Ecosystem Monitoring Program

Annual Report for Year 9

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Lower Columbia River Ecosystem Monitoring Program
Annual Report for Year 9 (October 1, 2012 to September 30, 2013)

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1 Executive Summary

Introduction

The Ecosystem Monitoring Program (EMP) is managed by the Lower Columbia Estuary Partnership (Estuary Partnership) and is an integrated status and trends program for the lower Columbia River. The study area extends from the mouth of the estuary to the Bonneville Dam. The program is designed to provide an inventory of the different types of habitats within the lower river, track trends in the overall condition of these habitats, provide a suite of reference sites for use as end points in the region's habitat restoration actions, and place findings from management actions 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 throughout the river and to evaluate the trends and impacts of management actions over time. When the EMP was created in 2004, most previous research had occurred in a small section of the lower river, close to the river mouth in Reaches A and B. There was a considerable lack of research and monitoring within the tidal freshwater section (Reaches C-H), resulting in little basic understanding of habitats, fish use and food web dynamics in this region. 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 throughout 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

The Estuary Partnership and its monitoring partners have focused on providing an inventory of salmon habitats (or "status") across the lower river stratifying by hydrogeomorphic reach (A–H) and including a growing number of fixed sites for interannual variability (or "trends"). Each year prior to 2013, three to four "status" sites, in a previously unsampled reach, were selected along with the continued sampling of a growing number of "trend sites." In 2013, the EMP partnership focused on the six "trend" sites only.

In 2013, 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 and salmon metrics - occurrence, composition, growth, diet, condition and residency), 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 active radiation, and isotope ratios; by USGS and OHSU); secondary productivity (zooplankton abundance and taxonomy; by OHSU) and mainstem abiotic conditions (nitrates, conductivity, temperature, dissolved oxygen, turbidity, chlorophyll *a* 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 2013, the EMP partnership monitored the six "trend" sites: Ilwaco in Reach A (sampled 2011-2013), Secret River in Reach B (sampled in 2012 and 2013), Welch Island in Reach B (sampled in 2012 and 2013), Whites Island site in Reach C (sampled in 2009-2013), Campbell Slough in Reach F (sampled from 2007-2013), Franz Lake Slough site in Reach H (sampled in 2008-2013), in order to examine temporal trends. In 2013, the Center for Coastal Margin Observation and Prediction (CMOP) at OHSU continued to monitor mainstem conditions via an *in situ* water quality monitoring platform (also monitored in 2012) to provide biogeochemistry of the tidal freshwater region of the lower river for comparison to the "trend" sites as well as biogeochemistry of the estuary, key for assessing hypoxia, ocean acidification and climate change impacts. 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).

2013 Summary of Results:

Mainstem abiotic conditions

- Biochemical measurements in the mainstem Columbia River at RM-122 provided valuable data upstream of the Willamette River confluence and when compared to conditions observed at the RM-53 platform, enhanced our understanding of the change in conditions within the lower river and estuary.
- Water temperature follows seasonal climate conditions (likely as a result of upstream processes) rather than being influenced by local effects or tributaries. Water temperatures greater than 19°C (threshold for suitable salmon habitat) occurred during more days in 2013 than in the previous two monitoring years.
- Fluxes and concentrations of colored dissolved organic matter (CDOM), nitrate, and turbidity changed significantly upstream to downstream of the Willamette River confluence and were observed to increase during winter storms at RM-53 but showed little or no response at RM-122.
- Chlorophyll *a* concentrations > 40 μg/L were measured during the 2013 spring bloom (March - May) and occurred during a period of relatively low rainfall. The seasonal flux of chlorophyll *a* was similar between RM-122 and RM-53, likely due to the difference in material that is derived in the watershed and is entering the river during runoff versus material that grows in the river. The small increase in flux between RM-122 and RM-53 may be best explained as the amount added by *in situ* growth between the two sites.
- Summer biomass of phytoplankton was typically lower than the preceding spring bloom and similar to 2012 biomass; phosphorus limitation was identified as a possible limiting factor for phytoplankton growth.

Off-channel abiotic conditions

- Similar to previous study years, Whites Island had the most suitable water quality conditions for juvenile salmonids during the 2013 monitoring period.
- A high frequency of hourly measurements exceeding Washington State thresholds was observed at Ilwaco; however, frequent flushing allowed for conditions to return to suitable ranges quicker than sites located farther upstream.
- Campbell Slough data show greater year-to-year variability due a longer period of monitoring (data have been collected since 2009) compared to other monitoring sites. Given that 2011 and 2012 were cooler years with higher than average water levels in the Columbia River, long-term data sets are valuable for exhibiting annual variability in the system.
- Generally, warmer water temperatures, lower dissolved oxygen concentrations, and higher maximum pH values were observed in 2013 compared to 2011 and 2012 at all sites where data were collected.
- Ilwaco (marine-influenced) nutrient concentrations were higher than at the freshwater-influenced sites located farther upstream.

Habitat Structure and Hydrology

Sediment

- Annual sediment accretion rates ranged from -1.7 (erosion) to 3.0 cm/year in 2013, with most values falling between 0.0 and 1.4 cm/year.
- Sediment accretion rates at the trend sites are variable between years. Cunningham Lake showed the lowest inter-annual variability (standard deviation [SD] ±0.26) and Franz Lake showed the highest inter-annual variability (SD ±1.74), as well as the highest accretion rates across all trend sites.

Hydrology

- Hydrologic patterns are highly variable throughout the estuary and affect vegetation assemblages, cover, and biomass production.
- Inter-annual variation in inundation patterns is much greater at sites located farther upstream where seasonal flooding can result in months of inundation during years of higher water flow. At the lower, tidally dominated sites, inundation occurs frequently, but for a short duration.

Vegetation Assemblages

- Native vegetative cover was higher than non-native cover at the three lower river sites (Iwaco and Secret River High and Low), with native species cover dominated by Lyngby's sedge (*Carex lyngbyei*).
- At all the sites where reed canarygrass (*Phalaris arundinacea)* occurred, an increase in cover of the species was noted between 2012 and 2013, particularly at Secret River and Whites Island.

- A higher number of species was observed at the Campbell Slough site in 2013 than in previous years, potentially due to local disturbances in recent years.

Elevation, Inundation and Vegetation Interactions

- Reed canarygrass has a large range of inundation tolerance and may withstand high seasonal inundation, although seems to not be tolerant of high tidal inundation.
- Inundation (as expressed by sum exceedance value, SEV) ranges at sites where reed canarygrass thrives ranged from 250 at Secret River to 2800 at Franz Lake, and it is suspected that the species is most competitive at the middle of the inundation range.

Food Web

Phytoplankton

- Total phytoplankton abundance inversely tracked the magnitude of Columbia River flow, and data suggest differences in the magnitude and composition of planktonic assemblages upstream versus downstream of the Willamette-Columbia confluence. However, the different habitat characteristics of the monitoring sites at the reach scale may somewhat obscure comparisons.
- Prior to the spring freshet, diatoms dominated the assemblages at all sites in the lower Columbia, and during low river flow periods other species including green algae (Class Chlorophyceae) and cyanobacteria (Class Cyanophyceae) increased in relative abundance.
- Plankton communities at Campbell Slough exhibited distinct temporal patterns in total and relative abundance that were associated with the higher nutrients and slower flushing. In the summer, elevated abundances of cyanobacteria (including harmful species capable of producing heptotoxins) were observed.
- Isotopic signatures of carbon and nitrogen in the dominant salmon prey (Diptera and Amphipods) suggested that phytoplankton made up an important food source at certain times of the year, particularly during periods of high biomass (i.e., spring “blooms”). The degree to which phytoplankton populations influence prey productivity remains unclear; however, given that salmon prey isotope signatures track those of phytoplankton suggests that there is a relationship that should be further explored.

Zooplankton

- Copepods dominated zooplankton assemblages at Ilwaco throughout the year. Other sites located farther upstream were more variable throughout the monitoring period with rotifers occasionally present in high abundance (particularly early in the season) and crustacean zooplankton (copepods and cladocerans) generally dominating assemblages later in the season.
- The highest zooplankton abundances were observed at Campbell Slough during the summer.

Macroinvertebrates

- In 2013, aquatic Diptera and Amphipods contributed the majority of overall prey consumed by fish in terms of biomass and numbers.
- Additional fish prey taxa that are energetically important include Trichoptera, Hemiptera (primarily aquatic but some terrestrial taxa as well), Cladocera, and Oligochaeta.

- A consistent difference between the high densities of invertebrates in the emergent vegetation habitat compared to the low densities in the open water habitat was observed during the 2013 surveys. Diptera abundances were related to the percent of live vegetation cover within emergent vegetation transects.

Fish

- 2013 data were consistent with previous years of monitoring that show tidal freshwater habitats in the LCRE as important for migration, feeding, and rearing for several species of salmonids.
- Fish assemblages in Reach A (Ilwaco Slough) include saltwater species not present at sampling sites elsewhere. In Reaches B and C, fish assemblages were similar with high proportions of stickleback and smaller numbers of mainly native species. At Campbell Slough and Franz Lake (Reaches F and H) higher proportions of non-native species were observed as well as potential predators of juvenile salmonids.
- Fish species diversity and richness was highest at Campbell Slough and Franz Lake than at the other sites located farther downstream, similar to previous monitoring years.
- Chum salmon dominated the salmon catch at Ilwaco Slough, while Chinook and coho salmon were observed in only in small numbers.
- Similar to previous monitoring years, Chinook salmon were the dominant species at Secret River, Welch Island, Whites Island and Campbell Slough. Small numbers of chum salmon were observed at Secret River and Welch Island as well as small numbers of coho salmon at Secret River. Small numbers of Chinook and coho salmon were observed at Franz Lake, although this site was sampled for only a limited portion of the season.
- Similar to previous monitoring years, unmarked fish were predominate in the lower reaches of the LCRE, while larger percentages of marked fish were observed in upstream reaches in 2013. Percentages of marked Chinook salmon were low at Secret River and gradually increased upstream to Campbell Slough, where 42% of Chinook salmon were marked. Coho salmon collected at Welch Island were unmarked. However, at Franz Lake only unmarked fish were found in 2013 which was different than previous years where both unmarked and marked coho and Chinook salmon were observed and may be related to sampling limitations in May and June during the peak hatchery release period.
- Unmarked Chinook salmon density was highest at Welch Island, and lowest at Ilwaco Slough and Franz Lake, with intermediate values at the other sites. Coho and chum salmon densities were low at most sites in 2013.
- Unmarked juvenile Chinook salmon were typically present from February through June or July, with the largest number of fish present in May and June. Marked Chinook salmon were found over the same time period, with highest densities observed in May and June, consistent with the period of hatchery releases. Chum salmon were present from February through April, while unmarked coho salmon were found in small numbers from February through December. Marked coho salmon were nearly all collected in May, which again is consistent with hatchery release timing. Almost no salmon were present in August or September.

- Secret River and Welch Island had the highest proportions of juvenile Chinook salmon fry, and fingerlings were most abundant at Campbell Slough and Franz Lake.
- As in past years, clear patterns in condition factor by site were not apparent, although some indications of increased condition factor of unmarked Chinook salmon at Secret River and Welch Island were noted in 2013. Limited sampling at these two sites so far (2012 and 2013 only), precludes solid interpretation of this finding.
- 2013 Chinook salmon stock data are not yet available, but 2012 data (which were not included in last year's report) show spatial variation along the river in Chinook salmon stock usage. Unmarked West Cascades fall Chinook are more abundant at the sites in Reaches A through C, with Upper Columbia summer/fall Chinook being the most prevalent stock at Campbell Slough in Reach F and Lemon Island in Reach G.

Acknowledgements

This study could not have been completed without the help of our partners. We are grateful to the Northwest Power and Conservation Council and the Bonneville Power Administration for funding through the Columbia Basin Fish and Wildlife Program. Whitney Temple and Jennifer Morace could not be listed as co-authors since at the time of this draft it had not gone through USGS peer-review. USGS collected and analyzed abiotic conditions at four of the trend sites and portions of the food web, including the stable isotope analysis. We thank them immensely for their collaborative work on this program. We also thank the land owners and managers who have allowed us to conduct research on lands they manage, including: Alex Chmielewski (Ridgefield National Wildlife Refuge) and Jim Clapp (Franz Lake National Wildlife Refuge). Finally, the Estuary Partnership's Science Work Group provided valuable input throughout the process and peer review on final drafts. The Science Work Group is composed of over 60 members, and is integral in ensuring the Estuary Partnership represents the best available science.

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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 Ecosystem Monitoring Program (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 in the Columbia River basin 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, provide a suite of reference sites for use as end points in the region's habitat restoration actions, and place findings from the other research and monitoring efforts 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).

With funding limitations, the Estuary Partnership and its monitoring partners have focused on providing an inventory of salmon habitats (or "status") across the lower river and including a growing number of fixed sites for interannual variability (or "trends"). The focus of the EMP has been on minimally disturbed tidally influenced emergent wetland sites. Between 2005 and 2012, three to four "status" sites, in a previously unsampled river reach, were selected along with the continued sampling of a growing number of "trend sites."

In 2012, the EMP scope was reduced to monitoring only the 6 trend sites: Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2013), Whites Island (2009-2013), and Franz Lake (2008-2009, 2011-2013), Ilwaco (2010-2013), Secret River (2010-2013), and Welch Island (2010-2013). Habitat structure and hydrology data began to be collected in 2005, fish data collection began in 2007, fish prey data collection began in 2008, and water quality data and food web data collection began in 2010. Data collection includes:

- Salmonid occurrence, composition, growth, diet, condition and residency
- Habitat structure, including physical, biological and chemical properties of habitats
- Food web characteristics, including primary and secondary productivity of habitats and in the mainstem lower river and
- Biogeochemistry of tidal freshwater region of the lower river for comparison to the biogeochemistry of the estuary, key for assessing hypoxia, ocean acidification and climate change impacts.

2.2 Activities Performed, Year 9 Contract (October 1, 2012-September 30, 2013)

Funding for the EMP by the NPCC/BPA supports the Estuary Partnership's Research Scientists. As part of 2012-2013 EMP tasks, funding supported the following:

- Coordinated development of the synthesis report for the Columbia River Estuary Ecosystem Classification.
- Facilitated discussions and development of study design for 2013-2014 monitoring efforts.
- 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, 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, OHSU 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 www.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 2013-2014 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 2013-2014.

Funds for monitoring contract also support the Research Scientist's work on the Estuary Partnership's Action Effectiveness Monitoring and Research (AEMR) program funded by BPA. For this program, the Research Scientist:

- Coordinated field data collection at four restoration sites and four reference sites
- Assisted with preparation of Level 3 AEMR Workshop
- Developed guidelines for project sponsors for designing Level 3 AEMR Plan
- Coordinated efforts to prioritize and identify two new restoration sites and two reference sites for 2013-2014 AEMR efforts
- Managed AEMR subcontracts with CREST and PNNL for 2012-2013
- Compiled AEMR data and field reports from subcontractors for the Restoration Program's 2012-2013 annual report to BPA
- Developed scopes of work for AEMR subcontractors for 2013-2014
- Coordinated with project sponsors to collect extensive Level 3 AEMR data related to the programmatic AEMR plan

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

- Coordination and communication among interested parties by staying abreast of research, monitoring and evaluation (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, AEMR, and Integrated Status and Trends Monitoring.

- Reviewed Data Reduction Protocols and Data Reduction Workbooks for the Oncor Database program

Funding for the EMP also provides partial support for the Estuary Partnership's GIS/Data Management Specialist. For the 2012-2013 EMP efforts, the GIS/Data Management Specialist participated in continued development of the Columbia River Estuary Ecosystem Classification as follows:

- Assisted UW and USGW in revising Level 4 (Ecosystem Complex) and Level 5 (Geomorphic Catena) map layers.
- Assisted with development of final synthesis report (in -progress), which will supersede the USGS Open File Report. Activities include writing, editing, meeting coordination, and field visits.

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

- Participated in development of Oncor database, with BPA, USACOE, PNNL, Sitka Technologies, PNAMP
- Provided field support for PNNL sampling crew during the 2013 field season
- Provided field support for NOAA sampling crew during the 2013 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 (NEP), established in Section 320 of the Clean Water Act. The EMP's study area encompasses that of the NEP (i.e., the Estuary Partnership) and includes all tidally influenced waters, 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, in tidally influenced shallow water emergent wetlands connected to the Columbia River.

The Estuary Partnership and monitoring partners use a multi-scaled stratification sampling design for the emergent wetland component of the EMP based on the Columbia River Estuary Ecosystem Classification (Classification). The 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 ecosystem attributes. The Classification 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. The EMP sampling design has been organized according to Level 3 of the Classification, which divides the LCRE into eight major hydrogeomorphic transitions (Figure 1). Previous habitat monitoring efforts for the EMP have concentrated on a different reach each year since 2004 for "status" monitoring, with a growing number of "trend" sites that are sampled every year (Table 1).

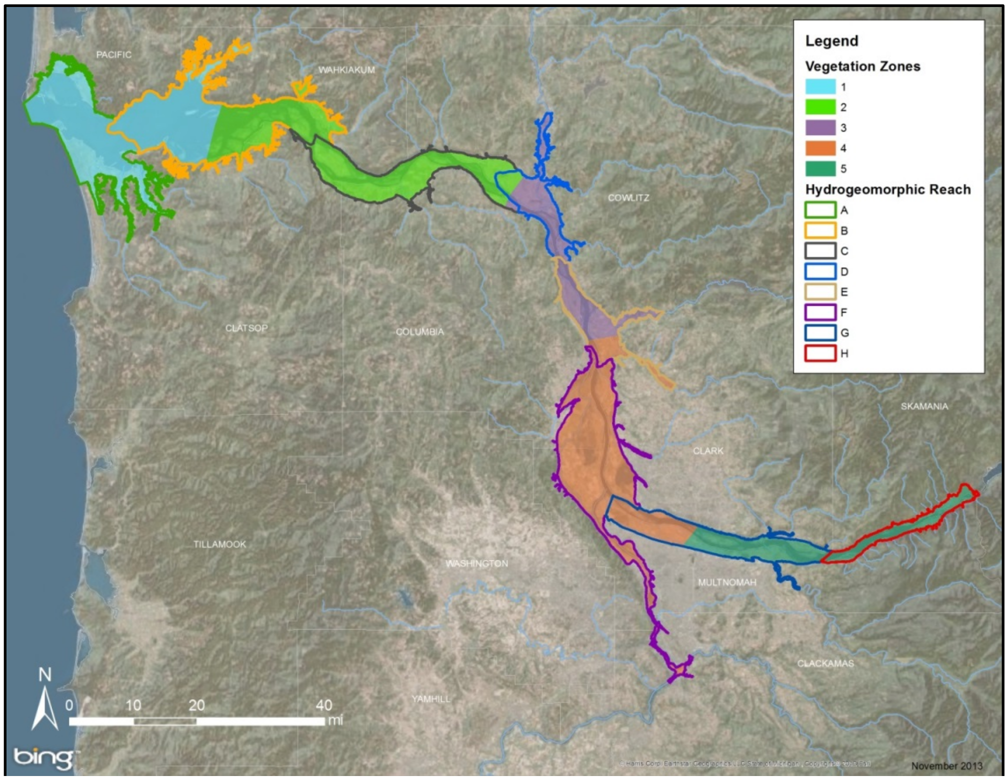


Figure 1. Lower Columbia River and estuary (LCRE) with hydrogeomorphic reaches (A-H) outlined and vegetation zones (1-5) specified by color.

2.4 Characterization of Emergent Wetlands in the LCRE

2.4.1 Sampling Effort, 2005-2013

The objective of the EMP is to characterize habitat structure and function of estuarine and tidal freshwater habitats and monitor salmon occurrence and health in those habitats in the LCRE. Because of limited funding, the EMP has largely concentrated on characterizing relatively undisturbed emergent wetlands that provide important rearing habitat for juvenile salmonids, and can serve as reference sites for restoration actions. Since 2007, we have co-located monitoring of habitat structure, fish, fish prey, and basic water quality metrics 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 2013, site is represented in bold.

Reach	Type of Site	Site	Site Code	Vegetation & Habitat	Fish &Prey	Abiotic Conditions	Food Web[1]	Mainstem abiotic conditions
A	Trend	**Ilwaco**	BBM	2011-2013	2011-2013	2011-2013	2011-2013	
B	Trend	**Secret River**	SRM	2008[2], 2012, 2013	2012, 2013		2012, 2013	
	Trend	**Welch Island**	WI2	2012, 2013	2012, 2013		2012-2013	
C	Status	Ryan Island	RIM	2009	2009			
	Status	Lord-Walker Island 1	LI1	2009	2009			
	Status	Lord-Walker Island 2[3]	LI2	2009				
	Trend	**Whites Island**	WHC	2009-2013	2009-2013	2009, 2011-2013	2011-2013	
	Status	Jackson Island	JIC	2010	2010			
	Status	Wallace Island	WIC	2010	2010			
	Status	Bradwood Landing	BSM	No access	2010			
D	Status	Cottonwood Island small slough	CI2	2005				
	Status	Cottonwood Island large slough	CI1	2005				
	Status	Dibble Slough	DSC	2005		2005		
E	Status	Sandy Island 1, 2	SI1 SI2	2007	2007			
	Status	Lewis River Mouth	NNI	2007				
	Status	Martin Island	MIM	2007				
F	Status	Sauvie Cove	SSC	2005				
	Status	Hogan Ranch	HR	2005				
	Status	Goat Island	GIC	2011	2011			
	Status	Deer Island	DIC	2011	2011			
	Status	Burke Island	BIM	2011	2011			
	Trend	**Cunningham Lake**	CLM	2005-2013	2007-2009			
	Trend	**Campbell Slough**	CS1	2005-2013	2007-2013	2008-2013	2010-2013[4]	
G	Status	Water Resources Center	WRC	2006				
	Status	McGuire Island	MIC	2006				
	Status	Old Channel Sandy River	OSR	2006			2006	
	Status	Chattam Island	CIC	2006				
	Status	Government/Lemon Island	GOM	2012	2012	2012		
	Status	Reed Island	RI2	2012	2012	2012		
	Status	Washougal Wetland	OWR	2012	2012	2012		
	Trend	**RM122**	-					2012,

								2013
H	Trend	**Franz Lake (slough)**	FLM	2008-2009, 2011-2013	2008-2009, 2011-2013	2011-2013	2011-2013	
	Status	Sand Island	SIM	2008	2008	2008		
	Status	Beacon Rock		2008	2008			
	Status	Hardy Slough	HC	2008	2008			

[1] In Reach B no sampling was conducted by USGS.
[2] Site sampled as part of the Reference Site Study; thus, only vegetation and habitat data were collected.
[3] 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.
[4] Phytoplankton and zooplankton only sampled from 2001 – 2013.

2.4.2 Site Descriptions

In 2013, the EMP focused solely on the “trend” sites that have been monitored for multiple years. The 2013 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.

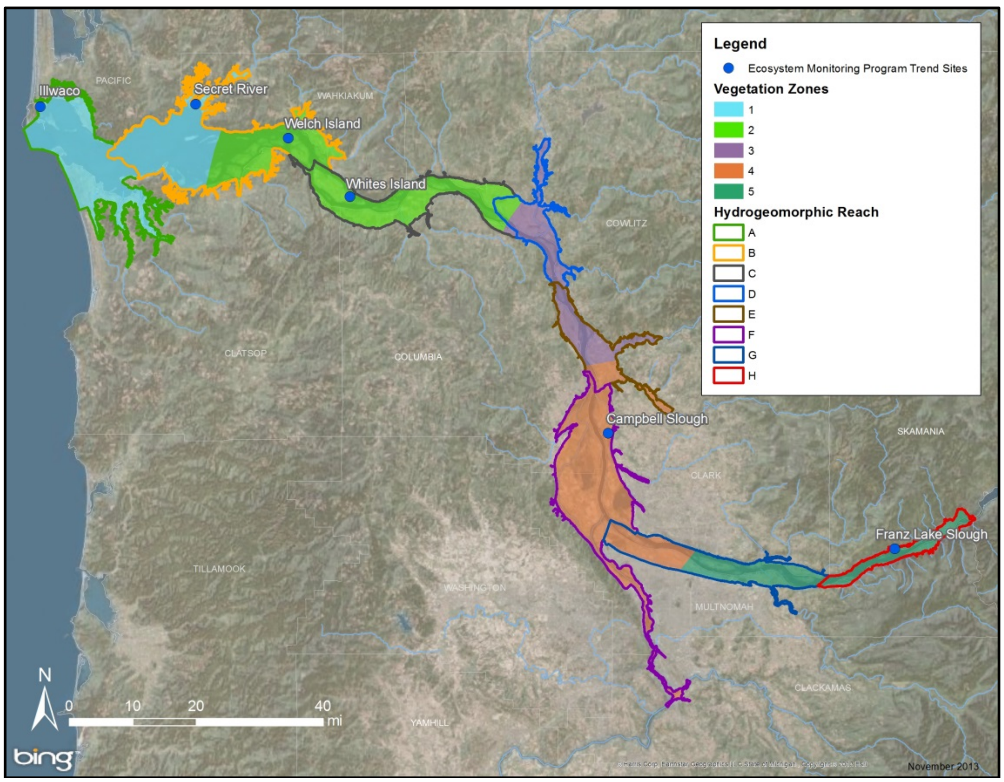


Figure 2. Map of the 2013 monitoring trend sites with the hydrogeomorphic and hydro-vegetation zones (hydro-vegetation zones as described in Borde et al. 2012)

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*) (Figure 3a). Being so close to the mouth of the Columbia River, the tidal channel is regularly inundated with brackish water (salinity < 10 Practical Salinity Units, PSU).

Secret River. The Secret River marsh, located in Grays Bay at the mouth of Secret River 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). Although the marsh was present on the historical maps from the late 1880's, 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, 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" (Figure 3) 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. 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 canarygrass 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.

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) 
e) 
f) 

g) 

Figure 3. 2013 Ecosystem Monitoring sites: (a) Ilwaco; (b) Secret River; (c) Welch Island; (d)Whites Island, Cut-Off Slough; (e) Cunningham Lake; (f) Campbell Slough; (g) Franz Lake.

2.4.3 Water Year

The 2013 water year was somewhat similar to the 29-year daily average, characterized by higher than average water surface elevation (WSE) in the winter (December – January) followed by a period of below average elevations in February and March (Figure 4). The spring freshet occurred earlier than the 29-year average with peak water levels occurring in April, May, and late June. Summer and fall WSE was similar to the long term average. Hydrographs for specific monitoring sites in 2013 are provided in Appendix A.

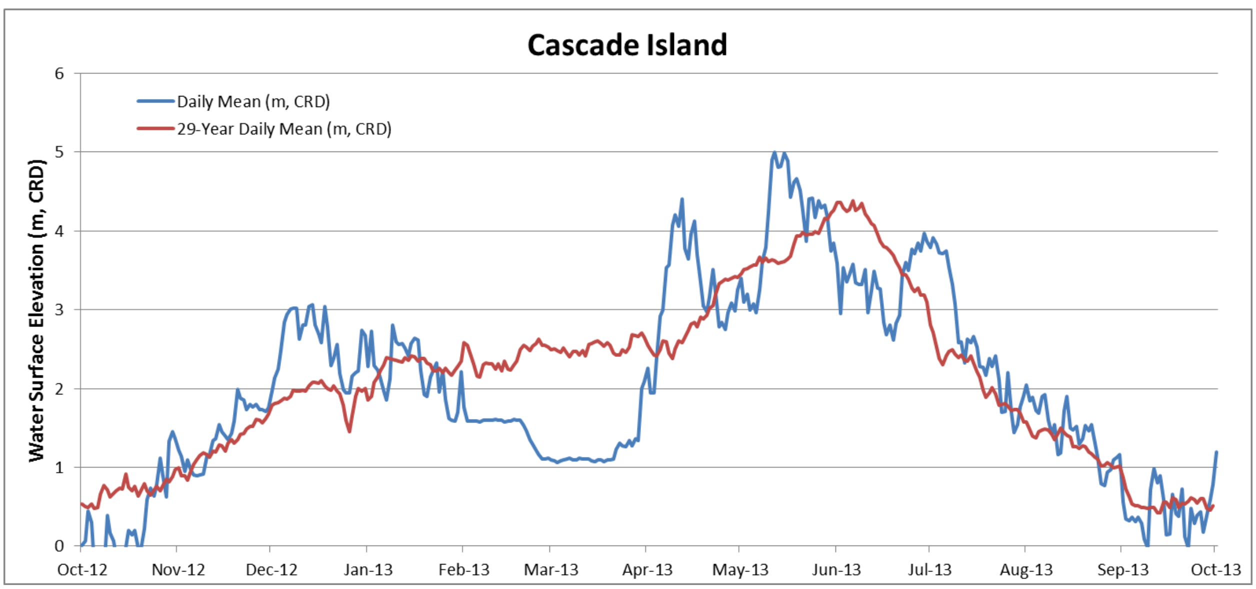


Figure 4. Water surface elevation at Cascade Island, just below Bonneville Dam (rkm 233), from October 2012 to October 2013 compared to the 29-year daily mean water surface elevation (Data from USGS National Water Information System at: http://waterdata.usgs.gov/nwis/).

3 Methods

3.1 Abiotic site conditions

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

In 2013, USGS monitored water quality and assessed food-web resources at four of the trend sites: Franz Lake (slough), Campbell Slough, Whites Island, and Ilwaco (Table 2). USGS has deployed a continuous water-quality monitor at Campbell Slough in the Roth Unit of the Ridgefield National Wildlife Refuge for six years (2008-2013). This site in Reach F has been sampled for vegetation since 2005 (PNNL) and for fish since 2007 (NOAA Fisheries). USGS has also deployed a continuous water-quality 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 2011-2013 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 during 2011-2013 and for vegetation (PNNL) and fish (NOAA Fisheries) in 2008-2009 and 2011-2013. 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 during 2011-2013.

Table 2. Site information for locations of water-quality monitors in 2013. *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”	April 2	June 30
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47’ 05”	-122° 45’ 15”	April 2	August 7
Whites Island	460939123201600	Birnie Slough, White’s Island, Columbia River, WA	C	46° 09’ 39”	-123° 20’ 16”	April 4	June 30
Ilwaco	461802124024400	Columbia R. at Port of Ilwaco Marina at Ilwaco, WA	A	46° 18' 02”	-124° 02' 43"	April 3	June 31

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 3 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 2013, the monitors were deployed during the first week of April through the last week of July, with visits every 2–4 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 3. 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.

3.1.2 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 filtered prior to processing. Nitrogen and phosphorus species were analyzed during six sampling events in 2013. Nitrate+nitrite, orthophosphate and total Kjeldahl nitrogen were determined according to EPA standard methods (EPA 1983a), ammonium was determined colorimetrically (APHA 1998), and total phosphorus was determined according to USGS (1989).

3.2 Habitat Structure

In 2013, PNNL collected field data on vegetation and habitat conditions at the seven trend sites (Figure 2). Details regarding these sites and other sites visited in 2012 are provided in Table 4. Detailed maps of the 2013 monitoring sites are presented in Appendix B.

Table 4. Site location and sampling dates for each site visited in 2013. "Full sampling" means we collected the entire suite of measurements outlined in Sections 1.3 and 2.1.

Site Name	Site Code	River kilometer (rkm)	Site Type	Sampling Date	Monitoring Activity
Ilwaco/Baker Bay	BBM	6	Trend	7/26	Full sampling
Secret River (low marsh)	SRM-L	37	Trend	7/25	Full sampling
Secret River (high marsh)	SRM-H	37	Trend	7/24	Full sampling
Welch Island 2	WI2	53	Trend	7/23	Full sampling
Whites Island	WHC	72	Trend	7/22	Full sampling
Cunningham Lake	CLM	145	Trend	7/29	Full sampling
Campbell Slough	CS1	149	Trend	7/27	Full sampling
Government/Lemon Island	GOM	181	Status	10/9	Sediment Stakes
Old Washougal River	OWR	195	Status	9/13	Sediment Stakes
Reed Island 2	RI2	204	Status	9/19	Sediment Stakes
Franz Lake	FLM	221	Trend	7/31	Full sampling

3.2.1 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 though specific data to do so is limited in the LCRE. 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 (released to the river in the form of 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).

3.2.2 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.

3.2.2.1 *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). In 2012, sedimentation stakes were installed at the Secret River, Welch Island, Government/Lemon Island, Washougal River, and Reed sites this year, and were measured again in 2013. All previously installed sediment accretion stakes at the trend sites were measured in 2013. The accretion or erosion rate is calculated by averaging the 11 measurements from each year and comparing the difference.

3.2.2.2 *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. The sensors have been downloaded and redeployed every year since 2010 for collection of a continuous dataset.

3.2.2.3 *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 and in February and August of 2013.

3.2.2.4 *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.

Emergent marsh (EM) zone delineation occurred in previous years as part of this and other studies (Jay et al. in review; Sagar et al. 2013; Borde et al. 2012) and is used here to evaluate vegetation patterns within the tidal wetlands of the LCRE. The zones boundaries are meant to be broad, with variation of the zone boundaries observed between years, however the following river kilometers are used to delineate the zones currently:

EM Zone	River Kilometer (rkm)
1	0 - 40
2	41 - 88
3	89 - 136
4	137 - 181
5	182 - 235

3.2.2.5 *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.

3.2.2.6 *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.

3.2.2.7 *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.

3.2.2.8 *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:

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

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.

3.2.2.9 *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).

3.3 Food web

3.3.1 Primary Productivity

3.3.1.1 *Phytoplankton/periphyton*

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^{o}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 5. 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 and 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 5. List of samples analyzed (X's) from six trend sites in the Lower Columbia River in 2013.

Site	Reach	Sample collection date	Zooplankton	Phytoplankton
ILWACO MARINA	A	4/18/13		X
	A	5/2/13	X	X
	A	5/16/13	X	
	A	5/30/13	X	
	A	6/27/13	X	
	A	7/11/13	X	
WELCH ISLAND	B	5/6/13	X	
	B	6/3/13	X	
	B	7/1/13	X	
SECRET RIVER	B	5/6/13	X	
	B	6/3/13	X	
	B	7/1/13	X	
WHITES ISLAND	C	4/17/13	X	X
	C	5/1/13	X	X
	C	5/15/13	X	X
	C	5/29/13	X	X
	C	6/26/13	X	X
	C	7/10/13	X	X
CAMPBELL SLOUGH	F	5/1/13	X	X
	F	5/13/13	X	X
	F	5/29/13	X	X
	F	6/24/13	X	X
	F	7/10/13	X	X
FRANZ LAKE SLOUGH	H	4/16/13	X	
	H	4/30/13	X	X
	H	5/14/13	X	X
	H	5/28/13	X	X
	H	6/25/13	X	X
	H	7/9/13	X	X

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 2011-2013 in the following experiments.

Phytoplankton Productivity

In 2013, phytoplankton productivity was estimated using the ^{14}C uptake approach described in previous annual reports, except the stable isotope ^{13}C as a tracer in place of ^{14}C (Hama et al. 1983; Sagar et al. 2013). 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.

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.

3.3.1.2 ***Emergent Vegetation***

Field Methods
From 2011 to 2013 above ground biomass was sampled to estimate the primary productivity at the six trend sites. Samples were collected in the summer during July or August during peak biomass and again in February during the winter low biomass period. 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 and 2013, 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 strata. Within the 1-m square biomass plot, a 0.1 m2 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.

Lab Methods
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, obvious root material was removed, and for the winter samples live and dead material was seperated. 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.

3.3.2 Secondary Productivity

3.3.2.1 *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 at four EMP sites (Ilwaco, Whites Island, Campbell Slough, and Franz Lake Slough). At Secret River and Welch Island (Reach B sites), the zooplankton samples were collected using a smaller net with a 60 μm mesh (0.3 m wide x 1 m long). A list of the collection sites and dates they were sampled is given in Table 5.

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:

$$Distance = \frac{Difference\ in\ counts \times Rotor\ Constant}{999999} \tag{1}$$

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:

$$Speed = \frac{Distance\ in\ meters \times 100}{Time\ in\ seconds} \tag{2}$$

The volume is determined as:

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

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.

Taxonomy

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

3.3.2.2 *Macroinvertebrate Prey Availability*

Open water and emergent vegetation

We quantified the density, diversity and size of invertebrate prey available to juvenile Chinook salmon across 18 sites between 2008 and 2013, focusing on months during which salmon were abundant (generally when n>5). To assess prey availability across the range of habitats sampled 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 collections), but occasionally one or three of each were collected depending on field conditions (Table 6).

Table 6. The number of invertebrate tow samples collected at each site per sampling event between 2008-2013.

	2008			2009		2010				2011			2012					2013				
	April	May	June	May	June	April	May	June	July	April	May	June	February	March	April	May	June	March	May	June	July	total tow samples
Ilwaco	0	0	0	0	0	0	0	0	0	2	8	4	0	0	0	0	0	0	0	0	0	14
Secret River	0	0	0	0	0	0	0	0	0	0	0	0	4	0	4	1	6	0	4	4	0	23
Welch Island	0	0	0	0	0	0	0	0	0	0	0	0	0	2	5	4	4	4	4	4	4	31
Ryan Island	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Bradwood Slough	0	0	0	0	0	4	4	4	4	0	0	0	0	0	0	0	0	0	0	0	0	16
Jackson Island	0	0	0	0	0	4	4	4	0	0	0	0	0	0	0	0	0	0	0	0	0	12
Whites Island	0	0	0	4	0	4	4	4	4	0	10	4	0	2	6	4	4	0	4	3	6	63
Wallace Island	0	0	0	0	0	4	4	4	4	0	0	0	0	0	0	0	0	0	0	0	0	16
Lord/Walker Island	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Burke Island	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4
Goat Island	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4
Deer Island	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	4
Campbell Slough	3	6	0	5	4	4	4	4	2	0	4	0	0	0	0	4	4	0	4	4	0	52
Lemon Island	0	0	0	0	0	0	0	0	0	0	0	0	0	3	4	4	2	0	0	0	0	13
Washougal	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	4	4	0	0	0	0	10
Sand Island	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Franz Lake	6	6	0	4	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	18
Hardy Slough	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
total tow samples	15	12	4	20	4	20	20	20	14	2	36	8	4	7	21	21	24	4	16	15	10	297

Taxonomists at Rhithron and Associates (Missoula, MT) processed all invertebrate samples. For tow samples, processing included sorting, identifying, counting, measuring and, for 2012 and 2013 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. For samples collected in 2012 and 2013, 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 most 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}=60) 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 (see section 3.3.3.1) collected concurrently.

3.3.3 Salmon Diet

3.3.3.1 *Chinook salmon diets and prey selectivity*

Chinook salmon were usually killed within an hour of collection (Table 7). 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 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.

Table 7. The number of juvenile Chinook diet samples collected at each site per sampling event between 2008 and 2013.

	2008			2009		2010					2011			2012					2013				
	April	May	June	May	June	April	May	June	July	August	May	June	July	February	March	April	May	June	March	May	June	July	total diet samples
Secret River	0	0	0	0	0	0	0	0	0	0	0	0	0	15	0	15	0	14	0	12	1	2	59
Welch Island	0	0	0	0	0	0	0	0	0	0	0	0	0	16	14	14	30	15	9	30	23	25	176
Ryan Island	0	0	0	9	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	19
Bradwood Slough	0	0	0	0	0	10	17	9	10	8	0	0	0	0	0	0	0	0	0	0	0	0	54
Jackson Island	0	0	0	0	0	19	15	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	42
Whites Island	0	0	0	10	0	16	14	18	19	13	10	25	2	0	13	10	11	15	0	15	13	0	204
Wallace Island	0	0	0	0	0	6	14	11	11	0	0	0	0	0	0	0	0	0	0	0	0	0	42
Lord/Walker Island	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
Burke Island	0	0	0	0	0	0	0	0	0	0	10	0	2	0	0	0	0	0	0	0	0	0	12
Goat Island	0	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	13
Deer Island	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	10
Campbell Slough	6	19	0	10	9	12	24	18	15	0	22	0	0	0	0	0	18	15	0	34	9	1	212
Lemon Island	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13	7	15	15	0	0	0	0	50
Washougal	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	15	18	36	0	0	0	0	69
Sand Island	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13
Franz Lake	15	7	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30
Pierce Island	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9
Hardy Slough	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13
total diet samples	43	26	13	43	19	63	84	64	55	21	65	25	4	31	40	61	92	110	9	91	46	28	1033

3.3.3.2 *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 an increase of one trophic level (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 ("permil"; ‰), so stable isotope analysis of nitrogen is useful in determining trophic position. The $^{13}C/^{12}C$ ratio usually changes by less than 1‰, 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 Chinook salmon (*Oncorhynchus tshawytscha*) muscle tissue and several potential food sources to provide information on the food web supporting juvenile salmonids. Juvenile salmonids were collected by NOAA Fisheries staff using a beach seine. 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). Starting 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. 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. In 2013, muscle, mucus, and liver samples were collected.

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.

Plants

Samples of 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.

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 2012-13, whole bodies were wrapped in foil and kept on ice until returned to the lab and stored in a freezer. 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* spp. 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.

3.4 Fish Use

3.4.1 Fish Community

In 2013, NOAA Fisheries monitored prey availability and habitat use by juvenile Chinook salmon and other fishes at the six trend sites, Franz Lake in Reach H (previously sampled in 2008, 2009, 2010, 2011, and 2012), Campbell Slough in Reach F (sampled from 2007-2012), Whites Island site in Reach C (sampled from 2009-2012), Secret River and Welch Island in Reach B (sampled in 2012) and Ilwaco in Reach A (sampled in 2011 and 2012), in order to examine year-to-year trends in fish use at the sites. Coordinates of the sites are shown in Table 8.

Table 8. 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
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 March 2013 through December 2013. 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. Fish sampling events for 2013 are shown in Table 9.

Table 9. Number of beach seine sets per month at 2013 EMP sampling sites. NS = not sampled. No sampling was conducted in October or November because of the federal government shut-down and related delays in transfer of project funds.

Site	March	April	May	June	July	August	Sept	Dec	Total
Ilwaco Slough	1	3	3	3	3	3	3	3	19
Secret River	3	2	2	3	3	3	3	NS[3]	19
Welch Island	1	3	1	2	2	3	3	NS[3]	15
Whites Island	3	3	1	3	3	3	3	NS[3]	19
Campbell Slough	NS[1]	NS[1]	1	3	3	3	3	3	13
Franz Lake	3	1	1	NS[2]	NS[1]	3	2	3	13
Grand Total	11	12	9	14	14	18	17	9	104

[1]Not sampled because site was closed due to nesting bald eagles
[2]Not sampled due to high water conditions
[3]Not fishable due to strong currents

When Chinook salmon were present, up to 30 individual juvenile Chinook salmon 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.

3.4.2 Salmon Metrics

3.4.2.1 *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.

3.4.2.2 *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 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:

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

3.4.2.3 *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 seven 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.

3.4.2.4 *Chemical Contaminants in Chinook salmon*

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.

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).

3.1 Mainstem conditions

3.1.1 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 11.070 N, 123 11.246 W). A second platform, funded by 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 34.618 N, 122 22.783 W). 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 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.

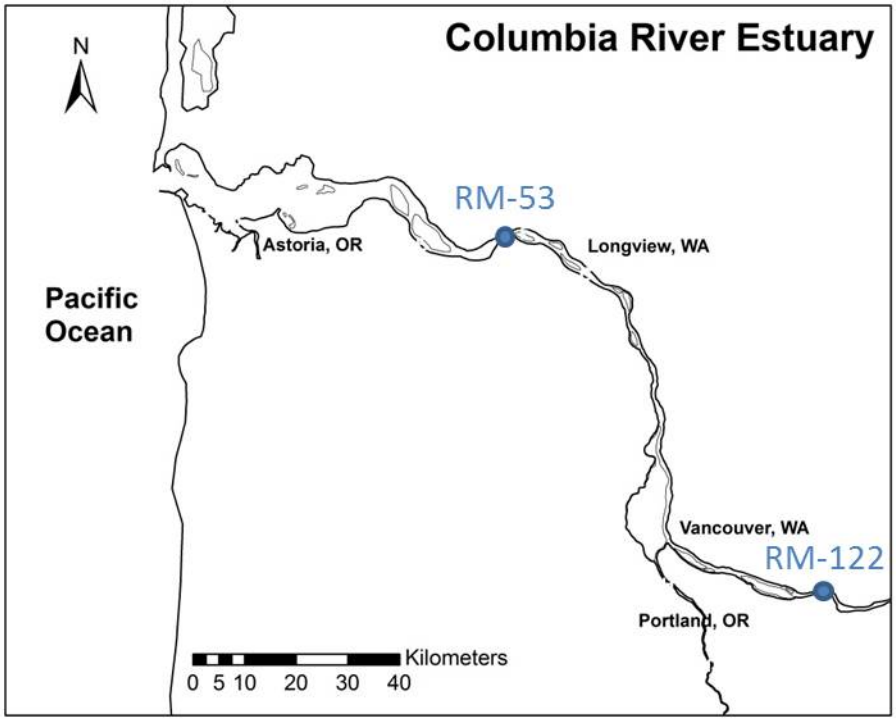


Figure 5. Station locations for the two *in situ* water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program.

3.1.2 Operation of RM-122 Platform at Port of Camas-Washougal

The instrument platform ran continuously from September 2012 through December 2013. Regular maintenance visits were conducted by OHSU personnel. The instruments were removed in December 2013 for factory maintenance, including new calibrations.

3.1.3 Sensor Configuration

Common instruments/sensors to both platforms are provided in Table 10. Sensors are configured to collect a sample and telemeter the data every hour. In addition to the parameters listed in Table 1, 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 (three consecutive measurements daily). During the sampling period, the Cycle-PO4 ran for 182 days between January 16, 2013 and September 12, 2013. The filter size on the instrument is 10 microns, which is significantly higher than traditional filtered samples (0.45 um). Therefore, data must be compared with caution as some phosphate removed by traditional sampling is measured by the Cycle-PO4.

Table 10. Description of the components on the sensor platforms.

Company	Sensor	Parameters
Satlantic	LOBO	Power distribution Sensor control Wireless communication Data management
Saltlantic	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 *a* Concentration

3.1.4 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 typically maintained at shorter intervals). 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. Maintenance trips include cleaning of all sensors and surfaces and performing any other needed maintenance. Additionally, water samples are collected for laboratory analysis of nutrients and chlorophyll *a*. The schedule of maintenance activities is given in Table 11.

Table 11. Maintenance dates, September 2012-December 2013.

RM-53	RM-122
9/5/2012	9/5/2012
12/4/2012 12:00	12/10/2012
1/8/2013 11:00	1/16/2013
2/12/2013 11:15	2/7/2013
3/26/2013 10:30	3/27/2013
4/23/2013 11:35	4/17/2013
5/21/2013 10:05	5/29/2013
6/18/2013 11:30	6/27/2013
8/20/2013 11:00	7/15/2013
12/4/2013 11:00	8/6/2013
	8/14/2013
	9/3/2013
	12/15/2013

3.1.5 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. Chlorophyll *a* measurements are used to correct the *in situ* fluorometer measurements. The discreet samples and the corresponding sensor data for nitrate and chlorophyll *a* are shown in Table 12.

Table 12. Comparison of *in situ* data with laboratory measurements of water samples.

Location/Parameter/# measurements	Correlation
RM-53/Nitrate/28	$y = 0.9x + 3$ $r^2 = 0.95$
RM-122/Nitrate/42	$Y = 0.9x + 1$ $r^2 = 0.99$
RM-53/Chl/13	$Y = 1.1x - 1$ $r^2 = 0.98$
RM-122/Chl/13	$Y = 0.9x + 1$ r2 = 0.93

3.1.6 Fluxes

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 6) reflects inputs from the tributaries of the lower Columbia River, including the Willamette, Cowlitz, and many smaller rivers.

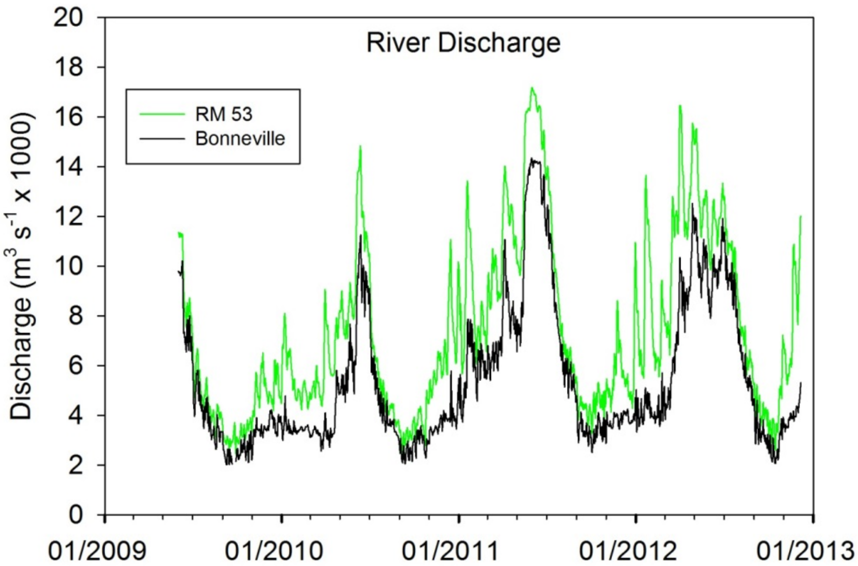


Figure 6. Columbia River daily discharge (m^3 d^{-1}) measured at RM-53 (green line) and Bonneville Dam (black line) during the period July 2009-December 2013. The difference in discharge between the two sites represents the input from tributaries in the Lower Columbia River.

4 Results

4.1 Mainstem conditions

4.1.1 Biogeochemical observations at RM-53 2009-2013

The data for all measured parameters and river discharge is shown in Figure 7. Figure 8 is the time series data expanded for 2013 to highlight the seasonal changes during the reporting period.
River discharge had a relatively large winter flux compared to the previous three years and the discharge associated with the spring freshet did not reach as high as levels as 2011 or 2012. Chlorophyll *a* levels during the spring bloom reached values larger than in the two previous years, and comparable to the high values measured in 2010. The summer chlorophyll *a* was similar to previous years in that there was overall less chlorophyll *a* than during spring, before the freshet. Turbidity, CDOM and nitrate displayed seasonal variability typical of high winter values associated with episodic events and low summer values associated with lower discharge, although inter-annual variability is evident. Temperature displayed seasonal variability consistent with previous years, including low temperatures during winter and high temperatures during summer. The number of days during 2013 that the mainstem river was above various thresholds for habitat quality are shown in Table 13. During 2013, there were more days above 19 °Celsius than in 2010 or 2012, but not 2009.

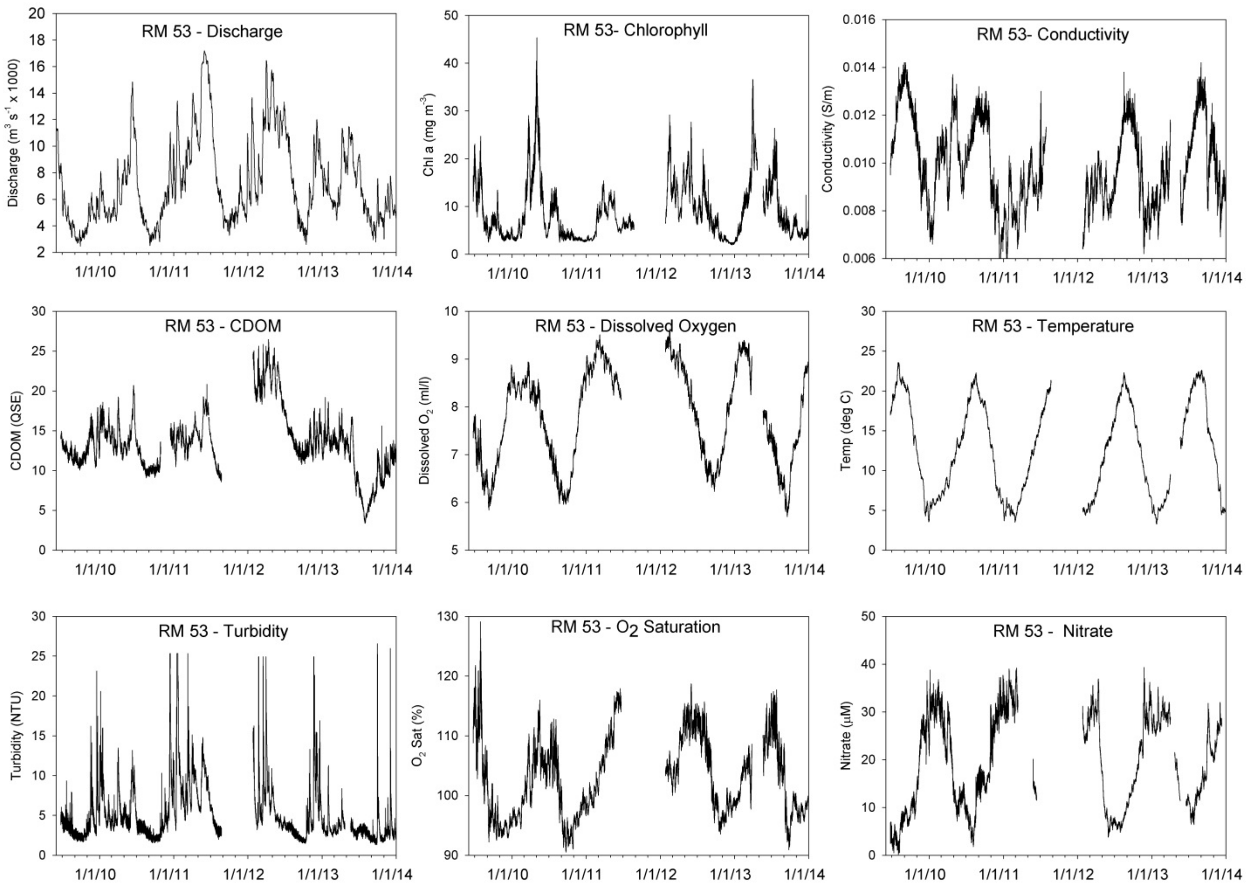


Figure 7. Time series of biogeochemical parameters measured hourly at RM-53 for the period July 2009 – December 2013. CDOM = colored dissolved organic matter.

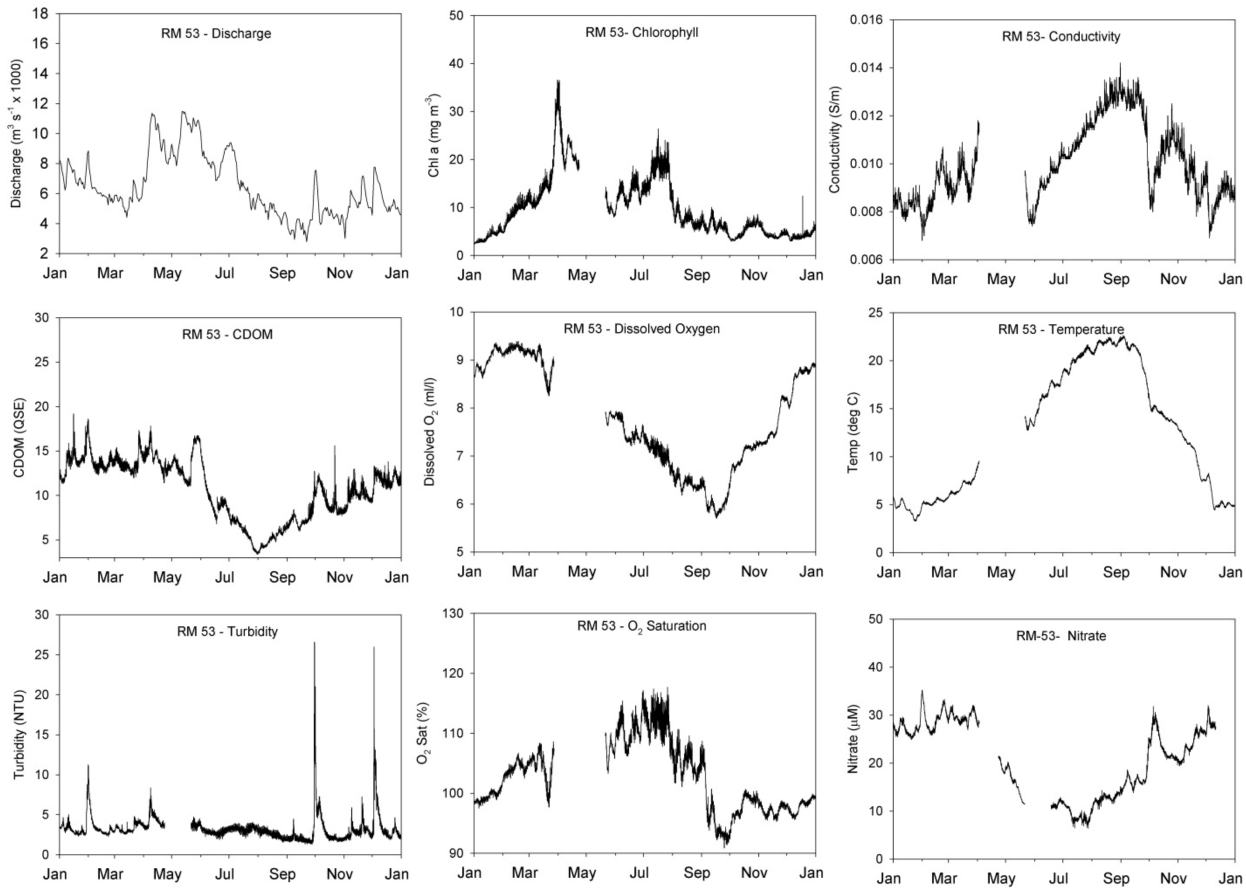


Figure 8. Time series of biogeochemical parameters measured hourly at RM-53 for 2013 only.

Table 13. Number of days with temperatures > 19°C measured in the Columbia River at RM-53.

	2009	2010	2012	2013
Range 19-21 °C	70	49	53	67
Above > 21° C	11	2	2	14
Total > 19°C	82	51	55	81

4.1.2 Biogeochemical observations at RM-122 for the reporting period

The data for all measured parameters and river discharge are shown in Figure 9. River discharge was highest during the freshet, with winter fluxes lower than those measured at RM-53. Chlorophyll *a* levels reached 40 $\mu g\ L^{-1}$ during the spring bloom and were lowest during winter months. CDOM levels were highest during the onset of the spring freshet and lowest during summer months. Dissolved oxygen changes were driven by temperature at the seasonal scale, and by biological productivity and possible super-saturation from spillage over dams at shorter time scales. Dissolved oxygen was never lower than 6 ml L^{-1}. Turbidity and nitrate displayed a seasonal cycle similar to RM-53, with highest concentrations measured during winter with lowest concentrations measured during summer.

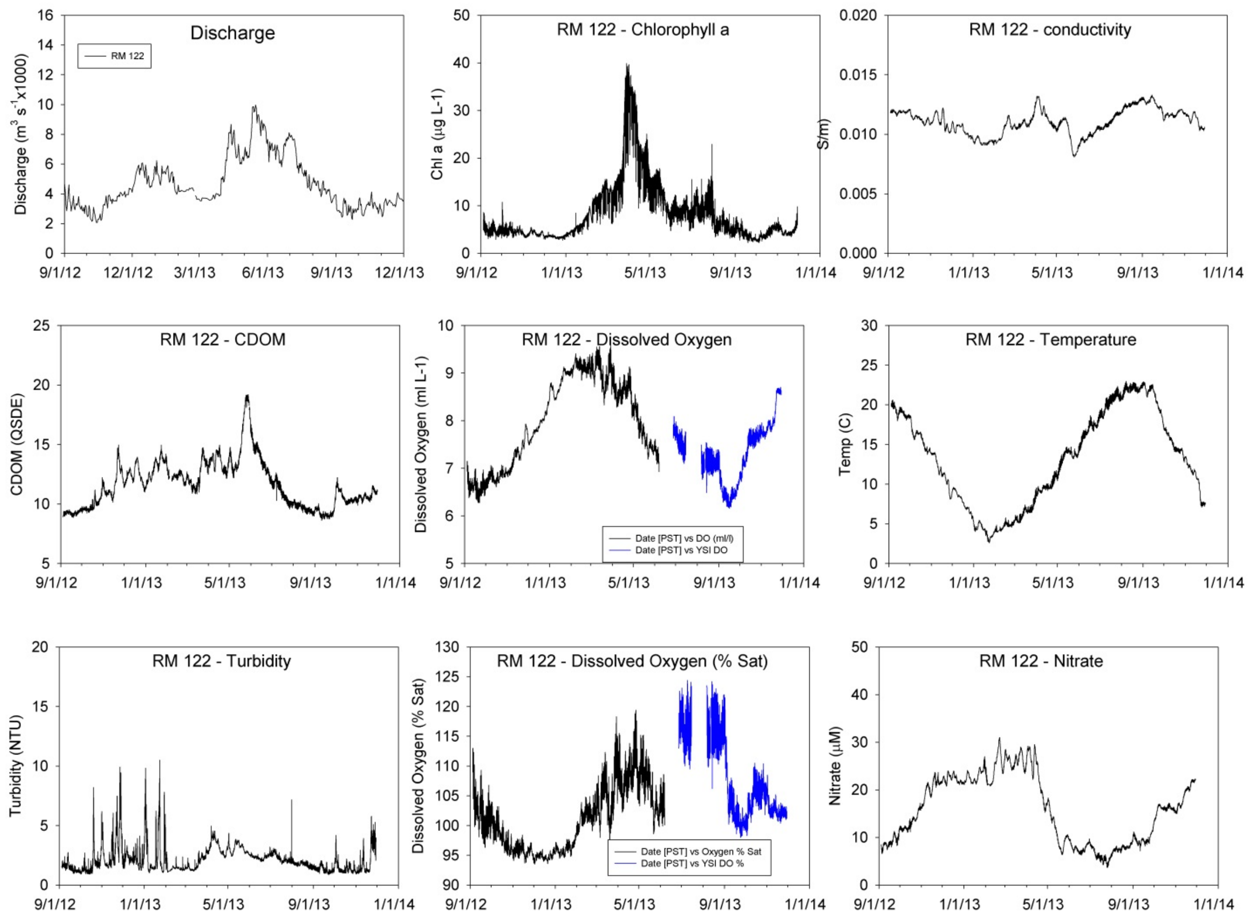


Figure 9. Time series of biogeochemical parameters measured hourly at RM-122 for the period September 2012– December 2013.

4.1.3 Comparison of RM-53 and RM/122 for the reporting period

The data for all measured parameters and river discharge are shown in Figure 10. River discharge was higher at RM-53 at all times as a result of tributary inputs downstream of RM-122, including the Willamette River. During winter months, the high discharge measured at RM-53 was not observed at Bonneville Dam. During the freshet, discharge was high at both locations indicating the large flux from the mainstem Columbia. During summer, the discharge was nearly equal for several months indicating the low contribution from tributaries during this time. Chlorophyll *a* levels were very similar at both sites, including during the spring bloom. One exception was during summer, when RM-53 showed higher levels during August. Conductivity was comparable at both locations, however during the winter, conductivity was lower at RM-53, correlating t with the higher discharge observed during this period. CDOM levels were higher at RM-53 during winter and lower following the spring freshet. Dissolved oxygen and temperature showed very similar trends at the seasonal scale, and the measurements were very similar in magnitude throughout the year. Turbidity and nitrate tended to be higher at RM-53, especially during the winter period when discharge was high. However in summer both variables at similar variability and concentration.

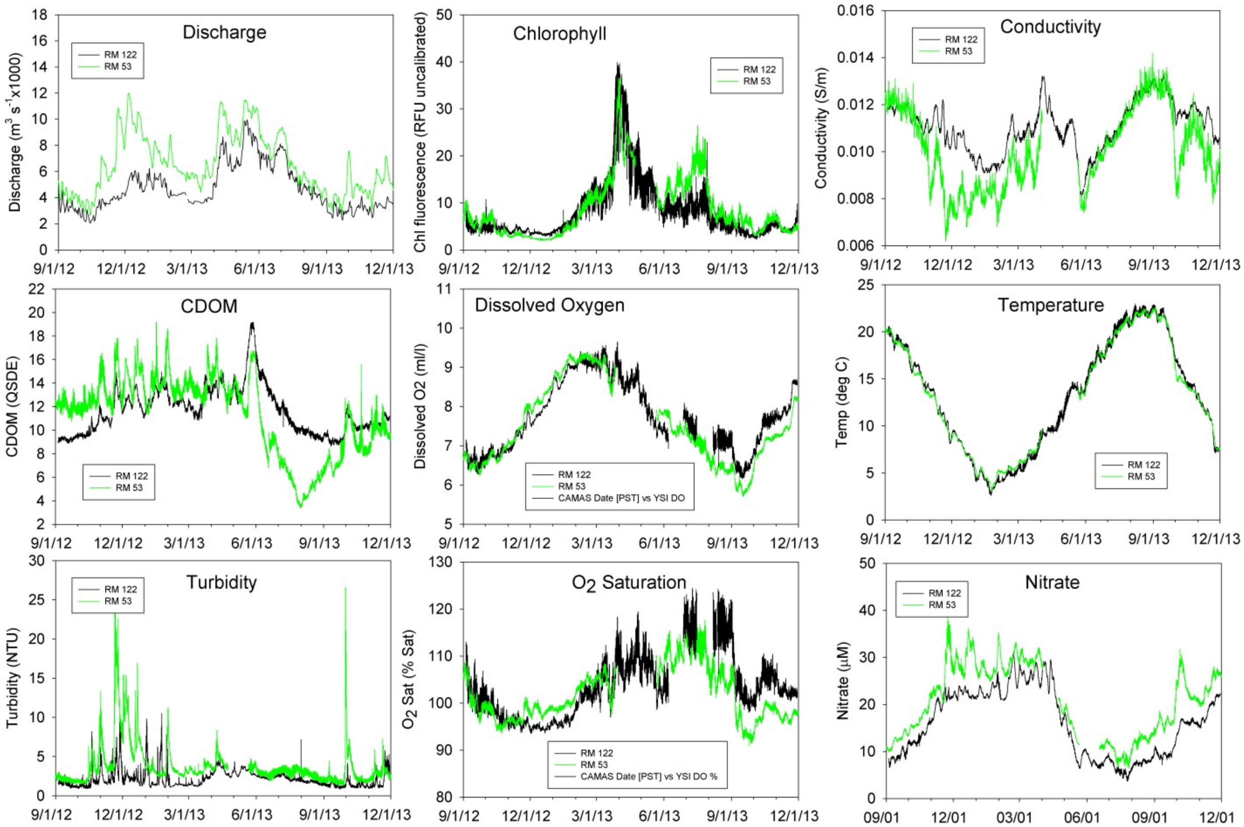


Figure 10. Comparison of RM53 and RM122 for the time period September 2012-December 2013.

4.1.4 Fluxes at RM-53 and RM-122 during reporting period

Nitrate flux was higher at RM-53 at all times of the year, consistent with high concentrations and higher river discharge (Figure 11). The highest nitrate fluxes were observed between December – February when discharge was high and associated with regional rainfall events. A similar pattern is observed for turbidity and CDOM during the winter period. However in spring and summer the turbidity and CDOM fluxes are more similar between the two sites, especially after the spring freshet.

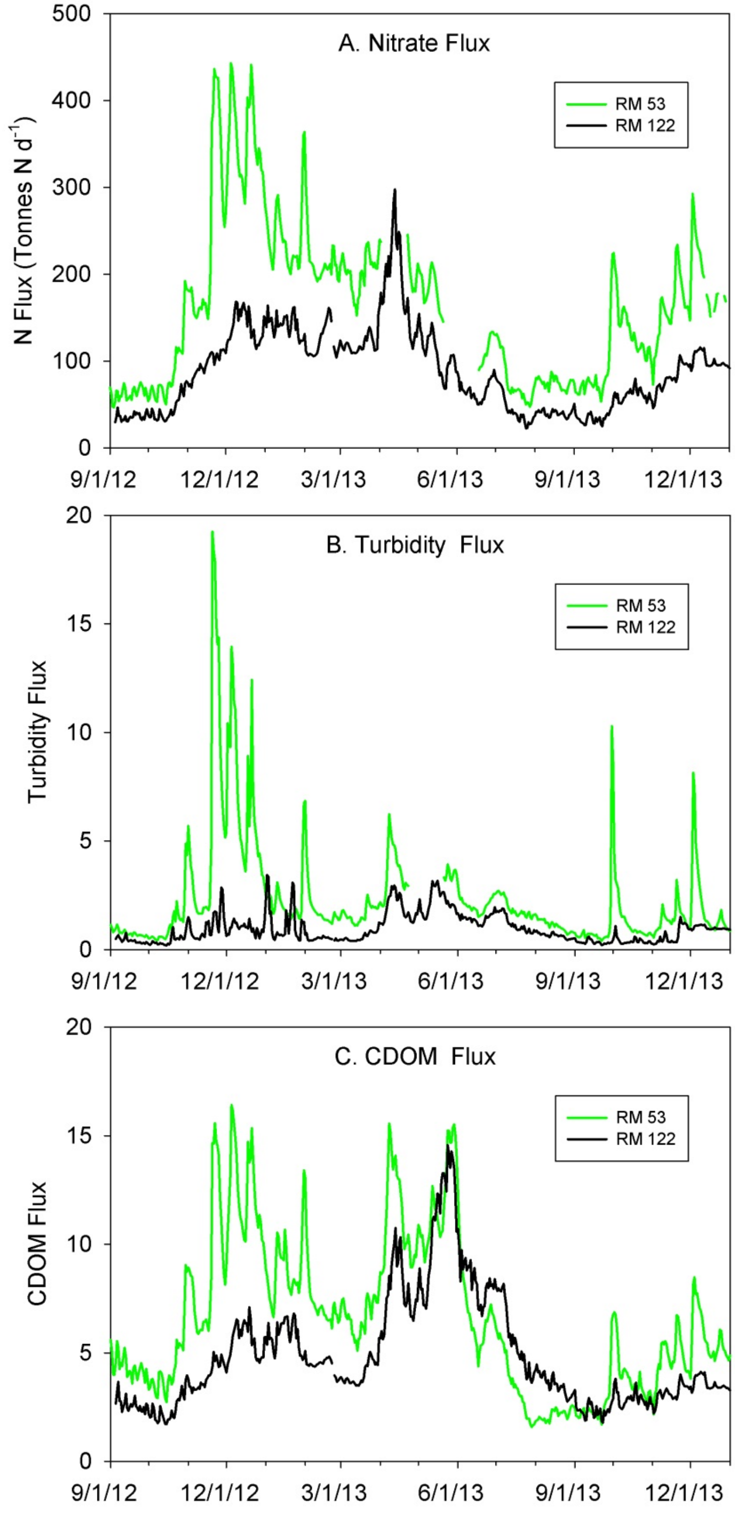


Figure 11. Flux calculations for nitrate, turbidity, and CDOM calculated from sensor measurements and river discharge.

4.1.5 Phosphate at RM-122 during the reporting period

Phosphate ranged from below detection (0.05 µM) in winter to approximately 0.4 µM in late summer (Figure 12). Based on these concentrations, the N:P ratio can be calculated for the nitrate and phosphate measurements, as shown in Figure 13. Phytoplankton typically require an N:P ratio of 16 for nutritional requirements, therefore nearly all year long phytoplankton are potentially limited in growth by phosphate concentration.

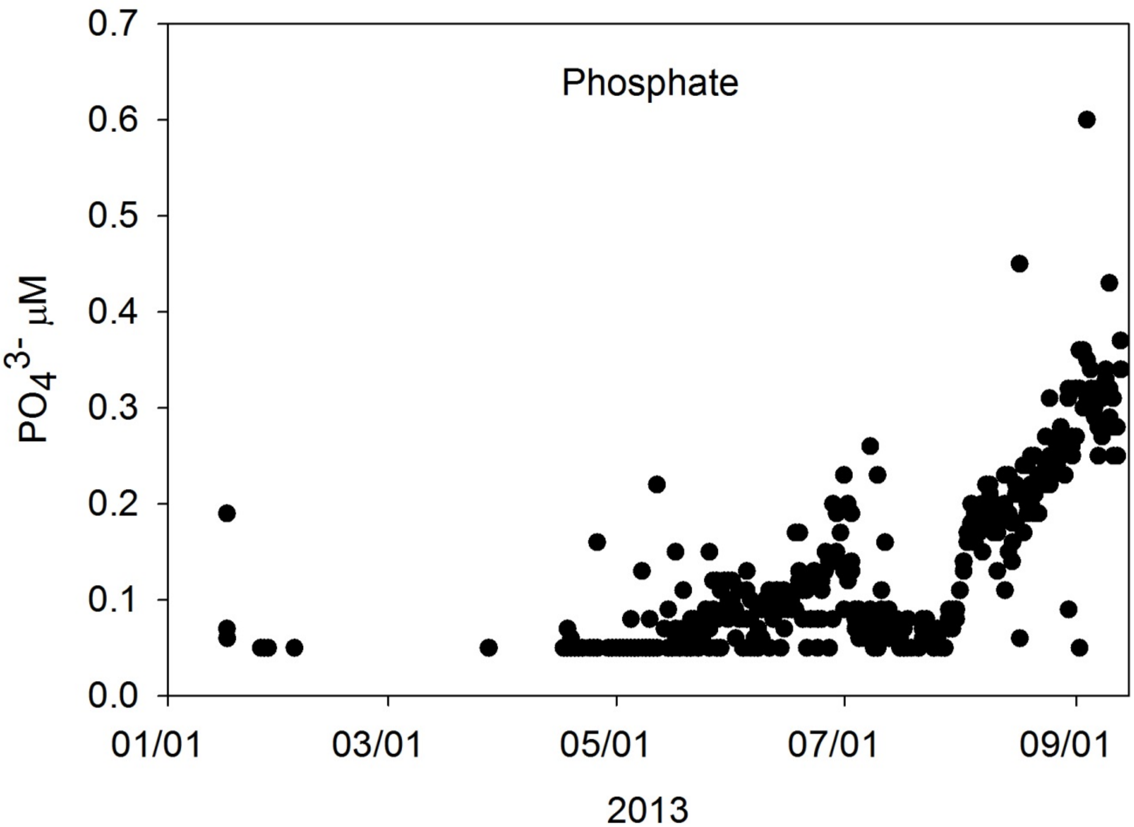


Figure 12. Phosphorus concentrations at RM-122 measured as ortho-phosphate using the CyclePO4 *in situ* instrument.

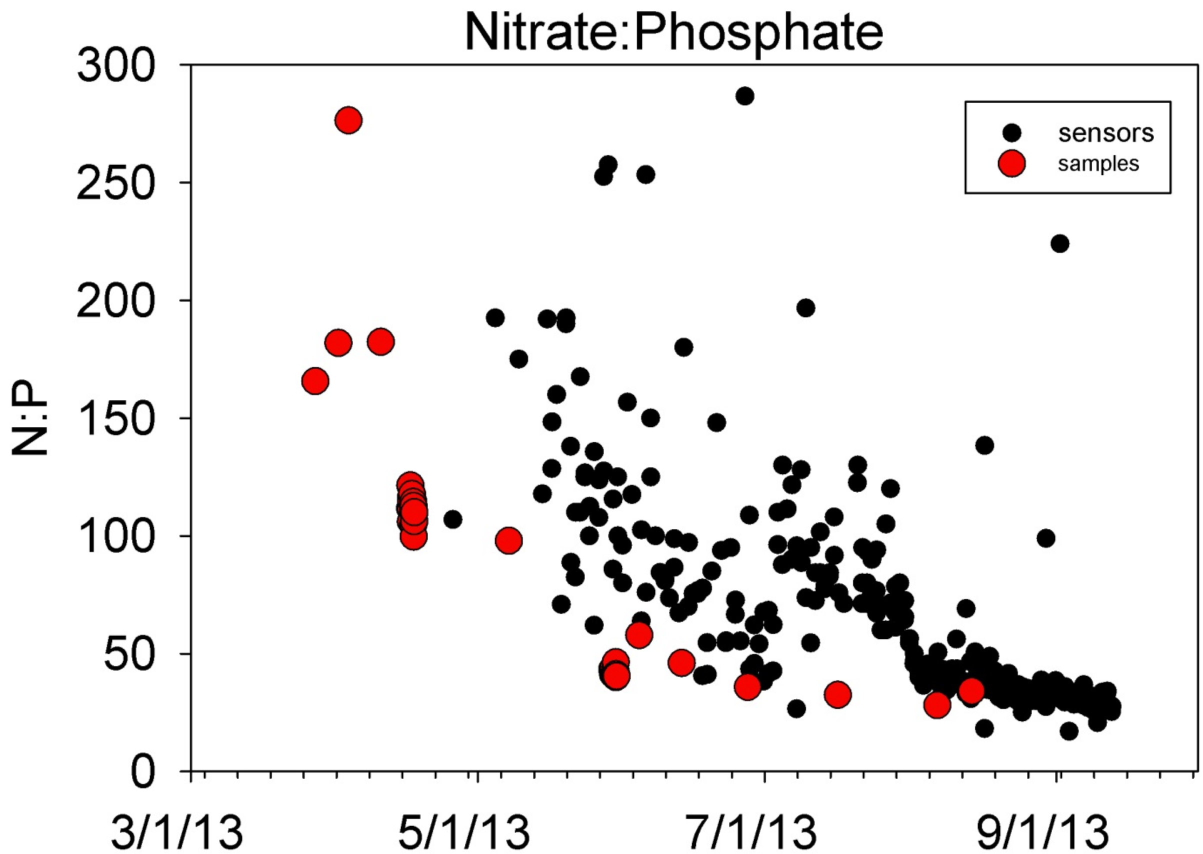


Figure 13. N:P at RM122 using measurements made by grab samples (large dots) and from *in situ* sensors (small dots).

4.2 Abiotic site conditions

4.2.1 Continuous water-quality

4.2.1.1 *Temperature*

Water temperature at all sites increased throughout the 2013 water-quality monitoring period (April–July; Figure 14). However, water temperature at all sites decreased during the snowmelt-driven higher Columbia River flows in May. The largest daily variation in temperature was at Ilwaco, due to intense tidal flushing at that site. The mean weekly-maximum water temperature exceeded the Washington state temperature standard (17.5° C) by early-mid June at all sites. That standard was exceeded starting in mid-April at Ilwaco, except during the May freshet (Figure 15). Among sites, Whites Island had the coolest mean weekly-maximum temperature.

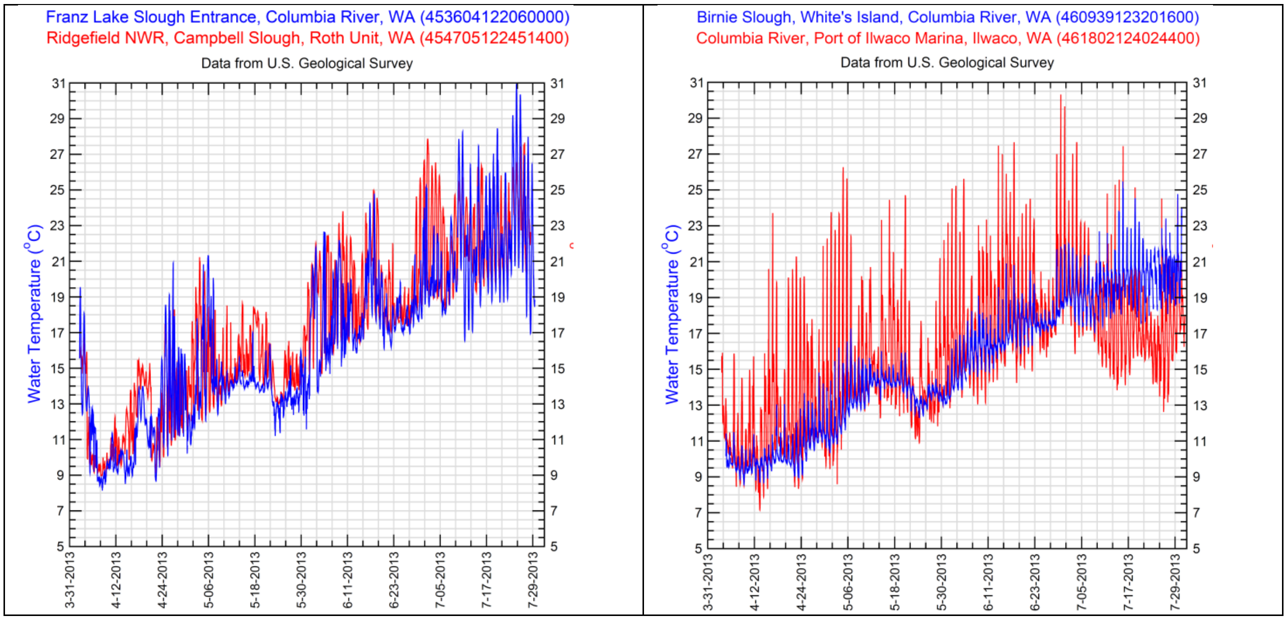


Figure 14. Continuous water temperature in April–July, 2013: a) Franz Lake Slough and Campbell Slough; b) Whites Island and Ilwaco. [°C, degrees Celsius].

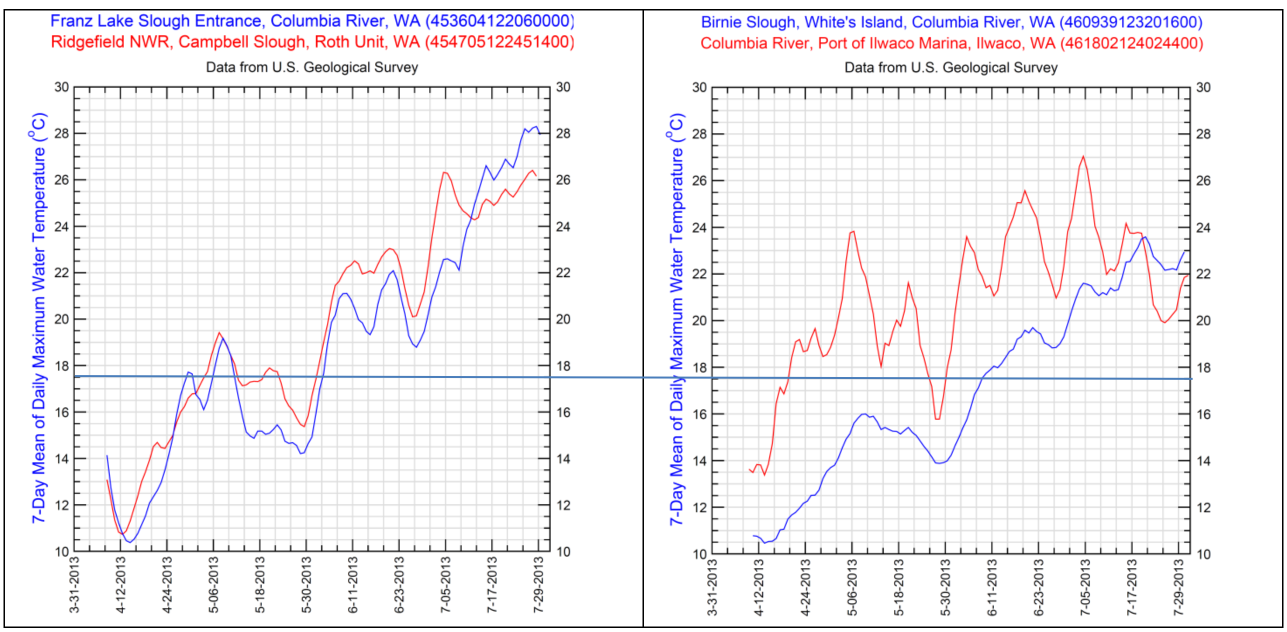


Figure 15. Mean weekly maximum water temperature in April–July, 2013: a) Franz Lake Slough and Campbell Slough; b) Whites Island and Ilwaco. [°C, degrees Celsius].

4.2.1.2 *Dissolved oxygen*

Median dissolved oxygen concentrations at all sites decreased gradually during the 2013 monitoring period. Daily variations in dissolved oxygen concentrations were the largest at all sites in mid- to late-July, when water depth was lowest (Figure 16). Among sites, the largest daily variation was at Ilwaco. The daily minimum dissolved oxygen was less than the 8.0 milligrams per liter (mg/L) Washington state threshold during the entire monitoring season (Ilwaco), from mid-June through July (Whites Island), intermittently, but mostly in June–July (Franz Lake) and intermittently throughout season (Campbell Slough) (Figure 17). However, the daily maximum was never less than 8.0 mg/L at any site in 2013 (Figure 18). Median daily dissolved oxygen was less than 8.0 mg/L during much of July (Franz Lake Slough), intermittently throughout the period (Campbell Slough), during parts of May and June (Ilwaco), and never at Whites Island (Figure 19). During those periods, at least half of the hourly measured values during the day had low dissolved-oxygen concentrations. However, since the daily maxima were never less than 8.0 mg/L, all sites had some dissolved oxygen conditions that were suitable for salmonids every day during the monitoring period.

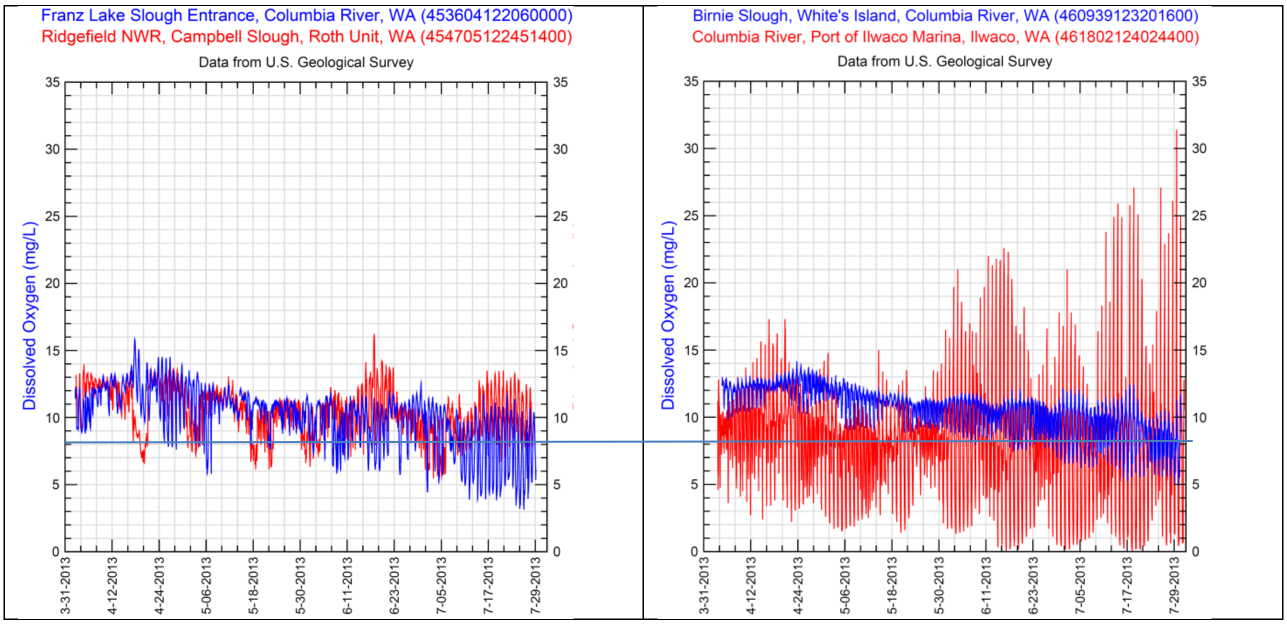


Figure 16. Continuous dissolved oxygen concentrations in April–July, 2013: a) Franz Lake Slough and Campbell Slough; b) Whites Island and Ilwaco. [mg/L, milligrams per liter]

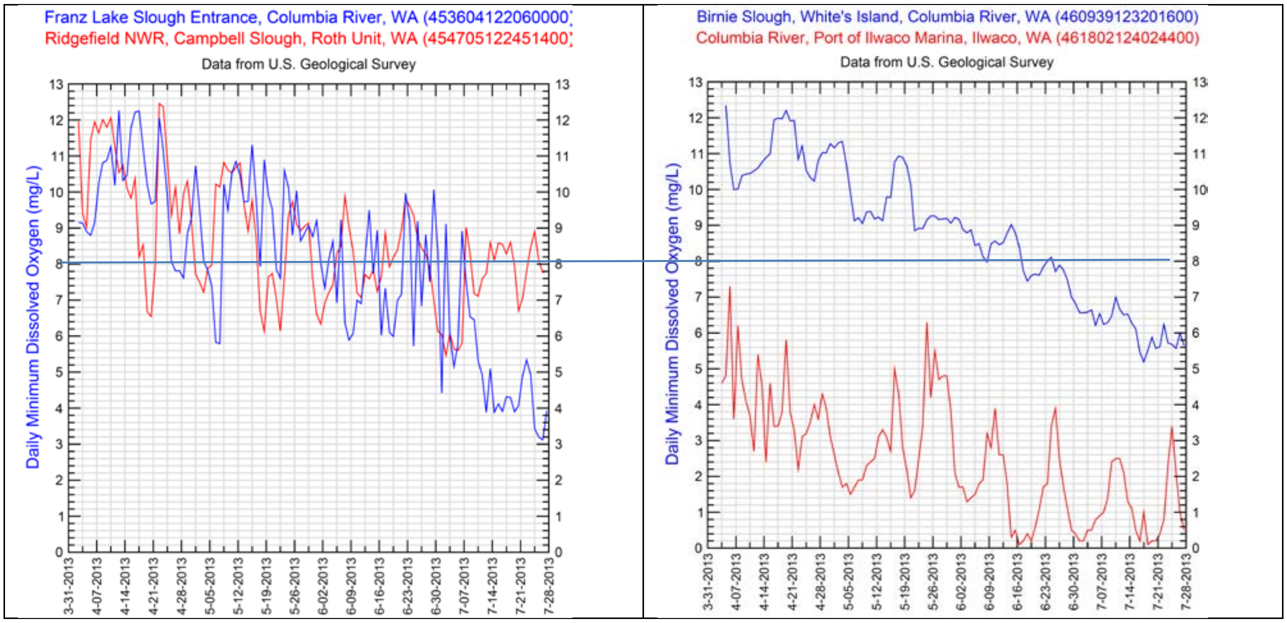


Figure 17. Daily minimum dissolved oxygen concentrations in April–July, 2013: a) Franz Lake Slough and Campbell Slough; b) Whites Island and Ilwaco. [mg/L, milligrams per liter]

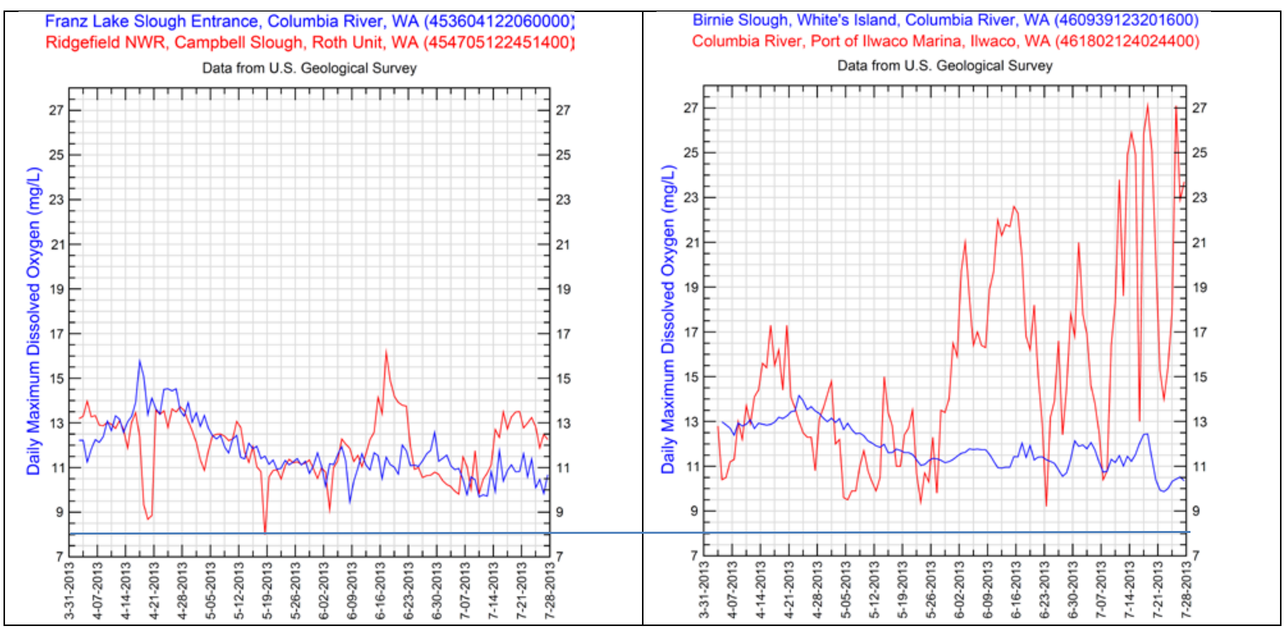


Figure 18. Daily maximum dissolved oxygen concentrations in April–July, 2013: a) Franz Lake Slough and Campbell Slough; b) Whites Island and Ilwaco. [mg/L, milligrams per liter]

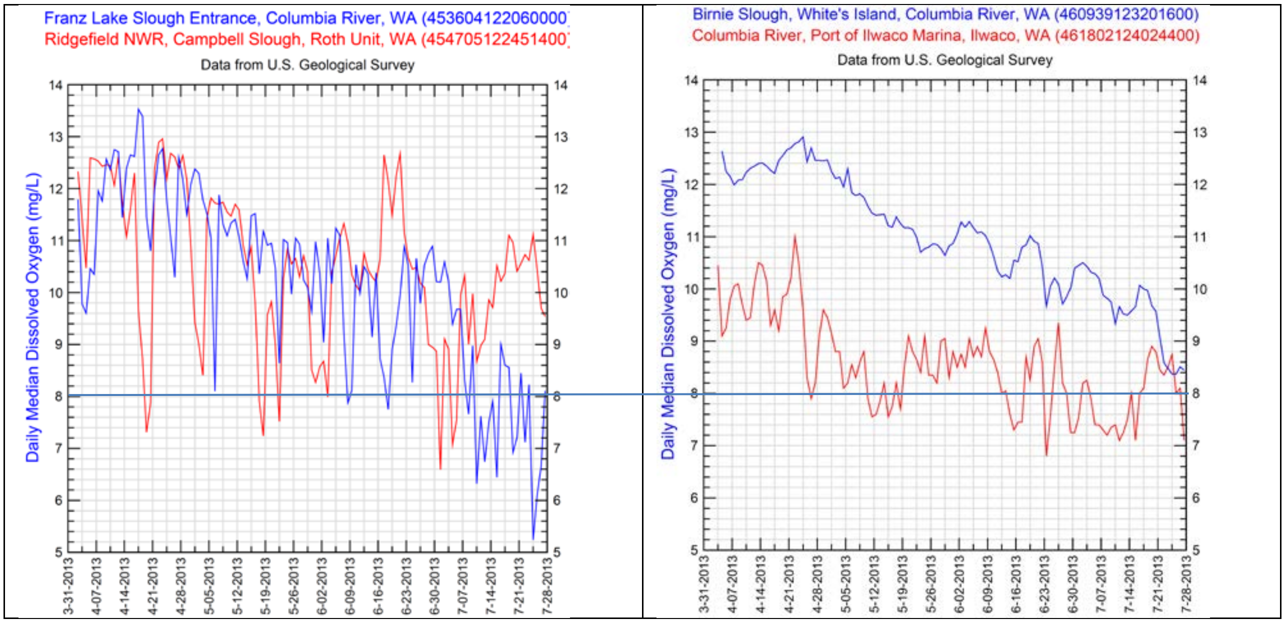


Figure 19. Daily median dissolved oxygen concentrations in April–July, 2013: a) Franz Lake Slough and Campbell Slough; b) Whites Island and Ilwaco. [mg/L, milligrams per liter]

4.2.1.3 *pH*

pH fluctuated throughout the 2013 monitoring period at all sites with no overall increasing or decreasing trend during the period (Figure 20). There were no pH data from Franz Lake in 2013 due to equipment malfunctions. The pH was never less than the minimum pH threshold of 6.5 in 2013 at any site. However, the maximum threshold of 8.5 was exceeded in mid-June and mid- to late July at Campbell Slough and periodically at Whites Island and Ilwaco. The large peak at Campbell Slough in June appears to be due to primary production, while the high values in July appear to be driven by temperature changes.

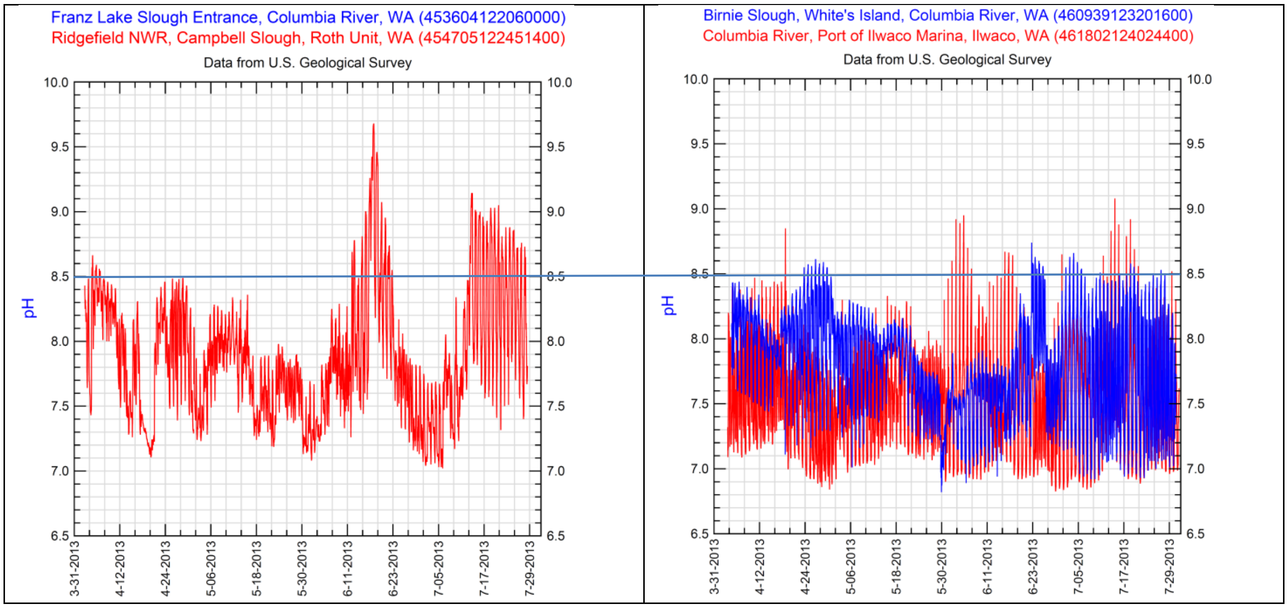


Figure 20. Continuous pH in April–July, 2013: a) Campbell Slough; b) Whites Island and Ilwaco.

4.2.1.4 *Specific conductance*

Specific conductance ranged from approximately 50 to 200 microsiemens per centimeter (µS/cm) at the three sites with more river(freshwater)-dominated source water (i.e., Franz Lake, Campbell Slough, and Whites Island) and ranged from approximately 2,000 to 32,000 µS/cm at the marine-influenced site (Ilwaco; Figure 21). Patterns in specific conductance highlight when the sites are influenced by freshwater sources such as Columbia River water, other water sources (Franz Lake for Franz Lake Slough, Campbell Lake for Campbell Slough), or the Pacific Ocean (Ilwaco).

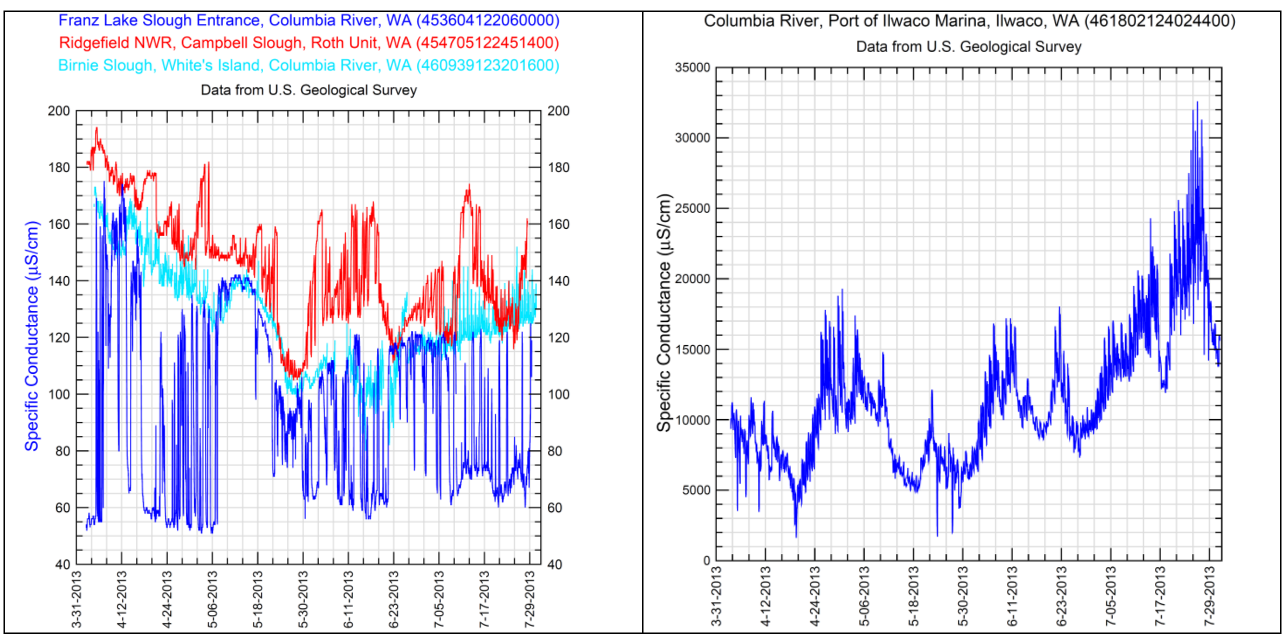


Figure 21. Continuous specific conductance in April–July, 2013: a) Franz Lake Slough, Campbell Slough, and Whites Island; b) Ilwaco. Note the different Y axis scales. [µS/cm, microsiemens per centimeter]

4.2.2 Nutrients

Ilwaco had the highest total nitrogen concentrations among the sites throughout the 2013 sampling period (Figure 22). The three more river-dominated sites (Franz Lake Slough, Campbell Slough, and Whites Island) had higher total nitrogen concentrations through early May, after which concentrations decreased during the spring freshet before increasing in early July. Combined nitrate and nitrite concentrations were highest at all sites in April, decreasing in early May at the three more upriver sites and in late May at Ilwaco (Figure 23). Total Kjeldahl Nitrogen (organic nitrogen plus ammonia) concentrations were highest at Ilwaco and Campbell Slough (Figure 24). Most TKN was in the form of organic nitrogen, although ammonia also contributed to TKN at Ilwaco and Whites Island.

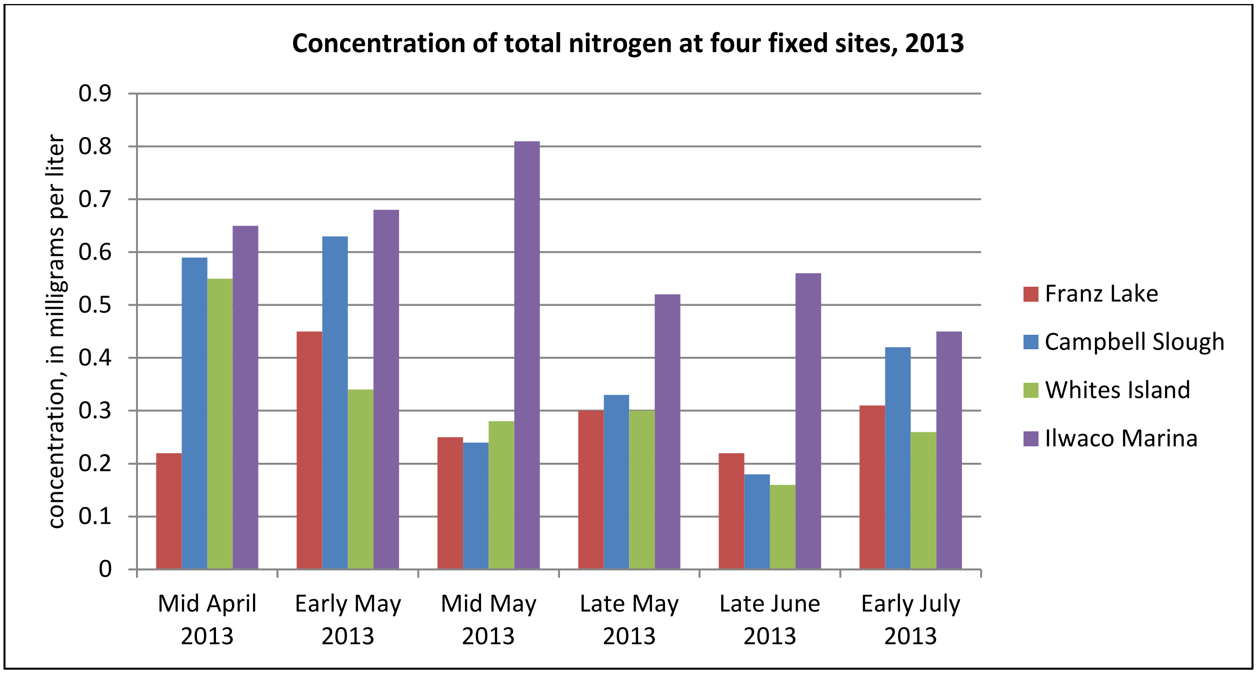


Figure 22. Total nitrogen concentration at four fixed sites in 2013

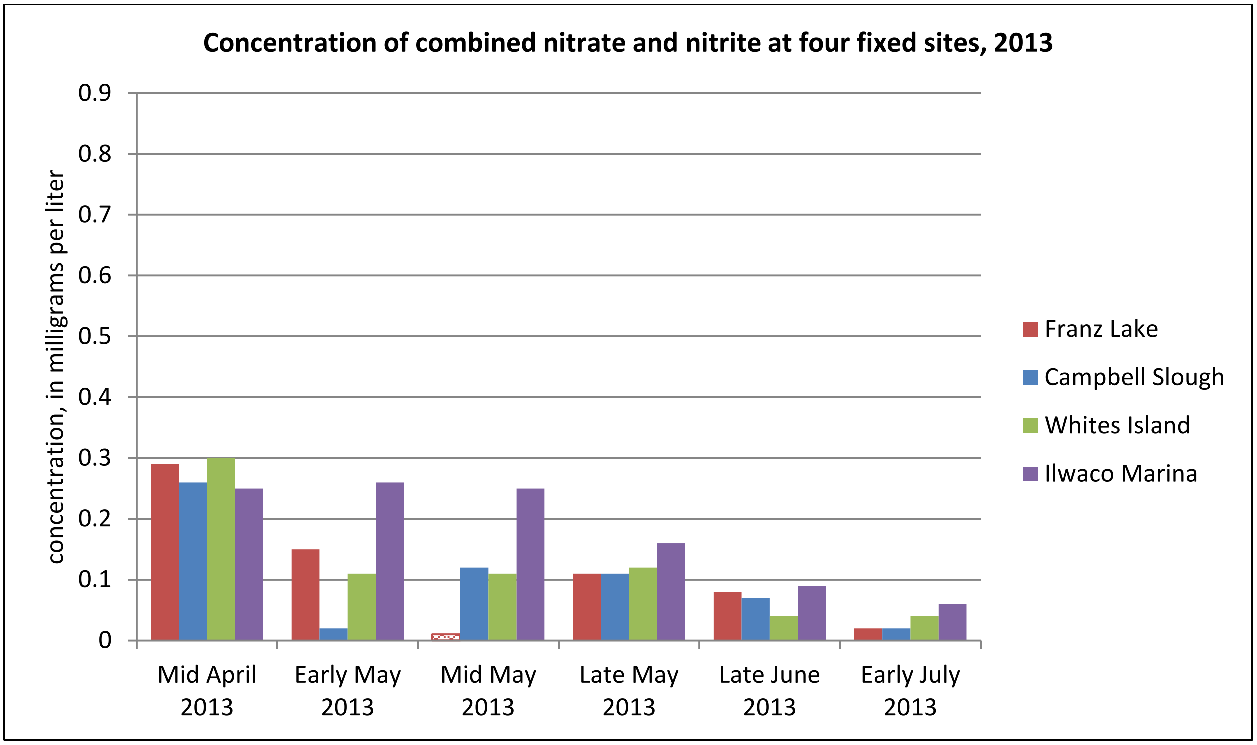


Figure 23. Combined nitrate and nitrite concentration at four fixed sites in 2013. Hashed columns indicate concentrations less than the detection limit.

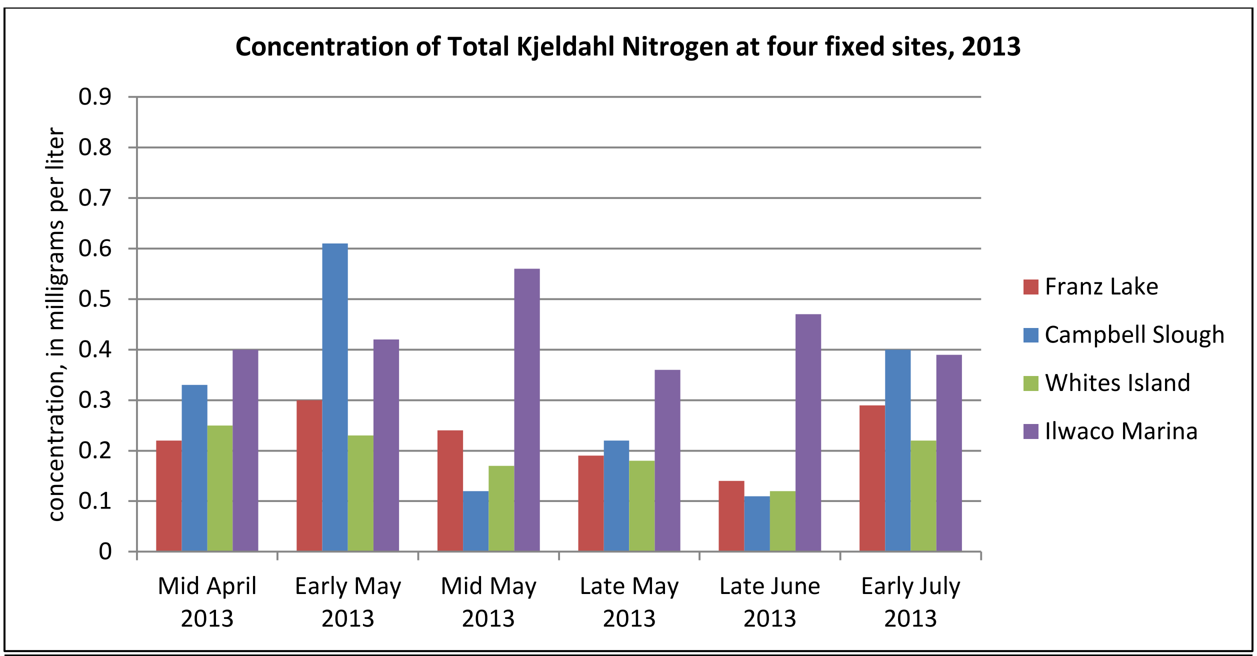


Figure 24. Total Kjeldahl Nitrogen concentration at four fixed sites in 2013

Ilwaco had the highest total phosphorus and orthophosphate concentrations among the four water-quality monitoring sites throughout the sampling period in 2013 (Figure 25). The more three river-dominated sites had very low orthophosphate concentrations throughout the season (Figure 26). The highest orthophosphate concentrations were in May (Ilwaco and Whites Island) and July (Franz Lake

Slough and Campbell Slough). As with total nitrogen, total phosphorus concentrations at Ilwaco peaked in mid-May. The highest total phosphorus concentrations at the three upriver sites were in early May.

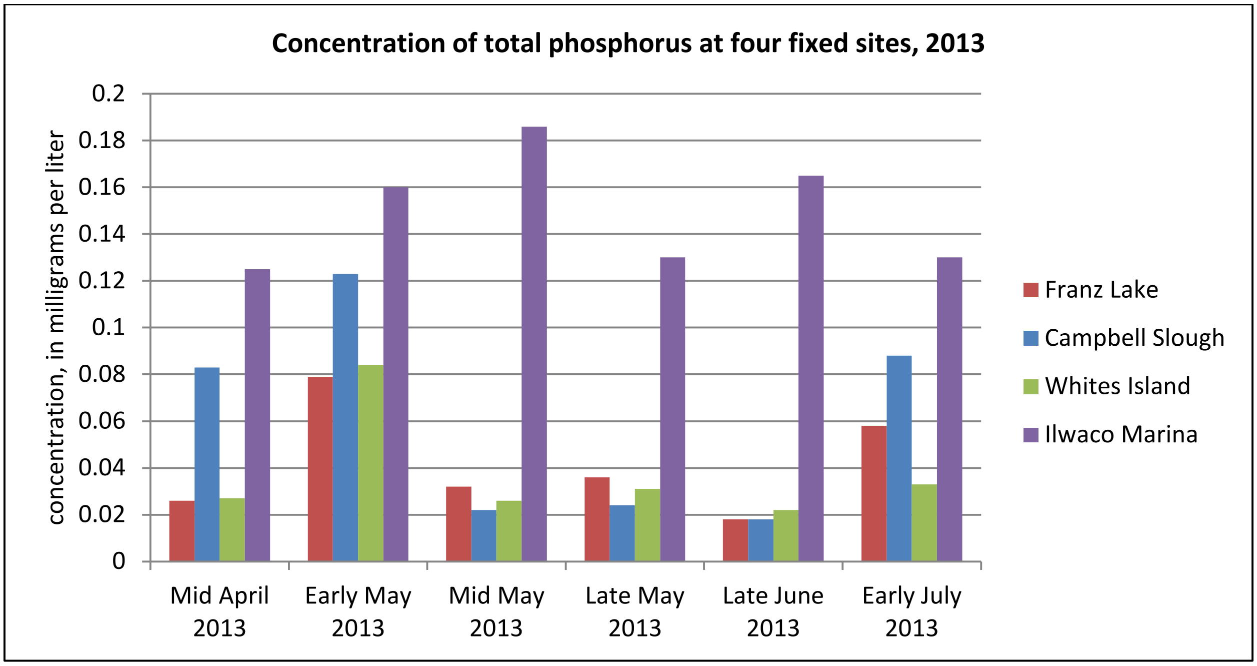


Figure 25. Total phosphorus concentration at four fixed sites in 2013

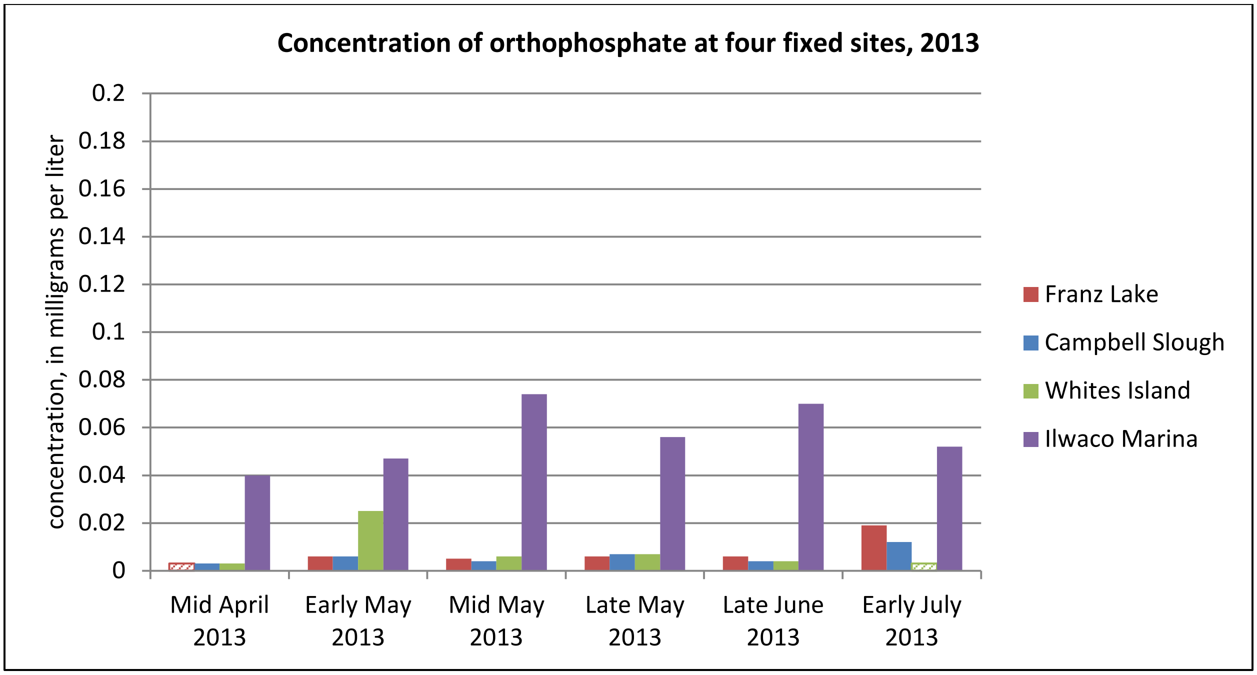


Figure 26. Orthophosphate concentration at four fixed sites in 2013. Hashed columns indicate concentrations less than the detection limit.

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.

4.3 Habitat Structure

4.3.1 Hydrology

Hydrographs from all the years in which WSE was sampled at the trend sites, including the 2013 water year, are provided in Appendix A. The following observations were made for these sites:

- The Ilwaco site results indicate that the WSE at rkm 6, is very minimally affected by the spring freshet, but is however elevated due to winter storm events and extreme high tides. Additionally, low water elevations are truncated at the site due to the elevation of the tidal channel above extreme low water.

- The Secret River site, at rkm 37, is also affected by winter storm events and not the spring freshet. WSEs at this site reach slightly higher and the tide range is greater than at the Ilwaco site in part due to the lower elevation of the tidal channel where the sensor is located. The low elevation marsh at the site is infrequently exposed and conversely, the high elevation marsh is infrequently inundated.

- The Welch Island site, located at rkm 53, is predominantly tidal, however slightly elevated WSE was detectable during the prolonged spring freshet in 2012. Winter storms are also drive higher water levels at this site.

- The hydrologic patterns at the Whites Island site (rkm 72) exemplify the mixed of hydrologic drivers in the LCRE. The tidal range is over 2m in most months, while elevated water levels are also experienced during winter storm events and during the spring freshet.

- The Cunningham Lake and Campbell Slough sites, at rkm 145 and149, respectively, have similar hydrologic patterns except the Cunningham Lake site has a slightly greater tidal range and slightly lower WSE during the freshet. The Campbell Slough WSE does not get as low as the Cunningham Lake site due to a weir located at the mouth of the Slough limiting drainage. The primary hydrologic driver at the site is the spring freshet, although in 2013 winter storms also increased the WSE at these sites. Both sites were inundated for approximately three months during the winter then again for three months during the spring freshet with the WSE nearly equal in magnitude for the two periods.

- The Franz Lake site has no discernable tidal signal and low water was maintained at the site by a beaver dam in the fall that washed out sometime in the winter. The winter and spring high WSEs are both discernable, however, the spring levels were considerably higher than the winter at this site. In 2013, the site was inundated for approximately one month in the winter and for approximately three months in the spring.

The cumulative inundation during the growing season, as measured by the sum exceedance value (SEV), is a means of comparing sites to each other and overtime. Inter-annual variation in inundation patterns is much greater at the upper River sites (Figure 27), where seasonal flooding can result in months of inundation during high water years. At the lower, tidally dominated sites, inundation occurs frequently, but for a short duration of a few hours. At the Whites Island site the impact of high water during the 2011 and 2012 spring freshets is slightly discernable in the SEV at the average marsh elevation, whereas the up-river sites have large differences in the SEV between years.

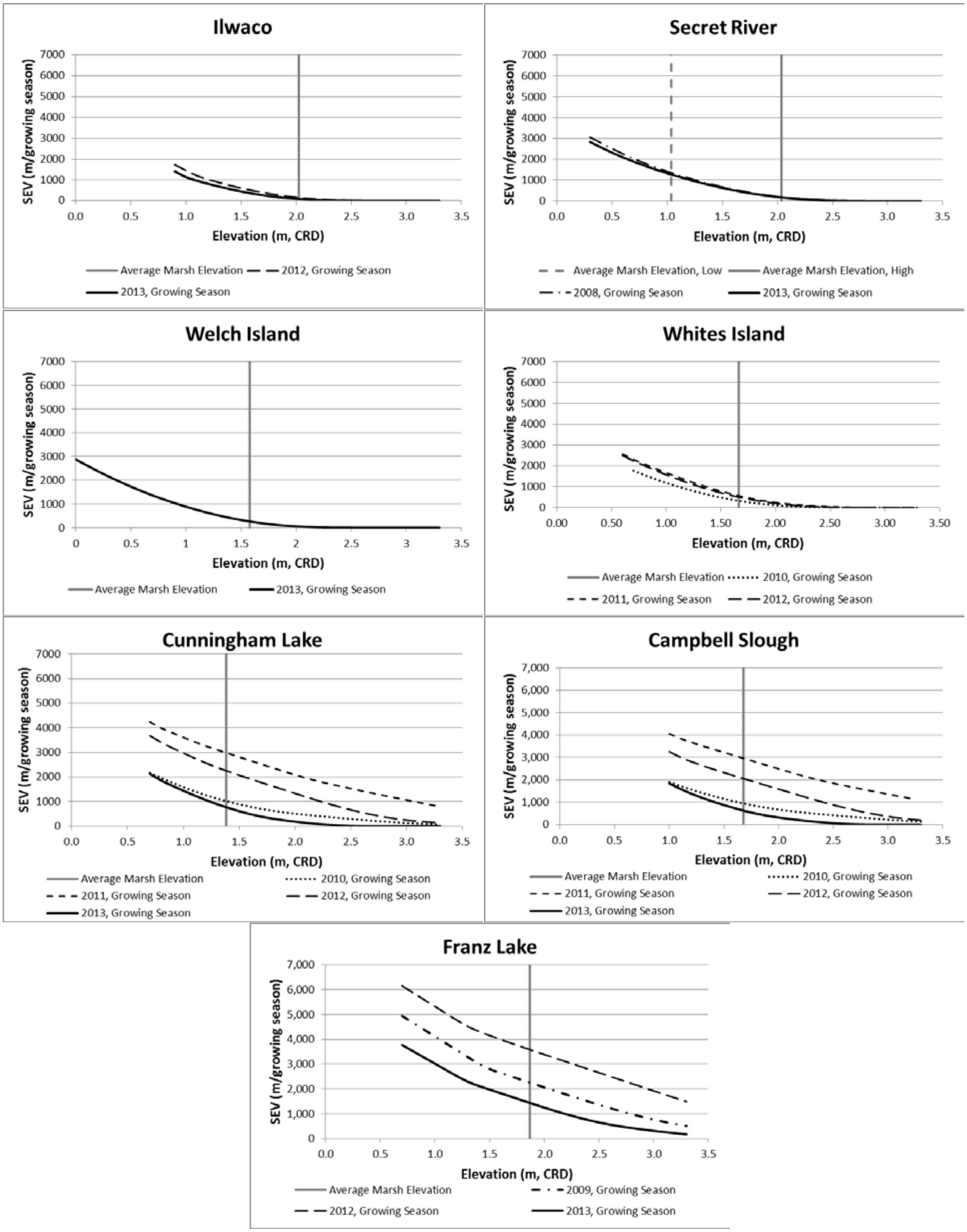


Figure 27. Growing season sum exceedance values (SEVs) for the trend monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the LCRE; the vertical line represents the average elevation at each site.

4.3.2 Sediment Accretion Rates

Annual sediment accretion rates ranged from -1.7 (erosion) to 3.0 cm/year in 2013, with most values falling between 0.0 and 1.4 cm/year (Table 14). These rates are consistent with those found at a larger number of reference sites in the LCRE as documented by Borde et al. (2012). Within this range, rates are variable and are not consistently related to elevation or river kilometer (rkm). Likewise, the data from the trend sites indicates that the rates are also variable between years (Table 15). The site with the lowest inter-annual variability is Cunningham Lake (SD ±0.26) and the highest is Franz Lake (SD ±1.74), which also had the highest accretion rates measured at the trend sites.

Table 14. Sediment accretion rates measured at sites in 2013.

Site	Rkm	Stake Elevation (m, CRD)	Days Deployed	Annual Accretion/ Erosion Rate (cm)
Ilwaco(BBM)	6	1.82	356	0.6
Secret River – low (SRM-L)	37	1.01	356	-1.7
Secret River – low (SRM-L)	37	0.98	356	0.3
Secret River – high (SRM-H)	37	2.15	355	1.4
Welch Island 2 (WI2)	53	1.66	356	0.8
Whites Island – mid (WHC-M)	72	1.34	356	1.2
Whites Island – high (WHC-H)	72	1.89	356	0.2
Cunningham Lake (CLM)	145	1.53	355	1.3
Campbell Slough (CS1)	149	1.56	351	0.2
Government Island GOM)	181	1.97	407	0.4
Old Washougal River Channel (OWR)	195	1.60	382	-1.0
Reed Island 2 (RI2)	204	1.58	386	-0.2
Franz Lake (FLM)	221	1.88	335	3.0

Table 15. Sediment accretion rates at the trend sites between 2008 and 2013. See Table 14 for definition of site codes.

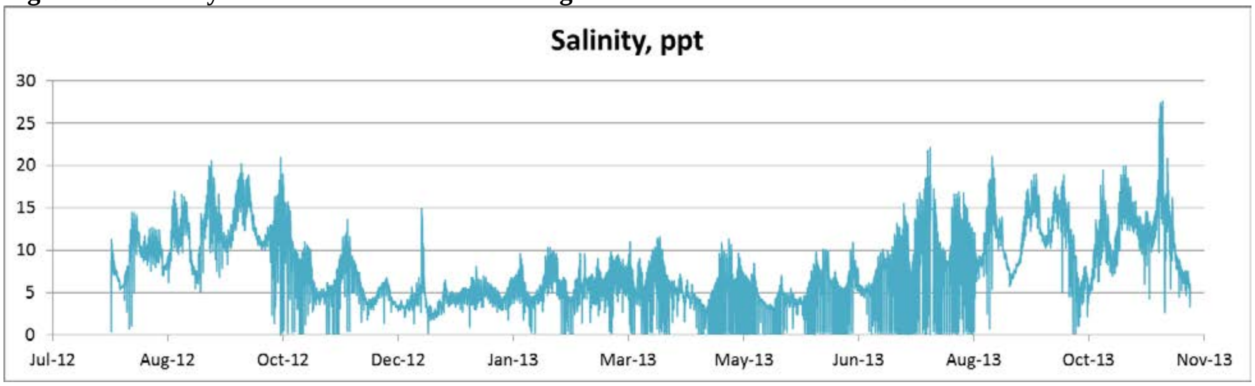
Site Code:	BBM	SRM-L	SRM-H	WI2	WHC-M	WHC-H	CLM	CS1	FLM
Elevation (m, CRD):	1.82	0.98	2.15	1.66	1.34	1.89	1.53	1.56	1.88
Year	Annual Rate (cm)								
08-09	ND[1]	ND	0.2	ND	ND	-1.2	ND	ND	0.5
09-10	ND	ND	2.8	ND	ND	1.0	1.9	0.4	ND
10-11	1.7	ND	0.9	ND	ND	0.1	1.6	1.7	3.0
11-12	0.1	ND	ND	ND	ND	0.9	1.4	0.9	-0.4
12-13	0.6	0.3/-1.7	1.4	0.8	1.2	0.2	1.3	0.2	3.0
Standard Deviation	0.82	1.41	1.10	NA	NA	0.88	0.26	0.67	1.74

1 ND No data.

4.3.3 Salinity

Salinity was measured at the Ilwaco site between August 2012 and November 2013 (Figure 28). The range was between 0.1 and 29.1 parts per thousand (ppt), with 62 percent of the measurements falling between 2 and 10 ppt. The variability is related to tidal patterns on a daily and monthly scale and also seasonally with the highest salinity occurring in the fall when precipitation and river flows are lowest.

Figure 28. Salinity at the Ilwaco site between August 2012 and November 2013.



4.3.4 Vegetation Species Assemblage

A list of all species and their cover values observed at the trend sites in 2013 is provided in Table C-1, Appendix C. The cover and elevation range of each species is also plotted by site and provided in Figure C-1, Appendix C. Based on the cover and species richness (Table 16), the vegetation assemblages observed at the 2013 monitoring sites can be broadly grouped into categories associated with the emergent marsh (EM) vegetation zones (Figure 2) as follows:

Zone 1	low species richness/high cover
Zone 2	high species richness/high cover
Zone 3	no data collected in 2013
Zone 4 and 5	low species richness/moderate cover

In the three lowest river sites, native vegetative cover was higher than non-native cover (Table 16), with native species cover dominated by Lyngby's sedge (*Carex lyngbyei*). The Zone 2 sites (Secret River, Welch Island, and Whites Island) have the highest number of species, ranging from 41 to 52 species observed (the latter is for the Secret River low and high marsh plots combined), with cover generally over 100 percent. The Secret River site (rkm 37) had lower cover of the native sedge and 35 percent cover of reed canarygrass. At the Whites Island site and the Zone 4 sites, non-native cover, predominantly reed canarygrass (*Phalaris arundinacea*; Table 17), was equal or greater than native cover. Reed canarygrass cover was very low at the Zone 5 site, where the native water smartweed (*Polygonum amphibium*) had the highest cover.

Table 16. Species richness and areal cover of native and non-native species at the 2013 monitoring sites.

Site	Rkm	# Native Species	Native Species % Cover	# Non-native Species	Non-native Species % Cover	Total # Species	Total % Cover
Ilwaco	6	17	101.2	2	1.5	19	102.8
Secret River - High	37	23	79.4	8	47.6	31	126.9
Secret River - Low	37	23	85.1	5	5.4	28	90.5
Welch Island	53	35	104.1	9	24.4	44	128.5
Whites Island	72	29	34.4	12	76.6	41	111.1
Campbell Slough	145	21	36.2	15	36.7	36	72.9
Cunningham Lake	149	12	30.9	3	40.1	15	71.0
Franz Lake	221	16	47.2	3	11.4	19	58.5

Trends in the dominant vegetation cover at the sites are depicted in Figure 29. Variability in the cover of the dominant species is most discernable in the upper river sites, with low overall cover occurring in 2006, 2008, 2011, and 2012, the highest water years of the sampling period. Additionally, at Campbell Slough cover was decreased in 2007 when cows were periodically present at the site. The Franz Lake site continued to have relatively low cover in 2013 following the high water years. Another discernable trend at all the sites where *P. arundinacea* occurred was an increase in cover of the species between 2012 and 2013 (Table 17). This trend is most notable at Secret River and Whites Island, where the cover increased 15 percent between years. However, at Whites Island the cover had been this high in a previous year (2011); without longer term data at Secret River we cannot discern whether this trend is an upward trajectory or inter-annual fluctuations. At the upper river sites cover in 2013 was not as high as it had been in previous years (2010).

Table 17. Percent cover of *Phalaris arundinacea* at the trend sites in 2012 and 2013.

		Percent Cover *Phalaris arundinacea*								
Site	**Rkm**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
Ilwaco	6	ND	ND	ND	ND	ND	ND	ND	0.0	0.0
Secret River–Low	37	ND	ND	ND	5.3	ND	ND	ND	0.0	0.0
Secret River–High	37	ND	ND	ND	10.4	ND	ND	ND	19.8	35.5
Welch Island	53	ND	ND	ND	ND	ND	ND	ND	5.9	9.8
Whites Island	72	ND	ND	ND	ND	43.0	47.8	56.8	42.0	56.5
Cunningham Lake	145	41.7	16.4	36.1	32.8	38.5	57.3	15.6	22.5	39.2
Campbell Slough	149	35.6	30.7	18.4	28.9	37.9	41.5	33.6	15.2	33.1
Franz Lake	221								5.0	11.2

Submerged aquatic vegetation (SAV) species occur at the lowest elevations of the sites, in the channels and in ponded depressions in the emergent vegetation. Vegetative cover for the SAVs is included in the emergent cover for all of the sites (Table C-1) and provided separately for the channels of five of the trend sites in Table C-2, Appendix C. Two of the sites (Cunningham Lake and Franz Lake) have the channel data included in the cover data for the rest of the site because the transects cross the channel. Horned pondweed (*Zannichellia palustris*) is the only SAV species that occurs at the Ilwaco site and is found in the tidal channel and in ponded areas of the marsh. At the Secret River low marsh site Canada waterweed (*Elodea canadensis*) accounted for 24 percent of the cover, occurring throughout the low marsh in small depressions that hold water at low tide. This is down from down from 35 percent the previous year (Figure 29). At all other sites, SAV species account for less than five percent of the cover in the emergent marsh area. In the channels of the Secret River, Welch Island, and Whites Island sites cover of SAV was dominated by the native species *Elodea* spp. and *Potamogeton richardsonii*, with 49, 85, and 23 SAV cover at the three sites, respectively.

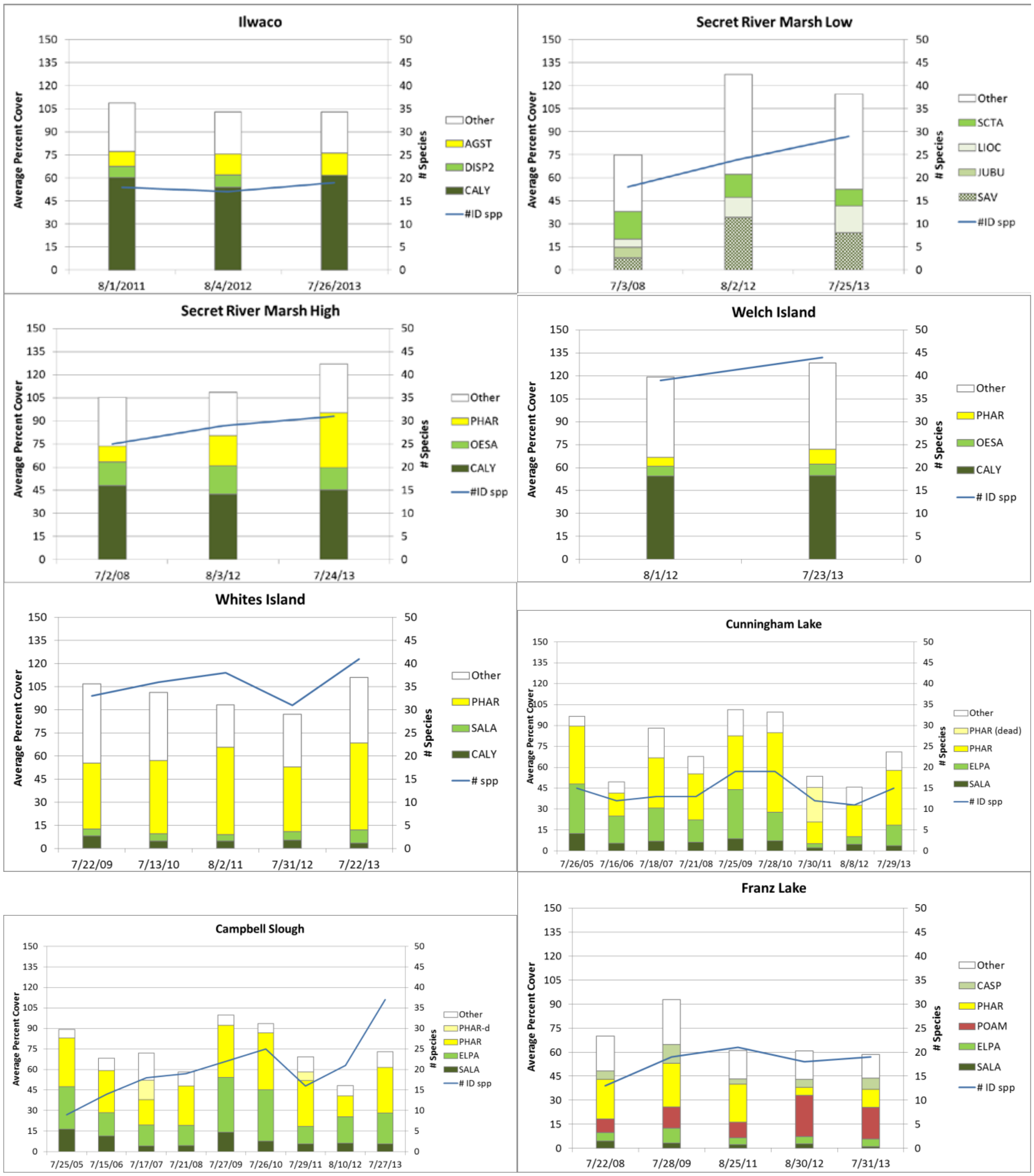


Figure 29. Average percent cover and number of identified species at the trend sites for all years monitored. Sites are presented in the order in which they occur in the River, starting at near the mouth.

4.3.5 Channel Morphology and Inundation

Inter-annual variability of cross section morphology is low as seen by the comparison of single cross section locations from the trend sites. Slight changes in channel morphology are discernable in the cross sections from Whites Island and Franz Lake. These changes are perhaps due to the higher flows that likely occurred at these sites during the high water year of 2012 and the shifting sediments that perhaps followed in 2013.

Channel measurements and the inundation frequency results are presented in Table 18. The channels are generally small with a cross sectional area less than 10 m^2. The Secret River channel was the largest, closer to 20 m^2 for most of the length. The channels vary in width from 1.3 m to 50.1 m; most becoming narrower with increasing elevation except the Ilwaco and Whites Island channels which are wider at the middle than at the mouth. Channel depth ranges from 0.3 m to 2.1 m, with most channels between 0.9 m and 1.2 m in depth. The thalweg elevation of the channels was consistently between 0.3 and 1.0 m and the channel bank between 1.0 and 2.0 m, relative to CRD.

Inundation frequency in the tidal channels of the 2013 sites was calculated during the year (August 2012 to August 2013) and during the peak juvenile salmon migration period (March- July 2013) for the following conditions:

- thalweg of each channel cross section with 50 cm of water or more
- channel bank of each cross section with 10 cm of water or more.

In the lower river, the inundation frequency is similar between the two time periods due to the limited effect of the spring freshet in 2013 in this part of the river. The Secret River channel was inundated more frequently than the Ilwaco or Welch Island channels due to the greater depth and lower bank elevation; the channel had greater than 50 cm of water at least 70 percent of the time and the bank was inundated to 10 cm nearly 60 percent of the time at the mouth and 15 to 45 percent farther up the channel. The other lower river channels were inundated less frequently, with the thalweg inundation generally between 40 and 65 percent and the bank inundation between 10 and 25 percent.

In the upper river, inundation frequency was higher during the peak fish migration period than during the year as a whole, and overall inundation frequencies were higher than in the lower river. During the freshet in 2013, the channels were inundated at least 74 percent of the time and the banks at least 68 percent of the time except the high bank at the mouth of the Franz Lake channel. These values are considerably lower than the inundation frequencies observed in the upper river during the 2012 spring migration period when the channels and the banks were inundated nearly 100 percent of the time.

Table 18. Physical channel metrics measured at each site. The channel mouth (indicated with a *) was measured in 2013; the year of full channel measurement is provided in parentheses after the site code. Inundation time percentages for one year (August 2012 - August 2013) and between 1 March and 31 July, 2013 (the peak juvenile Chinook salmon migration period). Cross sections are numbered starting at the mouth.

								Inundation			
		Physical Metrics						Year		March-July	
Site (year)	**Cross Section**	**Thalweg Elevation (m, CRD)**	**Bank Elevation (m, CRD)**	**Channel Depth (m)**	**Cross Section Area (m^2)**	**Channel Width (m)**	**Width:Depth Ratio**	**% Time WL > Thalweg + 50cm**	**% Time WL > Bank + 10cm**	**% Time WL > Thalweg + 50cm**	**% Time WL > Bank + 10cm**
BBM (11)	1*	0.93	1.71	0.78	4.20	6.80	8.70	45%	28%	42%	24%
	2	0.70	1.86	1.16	8.94	9.30	8.04	55%	22%	51%	18%
	3	0.90	2.12	1.22	9.73	10.10	8.27	46%	11%	43%	9%
	4	1.01	2.00	0.99	4.33	5.20	5.23	42%	16%	38%	13%
	5	1.17	2.26	1.09	1.58	2.70	2.48	35%	7%	31%	5%
SRM (12)	0*	0.15	1.04	0.89	10.6	23.9	26.9	80%	59%	77%	56%
	1	0.32	1.42	1.09	19.3	22.6	20.6	72%	46%	69%	43%
	2	-0.04	2.13	2.17	22.5	14.9	6.87	93%	18%	88%	15%
	3	-0.03	1.98	2.01	20.7	15.1	7.52	92%	24%	88%	20%
WI2 (12)	1*	0.19	1.58	1.39	15.3	20.0	14.4	65%	23%	66%	24%
	2	0.36	1.65	1.29	8.75	9.20	7.13	58%	21%	59%	22%
	3	0.71	1.80	1.09	3.96	5.09	4.67	44%	15%	45%	16%
	4	0.78	1.74	0.96	2.07	3.30	3.44	41%	17%	42%	18%
	5	1.31	1.62	0.31	0.42	1.32	4.27	18%	22%	19%	23%
WHC (11)	1*	0.35	1.43	1.08	22.5	39.6	36.7	ND	ND	ND	ND
	2	0.34	1.41	1.07	10.8	20.5	19.1	ND	ND	ND	ND
	3	0.61	1.53	0.92	11.1	36.2	39.5	ND	ND	ND	ND
	4	0.92	1.93	1.00	34.0	50.1	50.0	ND	ND	ND	ND
	5	0.44	1.45	1.01	1.90	2.83	2.80	ND	ND	ND	ND
CLM (13)	1	0.81	1.11	0.30	3.17	17.6	58.7	59%	67%	74%	80%
CS1 (13)	1	0.78	1.44	0.66	11.2	23.1	35.0	74%	52%	87%	68%
FLM (12)	0*	0.34	2.36	2.02	24.7	23.8	11.8	100%	24%	100%	46%
	BD†	0.99	1.58	0.60	7.00	17.0	28.6	54%	47%	75%	71%
	3	0.40	1.39	0.99	4.20	14.3	14.4	100%	53%	100%	75%
	4	0.85	1.45	0.60	6.20	13.2	22.0	60%	51%	76%	74%

ND = No Data. Water level sensor failed at the site in 2012-13; † The beaver dam was not present in 2013.

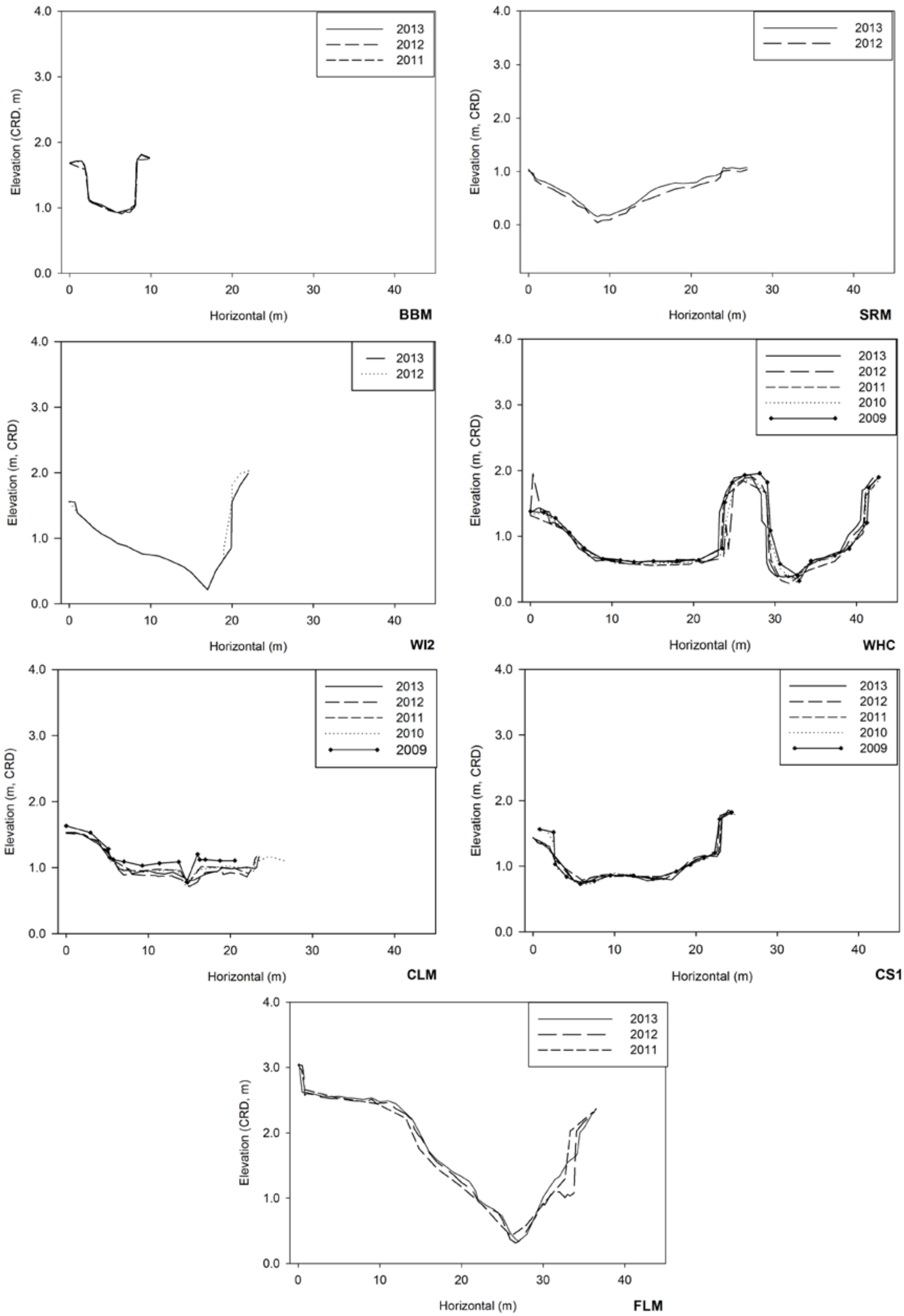


Figure 30. Elevations of the channel cross sections for the trend sites showing multiple years.

4.4 Food web

1.1.1 Net Productivity

1.1.1.1 Temporal Patterns

Primary Production

Quantity

Pelagic

In terms of chlorophyll *a*, phytoplankton abundance as estimated by chlorophyll *a* concentrations was highest in mid-April and mid-May at the three upriver sites (Figure 31), but concentrations were below the limit of detection in late June, except at Whites Island.

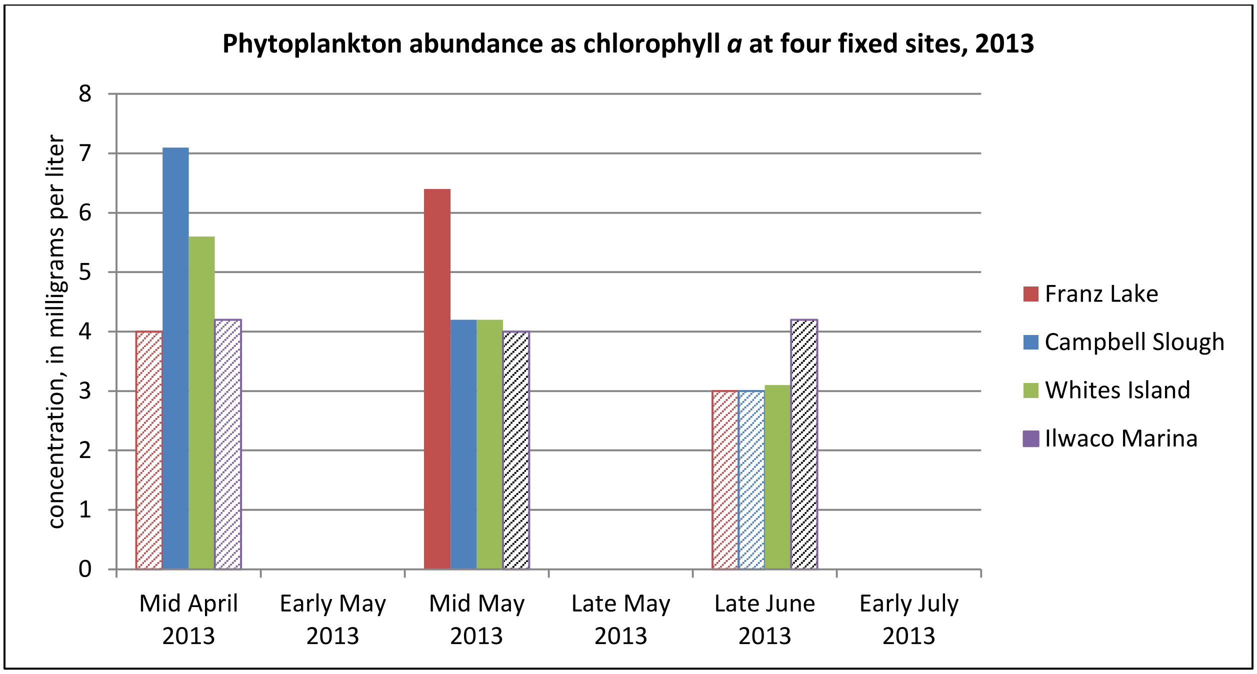


Figure 31. Phytoplankton abundance as chlorophyll *a* concentrations in 2013. Hashed columns indicate concentrations less than the detection limit.

Periphyton abundance (as chlorophyll *a*) was highest at Ilwaco in May and June 2013, although Whites Island also had high periphyton abundance during the sampling period (Figure 32).

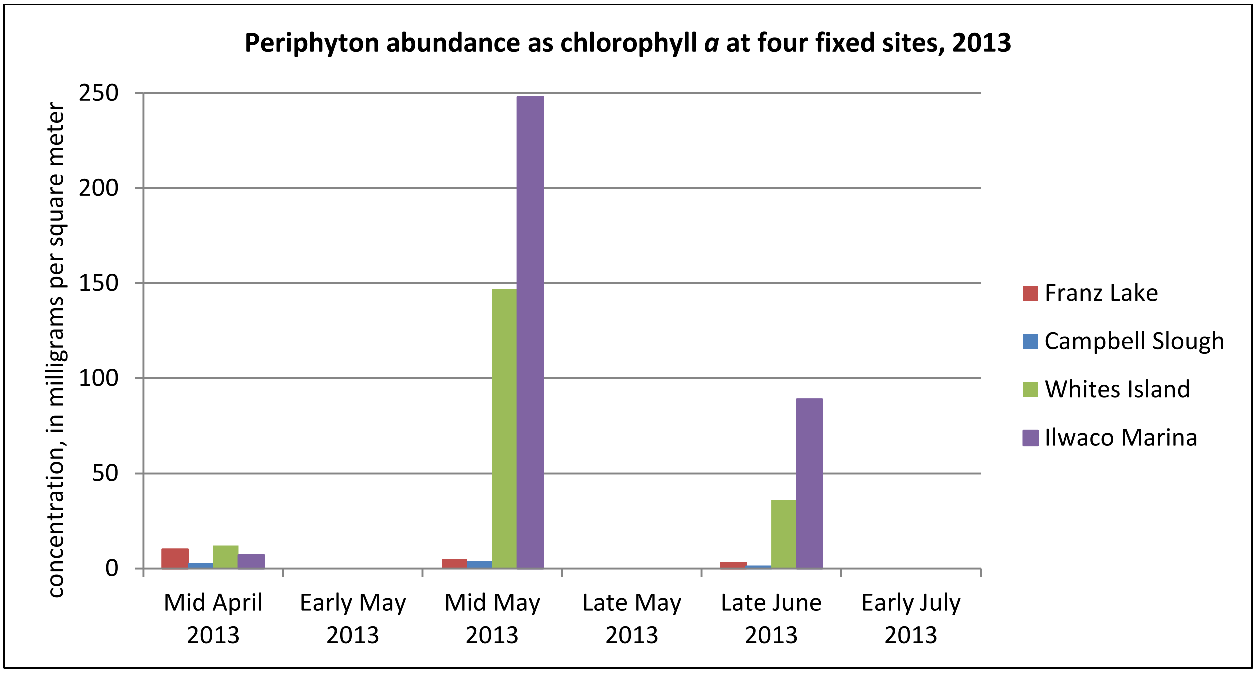


Figure 32. Periphyton abundance as chlorophyll *a* concentrations in 2013.

As in previous years (see Sagar et al. 2012), the total phytoplankton abundance (cells mL^{-1}) was lowest at Ilwaco (Figure 33), although fewer samples from Ilwaco were enumerated microscopically in 2013, so the comparison is limited. The total abundance of phytoplankton was greatest at Campbell Slough, where the greatest proportion of non-diatom species (especially cyanobacteria and green algae) was observed (Figure 33). With the exception of Ilwaco, the other sites showed a decrease in total phytoplankton abundance during May, which coincided with the spring freshet. The decline in phytoplankton abundance during this time was more dramatic at Campbell Slough and Franz Lake Slough compared to Whites Island.

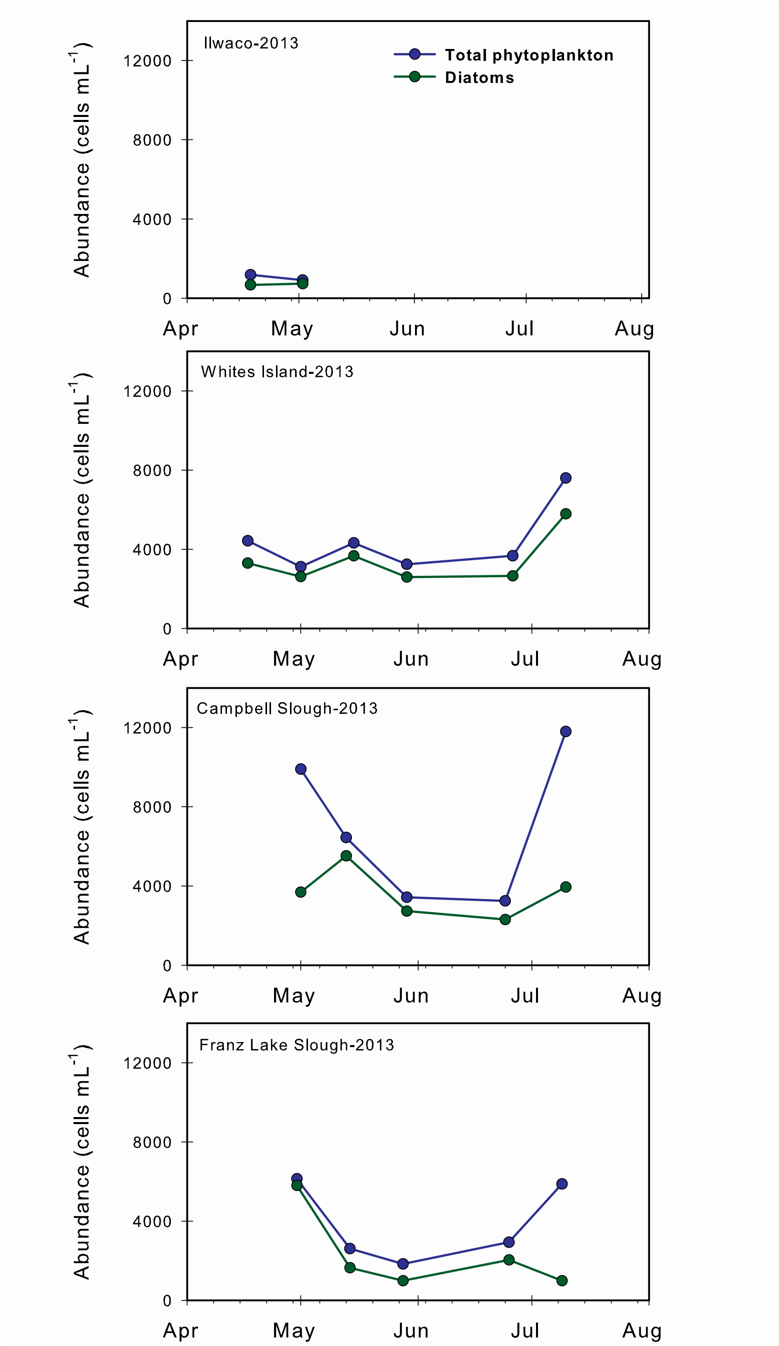


Figure 33. Total phytoplankton abundance (blue filled circles joined by lines) and diatom abundance (green circles joined by lines) at Ilwaco (Reach A), Whites Island (Reach C), Campbell Slough (Reach F), and Franz Lake Slough (Reach H) in 2013.

Vegetation

The above ground biomass estimates for the emergent wetland vegetation found in the low and high marsh strata are provided in Table 19 and in the SAV strata in Table 20. Temporal trends are difficult to discern because of limited sampling over the three year time period. Comparisons between all three years are limited to three sites for the high marsh strata, one site for the low marsh strata, and two sites for the SAV strata. This limited data can be used to make broad generalizations that should be taken with caution given the limited number of comparisons. Within the high marsh strata, summer biomass was the lowest in 2011, except at the Whites Island site, where it was lowest in 2012. The low marsh strata have too few observations among sites and between years to be meaningful. The results from the two sites with samples from the SAV strata in all three years indicate a decline over the 3-year period. However, two other sites measured in the last two years increased over that time period.

Table 19. Aboveground biomass of emergent wetland vegetation from high marsh and low marsh strata. The difference in biomass for each year is the summer biomass minus the winter biomass in g/m^2 and is an estimate of the amount of detritus production each year. Sites are ordered by distance from the mouth of the River.

		Biomass Dry Wt. (g/m^2)											
		Summer 2011		Winter 2012		Difference	Summer 2012		Winter 2013		Difference	Summer 2013	
Site	Strata	n	Avg Dry Wt. (SD)	n	Avg Dry Wt. (SD)		n	Avg Dry Wt. (SD)	n	Avg Dry Wt. (SD)		n	Avg Dry Wt. (SD)
Ilwaco (BBM)	high marsh	7	976 (421)	7	385 (133)	591.5	10	1175 (257)	10	254 (135)	921	10	1141 (429)
Secret River (SRM)	high marsh						5	1443 (148)	5	194 (210)	1248	9	1062 (386)
Welch Island (WI2)	high marsh						5	1141 (322)	9	272 (122)	870	9	1361 (647)
Whites Island (WHC)	high marsh	6	1152 (844)	5	517 (327)	635.2	8	740 (623)	8	346 (258)	393	9	1359 (834)
Campbell Slough (CS1)	high marsh	3	410 (356)	4	101 (64)	309.3						6	434 (67)
Franz Lake (FLM)	high marsh	8	203 (152)	12	245 (114)	-42.2	7	672 (557)	5	104 (107)	567	9	434 (317)
Ilwaco (BBM)	low marsh	1	24 (NA)										
Secret River (SRM)	low marsh						5	265 (71)	5	15 (15)	250	9	175 (124)
Welch Island (WI2)	low marsh						4	401 (362)					
Whites Island (WHC)	low marsh	2	88 (89)	3	6 (6)	79.0	3	114 (102)	3	10 (15)	104	6	163 (126)
Campbell Slough (CS1)	low marsh	5	278 (151)	4	3 (4)	274.3						11	56 (38)
Franz Lake (FLM)	low marsh			1	66 (NA)				2	30 (24)			

SD = Standard Deviation
NA = Not Applicable

Table 20. Aboveground biomass of submerged aquatic vegetation (SAV). The difference in biomass for each year is the summer biomass minus the winter biomass in g/m^2 and is an estimate of the amount of biomass contributed to the system each year. Sites are ordered by distance from the mouth of the River.

		Biomass Dry Wt. (g/m^2)												
		Summer 2011		Winter 2012		Difference	Summer 2012		Winter 2013		Difference	Summer 2013	
Site	Strata	n	Avg Dry Wt. (SD)	n	Avg Dry Wt. (SD)		n	Avg Dry Wt. (SD)	n	Avg Dry Wt. (SD)		n	Avg Dry Wt. (SD)
Ilwaco (BBM)	SAV	4	82 (91)	4	0.0 (0.0)	82	6	28 (38)	6	0.1 (0.1)	28	6	14 (30)
Secret River (SRM)	SAV						6	30 (12)	6	2.4 (5.1)	28	6	94 (77)
Welch Island (WI2)	SAV						4	98 (62)	4	6.2 (4.2)	92	6	173 (198)
Whites Island (WHC)	SAV	8	49.3 (65.0)	8	0.4 (0.5)	48.9	6	36 (76)	6	0.3 (0.6)	36	6	11 (20)
Campbell Slough (CS1)	SAV	8	0.4 (0.8)	8	0.0 (0.0)	0.4						6	9.3 (23)
Franz Lake (FLM)	SAV						6	4.0 (10)	6	0.0 (0.0)	4.0	6	0.2 (0.4)

Species Composition

Pelagic

The species composition of fluvial phytoplankton was generally dominated by diatoms (Class Bacillariophyceae; Figure 34) at all sites, although the proportion of non-diatom species was greatest at Campbell Slough. Franz Lake Slough also showed lower proportions of diatoms in the summer months compared to Whites Island in Reach B. Another difference among sites was the higher abundance of cyanobacteria species at Campbell Slough, both prior to and especially after, the spring freshet.

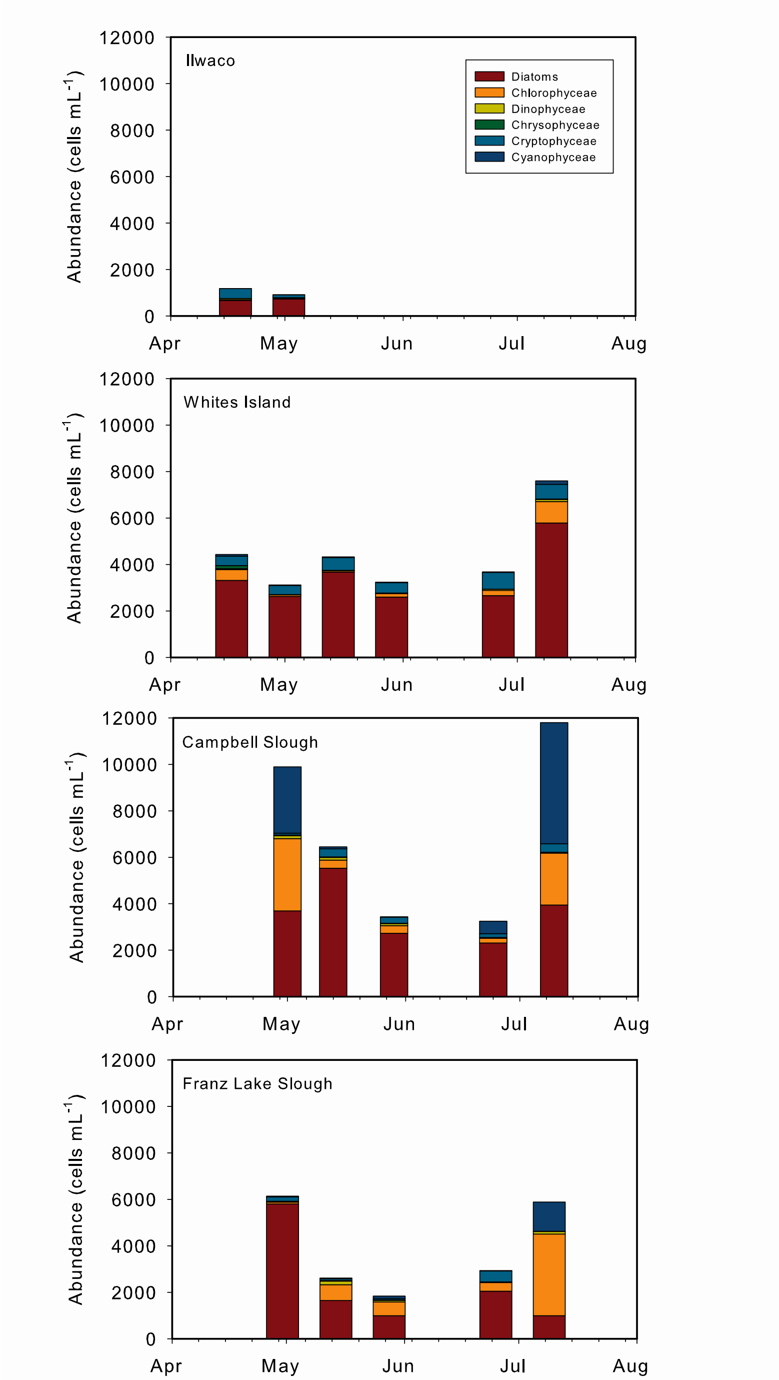


Figure 34. Relative abundance (proportions) of different taxonomic groupings of phytoplankton at Ilwaco (Reach A), Whites Island (Reach C), Campbell Slough (Reach F), and Franz Lake Slough (Reach H).

Vegetation

The samples for the vegetation component of primary production were taken from three primary marsh strata: high marsh, low marsh, and submerged aquatic vegetation (SAV), with 66 percent of the samples further categorized into species specific strata (Table 21). In general, the species comprising the vegetation biomass samples are the dominant species found in the LCRE. Vegetation species assemblages at the six study sites are described in detail in the annual reports from this study (e.g., Sagar et. al. 2013). Although dominant species were noted in many of the samples, frequently the samples were a mix of more than one species. The samples in the non-specific categories were either a mix of many species, with no dominant, or the dominant species were not indicated at the time of sampling. The high percentage of non-specific SAV samples is due to the fact that many of the samples contained no vegetation due to the patchy nature of the SAV in many of the channels.

Table 21. Dominant species in the vegetation biomass samples. The remaining samples that were not categorized by dominant species are lumped into the non-specific categories.

Dominant Species	Common Name	Species Code	Percent of Samples Containing Dominant Species
	Emergent Species		
Carex lyngbyei	Lyngby sedge	CALY	23
Eleocharis palustris	common spike rush	ELPA	5
Phalaris arundinacea	reed canarygrass	PHAR	17
Polygonum amphibium	water smartweed	POAM	3
Sagittaria latifolia	wapato	SALA	6
	Submerged Aquatic Species (SAV)		
Elodea spp.	waterweed	ELSP	4
Potamogeton richardsonii	Richardson's pondweed	PORI	4
Zannichellia palustris	Horned pondweed	ZAPA	5
	Non-specific Categories		
High marsh			5
Low marsh			6
SAV			24

Rates

Net ecosystem metabolism

New data not yet available – will update in next draft

Annual Detrital Contribution

Summer peak biomass is an estimate of the annual primary production at the site (MacDonald 1984). This annual production dies back every year and as it decomposes it becomes detritus, an important component of the juvenile salmonid food web. To estimate the detritus production, the winter standing stock is subtracted from this the summer peak standing stock, providing an estimate of the annual detritus production for the wetland. This value is presented in Table 19 and Table 20 as the difference between Summer and Winter biomass values. Since only two years of data are available at this time, limited temporal patterns are discernable regarding the annual detritus contribution of the marshes. At the Ilwaco (rkm 6) and Franz Lake (rkm 221) sites, annual detritus from the high marsh strata increased between years, while at the Whites Island (rkm72) there was a decrease between years.

Secondary Production

Quantity

The total abundance of zooplankton (excluding microzooplankton < 80 μm) at the six EMP fixed sites spanning Reach A to Reach H (Ilwaco to Franz Lake Slough) are shown in Figure 35. The data show that zooplankton abundances at Ilwaco did not show similar patterns to the other sites. At the Reach B, C, and F sites (Secret River, Welch Island, Whites Island, and Campbell Slough), total abundances of macrozooplankton were higher prior to the freshet than after it. This pattern was more exaggerated at Campbell Slough compared to the other sites. Abundances of zooplankton at Franz Lake Slough declined after mid-June, which represented a slight delay relative to the sites downstream of the Columbia-Willamette confluence (Reaches B-F). Once water levels were reduced relative to freshet conditions, the abundances of zooplankton increased at Whites Island, Campbell Slough, and Franz Lake Slough. The highest abundances of zooplankton were found at Campbell Slough, especially during the summer (post-freshet).

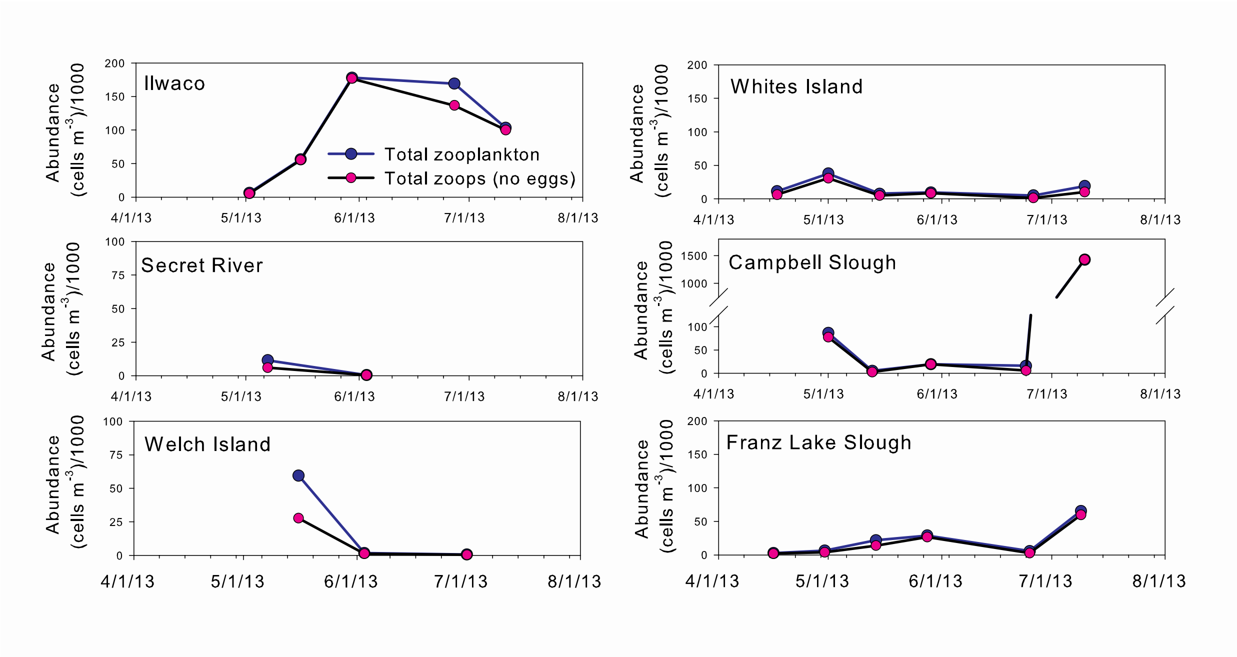


Figure 35. Abundances of zooplankton (total and total minus egg counts) at the six fixed EMP sites. Note the differences in scaling among the different sites that facilitate identification of patterns but obscure the differences in abundance between sites.

Species composition

Ilwaco (Reach A) showed a different pattern in species composition as well as in zooplankton abundance (Figure 36), with copepods dominating the assemblages through most of the time series. At Whites Island, rotifers were present at higher proportions prior to the freshet compared to post-freshet. Rotifers were present at higher proportions in Campbell Slough both before and after the freshet.

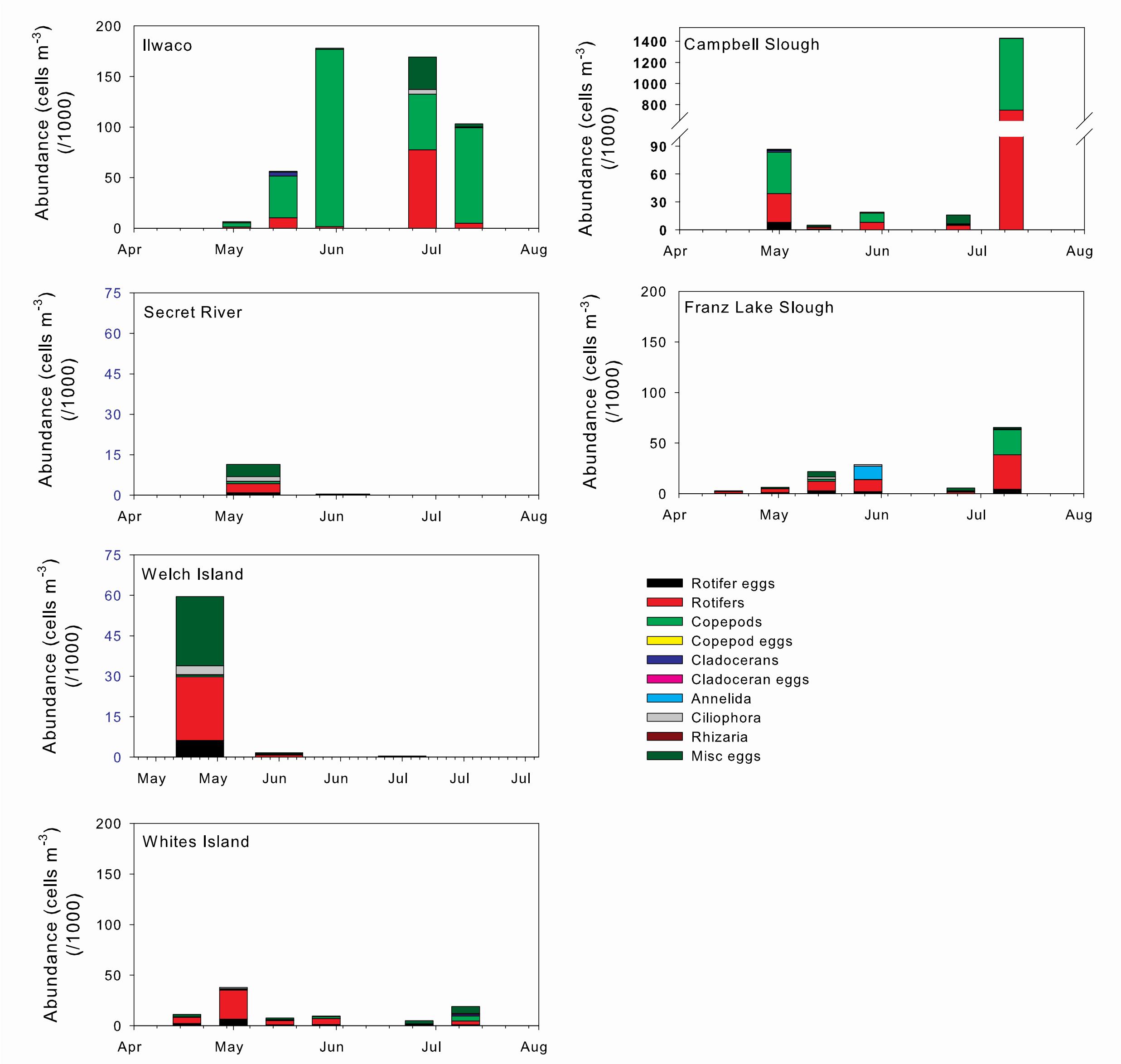


Figure 36. Relative abundances of different zooplankton taxa at the six fixed EMP sites in 2013.

4.4.1 Isotope ratios

Refer to 2013 EMP food web synthesis (Sagar et al. 2014).

4.4.2 Prey availability in emergent vegetation and open water habitats

In 2013, invertebrate tow samples were collected at Secret River, Welch Island, and Campbell Slough. Tow samples were not collected at Ilwaco Slough or Franz Lake in 2013 because of the lack of Chinook salmon at these sites, but data from earlier years are included in these results for comparison.
The diversity of prey available at the 2013 EMP sites included 27 invertebrate groups (referred to as taxonomic units, or TUs) represented across the tows from all sites (Figure 37). These TUs are primarily

orders, but for some taxa that are rare, multiple orders have been grouped together. For example, all Gastropoda (snails and slugs) are considered together rather than by the three individual orders that are present but rare. The overall mean number of TUs collected in tow samples per site per month was 8.4. Diversity was slightly greater in shoreline habitats than in open water habitats, but the different was not statistically significant (mean number of TUs, EV = 8.2, OW = 7.6 p=0.64). Diptera were caught in the majority of the tows (96%) and comprised 16% of the invertebrates caught in all tows in 2013 (n_{tows}=45). Hemiptera, primarily aquatic Corixidae but also some terrestrial bugs, were represented in 80% of tows, and made up 54% of the invertebrates caught in all tows. Hemiptera were especially abundant at Secret River, Welch Island, and Whites Island (Figure 37). Cladocerans and Copepods were also abundant at Campbell Slough, and in Franz Lake in some earlier samplings (Figure 37). Combined, these two groups were present in 80% of all tows and made up 14% of all invertebrates collected.

At Secret River, Welch Island, and Campbell Slough, invertebrate composition of the 2013 tows generally fit well with patterns observed in previous years. At Campbell Slough, however, the 2013 tows contained a higher percentage of Hemipterans and lower percentage of Dipterans than usual. In 2013, as in past years, invertebrate prey densities were much higher, from approximately 5 times to 30 times greater, in emergent vegetation tows than in open water tows at all sites (Figure 38). Invertebrate prey densities were somewhat higher at Whites Island than in previous years, but for other sites values were comparable to those previously observed.

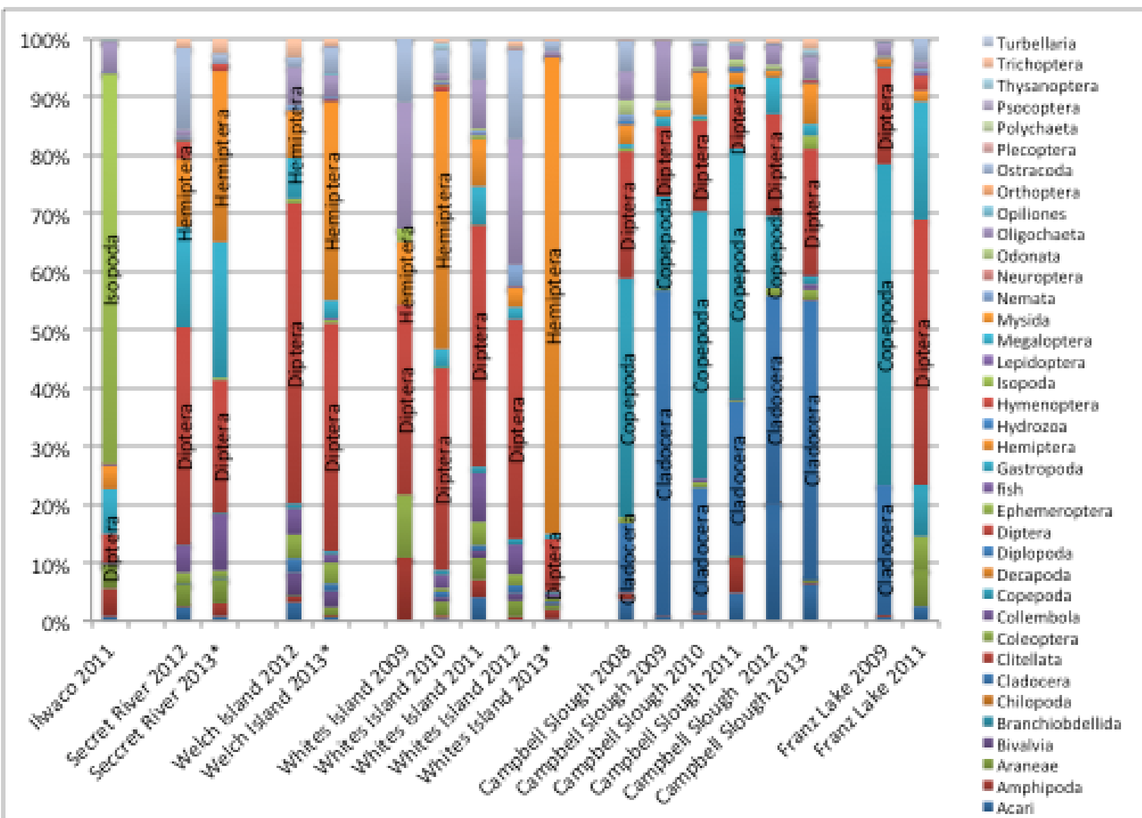


Figure 37. Composition (mean % by taxonomic unit) of invertebrates caught in all tows collected per site at sites sampled in 2013. Some taxa are labeled in each column to help orient the reader.

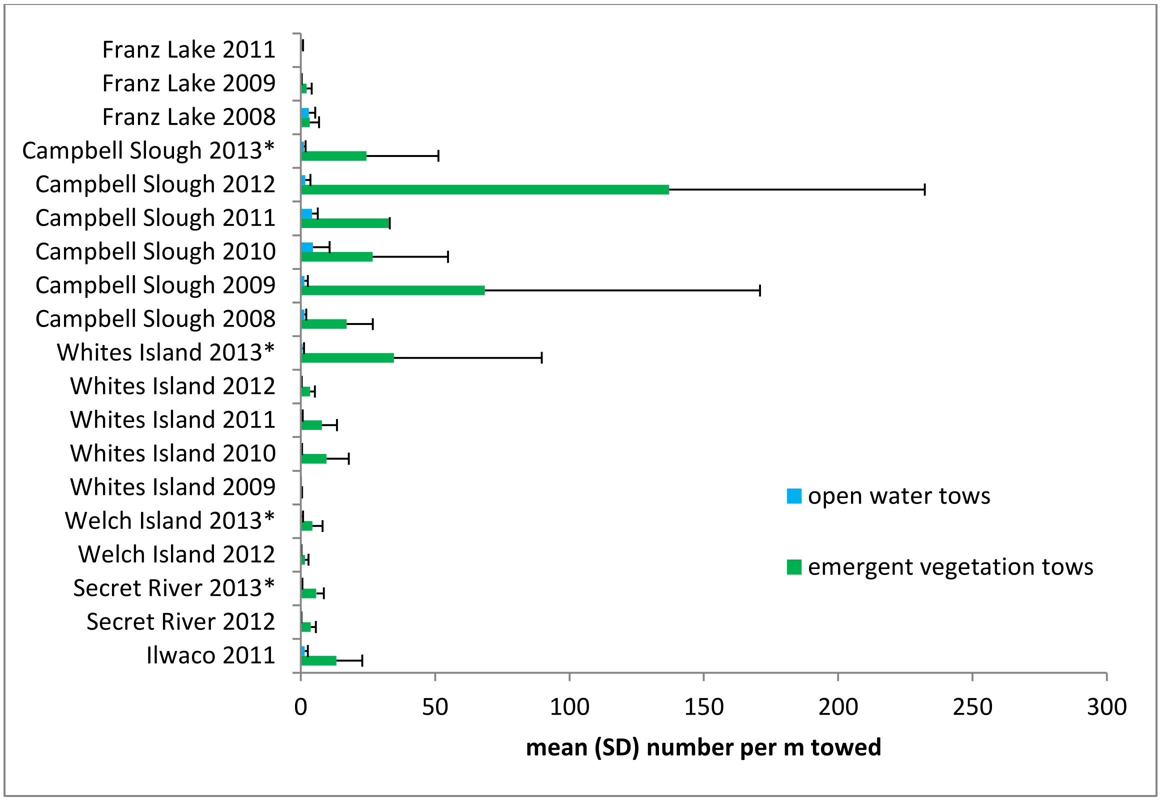


Figure 38. Mean (SD) number of invertebrates captured at each of the EMP trend sites by year. The 2013 samples are highlighted with asterisks (*). No data were collected at Franz Lake or Ilwaco in 2013 because too few salmon were present to obtain co-located diet data. Note the extreme variation within and among sites and the difference between abundances captured in the emergent vegetation tows and the open water tows.

4.5 Fish Use

4.5.1 Fish Community Composition

In 2013, fish communities at sites in Reach A, B, and C were dominated by three-spined stickleback (*Gasterosteus aculeatus*), which accounted for 90-95% of the total catch (Figure 39). Other species present at Ilwaco Slough in Reach A included two marine species, shiner perch (*Cymatogaster aggregata*) and Pacific staghorn sculpin (*Leptocottus armatus*), which together accounted for about 10% of the total catch. Juvenile salmon made up only 0.2% of the catch. At Secret River, Welch Island and Whites Island, other species present in addition to stickleback included juvenile Chinook salmon, chiselmouth (*Acrocheilus alutaceus*), various sculpins (*Cottus* sp.), and starry flounder (*Platichthys stellatus*), accounting for 3-6%. Species assemblages at Campbell Slough in Reach F and Franz Lake in Reach H were much more diverse (Figure 39). Prominent species at Campbell Slough included yellow bullhead (*Ameiurus natalis*), carp (*Cyprinus carpio*), tui chub (*Gila bicolor*), yellow perch (*Perca flavescens*), and American shad (Alosa sapidissima) as well as three-spined stickleback. At Franz Lake, the most abundant species were carp and chiselmouth.

Fish community composition in 2013 sampling was comparable our previous results from 2008-2012 (Figure 39). The dominance of stickleback at Ilwaco Slough, Secret River, Welch Island, and Whites Island observed in 2013 was consistent with earlier findings, as was the presence of saltwater species absent from other sites at Ilwaco Slough. The greater abundance of species such as bullhead, carp, tui chub, yellow perch, and American shad, at Campbell Slough, and the dominance of carp and chiselmouth at Franz Lake, were also consistent with previous sampling.

Species diversity (H') at the sampling sites in 2013 was relatively low at the sites in Reaches A-C, with substantially higher values at Campbell Slough in Reach F and Franz Lake in Reach H (Figure 40). These patterns are typical of those observed in previous sampling. However, diversity was significantly lower in 2013 at Ilwaco Slough and Franz Lake in comparison with other sampling years (2011-2012 for Ilwaco Slough, 2008-2012 for Franz Lake). Similarly, species richness (the number of species) at the sampling sites in 2013 was relatively low at the sites in Reaches A-C, with higher values at Campbell Slough in Reach F and Franz Lake in Reach H. These patterns are typical of those observed in previous samplings (Figure 40). However, at Franz Lake, species richness was significantly lower in 2013 ($p = 0.0033$) than the average for previous sampling years.

In 2013, percentages on non-native species and known salmon predators (largemouth and smallmouth bass, northern pike minnow, and walleye) in catches (Figure 41) were extremely low at Ilwaco Slough, Secret River, Welch Island, and Whites Island, with the percentage of non-native species ranging from 0.02 to 1.3% and the percentage of predators ranging from 0 to 0.12%. The percentages of non-native species and predators were substantially higher at Campbell Slough and Franz Lake. At these two sites, the percentage of non-native species ranging from 28 to 55% and the percentage of predators ranged from 2.5 to 6.4%. At all sites, the percentages of non-native and predatory fish species observed in 2013 were comparable to percentages observed in previous years (Figure 41).

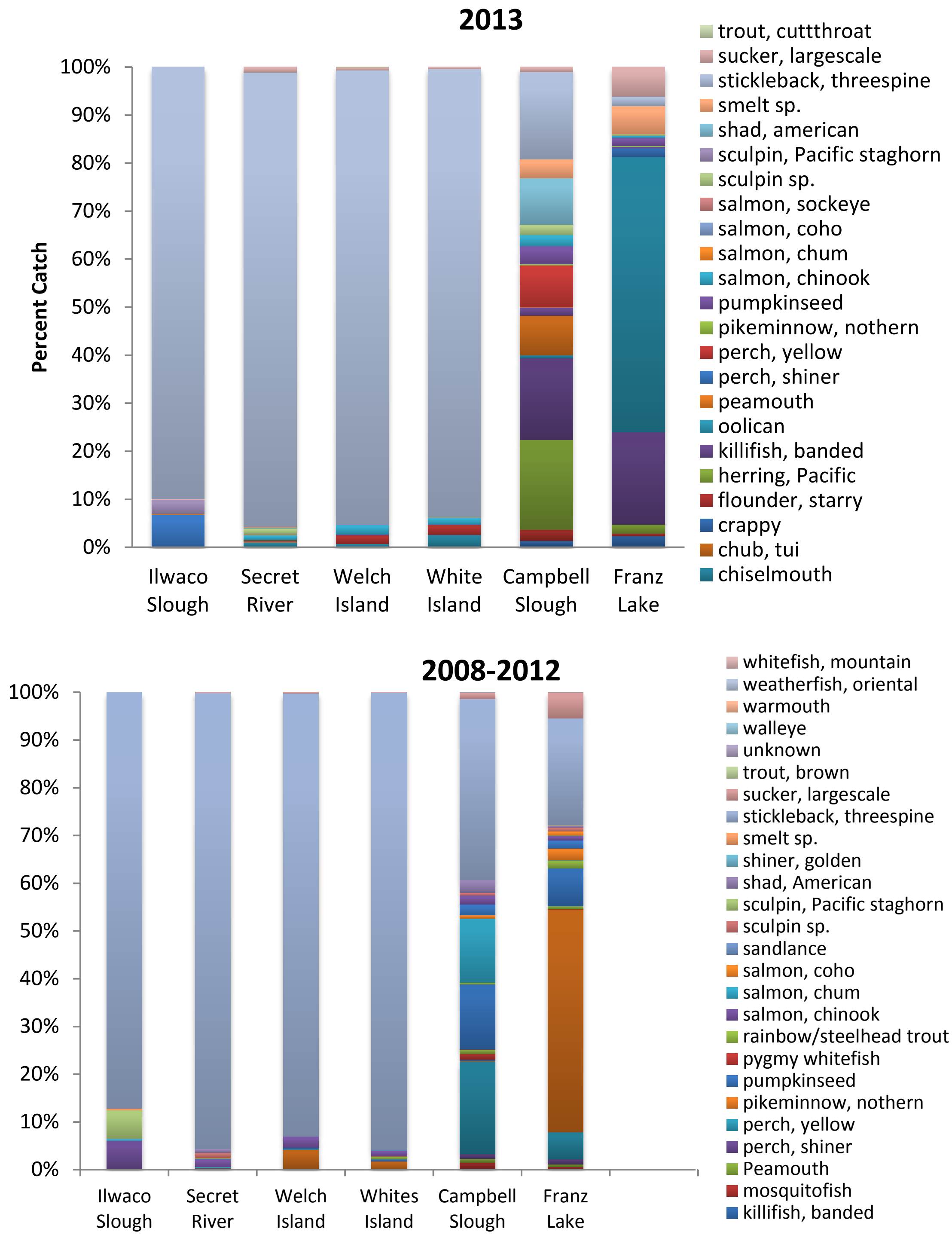


Figure 39. Fish community composition at EMP sites samples in a) 2013, and in b) previous sampling years 2008-2012).

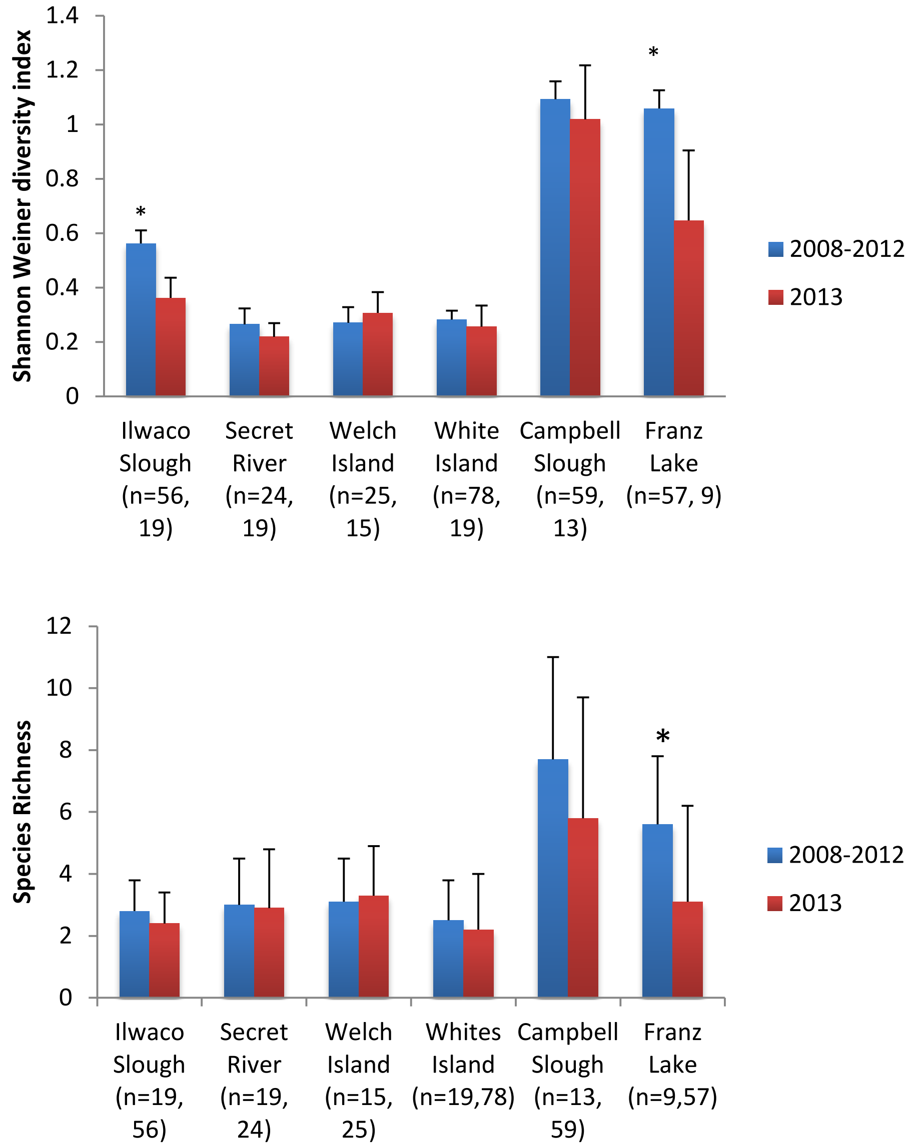


Figure 40. a) Shannon-Weiner diversity index and b) species richness (number of species) in mean values per tow at the EMP sampling sites in 2013 as compared to previous sampling (2008-2012). Asterisk indicates that the value for that site in 2013 is significantly different ($p < 0.05$) than the average value over the previous years of sampling.

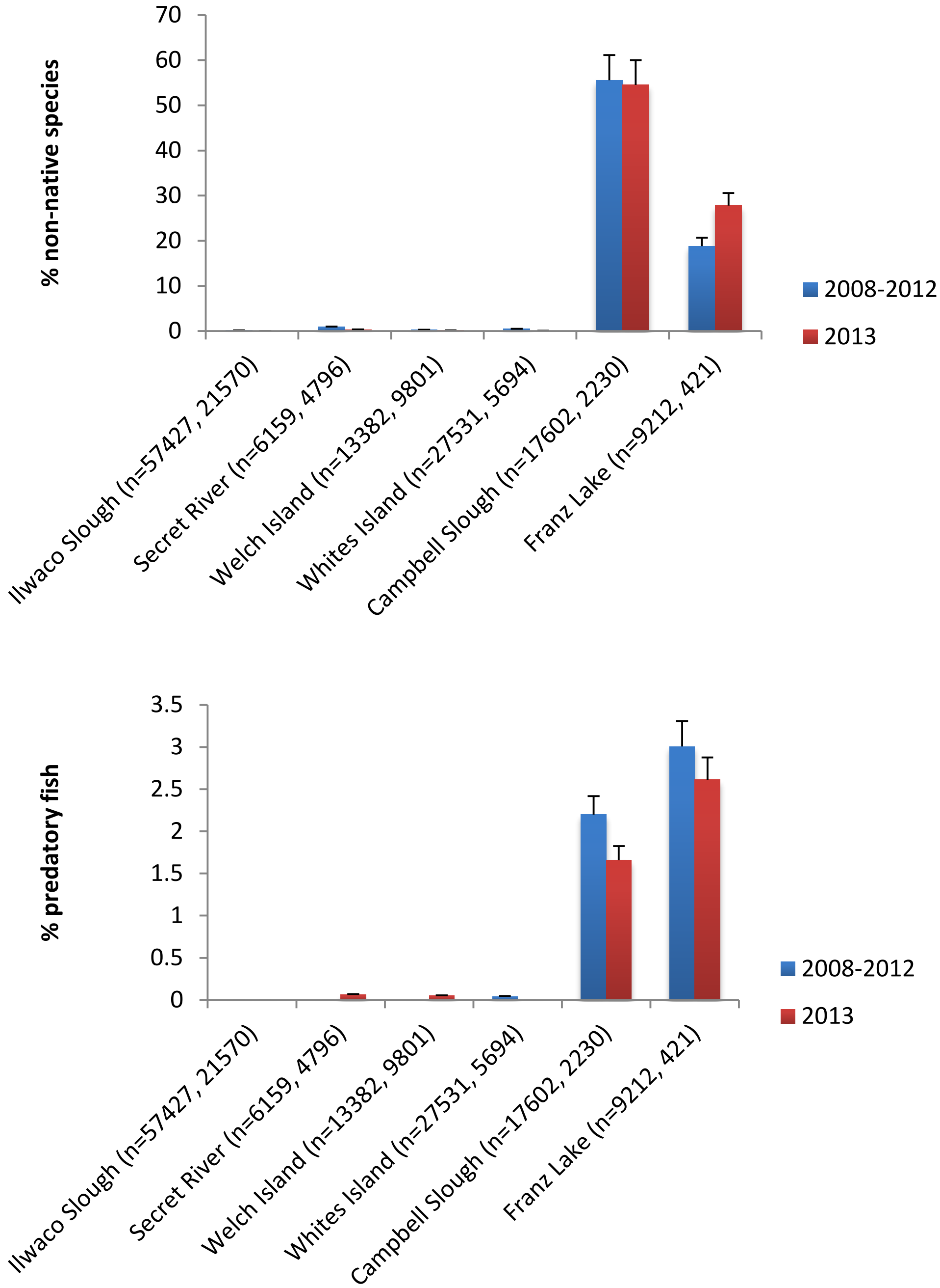


Figure 41. Percentages, based on total number of fish caught of a) non-native fish species and b) % of fish that are recognized predators of juvenile salmon (smallmouth and largemouth bass, northern pikeminnow, walleye) in 2013 as compared to 2008-2012.

4.5.1.1 *Salmon Species Composition*

In 2013, salmon species composition varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 42). At Ilwaco Slough, in Reach A, chum salmon were the only salmon species collected, accounting for 100% of the salmon catch. Chinook salmon were the dominant species at Secret River and Welch Island in Reach B, Whites Island in Reach C, and Campbell Slough in Reach F, making up 78% to 100% of catches. At all of the sites, unmarked, presumably wild fish were more abundant than marked hatchery fish. At Secret River, Welch Island, and Whites Island, 75-98% of Chinook salmon were unmarked, while at Campbell Slough, 58% were unmarked. In addition to Chinook salmon, small numbers of chum salmon were found at Secret River and Welsh Island, as well as some unmarked coho salmon at Secret River and one cutthroat trout at Welch Island. Sockeye salmon, though nowhere abundant, were observed for the first time in 2013, at Whites Island and Secret River. At Franz Lake, only two unmarked Chinook and two unmarked coho salmon were collected in 2013; sampling at the site was limited to collections in March, August, September, and December. The patterns of salmon species composition observed at the EMP sites in 2013 are very similar to those observed in previous sampling years (Figure 42), with chum salmon dominating at Ilwaco Slough, unmarked Chinook salmon dominating at Secret River, Welch Island, and Whites Island, and comparable proportions of marked and unmarked Chinook salmon at Campbell Slough. Typically Franz Lake supports a much more diverse mixture of salmon species, including both Chinook and coho salmon, than was apparent in 2013, but an accurate comparison is not possible because of the very limited sampling in 2013.

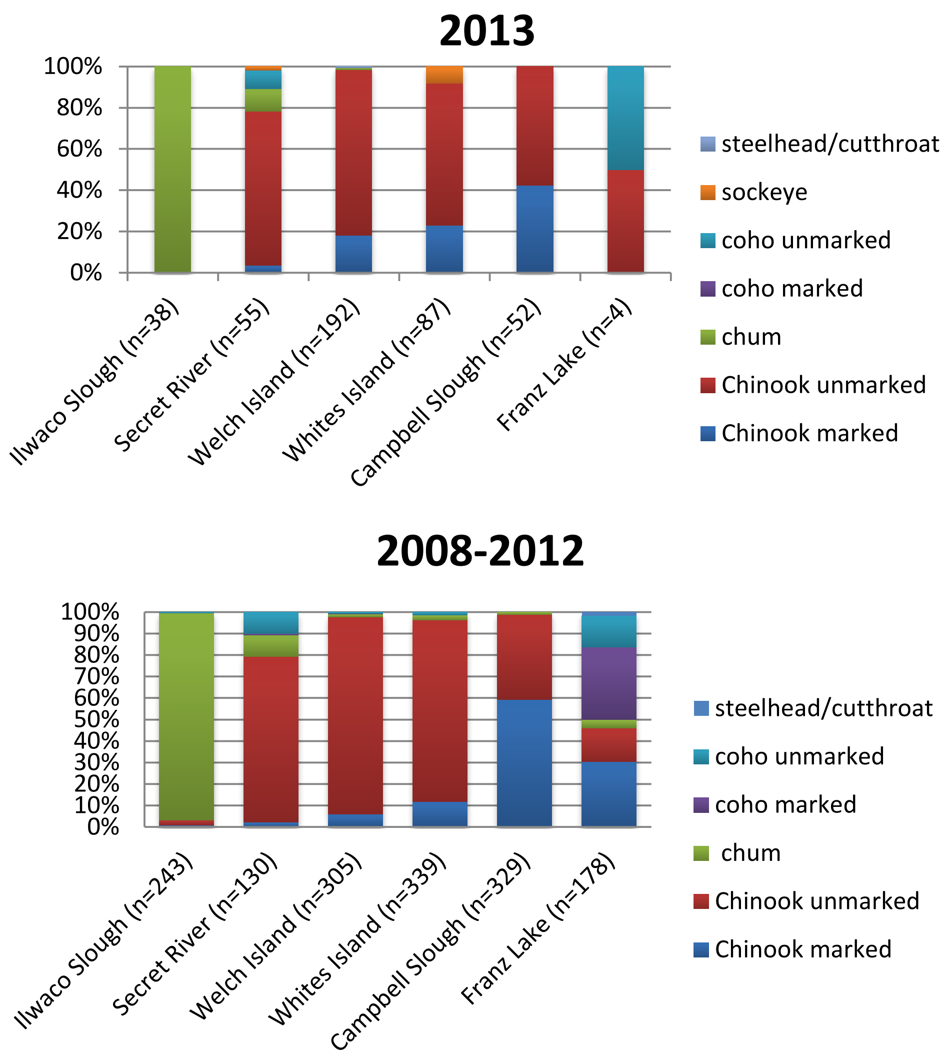


Figure 42. Percentages of salmon species collected at EMP sampling sites in 2013, as compared to percentages collected in previous sampling years.

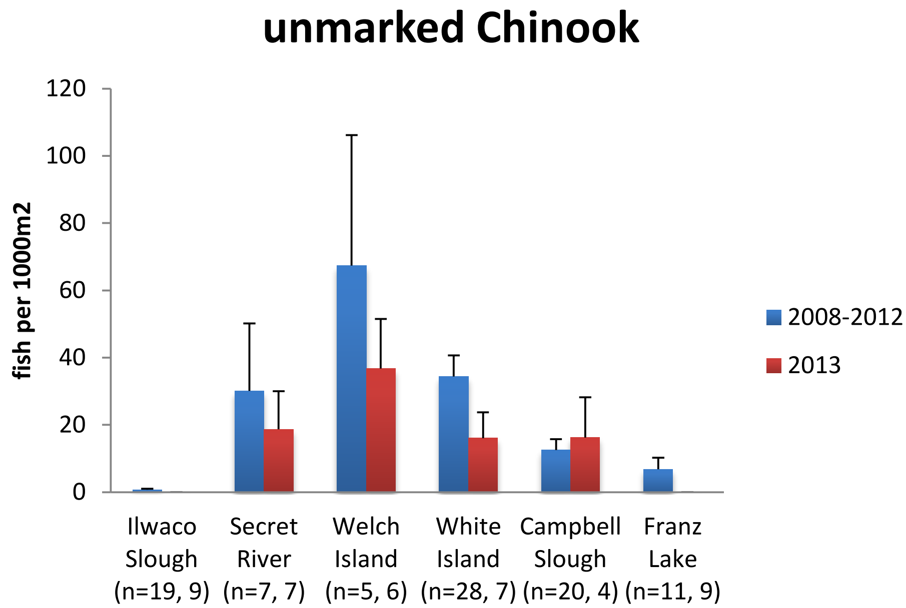
4.5.1.2 *Salmon Density*

Chinook salmon. Chinook salmon densities during the peak migration period (April-May) are shown in Figure 43. Franz Lake could not be sampled during these months due to high water so 2013 density for this site is not included. The density of unmarked Chinook salmon was highest at Welch Island, and lowest at Ilwaco Slough, with intermediate values at the other sites. In comparison with previous years, densities of unmarked Chinook salmon in 2013 tended to be lower at most of the sampling sites. The exception was Campbell Slough, where densities in 2013 were very similar to average levels observed previously. The density of marked Chinook salmon in 2013 (Figure 43) tended to increase with increasing distance from the mouth of the river, with highest values at Campbell Slough. At Ilwaco Slough, Secret River, and Welch Island, densities of marked Chinook were similar in 2013 to average values observed in the past. However, at Whites Island densities were higher, and at Campbell Slough lower, than average densities over previous years.

Coho salmon. As coho salmon are found throughout the sampling season, densities averaged over all sampling months are shown for this species. No marked coho salmon were caught in 2013, and unmarked coho were caught only at Secret River in Reach B. Consequently, coho salmon densities were lower in 2013 as compared to other sampling years (Figure 44). This was true even at Secret River, where some coho were collected. Exceptions are Campbell Slough, where no coho salmon have been caught in any sampling period, and Whites Island, where unmarked coho salmon have not been caught in any sampling period.

Chum salmon. Densities of chum salmon were quite low at most of the EMP sites in 2013, similar to other sampling years (Figure 45). However, relatively high densities of chum were observed at Ilwaco Slough in Reach A. In comparison to the average chum density over the past sampling years, the density in 2013 was somewhat higher.

Other species. The two other salmonid species found at the EMP sites in 2013 were sockeye salmon and cutthroat trout. Sockeye were found only at Whites Island and Secret River at average densities of 1.32 and 0.21 fish per 1000 m^2, respectively. Trout density was also very low, 0.27 fish per 1000 m2 at Welch Island, the only site where any trout were found in 2013. In past years, trout were also observed at Franz Lake, also at a low density of 0.26 fish per 1000 m^2.



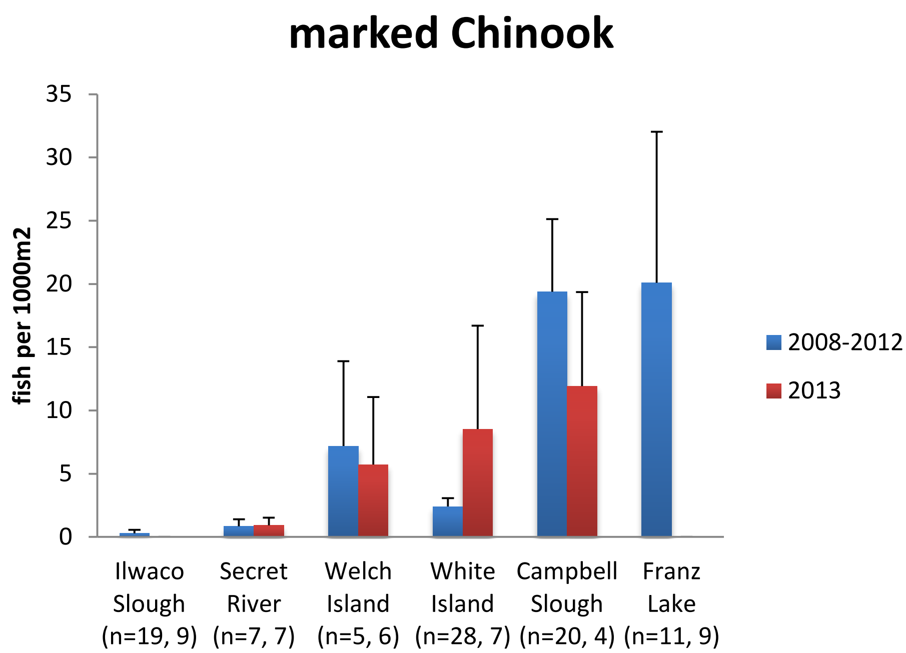
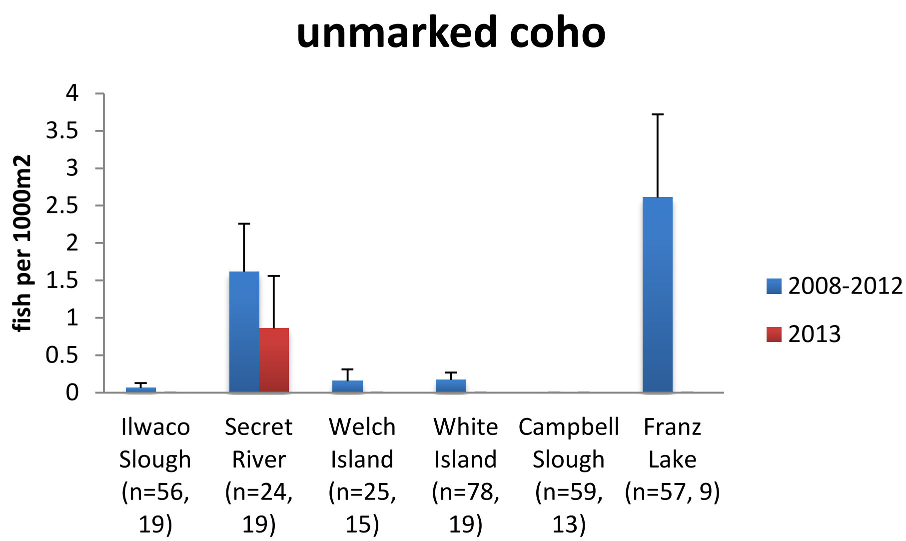


Figure 43. Densities (fish/1000 m^2) of a) unmarked and b) marked Chinook salmon at the 2013 EMP sampling sites as compared to average densities for the other years sampled. Only sampling events in the peak migration period (April-May) are included in these graphs. Values of n refer to numbers of beach seine sets in the earlier sampling period (2008-2012) and in 2013.



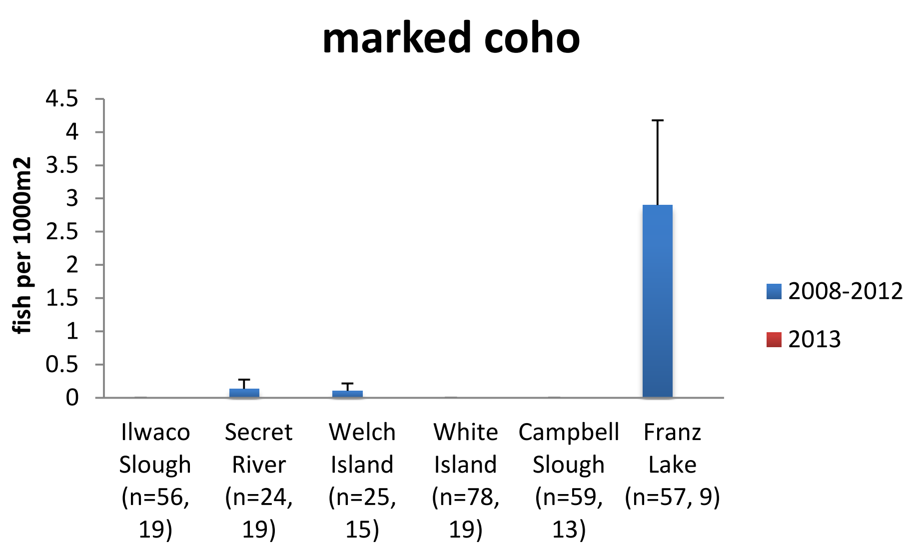


Figure 44. Densities (fish/1000 m2) of a) unmarked and b) marked coho salmon at the 2013 EMP sampling sites as compared to average densities for the other years sampled. Values of n refer to numbers of beach seine sets in the earlier sampling period (2008-2012) and in 2013. No marked coho salmon were caught in 2013.

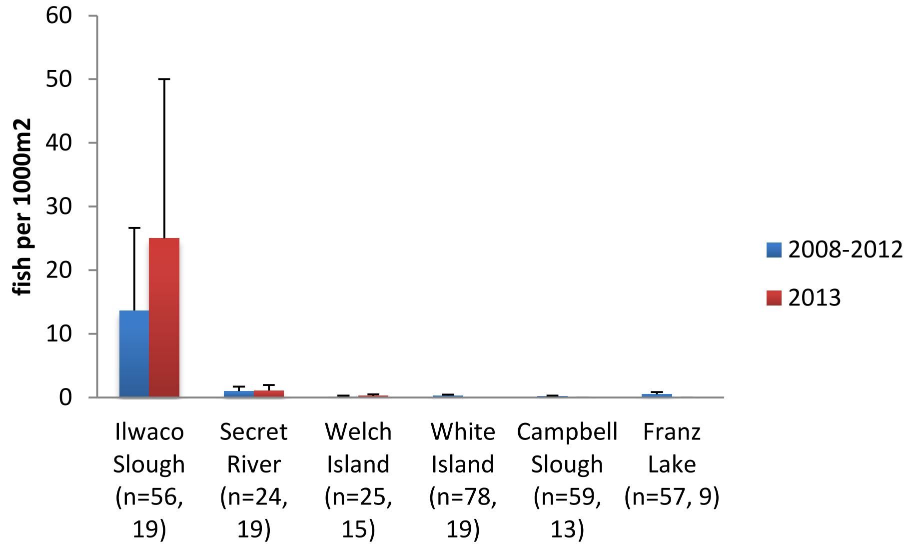


Figure 45. Densities (fish/1000 m^2) of chum salmon at the 2013 EMP sampling sites as compared to average densities for the other years sampled. Values of n refer to numbers of beach seine sets in the earlier sampling period (2008-2012) and in 2013.

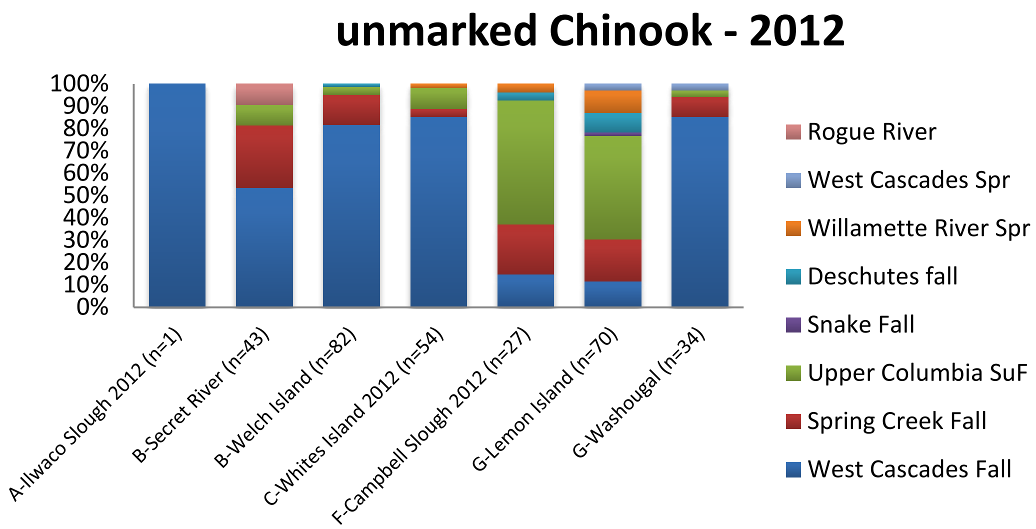
4.5.2 Salmon Metrics

4.5.2.1 *Genetic Stock Identification*

In this report we present the genetic stock data from 2012, which were not available for the 2013 annual report. Genetic samples from 2013 are now being analyzed. In 2012, genetics data were collected from Chinook salmon from three new sites, Secret River and Welch Island in Reach B, and Lemon Island and Washougal Wetland in Reach G. Among the unmarked fish (Figure 46), West Cascades fall Chinook were the most abundant stock at most sites. However, at Campbell Slough and Lemon Island, West Cascades fall Chinook were much less abundant, and the dominant stock was Upper Columbia summer/fall Chinook. Several other interior Columbia stocks were also found at Lemon Island, including Snake River and Deschutes River fall Chinook, as well as Willamette River spring Chinook. Rogue River Chinook were observed at the Secret River site. The stock composition patterns observed at Whites Island and Campbell Slough in 2012 were very similar to those found in previous sampling, with dominance of West Cascades fall Chinook at Whites Island and a more diverse assemblage of stocks at Campbell Slough (Figure 46). As only two fish have been analyzed so far from Ilwaco Slough, no real comparisons can be made, although the fish sampled in 2011 was a Spring Creek fall Chinook, while the fish sampled in 2012 was a West Cascades fall Chinook.

Among the marked fish sampled in 2012 (Figure 47), Spring Creek Group fall and West Cascades fall Chinook made up the majority of fish sampled at most sites. Lemon Island again was somewhat unusual in that a relatively high percentage of marked Upper Columbia summer/fall Chinook were found at this site. Other stocks observed in small numbers included West Cascades spring Chinook and Willamette River spring Chinook. The stock composition observed at Whites Island in 2013 was very similar to that of previous years (Figure 47). At Campbell Slough, the 2012 pattern was somewhat different as compared to the overall pattern based on sampling conducted from 2007 to 2011, as West Cascades fall Chinook were the most abundant stock in 2012, whereas Spring Creek fall Chinook were most abundant in previous samplings (Figure 47).

For the first time in 2012, we collected genetic stock information from Chinook salmon sampled in the fall and winter, and so have a more complete picture of seasonal patterns in stock habitat occurrence (Figure 48). For unmarked fish, these results showed a consistent presence of West Cascades fall Chinook from February through August. Spring Creek Group fall Chinook were most abundant early in the season, in February and March, then declined through the spring, summer and fall, whereas Upper Columbia summer/fall Chinook were less abundant from February through April, and found in the greatest number during the summer months. Too few salmon were found in September and October for genetic stock analyses to be performed, but in November and December, spring Chinook stocks (, West Cascades spring Chinook and Willamette River spring Chinook) dominated. Marked fish were present primarily at the sampling sites from April through August. In April and May, the majority of marked fish were spring Creek fall Chinook, but through later spring and summer, these fish became less abundant and West Cascades fall Chinook predominated. Some Upper Willamette Chinook were observed in April and May, and Upper Columbia summer/fall Chinook in June and July. Only two marked fish were analyzed from the fall and winter months (October and February) and both of these were West Cascades fall Chinook.



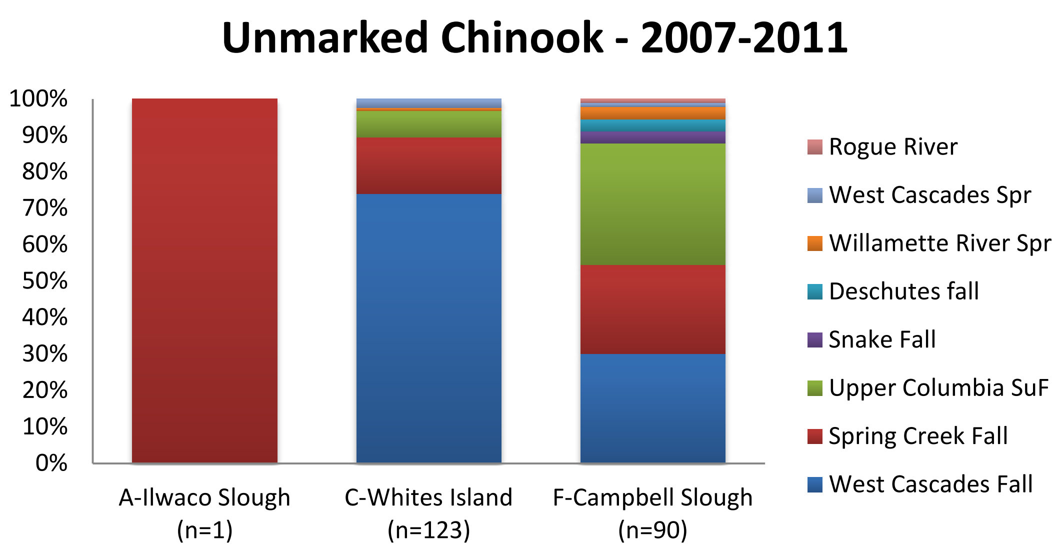
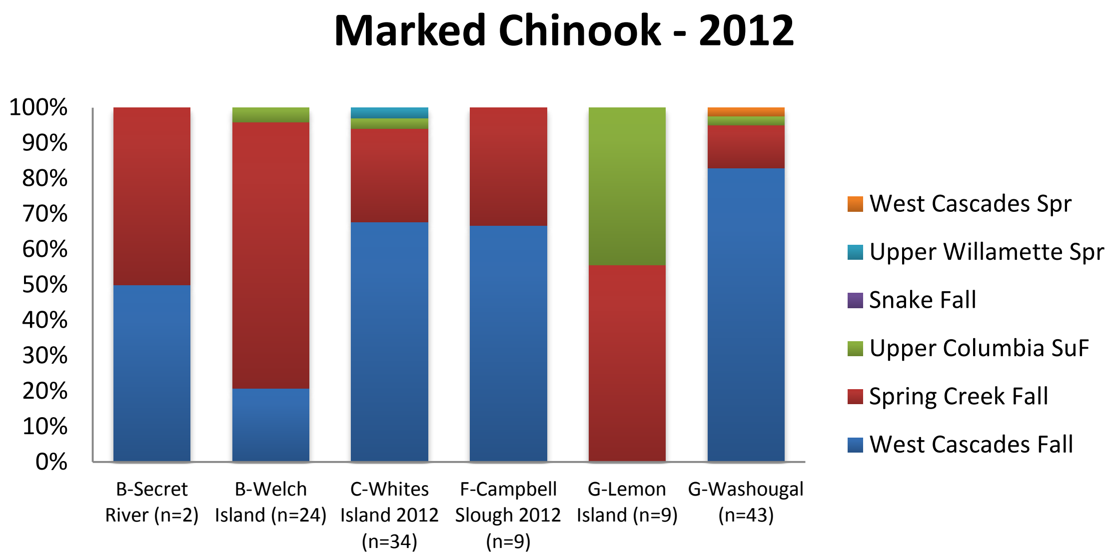


Figure 46. Stock distribution of unmarked Chinook salmon at EMP trend sites in 2012 as compared to 2007-2011.



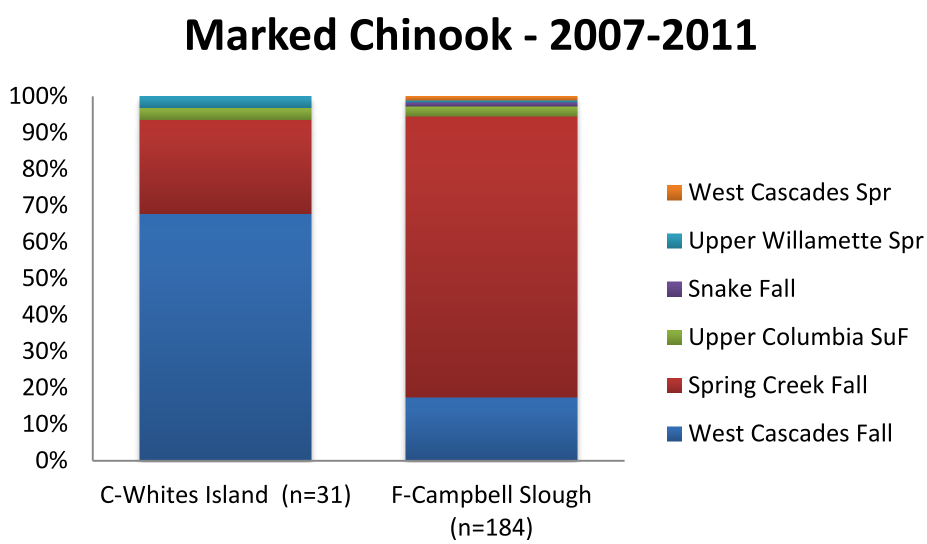


Figure 47. Stock composition of marked Chinook salmon at EMP trend sites in 2012 as compared to 2007-2011.

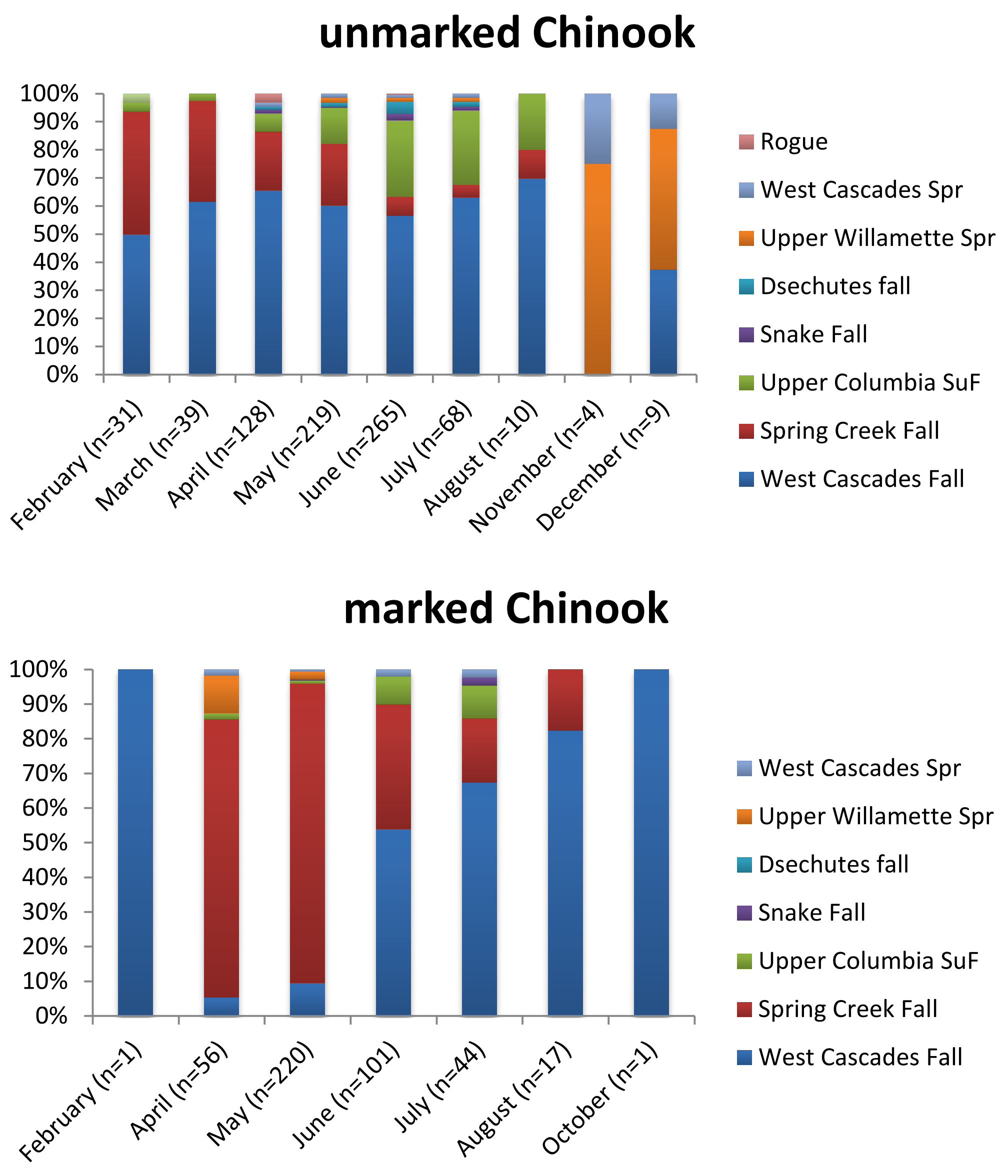


Figure 48. Stock composition of unmarked and marked Chinook salmon by month at EMP sampling sites.

4.5.2.2 *Salmon Size and Condition*

Chinook salmon

Length, weight, and condition factor. Length, weight, and condition of unmarked and marked juvenile Chinook salmon by month and sampling site for 2013 are shown in Figure 49 and Figure 50. For the umarked fish, length varied significantly with both month and site ($p < 0.0001$ for both factors). Length was smaller in the month of March ($p < 0.0001$) and larger in the month of June ($p = 0.0001$) than other months. Additionally, fish tended to be larger at Campbell Slough ($p < 0.0001$) and smaller at Welch Island ($p < 0.0001$) than other sites. Weight showed a similar pattern, varying significantly by both month and site ($p < 0.0001$ for both factors). Weight was lower in the month of March ($p < 0.0001$) and larger in the month of June ($p = 0.0012$) than other months. Additionally, fish tended to weigh more at Campbell Slough ($p < 0.0001$) and less at Welch Island ($p < 0.0001$) than other sites. Condition factor also varied with both month ($p < 0.0001$) and site ($p = 0.0127$). However, the pattern of variation was somwhat different. Condition factor was higher at Secret River ($p = 0.0030$) than at other sites, and lower in May ($p = 0.0451$) than in other months. For the marked fish, length and weight showed no significant variation by sampling month ($0.1415 < p < 0.4461$) but fish were signifcnatly larger and heavier at Campbell Slough ($0.0016 < p < 0.0043$) and smaller and lighter at Welch Island ($0.0004 < p < 0.0043$) compared to the overall average. No significant diffences in condition were found for either month ($p = 0.0825$) or site ($p = 0.7299$).

In comparison with early sampling years, the length, weight and condition of unmarked Chinook in 2013, showed similar patterns, with largest fish tending to occur at Campbell Slough (Figure 51). However, a few differences were apparent. Fish length was significantly larger at Welch Island ($p < 0.0001$) and Campbell Slough ($p = 0.0298$) in 2013 than in earlier years. Fish weight was significantly higher at Welch Island in 2013 ($p = 0.0371$). Condition factor was significantly higher at both Secret River ($p = 0.001$) and Welch Island ($p = 0.0003$) in 2013 than the average for earlier years of sampling. In comparison with early sampling years, the length and weight of marked Chinook in 2013 tended to be lower (Figure 52). Fish length was significantly lower at Welch Island ($p = 0.0071$) and Campbell Slough ($p < 0.0001$) in 2013 than in earlier years. Fish weight was also significantly lower in 2013 at Campbell Slough ($p = 0.0146$). No significant differences were observed between condition factor in 2013 and the earlier years of sampling.

Size class distribution. In 2013, no unmarked juvenile Chinook salmon were caught at Ilwaco Slough, but at Secret River and Welch Island, the majority of fish (83-84%) were fry, and the rest were fingerlings; no unmarked yearlings were caught at any sites in 2013 (Figure 53). At Whites Island, proportions of fry and fingerlings were comparable, with fry account for 62% and fingerlings for 38% of unmarked Chinook. At Campbell Slough and Franz Lake, too few unmarked Chinook were caught to accurately assess size class distribution, although the data suggest a shift to the larger size class at Campbell Slough. Only seven unmarked Chinook were caught at Campbell Slough, all of which were fingerlings. At Franz Lake, only one unmarked Chinook salmon was caught, and it was a fry.The patterns observed in 2013 were similar those seen in earlier years (Figure 53). Proportions of fry and fingerlings were almost identical at Secret River, Welch Island, and Whites Island. At Ilwaco Slough, fry also predominated. At Campbell Slough, fingerlings made up 65% of unmarked Chinook, lower than the 100% seen in 2013, but indicating a consistent trend toward larger fish in comparison to downstream sites. At Franz Lake, fry were the dominant life stage, making up 80% of unmarked Chinook captured. Thus the fact that the single Chinook caught at that site in 2013 was a fry is consistent with earlier sampling. In 2013, very few marked Chinook salmon were caught at any of the sampling sites with the exception of Campbell Slough. Of those collected, however, all were fingerlings, with the exception of a single fish, captured at Welch Island (Figure 54). In past years, the overhwelming majority of marked Chinook salmon have also been in the fingerlings, with other size classes only rarely represented (Figure 54).

Coho salmon

Length, weight, and condition factor. No marked coho salmon were caught in 2013, and Secret River was the only site where unmarked coho salmon were caught. Length, weight, and condition of unmarked juvenile coho salmon from Secret River in 2013 as compared to 2012 are shown in Figure 55, along with values for coho collected from other sites between 2008 and and 2012 for comparison. Both length and weight tended to be higher in unmarked coho collected from Secret River in 2013 than in 2013, but the differences were not statistically significant. However, In comparison with 2012, condition factor was significantly higher in 2013 than in 2013 (p = 0.0451).

Size class distribution. In 2013, coho salmon collected from Secret River were from a variety of size classes, with fry, fingerlings, and yearlings all observed (Figure 56). In comparison to 2012, a wider range of size classes was collected in 2013, as in 2012, only fry and yearling size classes were found.

Chum salmon

Length, weight, and condition factor. Chum salmon were caught at Ilwaco Slough, Welch Island, and Secret River in 2013. Length, weight, and condition of chum salmon these sites in 2013 as compared to 2012 are shown in Figure 57, along with values for chum collected from other sites between 2008 and and 2012 for comparison. As all chum salmon captured were within a very narrow size range,with 90% of fish between 36 and 49 mm, size class distribution was not determined.

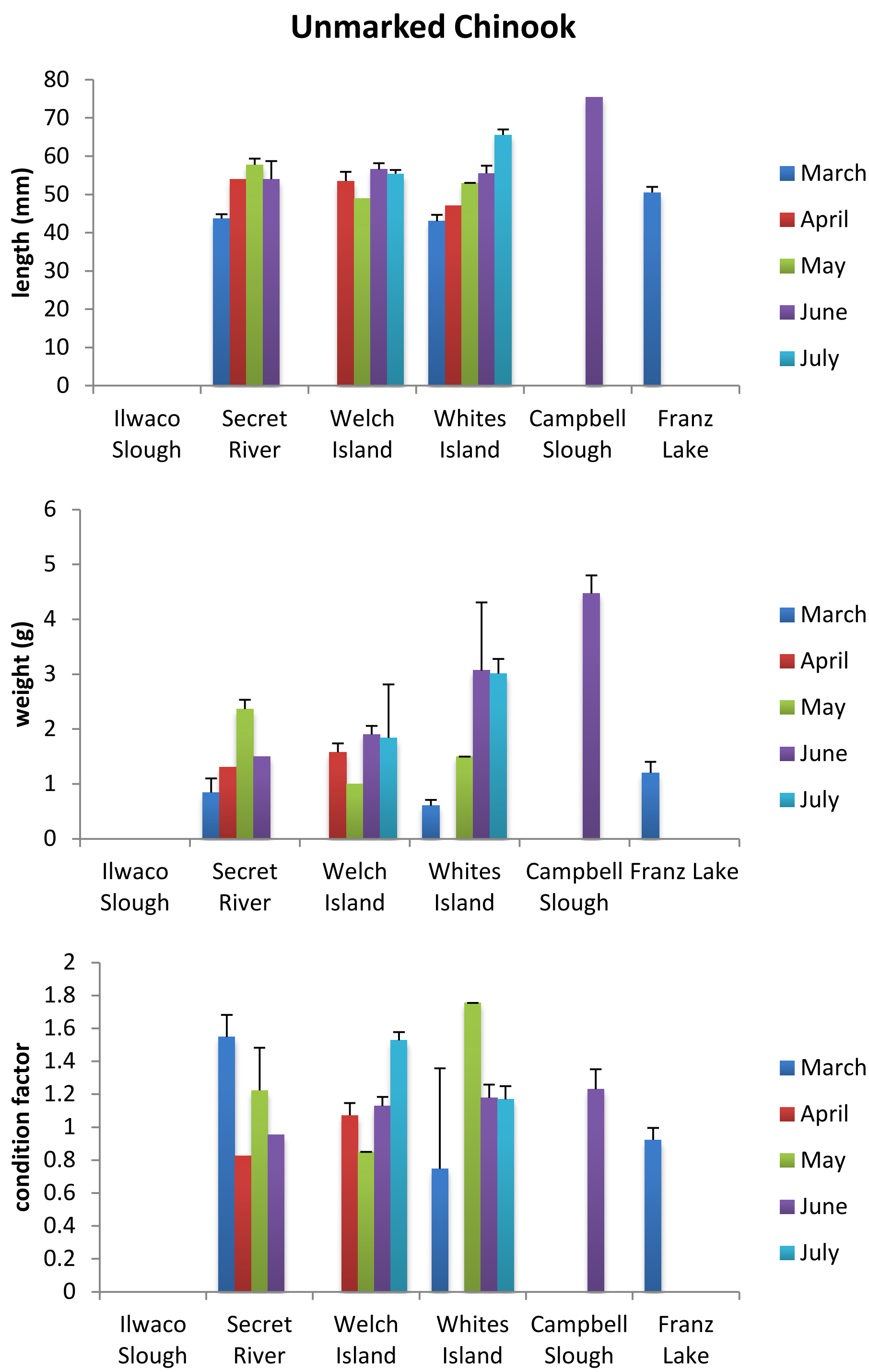


Figure 49. Length, weight and condition factor of unmarked Chinook salmon from the 2013 EMP sites.

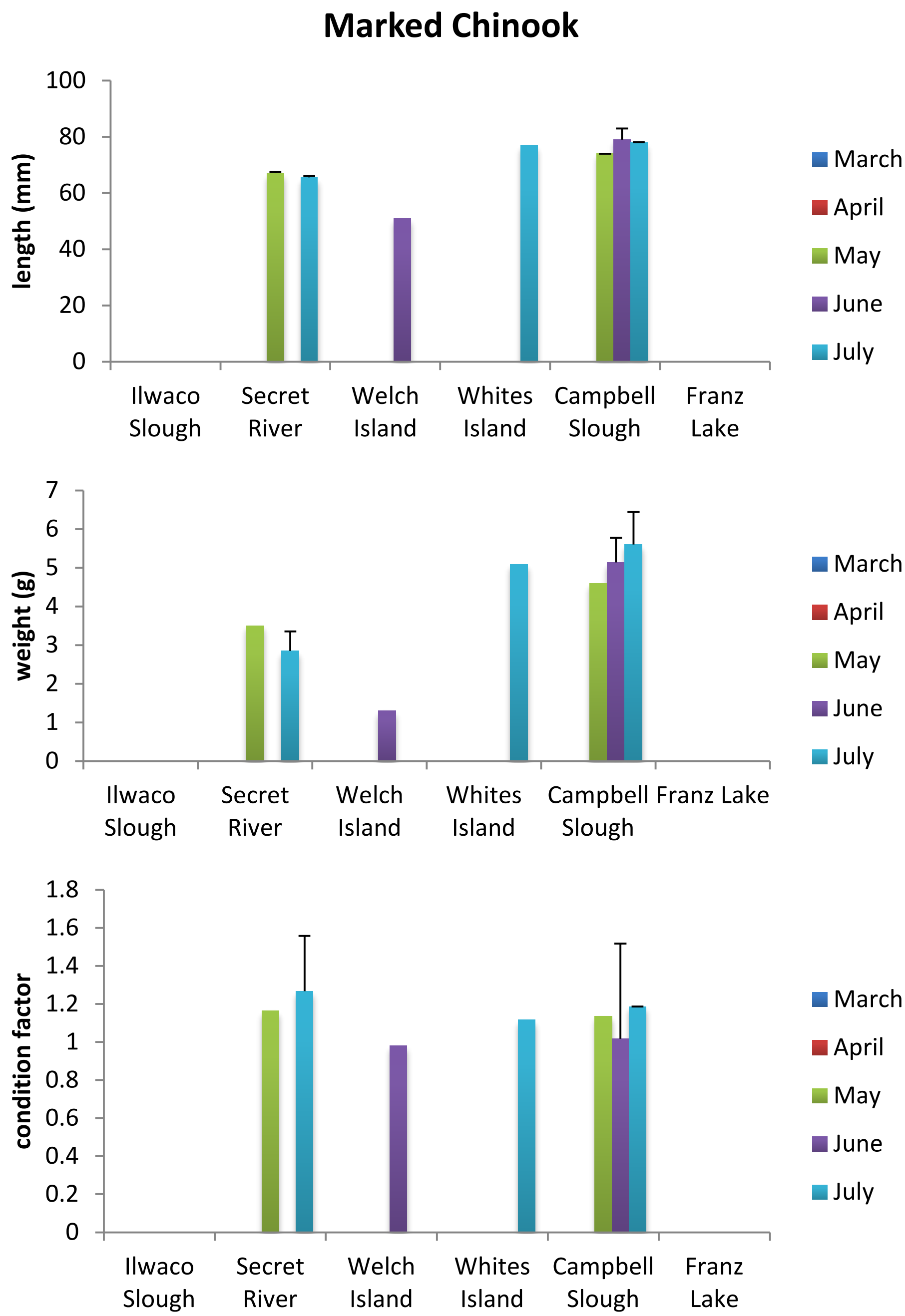


Figure 50. Length, weight and condition factor of marked Chinook salmon from the 2013 EMP sites.

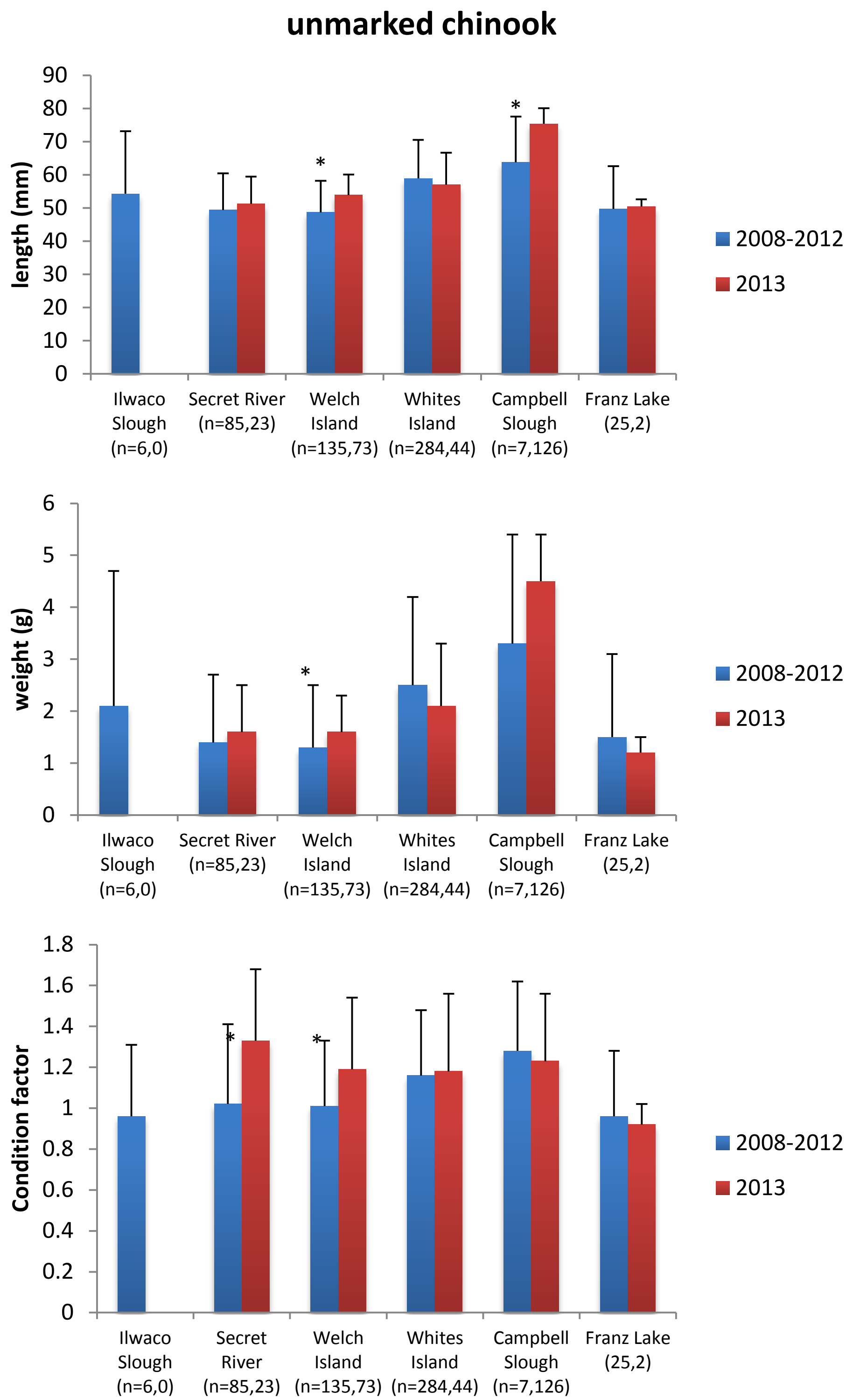


Figure 51. Mean length, weight, and condition of unmarked Chinook salmon at the 2013 at the 2013 EMP sampling sites as compared to average values for the other years sampled. Values of n refer to numbers of fish measured in the earlier sampling period (2008-2012) and in 2013. Asterisks indicate sites with significant differences ($p < 0.05$) between sampling periods

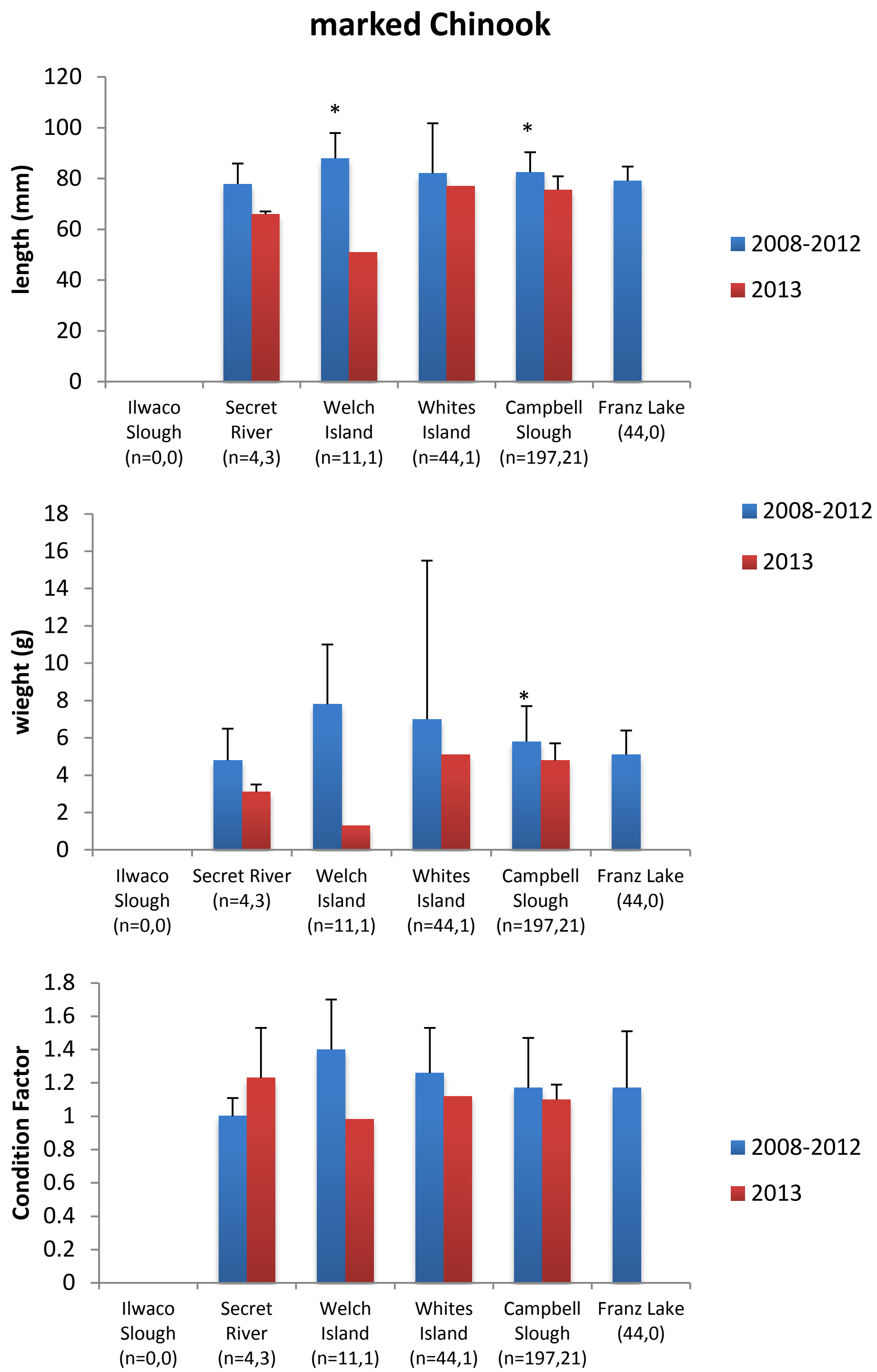
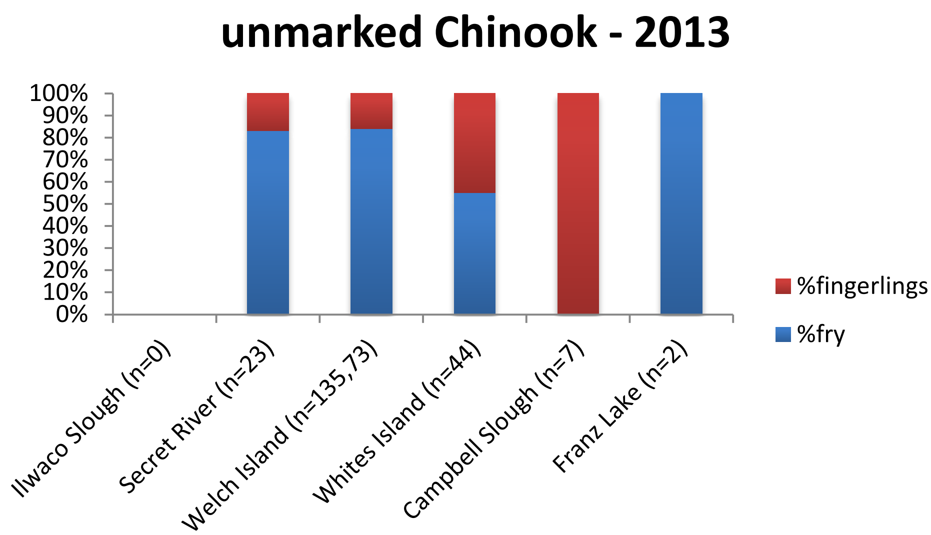


Figure 52. Mean length, weight, and condition of marked Chinook salmon at the 2013 at the 2013 EMP sampling sites as compared to average values for the other years sampled. Values of n refer to numbers of fish measured in the earlier sampling period (2008-2012) and in 2013. Asterisks indicate sites with significant differences ($p < 0.05$) between sampling periods.



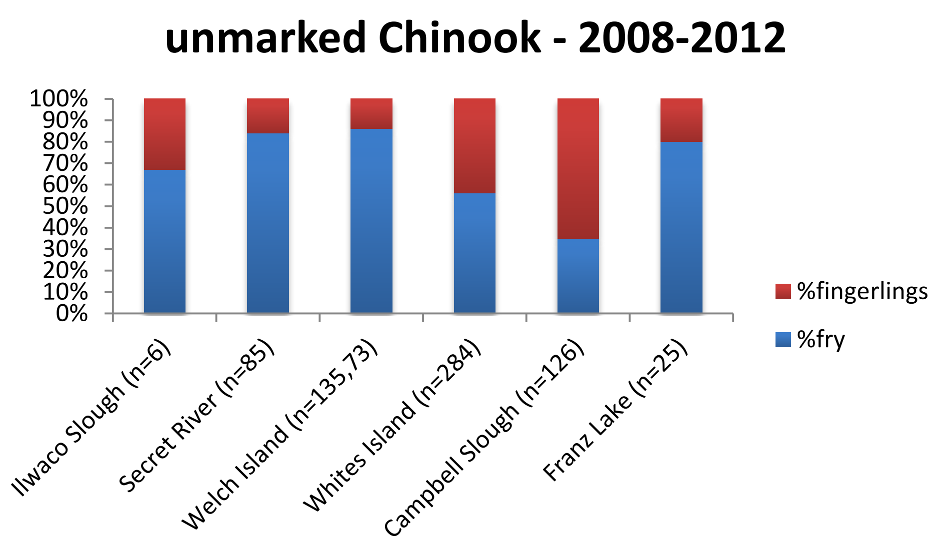
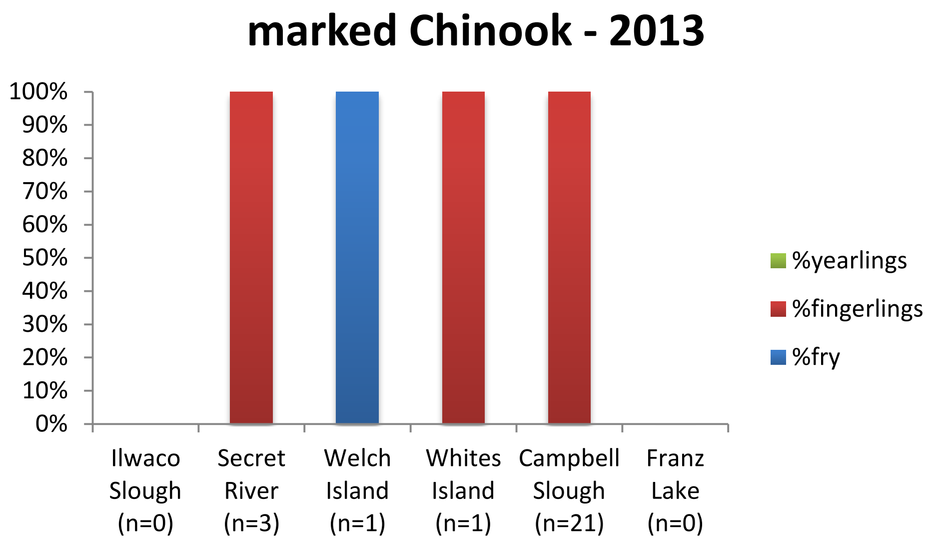


Figure 53. Size class distribution of unmarked juvenile Chinook salmon from the EMP sampling sites in 2013, as compared with other sampling years.



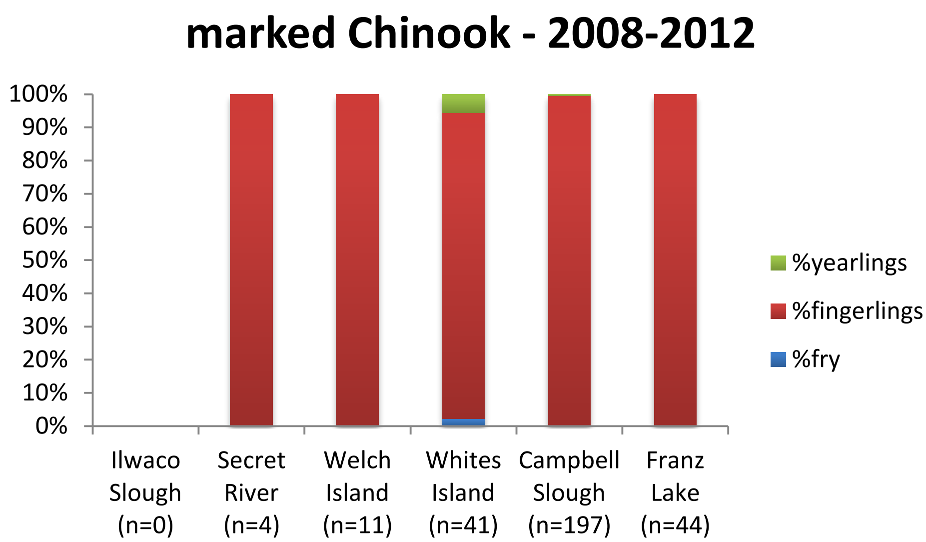


Figure 54. Size class distribution of marked juvenile Chinook salmon from the EMP sampling sites in 2013, as compared with other sampling years.

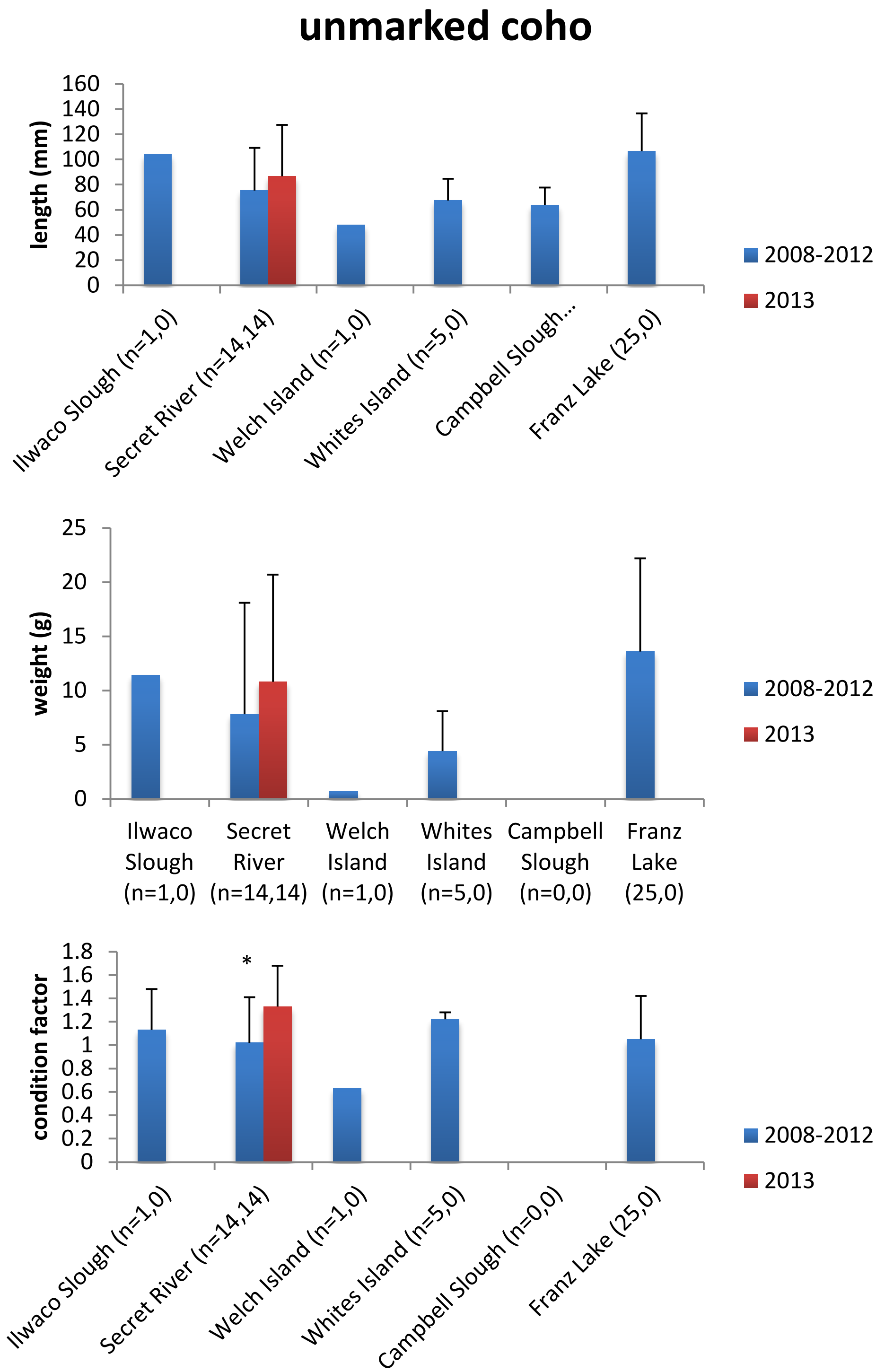


Figure 55. Mean length, weight, and condition of unmarked coho salmon at the 2013 EMP sampling sites as compared to average values for the other years sampled. Values of n refer to numbers of fish measured in the earlier sampling period (2008-2012) and in 2013. Asterisks indicate sites with significant differences ($p < 0.05$) between sampling periods

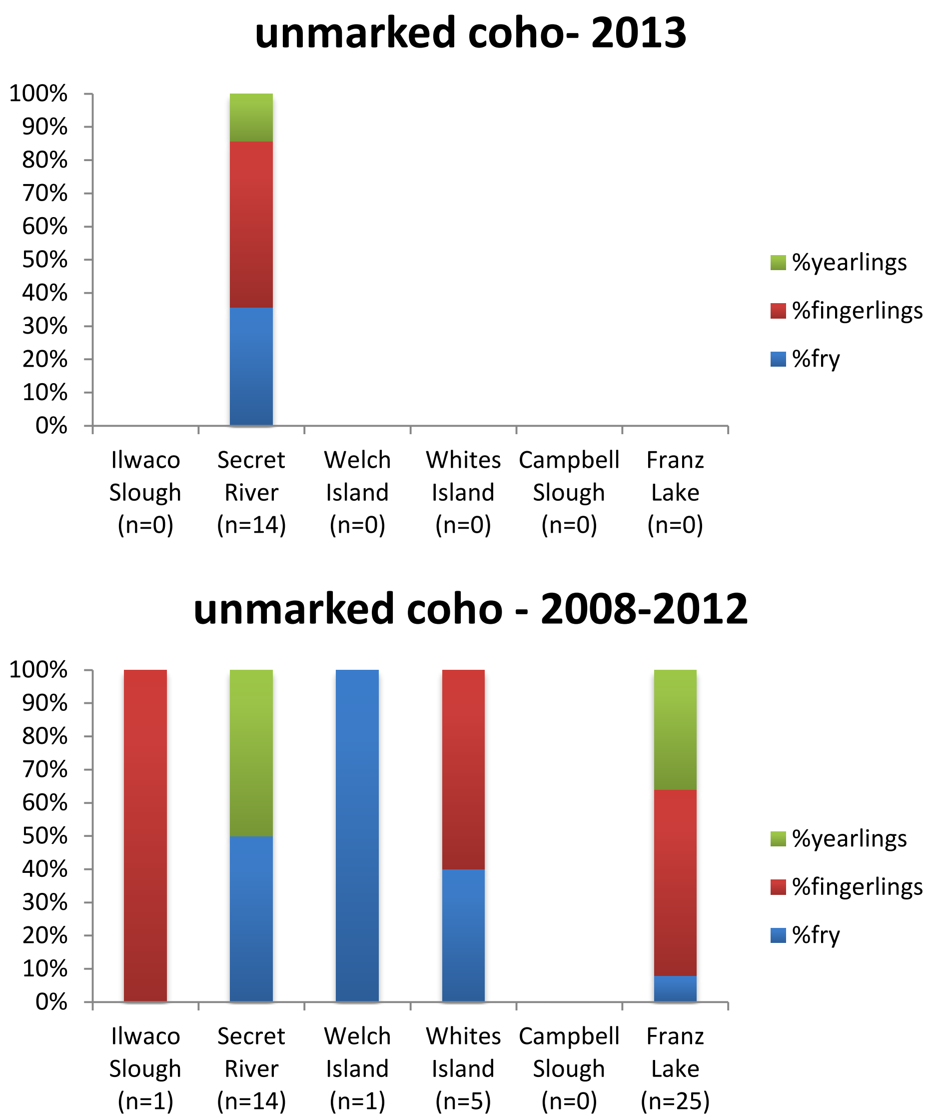


Figure 56. Size class distribution of marked juvenile coho salmon from the EMP sampling sites in 2013, as compared with other sampling years.

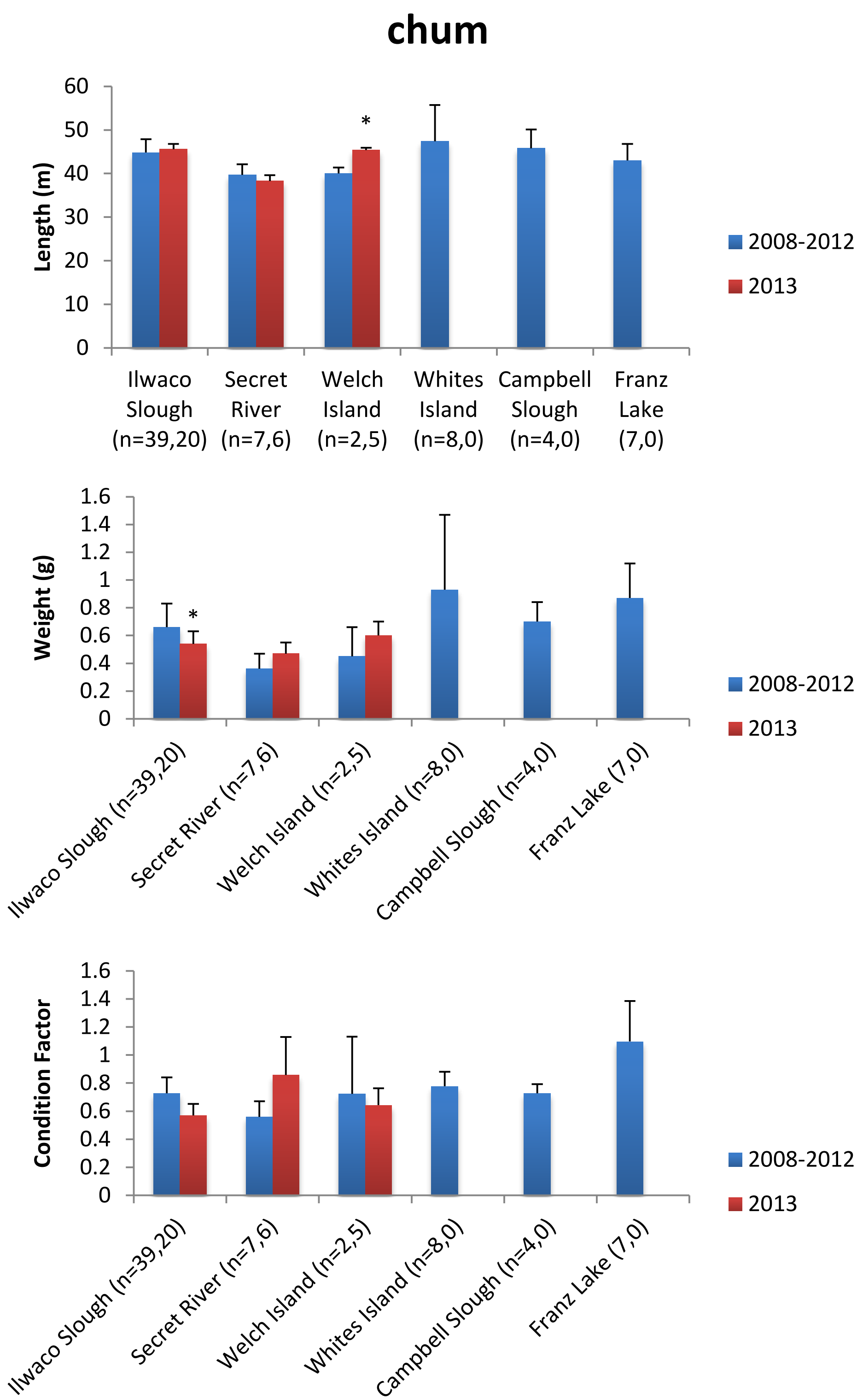


Figure 57. Size class distribution of chum salmon from the EMP sampling sites in 2013, as compared with other sampling years.

4.5.2.3 *Growth Analyses*

New data not yet available at the time of the writing of this report.

4.5.2.4 *Lipid content of juvenile Chinook salmon*

In this section lipid content data from 2012 are presented. Samples collected in 2013 have not been analyzed at this time. Whole body lipid content was measured both gravimetrically and by Iatroscan, but data presented here are from the gravimetric method because the largest number of samples had lipid content determined by this method. Latroscan values are highly correlated but slightly lower than gravimetric values.

Lipid content of marked vs. unmarked fish. For the samples collected in 2012, the lipid content of marked fish was significantly higher than the lipid content of unmarked fish (ANOVA, $p = 0.0234$). The mean lipid content (± SD) of samples from unmarked fish was 0.97 ± 0.23% (n=38), while the lipid content of samples from marked fish was 1.24 ± 0.10% (n=9).

Lipid content by season. In unmarked Chinook salmon, lipid content varied significantly by month of capture ($p = 0.0287$), with significantly higher values in fish collected in June than in April (Figure 58). Marked fish were collected only in May and June. While lipid content of marked fish was higher in June than in May, the difference was not statistically significant (ANOVA; p = 0. 2173; Figure 58).

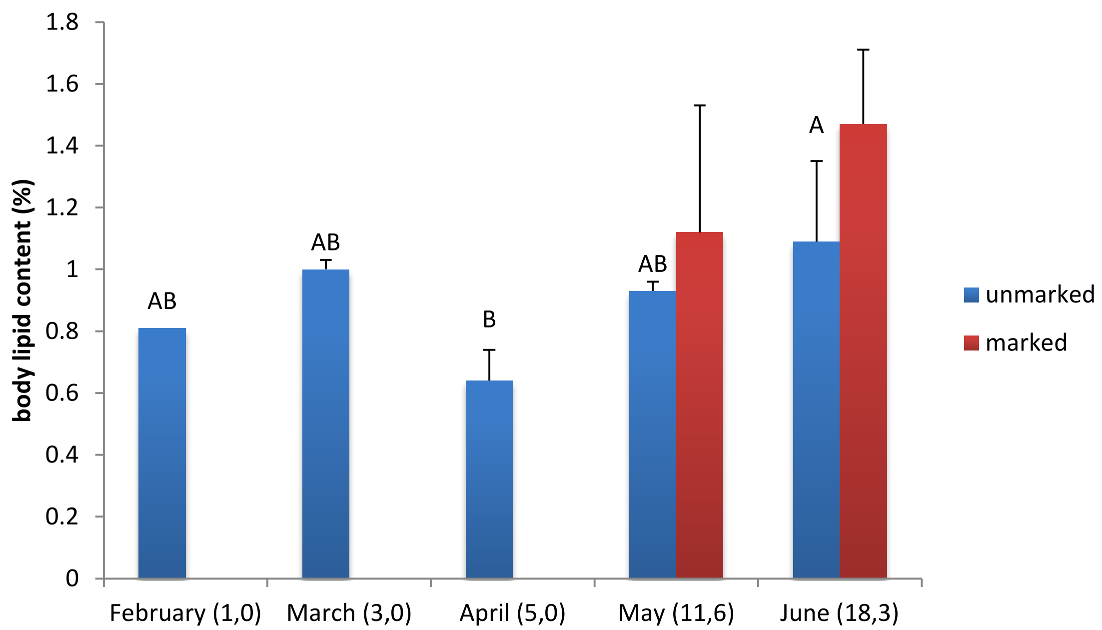


Figure 58. Mean % lipid content (SD) by month in marked and unmarked juvenile Chinook salmon collected from the EMP sampling sites in 2012. Marked and unmarked fish are compared separately. Values with different letter superscripts (upper case for unmarked fish, lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

Lipid content by site. In 2012, lipid content was measured in juvenile Chinook salmon from two new sites in Reach B, Secret River and Welch Island, and two new sites in Reach G, Lemon Island and Washougal (Figure 59). At Welch and Lemon Island and Washougal, samples were collected from both marked and unmarked fish, but at Secret River it was only possible to collect samples from unmarked fish. The average lipid content of unmarked Chinook salmon from Welch Island and Lemon Island (1.03-1.09%) was not especially high, but was comparable to levels observed at many other sites. Average lipid levels in fish from Washougal and Secret River (0.79-0.86%) were among the lowest values observed at the EMP sampling sites, and the value at Secret River was significantly lower (p = 0.0006) than the average value for unmarked Chinook salmon from Campbell Slough, where one of the highest mean lipid values was observed. The average lipid content of marked Chinook salmon from Welch Island, Lemon Island, and Washougal ranged from 1.35-1.47%, comparable to levels found in marked fish from most of the EMP sites.

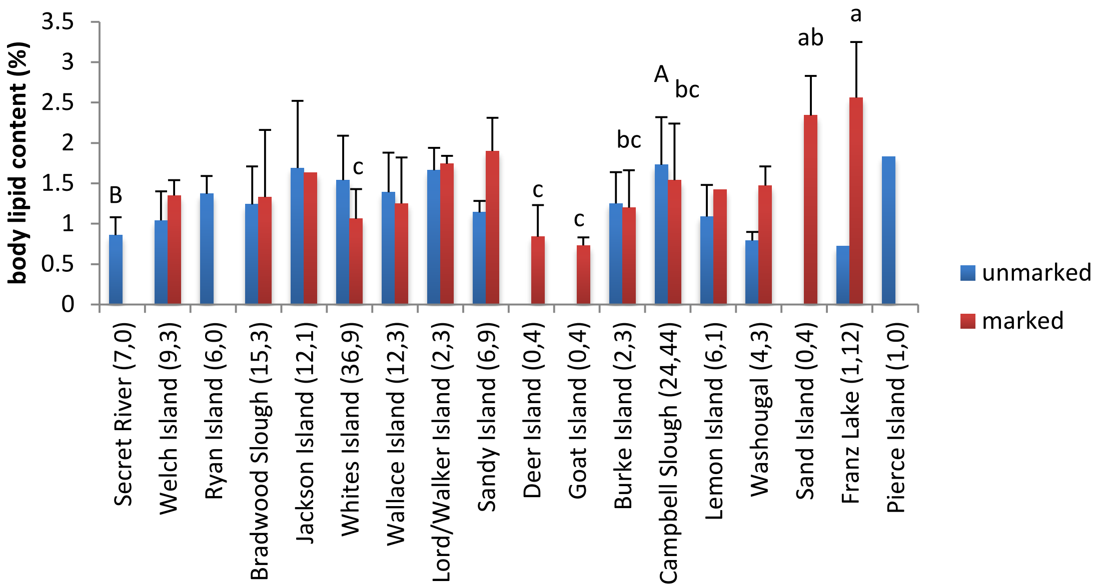


Figure 59. Mean % lipid content (SD) by site in marked and unmarked juvenile Chinook salmon from the EMP sampling sites. All data collected between 2007 and 2012 are included in this graph, so some sites include data from multiple years. Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

Trends in lipid content at fixed sites. Lipid content by year for marked and unmarked juvenile Chinook salmon collected between 2007 and 2012 at the EMP fixed sites is shown in Figure 60. Among unmarked Chinook salmon at both Campbell Slough and Whites Island, the lipid content of fish collected in 2012 was among the lowest. At White Island, the 2012 value was significantly lower than the lipid content measured in any previous year (p = 0.0049) while at Campbell Slough it was lower than values measured in either 2007 or 2010 (p = 0.0240). The lipid content of marked fish did not vary significantly by year at Whites Island (p = 0.4598), but at Campbell Slough the lipid content of marked fish was significantly lower than values measured in 2008 and 2010 (p < 0.0001).

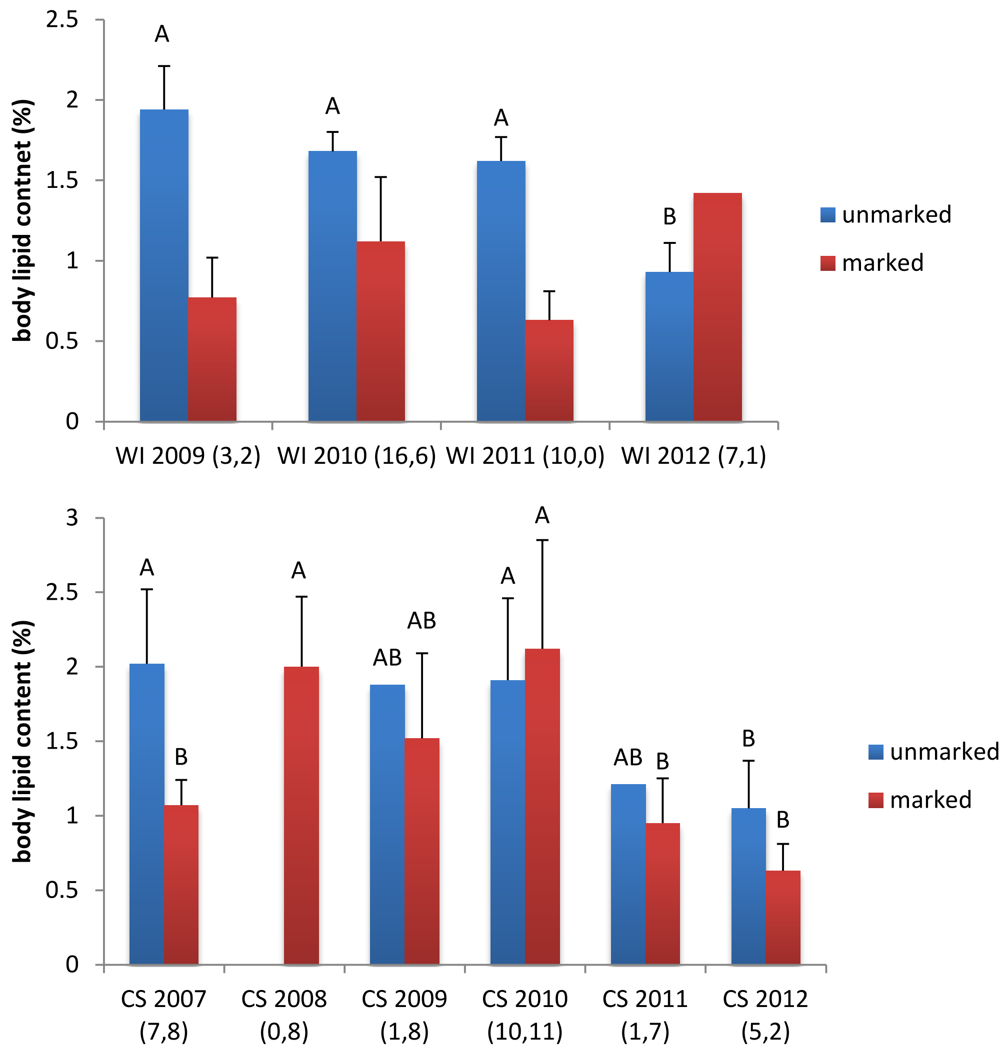


Figure 60. Temporal trends in mean % lipid content (SD) of marked and unmarked juvenile Chinook salmon from Whites Island (WI) and Campbell Slough (CS). Differences among years for marked and unmarked fish were tested separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

4.5.2.5 *Contaminants in juvenile Chinook salmon*

In this section, data on contaminant concentrations in bodies of juvenile Chinook salmon collected in 2012 are presented. Data are not yet available for the 2013 samples.

Contaminants in marked vs. unmarked fish. Overall, concentrations of persistent organic pollutants (POPs) and PAHs were very similar in marked and unmarked Chinook salmon collected in 2012. Average PCB, DDT, and PBDE concentrations (SD) in unmarked Chinook salmon (n=29) were 1417 ± 524 ng/g lipid, 1192 ± 97 ng/g lipid, and 605 ± 264 ng/g lipid, respectively. Average PCB, DDT, and PBDE concentrations (SD) for marked Chinook salmon (n=9) were 1430 ± 208 ng/g lipid, 1154 ± 173 ng/g lipid, and 567 ± 403 ng/g lipid, respectively. No significant differences were observed ($0.7418 < p < 0.9571$). Concentrations of PAHs in juvenile Chinook salmon bodies were slightly higher in marked than in unmarked fish, but the differences were not statistically significant ($p = 0.2959$). The average total PAH concentration in unmarked Chinook salmon collected in 2012 (n=30) was 23 ± 21 ng/g wet wt (weight) while for marked Chinook salmon (n=9), the average concentration was 31 ± 13 ng/g wet wt.

Contaminants by site. In 2012, POPs and PAHs were measured in juvenile Chinook salmon from two new sites in Reach B, Secret River and Welch Island, and two new sites in Reach G, Lemon Island and Washougal. At Welch and Lemon Islands and Washougal, samples were collected from both marked and unmarked fish, but at Secret River it was only possible to collect samples from unmarked fish.

In unmarked juvenile Chinook salmon, concentrations of PCBs (Figure 61) at all four of the new sites were not significantly different from concentrations in fish from the other EMP sites, with average values ranging from 1142 ± 137 ng/g lipid at Washougal to 1279 ± 184 ng/g lipid at Welch Island. These values are below the estimated threshold for toxic effects of PCBs in juvenile salmon of 2400 ng/g lipid (Meador et al. 2002). In marked fish, average PCB concentrations were also below the estimated toxicity threshold, ranging from 662 ± 206 ng/g lipid at Washougal to 1225 ± 169 ng/g lipid at Welch Island. Concentrations of PCBs in marked fish from Welch Island were comparable to those in marked fish from Sandy Island, where the highest PCB concentrations were observed. Concentrations of PCBs in marked fish from Washougal were significantly lower than those in marked fish from Sandy Island, as well as Welch and Lord/Walker Islands ($0.0001 \leq p \leq 0.0490$). Concentrations of PCBs in marked fish from Lemon Island did not differ significantly from those in marked fish from any of the EMP sites.

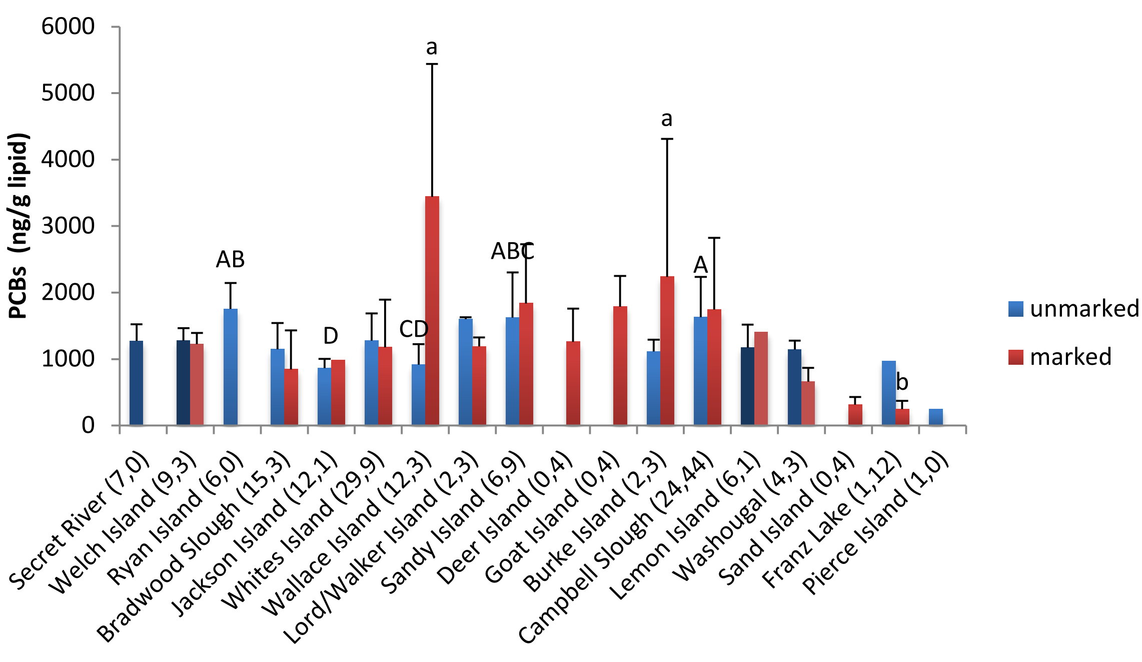


Figure 61. ∑PCB, concentrations (ng/g lipid) in bodies (minus stomach contents) of marked and unmarked juvenile Chinook salmon from the new 2012 EMP sampling sites, as compared to previously sampled sites. All data collected between 2007 and 2012 are included in this graph, so some sites include data from multiple years. Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

Concentrations of DDTs (Figure 62) in unmarked fish ranged from 502 ± 74 ng/g lipid at Washougal to 1474 ± 573 ng/g lipid at Welch Island, well below the estimated toxicity threshold of 5,000-6,000 ng/g lipid (Beckvar et al. 2005; Johnson et al. 2007). The DDT concentrations in unmarked fish from Washougal were significantly lower ($0.0001 < p < 0.0173$) than those in fish from most of the other EMP sites, while concentrations in fish from Lemon Island, Welch Island, and Secret Island were more comparable to those in fish from previously sampled EMP sites.
Concentrations of DDTs in marked fish were also relatively low at the new sites, ranging from 268 ± 55 ng/g lipid at Washougal to 1831 ng/g lipid at Lemon Island. Similar to unmarked fish, concentrations of DDTs in marked fish from Washougal were significantly lower than values observed in fish from several other EMP sites ($0.0001 < p < 0.0374$), while those in marked fish from Lemon Island, Secret Island, and Welch Island were not.

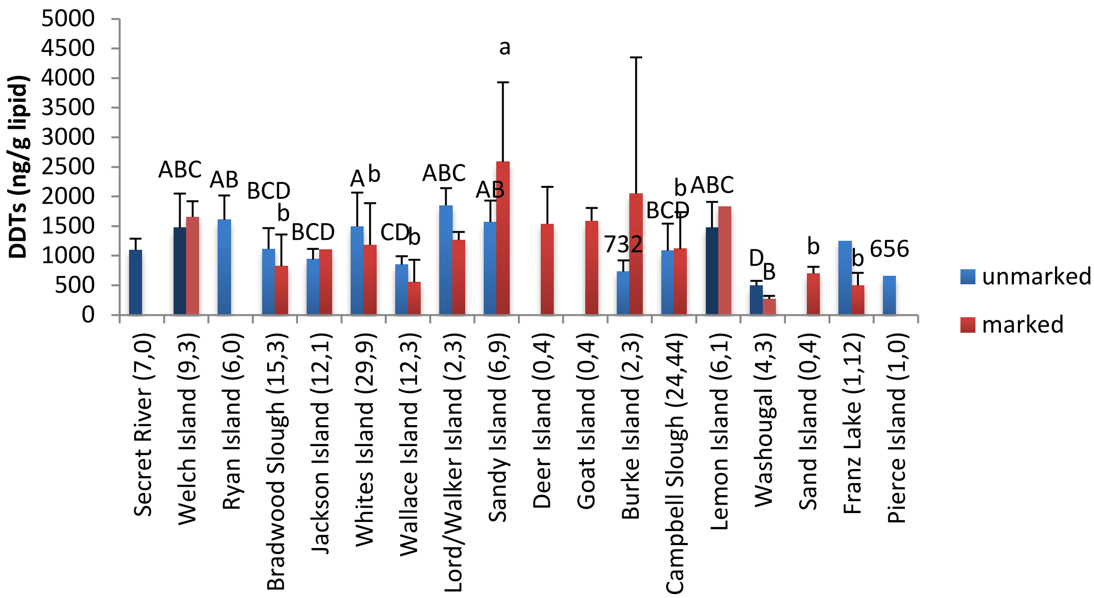


Figure 62. ∑DDT concentrations (ng/g lipid) in bodies (minus stomach contents) of marked and unmarked juvenile Chinook salmon from the 2012 EMP sampling sites, as compared to sites sampled previously. All data collected between 2007 and 2012 are included in this graph, so some sites include data from multiple years. Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, $\alpha = 0.05$).

Concentrations of PBDEs (Figure 63) in unmarked fish ranged from 397 ± 77 ng/g lipid at Lemon Island to 855 ± 246 ng/g lipid at Welch Island, while concentrations of PBDEs in marked fish ranged from 137 ± 63 ng/g lipid at Washougal to 1029 ± 182 ng/g lipid at Welch Island. The PBDE concentration in unmarked fish from Lemon Island was significantly lower ($p = 0.0055$) than maximum average concentration, found in unmarked fish from Burke Island, while the PBDE concentration in marked fish from Washougal was significantly lower ($p < 0.0001$) than the maximum concentrations found in marked fish from Sandy Island. No other significant differences in PBDE concentrations were found for fish from the new sampling sites. Average PBDE concentrations in both marked and unmarked juvenile salmon from Secret River, Lemon Island, and Washougal, and unmarked fish from Welch Island, were below the 940 ng/g lipid concentrations associated with immune dysfunction in Arkoosh et al. (2010). However, the average PBDE concentration of marked fish from Welch Island, at 1029 ng/g lipid, were slightly above this toxicity benchmark.

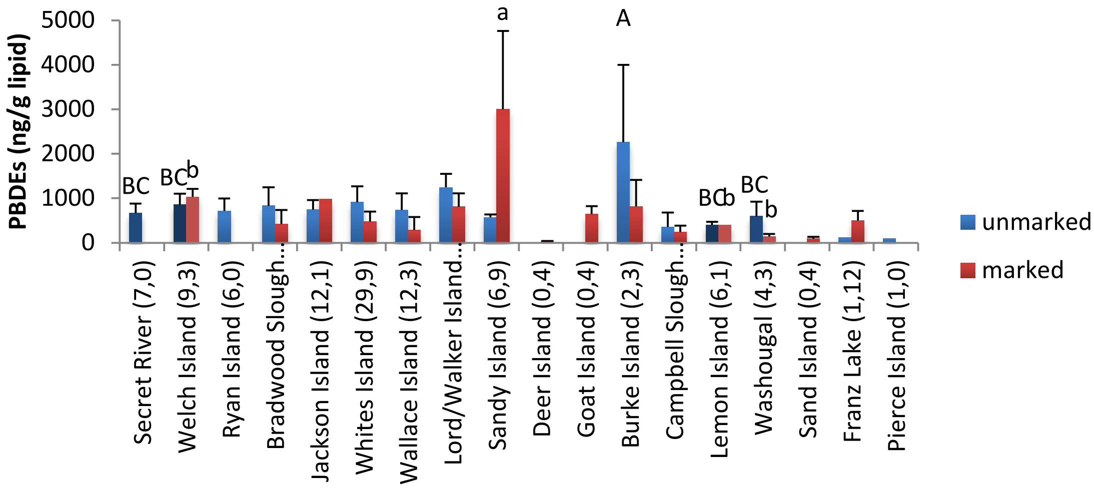


Figure 63. ∑PBDE, concentrations (ng/g lipids) in bodies (minus stomach contents) of marked and unmarked juvenile Chinook salmon from the 2012 EMP sampling sites, as compared to concentrations at other EMP sites. All data collected between 2007 and 2012 are included in this graph, so some sites include data from multiple years. Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

Average concentrations of PAHs (Figure 64) in unmarked juvenile Chinook salmon ranged from 13 ± 6 ng/g wet wt in fish from Lemon Island to 59 ± 42 ng/g wet wt in fish from Washougal, while PAH concentrations in marked fish ranged from 14 ng/g wet wt at Lemon Island to 41 ± 1 ng/g wet wt at Welch Island.

The PAH concentration in unmarked fish from Washougal was significantly higher ($0.0095 < p < 0.0484$) than those in fish from several other EMP sites, including Lemon Island and Secret River, while concentrations at the other sites were more comparable to PAH concentrations previously measured at most EMP sites. Concentrations of PAHs in marked fish from the new sampling sites did not differ significantly from concentrations measured at any of the other sites (Tukey-Kramer HSD multiple range test, $p \leq 0.05$). Because we have only recently started to measure PAH concentration in salmon bodies, we are uncertain of the body concentration associated with toxic effects of these compounds.

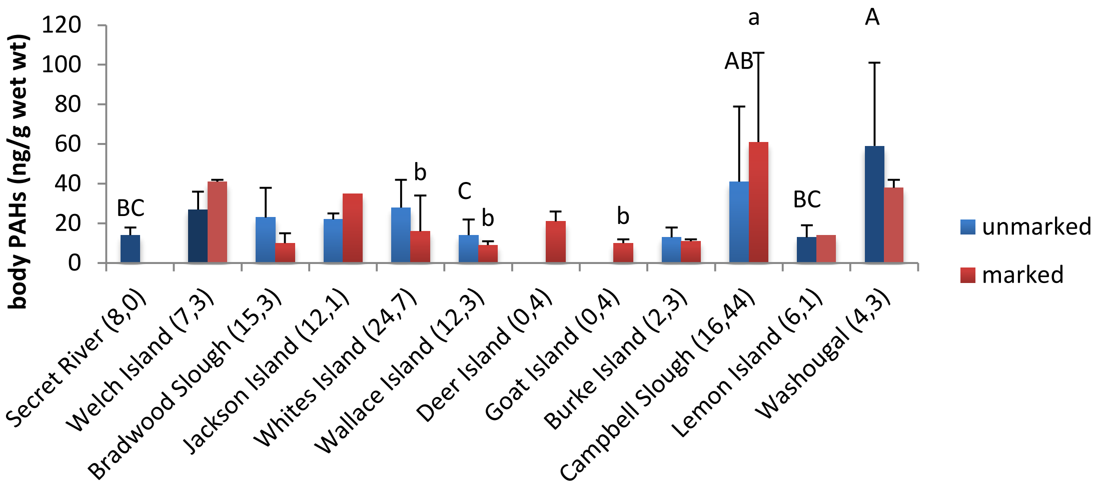


Figure 64. ∑PAH concentrations (ng/g wet wt) in bodies (minus stomach contents) of marked and unmarked juvenile Chinook salmon from the 2012 EMP sampling sites, as compared to concentrations in fish from other EMP sites. All data collected between 2007 and 2012 are included in this graph, so some sites include data from multiple years. Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, $\alpha = 0.05$).

Trends in contaminant concentrations at fixed sites. Of the fixed sites sampled in 2012, data on contaminant concentrations are available only for Campbell Slough. Too few fish were collected at Franz Lake or Ilwaco Slough for analysis, and concentrations in samples from Whites Island could not be measured because of equipment problems and resulting analytical difficulties.

Concentrations of POPs by year for marked and unmarked juvenile Chinook salmon collected at Campbell Slough between 2007 and 2012 are shown in Figure 65. For both marked and unmarked Chinook salmon, PCB concentrations were higher in 2012 than in 2011, with a statistically significant increase in the case of unmarked Chinook ($p = 0.0304$). In unmarked fish, concentrations of DDTs at Campbell Slough in 2012 were not significantly different from those measured in other years. In marked fish, concentrations were slightly higher in 2012 than in 2011 or several earlier years, but the differences were not statistically significant. Concentrations of PBDEs were somewhat higher in both unmarked and marked fish collected from Campbell Slough in 2012 than in 2011, but concentrations were not significantly different among years (Tukey-Kramer HSD multiple range test, $p < 0.05$).

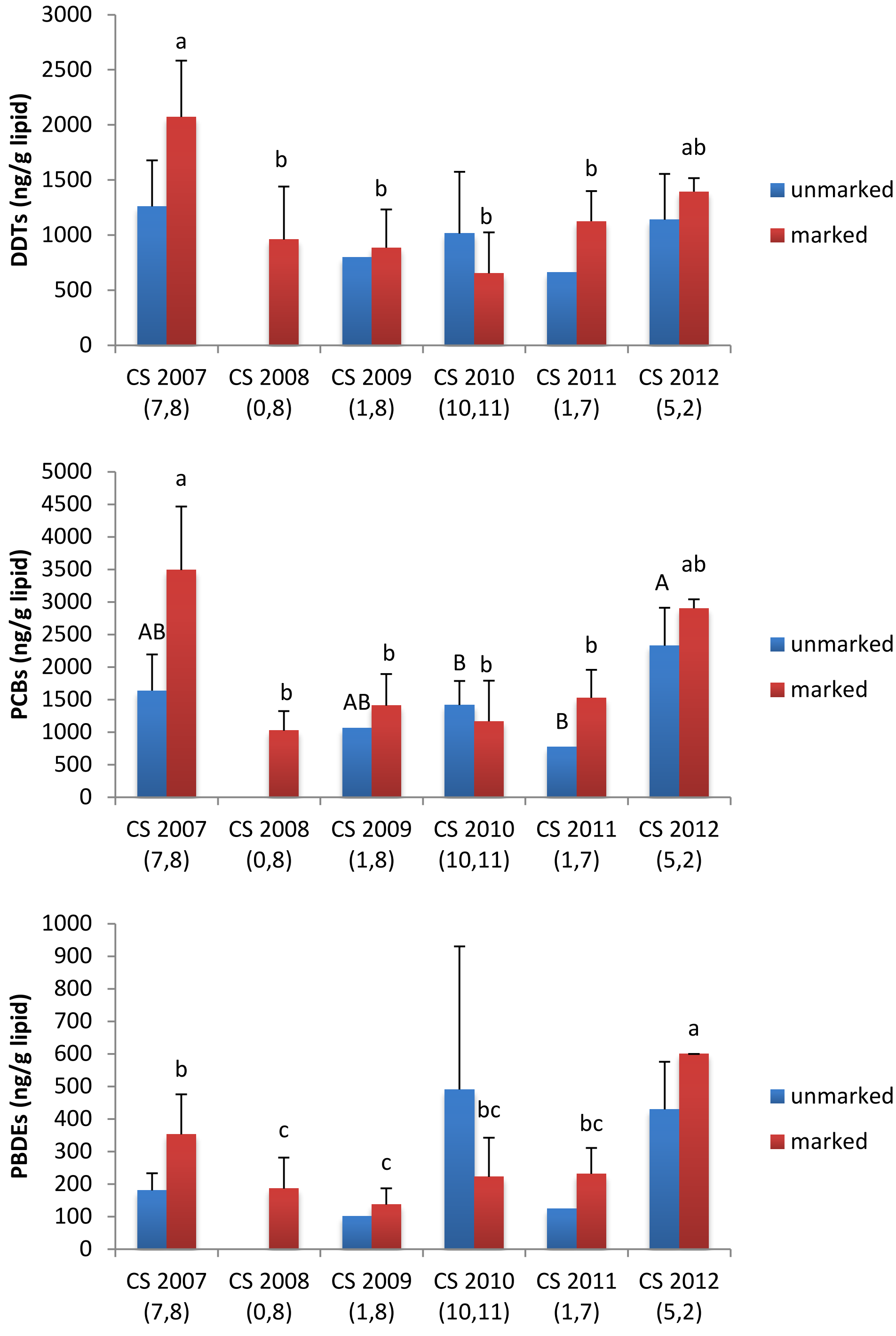


Figure 65. ∑DDT, ∑PCB, and ∑PBDE concentrations (ng/g lipid) in bodies of marked and unmarked juvenile Chinook salmon sampled from Campbell Slough in 2012, as compared to earlier years. Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

Concentrations of PAHs by year for marked and unmarked juvenile Chinook salmon collected at Campbell Slough between 2010 and 2012 are shown in Figure 66; PAH analyses were not conducted on salmon bodies prior to 2010. Concentrations of PAHs in bodies of both marked and unmarked salmon collected from Campbell Slough in 2012 were the lowest yet reported, significantly lower than concentrations measured in either 2010 or 2011 ($0.0011 < p < 0.0077$).

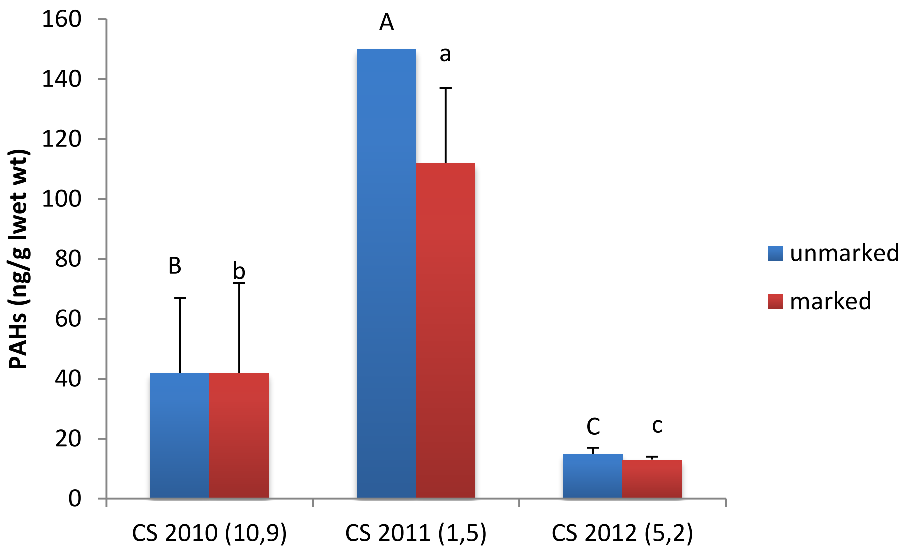


Figure 66. ∑PAH concentrations (ng/g wet wt) in bodies of marked and unmarked juvenile Chinook salmon sampled from Campbell Slough in 2012, as compared to earlier years. (PAHs in bodies were not measured prior to 2010). Marked and unmarked fish are compared separately; values with different letter superscripts (upper case for unmarked fish and lower case for marked fish) are significantly different (Tukey-Kramer HSD multiple range test, α = 0.05).

4.5.2.6 *PIT-tag array monitoring of juvenile salmon habitat occurrence*

Campbell Slough. A PIT (passive integrated transponder) tag detection system was installed in Campbell Slough on June 2, 2011. This system consists of a Destron-Fearing FS1001-MTS multiplexing transceiver, which simultaneously receives, records and stores tag signals from two antennas measuring 4' by 20'. The system is powered by a 470W solar array with battery backup and is also connected to a wireless modem that allows daily data downloads. In 2011 (first year of 'pilot' project) the system functioned reasonably well throughout much of the summer, however we discovered that some changes and improvements would be required if the system was going to perform better in subsequent years. For example, the antennas were anchored to the channel bottom without any mechanism for adjusting their height within the water column when water levels changed dramatically. Also, due to the initial location of the solar array, the system did not receive adequate solar exposure to keep the batteries fully charged, resulting in the system shutting down on several occasions.

In 2012 we addressed both of these issues, however we were unable to install and operate the antennas and receiver that summer due to scheduling conflicts and the presence of a Bald Eagle's nest near our site. We utilized this down year to move the solar panels (along with receiver, modem and batteries) to the north side of the slough, thereby eliminating the issue of poor solar exposure and insufficient power generation. We also reinstalled the antennas using a line and pulley system anchored to the channel bottom, allowing us to adjust the depth of the antennas within the water column if necessary.

For the 2013 sampling season, the system was installed and connected on May 9. Water depth at the site on day of installation was about 8' and the antennas were adjusted so that the top rail was floating just at the surface. For 2013, water levels at the site fluctuated moderately thoughout the spring and early summer, unlike during 2011 when we observed dramatic flooding. Only one depth adjustment was required (in early June) to keep the antennas well positioned in the water column thoughout the summer. Overall, the antennas, receiver, modem and batteries functioned perfectly and we had continuous and uninterrupted data collection though November 6. It should be noted however that since about mid-July, the antennas were floating on the surface due to very low water levels (approx. 2 feet). On November 13, the entire system was powered off as the batteries were completely discharged, as there was insufficient solar power generation for it to operate it.

In total, the system was operable and collecting tag data for nearly six months and recorded 48 detections, which corresponded to 15 unique tags. The first detection occurred on May 9 and the last detection was on August 20. Using the PTAGIS database we were able to determine species and site origination info for all but three of these tags. Most of the detected fish were hatchery Fall Chinook salmon, but we also detected one wild Chinook salmon, one hatchery summer steelhead and one Northern pikeminnow (Table 22). Nearly all hatchery Chinook salmon originated from the Spring Creek National Fish Hatchery (near White Salmon, WA) but one late release (July) came from the Little White Salmon Hatchery in Stevenson, WA (upstream of Bonneville Dam). In addition to the hatchery Chinook salmon, we detected one wild Chinook that was tagged as part of a barging study, one summer steelhead that originated from the Cottonwood Acclimation Pond (SE corner of WA state) and also a Northern pikeminnow that had been tagged and released near the mouth of the Lewis River.

Nearly half of all these tagged fish were detected multiple times at the site over a period of multiple days or weeks. One Chinook from the Spring Creek Hatchery was detected 17 times over a 12 day period. The longest time between detections of the same fish was 22 days and the shortest was 18 hours. The time for Chinook to travel from the Spring Creek Hatchery to the Campbell Slough site ranged from 2-5 weeks.

For 2014, we are planning to replace the battery bank with larger batteries and have the system powered up and operational in late January. We will plan to 'boost' the battery bank with a generator/charger system periodically until spring when the system should be self-sufficient on solar power generation. Site visits to adjust the antenna 'height' in the water column will be made as needed and be somewhat dependent upon presence/absence of an active Eagle's nest.

Horsetail/Oneonta Creek. A PIT tag detection system (Figure 67) was installed and fully operational in the culvert system at the confluence of Horsetail and Oneonta Creeks on May 9, 2013. This system consists of a Biomark FishTRACKER IS1001-MTS distributed Multiplexing Transceiver System (MTS), which powers 10 antenna units mounted within the culvert system at Horsetail/Oneonta Creek site (Columbia River, OR) beneath Interstate-84. The MTS unit receives, records and stores tag signals from these 10 antennas, which all measure approximately 6' by 6' and are mounted on both ends of the 5-barrel culvert system running under the freeway. The system is powered by an 840 watt (W) solar panel array and supported by a massive 24-volt, 800 amp-hour battery bank backup. The unit is also connected to a fiber optic wireless modem that allows for daily downloads of tag data and system voltage monitoring updates.

The successful installation of this system was the culmination of nearly 18 months of work towards this goal. This effort included obtaining permits from multiple groups/agencies, the fabrication and testing of antennas during the Fall/Winter 2012, installation of all 10 antennas in January 2013, installation of a 17' steel pole (requiring 38 bags of concrete) and all the solar components in February and final placement of the battery bank (16 batteries) and all system components in April. Final electrical connections, power-up and tuning of the entire system was completed on May 9.

In this first year of operation, the system functioned remarkably well through much of the calendar year; however we did experience some technical issues that led to a period of downtime for the system and interruptions in data collection. For example, after the initial set-up, the system functioned well until early June when a tripped circuit breaker on the solar input line caused a significant voltage drop in the batteries, forcing the MTS receiver into standby mode. At the time this occurred, we did not have the wireless modem installed and so only became aware of this problem on June 14 during a planned site visit. Corrective measures were made on June 14 and the wireless modem was installed at the same time, allowing for daily system updates.

We also encountered some problems with several of the ACN's (antenna control nodes), resulting in some periods of no data collection for several antennas. On one occasion, the faulty ACN was replaced within a couple weeks of discovery, however one other faulty ACN was not discovered for several months. This particular ACN was on the western most barrel on the north side, and corresponds to the culvert barrel that has a fish ladder type structure within it to facilitate upstream migration. For much of the summer it was observed that this antenna always failed the hourly internal 'test tag' protocol. It was initially believed that this failed test was most likely due to the barrel being completely filled in with sediments (as was observed in previous years). However, after the barrel was cleared of sediments in late summer the antenna continued to fail the hourly test tag and it was determined that the ACN was defective. Upon this discovery, that antenna was connected to a functioning node.

A further interruption of tag data collection occurred beginning July 18 when all five of the south side antennas were disconnected and removed to allow for restoration activities at the site. The restoration activities were completed in early September and these antennas were reinstalled and connected to the receiver on September 11, but not powered up because 4 of 5 barrels were dry. These antennas were finally powered back up on September 29, but three of them failed to come back on due to what was later determined to be a faulty wire connection. These three antennas were finally reconnected and back in operation on October 13.

In general, for this first year of operation the entire system (antennas, receiver, modem, solar panels and batteries) functioned exceptionally well and with just a couple exceptions, we had nearly continuous tag data collection for most antennas though November 26, despite the significant decline in solar power generation that started in late September. Finally, in early December the voltage drop in the battery bank backup accelerated dramatically and the entire system was powered off on December 4 to preserve the integrity of the batteries.

In total, the system was operable and collecting tag data for nearly seven months and recorded 591 detections, which corresponded to 72 unique tags. The first detection occurred on May 11 and the last one was on November 23. Using the PTAGIS database we were able to determine species and site origination info for all but two of these tags (Table 23). Nearly all of the fish detected from May through the end of July were juvenile salmonids that had been released earlier in the year. The exceptions were an adult sockeye and Chinook salmon that were tagged in the lower river for an upstream migration study. Most of the juvenile fish were hatchery Chinook salmon, comprising both spring and fall stocks, but we also detected some hatchery coho and a hatchery summer steelhead. There was also a juvenile wild Chinook salmon that had been tagged upriver as part of a survival study. The hatchery Chinook salmon originated from as far away as the Rapid River (tributary of the Salmon River) Hatchery near Riggins, ID and as nearby as the Little White Salmon Hatchery in Stevenson, WA (above Bonneville Dam). The coho salmon originated from hatcheries (or rearing ponds) near Winthrop and Wenatchee, WA.

After the last juvenile salmon detected on July 28, there were no detections for the entire month of August and most of September. Fish were finally detected again in late September on an almost daily basis though late November. These fish were predominantly adult Chinook and coho salmon that had been captured and tagged in the lower river as part of an upstream migration study. There were also a couple of juvenile wild steelhead that had been tagged upriver and released and a pair of juvenile hatchery coho salmon from the Kooskia National Fish Hatchery Hatchery in Idaho that had been released in February and April.

Of the 21 juvenile salmon and steelhead that were detected at the site in the spring and early summer, nearly all of them were encountered for only a day or two. From this group, it appears that only two individuals (hatchery coho, hatchery spring Chinook salmon) were able to successfully navigate all the way through the culvert. An adult sockeye salmon also transited the culvert system in mid-July and was able to make it through the eastern most barrel (without a fish ladder structure). In contrast to the spring/summer juveniles, nearly twice as many adult salmon were encountered at the site in autumn, and many were present for long periods of time. For example, one adult Fall Chinook salmon was detected 57 times over the span of a 15-day period. These adults were also far more successful in transiting through the culvert system, with at least 14 individuals (both coho and Chinook salmon) detected by a south side antenna. In most cases, these fish passed through the western most barrel with the fish ladder structure. Interestingly, the only two detected juvenile wild steelhead (released in spring) also were observed going all the way through the culvert in the fall. A more in depth analysis of these data to look at duration of time encountered at the site and also success in transiting the culvert structure will be presented in a future report.

For 2014, we need to replace a faulty ACN on the north side and also boost (charge) the battery bank backup in order to power it back up and make it fully operational. This will take place in late January and we plan to continue charging the battery bank as needed with a generator until springtime, when the system should be self-sufficient on solar power generation.

Figure 67. PIT tag array at Horsetail/Oneanta Creeks.

Table 22. Fish detected in 2013 at Campbell Slough PIT-tag array.

Species	No of fish detected	Months Present	Length (mm)
Juvenile hatchery fall Chinook	9	May, June	63-75
Juvenile hatchery summer steelhead	1	May	na
Juvenile wild Chinook	1	May	104
Nothern Pikeminnow	1	July	484

Table 23. Fish detected in 2013 at Horsetail Falls PIT-tag array.

Species	No of fish detected	Months present	length (mm)
Juvenile hatchery spring Chinook	11	April, May, June, July	99-128
Juvenile hatchery summer steelhead	2	July	n/a
Juvenile hatchery fall Chinook	4	May, July	74 - 80
Juvenile wild steelhead	2	Oct	147, 151
Juvenile hat coho	4	Sept, Oct, Nov	107 - 124
Adult spring Chinook	1	June	n/a
Adult fall Chinook	10	Sept, Oct, Nov	605 - 855
Adult coho	28	Sept, Oct, Nov	400 - 920
Adult sockeye	1	July	n/a
Adult hatchery fall Chinook	3	Sept, Oct, Nov	380 - 640
Adult hatchery coho	2	Sept, Oct, Nov	610 - 660

4.5.2.7 *Juvenile Chinook salmon diets*

In 2013, Chinook salmon diet samples were collected from Secret River, Welch Island, Whites Island, and Campbell Slough. At all four sites, Diptera were the most abundant or among the most abundant items, both by count (Figure 68) and by weight (Figure 69). Amphipods were also abundant in salmon diets at Secret River, Welch Island, and Whites Island. Generally Diptera were found at the greatest abundance, but at Secret River and Whites Island, amphipods were more abundant than Diptera in May. Other species observed at relatively high abundances included Trichoptera in June and July at Secret River, and Cladocera in May at Whites Island.

Electivity values for 2013 by month at the sampling sites (Table 24) 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. Amphipods were highly selected at Secret River, Welch Island, and Whites Island, while the preference for Diptera, though still evident in some months, was less marked. At Campbell Slough, Diptera were a selected prey but Amphipods but appear to have been avoided. Hemiptera and Cladocera were generally consumed at levels at or below those that would be expected given their abundances, but patterns for other taxa were some inconsistent. For example, Copepods and Hymenoptera appeared to be selected at Welch Island, but not at other sites, whereas Trichoptera were sometime preferred and sometimes avoided, depending on site and month. It should be noted that 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.

Diet composition patterns observed at the trend sites in May in 2013 were generally similar to those observed in fish collected in the same month in previous years (Figure 70 and Figure 71). Secret River was only sampled in May in 2013, so could not be included in the comparison. Diets of fish from Campbell Slough showed consistently high Diptera abundance, along with Cladocera and various other taxa. At Welch Island and Whites Island, diets were dominated by amphipods and Diptera, with Cladocera also present at Whites Island. While actual abundances of various prey items in diets varied from year to year, values for the most part remained within similar ranges. Some differences in diets from year to year include the presence of Nemata in diets at Welch Island in 2012 and at Whites Island in 2011 and 2013, but the absence of this taxon in 2013.

When electivity values for the month of May are compared across multiple years (Table 25), a consistent preference for Amphipods is observed at Welch Island and Whites Island, but the preference for Diptera varies from year to year. At Campbell Slough the pattern is the opposite, with a consistent preference for Dipterans, and a mixed response to Amphipods. Cladocera and Copepods were generally avoided, with inconsistent responses to other taxa.

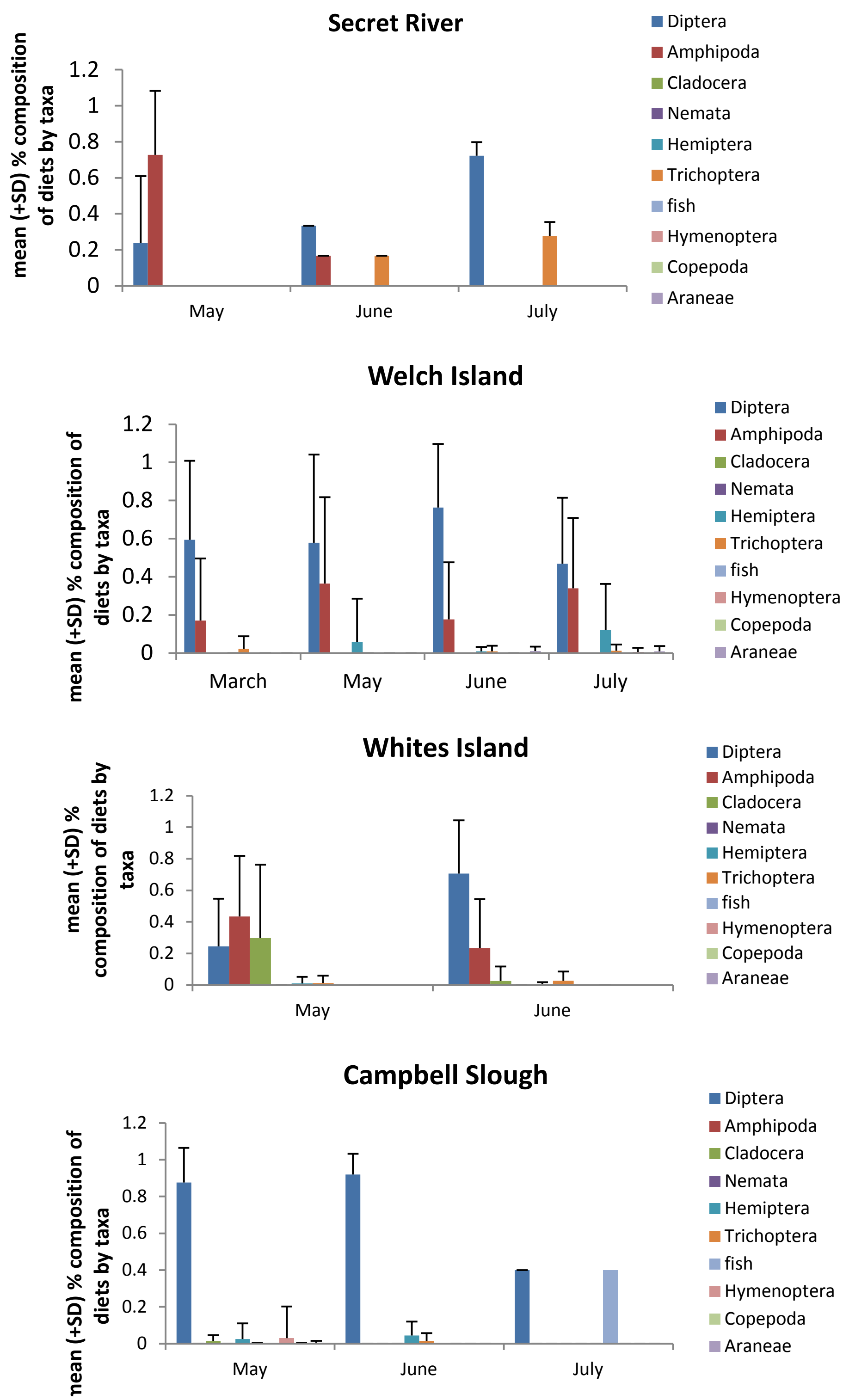


Figure 68. Mean (SD) percent composition (by abundance) of prey in juvenile Chinook stomachs collected from Secret River, Welch Island, Whites, Island, and Campbell Slough in each month sampled in 2013. The 10 orders of prey presented here include on average over 95% of the consumed invertebrate prey for all fish, not just fish from these sampling events. Because not all orders are shown, the bars may not sum to 100%.

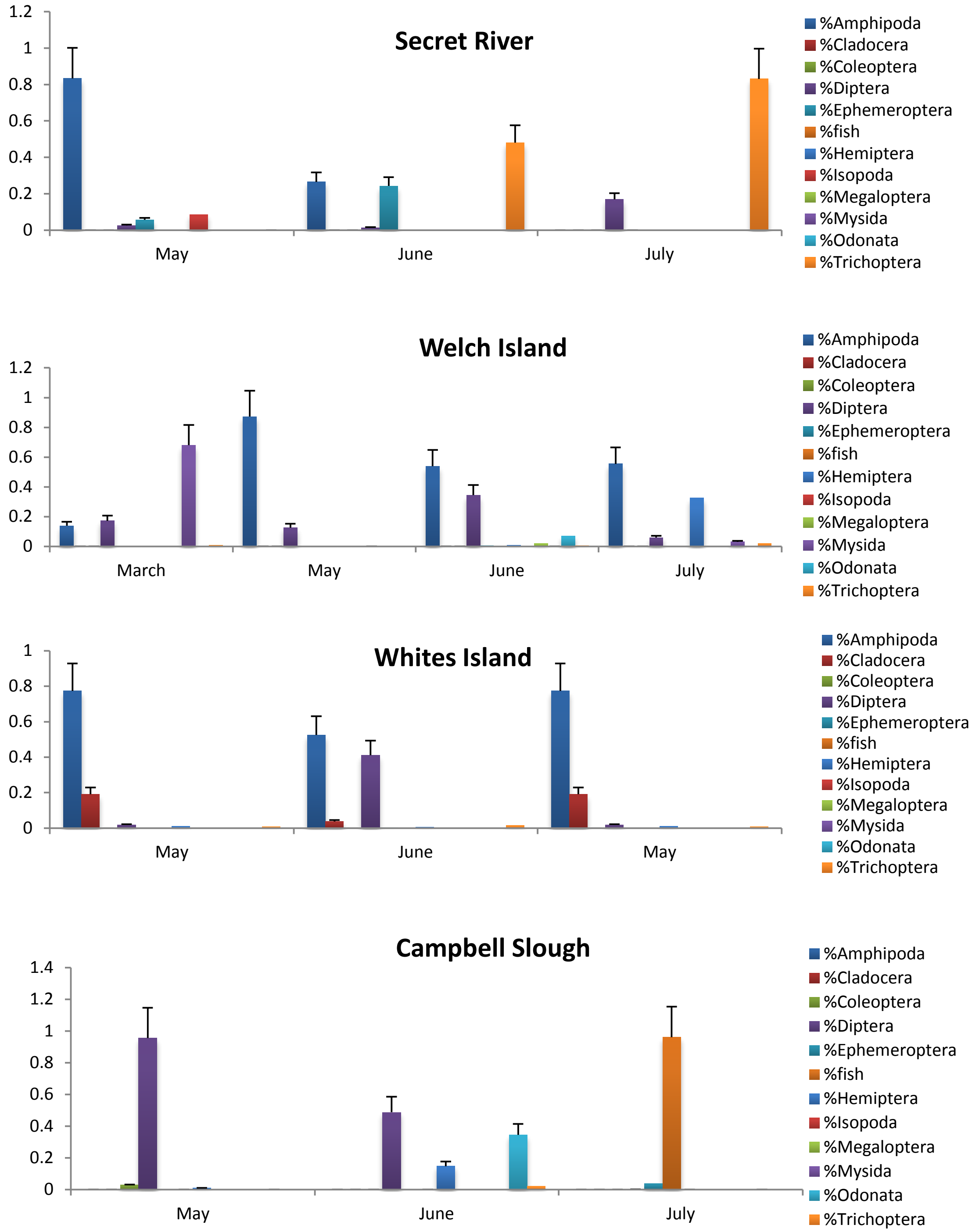


Figure 69. Mean (SD) percent composition (by weight) of prey in juvenile Chinook stomachs collected from Secret River, Welch Island, Whites, Island, and Campbell Slough in each month sampled in 2013. The 10 orders of prey presented here include on average over 95% of the consumed invertebrate prey for all fish, not just fish from these sampling events. Because not all orders are shown, the bars may not sum to 100%.

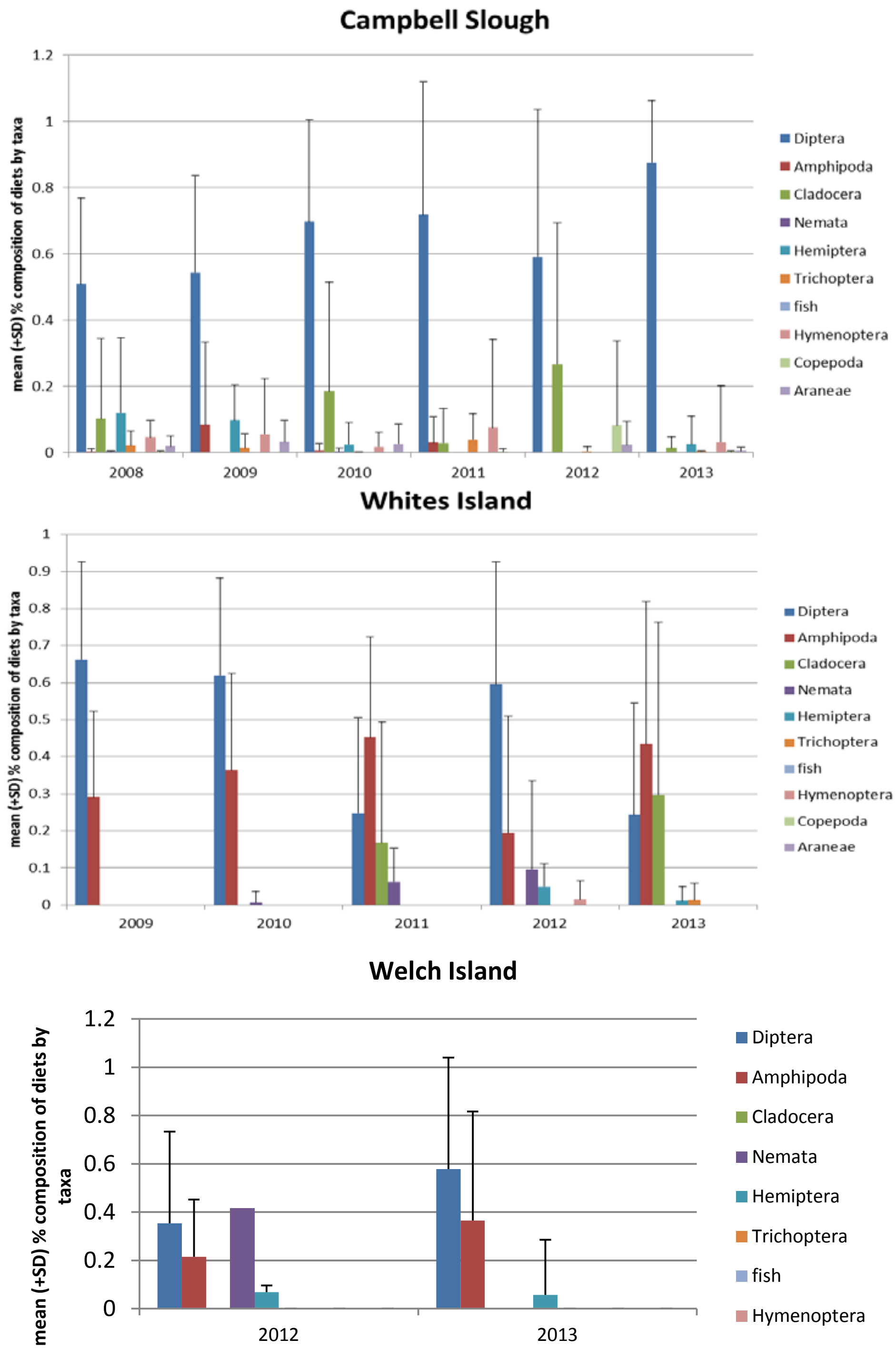
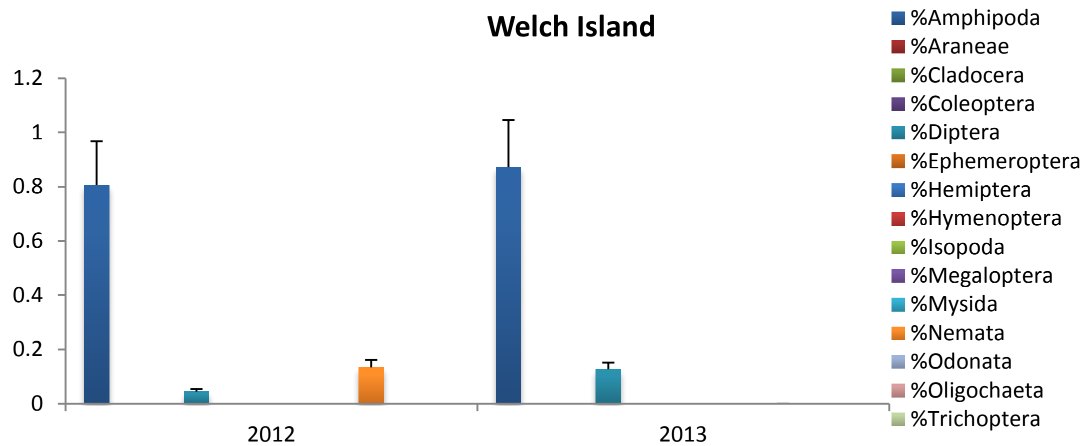
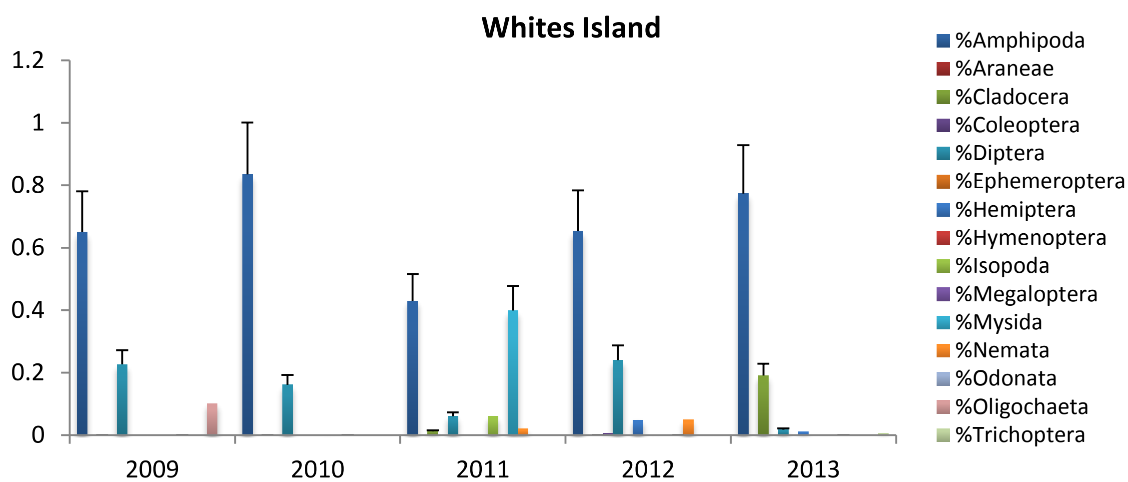


Figure 70. Mean (SD) percent composition (by abundance) of prey in juvenile Chinook stomachs collected from Campbell Slough (a) Whites Island (b) and Welch Island (c) each year in May. Secret River was not included as it was sampled in May only in 2013. The 10 orders of prey presented here include on average over 95% of the consumed invertebrate prey for all fish, not just fish from these sampling events. Because not all orders are shown, the bars may not sum to 100%.





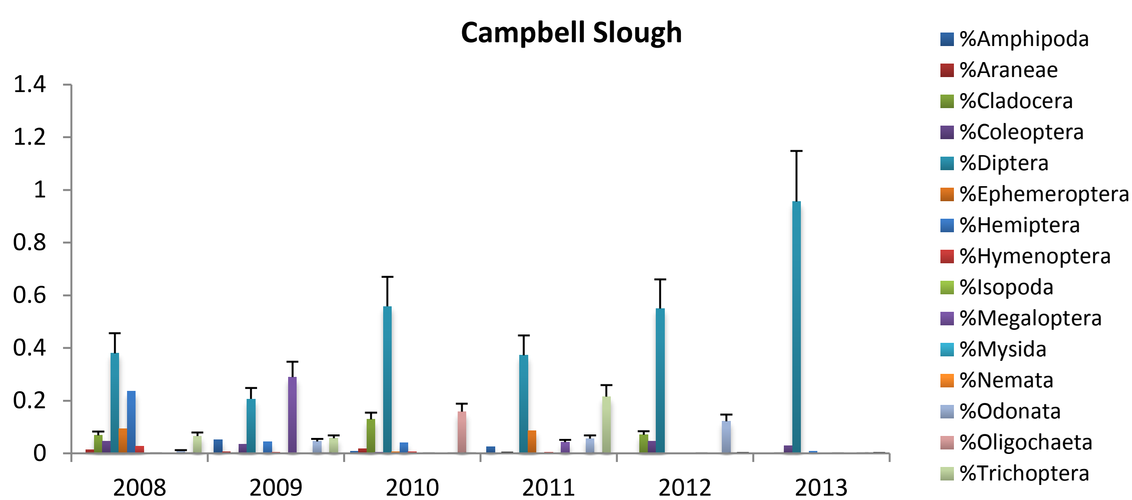


Figure 71. Mean (SD) percent composition (by weight) of prey in juvenile Chinook stomachs collected from Campbell Slough (a) Whites Island (b) and Welch Island (c) each year in May. Secret River was not included as it was sampled in May only in 2013.

Table 24. Ivlev's electivity values for Chinook salmon by site and by the month sampled in 2013.

Site	Month	Diptera	Amphipoda	Cladocera	Hemiptera	Trichoptera	Hymenoptera	Copepoda
Secret	May	-0.18	0.91	-1.00	-1.00		-1.00	-1.00
River	June	0.01	0.97		-1.00	0.72	-1.00	
Welch	March	0.07	1.00	-1.00	-1.00	-0.03		1.00
Island	May	-0.07	0.89	-1.00	0.56	-1.00	-1.00	1.00
	June	0.33	1.00		-0.88	.03		
	July	0.31	1.00	-1.00	-0.41	1.00	1.00	1.00
Whites	May	-0.38	0.71	1.00	-0.69	0.11	-1.00	-1.00
Island	June	0.27	0.66	-0.30	-0.98	1.00		-1.00
Campbell	May	0.31		-0.37	-0.31	-0.86	0.25	-0.53
Slough	June	0.51	-1.00	-1.00	-0.15	0.02	-1.00	-1.00

Table 25. Ivlev's electivity values for Chinook salmon by year for trend sites sampled in May.

Site	Year	Diptera	Amphipoda	Cladocera	Hemiptera	Trichoptera	Hymenoptera	Copepoda
Welch	2012	-0.44	1.00	-1.00	-0.28	-1.00		
Island	2013	-0.07	0.89	-1.00	0.56	-1.00	-1.00	1.00
Whites	2009	0.51	0.63		-1.00			
Island	2010	0.21	0.97	-1.00	-1.00	-1.00	-1.00	-1.00
	2011	-0.20	0.57	-1.00	-1.00			
	2012	0.34	0.72		1.00	-1.00	1.00	
	2013	-0.38	0.71	1.00	-0.69	0.11	-1.00	-1.00
Campbell	2008	0.38	-0.74	-0.20	0.56	0.92	1.00	-1.00
Slough	2009	0.16	0.30	-1.00	1.00	1.00	1.00	-1.00
	2010	0.22	0.70	0.44	-0.73	-0.83	0.52	-1.00
	2011	0.66	-0.12	-0.83	-1.00	0.85	0.96	-0.99
	2012	0.57	-1.00	-0.24	-1.00	-0.43	-1.00	-0.12
	2013	0.31		-0.37	-0.31	-0.86	0.25	-0.53

5 Discussion

5.1 Abiotic site conditions

In 2013, the three more river-dominated sites (Franz Lake Slough, Campbell Slough, and Whites Island) had similar water-quality characteristics that differed from those of Ilwaco. Ilwaco had the largest daily variation due to the strong tidal influence on that site. The spring freshet delivered cool water to all sites in late May, although the cooler temperatures persisted longer at Franz Lake Slough and Campbell Slough than at the sites that experience stronger tidal flushing due to their position in the estuary and proximity to the mainstem of the river (Whites Island and Ilwaco). There was not prolonged inundation of the two most upstream sites (Franz Lake Slough and Campbell Slough) in 2013 as occurred during and after the high flows of 2011 and 2012. The shallower channel depths in 2013 at Franz Lake Slough caused larger daily variation in water-quality parameters than in 2011 or 2012, particularly temperature and dissolved oxygen. At all sites, the largest magnitude daily variation in water-quality parameters was in mid to late July, when channel depths were the shallowest. Although all sites except Whites Island had daily median dissolved oxygen concentrations less than the 8.0 mg/L threshold (most notably Franz Lake Slough in July and Ilwaco in June–July), the daily maxima were never less than that threshold. Therefore, all the sites had dissolved oxygen concentrations that are suitable to salmonids during at least a portion of each day throughout the 2013 monitoring period. This was not the case at Campbell Slough during some previous monitoring years, particularly 2011, when dissolved oxygen concentrations were continuously low when the site was flooded during and after the freshet. As in previous years, Whites Island had the most suitable water-quality conditions for juvenile salmonids during the 2013 monitoring period. Ilwaco had the highest frequency of hourly measurements exceeding the thresholds, but the frequent flushing at that site also meant that conditions became suitable again much more quickly than at the slower-flushed upriver sites. Compared to 2011 and 2012, for which there are data from all sites, 2013 had generally warmer water temperatures, lower dissolved oxygen concentrations and higher maximum pH values. However, 2011 and 2012 were cool years with higher than normal Columbia River water levels. Therefore, Campbell Slough data (collected since 2009) show much more year-to-year variability during the period of record compared to the other three sites with data only from 2011–2013.

The three upstream sites had relatively similar nutrient conditions in 2013, particularly in comparison with Ilwaco. The higher nutrient concentrations at Ilwaco are typical of marine-influenced estuary sites compared to freshwater riverine sites. The low orthophosphate concentrations at the three upstream sites are typical for rivers, in which bioavailable orthophosphate is rapidly taken up by organisms, leaving little available in the water column. Orthophosphate concentrations in estuaries are often higher than in freshwaters because higher-pH seawater causes orthophosphate to desorb and become dissolved in the water column, as appears to be the case at Ilwaco. Compared to 2011-2012, nutrient concentrations were slightly lower in 2013 at the three upstream sites. In 2013, the three upstream sites had the highest nitrogen and phosphorus concentrations in mid-April or early May, then concentrations decreased during and after the freshet and remained low until early July.

5.2 Habitat Structure

Hydrology is the primary driver affecting the habitat structure of LCRE wetlands. The dynamic nature of the hydrology in this system has been previously shown to be linked to variation in vegetation species assemblages, cover, and biomass production (Sagar et al., 2013). In the lower estuary, the hydrologic patterns are driven primarily by tidal forces, with fluvial forces becoming increasingly dominant with distance from the mouth (Jay et al., in press). The hydrographs in Appendix A, provided for multiple years at the trend sites, offer a glimpse into seasonal and interannual variation and clearly show the shift from a dominance of tidal energy in the lower estuary to fluvial dominance in the upper estuary and lower River up to Bonneville dam. These hydrologic complexities likely affect other aspects of the habitat structure in addition to vegetation.

5.2.1 Sediment Accretion

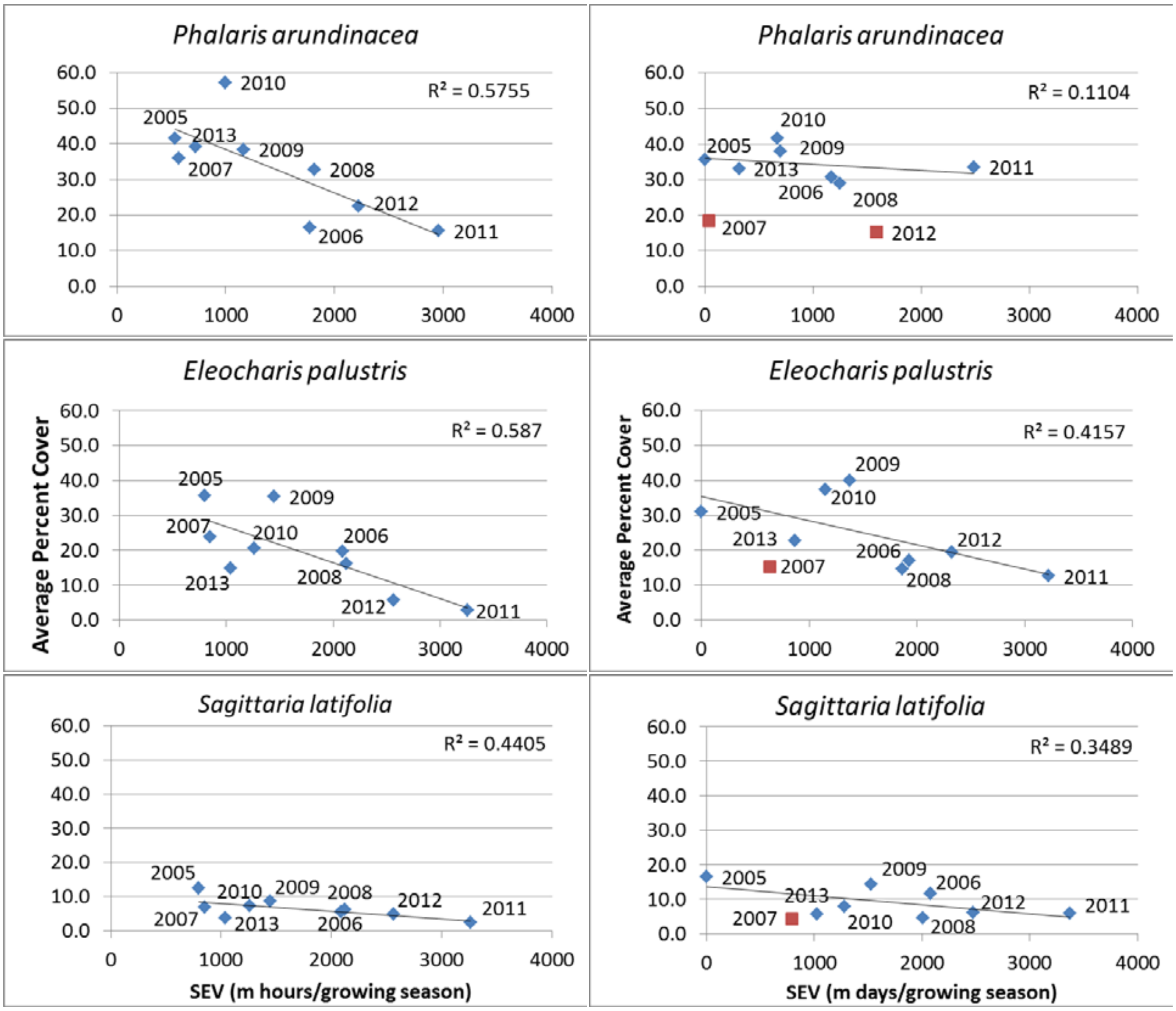
Sediment accretion rates are variable and are likely driven by multiple hydrologic factors including the sites location along the riverine-estuarine gradient, the magnitude and duration of the spring freshet, previous year flooding and sediment loading patterns, and the frequency and intensity of winter storms. Other factors likely play a role as well, including elevation, distance from the main stem, local geomorphology, and local disturbances. Examples of local disturbances include the placement of dredge material nearby and natural features such as the beaver dams at Franz Lake. Data from multiple years at the trend sites indicates that perhaps accretion rates are affected differently at the upper river sites compared to the lower river sites. With the sediment dynamics of the upper river wetlands affected more strongly by the Columbia River spring freshet and the lower river sediment dynamics affected also by winter storms that flush sediments from the tributaries into the River and adjacent wetlands. We hypothesize that in the upper river sediment accretes at a higher than average rate in high water years, then in subsequent years rates may be lower than average as the accumulated sediment is re-distributed. In the lower river sites, higher sediment accretion rates may occur in years when high intensity rain events have occurred.

5.2.2 Elevation, Inundation, and Vegetation Interactions

Species richness and cover is related to location in the River and the environmental conditions that affect each zone of the River differently. In Zone 1, the Ilwaco site had high cover and a relatively low number of species which can be attributed to the higher salinity that occur at the site. Elevation and salinity are likely the causes for the variation in reed canarygrass cover at the lower river sites. Salinity precludes the occurrence of reed canarygrass at the Ilwaco site. While at the Secret River and Welch Island sites, salinity is much lower and elevation is likely the primary driver for reed canarygrass occurrence. In general, reed canarygrass is more likely to occur at elevations greater than 1.5 m, CRD (Sagar et al. 2013). At the WI2 site elevations ranged from 0.9 m to 1.7 m, CRD, with reed canarygrass occurring only between 1.5 m and 1.7 m, CRD and therefore very little cover was observed there. Conversely, at the SRM high marsh site the whole area was within the range for reed canarygrass, with elevations ranging from 1.9 m to 2.2 m, CRD. At the WHC site, much of the site is also within the elevation range where reed canarygrass occurs and salinity is not a factor at this site.

The high water events in 2011 and 2012 affected the vegetation communities by reducing cover. We hypothesize that even in the lower river sites the effect of the high water in 2012 was evident. The lower cover of reed canarygrass at SRM, WI2, and WHC in 2012 and the higher occurrence of SAV at the SRM-L site in 2012 provide evidence of this At the SRM-H site the trend seems to be perhaps an increase in reed canarygrass cover when the results from 2008 are included in the analysis; however, 2008 was another relatively high water year which could have resulted in reduced reed canarygrass cover.

In Zone 4, cover was reduced during 2011 and 2012 the sites may not have fully recovered in 2013, especially at the lower elevations of the sites (Figure 72). At the Campbell Slough site, a higher number of species was observed in 2013 than in previous years, which is perhaps due to disturbances that have occurred at the site in recent years. In addition to flooding, the site has also been subject to periodic grazing by cows. While grazing and flooding have had a detrimental effect on the dominant species by reducing biomass and macrodetritus production, species diversity has increased in response to the disturbances. In one particular area of the Campbell Slough site, a cow trample path has developed and many small plants were observed in this area that had not been previously recorded at the site.



a) Cunningham Lake

b) Campbell Slough

Figure 72. 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.

In Zone 5, the shift of species dominance from reed canarygrass to the native water smartweed (*Polygonum amphibium*) appears to be directly attributable to the high water in 2011 and 2012. This species is often found in shallow water of lakes, ponds, and streams (WADOE 2001). The site had more cover of reed canarygrass in 2013 than the previous year, but the smartweed was still the dominant species, indicating that the effects of prolonged high water events can last beyond the year of impact.

We are still trying to understand the complex interaction between hydrologic patterns and reed canarygrass. The species has huge range of inundation tolerance and can withstand high seasonal inundation. However, it appears that it cannot withstand high tidal inundation; perhaps requiring a certain amount of exposure during the growing season. The inundation (as expressed by the SEV) ranges for when reed canarygrass was thriving at the trend sites are as follows:

SRM	250
WI2	300
WHC	500-800
CLM	900-1100
CS1	900-1100
FLM	2000-2800

We hypothesize that perhaps the species is most competitive in the middle ranges, when the growing season SEV is between 800 to 1800 m/growing season and while the species occurs and is even expanding at lower SEV's in the lower river, perhaps the competitive advantage is less. At Campbell Slough and Cunningham Lake, where the dominance of reed canarygrass has been highest, the greatest cover during our 9-year monitoring period occurred in 2010, when there was no winter high water and when growing season SEV was about 1000 m/growing season.

5.2.3 Tidal Channels

The tidal channels of LCRE wetlands provide many important functions including the transport of prey and macrodetritus out of the wetlands and the pathway for fish to access the sites for feeding and refuge. As such, it is important to understand the morphology and dynamics of the hydrologic patterns in these areas. We found that the channels at the trend sites have a similar morphology, however, as expected the frequency of inundation varies depending on the location in the river, the magnitude of the spring freshet, and the gradient of the channel. The lower river channels provide less frequent access to juvenile salmon than in the upper river during the peak migration period; however, the daily fluctuation in water levels likely provides more frequent access to the marsh channel interface than the more constant high water that occurs with the spring freshet in the upper river.

5.3 Food Web

Data collected as part of the EMP on emergent wetland plant biomass production is improving our understanding of the macrodetritus contribution from these emergent wetlands in the LCRE. Data from six sites along the estuarine gradient, collected over a three-year period, provide some insight to patterns and trends in biomass production and the resulting detritus contribution. Three wetland strata were sampled as part of this study: high marsh, low marsh, and submerged aquatic vegetation (SAV). At sites sampled, high marsh had the greatest plant biomass (average of 929 g/m^2), compared to low marsh and SAV (average of 249 g/m^2 and 42 g/m^2, respectively) and the four lower river sites had greater biomass than the two upper river sites (high marsh average of 1162 g/m^{2}and 426 g/m^2, respectively). This spatial trend was likely due to the effects of the two high water years in a row, which reduced the vegetation cover and biomass production in the upper river sites during all three years of the study. Additional data and discussion of biomass production and wetland macrodetritus contribution is provided in the most recent Ecosystem Monitoring synthesis report for this program (Sagar et al., 2014).

Similar to patterns in dissolved nutrient concentrations, the phytoplankton biomass estimated by chlorophyll *a* was higher in April and May 2013, and appeared not to have recovered from the high spring flows by late June. Periphyton abundance decreased over the course of the sampling season at Franz Lake Slough and Campbell Slough in 2013. However, at Whites Island and Ilwaco, where periphyton abundances were much higher than at the upstream sites, periphyton abundance peaked in May and remained high into late June. Generally, biomass levels associated with phytoplankton and periphyton (as estimated by chlorophyll *a* concentrations) were more consistent over the 2013 sampling season compared to 2011 or 2012. Overall, the pattern of higher phytoplankton biomass at the upstream sites and higher periphyton abundance at the downstream sites in 2013 is consistent with data from 2011–2012.

As in previous years (Sagar et al., 2013), the total abundance of phytoplankton inversely tracked the magnitude of Columbia River flow. Further, the data qualitatively suggest a difference in the magnitude and composition of planktonic assemblages upstream versus downstream of the Willamette-Columbia confluence (for example, when data from Franz Lake Slough was compared with the downstream sites); however, the different habitat characteristics at the reach scale of the EMP sites obfuscate the comparisons somewhat. In other systems, reach-scale differences in indicators of stream health (for example, Indices of Biotic Integrity of fish) are strongly influenced by geological features, making stream health less predictable at the reach scale compared to the watershed scale (Frimpong et al. 2005). A study of the effects of land use on physical habitats, macroinvertebrates, and fish communities found that local habitat differences were poor predictors of the health of aquatic biota in rivers and streams (Nerbonne and Vondracek, 2001). These studies emphasize that local differences in habitat may be poor predictors of stream or river health at the landscape scale over which migrating fish species are found. Highly motile organisms like fish likely use a variety of habitats that together comprise their effective habitat. Thus, a combination of characteristics integrated over space (and to some degree, time) may strongly influence the health of biotic communities within a river system and prudent management of these systems should therefore take an integrated approach that considers linkages between habitats at the reach and watershed scales.

Among the shallow-water EMP sites, the planktonic communities at Ilwaco were most different than the other sites in terms of abundance and species composition. Similar to the other sites, the phytoplankton community was dominated by diatoms (Class Bacillariophyceae) at Ilwaco; however, within the broad class level, the dominant species differed than those upstream. As opposed to the more upstream sites, many more benthic forms were observed. This is consistent with the higher rates of periphyton production, which could have been included epibenthic or epiphytic diatom species. The strong marine influence at Ilwaco in Reach A near the river mouth was associated with larger influxes of marine-derived nutrients, especially phosphate, which can be limiting to phytoplankton growth in the Columbia River in the spring and summer. The continuous hourly monitoring of nitrate and phosphate at the mainstem sites clearly showed a deficit in phosphate relative to nitrate during the growing season. Grow-out experiments using whole-water samples incubated in bottles clearly showed a positive growth response to the addition of ortho-phosphate, providing strong support that low levels of phosphosur limit pelagic primary production in the system. In other systems, for example the Clyde River in Australia, bacteria have been shown to outcompete phytoplankton for limiting nutrients, particularly when sufficient dissolved organic carbon is available to fuel bacterial growth (Hitchcock and Mitrovic, 2013).

Prior to the spring freshet, diatoms dominated the assemblages at all sites in the lower Columbia, while during periods of lower river flow velocities other species including green algae (Class Chlorophyceae) and cyanobacteria (Class Cyanophyceae) increased in relative abundance. Although the plankton communities at Campbell Slough were not as different from the other sites as were those from Ilwaco, they exhibited distinct temporal patterns in total and relative abundance that were associated with the higher nutrients and slower flushing at this site. In the summer, elevated abundances of cyanobacteria (including harmful species capable of producing hepatotoxins) were observed at Campbell Slough.

The isotopic signatures of carbon and nitrogen in the dominant salmon prey (dipterans and amphipods) suggested that phytoplankton made up an important food source at some times of the year, particularly during periods of high biomass (i.e., spring "blooms"). How much the size of the phytoplankton populations influences prey productivity is unclear; however, the fact that the isotope signatures track those of phytoplankton suggests that there is a relationship between the two that should be further explored. In a comparison of macroinvertebrate populations associated with native water stargrass (*Heteranther dubia*) vs. non-native Eurasian milfoil (*Myriophyllum spicatum*) in experimental ponds in Texas, it was found that the abundances of macroinvertebrates (dominated by chironomid species) were associated with the abundance of epiphyton biomass, which differed among the plant species (Balci and Kennedy, 2003). Thus, levels of phytoplankton (particularly epibenthic species) may provide an important link between primary production and the availability of preferred prey items for juvenile salmonids.

Similar to the case for phytoplankton, zooplankton abundances were higher during periods of lower river flow compared to high river flow. High water levels likely have two main functions in terms of an influence on planktonic community structure. First, the rapid flow and high volumes of water originating from snowmelt likely lead to the dilution of populations present in the system; second, high water levels function to connect backwater habitats to the river mainstem. The biggest spatial difference in zooplankton assemblages was the dominance of copepods at Ilwaco at all times of the year versus the other sites, which were more variable over time. At each of the other sites, rotifers (small zooplankton) were occasionally present at high abundance, particularly earlier in the season, while crustacean zooplankton (copepods and cladocerans) generally dominated later in the season. From the stable isotope signatures and stomach content analyses, the pelagic crustaceans (copepods and cladocerans) were consumed by juvenile salmon, although they did not seem to be preferred prey items.

The highest zooplankton abundances were observed at Campbell Slough in the summer, which was the site characterized by the highest proportion of cyanobacteria. Given the high densities of copepods and cladocerans present coincident with the cyanobacteria, it is quite possible that selective grazing of diatoms over less palatable species such as cyanobacteria caused a shift in the relative abundance of the different phytoplankton groups. Campbell Slough had the highest nutrient concentrations and lowest levels of dissolved oxygen, which are both associated with cultural eutrophication.

5.4. Fish Use

In 2013, our sampling was focused on revisiting the trend sites, Ilwaco Slough in Reach A, Secret River and Welch Island in Reach B, Whites Island in Reach C, Campbell Slough in Reach F, and Franz Lake in Reach H to collect additional information on temporal trends in these areas. As in past years, due to high water conditions and other problems with site access, our ability to monitor salmon occurrence was limited at Franz Lake, and to a lesser extent, at Campbell Slough, compromising our ability to assess trends in fish use accurately at these sites. This was especially a problem at Franz Lake, which we have not been able to sample between April and June, the peak period of juvenile Chinook salmon occurrence, since 2009. However, at the sites we have been able to sample consistently, our 2013 data show patterns similar to those we have observed in previously in this program (e.g., see Sagar et al. 2013).

In 2013, as in previous years, fish community characteristics showed distinctive spatial patterns along the river. At the Ilwaco Slough site in Reach A, catches contained saltwater species not present at the other sampling sites. Fish assemblages found at Secret River and Welch Island, in Reach B, and Whites Island in Reach C, were very similar, with high proportions of stickleback and smaller numbers of mostly native species. At Campbell Slough in Reach F and Franz Lake in Reach H, we observed higher proportions of non-native species, including carp, killifish, pumpkinseed, and yellow perch, as well as potential predators of juvenile salmon such as smallmouth and largemouth bass , walleye, and northern pikeminnow. Fish species diversity and species richness were also higher at Campbell Slough and Franz Lake than at Ilwaco Slough, Secret River, Welch Island, and Whites Island, a pattern we have observed in earlier samplings. Values for several of these parameters were lower at Franz Lake in 2013 than the average values observed in earlier years, but this may be because we were not able to sample at this site during the summer months when species diversity is greatest and proportions of non-native species are highest (see Sagar et al. 2013).

The salmon species found at the trend sites in 2013 were also generally similar to those seen in earlier samplings. Chum salmon dominated the salmon catch at Ilwaco Slough. Chinook and coho salmon were rarely found and then only in small numbers, even though the site was among those most consistently sampled between February until December. This finding, as well as similar observations in 2011 and 2012, suggests that, with the possible exception of outmigrant chum, Ilwaco Slough is not heavily utilized by juvenile salmon. This pattern is not characteristic of other sites that have been studied in Reach A. For example, other researchers report much higher abundances of Chinook salmon at sites such as Point Adams and West Sand Island (Roegner et al. 2008; Bottom et al. 2005). These sites, however, are not representative of the emergent marsh habitat found at Ilwaco Slough, so may not be an appropriate comparison.

As in past years, Chinook salmon were the dominant species at Secret River, Welch Island, Whites Island, and Campbell Slough; Secret River and Welch Island also supported small numbers of chum salmon, as well as coho salmon at Secret River. Coho salmon have also been found at Ilwaco Slough, Welch Island, and Whites Island in previous years, but never in large numbers, so their absence in 2013 was not unusual. Trends at Franz Lake cannot really be evaluated, as it was sampled only for a limited part of the season, but two Chinook and two coho salmon were found at the site. While this is a very small number of fish, it is consistent with earlier observations that the Franz Lake site supports both Chinook and coho salmon (Sagar et al. 2013).

The percentages of marked Chinook salmon of hatchery origin were very low at Secret River, and increased gradually upstream to Campbell Slough, where 42% of Chinook salmon were marked, a patterns consistent with observations in previous years. The coho salmon collected at Welch Island were also unmarked. This pattern is consistent with our earlier observations that unmarked, presumably wild fish predominate in the lower reaches, with increased percentages of hatchery fish upriver. Franz Lake was the exception to this pattern; as this site, only unmarked fish were found in 2013, although in earlier sampling has supported both marked and unmarked coho and Chinook salmon. Indeed, a more detailed analysis on variance in Ecosystem Monitoring data (Sagar et al. 2014) suggests that at Franz Lake, the proportion of marked coho and Chinook salmon in catches has decreased between 2008 and 2013. While this change could represent a trend in relative abundance of marked hatchery fish and unmarked, presumably wild fish, it is more likely due to the fact that since 2009 we have been unable to sample Franz Lake in May and June, when the majority of hatchery fish are released (Columbia River DART; http://www.cbr.washington.edu/dart/hatch.html).

Densities of Chinook, coho, and chum salmon at the trend sites in 2013 also generally conformed to previously observed patterns, though there were some exceptions. Of the species observed, unmarked

Chinook salmon were present at the highest densities, 20-40 fish per 1000 m^2 at most sites. Unmarked Chinook salmon density was highest at Welch Island, and lowest at Ilwaco Slough and Franz Lake, with intermediate values at the other sites. In comparison with previous years, densities of unmarked Chinook salmon in 2013 tended to be lower at the Reach B and C sites, though density measurements can be quite variable and the reason for this apparent decline is unclear. In contrast to the density of unmarked Chinook salmon, the density of marked Chinook salmon tended to increase with increasing distance from the mouth of the river, a pattern we have observed before, with highest values in 2013 Campbell Slough. In earlier years, the density of marked Chinook salmon has also been quite high at Franz Lake, but no marked Chinook were found in 2013, likely, as noted earlier, because of the site could not be sampled during peak hatchery releases. Coho salmon densities were low at all sites in 2013, as they have been at in previous years at most of the trend sites. Again, Franz Lake is an exception; the low densities of both marked and unmarked coho salmon are in contrast to data from earlier years, and most likely related to limited sampling. Like coho densities, densities of chum salmon were quite low at most of the EMP sites in 2013, as they have been in previous years. The exception was Ilwaco Slough, where relatively high densities of chum were observed. In comparison to the average chum density over the past sampling years, the density in 2013 was somewhat higher.

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. Almost no salmon were present in August or September. Marked Chinook salmon were found over the same time period, but only rarely except in May and June, when they were found at the highest densities, 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 found in small numbers from February through December. 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). Because of work interruptions associated with the government shut-down, we were unable to carry out a full fall and winter sampling in 2013, and no data are available for October or November. Only Ilwaco Slough and Franz Lake were successfully sampled in December; Campbell Slough could not be sampled because of access restrictions at the wildlife refuge, and Secret River and Welch Island could not be fished because strong tides and weather. However, the data we were able to collect confirmed the importance of late winter as a rearing and migration period for juvenile salmon, with unmarked coho salmon present at the Franz Lake site. 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), as is the presence of small number of Chinook and coho salmon in winter months (Sather et al. 2009; Johnson et al. 2011).

Data are not yet available for growth, lipid content, or contaminants, but significant differences were found among sites in size class distribution of unmarked Chinook salmon. As in previous years, Secret River and Welch Island had the highest proportions of juvenile Chinook salmon fry. 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. Fingerlings were most abundant at Campbell Slough and Franz Lake. This pattern is typical for Campbell Slough. However, in past years, Franz Lake has also supported a relatively high proportion of Chinook salmon fry, so the discrepancy may be due to our inability to sample the site very extensively during early spring when fry are most abundant. This problem may apply to a lesser extent at Campbell Slough as well, as we have been unable to access the site in February and March when fry are most common. As in past years, clear patterns in condition factor by site were not apparent, although there were indications of increased condition factor of unmarked Chinook salmon at Secret River and Welch Island in 2013. Because these two sites have only been sampled in 2012 and 2013 so far, it is hard to interpret this finding. These differences could easily be within the normal range of variability at this site. Marked Chinook salmon, as usual showed high uniformity is size and life history stage, with almost all fish in the fingerling size class. There was some variation by year even for the marked fish, with the average size at Campbell Slough being smaller in 2013 than in previous years.

Unmarked coho salmon were found only at Secret River and Franz Lake in 2013, but included fish in a range of size classes, a pattern that has also been seen in earlier samplings. Since they were found only in small numbers and not at all sites, trends in coho condition factor were difficult to evaluate, although coho condition factor, like Chinook condition factor appeared higher in 2013 at Secret River than in 2012. Again, the importance of this difference is uncertain, as we have only two years of data and little information on the range of variation at this site.

Chinook salmon stock data are not yet available for 2013, but the 2012 data, which were not included in last year's report, show that Chinook salmon stock usage of the EMP sampling sites changes spatially along the river. Among unmarked fish, West Cascades fall Chinook are more abundant at the sites in Reaches A through C, with Upper Columbia summer/fall Chinook being the most prevalent stock at Campbell Slough in Reach F and Lemon Island in Reach G. Surprisingly, West Cascades fall Chinook were the most abundant stock at Washougal, a pattern we have typically seen more frequently at sites in lower reaches. Too few Chinook salmon have been caught at Ilwaco Slough to evaluate stock composition, but at Whites Island and Campbell Slough stock composition patterns in 2012 were similar to previous years, with West Cascades fall Chinook dominating catches at Whites Island and Campbell Slough supporting a more diverse array of stocks. As we have observed previously (Sagar et al. 2013) marked Chinook salmon were for the most part West Cascades fall Chinook and Spring Creek Group fall Chinook, with West Cascades fall Chinook being more prevalent at Secret River and Welch Island in Reach B, and Spring Creek fall Chinook being most prevalent at the other sites. Lemon Island was unusual in that it had a high proportion of marked Chinook from the Upper Columbia River summer/fall stock. In comparison to stock composition patterns at the Whites Island, and Campbell Slough trends sites in previous years, those at Whites Island were similar, but at Campbell Slough, the proportion of Spring Creek fall Chinook was lower than usual. The addition of data from 2012 also support our earlier observation (Sagar et al. 2013) that stock distribution changes with sampling time for some stocks of both marked and unmarked Chinook salmon. Spring Creek 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.

In summary, for those trend sites that have been sampled over a consistent seasonal period, 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 from 2011 to 2013.

5.3.1 Fish Prey

In 2013, as in our early samplings, we found that aquatic Diptera and Amphipods contributed the majority of the consumed prey overall, in biomass and in numbers at the EMP 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.

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, while 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.

5.4 Mainstem conditions

5.4.1 Water quality

RM-122 biogeochemical measurements in the mainstem Columbia River provide important data upstream of the Willamette River, and when compared with the RM-53 platform can be used to better understand how conditions change throughout the entire lower river and estuary. For example, water temperature between the two sites were found to be nearly the same at all times of year, including during the warmest periods (Figure 10). This observation implies that local heating and cooling has no influence on the mainstem water temperature, and that the Willamette and other tributaries do not change the water temperature of the Columbia River as measured at RM-53. Instead, water temperature follows seasonal climate conditions and is likely a result of upstream processes. As noted in the 2012 annual report and elsewhere, water temperatures above the recognized threshold for suitable salmon habitat (19°C, Bottom et al. 2011) occur each summer in the mainstem portion of the river. Temperatures above 19°C occurred for more days in 2013 than in the previous two years (Table 13), a result that is correlated to lower discharge levels associated with the spring freshet as compared to 2011 and 2012 (Figure 8). Given that the temperature between the two sites is similar at all times of the year and that there is no gradient in temperature with water depth, we conclude that the observed periods when temperature surpasses 19°C represent the *lowest* temperatures in the mainstem lower Columbia River for these time periods. Therefore, if salmonids are seeking cold water refuges during this time, they would need to be out of the mainstem Columbia River and likely associated with localized regions of cold water input, perhaps from small mountainous rivers or underground springs.

In contrast to temperature, other water quality indicators change significantly between upstream and downstream of the Willamette River. For example, fluxes and concentrations of CDOM, nitrate, and turbidity increase during winter storms at RM-53 but show little or no response at RM-122 (Figure 10). This observation supports earlier evidence that winter storms cause increased precipitation and discharge to the lower Columbia River, but upstream of the Willamette River confluence the discharge is not significantly affected, either as a result of hydropower management or because precipitation falls as snow and therefore does not lead to increased runoff. The difference between fluxes at the two sites can be precisely quantified by summation of the daily nitrate flux calculation (i.e. from the data presented in Figure 11). For the period 09/2012-09/2013 the nitrogen flux of nitrate at RM-53 was 64.6 thousand metric tons. The N flux measured for the same time period at RM-122 was 34.8 thousand metric tons. Therefore, a 46% increase in nitrogen flux occurred downstream of RM-122. This finding has significant implications for predicting future water quality scenarios in the lower Columbia River and estuary, since tributaries are responsible for approximately half of the total nutrient input to the lower river nutrient budget. Similar calculations were made for other important biogeochemical parameters (Table 26). The data show similar results for turbidity (61% increase) and a lesser influence of CDOM (26% increase). The smallest change was in the chlorophyll *a* flux (10% increase), which reflects the different biogeochemical factors influencing chlorophyll *a* concentrations compared to the other parameters (discussed below). We can attribute most of the increased flux in Table 26 to the episodic events that occurred during winter of 2013, therefore these events have a significant contribution to the annual flux of material to the estuary and ocean. Similar findings have been reported for many of the smaller mountainous rivers in the Oregon coastal range (Wheatcroft et al. 2010) and suggest that year round monitoring is critical to understand movement of materials through rivers in the Pacific Northwest.

Table 26. Calculation of % increase in fluxes between RM-122 and RM-53 for the time period 09/2012-09/2013.

	RM-122	RM-53	% Increase
Nitrate	3.5E+04	6.5E+04	46
Turbidity	3.9E+11	9.9E+11	61
CDOM	2.0E+12	2.7E+12	26
Chlorophyll *a*	4.0E+07	4.4E+07	10

5.4.2 Chlorophyll *a*

The spring bloom associated with phytoplankton growth in the mainstem is a dominant feature of the entire Lower River during March-May as evidenced by chlorophyll *a* concentrations measured at both fixed stations. Concentrations > 40 μg/L were measured during the 2013 bloom and occurred during a period of relatively low rainfall for the region (Figure 10). The term “spring bloom” is used for many temperate latitude ecosystems since it is a common phenomenon driven by increased day length in spring that overcomes light limitation for phytoplankton growth. In most instances, blooms decline as a result of either exhaustion of nutrients or by herbivory from grazers and filter feeders. However in the Columbia River there are multiple other explanations for the decline, including: 1) episodic storms that increase turbidity and decrease light availability, 2) infection by chytrid fungi, and 3) increased discharge from the Columbia River freshet. Most likely several factors influence the spring bloom decline although more research is needed to address the relative contributions of these factors.

The seasonal flux of chlorophyll *a* was also similar at the two locations (Table 26). This is in contrast to nitrate, turbidity, and CDOM. The reason for the different behavior is due to the fundamental difference between material that is derived in the watershed and enters the river during runoff versus material that grows in the river. Since the large episodic storms that result in large runoff occur in winter when phytoplankton are low, there is only a small influence on the chlorophyll *a* flux. However, when runoff is low and phytoplankton are growing rapidly, fluxes are high. The small increase in flux between RM-122 and RM-53 is more easily explained as the amount added by *in situ* growth *between* RM-122 and RM-53.

5.4.3 Primary production

Following the freshet, there is a summer bloom that also persists throughout the river (as measured by the two observational platforms). Based on multiple years of data at RM-53 and supported by the 2013 data from RM-122, the summer biomass of phytoplankton is typically lower than the preceding spring bloom. Two likely reasons for this observation are nutrient limitation of growth and top-down control of biomass by grazing. It is important to distinguish between these two mechanisms, since nutrient limitation implies that increased growth of phytoplankton may occur under scenarios where nutrient loading increases, whereas grazing control of biomass implies that phytoplankton growth rates are normal but because of grazing the biomass does not increase. This would lead to a large transfer of carbon to the food web relative to a nutrient limited food web. In fact, there is evidence for both mechanisms to operate in the Columbia River. As described in the 2012 annual report (Sagar et al. 2013), Net Ecosystem Metabolism (NEM) can provide a direct measure of phytoplankton growth rates. This measure of growth is based on changes in dissolved oxygen concentration and thus does not depend on biomass measurements. A careful analysis of NEM was presented in the 2012 annual report for RM-53, and from those data it is evident that the summer period had much lower chlorophyll *a* concentrations than the spring bloom, however Gross Primary Production (GPP) was similar during the two time periods (Figure 73). This is strong support for grazing as an important factor in summer for controlling biomass concentrations, however direct measurement of grazing rates were not conducted. These measurements should be attempted in future work as a means to constrain the carbon flow from phytoplankton into the food web.

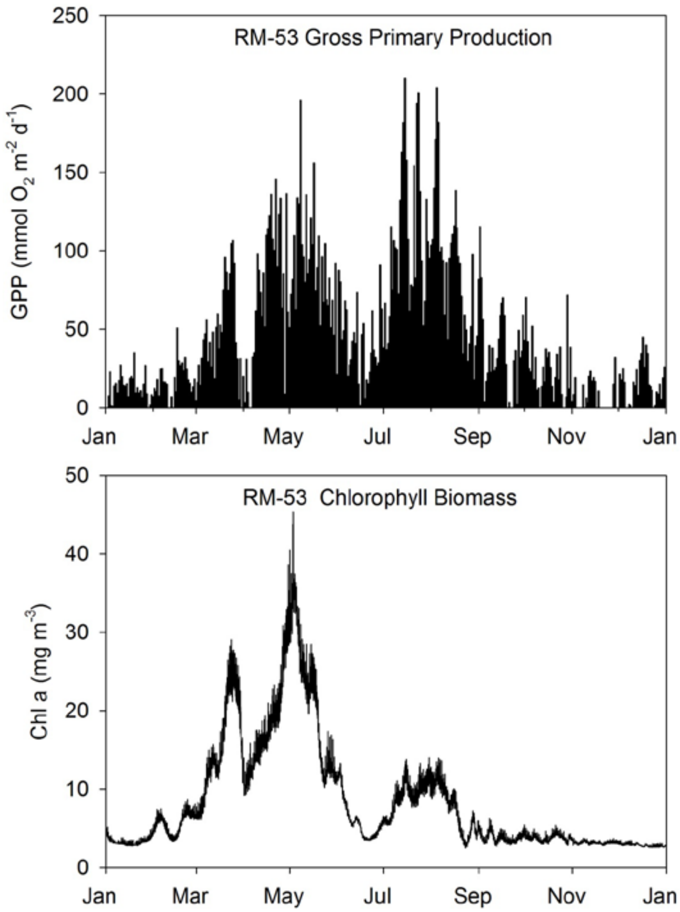


Figure 73. Comparison of Gross Primary Production GPP and chlorophyll *a* biomass at RM-53 for 2010.

Phosphorus limitation was also identified in the 2012 report as a probable limiting factor for phytoplankton growth. This has been supported by the RM-122 data (Figure 13) that indicate that nitrate to phosphate ratios are always above 16:1, which is the approximate ratio of phytoplankton nutrient requirements. Especially in winter and spring, phytoplankton would be predicted to always deplete phosphate before nitrate if growth were otherwise unrestricted. A simple experiment was conducted during the 2013 spring bloom to further test this hypothesis. Triplicate water samples from RM-122 were collected into 1 L bottles and incubated in natural light conditions to determine growth response to the addition of inorganic phosphorus. Each experiment bottle received 2 μM of phosphate and was allowed to grow for three days. The control bottles did not receive any phosphorus addition. The results clearly showed that the natural populations that were present were able to grow significantly higher with the presence of additional phosphorus (Figure 74), therefore supporting other evidence that phosphorus is a factor that limits phytoplankton biomass in the spring.

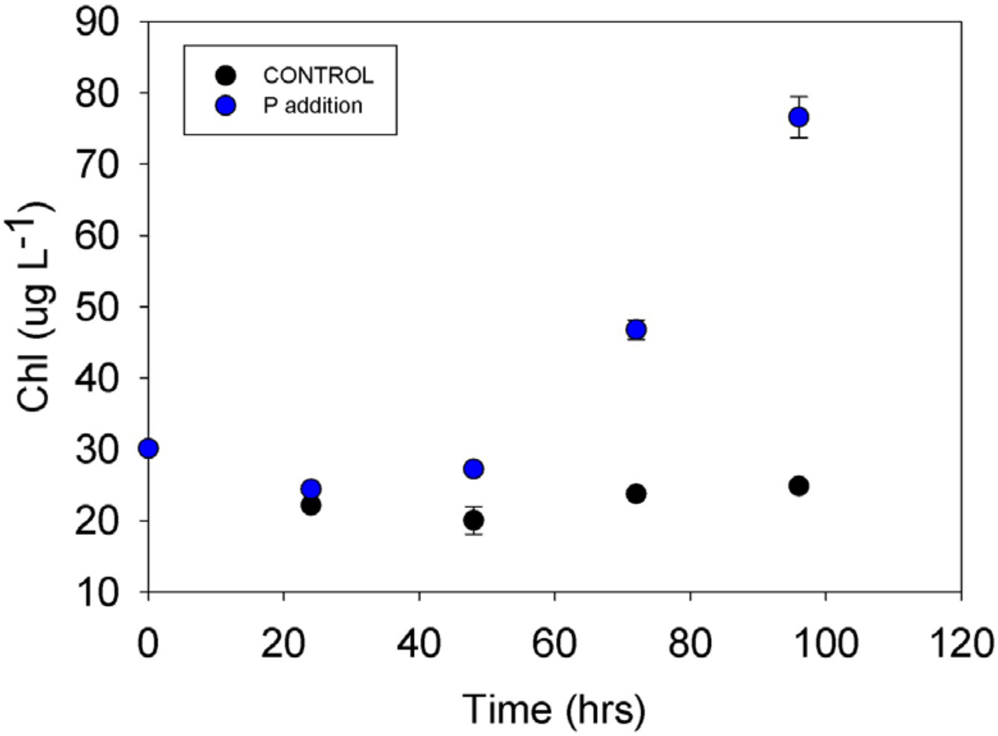


Figure 74. Results from phosphate-addition experiment at RM-122 in July 2013. Control bottles consisted of whole water samples incubated without addition of phosphate, while P addition bottles included an amendment of +2 µM phosphate. Error bars represent the standard deviation of triplicate bottles.

5.5 Summary and future recommendations

Understanding the links between abiotic conditions (i.e., water quality) and the sources of carbon for higher trophic levels is central to any ecosystem assessment or monitoring program. By accurately characterizing nutrients, energy flow, and water quality conditions such as dissolved oxygen concentrations, it is possible to better assess overall ecosystem function for targeted species such as Chinook salmon. Nitrate can cause water quality issues if it is used by plants and algae as a nutrient source, leading to eutrophication (Cloern 2001). Increased DOC and turbidity from runoff are associated with a range of changes to ecosystems including reduced algae growth, underwater vision for predators, and increased sedimentation (Cloern 2001).

In addition, more information about the temporal variability of the primary producers in the mainstem portion of the river (i.e., phytoplankton) and their prey is critical to inform future food web studies, including those associated with reed canarygrass and other emergent habitats where particulate material from the river likely accumulates on the benthos and is consumed by zooplankton and macroinvertebrates that colonize those habitats. The mechanisms and quantification of the phytoplankton – prey – salmon food chain are very poorly understood for the system. This food chain may also be the mechanisms through which contaminants are transferred from water to fish, and should be an active and complementary area of research.

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7 Appendices

Appendix A. Site Hydrographs

Hydrographs are in order by site location in the River, starting at the mouth.

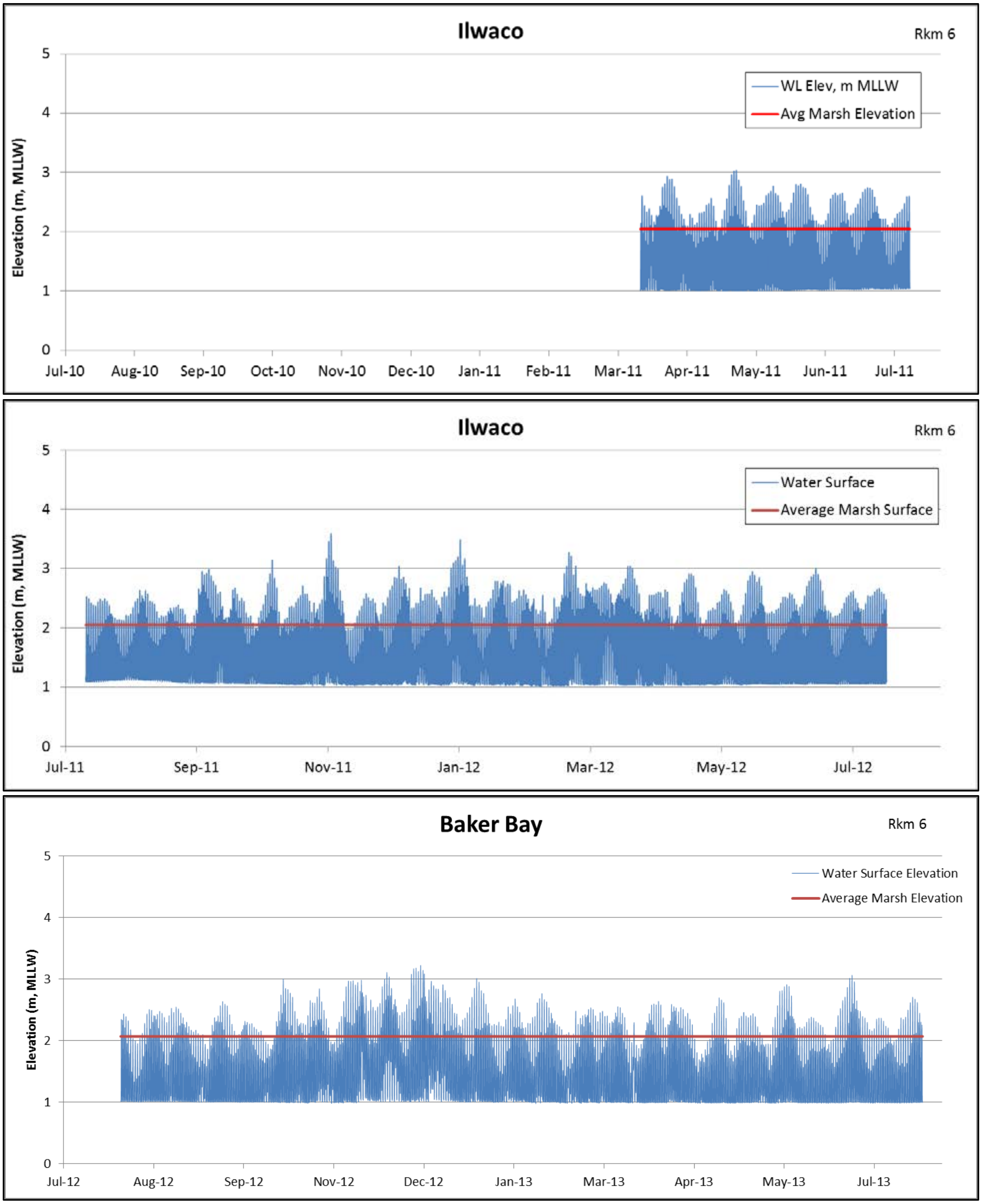


Figure A - 1. Water surface elevation data from the Ilwaco study site for the years 2011-2013. The red line represents the average elevation of the marsh sampling area.

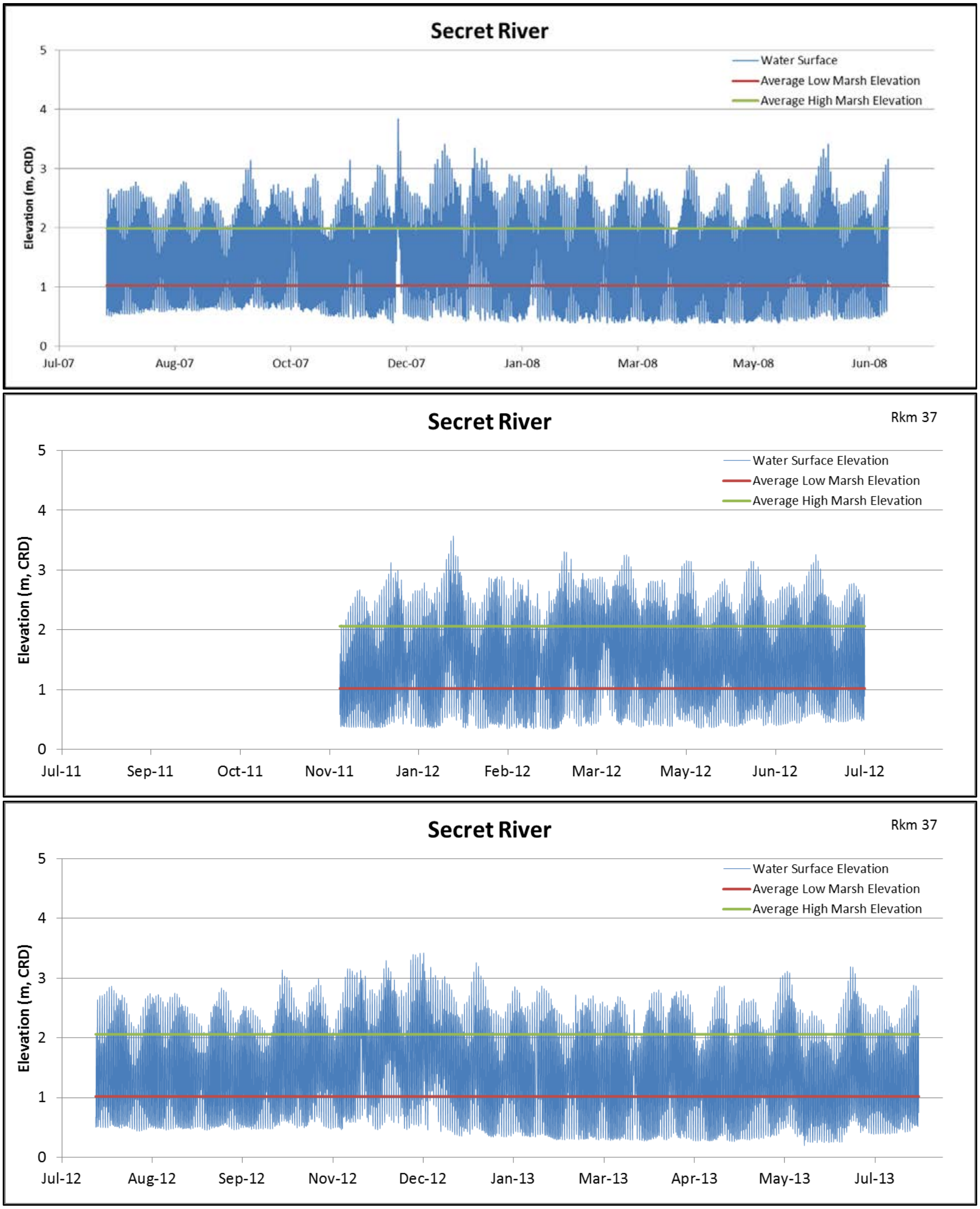


Figure A -2. Water surface elevation data from the Secret River study site for the years 2007-2008 and 2012-2013. The red line represents the average elevation of the marsh sampling area.

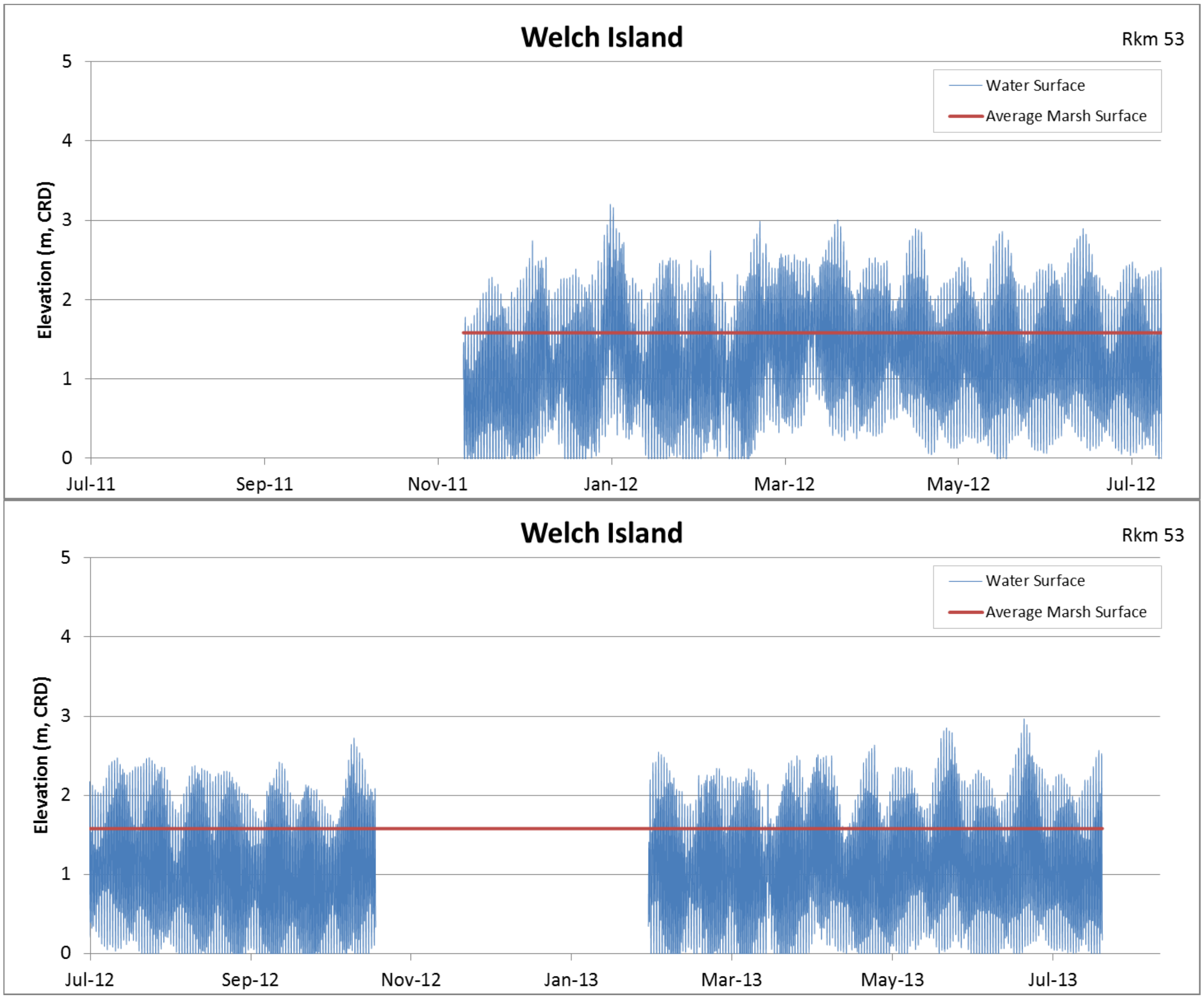


Figure A -3. Water surface elevation data from the Welch Island study site for the years 2012-2013. The red line represents the average elevation of the marsh sampling area. The sensor was displaced between early November and February.

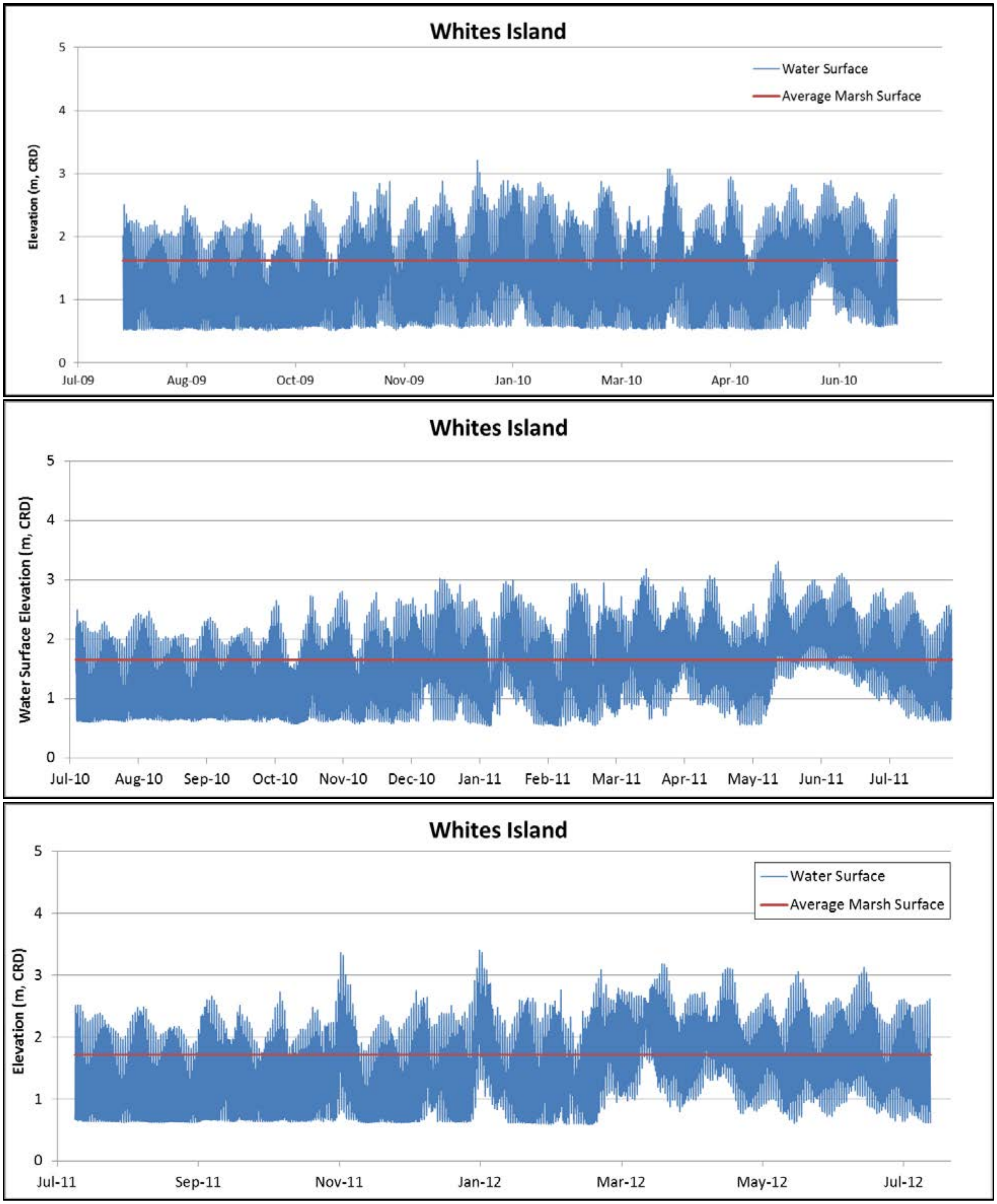


Figure A -4. Water surface elevation data from the Whites Island study site for the years 2009-2012. The red line represents the average elevation of the marsh sampling area. No data from 2013 due to sensor failure.

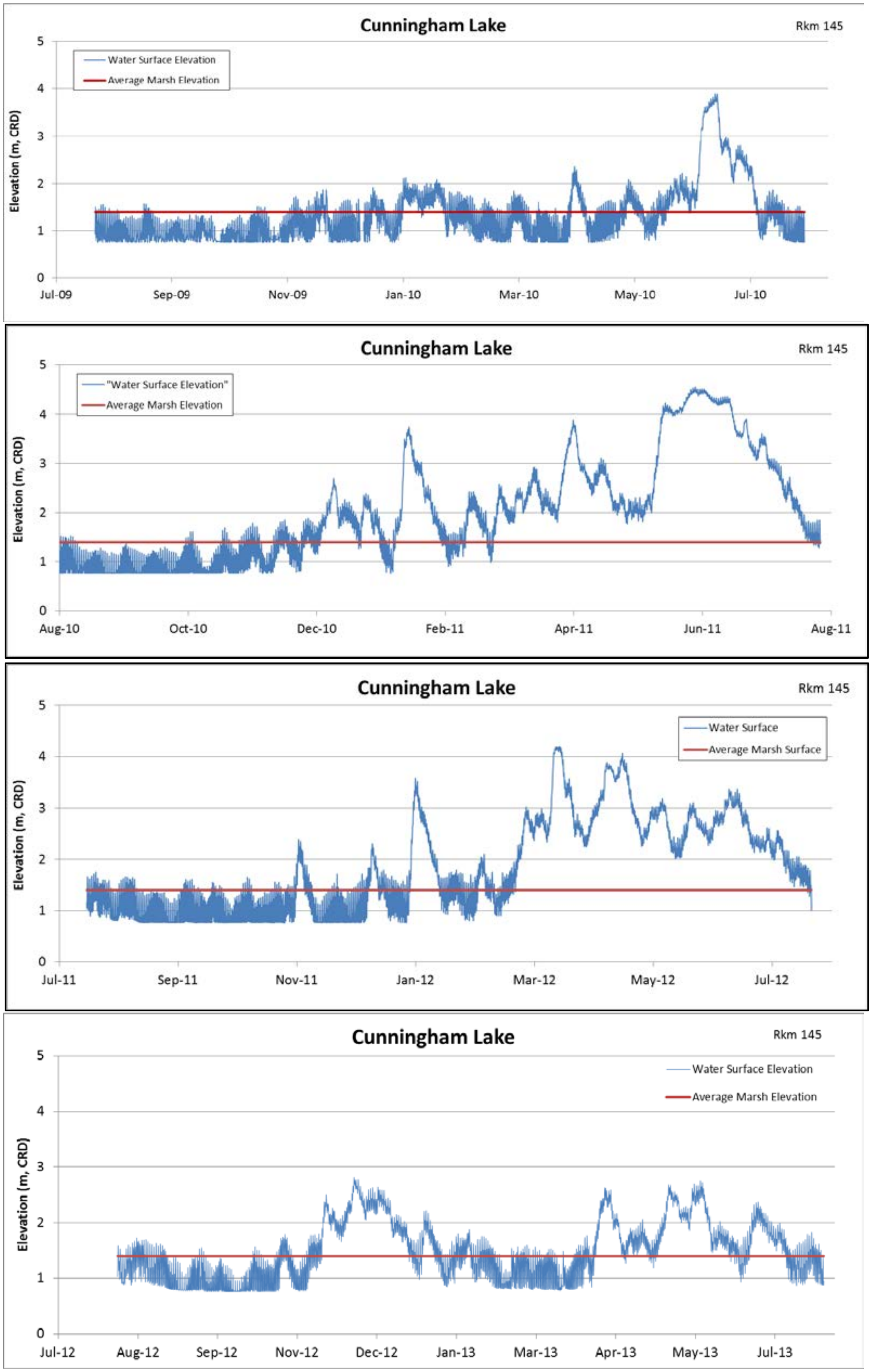
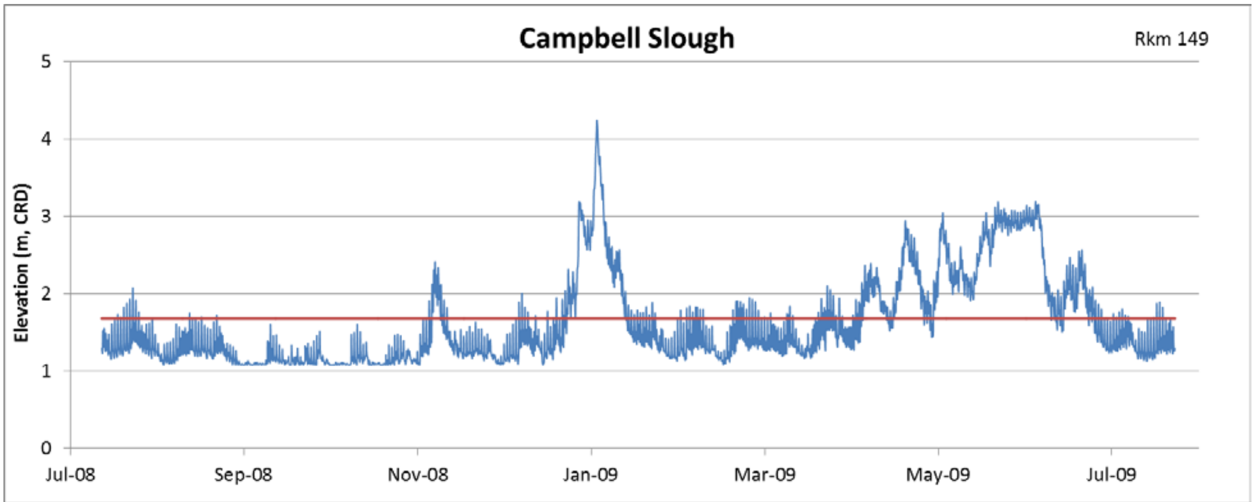
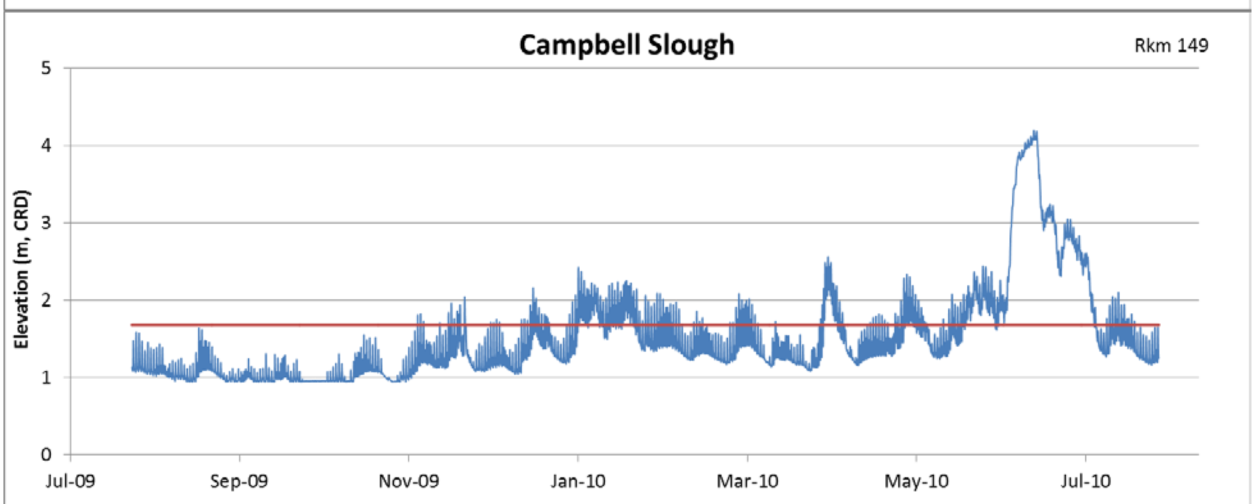
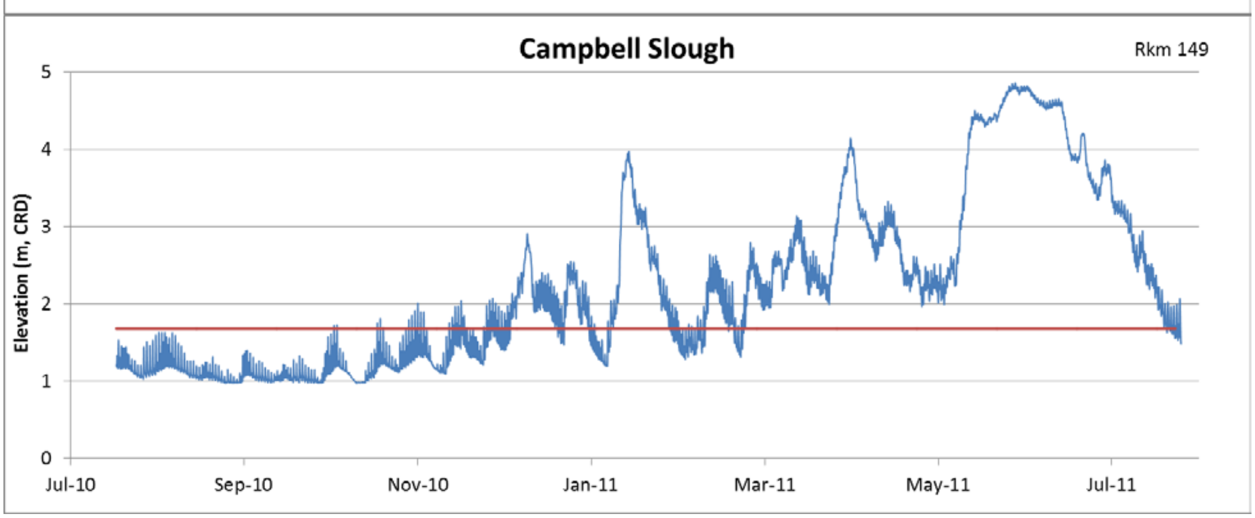


Figure A -5. Water surface elevation data from the Cunningham Lake study site for the years 2009-2013. The red line represents the average elevation of the marsh sampling area.


Campbell Slough
Rkm 149
Elevation (m, CRD)
5
4
3
2
1
0
Jul-08
Sep-08
Nov-08
Jan-09
Mar-09
May-09
Jul-09


Campbell Slough
Rkm 149
Elevation (m, CRD)
5
4
3
2
1
0
Jul-09
Sep-09
Nov-09
Jan-10
Mar-10
May-10
Jul-10


Campbell Slough
Rkm 149
Elevation (m, CRD)
5
4
3
2
1
0
Jul-10
Sep-10
Nov-10
Jan-11
Mar-11
May-11
Jul-11

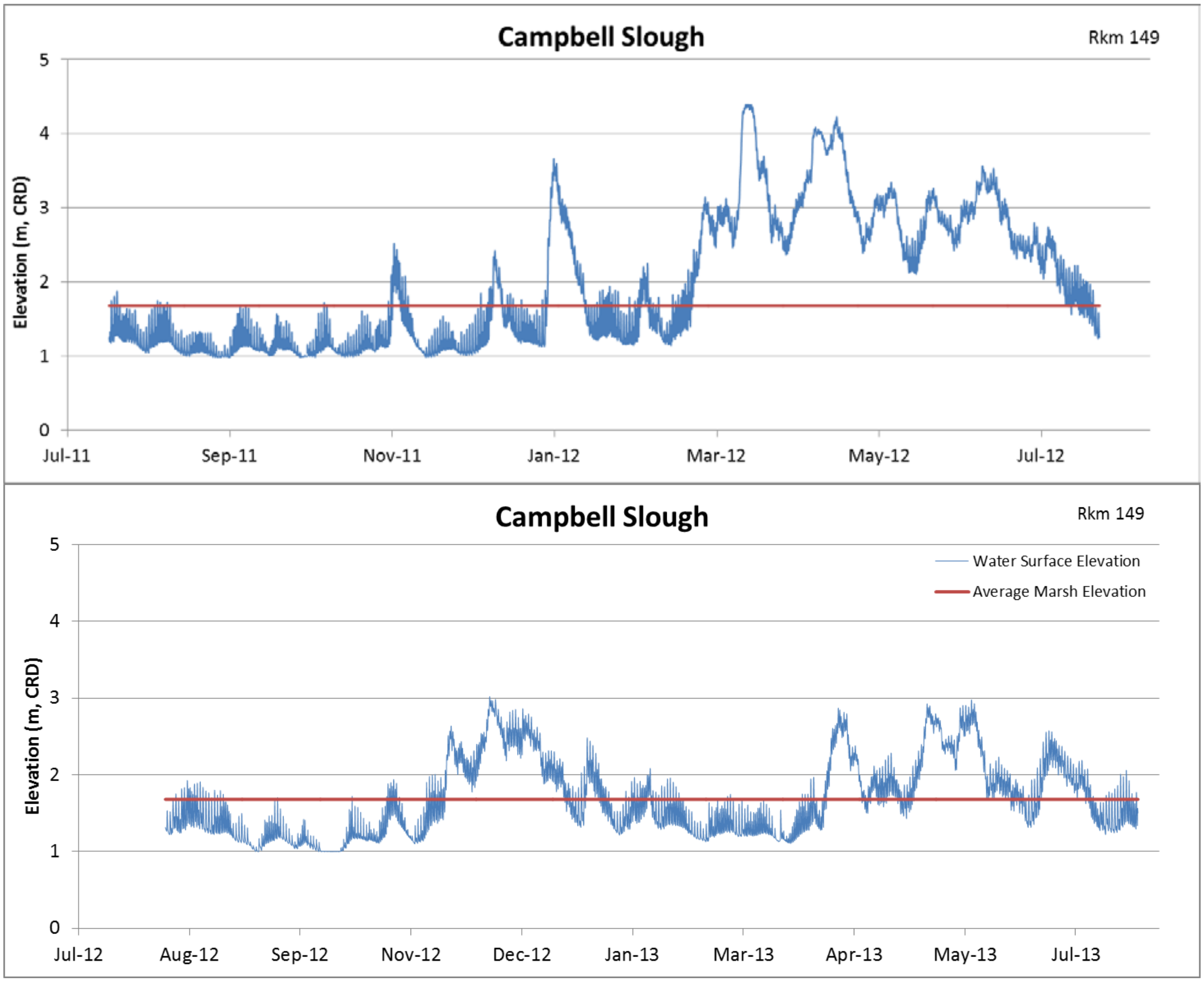


Figure A -6. Water surface elevation data from the Campbell Slough study site for the years 2008-2013. The red line represents the average elevation of the marsh sampling area.

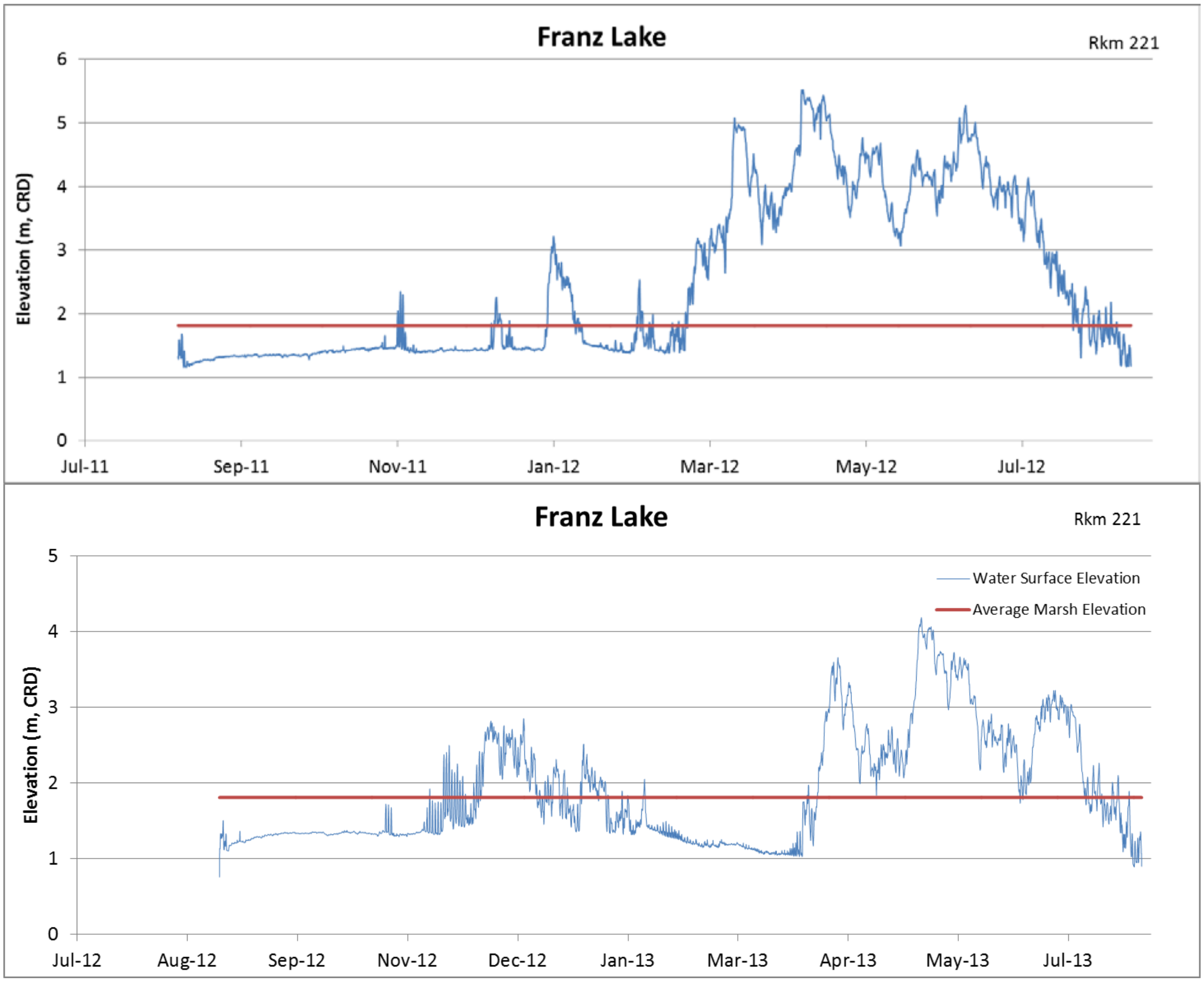
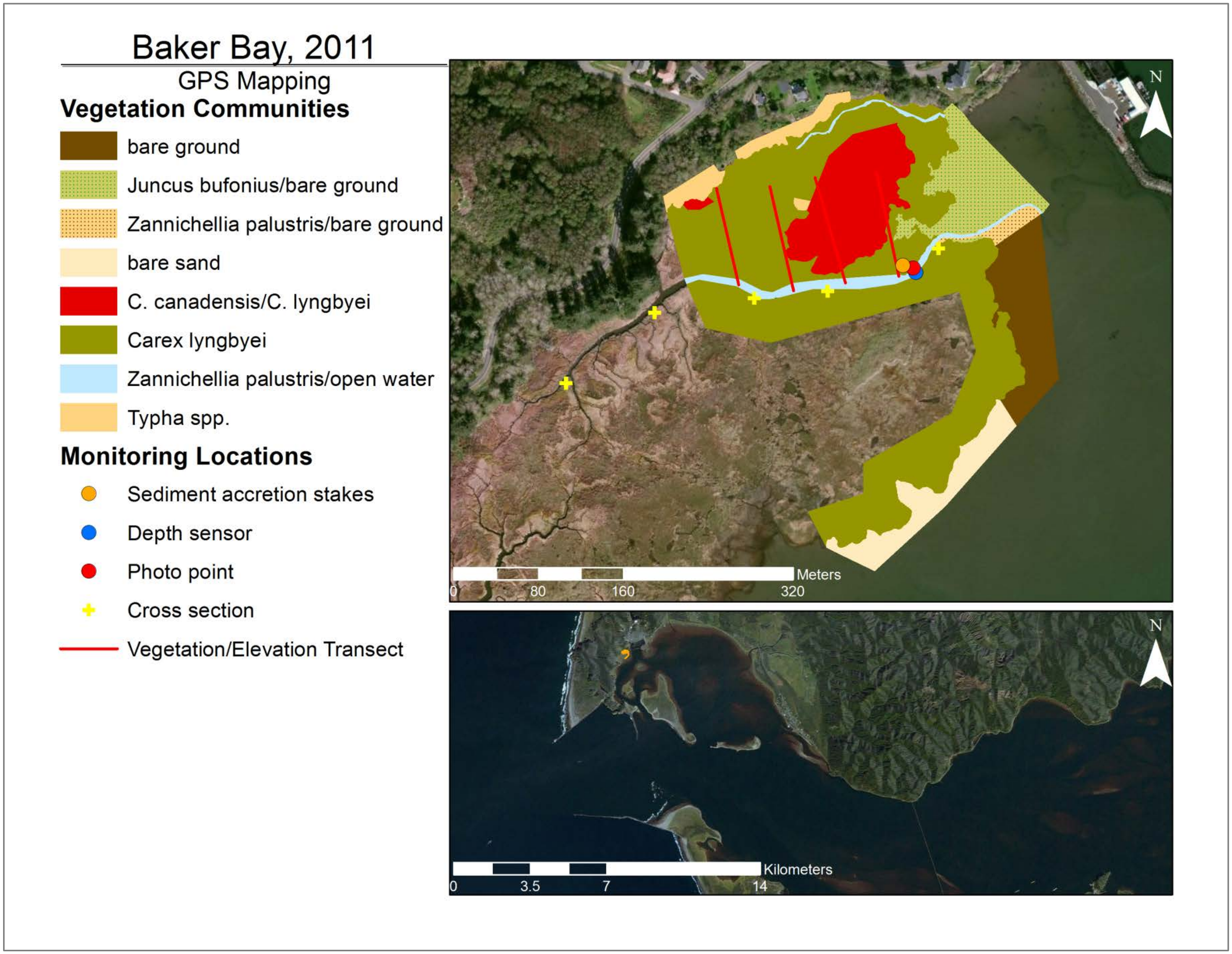


Figure A -7. Water surface elevation data from the Franz Lake study site for the years 2011-2013. The red line represents the average elevation of the marsh sampling area.

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 that appeared to have changed between 2011 and 2012 (Welch Island, Cunningham Lake and Franz Lake)*
- *Maps updated in 2013 because a larger area was mapped (Secret River)*


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
N
Meters
0 80 160 320
N
Kilometers
0 3.5 7 14

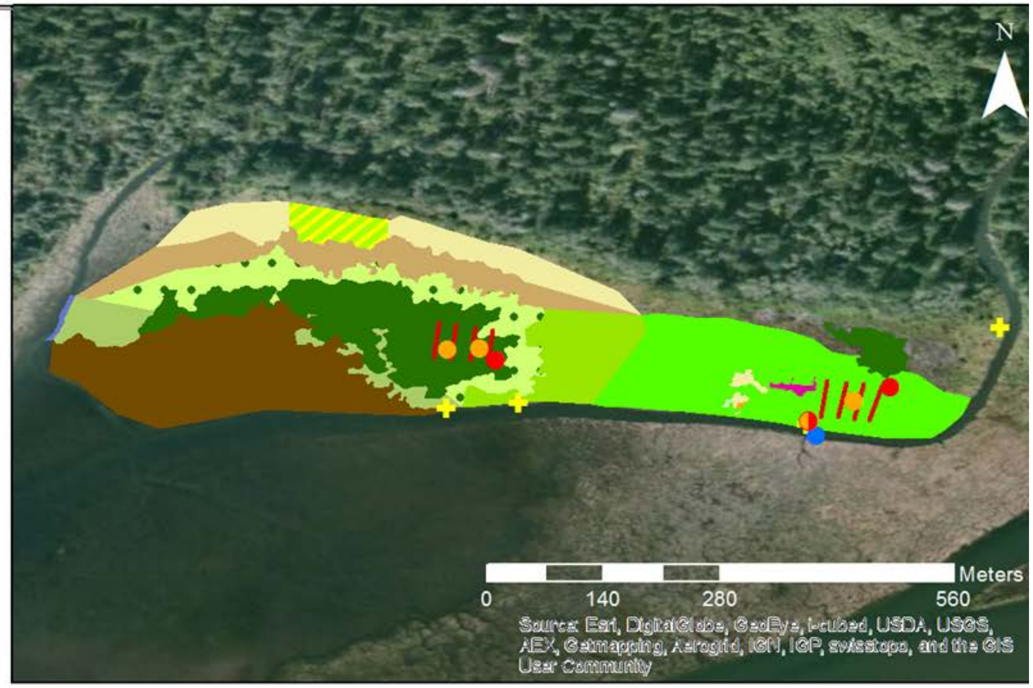
Secret River Marsh, 2013

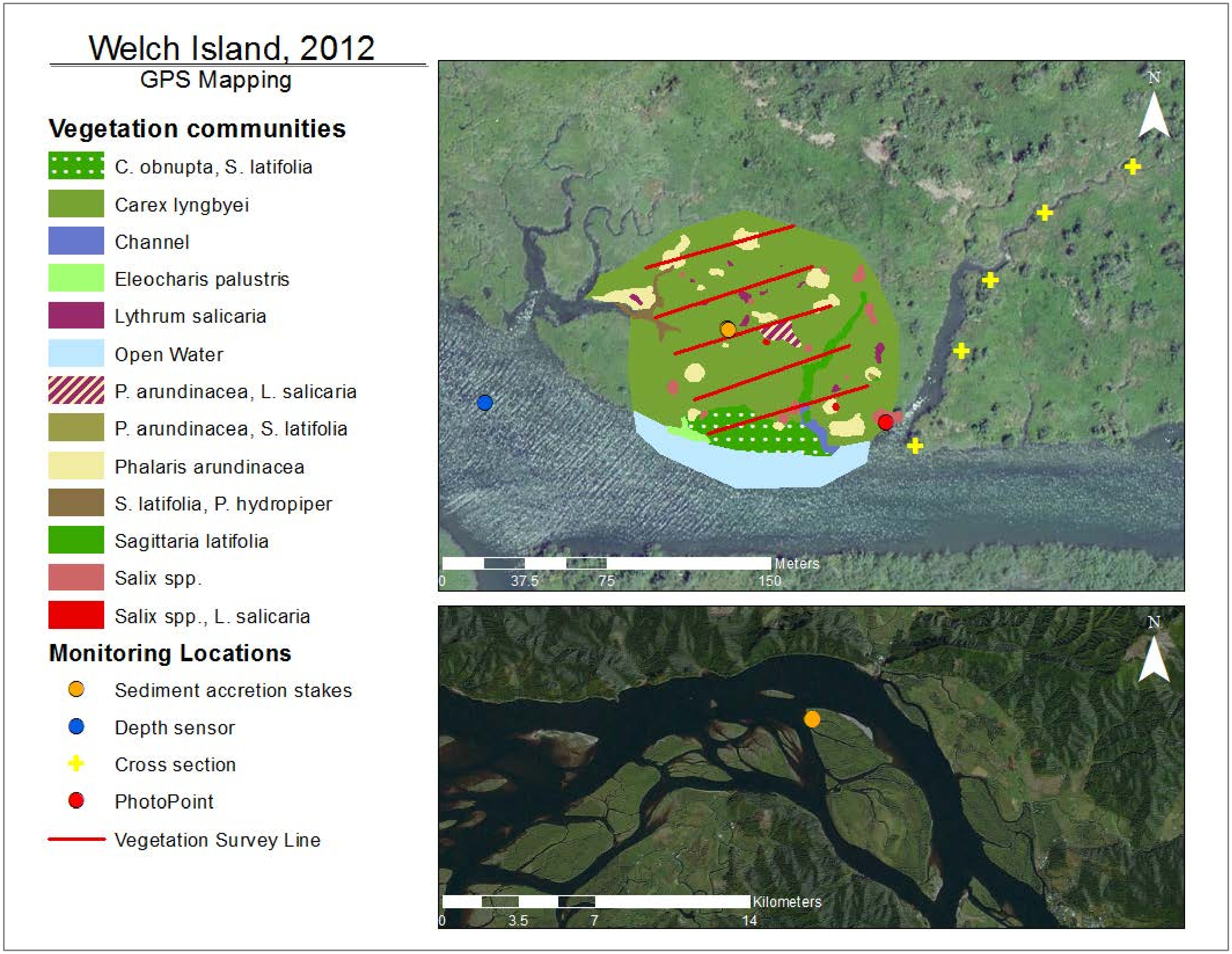
GPS Mapping

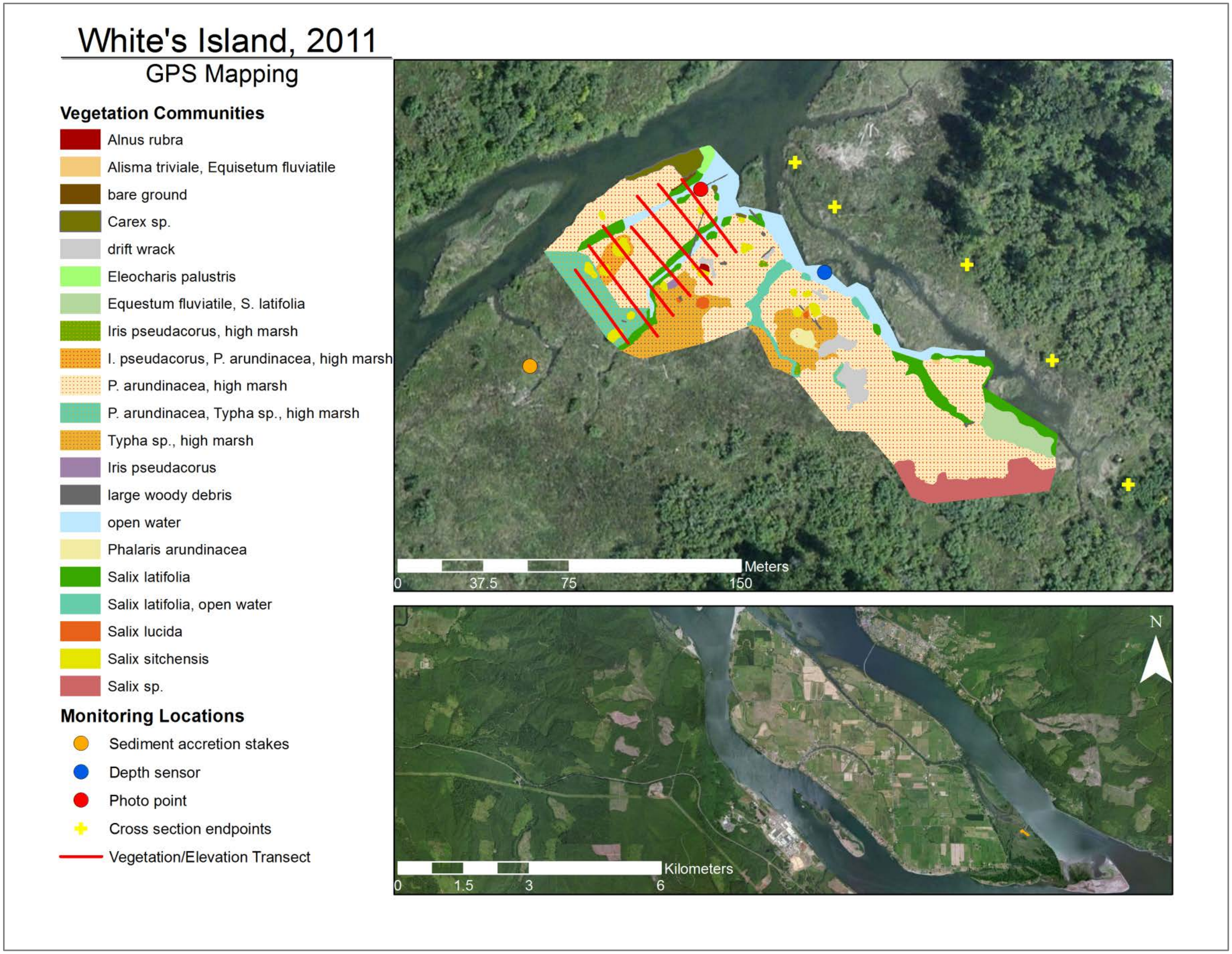
- Bare ground
- Carex lyngbyei/P. arundinacea
- Channel
- D. cespitosa and P. arundinacea
- Deschampsia cespitosa
- E. palustris and S. tabernaemontani
- Lythrum salicaria
- Mixed P. arundinacea
- Phalaris arundinacea
- Schoenoplectus tabernaemontani
- Sparganium eurycarpum
- Sparganium eurycarpum/P. arundinacea
- Submerged aquatic vegetation

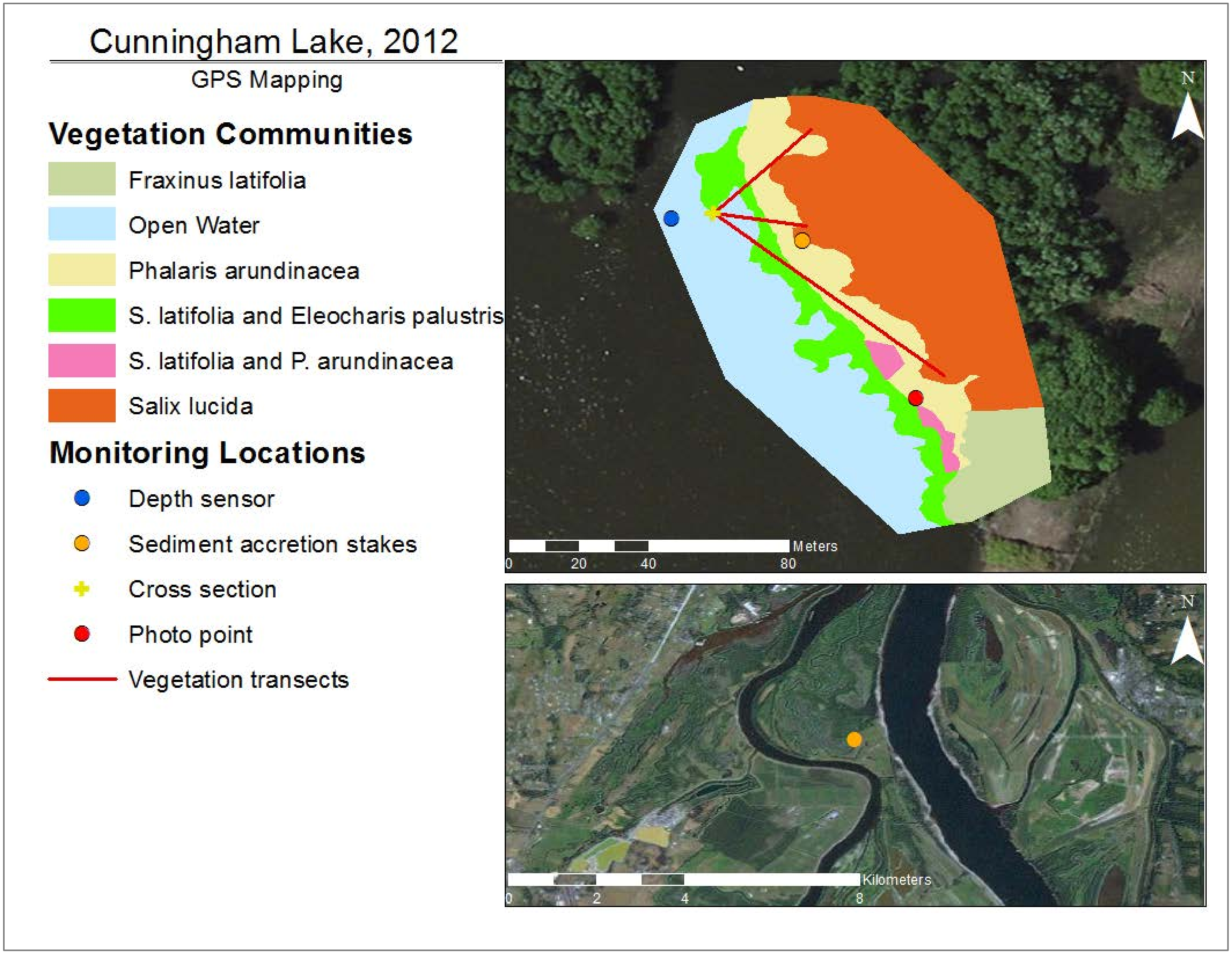
Monitoring Locations

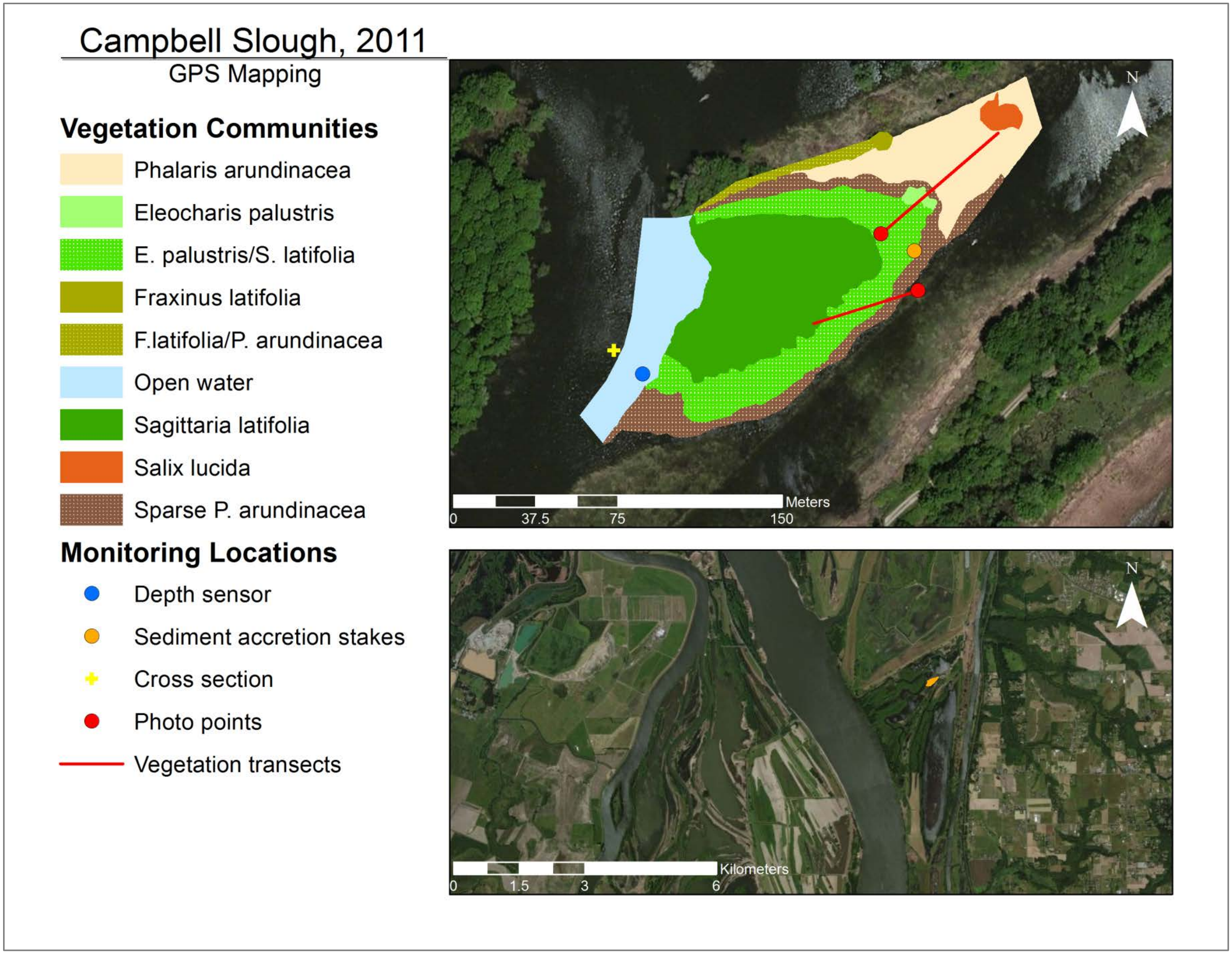
- Photo point
- Depth sensor
- Sediment accretion stakes
- Sediment accretion stakes/Photo point
- Cross section
- Vegetation/Elevation Transect

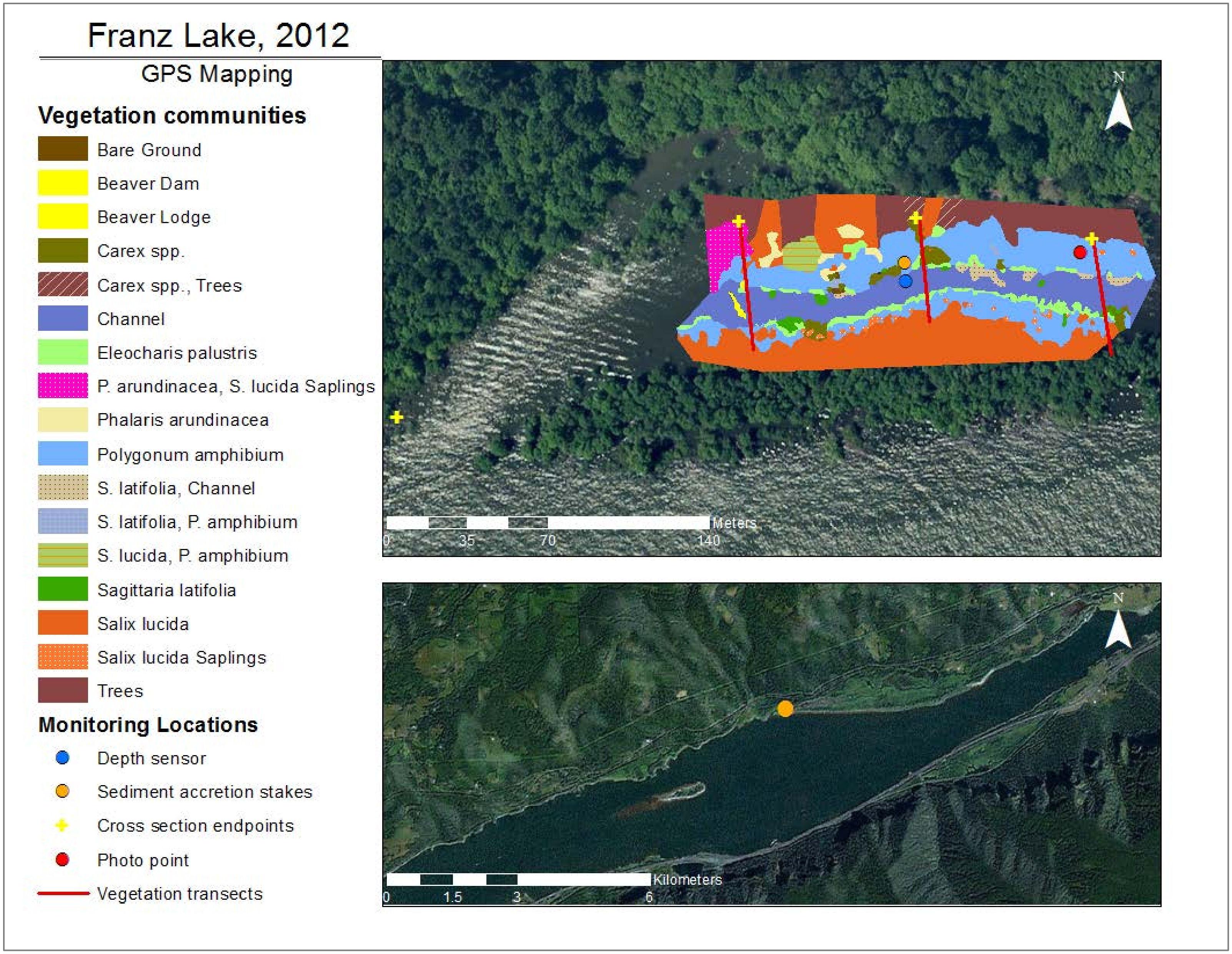



Welch Island, 2012
GPS Mapping
Vegetation communities
C. obnupta, S. latifolia
Carex lyngbyei
Channel
Eleocharis palustris
Lythrum salicaria
Open Water
P. arundinacea, L. salicaria
P. arundinacea, S. latifolia
Phalaris arundinacea
S. latifolia, P. hydropiper
Sagittaria latifolia
Salix spp.
Salix spp., L. salicaria
Monitoring Locations
Sediment accretion stakes
Depth sensor
Cross section
PhotoPoint
Vegetation Survey Line
N
0 37.5 75 150 Meters
N
0 3.5 7 14 Kilometers


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
Meters
0
37.5
75
150
N
Kilometers
0
1.5
3
6


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
N
Meters
0 20 40 80
N
Kilometers
0 2 4 8


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
N
Meters
0 37.5 75 150
N
Kilometers
0 1.5 3 6



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
N
Meters
0 35 70 140
N
Kilometers
0 1.5 3 6

Appendix C. Vegetation Species Cover

Table C.1. Site marsh elevation (in meters, relative to the Columbia River vertical datum CRD) and marsh vegetation species average percent cover from 2013. The three dominant cover classes are bolded in red for each site and non-native species are shaded in yellow. 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	Ilwaco	Secret-Low	Secret-High	Welch	Whites	Campbell	Cunningham	Franz
					Elevation (m, CRD)							
				Min	1.61	0.96	1.93	0.95	0.77	1.23	1.05	1.10
				Avg	2.00	1.04	2.08	1.58	1.65	1.70	1.39	1.85
				Max	2.38	1.22	2.20	1.72	2.10	2.72	1.65	2.33
					Average Percent Cover							
AGGI	*Agrostis gigantea*	redtop; black bentgrass	NI	no	0.0	0.1	1.8	0.8	0.1	0.0	0.2	0.0
AGST	*Agrostis stolonifera L.*	creeping bentgrass	FAC	no	**14.5**	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ALTR	*Alisma triviale*	northern water plaintain	OBL	yes	0.0	1.9	0.1	0.2	0.7	0.0	0.0	0.0
AREG	*Argentina egedii ssp. Egedii*	Pacific silverweed	OBL	yes	4.4	0.0	1.7	2.9	0.0	0.0	0.0	0.0
BICE	*Bidens cernua*	Nodding beggars-ticks	FACW+	yes	0.0	2.3	0.0	0.1	0.3	0.0	0.0	0.0
CAAM	*Castilleja ambigua*	paint-brush owl-clover; johnny-nip	FACW+	yes	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CAHE	*Callitriche heterophylla*	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.8	0.1	1.1	0.5	0.0	0.0	0.0
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	yes	**61.6**	7.8	**45.3**	**54.6**	3.4	0.0	0.0	0.0
CAPA	*Caltha palustris*	Yellow marsh marigold	OBL	yes	0.0	0.0	8.0	6.5	0.0	0.0	0.0	0.0
CASP	*Carex sp.*	Carex	mixed	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.3	7.0
CASP2	*Calamagrostis sp.*	reedgrass		yes	0.0	1.7	0.0	1.0	0.0	0.0	0.0	0.0
CEDE	*Ceratophyllum demersum*	Coontail	OBL	yes	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-Low	Secret-High	Welch	Whites	Campbell	Cunningham	Franz
COAR	*Convolvulus arvensis*	Morning glory; Field bindweed	UPL	no	0.0	0.0	0.8	0.0	2.3	0.0	0.0	0.0
DECE	*Deschampsia cespitosa*	Tufted hairgrass	FACW	yes	4.5	0.0	0.1	0.0	0.0	0.0	0.0	0.0
ELAC	*Eleocharis acicularis*	Needle spikerush	OBL	yes	2.3	0.0	0.0	0.0	0.0	0.0	0.1	0.2
ELCA	*Elodea canadensis*	Canada waterweed	OBL	yes	0.0	**23.8**	0.0	0.1	1.9	0.0	0.3	0.0
ELPA	*Eleocharis palustris*	Common spikerush	OBL	yes	0.3	5.1	0.0	2.7	1.5	**14.8**	**22.6**	4.7
ELPAR	*Eleocharis parvula*	Dwarf spikerush	OBL	yes	0.1	1.5	0.0	0.1	0.0	0.0	0.0	0.0
EPCI	*Epilobium ciliatum*	Willow herb	FACW-	yes	0.0	0.0	0.2	0.4	0.3	0.0	0.0	0.0
EQFL	*Equisetum fluviatile*	Water horsetail	OBL	yes	0.0	0.0	3.3	2.4	4.7	0.1	0.2	0.5
FOAN	*Fonrinalis antipyretica*	common water moss	OBL	yes	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
FRLA*	*Fraxinus latifolia*	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0
FUDI	*Fucus distichus*	Rockweed	OBL	yes	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GATR	*Galium trifidum L. spp. columbianum*	Pacific bedstraw	FACW	yes	0.0	0.0	1.1	0.4	1.3	0.0	0.0	0.0
GLGR	*Glyceria grandis*	American mannagrass	OBL	yes	0.0	0.0	0.3	0.7	1.1	0.0	0.0	0.0
GLMA	*Glaux maritima*	sea milkwort	FACW+	yes	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GLST	*Glyceria striata*	Fowl mannagrass	OBL	yes	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
GREB	*Gratiola ebracteata*	bractless hedgehyssop	OBL	yes	0.0	0.3	0.0	0.0	0.0	0.0	0.1	0.0
HEAU	*Helenium autumnale*	common sneezeweed	FACW	yes	0.0	0.0	0.5	0.0	0.0	0.0	0.1	0.1
HOLA	*Holcus lanatus*	Common velvetgrass	FAC	no	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0
HYSC	*Hypericum scouleri*	Western St. Johns wort	FAC	yes	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0
IMSP	*Impatiens capensis,Impatiens noli-tangere*	western touch-me-not, common touch-me-not, jewelweed	FACW	yes	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.0
IRPS	*Iris pseudacorus*	Yellow iris	OBL	no	0.0	0.0	0.3	1.3	3.4	0.0	0.0	0.0
ISSP	*Isoetes spp.*	quillwort	OBL	yes	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0
JUEF	*Juncus effusus*	Soft rush	FACW	mixed	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-Low	Secret-High	Welch	Whites	Campbell	Cunningham	Franz
JUOX	*Juncus oxymeris*	Pointed rush	FACW+	yes	0.0	0.8	0.1	1.2	0.1	0.0	0.2	0.0
JUTE	*Juncus tenuis*	slender rush, poverty rush	FACW-	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LAPA	*Lathyrus palustris*	Marsh peavine	OBL	yes	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
LEOR	*Leersia oryzoides*	Rice cutgrass	OBL	yes	0.0	0.0	0.3	1.1	0.0	0.0	0.1	0.8
LIAQ	*Limosella aquatica*	Water mudwort	OBL	yes	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0
LIOC	*Lilaeopsis occidentalis*	Western lilaeopsis	OBL	yes	1.8	**17.4**	0.0	0.0	0.0	0.0	0.0	0.0
LISC	*Lilaea scilloides*	Flowering quillwort	OBL	yes	0.0	0.4	0.0	0.4	0.0	0.0	0.0	0.0
LOCO	*Lotus corniculatus*	Birdsfoot trefoil	FAC	no	0.0	0.0	0.0	1.0	**5.9**	0.0	0.1	0.0
LUPA	*Ludwigia palustris*	False loosestrife	OBL	yes	0.0	0.0	0.0	0.0	0.0	2.1	4.2	1.2
LYAM	*Lysichiton americanus*	Skunk cabbage	OBL	yes	0.0	0.0	1.3	2.9	0.0	0.0	0.0	0.0
LYAM2	*Lycopus americanus*	American water horehound	OBL	yes	0.0	0.0	0.0	1.4	0.6	0.0	0.0	0.0
LYNU	*Lysimachia nummularia L.*	Moneywort, Creeping Jenny	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.0
LYSA	*Lythrum salicaria*	Purple loosestrife	FACW+	no	0.0	0.0	1.0	0.9	0.4	0.0	0.0	0.0
MEAR	*Mentha arvensis*	wild mint	FACW-	yes	0.0	0.0	0.1	1.8	0.0	0.0	0.8	0.0
MIGU	*Mimulus guttatus*	Yellow monkeyflower	OBL	yes	0.0	0.0	0.2	1.6	0.3	0.0	0.0	0.0
MYHI	*Myriophyllum hippuroides*	western milfoil	OBL	yes	0.0	0.0	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	7.6	**8.0**	4.2	0.0	0.0	0.0
MYSI	*Myriophyllum sibiricum*	northern milfoil, short spike milfoil	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
MYSP	*Myosotis laxa, M. scorpioides*	Small forget-me-not, Common forget-me-not	mixed	mixed	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
OESA	*Oenanthe sarmentosa*	Water parsley	OBL	yes	0.0	0.0	**14.6**	7.7	3.9	0.0	0.0	0.0
PAVI	*Parentucellia viscosa*	Yellow parentucellia	FAC-	no	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
PHAR	*Phalaris arundinacea*	Reed canarygrass	FACW	no	0.0	0.0	**35.5**	**9.8**	**56.5**	**39.2**	**33.1**	**11.2**
PLDI	*Platanthera dilatata*	white bog orchid	FACW+	yes	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-Low	Secret-High	Welch	Whites	Campbell	Cunningham	Franz
PLMA	*Plantago major*	common plantain	FACU+	no	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	19.9
POAN2	*Poa annua*	annual bluegrass	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
POCR	*Potamogeton crispus*	Curly leaf pondweed	OBL	no	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
POHY	*Polygonum hydropiper, P. hydropiperoides*	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0
PONA	*Potamogeton natans*	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
POPE	*Polygonum persicaria*	Spotted ladysthumb	FACW	no	0.0	1.3	0.1	1.1	0.0	0.8	0.3	0.0
POPE2	*Potamogeton pectinatus*	Sago pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
POPU	*Potamogeton pusillus*	Small pondweed	OBL	yes	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
PORI	*Potamogeton richardsonii*	Richardson's pondweed	OBL	yes	0.0	0.5	0.0	0.0	2.7	0.0	0.0	0.0
POSP	*Polygonum sp.*	Knotweed, Smartweed	mixed	mixed	0.0	2.6	0.7	1.4	0.2	0.0	0.4	0.0
RARE	*Ranunculus repens*	Creeping buttercup	FACW	no	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0
ROCU	*Rorippa curvisiliqua*	curvepod yellow cress	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
ROPA	*Rorippa palustris*	Marsh yellow-cress	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
RUAQ	*Rumex aquaticus*	Western dock	FACW+	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
RUAR	*Rubus armeniacus*	Himalayan blackberry	FACU	no	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
RUCR	*Rumex crispus*	Curly dock	FAC+	no	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0
RUMA	*Rumex maritimus*	Golden dock, seaside dock	FACW+	yes	0.0	0.0	0.5	0.2	0.0	0.0	0.0	0.0
SALA	*Sagittaria latifolia*	Wapato	OBL	yes	0.0	0.2	1.5	4.6	**8.5**	3.8	**5.7**	0.9
SALU*	*Salix lucida*	Pacific willow	FACW+	yes	0.0	0.0	0.0	0.0	0.0	**9.4**	0.7	**9.7**
SASI*	*Salix sitchensis*	Sitka willow	FACW	yes	0.0	0.0	0.0	0.9	1.8	0.0	0.0	0.0
SCAM	*Schoenoplectus americanus*	American bulrush, threesquare bulrush	OBL	yes	0.9	0.0	0.0	0.0	0.1	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-Low	Secret-High	Welch	Whites	Campbell	Cunningham	Franz
SCAR	*Schedonorus arundinaceus*	tall fescue	FAC-	no	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCMA	*Schoenoplectus maritimus*	Seacoast bulrush	OBL	yes	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCTA	*Schoenoplectus tabernaemontani*	Softstem bulrush, tule	OBL	Yes	0.0	**10.8**	0.0	0.0	0.0	0.0	0.0	0.0
SCTR	*Schoenoplectus triqueter*	Threesquare tule	OBL	no	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0
SISU	*Sium suave*	Hemlock waterparsnip	OBL	yes	0.0	0.0	0.2	2.8	0.1	0.0	0.0	0.0
SODU	*Solanum dulcamara*	Bittersweet nightshade	FAC+	no	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
SPAN	*Sparganium angustifolium*	Narrowleaf burreed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
SPCA	*Spergularia canadensis*	Canadian sandspurry	OBL	yes	T	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SPEU	*Sparganium eurycarpum*	giant burreed	OBL	yes	0.0	2.2	0.0	0.0	0.0	0.0	0.0	0.0
SYSU	*Symphyotrichum subspicatum*	Douglas aster	FACW	yes	3.0	0.3	0.3	1.9	0.0	0.0	0.0	0.0
TRMA	*Triglochin maritima*	Seaside arrowgrass	OBL	yes	5.3	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	T	0.0	0.0	0.0	0.0
TYAN	*Typha angustifolia*	Narrowleaf cattail	OBL	no	1.2	0.0	0.0	0.0	2.8	0.0	0.0	0.0
TYSP	*Typha angustifolia, T. latifolia*	Narrowleaf cattail, common cattail	OBL	mixed	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
VEAM	*Veronica americana*	American speedwell	OBL	yes	0.0	0.1	0.1	T	0.1	0.0	0.1	0.0
VEAN	*Veronica anagallis-aquatica*	water speedwell	OBL	yes	0.0	0.0	0.0	0.0	0.2	0.0	0.1	0.0
ZAPA	*Zannichellia palustris*	horned pondweed	OBL	yes	T	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other Cover												
Algae		algae			0.3	1.0	0.0	0.0	0.0	3.8	6.3	0.0
BG		bare ground			6.5	31.3	2.8	3.6	8.6	31.5	19.2	42.8
Detritus		detritus			1.6	0.3	2.3	1.3	0.4	0.1	0.1	1.7
DW		drift wrack			**12.3**	0.3	2.3	0.1	1.5	0.3	0.0	0.1
Litter		litter			0.0	1.0	0.0	0.0	0.0	1.7	0.1	2.9

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-Low	Secret-High	Welch	Whites	Campbell	Cunningham	Franz
LWD		large woody debris			0.0	0.0	0.0	0.0	3.8	0.4	0.0	0.9
MG		Mixed Grass	mixed	Grass	0.0	0.0	0.3	0.3	1.2	0.0	0.0	0.0
Moss		Moss		Moss	0.0	1.5	0.0	0.0	1.7	0.0	0.2	0.0
OW		open water			0.5	37.0	0.0	1.9	3.1	3.1	14.7	21.3
SAP		tree/shrub sapling		Tree/Shrub	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0
SD		standing dead			0.0	0.0	0.0	0.0	0.1	0.2	0.1	0.1
SH		shell hash			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
SMH		small mixed herbs		Herb	0.0	0.0	0.0	0.0	0.0	0.2	0.1	2.6

T = Trace

Table C.2. Site channel elevation (in meters, relative to the Columbia River vertical datum CRD) and submerged aquatic vegetation (SAV) species average percent cover from 2013. The three dominant cover classes are bolded in red for each site and non-native species are shaded in yellow. 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). Channel data was included in the marsh data for the Cunningham Lake and Franz Lake sites.

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret River	Welch	Whites	Campbell
					Elevation (m, CRD)				
				Min	0.90	0.13	0.21	0.14	0.67
				Avg	1.01	0.36	0.44	0.35	0.88
				Max	1.18	0.69	0.56	0.61	1.05
					Average Percent Cover				
AGGI	*Agrostis gigantea*	redtop; black bentgrass	NI	no	0.0	0.0	0.2	0.0	0.0
ALTR	*Alisma triviale*	northern water plaintain	OBL	yes	0.0	0.1	0.8	0.1	0.0
BICE	*Bidens cernua*	Nodding beggars-ticks	FACW+	yes	0.0	0.1	0.2	0.1	0.0
CASP2	*Calamagrostis sp.*	reedgrass		yes	0.0	**0.6**	0.0	0.0	0.0
CAHE	*Callitriche heterophylla*	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.1	2.0	0.0	0.0
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	yes	**3.5**	0.0	0.0	0.0	0.0
CEDE	*Ceratophyllum demersum*	Coontail	OBL	yes	0.0	0.1	**3.5**	0.0	0.0
ELPA	*Eleocharis palustris*	Common spikerush	OBL	yes	0.0	0.0	0.8	0.0	0.0
ELCA	*Elodea canadensis*	Canada waterweed	OBL	yes	0.0	**10.9**	0.0	**2.6**	0.0
ELNU	*Elodea nuttallii*	Nuttall's waterweed, western waterweed	OBL	yes	0.0	0.0	**61.7**	0.0	0.0
EQFL	*Equisetum fluviatile*	Water horsetail	OBL	yes	0.0	0.0	0.3	0.0	0.0
FOAN	*Fonrinalis antipyretica*	common water moss	OBL	yes	0.0	0.1	0.0	0.0	0.0
FUDI	*Fucus distichus*	Rockweed	OBL	yes	1.7	0.0	0.0	0.0	0.0
ISSP	*Isoetes spp.*	quillwort	OBL	yes	0.0	0.0	0.2	0.0	0.0
LIOC	*Lilaeopsis occidentalis*	Western lilaeopsis	OBL	yes	1.7	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret River	Welch	Whites	Campbell
LIAQ	*Limosella aquatica*	Water mudwort	OBL	yes	0.0	0.1	0.8	**0.7**	0.0
MYSC	*Myosotis scorpioides*	Common forget-me-not	FACW	no	0.0	0.0	0.3	0.0	0.0
MYHI	*Myriophyllum hippuroides*	western milfoil	OBL	yes	0.0	0.1	2.7	0.4	0.0
POHY	*Polygonum hydropiper, P. hydropiperoides*	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	0.1	0.0	0.0	0.0
POPE	*Polygonum persicaria*	Spotted ladysthumb	FACW	no	0.0	0.0	**3.5**	0.0	0.0
POSP	*Polygonum sp.*	Knotweed, Smartweed	mixed	mixed	0.0	**0.6**	0.0	0.0	0.0
POCR	*Potamogeton crispus*	Curly leaf pondweed	OBL	no	0.0	0.3	0.0	**0.7**	0.0
PONA	*Potamogeton natans*	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.0	**0.2**
POPE2	*Potamogeton pectinatus*	Sago pondweed	OBL	yes	0.0	0.0	0.0	0.0	**4.2**
PORI	*Potamogeton richardsonii*	Richardson's pondweed	OBL	yes	0.0	**37.6**	**17.3**	**19.3**	0.0
ULLA	*Ulva lactuca*	Sea lettuce	OBL	yes	0.2	0.0	0.0	0.0	0.0
ZAPA	*Zannichellia palustris*	horned pondweed	OBL	yes	**18.5**	0.0	0.0	0.4	0.0
Algae		algae			**2.5**	0.0	0.0	0.0	**13.3**
BG		bare ground			70.8	45.0	9.2	76.1	86.7
Detritus		detritus			0.8	0.0	0.0	0.0	0.0
Litter		litter			0.0	0.1	0.0	0.0	0.0
LWD		large woody debris			0.8	0.0	0.0	0.0	0.0
OW		open water			50.8	76.4	58.3	96.4	100.0
SH		shell hash			0.0	**0.6**	0.0	0.0	0.0

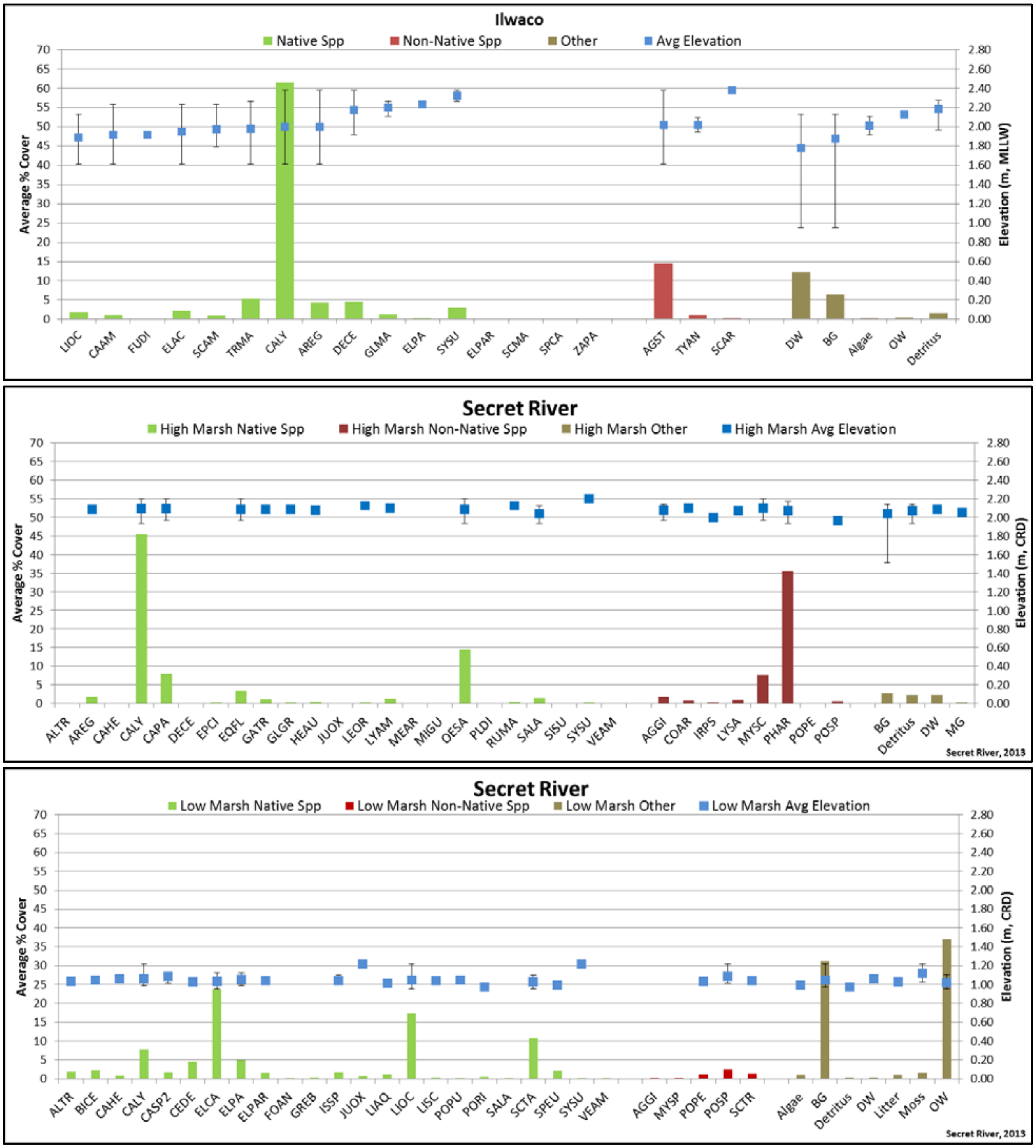


Figure C-1. Vegetation species cover and elevations for sites sampled in 2013. Sites are ordered by position in the LCRE, starting near the mouth. Species are in order by elevation. Bars represent the minimum and maximum elevations at which the vegetative species occurred within the sample area (Table C.1. for species names and percent cover data associated with codes along the x-axis).

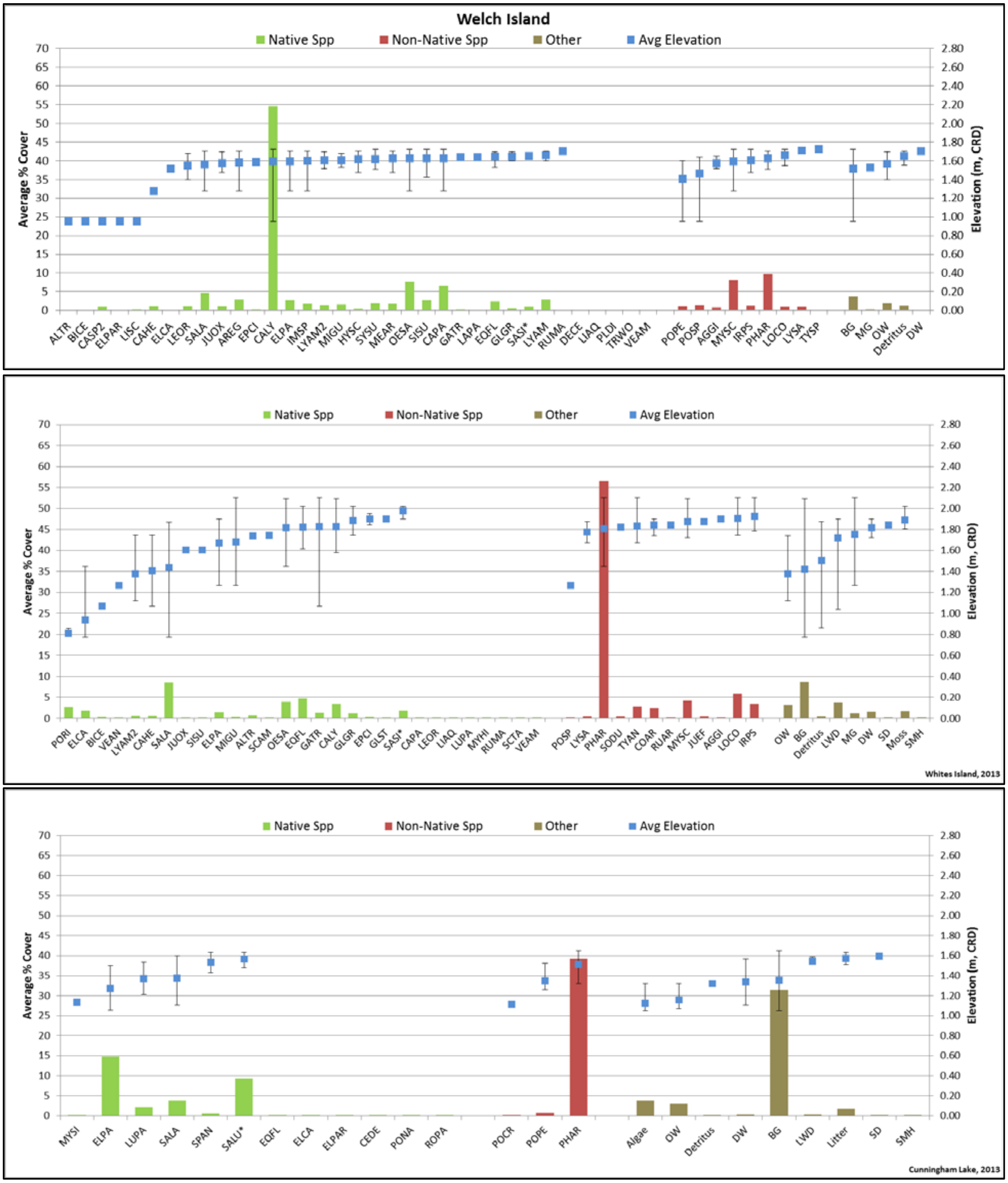


Figure C-1. Continued.

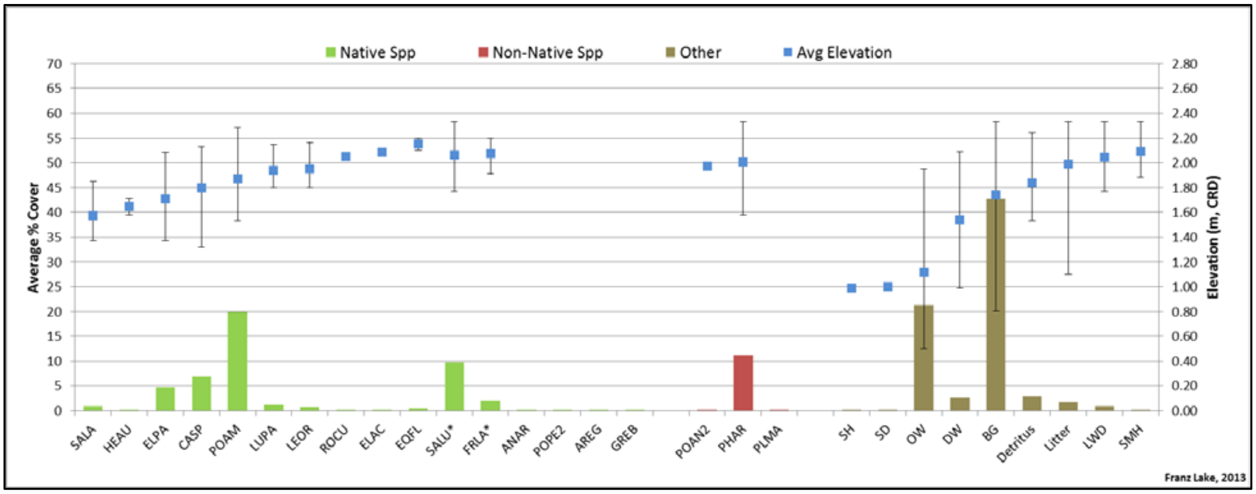


Figure C-1. Continued.