Action Effectiveness Monitoring for the Lower Columbia River and Estuary Habitat Restoration Program

Annual Report for Year 8

Project Number: 2003-011-00
Contract Number: 54907
Performance/Budget Period: October 1, 2011 – September 30, 2012
Annual Report

Technical Contact: Matthew Schwartz
Research Scientist
Lower Columbia Estuary Partnership
811 SW Naito Parkway, Suite 410
Portland, Oregon 97204
Phone: (503) 226-1565 x 239
mschwartz@estuarypartnership.org

BPA Project Manager: Anne Creason
Policy Analyst
Bonneville Power Administration
905 NE 11th Avenue
Portland, Oregon 97208
Phone: (503) 230-3859
amcreason@bpa.gov

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Matthew D. Schwartz*
Lyndal L. Johnson[1]
Sean Y. Sol[1]
O. Paul Olson[1]
Kate H. Macneale[1]
Paul M. Chittaro[1]
David J. Teel[1]
Gina M. Yitalo[1]
George L. Kral[2]
Melissa A. Rowe Soll[2]
April Silva[3]
Jason Smith[3]
Jina P. Sagar*

*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] Northwest Fisheries Science Center, NOAA-Fisheries
[2] Ash Creek Forest Management, Inc.
[3] Columbia River Estuary Study Taskforce

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1 Executive Summary

The Lower Columbia Estuary is designated as an estuary of national significance by the US Environmental Protection Agency. As one of 28 designated estuaries in the National Estuary Program, the Lower Columbia Estuary Partnership's (Estuary Partnership's) mission is "to preserve and enhance the water quality of the estuary to support its biological and human communities." The Estuary Partnership's Habitat Restoration Program's goal is to enhance, protect, and restore tidal wetlands and other key habitats in the lower Columbia River and estuary. This program provides a coordinated, ecosystem-based approach that implements the restoration actions of many partners in the lower Columbia River and estuary and allows partners to jointly leverage resources and expertise.

The Bonneville Power Administration (BPA) has funded the Estuary Partnership's Habitat Restoration Program for nine years. The focus of BPA's funding is the development and implementation of habitat restoration projects designed to benefit Endangered Species Act listed salmonids. To measure the success of habitat restoration actions and improve restoration practices, it is necessary to have a monitoring program which focuses on the outcomes of restoration projects. The Effectiveness Monitoring Program administered by the Estuary Partnership implements Action Effectiveness Monitoring (AEM) to address RPA 60 in the 2008 Draft Biological Opinion for the Federal Columbia River Power System (2008 Draft BiOp) based on the Estuary RME plan. The goal of this effort is to provide the Estuary Partnership, primary funding agencies (BPA and Environmental Protection Agency [EPA]), restoration partners (e.g., USACE and Columbia River Estuary Study Taskforce [CREST]), and others with information useful for evaluating the success of restoration projects. This Effectiveness Monitoring Program will focus on projects sponsored by the Estuary Partnership's Habitat Restoration Program.

This annual report documents AEM efforts implemented by the Estuary Partnership under BPA Project Number: 2003-011-00, Contract Number: 54907.

In spring 2012, The Estuary Partnership contracted NOAA Fisheries (NOAA), Ash Creek Forest Management (ACFM), and Columbia River Estuary Study Taskforce (CREST) to conduct pilot AEM at three sites (Mirror Lake, Sandy River Delta, and Fort Clatsop). These AEM sites represent different restoration activities (culvert enhancement to improve fish passage, large wood installation, re-vegetation, and culvert removal for tidal reconnection), habitats (bottomland forest, riparian forest, emergent wetland, and brackish wetland), and hydro-geomorphic reaches of the river (Reaches H, G, and A, ranging from tidal freshwater in Reach H, or the Columbia River George, to saltwater intrusion in Reach A, near Astoria, Oregon). To quantify effects of restoration actions, a Before/After Control Impact (BACI) statistical design was used when monitoring restoration sites. The BACI design uses a control or reference site to deal with both the spatial and temporal variation associated with ecological data collection (Osenberg et al. 2006). AEM at the Scappoose Bottomlands restoration area was concluded in 2011, due to landowner concerns.

<u>Summaries of AEM Results</u>

- NOAA Fisheries sampled fish and macroinvertebrates from April to October 2012 at five locations at the Mirror Lake restoration site. The sampling investigates site usage by fish, condition and stock of collected juvenile salmonids, and abundance and biomass of macroinvertebrates (**Section 3 Fish-passage Improvement and LWD AEM at Mirror Lake**).

- From 2008 – 2012, Chinook and coho salmon abundance did not show any clear increasing or decreasing trend. Fish community composition has changed with the proportion of sticklebacks in catches generally increasing. Fish community diversity has increased at the Confluence, Latoruell and Youngs Creek sampling sites, which is partly due to an influx of non-native species.
- No clear trends have been observed related to salmon health, condition, or realized function, however, values for growth, lipid content, and condition factor are comparable to reference sites in the same reach.
- There is no clear trend in habitat quality in terms of prey availability, but prey resources at the Lake and Culvert sites are similar in type and quantity to reference sites found in the same reach.
- Elevated summer temperatures are a consistent concern at the Culvert and Lake sites, but chemical contamination appears low.
- Marked Chinook at both sites were primarily from the Spring Creek Group fall Chinook stock, while unmarked fish belonged to a diverse array of stocks including Upper Columbia summer/fall Chinook, Snake River fall Chinook, and Deschutes River fall Chinook, as well as West Cascades fall and Spring Creek Group fall Chinook stocks from the Lower Columbia River ESU.

- ACFM returned to six restoration sites to collect data at 251 vegetation plots across 399 acres at the Sandy River Delta and Mirror Lake restoration sites to assess the success of invasive vegetation removal and native vegetation plantings at these restoration sites (**Section 4 Planting Success AEM at Mirror Lake and Sandy River Delta**).
    - Units monitored over the last five years showed positive trends of native species reestablishment, such as increased canopy cover, native plant density, and understory development. Natural regeneration, primarily through rhizomatous growth, has increased structural complexity and is driving restored units closer towards reference site conditions.
    - The presence of noxious weeds, however, threatens to reduce the success of these native plantings. Reed canarygrass (*Phalaris arundinacea*) growth has reduced plant diversity, and also reduces natural control on rodent populations, which have detrimental effects on woody plants due to increased browse. Himalayan blackberry, left unmanaged, may return to previous levels, which can inhibit self-sustaining native woody plant populations and the vegetation structure necessary for target fish and wildlife species

- CREST collected habitat (sediment accretion, channel cross-sections) and macroinvertebrate data at the Fort Clatsop restoration and reference sites, for the fourth year. In 2009, CREST began collecting water quality data at Ft. Clatsop restoration and reference sites. Fish and macroinvertebrates were sampled monthly between March and July at the Fort Clatsop site. Permit limitations restricted the ability to sample the fish community in 2012; however, a synthesis of fish sampling from 2007 to 2011 was completed. (**Section 5 Salmon, Salmon Prey, and Habitat AEM at Fort Clatsop South Slough & Alder Creek**)
    - A Pit Tag Array was installed at Ft. Clatsop in March of 2012. There were two detections in 2012, both upriver hatchery Chinook in (in April and June). The Chinook detected in April occupied the site for two hours, while the Chinook detected in June was not detected leaving the site.
    - Fish community data from 2007-2011 revealed a decline in the percentages of non-native and native, non-salmonid, fish species at South Slough and an increase in unmarked juvenile salmonids. Fish community data between South

Slough and Alder Creek demonstrated a similar pattern across years sampled, which indicates the restoration site and reference site are experiencing similar inter annual variability. However, differences in sampling gear and method preclude statistical comparison between years.

- In 2007, prior to tidal reconnection, peak salmonid abundance occurred earlier in the year at South Slough. Following the tidal reconnection juvenile salmonids were occupying the site later into mid to late summer. This increase in site use could be attributed to a decrease in water temperature as a result of tidal reconnection. Post-restoration temperatures were consistently lower at Sough Slough indicating tidal reconnection improved water quality. This is further reinforced by temperature trends from Alder Creek, where average monthly temperatures either remained consistent or increased.
- Chinook at both sites demonstrated a preference for isopod, corophium, and chironomid species. These species had the highest percentages for taxa consumed by Chinook, as well as the highest Index of Relative Importance value.
- Sediment accretion measurements recorded at South Slough between 2008 and 2012 reveal an overall net gain of sediment over the course of the monitoring period. Channel cross sections at South Slough over the same period of time show small changes in channel erosion and aggradation. There was an increase in erosion near the mouth of South Slough and an aggradation in the upper end of Slough.

2 Background on Estuary Partnership's Action Effectiveness Monitoring

The Estuary Partnership's mission is "to preserve and enhance the water quality of the estuary to support its biological and human communities." As part of this mission, the Estuary Partnership manages an umbrella Habitat Restoration Program Estuary Ecosystem Restoration Program that supports state, federal and local government restoration objectives. The Habitat Restoration Program's goal is to enhance, protect, and restore tidal wetlands and other key habitats in the lower Columbia River and estuary. This program provides a coordinated, ecosystem-based approach to implement restoration actions by many partners in the lower Columbia River and estuary and allows partners to jointly leverage resources and expertise. The Estuary Partnership's Comprehensive Conservation and Management Plan, completed in 1999 and updated in 2011, calls for enhancing, protecting, creating, or reclaiming 19,000 acres of wetland habitat, including at least 3,000 acres of tidally influenced habitat, by 2014. The Estuary Partnership has catalogued more than 16,614 acres of lower Columbia River habitat that have been acquired, protected, or restored throughout the region since 1999. Estuary Partnership funding has supported approximately 60 restoration projects, which have resulted in the restoration of 3,325 acres of habitat. The goal of the Action Effectiveness Program is to provide the Estuary Partnership, primary funding agencies (BPA and Environmental Protection Agency [EPA]), restoration partners (e.g., USACE and Columbia River Estuary Study Taskforce [CREST]), and others with information useful for evaluating the success of restoration projects. Such evaluations supported by AEM facilitate improvements in project design and management, increase the success of restoration projects for ESA listed salmonids, and address RPA 60 of the 2008 Draft BiOp.

In preparation for the Draft 2008 BiOp, the plan for "Research, Monitoring, and Evaluation for the Federal Columbia River Estuary Program" (Estuary RME) was prepared for the Bonneville Power Administration (BPA) by the Pacific Northwest National Laboratory (PNNL) in conjunction with National Oceanic and Atmospheric Administration (NOAA) Fisheries and the

US Army Corps of Engineers (USACE) with the collaboration of the Estuary Partnership (Johnson et al. 2008) as part of the Estuary and Oceanic Subgroup (EOS). The plan provided a framework to evaluate progress towards understanding, conserving, and restoring the estuary to benefit ESA listed salmonid species and outlines recommendations for AEM.

The Draft 2008 BiOp highlights the importance of estuarine habitat restoration for anadromous fish (Reasonable and Prudent Alternatives [RPA] 36-38). These restoration RPAs are to be implemented in conjunction with AEM identified in RPA 60. AEM is needed to "evaluate the effects of selected individual habitat restoration actions at project sites relative to reference sites and evaluate post-restoration trajectories based on project-specific goals and objectives" (NMFS, 2008).

The Effectiveness Monitoring Program focuses on projects sponsored by the Estuary Partnership's Habitat Restoration Program. On-the-ground AEM efforts collect the data needed to assess the performance and functional benefits of restoration actions in the lower Columbia River and estuary (LCRE).

The Estuary Partnership's objectives for the Effectiveness Monitoring Program are to:

- Improve restoration techniques to maximize impact of habitat restoration actions and better track long term project success
- Identify how restoration techniques address limiting factors for salmonids
- Determine the impact of restoration actions on salmon recovery at the site, landscape, ecosystem scale
- Use intensive monitoring to inform extensive monitoring efforts to improve multi-scale AEM

To meet AEM program objectives, the Estuary Partnership are engaged it the following tasks:

- Implementing AEM as outlined in the Estuary RME plan (Johnson et al. 2008) and following standardized monitoring protocols (e.g., Roegner et al. 2009) where applicable
- Developing long-term datasets for restoration projects and their reference sites
- Developing a programmatic plan for AEM to provide improved efficiency and coordination between stakeholders
- Disseminating data and results to facilitate improvements in regional restoration strategies
- Developing of a regional cooperative effort by all agencies and organizations participating in restoration monitoring activities to maximize the usefulness of monitoring data

Additionally, the Estuary Partnership aims for the Effectiveness Monitoring Program to complement our existing Ecosystem Monitoring Program (BPA project # 2003-007-00). The Ecosystem Monitoring Program implements monitoring activities to characterize undisturbed emergent wetlands and assess juvenile salmonid usage of those habitats. Several sites monitored by the Ecosystem Monitoring Program were included in the Estuary Partnership's Reference Site Study funded by BPA (Borde et al. 2012). One of objective of the study was to determine if structural data from multiple reference sites can be used to evaluate restoration action effectiveness. The concept of using multiple reference sites is important because a paired reference site is not always available at or near a restoration site. Borde et al. found that sediment accretion, elevation, inundation, water temperature, vegetation composition and similarity, channel morphology are useful metrics for evaluating restoration effectiveness between a restoration site and multiple reference sites. The Estuary Partnership's EMP continues to monitor many parameters likely to be included in AEM (e.g., vegetation, water quality, and salmon) and the collection of comparable datasets by the two programs (where possible) will continue to fill data gaps and add to our understanding of habitat conditions and juvenile salmonids in the lower river.

2.1 Site Selection

In January 2008, the Estuary Partnership and the Estuary and Oceanic Subgroup (EOS) identified sites for pilot AEM. The Estuary Partnership presented a sample of restoration projects supported with BPA funds as potential sites (Table 1). Projects included a variety of restoration activities implemented in different habitats and reaches of the river. EOS members recommended selecting sites to represent different restoration activities, habitats, and geographic reaches of the river. Other recommended considerations included:

- Baseline monitoring was conducted at the restoration site.
- Re-vegetation AEM in different habitats would provide useful data and be low in cost relative to AEM for projects such as like tidal reconnection.
- If possible, AEM should occur at sites where restoration actions are apt to continue for multiple years (indicating a financial investment in the project area).
- AEM at sites sponsored by BPA and partners would provide collaboration opportunities.
- Some (but not all) project managers would have the capacity to implement AEM in 2008.

EOS members recommended 4 projects for AEM (Mirror Lake, Sandy River Delta, Scappoose Bottomlands, and Fort Clatsop [Figure 1]) were first sampled in 2008 and 2009. These AEM sites represent different restoration activities (culvert enhancement to improve fish passage, large wood installation, re-vegetation, cattle exclusion, and culvert removal for tidal reconnection), habitats (bottomland forest, riparian forest, emergent wetland, and brackish wetland), and geographic reaches of the river (Reaches H, G, F, and A, ranging from tidal freshwater in Reach H, or the Columbia River Gorge, to saltwater intrusion in Reach A, near Astoria, Oregon).

Table 1. Sample of Estuary Partnership restoration projects funded by BPA presented as potential sites to EOS members. Recommended AEM sites are highlighted in gray.

Project Name	Restoration Activity	Year(s) When Restoration Occurred	Habitat Type	Reach	Baseline Monitoring
Mirror Lake	Improve fish passage; Large wood installation; Native plant revegetation	2007 – Present	Bottomland hardwood forest	H	Yes
Sandy River Delta	Native plant revegetation	2004 – 2006	Riparian forest	G	No
Stephens Creek	Floodplain reconnection; Native plant revegetation	2007 – Present	Floodplain	G	Yes
Salmon Creek	Large wood installation	2007 – Present	Riparian	F	TBD
Malarkey Ranch	Culvert removal	2004 – 2005	In stream	F	Yes
Scappoose Bottomlands	Cattle exclusion; Invasive removal; Native plantings	2004 – Present	Emergent wetland	F	Yes
Alder Creek	Culvert removal	2005 – 2006	In stream	F	Yes
Lewis River	Native plant revegetation	2007 – Present	Riparian	E	TBD
Sharnelle Fee	Dike breach	2005 – Present	Tidally influenced wetland	A	Yes
Lewis and Clark	Dike breach	2004 – 2006	Tidal estuarine habitat	A	Yes
Fort Clatsop	Culvert removal and bridge installation	2005 –Present	Brackish wetland	A	Yes

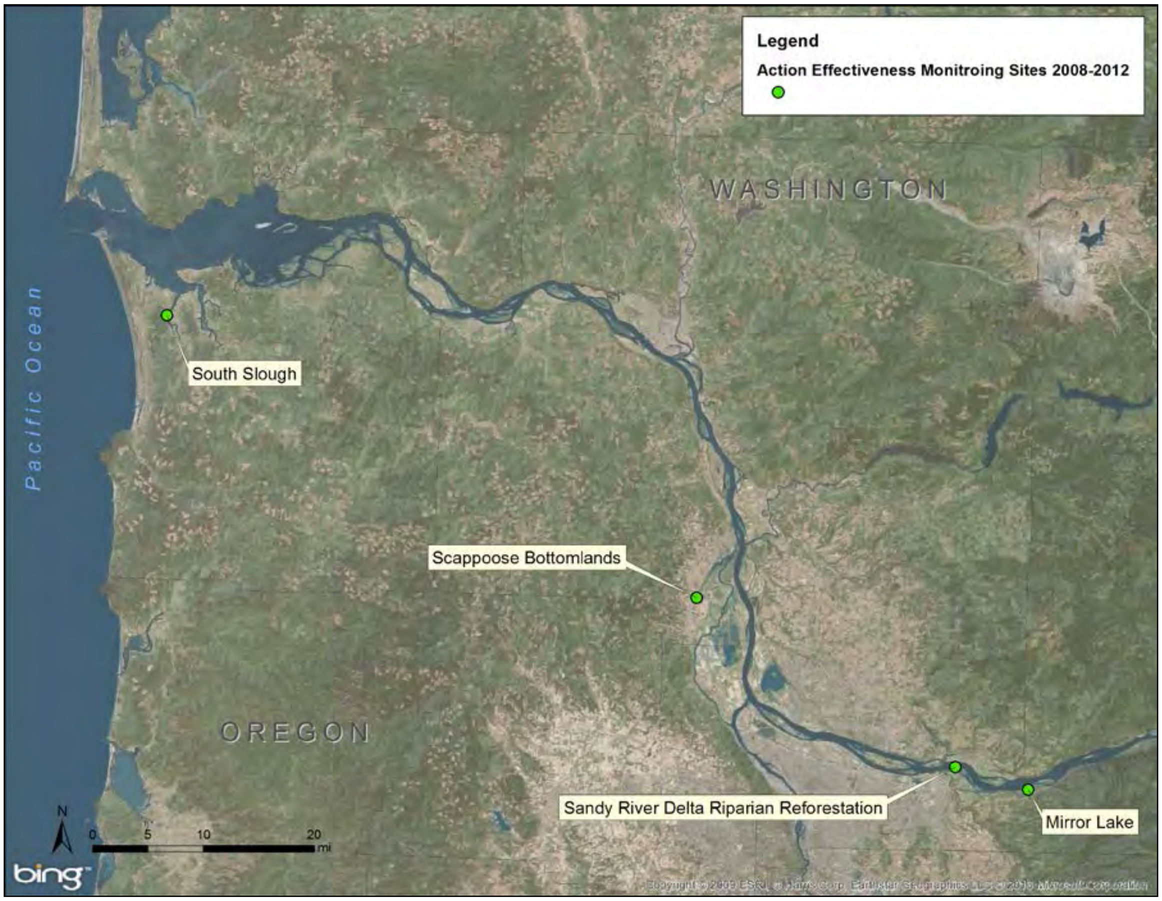


Figure 1. Restoration projects monitored from 2008-2012 as part the the Estuary Partnership's Action Effectiveness Monitoring Program.

2.2 Programmatic Action Effectiveness Monitoring and Research

In 2012, the Estuary Partnership, BPA, and US Army Corps began to update the AEM program applying lessons learned during initial AEM efforts. "*A Programmatic Plan for Restoration Action Effectiveness Monitoring and Research in the Lower Columbia River and Estuary"* (Johnson et al. 2013) was developed to determine the success of restoration actions not only at the site level, but also at landscape and estuary-wide scales. The intended outcome of this programmatic AEM plan is to achieve efficiency, coordination, and consistent conduct of AEM across the LCRE over the next six years of the FCRPS BiOp (2013-2018). In addition, this programmatic AEM guidance will be incorporated into technical proposals during the Estuary/Lower Columbia River categorical review within the Northwest Power and Conservation Council's Fish and Wildlife Program in early 2013. Regional stakeholders can use this programmatic approach to provide context for their project-specific AEM efforts and help project-level goals synchronize with landscape level efforts of the Columbia Estuary Ecosystem Restoration Program (CEERP) level where concern is for the collective ecological success of multiple restoration projects across multiple landscapes in the LCRE. Stakeholder research, monitoring, and evaluation (RME) plans involve using AEM to determine if their restoration actions were successful in meeting the project's objectives, identify improvements to restoration design and execution, and recognize cost efficiencies in AEM efforts. Overall, the programmatic approach to estuary AEM will be better coordinated with the broader estuary restoration effort through the Estuary Partnership, and with Columbia River tributary habitat AEM and the federal

RME effort under the 2008 Federal Columbia River Power System Biological Opinion (NMFS 2008).

3 Fish-passage Improvement and LWD AEM at Mirror Lake

The Mirror Lake Complex is located next to Rooster Rock State Park in the Columbia River Gorge, approximately 25 miles east of Portland, Oregon, and 15 miles west of Bonneville Dam on the Columbia River. The lake is connected to Rooster Rock State Park by I-84 culvert at the base of Rooster Rock. A boat ramp and dock are located at the lagoon at the base of Rooster rock, and the lagoon merges with the Columbia River. This area is used for fishing, and other recreational activities. The lake is used for fishing and recreation and the area contains some high quality emergent wetland habitat. Youngs Creek and Latourell Creek are two small streams that feed the emergent wetland at Mirror Lake. The surrounding area is degraded as a result of past land clearing, agriculture and grazing.

The US west coast is home to 119 plant and animal species that are federally listed as threatened or endangered, including Columbia River Pacific salmon (NMFS, 1998), and destruction or modification of critical habitat is thought to be a major factor contributing to salmon declines. In an effort to increase habitat for threatened or endangered juvenile salmon, a large number of restoration activities have been initiated in the lower Columbia River and estuary, including the habitat restoration and enhancement project in the Mirror Lake Complex. Observation of salmon at the lake and creek during a fish spawning survey prompted restoration and enhancement of Mirror Lake Complex (LCEP 2011). The stream bed in the upper reach of Youngs Creek had a cobble substrate suitable for salmon spawning, but access was blocked by a failing culvert in an earthen dam that was formerly used as farm access road. In 2005, the dam was replaced with 70 ft. bridge to give salmon species access to upstream spawning area; to reduce water temperature both above and below the bridge; and to improve the hydrology of Youngs Creek, which drains into Mirror Lake. Other activities included removing blackberries along the stream and planting native willows and cottonwoods. Below Mirror Lake itself, the culvert under I-84 allowed for backwater from the Columbia River to flood the site; however, flows were thought to be somewhat restricted compared to historical conditions. Consequently, as part of the restoration project, a number of improvements to the culvert were made to improve flow, increase water depth, and provide refugia for fish (LCEP 2011).

To determine the efficacy of the restoration activities conducted at the Mirror Lake Complex, an Action Effectiveness Monitoring Program was established, with monitoring objectives, metrics, analyses and criteria for success for each of the restoration actions conducted as part of the overall program (LCEP 2011). For fish and fish prey, the specific objectives and metrics include the following:

1. Increase the number of salmon in Mirror Lake Complex from pre-restoration levels, as indicated by an increase in catch per unit effort (CPUE) or density of salmon from pre-restoration levels
2. Restore fish community composition in the Mirror Lake Complex to conditions typical of undisturbed sites, as indicated by:
    - Fish species and species distribution more comparable to those at undisturbed reference sites
    - Fish species diversity and richness diversity more comparable to values at undisturbed reference sites after restoration
    - Fewer non-native species after restoration

3. Increase the diversity of salmon in Mirror Lake Complex from pre-restoration levels, or restore diversity to conditions comparable to those at similar, non-disturbed sites, as indicated by:
    - An increase in the number of salmon species and/or higher diversity present after restoration
    - A greater variety of sizes and life history stages of salmon present after restoration
    - A greater diversity of Chinook salmon stocks present after restoration
4. Increase salmon health and condition from pre-restoration levels, as indicated by improvement in lipid content, condition factor, and growth rates of Chinook salmon post-restoration
5. Improve water quality, including the summer temperature regime, at Mirror Lake complex sites
6. Improve wetland habitat quantity and quality as indicated by increased presence of invertebrate prey consumed by salmon

Since 2008, NOAA Fisheries has been tracking these metrics at the Mirror Lake Complex sites, as well collecting synoptic data on water temperature at the sites during fishing events. Chemical analyses of bodies and bile from juvenile Chinook salmon collected at the sites were also conducted as an additional indicator of habitat quality, and to complement toxics monitoring efforts throughout the estuary (e.g., LCEP 2007), although contaminants were not identified as a stressor of concern in the Mirror Lake Complex.

In this section we present our findings on trends in fish community composition, salmon habitat occurrence, and measures of salmon health and fitness between 2008 and 2012. The data from the Mirror Lake sites are also compared to comparable information from other, relatively undisturbed sites that have been sampled in Reach H of the Lower Columbia River as part of the Estuary Partnership's Ecosystem Monitoring Program (EMP; see Sagar et al. 2013) and can serve as reference sites.

3.1 Methods

3.1.1 Survey Site Description

Figure 2 shows the five areas of focused fish sampling at the Mirror Lake project area as described below. The coordinates and years when these sites were sampled are shown in Table 2

Culvert: This site is located immediately below the I-84 culvert and adjacent areas opposite the boat launch and associated docks (Figure 2, Figure 3 A, B). The area immediately below the culvert had very little to no vegetation associated with the banks or bottom. The banks are steep, and rocky, areas consisting of pebbles to small boulders. Bottom sediment is the same. The adjacent areas are dominated by grasses, with a steep bank (1.5 m) that drops off quickly. The bottom sediments are composed of very soft mud. In the summer of 2008, boulders were added to the culvert at I-84 to improve water flow for salmon passage.

Lake: This site is on the open water part of the lake near the I-84 culvert (Figure 2, Figure 3 C). The area is dominated by grasses from the high water mark to the low water edges, and by shrubs and blackberry vines along the bank above and at very high water levels. The Lake substrate consists of consolidated to soft-packed mud, with aquatic vegetation later in the season. The Lake is fed by waters from Latourell Creek and Youngs Creek. Its water level varies seasonally depending on the elevation of a beaver dam at its outlet and backwater from the Columbia River that inundates the site during spring runoff.

Youngs Creek: This site is located upstream of the Lake site (Figure 2, Figure 3 D). The creek varies from about 1.5 meters wide at low water level to about 5 meters wide at high water. The riparian area is dominated by reed canarygrass (*Phalaris arundinacea*) to the edge of the creek bed and in immediately adjacent areas, with a steep drop (1.5 meters) from the edge of the creek bank to the water. Bottom sediment is composed of very soft mud. From mid-June to late summer, the creek banks are overgrown with tall grasses, which overhang the banks, providing shade and cover for stream inhabitants. Between 2004 and 2007, before monitoring was initiated, a failing culvert (dam) at this site was replaced with a 70 ft bridge to give salmon species access to upstream spawning areas. Prior to restoration activities, very little large woody debris existed at this site and grasses provided the only available cover. To improve this situation, invasive plants along the creek were removed and native willows and cottonwoods were planted. In summer of 2008, large woody debris was added to Youngs Creek to improve salmon habitat.

Confluence and Latourell Creek. Beginning in 2010, these sites were surveyed intermittently for salmon observation. The Confluence site (Figure 3 E) is located at the confluence of Latourell Creek and Youngs Creek, downstream of the Youngs Creek site. Latourell Creek (Figure 3 F) is located 100 m downstream of Latourell Lake. Confluence site is slightly wider and shallower than Youngs Creek and but Latourell Creek is similar to Youngs Creek. Bottom sediments at both sites are composed of very soft mud, and the banks are overgrown with tall grasses.

Table 2. Coordinates and sampling years for Mirror Lake Complex sites.

Site Name	Latitude	Longitude	Years Sampled
Culvert	45° 32.606'N	122° 14.878'W	2008-2012
Lake	45° 32.562'N	122° 14.703'W	2008-2012
Confluence	45° 32.620'N	122° 13.727'W	2010-2012
Latourell Creek	45° 32.590'N	122° 13.190'W	2010-2012
Youngs Creek	45° 32.735'N	122° 12.275'W	2008-2012

Figure 2. Photo showing areas of fish collection at Mirror Lake. Photo provided by Google Earth.

A) 
B) 

C) 
D) 

E) 
F) 

Figure 3. Photos of fish sampling sites at the Mirror Lake project area. A) Culvert at high water, B) Culvert at low water, C) Lake, D) Youngs Creek, E) Confluence and F) Latourell Creek.

3.1.2 Fish Monitoring

Monitoring for fish and prey was generally initiated in April and continued on a monthly basis through August or September in 2008-2010, and as late as December in 2011 (Table 3). The Latourell Creek site was sampled only once per season because it was difficult to access, in 2008 as a reconnaissance sampling, and in August 2010, September 2011, and August 2012. Extremely high water levels precluded sampling effort at some sites during certain sampling events; and in 2011, permitting issues delayed sampling in 2011 until May.

Table 3. Fishing attempts made at Mirror Lake Complex Action Effectiveness Monitoring fishing sites.

	Month								
Site	Apr	May	June	Jul	Aug	Sept	Oct	Nov	Dec
Culvert 2008		3	1	3	2	1			
Culvert 2009	1	1	4	3	3	HW			
Culvert 2010	2	3	2	2	2	HW			
Culvert 2011	PI	1	HW	2	2	HW	2		
Culvert 2012	1	1	1	2	2	HW	2		
Lake 2008	3	2	2	2	3	3	2		
Lake 2009	3	2	4	3	3	HW			
Lake 2010	3	3	2	3	3	HW			
Lake 2011	PI	2	1	2	3	3	3	3	3
Lake 2012	2	2	1	1	3	3	3		
Confluence 2010	3	3		3					
Confluence 2011	PI	1		1		2			
Confluence 2012	1				1				
Latourell Cr. 2010					2				
Latourell Cr. 2011						2			
Latourell Cr. 2012					2				
Youngs Creek 2008	3	3	1	3	2	1			
Youngs Creek 2009	2	3	2	1	3	HW			
Youngs Creek 2010	3	3	HW	3	2	HW			
Youngs Creek 2011	PI	HW	HW	2	2	2	2		
Youngs Creek 2012	HW	HW	HW	HW	3	3	2		

HW= not sampled due to high water, PI=not sampled due to permit issues

Due to variation in topography, accessibility, and water levels among the restoration sites, several types of gear were used to sample the Mirror Lake sites. Depending on site conditions, fish were collected using either a Puget Sound beach seine (PSBS) (37 x 2.4 m, 10 mm mesh size), a modified PSBS (MPSBS, shortened to 7.5 x 2.4 m, 10 mm mesh size), or a modified block net (MBN) where the middle portion of the PSBS was used as a block net and a second net (2 x 1.5 m, 10 mm mesh size) was used as a fish chase net. PSBS sets were deployed using a 17 ft Boston Whaler or a 9 ft inflatable raft. The MPSBS was deployed on foot in shallow water where efficient boat deployment was not possible. The MBN was used to sample fish in small stream channels where fishing with the PSBS or MPSBS was not efficient or feasible. Up to three sets were performed at each site at each sampling time, as site conditions and sampling permit limitations allowed. At each sampling event, the coordinates of the sampling locations, the time of sampling, weather, habitat conditions, vegetation, and water temperature were recorded.

All fish in each set were identified to the species level and counted. Salmonids were examined for fin clips and coded wire tags (CWTs) in order to determine the proportions of marked fish (of known hatchery origin) and unmarked fish (potentially wild). Subsets of up to 30 juvenile Chinook salmon (Onchorhynchus tshawytscha), coho salmon (Onchorhynchus kisutch), chum salmon (Onchorhynchus keta), and steelhead trout (Onchorhynchus mykiss) from each set were measured (to nearest mm) and weighed (to nearest 0.1 g).

Additionally, when sufficient Chinook salmon were present, up to 30 individual juvenile Chinook from each were collected and sacrificed. The following samples were collected from necropsied fish: stomach contents for taxonomic analyses of prey in salmon diets; whole bodies (minus stomach contents) for measurement of lipid content and classes; otoliths for estimation of age and growth rates; and fin clips for genetic stock identification. If enough animals were captured, bile for measurement of metabolites of polycyclic aromatic hydrocarbons (PAHs), as well as stomach contents for measurement of PAHs and other persistent organic pollutants (POPs), including PCBs, DDTs and organochlorine pesticides, and PBDEs; and whole bodies (minus stomach contents) for measurement of bioaccumulative POPs were also collected.

Samples for chemical analysis were held on dry ice and transported to the NWFSC laboratory, where they were stored frozen at -80°C until analyses were performed. Stomach contents samples for taxonomic analysis were preserved in 70% ethanol. Fin clips for genetic analyses were collected and preserved in alcohol, following protocols described in Roegner et al. (2009). Otoliths for age and growth determination were also stored in 70% ethanol.

3.1.3 Sample analyses

3.1.3.1 Genetic stock identification.

Genetic stock identification (GSI) techniques (see Manel et al. 2005) were used to investigate the origins of juvenile Chinook salmon using the Mirror Lake Complex sites, as described in Teel et al. (2009) and Roegner et al. (2010). The stock composition of juveniles was estimated with a regional microsatellite DNA data set (Seeb et al. 2007) that includes baseline data for spawning populations from throughout the Columbia River basin (described in Teel et al. 2009). The overall proportional stock composition of Mirror Lake samples was estimated with the GSI computer program ONCOR (Kalinowski et al. 2007), which implements the likelihood model of Rannala and Mountain (1997). Probability of origin was estimated for the following regional genetic stock groups (Seeb et al. 2007; Teel et al. 2009): Deschutes River fall Chinook; West Cascades fall Chinook; West Cascades spring Chinook; Middle and Upper Columbia spring Chinook; Spring Creek Group fall Chinook; Snake River fall Chinook; Snake River spring Chinook; Upper Columbia River summer/fall Chinook; and Upper Willamette River spring Chinook. West Cascades and Spring Creek Group Chinook are Lower Columbia River stocks.

3.1.3.2 Fish community characteristics, catch per unit effort, and fish condition calculation.

Fish species diversity was calculated using the Shannon-Weiner diversity index (Shannon and Weaver 1949, Margalev 1958):

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

Where

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

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

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

This index was also used to calculated diversity of salmonid species, using data only for Chinook, coho, chum, steelhead trout, and cutthroat trout. Additionally, the diversity index was applied to size class data to calculate a size class diversity index for Chinook and coho salmon (Diefenderfer et al. 2011). For these calculations, size classes were used in the place of species, and the number of individuals within a size class in the place of number of individuals of a given species. Size classes used were fry (< 60 mm in length); fingerlings (60-130 mm in length) and yearlings (> 130 mm in length); see Fresh et al. 2005).

Fish catch per unit effort (CPUE) and density were calculated as described in Roegner et al. (2009). CPUE is the catch of the fish or animals in numbers taken in a defined unit of effort, in this case per beach seine set. Density is the number of fish captured in the area sampled by the fishing technique used, which we standardized to number per 1000 m^2. The area sampled by the fishing gear used was estimated as described in Roegner et al. 2009.

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:

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

3.1.3.3 Lipid Determination.

As part of our study we determined lipid content in salmon whole bodies. Lipid content can be a useful indicator of salmon health (Biro et al. 2004), and also affects contaminant uptake and toxicity (Elskus et al. 2005). Studies show that the tissue concentration of a lipophilic chemical that causes a toxic response is directly related to the amount of lipid in an organism (Lassiter and Hallam, 1990; van Wezel et al., 1995); in animals with a high lipid content, a higher proportion of the hydrophobic compound is associated with the lipid and unavailable to cause toxicity. Prior to analyses, salmon whole body samples from the field were composited by genetic reporting group and date and site of collection into a set of composite samples, each containing 3-5 fish each. In salmon whole bodies, composite samples from the total amount of extractable lipid (percent lipid) was determined by Iatroscan and lipid classes were determined by thin layer chromatography with flame ionization detection (TLC/FID), as described in Ylitalo et al. (2005).

3.1.3.4 Chemical Contaminants in Chinook salmon

3.1.3.4.1 Persistent organic pollutants in bodies

Composite body samples, with stomach contents removed, were extracted with dichloromethane using an accelerated solvent extractor. The sample extracts were cleaned up using size exclusion liquid chromatography and analyzed by gas chromatography/mass spectrometry (GC/MS) for PCB congeners; PBDE congeners; organochlorine (OC) pesticides including DDTs, hexachlorocyclohexanes (HCHs), chlordanes, aldrin, dieldrin, mirex, and endosulfans; and low (2-3 ring) and high (4-6 ring) molecular weight aromatic hydrocarbons as described by Sloan et al. (2004, 2006, 2010). 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.

3.1.3.4.2 PAH metabolites in salmon bile.

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

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

Biliary protein was measured according to the method described by Lowry et al. (1951). Biliary fluorescence values were normalized to protein content, which is an indication of feeding state and water content of the bile. Fish that have not eaten for several days exhibit higher biliary FAC values and higher protein content than fish that are feeding constantly and excreting bile more frequently (Collier and Varanasi 1991).

3.1.4 Prey Sampling

For the invertebrate prey sampling, the objective was to collect aquatic and terrestrial invertebrate samples and to identify the taxonomic composition and abundance of salmonid prey available at sites when juvenile salmonids were collected. These data will be compared with the taxonomic composition of prey found in stomach contents of fish collected concurrently. Because juvenile

Chinook salmon were the target species for diet analyses, prey sampling in 2012 was limited to the Culvert and Lake sites, where juvenile Chinook salmon were typically present.

In 2012, NOAA Fisheries invertebrate collections at the Culvert and Lake sites in the Mirror Lake project area. Open water column Neuston tows (2-3 tows at each site at each sampling time) collect prey available to fish in the water column and on the surface of open water habitats. For each tow, the net was towed for a measured distance of at least 100 m. Invertebrates, detritus, and other material collected in the net were sieved, and invertebrates were removed and transferred to a labeled glass jar or Ziploc bag. The jar or bag was then filled with 95% ethanol so that the entire sample was covered. Emergent vegetation Neuston tows (2-3 tows at each site at each sampling time) tows collect prey associated with emergent vegetation and available to fish in shallow areas. For each tow, the net was dragged through water and vegetation at the river margin where emergent vegetation was present and where the water depth was < 0.5 m deep for a recorded distance of 10 m. The samples were then processed and preserved in the same manner as the open water tows.

3.1.5 Statistical analyses

Analysis of variance (ANOVA) and Tukey-Kramer HSD (honestly significant difference (HSD) multiple range test were used to compare inter-annual differences in fish density, lipid content, condition factor, and related parameters at the Mirror Lake Complex sites. Values were also compared with those from fish of the same species collected at relatively undisturbed sites from the Lower Columbia River Reach H (the Columbia Gorge from river km 204-233; Simenstad et al. 2011) collected as part of the Estuary Partnership's Ecosystem Monitoring Program (see Sagar et al. 2013). Analyses were conducted with the JMP statistical package.

3.2 Results

3.2.1 Objective 1: Increase the number of salmon in Mirror Lake Complex from pre-restoration levels

Salmonid Catch per Unit Effort (Density), Seasonal Occurrence, and Percentage of Catch. Mean Chinook and coho salmon densities (as estimated from catch per unit effort) at the Mirror Lake Complex sites over the sampling season are shown in Figure 5 – 6, and densities by sampling event, including peak densities for each year, are shown in Table 4 – 6. Mean densities over the sampling season for the Culvert and Lake sites were calculated using data collected from April through September to produce a more standardized sampling period among years.

At the Culvert, Chinook salmon density showed no clear increasing or decreasing trends between 2008 and 2012 (Figure 4), although densities of unmarked Chinook were generally higher and more variable than densities of marked Chinook. In 2008, and 2010-2012, the average density of unmarked Chinook salmon at the Culvert ranged from 3.3 to 51.1 fish per 1000 m^2, values that were comparable to the average density for unmarked Chinook salmon at the Reach H EMP sites (12.6 fish per 1000 m^2). However, in 2009, the density of unmarked Chinook salmon at the Culvert (97.6 fish per 1000 m^2) was much higher than in any other year, and higher than the mean density at the Reach H EMP sites. Because of the high variability in densities, however, none of these values were significantly different ($p = 0.072$; One-way ANOVA).

Densities of marked Chinook salmon (Figure 4) averaged over the sampling season remained fairly stable from 2008 to 2012 (ranging from 1.2 fish per 1000 m^2 in 2008 to 29.1 fish per 1000 m^2 in 2012), and were comparable to the average density of marked Chinook salmon at the Reach

H EMP sites (2.0 fish per 1000 m^2). No statistically significant differences were observed among mean densities over the 2008-2012 sampling period ($p = 0.6210$; One-way ANOVA).

The period during which Chinook salmon were present at the Culvert site varied from year to year (Table 4). In most sampling years, unmarked juvenile Chinook were present only through June, but in 2010, unmarked Chinook salmon were present through August. In 2011 and 2012, sampling was extended through October, but no Chinook salmon were observed. Peak densities of unmarked Chinook salmon typically occurred in May or June, and ranged from a low of 16.4 fish per 1000 m^2 in May 2011 to a high value of 827 fish per 1000 m^2 in May 2009. Peak densities for 2008, 2010, and 2011 were not significantly different from each other or from the peak density at the Reach H EMP sites but the peak density in 2009 was significantly higher than the peak densities at the Culvert in any other year, as well as at the Reach H EMP sites (One-way ANOVA, and Tukey-Kramer HSD Multiple range test, $p \leq 0.05$).

Marked Chinook salmon were observed on an occasional basis, and the months when they occurred varied from year to year (Table 4). Peak densities of marked Chinook salmon ranged from a low of 5.6 fish per 1000 m^2 in August 2008 to a high of 245.7 fish per 1000 m^2 in April 2011. Peak densities of marked Chinook in 2011 and 2012 were significantly higher than in 2010, and densities in 2010 were significantly higher than in 2008, 2009, or at Reach H (Table 4, One-way ANOVA, and Tukey-Kramer HSD Multiple range test, $p < 0.05$),

Like Chinook salmon density, the average density of unmarked coho salmon at the Culvert was variable between 2008 and 2012, and comparable to the average density for unmarked coho salmon at the Reach H EMP sites (Figure 4). Density declined between 2008 and 2009, but tended to increase from 2009 to 2011, then decreased again in 2012, although no statistically significant differences were found among years ($p = 0.7752$).

Marked coho salmon, on the other hand, were found at the highest density in 2008 (28.2 fish per 1000 m^2), and then densities declined steadily from 2009 to 2011, when no marked coho salmon were found. However, a small number of marked coho were found in 2012 (3.2 fish per 1000m^2). In comparison to the average density of marked coho salmon at the Reach H sites (0.05 fish per 1000 m^2), the density at the Culvert was higher in 2008, but similar in subsequent years. The mean density of marked coho salmon was significantly higher at the Culvert in 2008 than the Reach H site mean ($p = 0.0151$; Figure 3).

The period of time that coho salmon were present at the Culvert site varied from year to year (Table 4). In 2008, 2009 and 2012, unmarked coho salmon were found only through May, but in 2010, unmarked coho salmon were present through July. In 2011, sampling was limited because of high water and could not be conducted in June. Unmarked coho salmon were present in May, and may have been present in June, but were absent from the site from July onward. Sampling extended until October in 2011, but no coho salmon were found at that time. The peak density of unmarked coho was highest in May of 2011 (194.6 fish per 1000 m^2), but because of the high variability in density and small number of sets, no significant differences were among years observed ($p = 0.4913$). Marked coho salmon were observed on an occasional basis, and the months when they occurred varied from year to year. The highest peak density of marked coho salmon at the Culvert was observed in May of 2008 (84.5 fish per 1000 m^2), and was significantly higher than peak densities at the Culvert in any other year and significantly higher than the peak density at the Reach H EMP sites ($p = 0.0001$).

At the Lake site (Figure 5), mean Chinook salmon densities were consistently lower than at the Culvert, but showed similar trends. As at the Culvert, Chinook salmon densities at the Lake neither increased nor decreased between 2008 and 2012, but were highest in 2009. The average density of unmarked Chinook salmon at the Lake ranged from 0.81 fish per 1000 m^2 in 2010 to 24.0 fish per 1000 m^2 in 2009. These values were comparable to those for unmarked Chinook salmon from the Reach H EMP sites. No significant differences were observed for mean densities of unmarked Chinook salmon among years or with the Reach H EMP sites ($0.1974 < p < 0.4128$). Average densities of marked Chinook salmon varied from no fish in 2010 to 10.87 fish per 1000 m^2 in 2012, values comparable to those observed for marked Chinook salmon at the Reach H EMP sites. No significant differences were observed for mean densities of marked Chinook salmon among years or with the Reach H EMP sites ($0.1026 < p < 0.5744$).

Unmarked Chinook salmon were typically present at the Lake site in May and June (Table 5); but in 2010 they were found in May only and in 2012 from April through June. Marked Chinook salmon were found during the same time frame. In 2011, sampling was extended through December at the Lake site, but no Chinook salmon were found at the site in the fall and winter months. At the Lake site, peak Chinook salmon densities were consistently lower than at the Culvert, but showed similar trends, with highest peak densities in 2009. The yearly peak density of unmarked Chinook varied from 3.8 fish per 1000 m^2 in May 2010 to 164 fish per 1000 m^2 in May 2009. These values were also comparable to those for unmarked Chinook salmon from the Reach H EMP sites. No significant differences were observed for peak densities of unmarked Chinook salmon among years or with the Reach H EMP sites ($p=0.1933$). Peak densities of marked Chinook salmon ranged from 3.6 fish per 1000 m^2 in May 2008 to 65.2 fish per 1000 m^2 in May 2012, values comparable to those observed for marked Chinook salmon at the Reach H EMP sites. No significant differences were observed for peak densities of marked Chinook salmon among years or with the Reach H EMP sites ($p = 0.6548$), although densities were higher in all years following 2008.

Mean densities of unmarked coho salmon at the Lake were very similar in 2008 and 2009 (ranging from 1.1 to 1.4 fish per 1000 m^2). Mean density increased in 2010 to 8.7 fish per 1000 m^2, then declined again in 2011 to levels comparable to those observed earlier, and in 2012, no unmarked coho were found at the Lake. No significant differences were observed for mean densities of unmarked coho salmon among years or with the Reach H EMP sites ($0.349 < p < 0.6538$), although the density of unmarked coho salmon at the Lake was generally low in comparison to the Culvert or Reach H EMP. Marked coho salmon were not found at the Lake in any sampling year.

Unmarked coho salmon were generally found at the Lake site in May and June (Table 5) and in 2010 were also present in April. In 2011, sampling was extended through December at the Lake site, and coho salmon were observed in October and December. Peak densities of unmarked coho salmon at the Lake were very similar in 2008 and 2009; 8.9 fish per 1000 m^2 in June 2008 and 8.4 fish per 1000 m^2 in May 2009). Peak density increased in 2010 (33.2 fish per 1000 m^2 in May 2010), then declined again in 2011 (2.3 fish per 1000 m^2, observed in October). No significant differences were observed for peak densities of unmarked coho salmon among years or with the Reach H EMP sites ($p=1.000$).

Chum salmon (Table 6) were found only at the Culvert site, with mean and peak densities comparable to those seen at the Reach H EMP sites. All chum salmon were collected in April. Steelhead trout (Table 6) were found occasionally at the Culvert in 2010 only, at densities comparable to the Reach H EMP sites. No significant differences were observed in either mean or peak densities of chum or steelhead trout at the Culvert among sampling years, or with the Reach H EMP sites ($p > 0.05$).

At the Confluence, Latourell Creek, and Youngs Creek, marked coho salmon were consistently absent from all three sites, but mean densities of unmarked coho salmon were generally much higher than at the Reach H EMP sites (Figure 6). The Confluence and Latourell Creek sites were sampled only from 2010 through 2012, but did show some changes in coho density between these three years. The mean density of coho at the Confluence was the highest in 2010 and decreased in subsequent years (from 731 fish per 1000 m^2 in 2010 to 2.4 fish per 1000 m^2 in 2012). However, the decrease in density was not statistically significant. Coho density at the Confluence in 2010 was significantly higher than the mean density at reach H sites, but not in other years. At Latourell Creek, the mean density of coho salmon was lower in 2011 than in 2010 (2950 fish per 1000 m^2 in 2010 as compared to 722 fish per 1000 m^2 in 2011), but the density increased in 2012 (1410 fish per 1000m^2). The interannual changes in coho density at Latourell Creek were significantly different, but in all years Latourell Creek densities were consistently significantly higher than those at Reach H EMP sites.

At Youngs Creek, the mean density of coho salmon declined significantly between 2008 (3536 fish per 1000m^2) and 2009 (268 fish per 1000m^2), but increased in 2010 (1213 fish per 1000m^2) and 2011 (4973 fish per 1000 m^2), then declined in again 2012 (2355 fish per 1000m^2) (Figure 6). These interannual changes were significantly different; moreover, the mean density of unmarked coho salmon at Youngs Creek was consistently higher than the mean density at the Reach H EMP sites (One-way ANOVA, and Tukey-Kramer HSD Multiple range test, $p \leq 0.05$).

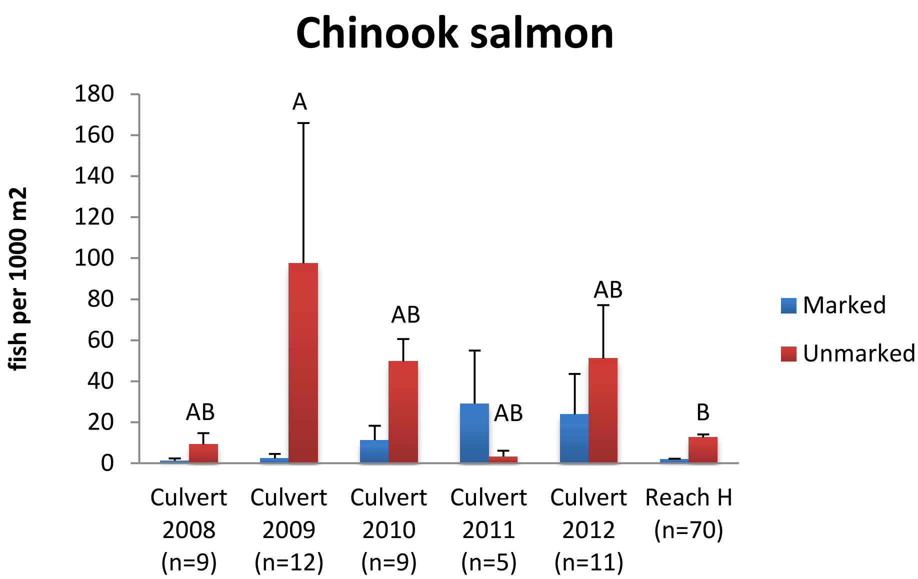
Unmarked coho salmon were consistently found at Youngs Creek throughout the sampling season from 2008 to 2012 (Table 7). Peak densities of unmarked coho salmon were much higher at Youngs Creek than at the Reach H EMP sites (Table 7), and followed a pattern similar to mean densities, with the lowest value at 550 fish per 1000 m^2 in August 2009 and the highest at 7750 fish per 1000 m^2 in July 2011. The peak density of unmarked coho salmon at Youngs Creek was significantly higher in 2011 than in any other sampling year, and in all years, peak densities of unmarked coho at Youngs Creek were significantly higher than the average peak densitiy at the Reach H EMP sites (One-way ANOVA, and Tukey-Kramer HSD Multiple range test, $p < 0.05$),

The Confluence and Latourell Creeks were less consistently sampled than Youngs Creek, so the seasonal presence of salmon at these sites is more difficult to evaluate, but unmarked coho salmon were found on most occasions when the sites were sampled (Table 8). At the Confluence site, peak coho density was significantly lower in 2011 and 2012 than in 2010 (Table 8, $p \leq 0.05$). Also, peak coho salmon density in 2010 was significantly higher than peak coho density at the Reach H EMP sites. At Latourell Creek, mean and peak densities were the same as sampling was limited to one event per year. Density was the highest in 2010, decreased in 2011 and increased again in 2012. Significant interannual variation in coho salmon density was observed at Latourell Creek. However, even in 2011 when the density was lowest, it was significantly higher than the peak density at the Reach H ($p = 0.001$; Table 8).

Chinook salmon were not found at Latourell Creek or Youngs Creek, but were caught for the first time at the Confluence in 2011 (Figure 7,Table 8). In 2011, both unmarked and marked Chinook salmon were present at the site in May only (10.96 fish per 1000 m^2 and 13.7 fish per 1000 m^2, respectively, Table 8); whereas in 2012, only unmarked Chinook were present in April (3.7 fish per 1000 m^2). Figure 7 shows the density of unmarked Chinook averaged over the sampling season. These values were somewhat lower than mean and peak densities for marked and unmarked Chinook salmon at the Reach H EMP sites. However, mean density values among years were not significantly different for either marked or unmarked Chinook, nor were they different from values for the Reach H EMP sites ($p > 0.05$).

At Youngs Creek (Table 7), steelhead trout were found during most sampling years, usually between July and October, at densities comparable to or slightly higher than the mean for the Reach H sites. However, steelhead densities showed no clear increasing or decreasing trends, and no significant differences among years or with Reach H EMP sites (p=0.2909; Table 7). Steelhead were not collected at the Confluence or Latourell Creek.

At both the Culvert and Lake sites, the proportion of salmonids in the total catch generally declined between 2008 and 2012 (Figure 8). At the Culvert the percentage of salmonids in the total catch changed from a high of 25.7% in 2008 to a low of 5.2% in 2011 increasing somewhat in 2012 to 10.7%. At the Lake it changed from a high of 4.5% in 2008 to a low of 0.70-0.78% in 2011 and 2012. At Youngs Creek, the percentage of salmonids in the total catch varied from 96.2% in 2010 to 53.4% and 58.5% in 2011 and 2012, but showed no consistent increasing or decreasing trends. The Confluence and Latourell Creek sites were sampled only from 2010 to 2012, but at both of these sites, the percentage of salmonids in the total catch was lower in 2012 than in 2010, declining from 25.2% to 0.02% at the Confluence and from 52.3% to 3.6% at Latourell Creek.



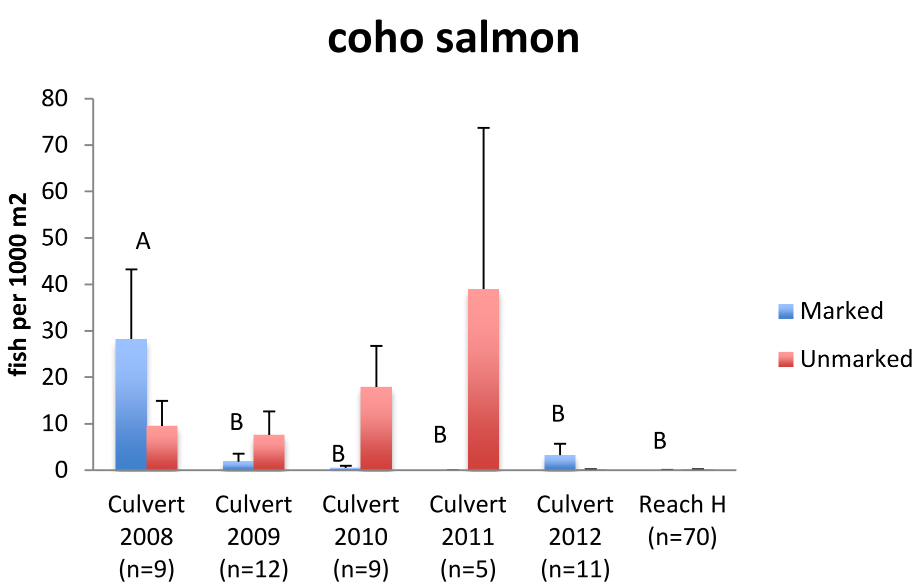
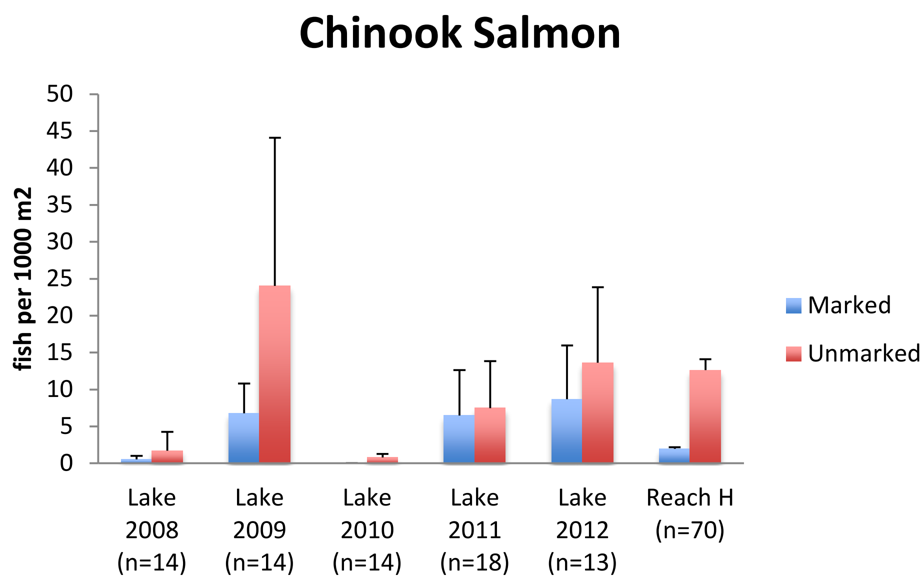


Figure 4. Mean densities of Chinook and coho salmon at the Mirror Lake Culvert site from 2008 to 2012, as compared to Reach H sites sampled as part of the Ecosystem Monitoring Program. Values of n indicate the number of sampling events during that year. Values with different letter superscripts are significantly different (One-way ANOVA, Tukey's HSD multiple range test, $p < 0.05$). No significant differences were observed among densities for unmarked coho salmon or marked Chinook salmon. Density values were weighted by area sampled when testing for differences among the means.

Table 4. Mean monthly densities of salmon species at the Culvert site from 2008 to 2012, as compared to mean monthly densities at the Reach H Ecosystem Monitoring Program sites. Values with different letter superscripts are significantly different peak monthly densities (1-way ANOVA, Tukey-Kramer HSD test, $p < 0.05$)

	n	Unmarked Chinook salmon density in fish per 1000 m^2 (mean ± SE)	Marked Chinook density in fish per 1000 m^2 (mean ± SE)	Unmarked coho salmon density in fish per 1000 m^2 (mean ± SE)	Marked coho density in fish per 1000 m^2 (mean ± SE)
Culvert May 2008	3	$22.0±15.4^B$	0	28.5±11.3	$84.5±26.0^A$
Culvert June 2008	1	7.0	0	0	0
Culvert July 2008	2	0	0	0	0
Culvert Aug 2008	2	0	$5.6 ± 5.6^B$	0	0
Culvert Sept 2008	2	0	0	0	0
Culvert April 2009	1	310	0	57.6	0
Culvert May 2009	1	827^A	25.7^B	33	22.1
Culvert June 2009	4	1.02 ± 1.03	1.03 ± 1.03	0	0
Culvert July 2009	3	0	0	0	0
Culvert Aug 2009	3	0	0	0	0
Culvert April 2010	2	25.4 ± 11.6	0	16.7 ± 2.8	0
Culvert May 2010	1	118^B	0	18.8	0
Culvert June 2010	2	48 ± 0.8	$50.6±4.8^B$	52.9 ± 36.6	0
Culvert July 2010	2	28 ± 15	0	1.6 ± 1.6	2.1 ± 2.1
Culvert Aug 2010	2	2.7 ± 2.7	0	0	0
Culvert May 2011	1	16.4^B	145.3^A	194.6	0
Culvert June 2011	0	NS	NS	NS	NS
Culvert July 2011	2	0	0	0	0
Culvert Aug 2011	2	0	0	0	0
Culvert Sept 2011	0	NS	NS	NS	NS
Culvert Oct 2011	2	0	0	0	0
Culvert April 2012	1	193	245.7^A	2.0	3.9
Culvert May 2012	1	274^B	17.5	0	31.2
Culvert June 2012	1	93.9	0	0	0
Culvert July 2012	2	0	0	0	0
Culvert Aug 2012	3	0	0	0	0
Culvert Sept 2012	3	0	0	0	0
Culvert Oct 2012	2	0	0	0	0
Reach H Oct	6	0	0	6.7±4.9	0
Reach H Nov	6	0	0	1.1±1.1	0
Reach H Dec	6	2.2±1.4	0	11.9±8.4	1.5±1.5
Reach H Feb	3	0	0	0	0
Reach H Mar	1	0	0	0	0
Reach H April	14	26.5±20.5	15.0±9.6	7.0±6.2	0
Reach H May	14	20.6±18.1	$24.2±15.7^B$	2.7±1.2	$23.4±9.0^B$
Reach H June	3	$52.64±35.6^B$	1.2±1.2	5.0±2.5	7.2±5.3
Reach H July	17	1.7±1.7	1.1±0.8	0.4±0.4	0
Reach H Aug	14	0	0	26.9±26.9	0
Reach H Sept	5	0	0	0	0



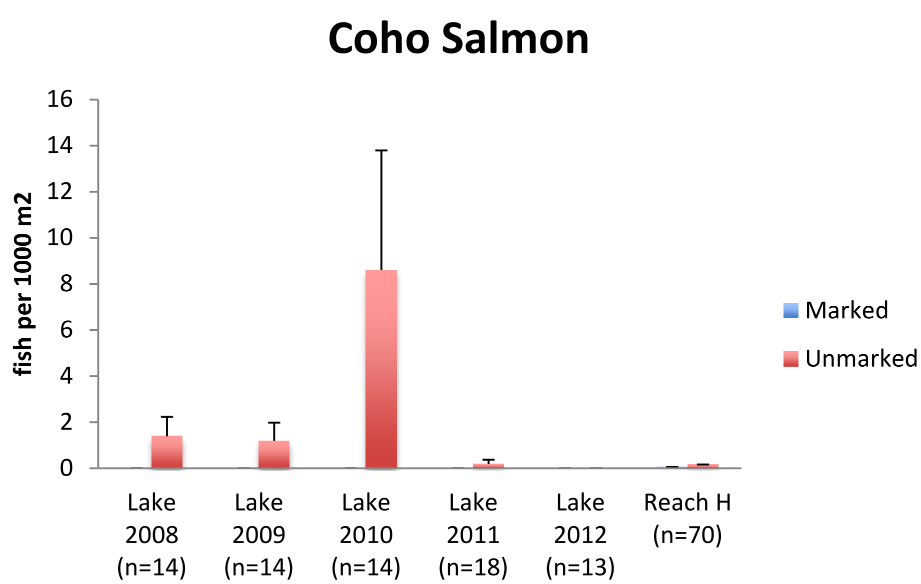


Figure 5. Mean densities of marked and unmarked Chinook and coho salmon at the Mirror Lake Complex Lake site from 2008 to 2012, as compared to Reach H sites sampled as part of the Ecosystem Monitoring Program. Values of *n* indicate the number of sampling events during that year. No marked coho salmon were collected at the Lake site. No significant differences were detected for any group of fish among years or with the Reach H sites (One-way ANOVA, $p > 0.05$). Density values were weighted by area sampled when testing for differences among the means.

Table 5. Mean monthly densities of salmon species at the Lake site from 2008 to 2012, as compared to mean monthly densities at the Reach H Ecosystem Monitoring Program sites.

Site	n	Unmarked Chinook salmon density in fish per 1000 m^2 (mean ± SE)	Marked chinook density in fish per 1000 m^2 (mean ± SE)	Unmarked coho salmon density in fish per 1000 m^2 (mean ± SE)	Marked coho density in fish per 1000 m^2 (mean ± SE)
Lake 2008 April	3	0	0	0	0
Lake 2008 May	2	18.3±18.3	3.6±3.6	0.91±0.91	0
Lake 2008 June	2	4.8±4.8	0	8.9 ± 1.2	0
Lake 2008 July	1	0	0	0	0
Lake 2008 Aug	3	0	0	0	0
Lake 2008 Sept	3	0	0	0	0
Lake 2009 April	3	0	0	0	0
Lake 2009 May	2	164.1±128.3	43.2 ± 5.5	8.4 ± 1.1	0
Lake 2009 June	4	2.1±2.1	2.1 ± 2.1	0	0
Lake 2009 July	3	0	0	0	0
Lake 2009 Aug	2	0	0	0	0
Lake 2009 Sept	0	NS	NS	NS	NS
Lake 2010 April	3	0	0	1.8 ± 1.0	0
Lake 2010 May	3	3.8±1.5	0	33.2 ± 22	0
Lake 2010 June	2	0	0	8.2 ± 0.5	0
Lake 2010 July	3	0	0	0	0
Lake 2010 Aug	3	0	0	0	0
Lake 2010 Sept	0	NS	NS	NS	NS
Lake 2011 April	0	NS	NS	NS	NS
Lake 2011 May	2	38.0±36.0	35.6±35.6		0
Lake 2011 June	1	6.4	2.1	2.13	0
Lake 2011 July	2	0	0	0	0
Lake 2011 Aug	3	0	0	0	0
Lake 2011 Sept	3	0	0	0	0
Lake 2011 Oct	3	0	0	2.3±2.3	0
Lake 2011 Nov	3	0	0	0	0
Lake 2011 Dec	3	0	0	1.4±1.4	0
Lake 2012 April	2	3.2±3.5	0	0	0
Lake 2012 May	2	89.0±71.4	65.2±47.6	0	0
Lake 2012 June	1	13.7	0	0	0
Lake 2012 July	1	0	0	0	0
Lake 2012 Aug	3	0	0	0	0
Lake 2012 Sept	3	0	0	0	0
Lake 2012 Oct	3	0	0	0	0
Reach H Feb	3	0	0	0	0
Reach H Mar	1	0	0	0	0
Reach H April	14	26.5±20.5	15±9.6	7±6.2	0
Reach H May	14	20.6±18.1	24.2±15.3	2.7±1.2	23.4±9.0
Reach H June	3	52.6±35.6	1.2±1.2	5.0±2.5	7.2±5.3
Reach H July	17	1.7±1.7	1.1±0.8	0.4±0.4	0
Reach H Aug	14	0	0	26.9±26.9	0
Reach H Sept	5	0	0	0	0
Reach H Oct	6	0	0	6.7±4.9	0
Reach H Nov	6	0	0	1.1±1.1	0
Reach H Dec	6	2.2±1.4	0	11.9±8.4	1.5±1.5

Table 6. Mean and peak chum salmon and steelhead trout densities at the Mirror Lake Complex Culvert and Lake sites between 2008 and 2012, as compared to mean and peak densities of these species at Reach H sites sampled as part of the Ecosystem Monitoring Program (EMP). There were no statistically significant differences in densities among sites and years, or between Mirror Lake Complex sites and Reach H sites (ANOVA, $p \leq 0.05$).

Site	Fish density per 1000 m2			
	Chum salmon		Steelhead trout	
	Mean over sampling season (± SE)	Peak monthly mean April (± SE)	Mean over sampling season (± SE)	Peak monthly mean May-Jul (± SE)
Culvert 2008	0 (n=9)	NS	0 (n=9)	0 (n=6)
Culvert 2009	2.3 ± 2.3 (n=12)	27.4 (n=1)	0 (n=12)	0 (n=8)
Culvert 2010	0.22 ± 0.22 (n=9)	0.97 ± 0.97 (n=2)	0.90 ± 0.90 (n=9)	1.6 ± 1.6^{A} (n=2)
Culvert 2011	0 (n=7)	NS	0 (n=7)	0 (n=3)
Culvert 2012	0 (n=13)	0 (n=1)	0 (n=13)	0 (n=4)
Reach H	1.54 ± 0.97 (n=89)	9.7 ± 5.8 (n=14)	0.13 ± 0.07 (n=89)	0^{B} (n=32)
	p = 0.9392	p=0.7133	p=0.1718	p=0.0514

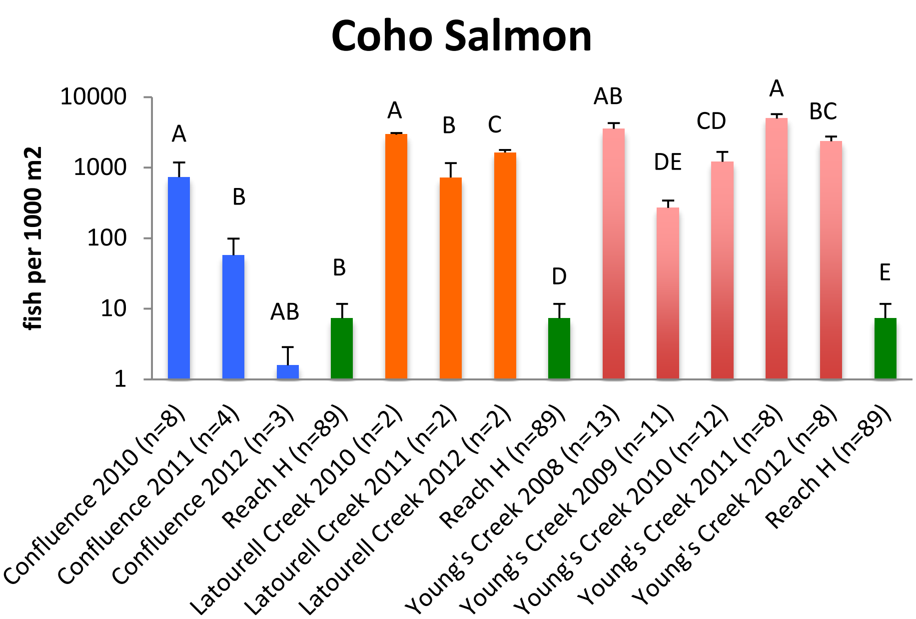


Figure 6. Mean densities of unmarked coho salmon at the Mirror Lake Complex Confluence and Latourell Creek sites in 2010-2012, and Youngs Creek site 2008-2012, as compared to Reach H sites sampled as part of the Ecosystem Monitoring Program. No marked coho salmon were collected at either of these sites. Fish densities are compared among years within sites and to the average density at the Reach H EMP sites. Values with different letter superscripts are significantly different (One-way ANOVA, Tukey's HSD multiple range test, $p < 0.05$). Density values were weighted by area sampled when testing for differences among the means.

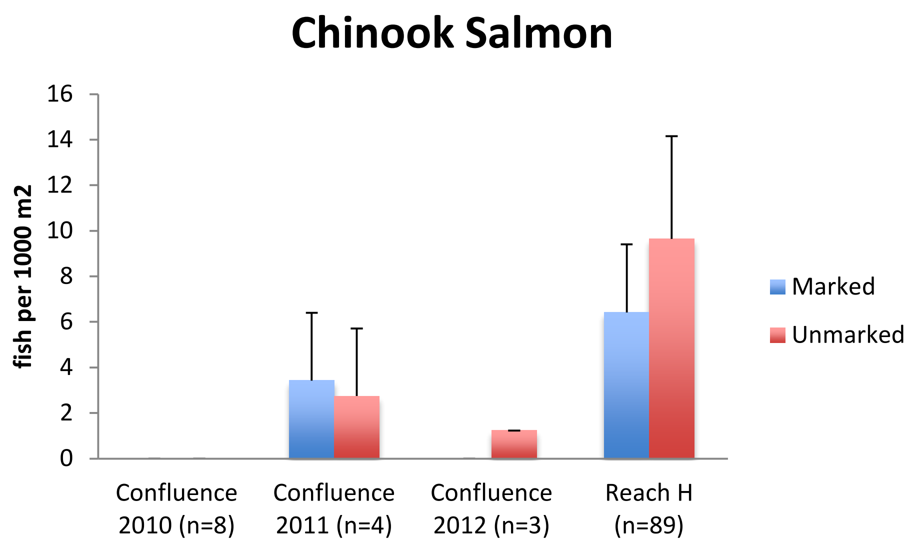


Figure 7. Mean densities of marked and unmarked Chinook salmon at the Mirror Lake Complex Confluence site in 2010 - 2012, as compared to Reach H sites sampled as part of the Ecosystem Monitoring Program. No significant differences were detected for either marked or unmarked Chinook between years or with the Reach H sites (One-way ANOVA, $p > 0.05$). Density values were weighted by area sampled when testing for differences among the means.

Table 7. Mean monthly densities of salmon species at Youngs Creek, the Confluence and Latourell Creek between 2008 and 2012

Site	n	Unmarked Chinook salmon density	Marked Chinook salmon density	Unmarked coho salmon density in fish per 1000 m^2 (mean ± SE)	Steelhead trout density in fish per 1000 m^2 (mean ± SE)
Youngs Creek 2008 April	1	0	0	NS	NS
Youngs Creek 2008 May	3	0	0	715±231	0
Youngs Creek 2008 June	0	0	0	NS	NS
Youngs Creek 2008 July	3	0	0	5798±487	22±22
Youngs Creek 2008 Aug	3	0	0	**6513±677**	22±22
Youngs Creek 2008 Sept	3	0	0	2299±455	**11±6.3**
Youngs Creek 2009 April	2	0	0	40 ± 10	0
Youngs Creek 2009 May	3	0	0	23 ± 3	**0**
Youngs Creek 2009 June	2	0	0	370 ± 70	0
Youngs Creek 2009 July	2	0	0	480 ± 200	0
Youngs Creek 2009 Aug	2	0	0	**550 ± 40**	0
Youngs Creek 2009 Sept	0	0	0	NS	NS
Youngs Creek 2010 April	3	0	0	117 ± 39	0
Youngs Creek 2010 May	3	0	0	222±79	0
Youngs Creek 2010 June	0	0	0	NS	NS
Youngs Creek 2010 July	3	0	0	**3310 ± 1091**	**3.3 ± 3.3**
Youngs Creek 2010 Aug	3	0	0	1203±797	0
Youngs Creek 2010 Sept	0	0	0	NS	NS
Youngs Creek 2011 April	0	0	0	NS	NS
Youngs Creek 2011 May	0	0	0	NS	NS
Youngs Creek 2011 June	0	0	0	NS	NS
Youngs Creek 2011 July	2	0	0	**7750±583**	0
Youngs Creek 2011 Aug	2	0	0	5870 ± 920	5.0 ± 5.0
Youngs Creek 2011 Sept	2	0	0	2120 ± 510	0
Youngs Creek 2011 Oct	2	0	0	4155 ± 15b	**10 ± 0**
Youngs Creek 2012 April	0	0	0	NS	NS
Youngs Creek 2012 May	0	0	0	NS	NS
Youngs Creek 2012 June	0	0	0	NS	NS
Youngs Creek 2012 July	0	0	0	NS	NS
Youngs Creek 2012 Aug	2	0	0	**3092±785**	5.0 ± 5.0
Youngs Creek 2012 Sept	3	0	0	2350±731	3.3±3.3
Youngs Creek 2012 Oct	3	0	0	1260±80	**0**
Reach H Feb	3	0	0	0	0
Reach H Mar	1	0	0	0	0
Reach H April	14	26.5±20.5	15.0±9.6	7.0±6.2	0
Reach H May	14	20.6±18.1	**24.2±15.7**	2.7±1.2	23.4±9.0
Reach H June	3	**52.64±35.6**	1.2±1.2	5.0±2.5	7.2±5.3
Reach H July	17	1.7±1.7	1.1±0.8	0.4±0.4	0
Reach H Aug	14	0	0	**26.9±26.9**	0
Reach H Sept	5	0	0	0	0
Reach H Oct	6	0	0	6.7±4.9	0
Reach H Nov	6	0	0	1.1±1.1	0
Reach H Dec	6	2.2±1.4	0	11.9±8.4	1.5±1.5

Table 8. Mean monthly densities of salmon species at Confluence and Latourell Creek between 2008 and 2012

Site	n	Unmarked Chinook salmon density	Marked Chinook salmon density	Unmarked coho salmon density in fish per 1000 m^2 (mean ± SE)	Steelhead trout density in fish per 1000 m^2 (mean ± SE)
Confluence 2010 April	2	0	0	0	0
Confluence 2010 May	3	0	0	210 ± 120	0
Confluence 2010 June	0	NS	NS	NS	NS
Confluence 2010 July	3	0	0	1741 ± 1151	0
Confluence 2010 Aug	0	NS	NS	NS	NS
Confluence 2010 Sept	0	NS	NS	NS	NS
Confluence 2011 April	0	NS	NS	NS	NS
Confluence 2011 May	1	10.96	13.7	24.7	0
Confluence 2011 June	0	NS	NS	NS	NS
Confluence 2011 July	1	0	0	0	0
Confluence 2011 Aug	0	NS	NS	NS	NS
Confluence 2011 Sept	2	0	0	103 ± 98	0
Confluence 2012 April	1	3.7	0	0	0
Confluence 2012 May	0	NS	NS	NS	NS
Confluence 2012 June	0	NS	NS	NS	NS
Confluence 2012 July	0	NS	NS	NS	NS
Confluence 2012 Aug	2	0	0	2.4±2.4	0
Confluence 2012 Sept	0	NS	NS	NS	NS
Confluence 2012 Oct	0	NS	NS	NS	NS
Latourell Cr 2010 Aug	2	0	0	2950± 210	0
Latourell Cr 2011 Sept	2	0	0	722 ± 622	0
Latourell Cr 2012 Aug	2	0	0	1410±220	0
Reach H Feb	3	0	0	0	0
Reach H Mar	1	0	0	0	0
Reach H April	14	26.5±20.5	15.0±9.6	7.0±6.2	0
Reach H May	14	20.6±18.1	24.2±15.7	2.7±1.2	23.4±9.0
Reach H June	3	52.64±35.6	1.2±1.2	5.0±2.5	7.2±5.3
Reach H July	17	1.7±1.7	1.1±0.8	0.4±0.4	0
Reach H Aug	14	0	0	26.9±26.9	0
Reach H Sept	5	0	0	0	0
Reach H Oct	6	0	0	6.7±4.9	0
Reach H Nov	6	0	0	1.1±1.1	0
Reach H Dec	6	2.2±1.4	0	11.9±8.4	1.5±1.5

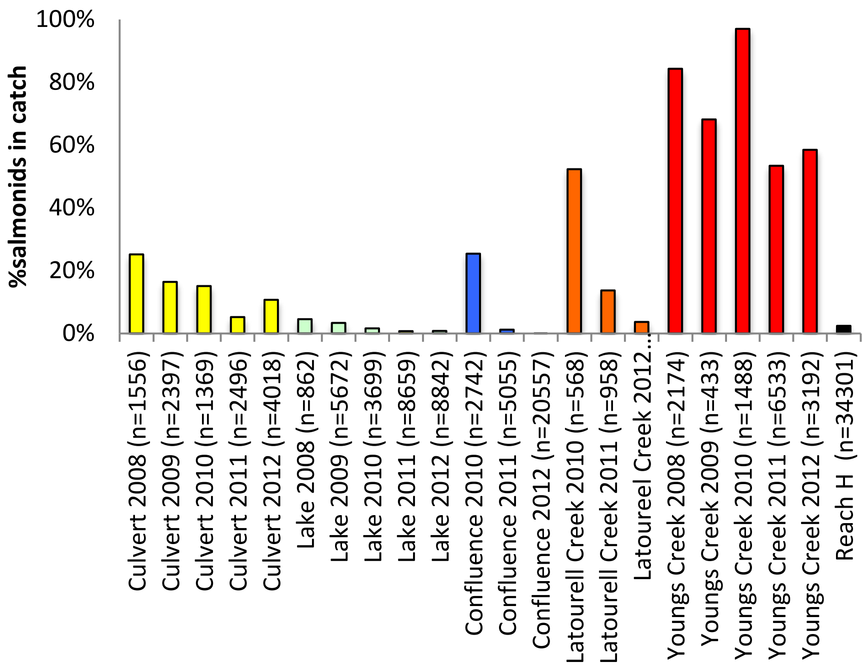


Figure 8. Percentage of salmonids in total catch at the Mirror Lake Complex sites from 2008 to 2012, as compared to Reach H sites sampled as part of the Ecosystem Monitoring Program (EMP). Values of *n* indicate the number of fish in the total catch for that site and year.

3.2.2 Objective 2: Restore fish community composition and characteristics in the Mirror Lake Complex to conditions present at undisturbed sites

Fish community species composition before and after restoration. While the specific fish species present at the Culvert and Lake sites remained about the same between 2008 and 2012, the proportions of different species at these two sites changed substantially (Figure 9). At both sites, the proportion of stickleback in catches increased steadily from 2008 to 2011, from 2.7% to 61% of the catch at the Culvert and from 29% to 90% of the catch at the Lake, and then declined somewhat in 2012, to 23% at the Culvert and 44% at the Lake. In contrast, proportions of some other species declined. For example, at the Culvert, the percentage of chiselmouth declined from 25% in 2008 to 1.3% in 2011, while the percentage of pumpkinseed declined from 19% to 1.2% over the same timeframe. At the Lake, chiselmouth declined from 23% of the catch in 2008 to 2.3% in 2011, and pumpkinseed declines from 22% to 0.9% of the catch. In 2012, however, with the decline of the sticklebacks in the catch, the percentages of these species increased somewhat. The percentages of pumpkinseed were 9% at the Culvert and 17% at the Lake, while the percentages of chiselmouth were 33% at the Culvert, and 30% at the Lake. Percentages of salmonids in the catch at these two sites also tended to fluctuate with the percentage of stickebacks present.

At Youngs Creek, species composition did not change as dramatically (Figure 9). Coho salmon and stickleback were the dominant species throughout the sampling period. While the proportion of stickleback varied from year to year, with highest values in 2009 and 2011, there was no increasing or decreasing trend. In 2012, native species including chiselmouth and largescale sucker was observed for the first time; while the non-native species mosquitofish was also observed. At the Confluence and Latourell Creeks, coho salmon and stickleback also were the dominant species from 2010 to 2012, but in 2011 a number of new species not noted before appeared at these sites (Figure 9). These included: bluegill, carp, mosquitofish, chiselmouth, chub, killifish, and largescale sucker, as well as Chinook salmon. Carp, chiselmouth, chub, mosquitofish, and pumpkinseed were also observed in 2012.

Species richness before and after restoration. Species richness (total number of species observed) remained fairly stable at the Culvert and Lake sites from 2008 to 2012 (Figure 10). However, at Confluence and Latourell Creek, the number of species increased markedly between 2010 and 2011, but decreased slightly in 2012 (Figure 10). At Youngs Creek, the number of species was changed little from 2008 to 2011, but increased somewhat in 2012.

Shannon Wiener Species Diversity Index before-and-after restoration. At both the Culvert and Lake sites, and especially at the Lake, species diversity generally declined from 2008 to 2011, but increased again in 2012 (Figure 10). At the Confluence diversity declined from 2010 to 2012, while at the Latourell Creek the diversity increased from 2010 to 2011 than declined sharply in 2012 (Figure 10). At Youngs Creek, species diversity varied between 2008 to 2012, but showed no clear increasing or decreasing trends (Figure 10).

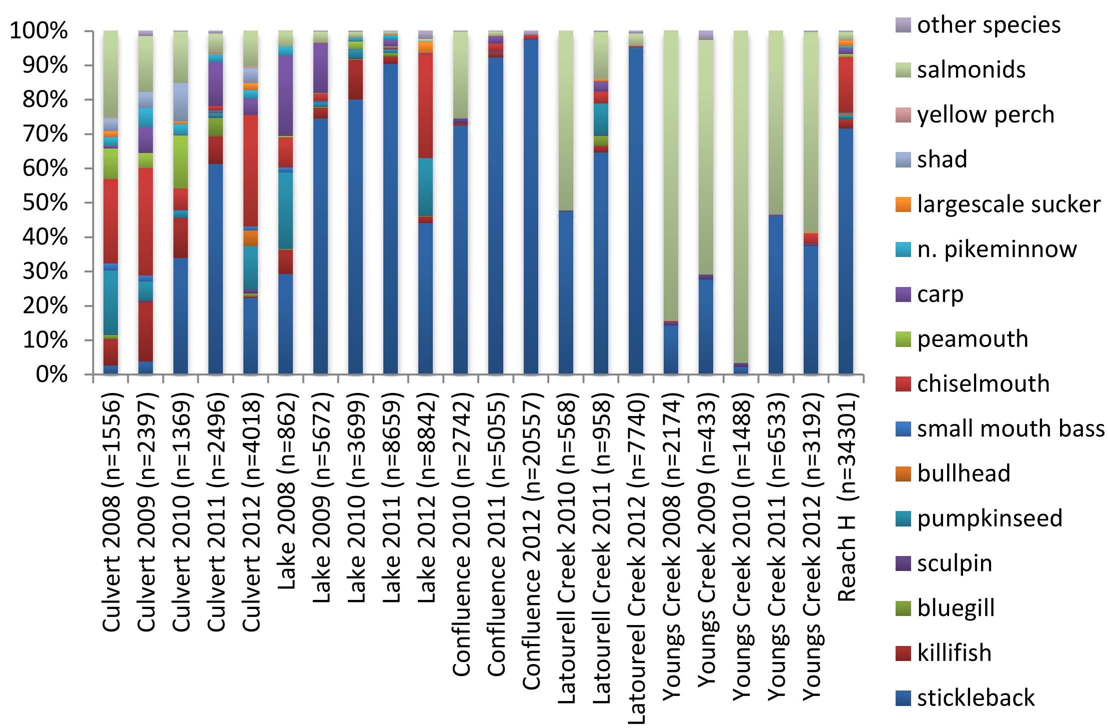


Figure 9. Fish community composition at Mirror Lake complex sites between 2008 and 2012, as compared to Reach H EMP sites. Values of *n* indicate the size of the total catch for that site and year.

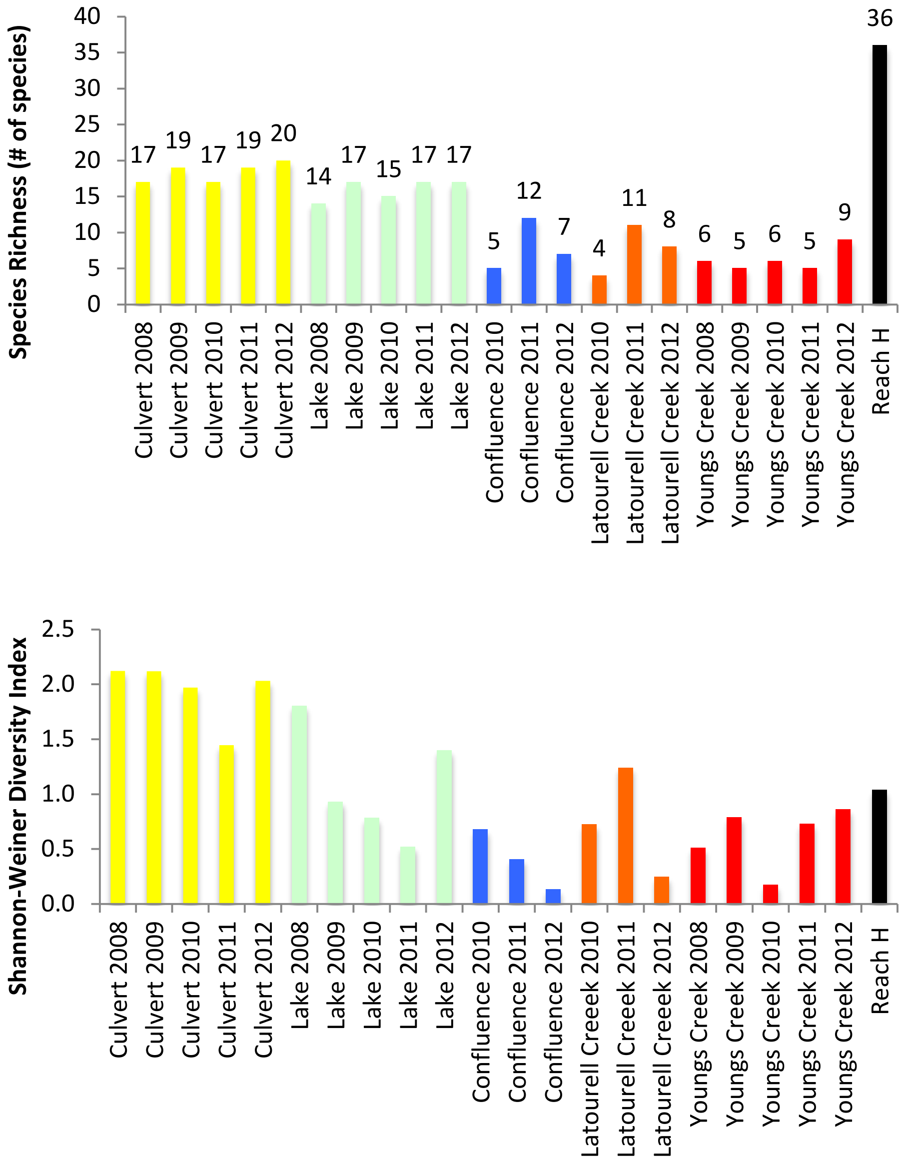


Figure 10. Species richness and Shannon-Wiener species diversity index at Mirror Lake complex sites from 2008 to 2012.

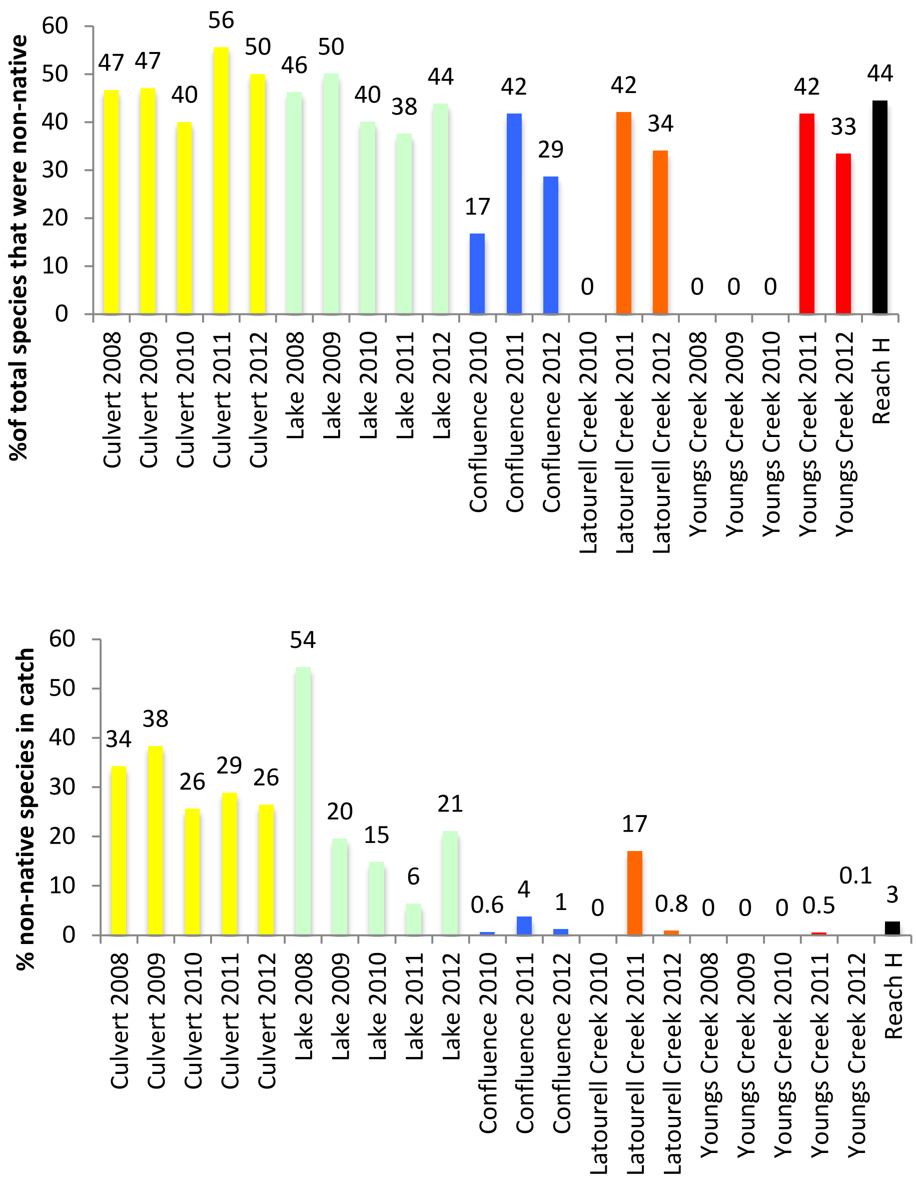


Figure 11. Percentages of non-native species caught at the Mirror Lake Complex sites between 2008 and 2012, calculated as the percentage of total species observed and as the percentage of the total catch.

% of native vs. non-native species before-and-after restoration. The percentage of the species captured that were non-native remained fairly constant at the Culvert and Lake sites between 2008 and 2012 (Figure 11). At the Confluence, the percentage of the non-native species increased from 17% in 2010 to 42% in 2011, but decreased to 29% in 2012. The Latourell Creek, no non-native species were caught in 2010, but the proportion increased to 42% 2011 and 34% in 2012. At Youngs Creek, no non-native species were observed between 2008 and 2010, but in 2011, nearly half of the species caught (42%) were non-native, and about a third (33%) were non-natives in 2012. By 2012, the proportions of non-native species caught at Mirror Lake complex sites were similar to the proportion observed at Reach H sites (44%).

While many non-native species were found at the Mirror Lake Complex sites, they rarely dominated catches (Figure 11). At the Culvert, non-native fish made up 26-38% of the total catch, with proportions slightly lower in 2010 and 2012 than in 2008 and 2009. At the Lake, however, the percentage of non-native fish in the catch declined steadily and dramatically from 54% in 2008 to 6% in 2011, then increased somewhat to 21% in 2012. These changes were paralleled by changes in the percentages of stickleback in catches at these two sites (Figure 11). At the Confluence, Latourell Creek, and Youngs Creek, non-native fish were either not present or made up a very low percentage of catches prior to 2011, but in 2011, the percentage of non-native fish in the catches increased at all three sites. The increase was especially dramatic at Latourell Creek, where 17% of the total catch was made up of non-native fish in 2011, as compared to no fish in 2010. In 2012, the numbers of non-native fish declined at these sites, to $\leq$1%. The percentage of non-native fish in the total catch at the Reach H sites was much lower than percentages observed than Culvert and Lake, but comparable to percentages observed at the Confluence, Latourell Creek and Youngs Creek.

3.2.3 Objective 3: Increase the diversity of salmon in Mirror Lake Complex from pre-restoration levels

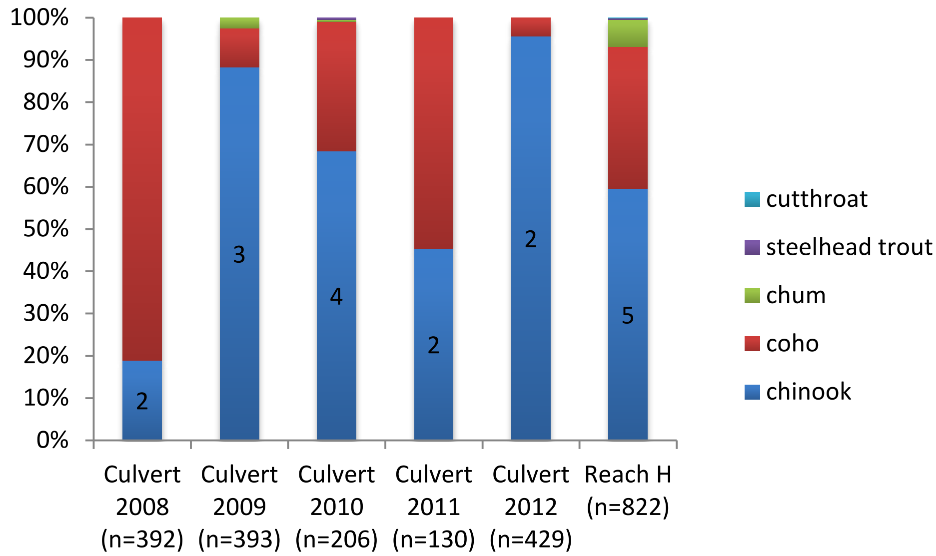
Number and diversity of salmonid species present before-and-after restoration. At the Culvert site (Figure 12), the salmon species observed included Chinook salmon, coho salmon, chum salmon, steelhead trout, and cutthroat trout. The number of species increased from 2 to 4 between 2008 and 2010, and then declined again to 2 in 2011 and in 2012. However, coho and Chinook salmon made up the majority of the salmonid catch in all years. At the Lake site (Figure 12), the salmonid species observed included Chinook salmon, coho salmon, steelhead trout, and cutthroat trout; no chum salmon were seen at the Lake. The number of species increased from 2 to 3 between 2008 and 2010, and then declined again to 2 in 2011, and 1 in 2012. However, as at the Culvert site, coho and Chinook salmon made up the majority of catches in all years. In comparison to the Ecosystem Monitoring sites in Reach H, the Culvert and Lake sites were used by fewer salmonid species (5 species vs. 1-4).

At the Confluence site (Figure 13), the salmonid species observed included Chinook salmon and coho salmon. In 2010, only coho salmon were observed at the site, but in 2011 and 2012, small numbers of Chinook salmon were observed as well. At Latourell Creek, from 2010 to 2012, only coho salmon were observed (Figure 13). At Youngs Creek, the salmonid species observed included coho salmon, steelhead trout, and cutthroat trout (Figure 13). The number of species varied from 1-3 between 2008 and 2012, but showed no increasing trend. In comparison to the Ecosystem Monitoring site in Reach H, the Confluence, Latourell Creek, and Youngs Creek were used by fewer salmonid species (5 species vs. 1-3).

Salmon species diversity, calculated using the Shannon-Weiner Diversity Index, varied widely among the Mirror Lake Complex sites, with highest values at the Culvert and Lake and lowest values at the Confluence, Latourell Creek, and Youngs Creek (Figure 14). Diversity index values at the Culvert ranged from 0.18 to 0.69, and tended to increase between 2008 and 2011, largely due to a more even distribution of Chinook and coho salmon in catches, but declined in 2012. At the Lake, diversity index values ranged from 0 to 0.57, with the highest value seen in 2008, and the lowest in 2012. At the Confluence, salmon diversity changed dramatically between 2010 and 2012, from to 0 to 0.5, because of the presence of Chinook salmon at the site in 2011 and in 2012. At both Latourell Creek and Youngs Creek, diversity was very low as only coho salmon made up and all of the salmon catches at these sites. Compared to the Reach H EMP sites, diversity was comparable at the Culvert, Lake and Confluence, but lower at Latourell Creek and Youngs Creek (Figure 14).

Percentage of marked hatchery salmon before and after restoration. Percentages of unmarked vs. marked Chinook salmon varied from year to year at the Culvert and Lake sites, but no clear trends were observed (Figure 15). At the Confluence, where Chinook salmon were observed for the first time in 2011, 40% of the 10 fish captured were unmarked, presumably wild fish. In 2012, 25% of the 4 Chinook salmon caught were unmarked. The percentages of unmarked fish for all years and sites were generally similar to the percentage of unmarked Chinook salmon found at the Reach H EMP sites (66%).

The Culvert site was the only site where marked, hatchery coho salmon were found. At the Culvert, the percentage of unmarked coho salmon increased steadily from 25% of the coho salmon catch in 2008 to 100% of the coho salmon catch in 2011 (Figure 15). In 2012, the however, the percentage of unmarked coho salmon at the Culvert decreased to 0.5%. From 2009-2011, the percentage of unmarked coho in catches at the Culvert was substantially higher than the percentage found at the Reach H EMP sites (49%).



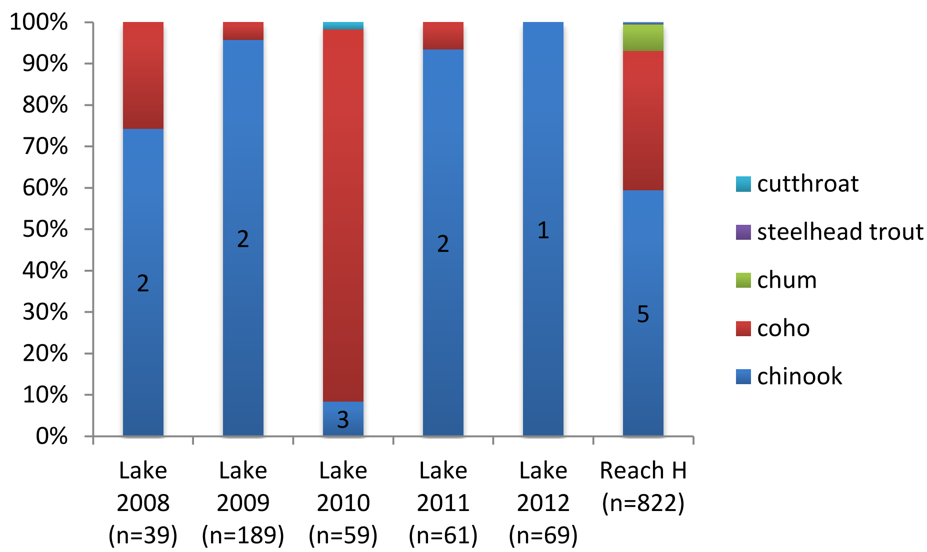


Figure 12. Proportions of different salmonid species at the a) Culvert and b) Lake sites between 2008 and 2012 and as compared to salmonid proportions at the Reach H Ecosysem Monitoring Project (EMP) sites in Reach H of the lower Columbia River. The total number of salmon species for each site and year are indicated on the bar graphs.

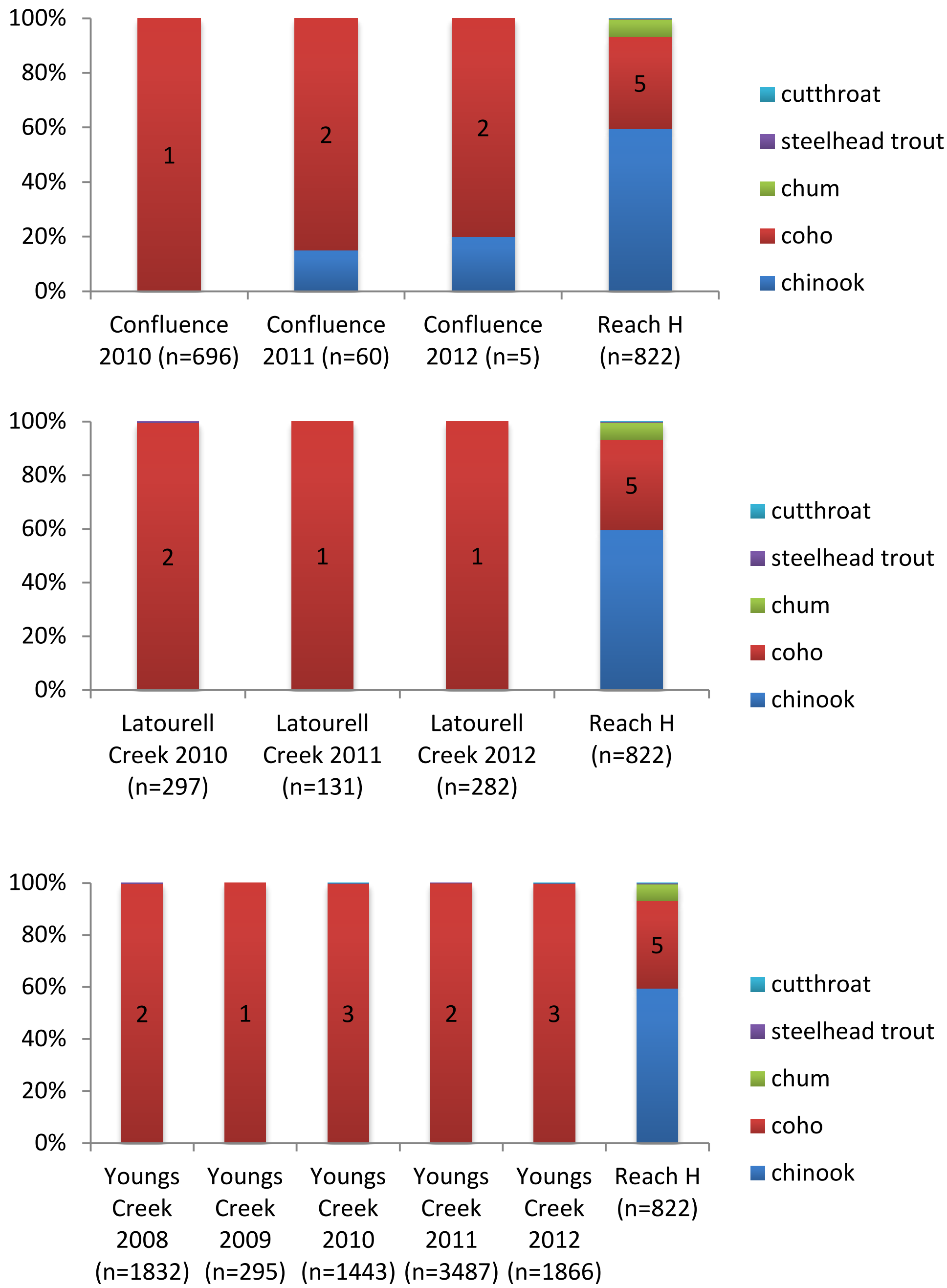


Figure 13. Proportions of different salmonid species at the Confluence, Latourell Creek and Youngs Creek sites between 2008 and 2012 and as compared to salmonid proportions at the Reach H Ecosysem Monitoring Project (EMP) sites in Reach H of the lower Columbia River. The total number of salmon species for each site and year is indicated on the bar graphs. The Confluence and Latourell Creeks were sampled from 2010 to 2012 only.

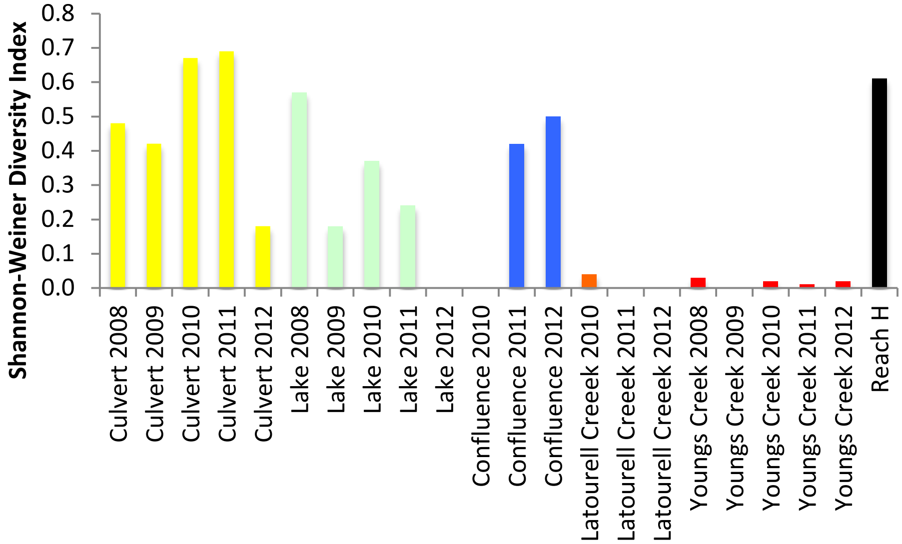
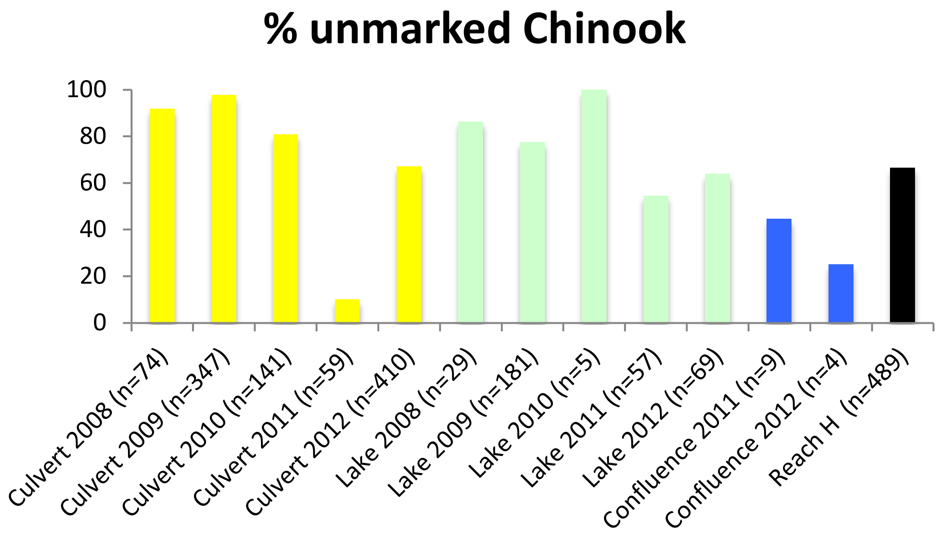


Figure 14. Shannon-Weiner diversity index values for salmonid species at the Mirror Lake Complex sites between 2008 and 2012, and at the Reach H Ecosystem Monitoring sites.



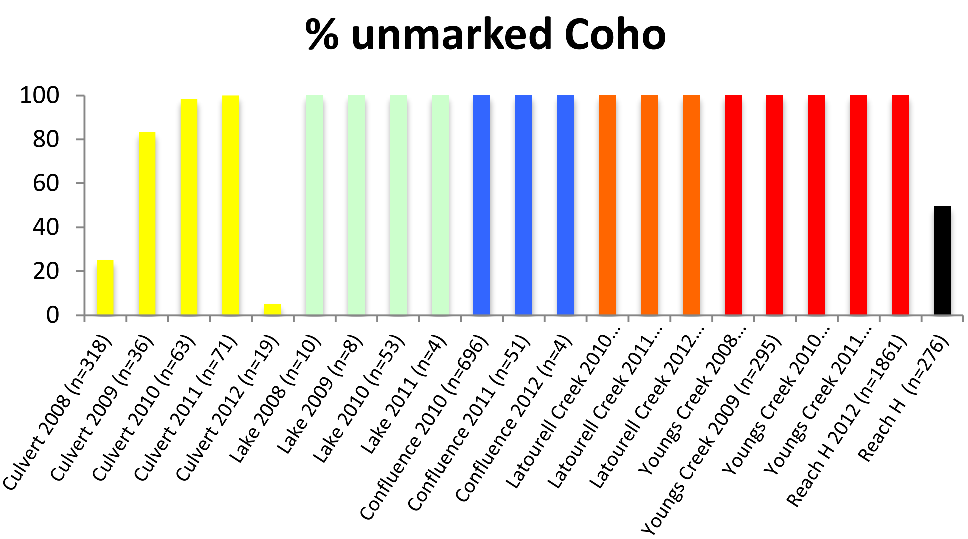


Figure 15. Percentages of unmarked Chinook and coho salmon at the Mirror lake Complex sites from 2008 to 2012.

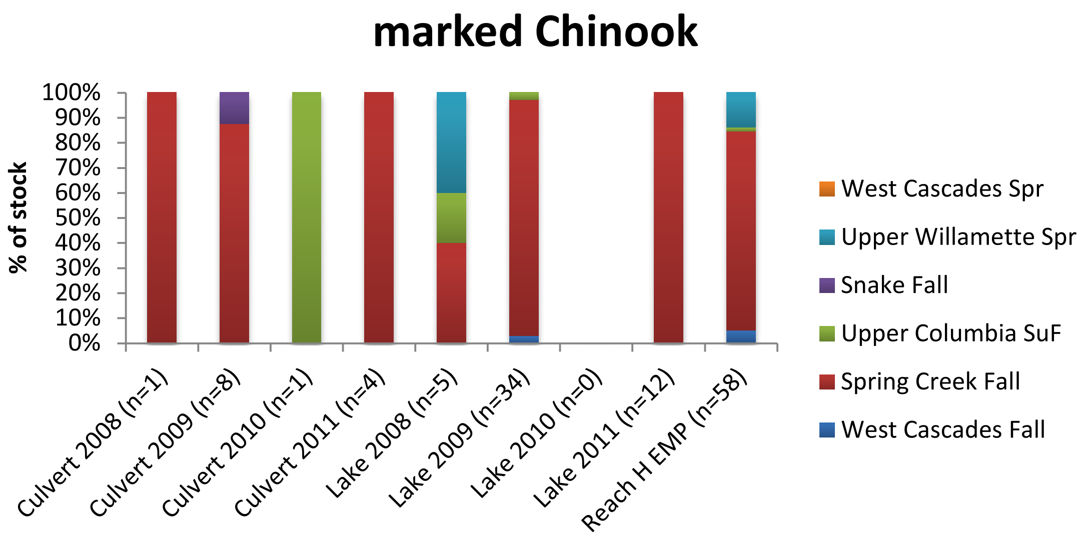
Genetic stock distribution before and after restoration. Genetic stock data are available only for Chinook salmon, which were found primarily at the Culvert and Lake sites, and only for 2008-2011; the data for 2012 are not yet available. The numbers of marked and unmarked Chinook salmon per site were often very small, which limits our ability to assess temporal changes in genetic stock diversity associated with restoration activities. However, at both the Lake and Culvert sites, the majority of marked Chinook salmon were Spring Creek Group fall Chinook, with smaller numbers of fish from other stocks (Figure 16). This is similar to the makeup of Chinook salmon from the EMP sites in Reach H.

Among the unmarked Chinook salmon, stock composition at both the Lake and Culvert was variable from year to year, and generally more different stocks were present at the Culvert than at the Lake (Figure 16). At the Culvert the most prominent genetic stock was generally Upper Columbia summer/fall Chinook. At the Lake, this stock was dominant in 2008 and 2009, but in 2010 and 2011, the two Lower Columbia fall Chinook stocks, West Cascades fall Chinook and Spring Creek Group fall Chinook, were most prominent. The stock composition at both sites was somewhat similar to that observed for unmarked Chinook salmon from the EMP sites in Reach H, except that West Cascades fall Chinook, absent from Reach H, were found at both the Culvert and Lake sites, and the Lake had a higher proportion of Spring Creek Group fall Chinook than the Reach H sites.

Salmon size class distribution before and after restoration. The size class and life stage distribution of unmarked Chinook salmon at the Mirror Lake Complex sites varied from site to site and year to year (Figure 17). Fry (< 60 mm) and fingerlings (60-130 mm) were the predominant size classes at all sites and years, although some yearlings (> 130 mm) were also observed.

At the Culvert, in most years the majority of unmarked juvenile Chinook collected were < 60 mm in length, but in 2010 a wider range of size classes was observed. At the Lake, the majority of fish were in the 50-60 mm size range from 2008 to 2010, but in 2011 and 2012 a higher proportion were 60-80 mm in length. The juvenile Chinook found at the Confluence in 2011 and 2012 were all 50-70 mm in length. In comparison to the Reach H EMP sites, the Culvert generally had a higher proportion of smaller size classes (< 50 mm), while the Lake and Confluence sites had higher proportions of larger size classes (50-90 mm). However, the size class distribution at the Culvert in 2010 was very similar to that of the Reach H EMP sites. Marked Chinook salmon at the Mirror Lake Complex sites were all within the fingerling size range (60 to 130 mm; Figure 17). At the Reach H EMP sites, the majority of marked juvenile Chinook collected were of comparable size (70-90 mm in length), but 19% of juvenile Chinook were yearlings > 130 mm in length.

Size class diversity (Figure 18) was generally higher for unmarked than marked Chinook salmon. Among unmarked Chinook salmon there were no clear temporal trends at either the Culvert or Lake, though values at the Lake tended to be higher, with the exception of 2010, when only 5 unmarked Chinook all of the same size class were collected. At the Culvert in 2010, and at the Lake in 2008 and 2011, diversity values were comparable to that of the Reach H EMP sites, but otherwise values were lower. Unmarked Chinook were found only in 2011 and 2012 at the Confluence, and the size class diversity was low. Among marked Chinook, size class diversity was highest at the Culvert in 2008, comparable to the values for the Reach H EMP sites, and then lower for the rest of the sampling period. At the Lake, on the other hand, values were relatively low in 2008 and 2009, but higher in 2011 and 2012, with the value in 2011 only slightly below the value for the Reach H EMP sites. (No marked Chinook were collected at the Lake in 2010). Marked Chinook were found at the Confluence only in 2011, and their size class diversity was relatively low.



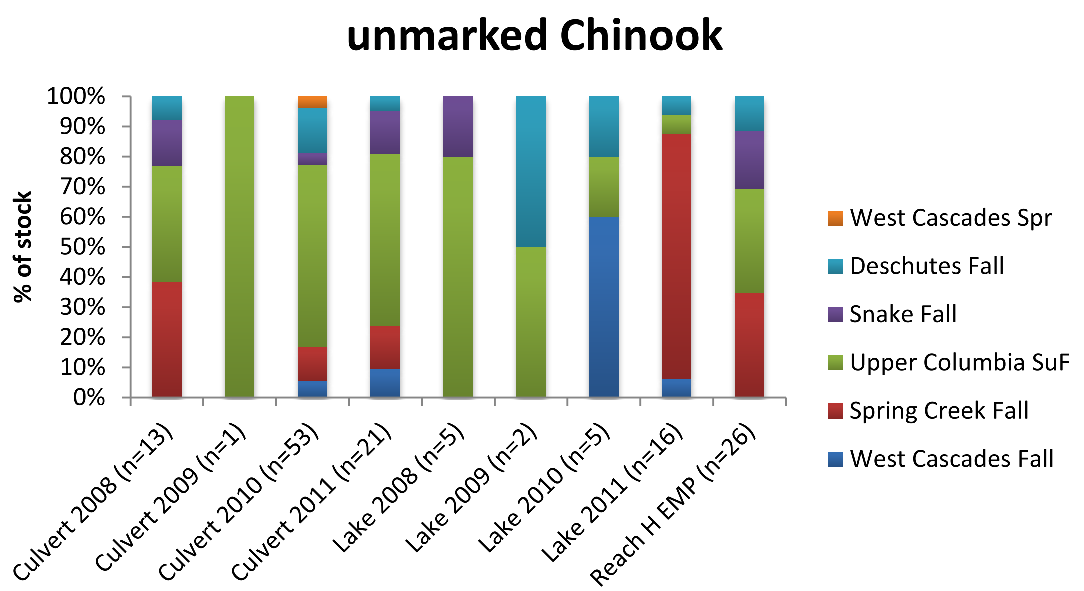
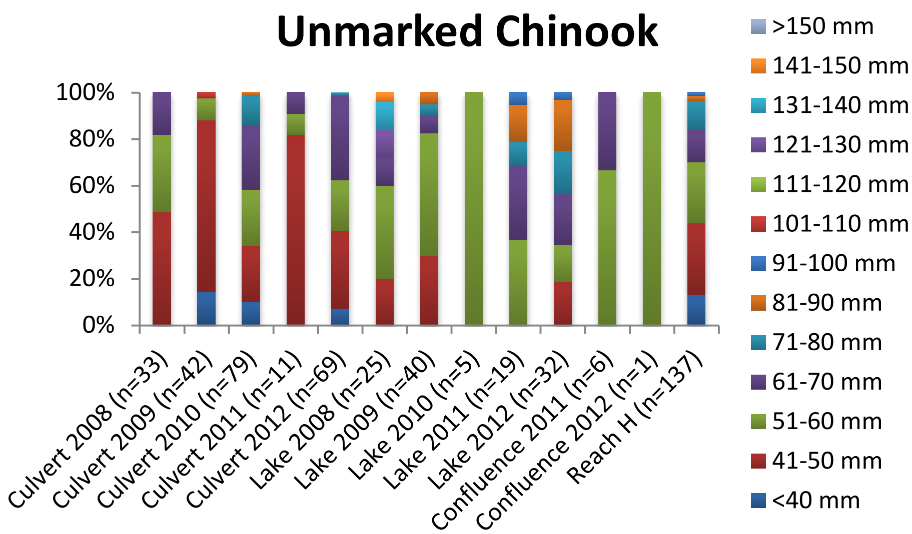


Figure 16. Genetic stocks of marked and unmarked juvenile Chinook salmon from Mirror Lake Complex sites, 2008-2011, as compared to juvenile Chinook salmon from Ecosystem Monitoring Project (EMP) sites in Reach H of the Lower Columbia River.



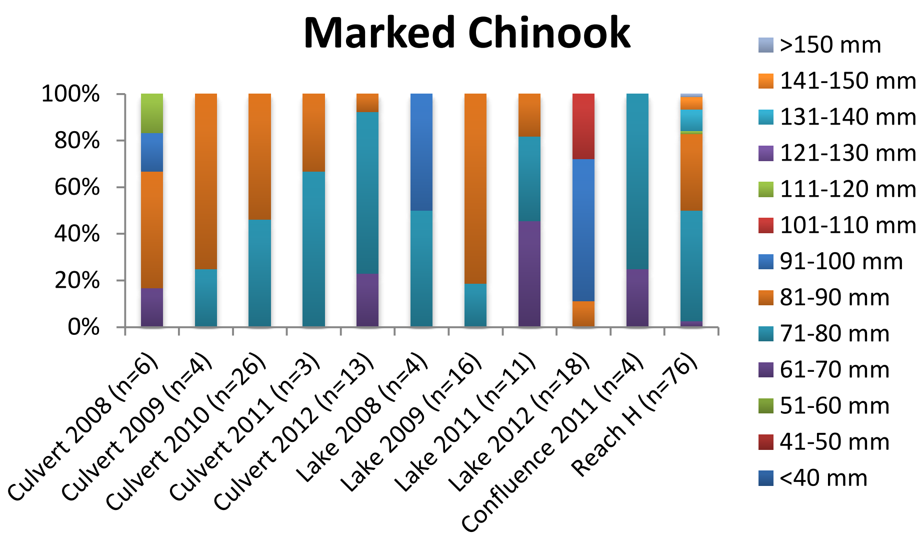
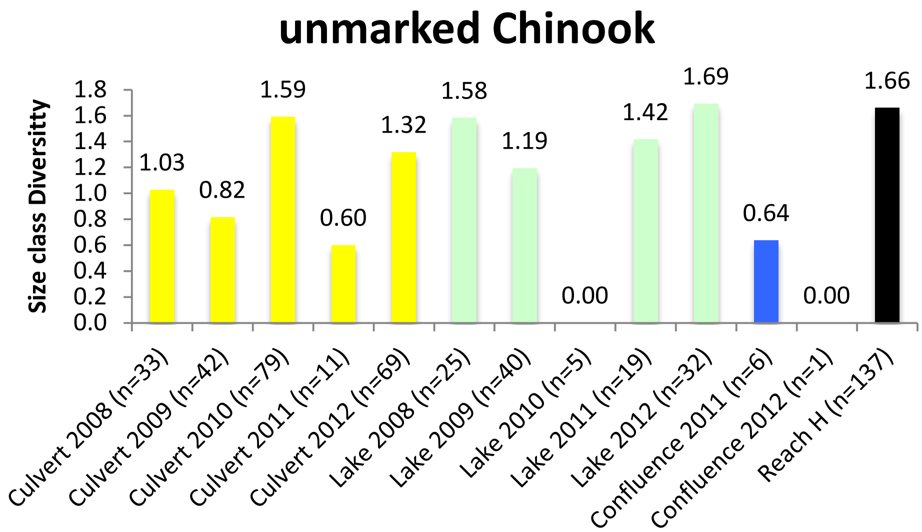


Figure 17. Size class distribution of unmarked and marked Chinook salmon from Mirror Lake Complex sites as compared to Reach H EMP sites.



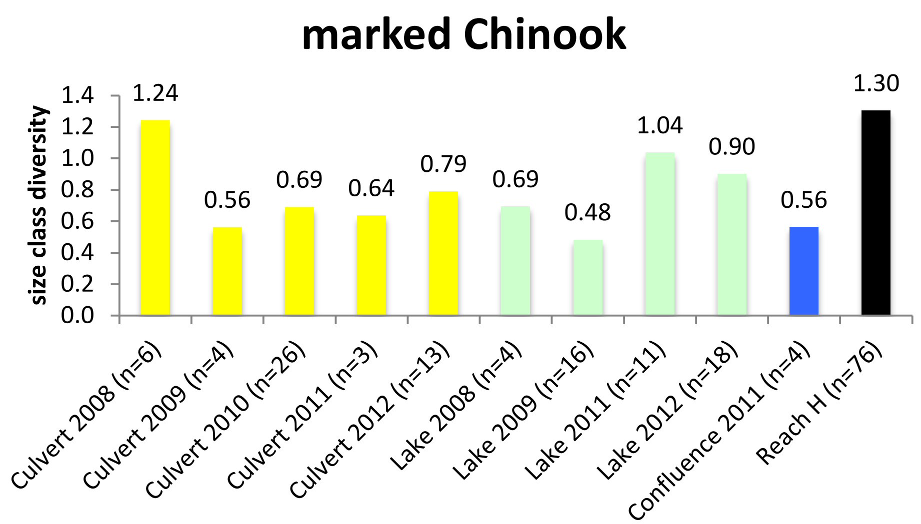
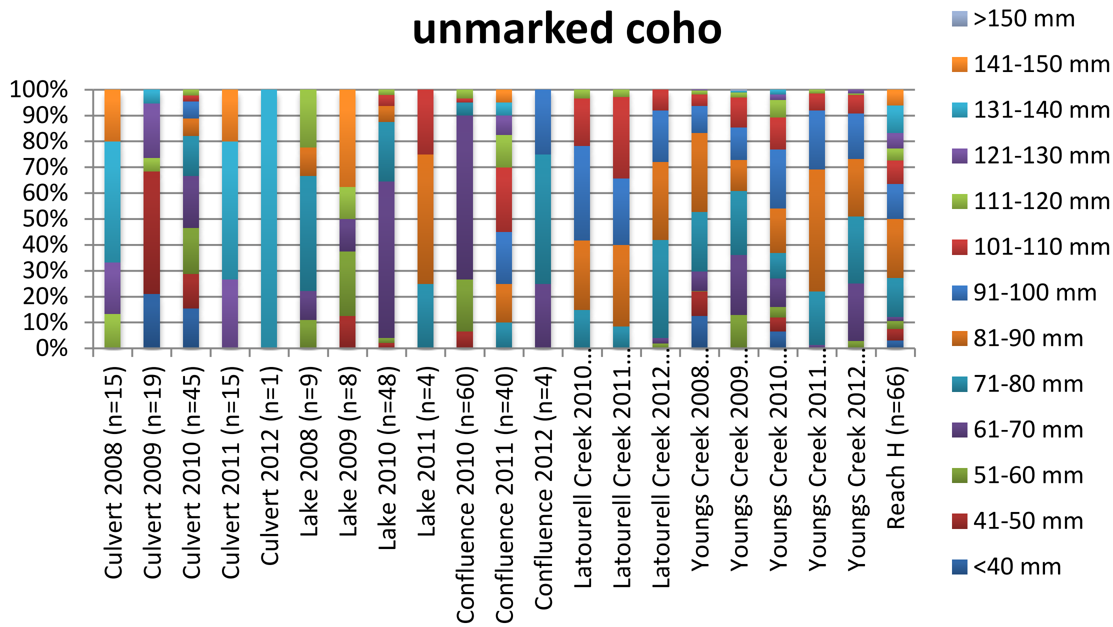


Figure 18. Size class diversity for unmarked and marked Chinook at the Mirror Lake Complex sites as compared to Reach H sites monitoring as part of the EMP.

For unmarked coho salmon, size class distribution also was quite variable from year to year and site to site (Figure 19). At the Culvert, in 2008, 2011, and 2012, the majority of unmarked coho were in larger size classes, from 130-150 mm in length, but in 2009 and 2010, a much higher proportion of fish were in smaller size classes (60-80 mm in length). Majority of unmarked coho at the Lake were between 60-90 mm with some in the size range of 100-120 mm. At the Confluence over 70-90% of the salmon collected were between 60-80 mm in length in 2010 and 2012, but in 2011, almost all salmon captured were > 80 mm in length, with the highest proportion in the 100-110 mm range. At Latourell Creek, size class distributions were very similar in 2010 and 2011, with the majority of fish between 80 and 110 mm in length, but the proportion of smaller fish (< 80 mm) was higher in 2012. At Youngs Creek, a wide range of size classes was represented in all sampling years, including both very small fish (40-60 mm) and fish above 120 mm in length, with about 70% of fish ≤ 90 mm in most years. Of the sites, Youngs Creek most closely approximated the Reach H EMP sites in size class distribution, although size class distributions at the Culvert in 2010 and the Confluence in 2011 were also similar to the Reach H EMP sites. Marked coho, found only at the Culvert site, were less variable in terms of size class distribution. With the exception of one smaller fish at the Culvert in caught in 2010, the majority of fish at both the Culvert and Reach H were in the 130-150 mm size range. At the Reach H EMP sites, the size range of marked coho was similar, although with a somewhat larger number of size classes.

Size class diversity of unmarked coho salmon varied from year to year, but showed no clear trends at any of the Mirror Lake Complex sites (Figure 20). At the Culvert, it was quite variable, ranging from 2.01 in 2010 to 1.01 in 2011. Values at the Lake were more consistent, but appeared to show a gradual decline from 1.49 in 2009 to 1.04 in 2011; the value slightly increased to 1.12 in 2012. At the Confluence, size class diversity was much higher in 2011 than 2010 of 2012, but at Latourell Creek it was in a similar range for all sampling years. At Youngs Creek size class diversity ranged from 1.32 to 2.19, with the highest value in 2010 and the lowest value in 2011. Compared to the Reach H EMP sites, size class diversity was generally lower at the Mirror Lake Complex sites, although values approaching those at Reach H were found at Youngs Creek in 2010, the Culvert in 2010, and the Confluence in 2011. Size class diversity for marked coho salmon (Figure 20) was generally much lower than for unmarked coho salmon. At the Culvert, the only site where marked coho were found, size class diversity declined steadily from 2008 to 2011, then increased in 2012, along with the number of marked coho collected at the site. Compared to the Reach H EMP sites, size class diversity of marked coho was similar in 2008, but lower in all other years.



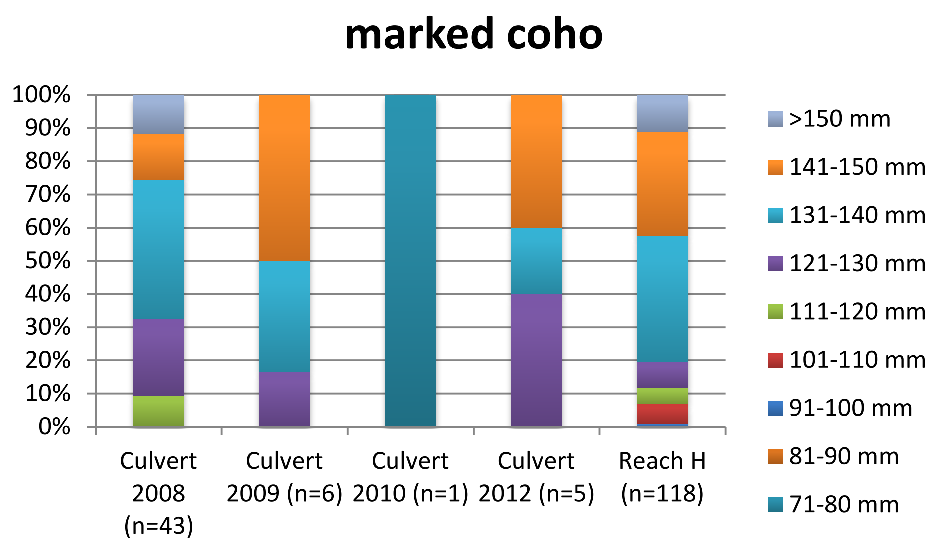
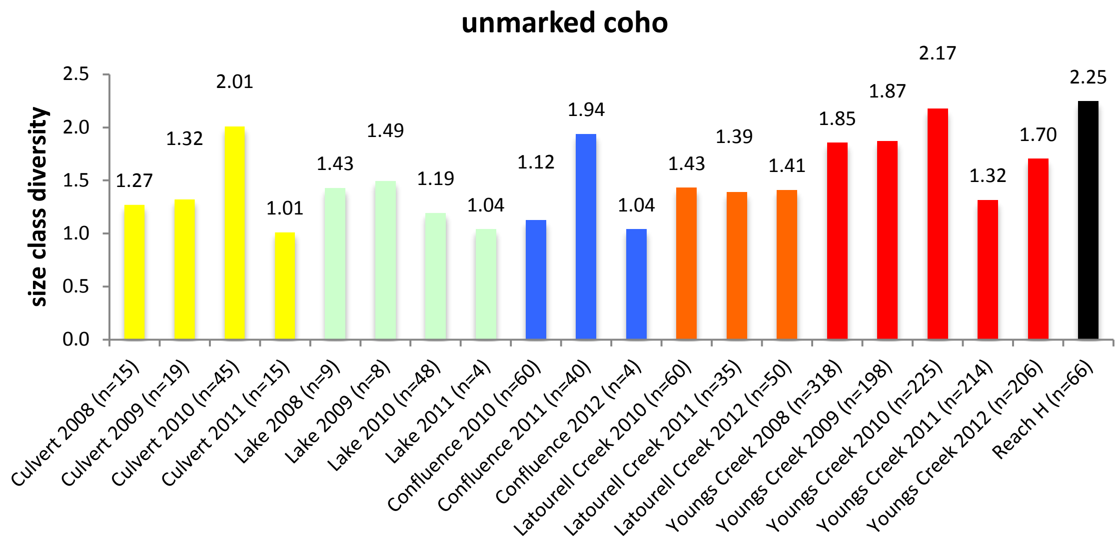


Figure 19. Size class distribution for unmarked and marked juvenile coho salmon from the Mirror Lake Complex sites as compared to Reach H sites monitored as part of the EMP.



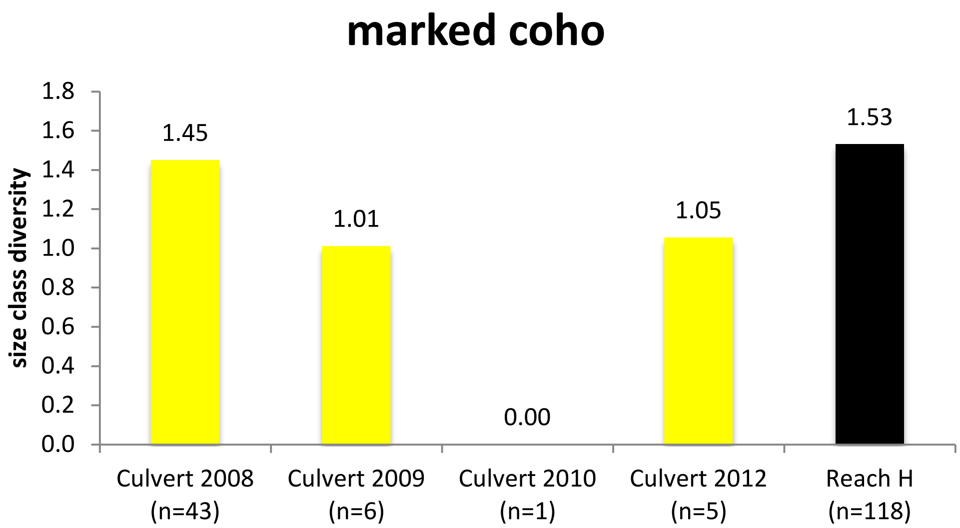


Figure 20. Size class diversity of unmarked and marked coho salmon from the Mirror Lake complex sites as compared to Reach H EMP sites.

3.2.4 Objective 4: Improve salmon health and condition in the Mirror Lake Complex

Salmon condition factor (K) before-and-after restoration. For unmarked Chinook salmon (Figure 21), there were significant interannual differences in K at both the Culvert and Lake sites ($p < 0.0001$), but no clear trends. At the Culvert, K in unmarked Chinook salmon was highest in 2008 and 2012, and somewhat lower in from 2009 to 2011 (Figure 21). In 2010, K was significantly lower than in either 2008 or 2012. Compared to the Reach H EMP sites, K was significantly lower at the Culvert in 2009, 2010, and 2011. At the Lake site, K was highest in 2008 and 2010, with somewhat lower values in 2009, 2011, and 2012. However, only the difference between 2008 and 2009 was statistically significant (Tukey's HSD, $p < 0.05$). K also tended to be lower than at Reach H EMP sites in 2009, 2011 and 2012, and higher in 2008 and 2010, but differences were not significant (Tukey's HSD, $p < 0.05$). At the Confluence, Chinook salmon K was significantly lower in 2012 than in 2011, and significantly lower than K at in salmon from the Reach H EMP sites as well (Tukey's HSD, $p < 0.05$). For marked Chinook salmon (Figure 22), there were no significant interannual differences or differences between Mirror Lake Complex sites and EMP sites for either the Culvert, Lake, or Confluence (Tukey's HSD $p>0.05$).

For unmarked coho salmon (Figure 23), there were no significant interannual differences in K within site or from the Reach H EMP sites at the Culvert, Lake, Confluence, or Latourell Creek, sites ($p > 0.05$). At Youngs Creek, K was highest in 2009 (significantly higher than values for 2008 or 2012) and lowest in 2012 (significantly lower than values for 2009 and 2010). However, none of the years had values significantly different from the average value of K for the Reach H EMP sites. Marked coho salmon (Figure 24) were found at the Culvert only. Values of K at this site were similar to those for marked coho salmon at the Reach H EMP sites, and no significant differences were found among sampling years ($p > 0.05$).

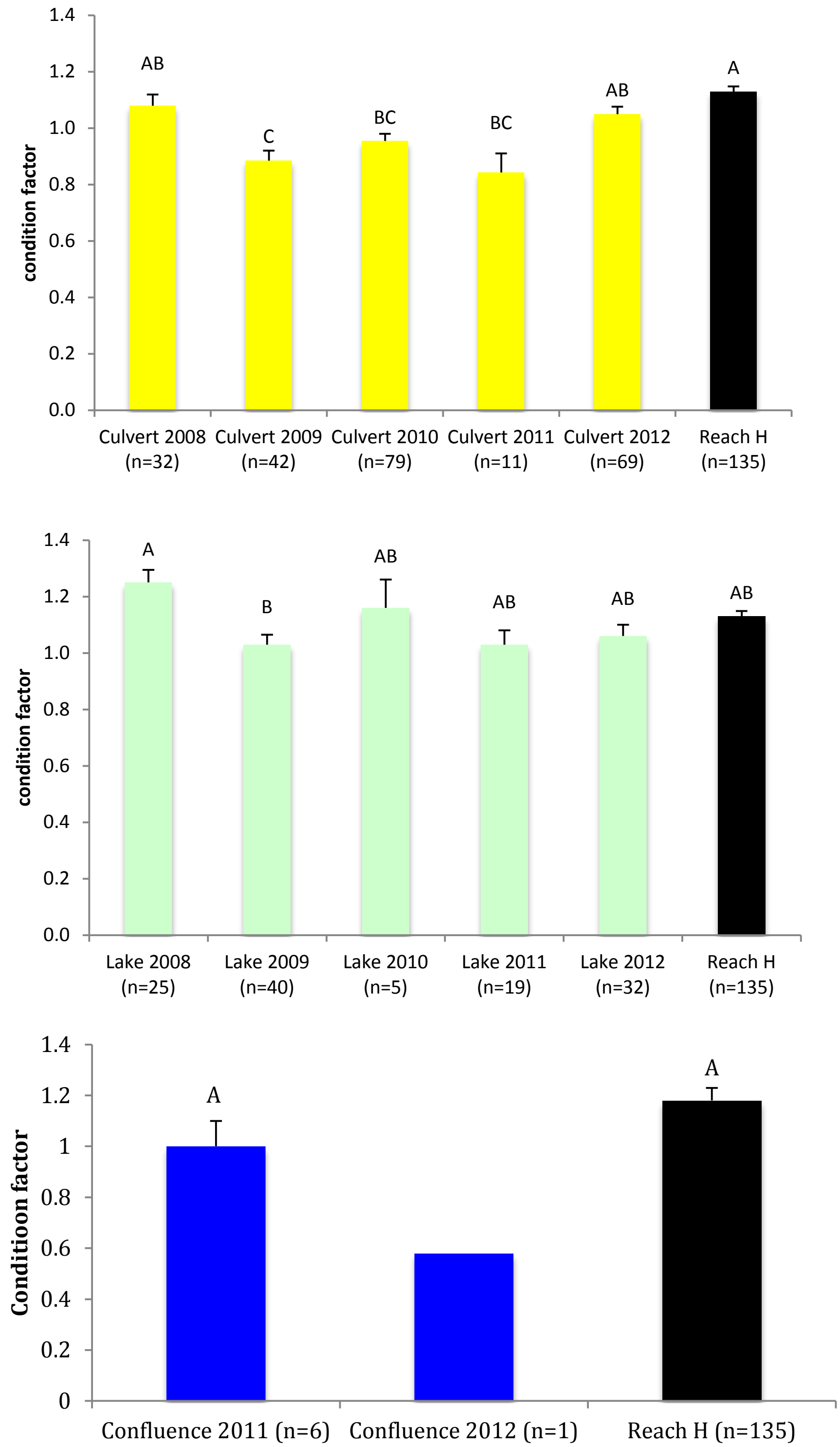


Figure 21. Condition factor (K) ± SE in unmarked juvenile Chinook salmon from the Culvert, Lake, and Confluence sites as compared to Reach H EMP. Values with different letter superscripts are significantly different (Tukey's HSD, $p < 0.05$).

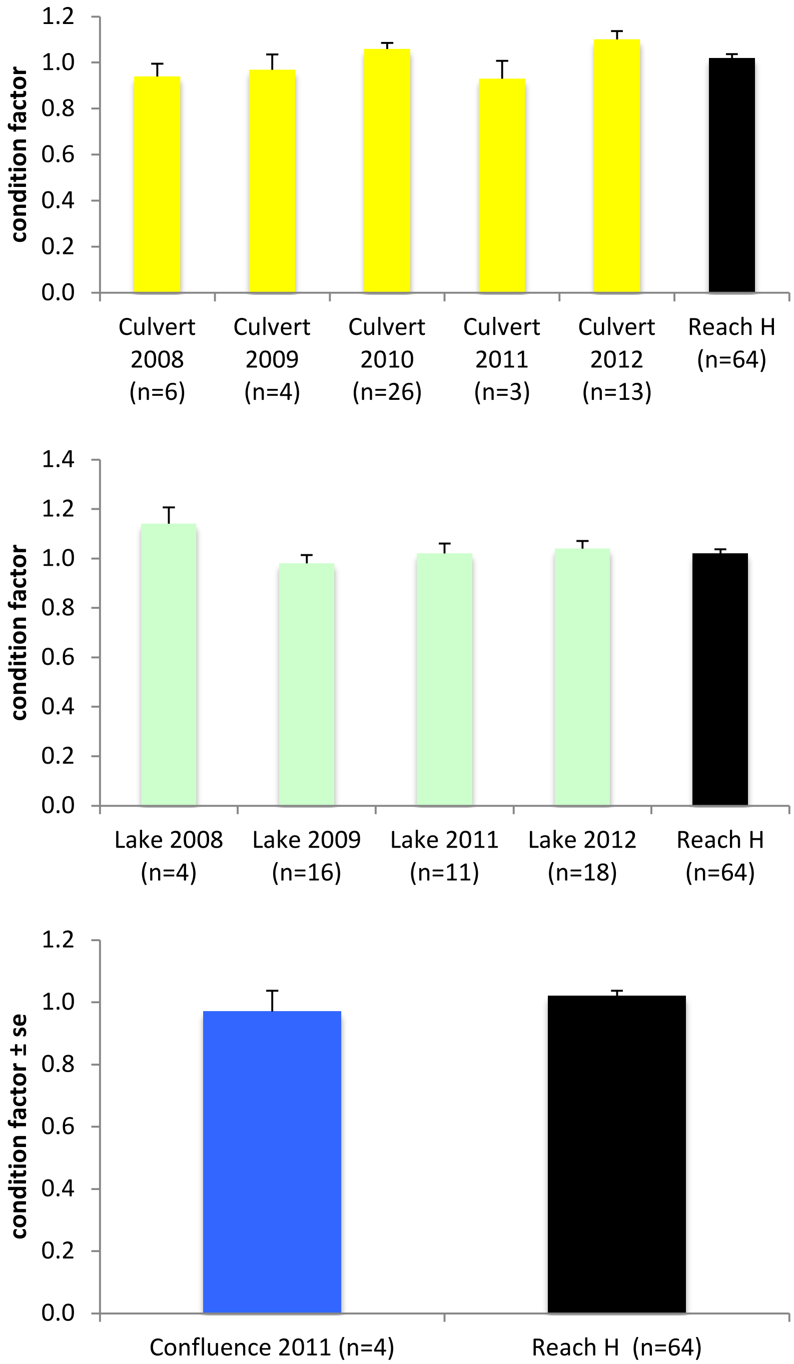


Figure 22. Condition factor (K) ± SE in marked juvenile Chinook salmon from the Culvert, Lake, and Confluence sites as compared to Reach H EMP. Values with different letter superscripts are significantly different (Tukey's HSD, $p < 0.05$).

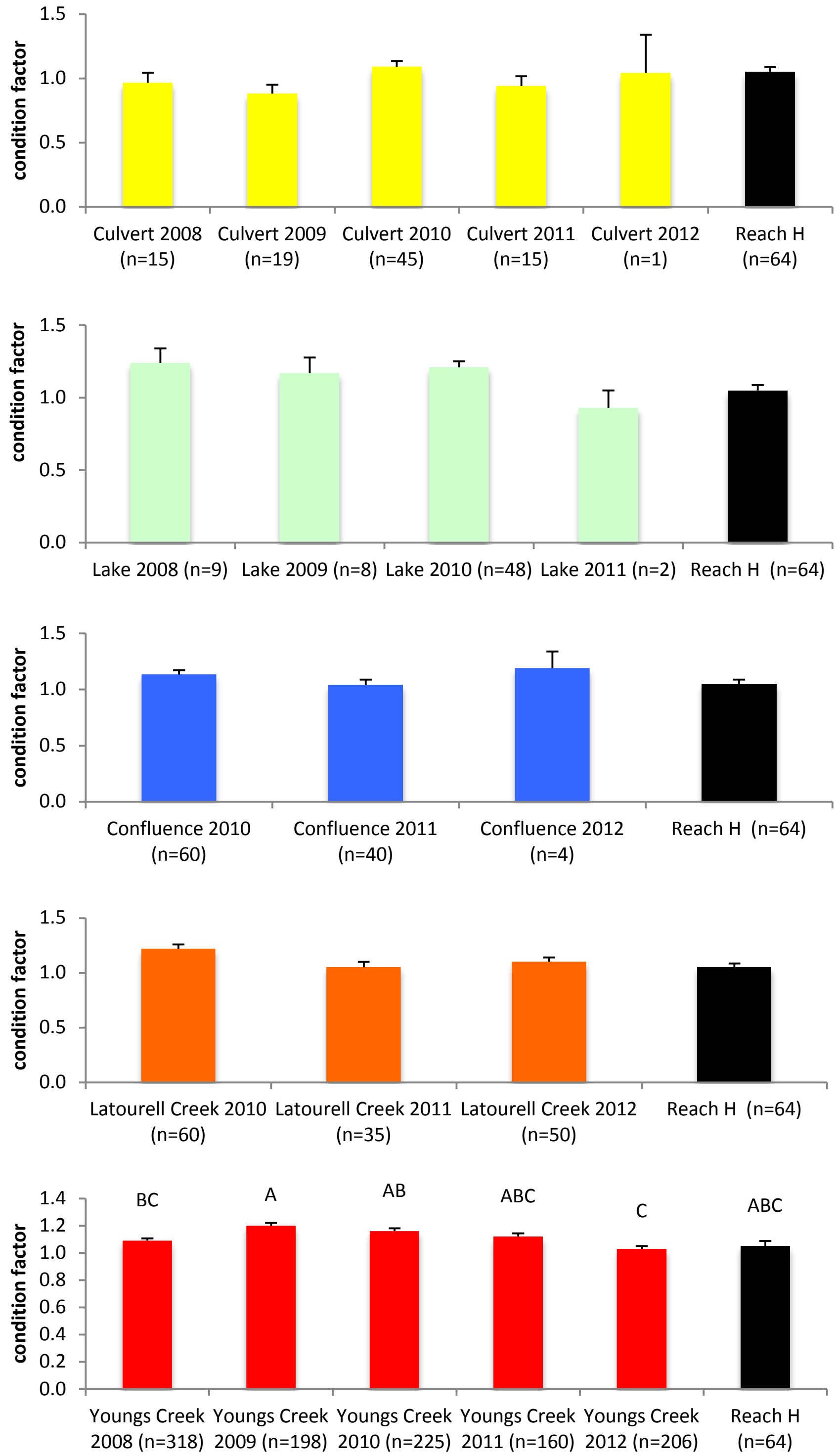


Figure 23. Condition factor (K) in unmarked juvenile coho salmon from the Culvert, Lake, Confluence, Latourell Creek, and Youngs Creek sites as compared to Reach H EMP. Values with different letter superscripts are significantly different (Tukey's HSD, $p < 0.05$).

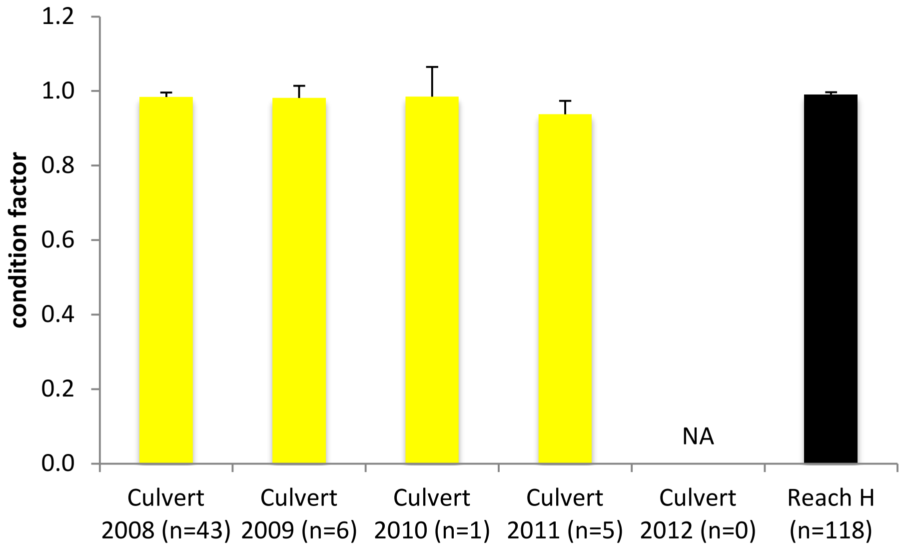


Figure 24. Condition factor (K) ±SE in marked juvenile coho salmon from the Culvert site as compared to Reach H EMP. No significant differences were observed for either site (Tukey's HSD, $p > 0.05$). Marked coho salmon were not found at the Lake, Confluence, Latourell Creek, or Youngs Creek.

Salmon growth rates before-and-after restoration. Growth rates were determined from otoliths collected from juvenile Chinook salmon from Mirror Lake Culvert and Lake sites between 2008 and 2010. Overall growth rates did not differ significantly between the two sites, although the mean growth rate was somewhat higher at the Lake than the Culvert (Figure 25). Also, growth rates in fish from the two Mirror Lake sites did not differ significantly from growth rates determined for juvenile Chinook salmon from other sites sampled in Reach H as part of the Ecosystem Monitoring Project (Figure 25).

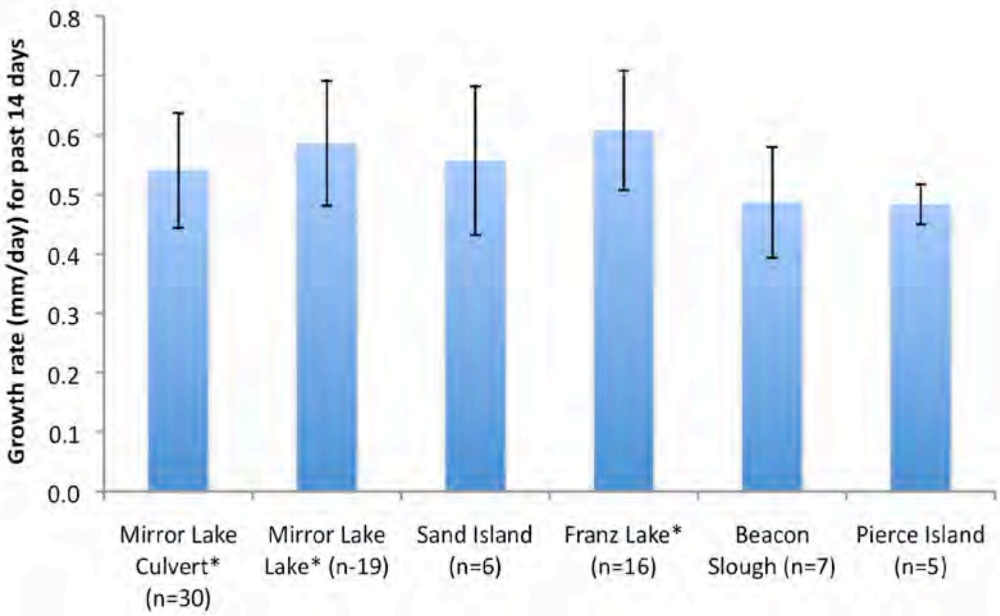


Figure 25: Mean daily somatic growth rate (mm/day) of Chinook salmon collected from several sites within Reach H. Whiskers represent standard deviation, and no significant differences were detected among sites.

There was some variation in growth rate from year to year at both the Culvert and Lake sites (Figure 26). Fish from the Culvert site had significantly slower growth in 2008 relative to 2009 and 2010, while fish from the Lake site showed significantly slower growth in 2008 relative to 2009 (only 2 fish collected in 2010 were analyzed and these were not included in the analysis; ANOVA = Culvert $F_{2,26}$=6.2; $p < 0.01$; Lake $F_{1,14}$=5.1; $p < 0.05$).

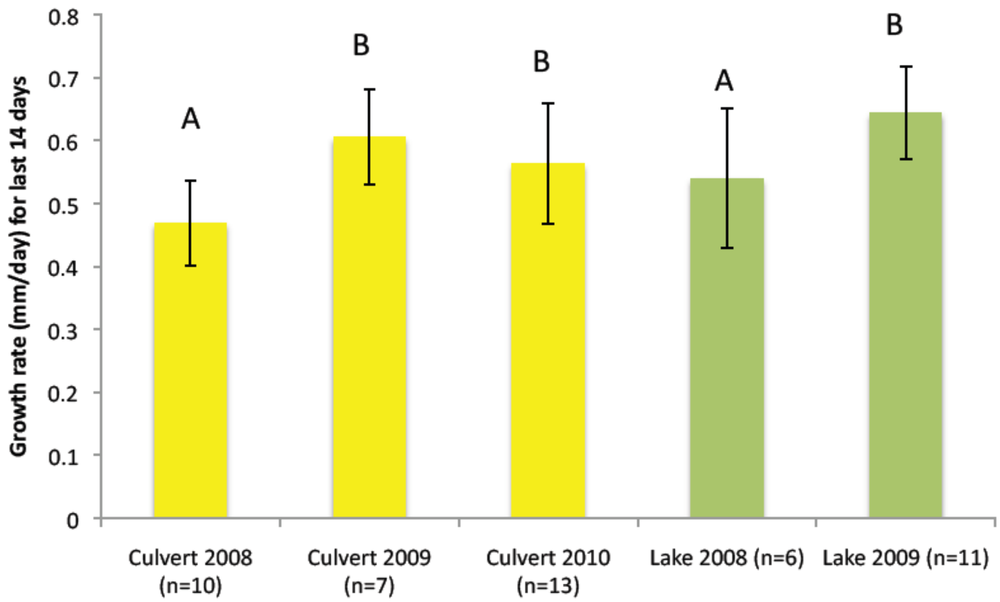
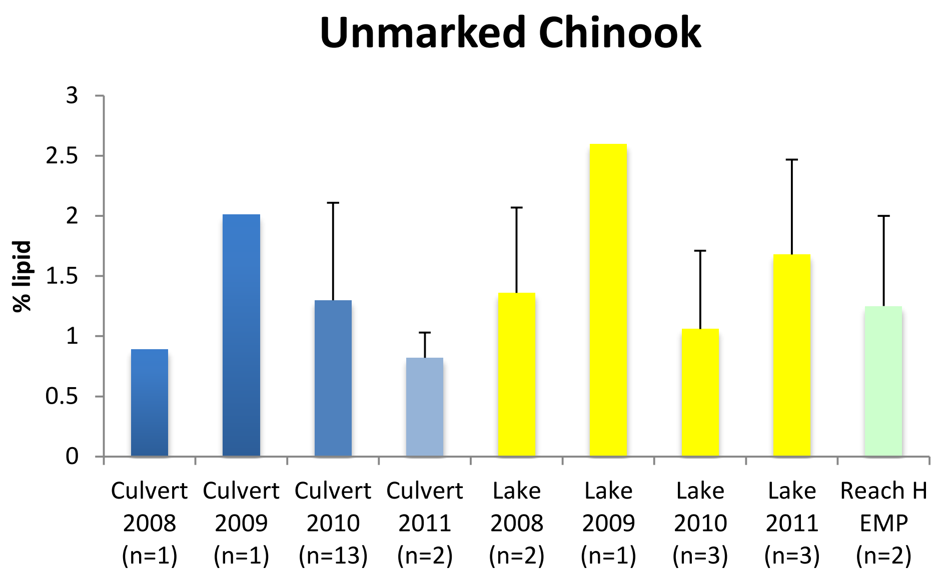


Figure 26. Daily growth rate as estimated from otolith examination at Mirror Lake Culvert and Lake sites from 2008 through 2010. Year within each site were compared separately. Values with different letter superscripts are significantly different (One-way ANOVA with Bonferroni post-hoc mean comparison test, $p < 0.05$). Whiskers represent standard deviation.

Salmon lipid content before-and-after restoration. Lipid content data are available for Chinook salmon only, at the Lake and Culvert sites. At this point we have lipid data for Chinook salmon samples collected from the Culvert and Lake sites from 2008 to 2011. Lipid content is calculated gravimetrically and also estimated using the Iatroscan method, which also provides data on lipid classes. The two values are highly correlated, although the gravimetric method generally provides slightly higher values than the Iatroscan method. Here we are reporting lipid content determined gravimetrically, as this information is available the largest number of samples. Because lipid content is significantly higher (1-way ANOVA, $p \leq 0.0001$) in marked fish than in unmarked fish (2.3% ± 0.7% vs. 1.3% ± 0.7%), these two groups of fish are considered separately. Mean lipid content in unmarked Chinook varied from 0.9 to 2.0% at the Culvert and 1.1 to 1.09% at the Lake, as compared to 0.88% for the EMP Reach H sites, while in marked Chinook mean lipid content varied from 0.8 to 2.04% at the Culvert and from 1.57 to 2.6% at the Lake, as compared to 1.3% for the EMP Reach H sites (Figure 27). However, year-to-year differences within the sites were not statistically significant for either marked or unmarked fish (Tukey-Kramer HSD multiple range test, $p \leq 0.05$) and showed no clear increasing or decreasing trends. Lipid levels in marked and unmarked Chinook salmon from the Culvert and Lake sites were also comparable to lipid values observed in marked and unmarked juvenile Chinook salmon sampled as in Reach H as part of the Ecosystem Monitoring Program (Sagar et al. 2013; Figure 27).



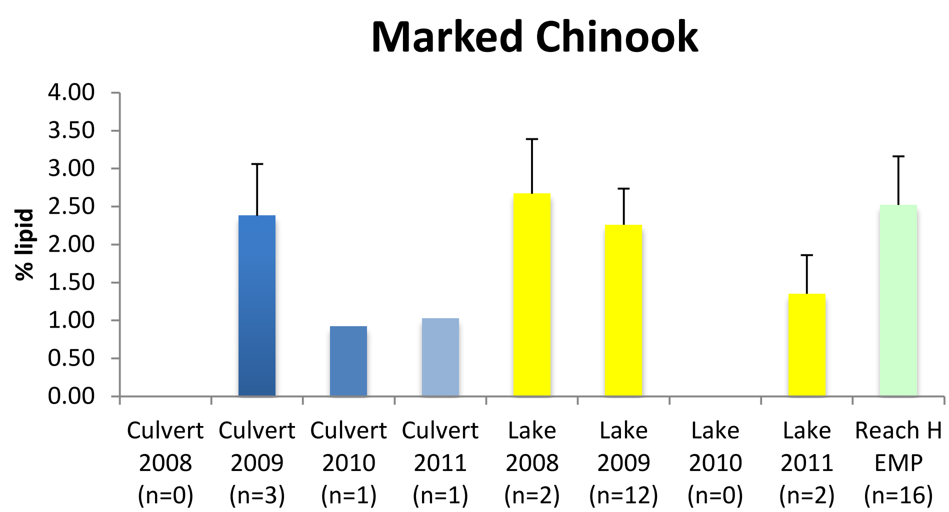


Figure 27. Lipid content in unmarked and marked juvenile Chinook salmon from the Mirror Lake Complex sites, as compared to juvenile Chinook salmon from Reach H sites monitored as part of the EMP. No significant differences were observed among years within the sites or with Reach H (Tukey-Kramer HSD multiple comparison test, $p \leq 0.05$) for either marked nor unmarked fish.

Stable Isotope Ratios in juveile Chinook salmon. In 2012, stable isotope ratios were determined on a subset of juvenile Chinook salmon collected in 2010 from the Lake and Culvert sites, as well as from several sites sampled as a part of the Ecosystem Monitoring Project (Table 9). While we do not have data evaluate changes in stable isotope ratios over the course of the restoration project, we can compare the two sites with the other sites sampled (in Reaches C-F). Additional data may be available for other Reach H sites and for the 2011 and 2012 samplings at Mirror Lake in the future.

Table 9. Stable isotope ratios in juvenile Chinook salmon collected in 2010 from the Mirror Lake Complex Lake and Culvert sites. Other sites sampled included Bradwood Slough, Jackson Island, Wallace Island in Reach C and Campbell Slough in Reach F (see Sagar et al. 2013). Two Mirror Lake sites are also included as representative of fish from Reach H.

	$\partial^{13}C$		$\partial^{15}N$	
	Unmarked	marked	Unmarked	marked
H-Mirror Lake Culvert	-22.8 ± 2.6 (n=11)	-19.8 ± 0.11 (n=2)	9.4 ± 1.5[b] (n=11)	12.4 ± 0.2 (n=2)
H-Mirror Lake Lake	-20.2 ± 0.5 (n=2)		8.6 ± 1.0[b] (n=2)	
Other sites	-22.6 ± 1.6 (n=40)	-20.8 ± 1.9 (n=33)	11.2 ± 0.7[a] (n=40)	12.2 ± 1.2 (n=33)
	$p = 0.1977$	$p = 0.4586$	$p < 0.0001$	$p = 0.8902$

No significant differences were found in $\partial^{13}C$ ratios between the Lake and Culvert sites or with other sites sampled for either unmarked or marked fish. For the marked fish, $\partial^{15}N$ ratios were also similar at the Culvert sites and other sites (no marked fish were sampled from the Lake site). However, $\partial^{15}N$ ratios were significantly lower in unmarked juvenile Chinook from both Mirror Lake sites than in juvenile Chinook that were sampled from the other sites.

Contaminant Concentrations in juvenile Chinook salmon from the Mirror Lake Complex sites. At this point, body chemistry data for PCBs, PBDEs, and DDTs are available for juvenile Chinook salmon collected from the Culvert and Lake sites in 2008, 2009, and 2010 (Figure 26). At both sites, DDTs were present at the highest concentrations, although low levels of PBDEs and PCBs were also detected. For all three classes of contaminants, concentrations tended to be higher in unmarked than in marked fish (2400 vs. 1500 ng/g lipid for DDTs; 550 vs. 1000 ng/s lipid for PCBs; and 130 vs. 440 ng/g lipid for PBDEs for samples from the Culvert and Lake combined), but the difference was statistically significant only for PBDEs ($p = 0.0148$). Also, concentrations of all three classes of contaminants tended to be higher in salmon from the Culvert and Lake sites than in salmon from the Reach H EMP sites, with differences particularly apparent among marked fish (Figure 28).

As for temporal trends, no statistically significant differences were detected among sampling years for either marked ($0.2459 < p < 0.8788$) or unmarked fish ($0.0506 < p < 0.2989$) from either the Culvert or Lake sites. at the Culvert, concentrations of PCBs and PBDEs were higher in the samples collected in 2010 than in 2008 and 2009, whereas DDTs were at intermediate levels compared to 2008 and 2009. At the Lake site, DDT and PCB concentrations were slightly lower in 2010 than in 2008 and 2009, while PBDEs were slightly higher. However, no statistically significant differences were detected among sampling years for either marked ($0.2459 < p < 0.8788$) or unmarked fish ($0.0506 < p < 0.2989$) from either the Culvert or Lake sites. Also, there were no statistically significant differences in concentrations of PBDEs, PCBs, or DDTs in unmarked juvenile Chinook salmon from the Culver and Lake sites and levels of these contaminants in unmarked juvenile Chinook salmon collected from Reach H as part of the Ecosystem Monitoring Program ($0.503 < p < 0.7125$). In marked fish, on the other hand, concentrations of DDTs were significantly higher in salmon collected from the Lake than in samples from collected from Reach H as part of the Ecosystem Monitoring Program ($p = 0.0009$). Also, concentrations of both PCBs and PBDEs were significantly higher in juvenile Chinook salmon from the Culvert site than in juvenile Chinook collected from the Reach H as part of the Ecosystem Monitoring Program ($0.0044 < p < 0.0333$).

Mean concentrations of PCBs, DDTs, and PBDEs in all years at both sites were below estimated toxic effect concentrations for juvenile salmon [2400 ng/g lipid for PCBs (Meador et al. (2002); 5000-6000 ng/g lipid for DDTs (Beckvar et al. 2005 as modified for lipid content by Johnson et al. 2007); and 940 ng/g lipid for PBDEs (Arkoosh et al. 2010)]. However, two of the 22 samples collected at the Culvert site in 2010 were above the effects threshold for all three classes of contaminants, and an additional two samples were above the effect threshold for PBDEs only. All four of these samples were from unmarked fish; two were West Cascades fall Chinook salmon and two were Spring Creek Group fall Chinook salmon.

We have also attempted to monitor exposure to polycyclic aromatic hydrocarbons (PAHs) in juvenile Chinook salmon from these sites by measuring concentrations of PAH metabolites in salmon bile, but have had difficulty obtaining adequate bile from these small fish to examine intersite differences or interannual trends in exposure levels. The limited data on concentrations of benzo[a]prene (BaP) and phenanthrene (PHN) and naphthalene (NPH) metabolites (measured as fluorescent aromatic compounds detected at BaP, NPH, and PHN wavelengths (FACs-BaP,FACs-NPH, and FACs-PHN) in bile of Chinook salmon from the Lake and Confluence sites are shown in Figure 29. Bile metabolites in juvenile Chinook salmon from the Reach H Ecosystem Monitoring sites are included for comparison. The bile sample from the Mirror Lake area includes samples from the Lake and Culvert sites combined because the volume of bile obtained was too small for separate samples to be run from each sites. Because the number of composite samples collected each year was so small, no statistically significant differences were observed among concentrations. However, levels of FACs-PHN, FACs-BaP, and FACs-NPH tended to be slightly lower at the Mirror Lake sites than in salmon collected from Reach H as part of the Ecosystem Monitoring Program.

As an alternate measure of PAH exposure, in 2010 and 2011 we measured PAHs in bodies of juvenile Chinook salmon from the Mirror Lake sites (Figure 30). Low molecular weight PAHs accounted for the majority of PAHs measured in the samples (95% at the Culvert site and 99% at the Lake site in 2010, and 99% at the Culvert and 96% at the Lake site in 2011). Total PAH concentrations were not significantly different (ANOVA, $p \leq 0.05$) in samples from the Lake and Culvert sites, or between the Lake and Culvert sites and other EMP sites where PAHs in salmon bodies were measured. Nor were concentrations change significantly at the sites between 2010 and 2011, although concentrations were slightly lower in 2011.

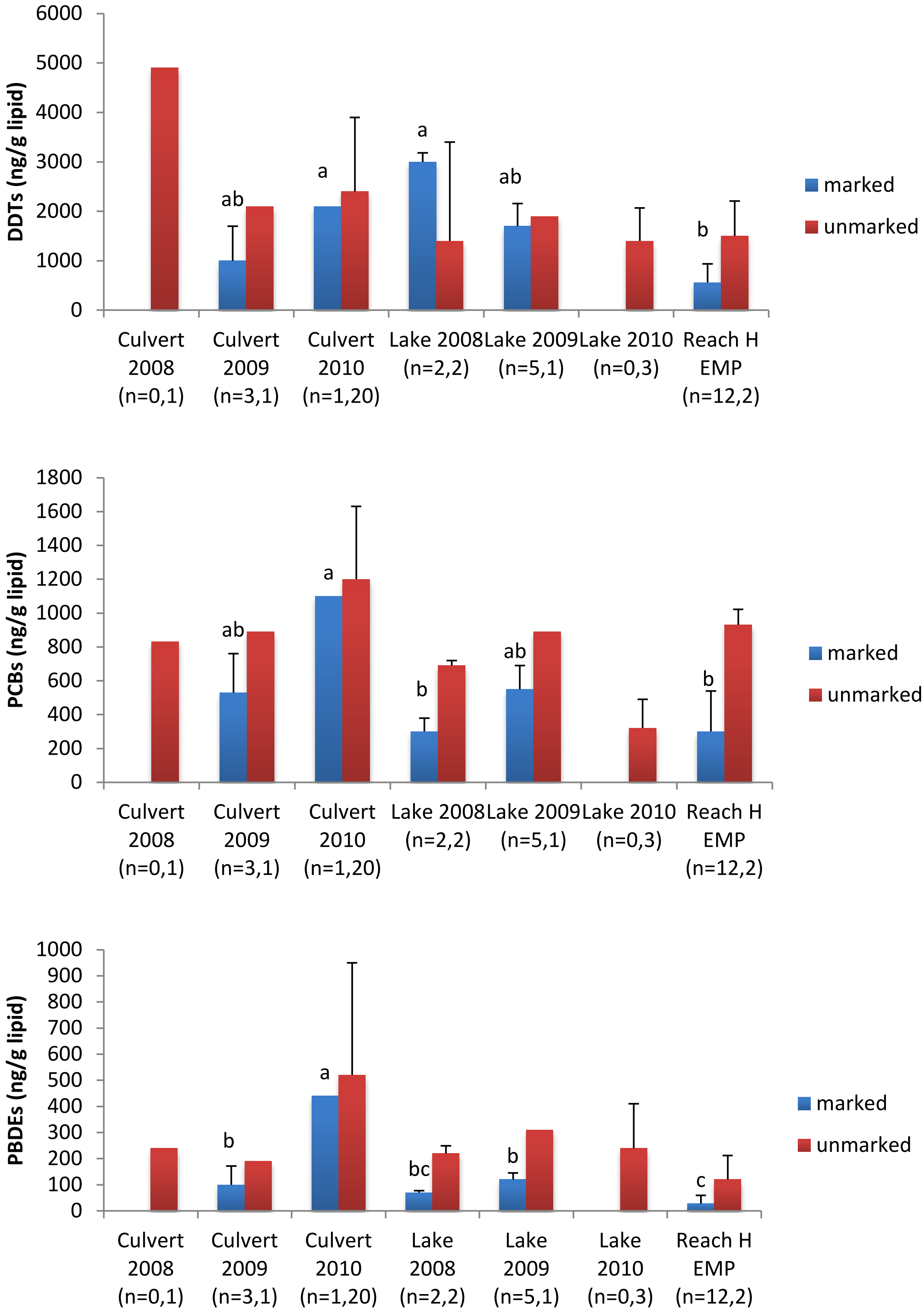


Figure 28. Concentrations of DDTs, PCBs, and PBDEs in juvenile Chinook salmon sampled from the Culvert and Lake sites at the Mirror Lake complex from 2008-2010.

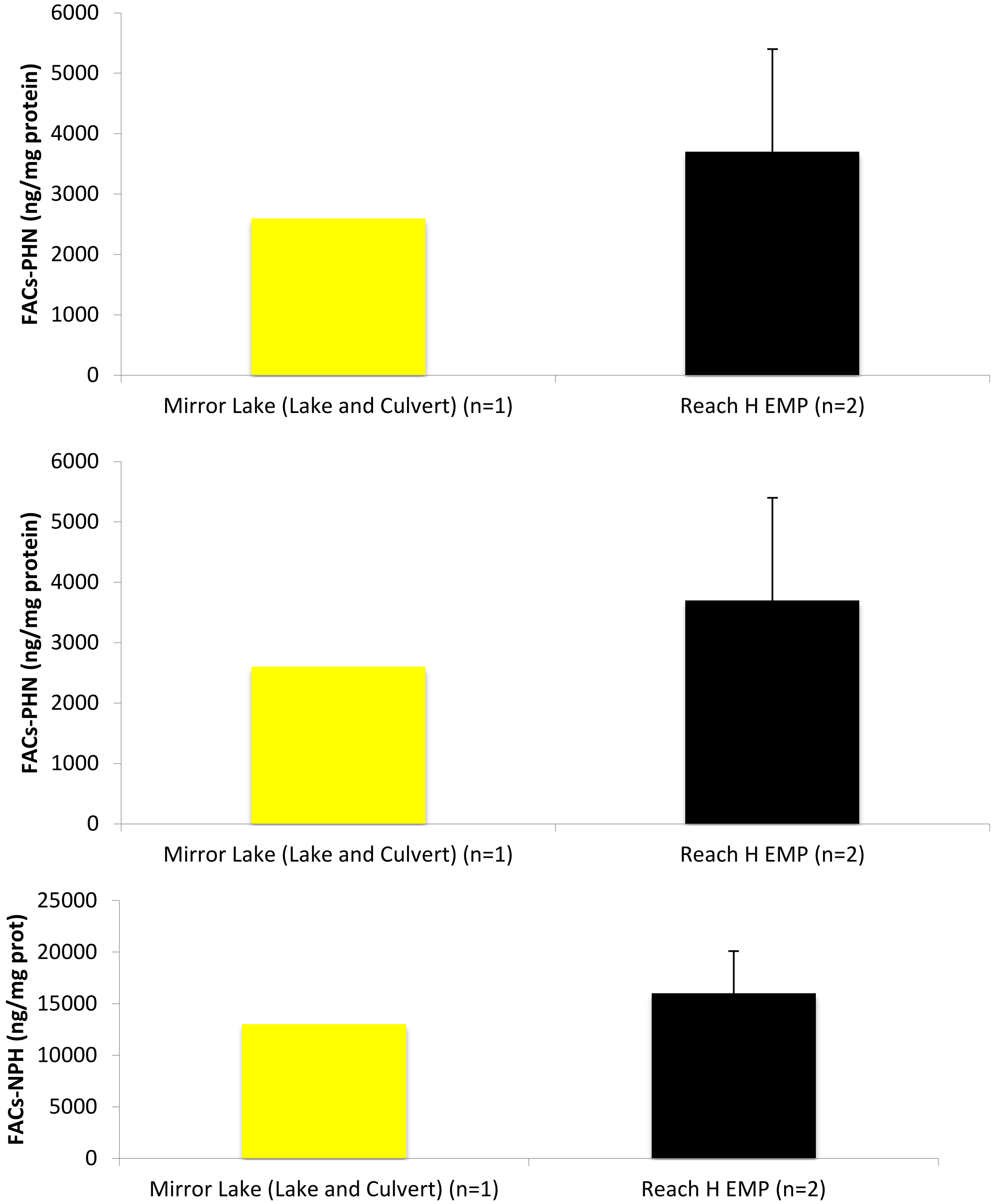


Figure 29. Concentrations of fluorescent aromatic compounds (FACs) measured at phenanthrene, benzo[a]pyrene, and naphthalene wavelengths (FACs-PHN, FACs-BaP, and FACs-NPH) in bile of juvenile Chinook salmon from Franz Lake and Campbell Slough. No significant differences were found in bile metabolite levels among sites.

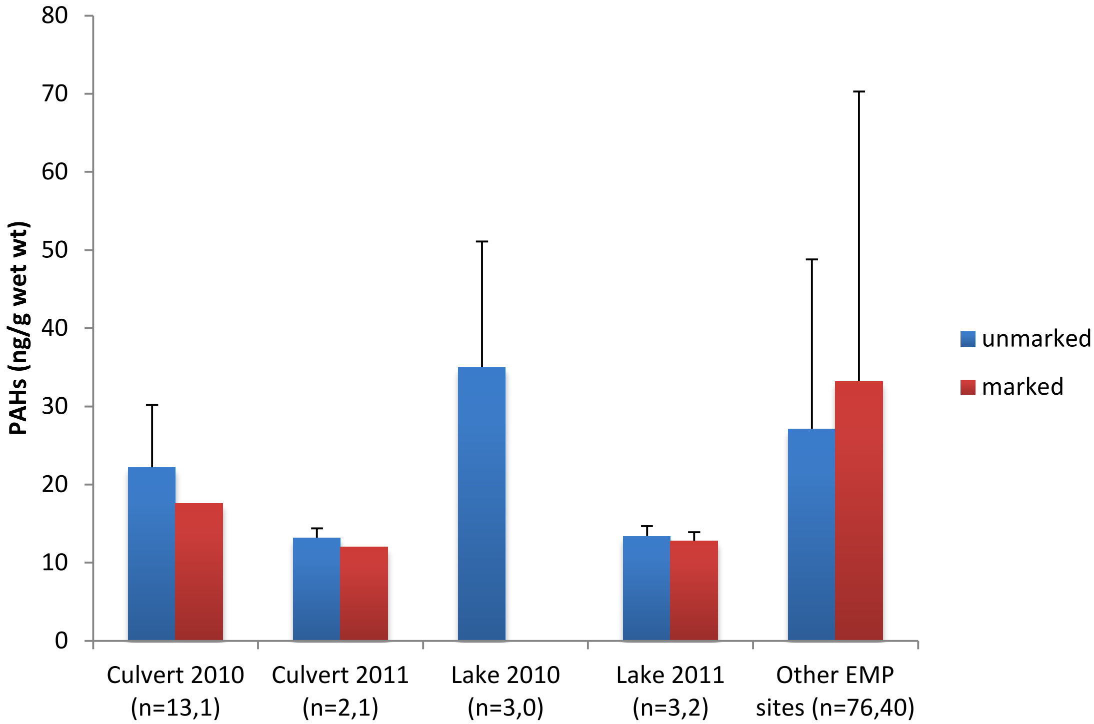


Figure 30. Concentrations of polycyclic aromatic hydrocarbons (PAHs) in bodies of juvenile Chinook salmon sampled from the Culvert and Lake sites at the Mirror Lake Complex in 2010 and 2011. In 2008 and 2009, PAH analyses were not performed on Chinook salmon bodies. Total PAH concentrations in salmon from the Culvert and Lake sites were not statistically different from each other, or from the mean concentration for Chinook salmon from other Ecosystem Monitoring Program (EMP sites) where body PAHs were measured.

3.2.5 Objective 5: Improve water quality in the Mirror Lake Complex Water Temperature and other Physical Factors

At the Culvert, in all sampling years, water temperature increased steadily throughout the sampling season from 9-17ºC in April and May to over 20ºC in July and August (Figure 31). In 2007, 2008, 2010, 2011, and 2012, the temperature ranges at the site were similar, although it was sampled only from May through July in 2007. In 2009, higher maximum temperatures were reached, with temperatures of 25-28ºC in July and August. Extended fall sampling in 2011 and 2012 showed that water temperatures declined to below 15ºC by October. At the Lake, the water temperature range was similar to that observed at Culvert, typically increasing from about 10ºC in April to 20-25ºC by July and August (Figure 31). Summer temperatures were somewhat higher in 2009 than in the other sampling years, with a maximum temperature in 2009 of 31ºC in August of 2009. Our more extended fall and winter sampling in 2011 and 2012 showed that water temperature at the Lake declines steadily beginning in August or September to less than 5ºC by November.

At Youngs Creek, Latourell Creek, and the Confluence, summer temperatures were much lower than at either the Lake or Culvert, remaining below 20ºC in all sampling years (Figure 31and Figure 32). At Youngs Creek, in 2008 and from 2010 through 2012, temperatures ranged from 7.8-13.7 ºC in April and May to 13-14.5ºC from June through September (Figure 31). As at the other sites, water temperatures at Youngs Creek in 2009 were unusually high, with a maximum temperature of 19.6ºC in August. Temperature ranges at Latourell Creek and the Confluence were very similar to those at Youngs Creek, with maximum summer temperatures between 14.7 and 16.8ºC (Figure 32). Aside from the unusually high water temperatures in 2009, no trends in water temperature were observed at any of the sampling sites.

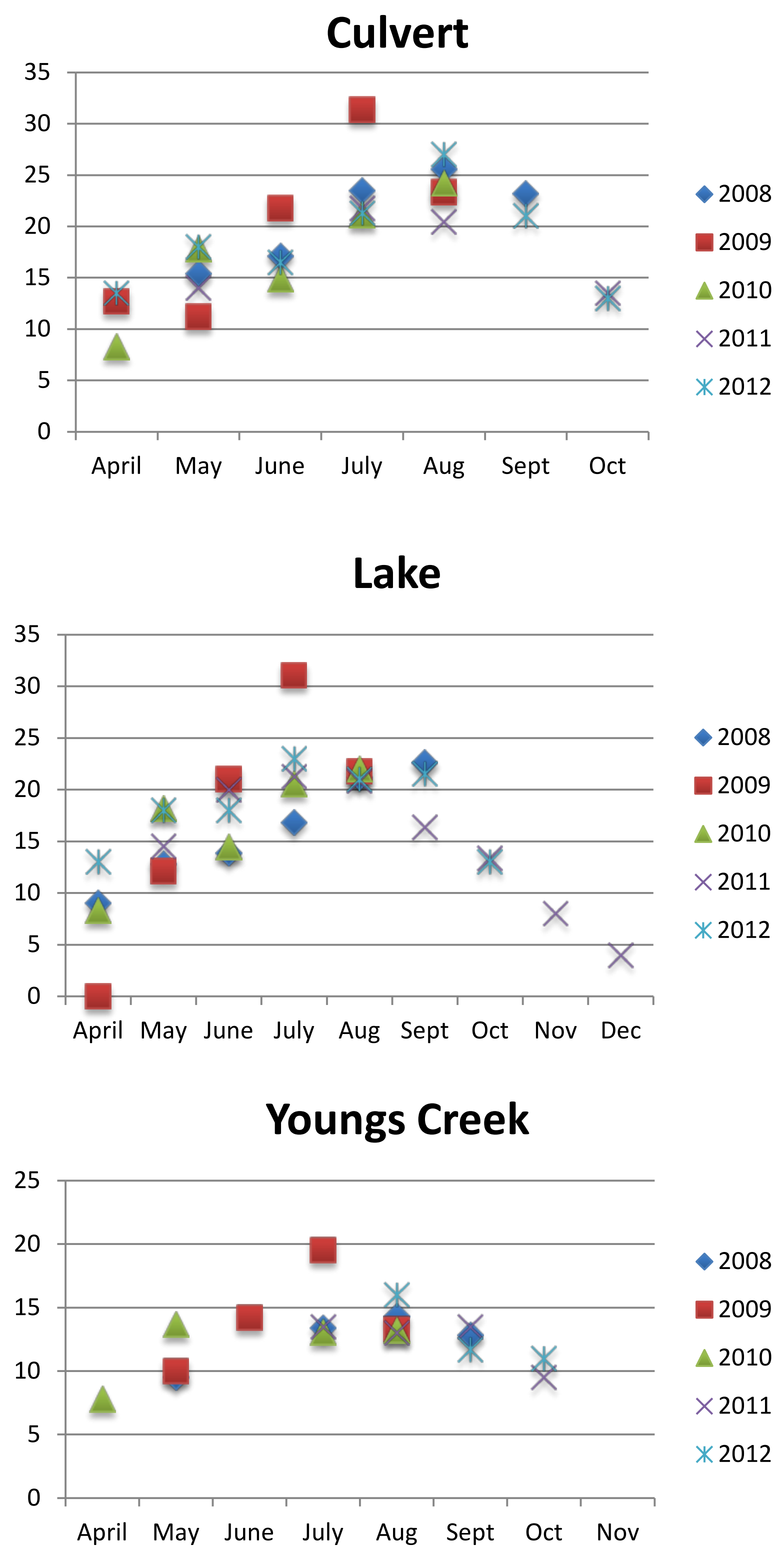
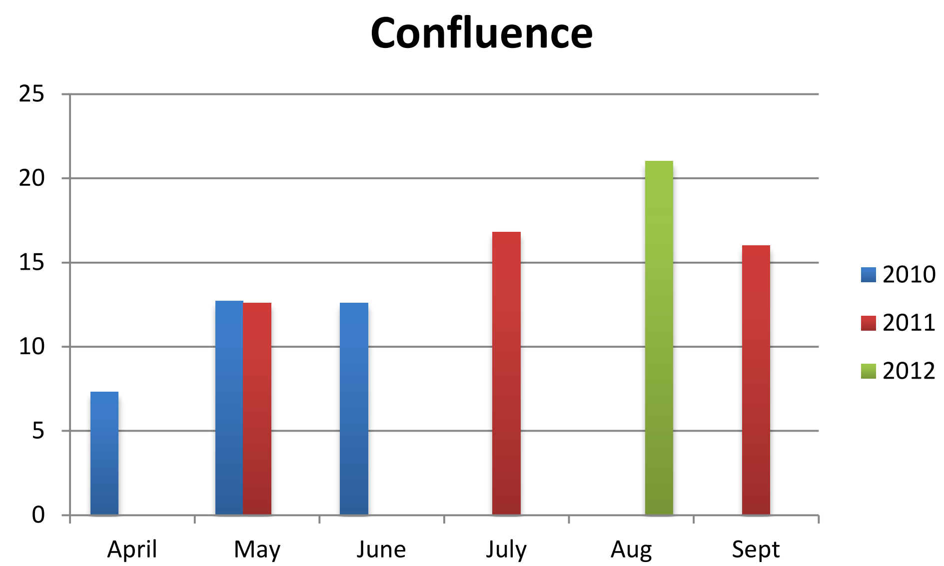


Figure 31. Seasonal water temperatures at Culvert, Lake, and Youngs Creek Mirror Lake Complex sites



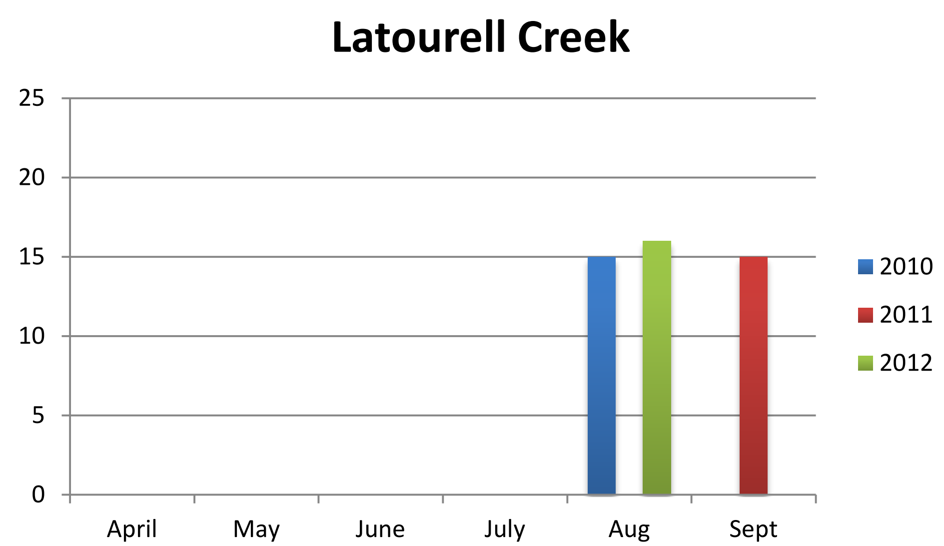


Figure 32. Water temperatures at Confluence and Latourell Creek Mirror Lake Complex sites

3.2.6 Objective 6: Improve wetland habitat quantity and quality, including availability of salmon prey.

Salmonid Prey Availability Surveys and Diet Analyses for Juvenile Chinook Salmon. Prey availability surveys were conducted at the Mirror Lake sites by sampling with benthic cores, terrestrial sweep nets, and Neuston tows to investigate the availability of salmonid prey species in benthic, terrestrial, and water column environments. At the Culvert and Lake, prey collections coincided with collections of juvenile Chinook salmon, so that when sufficient numbers of fish were collected the taxonomic composition and abundance of consumed prey could be compared with available prey. The number of tow samples and paired diet samples collected between 2008 and 2012 are shown in Table 10.

Table 10. Number of juvenile Chinook salmon diet samples and Neuston tow samples collected from Mirror Lake sites from 2008-2012. EV = emergent vegetation tows; OW = open water tows. The 2008-2011 tow samples have been processed and results are reported here; the 2011 and 2012 diet samples and 2012 tow samples are being processed currently.

		2008		2009	2010				2011		2012		
		May	June	May	April	May	June	July	May	June	April	May	June
Chinook salmon diets													
Lake		13	5	10	-	-	-	-	10	3	6	32	5
Culvert		14	-	7	18	10	10	-	10	-	26	20	12
Tow samples													
Lake	EV	3	3	2	-	2	-	-	2	-	2	2	1
	OW	2	3	2	-	2	-	-	2	3	2	2	2
Culvert	EV	2	-	2	2	2	2	2	2	-	1	2	-
	OW	3	1	2	2	2	2	2	2	-	2	2	2

Prey Availability. Results of the benthic core and terrestrial sweep sampling (conducted in 2008) are shown in Table 11and Table 12. Dominant macroinvertebrate species at all three sites included oligochate worms and Dipteran larvae and pupae. Dipterans (primarily adults) were also prominent in the terrestrial sweep samples collected in 2008, especially at the Lake, where they made up 73-78% of macroinertebrates collected. At Youngs Creek, Hemipterans dominated the terrestrial sweep samples, accounting for 68% of individuals collected

Table 11. Mean counts of macroinvertebrates from sediment cores collected in 2008. Note these are mean counts based on 4-5 samples per event. % indicates the proportion of each mean that is composed of that taxon; values are bolded if that taxon made up 10% or more of the mean for that event.

	Culvert			**Lake**			**Youngs Creek**					
	July			**July**			**May**			**July**		
Taxa	Mean	SD	%	Mean	SD	%	Mean	SD	%	Mean	SD	%
Acari	0.2	.45	< 1	0.4	0.89	1	0.20	0.45	1	-	-	-
Amphipoda	-	-	-	-	-	-	-	-	-	1.50	2.38	2
Annelida	8.8	15.17	**12**	4.80	6.40	**10**	-	-	-	1.25	1.50	1
Arachnida	6.00	8.34	8	0.60	1.34	1	0.80	1.79	2	1.50	1.91	2
Aphidoides	-	-	-	-	-	-	-	-	-	0.25	.50	< 1
Coleoptera	0.20	.45	< 1	0.40	0.55	1	0.20	0.45	1	0.25	.50	< 1
Collembola	0.20	.45	< 1	-	-	-	-	-	-	1.00	2.00	1
Copepoda	0.20	.45	< 1	3.20	4.92	7	1.60	2.61	5	4.25	4.03	**4**
Diptera	7.60	6.54	**11**	16.20	7.09	**34**	6.20	4.49	**19**	38.25	23.77	**39**
Ephemeroptera	-	-	-	0.60	0.89	1	0.20	0.45	1	-	-	-
Gastropoda	-	-	-	-	-	-	0.60	0.55	2	7.00	8.72	7
Hemiptera	0.40	0.55	1	-	-	-	7.40	15.99	**22**	0.25	0.50	**< 1**
Isopoda	-	-	-	0.20	0.45	< 1	-	-	-	-	-	-
Nematoda	3.00	3.08	4	2.80	4.09	6	1.00	1.73	3	0.25	0.50	**< 1**
Odonata	-	-	-	0.20	0.45	< 1	-	-	-	-	-	-
Oligochaeta	19.00	15.13	**27**	15.40	8.68	**32**	10.20	4.44	**31**	24.25	21.83	**24**
Ostracoda	0.20	.45	< 1	0.60	1.34	1	0.40	0.55	1	1.00	1.15	1
Pelecypoda	0.20	.45	< 1	-	-	-	3.40	3.51	**10**	13.25	10.87	**13**
Plecoptera	-	-	-	-	-	-	-	-	-	0.25	0.50	< 1
Polychates	0.20	.45	< 1	1.60	2.50	3	1.00	2.24	3	0.75	0.96	1
Trichoptera	-	-	-	-	-	-	-	-	-	3.75	5.56	4
Insect eggs	.25.20	19.94	**35**	1.20	1.30	0.20	0.45	1	1	-	-	-

Table 12. Mean counts of macroinvertebrates from terrestrial sweep net samples collected in 2008. Note these are mean counts based on 2-3 samples per event, with transects of 10 m each. % indicates the percentage of each mean that is composed of that taxon; values are bolded if that taxon made up 10% or more of the mean for that event.

	Culvert			**Lake**						**Youngs Creek**		
	May			**May**			**July**			**July**		
Taxa	Mean	SD	%	Mean	SD	%	Mean	SD	%	Mean	SD	%
Acari	1.00	0.00	1	1	1.73	< 1	-	-	-	0.33	.58	< 1
Arachnida	-	-	-	10.33	17.90	5	.67	1.15	1	0.33	.58	< 1
Aranea	51.50	.71	**33**	6.33	5.51	3	1.67	2.08	1	.33	.58	< 1
Coleoptera	12.50	3.54	8	11.33	9.71	6	6.67	7.02	6	3.33	1.53	4
Collembola	1.00	0.00	1	-	-	-	-	-	-	-	-	-
Diptera	67	60.81	**43**	139.67	159.84	**73**	68	60.10	**78**	18.67	15.54	**23**
Diplopoda	-	-	-	0.33	.58	< 1	-	-	--	-	-	-
Gastropoda	1.50	2.12	1	19.00	13.89	9	-	-	-	-	-	-
Hemiptera	20.50	19.09	**13**	2.67	3.06	1	11.67	3.51	**10**	55.33	38.76	**68**
Hymenopter	-	-	-	3.33	4.16	2	1.67	1.53	1	3.00	3.00	4
Odonata	0.50	0.71	< 1	0.33	0.58	< 1	-	-	-	-	-	-
Orthoptera	-	-	-	-	-	-	1.33	2.31	1	-	-	-
Trichoptera	-	-	-	-	-	-	1.00	1.73	1	-	-	-
Unknown	-	-	-	.67	1.15	< 1	0.67	1.15	1	0.33	.58	< 1

Data from the open water and emergent vegetation Neuston tow samples at Youngs Creek, collected in 2009, are shown in Table 13. The tow samples were quite variable, reflecting the diversity in composition and abundance of invertebrate taxa found at Youngs Creek over time. Dipteran species made up a significant proportion of most of the open water and emergent vegetation samples at the Culver and Lake sites. This was less true at Youngs Creek; while Dipterans were consistently present, and accounting for at least 10% of the samples, they were usually not the dominant organisms. Other groups that made up a high proportion of samples at Youngs Creek included Ephemeroptera, Acari (mites), amphipods, and oligochate worms.

The tow samples from the Culvert and Lake sites, like those from Youngs Creek, reflect the diversity in composition and abundance of invertebrate taxa found at these sites over time and between sites. Densities and diversity of invertebrates in the tows were generally high in 2008, low in 2009 (during the only sampling event in May), and intermediate in 2010 and 2011 (Figure 33-34, Table 14 and Table 15). We should note that the number of tows collected was small in all years (n=2 per sampling type for most sampling events).

Densities of potential prey varied greatly within and among sampling events at both Mirror lake sites (Figure 33). Much of this variation can be explained by the differences in densities among tows during individual sampling events (Figure 33). The composition of available prey varied considerably as well (Figure 34), with Diplostraca, Copepods, and Diptera typically dominating the samples.

Table 13. Mean counts of macroinvertebrates from Neuston net tows at Youngs Creek in 2009. Nets were towed through aquatic habitats that were either adjacent to emergent vegetation (along the margin of the habitat) or away from the margin in the open water. Note these are mean counts based on 2-3 samples per event, with emergent vegetation tows sampling 10 m each and open water tows sampling 50 m each. % indicates the proportion of each mean that is composed of that taxon; values are bolded if that taxon made up 10% or more of the mean for that event.

	Youngs Creek 2009														
	April			**May**						**July**			**August**		
	Open water			Emergent vegetation			Open water			Emergent vegetation			Emergent vegetation		
Taxa	Mean	SD	%	Mean	SD	%	Mean	SD	%	Mean	SD	%	Mean	SD	%
Acari	2.50	0.71	.10	2.00	0.73	0.03	3.67	2.31	0.22	2.00	3.46	0.02	25.50	34.65	0.06
Amphipoda	1.50	0.71	.06	13.00	10.82	0.21	2.33	2.008	0.14	65.00	91.79	0.67	27.50	38.89	0.06
Arachnida	-	-	-	-	-	-	-	-	-	-	-	-	1.00	1.41	0.13
Aranea	1.00	0.00	0.04	-	-	-	0.33	0.58	0.02	-	-	-	-	-	-
Cladocera	1.00	1.41	0.04	-	-	-	2.00	1.73	0.12	-	-	-	2.50	3.54	0.01
Coleoptera	1.00	1.41	0.04	3.33	1.58	**0.01**	-	-	-	8.00	11.36	0.08	7.00	2.83	0.03
Collembola	3.00	4.24	0.12	3.33	1.58	**0.01**	1.00	1.00	0.06	2.00	2.65	0.02	0.50	0.71	0.00
Diptera	3.00	1.41	0.12	3.33	1.15	**0.05**	2.00	1.73	**0.12**	-	-	-	59.00	62.23	**0.13**
Diplopoda	-	-	-	-	-	-	3.67	1.53	0.22	-	-	--	-	-	-
Ephemeroptera	5.50	7.78	0.22	33.33	18.04	0.55	-	-	-	-	-	-	3.50	4.95	0.01
Gastropoda	-	-	-	3.67	4.04	**0.06**	-	-	-	2.00	3.46	**0.02**	11.50	13.44	**0.03**
Hemiptera	0.50	0.71	0.02	1.00	1.73	**0.02**	-	-	-	13.00	2.00	**0.13**	44.00	52.33	**0.10**
Hymenoptera	-	-	-	-	-	-	-	-	-	4.00	4.58	0.04	-	-	-
Isopoda	-	-	-	-	-	-	-	-	-				0.50	0.71	0.00
Odonata	-	-	-	-	-	-	-	-	-	-	-	-	1.00	1.41	0.00
Oligochaeta	-	-	-	-	-	-	-	-	-	0.67	1.15	0.01	250.00	353.55	0.56
Ostracoda	1.50	0.71	0.06	-	-	-	1.33	2.31	0.08	-	-	-	2.50	3.54	0.01
Plecoperaa	--	-	-	0.67	1.15	0.01	-	-	-	0.33	0.58	0.00	6.50	9.19	0.01
Pelecypoda	-	-	-	0.33	0.58	0.01	-	-	-	-	-	-	-	-	-
Trichoptera	4.50	6.36	0.18	2.67	2.08	0.04	-	-	-	0.67	0.58	0.01	-	-	-
Unknown	-	-	-	-	-	-	0.67	1.15	0.04	-	-	-	2.50	3.54	.01

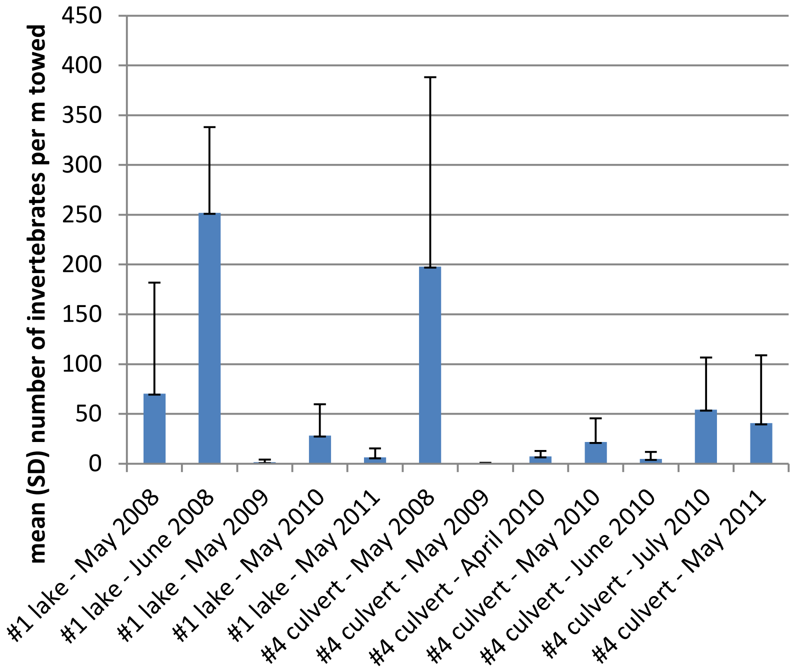


Figure 33. Mean (SD) number of invertebrates per m towed, with all emergent vegetation tows and open water tows averaged by sampling event.

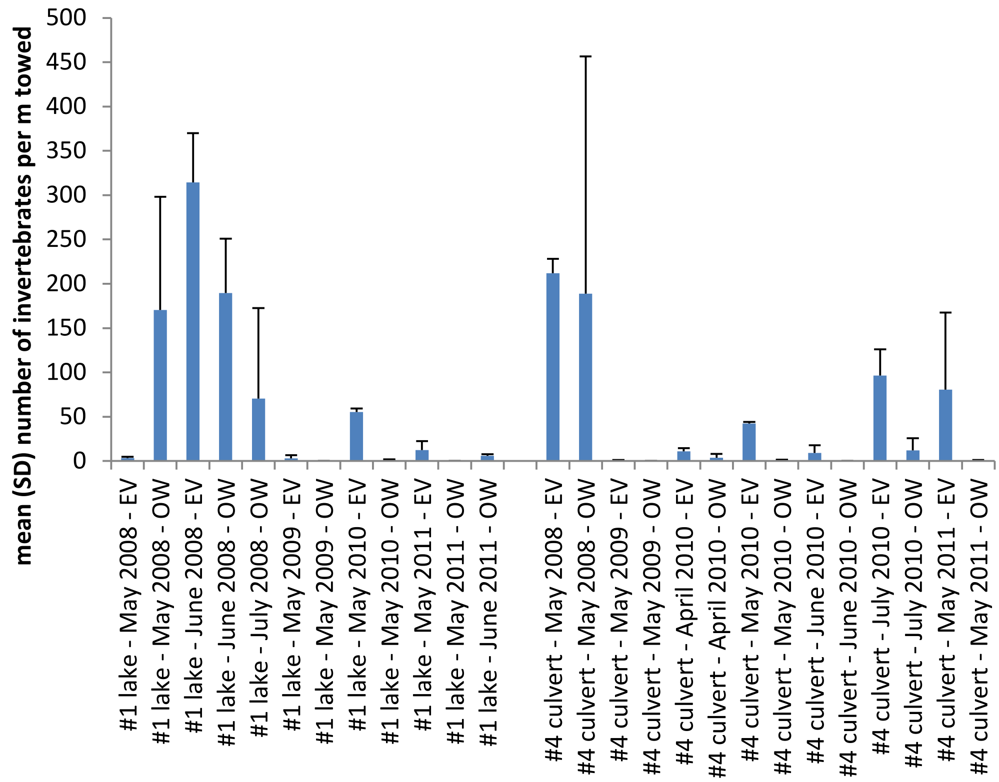


Figure 34. Mean (SD) number of invertebrates per m towed by sampling event and by the type of habitat sampled; EV = shoreline aquatic habitat with emergent vegetation, and OW = open water habitat. Note the high variation in counts within a sampling type as well as between sampling types across sampling events.

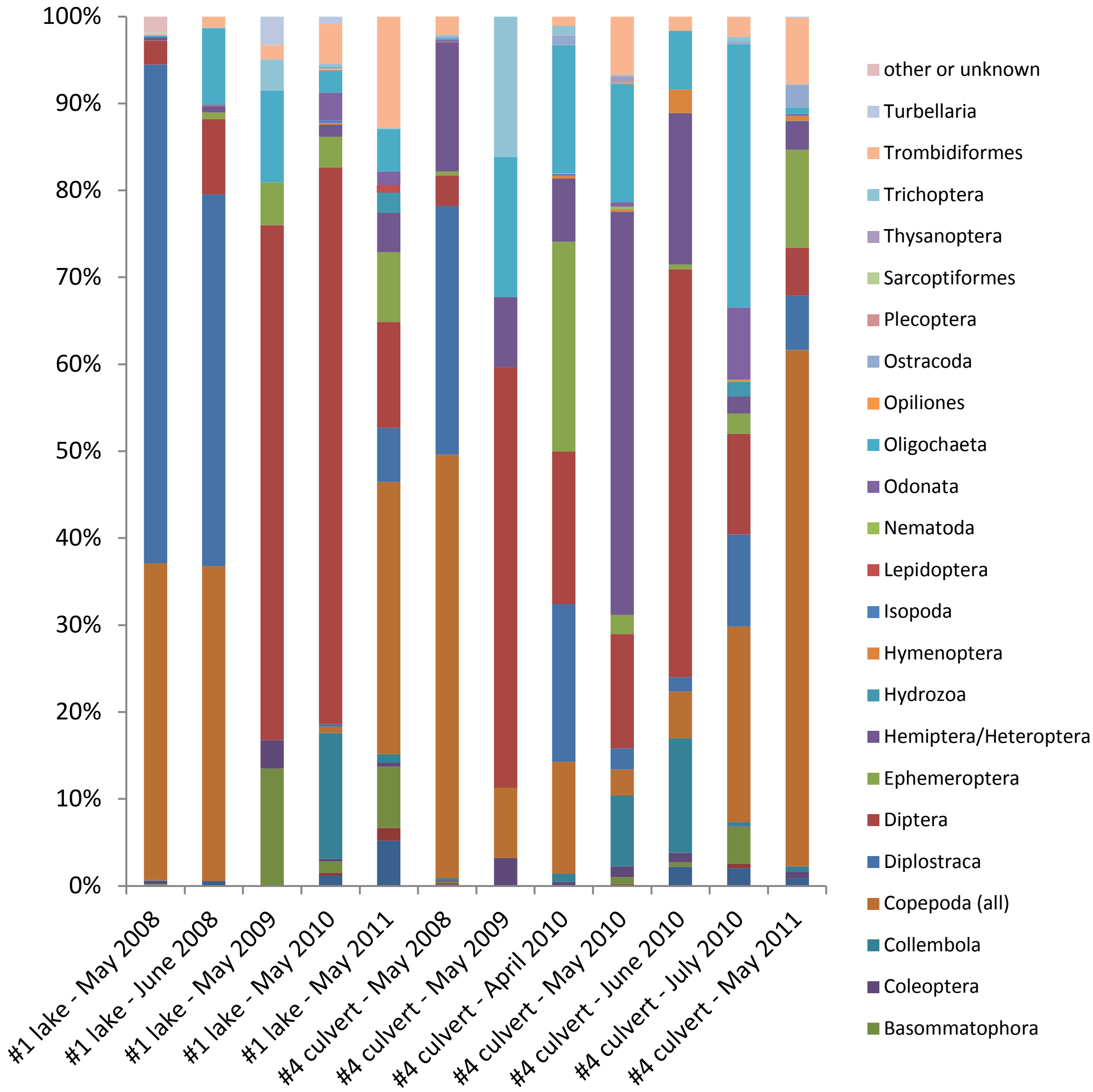


Figure 35. Mean proportions of invertebrate taxa collected in tow samples at Mirror Lake Complex Lake and Culvert sites between 2008 and 2010 by sampling event.

Salmon diet samples. We have identified and counted 4807 individual prey items in 87 Chinook salmon stomachs collected from the Mirror Lake Complex Lake and Culvert sites. On average, a typical juvenile Chinook salmon had 60.7 (SD 48.4) and 52.7 (SD 114.2) prey items in its stomach at the time of capture at the Lake and Culvert sites, respectively. The range in the number of prey items per stomach was greater for fish collected from the Culvert (0-718 individual prey items) compared to the Lake (6-196), but there was no overall difference in the number of prey consumed between the sites (Figure 35; t-test, $p = 0.72$) and no clear trend over time across sites (R^2 for trendline by year across sites = 0.12). These patterns are similar to those observed at other Columbia River sites sampled over the same period, where the variation among sampling events within a site was often as great as the variation among sites (Figure 36, Johnson et al. 2011).

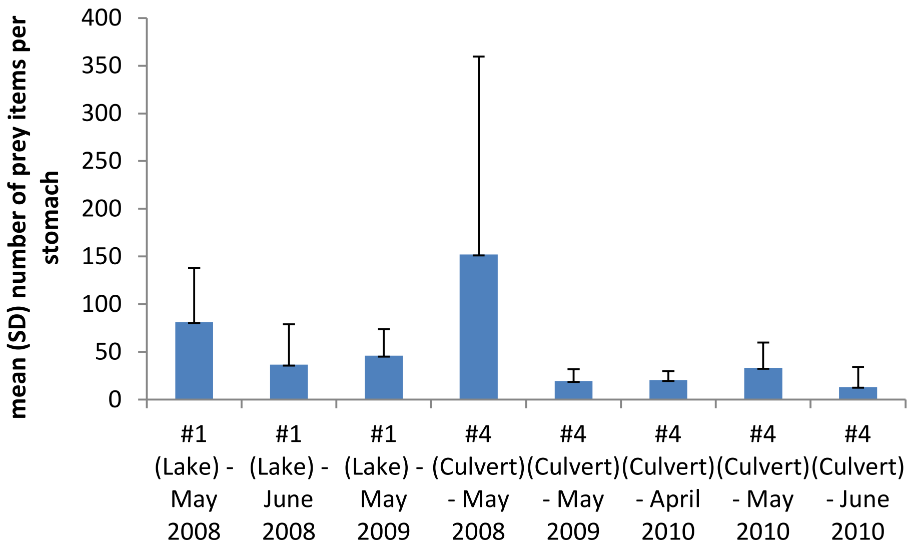


Figure 36. Mean (SD) number of prey items per juvenile Chinook stomach, by sampling event, for the Mirror Lake Complex Lake and Culvert sites.

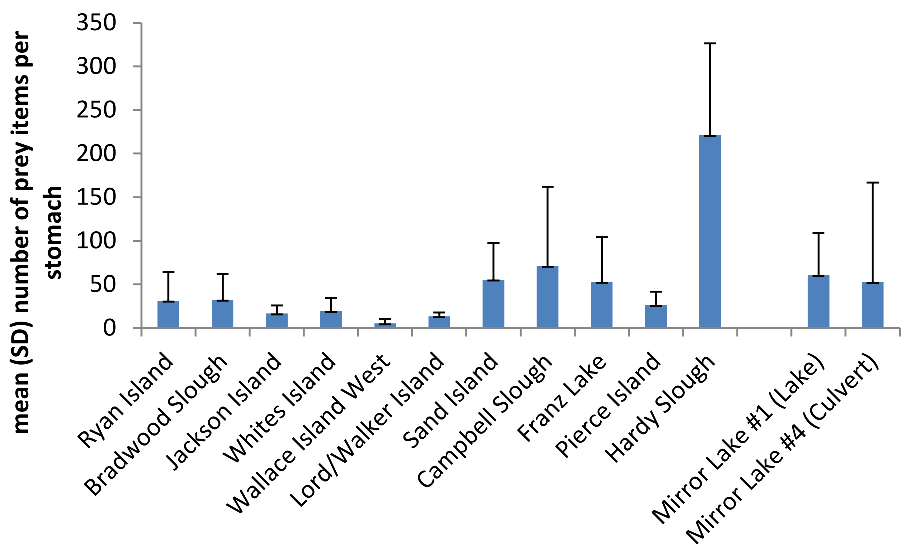


Figure 37. Mean (SD) number of prey items per juvenile Chinook stomach, averaged over all sampling events for each site, for Mirror Lake sites as well as other Lower Columbia River sites.

We identified 20 different orders of invertebrates from the stomach samples, and the dominant prey items in most stomachs were Diptera larvae and pupae, Diplostraca and Copepods (Figure 37). The overall proportions of prey items in diets from the Lake were quite similar to those from other sites along the river, with Diptera equaling 84% of the prey items by count (averaged over all sampling events). Diplostraca (Cladocera, primarily Daphnia spp.) were common in diets from the Lake in June 2008, but were less than 10% for all other sampling events at the two sites. Likewise, diets from Chinook salmon collected at the Culvert were similar to the Lake and other sites along the river (dominated by Dipterans), except for samples collected in May 2008 in which Copepods were dominant (60%). The vast majority of the Dipterans were aquatic midge (Chironomidae) larvae and pupae, and Copepods were primarily Cyclopoids with fewer Calanoids. Although Copepods and Cladocerans were numerically important in the diets from two sampling events, Dipterans are likely the most energetically important prey items given their larger size. Individual Chironomid larvae and pupae were on average 3x larger than individual Cladocera (preliminary blotted wet weight estimates).

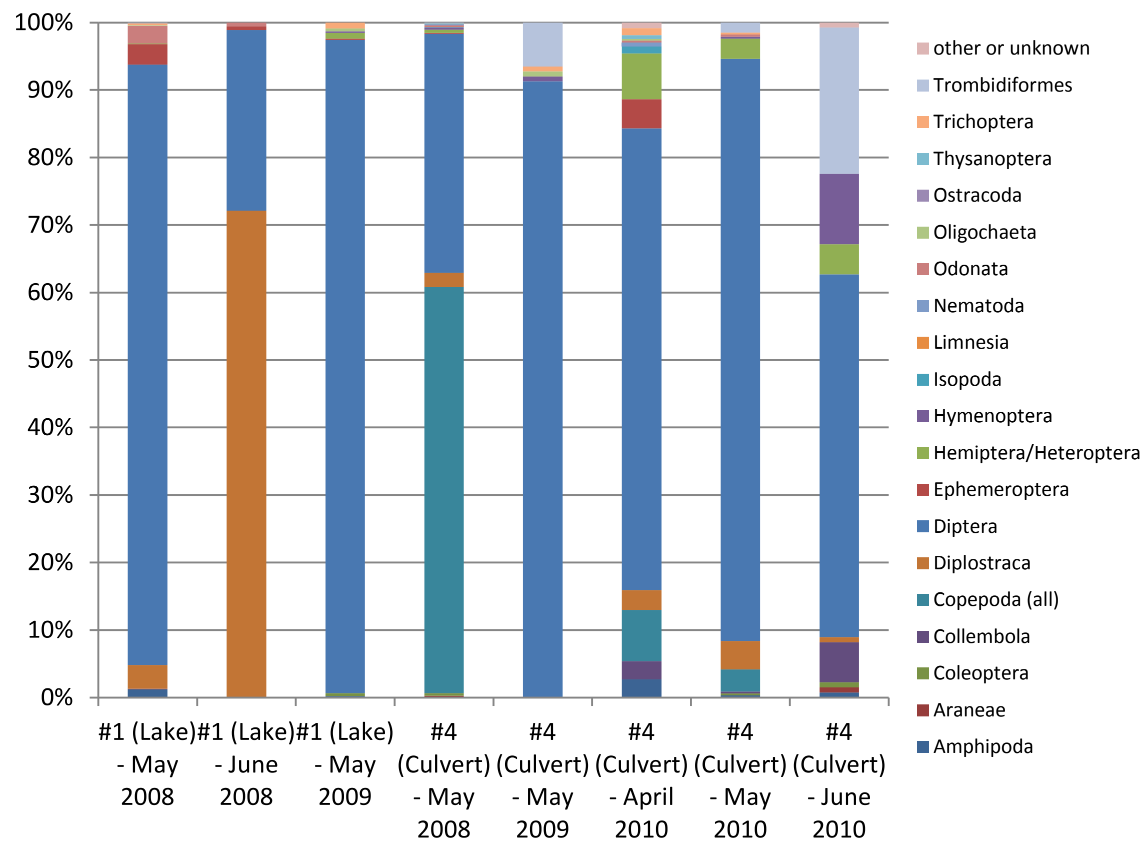


Figure 38. Mean proportions of prey taxa found in juvenile Chinook diets (by counts), for each sampling event at the Mirror Lake Complex Lake and Culvert sites.

Similarity analyses that use both the abundance and diversity of prey consumed by individual fish across these sampling events indicate that there was a significant difference in consumed prey among years but not between sites (PERMANOVA, 2 way test with site fixed and year random; p (perm) values were site=0.268, year=0.002, sitexyear=0.068).

Electivity analysis. Ivlev's prey electivity values, indicating which available prey taxa were preferred or avoided, show a consistent preference for Dipterans and avoidance of some of the other available taxa, even those that are quite abundant at the sites (Table 13 and Table 14). We suspect the preference for Diptera is largely explained by their relative abundance coupled with their size. Although Chironomids are quite small, and Chinook could consume much larger prey items, few larger prey items are plentiful if even available. For example, the five fish sampled in June 2008 at site #1 did consume a relatively high proportion of Cladocera (68% of their diets on average), but they did not consume the abundant Copepods.

Table 14. Mean (SD) numbers of invertebrates in Neuston tow samples (# per m towed) and from juvenile Chinook stomachs (# per stomach) from Mirror Lake site #1. Ivlev's electivity values are given to illustrate if salmon were preferentially selecting or avoiding invertebrates in their habitats. Positive values indicate a relative preference, negative values indicate prey were avoided relative to their availability. These taxa represent the 10 most abundant taxa present in the tows and/or the diets of fish.

	Mirror Lake # 1 - Lake								
	May-08			Jun-08			May-09		
	tows	diets	electivity	tows	diets	electivity	tows	diets	electivity
Amphipoda	0.01 (0.13)	1.00 (1.35)	0.79	1.37 (1.61)		-1		0.10 (0.32)	1
Coleoptera	0.27 (0.36)		-1	0.10 (0.24)		-1	0.05 (0.10)	0.20 (0.42)	-0.77
Collembola	0.04 (0.09)		-1			-1			
Copepoda	25.61 (39.81)		-1	91.18 (61.38)		-1			
Diplostraca	40.29 (1.90)	2.85 (4.67)	-0.88	107.74 (57.55)	26.4 (33.90)	0.26			
Diptera	1.9 (1.47)	72.20 (54.48)	0.94	21.84 (25.55)	9.8 (9.07)	0.51	0.90 (1.54)	44.60 (27.67)	0.24
Ephemeroptera	0.03 (0.04)	2.46 (3.41)	0.97	1.92 (3.85)	0.20 (0.45)	-0.17	0.08 (0.15)	0.10 (0.32)	-0.92
Hemiptera/ Heteroptera	0.26 (0.33)	0.08 (0.28)	-0.59	1.84 (2.40)		-1		0.40 (0.70)	1
Oligochaeta	0.15 (0.18)	0.08 (0.28)	-0.39	21.96 (25.22)		-1	0.16 (0.29)	0.20 (0.42)	-0.92
Trombidiformes	0.13 (0.18)	0.08 (0.28)	-0.33	3.19 (3.45)		-1	0.03 (0.05)		-1

Table 15. Mean (SD) numbers of invertebrates in Neuston tow samples (# per m towed) and from juvenile Chinook stomachs (# per stomach) from Mirror Lake site #4. Ivlev's electivity values are given to illustrate if salmon were preferentially selecting or avoiding invertebrates in their habitats. Positive values indicate a relative preference, negative values indicate prey were avoided relative to their availability. These taxa represent the 10 most abundant taxa present in the tows and/or the diets of fish.

	Mirror Lake #4 - Culvert														
	May-08			May-09			Apr-10			May-10			Jun-10		
	tows	diets	electivity	tows	diets	electivity	tows	diets	electivity	tows	diets	electivity	tows	diets	electivity
Amphipoda	0.23 (0.43)	0.07 (0.27)	-0.43				0.01 (0.01)	0.56 (1.46)	0.95	0.02 (0.02)	0.01 (0.32)	0.62	0.01 (0.14)	0.01 (0.32)	-0.48
Coleoptera	0.24 (0.23)	0.57 (0.94)	0.51	0.01 (0.02)		-1	0.03 (0.05)		-1	0.28 (0.43)	0.01 (0.32)	-0.62	0.05 (0.10)	0.01 (0.32)	-0.18
Collembola	0.51 (1.14)		-1				0.07 (0.09)	0.56 (1.76)	0.47	1.77 (3.29)	0.10 (0.32)	-0.93	0.61 (1.20)	0.80 (2.20)	-0.37
Copepoda	96.47 (90.81)	91.5 (197.87)	0.1	0.03 (0.05)		-1	0.94 (1.54)	1.56 (2.97)	-0.26	0.66 (1.10)	1.10 (2.02)	0.04	0.25 (0.27)		-1
Diplostraca	56.67 (110.59)	3.21 (10.91)	-0.86				1.32 (2.52)	0.61 (1.09)	-0.072	0.53 (0.81)	1.40 (2.84)	0.27	0.08 (0.15)	0.10 (0.32)	-0.36
Diptera	6.96 (5.67)	53.9 (49.42)	0.82	0.15 (0.24)	18.00 (12.64)	0.31	1.28 (1.44)	14.06 (7.79)	0.59	2.86 (2.97)	28.90 (23.73)	0.74	2.16 (3.53)	7.20 (9.26)	0.08
Ephemeroptera	0.84 (1.13)	0.21 (0.43)	-0.5				1.76 (3.17)	0.89 (3.07)	-0.7	0.48 (0.65)		-1	0.03 (0.05)		-1
Hemiptera/ Heteroptera	29.45 (40.31)	0.71 (0.99)	-0.94	0.03 (0.05)		-1	0.53 (0.85)	1.39 (3.01)	-0.04	10.10 (13.65)	1.00 (1.89)	-0.88	0.80 (1.23)	0.60 (0.97)	-0.58
Oligochaeta	0.4 (0.89)		-1	0.05 (0.1)	0.14 (0.38)	-0.91	1.08 (2.01)	0.06 (0.24)	-0.96	2.96 (3.31)		-1	0.31 (0.41)		-1
Trombidiformes	4.19 (6.30)	0.14 (0.36)	-0.92		1.29 (3.4)	1	0.08 (0.09)		-1	1.46 (1.86)	0.50 (1.08)	-0.64	0.08 (0.10)	2.90 (9.17)	0.86

3.3 Discussion

The goal of the Mirror Lake salmon and prey sampling is to evaluate the effectiveness of site enhancements on salmonid prey availability, salmonid occurrence, and salmonid health and condition at the Mirror Lake Complex restoration sites. This is being accomplished by 1) comparing data on fish assemblages, prey types and abundance, salmon habitat occurrence, and salmon health indicators before and after enhancements, and 2) comparing data from Mirror Lake with other relatively undisturbed monitoring sites in the Lower Columbia, such as the Ecosystem Monitoring sites in Reach H, to see whether the restoration activities are helping the sites to approach reference conditions. The restoration program has several objectives: 1) increase the number of salmon in the Mirror Lake Complex; 2) restore fish community composition in the Mirror Lake complex; 3) increase salmon diversity within the Mirror Lake complex; 4) improve salmon health, condition, and realized function of the Mirror Lake Complex; 5) improve water quality in the Mirror Lake Complex, with particular emphasis on water temperature; and 6) improve wetland habitat quantity and quality in the Mirror Lake Complex, including prey quality and quantity. In this section we will review each objective and discuss what progress has been made in realizing the objective over the past four years.

3.3.1 Objective 1: Increase the number of salmon in Mirror Lake Complex from pre-restoration levels.

Generally, we do not yet see increases in the number of salmon at the Mirror Lake sites as a result of restoration actions. At the Culvert and Lake sites, Chinook and coho salmon densities, based on catch per unit effort data, were quite variable between 2008 and 2012 but did not show any clear increasing trends. Similarly, the period of time during which Chinook and coho salmon were found at these sites varied from year to year, but showed no clear trends. At Youngs Creek and Latourell Creek, coho salmon density also varied from year to year but showed no consistent increase, while at the Confluence, coho densities declined between 2010 and 2012, for reasons that are not known. We did, however, observe higher Chinook salmon densities at the Confluence in 2011 and 2012 than in 2010, most likely as a result of the unusually high water conditions that made the site more accessible to Chinook salmon from the mainstem Columbia.

3.3.2 Objective 2: Restore fish community composition at the Mirror Lake Complex sites.

Over the past several years, we have seen changes in several measures of fish diversity at the Mirror Lake complex sites. One of the most obvious changes has been an increase in the proportion of stickleback in catches at all of the sampling sites. Increased dominance of stickleback is typically associated with a number of other changes in fish community indices, including declines in the Shannon-Weiner species diversity, an increased percentage of native fish in catches, and declines in the proportions of other species, including non-native fishes such as pumpkinseed and carp, as well as native fishes such as chiselmouth and salmon. These trends are clearly seen at the Culvert and Lake between 2008 and 2011, when the proportion of stickleback in catches was highest. In 2012, the proportion of stickleback in catches declined somewhat at these two sites, with associated increases in species diversity, and of other species (e.g., chiselmouth, pumpkinseed, and salmon) in catches. With the increased dominance of stickleback, fish community composition at the Lake and Culvert is more similar to the stickleback-dominated communities found at tidal freshwater sites in Reach C of the Lower Columbia, as well as at Hardy Slough in Reach H (Sagar et al. 2013) and at the nearby Sandy River delta (Sather et al. 2009). On the other hand, the large number of stickleback could have a negative effect on juvenile salmon, as there is evidence that their diet preferences overlap (Spilseth and Simenstad 2011).

Stickleback also showed increased over time in catches at the Confluence, Latourell Creek, and Youngs Creek. At Youngs Creek, as at the Culvert and Lake, this trend peaked in 2011, but at Latourell Creek and the Confluence, the proportion of stickleback in catches continued to increased, reaching the highest percentage in 2012. This dominance of stickleback was associated with particularly low species diversity values at Latourell Creek and the Confluence in 2012. Another major change observed at the Confluence, Latourell Creek and Youngs Creeks was increased species richness and percentages of invasive species at all three sites in 2011and 2012, as compared to earlier years, apparently due to a high water years that gave these species access to the sites. This effect was relatively minor at Youngs Creek in 2011, where species richness showed no clear trend, although carp, a non-native species were found here for the first time in 2011. In 2012, changes were clearer, with species richness increasing to 9 species, including two native species, chiselmouth and largescale sucker, present for the first time in 2012. At the Confluence and Latourell Creek, the number of species increased most markedly between 2010 and 2011. In 2012, with the increased dominance of stickleback at these sites, species richness, diversity, and percentage of non-native species all declined somewhat. However, values were still higher than in 2012, and several species observed for the first time in 2011 (e.g., chiselmouth, pumpkinseed) were found again in 2012. At this point is it not clear whether these non-native species will become established in the Mirror Lake Complex above the Lake site, or what effect they might have on salmon or other native fish species or other aquatic organisms at these sites. However, introductions of non-native species have been associated with declines in native fishes (Sanderson et al. 2009, ISAB 2008), and non-native species may negatively affect juvenile salmon as predators or as competitors for similar prey resources (Sanderson et al. 2009).

3.3.3 Objective 3: Increase salmon species richness and diversity.

The number of salmon species at the Mirror Lake Complex sites generally changed little from 2008 to 2012, with no clear trends to suggest that an increased number of species are using the sites. The one exception was at the Confluence site, where Chinook as well as coho salmon were observed for the first time in 2011, and were found again in 2012. Diversity, which takes into account both number of species and the evenness of their distribution, showed no increasing trends at the Culvert, Lake, Latourell Creek, or Youngs Creek, but was much higher at the Confluence in both 2011 and 2012 than in 2010, because of the presence of both coho and Chinook salmon. Salmon diversity was lower in 2012 at all the Mirror Lake Complex sites than at the Reach H sites sampled as part of the EMP.

Proportions of hatchery fish in catches at Mirror Lake sites showed no clear trends, though some variation was observed. In the case of Chinook salmon, the percentage of hatchery fish observed at the Culvert was much higher in 2011 than in 2008-2010, but the percentage decreased to more typical levels in 2012. At the Lake in 2011 and 2012, and at the Confluence in 2012, proportions of marked Chinook were somewhat higher than in earlier years. Hatchery coho were found only at the Culvert. Coho salmon catches at this site were increasingly dominated by unmarked fish from 2008-2011, but in 2012, the percentage of unmarked coho decreased dramatically, to a level below that observed in 2008.

Salmon size class and life stage diversity of unmarked Chinook and coho salmon at the Mirror Lake Complex sites was generally quite variable among sites and years, but no obvious trends emerged. Chinook salmon were mainly in the fry and fingerling size ranges, and the number of yearling-size fish encountered was small. In comparison to the Reach H EMP sites, the Culvert generally had higher proportions of smaller size classes (< 50 mm), especially in years when sampling was more intensive earlier in the season, while the Lake and Confluence had higher proportions of larger size classes (> 50). These trends were also seen in 2012 at the Lake and Confluence; at the Culvert, the size class distribution at the Culvert in 2012 was very similar to that of the Reach H EMP sites. Size class diversity at both the Culvert and Lake were also comparable to Reach H sites in 2012 Both marked Chinook and coho salmon were fairly uniform in size at all sites, with the majority of both species within the fingerling size range (70 to 100 mm), and in comparison to unmarked fish, size class diversity at all sites was relatively low. This is typical of hatchery-origin salmon, as has been noted in a number of other studies (Bottom et al. 2005; Roegner et al. 2008; Sather et al. 2009; Johnson et al. 2011), including the Ecosystem Monitoring Project (Sagar et al. 2013).

The genetic diversity of Chinook stocks using the Lake and Culvert sites did not change greatly between 2008 and 2011 (2012 data are not yet available), but was consistently comparable to patterns observed at other Reach H sites sampled as part of the EMP (Sagar et al. 2013), as well as at other nearby sites such as the Sandy River Delta (Sather et al. 2009; Johnson et al. 2011). Marked Chinook at both sites were primarily from the Spring Creek Group fall Chinook stock, while unmarked fish belonged to a diverse array of stocks including Upper Columbia summer/fall Chinook, Snake River fall Chinook, and Deschutes River fall Chinook, as well as West Cascades fall and Spring Creek Group fall Chinook stocks from the Lower Columbia River ESU.

3.3.4 Objective 4: Improve salmon health, condition, and realized function at the Mirror Lake Complex

Generally, our measures of salmon health and condition show no clear increasing or decreasing trends at the at the Mirror Lake Complex sites. In unmarked Chinook salmon, condition factor (K) at the Culvert was typically lower than at Reach H EMP sites, while values of K in unmarked Chinook salmon from the Lake and the Confluence were more similar to the Reach H sites, but in 2012, K for unmarked Chinook at both sites was comparable to the value for unmarked Chinook at Reach H. At the Confluence, K was lower in 2012 than in 2011, but only one fish was sampled. Among marked Chinook, no significant differences were observed among years or with Reach H at any of the Mirror Lake sites. Similarly, the lipid content of Chinook salmon from the Culvert and Lake sites (no data are available for the Confluence) was comparable to values observed for Chinook salmon in the Reach H EMP sites, and while it varied from year to year, showed no clear increasing or decreasing trends. Chinook salmon growth rates at the Lake and Culvert were also comparable to those of Chinook salmon from Reach H sites, and tended to be somewhat higher in 2009 and 2010 than in 2008.

We have no data on lipid content or growth rates for coho salmon. Condition factor (K) for coho salmon at the Culvert and Lake were generally comparable to those for Reach H EMP sites. At Youngs Creek, K values for coho salmon were generally as high or higher than values at Reach H EMP sites, although some variability was observed among years, with the lowest value was in 2012. At both the Confluence and Latourell Creek, K declined between 2010 and 2011, but increased 2012. The reasons for these changes are not known, but suggest that changes associated with restoration have not yet affected coho salmon health measures at these sites.

Chemical contaminants were not identified as risk factor for juvenile salmon at the Mirror Lake Complex sites, and the data on persistent organic pollutants in Chinook salmon from these sites confirm that exposure is generally minimal, with contaminant concentrations in bodies and bile are below estimated thresholds for health effects in juvenile salmon (Meador et al. 2002, 2008; Beckvar et al. 2005, Johnson et al. 2007; Arkoosh et al. 2010). Little change has been observed between 2008 and 2011, but concentrations are already at levels characteristic of relatively undisturbed sites (Johnson et al. 2007, 2013). Higher concentrations of DDTs and PBDEs were found in a few salmon samples from Culvert site, but these fish may have been exposed to these contaminants at other sites in the Columbia Basin, prior to entering the Mirror Lake Complex.

Overall, these results suggest that the health and condition of coho and Chinook salmon from the Mirror Lake Complex sites are comparable to that of these species at Reach H EMP sites, but we are seeing little change in response to habitat improvements at the site.

3.3.5 Objective 5: Improve water quality in the Mirror Lake Complex

As part of the fish sampling effort, relatively little data are collected on water quality; however synoptic temperature data are collected during fishing events at all of the sites. This information suggests that at both the Lake and Culvert sites, high summer temperatures may make these habitats unsuitable or stressful for juvenile salmon in the summer months. Optimal temperatures for juvenile Chinook salmon are below 16°C; temperatures above 16°C have been associated with reduced growth rates (Bisson and Davis 1976; Marine and Cech 2004), with increased morality occurring at temperatures exceeding 20°C (McCullough 1999). Similarly, juvenile coho salmon avoid streams with temperatures above 18°C (Welsh et al. 2001). The Willamette-Columbia River Technical Recovery Team recommends a 7-day average daily temperature of 16°C and a weekly mean temperature of 15°C as the maximum optimal temperatures for juvenile rearing of salmon stocks present in the Lower Willamette and Lower Columbia Rivers (Richter and Kolmes 2003). At the Culvert and Lake, water temperatures were consistently above 20°C in July and August, with temperatures above 30°C in some years. Between 2008 and 2012, no clear trends in water temperatures at these sites were been observed, with the exception of particularly high temperatures in 2009. This did seem to affect salmon occurrence at the sites, as juvenile Chinook salmon were present at both the Lake and Culvert for a shorter period in 2009 than in other years.

At the Confluence, Latourell Creek, and Youngs Creek, temperatures were consistently much cooler than at the Lake and Culvert, rarely rising above 15°C even in the summer months. There has been little sign of any change in temperature as a result of habitat improvements, but temperatures at these sites already appear to be favorable for juvenile salmonids. Overall, these results are not unexpected, as temperatures reductions are not expected until shading from planted trees develops, which will require a number of years (LCEP 2011).

3.3.6 Objective 6: Improve wetland habitat quantity and quality to increase availability of salmon prey

Concurrently collected diet and prey data are available only for Chinook salmon at the Lake and Culvert sites. Like salmonid occurrence data, diet and prey availability sampling for Chinook salmon at the Culvert and Lake reveal similarities between these two sites. Moreover, prey availability and consumption at the Lake and Culvert appeared to be quite similar to patterns seen at other Lower Columbia Rivers sites in Reach H sampled as part of the Ecosystem Monitoring Project (Sagar et al. 2013). The diversity and density of macroinvertebrates in the samples from the Culvert and Lake were quite variable, and there were no clear differences between the two sites in relative abundance of salmonid prey items. Prey densities in tow samples were substantially lower in 2009 than in 2008, and intermediate in 2010 and 2011, so no clear trends in prey quality or abundance were observed. However, because of the limited number of samples analyzed and the spatial and temporal variability in prey, it is hard to be certain that these results represent the true conditions at these sites.

One finding that is striking at these sites, and is consistent with other sites sampled in the region (Sather et al. 2009; Johnson et al. 2011, Sagar et al. 2013), is the relatively high proportion of Dipterans (primarily Chironomidae larvae and pupae) in the diets, and the consistent preference shown for this prey type in the electivity analyses. We suspect the preference for these prey items is largely explained by their relative abundance coupled with their size. Although Copepods and Cladocerans were numerically important in the diets from two sampling events (both in 2008), Dipterans are likely the most energetically important prey items, as Chironomid larvae and pupa are typically about three times larger than individual Cladocera (preliminary blotted wet weight estimates). While Chinook could consume much larger prey items than Chironomids, few larger prey items are plentiful if even available at the Lake and Culvert sites. This evidence of selectivity in prey items may be useful in evaluating the quality of prey resources at the Mirror Lake restoration sites. We did observe consistent dominance of Dipterans in all diet samples collected from the Lake and Culvert in 2009 and 2010, whereas in 2008 other taxa (Cladocerans and Copepods) dominated in two of the 2008 samplings. However, it was not clear that this was related to a chance in prey composition at the sites, based on the tow samples collected in 2009 and 2010.

In Chinook salmon diets, we saw no clear trends over time in the numbers of prey items in stomachs at either the Lake or Culvert site. These patterns were similar to those observed at other Columbia River sites sampled over the same period, where the variation among sampling events within a site was often as great as the variation among sites (Sagar et al. 2013). The numbers of prey items found in stomach contents of Chinook from the Lake and Culvert were also similar to the numbers found in juvenile Chinook salmon from most EMP sites (Sagar et al. 2013), suggesting that these two sites are providing food resources comparable to relatively undisturbed references areas.

Our prey availability data are more limited at Youngs Creek, but the results we have suggest that the macroinvertebrate community at this site is somewhat different from that at the Lake and Culvert sites. While many of the same taxa were present, the Youngs Creek samples generally contained lower proportions of Dipterans and higher proportions of other species, including Ephemeroptera, Acari (mites), amphipods, and oligochate worms than samples from the Culvert and Lake sites. As yet we have no information on coho salmon diets at the Young Creek site, so it is uncertain which of the macroinvertebrate species present might constitute preferred prey, or how densities of these prey items compare with other undisturbed sites. Other studies indicate that, like juvenile Chinook salmon, coho salmon consume Dipterans, especially Chironomid larvae, pupae, and adults, as well as other insects such as along Ephmeroptera, Trichoptera, Plecoptera; amphipods and oligochaetes may also be significant components of the diet when available (Gonzales 2006; Roegner et al. 2010; Allan et al. 2003; Hetrick et al. 1998). These taxa were often present in significant proportion in sediment core, terrestrial sweep, and Neuston tow samples from Young Creek, suggesting that appropriate prey items are available for the coho salmon utilizing the site. However, diet information for juvenile coho salmon from the Young Creek site, as well as comparative data on diets and prey availability from comparable reference sites, would be helpful in interpreting the quality of prey resources in this area and how they may be affecting by restoration activities.

3.4 Conclusions

Our key findings with respect to salmon abundance, fish community composition, salmon diversity, salmon health and condition, water quality, and prey are summarized in Table 16. Our results suggest that the restoration actions at the Mirror Lake Complex have contributed to changes in fish community composition at several sites. Improved passage at the Culvert, in combination with high water conditions, may also have facilitated juvenile Chinook salmon access to the Latourell Creek/Youngs Creek Confluence areas.

Table 16. Key finding from 2008-2012 salmon and salmon prey monitoring at the Mirror Lake complex Restoration sites.

Objective 1: Salmon abundance: - Chinook salmon abundance did not show any clear increasing or decreasing trends. - Coho salmon abundance did not show any clear increasing or decreasing trends.
Objective 2: Fish community composition - Fish community composition has changed and the proportion of sticklebacks in catches has generally increased. - Percentages of some native and non-native species in the catch have declined at the Lake and Culvert as stickleback have increased; however, species proportions are variable from year to year. - Fish community diversity has increased at the Confluence, Latourell Creek and Youngs Creek, but this is partly due to influx of non-native species
Objective 3: Salmon diversity - No clear trends to suggest an increase in salmon diversity, as indicated by species, size class, or genetic stock at the Culvert, Lake, Latourell Creek, or Youngs Creek sites. - There is increased salmon diversity at the Confluence because of the appearance of Chinook salmon in 2011 and 2012, in large part due to high water conditions.
Objective 4: Salmon health, condition, realized function - No clear trends, but at most sites values for growth, lipid content, and condition factor are comparable to Reach H EMP sites
Objective 5: Water quality: - No clear trends, but consistently high summer water temperatures at the Culvert and Lake sites, and cool temperatures suitable for salmon at the Confluence, Latourell Creek, and Youngs Creek
Objective 6: Improved habitat quality in terms of prey availability - No clear trends, but prey resources at the Lake and Culvert are similar in type and quantity to those found at Reach H EMP sites. - Prey composition at Youngs Creek is somewhat different from the Lake and Culvert and from Reach H EMP sites, but many taxa that are known prey items for coho are present. Data are lacking to assess temporal trends.

No clear trends were seen for other indicators. Elevated summer temperatures are a consistent concern at the Culvert and Lake sites, but chemical contamination appears low. While prey availability and fish condition indicators show no improvement, they were typically comparable to levels observed at relatively undisturbed sites sampled at part of the Ecosystem Monitoring Program. This suggests that the Mirror Lake Complex sites have adequate prey resources and that juvenile salmon using these sites are in good health.

Confounding factors

Several factors may limit our ability to evaluate the effectiveness of the restoration actions that have been carried out at the Mirror Lake Complex sites. First, several of the indicators measured would not be expected to change significantly within the time frame that we have been monitoring. These include several factors associated with recovery of riparian vegetation, such as water temperature and related water quality parameters, as well as prey quantity and quality.

Inherent variability in many measures due to changing weather and climatic conditions, seasonal and diurnal variation also make trend evaluation difficult, especially as the amount of sampling we can do is constrained by staff and funding limitations. Salmon density and patterns of occurrence in particular may be strongly affected by year class size, which is dependent on ocean conditions as well as habitat quality at sites utilized by juveniles. It may also vary substantially with yearly changes in weather conditions and water temperature regimes. Also, because of permit constraints, the number of juvenile salmon we can sample is also limited, and this may make it difficult to get reliable estimates of density, size class distribution, and genetic stock composition.

A related problem is that, although many of the indicators we measure vary with the sampling season, because of a variety of factors, including water levels, permit limitations, and site access, we are not always able to sample every site over the same time period each year. Consequently, the months during which data are collected varies from year to year. This complicates our evaluation of inter-annual trends.

Our ability to detect changes may also be constrained by the fact that some indicators of restoration success are already comparable to reference areas. For example, measures of salmon fitness at the Mirror Lake Complex sites are already approaching values at other relatively undisturbed sites in Reach H. Prey density and composition at the Lake and Culvert are also comparable to most of the Reach H reference sites. Consequently, the amount of improvement that can be expected may be limited.

The lack of appropriate reference sites for the Mirror Lake Complex is another confounding factor. For the Culvert and Lake, the Ecosystem Monitoring Program emergent marsh sites in Reach H are reasonable reference areas, but Confluence, Latourell Creek, and Youngs Creek are different habitat types, so comparison with those sites is not appropriate. Our ability to monitor changes in salmon health, condition and realized function as well as prey availability are also limited at Youngs Creek, Latourell Creek, and the Confluence because of our focus on Chinook salmon as a target species, when the dominant salmon species at these sites is coho salmon.

One final concern is that our findings to date suggest that improving access to the area for salmon may also facilitate the movement of non-native species into the Mirror Lake complex, and the long-term effects of such changes in fish community composition are uncertain.

Recommendations

One of the most dramatic changes we have observed are changes in fish community composition at the Mirror Lake Complex sites, including movement of non-native species into some sites, including the Confluence and Latourell Creeks. This suggest that this may be an especially important indicator to monitor in the future, so we can better understand long-term changes and potential consequences. The impacts of the increasing dominance of stickleback at the Lake site, and the potential for non-native species to become established at the Confluence and Latourell Creek are two areas of concern. Another indicator that may be especially important to continue following are apparent increases in coho density at Youngs Creek and Latourell Creek which may be related to habitat improvements.

Coho salmon are a major target species for several of the Mirror Lake Complex sites, but are not as intensively monitored as Chinook salmon. To better assess improvements in their health and condition, and in the quality of habitat at the Confluence, Latourell Creek, and Youngs Creek, it would be useful to sample coho salmon diets, and prey, and measure additional health measures in coho salmon such as lipid content and growth. Additionally, if possible it would be helpful to identify more appropriate reference sites for the Confluence, Latourell Creek, and Youngs Creek, so we can better evaluate whether salmon densities, fish community composition, and salmon health and condition are approaching conditions at relatively undisturbed sites.

Finally, our results to date suggest that long-term monitoring will be necessary to evaluate the effectiveness of restoration actions in the Mirror Lake Complex. Consequently, it would be helpful to consider the frequency of monitoring at these sites going forward into the future, as well as the suite of indicators that will be most useful for assessing the effectiveness of restoration actions at these sites.

4 Planting Success AEM at Mirror Lake and Sandy River Delta

In the fifth year of Action Effectiveness Monitoring (AEM) for the Estuary Partnership, Ash Creek Forest Management LLC (Ash Creek) monitored six ecological restoration projects at the Sandy River Delta and Mirror Lake in Multnomah County, Oregon. Due to their location in the active floodplain of the Columbia River, the Sandy River Delta and Mirror Lake provide habitat critical to sustaining healthy aquatic life. Restoration and stewardship of native vegetation are key to recovering and maintaining functional habitats for listed and rare native fish and wildlife in the Columbia region.

The goal of all of the Sandy River Delta projects is the recovery of native Columbia River floodplain forest and scrub plant communities with associated ecosystem function. Resulting native plant cover is expected to contribute to improved riparian function through small large wood recruitment in aquatic habitats, increased shading of aquatic habitats, and bank stabilization.

Restoration goals at Mirror Lake were defined in the Estuary Partnership conceptual plan (Estuary Partnerhsip 2009). Goals include improved salmon habitat in Latourell and Young Creeks by providing shade to help lower in-stream temperatures by creating overhanging, vegetation for organic matter input, and reestablishing trees as a source for long-term in-stream wood recruitment. Also, an increase in woody plants will promote beaver activity to provide additional waterfowl habitat and rearing habitat for Coho salmon.

Objectives for restoration sites are:

- Reduction in noxious weed cover to levels that enable native plants to become ‘free to grow,’ as shown in declining rates of suppression.
- Increasing density over time of planted and naturally occurring native woody species, measured as stems per species per plot and compared to reference site conditions.
- Future trending of native plant survival, captured in plant vigor estimates.
- Recovery of a native herbaceous layer and ongoing reduction in non-native plant cover, measured per Daubenmire (1952) in $1m^2$ plots taken in Comprehensive Monitoring years.
- Increased shade, taken at sites >5 years with densitometer in Comprehensive Monitoring years.

4.1 Survey Site Descriptions

4.1.1 Sandy River Delta

The Sandy River Delta (Delta, Figure 39), at the western terminus of the Columbia River Gorge and the eastern edge of the Portland Metropolitan region, represents one of the greatest opportunities and challenges for ecological restoration in northwest Oregon. Restoration at the Delta supports the Forest Service Sandy River Delta Plan (1995), a publicly reviewed and NEPA-approved restoration plan. The plan guides all restoration activities undertaken on site and over time is designed to re-create the largest contiguous hardwood forest in the lower Columbia region, as well as emergent wetland, prairie, scrub and savanna habitats comparable to pre-settlement conditions at the site. Native plant communities of the Sandy River Delta serve important habitat function for salmonids, waterfowl and neo-tropical migratory birds that travel the Pacific Flyway and other species native to western Oregon.

Identified as anchor habitat for salmonid recovery (Sandy River Basin Partners, 2008), the Delta supports important life stages of federally listed Evolutionarily Significant Units (ESUs) Lower Columbia River coho *(Oncorhynchus kisutch)*, Lower Columbia River Chinook (*O. tshawytscha*), Lower Columbia River steelhead (*O. mykiss*) and Columbia River chum (*O. keta*), as well as upriver salmonid populations (Sather, N. et al, 2009). The site also supports Pacific smelt (*Thaleichtys pacificus*), recently listed as Threatened by the National Marine Fisheries Service.

Restored native plant communities also serve important habitat functions for the many other important resident and migratory wildlife species that occur at the Delta. According to Robert Altman of the American Bird Conservancy and others, the high diversity of bird species that share the Delta is exceedingly unique in the local area and includes over 80 neo-tropical migrant species and over 20 waterfowl species (US Army Corps of Engineers, 2011). Bald eagle (*Haliaeetus leucocephalus*), Northern red legged frog (*Rana aurora aurora*) and painted turtle (*Chrysemys picta*) occur at the Delta, as do pileated woodpecker (*Dryocopus pileatus*), Western bluebird (*Sialia mexicana*), Vaux's swift (*Chaetura vauxi*) and Yellow-billed cuckoo (*Coccyzus americanus*). Thought to be extirpated from the region, the cuckoo was sighted on a restoration sites at the Delta that are managed through a collaborative effort with the Forest Service, Confluence Project and Ash Creek Forest Management (pers. comm, J. Withgott, July 2009, reported to Oregon Birders Online).

The Delta is also an important scenic, educational, and economic resource. Thousands of visitors walk and boat around this high-profile site each year; and hundreds of volunteers work in ecological stewardship and research on site. The Delta is a popular location for local birders, who have sighted a number of rare bird species. The Delta is also popular with dog walkers, cyclists, hunters, and other outdoor enthusiasts looking for an escape near the Metro center. Investment in restoration at the Delta also has enabled the local farming, nursery and forestry communities to participate in all aspects of restoration.

Historically, land conversion focused on ranching and farming in the early 20th century, which was followed by conservation by the USDA Forest Service in the 1990s. The result of historic land practices have opened the Delta to noxious weed infestation and associated loss of critical ecosystem processes. Weeds that inhibit native plant reestablishment have become established, resulting in nearly 100% invasive non-native monocultures of Himalayan blackberry (*Rubus armeniacus*) and reed canarygrass in significant portions of all units. It is well established that exclusion of native plants by noxious weeds causes decline in near and long term of production of habitat structures and function that can support life cycles of native fish and wildlife species (USDA Forest Service, 2011).

A restoration partnership between the US Forest Service, Army Corps of Engineers, Bonneville Power Administration, Oregon Watershed Enhancement Board, Estuary Partnership, Ducks Unlimited, Ash Creek Forest Management, and others has restored over 700 acres of diverse native habitat at the Delta. The partnership has removed over 700 acres of non-native vegetation, planted over 700,000 native trees and shrubs, and established large, contiguous tracts of self-sustaining, diverse assemblages of native plants. In the process, project partners have developed and refined approaches to restoration of large tracts that are successful and efficient, and have provided a valuable model for other projects along the Columbia River and elsewhere.

4.1.2 Mirror Lake

The Mirror Lake site is a 390-acre parcel located within Rooster Rock State Park approximately 10 miles east of Troutdale in the Columbia River Gorge (Figure 40). The site is unique, because it functions as one of few remaining large, contiguous tract of historic bottomland hardwood forest within the Columbia River floodplain. The site includes two lakes, two streams (Young and Latourell Creeks), expansive wetlands, and remnants of the historic bottomland hardwood forest. Historic records indicate that the area south of I-84 was dominated by Oregon ash (*Fraxinus latifolia*) up to 20 inches in diameter with two semi-open wetland prairies with scattered willows and larger willow-dominated bottomlands. (Estuary Partnership, 2009).

The site supports several evolutionarily significant units (ESUs) of federally listed salmonids, including spawning populations of Lower Columbia River Coho salmon and rearing and/or off-channel habitat for steelhead and Chinook salmon, potentially from both Lower Columbia and up-river ESUs (LCEP 2009). Numerous other species of interest also are found on-site, including great blue heron (*Ardea herodias*), red-legged frog (*Rana aurora*), lamprey (*Lampetra spp.*), sculpins (*Cottus spp.*) and bald eagle. The site's location at the western terminus of the Columbia River Gorge and within the Pacific Flyway also makes it a valuable scenic resource and important habitat for numerous neotropical migratory birds and waterfowl.

Beginning in the early 1900s, the majority of the property was cleared and used as farming and grazing land. The property was acquired for public open space, and Oregon Parks and Recreation Department (OPRD) ended cattle grazing practices in the early 1990s. Since that time, invasive species have established expansive, dominant communities that prevent significant recruitment of native species and the re-establishment of native riparian habitats. The primary invasive species of concern on the site are reed canarygrass, Himalayan blackberry, and Canada thistle (*Cirsium arvense*) (LCEP 2009).

The site has been a top restoration priority for OPRD since its acquisition, but resources have not been available to further OPRD's goals. In 2005, the Lower Columbia River Estuary Partnership funded construction of a bridge that provided access for restoration activities and replaced a culvert that was an impediment to fish passage. Oregon Department of Transportation initiated site preparation; Estuary Partnership completed phase-1 restoration (weed control and plant installation) on 29 acres and began AEM in 2008.

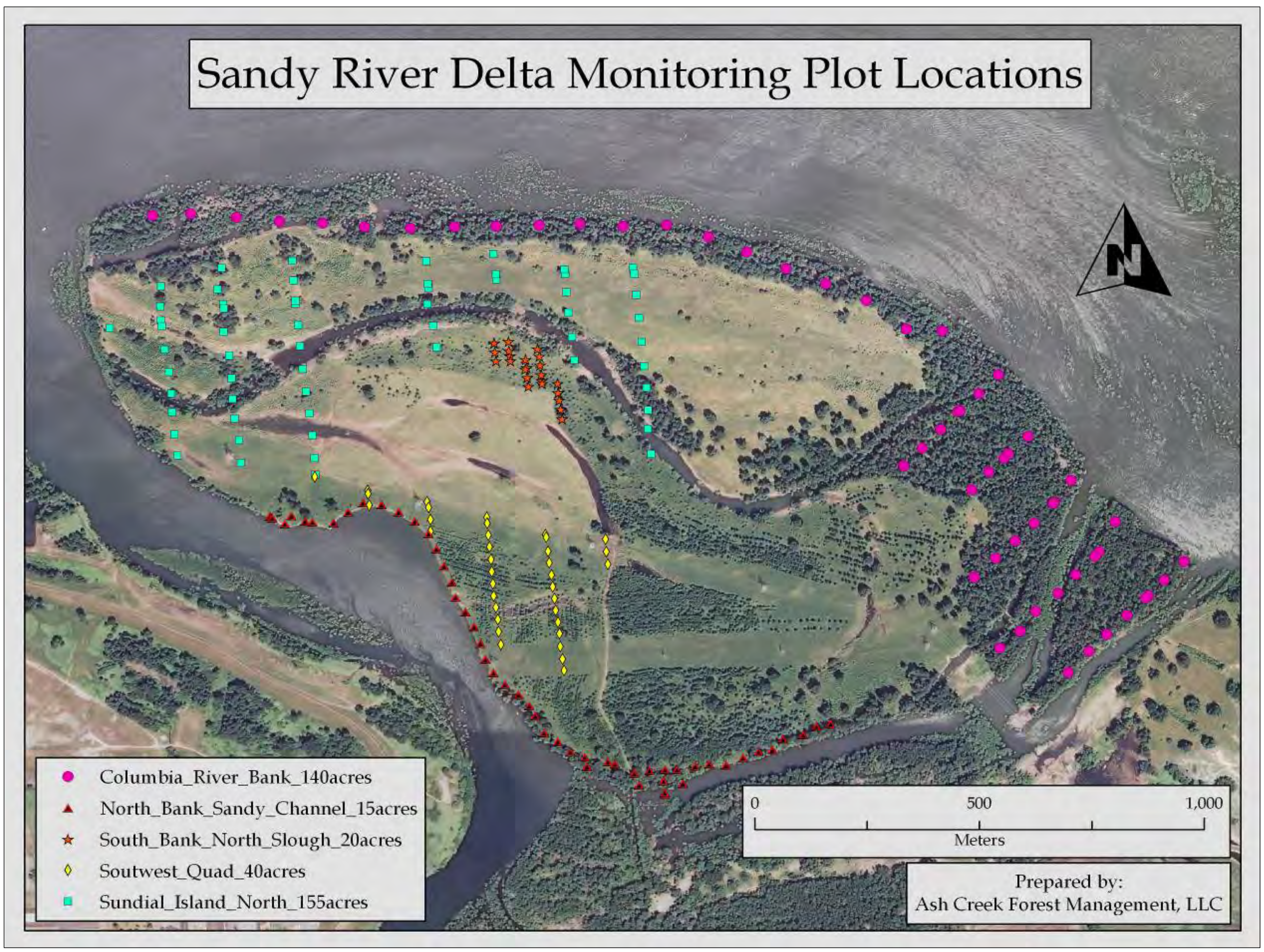


Figure 39. Monitoring plot locations, Sandy River Delta units.

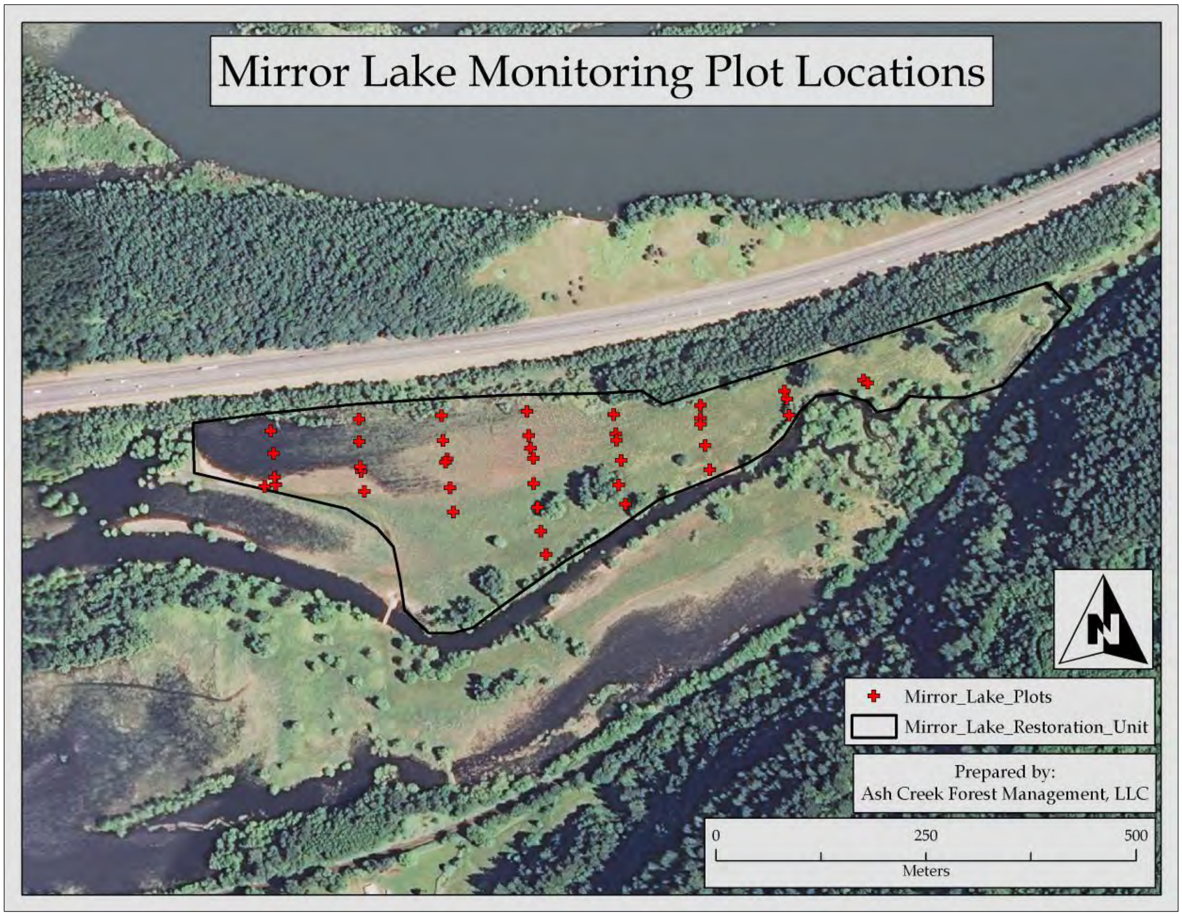


Figure 40: Mirror Lake Monitoring plot locations

4.2 Methods

Sampling protocol followed Roegner et al. (2009) *Protocols for Monitoring Habitat Projects in the Lower Columbia River and Estuary*. First-year monitoring initiated on five units in 2008 and on the Columbia River Bank in 2011 followed the Comprehensive monitoring protocol. Permanent transects and plots were established at each unit and spaced according to unit size to ensure sampling of the entire restoration area. At the 15-acre North Bank Sandy River, plots were established along a changing azimuth to capture interior and edge habitat. On all units, one-third of total plots per unit were randomly chosen to be permanent and marked with PVC, pink flagging and a pink marking whisker, and photos were taken to capture visual change over time. In 2009, 2010, 2011 and 2012 permanent plots were re-located and sampled. All other plots along each transect were installed systematically at pre-determined intervals based on unit size. Where a plot center landed on or near a boundary, the plot was transformed into a 5.6 m radius semicircle (noted in data). Where the middle of the woody plant (stem or stem cluster) was not within the plot radius, it was not included in the survey.

At each plot surveyed with the Comprehensive monitoring protocol, surveyors counted stems of all woody species, and recorded for each whether live or dead, whether natural, planted, or unknown; vigor and suppression (defined below) for each plant were also recorded. Canopy cover was measured with a densiometer, and the diameter of the tree nearest the center of the plot was recorded. A 1-m^2 quadrat was placed within the plot and cover of all herbaceous species within the quadrat was recorded. Surveyors noted specific habitat features for plots falling within existing forested areas or exhibiting other atypical conditions.

Animal damage, such as by deer, elk, or voles, was noted. To determine the effectiveness of bamboo installed around tree trunks on vole damage, more detailed data was taken on animal damage in the SW Quad and Sundial Island North where bamboo was installed in December 2011 around many cottonwoods.

At the Columbia River Bank, surveyed with the Rapid monitoring protocol, surveyors counted stems of woody vegetation by species and noted for each whether live or dead, natural, planted or unknown. Plant vigor and suppression, for each plant, and average height for each species was also recorded. Invasive species were listed and their cover in the 4 m radius plot was recorded.

For vigor ratings, 'Low vigor' defined a plant that was damaged or severely stressed due to shade, drought or competition; 'Medium vigor' indicated normal stress expected in recent plantings (discoloration of leaves, herbivory, etc); and 'High vigor' described plants in excellent condition and growing vigorously relative to species growth potential. Plants that were designated 'suppressed' were significantly shaded, crowded and/or overtopped by competing weedy vegetation to the extent that they would not be expected to grow out from underneath the suppressing plant.

For all units, the number of installed plants per hectare was calculated by dividing the total number of installed plants by number of hectares planted. The number of installed plants and percent survival were calculated as: Total number of live, installed plants counted on all sample plots (T) divided by number of plots sampled (n) to yield average of surviving, installed plants per plot (Tp); total per plot was multiplied by 200 (because a 4-m radius plot is $1/200^{th}$ ha) to estimate total number of live, installed plants per hectare (Th); this total was then divided by number of plants originally installed per hectare (i).

$$T / n = Tp$$
$$Tp * 200 = Th$$
$$Th / i = \%\ \text{survival}$$

Because natural recruitment is increasing on AEM units, we calculated total native plant stocking (total number of native stems) per the same formula and tracked these numbers as a measure of overall restoration effectiveness. We also estimated trends in natural recruitment by subtracting numbers of live installed plants from total live plants per plot.

4.3 Results

In July and August 2012 Ash Creek staff returned to five restoration units that were first monitored in 2008, and sampled the Columbia River Bank, first monitored in 2011; these six units totaled 399 acres (Table 17). Comprehensive monitoring protocol was used to survey the five units originally surveyed in 2008: Estuary Partnership's 15-acre "North bank Sandy Channel"; Estuary Partnership's 20-acre "South Bank/North Slough"; Estuary Partnership's and BPA's 40-acre "Southwest Quad" and US Army Corps of Engineers' 155-acre "Sundial Island North"- located at the Sandy River Delta - and the 29-acre "Mirror Lake" unit at Rooster Rock State Park. Rapid monitoring protocol was used to survey the 140 acre "Columbia River Bank" (Figure 39 and Figure 40). Table 18 lists the species that have been planted throughout the units.

Table 17: Restoration Units

Site Name	Acres
Sundial Island North	155
Southwest Quad	40
South Bank/North Slough	20
North Bank Sandy Channel	15
Mirror Lake	29

Table 18. Woody species installed at Sandy River Delta and Mirror Lake

Trees		Shrubs	
Scientific name	**Common name**	**Scientific name**	**Common name**
*Abies grandis**	Grand fir	*Cornus stolonifera*	Red-osier dogwood
Acer macrophyllum	Bigleaf maple	*Holodiscus discolor*	Oceanspray
Alnus rubra	Red alder	*Mahonia aquifolium*	Tall Oregon grape
Crataegus douglasii	Black hawthorn	*Lonicera involucrata*	Black twinberry
Fraxinus latifolia	Oregon ash	*Oemleria cerasiformis*	Indian plum
Populus trichocarpa	Black cottonwood	*Philadelphus lewisii*	Mock orange
Prunus emarginata	Bitter cherry	*Physocarpus capitatus*	Pacific ninebark
*Pseudotsuga menziesii**	Douglas-fir	*Ribes sanguineum*	Redflowering currant
Quercus garryana	Oregon white oak	*Rosa pisocarpa*	Swamp rose
Rhamnus purshiana	Cascara	*Rubus parviflorus*	Thimbleberry
Thuja plicata	Western redcedar	*Rubus spectabilis*	Salmonberry
		Salix lasiandra	Pacific willow
		Salix piperi	Piper willow
		Salix scouleriana	Scouler willow
		Sambucus cerulea	Blue elderberry
		Sambucus racemosa	Red elderberry
		Spiraea douglasii	Douglas spiraea
		Symphoricarpos albus	Snowberry

* planted at Mirror Lake only

Survival rates of installed plants showed an increase over 2011, on all sites except South Bank/North Slough, and were 61% at Mirror Lake and ranged from 85% to 111% on units at Sandy River Delta (Table 19). With the exception of the Columbia River Bank, the ratio of trees to total woody plants has decreased from 2011, indicating recruitment of shrubs via rhizomes and natural seeding at the units that have had several years to grow. When naturally occurring (non-planted) trees and shrubs were included in analyses, the total live, woody stems (stocking) increased markedly from 2011, ranging from 171% to 422% of the original number of plants installed, with numbers ranging from 3,914 to 16,980 per hectare (Table 19).

Most live stems measured in 2012 were designated 'medium vigor,' while suppression rates dropped to 1% (Table 20). Differences in methods of assessment in 2011 and 2012 may explain lower average suppression and lower plant vigor from 2011 numbers.

Canopy cover (Table 21) increased substantially in all units measured, increasing the most in the SW Quad, and the least at Sundial Island North. The North Bank/South Slough had the highest average DBH, while Mirror Lake had the lowest.

Reed canarygrass was the most abundant herbaceous species across all units except the South Bank/North Slough, found in 30%-90% of plots (Table 22). Other non-native grasses, such as velvetgrass (*Holcus lanatus*), and bentgrass (*Agrostis* sp.) also dominated the herbaceous understory. Another noxious weed managed across the units, Himalayan blackberry, was present in 33% to 85% of plots, but has declined in cover and occurrence on all units since 2008.

Surveys of the effectiveness of bamboo installed in the winter of 2011-2012 show mixed but promising results (Table 23). On both Sundial Island North and the Southwest Quad there were higher rates of vole damage among the cottonwood with bamboo but lower mortality rates in the bamboo-treated plants than among plants without bamboo.

Table 19. Plant survival and stocking by Restoration Unit.

UNIT	Sundial Island North	Southwest Quad	South Bank/North Slough	North Bank Sandy Channel	Mirror Lake	Columbia River Bank	Reference Site
Original number of plants installed per hectare	2010	3840	2150	4610	3444	5241	0
Plots per unit							
2008	50	30	20	50	38		30
2009	50	37	22	50	33		
2010	49	30	20	50	36		
2011	51	31	21	50	47	49	
2012	51	30	20	51	49	50	
Live, installed plants per hectare							
2008	1,228	3,240	1,540	2,588	3,100		0
2009	1,509	2,627	2,086	N/A*	3,362		
2010	2,400	3,246	2,450	N/A*	2,538		
2011	1,541	2,568	2,619	N/A*	1,821	5,269	
2012	1,710	3,527	2,390	N/A*	2,114	N/A*	
Percent installed plant survival							
2008	61%	84%	72%	56%	90%		
2009	75%	68%	97%	N/A*	98%		
2010	119%	86%	114%	N/A*	74%		
2011	77%	67%	122%	N/A*	53%	101%	
2012	85%	92%	111%	N/A*	61%	N/A*	
Total live woody stems per hectare (Stocking)							
2008	1,784	3,367	1,660	2,860	3,100		7,753
2009	2,396	2,795	2,196	3,793	3,396		
2010	2,514	3,453	2,630	4,312	2,591		
2011	3,753	3,265	2,981	4,112	2,128	9,898	
2012	3,914	16,980	4,770	16,337	5,898	12,808	
Percent stocking compared to installed plants							
2008	89%	88%	77%	62%	90%		
2009	119%	73%	102%	82%	99%		
2010	125%	90%	122%	94%	75%		
2011	187%	85%	139%	89%	62%	189%	
2012	195%	442%	222%	354%	171%	244%	
Ratio trees to total woody plants							
2008	75%	37%	75%	64%	51%		11%
2009	60%	33%	88%	33%	40%		
2010	78%	30%	71%	51%	49%		
2011	37%	29%	66%	40%	45%	18%	
2012	32%	6%	40%	14%	13%	22%	
Total trees per hectare							
2012	1263	953	1930	2325	784	2860	

*NOT AVAILABLE: Planted and natural vegetation no longer reliably distinguishable.

Table 20. Plant vigor and suppression

VIGOR	Low	Medium	High
Total live, installed trees and shrubs on surveyed plots, 2012	291	11,810	739
2008 ratio per rating	6%	87%	7%
2009 ratio per rating	2%	87%	11%
2010 ratio per rating	2%	88%	10%
2011 ratio per rating	3%	49%	48%
2012 ratio per rating	2%	92%	6%
SUPPRESSED	**Yes**	**No**	
Total live, installed trees and shrubs on surveyed plots, 2012	69	12,765	
2008 ratio per rating	25%	75%	
2009 ratio per rating	19%	81%	
2010 ratio per rating	26%	74%	
2011 ratio per rating	10%	90%	
2012 ratio per rating	1%	99%	

Table 21. Canopy cover and DBH across units.

UNIT	Sundial Island North	Southwest Quad	South Bank/North Slough	North Bank Sandy Channel	Mirror Lake	Columbia River Bank	Reference Site
% Canopy cover							
2008	31%	18%	17%	42%	12%	*	91%
2012	37%	42%	29%	56%	*	*	
DBH (cm)							
2012	6.2	3.2	3.0	10.3	1.7	*	

* Data not recorded

Table 22. Reed canarygrass and Himalayan blackberry presence and abundance across units.

UNIT	Sundial Island North	Southwest Quad	South Bank/North Slough	North Bank Sandy Channel	Mirror Lake	Columbia River Bank*
Reed Canarygrass						
2008 % presence	35%	83%	50%	56%	68%	NA
2012 % presence	43%	90%	30%	47%	67%	64%
2012 % cover where present	55%	62%	59%	45%	62%	26%
2012 % cover all plots	24%	56%	18%	21%	42%	17%
Himalayan Blackberry						
2008 % presence	90%	53%	90%	78%	68%	
2012 % presence	65%	43%	85%	45%	33%	84%
2012 % cover where present	10%	2%	3%	22%	3%	2%
2012 % cover all plots	1.0%	0.2%	0.6%	4.0%	0.1%	2%

* In CRB, presence/absence and % cover was measured in the 4m radius plots only

Table 23. Comparison of vole damage and mortality of cottonwood between plants with bamboo installed and plants without bamboo.

	No bamboo	Bamboo
Sundial Island North		
% of plants with vole damage	9.2%	48.4%
% mortality	16.3%	9.7%
SW Quad		
% of plants with vole damage	9.8%	13.0%
% mortality	12.2%	8.7%

4.4 Discussion

Trends observed from 2008 to 2012 indicate that planting units are moving towards restoration targets of reestablishment of floodplain forest and scrub habitats and reduction of noxious weed species. High survival rates and natural recruitment have contributed to a significant increase in overall density of native woody plant stems (stocking) measured on all units since monitoring was initiated in 2008 and 2011. Native stem densities as high as 400% of initial planting densities indicate that units are supporting natural recruitment of native woody plants. The large increase in stocking is mostly due to an increase in number of shrub stems, which are spreading via clonal growth. Seedlings of ash, red elderberry, hawthorn, and red-osier dogwood were also observed.

Further indication of success is evident when changes are compared with the reference site. The increasing canopy cover over the last 5 years is one indication that monitored units are developing towards reference site conditions. Most units are far from the 91% canopy closure seen in the reference site, the increase in cover since 2008 shows that that trees planted are surviving and filling in. The tree-to-woody plant ratio is decreasing, also becoming more similar to the reference site.

Sundial Island North (155 acres)

While installed plant survival appears to have fallen to 85% at this unit, trends in total stocking show a marked increase, from 1,228 per ha installed in 2007 to 3,914 per ha native woody stems measured in 2012. Overall stocking relative to initial planting density has increased steadily from 2008 to 2012, now at nearly 200%. Although high, stocking and canopy cover has not increased as much as in other units, perhaps because of the high amount of browse in this unit, at 53% of plants.

Southwest Quad (40 acres)

Originally restored in 2005 and interplanted before monitoring began in 2008, the overall stocking in the Southwest Quad shows a dramatic increase in the number of live stems, at over 440% of the installed number of plants. This change is most likely due to increased production of rhizomatous shoots and natural seedlings by shrubs, as the proportion of shrubs has increased 23%, from 71% to 94% of total stocking. The prevalence of shrubs reflects original project design, where shrubs were installed to accommodate BPA powerline corridors on the unit.

Canopy cover has increased 24% from 2008 to 2012, the greatest increase across all of the units. This is even more notable since over ¼ of the plots were in the powerline corridor, where no trees have been planted. Analyzing these areas separately, shrub stocking was nearly 3 times as dense in the powerline corridor than in the rest of the unit.

The Southwest Quad showed the highest prevalence of reed canarygrass, present in 90% of plots, and with an average cover of 56%. Because much of this unit is underneath the powerline corridor, where no trees can be planted, several years may be required to achieve the canopy cover necessary to shade out this noxious species.

Bamboo appears to have been more effective in this unit. There were 23 cottonwood surveyed that had bamboo installed. Three of these had vole damage (13%), and one of these with vole damage died, for a mortality rate with bamboo of 4.3%.

South Bank/North Slough (20 acres)
South Bank/North Slough shows over 200% woody plant stocking as compared to original restoration planting density. This increase in stocking indicates that natural regeneration is occurring on the unit. This is most likely due to less weed competition as the canopy fills in, and as plantings reproduce through vegetative growth. Stocking numbers were boosted by an interplanting in February 2010, where approx 400 plants per hectare were installed to improve plant densities in understocked portions of the unit. The tree to shrub ratio continued to decline in 2012, while canopy cover increased 12%, indicating progress toward reference site conditions. However, Himalayan blackberry was present in 85% of plots, indicating this unit is highly susceptible to reestablishment of this species.

North Bank Sandy Channel (15 acres)
Originally installed in 2006, this riparian restoration project now shows succession toward target, self-sustaining conditions, as indicated by increasing total woody plant stocking. 2012 showed a marked increase in stocking, a trend attributable to increased native seed production and germination, as well as rhizomatous spread of native trees and shrubs. Natural regeneration has made natural and installed plants indistinguishable in many areas. The low tree-to-shrub ratio (14%) and high canopy cover (56%) indicates recovery of healthy riparian conditions and progress toward reference site conditions.

The unit's border along the current mainstem of the Sandy has been especially hard hit by erosion, and several permanent plot locations appear to have partially or completely eroded into the river. To date, our analysis has used the original unit size, as measured in 2008; calculations with fewer hectares would produce even higher survival rates.

Mirror Lake (29 acres)
Originally planted in 2008, stocking rates were on the decline at Mirror Lake until 2012, which showed a substantial increase in woody plant density. This is due to an increase in shrubs, especially spiraea and swamp rose, via clonal regeneration. Shrubs increased in proportion from 55% in 2011 to 87% in 2012.

Stocking at the Mirror Lake site continues to suffer due to meadow vole, elk and deer activity. Two common tree species, Oregon ash and black cottonwood, showed high levels of browse at 68% and 45% respectively. 17% of ash had vole damage, and 19% of cottonwood had vole damage. Resulting stocking loss is delaying canopy closure at this site. Reed canarygrass cover has increased substantially since 2008, providing ample protection for voles, which are competing with the species that are meant to eventually shade out this noxious weed. Dominance of reed canarygrass, at among the highest levels among the units, has corresponded with a marked decline in herbaceous plant diversity since 2008. In 2008 Mirror Lake had the highest herbaceous species richness per plot, and has lost an average of 11 species per plot to have the 2nd lowest species richness of the units. Figure 41 illustrates that higher cover of reed canarygrass is correlated with lower species richness in this unit.

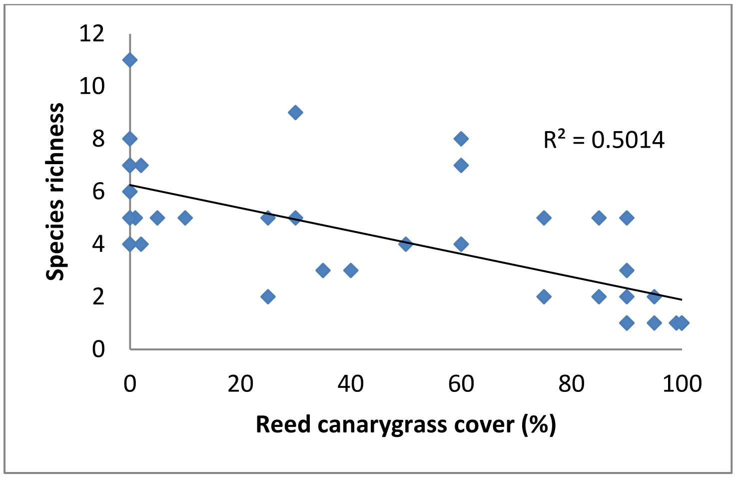


Figure 41**.** The relationship between reed canarygrass cover and richness of other herbaceous species at Mirror Lake in 2012.

Columbia River Bank (140 Acres)

The most recent addition to the AEM program, the Columbia River Bank (CRB), has been planted, interplanted, and managed for plant establishment and weed control for three years beginning in 2009. The second year of monitoring shows very high stocking levels, with over 200% stocking relative to installed plant density. High stocking may be attributed to the intact forest canopy at CRB, which provides favorable growing conditions for trees and shrubs native to the Delta. Thorough site preparation also appears to have supported installed plant establishment and released native plants, seed and rhizomes. The highly variable density of native plants at the Delta also can result in unexpectedly high numbers of natural and installed plants relative to initial planting density at CRB, where it is already difficult to distinguish planted individuals from natural ones. The apparent success on this unit shows how integral both intact forest canopy and thorough weed control are in the reestablishment of well-stocked and naturally reproductive floodplain forests. However, monitoring showed that Himalayan blackberry and reed canarygrass were pervasive this year, present in 84% and 64% of plots, respectively.

4.5 Conclusions

Monitoring in 2012 indicates that intensive site preparation, planting and stewardship treatments are resulting in native plant reestablishment of significant areas of functional Columbia River floodplain forest within the areas surveyed at the Sandy River Delta. Additional treatments to enhance native cover and control the most aggressive weeds continue to be necessary to meet target conditions on all units, but intensity and frequency of treatments on older units continue to decline. Mirror Lake, due to lack of management during the critical plant-establishment period, is supporting robust shrub reproduction, but is showing low plant survival, high weed cover and loss to animal damage.

The objective of restoration at the Delta and Mirror Lake is to move project sites toward reference site conditions, characterized by a closed canopy of mature native trees and a dense understory of native shrubs and herbaceous plants. Natural recruitment at units monitored has led to dramatic increases in numbers of shrubs. Increases in canopy cover on all units except Mirror Lake, demonstrate development of forest canopy towards target conditions. The presence of invasive species, however, while not currently reducing native plant vigor, should be monitored and controlled to prevent reestablishment to levels that inhibit native woody plant recruitment and growth.

It has been widely observed among restoration practitioners, where unmanaged, noxious weeds can dominate plant communities, forming a single-species mat that replaces native understory plants and inhibits natural recruitment of native woody species. Over time associated ecosystem functions critical to native fish and wildlife species decline, such as production of large wood, shade and prey, bank stabilization and in-stream complexity.

As closed forest canopy conditions develop on restoration sites, annual maintenance costs for plant establishment decline. As forest canopies increase, noxious weeds will be inhibited by shade and the diminishing presence of noxious weeds will continue to improve conditions for native plants to regenerate. These trends are apparent at the well-established restoration sites at the Delta, particularly one first planted in 2002, where a closed canopy of 60' tall cottonwood and ash shade an understory of planted and natural shrubs with volunteer fern, forbs and sedges. New tree recruitment is inhibited in these highly stocked stands, and regeneration is expected to occur in areas where tree-fall or flooding creates openings for seedling establishment.

Absent an invasive weed source, a disturbance that opens a well-developed canopy can result in recruitment of native plants, including canopy forming trees. Landscape-scale restoration, such as the almost 800 acres under management at the Sandy River Delta, has reduced local seed source, thereby creating favorable conditions at the Delta for the critical ecosystem functions of long-term canopy formation, large wood recruitment and shade production.

The scale and inter-connectedness of the Sandy River Delta and Mirror Lake restoration projects sets this work apart from other restoration efforts in the region. Entirely within the floodplain of the Columbia River, these sites encompass numerous stream banks, riparian areas, off-channel ponds, sloughs, and other habitats critical to sustaining healthy aquatic life. These sites offer the opportunity to manage healthy native vegetation over the remainder of this expansive, complex area - and to chart a course for management of similar areas in the estuary.

4.6 Recommendations

Low numbers of suppressed plants suggests that, while noxious weeds are still present on site, management treatments have been successful in controlling weeds that would otherwise overtop and slow or prevent native plant reestablishment. Himalayan blackberry has decreased in prevalence across all units from 2008 to 2012, indicating that weed control treatments have reduced this species. Blackberry is still present in 33%-84% of plots, however, and though generally low in cover this year where found, the common occurrence of this species is cause for concern. Reed canarygrass has decreased in prevalence across some units, but has increased in others, most notably at Sundial Island North and the Southwest Quad.

Invasive weeds like Himalayan blackberry and reed canarygrass are observed to reduce biological diversity and inhibit critical ecosystem functions, such as natural regeneration of native plant communities and associated large wood, shade production and soil stabilization. Analysis of our data from 2012 shows that as reed canarygrass cover increases, the richness of other herbaceous species declines (Figure 41). These observations suggest that continued maintenance and stewardship of all units is needed to achieve the goal of restoring naturally reproductive Columbia River floodplain forest and scrub. Additional vegetation management treatments are indicated for all units, shown in Table 24.

Sundial Island North (155 acres)
Blackberry spot spray is recommended until the unit achieves canopy closure. Fifteen acres in the easternmost portion of the unit are poorly stocked due to severe browse and weed competition and should be interplanted to ensure full canopy development.

Southwest Quad (40 acres)
Interplanting is recommended to boost unmaintained native plant populations. Himalayan blackberry spot spraying is recommended across the entire unit.

South Bank/North Slough (20 acres) & North Bank Sandy Channel (15 acres)
Only minimal Himalayan blackberry control effort is required on a biennial basis at this time.

Columbia River Bank (140 Acres)
High priority treatments at this unit are interplanting in understocked areas and continuing annual weed control treatments to keep Himalayan blackberry and reed canarygrass in check.

Mirror Lake (29 acres)
Due to regulatory and funding limitations, there has not been sufficient maintenance of this unit during plant establishment, which has resulted in lower success of the plantings at Mirror Lake. Significant areas of the unit are now understocked and dominated by weeds. More site preparation, intensive planting and several years of plant establishment maintenance are recommended to establish canopy on this site. The focus should be on tree species planting and maintenance, until canopy closure is achieved. Experimental measures to deter animal damage - such as bamboo staking to control vole damage, as tested at the Sandy River Delta - might prove beneficial for new plantings as well as plants previously installed on the Mirror Lake unit.

Table 24. Recommended 2013 maintenance treatments.

Task	Treatment Date	Sundial Island N	SW Quad	S Bank N Slough	N Bank Sandy Channel	Mirror Lake	CRB
Interplanting	2/1/2013		X			X	X
RCG and blackberry spot spray	5/1/2013					X	X
Mow	8/1/2013	X				X	
Blackberry spot spray	9/1/2013	X	X	X	X		X

5 Salmon, Salmon Prey, and Habitat AEM at Fort Clatsop South Slough & Alder Creek

In 2007 the Estuary Partnership and its partners replaced a failing tide gate with a bridge at the Lewis and Clark National Historical Parks' Fort Clatsop in order to reconnect South Slough (and 45 acres of diked pastureland) with the tidal influence of the Columbia River. The pre-existing culvert was inadequate to handle the volume of water inside the slough. Water was impounded during tidal fluctuations, increasing water velocity, delaying tidal response, and potentially limiting natural fluvial processes.

In 2007 the Columbia River Estuary Study Taskforce (CREST) implemented pre-project monitoring as a baseline for characterizing fish community assemblages, size class, and residency; and for water quality conditions including temperature, tidal range/depth, dissolved oxygen and conductivity. CREST performed effectiveness monitoring from 2008 - 2012 after restoration actions were complete as part of the Estuary Partnership's Action Effectiveness Monitoring Program. Permit limitations in 2012 prevented CREST from sampling the fish community but channel morphology, prey availability, and water quality were monitored. A Passive Integrated Transponder (PIT) tag array was installed in 2012 inside South Slough, and the adjacent elevated wetland underwent wetland enhancement in accordance with the in-water work period.

The objectives for this project are:

- Characterize fish species composition, size class, and spatial and temporal distribution in South Slough (restoration site) and the reference site, Alder Creek.
- Measure and characterize water quality parameters that may be limiting factors or limit habitat opportunity to the site for salmonids.
- Measure changes in sediment movement (erosion/aggradation) and landscape changes (channel morphology/riparian vegetation).
- Estimate terrestrial macroinvertebrate prey availability and prey selectivity.

5.1 Sample Site Descriptions

South Slough resides between steep hillsides and the mainstem Lewis and Clark River (Figure 42). A year round freshwater input drains off the hillsides into the slough and its adjacent wetlands. The wetland on the south side of the channel was altered by the placement of fill, elevating this side a couple of feet higher than the wetland on the northern bank of the slough. The site has been actively grazed with no restrictions from the riparian zone or slough. The difference in elevation on the southern wetland has resulted in differences in plant community, land use, and both inundation frequency and duration between the two wetlands adjacent to the slough. The plant community on the south side is dominated by non-native pasture grass and common rush (*Juncus balticus*), while to the north the wetland is dominated by native plants such as small fruited bull rush (*Scirpus microcarpus*), douglas spirea (*Spirea douglasii*), slough sedge (*Carex obnupta*), and soft rush (*Juncus effuses*).

The amount and diversity of accessible estuarine habitat impacts the abundance and productivity of ESA listed salmon populations in the Columbia River Estuary (Fresh et al. 2005). Prior to restoration an undersized culvert connected South Slough to the Lewis & Clark River mainstem. The culvert was located under the National Park Highway. The roadway and culvert posed an obstacle ("Passage/Flow Barrier" stressor) to fish passage and affected four controlling factors within South Slough: sediment, hydrodynamics, bathymetry/topography, and water quality. Reconnecting South Slough to tidal influence increases habitat opportunity for juvenile life stages of salmon migrating through the estuary. The landscape position in the Columbia River Basin, the size, and the diversity of habitats within the South Slough project site should contribute to the diversity of life history strategies of ESA listed salmon populations in the LCRE.

In August of 2012 wetland enhancements began in the southern area of the slough. Additional channels were excavated and large woody debris was added to increase habitat opportunity for ESA listed salmon observed on site. The remaining floodplain was lowered in some areas and raised in others to provide a diversity of habitat types. The area will continue to provide habitat for grazing elk (which are commonly observed in the early morning), resident and migratory song birds, waterfowl, amphibians, and fish species.

Alder Creek was selected as a reference site for South Slough based on its similar landscape position, tidal influence, and habitat characteristics. Alder Creek, located less than half a mile downriver from South Slough, was naturally breached around 50 years ago, allowing ample time to undergo the successional change resulting from tidal reconnection.

While a number of similarities exist between South Slough and Alder Creek, some key differences are present. South Slough resides at the toe-slope of a sub watershed within the larger Young's Bay Watershed, providing constant freshwater input that maintains water levels of at least 2 feet during low tide events. Alder Creek connects to the northern end of the wetland adjacent to South Slough through a small culvert. The freshwater input is impounded in comparison and the elevation of the channel result in 6-12 inches of water during low tides. These key differences are believed to impact habitat opportunity, fish community distribution, and water quality.

Figure 42. Ft Clatsop South Slough restoration site, and the reference site Alder Creek

5.2 Methods

The AEM program at South Slough was established as a Before-After-Control-Impact design (BACI) in order to quantify the effects of restoration efforts. Since 2007 CREST has monitored biological, chemical, and physical parameters at South Slough and Alder Creek (reference site) in order to observe the response of these parameters to the tidegate removal. A synthesis of monitoring results was conducted for data collected from 2007 to 2011 from South Slough and Alder Creek. When the synthesis was conducted, it was uncertain if monitoring would continue in 2012 at South Slough. Data was collected in 2012 and is presented, but is not included in the synthesis. Alder Creek was selected as a reference site based on its landscape position, tidal connectivity, and environmental conditions that could support juvenile salmonids; all of which are similar to South Slough.

In 2007 pre-project monitoring was conducted at South Slough and Alder Creek; parameters included fish community, landscape changes, and water quality. From 2008 - 2011 post-project monitoring was implemented at South Slough and Alder Creek. In 2012 monitoring include landscape changes and water quality, but excluded fish community due to a lack of fish collection permits. Metrics post-project included fish community, prey availability & selectivity, landscape changes, and water quality. All sample gear and fishing techniques were consistent with the methods described in "*Monitoring Protocols for Salmon Habitat Restoration Projects in Lower Columbia River and Estuary*" (Roegner et al. 2009).

5.2.1 Fish Community

Between 2007 and 2011, fishing methods and fishing gear were changed between South Slough and Alder Creek to improve overall fish capture. As a result of changes in methods, the degree of analysis possible between the two sites is limited. However, fish community composition, timing, size class, invertebrate prey availability and selectivity, landscape changes and water quality parameters can be directly compared.

In 2007, pre-restoration, a trap net with sanctuary bag was used at both South Slough and Alder Creek. Wing nets were used to funnel fish into the trap net and sanctuary bag. This method was more successful at Sough Slough than Alder Creek. Low water velocity at Alder Creek caused the bag to collapse in on itself, and fish were observed swimming out of the trap.

In 2008 the same trap net system was used at South Slough while a beach seine was implemented in place of the trap net at Alder Creek. Seining was tested as a method in order to find the most suitable fish capture technique for the specific site conditions at Alder Creek. A small boat pulled the net using a 3 horsepower motor. This method seemed to be more effective as it resulted in slightly larger fish catches. Access with a small, motorized boat was very difficult, however.

In 2009 and 2010, a modified trap net system was utilized at both sites to establish consistency in methods, while also utilizing the most effective fish capture and handling techniques possible. Wing nets were used to passively direct fish into a 5ft wide by 7ft tall, 20ft long sock net with a livebox attached instead of an open sanctuary bag. The method has remained the same at both sites since 2009. The livebox has appeared to discourage escapement and reduced stress to fish while holding.

In 2011 the methods were further improved by seining down the channel into the trap before the trap was pulled at the end of the sampling event. This was done at both South Slough and Alder Creek. A pole seine was stretched across the channel above the trap net, 102 meters at the restoration site and 92 meters at Alder Creek, and walked down to the mouth of the trap, encouraging fish into the net. The distance the channel was seined down depended on channel morphology, water depth, and large woody debris in the channel. Alder Creek drains faster than South Slough consequently fish from this site were processed first to avoid releasing the catch into unfavorable water quality conditions.

After capture, dip nets were used to transfer fish from the livebox or seine into black buckets. The water conditions inside the buckets were maintained near stream conditions, particularly in regards to temperature and dissolved oxygen (DO). Portable aerators were used to maintain DO levels. All non-salmonid fish were identified to species, with the first 30 measured and the remainder counted. Salmon were separated from the other fish species and processed first. All juvenile salmonids were measured and weighed, checked for tags and markings, then allowed to recover before they were released back into the stream. .

The different methods applied need to be taken into account when comparing data, particularly between South Slough and Alder Creek during 2008. While timing, size class, and species composition can still be compared between years and sites, the addition of seining down the channel in 2011 does bias density data for 2011.

5.2.2 PIT tag array

Passive Integrated Transponder (PIT) tag arrays are stationary antennas capable of sampling the entire width of culverts, streams, spillways, or fish ladders. In January of 2012 a PIT tag array was installed at South Slough. The array consists of two 20 ft x 4 ft antennas positioned near the bridge within South Slough. The antennas are positioned to ensure full coverage of the channel under normal (tides exceeding 8ft open a window of zero detection near the water surface above the array) tidal patterns. The antennas are powered and operated through the use of a multiplexing transceiver system, the MUX FS1001M. The transceiver is capable of powering up to six antennas, providing ample opportunity to monitor South Slough. The system requires implanting fish with a small PIT tag. Tagging procedures follow strict guidelines established by the Columbia Basin Fish and Wildlife Authority's PIT tag markings procedure manual. Each individual tag contains a code specific to each individual fish. Once a PIT-tagged fish swims through or in the nearby vicinity of an antenna the PIT tag number is detected by the antenna receiver which then records and stores the date, time of passage and PIT tag number unique to each individual fish. All anadromous PIT tag data detected by the South Slough array are entered and stored in the PIT Tag Information System (PTAGIS) data repository.

5.2.3 Genetic Analysis

To better understand fish usage between stocks caudal clips were taken for genetic analysis. Pre-labeled vials were loaded with non-denatured 95% ethanol for individual samples. Scissors were used to take the tip of the upper caudal fin and insert it into the prepared vial. Each sample was given an ID# that correlated to a particular fish. When necessary to collect samples, 1 to 2 salmon at a time were anesthetized in a buffered tricaine methanesulfonate (MS222) solution. Samples were sent to NOAA's NMFS Northwest Fisheries Science Center, Manchester Facility, for genetic analysis.

5.2.4 Prey Availability & Salmon Diet

Insect fall-out traps were utilized to evaluate the availability of terrestrial macroinvertebrates as a food source for juvenile salmonids at South Slough and Alder Creek. Five traps were placed at both South Slough and Alder Creek. Fall-out traps were made of rectangular plastic tubs secured loosely by string to three PVC pipes, and filled with an inch of soapy water. The set up was designed so that the traps could rise and fall with tidal influence. Tubs were placed on the bank near the trap net sites. The traps work by disrupting the flight ability of insects that land on the surface. Samples were collected after 48 hours and preserved in 90 percent Ethanol for lab analysis.

In conjunction with fall out traps, benthic core samples were taken to identify the benthic component of prey availability at both sites. These samples were collected using 2 inch diameter PVC pipe. One end is inserted approximately 4 inches into the sediment of the channel at or near low tide, and a rubber stopper is placed on the other end of the pipe creating a vacuum suction used to contain the sample in the pipe while it was removed from the substrate. Five samples were collected, each adjacent to a fall out trap. The samples were rinsed through a 500 micro millimeter mesh sieve and preserved in individual plastic jars with 95 percent Ethanol. Rose bengal, an inert stain, was applied to facilitate sorting invertebrates from other debris in the sample.

Macroinvertebrate data provides insight into how and when the site provides forage opportunities for juvenile salmonids. Five samples were collected monthly during March through July. In August construction activities inhibited sampling due to dramatic changes in vegetation onsite (site was mowed prior to excavation). Samples were not collected in August from the reference site as there was no comparison available from the restoration site.

5.2.5 *Sediment Accretion Stakes*

Reconnecting a site to tidal influence restores natural processes such as sediment and nutrient transport. Sediment accretion stakes allow for the simple measurement of sediment aggradation and erosion as a result of hydrologic re-connection. Sediment accretion stakes were installed at South Slough in 2008. In 2009 Pacific Northwest National Laboratories (PNNL) installed sediment accretion stakes at Alder Creek in the same manner as described above. Sediment accretion measurements are taken at both sites to measure the changes in soil erosion and/or deposition along the bank, and to compare the changes in sediment transport between South Slough and Alder Creek. Two level stakes were placed in an area adjacent to the channel where inundation was expected to occur, and set one meter apart. The stakes are leveled, a meter stick placed on top of both, and the distance measured from the meter stick to the ground at 10 cm intervals. Measurements were taken twice during the sampling season at both South Slough and Alder Creek to cumulatively reveal temporal and spatial shifts in sediment distribution.

5.2.6 *Channel Morphology*

Channel cross-sections were used to record changes in channel morphology as a product of changes in hydrology resulting from tidal reconnection. Cross sections consisted of 5 transects at each site that extended from bank to bank across the channel. The start and end points were marked with PVC and GPS coordinates recorded (Table 25). Start and end points were set back far enough on the bank so that they would not be lost to erosion as the channel changed over time. An auto-level and stadia rod was used to measure the elevation at set intervals. Intervals were selected based on site topography and the degree of change across the channel that is anticipated.

Table 25. Channel cross section GPS coordinates for South Slough and Alder Creek, 2012.

Site	Cross section 1	Cross section 2	Cross section 3	Cross section 4	Cross section 5
South Slough	N46°7’44.6” W123°52’46.7” N46°7’43.8” W123°52’47.6”	N46°7’43.7” W123°52’43.6”, N46°7’44.0” W123°52’43.0”	N46°7’44.7” W123°52’50.2”, N46°7’43.7” W123°52’50.7”	N46°7’44.3” W123°52’52.0”, N46°7’43.9” W123°52’51.0”	N46°7’43.1” W123°52’52.2”, N46°7’43.1” W123°52’52.2”
Alder Creek	N46° 7' 53.472" W123° 52' 44.544"	N46° 7' 53.7594" W123° 52' 44.4354"	N46° 7' 54.264" 123° 52' 44.7594"	N46° 7' 55.0554" W123° 52' 44.6514"	N46° 7' 56.5674" W123° 52' 44.58"

5.2.7 *Water Quality*

A multi-parameter TROLL 9500 Professional series water quality probe was installed at South Slough in 2007 and was used through the 2012 monitoring season. The probe collected data year round at one-hour intervals. The same brand and model of probe was installed at Alder Creek in 2009. In 2009 the Clark dissolved oxygen sensors on the probes were upgraded to an optical (RDO) sensor in order to reduce drift in data and decrease maintenance. The original water quality meter was deployed approximately 75 ft. upstream of the bridge, at N46°7’43.4” W123°52’42.9”, N46°7’43.7” W123°52’43.2”. As a result of theft in 2009, a new probe was deployed upstream of the trap net site and approximately 300m upstream of the bridge. The probe at South Slough is currently located at N45°51’53.2” W122°44’43.2.

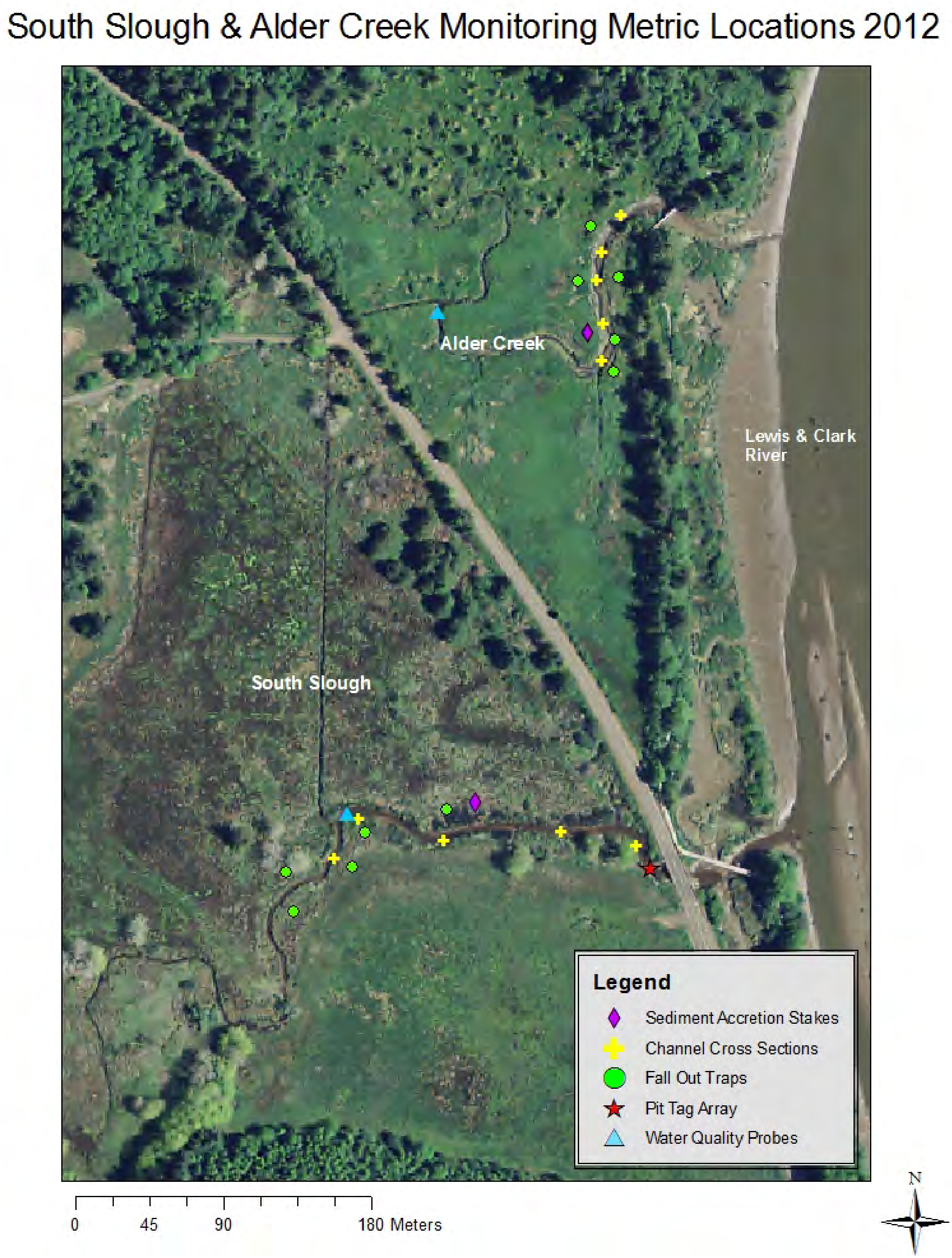


Figure 43. Monitoring metrics at South Slough and Alder Creek, 2012.

5.3 Results and Discussion

5.3.1 Prey Availability & Salmon Diet 2012

South Slough and Alder Creek had 43 species present in fall out traps over the course of the monitoring period (Table 26). The number of different prey species present demonstrated similar patterns during the sampling season at both South Slough and Alder Creek. Overall, species diversity increased as sampling season progressed. Species richness, expressed as the cumulative count of all individual macroinvertebrates, similarly increased in the last two months of the sampling season. These patterns are consistent with previous year's data, showing an increase in both diversity and abundance as the seasons transitioned from spring to summer. Terrestrial macroinvertebrate species assemblages demonstrate temporal variation with peak abundances and diversity in accordance with periods of high plant growth and abundance.

Fall out trap observations at South Slough revealed a gradual growth trend in both species richness and diversity throughout the 2012 monitoring season (Figure 44). In March a total of 27 macroinvertebrates were captured representing seven species. Four months later in July the same fall out trap locations yielded 836 individual macroinvertebrates representing 33 species. Alder Creek fall out trap samples also reflected a similar growth trend in macroinvertebrate species richness and diversity over the course of the sampling season.

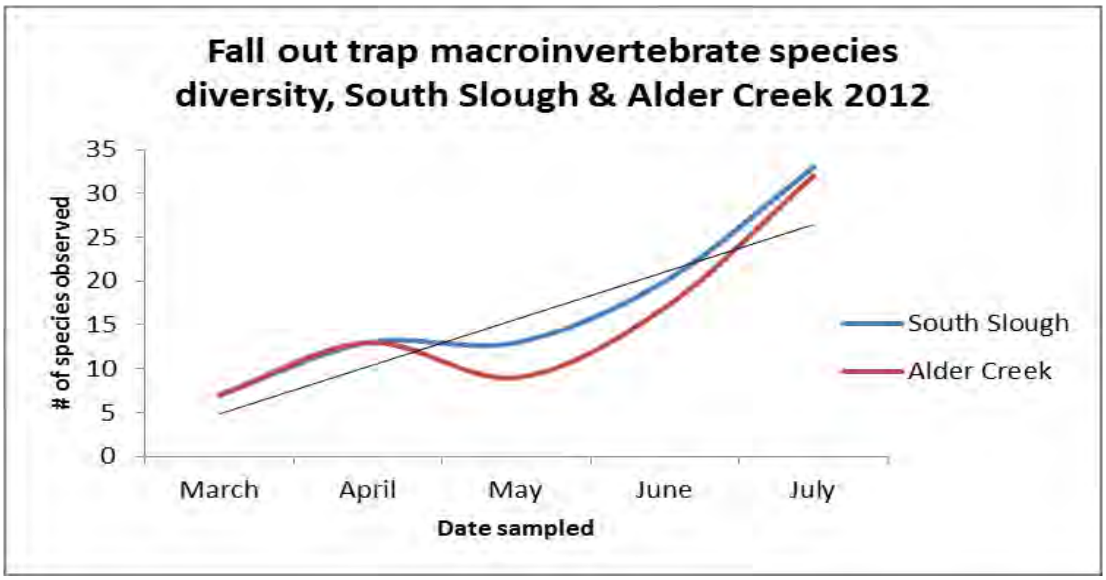


Figure 44. Temporal variation in macroinvertebrate species observed in fall out traps sampled at South Slough and Alder Creek, 2012.

The cumulative macroinvertebrate community observed between the two monitoring sites transitioned from 11 to 39 species from March to July in 2012. As a result, few species were found to be consistently present at both sites throughout the entirety of monitoring season. Chironomidae were the only species regularly observed at both sites during every month fall out trap sampling occurred. Sciaridae were observed at Alder Creek each month sampling occurred, but were not found in South Slough samples in March. Many other species were observed (Table 26), but few were regularly present throughout the season. The month of July represented the most productive month in terms of both diversity and abundance with 2,136 individual invertebrates captured in fall out traps belonging to 39 species. In June and July several additional species from the order Trichoptera and Diptera were found to be present in fall out trap samples, resulting in a noticeable increase in species richness and diversity at each of the two monitoring sites.

Table 26**.** Macroinvertebrate species observed in fall out traps at South Slough and Alder Creek, 2012. (*parenthesis represent the percentage of each taxa observed for each month relative to site sampled*)

TAXA	**MONTH**									
	MARCH		**APRIL**		**MAY**		**JUNE**		**JULY**	
	South Slough	Alder Creek	South Slough	Alder Creek	South Slough	Alder Creek	South Slough	Alder Creek	South Slough	Alder Creek
Acari							7 (5.3%)	14 (11%)	4 (<1%)	3 (<1%)
Acrididae									1 (<1%)	
Agromyzidae									1 (<1%)	1 (<1%)
Aphidoidea							7 (5.3%)	14 (11%)	95 (11.3%)	18 (1.4%)
Apoidea									1 (<1%)	
Araneae				2 (3%)	6 (10.7%)	2 (3%)	18 (13.7%)	5 (3.9%)	13 (1.5%)	9 (<1%)
Bibionidae										1 (<1%)
Braconidae				1 (1.4%)			8 (6.1%)		27 (3.2%)	2 (<1%)
Cecidomyiidae			1 (3%)	1 (1.4%)	4 (7%)	1 (1.5%)		7 (5.5%)	11 (1%)	18 (1.4%)
Ceratopogonidae				1 (1.4%)					4 (<1%)	17 (1.3%)
Chalcoidea						1 (1.5%)	5 (3.8%)	1 (<1%)	18 (2%)	43 (3%)
Chironomidae	7 (26%)	40 (40%)	11 (31%)	21 (29%)	26 (46%)	20 (30.7%)	43 (32.8%)	13 (10%)	160 (12%)	309 (24%)
Chloropidae	3 (11%)	2 (2%)	1 (3%)				5 (3.8%)		2 (<1%)	12 (<1%)
Cicadellidae			1 (3%)	8 (11%)	2 (4%)	28 (43%)	9 (6.8%)	41 (32%)	67 (8%)	76 (6%)
Coccinellidae									1 (<1%)	
Coleoptera	1 (4%)		3 (8%)		1 (2%)		4 (3%)	5 (3.9%)	4 (<1%)	9 (<1%)
Curculionidae					1 (2%)	2 (3%)				3 (<1%)
Cynipodea				1 (1.4%)						
Delphacidae										5 (<1%)
Dolichopodidae			1 (3%)				2 (1.5%)	9 (7%)	96 (11,5 %)	359 (28%)
Ephydridae	3 (11%)		1 (3%)					1 (<1%)	188 (22%)	227 (17%)
Gastropoda							2 (1.5%)	1 (<1%)	5 (<1%)	
Hemiptera									5 (<1%)	3 (<1%)
Ichneumonidae		4 (4%)	1 (3%)							5 (<1%)
Isopoda	2 (7%)		2 (6%)							

Lonchopteridae					1 (2%)	2 (3%)	1 (<1%)			
Muscidae										1 (<1%)
Mycetophilidae										1 (<1%)
Mymaridae				7 (10%)	3 (5%)	7 (10.8%)	2 (1.5%)	10 (7.9%)	9 (<1%)	97 (7.5%)
Odonata									1 (<1%)	
Phoridae		1 (1%)	1 (3%)	1 (1.4%)				2 (1.6%)	17 (2%)	3 (<1%)
Psychodidae	10 (37%)	49 (49%)	11 (31%)	23 (32%)			7 (5.3%)	1 (<1%)	36 (4%)	21 (1.6%)
Ptychopteridae					3 (5%)		2 (1.5%)		9 (<1%)	1 (<1%)
Sciaridae		1 (1%)	1 (3%)	4 (6%)	1 (2%)	2 (3%)	1 (<1%)	1 (<1%)	17 (2%)	8 (<1%)
Sciomyzidae	1 (4%)									4 (<1%)
Sphaeroceridae		3 (3%)			1 (2%)		3 (2.3%)		6 (<1%)	1 (<1%)
Staphylinidae			1 (3%)		3 (5%)		1 (<1%)		8 (<1%)	2 (<1%)
Syrphidae								1 (<1%)	3 (<1%)	15 (1.2%)
Tachinidae									3 (<1%)	4 (<1%)
Tenebrionidae					4 (7%)		2 (1.5%)			
Thysanoptera				1 (1.4%)			2 (1.5%)	1 (<1%)	9 (<1%)	15 (1.2%)
Tipulidae				1 (1.4%)					10 (1%)	7 (<1%)
Trichoptera									5 (<1%)	
Total number of individuals	**27**	**100**	**36**	**72**	**56**	**65**	**131**	**127**	**836**	**1300**

Observations of macroinvertebrates in fall out traps demonstrated similar trends in both species diversity and abundance at South Slough and Alder Creek. Conversely, benthic prey samples at Alder Creek revealed a far more prevalent and consistently abundant benthic invertebrate community than that observed at South Slough. Benthic core samples collected from Alder Creek in 2012 revealed 1231 individual invertebrates representing 16 species, while South Slough samples had 1145 individuals representing 13 species. Nematodes and Oligochaetes were the most abundant prey species in benthic core samples and were observed at both sites every month sampling occurred (Table 27).

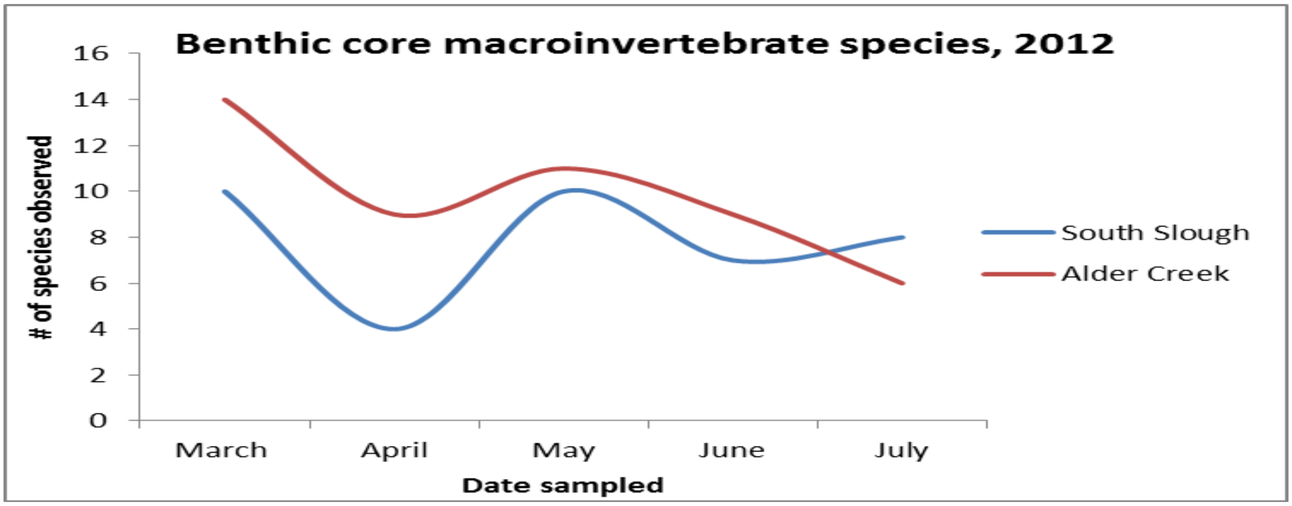


Figure 45. Temporal variation in macroinvertebrate species observed in benthic core samples at South Slough and Alder Creek, 2012**.**

Table 27. Macroinvertebrates observed in benthic core samples at South Slough and Alder Creek, 2012. (*parenthesis represent the percentage of each taxa observed for each month relative to site sampled*)

Taxa	March		April		May		June		July	
	South Slough	Alder Creek	South Slough	Alder Creek	South Slough	Alder Creek	South Slough	Alder Creek	South Slough	Alder Creek
Amphipoda		18 (3.8%)								
Anisogammaridae					3 (1%)					
Bivalvia				2 (1.2%)			3 (1.6%)		1 (<1%)	1 (<1%)
Ceratopogonidae	17 (5.4%)	8 (1.7%)	3 (2%)	3 (1.8%)	5 (1.5%)	21 (7.5%)	7 (3.8%)	5 (3.3%)		
Chironomidae	11 (3.5%)	10 (2.1%)		3 (1.8%)	5 (1.5%)	5 (1.7%)	8 (4.3%)	2 (1.3%)		
Cicadellidae										1 (<1%)
Corophium		7 (1.5%)		4 (2.5%)		1 (<1%)				
Eogammarus		1 (<1%)								
Gammaridae	2 (<1%)	3 (<1%)		4 (2.5%)	1 (<1%)	4 (1.4%)		3 (2%)		2 (<1%)
Gastropoda	145 (46.6%)	174 (37.4%)	81 (54%)	79 (48.7%)	129 (40.9%)	114 (40.5%)	75 (41.2%)	63 (42.2%)	17 (9%)	
Hyalla Sp.		1 (<1%)								
Isopoda	5 (1.6%)	8 (1.7%)			2 (<1%)	8 (2.8%)		2 (1.3%)	1 (<1%)	
Nematoda	36 (11.5%)	23 (4.9%)	8 (5.3%)	4 (2.5%)	14 (4.4%)	24 (8.5%)	22 (12%)	3 (2%)	19 (10.1%)	22 (12.6%)
Oligochaeta	89 (28.6%)	196 (42.1%)	58 (38.6%)	60 (37%)	154 (48.8%)	101 (35.9%)	66 (36.2%)	68 (45.6%)	128 (68.4%)	126 (72.4%)
Pelecypoda	1 (<1%)				1 (<1%)				1 (<1%)	
Polychaeta	1 (<1%)					1 (<1%)		1 (<1%)	10 (5.3%)	22 (12.6%)
Psychodidae		1 (<1%)								
Tipulidae		8 (1.7%)		3 (1.8%)		1 (<1%)	1 (<1%)		10 (5.3%)	
Trichoptera	1 (<1%)	7 (1.5%)			1 (<1%)	1 (<1%)		2 (<1%)		
Total	**311**	**465**	**150**	**162**	**315**	**281**	**182**	**149**	**187**	**174**

5.3.2 *Prey availability & Selectivity 2008-2011*

Prey availability samples were collected to coincide with fish community sampling events from 2008-2011. Five samples were collected monthly for both fall out traps and benthic cores. Species diversity peaked in May and abundance in July. Prey availability may be interpreted as a reflection of the production capacity of an area, and a resource for juvenile salmon as it likely influences their distribution. Stream type Chinook and coho that spend their first year in freshwater systems may benefit from the availability of invertebrates such as chironomids, which are one of the most abundant invertebrate species captured at South Slough and Alder Creek. Future research will attempt to include abundance of aquatic invertebrates in the water column (nueston net sampling) per a set area in order to quantitatively measure the actual productivity.

Table 15. Proportions of samples collected from marked and unmarked juvenile Chinook diets at South Slough (SS) and Alder Creek (AC), 2007 – 2011. **No samples were collected in 2007.*

Prey Utilization; Juvenile Chinook Salmon Gut Contents								
	2008		**2009**		**2010**		**2011**	
*Year	**SS**	**AC**	**SS**	**AC**	**SS**	**AC**	**SS**	**AC**
# of samples	2	0	3	0	3	1	2	8
# of Species	4	0	6	0	10	5	2	6
Taxa	Mean proportion by taxa							
Amphipod								1.70
Anisogammeridae					5.56			
Aphidoidea					5.56	33.33		
Chironomidae			30.43		2.78	22.22	11.63	7.50
Chloropidae					2.78			
Cicadellidae								0.80
Coleoptera								1.70
Corophium	29.00					33.33		1.70
Diptera spp.	5.90					11.11		
Formicidae			4.35					
Gastrapoda					2.78	11.11		
Gammaridae								
Isopoda	58.0		8.69		80.56		88.37	85.80
Mysidacea			4.35					
Nematoda			21.74					
Polycheata	5.90							
Ptychopteridae			30.43					

Diet samples from Chinook and coho were examined and analyzed at both sites. The diet samples between marked and unmarked Chinook, and between marked and unmarked coho were similar, and thus combined in order to analyze the largest group of samples possible. Percentages of individual taxa were calculated in relation to total taxa selected as prey, and the Index of Relative Importance (IRI) calculated for individual salmonid species by site. Prey availability was measured for individual taxa in relation to total taxa as well. Chinook at both sites demonstrated a preference for isopod, corophium, and chironomid species. These species had the highest percentages for taxa consumed by Chinook, as well as the highest IRI value. IRI values are based on percentages of frequency and weight of individual taxa in relation to sample totals. Oligocheats were selected less often, yet had a heavier weight and as a result had a similar IRI value to chironomids for coho at Alder Creek.

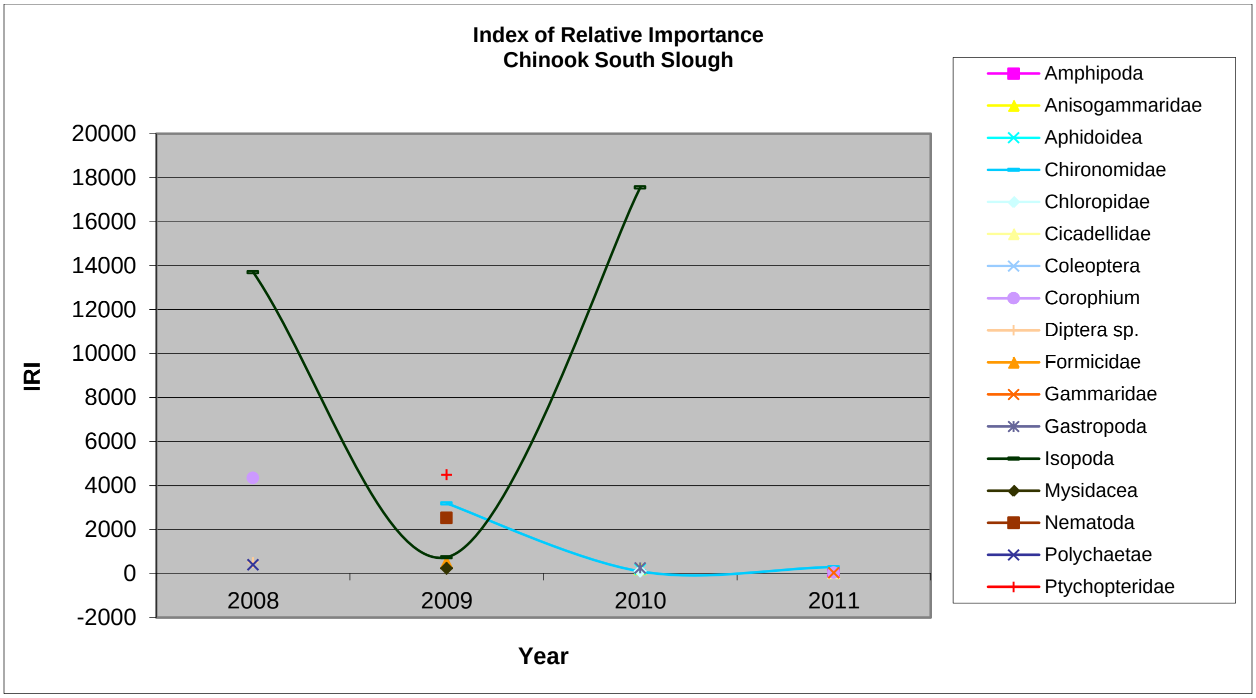


Figure 28. IRI for all Chinook at South Slough, 2008 – 2011.

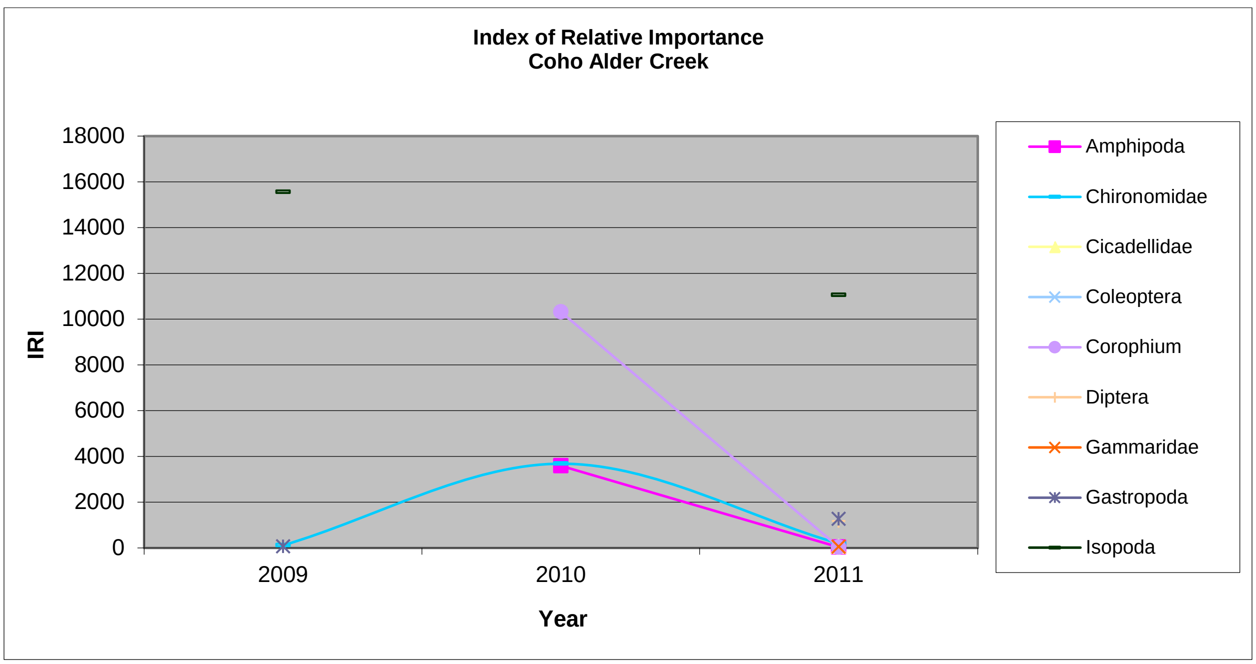


Figure 29. IRI for all Chinook at Alder Creek, 2009 – 2011. **No samples collected in 2008.*

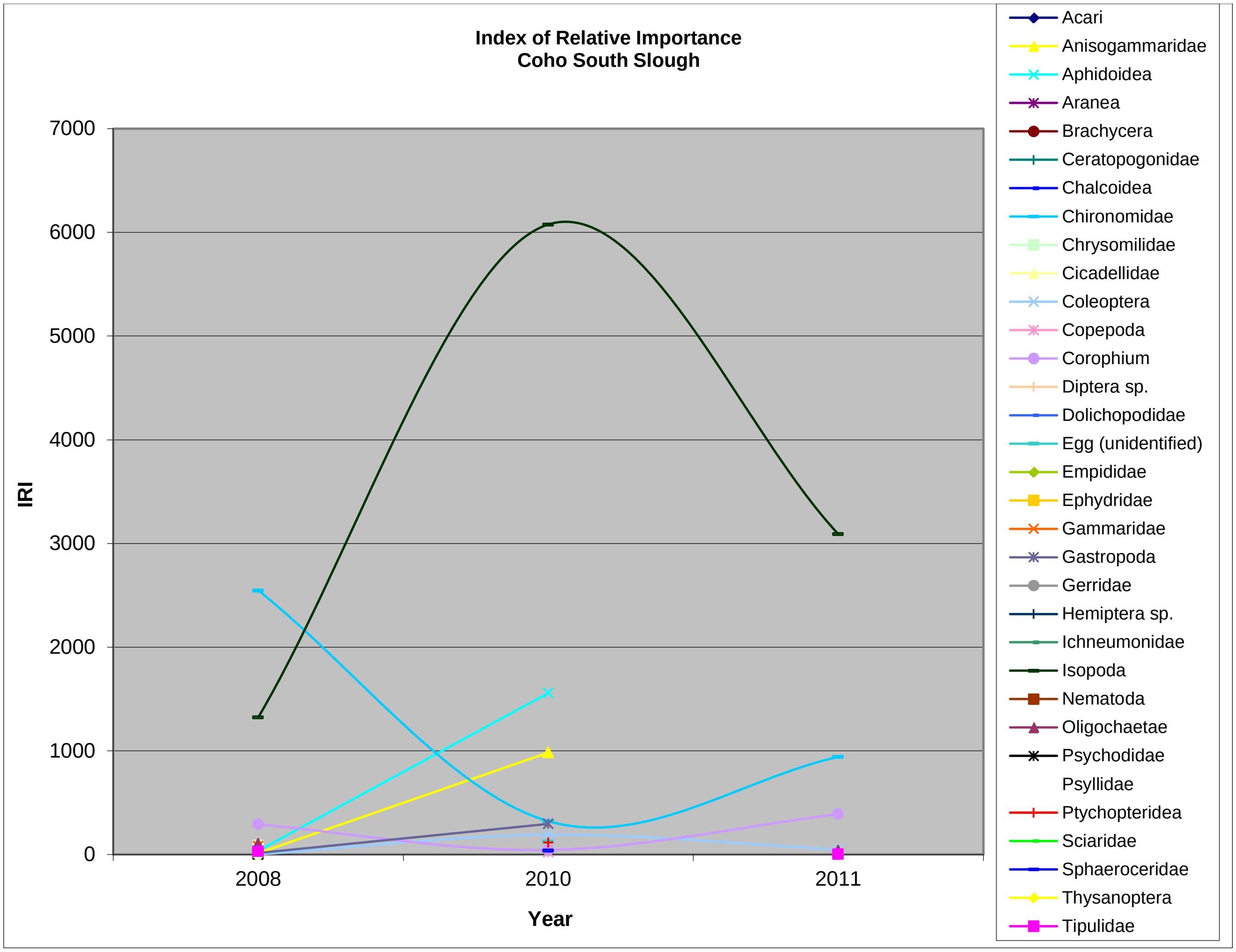


Figure 30. Index of Relative Importance for Coho at South Slough, 2008 – 2010. *No coho diets were collected in 2009 on account of the size of fish caught.

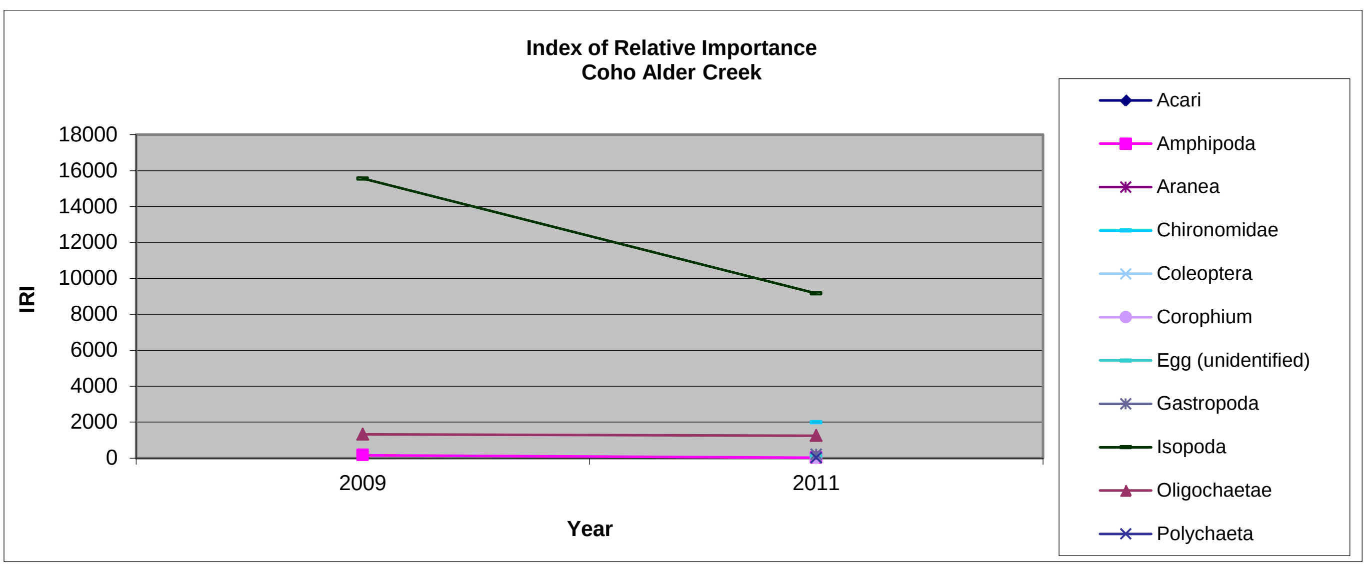


Figure 31. Index of Relative Importance for Coho at Alder Creek, 2008 – 2011. *No samples collected in 2008 or 2010.*

Chinook selected several prey taxa consistently at both sites over the years sampled. Coho however appear to be far more opportunistic feeders and preyed upon a greater diversity of invertebrate species. In comparing prey selectivity for coho between South Slough and Alder Creek, coho preyed upon a wider variety of species at South Slough. A greater number of diet samples from coho were acquired from South Slough than at Alder Creek, which may account for the differences between the two sites. While the prey selected at South Slough was more diverse, the dominant prey selected was consistent between the two sites. Coho at both sited demonstrated preferences for isopods and chironomids species.

Table 16. Proportions of samples collected from juvenile coho diets at South Slough (SS) and Alder Creek (AC), 2008-2011. *No samples were collected in 2007.*

Prey Utilization; Juvenile Coho Salmon Gut Contents								
	2008		2009		2010		2011	
*Year	**SS**	**AC**	**SS**	**AC**	**SS**	**AC**	**SS**	**AC**
Number of samples	21	0	4	1	7	0	14	10
Number of Species	29	0	6	2	11	0	9	10
Taxa	Mean proportion by taxa							
Acari	0.46							
Amphipod								0.87
Anisogammeridae	0.15				9.41			
Aphidoidea	1.38				15.29			
Aranea	0.77							0.87
Brachycera	0.31							
Braconidae							2.45	
Ceratopogonidae	0.77							
Chalcoidea	0.61							
Chironomidae	41.47		30.43		1.18		22.86	21.74
Chrysomellidae	0.61							
Cicadellidae	0.31							
Coleoptera	0.15				3.53		2.04	0.87
Copepoda					1.18			
Corophium	8.60				1.18		9.39	0.87
Diptera spp.	0.61							
Dolichopodidae	0.15						0.41	
Egg (unidentified)					7.06			7.82
Empididae	0.15							
Ephydridae	0.61							
Formicidae			4.35					
Gammaridae	0.92							
Gastrapoda	2.00				16.47			4.35
Gerridae	0.15							
Hemiptera	0.16							
Mysidacea			4.35					
Ichnuemoidea	0.31							
Isopoda	36.87		8.70	91.89	37.65		51.84	57.39
Nematoda	0.77		21.74					
Oligocheata	0.15			8.11			2.86	4.35
Polycheata								0.87
Psychodidae	0.15							
Psyllidae	0.15							
Ptychopteridae			30.43		2.53			
Sciaridae	0.31							
Sphaeroceridae					1.18			
Tipulidae	0.61						0.41	
Thysanoptera							0.41	

Chinook and coho at both sites predominantly preyed upon isopods and chironomids, with Chinook heavily selecting chorophium species as well. Chironomids were readily available and present at both sites during every year sampled. Isopods and corophium were not found in sampling events during 2007 – 2011, and so no conclusions can be drawn regarding the availability of density of aquatic invertebrates at either site.

More diet samples were acquired from coho than from Chinook between 2008 and 2011. These samples indicate that coho (39 prey taxa) are far more opportunistic feeders while Chinook (17 prey taxa) seem to be more selective feeders. Coho and Chinook both preyed the most heavily and consistently on isopods and chironomids. Without having the same number of samples we can only make inferences from the data. Gathering more Chinook diet samples may reveal greater diversity in prey selectivity.

Table 17. Total number of invertebrates captured as a percentage of the total number of all caught at South Slough (SS) and Alder Creek (AC), 2008 – 2011. **No samples were collected in 2007*

*Year	2008		2009		2010		2011	
Site	SS	AC	SS	AC	SS	AC	SS	AC
Number of samples	23	7	10	8	19	3	17	20
Number of Taxa	41	37	16	30	36	11	25	31
Taxa	Mean proportion by taxa							
Acari	0.50	0.20	0.23	0.68	0.04	0.78	-	0.3
Agromyzidae	0.05	0.10	-	-	-	-	-	-
Aphidoidea	2.42	0.10	12.95	0.23	66.30	-	53.4	0.4
Apidae	0.11							
Aranea	1.48	1.11	-	0.79	0.15	-	-	1.2
Biblionidae	-	-	-	0.34	0.02	-	-	-
Braconidae	0.79	-	1.36	0.34	0.23	-	0.3	0.2
Cantharidae	0.15	0.20	-	-	-	-	-	-
Carabidae	0.09	0.10	-	-	-	-	0.5	-
Cecidomyiidae	1.63	32.76	-	4.68	0.09	-	1.0	1.7
Ceratopogonidae	0.69	1.51	-	0.91	0.59	4.69	0.2	0.3
Chalcoidea	0.54	1.31	0.45	1.48	0.41	-	-	2.5
Chironomidae	51.73	45.26	52.50	38.99	5.54	6.25	22.3	29.4
Chloropidae	1.09	0.60	-	2.17	0.02	-	-	0.3
Cicadellidae	2.52	4.33	-	3.99	3.34	2.5	1.7	12.7
Ciculidae	-	-	-	-	-	2.5	-	-
Coccinellidae	-	-	-	-	0.11	-	-	-
Coenangrionidae	0.05	-	-	-	-	-	-	-
Coleoptera	-	0.30	-	0.91	0.08	1.25	-	0.6
Copepoda	-	-	-	-	0.02	-	0.2	-
Coroxidae	-	-	0.91	-	-	-	0.5	-
Curculionidae	0.05	-	-	-	-	-	0.2	0.2
Cynopoidae	0.09	0.10	-	-	-	-	0.2	-
Delphacidae	-	-	-	-	-	-	-	0.5
Diptera	0.25	0.20	-	-	0.11	5.00	-	-
Dolichopodidae	3.07	-	1.59	13.46	2.16	-	2.3	-
Donanciinae	0.05	-	-	-	-	-	-	-
Drosophilidae	0.05	-	-	-	-	-	-	20.6
Empididae	1.09	1.11	-	-	0.63	-	0.5	0.4
Ephydridae	9.15	0.40	8.41	0.79	2.50	1.25	0.5	8.6
Entomobryidae		0.91						
Formicidae	0.15	0.20	-	-	0.06	-	-	-
Gastrapoda	3.76	0.30	1.36	0.11	2.60	46.25	1.5	0.9
Hymenoptera	0.25	0.10	-	0.23	0.02	-	-	0.1
Hypogastridae		0.20						
Ichnuemoidea	5.49	0.10	-	1.25	0.11	-	0.5	0.3
Isopoda	0.05	-	1.14	0.11	0.11	-	0.2	-
Mesovellidae	-	-	-	-	-	-	0.2	-
Muscidae	0.05	-	-	0.68	-	-	-	0.3

Mymiridae	1.58	0.91	10.23	2.28	0.09	-	1.0	1.1
Nematocera				1.03				
Phoridae	-	0.10	-	0.91	0.06	1.25	-	0.2
Pscoptera	0.05	-	-	0.34	-	-	-	-
Psychodidae	3.51	3.43	4.10	9.69	0.56	13.75	5.0	14.1
Psyllidae		0.20						
Ptychopteridae	0.45	-	0.45	1.14	0.49	1.25	5.2	0.6
Sciaridae	1.6	1.11	0.45	5.82	0.28	6.25	0.7	0.6
Sialdidae	0.25	0.30	-	-	0.13	-	-	-
Sphaeroceridae	0.25	0.30	0.23	0.91	0.17	-	0.7	0.1
Sphecoidea	0.05	-	-	-	-	-	-	-
Staphylinidae	0.15	0.20	0.0.91	0.68	0.04	-	0.2	0.6
Syrphidae	0.05	0.20	-	0.11	1.77	-	-	0.1
Thysanoptera	-	0.71	-	2.05	0.09	-	-	0.6
Tipulidae	1.14	0.40	-	0.11	0.21	-	0.5	0.2
Trichoptera	1.88	-	-	-	0.04	-	-	2.8

5.3.3 Fish Community 2012

Permit limitations restricted the ability to sample the fish community in 2012. However, fish community was extensively monitored post-project during the years of 2008 through 2011.

5.3.4 Pit Tag Array 2012

The PIT tag array was installed in March of 2012, with the first detection occurring in the following month of April (Table 28). The PIT tagged fish was present within the site for two hours before being detected again leaving the slough. The advantage of having two antennas back to back is the ability to provide directional information in regards to movement. The direction the fish is going, either migrating into or out of the channel is determined by the order in which it is detected by the two antennas. Directional movement informs researchers how tagged individuals are utilizing habitat. Two antennas are also beneficial because they increase the likelihood of detection. A tagged fish detection is more likely to occur at certain orientations than others; a relational factor to the electrical field created by the array. Certain tag orientations do not result in the tag receiving an adequate charge to produce detection. Extreme high water events are another reason in which a tagged fish could go undetected. In this case the antennas maximum range is exceeded and cannot cover the entire tidal prism as a product of extremely high water volumes.

A second tag detection occurred in June (Table 28). The PIT tagged fish was detected entering the site but never leaving the site. The failure to detect the tagged fish leaving the site could have been attributed to several factors such as predation, or a larger tide providing windows of zero detection outside of the antennas range. Since June, no tagged fish have been identified utilizing South Slough. In August construction equipment damaged the electrical cord to the antenna array, temporarily disabling the system. In an effort to avoid additional risk or damage, the array was removed in early September to allow for restoration work in the channel. The array will be re-installed in October to continue year round data collection.

Table 28. PIT tag detections at South Slough 2012.

Detection Date	April 2012	June 2012
Capture method	Dip net	Dip net
File ID	JAR12086.SC3	JAR12087.SC5
Flags	AD (hatchery reared)	AD (hatchery reared)
Length (mm)	66 mm	63 mm
Organization	USF&WS	USF&WS
Release Site	Spring Creek Hatchery	Spring Creek Hatchery
River Kilometer Mile (Rkm)	269	269
Species	Chinook	Chinook
Tag Date	3-26-2012	3-27-2012
Tag ID	3D931C2DD7E4E8	3D9.1C2DD8D975
Release Date & Time	4-13-2012 10:15	3-27-2012 08:00

5.3.5 Genetic Analysis

Genetic samples from 2011 were processed at NOAA's NMFS Northwest Fisheries Science Center, Manchester Facility. Limited numbers of Chinook were observed at South Slough and Alder Creek so genetic samples were only taken from non-clipped or marked fish. Seven samples from South Slough and three samples from Alder Creek taken in 2011 were processed for evaluation. Several stocks were identified from these samples; those stocks were the Spring Creek Group Fall run (SCG_F), West Cascades Fall run (WC_F), Willamette Spring run (WR_Sp), Rogue, and coastal stock. Spring Creek group Chinook are a fall ("tule") stock originating from the Spring Creek National Fish Hatchery in the Columbia River Gorge area. This stock has been widely propagated throughout the lower Columbia River (Myers et al. 2006). The West Cascade spring and fall (only fall stock were observed) and the Willamette River spring stock groups are comprised of fish originating from several tributaries and hatcheries in the lower Columbia River (Myers et al. 2006). Rogue River stock was introduced to the Columbia River from southern Oregon in the 1980s as part of a continuing effort to enhance fisheries in off-channel areas (North et al. 2006).

Data from genetic analysis can provide insight into migration patterns and habitat usage by genetically diverse salmonid stocks in the Columbia River Basin. Juvenile subyearling Chinook salmon are smaller than their yearling counterparts and tend to use the estuary as juvenile rearing habitat to a much greater extent than other juvenile salmonids (Thorpe 1994). Salt marshes and tidal channels are important habitats due to their ability to provide a source of food and shelter for subyearling Chinook salmon rearing in the estuary (Healy 1982; Bottom et al. 2005). In the month of March there is an annual large pulse of juvenile fall tule Chinook salmon that is released as subyearlings from the Spring Creek National Fish Hatchery (RKM 269) upstream of Bonneville Dam. Rogue River stock was heavily used at Big Creek Fish Hatchery, and in the Clatsop Fisheries Net Pen project in Young's Bay. Genetic samples from Rogue and Spring Creek stock suggest that they are either escaping without ad-clips, and/or they are reproducing naturally in the Lower Columbia River after introduction through hatcheries.

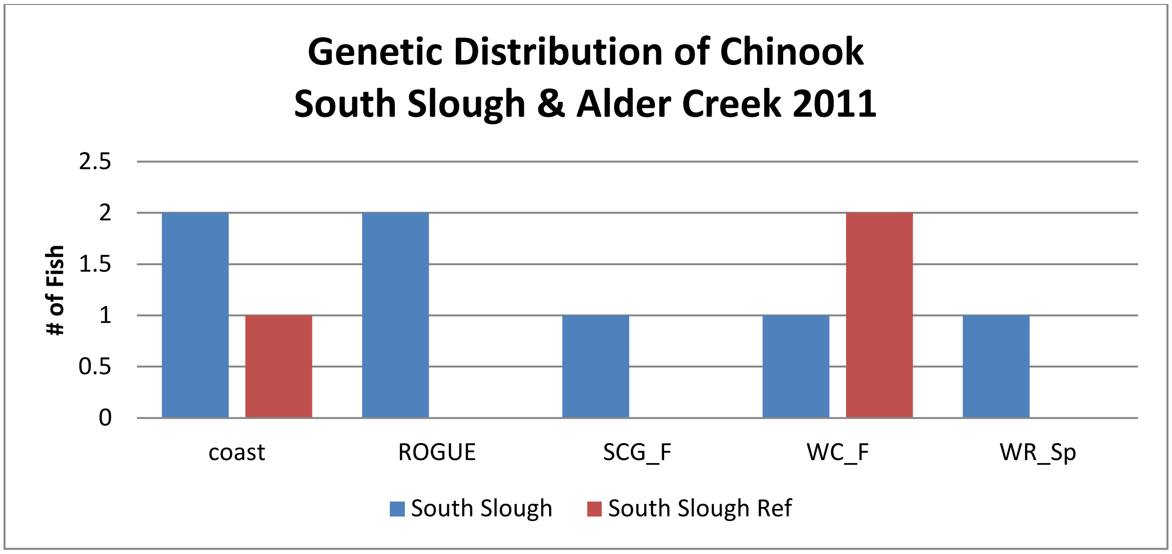


Figure 46. Genetic distribution of Chinook at South Slough and South Slough Ref (Alder Creek), 2011.

Genetic analysis of samples collected at South Slough and Alder Creek in 2011 showed that Coastal and West Cascade fall stocks were the dominant stocks sampled, each representing 30% of all Chinook sampled. Rogue River stock contributed 20% while Spring Creek Group fall and West Cascade spring each represented 10% of those sampled.

Table 29. Genetic stock of unclipped Chinook sampled at South Slough and Alder Creek, 2011.

Sample Date	Site	Species	Length (mm)	Weight (g)	Sample ID	Best Estimate	Probability
3-28-2011	South Slough	Chinook	46	0.76	SS11.007	WR_Sp	88%
4-12-2011	South Slough	Chinook	58	1.92	SS11.0016	Rogue	99%
5-24-2011	South Slough	Chinook	48	0.90	SS11.0403	WC_F	99%
5-24-2011	South Slough	Chinook	52	1.11	SS11.0402	Coast	98%
5-24-2011	South Slough	Chinook	57	1.63	SS11.0374	SCG_F	94%
6-9-2011	South Slough	Chinook	53	1.60	SS11.0513	Coast	99%
6-9-2011	South Slough	Chinook	58	1.88	SS11.0512	Rogue	99%
5-12-2011	Alder Creek	Chinook	37	0.54	SSR11.0039	WC_F	98%
5-12-2011	Alder Creek	Chinook	40	0.52	SSR11.0044	WC_F	99%
6-9-2011	Alder Creek	Chinook	65	2.65	SSR11.0111	Coast	69%

5.3.6 Synthesis of Fish Community Sampling 2007-2011

5.3.6.1 Sampling Frequency & Methods 2007-2011

Fish community sampling began in 2007 at South Slough (pre-restoration) and Alder Creek. Frequency of sampling in regards to months and events per month varied between years and between the two sites (Table 30). Over the course of 5 years the fish community sampling gear has been modified at both sites, and different methods have been attempted at Alder Creek. Fish community sampling occurred between January and August in 2007 through 2011. Events were implemented at a frequency of once to twice a month (Table 31). A trapnet was used consistently at South Slough, with a cod net and sanctuary bag during 2007 and 2008, and a sock net and livebox from 2009 to 2011. The change in trap net components reduced stress to salmonids and discouraged escapement. A trapnet with cod net and sanctuary bag was used at Alder Creek in 2007. An 80ft. long by 10ft. deep beach seine was used in 2008, and a trapnet with sock net and livebox from 2009 to 2011.

Table 30. Fish sampling methods used at South Slough and Alder Creek between 2007-2011

	South Slough		Alder Creek	
Year	**Method**	**Modification**	**Method**	**Modification**
2007	Trapnet	Sanctuary bag	Trapnet	Sanctuary bag
2008	Trapnet	Sanctuary bag	Seine	N/A
2009	Trapnet	Livebox, modified wing nets	Trapnet	Livebox
2010	Trapnet	Livebox	Trapnet	Livebox
2011	Trapnet	Livebox, seining down channel	Trapnet	Livebox, seining down channel

Sampling frequency was inconsistent between the years as a result of funding, permitting and weather conditions. Increased funding sources in 2007, 2008 and 2011 allowed sampling to be implemented twice a month, while in 2009 and 2010 sampling occurred once a month. During the months of January through March above average high water events inhibited trapnet set up at both sites in 2010.

At South Slough, the primary sampling method, the use of a fyke trap net, remained consistent between 2007 and 2010, with the addition of seining down the channel prior to pulling the trapnet in 2011. In 2009 small mesh panels were used as a substitute for wing nets in an attempt to reduce the ‘bagging’ effect observed in traditional wing nets resulting from the outgoing current. Constructing panels that could be easily placed and removed for fish sampling proved difficult because of the depth and velocity in the channel at South Slough, and gaps between the panels allowed for small areas of potential escapement from the trap. The use of panels was discontinued after materials were stolen from on site in late spring of 2009.

In 2011 seining was implemented in order to encourage any fish holding upstream in the channel downstream into the trap. A pole seine was used to span the width of the channel and walked from a distance upstream (102 meters at South Slough and 92 meters at Alder Creek) down to the mouth of the trapnet. It was discovered that large numbers of fish were able to resist the current and remain in the channel during ebb tide. The channel at Sough Slough holds enough water during low tide to allow fish to remain without stranding. The addition of seining to fish community sampling in 2011 impacts comparison of fish density between 2011 and previous years sampled. At Alder Creek in 2008 a beach seine was used in place of a trap net. This impacts the comparison of fish abundance/density as the area sampled using a beach seine represents a sub sampling of the sampled by trap net in successive years.

Table 31. Number of successful fishing events at South Slough and Alder Creek, gear used, and hours (TN) or pulls (BS) by month for 2007 through 2011. **A dash indicates that sampling did not occur that month.*

Number of successful fishing attempts, hours or area (m2) fished, and gear type																
(TN=trapnet, BS= beach seine)		2007			2008			2009			2010			2011*		
Site	Month	Number of Events	Hours Sampled	Gear	Number of Events	Hours Sampled	Gear	Number of Events	Hours Sampled	Gear	Number of Events	Hours Sampled	Gear	Number of Events	Hours Sampled	Gear
South Slough	Jan	1	3	TN	1	4.5	TN	-	-	TN	-	-	TN	-	-	TN
	Feb	2	9.5	TN	1	4.7	TN	1	3.7	TN	-	-	TN	-	-	TN
	Mar	2	5.5	TN	2	8.2	TN	1	3.5	TN	-	-	TN	2	7	TN
	Apr	2	6.8	TN	2	10.5	TN	2	13.3	TN	2	9	TN	2	12	TN
	May	-	-	TN	3	16.8	TN	1	5	TN	1	4	TN	2	11	TN
	Jun	1	2.5	TN	2	10	TN	1	4	TN	1	4	TN	1	4	TN
	Jul	-	-	TN	2	10	TN	1	5.5	TN	1	4	TN	1	4	TN
	Aug	-	-	TN	2	10	TN	-	-	TN	1	4	TN	-	-	TN
Alder Creek	Jan	-	-	TN	-	-	-	-	-	TN	-	-	TN	-	-	TN
	Feb	-	-	TN	-	-	-	-	-	TN	-	-	TN	-	-	TN
	Mar	-	-	TN	-	-	-	-	-	TN	-	-	TN	2	10	TN
	Apr	2	7	TN	3	N/A	BS	2	3.3	TN	2	7	TN	2	9	TN
	May	2	1.3	TN	3	N/A	BS	1	3.5	TN	1	2	TN	2	7	TN
	Jun	1	0.8	TN	4	N/A	BS	1	3.3	TN	1	3	TN	1	3	TN
	Jul	-	-	TN	4	N/A	BS	1	3.3	TN	1	2	TN	1	2	TN
	Aug	-	-	TN	2	N/A	BS	-	-	TN	1	2	TN	-	-	TN

5.3.6.2 Species Composition

During sampling between 2007 and 2011 a total of 15 different taxa of fish were caught at South Slough; 5 were salmonids, 7 were native non-salmonid species, and 3 were non-native species. A total of 12 different taxa were observed at Alder Creek; 3 were native salmonids, 7 were native non-salmonid species, and 2 were non-native species. Native fish species (including salmonids) comprised 80% of the total number of species observed at South Slough, and 83% for Alder Creek.

Six different native, non-salmonid species were observed at both South Slough and Alder Creek between 2007 and 2011. Of these, 5 were present at both sites: including three-spine stickleback, cottid sp., peamouth chub, shiner perch, and smelt. Large-scale suckers were only observed at Alder Creek. Lamprey were only observed at South Slough. Three-spine stickleback were the most abundant of all the species caught at both sites. Non-native fish species observed included sunfish sp., largemouth bass, and banded killifish. Banded killifish and sunfish were found at both sites, largemouth bass were sampled at South Slough on just one occasion during the 5 years of sampling. Overall the diversity and abundance of non-native species was consistently low for both sites over the 5 years sampled.

Table 32. Summary table showing number of non-salmonid, native fish species caught by month at South Slough and Alder Creek, 2007-2011.

Site	Month	Number of native species (non-salmonid)					Native species caught (total number)
		2007	2008	2009	2010	2011	
South Slough	January	2	1	-	-	-	Three spine stickleback, cottid, peamouth chub, shad, shiner perch, smelt, lamprey **2007:** (6,580) **2008:** (61,579) **2009:** (20,253) **2010:** (3,722) **2011:** (15,197)
	February	2	3	2	-	-	
	March	4	3	2	-	1	
	April	2	3	2	3	1	
	May	-	3	1	1	1	
	June	2	4	3	2	1	
	July	-	3	2	2	0	
	August	-	4	-	2	-	
Alder Creek	January	-	-	-	-	-	Three spine stickleback, cottid, peamouth chub, large scale sucker, shiner perch, smelt **2007:** (634) **2008:** (3,217) **2009:** (674) **2010:** (1,296) **2011:** (1,179)
	February	-	-	-	-	-	
	March	-	-	-	-	1	
	April	2	3	4	2	1	
	May	2	3	2	3	3	
	June	2	2	4	3	2	
	July	-	4	3	3	2	
	August	-	3	-	4	-	

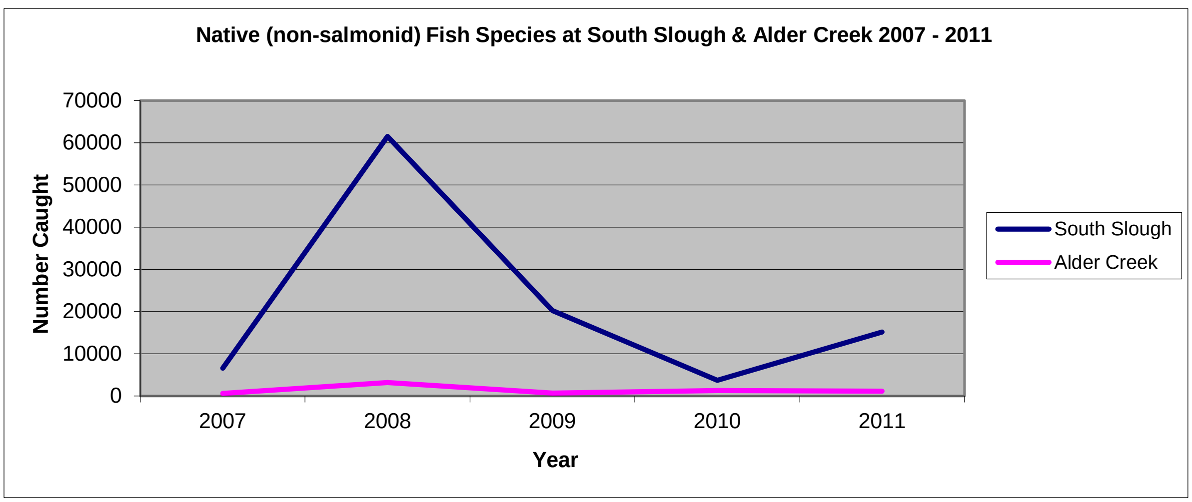


Figure 47. Total number of native, non-salmonid, fish species caught at South Slough and Alder Creek, 2007 – 2011.

Table 33. Summary table showing number of non-native species captured by month at South Slough and Alder Creek, 2007-2011.

Site	Month	Number of non-native species					Non-native species caught (total number)
		2007	**2008**	**2009**	**2010**	**2011**	
South Slough	**January**	10	0	-	-	-	Banded killifish, sunfish, large mouth bass **2007:** (7) **2008:** (61) **2009:** (4) **2010:** (16) **2011:** (5)
	February	0	0	0	-	-	
	March	0	0	0	-	0	
	April	0	1	1	0	0	
	May	-	1	0	0	1	
	June	1	2	0	1	1	
	July	-	1	0	1	0	
	August	-	3	-	2	-	
Alder Creek	**January**	-	-	-	-	-	Banded killifish, sunfish **2007:** (1) **2008:** (30) **2009:** (6) **2010:** (24) **2011**: (4)
	February	-	-	-	-	-	
	March	-	-	-	-	0	
	April	1	0	1	0	1	
	May	0	1	0	1	1	
	June	0	1	1	2	1	
	July	-	1	1	1	1	
	August	-	1	-	0	-	

Table 34. Summary table showing number of unmarked salmonid species captured by month at South Slough and Alder Creek, 2007-2011.

Site	**Month**	**Number of salmonid species**					Salmonid species caught **(total number)**
		2007	**2008**	**2009**	**2010**	**2011**	
South Slough	January	1	0	-	-	-	**2007:** Coho, chum (5) **2008:** Chinook, coho, chum, cutthroat, steelhead (56) **2009**: Chinook, coho, chum, cutthroat (11) **2010:** Chinook, coho (44) **2011:** Chinook, coho, chum, cutthroat (736)
	February	1	0	0	-	-	
	March	1	1	1	-	3	
	April	2	2	3	2	2	
	May	-	3	0	2	2	
	June	1	3	2	2	2	
	July	-	3	0	1	1	
	August	-	2	-	2	-	
Alder Creek	January	-	-	-	-	-	**2007:** Chinook, Coho (2) **2008:** Coho (6) **2009:** Chinook, coho, cutthroat (18) **2010:** Chinook, coho (4) **2011:** Chinook, coho (134)
	February	-	-	-	-	-	
	March	-	-	-	-	1	
	April	0	0	3	1	2	
	May	2	1	1	1	2	
	June	0	1	1	0	2	
	July	-	1	0	0	1	
	August	-	0	-	1	-	

The total numbers of native fish, salmonids, and non-native fish caught each month were similar between the years (post-restoration) despite the difference in sampling frequency and the addition of seining down the channel in 2011. The total numbers of species caught, however, vary greatly between the years and sites. Because sampling frequency and technique differed between years and between sites total numbers cannot be used reliably for statistical analysis, but instead general conclusions can be made regarding similarity in species present between the years and sites. For example, three-spine stickleback were captured during every event at both sites which indicates they are present throughout the sampling season, but their exact density or abundance between years cannot be compared because of the differences in sampling.

Three species were observed during every year sampled at both sites. These were coho, three spine stickleback and banded killifish. Five salmonid species were observed at South Slough between 2007 and 2011, Chinook (*Oncorhynchus tshawytscha*), coho (*O. kisutch*), chum (*O. keta*), cutthroat (*O. clarkii*), and steelhead (*O. mykiss*). Chinook, coho, and cutthroat were the only salmonid species observed at Alder Creek. Over this same period, coho and Chinook were the most abundant salmonid species sampled at both sites in all years sampled. Steelhead were only observed during 1 sampling event, at South Slough, for the entire 5 years of sampling. Three cutthroat were observed between 2009 and 2011; 1 caught in April of 2009, and 2 caught in May of 2011.

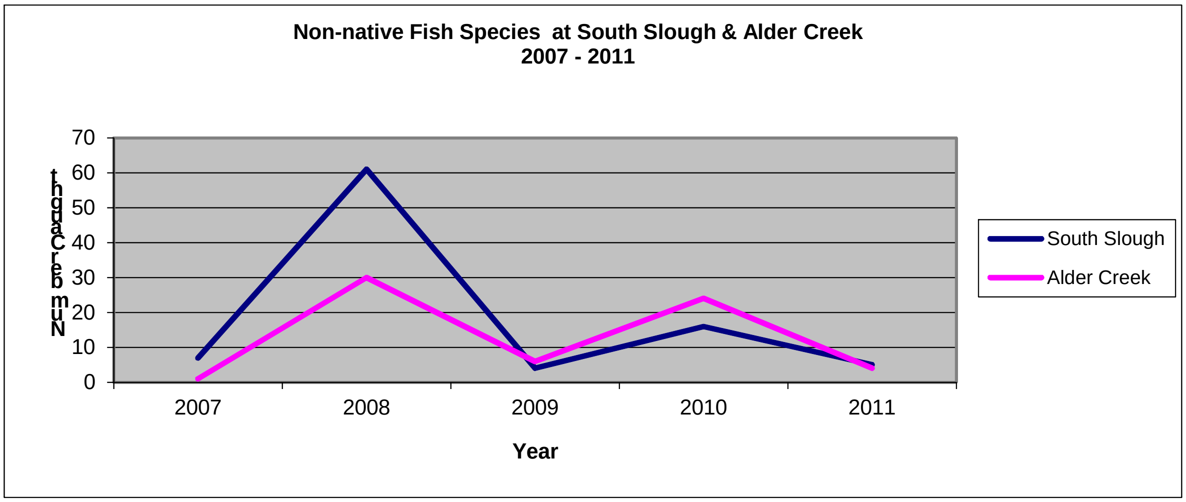


Figure 48. Number of non-native fish caught at South Slough and Alder Creek, 2007 – 2011.

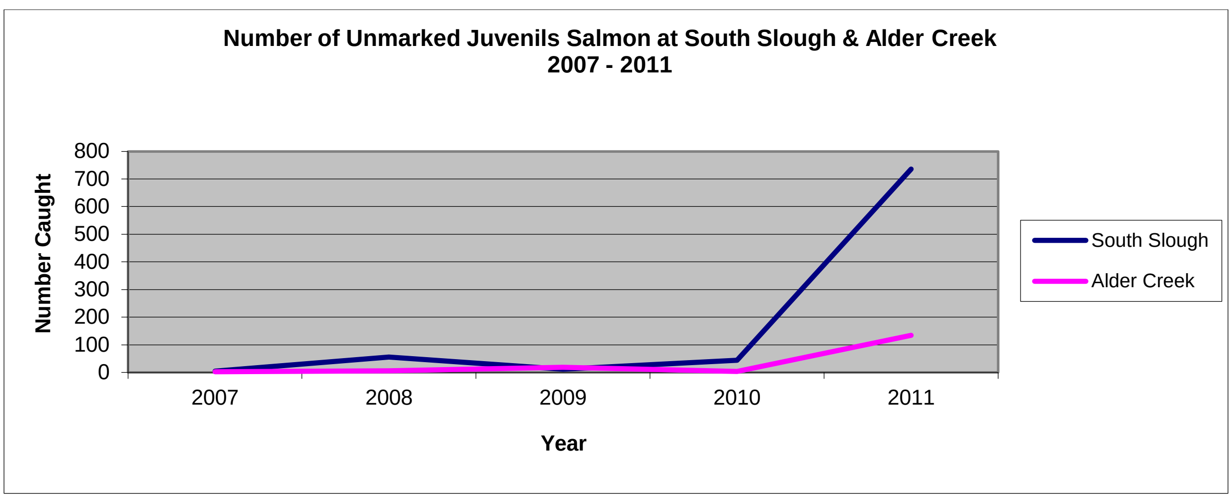


Figure 49. Number of salmonids caught at South Slough and Alder Creek, 2007 – 2011.

Table 35. Total number of fish species caught as a percentage of total number of species caught at South Slough and Alder Creek, 2007-2011.

	South Slough					Alder Creek				
	2007	**2008**	**2009**	**2010**	**2011**	**2007**	**2008**	**2009**	**2010**	**2011**
***Chinook**		0.02	0.01	0.10	0.10	0.50		1.70	0.20	0.90
Chum	0.02	0.01	0.02		0.006					
***Coho**	0.14	0.10	0.01	1.10	4.40	0.50	0.20	0.80	0.20	6.10
Cutthroat		0.001	0.005		0.006			0.30		
Steelhead		0.005								
Three spine stickleback	99.5	99.6	99.9	98.1	95.4	95.7	93.8	90.5	90.1	92.4
Peamouth chub	0.05	0.06	0.03	0.20	0.01		2.40	4.80	6.30	0.30
Cottid sp.	0.20	0.04		0.08		2.60	2.60	0.03	0.20	
Shiner perch	0.02	0.003						0.60	0.90	
Shad	0.05						0.03			
Smelt	0.02	0.004							0.08	
Large scale sucker							0.03	0.20	0.20	0.05
Lamprey			0.0001							
Banded killifish	0.10	0.03	0.02	0.40	0.03	0.50	0.90	0.90	1.60	0.05
Sunfish sp.		0.01		0.06	0.02				0.20	0.20
Largemouth bass		0.02								

**Includes both non-marked and ad-clipped salmon.*

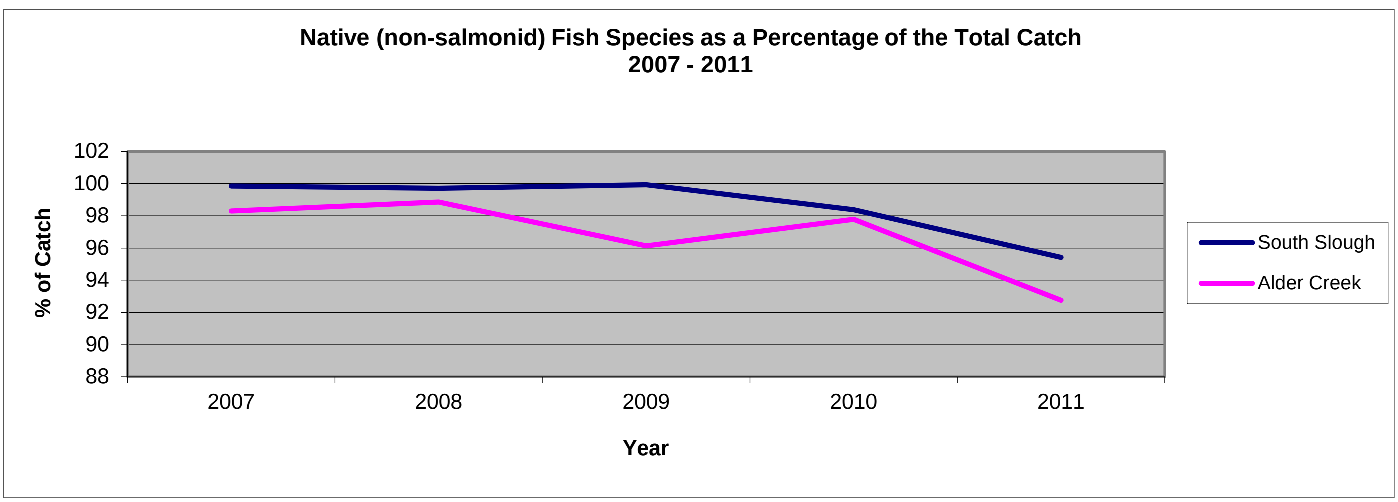


Figure 50. Native, non-salmonid, fish species as a percentage of the total catch at South Slough and Alder Creek, 2007 – 2011.

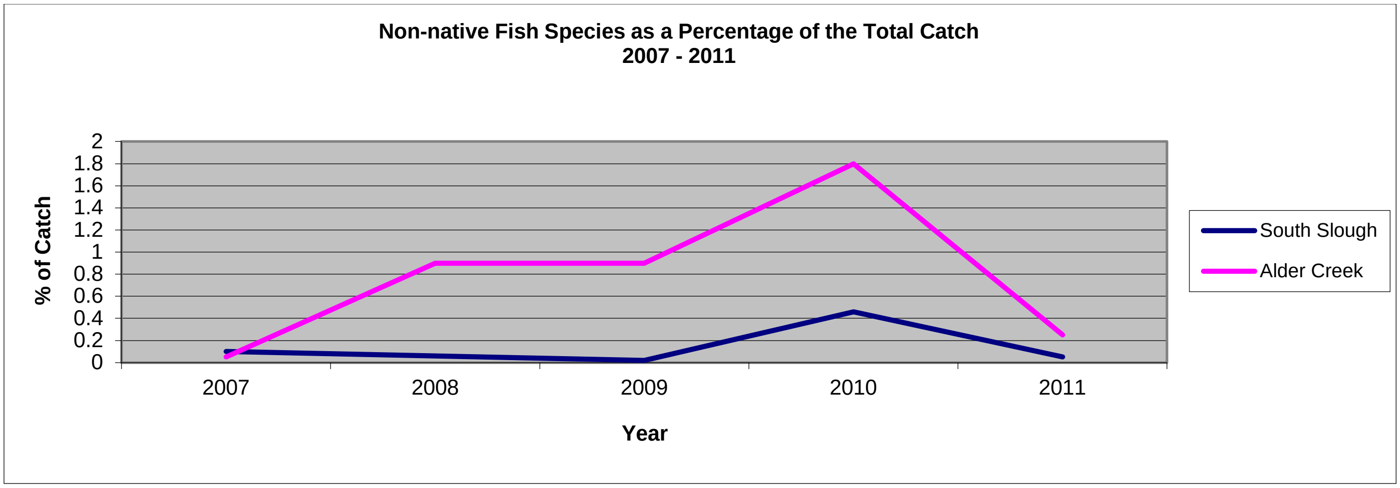


Figure 51. Non-native fish species as a percentage of the total catch at South Slough and Alder Creek, 2007 – 2011.

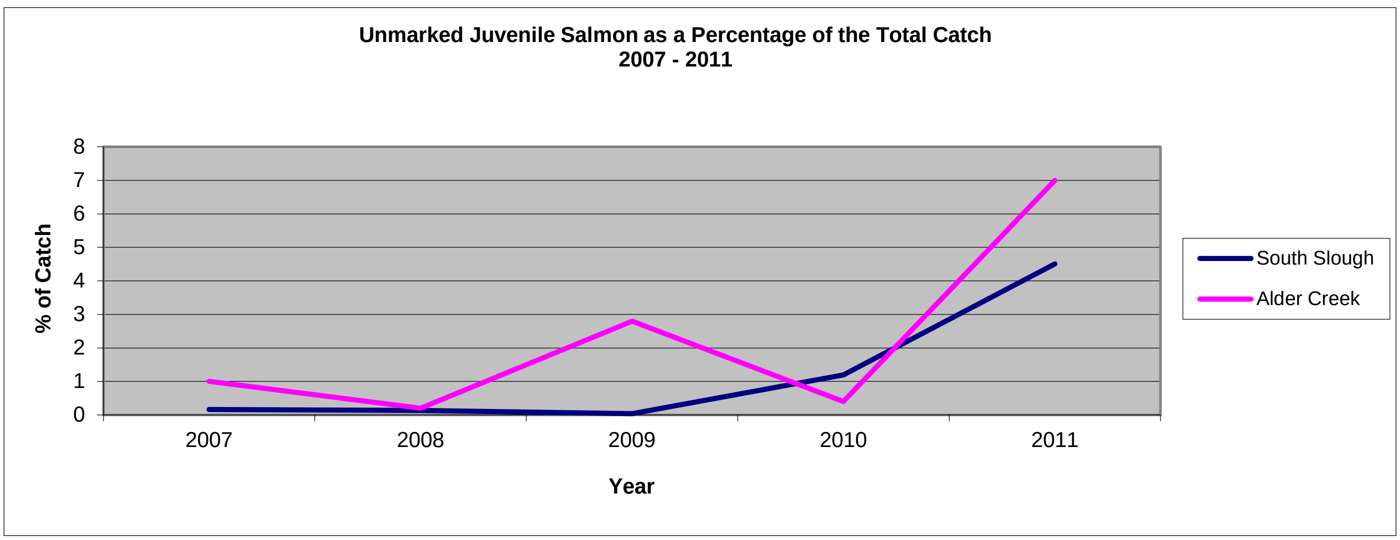


Figure 52. Unmarked juvenile salmon as a percentage of the total catch at South Slough and Alder Creek, 2007 – 2011.

Percentages of species relative to the entire catch were influenced by the change of methods in 2011. Alternative methods revealed large numbers of coho fingerlings present in the channel, increasing their percentage in relation to total catch. It was discovered that Chinook were present during more months of the sampling season in 2011 than in previous years. Aside from these two differences, the overall percentage numbers among sites appear similar to previous years, as were results from seining Alder Creek in 2008. All numbers were included for a visual comparison in the following figures.

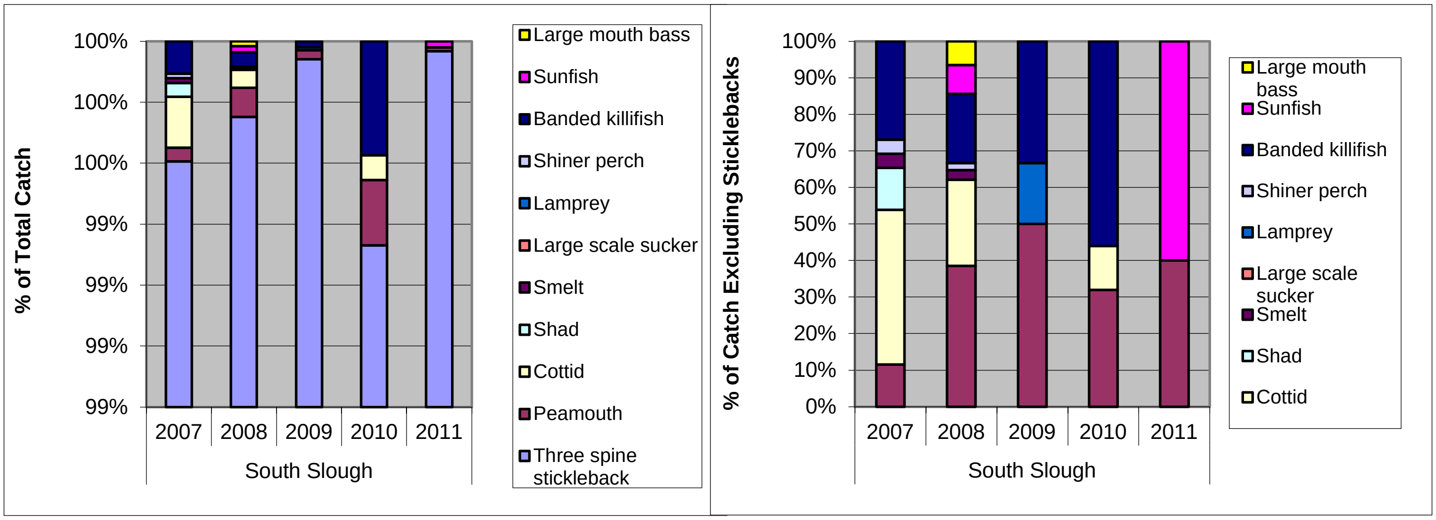


Figure 53. Proportions of different fish species caught at South Slough, 2007-2011

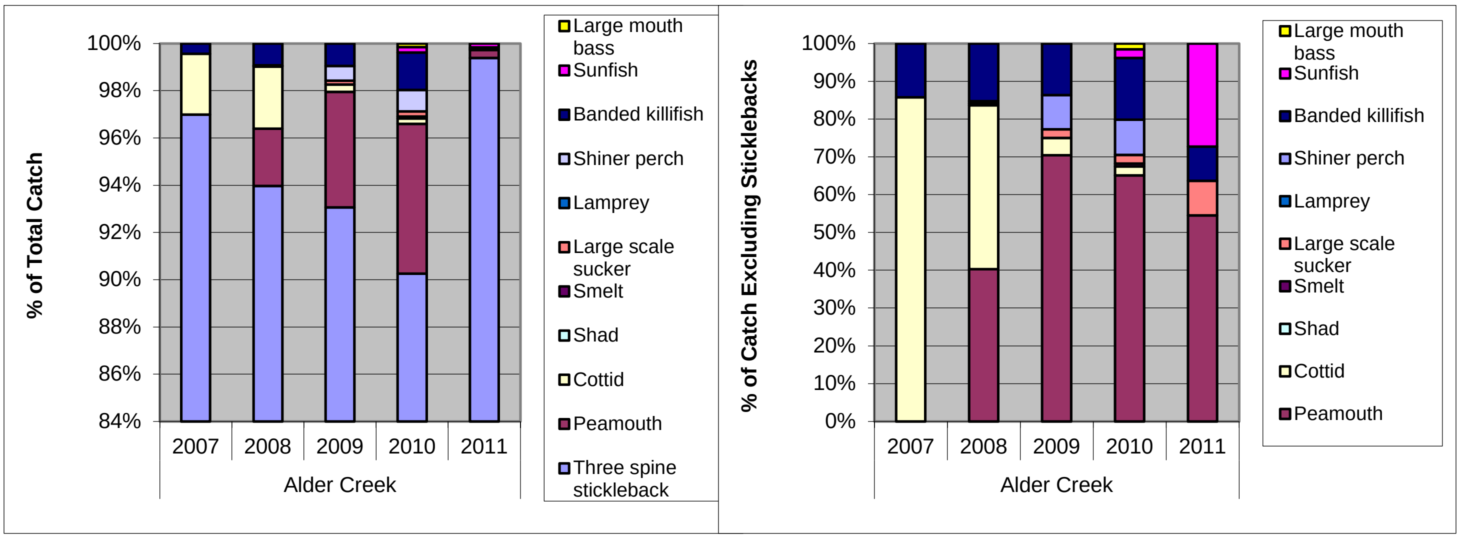


Figure 54. Proportions of different fish species at Alder Creek, 2007-2011

Seining in 2011 increased the percentage of salmon in relation to the total catch, which correlates to a decrease in the percentage of other species in the catch. 2011 data has been included with past years' percentages for comparative purposes, with the caveat that statistically the data cannot be compared. Regardless of the sampling method utilized, Three-spine sticklebacks were consistently the most abundant species; accounting for 90% to 99% of the total catch at both sites.

Because three-spine sticklebacks were such a large component of the total catch, it helps to remove them from the picture (Figure 53 and Figure 54) when looking at the composition of other fish species. Overall, native fish species were more abundant than non-native fish species at both sites, with peamouth and cottids representing the second and third most abundant species. Banded killifish and Sunfish species were the most common and abundant non-native fish species at Alder Creek, and banded killifish the most common and abundant at South Slough.

The addition of seining to the monitoring protocols in 2011 seemed to have a significant contribution to the total number of fish captured per sampling event, salmonids in particular. Prior to 2011 salmonids made up 1.91% of the total catch over a four-year period at South Slough. In 2011 alone this number increased to 4.62%. A similar increase was seen at Alder Creek (salmonids made up 4.1% of the cumulative total catch from 2007-2010 to 7%

in 2011) suggesting that the addition of hand seining to the fish sampling methods increased catch efficiency, providing a more accurate account of the abundance and percentages of salmon relative to the overall fish community.

Limited hatchery (marked) salmon were observed at South Slough and Alder Creek over the 5 year sampling period. During sampling in 2009 and 2011 marked Chinook were observed at South Slough and Alder Creek in 2009 and 2011. Marked coho were observed at Alder Creek in 2009.

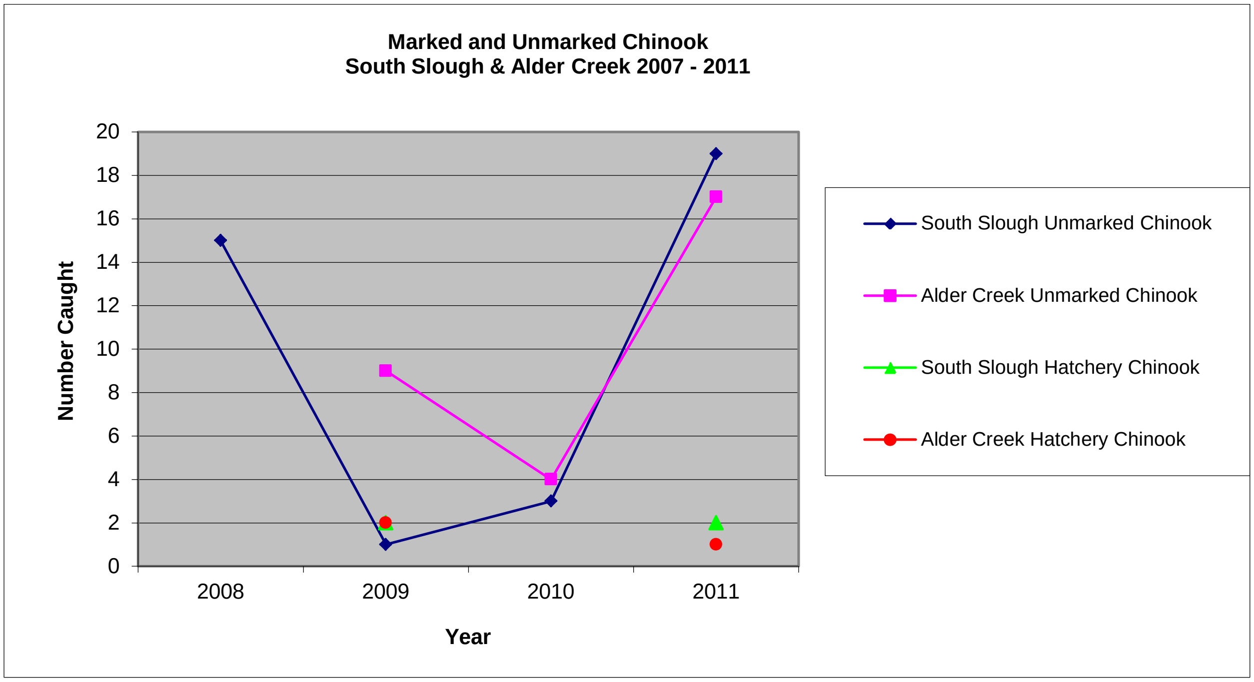


Figure 55. Numbers of marked and unmarked Chinook at South Slough and Alder Creek, 2007 – 2011. **No Chinook were caught in 2007.*

Sampling methods were different between South Slough and Alder Creek in 2008, and while they were consistent at both sites in 2011 the addition of seining differed from methods in previous years. Despite the change in methods the species present were relatively consistent and similar between sites. With the exception of 2009, coho were the most abundant salmonid species at both sampling sites. Over 5 years of sampling coho represented 94.4% of all salmonids captured at South Slough and 79.4% at Alder Creek.

Over that same 5 year period Chinook represented the next highest proportion of all salmonids with 2.8% at the South Slough and 18.8% at Alder Creek. Chum, cutthroat and steelhead in that order follow with a regressing trend. Coho and chum salmon were the only salmonid species caught at both sites pre-and post-restoration. Sampling at South Slough demonstrated an increase in salmonid species present within the tidal slough after restoration. The number of sampling events, hours per month and per year (Table 30) needs to be taken into consideration when comparing yearly totals, as the number of events differed between years and impacts the total catch numbers.

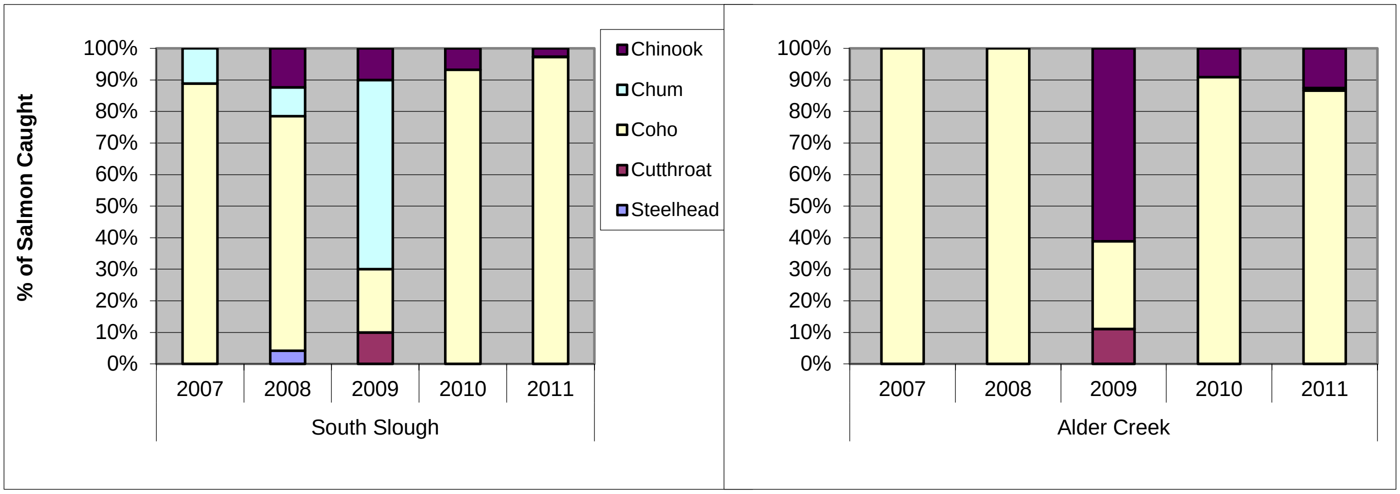


Figure 56. Proportions of unmarked juvenile salmonid species caught at South Slough and Alder Creek, 2007-2011.

5.3.6.3 Temporal Distribution

Timing of habitat use for fish species at South Slough was compared between 2007 through 2010, to 2011 data in order to determine if the addition of seining down as a modification to sampling methods biased the data. It was found that for Chinook and coho temporal distribution was 1 to 2 months different when including the 2011 data. Seining down the channel is believed to give a more accurate portrayal of species present and density as it covers the entire channel, discouraging escapement. Trapnetting without seining down the channel allows fish to actively avoid the trap only capturing a portion of what species and their total numbers may actually be present. The difference in methods does impact comparability of data between 2007 and 2010 to data from 2011. In 2011 with the modification to trapping methods Chinook salmon were most abundant in April, in contrast to data from 2007 – 2010 where they are most abundant in June. The same proved to be true for coho; in 2011 they were most abundant in May while between 2007 and 2010 they were most abundant in June. Chum were only present at South Slough during March and April for all years sampled.

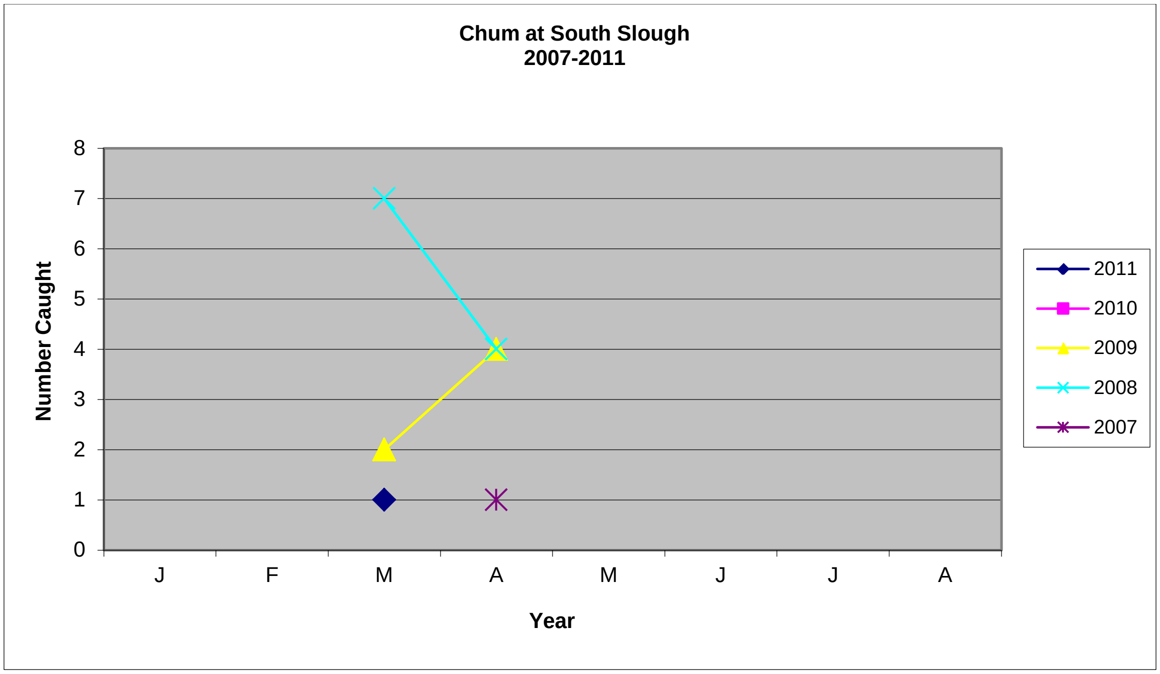


Figure 57. Temporal distribution of chum at South Slough, 2007 – 2011.

Unmarked Chinook were the most abundant between March and June at both South Slough and Alder Creek. Marked Chinook were observed in April of 2009 and in May of 2011 at both sites. 2011 was the only year that Chinook were observed in February and March, and with the modification in methods during 2011 it cannot be concluded whether or not this was an annual fluctuation in timing or the result of more efficient methods.

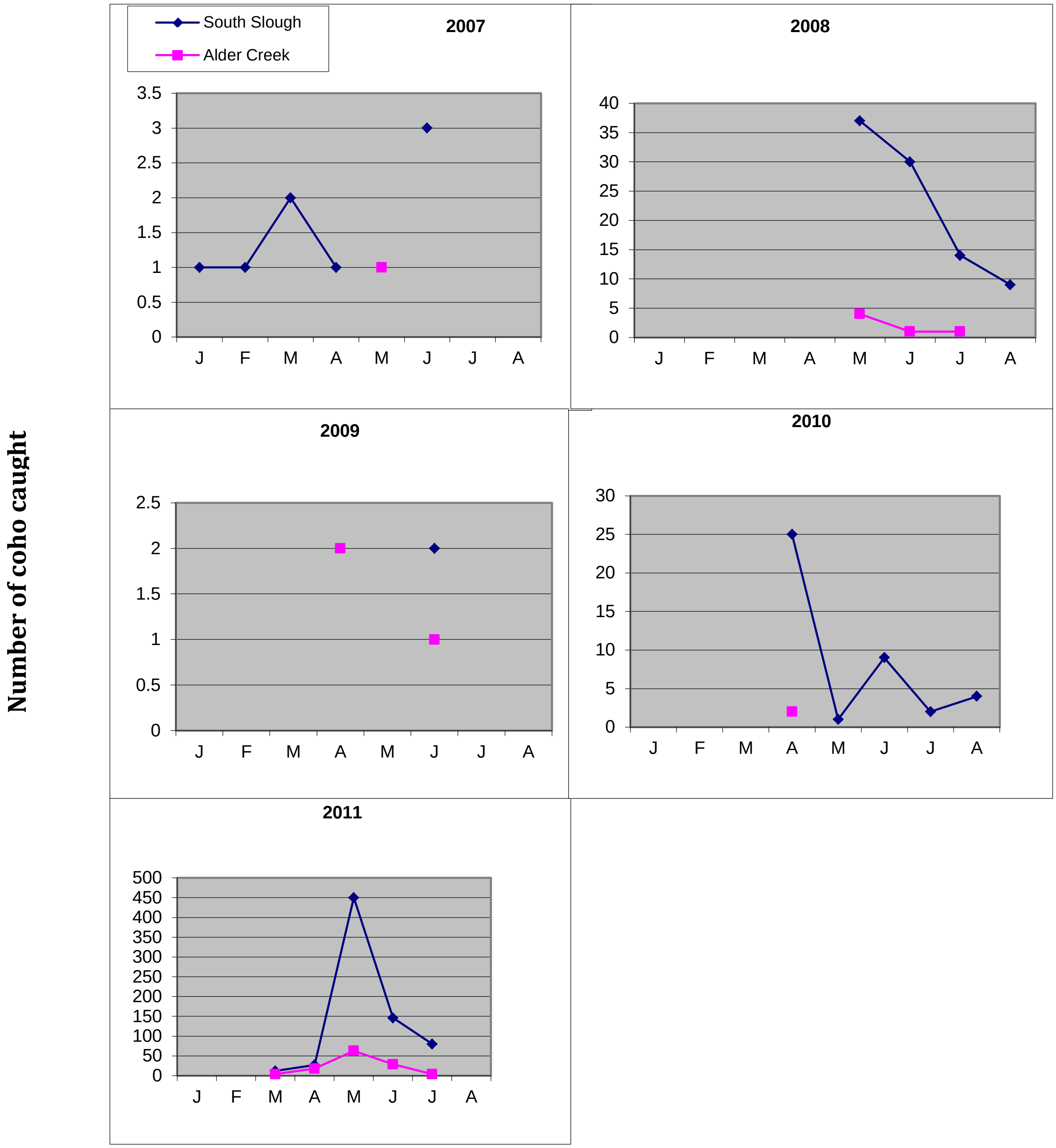


Figure 58. Temporal Distribution of unmarked Coho at South Slough and Alder Creek 2007 – 2011.

Coho exhibited a greater temporal range than Chinook at both sites. Coho were observed at South Slough during every month of the sampling season at least once during the 5 years sampled, and observed between March and July at Alder Creek. Abundance peaked in May at South Slough and Alder Creek.

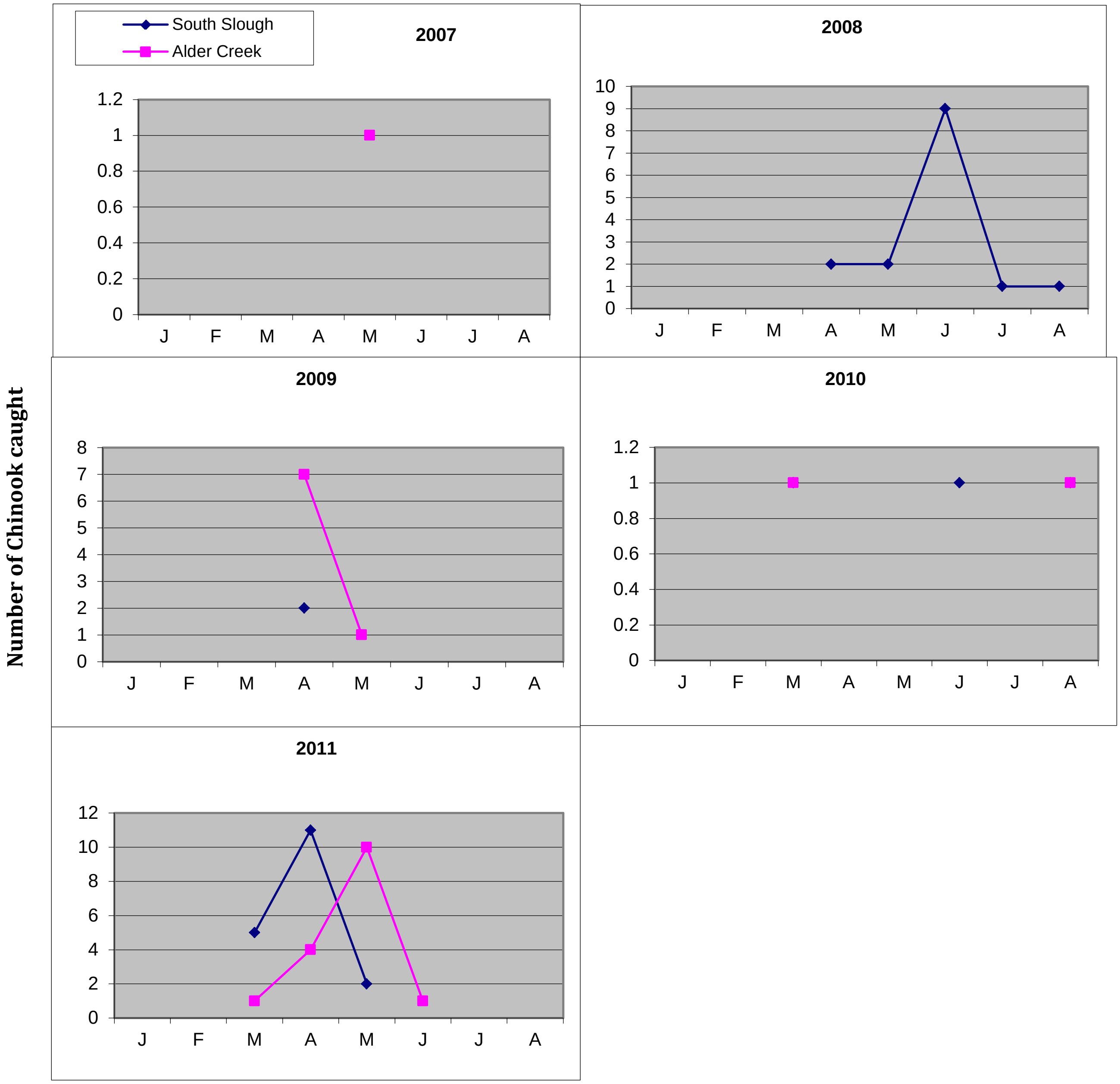


Figure 59. Temporal distribution of unmarked Chinook at South Slough and Alder Creek, 2007- 2011.

5.3.6.4 Size, Weight & Condition (K) Factor

The size of salmonids in terms of mean length and weight varied throughout the months and years of monitoring. The cumulative mean length of 5 years of Chinook, coho and chum data revealed that Chinook at South Slough averaged 71.8 mm while Chinook at Alder Creek averaged 64.7 mm; Coho at South Slough averaged 61.8 mm which was slightly smaller than coho observed at Alder Creek, which averaged 64.2 mm; Chum were very similar in size between the two sites averaging 45.1 mm at South Slough and 46 mm at Alder Creek. With the exception of Chinook, both sites produced similar mean lengths when averaged over the 5 year monitoring period.

No consistent trend presented itself from the size class data for juvenile coho and Chinook. Some years demonstrated a net increase in size across the season and other years demonstrated a net decrease in size. During 2008, 2010 and 2011 coho size increased between May and August at South Slough. At Alder Creek the only year coho exhibited a growth trend was in 2011. In all previous years at Alder Creek coho were either only present during one month of the sampling season or demonstrated a decrease in average size. Chinook at South Slough varied in size across the years, with larger individuals observed in April, and then again in July and August. However, between these months smaller individuals were caught. Similarly, in 2007 at South Slough and in 2009 at Alder Creek larger sized coho were observed early in the sampling season, followed by the presence of smaller sized coho. The change from larger individuals to smaller sized Chinook suggests the presence of different life history strategies utilizing South Slough throughout the year.

Coho sampled in South Slough in 2011 had a change in mean length from 38.5 mm to 60.8 mm from April to July, equivalent to a +22.3 mm change before leaving this habitat by August. Alder Creek coho demonstrated a similar trend in 2011 with a mean length changing from 43.5mm to 65.5 mm from March to July, equivalent to a +22 mm change in mean length before leaving the site by August. 2007 pre-restoration monitoring at South Slough revealed a change in coho mean lengths from 43.4 mm to 80.3 between March and June, equivalent to a +36.9 mm change before coho became absent from the system. Interestingly, pre-restoration monitoring at South Slough revealed an average mean length of coho at 76.7 mm, while cumulative post-restoration monitoring revealed an average of 58.1 mm. This may be attributed to a variety of factors such as; elevated velocity pre-restoration limiting access (tide gates and higher flows), sampling gear efficiency, statistical outliers or anomalies due to pre restoration reduced sample size (9 coho caught prior to and 700+ post restoration), etc.

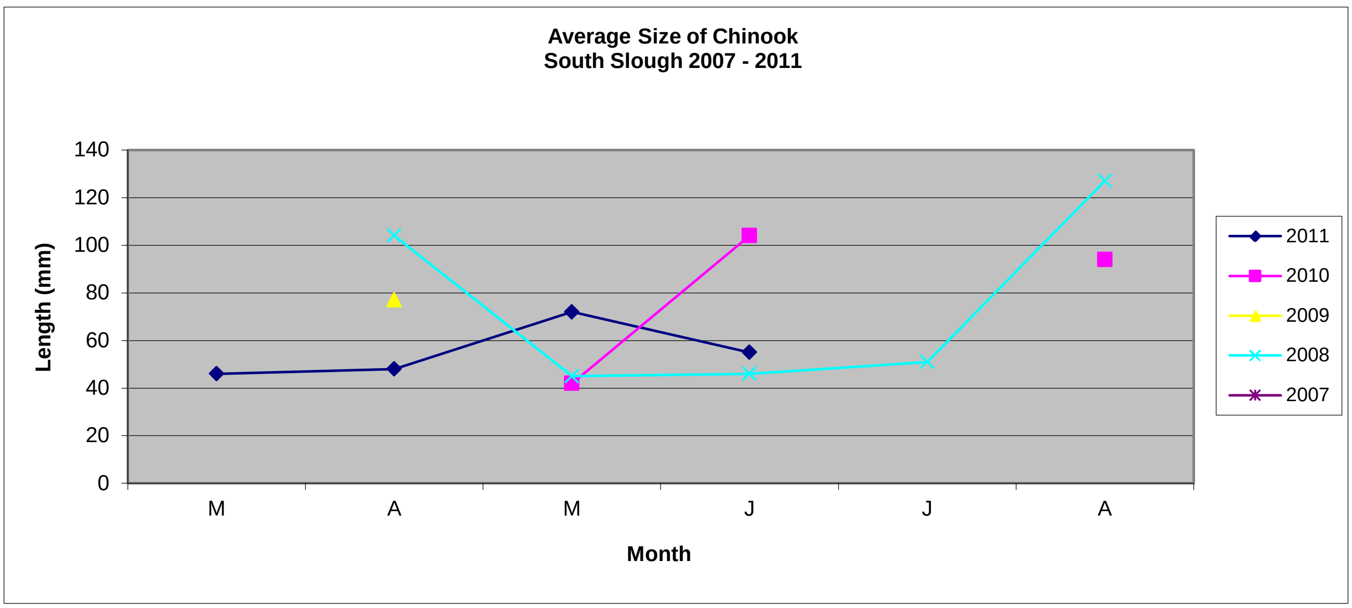


Figure 60. Average size by month of unmarked Chinook at South Slough, 2007 – 2011.

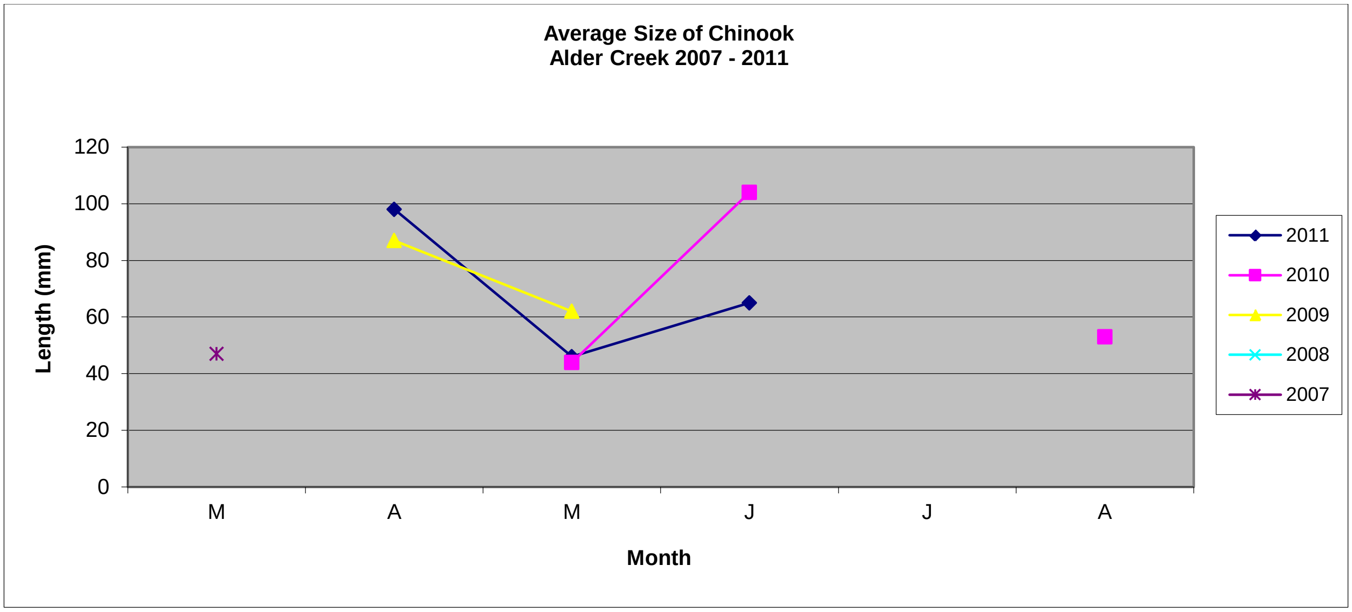


Figure 61. Average size by month of unmarked juvenile Chinook at Alder Creek, 2007 – 2011.

The presence of different size classes at South Slough and Alder Creek suggest that both stream type Chinook and coho are present. From January through April larger sized Chinook and coho were present (104-114 mm). Based on their size and timing these are most likely stream type yearlings utilizing the shallow water habitat on their way to the estuary to begin smoltification. The smaller sized Chinook, chum, and coho utilizing the habitat are likely a variety of ocean and stream type life history strategies, utilizing the habitat as they both migrate to the estuary at small sizes or search for forage and refuge during their prolonged stay in freshwater.

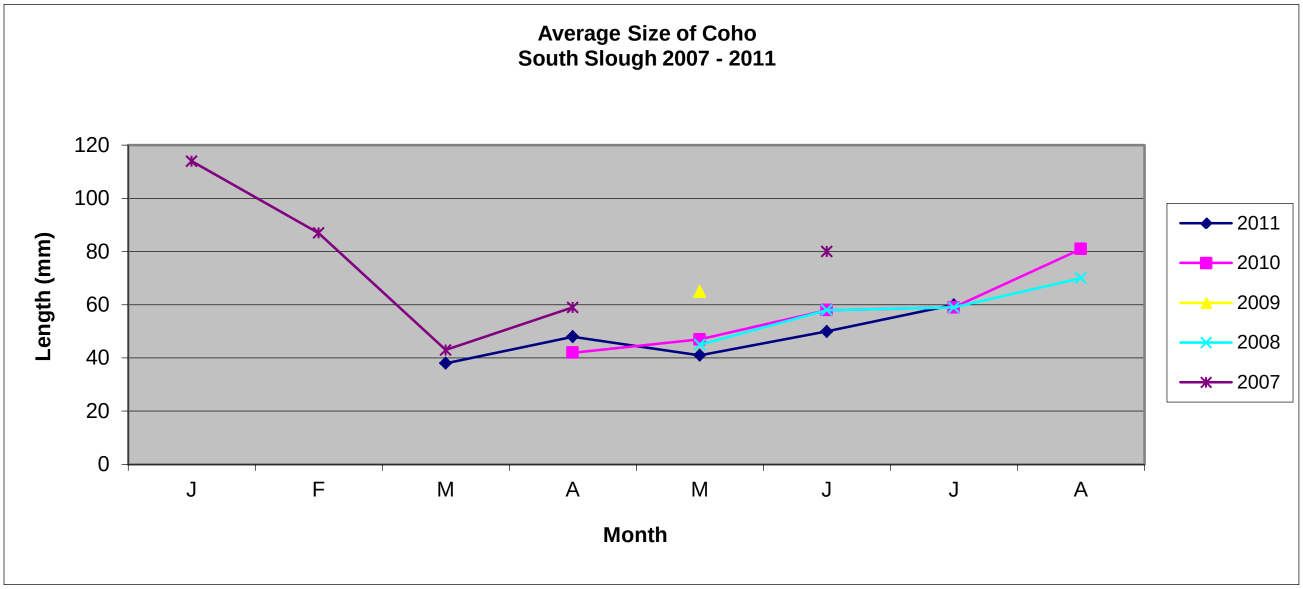


Figure 62. Average size of unmarked coho by month at South Slough, 2007 – 2011.

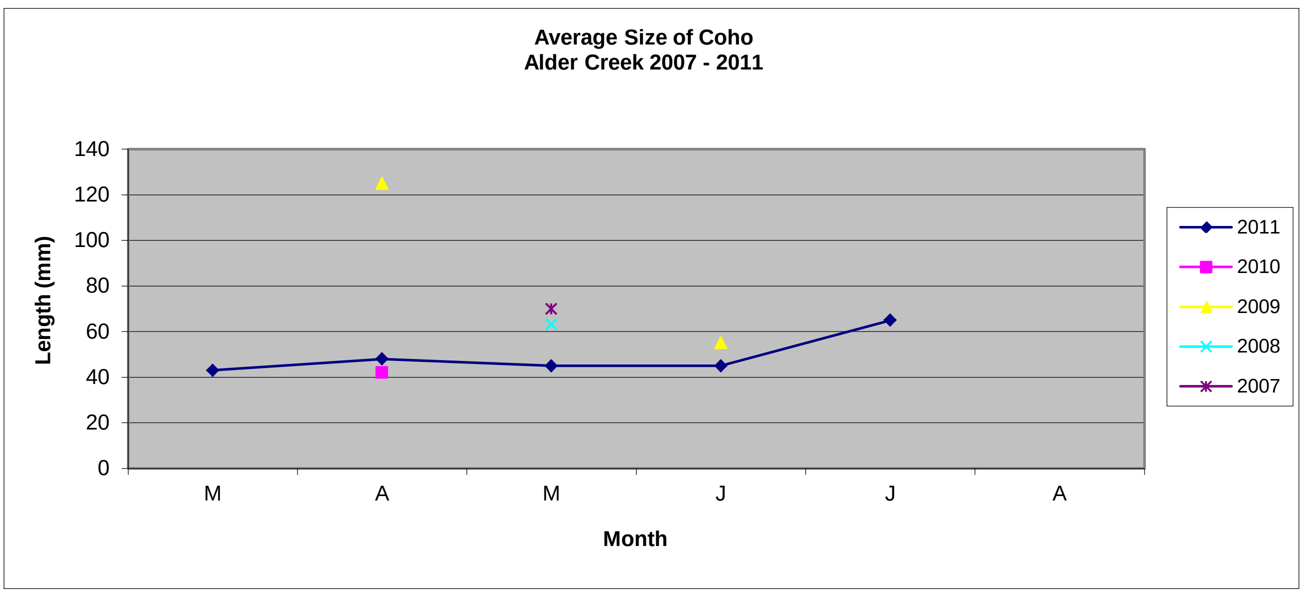


Figure 63. Average size of unmarked coho by month at Alder Creek, 2007 – 2011.

Table 36. Mean (SD) length of unmarked juvenile Chinook, coho, and chum between 2007 and 2011 at South Slough and Alder Creek, 2007 - 2011. *^Only one individual caught/month; dash indicates no event.*

		Mean (SD) length (mm) juvenile Chinook, coho, and chum														
		Chinook					**Coho**					**Chum**				
		2007	2008	2009	2010	2011	2007	2008	2009	2010	2011	2007	2008	2009	2010	2011
South Slough	Jan	0	0	-	-	-	114^	0	-	-	-	0	0	-	-	-
	Feb	0	0	0	-	-	87^	0	0	-	-	0	0	0	-	-
	Mar	0	0	0	-	46^	43.44 (±8.2)	0	0	-	38.5 (±1.9)	0	45 (±3)	48 (±1.4)	-	41^
	Apr	0	104 (±8.5)	77^	0	48 (±1.05)	59^	0	0	42.8 (±16.4)	48.5 (±24.7)	0	43 (±4.7)	52.7 (±3.2)	0	0
	May	0	45.5 (±4.9)	0	42^	72 (±33.5)	0	45.8 (±5.2)	0	47^	41.8 (±6.72)	-	0	0	0	0
	Jun	0	46.8 (±3.3)	0	104^	55.5 (±3.5)	80.3 (±3.2)	58.2 (±4.6)	65.5 (±16.2)	58.9 (±2.7)	50.8 (±8.89)	0	0	0	0	0
	Jul	-	51^	0	0	0	-	69.2 (±7.3)	-	59.7 ±0.9	60.83 ±5.36	-	0	0	0	0
	Aug	-	127^	-	94^	-	-	70 (±10.4)	-	81.3 (±2.2)	-	-	0	-	0	-
Alder Creek	**Jan**	-	0	-	-	-	-	0	-	-	-	-	-	-	-	-
	Feb	-	0	-	-	-	-	0	-	-	-	-	-	-	-	-
	Mar	-	0	-	-	0	-	0	-	-	43.5 (±5.3)	-	-	-	-	0
	Apr	0	0	87.3 (±39.1)	0	98.8 (±31.2)	0	0	125.5 (±3.5)	42.5 (±2.1)	48.6 (±26.9)	46^	0	0	0	0
	May	47^	0	62	44^	46.5 (±20.7)	70^	63.8 (±32.2)	0	0	45.5 (±19.1)	0	0	0	0	0
	Jun	0	0	0	104^	65^	0	0	55^	0	45.4 (±19.2)	0	0	0	0	0
	Jul	-	0	0	0	0	-	0	0	0	65.5 (±8.7)	-	0	0	0	0
	Aug	-	0	-	53^	-	-	0	-		-	-	0	-	0	-

Chinook salmon captured within the 5 years of monitoring exhibited virtually identical weights on average between South Slough and Alder Creek (6.28g and 6.29g respectively). It can be seen in the data that there are significantly reduced abundance of coho at Alder Creek in comparison to South Slough (Figure 59). Coho sampled at South Slough in 2010 had a change in mean weight; from 1.3g to 2.74g from April to July, equivalent to a +1.44g change in mean weight before leaving the site in August. In 2011, coho sampled within South Slough revealed a +2.11g change in mean weight before leaving the site in August. In 2011, Alder Creek demonstrated a +2.71g change in mean weight from March through July. Mean length and weight have significant implications on habitat suitability. Unfortunately mean weight data cannot be compared to pre-restoration data due to the fact that weights were not a parameter in the 2007 sampling season. These changes may, or may not accurately reflect growth, because we do not know whether larger or smaller fish may have differentially moved into or out of the sample reaches.

Table 37. Mean (SD) weight of unmarked juvenile Chinook, coho, and chum between 2008 and 2011 at South Slough and Alder Creek. (**Indicates no weight taken, ^indicates only one fish caught).*

	Mean (SD) weight (g) juvenile Chinook, coho, and chum												
		Chinook				**Coho**				**Chum**			
		2008	2009	2010	2011	2008	2009	2010	2011	2008	2009	2010	2011
South Slough	Jan	0	-	-	-	0	-	-	-	0	-	-	-
	Feb	0	0	-	-	0	0	-	-	0	0	-	-
	Mar	0	0	-	0.76^	0	0	-	0.49 (±0.11)	0.5 (±0.14)	0.95 (±0.07)	-	0.31^
	Apr	11.9 (±1.35)	*	0	0.94 (±0.85)	0	0	1.3 (±3.2)	2.55 (±6.12)	0.47 (±0.15)	0.97 (±0.29)	0	0
	May	*	0	0.51^	7.92 (±0.58)	1.03 (±0.60)	0	1.28^	1.06 (±1.01)	0	0	0	0
	Jun	0.94 (±0.23)	0	13.8	1.74 (±0.19)	2.24 (±0.76)	4.0 (±3.3)	2.51 (±0.47)	1.47 (±1.31)	0	0	0	0
	Jul	1.1^	0	0	0	3.99 (±1.59)	-	2.74 (±0.99)	2.6 (±0.86)	0	0	0	0
	Aug	18.2^	-	9.6^	-	4.53 (±0.91)	-	6.6 (±0.83)	-	0	-	0	-
Alder Creek	**Jan**	-	-	-	-	-	-	-	-	-	-	-	-
	Feb	-	-	-	-	-	-	-	-	-	-	-	-
	Mar	-	-	-	0	-	-	-	0.89 (±0.61)	-	-	-	0
	Apr	*	10.4 (±11.3)	0	13.78 (±8.6)	0	22.6 (±2.1)	*	0.49 (±0.13)	0	0	0	0
	May	*	2.09^	8.22^	5.68 (±8.24)	*	0	0	0.50 (±0.16)	0	0	0	0
	Jun	0	0	-	2.65^	0	1.77^	0	3.55 (±3.3)	0	0	0	0
	Jul	0	0	-	0	0	0	0	3.6 (±1.5)	0	0	0	0
	Aug	0	-	2.3^	-	0	-	-	-	0	-	0	-

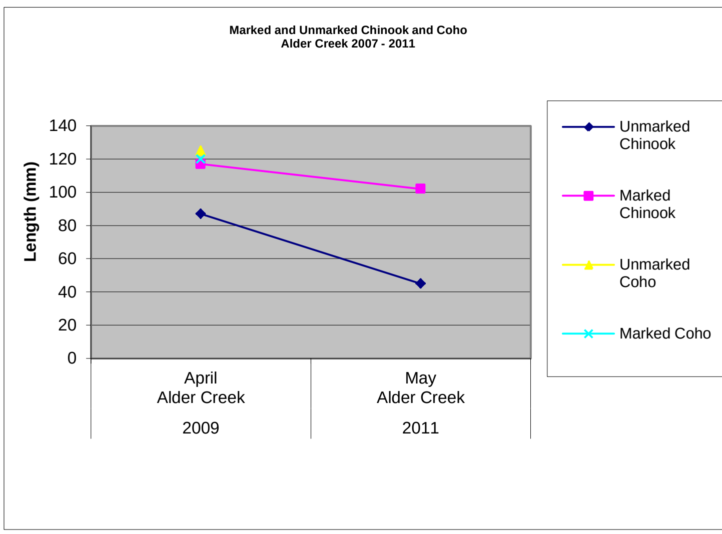
Table 38. Mean (SD) condition factor of juvenile Chinook, coho, and chum calculated from length and weight, between 2007 and 2011 at South Slough and Alder Creek. **Not enough data to calculate condition factor*

	Mean (SD) Condition Factor for juvenile Chinook, coho, and chum															
		Chinook					**Coho**					**Chum**				
		2007	2008	2009	2010	2011	2007	2008	2009	2010	2011	2007	2008	2009	2010	2011
South Slough	Jan	*	0	-	-	-	*	0	-	-	-	*	0	-	-	-
	Feb	*	0	0	-	-	*	0	0	-	-	*	0	0	-	-
	Mar	*	0	0	-	0.78*	*	0	0	-	0.87 (±0.16)	*	0.53 (±0.13)	0.86 (±0.02)	-	0.45 *
	Apr	*	1.07 (±0.71)	0.99^	0	0.77 (±0.38)	*	0	0	1.07 (±0.42)	0.87 (±0.34)	*	0.55 (±0.11)	0.65 (±0.09)	0	0
	May	*	*	0	*	0.91 (±0.12)	*	0.97 (±0.21)	0	1.23	0.93 (±0.21)	*	0	0	0	0
	Jun	*	0.92 (±0.15)	0	*	1.02 (±0.08)	*	1.12 (±0.33)	1.23 (±0.24)	1.23 (±0.18)	1.0 (±0.19)	*	0	0	0	0
	Jul	*	0.82^	0	0	0	*	1.14 (±0.12)	-	1.25 (±0.09)	1.12 (±0.14)	*	0	0	0	0
	Aug	*	0.89^	-	*	-	*	0.48 (±0.91)	-	1.23 (±0.11)	-	*	0	-	0	-
Alder Creek	**Jan**	*	-	-	-	-	*	-	-	-	-	*	-	-	-	-
	Feb	*	-	-	-	-	*	-	-	-	-	*	-	-	-	-
	Mar	*	-	-	-	0	*	-	-	-	0.98 (±0.26)	*	-	-	-	0
	Apr	*	*	0.94 (±0.15)	0	1.18 (±0.04)	*	0	1.14 (±2.12)	0.86 (±0.69)	0.95 (±0.24)	*	0	0	0	0
	May	*	*	0.88^	*	1.08 (±0.67)	*	*	0	0	0.81 (±0.18)	*	0	0	0	0
	Jun	*	0	0	0	0.96^	*	0	1.06^	0	1.09 (±0.17)	*	0	0	0	0
	Jul	*	0	0	0	0	*	0	0	0	1.24 (±0.09)	*	0	0	0	0
	Aug	*	0	-		-	*	0	-		-	*	0	-	0	-

Although salmonids sampled at South Slough from 2007 through 2011 were not consistently greater in length, they had the highest condition factor during most months sampled (Table 38). Coho had the highest condition factor within both sites. Chum sampled at South Slough had the lowest condition factor. The majority of salmon observed at South Slough and Alder Creek were unmarked fish. Marked Chinook were observed in only 4 sampling events over a 5 year period, and hatchery coho during just 1 event. With the exception of June 2009 unmarked Chinook and coho were present at the same time marked stock was observed. Marked origin juvenile salmon are generally larger in size than unmarked juvenile salmon due to the nature of their artificial environment (i.e. regular feeding, protection from predation). Average size and weight (SD) were compared between marked and unmarked Chinook and coho for the months and years hatchery fish were caught. Marked Chinook had a larger average size and weight than unmarked Chinook at both sites; marked coho, however, exhibited a smaller average size and weight than unmarked coho observed during the same month and year.

Table 39. Comparison of lengths and weights for marked and unmarked Chinook at South Slough and Alder Creek. **Only one fish caught.*

Unmarked and Marked Chinook Lengths and Weights (SD) at South Slough and Alder Creek										
		2009					**2011**			
	Year	**Length (SD)**		**Weight (SD)**			**Length (SD)**		**Weight (SD)**	
	Month	**Unmarked**	**Marked**	**Unmarked**	**Marked**		**Unmarked**	**Marked**	**Unmarked**	**Marked**
South Slough	**May**					**May**	72 (±33.5)	98.5 (±12.02)	7.92 (±0.58)	9.15 (±3.75)
	June	N/A	91.5 (±7.78)	N/A	8.63 (±2.02)	**June**				
Alder Creek	**April**	87.3 (±39.1)	117 (±26.91)	10.4 (±11.3)	15.67 (±13.54)	**April**				
	May					**May**	45.5 (±19.1)	102*	5.68 (±8.24)	14.10*



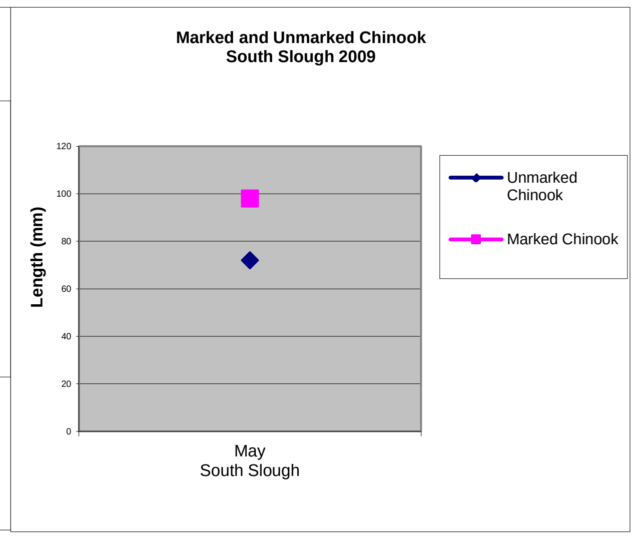


Figure 64. Lengths (mm) of marked and unmarked Chinook and coho at South Slough and Alder Creek, 2007 – 2011.

Table 40. Comparison of lengths and weights (SD) for hatchery/marked and unmarked coho at South Slough and Alder Creek.

Unmarked and Hatchery Coho Lengths and Weights (SD) at South Slough and Alder Creek					
	Year	**2009**			
		Length (SD)		**Weight (SD)**	
	Month	**Unmarked**	**Hatchery**	**Unmarked**	**Hatchery**
Alder Creek	**April**	125 (+3.5)	120.5 (±16.26)	22.3 (±2.1)	16.45 (±9.77)

5.3.6.5 Catch Per Unit Effort (CPUE)

CPUE is generally calculated for fish sampling conducted by beach seine as number of fish caught divided by the area fished in m^2. With the use of a trapnet however, the area fished includes the entire upstream portion of the channel above the net, making an accurate account of fish density difficult to calculate. Sampling inconsistency compounds this matter as it was conducted once a month during some years and twice a month during others. In order to standardize the data collected CPUE is calculated within this section as number of fish caught divided by hours fished. 2011 was excluded from comparison of previous years because the altering methods implemented were different making it statistically invalid in relation to prior data collected. In 2008 at Alder Creek, seining was used in place of trapnetting. As a result CPUE calculations South Slough. 2008 CPUE is only included in the South Slough data, excluding Alder Creek data in the following figures. CPUE was calculated for unmarked salmon only.

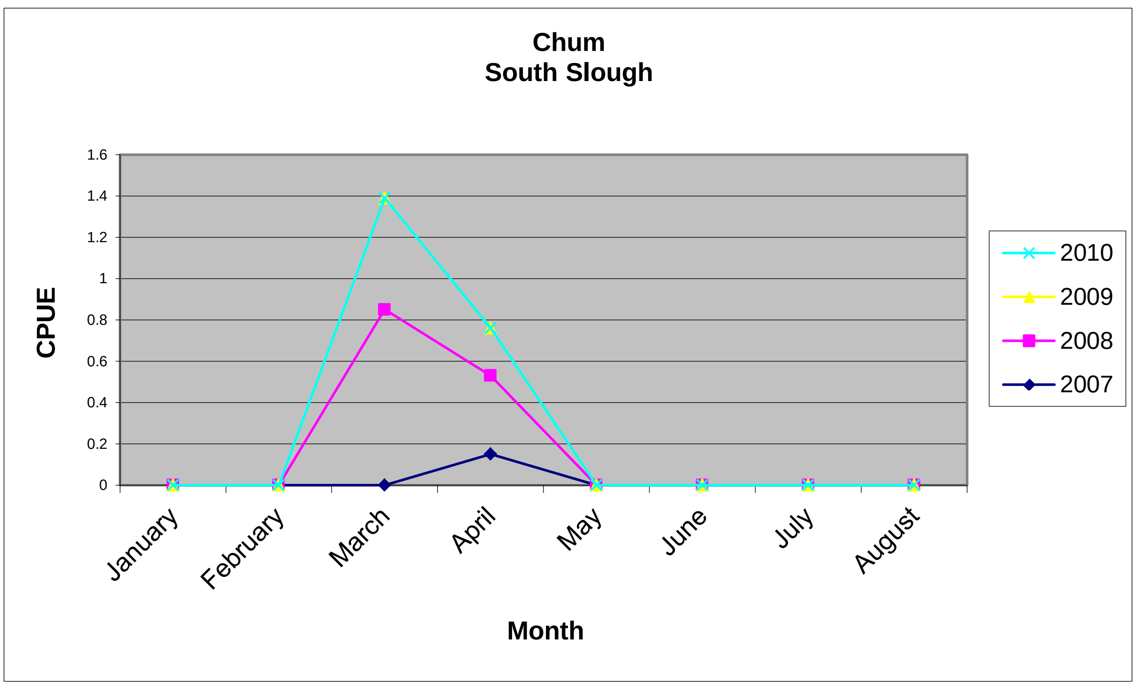


Figure 65. CPUE for chum at South Slough, 2007 – 2010.

CPUE was highest for chum in March and April in all years. Between 2007 and 2010, the CPUE for Chinook and coho was highest in June of 2010. At Alder Creek during 2007, 2009 and 2010 CPUE for unmarked Chinook was highest in May of 2007 and April in both 2009 and 2010. CPUE for unmarked coho at Alder Creek between these same years was highest during April of 2010.

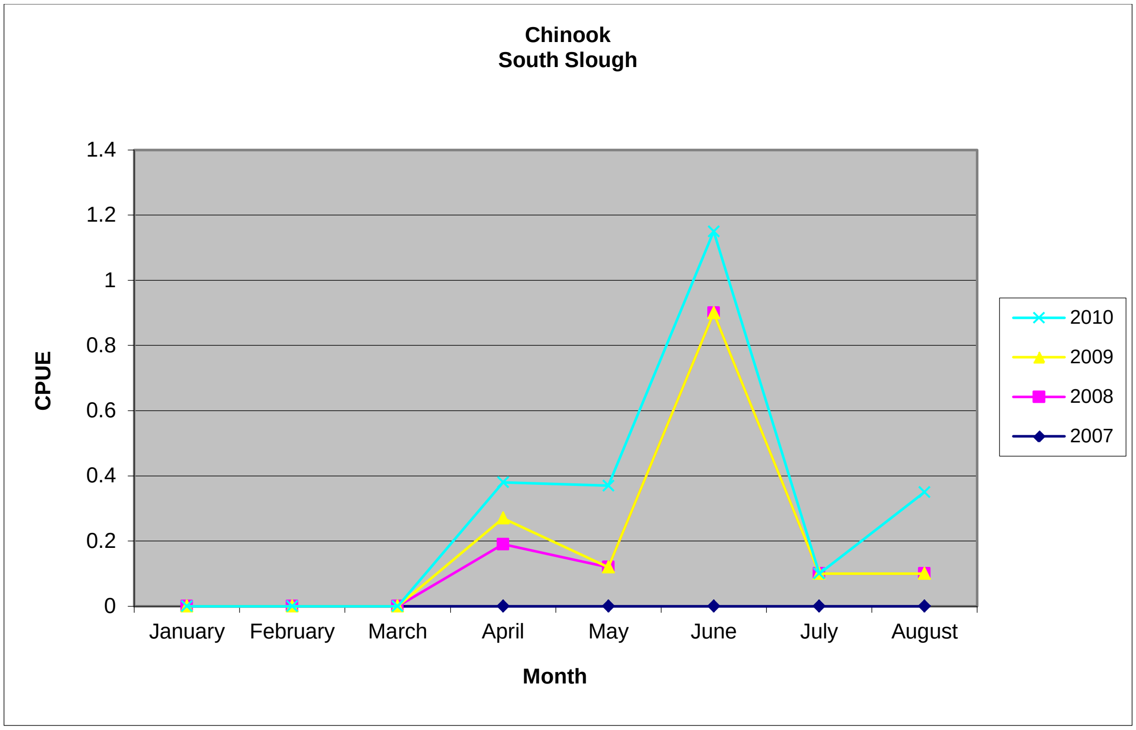


Figure 66. CPUE for unmarked Chinook at South Slough, 2007 – 2010.

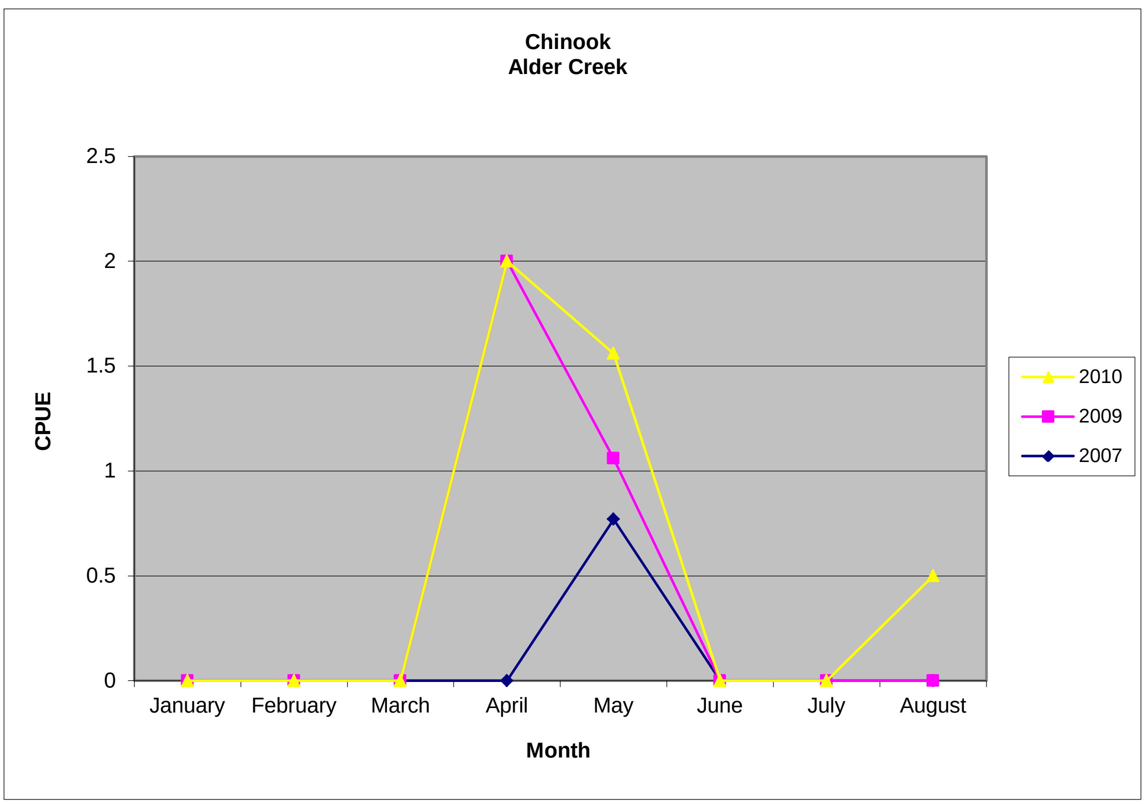


Figure 67. CPUE for unmarked Chinook at Alder Creek, 2007, 2009 & 2010.

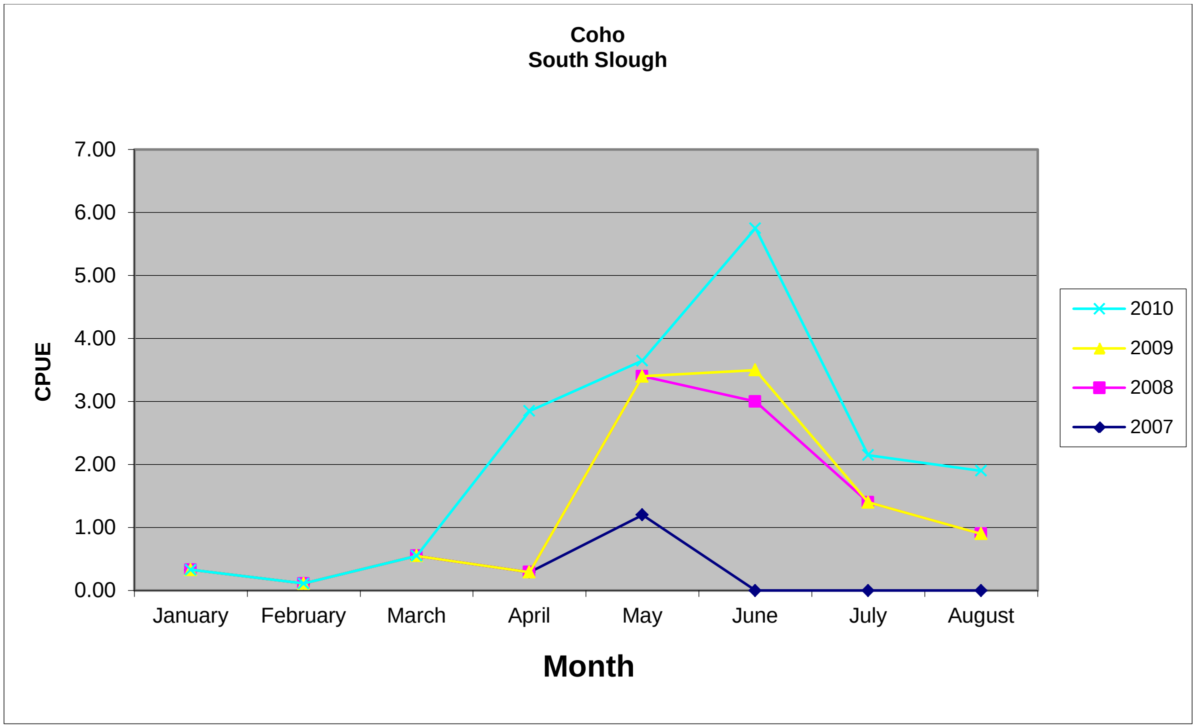


Figure 68. CPUE for unmarked coho at South Slough, 2007 – 2010.

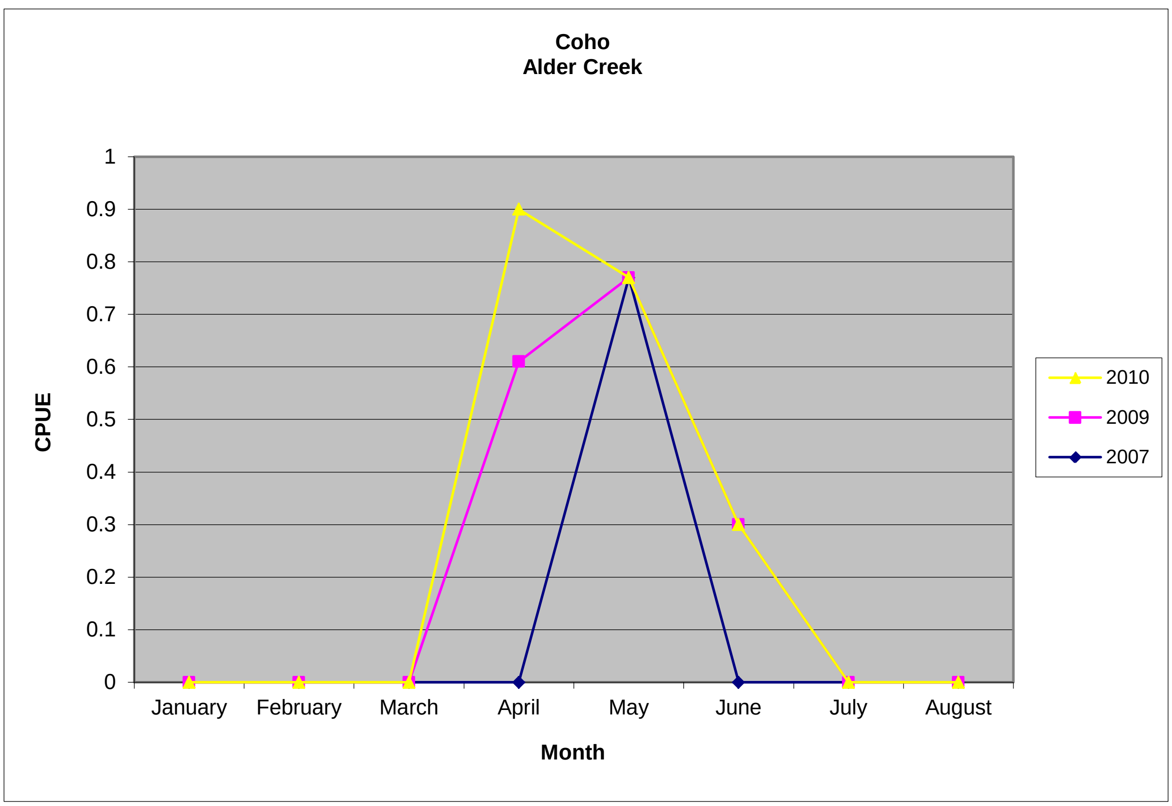


Figure 69. CPUE for unmarked coho at Alder Creek, 2007, 2009 & 2010.

Abundance of juvenile coho and Chinook was generally greater at South Slough. This may be due to the differences in habitat conditions at South Slough. The channel has a greater and more consistent freshwater input than Alder Creek, and the elevation appears to be lower allowing the channel to hold several feet of water at low tide while the Alder Creek channel drains to several inches. Greater depth theoretically increases habitat opportunity, supports healthier water quality conditions and shaded freshwater input provides lower temperatures and higher dissolved oxygen levels during low tide.

Table 41. Catch Per Unit Effort for Chinook, coho, and chum at South Slough and Alder Creek, 2007 - 2010 (measured by fish caught per hour). *Only unmarked salmon included.*

	CPUE for juvenile Chinook, coho, and chum*												
		Chinook					**Coho**					**Chum**	
		2007	**2008**	**2009**	**2010**	**2007**	**2008**	**2009**	**2010**	**2007**	**2008**	**2009**	**2010**
South Slough	Jan					0.33							
	Feb					0.11							
	Mar					0.55					0.73		
	Apr		0.19	0.08	0.10	0.15			3.70		0.39		
	May		0.12		0.30		2.21		0.30				
	Jun		0.91	0.50	0.30	1.20	3.54	0.50	2.30				
	Jul		0.10				1.41		0.80				
	Aug		0.10		0.30		0.91		1.00				
Alder Creek	Jan												
	Feb												
	Mar												
	Apr			3.0				1.20					
	May	0.80		0.29	0.50	0.80							
	Jun							0.31					
	Jul												
	Aug				0.50								

5.3.6.6 Species Diversity

Species diversity (Figure 70) as calculated by the Shannon-Weiner diversity index, was highest at Alder Creek each year throughout the 5 year monitoring period. A total of 15 different species were observed at South Slough, 5 of which were salmonids (Table 33, Table 34, and Table 35). A faintly elevated number in comparison to the 12 individual species observed at Alder Creek, 3 of which were salmonids. The lower indices observed at South Slough can partially be attributed to the larger numbers of three spine stickleback present in each catch. Between the two monitoring sites, three-spine stickleback comprise between 89% and 98% of the total catch on average. South Slough total catch averaged 98.3% Three-spine stickleback while Alder Creek averaged 92.5%. Consequently, while the number of different species present within the fish community at the restoration site was larger; the proportion of individual species within that community were dramatically uneven, resulting in a lower Shannon diversity index.

The number above each data plot in Figure 70 represents the total number of different species observed for that site and year. 2007 and 2008 are an example of how disproportionate numbers for individual species influence the species diversity value. In both 2007 and 2008 a greater diversity of species was observed at Sough Slough, however the diversity indices numbers are both lower for South Slough during those years. The indices values are in effect a reflection of the consistency in total abundance among all species caught. Three spine stickleback were generally caught in the thousands, while salmon and other native and non-native fish species were in the tens.

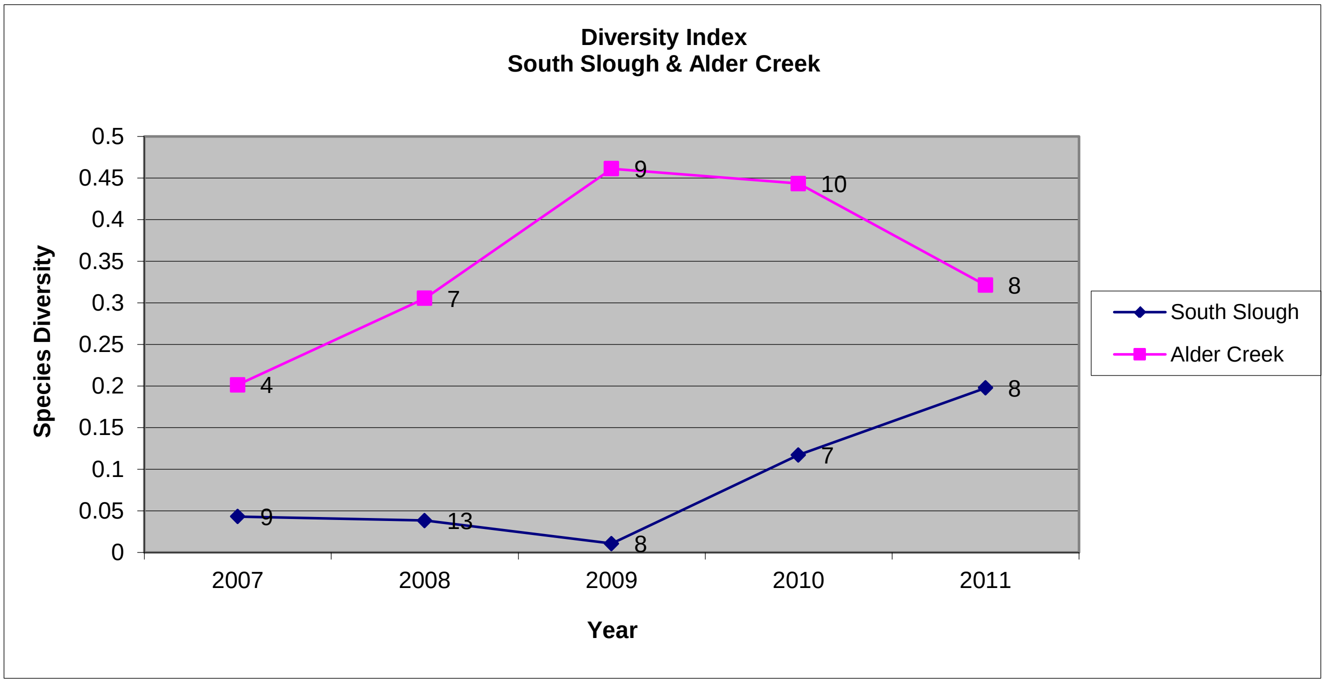


Figure 70. Diversity of fish community at South Slough and Alder Creek, 2007 – 2011. (Shannon-Weiner diversity index)

5.3.7 Habitat

5.3.7.1 Sediment Accretion 2012

Micro-topographic changes have been recorded at South Slough from 2008 to 2012 and at Alder Creek from 2009 to 2012. Sediment erosion and aggradation has occurred over time at varying rates throughout the monitoring period. Figure 71 below provides a visual representation of the micro-topographic trends occurring at South Slough and Alder Creek during this period of time. Each series line (years 2008-2012) within Figure 71 reflects the change in sediment elevation via erosion or aggradation that has occurred in comparison to measurements taken the previous year; i.e., series line 2008 represents the erosion and/or aggradation that has occurred since 2008. In comparison, Table 6 reflects the net change in sediment movement from 2011 to 2012 at each respective site.

Table 42. Sediment Accretion at South Slough and Alder Creek 2012.

Net Change in Sediment Movement at South Slough & Alder Creek Between 2011 & 2012.	
South Slough	**Alder Creek**
-7.7 (±1.79) cm	+7.2 (±1.35) cm

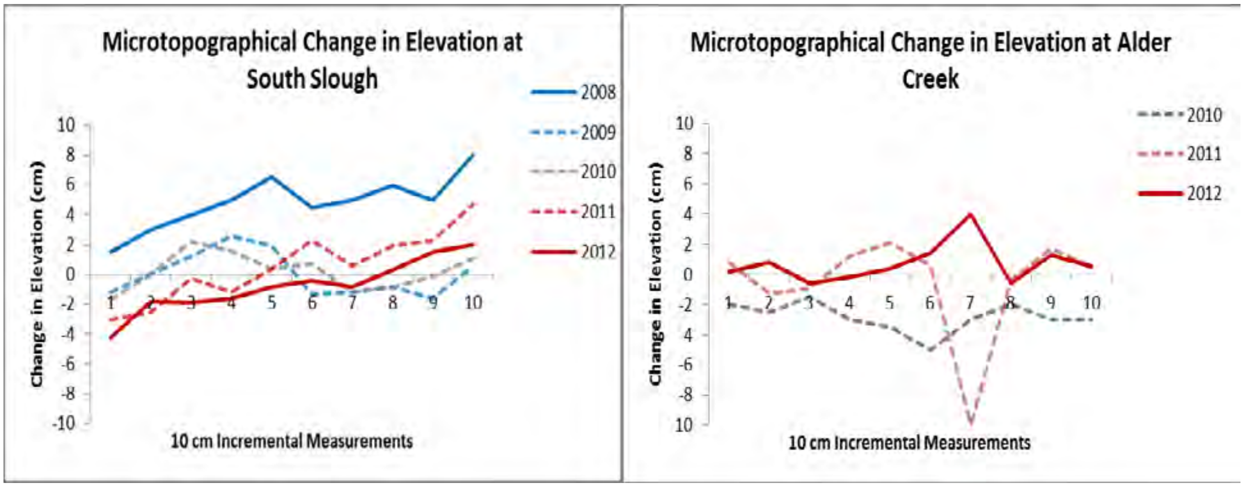


Figure 71. Sediment movement at South Slough and Alder Creek 2007 – 2012.

Sediment accretion measurements recorded at South Slough between 2008 and 2012 shown in Figure 71 reveal an overall net gain of sediment over the course of the monitoring period. With the exception of 2012 all other years have yielded a net gain (Table 42) in sediment movement. The Alder Creek sediment accretion station has experienced increased variability over the years in comparison to South Slough. From 2009 to 2011 Alder Creek has endured a net loss in sediment movement while in 2012 the site has experienced a net gain.

5.3.7.2 Sediment Accretion 2009-2011

Sediment accretion measurements were taken at both sites to measure the changes in soil erosion and/or aggradation along the bank, and to compare the changes in sediment transport between South Slough and Alder Creek. Micro-topographic changes have been seen at South Slough, with sediment eroding several centimeters in 2009, and aggrading in 2010 to 2011 (Table 43). The average change in soil depth was +2.2 (±1.49) cm in 2010, and +4.4 (±1.24) cm in 2011 at South Slough. Alder Creek experienced consistent aggradation at a rate of +28.5 (±0.97) cm per year between 2009 and 2010, and a mix of erosion and aggradation between 2010 and 2011 with an average change in soil depth of +5.7 (±3.49) cm per year.

Table 43. Sediment deposition/erosion, average (SD), changes at South Slough and Alder Creek, measured in centimeters.

Change in sediment deposition/aggradation in centimeters					
	South Slough			**Alder Creek**	
	2009	**2010**	**2011**	**2010**	**2011**
Mean (SD)	-0.2 (±1.49)	+2.2 (±1.24)	+4.4 (±2.39)	+2. 9 (±0.97)	+5.7 (±3.49)

5.3.7.3 Channel Morphology 2012

Figure 72 – Figure 81 below illustrate the channel width and depth starting downstream and progressing upstream at South Slough and Alder Creek. The cross sections start near the bridges at both sites where they connect to the Lewis & Clark River. In conjunction with year round freshwater input from the surrounding watershed the sub sea-level depth of the channel in South Slough allows the slough to hold water during all phases of the tidal cycle, including negative tides. This increases the habitat opportunity for ESA listed juvenile salmonids at South Slough. The channel demonstrated deepening near the bridge, likely the result of sediment buildup behind the tide gate moving with the reconnection of tidal connectivity. Overall the channel in South Slough has deepened

post restoration, providing increased slow moving shallow water habitat for salmonids. Combined with the prey availability the site has the potential to increase juvenile salmon viability by providing refugia and foraging opportunity.

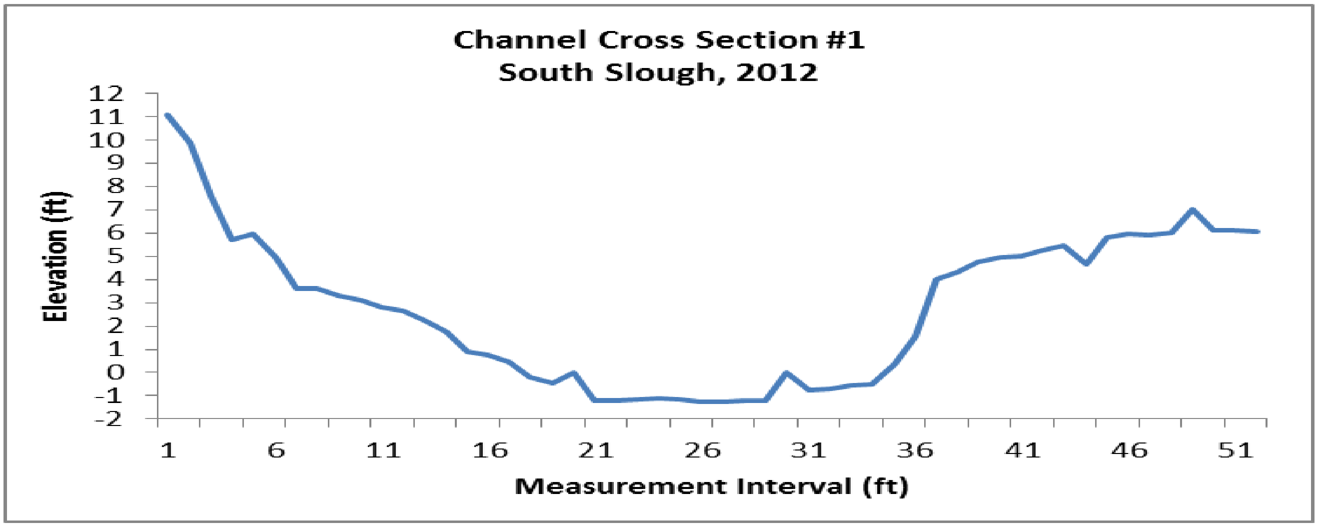


Figure 72. South Slough channel cross section #1, 2012.

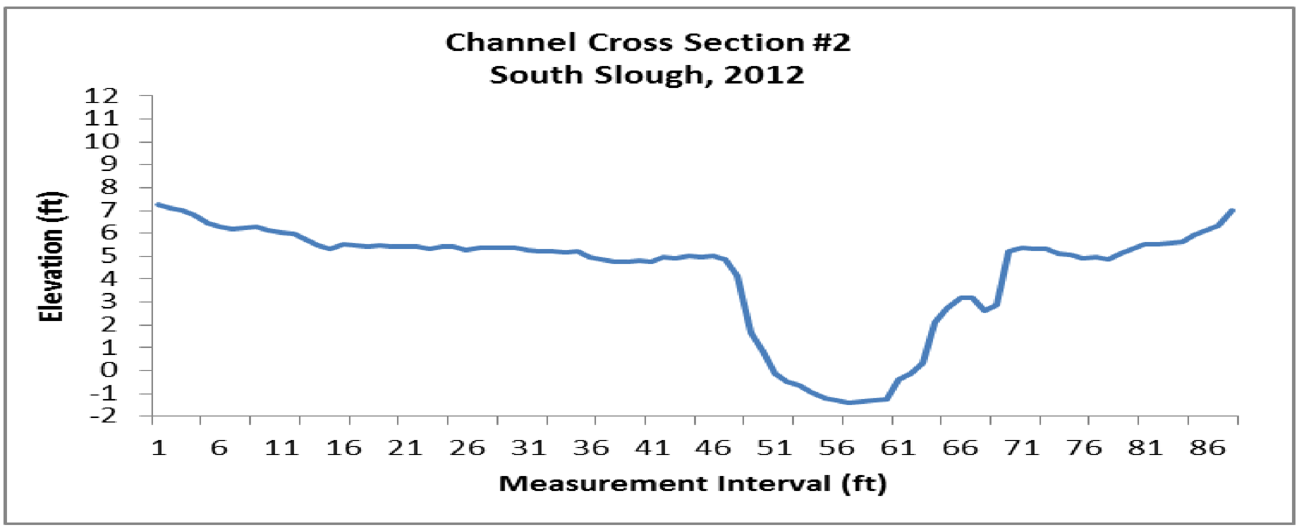


Figure 73. South Slough channel cross section #2, 2012.

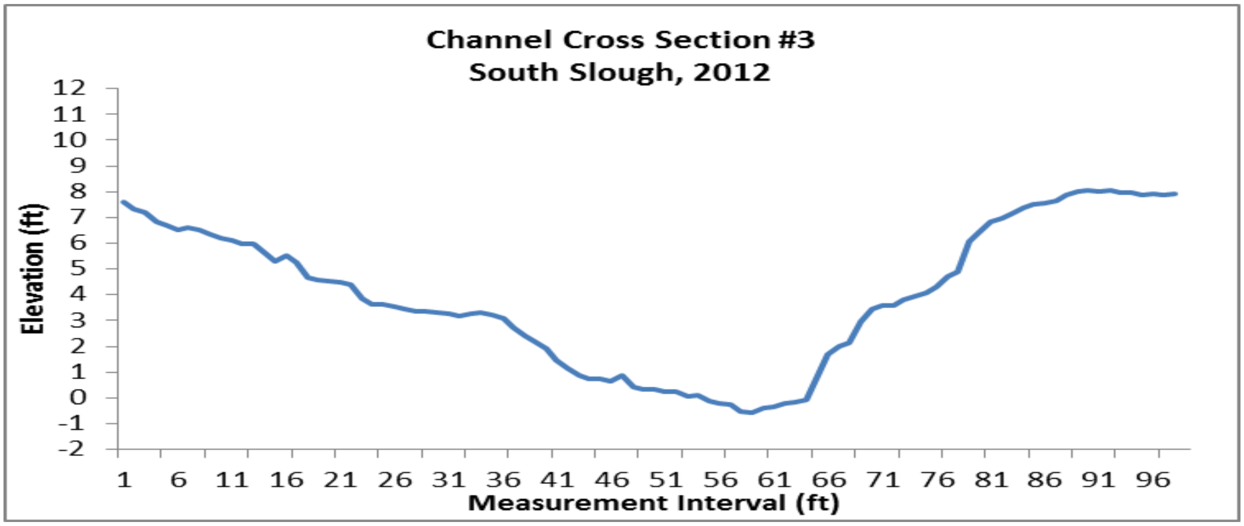


Figure 74. South Slough channel cross section #3, 2012.

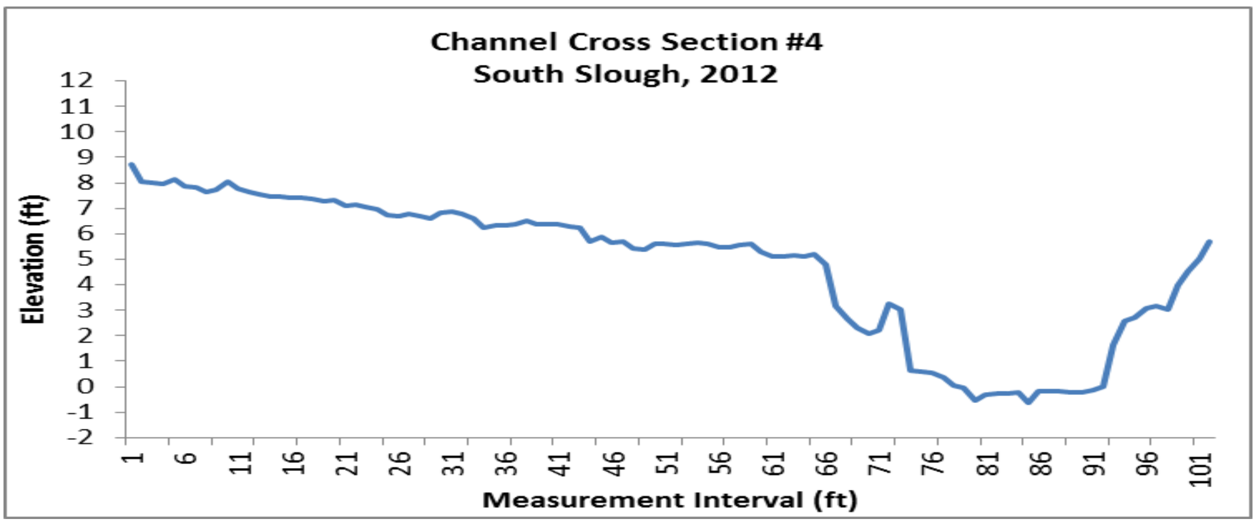


Figure 75. South Slough channel cross section #4, 2012.

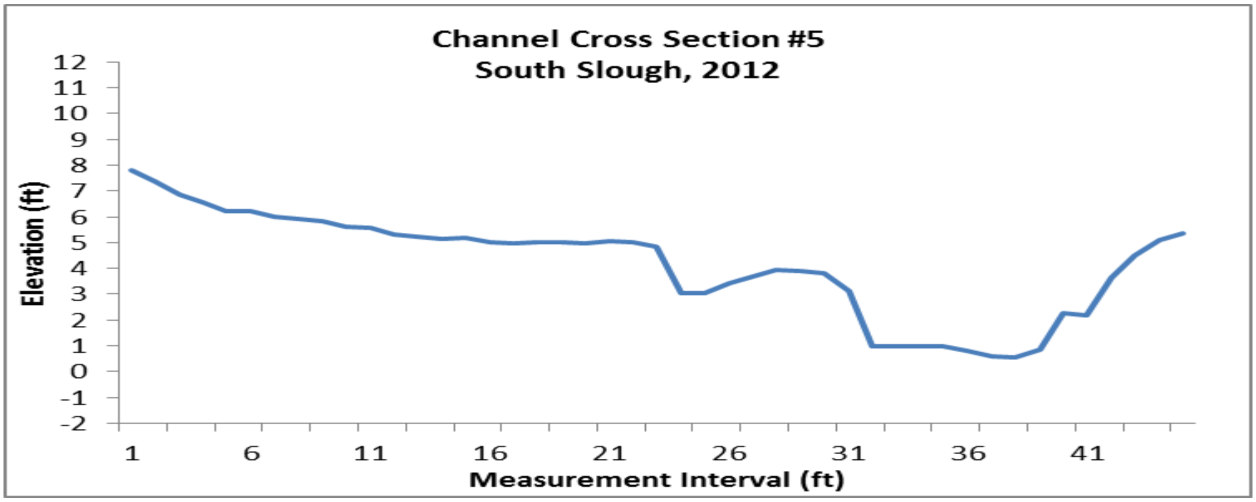


Figure 76. South Slough channel cross section #5, 2012.

Alder Creek channel cross sections reveal higher elevation channel depths in relation to those recorded at South Slough. The higher elevation of the channel at Alder Creek causes the channel to drain almost entirely during each of the two daily low tides. With the freshwater input provided by Alder Creek's watershed impounded by an undersized culvert, low tide events produce every shallow, reduced flow, high temperature water unsuitable to salmonids. As a result the habitat opportunity at Alder Creek is reduced compared to South Slough.

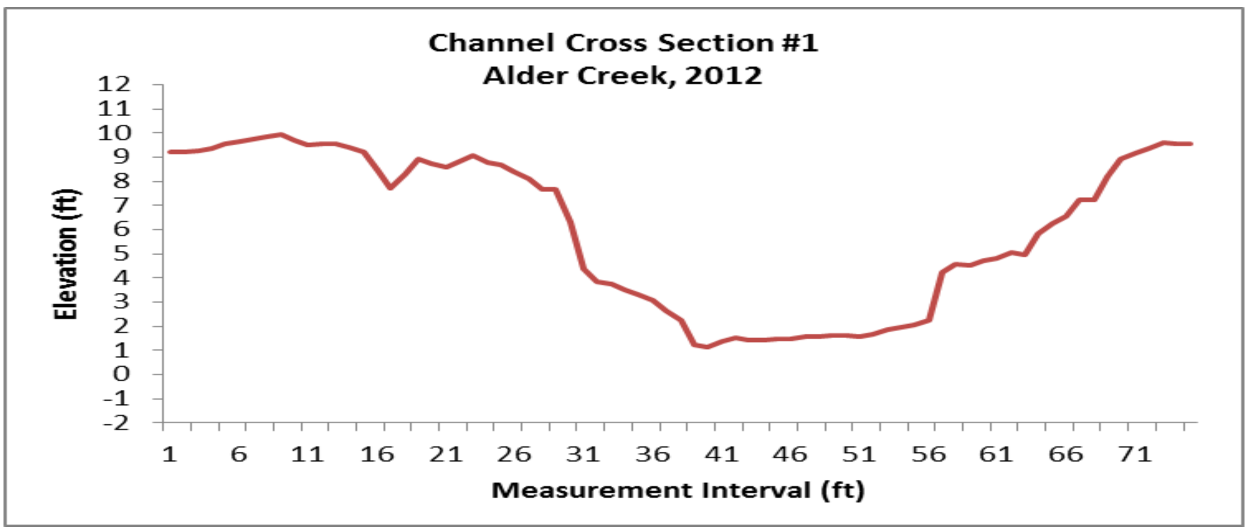


Figure 77. Alder Creek channel cross section #1, 2012.

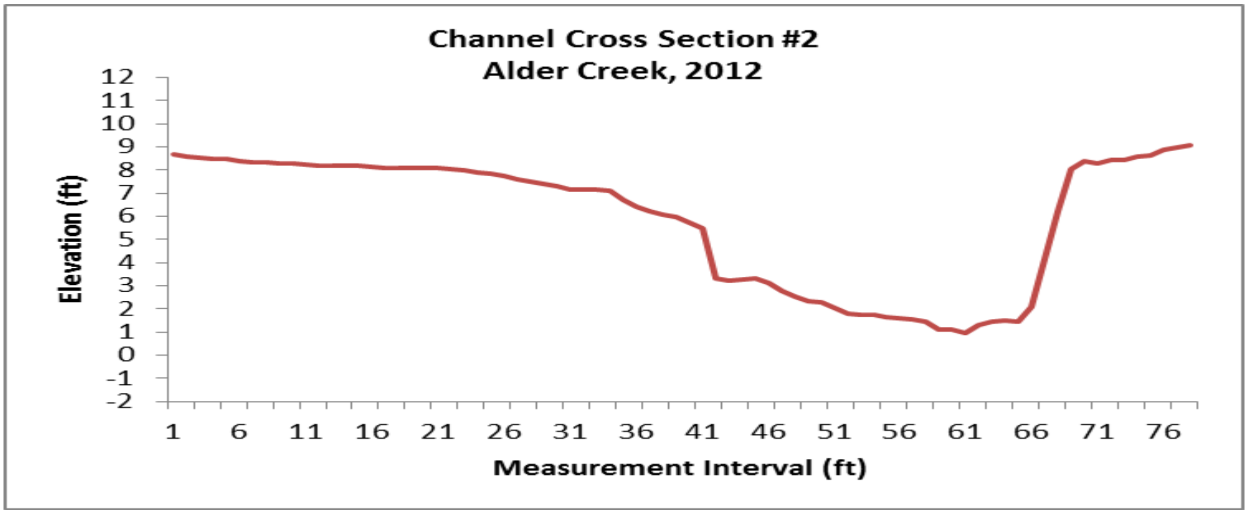


Figure 78. Alder Creek channel cross section #2, 2012.

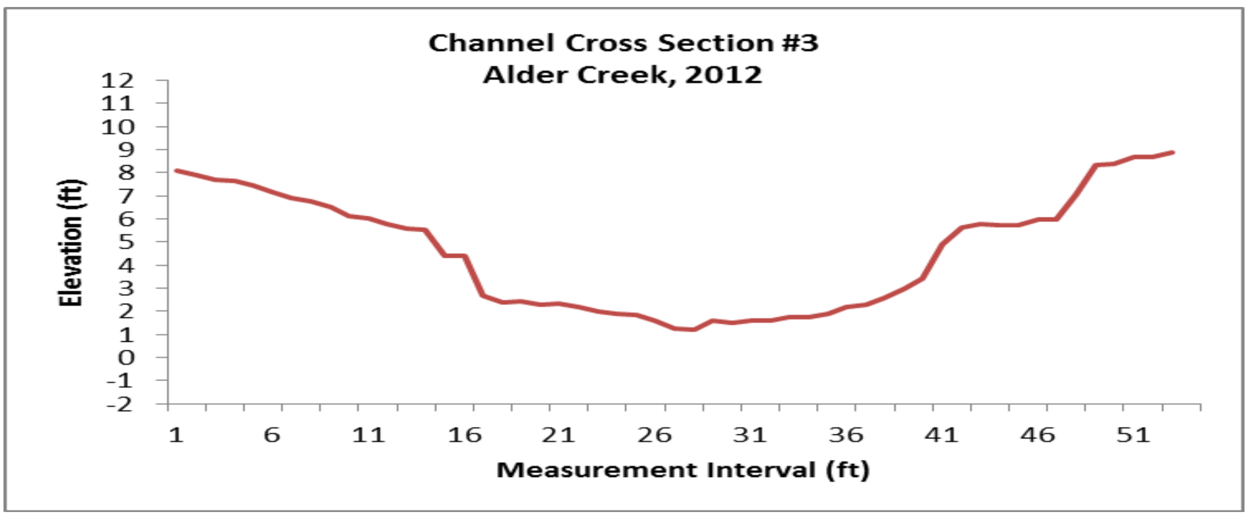


Figure 79. Alder Creek channel cross section #3, 2012.

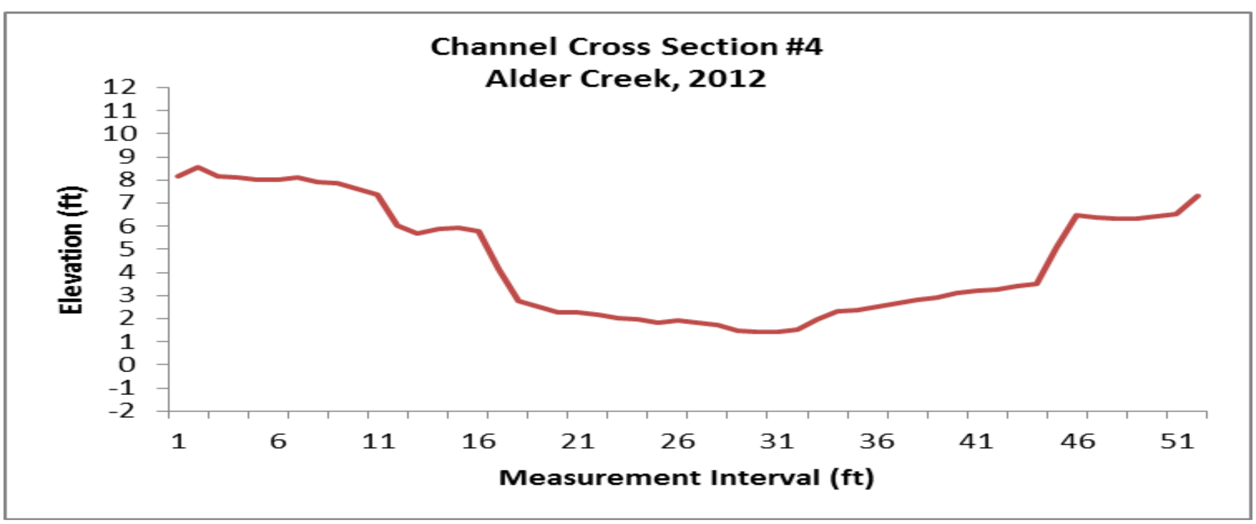


Figure 80. Alder Creek channel cross section #4, 2012.

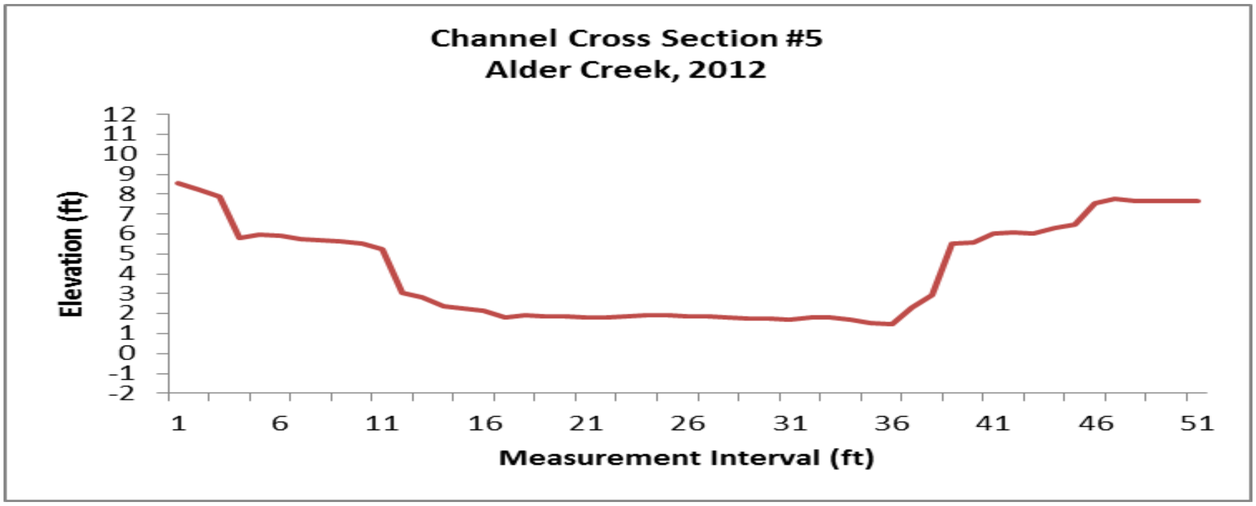


Figure 81. Alder Creek Channel cross section #5, 2012.

5.3.7.4 Channel Morphology 2007-2011

Measuring changes in channel depth and width reveals not only tidal impacts to off channel habitat but also the rate of successional change at tidal reconnection project sites. This assessment can be made through comparing a newly reconnected tidal channel to one that has experienced tidal influence for a long period of time (i.e. Alder Creek). The following series of figures (Figure 82 - Figure 92) illustrate the changes in channel width and depth starting downstream and working upstream. South Slough holds water at all times making it accessible to salmonids year round and during low and high water events, the lowest channel cross sections, those nearest the bridge, have undergone significant channel deepening, allowing the channel to hold more water during low water events which may result in improved access to the site. After restoration activities, access is present year round and has improved through a deeper channel at low water, which may potentially lower water velocity, as the head differential in the slough and outside is less. Lower water velocity theoretically allows small juvenile salmon easier access to the site. It should be noted however that water velocity measurements have not been taken and this hypothesis is based on the channel profile instead.

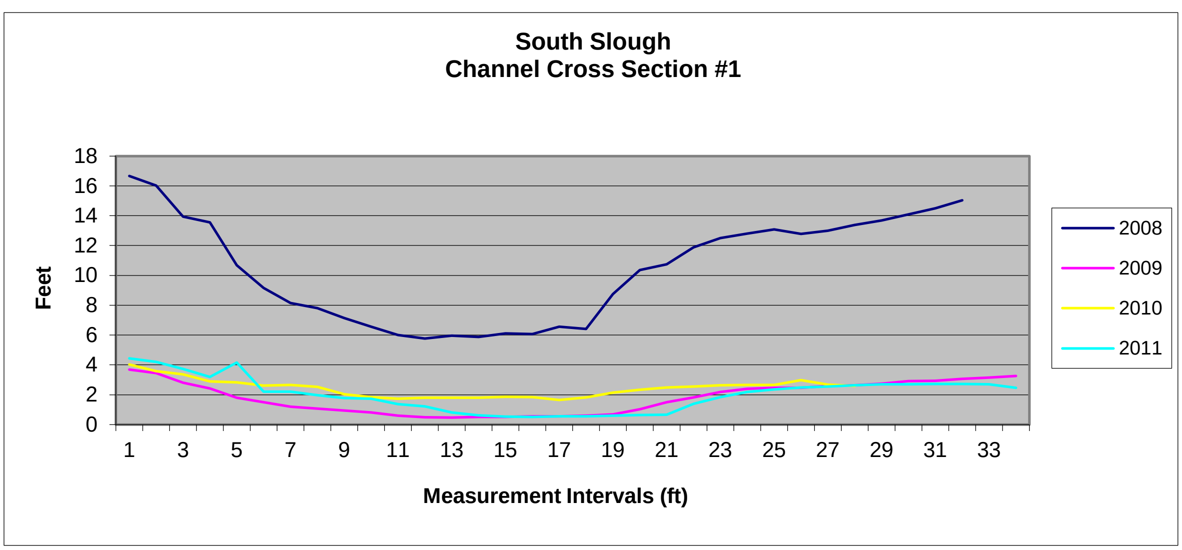


Figure 82. Downstream most channel cross section, #1, at South Slough, 2007-2011.

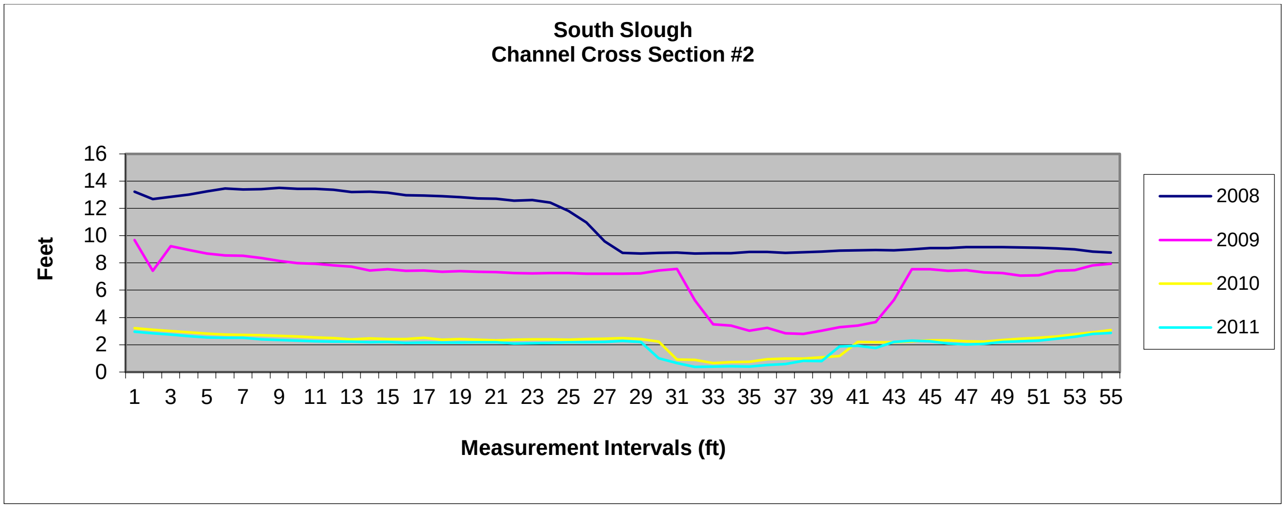


Figure 83. Channel cross section #2 at South Slough, 2007-2011.

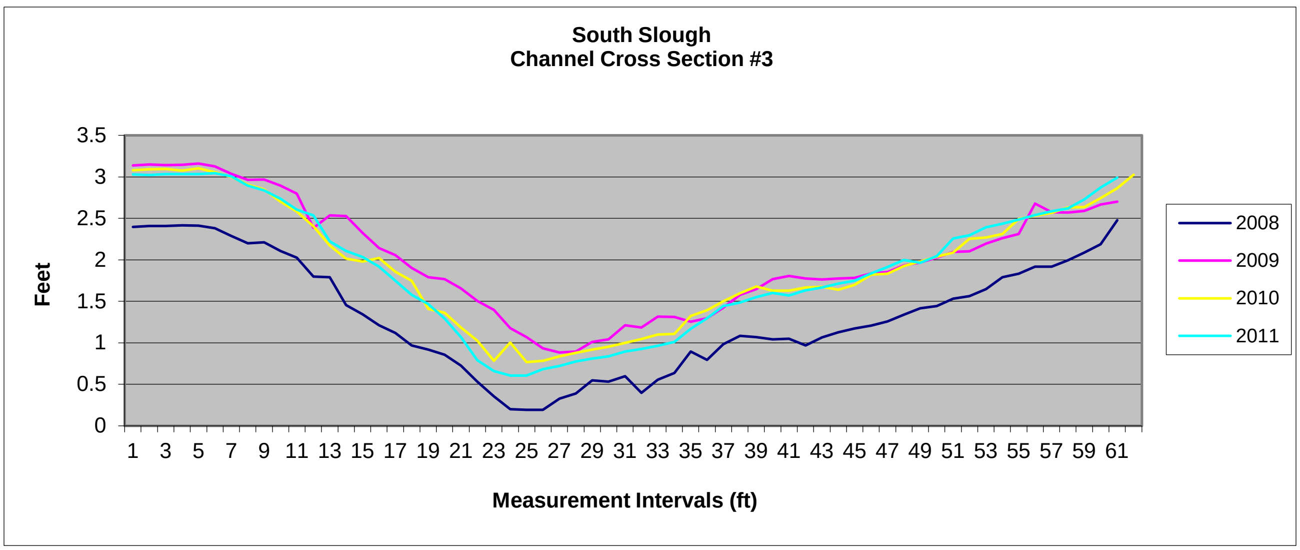


Figure 84. Channel cross section #3 at South Slough, 2008-2011.

Channel cross sections #3, #4 and #5 (Figure 85 and Figure 86) are taken upstream in the South Slough near the fish sampling site. These show changes in the channel bottom aggrading over the past five years, as opposed to the previous cross section (Figure 82 and Figure 83), which underwent significant erosion in the channel.

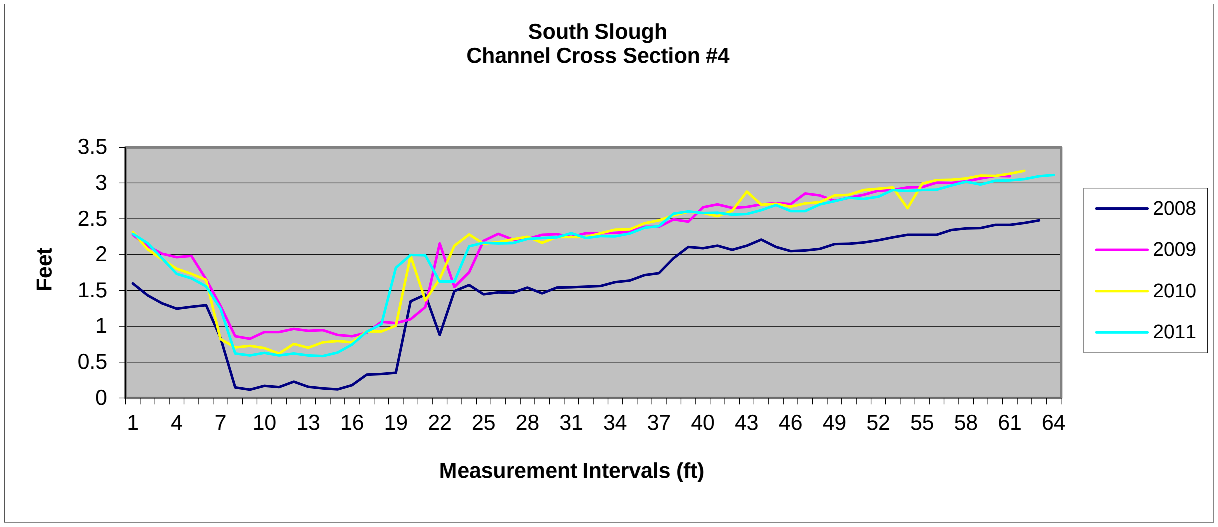


Figure 85. Channel cross section #4 at South Slough, 2007-2011.

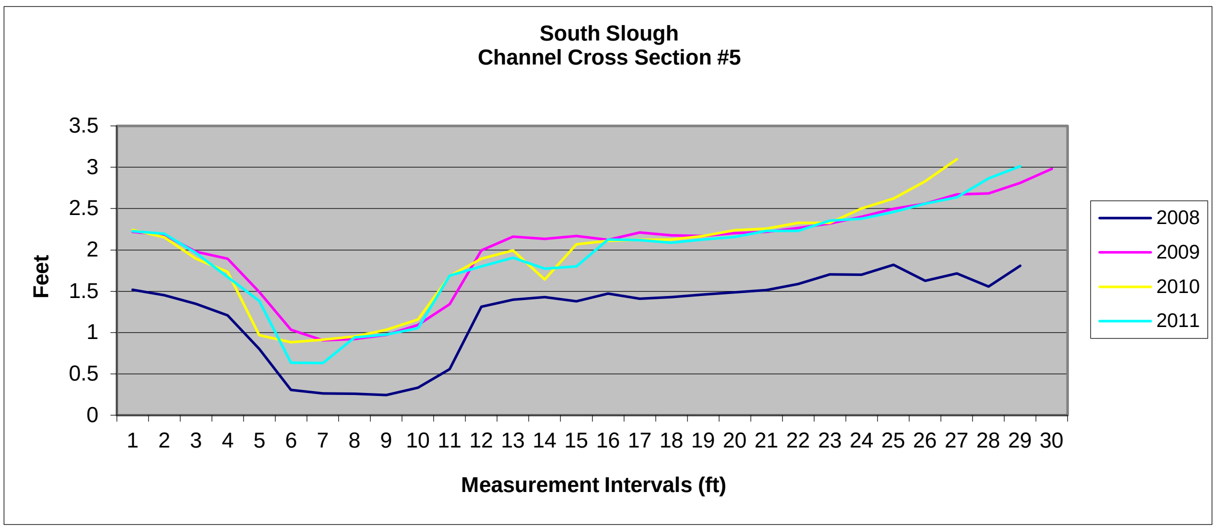


Figure 86. Upstream most channel cross section, #5, at South Slough, 2007-2011.

Channel cross sections at South Slough demonstrated surprising results. The degree of expected change in channel erosion and aggradation was small; however the actual amount of sediment movement was substantial. This signifies the continuous dynamic environment of tidal channels in the lower estuary. Unlike South Slough, Alder Creek has demonstrated some slight channel migration as well as consistent erosion along all five cross sections (Figure 87 -Figure 91). The level of change is in smaller increments, Sough Slough eroded down 6 ft in some areas, while Alder Creek eroded only 1 ft.

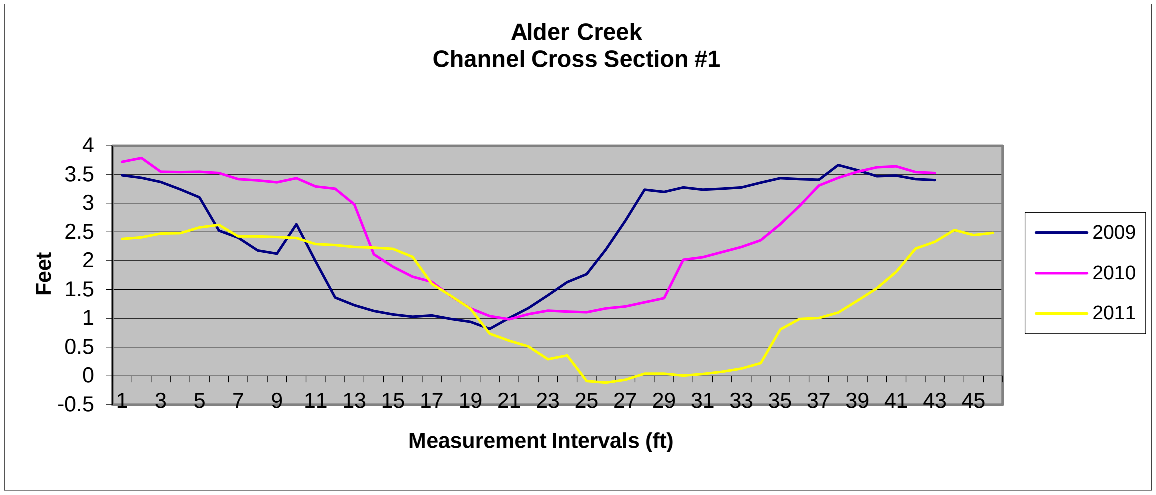


Figure 87. Downstream most channel cross section, #1, at Alder Creek, 2009-2011.

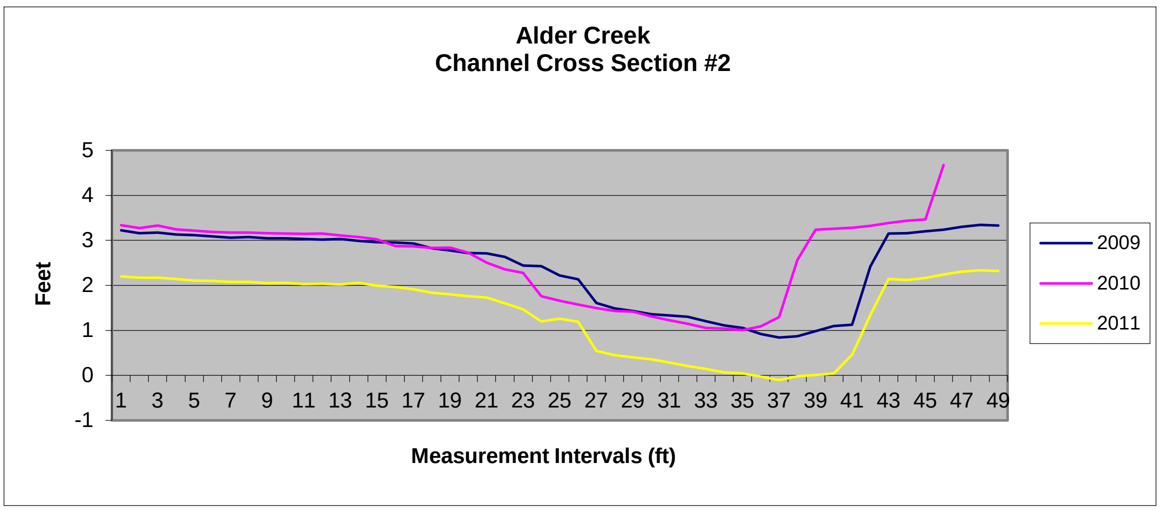


Figure 88. Channel cross section #2 at Alder Creek, 2009-2011.

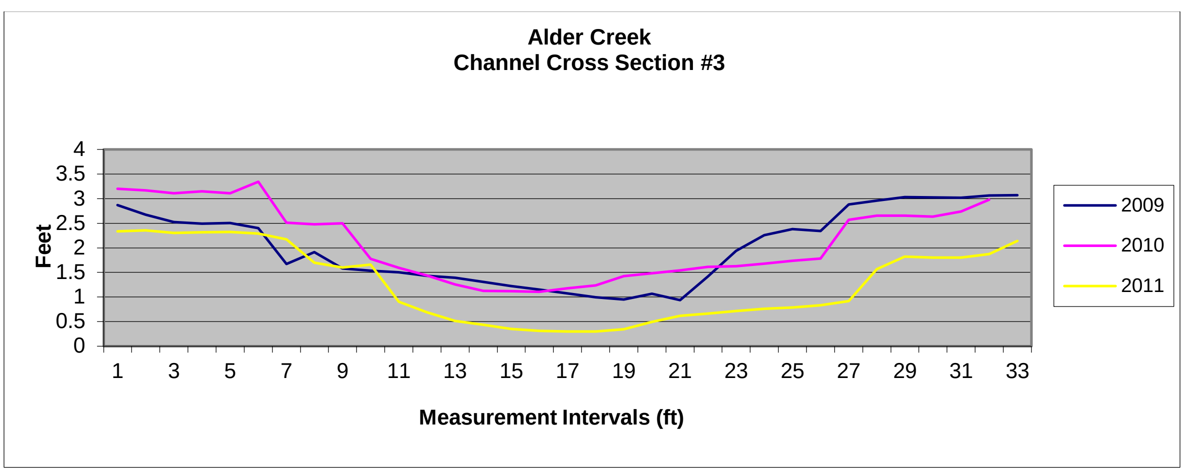


Figure 89. Channel cross section #3 at Alder Creek, 2009-2011.

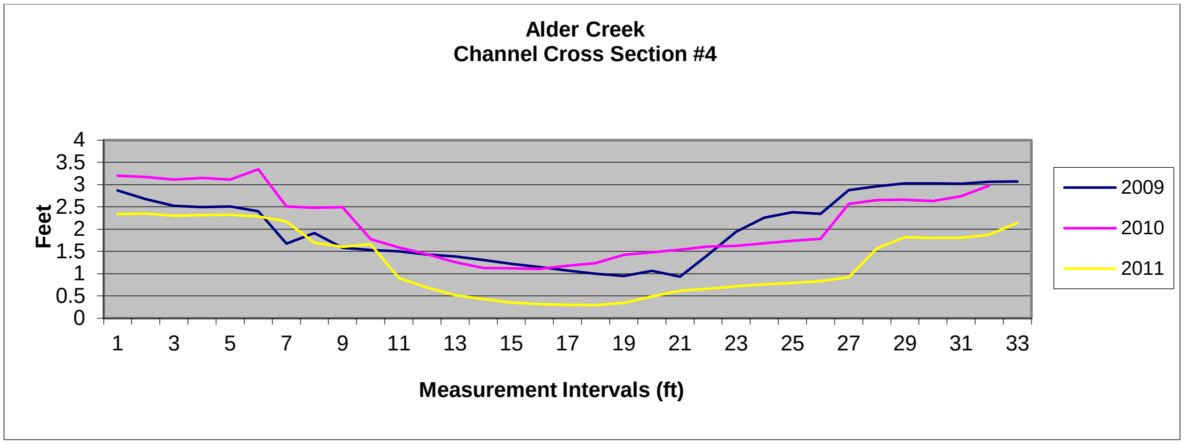


Figure 90. Channel cross section #4 at Alder Creek, 2009-2011.

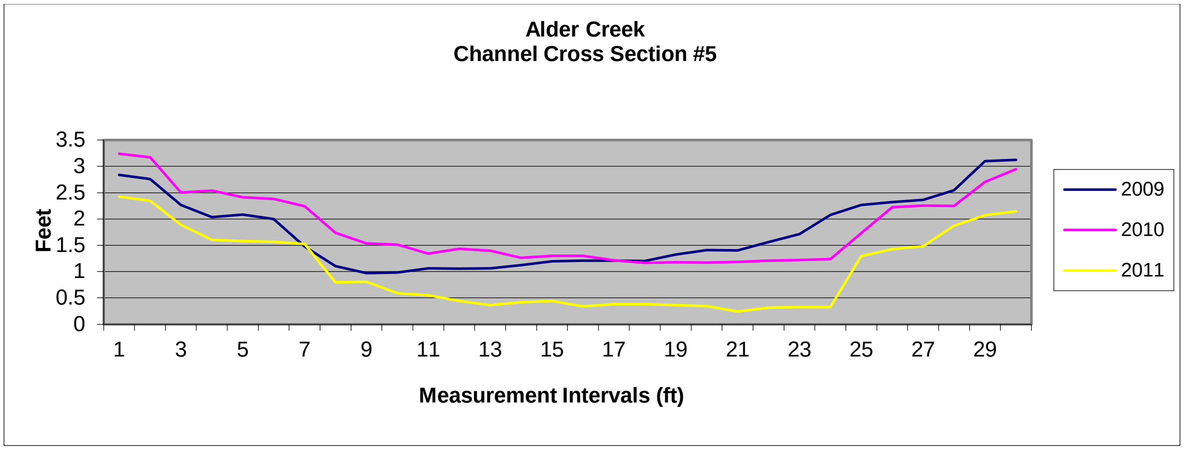


Figure 91. Upstream most channel cross section, #5, at Alder Creek, 2009-2011.

5.3.7.5 Water Quality 2012

5.3.7.5.1 Temperature

Juvenile salmonids have a water temperature range of around 0° Celsius to 24° Celsius (Johnson et. al. 2011). For salmonids and other fish species, no single environmental factor affects their development and growth rate more than water temperature (Bjornn 1991). Water temperature influences the metabolism, behavior, and mortality of fish and other organisms in their environment (Mihursky and Kennedy 1967). While many fish species may be capable of surviving at temperatures near the extremes of this suitable range, growth can be highly impacted. Many salmonids alter their behavior when encountering increases or decreases in temperature (Bjornn 1991). Low temperatures in the winter can result in casualties of stream type Chinook and coho, and high temperatures in the summer can limit the distribution and growth capacity of juvenile salmonids.

Prior to restoration activities South Slough had limited tidal connectivity and influence. In theory, this restricted the degree of water and nutrient exchange with the mainstem Lewis & Clark River and decreased the amount of sediment transport. However, without pre-restoration data, the magnitude of the effect on nutrient exchange and sediment transport related to limited tidal connectivity is unknown in South Slough. It is important to take the differences in site conditions into consideration when comparing South Slough and Alder Creek. South Slough is a much deeper channel than Alder Creek, which may explain the differences in water temperature between the two sites during similar times of the year. It is also important to note that water quality data was not collected at Alder Creek in 2007, while it can be assumed that annual variation in temperature at South Slough would remain relatively consistent from one year to the next we have no data to confirm this assumption.

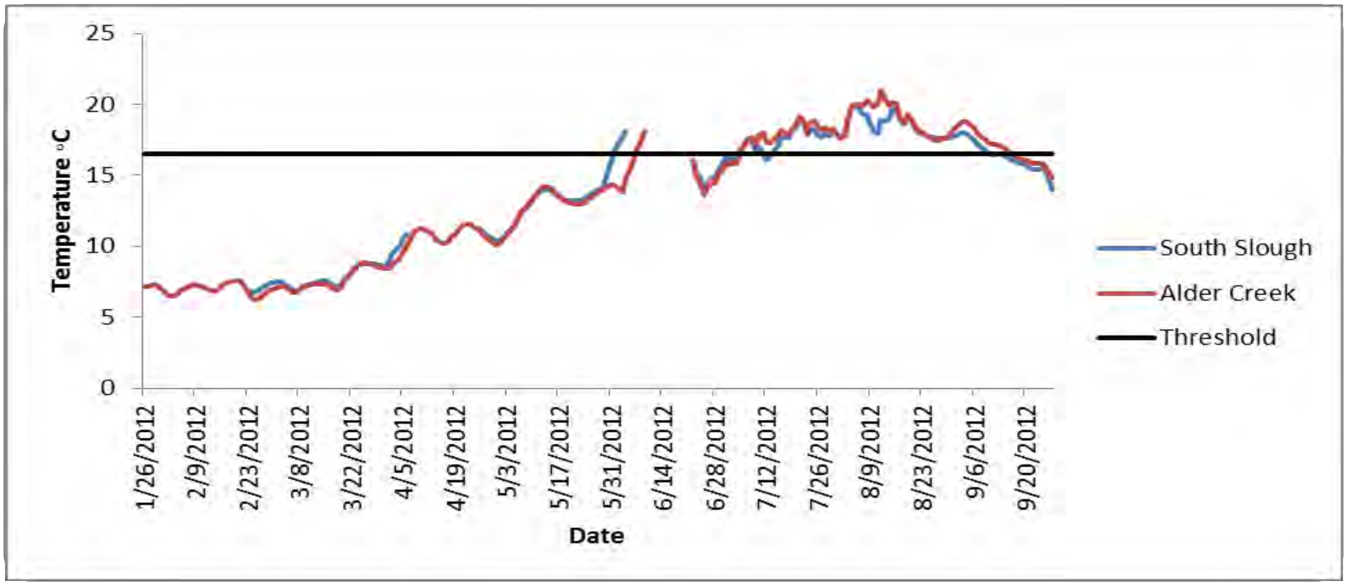


Figure 92. 7 day moving average of temperature (˚C) at South Slough and Alder Creek, 2012.

Table 44. Temperature maximum, minimum and averages (SD) for South Slough and Alder Creek, 2012. *Parentheses indicate standard deviation value.*

	South Slough		
Month	**Maximum Temp**	**Minimum Temp**	**Mean Temp (SD)**
February	8.53	4.09	6.81 (±0.95)
March	10.27	3.72	7.64 (±1.16)
April	13.53	7.13	10.02 (±1.68)
May	17.45	8.42	12.65 (±2.12)
June	18.04	11.46	14.48 (±1.53)
July	21.17	13.27	17.39 (±1.69)
August	22.31	13.93	18.64 (±1.69)
September	20.76	7.59	16.72 (±1.75)
	Alder Creek		
	Maximum Temp	**Minimum Temp**	**Mean Temp (SD)**
January	8.73	4.41	6.99 (±1.23)
February	9.56	4.27	6.98 (±0.95)
March	12.72	3.47	7.47 (±1.23)
April	15.4	6.31	10.36 (±1.68)
May	21.64	8.4	12.59 (±2.28)
June	21.26	10.52	14.31 (±1.68)
July	25.35	13.23	17.69 (±1.87)
August	24.78	14.23	19.09 (±1.87)
September	22.67	7.81	17.25 (±1.71)

Studies conducted by the Washington Department of Ecology suggest that the best estimate of threshold for a healthy summer rearing temperature ranged between 14.78-18.08°C with a mean value of 16.5°C. Utilizing nine lines of evidence (Lab growth studies at constant temperature, ranges identified in literature as optimal for growth, comparison test regimes with better growth, lab growth studies in fluctuating temperature regimes, field studies on growth, predation and competition, temperature preferences in lab, swimming performance and scope of activity, field distributions – healthy) the best estimate of threshold was determined to be 16.5°C, considered to be fully protective of summer juvenile rearing (Hicks 2000).

Temperature data for South Slough and Alder Creek reveal the 7-DMA to approach and surpass Washington Department of Ecology's best estimate of juvenile rearing threshold (16.5°C) between July and September 2012. Data exceeding this threshold does not come within the range considered as lethal (25°C) to salmonids, but does imply that both sites exceed the temperatures preferred by rearing juveniles in the warm summer months. It should be noted however that the maximum temperatures revealed in Table 44 are commonly seen at low tide when there is little water in the channel. During those times (specifically at Alder Creek) it is reasonable to assume that fish are either able to survive limited exposure to higher temperature or migrate in to the mainstem Lewis & Clark during low tide. Fish sampling in previous years lends evidence to this as coho remained in South Slough during low tide events throughout the sampling season while no salmonids were observed in Alder Creek during low tides.

5.3.7.5.2 Dissolved Oxygen

Dissolved oxygen (DO) is a measurement of the amount of oxygen dissolved in water. DO is essential to fish and other aquatic life forms, and as such is a critical component to the characterization of the health of an aquatic system. The optimal DO level for salmonids is 9mg/L. A level of 7-8 mg/L is generally considered acceptable, while 3.5-6 mg/L is considered poor (Bjornn 1991). Growth rates and food conversion efficiency of juvenile salmonids may be limited by concentrations less than 5 mg/L. Levels below 3.5 mg/L are potentially fatal to salmon. A level below 3 mg/L is stressful to most vertebrates and other forms of aquatic life (Bjornn 1991).

Dissolved oxygen was not captured at South Slough throughout the season due to repeated failure of the dissolved oxygen sensor, despite multiple attempts to calibrate and replace the sensor. Dissolved oxygen was captured at Alder Creek throughout the monitoring season. Dissolved oxygen levels at Alder Creek remained within acceptable levels throughout the majority of the monitoring season, only demonstrating brief instances in which levels dipped below the 5 mg/L threshold (Figure 20). Dissolved oxygen at Alder Creek averaged 8.42 mg/L (with a standard deviation of ±1.51) over the course of the monitoring season, an acceptable level for salmonids (Table 45).

Based on previous years' data during which South Slough consistently maintained higher DO concentrations than Alder Creek, we can surmise that DO levels demonstrated similar patterns in 2012. DO levels remained inside the acceptable levels for salmonids the majority of the time at Alder Creek. With greater depths and lower temperatures, South Slough most likely maintained acceptable levels of DO as well.

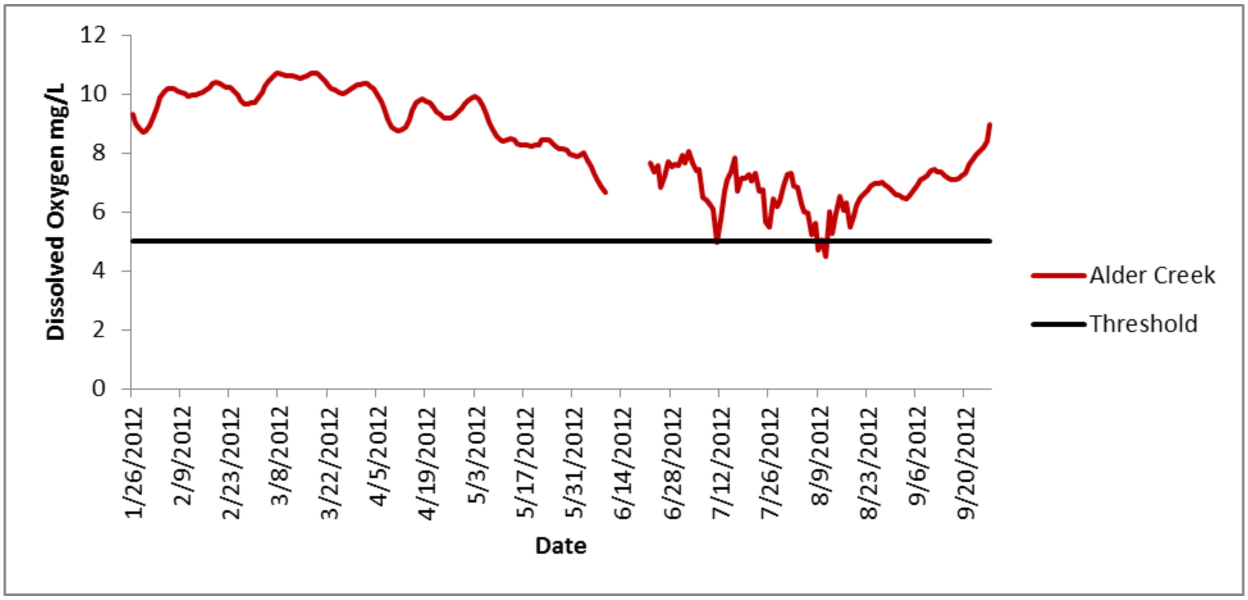


Figure 93. 7 day moving average for dissolved oxygen at Alder Creek, 2012.

Table 45. Dissolved Oxygen (DO) Maximum, minimum and mean (SD) recorded at Alder Creek, 2012. Parentheses indicate standard deviation values.

	Alder Creek		
Month	**Maximum DO levels**	**Minimum DO levels**	**Mean DO (SD) levels**
January	11.48	6.38	9.49 (±1.22)
February	12.17	6.19	9.91 (±1.23)
March	12.5	7.11	10.30 (±1.01)
April	11.82	5.27	9.55 (±1.14)
May	11.3	3.03	8.76 (±1.55)
June	12.11	4.08	7.73 (±1.45)
July	14.22	0.70	6.81 (±2.17)
August	13.34	-0.08	6.14 (±2.34)
September	13.03	1.94	7.08 (±2.08)
2012	14.22	-0.08	8.42 (±1.51)

5.3.7.6 Water Quality 2007-2011

Water quality parameters including temperature, dissolved oxygen (DO), depth (pressure), and conductivity have been measured pre and post restoration at South Slough to evaluate the changes resulting from reconnection of tidal influence and the ensuing benefits to salmon. Stream temperature, DO and habitat opportunity are limiting factors for salmon. Juvenile salmon have a water temperature range of around 0° Celsius to 24° Celsius (Johnson et. al. 2011). Low temperatures in the winter can result in casualties of stream type Chinook and coho, and high temperatures in the summer can limit the distribution and growth capacity of juvenile salmonids.

Prior to restoration activities South Slough experienced a muted tidal regime. The culvert at the mouth of the slough restricted water flow, reducing the tidal highs and lows. In theory, this restricted the degree of water and nutrient exchange with the mainstem Lewis & Clark River, and decreased the amount of sediment transportation (no data available to support this). Figure 94 demonstrates the restricted tidal activity pre-restoration, and the restored tidal regime post-restoration, with the caveat that the tidal cycles will not be identical as tidal fluctuations vary annually depending on the lunar cycle and alignment of the sun, moon and Earth. The tidal cycle becomes more pronounced post-restoration, demonstrating the restored tidal regime to the South Slough. Visual observations during field monitoring events attest to the restored tidal connection as well, with a delay in outgoing tides pre-restoration, and sinuous tidal ebb and flow post-restoration.

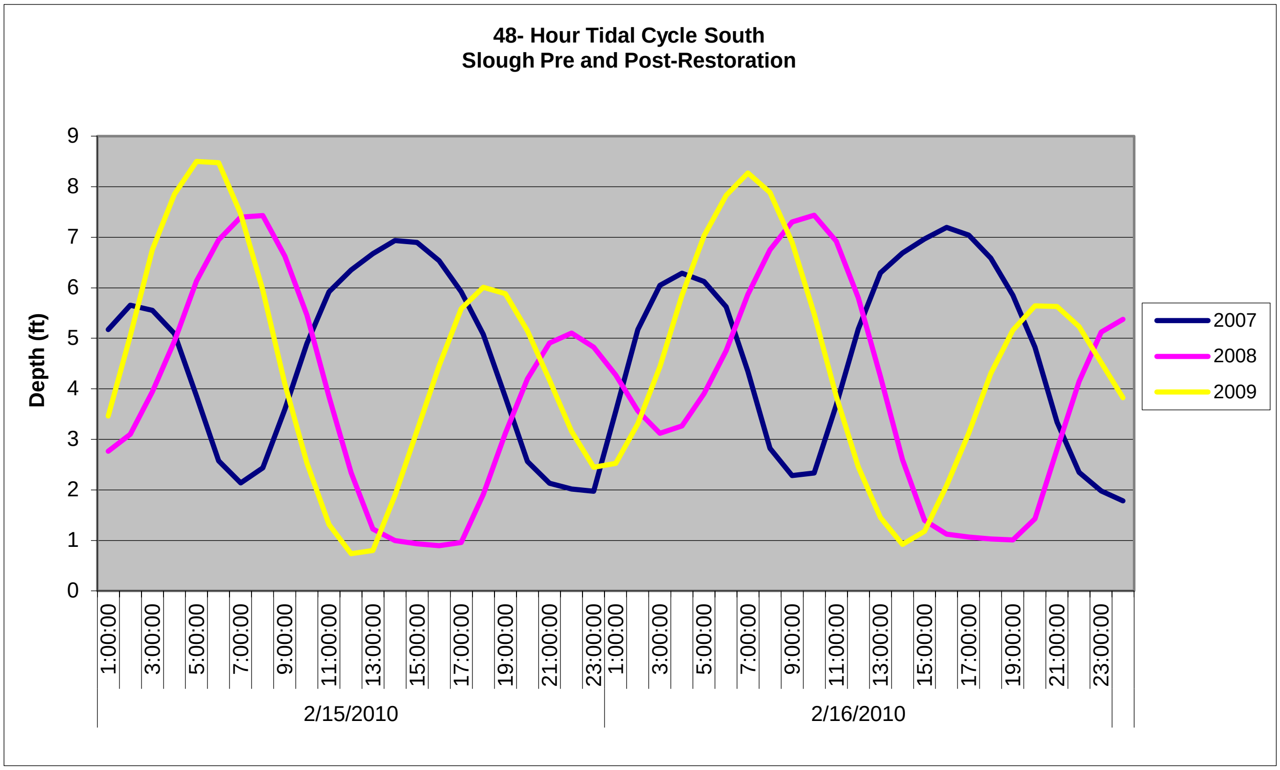


Figure 94. 48-Hour tidal cycle at South Slough between February 15th and 16th, 2007 – 2009. *Data from 2010 & 2011 were not available.*

Pre-restoration water quality data was not collected from Alder Creek, so no comparisons can be made. In order to assess the effects of restoration activities pre and post reconnection data from South Slough must be examined. Comparing post-restoration data from South Slough to Alder Creek as a reference for environmental conditions also assists in concluding whether or not changes can be attributed to restoration actions. Examining conductivity during tidal cycles helps illustrate the changes that have resulted at South Slough from restoration.   Figure 96 demonstrates the lack of natural tidal fluctuations pre-restoration, while 2010 data clearly shows the impact of a fully connected tidal influence on conductivity.

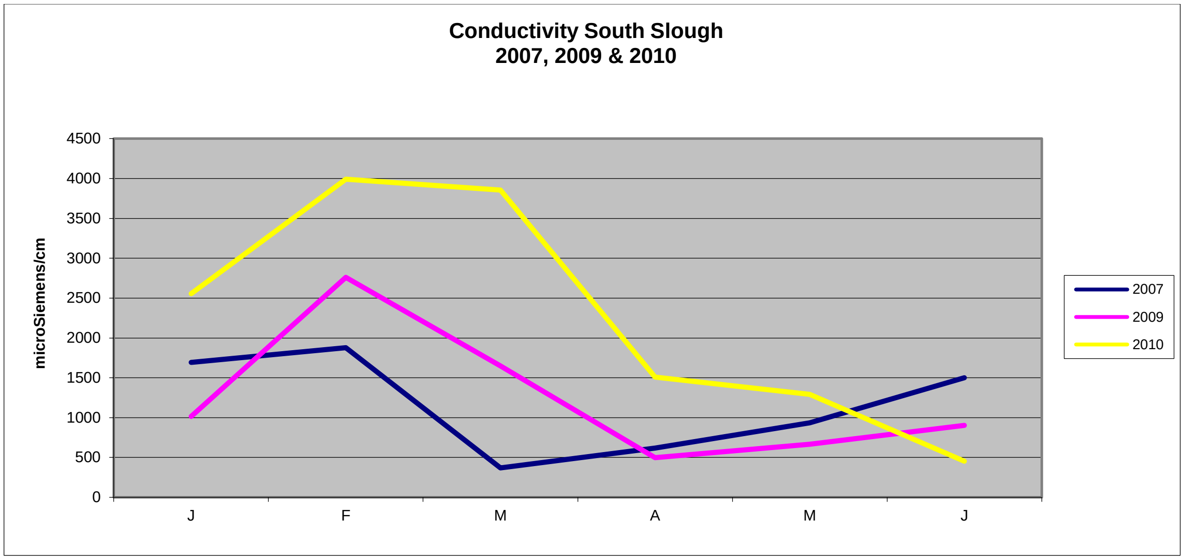


Figure 95. Conductivity at South Slough before and after restoration, 2007, 2009 and 2010.

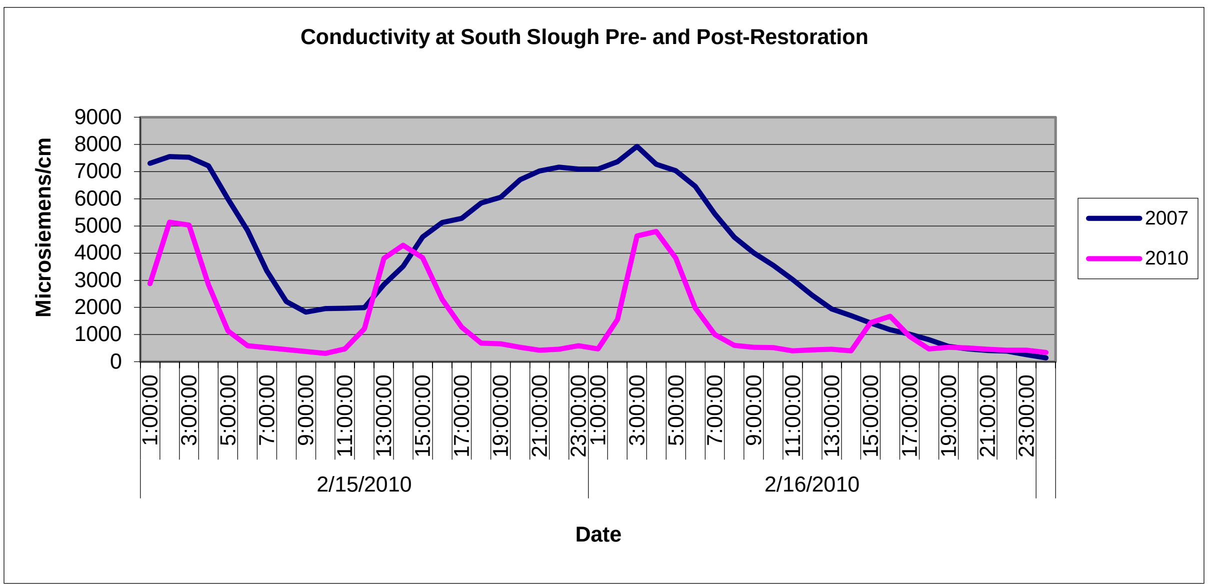


Figure 96. Conductivity at South Slough over a 48-hour period pre and post restoration, 2007 and 2010.

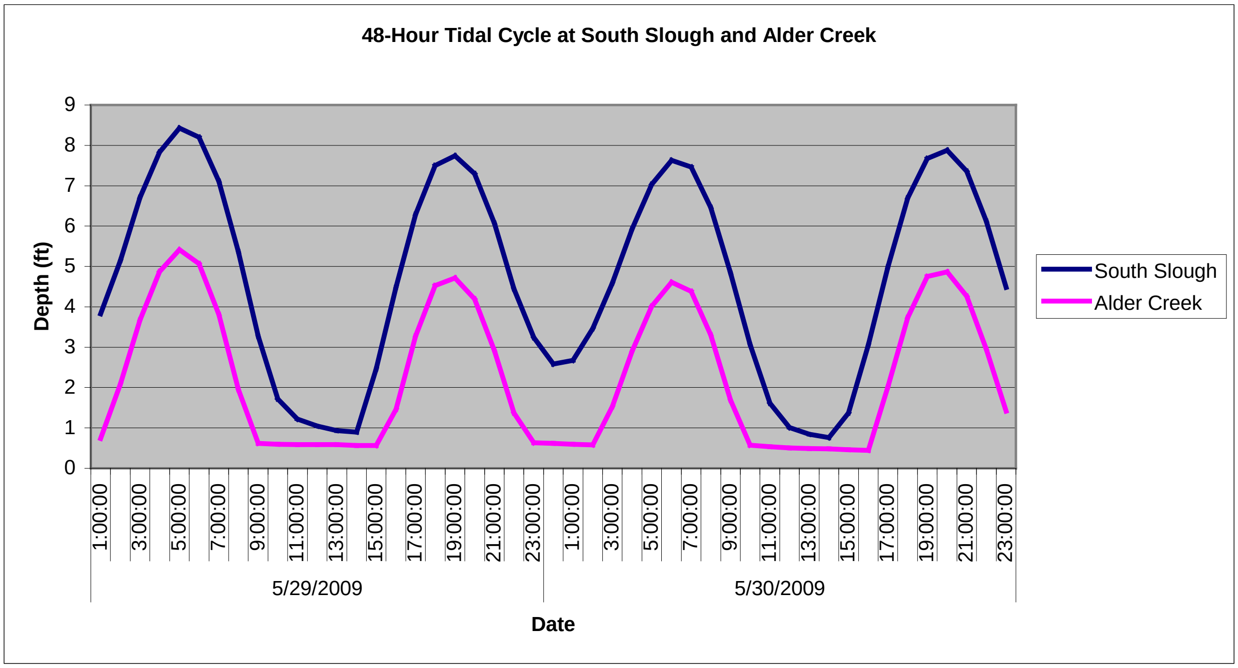


Figure 97. 48-Hour tidal cycle at South Slough and Alder Creek during May 29th – 30th, 2009.

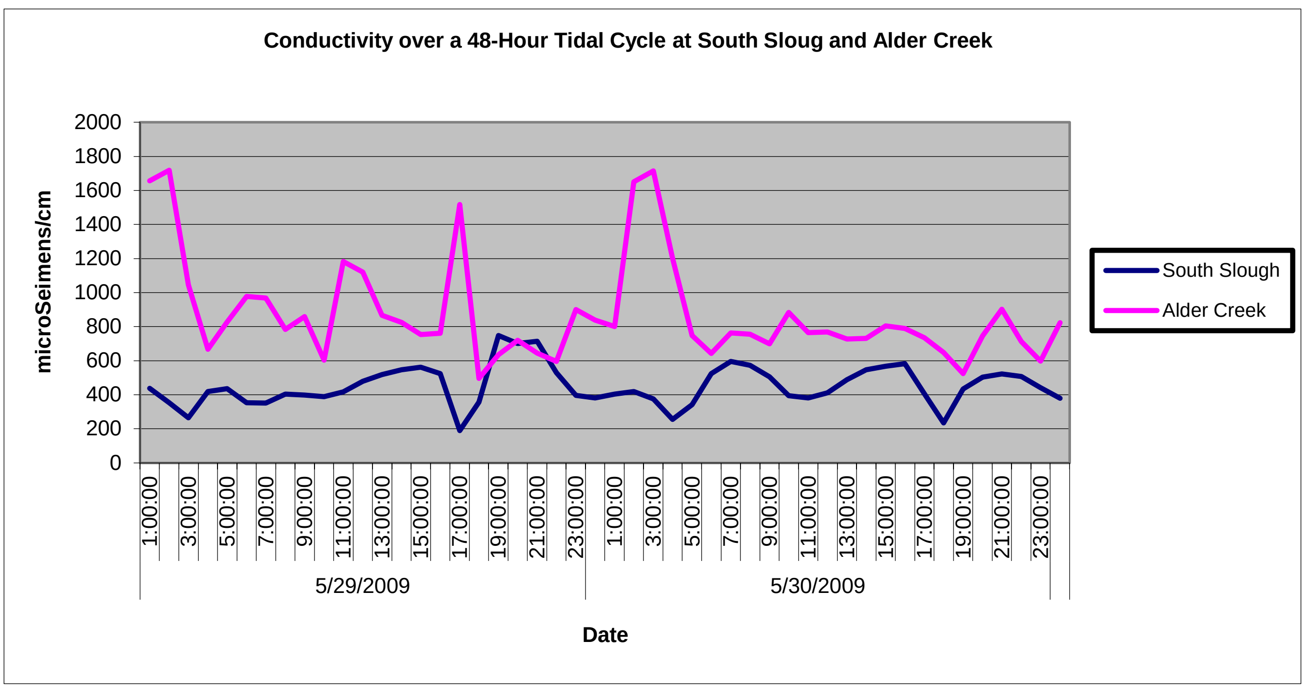


Figure 98. Actual Conductivity over a 48-hour tidal cycle at South Slough and Alder Creek, 2009.

South Slough has greater changes in water depth (Figure 97), yet smaller fluctuations in conductivity (Figure 98). This is most likely the result of differences in channel depth and freshwater input between the two sites. Conductivity data alone may not be as reliable an indicator as depth in evaluating the effect of treatment actions, as discharges to the stream can raise the conductivity by raising nitrate, chloride and phosphate levels. The wetlands directly adjacent to South Slough was previously used for livestock grazing, and when combined with restricted water exchange from the mainstem Lewis & Clark the nitrate level could have been substantial.

When comparing South Slough to Alder Creek the differences in site conditions must be taken into account. South Slough is a much deeper channel than Alder Creek, which may explain the differences in water temperature between the two sites during similar times of the year (Figure 99 and Figure 101). It is also important to note that water quality data was not collected at Alder Creek in 2007, while it can be assumed that annual variation in temperature at South Slough would remain relatively consistent from one year to the next we have no data to confirm this assumption. There are two gaps in South Slough water quality data. The data loss was attributed to stolen equipment in 2009, and due to equipment malfunctions in 2010 that required the probe to be sent back to the manufacturer.

Post restoration temperature maximums were consistently lower than 2007 temperatures at South Slough. Alder Creek maintained higher temperature maximums throughout the year, a reflection of the site conditions, the shallow channel and lack of woody vegetation along the riparian zone. Annual temperature changes impact many biological processes for juvenile and adult salmonids; including but not limited to feeding potential, growth rates, spawning, smoltification, hatching, out migration timing and success.

Temporal trends in 7-Day moving average (7-DMA) temperature time series during the period of time in which salmonids are in high abundance demonstrated similar temporal trends within and across all years post-restoration at South Slough. Each year, the 7-DMA temperatures approached or exceeded the generally acceptable tolerance range for salmonids in the late summer months at both sites. The data indicates that temperatures at South Slough post-restoration are more similar to temperatures of Alder Creek and at times even cooler due in part to differences in channel morphology and channel reconnection (Figure 101). Maximum temperatures at South Slough post-restoration remained 1 to 3 °C lower than temperatures of the adjacent Alder Creek, particularly in the later months of June through August. As mentioned previously, the channel at South Slough retains greater water depth than the channel at Alder Creek. South Slough also has a consistent freshwater input from a well shaded upstream reach, while Alder Creek is fed by water that travels in very shallow channel across a large wetland. The differences in upstream freshwater input may contribute to differences in water temperature in the summer months when air temperatures and solar radiation are greater.

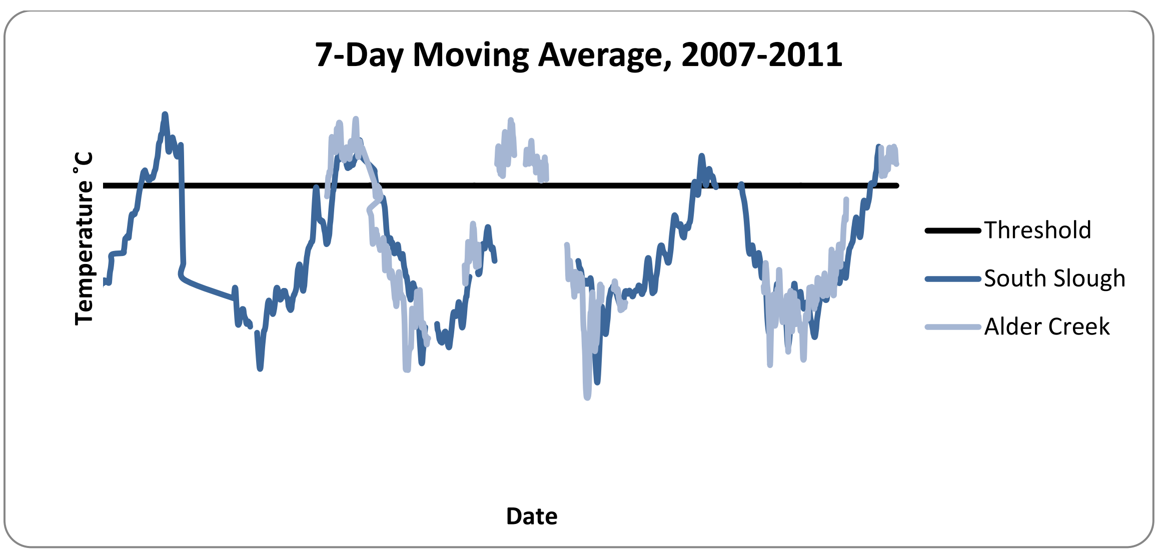


Figure 99. 7-day moving average for temperature at South Slough and Alder Creek, 2007-2011.

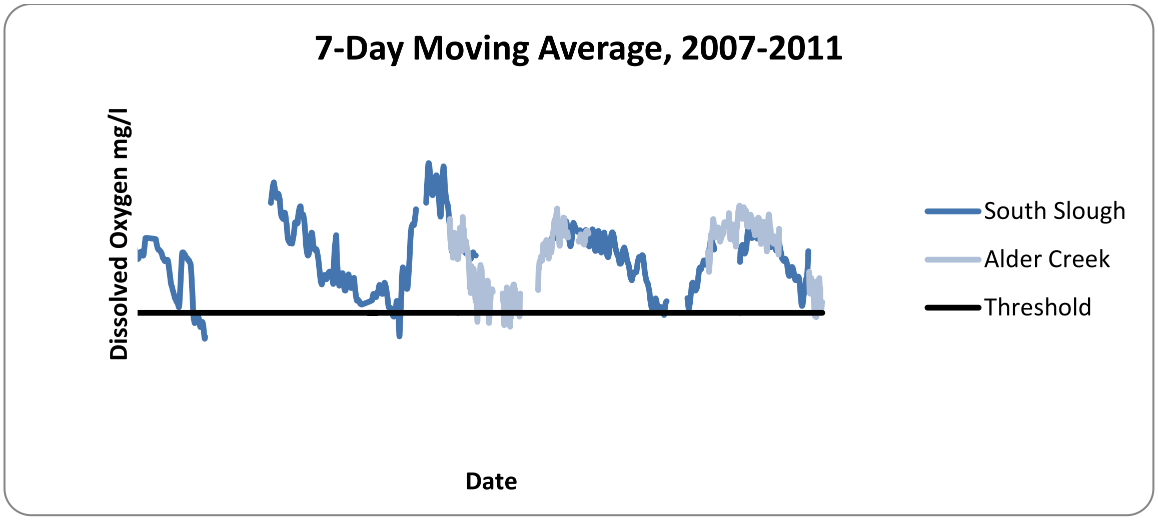


Figure 100. 7 day moving average for dissolved oxygen (DO) at South Slough and Alder Creek, 2007-2011.

The optimal DO level for salmonids is 8mg/l. A level of 7-8 mg/l is generally considered acceptable, while 3.5-6 mg/l is considered poor. Levels below 3.5 mg/l are likely fatal to salmon. A level below 3 mg/l is stressful to most vertebrates and other forms of aquatic life (Bjornn 1991).

Dissolved oxygen levels at South Slough remain within acceptable levels averaging 8.23 mg/l. Due to equipment failure dissolved oxygen levels at Alder Creek are not as well defined. During the time period in which data was gathered at Alder Creek the average dissolved oxygen has been recorded at 8.86 mg/l. A comparison of South Slough DO levels to that of Alder Creek reveals trends in 7-Day moving average (7-DMA) DO to be rather analogous (Figure 100). The lack of continuous DO data set pre and post restoration represses our ability to analyze the result of restoration on the sloughs DO levels. However, we can assume that the decline in water temperature resulting from re-connection would translate to higher levels of DO.

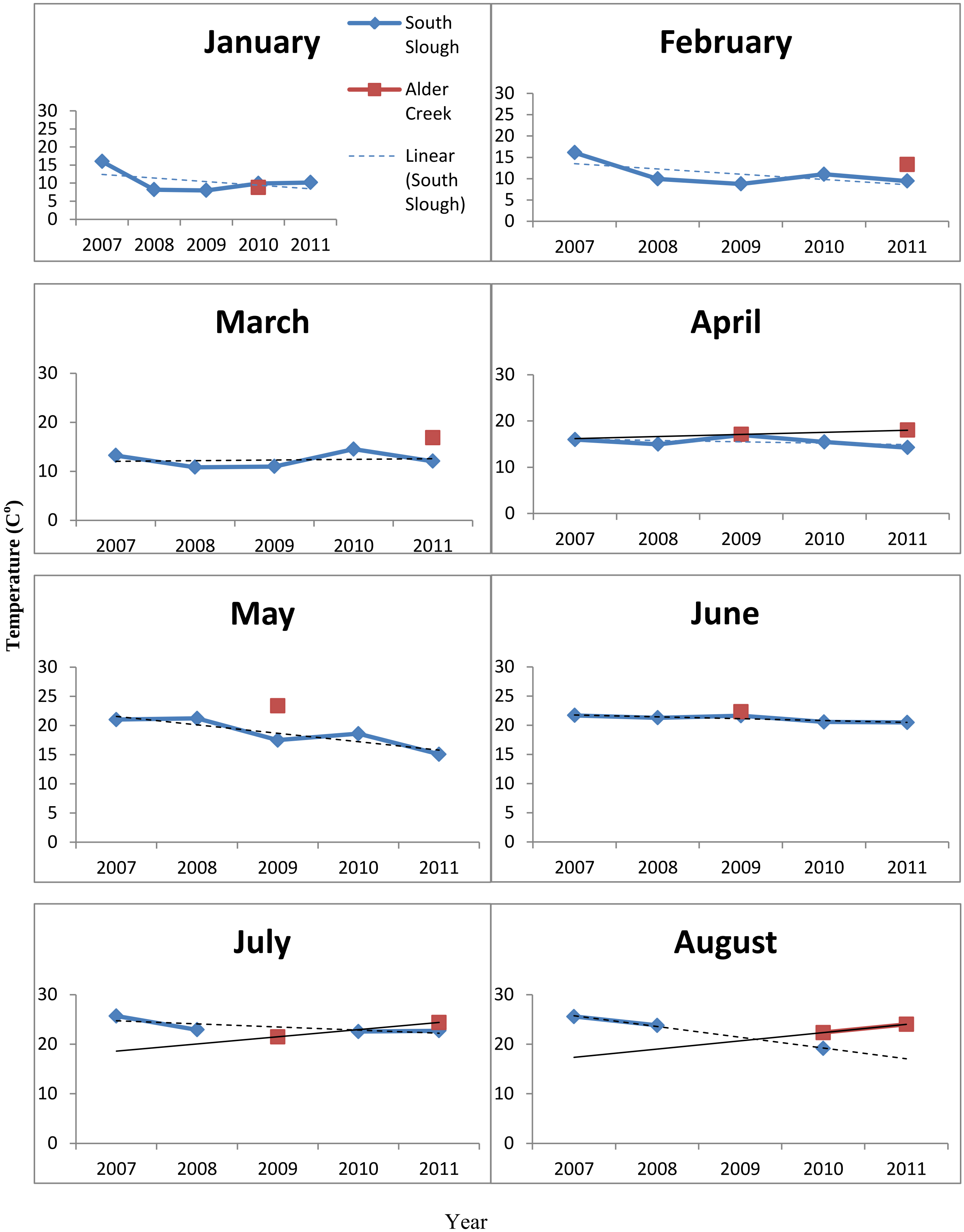


Figure 101. Temperature maximums for South Slough and Alder Creek, 2007 – 2011.

Table 46. Temperature maximums and average (SD) for South Slough and Alder Creek, 2007-2011. **No data collected in 2007.*

	In stream temperature ranges in South Slough and Alder Creek										
		Max Temp					**Mean (SD) Temp**				
		2007	**2008**	**2009**	**2010**	**2011**	**2007**	**2008**	**2009**	**2010**	**2011**
South Slough	Jan	16.0	8.19	8.0	9.92	10.17	4.36 (±0.77)	4.32 (±1.74)	5.17 (±0.91)	8.14 (±0.83)	8.53 (±0.67)
	Feb	16.11	9.95	8.76	11.02	9.43	7.03 (±2.83)	6.98 (±1.22)	12.77 (±1.93)	8.36 (±0.87)	6.53 (±1.43)
	Mar	13.22	10.83	10.98	14.52	12.04	8.89 (±1.74)	7.98 (±1.05)	11.35 (±2.26)	9.33 (±1.48)	8.09 (±1.20)
	Apr	15.96	14.97	16.95	15.46	14.23	11.98 (±1.54)	9.51 (±1.67)	11.17 (±1.81)	10.47 (±1.83)	9.54 (±1.29)
	May	21.0	21.22	17.48	18.59	15.06	14.75 (±2.03)	13.69 (±2.33)	12.5 (±1.62)	12.8 (±2.01)	12.67 (±1.35)
	Jun	21.68	21.28	21.63	20.56	20.48	17.39 (±0.77)	14.67 (±2.63)		14.29 (±2.27)	15.73 (±1.81)
	Jul	25.68	22.91		22.54	22.71	20.34 (±0.77)	18.46 (±1.64)		17.25 (±2.03)	18.65 (±2.21)
	Aug	25.57	23.8		19.13		19.29 (±0.77)	18.76 (±1.43)			
Alder Creek	Jan	*			8.81		*		6.37 (±1.24)	8.07 (±0.82)	6.52 (±1.62)
	Feb					13.29		8.88 (±0.56)		7.50 (±0.53)	7.38 (±1.46)
	Mar					16.83		8.28 (±0.69)			9.16 (±1.32)
	Apr			17.11		18.01			10.98 (±2.02)		11.54 (±2.17)
	May			23.36					11.79 (±1.74)		
	Jun			22.33				17.94 (±1.49)	17.33 (±1.62)		
	Jul			21.47		24.36		19.57 (±0.96)	19.03 (±2.03)		18.21 (±1.74)
	Aug			22.34		24.01		19.59 (±0.92)	18.86 (±1.55)		18.80 (±1.53)

5.4 Conclusions

5.4.1 Sampling Frequency & Methods

Fish community sampling methods were modified every year from 2007 - 2011 at South Slough and Alder Creek. This modification provided a more complete sample of fish density and community composition, but precludes statistical analysis across years. The benefit to the modification led to a more complete sampling of the sites and resulted in the discovery that coho fry were able to remain in the channel above the trap net during the ebb tide.

5.4.2 Species Composition

South Slough expressed a greater diversity of species than Alder Creek. Fish community trend data revealed a decline in the percentages of non-native and native, non-salmonid, fish species at South Slough and a subsequent increase in unmarked juvenile salmonids. Trend data between South Slough and Alder Creek demonstrated a similar pattern over the years sampled. From this similarity it can be concluded that the restoration site is responding in a similar pattern to the reference site in regards to fluctuations in species composition. Restoration actions cannot be determined to reduce the number of non-native fish species as a result of the similar downward trend in non-native species at both sites. Species composition demonstrated results similar to other studies conducted in the LCRE. Previous studies in the LCRE identified three-spine stickleback as the most abundant species encountered at sampling sites in the LCRE between 2007 and 2009, consistent with data collected from South Slough and Alder Creek between these same years (Johnson et. al. 2011).

5.4.3 Peak Abundance & Size Class

Peak abundance of salmonids showed more variability between the two sites. Coho peak abundance timing was consistent pre and post restoration, and Chinook were not observed at South Slough in 2007, so it cannot be concluded whether restoration activities influenced peak abundance timing for either species. Prior studies demonstrate the peak abundance for juvenile salmonids ranges between March and July for fry, and both ocean and stream type fingerlings (Bottom et. al. 2005). This is consistent with timing and abundance data at both South Slough post restoration, and at Alder Creek during all years sampled. In 2007, prior to tidal reconnection peak abundance occurred earlier in the year at South Slough, which was potentially due to higher temperatures pre-restoration (post-restoration water temperatures are lower than pre-restoration temperatures). The tidal reconnection of South Slough has resulted in lower temperatures, allowing juvenile salmon to successfully utilize the slough for longer periods during the year. This effectively increases the habitat opportunity of South Slough for ESA listed salmon species.

5.4.4 Catch per Unit Effort

CPUE was calculated as the number of fish divided by the amount of time (hours) fished for trap netting events. CPUE for South Slough can be compared between 2007 and 2010, between 2007, 2009 and 2010 at Alder Creek, and during 2007, 2009, 2010, and 2011 between South Slough and Alder Creek. CPUE was highest for chum in March and April; this is consistent with timing of chum at other CREST monitoring sites in the Gray's River, Lewis & Clark River and the Young's River. Chinook CPUE at Alder Creek was highest during March (2010) and June (2009) . Chinook CPUE at South Slough was highest in June (2010). Similar to Chinook, CPUE for coho was highest during different months at South Slough and Alder Creek. At South Slough CPUE for coho was highest during June (2010), and during March (2010) at Alder Creek.

5.4.5 Size Class and Temporal Distribution

Chinook and coho were observed in the greatest numbers during similar months at both sites, meaning large numbers of both species are utilizing and/or are dependant on the tidal shallow water habitat at the same times of the year. Between January and April larger sized Chinook and coho were present (104-114mm). Based on their size and timing these are most likely stream type yearlings utilizing the shallow water habitat on their way to the estuary to begin smoltification. The smaller sized Chinook, chum, and coho utilizing the habitat later in the sampling season are likely a variety of ocean and stream type life history strategies, utilizing the habitat as they both migrate to the estuary at small sizes or search for forage and refuge during their prolonged stay in freshwater. This is consistent with the overlapping life stages migrating through the LCRE at different times in their life cycle (Bottom et. al. 2005).

5.4.6 Species Diversity

While the number of species was greater at South Slough, Alder Creek exhibited a more uniform number of individuals per species across the total catch giving it a higher diversity index value (Shannon-Wiener). Species composition demonstrated similar trends at both South Slough and Alder Creek. The percentages of native, non-salmonid, and non-native fish species experienced similar peaks and declines in abundance across the sampling years, ultimately being at their lowest in 2011. Unmarked juvenile salmon increased in percentage in 2011. The trend data is consistently similar between South Slough and Alder Creek, both before and after restoration was completed. As a result of this similarity, the changes in percentages of salmonids, native non-salmonid, and non-native fish cannot be attributed to treatment effects at South Slough. However, the overall numbers of fish caught across the entire fish community is higher post-restoration at South Slough. This means that restoration did not increase the percentage of salmon relative to the entire catch, but instead increased the abundance of all fish species at South Slough including juvenile salmonids. As the vast majority of fish at South Slough are native species, the treatment effects have increased the habitat amount and opportunity for not only ESA listed salmonids in the Columbia River Estuary but for all native fish species.

5.4.7 Marked and Unmarked Salmonids

Marked (presumably hatchery), juvenile salmon represented a very small percentage of the total catch, and were not observed during every sampling year. There is limited artificial propagation of salmon in the Young's Bay Watershed, where South Slough and Alder Creek reside, although net pens are located in the Young's River and Young's Bay. Because of its location at the mouth of the Columbia River, marked juvenile salmon are not passing directly through Young's Bay during seaward migration, which is likely the reason very few marked fish are observed at South Slough and Alder Creek. With very few marked fish utilizing the Young's Bay Watershed it could be a valuable area for native salmon restoration as competition for resources with marked fish is limited. Prey Availability & Selectivity

5.4.8 Prey Availability and Selectivity

Chinook and coho at both South Slough and Alder Creek are selecting the same several taxa of invertebrates: chironomids, corophium, and isopods. Chinook and coho are also most abundant during the same months (April – June) at South Slough and Alder Creek. This may present competition for resources including forage and refuge. This data is consistent with previous data, finding that Chinook in particular demonstrate a high selectivity for corophium sp., from shallow water habitats and invertebrates from vegetated habitats during their months of peak abundance (Bottom et. al. 2005). These findings further state that juvenile salmon in brackish environments feed extensively on emergent insects, with selectivity towards all life stages of chironomids (consistent with Bottom et. al. 2005). South Slough and Alder Creek provide both of these habitat types, consisting of shallow water sloughs with vegetated riparian areas and aquatic vegetation in the channels. The coincidental timing and prey selectivity may also be the result of abundance in prey resources at both sites. Terrestrial prey most selected by Chinook and coho (chironomids) were present during every year sampled at both sites. Future sampling will include nueston nets in order to provide data on the presence, timing, and abundance of aquatic macroinvertebrates; of particular interest are isopods and corophium as they were two of the three most selected prey species.

5.4.9 Sediment Accretion and Channel Cross Sections

Sediment accretion stakes at South Slough demonstrated similar trends over time to Alder Creek, with overall net increases in sediment deposits. Channel cross sections from South Slough demonstrate erosion in the lower channel and slight sediment deposition in the upper channel. The most dramatic changes occurred in the first year of full tidal reconnection, demonstrating how quickly a restoration site can change after activities are complete. In contrast, Alder Creek demonstrated changes in channel morphology on a much smaller scale. The most significant and constant changes at Alder Creek were channel migration and erosion. It is expected that over time South Slough and Alder Creek will respond similarly to annual changes in sediment transport, however during the years sampled cross section data demonstrated very little similarity between the two sites. South Slough experienced minimal changes in channel morphology between 2010 and 2011, yet in that same year Alder Creek experienced the most significant changes seen during all the years monitored. Overall the trend data exhibits no clear similar trend in erosion or sediment deposition between South Slough and Alder Creek. It is expected that as the channel at South Slough reverts towards its historic condition that the two sites will demonstrate similar patterns in channel migration and sediment movement. It is also likely that differences in site conditions influence the channel morphology to an extent that their channel morphology changes will be consistently dissimilar. The freshwater input to South Slough may cause increased sediment deposition/erosion during high precipitation events, while the input to Alder Creek is limited and constricted by a culvert. Long term data will inform on the successional restoration of South Slough, and ultimately determine how similar the sites are in regards to sediment movement and changes in channel morphology.

5.4.10 Water Quality

Water quality sampling was not conducted in 2007 at Alder Creek so a pre-restoration baseline comparison is not possible. In order to distinguish between treatment effects and environmental fluctuations pre- and post-restoration data from South Slough were compared, and post-restoration data from South Slough and Alder Creek were compared. It is evident that water temperatures responded to the re-establishment of natural tidal hydrodynamics. The contrast in temperature data from 2007 to 2008 at South Slough revealed the benefits of a more complete tidal hydrological connection in terms of restoring natural water temperatures. The temperatures inside South Slough were much warmer than those in Alder Creek previous to restoration, which indicated restricted connection with water from the Lewis and Clark River. Post-restoration temperatures were consistently lower at Sough Slough indicating tidal reconnection improved water quality. This is further reinforced by temperature trends from Alder Creek, where average monthly temperatures either remained consistent or increased. Conductivity data exhibited similar changes post-restoration. There was not only an increase in conductivity after restoration actions were complete, but the restored tidal patterns showed increased conductivity during the incoming tide and decreased conductivity during the ebb flow (when freshwater is the primary source). Water quality data from Alder Creek is limited, and as such strong inferences from the data are not possible. It is surmised that since temperature, dissolved oxygen and tidal signature represent positive indicators of habitat improvement as a result of treatment effects that conductivity levels are as well.

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7 Appendices

Appendix A: Sandy River Delta Action Effectiveness Photo Monitoring Points

Sundial Island North
Photo point 1, Transect A

2008

2012

Sundial Island North
Photo point 4, Transect 3.


2008


2012

Sundial Island North
Photo point 6, Transect 5

2008

2012

Sundial Island North
Photo point 6, Transect 5

2008

2012

Southwest Quad
Photo comparisons


2008


2012

Southwest Quad
Photo comparisons

2008

2012

South Bank/North Slough
Photo point 2, Transect 2

2008

2012

South Bank/North Slough
Photo point 5, Transect 5

2008

2012

North Bank Sandy Channel
Photo point 6, Plot 47

2008

2012

North Bank Sandy Channel
Photo point 6, Plot 47

2008

2012

Mirror Lake
Photo point 5, Transect 5

2008

2012

Mirror Lake
Photo point 7, Transect 7

2008

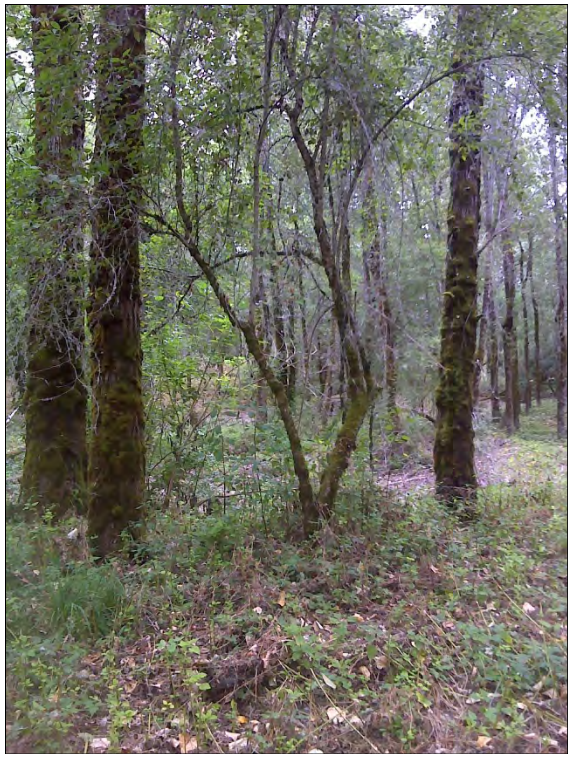
2012

Mirror Lake
Photo point 9, Transect 9

2008

2012

Columbia River Bank
Transect 5, Plot 3



2011

2012

Columbia River Bank
Plot 9

2011

2012

Columbia River Bank: Pre- and Post-Restoration Comparison Photos

2008: Pre-restoration

2008: After site preparation


2011: Post-restoration


2012: Post-restoration