Lower Columbia River Ecosystem Monitoring Program

Annual Report for Year 10

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Technical Contact: Amanda Hanson
Research Scientist
Lower Columbia Estuary Partnership
811 SW Naito Parkway, Suite 410
Portland, Oregon 97204
Phone: (503) 226-1565 x239
ahanson@estuarypartnership.org

BPA Project Manager: Jason Karnezis
Fish & Wildlife Project Manager
Bonneville Power Administration
905 NE 11th Avenue
Portland, Oregon 97208
Phone: (503)230-3098
jpkarnezis@bpa.gov

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Amanda C. Hanson[1]
Amy B. Borde[2]
Lyndal L. Johnson[3]
Tawnya D. Peterson[4]
Joseph A. Needoba[4]
Shon A. Zimmerman[2]
Matthew Schwartz[1]
Cynthia L. Wright[2]
Paul M. Chittaro[3]
Sean Y. Sol[3]
David J. Teel[3]
Gina M. Ylitalo[3]
Daniel Lomax[3]
Claudia E. Tausz[4]
Heida L. Diefenderfer[2]
Catherine A. Corbett[1]

Prepared by the Lower Columbia Estuary Partnership
with funding from the Bonneville Power Administration

Lower Columbia Estuary Partnership
811 SW Naito Parkway, Suite 410
Portland, OR 97204

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[1] Lower Columbia Estuary Partnership
[2] Pacific Northwest National Laboratories
[3] Northwest Fisheries Science Center, NOAA-National Marine Fisheries Service
[4] Oregon Health and Sciences University

1 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 out-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, 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 2014, data were collected on fish, habitat, hydrology, food web, abiotic site conditions, and mainstem river conditions at 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 Ilwaco Slough (rkm 6), Secret River (rkm 37), and Cunningham Lake (rkm 145) in 2014. 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. Data were collected and analyzed by regional experts from the National Oceanic and Atmospheric Administration National Marine Fisheries Service (NOAA-Fisheries; fish data), Battelle-Pacific Northwest National Laboratory (PNNL; habitat and hydrology data), United States Geological Survey (USGS; abiotic site conditions and food web data), and Oregon Health and Sciences University (OHSU; mainstem river conditions and food web data).

An understanding of conditions within the mainstem of the Columbia River is critical because shallow, off-channel habitats are influenced by flows in the mainstem. Water temperature in both the mainstem and at trends sites were generally less than established threshold values during the juvenile salmonid outmigration period, although daily average temperatures, at times, exceeded 19°C later in the summer (July – September). Data collected from an in situ mainstem water quality monitoring platform at river mile 122 show high dissolved nutrient concentrations in the winter and low concentrations in late summer. Nutrient ratios indicate that phosphorus may be a limiting factor for phytoplankton growth in the mainstem. In addition, reoccurring annual patterns of increased nitrate levels and decreased dissolved oxygen in early September coincides with the cessation of managed spill from Bonneville Dam. However, interpretation of the mainstem sensor data is limited because of the lack of information from upstream of the dam with which to compare it. That is, we cannot separate effects of downstream advection from in situ growth. While the data in their current form are valuable, additional data from upstream of the dam would refine the observations and make interpretations more robust.

Chlorophyll *a* concentrations and phytoplankton assemblages (diatom-dominated) were similar at Welch Island and Whites Island throughout the 2014 monitoring season. In contrast, phytoplankton assemblages at Campbell Slough and Franz Lake became increasingly dominated by cyanobacteria after the freshet when water levels decreased and temperatures increased. At these two sites, ratios of nitrogen to phosphorus were much lower than at the sites in Reach B and C, which tends to favor cyanobacteria over other types of phytoplankton. Zooplankton were most abundant at Campbell Slough, with peak abundance occurring in June and July. Peak chlorophyll *a* concentration coincided with high abundances of zooplankton and at all sites, zooplankton assemblages were dominated by small rotifers.

Vegetation composition and cover, sediment accretion, and hydrologic patterns were monitored at seven emergent wetland sites in the lower river. Sediment accretion rates were variable in time and space (-0.5 to 3.0 cm/yr). Accretion rates were generally greater at lower-elevations, closer to the channel; however,

the opposite effect may occur along a natural levee adjacent to a channel, such as at the Secret River high marsh site, where the accretion rates were greater along the higher elevation levee than in the lower elevation areas of the backmarsh behind the levee. At the three upper estuary trends sites, average annual sediment accretion rates were frequently greater than 1.0 cm per year, likely due to the greater sediment load of the river at the upper end of the estuarine gradient. Even though sediment loads are relatively low compared to historical levels, the current sediment loads appear to remain adequate for increasing wetland area in the lower river over time. Inundation magnitude varies spatially, generally increasing with distance from the river mouth. 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; however, a shift in vegetation community occurred at Franz Lake in recent years where the species dominance shifted from reed canarygrass, to water smartweed (*Polygonum amphibium*), a native species.

Fish data collected under the EMP have shown differences among genetic stocks for growth, lipid content, size, and presence across the trends sites. The data show that marked and unmarked Chinook salmon from various stocks, including Upper Columbia Fall, Snake River Fall, Spring Creek Fall, Deschutes, and West Cascades Chinook salmon use off-channel habitat in the lower river during outmigration. The majority of PIT tagged fish detections at Campbell Slough occurred during May 2014, which coincided with hatchery releases elsewhere in the region. Most of the detected fish were juvenile hatchery fall Chinook salmon from Spring Creek Hatchery, although wild and hatchery spring Chinook salmon, coho salmon, steelhead, and sockeye salmon were also detected by the array, some of which originated from hatcheries located at far away as Idaho. In the upper reaches of the lower river, fish species diversity and richness were greater and non-native and predatory fish were more common. Salmonid occurrence patterns were similar to previous years, except that chum salmon were not detected at all in 2014 at any of the EMP sites; however, small numbers of sockeye salmon have been detected at several EMP sites in recent years (Welch Island, Campbell Slough, and Franz Lake).

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. 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. 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 restoration design and translate to additional reference data for comparison to regional action effectiveness monitoring efforts.

Acknowledgments

This study could not have been completed without the help of our partners. We are grateful to the Northwest Power and Conservation Council and the Bonneville Power Administration for funding the Ecosystem Monitoring Program through the Columbia Basin Fish and Wildlife Program. Whitney Temple and Jennifer Morace could not be listed as co-authors because at the time of this report USGS peer-review was not complete. USGS collected and analyzed abiotic conditions at four of the trends sites and portions of the food web, including the stable isotope analysis. We thank them immensely for their collaborative work on this program. This effort could not have been completed without the help of numerous field assistants: we would like to thank Nichole Sather, Eric Fischer, and Allie Simpson from PNNL; Dave Piatt from USGS; and Daniel Evans from the Estuary Partnership. We also thank the land owners and managers who have allowed us to conduct research on lands they manage, including Alex Chmielewski (Ridgefield National Wildlife Refuge) and Paul Meyers (Lewis and Clark National Wildlife Refuge). Finally, the Estuary Partnership's Science Work Group provided valuable input throughout the process and peer review on final drafts. The Science Work Group is composed of over 60 members, and is integral in ensuring the Estuary Partnership represents the best available science.

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2 Introduction

2.1 Background

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. Emergent wetlands in the lower Columbia River cover a narrow elevation range (1.83 m, 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. 2015). 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 seasonally drive 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 of 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 have irreplaceable value 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 executes 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).

2.2 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 on habitats supporting juvenile salmonids, in tidally influenced shallow water emergent wetlands connected to the Columbia River.

The Estuary Partnership and monitoring partners use a multi-scaled stratification sampling design for 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

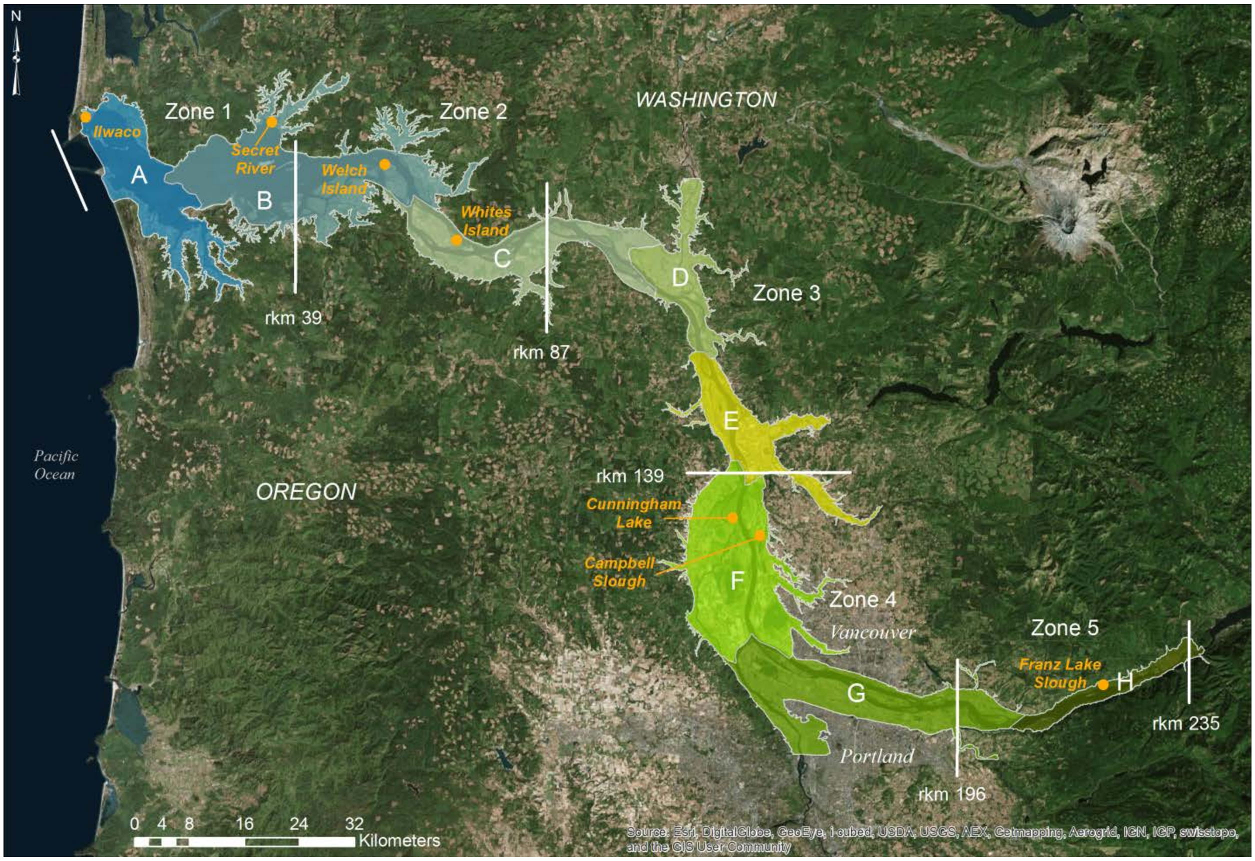


Figure 1. Lower Columbia River and estuary with hydrogeomorphic reaches (A-H) specified by color (Simenstad et al. 2011) and wetland zones (1-5) delineated by white lines (Jay et al. in revision). The 2014 EMP sites are shown in orange.

2.3 Characterization of Emergent Wetlands in the Lower Columbia River

2.3.1 Sampling Effort, 2005-2014

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 site conditions (i.e., conditions influencing productivity such as temperature, water clarity, dissolved oxygen, nutrients) sampling and analysis in both the mainstem Columbia River and trends sites to the EMP.

In 2012, 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 (2010-2014), Secret River (2010-2014), Welch Island (2010-2014), Whites Island (2009-2014), Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2014), and Franz Lake (2008-2009, 2011-2014). In 2014, a separate research study objective was undertaken by some EMP partners to address a critical uncertainty research question posed by the Expert Regional Technical Group (ERTG) and resulted in additional adjustments to the annual monitoring sampling plan. Thus, two EMP trends sites (i.e., Ilwaco Slough and Secret River) were not sampled for the majority of the annual monitoring metrics in 2014 (juvenile salmon data, food web dynamics, and abiotic conditions). Habitat and hydrology data, however, were collected at all trends sites including Ilwaco Slough and Secret River as well as at Cunningham Lake. This last site serves as a reference site for habitat and hydrology representative of Reach F sites because vegetation has been infrequently trampled by livestock at Campbell Slough in past years. In addition, some metrics typically collected on an annual basis at EMP sites were excluded for 2014, such as salmon diet, vegetation biomass, and benthic macroinvertebrate samples.

Activities Performed, Year 10 Contract (October 1, 2013 – September 30, 2014):

- Salmonid occurrence, composition, growth, condition 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 2014.

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat[1]	Fish &Prey	Abiotic Conditions	Food Web[2]
A	Trend	**Ilwaco Slough**	BBM	2011-2014	2011-2013	2011-2013	2011-2013
B	Trend	**Secret River**	SRM	2008[3], 2012-2014	2012, 2013		2012, 2013
	Trend	**Welch Island**	WI2	2012-2014	2012-2014[6]	2014	2012-2014
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	2009-2014	2009-2014[6]	2009, 2011-2014	2011-2014
	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	Lewis River Mouth	NNI	2007			
	Status	Martin Island	MIM	2007			
F	Status	Sauvie Cove	SSC	2005			
	Status	Hogan Ranch	HR	2005			
	Status	Goat Island	GIC	2011	2011		
	Status	Deer Island	DIC	2011	2011		
	Status	Burke Island	BIM	2011	2011		
	Trend	**Cunningham Lake**	CLM	2005-2014	2007-2009		
	Trend	**Campbell Slough**	CS1	2005-2014	2007-2014[6]	2008-2014	2010-2014[5]
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**	-			2012-2014	
H	Trend	**Franz Lake (slough)**	FLM	2008-2009, 2011-2014	2008-2009, 2011-2014[6]	2011-2014	2011-2014
	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.
[2] In Reach B abiotic conditions sampling was not conducted at Welch Island until 2014.
[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 at Lord-Walker 2.
[5] Phytoplankton and zooplankton only sampled from 2011 – 2014.
[6] In 2014, fish prey data were not collected for juvenile Chinook salmon diet and prey availability analyses.

2.3.2 Site Descriptions

In 2014, the EMP focused primarily on four of the six trends sites that have been monitored for multiple years. Habitat and hydrology data, however, 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). Coordinates for trends sites sampled in 2014 are listed in Table 2. The 2014 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 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.

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, 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, and in 2012 the site was heavily grazed and trampled by cows.

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 2014: (a) Ilwaco Slough; (b) Secret River; (c) Welch Island; (d) Whites Island; (e) Cunningham Lake; (f) Campbell Slough; (g) Franz Lake. Ilwaco Slough, Secret River, and Cunningham Lake were only sampled for habitat and hydrology metrics in 2014.

2.3.3 Water Year

The 2014 water year was characterized by lower than average water levels in the early winter months (January and February) followed by a period of higher than average water levels in late winter and early spring (Figure 3). This resulted in an early and rather prolonged spring freshet compared to the 29-year mean. Summer and fall flows were similar, and at times slightly higher than average conditions. Hydrographs for the monitoring sites in 2014 are provided in Appendix A.

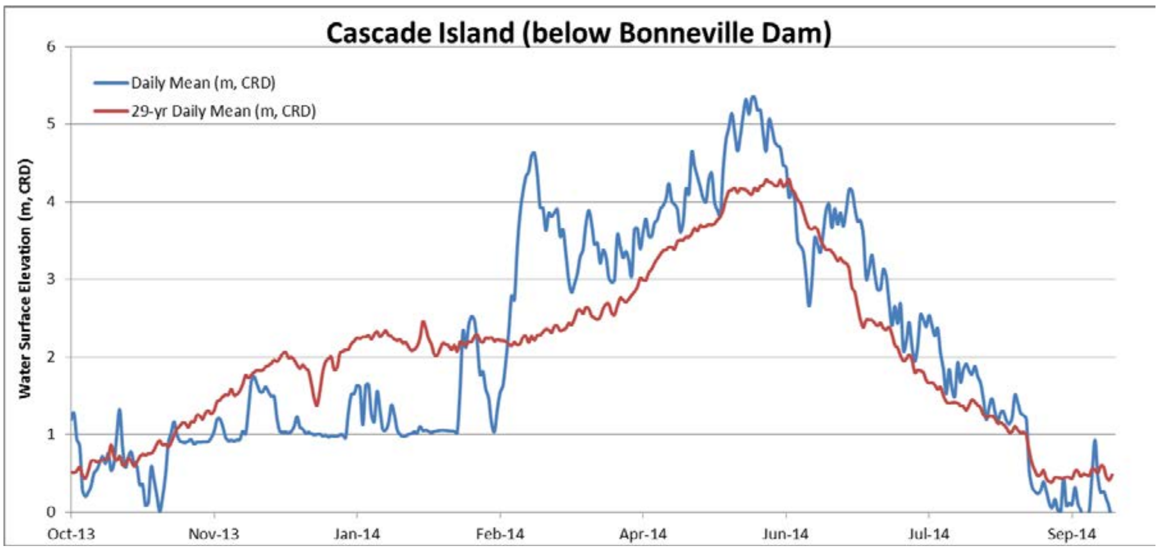


Figure 3. Water surface elevation at Cascade Island, just below Bonneville Dam (rkm 233), from October 2013 to October 2014 compared to the 29-year daily mean water surface elevation (Data from USGS National Water Information System at: http://waterdata.usgs.gov/nwis/).

3 Methods

3.1 Mainstem Conditions

3.1.1 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). 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 collecting unprecedented 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 for assessing ecosystem conditions that allows the comparison of conditions stemming from upstream to conditions in the estuary (i.e., conditions resulting from lower river tributaries). One such product is flux calculations for various inorganic or organic components such as nitrate 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 loading to the coastal zone, where many important ecological processes may be affected. Another product is the assessment of Net Ecosystem Metabolism, which provides a daily measure of the gross primary production and aerobic respiration occurring in the river as measured by hourly changes in dissolved oxygen.

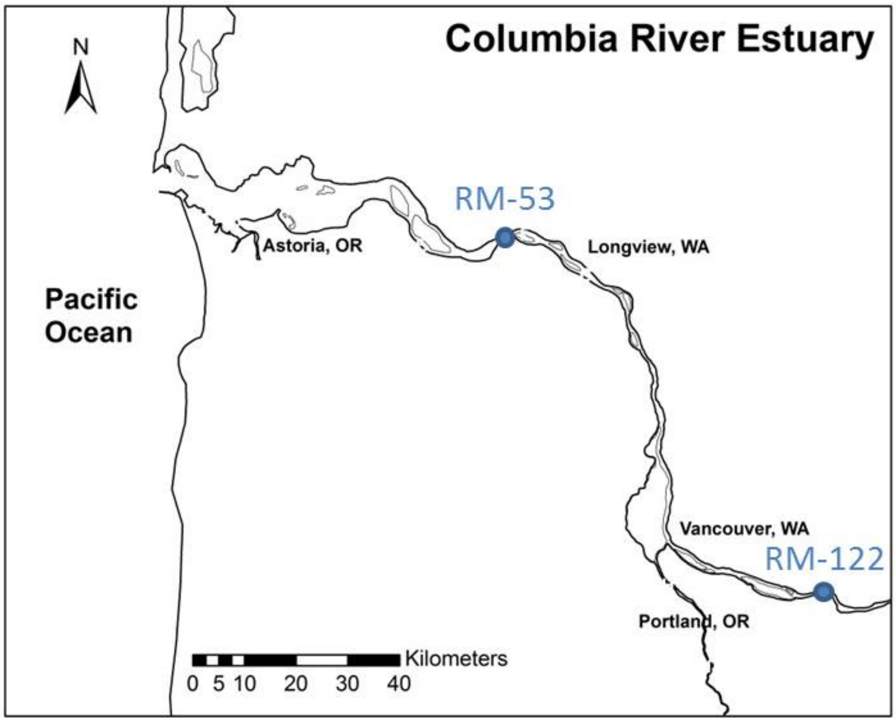


Figure 4. Station locations for the two in situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program.

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

The instrument platform ran continuously from September 2013 – December 2013, and from July 2014 – December 2014. In December 2013 the instruments were removed for service and calibration. The goal was to have the sensors returned and redeployed by April 2014. The sensors were shipped in early January but were not serviced and returned until June 2014. During the interim period, a Yellow Springs Instruments (YSI) 6600 V2 sonde was deployed as a back-up instrument. The first deployment occurred in April 2014 but only collected data from April 24 – May 4 due to battery issues. The sensor was redeployed June 11 – June 30. The Cycle-PO4 was operated continuously from August 11 – October 29, 2014.

3.1.3 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). During the sampling period, the Cycle-PO4 ran for 182 days between January 16, 2013 and September 12, 2013. The filter size on the instrument is 10 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 *a* Concentration

3.1.4 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, September 2013-December 2014.

RM-122
9/3/2013
12/15/2013
4/25/2014
6/11/2014
6/30/2014
7/11/2014
8/13/2014
10/29/2014

3.1.5 Quality Control

Initial sensor calibration was performed by the manufacturer. Each instrument is supplied with a certificate of calibration, and where appropriate, instructions for recalibration. For example, the 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	Correlation
RM-122/Nitrate/46	Y = 0.95x +1 r^2 = 0.99
RM-122/Chl/13	Y = 0.8x +1 r2 = 0.93

3.2 Abiotic site conditions

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

In 2014, USGS monitored water quality at four of the trends sites, Franz Lake, Campbell Slough, Whites Island, and Welch Island (Table 6).

Table 6. Locations of water quality monitors at trends sites in 2014.

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"	April 9	August 8
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47' 05"	-122° 45' 15"	April 11	August 6
Whites Island	460939123201600	Birnie Slough, White's Island, Columbia River, WA	C	46° 09' 39"	-123° 20' 16"	April 10	August 5
Welch Island	461518123285700	Unnamed Slough, Welch Island, Columbia River, OR	B	46° 15' 18.4"	-123° 28' 56.8"	April 10	August 5

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

The water quality monitors were YSI models 6600EDS and 6920V2, equipped with water temperature, specific conductance, pH, and dissolved oxygen probes. 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 outmigration period. In 2014, 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, 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 3.1).

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

Monitoring Metric	Range	Resolution	Accuracy
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

3.2.2 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 3.4.1.1.

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

3.3 Habitat Structure

In 2014, 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 2014 monitoring sites are presented in Appendix B.

Table 9. Site location and sampling dates for each site sampled in 2014. 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	6/27/2014
Secret River (low marsh)	SRM-L	37	Trend	7/14/2014
Secret River (high marsh)	SRM-H	37	Trend	7/14/2014
Welch Island 2*	WI2	53	Trend	8/1/2014
Whites Island*	WHC	72	Trend	7/31/2014
Cunningham Lake	CLM	145	Trend	7/18/2014
Campbell Slough*	CS1	149	Trend	7/18/2014
Franz Lake*	FLM	221	Trend	8/7/2014

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

3.3.1 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 2014 we focused our efforts on vegetation, 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).

3.3.2 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. A salinity sensor was added to the Ilwaco Slough site in 2011. Also starting in 2011, plant biomass was collected at all of the trends sites, excluding Cunningham Lake. 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 channel cross sections were not measured in 2014. Biomass collection was not part of the 2014 sampling effort due to funding constraints. Photo points were also designated at each site from which photographs were taken to document the 360-degree view each year.

3.3.2.1 *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 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. Sensor failure or loss has occurred, yet the sensors have been downloaded and redeployed every year since the initial deployment for collection of a nearly continuous dataset (Appendix A).

3.3.2.2 *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 2014. The accretion or erosion rate is calculated by averaging the 11 measurements along the one meter distance from each year and comparing the difference.

3.3.2.3 *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 2014, the sensor was cleaned and downloaded, and a verification sample was taken.

3.3.2.4 *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).

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 m^2 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.

3.3.2.5 *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). All trends sites were mapped in previous years and were not re-mapped in 2014 because no large-scale changes were observed at the sites.

3.3.2.6 *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 2014, elevation was re-measured at Ilwaco Slough, Secret River, and Cunningham Lake. Elevations from previous years were used at Welch Island, Whites Island, Campbell Slough, and Franz Lake. 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.

3.3.2.7 *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. 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:

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

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 trend 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 an high water years to predict results in higher water years.

3.3.2.8 *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.

3.4 Food Web

3.4.1 Primary Productivity

3.4.1.1 *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.

Table 10. List of samples analyzed (Xs) and data of collection from four trends sites in the Lower Columbia River in 2014.

Site	Reach	Date	Nutrients	Zooplankton	Phytoplankton
WELCH ISLAND	B	4/9/14	X	X	X
	B	5/7/14	X	X	X
	B	5/8/14	X	X	X
	B	6/3/14	X	X	X
	B	6/9/14	X	X	X
	B	6/25/14	X	X	X
	B	7/7/14	X	X	X
	B	7/15/14	X	X	X
WHITES ISLAND	C	4/9/14	X	X	X
	C	5/7/14	X	X	X
	C	5/8/14	X	X	X
	C	6/3/14	X	X	X
	C	6/9/14	X	X	X
	C	6/25/14	X	X	X
	C	7/7/14	X	X	X
	C	7/15/14	X	X	X
CAMPBELL SLOUGH	F	4/8/14	X	X	X
	F	5/6/14	X	X	X
	F	5/9/14	X	X	X
	F	6/4/14	X	X	X
	F	6/10/14	X	X	X
	F	6/14/14	X	X	X
	F	6/26/14	X	X	X
	F	7/8/14	X	X	X
	F	7/16/14	X	X	X
FRANZ LAKE SLOUGH	H	4/9/14	X	X	X
	H	5/9/14	X	X	X
	H	6/4/14	X	X	X
	H	6/10/14	X	X	X
	H	6/26/14	X	X	X
	H	7/8/14	X	X	X
	H	7/16/14	X	X	X

3.4.1.2 *Emergent Wetland Vegetation*

Field Methods

From summer 2011 to winter 2014 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 February during the winter low biomass period. For the emergent marsh biomass sampling, a 1 m^2 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 m^2 biomass plot, a 0.1 m^2 quadrat was placed in a randomly selected corner and all rooted vegetation, live or dead, was removed using shears. Each sample was placed in a uniquely numbered bag, and held in a cooler for the remainder of the sampling trip. For the submerged aquatic vegetation (SAV) plots, similar methods were employed with the exception of the placement of the plots. Either existing transects were extended past the baseline or new transects were created to reach the main slough. In some instances, an existing transect intersected the slough and an SAV plot was randomly placed along it. Depending on the width of the channel, either one or two SAV plots were randomly placed along each transect. Dominant vegetation species were recorded in field notebooks along with the corresponding biomass sample number.

Lab Methods

In the laboratory, the biomass samples were stored in a cold room until processing could begin. The samples were then individually rinsed of all non-organic material, obvious root material was removed, and for the winter samples live and dead material was separated. Pre-weighed pieces of tinfoil were used to secure the individual biomass samples, a wet weight was then measured, and the samples were placed in an oven set at 90°C for three to four days. When the samples were deemed completely dry, a second weight was then measured for each sample, and entered either into a datasheet or directly into a spreadsheet software program.

3.4.2 Secondary Productivity

3.4.2.1 *Zooplankton*

Secondary productivity (the rate of growth of consumers of primary production) was not measured directly, but was estimated from the abundance of 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.

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:

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

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

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

The volume is determined as:

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

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

Taxonomy

Zooplankton taxa were divided into one of the following groupings: rotifers, cladocerans, annelids, ciliates, and copepods, and 'other'. Eggs of rotifers, cladocerans, and copepods were enumerated separately.

3.4.3 Stable Isotope Ratios

The ratios of carbon and nitrogen stable isotopes in tissues of consumers reflect the stable isotope ratios of their food sources (Neill and Cornwell 1992; France 1995). Stable isotope ratios can be useful for determining major food sources, provided that the food sources have distinct isotopic ratios. This approach can reveal assimilated food sources over a longer time period than point-in-time techniques (e.g., gut content analysis) and a combination of the two approaches is often recommended. Stable isotope analysis of carbon and nitrogen can be used to assess the relative importance of algae and wetland plants to the food web supporting juvenile salmonids. Most carbon atoms have 12 neutrons (^{12}C), but approximately 1% of carbon atoms have 13 neutrons (^{13}C). Similarly, most nitrogen atoms have 14 neutrons (^{14}N), while 0.36% has 15 neutrons (^{15}N). Lighter isotopes are metabolized preferentially over heavier isotopes, so consumers at higher trophic levels (higher in the food web) become enriched in the heavier isotopes. Therefore, the ratios of heavy to light isotopes ($^{15}N/^{14}N$ and $^{13}C/^{12}C$) in the tissues of food sources, plus a small compensation for the metabolic loss of light isotopes ("trophic fractionation"), are reflected in the tissues of consumers. Typically, with an increase of one trophic level (i.e., from a plant to an herbivore or an herbivore to a carnivore), the $^{15}N/^{14}N$ ratio increases by 2.2 to 3.4 parts per thousand ("per mil"; ‰), so stable isotope analysis of nitrogen is useful in determining trophic position. The $^{13}C/^{12}C$ ratio usually changes by less than 1‰, making stable isotope analysis of carbon useful for determining inputs of primary producers when the different primary producers analyzed have distinct stable isotope ratios.

The stable isotope ratios of carbon and nitrogen were measured in juvenile Chinook salmon 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 (see Section 3.5). In 2010 and 2011, skinned muscle samples were collected for analysis because isotopic signatures in muscle typically have lower variability than other types of tissue and muscle is a good long-term integrator of food sources for stable isotope analysis. Alternatively, isotopic signatures of more metabolically active tissues such as liver, mucus, or blood are good media with which to examine relatively recent dietary sources because they turn over more quickly than muscle, otoliths, or scales (Phillips and Eldridge 2006; Michener and Kaufman 2007; Church et al. 2009; Buchheister and Latour 2010). We began collecting epidermal mucus in 2012 from a subset of juvenile salmonids (from which muscle samples were also collected) to test the suitability of mucus for this analysis. Epidermal mucus samples were collected from individual juvenile salmonids as described by Church et al. (2009) and composited in order to meet the minimum sample mass requirements for the analysis. In 2013 and 2014, muscle, mucus, and liver samples were collected.

Aquatic invertebrates were collected by USGS staff in open water and in emergent vegetation at the water's margin using opportunistic sampling. The aquatic midge, *Chironomidae*, and 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**
Chironomidae	*Chironomidae*	Particulate organic matter (POM)	Particulate organic matter (POM)
Corophium spp.	*Corophium spp.*	Periphyton	Periphyton
Gammarus spp.	*Gammarus spp.*	Vegetation	Vegetation
Hatchery food			

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 12. 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 for later processing.

Table 12. Vegetation and macroalgae species collected for stable isotope analysis 2010–2014, by site.

Ilwaco Slough	Welch Island	Whites Island	Campbell Slough	Franz Lake
Carex lyngbyei	*Carex lyngbyei*	*Alisma triviale*	*Elodea nuttalii*	*Phalaris arundinacea*
Cladophora columbiana	*Equisetum sp.*	*Carex lyngbyei*	*Eleocharus palustris*	*Polygonum amphibium*
Eleocharis parvula	*Lysichiton americanus*	*Elodea canadensis*	*Myriophyllum spicatum*	*Schoenoplectus americanus*
Fucus distichus	*Myriophyllum spicatum*	*Elodea nuttalii*	*Phalaris arundinacea*	
Lilaeopsis occidentalis	*Phalaris arundinacea*	*Equisetum sp.*	*Potamogeton crispus*	
Schoenoplectus americanus	*Potamogeton richardsonii*	*Iris pseudacorus*	*Potamogeton natans*	
Ulva lactuca	*Sagittaria latifolia*	*Myriophyllum spicatum*	*Sagittaria latifolia*	
Zannichellia palustris	*Schoenoplectus americanus*	*Phalaris arundinacea*		
	Zannichellia palustris	*Potentilla anserina sp. Pacifica*		
		Potamogeton crispus		
		Stuckenia pectinata		
		Potamogeton richardsonii		
		Sagittaria latifolia		
		Typha latifolia		

Frozen filters, salmon tissue, invertebrate, and plant material were freeze dried using a Labconco FreezeZone model 77520 lyophilizer (Labconco Corp., USA). Freeze-dried plants of the same species from the same sampling date were composited and ground using a clean coffee grinder. Freeze-dried invertebrates of the same taxa from the same collection site and collection date were composited, ground using a clean glass mortar and pestle, and subsampled when enough material was present. Otherwise, whole bodies of all individuals of the same taxa from the same site were composited into a single sample. Skinned muscle tissue samples from individual juvenile salmonids were analyzed separately; muscle tissue samples from different bodies were not composited. Epidermal mucus samples were composited from multiple juvenile salmonid bodies in order to have sufficient sample mass for analysis in 2012. In 2013 and 2014, epidermal mucus samples were collected individually so that stable isotope signatures can be linked among the muscle, liver, and mucus from individual fish.

Stable isotope ratios of carbon ($^{13}C/^{12}C$; "delta^{13}C"; "$\delta^{13}C$") and nitrogen ($^{15}N/^{14}N$; "$\delta^{15}N$") were measured 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 to the light isotope were measured, compared to universal standards, and reported in permil (‰) units. Vienna PeeDee Belemnite and air were used as the standards for carbon and nitrogen, respectively.

3.5 Fish Use

3.5.1 Fish Community

In 2014, NOAA Fisheries monitored habitat use by juvenile Chinook salmon and other fishes at four trends sites, Franz Lake in Reach H (previously sampled in 2008 – 2013), Campbell Slough in Reach F (sampled from 2007-2013), Whites Island site in Reach C (sampled from 2009-2013), and Welch Island in Reach B (sampled in 2012 and 2013), 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 February 2014 through July 2014, then again in November and December 2014. 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. Fish sampling events conducted in 2014 are shown in Table 13.

Table 13. Number of beach seine sets per month at EMP sampling sites in 2014. NS = not sampled. Sampling was not conducted in August to October due to funding constraints and related delays in transfer of project funds.

Site	Feb	Mar	Apr	May	Jun	Jul	Nov	Dec	Total
Welch Island	NS[3]	4	2	2	1	2	3	NS[3]	14
Whites Island	3	2	1	2	3	2	3	NS[3]	16
Campbell Slough	NS[1]	NS[1]	3	3	3	3	3	3	18
Franz Lake	3	1	1	NS[2]	NS[2]	NS[2]	2	2	9
Total	6	7	7	7	7	7	11	5	57

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

When 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.

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

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

N = the total number of all individuals

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

Catch per unit effort (CPUE) and fish density were calculated as described in Roegner et al. (2009), with fish density reported in number per 1000 m^2.

3.5.2 Salmon Metrics

3.5.2.1 *Genetic Stock Identification*

Genetic stock identification (GSI) techniques 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 was estimated with a regional microsatellite DNA data set (Seeb et al. 2007) that includes baseline data for spawning populations from throughout the Columbia River basin (described in Teel et al. 2009). The overall proportional stock composition of Lower Columbia River samples was estimated with the GSI computer program ONCOR (Kalinowski et al. 2007), which implemented the likelihood model of Rannala and Mountain (1997). Probability of origin was estimated for the following regional genetic stock groups: Deschutes River fall Chinook; West Cascades fall Chinook; West Cascades Spring Chinook; Middle and Upper Columbia Spring Chinook; Spring Creek Group fall Chinook; Snake River Fall Chinook; Snake River Spring Chinook; Upper Columbia River Summer/Fall Chinook; and Upper Willamette River Spring Chinook (Seeb et al. 2007; Teel et al. 2009). West Cascades and Spring Creek Group Chinook are Lower Columbia River stocks.

3.5.2.2 *Lipid Determination and Condition Factor*

As part of our study we determined lipid content in 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 Iatroscan and lipid classes were determined by thin layer chromatography with flame ionization detection (TLC/FID), as described in Ylitalo et al. (2005).

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

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

3.5.2.3 *Otoliths (Growth Rates)*

Otoliths were extracted 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:

$$La = d + \frac{Lc - d}{Oc} Oa$$

$$DG = \frac{Lc - La}{a}$$

where *La* and *Oa* 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, *Lc* and *Oc* are the fish length and otolith radius at capture, respectively.

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, Julian day, genetic stock, marked or unmarked, river kilometer, distance to channel center, 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.

3.5.2.4 *Chemical Contaminants in Chinook salmon*

Persistent Organic Pollutants in Bodies

Composite body samples (with stomach contents removed) were extracted with dichloromethane using an accelerated solvent extractor. The sample extracts were cleaned up using size exclusion liquid chromatography and analyzed by gas chromatography/mass spectrometry (GC/MS) for PCB congeners; PBDE congeners; organochlorine (OC) pesticides including DDTs, hexachlorocyclohexanes (HCHs), chlordanes, aldrin, dieldrin, mirex, and endosulfans; and low (2-3 ring) and high (4-6 ring) molecular weight aromatic hydrocarbons as described by Sloan et al. (2004, 2006). Summed PCBs were determined by adding the concentrations of 45 congeners (PCBs 17, 18, 28, 31, 33, 44, 49, 52, 66, 70, 74, 82, 87, 95, 99, 101/90, 105, 110, 118, 128, 138/163/164, 149, 151, 153/132, 156, 158, 170/190, 171, 177, 180, 183, 187, 191, 194, 195, 199, 205, 206, 208, 209). Summed DDT levels (∑DDTs) were calculated by summing the concentrations of *p,p'*-DDT, *p,p'*-DDE, *p,p'*-DDD, *o,p'*-DDD, *o,p'*-DDE and *o,p'*-DDT. Summed chlordanes (∑CHLDs) were determined by adding the concentrations of heptachlor, heptachlor epoxide, g-chlordane, a-chlordane, oxychlordane, *cis*-nonachlor, *trans*-nonachlor and nonachlor III. Summed hexachlorocyclohexanes (∑HCHs) were calculated by adding the concentrations of a-HCH, b-HCH, g-HCH, and lindane. Summed low molecular weight aromatic hydrocarbons (∑LAHs) were determined by adding the concentrations of biphenyl, naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, 2,6-dimethylnapthalene, acenaphthene, fluorene, phenanthrene; 1-methylphenanthrene, and anthracene. Summed high molecular weight aromatic hydrocarbons (∑HAHs) were calculated by adding the concentrations of fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[a]pyrene, benzo[e]pyrene, perylene, dibenz[a,h]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, indenopyrene, and benzo[ghi]perylene. Summed total aromatic hydrocarbons (∑TAHs) were calculated by adding ∑HAHs and ∑LAHs.

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

3.5.2.5 *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.

4 Results

4.1 Mainstem Conditions

Observations of various biogeochemical properties are made at RM-122 (Reach G) every hour, allowing for continuous surveillance of mainstem conditions during periods when sensors are operational. The data illustrate that variables such as turbidity vary at short timescales, differing substantially from day to day or month to month when different seasons or years are compared (Figure 5 and Figure 6). 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. Notably, early winter 2014 had more frequent and larger spikes in turbidity than did 2013. In contrast, variables such as temperature vary according to the same general trends year to year (warmer in summer, cooler in winter) and differences between years are subtler. The data show that the maximum mainstem river water temperatures at RM-122 were similar in 2013 and 2014 (Figure 5 and Figure 6).

Dissolved oxygen (DO) percent saturation and chlorophyll *a* provide information about primary productivity. When percent saturation of DO (relative to expected saturation levels that vary according to Henry's Law) exceeds 100%, it generally indicates inputs from photosynthesis. DO percent saturation exceeded 100% throughout August in both 2013 and 2014, indicating high rates of primary production prior to the end of the period corresponding to managed spill (approximately September 1). After September 1, 2014, there was a sharp decrease both in the daily fluctuations in %DO as well as the peak values in both 2013 and 2014, suggesting that this may be a regular feature in the mainstem river. It is unclear whether this feature reflects a change in the balance between primary production and respiration behind Bonneville Dam when water is held back.

Summer chlorophyll *a* concentrations remained relatively low and invariant during both 2013 (Figure 5) and 2014 (Figure 6), despite much larger changes in DO percent saturation. In both years there was a small peak in dissolved oxygen in autumn, which occurred earlier in 2014 (beginning of October) compared to 2013 (beginning of November).

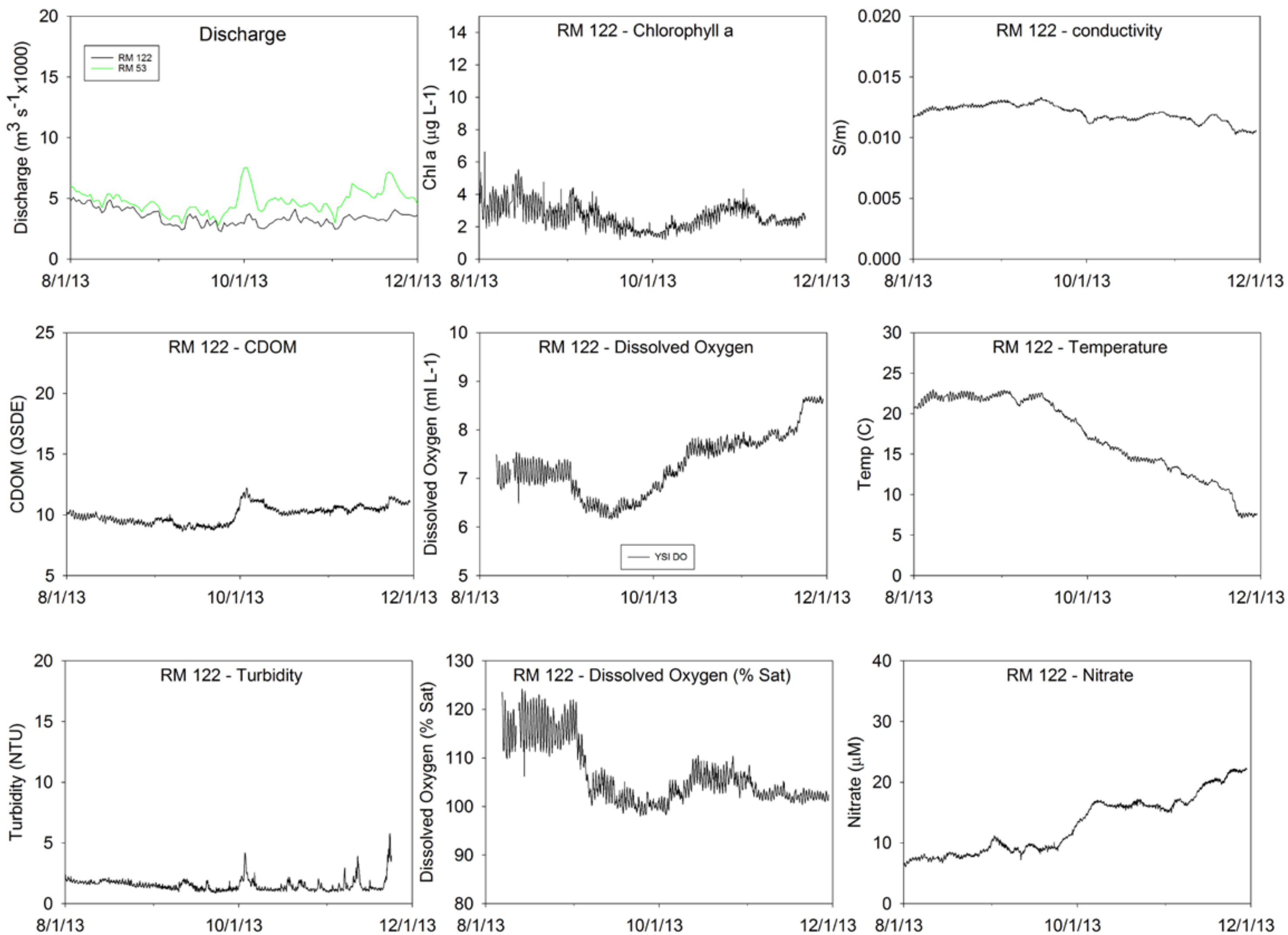


Figure 5. Time series data for mainstem river conditions metrics at RM-122 between 8/1/2013–12/1/2013.

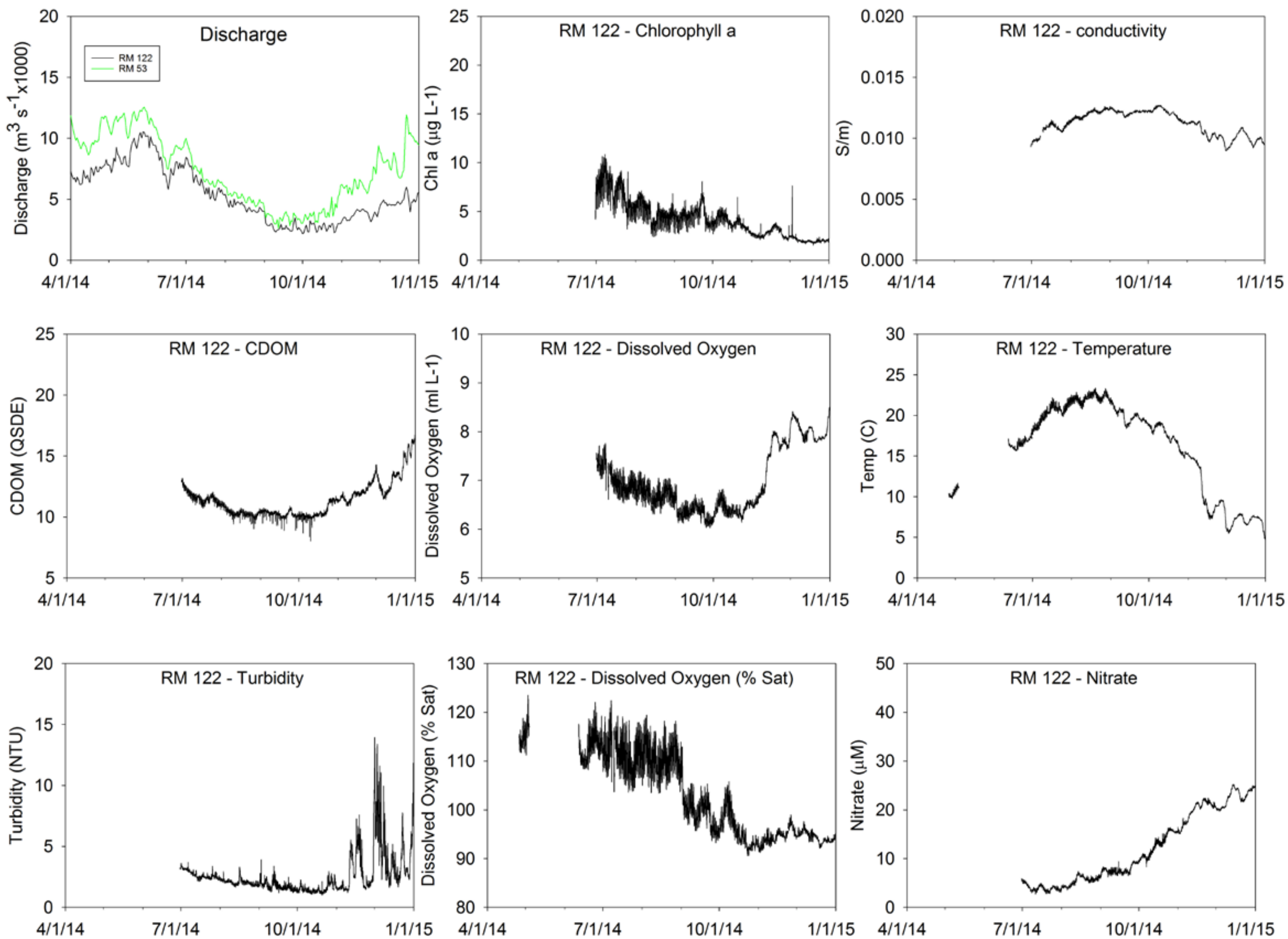


Figure 6. Time series data mainstem river conditions metrics collected at RM-122 between 4/1/2014–1/1/2015.

The average daily water temperatures observed at RM-122 in summer 2014 were higher than those observed in previous years at RM-53 (2009-2013; Table 14), with a 38-day stretch between July 28 and September 3, 2014 where temperatures exceeded 21^{o}C. Previous monitoring data show that water temperatures in the lower Columbia River are similar between RM-53 and RM-122 throughout the year and that the water column is well mixed (Sagar et al. 2015).

Table 14. Number of days with daily average temperatures >19°C or >21°C measured in the Columbia River mainstem at RM-53 (2009-2013) and RM-122 (2014).

Temperature range	2009	2010	2012	2013	2014
19-21 °C	70	49	53	67	30
> 21° C	11	2	2	14	42
Total > 19°C	82	51	55	81	72

Concentrations of nitrogen and phosphorus species were determined at RM-122 (Figure 7) in 2013 and 2014, and the data were divided by season: spring (Sp), summer (S), fall (F) and winter (W). Nitrate and ortho-phosphate concentrations were highest in winter and spring, while ammonium concentrations were highest in the late summer/autumn months. The ratio of available N and P (important in determining which nutrient could be limiting to phytoplankton growth) was highest in the spring, both for inorganic species (nitrate:phosphate) and total nutrient concentrations (TN:TP). Total nitrogen and phosphorus, which include dissolved and particulate forms, were highest in spring and autumn, respectively.

The sensor suite at Camas (RM-122) includes a Cycle-PO4, a reagent-based ortho-phosphate sensor that measured and reports hourly phosphate concentrations. The data show that both nitrate and phosphate are present at relatively low, constant concentrations during the summer and that they increase during the transition from to autumn to winter (Figure 8). Interestingly, nitrate levels increased somewhat abruptly on approximately September 1 when the managed spill ceased. Phosphate did not show this behavior.

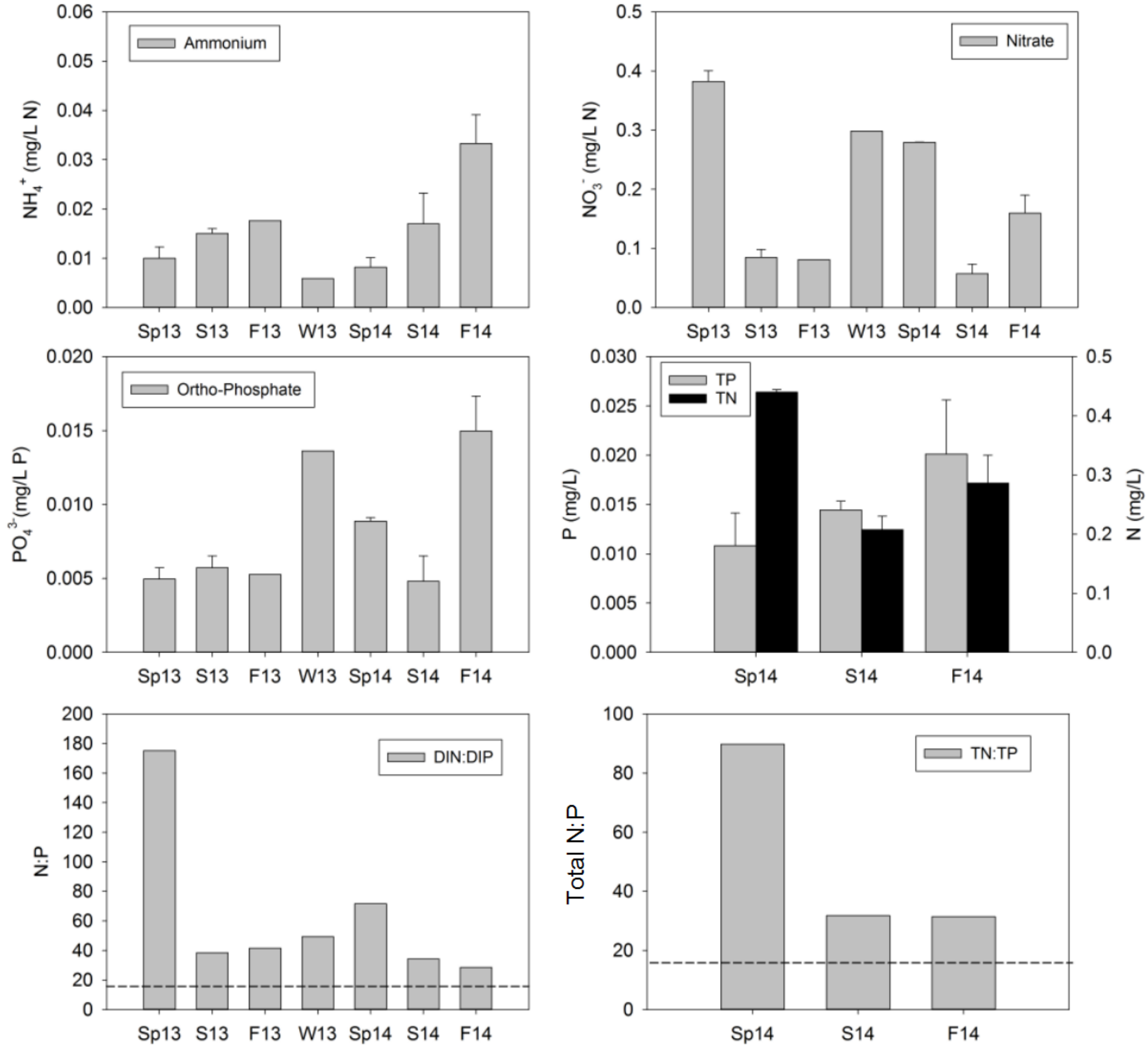


Figure 7. Nutrient sample data collected at RM-122 grouped into Spring (Sp), Summer (S), Fall (F) and Winter (W) for 2013 and 2014. TP = total dissolved phosphorus, TN = total dissolved nitrogen, DIN:DIP = dissolved inorganic nitrogen and phosphorus, TN:TP = total nutrient concentration. Dashed horizontal line indicates a 16N:1P ratio, that when exceeded, indicates a phosphorus limitation of phytoplankton growth.

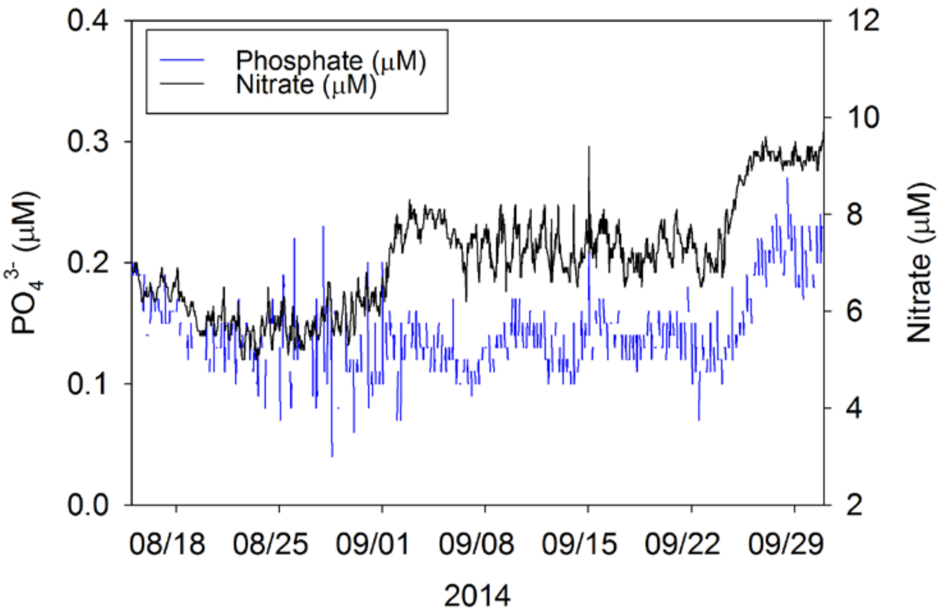


Figure 8. Time series of nitrate (black line) and phosphate (blue line) at RM-122 in 2014.

4.2 Abiotic Site Conditions

4.2.1 Continuous Water Quality

The magnitude and duration of the spring freshet has strong effects on water quality conditions of the mainstem, as well as in off-channel sites. Abiotic conditions were monitored at the trends sites between April and early August in 2014. During this period, peak Columbia River mainstem flows occurred May through early June and the peak freshet was higher than the 15-year mean flow (Figure 3). Also of note is the atypical peak in flow during March 2014, which could have flushed terrestrial organic matter into the river earlier in the spring than usual. To highlight mainstem river flows specific to the peak salmonid outmigration period, Figure 9 shows gauge height below Bonneville dam over multiple years. Compared to other water quality monitoring periods (April–July, 2011–2013), Columbia River streamflow below Bonneville Dam in 2014 was most similar to 2013, whereas 2011 and 2012 were characterized as high flow years with longer and larger freshet flows (Figure 9).

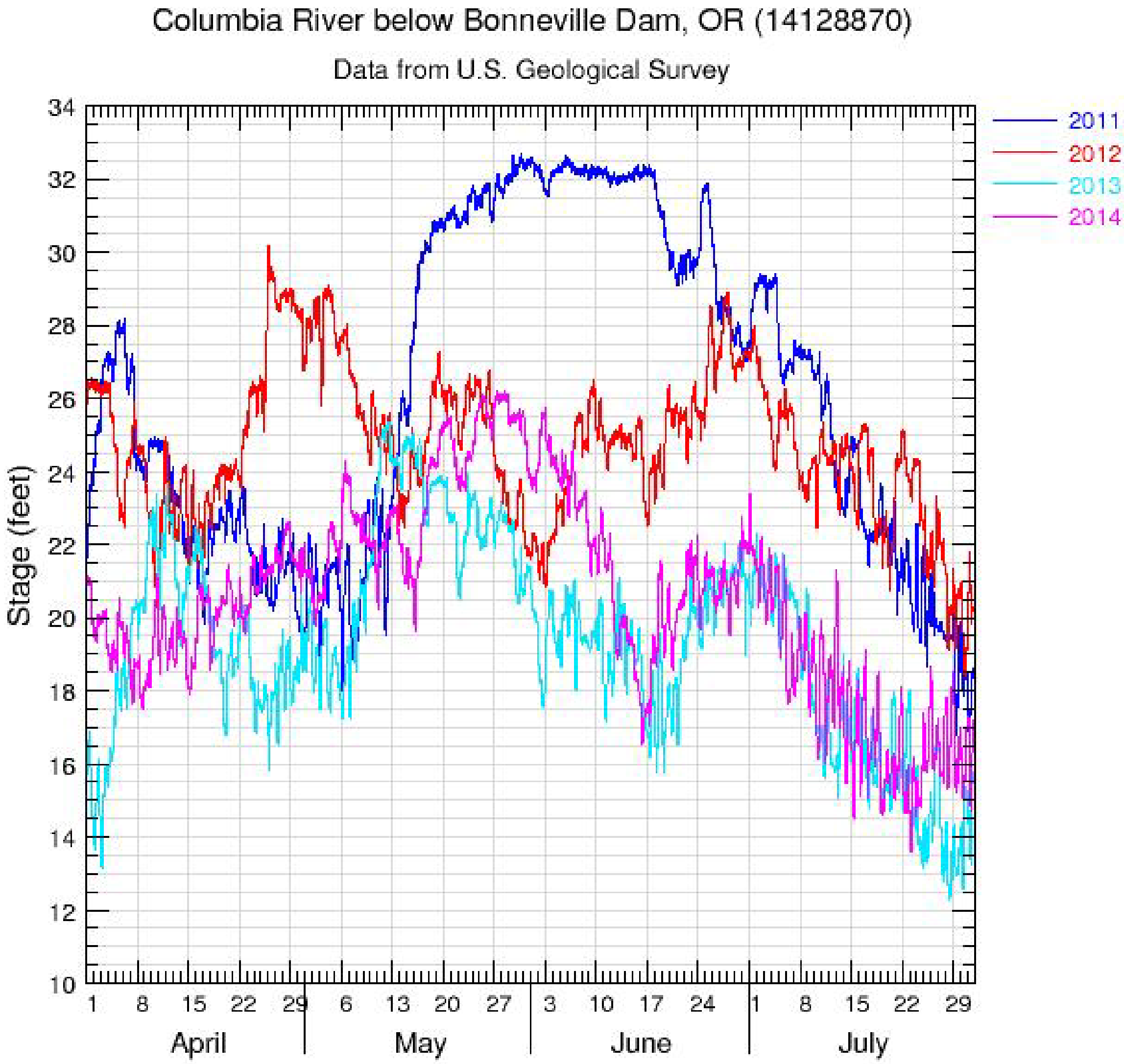


Figure 9. Gage height at the Columbia River below Bonneville Dam, Oregon USGS gaging station, April – July, 2011–2014. Data are from U.S. Geological Survey Data Grapher, accessed December 22, 2014: http://or.water.usgs.gov/cgi-bin/grapher/graph_by_yr_setup.pl?basin_id=columbia&site_id=14128870#step2.

4.2.1.1 *Temperature*

Water temperature at the four trends sites in 2014 (Franz Lake, Campbell Slough, Whites Island, and Welch Island) increased steadily during the April – July monitoring season (Figure 10). All sites exceeded the Washington State weekly maximum water temperature threshold of 17.5°C by early June (Table 15). Salmonids were caught at sites even during periods when the weekly maximum temperature threshold was exceeded, except on June 10 at Campbell Slough. Daily variability in water temperature fluctuated on approximately 1–2 week cycles and showed different patterns among the sites. Daily water temperature was most consistent throughout the season at Welch Island, the most downstream site, and Franz Lake, the most upstream site, showed the most seasonal variation in water temperature. Franz Lake water temperature variability was negatively associated with Columbia River flows below Bonneville Dam; water temperature varied more with lower river discharge and was more stable during the freshet period. During the most variable periods, Franz Lake water temperatures fluctuated up to approximately 11°C per day, in contrast with Campbell Slough and Whites Island (7-8°C per day), and Welch Island (approximately 5°C per day).

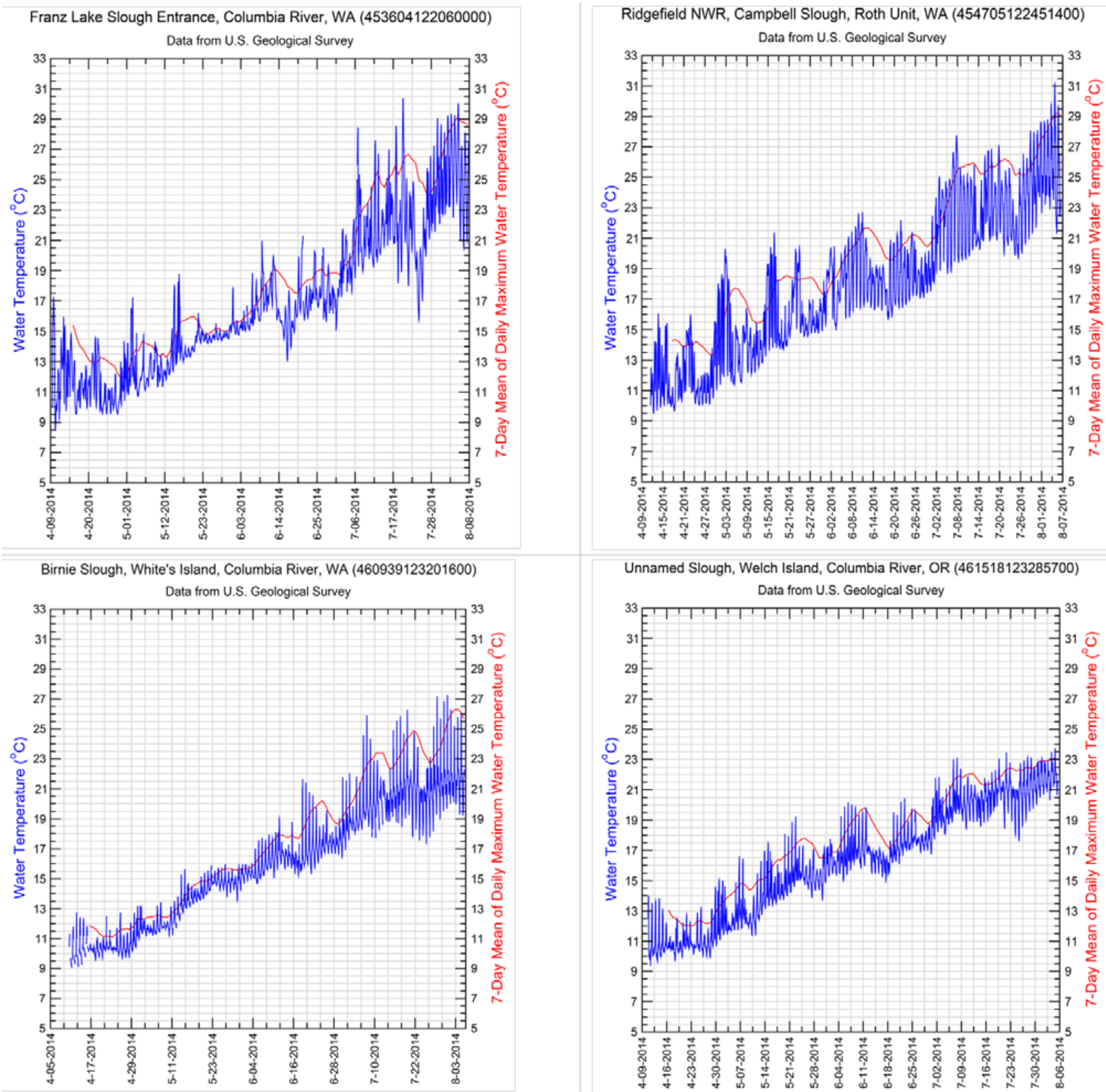


Figure 10. Continuous water temperature (blue) and average weekly maximum water temperature (red) at the four trends sites monitored for water quality in 2014.

Table 15. Dates during the 2014 monitoring period when Washington State weekly average maximum temperature threshold of 17.5°C was exceeded. Bolded dates indicate periods during which a fish sampling event occurred and salmonids were present. Italicized dates indicate periods during which a fish sampling event occurred and salmonids were not present.

Site	% of days exceeding threshold	Dates exceeding threshold*	Number of monitoring days
Franz Lake	59%	June 9–16 June 20–August 8	116
Campbell Slough	74%	May 5–**6**, 16–29 ***June 1–August 6***	112
Whites Island	51%	**June 10–August 5**	112
Welch Island	56%	May 23–26 **June 5–16** **June 20–August 5**	112

*Franz Lake was not sampled for fish May – July, 2014 due to high flow conditions.

4.2.1.2 *Dissolved Oxygen*

Dissolved oxygen patterns decreased overall throughout the 2014 monitoring period at Franz Lake, Whites Island, and Welch Island (Figure 11). Dissolved oxygen concentrations in Campbell Slough trended downward in April–May, but had a more consistent trend in June and July. The daily variability in dissolved oxygen typically increased later during the monitoring period. The daily minimum dissolved oxygen concentration was less than the 8.0 milligram per liter (mg/L) Washington State threshold at all sites during part of the monitoring period, particularly during and after June (Table 16). Salmonids were present during each month's fish sampling events at Whites Island and Welch Island, even in June and July when the daily minimum concentration was less than the 8.0 mg/L threshold.

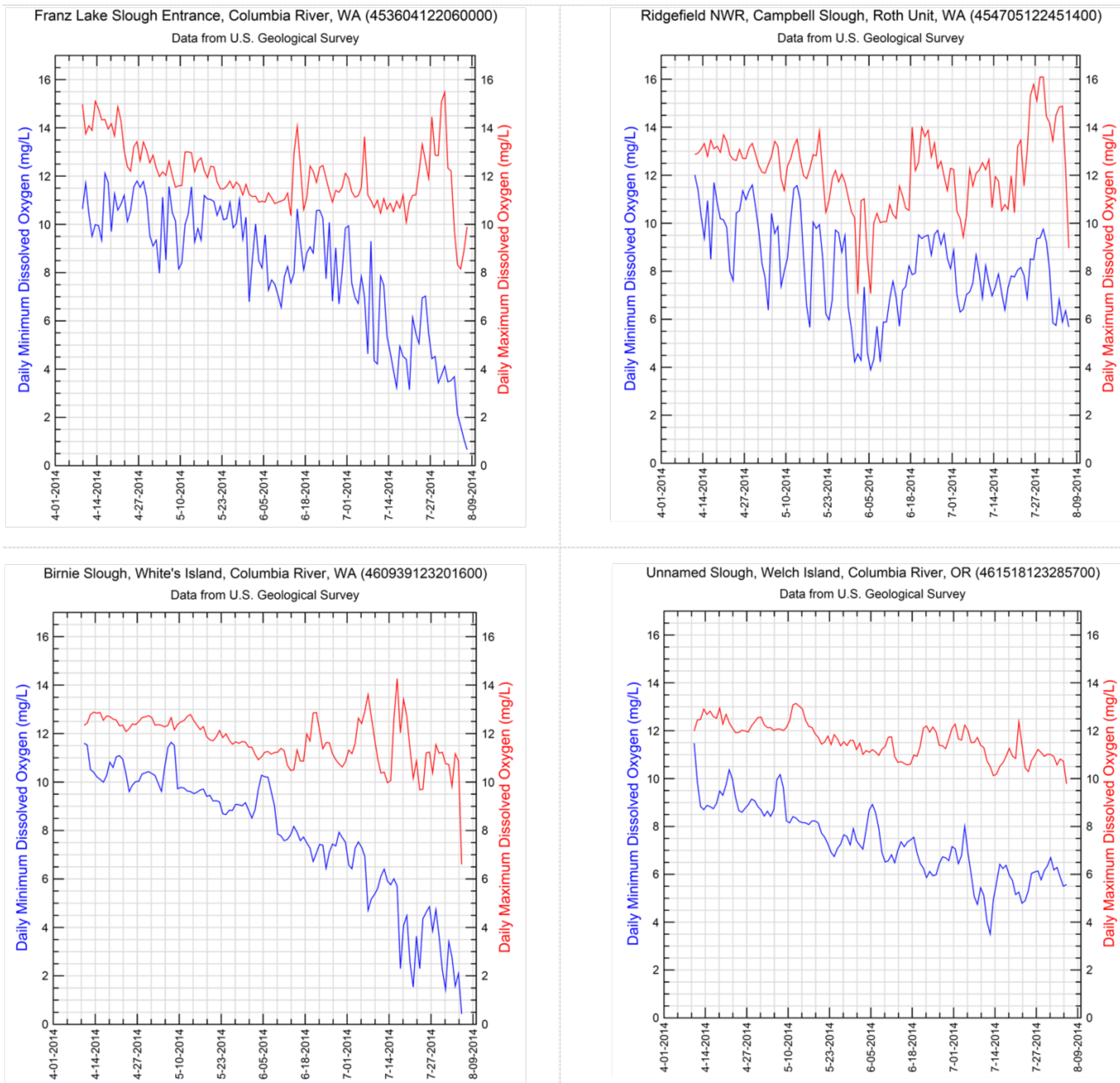


Figure 11. Daily minimum (blue) and maximum (red) dissolved oxygen (mg/L, milligrams per liter) concentrations monitored at the four trends sites in 2014.

Table 16. Dates during the 2014 monitoring period when Washington State daily minimum dissolved oxygen concentration threshold of 8.0 mg/L was not met. Bolded dates indicate periods during which a fish sampling event occurred and salmonids were present. Italicized dates indicate periods during which a fish sampling event occurred and salmonids were not present.

Site	% of days below threshold	Dates below threshold*	number of monitoring days
Franz Lake	39%	May 31 June 6–10 June 13, 24, 26, 28 July 2–7 July 9–August 7	121
Campbell Slough	55%	April 23 May 3, 4, 8, 16–17, 22–24 *May 29–June 16* June 18–19 July 2–7, 9, 10, 12–20, 23–24 August 1–6	118
Whites Island	57%	**June 9**–13 **June 15–August 5**	118
Welch Island	74%	May 20–June 3 **June 7–July3** **July 5–August 5**	118

*Franz Lake was not sampled for fish May – July, 2014 due to high flow conditions.

4.2.1.3 ***pH***

pH fluctuated, but overall showed fairly consistent trends during the 2014 monitoring period at most trends sites (Figure 12). The daily minima were never less than the minimum threshold of 6.5 during the monitoring period. The maximum threshold of 8.5 was exceeded at all the sites, mostly during later months of the monitoring period (Table 17). Salmonids were caught during the monthly fish sampling events even on days when pH exceeded the maximum threshold at Campbell Slough, Whites Island, and Welch Island.

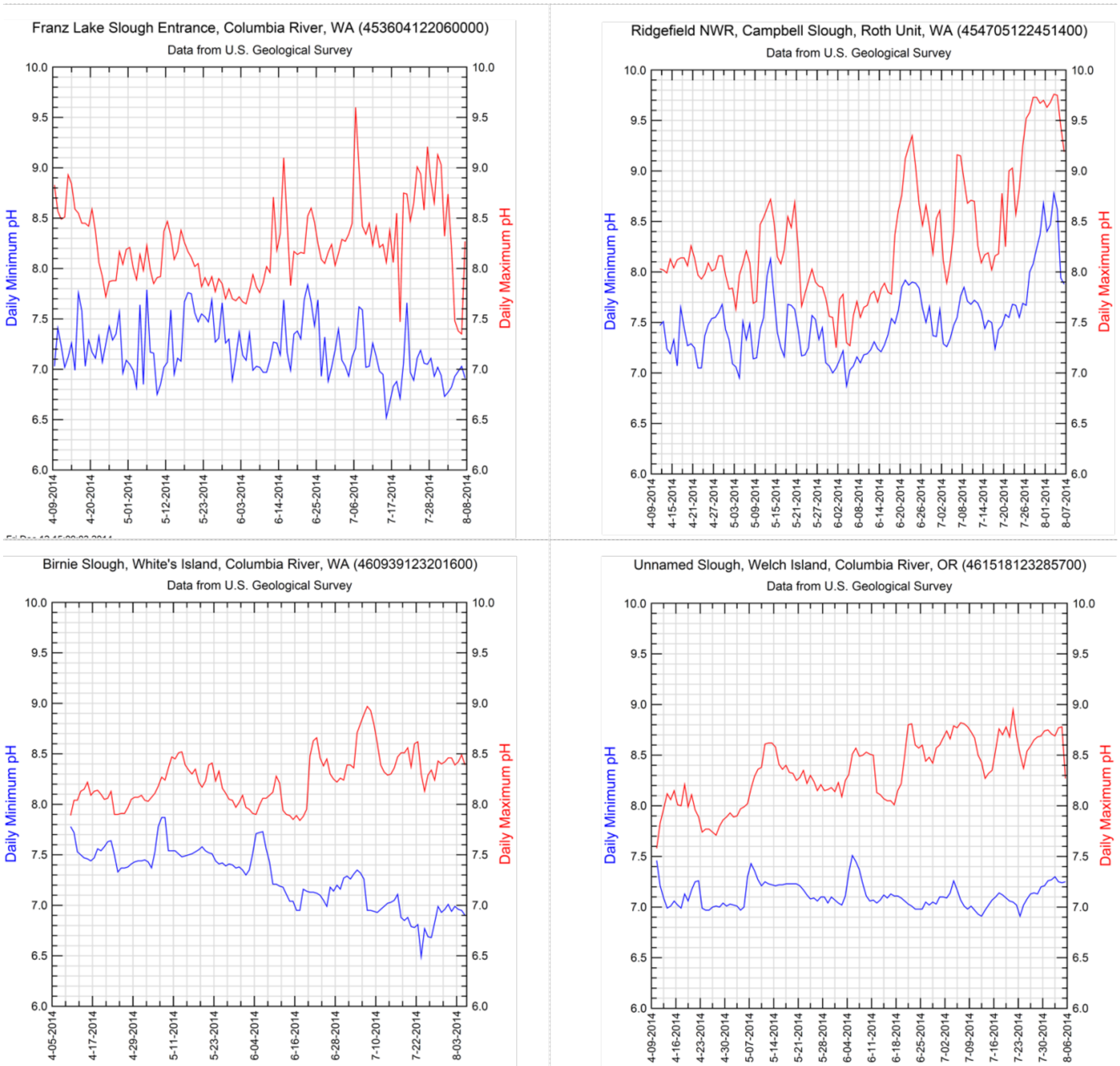


Figure 12. Daily minimum (blue) and maximum (red) pH monitored at the four trends sites in 2014. Data are in standard units.

Table 17. Dates during the 2014 monitoring period when Washington State maximum pH threshold 8.5 standard units was exceeded. Bolded dates indicate periods during which a fish sampling event occurred and salmonids were present. Italicized dates indicate periods during which a fish sampling event occurred and salmonids were not present.

Site	% of days exceeding threshold	Dates exceeding threshold*	number of monitoring days
Franz Lake	23%	April 9, 10, 13–16, 20 June 12, 15, 23 July 6, 7, 18, 20, 21 July 23–31 August 2	119
Campbell Slough	32%	May 12, 13, 18, 20 June 19–25, 27 July 1, **6–11,** 19 July 21–August 6	116
Whites Island	10%	June 21, 22 **July 4–10**, 19, 21–22	116
Welch Island	34%	May 11–14 June 6, 21–25 **June 29–July 10** July 16–22 July 26–August 4	116

*Franz Lake was not sampled for fish May – July, 2014 due to high flow conditions.

4.2.1.4 *Specific Conductance*

Specific conductance ranged from approximately 50 to 180 microSiemens per centimeter (µS/cm) during the 2014 monitoring period (Figure 13). Welch Island, which is the farthest site downstream and is not fed by other water sources than the Columbia River, had the least variability during the monitoring season.

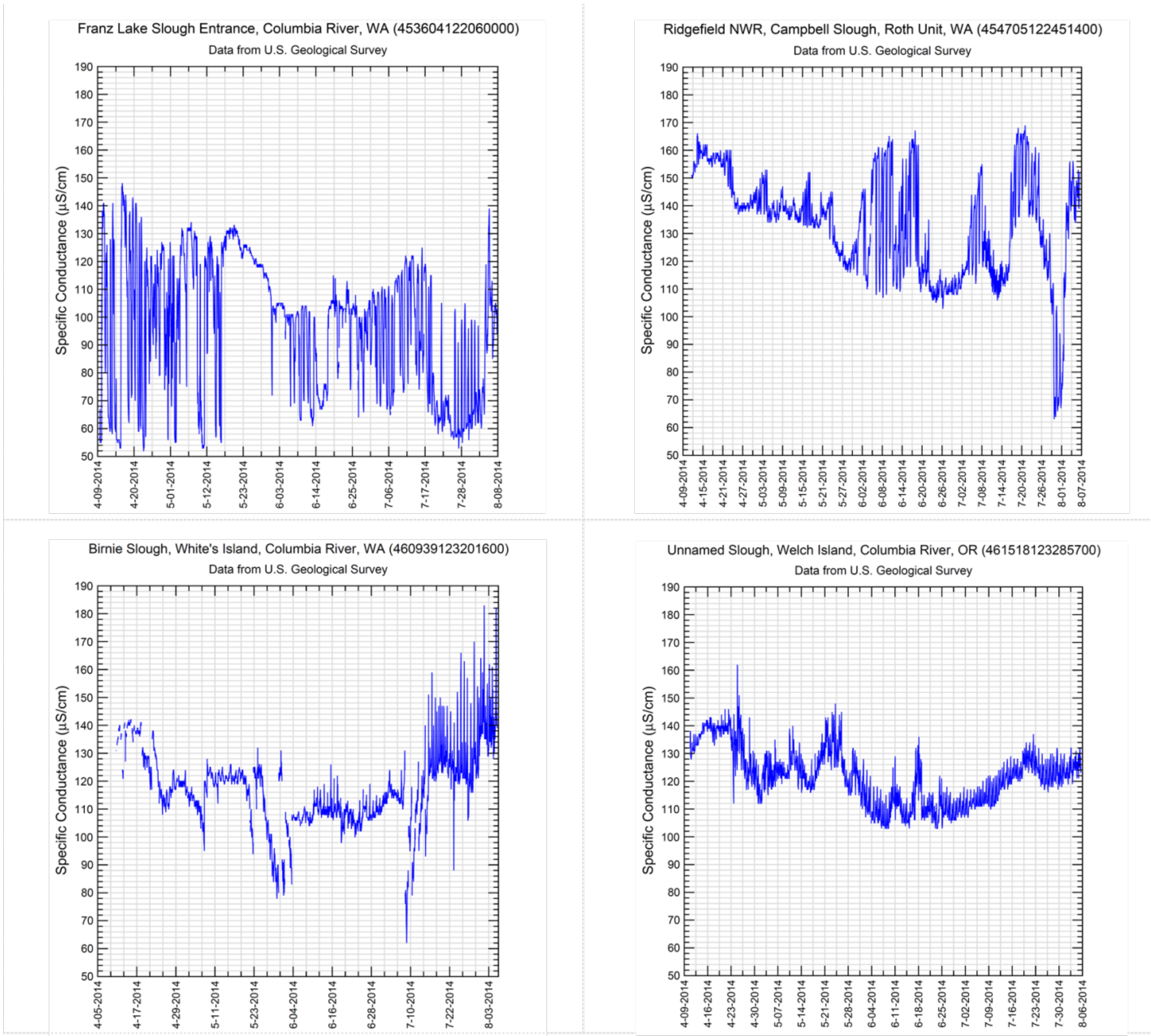


Figure 13. Continuous specific conductance (µS/cm) monitored at the four trends sites in 2014.

4.2.2 Nutrients

4.2.2.1 *Dissolved inorganic nutrients*

Dissolved nutrient concentrations included nitrate, nitrite, ammonium, ortho-phosphate, and silicic acid. Silicic acid data are not discussed here, since levels were always very high and they did not vary among sites. Dissolved nitrate concentrations varied according to the time of sampling ($p = 0.003$) and site (0.026), although a post hoc multiple comparisons test (One-Way ANOVA, Holm-Sidak test) did not reveal large enough differences between individual sites to be considered significant ($p > 0.05$ for all comparisons). Temporal differences were observed between early and late-season sampling dates, however, with early dates having significantly higher average nitrate concentrations (0.124 ± 0.115 mg L^{-1}) compared to later dates (0.034 ± 0.035 mg L^{-1} nitrate). Notably, the standard deviation of nitrate concentrations was higher earlier in the season compared to later (Figure 14). Differences among sites and dates for ammonium and ortho-phosphate concentrations were not significant ($p > 0.05$). Similarly, the standard deviations around average ammonium and ortho-phosphate concentrations were more similar over the season than they were for nitrate.

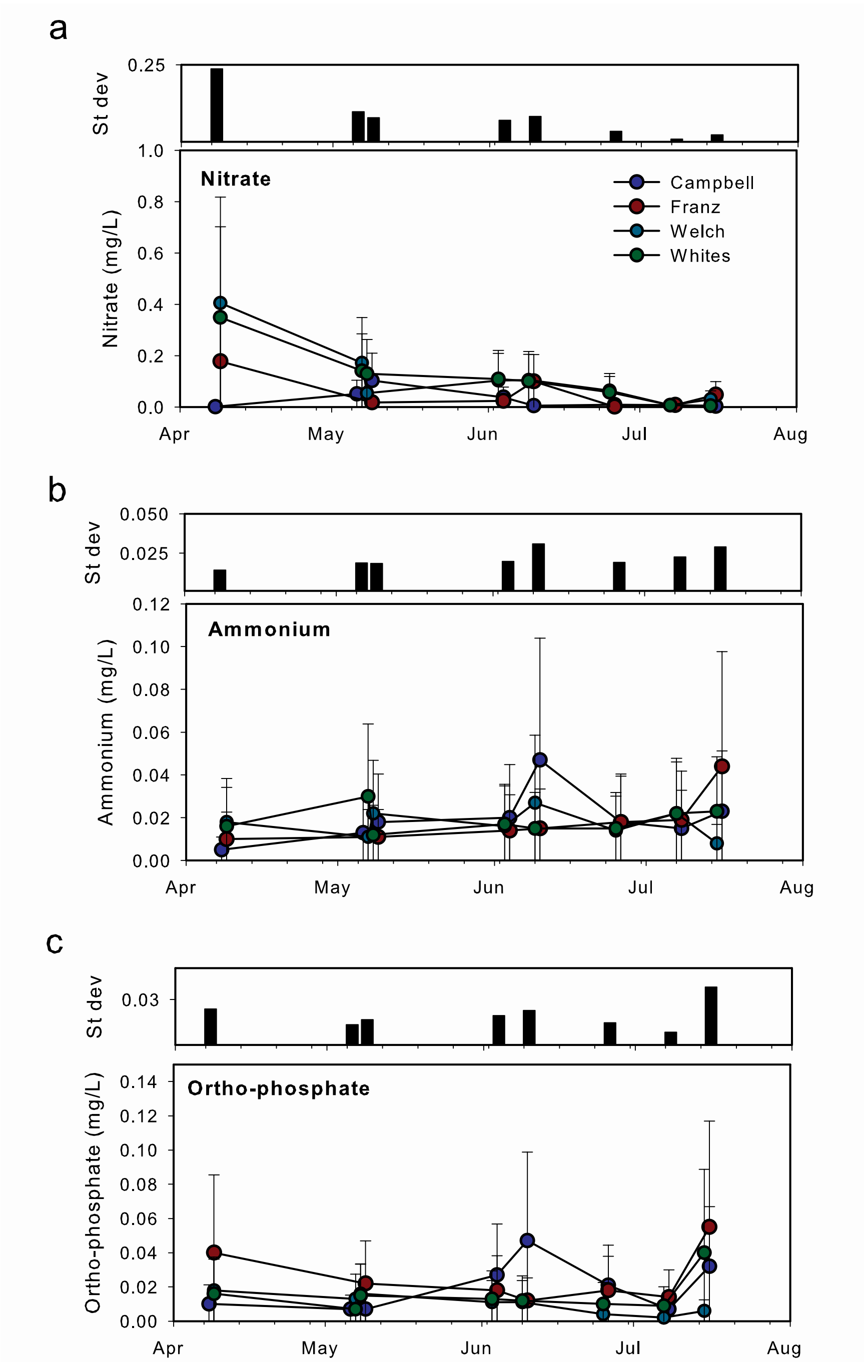


Figure 14. Concentrations of a) nitrate, b) ammonium, and c) ortho-phosphate at each of the four trends sites (Campbell Slough = blue symbols, Franz Lake = red symbols, Welch Island = cyan symbols, and Whites Island = green symbols) between April and August 2014. Standard deviations for each of the inorganic nutrient species is indicated in a panel above the corresponding line graph.

The ratio of summed dissolved inorganic nitrogen species [DIN = $\sum(NO_3^-, NO_2^-, NH_4^+)$] to ortho-phosphate (DIN:DIP) differed between sites ($p = 0.002$, Two-Way ANOVA), but not over time (Figure 15; $p = 0.215$). A Holm-Sidak post hoc multiple comparisons test revealed that there were differences in DIN:DIP values between Welch Island and Campbell Slough as well as Welch Island and Franz Lake

Slough ($p < 0.05$). DIN:DIP ratios were similar between Campbell Slough and Franz Lake Slough and values at Welch Island were similar to Whites Island. Despite the fact that there were significant differences in DIN:DIP between Whites Island and Franz Lake Slough prior to the month of July ($p = 0.049$; differences between Whites Island and Campbell Slough approached significance prior to July, $p = 0.051$), low DIN:DIP values in July at Whites Island rendered the overall temporal differences insignificant. DIN:DIP values averaged ~4-5 times higher at Welch Island compared to Campbell Slough or Franz Lake (32.4 ± 14.5 at Welch vs. 6.8 ± 9.8, 7.3 ± 7.1 at Campbell and Franz Lake Sloughs, respectively) and approximately three times higher at Whites Island (22.8 ± 17.1) compared to Campbell Slough or Franz Lake Slough. The low DIN:DIP ratios at Campbell Slough and Franz Lake Slough (i.e., <16:1) are indicative of nitrogen limitation of phytoplankton primary production.

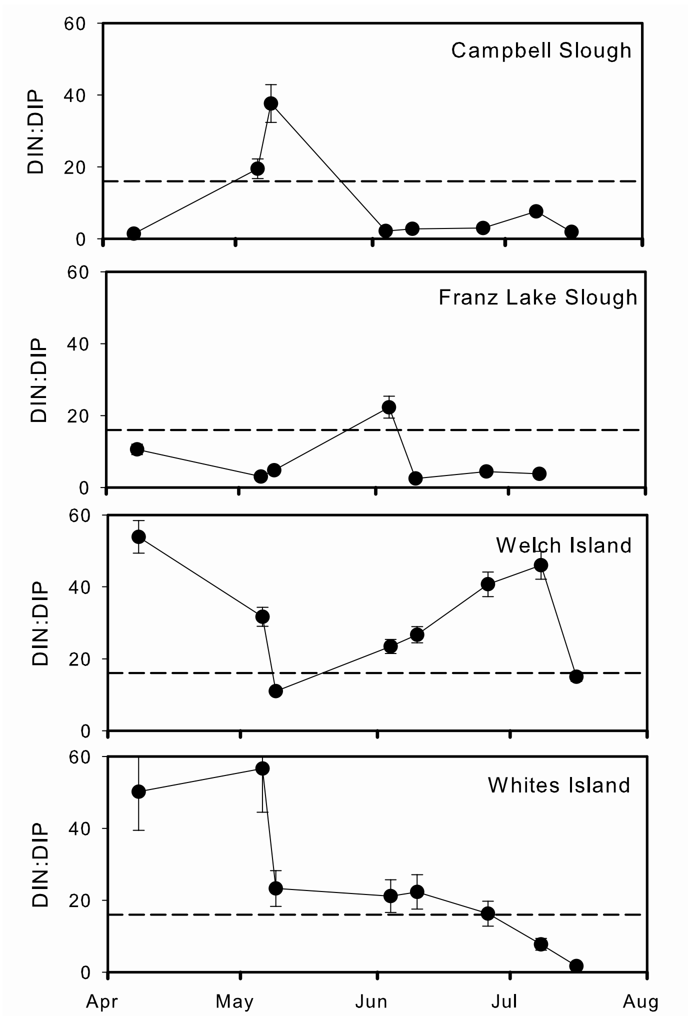


Figure 15. Dissolved inorganic nitrogen:phosphorus ratios at Campbell Slough, Franz Lake Slough, Welch Island, and Whites Island in 2014. DIN = dissolved inorganic nitrogen (NO_3^- + NO_2^-, + NH_4^+). The dotted line indicates an N:P value of 16:1. Levels higher than this indicate a tendency toward phosphorus limitation of phytoplankton growth, whereas a number below this level indicates nitrogen limitation of phytoplankton growth.

4.2.2.2 *Dissolved organic nitrogen and phosphorus*

Concentrations of dissolved nitrogen and phosphorus accounted for by the organic fraction are shown in Figure 16. Levels of dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) were much lower than the inorganic fractions described above. DON concentrations were relatively constant at Welch Island and Franz Lake Slough, with higher values observed at the latter site. In contrast, both Whites Island and Campbell Slough showed peaks in DON of similar magnitude in early June. The grey line in Figure 16 indicates the limit of detection for the phosphorus measurement, showing that most of the time DOP was present at levels below the limit of detection only at Franz Lake were concentrations high enough to quantify.

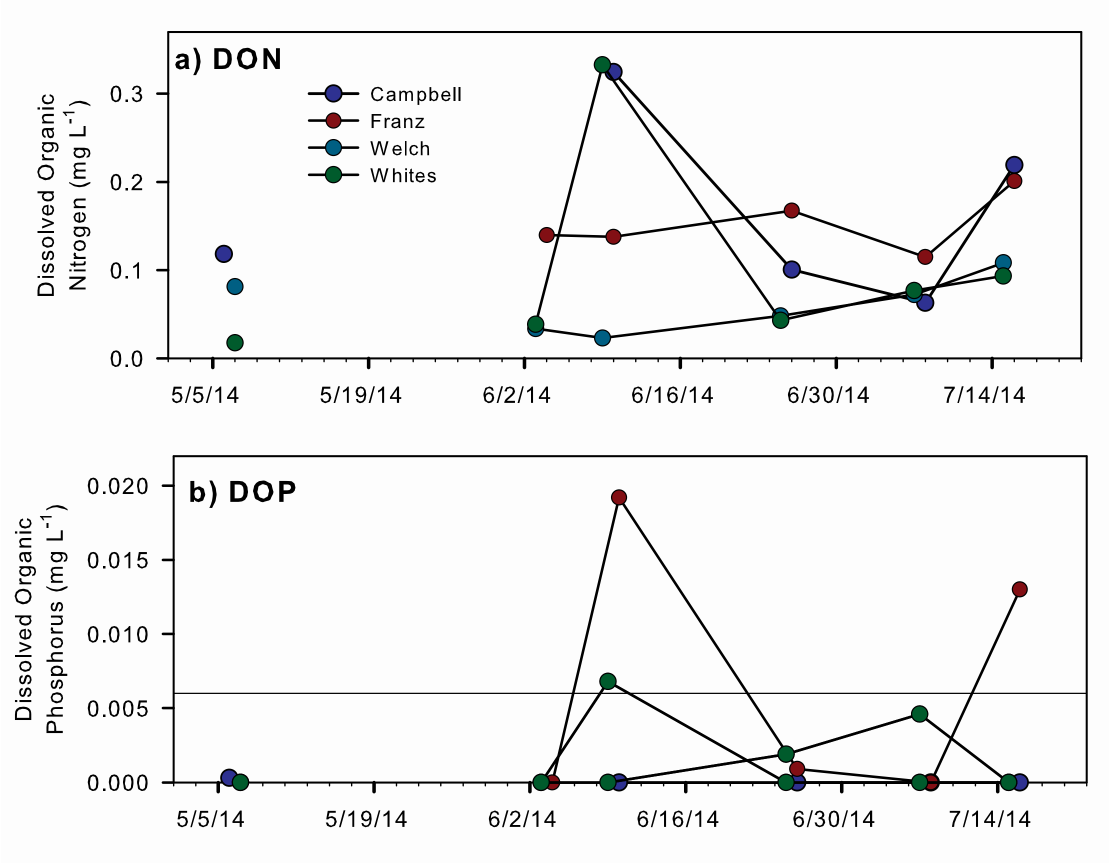


Figure 16. (a) Dissolved organic nitrogen (DON) and (b) dissolved organic phosphorus (DOP) at four trends sites (Campbell Slough, Franz Lake Slough, Welch Island, and Whites Island) in 2014.

4.3 Habitat Structure

4.3.1 Hydrology

Hydrology was monitored throughout the year at all EMP trends sites. Hydrographs from all the years in which water surface elevation was sampled at the trends sites, including the 2014 water year, are provided in Appendix A.

Although water surface elevation at Ilwaco Slough (rkm 6) is minimally affected by the spring freshet, it 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. The trends site at Secret River (rkm 37) is also affected by winter storm events and not by the spring freshet. Water surface elevations at this site are slightly higher and the tide range is greater than at Ilwaco Slough, partially due to the lower elevation of the tidal channel where the sensor is located. The low elevation marsh at the site is infrequently exposed and conversely, the high elevation marsh is infrequently inundated. The Welch Island site (rkm 53) is predominantly tidal, however slightly elevated water surface elevation was detectable during the prolonged spring freshet in both 2012 and 2014. Winter storms also drive higher water levels at Welch Island. The hydrologic patterns at the Whites Island site (rkm 72) exemplify the mix of hydrologic drivers in the lower river. The tidal range is greater than 2 m in most months, while elevated water levels also occur during winter storm events and the spring freshet.

The Cunningham Lake and Campbell Slough sites (rkm 145 and149, respectively) have similar hydrologic patterns except that Cunningham Lake has a slightly greater tidal range and slightly lower water surface elevation during the freshet. The Campbell Slough water surface elevation does not get as low as the Cunningham Lake site due to a weir located at the mouth of the slough, which limits drainage. The primary hydrologic driver at both sites is the spring freshet, although in 2013 and 2014 winter storms also increased the water surface elevation 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 water surface elevation 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.

The site at Franz Lake Slough does not exhibit a discernable tidal signal and low water was maintained at the site by a beaver dam in the fall that washed out sometime during in the winter months. The winter and spring high water surface elevations are both discernable; however, the spring levels were considerably higher than in the winter at this site. In 2013, the site was inundated for approximately one month in the winter and for approximately three months in the spring. In 2014, the site was inundated above the marsh surface beginning in February and continuing through June.

It is evident that the frequency of inundation at each site is dependent on the elevation, position along the tidal and riverine gradient, and seasonal and annual hydrologic conditions. The frequency of inundation at the average elevation of the sites in 2014 is shown in Figure 17. In the lower river, the percent time that high marshes are inundated is greater over the whole year ranging from 22 and 36 percent than it is 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 2014 due to its position at the lower end of the tidal wetland elevation range in the lower river. In 2014, the three up-river sites had growing season inundation frequencies of approximately 50 percent. The lower river high-marsh sites had frequencies ranging from 17 to 25 percent during the growing season.

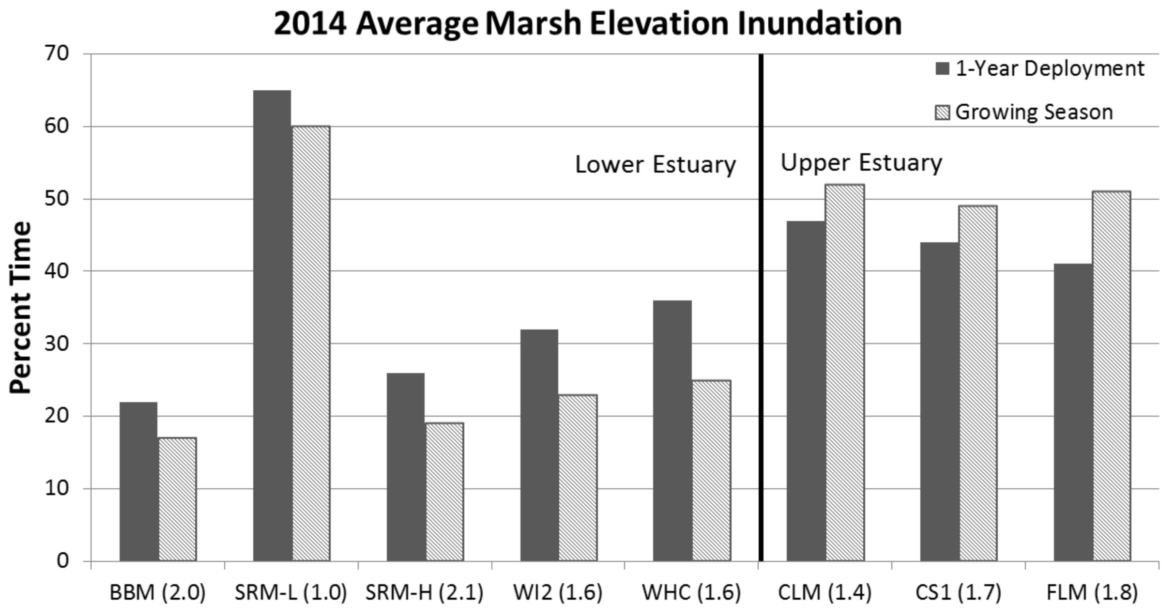


Figure 17. Inundation frequency at the seven trends sites in 2014; one-year deployment is from July 2013 to July 2014 and the growing season is from April-October. Site codes are defined in Table 1. Sites are ordered from left to right starting at the mouth (BBM = Ilwaco Slough, SRM-L = Secret River low marsh, SRM-H = Secret River high marsh, WI2 = Welch Island, WHC = Whites Island, CLM = Cunningham Lake, CS1 = Campbell Slough, FLM = Franz Lake). 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 general, cumulative inundation increases with increasing distance from the river mouth, with the highest inundation at the Franz Lake site (FLM; Figure 18). Inundation is consistently higher at Secret River than the three other lower estuary sites. The reason for this is not certain, but could be related to consistently higher tides at Secret River than at Welch Island and Whites Island, and the fact that the site is more affected by tributary run-off (i.e., Grays River) than Ilwaco Slough.

Inter-annual variation in inundation patterns is much greater at the upper estuary sites (Figure 19), where seasonal flooding can result in multiple 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 was slightly discernable in the SEV at the average marsh elevation, whereas the up-estuary sites have large differences in the SEV between years. At all sites measured, the SEV in 2014 was greater than it was in 2010 or 2013; similar to 2009 and 2010 values; and less than that measured in 2011 and 2012.

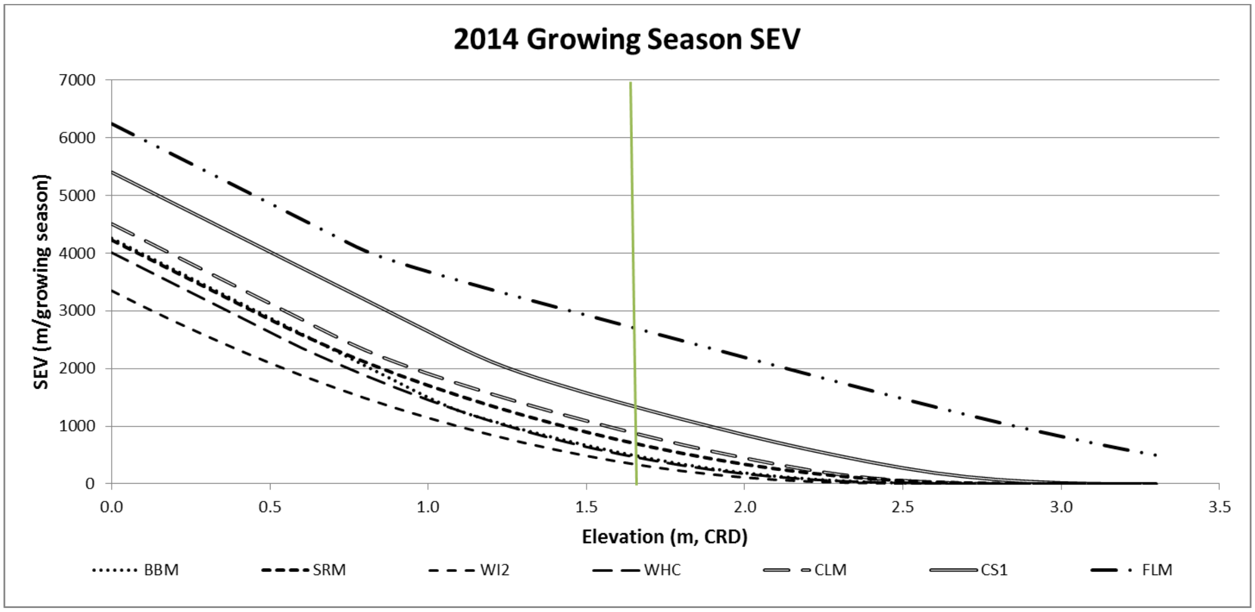


Figure 18. Growing season sum exceedance values (SEVs) for the 2014 trends monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the lower river; the vertical line represents the average elevation of all the trends sites. BBM = Ilwaco Slough, 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.

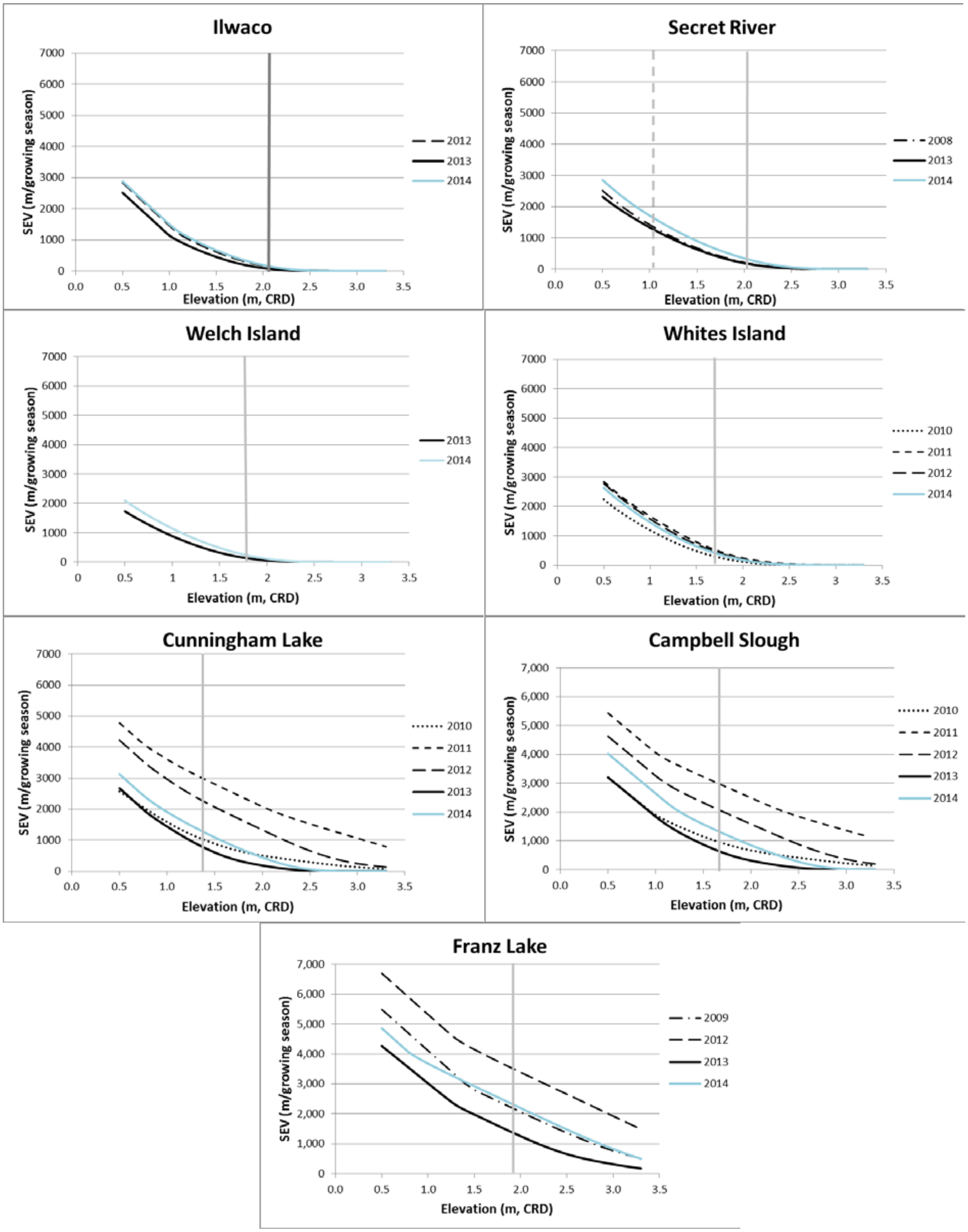


Figure 19. Annual growing season sum exceedance values (SEVs) for the trend monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the lower river. The vertical lines represent the average elevation at each site, with the elevation of the Secret River low marsh represented by an additional dashed line.

4.3.2 Sediment Accretion Rates

Annual sediment accretion rates for 2014 ranged from -1.6 (erosion) to 2.3 cm/year, with most values falling between 0.3 and 1.0 cm/year (Table 18). These rates are consistent with those found at a larger number of reference sites in the lower river as documented by Borde et al. (2012). Within this range, rates are variable between sites and between years (Table 18). The site with the lowest inter-annual variability is Welch Island (SD ±0.17) and the highest is Franz Lake (SD ±1.55), which also had the highest accretion rates measured of all trends sites. In 2011 and 2012, additional stake sets were deployed at varied elevations and distances from the primary tidal channel at Secret River and Whites Island in order to explore the effect of those variables on sediment accretion rates. Results varied depending on the site, with the lowest set (SRM-L) eroding at a rate of -1.8 cm per year and the sets closest to the channel (SRM-C and WHC-M) having the highest rates: 1.3 and 1.7 cm per year.

Table 18. Sediment accretion rates at the trends sites between 2008 and 2013. BBM = Ilwaco Slough, 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. SRM-C is a set of sediment accretion stakes near the channel at the Secret River high marsh site.

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
Average	**0.4**	**-1.8**	**0.9**	**1.3**	**0.7**	**1.7**	**0.3**	**1.3**	**0.9**	**1.4**
Standard Deviation	0.3	0.2	0.4	1.0	0.2	0.8	0.8	0.5	0.7	1.6

[1] ND No data.

4.3.3 Salinity

Salinity was measured at Ilwaco Slough between July 2011 and February 2015 (Figure 20). The range was between 0.1 and 30.7 parts per thousand (ppt). Only 10 records occurred above 25.5 ppt, and 30 ppt was exceeded for just one hour during January 2014. Just over 67 percent 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.

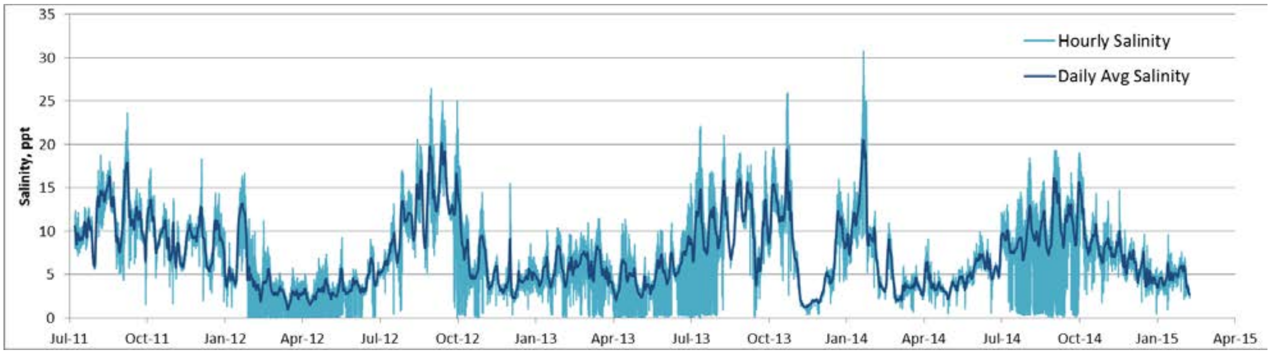


Figure 20. Hourly and daily average salinity at Ilwaco Slough between July 2011 and February 2015.

4.3.4 Vegetation Species Assemblage

A list of all species and percent cover values observed at the trends sites in 2014 is provided in Appendix C (Table C-1). The cover and elevation range of each species are also plotted by site and provided in Appendix C (Figure C-1).

Vegetation species assemblages vary temporally and spatially at the trends sites monitored as part of the EMP. Based on the cover and species richness (Table 19), the vegetation assemblages observed at the 2014 monitoring sites can be broadly grouped into categories associated with the emergent marsh (EM) vegetation zones (Figure 1) as follows:

Zone 1	low species richness/high cover
Zone 2	high species richness/high cover
Zone 3	no data collected in 2014
Zone 4 and 5	moderate species richness/moderate cover

In the three lowest river sites, native vegetative cover was higher than non-native cover (Table 19), with native species cover dominated by Lyngby's sedge (*Carex lyngbyei*). Total cover at these sites is over 100 percent except in the Secret River low marsh where high inundation (60% of the time) limits cover. The Zone 2 sites (Secret River, Welch Island, and Whites Island) have the highest number of observed species, ranging from 46 to 50 species (the latter is for the Secret River low and high marsh plots combined). The dominant wetland species and associated overall average cover for all years at the trends sites are provided in Table 20. At Whites Island and the Zone 4 sites, non-native cover, predominantly reed canarygrass (*Phalaris arundinacea*; Table 21), was nearly equal to or greater than native cover. Non-native cover was very low at the Zone 5 site, where the native water smartweed (*Polygonum amphibium*) and other indigenous species have out competed *P. arundinacea* the past three years (Figure 21).

Table 19. Species richness and areal cover of native and non-native species at the 2014 monitoring sites.

Site	Rkm	# Native Species	Native Species % Cover[1]	# Non-native Species	Non-native Species % Cover	Total # Species	Total % Cover
Ilwaco Slough	6	14	101.5	3	9.3	17	110.8
Secret River - High	37	24	81.9	9	29.1	33	110.9
Secret River - Low	37	26	70.0	6	7.6	32	77.6
Welch Island	53	34	94.1	12	16.0	46	110.1
Whites Island	72	28	32.7	17	61.0	45	93.7
Campbell Slough	145	15	24.6	10	31.8	25	56.4
Cunningham Lake	149	10	26.4	3	24.6	13	50.9
Franz Lake	221	18	62.2	5	9.1	23	71.3

[1]Cover 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 2014 results indicate that Lyngby's sedge (*C. lyngbyei*) was present at a higher percent cover at the trends sites than reed canarygrass (*P. arundinacea*), while previous studies have documented that *P. arundinacea* has the highest percent cover of emergent wetland species in the lower river (Sagar et. al., 2013). The reasons for this difference are 1) three of the seven study sites are located in the lower portion of the lower river, below rkm 72, where salinity and the tidally dominated hydrology reduce the probability of *P. arundinacea* occurrence (Borde et al. 2012; Sagar et al. 2013); and 2) three trends sites occur in the upper estuary, at or above rkm 145, where the effects of high water and other disturbances have reduced reed canarygrass cover for the past four years (Table 21 and Figure 21).

Table 20. Dominant vegetation species found at the seven trends sites sampled for habitat structure. Average percent cover throughout the study area was calculated by taking the average between years at each site then averaging all sites.

Species Code	Scientific Name	Common Name	Wetland Status	Category	Native	Avg. % Cover (SD)
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	Sedge	yes	20.05 (25.38)
PHAR	*Phalaris arundinacea*	Reed canarygrass	FACW	Grass	no	19.63 (16.4)
ELPA	*Eleocharis palustris*	Common spikerush	OBL	Sedge	yes	6.97 (8.59)
SALA	*Sagittaria latifolia*	Wapato	OBL	Herb	yes	3.34 (2.82)
OESA	*Oenanthe sarmentosa*	Water parsley	OBL	Herb	yes	3.29 (5.62)
POAM	*Polygonum amphibium*	Water ladysthumb, Water smartweed	OBL	Herb	yes	2.03 (5.71)

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 percent since

2012. The highest coverage has consistently been observed at the Whites Island site, where cover has been greater than 40 percent since the site was first monitored in 2009. Moderate cover between 20 and 35 percent was observed at the Secret River site since 2012; an increase from the 10 percent 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 percent, with lowest cover observed during years of high inundation and episodes of disturbance from cows. The Franz Lake site had consistently moderate cover of 33 to 34 percent in 2008 and 2009 followed by a decrease, starting in 2011 when high inundation favored the growth of a competitive native species, water smartweed (*P. amphibium*), which has persisted until 2014 (Figure 21).

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

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

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 SAV are reported with the emergent cover for all of the sites (Appendix C, Table C-1). Cover data for SAV are also provided separately for the channels located at 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. Horned pondweed (*Zannichellia palustris*) is the only SAV species that occurs at Ilwaco Slough, and is found in the tidal channel and in ponded areas of the marsh. At the Secret River low marsh site, Canada waterweed (*Elodea canadensis*) accounted for 17 percent of the cover in 2014, occurring throughout the low marsh in small depressions that hold water at low tide, though cover has decreased from 35 and 24 percent the previous two years (Figure 21). At all other sites, SAV species account for less than five percent of the cover in the emergent marsh area. In the channels of the Secret River, Welch Island, and Whites Island sites, SAV cover was 49, 85, and 23 percent at the three sites, respectively and was dominated by the native species *Elodea* spp. and *Potamogeton richardsonii*.

Trends in the dominant vegetation cover at the sites are depicted in Figure 21. A few notable changes in the species cover at the lower estuary sites include the disappearance of saltgrass (*Distichlis spicata*) from Ilwaco Slough during the past two years, the reduction of *C. lyngbyei* in 2014 at the Welch Island site, and the gradual reduction of *C. lyngbyei* at the Whites Island site since 2009. Variability in the cover of the dominant species is evident at sites in the upper reaches and at Secret River low marsh. At two of the upper estuary sites, cover greater than 80 percent occurred in the relatively low inundation years – 2005, 2007, 2009, and 2010. Cover at Campbell Slough in 2007 was an exception when cows were periodically present at the site. Cover in other years was likely affected by a combination of inundation and other disturbances. Inundation during the high water years of 2011 and 2012 strongly affected the cover at the upper estuary sites. At the Franz Lake site, a shift in vegetation dominance occurred during this time which continued until 2014.

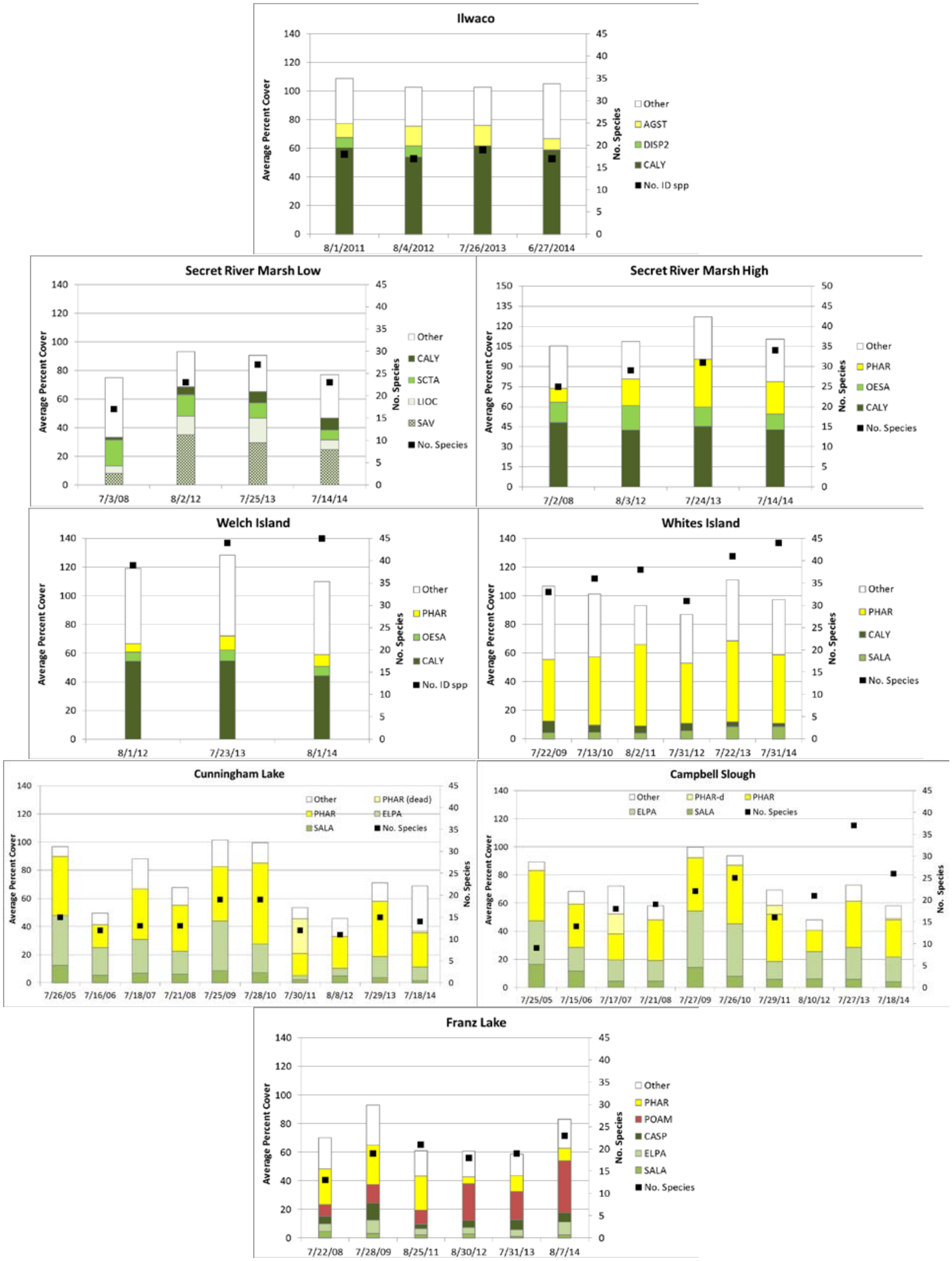


Figure 21. Average percent cover and number of identified species at the trends sites for all years monitored. Sites are presented in the order in which they occur in the River, starting at near the mouth.

A 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 22 and Table 22) with average similarity greater than 80 percent for the three high marsh sites below rkm 60. However, similarity between years was lower and more variable at the Secret River low marsh site. Average similarity between years significantly decreases with increasing distance from the river mouth (Figure 23; regression $p < 0.001$). The lowest average similarity was at the Cunningham Lake site (CLM) with 67.6 percent similarity (n=45) and the lowest single comparison was 46.8 percent also at CLM between 2005 and 2011. 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 23; regression $p = 0.001$).

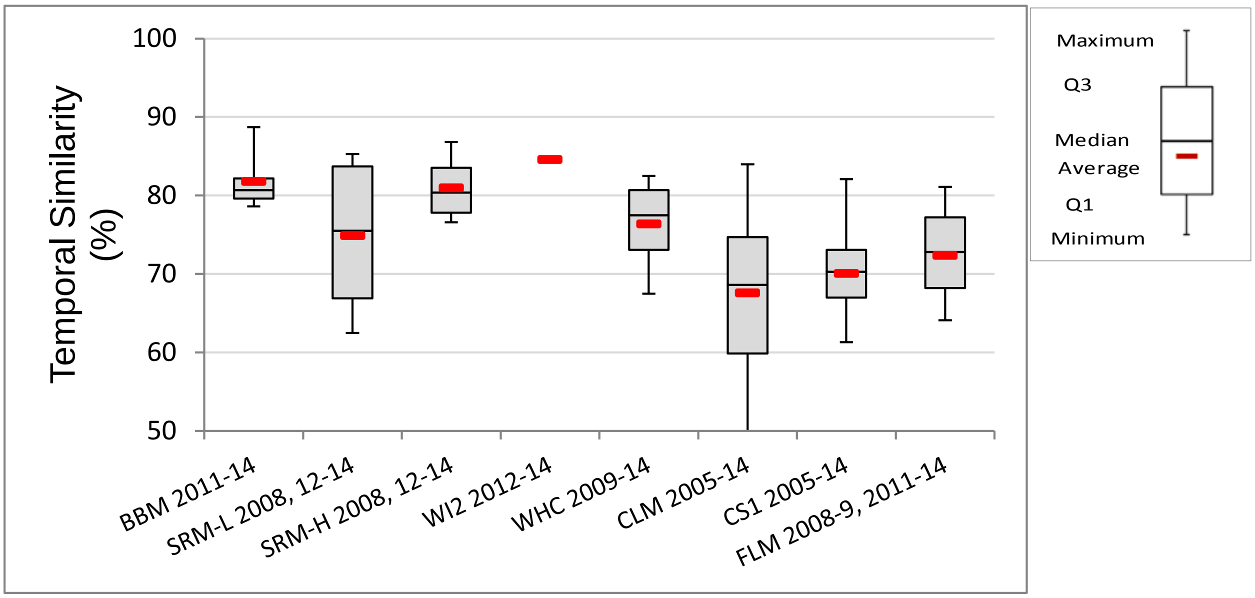


Figure 22. Pairwise similarity (%) of vegetation percent cover over time for each trend site. BBM = Ilwaco Slough, SRM-L = Secret River low marsh, SRM-H = Secret River high marsh, WI2 = Welch Island, WHC = Whites Island, CLM = Cunningham Lake, CS1 = Campbell Slough, FLM = Franz Lake. Sites are ordered from Reach A (river mouth) to Reach H.

Table 22. Descriptive statistics of the percent site similarity of vegetative cover with itself over time. Site codes are as follows: BBM = Ilwaco Slough, 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	*n*	Mean	StDev	Minimum	Q1	Median	Q3	Maximum
Ilwaco Slough	6	81.8	3.7	78.6	79.6	80.7	82.2	88.7
Secret River - low	6	74.9	10.3	62.5	66.9	75.5	83.7	85.3
Secret River - high	6	81.0	4.1	76.6	77.8	80.4	83.5	86.8
Welch Island	3	84.6	1.4	83.1	84.0	84.9	85.3	85.8
Whites Island	15	76.4	5.0	67.5	73.1	77.5	80.7	82.5
Cunningham Lake	45	67.6	9.4	46.8	59.9	68.6	74.7	84.0
Campbell Slough	45	70.1	4.8	61.3	67.0	70.3	73.1	82.1
Franz Lake	15	72.4	5.4	64.1	68.2	72.8	77.2	81.1

[1] The number of comparisons (*n*) is based on the number of years a site was monitored; for example, CLM was monitored for 10 years and 45 year-to-year comparisons could be made.

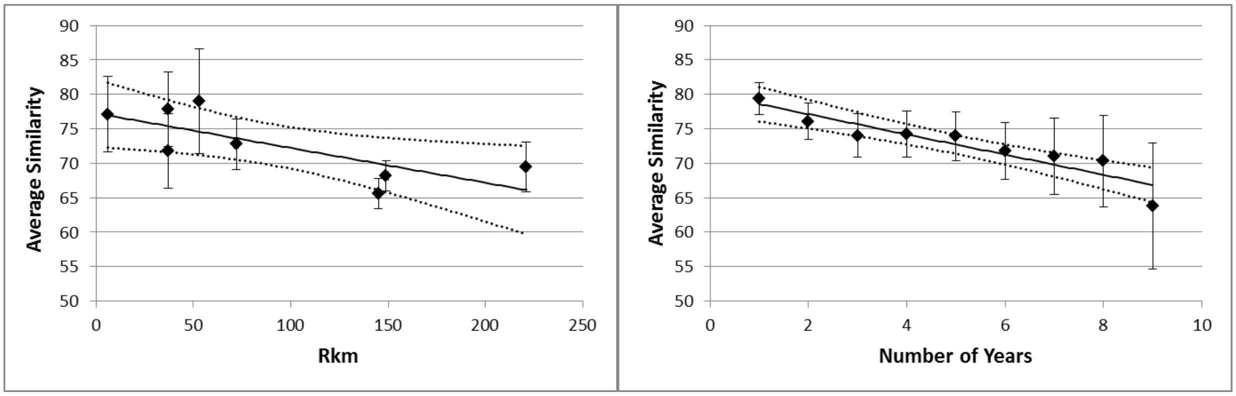


Figure 23. Average similarity of sites using the least square mean model for individual sites with the effect of the difference in years removed (left plot), and the average similarity between years with the effect of the different sites removed (right plot).

4.3.5 Channel Morphology and Inundation

Low inter-annual variability of channel morphology at the trends sites has been observed in years prior to 2014; therefore, channel cross section surveys were not conducted this year. Channel measurements from previous years are presented with the newly calculated inundation frequency results from 2014 in Table 23. The tidal channels measured at the sites are generally small, with cross sectional areas less than 10 m^2 (see Appendix B for locations of the measured channels). Five of the tidal channels surveyed are primary channels feeding directly into the Columbia River, while the channels at the Welch and Whites Island sites are secondary channels that feed into a larger tidal channel. The Secret River channel has the greatest area: close to 20 m^2 for most of its length. The channels vary in width from 1.3 m to 50.1 m; most becoming narrower with increasing elevation, with the exception of the Ilwaco Slough and Whites Island channels, which are 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.

Inundation frequency in the tidal channels in 2014 was calculated for the entire deployment year (August 2013 to August 2014) and during the peak juvenile salmon migration period (March to July 2014), for the following two conditions:

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

In the lower reaches, the inundation frequency was similar between the two time periods due to the reduced effect of the spring freshet in this part of the river (Table 23). The Secret River channel was inundated more frequently than the Ilwaco Slough or Welch Island channels due to the greater depth and lower bank elevation; the channel had greater than 50 cm of water at least 75 percent of the time and the bank was inundated to 10 cm about 60 percent of the time at the mouth and 20 to 50 percent farther up the channel. The other channels at lower reaches sites were inundated less frequently, with the thalweg inundation generally between 45 and 70 percent and the bank inundation between 15 and 30 percent, except at the uppermost cross section at the Ilwaco Slough site. The higher elevation Whites Island site had similar inundation frequencies to the Secret River site due to greater inundation during the spring freshet. Overall, the 2014 channel inundation frequencies in the lower reaches are approximately 5 percent higher than those observed in 2013.

Sites in the upper reaches had very high channel inundation frequencies during the peak fish migration period (>97 percent) compared to the year as a whole (48 to 85 percent). During the freshet in 2014, the banks were also inundated at least 96 percent of the time, except for the high bank at the mouth of the Franz Lake channel. These values are similar to the inundation frequencies observed in the upper reaches during the 2012 spring migration period, when the channels and the banks were inundated nearly 100 percent of the time, and are considerably higher than the inundation frequency during the same period in 2013.

Table 23. Physical channel metrics and inundation frequencies measured at each site. The channel mouth (indicated with an *) was measured in 2013; the year of full channel measurement is provided in parentheses after the site code. Inundation frequencies are calculated for one year (August 2013 - August 2014) and compared to results for five months between 1 March and 31 July 2014 (the peak juvenile Chinook salmon migration period). Cross sections are numbered starting at the mouth.

								Inundation			
				Physical Metrics				Year		March-July	
Site (year)	**Cross Section**	**Thalweg Elevation (m, CRD)**	**Bank Elevation (m, CRD)**	**Channel Depth (m)**	**Cross Section Area (m^2)**	**Channel Width (m)**	**Width:Depth Ratio**	**% Time WL > Thalweg + 50cm**	**% Time WL > Bank + 10cm**	**% Time WL > Thalweg + 50cm**	**% Time WL > Bank + 10cm**
Ilwaco Slough (11)	1*	0.93	1.71	0.78	4.20	6.80	8.70	47%	31%	48%	32%
	2	0.70	1.86	1.16	8.94	9.30	8.04	57%	24%	58%	25%
	3	0.90	2.12	1.22	9.73	10.10	8.27	48%	13%	49%	14%
	4	1.01	2.00	0.99	4.33	5.20	5.23	44%	18%	45%	19%
	5	1.17	2.26	1.09	1.58	2.70	2.48	37%	8%	38%	9%
Secret River (12)	0*	0.15	1.04	0.89	10.6	23.9	26.9	85%	61%	85%	64%
	1	0.32	1.42	1.09	19.3	22.6	20.6	75%	48%	77%	50%
	2	-0.04	2.13	2.17	22.5	14.9	6.87	96%	20%	97%	22%
	3	-0.03	1.98	2.01	20.7	15.1	7.52	96%	26%	97%	29%
Welch Island (12)	1*	0.19	1.58	1.39	15.3	20.0	14.4	69%	28%	75%	33%
	2	0.36	1.65	1.29	8.75	9.20	7.13	62%	25%	67%	30%
	3	0.71	1.80	1.09	3.96	5.09	4.67	47%	19%	52%	23%
	4	0.78	1.74	0.96	2.07	3.30	3.44	45%	21%	49%	26%
	5	1.31	1.62	0.31	0.42	1.32	4.27	23%	26%	27%	31%
Whites Island (11)	1*	0.35	1.43	1.08	22.5	39.6	36.7	75%	41%	88%	51%
	2	0.34	1.41	1.07	10.8	20.5	19.1	76%	42%	89%	52%
	3	0.61	1.53	0.92	11.1	36.2	39.5	63%	37%	75%	46%
	4	0.92	1.93	1.00	34.0	50.1	50.0	47%	19%	57%	27%
	5	0.44	1.45	1.01	1.90	2.83	2.80	71%	40%	84%	50%
Cunningham Lake (13)	1	0.81	1.11	0.30	3.17	17.6	58.7	51%	56%	97%	98%
Campbell Slough(13)	1	0.78	1.44	0.66	11.2	23.1	35.0	66%	50%	98%	96%
Franz lake (12)	0*	0.34	2.36	2.02	24.7	23.8	11.8	85%	33%	100%	77%
	3	0.40	1.39	0.99	4.20	14.3	14.4	80%	45%	99%	99%
	4	0.85	1.45	0.60	6.20	13.2	22.0	48%	44%	100%	98%

4.4 Food Web

4.4.1 Primary Production

4.4.1.1 *Emergent Wetland Vegetation*

Quantity

Some of the results presented here were previously reported in Sagar et al. (2014; 2015). However, the results are summarized again in the current report to provide context for the most recent data collected in the winter of 2014. The above ground biomass estimates for emergent wetland vegetation in the low and high marsh strata are provided in Table 24. Summer wetland biomass was positively correlated with elevation ($r= 0.60$, $p<0.01$) and negatively correlated with rkm ($r=-0.34$, $p=0.04$) and SEV ($r=-0.57$, $p<0.01$). The greatest biomass occurred on the high marsh, with statistically significant differences between each of the three marsh strata: high marsh, low marsh and SAV ($r^2=73\%$, $p<0.01$). High marsh had the greatest plant biomass (average of 929 g/m^2), compared to low marsh and SAV (average of 249 g/m^2 and 42 g/m^2, respectively). The highest summer emergent wetland biomass estimate was from Secret River in the high marsh during 2012 (1443 g/m^2) and the lowest estimate was from the low marsh at Campbell Slough in 2013 (56.3 g/m^2). The summer biomass from the submerged aquatic vegetation strata was typically the lowest of the three strata. 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). For the years sampled, summer biomass estimates in all strata decreased with increasing rkm. The four sites in the lower reaches had greater biomass than the two upper-most sites (high marsh average of 1162 $g/m^2$and 426 g/m^2, respectively). Estimated biomass was significantly different between the lower reaches (zones 1 and 2) and the upper reaches (zones 4 and 5); no sites were sampled in the middle part of the study area (zone 3; $r^2=77\%$, $p<0.01$).

Temporal trends are difficult to discern because of limited sampling over the three-year period. Comparisons between all three years are limited to three sites for the high marsh strata, one site for the low marsh strata, and two sites for the SAV strata. These limited data may be used to make broad generalizations that should be interpreted with caution given the limited number of comparisons. Within the high marsh strata, summer biomass was the lowest in 2011 at Ilwaco Slough and Franz Lake and lowest at the Whites Island site in 2012. The data from the low-marsh and SAV strata indicate that summer biomass values were variable between years.

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/m^2). Sites are ordered by distance from the river mouth.

		Summer Standing Stock 2011		Winter Standing Stock* 2012		Organic Matter Prod.	Summer Standing Stock 2012		Winter Standing Stock 2013		Organic Matter Prod.	Summer Standing Stock 2013		Winter Standing Stock 2014		Organic Matter Prod.
Site	**Strata**	**n**	**Avg Dry wt g/m^2 (SD)**	**n**	**Avg Dry wt g/m^2 (SD)**	**Avg Dry wt g/m^2**	**n**	**Avg Dry wt g/m^2 (SD)**	**n**	**Avg Dry wt g/m^2 (SD)**	**Avg Dry wt g/m^2**	**n**	**Avg Dry wt g/m^2 (SD)**	**n**	**Avg Dry wt g/m^2 (SD)**	**Avg Dry wt g/m^2**
Ilwaco Slough (BBM)	high marsh	7	976 (421)	7	385 (133)	591	10	1175 (257)	10	254 (135)	921	10	1141 (429)	10	227 (175)	914
Secret River (SRM)	high marsh	ND	ND	ND	ND	ND	5	1443 (148)	5	194 (210)	1248	9	1062 (386)	9	241 (151)	821
Welch Island (WI2)	high marsh	ND	ND	ND	ND	ND	5	1141 (322)	9	272 (122)	870	9	1361 (647)	9	365 (150)	996
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
Campbell Slough (CS1)	high marsh	3	410 (356)	4	101 (64)	309	ND	ND	ND	ND	ND	6	434 (67)	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
Ilwaco Slough (BBM)	low marsh	1	24 (NA)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Secret River (SRM)	low marsh	ND	ND	ND	ND	ND	5	265 (71)	5	15 (15)	250	9	175 (124)	9	9 (9)	166
Welch Island (WI2)	low marsh	ND	ND	ND	ND	ND	4	401 (362)	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
Campbell Slough (CS1)	low marsh	5	278 (151)	4	3 (4)	274	ND	ND	ND	ND	ND	11	56 (38)	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

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

Vegetative biomass samples were taken from three primary marsh strata: high marsh, low marsh, and submerged aquatic vegetation (SAV), with 66 percent of the samples categorized further into species-specific strata. In general, the species comprising the vegetation biomass samples are the dominant species found in the lower river. Although dominant species were noted in many of the samples, frequently the samples were a mix of more than one species. The samples in the non-specific categories were either a mix of many species (with no dominant) or the dominant species were not recorded at the time of sampling.

Dominance of *P. arundinacea* in the lower river has previously been documented (Sagar et. al. 2013) however, in this study *C. lyngbyei* was present as a dominant species in 45 of the summer samples, while *P. arundinacea* was only present as a dominant in 33 samples. This is primarily because four of the six study sites are located in the lower portion of the lower river, below rkm 89, where salinity and the tidally dominated hydrology reduce the probability of *P. arundinacea* occurrence (Borde et al. 2012; Sagar et al. 2013). *C. lyngbyei* was only present at the four lower lower river sites, occurring as a single dominant species at Ilwaco Slough (rkm 6) and mixed with other species at the Secret River site (rkm 37) and Welch Island site (rkm 53). The Whites Island site (rkm 72) had a contiguous patch of *C. lyngbyei* while the rest of the high marsh was a mix of *P. arundinacea* and other species. Thirty of the 33 summer samples with *P. arundinacea* came from the three sites located at the three sites located at or upriver from rkm 72. *P. arundinacea* was mixed with other species or absent from sites in the lower reaches. *Sagittaria latifolia* occurred in the samples from the four sites at rkm 53 and higher.

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 and depicted in Figure 24 as the difference between summer and winter standing stock values. Similar spatial patterns to those observed for summer biomass apply regarding the annual detrital contribution (summer – winter biomass) as well. In general, the annual detrital contribution was greater in the lower sites than at the upper sites, 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 (Figure 24). However, the implications of the spatial distribution of *C. lyngbyei* for this result must be considered, because this species only occurs in the lower portions of the study area, and therefore it is not affected by high water flooding effects (which occurred in two of three sample years). The two most common species in the samples were *C. lyngbyei* and *P. arundinaceae*. The detrital contribution for *C. lyngbyei*, across all sites where it was measured, was 858 g/m^2, while *P. arundinacea* was 345 g/m^2.

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/m^2).

		Summer Standing Stock 2011		Winter Standing Stock 2012		Organic Matter Prod.	Summer Standing Stock 2012		Winter Standing Stock 2013		Organic Matter Prod.	Summer Standing Stock 2013		Winter Standing Stock 2014		Organic Matter Prod.
Dominant Species	**Common Name**	***n***	**Avg Dry Wt g/m^2 (SD)**	***n***	**Avg Dry wt g/m^2 (SD)**	**Avg Dry wt g/m^2**	***n***	**Avg Dry wt g/m^2 (SD)**	***n***	**Avg Dry wt g/m^2 (SD)**	**Avg Dry wt g/m^2**	***n***	**Avg Dry wt g/m^2 (SD)**	***n***	**Avg Dry wt g/m^2 (SD)**	**Avg Dry wt g/m^2**
Carex lyngbyei	Lyngby's sedge	3	1049 (558)	5	331 (192)	718	10	1234 (377)	14	177 (115)	1057	7	1105 (290)	14	305 (154)	801
C. lyngbyei/ Agrostis spp.	Lyngby's sedge/ bentgrass	4	921 (370)	3	351 (194)	570	4	1009 (153)	6	236 (168)	773	6	1041 (527)	5	127 (98)	914
C. lyngbyei/ mixed spp.	Lyngby sedge/ high marsh	ND	ND	ND	ND	ND	8	1250 (288)	2	263 (35)	987	15	1291 (543)	5	261 (221)	1030
Phalaris arundinacea	reed canarygrass	9	578 (760)	13	306 (270)	272	ND	ND	6	353 (286)	NA	9	716 (718)	9	297 (251)	419
Polygonum amphibium	water smartweed	ND	ND	ND	ND	ND	3	747 (488)	1	274 (NA)	473	4	208 (192)	2	286 (296)	-79
Sagittaria latifolia	wapato	1	150 (NA)	4	4 (6)	146	4	91 (94)	3	10	(14)	12	101 (111)	6	10 (5)	91

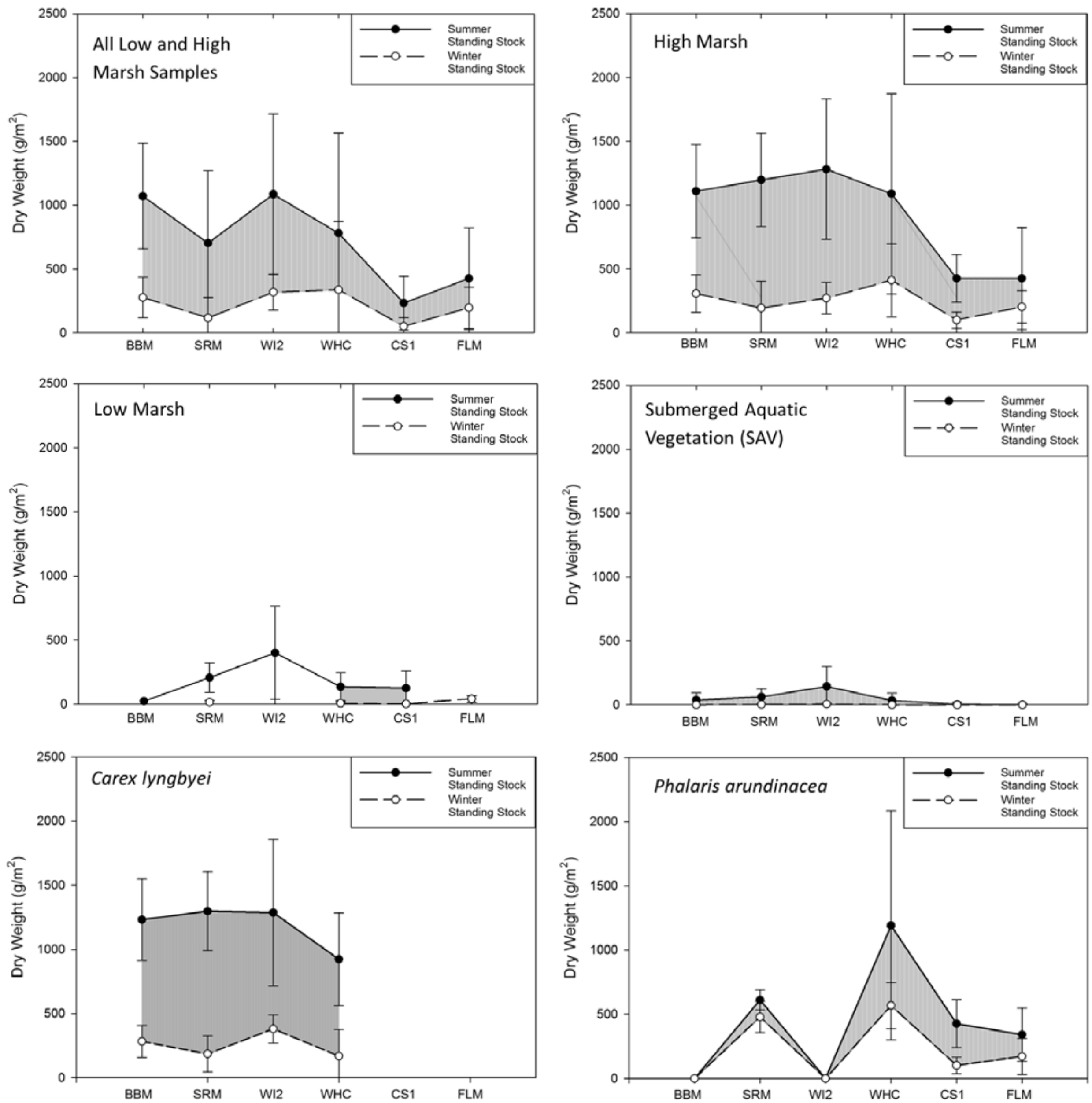


Figure 24. Average aboveground summer and winter standing stock for selected marsh and plant-species strata from three years of data collection (summer sampling in 2011, 2012, and 2013 and subsequent winters of 2012, 2013, and 2014). Shading represents the organic matter production, or detrital contribution, from each strata and site. Error bars represent ± 1 standard deviation.

4.4.1.2 *Pelagic*

Quantity

The quantity of pelagic primary production (amount of organic matter fixed in phytoplankton biomass) was estimated based on total chlorophyll *a* (a proxy for phytoplankton biomass) and from phytoplankton cell counts. Chlorophyll *a* ranged from 6.2 mg m-3 at Franz Lake Slough in April and 6.3 mg m-3 at Whites Island in July to 40.3 mg m-3 at Franz Lake Slough in June (Figure 25). There were no significant differences between chlorophyll *a* concentration among sites or over time, owing to the large within-site variations observed in the data set (p = 0.366 for time, p = 0.892 for site).

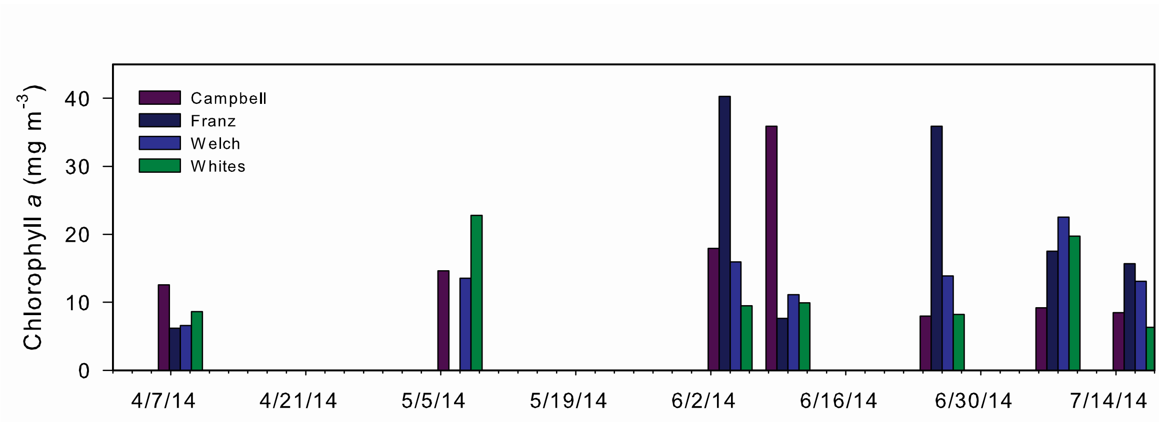


Figure 25. Chlorophyll *a* concentrations (mg m^{-3}) from discrete samples taken at the four trends sites (Campbell Slough, Franz Lake Slough, Welch Island, Whites Island).

Chlorophyll *a* concentrations were similar at Welch Island and Whites Island in a date-by-date comparison (Figure 26). During the summer months, chlorophyll *a* concentrations were higher at Welch Island compared to Campbell Slough; in contrast, chlorophyll *a* concentrations were lower at Welch Island than Franz Lake after the spring freshet.

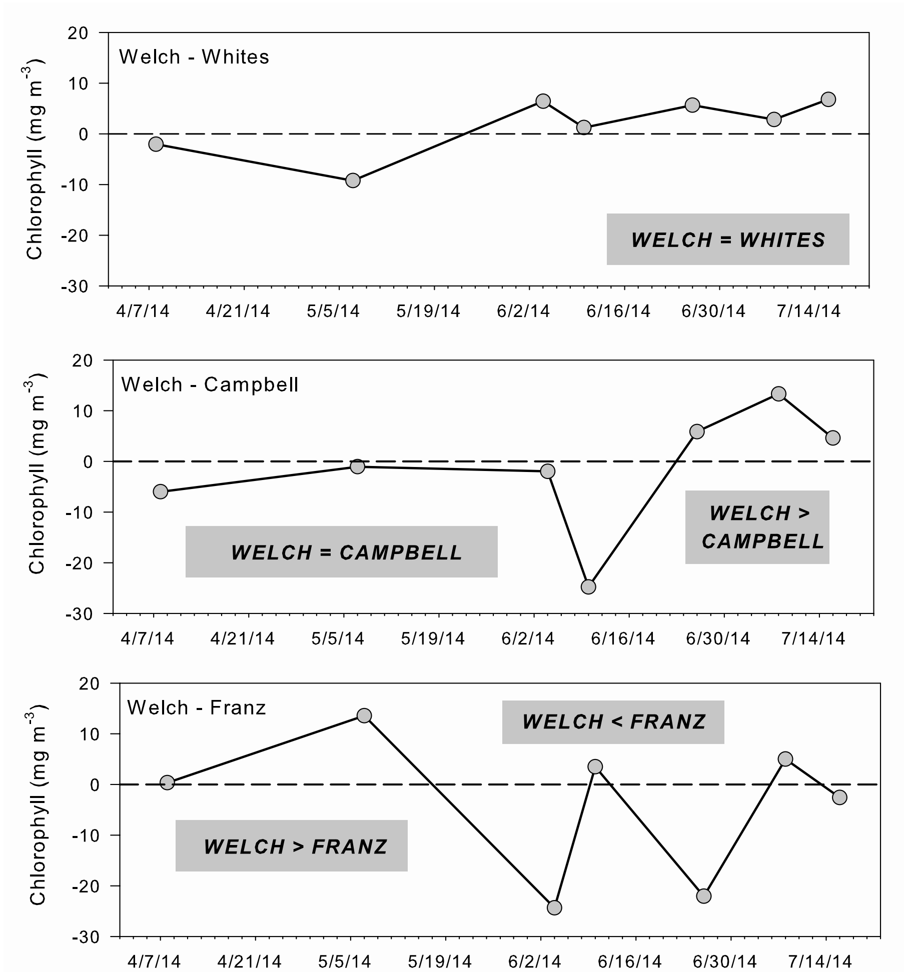


Figure 26. Differences in chlorophyll *a* concentration (mg m^{-3}) between Welch Island (0 km) and upstream sites (Whites Island, Campbell Slough, and Franz Lake Slough) over the course of the sampling season.

Species Composition

The phytoplankton assemblages at the two sites in Reaches B and C (Welch Island and Whites Island) were dominated by diatoms throughout the year, with smaller contributions from other groups including cryptophytes (Figure 27). Variations over time were very similar between Welch Island and Whites Island. In contrast, the phytoplankton assemblages were not always dominated by diatoms at Campbell Slough or Franz Lake Slough. Instead, green algae and cryptophytes were very abundant prior to the spring freshet. After the freshet, cyanobacteria became very abundant at both of these sites, accounting for a similar, if not greater, proportion of total cells than the diatoms.

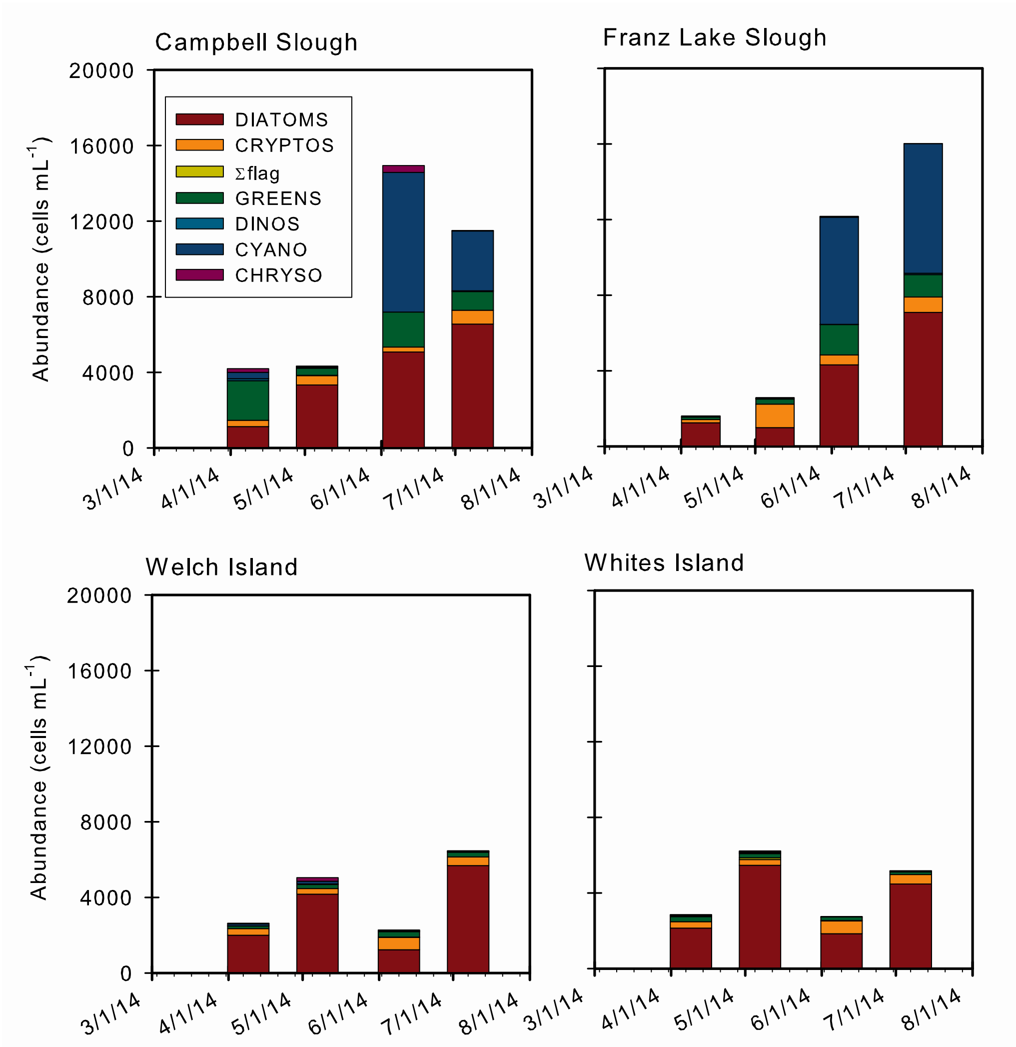


Figure 27. Histograms showing the relative proportion of different phytoplankton taxa at each of the four trends sites sampled in 2014. The height of the histogram corresponds to the total abundance of phytoplankton.

4.4.2 Secondary Production

4.4.2.1 *Quantity*

Zooplankton abundances were much higher in Campbell Slough than any of the other trends sites (Figure 28). At each of the sites, zooplankton were more abundant in June and July compared to April and May.

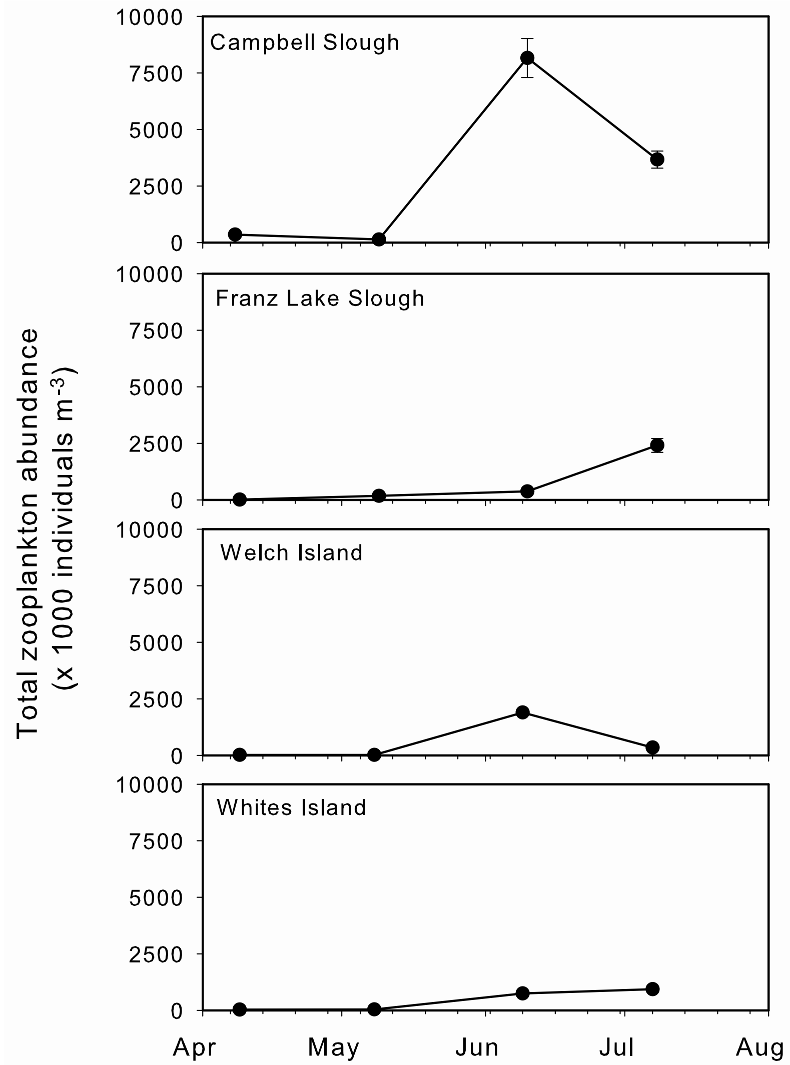


Figure 28. Total zooplankton abundances [∑(copepods, cladocerans, rotifers, other)] at each of the fixed sites sampled in 2014. Error bars represent one standard deviation; where they are not visible, standard deviation are smaller than the symbol.

4.4.2.2 *Species Composition*

By far, rotifers numerically dominated the zooplankton assemblage throughout the time series sampled and at each of the sites (Figure 29). The peaks in zooplankton abundance at Campbell Slough during summer were attributable to high numbers of rotifers and cladocerans (Figure 30).

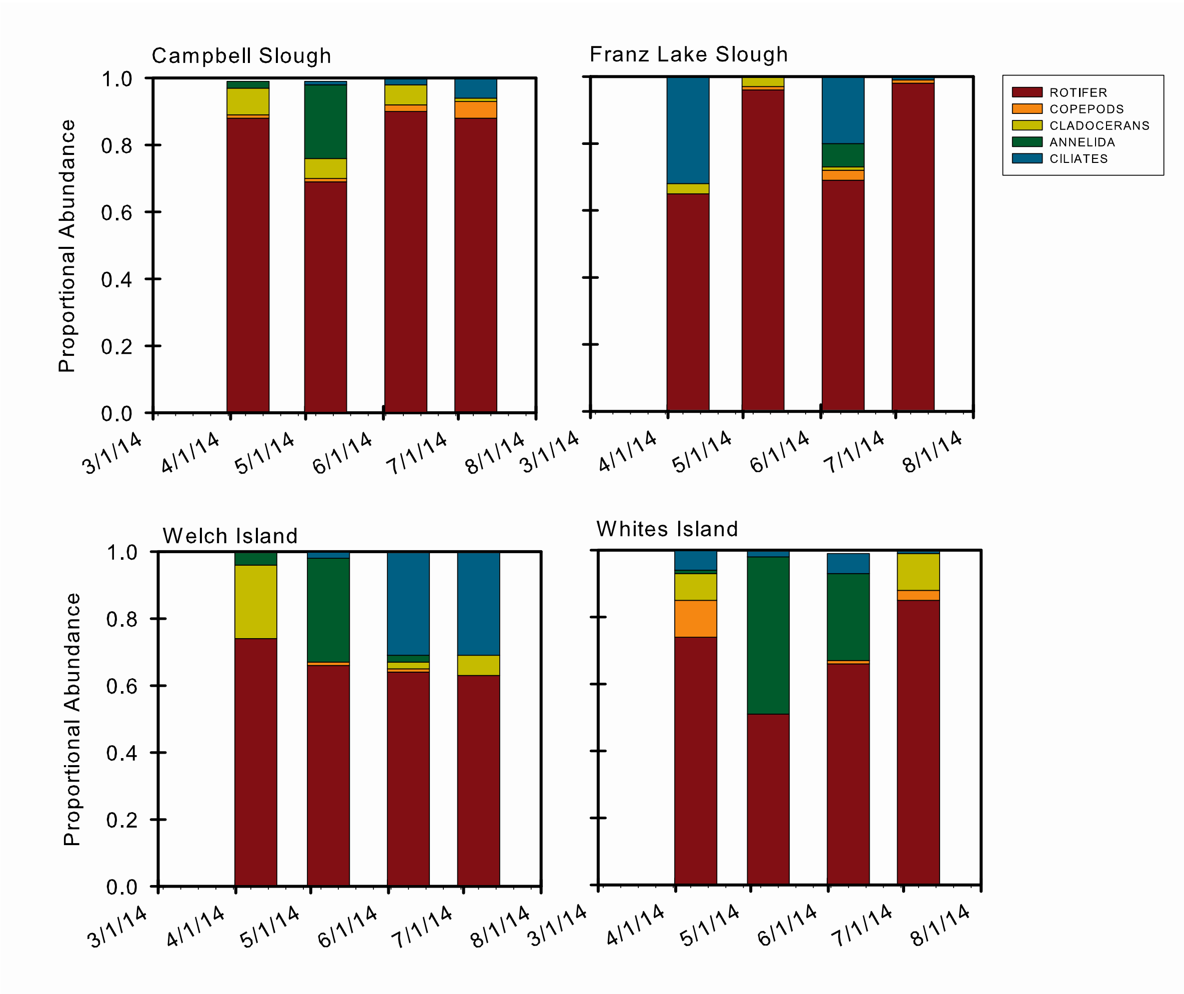


Figure 29. Histograms showing the relative proportion of different zooplankton taxa (rotifers, copepods, cladocerans, annelids, and ciliates) at each of the trends sites sampled in 2014. These data are based on abundance and not biomass.

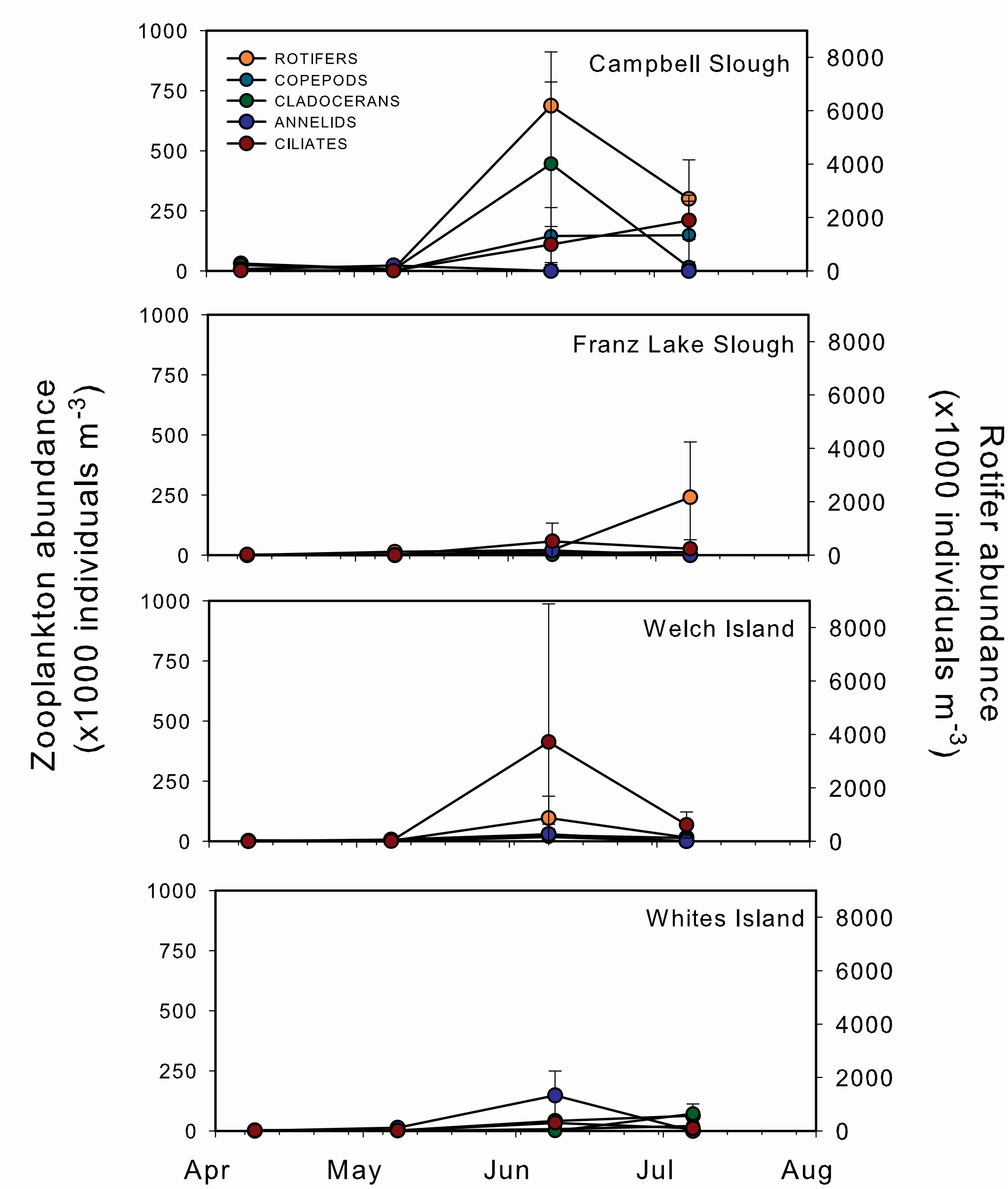


Figure 30. Abundances of each of the major zooplankton groups at each of the four fixed sites sampled in 2014. Since rotifers were so much more abundant than other taxa (especially at Campbell Slough), their abundances are displayed on a secondary axis with a higher maximum value.

4.4.3 Stable Isotope Ratios

Between 2010 and 2014, samples (n = 1100) were analyzed for stable isotope analysis. Samples of the same material collected from the same site and date were analyzed in duplicate or triplicate, depending on the amount of sample material available. For certain media, fewer replicates were analyzed for two reasons, 1) within-sample isotopic variability among replicate samples was determined to be extremely low based on the first few years of data (salmon muscle, for example); or 2) insufficient sample material for replication was available (salmon epidermal mucus from small fish, for example). Replicate sample data were averaged, yielding 582 samples for this analysis.

Mean isotope ratios of each sample type are shown in Figure 31 and data summarized by site are shown in Table 26. Generally, mean isotope ratios increased (i.e., became less negative for $\delta^{13}C$ and more positive for $\delta^{15}N$) from site to site along a downstream gradient for the site-specific samples (invertebrates and autotrophs). However, isotopic signatures of fish tissues varied by month of fish catch and by fish length (Table 27). Delta 15N signatures of fish tissues from this study decreased with increasing fish length (Figure 32).

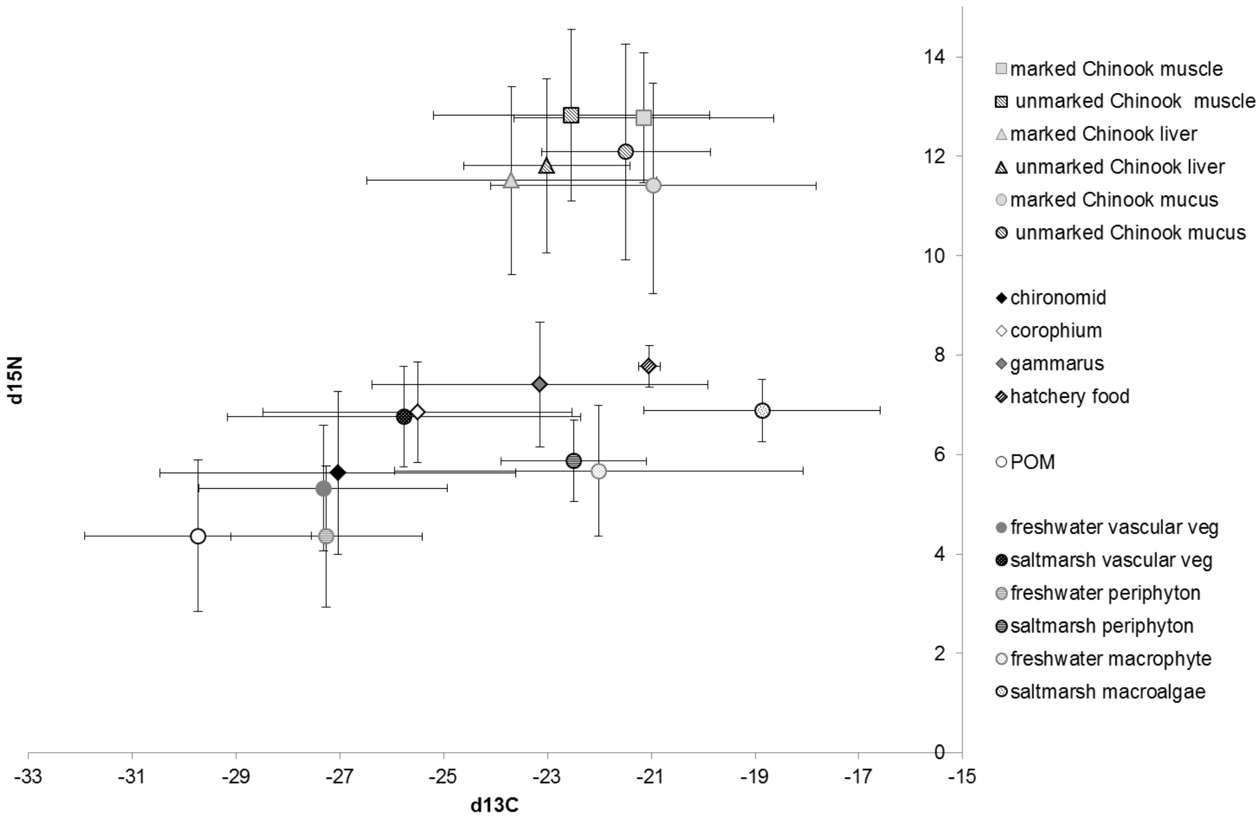


Figure 31. Carbon and nitrogen isotope bi-plot of marked and unmarked juvenile Chinook salmon tissues, invertebrate food sources, hatchery food, and autotrophs from trends sites, 2010-2014. Symbols represent mean values of all samples and error bars represent standard deviation. $d^{13}C$=delta ^{13}C; $d^{15}N$=delta ^{15}N; POM=particulate organic matter.

Table 26. Mean (± standard deviation) carbon and nitrogen isotope values for marked and unmarked juvenile Chinook salmon tissues, invertebrate prey, and autotrophs from fixed sites, 2010–2014. This table includes only live vegetation samples (not dead or decomposing). The C/N ratio listed is the atomic ratio, which is the mass ratio times 14/12. $\delta^{13}C$=delta 13C; $\delta^{15}N$=delta ^{15}N; C/N ratio=carbon-to-nitrogen ratio; n=sample size; POM=particulate organic matter.

Trophic Level	Sample Type	Ilwaco Slough				Welch Island				Whites Island			
		Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n	Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n	Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n
Chinook salmon	**Muscle**	-	-	-	0	-21 ± 1.1	13.8 ± 1.1	3.8	22	-22.5 ± 3.2	13.1 ± 1.4	3.9	34
	Marked	-	-	-	0	-	-	-	0	-20.7 ± 2	13.7 ± 1.4	3.9	6
	Unmarked	-	-	-	0	-21 ± 1.1	13.8 ± 1.1	3.8	22	-22.9 ± 3.3	13 ± 1.4	3.8	28
	Liver	-	-	-	0	-22.8 ± 1.1	11.9 ± 1.1	5.2	19	-22.3 ± 1.2	12.1 ± 1.8	5.6	20
	Marked	-	-	-	0	-	-	-	0	-21.5 ± 1.1	12.6 ± 2.1	6	4
	Unmarked	-	-	-	0	-22.8 ± 1.1	11.9 ± 1.1	5.2	19	-22.5 ± 1.2	11.9 ± 1.8	5.6	16
	Mucus	-	-	-	0	-21.5 ± 1	12.1 ± 1.7	4	18	-20.6 ± 1.5	12.5 ± 2.2	4	22
	Marked	-	-	-	0	-	-	-	0	-19.2 ± 1	13.1 ± 2.7	3.9	4
	Unmarked	-	-	-	0	-21.5 ± 1	12.1 ± 1.7	4	18	-20.9 ± 1.5	12.4 ± 2.2	4	18
Invertebrates	**Chironomid**	-	-	-	0	-25.5 ± 0.8	7.7 ± 0.6	5.6	2	-22.4 ± 2	7.1 ± 0.5	5.9	6
	Corophium	-22.7 ± 2.8	7 ± 1.6	5.6	5	-26.8 ± 0	7.1 ± 0.5	5.6	2	-27.5 ± 1.5	6.7 ± 0.5	6.3	5
	Gammarus	-20.8 ± 2.2	8.5 ± 0.6	5.2	9	-25.8 ± 2.2	7.1 ± 0.1	5.9	2	-23.9 ± 1.5	6.7 ± 0.6	5.9	5
Autotrophs	**Periphyton**	-22.5 ± 1.4	5.9 ± 0.8	10	11	-24.5 ± 3.6	5.8 ± 0.2	8.1	3	-26.9 ± 0.9	4.3 ± 1.7	14.9	10
	POM	-27.6 ± 0.9	6 ± 0.7	11.6	15	-28 ± 1.4	5.5 ± 0.5	8	10	-29.9 ± 1.3	4.7 ± 0.7	9.7	25
	Vegetation	-23.7 ± 4.6	6.7 ± 1	22.9	27	-27.1 ± 3	6 ± 1.2	26.4	19	-25.4 ± 4.1	5.4 ± 1.1	20	45

Trophic Level	Sample Type	Campbell Slough				Franz Lake			
		Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n	Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n
Chinook salmon	**Muscle**	-22.5 ± 2.7	12.1 ± 1.4	3.8	60	-19.4 ± 1.2	15.8 ± 1.8	4.1	4
	Marked	-21.2 ± 2.6	12.6 ± 1.2	3.8	28	-	-	-	0
	Unmarked	-23.7 ± 2.2	11.6 ± 1.4	3.9	32	-19.4 ± 1.2	15.8 ± 1.8	4.1	4
	Liver	-24.5 ± 2.4	10.9 ± 1.6	5.7	21	-22.1 ± 1.9	13.8 ± 2.8	6.1	4
	Marked	-24.6 ± 2.8	11.1 ± 1.7	5.9	10	-	-	-	0
	Unmarked	-24.4 ± 2	10.8 ± 1.7	5.6	11	-22.1 ± 1.9	13.8 ± 2.8	6.1	4
	Mucus	-22.1 ± 2.7	10.8 ± 1.5	3.8	26	-19.5 ± 0.8	16.3 ± 0.8	4.9	3
	Marked	-21.5 ± 3.4	10.8 ± 1.6	3.7	12	-	-	-	0
	Unmarked	-22.6 ± 1.9	10.7 ± 1.5	3.8	14	-19.5 ± 0.8	16.3 ± 0.8	4.9	3
Invertebrates	**Chironomid**	-27.3 ± 3.6	6.2 ± 0.5	5.6	16	-28.6 ± 2.3	4.5 ± 1.8	5.7	19
	Corophium	-27.1	6.5	6	1	-	-	-	0
	Gammarus	-23.4 ± 1.1	6.5 ± 0	5.7	2	-29.2 ± 2.4	5.7 ± 2.2	5.2	2
Autotrophs	**Periphyton**	-27.8 ± 1.5	4.2 ± 1.2	13	8	-28.1 ± 1.4	4.1 ± 1.4	13.4	9
	POM	-30.5 ± 2.8	4.3 ± 0.9	8.5	28	-30.7 ± 1.7	2.6 ± 1.5	7.1	25
	Vegetation	-27.8 ± 1.5	4.7 ± 1.7	20.5	29	-28.9 ± 1.1	4.6 ± 1	23.2	16

Table 27. Mean (± standard deviation) carbon and nitrogen isotope values for marked and unmarked juvenile Chinook salmon tissues from trends sites by month, 2010–2014. δ^{13}C=delta ^{13}C; δ^{15}N=delta ^{15}N; C/N ratio=carbon-to-nitrogen ratio; n=sample size.

Origin	Month	Tissue	Mean δ^{13}C	Mean δ^{15}N	C/N ratio	Mean total length (mm)	n
Marked	***April***	Muscle	-22.1 ± 3.7	12.4 ± 1.1	3.8	117.3	3
		Liver	-26.6 ± 4.1	10.4 ± 1.5	5.9	117.3	3
		Mucus	-22.5 ± 3.3	10.8 ± 1.3	3.8	117.3	3
	May	Muscle	-19.9 ± 1.4	13.4 ± 1.1	3.9	80.4	19
		Liver	-21.6 ± 1.2	13.6 ± 1.1	6.8	79.6	5
		Mucus	-18.2 ± 0.7	13.8 ± 1.7	3.8	79.6	5
	June	Muscle	-22.7 ± 2.7	12.2 ± 1	3.8	83.4	5
		Liver	-24.8 ± 1.5	10.4 ± 1.1	5.2	84	3
		Mucus	-22.2 ± 1.4	10.8 ± 0.9	3.9	84	3
	July	Muscle	-23 ± 2.8	11.6 ± 1.1	3.8	91.1	7
		Liver	-23.2 ± 1.8	10.3 ± 0.7	5.2	84	3
		Mucus	-22.1 ± 4	9.8 ± 0.8	3.6	90.4	5
	Total	*Muscle*	*-21.1 ± 2.5*	*12.8 ± 1.3*	*3.8*	*86.3*	*34*
		Liver	*-23.7 ± 2.8*	*11.5 ± 1.9*	*5.9*	*89.6*	*14*
		Mucus	*-21 ± 3.1*	*11.4 ± 2.1*	*3.7*	*90.9*	*16*
							64
Unmarked	***April***	Muscle	-20.9 ± 1.8	14.3 ± 2.3	4	57.2	16
		Liver	-22.7 ± 1.9	12.9 ± 2.2	5.8	59.2	13
		Mucus	-21.4 ± 1.7	13.7 ± 2.4	4.6	59.2	14
	May	Muscle	-20.5 ± 1.4	14.1 ± 1	3.9	63.6	20
		Liver	-22.2 ± 1.3	12.7 ± 1	5.9	59.9	17
		Mucus	-20.6 ± 1.4	13.2 ± 1.2	4.2	61.9	16
	June	Muscle	-24.2 ± 2.8	12 ± 0.9	3.8	70	25
		Liver	-24.2 ± 1.2	10.4 ± 0.7	4.9	65.5	11
		Mucus	-22.3 ± 1.2	10.6 ± 0.8	3.6	64.8	14
	July	Muscle	-23.5 ± 2.2	11.7 ± 1.1	3.8	75.8	25
		Liver	-23.5 ± 1	10.4 ± 0.3	4.8	70.9	9
		Mucus	-22.2 ± 1.8	9.9 ± 1	3.5	76.2	9
	Total	*Muscle*	*-22.5 ± 2.7*	*12.8 ± 1.7*	*3.9*	*66.5*	*86*
		Liver	*-23 ± 1.6*	*11.8 ± 1.8*	*5.5*	*62.9*	*50*
		Mucus	*-21.5 ± 1.6*	*12.1 ± 2.2*	*4*	*64.4*	*53*
							189

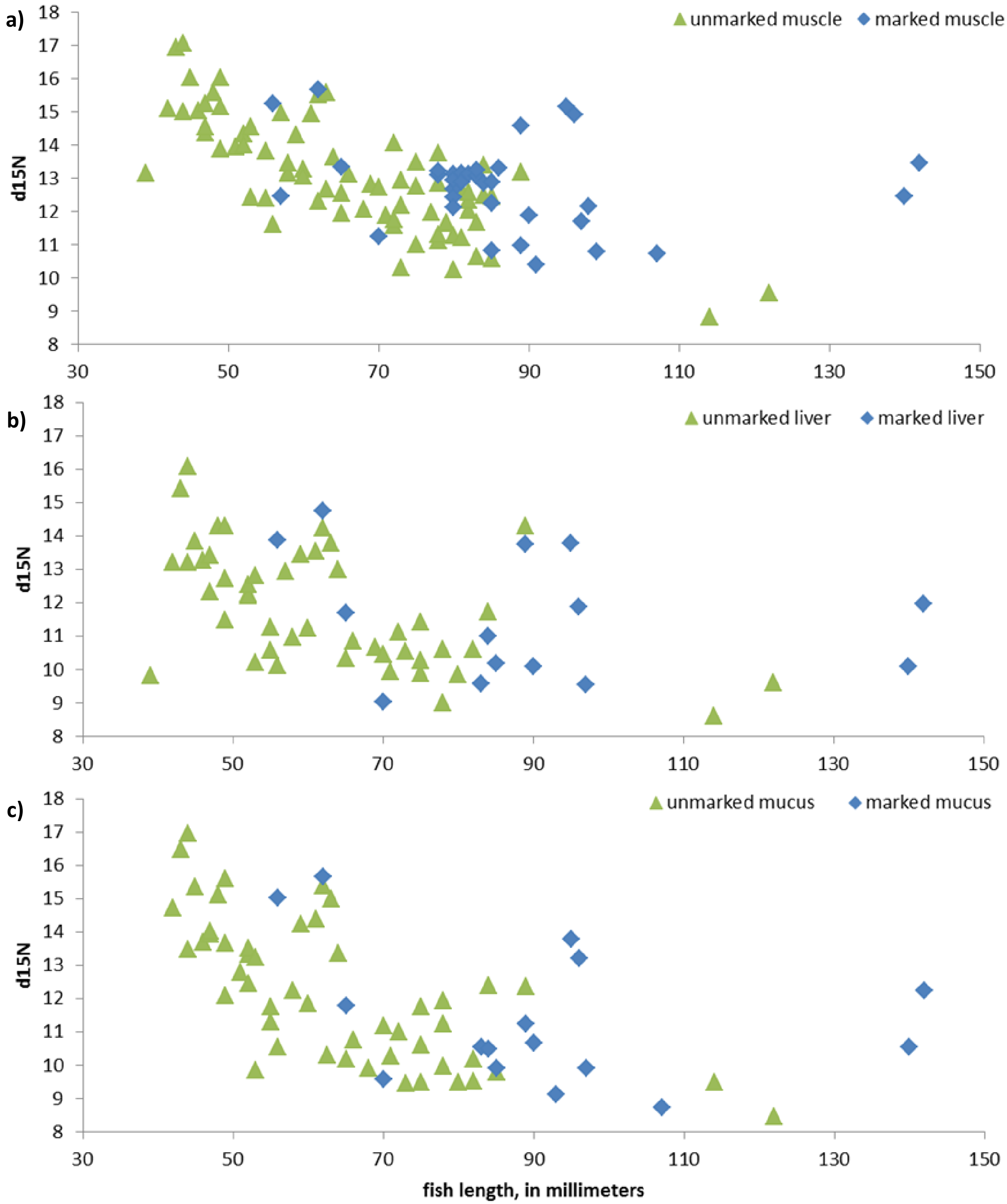


Figure 32. Delta-15N (d15N) of marked and unmarked juvenile Chinook salmon tissues a) muscle, b) liver, and c) epidermal mucus) as a function of total length (mm).

Carbon-to-nitrogen (C/N) ratios detected in juvenile Chinook salmon tissues, according to total length, are presented in Figure 33. Many fish in the 70-90 mm length range had C/N ratios of 3.5 to 4 for muscle and mucus samples (Figure 33) and C/N ratios for liver samples were more variable.

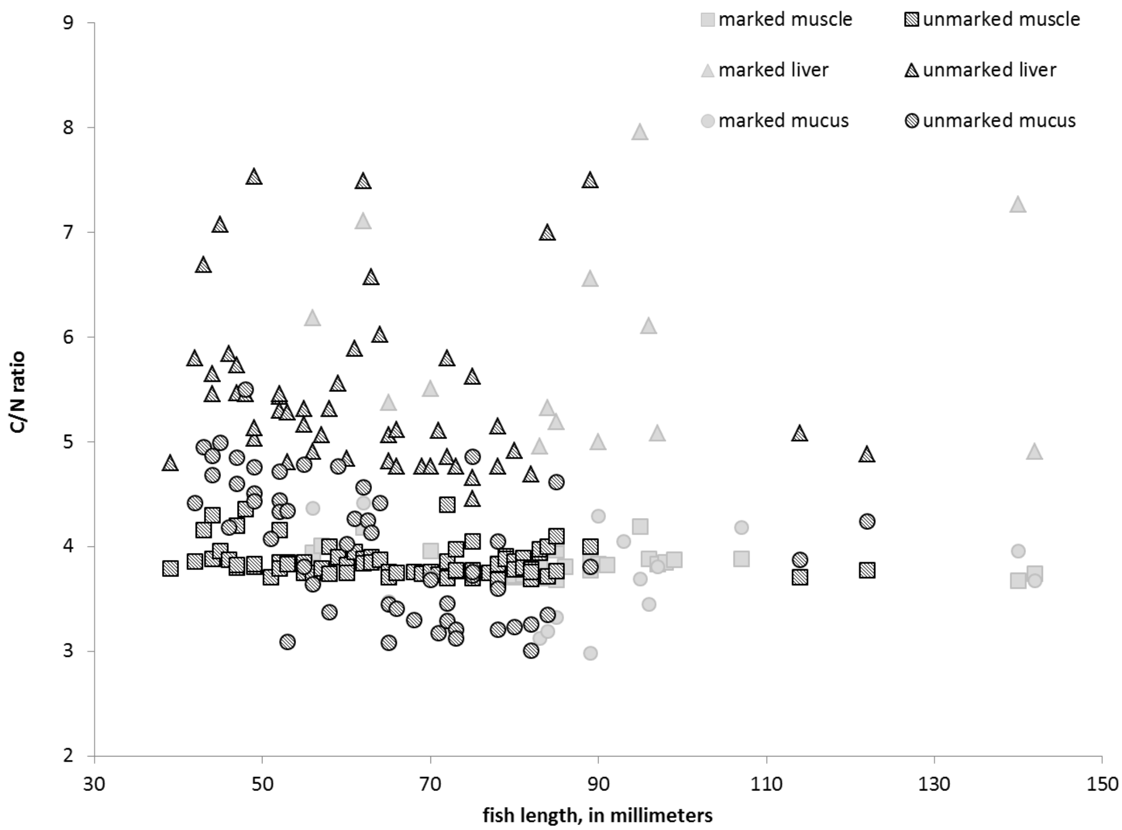


Figure 33. Carbon-to-nitrogen (C/N) ratios of marked and unmarked juvenile Chinook salmon tissues as a function of total fish length.

Isotopic signatures of autotrophs differed among sites along the estuarine gradient, with increased isotopic enrichment at downstream sites (Table 28). Carbon isotopic values varied more among the different vegetation types than it did among periphyton or particulate organic matter (POM) samples. The mean isotopic signatures of carbon and nitrogen from vegetation of each species are shown in Table 29.

Table 28. Mean carbon and nitrogen isotope values for autotrophs by site, 2010–2014. Values are the mean plus or minus one standard deviation. This table includes only live vegetation samples. The C/N ratio listed here is the molar ratio, which is the mass ratio times 14/12. $\delta^{13}C$=delta ^{13}C; $\delta^{15}N$=delta ^{15}N; C/N ratio=carbon-to-nitrogen ratio; n=sample size; POM=particulate organic matter.

Autotroph	Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n
Periphyton	*-26 ± 2.7*	*4.8 ± 1.4*	*12.4*	*41*
Franz Lake	-28.1 ± 1.4	4.1 ± 1.4	13.4	9
Campbell Slough	-27.8 ± 1.5	4.2 ± 1.2	13.0	8
Whites Island	-26.9 ± 0.9	4.3 ± 1.7	14.9	10
Welch Island	-24.5 ± 3.6	5.8 ± 0.2	8.1	3
Ilwaco Slough	-22.5 ± 1.4	5.9 ± 0.8	10.0	11
POM	*-29.7 ± 2.2*	*4.4 ± 1.5*	*8.8*	*103*
Franz Lake	-30.7 ± 1.7	2.6 ± 1.5	7.1	25
Campbell Slough	-30.5 ± 2.8	4.3 ± 0.9	8.5	28
Whites Island	-29.9 ± 1.3	4.7 ± 0.7	9.7	25
Welch Island	-28 ± 1.4	5.5 ± 0.5	8.0	10
Ilwaco Slough	-27.6 ± 0.9	6 ± 0.7	11.6	15
Vegetation	*-25.5 ± 3.9*	*5.7 ± 1.3*	*18.0*	*107*
Franz Lake	-28.6 ± 1.3	4.9 ± 1.1	15.2	8
Campbell Slough	-27.5 ± 1.5	4.9 ± 1.4	16.5	21
Whites Island	-25 ± 4.1	5.5 ± 1.1	17.7	40
Welch Island	-26.4 ± 3.4	6.1 ± 1.2	18.6	13
Ilwaco Slough	-23.3 ± 4.5	6.8 ± 0.9	20.1	25

Table 29. Mean (± standard deviation) carbon and nitrogen isotope values for vegetation by site and species, 2010–2014 (live vegetation samples only). The C/N ratio is the molar ratio, which is the mass ratio times 14/12. $\delta^{13}C$=delta ^{13}C; $\delta^{15}N$=delta ^{15}N; C/N ratio=carbon-to-nitrogen ratio; n=sample size; POM=particulate organic matter.

	Mean $\delta^{13}C$	Mean $\delta^{15}N$	C/N ratio	n
Ilwaco Slough	*-23.3 ± 4.5*	*6.8 ± 0.9*	*20.1*	*25*
Carex lyngbyei	-28.2 ± 0.9	6.9 ± 0.4	29.1	6
Cladophora columbiana	-16.9 ± 0.9	6.6 ± 1	10.3	2
Eleocharis parvula	-19.3	6.6	12.3	1
Fucus distichus	-20.2 ± 0.9	6.9 ± 0.6	17.7	6
Lilaeopsis occidentalis	-28.6	8.9	13.1	1
Schoenoplectus americanus	-27 ± 0.7	5.7 ± 1.1	29.3	4
Ulva lactuca	-19.4 ± 6.7	6.7 ± 0.8	9.7	2
Zannichellia palustris	-20.9 ± 1	7.4 ± 0.5	13.2	3
Welch Island	*-26.4 ± 3.4*	*6.1 ± 1.2*	*18.6*	*12*
Carex lyngbyei	-29.7	6.8	19.4	1
Equisetum sp	-25.5 ± 0.1	6.2 ± 1.9	17.5	2
Lysichiton americanus	-28.1	4.9	12.3	1
Myriophyllum spicatum	-22.8	6.5	12.1	1
Phalaris arundinacea	-28.6 ± 1.1	4.7 ± 0.8	40.9	2
Potamogeton richardsonii	-20.5 ± 1.3	7.3 ± 0.1	11.9	2
Sagittaria latifolia	-26.2	6.7	11.4	1
Schoenoplectus americanus	-28.9	6.9	11.1	1
Zannichellia palustris	-30.9	4.8	25	1
Whites Island	*-25 ± 4.1*	*5.5 ± 1.1*	*17.7*	*40*
Alisma triviale	-28.8 ± 0.8	6.3 ± 1.2	9.4	3
Carex lyngbyei	-28.4 ± 0.1	5.5 ± 1.1	29.6	2
Elodea canadensis	-23.6	3.8	9.1	1
Elodea nuttalii	-20.5	5.3 ± 0.6	10.4	2
Equisetum sp	-25.2 ± 0.3	6.2 ± 0.8	23.5	5
Iris pseudacorus	-29.2	6.4	26.3	1
Iris pseudacorus	-28.7 ± 0.5	4.8 ± 1.2	19.2	3
Myriophyllum spicatum	-19.1 ± 0.4	5 ± 0.4	12.1	3
Phalaris arundinacea	-28.6 ± 0.5	4.9 ± 1.3	25.2	7
Potentilla anserina sp. Pacifica	-29.1	3.9	12.5	1
Potamogeton crispus	-20 ± 0.8	6 ± 1.2	10.6	2
Stuckenia pectinata	-18.9	6.4	9.3	1
Potamogeton richardsonii	-19.6 ± 2.5	5.4 ± 1.3	11	5
Sagittaria latifolia	-26.4 ± 0.6	6.4 ± 0.8	15.5	3
Typha latifolia	-27.1	5.7	33.3	1
Campbell Slough	*-27.5 ± 1.5*	*4.9 ± 1.4*	*16.5*	*21*
Elodea nuttalii	-28.9	6.6	9.5	1
Eleocharus palustris	-28.7 ± 0.9	4.7 ± 0.6	15.9	6
Myriophyllum spicatum	-24.5	4.6	12.8	1

	Mean δ^{13}C	Mean δ^{15}N	C/N ratio	n
Phalaris arundinacea	-28.1 ± 0.6	4.6 ± 2.1	22.7	7
Potamogeton crispus	-24.2	7.2	12.2	1
Potamogeton natans	-27.6	3.5	13.1	1
Sagittaria latifolia	-26.1 ± 0.7	5.3 ± 1	11.3	4
Franz Lake	*-28.6 ± 1.3*	*4.9 ± 1.1*	*15.2*	*8*
Phalaris arundinacea	-29.6 ± 1.9	6.3 ± 0.1	16.3	2
Polygonum amphibium	-28.6 ± 0.8	4.5 ± 0.9	14.1	5
Schoenoplectus americanus	-27.0	4.2	18.5	1

4.5 Fish Use

4.5.1 Fish Community Composition

In 2014, fish communities at Welch Island in Reach B and Whites Island in Reach C were dominated by three-spined stickleback (*Gasterosteus aculeatus*), which accounted for 92-96% of the total catch, respectively (Figure 34). Other species present at these two sites in addition to stickleback included juvenile Chinook salmon (*Oncorhynchus tshawytscha*), sockeye salmon (*Oncorhynchus nerka*), banded killifish (*Fundulus diaphanus*), largescale sucker (*Catostomus macrocheilus*), and eulachon (*Thaleichthys pacificus).* The species assemblage at Campbell Slough in Reach F and Franz Lake in Reach H were much more diverse. Stickleback, while abundant, were less dominant than at sites in the lower reaches, accounting for 48% of the total catch at Campbell Slough and 16% at Franz Lake (Figure 34). In addition to stickleback, other prominent species at Campbell Slough included carp (*Cyprinus carpio*), banded killifish, yellow perch (*Perca flavescens*), freshwater sculpins (*Cottus* spp.), largescale sucker, and American shad (*Alosa sapidissima*). Bluegill (*Lepomis macrochirus*), chiselmouth (*Acrocheilus alutaceus*), pumpkinseed (*Lepomis gibbosus*), Chinook salmon, coho salmon (*Oncorhynchus kisutch*), sockeye salmon, and cutthroat trout (*Oncorhynchus clarkii*) were also observed. At Franz Lake, a variety of species were also present including Chinook salmon, coho salmon, sockeye salmon, stickleback, largescale sucker, carp, chiselmouth, northern pikeminnow (*Ptychocheilus oregonensis*), banded killifish, and freshwater sculpin species. At all sites, fish community composition in the 2014 sampling was generally comparable to previous monitoring results from 2008-2013 (Figure 34). The dominance of stickleback at Welch Island and Whites Island observed in 2014 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.

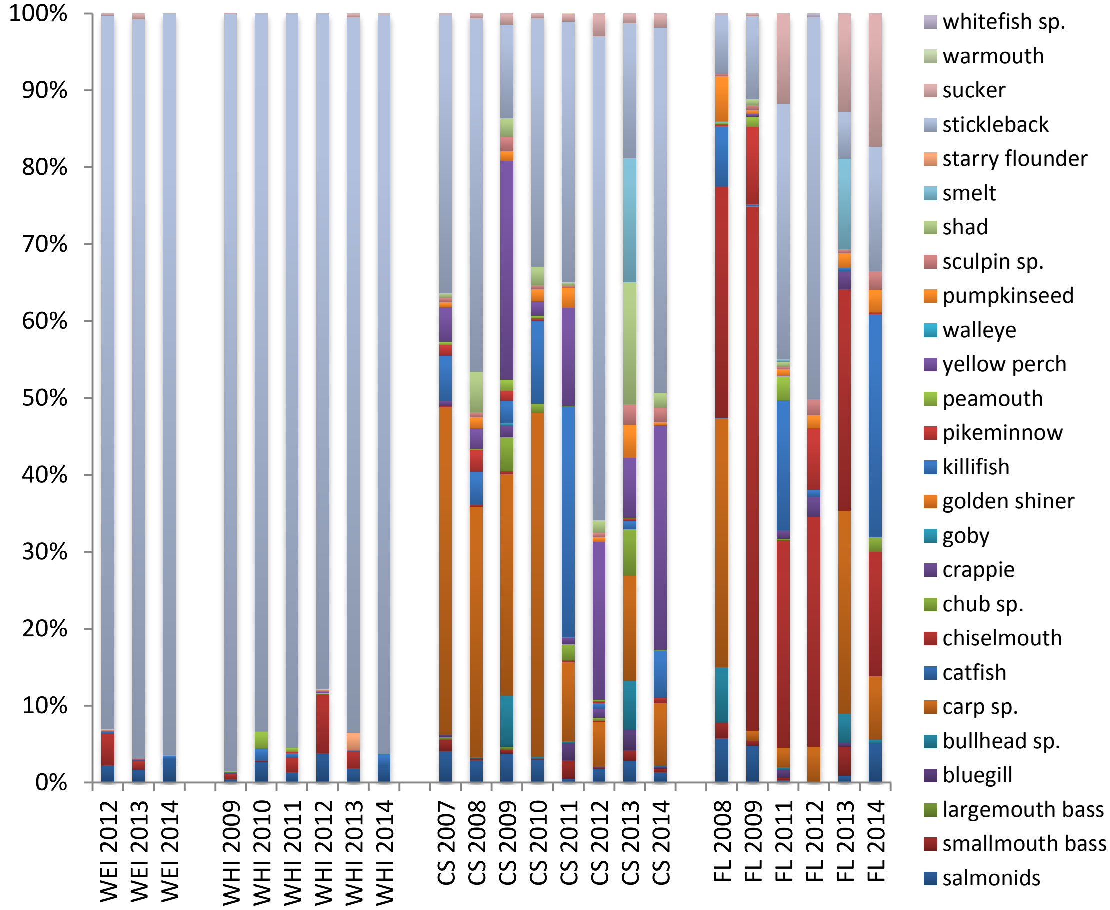


Figure 34. Fish community composition at the four EMP trends sites sampled in 2007-2014. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

In 2014, as in previous sampling years, significant differences were found in species richness among the trends sites ($p \leq 0.05$, ANOVA and Tukey's multiple range test), with mean species richness being lower at Whites Island and Welch Island than at Campbell Slough (Figure 35). Species diversity did not differ significantly among the sites. Within sites, significant differences were not observed among sampling years for either species diversity or species richness ($p \leq 0.05$, ANOVA and Tukey's multiple range test). Mean diversity was significantly higher at Campbell Slough and Franz Lake than at Welch Island or Whites Island (ANOVA, $p = 0.033$), while species richness was significantly higher at Campbell Slough than at the other three sampling sites (ANOVA, $p = 0.015$). For individual sites, there were no significant differences in species richness or species diversity by year, although species richness at Franz Lake tended to be low in comparison to previous years. Between 2008 and 2013, mean species richness at Franz Lake ranged from 6.9 to 9.0, while in 2014 it was 4.6.

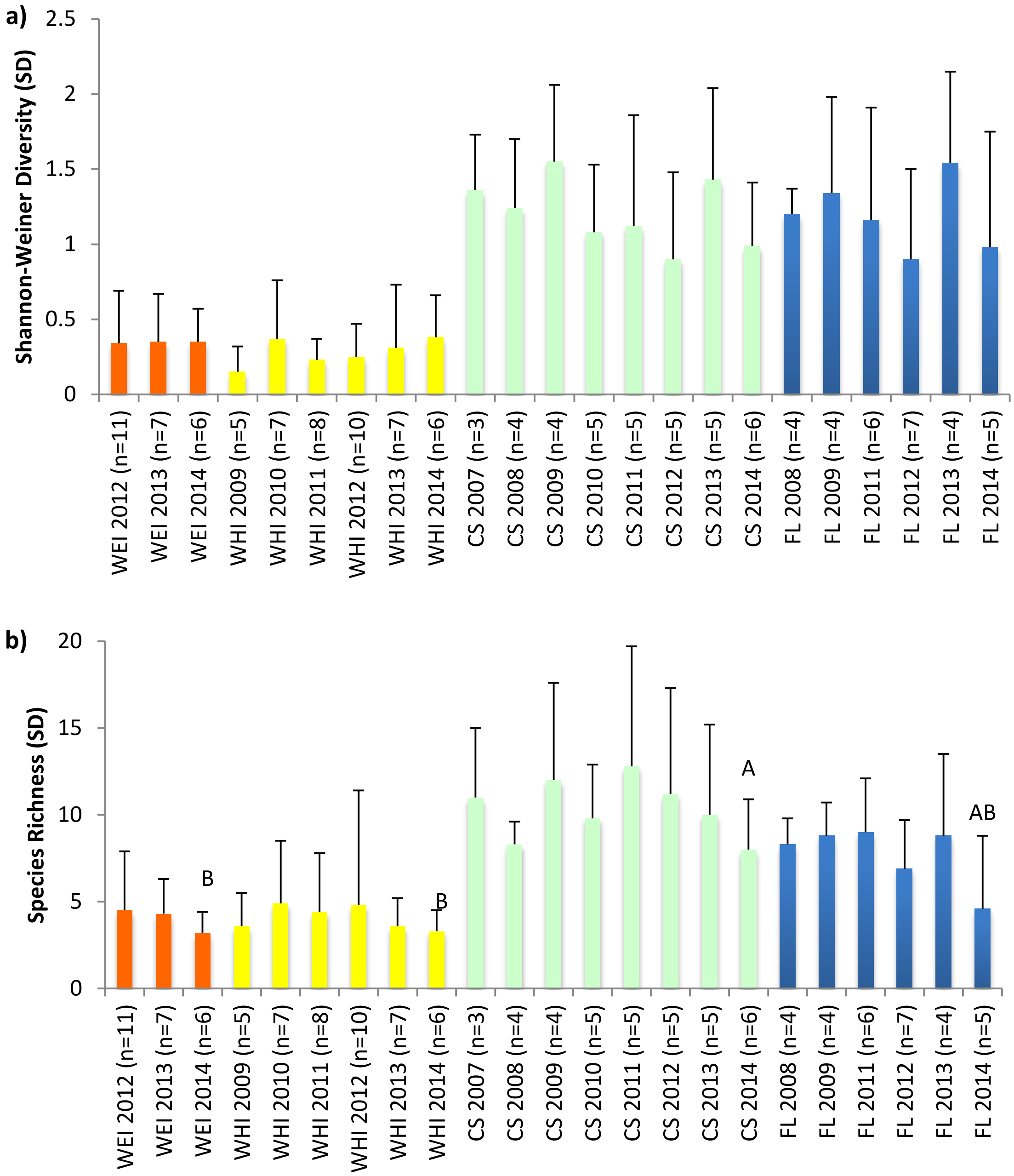
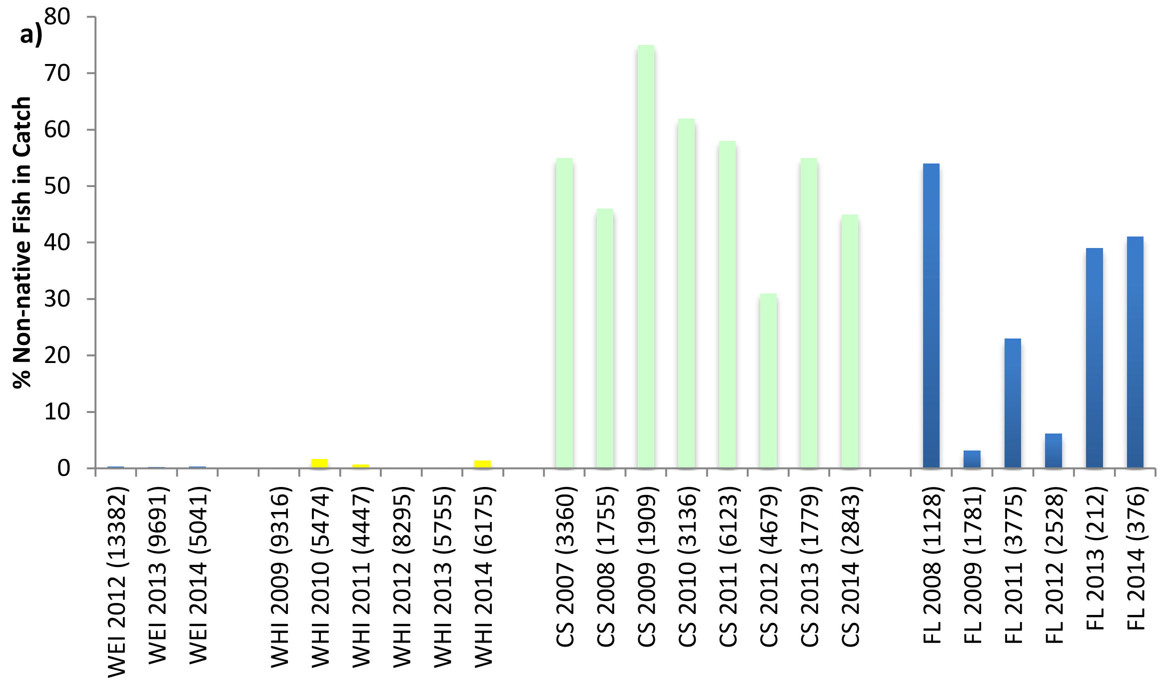


Figure 35. a) Shannon-Weiner diversity index and b) species richness (number of species) in mean (standard deviation, SD) values per sampling event (i.e., per monthly sampling event) at the EMP sampling sites in 2014 as compared to previous sampling years. Statistical differences are indicated by different letter superscripts. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

In 2014, non-native fish species made up only a small percentage of the catches at Welch Island and Whites Island (0.2% and 1.4%, respectively). The percentages of non-native species in catches were substantially higher at Campbell Slough (45%) and Franz Lake (41%; Figure 36). Predatory fish species known to prey on juvenile salmon such as largemouth bass (*Micropterus salmoides*) and smallmouth bass (*Micropterus dolomieu*), northern pikeminnow, and walleye (*Sander vitreus*, were absent at Welch Island and Whites Island (Figure 36) and made up only small percentages of the catches at Campbell Slough (0.5%) and Franz Lake (0.3%). At Welch Island, Whites Island, and Campbell Slough, the percentages of non-native fish species and juvenile salmon predators observed in 2014 were generally comparable to percentages observed in previous years, though the percentages of both groups of fish at Whites Island was slightly higher than typical values in the past. At Franz Lake, percentages of non-native species and predatory fish species have been quite variable from year to year, but with no clear trends. The percentage of non-native species in the catch in 2014 was comparable to most other years, while the percentage predatory fish was among the lowest values observed (Figure 36).



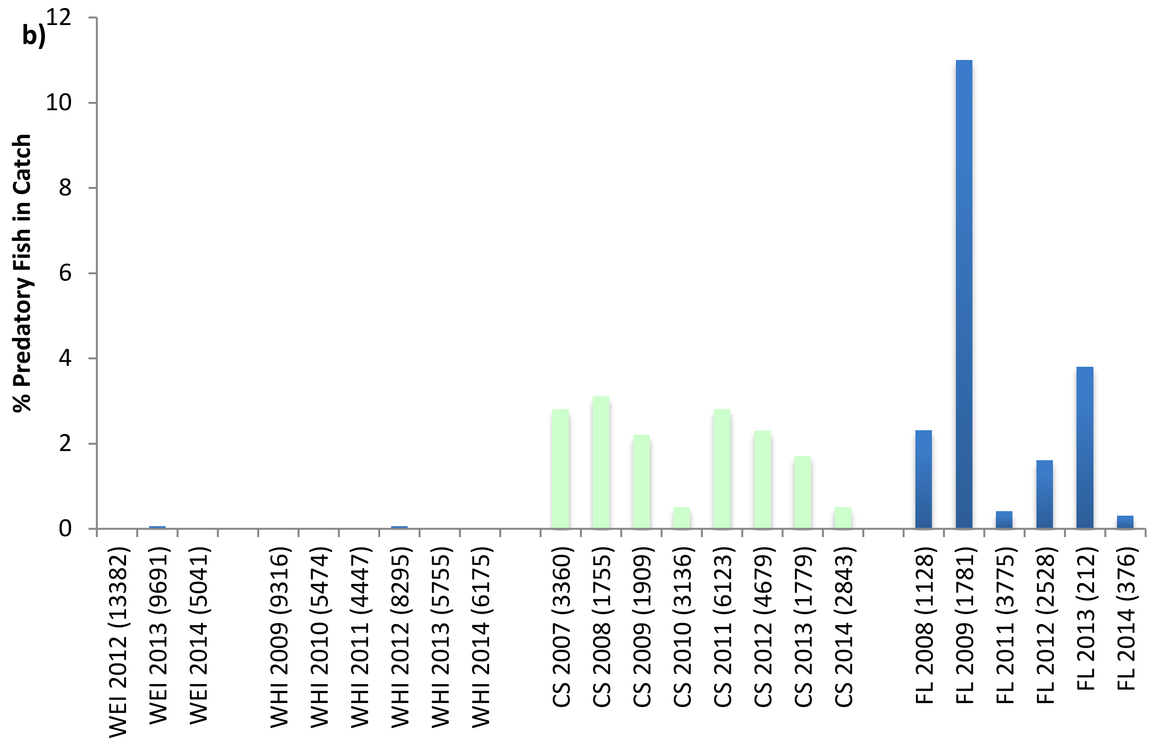


Figure 36. Percentages, based on total number of fish caught for a) non-native fish species and b) % of fish that are recognized predators of juvenile salmon (i.e., smallmouth and largemouth bass, northern pikeminnow, walleye) in 2014 as compared to previous sampling years. Numbers contained in parentheses represent total fish catch at a site within a given year. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

4.5.1.1 *Salmon Species Composition*

Similar to previous sampling years, salmon species composition in 2014 varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 37). Chinook salmon were the dominant species at Welch Island in Reach B, Whites Island in Reach C, and Campbell Slough in Reach F, comprising 90% to 100% of salmonid catches. At 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 38). This pattern is typical for Welch and Whites Island, but at Franz Lake, higher proportions of marked Chinook salmon have been collected in previous years. At Campbell Slough, the abundance of marked and unmarked Chinook salmon were similar, with 49% of Chinook salmon unmarked, a pattern similar to that observed in previous sampling years (Figure 38). In addition to Chinook salmon, small numbers of coho and sockeye salmon were found, as well as one cutthroat trout at Campbell Slough. No chum salmon were caught in 2014.

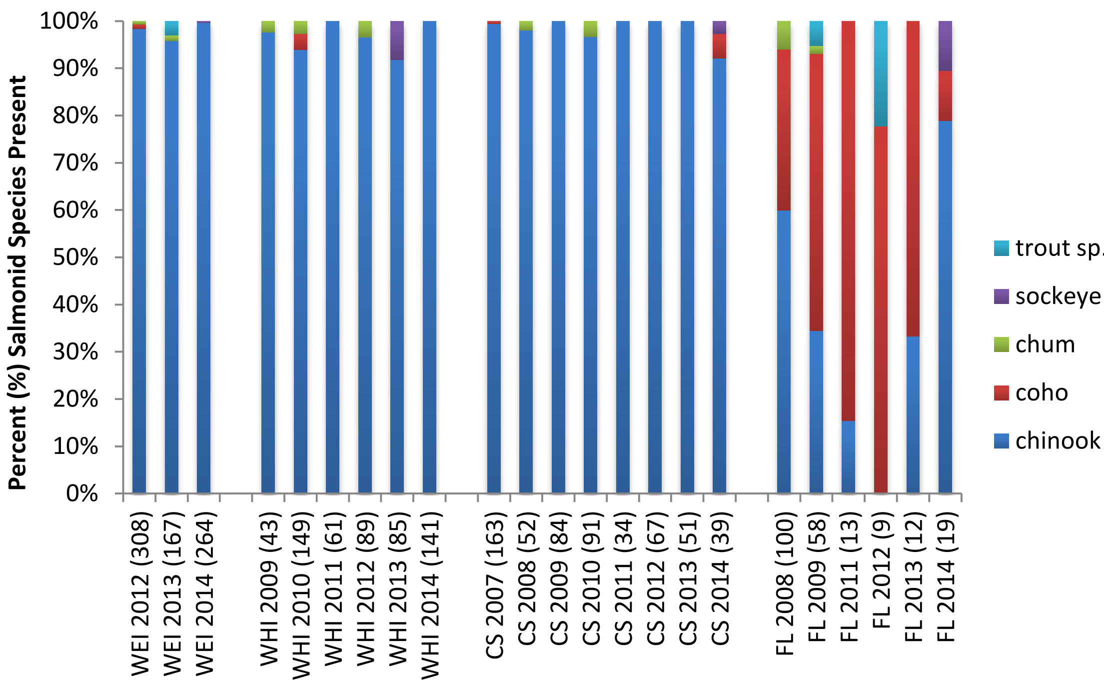


Figure 37. Percentage of salmonid species collected at EMP sampling sites in 2014, as compared to percentages collected in previous sampling years. Total number of salmonids captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

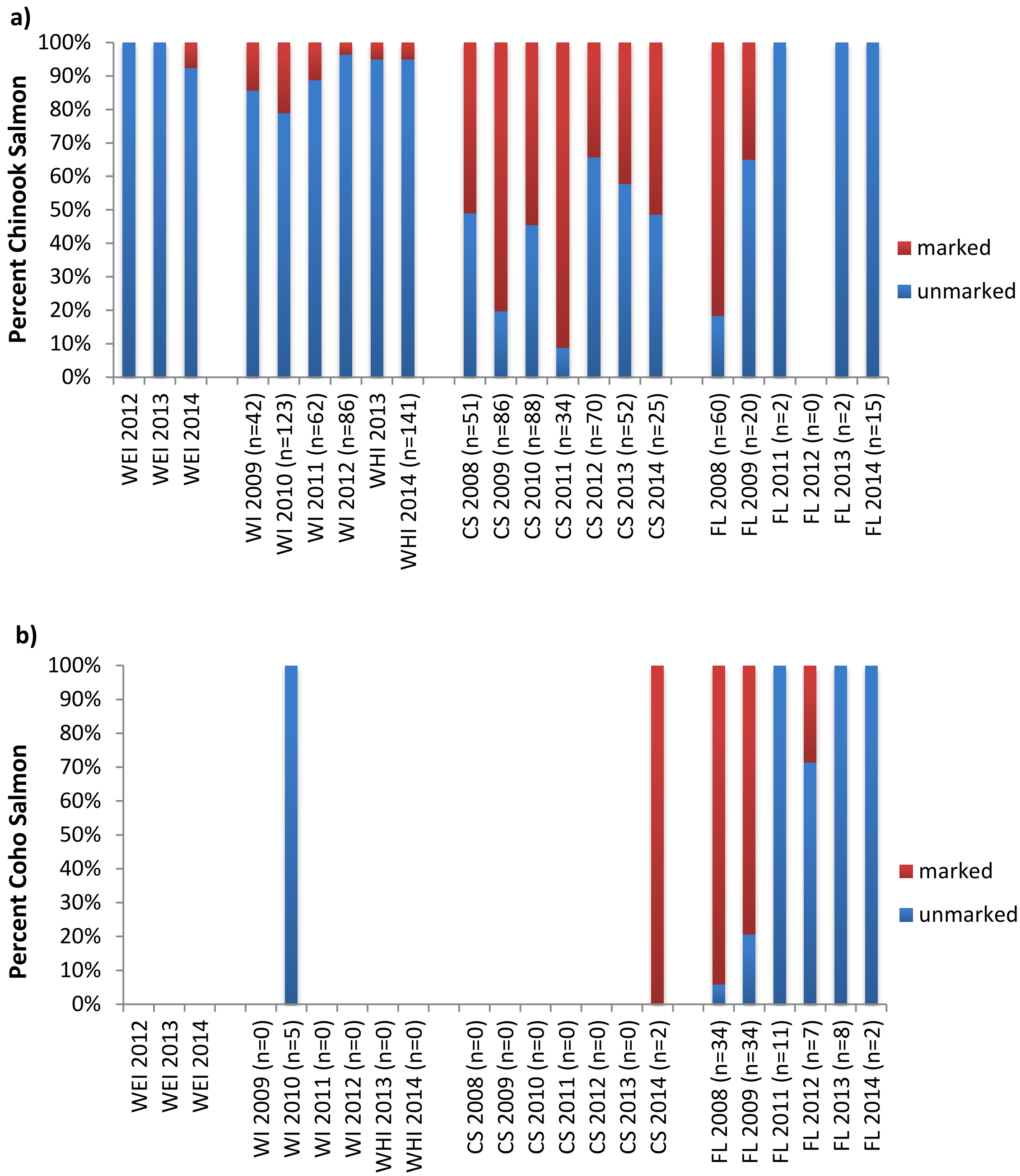


Figure 38. Percentage of marked and unmarked a) Chinook salmon and b) coho salmon captured at the EMP sampling sites in 2014, as compared to previous sampling years. Total number of the salmon species captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

4.5.1.2 *Salmon Density*

Chinook salmon. In 2014, unmarked Chinook salmon were found at the EMP sampling sites from February, when sampling began, through July, when sampling ceased. The highest average density of juvenile Chinook salmon was 68.8 fish per 1000 m^2 in April (Figure 39). Marked Chinook salmon were found in March, April, May, and July, with the highest average density of 13.2 fish per 1000 m^2 in July (Figure 39). Mean Chinook salmon densities by site and year are shown in Figure 40. The density of unmarked Chinook salmon was highest at Welch Island, and lowest at Campbell Slough, with intermediate values at the other sites. Densities of unmarked Chinook salmon in 2014 were generally within the same range as previous years. The densities of marked Chinook salmon in 2014 were generally lower than the densities of unmarked Chinook salmon, with the highest value observed at Welch Island. At Welch Island, densities of unmarked Chinook salmon tended to increase from 2012 to 2014, while at Campbell Slough, Whites Island, and Franz Lake, densities of marked Chinook salmon were in the lower range of observed values during annual sampling periods.

Coho salmon. Coho salmon densities by site and year are shown in Figure 41. Only four coho salmon were caught in 2014, two marked coho in May (1.07 fish per 1000 m^2) at Campbell Slough in Reach F and two unmarked coho (1.41 fish per 1000 m^2) at Franz Lake in November. Coho salmon have been captured only sporadically at Welch Island and Whites Island, so their absence in 2014 was not unusual as compared to previous years. At Campbell Slough, 2014 represents the first year that coho salmon have been captured since systematic sampling for salmon density began in 2008.

Chum salmon. Chum salmon were not caught in 2014, thus chum salmon densities at all sites in 2014 were low relative to previous years (Figure 41).

Sockeye salmon. Sockeye salmon were present at three of the four EMP sampling sites in 2014, Welch Island, Campbell Slough, and Franz Lake (Figure 8). They were not captured at Whites Island in 2014, although they were present at that site in 2013. Sockeye salmon were captured at the EMP sampling sites in April and May, with the highest density (4.41 fish per 1000 m^2) observed in April (Figure 41). Although densities were low (ranging from 0.53 to 4.42 fish per 1000 m^2, respectively), they represent an increase from years prior to 2013, when sockeye salmon were not observed at all (Figure 41).

Trout species. Similar to most previous sampling years, trout densities were very low in 2014 (Figure 41). Only one cutthroat trout was detected at Campbell Slough (0.08 fish per 1000 m^2). In past years, trout were also observed at Welch Island (0.58 fish per 1000 m^2) and Franz Lake (0.78-1.16 fish per 1000 m^2), but they were not observed at these sites in 2014.

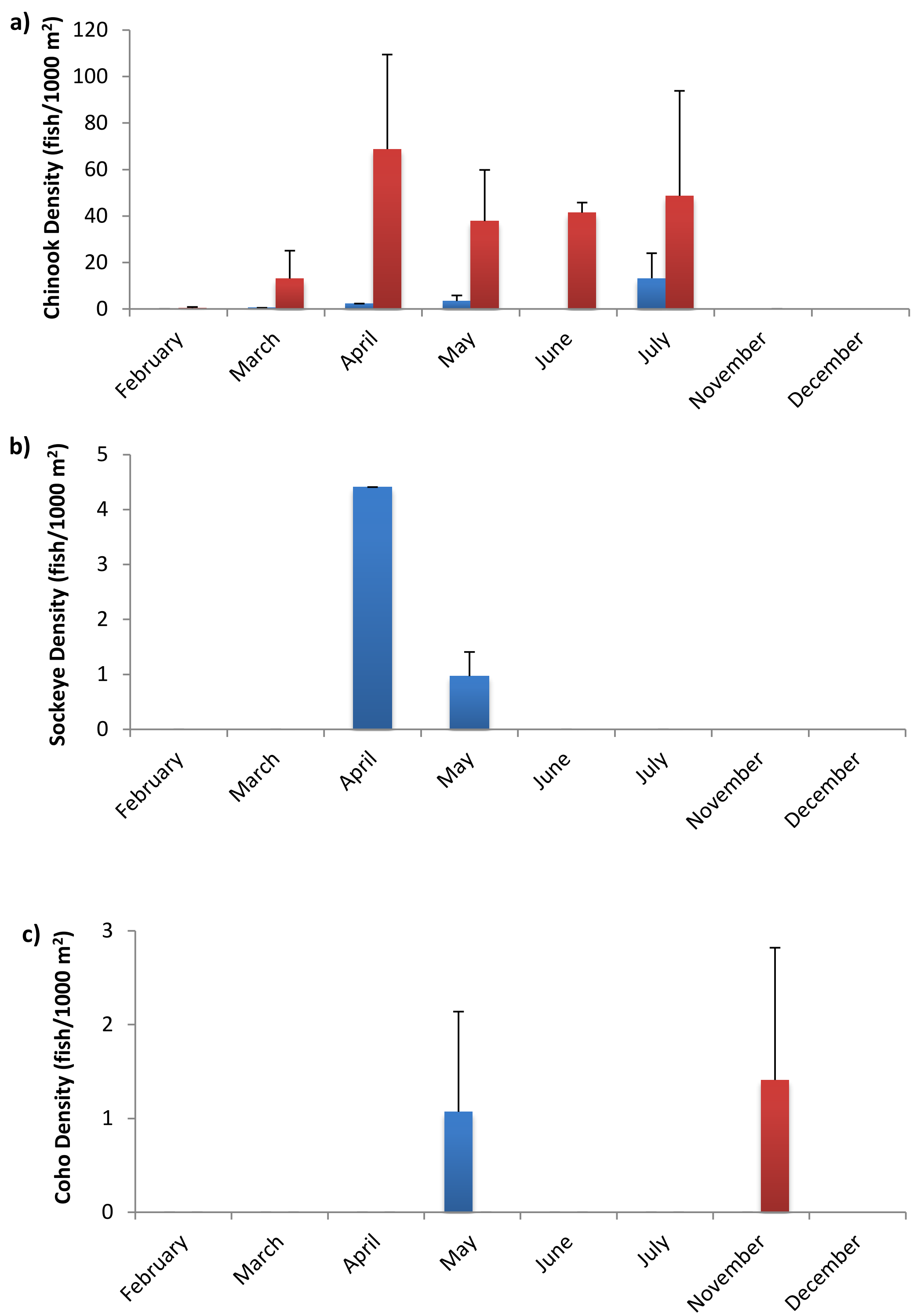


Figure 39. Marked (blue bars) and unmarked (red bars) juvenile a) Chinook salmon, b) sockeye salmon, and c) coho salmon densities (fish per 1000 m^2) by month during the 2014 sampling year.

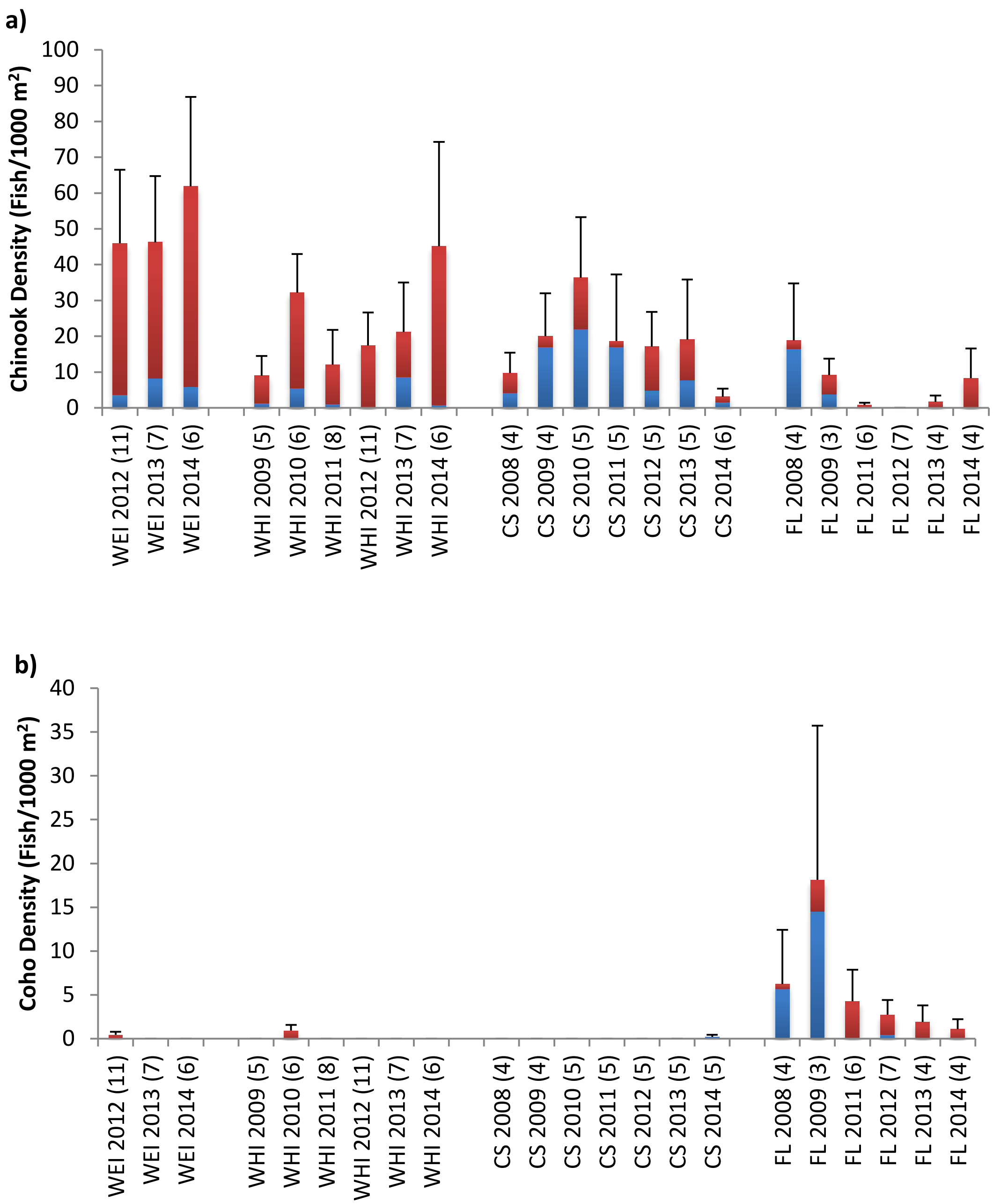


Figure 40. Marked (blue bars) and unmarked (red bars) juvenile a) Chinook salmon and b) coho salmon densities (fish per 1000 m^2) by trends site and year. Total number of salmonids captured per year at a site are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

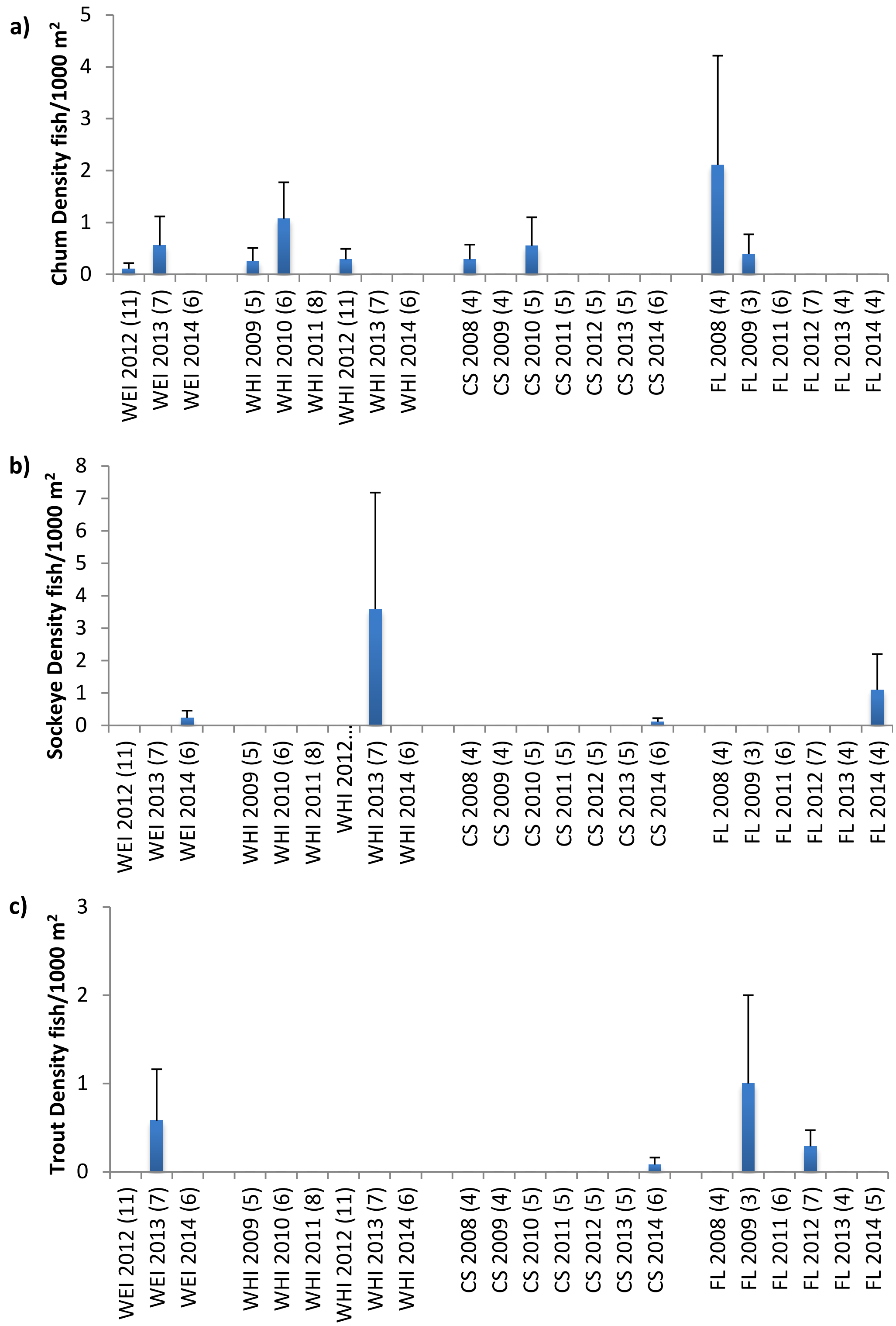


Figure 41. Juvenile a) chum salmon, b) sockeye salmon, and c) trout densities (fish per 1000 m^2) by year at trends sites. Total number of salmonids captured per year at a site are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

4.5.2 Salmon Metrics

4.5.2.1 *Genetic Stock Identification*

In this report we present the Chinook salmon genetic stock data collected in 2013, as genetic samples from the 2014 sampling year are currently undergoing analysis. In 2013, genetics data were collected from Chinook salmon at four of the six trends sites (Secret River, Welch Island, Whites Island, and Campbell Slough). Too few Chinook salmon were collected at Ilwaco Slough or Franz Lake to allow for genetic stock identification. Among unmarked fish West Cascades Fall Chinook were the most abundant stock at Secret River, Welch Island, and Whites Island, with Upper Columbia Fall Chinook becoming more prominent at Campbell Slough (Figure 42). Spring Creek Group Fall Chinook, as well as interior stocks such as Deschutes River Fall Chinook and Snake River Fall Chinook were also captured at the trends sites. At Secret River, some fish with genetic similarities to Rogue River Chinook were identified as well. The stocks present at the trends sites were generally similar over the sampling years, with no major changes in stock composition.

Among the marked fish Spring Creek Group Fall and West Cascades Fall Chinook made up the majority of fish captured at most sites in 2013, as in previous sampling years (Figure 42). Other stocks observed in small numbers included West Cascades Spring Chinook, Willamette River Spring Chinook, and Snake River Fall Chinook. The stock composition at the sampling sites did not vary greatly from year to year, aside from some differences in the presence or absence of the more rare stocks.

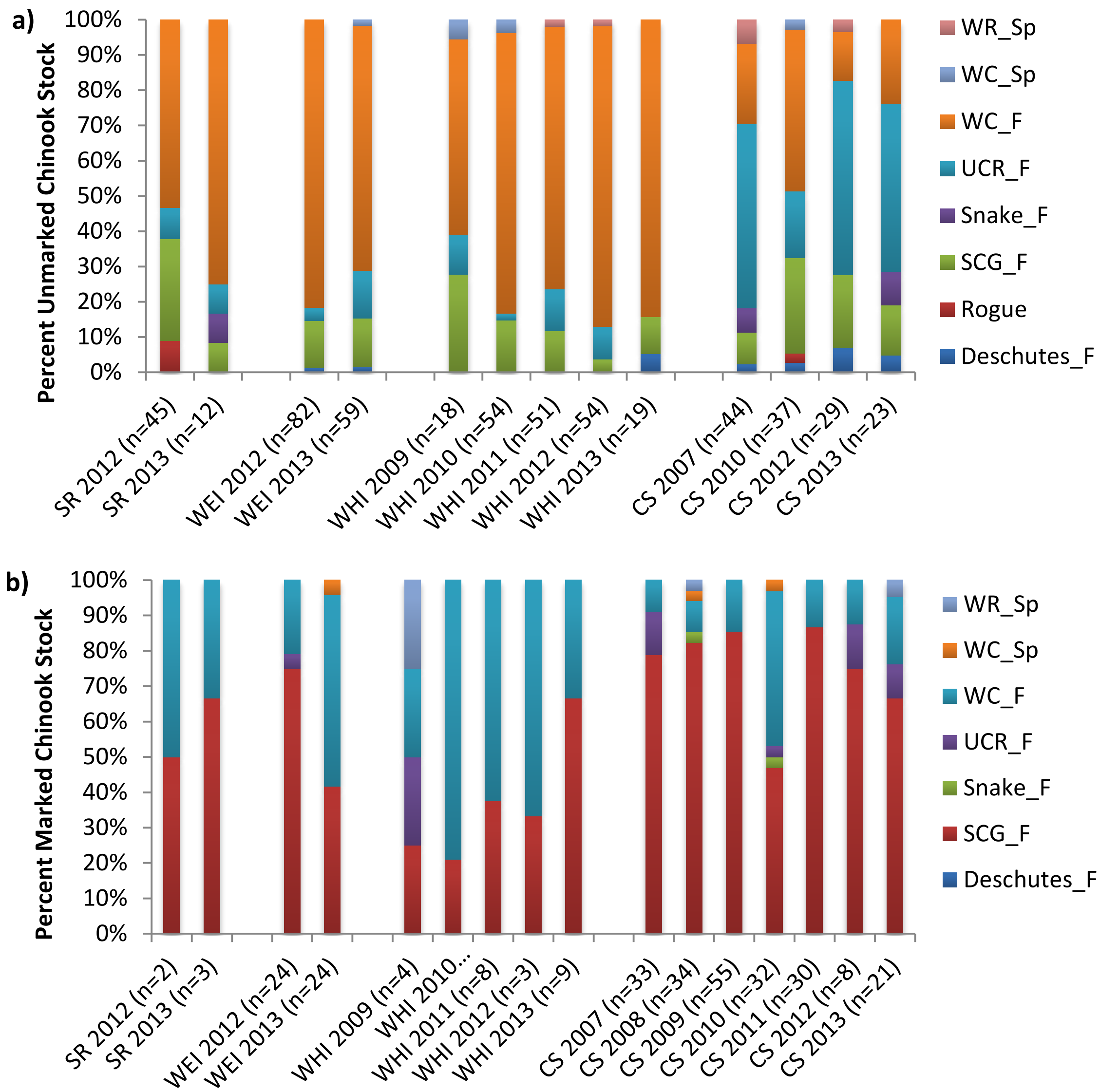


Figure 42. Genetic stock composition of a) unmarked and b) marked Chinook salmon at the trends sites in 2013, as compared to previous years. Sample sizes for each stock are presented in parentheses. Franz Lake and Ilwaco Slough are not shown, as no new data are available from these sites for temporal comparison. SR = Secret River, 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.

4.5.2.2 *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 2014, with the largest fish typically captured at Campbell Slough (Figure 43). Within sites, there was some variation by year. Significant differences in length among years were observed at both Campbell Slough ($p = 0.0004$) and Franz Lake ($p = 0.0026$), with fish being significantly larger at Campbell Slough and significantly smaller at Franz Lake in 2014 than in other years (Tukeys multiple range test, $p < 0.05$). At Welch Island and Whites Island, significant differences in length among years were also observed ($p < 0.0001$ for both sites), but the 2014 values were not especially high or low in comparison to other years. Significant differences in weight among years were also observed at both Campbell Slough ($p < 0.0001$) and Franz Lake ($p = 0.0272$). Fish were heavier at Campbell Slough and lighter at Franz Lake in 2014 than in other years (Tukeys multiple range test, $p < 0.05$). At Welch Island and Whites Island, significant differences in fish weight among years were also observed ($p < 0.0001$ for both sites); however, as with weight, 2014 values were not especially high or low in comparison to other years. Differences in condition factor were observed among years for Welch Island ($p = 0.0005$), Whites Island ($p < 0.0001$) and Campbell Slough ($p = 0.0002$), but not Franz Lake ($p = 0.1772$). However, the 2014 values were not especially high or low in comparison to other years at any of the sampling sites.

Too few marked Chinook salmon were caught at Welch Island, Whites Island, or Franz Lake in 2014 (or prior years) to allow for temporal comparison. At Campbell Slough, however, both length and weight varied with sampling year ($p < 0.0001$ for both metrics), with both length and weight being higher in 2014 than in other years (Figure 44). Condition factor also varied significantly among years ($p = 0.0001$), but the value in 2014 was not especially high or low in comparison to other years.

Size class distribution. At the trends sites in 2014, the majority of unmarked Chinook salmon were fry (71%), 28% were fingerlings, and less than 1% were yearlings (Figure 45). At Welch Island, fry predominated, making up 85% of unmarked Chinook salmon and fingerlings making up the remaining 15%. At Whites Island, proportions of fry and fingerlings were more comparable, with fry accounting for 60% and fingerlings for 40% of unmarked Chinook. At Campbell Slough, fingerlings predominated, making up 78% of the catch, while fry and yearlings each made up 11% of the catch. Campbell Slough was the only site where unmarked yearling Chinook salmon were observed, and this is the first year they have been observed at this site. At Franz Lake, all of the 15 unmarked Chinook that were caught in 2014 were fry. In comparison to previous years, the percentage of fry at Franz Lake was higher than usual, but somewhat lower than typically observed at Campbell Slough. At the Welch Island and Whites Island, the proportion of fry in 2014 was comparable to previous years.

Of the 31 marked Chinook salmon caught at the trends sites in 2014, 81% were fingerlings and the remaining 19% were yearlings (Figure 45). Marked Chinook salmon were not caught at Franz Lake and only one (a fingerling) was captured at Whites Island. At Welch Island, 71% of the marked Chinook salmon collected were fingerlings and 29% were yearlings. At Campbell Slough, 88% were fingerlings and 12% were yearlings. In comparison to previous sampling years, the proportion of yearlings encountered was greater in 2014 at Campbell Slough than during past years. At Welch Island, Whites Island, and Franz Lake, too few marked Chinook salmon were caught for temporal comparison.

Other salmon species

Only four coho salmon were caught in 2014, two marked fish from Campbell Slough in May, and two unmarked fish from Franz Lake in November. 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 (± SD) were 144.5 ± 0.6 mm; 27.6 ± 1.5 g; and 0.91 ± 0.06, respectively. The average length, weight, and condition factor of fish captured at Franz Lake (± SD) were 104.5 ± 4.9 mm; 12.1 ± 0.8 g; and 1.06 ± 0.08, respectively. Chum salmon were not caught at any of the trends sites in 2014. Four sockeye salmon were captured and measured in 2014 and the average length, weight, and condition factor of these fish (± SD) were 46.5 ± 9.3 mm; 0.8 ± 0.6 g; and 0.78 ± 0.08, respectively. One cutthroat trout was captured, which had a length, weight and condition factor of 160 mm, 42 g, and 1.03, respectively.

Franz Lake was the only site where coho have been caught consistently enough to compare size measurements by sampling year, and even at this site, only unmarked coho salmon were caught in all sampling years including 2014. Mean length, weight, and condition factor are shown for unmarked coho salmon from Franz Lake in Figure 46. Mean length (± SD) varied from 82 ± 11 mm in 2013 to 120 ± 34 mm in 2009, and the fish collected in 2014 being of intermediate size (105 ± 5 mm); length did not differ significantly among sampling years ($p = 0.154$). Weight, however, differed among years ($p = 0.0053$), with the highest values in 2009 and the lowest in 2013. Again, the weight of fish collected in 2014 was intermediate compared to other years. Condition factor did not show significant differences among sampling years, but the lowest values was observed in 2008 and the highest in 2014.

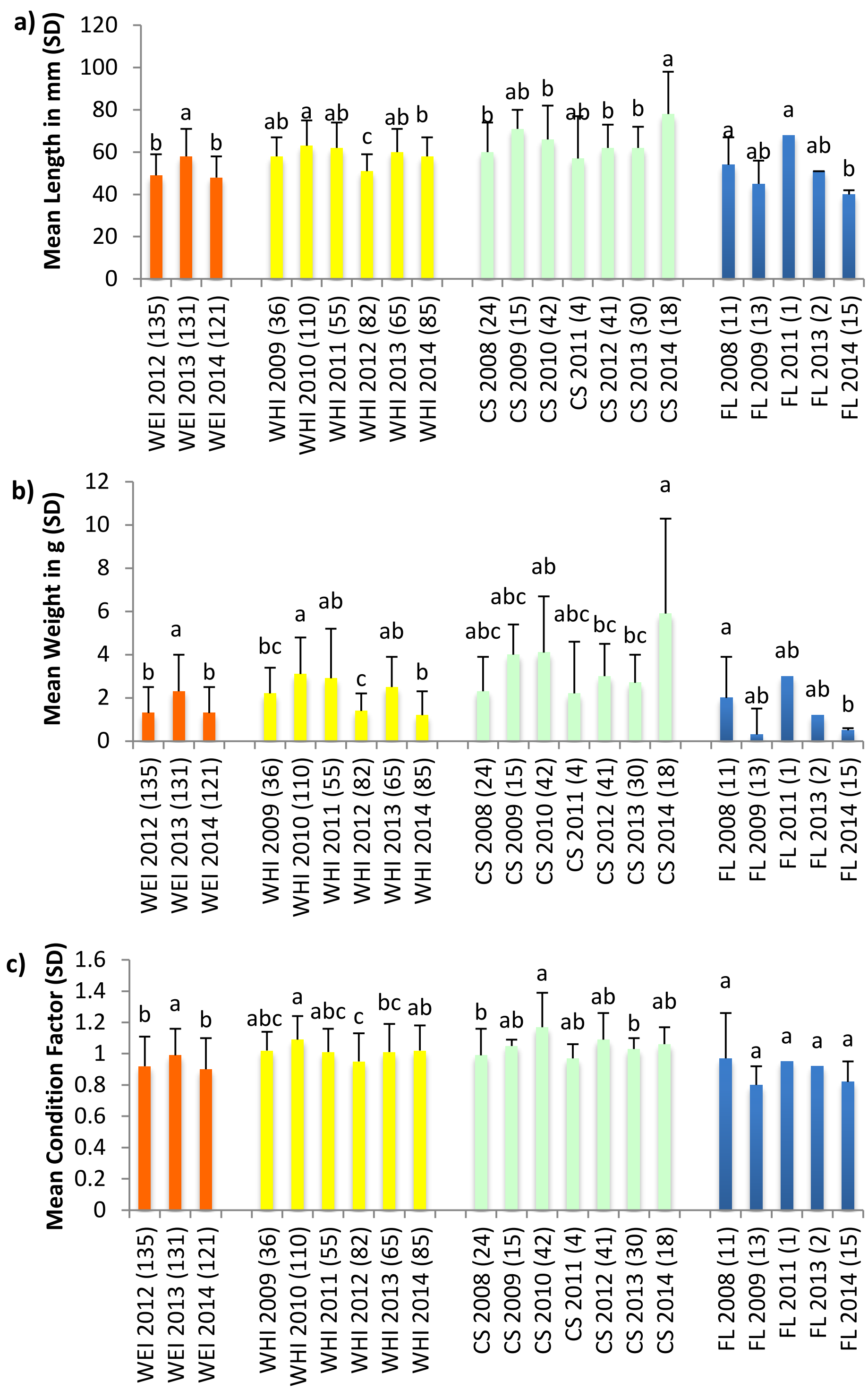


Figure 43. Mean (SD) a) length (mm), b) weight (g), and c) condition factor (± SD) of unmarked juvenile Chinook salmon at trends sites in 2014 as compared to previous years. Within the sites, values with different letter superscripts are significantly different (Tukey's multiple range test, $p < 0.05$). Total number of Chinook salmon captured per year at a site are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

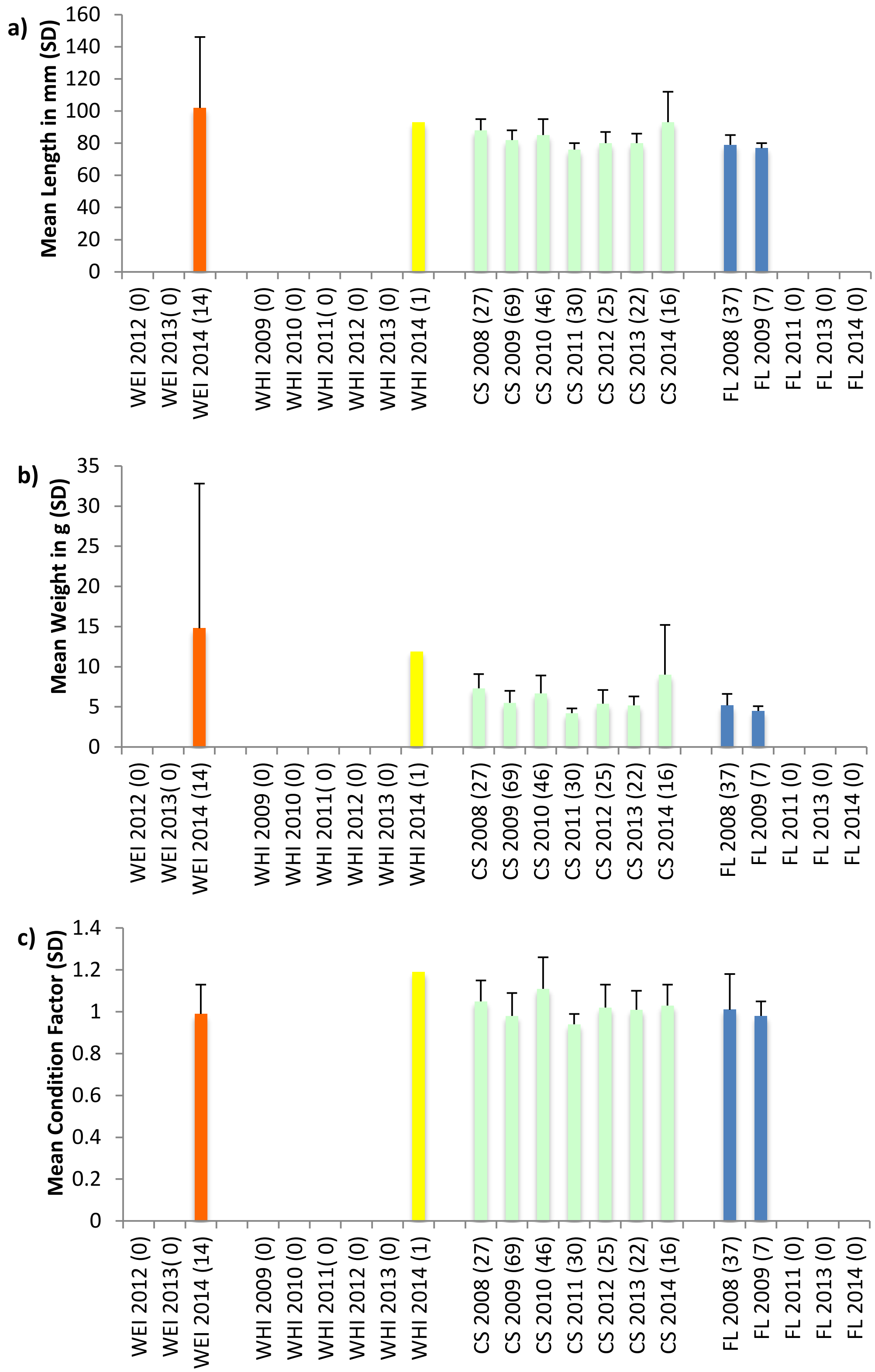


Figure 44. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of marked Chinook salmon at trends sites in 2014 and compared to previous sampling years. Total number of Chinook salmon captured per year at a site are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

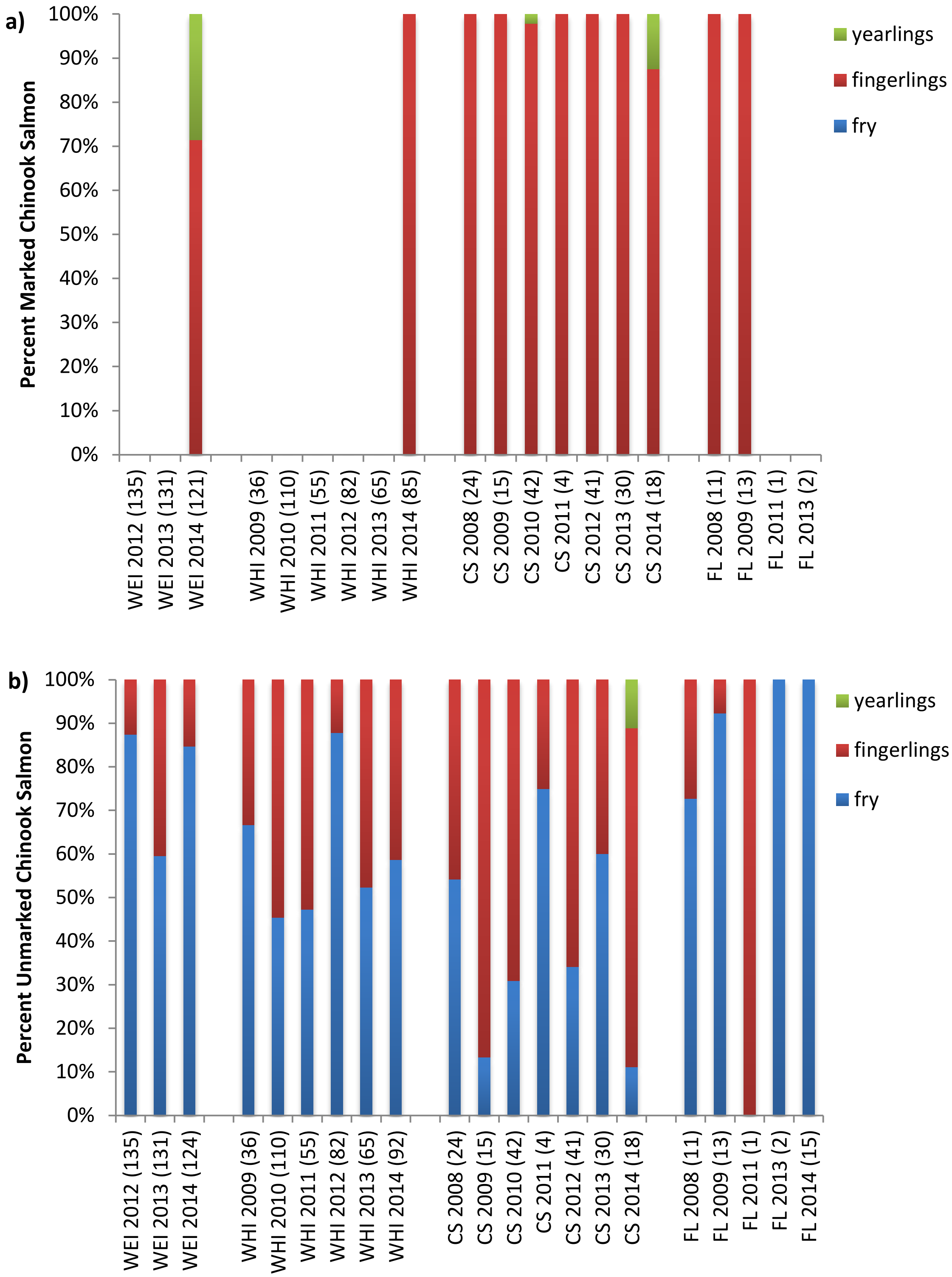


Figure 45. Size class distribution of a) marked and b) unmarked juvenile Chinook salmon captured at trends sites in 2014 and in previous sampling years. Total number of Chinook salmon captured per year at a site are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

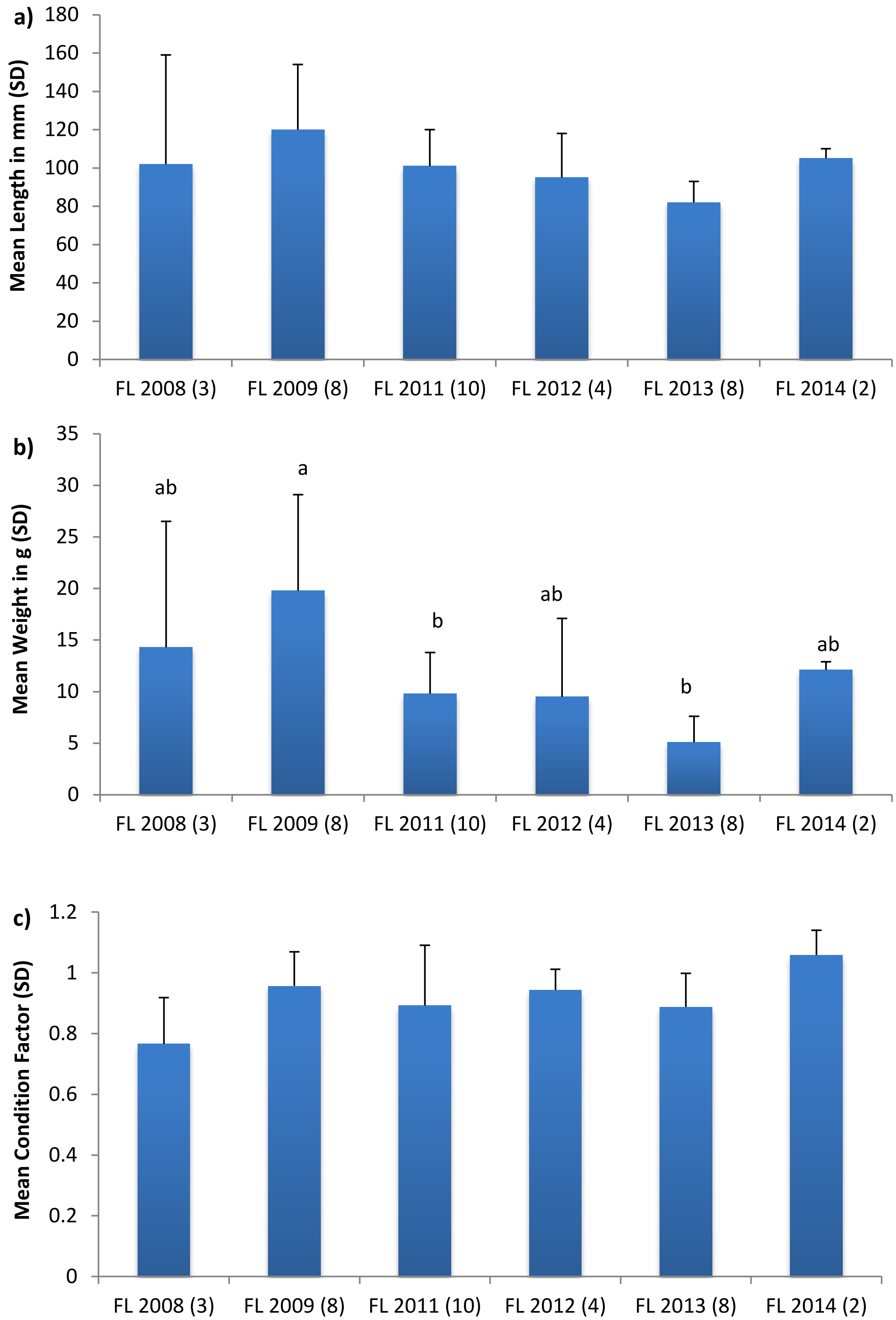


Figure 46. Mean a) length (mm), b) weight (g), and c) condition factor of unmarked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.

4.5.2.3 *Somatic Growth Analyses*

In the current report, growth data derived from otoliths collected from juvenile Chinook salmon between 2007 and 2012 are presented. 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 (Figure 47) and results from 2005 and 2007-2012 sampling years are reported here (Table 30). The number of otoliths collected from different Chinook salmon genetic stocks is presented in Table 31. ANOVA results indicated differences in somatic growth rates among sites ($F_{27,472} = 5.45$; $p < 0.001$); and Bonferroni post-hoc tests indicated differences among several sites (Appendix D, Table D.1; Figure 48). Our temporal analysis indicated that fish from three sites grew at significantly different rates among years (Campbell Slough $F_{1,58} = 19.6$, $p < 0.001$; Confluence Washington $F_{1,21} = 23.7$, $p < 0.001$, Franz Lake $F_{1,14} = 3.9$, $p = 0.067$) and Bonferroni post-hoc tests showed that fish collected at Campbell Slough in 2007 grew faster than those collected in 2011 and 2012 (Figure 49). Our analysis of whether somatic growth rates differed among genetic stocks indicated significant differences ($F_{6, 441} = 4.5$; $p < 0.001$; Figure 50). Bonferroni post-hoc tests determined that growth rates of Spring Creek Group Fall Chinook salmon were significantly greater than West Cascades Fall Chinook salmon ($p < 0.001$), and marginally significant than fish from Deschutes River Fall Chinook salmon ($p < 0.07$). Finally, marked fish grew significantly faster than unmarked fish ($F_{1,498} = 44.4$; $p < 0.001$; Figure 51).

Our analysis to assess how somatic growth rate varies according to seven variables indicated that four models were indistinguishable because they had a delta AIC of < 2 (Appendix D, Table D.2). Shared among all four models was the importance of reach, distance to channel center, sampling year, and whether the fish was marked or unmarked. The best model was the only four-factor model (the other three consisted of five- and six-factor models) such that reach, distance to channel center, marked or unmarked, and sampling year explained the greatest amount of variability in somatic growth rate.

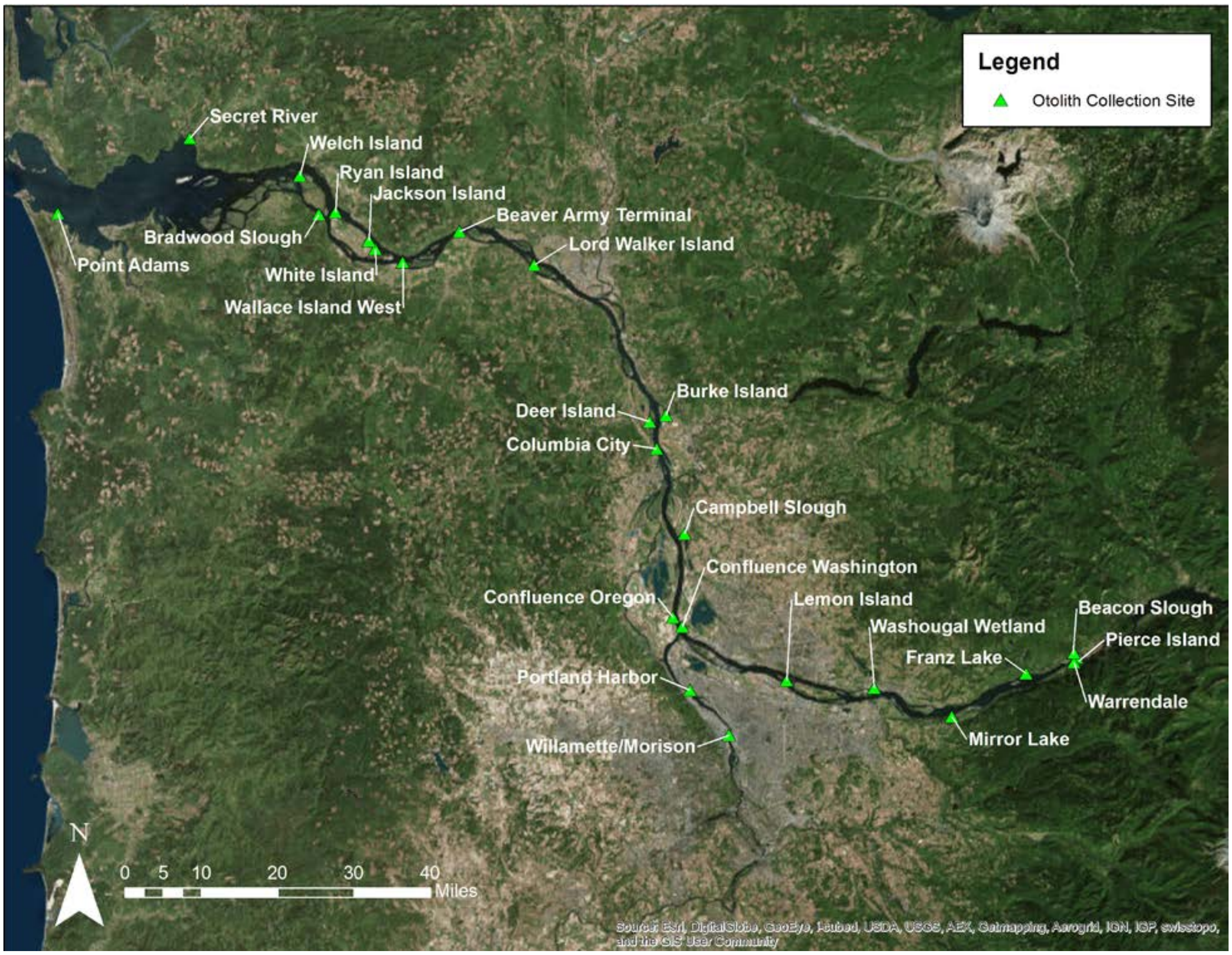


Figure 47. Map of sites where otoliths were collected from juvenile Chinook salmon in 2005 and 2007-2012. The 28 sites in the lower Columbia River included EMP status and trends sites, toxic contaminant monitoring sites, and action effectiveness monitoring sites.

Table 30. Sites and years from which juvenile Chinook salmon were collected and the number of otoliths processed for analysis of somatic growth rates. EMP trends sites are indicated in bold.

Site	Year							
	2005	2007	2008	2009	2010	2011	2012	Total
Beacon Slough			7					7
Beaver Army Terminal	17							17
Bradwood Slough					13			13
Burke Island						6		6
Campbell Slough		**14**	**6**	**11**	**13**	**10**	**6**	**60**
Columbia City	14							14
Confluence Oregon			11					11
Confluence Washington	11		12					23
Deer Island						7		7
Franz Lake			**11**	**5**				**16**
Goat Island						8		8
Jackson Island					13			13
Lemon Island							21	21
Lord Walker Island				4				4
Mirror Lake 1 (Lake)			6	10	2	9	12	39
Mirror Lake 4 (Culvert)			10	6	12	4	17	49
Pierce Island			5					5
Point Adams	16							16
Portland Harbor	16							16
Ryan Island				10				10
Sand Island			6					6
Secret River							**17**	**17**
Wallace Island West					14			14
Warrendale	12							12
Washougal							16	16
Welch Island							**21**	**21**
Whites Island				**8**	**14**	**9**	**17**	**48**
Willamette/Morrison	11							11
Total	97	14	74	54	81	53	127	500

Table 31. Genetic stocks for juvenile Chinook salmon collected from EMP sites for which growth rates from otoliths are available.

Genetic Stock	Code	Mean Fork Length (mm)	Sample Size
Deschutes River Fall	Desch_F	59.9	8
Interior Columbia River Spring	Int_Sp	67.0	1
Interior Columbia River Summer/Fall	Int_Su/F	70.6	12
Rogue River	Rogue	70.0	1
Snake River Fall	Snake_F	64.0	6
Spring Creek Group Fall	SCG_F	74.9	160
Upper Columbia River Summer/Fall	UCR_Su/F	63.9	71
Upper Willmette River Spring	WR_Sp	67.5	21
West Cascades Fall	WC_F	62.6	170
West Cascades Spring	WC_Sp	55.0	1
Not Assigned	N/A	63.5	49
	Total	**718.9**	**500**

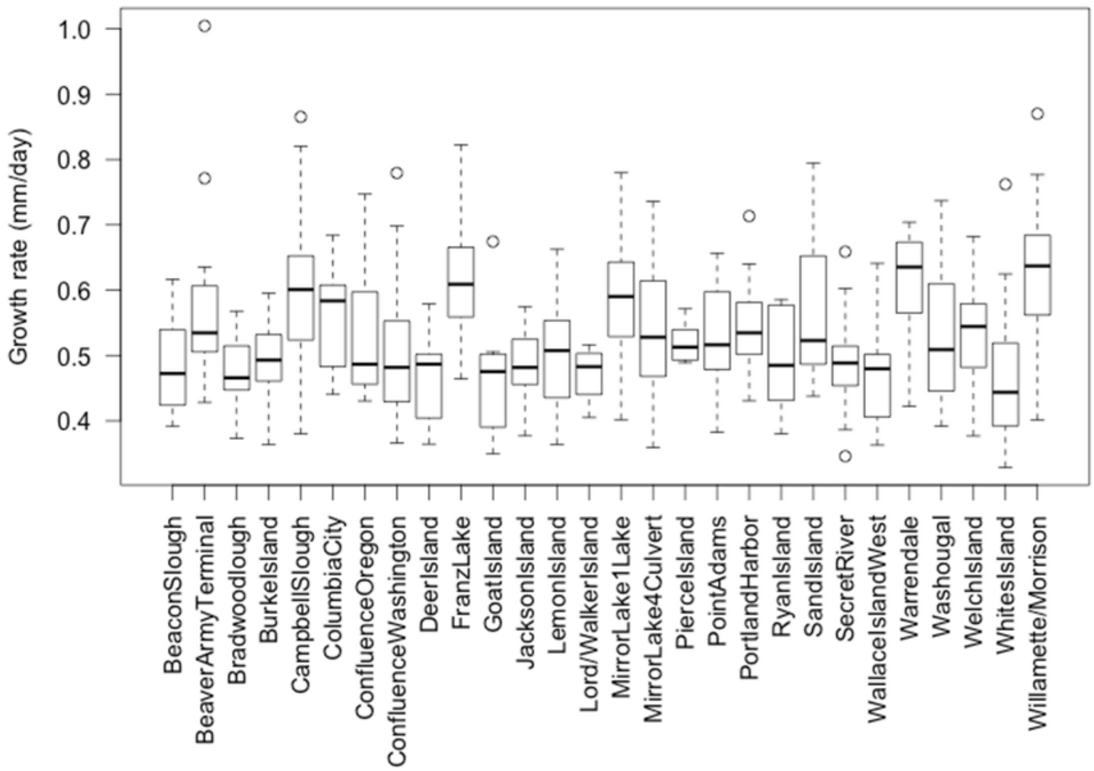


Figure 48. Somatic growth rates (mm/day) of fall Chinook salmon across monitoring sites. Bold lines represent median, box ends represent standard error, whiskers indicate standard deviation, and open circles represent outliers.

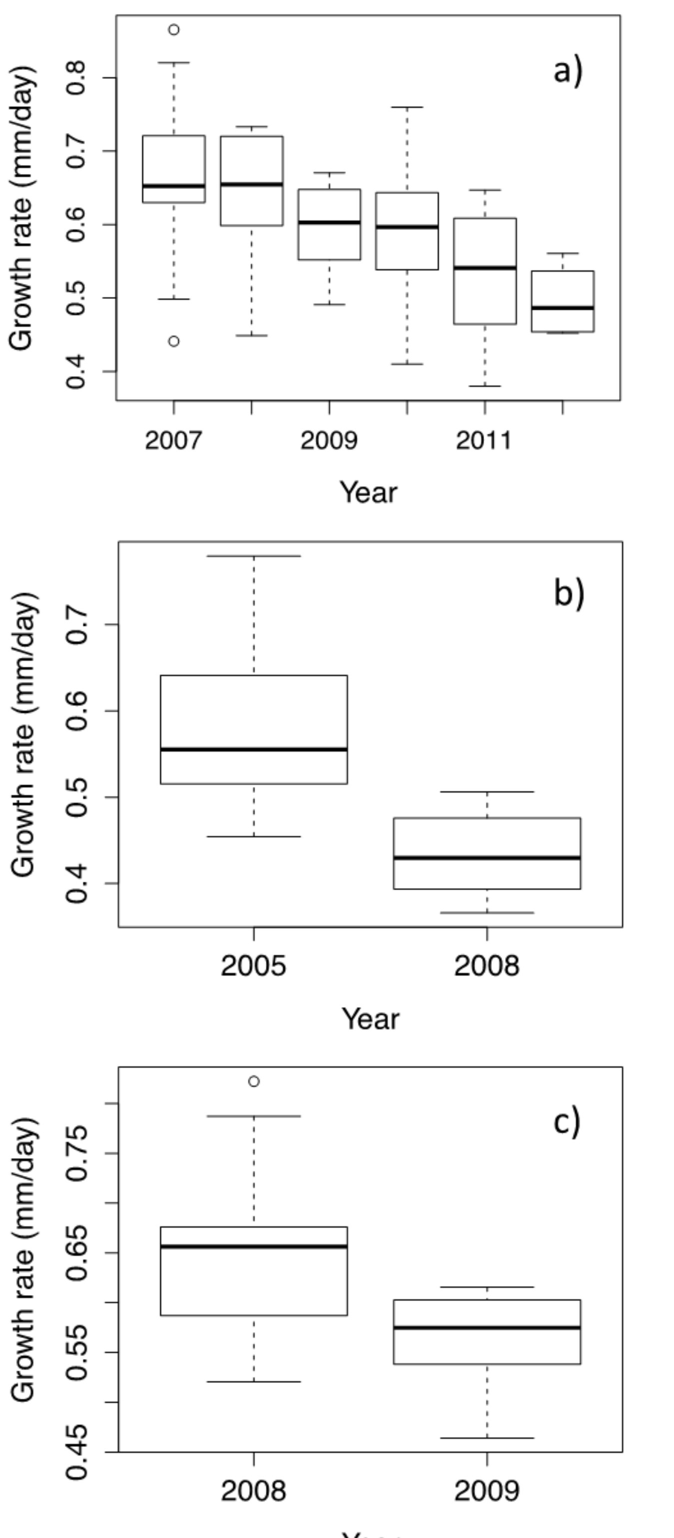


Figure 49. Growth rates (mm/day) for a) Campbell Slough, b) Confluence Washington, and c) Franz Lake that were sampled in multiple years and showed significant differences in growth rate among those years. Franz Lake showed marginal significant differences p=0.067). Thick lines represent median, boxes are quartiles, whiskers are minimum and maximum values, and circles are outliers.

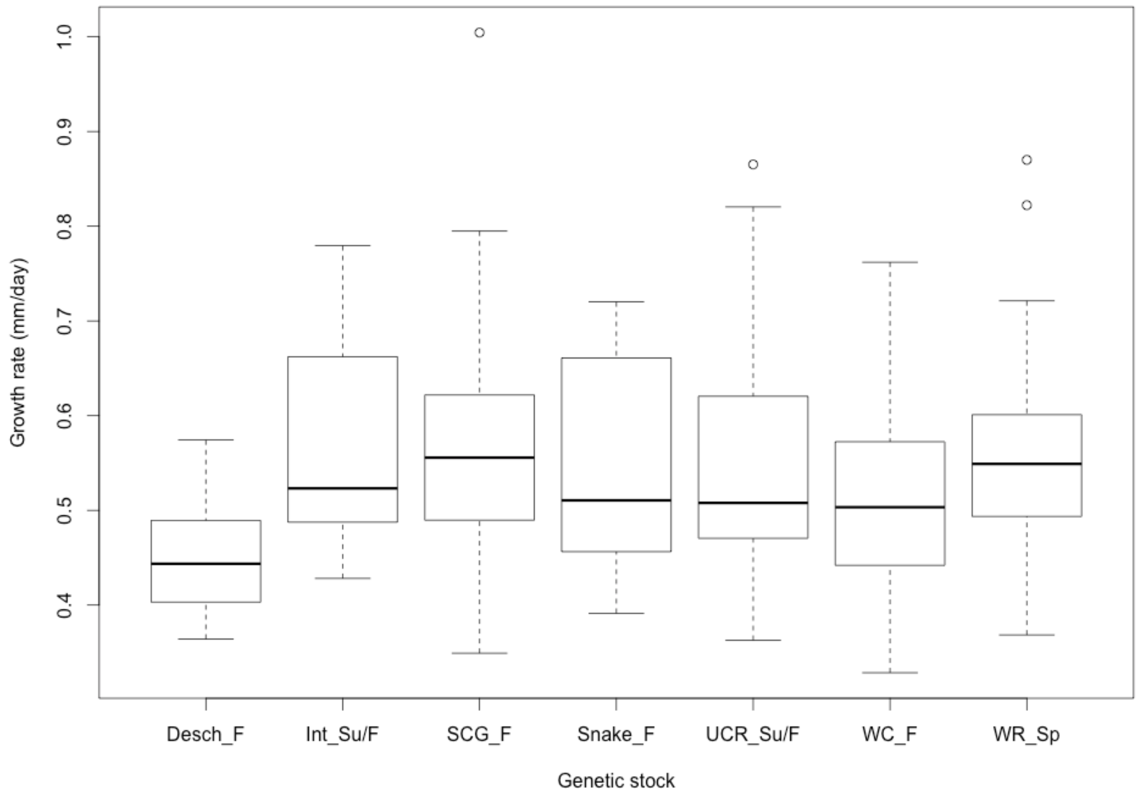


Figure 50. Mean juvenile Chinook salmon growth rate (mm/day) among seven genetic stocks (these seven stocks had samples sizes greater than n = 6; see Table 31). Bold lines represent median values, box ends represent standard error, whiskers indicate standard deviation, and circles represent outliers. Desch_F = Deschutes River Fall, Int_Su/F = Interior Columbia River Summer/Fall, Snake_F = Snake River Fall, UCR_Su/F = Upper Columbia River Summer/Fall, WC_F = West Cascades Fall, WR_Sp = Willamette River Spring.

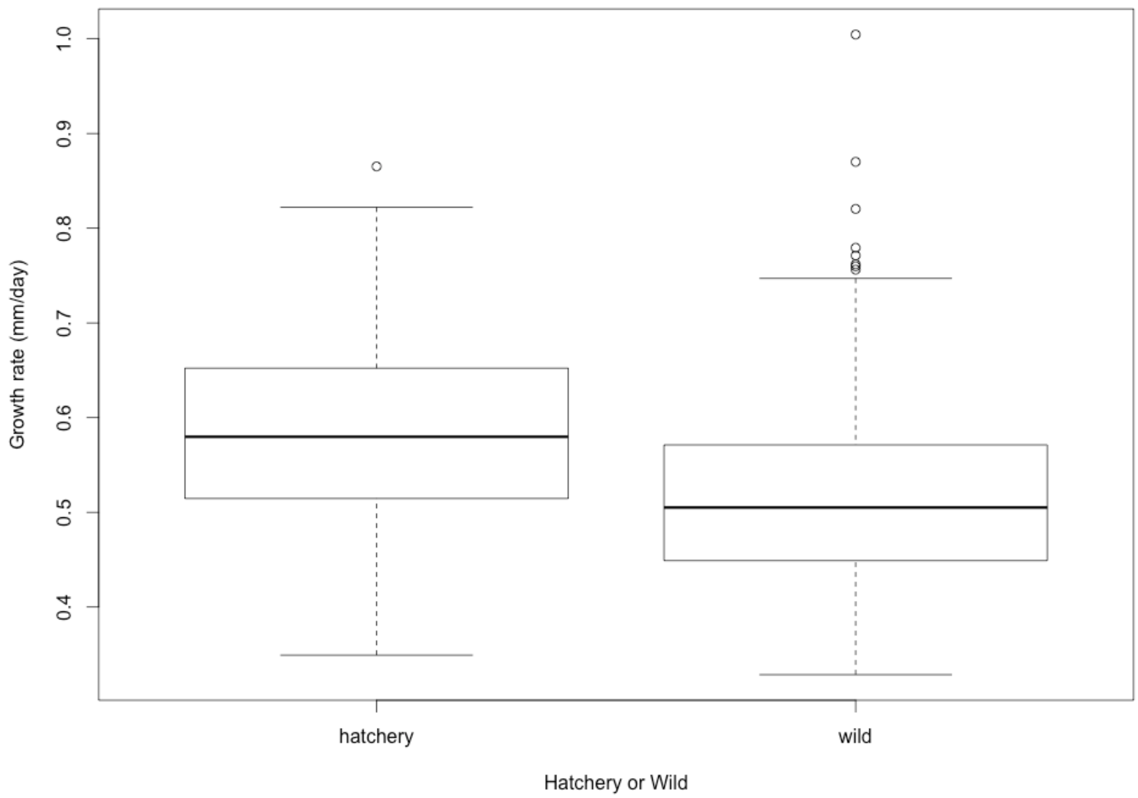


Figure 51. Mean growth rate (mm/day) for unmarked (wild, n=354) and marked (hatchery, n=146) Chinook salmon when all sites and years are pooled. Bold lines represent median values, box ends represent standard error, whiskers indicate standard deviation, and circles represent outliers.

4.5.2.4 *Lipid Content of Juvenile Chinook Salmon*

In the current report, lipid data collected from juvenile Chinook salmon between 2007 and 2013 will be presented. Lipid samples from 2014 are currently being analyzed and will be presented in a future report. The average lipid content of juvenile Chinook salmon collected in 2013 was 1.99 ± 0.96% for unmarked fish (n=19) and 2.00 ± 0.97% for marked fish (n=15). For both groups of fish, these are among the higher values observed since sampling began in 2007. Mean lipid content for marked fish ranged from 0.97% in 2012 to 2.02% in 2007, while unmarked fish lipid content has ranged from 1.00% in 2011 to 2.36% in 2008. Among 2013 sites, mean lipid content was highest in fish captured at Whites Island (3.5%) with mean values at the other sites ranging from 1.58% at Secret River to 1.79% at Welch Island (Figure 52).

Lipid content differed from year to year at Welch Island, Whites Island, and Campbell Slough (Figure 52). At Welch Island and Whites Island, the mean lipid content of salmon body samples was significantly higher in 2013 than in other years. At Secret River, the 2013 lipid values were higher than in 2012, although as only one sample was analyzed in 2013, differences could not be reliably evaluated. The high mean lipid content of the samples from Whites Island was especially unusual and not typical of the range of values seen in previous years. At Campbell Slough, the lipid content of the 2013 samples was comparable to levels measured previously, with no clear increasing or decreasing trends. At Franz Lake, data are available only for 2008 and 2009, thus, trends cannot be assessed.

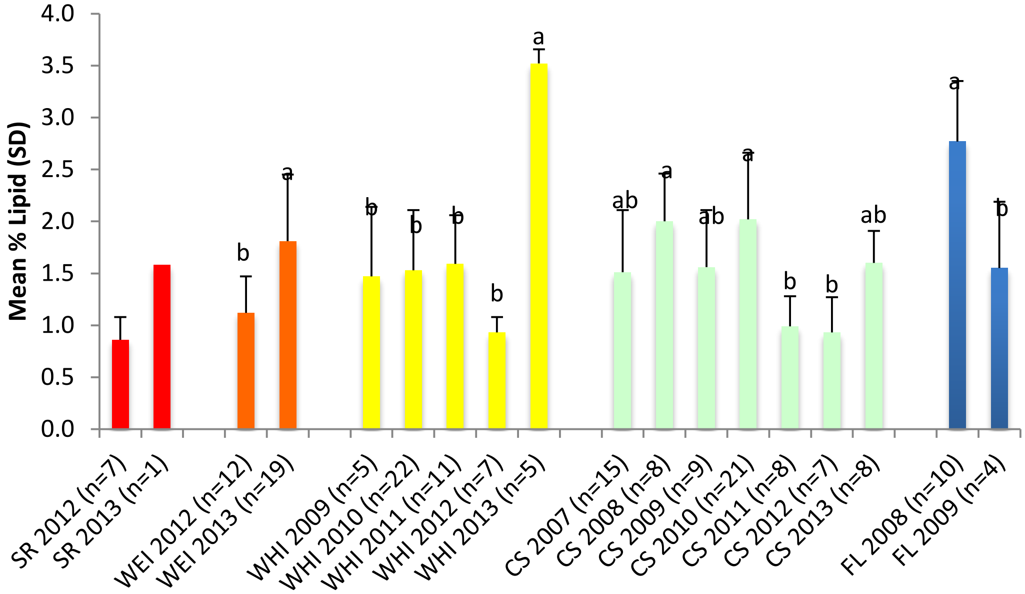


Figure 52. Mean lipid content (SD) of juvenile Chinook salmon from trends sites in 2013 as compared to earlier years. Samples from marked and unmarked fish are pooled as there was no consistent difference between these groups. Sample sizes presented in parentheses indicate the number of composite samples analyzed. Each composite is made up of 3-5 individual fish. Within each site, values with different letter superscripts are significantly different (ANOVA and Tukey's multiple range test, $p < 0.05$). SR = Secret River, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

4.5.2.5 *Contaminants in Juvenile Chinook Salmon*

In this report contaminant data from 2007-2013 will be presented, the samples from 2014 are currently being analyzed and data will be presented in a subsequent report. Overall, mean concentrations of the persistent organic pollutants DDTs, PCBS, and PBDEs were significantly different in juvenile Chinook salmon from the various trends sites (Figure 53). The mean concentration of DDTs was lowest at Franz Lake (560 ng/g lipid) and highest at Whites Island (1310 ng/g lipid), with intermediate values at the other sampling sites (1040-1060 ng/g lipid). In the case of PCBs, highest values were observed at Campbell Slough (1630 ng/g lipid) and lowest values at Franz Lake (300 ng/g lipid), with the other sites being intermediate (1060-1190 ng/g lipid). Concentrations of PBDEs were highest at Whites Island (740 ng/g lipid) with concentrations at Welch Island and Secret River in a similar range (600-640 ng/g lipid). Concentrations of PBDEs were significantly lower at Campbell Slough (270 ng/g lipid) and Franz Lake (31 ng/g lipid).

Within the sites, concentrations of these three classes of contaminants generally tended to be low in 2013 relative to previous years (Figure 53). Concentrations of DDTs, PCBs, and PBDEs were significantly lower at Welch Island and Whites Island than in previous years (Tukey's multiple range test, $p < 0.05$). Similarly, at Campbell Slough, concentrations of these contaminants in 2013 were among the lowest levels observed. At Franz Lake temporal comparisons were not possible, as data are available for 2008 and 2009 only.

In addition to DDTs, PCBs, and PBDEs, PAHs were measured in salmon bodies in 2013, and in 2010-2012 (Figure 54). Overall, mean concentrations of PAHs were highest in juvenile Chinook salmon from Campbell Slough (44 ng/g wet wt) and lowest at Secret River (12 ng/g wet wt), with intermediate concentrations at Welch Island and Whites Island (both 27 ng/g wet wt). The mean PAH concentration in salmon from Campbell Slough was significantly higher than concentrations at the other sites (Tukey's multiple range test $p < 0.05$). No data are available for Franz Lake, as too few Chinook salmon were collected for analyses to be performed. Within the sites, concentrations of PAHs in bodies of juvenile Chinook salmon did not vary by sampling year at Secret River, Welch Island, or Whites Island, but at Campbell Slough, concentrations were significantly lower in 2013, as well as 2012, than in previous sampling years (Tukey's multiple range test, $p < 0.05$).

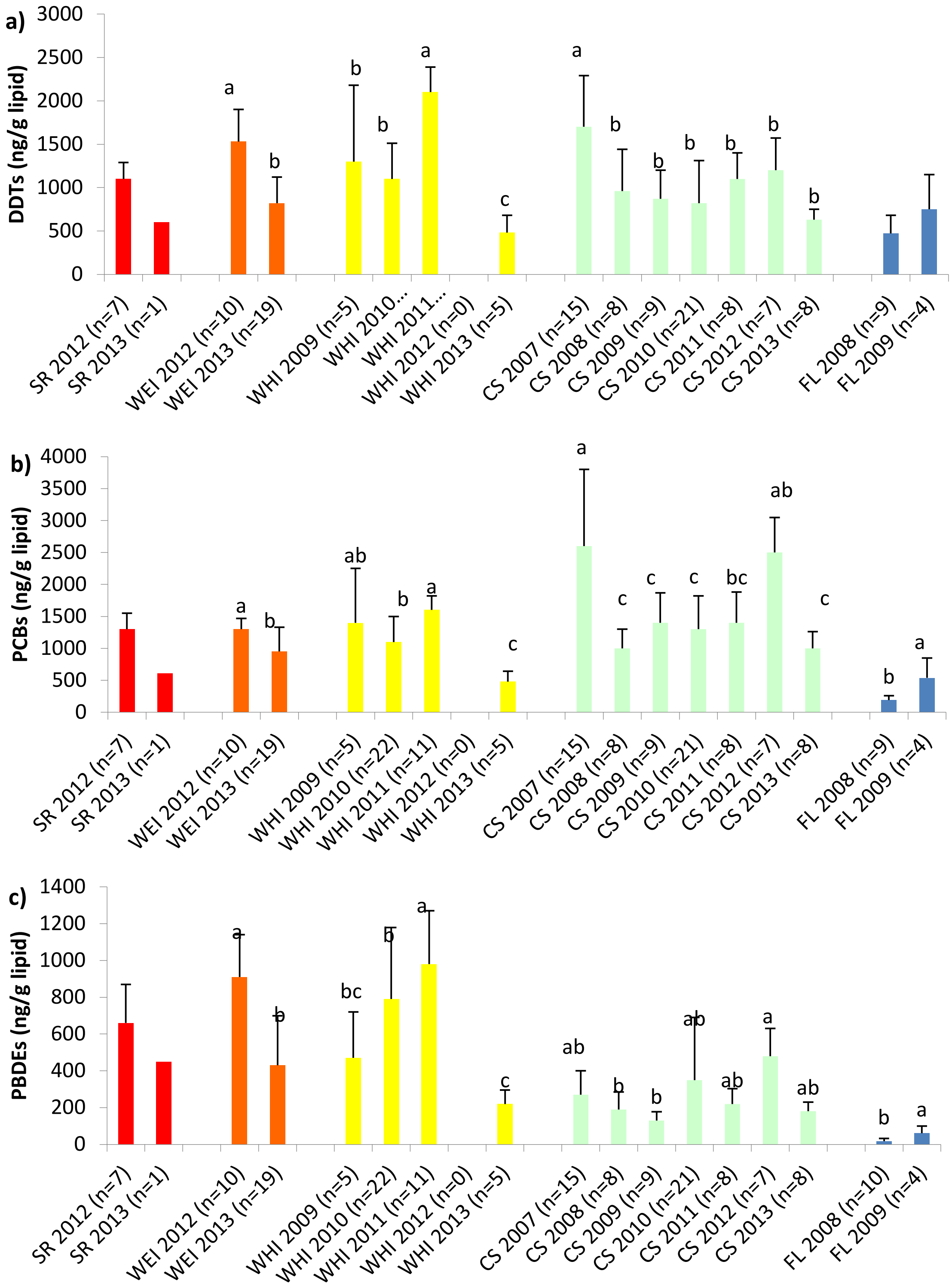


Figure 53. Mean concentrations in ng/g lipid (SD) of a) DDTs, b) PCBs, and c) PBDEs in juvenile Chinook salmon collected from trends sites in 2013 as compared to other years. Sample sizes indicate the number of composite samples of 3-5 fish analyzed. Within each site, values with different letter superscripts are significantly different (ANOVA and Tukey's multiple range test, $p < 0.05$). SR = Secret River, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

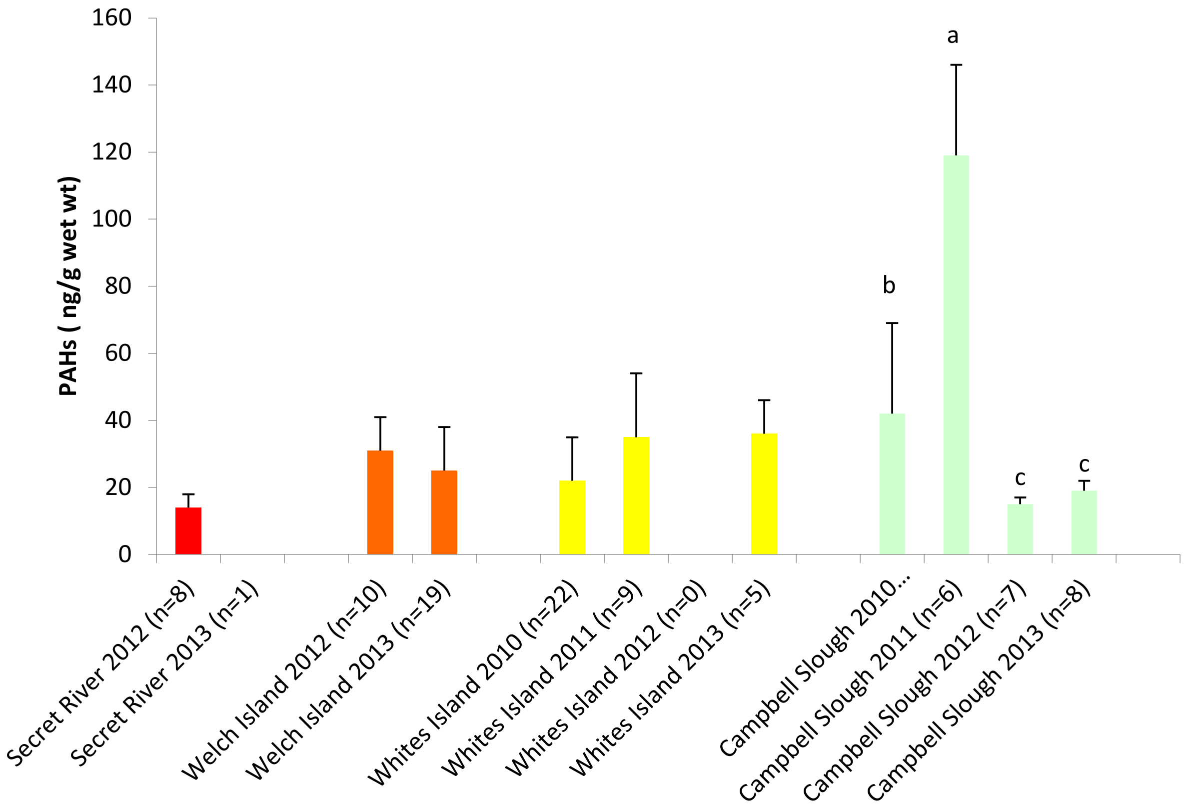


Figure 54. Mean concentrations (SD) in ng/g wet wt of PAHs in bodies of juvenile Chinook salmon collected from the trends sites in 2013, as compared to other years. Sample sizes indicate the number of composite samples (3-5 fish) analyzed. Within each site, values with different letter superscripts are significantly different (ANOVA and Tukey's multiple range test, $p < 0.05$). SR = Secret River, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

4.5.3 PIT-Tag Array Monitoring of Juvenile Salmon Residence

The passive integrated transponder (PIT) tag detection system located at Campbell Slough was powered up and operational on March 12, 2014. At this time, water depth in the channel was approximately 12 feet and the top of the antennas were approximately two feet below the surface. Throughout the 2014 monitoring period, water level fluctuated mostly within the 6 to 10 feet deep range (i.e., optimal depth for function of the antennae) until mid-July when it dropped and remained below six feet through the summer and much of the fall (water level values are determined relative to the nearest USGS gage station in Vancouver, WA; Figure 55). During this period of low water levels (less than 5 feet) after mid-July, the antennas were floating on the surface. On November 4, 2014 an antenna cable was severed (possibly by an aquatic rodent) and the system was completely shut down on December 11, 2014 due to insufficient solar power generation in the winter months and to allow for repairs to the damaged cable. Overall, the antennas, receiver, modem and batteries functioned correctly throughout the monitoring period, providing continuous and uninterrupted data collection until early November.

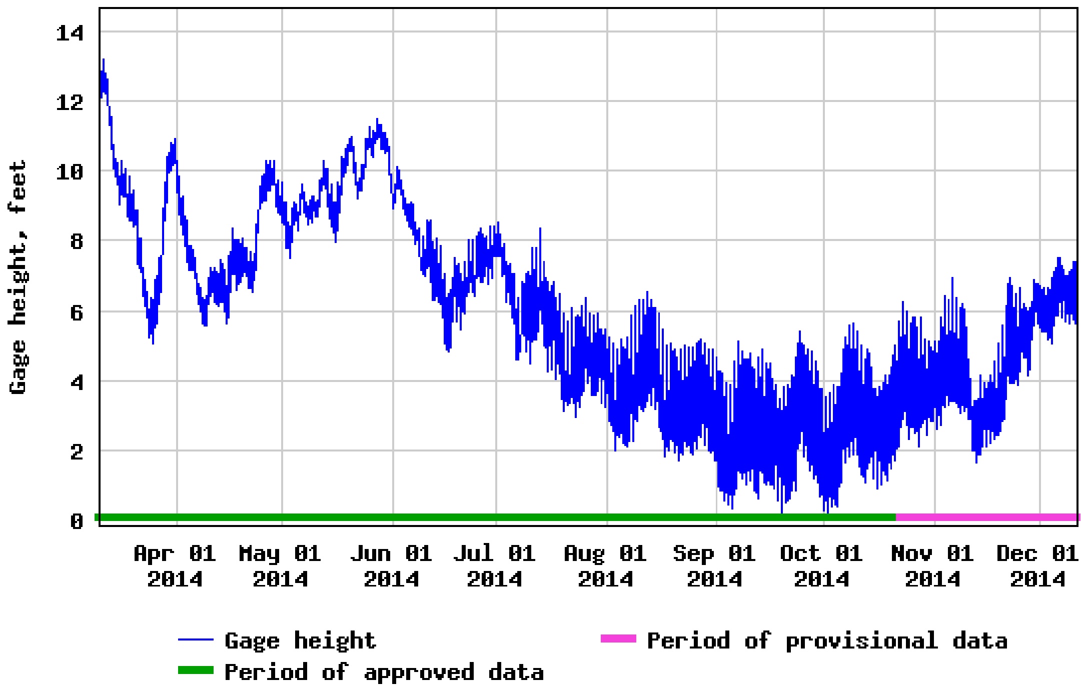


Figure 55. Seasonal water levels from the USGS gage 14144700 Columbia River at Vancouver, WA.

The PIT tag detection system was operable and collecting data for nearly eight months. During this time, the system recorded 55 detections, which corresponded to 31 unique tags. The first detection occurred on March 23, 2014 and the last detection was on June 12, 2014. Using the Columbia Basin PIT Tag Information System (PTAGIS) database we were able to determine species and site origination information for all but four of the tagged fish detected at Campbell Slough. As in 2013, most of the detected fish were hatchery fall Chinook salmon and a considerable number of northern pikeminnow (Table 32). Several wild Chinook salmon and hatchery spring Chinook salmon were also detected in 2014, as well as individual detections of hatchery summer steelhead, hatchery coho salmon, and hatchery summer sockeye salmon. All of the hatchery fall Chinook salmon were produced from the Spring Creek National Fish Hatchery (near White Salmon, WA), which is also where most of the Chinook salmon detected in 2013 originated. All northern pikeminnow detected by our array at Campbell Slough were tagged and released in the vicinity of Government Island (rkm 187). The three wild Chinook salmon originated from the Willamette River basin, two of which originated from near the Leaburg Dam on the McKenzie River in Oregon and the third was tagged or recaptured in the lower Willamette River near Sullivan Dam at Willamette Falls. The hatchery spring Chinook salmon came from the North Santiam River (Willamette Basin) in Oregon and the other from the Rapid River Hatchery (Snake River Basin) in northern Idaho. Other salmonid species detected at Campbell Slough included a hatchery coho salmon released below Roza Dam on the Yakima River, a hatchery summer sockeye salmon from Redfish Lake in Idaho, and one hatchery summer steelhead that originated from the Klickitat River.

Nearly half (14 of 31) of all the tagged fish were detected repeatedly at Campbell Slough over multiple days or weeks. The majority of the individuals that were repeatedly detected over days or weeks were hatchery fall Chinook salmon and northern pikeminnow (the two most abundant fish species detected by the array). Of the four fall Chinook salmon that were detected on multiple occasions, two fish went for more than two weeks between detections. The length of time between detections of the five northern pikeminnow tended to be only few days, but one fish was first detected in late April and then again on multiple occasions in early June, representing 46 days between the first and last observations. The other fish detected more than once included hatchery spring Chinook salmon, wild spring Chinook salmon, hatchery summer sockeye salmon, and hatchery summer steelhead. For each, all detections occurred within a 24 hour time frame.

Table 32. PIT-tagged fish detected in 2014 at the Campbell Slough PIT-tag array.

Species	# Fish Detected	Months Present	Length (mm)	Residency (days)	
				Range	Mean
Juvenile hatchery fall Chinook	11	April, May, June	58 - 92	1 - 17	4.3
Juvenile hatchery spring Chinook	2	March, May	70	1 - 2	1.5
Juvenile wild spring Chinook	3	March, April	96 - 123	1 - 2	1.3
Juvenile hatchery Coho	1	May	n/a	1	n/a
Juvenile hatchery summer Sockeye	1	May	n/a	1	n/a
Juvenile hatchery summer Steelhead	1	May	n/a	1	n/a
Northern Pike Minnow	8	April, May, June	257 - 336	1 - 46	7.9

The majority of detections occurred during the month of May, which coincided with the post-tagging release of many of these fish from hatcheries in late April and early May. The tagged pikeminnow were all released in April 2013 and timing of entry into Campbell Slough was likely to feed on migrating juvenile salmon. The first fish detected at the site in late March to early April were all spring Chinook salmon (wild and hatchery). Most of these fish had been tagged and released in summer or fall of 2013. As noted previously, most of the salmon detected were fall Chinook salmon from Spring Creek Hatchery near White Salmon, WA and it took approximately two to three weeks for most of these fish to migrate downstream to Campbell Slough; however, one salmon migrated that same distance in four days. Two salmon originating in Idaho, a sockeye from Redfish Lake and a spring Chinook salmon from Rapid River Hatchery, travelled the greatest distance to Campbell Slough. While it took the Chinook salmon over two months to reach our site, the sockeye salmon was detected at Campbell Slough less than two weeks after its release. The migration timing for the sockeye salmon suggests that it was transported downriver on a barge, but there were no additional detection records to confirm this. The fish that took the longest time period to migrate to Campbell Slough was a hatchery spring Chinook salmon from the North Santiam River, released in June 2013.

5 Discussion

5.1 Mainstem Conditions

Environmental conditions in all of the trends monitoring sites are influenced by mainstem river flow. This is particularly true during periods of high flow when connectivity between shallow water sites and the mainstem is greatest. It is therefore critical to characterize conditions in the mainstem in order to contextualize observations from the shallow water habitats. Based on the 2014 data, in addition to the data from past monitoring years, mainstem dissolved nutrient concentrations are highest during the winter and lowest in late summer. In 2014, we again observed very low concentrations of biologically available phosphorus in the mainstem, which persisted throughout the spring and summer. Although nitrate showed a seasonal decline from spring to summer, there was an abrupt increase in nitrate concentrations in early September that coincided with the cessation of managed flow from Bonneville Dam that cannot be explained using the data at hand.

Dissolved nitrogen: phosphorus ratios (N:P, mol:mol) can be used to infer nutrient limitation of phytoplankton growth, since the average elemental ratio of phytoplankton is 16N:1P. In the mainstem river, N:P ratios always exceeded 16:1, which is generally indicative of phosphorus limitation of phytoplankton growth. The ratios were particularly high in the spring and low in the summer.

The daily average temperatures in the mainstem river at RM-122 exceeded the threshold of 19°C (Bottom et al. 2011) for a total of 72 days in 2014, which is slightly less than the number observed in 2013 at RM-53 (81) and 2009 (82). The number of days having an average temperature exceeding 21°C (42), however, was higher than any of the years between 2009-2013. The year 2009, which had the highest total number of days where the average temperature exceeded 19°C had 11 days where the 21°C threshold was crossed, while 2013 had 14 days where the temperatures were higher than 21°C. Thus, 2014 can be classified as a warm year within the EMP time series. It is important to note that water temperatures in the lower river are generally consistent across reaches and the water column is well mixed (as shown by Sagar et al. 2015 at RM-53 and RM-122). Further analyses not included in this report are underway to determine how warm conditions in the mainstem manifest in the shallow water trends sites (Tausz et al. in prep.).

5.2 Abiotic Site Conditions

Among the 2014 trends sites where water quality monitoring was conducted, Welch Island had the smallest variations in dissolved oxygen, pH, and specific conductance throughout the April–July monitoring period. The temperature depression characteristic of the spring freshet was most prominent at Franz Lake and was not as strong at other sites compared to previous monitoring years. In 2013 and 2014, prolonged inundation did not occur at the two most upstream sites (Campbell Slough and Franz Lake Slough) the way it did during and after the high flows of 2011 and 2012. At all sites, the largest daily variations in water quality parameters were observed in mid-June through July, when channel depths were the shallowest during the April–July monitoring period. Presumably because of the shallower channel depths, there were larger daily variations in water quality parameters in 2013 and 2014 compared to 2011 or 2012. Although all sites except Welch Island had daily median dissolved oxygen concentrations less than the 8.0 mg/L threshold (in late May to early June at Campbell Slough and only during late July at Franz Lake and Whites Island), the daily maxima were almost never less than that threshold. The only exception was at Campbell Slough, where the maximum dissolved oxygen concentration was 7.1 mg/L on two days in early June. Otherwise, all the sites had dissolved oxygen concentrations suitable for salmonids during at least a portion of every day during the 2014 monitoring period. This was not the case at Campbell Slough during some previous monitoring years, particularly 2011, when dissolved oxygen concentrations were continuously low when the site was flooded during and after the freshet. Similar to previous monitoring years, Whites Island had the most suitable water quality conditions for juvenile salmonids during the 2014 monitoring period. During 2014 monthly fish sampling events, salmonids were captured during seining surveys at Campbell Slough, Whites Island, and Welch Island even on days when the water quality thresholds were not met. The fish sampling event on June 10 at Campbell Slough was the only sampling event when no salmonids were caught. On that day, neither the weekly maximum temperature nor the daily minimum dissolved oxygen threshold was met. Salmonids were caught in early April at Franz Lake during its only fish sampling event, before water quality data were available. However, temperature and dissolved oxygen trends in the days following fish sampling at Franz Lake indicate that water quality parameters would have been suitable for salmonids based on thresholds established by Washington State.

Nutrient concentrations at the trends sites were generally highest in spring and declined into the summer, particularly in the case of phosphorus. There was evidence that phosphorus was present at low enough concentrations to limit primary production at Welch Island and Whites Island. N:P ratios were below the 16:1 threshold often used as a proxy for nutrient replete conditions. In contrast, at Campbell Slough and Franz Lake Slough, the ratios were much lower, and generally fell below the 16:1 threshold. This is interesting because these were the two sites where cyanobacteria populations were present in the highest abundances. A likely explanation for the difference in cyanobacteria abundances between Campbell Slough and Franz Lake Slough (high) versus Welch Island and Whites Island (low) is that most cyanobacteria are capable of fixing atmospheric nitrogen (N2) using the enzyme, nitrogenase. Therefore, if nitrogen concentrations are low, but phosphorus is sufficient, this will tend to favor the proliferation of cyanobacteria rather than other types of phytoplankton. The ecological, biogeochemical, and food web consequences of differences between cyanobacteria-dominated vs. diatom-dominated communities have not been established in the Columbia River system. Efforts are underway to characterize features of the lower food web that could influence water quality using EMP data from 2011-present (Tausz et al. in prep.).

While the dissolved N:P ratios in the mainstem river always exceeded 16:1 (which is generally indicative of a system where primary production is regulated by phosphorus availability), at the four shallow water trends sites, the N:P ratio varied in space and time. At Whites Island and Welch island, the N:P ratio generally exceeded 16:1, except that the ratio declined and fell below the threshold at Whites Island in July. In contrast, at both Campbell Slough and Franz Lake Slough, N:P ratios generally fell below 16:1, with a few exceptions. Ratios below 16:1 are indicative of a system where primary production is regulated by the availability of biologically available nitrogen.

5.3 Habitat Structure

5.3.1 Hydrology

Hydrology in estuaries is often complex due to the mixing of riverine and tidal influences. In the Columbia River estuary this is particularly true with high riverine discharge combined with tidal influence for most of the 234 km extent of the estuary (Jay et al. 2015). The lower estuary is predominantly affected by tidal cycles, but also by winter storm runoff from the coastal subbasin of the Columbia River watershed (Jay et al. 2015). In contrast, in the upper estuary (above the Willamette River confluence) high flows occur during the spring freshet. This complexity results in spatial trends that are discernable in the trend site data gathered as part of this study. Predictably, inundation timing and magnitude differs throughout the estuary depending on the primary hydrologic drivers present. Inundation frequency (the percentage of hours in a year with standing surface water on the wetland) in the lower estuary is greater in the winter due to winter storms and runoff from western coastal tributaries (Figure 17). In the upper estuary, inundation duration is longer, but frequency of inundation events frequencies is lower, especially during the vegetation growing season.

Columbia River discharge is variable depending on annual precipitation and snow pack amount. Flows over the 20-year period from 1991 to 2011 at Beaver (rkm 89) varied between a minimum of 1,800 m^3/sec to a maximum of 24,500 m^3/sec (Jay et al. 2015). As a result, there is interannual variability in the inundation regime at wetland sites, particularly in the upper estuary which is more affected by river flow than tides. This variability is evident at the trends sites, as measured by the SEV (see Figure 17, Figure 18). The functional response to variable water levels is manifested in fluctuations of wetland capacity (Simenstad and Cordell 2000) as measured by vegetation cover, plant species composition, and organic matter production, as well as in differences in the frequency of opportunity for juvenile salmonids to access wetland channels. Specific ecosystem responses to the hydrologic regime are further discussed below.

5.3.2 Sediment Accretion Rates

Sediment accretion rates in marshes are influenced by numerous factors. In salt marshes, those factors include flooding regime, tidal range, elevation, distance to the marsh edge, sediment supply, storm activities, and vegetation composition (Richard 1978; Thom 1992; Ma et al. 2014). In riverine systems, additional factors that can contribute to accretion variability include dynamic river flows, distance from the river mainstem, proximity to tributary sediment sources, freshet magnitude, location along the riverine gradient, and biological disturbances such as beaver activity (Sherwood et al. 1990; Neubauer et al. 2002; Craft 2007). Sediment loading and transport in the lower river is related to discharge, but is complicated by dams and reservoirs in the basin, channel alteration, dredging, and land use changes in the sub-basins (Sherwood et al. 1990). Together these spatial, temporal, hydrologic, and biologic influences work in concert to affect the annual sediment accretion rates in the lower river.

Much of the variability in sediment accretion rates in the lower river can be explained by the factors described above. The lowest accretion rate with the lowest annual variability is the site closest to the mouth of the river (0.36 ± 0.26 at Ilwaco Slough, rkm 6) presumably due to a combination of the lower riverine sediment loads at the mouth, distance of the site from the mainstem (4.2 km), distance from a major tributary, and relatively high site elevation.

Sediment accretion observations from trends sites with multiple sets of accretion stakes indicate that sedimentation decreases with elevation and with distance from tidal channels. At the Whites Island site, the stakes closer to the tidal channel occur at a lower elevation (1.24 m, CRD) than those farther away from the channel (2.05 m, CRD) and as expected, the sediment accretion rates are greater at the lower-elevation stakes closer to the channel (on average 1.42 cm greater per year). However, a natural levee can result in the converse effect: higher elevation and higher accretion rates adjacent to a tidal channel, with lower elevation in the backmarsh behind the levee (Ma et al. 2014). This is the case at the Secret River high marsh, where one set of stakes is located on the natural levee adjacent to the channel (2.16 m, CRD) and the other is in the backmarsh at a slightly lower elevation (2.09 m, CRD). To date, the levee stakes had on average 0.41 cm higher accretion per year than the backmarsh, although variability was higher on the levee and we have fewer measurements in the backmarsh. Future data collection at these different elevations will confirm whether these trends will continue.

Lower elevation sites are typically expected to have higher sediment accretion rates (Richard 1978); however, the low marsh at Secret River is the only location in our study that has been eroding every year.

Interestingly, this is consistent with findings from a historical analysis that was done for reference wetland sites in the lower river (Diefenderfer et al. 2013; Borde et al. 2013), in which the Secret River site was the only site among 30 that had an increase in open water and a decrease in marsh surface compared to the historical maps from the late 1800's. The marsh is approximately half the size that it was on the historical maps (Borde et al. 2013), indicating that it has been eroding over the past 150 years. The timing and causes for this erosion are not known, but we speculate that it could be due to changes in hydrological dynamics (attributable to land use) in the Columbia River, or adjacent watersheds, wave action from storm events, or changes in sedimentation patterns and currents in the adjacent shallow bay.

At the three upper estuary trends sites, average annual sediment accretion rates are frequently greater than 1.0 cm per year, likely due to the greater sediment load of the river at the upper end of the estuarine gradient. At the Cunningham Lake and Campbell Slough sites, instances of rates below 1.0 cm per year may in part be attributable to animal disturbances. During several years cows were observed at the Campbell Slough site and in 2014 beavers had established a lodge and a trail near the stakes at the Cunningham Lake site. Animal activities such as these could compact the sediment between the stakes and result in erroneously low accretion rates.

The average annual accretion rate for all years at the trends sites is 0.75 cm. If the site nearest the mouth (Ilwaco Slough) and the eroding site (Secret River low marsh) are excluded, the annual average rate is 1.06 cm. Although the sediment load in the lower river is lower than historical levels (Sherwood et al. 1990; Naik and Jay 2011), it remains adequate to increase wetland elevations over time. The result is that wetlands are gradually prograding and expanding (Diefenderfer et al. 2013) and in at least some positions within the lower river they may have the potential to maintain adequate elevations despite increased inundation from sea level rise or other climate-driven changes. This remains to be explored in future research.

The implication of these observed sediment accretion rates for restoration is two-fold. First, it means that sites that have subsided prior to restoration have the potential to recover elevation over time. In fact, restoration may result in even higher accretion rates than those observed here (Diefenderfer et al. 2008). Related to this, the second implication is that these rates should be factored into restoration designs to ensure the site will meet long term goals. Specifically, considering the rate of elevation change could help determine the potential evolution of the plant community over time including the potential for reed canarygrass invasion. In concert with predictions about future elevation change at restoration and reference sites, there is a need to evaluate the potential for climate-related hydrologic change, and how these two drivers will alter the functional trajectories of lower Columbia River wetlands over time.

5.3.3 Salinity

Salinity in estuaries varies temporally and spatially depending on the timing and magnitude of freshwater inputs. In the lower Columbia River, salinity is variable along the estuarine gradient. The salinity gradient extends from the river mouth to the approximately rkm 21 during a strong spring freshet and up to rkm 43 during a weak spring freshet (Chawla et al. 2008), and even further during low flow periods. Salinity measured as part of this study, at Ilwaco Slough, indicates that salinity is variable on daily, monthly, seasonal, and annual scales. The mean salinity over a growing season varied less than 1 ppt during the three years of monitoring for this study. The plant community present at the site is adapted to this variability and is comprised of species that can tolerate fluctuations from freshwater to periods of greater salinity.

5.3.4 Vegetation Cover, Species Assemblage, and Organic Matter Production

Elevation and hydrology affect primary productivity in wetlands. Findings generated by the EMP are elucidating how spatial patterns of these factors across the lower river, as well as the plant species-specific mechanisms, cause variability in the amount of organic matter contributed to the food web by different plant communities. In high marshes (approx. 1.5 m CRD and higher) throughout the study area, we observed higher cover, summer standing stock, and organic matter production compared to low marshes. Hydrology had complex effects in sites in the upper estuary characterized by a fluvial-dominated hydrologic regime: both cover and summer standing stock were lower in high-water years, effects which appear to linger in out-years, also affecting potential organic matter production; conversely, cover is higher in moderate and average water years. The characteristic effects of low water years remain unknown because of the lack of low water years during the course of this study, however, we suspect low water years may benefit some species such as wapato (*Sagittaria latifolia*; a species that prefers less extreme inundation), and perhaps worse for species characteristic of high marshes due to the presumed lack of inundation at those elevations.

Although the EMP has previously documented the dominance of invasive reed canarygrass in the lower river (Sagar et al. 2013), this study, with more focus on lower estuary sites, has shown that the cover of Lyngby's sedge (*Carex lyngbyei*) can be similar to that of reed canarygrass if high seasonal flooding affects the latter species. Moreover, the data we have collected to date indicate greater organic matter production per unit area by *Carex lyngbyei* (858 g/m^2) than by reed canarygrass (345 g/m^2); however, data are lacking from sites in years where reed canarygrass was not affected by high inundation at the trends sites or trampling by cows at Campbell Slough. Whites Island is the only site that did not experience such disturbances and reed canarygrass organic matter production was an intermediate value to the species averages (623 g/m^2); however, at this site reed canarygrass is mixed with other species unlike its widely observed habit of monocultural dominance.

Given the well-documented spatial variation in hydrologic regime in the lower river (Sagar et al. 2013, Jay et al. 2015), it is unsurprising that this research also uncovered spatial variation in primary productivity. The study produced the following four key findings regarding the spatial variation of sites on the longitudinal river gradient from near the mouth to rkm 221.

1) High marsh sites in the lower reaches have higher cover, more *Carex* spp., and greater organic matter production.
2) The Whites Island site (rkm 72) has high cover, less *Carex* spp., and reduced organic matter production.
3) High water reduces cover at the Cunningham Lake (rkm 144) and Campbell Slough (rkm 145), and these sites have a high proportion of reed canarygrass in the high marsh. Together these effects likely would result in lower organic matter production; however, data from these sites are too limited to confirm this hypothesis.
4) Prolonged high water at the Franz Lake Slough site (rkm 221) during all three years of standing stock data collection resulted in a dominant vegetation shift from reed canarygrass to *Polygonum amphibium*; we have limited data on organic matter production by this species, but it appears to be less than reed canarygrass and the shift resulted in overall lower organic matter production at this site.

In summary, several implications of these findings can be generalized for the lower river. Reed canarygrass has lower organic matter production than some other native species, such as *Carex lyngbyei*, especially in high water years. Because of this, upper estuary sites, which have higher reed canarygrass cover and experience floods of greater magnitude and duration, produce less organic matter than lower estuary sites. High marshes have much higher organic matter production than low marshes. These findings suggest that if high organic matter production (in support of the salmonid food web) is a habitat restoration goal (Maier and Simenstad 2009), then high marshes should be one of the programmatic targets. However, it will be necessary to develop habitat restoration strategies to create diversity among habitats in order to allow higher potential organic matter production than monocultures of reed canarygrass would produce.

5.3.5 Channel Morphology and Inundation

The inundation data from the trends sites highlight potential differences in the relative timing of juvenile salmon habitat opportunity (i.e., access) between sites located in the upper and lower reaches. However, the differences between these two subareas of the lower river appear to be less attributable to the morphology of channels themselves and more to the landscape position of these wetlands relative to the dominance of fluvial versus tidal influences. Longitudinal trends (i.e., from near the mouth of the Columbia to rkm 221) were not discernable in the channel morphology metrics we measured at trends sites (thalweg and bank elevation; channel depth, width, cross-sectional area, and width:depth ratio). Nevertheless, the typical seasonal inundation pattern of upper reach sites versus the daily inundation pattern of lower river sites is important to consider relative to habitat opportunity for juvenile salmon stocks, which taken collectively are migrating year-round from natal streams throughout the basin (Bottom et al. 2008). The extended inundation we observed in the upper reaches during the spring freshet resulted in the channel thalweg *and* channel banks being inundated for most of the time during the peak juvenile salmon migration period. Still, the percent time the average marsh elevation is inundated (Figure 17) does not tell the whole story in regard to implications for access by aquatic organisms; at sites in the lower reaches, the frequency of inundation in the channel varied from 27 to 97 percent.

Comparative analysis of channels at the EMP trends sites is limited by the different landscape positions of these sites and the position of the surveyed channel reaches within the local channel networks feeding each site (i.e., the number of confluences between the surveyed reach and the mainstem river). Regarding landscape position, for example, the reach surveyed at the Cunningham Lake site is ~6 km from the mainstem Columbia, while the Campbell Slough site is only ~1.5 km away. Regarding network position, all sites are positioned directly on the mainstem river except Whites Island and Welch Island, which are each one confluence away. The channel at Welch Island (through most of the surveyed channel) followed a predictable pattern, getting narrower and shallower with distance from the channel mouth (e.g., Zeff 1999). Although all channels are alike in being “dead-end” sloughs, only a portion of each channel was included for each site survey. These factors provide some practical limits on interpretation of this data set. Nevertheless, these data serve as valuable reference points for sites with comparable positions in the landscape and channel networks throughout the lower river.

5.4 Food Web

Based on the last several years of observations (2011-present), the standing stocks of fluvial phytoplankton that support planktonic food webs were largest at the two trends sites located furthest from the Columbia mainstem (Campbell Slough and Franz Lake Slough) and smallest at the Reach B (Welch Island) and Reach C (Whites Island) sites (this study; Sagar et al. 2013; Sagar et al. 2014). Diatoms dominated the phytoplankton assemblage at Welch and Whites Islands throughout the sampling season, with smaller contributions by green algae and other taxa observed during the summer. Diatoms tend to be larger than some of the other taxa, and thus they generally result in greater concentrations of fixed organic matter, and generally constitute a more nutritious food source for higher trophic levels compared to other phytoplankton taxa (Campeau et al. 1994).

The peak chlorophyll *a* concentrations noted in June at Campbell Slough and Franz Lake Slough coincided with high abundances of cyanobacteria, which were present at increased proportional abundance in the late summer at these two sites. High abundances of cyanobacteria can lead to hypoxia and low pH associated with bloom die-off, decreased CO_2 availability and increased pH during periods of rapid growth, and reduced light penetration due to shading (Paerl et al. 2001; Havens 2008). Hypoxia following degradation of large algal blooms is often responsible for fish kills (Erickson et al. 1986; Lindholm 1991) and may reduce the diversity of benthic invertebrates (Jones 1987; Josefson and Widbom 1988). Shading of the water column by surface blooms of cyanobacteria can lead to loss of submerged aquatic vegetation (Havens 2008) as well as decreased visual clarity for benthic-feeding fish (Engström-Ost et al. 2006). The differences among sites in concentration of cyanobacteria versus other phytoplankton taxa reflect the fact that environmental conditions supporting growth differ among the sites, particularly with respect to available nutrients—both in terms of amount and in the ratio among them. For example, at the sites where cyanobacteria were abundant (Campbell Slough and Franz Lake Slough), the ratios of dissolved nitrogen: dissolved phosphorus were below the molar ratio typically found in the phytoplankton cells, indicating that nitrogen would tend to be the growth-limiting nutrient in the system. These conditions tend to favor cyanobacteria over other taxa.

Despite seeing distinct differences among sites, we are missing critical data about population dynamics upstream of the dam. For example, it might be that large populations of phytoplankton build up behind the dam and are transported downstream; alternatively, phytoplankton populations could increase substantially due to in situ growth. Because of this major unknown, there is uncertainty in interpretations of the daily changes in chlorophyll *a* and oxygen that provide data on net ecosystem metabolism. A solution would be to gather information about population dynamics of phytoplankton and nutrients upstream of Bonneville Dam in order to ensure that we can properly interpret the data from the in situ sensors.

Similar to other years, zooplankton abundances were higher at Campbell Slough than at the other fixed sites. Interestingly, the peak chlorophyll *a* concentration coincided with the highest abundances of zooplankton, particularly rotifers and cladocerans. In general, high abundances of zooplankton usually coincide with low concentrations of chlorophyll *a*, if phytoplankton are palatable. Since many cyanobacteria species are not readily grazed by zooplankton, the coincidence of chlorophyll *a* and zooplankton may reflect the resistance of these taxa to grazing pressure, as long as there was sufficient food to sustain zooplankton populations. Based on observations from previous years, although zooplankton do not tend to be preferred prey items for juvenile salmon at most times of year, they can constitute an important part of the diet in the mid to late summer (Sagar et al. 2013).

At the time of writing, estimates of food sources supporting juvenile salmon from stable isotope signatures were not complete. However, we can report trends in isotopic ratios of carbon and nitrogen at the four trends sites. Mean isotope ratios of carbon and nitrogen associated with invertebrates and autotrophs (plants, periphyton, and fluvial phytoplankton) at the trends sites became heavier along a downstream gradient, either reflecting the trend of nitrogen and carbon isotopic enrichment in marine compared to fresh waters (France 1994; Chaloner et al. 2002; Hobson et al. 2007) or indicating an increase in biomass at higher trophic levels, which typically bear heavier nitrogen isotope signatures. The latter could occur if production (the incorporation of organic matter at various trophic levels) increased along the downstream gradient. Ancillary evidence suggests that this could be the case downstream of Bonneville Dam; for example, the prevalence of parasitic infection of diatoms by chytrid fungi was shown to increase between river mile 53 (Beaver Army Terminal) and the downstream reaches of the estuary near Astoria (Maier 2014). A significant uncertainty in the EMP is the degree to which the standing stocks of lower food web organisms (fluvial phytoplankton, zooplankton) reflects in situ productivity versus downstream advection of organic matter produced in the impoundments behind the dam. It would be highly valuable to carry out sampling upstream of the dam to determine the balance between in situ primary and secondary production vs. downstream advection.

Isotopic signatures of autotrophs differed among sites along the estuarine gradient, with evidence of isotopic enrichment at downstream sites consistent with Herzka (2005). Carbon isotopic values varied more among vegetation types than for algae (periphyton or particulate organic matter), consistent with a previous study in the Columbia River estuary (Maier et al. 2011). Additionally, isotopic signatures and the C/N ratios of particulate organic matter from the current study are consistent with values for riverine phytoplankton in a review conducted by Finlay and Kendall 2007, indicating that these particulate organic matter data are a good proxy for phytoplankton. Due to the spatial variability among organic matter sources, a site-by-site approach would be most suitable for applying food web mixing models to these data.

Because salmon are transitory, the spatial pattern in isotopic signatures observed in the autotrophs did not hold for salmon tissues in this study. Instead, the isotopic signatures varied by month of capture and fish length, which likely reflects dietary shifts during growth and migration, with larger fish having lower $\delta^{15}N$ signatures in tissues. Figure 56 shows a generic representation of how isotopic signatures of fish tissues are expected to change after a dietary shift between isotopically distinct food sources (Herzka 2005). Similar changes have been shown after dietary shifts in laboratory studies, with the direction of change (isotopic enrichment or depletion of tissues) depending on the food sources used (Herzka 2005; Church et al. 2009). Other studies of feeding Pacific Northwest juvenile salmonids indicate that $\delta^{15}N$ decreases with size, age, or other metrics of growth, as tissues with enriched isotopic signatures reflecting marine-derived maternal influence or hatchery food influence turn over and begin to reflect less enriched freshwater food sources (Bilby et al. 1996; Kline and Willette 2002).

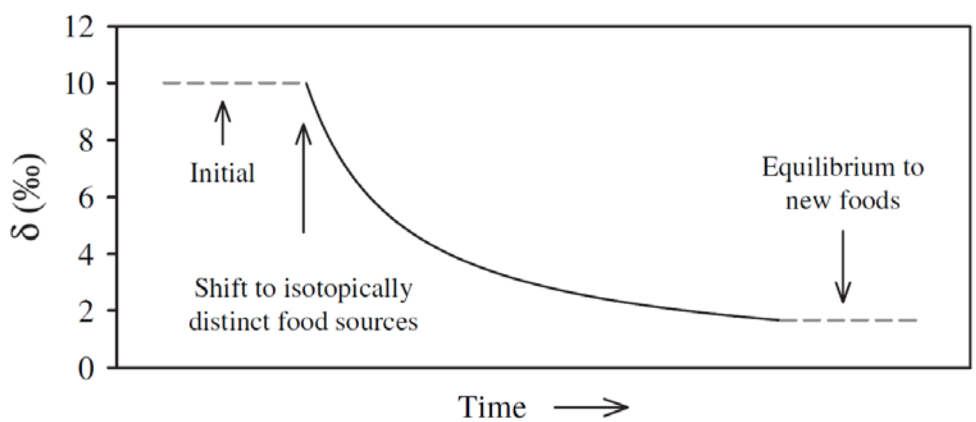


Figure 56. Generic representation of expected pattern of isotopic change in fish tissues over time following a dietary shift to an isotopically different food source (Herzka 2005)

5.5 Fish Use

In 2014, fish sampling focused on revisiting four trends sites: Welch Island in Reach B, Whites Island in Reach C, Campbell Slough in Reach F, and Franz Lake in Reach H to collect additional information on temporal trends in these areas. Similar to past years, our ability to monitor salmon occurrence was limited at Franz Lake, and to a lesser extent, at Campbell Slough due to high water conditions and other problems associated with site access, somewhat compromising our ability to accurately assess trends in fish use at these sites. High water levels between the months of April and June at Franz Lake in particular, have often precluded sampling during the peak period of juvenile Chinook salmon occurrence since 2009. However, at the trends sites where we have been able to sample consistently, the 2014 data show patterns similar to those we have observed previously under this project (e.g., Sagar et al. 2013b).

Patterns of salmon occurrence and fish community composition have remained relatively stable over years sampled at Welch Island, Whites Island, and Campbell Slough (i.e., sites that have been consistently sampled across seasons, year-to-year). Fish assemblages at Welch Island in Reach B and Whites Island in Reach C, are dominated by three-spined stickleback, include few non-native species or fish that are known salmon predators, and have low species richness and diversity. At Campbell Slough in Reach F, species richness and diversity are higher, but non-native species make up a significant proportion of catches and predatory fish such as smallmouth bass and northern pikeminnow are more common. Fish community patterns at Franz Lake have generally been similar to Campbell Slough, with greater species diversity and richness, higher proportions of non-native species, and higher proportions of predatory fish than Welch or Whites Island. Thus, annual sampling conducted at trends sites has shown a general increase in species richness and diversity, as well as greater occurrence of non-native and predatory species in the upper, freshwater dominated reaches compared to the lower and middle reaches.

Juvenile Chinook salmon was the dominant salmon species observed at Welch Island, Whites Island, and Campbell Slough. Unmarked Chinook salmon, especially of the smaller fry size class, have consistently dominated catches over time at Welch and Whites Islands, while at Campbell Slough, catches include substantial proportions of both marked and unmarked fish as well as a greater abundance of fingerlings than fry. Chinook salmon density has generally been higher at Welch Island and Whites Island than at Campbell Slough, although in some years total Chinook salmon densities at Campbell Slough approach those at Whites Island because of the presence of marked fish. Chum, coho, and sockeye salmon, as well as steelhead trout, are found only at low densities at these sites. Among unmarked fish, West Cascades Fall Chinook salmon are the dominant Chinook salmon stock observed at Welch Island and Whites Island, while at Campbell Slough, Upper Columbia Summer/Fall Chinook salmon are the most numerous, and other interior stocks, such as Snake River Fall Chinook salmon and Deschutes River Fall Chinook salmon, are also found (albeit in small numbers). As we have observed previously (Sagar et al. 2013b, 2014a, 2014b), marked Chinook salmon were, for the most part, West Cascades Fall Chinook salmon and Spring Creek Group Fall Chinook salmon, with West Cascades Fall Chinook salmon being most prevalent at Welch and Whites Island and Spring Creek Group Fall Chinook salmon most likely to occur at Campbell Slough.

Although patterns of salmon occurrence at Franz Lake have been more difficult to evaluate due to challenges associated with accessing the site during high water periods from 2011 to 2014, some consistent patterns have been observed. For example, coho salmon have consistently been captured in higher proportions at Franz Lake than at Welch Island, Whites Island, or Campbell Slough. Although Chinook salmon also use the site and account for a significant proportion of catches, other species such as chum salmon, sockeye salmon, and trout have been observed in small numbers. However, the proportions of marked and unmarked Chinook salmon and coho salmon in Franz Lake catches have varied from year to year. In 2008 and 2009 (i.e., the earliest sampling years for the program), many of the Chinook salmon and coho salmon captured at Franz Lake were marked fish, but more recently, unmarked (presumably wild) fish, of fry size class, have dominated catches. Indeed, a more detailed analysis on variance in EMP data (Sagar et al. 2015) suggested that the proportion of marked Chinook salmon captured at Franz Lake has decreased between 2008 and 2013, a trend that is further supported by marked Chinook salmon being absent from this site again in 2014. While this change could represent a trend in relative abundance of marked hatchery fish and unmarked, presumably wild fish, it is more likely due to the fact that since 2009 we have been unable to reliably sample Franz Lake in May and June, when the majority of hatchery fish are released (see Columbia River DART; http://www.cbr.washington.edu/dart/hatch.html). The Chinook salmon and coho salmon found at Franz Lake in 2014 were collected in April and November, prior to or after hatchery releases. If more comprehensive seasonal sampling could be performed at Franz Lake, it is possible that salmon occurrence patterns would be increasingly similar to those observed in 2008 and 2009, when the site supported a relatively high proportion of hatchery origin Chinook salmon and coho salmon. In addition, year-round sampling at Franz Lake could also allow for a more thorough analysis of trends in Chinook salmon stock occurrence at this site.

While salmon occurrence patterns in 2014 were generally similar to previous sampling years, there were a couple of unexpected findings. First, although chum salmon have been observed in small numbers at most sampling sites in past years, none were collected in 2014. This was somewhat unexpected, as all sites were sampled between February and April, the typical time of juvenile chum salmon outmigration (Salo 1991). It is not clear why chum salmon were not observed in 2014, but it may have simply been a matter of timing and the general low abundance of chum salmon at our sampling sites. Second, in both 2014 and 2013, small numbers of sockeye salmon were encountered at several of the trends sites, including Welch Island, Campbell Slough, and Franz Lake. The increasing presence of sockeye salmon at the trends sites is consistent with reports of higher returns of sockeye salmon in recent years (Williams et al. 2014) and may be an indication of beginnings of recovery for this stock.

In addition to monitoring Chinook salmon occurrence at the EMP trends sites, we also monitor several indicators of salmon health, condition factor, lipid content, growth rates, and chemical contaminants. This report contains our most comprehensive analysis of juvenile Chinook salmon somatic growth rates to date. These analyses indicate that salmon growth rates are influenced by multiple factors, among the most important being unmarked (i.e., wild) vs. marked (i.e., hatchery) origin, reach of collection, distance of the collection site from the mainstem, and sampling year. Overall, marked fish had higher growth rates than unmarked fish, consistent with trends toward somewhat higher condition factor and lipid content characteristic of hatchery fish. For those sites sampled over multiple years, growth rates tended to be higher in earlier sampling years. Also, while there was considerable variation in growth rates among sites, the highest growth rates tended to be in fish collected at sites located farther upriver, in Reach H, and from sites located farther from the mainstem. Stock of origin also influenced growth rate, with Spring Creek Group Fall Chinook salmon (a stock common among hatchery reared fish) having the highest growth rates, and Deschutes River Fall Chinook salmon the lowest growth rates. However, stock did not appear to be as important in explaining differences in growth rates as the other factors mentioned above.

Overall, among unmarked fish, condition factor has tended to be highest in fish collected from Campbell Slough and lowest in fish collected from Franz Lake (for all sampling years including 2014). Within sites, while there was variation from year to year, clear increasing or decreasing trends in condition factor were not evident. Among marked fish, condition factor was more uniform, showing no significant variation among sites, or across sampling years within sites. Condition factor also tended to be higher among marked fish than unmarked fish (the overall mean condition factor for the four trends sites sampled in 2014 were 0.995 ± 0.185 for unmarked fish, $n = 1044$ and 1.018 ± 0.129 for marked fish, $n = 292$), although differences were not statistically significant.

Lipid content showed some relationships that were similar to condition factor (note that lipid data presented in this report was collected in 2013). For example, among unmarked fish, percent lipid content (like condition factor) was highest in fish from Campbell Slough (1.7%) and lowest in fish from Franz Lake (1.0%). The relationship was different, however, among marked fish, with highest mean lipid content at Franz Lake (2.6%) and lower values at the other sites (1.5-1.6%). Because hatchery fish generally have high lipid levels prior to release (Johnson et al. 2010), such a pattern would be expected if the fish collected from Franz Lake were recent hatchery releases, while fish from the downstream sites were representative of fish that had experienced loss of lipid during outmigration. We also found that the lipid content of juvenile Chinook salmon collected in 2013 was relatively high compared to samples from previous sampling years, with average values of about 2% in both marked and unmarked fish, as compared to an overall average of approximately 1.5% for both groups. The lipid content of fish collected from Whites Island was especially high (3.5%), which did not appear to be due to the influence of a single outlier from an unusually high sample, as three of the five samples analyzed from Whites Island in 2013 had lipid levels above 3%. Nor was it clear that the fish with high lipid levels were recent hatchery releases, as one of the samples with high lipid content was a composite of unmarked, presumably wild fish. Lipid levels were also higher in salmon collected in 2013 than in the previous years at Secret River and Welch Island, although the average levels of 1.58% and 1.79% were well within the range of values typically observed. At Campbell Slough (i.e., the only site besides Whites Island where lipid data are available from multiple years), there was no clear temporal trend in lipid levels. Like growth rates, lipid levels in 2011 and 2012 were lower than in earlier sampling years, but not so for 2013. It is interesting to note that while trends in lipid levels in fish from Franz Lake cannot really be assessed with data from only 2008 and 2009, the decline in lipid levels from 2008 to 2009 is paralleled by lower growth rates in 2009 as compared to 2008.

Chemical contaminants were also measured in juvenile Chinook salmon samples in 2013, adding additional data to previous sampling efforts initiated in 2007 at Campbell Slough. Throughout this period, concentrations of contaminants in juvenile Chinook salmon collected from the EMP trends sites have been relatively low, with mean concentrations below estimated thresholds for adverse effects (Meador et al. 2002; Arkoosh et al. 2010, 2011; Johnson et al. 2007, 2013). However, concentrations of bioaccumulative contaminants (i.e., DDTs, PCBs, and PBDEs) in fish from the trends sites below the Vancouver/Portland area (i.e., Campbell Slough, Whites Island, Welch Island, and Secret River) were somewhat higher than levels measured in salmon from Franz Lake or other sites in Reaches G and H (see Sagar et al. 2013b, 2014a, 2014b; Johnson et 2013), suggesting some impact of urbanization and industrial activities on these fish. Moreover, some samples from the sites in Reach F and downstream had concentrations of PCBs or PBDEs above thresholds associated with toxic effects. Concentrations of PCBs were above the 2400 ng/g lipid effect threshold (Meador et al. 2002) in 17% of samples from Campbell Slough and 2% of samples from Whites Island, while concentrations of PBDEs were above the 940 ng/g lipid threshold (Arkoosh et al. 2010, 2011) in 3% of the samples from Campbell Slough, 13% of the samples from Secret River, 17% of the samples from Welch Island, and 28% of the samples at Whites Island. At those sites where sufficient data are available to evaluate trends, contaminant concentrations appear to be either declining slightly or remaining stable.

Since 2010, we have also been measuring PAHs in whole body samples of juvenile salmon as an alternative to assessing exposure from PAH metabolites in bile, because of the difficulty associated with collecting enough fish for bile analyses. Using the whole body method, we have observed some differences among sampling sites, with highest levels of PAHs in samples from Campbell Slough, and lowest levels in samples from Secret River. Trends were apparent at Whites Island and Campbell Slough, the sites that have been sampled the longest, with PAH concentrations increasing at Whites Island, but decreasing at Campbell Slough. However, there are some limitations to this technique. For example, only lower molecular weight PAHs are present in tissues at measurable concentrations, so this type of analysis may underestimate PAH exposure. More work is needed to determine how accurately these measures reflect exposure to both high and low molecular weight PAHs.

The information we have collected on salmon occurrence and residence time from the PIT tag array at Campbell Slough supports beach seine data indicating the importance of tidal freshwater habitats for juvenile salmon. As in 2013, the PIT tag array results from Campbell Slough indicated that hatchery Chinook salmon from locations as far away as the Rapid River Hatchery in the Snake River Basin in northern Idaho were using Campbell Slough for feeding and rearing. Fish detected in 2014 included Spring Creek Group fall Chinook salmon of hatchery origin, as well as both wild and hatchery Willamette River spring Chinook salmon and Snake River Chinook salmon. These data are consistent with genetics information from beach seine sampling, which also indicate that Campbell Slough supports not only Lower Columbia River fall Chinook salmon of local origin, but Willamette River and Interior Columbia stocks as well. The timing of detections, with the peak number recorded in May for fall Chinook salmon, is consistent with the patterns seen in beach seine surveys. For some fall Chinook salmon, the time between detections was over two weeks, suggesting some degree of residency at the site. Coho salmon and sockeye salmon, which were found in small numbers in beach seine catches, were occasionally detected in the PIT tag array as well, including a hatchery summer sockeye salmon from Redfish Lake in Idaho. The PIT tag array also detected the presence of fish that are rarely seen in beach seine catches, including northern pikeminnow, Snake River spring Chinook salmon, and steelhead. This suggests that while these fish may enter Campbell Slough near the mainstem, they may be less likely to migrate farther up the channel into our beach seining site to make use of this off-mainstem habitat. Detection timing for these fish is consistent with this hypothesis. For hatchery spring Chinook salmon, wild spring Chinook salmon, hatchery summer sockeye salmon and hatchery summer steelhead detected more than once, all detections occurred within a 24 hour time frame. The length of time between detections of northern pikeminnow also tended to be just a few days; however, one fish was first detected in late April and then again on multiple occasions in early June, representing 46 days between the first and last observations.

5.6 Adaptive Management & Lessons Learned

Quantifying the natural range of variation within a system and identifying the source of that variation improves our ability to predict how ecological components will respond to changing environmental conditions. Conditions at the relatively undisturbed EMP sites can be considered endpoints for restored areas and documentation of these conditions is useful for informing restoration site design. For example, data collected under the EMP has increased understanding of how sediment accretion rates vary spatially, both within sites and across the lower river, providing information that restoration practitioners can use to help evaluate potential restoration trajectories or site-scale performance. These data should be coupled with sediment accretion data collected at planned and early-stage restoration sites, and should be considered, along with predicted changes to the hydrologic regime, during project planning and site-scale adaptive management. Although the hydrologic regime is variable throughout the lower river, it is somewhat predictable (Jay et al. 2015). On this basis, the calculation and prediction of inundation at restoration sites, along with possible source and quantities of sediments from upstream watershed, is the most reliable way to evaluate available sediment accretion data to predict rates of accretion in order to determine the plant species most likely to occur and persist, and to determine target elevations (Diefenderfer et al. 2013). Thus, sediment accretion rates and hydrologic regime can inform predictions of restoration trajectories for the mosaic of plant communities at a site within a range of uncertainty (i.e., the types of communities occurring across the elevation gradient from low marsh to swamp; Diefenderfer et al. 2008). For example, at low-elevation sites that accrete to higher elevations, conditions may become more favorable for reed canarygrass establishment, but moderate elevations also support greater plant species diversity and therefore may restrict reed canarygrass establishment and dominance.

Although data collected by the EMP has improved the understanding of some components of estuary function, some data remain too limited for some habitat monitoring metrics to fully inform habitat restoration planning and evaluation (e.g., sediment accretion rates and reed canarygrass productivity). Continued collection of such data should be prioritized in order to fill knowledge gaps. Specifically, we recommend increased deployment of sediment accretion stakes at multiple elevations per site as well as technical evaluation of the accuracy and precision of this extremely low-cost method through side-by-side comparisons with other methods that are more established in the literature, such as surface elevation tables (SET) and marking with feldspar clay. By developing statistical relationships between the sediment accretion stake method used in the CEERP (Roegner et al. 2009) and other more widely published (but potentially more expensive or difficult to measure) methods, the CEERP would be able to draw on and benefit from the published literature to inform the program from project planning phases through site assessment and programmatic evaluation. Secondly, our research has shown that reed canarygrass, a non-native invasive plant, is the most prevalent plant species in the lower river (Sagar et al. 2013). Subsequent research through the EMP has demonstrated that this plant makes substantially lower contributions of macrodetritus to the salmonid food web than other native wetland plants present on the floodplain. We also hypothesize that reed canarygrass may reduce preferred salmon prey availability when compared to native species, such as *Carex lyngbyei*. Therefore, we are developing recommendations to support restoration practitioners in their attempts to thwart the spread of this plant and subsequent monocultural dominance of restoration sites (e.g., Johnson et al. 2012). However, the data on reed canarygrass standing stock and organic matter production in the upper reaches are still too limited to ascertain the relative effects of high, average, and low water years and we recommend continued sampling in this subarea of the lower river.

Phytoplankton dynamics in the lower river not only affect water quality in the mainstem and in emergent marsh habitats, but phytoplankton also provides an important food source for salmon prey (i.e., macroinvertebrates and zooplankton). We hypothesize that reservoir conditions and spill patterns at Bonneville Dam affects, to some degree, primary productivity in the lower river, as indicated by decreases in dissolved oxygen percent saturation, along with increased nitrate levels after cessation of regulated spill on September 1. This is a recurring trend observed from year-to-year. By documenting plankton community composition above Bonneville Dam, it would allow us to test this theory and help interpret the balance of in situ primary and secondary production with downstream advection.

Nutrient concentrations at the trends sites were generally higher early in the sampling season than later in the season. While dissolved nutrient ratios (nigrogen:phosporus) typically exceeded 16:1 in the mainstem, the nutrient ratio at the trends sites varied based on site location and time. The two lower river sites (Welch Island and Whites Island) generally exceeding the 16:1 limit (indicating phosphorus limitation) until later in the season (July), while the ratios generally fell below 16:1 at the two up-river sites. These lower nutrient ratios at Campbell Slough and Franz Lake may be due to relatively high abundances of cyanobacteria (capable of fixing atmospheric nitrogen). Thus, if nitrogen concentrations are limited, but phosphorus levels are adequate, cyanobacteria growth will be favored over phytoplankton (e.g., diatoms). The ecological consequences of cyanobacteria dominance in relation to food web dynamics, as well as possible implications of cyanotoxins in the lower river are currently unknown and requires future inquiry.

Fish data collected under the EMP show differences among genetic stocks in factors such as growth, lipid content, size, and presence across the trends sites. Beach seine and PIT tag detection data show that marked and unmarked chinook from West Cascade, Willamette River, and Interior Columbia River stocks use off-channel habitat in the lower river during outmigration (particularly in the tidal freshwater reaches), thus habitat management decisions should take into account a variety of salmonid species and life-history types. Creating and maintaining a diversity of habitats that will support multiple species and stocks during critical life stages (e.g., juvenile rearing) will contribute to multi-species recovery. Additionally, for the first time we have noted the presence of juvenile sockeye in fish surveys at EMP sites in 2013 and 2014, potentially indicating the beginnings of recovery for that species.

From the EMP data, we are seeing differences in fish, vegetation, and food web metrics across sites, which is beginning to provide a stronger understanding of how habitats in the lower river function spatially. As we continue to work towards distinguishing patterns in these data, we should be able to apply what we are learning at individual sites to the hydrogeomorphic reach scale. Spatial and temporal data show that fish from a variety of genetic stocks use EMP trends sites and indicate that fish originating from regions throughout the Columbia River basin are using emergent wetland habitats in the lower river for rearing and refuge during out-migration. In addition, the EMP food web sampling focuses on emergent marsh habitats and has not evaluated diets or source of prey items from juvenile salmonids captured in the river mainstem or tributaries. Conducting food web sampling in these other habitats would provide important data on how the lower river provides benefits to juvenile fish that typically spend less time rearing in the lower river than ocean-type Chinook salmon (such as juvenile steelhead or sockeye salmon).

The Estuary Partnership shares results from the monitoring program with other resource managers in the region. 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, EMP results are presented at scientific conferences, such as the Columbia River Estuary Workshop and Washington's Salmon Recovery 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 is used to compare results from the AEMR Program (see Schwartz et al. 2015). These are just some of the ways the results from this multi-faceted program are applied in resource management decisions.

6 References

Araya, Y.N., J. Silvertown, D.J. Gowing, K.J. McConway, H.P. Linder, and G. Midgley. 2010. A fundamental, eco-hydrological basis for niche segregation in plant communities. New Phytologist 189(1):1-6.

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

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

Bilby, R.E., B.R. Fransen, and P.A. Bisson. 1996. Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. Canadian Journal of Fisheries and Aquatic Sciences 53:164-173.

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

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

Borde A.B., V.I. Cullinan, H.L. Diefenderfer, R.M. Thom, R.M. Kaufmann, J. Sagar, and C. Corbett. 2012. Lower Columbia River and Estuary Ecosystem Restoration Program Reference Site Study: 2011 Restoration Analysis. PNNL-21433, prepared for the Lower Columbia River Estuary Partnership by Pacific Northwest National Laboratory, Marine Sciences Laboratory, Sequim, Washington.

Borde A.B., S.A. Zimmerman, R.M. Kaufmann, R.M. Thom, and C.L. Wright. 2013. Lower Columbia River and Estuary Habitat Monitoring Study, 2012. PNNL-22410, prepared for the Lower Columbia River Estuary Partnership by Pacific Northwest National Laboratory, Marine Sciences Laboratory, Sequim, Washington.

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

Bottom, D., A. Baptista, J. Burke, L. Campbell, E. Casillas, S. Hinton, D.A. Jay, M.A. Lott, G. McCabe, R. McNatt, M. Ramirez, G.C. Roegner, C.A. Simenstad, S. Spilseth, L. Stamatiou, D. Teel, and J.E. Zamon. 2011. Estuarine Habitat and Juvenile Salmon: Current and Historical Linkages in the Lower Columbia River and Estuary. Final Report 2002-2008 to U.S. Army Corps of Engineers.

Bottom D.L., C.A. Simenstad, J. Burke, A.M. Baptista, D.A. Jay, K.K. Jones, E. Casillas, and M. Schiewe. 2005. Salmon at River's End: The Role of the Estuary in the Decline and Recovery of Columbia River Salmon. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Technical Memorandum NMFS-NWFSC-68, Northwest Fisheries Science Center, Seattle, Washington.

Buchheister, A. and R.J. Latour. 2010. Turnover and fractionation of C and N stable isotopes in tissues of a migratory coastal predator, summer flounder (*Paralichthys dentatus*). Canadian Journal of Fisheries and Aquatic Sciences 67:445-461.

Chaloner, D.T., K.M. Martin, M.S. Wipfli, P.H. Ostrom, and G.A. Lamberti. 2002. Marine carbon and nitrogen in southeastern Alaska stream food webs: evidence from artificial and natural streams. Canadian Journal of Fisheries and Aquatic Sciences 59:1257-1265.

Church, M.R., J.L. Ebersole, K.M. Rensmeyer, R.B. Couture, F.T. Barrows, and D.L.G. Noakes. 2009. Mucus – a new tissue fraction for rapid determination of fish diet switching using stable isotope analysis. Canadian Journal of Fisheries and Aquatic Sciences 66:1-5.

Campeau, S., H.R. Murkin, and R.D. Titman. 1994. Relative Importance of Algae and Emergent Plant Litter to Freshwater Marsh Invertebrates. Canadian Journal of Fisheries and Aquatic Sciences 51(3):681-692.

Craft, C. 2007. Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and US tidal marshes. *Limnology and oceanography* 52(3):1220-1230.

Diefenderfer, H.L., A.B. Borde, and V.I. Cullinan. 2013. A Synthesis of Environmental and Plant Community Data for Tidal Wetland Restoration Planning in the Lower Columbia River and Estuary. PNNL-22667, prepared by the Pacific Northwest National Laboratory, Marine Sciences Laboratory, Sequim, Washington, for the U.S. Army Corps of Engineers, Portland District, Portland, Oregon.

Diefenderfer, H.L., A.M. Coleman, A.B. Borde, and I.A. Sinks. 2008. Hydraulic geometry and microtopography of tidal freshwater forested wetlands and implications for restoration, Columbia River, USA. Ecohydrology and Hydrobiology 8(2):339-361.

Engström-Ost, J., M. Karjalainen, and M. Viitasalo. 2006. Feedings and refuge use by small fish in the presence of cyanobacteria blooms. Environmental Biology of Fishes 76:109-117.

Elskus, A., T.K. Collier, and E. Monosson. 2005. Interactions between lipids and persistent organic pollutants in fish. *In:* Environmental Toxicology, T.P. Mommsen and T.W. Moon (Eds), Elsevier, San Diego. pp. 119-152.

Eriksson, J.E., J.A.O. Meriluoto, and T. Lindholm. 1986. Can cyanobacterial peptide toxins accumulate in aquatic food chains? Proceedings from the IV International Symposium of Microbial Ecology. Ljubljana, Jugoslavia. pp. 655-658.

Finlay, J.C. and C. Kendall. 2007. Stable isotope tracing of temporal and spatial variability in organic matter sources to freshwater ecosystems. *In* Stable Isotopes in Ecology and Environmental Science, R. Michener and K. Lajtha (eds.), 2nd edition. Blackwell Publishing, Inc. 566 pp.

France, R.L. 1994. Nitrogen isotopic composition of marine and freshwater invertebrates. Marine Ecology Progress Series 115:205-207.

France, R.L. 1995. Stable isotopic survey of the role of macrophytes in the carbon flow of aquatic food webs. Vegetatio [Belgium] 124:67-72.

Fulton, T. 1902. Rate of growth of sea fishes. Fisheries Scotland, Scientific Investigations, Report 20, pp. 326-439.

Hambrook Berkman, J.A. and M.G. Canova. 2007. Algal biomass indicators (ver. 1.0): U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chapter A7, section 7.4. Accessed November 29, 2011 from http://water.usgs.gov/owq/FieldManual/Chapter7/7.4.html.

Havens, K.E. 2008. Cyanobacteria blooms: effects on aquatic ecosystems. Advances in Experimental Medicine and Biology. 619:733-747.

Herzka, S.Z. 2005. Assessing connectivity of estuarine fishes based on stable isotope ratio analysis. Estuarine, Coastal and Shelf Science 64:58-69.

Hobson, K.A., R.J. Smith, and P. Sorenson. 2007. Applications of stable isotope analysis to tracing nutrient sources to Hawaiian Gobioid fishes and other stream organisms. *In* Biology of Hawaiian Streams and Estuaries, N.L. Evenhuis and J.M Fitzsimons (eds.). Bishop Museum Bulletin in Cultural and Environmental Studies, v.3, pp. 99-111.

Jay, D.A., K. Leffler, H.L. Diefenderfer, and A.B. Borde. 2015. Tidal-fluvial and estuarine processes in the lower Columbia River: I. along-channel water level variations, Pacific Ocean to Bonneville Dam. Estuaries and Coasts 38(2):415-433.

Jay, D.A., A.B. Borde, and H.L. Diefenderfer. In Revision. Tidal-fluvial and estuarine processes in the Lower Columbia River II: water level models, floodplain wetland inundation, and reach classification. Estuaries and Coasts. 00:00-00.

Johnson G.E., H.L. Diefenderfer, B.D. Ebberts, C. Tortorici, T. Yerxa, J.C. Leary, and J.R. Skalski. 2008. Research, Monitoring, and Evaluation for the Federal Columbia River Estuary Program. PNNL-17300, Pacific Northwest National Laboratory, Richland, Washington.

Johnson, G.E., H.L. Diefenderfer, R.M. Thom, G.C. Roegner, B.D. Ebberts, J.R. Skalski, A.B. Borde, E.M. Dawley, A.M. Coleman, D.L. Woodruff, S.A. Breithaupt, A.S. Cameron, C.A. Corbett, E.E. Donley, D.A. Jay, Y. Ke, K.E. Leffler, C.B. McNeil, C.A. Studebaker, and J.D. Tagestad. 2012. Evaluation of Cumulative Ecosystem Response to Restoration Projects in the Lower Columbia River and Estuary. PNNL-20296, prepared for the U.S. Army Corps of Engineers, Portland District, Portland, Oregon, by Pacific Northwest National Laboratory, Richland, Washington.

Johnson O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. Status Review of Chum Salmon from Washington, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC-32, Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, Washington.

Johnson, L.L., B.F. Anulacion, M.R. Arkoosh, O.P. Olson, C.A. Sloan, S.Y. Sol, J.A. Spromberg, D.J. Teel, G.K. Yanagida, and G.M. Ylitalo. 2013. Persistent organic pollutants in juvenile Chinook salmon in the Columbia Basin: implications for stock recovery. Transactions of the American Fisheries Society 142(1):21-40.

Johnson, L.L., M.L. Willis, O.P. Olson, R.W. Pearce, C.A. Sloan, G.M. Ylitalo. 2010. Contaminant concentrations in juvenile fall Chinook salmon from Columbia River hatcheries. North American Journal of Aquaculture 72(1):73-92.

Johnson, L.L., G.M. Ylitalo, C.A. Sloan, B.F. Anulacion, A.N. Kagley, M.R. Arkoosh, T.A. Lundrigan, K.Larson, M.D. Siipola, and T.K. Collier. 2007. Persistent organic pollutants in outmigrant juvenile Chinook salmon from the Lower Columbia Estuary, USA. Science of the Total Environment 374(2-3):342-366.

Jones, B.I. 1987. Lake Okeechobee eutrophication research and management. Aquatics 9:21–26.

Josefson, A.B. and B. Widbom. 1988. Differential response of benthic macrofauna and meiofauna to hypoxia in the Gullmar Fjord basin. Marine Biololgy 100:31–40.

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

Kline, T.C. and T.M. Willette. 2002. Pacific salmon (*Oncorhynchus* spp.) early marine feeding patterns based on 15N/14N and 13C/12C in Prince William Sound, Alaska. Canadian Journal of Fisheries and Aquatic Sciences 59:1626-1638.

Lassiter, R.R. and T.G. Hallam. 1990. Survival of the fattest: implications for acute effects of lipophilic chemicals on aquatic populations. Environmental Toxicology and Chemistry 9:585–595.

Lindholm, T., J.E. Eriksson, M. Reinikainen, and J.A.O. Meriluoto. 1992. Ecological effects of hepatoxic cyanobacteria. Environmental Toxicology 7(1):87-93.

Lower Columbia Estuary Partnership. 2007. Lower Columbia River and Estuary Ecosystem Monitoring: Water Quality and Salmon Sampling Report, Portland, OR.

Ma, Z., T. Ysebaert, D. van der Wal, D.J. de Jong, X. Li, and P.M. Herman. 2014. Long-term salt marsh vertical accretion in a tidal bay with reduced sediment supply. Estuarine, Coastal and Shelf Science 146:14-23.

Maier, M.A. 2014. Ecology of diatoms and their fungal parasites in the Columbia River, Ph.D. dissertation, Oregon Health & Science University, pp. 1-214.

Maier, G.O. and C.A. Simenstad. 2009. The role of marsh-derived macrodetritus to the food webs of juvenile Chinook salmon in a large altered estuary. Estuaries and Coasts 32:984-998.

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

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

McCullough, D.A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids with special reference to Chinook salmon. U.S. EPA, Region 10, Seattle, WA.

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

Michener, R.H. and L. Kaufman. 2007. Stable isotope ratios as tracers in marine food webs: an update. *In* Stable Isotopes in Ecology and Environmental Science, R. Michener and K. Lajtha (eds.), 2nd edition. Blackwell Publishing, Inc. 566 pp.

Myers, J.M., C. Busack, D. Rawding, A.R. Marshall, D.J. Teel, D.M. Van Doornik, and M.T. Maher. 2006. Historical population structure of Pacific salmonids in the Willamette River and lower Columbia River basins. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-73, 311 p.

Naik, P.K. and D.A. Jay. 2011. Distinguishing human and climate influences on the Columbia River: changes in mean flow and sediment transport. Journal of Hydrology 404(3)259-277.

National Marine Fisheries Service (NMFS). 2000. Biological Opinion: Reinitiation of Consultation on Operation of the Federal Columbia River Power System, Including the Juvenile Fish Transportation Program, and 19 Bureau of Reclamation Projects in the Columbia Basin. National Marine Fisheries Service – Northwest Region, Seattle, Washington. Available at http://www.salmonrecovery.gov/.

National Marine Fisheries Service (NMFS). 2008. Biological Opinion – Consultation on Remand for Operation of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(1)(A) Permit for Juvenile Fish Transportation Program. NOAA's National Marine Fisheries Service – Northwest Region, Seattle, Washington. Available at http://www.salmonrecovery.gov/.

Neill, C. and J.C. Cornwell. 1992. Stable carbon, nitrogen, and sulfur isotopes in a prairie marsh food web. Wetlands 12(3):217-224.

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

Neill, C. and J.C. Cornwell. 1992. Stable carbon, nitrogen, and sulfur isotopes in a prairie marsh food web. Wetlands 12(3):217-224.

Neubauer, S., I. Anderson, J. Constantine, and S. Kuehl. 2002. Sediment deposition and accretion in a mid-Atlantic (USA) tidal freshwater marsh. Estuarine, Coastal and Shelf Science 54(4):713-727.

Paerl, H.W., R.S. Fulton, P.H. Moisander, and J. Dyble. 2001. Harmful freshwater algal blooms with an emphasis on cyanobacteria. The Scientific World 1:76-113.

Phillips, D.L. and P.M. Eldridge. 2006. Estimating the timing of diet shifts using stable isotopes. Oecologia 147:195-203.

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

Richard, G. 1978. Seasonal and environmental variations in sediment accretion in a Long Island salt marsh. Estuaries 1(1): 29-35.

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

Roegner G.C., D.L. Bottom, A. Baptista, S. Hinton, C.A. Simenstad, E. Casillas, and K. Jones. 2004. Estuarine Habitat and Juvenile Salmon – Current and Historical Linkages in the Lower Columbia River and Estuary, 2003. Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, Washington.

Roegner, G.C., H.L. Diefenderfer, A.B. Borde, R.M. Thom, E.M. Dawley, A.H. Whiting, S.A. Zimmerman, and G.E. Johnson. 2009. Protocols for monitoring habitat restoration projects in the lower Columbia River and estuary. U.S. Department of Commerce. NOAA Technical Memorandum NMFS-NWFSC-97, 63 pp.

Roegner, G.C., A.M. Baptista, D.L. Bottom, J. Burke, L.A. Campbell, C. Elliot, S.A. Hinton, D.A. Jay, M. Lott, T.A. Lundrigan, R.A. McNatt, P. Moran, C.A. Simenstad, D.J. Teel, E. Volk, J.E. Zamon, and E. Casillas. 2008. Estuarine Habitat and Juvenile Salmon Current and Historical Linkages in the Lower Columbia River and Estuary, 2002-2004. Report by National Marine Fisheries Service to the U.S. Army Corps of Engineers Portland District, Seattle, Washington, 139 p.

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

Sagar, J.P., A.B. Borde, L.L. Johnson, J.L. Morace, T. Peterson, K.H. Macneale, W.B. Temple, J.A. Needoba, R.M Kaufmann, V.I. Cullinan, S.A. Zimmerman, R.M. Thom, C.L. Wright, P.M. Chittaro, O.P. Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, D. Lomax, A. Silva and M.A. Maier. 2013a. Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 8 (October 1, 2011 to September 30, 2012). Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration.

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

Sagar, J.P., A.B. Borde, L.L. Johnson, J.L. Morace, T. Peterson, K.H. Macneale, W.B. Temple, J.A. Needoba, R.M. Kaufmann, V.I. Cullinan, S.A. Zimmerman, R.M. Thom, C.L. Wright, P.M. Chittaro, O.P. Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, D. Lomax, A. Silva and M.A. Maier. 2014. Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 9 (October 1, 2012 to September 30, 2013). Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration, April 2014. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Sagar, J.P., A.B. Borde, L.L. Johnson, T.D. Peterson, J.A. Needoba, K.H. Macneale, M. Schwartz, A. Silva, C.A. Corbett, A.C. Hanson, V.I. Cullinan, S.A. Zimmerman, R.M. Thom, P.M. Chittaro, O.P. Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, M.A. Maier and C.E. Tausz. 2015. Juvenile Salmon Ecology in Tidal Freshwater Wetlands of the Lower Columbia River and Estuary: Synthesis of the Ecosystem Monitoring Program, Trends (2005–2013) and Food Web Dynamics (2011-2013), Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration.

Salo, E.O. 1991. Life history of chum salmon, *Oncorhynchus keta*. *In* Groot, C., and L. Margolis (eds.), Pacific salmon life histories, pp. 231-309. University of British Columbia Press, Vancouver, Canada.

Sandercock F.K. 1991. Life history of coho salmon, *Onchorhyncus kisutch*. *In* Groot, C., and L. Margolis (eds.), Pacific salmon life histories, pp. 395–446. University of British Columbia Press, Vancouver, Canada.

Sather, N.K., E.M. Dawley, G.E. Johnson, S.A. Zimmerman, A.J. Storch, A.B. Borde, D.J. Teel, C. Mallette, J.R. Skalski, R. Farr, and T.A. Jones. 2009. Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats in the Vicinity of the Sandy River Delta, Lower Columbia River, 2008. Annual Report, May 2009. Prepared for Bonneville Power Administration under an agreement with the U.S.

Department of Energy Contract DE-AC05-76RL01830. Pacific Northwest National Laboratory, Richland, Washington 99352.

Schwartz M.S., A.B. Borde. J. Smith, and N. Elasmar. 2015. Action Effectiveness Monitoring for the Lower Columbia River Estuary Habitat Restoration Program, October 2013 – September 2014. Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration.

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

Shannon, C.E. and W. Weaver. 1949. The mathematical theory of communication. The University of Illinois Press, Urbana, IL, 117pp.

Simenstad, C.A., Burke, J.L., O'Connor, J.E., Cannon, C., Heatwole, D.W., Ramirez, M.F., Waite, I.R., Counihan, T.D., and Jones, K.L. 2011. Columbia River Estuary Ecosystem Classification—Concept and Application. U.S. Geological Survey Open-File Report 2011-1228, 54 p.

Simenstad C.A. and J.R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. Ecological Engineering 15:283−302.

Sherwood, C.R., D.A. Jay, R.B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical Changes in the Columbia River Estuary. Progress in Oceanography 25:299-357.

Sloan, C.A., D.W. Brown, G.M. Ylitalo, J. Buzitis, D.P. Herman, D.G. Burrows, G.K. Yanagida, R.W. Pearce, J.L. Bolton, R.H. Boyer, and M.M. Krahn. 2006. Quality assurance plan for analyses of environmental samples for polycyclic aromatic compounds, persistent organic pollutants, fatty acids, stable isotope ratios, lipid classes, and metabolites of polycyclic aromatic compounds. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-77, 30 pp.

Sloan, C.A., D.W. Brown, R.W. Pearce, R.H. Boyer, J.L. Bolton, D.G. Burrows, D.P. Herman, and M.M. Krahn. 2004. Extraction, cleanup, and gas chromatography/mass spectrometry analysis of sediments and tissues for organic contaminants. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-59, 47 pp.

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

Thom, RM. 1992. Accretion rates of low intertidal salt marshes in the Pacific Northwest. Wetlands 12(3):147-156.

van Wezel A.P., D.A.M. de Vries, S. Kostense, D.T.H.M. Sijm, and Q. Opperhuizen. 1995. Intraspecies variation in lethal body burdens of narcotic compounds. Aquatic Toxicology 33:325–342.

Williams, J.G., S.G. Smith, J.K. Fryer, M.D. Scheurell, W.D. Muir, T.A. Flagg, R.W. Zabel, J.W. Ferguson, E. Casillas. 2014. Influence of ocean and freshwater conditions on Columbia River sockeye salmon *Oncorhynchus nerka* adult return rates. Fisheries Oceanography 23(3):210-224.

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

Zeff, M.L. 1999. Salt marsh tidal channel morphometry: applications for wetland creation and restoration. Restoration Ecology 7(2):205-211.

7 Appendices

Appendix A. Site Hydrographs

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

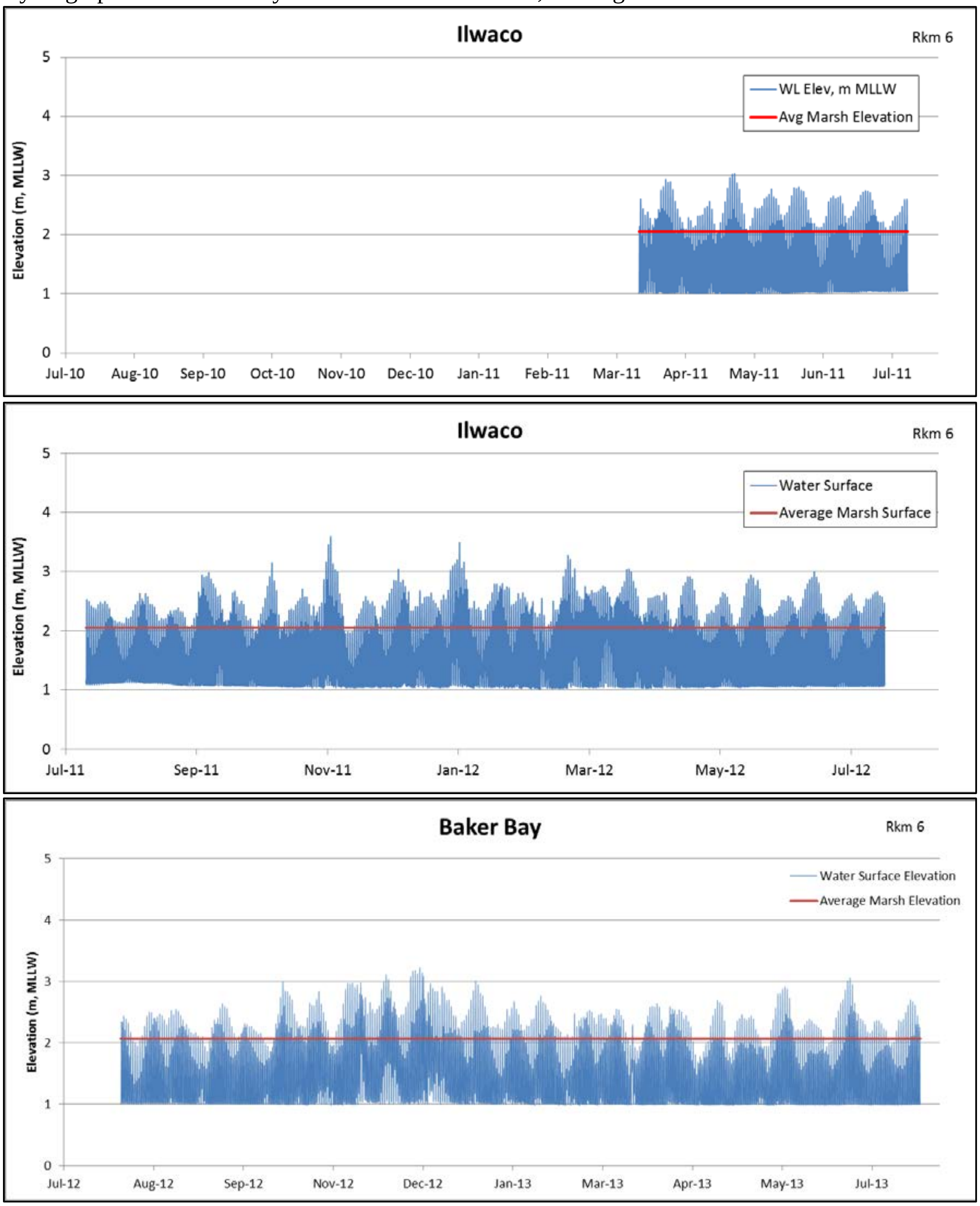
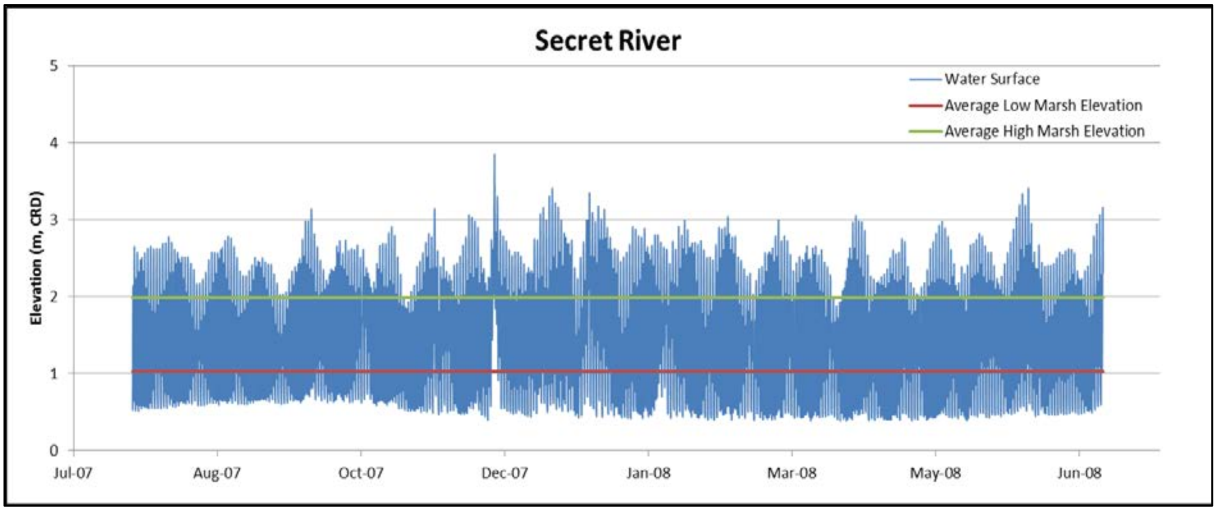
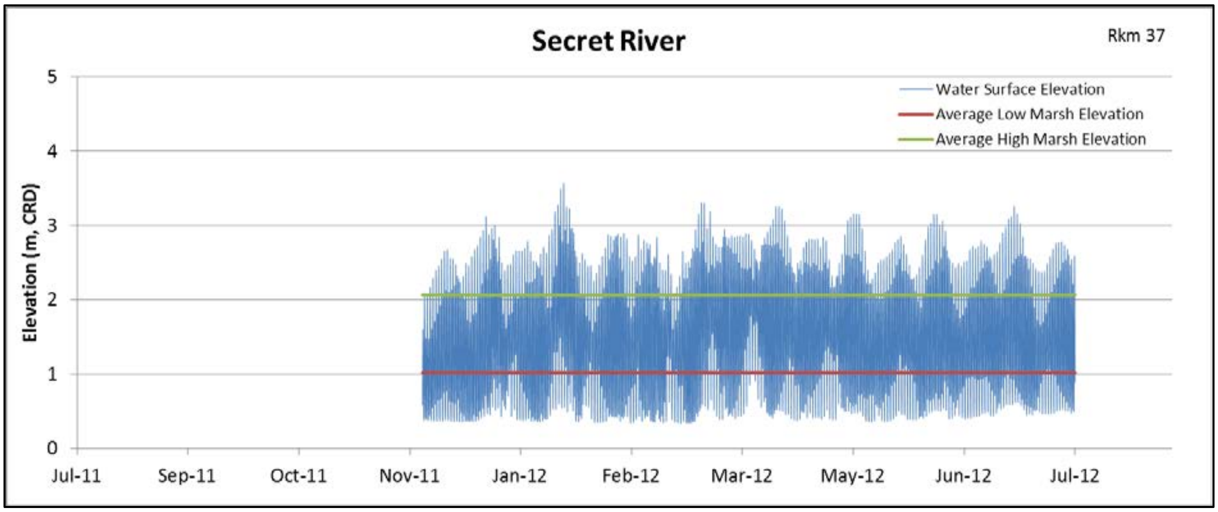
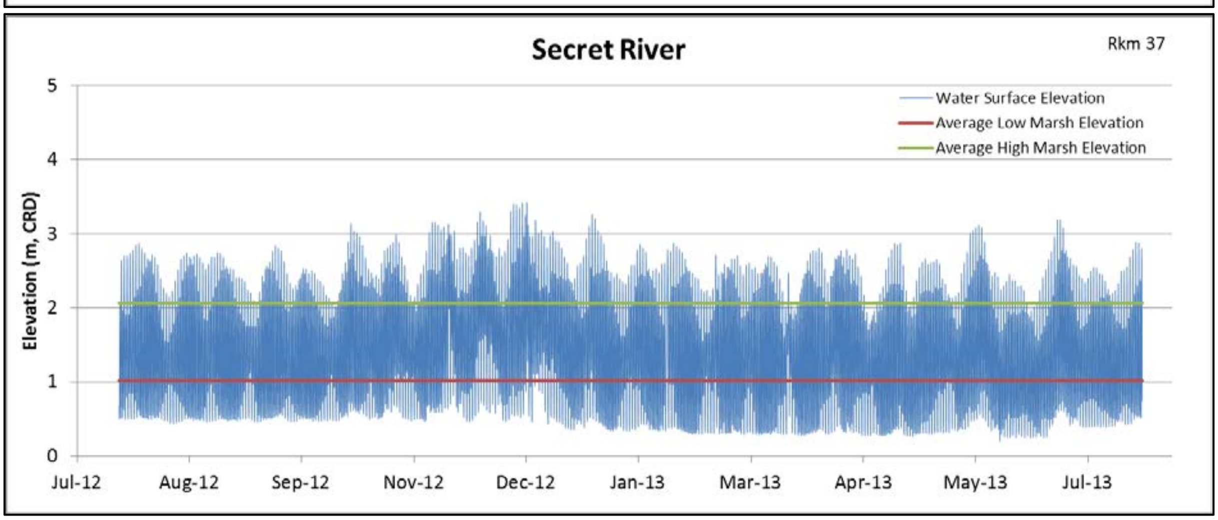


Figure A.1. Water surface elevation data from the Ilwaco Slough study site for the years 2011-2013. The red line represents the average elevation of the marsh sampling area.


Secret River
Water Surface
Average Low Marsh Elevation
Average High Marsh Elevation
Elevation (m, CRD)
0
1
2
3
4
5
Jul-07
Aug-07
Oct-07
Dec-07
Jan-08
Mar-08
May-08
Jun-08


Secret River
Rkm 37
Water Surface Elevation
Average Low Marsh Elevation
Average High Marsh Elevation
Elevation (m, CRD)
0
1
2
3
4
5
Jul-11
Sep-11
Oct-11
Nov-11
Jan-12
Feb-12
Mar-12
May-12
Jun-12
Jul-12


Secret River
Rkm 37
Water Surface Elevation
Average Low Marsh Elevation
Average High Marsh Elevation
Elevation (m, CRD)
0
1
2
3
4
5
Jul-12
Aug-12
Sep-12
Nov-12
Dec-12
Jan-13
Mar-13
Apr-13
May-13
Jul-13

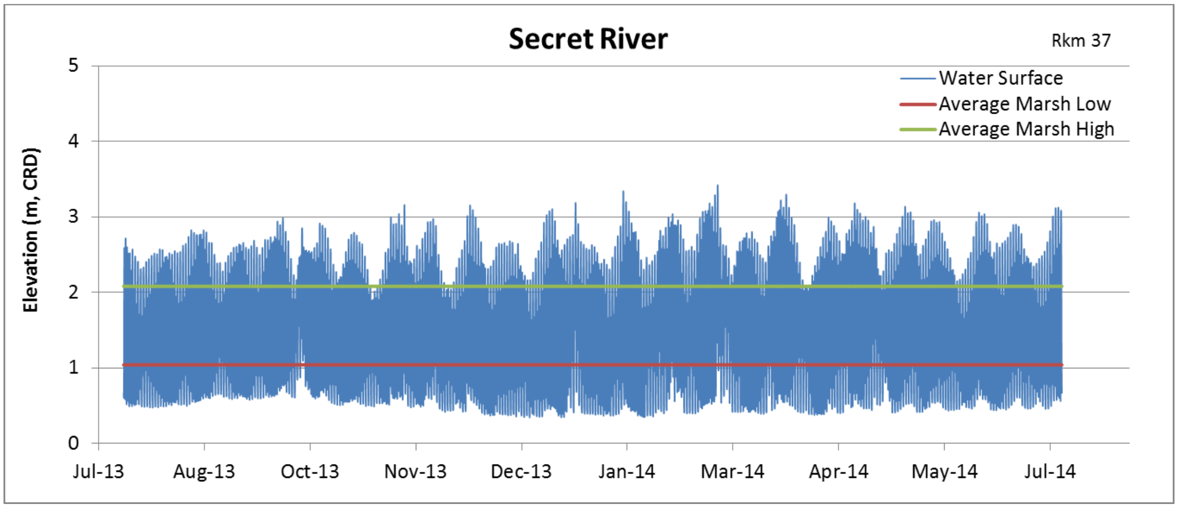


Figure A.2. Water surface elevation data from the Secret River study site for the years 2007-2008 and 2011-2014. The red line represents the average elevation of the marsh sampling area.

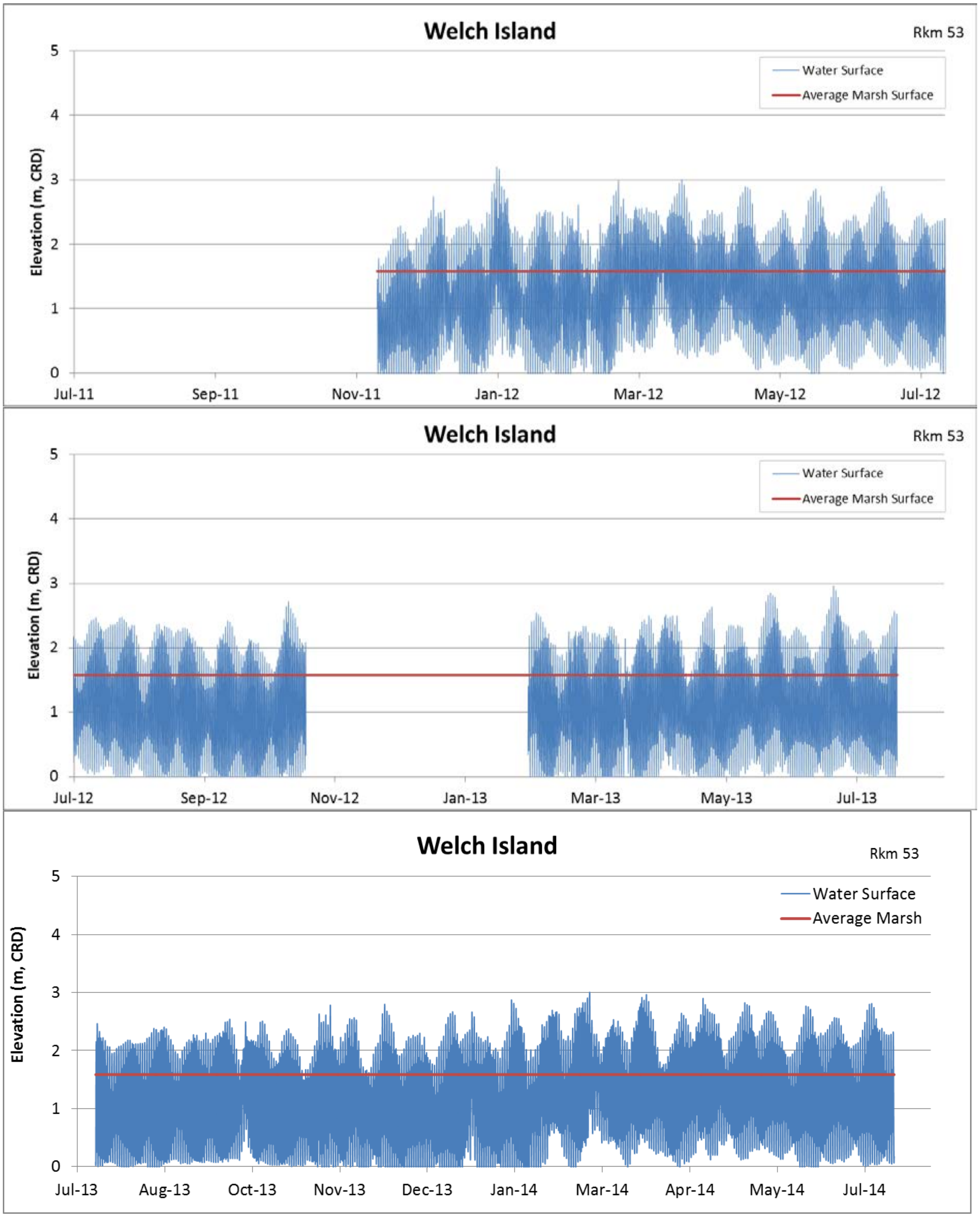
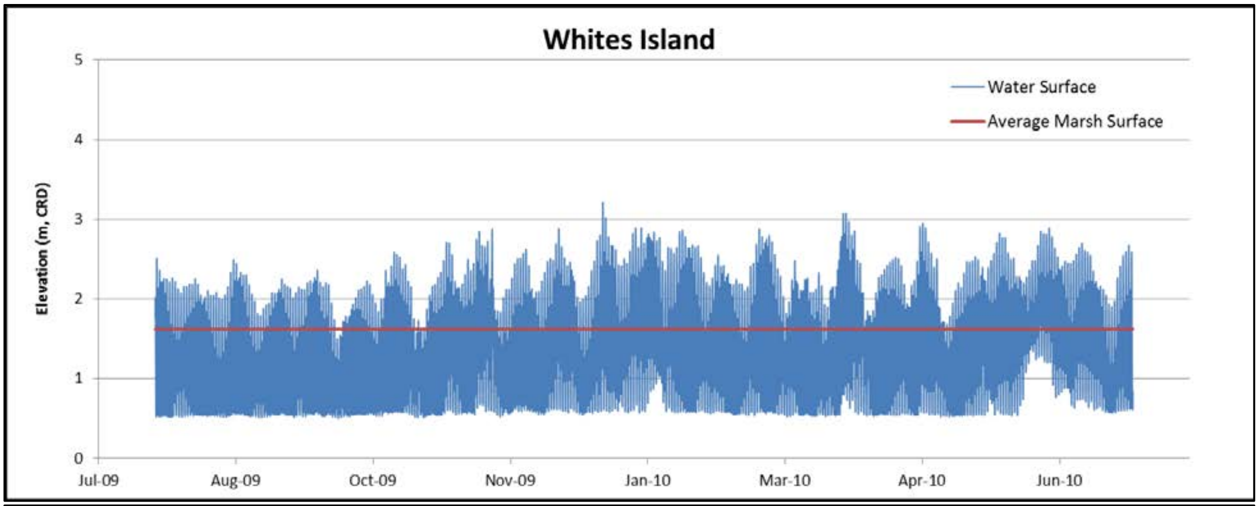
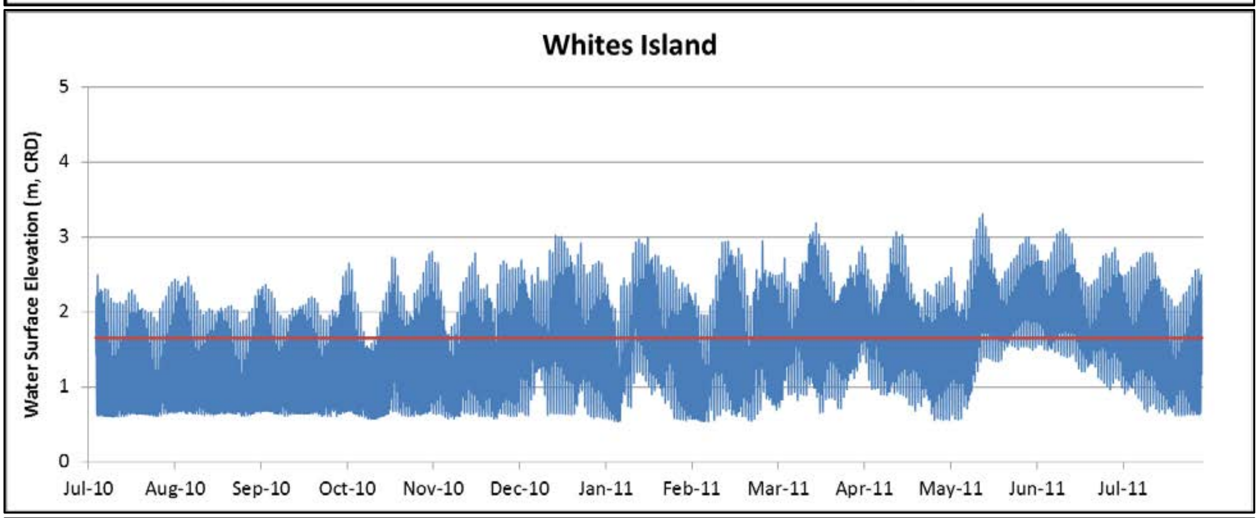
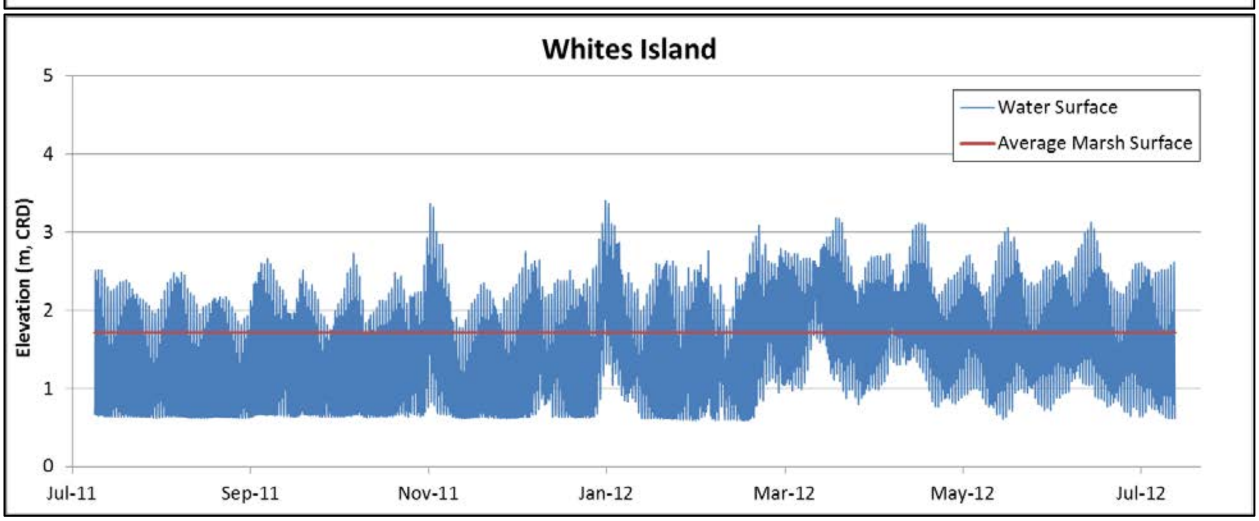


Figure A.3. Water surface elevation data from the Welch Island study site for the years 2012-2014. The red line represents the average elevation of the marsh sampling area. The sensor was displaced between early November 2012 and February 2013.


Whites Island
Water Surface
Average Marsh Surface
Elevation (m, CRD)
0
1
2
3
4
5
Jul-09
Aug-09
Oct-09
Nov-09
Jan-10
Mar-10
Apr-10
Jun-10


Whites Island
Water Surface Elevation (m, CRD)
0
1
2
3
4
5
Jul-10
Aug-10
Sep-10
Oct-10
Nov-10
Dec-10
Jan-11
Feb-11
Mar-11
Apr-11
May-11
Jun-11
Jul-11


Whites Island
Water Surface
Average Marsh Surface
Elevation (m, CRD)
0
1
2
3
4
5
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12

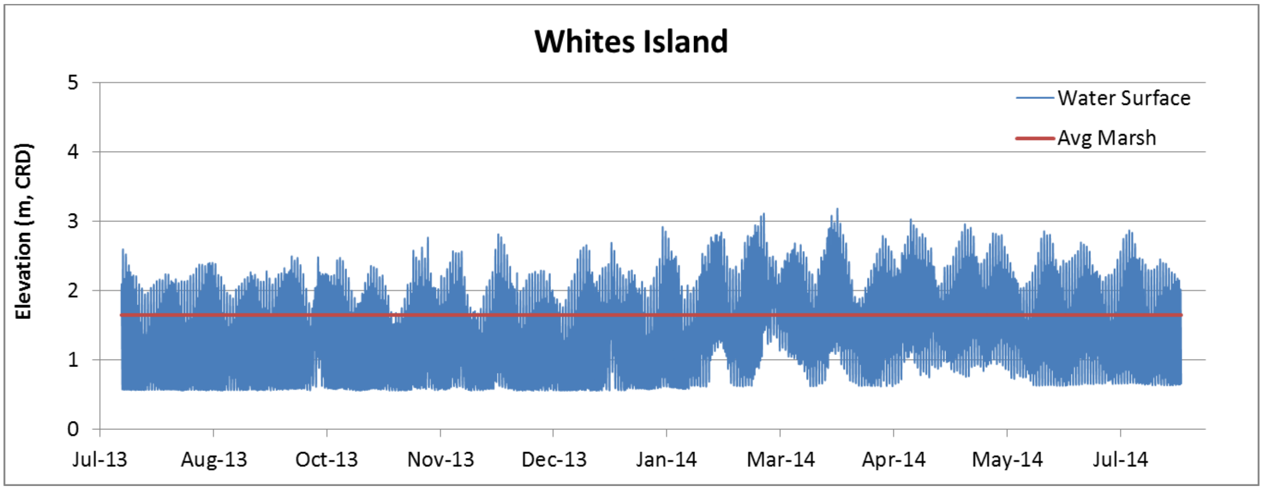
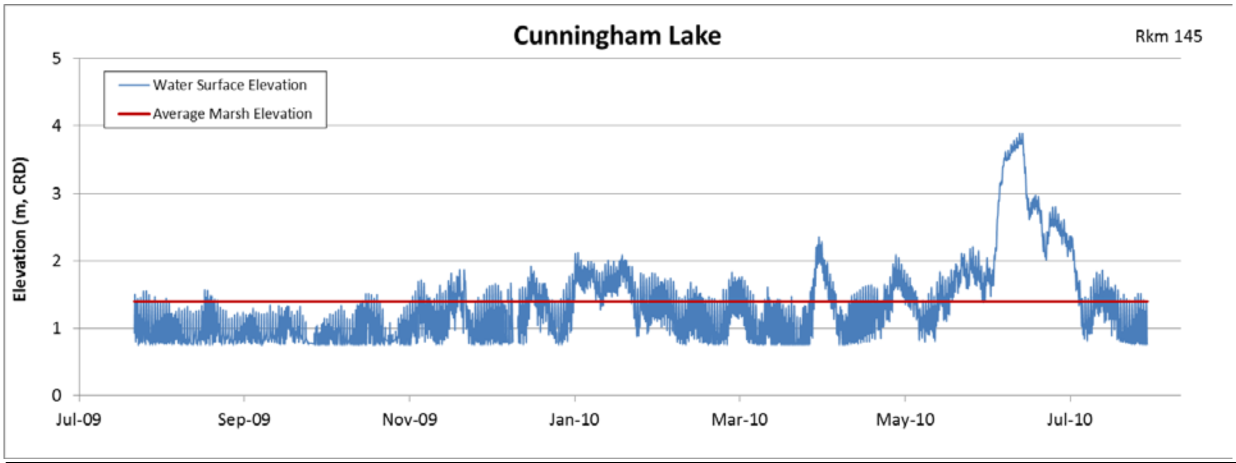
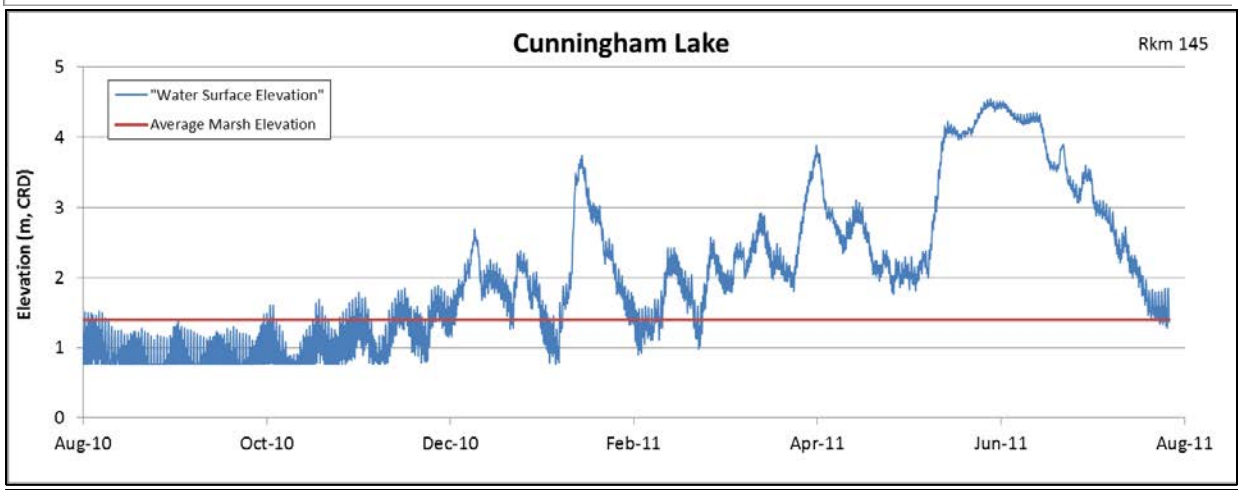
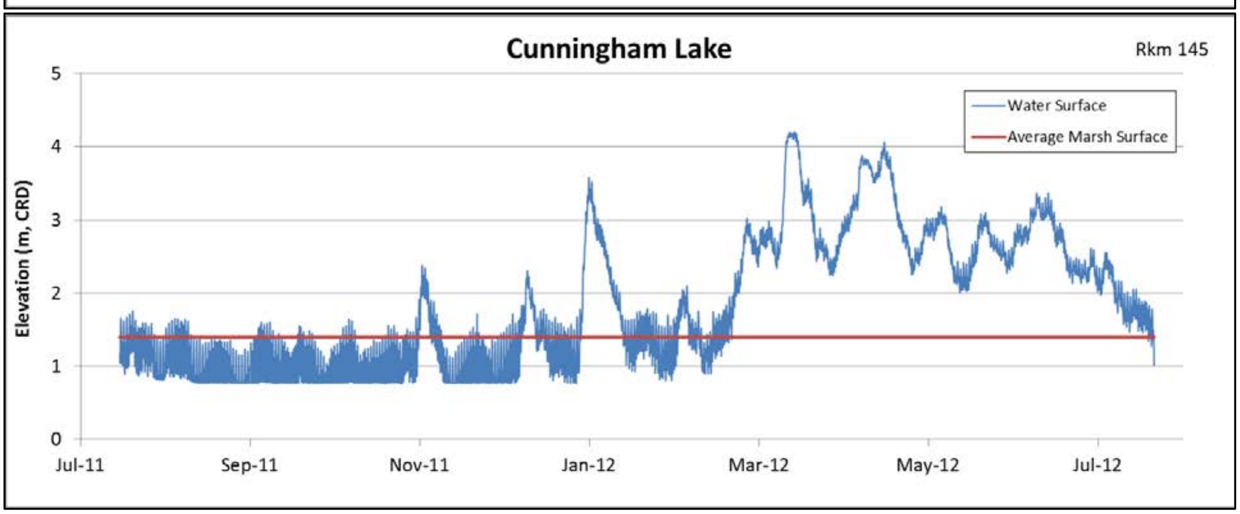


Figure A.4. Water surface elevation data from the Whites Island study site for the years 2009-2012 and 2013-2014. The red line represents the average elevation of the marsh sampling area. No data from 2013 due to sensor failure.


Cunningham Lake
Rkm 145
Water Surface Elevation
Average Marsh Elevation
Elevation (m, CRD)
5
4
3
2
1
0
Jul-09
Sep-09
Nov-09
Jan-10
Mar-10
May-10
Jul-10


Cunningham Lake
Rkm 145
"Water Surface Elevation"
Average Marsh Elevation
Elevation (m, CRD)
5
4
3
2
1
0
Aug-10
Oct-10
Dec-10
Feb-11
Apr-11
Jun-11
Aug-11


Cunningham Lake
Rkm 145
Water Surface
Average Marsh Surface
Elevation (m, CRD)
5
4
3
2
1
0
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12

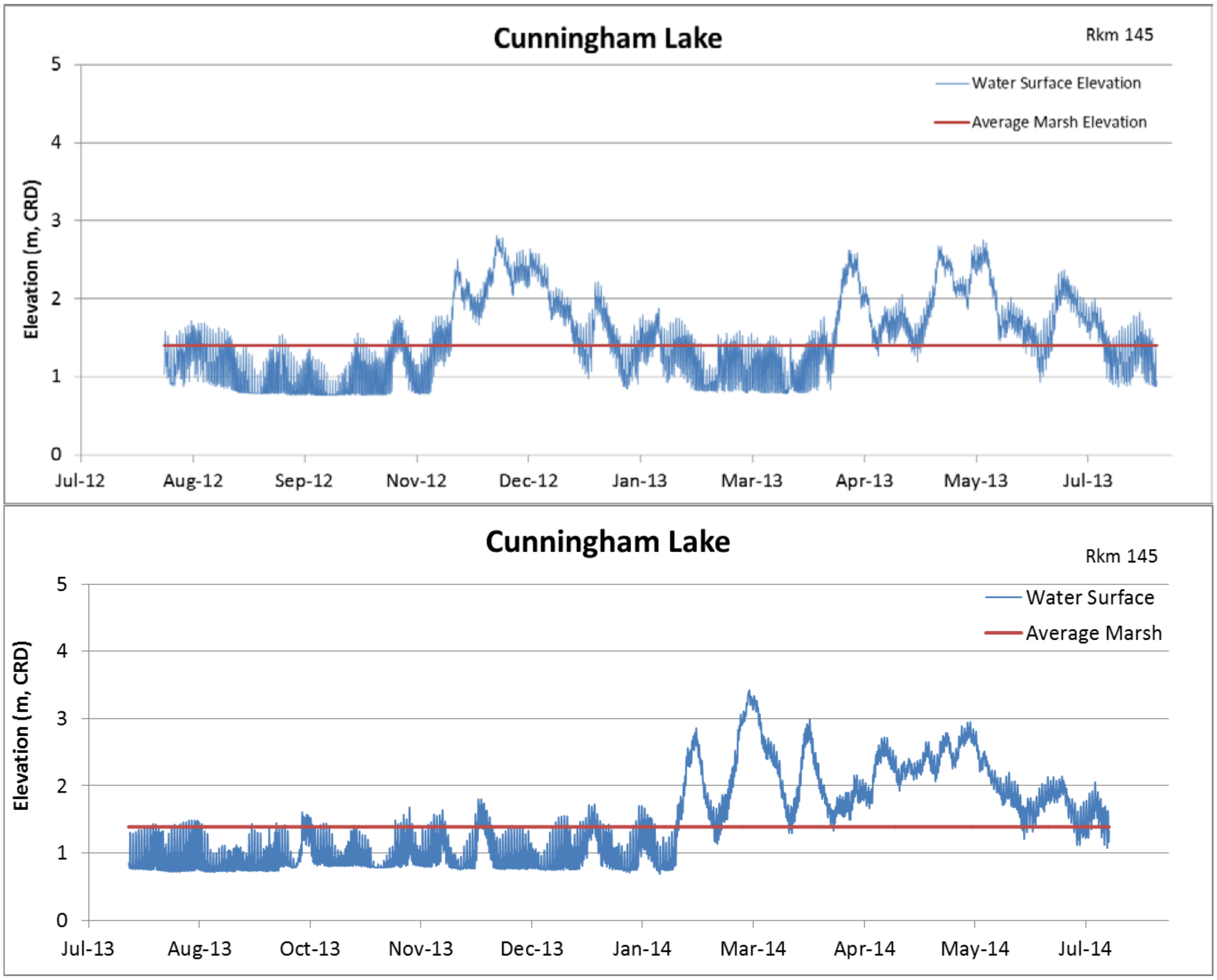
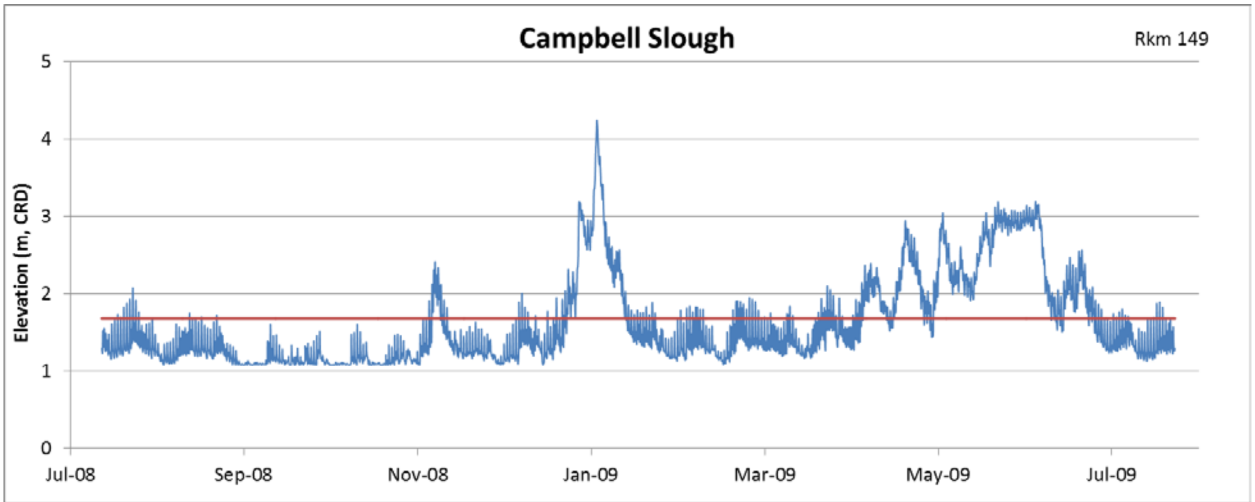
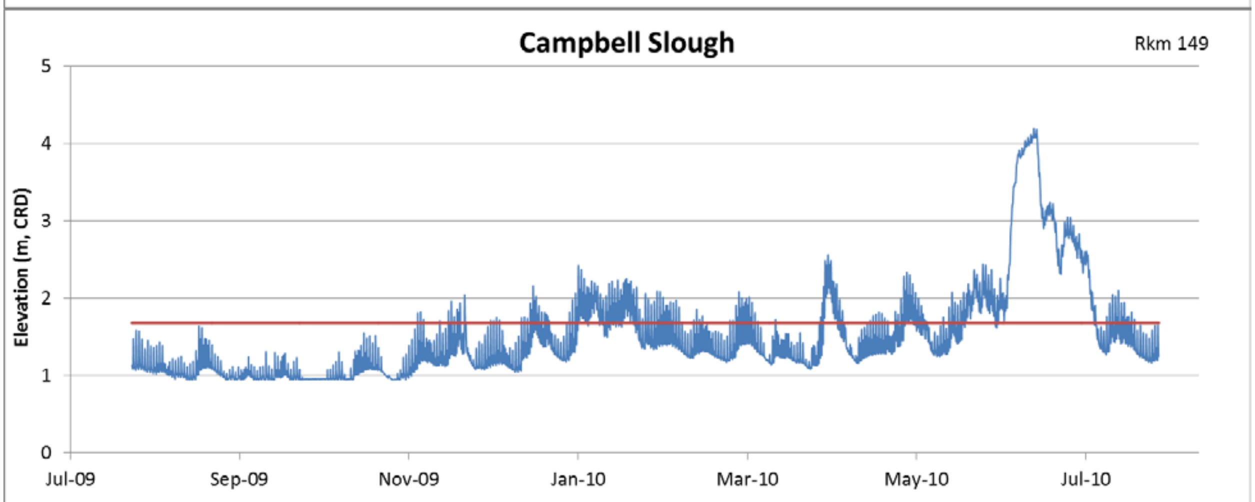
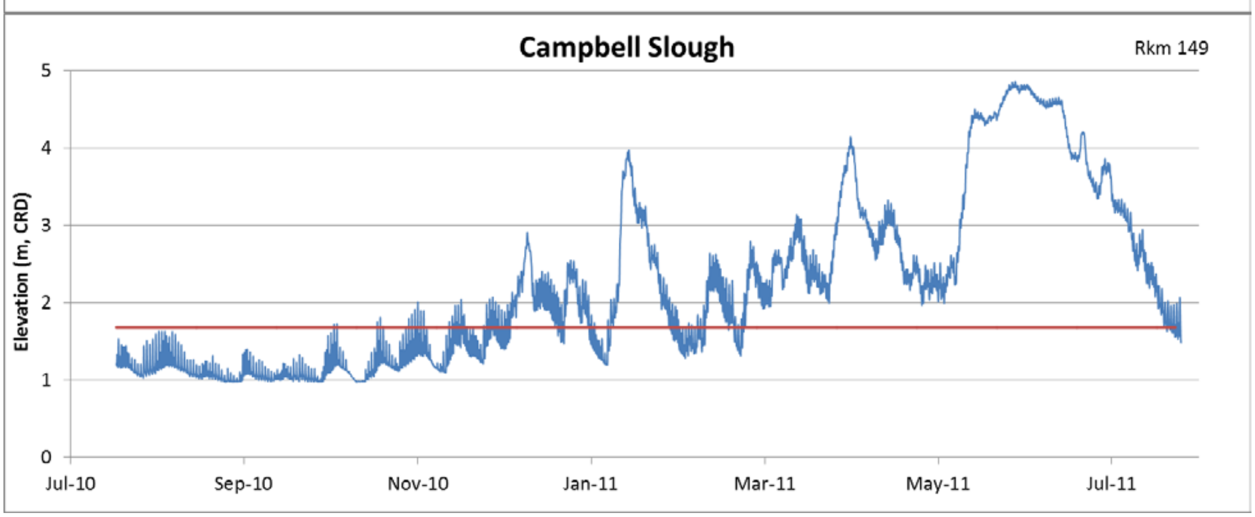


Figure A.5. Water surface elevation data from the Cunningham Lake study site for the years 2009-2014. The red line represents the average elevation of the marsh sampling area.


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


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


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

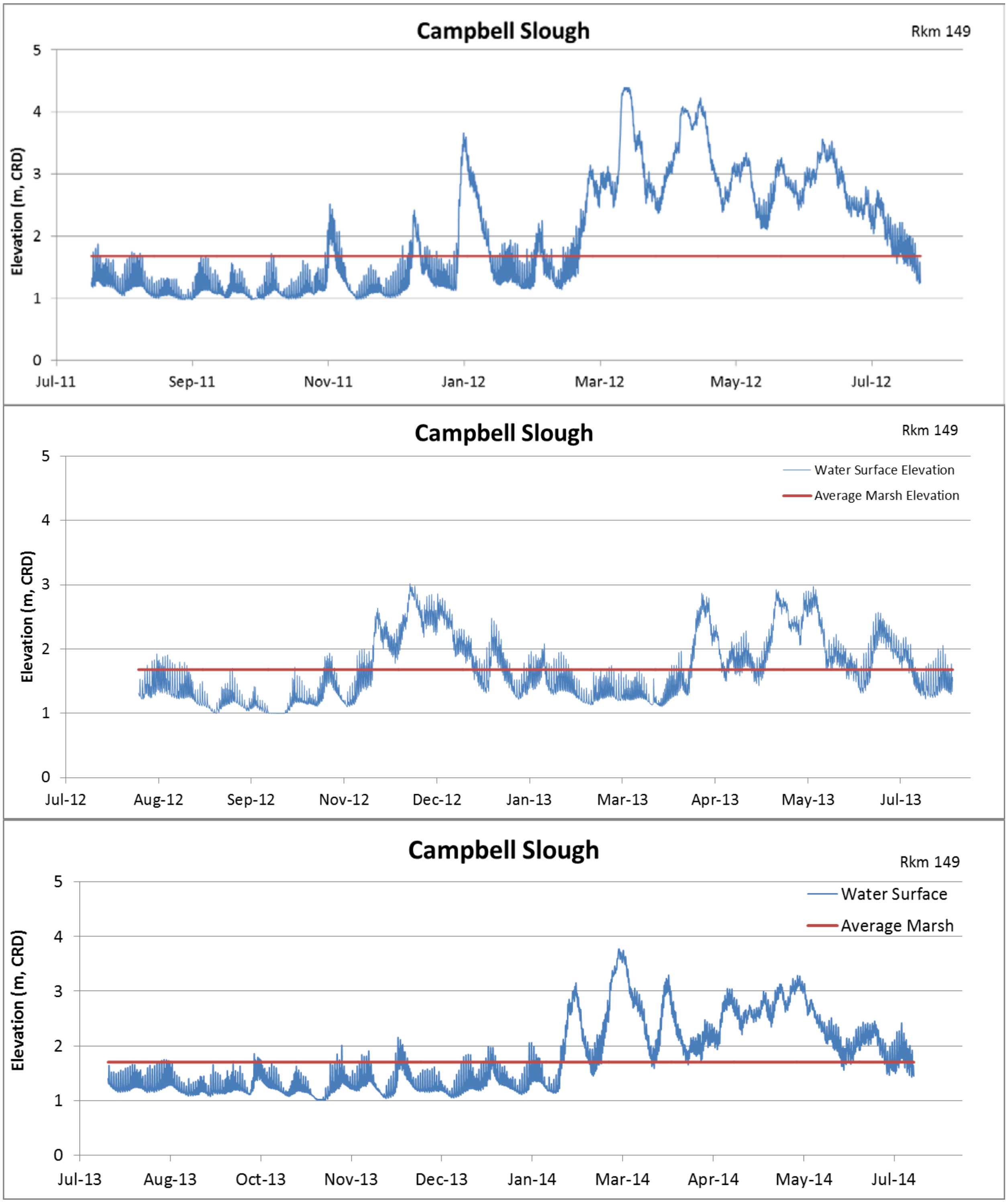
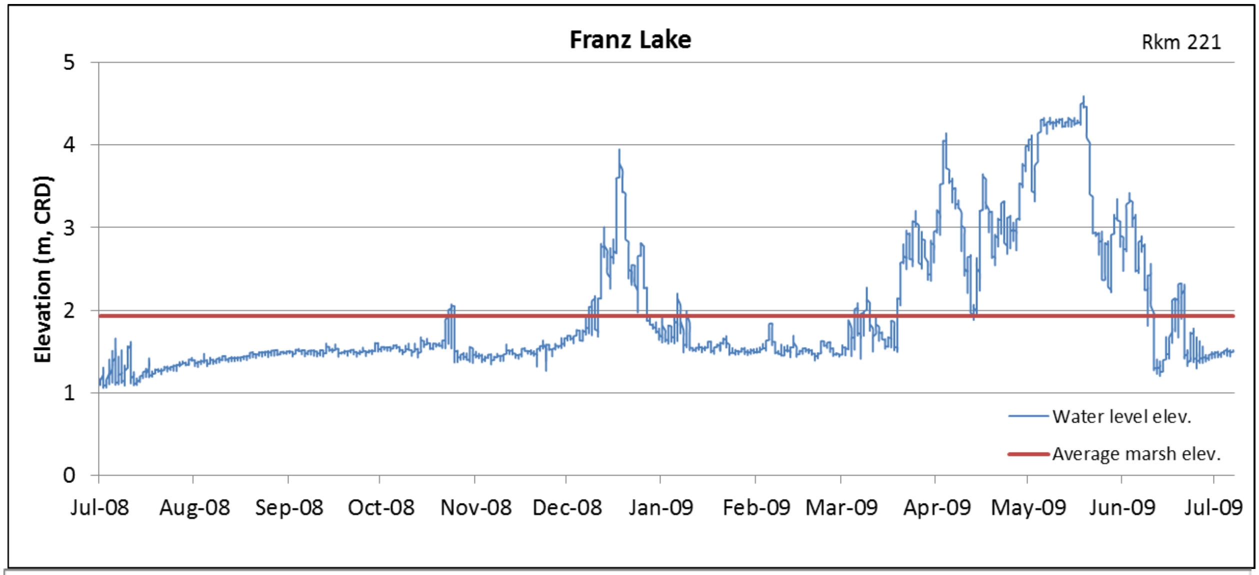
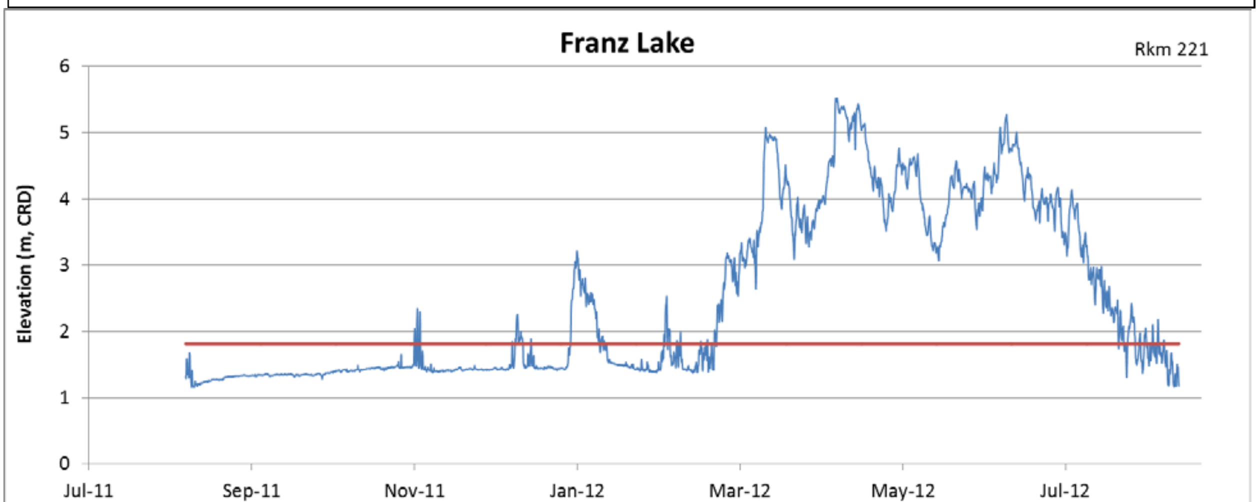


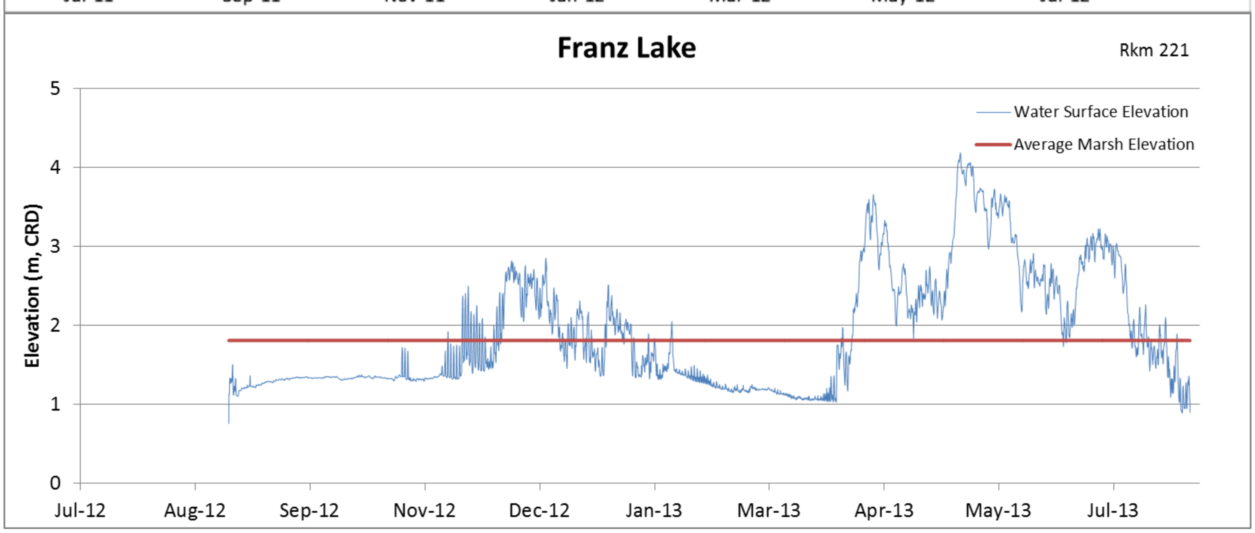
Figure A.6. Water surface elevation data from the Campbell Slough study site for the years 2008-2014. The red line represents the average elevation of the marsh sampling area.



Franz Lake
Rkm 221
Elevation (m, CRD)
5
4
3
2
1
0
Jul-08
Aug-08
Sep-08
Oct-08
Nov-08
Dec-08
Jan-09
Feb-09
Mar-09
Apr-09
May-09
Jun-09
Jul-09
Water level elev.
Average marsh elev.



Franz Lake
Rkm 221
Elevation (m, CRD)
6
5
4
3
2
1
0
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12



Franz Lake
Rkm 221
Elevation (m, CRD)
5
4
3
2
1
0
Jul-12
Aug-12
Sep-12
Nov-12
Dec-12
Jan-13
Mar-13
Apr-13
May-13
Jul-13
Water Surface Elevation
Average Marsh Elevation

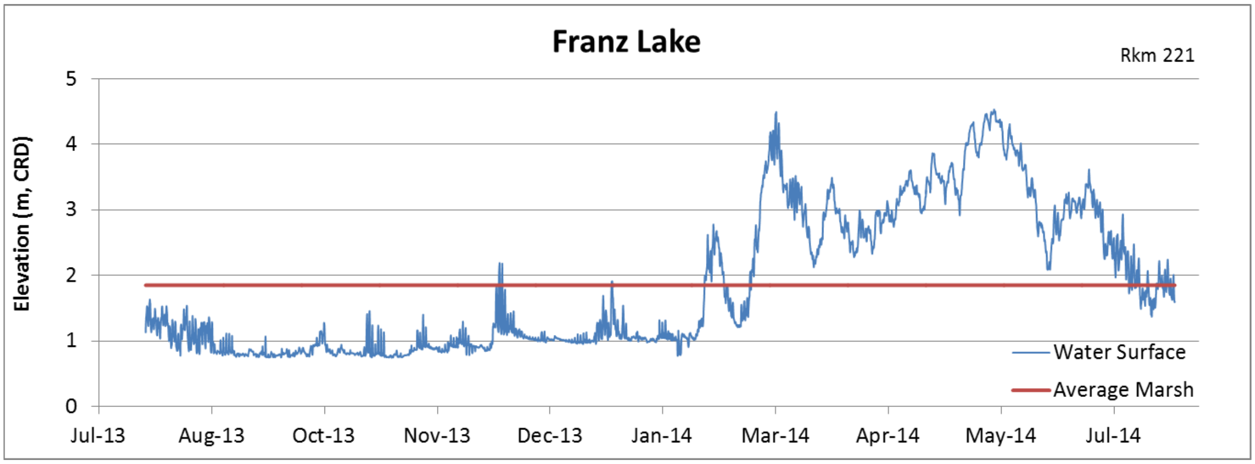
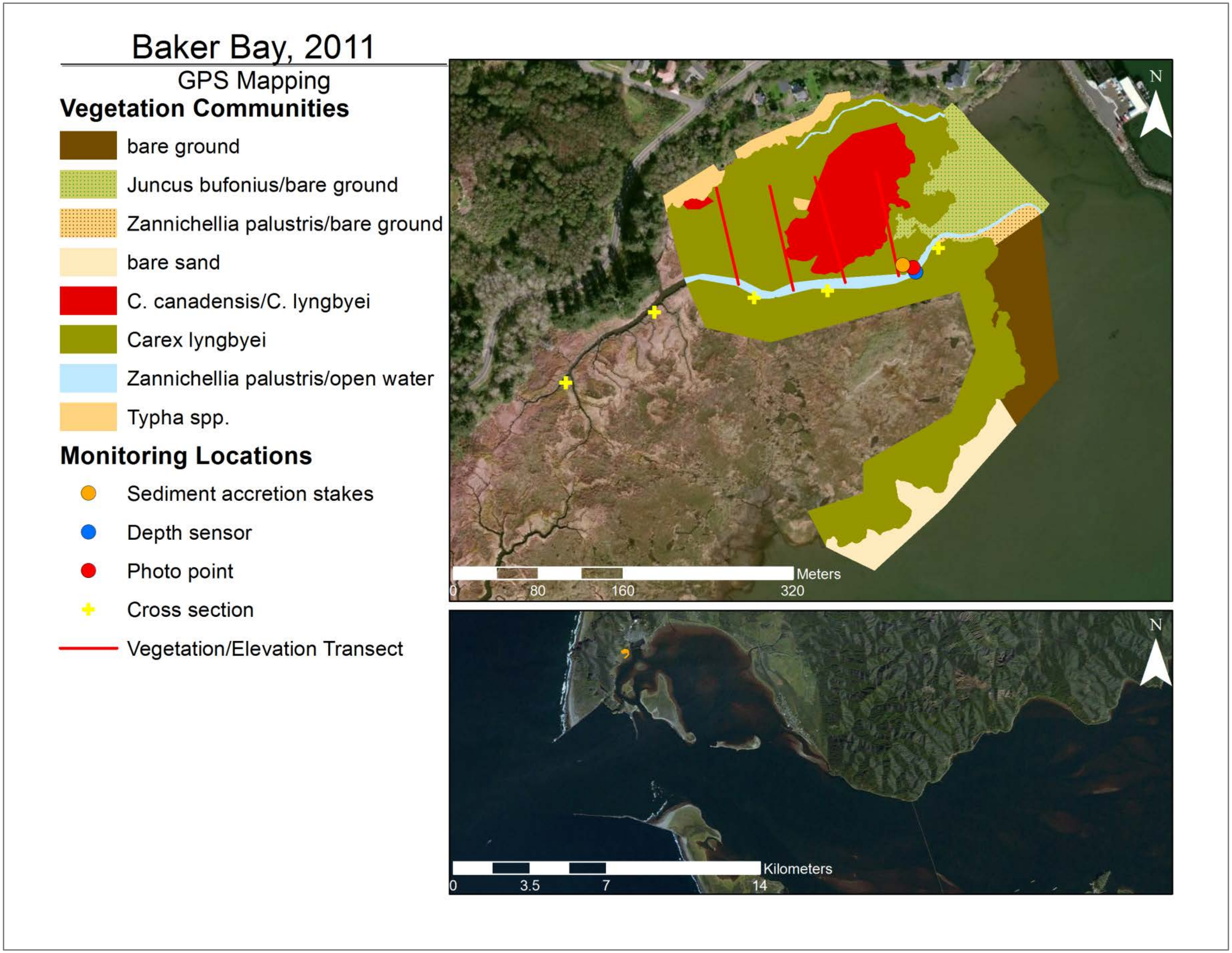


Figure A.7. Water surface elevation data from the Franz Lake study site for the years 2008-2009 and 2011-2014. The red line represents the average elevation of the marsh sampling area. Note the scale difference for the 2011-2012 plot.

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

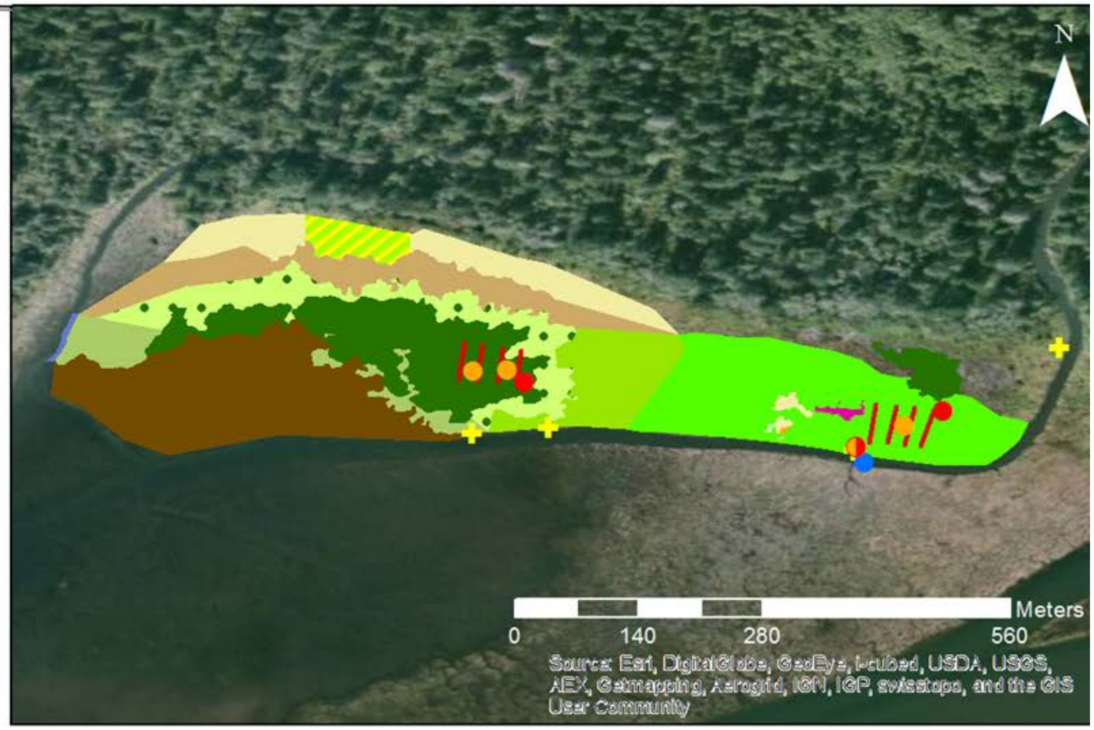
Secret River Marsh, 2013

GPS Mapping

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

Monitoring Locations

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



Welch Island, 2012

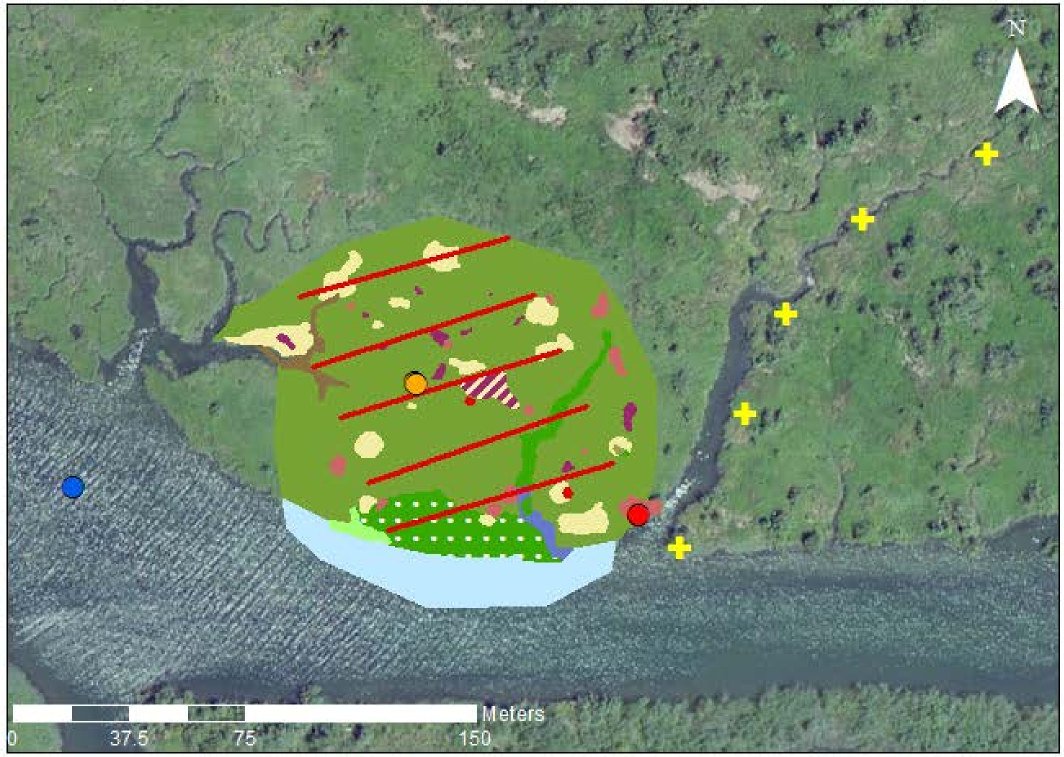
GPS Mapping

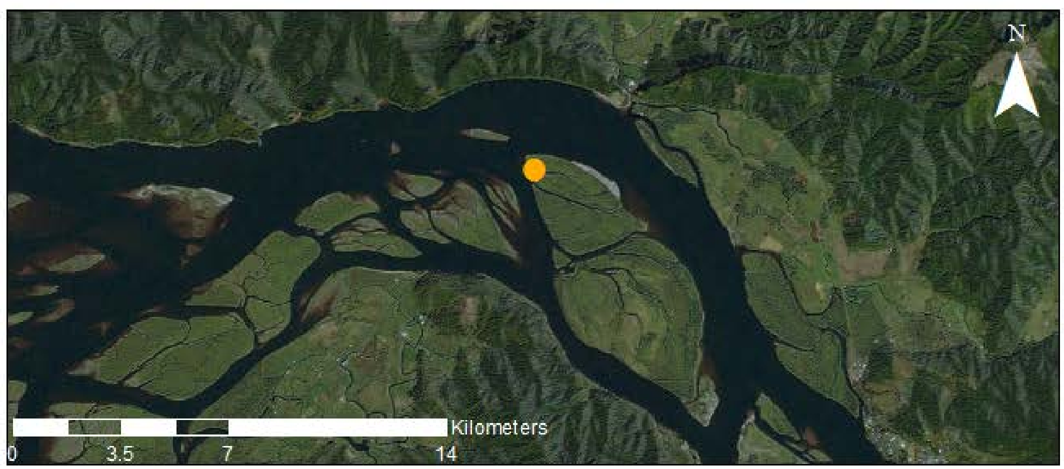
Vegetation communities

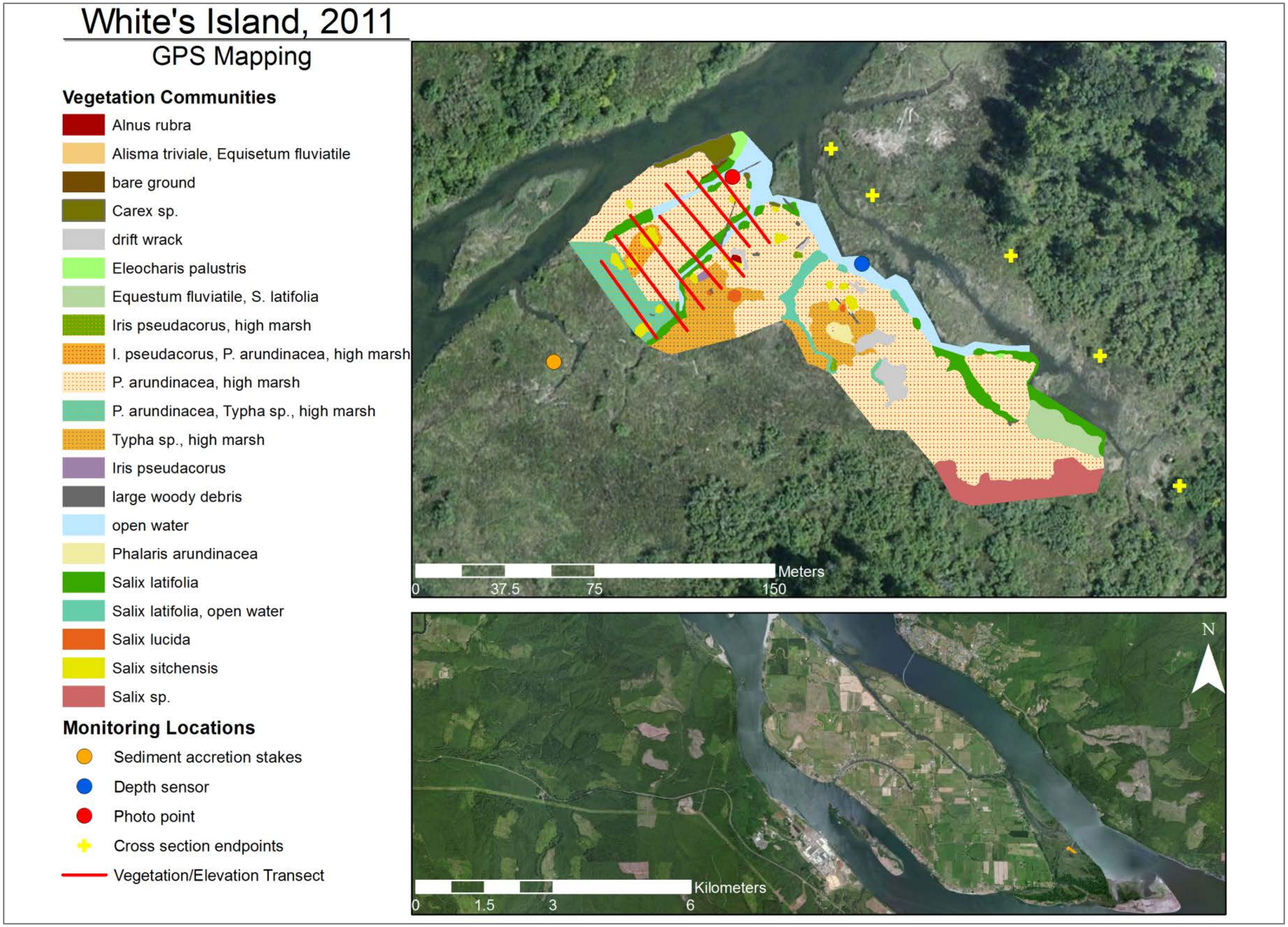
- C. obnupta, S. latifolia
- Carex lyngbyei
- Channel
- Eleocharis palustris
- Lythrum salicaria
- Open Water
- P. arundinacea, L. salicaria
- P. arundinacea, S. latifolia
- Phalaris arundinacea
- S. latifolia, P. hydropiper
- Sagittaria latifolia
- Salix spp.
- Salix spp., L. salicaria

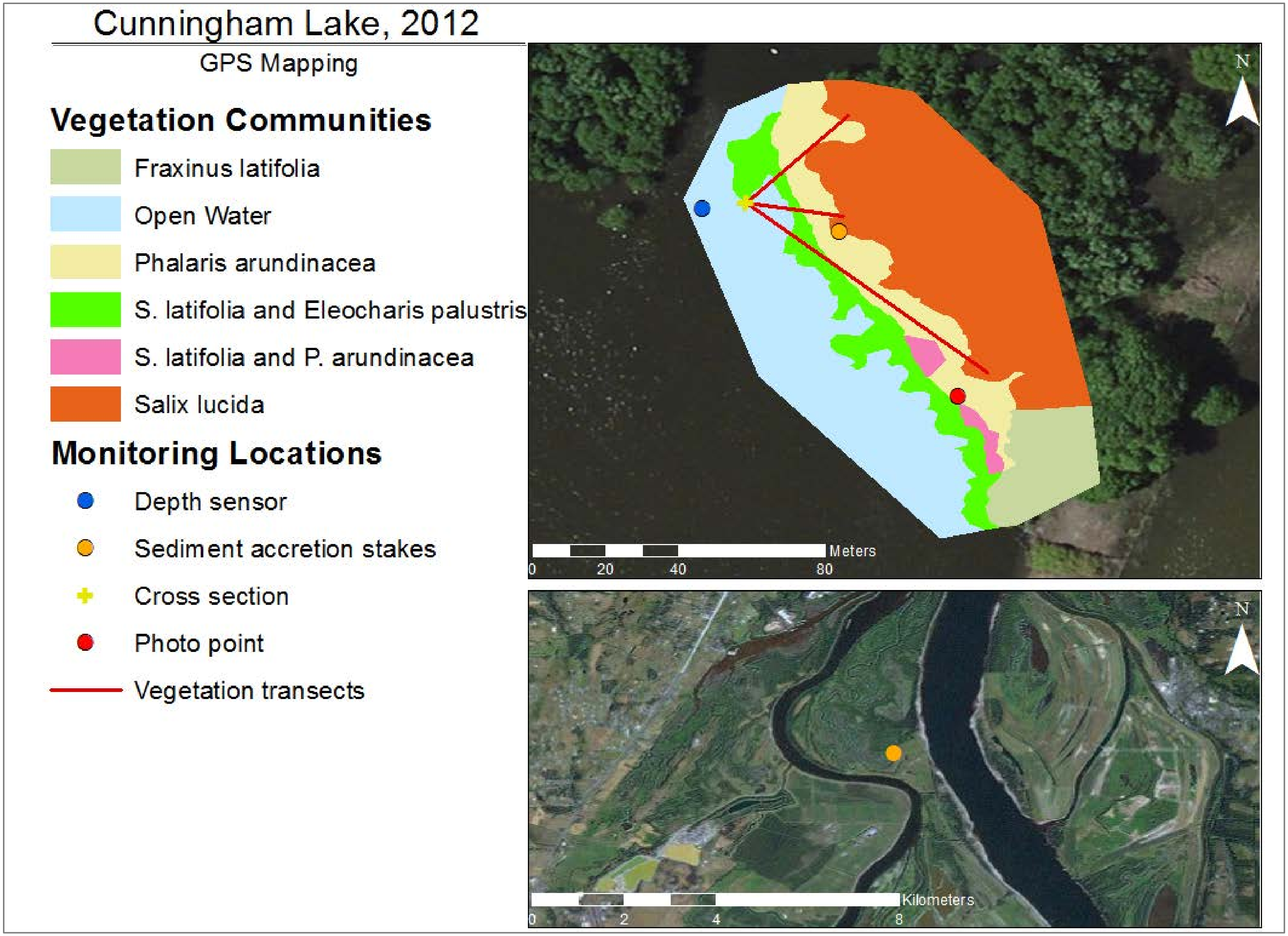
Monitoring Locations

- Sediment accretion stakes
- Depth sensor
- Cross section
- PhotoPoint
- Vegetation Survey Line

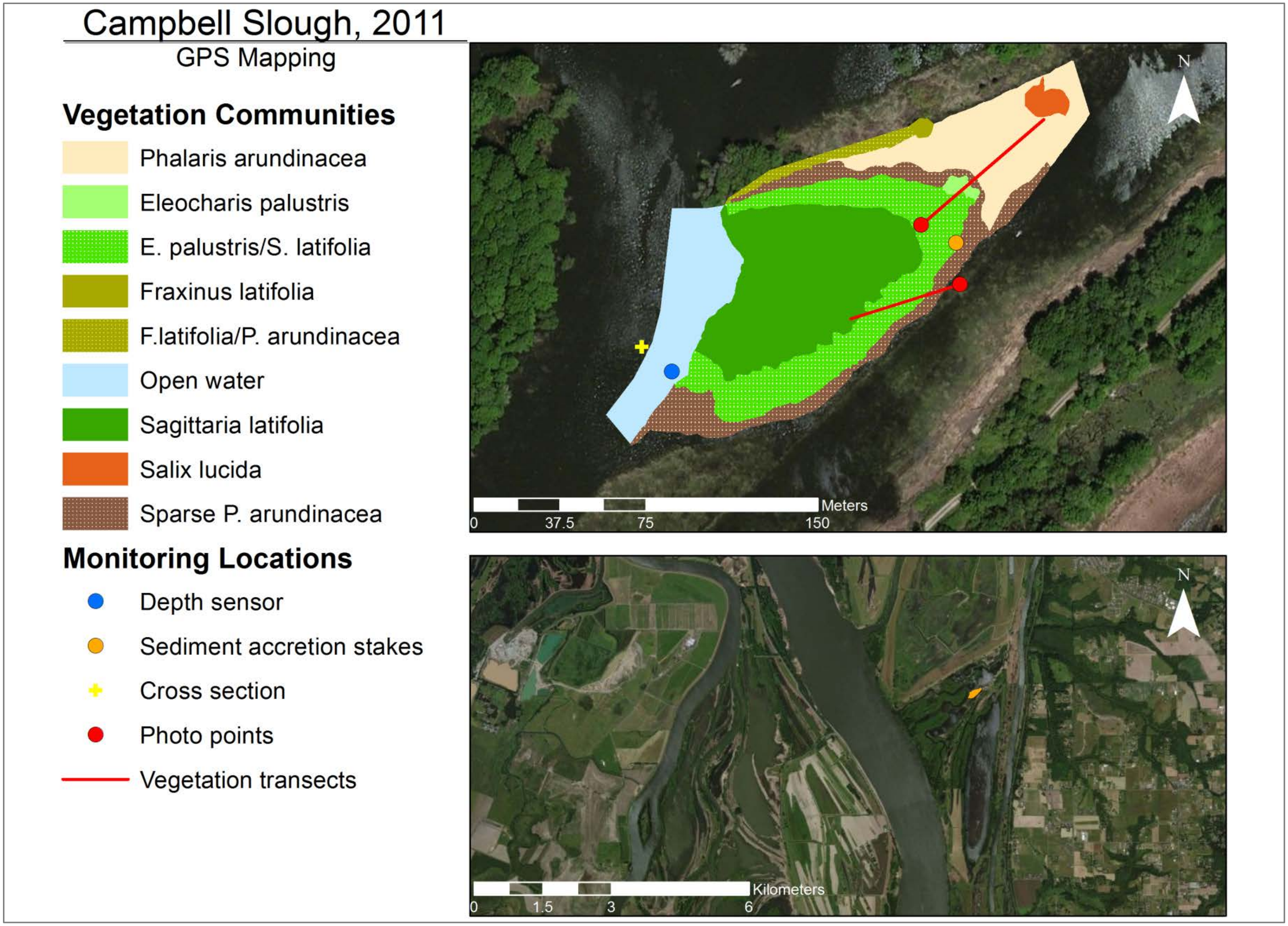


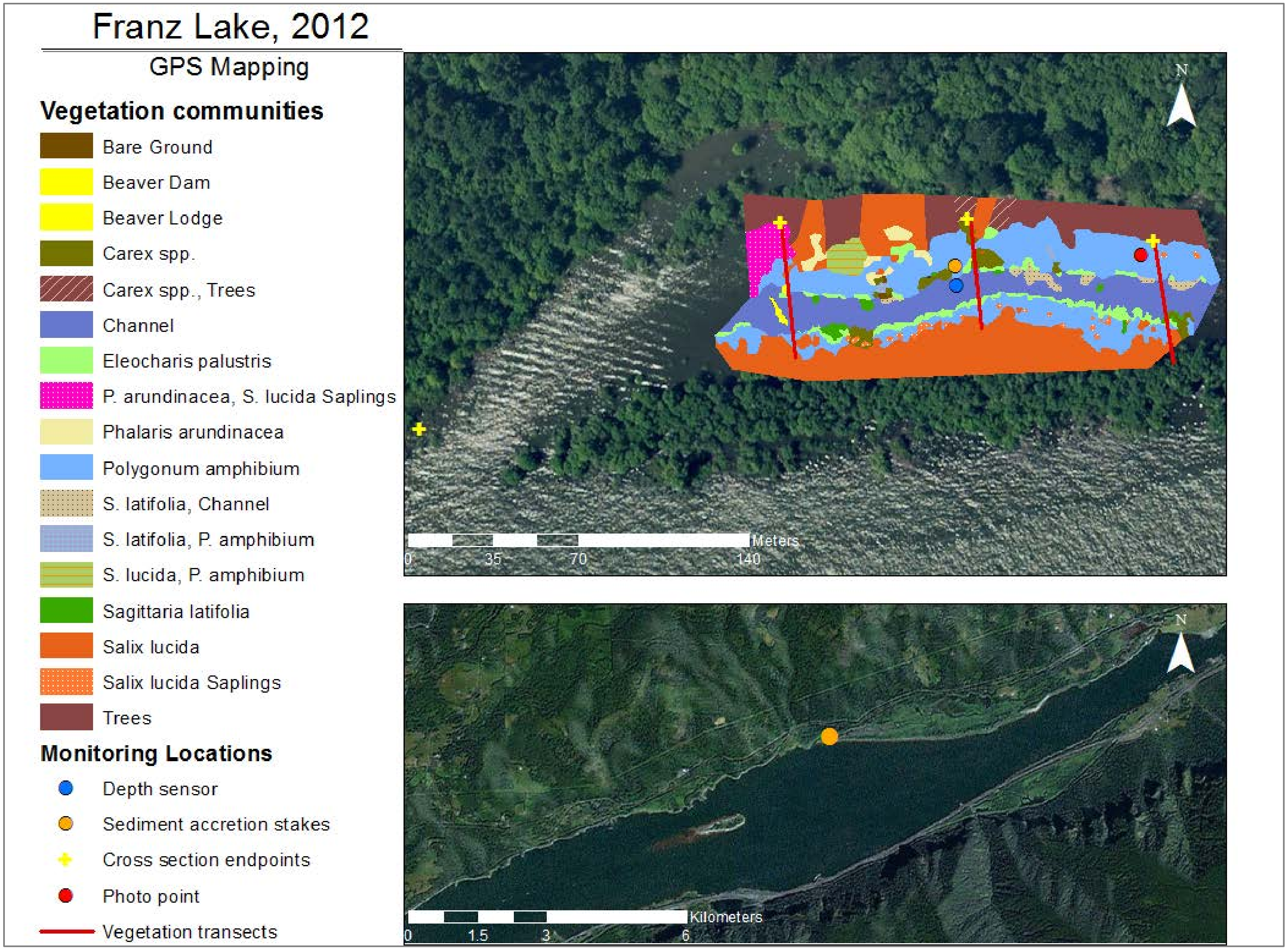



White's Island, 2011
GPS Mapping
Vegetation Communities
Alnus rubra
Alisma triviale, Equisetum fluviatile
bare ground
Carex sp.
drift wrack
Eleocharis palustris
Equestum fluviatile, S. latifolia
Iris pseudacorus, high marsh
I. pseudacorus, P. arundinacea, high marsh
P. arundinacea, high marsh
P. arundinacea, Typha sp., high marsh
Typha sp., high marsh
Iris pseudacorus
large woody debris
open water
Phalaris arundinacea
Salix latifolia
Salix latifolia, open water
Salix lucida
Salix sitchensis
Salix sp.
Monitoring Locations
Sediment accretion stakes
Depth sensor
Photo point
Cross section endpoints
Vegetation/Elevation Transect
Meters
0
37.5
75
150
N
Kilometers
0
1.5
3
6



Cunningham Lake, 2012
GPS Mapping
Vegetation Communities
Fraxinus latifolia
Open Water
Phalaris arundinacea
S. latifolia and Eleocharis palustris
S. latifolia and P. arundinacea
Salix lucida
Monitoring Locations
Depth sensor
Sediment accretion stakes
Cross section
Photo point
Vegetation transects
N
Meters
0
20
40
80
N
Kilometers
0
2
4
8


Campbell Slough, 2011
GPS Mapping
Vegetation Communities
Phalaris arundinacea
Eleocharis palustris
E. palustris/S. latifolia
Fraxinus latifolia
F.latifolia/P. arundinacea
Open water
Sagittaria latifolia
Salix lucida
Sparse P. arundinacea
Monitoring Locations
Depth sensor
Sediment accretion stakes
Cross section
Photo points
Vegetation transects
N
Meters
0
37.5
75
150
N
Kilometers
0
1.5
3
6


Franz Lake, 2012
GPS Mapping
Vegetation communities
Bare Ground
Beaver Dam
Beaver Lodge
Carex spp.
Carex spp., Trees
Channel
Eleocharis palustris
P. arundinacea, S. lucida Saplings
Phalaris arundinacea
Polygonum amphibium
S. latifolia, Channel
S. latifolia, P. amphibium
S. lucida, P. amphibium
Sagittaria latifolia
Salix lucida
Salix lucida Saplings
Trees
Monitoring Locations
Depth sensor
Sediment accretion stakes
Cross section endpoints
Photo point
Vegetation transects
N
Meters
0 35 70 140
N
Kilometers
0 1.5 3 6

Appendix C. Vegetation Species Cover

Table C.1. Site marsh elevation (in meters, relative to the Columbia River vertical datum CRD) and marsh vegetation species average percent cover from 2014. 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 Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
					Elevation (m, CRD)							
				Min	1.61	0.96	1.93	0.95	0.77	1.23	1.05	1.10
				Avg	2.00	1.04	2.08	1.58	1.65	1.70	1.39	1.85
				Max	2.38	1.22	2.20	1.72	2.10	2.72	1.65	2.33
					Average Percent Cover							
AGGI	*Agrostis gigantea*	redtop; black bentgrass	FAC	no	0.0	0.6	0.0	1.2	0.7	0.0	0.8	0.0
AGSP	*Agrostis sp.*	bentgrass	mixed	mixed	0.0	0.0	T	T	0.6	0.0	0.0	0.0
AGST	*Agrostis stolonifera L.*	creeping bentgrass	FAC	no	**7.9**	1.8	0.0	0.0	T	0.0	0.0	0.0
ALTR	*Alisma triviale*	northern water plaintain	OBL	yes	0.0	T	5.1	0.3	0.3	0.0	0.0	0.0
AREG	*Argentina egedii ssp. Egedii*	Pacific silverweed	OBL	yes	5.3	2.7	T	2.6	T	0.0	0.0	T
BICE	*Bidens cernua*	Nodding beggars-ticks	OBL	yes	0.0	0.9	2.4	3.2	0.5	0.0	0.0	0.0
CAAM	*Castilleja ambigua*	paint-brush owl-clover; johnny-nip	FACW	yes	4.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CAHE	*Callitriche heterophylla*	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.1	1.5	0.3	0.3	0.0	0.2	0.0
CALE	*Carex lenticularis*	lakeshore sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	yes	**58.9**	**42.8**	**8.3**	**44.2**	2.3	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
CAPA	*Caltha palustris*	Yellow marsh marigold	OBL	yes	0.0	8.4	0.0	3.0	0.7	0.0	0.0	0.0
CARE	*Carex retrorsa*	knotsheath sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
CASE	*Calystegia sepium*	Hedge false bindweed	FAC	no	0.0	0.5	0.0	0.0	1.1	0.0	0.0	0.0
CASP	*Carex sp.*	Carex	mixed	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.4
CASP2	*Callitriche sp.*	water-starwort	OBL	mixed	0.0	0.0	2.1	0.5	T	0.0	0.0	0.0
CEDE	*Ceratophyllum demersum*	Coontail	OBL	yes	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0
DECE	*Deschampsia cespitosa*	Tufted hairgrass	FACW	yes	1.2	T	0.0	0.1	0.0	0.0	0.0	0.0
DIAC	*Dichanthelium acuminatum*	western panicgrass	FAC	yes	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
ECCR	*Echinochloa crus-galli*	barnyardgrass	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
ELAC	*Eleocharis acicularis*	Needle spikerush	OBL	yes	2.3	0.0	1.7	0.0	T	0.1	T	0.7
ELCA	*Elodea canadensis*	Canada waterweed	OBL	yes	0.0	0.0	16.6	0.5	5.3	3.1	T	0.0
ELNU	*Elodea nuttallii*	Nuttall's waterweed, western waterweed	OBL	yes	0.0	0.0	3.9	0.0	0.0	0.0	0.0	0.0
ELPA	*Eleocharis palustris*	Common spikerush	OBL	yes	0.0	0.3	4.6	4.0	1.2	9.6	17.8	8.9
EPCI	*Epilobium ciliatum*	Willow herb	FACW	yes	0.0	0.3	0.0	1.9	0.5	0.0	0.0	0.0
EQAR	*Equisetum arvense*	field horsetail	FAC	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7
EQFL	*Equisetum fluviatile*	Water horsetail	OBL	yes	0.0	2.9	0.0	1.2	4.9	0.0	0.0	0.0
EQPA	*Equisetum palustre*	marsh horsetail	FACW	yes	0.0	0.0	0.0	0.0	0.0	2.3	T	0.2
FOAN	*Fonrinalis antipyretica*	common water moss	OBL	yes	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0
FRLA	*Fraxinus latifolia*	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
FRLA*	*Fraxinus latifolia*	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.2
FUDI	*Fucus distichus*	Rockweed	OBL	yes	5.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
GATR	*Galium trifidum L. spp. columbianum*	Pacific bedstraw	FACW	yes	0.0	0.6	0.0	0.6	0.5	0.0	0.0	0.0
GLGR	*Glyceria grandis*	American mannagrass	OBL	yes	0.0	0.0	0.0	T	T	0.0	0.0	0.0
GLMA	*Glaux maritima*	sea milkwort	OBL	yes	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GLST	*Glyceria striata*	Fowl mannagrass	OBL	yes	0.0	T	0.0	0.0	0.0	0.0	0.0	0.0
GNUL	*Gnaphalium uliginosum*	Marsh cudweed	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
GREB	*Gratiola ebracteata*	bractless hedgehyssop	OBL	yes	0.0	0.0	0.2	0.0	0.0	0.0	0.0	1.8
GRNE	*Gratiola neglecta*	American Hedge-hyssop	OBL	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
HEAU	*Helenium autumnale*	common sneezeweed	FACW	yes	0.0	0.3	0.0	0.0	0.0	0.0	0.1	0.4
HYSC	*Hypericum scouleri*	Western St. Johns wort	FACW	yes	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
IMSP	*Impatiens capensis,Impatiens noli-tangere*	western touch-me-not, common touch-me-not, jewelweed	FACW	yes	0.0	0.0	0.0	2.6	0.2	0.0	0.0	0.0
IRPS	*Iris pseudacorus*	Yellow iris	OBL	no	0.0	0.0	0.0	0.6	2.0	0.0	0.0	0.0
ISSP	*Isoetes spp.*	quillwort	OBL	yes	0.0	0.0	2.8	0.0	0.0	0.0	0.0	0.0
JUAR	*Juncus arcticus Wild. ssp. littoralis*	mountain rush	FACW	yes	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
JUOX	*Juncus oxymeris*	Pointed rush	FACW	yes	0.0	0.0	0.5	1.4	0.1	0.0	0.1	0.0
JUSP	*Juncus spp.*	Rush	mixed	mixed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
LAPA	*Lathyrus palustris*	Marsh peavine	OBL	yes	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
LEOR	*Leersia oryzoides*	Rice cutgrass	OBL	yes	0.0	0.0	0.0	0.8	1.5	0.0	1.0	2.6
LIAQ	*Limosella aquatica*	Water mudwort	OBL	yes	0.0	0.0	1.9	0.3	0.0	0.0	0.0	0.0
LIOC	*Lilaeopsis occidentalis*	Western lilaeopsis	OBL	yes	6.6	0.0	6.8	0.0	0.0	0.0	0.0	0.0
LISC	*Lilaea scilloides*	Flowering quillwort	OBL	yes	0.0	0.0	1.0	T	0.0	0.0	0.0	0.0
LOCO	*Lotus corniculatus*	Birdsfoot trefoil	FAC	no	0.0	T	0.0	1.0	1.8	0.0	0.2	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
LUPA	*Ludwigia palustris*	False loosestrife	OBL	yes	0.0	0.0	0.0	0.0	0.0	2.9	0.2	0.0
LYAM	*Lysichiton americanus*	Skunk cabbage	OBL	yes	0.0	1.0	0.0	1.5	0.0	0.0	0.0	0.0
LYAM2	*Lycopus americanus*	American water horehound	OBL	yes	0.0	0.4	0.0	0.3	1.0	0.0	0.0	0.0
LYNU	*Lysimachia nummularia L.*	Moneywort, Creeping Jenny	FACW	no	0.0	0.0	0.0	0.0	0.0	T	2.3	0.0
LYSA	*Lythrum salicaria*	Purple loosestrife	OBL	no	0.0	0.3	0.0	0.2	T	0.0	0.0	0.0
MEAR	*Mentha arvensis*	wild mint	FACW	yes	0.0	0.0	0.0	1.7	0.0	0.0	0.0	T
MIGU	*Mimulus guttatus*	Yellow monkeyflower	OBL	yes	0.0	0.4	T	2.6	0.1	0.0	0.0	0.0
MYLA	*Myosotis laxa*	Small forget-me-not	OBL	yes	0.0	5.5	0.0	3.4	0.0	0.0	0.0	0.0
MYSC	*Myosotis scorpioides*	Common forget-me-not	FACW	no	0.0	0.3	0.0	1.7	2.5	0.0	0.0	0.0
MYSI	*Myriophyllum sibiricum*	northern milfoil, short spike milfoil	OBL	yes	0.0	0.0	0.0	0.0	0.0	4.0	0.0	0.0
MYSP	*Myosotis laxa, M. scorpioides*	Small forget-me-not, Common forget-me-not	mixed	mixed	0.0	0.0	0.0	0.0	1.2	0.0	T	0.0
MYSP2	*Myriophyllum spp.*	Milfoil	OBL	mixed	0.0	0.0	0.3	0.0	0.0	0.0	0.2	0.0
OESA	*Oenanthe sarmentosa*	Water parsley	OBL	yes	0.0	11.6	T	6.6	2.7	0.0	0.0	0.0
PHAR	*Phalaris arundinacea*	Reed canarygrass	FACW	no	0.0	24.3	0.0	8.3	48.0	24.3	26.6	8.8
PLDI	*Platanthera dilatata*	white bog orchid	FACW	yes	0.0	T	0.0	T	0.0	0.0	0.0	0.0
PLMA	*Plantago major*	common plantain	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	T	T
POAM	*Polygonum amphibium*	water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	36.5
POCR	*Potamogeton crispus*	Curly leaf pondweed	OBL	no	0.0	0.0	1.1	0.0	T	0.0	0.4	0.0
POHY	*Polygonum hydropiper, P. hydropiperoides*	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	1.3	0.1	1.5	0.2	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
PONA	*Potamogeton natans*	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.4	0.3	0.0
POPE	*Polygonum persicaria*	Spotted ladysthumb	FACW	no	0.0	T	3.9	0.5	0.3	0.3	0.2	0.0
POPU	*Potamogeton pusillus*	Small pondweed	OBL	yes	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0
PORI	*Potamogeton richardsonii*	Richardson's pondweed	OBL	yes	0.0	0.0	0.1	0.0	1.0	0.0	0.0	0.0
POSP	*Polygonum sp.*	Knotweed, Smartweed	mixed	mixed	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0
RARE	*Ranunculus repens*	Creeping buttercup	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
ROCU	*Rorippa curvisiliqua*	curvepod yellow cress	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
RUAR	*Rubus armeniacus*	Himalayan blackberry	FACU	no	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
RUMA	*Rumex maritimus*	Golden dock, seaside dock	FACW	yes	0.0	0.4	0.0	0.1	0.0	0.0	T	0.0
SALA	*Sagittaria latifolia*	Wapato	OBL	yes	0.0	1.6	0.2	5.4	8.6	1.5	3.8	2.1
SALU	*Salix lucida*	Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8
SALU*	*Salix lucida*	Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	16.7	T	8.8
SASI	*Salix sitchensis*	Sitka willow	FACW	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.7
SASI*	*Salix sitchensis*	Sitka willow	FACW	yes	0.0	0.0	0.0	0.0	3.6	0.0	0.0	2.8
SCAM	*Schoenoplectus americanus*	American bulrush, threesquare bulrush	OBL	yes	4.5	0.0	0.0	0.0	T	0.0	0.0	0.0
SCAR	*Schedonorus arundinaceus*	tall fescue	FAC	no	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCMA	*Schoenoplectus maritimus*	Seacoast bulrush	OBL	yes	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCTA	*Schoenoplectus tabernaemontani*	Softstem bulrush, tule	OBL	Yes	0.0	T	6.9	0.0	T	0.0	0.0	T
SISU	*Sium suave*	Hemlock waterparsnip	OBL	yes	0.0	1.1	0.0	3.5	0.2	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
SODU	*Solanum dulcamara*	Bittersweet nightshade	FAC	no	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
SPAN	*Sparganium angustifolium*	Narrowleaf burreed	OBL	yes	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0
SPEU	*Sparganium eurycarpum*	giant burreed	OBL	yes	0.0	0.0	2.3	0.0	0.0	0.7	0.0	0.0
SYSU	*Symphyotrichum subspicatum*	Douglas aster	FACW	yes	0.6	0.9	0.3	1.5	0.1	0.0	0.0	0.0
TRMA	*Triglochin maritima*	Seaside arrowgrass	OBL	yes	6.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TRWO	*Trifolium wormskioldii*	Springbank clover	FACW	yes	0.0	0.0	0.0	T	0.0	0.0	0.0	0.0
TYAN	*Typha angustifolia*	Narrowleaf cattail	OBL	no	1.0	0.0	0.0	T	2.0	0.0	0.0	0.0
VEAM	*Veronica americana*	American speedwell	OBL	yes	0.0	0.0	0.0	0.0	T	0.0	0.5	0.0
VEAN	*Veronica anagallis-aquatica*	water speedwell	OBL	yes	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0
ZAPA	*Zannichellia palustris*	horned pondweed	OBL	yes	1.1	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Other Cover												
Algae		algae			3.9	0.0	4.5	0.0	T	18.3	38.0	0.1
BG		bare ground			5.0	1.3	25.5	6.3	9.3	36.7	44.9	26.4
Detritus		detritus			3.4	7.6	0.8	0.4	2.5	6.8	0.3	0.7
dSALU*	*Salix lucida*, dead	Pacific willow, dead	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0
DW		drift wrack			2.4	0.1	0.0	0.1	3.8	T	0.0	0.0
Litter		litter			T	0.0	0.0	T	0.4	1.4	T	2.3
LWD		large woody debris			0.0	0.3	0.0	0.0	2.1	2.5	0.0	2.5
Moss		moss			0.0	0.0	0.0	0.0	0.8	0.0	0.7	0.0
OW		open water			5.6	4.6	35.8	5.0	11.0	46.8	34.8	2.3
PHAR-d	*Phalaris arundinacea*, dead	Reed canarygrass, dead	FACW	no	0.0	0.0	0.0	0.0	1.2	1.3	0.8	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco Slough	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
SAP		saplings			0.0	0.0	0.0	T	0.0	0.0	T	0.0
SD		standing dead			0.0	0.1	0.3	0.0	0.1	0.0	0.0	0.2
SH		shell hash			0.0	0.0	0.0	0.0	T	0.0	0.0	0.0
SMH		small mixed herbs			T	0.0	0.0	T	0.1	T	0.0	4.4
SWD		small woody debris			0.0	0.0	0.0	0.0	0.0	2.1	0.0	0.0

T = Trace

Table C.2. Site channel elevation (in meters, relative to the Columbia River vertical datum CRD) and submerged aquatic vegetation (SAV) species average percent cover from 2014. The three dominant cover classes are bolded in red for each site and non-native species are shaded in yellow. Overhanging tree and shrub species are not included in identification of dominant cover. Species are sorted by their four letter code (1st two letters of genus and 1st two letters of species). Channel data was included in the marsh data for the Cunningham Lake site.

Code	Scientific Name	Common Name	Wetland Status	Native	Secret River	Welch	Whites	Campbell	Franz
					Elevation (m, CRD)				
				Min	0.13	0.21	0.14	0.67	0.50
				Avg	0.36	0.44	0.35	0.88	1.02
				Max	0.69	0.56	0.61	1.05	1.64
					Average Percent Cover				
ALTR	Alisma triviale	northern water plaintain	OBL	yes	0.0	0.2	0.0	0.0	0.0
BICE	Bidens cernua	Nodding beggars-ticks	OBL	yes	0.0	0.3	0.0	0.0	0.0
CAHE	Callitriche heterophylla	Water starwort; Twoheaded water starwort	OBL	yes	0.0	**4.3**	T	0.0	0.0
CASP2	Callitriche sp.	water-starwort	OBL	mixed	0.0	0.0	T	0.0	0.0
CEDE	Ceratophyllum demersum	Coontail	OBL	yes	0.0	0.3	0.0	0.0	0.0
ELAC	Eleocharis acicularis	Needle spikerush	OBL	yes	0.1	0.0	0.0	0.0	0.0
ELCA	Elodea canadensis	Canada waterweed	OBL	yes	**2.6**	**41.7**	0.5	0.0	0.0
ELPA	Eleocharis palustris	Common spikerush	OBL	yes	0.0	1.0	**2.2**	0.0	**0.4**
ELPAR	Eleocharis parvula	Dwarf spikerush	OBL	yes	0.0	0.2	0.0	0.0	0.0
EQFL	Equisetum fluviatile	Water horsetail	OBL	yes	0.0	0.3	0.0	0.0	0.0
IRPS	Iris pseudacorus	Yellow iris	OBL	no	0.0	0.0	T	0.0	0.0
LEOR	Leersia oryzoides	Rice cutgrass	OBL	yes	0.0	0.0	0.0	0.0	**0.1**
LIAQ	Limosella aquatica	Water mudwort	OBL	yes	0.0	0.2	T	0.0	0.0
MYSP	Myosotis laxa, M. scorpioides	Small forget-me-not, Common forget-me-not	mixed	mixed	0.0	**5.8**	**0.7**	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Secret River	Welch	Whites	Campbell	Franz
POAM	Polygonum amphibium	water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.1
POCR	Potamogeton crispus	Curly leaf pondweed	OBL	no	0.1	0.0	0.6	0.2	0.0
PONA	Potamogeton natans	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.2	0.0
POPE	Polygonum persicaria	Spotted ladysthumb	FACW	no	0.0	2.5	0.0	0.0	0.0
POPU	Potamogeton pusillus	Small pondweed	OBL	yes	0.1	0.2	0.0	0.2	0.0
PORI	Potamogeton richardsonii	Richardson's pondweed	OBL	yes	55.6	2.8	27.3	0.0	0.0
POZO	Potamogeton zosteriformis	Eelgrass pondweed	OBL	yes	0.0	0.0	T	0.0	0.0
SALA	Sagittaria latifolia	Wapato	OBL	yes	0.0	0.2	T	0.0	4.3
ZAPA	Zannichellia palustris	horned pondweed	OBL	yes	0.0	0.0	0.5	0.0	0.0
Other Cover									
Algae		algae			16.3	5.8	18.3	7.5	20.1
BG		bare ground			38.1	47.5	61.4	50.0	95.7
Detritus		detritus			0.6	0.0	T	0.0	0.0
DW		drift wrack			0.0	0.0	0.4	0.0	0.0
Litter		litter			0.0	0.0	0.6	0.0	0.3
OW		open water			81.3	11.7	64.4	100	100
SH		shell hash			0.8	0.0	0.9	0.0	0.0
SMH		small mixed herbs			0.0	0.0	T	0.0	0.0

T = Trace

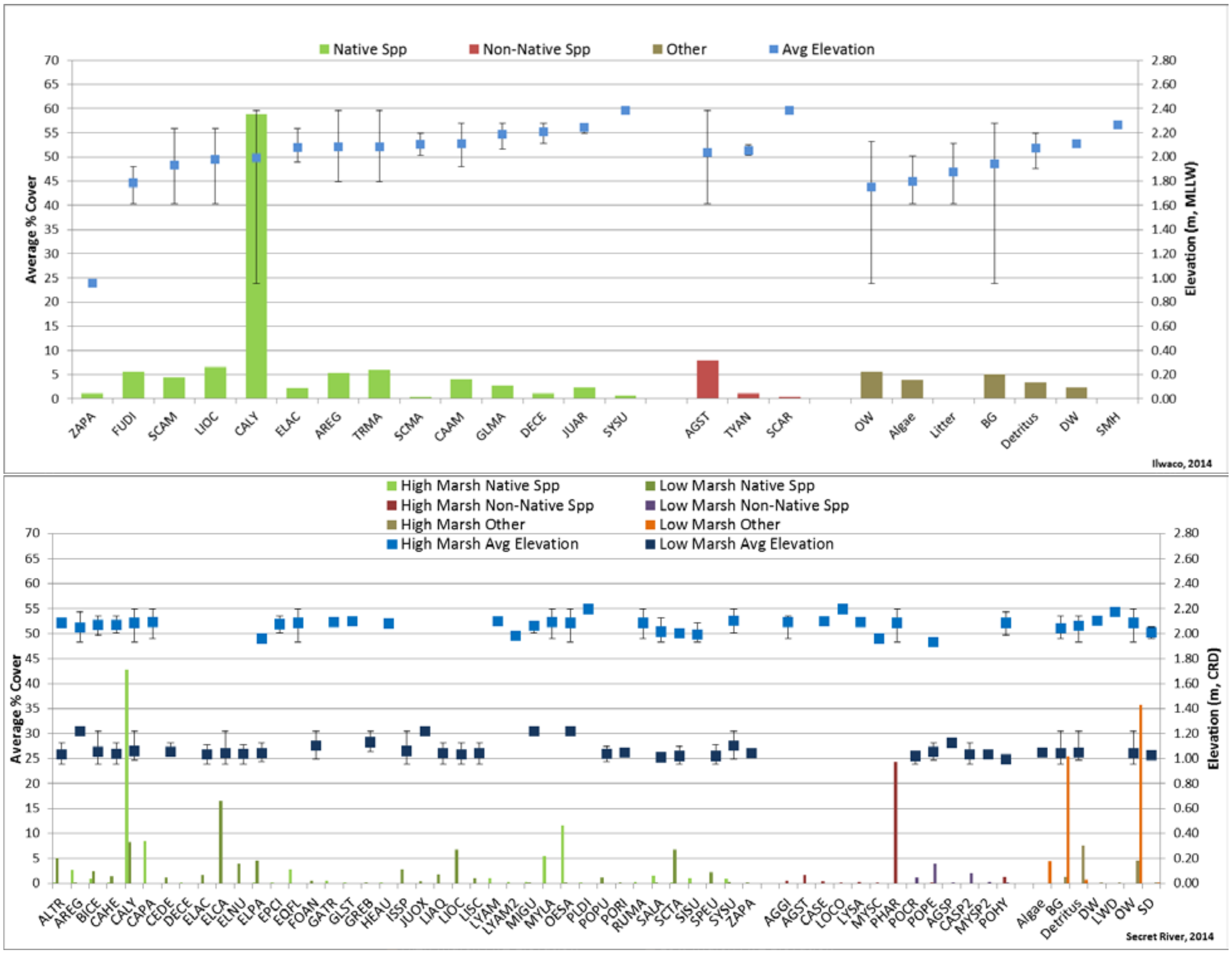


Figure C.1. Vegetation species cover and elevations for sites sampled in 2014. Sites are ordered by position in the lower river, starting near the mouth, and are identified in the lower right corner of the plots. Species are in order by elevation. Bars represent the minimum and maximum elevations at which the vegetative species occurred within the sample area (See Table C.1 for species names and percent cover data associated with codes along the x-axis).

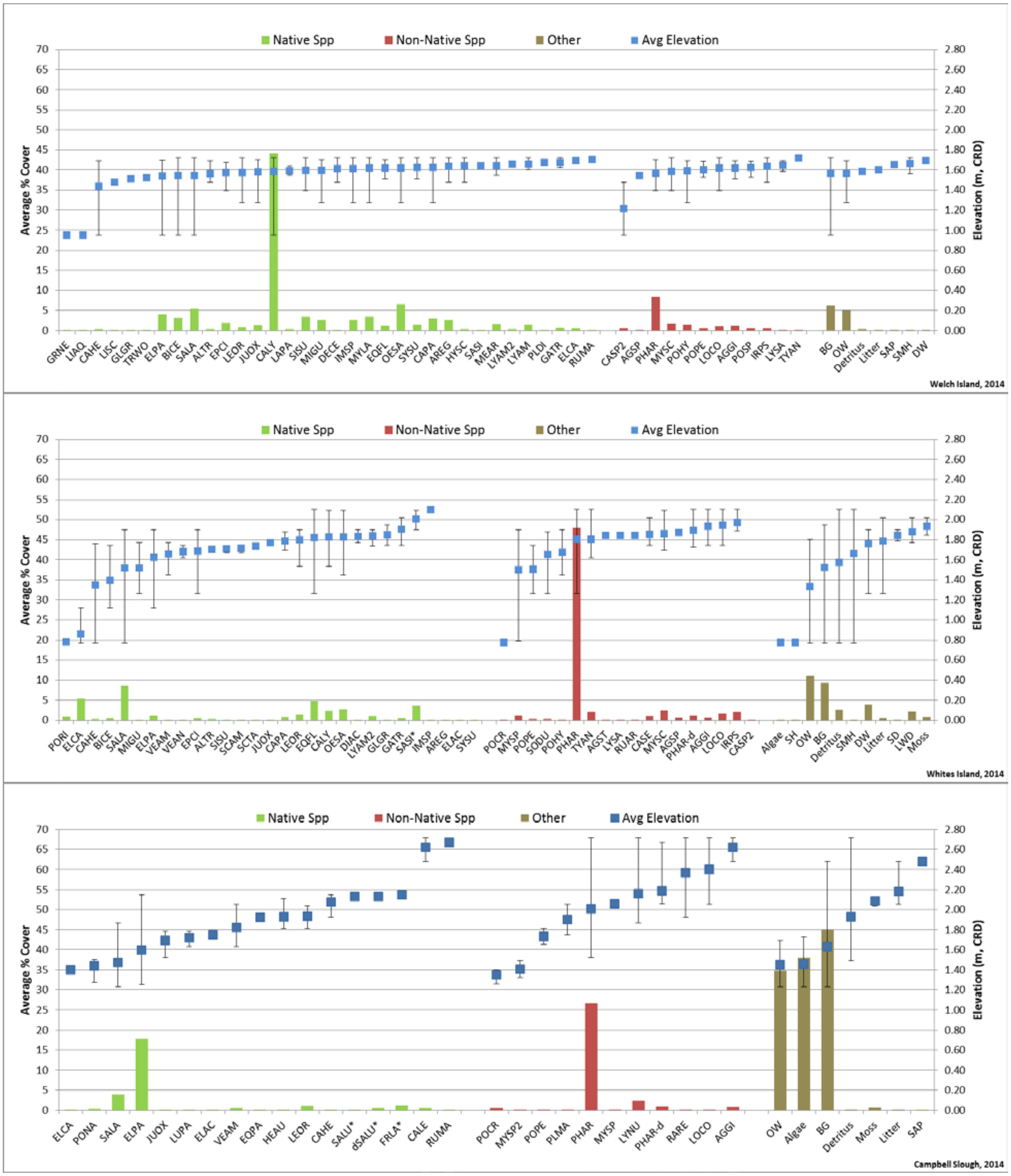
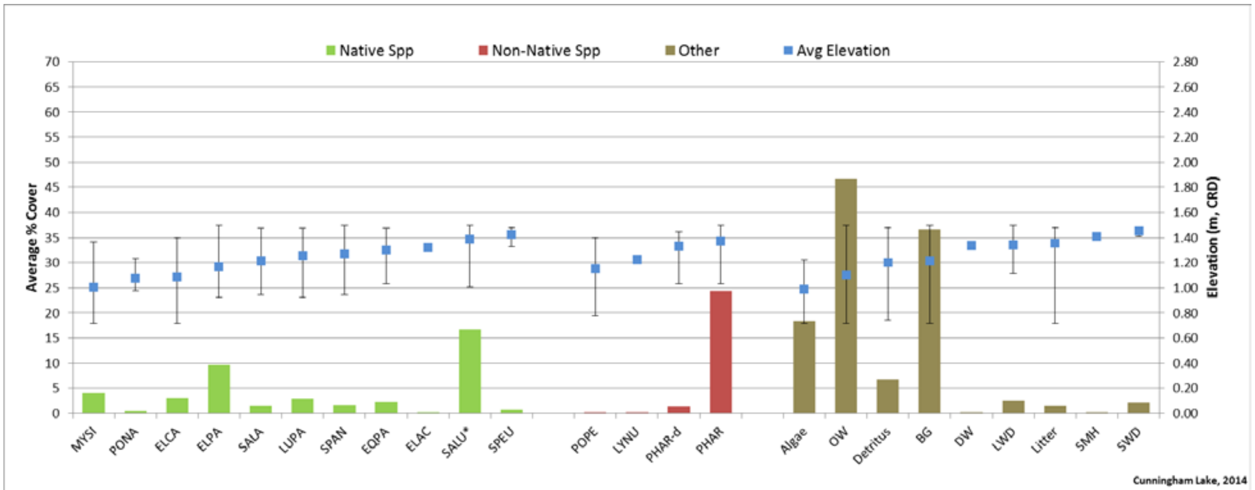


Figure C-1. cont.



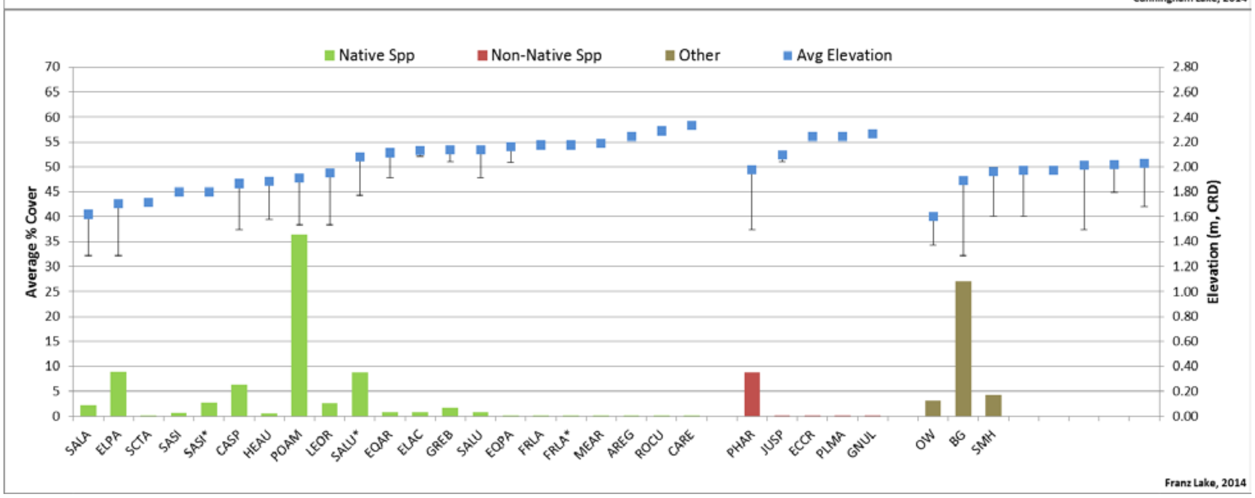


Figure C-1. cont.

Appendix D. Annual photo points from EMP trends sites

Photo points taken at the EMP trends sites during winter and summer sampling seasons.

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

Ilwaco Slough – PP1

27 June 2014

Secret River – PP1 [HIGH MARSH]

5 February 2010

2 August 2012

9 August 2013

Secret River – PP2 [LOW MARSH]

1 December 2011

2 August2012

15 July 2014

Secret River – PP3 [CHANNEL]

1 December 2011

15 July 2014

Welch Island – PP1

1 August 2012

3 February 2013

Welch Island – PP1

23 July 2013

1 August 2014

Whites Island – PP1

22 July 2009

13 July2010

2 August 2011

15 February 2012

Whites Island – PP1

31 July 2012

5 February 2013

22 July 2013

Whites Island – PP1

4 February 2014

31 July 2014

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

Campbell Slough – PP1

29 July 2005

15 July 2006

5 September 2006

Campbell Slough – PP1

17 July 2007

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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

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

Campbell Slough – PP2

22 July 2008

28 July 2009

25 August 2011

14 February 2012

Campbell Slough – PP2

21 July 2012

30 August 2012

11 October 2012

6 February 2013

Campbell Slough – PP2

31 July 2013

12 February 2014

7 August 2014

Appendix E. Fish Data Tables

Table E.1. Results of Bonferroni post-hoc tests showing significant p-values from pairwise comparison of somatic growth rates at all sites.

	BeaconSlough	BeaverArmyTerminal	Bradwoodlough	BurkeIsland	CampbellSlough	ColumbiaCity	ConfluenceOregon	ConfluenceWashington	DeerIsland	FranzLake	GoatIsland	JacksonIsland	LemonIsland	Lord/WalkerIsland	MirrorLake1Lake	MirrorLake4Culvert	PierceIsland	PointAdams	PortlandHarbor	RyanIsland	SandIsland	SecretRiver	WallaceIslandWest	Warrendale	Washougal	WelchIsland	WhitesIsland
Beaver Army Terminal	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bradwood Slough	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Burke Island	-	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Campbell Slough	-	-	0.01	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Columbia City	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Confluence Oregon	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Confluence Washington	-	-	-	-	0.06	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Deer Island	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Franz Lake	-	-	0.01	-	-	-	-	0.05	0.06		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Goat Island	-	-	-	-	-	-	-	-	-	0.06		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Jackson Island	-	-	-	-	-	-	-	-	-	0.05	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lemon Island	-	-	-	-	0.02	-	-	-	-	0.02	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lord/Walker Island	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-	-	-
MirrorLake1 Lake	-	-	0.04	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-	-
MirrorLake4 Culvert	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-	-
Pierce Island	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-	-
Point Adams	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-	-
Portland Harbor	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-	-
Ryan Island	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-	-
Sand Island	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-	-
Secret River	-	-	-	-	0.02	-	-	-	-	0.02	-	-	-	-	-	-	-	-	-	-	-		-	-	-	-	-
Wallace Island West	-	-	-	-	0.01	-	-	-	-	0.00	-	-	-	-	0.02	-	-	-	-	-	-	-		-	-	-	-
Warrendale	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-	-
Washougal	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-	-
Welch Island	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		-
Whites Island	-	0.01	-	-	0.001	-	-	-	-	0.00	-	-	-	-	0.00	0.02	-	-	-	-	-	-	-	0.001	-	-	
Willamette/Morrison	-	-	0.01	-	-	-	-	0.06	0.05	-	0.05	0.05	0.02	-	-	-	-	-	-	-	-	0.02	0.01	-	-	-	0.001

Table E.2. Generalized linear model results from the 10 best models that assess which variables account for the greatest variability in somatic growth rate (GR). Variables used in these models are collection year (YR), Julian day (CalDate), genetic stock (Stock), marked/unmarked (Wild), river kilometer (RiverKM), distance to channel center (ShoreDist), and river reach (Reach). Delta AIC is the difference between each model and the model with the lowest AIC.

Model	AIC	Delta AIC
GR ~Reach + ShoreDist + Wild + YR	-971.3301	0
GR ~Reach + RiverKM + ShoreDist + Wild + YR	-971.1451	0.185
GR ~CalDate + Reach + RiverKM + ShoreDist + Wild + YR	-969.5128	1.8173
GR ~CalDate + Reach + ShoreDist + Wild + YR	-969.4344	1.8957
GR ~RiverKM + ShoreDist + Wild + YR	-968.578	2.7521
GR ~CalDate + RiverKM + ShoreDist + Wild + YR	-966.6527	4.6774
GR ~Reach + Wild + YR	-965.2534	6.0767
GR ~CalDate + Reach + Wild + YR	-963.447	7.8831
GR ~Reach + RiverKM + Wild + YR	-963.2559	8.0742
GR ~CalDate + Reach + RiverKM + Wild + YR	-961.4667	9.8634