

Lower Columbia River Ecosystem Monitoring Program

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**Lower Columbia River Ecosystem Monitoring Program
Annual Report for Year 11 (October 1, 2014 to September 30, 2015)**

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Abstract

The Ecosystem Monitoring Program (EMP) is managed by the Lower Columbia Estuary Partnership and is an integrated status and trends program for the lower Columbia River. The EMP aims to collect key information on ecological conditions for a range of habitats throughout the lower river characteristic of those used by migrating juvenile salmon and provide information toward the recovery of threatened and endangered salmonids. The program inventories the different types of habitats within the lower river, tracks trends in the overall condition of these habitats over time, provides a suite of reference sites for use as end points in regional habitat restoration actions, and places findings from management actions into context with the larger ecosystem. The EMP is implemented through a multi-agency collaboration, focusing sampling efforts on examining temporal trends within a study area that extends from the mouth of the river to Bonneville Dam. In 2015, data were collected on fish, habitat, hydrology, food web, abiotic site conditions, and mainstem river conditions at Ilwaco Slough (rkm 6), Welch Island (rkm 53), Whites Island (rkm 72), Campbell Slough (rkm 149), and Franz Lake (rkm 221). Habitat and hydrology data were the only metrics collected at Secret River (rkm 37), and Cunningham Lake (rkm 145) in 2015. The trends sampling sites are minimally disturbed, tidally influenced freshwater emergent wetlands with backwater sloughs that represent a subset of the eight hydrogeomorphic reaches across the lower river. In addition, fish community data were collected in the lower Grays River and lower Lewis River (food web metrics were also collected in the lower Grays River) to explore usage of lower tributary habitats by migrating juvenile salmon.

An understanding of conditions within the mainstem of the Columbia River is critical for understanding the connectedness between shallow off-channel habitats and mainstem river flow. Water temperatures at the trends sites were generally higher in 2015 relative to long term mean temperatures, particularly in the three most upstream (river-dominated) sites in June and July (Whites Island, Campbell Slough, and Franz Lake). Reduced river flow contributed to higher salinity and pH at Ilwaco Slough in 2015 relative to long term mean values in June and July. In the mainstem, nitrate concentrations peaked in winter, followed by a drop in early-to-mid summer.

Hydrologic patterns, sediment accretion, and vegetation composition and cover were monitored at seven emergent wetland sites in the lower river. Cumulative inundation (i.e., sum exceedance value; SEV) varies spatially and in most years generally increases with distance from the river mouth. However, in 2015 Secret River experienced the greatest cumulative growing season SEV, whereas the sites in the upper reaches showed lower SEVs than in past sampling years. Sediment accretion rates were variable in time and space (-1.6 to 2.0 cm/year on average across years). Accretion rates were most consistent over time at Welch Island (0.7 ± 0.1 cm) and most variable across years at Campbell Slough and Franz Lake. The only site that consistently eroded over time is Secret River low marsh, averaging -1.6 cm/year. Emergent wetland vegetation cover and composition are related to hydrologic patterns, with non-native reed canarygrass (*Phalaris arundinacea*) being the most dominant species of vegetation (23.2%), although the average cover of *Carex lyngyei* was only slightly less at 19.5%. An increase in cover of the native species water smartweed (*Polygonum amphibium*) and wapato (*Sagittaria latifolia*) was observed in 2015 at the upper most sites.

Similar to previous years, diatoms dominated the phytoplankton community at the trends sites in 2015, particularly in the spring (April-May). However, the composition of the diatom community varies among sites, season, and years. At Whites Island, diatoms are typically dominant throughout the sampling season, while data from Campbell Slough and Franz Lake (in the upper, river-dominated reaches) showed high abundances of cyanobacteria during the summer months (June-July). The low flows observed in 2015 reduced flushing at the upper sites, leading to nutrient ratio alteration and a proliferation of

cyanobacteria populations, which may include toxin-producing species and represent a less nutritious food source for salmon prey.

Benthic invertebrate results show a transition in the macroinvertebrate community along the estuarine gradient. Ilwaco Slough had a greater presence of amphipods and isopods, whereas samples from upstream sites (i.e., Campbell Slough and Franz Lake) contained fewer dipterans and a higher abundance of other insects such as Collembola. The density and biomass of invertebrates captured in neuston samples were higher in emergent vegetation than in open water habitats, except for at Whites Island in June where more individuals were collected in open water compared to emergent vegetation. In the lower reaches (Ilwaco Slough, Welch Island, and Whites Island), juvenile Chinook salmon diets were dominated by amphipods, whereas chironomids and other diptera were preferred prey in the upper reaches (Campbell Slough and Franz Lake). Stomach fullness was greater in fish captured at lower reach sites where fish fed predominantly on amphipods compared to Franz Lake where chironomids were dominant in diets but stomachs were less full.

Lower Columbia River ESU stocks (West Cascades fall Chinook and Spring Creek group fall Chinook) were the dominant stocks at the trends sites, but interior stocks including Upper Columbia summer/fall Chinook and Snake River fall Chinook were also observed, particularly at Campbell Slough and Franz Lake. Differences in salmon presence at some of the trends sites were noted in 2015 compared to past sampling years, including low densities of Chinook salmon fry at Welch and Whites Island (where fry are typically caught in large numbers), absence of juvenile chum salmon in March and April, and a complete absence of Chinook salmon in July and September. The unusually warm water temperatures experienced in 2015 may have contributed to differences in fish densities and presence at the trends sites compared to previous years. Unmarked Chinook salmon dominated catches at all of the trends sites, except for Campbell Slough where more marked Chinook salmon were observed. Growth rates varied according stock and river reach (i.e., fish from upriver sites had higher growth rates than fish at the downriver sites). Sampling in the lower reaches of two tributaries showed chum salmon, coho salmon and Chinook salmon presence in the lower Grays River and a dominance of unmarked Chinook salmon fry in the lower Lewis River.

The Ecosystem Monitoring Program produces essential baseline information on ambient environmental conditions and yields insight into the cumulative effects of existing and new management actions and anthropogenic impacts. EMP data are useful for making comparisons to changing conditions, enhancing our understanding of fish habitat use, and determining whether water quality and habitat characteristics are meeting the needs of migrating juvenile salmonids. In addition, the relatively undisturbed conditions at the EMP trends sites should be considered end points for ecological function of habitats undergoing restoration, and findings can inform regional habitat restoration design and translate to additional reference data for comparison to action effectiveness monitoring efforts. Quantifying sources of variability in fish, habitat, and food web metrics allow for increased predictability for how biological components will respond to changes in environmental conditions, particularly as a result of climate change. The overall lesson learned from the 2015 results was that the uncharacteristically low flows and warmer water temperatures affected the fish community, food web, and habitat structure in the lower river. Such conditions lead to lower inundation, changes to the food web dynamics, and impaired water quality which reduced the suitability of some of the monitoring sites (upper, river-dominated sites particularly) for juvenile salmon use, especially during the late spring and early summer.

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1 Introduction

1.1 Background

The Columbia River historically supported diverse and abundant populations of fish and wildlife and is thought to have been one of the largest producers of Pacific salmonids in the world (Netboy 1980). Anthropogenic changes since the 1860s including dike construction, land use conversion, and the construction of the hydropower system in the Columbia River basin have resulted in alterations to the hydrograph (i.e., timing, magnitude, duration, frequency, and rate of change in river flows); degraded water quality and increased presence of toxic contaminants; introduction of invasive species; and altered food web dynamics. Subsequently, these changes within the Columbia River basin have significantly reduced the quantity and quality of habitat available to fish and wildlife species.

Threatened and endangered salmonids use shallow water wetland habitats of the lower Columbia River for rearing and refugia, with some stocks utilizing these habitats for long time periods before completing their migratory journey to the ocean (Bottom et al. 2005; Fresh et al. 2005, 2006; Roegner et al. 2008). Traditionally, fish and fish habitat research and monitoring efforts were concentrated in the lower reaches of the estuary (nearest the mouth of the river), leaving knowledge gaps in the basic understanding of fish habitat use and benefits within the upper, freshwater-dominated reaches. The quantity and quality of available habitats affects the diversity, productivity, and persistence of salmon populations (Fresh et al. 2005). Degradation and loss of estuarine habitats can threaten salmon population viability, thus highlighting the importance of identifying limiting factors to salmon survival and filling key knowledge gaps across the habitat gradient of the lower Columbia River to promote salmon recovery.

Tidal emergent wetland vegetation provides rearing and refuge habitat for juvenile fish and a source of organic matter to the mainstem and to downstream habitats, while tidal channels provide access to wetlands and to foraging opportunities. The majority of emergent wetlands in the lower Columbia River cover a narrow elevation range (0.8 – 2.6 m, relative to the Columbia River Datum), thus annual fluctuations in hydrology drive the spatial and temporal variability of wetland vegetation (i.e., cover and species composition) and affect wetland inundation (Sagar et al. 2013). Vegetation species composition in the lower river is spatially variable with the middle reaches showing the greatest species diversity; although some areas are dominated by the non-native reed canarygrass (*Phalaris arundinacea*), particularly in the river-dominated upper reaches (Sagar et al. 2013). Quantifying the variability in habitat metrics allows for greater predictability of how biota respond to changing environmental conditions and improves our understanding of how the lower river functions ecologically.

Salmonids occupy the upper trophic levels in the Columbia River system and they spend portions of their life cycle in fresh water, estuarine water, and oceanic water. Thus, threats to their survival could arise from a variety of sources or stressors occurring at any one of several life stages or habitat types. Large-scale changes to the ecological characteristics of the lower Columbia River food web as a consequence of wetland habitat loss have resulted in a reduction of macrodetritus inputs to the system that historically formed the basis of the aquatic food web (Sherwood et al. 1990). Currently, it is believed that organic matter derived from fluvial phytoplankton (rather than macrodetritus) may be a seasonal driver of the salmon food web (Maier and Simenstad 2009). The consequences of this apparent shift in the type of organic matter fueling food web dynamics are uncertain and the understanding of food web shifts requires detailed examination of interactions between multiple trophic levels and environmental conditions. Studying the abundance and assemblage of phytoplankton and zooplankton over space and time provides important information on diets of preferred salmon prey (i.e., chironomids and benthic amphipods). In

turn, understanding the abiotic conditions characteristic of emergent wetlands, and in the river mainstem are essential for elucidating patterns in primary and secondary productivity in the lower river.

The Lower Columbia Estuary Partnership (Estuary Partnership), as part of the Environmental Protection Agency (EPA) National Estuary Program, is required to develop and implement a Comprehensive Conservation and Management Plan. This Management Plan specifically calls for sustained long-term monitoring to understand ecological condition and function, evaluate the impact of management actions over time (e.g., habitat restoration), and protect the biological integrity in the lower Columbia River. The Estuary Partnership implements long-term monitoring through the Ecosystem Monitoring Program (EMP). Ultimately, the goal of the EMP is to track ecosystem condition over time, but also to allow researchers and managers the ability to distinguish between variability associated with natural conditions and any variability resulting from human influence. The EMP partnership collects on-the-ground data from relatively undisturbed emergent wetlands to provide information about habitat structure, fish use, abiotic site conditions, salmon food web dynamics, and river mainstem conditions to assess the biological integrity of the lower river, enhance our understanding of estuary function, and support recovery of threatened and endangered salmonids. The creation and maintenance of long-term datasets are vital for documenting the history of change within important resource populations. Therefore, through this program, we aim to assess the status (i.e., spatial variation) and track the trends (i.e., temporal variation) in the overall condition of the lower Columbia River, provide a better basic understanding of ecosystem function, provide a suite of reference sites for use as end points in regional habitat restoration actions, and place findings from other research and monitoring efforts (e.g., action effectiveness monitoring) into context with the larger ecosystem.

Ecosystem-based monitoring of the fish habitat conditions in the lower river is a regional priority intended to aid in the recovery of the historical productivity and diversity of fish and wildlife. The EMP is funded by the Northwest Power and Conservation Council/Bonneville Power Administration (NPCC/BPA) and a primary goal for the action agencies (i.e., the BPA and US Army Corps of Engineers) is to collect key information on ecological conditions for a range of habitats and whether the habitats in the lower river are meeting the needs of outmigrating juvenile salmonids for growth and survival. Such data provide information toward implementation of the 2008 Federal Columbia River Power System (FCRPS) Biological Opinion (BiOp; NMFS 2008). Specifically, NPCC/BPA funding for this program focuses on addressing BPA's Columbia Estuary Ecosystem Restoration Program (CEERP) goal of improving habitat opportunity, capacity and realized function for aquatic organisms, specifically salmonids.

The EMP addresses Action 28 of the Estuary Partnership Comprehensive Conservation and Management Plan; 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 implements the EMP by engaging regional experts at Battelle-Pacific Northwest National Laboratory (PNNL), National Oceanic and Atmospheric Administration National Marine Fisheries Service (NOAA-Fisheries), United States Geological Survey (USGS), and Oregon Health and Sciences University (OHSU).

1.2 Study Area

The lower Columbia River and estuary is designated as an "Estuary of National Significance" by the Environmental Protection Agency (EPA) and as such, it is part of the National Estuary Program (NEP) established in Section 320 of the Clean Water Act. The EMP study area encompasses that of the NEP (a.k.a., the Estuary Partnership), including all tidally influenced waters, extending from the mouth of the Columbia River at river kilometer (rkm) 0 to Bonneville Dam at rkm 235 (tidal influence is defined as historical tidal influence, relative to dam construction in the 1930s). The Estuary Partnership and

monitoring partners collect data for the EMP from 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 sampling 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 lower river. The primary purpose of the Classification is to enable management planning and systematic monitoring of diverse ecosystem attributes. The Classification also provides a utilitarian framework for understanding the underlying ecosystem processes that create the dynamic structure of the lower river. 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 lower river ecosystems. The EMP sampling design has been organized according to Level 3 of the Classification, which divides the lower river into eight major hydrogeomorphic reaches (Figure 1).

More recently, subsequent to the development of the sampling design, data collected as part of the EMP and other studies (Borde et al. 2012) have been used to define five wetland zones based on spatial variation of the hydrologic regime and vegetation patterns observed in the lower river (Jay et al. 2015; in revision). Vegetation species assemblages vary temporally and spatially at the trends sites, therefore vegetation assemblages are broadly grouped into categories, or emergent marsh (EM) zones, based on vegetation cover and species richness. EM zone delineation occurred in previous years as part of this and other studies (Jay et al. in revision; Sagar et al. 2013; Borde et al. 2012) and is used here to evaluate vegetation patterns within the tidal wetlands of the lower river because they are more representative of vegetation patterns than hydrogeomorphic reach. The zone boundaries are meant to be broad, and variation of the zone boundaries is observed between years. The following river kilometers are currently used to delineate the zones:

EM Zone	River Kilometer (rkm)
1	0 – 39
2	39 - 88
3	89 - 136
4	137 - 181
5	182 - 235

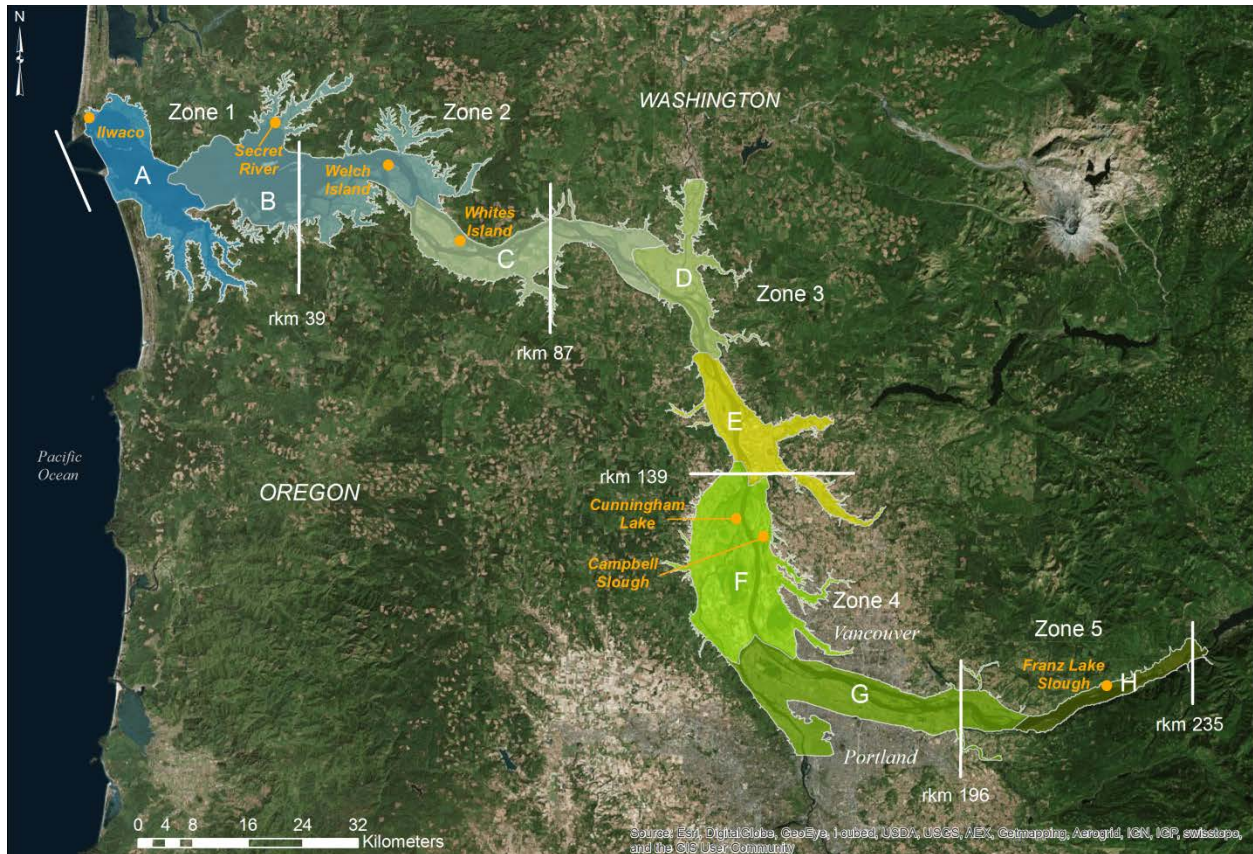


Figure 1. Lower Columbia River and estuary with hydrogeomorphic reaches (A-H) specified by color (Simenstad et al. 2011) and wetland zones (1-5) delineated by white lines (Jay et al. in press). The 2015 EMP trends sites are shown in orange.

1.3 Characterization of Emergent Wetlands in the Lower Columbia River

1.3.1 Sampling Effort, 2005-2015

The objective of the EMP is to characterize habitat structure and function of estuarine and tidal freshwater habitats within the lower river in order to track ecosystem condition over time, determine ecological variability in these habitats, and provide a better understanding of ecosystem function. The EMP has largely focused on characterizing relatively undisturbed tidally-influenced emergent wetlands that provide important rearing habitat for juvenile salmonids, which also serve as reference sites for restoration actions. 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 assessing interannual variability (or “trends”). Between 2005 and 2012, three to four status sites in a previously unsampled river reach (as denoted in the Classification described above) were selected for sampling each year, along with continued sampling of a growing number of trends sites (Table 1). Since 2007, we have conducted co-located monitoring of habitat structure, fish, fish prey, and basic water quality metrics at multiple emergent wetland sites throughout the lower river. In 2011, the Estuary Partnership added food web and abiotic conditions (i.e., conditions influencing productivity such as temperature, turbidity, dissolved oxygen, nutrients) sampling and analysis in both the mainstem Columbia River and at the trends sites.

In 2013, the EMP sampling scheme was adjusted to no longer include data collection at status sites and monitoring efforts focused solely on the six trends sites. The six trends sites are: Ilwaco Slough (2010-2015), Secret River (2010-2015), Welch Island (2010-2015), Whites Island (2009-2015), Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2015), and Franz Lake (2008-2009, 2011-2015). In 2015, an additional objective was undertaken by some EMP partners to address a question regarding usage of lower tributary habitats by juvenile salmon. Thus, fish community data were collected in the lower Grays River and lower Lewis River and food web metrics were collected in the lower Grays River. Habitat and hydrology data were collected at Cunningham Lake (in addition to the trends sites) as a reference site for habitat and hydrology representative of Reach F sites because vegetation has been periodically trampled by livestock at Campbell Slough in past years. Methods from the protocol Lower Columbia River Habitat Status and Trends (v1.0, [ID 85](#)) were used to monitor the status and trends of specified metrics.

Activities Performed, Year 11 Contract (October 1, 2014 – September 30, 2015):

- Salmonid occurrence, community composition, growth, condition, diet, prey availability, and residency
- Habitat structure, including physical, biological and chemical properties of habitats
- Food web characteristics, including primary and secondary production of shallow water 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.

Table 1. Summary of sampling effort by site and year(s) conducted at EMP sampling sites. Bold text indicates that data were collected in 2015.

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat¹	Fish & Prey⁶	Abiotic Conditions	Food Web⁵
A	Trend	Ilwaco Slough	BBM	2011-2015	2011-2013, 2015	2011-2013, 2015	2011-2013, 2015
B	Trend	Secret River	SRM	2008 ³ , 2012-2015	2012, 2013		2012, 2013
	Tributary	Grays River, lower	-		2015		2015
	Trend	Welch Island	WI2	2012-2015	2012-2015	2014 ²	2012-2015
C	Status	Ryan Island	RIM	2009	2009		
	Status	Lord-Walker Island 1	LI1	2009	2009		
	Status	Lord-Walker Island 2 ⁴	LI2	2009			
	Trend	Whites Island	WHC	2009-2015	2009-2015	2009, 2011-2015	2011-2015
	Status	Jackson Island	JIC	2010	2010		
	Status	Wallace Island	WIC	2010	2010		
	Status	Bradwood Landing	BSM		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	Deer Island	DIC	2011	2011		
	Status	Martin Island	MIM	2007			
	Status	Goat Island	GIC	2011	2011		
	Status	Burke Island	BIM	2011	2011		
	Tributary	Lower Lewis River	-		2015		
	Status	Lewis River Mouth	NNI	2007			
F	Status	Sauvie Cove	SSC	2005			
	Status	Hogan Ranch	HR	2005			
	Trend	Cunningham Lake	CLM	2005-2015	2007-2009		
	Trend	Campbell Slough	CS1	2005-2015	2007-2015	2008-2015	2010-2015
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	-			2012-2015	
H	Trend	Franz Lake (slough)	FLM	2008-2009, 2011-2015	2008-2009, 2011-2015	2011-2015	2011-2015
	Status	Sand Island	SIM	2008	2008	2008	
	Status	Beacon Rock		2008	2008		
	Status	Hardy Slough	HC	2008	2008		

¹ Vegetation biomass data were not collected at any EMP sites in 2014. Only the four upstream trends sites were sampled for biomass in 2015.

² Abiotic conditions sampling was conducted at Welch Island in 2014 only.

³ Site sampled as part of the Reference Site Study; thus, only vegetation and habitat data were collected.

⁴ Lord-Walker Island 2 was sampled by the EMP in conjunction with the Reference Site Study; thus, only vegetation and habitat data were collected.

⁵ Phytoplankton and zooplankton only sampled from 2011 – 2015.

⁶ Fish prey data were not collected for juvenile Chinook salmon diet and prey availability analyses in 2014.

1.3.2 Site Descriptions

In 2015, the EMP focused primarily on the six trends sites that were monitored over multiple years. Habitat and hydrology data were collected at all six trends sites plus Cunningham Lake, which is typically sampled for habitat and hydrology metrics as a control site due to livestock grazing activities that occasionally occur at the Campbell Slough site (Table 1). The lower reaches of two tributaries (Grays River and Lewis River) were sampled in 2015 for fish metrics to determine whether juvenile salmon of upriver origin were utilizing the lower portions of the tributaries during outmigration. Food web metrics were also collected in the lower Grays River in 2015 to correspond with fish collection surveys. Coordinates for trends sites sampled in 2015 are listed in Table 2. The 2015 trends monitoring sites are described in order below, starting at the mouth of the Columbia and moving upriver towards Bonneville Dam (Figure 1). Maps of the sites, including vegetation communities, are provided in Appendix B and photo points from all sampling years are provided in Appendix D.

Ilwaco Slough. This site is located in Reach A, EM Zone 1 at river kilometer (rkm) 6, southeast of the entrance of Ilwaco harbor, in Baker Bay, WA. The property is currently owned by Washington Department of Natural Resources. The site has developed in the past century as the bay 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. Ilwaco Slough 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 2a). 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). Selected as a long-term monitoring site in 2011, Ilwaco Slough was sampled for all EMP metrics until 2013. In 2014, only habitat and hydrology data were collected at this site, but sampling of the entire suite of monitoring metrics resumed in 2015.

Secret River. The Secret River marsh, located in Reach B, EM Zone 1 in Grays Bay at the mouth of Secret River at rkm 37, is an extensive marsh owned by the Columbia Land Trust. The site was 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 since then. The cause of this erosion is unknown. 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. Secret River was selected as a long-term monitoring site in 2012 and was sampled for all EMP metrics that year and in 2013. In 2014 and 2015, only habitat and hydrology data were collected at this site.

Welch Island. The monitoring site on Welch Island is located in Reach B, EM Zone 2 on the northwest (downstream) corner of the island at rkm 53, which is part of the Lewis and Clark National 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 historical late-1800's maps; however, the island has expanded since then and wetland vegetation has developed where there was previously open water near the location of the study site. 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 Reach C, EM Zone 2 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 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 connection to the river mainstem was approximately 0.7 km from the monitoring site. The site is characterized by high marsh, some willows, scattered large wood, and numerous small tidal channels.

Cunningham Lake. Cunningham Lake is a floodplain lake located in Reach F, EM Zone 4 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 2) 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 near 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. This site has been sampled exclusively for habitat and hydrology data.

Campbell Slough. The Campbell Slough site is located in Reach F, EM Zone 4 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. The US Fish and Wildlife Service manages the impact of reed canarygrass within the extensive refuge by allowing cattle grazing in some areas. The site is usually fenced off from cattle except for times during and immediately after high freshets, which can cause holes in the fencing due to high flows and occasional woody debris. 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. Since 2012 the site has been periodically grazed and trampled by cows, affecting primarily in the upper marsh portion of the site that is dominated by reed canarygrass.

Franz Lake. The long-term monitoring site located in Reach H, EM Zone 5, 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 is 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.

Table 2. Coordinates of the trends sites sampled 2014.

Site Name	Latitude	Longitude
Ilwaco Slough	46° 18.035'N	124° 2.784'W
Secret River	45° 9.561'N	122° 20.408'W
Welch Island	45° 47.032'N	122° 45.291'W
Whites Island	45° 9.561'N	122° 20.408'W
Cunningham Lake	45° 48.448'N	122° 48.285'W
Campbell Slough	45° 47.032'N	122° 45.291'W
Franz Lake	45° 36.035'N	122° 6.184'W



a) Ilwaco Slough



b) Secret River



c) Welch Island



d) Whites Island



e) Cunningham Lake



f) Campbell Slough



g) Franz Lake Slough

Figure 2. Ecosystem Monitoring sites sampled in 2015: (a) Ilwaco Slough; (b) Secret River; (c) Welch Island; (d) Whites Island; (e) Cunningham Lake; (f) Campbell Slough; (g) Franz Lake.

1.3.3 Water Year

The 2015 water year was characterized by average water levels in the fall of 2014, higher than average winter water levels, and lower than average spring and summer water levels as indicated by measurements just below Bonneville Dam (Figure 3). Peak water levels occurred in February rather than in the late spring and early summer. Winter high water was even more pronounced in December in the lower part of the estuary as runoff from the west side of the Cascade Range contributed to increased flows in the tributaries (see Section 3.1 Mainstem Conditions for more information). Hydrographs of all measured years at the trends sites, including 2015, are provided in Appendix A.

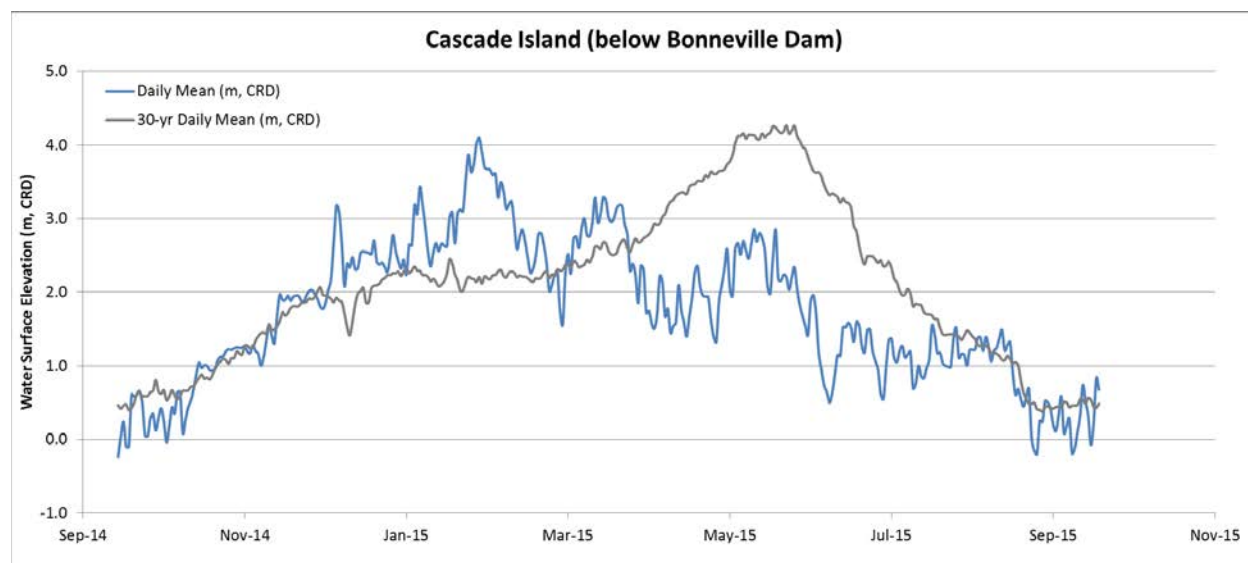


Figure 3. Water surface elevation at Cascade Island, just below Bonneville Dam (rkm 233), from October 2014 to October 2015 compared to the 29-year daily mean water surface elevation (Data from USGS National Water Information System at: <http://waterdata.usgs.gov/nwis/>).

2 Methods

2.1 Mainstem Conditions

2.1.1 Overview

The Center for Coastal Margin Observation and Prediction (CMOP) at Oregon Health & 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; Figure 4). 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; Figure 4). The monitoring protocol can be found on monitoringmethods.org ([Protocol ID 459](#)). 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 in near-real

time 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 capturing spatial and temporal resolution of basic water quality and biogeochemical observations for the mainstem Columbia River, an outcome of this effort is to provide daily estimates of parameters necessary for the assessment of ecosystem conditions at sites upstream and downstream of the Willamette-Columbia confluence. Knowledge of daily conditions at these sites allows the identification of contributions from lower river tributaries. Availability of these data enables the calculation of fluxes of various inorganic and organic components, such as nitrate concentration or phytoplankton biomass. Knowledge of nutrients and organic matter flux for a large river is important for a variety of applications, including assessment of pollution, indications of eutrophication, and quantification of material loading to the coastal zone, where many important ecological processes may be affected. Another product is the assessment of Net Ecosystem Metabolism (NEM), 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. NEM is often used by managers to identify changes or impairments to water quality (Caffrey 2004).

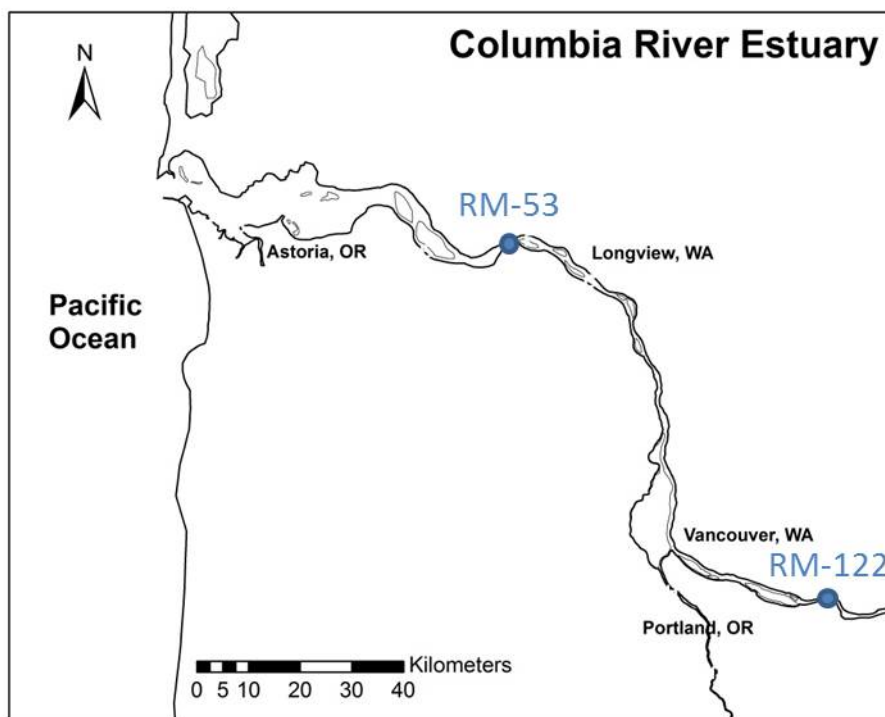


Figure 4. Station locations for the two in situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program. RM-53 (river mile 53) is Beaver Army Terminal, while RM-122 (river mile 122) is located in Camas, WA.

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

The instrument platform ran continuously from January 2015 – December 2015. In previous monitoring years, the instrument platform ran continuously from September 2013 – December 2013, and from July 2014 – December 2014. In December 2013 the instruments were removed for service and calibration and

were redeployed in June 2014 (a Yellow Springs Instruments (YSI) 6600 V2 sonde was deployed as a back-up instrument in the interim).

2.1.3 Sensor Configuration

Instruments and sensors common to both platforms are described in Table 3. Sensors are configured to collect a sample and telemeter the data every hour. In addition to the parameters listed in Table 3, 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). The filter size on the instrument is 10 μm , which is significantly higher than traditional filtered samples (0.45 μm). Therefore, data must be compared with caution, since some phosphate removed by traditional sampling is measured by the Cycle-PO4.

Table 3. Description of the components on the sensor platforms located at RM-53 and RM-122.

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

2.1.4 Sensor Maintenance

The sensors are designed to operate autonomously, at high temporal resolution (hourly), and over long periods between maintenance (estimated at three months, although sensors are typically maintained at shorter intervals). This is achieved through a design that maximizes power usage and minimizes biofouling. 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 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 4.

Table 4. Sensor maintenance dates at River Mile 122 in 2015.

RM-122	RM-53
2/6/2015	1/27/2015
2/20/2015	2/23/2015
3/6/2015	4/21/2015
3/27/2015	12/1/2015
4/13/2015	
5/4/2015	
5/29/2015	
7/1/2015	
7/21/2015	
7/29/2015	
8/6/2015	
8/17/2015	
8/24/2015	
9/4/2015	
9/30/2015	
11/7/2015	
12/17/2015	

2.1.5 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 Seabird SUNA for nitrate measurements operates with a calibration file determined at the factory under strictly controlled environmental conditions but 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 the laboratory at OHSU 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 5.

Table 5. Comparison of in situ data with laboratory measurements of water samples.

Location/Parameter/# measurements	Regression equation
RM-122/Nitrate/46	$Y = 0.95x + 1 \quad r^2 = 0.99$
RM-122/Chl/13	$Y = 0.8x + 1 \quad r^2 = 0.93$

2.2 Abiotic site conditions

2.2.1 Continuous Water Quality Data (Temperature, DO, pH, Conductivity)

In 2015, water quality was monitored at four of the trends sites, Franz Lake, Campbell Slough, Whites Island, and Ilwaco (Table 6). The monitoring protocol can be found on monitoringmethods.org ([Method ID 816](#)). Figure 5 shows how the sensors were deployed to ensure ready access for servicing and data downloads.

Table 6. Locations of water quality monitors at trends sites in 2015.

Site name*	USGS site number	USGS site name*	Reach	Latitude	Longitude	Monitor deployment date	Monitor retrieval date
Franz Lake	453604122060000	Franz Lake Slough Entrance, Columbia River, WA	H	45° 36' 04"	-122° 06' 00"	6/10/15	8/24/15
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47' 05"	-122° 45' 15"	4/29/15	8/24/15
Whites Island	460939123201600	Birnie Slough, White's Island, Columbia River, WA	C	46° 09' 39"	-123° 20' 16"	6/17/15	8/27/15
Ilwaco Slough			A	46° 18' 19"	-124° 02' 06"	6/8/15	8/26/15
Welch Island	461518123285700	Unnamed Slough, Welch Island, Columbia River, OR	B	46° 15' 18.4"	-123° 28' 56.8"	n/a	n/a

*Site names used in this report differ from official USGS site names to be consistent with site names used by other EMP partners.



Figure 5. Images showing deployment of water quality monitors (YSI sondes) at study sites.

The water quality monitors were YSI models 6600EDS and 6920V2, equipped with water temperature, specific conductance, pH, and dissolved oxygen probes. In addition, a fluorometer was installed at Campbell Slough capable of detecting and monitoring chlorophyll and phycocyanin. Table 7 provides information on the accuracy and effective ranges for each of these probes. The deployment period for these monitors was set to characterize water quality at the trends sites during the juvenile salmonid migration period. In 2015, the monitors were deployed from early April through early August, with visits approximately every four weeks to change the batteries, clean and calibrate the instruments, and make any necessary adjustments. In this report, given that the majority of the trends sites are located within Washington State; site-specific 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, available at <http://www.ecy.wa.gov/programs/wq/swqs/criteria.html>. Note that water temperature standards set by the Washington Department of Ecology (threshold of 17.5°C) are more conservative than those outlined by the maximum proposed by Bottom et al. (2011) used for comparisons in the mainstem conditions section of this report (Section 2.1).

Table 7. Range, resolution, and accuracy of water quality monitors deployed at four trends sites. ft, feet; m, meters; °C, degrees Celsius; µS/cm, microsiemens per centimeter; mg/L, milligrams per liter.

Monitoring Metric	Range	Resolution	Accuracy
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

2.2.2 Nutrients (N, P)

Nitrogen and phosphorus are nutrients that are often present at low enough concentrations to limit plant and phytoplankton growth in aquatic environments relative to other growth requirements. To analyze water column nutrient concentrations, 1 L water grab samples were collected from representative areas within the sites and subsampled prior to processing. Three fractions were determined from the subsamples: (1) dissolved inorganic species of nitrogen and phosphorus (nitrate, nitrite, ortho-phosphate, ammonium), (2) total dissolved nitrogen and phosphorus (TDN, TDP), and (3) total nitrogen and phosphorus (TN, TP). Nitrate+nitrite and orthophosphate were determined according to EPA standard methods (EPA 1983a), ammonium was determined colorimetrically (APHA 1998), and total phosphorus was determined according to USGS (1989). Detection limits for each ion or species are given in Table 8. The dates corresponding to sample collection are discussed in Section 2.4.1.2. The monitoring protocol can be found on monitoringmethods.org ([Method ID 1591](#)).

Table 8. Detection limits for colorimetric analysis of nitrogen and phosphorus species. TDN = total dissolved nitrogen, TN = total nitrogen, TDP = total dissolved phosphorus, TP = total phosphorus.

Ion or element	Detection limit (mg/L)
Ammonium	0.00280134
Nitrate + Nitrite	0.00700335
Nitrite	0.00140067
TDN	0.01540737
TN	0.1960938
Phosphate	0.00619476
TDP	0.00619476
TP	0.9601878
Silicic acid	0.0280855

2.3 Habitat Structure

In 2015, PNNL collected field data on vegetation and habitat conditions at the seven trends sites (Figure 1). Monitoring dates are provided in Table 9 and detailed maps of the 2015 monitoring sites are presented in Appendix B.

Table 9. Site location and sampling dates for each site sampled in 2015. All habitat and hydrology metrics were sampled at these sites except as otherwise noted.

Site Name	Site Code	River kilometer (rkm)	Site Type	Sampling Date
Ilwaco Slough (Baker Bay)*	BBM	6	Trend	8/2/15
Secret River (low marsh)*	SRM-L	37	Trend	8/1/15
Secret River (high marsh)*	SRM-H	37	Trend	8/1/15
Welch Island 2	WI2	53	Trend	7/31/15
Whites Island*	WHC	72	Trend	7/30/15
Cunningham Lake	CLM	145	Trend	7/28/15
Campbell Slough	CS1	149	Trend	7/29/15
Franz Lake	FLM	221	Trend	7/27/15

* Elevation data or channel cross section data were not collected.

2.3.1 Habitat Metrics Monitored

The habitat metrics in this study were monitored using standard monitoring protocols developed for the lower Columbia River (Roegner et al. 2009). Five metrics are included in this part of the monitoring program; however, in 2015 we focused our efforts on vegetation, elevation, hydrology, and sediment accretion. These metrics have been determined to represent important structural components, which can be used to assess habitat functions, although the data required to do so are limited in the lower river. 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). In the lowest part of the estuary, salinity is also an important factor determining vegetation composition and distribution. 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 site. Evaluating vegetative composition and species cover provides an indication of the condition of the site. Vegetation composition is important for 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 contributions to the biodiversity of the Columbia River estuarine ecosystem. Likewise, vegetative biomass is being collected at the trends 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 (i.e., habitat opportunity; Simenstad and Cordell 2000) and export of prey, organic matter, and nutrients. This information is also necessary to develop the relationship between channel cross-sectional dimensions and marsh size, which aids in understanding the channel dimensions necessary for a self-maintaining restored area (Diefenderfer and Montgomery 2009).

2.3.2 Annual Monitoring

The monitoring frequency for the habitat metrics depends on the variability of the metric between years. The composition, cover, and elevation of vegetation have been monitored annually since 2005. Since 2009, we have also measured channel cross sections, water surface elevation, and sediment accretion rates. Beginning in 2011, plant biomass was collected at all of the trends sites, excluding Cunningham Lake. In 2015, biomass was collected at the four upstream sites, including Cunningham Lake to maximize collection at sites with reed canarygrass. Sediment samples were collected once from each site to characterize sediment grain size and total organic content, but are not repeatedly collected. Similarly, vegetation community mapping methods were used to characterize the landscape at the site. After repeated mapping at each site, we determined that large-scale changes were not occurring between years; therefore this effort is no longer repeated during annual monitoring at trends sites unless vegetation changes are observed. Low inter-annual variability of channel morphology at the trends sites has been observed in prior sampling years, thus only the cross section at the channel mouth was measured in 2015. Photo points were also designated at each site from which photographs were taken to document the 360-degree view each year.

2.3.2.1 Hydrology

In 2009, pressure transducers (HOBO Water Level Data Loggers, Onset Computer Corporation) were deployed at Whites Island, Cunningham Lake, Campbell Slough, and Franz Lake as a means of logging hourly *in situ* water level data for one year. During the fall of 2010, a sensor was deployed at Ilwaco Slough that turned out to be faulty and was replaced in April 2011. Sensors were deployed at the Welch Island and Secret River sites in 2012. Occasionally sensor failure or loss has occurred; however, the sensors have been downloaded and redeployed every year since the initial deployment for collection of a nearly continuous dataset (Appendix A). The sensors were surveyed for elevation so that depth data could be converted to water surface elevation and evaluated against wetland elevations. The water surface elevation data was used to calculate the following annual hydrologic metrics for each site:

- Mean water level (MWL) – the average water level over the entire year
- Mean lower low water (MLLW) – the average daily lowest water level
- Mean higher high water (MHHW) – the average daily highest water level
- Annual water level range – the average difference between the daily high and low water levels
- Annual maximum water level – the maximum water level reached during the year

The monitoring protocol can be found on monitoringmethods.org ([Method ID: 3982](#)).

2.3.2.2 Sediment Accretion Rate

At each site, PVC stakes placed one meter apart 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 or pins, are used to determine gross annual rates of sediment accretion or erosion (Roegner et al. 2009). All previously installed sediment accretion stakes at the trends sites were measured in 2015. The accretion or erosion rate is calculated by averaging the 11 measurements along the one meter distance from each year and comparing the difference. The monitoring protocol can be found on monitoringmethods.org ([Method ID 818](#)).

2.3.2.3 *Salinity*

In order to better assess the influence of salinity on habitat, a conductivity data logger (Onset Computer Corporation) was deployed at the Ilwaco Slough site in August of 2011. The data logger records conductivity and temperature within the slough and derives salinity from those two measurements based on the Practical Salinity Scale of 1978 (see Dauphinee 1980 for the conversion). In February and August of 2015, the sensor was cleaned and downloaded, and a verification sample was taken. The monitoring protocol can be found on monitoringmethods.org ([Method ID 816](#)).

2.3.2.4 *Vegetation Species Assemblage*

The vegetation sampling 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 study area, 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 are 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). The monitoring protocol can be found on monitoringmethods.org ([Method ID 822](#)).

Along each transect, vegetative percent cover was evaluated at 2 – 10 m intervals. This interval and the transect lengths were based on the marsh size and/or the homogeneity of vegetation. At each interval on the transect tape, a 1 m² quadrat was placed on the substrate and percent cover was estimated by observers in 5% increments. If two observers were collecting data, 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.

2.3.2.5 *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 (Appendix B). Four trends sites (Ilwaco, Cunningham Lake, Campbell Slough, and Franz Lake) were re-mapped in 2015 because community changes were observed at those sites.

2.3.2.6 *Elevation*

In previous years, elevation was measured at all trends sites, corresponding to each of the following metrics: vegetation quadrats, the water level sensor, sediment accretion stakes, vegetation community boundaries, and in the channels. In 2015, elevation was re-measured at Welch Island, Cunningham Lake, Campbell Slough, and Franz Lake. Elevations from previous years were used at Ilwaco, Secret River, and Whites Island. 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 reviewed. 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 all data. 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. The monitoring protocol can be found on monitoringmethods.org ([Method ID 818](#)).

2.3.2.7 *Inundation*

The data from the water level sensors were used to calculate inundation metrics from the marsh and channel elevations collected at the sites. 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 (year-round, generally July to the following July) and 2) the period from March 1 through July 31, which represented the peak juvenile Chinook migration period in the lower river, as determined from data collected as part of this Ecosystem Monitoring Program and other studies (Bottom et al. 2005; Sather et al. 2009; Johnson et al. 2011).

In order to better assess hydrologic patterns and to make sites comparable over time and space, we calculated a single measurement that incorporates magnitude and duration of surface water flooding. Following work conducted in the US and in Europe (Simon et al. 1997; Gowing et al. 2002; Araya et al. 2010) we calculated the sum exceedance value (SEV) using the following equation:

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

where n is the number of hours present in the time period evaluated, and h_{elev} is the hourly water surface elevation above the marsh elevation. This differs from previous lower river 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 the same time periods must be used to compare calculations from past and present data.

For the trends analysis, the SEV was calculated for the average elevation of the three to five species that 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 and high water years to predict results in higher water years. The monitoring protocol can be found on monitoringmethods.org ([Method ID 954](#)).

2.3.2.8 *Vegetation Similarity Analysis at Trends 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 trends sites by using Primer™. Percent cover data were arc-sin, square-root transformed, but were not standardized, prior to analyses.

2.4 Food Web

2.4.1 Primary Productivity

2.4.1.1 *Emergent Wetland Vegetation*

Field Methods

From summer 2011 to winter 2015 above ground biomass was sampled to estimate the primary productivity at the six trends sites. Samples were collected in the summer during July or August during peak biomass and again in January or February during the winter low biomass period. For the emergent marsh biomass sampling, a 1 m² 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. Starting in 2012, the biomass was randomly sampled within distinct vegetation strata as determined by plant species dominance, to 1) more clearly associate the samples with vegetation type and 2) reduce the variability between samples within strata. Within the 1 m² biomass plot, a 0.1 m² quadrat was placed in a randomly selected corner and all rooted vegetation, live and dead, was removed using shears. Each sample was sorted to separate the primary strata species from other species and to distinguish live from dead then placed in a uniquely numbered bag and held in a cooler for the remainder of the sampling trip.

Dominant vegetation species were recorded in field notebooks along with the corresponding biomass sample number. Submerged aquatic vegetation (SAV) plots were sampled in 2011-2013 using similar methods, however due to the relatively low contribution of this strata to the overall macrodetritus production collection did not continue in subsequent years. The monitoring protocol can be found on monitoringmethods.org ([Method ID 1608](#)).

Laboratory 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 and obvious root material was removed. Pre-weighed pieces of tinfoil were used to secure the individual biomass samples, a wet weight was then measured, and the samples were placed in an oven set at 90°C for at least four days. When the samples were deemed completely dry, a dry weight was measured for each sample.

2.4.1.2 *Phytoplankton*

Abundance

Phytoplankton abundance was estimated in two ways: (1) from pigment concentrations, and (2) by direct counts using light microscopy. Phytoplankton abundance can be estimated by measuring the concentration of chlorophyll *a*, a photosynthetic pigment that is common to all types of phytoplankton. Water samples were collected into 1 L brown HDPE bottles and sub-sampled prior to processing. A subsample of water (typically between 60-300 mL) was filtered onto a 25 mL glass-fiber filter (GF/F) for chlorophyll *a* and kept frozen (-20°C) pending analysis. Chlorophyll *a* was determined fluorometrically using a Turner Designs Trilogy fluorometer according to the non-acidification method, which is highly selective for chlorophyll *a* even in the presence of chlorophyll *b* (Welschmeyer 1994).

Phytoplankton abundance was also determined by enumeration of individual cells using inverted light microscopy. The dates corresponding to sample collection for determination of nutrient concentrations, zooplankton abundance, and phytoplankton abundance are shown in Table 10. Duplicate 100 mL whole water samples were collected from each of the trends sites on the dates shown in Table 10. 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. The monitoring protocol can be found on monitoringmethods.org ([Method ID 1589](#) and [1590](#)).

Table 10. List of samples analyzed (Xs) and data of collection from four trends sites in the Lower Columbia River in 2015.

Site	Reach	Date	Nutrients	Zooplankton	Phytoplankton
ILWACO SLOUGH	A	4/6/15	X	X	X
	A	5/11/15	X	X	X
	A	6/23/15	X	X	X
	A	7/10/15	X	X	X
	A	7/28/15	X	X	X
	A	8/26/15	X	X	X
WELCH ISLAND	B	4/6/15	X	X	X
	B	5/20/15	X	X	X
	B	6/17/15	X	X	X
	B	7/15/15	X	X	X
	B	7/30/15	X	X	X
	B	8/24/15	X	X	X
GRAYS RIVER	B	4/7/15	X	X	X
	B	5/11/15	X	X	X
	B	6/23/15	X	X	X
	B	7/10/15	X	X	X
	B	7/28/15	X	X	X
WHITES ISLAND	C	4/7/15	X	X	X
	C	5/20/15	X	X	X
	C	6/17/15	X	X	X
	C	7/15/15	X	X	X
	C	7/30/15	X	X	X
	C	8/27/15	X	X	X
CAMPBELL SLOUGH	F	4/8/15	X	X	X
	F	5/12/15	X	X	X
	F	6/16/15	X	X	X
	F	7/13/15	X	X	X
	F	7/29/15	X	X	X
	F	8/24/15	X	X	X
FRANZ LAKE SLOUGH	H	4/8/15	X	X	X
	H	5/12/15	X	X	X
	H	6/18/15	X	X	X
	H	7/13/15	X	X	X
	H	7/29/15	X	X	X
	H	8/24/15	X	X	X

Multivariate Statistical Analyses

Nonmetric Multi-dimensional Scaling (NMDS) and Canonical Analysis of Principal Coordinates (CAP) routines were performed using PRIMER-E v.7 with PERMANOVA+. NMDS is a multivariate technique that identifies the degree of similarity among biological communities within a group of samples in a data set. In NMDS, samples are typically represented in 2-dimensional ordination space using distance between sample points as a measure of similarity of biological communities; short distances represent relatively high similarity between samples, while longer distances represent relatively low similarity between samples.

Major phytoplankton taxa were selected for multivariate analyses if their abundance constituted at least 10% of total phytoplankton abundance in any sample. Taxa that did not meet these criteria were excluded from analysis. Two NMDS analyses were run for this study that included (i) all major phytoplankton taxa

(NMDS_{total}) and (ii) only major diatom taxa (NMDS_{diatom}). Abundances for 25 major phytoplankton taxa (NMDS_{total}) and 10 major diatom taxa (NMDS_{diatom}) were standardized by sample and the data were square-root transformed in order to achieve a normal distribution of the data prior to analysis. Canonical Analysis of Principal Coordinates (CAP) is an analytical technique that uses canonical correlation to determine the degree to which environmental factors explain variability among biological communities. A Bray-Curtis resemblance matrix was assembled using the standardized, square-root transformed phytoplankton abundance data and six environmental variables including NO₂-+NO₃-, NH₄⁺, PO₄³⁻, mean daily water temperature, mean daily dissolved oxygen saturation, and mean daily discharge (at Bonneville Dam). Environmental data were normalized prior to analysis to compare variables at the same scale. Samples with missing environmental data were excluded from multivariate analyses. A total of 70 samples were analyzed in both NMDS analyses, and a total of 38 samples were included for CAP.

2.4.2 Secondary Productivity

2.4.2.1 Zooplankton

Secondary productivity (the rate of growth of consumers of primary production) was not measured directly, but was estimated from the abundance of pelagic zooplankton. The samples were collected from near the surface of the water (< 1 m depth) using an 80 µm nylon mesh net with a mouth diameter of 0.5 m and a length of 2 m at four trends sites (Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). A list of the collection dates and sampling sites are given above in Table 10. The monitoring protocol can be found on monitoringmethods.org ([Method ID 4189](#)).

Abundance

Zooplankton abundances collected via net tow were determined at each of four trends sites (Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). The net was fully submerged under the water and was dragged back and forth from a small boat through the water for approximately 3-5 min or over approximately 100 m. 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 volume of water passing through the net was determined 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:

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

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:

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

The volume is determined as:

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

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 was determined by multiplying the concentration factor by the final volume of concentrated sample examined under the microscope.

Taxonomy

Zooplankton taxa were broadly categorized into one of the following groupings: rotifers, cladocerans, annelids, ciliates, and copepods, and 'other'. Within these groups, individuals were identified to genus or species where possible (rotifers, cladocerans, ciliates, annelids), or to order (copepods). Eggs of rotifers, cladocerans, and copepods were enumerated separately.

2.4.3 Stable Isotope Ratios

The ratios of carbon (C) and nitrogen (N) stable isotopes in tissues of consumers reflect the stable isotope ratios (SIR) of their food sources (Neill and Cornwell 1992; France 1995). Therefore, SIR are useful in the determination of major food sources, as long as the latter have distinct isotopic ratios that allow them to be distinguished. Within the scope of the EMP, SIR analysis is used to estimate the relative importance of food sources including algae and wetland plants to the food web supporting juvenile salmonids at trends sites including Ilwaco, Whites Island, Campbell Slough, and Franz Lake Slough. SIR are suitable for identifying food sources assimilated over a longer time frame compared to point-in-time techniques such as gut content analysis; ideally, a combination of the two approaches provides the best indicator of diet.

C and N isotope ratios yield different information: since the $^{13}\text{C}/^{12}\text{C}$ ($\delta^{13}\text{C}$) ratio varies by only a small amount (<1‰) during the assimilation of organic matter, it is used to identify the primary source of organic matter (i.e., primary producers). In contrast, the ratio of $^{15}\text{N}/^{14}\text{N}$ ($\delta^{15}\text{N}$) changes markedly with trophic level, increasing by 2.2 to 3.4 parts per thousand (per mil, or ‰) with an increase of one trophic level (i.e., from a plant to an herbivore or an herbivore to a carnivore). Thus, $\delta^{15}\text{N}$ values are useful in determining trophic position.

The SIR of C and N were measured in juvenile Chinook salmon muscle tissues and several potential food sources to provide information on the food web supporting juvenile salmonids (Table 11). Juvenile salmon were collected by NOAA Fisheries staff during monthly beach seine sampling and frozen (see Section 2.6). Skinned muscle samples were collected for analysis since SIR signatures are more homogeneous within muscle tissue and since muscle is a good long-term integrator of food source.

Aquatic invertebrates were collected using a 250 μm mesh net with a rectangular opening in emergent vegetation at the water's margin. The aquatic midge, Chironomidae, and amphipods were selected because they have been found to be preferred food sources for juvenile salmonids in the lower Columbia River (Maier and Simenstad 2009; Sagar et al. 2013, 2014, 2015). Two amphipod taxa, *Corophium* spp. and *Gammarus* spp., were available from most sites and were also collected for analysis. Most invertebrate specimens were found attached to submerged portions of vegetation. Invertebrates were collected by rinsing the exterior of the vegetation with deionized water and removing the invertebrates from the rinse water using clean forceps. Invertebrate samples were then rinsed with deionized water to remove algae or other external particulate matter. Salmon and aquatic invertebrate samples were frozen for later processing.

Table 11. Potential food sources for marked and unmarked juvenile Chinook salmon and invertebrate consumers.

Fish		Invertebrates
Marked Chinook salmon	Unmarked Chinook salmon	Chironomidae, amphipods, zooplankton
Chironomidae	Chironomidae	Particulate organic matter (POM)
<i>Corophium</i> spp.	<i>Corophium</i> spp.	Periphyton
<i>Gammarus</i> spp.*	<i>Gammarus</i> spp.*	Live vegetation
Zooplankton	Zooplankton	Dead vegetation
Hatchery food*		

*Not analyzed in 2015

A variety of autotrophs were sampled to characterize the range of potential food sources for invertebrates. Samples of terrestrial and emergent vegetation, aquatic macrophyte, and macroalgae species were collected from representative areas within each site Table 11. Vegetation 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. Samples of particulate organic matter (POM) and periphyton were filtered onto 25 mm glass-fiber GF/F filters and were frozen (-20°C) for later processing.

Frozen filters, salmon tissue, invertebrate, and plant material were freeze dried using a Labconco FreezeZone 2.5L benchtop freeze dry system (Labconco Corp., USA). Plants were categorized as live or dead during field collections based on whether they were attached and by their physical appearance; mixtures of live plants from the same sampling date were composited and ground using a mortar and pestle, as were mixtures of dead vegetation (designated when plant material was detached rather than rooted). Freeze-dried invertebrates of the same taxa from the same collection site and collection date were composited, ground using a clean 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 composited into a single sample. Skinned muscle tissue samples from individual juvenile salmonids were analyzed separately by individual; muscle tissue samples from different bodies were not composited.

SIR of carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) were determined at the UC Davis Stable Isotope Facility using a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (Sercon Ltd., Cheshire, UK). The atomic ratios of the heavy isotope (^{13}C , ^{15}N) to the light isotope (^{12}C , ^{14}N) were compared to universal standards (Vienna PeeDee Belemnite and air for C and N, respectively) and reported in per mil (‰) units.

2.5 Macroinvertebrates

2.5.1 Salmon Prey Availability Sampling

2.5.1.1 Open Water and Emergent Vegetation

To assess availability of salmon prey at the trends sites, we conducted neuston tows in both open water (OW; in the center of the channel) and emergent vegetation (EV; along edge of the wetland channel among vegetation). For OW samples, a Neuston net (250 μm mesh) was deployed from a boat for an

average distance of 100 m and positioned to sample the top 20 cm of the water column. For EV samples, the Neuston net was pulled through a 10 m transect parallel to the water's edge in water at least 25 cm deep to enable samples from the top 20 cm of the water column. Neuston tows were taken concurrently with monthly beach seine collections when juvenile Chinook salmon were present at a site (i.e., captured during seine sets). Two OW and two EV samples were collected at each site per month; although, occasionally one or three tows were performed in each habitat type depending on field conditions (Table 12). Samples were preserved in 10% formalin until delivered to the laboratory for processing. The monitoring protocol can be found on monitoringmethods.org ([Method ID 1622](#)).

Table 12. The number of invertebrate tow samples (OW and EV) collected at each site per sampling event, 2008-2013 and 2015.

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Hardy Slough	Total Tow Samples
2008	April	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	6	6	0	15
	May	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	6	0	12
	June	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	4
2009	May	0	0	0	3	0	0	4	0	4	0	0	0	5	0	0	0	4	0	20
	June	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	4
2010	April	0	0	0	0	4	4	4	4	0	0	0	0	4	0	0	0	0	0	20
	May	0	0	0	0	4	4	4	4	0	0	0	0	4	0	0	0	0	0	20
	June	0	0	0	0	4	4	4	4	0	0	0	0	4	0	0	0	0	0	20
	July	0	0	0	0	4	0	4	4	0	0	0	0	2	0	0	0	0	0	14
2011	April	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	May	8	0	0	0	0	0	10	0	0	4	4	4	4	0	0	0	2	0	36
	June	4	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	8
2012	February	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
	March	0	0	2	0	0	0	2	0	0	0	0	0	0	3	0	0	0	0	7
	April	0	4	5	0	0	0	6	0	0	0	0	0	0	4	2	0	0	0	21
	May	0	1	4	0	0	0	4	0	0	0	0	0	4	4	4	0	0	0	21
	June	0	6	4	0	0	0	4	0	0	0	0	0	4	2	4	0	0	0	24
2013	March	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
	May	0	4	4	0	0	0	4	0	0	0	0	0	4	0	0	0	0	0	16
	June	0	4	4	0	0	0	3	0	0	0	0	0	4	0	0	0	0	0	15
	July	0	0	4	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	10
2015	April	5	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	6	0	17
	May	0	0	2	0	0	0	4	0	0	0	0	0	2	0	0	0	5	0	13
	June	0	0	6	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	10
Total Tow Samples		19	23	39	3	16	12	71	16	4	4	4	4	60	13	10	6	29	4	337

2.5.1.2 *Benthic Macroinvertebrates*

To characterize the benthic macroinvertebrate assemblage, benthic core sites were selected to correspond to locations directly adjacent those where the fish community, food web metrics, and vegetation were sampled. Benthic cores were collected monthly at the trends sites (n = 5 per site) between April and July. Cores were collected to a depth of 10 cm by driving a 2 inch diameter PVC pipe into the ground at each sampling location. Each core was then placed in a jar and fixed in 10% formalin. Core samples were collected at low tide from exposed sediments and among emergent vegetation. The monitoring protocol can be found on monitoringmethods.org ([Method ID 1593](#)).

2.5.1.3 *Laboratory Methods*

Invertebrates collected in neuston tows and benthic cores were identified in the lab using high-resolution optical microscopy and taxonomic references (Mason 1993, Kozloff 1996, Merritt and Cummins 1996, Thorp and Covich 2001, Triplehorn and Johnson 2005). Most individuals were identified to family, although some groups/individuals were identified to coarser (e.g., order) levels. For each sample, the number of individuals in each taxonomic group was counted, then each group was blotted dry and weighed to the nearest 0.0001 g.

Analysis of neuston tow data included all invertebrates except the single-celled protist Foraminifera, which were abundant in samples from Ilwaco Slough in April. In benthic core samples, taxa that were not aquatic and/or benthic in their ecology (e.g., adult flies) were considered contaminants and were excluded from analyses of benthic core data.

2.5.2 *Salmon Diet*

2.5.2.1 *Field Data Collection*

When juvenile Chinook were captured at a site, fish were typically euthanized within an hour of collection. If fish were not processed immediately, they were kept on ice until later in the same day when stomachs were extracted. Whole stomach samples were preserved in 10% formalin until delivered to the laboratory for processing. The total number of diet samples collected at EMP sites since 2008 is provided in Table 13.

2.5.2.2 *Laboratory Methods*

Organisms in the diets were identified in most cases to family level, although some groups/individuals were identified to coarser (e.g., order) levels, and crustaceans were usually identified to genus or species. Some contents were unidentifiable due to digestion. Each prey taxon was counted, blotted on tissue, and weighed to the nearest 0.0001 g.

Table 13. The number of Chinook salmon diet samples collected at each site per sampling event, 2008-2013, 2015.

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Pierce Island	Hardy Slough	Total Tow Samples
2008	April	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	13	15	9	0	43
	May	0	0	0	0	0	0	0	0	0	0	0	0	19	0	0	0	7	0	0	26
	June	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13	13
2009	May	0	0	0	9	0	0	10	0	6	0	0	0	10	0	0	0	8	0	0	43
	June	0	0	0	10	0	0	0	0	0	0	0	0	9	0	0	0	0	0	0	19
2010	April	0	0	0	0	10	19	16	6	0	0	0	0	12	0	0	0	0	0	0	63
	May	0	0	0	0	17	15	14	14	0	0	0	0	24	0	0	0	0	0	0	84
	June	0	0	0	0	9	8	18	11	0	0	0	0	18	0	0	0	0	0	0	64
	July	0	0	0	0	10	0	19	11	0	0	0	0	15	0	0	0	0	0	0	55
	August	0	0	0	0	8	0	13	0	0	0	0	0	0	0	0	0	0	0	0	21
2011	May	0	0	0	0	0	0	10	0	0	10	13	10	22	0	0	0	0	0	0	65
	June	0	0	0	0	0	0	25	0	0	0	0	0	0	0	0	0	0	0	0	25
	July	0	0	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	0	4
2012	February	0	15	16	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	31
	March	0	0	14	0	0	0	13	0	0	0	0	0	0	13	0	0	0	0	0	40
	April	0	15	14	0	0	0	10	0	0	0	0	0	0	7	15	0	0	0	0	61
	May	0	0	30	0	0	0	11	0	0	0	0	0	18	15	18	0	0	0	0	92
	June	0	14	15	0	0	0	15	0	0	0	0	0	15	15	36	0	0	0	0	110
2013	March	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9
	May	0	12	30	0	0	0	15	0	0	0	0	0	34	0	0	0	0	0	0	91
	June	0	1	23	0	0	0	13	0	0	0	0	0	9	0	0	0	0	0	0	46
	July	0	2	25	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	28
2015	April	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
	May	0	0	15	0	0	0	15	0	0	0	0	0	15	0	0	0	4	0	0	49
	June	0	0	7	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	20
Total Tow Samples		6	59	198	19	54	42	232	42	6	12	13	10	227	50	69	13	34	9	13	1108

2.5.3 Salmon Prey Data Analysis

Descriptive statistical analysis of the whole invertebrate community was calculated, in addition to specific analyses of the order Diptera (flies) and amphipod taxa that have been shown to be important prey of juvenile Chinook salmon in the lower Columbia River (Lott 2004, Spilseth and Simenstad 2010). For benthic cores, the density and biomass of taxa in each sample were calculated as the total count or weight for a given taxon divided by the core volume (# individuals/m³, g/m³). For neuston tows, the density and biomass of taxa in each sample were calculated as the total count or weight for a given taxon divided by the meters towed (# individuals/m towed, mg/m towed). To compare taxa densities and biomass between study sites, density and biomass data for each taxon were summed across replicate samples taken within a given site each month, and then divided by the number of replicates to give an average total density and biomass at each sampling site per month.

Multivariate analyses were used to examine differences in the invertebrate assemblage between sites using the PRIMER (Plymouth Routines In Multivariate Ecological Research) software package developed at the Plymouth Marine Laboratory (Clarke and Warwick 1994; Clarke and Gorley 2006). Taxa were initially combined into broad groups for analysis of community composition. Similarity indices were calculated for the average site abundance of each invertebrate taxon using the Bray-Curtis similarity coefficient as a measure of distance between sites. The density data were square root transformed prior to analysis. A non-metric, multi-dimensional scaling (MDS) ordination plot was used to show similarity. The MDS plots observations as points such that those close together represent samples similar in community composition, and points far apart correspond to different composition values.

Diet composition was assessed as the percent of the total index of relative importance (% IRI) for each taxon, as calculated in Liao et al. 2001, where:

$$IRI_i = (P_{i, \text{numeric}} + P_{i, \text{gravimetric}}) \times FO_i$$

and % IRI is the percentage of the total IRI for prey taxa i . In the equation, P is the percent numeric and gravimetric composition of total prey and FO is the percent frequency of occurrence of prey i . This index is recommended because it accounts for prey weight and numbers, as well as the likelihood of taxa appearing in the diet of individuals (frequency of occurrence; Liao et al. 2001). Because the index incorporates taxa counts, items that were not countable (e.g., plant matter, unidentifiable, highly-digested material), were removed from descriptive analyses of diet composition.

Instantaneous ration (IR) was calculated as a measure of fish condition or fitness. IR is the ratio of the total diet weight to the total fish weight. Total diet weight was calculated as the sum of the weights of all individual taxa counted in the diet, except that only nutritious diet items were included in IR calculations; sediment and plant matter were excluded. For descriptive analyses, IR was calculated for each individual salmon diet, and averaged across all fish within a given habitat and month.

Ivlev's electivity index (E) was used to assess selection by juvenile Chinook of prey items. This index compares the relative availability of prey in the environment and their relative contribution to diets, such that:

$$E_i = (r_i - P_i) / (r_i + P_i)$$

where r_i is the relative abundance of a prey in the diet and P_i is the prey's relative abundance in the ecosystem. Electivity is scaled between -1 and 1, where -1 indicates total avoidance of a prey; 0 indicates

that a prey is taken in proportion to its abundance in the ecosystem; and 1 indicates total preference for a prey. Neuston tow density data was used to represent prey abundance. Where both emergent vegetation and open water samples were collected, an average density over both habitats was used to calculate electivity.

2.6 Fish

2.6.1 Fish Community

In 2015, NOAA Fisheries monitored habitat use by juvenile Chinook salmon and other fishes at five trends sites, Franz Lake in Reach H (previously sampled in 2008 – 2014), Campbell Slough in Reach F (sampled from 2007-2014), Whites Island site in Reach C (sampled from 2009-2014), Welch Island in Reach B (sampled from 2012-2014), and Ilwaco Slough in Reach A (sampled from 2011-2014), in order to examine year-to-year trends in fish habitat use in the lower river. Coordinates of the sampling sites are shown in Table 2.

Fish were collected from March 2015 through July 2015, then again in September, November and December 2015. Because of high water temperatures fishing was not permitted at any sites in August and only at Ilwaco Slough in September. Fish were collected using a Puget Sound beach seine (PSBS; 37 x 2.4 m, 10 mm 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. All captured fish were identified to the species level and counted. Salmonid species (up to 30 specimens) were measured (fork length in mm) and weighed (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. The monitoring protocol can be found on monitoringmethods.org ([Method ID 826](#)). Fish sampling events conducted in 2015 are shown in Table 14.

Table 14. Number of beach seine sets per month at EMP sampling sites in 2015. Grays and Lewis River Tributary sites were sampled only from April through June. NS = not sampled.

Site	Mar	Apr	May	Jun	Jul	Sep	Nov	Dec	Total
Ilwaco Slough	NS ¹	3	3	3	3	3	3	3	21
Welch Island	3	1	1	3	3	NS ²	NS ³	NS ³	14
Whites Island	NS ¹	1	1	3	3	NS ²	NS ³	NS ³	8
Campbell Slough	NS ¹	3	3	3	3	NS ²	3	3	18
Franz Lake	NS ¹	3	1	3	3	NS ²	3	3	16
Grays River Lower	NS	1	1	1	NS	NS	NS	NS	3
Grays River Upper	NS	1	1	1	NS	NS	NS	NS	3
Lewis River Lower	NS	1	1	1	NS	NS	NS	NS	3
Lewis River Upper	NS	1	1	NS ¹	NS	NS	NS	NS	2
Total	3	11	9	15	15	3	9	9	85

¹Not sampled due to permit issues

²Not sampled due to high water temperatures

³Not fishable due to strong currents

In addition to the trends sites, Columbia River tributary sites in the Grays River and Lewis River were also sampled from April through June as a pilot project to explore use of tributary sites by juvenile salmonids. At these sites, only one beach seine set was conducted per sampling event. Fish sampling events at these sites are shown in Table 14.

When juvenile Chinook salmon were captured, up to 30 individuals were collected for necropsy at each field site during each sampling effort. Salmon fork length were measured (to the nearest mm) and weighed (to the nearest 0.1 g), then euthanized by anesthesia with a lethal dose of MS-222. For each juvenile Chinook salmon, the following samples were collected: 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) and stomach contents for measurement POPs, including PAH, DDTs, PCBs, PBDEs, and various organochlorine pesticides. At the tributary sites, fin clips were taken for genetics, but no other samples were collected and fish were released.

Samples for chemical analyses were frozen and stored at -80°C until lab 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 richness (S ; the number of species present) and fish species diversity for each site were calculated by month and year. Fish species diversity was calculated using the Shannon-Weiner diversity index (Shannon and Weaver 1949):

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

Where

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

N = the total number of all individuals

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

Catch per unit effort (CPUE) and fish density were calculated as described in Roegner et al. (2009), with fish density reported in number per 1000 m².

2.6.2 Salmon Metrics

2.6.2.1 Genetic Stock Identification

Genetic stock identification (GSI) techniques were used to investigate the origins of juvenile Chinook salmon captured in habitats of the Lower Columbia River Estuary (Manel et al. 2005; Roegner et al.

2010; Teel et al. 2009). Juvenile Chinook salmon stock composition and individual assignment were estimated by using a regional microsatellite DNA baseline data set of known-origin reference populations (Seeb et al. 2007). The specific baseline data we used included 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 by the conditional maximum likelihood methods of Rannala and Mountain (1997), implemented in the computer program ONCOR (Kalinowski et al. 2007). We also used the CML method to assign individual fish to the most likely stock of origin according to the maximum a posteriori probability of membership. Individual fish assigning with lower probability than 0.8 were deemed of uncertain origin and were omitted from consideration of stock-specific ecological traits (see below, Moran et al. 2014 for methodological detail). Probability of origin was estimated for the following regional genetic stock groups: West Cascades fall Chinook, West Cascades spring Chinook, Middle and Upper Columbia spring Chinook, Spring Creek Group fall Chinook, Deschutes River fall Chinook, Snake River fall Chinook, Snake River spring Chinook, Upper Columbia River summer/fall Chinook, and Upper Willamette River spring Chinook (Seeb et al. 2007; Teel et al. 2009). West Cascades and Spring Creek Group Chinook are Lower Columbia River stocks. The monitoring protocol can be found on monitoringmethods.org ([Method ID 948](#)).

2.6.2.2 *Lipid Determination and Condition Factor*

As part of our study we determined lipid content in Chinook 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, whole body samples from salmon collected in the field were composited by genetic reporting group, date, and site of collection into a set containing 3-5 fish each. Using the composited salmon whole body samples, the total amount of extractable lipid (percent lipid) was determined by Iatrosan 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 (K ; Fulton 1902; Ricker 1975) was calculated as an indicator of fish health and fitness, using the formula:

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

The monitoring protocol can be found on monitoringmethods.org ([Method ID 952](#)).

2.6.2.3 *Otoliths (Growth Rates)*

Otoliths were dissected from juvenile Chinook salmon collected at EMP status and trends sampling sites (including toxic contaminant sampling sites; Lower Columbia Estuary Partnership 2007), as well as Action Effectiveness Monitoring sites from May to June in 2005 and 2007-2012 ($n = 28$ sites). Otolith data collected from action effectiveness monitoring sites and the toxic contaminant study in addition to EMP status and trends sites to allow for the most comprehensive analysis possible. Otoliths from fish ranging in fork length from 37-111 mm (mean = 67 mm, SD = 13 mm) were processed for microstructural analysis of recent growth. Specifically, left sagittal otoliths were embedded in Crystal Bond® and polished in a sagittal plane using slurries (Buehler®'s 600 grit silicon carbide, 5.0 alumina oxide, and 1.0 micropolish) and a grinding wheel with Buehler® 1500 micropolishing pads. Polishing ceased when the core of the otolith was exposed and daily increments were visible under a light microscope. Polished otoliths were photographed using a digital camera (Leica DFC450) mounted on a

compound microscope (Zeiss®). Using Image Pro Plus® (version 7, Mediacybernetics) the average daily growth rate for each individual was determined (i.e., mm of fish length/day) for the last seven days of their life. A total of 500 otoliths were analyzed. Daily growth rate (DG, mm/day) was determined using the Fraser-Lee equation:

$$L_a = d + \frac{L_c - d}{O_c} O_a$$

$$DG = \frac{L_c - L_a}{a}$$

where L_a and O_a represents fish length and otolith radius at time a (i.e., last seven days), respectively, d is the intercept (11.9 mm) of the regression between fish length and otolith radius, L_c and O_c are the fish length and otolith radius at capture, respectively. The monitoring protocol can be found on monitoringmethods.org ([Method ID 949](#)).

Analysis of Variance (ANOVA) was used to determine whether growth rates differed among sites, and if significant differences were detected, a Bonferroni post-hoc test was used to identify which sites differed. Since several sites were repeatedly sampled for fish over multiple years, we also used an ANOVA to assess whether somatic growth rate varied annually within each of the following sites: Campbell Slough, Franz Lake, Mirror Lake #1, Mirror Lake #4, (Schwartz et al. 2013) and Confluence Washington (a site that was sampled as part of the studies conducted by NOAA for the Lower Columbia Estuary Partnership in 2007, a 2008 Portland Harbor Natural Resource Damage Assessment Trustees project, and a 2013 PAH study). We also used an ANOVA to determine if somatic growth rate differed among fish grouped according to genetic stock and whether fish were marked or unmarked. Lastly, we used generalized linear models to assess how somatic growth rate (response variable) varied according to seven predictor variables: collection year, genetic stock, marked or unmarked, river kilometer, and river reach. We ran 128 models (including a null model with no effects) representing all possible combinations of the aforementioned seven variables. All model parameters were estimated by maximizing the likelihood function. To compare models, Akaike's Information Criterion (AIC; Akaike 1973; Burnham and Anderson 2002) was calculated for each model, such that smaller AIC values indicated "better" models. When comparing two models, the difference in AIC values (delta AIC) was computed, and according to Burnham and Anderson (2002), a delta AIC of less than 2 indicates little difference between competing models; a delta AIC of 2–10 indicates moderate support for a difference between the models, and a delta AIC of greater than 10 indicates strong support.

2.6.2.4 *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. (2006, 2014). 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 (Σ 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 (Σ 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 (Σ HCHs) were calculated by adding the concentrations of *a*-HCH, *b*-HCH, *g*-HCH, and lindane. Summed low molecular weight aromatic hydrocarbons (Σ LAHs) were determined by adding the concentrations of biphenyl, naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, 2,6-dimethylnaphthalene, acenaphthene, fluorene, phenanthrene; 1-methylphenanthrene, and anthracene. Summed high molecular weight aromatic hydrocarbons (Σ 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 (Σ TAHs) were calculated by adding Σ HAHs and Σ 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. The monitoring protocol can be found on monitoringmethods.org ([Method ID 950](#)).

2.6.2.5 *PIT Tag Array*

A passive integrated transponder (PIT) tag detection system was installed at Campbell Slough in June 2011, approximately 150 m into the slough channel from the mainstem Columbia River. The 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 for daily data downloads. The array is intended to monitor presence and to estimate residency of PIT tagged fish in Campbell Slough.

3 Results

3.1 Mainstem Conditions

3.1.1 Continuous data from the mainstem

3.1.1.1 *Discharge at Beaver Army Terminal (RM-53)*

Columbia River flow fluctuates from year to year based on snowmelt and local precipitation. Compared to years with high (1997) and low (2001) daily averaged river discharge ($\text{m}^3 \text{s}^{-1}$), 2014 and 2015 were relatively low, but higher than values observed in 2001 during the early part of the season (Figure 6). Because there was no significant freshet observed in 2015, river discharge (volume flux) around the typical time of the freshet was similar to that seen in 2001. However, the summer low flows in both 2014 and 2015 were similar to the long-term average, while the low flows in 2001 were lower than average. The repercussions of two consecutive low-discharge years on system ecology are presently unknown.

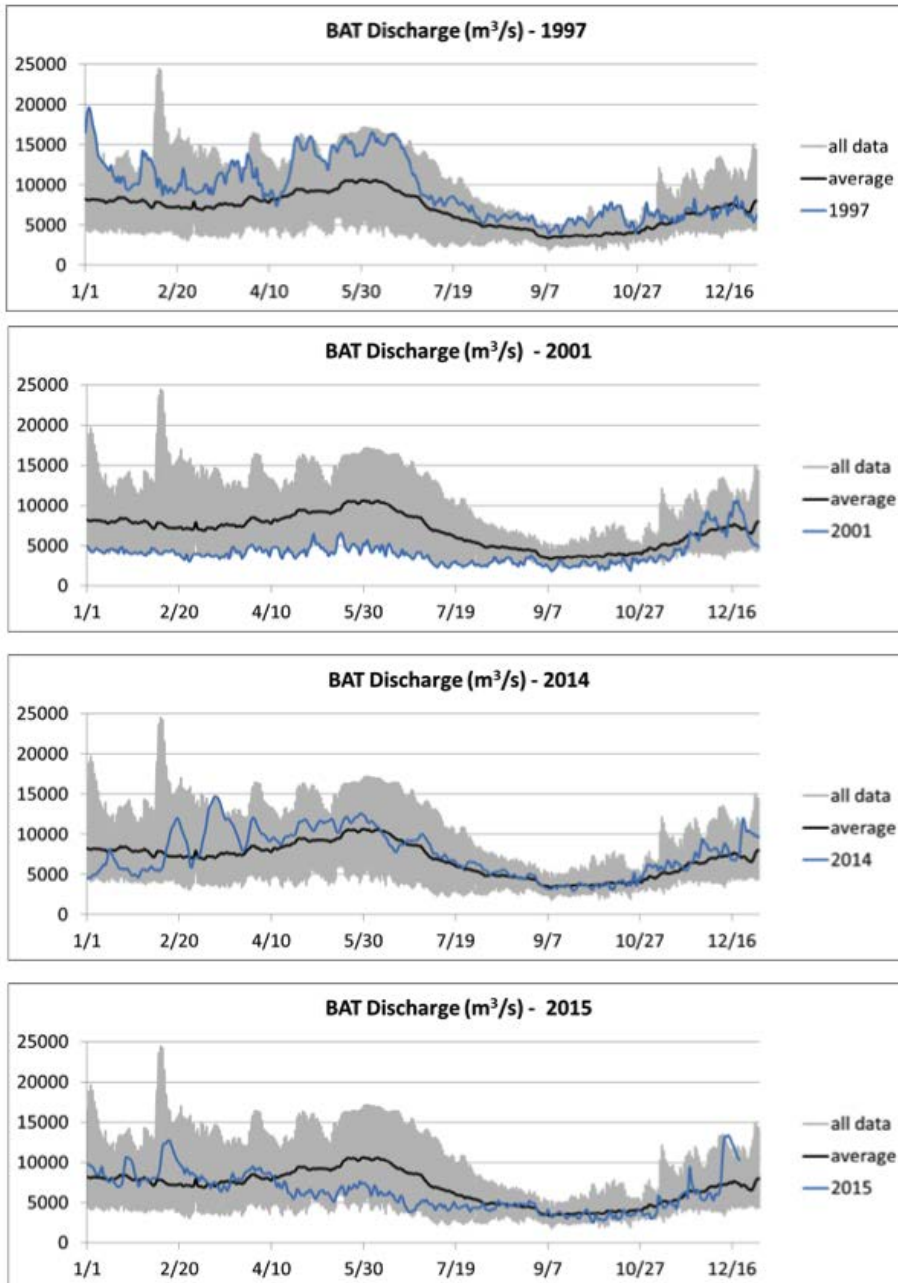


Figure 6. Daily average river discharge at Beaver Army Terminal (BAT) during years with high (1997) and low (2001) flow, relative to the long-term average (shown in the solid black line). Both 2014 and 2015 had low daily average river discharge volumes compared to the long-term mean (two lower panels). Low river discharge values in both 2014 and 2015 were similar to those in 2001; however, early season values were higher in 2014 and 2015 compared to 2001.

3.1.1.2 *Water temperature at Beaver Army Terminal (RM-53)*

Daily river discharge was lower in 2001 compared to 2014 or 2015; however, the daily average water temperatures were higher in 2015 compared to 2014 and to the year 2001, when discharge was very low. In 2001, daily temperatures in the mainstem were about average, with the exception of a few weeks in

May where temperatures were high. In contrast, temperatures in 2015 were higher than the long-term average throughout the spring and summer until approximately September (Figure 7).

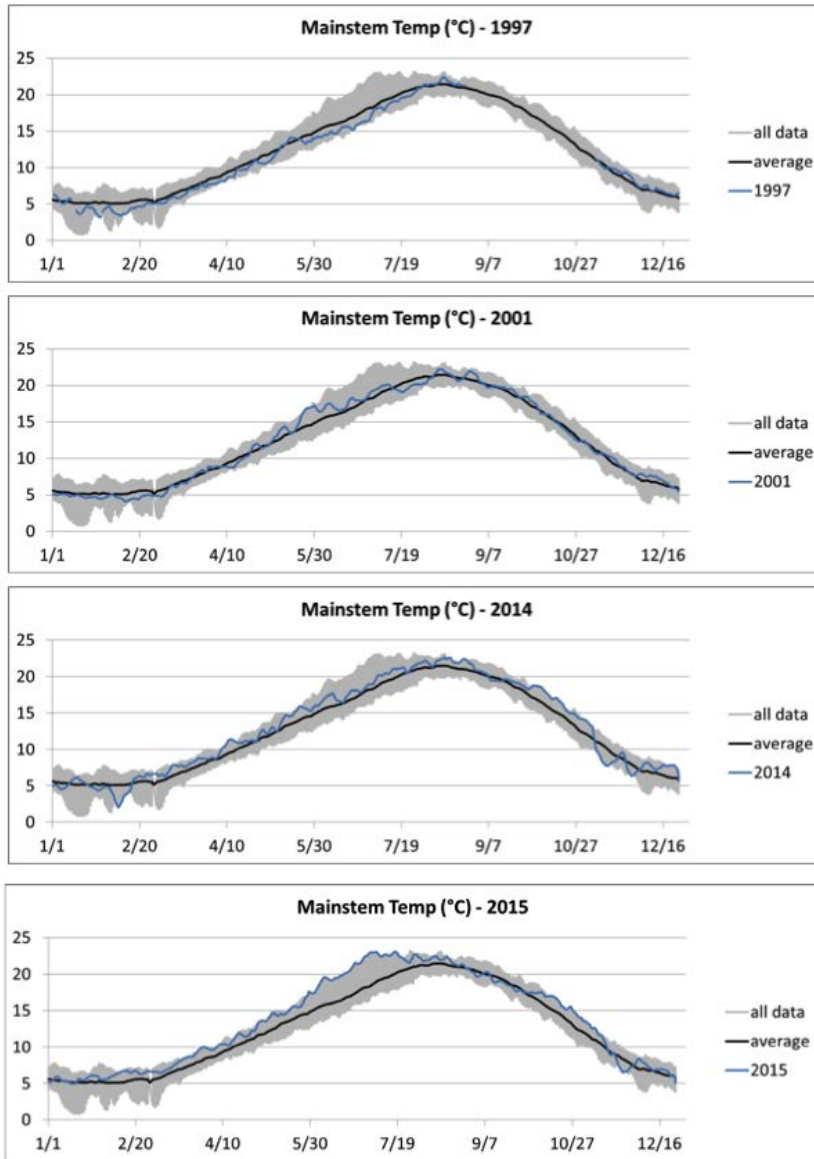


Figure 7. Daily average water temperatures at Beaver Army Terminal (BAT) for years with high flow (1997) and low flow (2001, 2014, 2015).

The total number of days where the daily average river temperature exceeded 19°C was 102 at Camas, WA and 105 at Beaver Army Terminal (Table 15). There were similar numbers of days where temperatures exceeded 21°C at both sites (66 and 68 days at Camas and Beaver Army Terminal, respectively). When these data are compared with previous observations from 2009-2014, the total number of days exceeding 19°C was 21–51 days more in 2015 than the last five years (Table 16).

Table 15. Number of days in 2015 where average daily water temperature was greater than 16°C, 19°C, and 21°C at Camas, WA (River Mile 122) and Beaver Army Terminal (BAT, River Mile 53).

Site	No. days, temperature >16°C	No. days, temperature >19°C	No. days, temperature >21°C
Camas, WA	150	102	66
BAT	152	105	68

Table 16. Number of days with daily average water temperatures greater than 19°C or 21°C in the Columbia River at Beaver Army Terminal (BAT, River Mile 53).

Temperature range	2009	2010	2012	2013	2014	2015
19-21°C	70	49	53	67	30	37
>21°C	11	2	2	14	42	68
Total >19°C	81	51	55	81	72	102

3.1.1.3 *Water quality parameters in the mainstem*

In 2015, winter turbidity and winter nitrate concentrations were lower relative to observations from previous years (see Sagar et al. 2014). Chlorophyll *a* concentrations were also lower than the long-term observations. Dissolved oxygen percent saturation levels were similar to other years, but maximum DO saturation occurred slightly earlier in the spring in 2015 compared to some other years (e.g., mid-April in 2015 compared to early June in 2012).

Seasonality in other parameters at the Camas, WA mooring such as chlorophyll *a* (which is representative of phytoplankton standing stocks and used as a metric of water quality) and dissolved oxygen saturation also differed in 2015 compared to other years (Figure 10). The peak in percent saturation of dissolved oxygen relative to the atmosphere occurred earlier in 2015 (mid-late April) compared to, for example, 2012 (June) or 2013 (Aug-September). In contrast, the peak in chlorophyll *a* in 2015 (mid-May) occurred somewhat later than in other years: late March (2010), mid-February (2012), late March (2013).

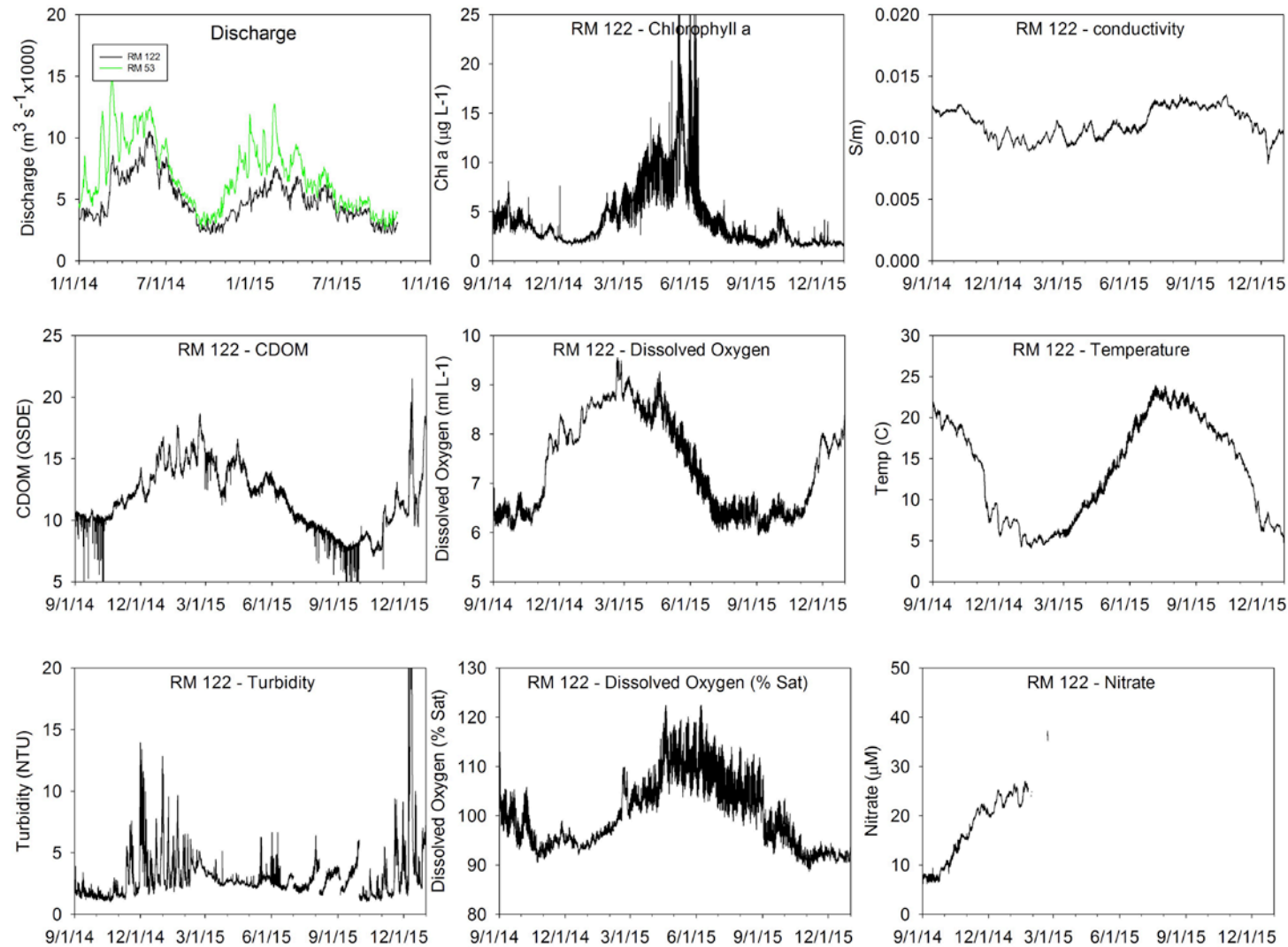


Figure 8. Parameters determined at the Camas, WA mooring (RM-122) in 2015, including chlorophyll *a*, conductivity, colored dissolved organic matter (CDOM), dissolved oxygen, temperature, turbidity, dissolved oxygen percent saturation relative to the atmosphere, and nitrate.

Peak chlorophyll *a* concentrations in 2015 at the Camas mooring were lower than in previous years, for example 2013 (Figure 9). The highest concentration of chlorophyll *a* was observed a little later in 2015 (~late May), which is typically when the spring freshet would dampen the signal due to flushing and dilution of standing stocks (Maier 2014; Maier et al. in review).

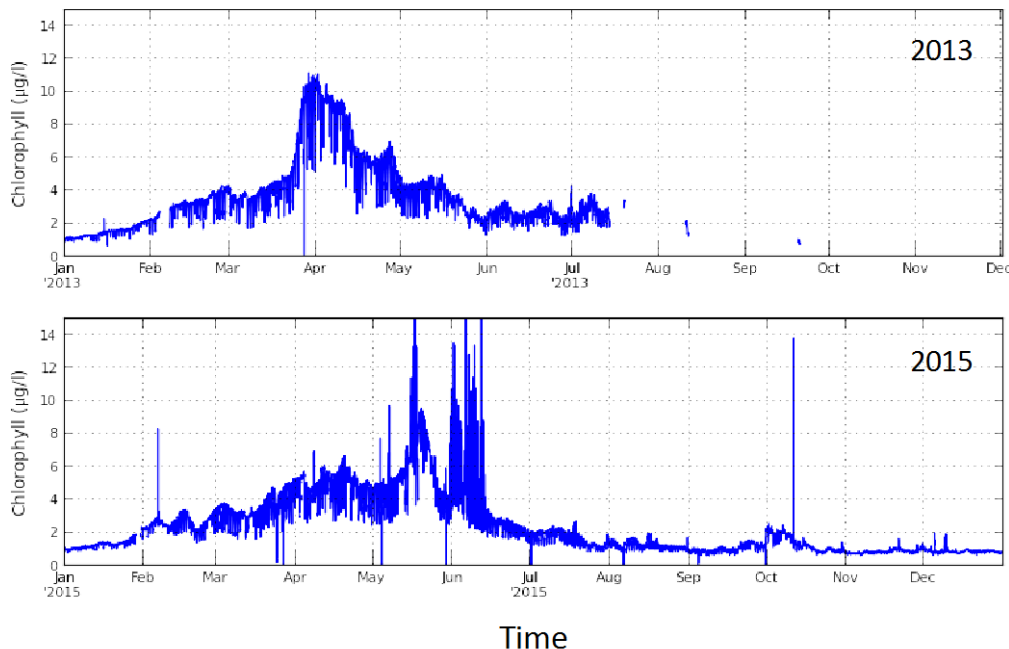


Figure 9. Hourly chlorophyll *a* concentrations observed at Camas, WA (RM 122) in 2013 (upper panel) compared to 2015 (lower panel). Values were higher in 2013 during the typical spring bloom period (March-May) compared to 2015. Peak chlorophyll concentrations in 2015 were observed not during the typical bloom period, but during the time window when the spring freshet typically occurs (May-June).

3.1.2 Discrete samples from the mainstem

Discrete water samples for the analysis of dissolved nutrients were collected at Beaver Army Terminal (RM 53) and at Camas, WA (RM 122) during servicing trips listed in Table 4.

3.1.2.1 *Camas, WA*

At Camas, nitrate concentrations in 2015 reached a peak in the winter, which was followed by a steady drawdown to a minimum of $<0.10 \text{ mg L}^{-1}$ in the early-to-mid summer. Seasonal variations in the concentration of both ortho-phosphate and ammonium were small, although an increase in the concentration of ammonium (NH_4^+) was observed in mid-August (Figure 10).

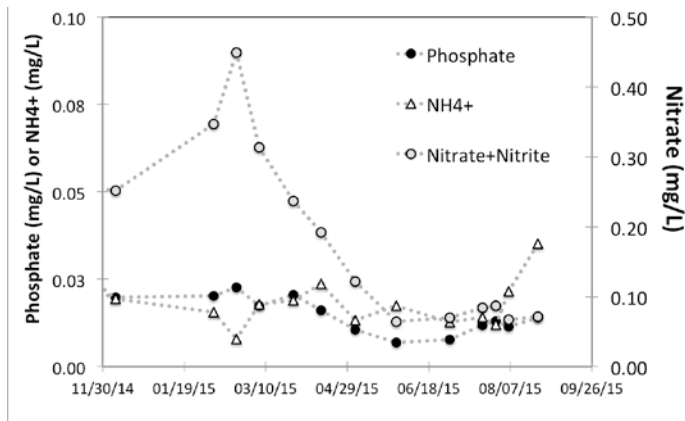


Figure 10. Time series of dissolved nutrient concentrations at the Port of Camas, WA.

3.1.2.2 Beaver Army Terminal (RM 53)

Similar to Camas, nitrate concentrations at Beaver Army Terminal in 2015 reached a peak in the winter, and declined through the spring and summer to a minimum of $\sim 0.10 \text{ mg L}^{-1}$ (Figure 11). Seasonal variations in the concentration of both ortho-phosphate and ammonium were small and an increase in the concentration of ammonium (NH_4^+) was observed in mid-August.

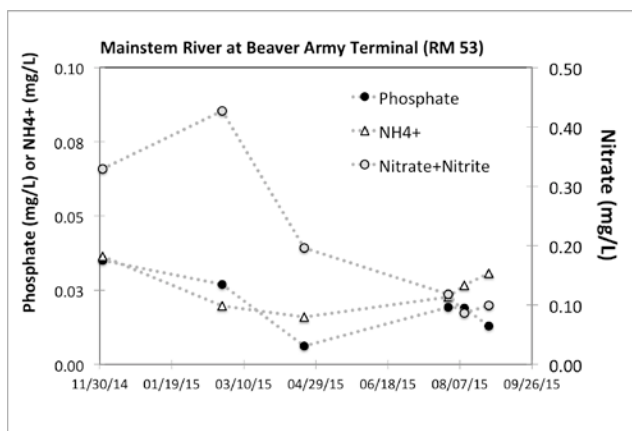


Figure 11. Concentrations of nitrate (as nitrate+nitrite), ortho-phosphate, and ammonium at Beaver Army Terminal in 2015.

3.2 Abiotic Site Conditions

3.2.1 Continuous Water Quality

YSI sondes were deployed at four trends sites in 2015: Ilwaco Slough, Whites Island, Campbell Slough, and Franz Lake Slough. The data were collected every hour during the deployments and daily averages for the following parameters were computed: temperature, salinity or conductivity, pH, and dissolved oxygen (reported as percent saturation). In each of the plots, the daily average for each parameter is shown for all available data to place data from 2015 into context. Note that the deployment period in 2015 was extended into August, and therefore the August long-term averages are more similar to 2015 values compared to the other months in the study.

3.2.1.1 *Ilwaco Slough*

At Ilwaco (Reach A), temperatures were close to the long-term average (Figure 12). With the exception of high values at the beginning of the 2015 deployment and low values at the end of June, the percent saturation of oxygen was similar to other years as well. However, substantial differences in salinity and pH were observed at Ilwaco in 2015 compared to the long-term mean values. Salinity was considerably higher during the months of June and July in 2015—with larger short-term fluctuations—compared to the long-term mean; salinity is typically higher in August as river flow subsides, but the onset of high salinities was earlier than average. In addition, pH was higher during June and July of 2015 compared to the long-term mean.

3.2.1.2 *Whites Island*

At Whites Island (Reach C), average daily temperatures were greater in 2015 relative to the long-term mean in June and July (Figure 13). Similarly, conductivity was higher during June and July in 2015 at Whites Island compared to the long-term mean. pH was lower in 2015 at Whites Island compared to the long-term mean. The difference was substantial enough to influence the temporal trends in the long-term data. Finally, dissolved oxygen saturation was slightly higher than the long-term mean between late July and mid-August in 2015.

3.2.1.3 *Campbell Slough*

Data from Campbell Slough showed that the water was warmer in 2015 than the long-term average in early May, in June, and in July (Figure 14). Conductivity, pH, and oxygen saturation were elevated in June–July 2015 compared to the long-term mean.

3.2.1.4 *Franz Lake Slough*

Water temperatures at Franz Lake Slough were more than five degrees Celsius higher than the long-term daily average during late June and July (Figure 15). Daily average conductivity was lower than the long-term average, while pH was slightly higher during the same time period. The percent saturation of dissolved oxygen was higher in mid-July and in early August, but lower than the long-term mean in late June.

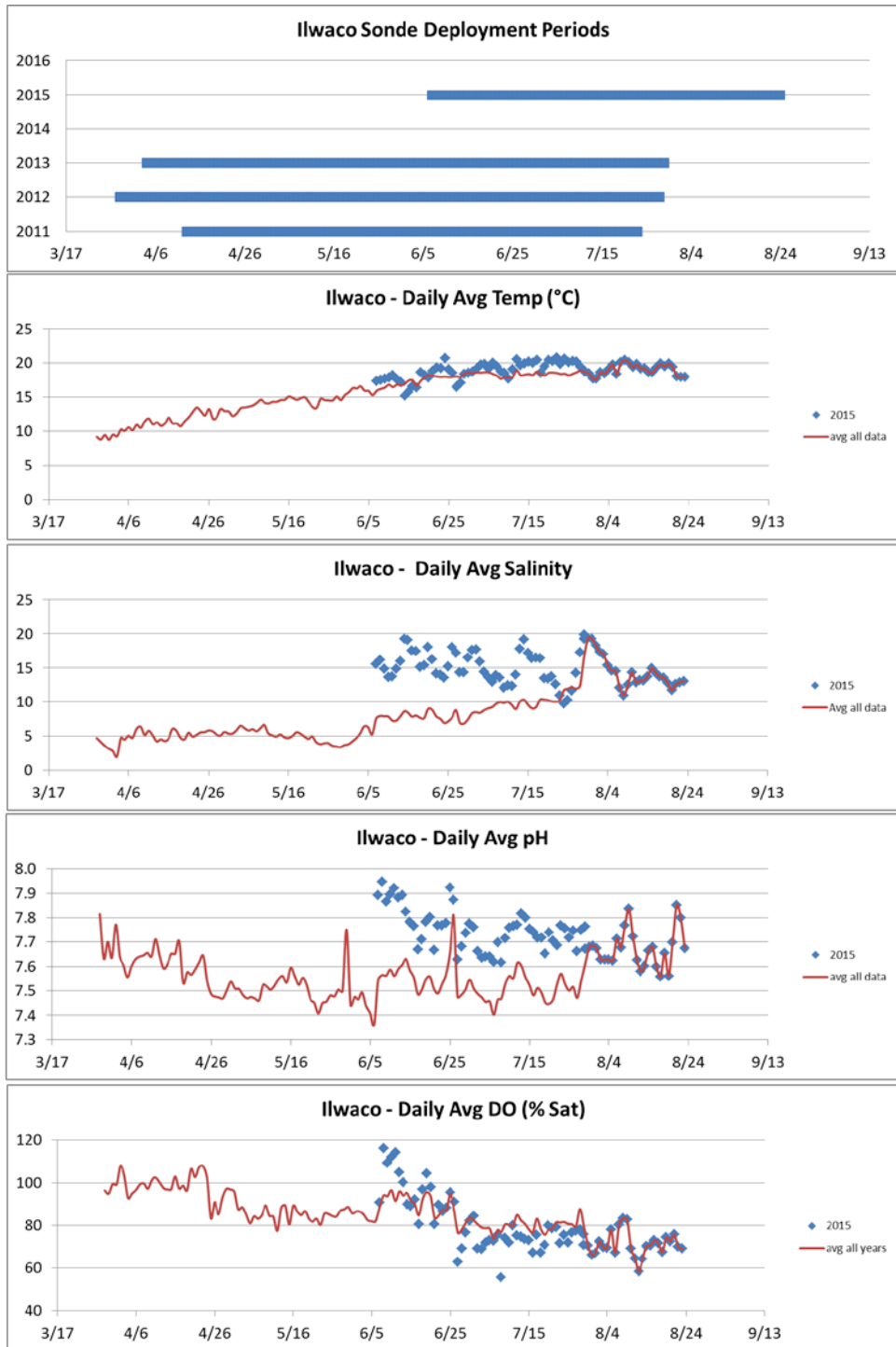


Figure 12. Time series showing continuous data from a YSI sonde deployed at Ilwaco Slough in Baker Bay (Reach A) in 2015 (blue diamonds) and the long-term average (red lines). Top panel shows the period over which the sonde was deployed during each year between 2011 and 2015.

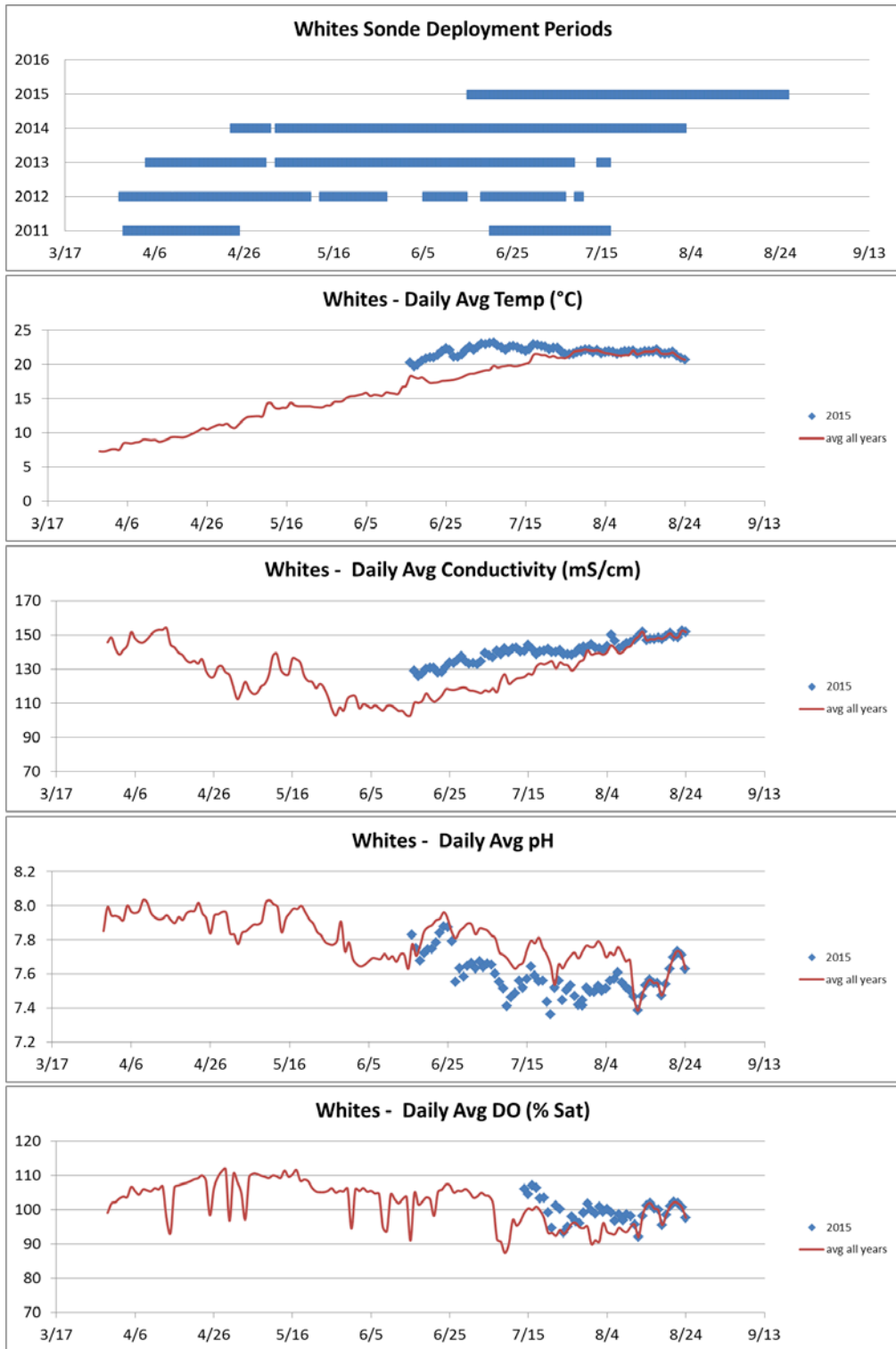


Figure 13. Time series showing continuous data from a YSI sonde deployed at Whites Island (Reach C) in 2015 (blue diamonds) and the long-term average (red lines). Top panel shows the period over which the sonde was deployed during each year between 2011 and 2015.

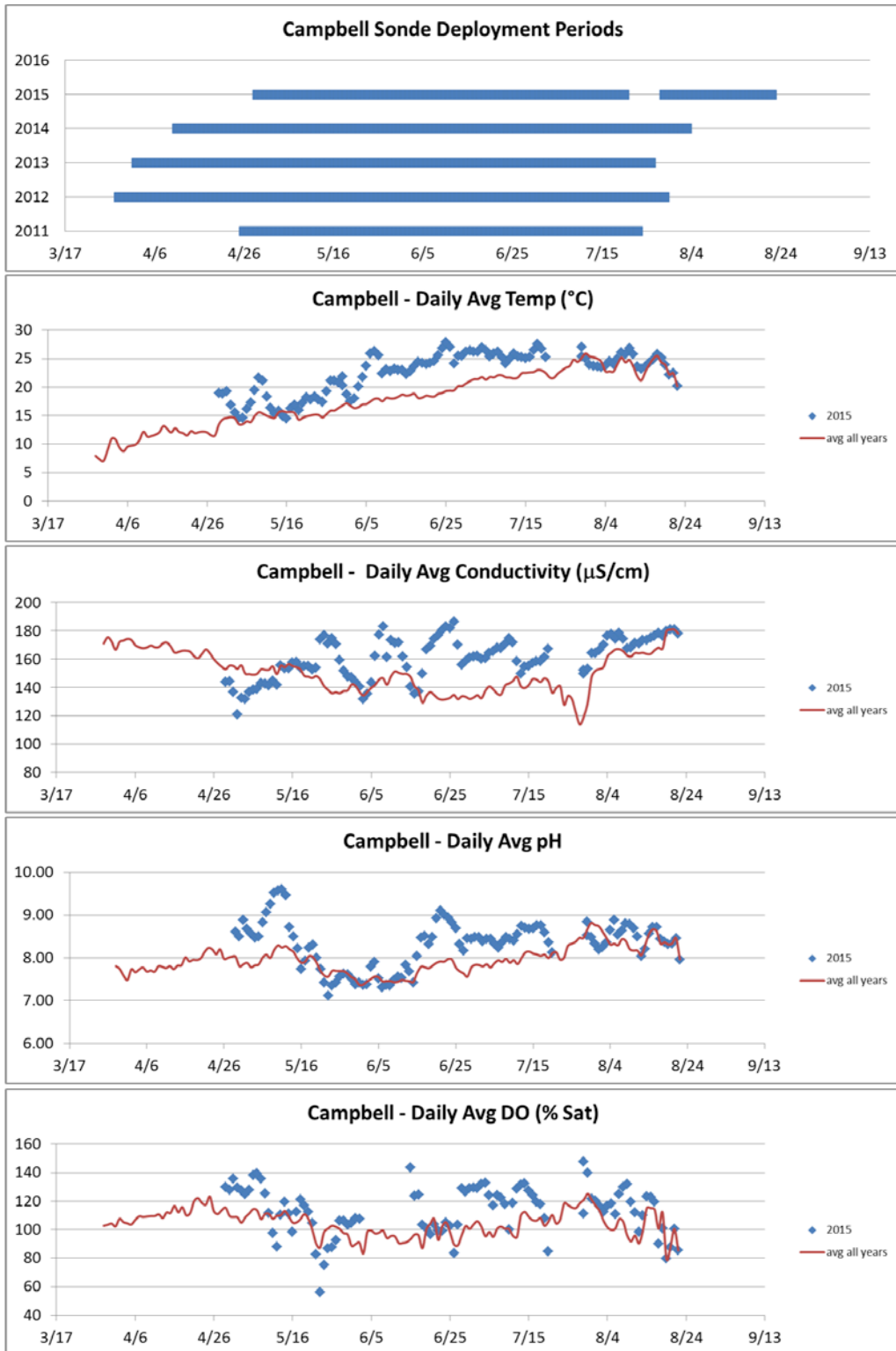


Figure 14. Time series showing continuous data from a YSI sonde deployed at Campbell Slough (Reach F) in 2015 (blue diamonds) and the long-term daily average (red lines). Top panel shows the period over which the sonde was deployed during each year between 2011 and 2015.

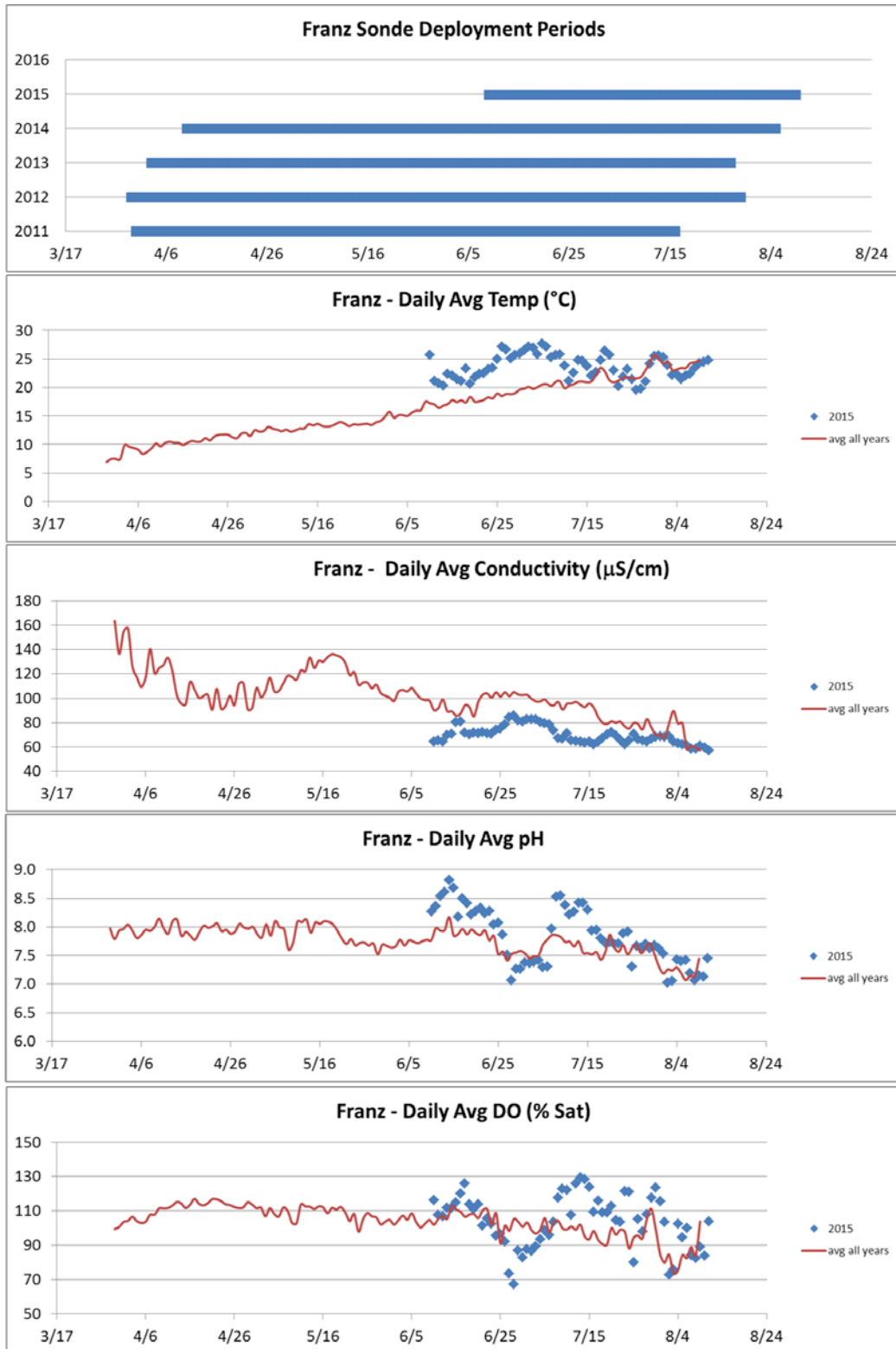


Figure 15. Time series showing continuous data from a YSI sonde deployed at Franz Lake Slough (Reach H) in 2015 (blue diamonds) and the long-term daily average (red lines). Top panel shows the period over which the sonde was deployed during each year between 2011 and 2015.

3.2.2 Nutrients

3.2.2.1 Dissolved inorganic nutrients (nitrate+nitrite)

Nitrate concentrations were relatively low at all sites in 2015 after the spring period (~April-early May; Figure 16). A clearer picture of how nitrate concentrations in 2015 compared to previous years is given in Figure 17 and Figure 18. The first shows the mean nitrate concentration at five trends sites calculated for each month of sampling for (a) 2011-2015, inclusive, and (b) 2015. These data show that nitrate values were generally lower than average at all of the sites below the confluence of the Willamette River (Figure 17). Figure 18 shows the same data plotted as the nitrate anomaly in 2015; that is, the 5-year average values determined each month at each site were subtracted from monthly averages determined in 2015. Plotted this way, it is clear that nitrate values at Whites Island and Campbell Slough were below the long-term average during the spring, whereas values at Franz Lake Slough were above average during the same time period.

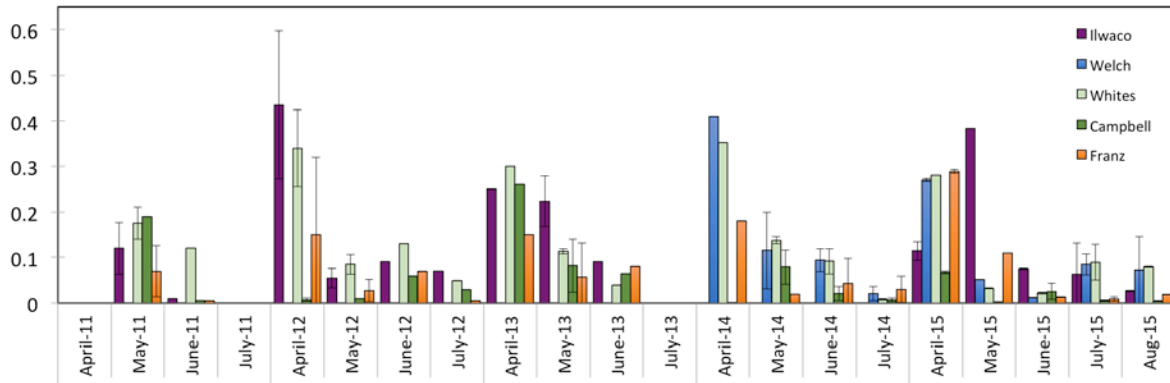


Figure 16. Nitrate concentrations (mg L^{-1}) at five trends sites (Ilwaco, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough) between 2011 and 2015. Error bars represent one standard deviation of the mean values determined for each sampling month.

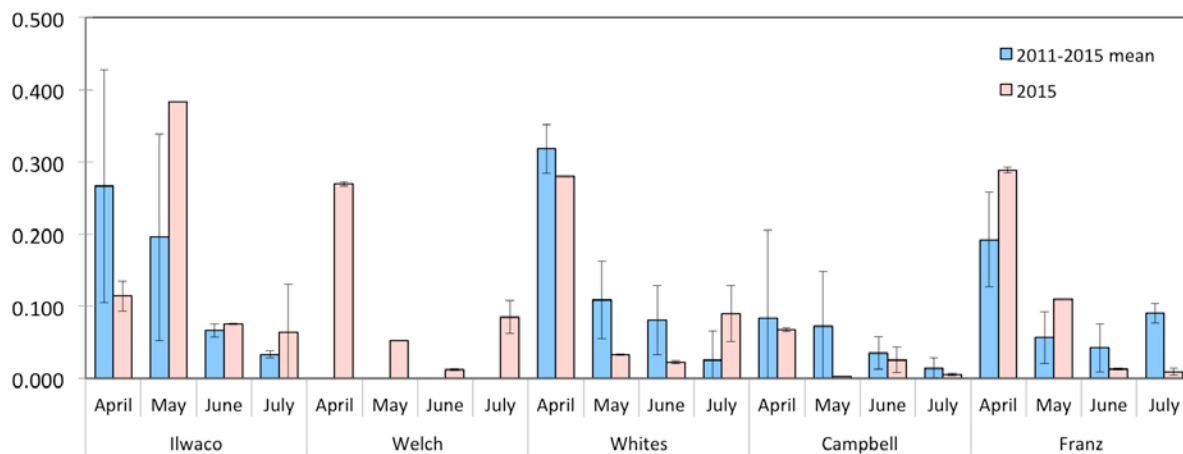


Figure 17. Mean monthly nitrate values (\pm one standard deviation) at each of five trends sites for two time periods: (1) 2011-2015, inclusive and (2) 2015 only.

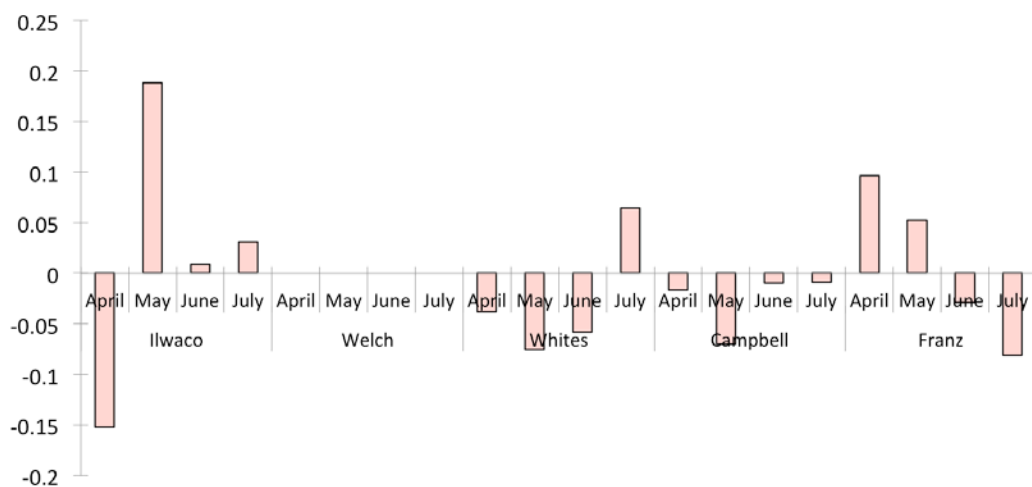


Figure 18. 2015 nitrate anomaly calculated relative to 5-year monthly mean values (2011-2015). Positive values indicate higher monthly average values in 2015 relative to the long-term mean, while negative numbers indicate lower monthly average values in 2015 relative to the long-term mean.

3.2.2.2 *Dissolved inorganic phosphorus (ortho-phosphate, or soluble reactive phosphorus)*

Ortho-phosphate concentrations were higher in 2015 compared to the other years comprised in the monitoring program (Figure 19), with the highest average monthly values occurring in May in Campbell Slough and consistently high values observed at Franz Lake Slough.

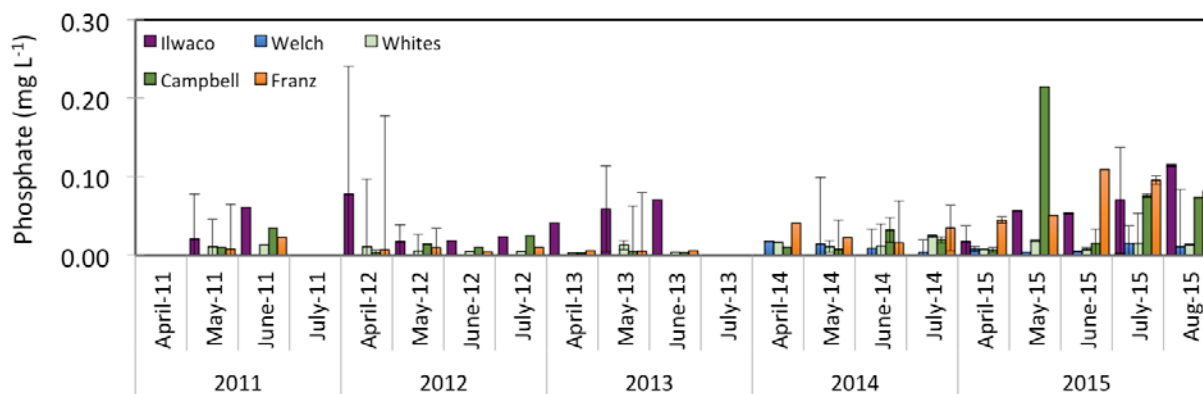


Figure 19. Average monthly concentrations of ortho-phosphate at five trends sites (Ilwaco, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). Error bars represent one standard deviation of the mean determined for each month of samples.

Average monthly ortho-phosphate concentrations in the year 2015 were higher relative to the five-year mean values at Campbell Slough in May and July and at Franz Lake Slough in June and July (Figure 20). Otherwise, phosphate concentrations were similar or lower than the five-year mean.

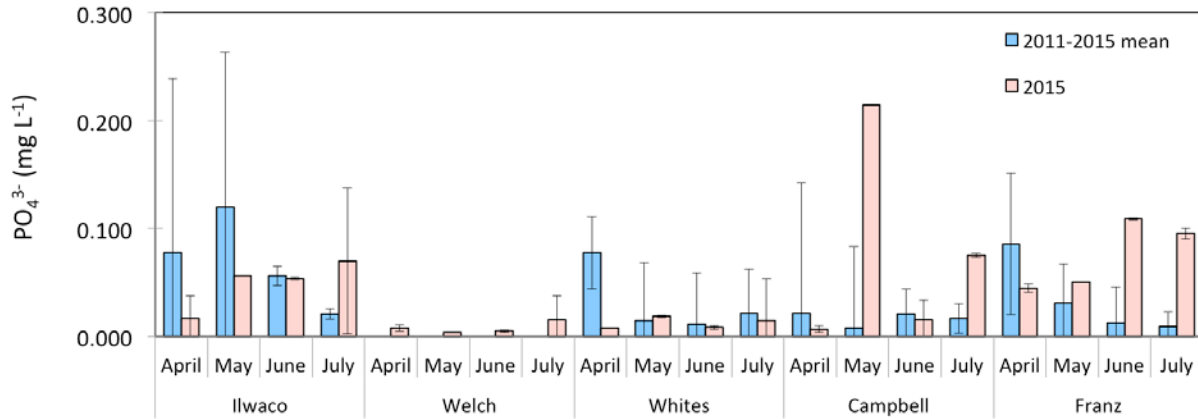


Figure 20. Average monthly concentrations of ortho-phosphate calculated over a five-year period (2011-2015) compared to monthly averages determined in 2015.

Anomalies in 2015 phosphate concentrations (i.e., average monthly values relative to the five-year mean) are shown in Figure 21. A large positive phosphate anomaly was observed at Campbell Slough in May while at Ilwaco and Whites Island spring concentrations were lower than the five-year mean.

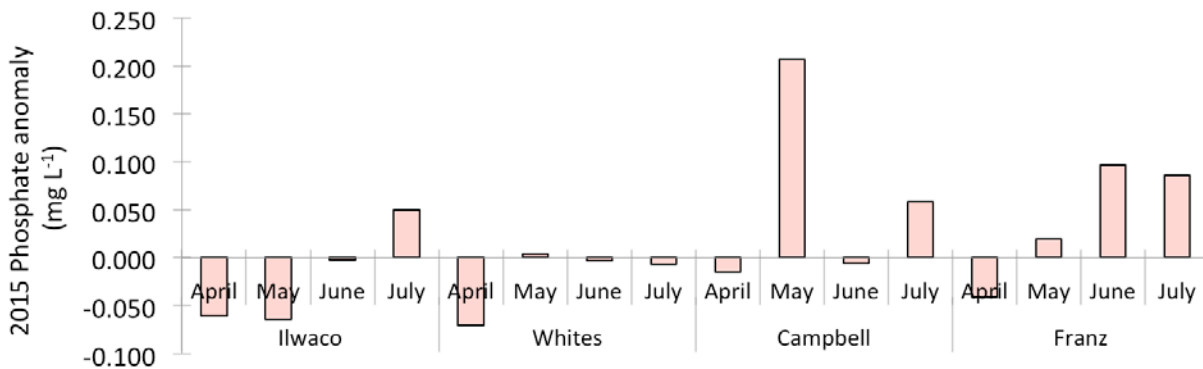


Figure 21. Anomaly in monthly average ortho-phosphate concentrations for the year 2015 compared to five-year monthly mean values.

3.2.2.3 *Dissolved organic nitrogen and phosphorus*

In 2015, dissolved organic nitrogen (DON) concentrations increased from spring to summer at Ilwaco, Welch Island, and Franz Lake while at Grays River and Whites Island there was no temporal trend in DON. Campbell Slough had high DON concentrations in May, July, and August (Figure 22).

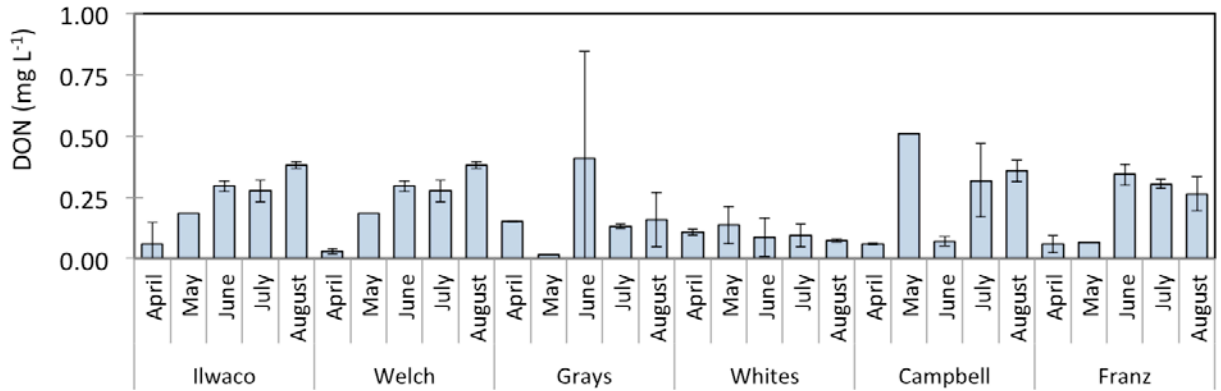


Figure 22. Monthly dissolved organic nitrogen concentrations (mg/L) at trends sites (Ilwaco, Welch Island, Grays River, Whites Island, Campbell Slough, and Franz Lake Slough) during the spring and summer months of 2015.

Dissolved organic phosphorus concentrations (determined as the difference of total dissolved phosphorus and soluble reactive phosphorus, or ortho-phosphate) was below the analytical limits of detection throughout the 2015 study period, with the exception of monthly average samples taken from Ilwaco in May, June, and July. Values of DOP for those dates were 0.052 mg L⁻¹ (May), 0.011 ±0.005 mg L⁻¹ (June), and 0.008 ±0.043 mg L⁻¹ (July).

3.2.2.4 *Particulate carbon, nitrogen, and phosphorus*

Particulate organic carbon concentrations were determined on the samples collected for determination of stable isotope ratios (C, N). The analysis provides information about the quantities of carbon and nitrogen for computation of C:N ratios of particulate matter, which can vary under different environmental conditions, such as under nutrient limitation of growth of primary producers. Seasonal increases in C:N values were largest at Campbell Slough and Franz Lake Slough, with the highest C:N values observed in the summer months (Figure 23).

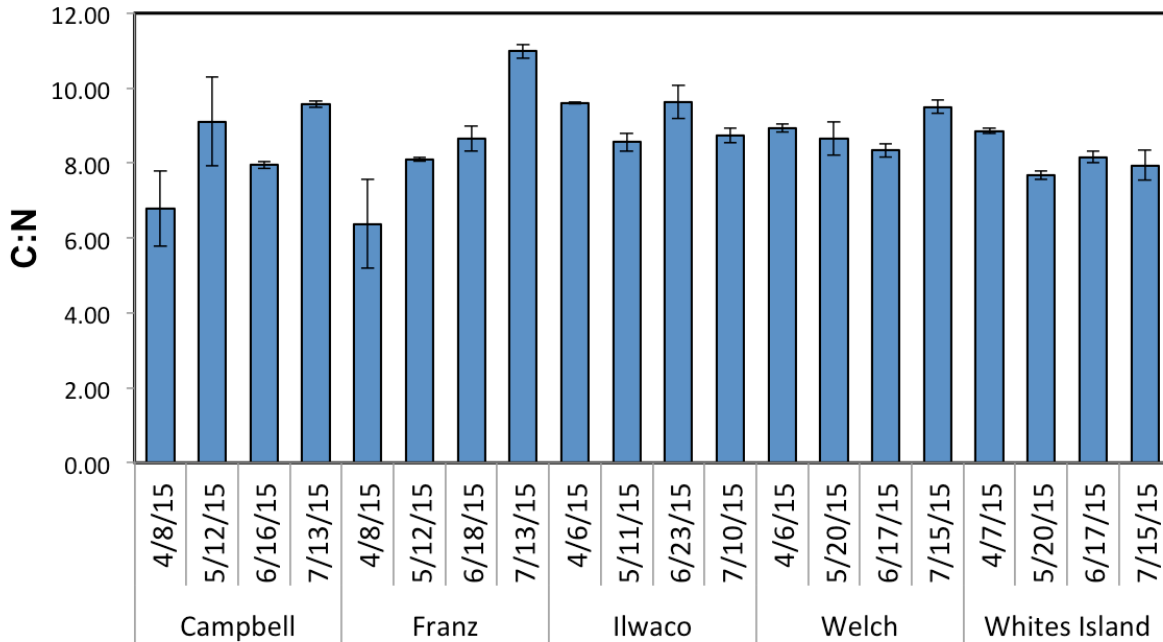


Figure 23. Carbon-to-nitrogen (C:N) ratios determined at five trends sites (Campbell Slough, Franz Lake Slough, Ilwaco channel, Welch Island, and Whites Island) in 2015. Error bars represent one standard deviation from the mean of three replicates.

Particulate nitrogen concentrations were consistently higher at Campbell Slough compared to the other trends sites, where the values showed only small differences through the spring and early summer (Figure 24). Aside from a high value in April at Welch Island, the sites in Reaches B and C (Welch Island, Grays River, and Whites Island) had relatively low concentrations of particulate nitrogen (Figure 24).

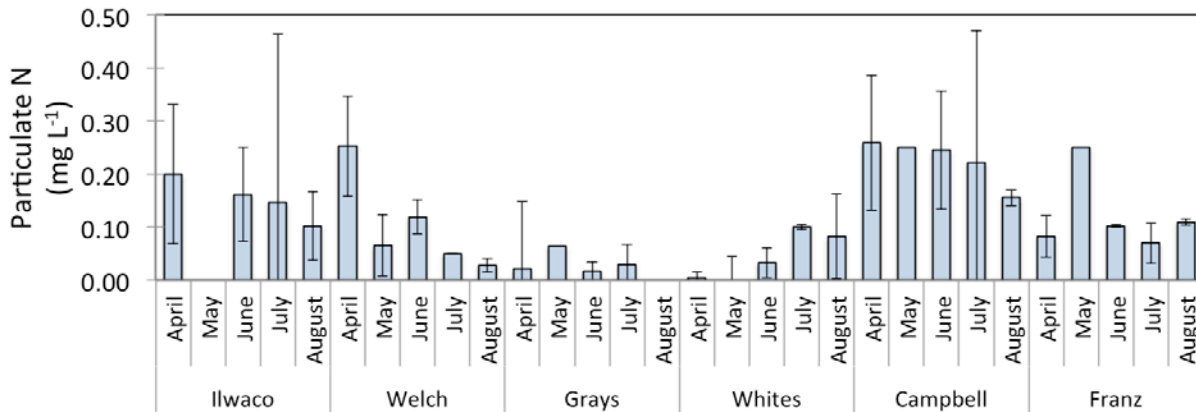


Figure 24. Average monthly concentration of particulate nitrogen (mg L⁻¹) at the trends sites in 2015. Error bars represent one standard deviation of the mean.

The concentrations of phosphorus associated with particles showed similar trends in 2015 as observed for particulate nitrogen, with the highest values occurring at Campbell Slough, Ilwaco, and Franz Lake

Slough (Figure 25). Concentrations were much lower at Welch Island, Grays River, and Whites Island, with the exception of high values observed in June at Grays River.

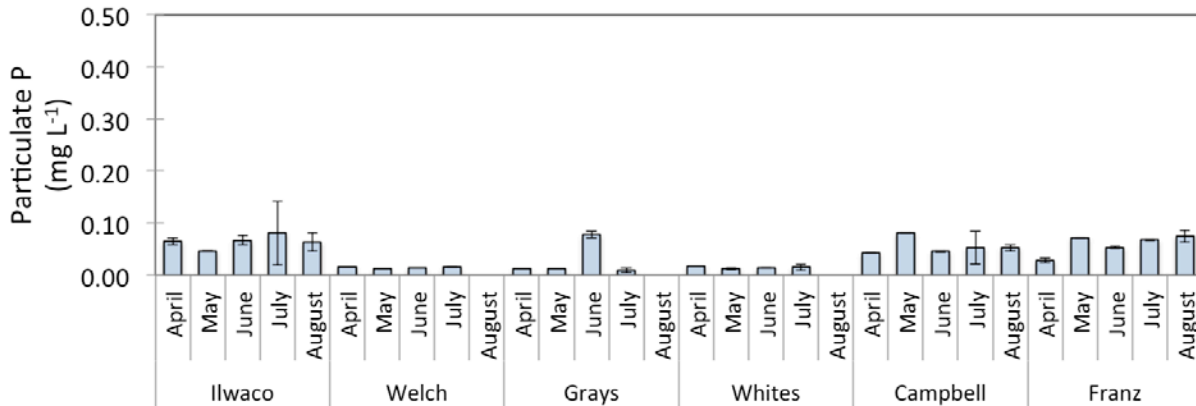


Figure 25. Average monthly concentrations of particulate phosphorus (mg L^{-1}) at the trends sites in 2015. Error bars represent one standard deviation of the monthly mean.

A comparison of nitrogen and phosphorus in particles versus dissolved fractions indicates whether nutrients are tied up in biomass (or inorganic particles) or if they may be available for consumption by primary producers. The ratios of particulate to dissolved nitrogen during 2015 were highest in April at Ilwaco, highest in April and June at Welch Island and Campbell Slough, highest in July and August at Whites Island, and highest in May at Franz Lake Slough (Figure 26).

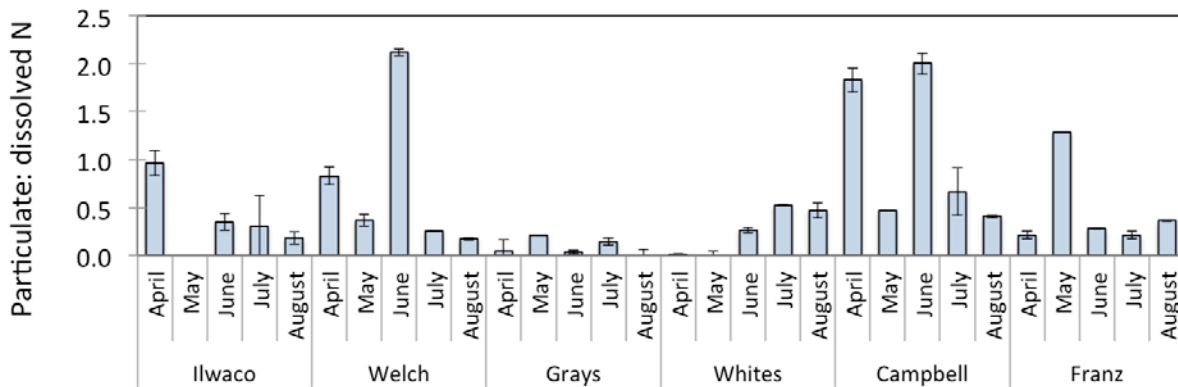


Figure 26. Ratio of particulate to dissolved nitrogen at the trends sites in 2015. Error bars represent one standard deviation of mean monthly values.

Ratios of particulate to dissolved phosphorus were much higher than the ratios of particulate to dissolved nitrogen. Concentrations of particulate phosphorus were more than five times greater than dissolved phosphorus in June at Grays River in Reach B and in April at Campbell Slough in Reach F. The site with the lowest ratios of particulate to dissolved P was Franz Lake Slough (Figure 27).

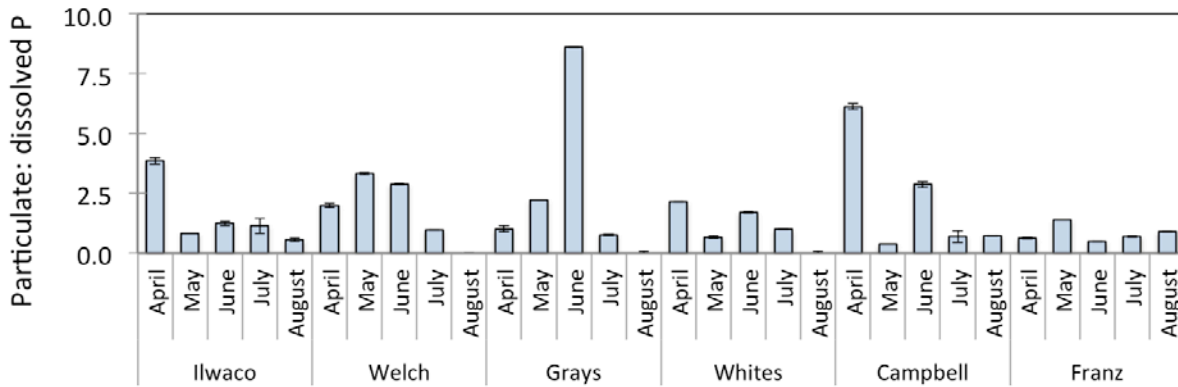


Figure 27. Ratios of particulate to dissolved phosphorus calculated from monthly mean values of particulate phosphorus and total dissolved phosphorus at the trends sites in 2015. Error bars represent one standard deviation of the monthly mean ratio values.

Ratios of nitrogen to phosphorus are often indicative of which nutrient may limit primary production; in general, cellular ratios of N:P tend toward values of 16:1 in the absence of nutrient limitation (Redfield et al. 1963). Ratios higher than 16 can be suggestive of phosphorus-limitation of primary production, while values less than 16 can be indicative of nitrogen limitation. The data from the trends sites suggest that phosphorus availability was limiting phytoplankton growth in April and June at Welch Island and in July at Whites Island (Figure 28). When N:P values fall below 16, this could indicate N limitation, but it could also reflect the fact that phosphorus readily adsorbs to particles and therefore the signal observed in the particulate measurements may be abiotic.

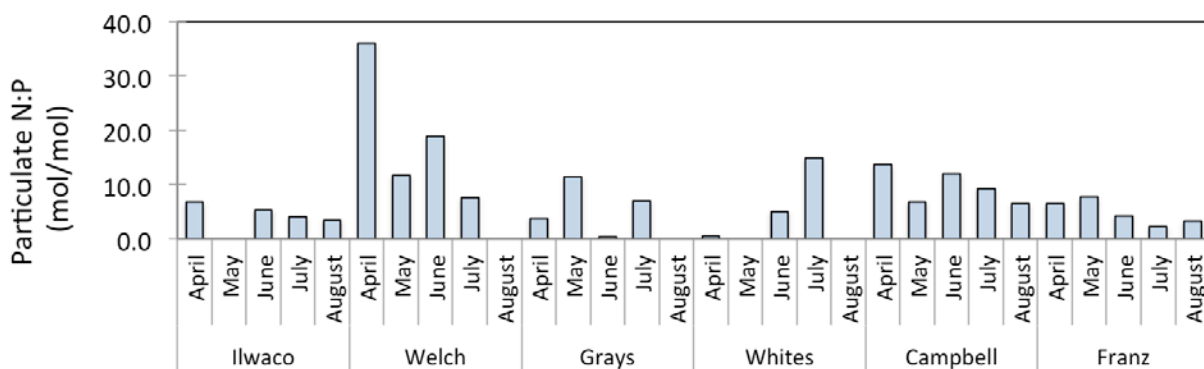


Figure 28. Molar ratio of particulate nitrogen and phosphorus at the trends sites in 2015.

When seasonally integrated values were examined, the concentration of particulate nitrogen was highest at Campbell Slough, where the value integrated over the study period (beginning of April to the end of

August) was $34.5 \text{ mg L}^{-1} \text{ season}^{-1}$ compared to much lower values at Whites Island ($6.9 \text{ mg L}^{-1} \text{ season}^{-1}$) or Grays River ($5.0 \text{ mg L}^{-1} \text{ season}^{-1}$; Figure 29). Although not as high as Campbell Slough, both Ilwaco and Franz Lake Slough had relatively high concentrations of particulate nitrogen during 2015 ($24.7 \text{ mg L}^{-1} \text{ season}^{-1}$ and $18.8 \text{ mg L}^{-1} \text{ season}^{-1}$, respectively).

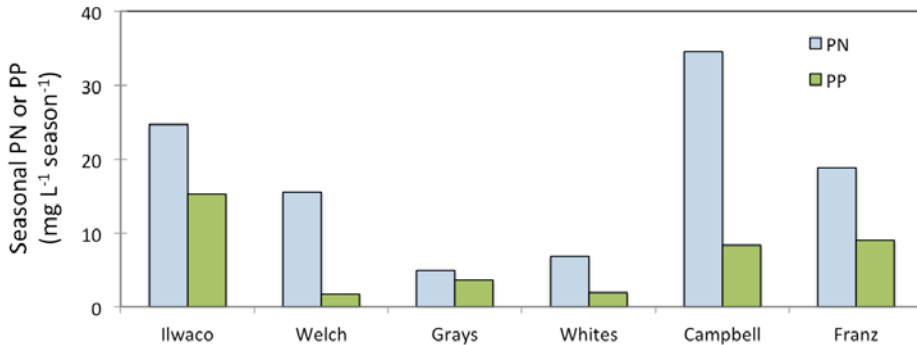


Figure 29. Seasonally integrated values of particulate nitrogen (PN) and particulate phosphorus (PP) at the trends sites in 2015. Integrated values were estimated by taking monthly mean values and multiplying by the number of days in the month and then summing the monthly values for April, May, June, July, and August.

3.3 Habitat Structure

3.3.1 Hydrology

Hydrologic patterns vary from year to year at all but the most tidal sites. In 2015, winter water levels were elevated by three flood events, but no spring freshet was discernable at any of the sites. Mean water levels over the year were similar between all sites, ranging from 1.4 m at Whites Island to 1.7 m at Franz Lake (Table 17). Hydrographs from all the years in which water surface elevation (WSE) was sampled at the trends sites, including the 2015 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 minimally affected by the spring freshet, but is elevated by winter storm events and extreme high tides. Additionally, low-water elevation measurements are truncated at the site because the elevation of the tidal channel is above that of extreme low water. Average tidal range at the site was 1.48 m in 2015 (Table 17).
- The Secret River site, at rkm 37, is also affected by winter storm events although minimally by the spring freshet. In 2015, the WSEs and the average tidal range at this site were greater than at the Ilwaco site (Table 17). The tide range was 2.15 m in part due to the lower elevation of the tidal channel where the sensor is located. The low-elevation marsh at the site is exposed only during low tide and conversely, the high-elevation marsh is inundated only during high tide each day.
- The Welch Island site, located at rkm 53, is predominantly tidal; however, slightly elevated WSE was detectable during prolonged spring freshet conditions in 2012 and 2014. Winter storms also drive higher water levels at this site, particularly elevating the low tide levels. Tidal range is greatest at this site at 2.24 m (Table 17), primarily due to the depth of the tidal channel below the

extreme low water level. Sensor failure precluded measurements at the site from April to July 2015.

- The hydrologic pattern at the Whites Island site (rkm 72) exemplifies the mix of hydrologic drivers in the lower river. The average tidal range was 1.72 m in 2015 (Table 17), while elevated water levels also occurred during winter storm events. In previous years, elevated water levels were also observed during the spring freshet.
- The Cunningham Lake and Campbell Slough sites, at rkm 145 and 149, respectively, have similar hydrologic patterns except that Cunningham Lake site has a slightly greater tidal range and slightly lower WSE during flood events (Table 17). The sensor at Cunningham Lake is in the very upper reach of the channel and is therefore elevated above the lowest water levels. The Campbell Slough sensor is in a deeper channel, however a weir located at the mouth of the slough limits drainage. In most years, the primary hydrologic driver at both sites is the spring freshet, although in 2013 to 2015 winter storms also increased the WSE at these sites. In 2013, 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. In 2014, the peak water levels in January to March exceeded those during the spring freshet between April and June and in 2015 three winter peaks far exceeded the indiscernible spring freshet.
- The Franz Lake site, at rkm 221, has a small tidal signal (on average 0.24 m; Table 17) which is difficult to distinguish from diurnal variation from dam operations (Jay et al. 2015). Low water was maintained at the site by a beaver dam in the fall that washed out sometime in the winter and was rebuilt in the following summer. In most years, the winter and spring high WSEs are both discernable, however, the spring levels are usually considerably higher than those in winter. In 2015, the site was inundated above the marsh surface most of the winter and early spring then only periodically in May after that.

Table 17. Water surface elevation (WSE) metrics calculated at each site for the sensor deployment period ending in 2015. All metrics are in meters, relative to the Columbia River Datum (CRD). MWL = mean water level; MLLW = mean lower low water; MHHW = mean higher high water.

Site	rkm	MWL	MLLW	MHHW	Avg Tidal Range (m)	Max WSE	Date of Max WSE	Period of Record
Ilwaco Slough ¹	6	1.53	1.04	2.47	1.43	3.40	10/25/2014	Aug 2014 - Jul 2015
Secret River	37	1.52	0.52	2.68	2.15	3.54	12/20/2014	Aug 2014 - Jul 2015
Welch Island	53	1.47	0.35	2.59	2.24	3.41	12/21/2014	Aug 2014 - Apr 2015
Whites Island	72	1.41	0.66	2.38	1.72	3.32	12/21/2014	Aug 2014 - Jul 2015
Cunningham Lake	145	1.46	1.24	1.72	0.49	3.09	12/22/2014	Aug 2014 - Jul 2015
Campbell Slough	149	1.56	1.40	1.80	0.41	3.22	2/9/2015	Aug 2014 - Jul 2015
Franz Lake	221	1.71	1.60	1.84	0.24	3.57	2/12/2015	Aug 2014 - Jul 2015

¹Data between 11 Nov 2014 and 12 Feb 2015 were not used in these calculations due to sensor movement during that time.

It is evident that the frequency of inundation at each site is dependent on the elevation, the position along the tidal and riverine gradient, and the seasonal and annual hydrologic conditions. The frequency of inundation at the average elevation of the sites in 2015 is shown in Figure 30. At all sites in 2015, the percent of time that the high marshes were inundated was greater over the whole year, ranging from 25-41%, than it was during the growing season, driven by higher winter water levels. Inundation at the Secret River low marsh site had the highest inundation frequency of all the sites monitored in 2015 due to its position at the lower end of the tidal-wetland elevation range in the lower river. In 2015, the three up-river sites had very low growing-season inundation frequencies of 3-11%; in contrast to frequencies of about 50% the previous year. The lower river high-marsh sites had frequencies similar to previous years, ranging from 15-25% during the growing season.

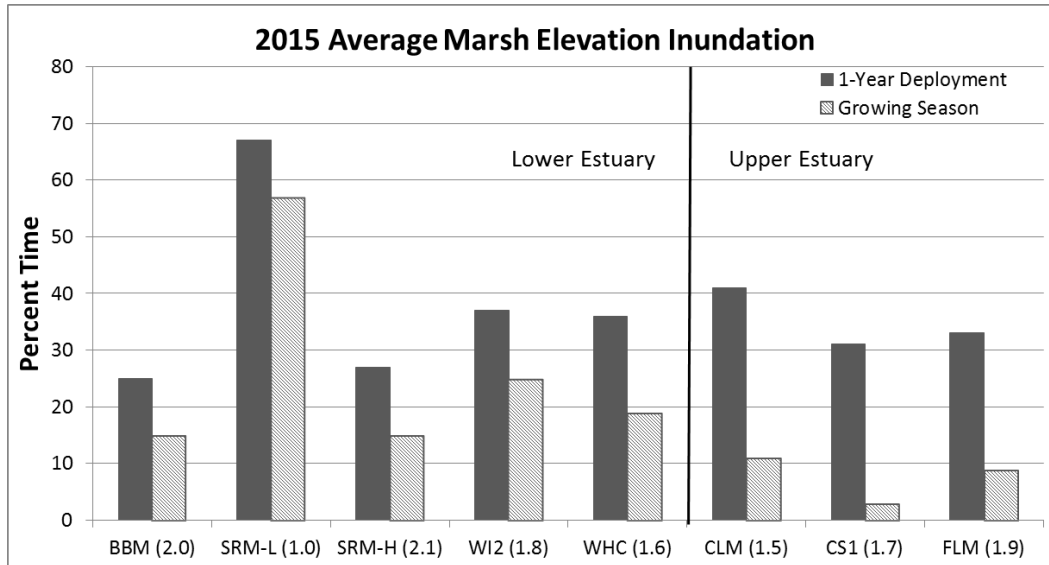


Figure 30. Inundation frequency at the seven trends sites in 2015; one-year deployment is from July 2014 to July 2015 and the growing season is from April to October. Site codes are defined in Table 1. Sites are ordered from left to right starting at the mouth. Average site elevations are given in parentheses after the site codes. All sites are high marshes with the exception of Secret River Low Marsh (SRM-L), where the highest inundation occurred.

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 over time. In most years, cumulative inundation increases up-estuary, with the highest inundation at the Franz Lake site. However, in 2015 the highest inundation occurred at the Secret River site (SRM; Figure 31). Inundation is consistently higher at Secret River than the other lower estuary sites. The reason for this is not certain, but could have to do with consistently higher tides at Secret River than at Whites Island, and the fact that the site is more affected by tributary run-off than the Ilwaco site. The lowest inundation in 2015 occurred at Cunningham Lake, where tidal influence is less than at the estuary sites in the lower reaches and fluvial influence is less than at the up-river sites.

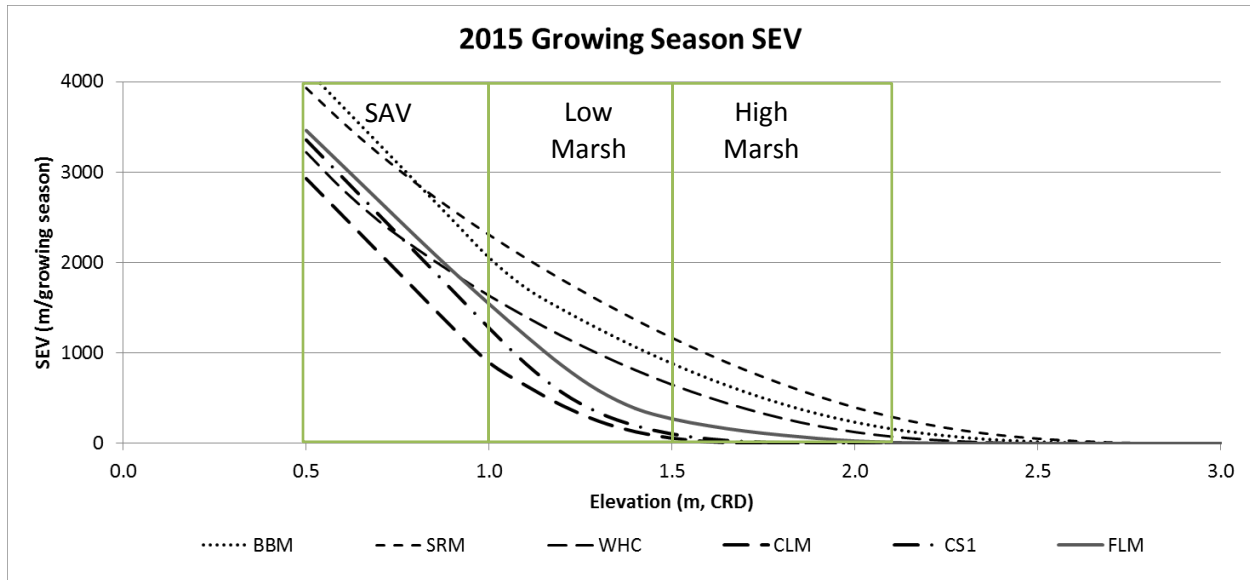


Figure 31. 2015 growing season sum exceedance values (SEVs) for the trends monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the lower river; the vertical lines represent the approximate boundaries between vegetation communities at the trends sites.

Inter-annual variation in inundation patterns is much greater at the upper estuary sites (Figure 32), 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 discernible in the SEV at the average marsh elevation, whereas the up-estuary sites have large differences in the SEV between years. At the up-river sites, the SEV in 2015 was very low, with no inundation at the average site elevations during the growing season. Inundation was similar to other years at the lower estuary sites.

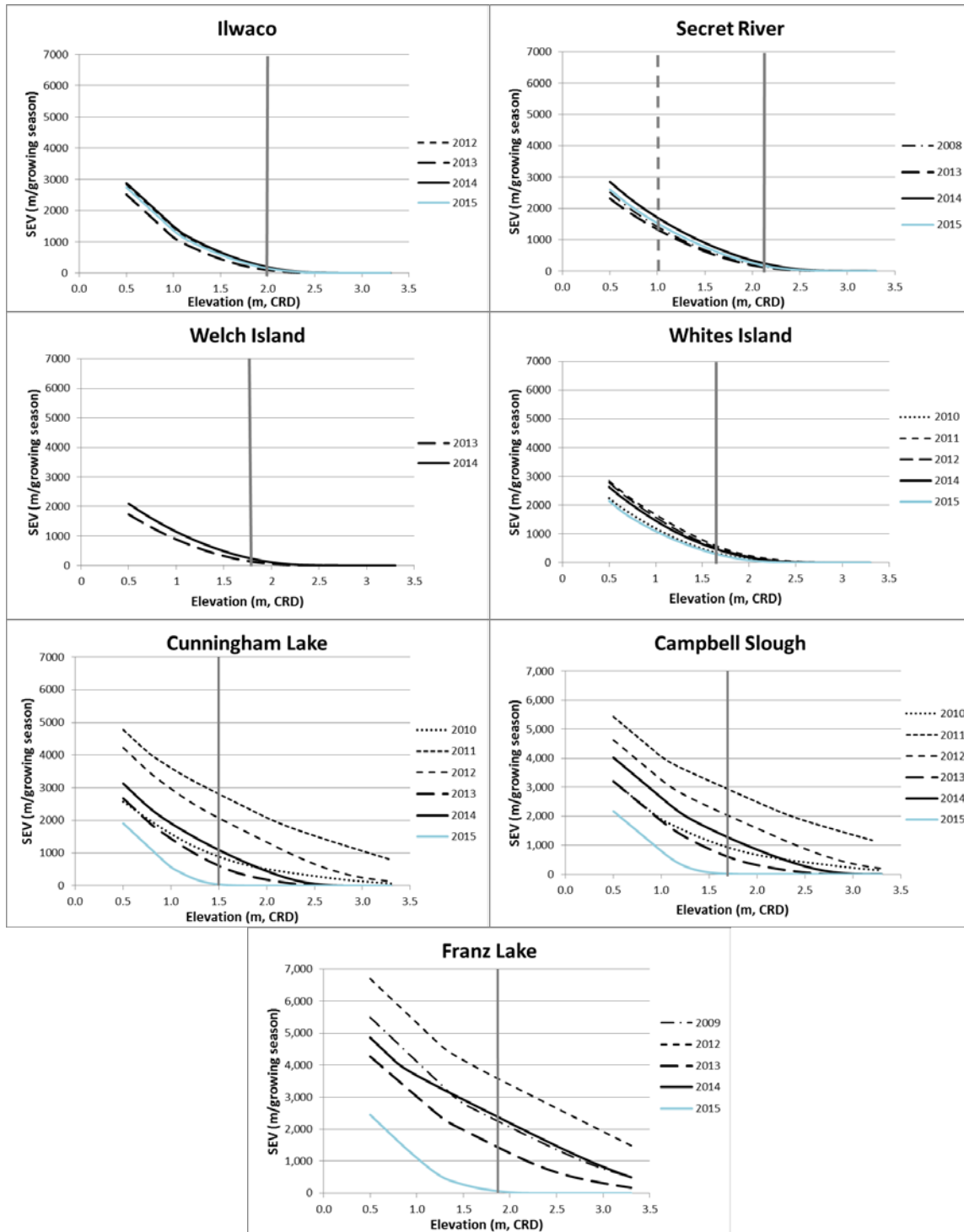


Figure 32. Annual growing season sum exceedance values (SEVs) for the trends monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the lower river. The vertical line represents the average elevation at each site, with the elevation of the Secret River low marsh represented by an additional dashed line. Sites are ordered with the site nearest the Columbia River mouth in the upper left panel.

3.3.2 Sediment Accretion Rates

Average sediment accretion at the trends sites ranges from 0.3 cm to 2.0 cm per year (Table 18). The Secret River low marsh stakes, however, have consistently measured erosion every year, averaging -1.6 cm per year. The site with the least variability over time is the Welch Island site (0.7 ± 0.1 cm), while the greatest variation between years was observed at the Campbell Slough and Franz Lake sites. The Whites Island mid-elevation stake measurements have been steadily accreting increasing amounts over the three year period and have the highest average accretion of 2.0 cm per year.

The annual sediment stake data measured in 2015 generally followed the trends observed in previous years with a few notable exceptions. The Secret River high marsh measurements indicated over 2 cm of accretion, however the stakes were no longer level and while measurements were adjusted to maintain a level plane, it is possible that both stakes may have “slumped”. A new set was deployed at this location to replace the potentially erroneous stakes. The Cunningham Lake and Campbell Slough sediment stake data indicate that erosion has occurred at these sites; however, in both cases, the erosion is likely caused by animal activity, with a beaver trail just outside the Cunningham Lake stakes and cow prints observed near the Campbell Slough stakes. New stakes were deployed at these sites to avoid the beaver trail and to measure accretion at additional elevations within the site.

Table 18. Sediment accretion rates at the trends sites between 2008 and 2015. See Table 1 for site code definitions; SRM-C is a set of stakes on the channel bank at the Secret River high marsh site and WHC-M and WHC-H represent mid-elevation and high-elevation marsh locations, respectively.

Site Code:	BBM	SRM-L	SRM-H	SRM-C	WI2	WHC-M	WHC-H	CLM	CS1	FLM
Elevation (m, CRD):	1.82	1.01	2.09	2.16	1.66	1.34	1.89	1.53	1.56	1.88
Year	Annual Rate (cm)									
08-09	ND ¹	ND	ND	0.2	ND	ND	-1.2	ND	ND	0.5
09-10	ND	ND	ND	2.8	ND	ND	1.0	1.9	0.4	ND
10-11	1.7	ND	ND	0.9	ND	ND	0.1	1.6	1.7	3.0
11-12	0.1	-2	ND	ND	ND	ND	0.9	1.4	0.9	-0.4
12-13	0.6	-1.7	1.1	1.4	0.8	1.2	0.2	1.3	0.2	3.0
13-14	0.3	-1.6	0.6	1.0	0.6	2.3	0.8	0.5	1.5	0.7
14-15	1.0	-0.9	2.4	0.7	0.7	2.7	0.0	-0.5	-2.4	1.2
Average	0.5	-1.6	1.4	1.2	0.7	2.0	0.3	1.0	0.4	1.3
Std Dev	0.4	0.5	0.9	0.9	0.1	0.8	0.8	0.9	1.5	1.4

¹ ND No data.

3.3.3 Salinity

Salinity was measured at the Ilwaco site between July 2011 and April 2015 (Figure 33). The range was between 0.1 and 30.7 parts per thousand (ppt). Only 10 records occurred above 25.5, and 30 ppt was exceeded for just one hour during January 2014. Just over 67% of the measurements were between 2 and 10 ppt. High daily variability occurred primarily during the spring of 2012 and 2013 and in the late summer of 2014. Average annual salinities during the growing seasons (April 22 to October 12) of 2012 to 2014 are between 7.6 and 8.1 ppt. Unfortunately, sensor failure prohibited salinity measurements being taken in summer 2015 when freshwater input was likely low. Measurements in April 2015 indicated that the salinity was higher in that month on average (13.9 ppt) than the average measured in the previous three years (11.6 ppt in 2012; 12.6 ppt in 2013; 13.3 ppt in 2014). See section 3.2.1.1 for measurements of salinity during summer 2015.

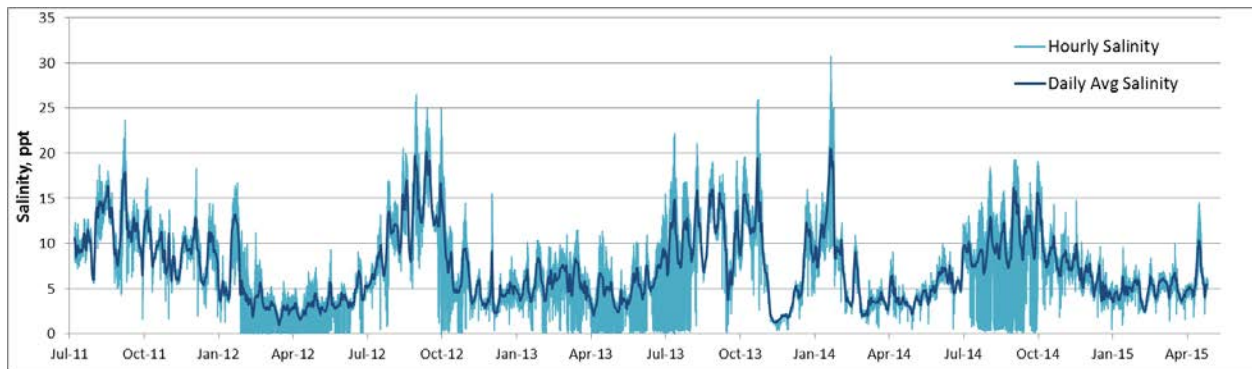


Figure 33. Salinity measurements logged at the Ilwaco site from July 2011 through April 2015.

3.3.4 Vegetation Species Assemblage

A summary of the number and aerial cover of native and non-native species at the trends sites in 2015 is provided in Table 19. The highest number of species overall and the highest number of native species were found at Welch Island. The lowest number of species occurred at Ilwaco Slough and at Cunningham Lake; however, there were more non-native species at Cunningham Lake. Many sites had aerial cover greater than 100 percent in 2015 (Table 19). The highest cover was measured at Welch Island and the lowest total cover in 2015 was measured at the Ilwaco Slough site.

Table 19. Species richness and areal cover of native and non-native species at the 2015 monitoring sites.

Site	Rkm	# Native Species	Native Species % Cover ¹	# Non-native Species	Non-native Species % Cover	Total # Species	Total % Cover
Ilwaco Slough	6	17	67.9	3	10.0	20	77.9
Secret River - High	37	24	77.1	10	37.8	34	114.9
Secret River - Low	37	23	79.9	4	4.9	27	84.8
Welch Island	53	36	103.0	11	17.3	47	120.2
Whites Island	72	31	40.0	13	65.9	44	105.9
Campbell Slough	145	22	62.8	12	32.4	34	95.2
Cunningham Lake	149	14	57.7	6	53.7	20	111.4
Franz Lake	221	22	83.3	7	16.9	29	100.2

¹Cover values include only live herbaceous vegetation and woody species that are not solely overhead; overhanging tree cover is not included. Cover values are not relative but absolute and therefore can exceed 100% where there is more than one vertical layer in the plant community.

The aerial cover of the six most common species found at the trends sites are summarized in Table 20. During the monitoring period prior to 2015, the cover of *Carex lyngbyei* and *Phalaris arundinacea* were very similar. In 2015 the latter was slightly higher than *C. lyngbyei*, primarily due to an increase in *P. arundinacea* cover. Overall, there was an increase in *Sagittaria latifolia* and in *Polygonum amphibium* in 2015 compared to the average of previous years.

Table 20. Common vegetation species found at the seven trends sites sampled for habitat structure. Average percent cover estuary-wide was calculated by taking the average between years (through 2014) at each site then averaging all sites. The 2015 data was averaged by site then all sites averaged.

Species Code	Scientific Name	Common Name	Wetland Status	Category	Native	All Years Avg. % Cover (SD)	2015 Avg. % Cover (SD)
CALY	<i>Carex lyngbyei</i>	Lyngby sedge	OBL	Sedge	yes	20.1 (25.4)	19.5 (24.0)
PHAR	<i>Phalaris arundinacea</i>	Reed canarygrass	FACW	Grass	no	19.6 (16.4)	23.2 (21.3)
ELPA	<i>Eleocharis palustris</i>	Common spikerush	OBL	Sedge	yes	6.97 (8.59)	7.71 (10.3)
SALA	<i>Sagittaria latifolia</i>	Wapato	OBL	Herb	yes	3.34 (2.82)	9.80 (9.59)
OESA	<i>Oenanthe sarmentosa</i>	Water parsley	OBL	Herb	yes	3.29 (5.62)	1.04 (1.58)
POAM	<i>Polygonum amphibium</i>	Water ladysthumb, Water smartweed	OBL	Herb	yes	2.03 (5.71)	5.12 (14.5)

Reed canarygrass (*P. arundinacea*) is present at six of the seven trends sites in the lower river (Table 21), with the extent of coverage varying depending on location and annual environmental conditions. The lowest cover was observed at the Welch Island site, where cover has remained less than 10% since 2012. The highest coverage has consistently been observed at the Whites Island site, where cover has been greater than 40% since the site was first monitored in 2009. Moderate cover between 20 and 35% was observed at the Secret River site since 2012; an increase from the 10% cover observed in 2008. The upper estuary sites have had variable cover over the monitoring period. *P. arundinacea* cover at the Cunningham Lake and Campbell Slough sites ranged from 15 to 57%, with lowest cover observed during

years of high inundation and episodes of disturbance from cow grazing. The Franz Lake site had consistently moderate cover of 33 to 34% in 2008 and 2009 followed by a decrease, starting in 2011 when high inundation favored the growth of a competitive native species, *Polygonum amphibium*, which has persisted until 2015 (Figure 34).

Table 21. Average percent cover of *Phalaris arundinacea* at the trends sites between 2005 and 2014.

Site	Rkm	Average Percent Cover <i>Phalaris arundinacea</i>										
		2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
Ilwaco	6	ND	ND	ND	ND	ND	ND	ND	0.0	0.0	0.0	0.0
Secret River– Low	37	ND	ND	ND	5.3	ND	ND	ND	0.0	0.0	0.0	0.0
Secret River– High	37	ND	ND	ND	10.4	ND	ND	ND	19.8	35.5	24.3	26.3
Welch Island	53	ND	ND	ND	ND	ND	ND	ND	5.9	9.8	8.3	8.3
Whites Island	72	ND	ND	ND	ND	43.0	47.8	56.8	42.0	56.5	48.0	53.9
Cunningham Lake	145	41.7	16.4	36.1	32.8	38.5	57.3	15.6	22.5	39.2	24.3*	52.0
Campbell Slough	149	35.6	30.7	18.4	28.9	37.9	41.5	33.6	15.2	33.1	26.6	29.5
Franz Lake	221	ND	ND	ND	33.0	34.3	ND	26.5	5.8	13.8	8.8	15.5

*A different sampling design was used at Cunningham Lake in 2014, so results are not directly comparable to the other years.

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 data for the SAV species are reported with the emergent cover for all of the sites (Appendix C, Table C-1). Cover data for SAV species are also provided for the channels of six of the trends sites (Appendix C, Table C-2); at Cunningham Lake the channel is very small and not distinguishable from the adjacent flats that are included in the transects. In previous years horned pondweed (*Zannichellia palustris*) occurred at the Ilwaco site, where it was found in the tidal channel and in ponded areas of the marsh. In 2015 however, the SAV in the channel was replaced by *Zostera japonica*, a marine species that requires higher salinity than the pondweed. At the Secret River low marsh site, waterweed (*Elodea* spp.) accounted for 27% of the cover in 2015, occurring throughout the low marsh in small depressions that hold water at low tide (Figure 34). At all other sites, SAV species account for less than 5% of the cover in the emergent marsh area. In channels of the lower river, SAV species are more prevalent than in those of the upper river sites and are dominated by the native species *Elodea* spp. and *Potamogeton richardsonii*. The SAV cover in the channel at the Secret River site was 78% up from 49% in 2013, at Welch Island SAV in the channel was 30% down from 85% in 2013, and at Whites Island 38% up from 23%.

Annual vegetation cover is depicted in Figure 34. Variability in the cover of the dominant species is particularly evident at upper estuary sites. At Cunningham Lake and Campbell Slough, cover greater than 80% occurred in the relatively low inundation years (2005, 2007, 2009, 2010, and 2015). Cover at Campbell Slough in 2007 was an exception when cows were periodically present at the site. Cover in other years was affected primarily by inundation, especially during the high water years of 2011 and 2012. At the Franz Lake, a shift in vegetation dominance occurred during this time which continued until 2015. Total cover in 2015 was very high at the three upper estuary sites. Wapato (*Sagittaria latifolia*) had the highest cover measured during the monitoring period at three of the four sites where it occurs. The fourth site, Franz Lake, has had low cover since 2008 due to inundation from the beaver dam at the site. At Ilwaco Slough, cover was the lowest measured during the monitoring period. At two of the sites where

cover varies, there is a positive relationship between the number of species and the aerial cover at the site (Cunningham Lake, $R^2 = 0.70$; Franz Lake, $R^2 = 0.52$).

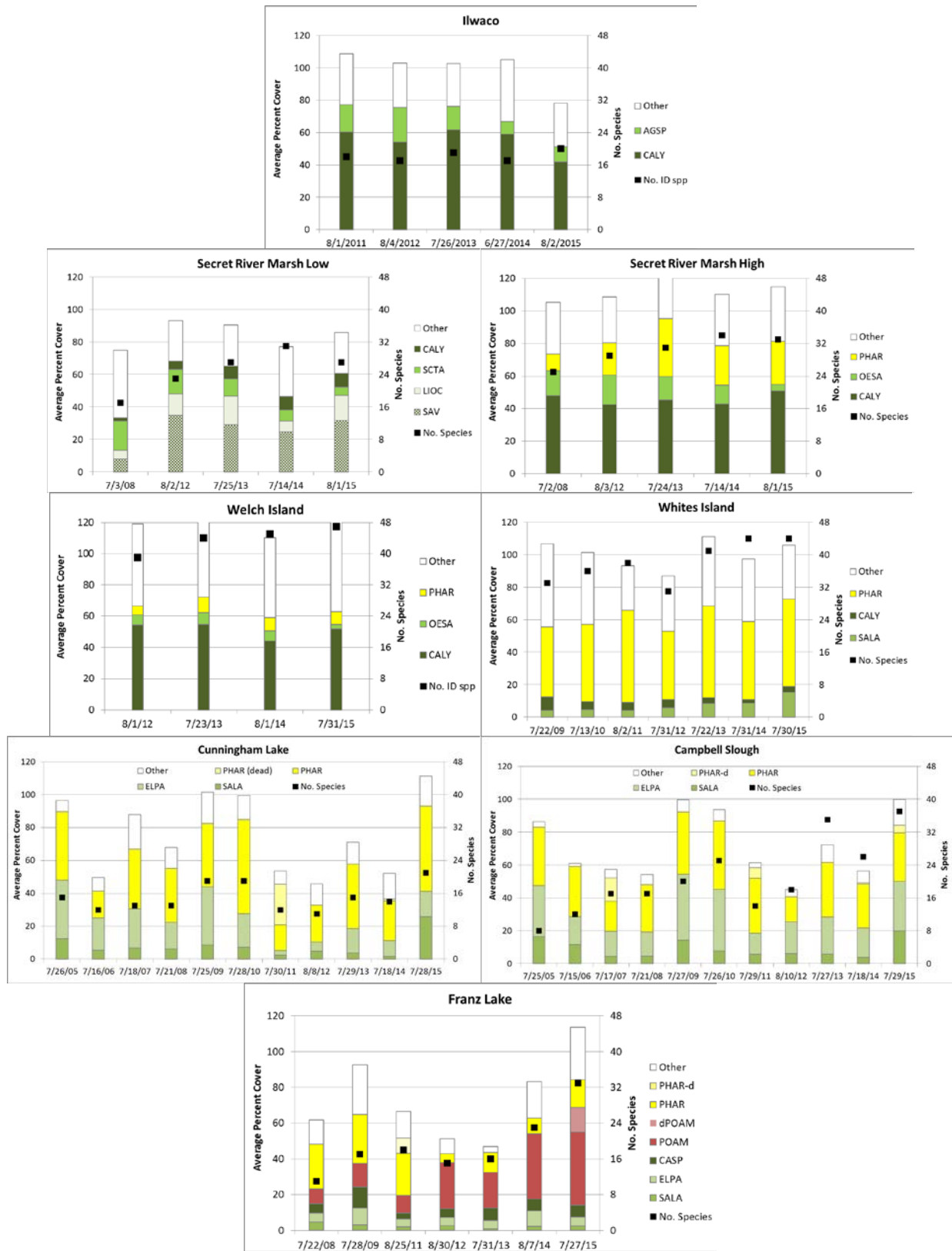


Figure 34. Average percent cover and number of identified species at the trends sites for all years monitored. Sites are presented in the order in which they occur in the river, starting near the mouth.

A weighted similarity analysis of vegetation cover between years at each site was conducted to evaluate change over time and differences between sites. In general, the similarity between years at the trends sites was the greatest at the lower estuary sites (Figure 35 and Table 22) with average similarity between 76 and 82 percent similarity for the four marsh sites below rkm 72. The lowest average similarity was at the three upper estuary sites with 69 percent similarity. Average similarity between years significantly decreases moving up river (Figure 36; regression $p < 0.001$). As the span between years increases, the pairwise similarity for a given site decreases. Thus, for those trends sites observed over a greater number of years, the average similarity decreased significantly with an increasing number of years between observations (Figure 36; regression $p = 0.001$). Similarity was slightly higher for the comparison between the 10 years at Cunningham Lake and Campbell Slough compared to the pairwise comparison of nine years.

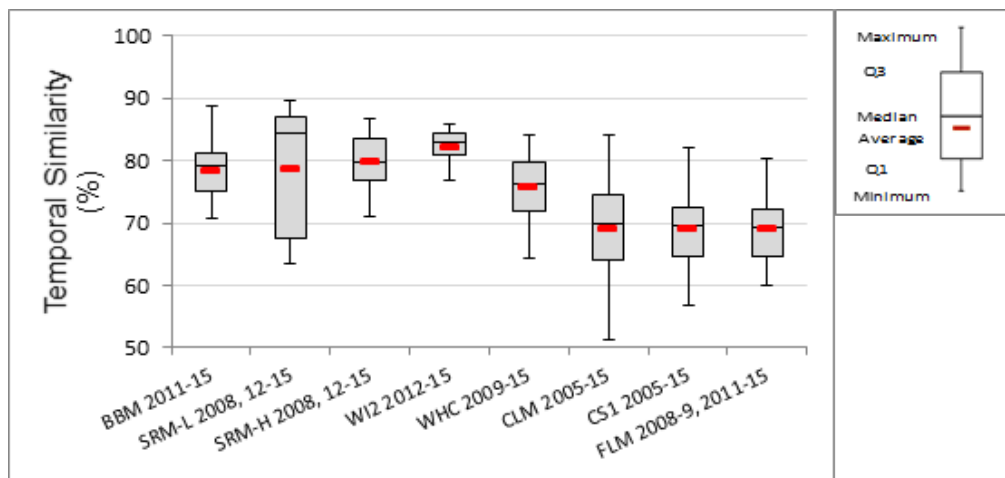


Figure 35. Box plot of the Bray-Curtis similarity calculated between years at the trends sites. Site codes are as follows: BBM = Ilwaco, SRM-L = Secret River low marsh, SRM-H = Secret River high marsh, WI2 = Welch Island, WHC = Whites Island, CLM = Cunningham Lake, CS1 = Campbell Slough, FLM = Franz Lake. Sites are ordered from the mouth to the upper estuary.

Table 22. Descriptive statistics of the percent site similarity of vegetative cover at a site over time. Site codes are as follows: BBM = Ilwaco, SRM-L = Secret River low marsh, SRM-H = Secret River high marsh, WI2 = Welch Island, WHC = Whites Island, CLM = Cunningham Lake, CS1 = Campbell Slough, FLM = Franz Lake. Sites are ordered from the mouth to the upper estuary.

Site	<i>n</i>	Mean	StDev	Minimum	Q1	Median	Q3	Maximum
BBM	10	78.6	5.2	70.7	75.0	79.1	81.2	88.7
SRM-L	10	78.6	10.6	63.4	67.5	84.5	87.1	89.5
SRM-H	10	79.8	4.7	71.0	76.9	79.7	83.5	86.8
WI2	6	82.3	3.3	76.9	80.9	83.0	84.4	85.8
WHC	21	75.7	5.5	64.5	71.8	76.2	79.7	84.1
CLM	55	69.0	7.8	51.4	64.2	69.7	74.5	84.0
CS1	55	69.0	5.5	56.9	64.7	69.5	72.4	82.1
FLM	21	69.3	6.3	60.1	64.7	69.2	72.1	80.5

¹ The number of comparisons (*n*) is based on the number of years a site was monitored; for example, CLM was monitored for 11 years and 55 year-to-year comparisons could be made.

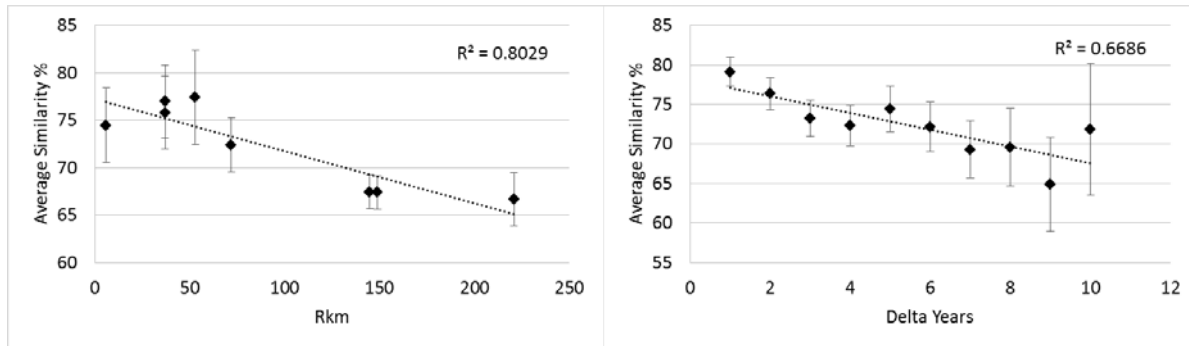


Figure 36. Average similarity of sites using the least square mean model for individual sites with the effect of the difference in years removed (left plot), and the average similarity between years with the effect of the different sites removed (right plot).

3.3.5 Channel Morphology and Inundation

Low inter-annual variability of channel morphology at the trends sites has been observed in years prior to 2015; therefore, only the channel mouth cross section was surveyed this year. Channel measurements from previous years are presented with the newly calculated inundation frequency results from 2015 in Table 23. The tidal channels measured at the sites were generally small, with cross sectional areas less than 10 m² (see Appendix B for locations of the measured channels). Five of the tidal channels surveyed were primary channels feeding directly into the Columbia River, while the channels at the Welch Island and Whites Island sites were secondary channels that feed into a larger tidal channel. The Secret River channel had the greatest area: close to 20 m² for most of its length. The channels varied in width from 1.3 m to 50.1 m; most becoming narrower with increasing elevation, with the exception of the Ilwaco and Whites Island channels, which were slightly wider at the middle than at the mouth. Channel depth ranged 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.

In 2015, three of the channels (Secret River, Campbell Slough, and Franz Lake) were inundated to a depth greater than 50 cm at least 69% of the time, while the other three (Ilwaco Slough, Whites Island, and Cunningham Lake) ranged between 36 and 75%. The 2015 channel inundation frequency during the spring/summer peak salmonid migration period was slightly lower ($\leq 10\%$) than the inundation frequency during the entire one year deployment period (Table 23), except at Franz Lake, where inundations were slightly higher in the spring/summer period. This is in contrast to previous years when inundation frequencies were usually much greater in the spring/summer in the upper portion of the estuary. The lower part of the channel at Franz Lake (e.g., cross sections 0 and 3) was probably not actually inundated 98-100% of the time. The frequencies are elevated due to the location of the sensor upstream from a beaver dam; water levels below the beaver dam are more variable and are approximately 0.7 m lower than the beaver dam induced water level at the sensor. The beaver dam appears to wash out every year then is gradually built up from an elevation of approximately 0.8 m to 1.5 m CRD (see Appendix A for annual hydrographs of Franz Lake).

The channel banks were inundated between 6 and 67% in 2015 (Table 23). The lowest inundation frequency occurred at Ilwaco Slough, Campbell Slough, and Franz Lake. The highest frequency was at Secret River, Whites Island, and Cunningham Lake. The low frequency at the upper estuary sites is in stark contrast to previous years when inundation frequencies during the spring/summer were often near 100%.

Table 23. Physical channel metrics and inundation frequencies measured at each site. The channel mouth (indicated with an *) was measured in 2015; the year of full channel measurement is provided in parentheses after the site code. Inundation frequencies are calculated for one year (August 2014 - August 2015) and compared to results for five months between 1 March and 31 July 2015 (the peak juvenile Chinook salmon migration period). Cross sections are numbered starting at the mouth.

Physical Metrics								Inundation			
Site (year)	Cross Section	Thalweg Elevation (m, CRD)	Bank Elevation (m, CRD)	Channel Depth (m)	Channel Width (m)	Width: Depth Ratio	Cross Section Area (m ²)	Year		March-July	
								% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm	% Time WL > Thalweg + 50cm	% Time WL > Bank + 10cm
BBM (11)	1*	0.93	1.62	0.69	6.3	9.2	3.7	50	37	45	34
	2	0.70	1.86	1.16	9.30	8.04	8.94	60	27	55	23
	3	0.90	2.12	1.22	10.10	8.27	9.73	51	15	47	10
	4	1.01	2.00	0.99	5.20	5.23	4.33	46	20	42	15
	5	1.17	2.26	1.09	2.70	2.48	1.58	40	10	36	6
SRM (12)	0*	0.10	0.92	0.82	23.1	28.1	9.3	89	67	84	64
	1	0.32	1.42	1.09	22.6	20.6	19.3	76	49	72	46
	2	-0.04	2.13	2.17	14.9	6.87	22.5	96	21	94	18
	3	-0.03	1.98	2.01	15.1	7.52	20.7	96	27	93	24
WI2 (12)	1*	0.29	1.74	1.45	20.0	13.8	14.8	NA	NA	NA	NA
	2	0.36	1.65	1.29	9.20	7.13	8.75	NA	NA	NA	NA
	3	0.71	1.80	1.09	5.09	4.67	3.96	NA	NA	NA	NA
	4	0.78	1.74	0.96	3.30	3.44	2.07	NA	NA	NA	NA
	5	1.31	1.62	0.31	1.32	4.27	0.42	NA	NA	NA	NA
WHC (11)	1*	0.57	1.01	0.45	33.8	75.9	8.0	74	62	71	59
	2	0.34	1.41	1.07	20.5	19.1	10.8	75	43	73	41
	3	0.61	1.53	0.92	36.2	39.5	11.1	62	37	59	35
	4	0.92	1.93	1.00	50.1	50.0	34.0	47	19	45	17
	5	0.44	1.45	1.01	2.83	2.80	1.90	70	41	67	39
CLM (15)	1	0.82	1.17	0.34	17.3	50.5	2.5	55	61	52	59
CS1 (15)	1	0.77	1.49	0.73	23.0	31.6	19.0	69	19	75	7
FLM (12)	0*	0.34	2.23	1.89	23.2	12.2	21.3	99	13	100	9
	3	0.40	1.39	0.99	14.3	14.4	4.20	98	51	100	53
	4	0.85	1.45	0.60	13.2	22.0	6.20	76	45	84	49

3.4 Food Web

3.4.1 Primary Production

3.4.1.1 *Emergent Wetland Vegetation*

Some of the results presented here were previously reported in Sagar et al. (2014), Hanson et al. (2014), and Hanson et al. (2015); however, they are summarized here again to provide context for the most recent data collected in the summer of 2015 and winter of 2016.

Quantity

The above ground biomass estimates for the emergent wetland vegetation found in the low and high marsh strata are provided in Table 24. In the years prior to 2015, the greatest biomass occurred in the high marsh, with statistically significant differences between each of the three marsh strata: high marsh, low marsh and SAV ($r^2=73\%$, $p < 0.01$). In 2015, the biomass in the high marsh was still the greatest (average of 1069 g/m^2). However, the low marsh average of 448 g/m^2 was greater than that measured in previous years (average of 249 g/m^2).

In 2015, one lower estuary site was sampled in zone 2 (see Figure 1 for zone delineations), two sampled in zone 4, and one in zone 5. Prior to 2015 six sites were sampled and summer biomass estimates in all strata decreased with increasing rkm. The four lower estuary sites had greater biomass than the two upper estuary sites (high marsh average of 1162 g/m^2 and 426 g/m^2 , respectively). In 2015, this trend continued in the high marsh biomass estimates, with a decrease from 1281 g/m^2 at Whites Island to 893 g/m^2 at the Franz Lake site. However, in the low marsh stratum, the biomass estimates were more consistent at the sites sampled, with the highest amount measured at the Campbell Slough site in the upper estuary.

The submerged aquatic vegetation (SAV) stratum was not sampled in 2015 due to the low organic matter production estimates measured in previous years. When sampled, the SAV summer biomass was usually the lowest of the three strata averaging 42 g/m^2 . The highest SAV biomass estimate was at Welch Island in 2013 (173 g/m^2) and the lowest was at Franz Lake Slough in 2013 (0.2 g/m^2).

Temporal trends are difficult to discern because of sampling limitations over the four-year period. Comparisons between all four years are limited to two sites for the high marsh strata and one site for the low marsh strata. The high marsh at the Whites Island site had consistent biomass between years, except in 2012 when both summer and winter standing stock were lower. In contrast, the Franz Lake site was more variable between years, with the summer standing stock highest in 2012 and 2015. Winter standing stock was considerably higher at the site in 2015 however, resulting in lower organic matter production compared to 2012. The low marsh site had higher summer biomass in 2015 than in previous years, while the winter standing stock remained low.

Table 24. Average aboveground standing stock of emergent wetland vegetation from high marsh and low marsh strata. Organic matter production for each year is calculated as the summer standing stock minus remaining winter standing stock (g/m²). Sites are ordered by distance from the CR mouth.

		Avg Dry wt g/m ² (SD)																							
Site	Strata	Summer Standing Stock 2011			Winter Standing Stock* 2012			Summer Standing Stock 2012			Winter Standing Stock 2013			Summer Standing Stock 2013			Winter Standing Stock 2014			Summer Standing Stock 2015			Winter Standing Stock 2016		
		n			n			n			n			n			n			n			n		
Ilwaco (BBM)	high marsh	7	976 (421)	7	385 (133)	591	10	1175 (257)	10	254 (135)	921	10	1141 (429)	10	227 (175)	914	ND	ND	ND	ND	ND	ND	ND	ND	ND
Secret R. (SRM)	high marsh	ND	ND	ND	ND	ND	5	1443 (148)	5	194 (210)	1248	9	1062 (386)	9	241 (151)	821	ND	ND	ND	ND	ND	ND	ND	ND	ND
Welch Is. (WI2)	high marsh	ND	ND	ND	ND	ND	5	1141 (322)	9	272 (122)	870	9	1361 (647)	9	365 (150)	996	ND	ND	ND	ND	ND	ND	ND	ND	ND
Whites Is. (WHC)	high marsh	6	1152 (844)	5	517 (327)	635	8	740 (623)	8	346 (258)	393	9	1359 (834)	9	670 (873)	689	12	1281 (463)	12	497 (274)	784	ND	ND	ND	
Cunningham Lake (CLM)	high marsh	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	7	1008 (265)	7	319 (171)	689	ND	ND	ND	
Campbell Sl. (CS1)	high marsh	3	410 (356)	4	101 (64)	309	ND	ND	ND	ND	ND	6	434 (67)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
Franz Lake (FLM)	high marsh	8	203 (152)	12	245 (114)	-42	7	672 (557)	5	104 (107)	567	9	434 (317)	9	234 (222)	200	12	893 (719)	12	510 (238)	383	ND	ND	ND	
Ilwaco (BBM)	low marsh	1	24 (NA)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
Secret R. (SRM)	low marsh	ND	ND	ND	ND	ND	5	265 (71)	5	15 (15)	250	9	175 (124)	9	9 (9)	166	ND	ND	ND	ND	ND	ND	ND	ND	
Welch Is. (WI2)	low marsh	ND	ND	ND	ND	ND	4	401 (362)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	
Whites Is. (WHC)	low marsh	2	88 (89)	3	6 (6)	79	3	114 (102)	3	10 (15)	104	6	163 (126)	6	9 (5)	153	6	441 (407)	6	32 (34)	409	ND	ND	ND	
Cunningham Lake (CLM)	low marsh	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	6	296 (212)	6	7 (6)	289	ND	ND	ND	
Campbell Sl. (CS1)	low marsh	5	278 (151)	4	3 (4)	274	ND	ND	ND	ND	ND	11	56 (38)	ND	ND	ND	12	527 (352)	12	40 (70)	487	ND	ND	ND	
Franz Lake (FLM)	low marsh	ND	ND	1	66 (NA)	ND	ND	ND	2	30 (24)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	

OM = Organic Matter; SD = Standard Deviation; ND = No Data; NA = Not Applicable

* Winter standing stock includes only plant material from the previous year. New, live shoots were excluded.

Species Composition

In general, the species comprising the vegetation biomass samples are the dominant species found in the lower river. Table 25 provides a summary of the biomass results for the dominant species in the sample sites. Although dominant species were targeted, occasionally the samples were a mix of more than one species.

In 2015, the four trends sites that were sampled for biomass were located in the portion of the lower river with a high probability of *P. arundinacea* occurrence (Borde et al. 2012; Sagar et al. 2013). The Whites Island site (rkm 72) is near the upper limit of *C. lyngbyei*, with only a small contiguous patch, while the rest of the high marsh was a mix of *P. arundinacea* and other species. Of the other three sites, only Cunningham Lake had a homogeneous stand of *P. arundinacea*. The Campbell Slough site had been grazed and therefore the high marsh was not sampled and the Franz Lake site has an area of *P. arundinacea* that is mixed with *Polygonum amphibium*. In 2015 sampling was conducted to enable quantification of the target species relative to the other species in the sample; only samples that were comprised predominantly of that species were included in the averages presented in Table 25. The low marsh species, *Eleocharis palustris* and *Sagittaria latifolia*, were present at all four of the sample sites in 2015.

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 organic matter (detritus), an important component of the juvenile salmonid food web. To estimate detrital production, the winter standing stock is subtracted from the summer peak standing stock, providing an estimate of the annual detritus production for the wetland. These estimates are presented in Table 24 and Table 25 as the difference between summer and winter standing stock values. Similar spatial patterns to those observed for summer biomass also apply regarding the annual detrital contribution. In general, the annual detrital contribution was greater in the lower estuary than in the upper estuary, although an increase was observed at the Franz Lake site in 2012 (Table 24). A similar pattern is apparent when the detrital contribution from individual strata or species is evaluated (Table 25) in part due to *C. lyngbyei* only occurring in the lower estuary. In 2015, the species with the highest summer standing stock and the greatest organic matter production was *C. lyngbyei* followed by *P. amphibium*; however, the latter had a much higher percentage of winter standing stock remaining and therefore less organic matter contributed to the system over the winter (Table 25). Similarly, approximately half the summer standing stock of *P. arundinacea* remained in the winter, resulting in lower organic matter contribution than the other high marsh species. Low marsh species have lower biomass production in the summer, however most of the plants die back in the winter, contributing most of the organic matter to the system by early spring. The two most common species in the samples were *C. lyngbyei* and *P. arundinacea*. The average annual detritus contribution for *C. lyngbyei* across all sites and all years was 1161 g/m², while *P. arundinacea* was 627 g/m².

Table 25. Average aboveground summer and winter standing stock of the dominant species of emergent wetland vegetation. Annual organic matter production is calculated as the summer standing stock minus remaining winter standing stock (g/m²).

		Avg Dry wt g/m ² (SD)																			
Scientific Name	Common Name	2011			2012			2013			2014			2015			2016				
		Summer Standing Stock	Winter Standing Stock*	OM Prod	Summer Standing Stock	Winter Standing Stock	OM Prod	Summer Standing Stock	Winter Standing Stock	OM Prod	Summer Standing Stock	Winter Standing Stock	OM Prod	Summer Standing Stock	Winter Standing Stock	OM Prod	Summer Standing Stock	Winter Standing Stock	OM Prod		
		<i>n</i>	<i>n</i>		<i>n</i>	<i>n</i>		<i>n</i>	<i>n</i>		<i>n</i>	<i>n</i>		<i>n</i>	<i>n</i>		<i>n</i>	<i>n</i>			
<i>Carex lyngbyei</i>	Lyngby sedge	3	1049 (558)	5	331 (192)	718	10	1234 (377)	14	177 (115)	1057	7	1105 (290)	14	305 (154)	801	3	1789 (183)	3	200 (87)	1589
<i>C. lyngbyei/Agrostis spp</i>	Lyngby sedge/ bentgrass	4	921 (370)	3	351 (194)	570	4	1009 (153)	6	236 (168)	773	6	1041 (527)	5	127 (98)	914	ND	ND	ND	ND	ND
<i>C. lyngbyei/mixed spp</i>	Lyngby sedge/ high marsh	ND	ND	ND	ND	ND	8	1250 (288)	2	263 (35)	987	15	1291 (543)	5	261 (221)	1030	ND	ND	ND	ND	ND
<i>Phalaris arundinacea</i>	reed canary grass	9	578 (760)	13	306 (270)	272	ND	ND	6	353 (286)	NA	9	716 (718)	9	297 (251)	419	18	850 (450)	20	437 (206)	413
<i>Polygonum amphibium</i>	water smartweed	ND	ND	ND	ND	ND	3	747 (488)	1	274 (NA)	473	4	208 (192)	2	286 (296)	-79	6	1412 (645)	6	619 (303)	793
<i>Eleocharis palustris/Sagittaria latifolia</i>	spike rush/ wapato	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	6	65 (30)	ND	ND	ND	11	332 (170)	12	43 (68)	289
<i>Sagittaria latifolia</i>	wapato	1	150 (NA)	4	4 (6)	146	4	91 (94)	3	10 (15)	81	11	110 (112)	6	10 (5)	100	9	505 (437)	10	5 (8)	500

3.4.1.2 Pelagic

Chlorophyll *a* concentrations were determined at six trends sites (Ilwaco Slough, Welch Island, Grays River, Whites Island, Campbell Slough, and Franz Lake Slough) over the spring-summer months (Figure 37). The lowest values were observed at Franz Lake Slough and Campbell Slough (with the exception of a high concentration in July). The highest chlorophyll *a* concentrations were seen at Welch Island and Whites Island in reaches B and C, respectively.

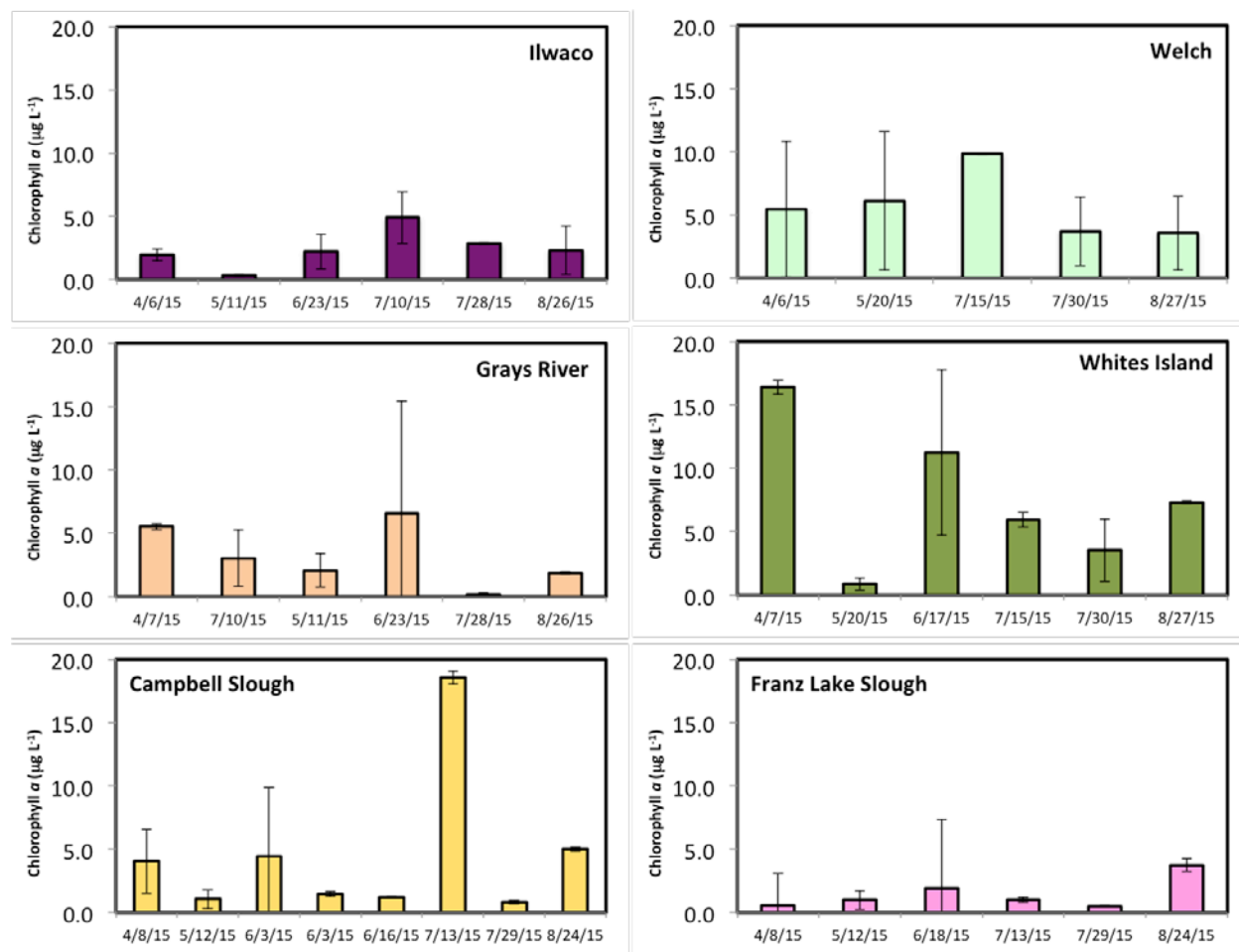


Figure 37. Time series of chlorophyll *a* concentrations ($\mu\text{g L}^{-1}$) at six trends sites in 2015. Error bars represent one standard deviation of the mean of three replicates.

The phytoplankton species composition was numerically dominated by diatoms (Class Bacillariophyceae) at Whites Island throughout the time series (Figure 38). In contrast, cyanobacteria were numerically dominant at Campbell Slough and Franz Lake Slough. Twenty-five major taxa were noted, including 10 diatoms (Bacillariophyceae), two cryptophytes, eight chlorophytes, four cyanobacteria and one “other” (unidentified dinoflagellate) were identified based on relative contribution to total phytoplankton abundance. Common chlorophyte taxa included *Scenedesmus* sp., *Micratinium* sp., *Ankistrodesmus* sp., and several colonial forms, including *Eudorina* sp., *Dictyosphaerium* sp., *Gloeocystis* sp., and *Sphaerocystis* sp. For multivariate analyses, colonial chlorophytes (excluding *Micratinium* sp. and cf. *Spondylosium* sp.) were grouped into one taxonomic category due to uncertainty in identification. Several small, unicellular flagellated chlorophytes were also common but were difficult to identify, and were

therefore also summed into one taxonomic group. Small, unicellular non-flagellated chlorophytes were grouped into a separate taxonomic group. The dominant major cyanobacteria observed included *Microcystis* sp., *Dolichospermum* sp., *Merismopedia* sp., and unidentified filamentous cyanobacteria.

Consistent seasonal and inter-site differences exist among phytoplankton assemblages over the five-year period (2011–2015). Diatoms (Class Bacillariophyceae) accounted for a large proportion of total phytoplankton abundance during the spring (April–May) at Whites Island and Franz Lake Slough every year (Figure 38). In contrast, Campbell Slough had high diatom abundances in the spring (April–May) of 2011, 2012, and 2015, but not 2013 and 2014. At Whites Island, the dominance of diatoms continued through the summer months during every year of the study, while Franz Lake Slough and Campbell Slough had high abundances of cyanobacteria (at least 10,000 cells mL⁻¹) during the summer months (June and July) almost every year. Other phytoplankton taxa, consisting mostly of cryptophytes (Class Cryptophyceae) and green algae (Class Chlorophyceae) were abundant (up to about 3,000 cells mL⁻¹) at various points throughout this study, but rarely constituted a majority.

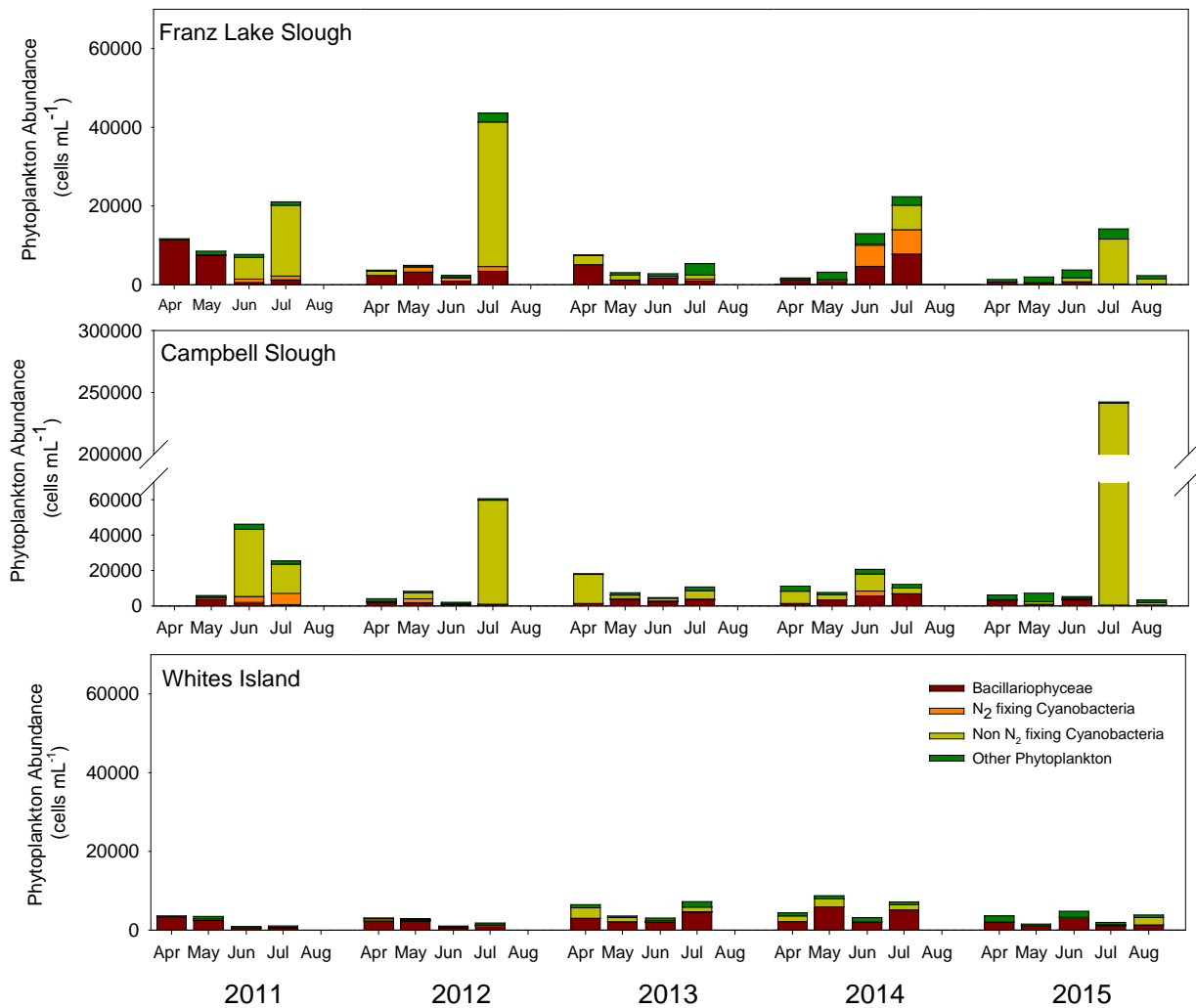


Figure 38. Time series of phytoplankton community at all three study sites (cells mL⁻¹, left). Dark red bars represent Bacillariophyceae (diatoms), orange bars represent nitrogen-fixing cyanobacteria, light-green bars represent non-nitrogen-fixing cyanobacteria and dark green bars represent all other phytoplankton.

Total cyanobacteria abundances at Campbell Slough and Franz Lake Slough followed similar seasonal patterns (Figure 38), with high abundances typically occurring in June and July. The timing of elevated abundances of cyanobacteria was similar at both sites, with high abundances coinciding with warm water temperatures in the mid to late summer.

Diatom community composition varied within sites and between seasons at all three sites. In the two high water years (2011 and 2012), *Asterionella formosa* accounted for a large proportion of the diatom assemblage during the spring months leading up to the freshet (April-May; Figure 39). *Aulacoseira* spp. were present at high abundances, but tended to occur slightly later in the season than *A. formosa*, usually in May and June and occasionally into July at Franz Lake Slough (2012 and 2014). Small (<20 µm) centric diatoms, both solitary and colonial, were sometimes abundant and tended to be present at high abundances during early spring (notably in April 2011 at Franz Lake Slough). *Skeletonema potamos* was abundant each year at all three sites, although to a lesser extent at Franz Lake Slough compared to Campbell Slough or Whites Island. This species tended to dominate the diatom community later in the summer, between June and July.

The diatom assemblages were dominated by one or two species at a time during 2011 and 2012; in contrast, several diatom species tended to co-dominate in 2013 and 2014 (Figure 39). *A. formosa* was consistently most abundant (>1,000 cells mL⁻¹) during early spring from 2011-2014, particularly at Whites Island; however, this species was present at low abundance during the spring of 2015 when discharge and precipitation were low. During most years, species of the diatom genus *Aulacoseira* increase in abundance following blooms of *A. formosa*. Interestingly, whereas *A. formosa* was not abundant in the system in 2015, *Aulacoseira* spp. were very abundant throughout 2015 at Whites Island. Diatom abundance and diversity at Franz Lake Slough was very low in 2015 where the assemblage was comprised almost entirely of small (<10 µm) *Nitzschia* sp.

The mean total abundance of phytoplankton (mean of total abundance during all sampling dates in one season) at Whites Island was lower during high-water years (2011 and 2012) than in years where discharge was closer to the 10-year average (2013 and 2014). In Campbell Slough and Franz Lake Slough, the opposite pattern was observed, although high abundances during 2011 and 2012 were primarily the result of high cyanobacteria abundances in the summer months. Total cyanobacteria abundance at both Franz Lake Slough and Campbell Slough was higher during the summers of 2011 and 2012, with relatively low abundances during summer months in 2013 and 2014.

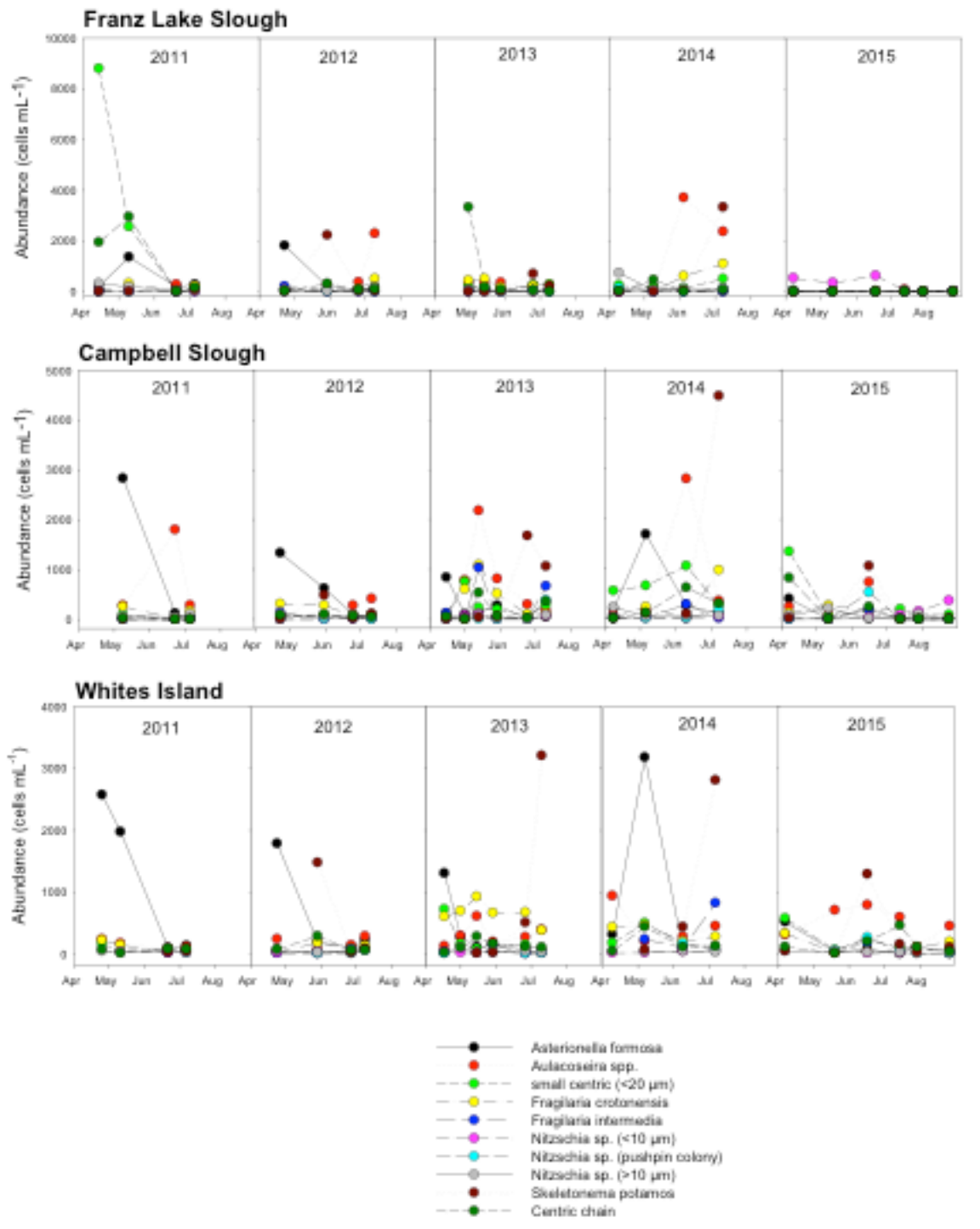


Figure 39. Time series of major diatom taxa at Franz Lake Slough, Campbell Slough and Whites Island between April 1 and August 31, 2011-2015.

High abundances of cyanobacteria were observed primarily in the late summer after the annual freshet subsided. Although total cyanobacteria abundances at Campbell Slough and Franz Lake Slough followed similar seasonal patterns, population dynamics of particular species differed between the two sites and

among the years examined. For example, in June and July 2014, higher abundances of *Dolichospermum* sp. were observed at Franz Lake Slough compared to Campbell Slough. Perhaps the most notable difference among the two sites was the very high abundance (monthly average of about 237,000 cells mL⁻¹) of *Merismopedia* sp. observed at Campbell Slough in July 2015, which was reduced to 1,130 cells mL⁻¹ by August 2015. *Merismopedia* sp. was absent in Whites Island during that time period and present in Franz Lake Slough only at very low abundance (<50 cells mL⁻¹).

At both Franz Lake Slough and Campbell Slough, cyanobacteria populations tended to be dominated by non-nitrogen-fixing cyanobacteria (primarily *Microcystis* sp.), although nitrogen-fixing cyanobacteria (primarily *Dolichospermum* sp.) were present during most years and were occasionally very abundant. For example, *Dolichospermum* sp. was present at >10,000 cells mL⁻¹ in June and July of 2014 in Franz Lake Slough. This taxon was more prevalent in both Campbell Slough and Franz Lake Slough during high-water years (2011 and 2012) compared to average or low-water years.

No statistically significant relationships were observed between cyanobacteria abundances and Σ DIN at any of the three sample sites (data not shown). At Campbell Slough and Franz Lake Slough, high abundances of cyanobacteria (>20,000 cells mL⁻¹) were observed when DIP was moderately high (between 0.01 and 0.10 mg L⁻¹), but not when DIP was above 0.10 mg L⁻¹ (low flow periods; Figure 40). High abundances of cyanobacteria (>20,000 cells mL⁻¹) were observed only when Σ DIN:DIP was low (<10), with the exception of Campbell Slough during May, 2013 when high Σ DIN was observed and Σ DIN:DIP was >200 (Figure 41).

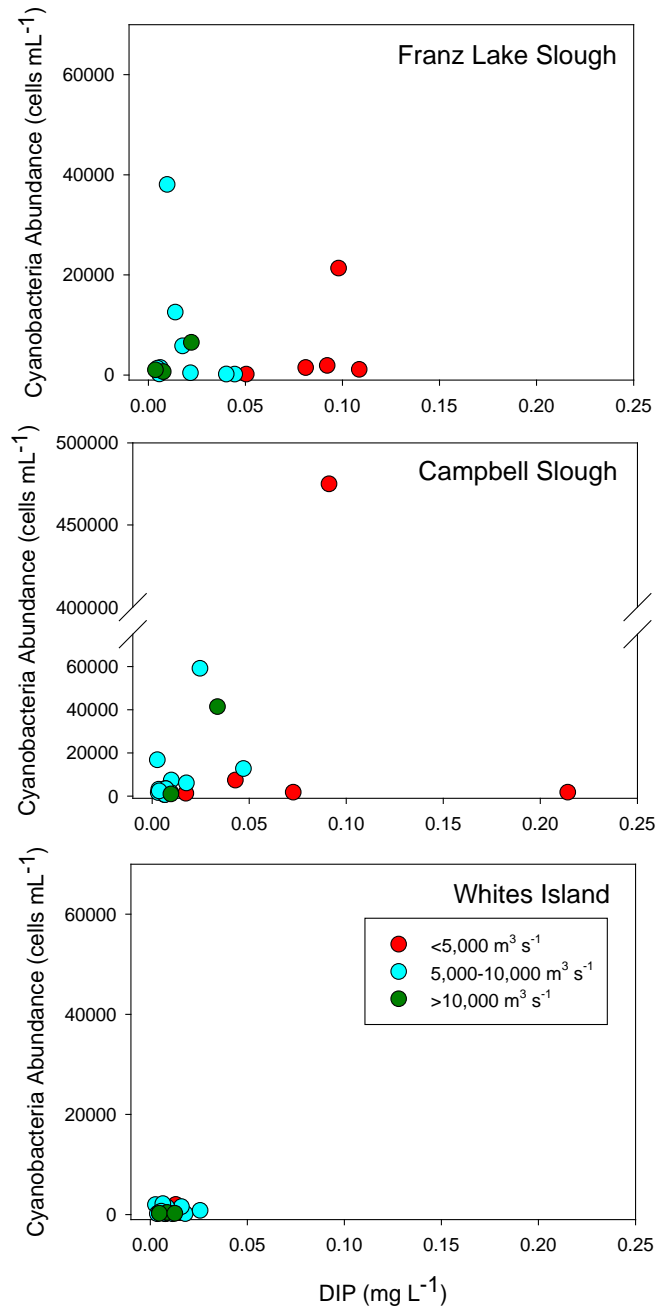


Figure 40. Scatter plots showing cyanobacteria abundance (cells mL⁻¹) in relation to DIP concentrations (mg L⁻¹) at the three study sites. Samples were grouped according to mean daily discharge at Bonneville Dam (indicated by color coding). Red circles indicate mean daily discharge <5,000 m³ s⁻¹; blue circles indicate mean daily discharge between 5,000-10,000 m³ s⁻¹; green circles indicate mean daily discharge >10,000 m³ s⁻¹.

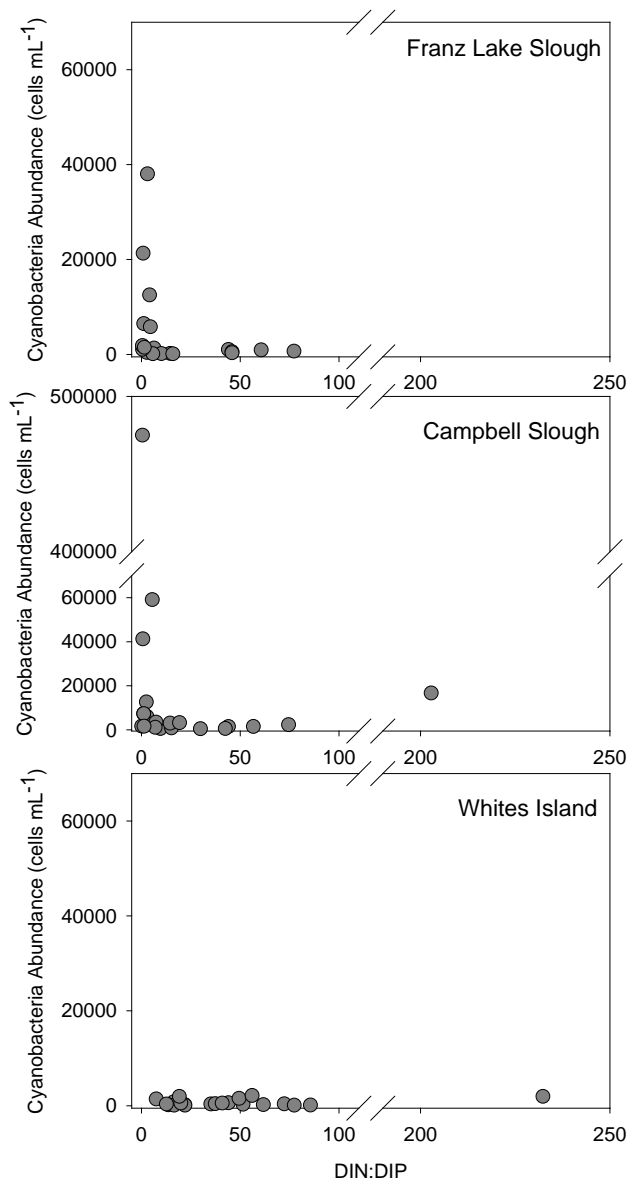


Figure 41. Scatter plots showing cyanobacteria abundance (cells mL⁻¹) in relation to molar ratio of DIN:DIP at the three study sites.

3.4.2 Multivariate Statistical Analyses of Phytoplankton Communities

Two Non-Metric Multidimensional Scaling (NMDS) analyses were performed: one grouping all major phytoplankton taxa and one grouping only diatoms. In the first NMDS (NMDS_{total}), ordination plots were assembled with a 2D stress level of 0.19 and a 3D stress of 0.13. In 2D ordination space, samples from Whites Island were located relatively close to each other, while samples at both Campbell Slough and Franz Lake Slough were more distantly distributed (Figure 42). This distribution pattern indicates high intra-site similarity among phytoplankton assemblages at Whites Island and low similarity at the other two sites. The distribution of samples in the 2D plot revealed a horizontal gradient in terms of river discharge, with distinct taxa associated with different river discharge magnitudes. *Merismopedia* sp., small *Nitzschia* sp. (<10 µm), and unicellular non-flagellated chlorophytes were higher in abundance at

low discharge ($<5,000 \text{ m}^3 \text{ s}^{-1}$), while the colonial diatom *Asterionella formosa* and non-colonial small centric diatoms ($<20 \mu\text{m}$) were dominant during higher discharge periods.

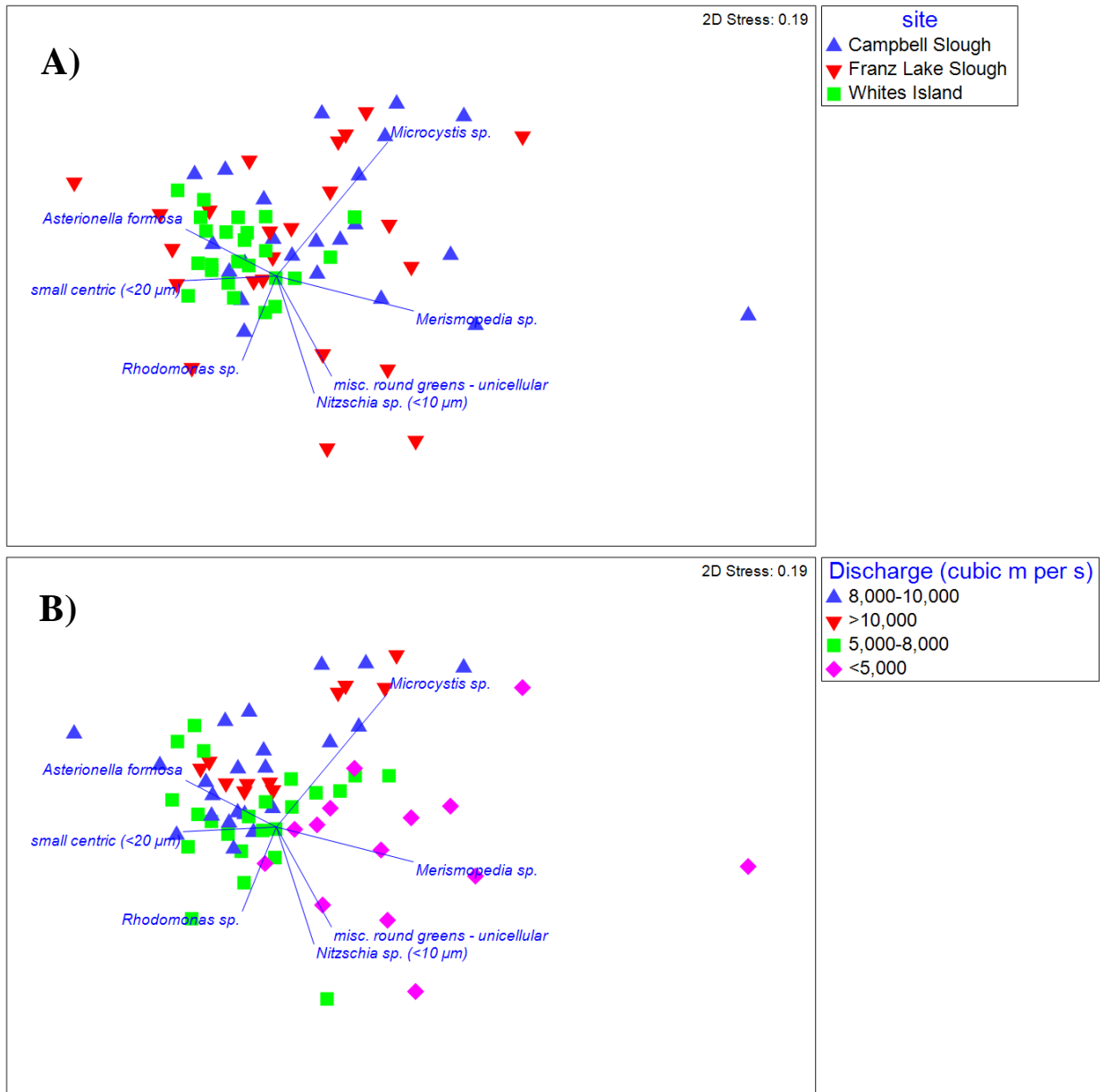


Figure 42. Two-dimensional nonmetric multidimensional scaling (NMDS_{total}) plot based on Bray-Curtis similarities between densities of phytoplankton taxa determined at trends sites between 2011 and 2015. Significant correlations between taxa and sites ($r > 0.4$) are indicated by vectors. Samples are colored by A) site and B) discharge category.

Given that diatoms are also valuable indicators of water quality, we performed NMDS on major diatom taxa separately from all other taxa (Figure 43). Three major taxa—*Asterionella formosa*, *Skeletonema potamos*, and small ($<10 \mu\text{m}$) *Nitzschia sp.*—were identified with Spearman correlations of $r > 0.6$. The samples were distributed along a gradient from high to low and also along a temporal gradient showing a

distinct separation between spring samples (April and May) and summer samples (June, July and August). There was an inverse relationship between *A. formosa* and *S. potamos* in terms of seasonal abundances, with *A. formosa* dominating during the spring and *S. potamos* dominating during the summer. Small (<10 μm) *Nitzschia* sp. were positively associated with periods of low discharge (<5,000 $\text{m}^3 \text{s}^{-1}$), while *A. formosa* and *S. potamos* were both associated with moderate to high discharge (>5,000 $\text{m}^3 \text{s}^{-1}$).

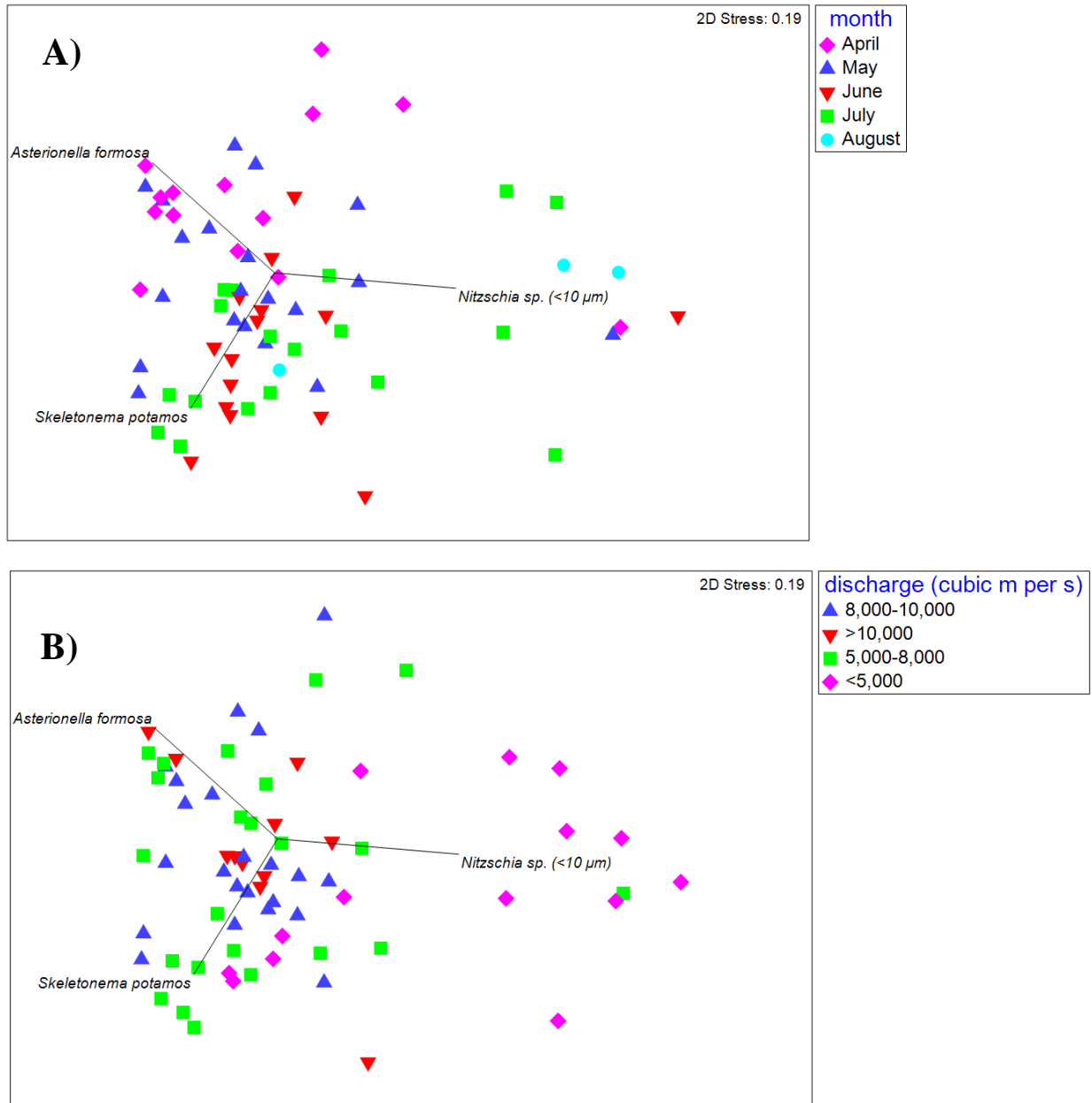


Figure 43. Two dimensional nonmetric multidimensional scaling plot based on Bray-Curtis similarities between densities of diatom taxa determined at trends sites between 2011 and 2015 (NMDS_{diatom}). Significant correlations between taxa and sites ($r > 0.6$) are indicated by vectors. Samples are colored by (A) month and (B) discharge category.

Relationships between taxonomic assemblages of phytoplankton and environmental variables were explored using Canonical Analysis of Principal Coordinates (CAP). CAP produced two pairs of canonical axes, which explained 79% (CAP1) and 50% (CAP2) of the variance in correlations between phytoplankton abundance and abiotic variables (Figure 44). Water temperature and PO_4^{3-} were positively correlated to the first CAP axis ($r = 0.600$ and $r = 0.605$, respectively), while mean daily discharge was negatively correlated with the first CAP axis ($r = -0.449$). PO_4^{3-} was positively correlated with the second CAP axis ($r = 0.551$), while water temperature was negatively correlated with the second CAP axis ($r = -0.640$). All other environmental variables were not strongly correlated with either CAP axis. Along the first CAP axis (CAP1), there was an overall pattern of higher discharge samples on the negative side of the axis and lower discharge samples on the positive side of the axis. Samples in the lowest discharge threshold category ($<5,000 \text{ m}^3 \text{ sec}^{-1}$) were taken primarily in 2015.

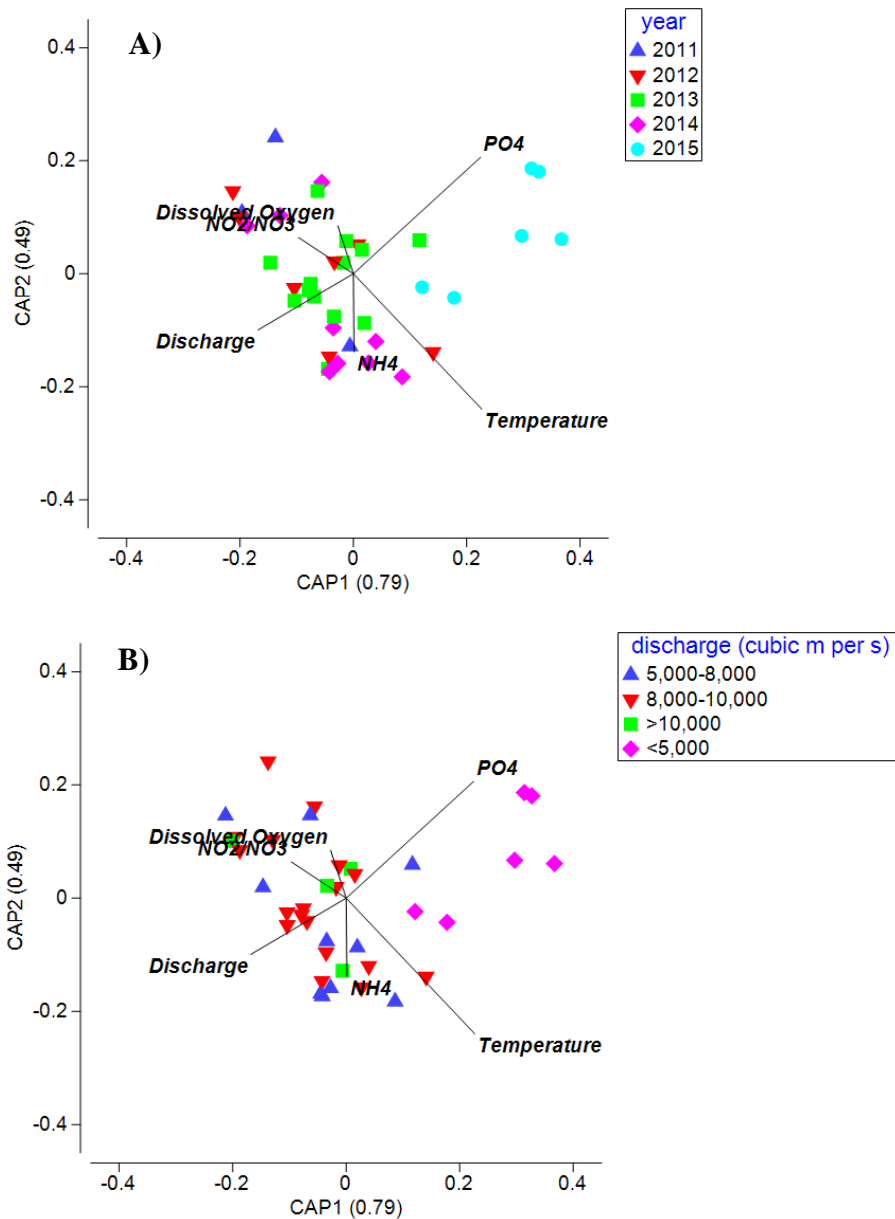


Figure 44. CAP ordination diagram showing all six environmental variables with samples colored coded by A) year and B) discharge thresholds ($\text{m}^3 \text{ s}^{-1}$).

When particular species were examined in relation to environmental variables within the NMDS ordination, clear differences were observed (Figure 45). The highest relative abundances of the spring diatom, *Asterionella formosa*, were observed when levels of dissolved oxygen, nitrate, and river discharge were high and temperatures were low. In contrast, the highest relative abundances of the summer diatom, *Skeletonema potamos*, was observed when temperatures and ammonium concentrations were high. The two cyanobacteria species (*Microcystis* sp. and *Dolichospermum* sp.) were both observed when temperature and ammonium concentrations were high; however, there was a stronger correlation between ortho-phosphate concentration and the relative abundance of *Microcystis* sp. compared to *Dolichospermum* sp. (Figure 45).

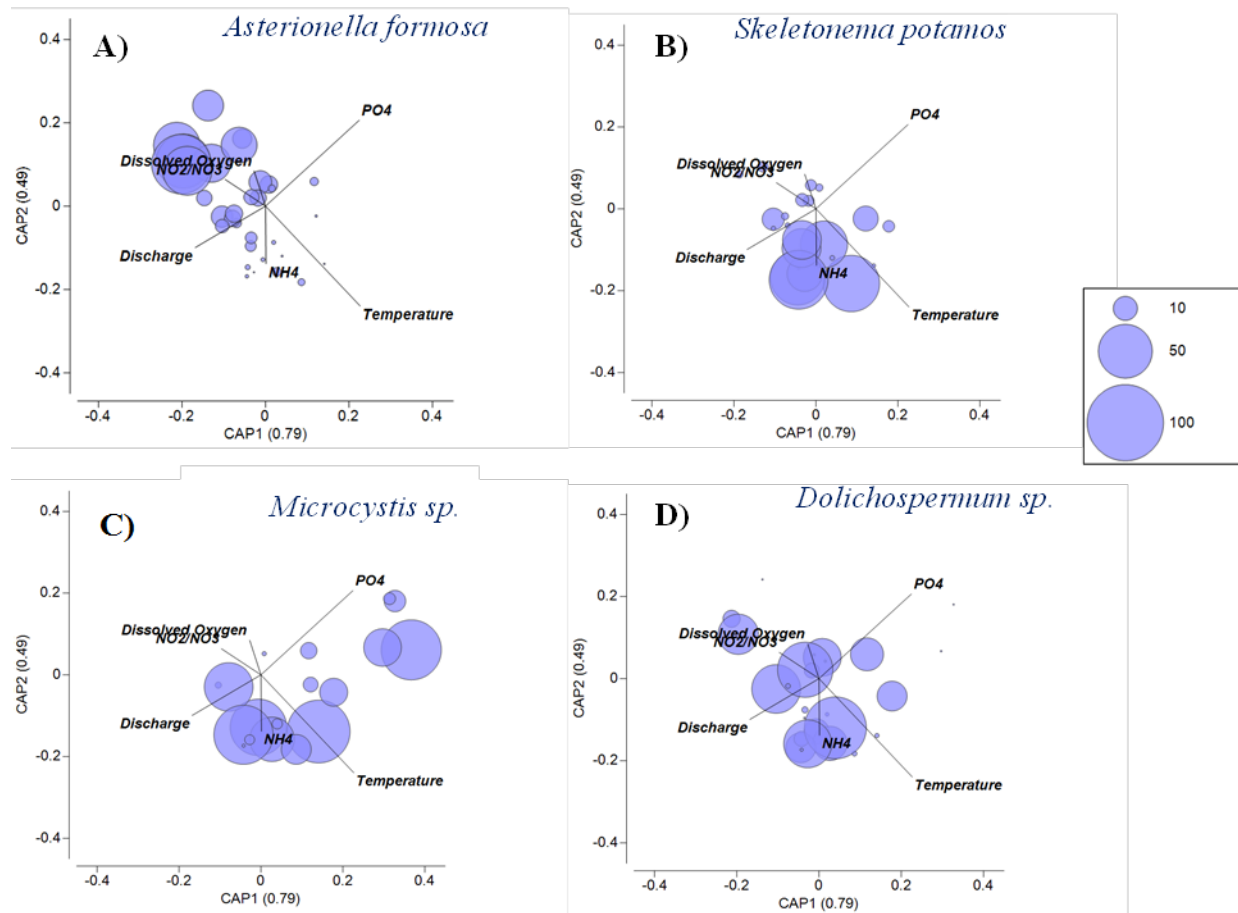


Figure 45. Bubble plots showing relative abundance of A) *Asterionella formosa*, B) *Skeletonema potamos*, C) *Microcystis* sp., and D) *Dolichospermum* sp. overlaid onto the CAP ordination diagram. Bubble sizes correlate to relative abundance (0–100%) of taxa in a sample.

3.4.3 Secondary Production

Zooplankton abundances and species composition were determined at the trend sites in 2015. Zooplankton tended to be more abundant later in the season at sites in Reaches A-C (Ilwaco, Grays River, Welch Island, and Whites Island; Figure 46). At both Campbell and Franz Lake Slough, abundances were

more variable. With a few exceptions, rotifers were numerically dominant. Rotifers are much smaller than the crustacean zooplankton (copepods and cladocerans); therefore, their contributions to organic carbon are also smaller on a per cell basis.

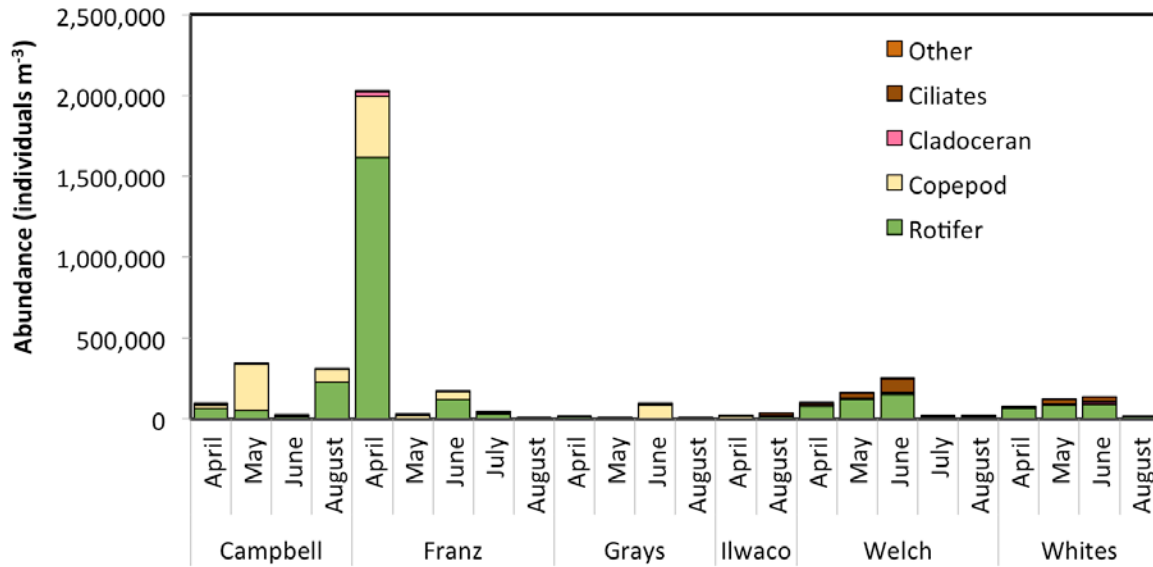


Figure 46. Zooplankton abundance at trend sites (Campbell Slough, Franz Lake Slough, Grays River, Ilwaco Slough, Welch Island, and Whites Island) in 2015.

According to percent contribution, copepods were present at higher relative abundance in May at Campbell Slough and Franz Lake Slough compared to the summer period (Figure 47). Copepods accounted for a higher percentage of total zooplankton throughout the spring and summer (May-August) at the Grays River site and in April at Ilwaco compared to other sites.

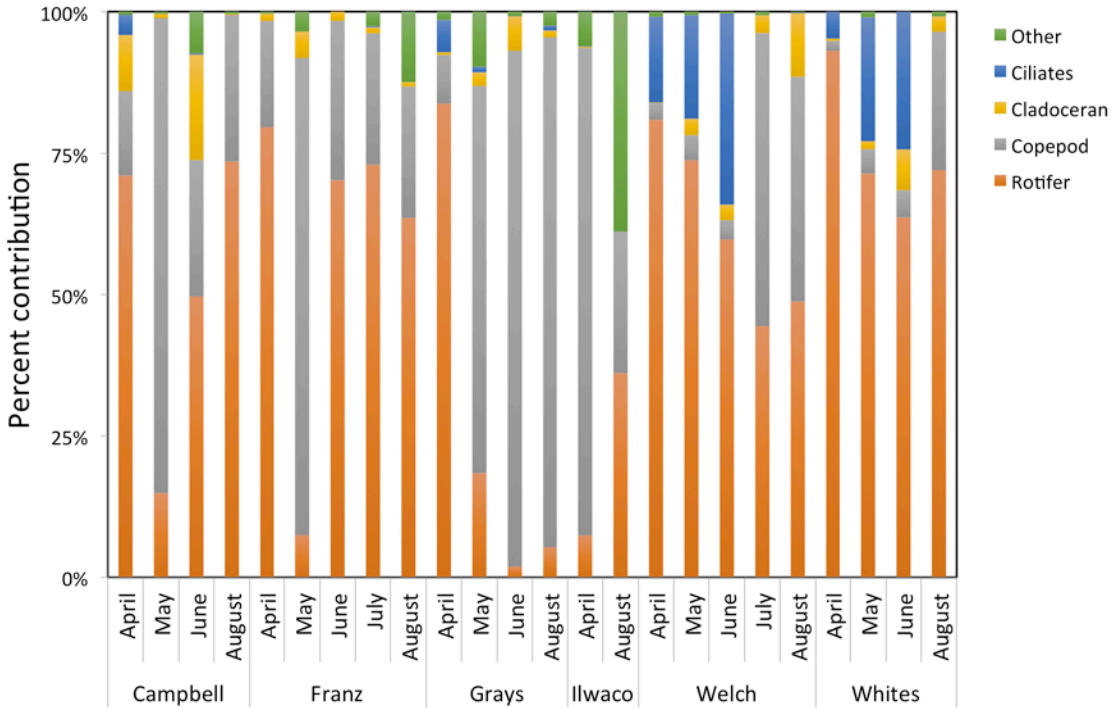


Figure 47. Percent contribution of different zooplankton taxa at the trends sites in 2015.

The percent contribution of crustacean zooplankton (copepods and cladocerans) to total zooplankton abundance varied from less than 10% to more than 90% (Figure 48), indicating high variability among the samples. The larger crustacean zooplankton contain larger amounts of organic matter relative to smaller taxa, and can provide a direct food supply to juvenile salmon (Merz 2001).

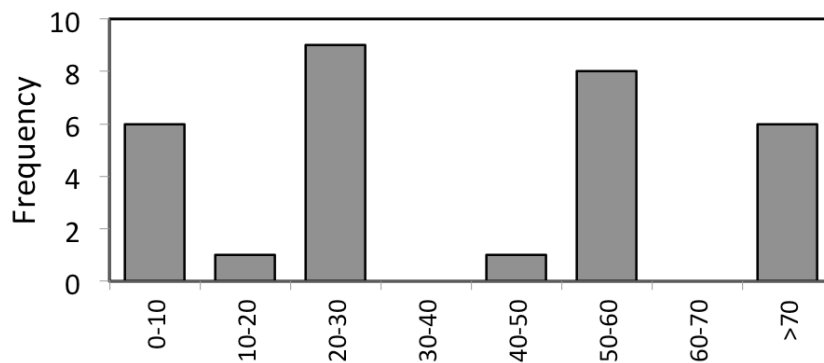


Figure 48. Frequency histogram showing the percent contributions of crustacean zooplankton to total zooplankton across all trend sites in 2015.

3.4.4 Stable Isotope Ratios

The stable isotope data from particulate matter, live and dead vegetation, and zooplankton in 2015 (Figure 49, Figure 50) have not yet been incorporated into a mixing model to explain isotopic signatures of salmon muscle (Figure 51, Figure 52) since not all the data have been received from the analytical facility. Results from the mixing model will be presented in a future report.

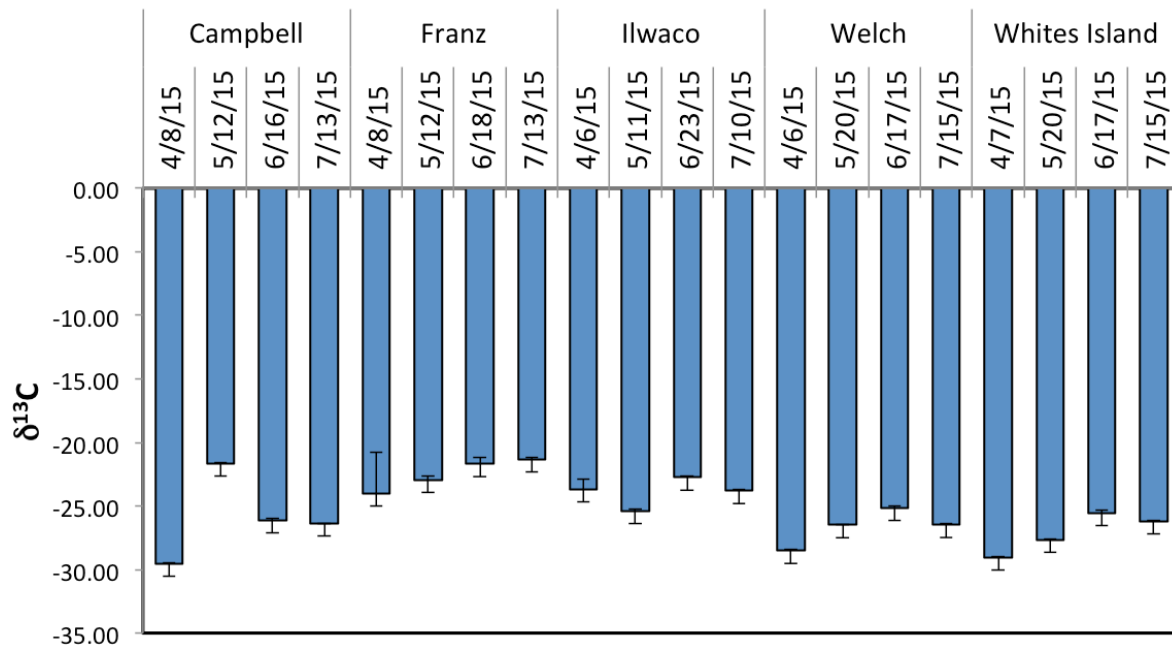


Figure 49. $\delta^{13}\text{C}$ values relative to a Pee Dee Belemnite standard for particulate matter at trends sites (Campbell Slough, Franz Lake Slough, Ilwaco, Welch Island, and Whites Island) during the months of April, May, June, and July 2015.

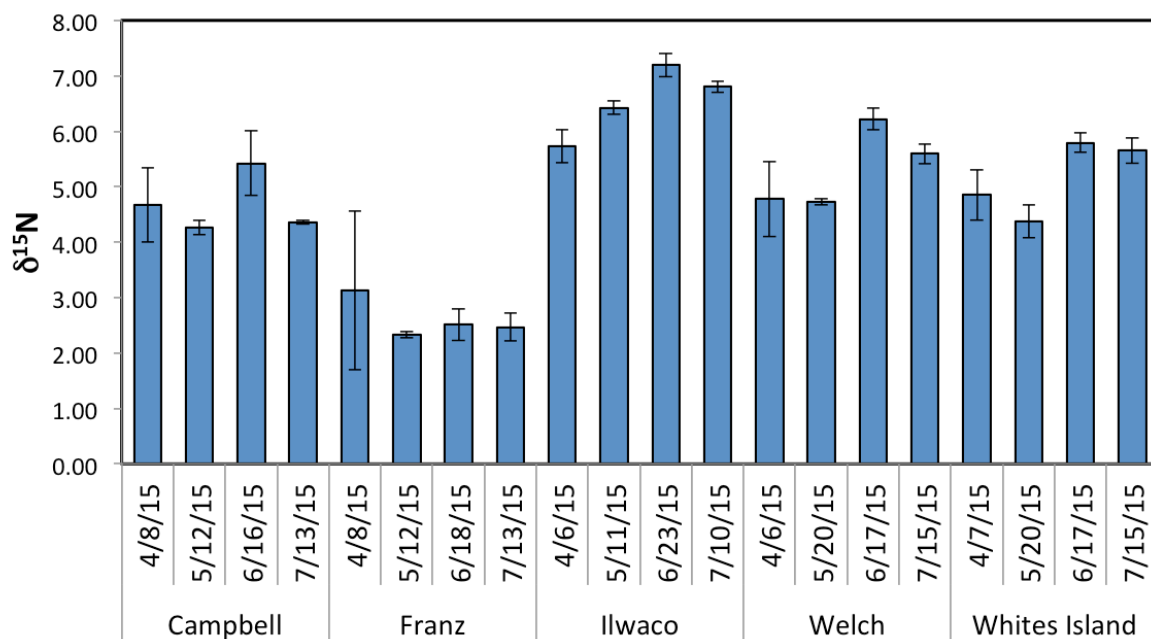


Figure 50. $\delta^{15}\text{N}$ values relative to air for particulate matter at trends sites (Campbell Slough, Franz Lake Slough, Ilwaco, Welch Island, and Whites Island) during the months of April, May, June, and July 2015).

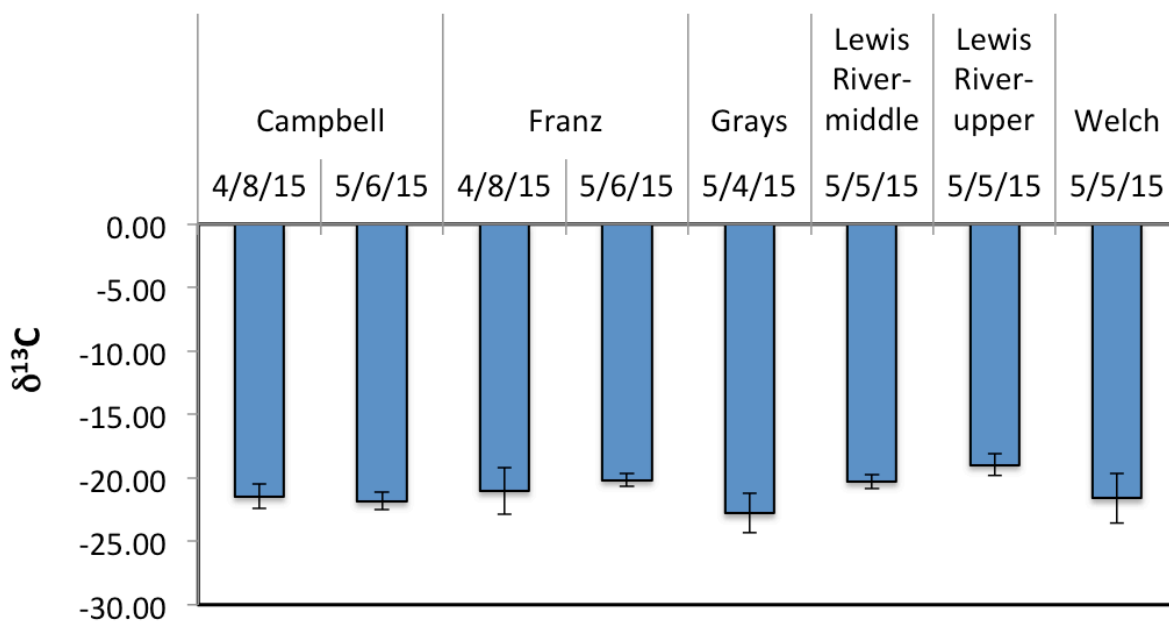


Figure 51. $\delta^{13}\text{C}$ values from salmon muscle tissue relative to a PeeDee Belemnite standard. Values are the average of replicate samples (3-10 individuals).

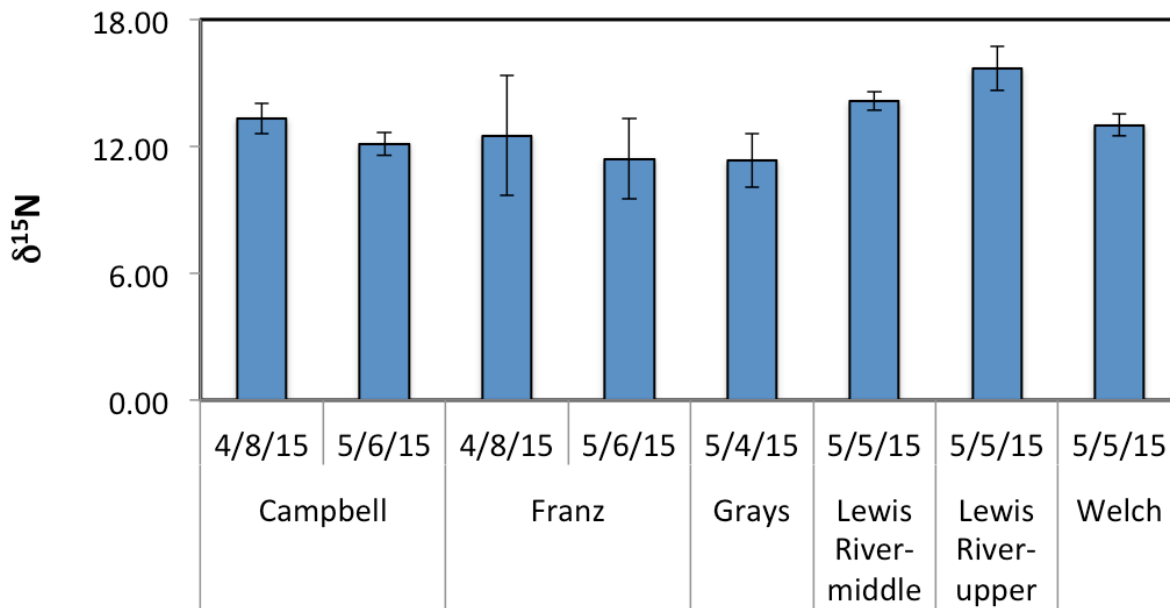


Figure 52. $\delta^{15}\text{N}$ values from salmon muscle relative to air. Values are the average of replicate samples (3-10 individuals).

3.5 Macroinvertebrates

3.5.1 Salmon Prey Availability

3.5.1.1 *Benthic*

The average monthly density of all benthic invertebrate taxa combined was greatest from Ilwaco Slough (Figure 53). Average densities tended to be similar from sites in the lower estuarine reaches B and C (Grays River, Welch Island, and Whites Island). Campbell Slough and Franz Lake had a similar decline in density in June and July. The average biomass of all benthic taxa tended to be greatest from Welch Island and Whites Island.

Dipterans occurred in benthic cores from all sites sampled (Figure 54). There was not a consistent temporal trend in densities between sites; however, biomass tended to be greatest in June or July at most sites. Dipteran density was relatively low at Franz Lake in all months, and biomass was low at both Campbell Slough and Franz Lake in all months sampled. Amphipods occurred at Ilwaco Slough in all months, while only two individuals appeared at Whites Island in May, and no amphipods occurred at any of the other sites.

Annelid worms dominated the benthic assemblages at all sites (Figure 55). The invertebrate composition at Ilwaco Slough was distinguished from other sites by the presence of amphipods and isopods (Figure 56). The invertebrate composition was fairly similar between Grays River, Welch Island, and Whites Island in having secondary contributions by Diptera, gastropods, bivalves, and nematode worms. Benthic

assemblages at Campbell Slough and Franz Lake varied by month, but both sites had greater densities of Collembola and other insect taxa in some months, compared to the more downriver sites.

Average density of all invertebrate taxa was compared to benthic samples from past years where similar months were sampled (Figure 57). Densities in 2015 were generally consistent with those from previous years. Average monthly composition was also consistent within a site across years (Figure 58).

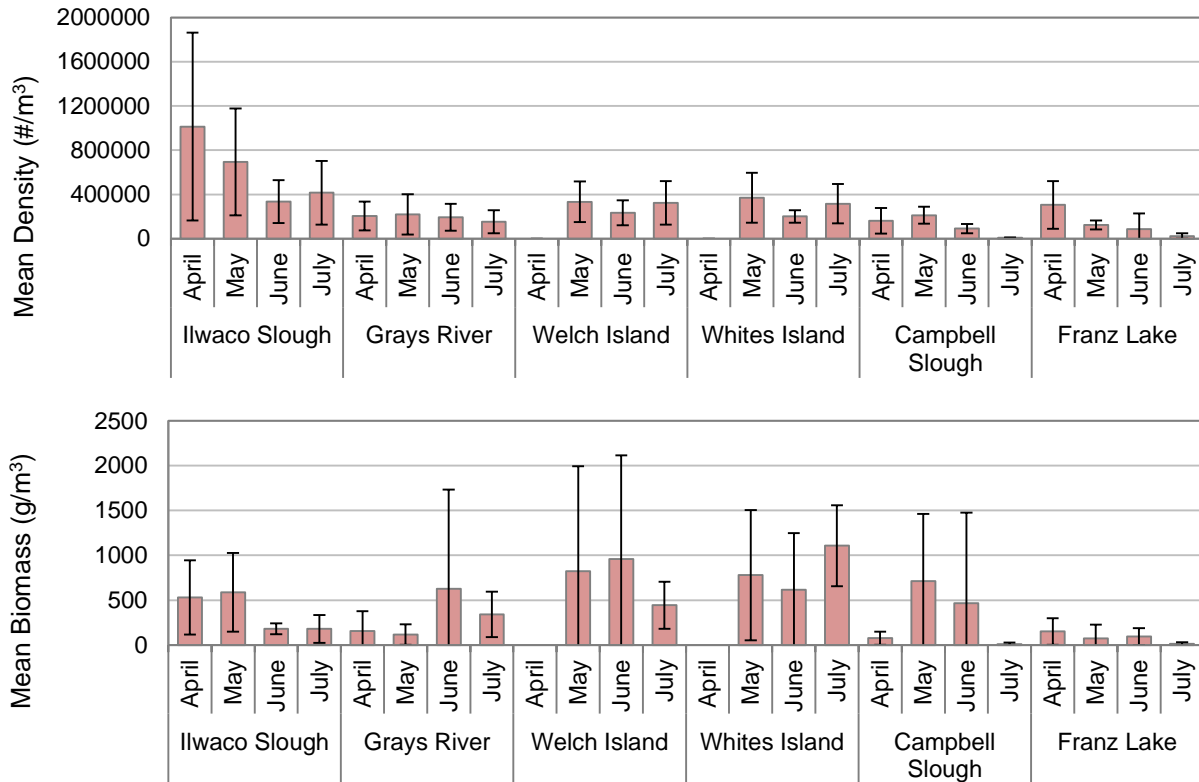
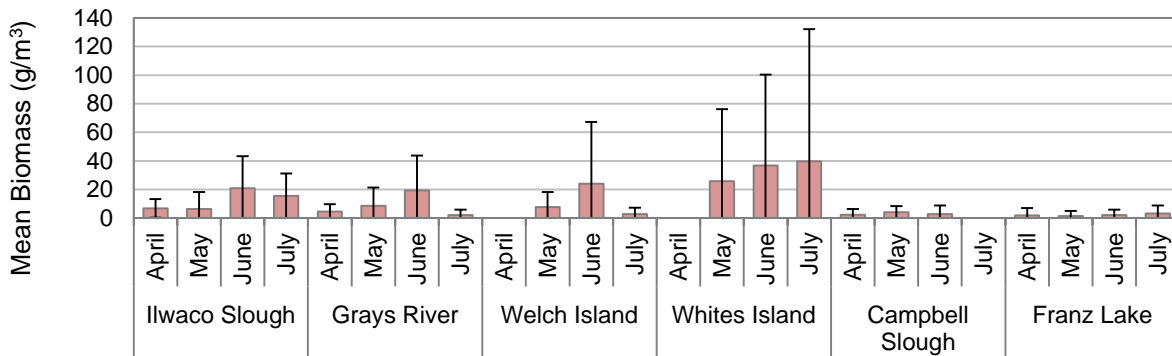
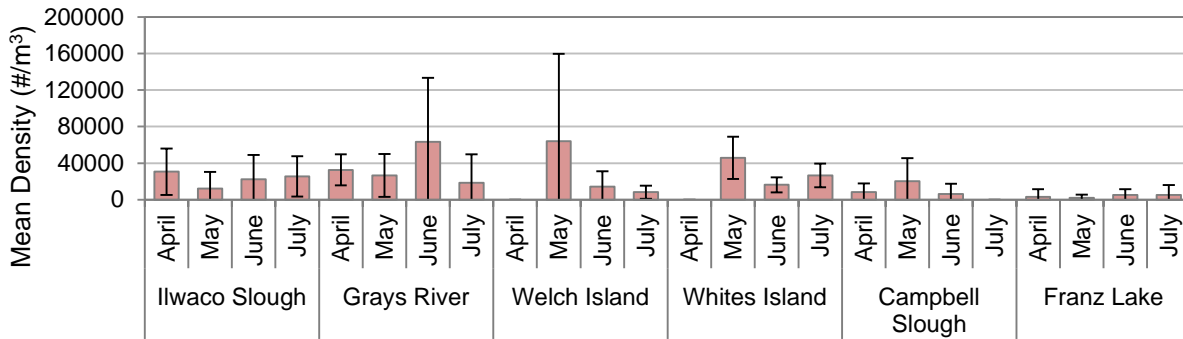


Figure 53. Average density (count/m³) and biomass (g/m³) of all benthic invertebrate taxa by sample site and month in 2015. Error bars represent 95% confidence intervals. Benthic cores were not collected from Welch Island and Whites Island in April.

Diptera



Amphipoda

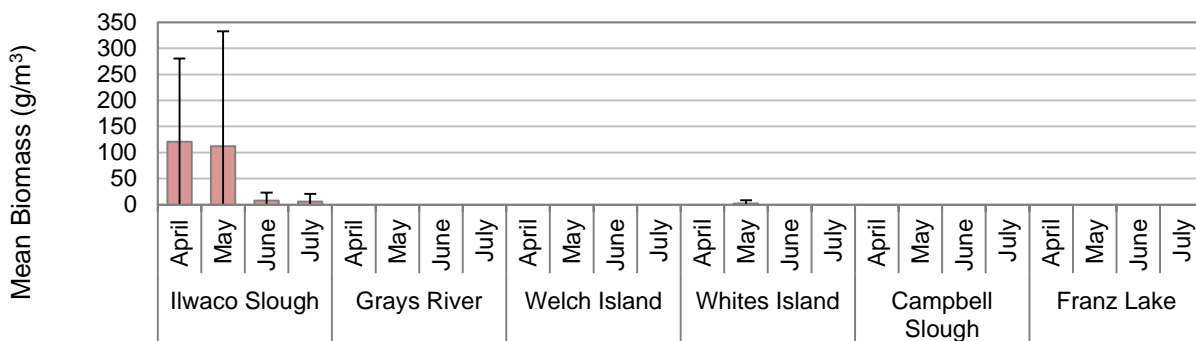
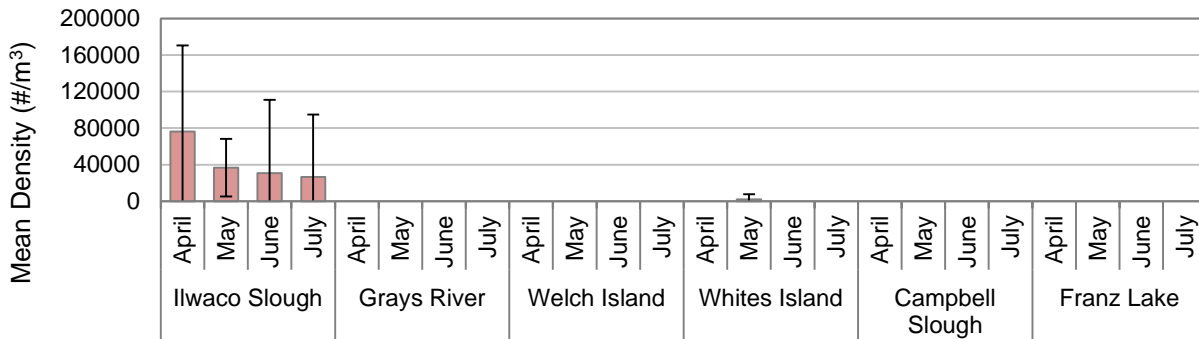


Figure 54. Average density (count/m³) and biomass (g/m³) of immature Diptera (including Chironomidae) and Amphipoda by sample site and month in 2015. Error bars represent 95% confidence intervals. Benthic cores were not collected from Welch Island and Whites Island in April.

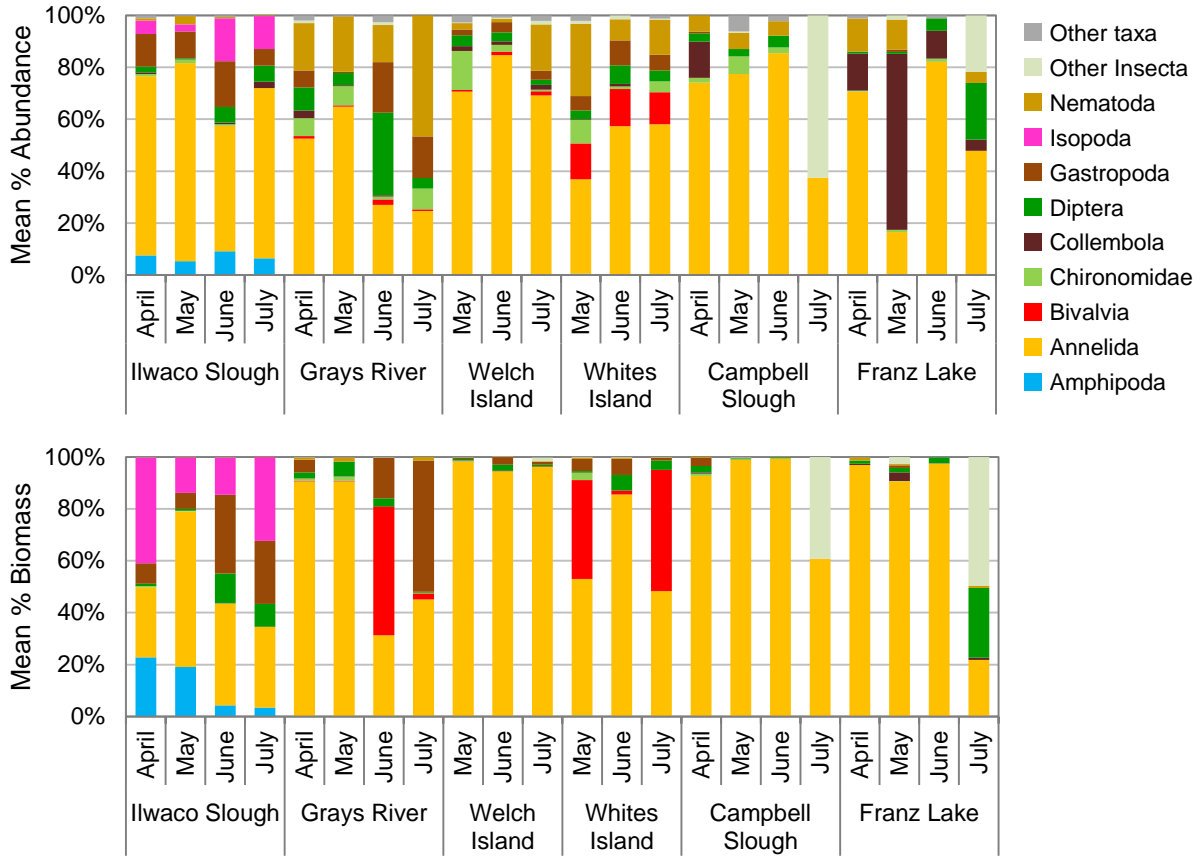


Figure 55. Average composition of benthic invertebrate taxa at trends sites during a sample month. Average percent abundance (top) and average percent biomass (bottom) for each taxonomic group in 2015 are shown.

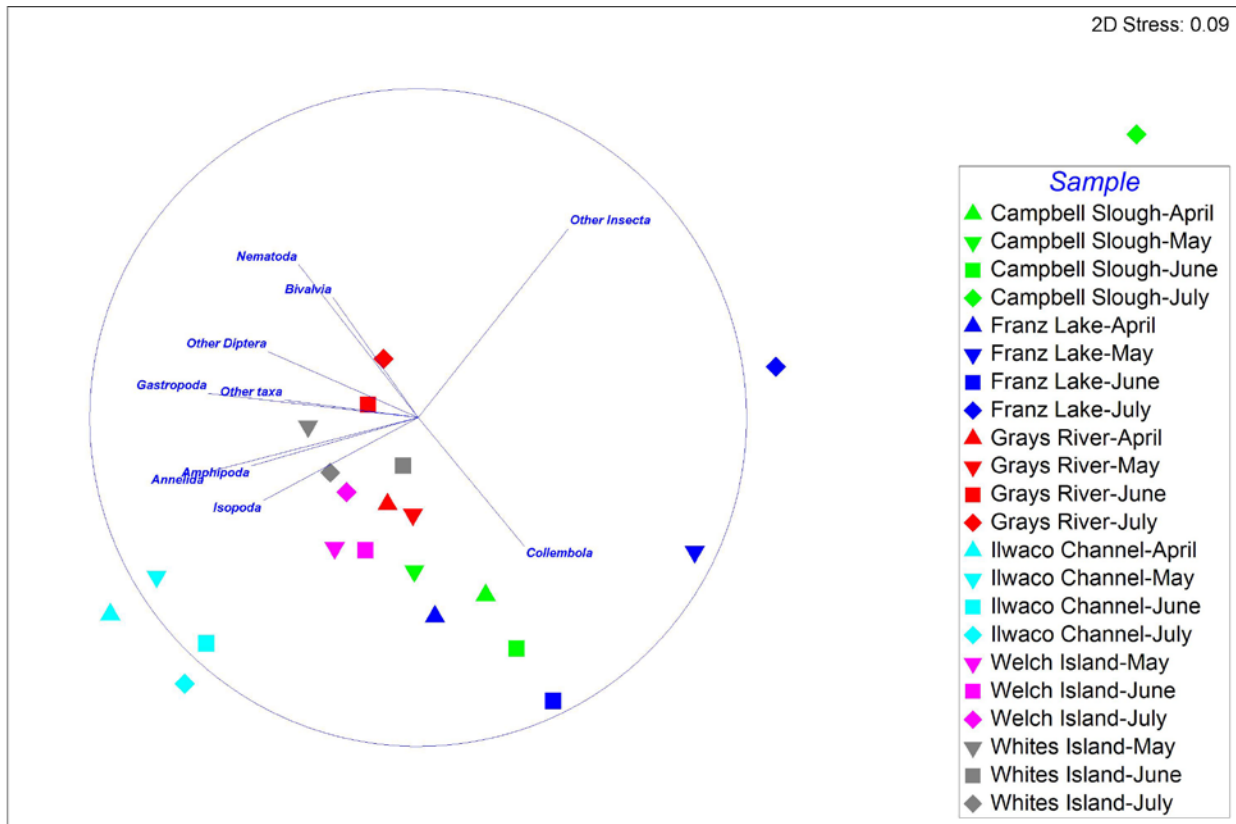


Figure 56. Two-dimensional nonmetric multidimensional scaling (MDS) plot based on Bray-Curtis similarities between monthly densities of benthic taxa in 2015. Significant correlation with variables (Pearson $R > 0.4$) are represented as blue vectors.

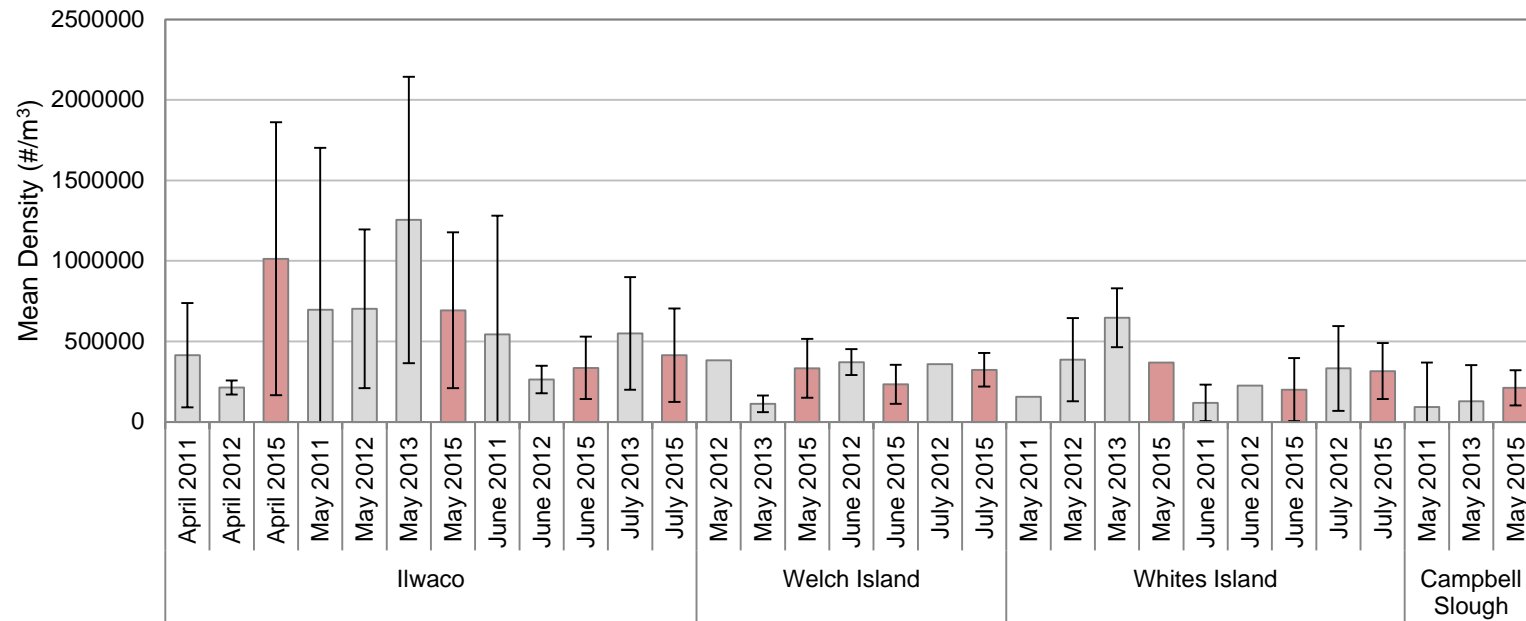


Figure 57. Comparisons across years of average density (count/m³) of all benthic invertebrate taxa by sample site and month. Red bars highlight samples from 2015. Error bars represent 95% confidence intervals.

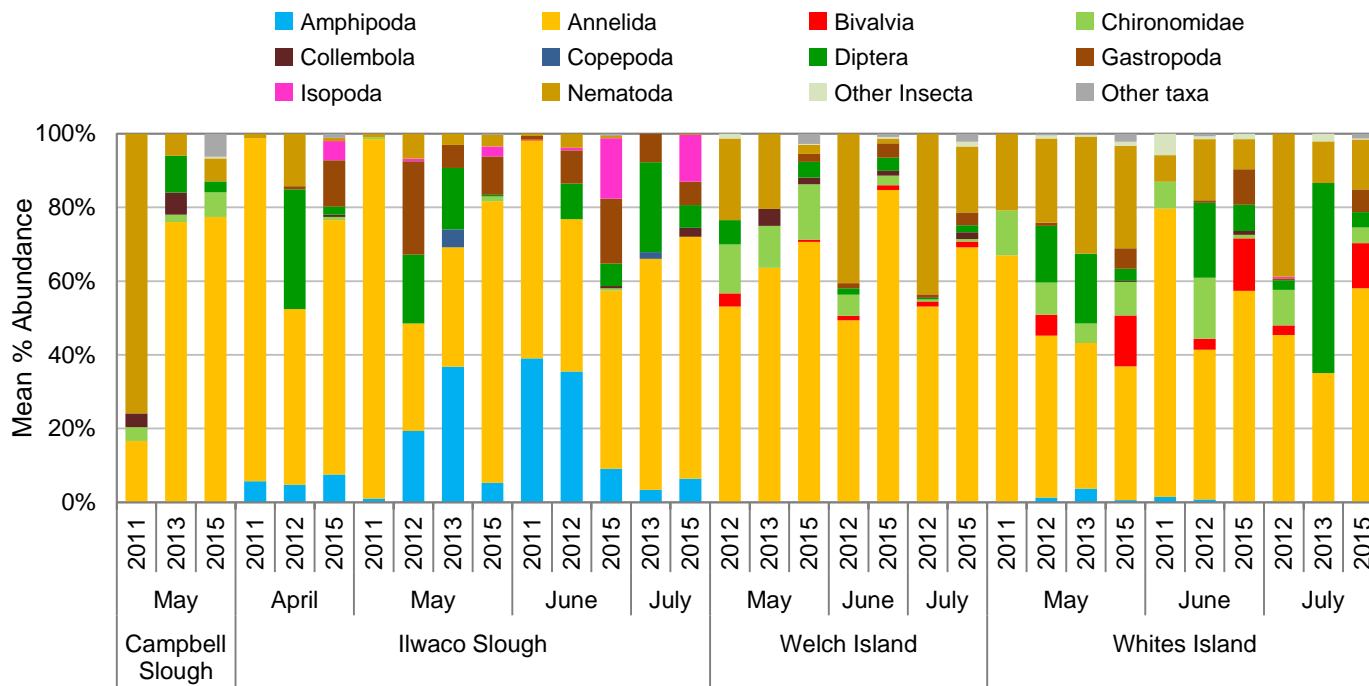


Figure 58. Comparison of average percent abundance of benthic invertebrate taxa at trends sites across sampling years.

3.5.1.2 *Neuston*

The average density of all invertebrate taxa collected by neuston tows was greater in emergent vegetation habitats than in open water habitats, except from Whites Island in June when average density was 1.75 times greater in open water compared to emergent vegetation (Figure 59). April densities were very high in the emergent vegetation at Campbell Slough and Franz Lake. At that time, average density was 59.9 and 84.3 times greater in the emergent vegetation than in the open water at Campbell Slough and Franz Lake, respectively. Otherwise, average density ranged between 1.28 times and 4.63 times greater in the emergent vegetation than in the open water.

The average biomass of all invertebrate taxa was consistently greater in emergent vegetation habitats than in open water habitats (Figure 59). Average biomass ranged between 8.73 and 370.56 times greater in the emergent vegetation than in the open water. Average biomass in the emergent vegetation was high from all sites sampled in April compared to other monthly samples; however, comparison between months was not possible for the Ilwaco Slough or Campbell Slough sites. Biomass was generally low in the open water, ranging between 0.2 and 2.9 mg per meter towed. Neuston was not sampled from the emergent vegetation at Welch Island or Campbell Slough in May; therefore, comparisons between habitat types were not possible for these samples.

Diptera, including Chironomidae, occurred at all of the sites on each date sampled. The average density and biomass of Diptera was consistently greater in emergent vegetation habitats than in open water habitats (Figure 60). Average Diptera density ranged between 14.82 and 178.38 times greater in the emergent vegetation than in the open water. Average Diptera biomass similarly ranged between 13.21 and 150.11 times greater in the emergent vegetation than in the open water. Diptera density was greatest from Franz Lake in April, when an average of 44 individuals were collected per meter towed, representing 10 percent of the total invertebrate abundance. Diptera were also abundant from Whites Island in May, representing 73 percent of the total invertebrate density and biomass. Diptera density was moderate (4.1 individuals per meter towed) from the open water at Campbell Slough in May; however, the emergent vegetation was not sampled and therefore comparison between habitat types was not possible.

Amphipods were most abundant in neuston tows from Ilwaco Slough in April, where average density and biomass were greater in the emergent vegetation than in the open water (Figure 61). Amphipods were also observed at other sites, but in lower abundances than at Ilwaco Slough.

The composition of neuston samples varied among sites, months, and habitats (Figure 62). In general, Ilwaco Slough had a greater proportion of amphipods, isopods, and annelid and nematode worms than the other sites. Major taxa contributing to abundances in the neuston assemblage at the other sites included cladocerans, copepods, and chironomids. Although not numerically abundant, the large body size of amphipods, Aranae (spiders), gastropods, Hemiptera, and other insects (particularly caddisflies and beetles) made a relatively large contribution to the proportional biomass when they were present. The open water had a higher average proportional abundance and biomass of cladocerans and amphipods, while isopods, insect taxa, and Collembola accounted for greater proportions of the emergent vegetation composition. The high counts of invertebrates in the emergent vegetation at Campbell Slough and Franz Lake in April were not dominated by the same major taxa. At that time, Campbell Slough was primarily comprised of cladocerans, nematode worms, ostracods, and copepods. Franz Lake was strongly dominated by copepods, though chironomids were also abundant.

The MDS plot distributes observations of the proportional abundance of invertebrate taxa to examine which neuston samples were similar in community composition (Figure 63). Ilwaco Slough samples are distinguished from the other sites primarily by higher proportions of amphipods and annelid and nematode worms. There appears to be some separation of Welch Island and Campbell Slough from

Whites Island and Franz Lake. Welch Island and Campbell Slough samples tended toward higher proportions of cladocerans and nematode worms, while Whites Island and Franz Lake were characterized by a greater presence of copepods and insect taxa (including chironomids and other Diptera, as well as Hemiptera). Whites Island was the only site where the invertebrate composition in emergent vegetation samples was distinct from that in open water samples.

Average densities of all invertebrate taxa were compared to neuston samples from past years where similar months and habitats were sampled (Figure 64). Densities from 2015 were consistently greater than those previously recorded from both the emergent vegetation and open water. Average monthly composition varied across years (Figure 65). For example, Diptera (including chironomids) tended to comprise a smaller portion of the open water assemblage in 2015 compared to previous years. The percent of Diptera abundance was also lower from the emergent vegetation at Welch Island in 2015; however, the percent abundance of Diptera was similar across years from the emergent vegetation at Whites Island and Franz Lake. Cladocerans comprised a greater proportion of the open water assemblage at Welch Island, Whites Island, and Franz Lake in 2015 compared to previous years.

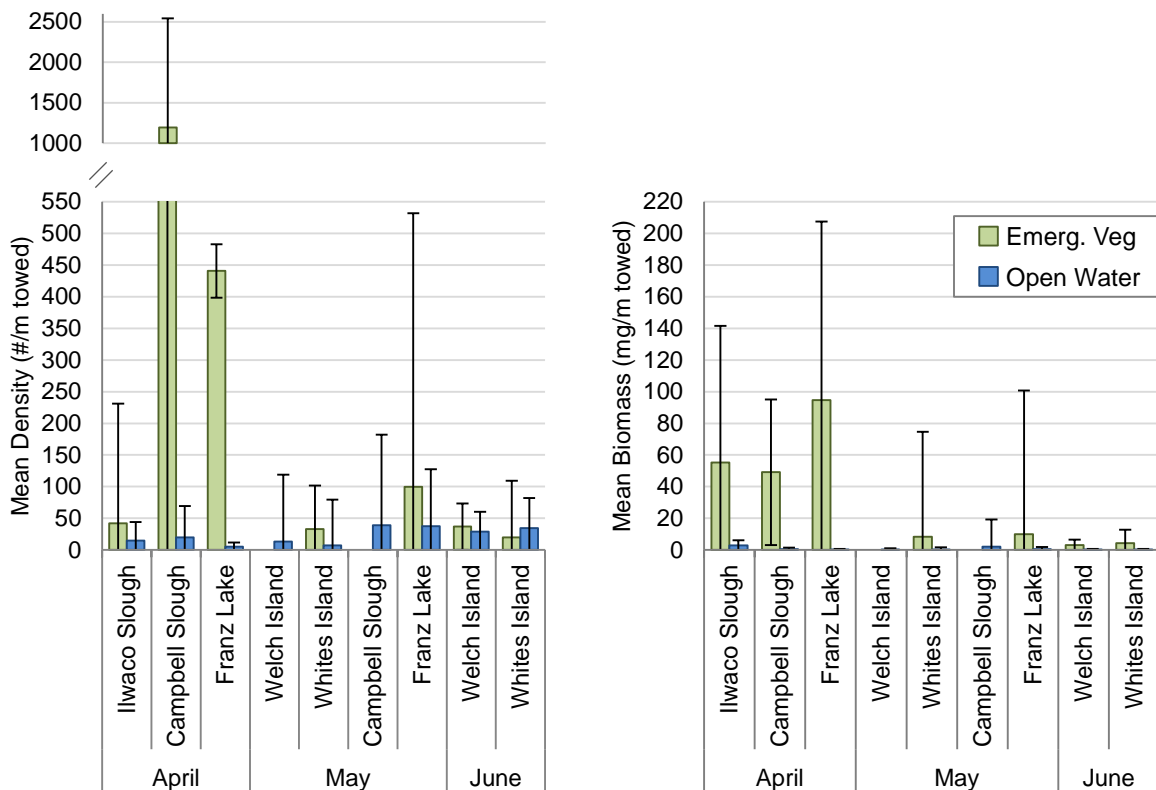


Figure 59. Average density (left) and biomass (right) of all invertebrate taxa collected by neuston tow in emergent vegetation (green) and open water (blue) in 2015. Error bars represent 95% confidence intervals. Emergent vegetation was not sampled in May at Welch Island and Campbell Slough.

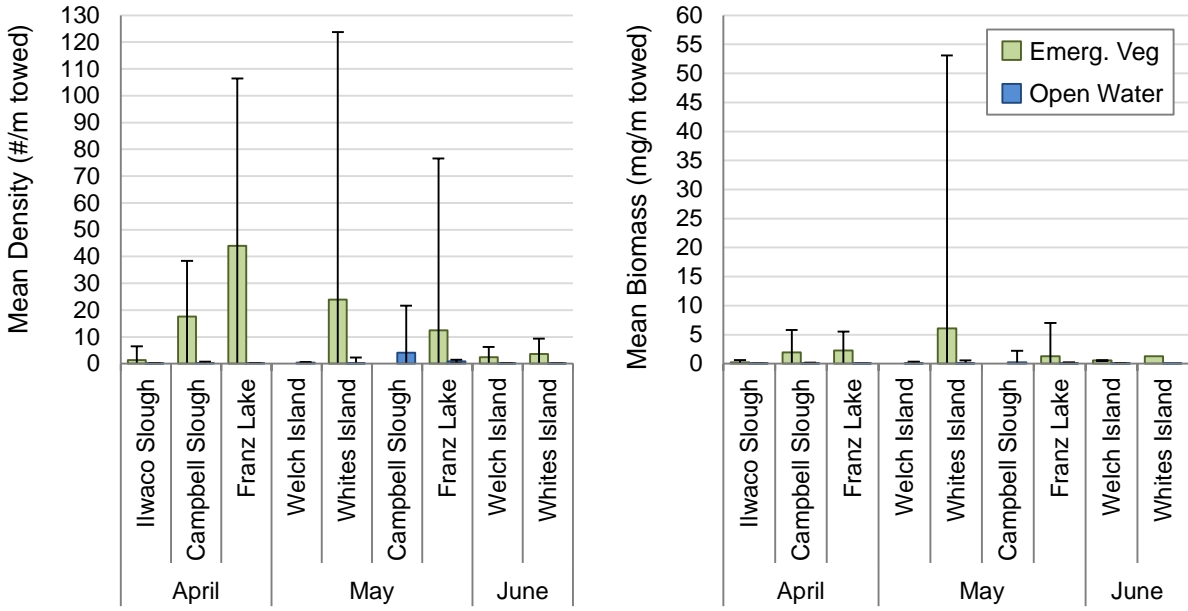


Figure 60. Average density (left) and biomass (right) of Diptera (including Chironomidae) taxa collected by neuston tow in emergent vegetation (green) and open water (blue) in 2015. Error bars represent 95% confidence intervals. Emergent vegetation was not sampled in May at Welch Island and Campbell Slough.

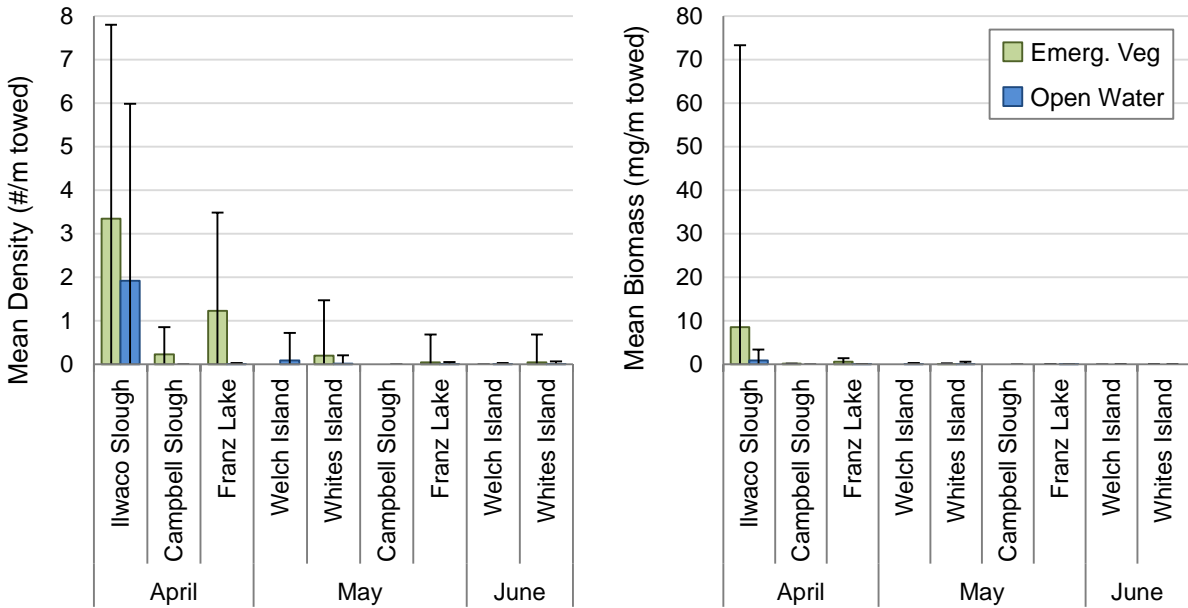


Figure 61. Average density (left) and biomass (right) of amphipoda collected by neuston tow in emergent vegetation (green) and open water (blue) in 2015. Error bars represent 95% confidence intervals. Emergent vegetation was not sampled in May at Welch Island and Campbell Slough.

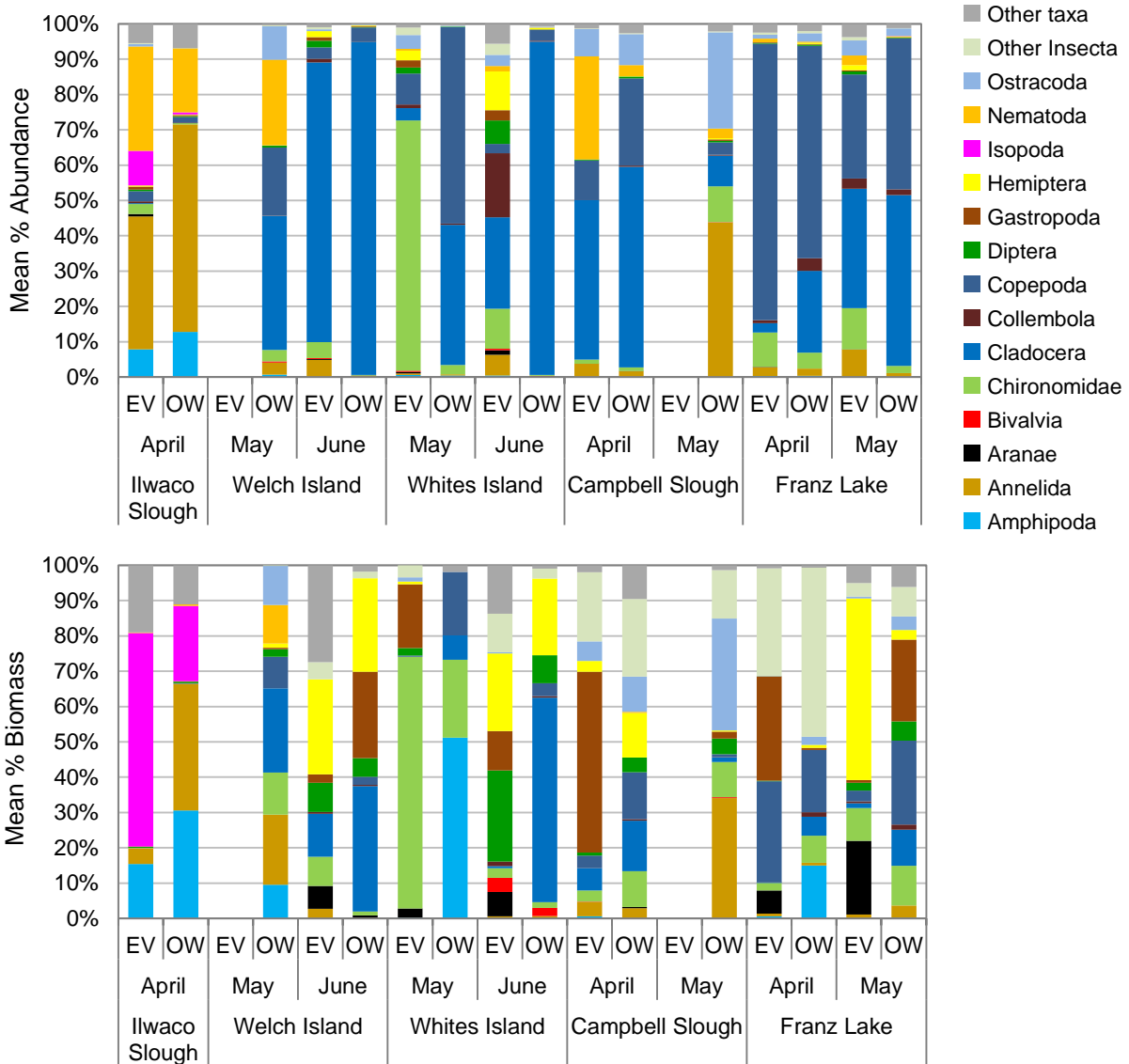


Figure 62. Average composition of neuston tows in emergent vegetation (EV) and open water (OW) during a sample month, by average percent abundance (top) and average percent biomass (bottom) for each taxonomic group in 2015.

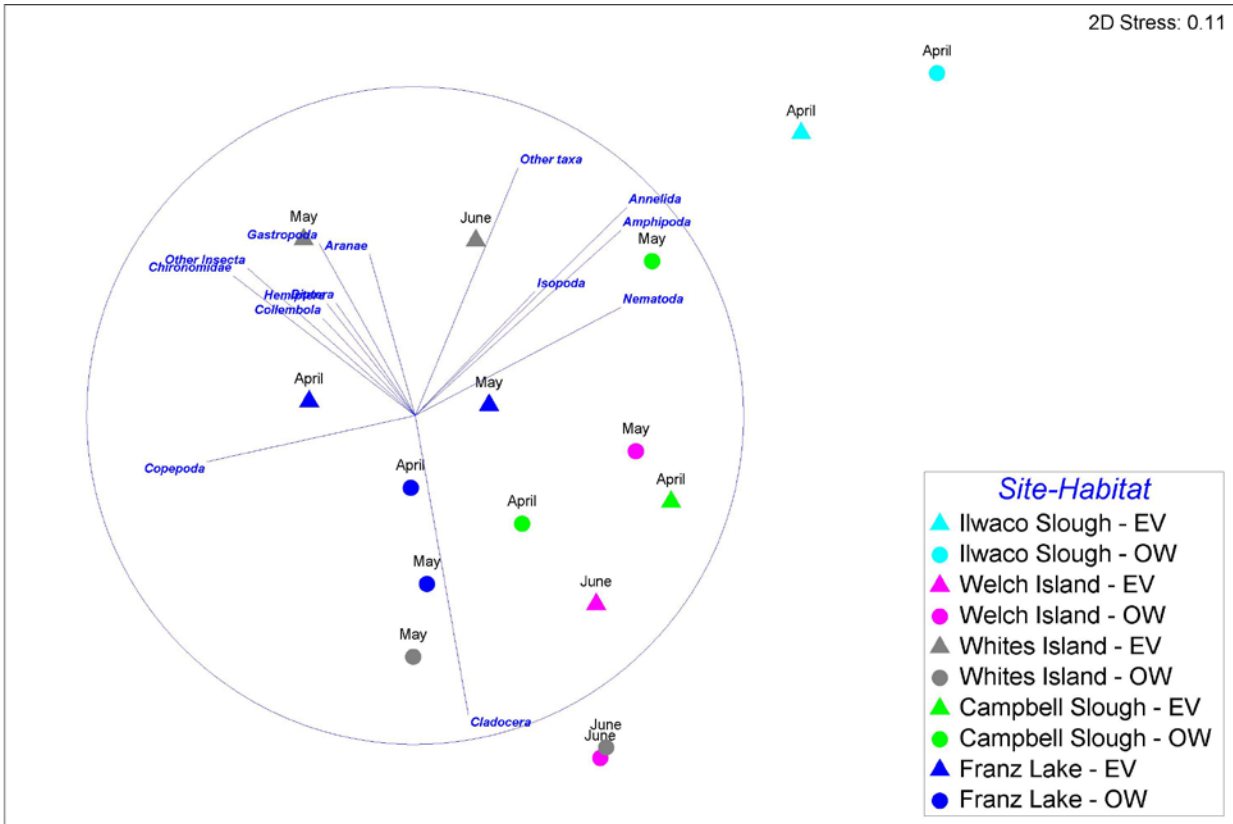


Figure 63. Two-dimensional nonmetric multidimensional scaling (MDS) plot based on Bray-Curtis similarities between samples of neuston tow taxa average proportional abundance in 2015. Significant correlation with variables (Pearson R > 0.4) are represented as blue vectors.

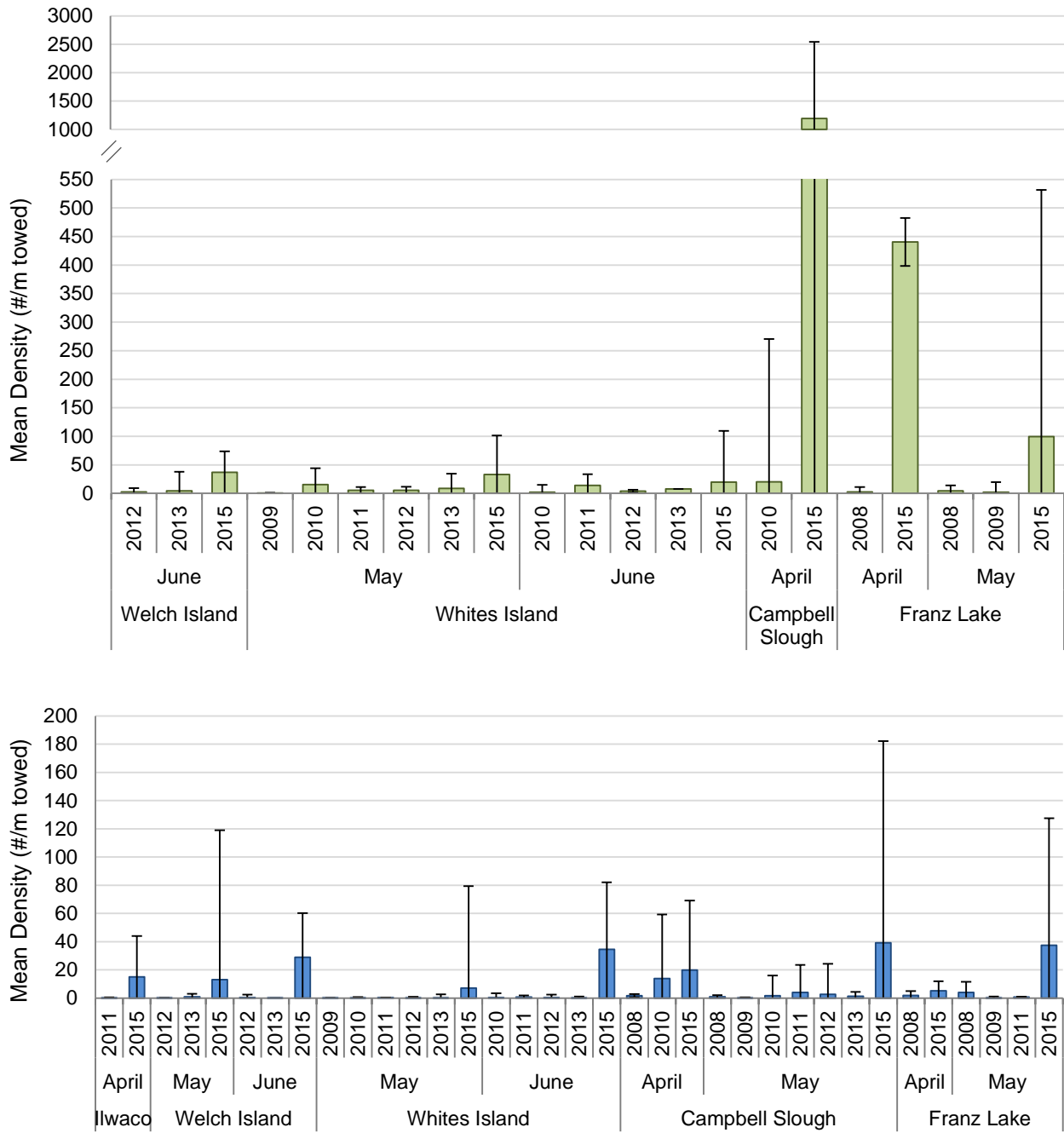


Figure 64. Comparisons across years of average density per meter towed of all invertebrate taxa collected by neuston tow in emergent vegetation (green) and open water (blue). Error bars represent 95% confidence intervals.



Figure 65. Comparisons across years of average percent abundance of neuston invertebrate taxa collected in open water (OW) and emergent vegetation (EV).

3.5.2 Salmon Diet

The dominant prey items for juvenile Chinook salmon were amphipods and emergent/adult Chironomidae (Figure 66). Amphipods comprised almost all of the prey consumed from Ilwaco Slough samples taken in April. Diets from Welch and Whites Island both demonstrated a shift away from amphipods from May to June to a higher % IRI of chironomids and other insect taxa, including Hemipterans, which corresponded to their increase in the neuston assemblage at this time. Diets collected from Campbell Slough and Franz Lake in May were dominated by chironomids and other Diptera taxa. Instantaneous ration values were on average highest from Ilwaco Slough (mean IR = 0.015, 95% CI = 0.0045) and lowest from Franz Lake (mean IR = 0.0021, 95% CI = 0.0026).

Electivity values for 2015 indicate juvenile Chinook salmon consumed amphipods in proportions higher than would be expected given their abundance in samples taken from habitats at all sites (Figure 67). Insect taxa, were relatively rare in neuston samples from Ilwaco Slough, but were avoided altogether by juvenile salmon collected from this site and thus received negative electivity values. Chironomids were positively selected for by fish from Welch Island, Campbell Slough, and Franz Lake. While chironomids were consumed at Whites Island in May they were abundant in the environment at that time, which resulted in a low electivity value.

Comparison of electivity values from four sites sampled between 2008 and 2015 indicates a consistent high selection rate of amphipods by juvenile Chinook salmon at Welch and Whites Island (Figure 68). At these sites there were several instances in which amphipods occurred in diets but did not occur in neuston samples (represented by an electivity value of 1). Electivity results were not as consistent at Campbell Slough and Franz Lake, where negative selection occurred in several years, including several instances in which amphipods did not occur at all in diets (represented by an electivity value of -1). While contribution of amphipods to the % IRI of fish from Campbell Slough and Franz Lake in 2015 was minimal, their very low presence in the environment resulted in a high selection rate indicated by electivity values of or near 1. For dipterans, electivity values tended to be positive except at Whites Island, which had the largest incidence of negative electivity values. In most cases, electivity for dipterans (including chironomids) was higher in 2015 than in previous years.

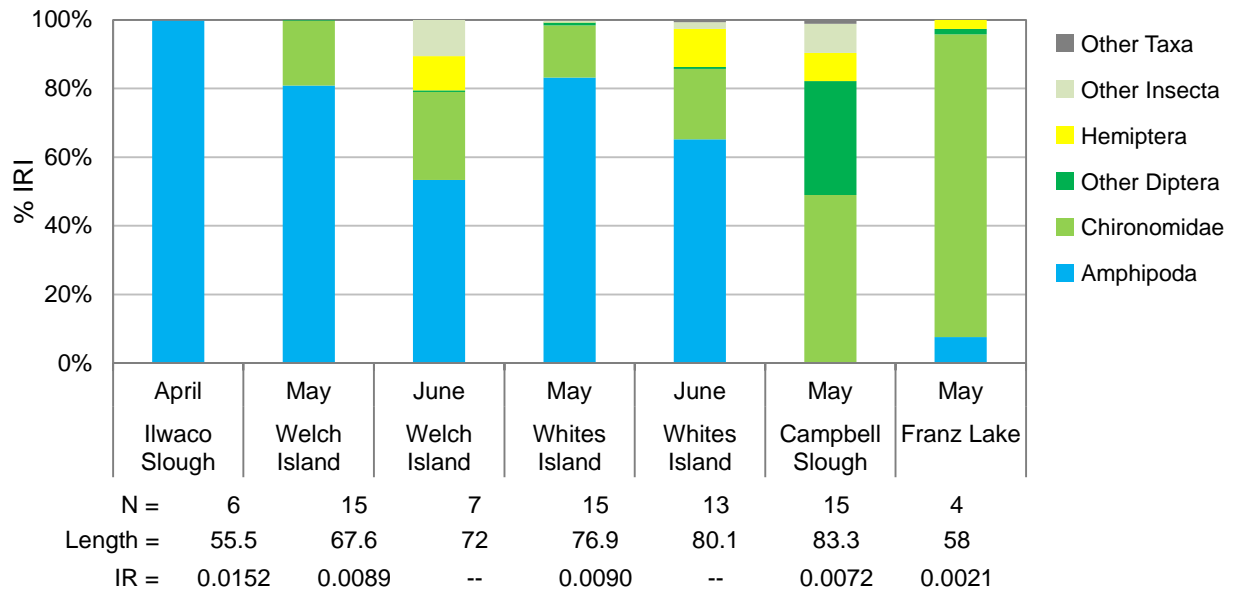


Figure 66. Contribution of prey to juvenile Chinook diets by % IRI. Number of fish sampled, average fish fork length (mm), and average instantaneous ration (IR) are listed below each bar.

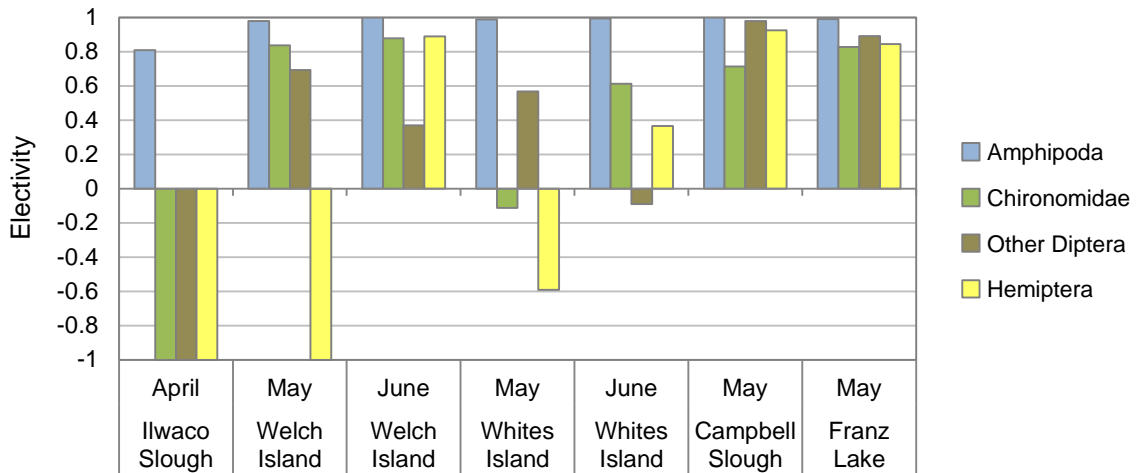
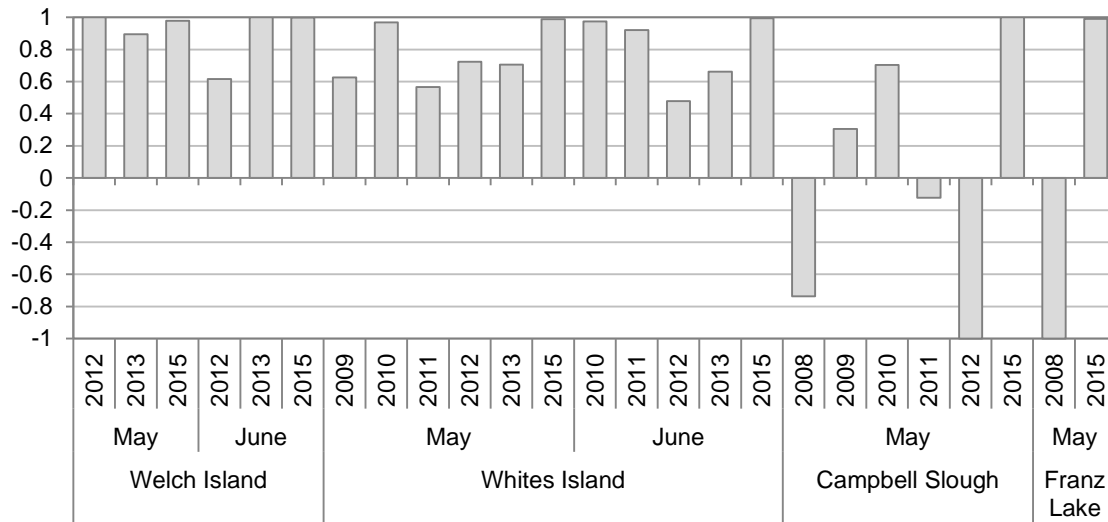


Figure 67. Ivlev's electivity values of selected prey taxa for juvenile Chinook collected in 2015.

Electivity of Amphipoda



Electivity of Diptera

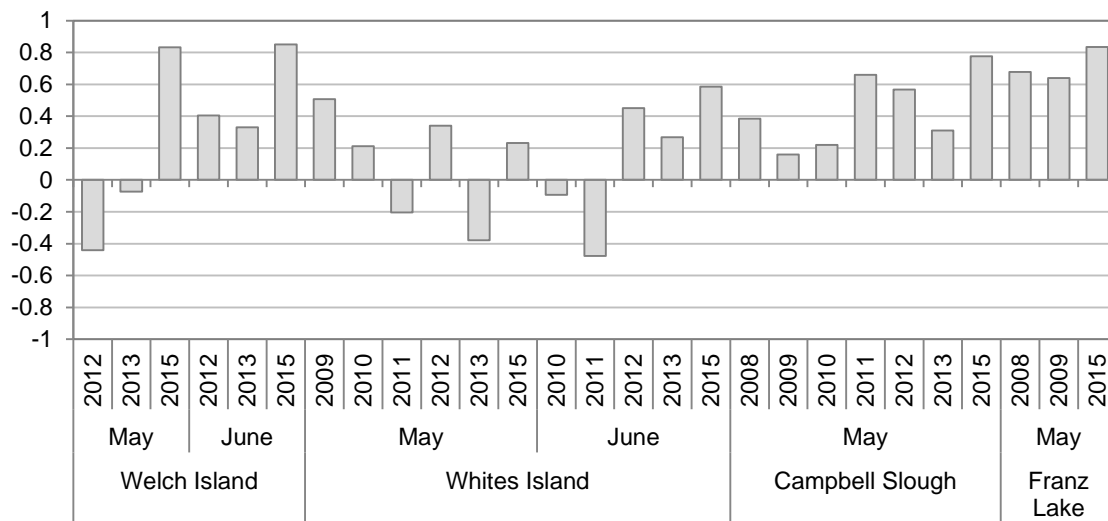


Figure 68. Comparison of Ivlev’s electivity values of Amphipoda and all Diptera for juvenile Chinook salmon collected from sites sampled between 2008 and 2015.

3.6 Fish

3.6.1 Fish Community Composition

In 2015, fish communities at Ilwaco Slough in Reach A, Welch Island in Reach B and Whites Island in Reach C were dominated by three-spined stickleback (*Gasterosteus aculeatus*), which accounted for 72% of the total catch at Ilwaco Slough, 98% of the catch at Welch Island, and 95% of the catch at Whites Island (Figure 69). Other species present at Ilwaco Slough included banded killifish (*Fundulus diaphanous*), shiner perch (*Cymatogaster aggregata*), and staghorn sculpin (*Leptocottus armatus*), the latter two of which are saltwater species not found at other sites. Each of these species accounted for 8-

11% of the total catch. Salmonids were present but only in very small numbers. At Welch Island and Whites Island, species present that accounted for more than 1% of the total catch included juvenile salmonids and chiselmouth (*Acrocheilus alutaceus*). The species assemblages at Campbell Slough in Reach F and Franz Lake in Reach H were much more diverse. Stickleback, while abundant, were less dominant than at the lower river sites, accounting for 27% of the total catch at Campbell Slough and 4% at Franz Lake (Figure 69). In addition to stickleback, other prominent species at Campbell Slough, in order of abundance, included carp (*Cyprinus carpio*), banded killifish, tui chub (*Gila bicolor*), yellow perch (*Perca flavescens*), juvenile salmonids, pumpkinseed (*Lepomis gibbosus*), and yellow bullhead (*Ameiurus natalis*). At Franz Lake, a variety of species in addition to stickleback were present including, in order of abundance, chiselmouth, banded killifish, largescale sucker (*Catostomus macrocheilus*), pumpkinseed, northern pikeminnow (*Ptychocheilus oregonensis*), smallmouth bass (*Micropterus dolomieu*) carp, juvenile salmonids, peamouth (*Mylocheilus caurinus*), and bluegill (*Lepomis macrochirus*). At all sites, fish community composition in the 2015 sampling was generally comparable to previous monitoring results from 2007-2014 (Figure 69). The dominance of stickleback at Welch Island and Whites Island observed in 2015 was consistent with earlier findings, as was the wider range of species present at Campbell Slough and Franz Lake; although specific species presence and species proportions differed somewhat from year to year.

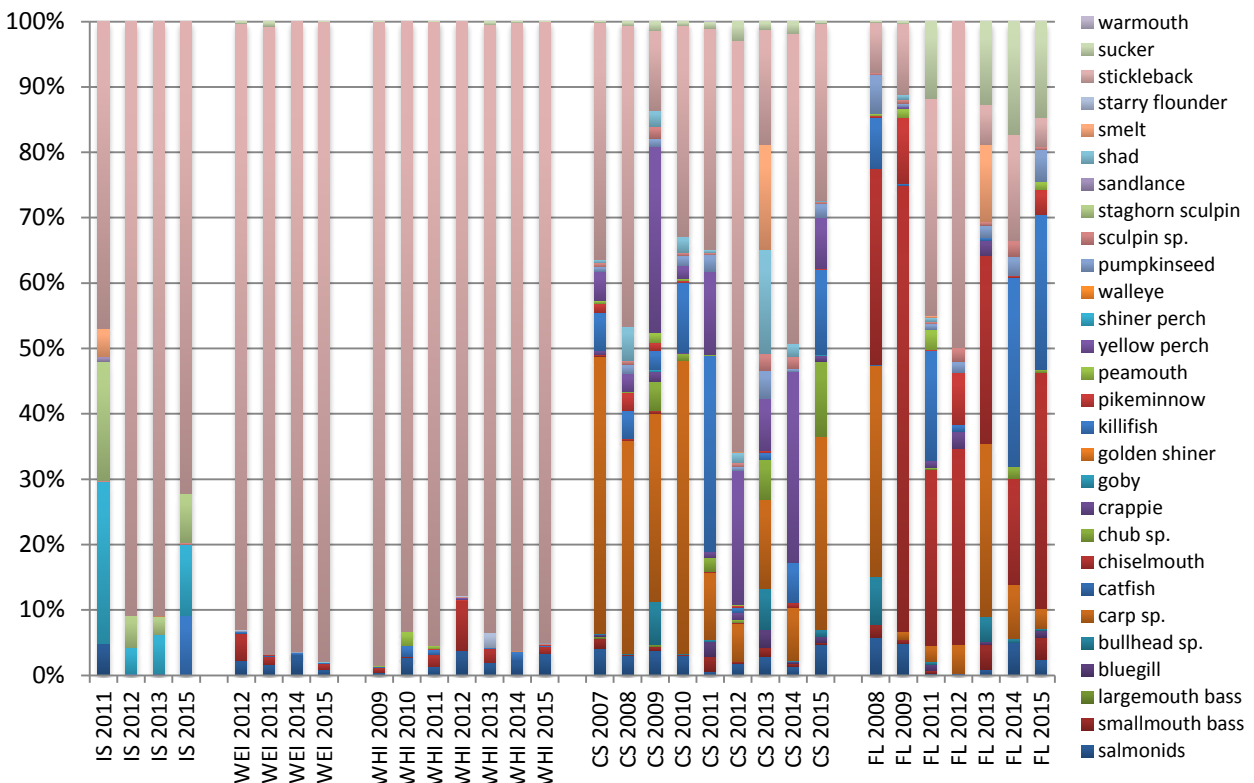


Figure 69. Fish community composition at the five EMP trends sites sampled in 2007-2015. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

In 2015, as in previous sampling years, significant differences were found in species richness (ANOVA, $p = 0.0073$) and species diversity ($p = 0.0004$), among the trends sites (Figure 70). Mean diversity was significantly higher at Franz Lake than at Campbell Slough, Welch Island, Whites Island, or Ilwaco Slough, and higher at Campbell Slough than at Welch Island and Whites Island ($p < 0.05$). Species richness was significantly higher at Campbell Slough and Franz Lake than at Ilwaco Slough, Welch

Island or Whites Island ($p < 0.05$). For individual sites, there were no significant differences in species richness or species diversity by year.

In 2015, non-native fish species made up only a small percentage of the catches at Welch Island and Whites Island (0.2% and 0.3%, respectively). The percentages of non-native species in catches were substantially higher at Ilwaco Slough (9%), Campbell Slough (56%), and Franz Lake (36%; Figure 71). Predatory fish species known to feed on juvenile salmon, such as largemouth bass (*Micropterus salmoides*), smallmouth bass, northern pikeminnow, and walleye (*Sander vitreus*) were absent at Ilwaco Slough and Welch Island (Figure 71), and made up only small percentages of the catches at Whites Island (0.2%) and Campbell Slough (0.4%). They were somewhat more abundant at Franz Lake, where they made up 6.8% of the catch. At Welch Island, Whites Island, Campbell Slough, and Franz Lake, the percentages of non-native fish species observed in 2015 were generally comparable to percentages observed in previous years. However at Ilwaco Slough, the percentage was high compared to previous years because of greater number of banded killifish caught at the site in 2015.

At Ilwaco Slough, Whites Island, and Welch Island, the percentages of predatory fish species observed in 2015 were generally comparable to percentages observed in previous years. At Campbell Slough the percentage was somewhat lower, and at Franz Lake somewhat higher than in most other years. However, at both Campbell Slough and Franz Lake, percentages of non-native species and predatory fish species have been variable from year to year, but with no clear trends increasing or decreasing trends observed (Figure 71).

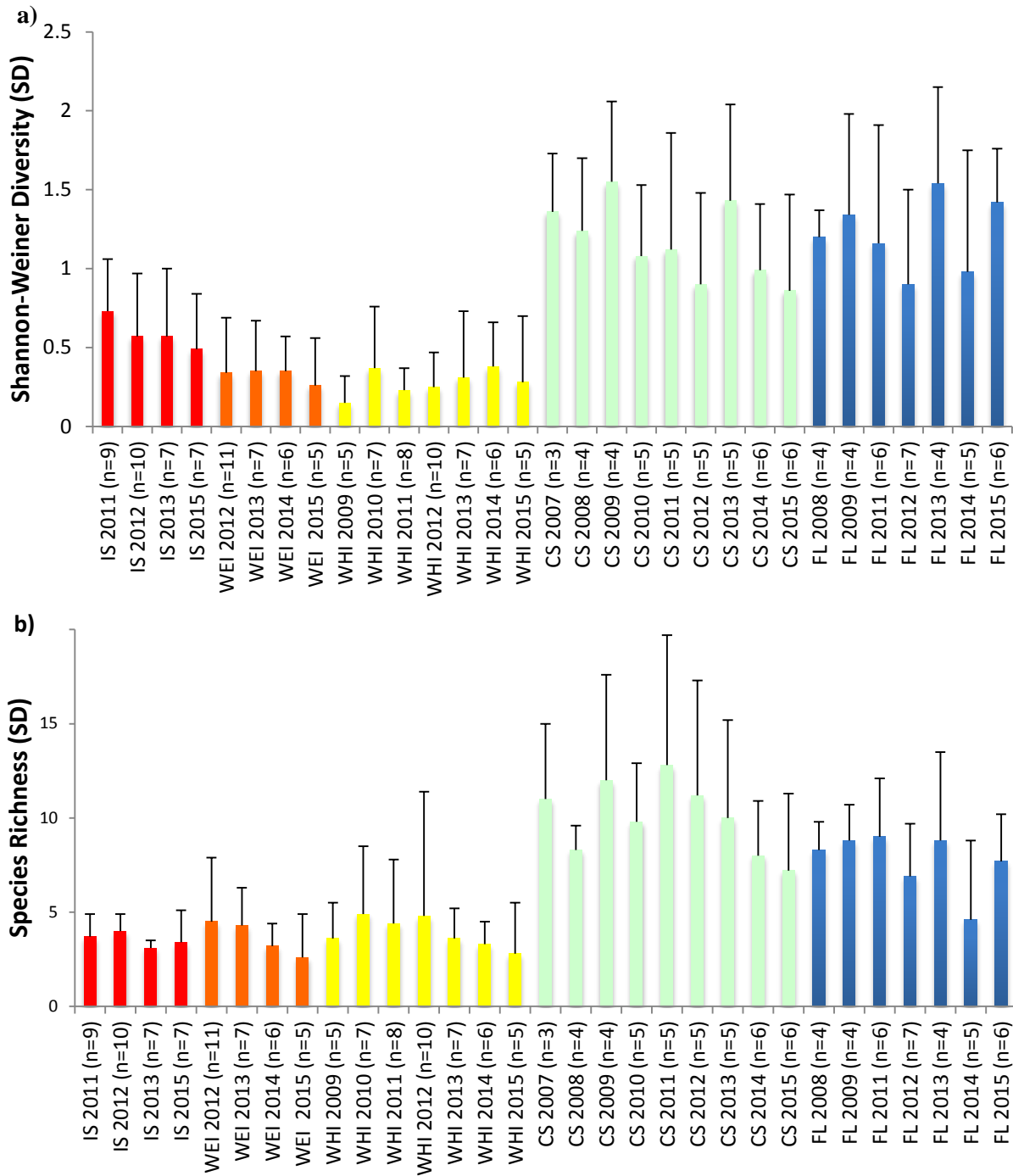


Figure 70. a) Shannon-Weiner diversity index and b) species richness (number of species) in mean (standard deviation, SD) values per sampling event (i.e., per monthly sampling event) at the EMP sampling sites in 2015 as compared to previous sampling years. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

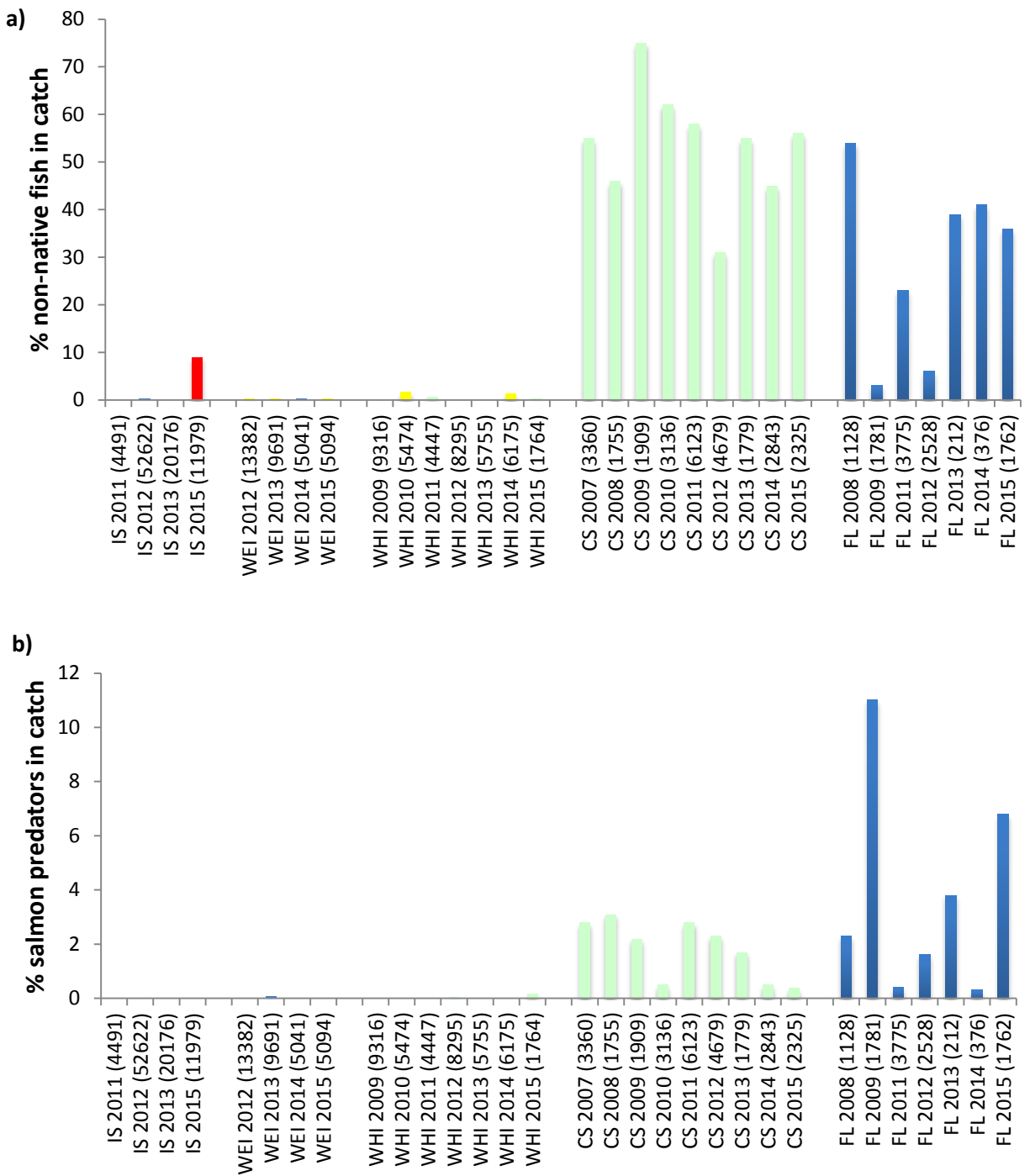


Figure 71. Percentages (based on total number of fish caught) of a) non-native fish species and b) % of fish that are recognized predators of juvenile salmon (i.e., smallmouth and largemouth bass, northern pikeminnow, walleye) in 2015 and in previous sampling years. Numbers contained in parentheses represent total fish catch at a site within a given year. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.1.1 Salmon Species Composition

Similar to previous sampling years, salmon species composition in 2015 varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 72). Chinook salmon were the dominant salmon species at Ilwaco Slough in Reach A, Welch Island in Reach B, Whites Island in Reach C, and Campbell Slough in Reach F, comprising 90% to 100% of salmonid catches. Chinook salmon were also the dominant species at Ilwaco Slough and Franz Lake, which is not necessarily typical for these sites. It should be noted, however, that at Ilwaco Slough the salmonid catch was represented by only seven fish, six of which were Chinook salmon. At Ilwaco Slough, Welch Island, Whites Island, and Franz Lake, unmarked (presumably wild) fish were more abundant than marked hatchery fish, accounting for 92-100% of the Chinook salmon collected (Figure 73). This pattern is typical for Welch and Whites Islands, and to an extent at Ilwaco Slough, although the number of Chinook at this site has generally been very low. At Franz Lake, this pattern has been evident since 2011, but higher proportions of marked Chinook salmon were collected in earlier sampling years. At Campbell Slough, marked Chinook salmon predominated in 2015, with 33% of Chinook salmon unmarked. Such a mixture of marked and unmarked Chinook salmon is the typical pattern for this site (Figure 73). In addition to Chinook salmon, small numbers of unmarked coho and chum salmon were found. No trout or sockeye salmon were caught in 2015.

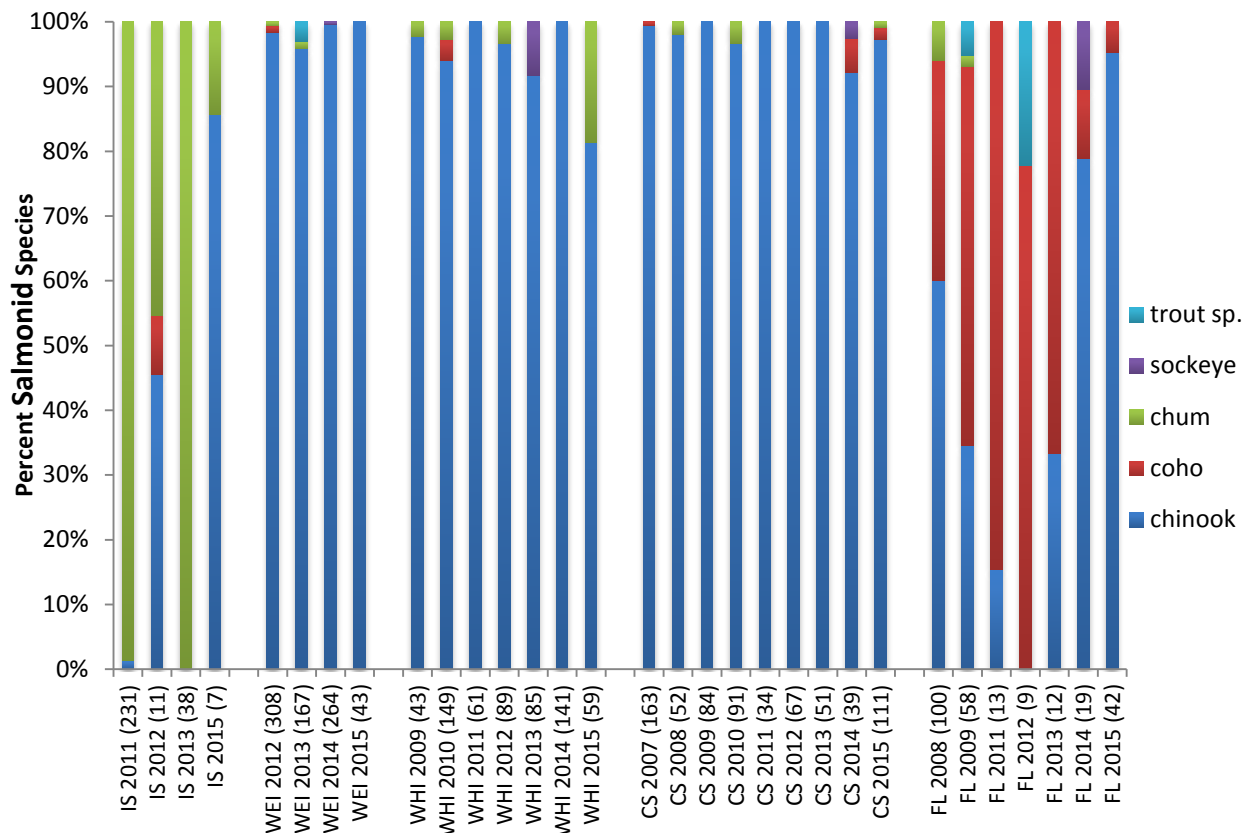


Figure 72. Percentage of salmonid species collected at EMP trends sites in 2015, as compared to percentages collected in previous sampling years. Total number of salmonids captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

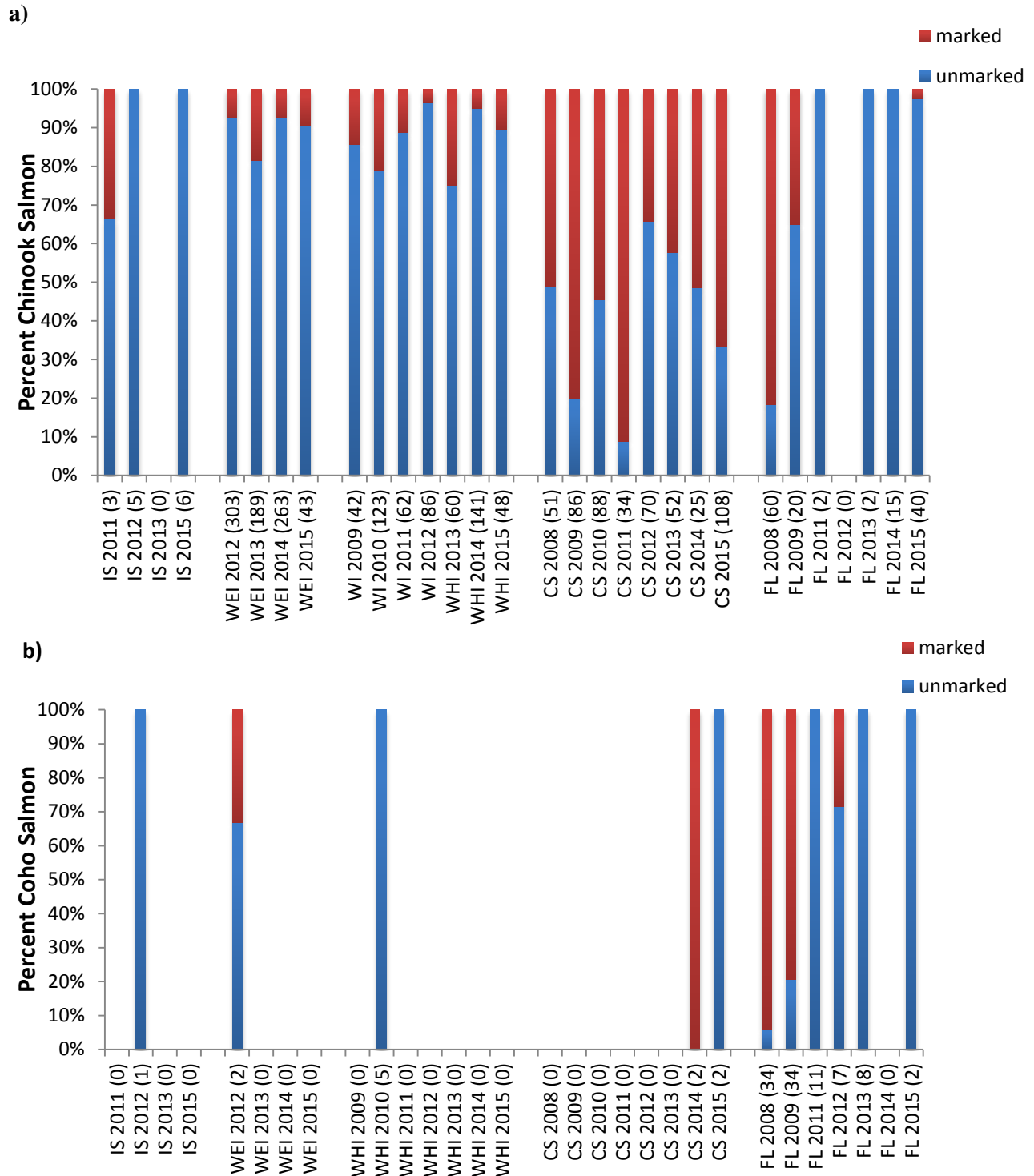


Figure 73. Percentage of marked and unmarked a) Chinook salmon and b) coho salmon captured at the EMP sampling sites in 2015, as compared to previous sampling years. Total number of the specified salmon species captured at a given site and year are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.1.2 *Salmon Density*

Chinook salmon. In 2015, unmarked Chinook salmon were captured at the EMP trends sites from March, when sampling began, through June. Chinook salmon were not observed in August, September, November, or December (sampling was not conducted in October) at the sites that were accessible and suitable for sampling during those months. The highest average density of unmarked juvenile Chinook salmon was 34.9 fish per 1000 m² in May (Figure 74). Mean Chinook salmon densities by site and year are shown in Figure 75. The density of unmarked Chinook salmon was highest at Campbell Slough and lowest at Ilwaco Slough, with intermediate values at the other sites. Densities of unmarked Chinook salmon in 2015 were generally within the same range as previous years at Ilwaco Slough, Campbell Slough and Franz Lake, but lower than usual at Welch Island and Whites Island. Marked Chinook salmon were captured in May and June, with the highest average density of 28.6 fish per 1000 m² in May (Figure 74). The densities of marked Chinook salmon in 2015 were generally lower than the densities of unmarked Chinook salmon, with the highest density observed at Campbell Slough.

Coho salmon. In 2015, no marked coho salmon were collected, and unmarked coho salmon were present only in April at a density of 0.46 fish/1000 m² (Figure 74). Coho salmon densities by site and year are shown in Figure 75. Only six coho salmon were caught in 2015, two at Campbell Slough in Reach F and four at Franz Lake (< 0.1 fish per 1000 m² at both sites). Coho salmon have been captured only sporadically at Ilwaco Slough, Welch Island and Whites Island, so their absence in 2015 was not unusual compared to previous years. At Campbell Slough, 2015 represents the first year that coho salmon have been captured since systematic sampling for salmon density began in 2008. At Franz Lake, the only site where coho salmon are consistently sampled, coho salmon density was at its lowest reported level in 2015 and has shown a consistent decline since 2011. Marked coho salmon, which were common at the site in 2008 and 2009, have not been observed since 2012. The absence of coho salmon in the catches at Franz Lake may have been due to the inability to sample the site during high flow in spring when juvenile coho salmon are typically expected to migrate through the area or are released from hatchery facilities.

Chum salmon. In 2015, chum salmon were observed at the EMP trends sites in April and May, with the highest average density in May (4.1 fish per 1000 m²; Figure 74). Chum salmon were present at Whites Island and Campbell Slough in 2015 (Figure 76). Since the beginning of this long-term study chum salmon have been found at all the sampling sites at varying densities, although not consistently. Chum salmon have not been observed at Franz Lake since 2009.

Sockeye salmon and trout species. Sockeye salmon and trout were not caught in 2015, thus densities at all sites in 2015 were low relative to previous years (Figure 76).

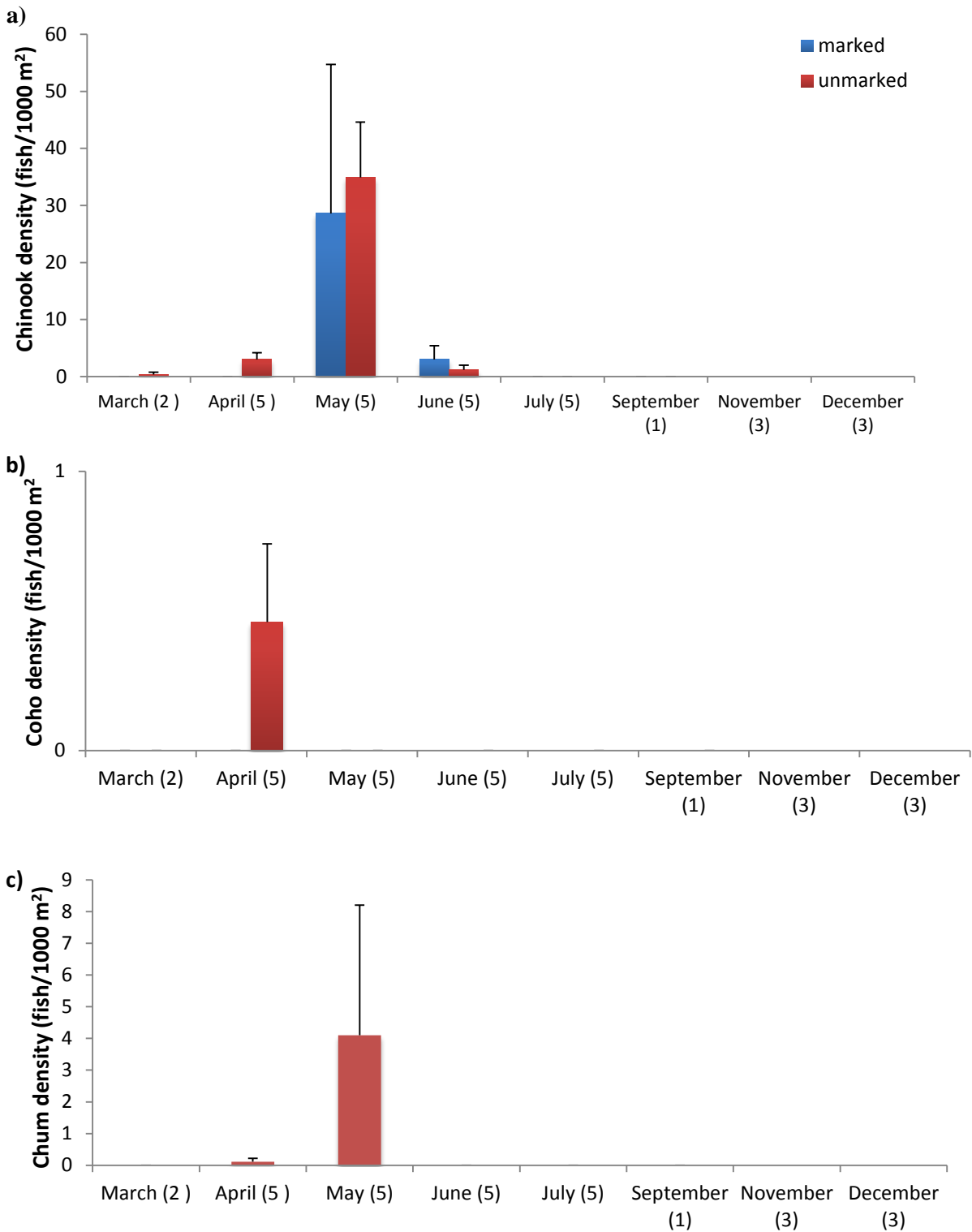


Figure 74. Marked (blue bars) and unmarked (red bars) juvenile a) Chinook salmon, b) coho salmon, and c) chum salmon densities (fish per 1000 m²) by month during the 2015 sampling year (all sites combined).

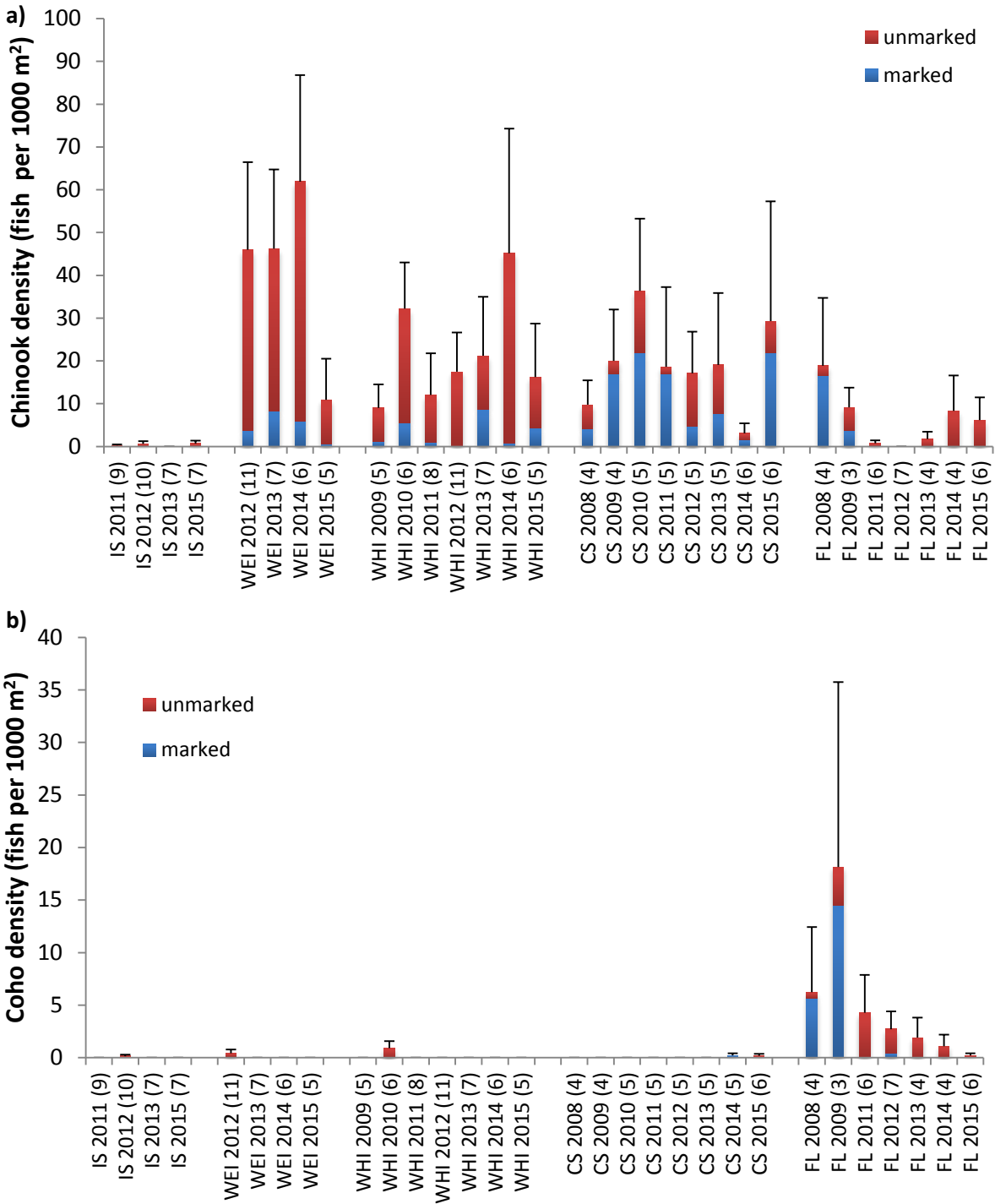


Figure 75. Marked (blue bars) and unmarked (red bars) juvenile a) Chinook salmon and b) coho salmon densities (fish per 1000 m²) by trends site and year. Total number of salmonids captured per year at a site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

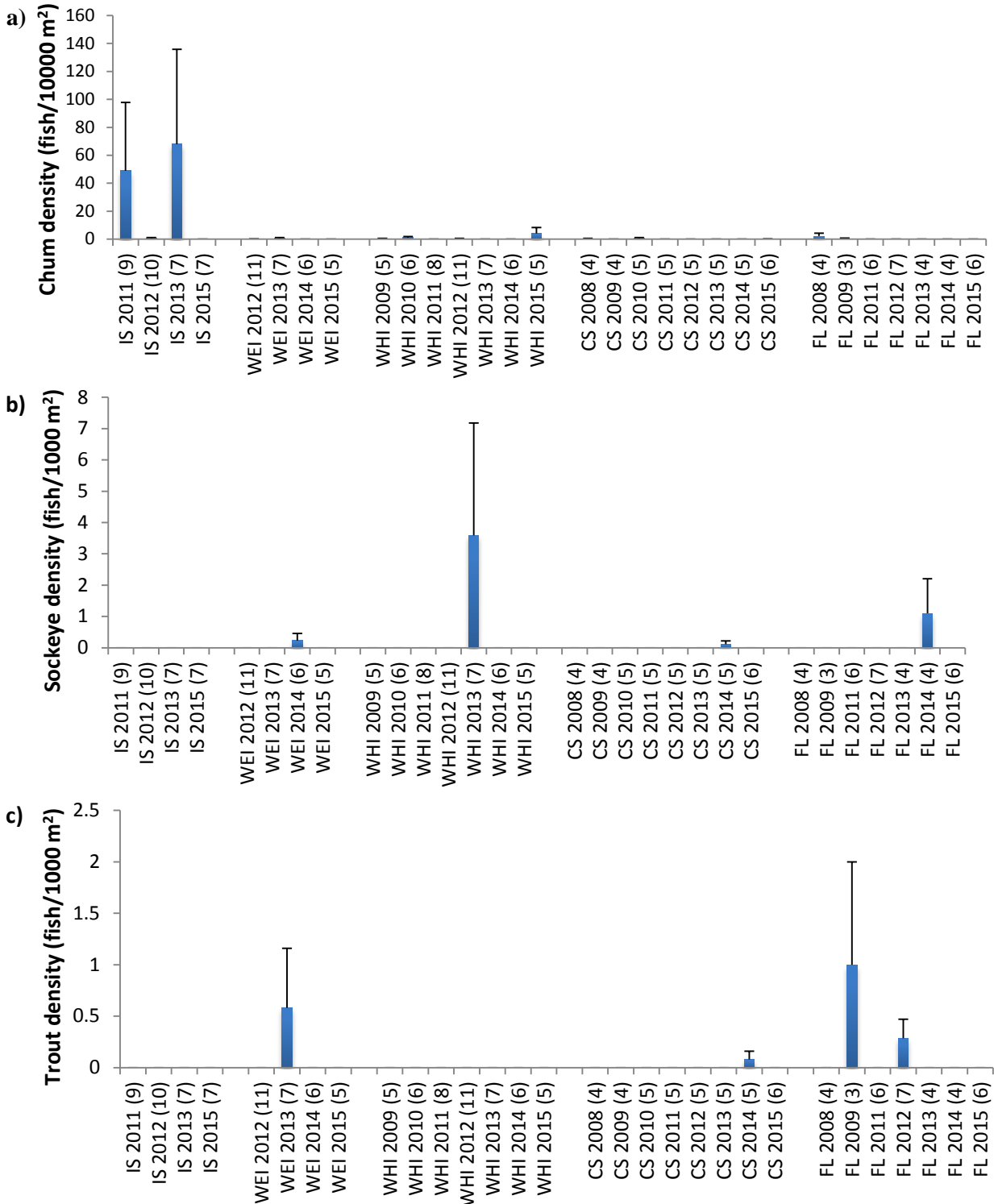


Figure 76. Juvenile a) chum salmon, b) sockeye salmon, and c) trout densities (fish per 1000 m²) by year captured at trends sites. Total number of salmonids captured per year at a site is presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.2 Salmon Metrics

3.6.2.1 *Genetic Stock Identification*

In this report we present the Chinook salmon genetic stock data collected in 2014, as genetic samples from the 2015 sampling year are currently undergoing analysis. In 2014, genetics data were collected from Chinook salmon at Welch Island, Whites Island, Campbell Slough, and Franz Lake. Too few Chinook salmon were collected at Ilwaco Slough to allow for meaningful stock composition estimates. Among unmarked fish, West Cascades fall Chinook were the most abundant stock at Welch Island and Whites Island, while Spring Creek Group fall Chinook were more prominent at Campbell Slough (Figure 77). Interior stocks such as Snake River fall Chinook and Upper Columbia summer/fall Chinook were also captured at the trends sites. The stocks present at the trends sites were generally similar over the sampling years, although in 2014 the interior stocks were somewhat less abundant at the Welch Island, Whites Island, and Campbell Slough sites. In 2014 we obtained genetic stock information on unmarked Chinook salmon from Franz Lake for the first time. Snake River fall Chinook and Upper Columbia River summer/fall Chinook salmon were the predominant stocks at Franz Lake.

In 2014, marked Chinook salmon were collected and analyzed for genetic stock at Welch Island, Whites Island, and Campbell Slough. All marked fish collected in 2014 were either Spring Creek Group fall or West Cascades fall Chinook salmon. West Cascades fall Chinook were most common at Welch Island and Whites Island, whereas Spring Creek Group fall Chinook were most common at Campbell Slough (Figure 77). As with unmarked fish, the stock composition of marked fish at the sampling sites did not vary greatly from year to year, although in 2014 the number of fish from interior stocks was especially low.

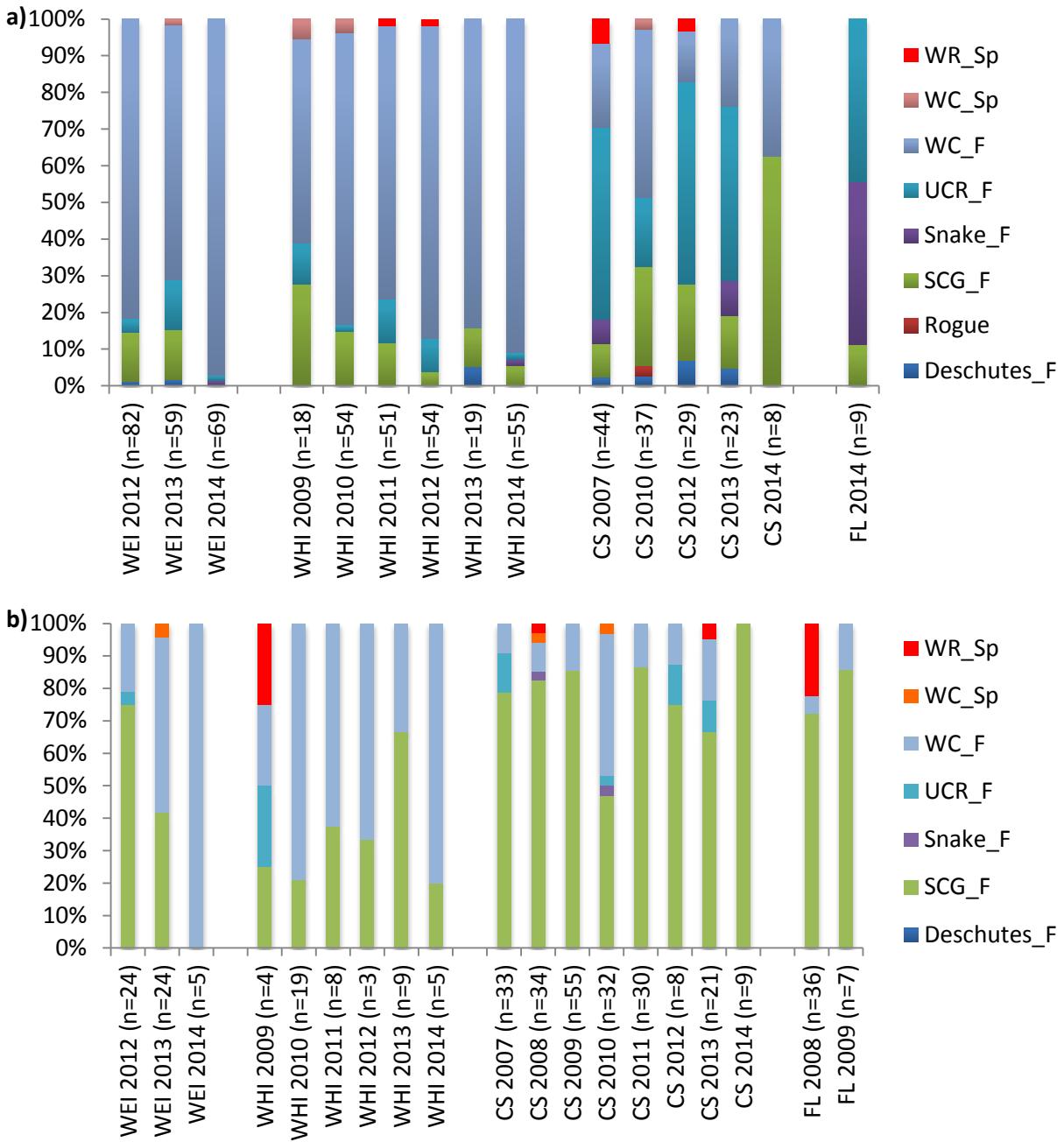


Figure 77. Genetic stock composition of a) unmarked and b) marked Chinook salmon at the trends sites in 2014, as compared to previous years. Sample sizes for each site are presented in parentheses. Ilwaco Slough is not shown, as no new data are available from this site for temporal comparison. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough. Chinook salmon stocks: WR_Sp = Willamette River spring, WC_Sp = West Cascade spring, WC_F = West Cascade fall, UCR_F = Upper Columbia River fall, Snake_F = Snake River fall, SCG_F = Spring Creek Group fall, Rogue = Rogue River, Deschutes_F = Deschutes River fall.

3.6.2.2 *Salmon Size and Condition*

Chinook salmon

Length, weight, and condition factor. In comparison with previous sampling years, the length, weight and condition of unmarked Chinook salmon showed similar patterns in 2015, with the largest fish typically captured at Campbell Slough (Figure 78). Within sites, there was some variation among years. Significant differences in length among years were observed at Welch Island ($p < 0.0001$), Whites Island ($p < 0.0001$), Campbell Slough ($p < 0.0001$) and Franz Lake ($p = 0.0003$), with the 2015 fish being among the largest at all sites (Tukeys multiple range test, $p < 0.05$). At Ilwaco Slough, significant differences in length among years were also observed ($p = 0.0223$), but the 2015 values were not significantly higher or lower in comparison to other years. Significant differences in weight among years were also observed at Welch Island ($p < 0.0001$). Chinook salmon captured at Whites Island ($p < 0.0001$) and Campbell Slough ($p < 0.0001$) in 2015 were among the heaviest collected at these sites (Tukeys multiple range test, $p < 0.05$). Significant differences in fish weight among years were also observed at Ilwaco Slough ($p = 0.0052$) and Franz Lake ($p = 0.0198$); however, the 2015 values were not especially high or low in comparison to other years. Differences in condition factor were observed among years for Welch Island ($p = 0.0007$), Whites Island ($p < 0.0001$) and Campbell Slough ($p = 0.0002$), but the 2015 values were not especially high or low in comparison to other years. At Franz Lake ($p = 0.3162$) and Ilwaco Slough ($p = 0.9575$) condition factor did not differ significantly among years.

Marked Chinook salmon were not caught at Ilwaco Slough in 2015. At Welch Island, marked Chinook salmon were caught in 2014 and 2015, but there was no significant difference in length between the two years ($p = 0.2614$; Figure 79) and weights were not collected for the 2015 fish, so weight and condition could not be evaluated. Similarly, at Franz Lake, where marked Chinook salmon were caught in 2008, 2009, and 2015, no significant differences were observed across sampling years for length, weight, or condition factor ($0.1520 < p < 0.7178$). At Whites Island, where marked Chinook salmon were caught in both 2014 and 2015, length ($p = 0.1462$), weight ($p = 0.0535$), and condition factor ($p = 0.0049$) all tended to be lower in 2015 than in 2014, although the difference was significant only in the case of condition factor. At Campbell Slough, length, weight, and condition factor all varied with sampling year ($p < 0.0001$ for all three metrics), but values in 2015 were not especially high or low in comparison to previous years.

The overall mean length of unmarked fish from the five trends sites sampled in 2015 was 55 ± 15 mm for unmarked fish ($n = 237$) as compared to 80 ± 6 mm for marked fish ($n = 46$), with the values showing a significant difference ($p < 0.0001$). The overall mean weight of unmarked fish from the five trends sites sampled in 2015 was 2.0 ± 2.1 g ($n = 232$) as compared to 5.5 ± 1.4 g for marked fish ($n = 32$), with the values showing a significant difference ($p < 0.0001$). The overall mean condition factor of unmarked fish from the five trends sites sampled in 2015 was 0.912 ± 0.209 for unmarked fish ($n = 232$) as compared to 1.04 ± 0.072 for marked fish ($n = 32$), with the values also showing a significant difference ($p = 0.0005$).

Size class distribution. At the trends sites in 2015, the majority of unmarked Chinook salmon were fry (60%), 40% were fingerlings, and none were yearlings (Figure 80). At Ilwaco Slough, the one unmarked Chinook salmon collected was a fry. At Welch Island, fry predominated, making up 76% of unmarked Chinook salmon and fingerlings making up the remaining 24%. At Whites Island, no fry were collected in 2015, and 100% of sampled fish were fingerlings. At Campbell Slough, fingerlings predominated, comprising 86% of the catch, while fry made up 14% of the catch. At Franz Lake, 83% of the 35 unmarked Chinook that were caught in 2015 were fry, with fingerlings making up the remaining 17%. In comparison to previous years, the percentage of fry at Welch Island and Whites Island was lower than in previous years, while at Franz Lake, the proportion of fry in 2015 was among the highest observed (Heterogeneity G test, $p < 0.05$). At Ilwaco Slough, the proportion of fry in 2015 was relatively higher,

while at Campbell Slough it was relatively low, but in neither case were the 2015 proportions significantly higher or lower than the overall average for those sites (Heterogeneity G-test, $p < 0.05$).

Of the 45 marked Chinook salmon caught at the trends sites in 2015, 100% were fingerlings; no yearlings were observed (Figure 80). In comparison to previous sampling years, the proportion of yearlings encountered did not differ significantly from previous years at Welch Island, Whites Island, Campbell Slough, or Franz Lake. No marked Chinook salmon have been caught at Ilwaco Slough.

Other salmon species

Only four coho salmon were caught in 2015, two unmarked fish from Campbell Slough and two unmarked fish from Franz Lake, all caught in April. These are the only coho salmon for which length and weight data are available for these sites, thus seasonal comparisons cannot be made. The average length, weight, and condition factor of fish captured at Campbell Slough (\pm SD) were 151 ± 1 mm; 33.5 ± 1.5 g; and 0.97 ± 0.07 , respectively. The average length, weight, and condition factor of fish captured at Franz Lake (\pm SD) were 115 ± 3 mm; 16.2 ± 1.9 g; and 1.06 ± 0.05 , respectively. Eleven chum salmon were captured and measured in 2015, one at Campbell Slough and 10 at Whites Island, all caught in May. The average length, weight, and condition factor of these fish (\pm SD) were 58 ± 6 mm; 1.8 ± 0.9 g; and 0.87 ± 0.32 , respectively. The chum salmon collected in 2015 were somewhat larger than those collected in previous years, which were collected primarily in April, and whose average length and weight ranged from 43-48 mm and 0.6-1.0 g. The 2015 value for condition factor was intermediate, between a high of 1.10 in 2008 and a low of 0.58 in 2013. Sockeye salmon and trout were not caught at any of the trends sites in 2015.

Franz Lake was the only site where coho have been sufficiently numerous to compare size among years, and even at this site, only unmarked coho salmon were caught in all sampling years including 2015. Mean length, weight, and condition factor are shown for unmarked coho salmon from Franz Lake in (Figure 81). Mean length (\pm SD) varied from 82 ± 11 mm in 2013 to 120 ± 34 mm in 2009, with the fish collected in 2015 being of intermediate size (115 ± 3 mm); length did not differ significantly among sampling years ($p = 0.1802$). Weight, however, differed among years ($p = 0.0064$), with the highest values in 2009 and the lowest in 2013. Again, the weight of fish collected in 2015 was intermediate compared to other years. Condition factor did not show significant differences among sampling years ($p = 0.2071$), but the lowest values were observed in 2008 and the highest in 2014 and 2015.

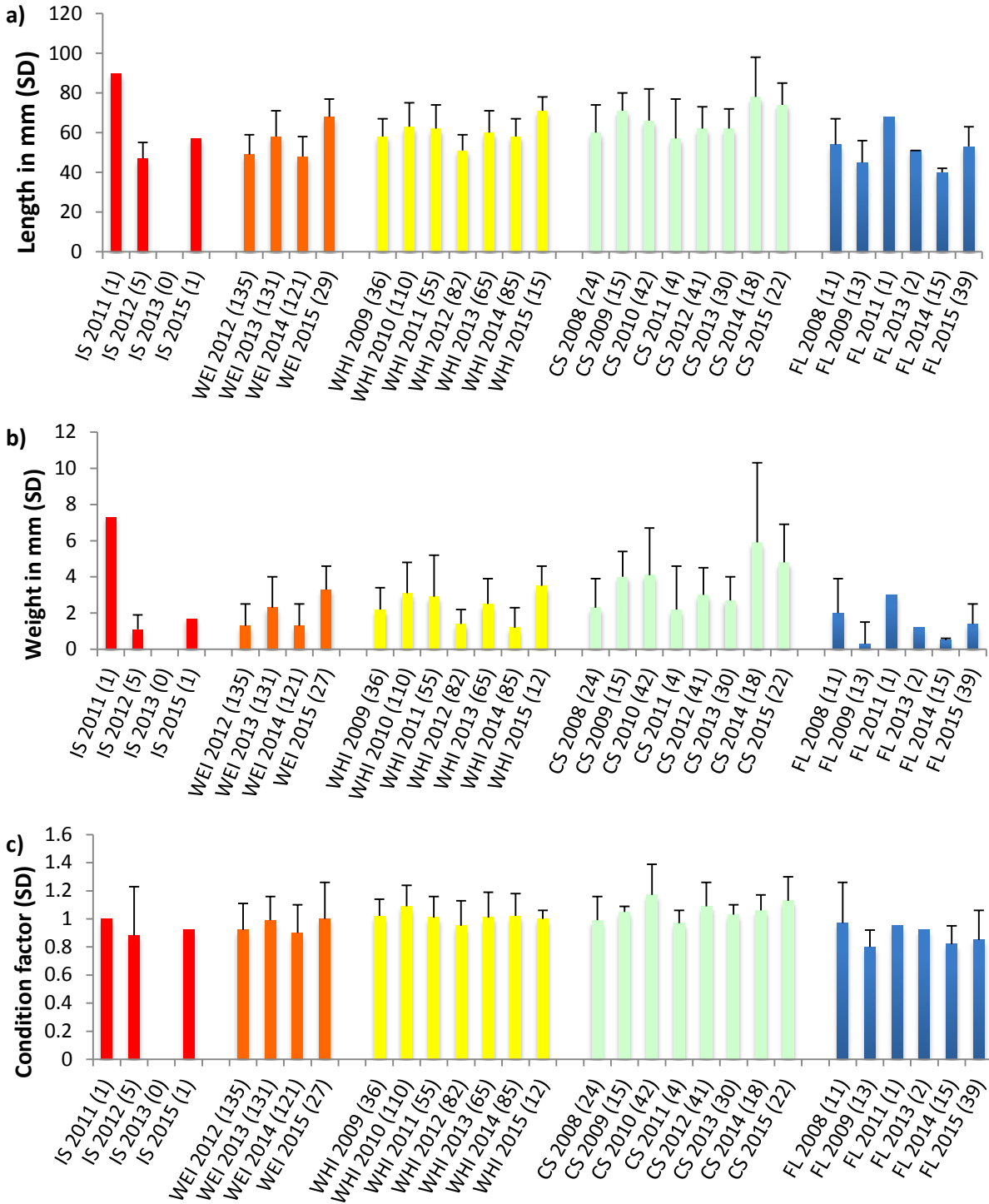


Figure 78. Mean (SD) a) length (mm), b) weight (g), and c) condition factor (\pm SD) of unmarked juvenile Chinook salmon at trends sites in 2014 as compared to previous years. Within the sites, values with different letter superscripts are significantly different (Tukey's multiple range test, $p < 0.05$). Total number of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

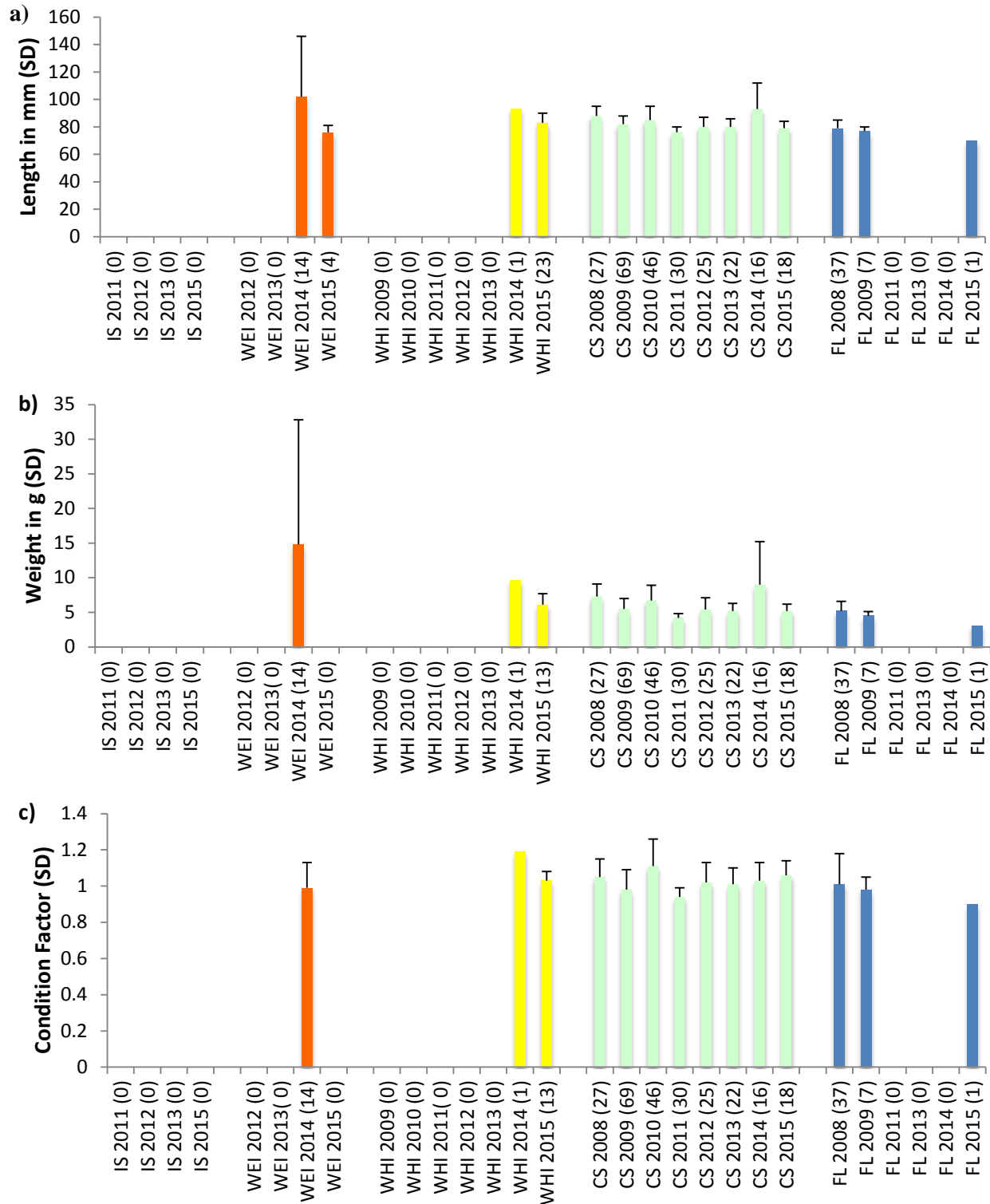


Figure 79. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of marked Chinook salmon at trends sites in 2015 compared to previous sampling years. Total number of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

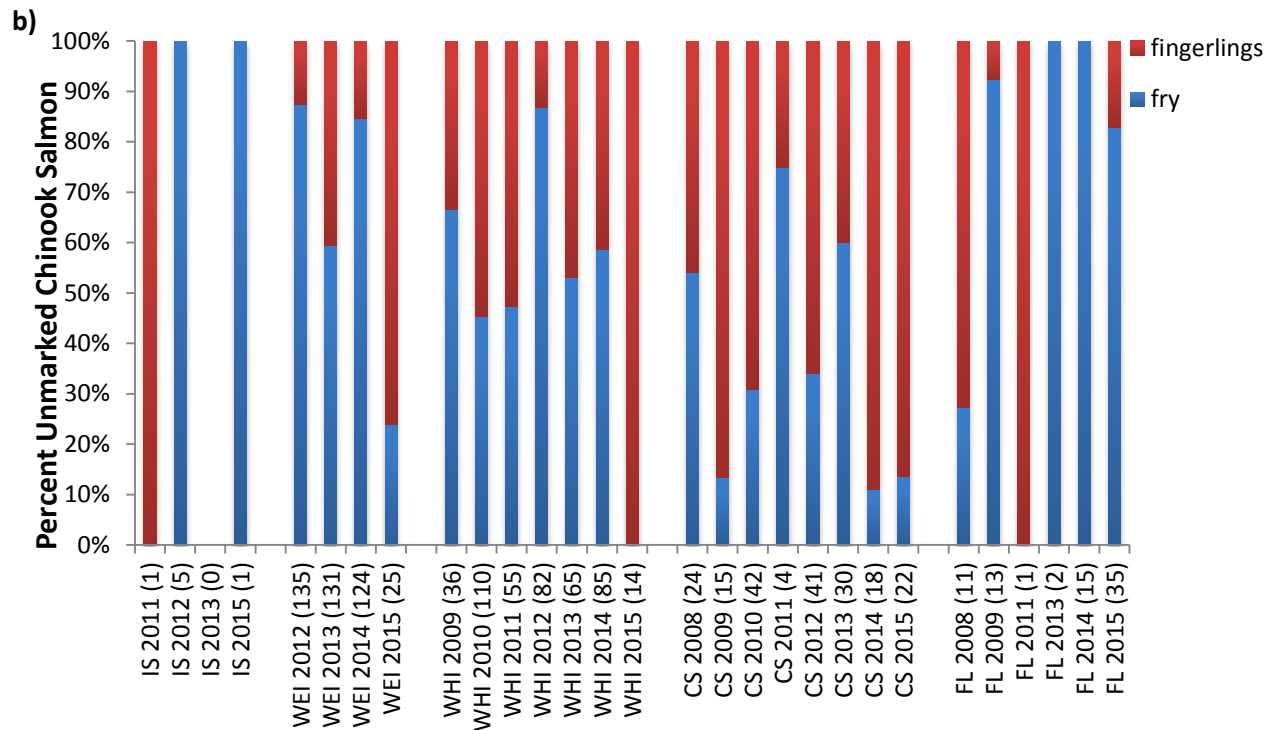
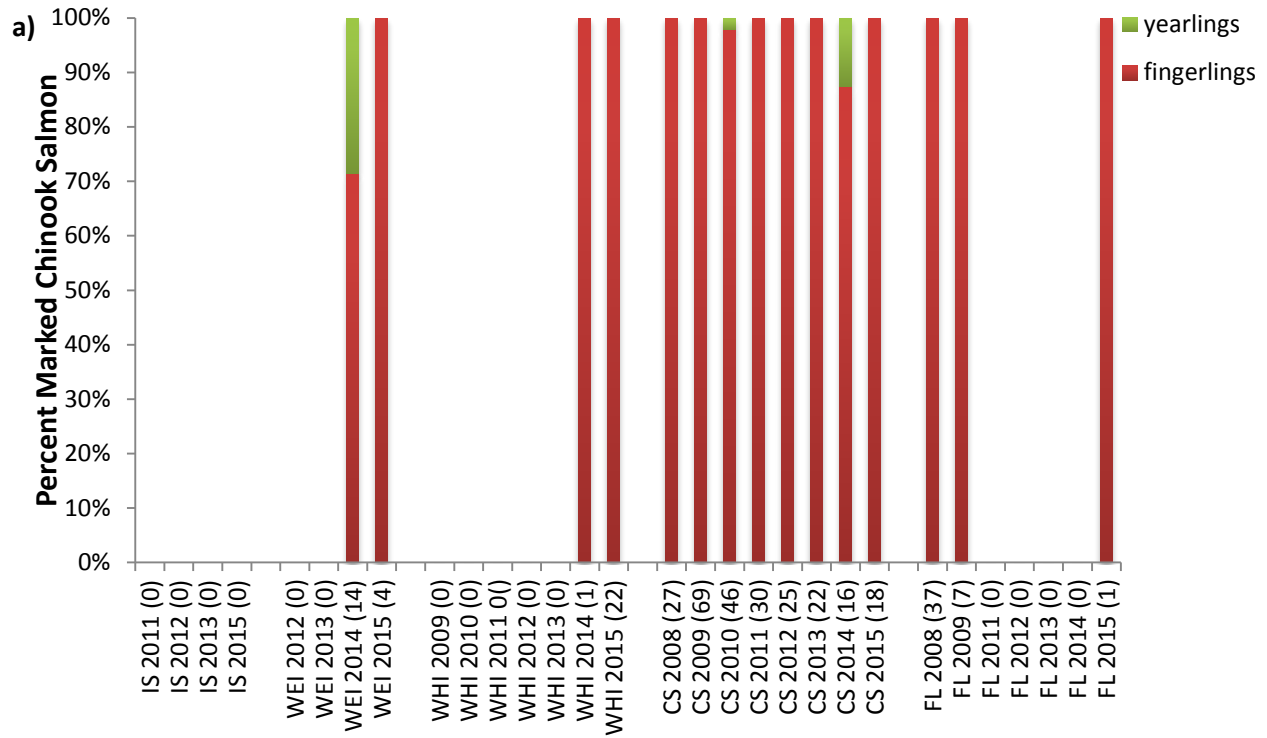


Figure 80. Size class distribution of a) marked and b) unmarked juvenile Chinook salmon captured at trends sites in 2015 and in previous sampling years. Total number of Chinook salmon captured per year at a site are shown in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

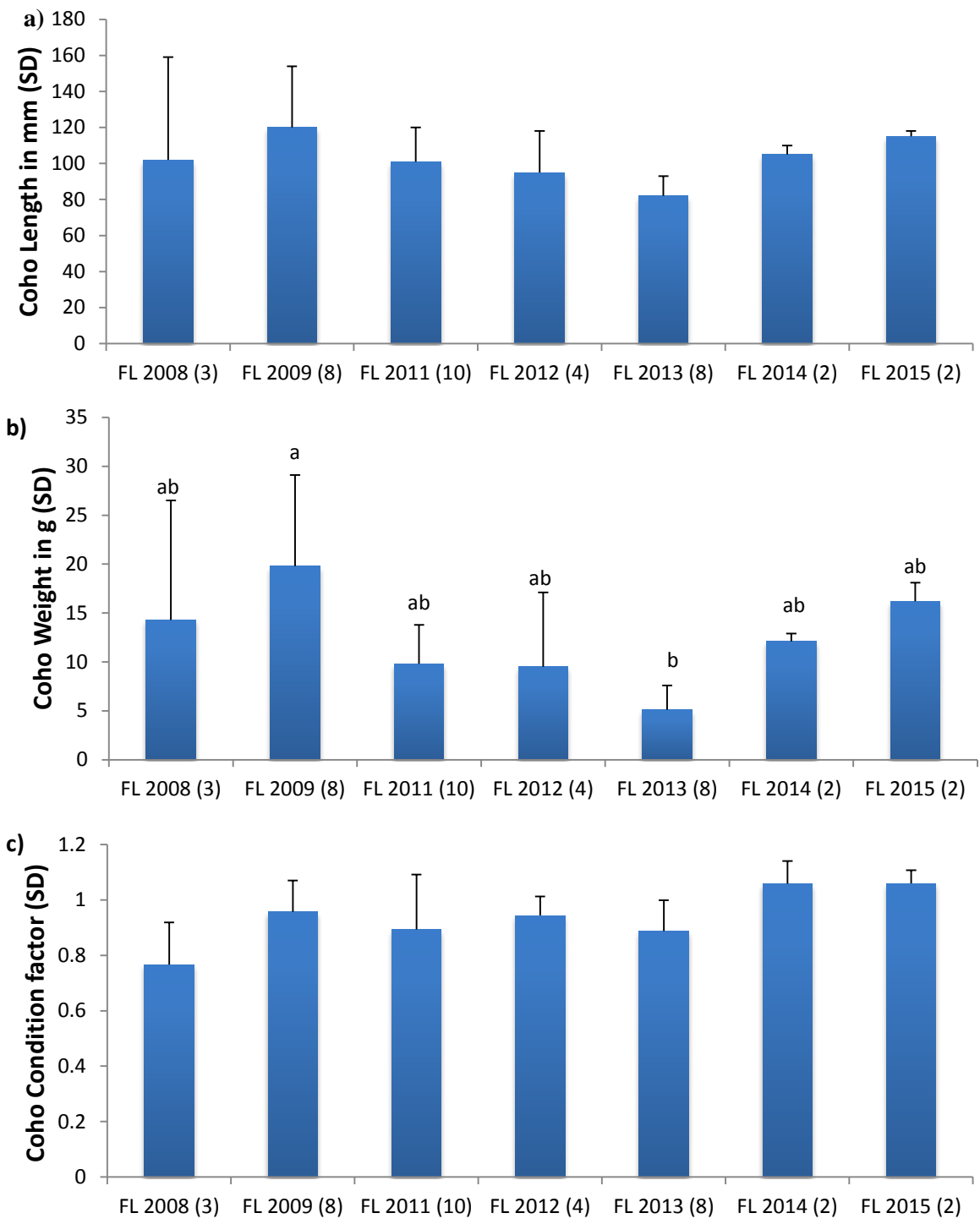


Figure 81. Mean a) length (mm), b) weight (g), and c) condition factor of unmarked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.

3.6.2.3 *Somatic Growth Analyses*

Otoliths were collected from juvenile Chinook salmon at 28 sites in the lower Columbia River from EMP status and trends sites, toxic contaminant monitoring sites, and action effectiveness monitoring sites. Results from 2005 and 2007-2013 sampling years are reported here. Analyses of otoliths collected in 2014 and 2015 have not yet been completed.

Variability in somatic growth rates among the sites for the seven years of available data are shown in Figure 82. ANCOVA indicated marginally significant (2005 and 2013), and significant differences in growth rate among sites when accounting for fish size (fork length was a covariate). A standard ANCOVA was used for all years except 2013 where a separate slopes model was used because a significant interaction was detected between site and fork length. In general, growth rates tended to be higher in fish from upstream sites than downstream sites. An exception was the Columbia Confluence Washington site sampled in 2008, which had the lowest growth rate of all sites, although its position in the lower river was intermediate.

Variability in growth rate (mm/day) by hydrogeomorphic reach for each of the seven sampling years is shown in Figure 83. ANCOVA indicated significant differences in growth rate among reaches, when accounting for fish size (fork length was a covariate), for fish collected in all years except 2005 and 2008. A standard ANCOVA was used for analyses except 2010 and 2012 where an ANOVA and separate slopes model were used, respectively. The separate slopes model was used because a significant interaction was detected between site and fork length. In four of the seven sampling years, fish from reaches F and H grew significantly faster than those collected from Reaches B or C.

For the three sites for which data were available for four years or more (Campbell Slough, Mirror Lake, and Whites Island), annual variability in growth rate was compared (Figure 84). All sites showed significant variability in growth rate by sampling year. At Campbell Slough, growth rates were significantly lower in 2012 than in 2007. At Whites Island, growth rates were significantly lower in 2011 than in 2013; and at Mirror Lake, growth rates were significantly lower in 2011 than in 2008.

Figure 85 shows variability in growth rate by genetic stock. Fish were included in this analysis if their probability of assignment was greater than or equal to 0.8, and populations were included if their sample size was 10 fish or greater. ANCOVA indicated a significant difference ($p < 0.001$) in growth rate among populations when we account for differences in fork length, with West Cascades fall Chinook salmon showing significantly lower growth rates than Spring Creek Group fall Chinook or Upper Willamette River spring Chinook.

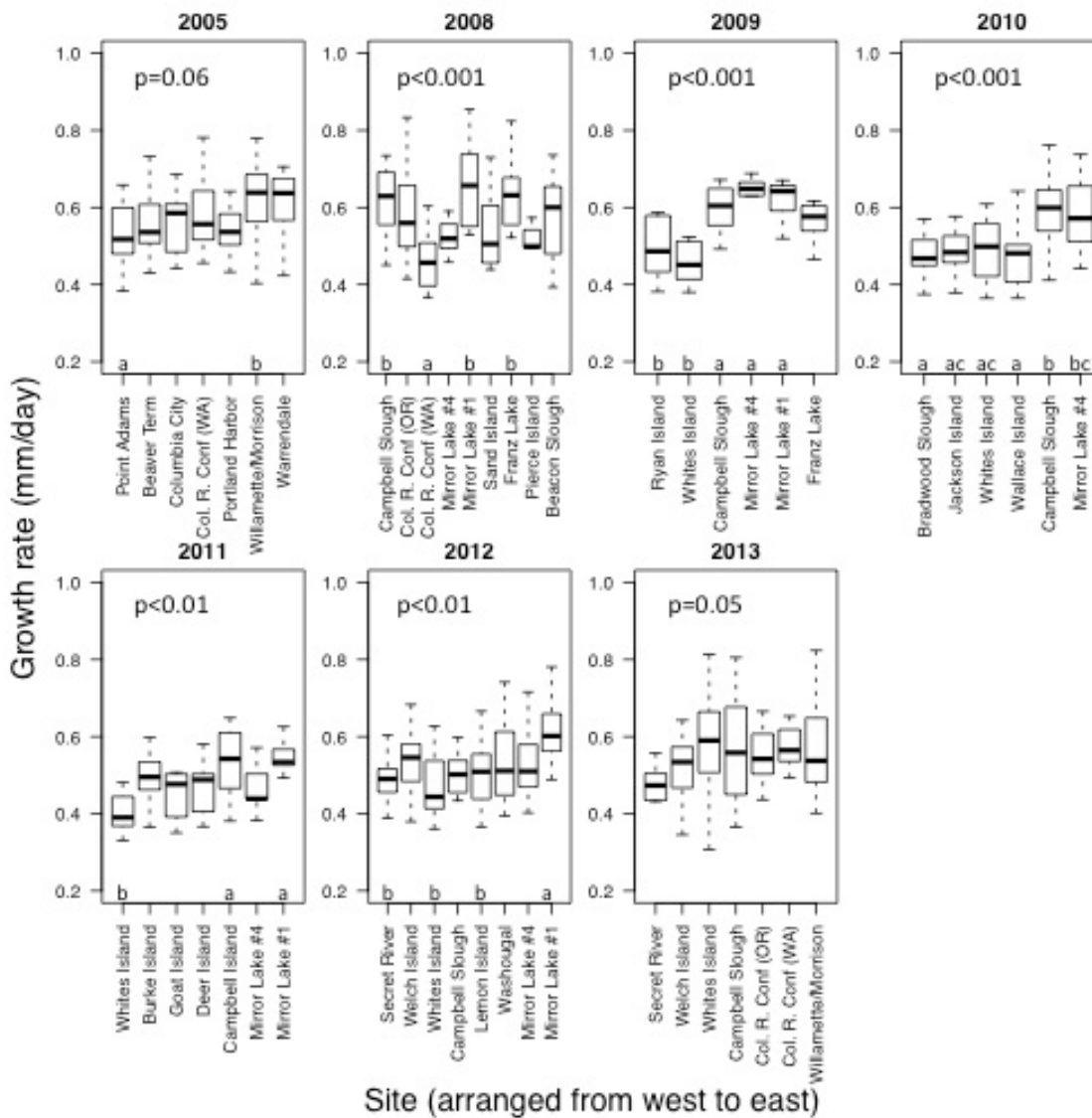


Figure 82. Somatic growth rate (mm/day) from otolith collection sites across seven sampling years. P-values are provided in each plot and unique letters indicate sites that were significantly different as determined from bonferroni post-hoc analysis. Median, interquartile range, and extreme values are indicated as the horizontal line, box, and whiskers, respectively.

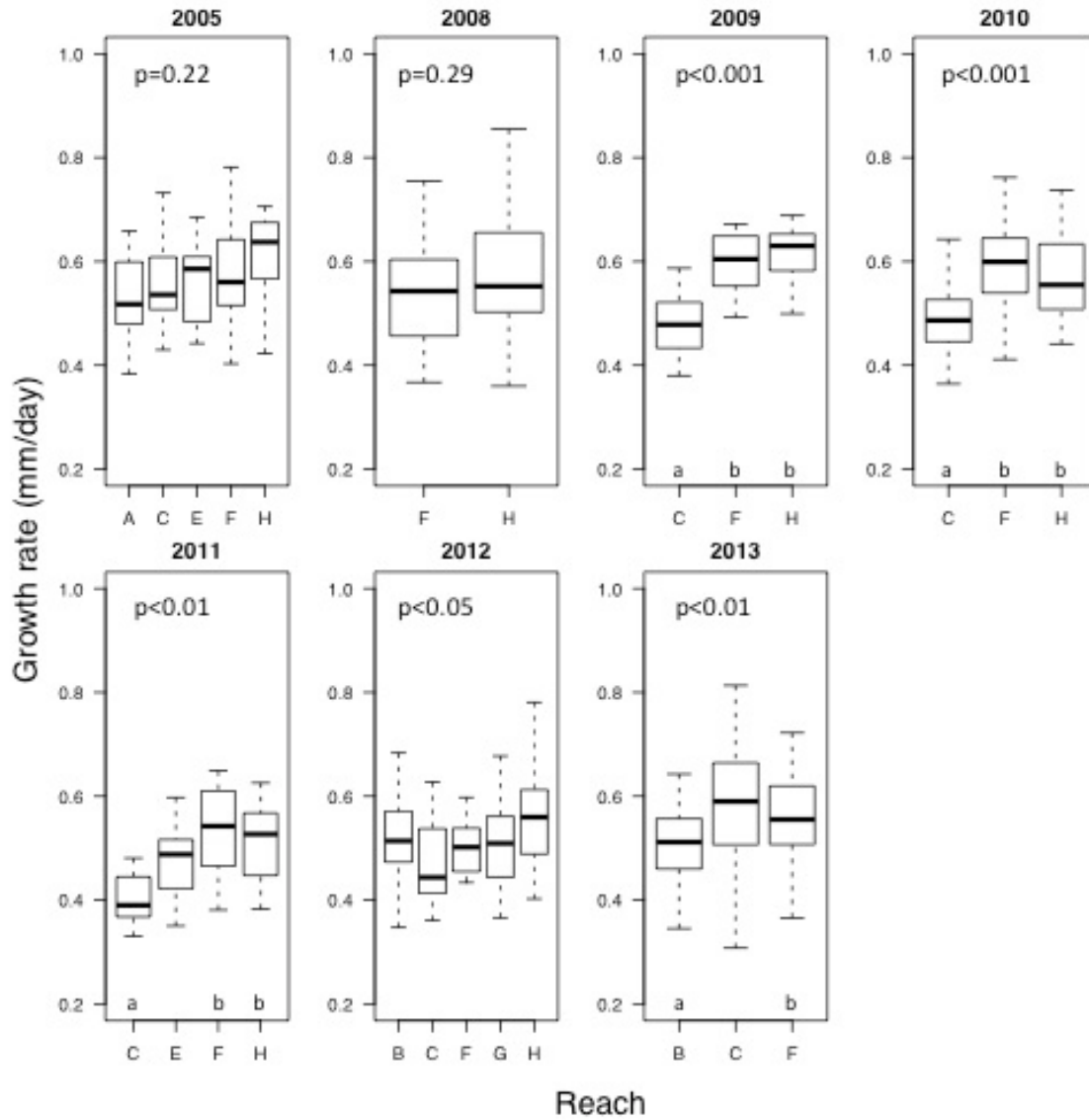


Figure 83. Somatic growth rate (mm/day) from different hydrogeomorphic reaches across seven sampling years. P-values are provided in each plot and unique letters indicate reaches that were significantly different as determined from bonferroni post-hoc analysis. Median, interquartile range, and extreme values are indicated as the horizontal line, box, and whiskers, respectively.

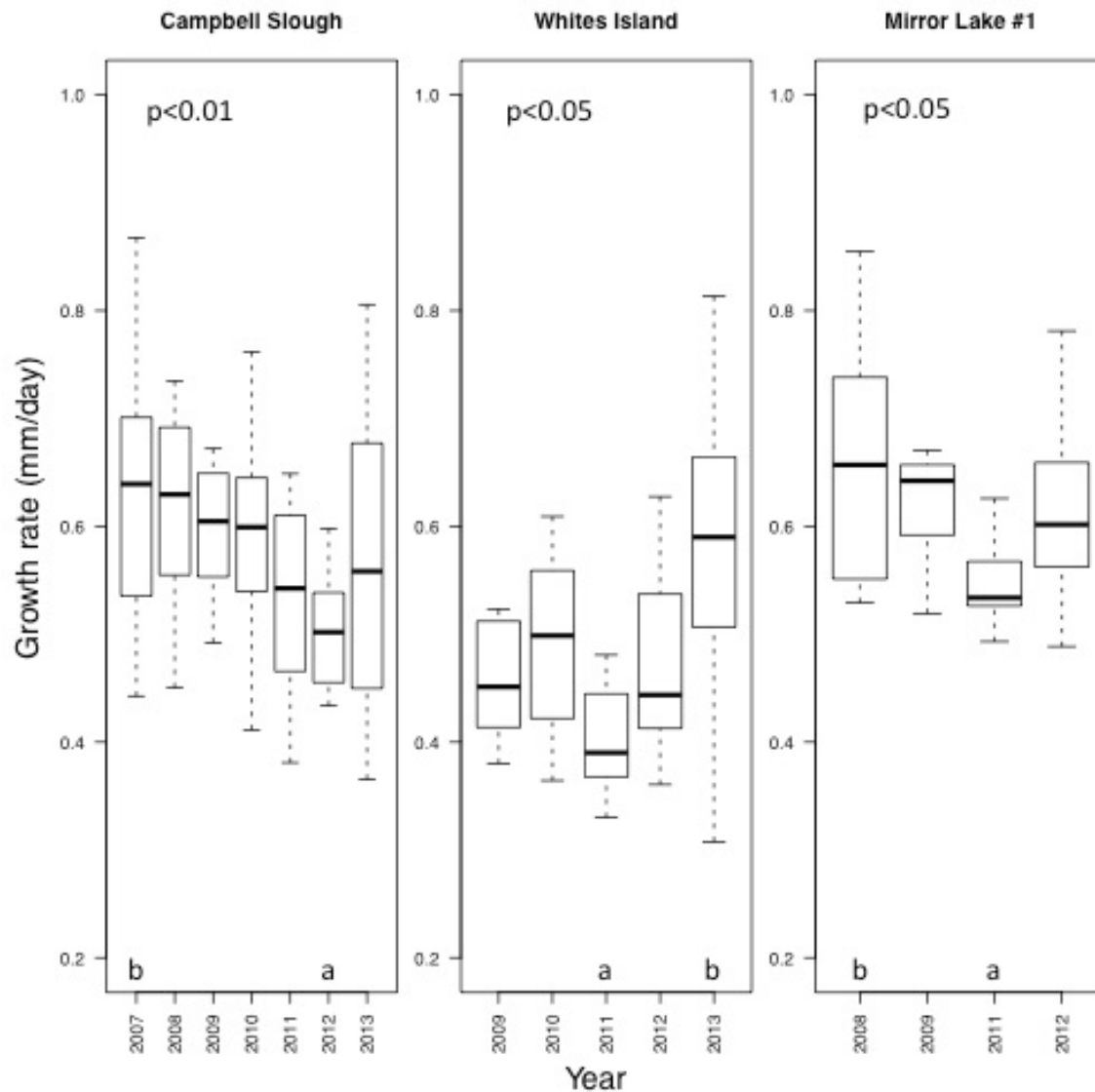


Figure 84. Somatic growth rate (mm/day) of juvenile Chinook salmon from across otolith collection years. P-values are provided in each plot and unique letters indicate years that were significantly different as determined from bonferroni post-hoc analysis. Median, interquartile range, and extreme values are indicated as the horizontal line, box, and whiskers, respectively.

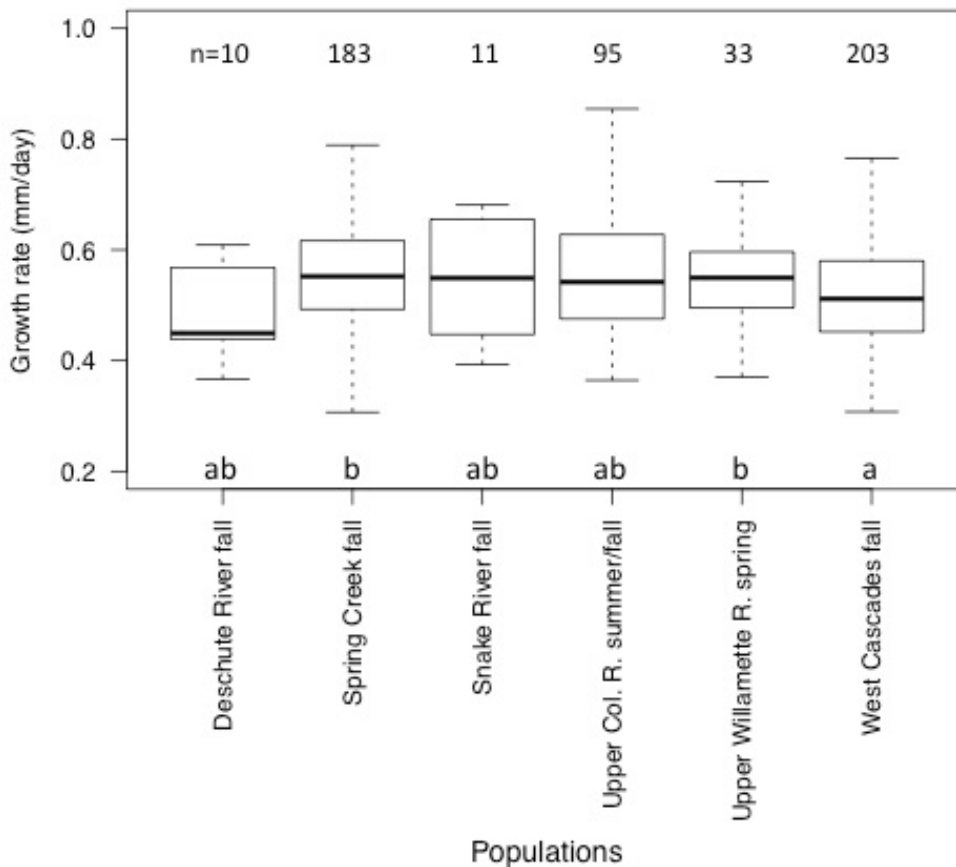


Figure 85. Somatic growth rate (mm/day) among genetically assigned populations. Significant differences between populations were determined from bonferroni post-hoc analysis and are indicated by unique letters. Median, interquartile range, and extreme values are indicated as line, box, and whiskers, respectively.

3.6.2.4 *Lipid Content of Juvenile Chinook Salmon*

In the 2014 annual report, we presented data on lipid content of juvenile Chinook salmon between 2007 and 2013. Lipid samples from 2014 are currently being analyzed and will be presented in a future report. The 2015 samples will be analyzed when genetics data are available for the 2015 fish.

3.6.2.5 *Contaminants in Juvenile Chinook Salmon*

In the 2014 report, contaminant data from 2007-2013 were presented. The samples from 2014 are currently being analyzed and data will be presented in a future report. The samples from 2015 will be analyzed when genetics data from the 2015 fish are available.

3.6.3 *Juvenile Salmon in Columbia River Tributaries*

In 2015, we conducted a pilot study to monitor the occurrence of juvenile salmon in the lowest reaches of two tributaries of the Columbia River, the Grays River and the Lewis River. These sites were sampled by beach seine in April, May, and June.

Fish community. Three spined stickleback and juvenile salmon were the predominant species at both sites during most sampling months. Other species present included starry flounder (*Platichthys stellatus*), various sculpin species (*Cottus* sp.), banded killifish, American shad (*Alosa sapidissima*), yellow perch, peamouth, and northern pikeminnow. Juvenile salmon were also present and made up an especially high proportion of the catch in the lower Lewis River (Figure 86).

Salmon. Marked hatchery salmon (of any species) were not observed at either the Grays River or the Lewis River sites. In the Grays River, chum, coho and Chinook salmon were all present (Figure 87). Chum salmon were collected in April, coho salmon were collected in May, and Chinook salmon were present in both months. In the Lewis River, Chinook salmon dominated the catches in all sampling months, although small numbers of coho salmon were also found in the lower Lewis River in June (Figure 87). Chinook salmon densities were consistently higher in the Lewis River than in the Grays River, and reached peak levels in the Lewis River in May and June. Coho and chum salmon densities, on the other hand, were highest in the Grays River (Figure 88).

The size class distribution of juvenile Chinook salmon from the tributary sites, as compared to the Columbia River trends sites, is shown in Figure 89 (this type of analysis was not performed on coho and chum salmon because the number of these fish collected at the tributary and trends sites was too small.) The proportion of fry was higher at the tributary sites than in most of the EMP trends sites along the Columbia River, the exception of Franz Lake where fry also dominated the catch.

Water temperature. Water temperatures at the Grays River sites did not differ greatly from water temperatures at the EMP trends sites along the Columbia River (Figure 90), at the Lewis River sites, water temperatures were consistently several degrees lower than those at the EMP trends sites. In April, water temperatures in the Lewis River ranged from 7.1-8.2°C compared to an average of 11.1°C at the trends sites; in May, temperatures were 9.3-9.4°C compared to 13.2°C; while in June temperatures were 14.7°C in the Lewis River compared to 20.5°C in the trends sites (Figure 90).

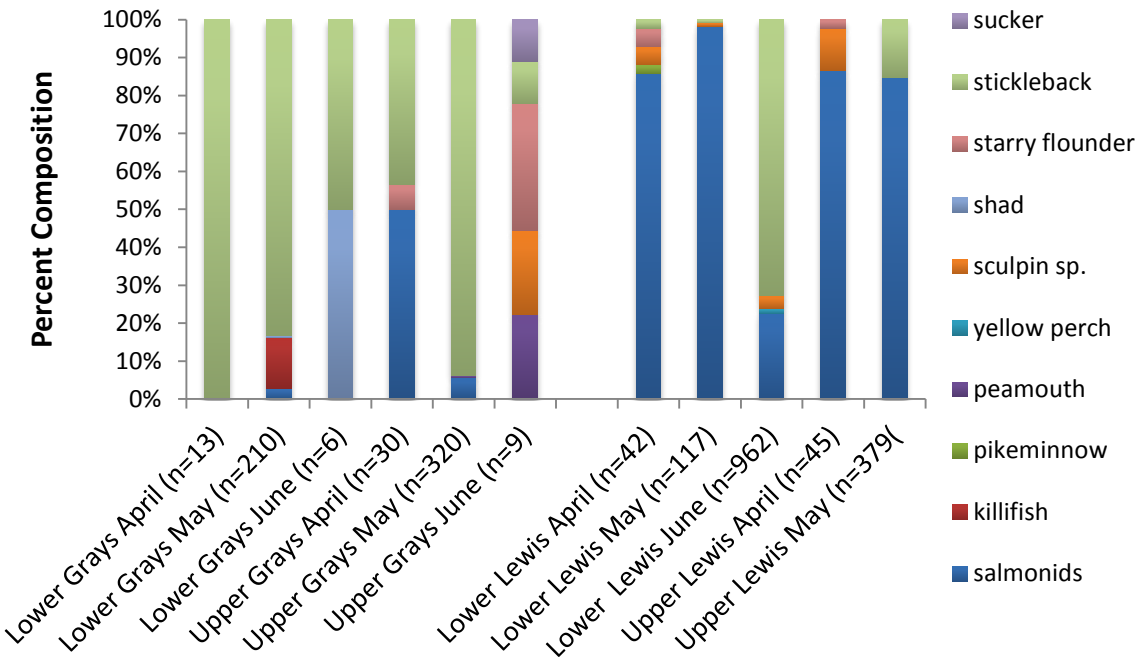


Figure 86. Fish community composition at the Grays River and Lewis River tributary sites in April, May and June 2015.

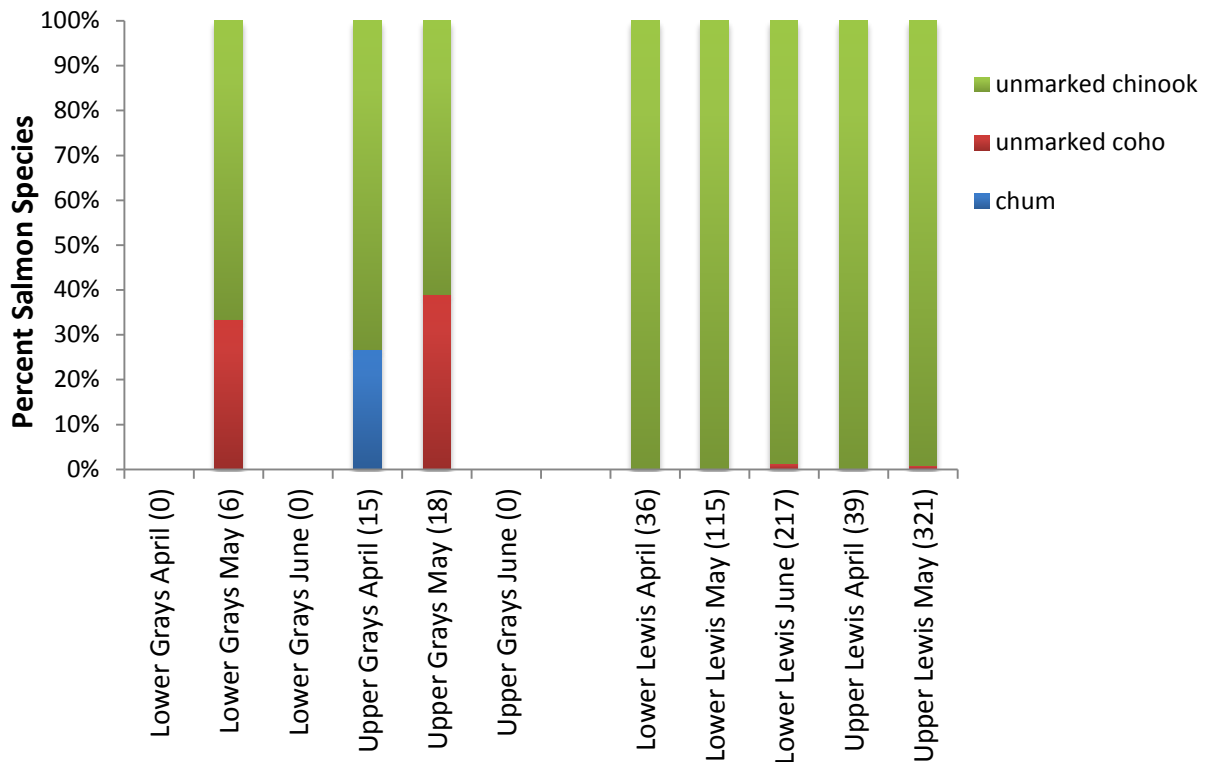


Figure 87. Percent of salmon species at the Grays River and Lewis River sites in April, May and June 2015.

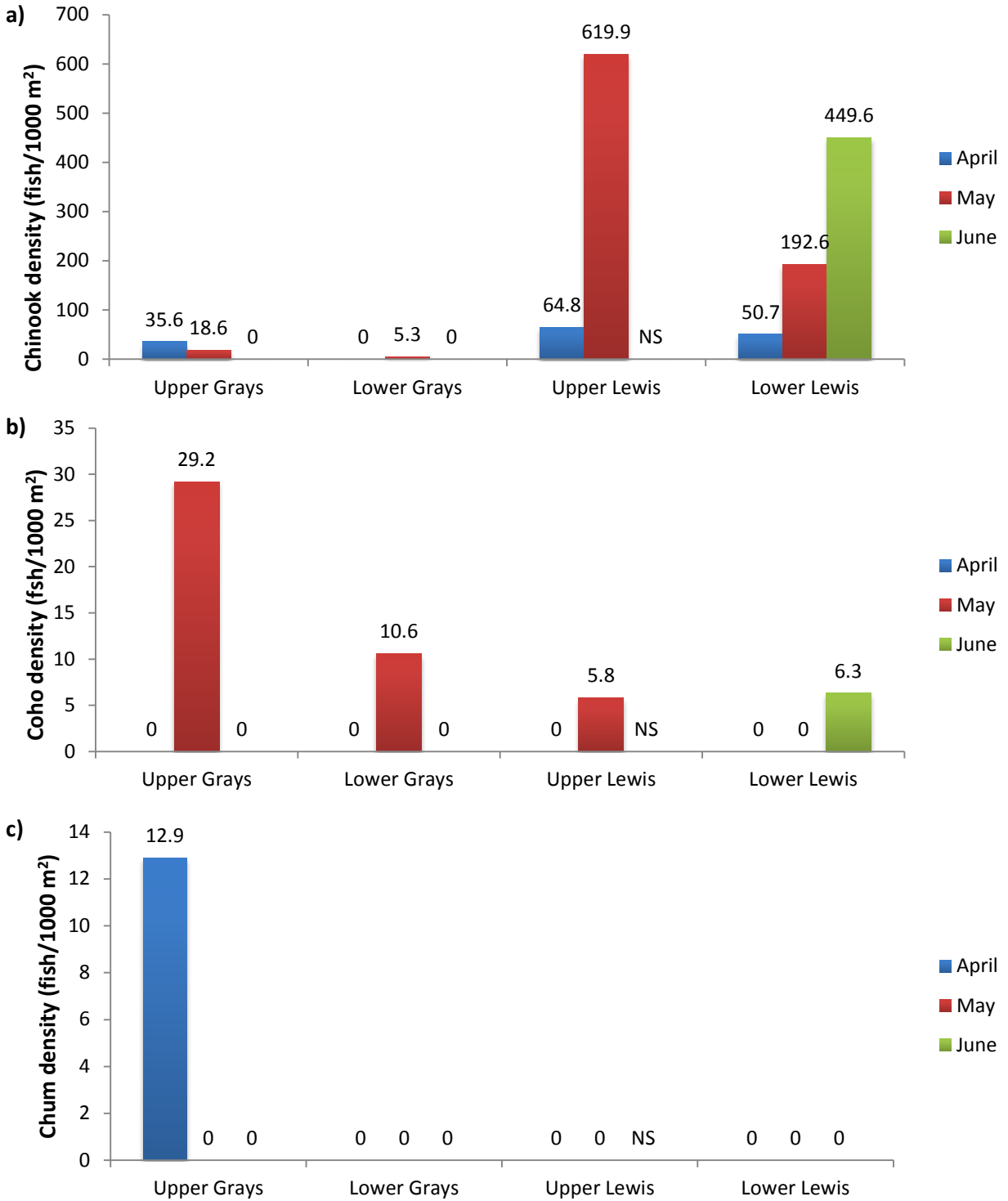


Figure 88. a) Chinook salmon b) coho salmon and c) chum salmon densities (fish per 1000 m²) at the Grays River and Lewis River tributary sites. Sampling occurred in April, May, and June 2015.

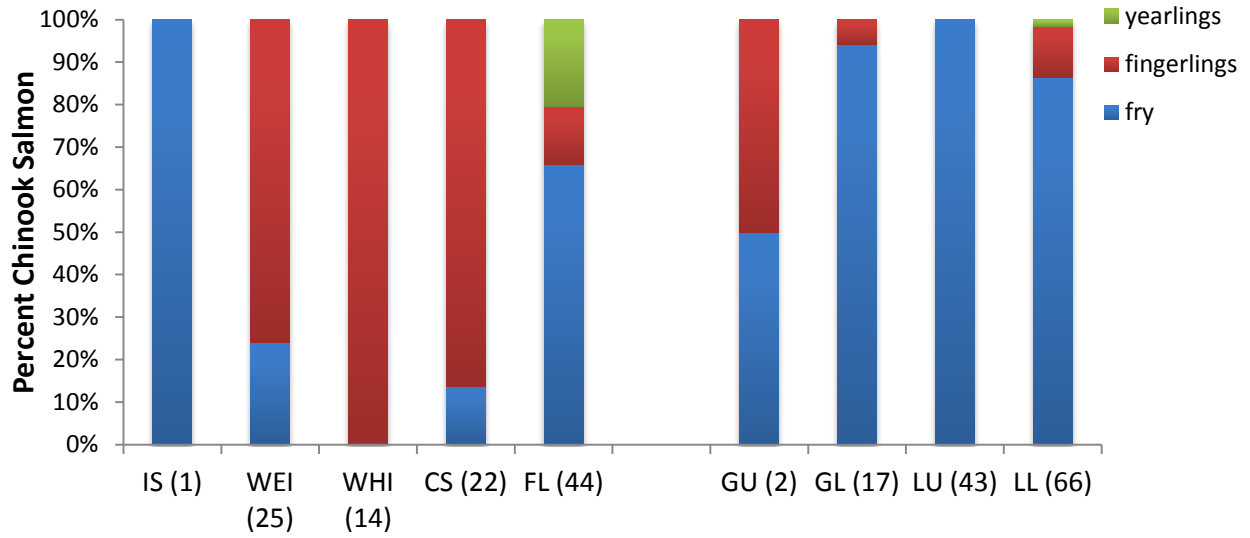


Figure 89. Size class distribution of juvenile Chinook salmon at tributary sites compared to EMP trends sites. IS = Ilwaco Slough, WEI = Welch Island; WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake, GU = Upper Grays River, GL = Lower Grays River, LU = Upper Lewis River, LL = Lower Lewis River.

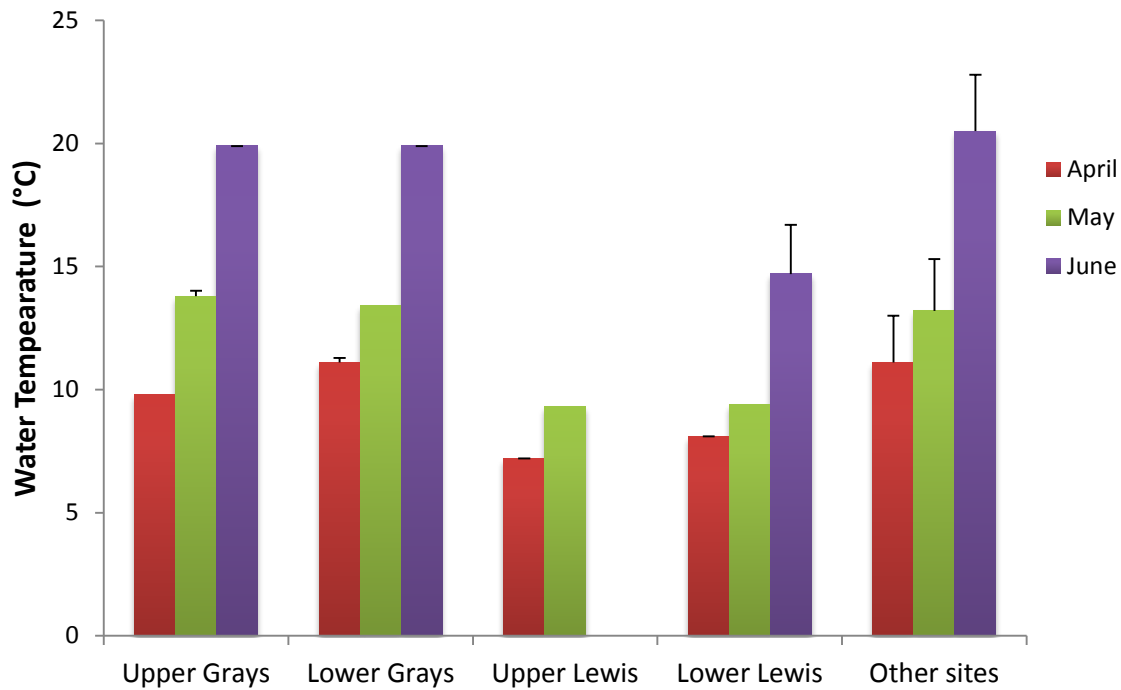


Figure 90. Water temperatures at the Lewis River and Grays River tributary sites as compared to the EMP trends sites.

3.6.4 PIT-Tag Array Monitoring of Juvenile Salmon Residence

The PIT (passive integrated transponder) tag detection system at Campbell Slough was operational between early March and early December 2015. Water depth at the site on day the array was powered up in March was unseasonably low (approximately 5 feet at the USGS Vancouver gage, 14144700) compared to previous years and the antennas were observed to be floating on the surface (Figure 91). The array operates at optimal functionality at water depth between 6 and 10 feet (on the USGS Vancouver gage); however, water levels remained very low (between 3 – 6 feet) for most of the spring. Thus, these low water levels resulted in the antennas floating on the surface of the water, operating with poor tag detection efficiency during this time. In addition, the antenna cable was severed (by aquatic rodents) in late spring, rendering the antenna temporarily inoperable.

Although the system was partially operable and collecting tag data for nearly nine months, the array recorded only one tag detection in 2015, likely due to the unusually low water levels. This detection occurred in mid-November, when water levels were nearly 7 feet on the USGS Vancouver gage. The fish was a hatchery spring Chinook (fork length = 80 mm) released from the North Santiam River in early August. It was detected at the Sullivan Dam bypass facility (Willamette Falls, Oregon City) on November 13, and then took only an additional five days to reach Campbell Slough.

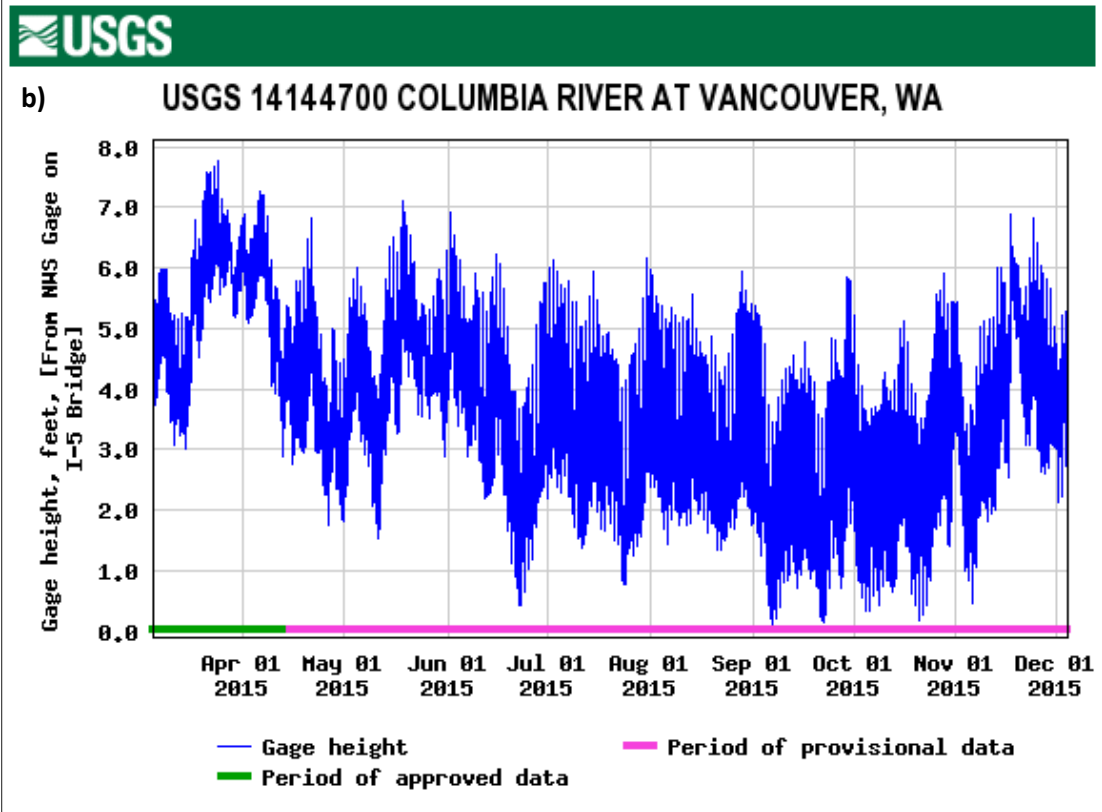
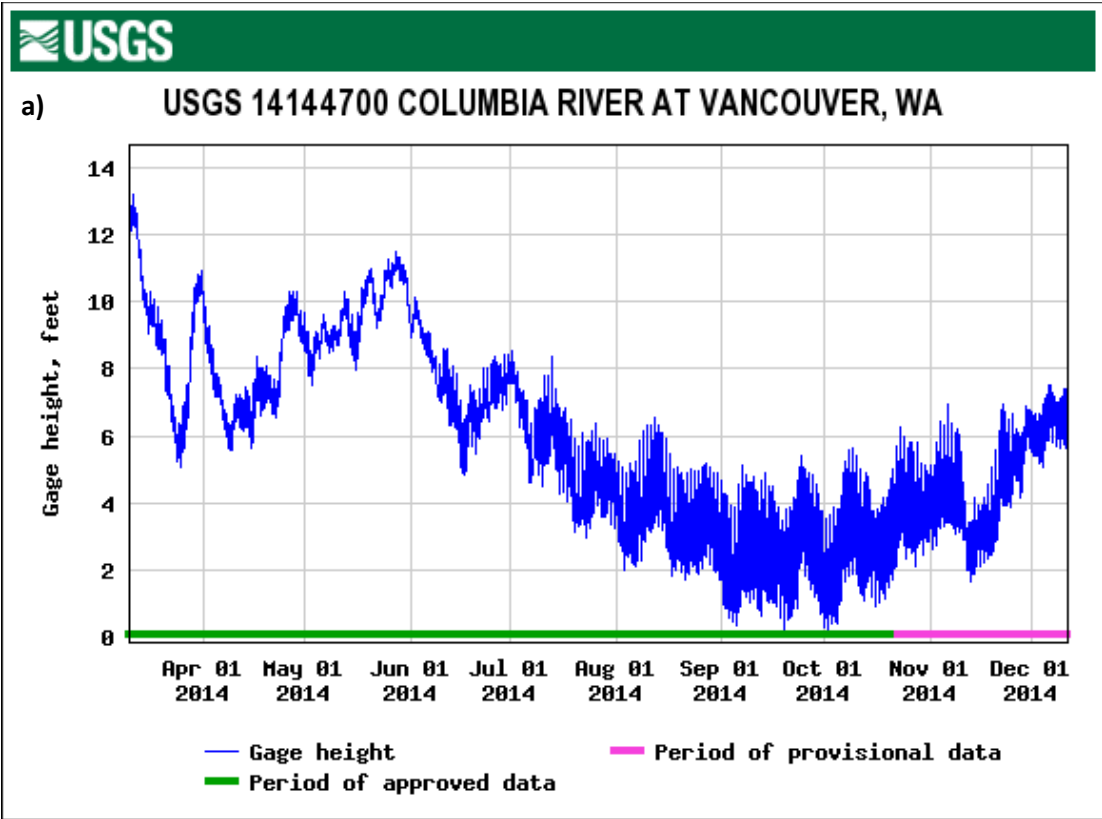


Figure 91. A comparison of seasonal water levels at the Vancouver, WA USGS gage (14144700) in a) 2014 and b) 2015.

4 Discussion

4.1 Mainstem Conditions

In 2015, river discharge was lower than usual during the period corresponding to the spring freshet, but was otherwise close to the 10-year average. Water temperatures, however, were much higher than normal, with more days having average temperatures exceeding 21°C than in previous years (2009-2014).

Turbidity and CDOM concentrations tend to reflect the input of terrestrial material, and therefore peaks in these values are indicative of storm or rainfall events. Low winter turbidity and nitrate concentrations were observed in 2015 compared to previous years, which reflects reduced runoff. The seasonality of Columbia River nitrate flux was distinctly different in 2015 compared to previous years and was manifest in lower winter values and higher summer values. Due to low Columbia River discharge, the relative importance of the Willamette River to total flow was likely larger in 2015, probably accounting for higher nitrate concentrations later in the year than in previous years at Beaver Army Terminal (RM 53), which is located downstream of the Willamette-Columbia confluence.

Variation in the timing of events such as the spring phytoplankton bloom and the peak in zooplankton abundance have potentially important implications for ecosystem function. In 2015, the peak in chlorophyll *a* was relatively late (mid-May) compared to the usual window between early March and late April (Maier et al. in review). A late peak in chlorophyll *a* was also observed in 2012, although in that year there was a separate early peak in mid-February. In both 2012 and 2015, higher-than-average flows were observed in late-January/early February and the peak in percent dissolved oxygen saturation preceded the chlorophyll *a* peak.

4.2 Abiotic Site Conditions

At each of the trends sites with the exception of Ilwaco Slough, the water was warmer in 2015 compared to the five-year mean (2011-2015). At Ilwaco Slough, higher-than-average salinities were observed in the summer, which reflected the reduced freshwater flows and corresponding salt intrusion. Nitrate concentrations were generally lower in 2015 compared to the five-year monthly means, with a few exceptions that included a high value in July at Whites Island and a high value in April at Franz Lake Slough. Lower-than-average nitrate concentrations were noted elsewhere, particularly at Franz Lake Slough in June and July. In contrast, ortho-phosphate concentrations were nearly average with the exception of high values observed at Franz Lake Slough in June and July and in May and July at Campbell Slough. The differing directions of the anomalies in nitrate vs. phosphate resulted in highly variable N:P ratios, which elsewhere has been shown to influence the structure of phytoplankton communities if one of these essential nutrients becomes depleted (Turner et al. 1998).

4.3 Habitat Structure

The hydrologic patterns and weather conditions in 2015 resulted in vegetation responses at some sites not previously measured during the 11 year monitoring period of this study. Responses were observed at the community level and at the species scale. Some species responded to the warm dry conditions with an increase in productivity, while others were adversely affected. In general, the lower and mid-marsh species thrived under these conditions, while higher marsh species response varied depending on site conditions. The most obvious increase was in wapato (*Sagittaria latifolia*), however more subtle increases in nodding beggars-tick (*Bidens cernua*) were observed in the lower estuary and rice cutgrass (*Leersia*

oryzoides) in the upper estuary. Productivity of the non-native invasive reed canarygrass (*P. arundinacea*) appeared unaffected by the change in hydrologic conditions.

The vegetation at Ilwaco Slough experienced early season senescence resulting in lower cover than measured at the site over the past five years. This may perhaps have been brought on by a combination of higher than normal temperatures, solar radiation, and salinity. Climate data recorded at the Astoria, Oregon airport indicates that the average temperature in July was 17.8°C, which was 2.1°C above the 30 year average. Additionally, there were 19 sunny days in the month of July prior to our visit. Perhaps the greatest environmental condition contributing to the early senescence is salinity. The monthly averages for May through July were considerably higher than in previous years (see Section 3.3.3). The effects of the higher salinity were also observed in the submerged aquatic vegetation, which in the past had been singularly *Zannichellia palustris*, a brackish or freshwater species. In 2015, however, the only species present was *Zostera japonica*, a non-native seagrass common in the outer coast estuaries and Puget Sound. This species has a wide salinity tolerance and is considered a euryhaline species in some west coast estuaries (Shafer et al. 2011), tolerating variable salinities similar to those found at Ilwaco Slough. Interestingly, *Z. japonica* has been shown to germinate at higher rates in lower salinity environments (Kaldy et al. 2015) and then thrives in salinities around 20 ppt (Shafer et al. 2011). This species can be invasive on intertidal mudflats and may become more widespread if salinity conditions continue to increase during the summer months due to sea level rise or reduced river flows.

In contrast to Ilwaco Slough, *C. lyngbyei* and *P. arundinacea* cover in the high marsh increased at the Secret River site compared to previous years. The low marsh site was similar to previous years, with a decrease in softstem bulrush (*Schoenoplectus tabernaemontani*) over time. The Welch Island site is very stable; none of the vegetation communities have varied much in their cover in the four years of monitoring. These two sites are the most stable of the trends monitoring sites. They are dominated by tidal hydrology, thus hydrologic variability is stable. The sites have the highest species diversity, likely due to these stable environmental conditions (Sharpe and Baldwin 2009; Janousek and Folger 2014).

At the Whites Island site, *P. arundinacea* is variable between 43% and 57%, but generally has increased over time. Conversely, *C. lyngbyei* has lower cover but is also variable and has decreased over the past seven years ($R^2 = 0.72$). This site is near the upper extent for *C. lyngbyei* in the lower river (Hanson et al. 2016), so *P. arundinacea* likely has a competitive advantage. Additionally, a portion of the *Carex* spp. at the site, especially that at slightly higher elevations (~2.0 m, CRD), is likely *Carex obnupta*. This is consistent with winter observations at the site, when *C. obnupta* has not died back, and is similar to patterns observed at other locations in the Columbia River estuary (Christy and Putera 1992) and in outer coast estuaries (Janousek and Folger 2014). At this site in the summer, the two species are difficult to distinguish from each other either due to lack of flowering parts or due to a possible hybridization between the two species and may have been misidentified as *C. lyngbyei*. In any case, overall cover of *Carex* spp. at the site is declining and *P. arundinacea*, which is already dominant at the site, may be increasing.

The upper estuary sites had the highest vegetative cover measured over the monitoring period. Cover of wapato (*Sagittaria latifolia*) was exceptionally high in 2015, with the size of the plants often exceeding one meter. Overall cover would have likely been higher at the Campbell Slough site, except that cows had grazed the *P. arundinacea*. We assume that high cover condition is a direct result of the low water levels experienced in the late spring and summer. High solar radiation may also have contributed to the high growth levels, although this may be less of a factor than water levels. When the water is over the plants they are getting much less light than when they are exposed, regardless of sky conditions. At Cunningham Lake and Campbell Slough, tidal inundation ensured that the sites were inundated most days for short periods of time, allowing the plants to thrive without the stress of excessive inundation.

The Franz Lake site experienced a species shift in 2012 after two consecutive years of high inundation. The former dominant species was *P. arundinacea*, a facultative wetland species that can tolerate some level of tidal and seasonal inundation. The replacement species was *Polygonum amphibium*, an obligate wetland species that is often found on shorelines, in shallow water, along streams and lakes (WADOE 2001). To survive high inundation levels the plant grows very tall (two or more meters) with large hollow stems that float on the surface of the water. When the water recedes the plant is prostrate on the ground. The species has persisted for three years after the high inundation years, including 2015, when inundation during the spring and summer at the Franz Lake site was very low. The plant has the ability to change its morphology, becoming smaller (~ 1 m), with narrow stems and more upright growth habit in response to changing inundation (Partridge 2001). This adaptable plant is currently outcompeting *P. arundinacea* and given its wide tolerance for changing hydrology, it is likely to continue to do so.

4.4 Food Web

4.4.1 Primary Production

4.4.1.1 Emergent Wetland Vegetation

Emergent wetland vegetation has been shown to be an important component of the detrital based juvenile salmonid food web in the lower river (Simenstad et al. 1992; Maier and Simenstad 2009; Howe and Simenstad 2015). The quantity of macrodetritus from wetlands has likely declined from historical levels due to wide spread diking and development of the floodplain (Simenstad et al. 1992; Kukulka and Jay 2003; Cannon 2015). Interannual hydrologic variability results in variable quantities of emergent marsh detritus, particularly in the upper estuary, with high flow years corresponding to reduced amounts of macrodetritus production (current study and Hanson et al. 2015). Additionally, vegetation species composition can have an effect on the quantity and quality of macrodetritus inputs in the lower river (Hanson et al. 2016).

In 2015, productivity and the resulting summer standing stock of the low marsh vegetation was higher than previously measured likely due to low inundation levels. Productivity in the high marsh was also high, but winter breakdown was variable depending on the species composition. Sites with high cover of *P. arundinacea* in the high marsh contributed less macrodetritus to the system than sites with higher native cover. The site with high native water smartweed (*P. amphibium*) cover was intermediate in organic matter contribution between the high contribution from sedge dominated marshes and lower contribution from *P. arundinacea* marshes. Inundation appears to affect the phenology of *P. amphibium*, such that in low inundation years, such as 2015, and the plant is smaller, more structurally rigid, likely with higher lignon (carbon) content.

Cover and productivity of *P. arundinacea* in the upper estuary above rkm 70 is high (Sagar et al. 2015) and few species can compete. Consequences of invasion include reduced macrodetrital inputs, lower quality of food value, and a shift in the timing of those inputs (Hanson et al. 2016). Our results corroborate these findings indicating that native sedge species, such as *C. lyngbyei*, are more productive and breakdown more quickly and completely in the winter.

While herbivore species have been shown to be adversely affected by invasive grasses (Gratton and Denno 2006; Spyreas et al. 2010), little evidence can be found for the effects on detritivores. Hanson et al. (2016) indicates that some detritivore species in the lower river may have a higher abundance in *C. lyngbyei* than in *P. arundinacea*. This may be particularly true where monotypic stands of *P. arundinacea* result in reduced detritus quality for detritivores. Hanson et al. (2015) also indicates that timing of detrital

input from the two species may also be different resulting in two contrasting food web scenarios as follows:

Native vegetation species scenario

Winter breakdown and decomposition → high food quality and availability for detritivorous juvenile salmonid prey in the spring → increased abundance of prey species during peak juvenile salmonid migration.

Invasive vegetation species scenario

Partial winter breakdown and decomposition → lower food quality and availability for detritivorous juvenile salmonid prey in the spring → lower abundance of prey species during peak juvenile salmonid migration.

To determine the validity of these contrasting processes future studies should focus on the timing of the detrital contribution *P. arundinacea* over the course of a full year or more to better understand the timing this critical component of the food web.

4.4.1.2 *Pelagic*

The mean spring-summer abundance of phytoplankton (i.e., the mean total abundance during all sampling dates during the time frame of sampling) at Whites Island was lower during high water years (2011 and 2012) than in years when discharge was closer to the 10-year average (2013 and 2014). The opposite pattern was observed at Campbell Slough and Franz Lake Slough, although some of the highest numerical abundances observed at these two sites in the summer months of 2011 and 2012 were associated with cyanobacteria, which are often much smaller than other phytoplankton and therefore contribute less to carbon biomass on a per cell basis. When considering only diatoms, total abundances in Campbell Slough were higher in 2013 and 2014 than 2011 and 2012, while mean total diatom abundance varied between years at Franz Lake Slough (highest in 2011). Total cyanobacteria abundance at both Franz Lake Slough and Campbell Slough was higher during the summers of 2011 and 2012, with relatively lower abundances during summer months in 2013 and 2014.

As in previous years, the 2015 phytoplankton species composition was largely dominated by diatoms during the spring-summer months at the trends sites. However, high abundances of small non-nitrogen fixing cyanobacteria were observed at Campbell Slough and Franz Lake Slough. The diatom community composition varied among sites and among seasons and years. For example, during two high-water years (2011 and 2012), the diatom *Asterionella formosa* dominated the diatom assemblage during the spring months leading up to the freshet (April–May) followed by *Aulacoseira* spp. (May–June). Small (<20 µm) centric diatoms, both solitary and colonial, were sometimes abundant and tended to be present at high abundances during the early spring (notably in April 2011 at Franz Lake Slough). *Skeletonema potamos* was abundant each year at all three sites, although to a lesser extent at Franz Lake Slough compared to Campbell Slough or Whites Island. This species tended to dominate the diatom community later in the summer (June–July).

Changes in nutrient supply and turbidity influence the size and type of standing stocks of primary producers, which ultimately influence food web characteristics and energy flow through aquatic ecosystems. In 2015, we observed high densities of the cyanobacterium, *Merismopedia* in Campbell Slough and Franz Lake Slough, which was not noted in past years. Cyanobacteria do not tend to be efficiently grazed by invertebrates and provide little nutritional value to consumers.

Diatom assemblages were dominated by a small number of species (1–2 species) during 2011 and 2012 in contrast to 2013 and 2014, where several diatom species co-dominated, particularly at Campbell Slough and Whites Island (Tausz 2015). *A. formosa* was consistently most abundant ($>1,000$ cells mL^{-1}) during early spring at Whites Island from 2011–2014, but it was present at low abundance during the spring of 2015. Instead, *Aulacoseira* sp. were present at high abundances throughout 2015 at Whites Island. Diatom abundance and diversity at Franz Lake Slough was very low in 2015, and was comprised almost entirely of small (<10 μm) *Nitzschia* sp.

4.4.2 Secondary Production

Zooplankton were more abundant in Campbell Slough and Franz Lake Slough, similar to previous years. However, while zooplankton abundances are generally higher in summer rather than the spring, zooplankton—including crustaceans (copepods and cladocerans)—were present at high densities in April at Campbell Slough. Changes in the springtime abundance of macrozooplankton could influence both the lower food web (by reducing standing stocks of primary producers through grazing) and higher trophic levels, since they can be directly consumed by fish (Merz 2001).

4.5 Macroinvertebrates

Multidimensional scaling of benthic samples provided some evidence of a transition in the macroinvertebrate community along the estuarine gradient. Annelid worms were abundant across all sites; however, the species composition at Ilwaco Slough was distinguished from other sites by a greater presence of amphipods and isopods. Further upriver, Campbell Slough and Franz Lake were characterized by lower densities of chironomids and other dipterans, and a greater presence of Collembola and insects other than Diptera compared to the other sites.

Neuston samples varied among sites, months, and the two habitats sampled. The densities and biomass of invertebrates were higher in the emergent vegetation habitats than in the open water habitats, except for average densities from Whites Island in June where approximately 20 individuals were collected per meter towed in emergent vegetation and approximately 35 individuals were collected per meter towed in open water. Cladocerans strongly dominated open water counts from Whites Island in June, while the emergent vegetation was characterized by a more diverse assemblage of cladocerans, chironomids and other Diptera, Hemiptera, and Collembola, among other taxa. Overall, average density of all invertebrates was 22.31 times greater in the emergent vegetation than in the open water. This difference was very near that previously reported for the EMP sites, where overall densities in the emergent vegetation were 20.3 times those of the open water (Sagar et al. 2015). The difference in density between habitat types, however, varied considerably in 2015 and was often well below the 22.31 average (e.g. the median difference was 2.82).

Average densities of all invertebrate taxa collected by neuston tows in 2015 were consistently greater than those from previous years in both the emergent vegetation and open water habitats. While interannual variations in abundance of invertebrate species are often observed (Boesch et al. 1976, Desmond et al. 2002), the consistency across all sites and months where a comparison was possible is notable. The same pattern was not seen in comparison of benthic samples to previous years, where average monthly densities were generally consistent across years.

In 2015, as in previous sampling from the EMP sites, amphipods and chironomids had the highest contribution to the index of relative importance (IRI) of juvenile Chinook salmon diets. Amphipods in particular dominated diets at lower reach sites (Ilwaco Slough, Welch Island, and Whites Island). High proportions of amphipods occurred in juvenile Chinook diets despite their relatively low presence in benthic core and neuston samples from Welch Island and Whites Island. Also, while contribution of

amphipods to the IRI of fish from Campbell Slough and Franz Lake was minimal, their very low presence in the environment resulted in high selection values for this prey. Fish primarily consumed chironomids and other dipteran taxa from Campbell Slough and Franz Lake, and electivity values from these sites indicated a strong preference for these taxa as well as for hemipterans.

The instantaneous rations for juvenile Chinook were similar to those found in a separate study of juvenile salmon diets in the Columbia River estuary (Spilseth 2008). The highest rations were found from sites where fish fed predominantly on amphipods, particularly Ilwaco Slough in April. The lowest rations were seen at Franz Lake in May where chironomids dominated diets. Chironomids and other Diptera are highly selected prey and have been shown to be almost twice as energy rich as crustacean prey, such as amphipods (Gray 2005). Abundance of dipterans was low in benthic core samples from Franz Lake, although their density in neuston samples was high in April and moderate in May. Therefore, while fish consumed high energy prey at Franz Lake, prey quantities or foraging access may potentially inhibit feeding intensity at this site.

4.6 Fish

Trends Sites

In 2015, our sampling focused on revisiting five trends sites (Ilwaco Slough in Reach A, 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 changes in salmon occurrence and health in these areas. In addition, we conducted a pilot study to assess salmon occurrence in two tributaries of the Columbia River, the Grays River and the Lewis River, which could potentially provide valuable habitat for migrating juvenile salmon.

At the trends sites, many of the patterns we observed in earlier years (Sagar et al. 2013, 2015; Hanson et al. 2014) remained consistent in 2015. Patterns of fish community composition remained relatively stable, with low species diversity and richness and few non-native or predatory fish species at Ilwaco Slough, Welch Island, and Whites Island. As in previous years, we observed higher species richness and diversity, as well as higher proportions of non-native species and predatory fish species at Campbell Slough and Franz Lake. The proportion of non-native species, however, was higher than usual at Ilwaco Slough in 2015, with a large number of banded killifish caught at the site.

In 2015, as in previous years, Chinook salmon were the dominant salmonid species at all sites. Unmarked fish dominated catches at Ilwaco Slough, Welch Island, Whites Island, and Franz Lake, while higher proportions of marked hatchery fish were observed at Campbell Slough. We do not yet have genetics data from fish collected in 2015, but Chinook stock composition in 2014 (the most recent year from which results are available) was consistent with previous years. West Cascades fall Chinook and Spring Creek group fall Chinook, both from the Lower Columbia River ESU, were the dominant stocks, but interior stocks including Upper Columbia summer/fall Chinook and Snake River fall Chinook were also observed, especially at Campbell Slough and Franz Lake.

We did, however, observe some unusual patterns of juvenile salmon occurrence in 2015, possibly associated with the prolonged summer drought and high water temperatures observed throughout the region during this sampling season. For example, juvenile salmon densities were lower than in previous years at Welch Island and Whites Island and the proportion of fry observed was lower as well. The large numbers of unmarked Chinook salmon fry that are typically caught at these sites early in the sampling season were not present. Juvenile chum salmon were also absent from our sampling sites in March and April, when they are usually most abundant (Sagar et al. 2015, Hanson et al. 2015), but were found in small numbers in May. Additionally, we observed especially low densities of juvenile salmon during the

summer months. Densities were lower in June compared to most years, and Chinook salmon were absent from all sites in July and September, when densities are typically low but a small number of fish are generally present. The absence of juvenile Chinook salmon after June is likely related to the higher water temperatures during the summer. By June, the water temperatures at Welch and Whites Island were 19°C, while in past years temperatures were in the 13-17°C range. At Campbell Slough and Franz Lake, the temperature was greater than 22°C by June. Juvenile salmon were also absent from all sites in November and December. While salmon are not generally abundant in these months, in past years we typically observed small numbers of coho salmon and spring Chinook salmon (Sagar et al. 2013, 2015; Hanson et al. 2014) at some of the trends sites. These fish were not present in 2015.

In addition to monitoring Chinook salmon occurrence at the EMP trends sites, we also monitor several indicators of salmon health, including condition factor, lipid content, growth rates, and chemical contaminants. For most of these parameters we do not yet have new data to present for fish collected in 2014 and 2015. However, we did measure condition factor, and found no evidence of reduced condition in juvenile salmon sampled in 2015, in spite of the high summer temperatures. Condition factor was within the normal range at all of the sampling sites. Overall, condition factor tends to be highest in fish collected from Campbell Slough and lowest in fish collected from Franz Lake, and this pattern persisted in 2015. Thus, altered condition factor associated with the high temperatures in 2015 was not clearly evident.

Similarly, Roegner and Teel (2014) found little evidence for reduced condition in juvenile Chinook salmon from tidal freshwater sites in the Lower Columbia River, although they did observe reduced density during periods of high summer temperature. They presented two possible explanations for this observation. First, they suggested that fish experience higher metabolic rates under elevated temperatures, which could be compensated for by increased food intake, allowing positive condition to be maintained. In support of this hypothesis they cited evidence that growth can be maintained at temperatures above 19°C, if oxygen and food ration are sufficient (Brett et al. 1982; Clarke and Shelbourn 1986; Marine and Cech 2004). Such conditions generally occur in salmon rearing and migration habitats and adequate food sources would likely be available during the summer period (Bottom et al. 2011). Some other studies (e.g., Sommer et al. 2001) also support a positive relationship of temperature on growth and condition of juvenile salmon, assuming food resources are adequate. However, it is unclear whether this would be true under the extreme temperature conditions of a drought year. A second explanation proposes that Columbia River Chinook salmon stocks have different migration periods and are thus exposed to – and tolerant of – different thermal regimes. Roegner and Teel (2014) found that the majority of fish sampled during periods of high temperature had originated from genetic stocks that historically had a summer juvenile migration period (West Cascade tributaries fall and upper Columbia River summer/fall stocks; Howell et al. 1985). It might be that these summer migrating stocks are better adapted to high temperatures than stocks passing through the estuary earlier in the year.

Based on 2015 sampling results, even those stocks usually captured later in the summer were absent from tidal freshwater sites, suggesting that extreme temperatures triggered an earlier migration time from off-channel habitats to the ocean, even for stocks that might be more tolerant. The lack of any clear impact on condition factor may be because all juveniles migrated out of the estuary as temperatures became too extreme for each stock to maintain positive condition. Further examination of the genetic data for fish collected in 2015 may reveal if stock composition was different in this year of unusually high summer temperatures, as well as how lipid content and somatic growth rates are influenced by these unusual weather patterns. It should also be noted that the failure to observe poor condition in juvenile Chinook salmon during periods of high temperature does not necessarily mean that growth and survival will not be affected later on. In fact, in a study from British Columbia, Holtby (1988) found that warmer stream temperatures associated with clear-cut logging were actually beneficial to juvenile coho salmon in the short term, contributing to higher growth rates and increased overwinter survival. However, marine

survival of these fish was reduced, presumably because the earlier smolt migration to the ocean of these rapidly growing individuals was not optimally timed for availability of ocean prey and avoidance of predators (Holtby 1988; Mantua et al. 2010). Moreover, ocean conditions themselves could be detrimental during warm periods. Indeed, Daly and Brodeur (2015) found that warm ocean regimes were associated with decreased condition factor and smaller body size in yearling Chinook salmon collected in the Columbia River plume and coastal Pacific Ocean waters. Continuing work will help us to close the loop on this important life history transition and should help resolve an apparent contradiction. It stretches credibility that high temperatures have nothing to do with condition factor in migrating juvenile salmonids. Apparently, this effect is manifest sometime after ocean entry.

Somatic growth data for juvenile Chinook salmon collected in 2015 are currently under analysis and data are not currently available so we can only speculate on how extreme temperatures will influence salmon growth and migration timing. Our expectation is that fish condition remained consistent within the tidal freshwater rearing habitats, but that fish moved out of the estuary earlier.

In this report we provide an updated analysis of juvenile Chinook salmon somatic growth rates including samples from 2013. As in the most recent annual report for this program (Hanson et al. 2014), we learned that salmon growth rates are influenced by multiple factors, including fish length, month of collection, unmarked vs. marked origin, genetic stock, reach or site of collection, distance from the mainstem, and sampling year. Growth rates tended to be higher in larger fish, a relationship which accounted for overall higher growth rates in marked than unmarked fish. Even after accounting for differences in fish length, there were differences with site or reach of collection, with lowest growth rates consistently being found in fish collected from sites in Reaches B and C. There was also a significant difference in growth rates among stocks, after accounting for differences in fork length, with West Cascades fall Chinook salmon showing significantly lower growth rates compared to Spring Creek Group fall Chinook or Upper Willamette River spring Chinook salmon. The West Cascades fall Chinook salmon stock is typically the most commonly observed in Reaches B and C, thus the predominance of this stock in the area may contribute to lower growth rates in these reaches. Clearly some of these effects are confounded and further sample collection and analysis over a range of environmental conditions is required to dissect the principal drivers of juvenile growth among Chinook salmon stocks.

Two EMP sites, Campbell Slough and Whites Island, have growth rate data available for four years or more and we identified some compelling trends. At Campbell Slough, growth rates have declined since the site was first sampled in 2007, while at Whites Island, growth rates showed an increase in 2013. Continued data collection is necessary to determine whether these differences truly represent increasing or decreasing trends, or simply reflect the year-to-year variability in growth at these sites.

Tributary Sites

In addition to the annual sampling efforts at the trends sites, in 2015 we sampled juvenile salmon in two tributaries of the Columbia River (lower Grays River and lower Lewis River) to collect preliminary information on how these areas function as rearing habitat for threatened and endangered juvenile Columbia River salmon. We found that both of these areas were being used by juvenile salmon, with especially high densities of juvenile Chinook salmon in the lower Lewis River. These fish were predominantly fry and of smaller size than those found at the trends sites elsewhere in the lower Columbia River study area. Although genetics data for these fish are not yet available, the small size of the juvenile Chinook salmon suggests that they are most likely locally produced and are almost certainly part of the Lower Columbia River ESU. The residence time of juvenile Chinook salmon from the area is not known, but we do know that they were present from April through June.

Interestingly, we found that water temperatures were consistently lower at the Lewis River site than at either the Grays River or the EMP trends sites. This was especially significant during the summer months, when temperatures in the Lewis River were 14.8°C in June, whereas temperatures at the trends sites were above 20°C and unsuitable for juvenile salmon. In addition, mainstem water temperature recorded at RM-122 were 19.6°C on June 15, also considerably higher than temperatures observed in the lower Lewis River on that same day. These findings suggest that tributary habitat such as that found in the lower Lewis River is not only being utilized by juvenile salmon, but could serve as an area of cool water refugia during periods of summer drought.

4.7 Implications for Climate Change

Exceptionally high temperatures characterized regional conditions in 2015. A persistent atmospheric ridge over the northeast Pacific resulted in a large warm temperature anomaly in the ocean (>3 standard deviations higher than normal), which extended from Mexico to Alaska (Bond et al. 2015). These anomalous conditions had severe ecological impacts in the ocean, including low primary productivity (Whitney 2015), range shifts in copepod species (Peterson et al. 2016), death and starvation in seabird (Cassin's auklet) populations (Opar 2015), whale (NOAA 2016a) and sea lion mortality events (NOAA 2016b) between 2013 and 2015, and an unprecedented margin-wide harmful algal bloom (NOAA 2016c). These harmful ecosystem effects (i.e., low biological productivity) arose from the consequences of a warm ocean: enhanced stratification, low nutrient availability, and likely increased metabolic rates. In the Columbia River, warming would be expected to have similarly harmful consequences, including thermal stress and increased metabolic rate among aquatic organisms, while changes in runoff influence turbidity and nutrient supply to the river. Such anomalies observed in 2015 may be indications of how processes may change in the future with climate change.

Environmental conditions observed in 2015 seemed to affect vegetation species composition and plant morphology at various locations in the lower river. For example, lower and mid-marsh species thrived under the warmer, lower flow conditions, while higher marsh species response varied depending on site conditions. Cover of wapato (*S. latifolia*) and nodding beggars-tick (*B. cernua*) increased the lower reaches and rice cutgrass (*L. oryzoides*) increased in the upper reaches. We also noted that the wapato at some sites was unusually tall and water smartweed (*P. amphibium*) was smaller than usual with narrow stems and more upright growth habit in response to reduced inundation. At Ilwaco Slough, an early season senescence resulted in lower cover at that site compared to the past five years of observation and may have been a result of higher than average temperatures, solar radiation, and salinity. Salinity values, in particular, were higher in 2015 than in past years. We noted the presence of the invasive *Z. japonica*, a euryhaline species found in some west coast estuaries and is capable of tolerating variable salinities (thriving in salinities around 20 ppt), similar to those found at Ilwaco Slough. This species may become more widespread if salinity conditions continue to increase during the summer months due to sea level rise or reduced river flows. Reduced inundation (as a result of lower flows and freshet timing shifts), warm temperatures, and increased salinity in the lowest reaches of the river could have a large effect on vegetation community and habitat structure, thus potentially greatly affecting the base of the food web in the lower river.

The drought year of 2015 was distinctly different than previous years in terms of river temperature (higher than normal) and in water level (lower than normal). At reduced water level, off-channel habitats become disconnected from the mainstem, as evidenced by the development of distinct biogeochemical characteristics and patterns in plankton community composition. With higher temperatures, the threat of cyanobacteria blooms becomes greater, which has the potential to negatively influence food webs in three ways. First, cyanobacteria constitute a low-quality food source for herbivores compared to diatoms, which possess high concentrations of essential fatty acids. Second, many cyanobacteria species produce potent

hepato- and neurotoxins, which could negatively influence the health of juvenile salmon. Finally, cyanobacteria tend to be found near the water surface, as opposed to diatoms, which are denser and thus more likely to be found near the bottom. Since many chironomids are bottom feeders, a reduction in the supply of sinking diatoms many result in a reduced supply of organic matter to fuel growth among these preferred salmon prey.

In addition, poor water quality in the form of reduced dissolved oxygen, large fluctuations in pH, and high concentrations of chlorophyll may negatively impact the survival of juvenile salmon. Poor water quality tends to be observed where flushing rates are low; thus, at low river discharge, threats from poor water quality become more severe. While operations at the hydroelectric dam can ensure that water flows remain high enough to overcome extreme threats at present, in the future additional demands for water withdrawal from the Columbia River during extensive drought periods may overwhelm the system and result in off-channel areas being subjected to slow flushing and impaired water quality.

In combination with ongoing land-use changes, the Columbia River is expected to experience changes in freshwater flow and temperature due to climate change, which will elicit additional challenges to the maintenance of key estuarine habitats (Bottom 2005, ISAB 2007). The ISAB Climate Change Report (2007) describes sea level rise, sediment deposition, and changes in the upstream extension of the salt wedge as climate change impacts that could alter estuarine ecosystems. Climate change effects specific to the macroinvertebrate community in the lower Columbia River and estuary have not been modeled, and it can be difficult to isolate the influence of climate change on benthic communities when other stressors such as eutrophication and non-native species introductions are developing concurrently (Schückel and Kröncke 2013). However, we can speculate some general outcomes based on observed effects of climate change on wildlife. Such effects, particularly those applicable to macroinvertebrates, include: changes in species distributions, community changes (composition changes and increased spread of invasive species), alterations to life-history traits for particular species (emergence timing, sex ratios), and the direct loss of habitat (Hogg and Williams 1996, Mawdsley et al. 2009, Filipe et al. 2013). Studies on the macroinvertebrate response to climate change often conclude that effects are species-specific with no overall consequence on entire taxonomic groups (Hogg and Williams 1996, Burgmer et al. 2007, Lawrence et al. 2010, Filipe et al. 2013). Warmer water and altered hydrodynamics may force spatial or temporal contractions of specific species with narrow tolerances, while at the same time allowing for expansions of others that can tolerate a wider range of conditions, making it difficult to predict outcomes at individual sites. Indeed, results varied in a study that modeled the predicted response of freshwater macroinvertebrates to the independent and combined effects of climate change and land-use change, some scenarios resulted in increases in richness, while others showed no change or declines (Mantyka-Pringle et al. 2014).

Chironomids are able to adapt to a variety of ecosystems and conditions, characteristics that are consistent with the ubiquitous nature of the family (Cranston 1995, Ferrington 2008). In the Columbia River estuary, this diversity of characteristics is exemplified by distinct patterns in habitat associations and emergence timing of different chironomid genera (Ramirez 2008). Diversity in the chironomid family might translate into an overall resilience of these insects to changing conditions. However, in a recent study investigating the effects of the invasive Reed Canarygrass on macroinvertebrates in the lower Columbia River, densities and biomass of chironomids as well as other Diptera tended to decline in areas dominated by the invasive species (Hanson et al. 2016). Also, in an experiment that replicated expected global warming, a relatively small increase in the temperature of a natural stream resulted in decreased chironomid densities (Hogg and Williams 1996). Research from the western United States and other regions, mainly Europe, have documented recent changes in chironomid communities in alpine and subalpine lakes that correspond with global warming (Battarbee et al. 2002; Solovieva et al. 2005; Porinchu et al. 2007a, 2010; Reinemann et al. 2014). Radiative forcing that accompanies the changing temperature has been implicated in some of these studies as a driver of the observed shift in chironomid community

composition (Solovieva et al. 2005; Porinchu et al. 2010; Larocque-Tobler et al. 2011). Presumably, Columbia River estuary wetland habitats will experience changes in chironomid communities related to warming climate.

Changes in the composition of insect assemblages occur quickly in space and time, providing a sensitive indication of variation in ecosystem conditions, yet also making interpretation of community patterns particularly susceptible to decisions on the scale of observation. To effectively monitor change, consideration of different spatial scales and indicators (richness, community composition, species abundance) need to be made. However, to analyze tolerances of different species would require extensive research and knowledge. For example, when sampled comprehensively, chironomid richness often approaches 80 or more species per site (Ferrington 2008). Thus, while it may be possible to discern effects of climate change on density and biomass of chironomids and other insects by comparing with previous data, measures of species richness will probably not be possible because chironomids have not been identified past family level in most studies in the Columbia River and elsewhere in the region. Some would argue a more strategic approach would be to shift the focus from specific ecosystem assemblages to identifying and maintaining ecosystem functions and processes (Harris et al. 2006).

The 2015 weather pattern of low snow pack and high summer temperatures are likely to become increasingly frequent as a consequence of climate change. These altered weather patterns are predicted to have a variety of adverse effects on Pacific Northwest salmon (Beechie et al. 2013; Mantua et al. 2010). Some of the likely impacts on migrating juvenile salmon are listed below:

- Further increases in summer water temperatures might affect juvenile salmon growth and increase mortality rates.
- Thermal stress caused by higher summer temperatures and reduced flows could lead to earlier juvenile migration to the ocean, which may not be optimally timed to align with ocean prey abundance and predators presence, thus decreasing marine survival.
- Low summer river flows might contribute to increased temperatures and pose risks to the quality and quantity of favorable rearing habitat, decreasing rearing habitat capacity for juvenile salmonids.
- Increased winter flooding and higher peak winter flows may increase scour of eggs from gravel, and could also increase the mortality of juveniles by reducing the availability of slow-water rearing habitats and displacing juveniles from their preferred habitats.
- Loss of spring snowmelt may alter survival of eggs or emergent fry and negatively impact the migration success from formerly snowmelt dominant streams where seaward migration timing has evolved to match the timing of the freshet.
- Elevated water temperatures in spring confer a competitive advantage to warm water predators, such as smallmouth bass, that can consume significant numbers of juvenile salmonids.

In 2015, we clearly observed how low summer flows and increased summer temperatures decreased the rearing capacity of the tidal freshwater emergent marsh areas that we sample in the EMP. By June, temperatures at all trends sites were above optimal levels for juvenile salmon, and low water levels reduced the amount of available habitat for juvenile salmon at some sites. The reasons are not clear for the lack of early spring fry at Welch Island and Whites Island, the apparent delay in outmigration of chum through our sites, and the absence of coho and spring Chinook salmon in November and December, but these outcomes would be consistent with some predicted climate change implications. For example, warmer winter temperatures could have led to premature emergence and higher mortality for alevins. Alternatively, loss of spring snowmelt might have altered survival of eggs or emergent fry, negatively impacting the success of juvenile salmon migration. Whatever the case, our results suggest that factors associated with climate change, such as increased summer temperatures and low flows, reduced

snowpack, and warmer winter temperatures, will not be favorable to salmon productivity and will make recovery of listed salmon a greater challenge. Our results suggest that these effects will be particularly acute in the critical tidal freshwater rearing habitats of the Lower Columbia River.

5 Adaptive Management & Lessons Learned

Long-term monitoring data provide a range of natural variability to which action effectiveness monitoring data can be compared. By collecting similar metrics between the Action Effectiveness Monitoring Program (AEM) and the EMP (i.e., benthic macroinvertebrates, water temperature, hydrology, channel cross sections, and vegetation cover), we are able to use EMP trends sites as reference sites, allowing for the placement of results from restoration actions and AEM into context with reference conditions. For example, in 2015 the Estuary Partnership began collecting benthic macroinvertebrate samples as part of Level 2 AEM. Given that the availability of preferred salmon prey (e.g., Chironomidae) during the juvenile salmon migration period is essential for fish survival, the benthic macroinvertebrate data collected under the monitoring program provide a better understanding of when prey are most abundant and which prey taxa are available to juvenile salmon at restoration sites before and after construction (see additional detail in Schwartz et al. 2016). Integrating EMP and AEM benthic macroinvertebrate results provides a much broader spatial and temporal representation of the community structure and prey availability in the lower river.

EMP results are useful for developing restoration strategies and designing projects. For example, EMP habitat and hydrology data from Cunningham Lake and Campbell Slough are being used to build a model for the planning phase of a habitat restoration project at Buckmire Slough (Clark County, WA). The model is intended to predict the inundation regime and vegetation zones that will develop at the site post-restoration.

One priority in the region is to determine methods to control *P. arundinacea* in an effort to reduce the likelihood of invasion at restoration projects. The effect of this species on the food web is not well quantified, but likely has changed the dynamics of the timing and prey availability in two thirds of the estuary where it dominates. We recommend studying food web processes in future studies by focusing on the timing of the detrital contribution from *P. arundinacea* over the course of a full year (or longer) to better understand the timing of decomposition and availability to salmon prey.

Monitoring mainstem and site-specific abiotic conditions are useful for detecting changes or impairments to water quality. Understanding flux of inorganic and organic material is important for pollution assessment, detecting potential eutrophication, and evaluating nutrient loading, which may affect ecological processes. Through the EMP (along with partner programs such as CMOP), we have been able to monitor conditions over multiple years in the lower Columbia River that show annual trends and potential effects on aquatic life and food web dynamics. Establishment and continued maintenance of such long-term datasets are invaluable to resource managers to track how water quality changes over time, allowing for decisions to be made about water management and aquatic species conservation.

The 2015 results showed some differences in the occurrence and abundance of juvenile salmon. The warmer than average water temperatures that occurred during the study period suggest a limited suitability of some of the habitats in our study area for juvenile salmon under such conditions. In particular, we observed fewer fish at some of the sites, with abundances decreasing earlier in the season than usual as water temperatures warmed above 22°C in June. Our results highlight the importance of cold water refugia areas to both juvenile salmon (particularly for later migrants such as subyearling

Chinook that pass through the lower river later in the summer) and for adults migrating upstream to spawn. Protection of existing cool water refugia areas and employing restoration efforts that create additional cool water refuge areas are needed in the Columbia River to benefit salmon and promote survival. Further data collection under the EMP as well as other cold water refugia surveying efforts will inform where cold water restoration is needed and how to best implement such actions.

Fish genetics data collected under the EMP show temporal and spatial patterns of juvenile Chinook salmon habitat use in the lower Columbia River for stocks from throughout the Columbia Basin. As we continue to learn more about the migration timing of various juvenile Chinook salmon stocks and how these stocks use and benefit from lower river habitats, we will also improve our ability to manage these stocks. Ensuring suitable habitats are available for upriver stocks to use in the lower river and estuary will contribute to health and survival of these populations.

Hydrology data from long-term monitoring provide an indication of how the river functions over time and a “relative baseline” to which future data may be compared given the predicted changes associated with climate change (e.g., reduced snowpack, freshet shifts, and resulting effects on ecological function). Differences in the physical, biogeochemical, and ecological characteristics of habitats used by juvenile salmon during the drought year of 2015 compared to other years may offer insight into how environmental factors (e.g., warmer water temperatures, lower flows) play into juvenile salmon survival. Patterns of juvenile salmon occurrence in 2015 may offer some insight into changes that could be expected with climate change, such as alterations in juvenile salmon migration timing and habitat use. It would be worthwhile to perform a modelling exercise using long term monitoring data to help predict how the ecological conditions of the lower Columbia River may respond to a changing climate. If 2015 conditions are an indication of what could occur in the future, it would mean adjusting how resources in the lower river are managed (e.g., habitat restoration techniques, fisheries management, invasive species monitoring and control, flow management).

The EMP is a multi-faceted program and results are shared with other resource managers in the region and applied to management decisions. The Science Work Group is composed of over 60 individuals from the lower Columbia River basin representing multiple regional entities (i.e., government agencies, tribal groups, academia, and private sector scientists) with scientific and technical expertise who provide support and guidance to the Estuary Partnership. Results from the EMP are annually presented and discussed at a Science Work Group meeting. In addition, during the past year, EMP results were presented at scientific conferences including the Washington State Salmon Recovery Conference, American Fisheries Society Annual Meeting, Coastal & Estuarine Research Federation Biennial Conference, USACE Anadromous Fish Evaluation Program Annual Review, and the Columbia River Estuary Conference. Data are often provided to restoration practitioners for use in restoration project design and project review templates (e.g., ERTG templates). Finally, data from the EMP are used as reference conditions for comparison to results from the AEM Program (see Schwartz et al. 2016).

6 References

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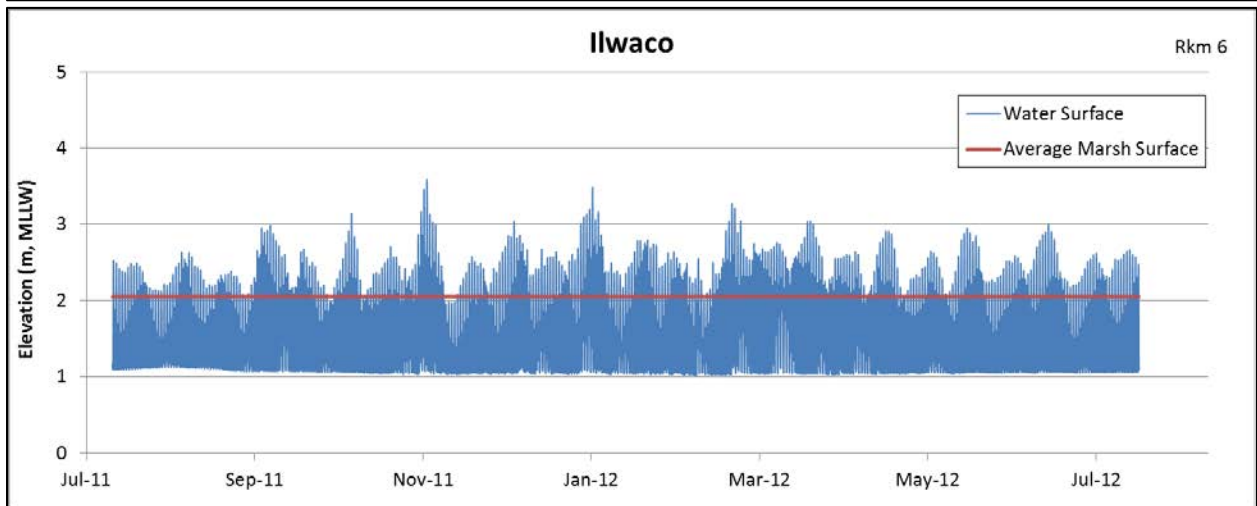
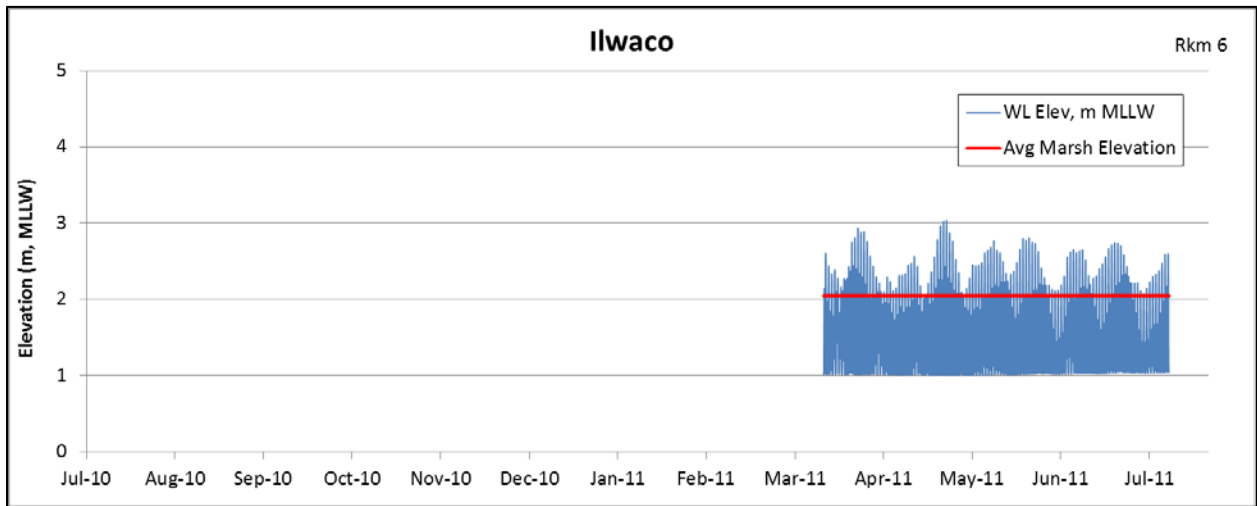
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7 Appendices

Appendix A. Site Hydrographs

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



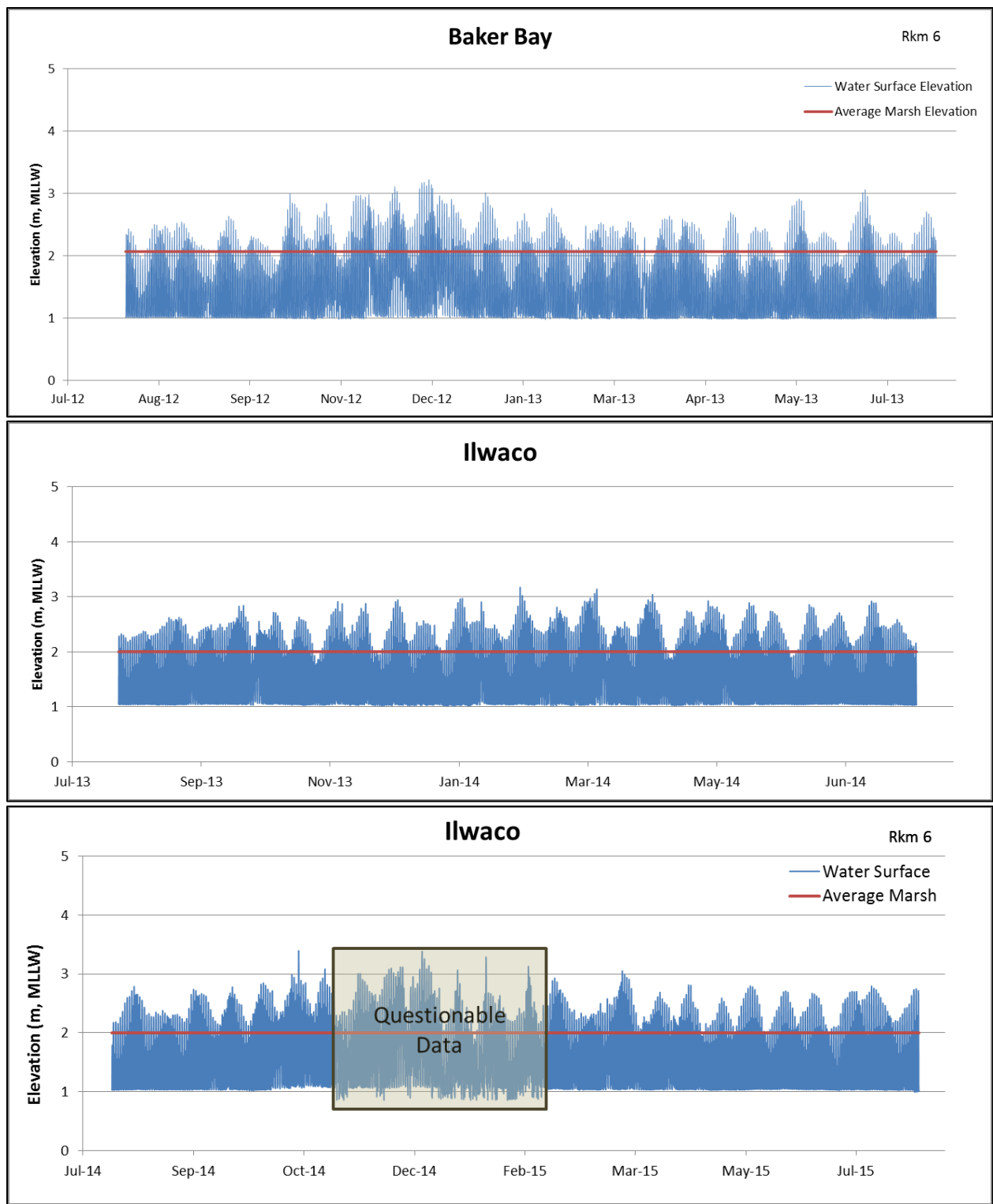
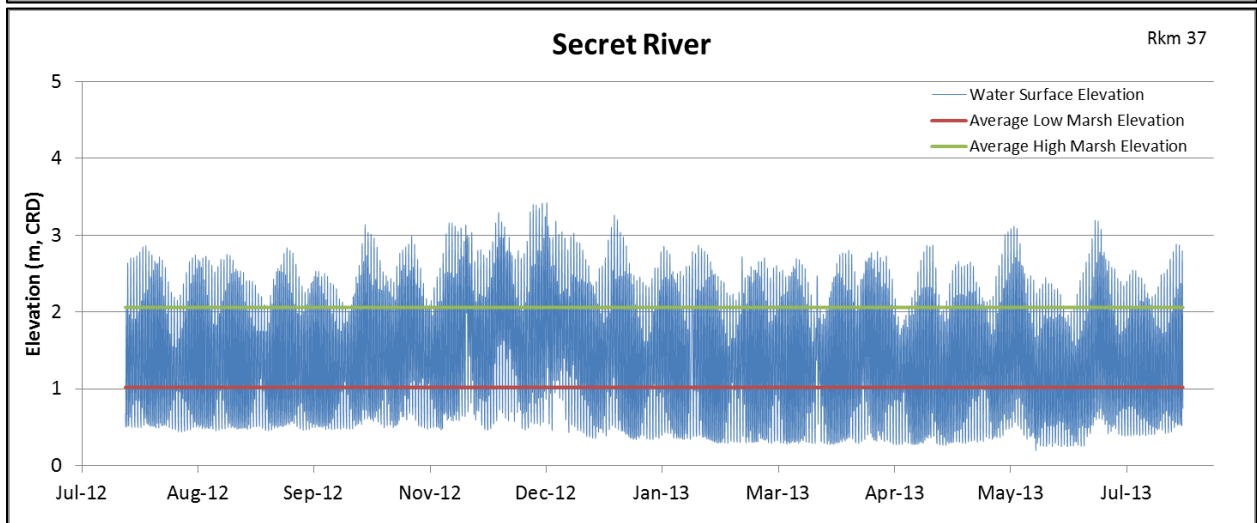
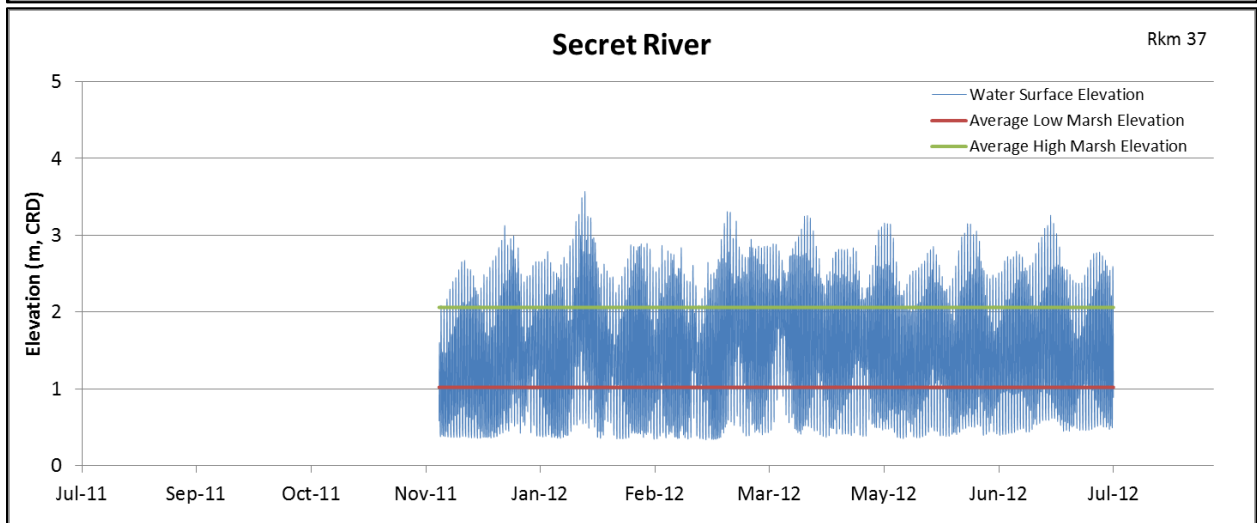
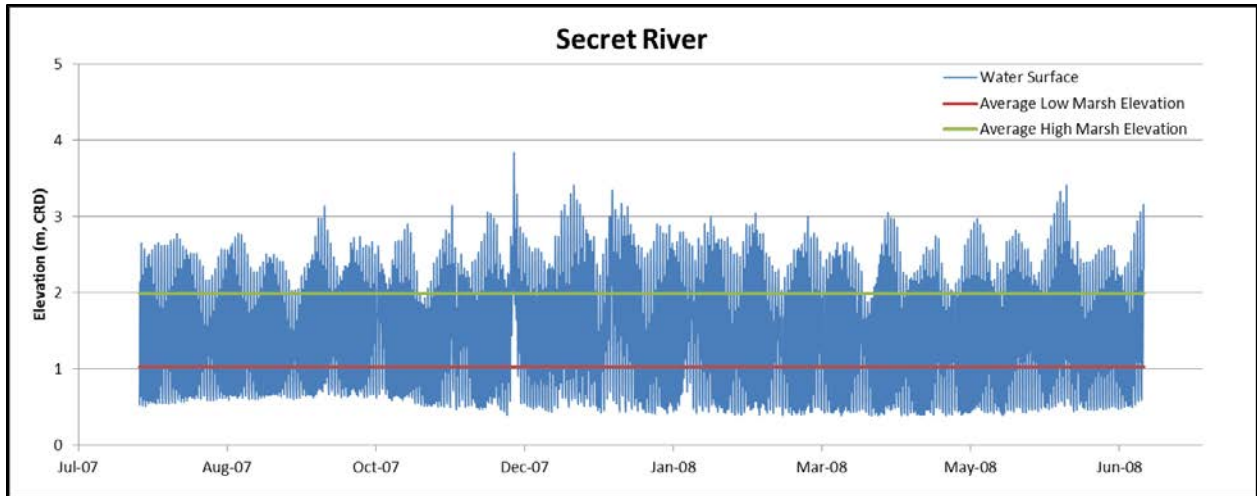


Figure A - 1. Water surface elevation data from the Ilwaco study site for the years 2011-2015. The red line represents the average elevation of the marsh sampling area. In Nov 2014-Fb 2015, the sensor appears to have become disconnected from the deployment post; measurements from this time should not be used in calculations.



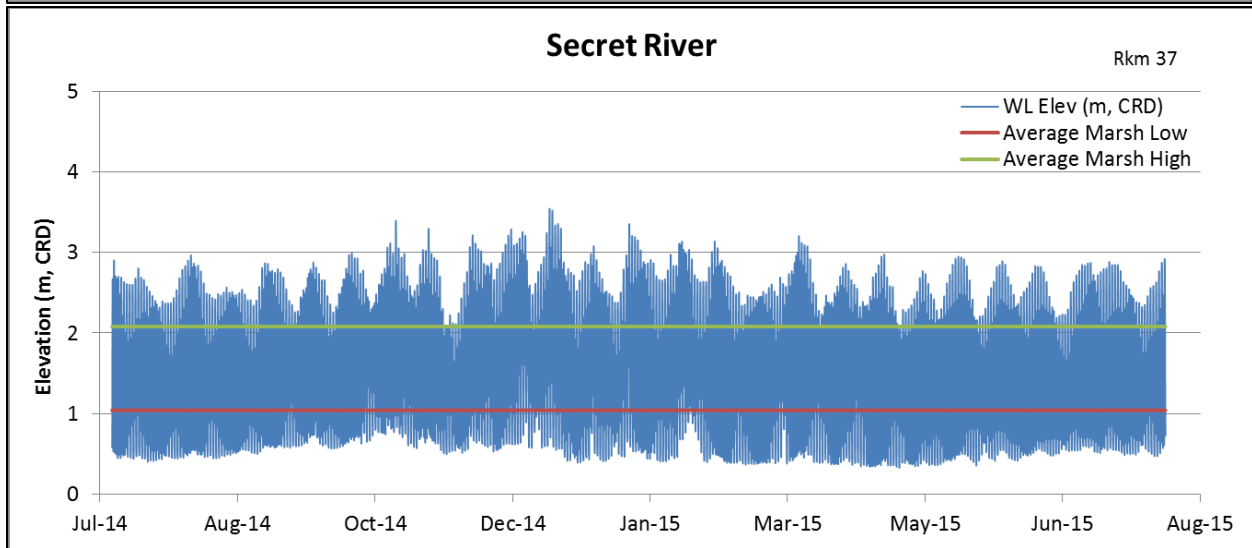
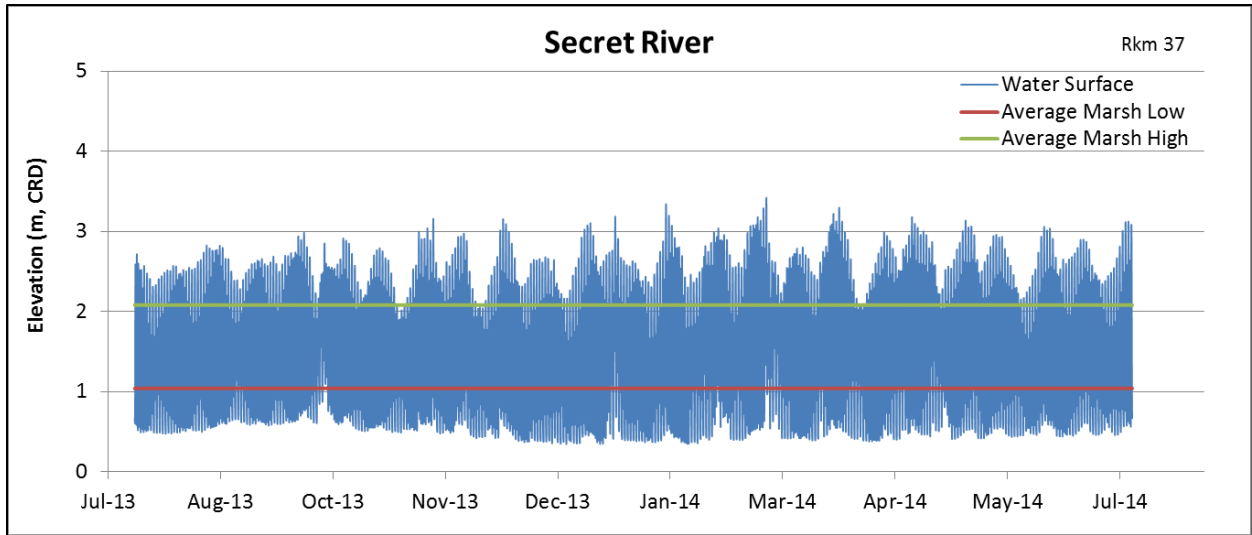
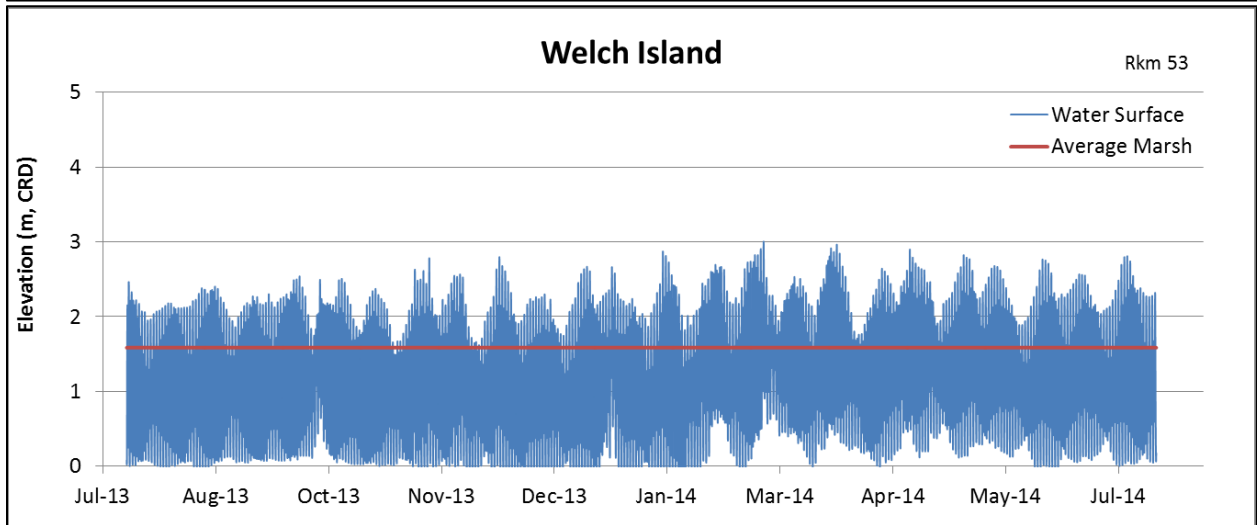
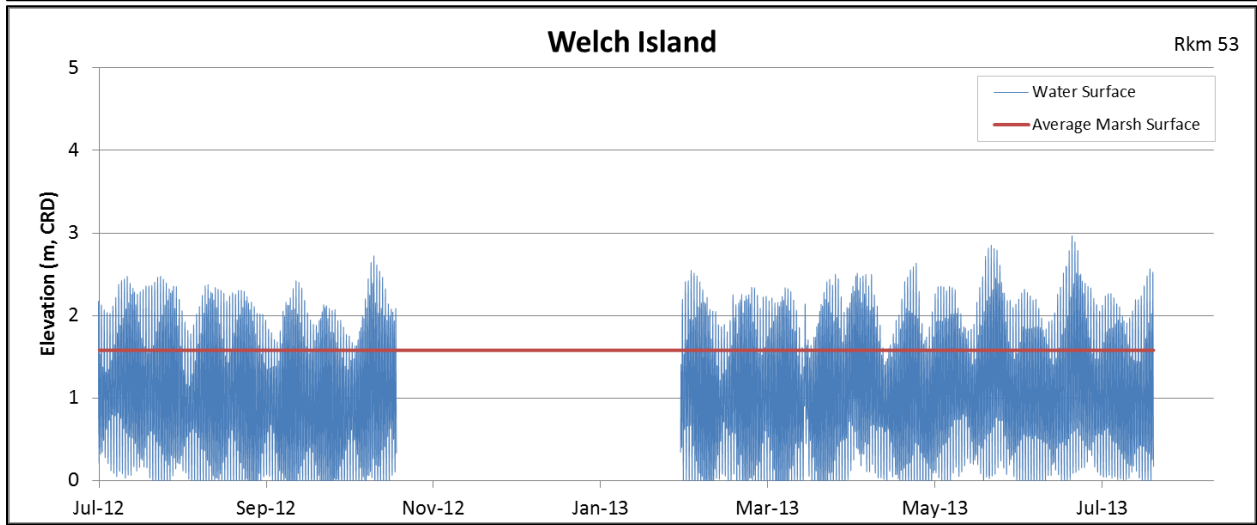
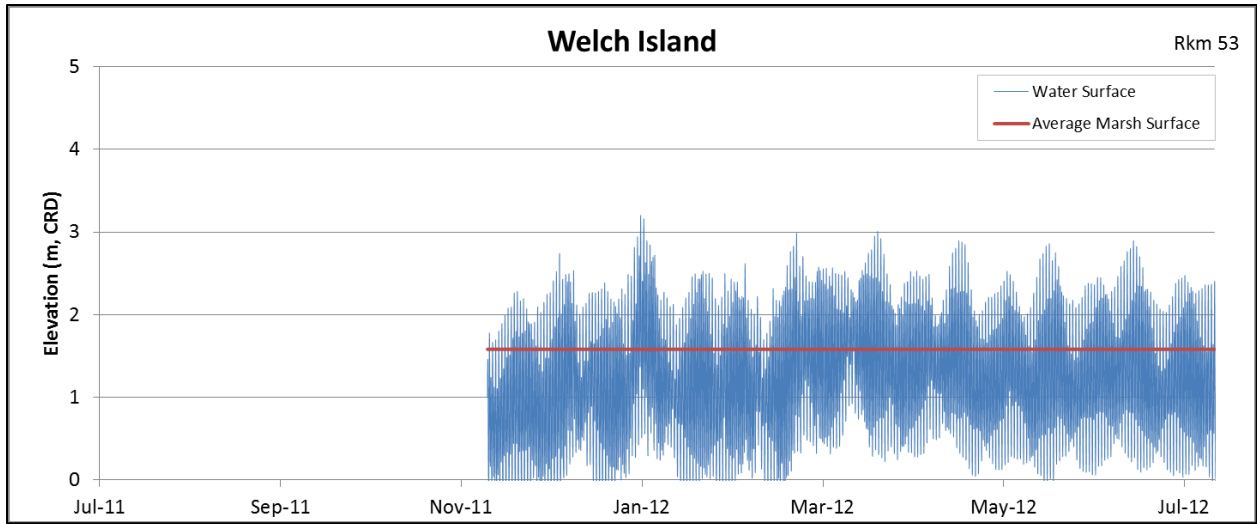


Figure A -2. Water surface elevation data from the Secret River study site for the years 2007-2008 and 2011-2015. The red line represents the average elevation of the marsh sampling area.



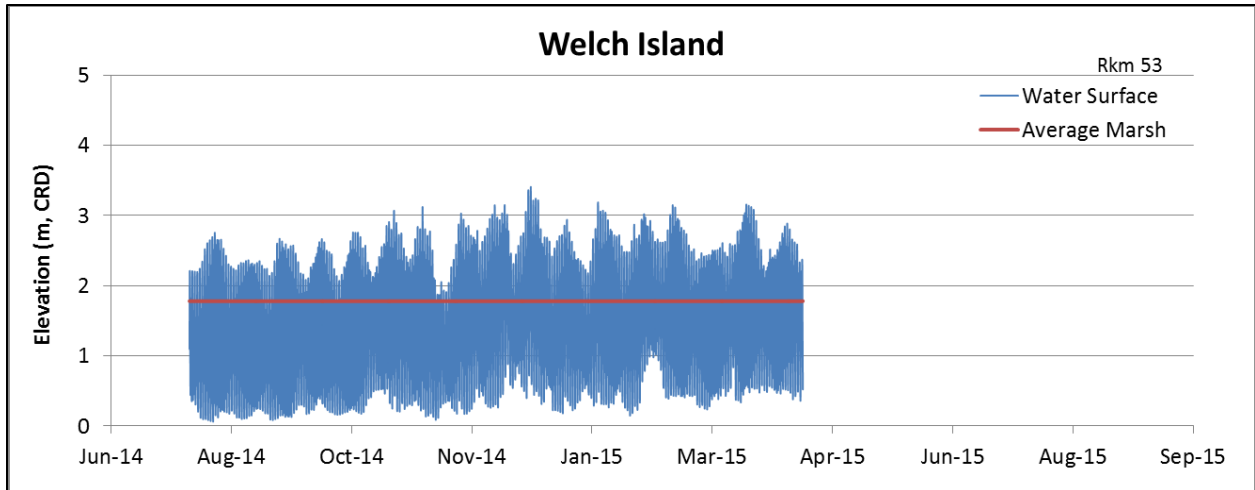
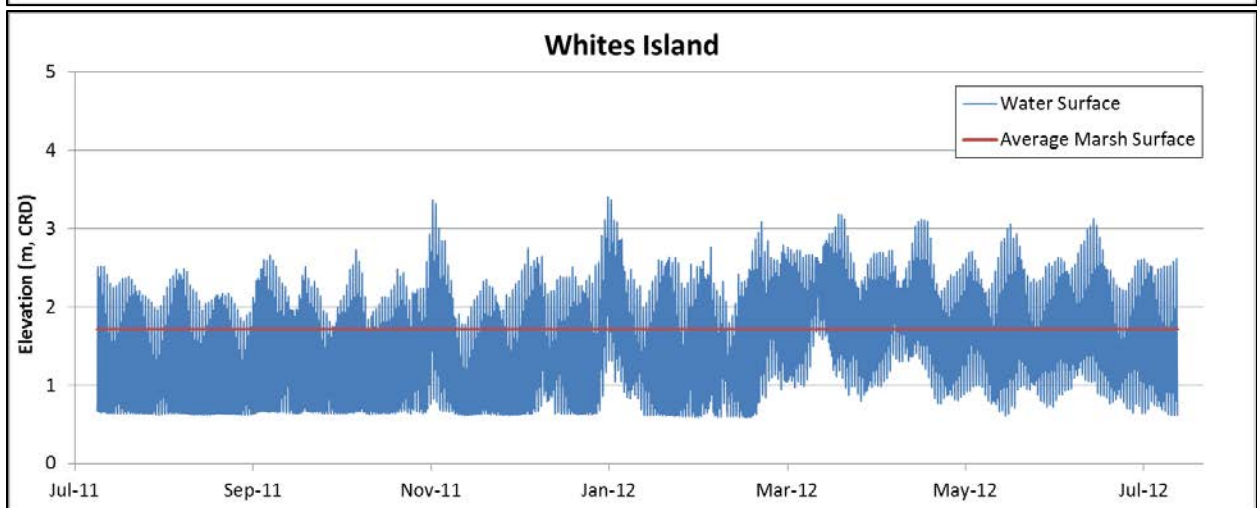
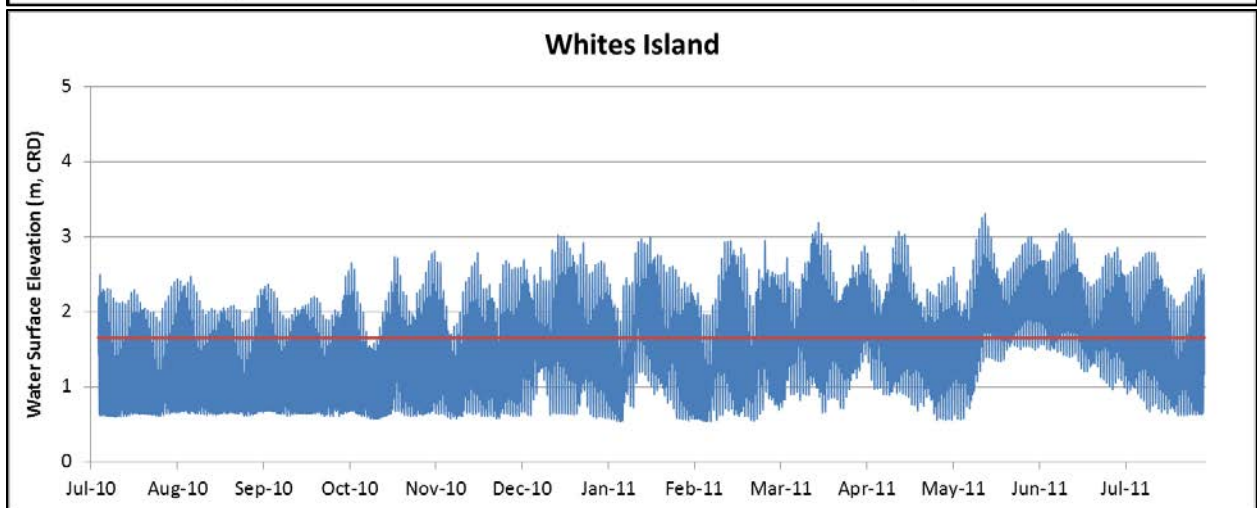
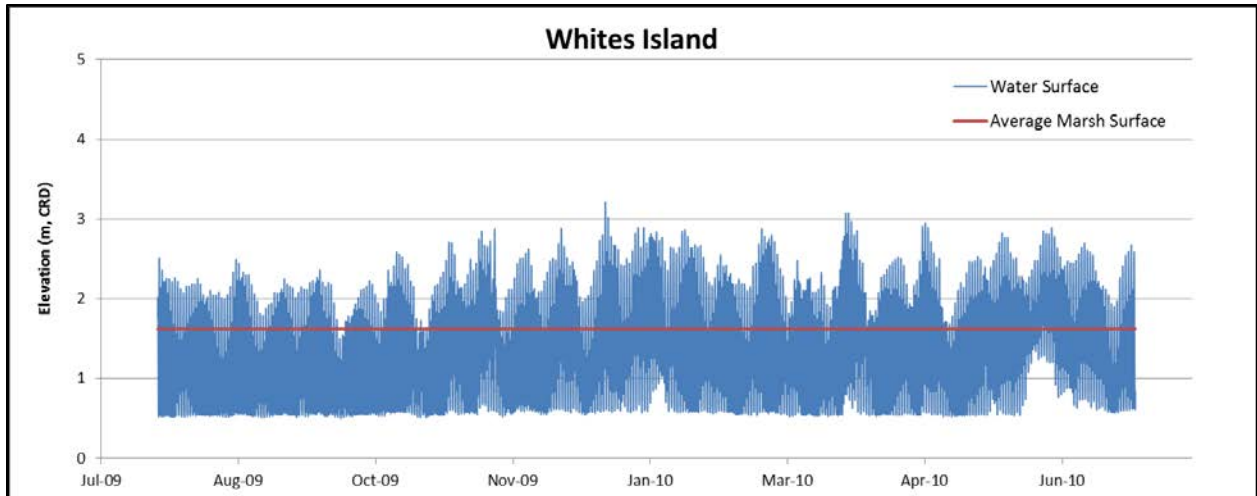


Figure A -3. Water surface elevation data from the Welch Island study site for the years 2012-2015. The red line represents the average elevation of the marsh sampling area. The sensor was displaced between early November 2012 and February 2013.



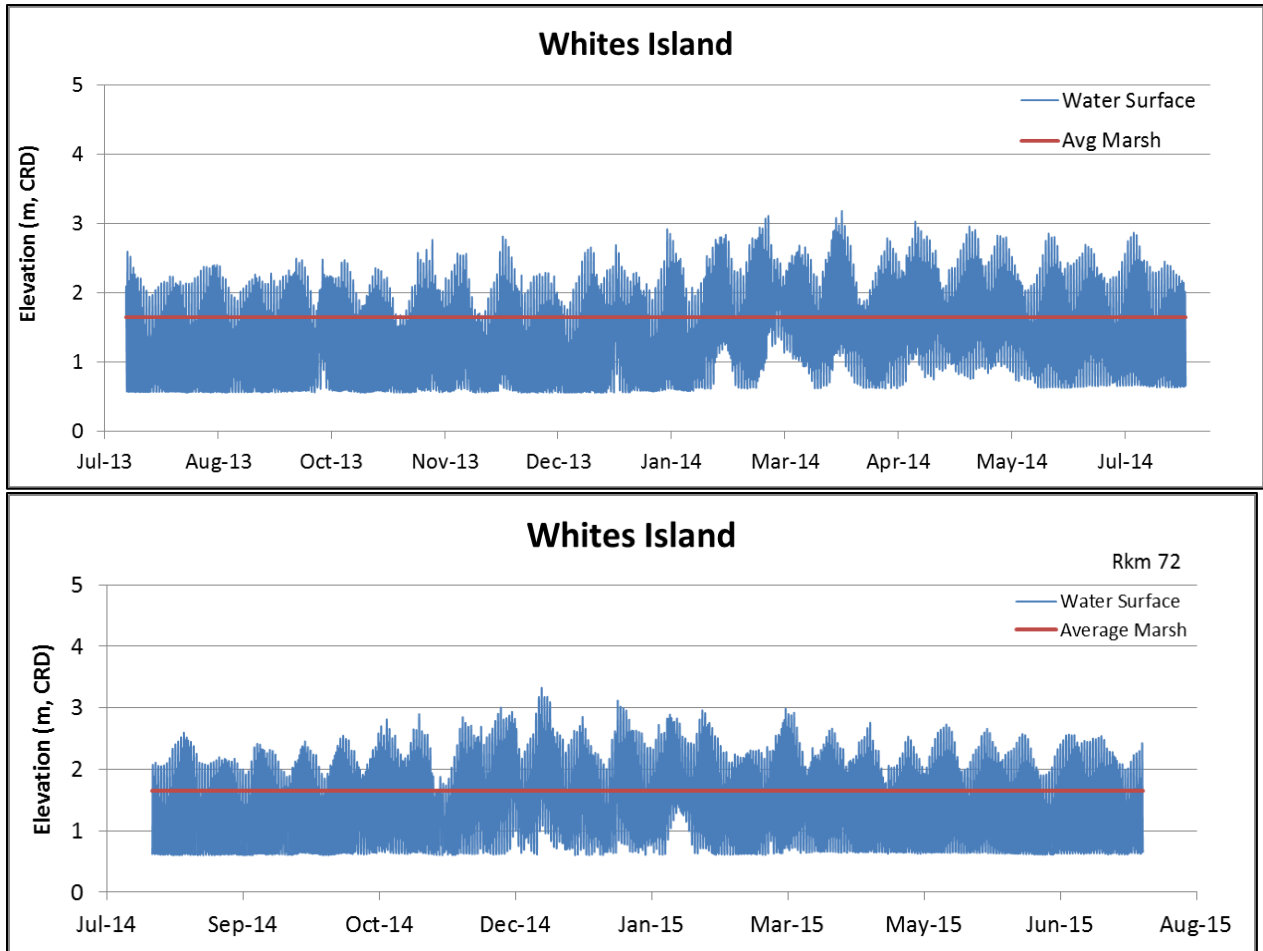
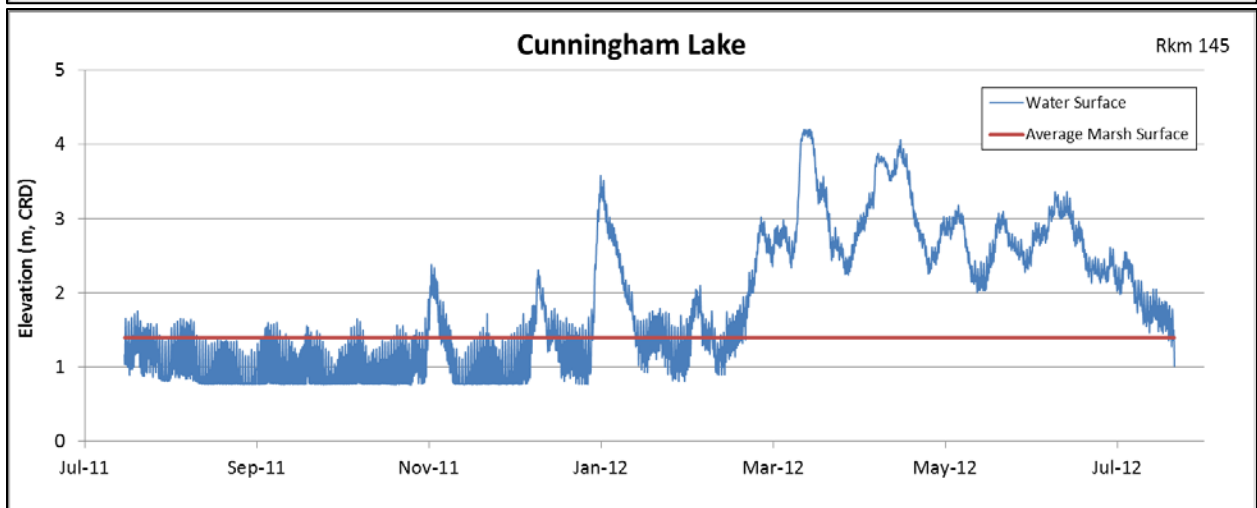
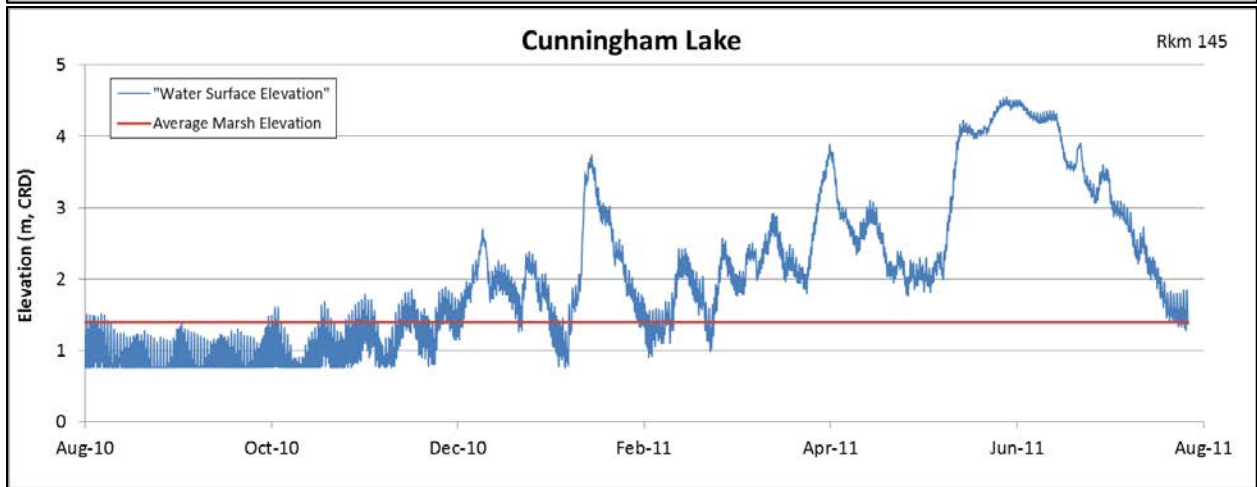
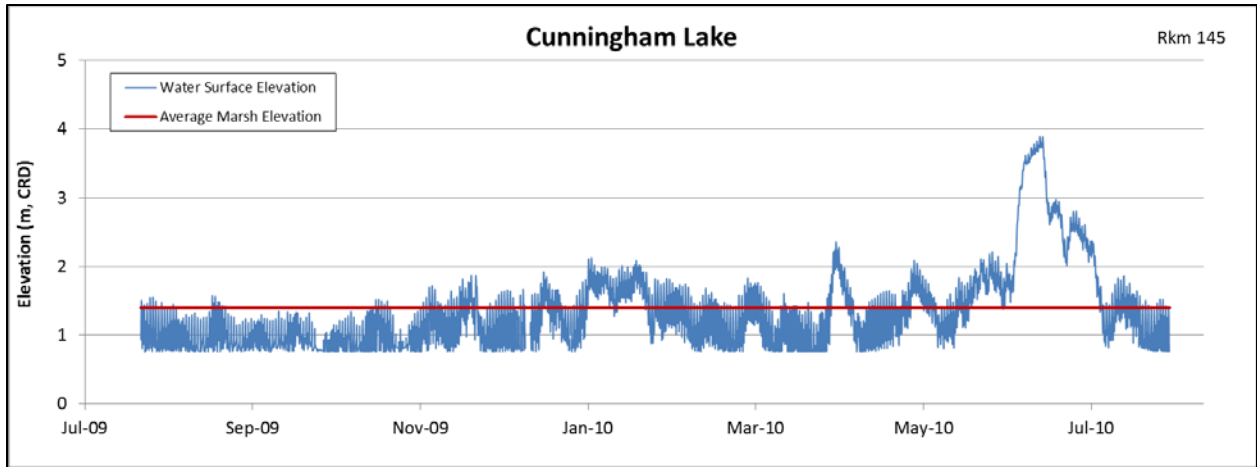


Figure A -4. Water surface elevation data from the Whites Island study site for the years 2009-2012 and 2013-2015. The red line represents the average elevation of the marsh sampling area. No data from 2013 due to sensor failure.



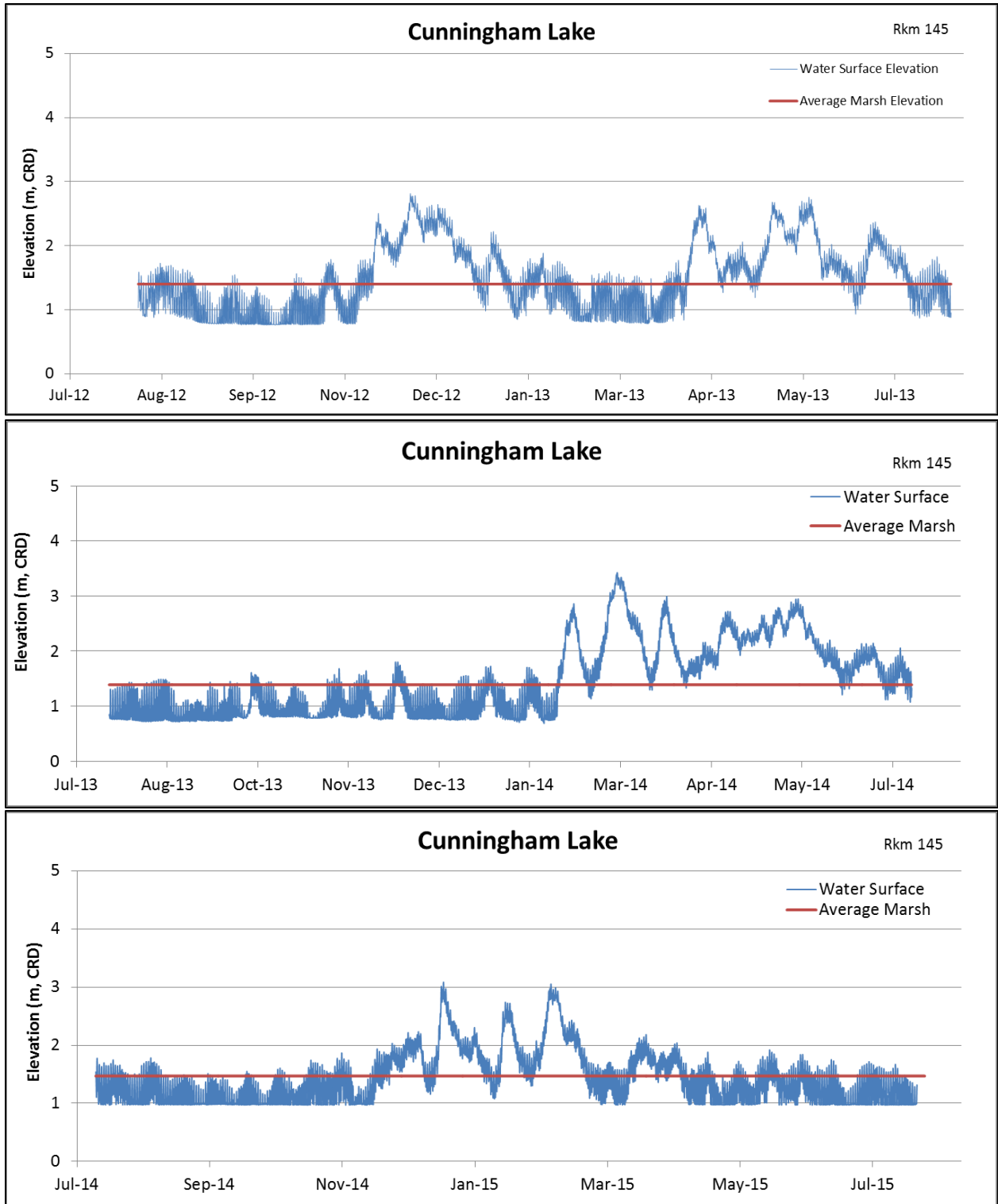
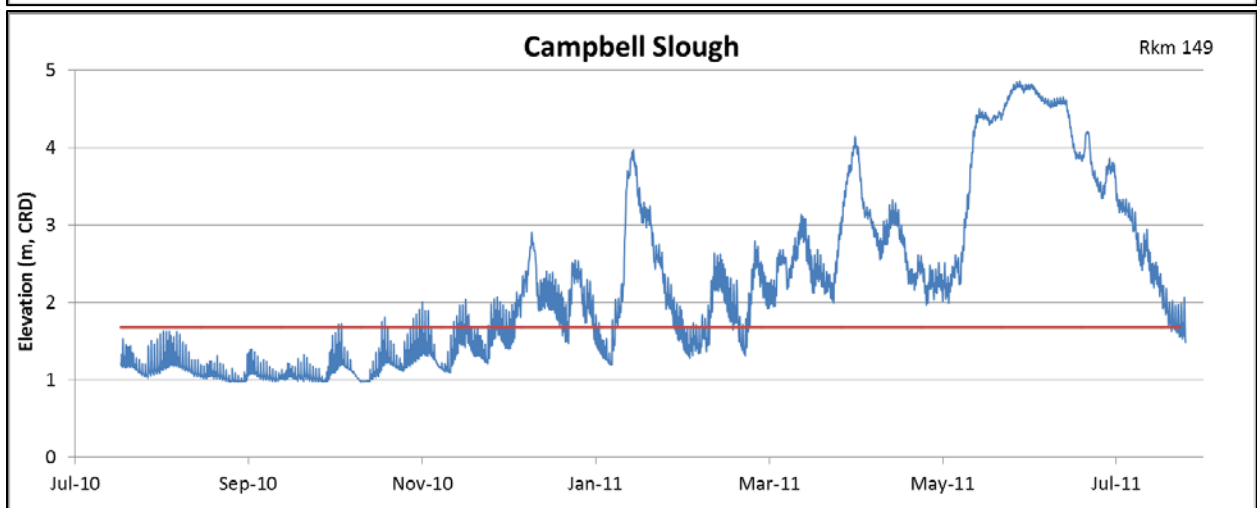
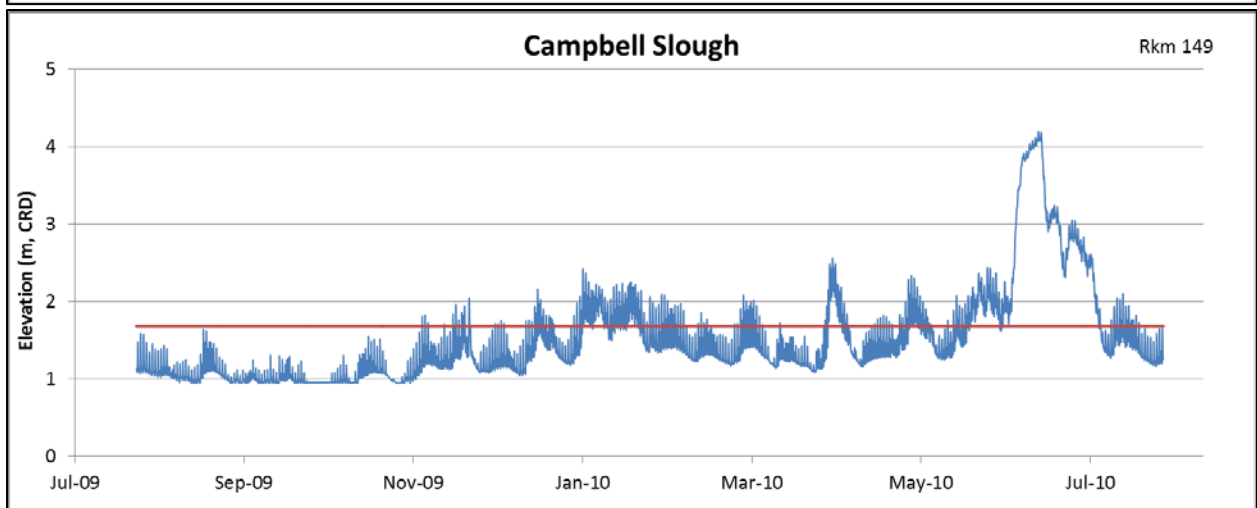
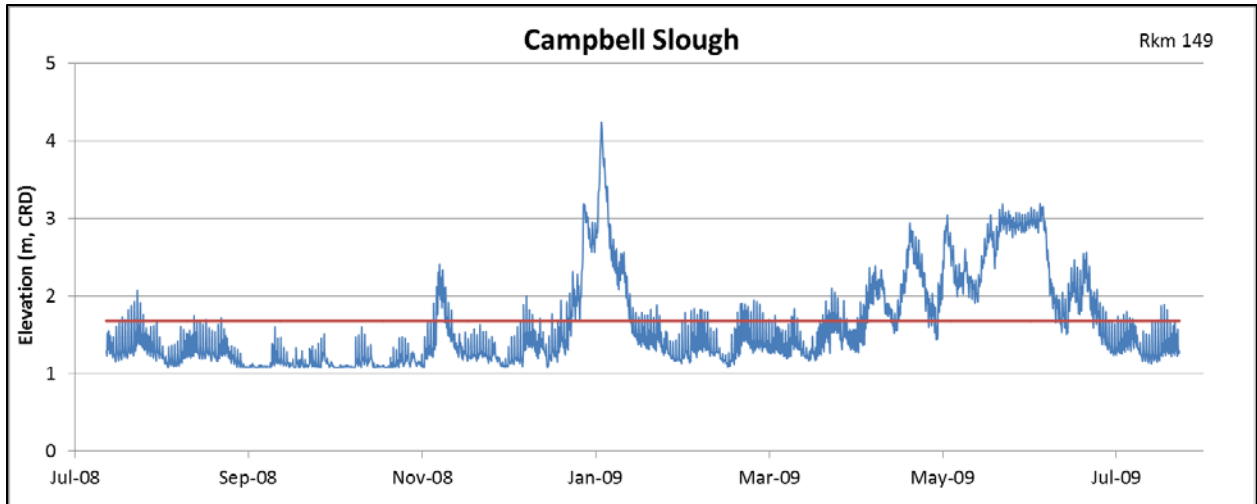
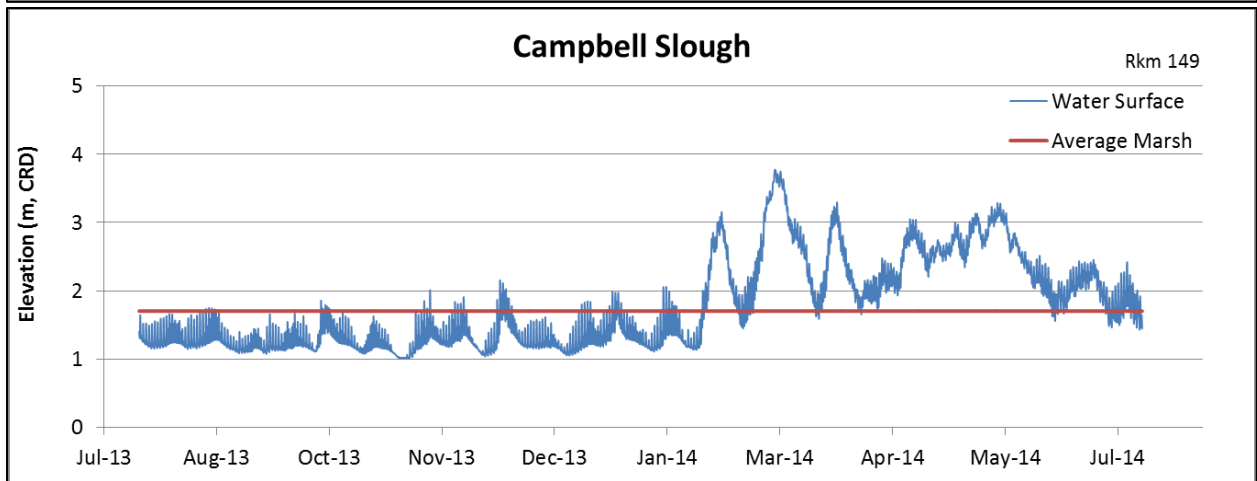
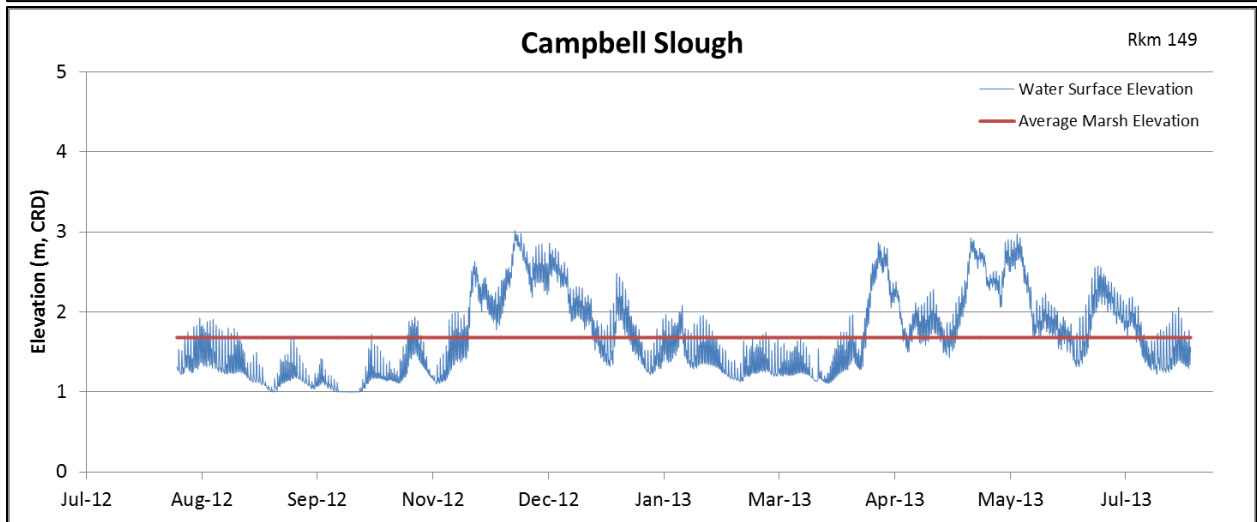
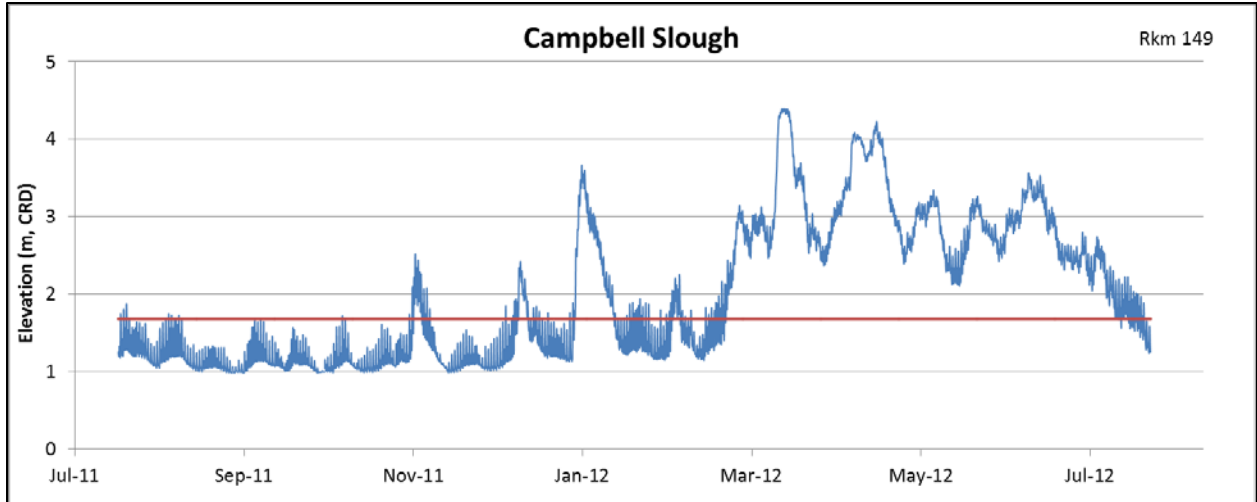


Figure A -5. Water surface elevation data from the Cunningham Lake study site for the years 2009-2015. The red line represents the average elevation of the marsh sampling area.





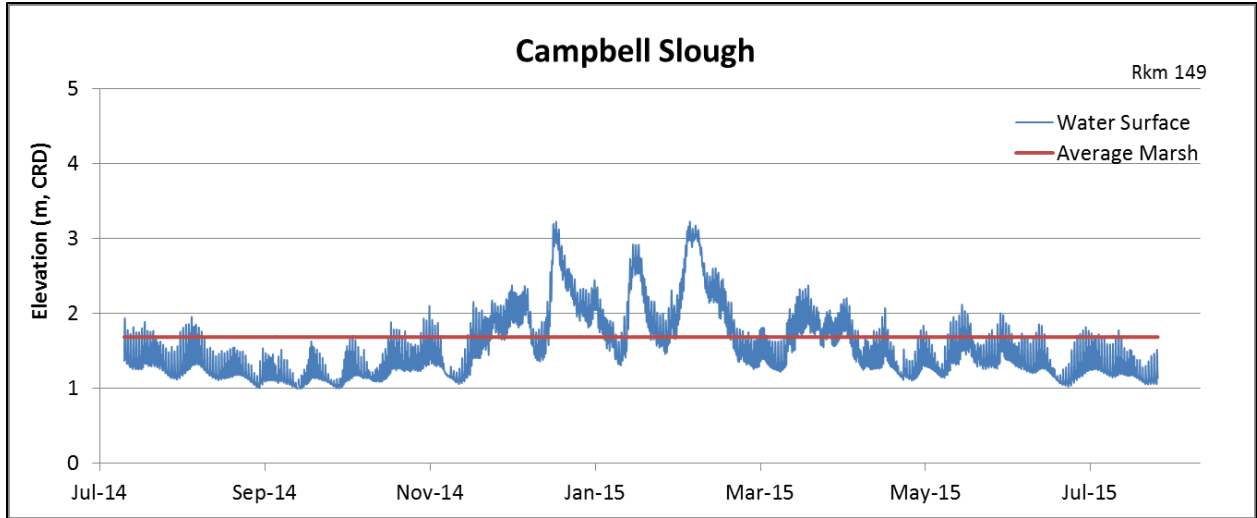
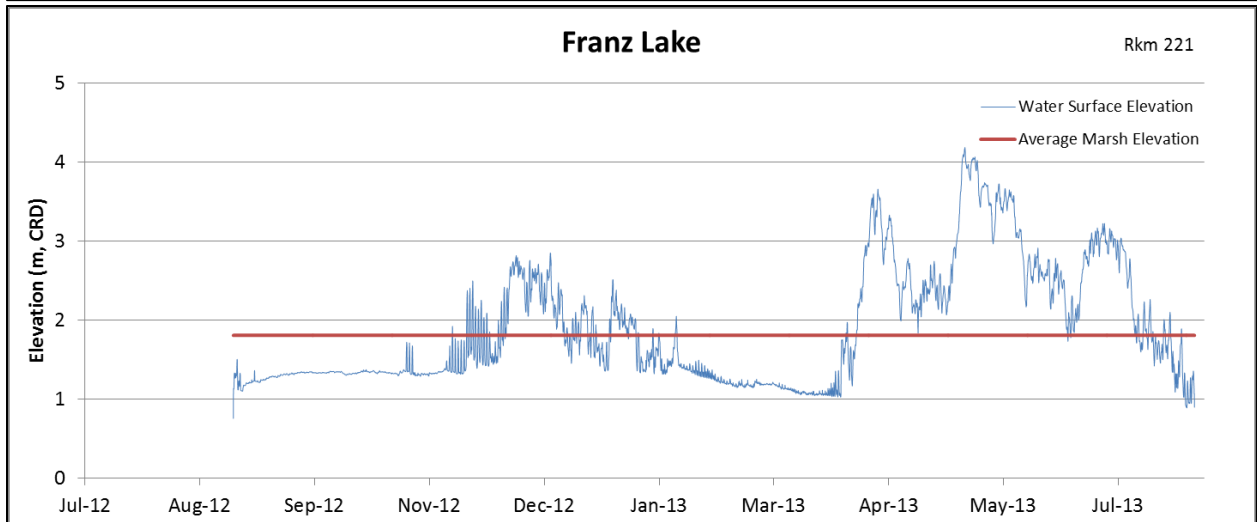
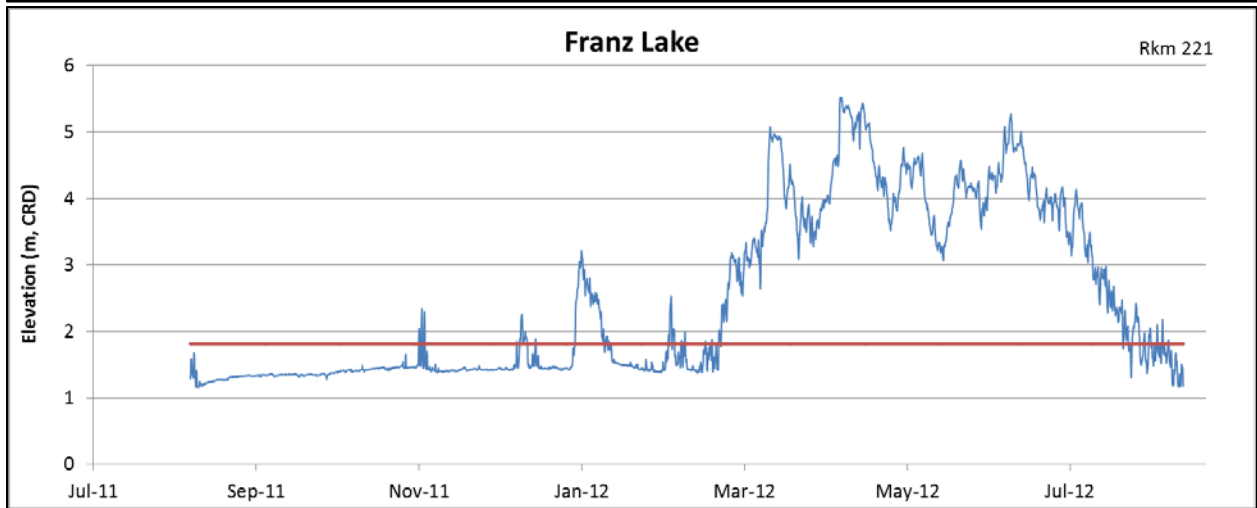
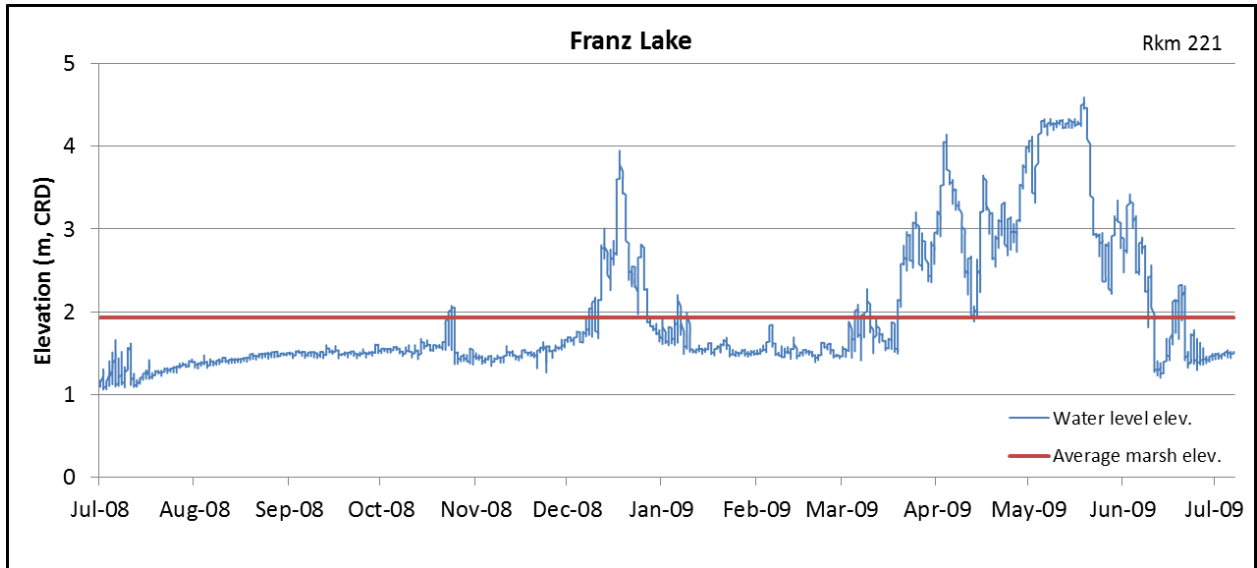


Figure A -6. Water surface elevation data from the Campbell Slough study site for the years 2008-2015. The red line represents the average elevation of the marsh sampling area.



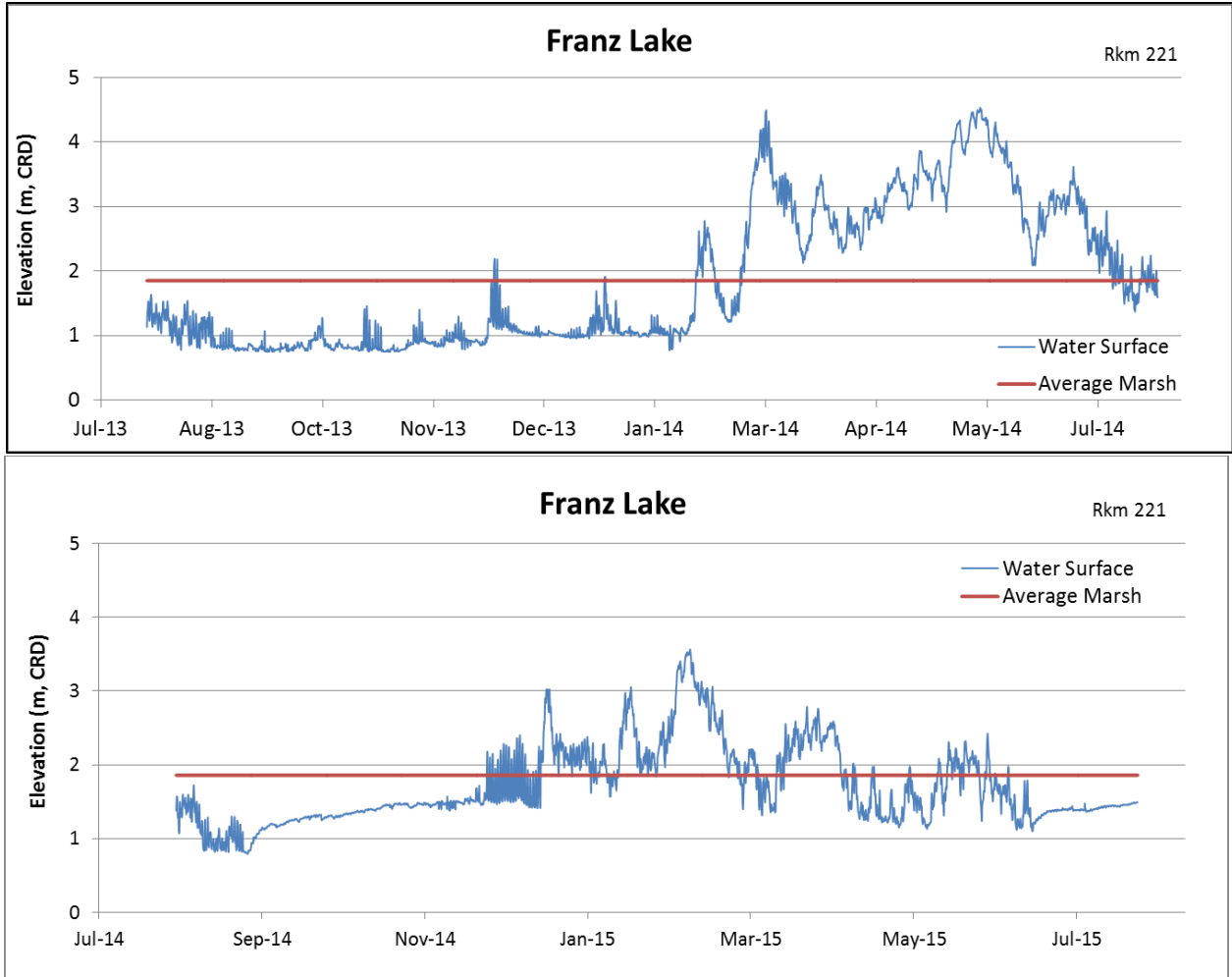


Figure A -7. Water surface elevation data from the Franz Lake study site for the years 2008-2009 and 2011-2015. The red line represents the average elevation of the marsh sampling area. Note the scale difference for the 2011-2012 plot.

Appendix B. Site Maps

NOTE: Sites that have been previously mapped (trends 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 trends sites that had no observable change (Ilwaco Slough, 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

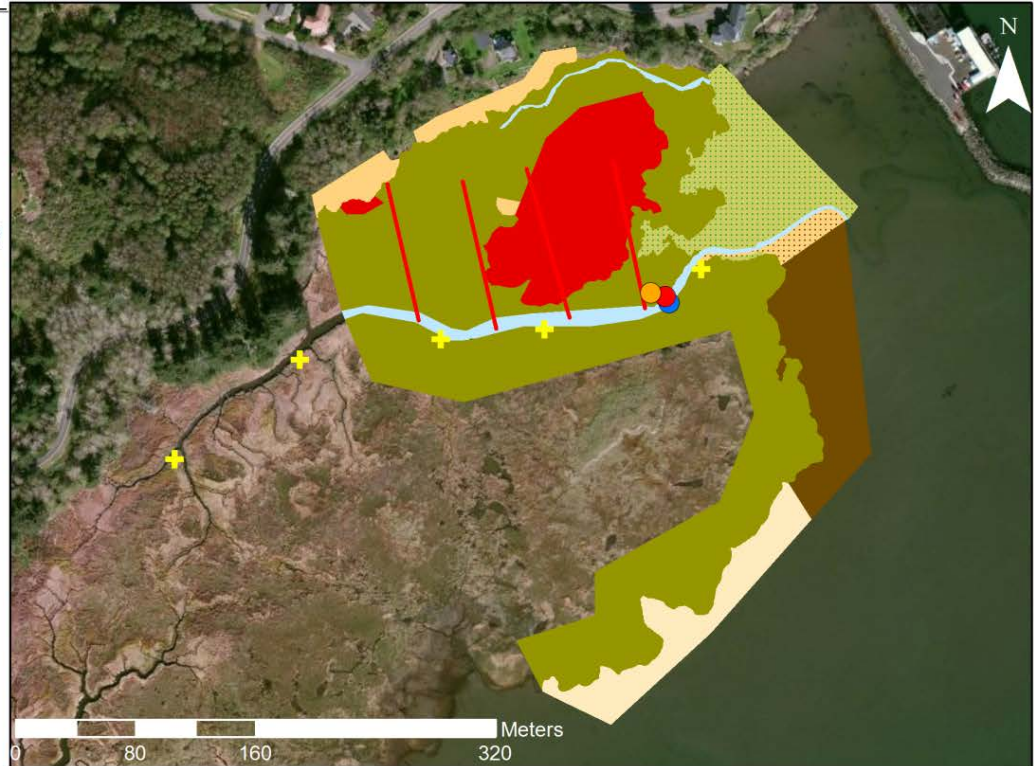
GPS Mapping

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









Secret River Marsh, 2013

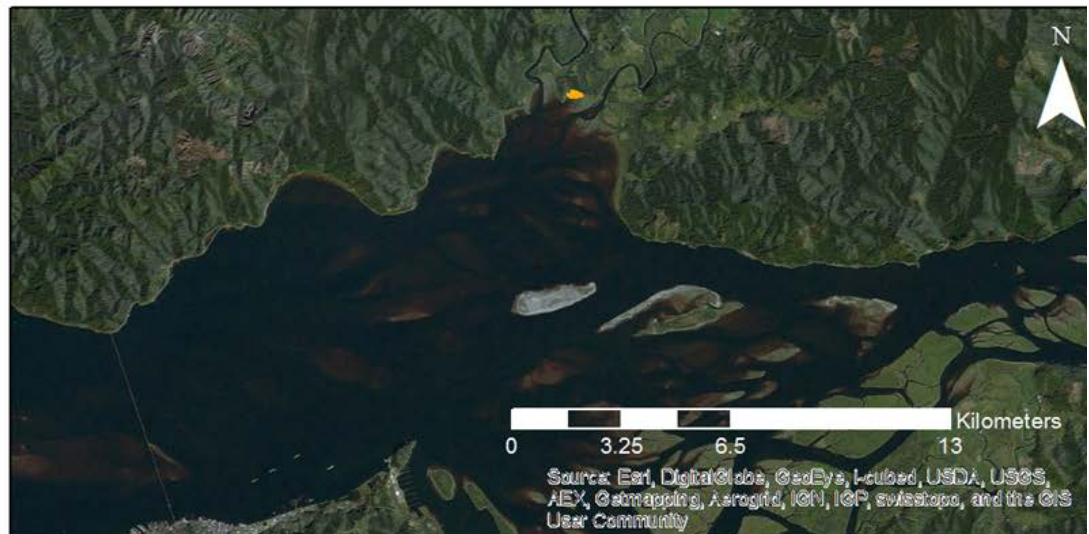
GPS Mapping

-  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

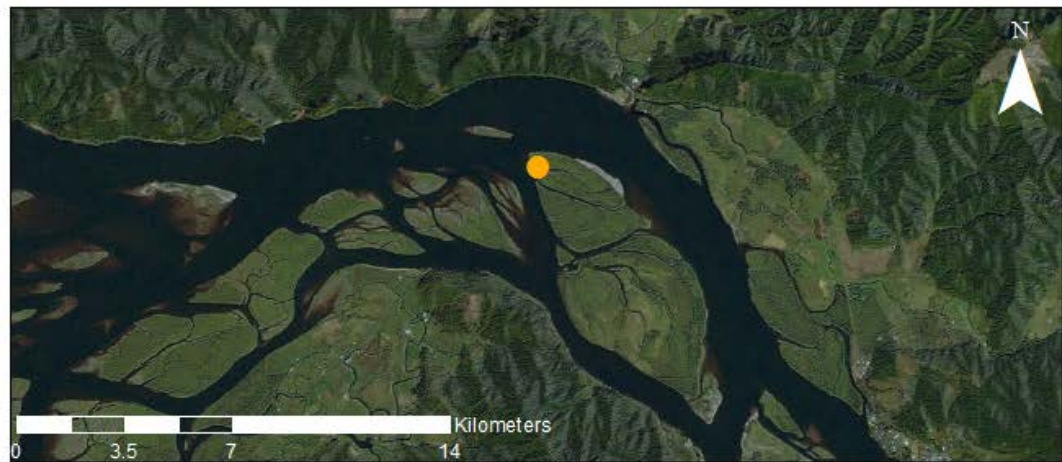
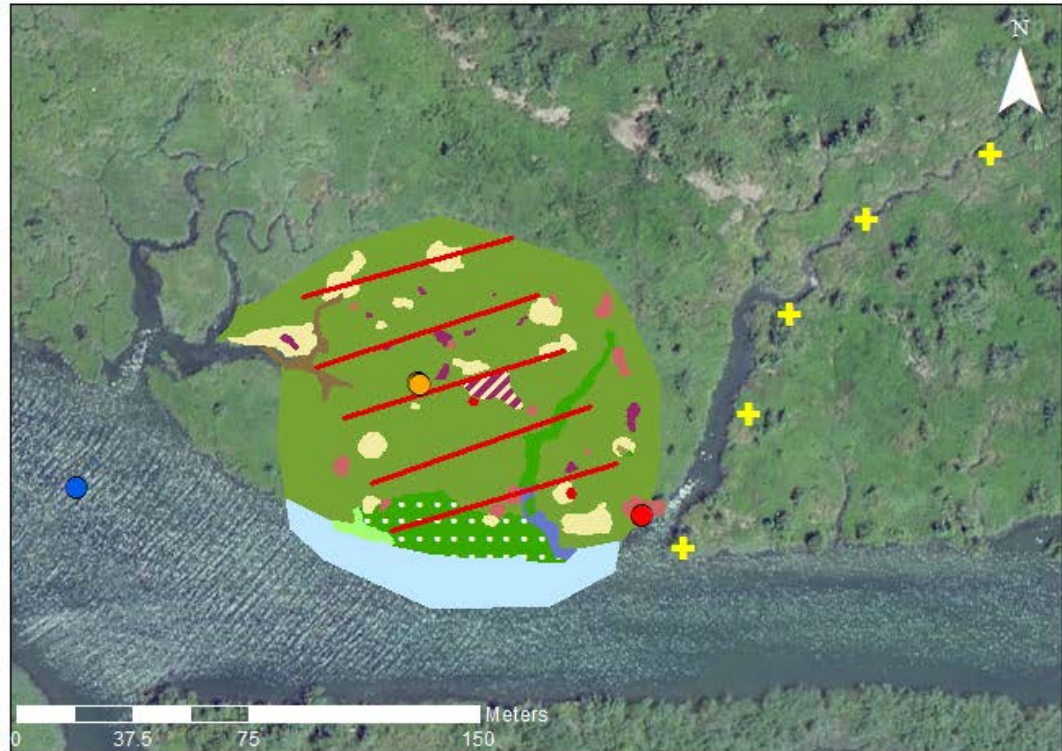
GPS Mapping

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



White's Island, 2011

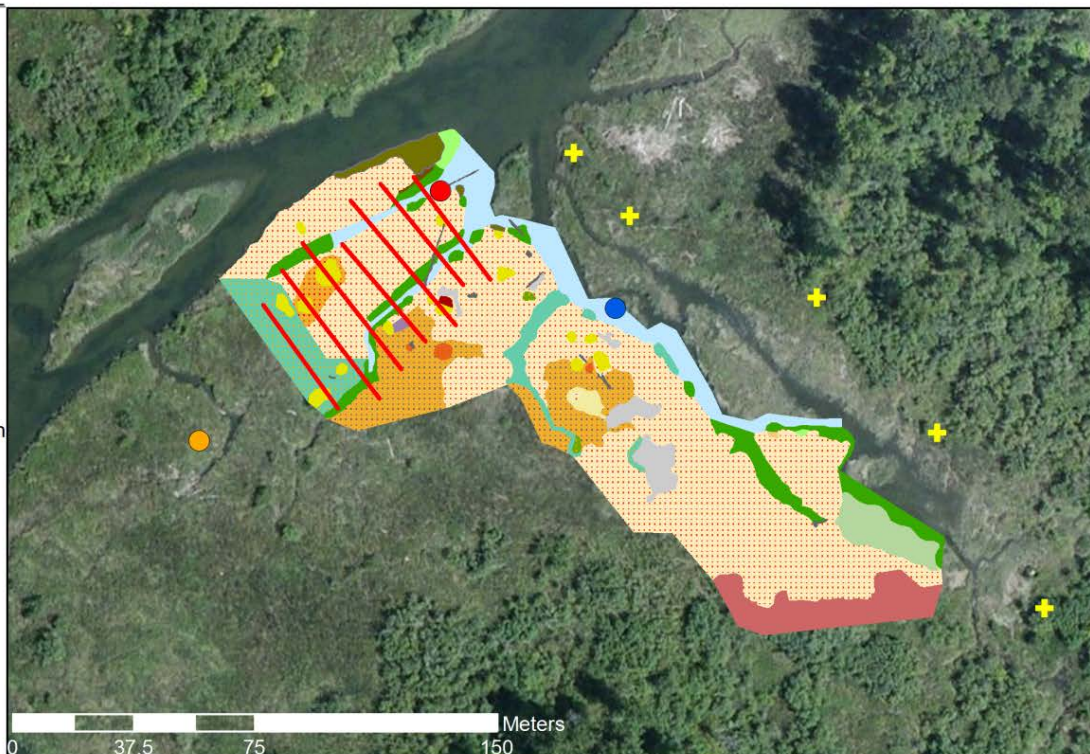
GPS Mapping

Vegetation Communities

- Alnus rubra*
- Alisma triviale*, *Equisetum fluviatile*
- bare ground
- Carex* sp.
- drift wrack
- Eleocharis palustris*
- Equisetum 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








Cunningham Lake, 2012

GPS Mapping

Vegetation Communities

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

Monitoring Locations

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








Campbell Slough, 2011

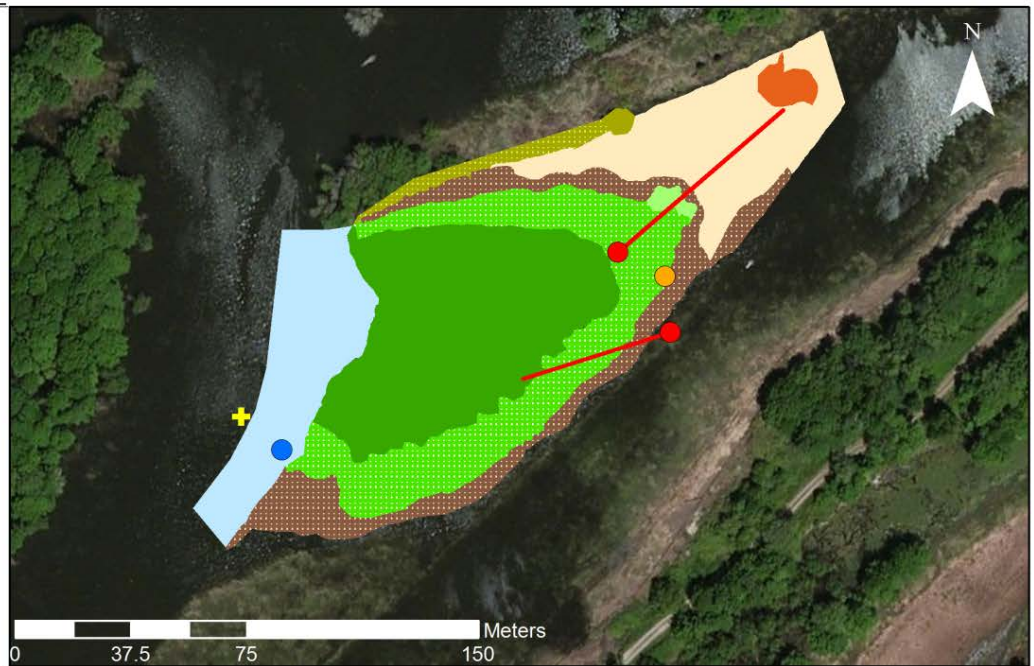
GPS Mapping

Vegetation Communities

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

Monitoring Locations

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








Franz Lake, 2012

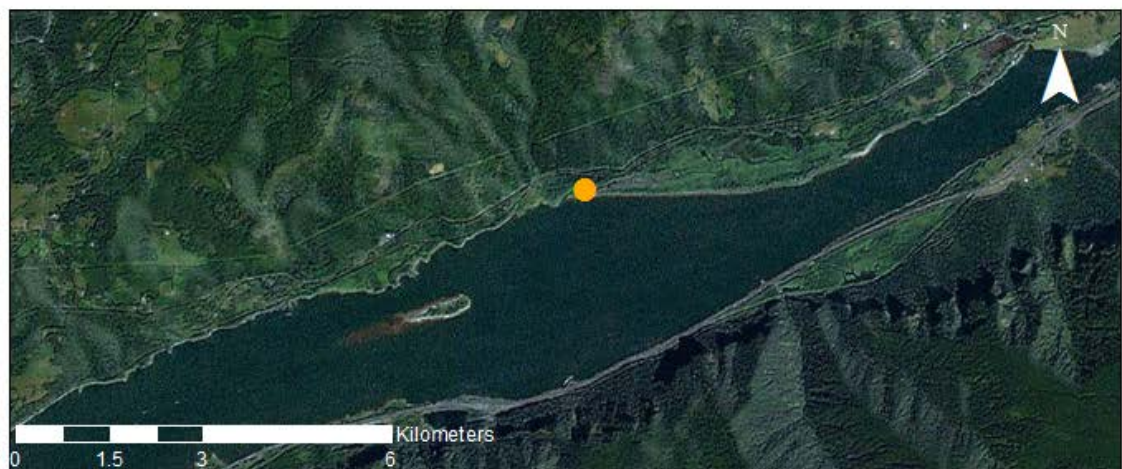
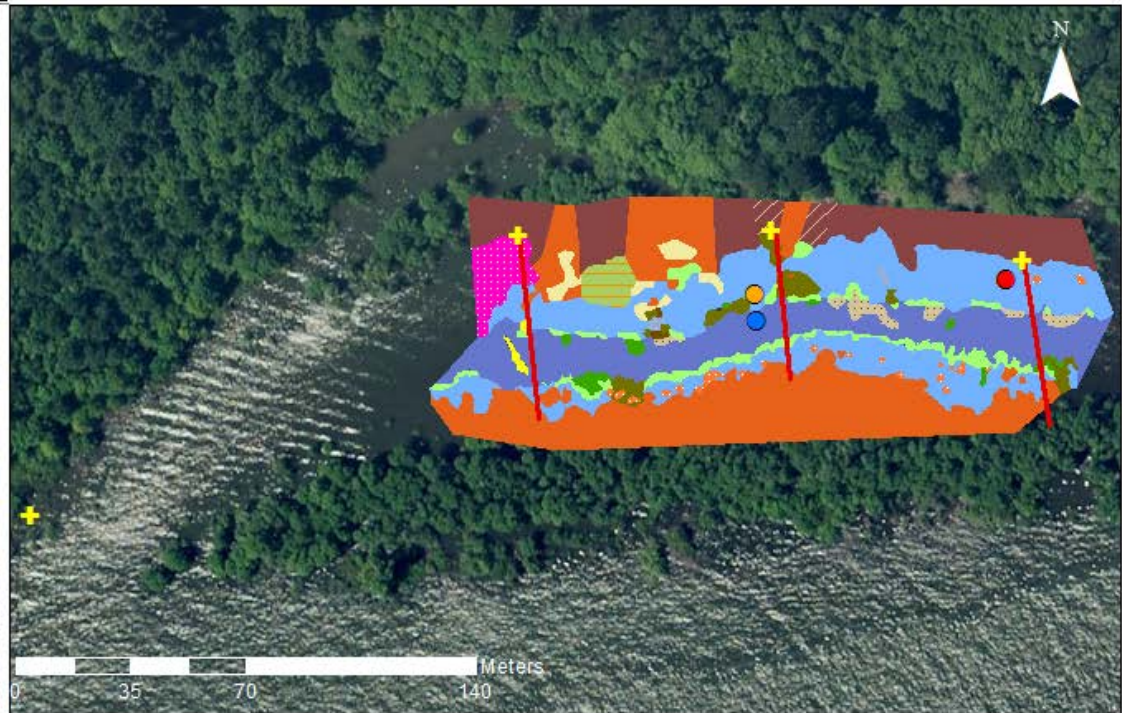
GPS Mapping

Vegetation communities

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

Monitoring Locations

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



Appendix C. Vegetation Species Cover

Table C.1. Site marsh elevation (in meters, relative to the Columbia River vertical datum CRD) and marsh vegetation species average percent cover from 2015. 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-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz	
													Elevation (m, CRD)
					Min	1.61	0.96	1.93	1.07	0.77	1.11	1.21	1.21
					Avg	2.00	1.04	2.08	1.78	1.65	1.47	1.68	1.86
					Max	2.38	1.22	2.20	1.93	2.10	1.73	2.69	2.29
Average Percent Cover													
AGGI	<i>Agrostis gigantea</i>	redtop; black bentgrass	FAC	no	0.0	0.6	0.0	1.1	0.3	0.0	0.0	0.2	
AGSP	<i>Agrostis sp.</i>	bentgrass	mixed	mixed	9.3	0.0	0.1	0.0	0.0	0.0	0.1	0.0	
ALTR	<i>Alisma triviale</i>	northern water plantain	OBL	yes	0.0	0.1	1.3	0.3	0.3	0.0	0.0	0.0	
AMFR	<i>Amorpha fruticosa</i>	indigo bush	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	
AREG	<i>Argentina egedii ssp. Egedii</i>	Pacific silverweed	OBL	yes	2.1	1.9	0.0	3.7	0.0	0.0	0.0	0.3	
BICE	<i>Bidens cernua</i>	Nodding beggars-ticks	OBL	yes	0.0	0.5	3.6	0.2	1.2	0.2	0.0	0.0	
BIFR	<i>Bidens frondosa</i>	devil's beggartick	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	
CAAM	<i>Castilleja ambigua</i>	paint-brush owl-clover; johnny-nip	FACW	yes	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
CAAP	<i>Carex aperta</i>	Columbia sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.2	
CAHE	<i>Callitriche heterophylla</i>	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.1	0.9	0.1	0.5	0.0	1.6	0.0	
CAHE2	<i>Callitriche hermaphroditica</i>	northern water-starwort	OBL	yes	0.0	0.0	0.7	0.1	0.0	0.0	0.0	0.0	
CALE	<i>Carex lenticularis</i>	lakeshore sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	
CALY	<i>Carex lyngbyei</i>	Lyngby sedge	OBL	yes	41.6	51.0	8.6	51.7	3.5	0.0	0.0	0.0	
CAOB	<i>Carex obnupta</i>	Slough sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
CAPA	<i>Caltha palustris</i>	Yellow marsh marigold	OBL	yes	0.0	5.6	0.0	3.3	0.8	0.0	0.0	0.0
CARE	<i>Carex retrorsa</i>	knotsheath sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
CASE	<i>Calystegia sepium</i>	Hedge false bindweed	FAC	no	0.0	0.3	0.0	0.0	2.5	0.0	0.0	0.0
CEDE	<i>Ceratophyllum demersum</i>	Coontail	OBL	yes	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0
CYOD	<i>Cyperus odoratus</i>	rusty flatsedge	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1
DECE	<i>Deschampsia cespitosa</i>	Tufted hairgrass	FACW	yes	4.2	0.0	0.0	0.5	0.0	0.0	0.0	0.0
ELAC	<i>Eleocharis acicularis</i>	Needle spikerush	OBL	yes	0.4	0.0	1.8	0.5	0.5	0.5	0.2	0.0
ELCA	<i>Elodea canadensis</i>	Canada waterweed	OBL	yes	0.0	0.0	13.5	3.0	6.1	0.0	0.2	0.0
ELNU	<i>Elodea nuttallii</i>	Nuttall's waterweed, western waterweed	OBL	yes	0.0	0.0	13.5	0.0	0.0	0.0	0.0	0.0
ELOV	<i>Eleocharis ovata</i>	Ovoid spikerush	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0
ELPA	<i>Eleocharis palustris</i>	Common spikerush	OBL	yes	0.0	0.8	5.0	4.1	1.4	15.3	30.1	5.0
ELPAR	<i>Eleocharis parvula</i>	Dwarf spikerush	OBL	yes	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
EPCI	<i>Epilobium ciliatum</i>	Willow herb	FACW	yes	0.0	0.8	0.0	0.4	0.5	0.0	0.0	0.0
EQFL	<i>Equisetum fluviatile</i>	Water horsetail	OBL	yes	0.0	1.7	0.0	1.4	3.4	0.0	0.0	0.0
EQPA	<i>Equisetum palustre</i>	marsh horsetail	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.6
FRLA*	<i>Fraxinus latifolia</i>	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	1.3	2.9
GATR	<i>Galium trifidum</i> L. spp. <i>columbianum</i>	Pacific bedstraw	FACW	yes	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0
GATR2	<i>Galium triflorum</i>	fragrant bedstraw	FACU	yes	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0
GATR3	<i>Galium trifidum</i>	small bedstraw	FACW	yes	0.0	0.0	0.0	0.9	0.2	0.0	0.0	0.0
GLGR	<i>Glyceria grandis</i>	American mannagrass	OBL	yes	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
GLMA	<i>Glaux maritima</i>	sea milkwort	OBL	yes	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GNUL	<i>Gnaphalium uliginosum</i>	Marsh cudweed	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.7
GREB	<i>Gratiola ebracteata</i>	bractless hedgehyssop	OBL	yes	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
GRNE	<i>Gratiola neglecta</i>	American Hedge-hyssop	OBL	yes	0.0	0.0	0.0	T	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
HEAU	<i>Helenium autumnale</i>	common sneezeweed	FACW	yes	0.0	0.5	0.0	0.0	0.0	0.0	0.4	1.2
HYSC	<i>Hypericum scouleri</i>	Western St. Johns wort	FACW	yes	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0
IMSP	<i>Impatiens capensis, Impatiens noli-tangere</i>	western touch-me-not, common touch-me-not, jewelweed	FACW	yes	0.0	0.6	0.0	5.1	0.8	0.0	0.0	0.1
IRPS	<i>Iris pseudacorus</i>	Yellow iris	OBL	no	0.0	0.0	0.0	2.0	1.5	0.1	0.0	0.0
ISCE	<i>Isolepis cernua</i>	low bulrush	OBL	yes	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ISSP	<i>Isoetes spp.</i>	quillwort	OBL	yes	0.0	0.0	0.4	0.1	0.0	0.0	0.0	0.0
JUAC	<i>Juncus acuminatus</i>	Tapertip rush	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
JUAR	<i>Juncus arcticus Wild. ssp. littoralis</i>	mountain rush	FACW	yes	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
JUOX	<i>Juncus oxymiris</i>	Pointed rush	FACW	yes	0.0	0.0	0.3	2.4	0.2	0.0	0.0	0.0
JUTE	<i>Juncus tenuis</i>	slender rush, poverty rush	FAC	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
LAPA	<i>Lathyrus palustris</i>	Marsh peavine	OBL	yes	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
LEOR	<i>Leersia oryzoides</i>	Rice cutgrass	OBL	yes	0.0	0.1	0.0	1.2	0.9	4.0	0.5	7.3
LIAQ	<i>Limosella aquatica</i>	Water mudwort	OBL	yes	0.0	0.0	1.6	0.1	0.1	0.0	0.1	0.0
LIDU	<i>Lindernia dubia</i>	yellow seed false pimpernel	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0
LIOC	<i>Lilaeopsis occidentalis</i>	Western lilaeopsis	OBL	yes	3.3	0.0	15.3	0.0	0.0	0.0	0.0	0.0
LISC	<i>Lilaea scilloides</i>	Flowering quillwort	OBL	yes	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.0
LOCO	<i>Lotus corniculatus</i>	Birdsfoot trefoil	FAC	no	0.0	0.1	0.0	0.9	1.8	0.0	0.4	0.0
LUPA	<i>Ludwigia palustris</i>	False loosestrife	OBL	yes	0.0	0.0	0.0	0.0	0.0	3.2	5.7	0.0
LYAM	<i>Lysichiton americanus</i>	Skunk cabbage	OBL	yes	0.0	0.8	0.0	2.5	0.0	0.0	0.0	0.0
LYAM2	<i>Lycopus americanus</i>	American water horehound	OBL	yes	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
LYNU	<i>Lysimachia nummularia L.</i>	Moneywort, Creeping Jenny	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0
LYSA	<i>Lythrum salicaria</i>	Purple loosestrife	OBL	no	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
MEAR	<i>Mentha arvensis</i>	wild mint	FACW	yes	0.0	0.1	0.0	1.3	0.0	0.0	0.0	0.4
MESP	<i>Mentha spp.</i>	Mint (field mint, spearmint)	mixed	mixed	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0
MIGU	<i>Mimulus guttatus</i>	Yellow monkeyflower	OBL	yes	0.0	0.2	0.0	4.3	0.5	0.0	0.0	0.0
MUKE	<i>Murdannia keisak</i>	wart-removing herb	OBL	no	0.0	0.7	0.1	0.8	0.0	0.0	0.0	0.0
MYLA	<i>Myosotis laxa</i>	Small forget-me-not	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
MYSC	<i>Myosotis scorpioides</i>	Common forget-me-not	FACW	no	0.0	6.3	0.0	2.6	2.1	0.0	0.0	0.0
MYSI	<i>Myriophyllum sibiricum</i>	northern milfoil, short spike milfoil	OBL	yes	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0
MYSP	<i>Myosotis laxa, M. scorpioides</i>	Small forget-me-not, Common forget-me-not	mixed	mixed	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0
MYSP2	<i>Myriophyllum spp.</i>	Milfoil	OBL	mixed	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0
OESA	<i>Oenanthe sarmentosa</i>	Water parsley	OBL	yes	0.0	4.0	0.0	2.8	1.5	0.0	0.0	0.0
PADI	<i>Paspalum distichum</i>	Knotgrass	FACW	yes	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
PHAR	<i>Phalaris arundinacea</i>	Reed canary grass	FACW	no	0.0	26.3	0.0	8.3	53.9	52.0	29.5	15.5
PLDI	<i>Platanthera dilatata</i>	white bog orchid	FACW	yes	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0
PLMA	<i>Plantago major</i>	common plantain	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
POAM	<i>Polygonum amphibium</i>	water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.1	40.9
POAN2	<i>Poa annua</i>	annual bluegrass	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
POHY	<i>Polygonum hydropiper, P. hydropiperoides</i>	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	0.8	0.0	0.5	0.7	0.7	0.3	0.0
PONA	<i>Potamogeton natans</i>	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
POPE	<i>Polygonum persicaria</i>	Spotted ladysthumb	FACW	no	0.0	0.3	3.9	1.1	0.0	0.9	0.1	0.0
PORI	<i>Potamogeton richardsonii</i>	Richardson's pondweed	OBL	yes	0.0	0.0	1.0	0.0	0.5	0.0	0.0	0.0
RARE	<i>Ranunculus repens</i>	Creeping buttercup	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
RASC	<i>Ranunculus sceleratus</i>	Celery-leaved buttercup	OBL	yes	0.0	0.0	0.0	0.0	0.0	T	0.0	0.0

Code	Scientific Name	Common Name	Wetland		Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
			Status	Native								
ROPA	<i>Rorippa palustris</i>	Marsh yellow-cress	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1
RUAQ	<i>Rumex aquaticus</i>	Western dock	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
RUAR	<i>Rubus armeniacus</i>	Himalayan blackberry	FACU	no	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
RUCR	<i>Rumex crispus</i>	Curly dock	FAC	no	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
RUMA	<i>Rumex maritimus</i>	Golden dock, seaside dock	FACW	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
SALA	<i>Sagittaria latifolia</i>	Wapato	OBL	yes	0.0	5.3	0.3	9.2	15.3	25.9	19.8	2.8
SALU	<i>Salix lucida</i>	Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.6
SALU*	<i>Salix lucida</i>	Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	7.1	0.2	13.9
SASI*	<i>Salix sitchensis</i>	Sitka willow	FACW	yes	0.0	0.0	0.0	0.4	1.2	0.0	0.0	0.0
SCAM	<i>Schoenoplectus americanus</i>	American bulrush, threesquare bulrush	OBL	yes	4.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCAR	<i>Schedonorus arundinaceus</i>	tall fescue	FAC	no	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCMA	<i>Schoenoplectus maritimus</i>	Seacoast bulrush	OBL	yes	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCTA	<i>Schoenoplectus tabernaemontani</i>	Softstem bulrush, tule	OBL	Yes	0.0	0.1	5.1	0.0	0.0	0.0	0.0	0.0
SISU	<i>Sium suave</i>	Hemlock waterparsnip	OBL	yes	0.0	0.4	0.0	1.3	0.0	0.0	0.0	0.0
SODU	<i>Solanum dulcamara</i>	Bittersweet nightshade	FAC	no	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0
SPAN	<i>Sparganium angustifolium</i>	Narrowleaf burreed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
SPEU	<i>Sparganium eurycarpum</i>	giant burreed	OBL	yes	0.0	0.0	3.9	0.0	0.0	0.0	0.0	0.0
SYEA	<i>Symphyotrichum eatonii</i>	Eaton's aster	FAC	yes	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0
SYSU	<i>Symphyotrichum subspicatum</i>	Douglas aster	FACW	yes	1.0	0.3	0.3	0.7	0.1	0.0	0.0	0.0
TRMA	<i>Triglochin maritima</i>	Seaside arrowgrass	OBL	yes	5.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TRWO	<i>Trifolium wormskioldii</i>	Springbank clover	FACW	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
TYAN	<i>Typha angustifolia</i>	Narrowleaf cattail	OBL	no	0.5	0.0	0.0	0.0	2.2	0.0	0.0	0.0
VEAM	<i>Veronica americana</i>	American speedwell	OBL	yes	0.0	0.0	0.1	0.1	0.2	0.4	0.1	0.8
ZAPA	<i>Zannichellia palustris</i>	horned pondweed	OBL	yes	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
Other Cover												
	Algae	algae			6.0	0.0	13.8	0.0	0.0	0.0	0.0	0.0
	BG	bare ground			19.7	5.3	22.6	4.3	4.4	15.7	6.7	9.9
dCALY	<i>Carex lyngbyei</i> , dead	Lyngby sedge, dead	OBL	yes	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
dAGSP	<i>Agrostis sp.</i> , dead	bentgrass, dead	mixed	mixed	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
dELPA	<i>Eleocharis palustris</i> , dead	Common spikerush, dead	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	5.4	0.0
	Detritus	detritus			0.8	2.6	0.0	0.0	2.7	0.6	2.4	7.8
dPHAR	<i>Phalaris arundinacea</i> , dead	Reed canary grass, dead	FACW	no	0.0	0.0	0.0	0.0	0.4	0.0	4.6	0.0
dPOAM	<i>Polygonum amphibium</i> , dead	water ladysthumb, water smartweed, dead	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.8
	DW	drift wrack			0.2	5.3	0.8	0.1	0.0	0.0	0.0	0.0
	FGA	Filamentous green algae			0.0	0.0	0.0	0.0	0.0	3.8	0.0	0.0
FUDI	<i>Fucus distichus</i>	Rockweed	OBL	yes	10.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Litter	litter			0.0	0.0	0.0	0.0	0.0	0.2	0.1	5.3
	LW	live wood rooted in quadrat			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
	LWD	large woody debris			0.0	0.1	0.0	0.0	1.4	0.0	0.1	1.0
	MOSS	Moss			0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
	OW	open water			9.4	2.5	68.5	22.3	4.8	2.5	6.5	9.3
	Rock	rock			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	SD	standing dead			6.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	SMH	small mixed herbs			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
	SWD	small woody debris			0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.4
ULLA	<i>Ulva lactuca</i>	Sea lettuce	OBL	yes	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0

T = Trace

* = Overhead vegetation

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 2015. The three dominant cover classes are bolded in red for each site and non-native species are shaded in yellow. 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 site.

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret	Welch	Whites	Campbell	Franz	
											Elevation (m, CRD)
					Min	0.90	0.13	0.46	0.14	-0.07	0.68
					Avg	1.01	0.36	0.62	0.35	0.69	0.93
					Max	1.18	0.69	0.95	0.61	0.91	1.19
					Average Percent Cover						
ALTR	<i>Alisma triviale</i>	northern water plaintain	OBL	yes	0.0	0.0	1.3	1.1	0.0	0.0	
BICE	<i>Bidens cernua</i>	Nodding beggars-ticks	OBL	yes	0.0	0.0	9.3	1.1	0.0	0.0	
CAHE	<i>Callitriche heterophylla</i>	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.0	10.0	1.1	0.0	0.0	
CAHE2	<i>Callitriche hermaphroditica</i>	northern water-starwort	OBL	yes	0.0	0.0	0.3	0.0	0.0	0.0	
CALY	<i>Carex lyngbyei</i>	Lyngby sedge	OBL	yes	0.2	0.0	0.0	0.0	0.0	0.0	
CEDE	<i>Ceratophyllum demersum</i>	Coontail	OBL	yes	0.0	0.1	0.3	0.0	0.0	0.0	
DIAC	<i>Dichanthelium acuminatum</i>	western panicgrass	FAC	yes	0.0	0.0	0.0	0.0	0.0	0.1	
ELAC	<i>Eleocharis acicularis</i>	Needle spikerush	OBL	yes	0.0	0.1	16.0	0.0	0.0	0.0	
ELCA	<i>Elodea canadensis</i>	Canada waterweed	OBL	yes	0.0	20.6	27.5	13.9	5.8	0.0	
ELNU	<i>Elodea nuttallii</i>	Nuttall's waterweed, western waterweed	OBL	yes	0.0	0.4	0.0	0.0	0.0	0.0	
ELPA	<i>Eleocharis palustris</i>	Common spikerush	OBL	yes	0.0	0.0	7.8	0.0	0.0	0.0	
ELPAR	<i>Eleocharis parvula</i>	Dwarf spikerush	OBL	yes	0.2	0.0	0.0	0.0	0.0	0.0	
EQFL	<i>Equisetum fluviatile</i>	Water horsetail	OBL	yes	0.0	0.0	0.2	0.0	0.0	0.0	
GRNE	<i>Gratiola neglecta</i>	American Hedge- hyssop	OBL	yes	0.0	0.0	0.2	0.0	0.0	0.0	
LIAQ	<i>Limosella aquatica</i>	Water mudwort	OBL	yes	0.0	0.0	1.2	0.1	0.0	0.0	
LISC	<i>Lilaea scilloides</i>	Flowering quillwort	OBL	yes	0.0	0.1	0.0	0.0	0.0	0.0	

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret	Welch	Whites	Campbell	Franz
MYSC	<i>Myosotis scorpioides</i>	Common forget-me-not	FACW	no	0.0	0.3	0.0	0.0	0.0	0.0
MYSI	<i>Myriophyllum sibiricum</i>	northern milfoil, short spike milfoil	OBL	yes	0.0	0.0	0.0	1.8	0.0	0.0
MYSP2	<i>Myriophyllum spp.</i>	Milfoil	OBL	mixed	0.0	0.0	0.0	0.0	4.3	0.0
POAM	<i>Polygonum amphibium</i>	water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.1
POCR	<i>Potamogeton crispus</i>	Curly leaf pondweed	OBL	no	0.0	11.4	0.0	0.0	30.0	0.0
POPE	<i>Polygonum persicaria</i>	Spotted ladysthumb	FACW	no	0.0	0.0	7.7	0.0	0.0	0.0
POPU	<i>Potamogeton pusillus</i>	Small pondweed	OBL	yes	0.0	0.1	0.3	0.0	0.3	0.0
PORI	<i>Potamogeton richardsonii</i>	Richardson's pondweed	OBL	yes	0.0	45.0	2.0	20.8	0.0	0.0
POZO	<i>Potamogeton zosteriformis</i>	Eelgrass pondweed	OBL	yes	0.0	0.0	0.0	1.8	0.0	0.0
SALA	<i>Sagittaria latifolia</i>	Wapato	OBL	yes	0.0	0.0	0.3	0.1	0.0	0.9
VEAM	<i>Veronica americana</i>	American speedwell	OBL	yes	0.0	0.0	0.2	0.0	0.0	0.0
ZAPA	<i>Zannichellia palustris</i>	horned pondweed	OBL	yes	0.0	0.9	0.2	0.0	0.0	0.0
ZOJA	<i>Zostera japonica</i>	Japanese eelgrass	OBL	no	6.6	0.0	0.0	0.0	0.0	0.0
Other Cover										
Algae		Algae			1.6	0.0	0.0	0.0	0.0	0.0
BG		bare ground			93.0	29.4	26.0	57.5	61.7	99.1
Detritus		detritus			0.2	0.1	0.0	0.0	0.0	0.0
DW		drift wrack			1.0	0.0	1.7	0.0	0.0	0.0
FGA		Filamentous green algae	OBL		0.0	2.5	0.0	0.0	2.5	0.0
OW		open water			37.0	75.0	4.2	0.0	0.0	100

Appendix D. Annual photo points from EMP trends sites

Ilwaco Slough – PP1

31 July 2011



15 February 2012



4 August 2012



4 February 2013



Ilwaco Slough – PP1

26 July 2013



20 September 2013



3 February 2014



27 June 2014



Ilwaco Slough – PP1

2 August 2015



Secret River – PP1 [HIGH MARSH]

5 February 2010



2 August 2012



9 August 2013



1 August 2015



Secret River – PP2 [LOW MARSH]

1 December 2011



2 August 2012



15 July 2014



1 August 2015



Secret River – PP3 [CHANNEL]

1 December 2011



15 July 2014



1 August 2015



Welch Island – PP1

1 August 2012



3 February 2013



23 July 2013



1 August 2014



31 July 2015



Whites Island – PP1

22 July 2009



13 July 2010



2 August 2011



15 February 2012



31 July 2012



5 February 2013



22 July 2013



Whites Island – PP1

4 February 2014



31 July 2014



30 July 2015



Cunningham Lake – PP1

26 July 2005



18 July 2007



21 July 2008



Cunningham Lake – PP1

25 July 2009



17 May 2010



28 July 2010



Cunningham Lake – PP1

30 July 2011



8 August 2012



29 July 2013



Cunningham Lake – PP1

18 July 2014



28 July 2015



Campbell Slough – PP1

29 July 2005



15 July 2006



5 September 2006



Campbell Slough – PP1

17 July 2007



26 July 2010



29 July 2011



Campbell Slough – PP1

15 February 2012



21 July 2012



10 August 2012



Campbell Slough – PP1

27 July 2013



18 July 2014



29 July 2015



Campbell Slough – PP2

25 July 2005



27 July 2009



26 July 2010



29 July 2011



10 Aug 2012



Campbell Slough – PP2

27 July 2013



18 July 2014



29 July 2015



Campbell Slough – PP2

22 July 2008



28 July 2009



25 August 2011



14 February 2012



21 July 2012



Campbell Slough – PP2

30 August 2012



11 October 2012



6 February 2013



31 July 2013



Campbell Slough – PP2

12 February 2014



7 August 2014



27 July 2015

