Development of New Information to Inform Fish Passage Decisions at the Yale and Merwin Hydro Projects on the Lewis River

Annual Progress Report

August 2015

Robert Al-Chokhachy[1], Mark Sorel[2], Dave Beauchamp[3], and Christopher Clark[4]

[1]U.S. Geological Survey, Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, MT 59715, ral-chokhachy@usgs.gov

[2]Univeristy of Washington, School of Aquatic and Fishery Sciences, Box 355020, Seattle, WA 98195-5020

[3]U.S. Geological Survey, Washington Cooperative Fish and Wildlife Research Unit, School of Aquatic and Fisheries Sciences, University of Washington, Box 355020, Seattle, WA 98195-5020

[4]Montana State University, Department of Ecology, Bozeman, MT, 59715

Table of Contents   **Page Number**

Introduction

The reintroduction of extirpated salmonids to historically-occupied areas is becoming increasingly common as a conservation and recovery strategy (Marcot et al. 2012; Temple and Pearsons 2012; Anderson et al. 2014). Often reintroductions are implemented after the factors which originally led to species extirpation have been reduced, eliminated, or mitigated. For species of Pacific Salmon (*Oncorhynchus* spp.) and steelhead (i.e., anadromous; *O. mykiss*), addressing barriers to migration, which have been a primary factor in the declines and extirpation of many populations (Sheer and Steel 2006; McClure et al. 2008), has been an integral component of recovery efforts. Mitigation has included barrier removals (e.g., Weigel et al. 2013), developing fish passage opportunities (e.g., Kiffney et al. 2009), and/or actively trapping and hauling juvenile and adult anadromous salmonids around barriers (e.g., Serl and Morrill 2010).

With any reintroduction there are a number of concerns regarding the ecological impact of the reintroduction efforts (see Anderson et al. 2014 for review). Anderson et al. (2014) identify three main tenets to consider when assessing reintroductions: 1) potential benefits if reintroduction is successful; 2) biological risk through interactions of reintroduced strains with existing populations; and 3) factors potentially limiting a successful reintroduction. Here we focus on the 2nd and 3rd factors outlined by Anderson et al. (2014) for the Upper North Fork Lewis River in Washington. The Upper North Fork Lewis River historically contained wild populations of Coho Salmon, Chinook salmon, and Steelhead. These populations were extirpated with the completion of the Merwin (1932), Yale (1953), and Swift (1958) hydropower facilities, which were built without full passage. However, recent licenses issued by the Federal Energy Regulatory Commission (FERC) require the installation and operation of upstream fish passage facility at Merwin dam and a downstream fish passage facility at Swift dam. The licenses were developed in consultation with the National Marine Fisheries Service (NMFS) and the United States Fish and Wildlife Service (USFWS 2006; NMFS 2007). The overarching goal of this fish re-introduction is to establish viable, self-sustaining, naturally-reproducing, harvestable populations of spring Chinook, winter steelhead, and Coho Salmon at levels higher than minimum viable populations.

Study area

The study area for this project includes portions of the Upper North Fork Lewis River in Washington including areas upstream of Merwin Dam (Figure 1). Coastal Cutthroat Trout (*O. clarkii clarkii*) are distributed throughout much of study area as well as unique populations of Rainbow Trout (*O. mykiss*) and Bull Trout (*Salvelinus confluentus*). Each of the three reservoirs is unique in their assemblage of aquatic species and physical characteristics. Lake Merwin is characterized by a high abundance of Northern

Pikeminnow (*Ptychocheilus oregonensis*), and is stocked with Kokanee (*O. nerka*), Rainbow Trout, and Tiger Muskie (*Esox masquinongy* x *E. Lucius*) for sport fisheries. Yale Lake supports a self-sustaining population of Kokanee, as well as Northern Pikeminnow. Both Yale Lake and Swift Reservoir host populations of Bull Trout which, like Chinook, Coho and Steelhead, are listed under the Endangered Species Act (ESA). While no persistent Bull Trout population currently exists in Lake Merwin, individual Bull Trout often occur within the reservoir through fallout from Yale Lake. Swift Reservoir is supplemented with Chinook Salmon (*O. tshawytscha*), Coho Salmon (*O. kisutch*), Steelhead Trout, and is annually stocked with 20,000 pounds of Rainbow Trout for sport harvest. Yale Lake also contains small populations of Chinook, Coho, and Steelhead from downstream movements over Swift dam. All three reservoirs have populations of naturally producing Rainbow and Cutthroat Trout, Mountain Whitefish (*Prosopium williamsonii*) as well as Largescale Suckers (*Catostomus macrocheilus*) (Table 1).

Project objectives

The reintroduction within the North Fork Lewis River has already commenced (2012) for the portions of the basin upstream of Swift Dam. Here, the specific objectives of this project were to collect new information to inform passage decisions specifically at the Yale and Merwin Hydro projects. In particular, this project focuses specifically on: 1) collecting data to ground-truth the amount and quality of habitat for anadromous species spawning and rearing in tributaries to Yale Lake and Lake Merwin; 2) assessing adult potential for spawning success in tributaries to Yale Lake and Lake Merwin; 3) evaluating juvenile production potential and emigration success; 4) evaluating likely predator impacts in Lake Merwin; and 5) assessing anadromous/resident interactions in Yale Lake and Lake Merwin and the tributaries to these reservoirs. In 2013, we initiated field research to help inform passage decisions related to these specific objectives and the preliminary results of this project are presented below. This document is organized based on specific tasks (2 through 6).

Acknowledgements

References

Anderson, J. H., G. R. Pess, R. W. Carmichael, M. J. Ford, T. D. Cooney, C. M. Baldwin, and M. M. McClure. 2014. Planning Pacific Salmon and Steelhead Reintroductions Aimed at Long-Term Viability and Recovery. North American Journal of Fisheries Management 34(1):72-93.

Kiffney, P. M., G. R. Pess, J. H. Anderson, P. Faulds, K. Burton, and S. C. Riley. 2009. Changes in fish communities following recolonization of the Cedar River, WA, USA by Pacific salmon after 103 years of local extirpation. River Research and Applications 25(4):438-452.

Marcot, B. G., C. S. Allen, S. Morey, D. Shively, and R. White. 2012. An Expert Panel Approach to Assessing Potential Effects of Bull Trout Reintroduction on Federally Listed Salmonids in the Clackamas River, Oregon. North American Journal of Fisheries Management 32(3):450-465.

McClure, M. M., S. M. Carlson, T. J. Beechie, G. R. Pess, J. C. Jorgensen, S. M. Sogard, S. E. Sultan, D. M. Holzer, J. Travis, B. L. Sanderson, M. E. Power, and R. W. Carmichael. 2008. Evolutionary consequences of habitat loss for Pacific anadromous salmonids. Evolutionary Applications 1(2):300-318.

NMFS. 2007. Biological opinion and Magnuson-Stevens Fishery Conservation and Management Act Consultion; operation of PacifiCorp and Cowlitz PUD's Lewis River Hydroelectric Project (FERC Nos. 935, 2071, 2111, and 2213). National Oceanic and Atmospheric Administration: National Marine Fisheries Service, 2005/05891, Seattle, Washington.

Serl, J., and C. Morrill. 2010. Summary report for the 1996 to 2009 seasonal operation of the Cowlitz Falls fish facility and related Cowlitz Falls anadromous reintroduction program activities. Washington Department of Fish and Wildlife.

Sheer, M. B., and E. A. Steel. 2006. Lost watersheds: Barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and Lower Columbia River basins. Transactions of the American Fisheries Society 135(6):1654-1669.

Temple, G. M., and T. N. Pearsons. 2012. Risk management of non-target fish taxa in the Yakima River Watershed associated with hatchery salmon supplementation. Environmental Biology of Fishes 94(1):67-86.

USFWS. 2006. Biological Opinion for the Federal Energy Regulatory Commission relicensing of the Lewis River Hydroelectric Projects: Merwin (No. 935), Yale (No. 2071), Swift No. 1 (No. 2111), and Swift No. 2 (No. 2213). U.S. Fish and Wildlife Service, FWS Reference Number: 1-3-06-F-0177, Lacey, Washington.

Weigel, D. E., P. J. Connolly, K. D. Martens, and M. S. Powell. 2013. Colonization of Steelhead in a Natal Stream after Barrier Removal. Transactions of the American Fisheries Society 142(4):920-930.

Table 1. Relative abundance of fish species and predominant summer crustacean zooplankton found in each reservoir.

	Swift	Yale	Merwin
Northern Pikeminnow	na	High	High
Largescale Suckers	High	High	High
Kokanee	na	High	High
Rainbow Trout	Moderate	Rare	Moderate
Cutthroat Trout	Rare	Rare	Rare
Bull Trout	Rare	Rare	Rare
Hybrid Trout	Rare	Unknown	Rare
Tiger Muskie	na	na	Moderate
Sculpin	Moderate	Moderate	Moderate
Threespine Stickleback	Rare	Rare	Rare
Mountain Whitefish	Rare	Rare	Rare
Bluegill Sunfish	na	na	Rare
Largemouth Bass	na	na	Rare
Juvenile Coho Salmon	Rare	na	na
Juvenile Chinook Salmon	Rare	Rare	Rare
Predominant summer zooplankton	Daphnia	Leptodora	Daphnia

Figure 1. A general map of the study area for stream and reservoir investigations; the study area included portions of the North Fork Lewis River upstream of Merwin Dam.
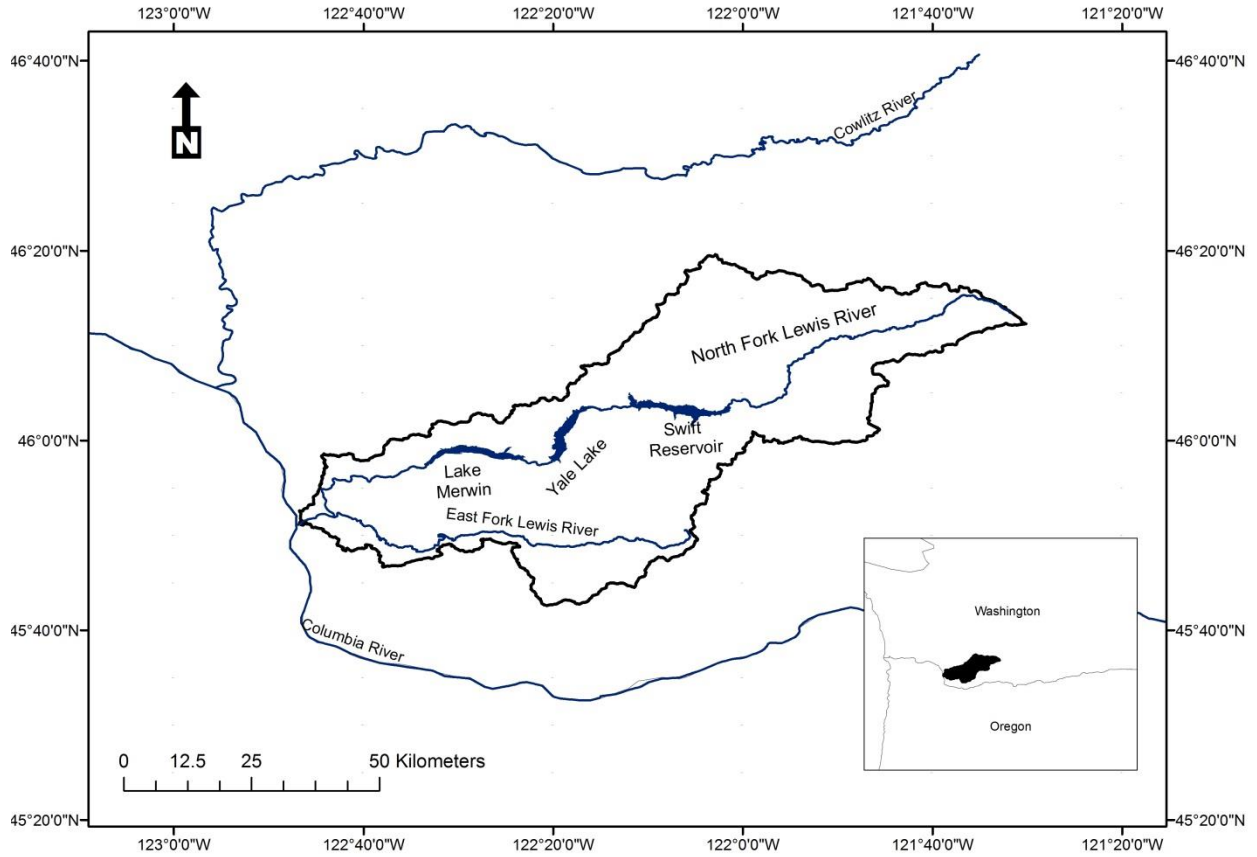
Task 2: Assessment of habitat in Lake Merwin and Yale lake

During 2013 we quantified the extent of available habitat for potential reintroductions of anadromous species in tributaries to Lake Merwin and Yale Lake. We inventoried all streams with at least 25 m of available habitat including Cape Horn Creek, Jim Creek, Indian George Creek, Buncombe Hollow Creek, Brooks Creek, Lower Speelyai, Siouxon Creek, Upper Speelyai Creek, West Tributary Speelyai, West fork Speelyai, Dog Creek, Ole Creek, Rain Creek, Cougar Creek, Panamaker Creek, and the Swift Bypass Channel (Table 1). In 2014 we evaluated the quality of habitat for *Oncorhynchus* spp. reintroductions in each of these study streams. Information regarding the extent and quality of habitat was subsequently integrated with the Ecosystem Diagnosis and Treatment method (EDT; Lestelle et al. 1996) for species-specific estimates of production potential in the Lake Merwin and Yale Lake basins.

Methods

Habitat Quality

During late July and early August of 2014 we conducted continuous habitat surveys from each tributary mouth to the upstream migration barrier. We used a census survey design to collect continuous habitat data for a comprehensive assessment of habitat and to minimize bias in assessments due to site-to-site variability (Urquhart et al. 1998; Larsen et al. 2004; Anlauf et al. 2011). To ensure consistency with regional approaches, we conducted our surveys using protocols derived from the Columbia Habitat Monitoring Program (CHaMP 2013) that are currently employed in anadromous salmon and steelhead research and monitoring in the Columbia River Basin (N. Bouwes, Unpublished data). In order to allow for rapid continuous surveys our approach was a modification from the original CHaMP 2013 method (hereafter CHaMP Lite). We geo-referenced each habitat unit and the data associated with that unit while conducting our surveys.

Reach Length

We established reach lengths in each stream using an average measure of bankfull width. We determined the minimum reach length for each stream by taking the average bankfull width of the first 10 habitat units (Table 1), and multiplying it by twenty (Kershner et al. 2004; CHaMP 2013). The first habitat unit boundary encountered after this minimum length served as the start of the next reach length.

Habitat Unit Classification

We identified habitat units using a two tiered hierarchical system (Hawkins et al. 1993). Tier I units were distinguished by gradient, relative stream velocity and turbulence and

included three classes: fast water turbulent, fast water non-turbulent, and slow water pool. Tier I fast water turbulent units were further subdivided into Tier II classes. These subclasses were differentiated by hydraulic properties and included riffles, rapids, cascades, and falls. Tier I slow water pools were further subdivided into Tier II classes based on the primary process that formed the unit and included scour pool, plunge pool, dam pool, beaver pool, and off channel pool. Tier I fast water non-turbulent units were referred to as glides.

During each sampling event, we evaluated stream physical habitat attributes that are important for *Oncorhynchus* spp. We geo-referenced each habitat unit and estimated (1) unit dimensions consisting of wetted and bankfull width and length; (2) the total number of pieces of large woody debris (LWD) exceeding 10 cm in diameter and 1m in length; (3) percent substrate composition of each size class to the nearest 5 percent; (4) percent fine sediment in pool tails; and (5) riparian condition according to Blair et al. (2008).

Habitat Unit Dimensions

We measured wetted and bankfull widths at two transects located approximately at 1/3rd and 2/3rd total unit length, and unit length from bottom to top (Hankin and Reeves 1988) using a laser range finder. We recorded three depth measurements spaced equidistantly across each transect for a total of 6 depth measurements, which we averaged for each unit.

We used the individual measurements from transects for an estimate of average wetted and bankfull width and depth for each habitat unit. Additionally, we calculated the surface area of each habitat unit through estimates of average width and unit length. Computing estimates of surface area enabled us to quantify the total surface area for each stream and the proportional surface area for any type of habitat unit.

For all pools we also calculated the residual pool depth as a measure of pool volume (Lisle 1987). We calculated the residual pool depth in each unit by taking the difference between the pool tail depth, located at pool tail crest, and the maximum depth (Kershner et al. 2004). We identified the pool crest visually as the point at which there is a break or transition in stream channel slope.

Large Woody Debris (LWD)

The importance of stream habitat diversity is undeniable for aquatic biota (Lonzarich and Quinn, 1995). Structural elements (e.g., woody debris) often control geomorphic processes (Abbe and Montgomery 2003; Montgomery et al. 2003) and can benefit salmonid communities (Fausch and Northcote 1992; Cederholm et al. 1997; Johnson et al. 2005). Here, we enumerated all large woody debris (LWD) equal to or greater than

1m within the bankfull channel; we further delineated pieces based on diameter as either *large* (diameter greater than 50 cm) or *small* (10cm – 50cm). If the stem was cracked or broken it was treated as one piece if any portions were still connected. If the piece was broken completely and both pieces met criteria, then they were both counted. Where pieces extended outside of the bankfull prism, we only considered the portions of pieces within the prism. If a piece extended across multiple habitat unit boundaries, we considered these as separate pieces in our counts for each unit, but only where the criteria for LWD were met.

We summarized LWD by obtaining reach based estimates of pieces per unit surface area (Count / (Average Width * Reach Length) for both size categories. This approach enabled us to estimate the LWD density of each size category separately as well as a sum across the large and small categories to get a total estimate of LWD density for a given reach.

Substrate Composition

We classified surface sediment composition using a two-tiered visual estimation approach (Buffington and Montgomery 1999) to quantify the dominant and sub-dominant size classes of substrate in habitat units. First we estimated extent of each habitat unit (to the nearest 5%) with different sediment characteristics. We modified the size class categories from Lane et al. (1947) and Church et al. (1987), which were generally consistent with the CHaMP protocol, and included: exposed bedrock surface, boulder (>256mm), large cobbles (127.1 – 256mm), small cobbles (64.1 – 127mm), gravel (2 – 64mm), and fines (<2mm).

Next, we used this information to quantify the dominant and subdominant sediment characteristics of each habitat unit (Buffington and Montgomery 1999).

Given concerns in bias for visual sediment estimates (e.g., McHugh and Budy 2005), we also conducted systematic pebble counts (Wolman 1954) at transects in the first ten habitat units of each stream. We placed transects at approximately 1/2 the total habitat length and we selected particles at ten equally spaced sampling points spanning the width of the wetted channel. Substrate particles were selected while turning eyes away and extending finger down and picking up the first particle touch at the tip of the surveyors boot (Wolman 1954; Kondolf 2000) and measured the b-axis of each particle (Kershner at al. 2004). We then regressed the sediment data from the ocular estimates against the sediment measurement from the pebble counts to quantify observer error and bias in the ocular estimates. The resulting regression equation ($R^2 = 0.51$) was then used to correct all ocular estimates of substrate across streams.

For all pool units, we also quantified the substrate characteristics of pool tailouts for measures of the quality of salmonid spawning habitat (McHugh and Budy 2005). We

visually identified pool tailouts as 20% of the channel unit upstream of the pool tail crest. We surveyed pool tailouts in the same manner but in addition to the pool habitat unit. Pool tailouts were not measured individually for width or length. The surface area of a pool tailout was estimated to be on average 20 percent of the surface area of the pool per EDT instructions. We estimated the proportion of fine sediment within each pool tailout using the same two-tiered visual estimation approach. To ensure accurate summaries of pool tail fines at the reach scale we used a weighted approach. The weighted average of percent fines over pool tails within a reach, with weights proportional to the surface area of each pool tail were summed to obtain a reach based estimate of percent fines in pool tails.

Riparian Function

Riparian forests exert strong controls on stream microclimates (Gregory et al. 1991; Caissie 2006), in-channel morphology (Bilby and Ward 1991; Montgomery et al. 1995), nutrient cycling (McClain et al. 2003), and water quality (Gilliam 1994; Naiman and Decamps 1997). We evaluated the riparian condition and assigned a riparian index rating for each habitat unit. We adopted an index rating from Blair et al. (2008) ranging from zero (no anthropogenic alterations with strong linkages to floodplain) to 4 (complete severing of floodplain stream linkages).

In order to obtain accurate estimates of riparian condition at the reach scale, we again used a weighted approach. The weighted average of riparian condition rating was calculated by using a weighted term that was proportional to the surface area of that habitat unit to the total surface area of the reach and multiplying it by the riparian condition rating. These weighted riparian condition ratings were then summed to the reach scale, and rounded to the nearest whole number.

In addition to habitat characteristics, we also quantified attributes that drive habitat forming processes and water quality. At the stream level we measured (1) stream gradient; (2) ambient stream temperature; (3) stage discharge relationships; and (4) dissolved oxygen (DO).

Longitudinal Gradient Survey

We initially collected gradient profiles for each tributary using a Trimble GeoXH 6000 which is capable of collecting high resolution elevation data (10cm). Before each gradient survey, we surveyed an established reference location with constant x, y, and z coordinates to use as a survey control station. This reference was Yale Lake monument (monument ID 6791; Washington State Department of Transportation). We used the reference survey data to obtain estimates of bias and accuracy. By incorporating reference surveys estimates we are able to evaluate and correct for potential bias in our

elevation data. Our reference surveys were found to deviate from the control station (mean: 0.72 cm, SD: 0.41).

We analyzed the longitudinal gradient data in ArcGIS to obtain estimates of gradient at the reach scale. We snapped the reach boundaries for each stream to the gradient profile, and calculated gradient as the difference between reach boundary elevations divided by the length (rise / run).

Temperature and discharge

Given the importance of stream flow and thermal profiles in driving fundamental biological processes, we collected stream temperature and discharge data in each survey stream. Where discharge data were not available from existing monitoring stations we quantified stage discharge relationships for estimates of summer and early autumn hydrologic regimes (June – October). We installed pressure transducers (Solinst Levelogger) near the mouth of each stream and collected hourly measurements of the stage (i.e., river height). In addition, we installed a barometric pressure logger to control for basin-specific changes in barometric from ambient weather that can influence stage measurements. During the period of deployment we measured discharge at least three times per stream to establish stream-specific stage-discharge relationships

We collected stream temperature data with each pressure transducer and placed additional temperature loggers at the upstream barrier in all study streams to measure ambient stream temperature profiles. All temperature loggers were tested and calibrated before deployment (Dunham et al. 2005). Stream temperature was collected in each study stream from June 1st through the end of October. Temperature data was collected at one hour intervals using Onset Tidbit v2 (range -20 to 50°C, accuracy ± 0.53°C from 0° to 50°C), and Onset HOBO Pendant (range -20° to 70°C, accuracy ± 0.2°C from 0° to 50°C) data loggers. Data loggers were housed in white PVC housing to avoid bias from direct solar radiation (sensu Isaak and Horan 2011).

For both the upstream and downstream temperature logging devices we calculated the mean temperate for each day. We then averaged these two numbers to get an overall daily temperature and then calculated the mean and range from these numbers for June through October.

Dissolved Oxygen

We measured dissolved oxygen at the end of August using a YSI multimeter model (Pro 2030). Measurements were taken in fast moving water at the top and bottom of each study stream. We summarized dissolved oxygen at the reach level, by correcting for lapse change which we estimated by differencing our top and bottom measurements then dividing by total length of the stream. This provided a per meter rate of change

which could then be multiplied by each reach length and added to the previous reach based estimate of dissolved oxygen.

Results and discussion

Surveys of available habitat indicated a total of 36.2 km of available habitat in tributaries to Yale Lake and Lake Merwin (Table 1; Figure 1). The majority (28.0 km) were in tributaries to Yale Lake with Siouxon Creek (and tributary; total = 6.4 km), Speelyai Creek (and tributaries; total = 7.5 km), the Swift Bypass Channel (4.9 km), and Cougar Creek (2.7 km) having the majority of available habitat. Within Lake Merwin, Brooks Creek (including tributary B1; 4.1 km) and Indian George Creek (1.5 km) contain the most available habitat. Brooks Creek (10,318 m^2) in Lake Merwin and Cougar Creek (35,565 m^2), Siouxon Creek (96,753 m^2), Speelyai Creek (45,346 m^2), and the Swift Bypass channel (119,180 m^2) in Yale Lake contained the most amount of habitat based on surface area (Table 2).

The complexity of stream habitat based on stream habitat type and the depth of stream habitat varied considerably across streams during the summer (July – August) when surveys occurred (Table 3). Across all streams, the density of small LWD (diameter 10cm – 50cm) was substantially greater than large LWD (diameter > 50 cm; Table 4). We observed the highest densities of LWD in Cape Horn, Jim and Lower Speelyai Creeks (Lake Merwin) and Dog, NF Siouxon, Ole, and Panamaker Creeks (Yale Lake; Table 4). In general, we found little evidence of fine sediment in tributaries to either reservoir (Table 5), suggesting sediment is unlikely limiting salmonid populations (Tappel and Bjornn 1983; McHugh et al. 2004).

Average temperatures ranged considerably across tributaries and the differences across years varied by stream (Tables 6, 7). The largest range of temperatures occurred during August. Across years, we found generally consistent temperatures during June and July, but 2014 September and October temperatures were dramatically warmer than 2013 in all streams except Cougar Creek. Average summer (July-August) temperatures in Buncombe Hollow, Dog Creek, Siouxon Creek, and Speelyai Creek (2014) were generally higher than preferred juvenile rearing temperatures for Coho and Chinook salmon (ODEQ 1995). As expected, measures of dissolved oxygen were largely correlated with observed temperatures (Table 8).

Riparian conditions and stream gradients varied substantially across tributaries. Overall, we found little evidence of excessive alteration in riparian conditions in the majority of tributaries (Table 8). However, substantial riparian degradation was found in Lower Speelyai Creek (Lake Merwin), Dog Creek, Brooks Creek, and the Swift Bypass Channel. Aside from Cougar Creek and the Swift Bypass Channel, average stream

gradients in all tributaries exceeded 2% with the highest gradients (~10%) in tributaries to Speelyai Creek.

Together, these data suggest habitat conditions, aside from some thermal constraints during the summer months and riparian degradation in some tributaries, do not appear to be limiting salmonid populations. More limiting, however, appears to be the habitat availability, particularly across tributaries to Lake Merwin, which suggests little potential for establishing and maintaining viable populations of anadromous fishes in this reservoir (McElhany et al. 2000; Nickelson 2000). Ultimately, assessing the potential for viable populations will require consideration not only of habitat availability, but also biotic interactions between anadromous fishes and extant species (e.g., Zimmerman 1999).

References

Abbe, T. B., and D. R. Montgomery. 2003. Patterns and processes of wood debris accumulation in the Queets river basin, Washington. Geomorphology 51(1-3):81-107.

Achord, S., R. W. Zabel, and B. P. Sandford. 2007. Migration timing, growth, and estimated parr-to-smolt survival rates of wild snake river spring-summer Chinook salmon from the salmon river basin, Idaho, to the Lower Snake River. Transactions of the American Fisheries Society 136(1):142-154.

Aitken, J. K., C. K. Cook-Tabor, and R. C. Wunderlich. 1996. Travel time of coho salmon and steelhead smolts emigrating through Howard Hanson Reservoir, King County, Washington. U.S. Fish and Wildlife Service, Olympia, Washington.

Anderson, J. H., G. R. Pess, R. W. Carmichael, M. J. Ford, T. D. Cooney, C. M. Baldwin, and M. M. McClure. 2014. Planning Pacific Salmon and Steelhead Reintroductions Aimed at Long-Term Viability and Recovery. North American Journal of Fisheries Management 34(1):72-93.

Anlauf, K. J., D. W. Jensen, K. M. Burnett, E. A. Steel, K. Christiansen, J. C. Firman, B. E. Feist, and D. P. Larsen. 2011. Explaining spatial variability in stream habitats using both natural and management-influenced landscape predictors. Aquatic Conservation-Marine and Freshwater Ecosystems 21(7):704-714.

Beauchamp, D. A., C. Luecke, W. A. Wurtsbaugh, H. G. Gross, P. E. Budy, S. Spaulding, R. Dillenger, and C. P. Gubala. 1997. Hydroacoustic assessment of abundance and diel distribution of sockeye salmon and kokanee in the Sawtooth Valley Lakes, Idaho. North American Journal of Fisheries Management 17:253-267.

Beauchamp, D. A., D. Parrish, and P. Whaley. 2009. Salmonids/coldwater species in large standing waters. Pages 97-117 in S. A. Bonar, D. Willis, and W. A. Hubert, editors. Standard Sampling Methods for North American Freshwater Fishes. American Fisheries Society, Bethesda, Maryland.

Bilby, R. E., and J. W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and 2nd-growth forests in southwesetern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48(12):2499-2508.

Blair, G. R., L. C. Lastelle, and L. E. Mobrand. 2008. The ecosystem diagnosis and treatment model: a tool for assessing salmonid performance potential based on habitat conditions. American Fisheries Society Symposium 71:22.

Brakensiek, K. E., and D. G. Hankin. 2007. Estimating overwinter survival of juvenile Coho salmon in a northern California stream: Accounting for effects of passive integrated transponder tagging mortality and size-dependent survival. Transactions of the American Fisheries Society 136(5):1423-1437.

Buffington, J. M., and D. R. Montgomery. 1999. A procedure for classifying textural facies in gravel-bed rivers. Water Resources Research 35(6):1903-1914.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51(8):1389-1406.

Candy, J. R., and T. D. Beacham. 2000. Patterns of homing and straying in southern British Columbia coded-wire tagged chinook salmon (Oncorhynchus tshawytscha) populations. Fisheries Research 47(1):41-56.

Cederholm, C. J., R. E. Bilby, P. A. Bisson, and B. Bumstead. 1997. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. North American Journal of Fisheries Management 17:947-963.

CHaMP. 2013. Columbia Habitat Monitoring Program. Scientific protocl for salmonid habitat surveys with the Columbia Habitat Monitoring Program. Prepared by the Integrated Status and Effectiveness Monitoring Program, Wauconda, Washinton.

Dilley, S. J. 1994. Horizontal and vertical distribution of juvenile salmonids in Howard Hanson Reservoir. U.S. Fish and Wildlife Service, Olympia, Washington.

Dunham, J., G. Chandler, B. Rieman, and D. Martin. 2005. Measuring stream temperature with digital data loggers: a user's guide. U.S. Department of Agriculture, Forest Service, Gen. Tech. Rep. RMRS GTR-150WWW, Fort Collins, Colorado.

Durkin, J. T., P. D. L., and R. F. Raleigh. 1966. Distribution and movement of juvenile salmon in Brownlee Reservoir, 1962-1965. Bureau of Commercial Fisheries Biological Laboratory, Seattle, Washington.

Eggers, D. M. 1978. Limnetic feeding behavior of juvenile sockeye salmon in Lake Washington and predator avoidance. Limnology and Oceanography 23:1114-1125.

Fausch, K. D., and T. G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693.

Gilliam, J. W. 1994. Riparian wetlands and water quality. Journal of Environmental Quality 23(5):896-900.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. Bioscience 41(8):540-551.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, L. M. Decker, S. V. Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18(6):3-12.

Isaak, D. J., and D. L. Horan. 2011. An Evaluation of Underwater Epoxies to Permanently Install Temperature Sensors in Mountain Streams. North American Journal of Fisheries Management 31(1):134-137.

Johnson, S. L., J. D. Rodgers, M. F. Solazzi, and T. E. Nickelson. 2005. Effects of an increase in large wood on abundance and survival of juvenile salmonids (Oncorhynchus spp.) in an Oregon coastal stream. Canadian Journal of Fisheries and Aquatic Sciences 62(2):412-424.

Kershner, J. L., B. B. Roper, N. Bouwes, R. Henderson, and E. Archer. 2004. An analysis of stream habitat conditions in reference and managed watersheds on some federal lands within the Columbia River basin. North American Journal of Fisheries Management 24(4):1363-1375.

Kiffney, P. M., G. R. Pess, J. H. Anderson, P. Faulds, K. Burton, and S. C. Riley. 2009. Changes in fish communities following recolonization of the Cedar River, WA, USA by Pacific salmon after 103 years of local extirpation. River Research and Applications 25(4):438-452.

Kondolf, G. M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129(1):262-281.

Larsen, D. P., P. R. Kaufmann, T. M. Kincaid, and N. S. Urquhart. 2004. Detecting persistent change in the habitat of salmon-bearing streams in the Pacific Northwest. Canadian Journal of Fisheries and Aquatic Sciences 61(2):283-291.

Lestelle, L. C., L. E. Mobrand, J. A. Lichatowich, and T. S. Vogel. 1996. Applied ecosystem analysis-a primer. Bonneville Power Administration, Project Number 9404600, Portland, Oregon.

Marcot, B. G., C. S. Allen, S. Morey, D. Shively, and R. White. 2012. An Expert Panel Approach to Assessing Potential Effects of Bull Trout Reintroduction on Federally Listed Salmonids in the Clackamas River, Oregon. North American Journal of Fisheries Management 32(3):450-465.

McClain, M. E., E. W. Boyer, C. L. Dent, S. E. Gergel, N. B. Grimm, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, W. H. McDowell, and G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6(4):301-312.

McClure, M. M., S. M. Carlson, T. J. Beechie, G. R. Pess, J. C. Jorgensen, S. M. Sogard, S. E. Sultan, D. M. Holzer, J. Travis, B. L. Sanderson, M. E. Power, and R. W. Carmichael. 2008. Evolutionary consequences of habitat loss for Pacific anadromous salmonids. Evolutionary Applications 1(2):300-318.

McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of Evolutionary Significant Units. National Marine Fisheries Service, Seattle, Washington.

McHugh, P., and P. Budy. 2005. A comparison of visual and measurement-based techniques for quantifying cobble embeddedness and fine-sediment levels in salmonid-bearing streams. North American Journal of Fisheries Management 25(4):1208-1214.

McHugh, P., P. Budy, and H. Schaller. 2004. A model-based assessment of the potential response of Snake River spring-summer Chinook salmon to habitat improvements. Transactions of the American Fisheries Society 133(3):622-638.

Montgomery, D. R., J. M. Buffington, R. D. Smith, K. M. Schmidt, and G. Pess. 1995. Pool spacing in forest channels. Water Resources Research 31(4):1097-1105.

Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. Ecology and Management of Wood in World Rivers 37:21-47.

Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28:621-658.

Nickelson, T. E. 2000. Population assessment: Oregon Coast coho salmon ESU. Oregon Department of Fish and Wildlife, Portland, Oregon.

NMFS. 2007. Biological opinion and Magnuson-Stevens Fishery Conservation and Management Act Consultion; operation of PacifiCorp and Cowlitz PUD's Lewis River Hydroelectric Project (FERC Nos. 935, 2071, 2111, and 2213). National Oceanic and Atmospheric Administration: National Marine Fisheries Service, 2005/05891, Seattle, Washington.

ODEQ. 1995. Temperature: 1992-1994 water quality standars review. Final issue paper. Oregon Department of Environmental Quality.

PacifiCorp/Cowlitz-PUD. 2001. Migratory behavoir of radio-tagged juvenile Coho Salmon through Swift Reservoir, 2001. PacifiCorp, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2002. Migratory behavoir of radio-tagged juvenile Chinook Salmon through Swift Reservoir, 2002. PacifiCorp/Cowlitz PUD, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2004. Behavoir of salmonid smolts at Swift Dam using 3-dimensional tracking with acoustic tags. PacifiCorp/Cowlitz-PUD, AQU 17, Portland, Oregon.

Pess, G. R., P. M. Kiffney, M. C. Liermann, T. R. Bennett, J. H. Anderson, and T. P. Quinn. 2011. The Influences of Body Size, Habitat Quality, and Competition on the Movement and Survival of Juvenile Coho Salmon during the Early Stages of Stream Recolonization. Transactions of the American Fisheries Society 140(4):883-897.

Quinn, T. P. 1993. A review of homing and straying of wild and hatchery-produced salmon. Fisheries Research 18:29-44.

Serl, J., and C. Morrill. 2010. Summary report for the 1996 to 2009 seasonal operation of the Cowlitz Falls fish facility and related Cowlitz Falls anadromous reintroduction program activities. Washington Department of Fish and Wildlife.

Sheer, M. B., and E. A. Steel. 2006. Lost watersheds: Barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and Lower Columbia River basins. Transactions of the American Fisheries Society 135(6):1654-1669.

Tappel, P. D., and T. C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management 3:123-135.

Temple, G. M., and T. N. Pearsons. 2012. Risk management of non-target fish taxa in the Yakima River Watershed associated with hatchery salmon supplementation. Environmental Biology of Fishes 94(1):67-86.

Urquhart, N. S., S. G. Paulsen, and D. P. Larsen. 1998. Monitoring for policy-relevant regional trends over time. Ecological Applications 8(2):246-257.

USFWS. 2006. Biological Opinion for the Federal Energy Regulatory Commission relicensing of the Lewis River Hydroelectric Projects: Merwin (No. 935), Yale (No. 2071), Swift No. 1 (No. 2111), and Swift No. 2 (No. 2213). U.S. Fish and Wildlife Service, FWS Reference Number: 1-3-06-F-0177, Lacey, Washington.

Weigel, D. E., P. J. Connolly, K. D. Martens, and M. S. Powell. 2013. Colonization of Steelhead in a Natal Stream after Barrier Removal. Transactions of the American Fisheries Society 142(4):920-930.

Wolman, M. G. 1954. A method of sampling coarse river-bed material. Transactions American Geophysical Union 35:951-956.

Zimmerman, M. P. 1999. Food habits of smallmouth bass, walleyes, and northern pikeminnow in the Lower Columbia River Basin during outmigration of juvenile anadromous salmonids. Transactions of the American Fisheries Society 128:1036-1054.

Table 1. The total stream length, average bankfull width (SD), wetted width (SD), and number of sample reaches (n) in each tributary stream to Lake Merwin and Yale Lake. Note the number of sample reaches corresponds to all subsequent tables.

	Stream	Length (km)	Bankfull width (m)	Wetted width (m)	N
Lake Merwin	Brooks Creek	4.1	7.7 (2.8)	3.3 (0.6)	17
	Buncombe Hollow Creek	1.1	5.6 (2.0)	2.9 (0.2)	9
	Cape Horn Creek	0.5	13.4 (0.6)	6.0 (0.6)	2
	Indian George Creek	1.5	10.2 (1.5)	3.9 (0.2)	7
	Jim Creek	0.5	12.3 (0)	5.29 (0)	2
	Lower Speelyai	0.3	16.6 (0)	6.5 (0)	1
	Rock Creek	0.2	-	-	-
		8.2			**38**
Yale Lake	Cougar Creek	3.9	16.1 (4.3)	8.1 (1.7)	8
	Dog Creek	0.3	15.9 (0)	5.6 (0)	1
	North Siouxon Creek	0.7	24.7 (0.4)	14.5 (0.1)	2
	Ole Creek	1.7	13.8 (2.0)	5.7 (0.4)	6
	Panamaker Creek	0.4	10.7 (0.7)	4.5 (1.0)	2
	Siouxon Creek	6.1	28.4 (5.0)	15.6 (3.0)	8
	Speelyai Creek	6.0	17.3 (14.4)	7.1 (1.8)	23
	Swift Bypass Channel	6.5	20.8 (6.1)	16.2 (5.6)	11
	West Fork Speelyai Creek	1.3	13.3 (1.6)	6.0 (0.5)	5
	West Tributary Speelyai Creek	1.1	9.9 (1.9)	5.4 (0.6)	5
Total		**28.0**			**71**

Table 2. The total surface area (SA) by reach and the average (SD) SA in pool habitat, pool tailouts (i.e., spawning habitat), and glides in each tributary stream to Lake Merwin and Yale Lake.

Reservoir	Stream	Reach SA (m^2)	Pool SA (m^2)	Pool tailout SA (m^2)	Glide SA (m^2)
Lake Merwin	Brooks Creek	10,318	90 (40)	18 (8)	69 (53)
	Buncombe Hollow Creek	3,373	119 (114)	24 (23)	7 (10)
	Cape Horn Creek	3,103	530 (13)	38 (3)	65 (65)
	Indian George Creek	6,165	142 (57)	74 (11)	64 (48)
	Jim Creek	2,692	604 (89)	46 (18)	90 (26)
	Lower Speelyai	1,738	120 (0)	45 (0)	414 (0)
Total		**27,389**			
Yale Lake	Cougar Creek	35,565	319 (327)	64 (65)	444 (425)
	Dog Creek	1,647	386 (0)	77 (0)	27 (0)
	North Siouxon Creek	9,688	1,147 (473)	229 (95)	0 (0)
	Ole Creek	9,975	495 (175)	99 (35)	361 (318)
	Panamaker Creek	1,866	285 (161)	57 (32)	120 (120)
	Siouxon Creek	96,753	3,452 (1,622)	690 (324)	1,052 (729)
	Speelyai Creek	45,346	329 (198)	66 (40)	299 (349)
	Swift Bypass Channel	119,180	2,683 (2815)	537 (563)	4,823 (2,981)
	West Fork Speelyai Creek	8,392	168 (58)	34 (12)	24 (38)
	West Tributary Speelyai Creek	6,208	299 (98)	60 (20)	0 (0)
Total		**334,622**			

Table 3. Average depth (SD) by habitat type and stream in each tributary stream to Lake Merwin and Yale Lake.

Reservoir	Stream	Cascade	Glide	Rapid	Riffle	Beaver pool	Dam pool	Plunge pool	Scour pool
						Average depth (m)			
Lake Merwin	Brooks Creek	0.15 (0.07)	0.2 (0.46)	0.12 (0.04)	0.10 (0.03)	0.21 (0.01)	0.24 (0.08)	0.96 (4.47)	0.26 (0.10)
	Buncombe Hollow Creek	0.10 (0.02)	0.19 (0.01)	0.13 (0.07)	0.10 (0.04)	-	0.29 (0.16)	0.33 (0.27)	0.29 (0.11)
	Cape Horn Creek	0.30 (0.03)	0.24 (-)	0.22 (0.05)	0.21 (-)	-	-	0.61 (0.28)	0.43 (0.17)
	Indian George Creek	0.18 (0.06)	0.16 (0.03)	0.16 (0.04)	0.12 (0.07)	-	0.28 (-)	0.55 (0.51)	0.32 (0.12)
	Jim Creek	0.22 (0.05)	0.2 (0.03)	0.17 (0.08)	0.13 (0.02)	-	0.33 (0.21)	0.33 (0.28)	0.50 (0.37)
	Lower Speelyai	0.19 (0.05)	0.17 (0.05)	0.13 (0.01)	0.18 (-)	-	-	0.22 (0.04)	0.26 (0.05)
Yale Lake	Cougar Creek	-	0.45 (0.15)	0.54 (1.12)	0.23 (0.08)	-	0.58 (0.41)	0.87 (0.55)	0.68 (0.30)
	Dog Creek	0.17 (0.04)	0.20 (-)	0.18 (0.02)	0.13 (-)	-	-	1.54 (1.79)	0.36 (0.14)
	North Siouxon Creek	0.43 (0.08)	-	0.26 (0.07)	-	-	-	1.04 (0.28)	-
	Ole Creek	0.20 (0.05)	0.28 (0.31)	0.16 (0.05)	0.10 (0.02)	-	-	0.72 (1.0)	0.53 (0.33)
	Panamaker Creek	0.20 (0.09)	0.20 (0.01)	0.20 (0.06)	-	-	-	1.15 (0.89)	0.51 (0.41)
	Siouxon Creek	0.67 (0.25)	0.52 (0.1)	0.42 (0.1)	0.34 (0.1)	-	-	0.56 (10.08)	1.44 (1.37)
	Speelyai Creek	0.26 (0.06)	0.39 (0.54)	0.27 (0.32)	0.24 (0.07)	-	0.98 (-)	0.53 (0.48)	0.70 (0.54)
	Swift Bypass Channel	0 (0.25)	0.67 (0)	0.48 (0.18)	0.28 (0.15)	1.37 (1.06)	0.49 (0.31)	0.69 (0.40)	0.70 (0.69)
	W. Fork Speelyai Creek	0.26 (0.06)	-	0.24 (0.04)	-	-	-	1.07 (1.93)	-
	W. Tributary Speelyai Creek	0.22 (0.06)	0.24 (0.05)	0.51 (1.25)	0.16 (0.03)	-	-	0.53 (0.33)	0.38 (0.25)

Table 4. The average (SD) density of large woody debris (LWD) delineated by large (lg: diameter >50 cm) and small (sm; 10<diameter<50 cm) pieces (#/m^2), and percent surface fines in pool tailouts in each tributary stream to Lake Merwin and Yale Lake.

.

Reservoir	Stream	LWD density (lg)	LWD density (sm)	% fines in tailout
Lake Merwin	Brooks Creek	0.15 (0.09)	0.79 (0.56)	5.4 (5.3)
	Buncombe Hollow Creek	0.02 (0.03)	0.07 (0.07)	12.3 (13.3)
	Cape Horn Creek	0.31 (0.09)	1.91 (0.18)	2.0 (0.5)
	Indian George Creek	0.17 (0.06)	0.96 (0.36)	2.5 (1.5)
	Jim Creek	0.41 (0.07)	2.13 (0.01)	0.6 (0.0)
	Lower Speelyai	0.40 (0)	1.59 (0)	4.9 (0)
Yale Lake	Cougar Creek	0.22 (0.18)	0.42 (0.32)	0.2 (0.4)
	Dog Creek	0.10 (0)	2.97 (0)	1.0 (0)
	North Siouxon Creek	0.39 (0.22)	1.05 (0.57)	2.3 (2.3)
	Ole Creek	0.16 (0.17)	0.99 (1.09)	0.2 (0.4)
	Panamaker Creek	0.40 (0.20)	1.68 (0.12)	5.2 (4.0)
	Siouxon Creek	0.16 (0.08)	0.43 (0.19)	0.3 (0.6)
	Speelyai Creek	0.17 (0.34)	0.40 (0.54)	3.8 (9.9)
	Swift Bypass Channel	0.03 (0.08)	0.12 (0.12)	0 (0)
	W. Fork Speelyai Creek	0.22 (0.18)	0.44 (0.26)	2.0 (3.0)
	W. Tributary Speelyai Creek	0.11 (0.09)	0.35 (0.31)	1.2 (1.7)

Table 5. Average size (SD) of streambed sediment particles for different habitat units found in each tributary stream to Yale Lake and Lake Merwin. Note: a dash indicates no habitat unit present.

	Average particle size (mm)								
	Beaver Pool	Cascade	Dam Pool	Glide	Off Channel	Plunge Pool	Rapid	Riffle	Scour Pool
Brooks Ck	85.7 (6.1)	157.2 (58.6)	82.8 (2.0)	84.5 (4.5)	-	92.1 (28.3)	102.7 (32.4)	85.5 (3.5)	85.1 (8.3)
Buncombe Hollow Ck	-	61.9 (0)	80.8 (0.8)	87.1 (6.0)	-	80.7 (11.0)	87.1 (12.7)	82 (0.8)	83.9 (7.9)
Bypass Channel	87.6 (3.6)	0 (0)	83.2 (3.2)	141 (54.5)	136.3 (69.1)	163 (59.9)	159 (59.2)	112.1 (32.0)	110.2 (48.0)
Cape Horn Ck	-	110.5 (65)	-	89.9 (-)	-	98.2 (41.7)	118.8 (52.8)	87.7 (-)	87.1 (3.0)
Cougar Ck	-	0 (0)	83.5 (3.0)	89.5 (5.6)	-	92.3 (19.8)	100.3 (27.5)	86.2 (6.9)	88.2 (6.6)
Dog Ck	-	184.7 (35.9)	-	94 (0.9)	-	76.3 (20.3)	138.7 (70.0)	92.1 (-)	163.3 (52.8)
Indian George Ck	-	117 (58.7)	81.6 (-)	103.7 (29.2)	-	100.7 (51.1)	100.6 (22.5)	90.0 (2.0)	84.9 (3.3)
Jim Ck	-	99.4 (44.6)	71.5 (13.5)	142.7 (67.1)	-	143.9 (57.4)	101.3 (21.4)	87.9 (4.2)	93.6 (27.5)
Lower Speelyai	-	152.4 (42.8)	-	83.9 (2.9)	-	82 (1.0)	95.5 (30.1)	82.8 (0.5)	111.3 (43.8)
North Siouxon Ck	-	251.7 (64.7)	-	-	-	139.6 (65.2)	136.2 (105)	-	-
Ole Ck	-	191.5 (67.3)	-	113.6 (36)	75 (18.5)	140.8 (53.2)	137.9 (34.0)	98.5 (26)	116.7 (37.5)
Panamaker Ck	-	117.8 (40.2)	-	97.4 (3.5)	-	125.3 (61.0)	156.1 (52.3)	-	92.0 (1.8)
Siouxon Ck	-	169.8 (94)	-	127.5 (50.9)	-	118.1 (53.4)	178.1 (81.2)	146.1 (77.3)	155.8 (56.9)
Speelyai Ck	-	166.1 (65.8)	137.2 (-)	122.2 (49.0)	85.8 (3.5)	134.8 (65.1)	181.8 (59.9)	115.1 (28.9)	137.2 (53.3)
W. Fork Speelyai Ck	-	211.9 (74.8)	-	-	-	147.6 (66.3)	155.2 (43.8)	-	-
W. Trib. Speelyai Ck	-	174.1 (50.0)	-	96.7 (8.1)	81.1 (-)	162.8 (57.0)	172.1 (49.5)	96.7 (2.4)	145.2 (48.3)

Table 6. The average and range of stream temperatures by month from July through October for Clear Creek (Swift Reservoir) and in each tributary stream to Lake Merwin and Yale Lake in 2013.Note: the logger and data for Siouxon Creek and Cape Horn Creeks were lost during the high flow event during September.

Reservoir	Stream	July		August		September		October	
		Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
Lake Merwin	Brooks Creek	13.0	10.6 - 15.5	13.0	11.4 - 15.1	12.6	10.1 - 15.3	9.5	7.7 - 11.2
	Buncombe Hollow	16.3	14.2 - 18.7	16.4	14.2 - 18.4	14.7	11.3 - 17.6	10.2	8.3 - 11.6
	Indian George	14.3	11.5 - 16.5	15.2	13.5 - 16.7	13.8	9.7 - 16.7	9.0	7.4 - 10.6
	Jim Creek	13.8	11.2 - 15.7	14.6	12.9 - 15.8	13.4	10.0 – 16.0	9.1	7.4 - 10.4
Yale Lake	Cougar Creek	6.8	6.2 - 7.9	6.8	6.4 - 7.9	6.8	6.2 - 8.1	6.8	6.5 - 7.2
	Dog Creek	13.0	10.5 - 14.9	17.0	12.9 - 23.3	14.6	9.1 - 20.5	9.3	8.2 - 10.7
	Ole Creek	13.8	11.6 - 15.8	14.8	13.2 – 16.0	13.2	8.9 - 15.6	8.4	7.5 - 9.3
	Panamaker Creek	12.8	10.5 - 14.9	14.0	12.2 - 16.5	na	na	na	na
	Speelyai Creek	15.3	11.6 - 19.2	16.2	13.6 - 19.5	13.7	9 - 18.5	8.7	7.1 - 10.4
Swift Reservoir	Clear Creek	14.0	10.7 - 17.8	14.5	12.6 - 17.9	13.1	7.6 - 17.2	7.3	6.1 - 8.9

[1]July values include period from July 10 – July 31st.

Table 7. The average and range of stream temperatures by month from June through October for Clear Creek (Swift Reservoir) and in each tributary stream to Lake Merwin and Yale Lake in 2014.

Reservoir	Stream	June Avg.	June Range	July Avg.	July Range	August Avg.	August Range	September Avg.	September Range	October Avg.	October Range
Lake Merwin	Brooks Creek	16.5	9.8 - 21.9	13.8	12.4 - 19.4	13.8	12.8 - 14.8	12.4	11.9 - 13.3	11.6	10.4 - 12.5
	Buncombe Hollow Creek	13.7	11.4 - 15.1	17.0	14.9 - 19.1	17.5	15.6 - 19.0	14.8	13.5 - 16.2	12.6	11.4 - 14.0
	Cape Horn Creek	11.2	9.5 - 12.4	14.2	12.4 - 15.8	14.9	13.5 - 16.2	12.9	11.7 - 14.2	11.3	9.7 - 13.0
	Indian George Creek	11.7	9.8 - 13.1	15.1	13.1 - 16.7	16.0	14.7 - 17.1	14.0	12.9 - 15.2	11.8	9.9 - 13.7
	Jim Creek	11.8	10.1 - 13.0	14.9	13.1 - 16.4	15.6	14.3 - 16.6	13.6	12.6 - 14.7	12.3	10.2 - 13.9
	Lower Speelyai	-	-	-	-	-	-	-	-	-	-
Yale Lake	Cougar Creek	7.0	6.5 - 7.7	7.3	7.1 - 7.7	7.4	7.1 - 7.6	7.0	6.7 - 7.2	7.2	6.6 - 8.4
	Dog Creek	9.9	9.2 - 10.7	12.7	11.2 - 14.1	16.5	14.1 - 19.8	15.1	12.8 - 18.2	11.7	9.2 - 15.7
	North Siouxon Creek	10.7	9.0 - 12.1	13.8	11.4 - 15.6	15.3	14.1 - 16.3	13.3	12.4 - 14.0	12.3	10.9 - 14.0
	Ole Creek	10.7	9.6 - 11.7	13.4	11.5 - 14.7	14.7	13.6 - 15.4	12.9	11.8 - 13.7	11.1	9.0 - 12.6
	Panamaker Creek	10.6	9.4 - 11.7	12.7	11.1 - 13.9	14.4	13.7 - 14.9	14.8	12.9 - 17.8	11.5	8.7 - 15.2
	Siouxon Creek	11.4	9.4 - 13.2	15.4	12.4 - 17.7	16.9	14.9 - 18.6	14.0	12.5 - 15.2	12.3	10.9 - 14.0
	Speelyai Creek	11.6	9.6 - 13.3	15.0	12.8 - 17.0	16.8	15.0 - 17.8	14.6	13.1 - 15.6	11.6	9.1 - 14.0
	Swift Bypass Channel	10.5	9.5 - 11.8	12.3	11.0 - 13.1	13.4	12.4 - 14.3	13.3	12.4 - 14.0	12.1	10.3 - 13.2
	W. Fork Speelyai Creek	10.8	9.0 - 12.1	13.6	11.6 - 15.3	15.0	13.4 - 15.9	13.0	11.7 - 14.4	10.8	8.7 - 13.0
	W. Tributary Speelyai Creek	11.3	9.1 - 12.8	14.4	12.3 - 16.4	15.5	13.9 - 16.9	13.3	12.1 - 14.6	12.2	10.6 - 14.0
Swift Reservoir	Clear Creek	10.4	8.1 – 14.4	14.4	10.1 – 19.1	15.6	13.2 – 19.3	13.5	10.8 – 16.7	10.7	6.8 – 14.8

26

Table 8. The average (SD) measures of dissolved oxygen, reach gradient, and riparian function in each tributary stream to Lake Merwin and Yale Lake.

Reservoir	Stream	Dissolved oxygen (ml/l)	Gradient (%)	Riparian function
Lake Merwin	Brooks Creek	9.84 (0.26)	4.4 (2.4)	2.7 (0.7)
	Buncombe Hollow Creek	8.50 (0.92)	3.2 (1.8)	1.6 (1.5)
	Cape Horn Creek	9.54 (0.15)	5.2 (0.9)	0.1 (0.1)
	Indian George Creek	9.65 (0.02)	5.7 (1.8)	0.9 (1.3)
	Jim Creek	9.24 (0.05)	4.7 (1.1)	0 (0)
	Lower Speelyai	9.89 (0)	2.8 (0)	4 (-)
Yale Lake	Cougar Creek	12.29 (0.04)	1.6 (0.9)	1.1 (1.6)
	Dog Creek	8.83 (0)	4.9 (0)	2.8 (-)
	North Siouxon Creek	9.75 (0.08)	7.0 (2.7)	1.8 (0)
	Ole Creek	9.46 (0.25)	2.0 (1.7)	0.9 (1.1)
	Panamaker Creek	9.80 (0.16)	-	0 (0)
	Siouxon Creek	9.73 (0.06)	2.1 (2.3)	0.1 (0.2)
	Speelyai Creek	9.56 (0.36)	6.2 (4.4)	0.9 (1.4)
	Swift Bypass Channel	-	0.7 (0.5)	2.1 (0.2)
	West Fork Speelyai Creek	9.68 (0.05)	10.7 (2.5)	0 (0)
	West Tributary Speelyai Creek	9.86 (0.03)	9.0 (5.6)	0 (0)

Figure 1. Stream habitat potentially available to anadromous salmon (blue) due to upstream barriers (black circles) in tributaries to Yale Lake and Lake Merwin.
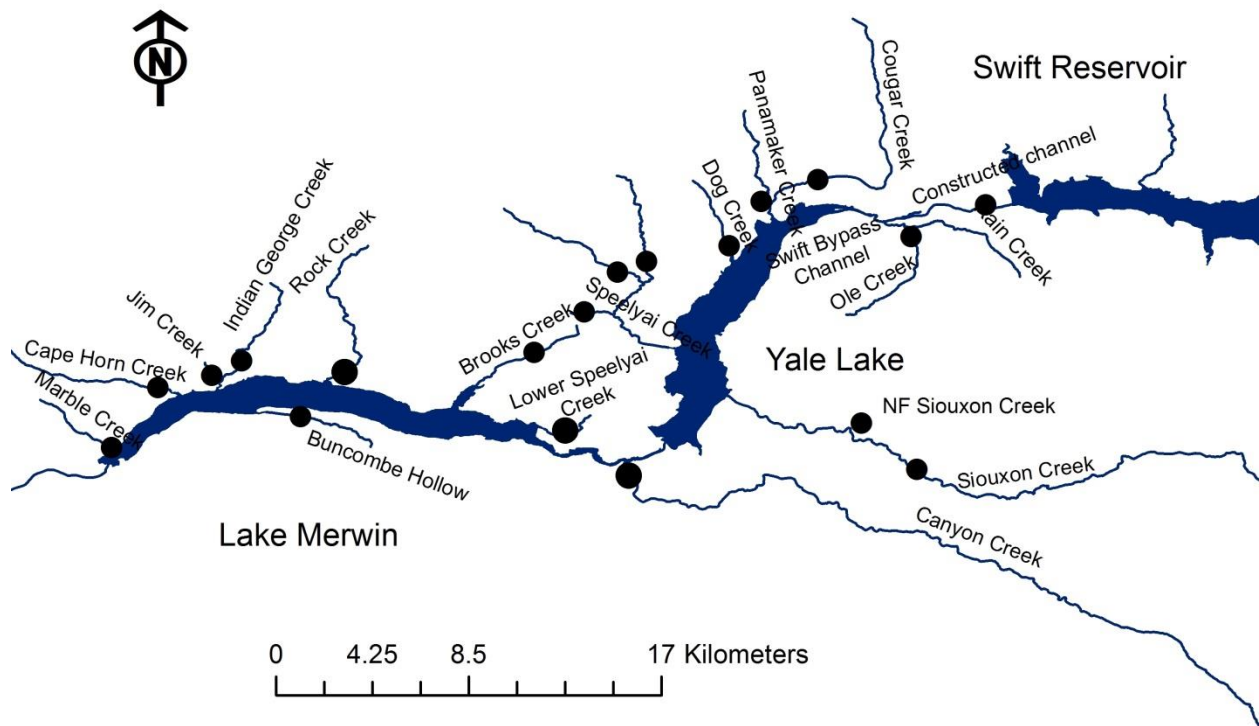
Task 3: Assess adult potential for spawning success

Understanding the ability of reintroduced anadromous species to successfully reproduce in the tributaries to Yale Lake and Lake Merwin is an essential component of the reintroduction program. Under current management plans, anadromous adults will be released into the reservoir systems near the dam facilities; with this, there remains considerable uncertainty in how these fish will sort and utilize available stream habitat. Here, we employed a set of test Coho Salmon introduced into Lake Merwin (based on availability of surplus salmon) to evaluate tributary use and potential for recruitment.

Methods

During 2014 we assessed the adult potential for spawning success in tributaries to Lake Merwin using a test release of adult Coho Salmon. On September 23, 2014 a set of 280 adults were released into the reservoir at Cresap Bay. Beginning in the 2nd week of October, which is the approximate start of Coho spawning observed in the Lewis River upstream of Merwin Dam (*see* Task 6), and continuing through the first week of November we tracked the adult spawning locations using bi-weekly redd and carcass surveys in each of the tributaries. Spawning surveys were continuous and included all habitat available in each tributary. During each survey, we georeferenced the occurrence of spawning activity and delineated observations into three categories: 1) carcasses; 2) observed redds; and 3) staging. We considered fish as "staging" where no redd was clearly observed in the vicinity and fish, and staging fish were not delineated based on male/female pairings.

Results and discussion

During surveys Coho spawning activity was observed in Cape Horn Creek, Jim Creek, Indian George Creek, and Brooks Creek (Figure 1). Spawning activities were observed up to the existing barriers in Cape Horn Creek and Jim Creek, but only partially upstream in Indian George Creek and Brooks Creeks. No spawning activity was observed in Buncombe Hollow. However, use of existing tributaries is likely to vary considerably given the lack of natal homing in the test fish used in this study (Quinn 1993; Candy and Beacham 2000). Follow-up surveys to characterize evidence of successful spawning (i.e., recruitment) will be conducted in each tributary during the 2015 field season.

References

Candy, J. R., and T. D. Beacham. 2000. Patterns of homing and straying in southern British Columbia coded-wire tagged chinook salmon (Oncorhynchus tshawytscha) populations. Fisheries Research 47(1):41-56.

Quinn, T. P. 1993. A review of homing and straying of wild and hatchery-produced salmon. Fisheries Research 18:29-44.

Figure 1. Locations of Coho Salmon release site (star), existing barriers to anadromous species (black triangles), and spawning activity including observed carcass locations (hollow circles), adults observed staging (grey circles), and redds (black circles) during fall spawning surveys in Lake Merwin tributaries.
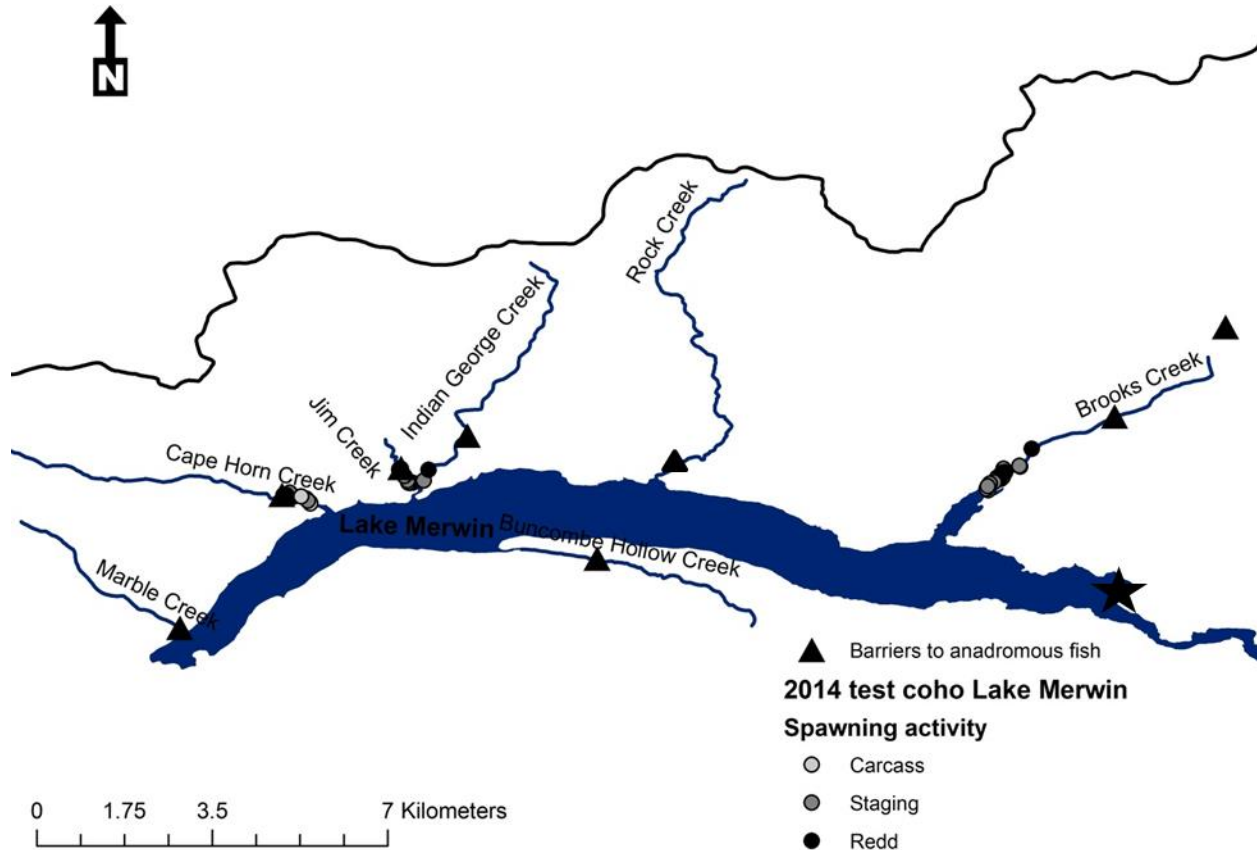
Task 4: Assess juvenile production and emigration success

Given the differences in habitat and resident fishes in each of the reservoirs, an important step of the reintroduction is to evaluate the survival and behavior of smolts in both stream and reservoir environments. Furthermore, understanding the timing of anadromous species emigrations to reservoirs, particularly as it relates to ambient conditions (e.g., streamflow), will provide critical insight into the strength of these factors relative to intra-annual cycles (i.e., seasons). Due to the lack of anadromous fish currently present in Yale Lake and Lake Merwin, it was necessary to thoroughly evaluate these tasks via assessments in Swift Reservoir in combination with data collected from test smolts in Yale Lake. Ultimately linking information regarding emigration timing, travel times within reservoirs, and results from Tasks 5 and 6 will allow for a more comprehensive assessment of the factors that may be limiting successful reintroductions.

Methods

Stream habitat

Wild and native fish

Continuing work initiated during 2013 we again focused our efforts on Clear creek to understand juvenile production and emigration success. We contracted our study area in 2014 by restricting our efforts to reaches that were upstream of the Biomark PIT-tag antenna system (installed in August 2014; latitude 46.11818, longitude -121.9995) to allow for detection of fish moving downstream past the antennas. This resulted in a study area of approximately 10.5 km. We continued using a systematic sampling design with reach lengths of a minimum of 250 m and a minimum target of 33% sampling rate (Figure 1). Reach lengths varied, however, as we began each reach at a pool tail and continued sampling until the first pool beyond 250 m. In 2014, we only surveyed those reaches above our PIT-tag antenna.

After evaluating a variety of sampling techniques in 2013, we determined backpack electrofishing to be the most efficient technique and this method was used exclusively during 2014. All sampling occurred within the first two weeks of September and we continued marking wild Coho Salmon. Once captured we anesthetized fish using AQUI-S (approximately 1.5 ml to 8 liters of water), and recorded weights and lengths (fork length). We marked all salmonids >80mm with a 12-mm full duplex PIT-tag in the dorsal sinus. Upon completion of marking, we placed fish in recovery pens within the sample reach; once equilibrium was fully regained, we returned fish to the sample reach. We spatially referenced each marked fish with a specific sampling reach to allow for analyses of spatial and temporal movement patterns.

Ultimately integrating data from individual marking and recapture events will allow for robust estimates of production to the Swift floating fish collector (FSC) in Swift Reservoir. We computed initial survival estimates for wild Coho during the tributary phase using individual mark-recapture information including marking, antenna recapture events, and recaptures at the FSC. We used a Cormack-Jolly-Seber mark-recapture model, which is commonly used for anadromous species survival estimates (Achord et al. 2007; Brakensiek and Hankin 2007), to estimate tributary survival for fish marked in 2013; we did not include fish marked in 2014 as emigration and rearing continue, which would likely bias estimates of survival. For these initial estimates, we did not include covariates (e.g., length, condition, etc.) in our analyses, but factors likely influencing survival (e.g., Pess et al. 2011) will be evaluated in our final analysis.

Acclimation Chinook and Coho Salmon

Hatchery acclimation salmon remain an important part of the reintroduction efforts in the Upper Lewis River. Starting in 2013, a portion of acclimation smolts have been PIT-tagged (Table 1) providing information on emigration patterns and success. Here, we used date of release, antennae recapture information, and date at which fish are captured at the FSC to estimate residence time and timing of emigration from tributaries, travel times across the reservoir to the collector, and proportion of individuals reaching the collector.

Reservoir habitat

To evaluate smolt movement behavior and migration timing through Yale Lake, and to locate smolt aggregations in the forebay to guide the siting of a future smolt collector, Coho Salmon smolt movements were assessed before, during and after a test release using hydroacoustics. A total of 5,000 hatchery raised Coho Salmon smolts were released into Yale Lake at Yale Park (middle of reservoir; Figure 2) on April 8, 2014 at 09:50 h.

Hydroacoustic surveys of 2014 smolt test release in Yale Lake

The hydroacoustics system used for all surveys consisted of two multiplexed split-beam 200 kHz transducers towed in side-looking (12.8° full beam angle) and down-looking (6.8° full beam angle) orientations. The transducers were connected to a Biosonics DE-6000 scientific echo sounder with the following data acquisition parameters: bottom threshold = 70 m, ping rate = 2-3 pps, pulse width = 0.3 ms, and target strength detection threshold = -65 dB. Surveys consisted of transects extending the length of the east and west banks, and along the center axis of the reservoir, as well as a zig-zag transect going between banks through the forebay and within the lower ¼ of the reservoir. We divided transects running the length of the reservoir into five distinct analysis regions based on minutes of latitude (Figure 2). These divisions were designed

to obtain finer spatial resolution in fish distribution, yet maintain enough sampling power within each spatial unit to obtain reliable estimates of fish densities. Most transects were surveyed during daylight, but some regions of the reservoir were surveyed during dusk (defined as 1 hour after sunset) and at night. Light levels (lux and microeinsteins: [uE]·s^{-1}·m^{-2}) were recorded at the start, end, and periodically throughout each transect. Light intensity is important, as fish like Kokanee (*O. nerka*), the primary pelagic planktivore in Yale Lake, often exhibit diel vertical migrations and their distribution is highly sensitive to time of day and light conditions (e.g., Eggers 1978).

Pre-release surveys were conducted in Yale Lake in order to establish the background distribution and density of resident fish targets on April 7, 2014, and the morning of April 8, 2014. The daylight component of the survey conducted on April 7, 2014 consisted of transects along both banks and a zig-zag near the dam (Table 2, Figure 2). Also on April 7, 2014, a survey of the middle axis was conducted at dusk, and a second zig-zag transect was conducted at nighttime. A short transect along the middle axis was conducted before the release on the morning of April 8, 2014 to examine the daytime distribution of fish, and these daylight surveys were used in place of the dusk surveys for comparison with post-release transects conducted in these areas during daylight.

The boat launch at Yale Park and surrounding areas were surveyed while fish were being released, to directly assess the target strengths of smolts as registered by the hydroacoustics system. Numerous direct sightings of the released smolts swimming through water while being sampled by the side-looking transducer were linked to specific locations on the corresponding echograms (Figure 3). Post-release surveys were conducted along the same tracks as the pre-release surveys, along both banks, the center axis, and in a zig-zag pattern within the forebay and lower ¼ of the reservoir. We surveyed transects following the smolt release on April 8, 2014 during daylight beginning at 13:58 h and concluding at 17:06 h (Table 1). We surveyed an additional zig-zag transect in the lower ¼ of the reservoir from 20:31 h to 21:01 h to examine the density of fish near the dam at night.

A third survey was conducted on April 9, 2014, the day after the smolts were released. All regions were surveyed during daylight beginning at 07:25 h and concluding at 10:43 h (Table 1). An additional transect was conducted at dusk along the middle-axis in Region 4 (middle/upper reservoir) from 19:28 h to 19:47 h to qualitatively examine whether smolts had left the vicinity of the release site. Lastly, we surveyed the same zig-zag transect within the lower ¼ of the reservoir during the night from 20:45 h to 21:18 h to assess whether smolts shifted to lower regions of the reservoir or were aggregated near the dam.

All echograms were visually examined and regions of background noise and non-fish targets were excluded from the analysis. Standard echo counting techniques

(Beauchamp et al. 1997b; Beauchamp et al. 2009) with EchoView 5.4 software (Myriax Pty. Ltd.) were used to evaluate depth specific densities of fish targets. Targets measured by the side-looking transducer were used to estimate fish density within the upper 5-20 m of the water column over a range of 0-40 m from the survey vessel. Target strength frequency histograms were examined for groups of smolts sampled during the release at the boat launch to determine the range of target strengths corresponding to the smolts as seen by the side-looking transducer. Fish densities were calculated for targets with strengths within the range determined for the smolts (-55 to -43 dB) on the side-looker. Water depths were too shallow to effectively sample smolts during the release with the down-looking transducer. Therefore, we used the average fork length of test released Coho smolts in 2013 (data were unavailable for 2014) to determine the likely range of target strengths expected on the down-looking transducer based on the fish length-to-target strength relationship of Love (1971). The average fork length of test released Coho in 2013 was 127 mm (N = 663, ranged 84–178 mm, SD = 10.3 mm), so densities were calculated for targets with strengths equivalent to 75-175 mm total length fish (-42 to -49 dB). Echograms from both the side-looking and down-looking transducers were visually examined for aggregations of smolt-sized targets, especially within the vicinity of the forebay.

Without real-time direct fish sampling during these hydroacoustics surveys to ground truth observed fish targets, or telemetry data from individually tracked Coho Salmon smolts, the interpretation of these data were limited to only qualitative descriptions of how smolt-sized fish targets are distributed in the reservoir across time and space. Any noticeable increase in the density of smolt-sized targets within different regions or depths after their release would only be suggestive that Coho smolts are occupying or moving through those areas.

Results and Discussion

Stream habitat

Wild Coho juveniles

During 2013, we sampled a total of 20 reaches on Clear Creek for a total of 5.4 km and in 2014 we sampled 17 reaches totaling 4.7 km. Although the size of Coho captured across years was generally similar (Figure 4), Coho captured in 2013 (average = 88.5 mm; SD = 8.3) were nearly 10 mm larger than fish captured in 2014 (average = 79.6 mm; SD = 8.1).   The number of juvenile Coho captured by reach varied considerably within and across years and juvenile Coho were captured as high as 7.0 river km upstream from the confluence with the Muddy River (Figure 5). Within the sample reaches we PIT-tagged 357 individual Coho in 2013 and 883 individuals in 2014.

Through May 31, 2014, antenna recapture data suggest considerable differences in migration of wild Coho across years. In 2013, the majority of marked fish emigrated during September (Figure 6), but fish marked in 2014 demonstrated no obvious peak, with diffused emigration throughout much of the spring 2014. Across both years, the majority of fish emigrating in early fall were marked in the most downstream reaches (Figure 7). Coho emigrating during the period of October 1 through May displayed a wider range of location of origin (i.e., tagging). Similar to Pess et al. (2011) we found the largest numbers of fish emigrating during the early fall and late spring (Figure 8). In contrast to Pess et al. (2011), however, we observed no clear trends in emigration date when considering Coho size (not shown).

Overall, relatively few PIT-tagged wild Coho have reached the FSC. Initial estimates of Coho survival from mark-recapture analysis (2013 data only) suggest tributary survival is high (Φ= 0.62, 0.34 – 0.85), and likely driven by the high proportion of fish in 2013 that emigrated in the fall of 2013. The high summer and early fall survival is consistent with recent apparent survival estimates from Pess et al. (2011). Of the 357 wild Coho marked in 2013, 28 (7.8%) have reached the FSC; estimates from 2014 marked fish are not available at this time. Individual marking data suggests Coho reside in Swift Reservoir for nearly 4 months, with the median number of days between emigrating from Clear Creek and collected at the FSC as 121 days (range = 27 – 347 days).

Acclimation Chinook and Coho Salmon

As outlined in Task 6, the majority (60%) of acclimation Spring Chinook emigrate from Clear Creek within the first 7 days after release and 98% within 60 days. When considering all PIT-tagged Spring Chinook, there has been a bimodal distribution of the total time since release to the FSC with an initial peak at 75 days after release and a second peak at approximately 9 months after release (Figure 9). The median days between initial release and collection at the FSC were generally consistent across release locations for 2013 (Clear Creek = 67 days, Crab Creek = 57 days, Muddy River = 75 days; Figure 10), but we observed considerable differences in 2014 for Clear Creek (259 days). For Spring Chinook released in Clear Creek we found the median residence time in Swift reservoir, which we calculated as the time since emigrating from Clear Creek (i.e., PIT-tagged individuals past the antenna), as 69 days (range = 5 – 320; Figure 11). However, nearly 30% of the Spring Chinook resided in Swift Reservoir for more than 170 days.

Overall, the proportion of juvenile salmon collected at the FSC has been relatively low across species and years (Lewis River Fish Passage Program Annual Report 2014). Nearly 14% of the hatchery juvenile Coho released in Clear Creek in 2013 successfully emigrated to the FSC. However, only 1.4% (2013), 2% (2014), and 1.7% (2015; early

estimate) of the acclimation Spring Chinook (PIT-tagged) have successfully emigrated to the FSC.

Reservoir habitat

Targets identified in hydroacoustic surveys in Yale Lake were predominantly moving downstream after initial release. Smolt-sized targets were found to be using open water (mid-channel) with some skew toward the west bank, the same side as the release. The densities of "smolt-like" targets detected in the surface-oriented side-looking data and down-looking data were approximately equal.

Density of smolt-sized targets from side-looking hydroacoustics

When compared to the pre-release surveys in Yale Lake, densities of surface-oriented smolt-sized targets increased in the afternoon following the morning test release of Coho Salmon smolts on April 8, 2014. Specifically, the densities of smolt-sized targets increased in the lower half of the reservoir (Regions 1-3) along the west bank (Figure 12). Densities also increased by a factor of 9 along the middle axis in the region closest to the dam (Region 1), and in the zig-zig transect in the lower ¼ of the reservoir by a factor of 13 during daylight and by 3-fold at night. Densities on the day after the release increased relative to pre-release densities in Regions 2 and 3 along the west bank. Densities also increased relative to pre-release in the lower ¼ zig-zag transects during both daylight and night.

On the day following release, densities of surface-oriented smolt-sized targets were similar or slightly higher than the previous day along the west bank in Regions 3 and 2, but were generally lower in the middle axis and other regions of the west bank, with no notable pattern on the east bank (Figure 13). Densities estimated from the zig-zig transect in the lower ¼ of the reservoir declined by 68% during the daylight survey, and the night zig-zag transect remained at similar levels to the day of the release. Aggregations of smolt-sized targets were observed on hydroacoustic echograms on the day after the release near the boat launch in Region 4 during the daylight survey and the additional dusk survey, although the densities in this area did not increase. Aggregations were also observed near the dam during daylight on the day following the release, with the appearance of higher concentrations along the west shoreline in Region 1, directly in front of Saddle dam, and directly in front of the spillway (Figure 3). Similar patterns were observed in echograms from nighttime near the dam on the day after the release.

Densities of smolt-sized targets from the side-looking hydroacoustic surveys largely provide inference for the upper 20 m of the reservoir and suggest relatively rapid downstream movements through Yale Lake. Our results from Yale Lake are consistent with previous efforts in Swift reservoir, where acoustically tagged Coho Salmon smolts

were detected most commonly in the upper portions of the water column near Swift Dam (PacifiCorp/Cowlitz-PUD 2004).

Density of smolt-sized targets from down-looking hydroacoustics

Densities of deeper Coho Salmon smolt-sized targets detected by the down-looking transducer increased during daylight surveys in the afternoon on the day of the release, compared to the pre-release surveys, in different depths and regions. Densities increased from zero to ~0.5 targets/1,000 m^3 along the east bank in Regions 4 and 5 (Figure 13). We observed 24-fold increase in Region 2 at 50-60 m depths along the west bank. Densities also increased along the middle axis in the region closest to the dam (Region 1) by 210% at 20-30 m; 202% at 30-40 m; 251% at 40-50 m; and by a factor of 6 at 50-60 m. The lower ¼ zig-zag transect increased by 134% at 30-40 m in the daylight survey, and 115% in the night survey, however, densities in most other depths decreased in the zig-zag transects (Figure 14).

Densities on the day after the release increased relative to the pre-release in Regions 1 and 2 along the middle axis at depths below 30 m, at 10-40 m in Region 3 along the West Bank, and at 40-50 m in Region 2 along the east bank (Figure 13). Coho smolt densities also increased at 30-40 m depths in the lower ¼ zig-zag transect during daylight (Figure 14).

Densities on the day after the release changed relative to the afternoon and night of the release day in a number of regions as well. We observed an increase in densities in Region 3 along the west bank from zero to 1.3 targets/1,000 m^3 at 30-40 m, and Region 2 along the west bank increased by a factor of 4 at 30-40 m (Figure 13). We observed a 135% increase at 30-40 m in Region 1 along the middle axis, as well as a 145% increase at 40-50 m in this area. The lower ¼ zig-zag transects during daylight increased by 153% at 20-30 m, 127% at 30-40 m, and 184% at 50-60 m (Figure 14). In contrast, densities decreased by approximately 50% in most depths in the lower ¼ zig-zag transects at night. Visual observation of aggregations of smolt-sized targets detected in Region 4 near the boat launch and increases in densities in Regions 4 and 5 along the East Bank suggest that a portion of the smolts remained in the upper reservoir until at least the day following the release.

The results from monitoring Yale Lake indicate that Coho Salmon smolts have a relatively short migration time through the reservoir, a pattern similar to previous studies (PacifiCorp/Cowlitz-PUD 2001). The increase in densities of smolt-sized targets in lower regions on the day of the release suggests that smolts were capable of transiting the lower 1/2 of Yale Lake in less than a day. A portion of the test release smolts likely moved down reservoir along the west bank (the bank where they were released) immediately upon release, based on the increased densities of smolt-sized targets. It appears many smolts were able to reach the dam, and distributed to the middle axis in

Region 1 and the east bank in Region 2 within 12 hours of being released. Further increases in densities along the west bank and lower reservoir regions on the day following the release suggest that smolts continued to migrate down the reservoir. Smolts appeared to migrate to the lower ¼ of the reservoir where they were detected on the side-looker in both post-release surveys during night.

The findings of this study are consistent with the results of radio telemetry tracking of Coho Salmon smolts in Swift reservoir, which found that more than half of tagged fish released in the upper reservoir were detected at the dam within 4 days, and the greatest number of detections by time interval was for the period of 1-2 days after release (PacifiCorp/Cowlitz-PUD 2001). The majority of detections in mobile surveys were along the shore where they were released, generally near river inlets, in coves, and in the vicinity of woody debris. Detections on the antenna array at Swift dam revealed that virtually all smolts approached the intake structure in the upper 13.8 meters of the water column, typically from the north side at first and then resided on the south side. Results from telemetry of Chinook Salmon smolts revealed that they also approached the intake in the upper 13.8 m of the water column, with a mean travel time of 5.5 days from release at Eagle Cliff Park to detection at the project intake (PacifiCorp/Cowlitz-PUD 2002). These results indicate that the densities observed by the side-looking transducer may be more important for identifying smolt distribution than densities observed by the down-looking transducer.

We acknowledge several limitations to the hydroacoustic analyses. Several environmental conditions potentially biased our results or limited our ability to interpret results within the context of smolt movement through the reservoir. The pre-release survey along the middle axis in Regions 3-5 represented crepuscular fish densities, while the post-release surveys represent daylight densities, making direct comparison difficult. The propensity of kokanee to migrate vertically across the diel sequence likely added some variability into our results, particularly during dusk and night periods, which could have masked the signal of the relatively small release of 5,000 test smolts. Furthermore, noise from wind and rain, as well as the tops of standing timber masked the signal of fish targets for portions of the side-looking data. However, the consistency of such our results with other studies indicating use of relatively shallow habitats in reservoirs (Durkin et al. 1966; Dilley 1994) and relatively rapid movement through the reservoirs observed from Chinook smolts (PacifiCorp/Cowlitz-PUD 2002) supports our findings.

Summary of findings from stream and reservoir habitat

In merging our results from stream and reservoir studies we found conflicting patterns of behavior. In particular, information from PIT-tagged wild Coho smolts and acclimation Chinook, which covered both stream and reservoir habitat, suggests relatively long

periods of residence time in Swift Reservoir and relatively low collection efficiencies (i.e., proportion of marked fish that are collected at the FSC). Concomitantly, data from hydroacoustic surveys in Yale Lake with Coho Salmon (this study) and previous radiotelemetry studies in Swift Reservoir with Chinook (PacifiCorp/Cowlitz-PUD 2002) and other studies (e.g., Aitken et al. 1996) suggest reservoir travel times to be relatively rapid. Together, these apparently conflicting data suggest smolt migrations through the reservoirs may be rapid, but actually finding the collector may be difficult, thus leading to higher chance of residualization.   This pattern is supported by a recent study of hydroacoustic tagged fish in Swift Reservoir exhibited the shortest travel times to the FSC with even relatively small differences in distances away from the dam forebay (PacifiCorp/Cowlitz-PUD 2004). Continued data collection of residualization and travel times of PIT-tagged fish from release to the FSC in addition to behavioral data near the FSC will help further resolve such patterns.

References

Abbe, T. B., and D. R. Montgomery. 2003. Patterns and processes of wood debris accumulation in the Queets river basin, Washington. Geomorphology 51(1-3):81-107.

Achord, S., R. W. Zabel, and B. P. Sandford. 2007. Migration timing, growth, and estimated parr-to-smolt survival rates of wild snake river spring-summer Chinook salmon from the salmon river basin, Idaho, to the Lower Snake River. Transactions of the American Fisheries Society 136(1):142-154.

Aitken, J. K., C. K. Cook-Tabor, and R. C. Wunderlich. 1996. Travel time of coho salmon and steelhead smolts emigrating through Howard Hanson Reservoir, King County, Washington. U.S. Fish and Wildlife Service, Olympia, Washington.

Anderson, J. H., G. R. Pess, R. W. Carmichael, M. J. Ford, T. D. Cooney, C. M. Baldwin, and M. M. McClure. 2014. Planning Pacific Salmon and Steelhead Reintroductions Aimed at Long-Term Viability and Recovery. North American Journal of Fisheries Management 34(1):72-93.

Anlauf, K. J., D. W. Jensen, K. M. Burnett, E. A. Steel, K. Christiansen, J. C. Firman, B. E. Feist, and D. P. Larsen. 2011. Explaining spatial variability in stream habitats using both natural and management-influenced landscape predictors. Aquatic Conservation-Marine and Freshwater Ecosystems 21(7):704-714.

Beauchamp, D. A., C. Luecke, W. A. Wurtsbaugh, H. G. Gross, P. E. Budy, S. Spaulding, R. Dillenger, and C. P. Gubala. 1997. Hydroacoustic assessment of abundance and diel distribution of sockeye salmon and kokanee in the Sawtooth Valley Lakes, Idaho. North American Journal of Fisheries Management 17:253-267.

Beauchamp, D. A., D. Parrish, and P. Whaley. 2009. Salmonids/coldwater species in large standing waters. Pages 97-117 *in* S. A. Bonar, D. Willis, and W. A. Hubert, editors. Standard Sampling Methods for North American Freshwater Fishes. American Fisheries Society, Bethesda, Maryland.

Bilby, R. E., and J. W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and 2nd-growth forests in southwesetern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48(12):2499-2508.

Blair, G. R., L. C. Lastelle, and L. E. Mobrand. 2008. The ecosystem diagnosis and treatment model: a tool for assessing salmonid performance potential based on habitat conditions. American Fisheries Society Symposium 71:22.

Brakensiek, K. E., and D. G. Hankin. 2007. Estimating overwinter survival of juvenile Coho salmon in a northern California stream: Accounting for effects of passive integrated transponder tagging mortality and size-dependent survival. Transactions of the American Fisheries Society 136(5):1423-1437.

Buffington, J. M., and D. R. Montgomery. 1999. A procedure for classifying textural facies in gravel-bed rivers. Water Resources Research 35(6):1903-1914.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51(8):1389-1406.

Candy, J. R., and T. D. Beacham. 2000. Patterns of homing and straying in southern British Columbia coded-wire tagged chinook salmon (Oncorhynchus tshawytscha) populations. Fisheries Research 47(1):41-56.

Cederholm, C. J., R. E. Bilby, P. A. Bisson, and B. Bumstead. 1997. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. North American Journal of Fisheries Management 17:947-963.

CHaMP. 2013. Columbia Habitat Monitoring Program. Scientific protocl for salmonid habitat surveys with the Columbia Habitat Monitoring Program. Prepared by the Integrated Status and Effectiveness Monitoring Program, Wauconda, Washinton.

Dilley, S. J. 1994. Horizontal and vertical distribution of juvenile salmonids in Howard Hanson Reservoir. U.S. Fish and Wildlife Service, Olympia, Washington.

Dunham, J., G. Chandler, B. Rieman, and D. Martin. 2005. Measuring stream temperature with digital data loggers: a user's guide. U.S. Department of Agriculture, Forest Service, Gen. Tech. Rep. RMRS GTR-150WWW, Fort Collins, Colorado.

Durkin, J. T., P. D. L., and R. F. Raleigh. 1966. Distribution and movement of juvenile salmon in Brownlee Reservoir, 1962-1965. Bureau of Commercial Fisheries Biological Laboratory, Seattle, Washington.

Eggers, D. M. 1978. Limnetic feeding behavior of juvenile sockeye salmon in Lake Washington and predator avoidance. Limnology and Oceanography 23:1114-1125.

Fausch, K. D., and T. G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693.

Gilliam, J. W. 1994. Riparian wetlands and water quality. Journal of Environmental Quality 23(5):896-900.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. Bioscience 41(8):540-551.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, L. M. Decker, S. V. Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18(6):3-12.

Isaak, D. J., and D. L. Horan. 2011. An Evaluation of Underwater Epoxies to Permanently Install Temperature Sensors in Mountain Streams. North American Journal of Fisheries Management 31(1):134-137.

Johnson, S. L., J. D. Rodgers, M. F. Solazzi, and T. E. Nickelson. 2005. Effects of an increase in large wood on abundance and survival of juvenile salmonids (Oncorhynchus spp.) in an Oregon coastal stream. Canadian Journal of Fisheries and Aquatic Sciences 62(2):412-424.

Kershner, J. L., B. B. Roper, N. Bouwes, R. Henderson, and E. Archer. 2004. An analysis of stream habitat conditions in reference and managed watersheds on some federal lands within the Columbia River basin. North American Journal of Fisheries Management 24(4):1363-1375.

Kiffney, P. M., G. R. Pess, J. H. Anderson, P. Faulds, K. Burton, and S. C. Riley. 2009. Changes in fish communities following recolonization of the Cedar River, WA, USA by Pacific salmon after 103 years of local extirpation. River Research and Applications 25(4):438-452.

Kondolf, G. M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129(1):262-281.

Larsen, D. P., P. R. Kaufmann, T. M. Kincaid, and N. S. Urquhart. 2004. Detecting persistent change in the habitat of salmon-bearing streams in the Pacific Northwest. Canadian Journal of Fisheries and Aquatic Sciences 61(2):283-291.

Lestelle, L. C., L. E. Mobrand, J. A. Lichatowich, and T. S. Vogel. 1996. Applied ecosystem analysis-a primer. Bonneville Power Administration, Project Number 9404600, Portland, Oregon.

Marcot, B. G., C. S. Allen, S. Morey, D. Shively, and R. White. 2012. An Expert Panel Approach to Assessing Potential Effects of Bull Trout Reintroduction on Federally Listed Salmonids in the Clackamas River, Oregon. North American Journal of Fisheries Management 32(3):450-465.

McClain, M. E., E. W. Boyer, C. L. Dent, S. E. Gergel, N. B. Grimm, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, W. H. McDowell, and G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6(4):301-312.

McClure, M. M., S. M. Carlson, T. J. Beechie, G. R. Pess, J. C. Jorgensen, S. M. Sogard, S. E. Sultan, D. M. Holzer, J. Travis, B. L. Sanderson, M. E. Power, and R. W. Carmichael.

2008. Evolutionary consequences of habitat loss for Pacific anadromous salmonids. Evolutionary Applications 1(2):300-318.

McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of Evolutionary Significant Units. National Marine Fisheries Service, Seattle, Washington.

McHugh, P., and P. Budy. 2005. A comparison of visual and measurement-based techniques for quantifying cobble embeddedness and fine-sediment levels in salmonid-bearing streams. North American Journal of Fisheries Management 25(4):1208-1214.

McHugh, P., P. Budy, and H. Schaller. 2004. A model-based assessment of the potential response of Snake River spring-summer Chinook salmon to habitat improvements. Transactions of the American Fisheries Society 133(3):622-638.

Montgomery, D. R., J. M. Buffington, R. D. Smith, K. M. Schmidt, and G. Pess. 1995. Pool spacing in forest channels. Water Resources Research 31(4):1097-1105.

Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. Ecology and Management of Wood in World Rivers 37:21-47.

Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28:621-658.

Nickelson, T. E. 2000. Population assessment: Oregon Coast coho salmon ESU. Oregon Department of Fish and Wildlife, Portland, Oregon.

NMFS. 2007. Biological opinion and Magnuson-Stevens Fishery Conservation and Management Act Consultion; operation of PacifiCorp and Cowlitz PUD's Lewis River Hydroelectric Project (FERC Nos. 935, 2071, 2111, and 2213). National Oceanic and Atmospheric Administration: National Marine Fisheries Service, 2005/05891, Seattle, Washington.

ODEQ. 1995. Temperature: 1992-1994 water quality standars review. Final issue paper. Oregon Department of Environmental Quality.

PacifiCorp/Cowlitz-PUD. 2001. Migratory behavoir of radio-tagged juvenile Coho Salmon through Swift Reservoir, 2001. PacifiCorp, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2002. Migratory behavoir of radio-tagged juvenile Chinook Salmon through Swift Reservoir, 2002. PacifiCorp/Cowlitz PUD, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2004. Behavoir of salmonid smolts at Swift Dam using 3-dimensional tracking with acoustic tags. PacifiCorp/Cowlitz-PUD, AQU 17, Portland, Oregon.

Pess, G. R., P. M. Kiffney, M. C. Liermann, T. R. Bennett, J. H. Anderson, and T. P. Quinn. 2011. The Influences of Body Size, Habitat Quality, and Competition on the Movement and Survival of Juvenile Coho Salmon during the Early Stages of Stream Recolonization. Transactions of the American Fisheries Society 140(4):883-897.

Quinn, T. P. 1993. A review of homing and straying of wild and hatchery-produced salmon. Fisheries Research 18:29-44.

Serl, J., and C. Morrill. 2010. Summary report for the 1996 to 2009 seasonal operation of the Cowlitz Falls fish facility and related Cowlitz Falls anadromous reintroduction program activities. Washington Department of Fish and Wildlife.

Sheer, M. B., and E. A. Steel. 2006. Lost watersheds: Barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and Lower Columbia River basins. Transactions of the American Fisheries Society 135(6):1654-1669.

Tappel, P. D., and T. C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management 3:123-135.

Temple, G. M., and T. N. Pearsons. 2012. Risk management of non-target fish taxa in the Yakima River Watershed associated with hatchery salmon supplementation. Environmental Biology of Fishes 94(1):67-86.

Urquhart, N. S., S. G. Paulsen, and D. P. Larsen. 1998. Monitoring for policy-relevant regional trends over time. Ecological Applications 8(2):246-257.

USFWS. 2006. Biological Opinion for the Federal Energy Regulatory Commission relicensing of the Lewis River Hydroelectric Projects: Merwin (No. 935), Yale (No. 2071), Swift No. 1 (No. 2111), and Swift No. 2 (No. 2213). U.S. Fish and Wildlife Service, FWS Reference Number: 1-3-06-F-0177, Lacey, Washington.

Weigel, D. E., P. J. Connolly, K. D. Martens, and M. S. Powell. 2013. Colonization of Steelhead in a Natal Stream after Barrier Removal. Transactions of the American Fisheries Society 142(4):920-930.

Wolman, M. G. 1954. A method of sampling coarse river-bed material. Transactions American Geophysical Union 35:951-956.

Zimmerman, M. P. 1999. Food habits of smallmouth bass, walleyes, and northern pikeminnow in the Lower Columbia River Basin during outmigration of juvenile anadromous salmonids. Transactions of the American Fisheries Society 128:1036-1054.

Table 1. The number of PIT-tagged acclimation Coho Salmon and Spring Chinook Salmon released by location and year in the Lewis River upstream of Swift Dam.

Year	Species	Clear Creek	Crab Creek	Muddy River
2013	Chinook	1,750	750	1,750
	Coho			2,000
2014	Chinook	7,576[1]		
2015	Chinook	3,400	3,300	3,300

[1]Includes fish from Muddy River as acclimation ponds at *the* Muddy River site were unusable.

Table 2. Hydroacoustic transects conducted to evaluate Coho Salmon smolt movement and distribution in Yale Lake, including release status (i.e., pre-, during, or post-release), date, location, start and end times, and diel period.

Release status	Date	Transect	Start time	End time	Period
Pre-release	7-Apr	West bank	16:28	17:00	Daylight
		Zig-zag	17:00	17:35	Daylight
		East bank	17:40	18:40	Daylight
		Middle axis	18:47	19:50	Dusk
		Zig-zag	20:33	22:03	Nighttime
Pre-release	8-Apr	Middle Axis	9:21	9:40	Daylight
Release	8-Apr	Boat launch	9:50	10:10	Daylight
Post-release	8-Apr	Zig-zag	13:58	14:35	Daylight
		East bank	14:35	15:35	Daylight
		West bank	15:35	16:40	Daylight
		Middle axis	16:41	17:06	Daylight
		Zig-zag	20:31	21:01	Nighttime
Post-release	9-Apr	West bank	7:25	8:18	Daylight
		Zig-zag	8:18	8:36	Daylight
		East bank	8:37	9:39	Daylight
		Middle axis	9:40	10:43	Daylight
		Middle axis	19:28	19:47	Dusk
		Zig-zag	20:45	21:18	Nighttime

Figure 1. Sample reaches (light grey circles) used in the systematic sampling design for PIT-tagging Coho Salmon in Clear Creek with location of the PIT-tag antenna (grey box). Inset shows the location of the sampling area in relation to Swift Reservoir.
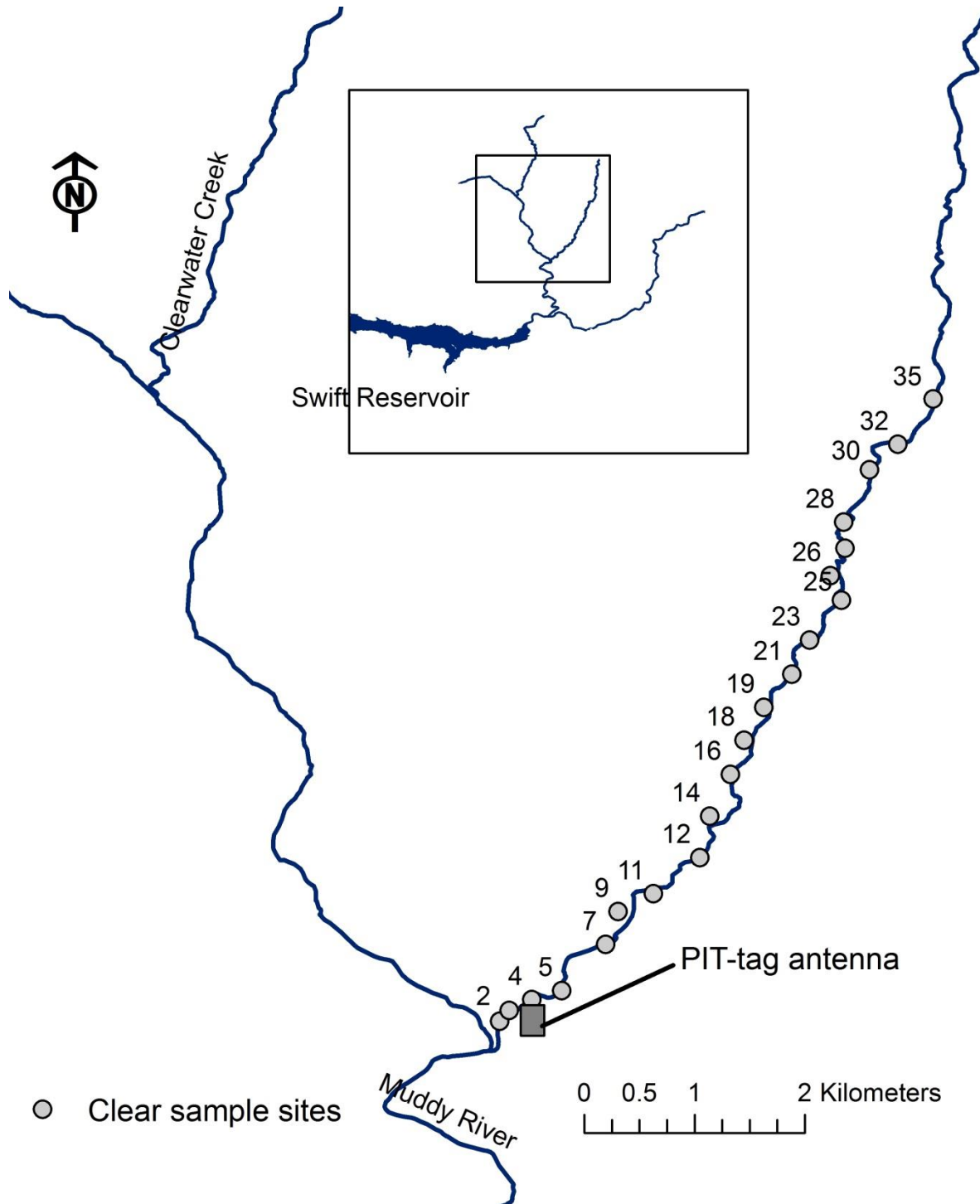
Figure 2. Yale Lake, with labeled regions that were used to evaluate densities of Coho Salmon smolt-sized targets in hydroacoustic surveys (black lines) and the release location of smolts (star; Yale Park). Region 1 contained all parts of the reservoir below 45° 59'N. Hydroacoustics transects were replicated for the pre-release, release day, and day after release surveys.
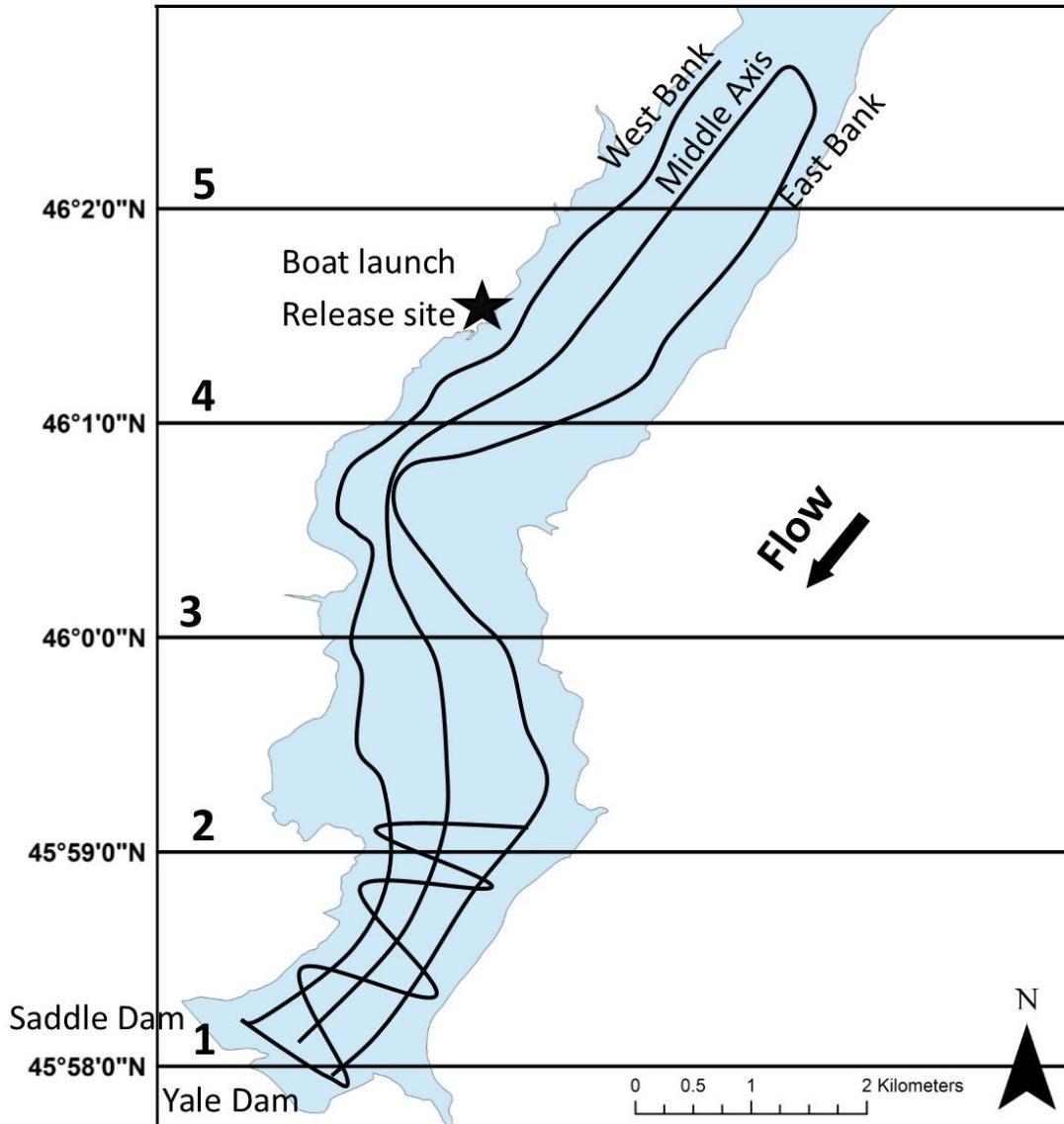
Figure 3. Echograms from the side-looking transducer showing smolt-sized targets in Yale Lake. Top panel shows the boat launch while fish were being released, and bottom panel shows an aggregation of smolt-sized targets seen along the west side of Yale Dam in a zig-zag transect.
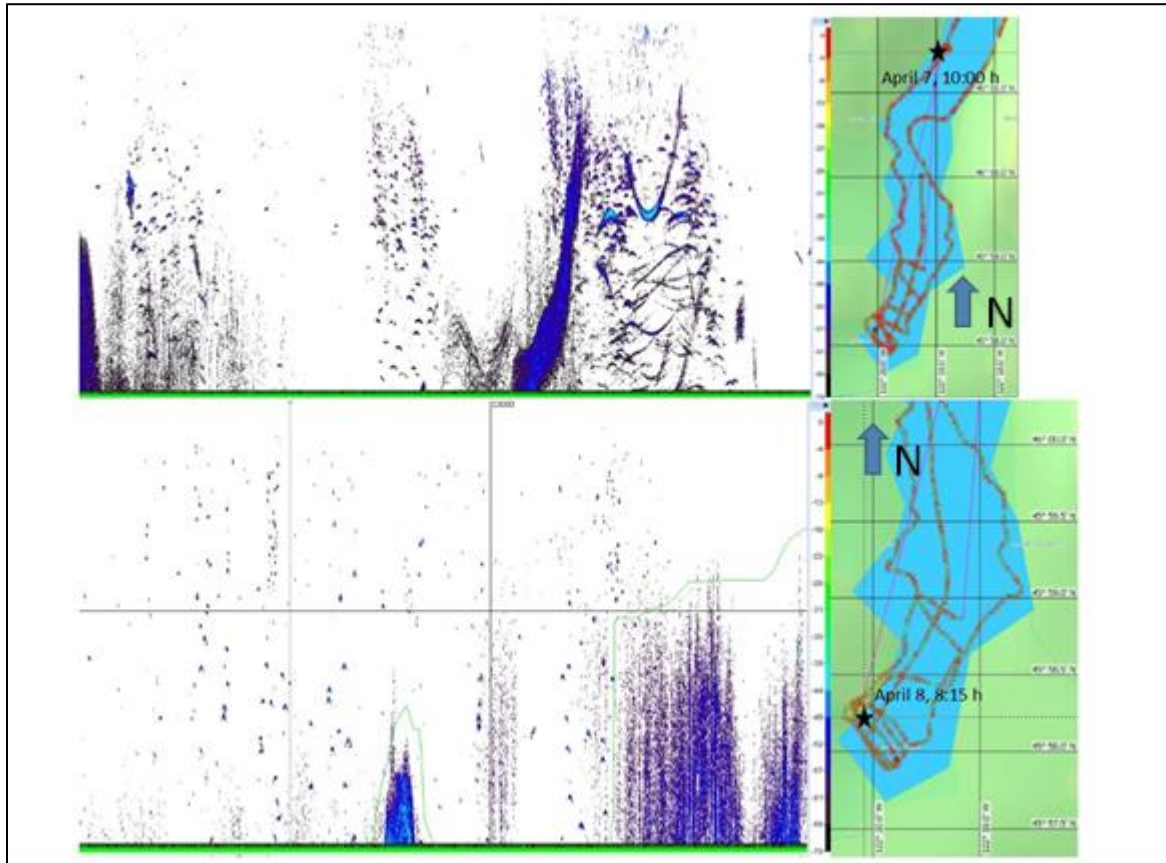
Figure 4. Length frequency of Coho Salmon captured in Clear Creek during September surveys in 2013 (a) and 2014 (b).
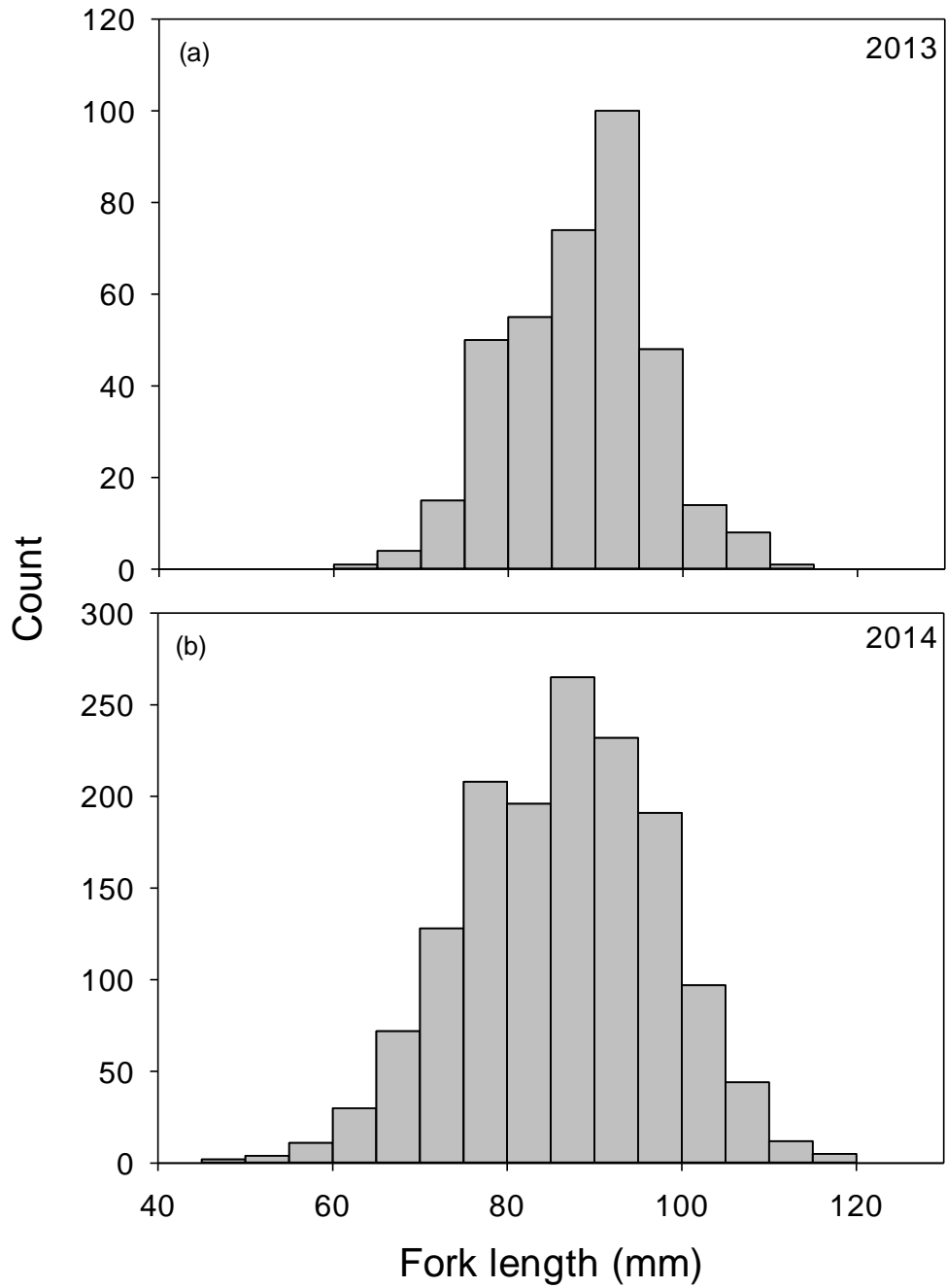
Figure 5. The number of PIT-tagged juvenile Coho by reach in 2013 (black) and 2014 (grey). Refer to Figure 1 for location of sample sites but sites progress from downstream to upstream (left to right).
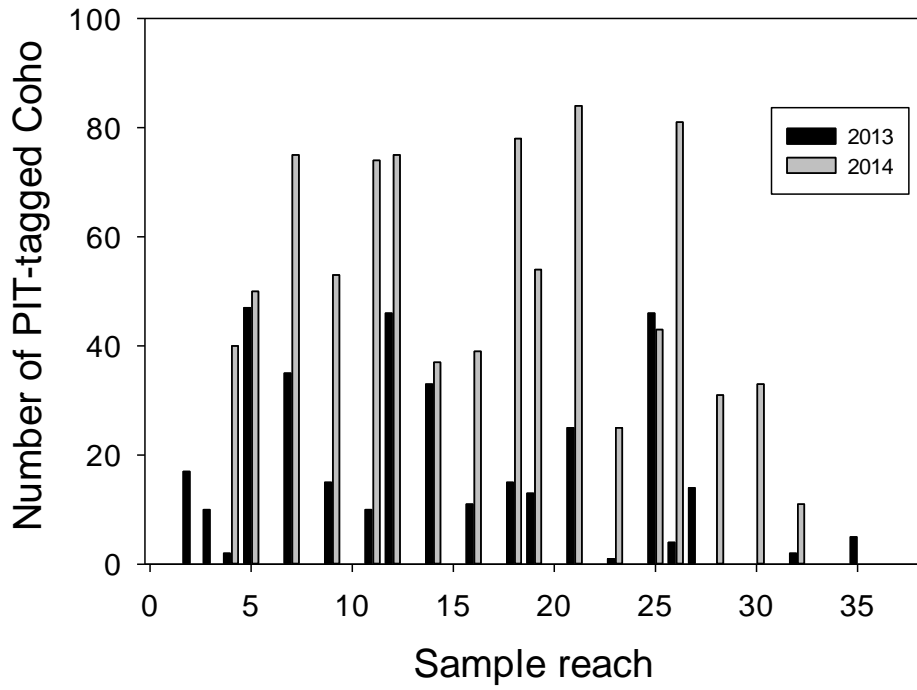
Figure 6. Emigration timing of PIT-tagged wild Coho Salmon downstream past the fixed antennas on Clear Creek (a) and the average daily hydrologic stage (black line) and stream temperature at the Clear Creek antennas in 2013 and 2014 (b). Note the dashed black line is estimated from correlations with the Muddy River USGS gaging station as the pressure transducer in Clear Creek was not initially operational.
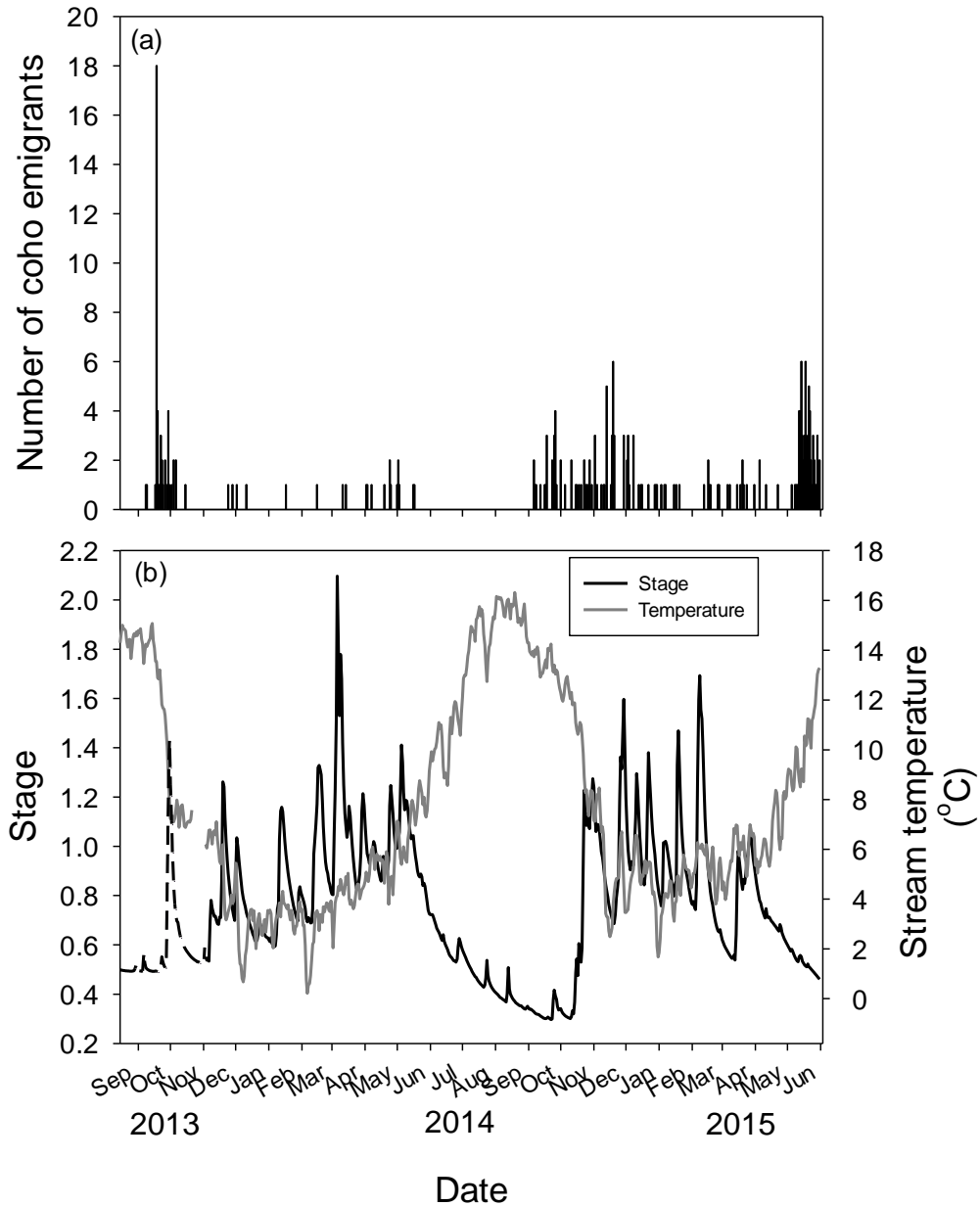
Figure 7. Number of days at large (i.e., time between PIT-tagging and emigration) for fish emigrating past the fixed PIT-tag antenna for fish marked in different sampling reaches. Note: increasing reach number corresponds to distance from river mouth (see Figure 1).
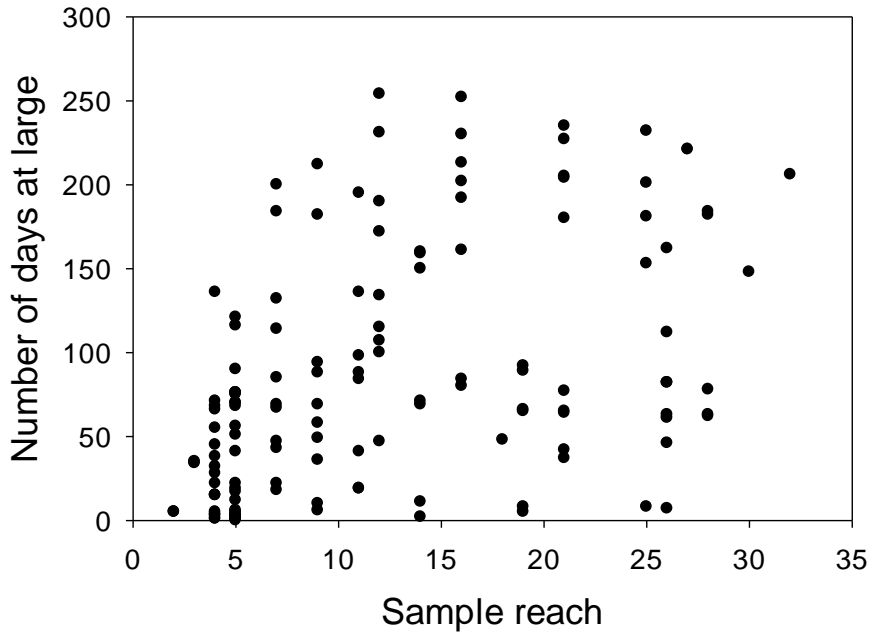

Figure 8. The number of days between PIT-tagging and emigration past the PIT-tag antenna on Clear Creek from 2013 and 2014. Note: PIT-tagging typically occurred during late August and early September in both years.
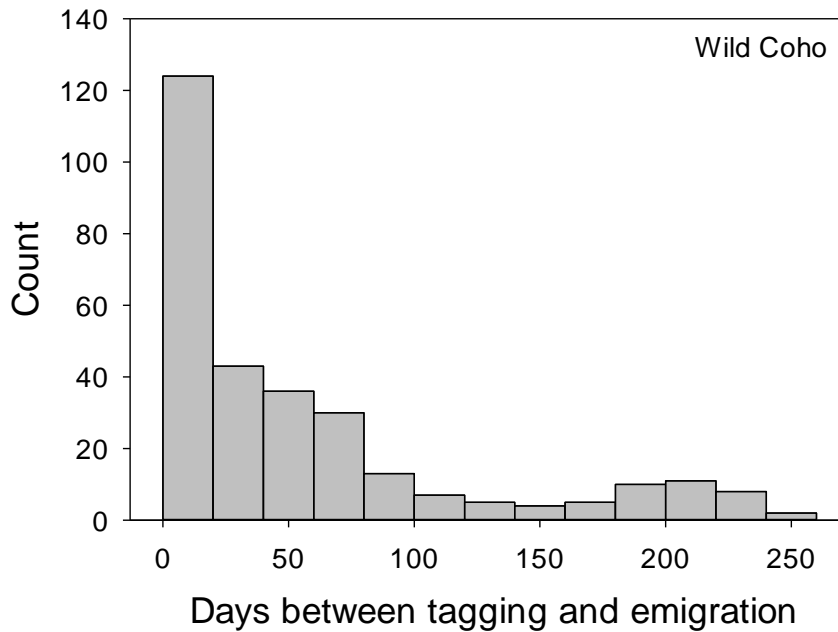
Figure 9. Total travel time for acclimation Spring Chinook (PIT-tagged) from release in tributaries (Crab Creek, Muddy River, Clear Creek) to the Floating Surface Collector in Swift Reservoir in 2013 and 2014.
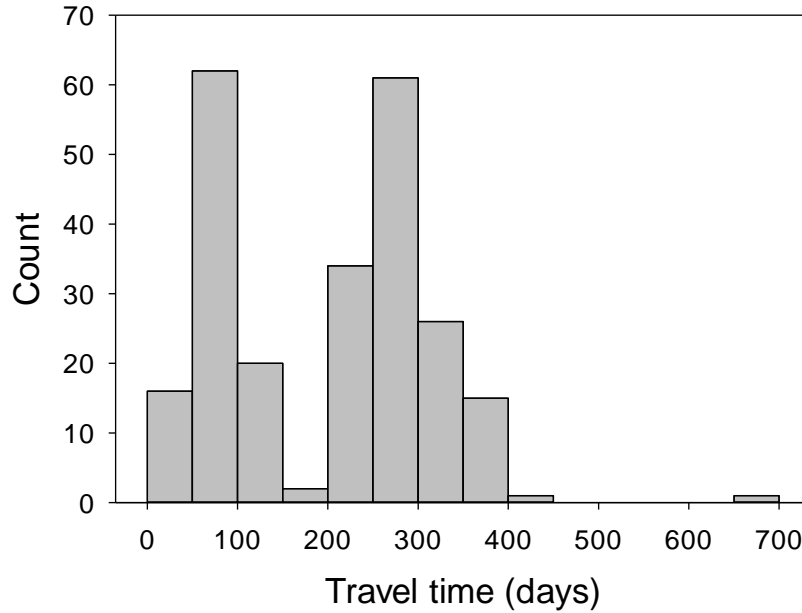

Figure 10. The total number of days between release and collection at the Floating Surface Collector in Swift Reservoir for acclimation Spring Chinook (PIT-tagged only) across different release locations and years. Note: multiple years of releases were only available for Clear Creek and Crab Creek, and the 2015 data are incomplete.
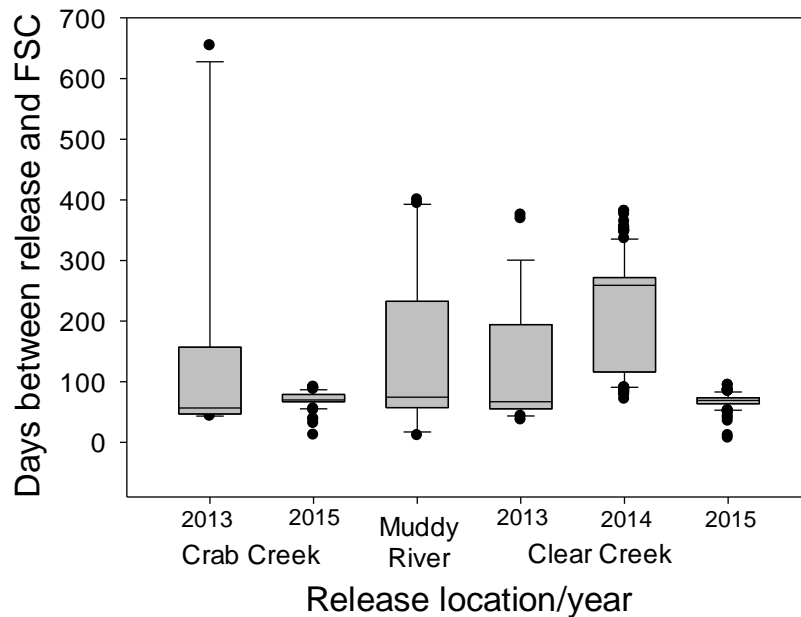
Figure 11. A frequency distribution of the number of days PIT-tagged Spring Chinook (acclimation) spent between emigration past the fixed PIT-tag antenna on Clear Creek (likely in Swift Reservoir) and collection at the Floating Surface Collector in Swift Reservoir.
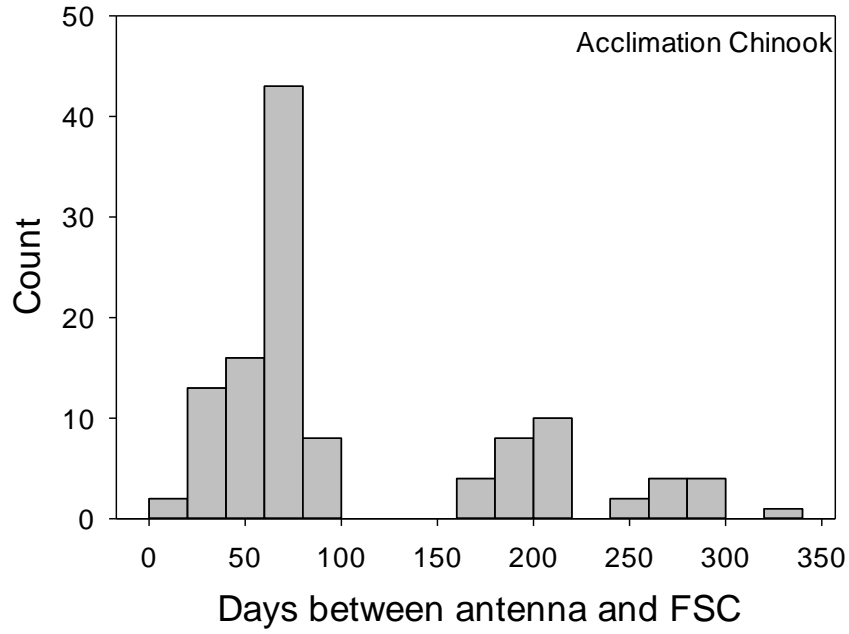
Figure 12. Coho Salmon smolt sized targets per 1,000 m^3 observed by the side-looking transducer in Yale Lake pre-release, the day of release, and post-release of 5,000 test smolts. Panels are arranged in columns from west (left), mid-axis (center) and east (right) transects (labeled on top), and rows based on upper to lower regions in the reservoir (region 1 closest to the dam). The lower ¼ zig-zag panels represent densities estimated from a series of zig-zag transects within the lower ¼ of the reservoir. Bars with an asterisk represent pre-release surveys conducted during dusk, while all other surveys were conducted under daylight unless otherwise noted.
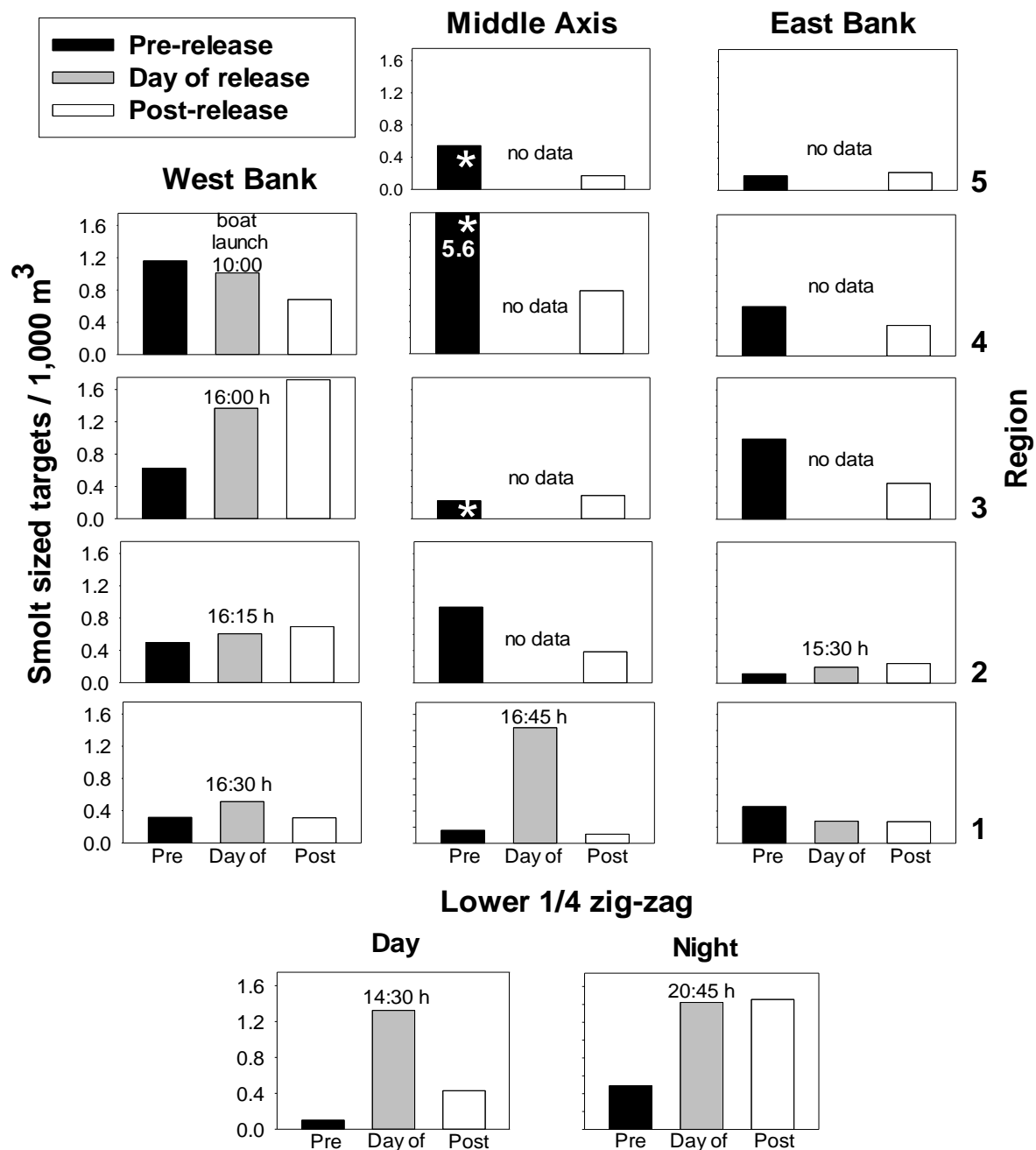
Figure 13. Smolt-sized targets per 1,000 m^3 observed by the down-looking transducer in 10 m depth bins in Yale Lake pre-, the day of, and post-release of 5,000 test smolts. Panels are arranged in columns based on transects (labeled on top), and rows based on regions (labeled at lower right of row). Lower numbered regions are lower in the reservoir (i.e. closer to the dam). Panels with an asterisk represent pre-release surveys conducted during dusk, while all other surveys were conducted under daylight.

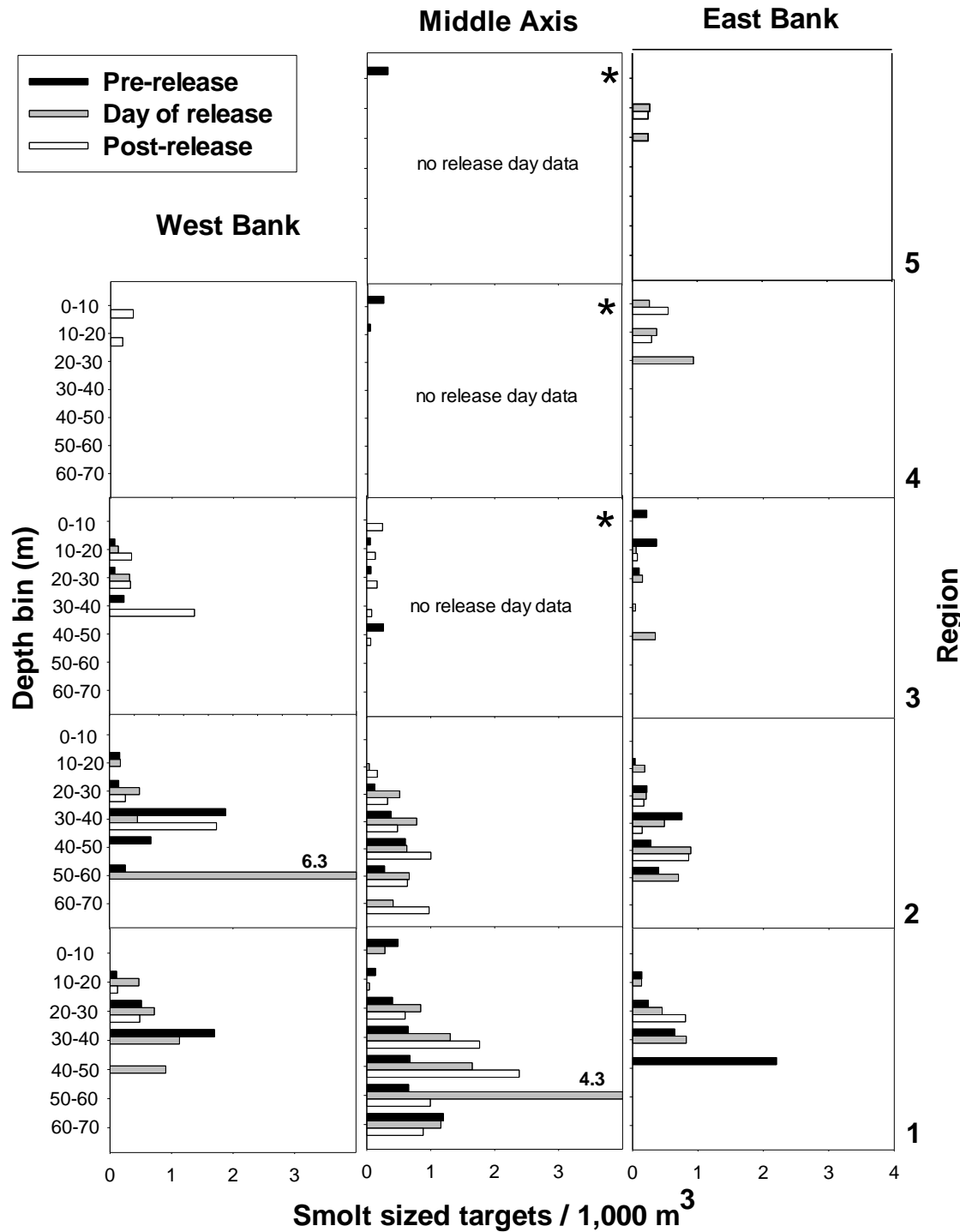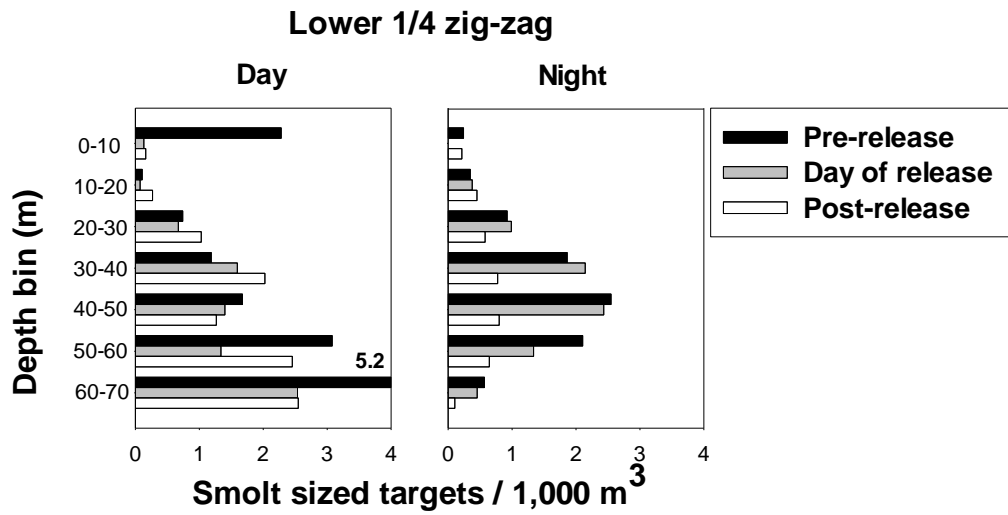Figure 14. Coho Salmon smolt-sized targets per 1,000 m^3 observed by the down-looking transducer in 10 m depth bins during zig-zag transects conducted in the lower ¼ of Yale Lake during daylight and night surveys (no crepuscular survey conducted). Surveys were conducted pre-, during, and post-release of 5,000 Coho smolts at Yale Park.

Task 5: Evaluate Lake Merwin predator impacts

Northern Pikeminnow (*Ptychocheilus oregonensis)* were identified as an abundant predator of juvenile salmon in Lake Merwin in the 1950s and 1960s, during an experiment to determine if the reservoir could be used to rear juvenile hatchery Coho Salmon (Hamilton et al. 1970). The abundance of predatory sized Northern Pikeminnow (≥200 mm) was estimated around 350,000 in 1961; however, the population has not since been assessed. Additionally, the Washington Department of Fish and Wildlife began stocking approximately 1,000 Tiger Muskie annually in 1995 to limit the population of Northern Pikeminnow, but the efficacy of this program has not been formally evaluated. The contemporary abundance, diet, growth, and temporal-spatial distribution of Northern Pikeminnow and kokanee are being evaluated to gauge how Northern Pikeminnow might affect populations of reintroduced anadromous salmonids. To achieve this objective, we characterized the temporal-spatial dimensions of the thermal environment, food supply, and the distribution, size, age, and diet of key predators and prey, and mapped the overall trophic structure of the food web through stable isotope analysis. These data will inform bioenergetics simulations constructed to estimate the seasonal and size-specific consumption rates and predation impact of Northern Pikeminnow on salmonids and alternative prey fish species (Beauchamp et al. 1997a).

Site description

Lake Merwin is approximately 23.3 km long (PacifiCorp 2004) with a maximum depth of 73 m and a mean depth of 31 m (PacifiCorp; unpublished data). It is oligotrophic, with a mean secchi depth of roughly 5 m and total phosphorus of <0.02 mg/l (PacifiCorp, and Cowlitz 2004c) (Table 1). Surveys suggest that there is a total of 6 km of anadromous salmonid habitat available in tributaries to Lake Merwin (Al-Chokhachy et al. 2014).

Thermal stratification in Lake Merwin began in May and peaked in August in 2014; rapid destratification occurred during October and the water column became isothermal in November (Figure 1). The epilimnion encompassed the upper 8 m of the water column, the metalimnion ranged from roughly 8-18 m, and the hypolimnion was approximately 18 m and deeper. Epilimnetic temperatures began at 14-16°C in May and peaked at 22-24°C in August and July. The hypolimnion warmed from 5-6°C in April to 15-18°C in October.

The fish assemblage in Lake Merwin consists of a large population of Northern Pikeminnow, and Largescale Suckers, and moderate to small populations of Kokanee, Cutthroat Trout, Rainbow Trout, Sculpin, Pumpkinseed Sunfish (*Lepomis gibbosus)*, Coho Salmon, Chinook Salmon, Bull Trout, Tiger Muskie, and Largemouth Bass (*Micropterus salmoides*) (Table 2). The Kokanee population is supplemented with

approximately 45,000 hatchery fingerlings in October and 48,000 yearlings in spring (PacifiCorp and Cowlitz-PUD 2014). Natural production of Kokanee from tributaries of Lake Merwin is likely minimal. However, recruitment from Yale Lake (directly upstream of Lake Merwin) is probably the primary source of recruitment of naturally-produced Kokanee (PacifiCorp and Cowlitz-PUD 2004a). Large (300-600 mm) hatchery rainbow trout are stocked periodically as available to provide fishing opportunities, and Coho Salmon and Chinook Salmon smolts are also present through spill from Yale Lake.

Methods

Thermal experience and depth distribution

Temperature profiles were recorded monthly from April to November at 1 meter intervals from the surface to 30 m and every 2 m thereafter down to 60 m at two stations located at 1/3 and 2/3 of the length of the reservoir (Figure 2), and measurements were averaged between stations. Temperature profiles recorded by PacifiCorp in November 1999, January 2000, and March 2000 in the lower reservoir were used to fill data gaps and complete the characterization of the seasonal thermal regime for periods outside of our monthly sampling (April to November). Catch rates in depth-stratified gill nets were used to infer seasonal depth distributions of fish, which will be converted to thermal experience based on the vertical temperature profiles for bioenergetics modeling.

Fish sampling

Fish were systematically sampled in June, late July, and November of 2013 to obtain biological samples, seasonal fish distribution data, and population size structure. Paired small mesh (2.5, 3.2, 3.8, 5.1, 6.4, and 7.6 cm stretched mesh) and large mesh (8.9, 10.2, 11.4, 12.7, and 15.2 cm stretched mesh) sinking gill nets were fished perpendicular to shore for approximately 24 hours. Nets were set at a minimum of three sites around the reservoir per season, and were fished in three depth strata: 1-15 m; 16-30 m; and >30 m deep to approximate the depths above, within, and below the thermocline during peak summer stratification. In addition to systematic gillnetting in 2013, samples were gathered opportunistically by gill nets targeting Northern Pikeminnow as part of the mark-recapture study and other approaches including the use of Merwin traps (Hamilton et al. 1970; Lynch 1993), angling, dip net, minnow trap, angler surveys, and snorkeler hand collection in 2013 and 2014.

We identified and counted fish by species and recorded a subset of fork lengths to the nearest mm and body mass to the nearest g. Living fish were either released or euthanized according to UW-IACUC protocol #3286-21. Stomachs, scales from the preferred region, fin tissue, and some whole bodies were collected, placed on ice in the field and later frozen for preservation and analysis.

The length frequency of Northern Pikeminnow captured in gill nets was adjusted for effort by mesh size, encounter probability (Spangler and Collins 1992), and gill net size-selectivity (Hansen et al. 1997). Only catches from mesh sizes 5.1 cm (2-inch stretch mesh) and above were used for calculation of the contemporary size structure, as they made up the majority of the effort and the catch of fish ≥200 mm. Catches from Merwin traps were also used for calculating size structure, and were adjusted for encounter probability only, as they are large trap nets that allow for equal capture of fish across a wide range of sizes. Separate length frequencies were created for fish identified as male and female. Kolmogorov-Smirnov tests were used to compare the adjusted and unadjusted length distributions for each gear type, and to compare between gear types and sexes. The weighted averages of the length distributions from gill nets and Merwin traps were also compared between the contemporary study and the study conducted in 1958-1963 (Hamilton et al. 1970). Data from both studies were adjusted based on the selectivity of gill nets observed in this study (mesh sizes and fork lengths for individual fish were not available for the older study), and encounter rate. Only catches from similar mesh sizes (3.75-10 cm) and areas (main reservoir and Speelyai Bay) were included from both studies.

Northern Pikeminnow mark-recapture

Northern Pikeminnow ≥200 mm were tagged with 2.54-cm numbered Floy© tags. In 2013, we captured fish for tagging using short-term gill nets (5.1, 6.4, and 7.6 cm stretched mesh) set nearshore for 30 min to 2 h. Catch rates were greatest at dusk on the north side of the reservoir around Indian George Creek and Speelyai Bay (Figure 2), so we preferentially netted in these areas at dusk. Our initial goal was to tag and release 2,000-5,000 marked fish in one season; however, difficulty in capturing sufficient numbers necessitated an extended tagging period over two years: June-August 2013 and May-July 2014. In 2014, we exclusively used Merwin traps to capture fish for tagging (n = 4 locations; Figure 3. Northern Pikeminnow were anesthetized in MS-222 buffered with baking soda, and tagged.

A total of 98 tagged Northern Pikeminnow were held in net pens for 48 hours to evaluate short term tagging mortality and tag retention. No tag loss and approximately 10% mortality was observed in the holding trials. However, lacerations and evisceration observed on the carcasses of mortalities, and the presence of otters and piscivorous birds in the area suggested that these fish may have been predated upon rather than injured in the capture and tagging process. Therefore, short-term tagging mortality was assumed negligible.

Northern Pikeminnow were captured and examined for tags in August and September 2014 with small mesh (5.1, 6.4, 7.6 cm stretched mesh) and large mesh (8.9, 10.2, 11.4, 12.7, and 15.2 cm stretched mesh) gill nets set methodically around the perimeter

of the reservoir and soaked for approximately 24 hours. We assumed that fish captured and tagged during their spawning migration in June had redistributed throughout the reservoir by August. All captured individuals were measured for fork length and examined for tags, and a subset were weighed. Another subset of stomach samples (including some bycatch) were frozen for analysis. Additional individuals were captured by a separate investigator from Meridian Environmental Inc. using an electrofishing boat in the inlet to the reservoir during August 2014.

We estimated the abundance of Northern Pikeminnow using the Chapman estimator (Seber 1982), with confidence intervals based on the table by Chapman (1948; reproduced in Seber 1982) for studies with fewer than 50 recaptures. This estimator does not account for recruitment or death during the course of the study; however, mortality certainly occurred given the long period of time between tagging and recapture. Therefore, an annual mortality rate was assessed for individuals marked in 2013 with a catch curve (Miranda and Bettoli 2007). We estimated mortality based on catch-at-age from an age-length key, assuming a constant recruitment rate for all year classes involved (Isely and Grabowski 2007). Population modeling and abundance estimation are ongoing.

Stable isotope analysis

We used stable isotope analyses for $\delta^{13}C$ and $\delta^{15}N$ to describe the trophic positions and predominant energy pathways associated with different size classes of Northern Pikeminnow and other fish and invertebrates from a range of feeding guilds and trophic levels in the reservoir food web (Vander Zanden and Rasmussen 1999; McIntyre et al. 2006). Zooplankton were collected in front of the boat exclusion barrier near Merwin Dam in May 2014 using a Clarke-Bumpus sampler (with 154 µ mesh conical net). Aquatic Juga snails (*Juga hemphilli*) were collected by hand near the mouth of Buncombe Hollow Creek in June 2013. Filter feeding Asian clams (*Corbicula fluminea*) were collected by hand or came up on the anchors of gill nets in various locations around the reservoir June-July 2013. Lastly, adult insects were collected by towing a neuston net near Speelyai Bay in September 2014. All samples were frozen for storage, then dried at 60 °C for 48 h, homogenized, and weighed in aluminum capsules for analysis. Zooplankton were separated by genus, and muscle tissue was removed from the shells of snails and clams, and the exoskeletons of crayfish. For fish samples, fin tissue was clipped from three randomly selected individuals within each 50-mm size class of Northern Pikeminnow and other fish species. Isotope samples were analyzed using a Costech Elemental Analyzer, Conflo III, MAT253 for continuous flow based measurement of solid organic material. The reference material was Vienna Pee Dee belemnite for carbon and atmospheric N_2 for nitrogen.

Diet analysis

We analyzed the diets of Northern Pikeminnow captured in different seasons as inputs for a bioenergetics model. Seasons were defined as: spring (April-June; months leading up to thermal stratification); summer (July-September; thermally stratified period) and fall (October-December; breakdown of thermal stratification). Stomach contents were identified under a dissecting microscope to species for prey fish and to order for invertebrates. The blotted wet weight of each prey type was recorded to the nearest 0.001 g. Rare invertebrates such as arachnids and ostracods were pooled with insects for simplicity. Lengths of prey fish were measured or reconstructed from diagnostic bones. We calculated diet proportions by weight (Chipps and Garvey 2007) for two size classes of Northern Pikeminnow (200-299 mm and 300-599 mm), the breakpoint reflecting a transition to piscivory based on $\delta^{15}N$ signatures. Sample sizes were 9 to 29 non-empty stomachs per size class each season based on availability of samples and variability of diet compositions.

Age and growth

Age and growth of Northern Pikeminnow were determined from scales and used to measure annual growth increments for different age-classes in the bioenergetics model. Opercles were also analyzed for a subset of individuals to corroborate the scale-based ages. Scales were measured along an axis 20° off the scale's anterior-posterior axis from the center of the focus out to the edge of each annulus and additionally to the margin of the scale using a dissecting microscope with a digital camera and Image Pro$^{©}$ analysis software. Fork length-at-age was back-calculated using the Fraser-Lee Method and an empirically derived estimate of the "a" parameter based on the scale radius to fork length relationship (Isely and Grabowski 2007). Growth in fork length was characterized by a von Bertalanffy growth function fit to the raw back-calculated length-at-age data. A likelihood ratio test implemented by the "vblrt" function in the *fishmethods* package in R (Nelson 2014) indicated that a von Bertalanffy growth model specific to each sex described the length-at-age data significantly better than a single model for both sexes ($X^2 = 12.88$, DF = 3, *P* = 0.005). Therefore, we fit a von Bertalanffy growth function separately for male and female Northern Pikeminnow. Fork length-at-age was converted to weight-at-age using length-weight relationships developed for Northern Pikeminnow in Lake Merwin. Separate relationships were developed for individuals <300 mm and ≥300 mm due to the inability of a single power function to fit length and weight measurements over the entire range of fork lengths observed. The fork length (FL, mm) to weight (W, g) relationship for <300 mm Northern Pikeminnow was (n = 2,941; range 34-299 mm; r^2 = 0.97; *P* = 0.00):

$$W = 0.000012 * FL^{2.979}.$$

The relationship for ≥300 mm individuals was (n = 354; range 300-574 mm; r^2 = 0.97; *P* = 0.00):

$$W = 0.0000015 * FL^{3.348}$$

Results and discussion

Distribution and abundance of Northern Pikeminnow

Northern Pikeminnow depth distribution varied by season, with a shallower distribution (0 to 15 m) during stratified conditions in June-September and a deeper (>15 m) distribution during isothermal conditions in November. The catch of 200-299 mm individuals per gill net set in the upper 15 m was approximately twice that of sets in 16-30 m depths during late spring (June) and summer (July –September) (Figure 4). No individuals ≥300 mm were caught in sets below 15 m during spring, and the catch of these larger predators was an order of magnitude higher in the shallow sets than the deeper sets during summer. In fall, the greatest catch of both size classes was greatest in 16-30 m depths. Catch rates of fish ≥300 mm were 5 times greater in 16-30 m than 0-15 m.

The length frequency distribution of Northern Pikeminnow was skewed toward smaller (200-299 mm) individuals, and exhibited a decrease in modal fork length between 1958-1963 and 2013-2014 (Figure 5). There was no significant difference between the adjusted and unadjusted length distributions from gill net samples in 2013-2014 (D = 0.313, *P* = 0.43), or the adjusted and unadjusted length distributions from Merwin trap samples (D = 0.25, *P* = 0.70; Figure 6). There was, however, a significant difference between the adjusted size distributions from these two gear types (D = 0.0687, P = 0.001). The difference between the present and historic length frequencies was not statistically significant (D = 0.313, *P* = 0.415; Figure 5). However, the difference between the adjusted length frequencies was significantly different between males and females in the present study (D = 0.75, *P* = 0.00).

A total of 1,112 Northern Pikeminnow ≥200 mm were tagged in 2013, and an additional 956 fish were tagged in June 2014. Merwin traps used in 2014 (n = 4; Figure 3) were most effective during the peak of the spawning migration in June. Specifically, the trap set at Cresap Creek captured approximately 95% of all fish tagged in 2014. Despite high capture numbers at this location, the extensive periods between marking and recapture events as well as the considerable movement patterns exhibited by Northern Pikeminnow (i.e., population mixing; Isaak and Bjornn 1996) suggest such patterns are unlikely to result in significant bias in our estimates.

In August and September 2014, 2,285 fish were examined for marks in the main reservoir, 12 of which were tagged. Out of these 12 recaptures, 4 were tagged in 2013 and 8 were tagged in 2014. Additionally, 185 Northern Pikeminnow ≥200 mm were captured and examined for tags in the inlet to the reservoir from August through November 2014, one of which was tagged in 2014.

All of the recaptured fish in 2014 that were marked in 2013 were captured along the north bank in the vicinity of Indian George Creek and Speelyai Bay, where all but one fish was tagged. One fish was tagged near Canyon Creek in the upper reservoir in 2013 but was recaptured in the lower reservoir near Indian George Creek in 2014. All of the recaptured fish that were marked in 2014 were captured in the upper quarter of the reservoir, near areas where they were tagged. While the trend of fish recaptured in proximity to the tagging site suggests potential site fidelity, Northern Pikeminnow movements can be highly variable and influenced by ambient hydrologic conditions (Isaak and Bjornn 1996; Tabor et al. 2010). We acknowledge a bias in the size of recaptured fish toward ≥300 mm individuals, which was likely the result of greater natural mortality for 200-299 mm fish and different capture probabilities for the two size classes between marking and recapture.

The Chapman estimator produced a preliminary population estimate of 314,000 (95% CI = 175,000-583,000) Northern Pikeminnow ≥200 mm, after applying a 27% annual mortality rate to fish tagged in 2013 based on the catch curve analysis. In contrast, the preliminary population abundance of Northern Pikeminnow ≥300 mm based exclusively on the number of fish marked and examined of this size (113 fish tagged, 361 examined for marks, and 6 recaptures), is only 5,200 (95% CI = 2,200-13,900). This estimate is markedly less than the abundance of ≥300 mm individuals that would be expected based on the estimate for the entire population and the length frequency distribution. This discrepancy in preliminary population estimates between the two methods of calculation is due to the difference in recapture probabilities for 200-299 mm and ≥300 mm Northern Pikeminnow. Further investigation into potential biases from size selectivity of gear used each period, size selective mortality, and spatial variability in sampling effort and catch rates are still being investigated and will be incorporated in future abundance estimates. Recaptures from sampling periods other than summer 2014 may be incorporated in future abundance estimates as well.

Stable isotope analysis

Primary consumers exhibited disparate $\delta^{13}C$ values, consistent with expected patterns of isotopic fractionation for benthic versus pelagic feeding in lentic environments. Zooplankton and filter-feeding clams rely on pelagic or pelagically derived carbon sources, and share similar $\delta^{13}C$ values with their presumptive consumers: salmonids, small Cottids and small Northern Pikeminnow (Figure 7; Table 3). Benthic herbivores (snails) and omnivores (crayfish) show much higher $\delta^{13}C$ than zooplankton and clams. Medium and large Cottids, small Largescale Suckers, Tiger Muskie, Largemouth Bass, and large Northern Pikeminnow relied more heavily on benthic energy pathways than pelagic, although large Northern Pikeminnow appear to also utilize a relatively small proportion of pelagic prey.

Northern Pikeminnow become increasingly piscivorous and reliant on benthically derived energy with increasing size (Figure 8). Small Northern Pikeminnow <200 mm exhibited lower average $\delta^{15}N$ and $\delta^{13}C$ signatures than the larger size classes, suggesting a diet of zooplankton, other invertebrates, and plant material. In addition to zooplankton and other invertebrates, medium sized Northern Pikeminnow (200-299 mm) likely consumed some benthic fishes. Large Northern Pikeminnow ≥300 mm appear to consume a combination of pelagic and benthic fishes, including cannibalizing smaller Northern Pikeminnow, and crayfish. Size-specific feeding patterns of Northern Pikeminnow informed by stable isotopes were largely supported by corresponding diet analyses.

Diet analysis

Northern Pikeminnow exhibited seasonal and ontogenetic shifts in diet composition with an increasing reliance on prey fish at larger sizes, in agreement with the stable isotope patterns. The spring and summer diets of 200-299 mm Northern Pikeminnow contained 73-83% zooplankton, and smaller proportions of sculpin, crayfish, and plant material (Figure 9). The principal prey of the 200-299 mm size class shifted in fall from zooplankton to plant material, and the proportion of sculpin in the diet increased. The average size of sculpin observed in the diets of 200-299 mm Northern Pikeminnow was 52 mm, and ranged 39–83 mm (n = 5). No salmonids were observed in the diets of the 200-299 mm size class of Northern Pikeminnow. The diets of the largest (300-599 mm) size class of Northern Pikeminnow contained primarily fish and crayfish. Their spring diets were largely composed of crayfish and smaller Northern Pikeminnow in approximately equal parts. During summer, crayfish still represented about 50% of the diet, but the proportion of Northern Pikeminnow decreased, and the proportions of sculpin and salmonids increased. The proportions of sculpin and salmonids continued to increase during fall.

The maximum fork length of prey fish consumed increased with the fork length of predatory Northern Pikeminnow (Figure 10). Salmonids were the largest prey fish consumed and occasionally exceeded 50% of the predator's fork length when reconstructed from diagnostic bones. Four 100-150 mm fork length salmonids were found in the guts of 300-399 mm Northern Pikeminnow. Six 200-325 mm salmonids were observed in ≥400 mm Northern Pikeminnow diets. As prey, the average fork length of Northern Pikeminnow eaten by 300-399 mm cannibalistic Northern Pikeminnow was 143 mm (range: 143-144, n = 2), and 164 mm for predators > 400 mm (range: 101–252 mm, n = 10). The average fork length of sculpin found within ≥300 mm Northern Pikeminnow was 46.6 mm (range: 20-138 mm; n = 16), however this mean was highly influenced by one individual that consumed 12 small sculpin between 19-32 mm.

Age and growth

Northern Pikeminnow in Lake Merwin exhibited a strong linear relationship between fork length (FL, mm) and scale radius (SR, mm) for the combined sexes (n = 101, fork length range: 34-560 mm; r^2 = 0.97):

$$FL = 95.371 * SR + 1.9455$$

The greatest scale-based age observed was 14 years (431 mm male and 542 mm female), however, 76% of the population of Northern Pikeminnow ≥200 mm were age 3 and 4 based on relative catch at age developed using an age-length key. The von Bertalanffy growth parameters fit for females were L_∞ = 786.64, K = 0.0665, and t_0 = -2.0139 (n = 34). Corresponding parameters for the male model were L_∞ = 689.23, K = 0.0599, and t_0 = -3.0117 (n = 38; Figure 11).

Overall, our results show that Northern Pikeminnow are most densely distributed in the upper 15 m of the water column during stratification. The size structure is highly skewed toward smaller individuals. There was not a statistically significant change in the size structure of Northern Pikeminnow from the historical study, suggesting no large response to Tiger Muskie or other changes. However, the high mortality of 200-299 mm Northern Pikeminnow, which are likely prey items for large Tiger Muskie (Baker et al. 2015), suggests Tiger Muskie may be at least partially controlling the population of Northern Pikeminnow. The apparent mortality in this size group also rendered it difficult to accurately estimate abundance, and further work is underway to generate additional estimates of the abundance of predatory sized fish. Stable isotopes and diet analysis both suggest that ≥300 mm fish have significantly higher per-capita consumption of fish. The abundance of this size class may be most important for estimating the potential predation impact on reintroduced juvenile salmonids. Bioenergetics modeling is currently underway to evaluate the magnitude of this impact.

Logical next steps for this task include relating the growth trajectories for Northern Pikeminnow to ecological processes in Lake Merwin such as: size frequency related to age at maturity, cannibalism, and piscivory on salmonids and other fishes. We will relate the vulnerable size range of Northern Pikeminnow to cannibalism to growth rate to estimate how long [how many age classes] are subjected to cannibalism. Bioenergetics simulations of cannibalism and predation by Tiger Muskie will be used to explore the relative capability of each predator to regulate the Northern Pikeminnow population and maintain the observed size structure.

References

Al-Chokhachy, R., M. Sorel, D. A. Beauchamp, C. Clark, and E. D. Lowery. 2014. Development of New Information to Inform Fish Passage Decision at the Yale and Merwin Hydro

Projects on the Lewis River. Annual Report submitted to PacifiCorp. U.S. Geological Survey, Bozeman, Montana.

Baker, W. P., M. J. Divens, and B. D. Bolding. 2015. Tiger muskellunge growth, condition, diet, and effect on target prey species in two Eastern Washington lakes: progress report 2001 - 2006. Washington Department of Fish and Wildlife, FPT 15-02, Olympia, Washington.

Beauchamp, D. A., C. Luecke, W. A. Wurtsbaugh, H. G. Gross, P. Budy, S. Spaulding, R. Dillenger, and C. P. Gubala. 1997. Hydroacoustic assessment of abundance and diel distribution of sockeye salmon and kokanee in the Sawtooth Valley Lakes, Idaho. North American Journal of Fisheries Management 17(253-267).

Chipps, S. R., and J. E. Garvey. 2007. Assessment of diets and feeding patterns. Pages 473-514 in C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

Hamilton, J. A. R., L. O. Rothfus, M. W. Erho, and J. D. Remington. 1970. Use of a hydroelectric reservoir for rearing of coho salmon (*Oncorhynchus kisutch*). Washington Department of Fisheries, Research Bulletin No 9., Olympia, Washington.

Hansen, M. J., C. P. Madenjian, J. H. Selgeby, and T. E. Helser. 1997. Gillnet selectivity for lake trout (Salvelinus namaycush) in Lake Superior. Canadian Journal of Fisheries and Aquatic Sciences 54(11):2483-2490.

Isaak, D. J., and T. C. Bjornn. 1996. Movement of northern squawfish in the tailrace of a lower Snake River dam relative to the migration of juvenile anadromous salmonids. Transactions of the American Fisheries Society 125(5):780-793.

Isely, J. J., and T. B. Grabowski. 2007. Age and growth. Pages 187-228 in C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of inland fisheries data. American Fisheries Society, Bethesda, Maryland.

Lynch, J. M. 1993. Evaluation of the Merwin trap as a means of northern squawfish (*Ptychocheilus oregonensis*) control in the Columbia River. University of Washington, Seattle, Washington.

McIntyre, J. K., D. A. Beauchamp, M. M. Mazur, and N. C. Overman. 2006. Ontogenetic trophic interactions and benthopelagic coupling in Lake Washington: Evidence from stable isotopes and diet analysis. Transactions of the American Fisheries Society 135(5):1312-1328.

Miranda, L. E., and P. W. Bettoli. 2007. Mortality. Pages 229-278 in C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

Nelson, A. G. 2014. fishmethods: fishery science methods and models in R. R package version 1.7-0.

PacifiCorp, and Cowlitz-PUD. 2004. Report on life history, habitat requirements, and distribution of aquatic analysis species (AQU 1). PacifiCorp and Cowlitz PUD, Portland, Oregon.

PacifiCorp, and Cowlitz-PUD. 2014. Lewis River hatachery and supplementation plan. PacifiCorp and Cowlitz PUD, Portland, Oregon.

Seber, G. A. F. 1982. The estimation of animal abundance. Griffin, London, England.

Spangler, G. R., and J. J. Collins. 1992. Lake Huron fish community structure based on gill-net catches corrected for selectivity and encounter probability. North American Journal of Fisheries Management 12:585-597.

Tabor, R. A., S. T. Sanders, M. T. Celedonia, D. W. Lantz, S. Damm, T. M. Lee, Z. Li, and B. E. Price. 2010. Spring/summer habitat use and seasonal movement patterns of predatory fishes in the Lake Washington Ship Canal. U.S. Fish and Wildlife Service, Lacey, Washington.

Vander Zanden, M. J., and J. B. Rasmussen. 1999. Primary consumer delta C-13 and delta N-15 and the trophic position of aquatic consumers. Ecology 80(4):1395-1404.

Table 1. Limnological and morphometric characteristics of Lake Merwin (PacifiCorp 2004, PacifiCorp and Cowlitz PUD. 2004, PacifiCorp Unpublished data).

Attribute	Estimate
Surface area (km^2)	15.7
Length (km)	23.3
Mean depth (m)	31
Maximum depth (m)	73
Mean Secchi depth transparency (m)	~5
Water residence time (days)	118.4
Total phosphorus (mg/L)	<.02
Trophic status	Oligotrophic
Months stratified	April-Sept.
Depth of thermocline (m)	~8
Ice cover	None
Range of surface temps (°C)	23-5
Maximum surface elevation (m msl)	73

Table 2. Relative abundance of fish species present in Lake Merwin based on field sampling.

Species	Relative abundance
Kokanee Salmon (*Oncorhynchus nerka*)	High*
Rainbow/Steelhead Trout (*O. mykiss*)	Moderate*
Bull Trout (*Salvelinus confluentus*)	Rare
Cutthroat Trout (*O. clarkii clarkii*)	Rare
Juvenile Coho Salmon (O. *kisutch*)	Rare*
Juvenile Chinook Salmon (*O. tshawytscha*)	Rare*
Northern Pikeminnow (*Ptychocheilus oregonensis*)	High
Largescale Suckers (*Catostomus macrocheilus*)	High
Sculpin (*Cottus* spp.)	High
Threespine Stickleback (*Gasterosteus aculeatus*)	Rare
Tiger Muskie (*Esox Lucius x masquinongy*)	Moderate*
Pumpkinseed Sunfish (*Lepomis gibbosus*)	High
Largemouth Bass (*Micropterus salmoides*)	Rare

* hatchery supplemented population

Table 3. Mean δ ^{15}N and δ ^{13}C stable isotope values for fish and invertebrates sampled in the Lewis River reservoirs by location and size class. Abbreviations PS = Pumpkinseed Sunfish, CCT = Coastal Cutthroat Trout, RBT = Rainbow Trout, LMB = Largemouth Bass, LSS = Largescale Sucker, NPM = Northern Pikeminnow, and SB = Threespine Stickleback.

	Spp.	Size Class (mm)	N	Mean δ ^{15}N	Minimum δ ^{15}N	Maximum δ ^{15}N	Mean δ ^{13}C	Minimum δ ^{13}C	Maximum δ ^{13}C
Merwin	PS	100-199	3	5.860	5.637	6.201	-25.895	-26.350	-25.592
	Clam	-	4	3.231	2.607	3.848	-28.656	-30.751	-25.627
	Coho	200-299	4	6.288	5.888	6.723	-26.236	-27.546	-24.334
	Copepods	-	1	3.160	3.160	3.160	-28.711	-28.711	-28.711
	Sculpin	<100	9	6.112	5.307	6.685	-25.386	-27.959	-21.971
		100-199	3	8.068	6.839	8.946	-23.354	-24.257	-22.855
	Crayfish	-	2	4.054	2.801	5.306	-21.718	-24.198	-19.238
	CCT	100-299	3	7.318	7.037	7.844	-26.639	-29.092	-23.817
	Daphnia	0	1	-0.758	-0.758	-0.758	-31.270	-31.270	-31.270
	Diptera	0	1	1.726	1.726	1.726	-26.537	-26.537	-26.537
	Hemiptera	0	1	1.194	1.194	1.194	-27.648	-27.648	-27.648
	Holopedium	0	1	0.695	0.695	0.695	-30.916	-30.916	-30.916
	RBT	<299	7	5.605	4.992	6.489	-24.342	-26.121	-22.133
		300-399	7	9.186	6.544	10.757	-20.956	-26.262	-17.692
		500-699	6	13.081	12.329	13.656	-17.658	-18.230	-17.177
	Kokanee	<399	9	6.423	5.232	7.573	-26.713	-28.186	-25.062
	LMB	300-499	2	9.084	8.585	9.582	-22.085	-22.768	-21.402
	LSS	<200	4	5.841	5.415	6.286	-24.051	-24.763	-23.791
		300-499	5	6.182	5.849	6.380	-27.943	-29.670	-25.710
	NPM	<200	9	5.451	4.639	6.963	-27.315	-29.245	-26.005
		200-299	6	6.519	6.179	7.626	-25.242	-25.543	-24.708
		300-399	5	8.640	7.353	9.627	-24.584	-26.729	-23.479
		400-599	9	9.348	8.278	10.048	-23.714	-24.899	-21.826
	Juga snail	-	4	2.576	1.993	3.135	-20.409	-22.691	-16.770
	Tiger Muskie	400-799	11	8.333	7.546	9.275	-24.513	-25.396	-23.345
		800-1099	9	9.608	8.902	10.236	-24.048	-25.041	-22.930
Yale	Bull Trout	300-399	2	7.923	7.913	7.933	-24.729	-26.582	-22.876
		400-799	11	9.385	8.844	10.164	-24.682	-27.557	-22.203
	Copepods	-	1	3.684	3.684	3.684	-31.088	-31.088	-31.088
	Sculpin	<100	5	6.189	5.285	7.603	-25.036	-28.558	-21.887
		100-199	1	7.057	7.057	7.057	-22.094	-22.094	-22.094
	Crayfish	-	2	3.037	2.568	3.507	-23.325	-26.952	-19.697
	CCT	200-399	7	7.499	5.706	8.549	-24.756	-27.164	-22.960

	Daphnia	-	2	0.463	-0.126	1.053	-31.221	-31.654	-30.789
	Holopedium	-	1	0.872	0.872	0.872	-31.094	-31.094	-31.094
	Plankton	-	1	1.034	1.034	1.034	-26.575	-26.575	-26.575
	Kokanee	<299	13	6.266	5.460	6.897	-30.441	-33.570	-27.435
	LSS	200	4	4.664	4.148	5.021	-21.376	-24.634	-19.008
		300	5	6.128	5.707	6.956	-26.099	-27.727	-24.416
	NPM	<200	8	4.753	4.207	5.263	-26.657	-29.247	-25.138
		200-299	6	6.304	5.640	6.988	-25.143	-25.543	-24.682
		300-399	6	8.083	7.431	9.427	-23.124	-26.712	-21.698
		400-599	4	8.564	7.962	9.356	-24.831	-26.950	-21.637
	RBT	200	4	6.095	5.182	7.708	-24.074	-26.922	-19.334
	SB	<100	5	6.334	4.580	7.355	-28.276	-30.050	-26.715
Swift	Bull Trout	100-299	4	8.618	7.146	11.898	-23.433	-24.877	-22.047
		300-699	16	10.417	8.664	11.846	-24.100	-27.718	-21.579
	Caddisfly	-	1	-0.297	-0.297	-0.297	-24.516	-24.516	-24.516
	Chinook	100-299	7	5.504	4.817	7.126	-26.201	-27.380	
	Coho	<200	8	5.405	4.398	6.721	-25.534	-27.146	-23.786
		200-299	5	6.454	5.441	7.224	-26.336	-26.931	-25.486
	Copepods	-	1	3.675	3.675	3.675	-29.048	-29.048	-29.048
	Sculpin	<100	5	4.856	3.796	7.048	-25.573	-29.037	-22.757
		100-199	3	6.243	4.360	8.016	-25.588	-27.891	-23.361
	Crayfish	-	4	3.911	3.118	5.608	-23.728	-25.457	-22.432
	CTT	200-399	11	7.045	5.832	8.201	-24.368	-25.813	-22.616
	Daphnia	-	1	0.708	0.708	0.708	-28.709	-28.709	-28.709
	RBT	100-199	5	4.924	2.845	6.378	-24.982	-25.904	-23.550
		200-299	9	6.236	5.120	8.003	-25.191	-27.289	-23.040
		300-399	11	7.595	5.245	11.187	-23.379	-27.230	-19.048
		500-599	3	11.410	10.194	12.211	-18.278	-20.058	-17.073
	LSS	<500	8	4.584	3.563	6.652	-24.382	-25.928	-22.315
	Snails	-	1	2.415	2.415	2.415	-23.750	-23.750	-23.750
	MWF	200-399	4	5.387	4.921	5.844	-21.953	-22.420	-20.770
	SB	400	1	7.498	7.498	7.498	-20.548	-20.548	-20.548

Figure 1. Isoclines of average water temperature (°C) from two stations 1/3 and 2/3 along the length of Swift, Yale, and Lake Merwin during April-October 2014, and November 2014 in Yale Lake. In Swift Reservoir, a profile from November 1999 and a partial profile from November 2013 were combined, and profiles inferred from tailrace temperatures from February and December 2013 were incorporated to fill out the seasonal thermal regime. Supplementary profiles from the lower half of Yale Lake from December 1997 and February 1998, and a composite of a November 1999 and a partial profile from November 2013, and January and March 2000 from the lower half of Lake Merwin were also included.
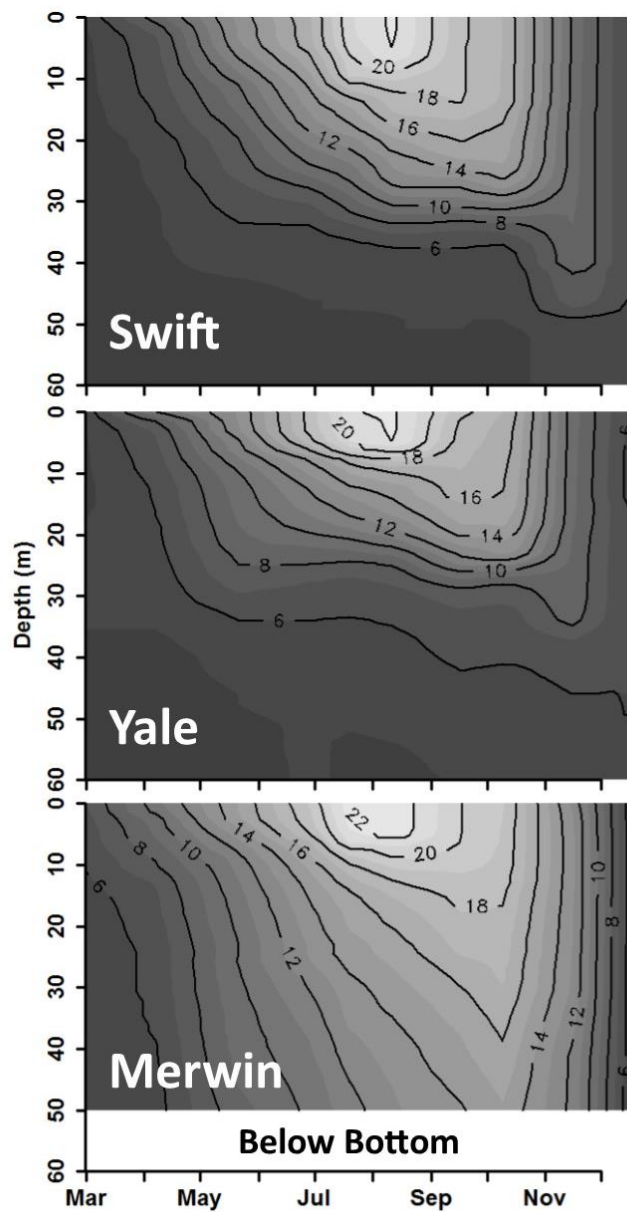
Figure 2. Reservoirs in the North Fork Lewis River illustrating the location of limnology stations (stars) where temperature and zooplankton were sampled, and the locations of traps in Lake Merwin (circles); inset demonstrates the location of the study area.
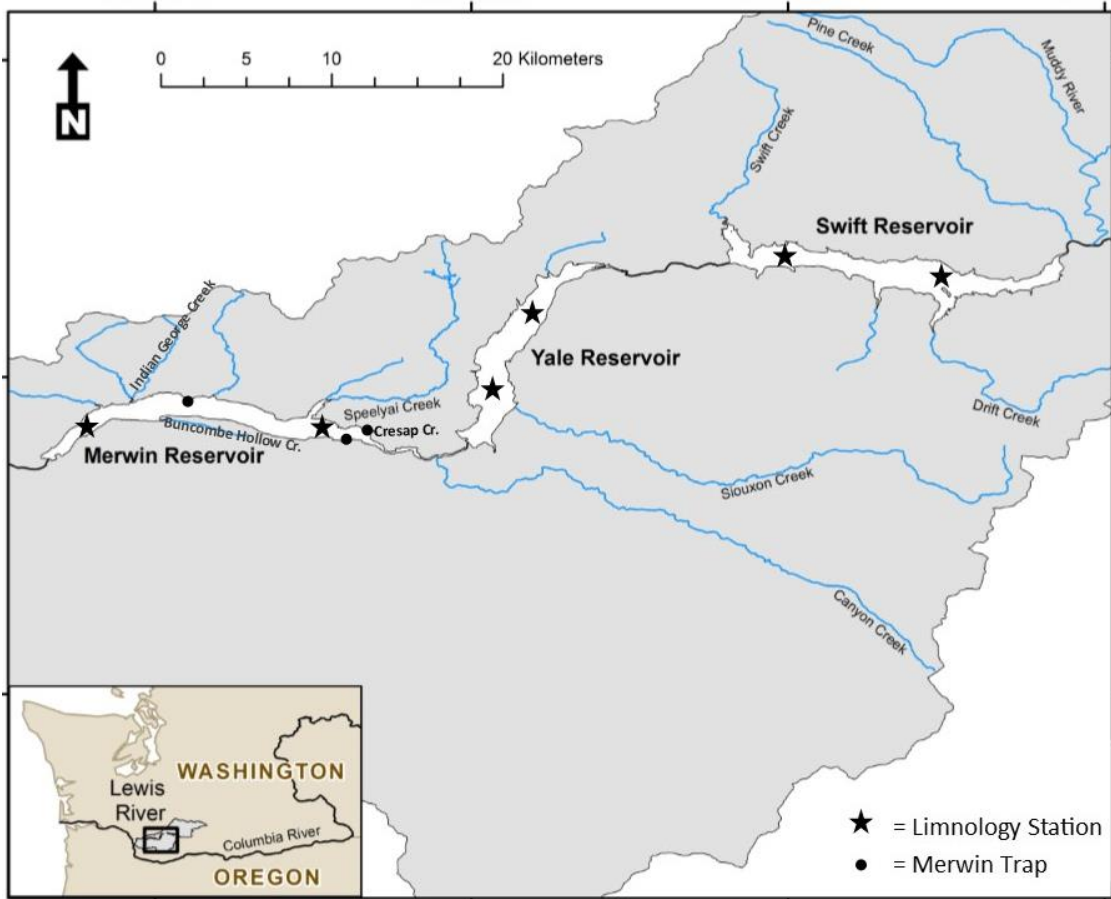
Figure 3. Limnological stations (star) and Merwin Trap locations (black circles) within Lake Merwin.
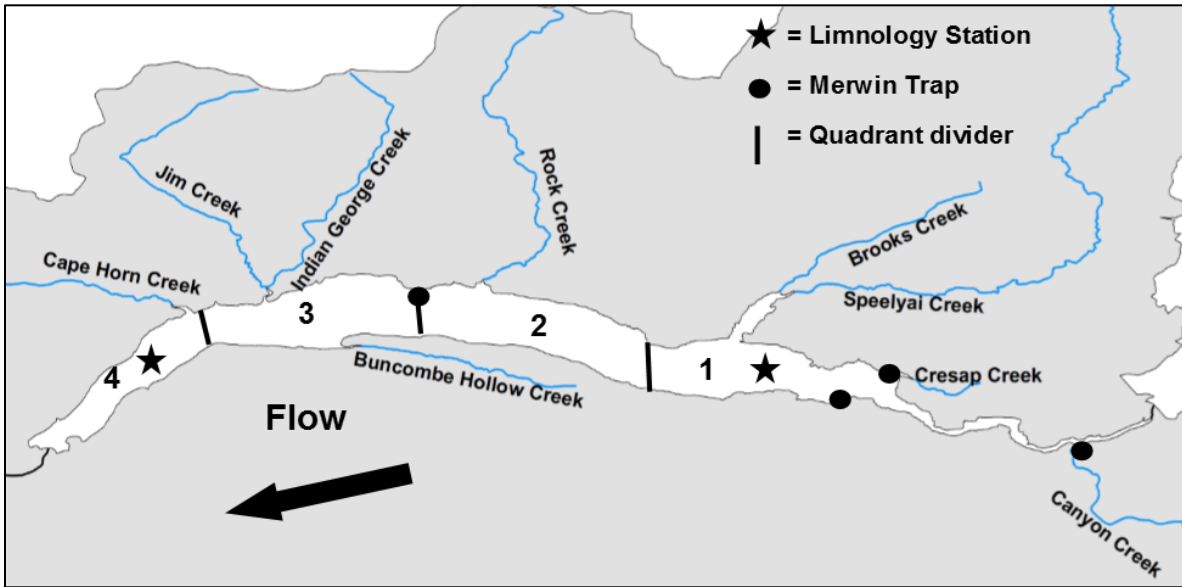

Figure 4. Catch-per-unit-effort for Northern Pikeminnow ≥200 mm caught in sinking horizontal gill nets by depth and season in Lake Merwin.
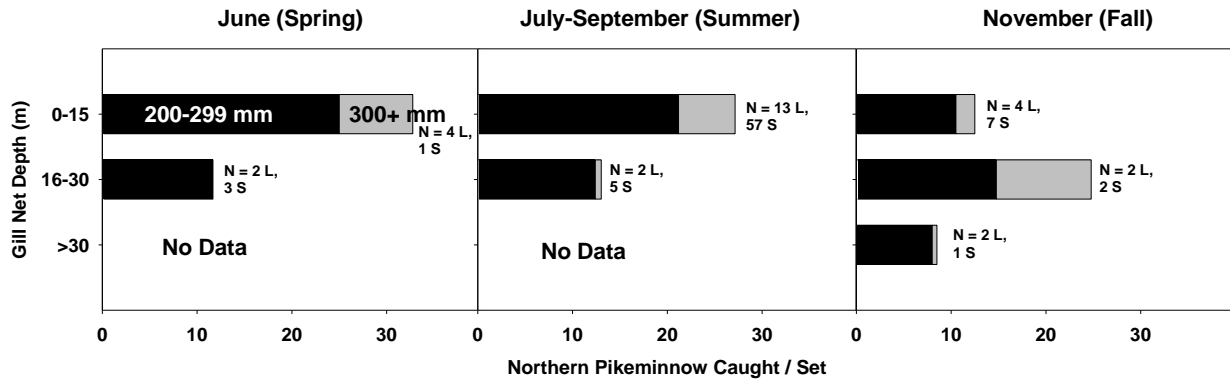
Figure 5. Length frequency distributions of male and female Northern Pikeminnow captured in gill nets and traps in Lake Merwin between 2013-2014 and 1958-1963 (Hamilton et al. 1970). The contemporary gill netting included larger mesh sizes than the historical data, which were included in the 2013-2014 distribution to best characterize the size distribution of the population. Efforts were made to standardize the length frequencies by including only fish captured in similar areas.
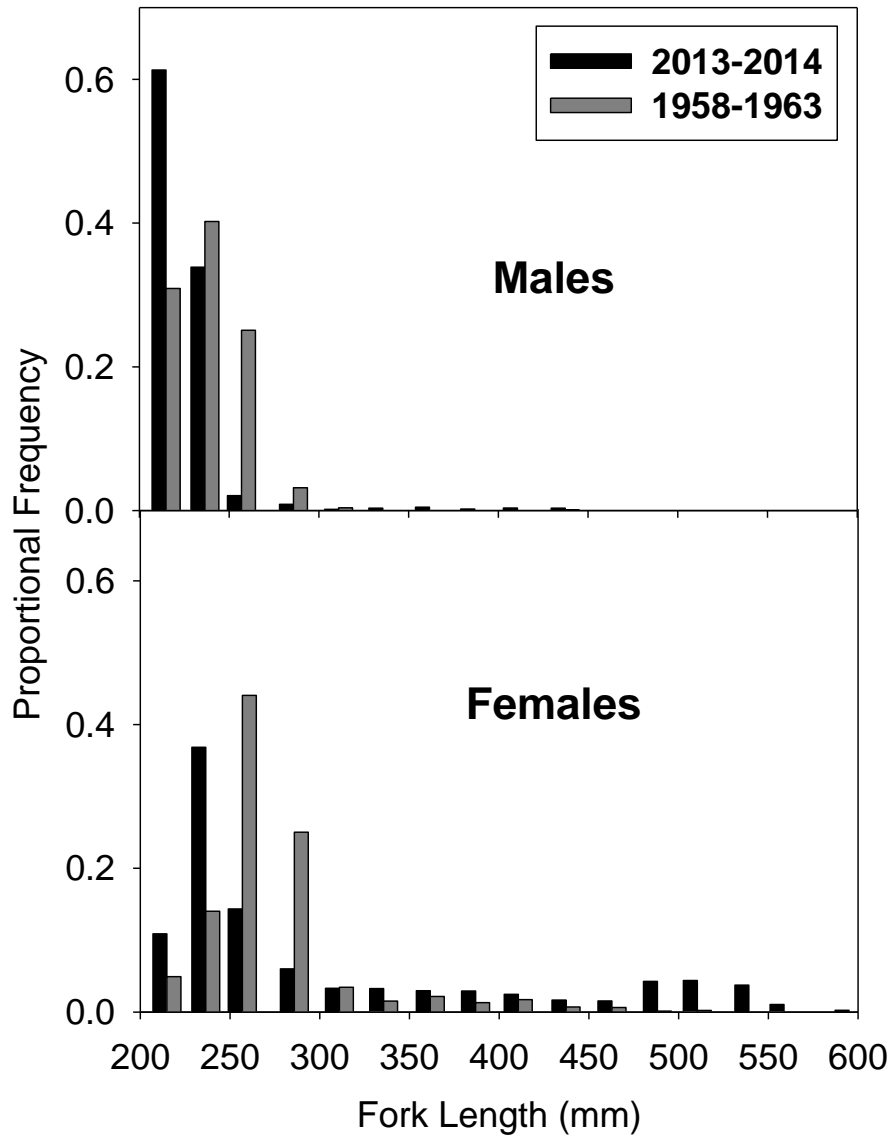
Figure 6. Length frequency histograms from raw catch data and data adjusted for size selectivity. The top panel shows Merwin trap data which was adjusted for size-selective encounter rate only, and the bottom panel shows gill net data that was adjusted for size selectivity of gill-net mesh sizes as well as encounter rate. Adjusted and unadjusted size distributions were not statistically significant for both gear types.
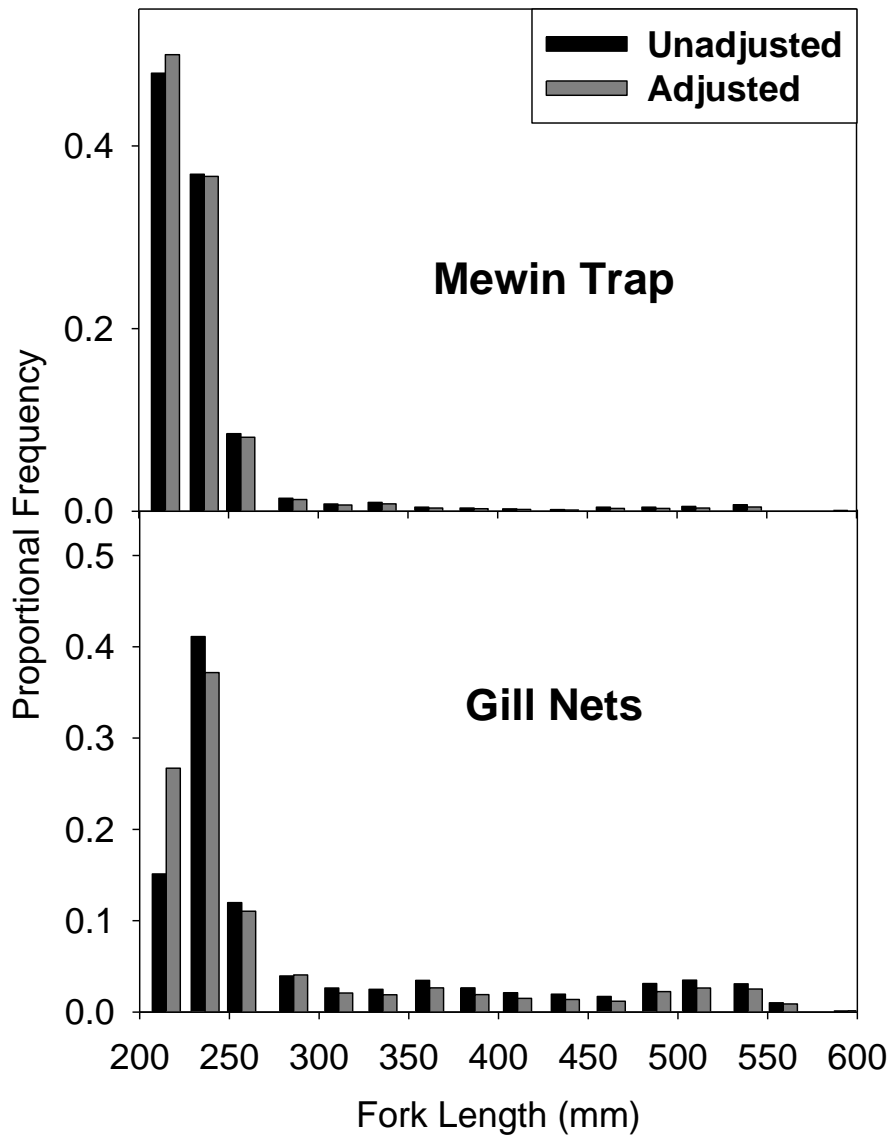
Figure 7. Average δ¹⁵N and δ¹³C signatures (error bars = 2 SE) for organisms sampled in Lake Merwin where: CCT = Coastal Cutthroat Trout; COHO = Coho Salmon; COT = Sculpin; Crayfish (*Pacifastacus leniusculus*); KOK = Kokanee; LMB = Largemouth Bass; LSS = Large-Scale Sucker; NPM = Northern Pikeminnow; RBT = Rainbow Trout; PS = Pumpkinseed; Snail; and TM = Tiger Muskie.
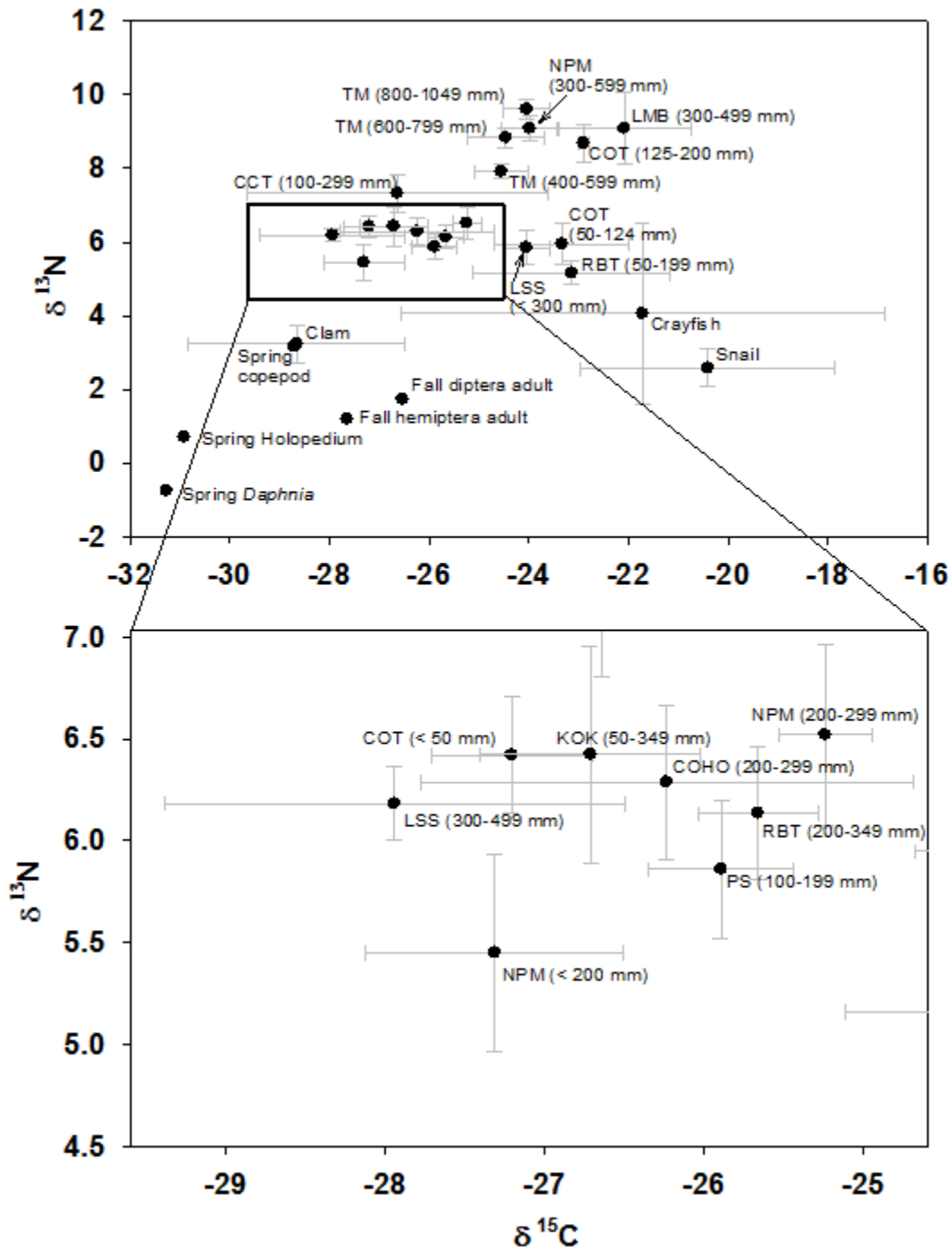
Figure 8. Stable isotope signatures $\delta^{13}C$ (top) and $\delta^{15}N$ (bottom) from Northern Pikeminnow as a function of fork length from sampling conducted during June-November (2013) in Lake Merwin.
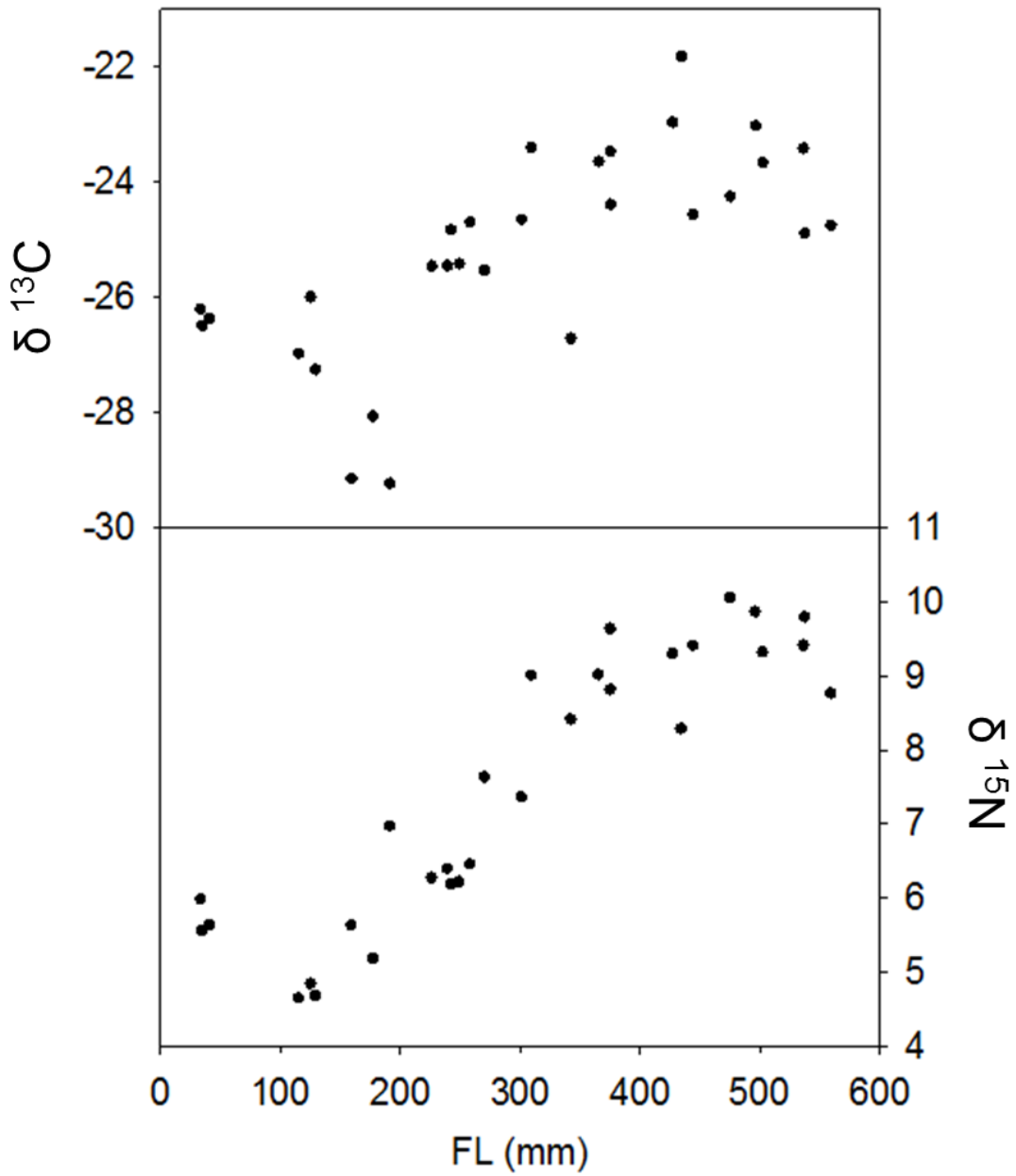
Figure 9. Average seasonal diet proportions by wet weight of two size classes of Northern Pikeminnow in Lake Merwin during spring (May-June), summer (July-September), and fall (November).
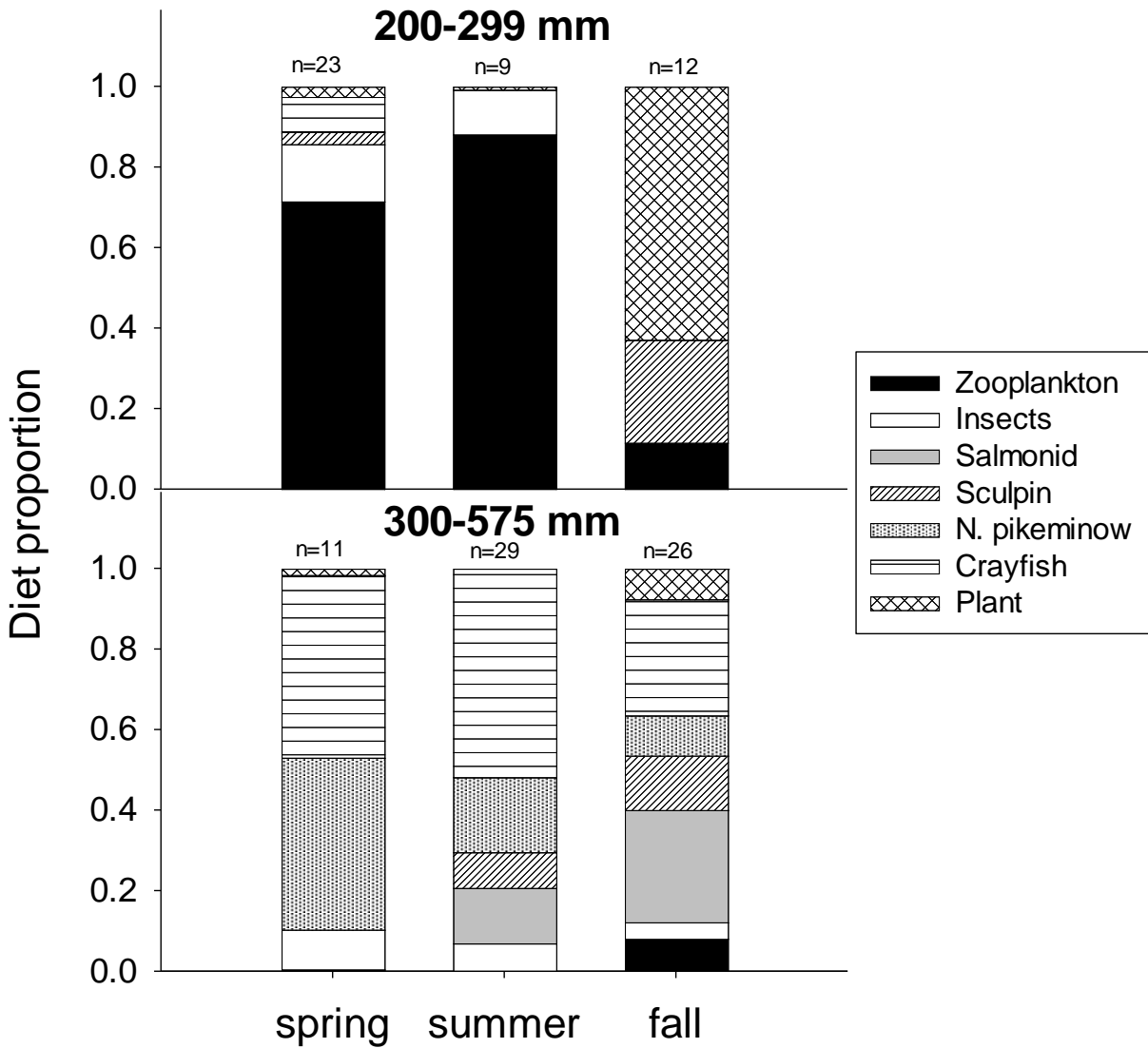
Figure 10. The relationship between Northern Pikeminnow fork length and prey fish fork length from Lake Merwin. Lines represent 50% (solid) and 40% (dashed) of the predator's fork length.
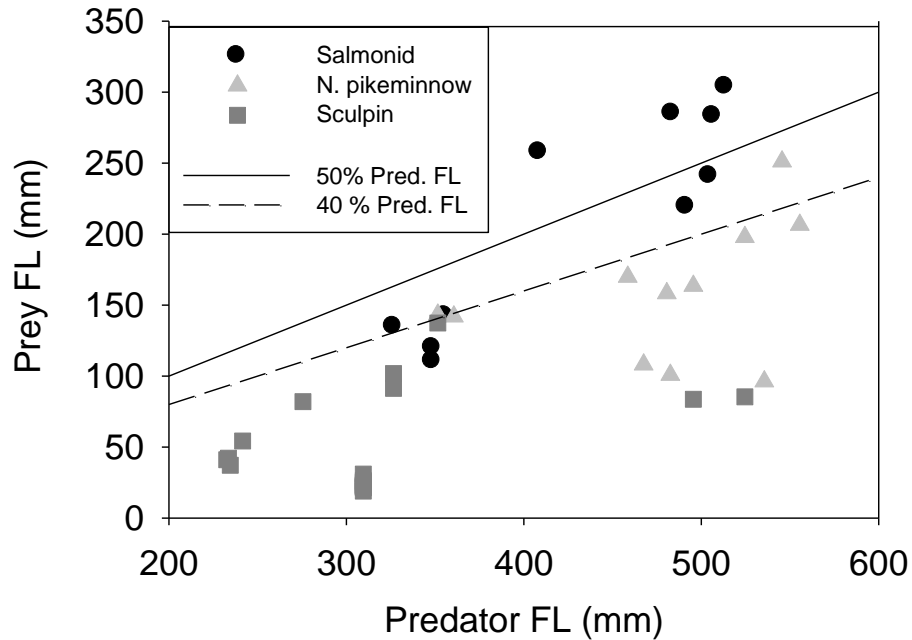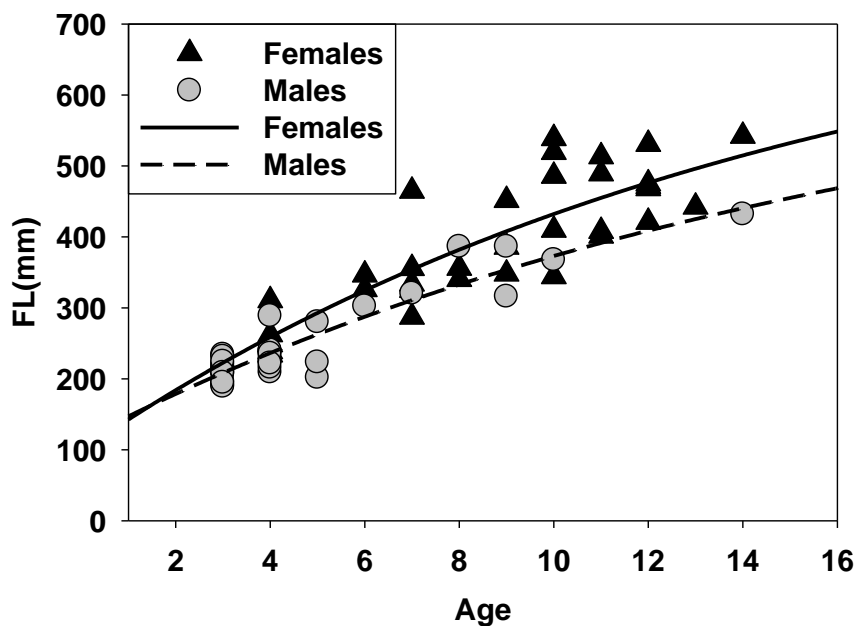

Figure 11. Length at age and von Bertalanffy growth curves for male (grey circles, dashed line) and female (black triangles, solid line) Northern Pikeminnow in Lake Merwin.

Task 6: Anadromous-resident fish interactions

The goal of this task is to assess the effects of anadromous fish introduction on resident species, and, conversely, assess the effects of resident fish on the reintroduced anadromous fish. Juvenile anadromous salmonids will utilize reservoirs as migratory corridors and rearing habitats to varying degrees. Therefore, the magnitude and duration of interactions with resident species will vary among anadromous salmonids. By quantifying existing seasonal, size-structured food web interactions for the key fishes and invertebrates likely to interact directly or indirectly with anadromous salmonids in the three reservoirs, potential limiting factors can be evaluated with regard to whether production or survival are inordinately constrained by temporal availability, quality, or accessibility of food, competition, or predation, and the role of environmental conditions.

Since reintroduction of anadromous salmonids is already underway in Swift Reservoir, the trophic relationships among potential predators, competitors, and prey can be compared based on stable isotope patterns from samples taken before the reintroduction. Stable isotope analysis can help evaluate dietary overlap or partitioning between salmon and resident trout and also examine the role of larger trout as potential salmon predators.

Methods

Stream habitat
During 2014 we continued studies to quantify resident-anadromous interactions in the study area. Given that reintroductions have not occurred in the Lake Merwin and Yale Lake, efforts were largely focused in tributaries to Swift Reservoir. Our specific objectives were to continue to quantify: 1) the distributional overlap between Bull Trout and juvenile Coho Salmon; 2) the spatial and temporal overlap between hatchery released juveniles (i.e., juvenile Chinook); and 3) identify potential community interactions. Data will continue to be collected to assess anadromous-resident interactions in stream habitat during 2015. Methods to assess such interactions will continue through 2015 and will include assessments of redd superimposition by anadromous species on Bull Trout (Pine Creek and P8), and an overall integration of measures of stream productivity, species-specific diet data, isotope data, and bioenergetics modeling.

Distributional overlap

Similar to 2013, we quantified distributional overlap of Bull Trout and juvenile Coho using primarily snorkel surveys in Pine Creek and P8. We selected these streams due to the relatively high abundance of Bull Trout in these streams in comparison to other streams occupied by Bull Trout (e.g., Rush Creek). While snorkeling studies can

substantially underestimate the abundance of Bull Trout, particularly during daytime studies (Thurow et al. 2006), we selected daytime snorkeling surveys due to the understanding that Bull Trout commonly use Pine Creek and P8 and the need to quantify juvenile Coho distribution, which is unlikely to have pronounced diel bias. To assess the distribution of both species, we used a systematic sampling approach (random start) as both species are likely clumped in distribution (e.g., Al-Chokhachy et al. 2009; Figure 4). Reach lengths varied by stream with reaches of at least 200 m used in Pine Creek and 150 m in P8. We partitioned the entire study area into reaches and sampled every 3rd reach from the random start for an approximate annual sample rate of 33%. Each year subsequent to 2013 we sampled an adjacent set of systematic sample reaches to effectively sample the entire study area during the duration of this study (i.e., 2013-2015).

During snorkel surveys, two individuals progressed upstream, remaining relatively parallel to each other; all side channels and channel units were included in each survey. All fish were recorded into 50-mm categories, and reported to a third individual on the streambank. Snorkelers carried flashlights to improve visibility in shaded areas and in sections with large numbers of woody debris.

In addition to overlap with Bull Trout, we considered the potential interactions between acclimation Chinook and wild Coho in Clear Creek. Specifically, we evaluated the spatial and temporal overlap of acclimation fish using location of tagging information and emigration patterns (i.e., through recapture events at the fixed antennae on Clear Creek) of marked Coho (Task 4) and PIT-tagged acclimation Chinook. The migration timing data will help identify 'residence' time of acclimation Chinook and migration patterns of wild Coho in response to the population of acclimation fish.

Foodweb and community interactions

During 2014, we continued to assess community interactions using a combination of stable isotope data and diet analyses. We revisited sites previously sampled during 2009 as part of a 'baseline' assessment of community interactions prior to large-scale anadromous reintroductions in the Upper Lewis River (Figure 5). We focused our efforts on tributaries to Swift Reservoir, given that anadromous reintroductions have been initiated and included sites on: Swift Creek, Drift Creek, Rush Creek, P8, Cussed Hollow, and the Upper Lewis (above Lower Falls; Figure 5). In addition, we also collected diet and isotope data at two sites on Clear Creek upstream of the confluence with Muddy River. At each site, we collected samples during June-July, mid to late August, and in early November after the onset of Coho spawning.

At each site, we sampled macroinvertebrates, collected tissue samples from fish, and collected diet data. We used a variety of sampling techniques to capture fish including

backpack electrofishing, electrofishing down to a seine, snorkel herding fish into a seine, and minnow traps. Here we did not constrain our sampling approaches to only electrofishing methods, which were used during baseline efforts, as we were interested in the most efficient methods for capturing appropriate sample sizes across species. However, differences in sampling methods are unlikely to affect results from isotopic and diet studies.

Once captured, we anesthetized fish with AQUI-S, weighed and measured each fish and collected tissue and diet samples from each species. Tissue samples were taken from the anal fin (3-mm x 3 mm) and frozen for isotopic analyses. To collect diet data, we used gastric lavage methods for all fish >60 mm. After sampling, we placed fish in flow-through recovery tanks within the sample stream; upon full recovery of equilibrium fish were released at their point of capture. We sampled for macroinvertebrates using a drift nets set in the thalweg at the bottom of each reach with set times of approximately one hour. We elected not to use Serber approaches, which were used in the 2009 baseline surveys, to minimize sorting of items, and given the fact that most fishes in the foodweb study are drift feeders. However, differences in methods are unlikely to influence interpretations of isotopic analyses.

All drift, diet, and fin clips for isotope samples were placed in water and frozen until analyzed. All diet and isotope samples were sorted to family and grouped based on functional feeding groups. Isotopic samples were prepared following the University of California, Davis Stable Isotope Facility protocol (http://stableisotopefacility.ucdavis.edu/). Briefly, fin samples were dried at temperatures between 50-60°C, set aside in a dehydrator to allow for acclimation to humidity, ground using a mortar and pestle and encapsulated in a tin capsule for analyses. All isotopic analyses were conducted by the UC Davis Stable Isotope Facility.

Redd superimposition

Given that both Bull Trout and Coho are fall-spawners, we investigated the potential for and evidence of superimposition of spawning redds (e.g., Weeber et al. 2010). First, we evaluated the potential for superimposition using existing redd data for Bull Trout and Coho. Here, we quantified the date of each redd observed for each species during recent (2012- current) surveys in the upper Lewis and calculated the proportion of Coho redds that occurred after the onset of Bull Trout spawning.

Next, in 2014 we assessed superimposition in Pine Creek given the abundance of Bull Trout and the abundance of reintroduced adult Coho in Swift Reservoir. Due to the potential for misidentifying Bull Trout redds (Dunham et al. 2001), we used existing redd location information for Bull Trout (i.e., flagged redds) from local biologists. Starting in mid-October, we conducted biweekly redd surveys to identify Coho redds superimposed

on Bull Trout redds. With high levels of suspended and delivered fine sediment in Pine Creek during the fall, accurately identifying the boundaries of existing Bull Trout redds can be challenging. As such, we delineated the boundaries of each redd using painted rocks (white) and characterized superimposition where Coho redds overlapped within the perimeter of Bull Trout redds.

Reservoir habitat
Carrying capacity of reservoir habitat: zooplankton supply and demand

The carrying capacities of Swift, Yale and Merwin reservoirs for reintroduced anadromous salmonids were investigated to evaluate whether the reservoirs represent viable and profitable rearing habitat. Our approach to assess potential anadromous-resident fish interactions and carrying capacity in each reservoir is similar to that used to address Task 5, and therefore, some methods and data used in Task 5 overlap with Task 6. Seasonal temperature profiles and depth distributions determined by catches in gill nets and hydroacoustic surveys were combined to estimate the thermal experience of key species. Seasonal diets have been characterized for key resident fishes in addition to Northern Pikeminnow. Monthly zooplankton biomass and production were estimated based on zooplankton densities, length measurements, and egg counts. Monthly consumption demand estimates for resident salmonids were compared to monthly zooplankton biomass and production in each reservoir to estimate the amount of prey available to support additional juvenile anadromous salmonids.

Fish sampling

We deployed short term gill nets for 30 minutes to 2 hours in Yale and Swift Reservoirs in August and November 2013 and May through November 2014 to obtain biological samples of Bull Trout and other resident fishes. We only set gill nets for short periods to reduce the risk of harming ESA listed Bull Trout. When Bull Trout were captured, stomach contents were collected using non-lethal gastric lavage, and other non-lethal samples (fin clip and scales) were taken before being released. See methods for Task 5 above for description of fish sampling in Lake Merwin. For small pelagic fishes, mid-water trawling was conducted in June and August 2013 in all three reservoirs but was restricted to the epilimnion due to submerged standing timber. Trawling was therefore ineffective in catching fish >50 mm which were in very low densities in the epilimnion. Creel sampling was effective for collecting length and weight data, approximate depth distribution, and samples of fish (diet, scale, stable isotope) in Yale and Swift Reservoirs, as was trolling in Swift Reservoir.

Stable isotope analysis

We characterized the food web structure of Swift and Yale Lakes with stable isotopes to identify key trophic pathways supporting or potentially limiting juvenile salmonids and other key resident fishes. Zooplankton used for stable isotope analysis were collected in July 2014 at the lower limnology site (1/3 of the way up from the dam) in Swift Reservoir (Task 5, Figure 2), and with vertical hauls using a conical net in Yale Lake in June, August, and November 2009 by PacifiCorp biologists. Minnow traps, and hand collection while snorkeling were used to obtain sculpin (*Cottus* spp.), Signal Crayfish (*Pacifastacus leniusculus*), and snails (Unidentified Spp.) in Yale and Swift Reservoirs, although no snails were available in Yale Lake. PacifiCorp also collected stable isotope samples in all three reservoirs in 2009 and 2010, of which a subsample were analyzed to fill data gaps and for comparison with contemporary samples. We removed caudal fin tissue from a range of fish species and size classes in each reservoir, and dried and ground approximately 5-10 samples from each species and size class for analysis (Task 5, Table 3). Samples were analyzed in the same fashion as those described in Task 5.

Thermal experience and depth distribution

We measured vertical temperature profiles and depth distribution of specific fish species in each reservoir to determine thermal experience for bioenergetics simulations. The depth distributions and thermal structure were also used to evaluate the potential extent of temporal-spatial overlap between key predators and prey. Vertical temperature profiles were measured monthly at two stations in Yale and Swift Reservoirs from April to October 2014 (and November 2014 in Yale) in the same fashion as described in Task 5. Depth-specific temperatures were averaged between the two stations. We used temperature profiles taken in November 1999 at the lower site in Swift reservoir by PacifiCorp biologists with a Hydrolab[©] Surveyor 2 or 3 (PacifiCorp and Cowlitz-PUD 2004b) , temperature measurements taken in the Swift tailrace in February and December 2013 (PacifiCorp, unpublished data), and temperature profiles taken by PacifiCorp biologists at the lower site in Yale Lake in March 1997 and January 1998 (PacifiCorp 1999) to fill data gaps and provide information on the thermal regime throughout an entire year.

Zooplankton

We measured the monthly density of *Daphnia* spp. in each reservoir to estimate the food supply available for resident and different population sizes of reintroduced anadromous salmonids. We conducted 2-3 depth-stratified oblique tows per month at each station where temperature was taken during April-October 2014 (and November 2014 in Yale Lake). Zooplankton were sampled with a Clarke-Bumpus sampler fitted with a 154 micron mesh conical net. We substituted tows from November 2013 in Swift and Merwin as we were not able to access these reservoirs in November 2014. The

tows were conducted at depths that represented the epilimnion, metalimnion, and hypolimnion (when available) during stratification and similar depths during spring and fall. The shallow depths of the upper reservoir stations, and the presence of standing timber, precluded us from conducting hypolimnetic tows at the upper stations in each reservoir each month. We identified and enumerated crustacean zooplankton (excluding immature copepods and nauplii) in subsamples of each tow, and scanned the entire contents of each tow for rare large bodied organisms (e.g., Leptodora and Chironomid pupae). We measured the body length of the first thirty individuals of each genus encountered in each subsample using ImagePro$^{©}$ digital image analysis software. Cladocerans were measured from the top of the helmet to the base of the tail spine, and copepods were measured from the top of the helmet to the base of the caudal rami. The number of *Daphnia* and Copepod eggs, as well as the number of individuals containing eggs were also enumerated to estimate zooplankton production rates.

Diet analysis

Diet analysis was conducted to provide input data for bioenergetics modeling which was used to evaluate trophic interactions between resident and anadromous fish. Diet analysis was conducted for kokanee in Merwin and Yale Lakes, and hatchery rainbow trout and spring Chinook Salmon in Swift Reservoirs in the same manner as described in Task 5 for Northern Pikeminnow in Lake Merwin. Diet analysis was also conducted for other resident species. Diet composition data is available for Northern Pikeminnow in Yale Lake, and less abundant salmonids such as Bull Trout and Mountain Whitefish in Swift Reservoir and Yale Lake. Sculpin, wild Rainbow Trout, and Cutthroat Trout were sampled in all three reservoirs.

Results and discussion

Stream habitat

Distributional overlap of Bull Trout and Coho

During 2014 we conducted 12 snorkel surveys in Pine Creek and P8 totaling 2.86 km of stream habitat (Figure 1). Including 2013 we have conducted surveys in 6.1 km of stream habitat. Although considerably more juvenile Coho were observed in 2014 (n = 40) than in 2013 (n = 2), the chance of high distributional overlap appears to be minimal during early stages of the reintroduction. The highest numbers of juvenile Coho observed during 2014 were proximate to P3, a small tributary where Coho adults were observed during 2013 fieldwork (unpublished data). During 2015, additional surveys will be conducted to: (1) sample previously unsampled locations in Pine Creek and P8, (2) extend sampling to previously unsampled Bull Trout habitat including P3 and Rush

Creek, and (3) conduct surveys during late spring and summer to evaluate changes in overlap during different temporal periods.

Similar to 2013 the majority of fish observed during snorkel surveys were coastal cutthroat trout and YOY *O. mykiss* spp. and Coastal Cutthroat trout (Figure 2a,b). The majority of Bull Trout were observed in P8 and the reaches in Pine Creek near the confluence with P8. Coho made up a small portion of the total number of fish observed in each reach.

Distributional overlap of acclimation Spring Chinook and juvenile Coho

The temporal overlap between acclimation Spring Chinook and juvenile Coho and other species appears to be relatively low (Figure 3). The median days between release and emigration past the antenna in Clear Creek is 6 days (range = 3 – 362). Approximately 60% of the 6,150 fish observed at the antenna emigrated with 7 days or less. However, over 35% of the fish remained in Clear Creek for more than 50 days. This group of fish all occurred during 2014; it is unclear if this group of fish that resided longer in Clear Creek were direct-released into the river or if fish were placed into the acclimation ponds adjacent to Clear Creek.

Foodweb and community interactions

At the time of this report, we have completed initial foodweb analyses with stable isotopes for P8 (tributary to Pine Creek; Figure 1), one of the major Bull Trout spawning and rearing tributaries in the Swift Reservoir basin. Initial results suggest juvenile Bull Trout (age-2 or less; Al-Chokhachy and Budy 2008) and YOY Coho feed at similar trophic levels during the spring, but utilize different food resources (i.e., different carbon signatures; DeNiro and Epstein 1978; Figure 4a). The Bull Trout feeding niche overlapped most substantially during the spring and fall with juvenile (<150 mm) Coastal Cutthroat Trout (Figure 4a,b). Both species appear to utilize a variety of food resources during the spring including Dipteran larvae, Trichopteran larvae, Lepidopterans, and adult Plecopterans but more restricted forage base during the fall.

Coho juveniles overlapped with larger (>150 mm) Coastal Cutthroat Trout in both trophic position and carbon sources during the spring, a pattern consistent with previous studies (Bilby et al. 1996); no Coastal Cutthroat Trout or Coho were captured during fall surveys. Additional diet data currently being analyzed in addition to the 2009 baseline data will help refine overlap in foodweb interactions and provide insights into potential competitive interactions between juvenile Coho and resident salmonids.

Redd superimposition

Since 2012, adult Coho Salmon have demonstrated a broad distribution of spawning activity and substantial evidence of spawning in the Pine Creek/P8 area (Figure 5). Based on existing redd data, we found that Coho spawning initiated after the majority of Bull Trout spawning has occurred each year (60%; Figure 6). Furthermore, we found that 80% of the Coho spawning occurred after end of documented Bull Trout spawning. The observed timing is consistent or earlier than reported for many populations of Coho Salmon in the Pacific Northwest (Weitkamp et al. 1995) suggesting a high chance of temporal overlap and superimposition.

During 2014 we conducted multiple redd surveys per week beginning in early October and continuing through early November. Based on existing Bull Trout redd information, and Coho surveys we documented 2 confirmed cases of redd superimposition in P8. Additional surveys will be conducted in 2015 to further evaluate the extent of superimposition.

Reservoir habitat

Carrying capacity of reservoir habitat: zooplankton supply and demand

Stable isotope analysis.—Similar to Lake Merwin, benthic and pelagic energy pathways were important for supporting the food webs of Swift and Yale Lakes (See Table 3, Task 5 for sample sizes for all isotopic analyses). The δ^{13}C values of Coho and Chinook Salmon in Swift Reservoir were between those of benthically feeding snails and pelagically feeding *Daphnia,* suggesting a mixed diet of benthic invertebrates and zooplankton (Figure 7). Hatchery Rainbow Trout in Swift Reservoir showed similar δ^{13}C values to snails and crayfish, suggesting a heavier reliance on benthic production than did the juvenile salmon. Bull Trout were the top predator in Swift Reservoir (300-699 mm fish δ^{15}N =10.4‰, ranged: 8.7-11.8‰) and consumed a mix of benthically and pelagically derived energy sources. Their trophic position varied ontogenetically somewhat based on minor shifts in δ^{13}C and δ^{15}N isotope values across size classes.

Kokanee in Yale Lake relied heavily on pelagic production, as evidenced by the similarity in δ^{13}C values to their presumptive prey, *Daphnia* and other zooplankton (Figure 8). The trophic position of large Northern Pikeminnow (≥300 mm) and Bull Trout, the top predators in Yale Lake, suggested that both relied heavily on benthic prey (e.g. trout, sculpins, smaller Northern Pikeminnow, largescale suckers, or crayfish), and pelagic fish like kokanee to a much lesser degree.

Thermal experience and depth distribution

The thermal structures of Swift and Yale Lakes in 2014 were similar, but differed from Lake Merwin. Thermal stratification started in May, peaked in August, and persisted into November in Swift and Yale Lakes (Task 5, Figure 1). Epilimnetic temperatures

fluctuated between 13-14°C in May, and 22-23°C in August. The depth of the epilimnion ranged 5-10 m deep between May and August. The extent of the metalimnion also varied, spanning from the bottom of the epilimnion to 30-50 m across the growing season. Unlike in Lake Merwin, the hypolimnion in Swift and Yale Lakes remained at approximately 5°C throughout the year.

Zooplankton

Daphnia densities and average body size fluctuated in all three reservoirs. A strong late spring peak occurred in all reservoirs between May and July, with a smaller secondary peak in September-October (Figure 9). In general, there was moderate to high densities of edible sized *Daphnia* (2.5-26 per L), a key food item for juvenile salmonids, in the epilimnion and metalimnion throughout the growing season in all three reservoirs. Densities in the epilimnion were generally twice that of the metalimnion, and hypolimnetic densities were always below 1 *Daphnia* /L with the exception of May-August in Lake Merwin.

Diet analysis

Kokanee were the dominant salmonid (and pelagic planktivore) in Yale and Merwin Reservoirs based on catches in gill nets and creel sampling. Hatchery Rainbow Trout and juvenile Chinook Salmon (including mostly residualized smolts from hatchery releases) were the most common planktivores observed in catches in Swift Reservoir. Diets were similar across invertebrate-feeding salmonids in each reservoir, but varied by season (Table 1). Cladocerans represented approximately 50-60% of the diet in spring, but increased to 75-100% during summer and 85-90% in fall.

Together, our results suggest access to the epilimnion by juvenile salmonids was likely restricted somewhat by warm surface water temperatures during mid-July into September. However, we found moderate densities of *Daphnia* (1-9 /L) in the cooler metalimnion throughout the growing season in all three reservoirs. Epilimnetic access to zooplankton is likely much less limited in Swift and Yale than Merwin, however, metalimnetic zooplankton densities in Merwin were fairly high. Monthly and depth-specific densities of *Daphnia* will be used to calculate the total biomass and production rates of *Daphnia* available to juvenile salmonids based on weight-length regressions specific to *Daphnia*, empirical counts of eggs per female, temperature and reservoir volume for ongoing evaluations of the carrying capacity of each reservoir.

Overall our results indicated that the seasonal supply and depth-distribution of *Daphnia* are important factors regulating the carrying capacity of these reservoirs for resident and juvenile anadromous salmonids, which will share zooplankton resources. The diet

proportions presented here will be used in bioenergetics simulations of consumption demand to evaluate whether there is sufficient food to support additional reintroduced salmonid populations.

References

Abbe, T. B., and D. R. Montgomery. 2003. Patterns and processes of wood debris accumulation in the Queets river basin, Washington. Geomorphology 51(1-3):81-107.

Achord, S., R. W. Zabel, and B. P. Sandford. 2007. Migration timing, growth, and estimated parr-to-smolt survival rates of wild snake river spring-summer Chinook salmon from the salmon river basin, Idaho, to the Lower Snake River. Transactions of the American Fisheries Society 136(1):142-154.

Aitken, J. K., C. K. Cook-Tabor, and R. C. Wunderlich. 1996. Travel time of coho salmon and steelhead smolts emigrating through Howard Hanson Reservoir, King County, Washington. U.S. Fish and Wildlife Service, Olympia, Washington.

Al-Chokhachy, R., and P. Budy. 2008. Demographic Characteristics, Population Structure, and Vital Rates of a Fluvial Population of Bull Trout in Oregon. Transactions of the American Fisheries Society 137(6):1709-1722.

Al-Chokhachy, R., P. Budy, and M. Conner. 2009. Detecting declines in the abundance of a bull trout (Salvelinus confluentus) population: understanding the accuracy, precision, and costs of our efforts. Canadian Journal of Fisheries and Aquatic Sciences 66(4):649-658.

Al-Chokhachy, R., M. Sorel, D. A. Beauchamp, C. Clark, and E. D. Lowery. 2014. Development of New Information to Inform Fish Passage Decision at the Yale and Merwin Hydro Projects on the Lewis River. Annual Report submitted to PacifiCorp. U.S. Geological Survey, Bozeman, Montana.

Anderson, J. H., G. R. Pess, R. W. Carmichael, M. J. Ford, T. D. Cooney, C. M. Baldwin, and M. M. McClure. 2014. Planning Pacific Salmon and Steelhead Reintroductions Aimed at Long-Term Viability and Recovery. North American Journal of Fisheries Management 34(1):72-93.

Anlauf, K. J., D. W. Jensen, K. M. Burnett, E. A. Steel, K. Christiansen, J. C. Firman, B. E. Feist, and D. P. Larsen. 2011. Explaining spatial variability in stream habitats using both natural and management-influenced landscape predictors. Aquatic Conservation-Marine and Freshwater Ecosystems 21(7):704-714.

Baker, W. P., M. J. Divens, and B. D. Bolding. 2015. Tiger muskellunge growth, condition, diet, and effect on target prey species in two Eastern Washington lakes: progress report 2001 - 2006. Washington Department of Fish and Wildlife, FPT 15-02, Olympia, Washington.

Beauchamp, D. A., C. Luecke, W. A. Wurtsbaugh, H. G. Gross, P. Budy, S. Spaulding, R. Dillenger, and C. P. Gubala. 1997a. Hydroacoustic assessment of abundance and diel distribution of sockeye salmon and kokanee in the Sawtooth Valley Lakes, Idaho. North American Journal of Fisheries Management 17(253-267).

Beauchamp, D. A., C. Luecke, W. A. Wurtsbaugh, H. G. Gross, P. E. Budy, S. Spaulding, R. Dillenger, and C. P. Gubala. 1997b. Hydroacoustic assessment of abundance and diel distribution of sockeye salmon and kokanee in the Sawtooth Valley Lakes, Idaho. North American Journal of Fisheries Management 17:253-267.

Beauchamp, D. A., D. Parrish, and P. Whaley. 2009. Salmonids/coldwater species in large standing waters. Pages 97-117 in S. A. Bonar, D. Willis, and W. A. Hubert, editors.

Standard Sampling Methods for North American Freshwater Fishes. American Fisheries Society, Bethesda, Maryland.

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

Bilby, R. E., and J. W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and 2nd-growth forests in southwesetern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48(12):2499-2508.

Blair, G. R., L. C. Lastelle, and L. E. Mobrand. 2008. The ecosystem diagnosis and treatment model: a tool for assessing salmonid performance potential based on habitat conditions. American Fisheries Society Symposium 71:22.

Brakensiek, K. E., and D. G. Hankin. 2007. Estimating overwinter survival of juvenile Coho salmon in a northern California stream: Accounting for effects of passive integrated transponder tagging mortality and size-dependent survival. Transactions of the American Fisheries Society 136(5):1423-1437.

Buffington, J. M., and D. R. Montgomery. 1999. A procedure for classifying textural facies in gravel-bed rivers. Water Resources Research 35(6):1903-1914.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51(8):1389-1406.

Candy, J. R., and T. D. Beacham. 2000. Patterns of homing and straying in southern British Columbia coded-wire tagged chinook salmon (Oncorhynchus tshawytscha) populations. Fisheries Research 47(1):41-56.

Cederholm, C. J., R. E. Bilby, P. A. Bisson, and B. Bumstead. 1997. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. North American Journal of Fisheries Management 17:947-963.

CHaMP. 2013. Columbia Habitat Monitoring Program. Scientific protocl for salmonid habitat surveys with the Columbia Habitat Monitoring Program. Prepared by the Integrated Status and Effectiveness Monitoring Program, Wauconda, Washinton.

Chipps, S. R., and J. E. Garvey. 2007. Assessment of diets and feeding patterns. Pages 473-514 *in* C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

DeNiro, M. J., and S. Epstein. 1978. Influence of diet on the distribution of carbon isotopes in animals. Geochimica et Cosmochimica Acta 42:495-506.

Dilley, S. J. 1994. Horizontal and vertical distribution of juvenile salmonids in Howard Hanson Reservoir. U.S. Fish and Wildlife Service, Olympia, Washington.

Dunham, J., G. Chandler, B. Rieman, and D. Martin. 2005. Measuring stream temperature with digital data loggers: a user's guide. U.S. Department of Agriculture, Forest Service, Gen. Tech. Rep. RMRS GTR-150WWW, Fort Collins, Colorado.

Dunham, J., B. Rieman, and K. Davis. 2001. Sources and magnitude of sampling error in redd counts for bull trout. North American Journal of Fisheries Management 21(2):343-352.

Durkin, J. T., P. D. L., and R. F. Raleigh. 1966. Distribution and movement of juvenile salmon in Brownlee Reservoir, 1962-1965. Bureau of Commercial Fisheries Biological Laboratory, Seattle, Washington.

Eggers, D. M. 1978. Limnetic feeding behavior of juvenile sockeye salmon in Lake Washington and predator avoidance. Limnology and Oceanography 23:1114-1125.

Fausch, K. D., and T. G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693.

Gilliam, J. W. 1994. Riparian wetlands and water quality. Journal of Environmental Quality 23(5):896-900.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. Bioscience 41(8):540-551.

Hamilton, J. A. R., L. O. Rothfus, M. W. Erho, and J. D. Remington. 1970. Use of a hydroelectric reservoir for rearing of coho salmon (*Oncorhynchus kisutch*). Washington Department of Fisheries, Research Bulletin No 9., Olympia, Washington.

Hansen, M. J., C. P. Madenjian, J. H. Selgeby, and T. E. Helser. 1997. Gillnet selectivity for lake trout (Salvelinus namaycush) in Lake Superior. Canadian Journal of Fisheries and Aquatic Sciences 54(11):2483-2490.

Hawkins, C. P., J. L. Kershner, P. A. Bisson, M. D. Bryant, L. M. Decker, S. V. Gregory, D. A. McCullough, C. K. Overton, G. H. Reeves, R. J. Steedman, and M. K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18(6):3-12.

Isaak, D. J., and T. C. Bjornn. 1996. Movement of northern squawfish in the tailrace of a lower Snake River dam relative to the migration of juvenile anadromous salmonids. Transactions of the American Fisheries Society 125(5):780-793.

Isaak, D. J., and D. L. Horan. 2011. An Evaluation of Underwater Epoxies to Permanently Install Temperature Sensors in Mountain Streams. North American Journal of Fisheries Management 31(1):134-137.

Isely, J. J., and T. B. Grabowski. 2007. Age and growth. Pages 187-228 *in* C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of inland fisheries data. American Fisheries Society, Bethesda, Maryland.

Johnson, S. L., J. D. Rodgers, M. F. Solazzi, and T. E. Nickelson. 2005. Effects of an increase in large wood on abundance and survival of juvenile salmonids (Oncorhynchus spp.) in an Oregon coastal stream. Canadian Journal of Fisheries and Aquatic Sciences 62(2):412-424.

Kershner, J. L., B. B. Roper, N. Bouwes, R. Henderson, and E. Archer. 2004. An analysis of stream habitat conditions in reference and managed watersheds on some federal lands within the Columbia River basin. North American Journal of Fisheries Management 24(4):1363-1375.

Kiffney, P. M., G. R. Pess, J. H. Anderson, P. Faulds, K. Burton, and S. C. Riley. 2009. Changes in fish communities following recolonization of the Cedar River, WA, USA by Pacific salmon after 103 years of local extirpation. River Research and Applications 25(4):438-452.

Kondolf, G. M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129(1):262-281.

Larsen, D. P., P. R. Kaufmann, T. M. Kincaid, and N. S. Urquhart. 2004. Detecting persistent change in the habitat of salmon-bearing streams in the Pacific Northwest. Canadian Journal of Fisheries and Aquatic Sciences 61(2):283-291.

Lestelle, L. C., L. E. Mobrand, J. A. Lichatowich, and T. S. Vogel. 1996. Applied ecosystem analysis-a primer. Bonneville Power Administration, Project Number 9404600, Portland, Oregon.

Lynch, J. M. 1993. Evaluation of the Merwin trap as a means of northern squawfish (*Ptychocheilus oregonensis*) control in the Columbia River. University of Washington, Seattle, Washington.

Marcot, B. G., C. S. Allen, S. Morey, D. Shively, and R. White. 2012. An Expert Panel Approach to Assessing Potential Effects of Bull Trout Reintroduction on Federally Listed Salmonids in the Clackamas River, Oregon. North American Journal of Fisheries Management 32(3):450-465.

McClain, M. E., E. W. Boyer, C. L. Dent, S. E. Gergel, N. B. Grimm, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, W. H. McDowell, and G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6(4):301-312.

McClure, M. M., S. M. Carlson, T. J. Beechie, G. R. Pess, J. C. Jorgensen, S. M. Sogard, S. E. Sultan, D. M. Holzer, J. Travis, B. L. Sanderson, M. E. Power, and R. W. Carmichael.

2008. Evolutionary consequences of habitat loss for Pacific anadromous salmonids. Evolutionary Applications 1(2):300-318.

McElhany, P., M. H. Ruckelshaus, M. J. Ford, T. C. Wainwright, and E. P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of Evolutionary Significant Units. National Marine Fisheries Service, Seattle, Washington.

McHugh, P., and P. Budy. 2005. A comparison of visual and measurement-based techniques for quantifying cobble embeddedness and fine-sediment levels in salmonid-bearing streams. North American Journal of Fisheries Management 25(4):1208-1214.

McHugh, P., P. Budy, and H. Schaller. 2004. A model-based assessment of the potential response of Snake River spring-summer Chinook salmon to habitat improvements. Transactions of the American Fisheries Society 133(3):622-638.

McIntyre, J. K., D. A. Beauchamp, M. M. Mazur, and N. C. Overman. 2006. Ontogenetic trophic interactions and benthopelagic coupling in Lake Washington: Evidence from stable isotopes and diet analysis. Transactions of the American Fisheries Society 135(5):1312-1328.

Miranda, L. E., and P. W. Bettoli. 2007. Mortality. Pages 229-278 in C. S. Guy, and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

Montgomery, D. R., J. M. Buffington, R. D. Smith, K. M. Schmidt, and G. Pess. 1995. Pool spacing in forest channels. Water Resources Research 31(4):1097-1105.

Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. Ecology and Management of Wood in World Rivers 37:21-47.

Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28:621-658.

Nelson, A. G. 2014. fishmethods: fishery science methods and models in R. R package version 1.7-0.

Nickelson, T. E. 2000. Population assessment: Oregon Coast coho salmon ESU. Oregon Department of Fish and Wildlife, Portland, Oregon.

NMFS. 2007. Biological opinion and Magnuson-Stevens Fishery Conservation and Management Act Consultion; operation of PacifiCorp and Cowlitz PUD's Lewis River Hydroelectric Project (FERC Nos. 935, 2071, 2111, and 2213). National Oceanic and Atmospheric Administration: National Marine Fisheries Service, 2005/05891, Seattle, Washington.

ODEQ. 1995. Temperature: 1992-1994 water quality standars review. Final issue paper. Oregon Department of Environmental Quality.

PacifiCorp. 1999. Final technical report aquatic resources-Yale Hydroelectric Project FERC Project No. 2071. Appendix 2.1-1. PacifiCorp, Portland, Oregon.

PacifiCorp, and Cowlitz-PUD. 2004a. Report on life history, habitat requirements, and distribution of aquatic analysis species (AQU 1). PacifiCorp and Cowlitz PUD, Portland, Oregon.

PacifiCorp, and Cowlitz-PUD. 2004b. Water quality monitoring and assessments study. Final technical report. Pacificorp and Cowlitz PUD, Portland, Oregon.

PacifiCorp, and Cowlitz-PUD. 2014. Lewis River hatachery and supplementation plan. PacifiCorp and Cowlitz PUD, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2001. Migratory behavoir of radio-tagged juvenile Coho Salmon through Swift Reservoir, 2001. PacifiCorp, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2002. Migratory behavoir of radio-tagged juvenile Chinook Salmon through Swift Reservoir, 2002. PacifiCorp/Cowlitz PUD, Portland, Oregon.

PacifiCorp/Cowlitz-PUD. 2004. Behavoir of salmonid smolts at Swift Dam using 3-dimensional tracking with acoustic tags. PacifiCorp/Cowlitz-PUD, AQU 17, Portland, Oregon.

Pess, G. R., P. M. Kiffney, M. C. Liermann, T. R. Bennett, J. H. Anderson, and T. P. Quinn. 2011. The Influences of Body Size, Habitat Quality, and Competition on the Movement and Survival of Juvenile Coho Salmon during the Early Stages of Stream Recolonization. Transactions of the American Fisheries Society 140(4):883-897.

Quinn, T. P. 1993. A review of homing and straying of wild and hatchery-produced salmon. Fisheries Research 18:29-44.

Seber, G. A. F. 1982. The estimation of animal abundance. Griffin, London, England.

Serl, J., and C. Morrill. 2010. Summary report for the 1996 to 2009 seasonal operation of the Cowlitz Falls fish facility and related Cowlitz Falls anadromous reintroduction program activities. Washington Department of Fish and Wildlife.

Sheer, M. B., and E. A. Steel. 2006. Lost watersheds: Barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and Lower Columbia River basins. Transactions of the American Fisheries Society 135(6):1654-1669.

Spangler, G. R., and J. J. Collins. 1992. Lake Huron fish community structure based on gill-net catches corrected for selectivity and encounter probability. North American Journal of Fisheries Management 12:585-597.

Tabor, R. A., S. T. Sanders, M. T. Celedonia, D. W. Lantz, S. Damm, T. M. Lee, Z. Li, and B. E. Price. 2010. Spring/summer habitat use and seasonal movement patterns of predatory fishes in the Lake Washington Ship Canal. U.S. Fish and Wildlife Service, Lacey, Washington.

Tappel, P. D., and T. C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management 3:123-135.

Temple, G. M., and T. N. Pearsons. 2012. Risk management of non-target fish taxa in the Yakima River Watershed associated with hatchery salmon supplementation. Environmental Biology of Fishes 94(1):67-86.

Thurow, R. F., J. T. Peterson, and J. W. Guzevich. 2006. Utility and Validation of Day and Night Snorkel Counts for Estimating Bull Trout Abundance in First- to Third-Order Streams. North American Journal of Fisheries Management 26(1):217-232.

Urquhart, N. S., S. G. Paulsen, and D. P. Larsen. 1998. Monitoring for policy-relevant regional trends over time. Ecological Applications 8(2):246-257.

USFWS. 2006. Biological Opinion for the Federal Energy Regulatory Commission relicensing of the Lewis River Hydroelectric Projects: Merwin (No. 935), Yale (No. 2071), Swift No. 1 (No. 2111), and Swift No. 2 (No. 2213). U.S. Fish and Wildlife Service, FWS Reference Number: 1-3-06-F-0177, Lacey, Washington.

Vander Zanden, M. J., and J. B. Rasmussen. 1999. Primary consumer delta C-13 and delta N-15 and the trophic position of aquatic consumers. Ecology 80(4):1395-1404.

Weeber, M. A., G. R. Giannico, and S. E. Jacobs. 2010. Effects of Redd Superimposition by Introduced Kokanee on the Spawning Success of Native Bull Trout. North American Journal of Fisheries Management 30(1):47-54.

Weigel, D. E., P. J. Connolly, K. D. Martens, and M. S. Powell. 2013. Colonization of Steelhead in a Natal Stream after Barrier Removal. Transactions of the American Fisheries Society 142(4):920-930.

Weitkamp, L. A., T. C. Wainwright, G. J. Bryant, G. B. Milner, D. J. Teel, R. G. Kope, and R. S. Waples. 1995. Status review of coho salmon from Washington, Oregon, and California. National Marine Fisheries Service, NOAA Technical Memorandum NMFS-NWFSC-24, Seattle, Washington.

Wolman, M. G. 1954. A method of sampling coarse river-bed material. Transactions American Geophysical Union 35:951-956.

Zimmerman, M. P. 1999. Food habits of smallmouth bass, walleyes, and northern pikeminnow in the Lower Columbia River Basin during outmigration of juvenile anadromous salmonids. Transactions of the American Fisheries Society 128:1036-1054.

Table 1. Seasonal diet proportions (n= total number of diets) for invertebrate feeding salmonids including (CHK = Spring Chinook), RBT_h= hatchery Rainbow Trout), and KOK = Kokanee salmon) captured in Lake Merwin, Yale Lake, and Swift Reservoir June-November 2013 and May-November 2014 including cladocerans (Cladoc.), copepods (Copep.), terrestrial insects (Terr. ins), aquatic insects (Aqua. Ins.) and fish.

Location	Spp.	Size class	Season	n	Cladoc.	Copep.	Terr. ins.	Aqua. ins.	Fish
Swift	CHK	100-299	Spr	4	0.546	0.000	0.100	0.104	0.250
	CHK	100-299	Sum	7	1.000	0.000	0.000	0.000	0.000
	CHK	100-299	Fall	11	0.855	0.000	0.009	0.136	0.000
	RBT_h	200-399	Spr	10	0.593	0.000	0.046	0.360	0.000
	RBT_h	200-299	Sum	4	0.999	0.001	0.000	0.000	0.000
	RBT_h	200-399	Fall	13	0.893	0.002	0.049	0.056	0.000
Yale	KOK	0-299	Spr	14	0.684	0.084	0.043	0.189	0.000
	KOK	0-299	Sum	14	0.755	0.162	0.000	0.083	0.000
	KOK	100-199	Fall	17	0.879	0.062	0.059	0.000	0.000
Merwin	KOK	0-399	Spr	17	0.536	0.018	0.202	0.243	0.000
	KOK	200-399	Sum	11	0.908	0.001	0.018	0.073	0.000
	KOK	100-399	Fall	13	0.857	0.000	0.043	0.100	0.000

Figure 1. The location of Coho redds (red circles; 2012-2014) in Pine Creek and tributary to Pine Creek (P8), snorkel surveys used to identify overlap in the distribution of Bull Trout and Coho during 2013 and 2014 (grey), and the number of juvenile Coho (number by reach) detected in surveys.
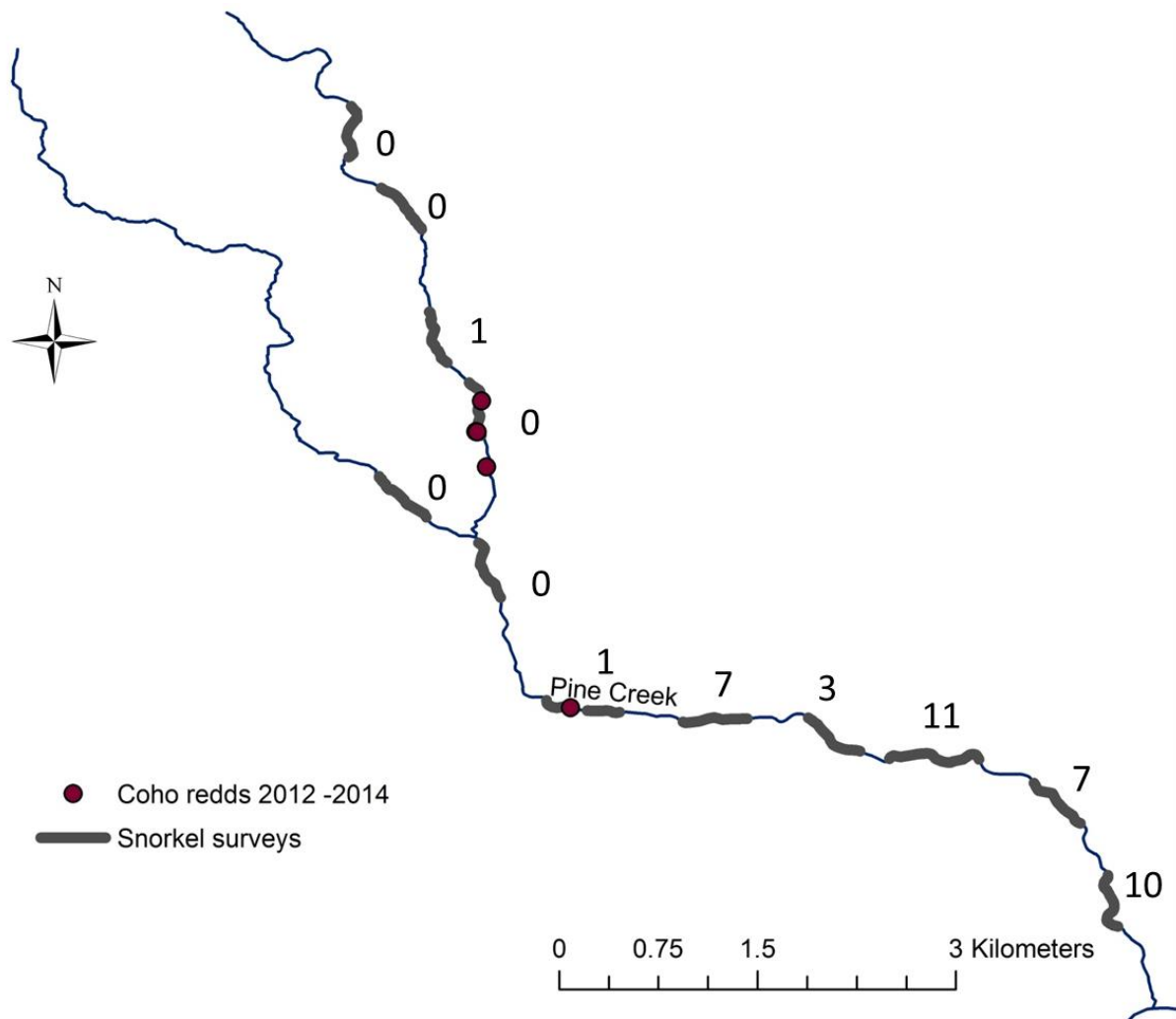
Figure 2. The proportion of Bull Trout (black), Coastal Cutthroat Trout (grey hatched), YOY Coho (white), Mountain Whitefish (light grey), and YOY *Oncorhynchus* spp. (Rainbow/Steelhead and Coastal Cutthroat Trout; dark grey) observed during snorkel surveys in different reaches (denoted by column) in Pine Creek and P8 during 2013 and 2014. Note: reaches are ordered in a downstream to upstream manner and 2013 and 2014 reaches are not the same-see methods).
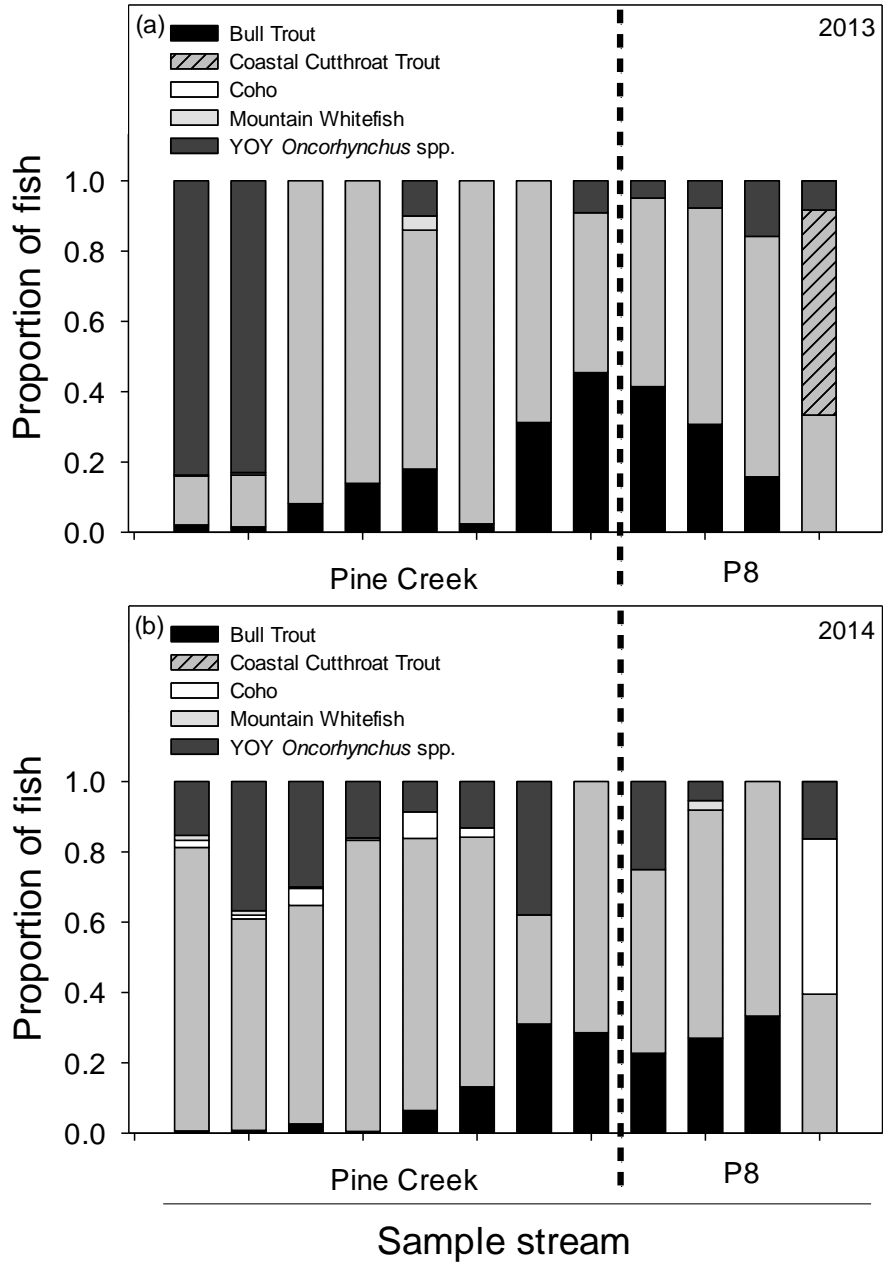
Figure 3. The cumulative frequency of the days between release and emigration past the Clear Creek PIT-tag antenna for acclimation Spring Chinook released during the spring of 2014 and 2015.
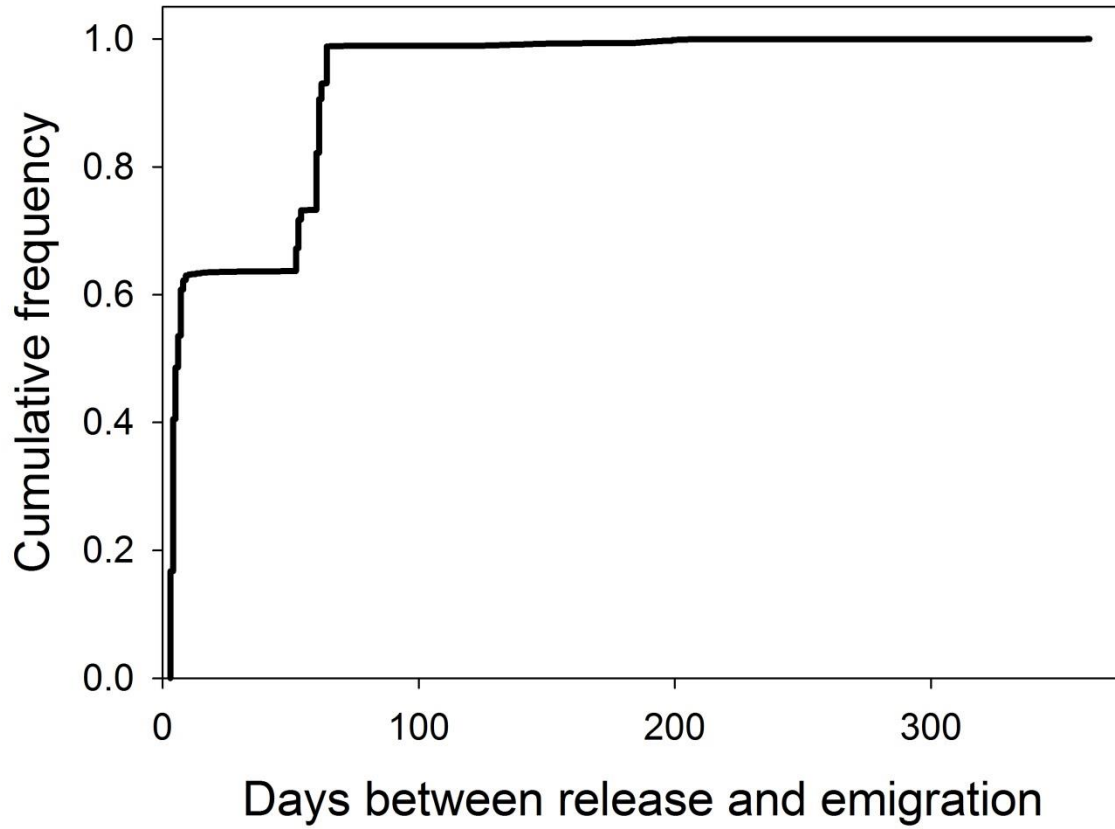
Figure 4. Stable isotope results during summer (a) and fall (b) for different size classes of fishes from P8, a main tributary to Pine Creek and the major Bull Trout spawning tributary in the Swift Reservoir portion of the upper Lewis River.
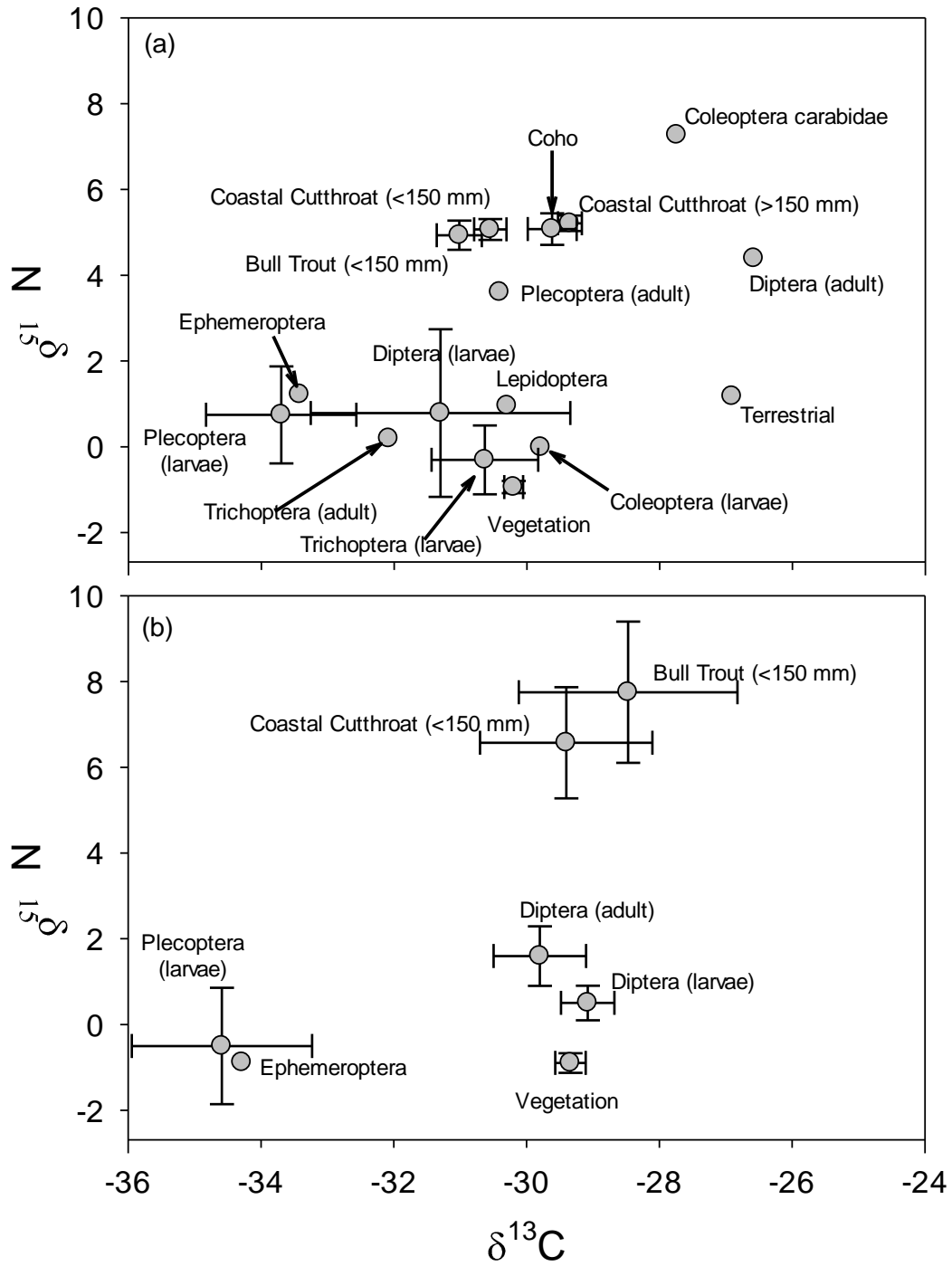
Figure 5. The distribution of Coho Salmon redds in the Upper Lewis Basin from surveys conducted in 2012 to 2014.
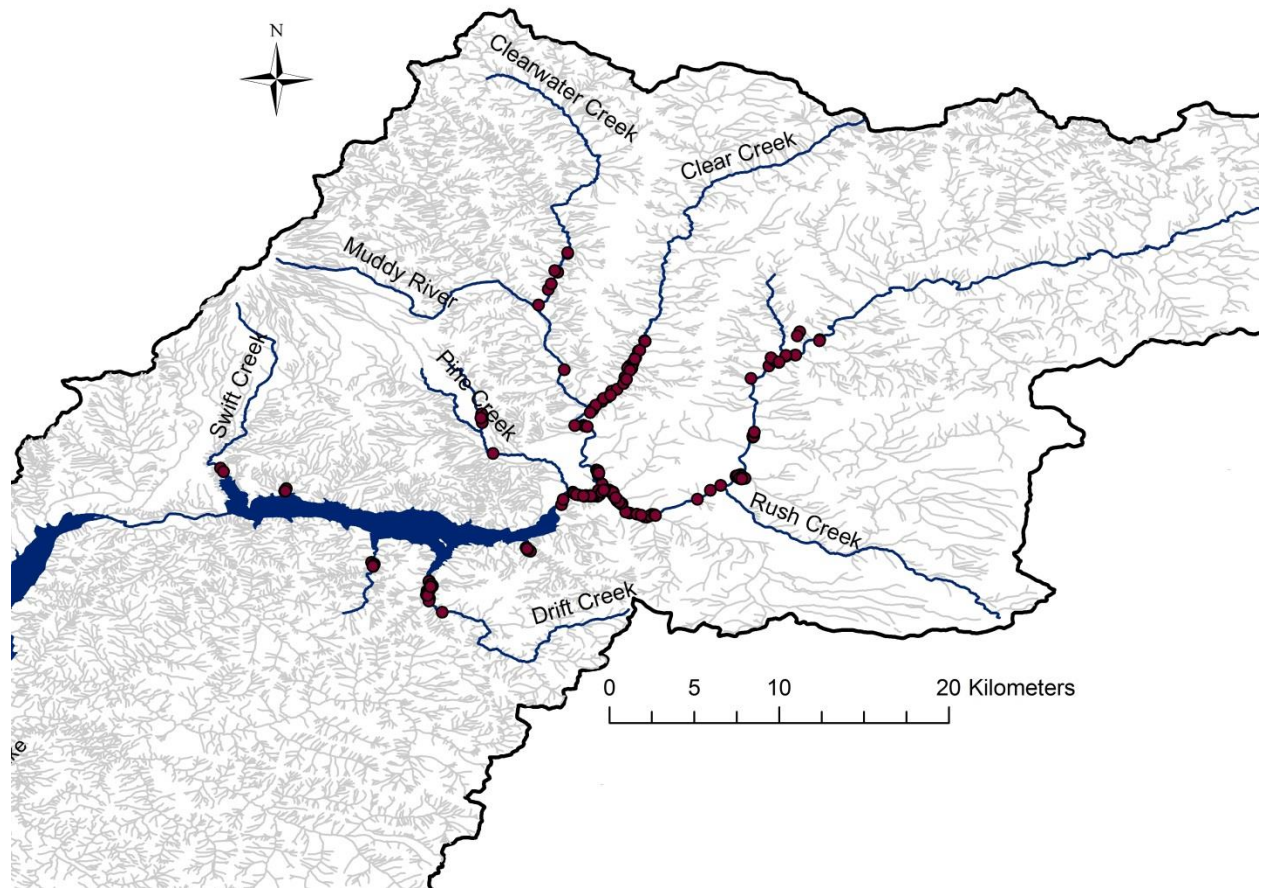
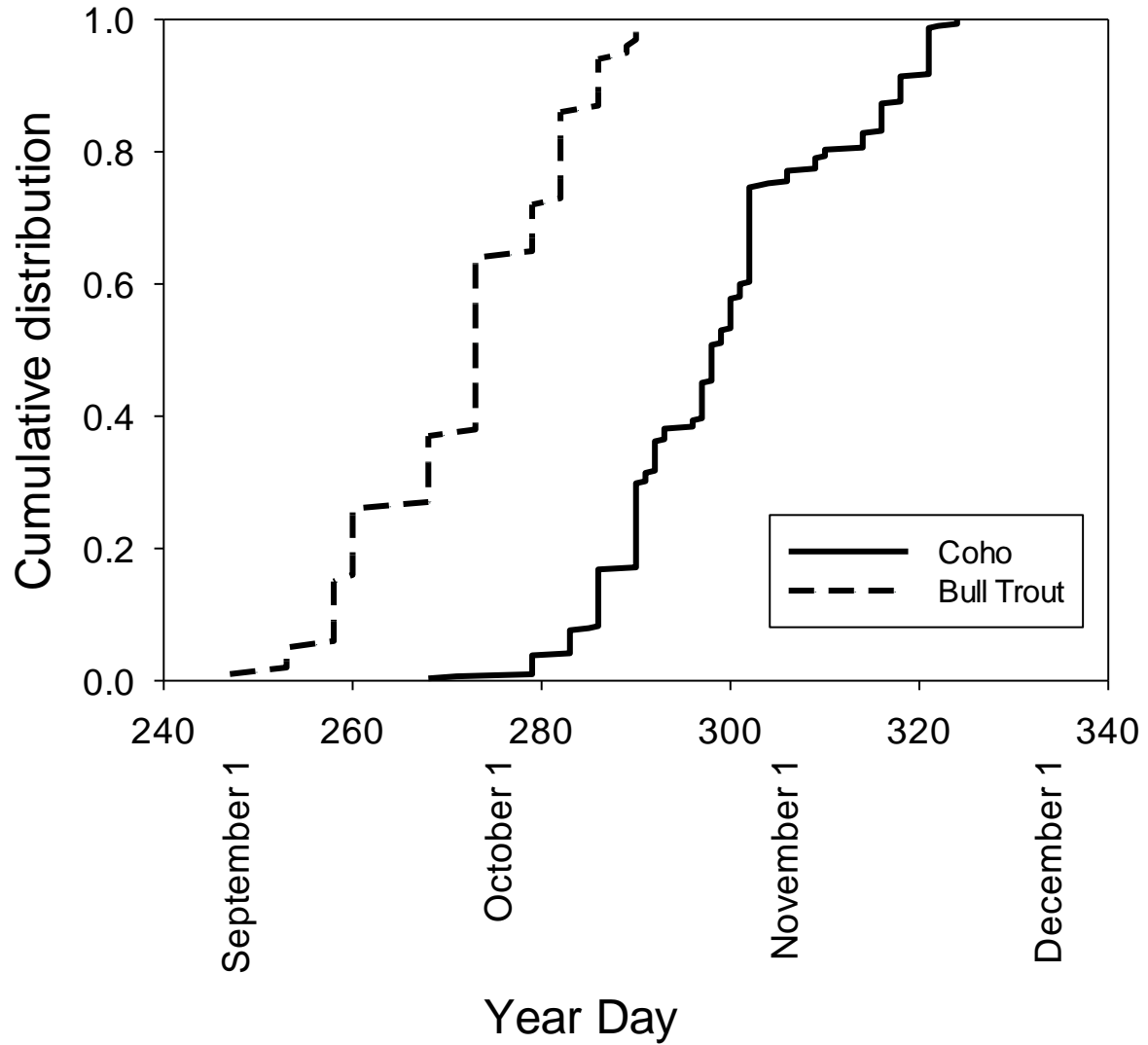Figure 6. The cumulative frequency distributions of the timing (shown as Year Day) of Coho (solid) and Bull Trout (dashed) redds in the Upper Lewis River.

Figure 7. Average δ ^{15}N and δ ^{13}C values (2SE) for organisms sampled in Swift Reservoir where BT$_{pre}$ = Bull Trout sampled in 2010 before significant reintroduction of anadromous salmonids; BT$_{post}$ = Bull Trout sampled in 2013-2014 after the reintroduction had begun; CCT = Coastal Cutthroat Trout; CHK = spring Chinook Salmon; COHO = Coho Salmon; COT = Sculpin; LSS = Large-Scale Sucker; MW = Mountain Whitefish; RBT = rainbow trout; RBT$_h$ = hatchery Rainbow Trout; and SB = Threespine Stickleback.

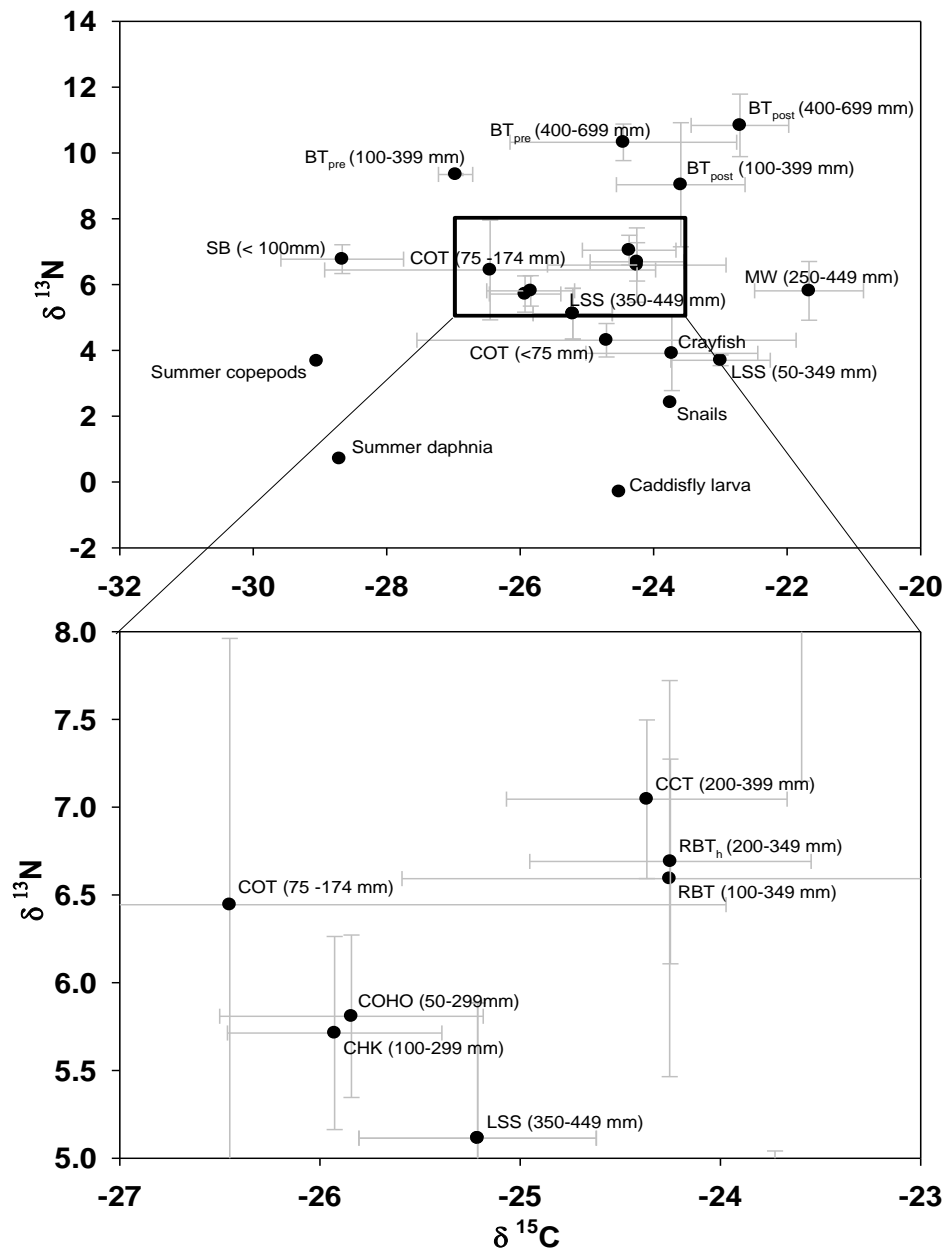Figure 8. Average δ ^{15}N versus δ ^{13}C (2 SE) for organisms sampled in Yale Lake where: BT = Bull Trout; CCT = Coastal Cutthroat Trout; COT = Sculpin; KOK = Kokanee; LSS = Large-Scale Sucker; NPM = Northern Pikeminnow; RBT = Rainbow Trout; and SB = Threespine Stickleback.
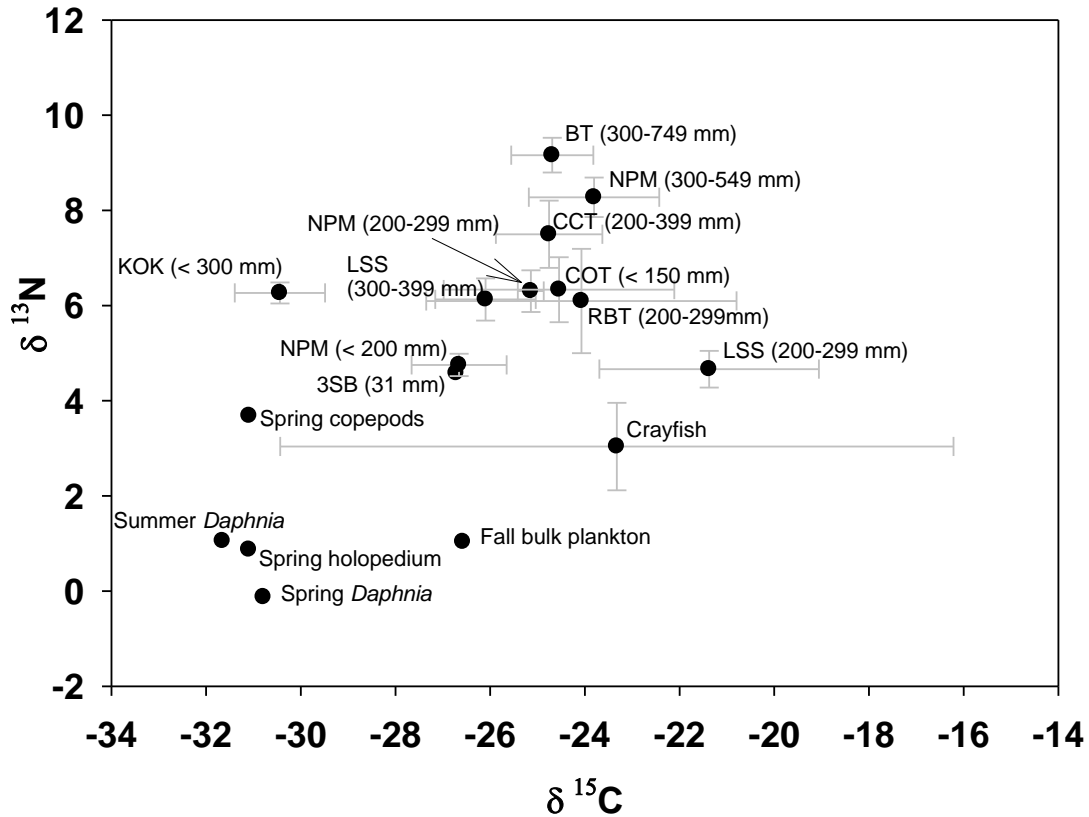
Figure 9. The average monthly density of *Daphnia* (±1 SE) in three depth strata from two stations located approximately 1/3 and 2/3 along the length of Swift, Yale, and Merwin Reservoirs between April and November 2014. The average length of *Daphnia* (±1 SE) measured each month is also included. *Data from the lower sampling site in Swift and Merwin Reservoirs collected in November 2013 are included to show a complete growing season, as samples could not be collected in these reservoirs in November 2014.