously. L=mean value is significantly lower and H= the mean value is significantly higher than the overall mean after adjusting for the effect of sampling month (p ≤ 0.05).

Trends in size and condition at fixed sites

Trends at the fixed sites incorporating the 2012 data could be evaluated only at Ilwaco Slough and Whites Island. At Franz Lake Slough and Campbell Slough, no chum salmon were collected in 2012.

At Ilwaco Slough (Figure 116) the length of juvenile chum was significantly greater in 2012 than in 2011 ($p < 0.05$), but neither length nor condition changed significantly. The difference in fish length between the two years could be attributed in part to the fact that all the chum collected in 2011 were caught in April, while the 2012 catch included fish collected in February and March as well.

At Whites Island (Figure 116), length and weight both declined significantly from 2009 to 2012, but CF did not change. As at Ilwaco Slough, these changes can be attributed at least in part to the month when the fish were caught. In 2009 and 2010, the fish were collected in April and May, while in 2012, they were collected in March and April (in 2011 no chum salmon were caught at Whites Island).

4.5.2.3   *Growth Analyses*

Data on Chinook salmon growth rates are not yet available

4.5.2.4   *Lipid content of juvenile Chinook salmon*

These data are not yet available

4.5.2.5   *Contaminants in juvenile Chinook salmon*

These data are not yet available
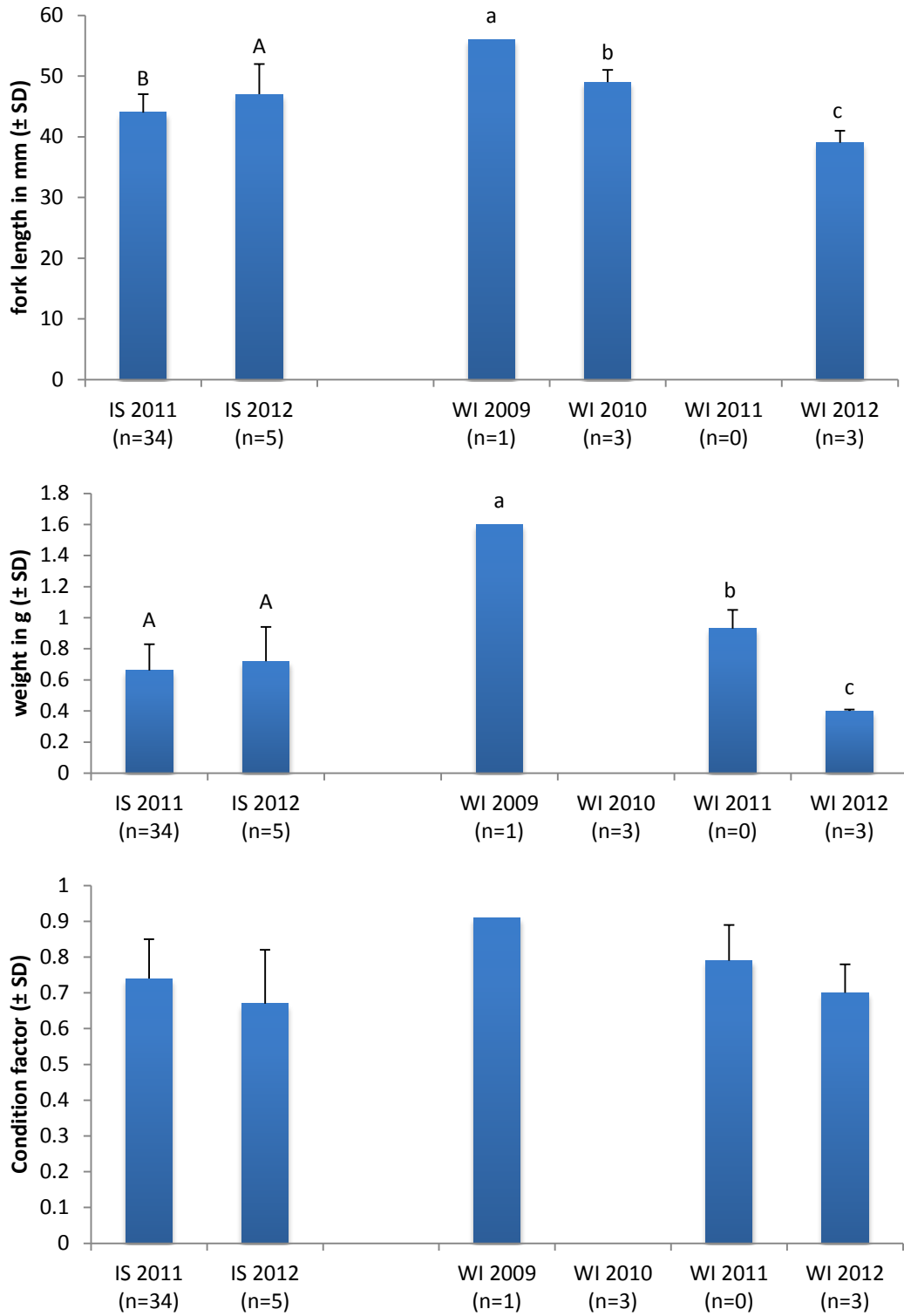
Figure 116. Mean length, weight, and condition by sampling year in juvenile chum salmon from Iwaco Slough (IS) and Whites Island (WI). The two sites are compared separately; values with different letter superscripts are significantly different ($p \leq 0.05$) from those for other year within that site.

4.5.2.6 *Juvenile Chinook salmon diets*

The juvenile Chinook salmon stomachs contained on average 21.7 (SD 40.5) invertebrates and weighted on average 32.1 mg (SD 32.8, wet weight). Diptera were by far the most represented order in the diets; they were on average 61.5% of all diet items by count, and 43.1% of the biomass consumed. The ranks of other taxa in the diets vary somewhat because of the differences in the relative sizes of the invertebrates (Table 30). Amphipods were relatively rare by count (11.7%), but because of their large size, they composed 20.3% of the biomass on average (Table 30). The variation in these proportions is quite high both within and across the space and time during which these were collected (Table 30, Figure 117). The standard deviations presented in Figure 118 illustrate that juvenile Chinook within a site can be feeding on very different prey items. For example, even for abundant prey types such as Diptera and Amphipoda, the variation in the percent of the diets made up by those two taxa can be greater than the mean values themselves. Likewise, the composition of prey in the diets of Chinook salmon caught at the same sites across multiple months suggests there is a wide variety of prey consumed within and across the times sampled.

Electivity values indicate juvenile Chinook consumed Amphipods and Dipterans at a rate higher than would be expected at some sites given the abundance of those taxa in the habitats sampled (Figure 119a and b, Table 31). Amphipods were highly selected especially at sites in the lower reaches, but appear to have been avoided at sites upstream of Deer Island (Figure 119a and b, Table 31). Hymenoptera were often rare but, when present, were selected at a high rate, especially at Whites and Wallace Islands (Table 31, Figure 119a). Other prey taxa were generally consumed at or below levels that would be expected given their abundances. The Ivlev's electivity value can appear skewed when abundances are low, and extreme values of -1 and 1 should be compared with actual counts. For example, if there were one Trichoptera larvae collected in a tow and none found in diets, the value would be -1. Likewise, if there had been 10,000 collected in tows and none in the diets, the value still would have been -1.

Table 30. Mean (SD) prey proportions of juvenile Chinook diets for all taxa, averaged over all fish captured in 2010-2012. Invertebrates that could not be identified or lacked a head were not counted, but were likely included in the "unknown" composite when weighed. The percent of the total mass of the average stomach that was "unknown" was 6.1%.

taxa	proportion of diets by biomass	SD of proportion by biomass	rank by biomass	proportion of diets by count	SD of proportion by abundance	rank by abundance
Diptera	0.431	0.273	1	0.615	0.373	1
Amphipoda	0.203	0.265	2	0.117	0.253	2
unknown	0.061	0.087	3	NA	NA	NA
Trichoptera	0.051	0.129	4	0.017	0.079	6
Hemiptera	0.047	0.102	5	0.034	0.112	5
Fish	0.045	0.181	6	0.017	0.116	7
Cladocera	0.027	0.071	7	0.067	0.225	3
Oligochaeta	0.027	0.111	8	0.001	0.018	19
Nemata	0.025	0.063	9	0.057	0.188	4
Mysida	0.013	0.058	10	0.001	0.019	21
Ostracoda	0.012	0.067	11	0.006	0.063	13
Hymenoptera	0.011	0.048	12	0.010	0.066	9
Ephemeroptera	0.009	0.022	13	0.007	0.054	12
Odonata	0.007	0.021	14	0.004	0.046	14
Isopoda	0.006	0.014	15	0.003	0.026	16
Copepoda	0.005	0.015	16	0.014	0.098	8
Collembola	0.004	0.010	17	0.009	0.054	10
Coleoptera	0.003	0.014	18	0.004	0.022	15
Megaloptera	0.003	0.011	19	0.000	0.007	25
Araneae	0.003	0.007	20	0.009	0.051	11
Lepidoptera	0.002	0.013	21	0.001	0.009	22
Plecoptera	0.002	0.014	22	0.001	0.008	23
Psocoptera	0.002	0.012	23	0.002	0.016	17
Rhynchobdellida	0.000	0.003	24	0.000	0.000	29
Trombidiformes	0.000	0.001	25	0.001	0.013	24
Chilopoda	0.000	0.000	26	0.002	0.039	18
Thysanoptera	0.000	0.000	27	0.001	0.020	20
Glossiphoniidae	0.000	0.000	28	0.000	0.001	26
Veneroida	0.000	0.000	29	0.000	0.000	27
Geophilomorpha	0.000	0.000	30	0.000	0.000	28
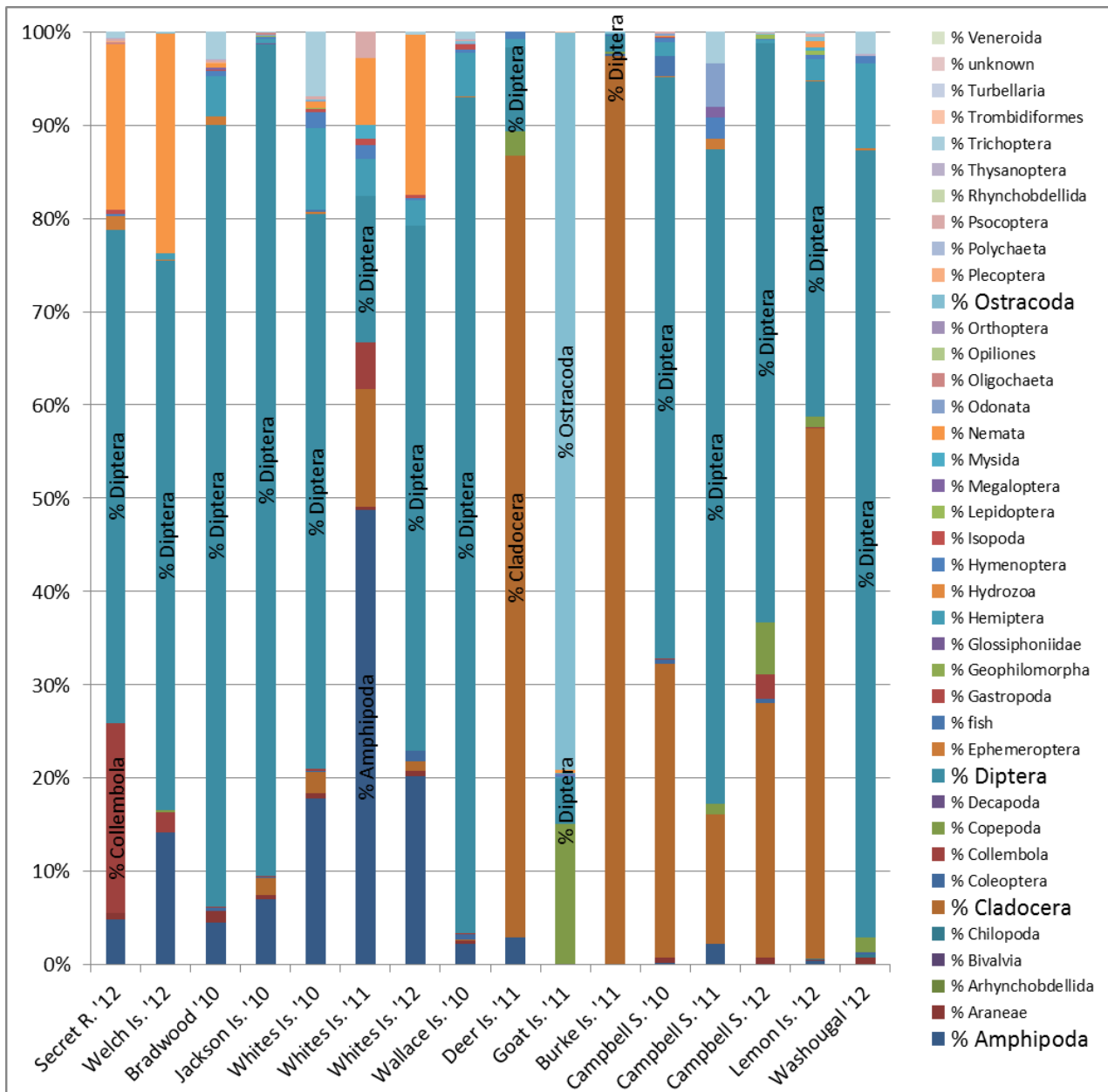
Figure 117. Mean composition of diets by site and year (using abundance values). Diptera and some other taxa are noted to orient readers. Several taxa that were often abundant are bolded in the legend.

Figure 118. Mean (SD) percent composition (by abundance) of juvenile Chinook diets of three sites sampled over several months in 2012. The nine orders of prey presented here include on average over 93% of the consumed invertebrate prey. Because not all orders are shown, the bars may not add to 100%.

Table 31. Ivlev's electivity values for juvenile Chinook collected from sites sampled in 2010-2012.

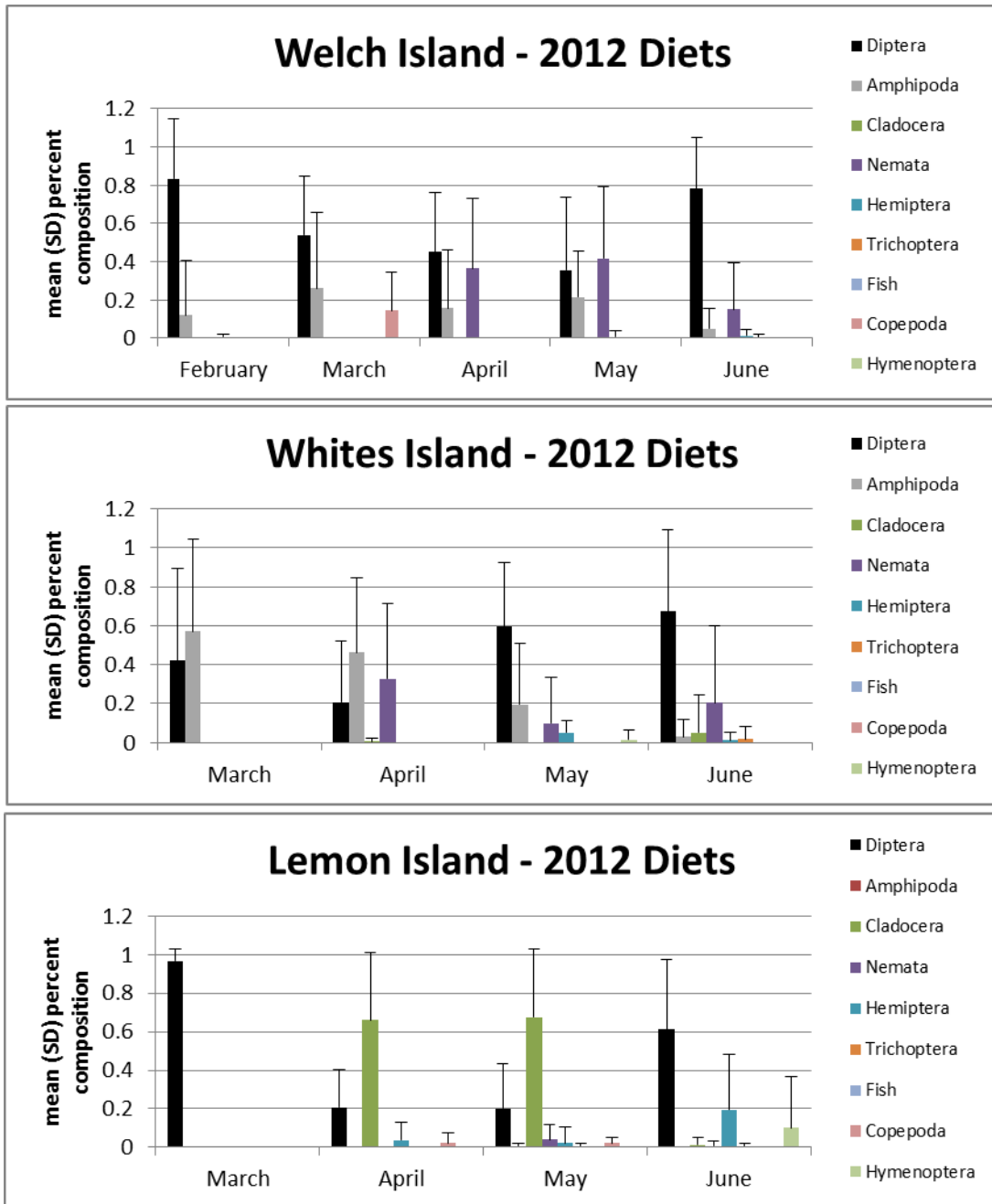| site	year	month	Diptera	Amphipoda	Cladocera	Hemiptera	Trichoptera	Copepoda	Hymenoptera
Secret River	2012	February	0.26	1.00		-1.00			
	2012	April	-0.23	0.98		-1.00	0.36	-1.00	-1.00
	2012	June	0.12	0.89	-1.00	-1.00	-0.78		-0.90
Welch Island	2012	March	0.36	0.41	-1.00			1.00	
	2012	April	0.42	1.00	-1.00	-1.00		-1.00	
	2012	May	-0.53	1.00		-0.72	-1.00		
	2012	June	0.27	0.55	-1.00	-0.57	-0.83	-1.00	
Bradwood Landing	2010	April	0.61	0.92		-0.59			
	2010	May	0.20	1.00		-0.64	-0.87		1.00
	2010	June	0.33	1.00	-1.00	-0.53			-0.27
	2010	July	0.67	0.71	-1.00	-0.68	0.99	-1.00	0.77
Jackson Island	2010	April	0.18	1.00	-0.48		-1.00	-1.00	
	2010	May	-0.02	0.85	-0.62	1.00	0.54	-1.00	0.25
	2010	June	0.15			-0.27	-1.00		1.00
Whites Island	2010	April	-0.14	1.00	0.98	-1.00		-1.00	
	2010	May	0.22	0.95	-1.00	-1.00	-1.00	-1.00	-1.00
	2010	June	-0.16	0.99	-1.00	-1.00	0.78	-1.00	0.46
	2010	July	0.45	0.91	-1.00	-0.43	1.00	-1.00	0.30
	2011	May	-0.43	0.52	0.81	-1.00	-1.00	-1.00	
	2011	June	-0.46	0.97	-1.00	-0.78		-1.00	1.00
	2012	March	0.24	0.81	-1.00			-1.00	
	2012	April	-0.47	0.94	0.80	-1.00	-1.00	-1.00	-1.00
	2012	May	0.26	0.97		1.00	-1.00	-1.00	1.00
	2012	June	0.43	0.74	-0.25	-0.60	0.24	-1.00	
Wallace Island	2010	April	0.21	0.89	-0.50	-1.00		-1.00	
	2010	May	0.06	0.22	-1.00	-0.80	-1.00	-1.00	0.11
	2010	June	0.32	1.00	-1.00	-0.81			
	2010	July	0.52	0.50	-1.00	0.44	1.00	-1.00	-0.11
Deer Island	2011	May	0.67	1.00	0.12	0.98	-1.00	-0.81	0.99
Goat Island	2011	May	-0.54	-1.00	-0.99	-0.99	-1.00	-0.37	1.00
Burke Island	2011	May	-0.84	-1.00	0.82	-0.91	-1.00	-0.99	-0.67
Campbell Slough	2010	April	0.89		-1.00	0.49		-1.00	
	2010	May	0.01	0.69	0.70	-0.84	-0.76	-1.00	-0.19
	2010	June	0.71	-1.00	-0.99	-0.75		-1.00	1.00
	2010	July	0.41		-1.00	0.64		-1.00	-1.00
	2011	May	0.75	-0.45	-0.32	-1.00	0.73	-0.95	0.97
	2012	May	0.20	-1.00	0.23	-1.00	-0.63	0.04	-1.00
	2012	June	0.48		-1.00	-0.63	-1.00	-1.00	0.95
Lemon Island	2012	March	0.51			-1.00			
	2012	April	-0.03	-1.00	0.77	-0.81	-1.00	-0.78	
	2012	May	0.37	0.27	0.70	-0.98	0.55	-0.85	
	2012	June	0.67		-0.92	0.46	1.00	-1.00	0.33
Washougal	2012	April	0.48		-1.00	-1.00	-1.00	0.02	0.71
	2012	May	0.07	-1.00	-1.00	0.82	-1.00	-1.00	-1.00
	2012	June	0.25		-1.00	0.45	1.00	-1.00	0.15
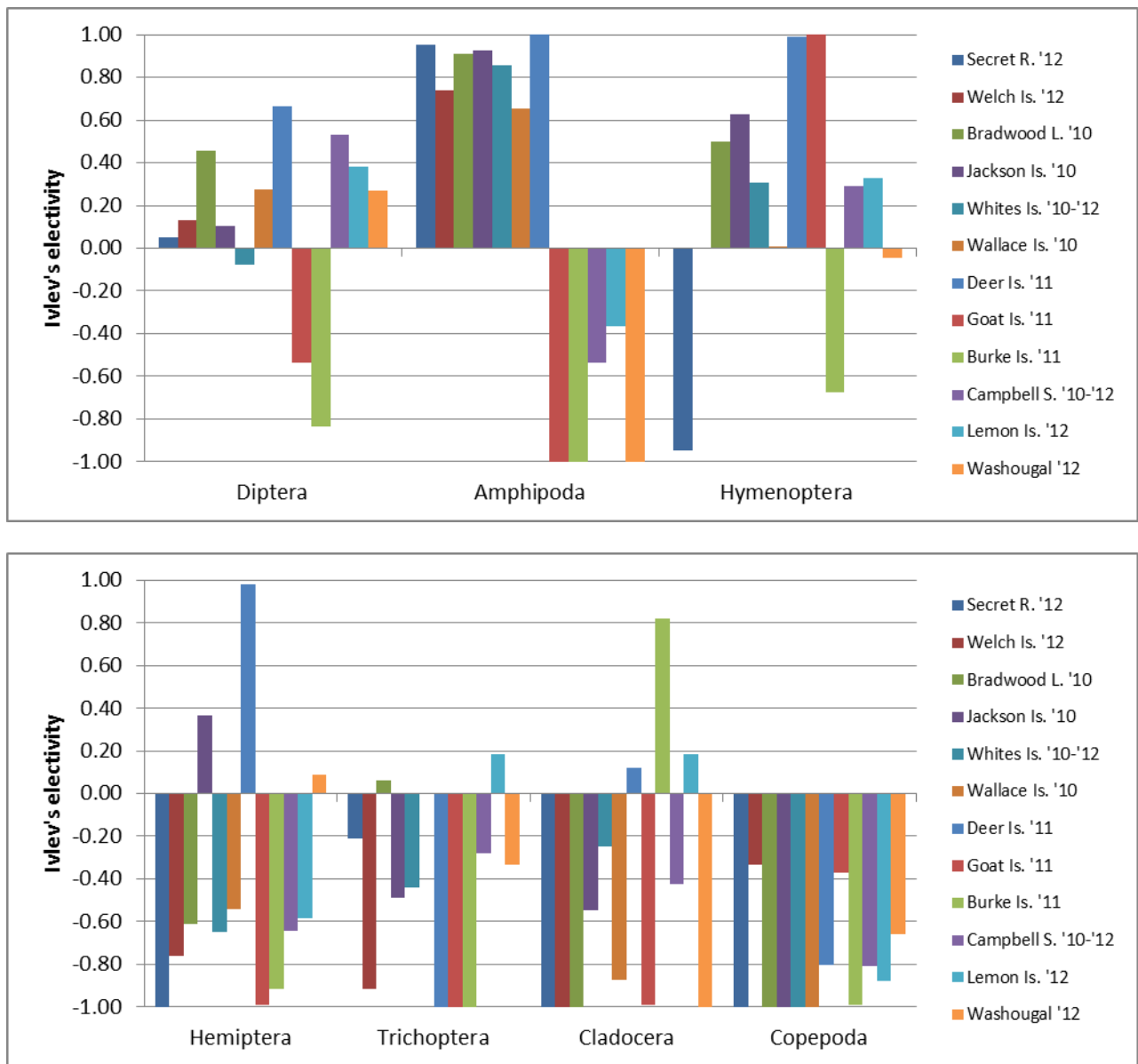
Figure 119 a and b. Ivlev's electivity values for juvenile Chinook collected from sites sampled in 2010-2012.

5 Discussion

5.1 Mainstem conditions

Several years of observations from RM-53 allow for some generalizations to be made concerning the biogeochemistry and water quality of the mainstem Columbia River. One important observation was that water temperatures above the recognized threshold for suitable salmon habitat (Bottom et al. 2011) occur each summer in the mainstem portion of the river. The high resolution data allows for an exact tally of the number of days above 19°C that showed 2009 to have significantly more days than 2010 or 2012 (Table 11). The depth where the sensor platform is located is over 20 meters deep and the

water column at this site is well mixed - therefore no localized temperature gradients develop in the water column. Therefore, this site likely represents the coolest temperatures anywhere in the Lower Columbia River, since shallow fringes will undergo localized warming relative to the mainstem. This observation suggests that during these periods there are no habitat refuges below 19 °C in the entire lower Columbia River above the saltwater interface.

One important correlation observed every winter at RM-53 was the increase in discharge and the concomitant increase in nitrate concentration, turbidity, and CDOM that lasted between days and weeks. These episodic discharge events were typically caused by large storms from the Pacific Ocean that resulted in significant precipitation in the Columbia River watershed. Furthermore, these events were more closely associated with increased discharge in lower Columbia River tributaries compared to the flow of the mainstem measured at Bonneville Dam, as illustrated by the difference in discharge between RM-53 and Bonneville Dam (Figure 7). Part of the reason for this pattern can be attributed to the runoff that directly results from precipitation in the lower elevation regions of the lower Columbia River, whereas precipitation in the higher elevations (above Bonneville Dam) adds to the region's snowpack and does not immediately contribute to discharge. In addition, regulation of discharge by Bonneville Dam (and dams upstream of Bonneville) acts to decouple winter precipitation from discharge. The Willamette River is the largest tributary in the lower Columbia River, has fewer dams, and its watershed has many low elevation regions, therefore this tributary is likely an important source of water and materials detected at RM-53 during these episodic events.

Nitrate flux at RM-53 was determined for the 2009-2010 water year to be approximately 47 thousand tonnes N, and occurs mostly during winter and before the spring freshet. This value agrees with historical estimates by the USGS (see Aulenbach 2006) using a more coarse temporal resolution for estimating fluxes that indicate that the Columbia River flux varied between 32 – 117 thousand tonnes per year during 1975-2002. These values are similar to the Susquehanna River (29-106 thousand tonnes yr^{-1}) and an order of magnitude smaller than the Mississippi River (495-1820 thousand tonnes yr^{-1}) for the same time period. Both the flux in the Susquehanna and the Mississippi are associated with deleterious environmental conditions related to eutrophication; however these same issues are less apparent in the Columbia River estuary and the coastal ocean. The reasons for the lower expression of eutrophication at the coastal zone is likely due to the seasonal timing of the majority of nitrate flux, which occurs during the winter period when algae growth is relatively low, and by the dominant forcing associated with ocean upwelling along the Oregon and Washington coast that dictate much of the ecosystem biogeochemical conditions during spring and summer.

Since the installation of an observatory node at RM-122 a more quantitative estimate of nitrate loading can be attributed to the lower Columbia River tributaries, since most of the major tributaries enter the mainstem Columbia downstream of this station. Although we are still early in the project, data from this season's early precipitation events clearly showed that nitrate flux was strongly associated with discharge and high nitrate concentrations occurring downstream of RM-122, with the highest values (> 75% of total nitrate flux) occurring in late November (Figure 18).

Another important observation from the mainstem river measurements is that fluvial phytoplankton exhibit seasonal patterns consistent with the general temperate ecological pattern observed for freshwater and marine primary production. The temperate zone pattern typically includes a spring bloom in phytoplankton caused by increased daylight that relieves light limitation of phytoplankton growth, followed by a bloom crash caused by nutrient exhaustion. The summer period is characterized by a lower biomass of algae that grow using recycled nutrients, and a fall bloom that results from

additional nutrient inputs from late season storms. The Columbia River phytoplankton population exhibits many of these general characteristics, with a few exceptions. It does not appear that nutrients lead to a bloom crash during the spring; rather the spring freshet can restrict phytoplankton biomass, likely as a result of high turbidity and high turbulence. Summer blooms are associated with low nutrients, but it remains unclear what limits phytoplankton biomass. Since nitrate always remains above 5 μM, it is likely another nutrient, such as phosphate, that limits growth rate.

Net ecosystem metabolism measurements corroborate the chlorophyll a measurements and indicate that primary production is highest during spring and summer in the mainstem portion of the river. Rates of primary production compared to estimates of biomass suggest that the doubling time of algal biomass is on the order of days (data not shown). Since biomass does not continuously increase during the same periods of high positive primary production (especially in summer), there must be considerable grazing and transfer of primary production into the food web of the river and coastal zone. These observations of algal abundance and phytoplankton growth are in agreement with earlier studies that clearly show riverine phytoplankton are abundant (Sullivan et al. 2001; Small and Prahl 2004) and that the Columbia River is likely 'greener' than similar big rivers of the world.

The flux of phytoplankton POC measured at RM-53 represents an important source of carbon to the saltwater region of the Columbia River estuary (Reach A). Unlike the nitrate flux which occurs mainly in winter, the POC flux occurs in spring and summer when conditions in the coastal zone are more biologically active. The water column of Reach A is typically considered to be net heterotrophic – that is, more organic carbon is respired that is produced (Small and Prahl 2004), and the allochthonous source is in large part from Columbia River phytoplankton that lyse upon reaching the saltwater environment. Hypoxia has recently been measured in the Columbia River estuary (e.g. Roegner et al. 2011) and together with ocean acidification is likely to persist for the next decades. Since both of these processes are further exacerbated by the respiration of phytoplankton POC from the Columbia River, it will be important in future years to quantify the POC flux (e.g. Figure 11and Figure 15) in relation to the timing of coastal upwelling and subsequent hypoxia and ocean acidification conditions in the estuary.

5.2   Abiotic site conditions

Generally, salmonids were caught during NOAA's fish sampling when water column conditions were cool, well-oxygenated, and moderate in pH. Juvenile Chinook salmon, for example, were typically present from February, when sampling was initiated, until July when temperatures and other water quality conditions were typically outside the range considered suitable for these fish.

However, salmonids were occasionally caught despite stressful water-quality conditions at Campbell Slough in 2010 and 2012. Chinook were present at Campbell Slough on July 19, 2012, despite water temperatures ranging from approximately 19 to 21 °C. However, despite the high temperatures, dissolved oxygen concentrations were suitable for salmonids. During NOAA's June and July 2010 fish sampling, Chinook were also present at Campbell Slough, even though dissolved oxygen concentrations had been continuously less than 8 mg/L for days. During the July sampling, the water temperature had been continuously greater than 17.5 °C for seven days, so it was particularly surprising for Chinook to be present. Chinook were also present at Ilwaco during the May and June fish sampling events in 2012. It should be noted, however, that the number of salmon present during these stressful periods was typically very small, with only one or two fish being encountered during sampling. Higher salmon densities were consistently observed during the late winter and spring when water quality conditions

were more favorable. The occasional presence of juvenile salmon at times when water quality conditions are unfavorable suggests that in some cases salmon may have been in the area only transiently, and so would not be greatly affected by unfavorable conditions, or that there may be refugia into which fish may move when water temperatures become too high or dissolved oxygen too low.

Water-quality parameters measured at four of the trend sites showed some predictable seasonal patterns during the spring–summer monitoring period. Generally, water temperatures at all sites increased during the monitoring period and became warmer than is suitable for juvenile salmon by July. Because warmer water can hold less dissolved oxygen than cooler water, the observed overall pattern of decreasing dissolved oxygen concentrations later in the monitoring period was also expected. However, dissolved oxygen concentrations and pH also vary as a result of primary production and decomposition in the water column. Dissolved oxygen concentrations dipped below the acceptable threshold for juvenile salmon health frequently at Ilwaco, intermittently at Franz Lake Slough and Campbell Slough, and only late in the season at Whites Island. Based on the patterns in temperature and pH, the frequent dips in dissolved oxygen at Ilwaco and the end-of-season declines at Whites Island appear to be due to the tidal flushing leaving the channel very shallow and prone to large temperature increases. Conversely, the intermittent dips in dissolved oxygen patterns at Franz Lake Slough and Campbell Slough matched closely with pH, so it appears that dissolved oxygen patterns at those sites is more driven by primary production and decomposition, particularly early in the monitoring season (April–May).

Although only two years' data are available for three of the four sites, it is clear that within-site differences between years result from differences in weather (particularly temperature) and Columbia River flow. Differences in water-quality parameters across the four trend sites strongly depends on the magnitude of tidal flushing, with Ilwaco, the most tidally dominated site, showing less seasonal variation and larger-magnitude differences at a shorter time scale (within a day). In contrast, the more river-dominated, upstream sites showed less short-term or daily variability, but changed more from the start to the end of the monitoring season. This suggests that the upriver sites (Franz Lake Slough and Campbell Slough) may have suitable conditions for juvenile salmon for longer periods, but that when conditions become unsuitable, they are slower to recover to because of the slower flushing at those sites. Cut-Off Slough at the Whites Island site stands out among the monitoring sites having had the most consistently suitable water-quality conditions for juvenile salmon during the monitoring periods in 2011 and 2012. Among the four water-quality monitoring sites, it was the site with juvenile salmon caught most consistently and in the highest numbers during NOAA's fish sampling. This would be expected given the favorable water-quality conditions and its short distance off of the main stem of the river.

5.3   Habitat Structure

Sites in our study have TOC concentrations below those found in other estuarine marsh systems. While some of the 2012 percentages are slightly higher than those from previous years, the difference is not comparable to the differences we found from other studies where TOC ranged from 13% to 30% (e.g. Elliot 2004). Interestingly, the TOC was higher in the Ilwaco samples than in the Secret River samples; however the amount decreased more rapidly with depth at the Ilwaco site. This pattern could be related to the difference in age of the sites, with the Ilwaco site having developed sometime in the past century and the Secret River site present on the historical maps of the late 1800's.

The concentration of TOC in soils and sediment is a function of plant production, the accumulation of biomass in the belowground, decomposition of leaf litter on site, sedimentation rate, microbial activity,

relase of CO2 to the atmosphere and other factors. We speculate several reasons for this condition. First, accretion rates appear to be somewhat high relative to many other tidal marsh sites in the region. This could dilute TOC retained in the sediment. Second, we suspect that organic matter that could deposit in the sediments is removed via dynamic water flows during the spring freshet, thus removing a source of deposited organic matter to the belowground pool. Third, because the TOC concentrations decline dramatically below 10-15cm deep in the soil, decomposition is rapid, the sites are relatively new in development, or less belowground production takes place and less surface detritus remains long enough on the surface of the sediment to be buried. The pulsed, short term, nature of primary production may favor above ground development as opposed to belowground root and rhizome development.

Sediment accretion rates are variable in time and space within a fairly narrow range of rates (-0.5 to 3.0 cm/yr). These differences are likely due to a combination of factors at the sites including vegetation, elevation, and flooding impacts. The variability in the rates at the Whites Island site between 2008-09 and 2009-2010 may be in part due to a change in vegetation from forget-me-not (*Myosotis scorpioides*) to include more reed canary grass (*Phalaris arundinacea*). The highest rate was measured at FLM in 2011, which would be expected following the extended high water period at this site. Likewise, rates greater than 1.0 cm/year were also observed at CS1 and CLM where the high water likely increased sedimentation rates also. In 2012 sediment accretion rates were lower than the rates in 2011 at all three upriver sites. This suggests that perhaps after high water years, such as 2011, there is movement of the sediment that accreted during the flood causing lower rates the following year or even erosion as is the case at Franz Lake in 2012.

Our results indicate that inundation frequencies vary in the estuary, generally increasing with rkm. In addition, winter high water results in greater inundation frequency in the lower portion of the LCRE during this time. The reason for this pattern is the greater input from "west-side" rivers to these lower zones in the winter time, whereas the upper part of the LCRE is hydrologically driven by the spring snow melt in the upper Columbia basin (Jay et al. in review). This pattern is consistent between years, however the location of where the transition between winter high-water dominance and spring high-water dominance is variable depending of annual conditions.

Elevation of emergent wetlands in the LCRE covers a very narrow range (Sager et al. 2013). Elevations measured in 2012 confirm this finding, with sites measured relative to CRD throughout the estuary covering a range of only 1.83 m. The results of our analysis of elevation boundaries for a few key species indicate that indeed the plants were likely affected by the higher water however the cumulative effects of two high water years was required to result in migration of elevation boundaries that were observed in 2012 at Cunningham Lake and Campbell Slough. These findings imply that climate change and the possible higher and/or longer inundation periods could have an effect on vegetation communities in the LCRE.

The SEV is a means of quantifying inundation, allowing comparisons between sites, evaluating interannual variability, and examining the effects of inundation on vegetation composition, cover and biomass production. The annual results of growing season SEV allow a comparison spatially and show the dramatic increase in inundation with distance from the mouth of the River. As rkm increases so too does the fluvial dominance in driving hydrologic patterns. In these upper portions of the River we show that high water levels in 2011 and 2012 resulted in lower vegetative cover, lower biomass, and at one site a shift in vegetation community. Periods of high water during the growing season can suppress growth via hypoxia as well as light attenuation.   Hypoxia reduces the ability of plants to extract

nutrients from the soils. Light attenuation reduces the productivity to respiration ratio thus reducing the ability of the plant to accumulate biomass.

A shift in vegetation community occurred at Franz Lake Slough in 2012. The species dominance shifted from reed canary grass (*Phalaris arundinacea*) to water smartweed (*Polygonum amphibium*). This shift was not observed in other upriver sites with similar or greater inundation, therefore there must also be a site specific driver for the shift. One plausible cause may be the prior presence of the species at the site and the high water conditions in 2011 and 2012 were conducive to the spread of this native aquatic species. The morphology of the plant allows it to grow 2-3 meters tall or greater with stems and leaves that float on the surface of the water allowing it to continue to photosynthesize in high water conditions (Figure 120). This species shift is another indication that prolonged alteration in hydrology, as might be expected from climate change or different river management, could have dramatic effects on the fluvial dominated wetlands of the LCRE.

| 21 July 2012 | 30 August 2012 |

Figure 120. The Franz Lake Slough site during high water (left) and low water (right) conditions showing dominance of water smartweed (*Polygonum amphibium*) at the site in 2012.

5.4 Food web

In 2011 and 2012, the abundance and productivity rates of periphyton (attached algae) were highest at Ilwaco and generally lower at sites moving upstream. Conversely, phytoplankton (suspended algae) abundance and productivity rates were generally greater at upstream sites compared to downstream. Seasonal patterns in algal abundance and productivity differed across sites and years, but the analysis only includes two years of data thus far. The high abundance of periphyton at Ilwaco was due to the presence of large, filamentous periphyton that were not present at the upstream sites. The strong tidal action at Ilwaco would favor the production of attached algae, whereas the sites that are less impacted by tidal flushing have conditions that are more amenable to phytoplankton production. The results from 2011 and 2012 suggest that the production and availability of autochthonous organic matter varies across sites and over time during the April–July monitoring season and across years. However, it is not possible to determine what this variability means in terms of how the sites differ in providing organic matter that is actually utilized in the food web, until the stable isotope food web data are available.

The phytoplankton assemblage was consistently dominated by diatoms at most of the sites, similar to observations from 2011 (Sagar et al. 2013; data not shown). The patterns in total abundance of phytoplankton at the different sites as determined by cell counts agreed reasonably well with the chlorophyll data. Total abundance was higher prior to or in the early stages of the freshet in all cases except for Secret River. At a few sites, a generalized grouping of "flagellate" species, most of which are small (<10 μm) exceeded the proportional abundance of diatoms, especially at Campbell Slough (throughout the set of observations) and Franz Lake Slough in mid-May. A working hypothesis is that the flushing of shallow water habitats by high Columbia River water levels tends to produce a habitat more favorable for diatom population growth rather than one where flagellate species have a competitive advantage. Notably, a shift from diatoms to flagellate species tends to occur in environments characterized by high nutrient loads (Turner et al. 1998). A shift toward flagellate species may also result from a difference in water column structure, with calm, poorly flushed waters promoting the sinking of heavily silicified diatoms to the river bed, which could lead to an accumulation of organic matter in poorly flushed areas. The higher ratios of flagellates to diatoms coincided with higher concentrations of both organic nitrogen (measured as TKN) and organic phosphorus, which likely reflect the higher competitive abilities for organic nutrients of flagellates relative to diatoms. Although diatoms tend to be associated with high primary and secondary productivity (as compared to many flagellate taxa), it may be that environments that favor retention of diatoms (for example, through sinking and slow water exchange) may prove to be good areas for benthic taxa.

At the Ilwaco marina site, there was an increasing prevalence of cryptophyte taxa over time among the (albeit few) observations. Cryptophytes are consumed as prey for photosynthetic ciliates that bloom in the marine-influenced areas of the Columbia River estuary each summer (August – October). Blooms of the ciliate, *Mesodinium rubrum*, may have an important influence on estuarine habitats through their intense rates of photosynthesis and growth (Herfort et al., 2012) and the corresponding influence on dissolved oxygen levels and pH (Tawnya Peterson, OHSU, unpubl. data).

At both Campbell Slough (Reach F) and Franz Lake Slough (Reach H), the minimum oxygen saturation values were associated with declining phytoplankton abundances, lower chlorophyll concentrations, and moderate levels of dissolved nitrogen and phosphorus observed later in the year (~mid-July). Periphyton biomass was relatively high at Campbell Slough in the mid-summer (July) and nutrient addition experiments indicated that primary production by periphyton was co-limited by both nitrogen and phosphorus at that time. Diversity among zooplankton species (as estimated by the Shannon-Weiner diversity index) was relatively low compared to other times of year or other sites, although overall the diversity of fish species was high at this site, although it should be noted that Campbell Slough had a high proportion of non-native fish species.

The open water zooplankton species assemblages differed markedly across sites, with those occurring downstream of Beaver Army Terminal (BAT; river mile 53) but upstream of the estuary (ie, Secret River, Welch Island, and Whites Island) being dominated by rotifers throughout the time series and those located upstream of BAT being numerically dominated by rotifers only prior to the period of elevated discharge. Contributions by larger species, including copepods and cladocerans were greater at Campbell Slough and Franz Lake Slough compared to the downstream sites, particularly later in the year. Interestingly, the zooplankton assemblage was entirely dominated by copepods at Ilwaco throughout the series of observations. This was also observed in 2011 (Sagar et al. 2012).

In concert with juvenile salmon gut content analyses from NOAA, the results of the stable isotope food web analysis will provide information about which invertebrate species and types of organic matter contribute to the juvenile salmonid food web.

Aquatic Diptera and Amphipods contribute 63% of the consumed prey biomass overall at these 12 sites. Additional prey taxa that are energetically important include Trichoptera, Hemiptera (primarily aquatic but some terrestrial taxa as well), small fish, Cladocera and Oligochaeta. Almost all of these are of aquatic origin, and most are consumed in their aquatic life stage. Although the feeding habitats of these taxa vary considerably, most are collector gatherers, or scavengers, consuming detritus or fine particulate organic matter and the associated microbial communities associated with that material.

Sediment-associated (benthic) macroinvertebrate taxa differed according to site, with the seawater-influenced site (Ilwaco) supporting Corophium spp., in contrast to sites further upstream. All sites were dominated by oligochaete and nematode worms. The contribution of Ceratopogomonidae (Order Diptera) was higher at Whites Island than the other sites examined.

The limited data available on stable isotope ratios in juvenile Chinook salmon bodies suggests that there may be some differences in diet among marked and unmarked fish, and fish of different stocks and from different sites. Most notable trends were lower ∂^{15}N values in fish from Reach H and from Interior Columbia River stocks (Upper Columbia summer/fall Chinook, Snake River fall Chinook, and Deschutes River fall Chinook). ∂^{15}N values also tended to be lower in unmarked fish than in marked fish. These findings suggest that marked (hatchery) fish may be feeding on items from sources higher in the trophic levels than unmarked, presumably wild fish. This would be consistent with the use of hatchery feeds derived from fish meal and fish oils, and also could be reflective of the somewhat larger size of most of the hatchery fish collected from field sites. The results also suggest that fish from the lower part of the estuary may be feeding on items from higher in the trophic levels than those originating in the interior basin or rearing at sites in the Gorge. Another possibility is that fish from lower in the estuary are exposed to nitrogen from other sources, such as sewage (Hansson et al. 1997), as there is evidence that fish from this area are exposed to wastewater compounds (Sagar et al. 2013).

The consistent difference between the high densities of invertebrates in the emergent vegetation habitat compared to the low densities in the open water habitat is one of the most enduring and striking result of these surveys. While we did find a relationship between Diptera abundances and the percent of live vegetation cover within emergent vegetation transects, other factors are likely contributing to the production of invertebrates in these emergent vegetation habitats. Although not explicitly evaluated, we propose inundation patterns and rates of primary production may be additional factors that contribute to the local production of detritus and FPOM, and potentially the secondary production of Diptera and Amphipods in these shoreline habitats.

5.5   Fish Use

Due to the extreme high water conditions in 2012, and the difficulty of sampling some sites, our ability to monitor salmon occurrence in 2012 was limited, especially at Franz Lake Slough, Campbell Slough, and Reed Island. Chinook salmon were found at all of the sampling sites with the exception of Franz Lake Slough, which was not accessible from April through July, the period when juvenile Chinook salmon are most abundant. Previous sampling has shown that Chinook salmon do utilize the site (e.g., Sagar et

al. 2013). Coho salmon were found at all the 2012 sampling sites except for Campbell Slough and Whites Island, though the largest numbers were seen at the sites in Reach G. This is consistent with our earlier findings indicating that coho salmon are more abundant upriver (e.g., in Reach H) than in reaches closer to the river mouth (Sagar et al. 2013). Chum salmon were found at Ilwaco Slough, Secret River, Welch Island, and Government/Lemon Island. All chum as well as the majority of Chinook and coho salmon from most of the sampling sites were unmarked, presumably wild fish. However, at the Washougal site, there were large catches of marked (hatchery) Chinook and coho in May and June. This is not surprising as the Washington State Department of Fisheries Washougal Hatchery is located nearby and releases both Chinook and coho juveniles into the Washougal River (Columbia River DART; http://www.cbr.washington.edu/dart/hatch.html).

We do not yet have estimates of salmon densities at the new sampling sites, but catch numbers indicate that both of the new Reach B sites, Secret River and Welch Island, as well as two of the new Reach G sites, Government/Lemon Island and Washougal, support significant numbers of unmarked Chinook salmon from February or March through July, as well as smaller numbers of unmarked coho and chum. Because it was inaccessible through so much of the sampling season, we could not evaluate salmon occurrence at Reed Island in Reach G.

Ilwaco Slough, Whites Island, Campbell Slough, and Franz Lake Slough were all resampled in 2012 to evaluate trends in juvenile salmon occurrence. Because Franz Lake Slough was so rarely accessible, trends at this site could not be assessed. At Whites Island and Campbell Slough, patterns of salmon occurrence were comparable to those observed in earlier years (Sagar et al. 2013). Although Ilwaco Slough was consistently sampled from February through October, and several salmon species were observed there, salmon numbers were consistently low. In 2011, there was a large catch of chum salmon at the site in April, but no other salmon were found (Sagar et al. 2012). These data suggest that, with the possible exception of outmigrant chum, this site is not heavily utilized by juvenile salmon.

At those sites that could be fished throughout the sampling period, unmarked juvenile Chinook salmon were typically present from February, when sampling began, through June or sometimes July, with the largest number of fish present in May and June. Marked Chinook were found over the same time period, but only rarely in February, March, April or July; almost all marked Chinook were caught in May and June, consistent with the period of hatchery releases (Columbia River DART; http://www.cbr.washington.edu/dart/hatch.html). Chum salmon were present from February through April, while unmarked coho salmon were in small numbers from February through October. Marked coho were nearly all collected in May, which again is consistent with the usual times of hatchery releases (Columbia River DART; http://www.cbr.washington.edu/dart/hatch.html).

The more extended fall and winter sampling, which we carried out for the first time in 2011-2012, establishes the importance of late winter and early spring as a rearing and migration period for juvenile salmon, particularly for unmarked juvenile Chinook and chum salmon. It is also significant that unmarked Chinook salmon were found in February and March at sites throughout the estuary, at Secret River and Welch Island in Reach B, as well as Government/Lemon Island and Washougal in Reach G. The presence of unmarked Chinook salmon in February and March is consistent with findings of other researchers in the estuary (Sather et al. 2009; Johnson et al. 2011; Roegner et al. 2012).

Fish community characteristics showed distinctive spatial patterns along the river. Stickleback were consistently one of the dominant species in all reaches, but the other species present varied considerably. The Ilwaco Slough site in Reach A was very different from other sites, as catches here

contained many saltwater species not present at any of the other sampling sites. We also observed higher proportions of non-native species, including carp, killifish, pumpkinseed, and yellow perch, in reaches farther from the mouth of the estuary, particular between Reaches E and H. Fish assemblages found at the new Reach B sites, Secret River and Welch Island, were similar to those observed at the Reach C site, while those at the new Reach G sites, Government/Lemon Island, Washougal, and Reed Island, were comparable to those found at the Reach H sites.

Fish community characteristics also showed a tendency to vary seasonally. Salmonids and other native species were most prevalent earlier in the sampling season, but later in the season, as temperatures increased, non-native species such as carp, crappie, killifish, and yellow perch, became more prevalent. Some native resident fish species, including chiselmouth and white sucker, were found more frequently during the summer months.

Data are not yet available for the 2012 sites for growth, lipid content, or contaminants, but significant differences were found among sites in size class distribution, length, weight, and condition factor of unmarked Chinook salmon. Welch Island and Washougal had particularly high proportions of juvenile Chinook salmon fry, less than 50 mm in length. This could be an indication that these sites have particularly high levels of natural salmon production in tributary river systems, or that these habitats are especially suitable for fry stage migrants. There were no very clear patterns in condition factor by reach or spatially along the river. However, at the Washougal site in Reach G, CF was significantly lower than the overall average for both marked and unmarked juvenile Chinook, even after accounting for the effect of sampling month.

In general, marked Chinook salmon were most abundant in catches at the upriver sites, in Reaches E-F, although there is some variation from site to site, and from year to year at the trend sites. Marked salmon were not especially abundant at the majority of 2012 sites. At the Reach B sites, sampled for the first time in 2012, marked salmon made up less than 10% of the Chinook catch, comparable to levels at the sites in Reach C. Similarly, at Government/Lemon Island in Reach G, only about 7% of the Chinook catch were marked fish. Washougal wetland, in Reach G, was an exception; here, marked salmon made up 45% of the Chinook catch. Reed Island could not be sampled for much of the year, so catch composition here could not be evaluated. However, the one Chinook salmon that was found there was marked.

Coho salmon were rarely caught in Reaches A-F, and those that were found were typically unmarked. However, both marked and unmarked coho salmon were present at the 2012 sampling sites in Reach G, as well as previously sampled sites in Reach H. At the Reach H sites, with the exception of Hardy Slough, the majority of coho salmon were marked fish. The same was true of the Washougal site in Reach G, but at both Government/Lemon Island and Reed Island, the majority of coho salmon were unmarked salmon. Interestingly, at Franz Lake Slough, the only site in Reach H that has been sampled in multiple years, the proportion of unmarked coho salmon in catches has increased between 2008 and 2011.

The high proportion of marked Chinook and coho salmon at Washougal wetland is probably related to the proximity of the Washington State Department of Fish and Wildlife Washougal Salmon hatchery nearby. This hatchery releases both Chinook and coho juvenile into the Washougal River (Columbia River DART; http://www.cbr.washington.edu/dart/hatch.html), which likely contributed to the large catches of marked fish at this site in May and June.

The low proportion of marked coho and Chinook salmon in catches has been very consistent from year to year at Whites Island in Reach C, but more variable at Campbell Slough and Franz Lake Slough in Reaches F and H. This could due in part to our difficulty in accessing these sites during portions of the sampling season. Franz Lake Slough, for example, could not be sampled between March and July in 2012, and Campbell Slough could not be sampled until May. This variation in sampling time will influence the percentages of marked fish in catches.

Chinook salmon stock usage of the EMP sampling sites changes spatially along the river. Among unmarked fish, West Cascades fall Chinook are more abundant in Reaches C and E (data on reaches A and B are not yet available), with Spring Creek Group fall Chinook becoming more prevalent in reaches F-H. While they are present throughout the Lower Columbia River, Chinook stocks from the Interior Columbia, such Upper Columbia summer/fall and Snake River fall Chinook, tend to be more common in Reach H. The spatial pattern for marked fish is fairly similar, with West Cascades fall Chinook being most common in Reach C, and Spring Creek fall Chinook being most common at most of the sites in Reach E-H.

Stock distribution changed with sampling time for some stocks of both marked and unmarked Chinook salmon. Spring Creek Group fall Chinook and Upper Columbia River summer/fall Chinook showed the clearest seasonal patterns in unmarked fish; Spring Creek Group fall Chinook were more abundant early in the sampling season, while Upper Columbia summer/fall Chinook were found in greater numbers later, in June, July, and August. Among the marked fish, Spring Creek Group fall Chinook dominated in April and May, and West Cascades fall Chinook became more abundant later in the sampling season.

For those sites that have been sampled in multiple years, patterns of salmon occurrence and fish community composition are fairly consistent.   At Ilwaco Slough, for example, aside from a large catch of chum salmon at a single sampling event in 2011, few salmon were caught, suggesting this site may be limited in its ability to support juvenile salmon. At Whites Island, unmarked Chinook salmon, especially smaller fry and fingerling size classes, consistently dominated catches. At Campbell Slough, Chinook salmon were also the dominant salmon species, but catches included substantial proportions of marked as well as unmarked fish. Year to year variability at Franz Lake Slough has been difficult to evaluate because of problems accessing the site for much of the sampling season in both 2011 and 2012.

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7 Appendix
Appendix A. Site Hydrographs
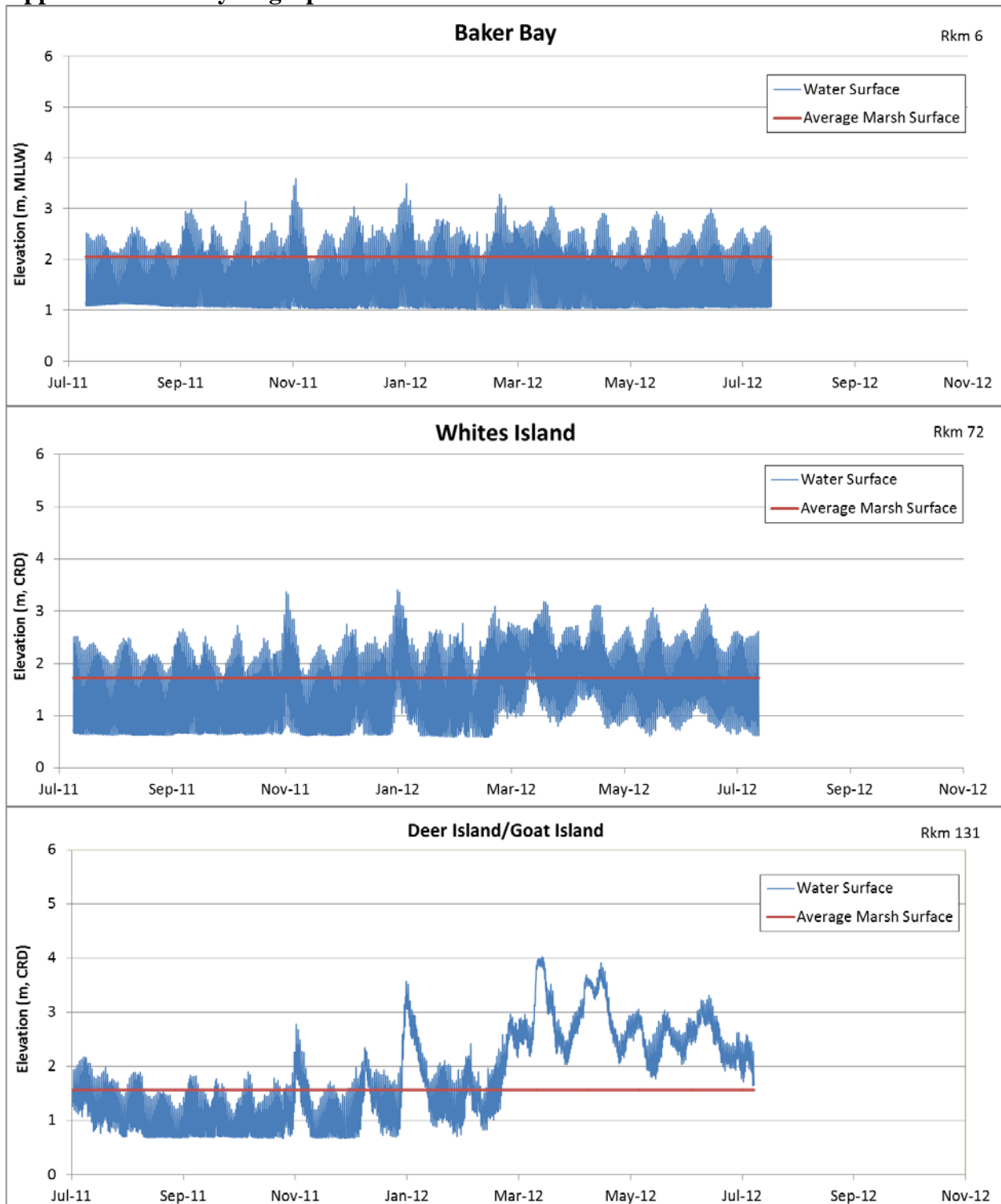
Figure A – 1. Water surface elevation data from the study sites where sensors were deployed 2011-2012. The red line represents the average elevation of the marsh sampling area.
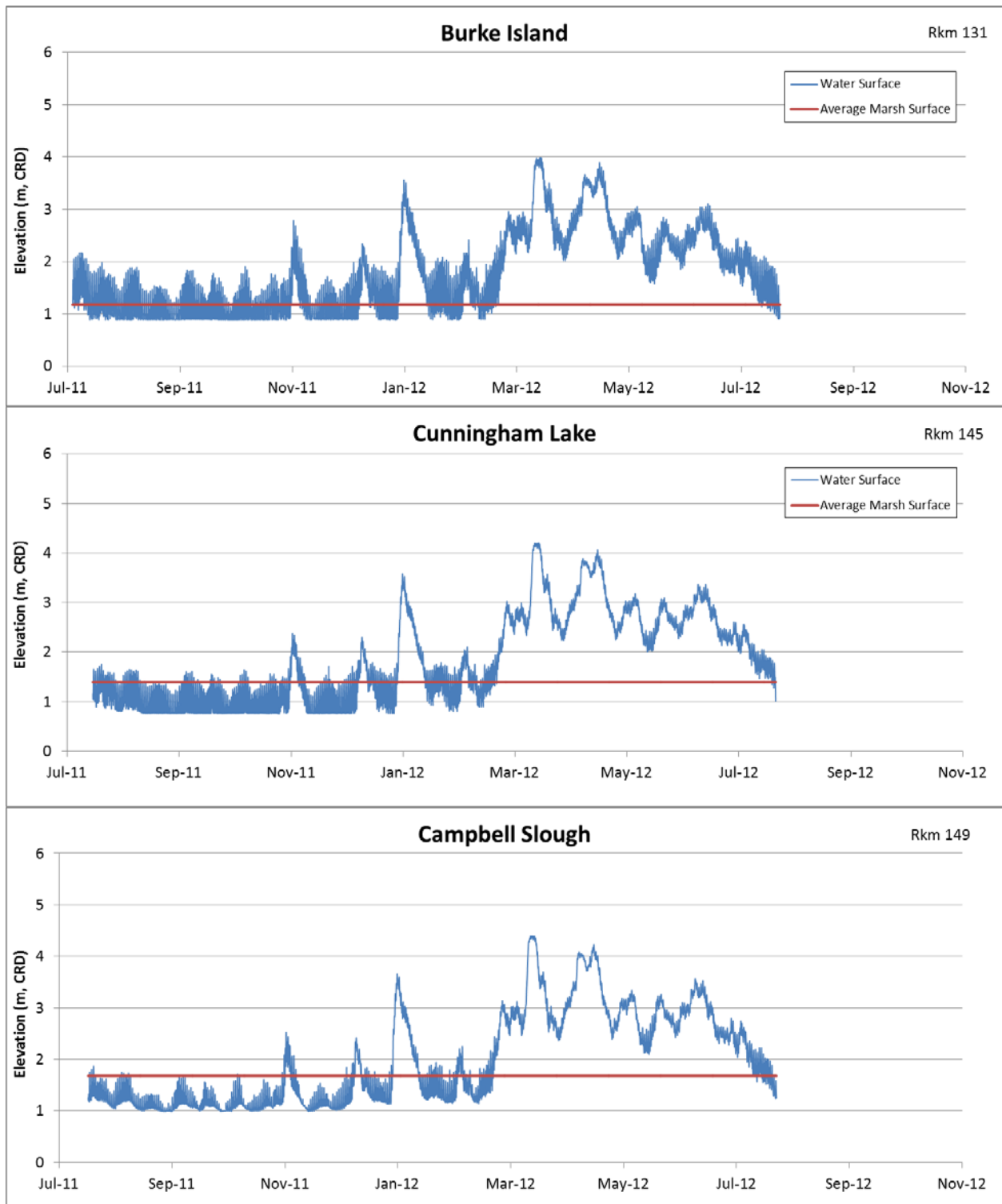
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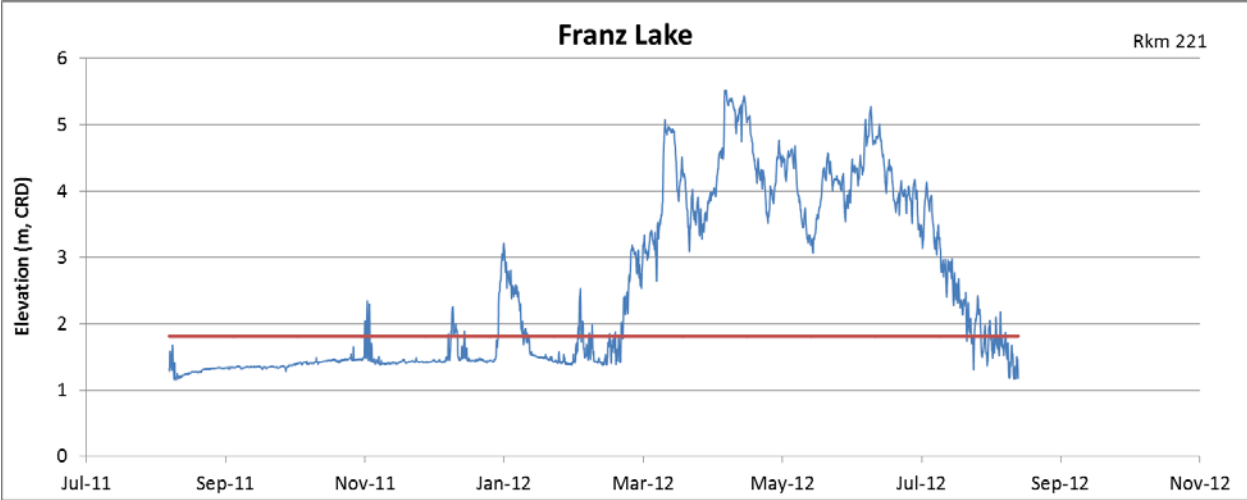
Figure A-1. Continued

Appendix B. Site Maps
NOTE: Sites that have been previously mapped (trend sites) and where no obvious changes were observed, were not re-mapped this year. Therefore, in this Appendix we include the following:

- Maps from 2011 for the trend sites that had no observable change (Ilwaco, Whites Island, and Campbell Slough)

- Maps from 2012 for the trend sites that did appear to change (Cunningham Lake and Franz Lake Slough)

- New sites for 2012 (Secret River, Welch Island, Government/Lemon Island, Washougal River, and Reed Island).

Baker Bay, 2011
GPS Mapping

Vegetation Communities
- bare ground
- Juncus bufonius/bare ground
- Zannichellia palustris/bare ground
- bare sand
- C. canadensis/C. lyngbyei
- Carex lyngbyei
- Zannichellia palustris/open water
- Typha spp.

Monitoring Locations
- Sediment accretion stakes
- Depth sensor
- Photo point
- Cross section
- Vegetation/Elevation Transect

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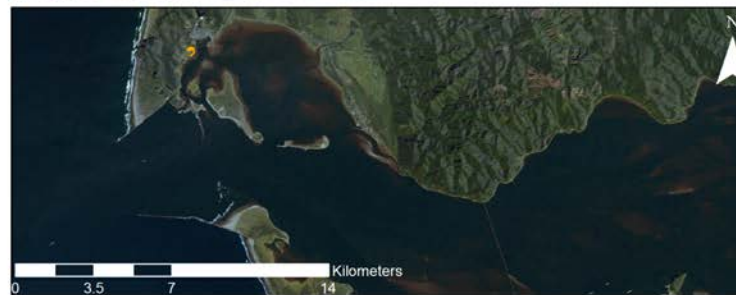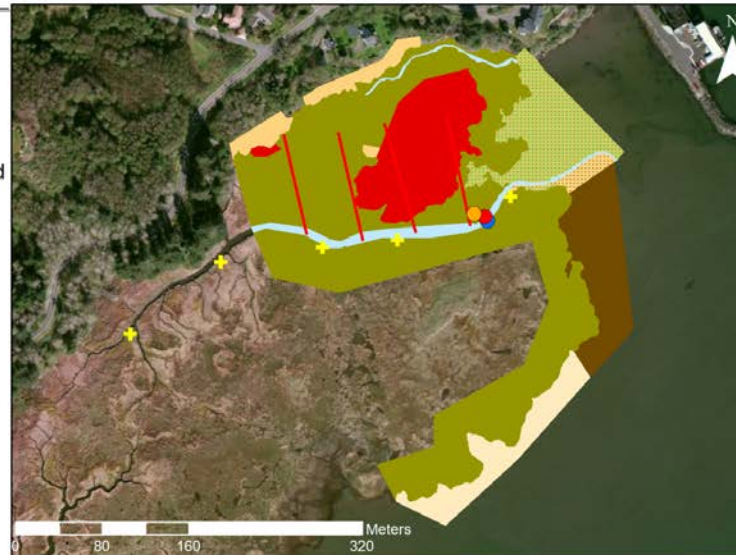
Secret River Marsh, 2012
GPS Mapping

Vegetation communities
- Channel
- D. cespitosa and P. arundinacea
- Deschampsia cespitosa
- Eleocharis palustris and S. tabernaemontani
- Submerged aquatic vegetation
- Bare ground
- Mixed P. arundinacea
- Phalaris arundinacea
- Schoenoplectus tabernaemontani

Monitoring locations
- Depth sensor
- Sediment accretion stakes
- Cross section
- Photo point
- Vegetation/Elevation Transect
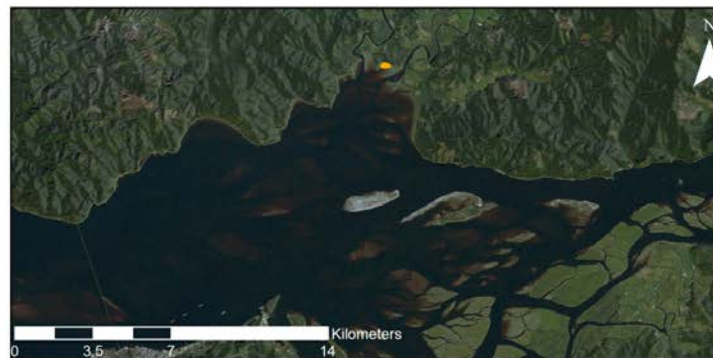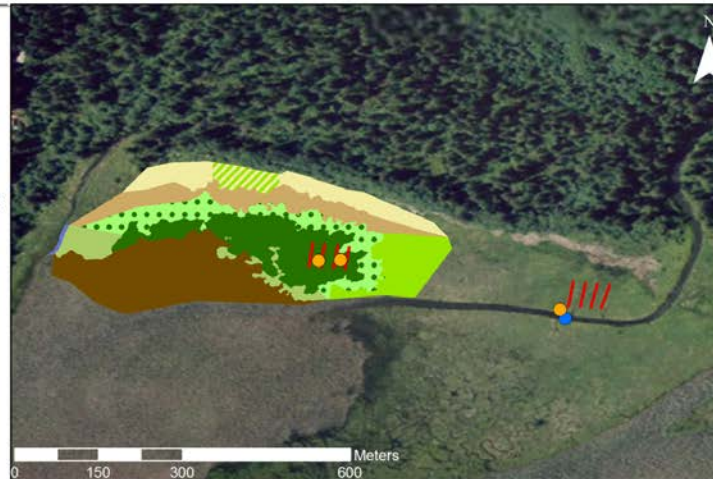
Welch Island, 2012
GPS Mapping

Vegetation communities

- C. obnupta, S. latifolia
- Carex lyngbyei
- Channel
- Eleocharis palustris
- Lythrum salicaria
- Open Water
- P. arundinacea, L. salicaria
- P. arundinacea, S. latifolia
- Phalaris arundinacea
- S. latifolia, P. hydropiper
- Sagittaria latifolia
- Salix spp.
- Salix spp., L. salicaria

Monitoring Locations

- Sediment accretion stakes
- Depth sensor
- Cross section
- PhotoPoint
- Vegetation Survey Line
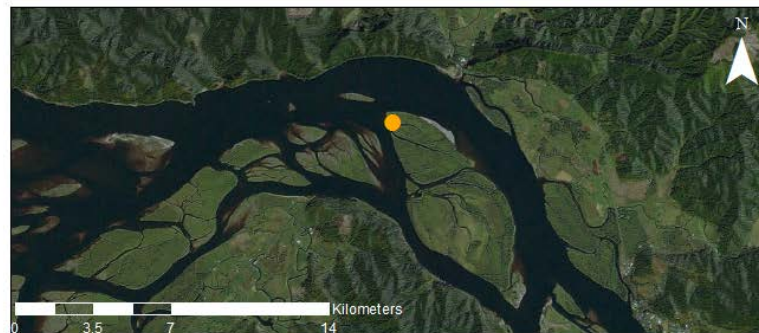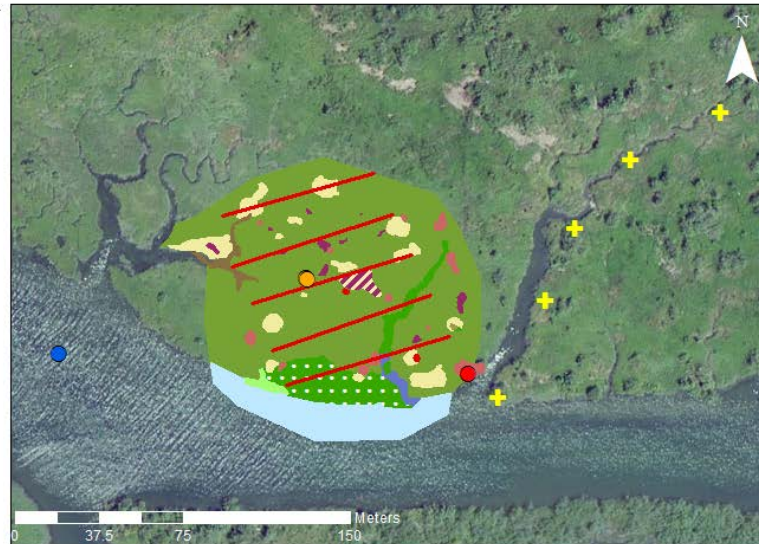
White's Island, 2011
GPS Mapping

Vegetation Communities
- Alnus rubra
- Alisma triviale, Equisetum fluviatile
- bare ground
- Carex sp.
- drift wrack
- Eleocharis palustris
- Equestum fluviatile, S. latifolia
- Iris pseudacorus, high marsh
- I. pseudacorus, P. arundinacea, high marsh
- P. arundinacea, high marsh
- P. arundinacea, Typha sp., high marsh
- Typha sp., high marsh
- Iris pseudacorus
- large woody debris
- open water
- Phalaris arundinacea
- Salix latifolia
- Salix latifolia, open water
- Salix lucida
- Salix sitchensis
- Salix sp.

Monitoring Locations
- Sediment accretion stakes
- Depth sensor
- Photo point
- Cross section endpoints
- Vegetation/Elevation Transect

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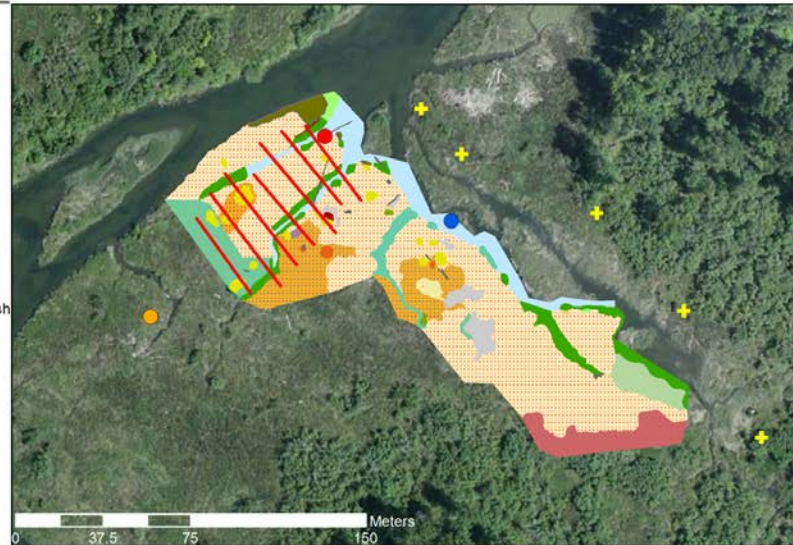
Cunningham Lake, 2012

GPS Mapping

Vegetation Communities

- Fraxinus latifolia
- Open Water
- Phalaris arundinacea
- S. latifolia and Eleocharis palustris
- S. latifolia and P. arundinacea
- Salix lucida

Monitoring Locations

- Depth sensor
- Sediment accretion stakes
- Cross section
- Photo point
- Vegetation transects

Campbell Slough, 2011
GPS Mapping

Vegetation Communities
- Phalaris arundinacea
- Eleocharis palustris
- E. palustris/S. latifolia
- Fraxinus latifolia
- F.latifolia/P. arundinacea
- Open water
- Sagittaria latifolia
- Salix lucida
- Sparse P. arundinacea

Monitoring Locations
- Depth sensor
- Sediment accretion stakes
- Cross section
- Photo points
- Vegetation transects
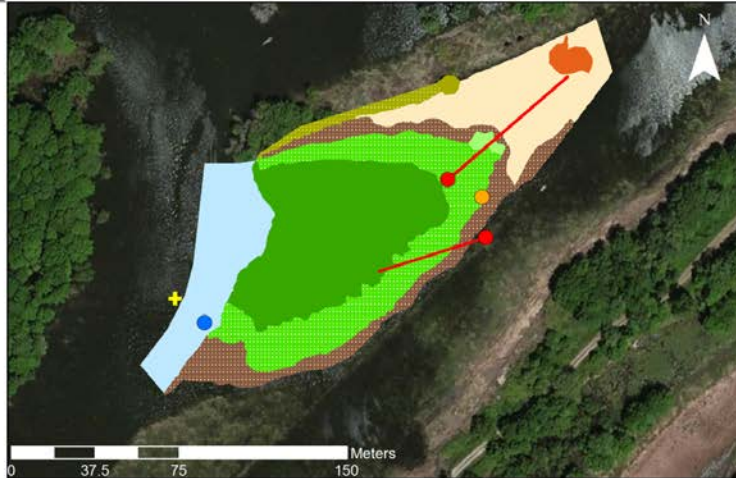
Government Island, 2012
GPS Mapping

Vegetation communities
- Bare Ground
- Eleocharis palustris
- Eleocharis palustris, Bare Ground
- Eleocharis palustris, Channel
- Open Water
- Small Mixed Herbs
- Small Mixed Herbs, Helenium autumnale
- Trees

Monitoring locations
- Depth sensor
- Sediment accretion stake
- Cross section endpoint
- Photo point
- Vegetation transects

Washougal, 2012

GPS Mapping

Vegetation communities
- Bare Ground
- Carex spp.
- Channel
- E. palustris, Open Water
- E. palustris, P. amphibium
- Eleocharis palustris
- P. amphibium, Open Water
- Phalaris arundinacea
- Polygonum amphibium
- Sagittaria latifolia, Open Water
- Salix lucida
- Spiraea douglasii
- Stunted P. arundinacea, P. amphibium
- Stunted Phalaris arundinacea

Monitoring Locations
- Depth sensor
- Sediment accretion stake
- Cross section
- PhotoPoint
- Vegetation transect

Reed Island, 2012
GPS Mapping

Vegetation communities
- Carex spp.
- E. palustris, Bare Ground
- Eleocharis palustris
- Open Water
- Phalaris arundinacea
- S. lucida, Carex spp.
- Salix lucida
- Salix spp. Saplings
- Iris pseudacorus

Monitoring locations
- Depth sensor
- Sediment accretion stakes
- Cross section
- Photo point
- Vegetation transect

0   45   90   180   Meters

0   1.5   3   6 Kilometers
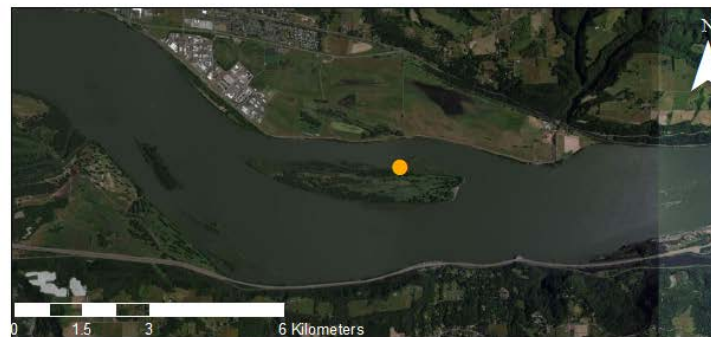
Franz Lake, 2012
GPS Mapping

Vegetation communities
- Bare Ground
- Beaver Dam
- Beaver Lodge
- Carex spp.
- Carex spp., Trees
- Channel
- Eleocharis palustris
- P. arundinacea, S. lucida Saplings
- Phalaris arundinacea
- Polygonum amphibium
- S. latifolia, Channel
- S. latifolia, P. amphibium
- S. lucida, P. amphibium
- Sagittaria latifolia
- Salix lucida
- Salix lucida Saplings
- Trees

Monitoring Locations
- Depth sensor
- Sediment accretion stakes
- Cross section endpoints
- Photo point
- Vegetation transects
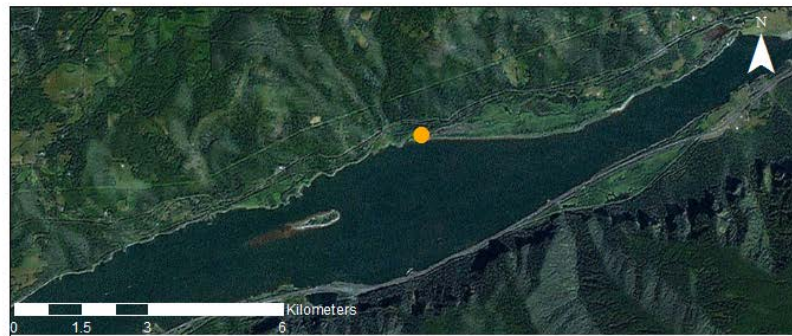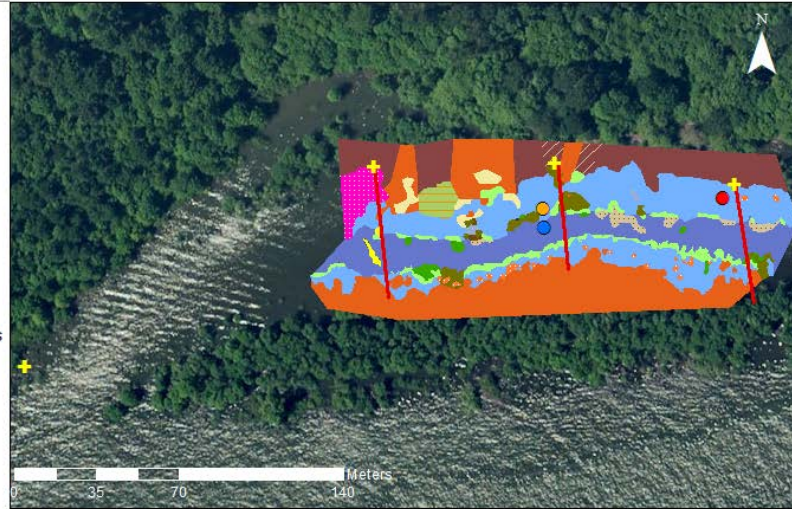
Appendix C. Vegetation Species Cover

Table C.1. **Site elevation (in meters, relative to the Columbia River vertical datum CRD) and vegetation species average percent cover from 2012. The three dominant cover classes are bolded in red for each site and invasive species are shaded in yellow (not necessarily non-native species). Overhanging tree and shrub species are not included in identification of dominant cover. 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	BBM	SRM-L	SRM-H	WI2	WHC	CS1	CLM	GOM	OWR	RI2	FLM
					Elevation (m, CRD)										
				Min	1.69	0.93	1.94	0.89	0.83	1.17	1.04	1.29	0.84	0.86	0.92
				Avg	2.05	1.02	2.06	1.57	1.72	1.68	1.40	1.95	1.35	1.48	1.81
				Max	2.43	1.20	2.17	1.72	2.09	2.66	1.65	2.64	1.85	2.15	2.28
Code	Scientific Name	Common Name	Wetland Status	Native	Average Percent Cover										
AGGI	*Agrostis gigantea*	redtop; black bentgrass	NI	no	0.0	0.0	1.1	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ALTR	*Alisma triviale*	northern water plaintain	OBL	yes	0.0	2.4	0.3	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
AMFR	*Amorpha fruticosa*	indigo bush	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
BESY	*Beckmannia syzigachne*	American sloughgrass	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.5	0.0	0.0
BICE	*Bidens cernua*	Nodding beggars-ticks	FACW+	yes	0.0	1.7	0.3	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
CAAM	*Castilleja ambigua*	paint-brush owl-clover; johnny-nip	FACW+	yes	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CAAT	*Carex athrostachya*	slender-beak sedge	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**8.4**	0.0	0.0	0.0
CACA	*Calamagrostis canadensis*	bluejoint	FACW+	yes	**13.7**	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CAHE	*Callitriche heterophylla*	Twoheaded water starwort	OBL	yes	0.0	2.6	0.0	0.6	0.4	0.0	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	BBM	SRM-L	SRM-H	WI2	WHC	CS1	CLM	GOM	OWR	RI2	FLM
CAHE2	*Callitriche hermaphroditica*	northern water-starwort	OBL	yes	0.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	yes	54.0	5.3	**42.5**	**54.4**	**5.3**	0.0	0.0	0.0	0.0	0.0	0.0
CAOB	*Carex obnupta*	Slough sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**8.4**	**4.9**
CAPA	*Caltha palustris*	Yellow marsh marigold	OBL	yes	0.0	0.0	8.3	5.6	0.1	0.0	0.0	0.0	0.0	0.0	0.0
CASE	*Calystegia sepium*	Hedge bindweed	FAC	no	0.0	0.0	0.8	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0
CASP	*Carex sp.*	Carex	mixed	yes	0.0	0.0	0.0	0.0	0.2	0.4	0.0	0.0	0.3	0.0	0.8
CEDE	*Ceratophyllum demersum*	Coontail	OBL	yes	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
COCO	*Cotula coronopifolia*	common brassbuttons	FACW+	no	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CYST	*Cyperus strigosus*	Strawcolor flatsedge; nutsedge	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0
DECE	*Deschampsia cespitosa*	Tufted hairgrass	FACW	yes	0.9	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
DISP2	*Distichlis spicata*	saltgrass	FACW	yes	**7.8**	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ELAC	*Eleocharis acicularis*	Needle spikerush	OBL	yes	2.2	0.5	0.0	0.0	0.0	0.0	0.0	2.2	3.7	0.1	0.4
ELCA	*Elodea canadensis*	Canada waterweed	OBL	yes	0.0	**33.8**	0.0	0.4	0.0	0.4	1.8	0.0	0.0	0.0	0.0
ELNU	*Elodea nuttallii*	Nuttall's waterweed, western waterweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
ELPA	*Eleocharis palustris*	Common spikerush	OBL	yes	0.0	6.2	0.5	3.6	2.0	**19.3**	**5.5**	**11.1**	**15.9**	**19.9**	4.5
EPCI	*Epilobium ciliatum*	Willow herb	FACW-	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	3.9	0.0	0.0	0.0

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Code	Scientific Name	Common Name	Wetland Status	Native	BBM	SRM-L	SRM-H	WI2	WHC	CS1	CLM	GOM	OWR	RI2	FLM
EQFL	*Equisetum fluviatile*	Water horsetail	OBL	yes	0.0	0.0	3.6	2.1	5.2	0.1	0.0	0.0	0.0	0.0	0.0
EQPA	*Equisetum palustre*	marsh horsetail	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.4	0.3
FRLA	*Fraxinus latifolia*	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
FRLA*	*Fraxinus latifolia*	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	2.8	0.0	0.0
FUDI	*Fucus distichus*	Rockweed	OBL	yes	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GASP	*Galium spp*	Pacific bedstraw; cleavers; small bedstraw	mixed	yes	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0
GATR	*Galium trifidum var. pacificum; Galium trifidum L. spp. columbianum*	Pacific bedstraw	FACW	yes	0.0	0.0	1.8	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GLGR	*Glyceria grandis*	American mannagrass	OBL	yes	1.2	0.0	0.9	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0
GNUL	*Gnaphalium uliginosum*	Marsh cudweed	FAC+	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0
GREB	*Gratiola ebracteata*	bractless hedgehyssop	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0
HEAU	*Helenium autumnale*	common sneezeweed	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.1	0.1
HYAN	*Hypericum anagalloides*	Tinker's penny	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
HYSC	*Hypericum scouleri*	Western St. Johns wort	FAC	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
IMSP	*Impatiens capensis,Impatiens noli-tangere*	Spotted touch-me-not, Common touch-me-not	FACW	yes	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0

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IRPS	*Iris pseudacorus*	Yellow iris	OBL	no	0.0	0.0	0.0	0.6	3.1	0.0	0.0	0.0	0.0	0.0	0.0
JUAR	*Juncus arcticus Wild. ssp. littoralis*	mountain rush	No	yes	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
JUOX	*Juncus oxymeris*	Pointed rush	FACW+	yes	0.0	0.3	0.1	1.4	0.1	0.2	0.0	0.0	0.0	0.1	0.0
LEMI	*Lemna minor*	Duckweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LEOR	*Leersia oryzoides*	Rice cutgrass	OBL	yes	0.0	0.0	0.3	1.1	0.7	0.0	0.0	0.1	0.0	**1.8**	0.9
LIAQ	*Limosella aquatica*	Water mudwort	OBL	yes	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LIOC	*Lilaeopsis occidentalis*	Western lilaeopsis	OBL	yes	2.9	**13.1**	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LISC	*Lilaea scilloides*	Flowering quillwort	OBL	yes	0.0	1.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LOCO	*Lotus corniculatus*	Birdsfoot trefoil	FAC	no	0.0	0.0	0.0	0.8	3.2	0.0	0.0	0.0	0.0	0.0	0.0
LUPA	*Ludwigia palustris*	False loosestrife	OBL	yes	0.0	0.0	0.0	0.5	0.0	0.2	0.4	0.3	0.0	0.0	0.0
LYAM	*Lysichiton americanus*	Skunk cabbage	OBL	yes	0.0	0.0	1.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LYAM2	*Lycopus americanus*	American water horehound	OBL	yes	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LYNU	*Lysimachia nummularia*	Moneywort, Creeping Jenny	FACW	no	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
LYPO	*Lythrum portula*	spatula-leaf loosestrife	NI	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.3	0.0	0.0	0.0
LYSA	*Lythrum salicaria*	Purple loosestrife	FACW+	no	0.0	0.0	2.1	0.3	0.1	0.0	0.0	0.0	0.0	0.0	0.0
MAVE	*Marsilea vestita*	Hairy waterclover	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0
MEAR	*Mentha arvensis*	wild mint	FACW-	yes	0.0	0.0	0.3	1.4	0.0	0.0	0.0	0.0	0.0	0.7	0.0

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MEPU	*Mentha pulegium*	pennyroyal	OBL	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.1	0.0
MIGU	*Mimulus guttatus*	Yellow monkeyflower	OBL	yes	0.0	0.0	0.7	2.5	0.4	0.0	0.0	0.0	0.0	0.0	0.0
MYLA	*Myosotis laxa*	Small forget-me-not	OBL	yes	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0
MYSC	*Myosotis scorpioides*	Common forget-me-not	FACW	no	0.0	0.0	2.9	5.2	2.8	0.0	0.0	0.0	0.0	0.0	0.0
MYSP2	*Myriophyllum spp.*	Milfoil	OBL	mixed	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
MYSP3	*Myriophyllum spicatum*	Eurasian water milfoil	OBL	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
OESA	*Oenanthe sarmentosa*	Water parsley	OBL	yes	0.0	0.0	18.3	6.3	4.2	0.0	0.0	0.0	0.0	0.0	0.0
PHAR	*Phalaris arundinacea*	Reed canary grass	FACW	no	0.0	0.0	19.8	5.9	42.0	15.2	22.5	0.0	3.8	0.0	5.0
PHAR-d		Reed canary grass, dead		#N/A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.3	0.0	0.0
PLDI	*Platanthera dilatata*	white bog orchid	FACW+	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PLMA	*Plantago major*	common plantain	FACU+	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
POAM	*Polygonum amphibium*	water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	25.8
POAN	*Potentilla anserina ssp. Pacifica/Argentina egedii ssp. Egedii*	Pacific silverweed	OBL	yes	3.8	0.0	0.7	1.2	0.0	0.1	0.0	0.0	0.0	0.0	0.2
POAN2	*Poa annua*	annual bluegrass	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.9	0.0	0.0	0.0

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POCR	*Potamogeton crispus*	Curly leaf pondweed	OBL	no	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.2	0.0
POHY	*Polygonum hydropiper, P. hydropiperoides*	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	0.1	0.1	1.9	0.2	0.0	0.0	0.0	0.0	0.0	0.0
PONA	*Potamogeton natans*	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.0	0.0	0.0	0.0
POPE	*Polygonum persicaria*	Spotted ladysthumb	FACW	no	0.0	3.2	0.1	2.2	0.1	0.0	0.4	0.5	0.2	0.4	0.0
POPU	*Potamogeton pusillus*	Small pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0
PORI	*Potamogeton richardsonii*	Richardson's pondweed	OBL	yes	0.0	0.3	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0
PORI2	*Potentilla rivalis*	Brook cinquefoil	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
RARE	*Ranunculus repens*	Creeping buttercup	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
RINA	*Ricciocarpos natans*	Purple fringed liverwort	na	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
ROPA	*Rorippa palustris*	Marsh yellow-cress	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.7
RUMA	*Rumex maritimus*	Golden dock, seaside dock	FACW+	yes	0.0	0.0	0.1	0.2	0.0	0.2	0.0	0.0	0.0	0.0	0.0
SALA	*Sagittaria latifolia*	Wapato	OBL	yes	0.0	0.1	1.1	4.4	**5.7**	**6.1**	**4.8**	0.6	0.7	0.8	2.7
SALU	*Salix lucida*	Pacific willow	FACW+	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0
SALU*	*Salix lucida*	Pacific willow	FACW+	yes	0.0	0.0	0.0	0.0	0.0	2.6	9.7	0.0	2.5	6.9	5.2
SASI*	*Salix sitchensis*	Sitka willow	FACW	yes	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
SASP	*Salix spp.*	Willow	mixed	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	BBM	SRM-L	SRM-H	WI2	WHC	CS1	CLM	GOM	OWR	RI2	FLM
SCAM	*Schoenoplectus americanus*	American bulrush, threesquare bulrush	OBL	yes	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCMA	*Schoenoplectus maritimus*	Seacoast bulrush	OBL	yes	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCTA	*Schoenoplectus tabernaemontani*	Softstem bulrush, tule	OBL	Yes	0.0	**15.0**	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.1
SISU	*Sium suave*	Hemlock waterparsnip	OBL	yes	0.0	0.3	0.8	5.6	0.6	0.0	0.0	0.0	0.0	0.0	0.0
SODU	*Solanum dulcamara*	Bittersweet nightshade	FAC+	no	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
SPAN	*Sparganium angustifolium*	Narrowleaf burreed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0
SPEU	*Sparganium eurycarpum*	giant burreed	OBL	yes	0.0	2.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SYSU	*Symphyotrichum subspicatum*	Douglas aster	FACW	yes	1.5	0.2	0.4	1.7	0.0	0.0	0.0	0.5	0.0	1.2	0.0
TRMA	*Triglochin maritima*	Seaside arrowgrass	OBL	yes	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TRWO	*Trifolium wormskioldii*	Springbank clover	FACW+	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TYAN	*Typha angustifolia*	Narrowleaf cattail	OBL	no	1.2	0.0	0.0	0.0	3.9	0.0	0.0	0.0	0.0	0.0	0.0
VEAM	*Veronica americana*	American speedwell	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
VEAN	*Veronica anagallis-aquatica*	water speedwell	OBL	yes	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
VESP	*Veronica spp.*	speedwell	OBL	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
XAST	*Xanthium strumarium*	rough cocklebur	FAC	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0

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ZAPA	*Zannichellia palustris*	horned pondweed	OBL	yes	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Algae		algae			1.0	0.0	0.0	0.0	0.0	10.8	0.0	2.9	35.3	13.1	3.1
BG		bare ground			4.6	13.8	2.5	4.5	14.9	55.3	61.8	40.0	60.3	63.1	39.5
Detritus		detritus			0.0	0.0	0.5	0.0	0.9	2.1	4.9	0.0	0.6	0.1	3.3
DW		Drift wrack			1.3	1.3	0.0	0.2	4.0	0.0	0.0	0.0	0.0	0.4	0.0
LW		Live wood			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
LWD		Large woody debris			0.0	0.0	1.5	0.0	2.6	0.0	0.8	0.2	0.6	0.2	1.5
Seedlings		Very small herbaceous seedlings (unidentifiable)			0.0	0.0	0.0	0.0	0.0	0.7	0.0	6.5	5.6	0.2	4.3
Litter		Litter			0.0	0.0	0.0	0.0	0.0	0.4	1.8	2.6	1.9	0.8	4.3
Moss		Moss			0.0	0.8	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0
OW		Open water			4.3	24.5	0.0	0.2	3.2	27.5	1.2	4.4	27.8	26.0	21.8
UID Grass 1		UID grass			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0